11-13-2017 10:00 AM

Identification of Thresholds in Benthic Macroinvertebrate Communities Associated with Agricultural Land Cover

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

Abstract

Agricultural land use affects benthic macroinvertebrate (BMI) community structure but riparian forest may mitigate its impact to within a specific threshold. BMI communities were sampled in small streams within the Grand River, Thames River, and Long Point watersheds in southwestern Ontario. The study assessed the location and amount of agricultural land use associated with variation in BMI assemblage structure. Three land use distribution scenarios were evaluated to isolate specific ranges of agricultural land use at either the riparian corridor or catchment scale, with the adjoining scale covering as wide a gradient of agricultural land use as possible. I did identify thresholds but the amount of variation associated with my thresholds would not enable us to suggest specific target ranges for land use managers looking to incorporate them into their stream biomonitoring programs. Further studies that assess various surficial material and share wider gradients will improve upon my findings.

Keywords: benthic macroinvertebrates, agriculture, riparian corridor, catchment, threshold, taxonomic composition.

Acknowledgments

I would like to thank my supervisor Dr. Adam Yates, who offered me an opportunity to complete a Master of Science degree when I never had any set intentions or plans to pursue one. I was randomly applying for jobs late at night one evening in 2013, and Adam had asked Chris Jones at the Ontario Ministry of Environment and Climate Change office in Dorset, Ontario to forward an email to anyone on his OBBN (Ontario Benthos Biomonitoring Network) list. I saw that email late that night, and responded to it, and Adam contacted me early the following morning.

Prior to committing to an important life changing decision, I met with one of my two mentors, Denis McGee from Sir Sandford Fleming College who thought that Adam's proposal that included benthic macroinvertebrate assessment and GIS work would be an excellent opportunity for me to pursue and learn many new skills. Another mentor of mine from Sir Sandford Fleming College, John Knight, had already previously suggested that I consider pursuing a Master of Science degree. I would thus like to thank both Denis and John for providing me with an amazing experience during my two years at Sir Sandford Fleming College. They provided me with their bountiful knowledge, guidance, and thought-provoking insight, always challenging me to question everything. I learned more in those two years with those gentlemen and other excellent professors than in any other two academic years throughout my life journey. I was also going through some very trying times in my personal life, and my time spent at Sir Sandford Fleming College provided me with a whole new focus to get back to my original biology roots. I

would also like to thank my hockey teammates during my time at Sir Sandford Fleming College who were also a foundation of support, and our amazing playoff runs only added to my overwhelming positive experience.

Dr. Adam Yates also made my decision to pursue a Masters' degree a little easier by providing a potential plan to receive funding through the Canadian Rivers Institute. Not only was I fortunate to receive a substantial stipend from their WATER (Watershed and Aquatics Training in Environmental Research) program which was funded by an NSERC (Natural Sciences and Engineering Research Council of Canada) CREATE (Collaborative Research and Training Experience) Program, but they also provided me with numerous field and online courses, conferences, and webinars that all contributed towards building upon my knowledge base and skill set. Even more important, were the lifelong friendships that were created from the amazing people I met throughout the course work I completed with the Canadian Rivers Institute. These people included the likes of both Chris Keung and Katrina Krievins who became trusted canoe partners throughout the adversity we faced in the journeys I made traversing the Little Jocko River and Magnetawan River with Chris, and the Amable du Fond River with Katrina. The Canadian Rivers Institute also provided me with other life firsts which included flying in an airplane, seeing one of the world's oceans for my first time, and then even jumping into the freezing cold North Atlantic Ocean in May three times which was the coldest water I had ever experienced in my life. Each of these experiences fell within the one week we spent in Fredericton, New Brunswick which was also a first, since I had never been east of the province of

Quebec until then. Finally, I also had the opportunity to see the main Laurentian Mountains during a second trip that followed our Fredericton course work, when we drove to a Canadian Rivers Institute research station in Sacre Coeur, Quebec. Again, we not only re-kindled our newly developed friendships with our colleagues, but I had never seen mountains before, and had also never seen that eastern portion of the St. Lawrence River. I even had the opportunity to see porpoises and Beluga Whales while visiting beautiful Tadoussac, Quebec where the Saguenay River empties into the St. Lawrence River. I had never seen marine mammals in their wild environment until then.

I would also like to acknowledge my great labmates who provided so much of the technical help I required in learning new skills during my Masters' program. Ed Krynak taught me how to conduct my stream sampling; prepare samples in the field; sort and count my benthic macroinvertebrates in the laboratory; sub-sampling procedures; washing samples in the laboratory; integrating my stream sites with Google Earth on my computer and on our automobile GPS; as well as the difficult steps involved in conducting my taxonomic adjustment process. Nolan Pearce was instrumental in helping me learn how to run one of my two statistical programs called SiZer through R software, as well as providing a wealth of other handy tricks applied to Microsoft Word, Excel, and PowerPoint. Sarah McKenzie also provided me with countless quick technical fixes when facing problems with Word and Excel. A thank you to Craig Irwin as well, especially for being there with me in the final days leading up to my thesis defence. Craig's calming presence really helped me to relax as I internally dealt with so much of the anxiety that I had built up. Finally, a thank you to Roger Holmes who assisted me with my field work throughout September and October of 2014, and to all of my other labmates who were all in some way, shape, or form, a part of my overall Masters' experience.

I would like to thank the three examiners at my thesis defence; Dr. Nusha Keyghobadi and Dr. Peter Ashmore from the Biology Department and Geography Department respectively, at the University of Western Ontario, and Dr. Bob Bailey from the University of Ontario Institute of Technology, who were all instrumental in providing thorough revisions that contributed to the overall completed thesis. Dr. Peter Ashmore was also part of my Masters' Thesis Committee that oversaw the annual progress I was making in my research. He was also one of my excellent teachers who co-taught the course, "Philosophies of Geography" with Dr. Jeff Hopkins, where both men consistently challenged me to think beyond accepted practices that I am accustomed to in science. Dr. Bob Bailey introduced me to R software as my teacher for the course, "Design, Analysis & Interpretation of Quantitative Biological Research". R software was a key component to the analysis of my data.

Other department colleagues of mine were instrumental as well. Hossein Hosseini played a key role in figuring out a difficult problem I had encountered with my GIS work that saved me from a lot of stress. Hossein also taught me a wide assortment of new GIS skills. Team Ghana, led by Roger Antabe, were always there by my side with their friendship, laughs, and social gatherings. Joe Smrekar was always available anytime I needed any sort of technical assistance with computers, room codes, or print credits. My lab coordinator Erika Hill was always around as both a friend and a colleague who always made sure I had what I needed in the laboratory. Lori Johnson has long been my own Mother while at school, who made sure I was well organized with important university administrative work, assisted with financial and tax information, ensured I found an amazing place to live, and always provided me with support and encouragement. Finally, a thank you to Dr. Dan Shrubsole who also contributed to my overall graduate student experience as Dr. Shrubsole served as the Geography Department Chair for most of my time as a graduate student, playing a role in the process of accepting me into the Master of Science program, and was always providing stories, laughs, advice, and encouragement.

I would also like to acknowledge the Environment and Sustainability portion of my Masters' degree. I graduated from the Centre for Environment and Sustainability's Collaborative Program after three years of work and guidance from my teacher Dr. Desmond Moser. My entire graduate student experience was only further enhanced through the many great people, fellow students, and teachers I engaged throughout the program. Dr. Hugh Henry from the Biology Department served as both an excellent teacher of mine for the course; "Global Change Biology" and acted as my supervisor for the Environment and Sustainability Collaborative Program. I also want to extend a big thank you to Dr. Abednego Aryee who I had never met previously, and he interviewed me at a time when I was miserably short on finances, and I was fortunate to have him hire me as his Teaching Assistant for his Environment and Sustainability graduate course; "Planning and Management". Additional thanks are extended to Dr. Phil Stooke,

Wendy Dickinson, and Mark Moscicki, who also had me serve as a Teaching Assistant in their respective courses.

A special thank you goes to the many First Nations' communities that contributed to my learning and advancement, further enhancing my principles and values in respecting Mother Earth and Her many river systems that I love to explore. A thank you to the Canadian Water Network who I was largely put into contact with by the Canadian Rivers Institute. The week-long "Paddling Together: Integrative Traditional and Western Water Knowledge" conference in North Bay, Ontario forged instant friendships. The guidance from both University of Guelph's Dr. Khosrow Farahbakhsh and Dr. Sherilee Harper, was both genuine and thought provoking, and the Canadian Water Network's Liana Kreamer who coordinated the entire event, all contributed towards making this conference an unforgettable life experience. I relished the opportunity to return to my home of North Bay, Ontario that I am immensely fond of, and once again, workalongside both my Nipissing First Nation and Dokis First Nation Brothers and Sisters. My thanks, thoughts, and appreciation also extend to both Clint Jacobs from Walpole Island First Nation and Paul General from Six Nations' who welcomed me into their communities and provided me with further teaching throughout my time spent with the Environment and Sustainability Collaborative Program. Finally, a special thank you to Roger Jacklin of Magnetawan First Nation who I met on my journey through his territory on the Magnetawan River and who has since become a genuine friend.

Finally, I would like to give a heartfelt thank you to my loving family; my Mom and Dad, my brother Chris, and my sister Amy. I would never have even started this journey, let alone maneuver my way through it, if it was not for all of your unconditional love and support. Also, much love to my foundational North Bay contingent; the Graniteville Glory boys Billy, Kyle, Jason, and Derek, and numerous other close friends, largely from North Bay, who have all been right by my side since the beginning. These friendships are especially important when life throws the unexpected your way and the only way you manage to guide your ship through those stormy seas is from those dear friends and loved ones being right there with you to take all of it on. I love you all so very much.

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List of Abbreviations

BMI – benthic macroinvertebrate

BMP – best management practice

CV – coefficient of variation

DEM – digital elevation model

DipteraRich – Diptera richness

EPT – Ephemeroptera, Plecoptera, Trichoptera

EPTRich – Ephemeroptera, Plecoptera, Trichoptera Richness

FBIRes – Hilsenhoff Family Biotic Index Residuals

GIS – geographic information system

GRCA – Grand River Conservation Authority

HFBI or FBI – Hilsenhoff Family Biotic Index

MultiovoltinismRes – Multivoltinism Residuals

nTaxa – Community Richness

SegReg – segmented regression

SiZer – significant zero crossings

%Small – Percent Small Body Size

1.0 Introduction

1.1 Agricultural Land Use

Agricultural development has the potential to modify landscapes when conducted at large scales. The agricultural regions of North America have been developed for the past two hundred years (Sharitz *et al.,* 1992), exemplified by the conversion of more than 40% of the six largest river basins in the United States into agricultural land use (Allan, 2004). Landscapes where the soil is fertile are now used for intensive row crop cultivation, whereas more shallow soils are being managed for lower intensity pasture agriculture (Yates and Bailey, 2010). When the land is used for crop cultivation, tillage practices disturb the soil matrix (Sallenave and Day, 1991), and fertilizers and manure applications rich in phosphorus and nitrogen, are added to the soil in the spring and fall seasons, whereas pesticides and herbicides are added to the crops throughout the growing season (Skinner *et al.,* 1997). Low order streams in the headwaters of watersheds are often channelized to enhance drainage and many smaller streams, both intermittent and permanent flowing, have been buried (Yates *et al.,* 2007). Tile drain systems are increasingly installed beneath the soil surface of agricultural properties to accelerate soil drainage and lower the groundwater table to increase the amount of arable lands (Prestegaard *et al.,* 1994). The described alterations of landscapes for agricultural uses results in a wide array of stressors being applied to river ecosystems (Allan, 2004).

Common agricultural stressors include increased nutrient and fine sediment loads to streams (Riley *et al.,* 2003; Skinner *et al.,* 1997). These stressors find their way to the stream through increased surface and subsurface runoff, particularly where riparian vegetation cover has been removed (Allan, 2004). Nutrient concentrations are significantly larger in streams exposed to agricultural land use, particularly for species of nitrogen and phosphorus (Johnson *et al.,* 1997). Sediment yields increase significantly in landscapes that have expansive agricultural land use, leading to reduced water clarity as well as deposition of sediments on the streambed, resulting in loss of interstitial spaces between larger substrate particles (Burdon *et al.,* 2013; Larsen *et al.,* 2009). Pesticides (i.e., insecticides, herbicides, fungicides) used to protect agricultural crops also find their way into streams through surface runoff and groundwater (Skinner *et al.,* 1997). Agricultural land use can also impact stream hydrology but the effects are variable depending on evapotranspiration rates of crops, extent of drainage and irrigation systems, and changes to soil infiltration capacity (Allan, 2004). When there has been significant loss of wetland areas, or enhancement of drainage ditches, it is common for stream flows to increase in both magnitude and frequency during storm events (Allan, 2004). Effects of drainage enhancement are magnified through removal of natural vegetation, leading to reduced soil infiltration and increased surface runoff, resulting in accelerated channel incision and bank erosion (Prestegaard *et al.,* 1994). Overgrazing and draining of wetlands leads to significant reductions in water retention within the watershed and results in water that is quickly routed downstream, leaving the river more susceptible to higher frequency of extreme floods, and reducing base flows as channels widen (Poff *et al.,* 1997). Higher peak flows will result in scouring the stream bottom, as well as eroding the stream banks, further contributing increases in sediments to the stream, and leading to a straightening or channelized affect on the stream (Schlosser, 1991). This leads to a loss of diversity in stream substrate, depth, and flow, which otherwise would enable a stream to create its diverse range of riffle, run, and pool habitats (Schlosser, 1991). The low flow conditions that follow these peak events leave a slow flowing, widened channel that enables sediments to drop out of the water column and homogenize stream substrate (Naiman and Decamps, 1997). Each of these aforementioned stressors impact the ecology of the stream with the effects on stream benthic macroinvertebrate (BMI) communities being particularly well documented (Wang *et al.,* 2007; Liess *et al.,* 2012; Larsen *et al.,* 2009; Elbrechet *et al.,* 2016).

When naturally vegetated landscapes are replaced with agricultural land use, stream biota is often affected (Allan, 2004). The elevated nutrient loadings to streams associated with agricultural land use have been found to have particularly adverse ecological effects on stream ecosystems, including eutrophication (Evans-White *et al.,* 2009). Eutrophication leads to increases in algal and macrophyte biomass (Wang *et al.,* 2007) and thus, increases in primary production (Liess *et al.,* 2009). Because BMI communities depend upon food availability from primary production, nutrients such as nitrogen and phosphorus are more likely to influence BMI through these indirect pathways (Richards *et al.,* 1993). Consequently, with increases in algae and macrophytes, nocturnal respiration leads to decreases in dissolved oxygen (Miltner and Rankin, 1998) and increases in pH, which negatively affects sensitive BMI (e.g. Trichoptera) and fish communities by impeding their growth or even leading to their death if oxygen depletion and pH levels are severe enough (Dodds and Welch, 2000). Changes in the water chemistry can result in alterations to the structure of the benthic algal, BMI, and fish communities (Lange *et al.,* 2011), and associated alterations of food web dynamics (Weigel and Robertson, 2007). As a result, increased nutrient loadings have been observed to cause decreases in biodiversity (Liess *et al.,* 2012). Sediments can also have significant effects upon biological communities, such as fish and BMI, with declines in taxa abundance and richness (Feld, 2013). In fact, inputs of fine sediments are viewed as a leading cause of impairment to stream biota in streams of numerous countries (Burdon *et al.,* 2013). Deposition of suspended sediments has been observed to be a leading contributor towards altered density, biomass and composition of BMI communities through loss of habitat variability, as fine sediments bury coarser substrate particles (Richards *et al.,* 1993). For example, Larsen *et al.* (2009) observed as much as a 25% decrease in the richness of sensitive BMI (Emphemeroptera, Plecoptera, and Trichoptera) taxa, at sites where fine particles covered 30% of the substrate compared to sites that were free of fine sediments. Higher sediment loads also lead to greater turbidity, which is negatively associated with light and can thus limit primary production (Richards *et al.,* 1993). Even low levels of turbidity can result in significant reductions of primary productivity (Lloyd *et al.,* 1987). In fact, sedimentation often results in decreased productivity across all trophic levels in aquatic ecosystems, altering the structure of plant, BMI, and fish communities (Karr and Schlosser, 1978). Fine sediments are also damaging to the delicate respiratory structures of sensitive BMI that respire using gills. Similarly, fine sediments can also stress or, in extreme cases, cause direct mortality to fish by clogging their opercular cavity and gill filaments, while also interfering with reproductive processes (Karr and Schlosser, 1978) through burying of spawning sites and associated interference with larval fish development (Berkman and Rabeni, 1987). BMI requiring temporary attachment to particle-free surfaces in the substratum or ones that swim through the water column, can also be impacted by the abrasive nature of fine sediments (Larsen *et al.,* 2011). Increased rates of sedimentation from agricultural landscapes will also be associated with elevated levels of any other compounds such as pesticides adsorbed to sediment. For example, in a paired watershed study conducted by Sallenave and Day (1991), production and density of Caddisfly was consistently lower in the watershed subjected to higher application rates of the pesticides atrazine, metolachlor, and EPTC. Skinner *et al.* (1997) also reported that pesticides entering streams from agricultural practices might be affecting the residing flora and fauna (Skinner *et al.,* 1997). However, because agricultural pesticide concentrations are rarely measured in research studies assessing the effects of agricultural land use on stream biota, the impact of pesticides may be more significant than is currently recognized (Allan, 2004).

Agricultural practices influence stream biota both directly or indirectly by affecting patterns of surface runoff and stream flow (Stewart *et al.,* 2000). Row crop agriculture is often associated with increases in hydrologic variation, which has been associated with declines in merovoltine (long-lived) and large-bodied

BMI (Poff and Ward, 1989). Many fish species depend upon the natural high and low seasonal flow of streams to initiate various reproductive or life stages such as migration, spawning, egg hatching, and feeding cycles (Poff *et al.,* 1997), and channelization of streams can interfere with these natural fluctuations in flow throughout the seasons. Flow rates disrupted by irrigation practices for agricultural use, also result in streams no longer experiencing the same depositional and erosion patterns that create the diverse linear habitats that are present in many forested headwater streams (Statzner and Higler, 1986). Channelization is also associated with reduced ecological diversity, as BMI biomass and fish biodiversity are both positively associated with increasing stream sinuosity (Karr and Schlosser, 1978). Loss of habitat diversity with channelization has also been shown to influence functional diversity. For example, Berkman and Rabeni (1987) observed that loss of riffle habitat resulted in reduced abundances of fish from feeding guilds that consumed benthic insectivores and herbivores. Clearly, a variety of factors contributes to changes in the stream biota, and yet these factors differ between streams, depending upon both the type and intensity of agricultural land (Larsen *et al.,* 2009). There are also stressors to the stream biota that are not a direct consequence of agricultural land use, but rather stem from the absence of riparian vegetation that is often removed as part of agricultural land development.

1.2 Riparian Corridors

Riparian corridors are the interface between aquatic and terrestrial ecosystems and perform an array of critical functions that determine the physicochemical and ecological conditions of adjacent aquatic ecosystems (Naiman and Decamps, 1997; Naiman *et al.,* 1988; Naiman *et al.,* 1993). For example, forested riparian corridors reduce the amount of solar radiation reaching the stream, thus reducing average stream temperatures, as well as fluctuations in temperature (Osborne and Kovacic, 1993). The riparian canopy also regulates the amount of primary production within the stream ecosystem through the interception of photosynthetically active radiation (PAR). The riparian canopy also acts as a protective barrier by minimizing wind speeds and precipitation near ground level, while increasing long-wave radiation at the surface, resulting in reduced fluctuations in air temperature, and a more stable thermal and moisture regime (Moore *et al.,* 2005). Plants in the riparian corridor provide an abundance of allochthonous energy inputs into the stream ecosystem. Indeed, most of the energy in forested headwater streams is derived from the riparian vegetation in the form of organic matter (Karl and Schlosser, 1978). If this material does not leach out dissolved organics into the stream, then it, along with all of the microorganisms living on it, is shredded, consumed, and digested by benthic macroinvertebrates, which egest a fine organic material that can be further utilized by other invertebrates. As well as providing an integral energy source to the stream ecosystem, riparian vegetation can alter stream habitat and stream flow through the addition of coarse woody debris (Naiman and Descampes 1997). Cumulatively, the effects of the riparian vegetation serve as a key control towards the ecological structure and function of stream ecosystems.

Linkages between the presence of riparian vegetation and the structure and function of stream communities are well documented, particularly in regions where forest dominates riparian corridors. Numerous studies point to the consequence of replacing native forests with agricultural land cover types, which is often a reduction in BMI diversity (Quinn and Hickey, 1990). Riparian forests provide shade to the stream, resulting in cooler water temperatures and increased habitat cover, which have been shown to be positively associated with increases to diversity in the benthic macroinvertebrate community, including the more sensitive EPT (Ephemeroptera, Plectopera, Trichoptera) taxa (Rios and Bailey, 2006). Cooler temperatures have also been connected to increases in abundance of coldwater fish species such as Salmonids (Clews and Ormerod, 2010). Naiman *et al.* (1993) also pointed out that sensitive BMI abundances increased in streams with riparian forests. Rios and Bailey (2006) and Yates *et al.* (2007) also found that BMI taxa that were more tolerant of organic pollution, increased in the absence of riparian cover and were associated with increased agricultural land use in the riparian corridor. Likewise, Clews and Ormerod (2010) noted a decrease in taxa richness with the loss of riparian vegetation, as well as decreased abundance of more sensitive BMI families (e.g. Perlodidae, Sericostomadidae) and Salmonids (e.g. Atlantic Salmon, Brown Trout). Lenat and Crawford (1994) also observed declines in fish species' richness, biomass, and absence of intolerant species with the removal of riparian vegetation. In addition to impacts on consumer communities, the riparian canopy has also been shown to regulate the amount of primary production within the stream ecosystem. For example, a field experiment by Kiffney *et al.* (2004) found that light limitation was the dominant control of variation in periphyton biomass. Shading effects on periphyton biomass in streams associated with pastoral agriculture have been particularly important when seeking a leading cause to changes within the BMI community, and has been suggested as the most critical factor contributing to Chironomid density (Quinn *et al.,* 1997). Thus, the loss of riparian forest has important indirect effects on stream biota through the mediation of both light and thermal radiation to streams (Allan, 2004). Removal of riparian forests also reduces allochthonous inputs of organic matter to the stream, limiting the availability of coarse particulate organic matter in the form of leaf litter and woody debris (Naiman *et al.,* 1993). The deprivation of organic matter coupled with the increase in light availability can shift the base of the stream food web from one dominated by detritivores to one that is largely driven by autotrophs (Vannote, 1980). This shift in basal food resource can have a subsequent effect on the composition of primary consumers as collector-gather taxa are replaced by grazing taxa (Clews and Ormerod, 2010; Vannote, 1980). Sweeney (1993) concluded that the absence of riparian forests along stream channels may be the single, most important factor linked to human influence, that affects both the structure and function of BMI communities.

Riparian corridors have been found to both act as an effective buffer between agricultural activities in the catchment, and receiving aquatic ecosystems such as streams (Harding *et al.,* 1998; Verhoeven *et al.,* 2006; Allan, 2004). Indeed, riparian corridors appear to be particularly effective at intercepting agricultural pollutants, such as nutrients and sediments, being transported in surface and subsurface runoff (Osborne and Kovacic, 1993; Richardson *et al.,* 2012; Jones *et al.,* 2010). Riparian corridors remove and retain pollutants through processes of sedimentation of suspended solids; adsorption and fixation onto soil particles; precipitation and flocculation of both soluble and particulate nutrients; consumption and metabolism of organics by plant and microbial communities; and microbial conversion of nutrients to gases that are released to the atmosphere (Sharitz *et al.,* 1992). Sediments, and nutrients bound to sediments, are captured through sedimentation processes induced by riparian vegetation increasing surface roughness, thereby slowing the flow of runoff causing sediment particles to settle out of overland flows. Soluble nutrients (e.g., phosphorus and nitrogen) in subsurface flow can be taken up by the microbial communities associated with the root systems of riparian vegetation, increasing their residence time and thus, levelling off their peak concentrations entering streams during rain events (Verhoeven *et al.,* 2006). Further, soils in riparian corridors are often hotspots for denitrification, resulting in increased soil moisture levels, and generating anaerobic soil conditions that reduce nitrogen forms to nitrogen gas, that is then emitted to the atmosphere (Verhoeven *et al.,* 2006). Soluble nutrients will also be adsorbed by both the organic and inorganic soil particles, as well as taken up by the riparian plants themselves, preventing these nutrients from reaching the stream (Osborne and Kovacic, 1993). Past studies have shown, however, that the filtering effects of riparian vegetation is strongly associated with the width of the corridor. For example, riparian corridors of 19 to 50 m in width were associated with removal of 68% to 100% of nitrogen transported in runoff, respectively

(Petersen *et al.,* 1992). Likewise, Sweeney and Newbold (2014) reported that riparian forest corridor widths of at least 30 m or more were required to achieve significant results in the interception of nitrate from the catchment. They also found that riparian forest corridor widths between 10 and 30 m are capable of trapping approximately 65% (low 10 m end) and 85% (high 30 m end) of sediments (Sweeney and Newbold 2014). Riparian corridor widths of 30 m or greater have also been reported as necessary for maintaining in-stream fish communities (Barton 1985). Greater widths were recommended by Thomas *et al.* (2016), who concluded that planting riparian deciduous forests that exceed 60 m in width along temperate headwater streams, would modify BMI community functionality, increase biomass, and also increase allochthonous inputs, that could all lead to potentially increased resilience within the stream biota. Maintenance of sufficiently wide, vegetated riparian corridors along streams in agricultural catchments, may thus mitigate harmful effects of agricultural land use, and enable the maintenance of ecosystem structures and functions that are similar to pristine, reference streams.

1.3 Spatial Scales

In his seminal paper, Hynes (1975) coined the phrase, “the valley rules the stream”, thus becoming one of the first to recognize the connection between the characteristics of the stream and its associated watershed. The work by Hynes was followed up by that of others further identifying the hierarchical structure of stream ecosystems (e.g., Frissell *et al.,* 1986), anchoring stream ecosystem components in a nested fashion. This improved framework for studying the connections between landscape attributes and aquatic ecosystem conditions, coupled with the advent of geographic information systems (GIS), has led to an explosion of research that has clearly established the validity of Hynes' central tenet. Included in this body of research have been several studies that have established the relative strength of association between the stream ecosystem and different spatial scales of the catchment, including the riparian corridor and upland catchment areas (e.g., Allan and Johnson, 1997; Yates and Bailey 2010; Waite *et al.,* 2010; Strayer *et al.,* 2003; Townsend *et al.,* 2003). The impetuses for this work largely stem from Allan and Johnson (1997), suggesting that spatially conceptualizing the stream using a hierarchical approach of landscape scales, will allow for a better understanding of the complex interactions between landscape and stream. For example, Allan *et al.* (1997) point out that specific processes, such as organic inputs into the stream, will depend more upon a localized riparian corridor scale, but others, such as hydrologic regimes, that directly affect sediment delivery and channel conditions, are more directly influenced by how water is delivered over a larger landscape scale. Indeed, several recent studies have found an inverse exponential relationship between the distance of land use from a watercourse and its influence on metrics of stream condition (e.g. Van Sickle and Johnson, 2008; Peterson *et al.,* 2011; Sheldon *et al.,* 2012; Yates *et al.,* 2014), although there are exceptions (e.g. Roth *et al.,* 1996). For example, agricultural land use within a narrow riparian stream corridor was most strongly associated with a biotic integrity index of fish (Van Sickle and Johnson, 2008). Likewise, measures of water quality, fish, and macroinvertebrate communities were most strongly associated with forested areas close to the stream (Sheldon *et al.,* 2012). Consequently, there is strong empirical evidence supporting the idea that the riparian corridor is disproportionately important to the condition of the stream relative to surrounding catchment areas. However, these past studies have been designed to identify the landscape scale with the strongest association with stream biota. What has yet to be elucidated are the minimum amounts of intact riparian vegetation required to maintain the buffering effects provided by the riparian corridor. Moreover, the amount of anthropogenic development within the upland areas of the catchment at which the buffering capacity of riparian vegetation is exceeded, has not been clearly established. Answers to these questions are critical as managers struggle to make informed decisions aimed at balancing increasing land use pressures with maintenance of river ecosystem health.

1.4 Benthic Macroinvertebrate Communities

Bioassessment and stream monitoring programs have long applied the concept of using an indicator describing ecological structure or function to assess stream conditions (Young and Collier, 2009; Death *et al.,* 2009; Young *et al.,* 2008). Benthic macroinvertebrate (BMI) assemblages are the most common and widely used biological indicator in stream monitoring programs throughout the world (Waite, 2014). BMI are widely applied indicators because they are ubiquitous in stream environments, are easily collected, have an ecology that is generally well understood, are sensitive to many common stressors, and have widely accepted methods for analysis and interpretation of assemblage status (Yates *et al.,* 2014). Consequently, BMI assemblages now form the foundation of stream biomonitoring programs around the world (Death *et al.,* 2009), including in Canada (Reynoldson *et al.,* 1997). To date, most biomonitoring programs have focused on the use of structural measures of BMI assemblages describing taxonomic diversity and composition such as total taxonomic richness, combined richness or relative abundance of specific taxon groups, including Emphemeroptera, Plecoptera, and Trichoptera (EPT) taxa and Diptera taxa, amongst other metrics (Pollard and Yuan, 2010). More recently, there has been increasing interest in expanding biomonitoring programs to include the use of functional measures, often called traits, of the BMI assemblage (Culp *et al.,* 2011).

Traits are the morphological, physiological and ecological attributes of organisms or their taxonomic group, describing their physical and functional roles within an ecosystem (Baird *et al.,* 2008). The rationale for applying traits to biomonitoring is grounded in Southwood's (1977, 1988) habitat templet model. Southwood's model predicts that where environmental conditions (i.e., habitat) are similar, the composition of species' traits should converge, even where the taxonomy of the community varies across larger biogeographical regions (Poff *et al.,* 2006). Based on this idea of environmental "filtering" (sensu Poff *et al.,* 2006), traits should be a predictor of how disturbances or stressors may alter a particular community (Menezes *et al.,* 2010). Several recent studies have applied trait-based approaches towards assessment of anthropogenic activities (Statzner and Beche, 2010; Poff *et al.,* 2006; Doledec *et al.,* 2006). For example, Statzner *et al.*, (2005) used multiple BMI traits (e.g. body size; body form) as indicators for anthropogenic stresses on large European rivers. They found that individual traits could be used to differentiate between different types of anthropogenic impacts across large geographical areas. Likewise, Young and Collier (2009) found a linear inverse relationship between BMI sensitivity to organic pollution and a land use stress gradient. Yates and Bailey (2011) conducted a study where they observed functional trait metrics to share the weakest relationships with gradients of human activity, but pointed out that researchers have indicated that functional BMI traits do respond to anthropogenic activities (Statzner *et al.,* 2001) and have even outperformed compositional traits in their sensitivity to changes in the ecosystem (Dolodec *et al.,* 2006). In fact, Dolodec *et al.* (2006) specifically found functional traits responding to intensive agricultural practices when they observed population resilience traits (e.g. short generation time; asexual reproduction) becoming more prevalent. These studies thus indicate that traits could be an effective approach to detecting human impacts from agricultural activities (Statzner and Beche, 2010; Culp *et al.,* 2011; Lange *et al.,* 2014).

1.5 Ecological Thresholds

Groffman *et al.* (2006) define a threshold as the point at which there is an abrupt change in an ecosystem quality, property or phenomenon, or where small changes in an environmental driver (e.g., agricultural land use) produce large responses in the ecosystem (e.g., BMI community composition). Thresholds may take on a continuous functional relationship, where the relationship changes to a positive or negative one, either from a neutral state or from an opposing positive or negative state. Thresholds may also be discontinuous, where the relationship experiences a sudden shift from one function to another (step function) (Dodds, *et al.,* 2010). There are numerous statistical methods available to detect thresholds. For example, there are model-fitting approaches that include a variety of piecewise regression models (sometimes referred to as segmented regression) (Daily *et al.,* 2012) or varying polynomial spline models (Andersen *et al.,* 2008). Alternatively, there are data-partitioning methods that seek to identify thresholds by minimizing residual variance through repetitive partitioning. Lastly, thresholds can be identified through data partitioning using nonparametric change point analysis (Evans-White *et al.,* 2009; Andersen *et al.,* 2008). Chaudhurri and Marron (1999) developed a hybrid approach that utilized both model-fitting and data-partitioning, based on the second derivative of polynomial regressions, called SiZer (significant zero crossings). Soon after, SiZer began to find use in ecological threshold detection among BMI communities (Daily *et al.,* 2012). For example, Sonderegger *et al.* (2009) demonstrated how SiZer could be used to detect ecological thresholds along a single explanatory variable by assessing mayfly abundance over time in the Arkansas River. Clements *et al.* (2010) also demonstrated the application of SiZer, in finding distinct threshold responses of BMI communities, including multiple thresholds, as well as serving as an exploratory analysis for assessing datasets for potential thresholds at varying resolutions. However, despite the number of methods that exist for threshold analysis, further research is needed to establish the conditions under which different techniques are most robust (Dodds *et al.,* 2010). Indeed, assessing the complexity of ecological systems means that thresholds are often none other than spurious detection or random fluctuations in the data (Andersen *et al.,* 2008).

Selection of an analysis for identifying thresholds must therefore take into consideration the specific environmental and experimental constraints to reduce the probability of spurious detections (Daily *et al.,* 2012). Moreover, it has been recommended that researchers apply multiple techniques, enabling results to be further validated through cross comparisons (Dodds *et al.,* 2010).

Environmental thresholds are increasingly being used to inform land management decisions, as a threshold can serve as an effective target that will ensure protection of a known level of ecosystem integrity (Dodds *et al.,* 2010; Clements *et al.,* 2010; Hilderbrand *et al.,* 2010). Moreover, identifying thresholds can be applied towards the establishment of stream ecosystem reference conditions within anthropogenically-dominated landscapes where "pristine" conditions are absent, by providing empirically based and ecologically relevant criteria for identifying best available landscape conditions (Toms and Villard, 2015). To date, land use thresholds have typically been sought at the catchment level (Allan, 2004). Indeed, there has been substantial research linking land use thresholds to biotic integrity of stream ecosystems at the catchment scale, although most of these studies have focused on urban environments (e.g., King and Baker, 2010; Hilderbrand *et al.,* 2010), and less so in agricultural landscapes. A study by Utz *et al.* (2009) is perhaps the most spatially extensive determination of agricultural land use thresholds, looking at several regions across the conterminous United States. Results of the Utz *et al.* (2009) study indicated that the percent coverage of agricultural land use inducing a threshold response in key BMI taxonomic groups, varied from as low as 21% coverage in some regions to greater than 80% coverage in others. However, knowledge of the precise thresholds at which agricultural land use at either the catchment or riparian corridor scale overwhelms mitigating effects of riparian vegetation is still lacking.

2.0 Research Questions and Hypotheses

The purpose of this study is to describe the amount and nature of variation in the structure and function of BMI assemblages in Southwestern Ontario streams exposed to varying patterns of agricultural land use at the riparian corridor and catchment scales and to assess if the location and amount of agricultural land use is associated with variation in BMI assemblage structure.

I will address this problem by answering the following three questions:

(1) Is there an association between stream BMI assemblage condition and variation in agricultural land use at the catchment scale when the riparian corridor scale has minimal agricultural cover?

Hypothesis:

BMI assemblages will exhibit a threshold response to variation in agricultural land use at the catchment scale when the riparian corridor scale has minimal agricultural cover, such that the BMI assemblage condition will be constant prior to the threshold and decline following the threshold. The threshold will coincide with the amount of agricultural land use at the catchment scale that exceeds the riparian corridor's capacity to mitigate agricultural effects (Figure 2.1a).

(2) Is there an association between stream BMI assemblage condition and variation in agricultural land use at the catchment scale when the riparian corridor scale is primarily agricultural?

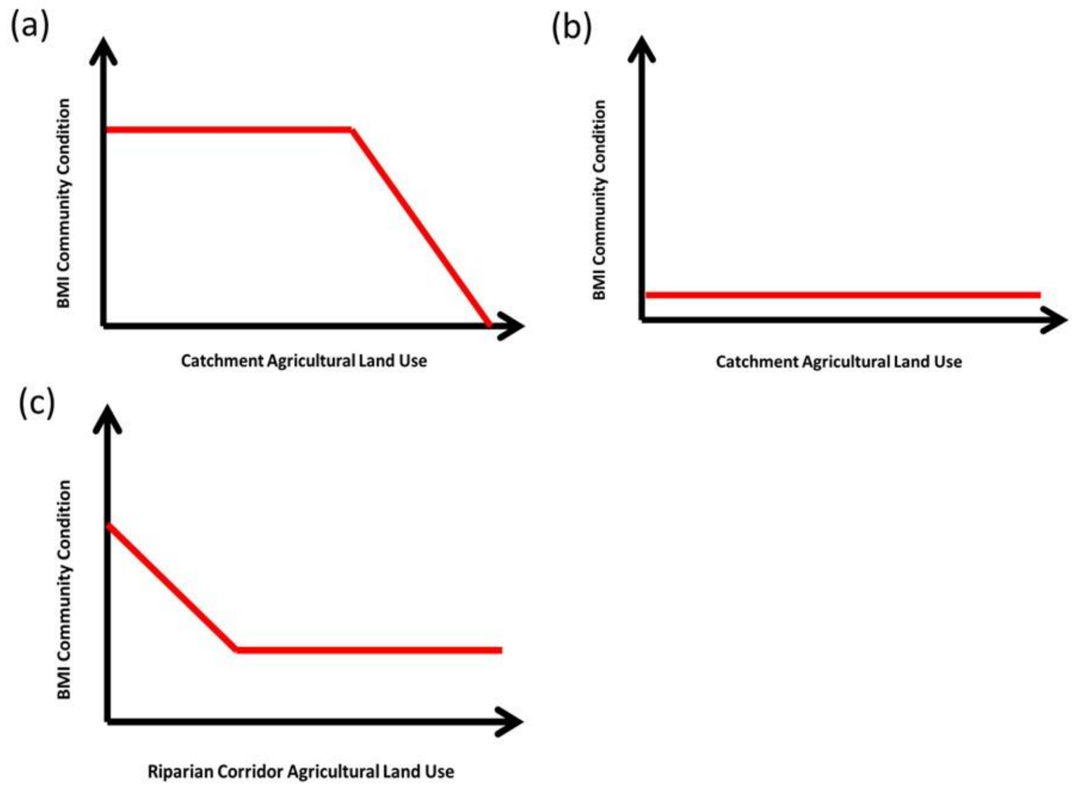


Figure 2.1 (a) Hypothesis #1 – Threshold response when agricultural land use at the catchment scale exceeds mitigating capacity of a minimally disturbed riparian corridor. (b) Hypothesis #2 – No association between variation in BMI assemblages and agricultural land use when the riparian corridor is disturbed. (c) Hypothesis #3 – Threshold response when agricultural land use within the riparian corridor is minimal enough that it can mitigate the effects of high percentages of agricultural land use within the surrounding catchment.

Hypothesis:

There will be no association between variation in BMI assemblages and agricultural land use in streams exposed to mostly agriculture at the riparian corridor scale. The riparian corridor has been disturbed beyond a point where mitigation of the effects of agricultural land use on the stream ecosystem can occur (Figure 2.1b).

(3) Is there an association between stream BMI assemblage condition and variation in agricultural land use at the riparian corridor scale when the catchment scale has primarily agricultural land cover?

Hypothesis:

BMI assemblages will exhibit a threshold response to variation in agricultural land use at the riparian corridor scale when the catchment scale has primarily agricultural land cover, such that the BMI assemblage condition will decline with increasing agricultural land cover prior to the threshold and be constant following the threshold. The threshold will coincide with the amount of agricultural land use that exceeds the riparian corridor's capacity to mitigate agricultural effects from agricultural activities at the catchment scale (Figure 2.1c).

3.0 Methods

3.1 Study Area

I conducted my study in southwestern Ontario, Canada (Figure 3.1). The region is surrounded by North America's Laurentian Great Lakes. Regional temperatures reach average highs approximating 27°C in July and average lows in January of -10°C (Environment Canada and Climate Change, 2016). Average annual precipitation is approximately 1025 mm with monthly averages ranging from 35 mm to 163 mm (Environment Canada and Climate Change, 2016). The physiography of southwestern Ontario is comprised of a wide assortment of glacial deposits overlying carbonate rich Paleozoic bedrock (Yates and Bailey, 2011). Retreating ice sheets have left a landscape characterized by an assortment of moraines, glacial outwash plains, and hills with steep irregular slopes. Physiography in the northern areas of the region is characterized by generally flat lands, consisting of poorly drained silty, clay-rich soils. Central sectors consist of an assortment of moraines, hills, gravel outwash plains, and large sand and gravel deposits. The southern and most western portions are dominated by heavy clay soils that drain poorly (MNDM, 2016; Grand River, 2014). Land use in the region is characterized by a patchwork of deciduous forests in an otherwise agriculturally dominated landscape, largely comprised of a mixture of rowcrops such as corn and soybean, and high-density livestock farms including beef, dairy, and poultry. Hydrologic regimes have been modified by widespread tile drainage and channelization (Yates and Bailey, 2011).

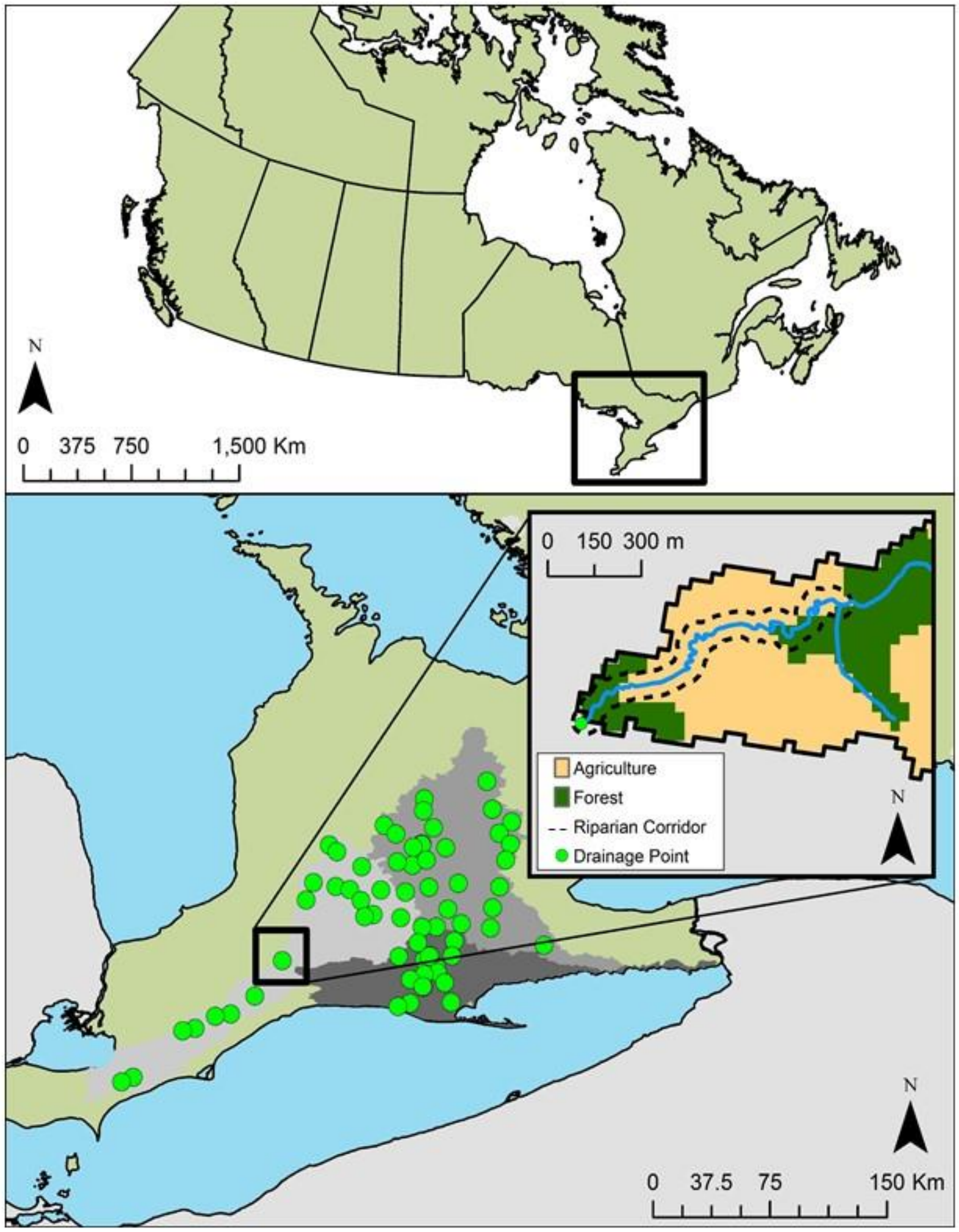


Figure 3.1 Location of study area in the southwestern portion of the province of Ontario. Green circles indicate the stream sites found within the Grand River, Thames River, and Long Point watersheds.

3.2 Study Design

This study assessed the responses of benthic macroinvertebrates to agricultural land use at two spatial scales; the catchment and riparian corridor scales. For the purpose of this study, the catchment scale was defined as the entire land area draining into a specified drainage point (Figure 3.2a). The riparian corridor scale was defined as a 40 metre-wide corridor on either side of the stream, running the entire length of the stream segment upstream of the drainage point (Figure 3.2b) (Banuelos and Yates, 2013). A stream segment was defined as a portion of stream situated between the specified downstream drainage point and the next upstream confluence.

The current study assessed benthic macroinvertebrate responses to three different scenarios of land use patterns within the catchment and riparian corridor areas. Scenario #1 included streams with minimal agricultural land use in the riparian corridor and a gradient of agricultural land cover in the surrounding catchment. In Scenario #2, streams are characterized by a range of agricultural land cover at the catchment scale but have a riparian corridor dominated by agricultural land use. Streams in Scenario #3, are exposed to a gradient in agricultural land cover at the riparian corridor scale, whereas the catchment scale is dominated by agricultural land cover.

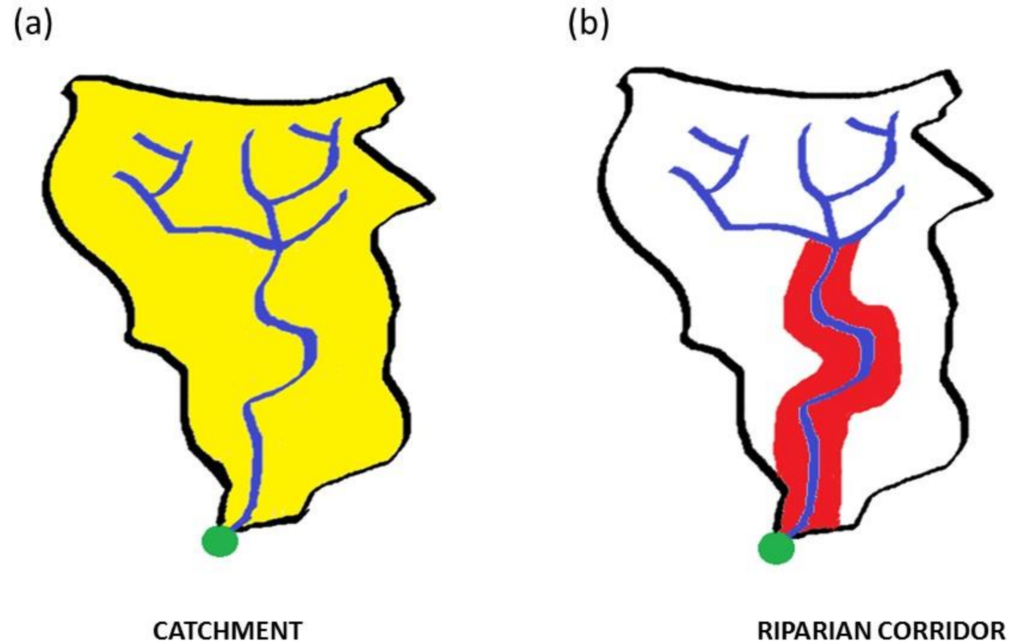


Figure 3.2 (a) An example of a catchment. Note that the area highlighted in yellow is the entire landscape that will drain all water (shaded in blue) within it through to the identified drainage point which is identified by the green shaded sphere. (b) An example of a riparian corridor highlighted in red. The riparian corridor includes 40 metres on either side of the stream.

3.3 Site Selection

Sites sampled in this study were selected from a database that included hundreds of sites in the Grand River Watershed generated by Banuelos and Yates (2013). Banuelos and Yates (2013) used a 1:50,000 scale river network to identify drainage points at each stream confluence. Catchment and riparian corridor boundaries were delineated for each drainage point using a 20 metre resolution digital elevation model (DEM). The delineated catchments and riparian corridor were intersected with a collection of GIS layers describing both anthropogenic (i.e., land use types) and natural (i.e., topography and surficial geology) landscape features (Banuelos and Yates, 2013).

I applied an eight-step multi-criteria approach to select sites from the database generated by Banuelos and Yates (2013) that resulted in maximum variation in agricultural land use at either the catchment or riparian corridor within each of the three scenarios, while minimizing variation in natural catchment conditions (Table 3.1). This analysis resulted in 22 sites being retained that were similar in catchment area and had predominately sandy soils. To increase the number of sites and the range of the agricultural gradients in each scenario, an additional 46 southwestern Ontario sites were incorporated from a previous study by Yates and Bailey (2010). Each of the additional 46 sites met the site selection criteria listed in Table 3.1. Sites were located within the Grand, Thames and Long Point Bay area drainage basins (Figure 3.1). Table 3.2 illustrates the land use patterns that were established for all three scenarios in this study. Scenario #1 had 24 sites with no more than 13% agricultural land cover in the riparian corridor.

Table 3.1. Multi-criteria approach in site selection that maximized variation in agricultural land use in the specified spatial scale, and minimized variation in natural catchment conditions.

Step 1	Catchments containing urban land use were removed.
Step 2	Remaining catchments with areas between 5 km^2 and 15 km^2 were selected to ensure a large enough catchment to provide perennial streams, yet small enough to ensure general comparability in attributes associated with catchment area (e.g., discharge and channel size).
Step 3	Surficial geology among the remaining catchments was classified to eliminate the confounding effects of surficial geology with agricultural land use.
Step 4	Variation of agricultural land use for each classification of surficial geology was assessed. The dominant surficial geology texture that was most strongly associated with the widest gradient of agricultural land use was determined to be sand. All other sites with differing stream bed types were removed.
Step 5	Remaining stream sites were placed into their respective land use scenarios ensuring that land use gradients at each of the spatial scales were being represented by the widest breadth of agricultural land use possible. Ensuring there were more than 20 sites per scenario, the resulting control scales (defined range of agricultural land use) for Scenario #1 were 15% or less agricultural land use in the riparian corridor; 75% or greater agricultural land use in the riparian corridor in Scenario #2; and 80% or greater agricultural land use in the catchment in Scenario #3.
Step 6	Nested sites (i.e., sites connected by a stream that share the same catchment) were identified. Among those nested sites, the one that was most suitable for sampling, provided optimal accessibility, and demonstrated all previous criteria, was chosen. Nested sites were eliminated to prevent sampling members of the same BMI community potentially migrating between sites.
Step 7	Candidate sites were evaluated by field visits to eliminate those that did not present the land use or substrate conditions indicated by the database or were associated with hazards or safety issues that would inhibit sampling.
Step 8	The “Channel Alteration” scores from the United States Environmental Protection Agency’s rapid habitat assessment protocol specific to low gradient streams (Plafkin *et al.,* 1989) were assessed. Any sites containing alteration scores less than 10 were removed. This was done to account for channelization as a possible confounding variable to the effects of agricultural land use on BMI.

Table 3.2. Percentage of land cover area that is agricultural land use at both the riparian corridor and catchment scales for all three scenarios.

		Site Count	Mean	Standard Deviation	*CV	Median	Minimum	Maximum
Scenario #1		24						
Riparian	**≤15%Ag**		4%	4%	103%	4%	0%	13%
Catchment	**Gradient**		73%	10%	14%	75%	48%	85%
Scenario #2		20						
Riparian	**≥75%Ag**		89%	9%	10%	91%	75%	100%
Catchment	**Gradient**		81%	13%	16%	84%	44%	96%
Scenario #3		43						
Riparian	**Gradient**		49%	31%	63%	50%	0%	95%
Catchment	**≥80%Ag**		87%	4%	5%	87%	80%	96%

*Note CV = Coefficient of Variation.

Scenario #2 had 20 sites that had greater than 75% agricultural land cover in the riparian corridor. Where both Scenario #1 and Scenario #2 had a gradient of agricultural land cover in the catchment, Scenario #3 had 43 sites with ≥80% agricultural land cover in the catchment and then a gradient of agricultural land cover in the riparian corridor. However, since Scenario #3 assesses an agricultural land cover gradient in the riparian corridor, 7 of its sites were used in Scenario #1 and another 12 sites also used in Scenario #2.

3.4 Field Sampling Protocol

All benthic macroinvertebrate (BMI) communities included in this study were sampled using the CABIN (Canadian Aquatic Biomonitoring Network) protocol (Reynoldson *et al.,* 2012). I conducted the sampling prior to the 46 stream sites that were added later from the Yates and Bailey (2010) study. The process uses a D-frame net equipped with 400 micron (µm) mesh, as the sampler disturbs the stream substrate, traveling upstream over a three minute period. All habitats were sampled in proportion to their occurrence within the defined sampling reach (i.e. six times the bankfull width).

Material collected during the kick sample was elutriated at the site to separate inorganic substrate from invertebrates and organic material. All invertebrates and organic material were then transferred to a sampling bottle, adding 10% buffered formalin as a preservative. Following transport to the lab and a minimum week fixation period, all samples were transferred from their formalin preservative to 75% ethanol. Samples were homogenized prior to being placed into a Marchant Box containing 100 grid cells. Subsampling was carried out by random selection of cells and enumeration of all bugs in the cell until a minimum of 5 cells or 300 individuals were subsampled, whichever required more of the sample to be processed. All enumerated taxa were identified to the lowest taxonomic unit practical, usually genus or family. To ensure taxonomic resolution remained consistent for all stream sites, a rule similar to that used by Vlek *et al.* (2004) was applied. Under this rule, if more than 20% of individuals in a taxon were identified to family level, than those individuals from the lower genus level would be elevated to the family level. In contrast, if less than 20% of individuals in a taxon were identified to the family level, than only individuals of that taxon that were identified to the genus level would be retained for analysis. However, in cases where less than 20% of individuals were at the family level, 100% of the sites where that family was collected were required to have at least one individual at the genus level for the taxon to be adjusted to the genus level, thus ensuring that diversity counts at individual sites were not artificially reduced through taxonomic adjustments.

The habitat of each sampling reach was assessed using the United States Environmental Protection Agency's rapid habitat assessment protocol (Barbour *et al.,* 1999) specific to low gradient streams. This habitat assessment serves to qualitatively index 10 habitat traits on a scale of 0 to 20. These traits include epifaunal and pool substrate; pool variability; sediment deposition; channel characteristics; bank stability; and riparian vegetation (Appendix A).

3.5 Data Analysis

The BMI assemblages for each site were summarized by calculating 12 metrics (Appendix B) describing aspects of taxonomic composition, diversity; and 4 types of functional traits (i.e., feeding groups, habitat use, life history strategy, and tolerance). Compositional metrics were %EPT (EPT – Ephemeroptera, Plecoptera, and Trichoptera) and %Diptera, and diversity metrics were total community richness, number of EPT taxa, and number of Diptera taxa. For the functional traits, there were two feeding (i.e., %Herbivores and %Shredders), two life history strategy (i.e., %Multivoltinism and %Small Body Size), two habitat (i.e., %Clingers and %Burrowers), and one tolerance (i.e., Hilsenhoff Family Biotic Index) metric calculated. Metrics describing both structural and functional attributes were calculated to account for the wide spectrum of ecological characteristics and associated variation in sensitivity to land use within the invertebrate assemblage.

Following the calculation of all 12 metrics, descriptive statistics (e.g. mean, standard deviation) were calculated for all 10 habitat traits and the 12 BMI metrics. The aim of this study was to identify thresholds in the associations between the BMI communities and agricultural land use. To accommodate this goal, the BMI metrics were transformed so that each metric more closely approximated a normal distribution and thus improves the researcher’s ability to visualize emerging patterns in the data whenever there is a substantive change (McDonald, 2014). Base-10 log transformations were applied to all diversity and tolerance metrics, whereas square-root transformations were applied to percent-based metrics describing community composition and functional trait metrics (McDonald, 2014).

Least squares linear regression analyses using SYSTAT 13 (SYSTAT, 2015) were performed to identify associations between variation in agricultural land cover within the control scale and the BMI community metrics for each of the three scenarios. Although this variation in the control scale was small (i.e, <15% agricultural land use) among catchments within each of the three scenarios, there was the potential that land cover differences within the control scale could be influencing BMI community metrics, and would thus confound the associations with agricultural land use at the landscape scale that was comprised of the agricultural land use gradient. Least squares linear regressions assessed the association between each BMI metric and the controlled agricultural land use at a significance level of $p < 0.1$. I increased the significance level from the more widely implemented $p < 0.05$ to increase the likelihood of identifying and consequently removing any effect of agricultural land use within the control scale. Residuals from the resulting significant associations were retained as the corrected biological metric for use in threshold detection analyses.

A variety of statistical methods can be used to detect thresholds (Dodds *et al.,* 2010) but their statistical performance remains poorly understood (Daily *et al.,* 2012). It is thus increasingly common practice for ecologists to implement multiple statistical methods when analyzing data for thresholds to increase confidence in the observed outcome (Daily *et al.,* 2012; Dodds *et al.,* 2010; Andersen *et al.,* 2009). Potential thresholds in the response of the 12 BMI community metrics to agricultural land use patterns in each of the three scenarios were thus analyzed using two exploratory statistical approaches; “significant zero crossings” (SiZer) (Sonderegger, 2015) and “segmented regression” (SegReg) (Oosterbaan, 2017). My approach enabled me to perform a comparison of outcome consistency between two different statistical methods.

SiZer applies a non-parametric smoother to the agricultural land use-BMI metric association (Clements *et al.,* 2010), and then analyzes the derivatives of the smoothed curve to detect potential thresholds (Sonderegger *et al.,* 2009). A visual image referred to as a map of the first derivative is produced (Figure 3.3). A potential threshold will be marked where the function’s derivative changes significantly ($p < 0.05$) (Clements *et al.,* 2010), and the derivative will be categorized as positive (blue), negative (red), or zero (purple) (Sonderegger *et al.,* 2009). To examine the first derivative for potential thresholds, the map is interpreted by reading chosen bandwidths on the y-axis from left to right along a horizontal plain through the derivative map. The SiZer test utilizes locally weighted polynomial regression that serves the purpose of smoothing the data with 95% confidence intervals (Clements *et al.,* 2010). The smoothing parameter “h,” referred to as the bandwidth, controls the amount of smoothing of the weighted function (Chaudhuri and Marron, 1999). There is no single best method for selecting a range of bandwidths for optimal data smoothing (Sonderegger, 2015). Too high a bandwidth will over-smooth the data and a potential threshold will be missed. If the bandwidth is too small, the weighted function will be influenced too often by too few data points and could reveal false thresholds (Sonderegger *et al.,* 2009). Interpretation of the first derivative map is achieved through examination of a range of bandwidths, making note of significant changes, and determining if such evidence coincides with a known ecological concept (e.g., %Shredders should decrease with increasing agricultural land use in the riparian corridor) (Clements *et al.,* 2010). As I was not interested in BMI community variation with each percentage change in agricultural land use, but rather what I will define as major thresholds occurring over a widespread gradient of agricultural land use, I chose to assess the bandwidth between -0.5 and 0.5. The selected range of bandwidth was based on advice by Sonderegger *et al.* (2009) who suggested this range for detection of large-scale thresholds when conducting similar analyses to detect long-term temporal changes in the recovery of mayfly populations following stream restoration. Thus, potential thresholds in my study were noted whenever the first derivative map displayed a change in colour at a specific bandwidth that fell between bandwidths -0.5 and 0.5. Wherever there was the most notable change, that particular bandwidth would be recorded. Linear relationships could also be interpreted if the entire range of bandwidths between -0.5 and 0.5 read through a solid block of “blue” (curve is significantly increasing with respect to the agricultural land use-metric association) or “red” (curve is significantly decreasing). If the entire range of bandwidths displayed as “purple”, then I interpreted there to be no association between agricultural land use and the specified metric. All SiZer analyses were conducted using R software (Sonderegger, 2015; R Core Team, 2015).

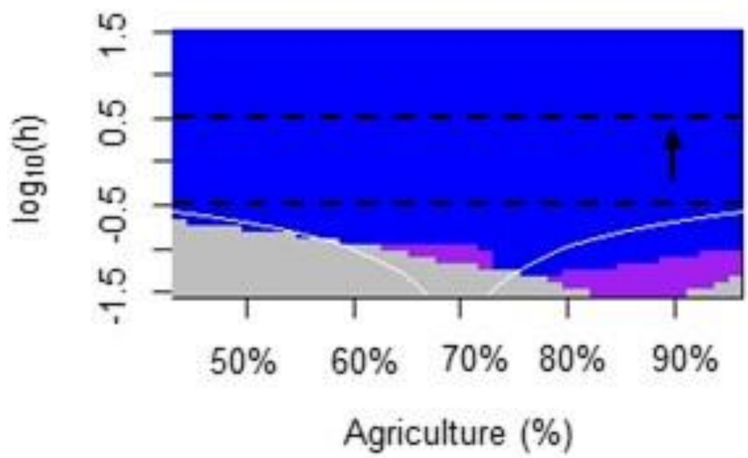
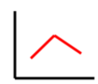
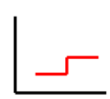


Figure 3.3. An example of a map of the first derivative as displayed in SiZer. The black dashed lines represent the range of bandwidths the researcher is interested in observing. One reads the image from left to right along the horizontal plane selected by the specific bandwidth (h) one chooses to read. If the blue shade appears along that plane, that indicates a positive derivative. The arrow is only further acting as a reminder of the direction of the derivative. This specific example does not include any red shading, but if it did, a red shade would indicate a negative derivative. A purple shade indicates no direction or zero. The grey shade indicates insufficient data at that point.

Segmented regression, sometimes referred to as piecewise regression is a regression analysis where the independent variable is partitioned into intervals around thresholds in the data, and each interval is then fitted with a separate line segment (Oosterbaan, 2017; Nordin *et al.,* 2009). In this study, I conducted segmented regression analyses using SegReg 1.7.0.0 (Oosterbaan, 2017). The SegReg program assigns the associations between the independent and dependent variables to one of seven function types (defined as function 0 through to function 6) (Table 2.5). These seven functions represent different possible structures of the relationships between the independent and dependent variables, including no relationship (Type 0); a line relationship (Type 1); and 5 different types of thresholds that the dependent variable could display in response to the independent variable. Types 2 through 4 represent threshold responses where the rate of change in the dependent variable changes significantly about the threshold. In contrast, Types 5 and 6 represent a threshold describing a state change in the data where the mean of the response variable differs about the identified threshold. SegReg establishes the function type that best fits the relationship between dependent and independent variables through a three-step process. First, the program assesses the strength (measured as R^2 value) of a linear association between the independent and dependent variable. This relationship is then used as a null model against which Types 2 through 6 are compared. If the fit of one or more of the threshold functions is more significant than the linear model, than the threshold function type with the largest coefficient of explanation (E) is retained. If neither a threshold function or the linear function results in a significant association between the dependent and independent variables then Type 0 is assigned, indicating no significant relationship. Significance is assessed using F-tests and 90% confidence interval (Oosterbaan, 2005). The selected function type and threshold results when the SegReg program has iteratively tested all possible thresholds within the association between the independent and dependent variables. The optimal threshold is that which exhibits the smallest confidence interval. The selected function type is that which maximizes the coefficient of explanation and passes a test of significance using an F-test with an alpha value of 0.05 (Werner *et al.,* 2014).

Table 3.3. Function type categories used in SegReg to assess and describe associations between benthic macroinvertebrate functional metrics and agricultural land.

Type Category	**Descriptor**	**Visual**
Type 0	No significant association; represented by a flat line. There is no threshold.	
Type 1	Represented by either a positive or negative sloping line. There is no threshold.	
Type 2	There are two segments that are connected at a threshold and each segment is sloping either positively or negatively.	
Type 3	Represented by a flat line until a threshold is established and followed by either a positive or negative sloping line. The threshold is accompanied with a confidence block.	
Type 4	A positive or negative sloping line that reaches a threshold and then flattens. The threshold is accompanied with a confidence block.	
Type 5	A flat line reaches a threshold and is followed with either a 90 degree shift upwards or downwards before the second segment continues flat again.	
Type 6	There are two disconnected segments but at least one of them is sloping positively or negatively either towards or away from the threshold.	

4.0 Results

4.1 Scenario #1

Scenario #1 was comprised of 24 sites that had at least 85% forest cover in the riparian corridor, and had limited evidence of channelization in the sampling reach. Habitat assessments showed that overall habitat quality scores for Scenario #1 sites were generally in the optimal category with a mean score of 152 (s = 25) (Table 4.1). The most optimal habitat score outside of the controlled for channelization category was channel flow status ($\bar{x}$ = 19, s = 2) followed by vegetative protection and riparian corridor width ($\bar{x}$ = 18, s = 3 and $\bar{x}$ = 18, s = 4, respectively). Epifaunal substrate/available cover and channel sinuosity tied for the lowest mean scores width ($\bar{x}$ = 11, s = 6 and $\bar{x}$ = 11, s = 4, respectively). Of the ten categories, epifaunal substrate/available cover had the greatest coefficient of variation (CV) at 54%, whereas channel alteration had the smallest CV at 9%. Pool characteristics (i.e., pool substrate characterization and pool variability) were the only two habitat categories that exhibited scores ranging from 0 to 20. However, epifaunal substrate/available cover, sediment deposition, and channel sinuosity also had scores that nearly spanned the scoring gradient (1 to 20, 3 to 20 and 2 to 19, respectively).

Sites sampled as part of Scenario #1 had a median of 21 taxonomic groups identified, but the range exceeded the median with the minimum number of taxa collected being 12, and the maximum number of taxa being 36 (Table 4.2). Dipterans, on average, comprised 45% (s = 24%) of the individuals sampled, which was nearly 3-fold greater than the relative abundance of EPT ($\bar{x}$ = 17%, s =

Table 4.1. Descriptive statistics for United States Environmental Protection Agency (USEPA) rapid habitat assessment categories for 24 southwestern Ontario streams used in Scenario #1 (15AgR).

Habitat Assessment Category	Mean	Standard Deviation	*CV	Median	Minimum	Maximum
Epifaunal Substrate/Available Cover	11	6	54%	13	1	20
Pool Substrate Characterization	12	5	40%	15	0	20
Pool Variability	13	6	48%	16	0	20
Sediment Deposition	15	4	29%	16	3	20
Channel Flow Status	19	2	10%	20	13	20
Channel Alteration	20	2	9%	20	11	20
Channel Sinuosity	11	4	36%	11	2	19
Bank Stability	15	4	29%	16	6	20
Vegetative Protection	18	3	15%	18	10	20
Riparian Vegetative Corridor Width	18	4	19%	20	6	20
Total Habitat Assessment Score	152	25	16%	155	95	193

*Note CV = Coefficient of Variation.

Table 4.2. Descriptive statistics for benthic macroinvertebrate metrics for 24 southwestern Ontario streams used in Scenario #1 (15AgR).

BMI Metric	Mean	Standard Deviation	*CV	Median	Minimum	Maximum
Diversity Metrics						
nTaxa	22	6	25%	21	12	36
EPTRich	6	3	48%	6	2	14
DipteraRich	8	2	29%	8	3	12
Community Composition Metrics						
%EPT	17%	12%	74%	14%	1%	46%
%Diptera	45%	24%	53%	43%	6%	91%
Functional Trait Metrics						
%Small	87%	10%	12%	90%	51%	99%
%Multivoltinism	17%	10%	58%	17%	2%	32%
%Shredders	4%	5%	106%	3%	0%	18%
%Herbivores	3%	5%	199%	0%	0%	23%
%Burrowers	38%	24%	64%	35%	3%	81%
%Clingers	32%	21%	67%	27%	5%	74%
FBI	6	1	16%	6	4	7

*Note CV = Coefficient of Variation.

12%). However, on average, only 2 more Dipteran taxa were collected per site than EPT taxa. Together, Diptera and EPT taxa accounted for more than half of the average richness of the communities ($\bar{x}$ = 22, *s* = 6). More than half of the stream communities in Scenario #1 were comprised of more than 90% small-bodied taxa and no site had less than 50% small-bodied taxa. In contrast, multivoltine taxa comprised an average of 17% (*s* = 10%) of the individual BMIs collected at the sites. Shredders and herbivores were rare (i.e., < 5%) or absent from most sites (medians = 3% and 0%, respectively) but were the metrics that presented the greatest variability with CV's (106% and 199%, respectively). In contrast, burrowers and clingers were present at all sites and exhibited ranges of relative abundances of about 70%. The FBI scores showed little variability (CV = 16%; $\bar{x}$ = 6; *s* =1) and indicated that the majority of communities were fairly tolerant to organic pollution.

4.2 Scenario #2

Scenario #2 was comprised of 20 sites with greater than 75% of the riparian corridor being used for agricultural activities. Overall, habitat quality at Scenario #2 represented conditions at the lower end of the suboptimal category with an average total score of 129 (*s* = 24). Assessments of habitat quality at these sites reflected the proximity of agricultural land use with riparian vegetation corridor width averaging 8 (*s* = 7) out of a possible 20 (Table 4.3). An average score of 8, the lowest average for any of the 10 categories, was also shared by channel sinuosity (*s* = 5) and pool variability (*s* = 6). In contrast, channel flow status presented the largest mean score ($\bar{x}$ = 19, *s* = 1) and was largely invariate among

Table 4.3. Descriptive statistics for United States Environmental Protection Agency (USEPA) rapid habitat assessment categories for 20 southwestern Ontario streams used in Scenario #2 (75AgR).

Habitat Assessment Category	Mean	Standard Deviation	*CV	Median	Minimum	Maximum
Epifaunal Substrate/Available Cover	10	4	44%	10	1	17
Pool Substrate Characterization	12	5	46%	13	0	19
Pool Variability	8	6	75%	6	0	16
Sediment Deposition	15	5	35%	17	2	20
Channel Flow Status	19	1	5%	20	17	20
Channel Alteration	16	4	22%	15	10	20
Channel Sinuosity	8	5	59%	8	0	18
Bank Stability	15	4	25%	16	4	20
Vegetative Protection	15	3	22%	15	8	20
Riparian Vegetative Corridor Width	8	7	88%	5	2	20
Total Habitat Assessment Score	129	24	18%	119	95	184

*Note CV = Coefficient of Variation.

the sampled sites (CV = 5%). Outside of channel flow status, large rates of variability (i.e., CV > 40%) were established for half of the habitat assessment categories with pool variability and riparian vegetative corridor width having CV's equal to, or exceeding, 75%. Furthermore, all habitat categories except channel flow status had a range of at least 10, and seven categories had ranges of 16 or greater.

BMI communities in Scenario #2 averaged 19 taxa (s = 6), following taxonomic adjustments (Table 4.4). Over half of total taxa were either Dipterans ($\bar{x}$ = 6, s = 3) or EPT ($\bar{x}$ = 4, s = 3). However, the relative abundances of Dipterans and EPT were variable among the sampled sites with CV's of 73% and 106%, respectively. More than half of the sites were comprised of 98% small-bodied BMI, whereas the other life history trait, %Multivoltinism, had a median of 27% (range = 69%). The two largest CV's were associated with the two feeding habitat trait metrics at 237% for %Shredders and 142% for %Herbivores. Each of the habitat scores displayed minimal variability (CV = 19%) with a mean score of 6 (s = 1), indicating that most of the communities were fairly tolerant of organic pollution.

4.3 Scenario #3

Scenario #3 incorporated 43 sites that had at least 80% agricultural land use in the catchment, but these sites ranged from 0 to 95% agriculture in the riparian corridor. Total habitat scores from Scenario #3 sites also shared a wide range with a minimum score of 88 and a maximum score of 195 (Table 4.5). The large overall habitat quality range was reflected in individual category scores with 6 categories (i.e., epifaunal substrate/available cover, pool substrate

Table 4.4. Descriptive statistics for benthic macroinvertebrate metrics for 20 southwestern Ontario streams used in Scenario #2 (75AgR).

BMI Metric	Mean	Standard Deviation	*CV	Median	Minimum	Maximum
Diversity Metrics						
nTaxa	19	6	32%	18	11	33
EPTRich	4	3	86%	3	0	11
DipteraRich	6	3	49%	6	2	13
Community Composition Metrics						
%EPT	18%	19%	106%	18%	0%	59%
%Diptera	33%	24%	73%	27%	1%	72%
Functional Trait Metrics						
%Small	92%	12%	14%	98%	52%	100%
%Multivoltinism	30%	25%	82%	27%	2%	71%
%Shredders	3%	8%	237%	0%	0%	30%
%Herbivores	2%	3%	142%	1%	0%	8%
%Burrowers	35%	28%	79%	27%	1%	88%
%Clingers	30%	26%	88%	21%	1%	96%
FBI	6	1	19%	6	4	7

*Note CV = Coefficient of Variation.

Table 4.5. Descriptive statistics for United States Environmental Protection Agency (USEPA) rapid habitat assessment categories for 43 southwestern Ontario streams used in Scenario #3 (80AgC).

Habitat Assessment Category	Mean	Standard Deviation	CV	Median	Minimum	Maximum
Epifaunal Substrate/Available Cover	11	5	43%	11	1	20
Pool Substrate Characterization	12	5	42%	13	0	20
Pool Variability	10	6	64%	10	0	20
Sediment Deposition	15	4	25%	16	7	20
Channel Flow Status	19	2	8%	20	13	20
Channel Alteration	18	3	17%	20	10	20
Channel Sinuosity	10	4	44%	10	0	20
Bank Stability	14	4	29%	14	2	20
Vegetative Protection	16	4	25%	16	4	20
Riparian Vegetative Corridor Width	13	7	54%	15	2	20
Total Habitat Assessment Score	139	24	17%	138	88	195

*Note CV = Coefficient of Variation.

characterization, pool variability, channel sinuosity, bank stability, and riparian vegetative corridor width) having ranges of 18 or more. However, the median category scores for pool substrate characterization, bank stability, and riparian vegetative corridor width were 13 or greater. Channel flow status and channel alteration exhibited the highest mean scores ($\bar{x}$ = 19, s = 2 and $\bar{x}$ = 18, s = 3, respectively). These two categories were also the least variable with CV's of 8% and 17%, respectively. The smallest mean scores were found for pool variability and channel sinuosity ($\bar{x}$ = 10, s = 6, and $\bar{x}$ = 10, s = 4, respectively), but pool variability was also the most variable category among the sites in Scenario #3 (CV = 64%).

Sites included in Scenario #3 averaged 21 different taxa (s = 6) (Table 4.6). On average, almost 60% of these taxa were either Dipterans ($\bar{x}$ = 7, s = 2) or EPT ($\bar{x}$ = 5, s = 3) taxa. However, DipteraRich and EPTRich both had ranges of at least 10. Furthermore, the relative abundances of Dipterans and EPT were highly variable with CV's of 55% and 86%, respectively. The least variable metric was %Small with a CV of 10% and a median of 96%. However, the other life history metric, %Multivoltinism, was more variable (CV = 75%), and ranged from 3% to 71% with a median of 21%. Variability was most dominant in the two feeding traits (%Shredders, CV = 171%; %Herbivores, CV = 172%). These two feeding trait categories were also rare among the Scenario #3 sites with medians of 1%. The two habitat traits exhibited wide ranging relative abundances with %Burrowers having a minimum of 4% and a maximum of 93%, and %Clingers having a minimum of 0% and a maximum of 74%. Median relative abundances of both

Table 4.6. Descriptive statistics for benthic macroinvertebrate metrics for 43 southwestern Ontario streams used in Scenario #3 (80AgC).

BMI Metric	Mean	Standard Deviation	*CV	Median	Minimum	Maximum
Diversity Metrics						
nTaxa	21	6	28%	21	10	34
EPTRich	5	3	54%	4	0	12
DipteraRich	7	2	37%	6	2	12
Community Composition Traits						
%EPT	18%	15%	86%	18%	0%	59%
%Diptera	41%	23%	55%	37%	2%	87%
Functional Trait Metrics						
%Small	93%	9%	10%	96%	50%	100%
%Multivoltinism	27%	21%	75%	21%	3%	71%
%Shredders	4%	7%	171%	1%	0%	35%
%Herbivores	2%	4%	172%	1%	0%	16%
%Burrowers	37%	26%	69%	27%	4%	93%
%Clingers	28%	22%	80%	23%	0%	74%
FBI	6	1	18%	6	4	8

*Note CV = Coefficient of Variation.

habitat traits were 4% different at 27% and 23% for %Burrowers and %Clingers, respectively. The FBI score averaged 6 at the Scenario #3 sites and exhibited little variability among sites (CV = 18%), indicating most communities were fairly tolerant of organic pollution.

4.4 Threshold Detection Scenario #1

Linear regression analyses conducted to assess relationships between BMI metrics and agricultural land use within the control scale (i.e., riparian corridor) identified significant associations ($p < 0.1$) for both %Clingers and the FBI. Residuals for %Clingers (i.e., %ClingersRes) and FBI (i.e., FBIRes) were thus used in the threshold analysis along with the raw data for the other 11 BMI metrics. No thresholds were identified by the SiZer analysis within the described range of agricultural land use (48% to 85%) at the catchment scale for any of the calculated BMI metrics (Table 4.7). However, four of the BMI metrics exhibited linear associations with the proportion of agricultural land use within the catchment. DipteraRich exhibited an increasing linear association with increased agriculture in the catchment (read at $\log_{10}(h) = 0.4$; Figure 4.1a). DipteraRich was the only increasing linear association observed among all BMI metrics in Scenario #1. %EPT was the only community composition metric that exhibited an association ($\log_{10}(h) = 0.5$; Figure 4.1c). Feeding traits were the only functional trait metrics that exhibited associations with increasing agriculture in the catchment. Decreasing linear associations were observed in both %Shredders ($\log_{10}(h) = 0.4$) and %Herbivores ($\log_{10}(h) = 0.05$) (Figures 4.2e & g). In contrast to the five associations observed using SiZer, SegReg identified only one significant

Table 4.7. Statistical analyses describing associations between BMI metrics and increasing agricultural land use in the catchment using both SiZer and SegReg for 24 southwestern Ontario streams used in Scenario #1 (15AgR).

BMI Metric	SiZer			SegReg			
	Threshold	**Before**	**After**	**Threshold**	**Before**	**After**	**Type**
Diversity Metrics							
nTaxa	-----------------	-----------	----------	-----------------	-----------	----------	0
EPTRich	-----------------	-----------	----------	-----------------	-----------	----------	0
DipteraRich	-----------------	Increase	----------	-----------------	-----------	-----------	0
Community Composition Metrics							
%EPT	-----------------	Decrease	----------	-----------------	-----------	-----------	0
%Diptera	-----------------	-----------	----------	-----------------	-----------	-----------	0
Functional Trait Metrics							
%Small	-----------------	-----------	----------	-----------------	-----------	-----------	0
%Multivoltinism	-----------------	-----------	----------	-----------------	-----------	-----------	0
%Shredders	-----------------	Decrease	----------	-----------------	-----------	-----------	0
%Herbivores	-----------------	Decrease	----------	68%	Greater	Smaller	5
%Burrowers	-----------------	-----------	----------	--------------	-----------	-----------	0
%ClingersRes	-----------------	-----------	----------	-----------------	-----------	-----------	0
FBIRes	-----------------	-----------	----------	-----------------	-----------	-----------	0

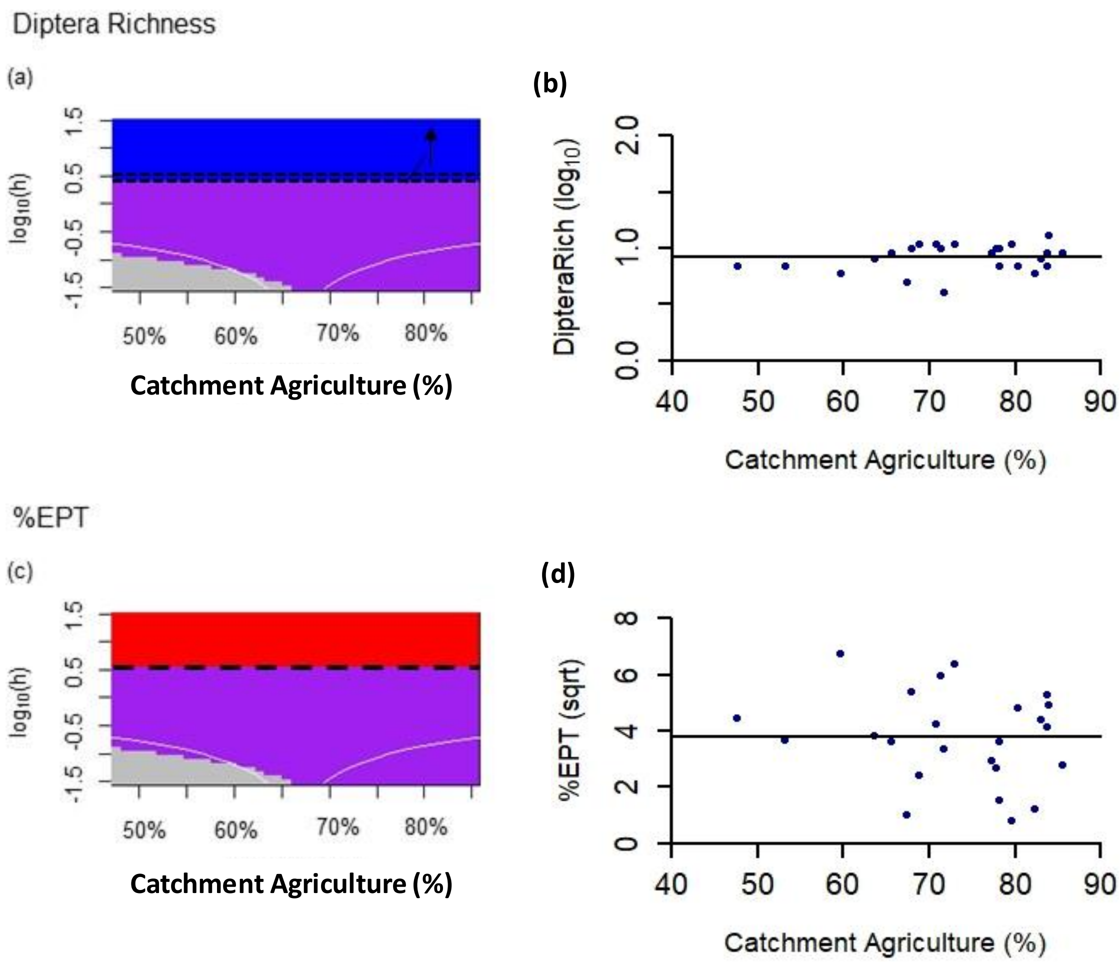


Figure 4.1. Plots indicating results of threshold analyses in Scenario #1 (15AgR) for Diptera Richness (a and b) and %EPT (c and d) using SiZer (a and c) and SegReg (b and d). Sizer plots show first derivative map displaying blue shading (positive association), red shading (negative association), purple shading (no association), and grey shading (insufficient data). Bandwidths marked by black dashed lines. Black arrow further emphasizes direction of association. SegReg plots present solid black line of best fit, and blue data points representing sample stream sites.

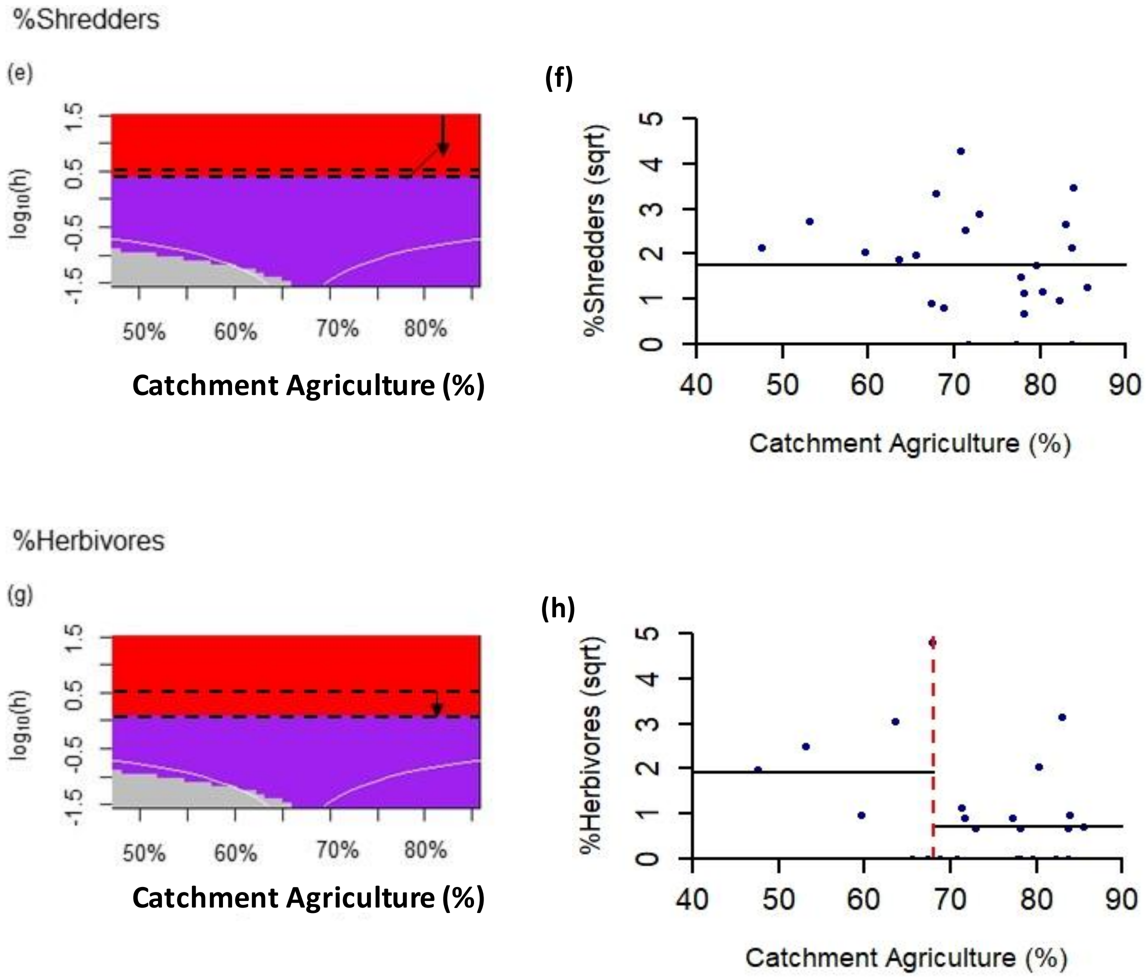


Figure 4.2. Plots indicating results of threshold analyses in Scenario #1 (15AgR) for %Shredders (e and f) and %Herbivores (g and h) using SiZer (e and g) and SegReg (f and h). Sizer plots show first derivative map displaying red shading (negative association), purple shading (no association), and grey shading (insufficient data). Bandwidths marked by black dashed lines. Black arrow further emphasizes direction of association. SegReg plots present solid black line of best fit, and blue data points representing sample stream sites. Threshold point indicated by red dashed line.

association between the BMI metrics and the proportion of agricultural land use at the catchment scale. This association was with %Herbivores which was found to exhibit a Type 5 threshold, where the relative abundance of herbivores was observed to shift from a mean relative abundance of approximately 2% to about 0.8% when agricultural cover in the catchment exceeded 68% (EC = 0.196, $F_{2,21}$ = 2.573, p = 0.1; Figure 4.2h).

4.5 Threshold Detection Scenario #2

Preliminary linear regression analyses relating agricultural land use within the control scale for Scenario #2 to the calculated BMI metrics did not identify any significant associations (p > 0.1). As a result, raw metric values were used for all threshold analyses. Agricultural land use in the catchment ranged between 44% and 96% in Scenario #2. No thresholds and only one association were identified among the diversity and community composition metrics using the SiZer and SegReg analyses. In contrast, 6 of the 7 functional trait metrics did exhibit associations with the proportion of agricultural land use within the catchment (Table 4.8). SiZer analysis indicated that EPTRich was negatively associated with increasing agricultural land use in the catchment ($\log_{10}(h)$ = 0.25; Figure 4.3a). Positive linear associations were observed from the SiZer analyses for both life history traits $\log_{10}(h)$ = 0; Figures 4.3c & 4.4e). In contrast, when assessed using the SegReg analysis, %Small was positively associated (Type 1 function) with agricultural land cover (R^2 = 0.638, $F_{1,18}$ = 31.657, p = 0.001; Figure 4.3d). A Type 5 function and an associated threshold of 84% agricultural cover was observed when SegReg analysis was conducted on %Multivoltinism, which increased from

Table 4.8. Statistical analyses describing associations between BMI metrics and increasing agricultural land use in the catchment using both SiZer and SegReg for 20 southwestern Ontario streams used in Scenario #2 (75AgR).

BMI Metric	SiZer			SegReg			
	Threshold	Before	After	Threshold	Before	After	Type
Diversity Metrics							
nTaxa	-----------------	-----------	----------	-----------------	-----------	----------	0
EPTRich	-----------------	Decrease	----------	-----------------	-----------	----------	0
DipteraRich	-----------------	-----------	----------	-----------------	-----------	-----------	0
Community Composition Metrics							
%EPT	-----------------	-----------	----------	-----------------	-----------	-----------	0
%Diptera	-----------------	-----------	----------	-----------------	-----------	-----------	0
Functional Trait Metrics							
%Small	-----------------	Increase	----------	-----------------	Increase	N/A	1
%Multivoltinism	-----------------	Increase	----------	84%	Smaller	Greater	5
%Shredders	-----------------	-----------	----------	-----------------	-----------	-----------	0
%Herbivores	85%	Decrease	----------	83%	Decrease	-----------	4
%Burrowers	-----------------	Increase	----------	--------------	-----------	-----------	0
%Clingers	-----------------	Decrease	----------	72%	-----------	Decrease	3
FBI	-----------------	Increase	----------	-----------------	Increase	N/A	1

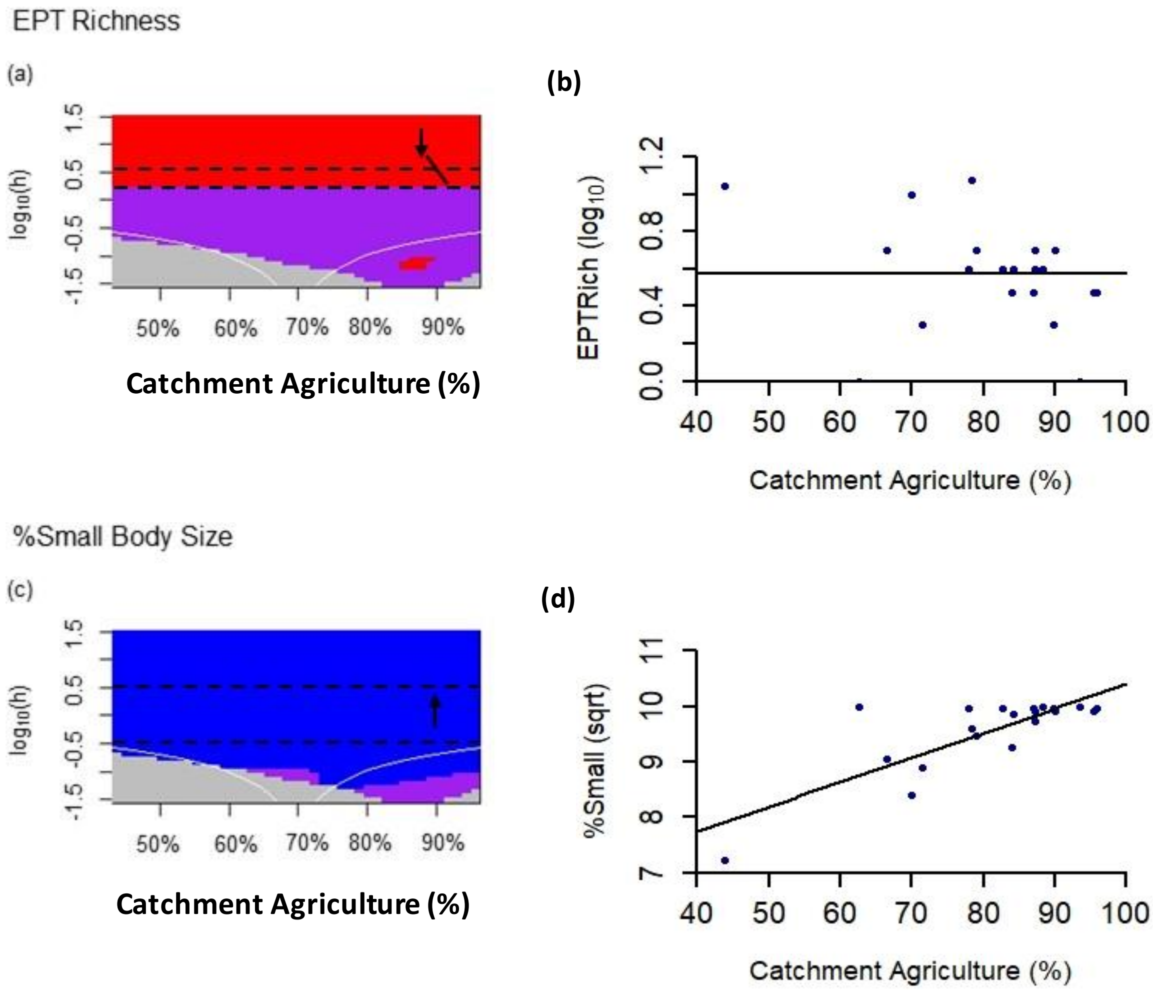


Figure 4.3. Plots indicating results of threshold analyses in Scenario #2 (75AgR) for EPT Richness (a and b) and %Small Body Size (c and d) using SiZer (a and c) and SegReg (b and d). Sizer plots show first derivative map displaying blue shading (positive association), red shading (negative association), purple shading (no association), and grey shading (insufficient data). Bandwidths marked by black dashed lines. Black arrow further emphasizes direction of association. SegReg plots present solid black line of best fit, and blue data points representing sample stream sites.

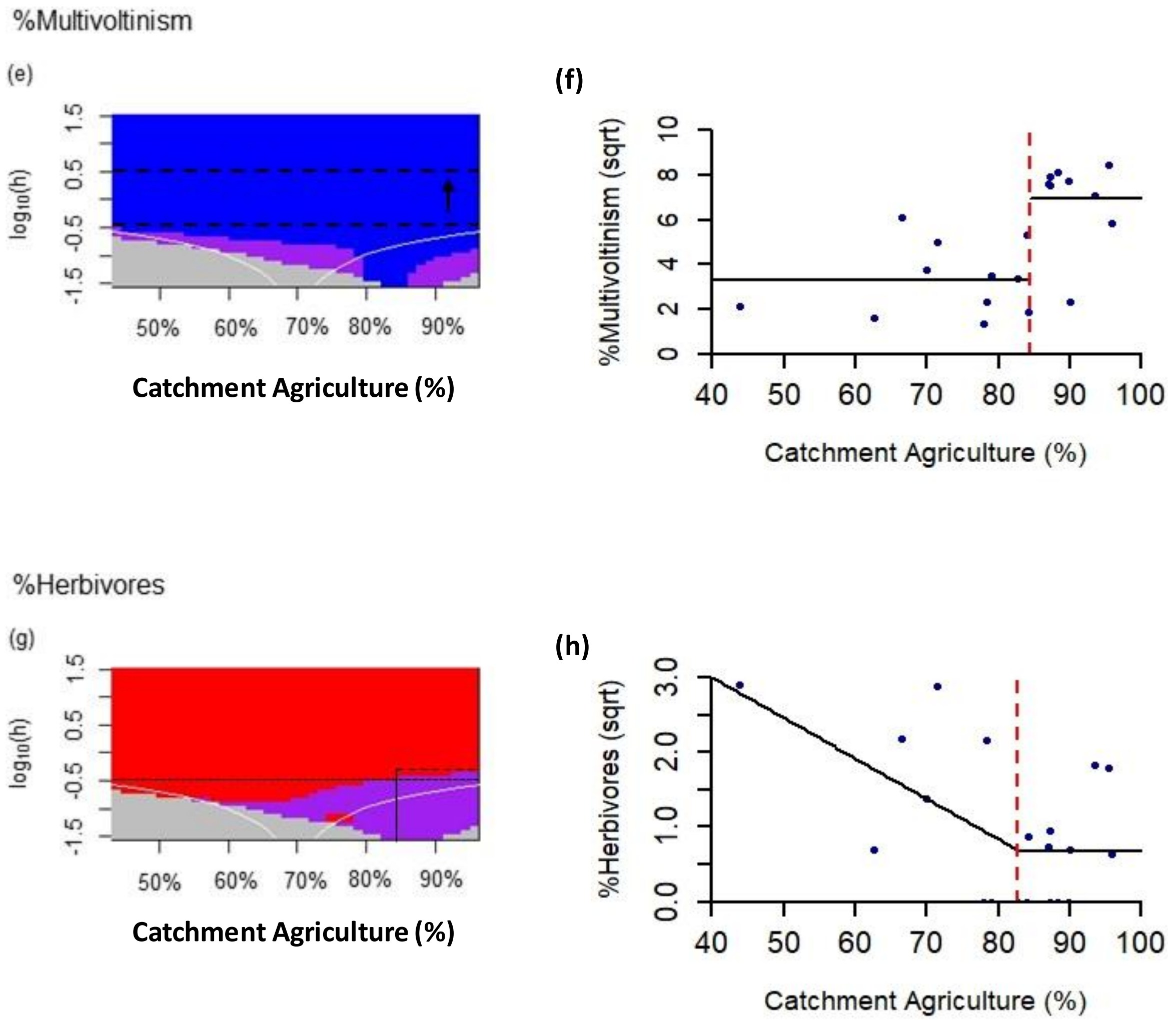


Figure 4.4. (e) Plots indicating results of threshold analyses in Scenario #2 (75AgR) for %Multivoltinism (e and f) and %Herbivores (g and h) using SiZer (e and g) and SegReg (f and h). Sizer plots show first derivative map displaying blue shading (positive association), red shading (negative association), purple shading (no association), and grey shading (insufficient data). Bandwidths marked by black dashed lines and threshold point represented by a solid black line. Black arrow further emphasizes direction of association. SegReg plots present solid black line of best fit, and blue data points representing sample stream sites. Threshold point indicated by red dashed line.

a mean relative abundance of 34% to a mean of approximately 70% when agricultural land cover exceeded 84% (EC = 0.543, $F_{2,17}$ = 10.109, p = 0.001; Figure 4.4f). Both the SiZer and SegReg analyses presented similar results for %Herbivores with decreasing associations that had thresholds of 85% ($\log_{10}(h)$ = -0.4) and 83% (90% confidence interval: 75% to 90%; EC = 0.347, $F_{3,16}$ = 2.848, p = 0.07), respectively (Figures 4.4g & h). The habitat functional traits exhibited opposing associations identified with the SiZer analyses with %Burrowers displaying an increasing linear association ($\log_{10}$(h) = 0.5) and a decreasing one with %Clingers ($\log_{10}(h)$ = 0; Figures 4.5i & k). However, only %Clingers exhibited a significant association in the SegReg analysis with a Type 3 function indicating decreasing relative abundance of clingers after agricultural land cover exceeded 72% (90% confidence interval: 65% to 80%) coverage at the catchment scale (EC = 0.404, $F_{3,16}$ = 3.647, p = 0.035; Figure 4.5l). Both SiZer and SegReg analyses identified positive linear associations between agricultural land cover and the Hilsenhoff Family Biotic Index ($\log_{10}(h)$ = 0 and R^2 = 0.478, $F_{1,18}$ = 16.581, p = 0.001, respectively; Figures 4.6m & n).

4.6 Threshold Detection Scenario #3

Linear regression analyses conducted on all BMI metrics in Scenario #3 identified significant associations between variation in agricultural land use within the control scale (i.e., catchment scale) for %Multivoltinism and the FBI. Residual values (i.e., %MultivoltinismRes and FBIRes) were thus used in threshold analyses. Raw metric scores were used in threshold analyses for the remaining 11

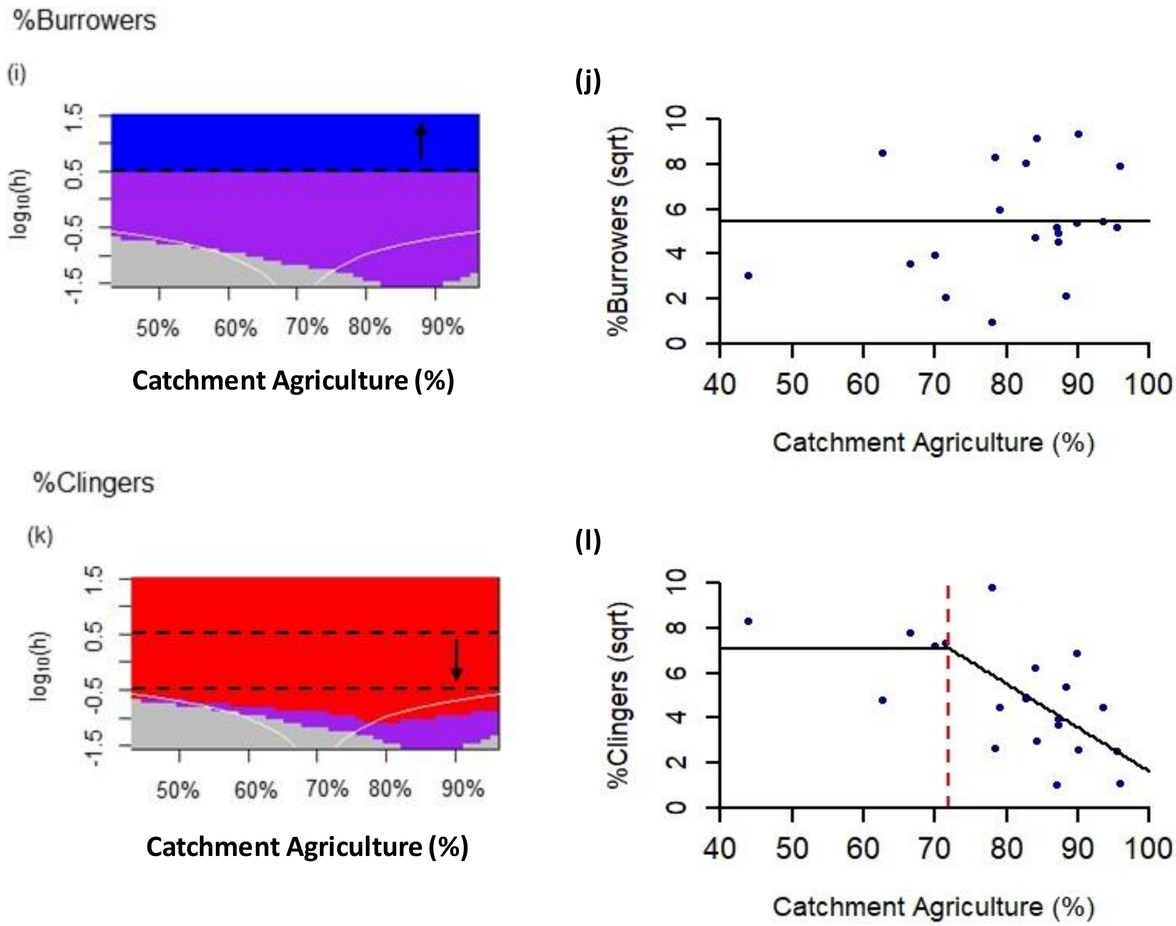


Figure 4.5. Plots indicating results of threshold analyses in Scenario #2 (75AgR) for %Burrowers (i and j) and %Clingers (k and l) using SiZer (i and k) and SegReg (j and l). Sizer plots show first derivative map displaying blue shading (positive association), red shading (negative association), purple shading (no association), and grey shading (insufficient data). Bandwidths marked by black dashed lines. Black arrow further emphasizes direction of association. SegReg plots present solid black line of best fit, and blue data points representing sample stream sites. Threshold point indicated by red dashed line.

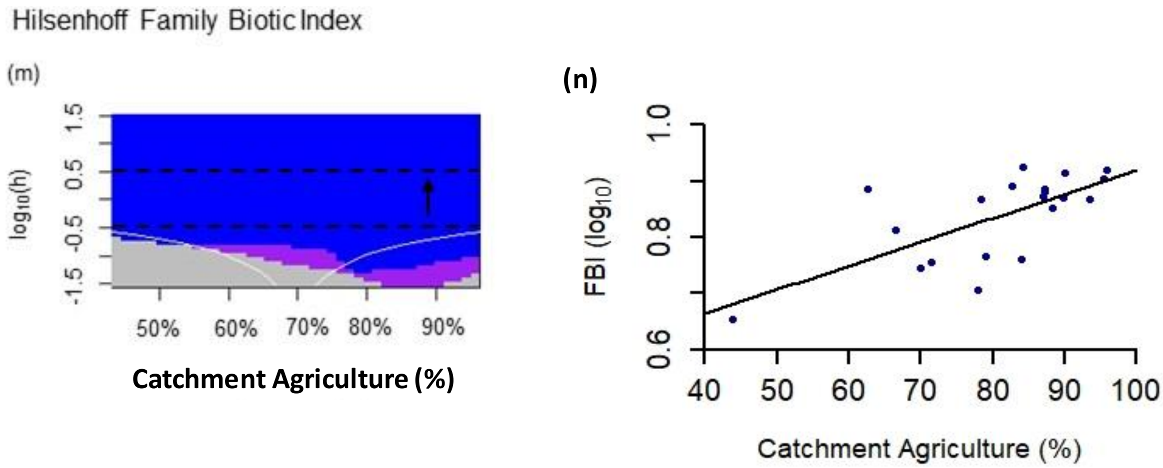


Figure 4.6. Plots indicating results of threshold analyses in Scenario #2 (75AgR) for FBI (m and n) using SiZer (m) and SegReg (n). Sizer plots show first derivative map displaying blue shading (positive association), purple shading (no association), and grey shading (insufficient data). Bandwidths marked by black dashed lines. Black arrow further emphasizes direction of association. SegReg plots present solid black line of best fit, and blue data points representing sample stream sites.

BMI metrics. In Scenario #3, agricultural land use in the riparian corridor ranged between 0% and 95%. Results from the SiZer and SegReg analyses revealed negative associations for all three diversity trait metrics (Table 4.9). Five of the seven functional trait metrics also identified significant associations with the proportion of agricultural land use within the riparian corridor. No significant associations were identified for the community composition metrics by either analysis. SiZer and SegReg analyses resulted in comparable results among all diversity metrics in terms of identifying thresholds that were followed by decreasing diversity with increasing agricultural land cover. However, the amount of agricultural cover in the riparian corridor associated with the thresholds differed between the two statistical programs by as much as 74% (DipteraRich) and by no less than 42% (nTaxa) (Figures 4.7a through 4.8f). Furthermore, neither statistical analysis identified associations between agricultural cover in the riparian corridor and the community composition metrics. The SiZer and SegReg analyses both identified thresholds that were followed by positive associations for %Small, and similar to what was observed for the diversity metrics, the thresholds were different between analyses; 18% ($\log_{10}(h) = -0.3$) and 44% (90% confidence interval: 35% to 51%) coverage at the riparian corridor scale (EC = 0.176, $F_{3,39} = 2.805$, $p = 0.057$) for SiZer and SegReg, respectively (Figures 4.8g & h). Agricultural cover in the riparian corridor was not found to be associated with %Herbivores but did exhibit a threshold effect on %Shredders, based on both the SiZer and SegReg analyses. %Shredders was observed to decline following thresholds of 46% ($\log_{10}(h) = -0.25$) and 59% (90% confidence interval: 50% to 70%) coverage at the

Table 4.9. Statistical analyses describing associations between BMI metrics and increasing agricultural land use in the riparian corridor using both SiZer and SegReg for 43 southwestern Ontario streams used in Scenario #3 (80AgC).

BMI Metric	SiZer			SegReg			
	Threshold	Before	After	Threshold	Before	After	Type
Diversity Metrics							
nTaxa	20%	-----------	Decrease	62%	----------	Decrease	3
EPTRich	8%	-----------	Decrease	53%	----------	Decrease	3
DipteraRich	6%	-----------	Decrease	75%	----------	Decrease	3
Community Composition Metrics							
%EPT	---------------	-----------	-----------	---------------	----------	-----------	0
%Diptera	---------------	-----------	-----------	---------------	----------	-----------	0
Functional Trait Metrics							
%Small	18%	-----------	Increase	44%	----------	Increase	3
%MultivoltinismRes	28%	-----------	Increase	77%	Smaller	Greater	5
%Shredders	46%	-----------	Decrease	59%	----------	Decrease	3
%Herbivores	---------------	-----------	-----------	--------------	----------	-----------	0
%Burrowers	---------------	-----------	-----------	--------------	----------	-----------	0
%Clingers	25%, 90%	Flat, Decrease	Decrease, Flat	53%	Greater	Smaller	5
FBIRes	29%	-----------	Increase	53%	Smaller	Greater	5

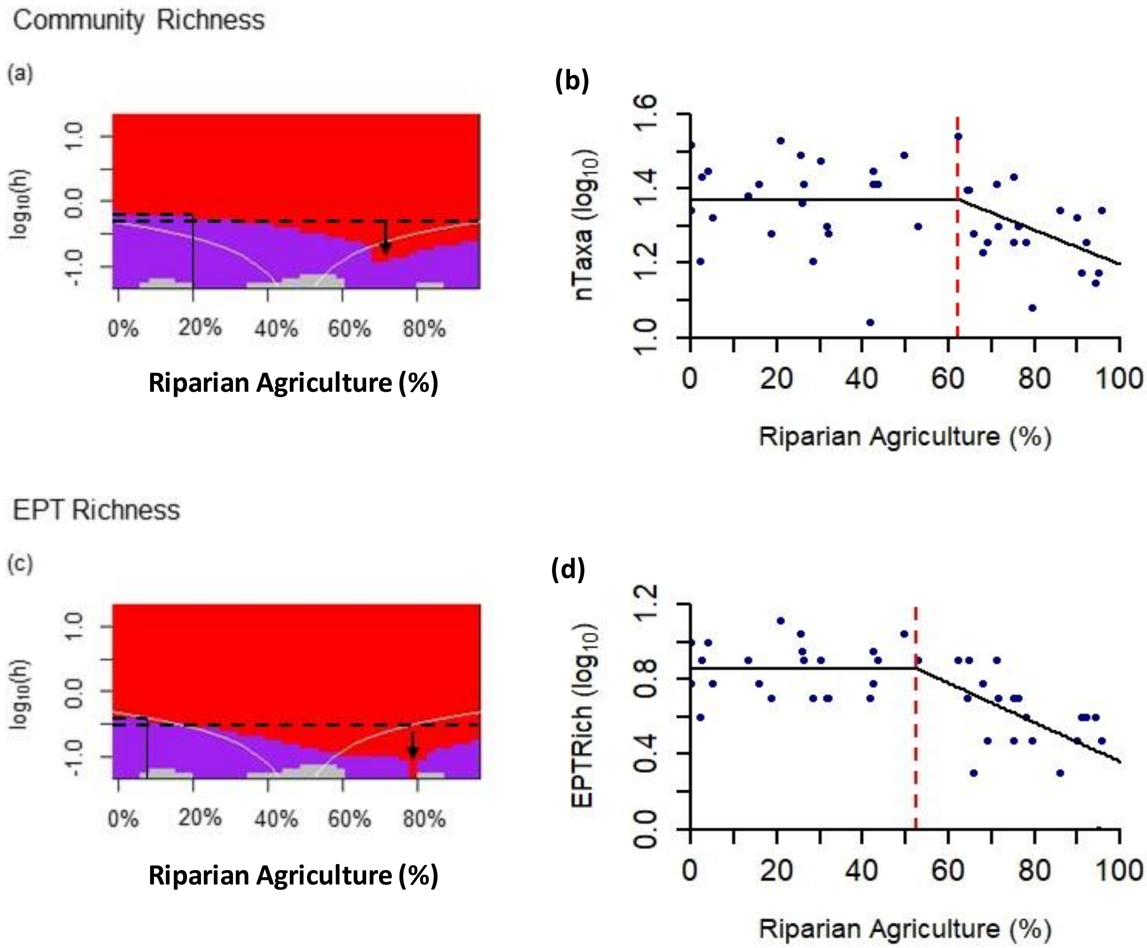


Figure 4.7. Plots indicating results of threshold analyses in Scenario #3 (80AgC) for Community Richness (a and b) and EPT Richness (c and d) using SiZer (a and c) and SegReg (b and d). Sizer plots show first derivative map displaying red shading (negative association), purple shading (no association), and grey shading (insufficient data). Bandwidths marked by black dashed lines and threshold point represented by a solid black line. Black arrow further emphasizes direction of association. SegReg plots present solid black line of best fit, and blue data points representing sample stream sites. Threshold point indicated by red dashed line.

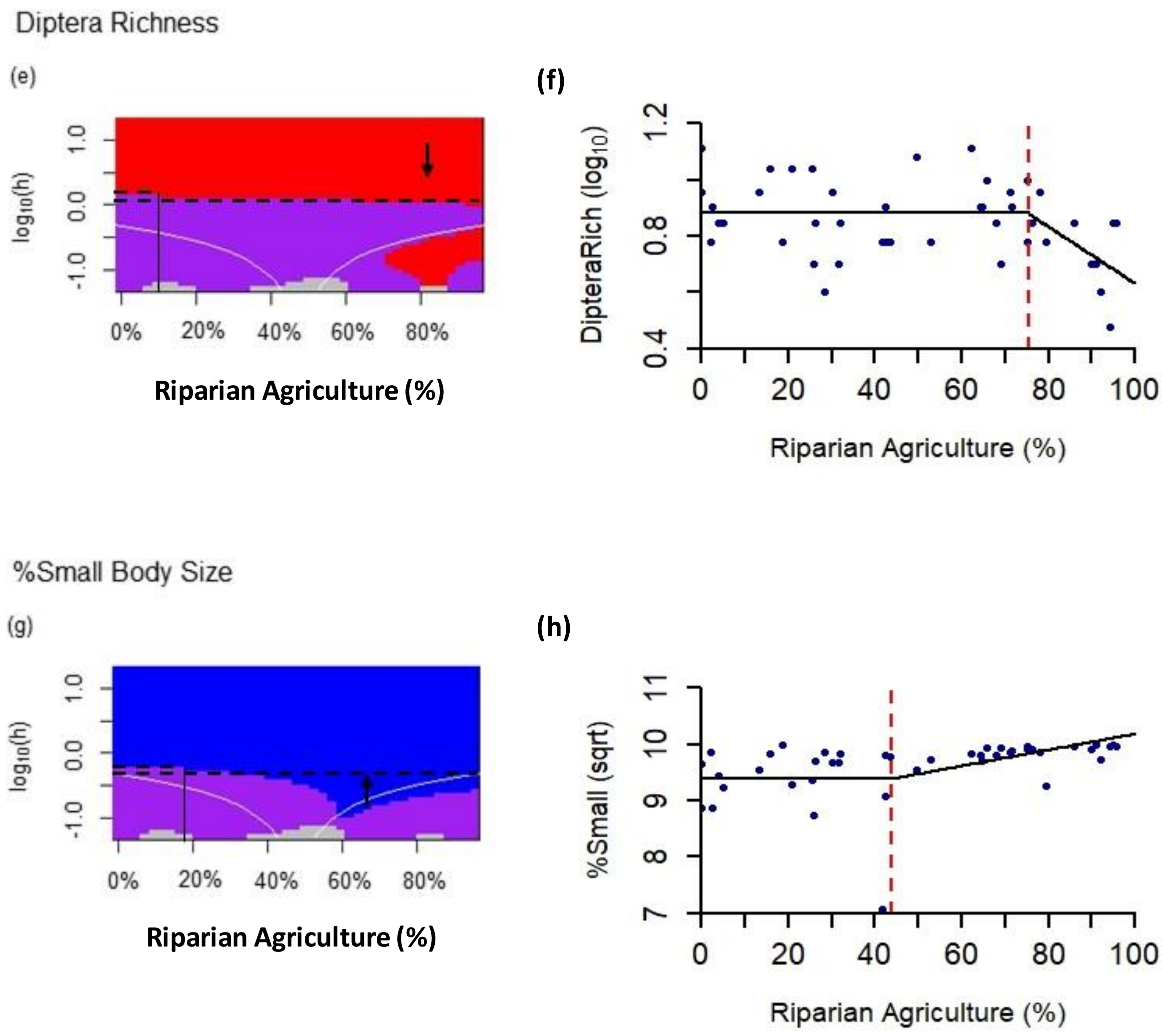


Figure 4.8. Plots indicating results of threshold analyses in Scenario #3 (80AgC) for Diptera Richness (e and f) and %Small Body Size (g and h) using SiZer (e and g) and SegReg (f and h). Sizer plots show first derivative map displaying blue shading (positive association), red shading (negative association), purple shading (no association), and grey shading (insufficient data). Bandwidths marked by black dashed lines and threshold point represented by a solid black line. Black arrow further emphasizes direction of association. SegReg plots present solid black line of best fit, and blue data points representing sample stream sites. Threshold point indicated by red dashed line.

riparian corridor scale (EC = 0.196, $F_{3,39}$ = 3.168, p = 0.039) based on SiZer and Segreg, respectively (Figures 4.9k & l). Similarly, significant associations were only found in one of the two habitat traits, as %Burrowers did not reveal any significant observations, whereas the SiZer ($\log_{10}(h)$ = -0.4) and SegReg (EC = 0.19, $F_{2,40}$ = 4.629, p = 0.016) analyses identified thresholds within the response of %Clingers to agricultural land cover in the riparian corridor. SiZer denoted a threshold at approximately 25% agricultural land cover in the riparian corridor beyond which %Clingers declined prior to a second threshold at 90% agricultural land cover, where additional agricultural land cover was not associated with further change in composition (Figure 4.10m). In contrast, the SegReg analysis indicated the response of %Clingers was best described by a Type 5 function characterized by a downward shift from a mean relative abundance of approximately 5.7% to a mean of 3.8% at a threshold of 53% agricultural land cover in the riparian corridor (EC = 0.19, $F_{2,40}$ = 4.269, p = 0.016; Figure 4.10n). SiZer and SegReg analyses found positive associations with %MultivoltinismRes, identifying thresholds of 28% ($\log_{10}(h)$ = -0.35) and 77% (90% confidence interval: 75% to 80%; EC = 0.193, $F_{3,39}$ = 3.118, p = 0.041) coverage at the riparian corridor scale, respectively (Figures 4.9i & j). Lastly, the FBIRes was found to be associated with agricultural land cover in the riparian corridor by both analyses. However, similar to %Clingers, the SiZer analysis identified a threshold after which FBIRes increased with agricultural cover, whereas the SegReg analysis indicated that a Type 5 function best fit the variation in the sampled streams (mean before = -0.02; mean after =

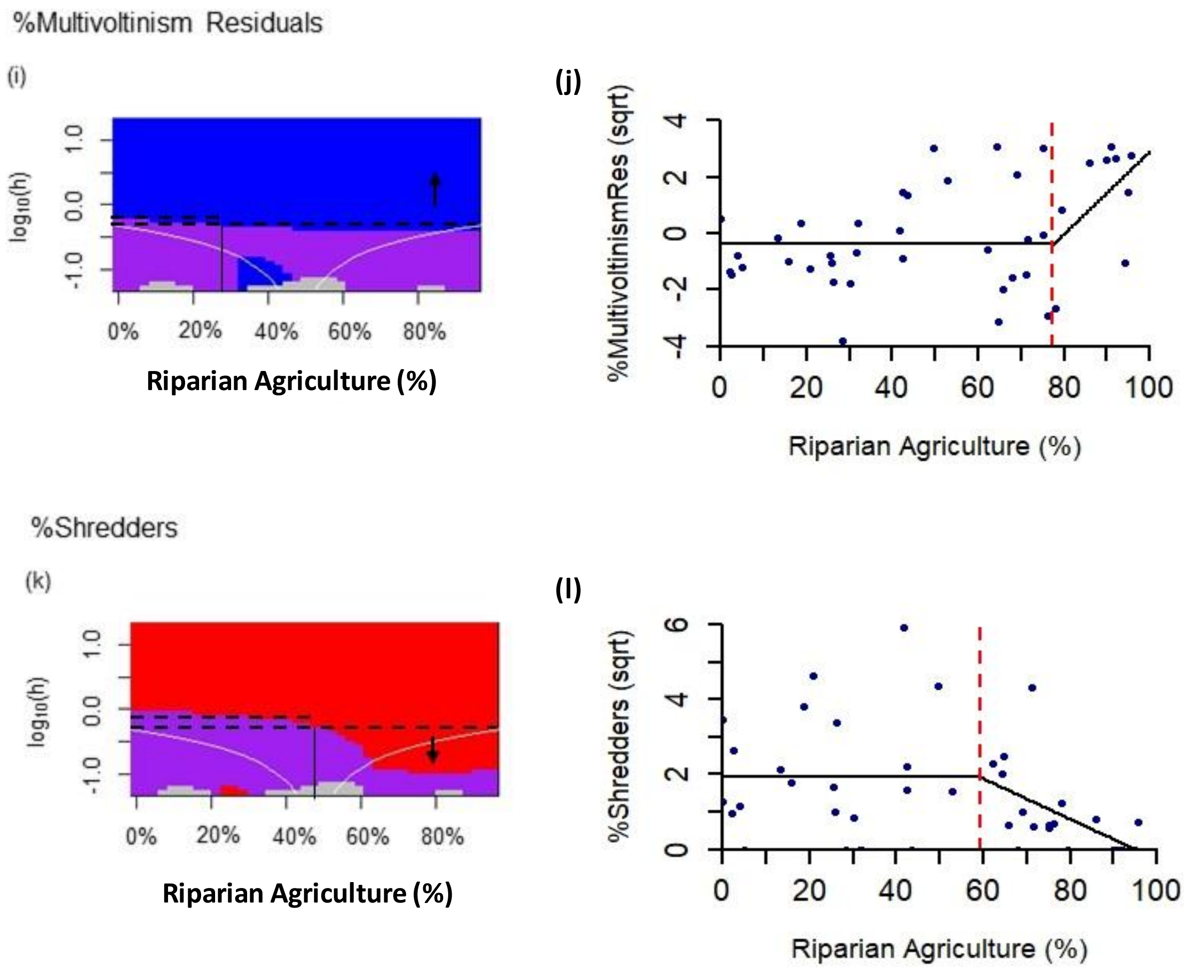


Figure 4.9. Plots indicating results of threshold analyses in Scenario #3 (80AgC) for %Multivoltinism Residuals (i and j) and %Shredders (k and l) using SiZer (i and k) and SegReg (j and l). Sizer plots show first derivative map displaying blue shading (positive association), red shading (negative association), purple shading (no association), and grey shading (insufficient data). Bandwidths marked by black dashed lines and threshold point represented by a solid black line. Black arrow further emphasizes direction of association. SegReg plots present solid black line of best fit, and blue data points representing sample stream sites. Threshold point indicated by red dashed line.

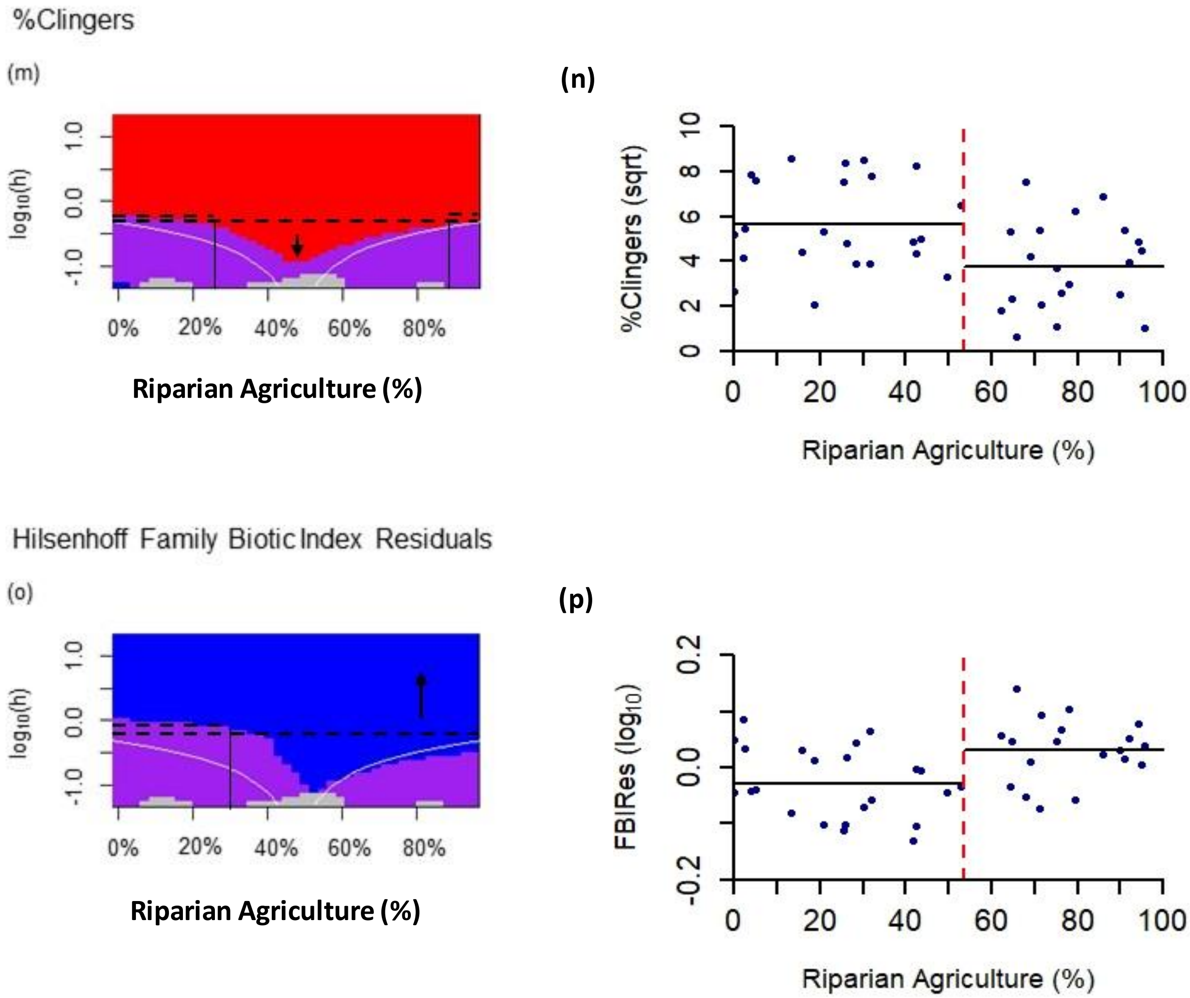


Figure 4.10. Plots indicating results of threshold analyses in Scenario #3 (80AgC) for %Clingers (m and n) and FBI Residuals (o and p) using SiZer (m and o) and SegReg (n and p). Sizer plots show first derivative map displaying blue shading (positive association), red shading (negative association), purple shading (no association), and grey shading (insufficient data). Bandwidths marked by black dashed lines and threshold point represented by a solid black line. Black arrow further emphasizes direction of association. SegReg plots present solid black line of best fit, and blue data points representing sample stream sites. Threshold point indicated by red dashed line.

0.02). Threshold values also differed between the analyses with SiZer indicating a threshold of 29% ($\log_{10}(h)$ = -0.2) and SegReg indicating a threshold of 53% (EC = 0.21, $F_{2,40}$ = 5.278, p = 0.009; Figures 4.10o & p).

5.0 Discussion

My study sought to identify scale-specific thresholds for agricultural land cover within the catchment and riparian corridor areas associated with changes in BMI community structure and function. Results of my assessments of the three land use scenarios generally supported my hypotheses. In particular, only a small number of primarily functional metrics were associated with land cover variation in the catchment scale (Scenarios 1 and 2), but even less so when the riparian corridor was covered in forest (Scenario 1). In contrast, BMI metrics describing community structure and function were frequently associated with effects of increasing agricultural cover in the riparian corridor (Scenario 3). When integrated, the findings from my land use pattern scenarios indicate that land use in the riparian corridor is disproportionately important to the ecological integrity of small streams exposed to agricultural activity.

5.1 Scenario #1

My results did not fully support my hypothesis for Scenario #1 in that I observed only one threshold response between BMI metrics and variation in agricultural land cover at the catchment scale when forest cover dominated the riparian corridor. This finding indicates that riparian vegetation can maintain BMI community conditions even when agricultural land cover in the upland areas of the catchment nears 100%. My research thus adds to a large body of literature that provides evidence indicating forested riparian vegetation serves a key role in buffering stream biota from the effects of surrounding agricultural activity (e.g., Thomas *et al.,* 2016; Naiman *et al.,* 1993; Ormerod *et al.,* 1993; Osborne &

Kovacic, 1993; Naiman *et al.,* 1988). For example, Thomas *et al.* (2016) pointed out that deciduous riparian trees in headwater streams provide shade, thus cooling water temperatures and enhance resilience by providing more diverse habitat and energetic subsidies (e.g. leaf litter for herbivores to feed on). Osborne and Kovacic (1993) showed that riparian forests were also efficient at reducing nitrogen concentration in shallow subsurface water prior to entering a stream. Protection, maintenance and restoration of riparian corridors are thus likely to be an important part of management schemes seeking to increase agricultural production without compromising the health of nearby stream ecosystems.

The lack of associations between the BMI metrics and agricultural land cover may also be explained by the incomplete agricultural land cover gradient (i.e., 44% to 96%) encompassed by the catchments of my study region. The lack of representation of sites exposed to catchment land use below 44% limited my ability to detect thresholds for BMI metrics that may have exhibited a response to increases in agriculture at smaller amounts of cover than I was able to test. Previous studies assessing BMI response to agricultural land use have noted thresholds at proportions of catchment cover not included in this study (see review by Allan 2004). For example, Quinn and Hickey (1990) found a threshold of 30% catchment agriculture in New Zealand streams exposed to a gradient of agricultural cover. Likewise, Utz *et al.* (2009) found agricultural thresholds for some individual BMI taxa as low as 21% agricultural catchment cover. Lastly, Wang *et al.* (1997) found no major changes in BMI compositional metrics until agricultural cover exceeded 50% at the catchment scale. The findings of these other studies suggest that in the agriculturally dominated landscape of southern Ontario, all streams may have already exceeded the amount of catchment scale agriculture that would overwhelm the buffering capacity of the riparian corridor, leaving a homogeneous, tolerant and taxonomically depauperate set of communities insensitive to additional agricultural land cover. However, because previous studies in other regions have not specifically assessed catchment scale effects of agricultural land cover on BMI composition and function where the riparian corridor is intact, further research is needed to establish a complete understanding of the buffering capacity of riparian vegetation.

The relative abundance of herbivores was the only metric both threshold analyses methods (i.e., SiZer and SegReg) identified as being associated with agricultural land cover at the catchment scale when riparian corridors were comparatively undisturbed (i.e., ≤15% agricultural cover). Both threshold analyses indicated that the abundance of herbivores was negatively affected by increasing agriculture in the catchment, although only the SegReg analysis identified a threshold. The inverse relationship between herbivore abundance and agricultural land cover in the catchment contradicts what has been generally observed in prior studies (Liess *et al.,* 2012; Fuller *et al.,* 1986; Robinson & Minshall, 1986). However, Liess *et al.* (2012) pointed out that lower food quality (streams higher in periphyton C:N) would mediate the affects of increasing herbivore abundance coinciding with increases in agricultural activity, and likely more so when coupled with a wide spectrum of stressors (e.g. fine sediment inputs; pesticides; changing hydrologic regimes) associated with agricultural activity. My decreasing herbivore abundance may also have simply been a result of sensitive herbivores responding to additional agricultural stressors. This hypothesis is consistent with the fact that other studies have observed herbivore abundance increases coinciding with increases in agriculture where riparian forest cover is largely absent, and therefore increases in sunlight reaching the stream leads to increases in primary production. My study was designed so that observations made in Scenario #1 were specific to an intact riparian corridor where my stream sites were significantly more shaded. Indeed, Delong and Brusven (1998) also observed an inverse relationship between herbivore abundance and algal biomass that counters much of the literature. They attributed their finding to the specific BMI taxa living in their observed stream communities, noting they had different Ephemeroptera and Coleoptera that could have been feeding on detritus as an alternate food source during periods of low algal abundance. The SiZer analysis also detected negative linear associations in %EPT and %Shredders and a positive linear association in DipteraRich. Although the associations may be spurious because they were also not detected by the SegReg analysis, the associations did correspond with my predicted directions of response for these metrics. For example, I expected to see decreases in EPT and shredder abundance with increases in agricultural cover, as pollution-sensitive EPT, many of which are shredders, would be less tolerant to the effects of agriculture such as increasing nutrients (Elbrecht *et al.,* 2016) and fine sediments (Burdon *et al.,* 2013; Niyogi *et al.,* 2007). In contrast, dipterans are generally more tolerant to stressors associated with agricultural land use and are known to increase in diversity in streams exposed to intensive agricultural land use (Hilsenhoff, 1977).

5.2 Scenario #2

I hypothesized that BMI communities collected in streams with substantial amounts of agricultural land cover in the riparian corridor (i.e., ≥ 75%) would exhibit similar composition irrespective of the amount of agricultural land cover in the catchment area. This prediction was largely supported by the diversity and community composition metrics, as all but EPTRich were found to be unassociated with agricultural land cover at the catchment scale. The sensitivity of many EPT taxa to common agricultural stressors has been widely reported and indeed the metric has frequently shown a response to agricultural land cover (e.g., Lange *et al.,* 2014; Yates *et al.,* 2014; Lenat and Barbour, 1994). For example, Burdon *et al.* (2013) presented results where EPT showed a strong nonlinear association with fine sediments in streams where loss of riparian vegetation contributed to the streambed exceeding 20% fines. EPT taxa have also been linked to several other stressors linked to agriculture such as temperature (Sponseller *et al.,* 2001; Sweeney, 1993); dissolved oxygen (Weigel *et al.,* 2003), nutrients (Hilsenhoff, 1987) and insecticides (Wallace *et al.,* 1996).

In contrast to my results for taxonomic diversity and composition, I found that six of the seven functional trait metrics exhibited a relationship with variation in agricultural cover at the catchment scale. It is important to note that for metrics where thresholds were observed, the threshold occurred at higher percentages of agricultural cover in the catchment (i.e., 72% to 85%). This finding suggests that even when the riparian corridor is dominated by agriculture, the land cover patterns in the catchment appear to influence the functional attributes of stream BMI communities. Two potentially complementary mechanisms could explain how the pattern of land use at the catchment scale is influencing the BMI traits. First, the proportionally small remnants of forest cover in the catchment may be serving to intercept a portion of agricultural runoff from upland areas or are providing key watershed processes such as infiltration and organic matter processing that maintain stream ecological conditions (Allan *et al.,* 1997; Roth *et al.,* 1996). Second, in watersheds with less agricultural activity, stream biota are exposed to a reduced level of stress resulting in conditions that select for different assemblages of BMI traits. However, similar to Scenario #1, the lack of representation of catchments with lower proportions of agricultural land cover (i.e., < 44%) limits my ability to provide more definitive insight into the likely mechanisms. Consequently, I was unable to detect ecological changes in metrics that may have been highly sensitive to agriculture land use in the catchment in the absence of riparian vegetation. Indeed, other studies have shown that many taxonomic changes occur at low levels of human activity, as the most sensitive species are extirpated (Yates and Bailey, 2011; Lenat and Crawford, 1994). This may explain why I only detected differences in trait-based metrics, which showed thresholds at higher levels of agricultural land cover (i.e., 72% to 85%), as opposed to diversity based metrics. Despite this limitation, my study does demonstrate that increased agricultural cover at the catchment scale may lead to the additional loss of ecological function in streams even after riparian vegetation has been removed.

My observation of a continued importance of the amount of agricultural land cover in the catchment in the absence of riparian forest cover has implications for regional land management strategies. First, it suggests that taking steps to conserve remaining forested areas in the upland areas of the catchment could protect existing levels of functional traits and functional diversity within the stream biota where agricultural land use is below identified thresholds. Second, the identification of independent effects of agricultural land use outside the riparian corridor indicates that implementation of best management practices (BMPs) in the catchment area may enhance ecological conditions in the stream. BMPs refer to any mechanism (e.g., improved manure storage; conservation tillage) employed with the task of mitigating the effects of agricultural practices on surrounding aquatic ecosystems (Yates *et al.,* 2007). Indeed, past studies have shown that BMPs in the catchment area can improve instream ecological conditions. For example, Selbig *et al.* (2004) noted that BMI communities responded positively to erosion-control and storm-runoff BMP's. Likewise, Wang *et al.* (2002) observed that sufficient BMP implementation at the catchment scale was essential for restoration of coldwater fish communities. Furthermore, because my study found evidence of ecological thresholds at high amounts of agricultural land cover, BMPs could be particularly effective for enhancing ecological conditions in catchments where land use exceeds the identified ecological thresholds. BMPs could thus provide a mechanism for reducing stream exposure to stressors without reducing the extent of agriculture land use (Yates *et al.,* 2007; Moore and Palmer, 2005).

5.3 Scenario #3

Analyses of the third scenario of an agricultural land cover gradient at a riparian corridor scale (i.e., 0% to 95%), surrounded by an agriculturally dominated catchment (i.e., ≥80%) also generated findings that supported my hypothesis. Indeed, as I hypothesized, this scenario resulted in the identification of the largest number of land use thresholds. Specifically, all diversity metrics exhibited thresholds that shared negative associations with increasing agricultural land cover in the riparian corridor. Of these three metrics, I had expected DipteraRich to increase but the overall community and EPT richness to decline. However, my findings suggest that losses of Diptera and EPT taxa occurred with declining riparian forest cover. However, I did not observe any associations between variation in agricultural cover in the riparian corridor and the community composition metrics. In contrast, five of the seven functional traits were identified to exhibit threshold responses. My observations of numerous associations between both taxonomic richness and functional trait metrics with increasing agricultural land cover in the riparian corridor, support past findings that riparian corridor conditions are strongly linked with instream ecological conditions. Management efforts to preserve existing riparian forests as well as restoration of devegetated riparian corridors should thus be increased.

The land use thresholds observed in Scenario #3 varied with the statistical analysis used. I saw relatively lower thresholds with the SiZer analysis, where all but one functional trait metric had thresholds below 30% agricultural land use in the riparian corridor, and where all diversity metrics had thresholds of no more than

20% agricultural land use. The taxonomic and functional trait metrics assessed in the SegReg analysis responded in the same direction as that of the SiZer analysis but revealed significantly higher thresholds (44% to 77% agricultural land use), suggesting the importance of implementing various means of threshold analyses. Of these two sets of thresholds, the percentages derived from the SiZer analysis are most similar with past studies. For example, King *et al.* (2005) observed strong changes in BMI composition if there was more than 22% developed land within a 250 m wide riparian corridor. Similarly, in a study of forest harvesting effects, Nordin *et al.* (2009) observed a significant increase in negative responses of BMI community indicators when 30% of the riparian forest was harvested along a 10 m wide headwater stream corridor. Furthermore, the SiZer analysis identified a clear distinction between the taxonomic richness metrics, EPTRich and DipteraRich, which had small thresholds (8% and 6% agricultural land use, respectively), when compared to the functional trait metrics that revealed thresholds ranging from 18% to 46% agricultural land use. Thus, functional trait metrics appear to have potential use for detecting changes in more advanced stages of agricultural development in the riparian corridor that taxonomic metrics might not detect.

5.4 Disproportionate Importance of Riparian Corridor

Most studies that have assessed stream communities at multiple landscape scales have been observing agricultural land cover gradients in both the riparian corridor and catchment simultaneously (e.g., Pearson *et al.,* 2016; Richards *et al.,* 1997; Roth *et al.,* 1996). However, Allan (2004) proposed that the majority of these past catchment-scale studies were merely reporting a trade-off affect where biological metrics were negatively associated with agricultural land use in the catchment and positively associated with forest cover in the riparian corridor. Allan (2004) further suggested that an alternative approach to a study design assessing a matrix of land use and spatial scales would be required to achieve a greater understanding of the influence that land use has on the ecological integrity of the stream. In accordance with this criticism, I configured my study design to ensure isolation of the riparian corridor and catchment landscape scales, thus independently establishing the role of each scale and allowing direct comparison of the effects of the riparian corridor and catchment scales. Integrating the results of Scenario #1 and #2, where I assessed streams with an agricultural land use gradient in the catchment, I saw comparatively few associations between agricultural land use and BMI metrics, relative to Scenario #3 when the agricultural land use gradient was in the riparian corridor. My finding of more numerous and stronger associations between ecological conditions and agriculture in the riparian corridor is consistent with several past studies. For example, Van Sickle and Johnson (2008) found that a fish IBI (Index of Biotic Integrity) was most strongly associated with agricultural land use within a narrow riparian corridor as opposed to larger landscape scales further from the stream. Lammert and Allan (1999) also found that land use within a 100 m riparian corridor was significantly related to the biotic integrity of both fish and BMI, whereas land use in the surrounding catchment showed no relationship. Likewise, Peterson *et al.* (2011) conducted a study where BMI metrics were most strongly associated with distance weighting models that attributed the greatest influence of land use to areas within the riparian corridor. In addition to greater number of metrics responding to increases in agricultural land use in the riparian corridor compared to the catchment area, there was also a disparity in the amount of agriculture that triggered a threshold response. Thresholds associated with agriculture in the riparian corridor were substantially lower (i.e., less than 46%) compared to thresholds for catchment land use (i.e., greater than 72%). Although the shortened catchment gradient limits my ability to make a definitive conclusion as to whether the difference in threshold levels is indicative of the disproportionate importance of land use in the riparian corridor, a study by Fitzpatrick *et al.* (2001) does support this interpretation of my data. Fitzpatrick *et al.* (2001) found that 10% agricultural land cover in the riparian corridor was associated with decreased fish IBI scores and further to that, if agricultural land use remained below 10% in the riparian corridor, catchments with agricultural land cover between 50% and 60% still maintained high fish IBI scores. Overall, my findings provide clear evidence that watershed managers are likely to best achieve river health goals by focusing landscape rehabilitation and protection efforts on the riparian corridor. Indeed, given the disproportionate response of BMI to agricultural activity in the riparian corridor it is likely that resources spent on restoration of riparian vegetation will provide greater benefits to stream health than equal amounts of resources spent on the upland areas of the catchment.

5.5 Taxonomic versus Trait Metrics

Throughout my study I observed that BMI functional trait metrics were more frequently associated with changes in agricultural land cover at both the catchment and riparian corridor scales than taxonomic metrics. In fact, of the 19 associations

I observed between BMI metrics and agricultural land use, 13 described a functional trait. My finding of strong responses of functional traits to land use is consistent with predictions of the Habitat Templet Theory (Townsend & Hildrew 1994; Southwood 1977) as well as other studies comparing the frequency and strength of associations of taxonomic and trait-based measures of BMI composition with agricultural land use (e.g., Doledec *et al.,* 2006; Doledec *et al.,* 2011). Indeed, Doledec *et al.* (2006) also found that functional traits provide increased sensitivity to the effects of land use and thus serve as an effective mechanism for monitoring the varying responses between low and high agricultural pressures. My findings thus support the calls for land use managers to include functional traits in stream biomonitoring programs to complement taxonomic metrics (Culp *et al.,* 2010). Moreover, my finding that eight of the eleven associations I observed between BMI metrics and agricultural land use at the catchment scale were with functional traits, suggest that trait-based metrics may be particularly effective at detecting the effects of catchment scale changes in land use in regions where human activity is already pervasive. This conclusion is concordant with Young and Collier (2009) who also demonstrated that functional traits would be a useful tool in biomonitoring programs for their ability to detect subtle changes at intermediate stages of the land use gradient. Furthermore, my finding that many functional traits exhibited change at greater amounts of agricultural land cover (i.e., 18% to 46%) in the riparian corridor than taxonomic metrics (i.e., 6% to 20%), albeit just from the SiZer analysis, suggests functional traits could provide complementary information to taxonomic metrics at the riparian corridor scale. Together my findings indicate that BMI trait based metrics are sensitive to differences in agricultural land use in settings where all streams are exposed to extensive agricultural cover. These metrics could thus be important biomonitoring tools in an agriculturally dominated region, such as southwestern Ontario, where land use managers need indicators sensitive to the predicted future intensifications of land use associated with growing global food demand (Genito *et al.,* 2002). These trait-based metrics may also be applicable for assessment of riparian corridor restoration projects, as my findings indicate that changes in trait composition will be detectable when moderately sized areas of the riparian corridor have been restored. Trait-based metrics could thus provide managers with earlier evidence of the effectiveness of restoration efforts.

5.6 Application of Land Cover Thresholds

My study identified several metrics that exhibited a threshold response to agricultural land cover at both the riparian corridor and catchment landscape scales. Thresholds can be effective for management activities as they provide empirically based targets for land use planning. However, although thresholds were identified by my study, caution is required in applying the thresholds as management targets for land use planning because of the substantial amount of uncertainty around the specific threshold values I observed. For example, I found error terms from thresholds identified by the SegReg analysis to encompass a minimum of 9% to a maximum of 20%. Thresholds analyses have often been found to result in substantial uncertainty (e.g. measurement errors, variability in the subject being assessed, inflated Type I errors) (Toms and Villard, 2015; Andersen *et al.,* 2009). For example, Dodds *et al.* (2010) noted a range of uncertainty of nearly 30% for a threshold they identified in Total Phosphorus concentrations they identified using a threshold regression model. Utz *et al.* (2009) attributed the substantial uncertainty they observed in land use thresholds to inherent variation in localized variables such as instream habitat and condition of the riparian vegetation as well as variables interacting with one another, concluding these environmental phenomena will all add statistical noise to ecological data and make threshold detection difficult. In addition to the substantial uncertainty around individual threshold values, there were also large differences between specific BMI metric thresholds identified by the two threshold analyses, SiZer and SegReg, used in my study. For example, results of my assessment of effects of increased agricultural land cover in the riparian corridor (i.e., Scenario #3) identified thresholds between 6% and 46% using SiZer but from 44% to 77% using SegReg. Dodds *et al.* (2010) also noted substantial differences among thresholds associated with the statistical technique applied to the dataset. Although further research and development of statistical techniques for identifying thresholds would likely help address the issue of consistency among techniques, I also recommend increased levels of control and inclusion of extraneous variables in studies aimed at identifying thresholds. In my study the increased control of extraneous variables, such as catchment size and physiography came at the expense of sample size. Sample size has been shown to be associated with increased uncertainty around threshold values, although SiZer has been shown to be robust with sample sizes as low as 30 (Daily *et al.,* 2012), which may explain the differences observed between thresholds identified by SiZer and SegReg. Daily *et al.* (2012) also indicated that in addition to sample size, the frequency of observations across the gradient being assessed, as well as the parameters that researchers apply to their model, will all have an affect on the rate of threshold detection and the threshold value. Expanding my controlled design to include greater numbers of samples may thus assist in refining my thresholds, allowing for more ready application to management strategies. Despite the described limitations of the identified thresholds, I do believe that my thresholds could immediately be applied by managers as general guidelines for land use management using the precautionary principle. Such action would ensure protection for remaining riparian vegetation while also providing expectations against which benefits of BMP implementation and restoration activities could be assessed.

6.0 Future Research

Designing my study to determine independent thresholds in agricultural land cover for the riparian corridor and catchment areas that initiate change in the composition of stream BMI communities, has enabled me to provide managers with valuable, but preliminary, tools and targets for land use planning. However, additional research is needed to generate further understanding regarding the importance of agricultural land use acting at different scales, as well as to refine and increase the applicable scope of the generated thresholds. Moreover, I believe there is an opportunity to apply the design and findings of my study towards research aimed at objectively establishing stream ecosystem reference conditions within agriculturally-dominant landscapes where “pristine” conditions are absent, by providing empirically based and ecologically relevant criteria for identifying best available landscape conditions. Indeed, my findings suggest that catchments with undisturbed riparian corridors may serve as potential reference sites and I encourage future research to test this hypothesis. This knowledge will advance bioassessment practices and enable a wide application of reference condition assessment based approaches in regions exposed to extensive development pressures. Furthermore, because my study focused on sand substrate, I recommend that varying substrates (e.g. silt, clay, cobble) be assessed to test the regional applicability of my findings. Such research would enable a governing authority to apply specific metrics that are best predicted by regional physiographies. I also note my inability to apply a lower spectrum of the agricultural land use gradient in the catchment in Scenarios #1 and #2, a limitation that may have resulted in us not detecting catchment thresholds with either a disturbed or undisturbed riparian corridor. Research broadening this range would thus be useful in ensuring that the findings of this study are complete. Similarly, my study was unable to implement a fourth scenario incorporating an agricultural land use gradient at the riparian corridor scale when the catchment is largely forested, although I recognize catchments fitting the criteria of such a scenario are likely to be rare in agricultural environments. Overall, continuing research to establishing scale-specific relationships between land use and stream ecological conditions will enhance the ability of government agencies, conservation authorities, and municipalities to draft informed land use policies that aim to achieve a balance between maximizing agricultural development and conservation of stream ecosystem, and the many services they provide.

7.0 Conclusions

My study provided findings that related to the role of agricultural land use at both the riparian corridor and catchment landscape scales. By designing my study to assess three different landscape scale scenarios, the conclusions derived from each of the three scenarios were as follows:

Scenario #1:

- Few associations between BMI metrics and agricultural land use in the catchment, suggesting the riparian corridor may be playing an integral role in buffering the stream biota from the affects of the surrounding agricultural activity.

Scenario #2:

- Numerous associations, including multiple thresholds between the BMI functional traits and agricultural land use in the catchment. The thresholds were also occurring at higher percentages in the catchment (i.e., 72% to 85%), suggesting that even when the riparian corridor is dominated by agriculture, the land cover patterns in the catchment appear to influence the functional attributes of stream BMI communities.

Scenario #3:

- All diversity metrics exhibited thresholds that shared negative associations with increasing agricultural land cover in the riparian corridor, and five of the seven functional traits exhibited threshold responses. In contrast, no associations were observed among community composition metrics. These observations suggest that riparian corridor conditions may be strongly linked to instream ecological conditions.

More cumulatively, synthesizing the information from each of the three scenarios, I presented three major conclusions:

(1) The land cover in the riparian corridor is disproportionately important to stream benthic macroinvertebrate community conditions than that in the catchment.

(2) Trait-based metrics may be particularly effective indicators for detecting the effects of catchment scale changes in land use in agriculturally-dominant regions similar to that of southwestern Ontario. Trait-based metrics should be applied into current stream biomonitoring programs as a complement to the taxonomic composition metrics that are already in practice.

(3) The large degree of uncertainty surrounding many of the identified thresholds, as well as the sometimes substantial differences between the amount of agricultural cover identified by SiZer and SegReg to be the threshold, indicates caution must be applied in adopting these thresholds for management. However, I do think that my thresholds could be immediately applied by land use managers to provide general targets for riparian restoration and protection. These targets could then be refined as additional information becomes available through future research.

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Appendix A

United States Environmental Protection Agency's rapid habitat assessment protocol specific to low gradient streams.

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 __________

Appendix B

Metrics used to describe diversity, composition and functional attributes in benthic macroinvertebrate communities in southern Ontario streams.

Metric	Definition	Predicted Response	Citation
Community Composition Metrics			
%EPT	EPT is a taxonomic abbreviation for the Orders Ephemeroptera (Mayflies); Plecoptera (Stoneflies); and Trichoptera (Caddisflies). These three orders are widely viewed as being generally sensitive to disturbance. %EPT refers to the percentage of individuals that fall within any of the three EPT Orders relative to the total number of individuals within the entire community. %EPT = EPT Individuals / Total Individuals in Community x 100.	Negative	(Paller *et al.,* 2017; Barbour *et al.,* 1999).
%Diptera	Diptera is the invertebrate order that refers to the “true flies”. %Diptera refers to the percentage of total Diptera individuals relative to the total number of individuals within the entire community. %Diptera. = Diptera Individuals / Total Individuals in Community x 100.	Positive	(Bouchard, 2004; Barbour *et al.,* 1999).
*Diversity Metrics			
nTaxa	nTaxa is abbreviated from “Taxa Richness”. nTaxa refers to the presence of each unique macroinvertebrate taxon in a community (sample site). The presence of each unique taxon receives a score of “1”. The total score will amount to the Taxa Richness.	Positive	(Barbour *et al.,* 1999; Resh *et al.,* 1995).
EPTRich	EPTRich is abbreviated from “EPT Richness”. EPTRich refers to the presence of each unique EPT taxon in a community. The presence of each unique taxon receives a score of “1”. The total score will amount to the EPT Richness.	Negative	(Paller *et al.,* 2017; Barbour *et al.,* 1999).

Metric	Definition	Predicted Response	Citation
DipteraRich	DipteraRich is abbreviated from "Diptera Richness". DipteraRich refers to the presence of each unique Diptera taxon in a community. The presence of each unique Diptera taxon receives a score of "1". The total score will amount to the Diptera Richness.	Positive	(Barbour *et al.,* 1999).
Functional Trait Metrics			
%Small	%Small is abbreviated from %Small Body Size. For a benthic macroinvertebrate to be classified as "small", the length of the macroinvertebrate must be less than 9 mm. %Small refers to the percentage of "small" individuals relative to the total number of individuals within the entire community. %Small Body Size = Total Small Body Size Individuals / Total Individuals in Community x 100.	Positive	(EPA, 2012).
%Multivoltinism	Multivoltinism is defined as any macroinvertebrate that experiences more than one generation per year. %Multivoltinism refers to the percentage of multivoltinistic individuals relative to the total number of individuals within the entire community. %Multivoltinism = Total Multivoltinistic Individuals / Total Individuals in Community x 100.	Positive	(EPA, 2012).
%Shredders	Shredders are benthic macroinvertebrates that have evolved specialized mouthparts and feeding behaviour for shredding leaves. %Shredders refers to the percentage of shredding individuals relative to the total number of individuals within the entire community. %Shredders = Total Shredding Individuals / Total Individuals in Community x 100.	Negative	(Merritt and Cummins, 1996).
%Herbivores	Herbivores are benthic macroinvertebrates that feed upon algae or plants. %Herbivores refers to the percentage of herbivore individuals relative to the total number of individuals within the entire community.	Positive	(Merritt and Cummins, 1996).

Metric	Definition	Predicted Response	Citation
	%Herbivores = Total Herbivore Individuals / Total Individuals in Community x 100.		
%Clingers	Clingers are benthic macroinvertebrates that spend most of their time "clinging" or latching on to varying stream substrate. %Clingers refers to the percentage of clinger individuals relative to the total number of individuals within the entire community. %Clingers = Total Clinger Individuals / Total Individuals in Community x 100.	Negative	(Merritt and Cummins, 1996).
%Burrowers	Burrowers are defined as benthic macroinvertebrates that burrow within the fine sediments of streams, normally associated with pools. %Burrowers refers to the percentage of burrowing individuals relative to the total number of individuals within the entire community. %Burrowers = Total Burrowing Individuals / Total Individuals in Community x 100.	Positive	(Merritt and Cummins, 1996).
FBI	FBI is abbreviated from "Hilsenhoff Family-Level Biotic Index". It is an average of values regarding tolerance to organic pollution for a specified group of arthropod families in a community found within the western Great Lakes region. $\sum \frac{ni \times ai}{N}$ = HFBI; where n = number of individuals in the taxa (*i*); a = tolerance value of the given taxa (*i*); and N = total individuals in the community.	Positive	(Hilsenhoff, 1988).

*Note: All diversity metrics were based upon the taxonomically adjusted data as explained in the field sampling protocol.

Appendix C

A complete list of sample sites for each of the three scenarios that comprised my study design. The appendix lists the date the site was sampled, GPS coordinates, and both riparian and catchment land cover at each site.

Site Code	Sample Date	Easting	Northing	Datum	Riparian Ag (%)	Catchment Ag (%)
Scenario #1 (15AgR)						
GR0930	10/25/2006	562153	4845421	NAD83	7%	60%
GR1000	9/26/2007	571155	4837658	NAD83	11%	64%
GR1057	9/20/2007	534846	4833683	NAD83	6%	78%
GR1068	10/3/2006	565419	4830598	NAD83	0%	72%
GR1213	10/6/2006	540629	4821307	NAD83	4%	78%
GR1406	11/6/2007	565993	4797144	NAD83	6%	48%
*GR1489_2006	10/10/2006	532913	4796575	NAD83	13%	84%
LP0482	10/3/2007	543923	4754083	NAD83	11%	77%
*LP0507	9/28/2007	533200	4753711	NAD83	2%	82%
*LP0591	9/16/2006	530898	4751431	NAD83	2%	83%
*LP0630_2006	9/18/2006	536618	4747062	NAD83	5%	84%
LP0691	9/18/2006	530966	4742379	NAD83	4%	66%
LP0725	9/13/2006	524341	4739377	NAD83	4%	78%
LP0738	10/16/2006	530370	4734916	NAD83	4%	73%
LP0749_2007	10/2/2007	530361	4734916	NAD83	10%	71%
LP0895	10/27/2006	518577	4722498	NAD83	1%	68%
*TR1616_2006	9/15/2006	432189	4716726	NAD83	4%	80%
*GR164089	10/28/2014	527441	4857571	NAD83	0%	85%
GR167083	10/17/2014	540062	4828513	NAD83	0%	71%
GR168833	10/22/2014	531729	4813454	NAD83	1%	67%
GR169911	10/15/2014	546828	4801802	NAD83	0%	53%
GR170120	10/15/2014	546654	4799170	NAD83	0%	80%
*GR170454	10/10/2014	532925	4796573	NAD83	0%	84%
GR171589	10/28/2014	548149	4788548	NAD83	0%	69%
Scenario #2 (75AgR)						
GR1137	10/3/2006	570755	4823749	NAD83	96%	44%
**GR1194	10/10/2006	529551	4823761	NAD83	95%	87%
GR1248	2006-10-02	525393	4821560	NAD83	96%	78%

Site Code	Sample Date	Easting	Northing	Datum	Riparian Ag (%)	Catchment Ag (%)
GR1341	9/27/2006	518038	4812424	NAD83	79%	72%
**GR1995	11/6/2006	587202	4761031	NAD83	75%	87%
**LP0344	9/20/2006	545024	4763314	NAD83	79%	84%
**LP0555	9/18/2006	518939	4753566	NAD83	91%	88%
**TR0644	9/25/2006	478826	4799579	NAD83	92%	87%
**TR0664	9/29/2006	495933	4795117	NAD83	95%	94%
TR1358	9/15/2006	464017	4751021	NAD83	100%	67%
**TR2019	10/24/2007	392727	4679202	NAD83	75%	96%
**TR2079	10/24/2007	387091	4676377	NAD83	90%	96%
**GR164569	10/28/2014	530469	4851703	NAD83	78%	84%
**GR166968	10/22/2014	535618	4829505	NAD83	86%	90%
GR168876	10/17/2014	517197	4813279	NAD83	97%	79%
GR169251	10/3/2014	548618	4810022	NAD83	79%	70%
GR169281	10/17/2014	525773	4809588	NAD83	100%	78%
**GR169473	10/6/2014	518412	4806924	NAD83	94%	83%
GR174326	10/14/2014	528446	4775929	NAD83	100%	63%
**GR175144	10/13/2014	543886	4773327	NAD83	76%	90%
Scenario #3 (80AgC)						
GR1012	10/10/2006	530160	4844321	NAD83	43%	84%
**GR1194	10/10/2006	529551	4823761	NAD83	95%	87%
GR1211	10/10/2006	529433	4823075	NAD83	64%	91%
*GR1489_2006	10/10/2006	532913	4796575	NAD83	13%	84%
GR1536	11/2/2006	521967	4793561	NAD83	71%	89%
GR1632_2006	9/22/2006	541820	4783067	NAD83	42%	87%
GR1776	9/21/2006	548196	4774107	NAD83	26%	92%
GR1882	9/21/2006	536538	4772003	NAD83	68%	89%
GR1926	9/21/2006	529783	4771694	NAD83	26%	88%
**GR1995	11/6/2006	587202	4761031	NAD83	75%	87%
**LP0344	9/20/2006	545024	4763314	NAD83	79%	84%
LP0397_2006	9/20/2006	527867	4762144	NAD83	69%	86%
*LP0507	9/28/2007	533200	4753711	NAD83	2%	82%
LP0520	10/30/2007	533848	4754171	NAD83	42%	87%

Site Code	Sample Date	Easting	Northing	Datum	Riparian Ag (%)	Catchment Ag (%)
**LP0555	9/18/2006	518939	4753566	NAD83	91%	88%
*LP0591	9/16/2006	530898	4751431	NAD83	2%	83%
*LP0630_2006	9/18/2006	536618	4747062	NAD83	5%	84%
TR0606_2007	9/18/2007	510429	4794823	NAD83	32%	87%
TR0643_2007	10/12/2007	489344	4797094	NAD83	44%	93%
**TR0644	9/25/2006	478826	4799579	NAD83	92%	87%
**TR0664	9/29/2006	495933	4795117	NAD83	95%	94%
TR0827_2007	10/31/2007	475397	4788729	NAD83	28%	95%
TR0885	9/14/2006	506874	4779585	NAD83	30%	86%
TR0893	9/17/2006	519822	4777691	NAD83	53%	83%
TR1443	11/3/2006	450926	4729122	NAD83	19%	81%
TR1587	9/15/2006	439456	4717987	NAD83	32%	85%
*TR1616_2006	9/15/2006	432188	4716726	NAD83	4%	80%
TR1704	11/7/2007	422524	4709267	NAD83	71%	82%
**TR2019	10/24/2007	392727	4679202	NAD83	75%	96%
**TR2079	10/24/2007	387091	4676377	NAD83	90%	96%
GR163749	10/29/2014	555632	4863476	NAD83	65%	95%
*GR164089	10/28/2014	527441	4857571	NAD83	0%	85%
**GR164569	10/28/2014	530469	4851703	NAD83	78%	84%
GR165043	10/28/2014	527657	4847340	NAD83	26%	87%
**GR166968	10/22/2014	535618	4829505	NAD83	86%	90%
GR168908	10/17/2014	516612	4813072	NAD83	50%	81%
**GR169473	10/6/2014	518412	4806924	NAD83	94%	83%
GR169535	10/6/2014	511839	4806026	NAD83	66%	80%
GR170285	10/10/2014	532425	4797753	NAD83	21%	83%
GR170335	10/15/2014	537798	4797521	NAD83	62%	90%
*GR170454	10/10/2014	532925	4796573	NAD83	0%	84%
**GR175144	10/13/2014	543886	4773327	NAD83	76%	90%
GR175197	10/13/2014	536412	4772957	NAD83	16%	89%

* Denotes a site that applied to both Scenario #1 and Scenario #3.
** Denotes a site that applied to both Scenario #2 and Scenario #3.

Curriculum Vitae

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Citizenship: Canadian

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Master of Science in Geography with Environment and Sustainability Degree Candidate (currently enrolled); September 2014 to December 2017, The University of Western Ontario, Department of Geography, London, Ontario; (Supervisor Dr. Adam G. Yates)

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- GEOGRAPHY 2090A: Space Exploration (September, 2014 to December, 2014)

Watershed Assessment Technician: June, 2014 to September, 2014; the University of Western Ontario, Geography Department

Watershed Assessment Technician: May, 2012 to September, 2013; Lake Simcoe Region Conservation Authority

Awards:

Collaborative Environment and Sustainability Graduate Student Travel Award – The University of Western Ontario (2016); Issued by the Collaborative Environment and Sustainability Program

NSERC CREATE Grant – The University of Western Ontario (2014); Issued by the Watershed and Aquatics Training in Environmental Research (WATER) Program within the Canadian Rivers Institute

Danny Fitzgerald Memorial Award – Sir Sandford Fleming College (2012); Presented to the Fish & Wildlife Technician graduate who has demonstrated outstanding proficiency and enthusiasm for a career in the Fish & Wildlife field

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Volunteer Experience:

Parks Canada - St. Lawrence Islands National Park: October 2012

- Updated a herbarium database.
- Planted pitch pine trees and deerbeery (species at risk).
- Assisted biologist by collecting ticks for Lyme Disease research.
- Assisted with Emerald Ash Borer research.

Canadian Wildlife Service, Environment Canada: November 2011

- 2011 Species Composition Survey supervised by Norman North.
- Identified waterfowl wing and body samples; tagged, bagged, and labeled them.
- Aged and sexed waterfowl wing and body samples.

Muskellunge Hatchery, Manager Mark Newell, Sir Sandford Fleming College: October 2011 to November 2011

- Worked with the Ontario Ministry of Natural Resources and Muskies Canada to stock Muskellunge in the mouth of the Talbot River in Lake Simcoe.
- Injected metallic tags into the operculum of juvenile Muskellunge.

Atlantic Salmon Hatchery, Manager Chris Westcott, Sir Sandford Fleming College: October 2011

- Assisted with the Ontario Ministry of Natural Resources to transport and stock Atlantic Salmon into two Lake Ontario streams in the Cobourg, Ontario area.

Conferences, Seminars, and Professional Associations:

- **Centre for Environment and Sustainability EnviroCon (Theme: "What will Canada's (and Western's) Environment Look Like in 150 Years?"): March 8, 2017; the University of Western Ontario; London, Ontario; includes the Challenges in Sustainability Panel Discussion – "Measure to Manage: How to Measure Environmental Quality, as We Work Toward Sustainability?"**

- **Challenges in Sustainability Panel Discussion – "How can We Learn from Indigenous Approaches to Environment and Sustainability?": January 27, 2017; Richard Ivey School of Business, the University of Western Ontario; London, Ontario**

- **Challenges in Sustainability Panel Discussion – "How can Sustainability be a Shared Value for Consumers and Corporations?": December 2, 2016; Richard Ivey School of Business, the University of Western Ontario; London, Ontario**

- **International Association for Great Lakes Research (IAGLR) Conference: June 6-10, 2016; the University of Guelph; Guelph, Ontario**

- **Agricultural Impacts on Water (Canadian Water Network): March 23, 2016; the University of Guelph; Guelph, Ontario**

- **Centre for Environment and Sustainability EnviroCon: March 9, 2016; the University of Western Ontario; London, Ontario; includes the Challenges in Sustainability Panel Discussion – "Do We Stand A Chance? Translating Environmental Science To Policy"**

- **Challenges in Sustainability Panel Discussion – "How Do We Bridge The Inequality Gap While Striving Towards Global Sustainability?": January 29, 2016; Richard Ivey School of Business, the University of Western Ontario; London, Ontario**

- **The Paris Climate Conference – Behind Closed Doors: December 16, 2015; The University of Western Ontario; London, Ontario;** Dr. Radoslav Dimitrov; Government Delegate for the European Union and Republic of Bulgaria

- **Challenges in Sustainability Panel Discussion – "How Can We Conserve Biodiversity in the Face of the 6th Mass Extinction?": November 13, 2015; Richard Ivey School of Business, the University of Western Ontario; London, Ontario**

- **Challenges in Sustainability Panel Discussion – "Two Degrees of Separation: What Lies Between Human Behaviour and the Climate Threshold?": October 2, 2015; Richard Ivey School of Business, the University of Western Ontario; London, Ontario**

- **12th Annual Earth Day Colloquium; April 9, 2015; the University of Western Ontario; London, Ontario**

- **Global Climate Governance and Canadian Policy: Looking Forward to Paris 2015; March 27, 2015; CIGI Campus / Balsillie School of International Affairs; Waterloo, Ontario**

- **Challenges in Sustainability Panel Discussion – "Interdisciplinarity: Buzzword, Baloney, or Saviour?": March 6, 2015; Richard Ivey School of Business, the University of Western Ontario; London, Ontario**

- **Challenges in Sustainability Panel Discussion – "Is Canada Water-secure?": February 6, 2015; Richard Ivey School of Business, the University of Western Ontario; London, Ontario**

- **Challenges in Sustainability Panel Discussion – "Can Ontario Transition to Renewable Energy?": January 16, 2015; Richard Ivey School of Business, the University of Western Ontario; London, Ontario**

- **Canadian Aquatic Biomonitoring Network (CABIN): December 2-3, 2014; Guelph, Ontario**

- **Challenges in Sustainability Panel Discussion – "Is a Zero-waste London possible?": November 14, 2014; Richard Ivey School of Business, the University of Western Ontario; London, Ontario**

- **What is Sustainability?: September 23, 2014; Richard Ivey School of Business, the University of Western Ontario; London, Ontario;** Dr. Tima Bansal, Canada Research Chair in Sustainability and Director of the Centre for Building Sustainable Value

- **Canadian Rivers Institute (CRI): Watershed and Aquatics Training in Environmental Research Program (WATER): September 2, 2014 to present; Fredericton, New Brunswick;** CRI Student Representative for the University of Western Ontario
- **Paddling Together: Integrative Traditional and Western Water Knowledge (Canadian Water Network): August 25-29, 2014; North Bay, Ontario**

Certifications and Licenses:

Environmental Science

- **NSERC Collaborative Research and Training Experience Program (CREATE) WATER Certificate;** Issued by the Canadian Rivers Institute on June 7, 2017

- **Watershed and Aquatics Training in Environmental Research Certificate;** Issued by the Canadian Rivers Institute on December 1, 2016

- **Aquatic Environmental Techniques Certificate;** Issued by the Canadian Rivers Institute on December 1, 2016

- **Professional Science Certificate;** Issued by the Canadian Rivers Institute on December 1, 2016

- **Certificate of Completion for the course Canadian Aquatic Biomonitoring Network (CABIN); Classification: Project Manager**
  - Completion of Module 1: Introduction to CABIN
  - Completion of Module 2: Field Sampling Using Standard CABIN Protocols
  - Completion of Module 3: Sample Processing and Taxonomy
  - Completion of Module 4: Study Design and the Statistics of Model Building
  - Completion of Module 5: Assessment and Reporting Using Standard CABIN Protocols
  - Completion of field training – Lowville, Ontario at Bronte Creek – June 22, 23, 2016 (supervised by Timothy Pascoe – Environmental Scientist with Environment & Climate Change Canada)

- **Certificate of Completion for the course Practical Hydrology, Hydrometry, and Geomorphology;** Issued by the Canadian Rivers Institute on July 31, 2015

- **Backpack Electrofishing Certification;** Issued by the Canadian Rivers Institute on May 25, 2015

- **Swiftwater Safety Rescue Technician Level 2;** Issued by Rescue Canada and Instructor Rob Lemmon through the Canadian Rivers Institute on May 23, 2015; Expires on May 23, 2018

- **Advanced Wilderness and Remote First Aid Certification;** Issued by the Canadian Rivers Institute on May 22, 2015

- **Certificate of Completion for the course Ontario Fish Identification Workshop;** Issued by the Royal Ontario Museum on May 6, 2015

- **Certificate of Completion for the course Integrated Forum: Ecosystem Management;** Issued by the Canadian Rivers Institute on April 26, 2015

- **Certificate of Completion for the course Benthic Macroinvertebrate Identification Workshop;** Issued by the Canadian Rivers Institute on January 24, 2015

- **Firearms Possession and Acquisition Licence (PAL) for Non-Restricted Firearms;** Issued by the Chief Firearms Officer of Ontario under the authority of the Firearms Act, Statutes of Canada; Expires on September 1, 2019

- **Ontario Hunting and Fishing Licences and Outdoors Card;** Updated and issued by the Ontario Ministry of Natural Resources; Originally issued by the Ontario Ministry of Natural Resources – 2000; Expires on December 31, 2018

- **Class 2 Backpack Electrofishing Training Course Certification;** Under new certification regulations – completion of course in full compliance with the curriculum guidelines recommended by the Ontario Ministry of Natural Resources (OMNR Policy F1.3.01.01) certification is valid for three years after the date of course completion; Issued on April 15, 2012; Issued on April 20, 2013; Originally issued by the Ontario Ministry of Natural Resources – July, 1995

- **Fur Harvest, Fur Management & Conservation Course Certification;** Issued by the Manager, Wildlife Policy Section, Ministry of Natural Resources; and the Ontario Fur Managers Federation President; Issued on April 2, 2013

- **Ice Safety / Rescue Certificate;** Issued on February 23, 2013

- **North American Wildlife Technology Association Technician Designation;**; Issued on June 1, 2012

- **Ontario Benthos Biomonitoring Network Certification;** Having completed Ontario Benthos Biomonitoring Network course requirements (Theory and Identification Test 90% competency); Issued on April 23, 2012

- **Water Safety Exercise Certification;** Issued March, 2012

- **Certificate of Completion for the course Radio and Ultrasonic Telemetry for Fish and Wildlife;** Issued December, 2011

- **S-100 Forest Fire Fighting Certification;** Issued by the Ontario Ministry of Natural Resources, Sir Sandford Fleming College, and Canadore College; June, 1999

- **General Radio Telephone Operator's Certificate (Aeronautical);** Issued by the Ontario Ministry of Natural Resources – June, 1995

- **Bear Safety Course Certification;** Issued by the Ontario Ministry of Natural Resources – June, 1995

Education

- **The Teaching Assistant Training Program (TATP) Certification;** Issued by the University of Western Ontario on August 17, 2014

- **Ontario College of Teachers Certificate of Qualification and Registration;** Updated and Issued on January 3, 2017; Originally issued on June 23, 2003

Professional Skills Development

- **Certificate of Completion for the course Understanding Personality Profiles;** Issued by the Canadian Rivers Institute on August 20, 2015

- **Certificate of Completion for the course Time Management;** Issued by the Canadian Rivers Institute on June 30, 2015

- **Certificate of Completion for the course Effective Communications;** Issued by the Canadian Rivers Institute on February 26, 2015

- **Certificate of Completion for the course Building and Understanding Learning Cultures;** Issued by the Canadian Rivers Institute on February 12, 2015

- **Certificate of Completion for the course Technical Writing I: The Basics;** Issued by the Canadian Rivers Institute on December 30, 2014

First Aid and Safety

- **WHMIS *NEW*;** Issued by the University of Western Ontario on July 7, 2016; Expires on July 7, 2019

- **Comprehensive WHMIS Training Certificate of Completion;** Updated and issued by the University of Western Ontario on June 23, 2014; Expires on June 23, 2017; Originally issued by the Ontario Ministry of Natural Resources – June, 1995

- **General Laboratory Safety and Hazardous Waste Management Training Certificate of Completion;** Issued by the University of Western Ontario on August 21, 2014

- **Emergency First Aid & CPR (C);** Successful completion of training; Issued by Instructor M. Smith on June 20, 2013

- **Worker Health and Safety Awareness Certificate of Completion;** Issued by the University of Western Ontario on July 2, 2014

- **Accessibility at Western (AODA) – Accessibility in Teaching Certificate of Completion;** Issued by the University of Western Ontario on June 24, 2014

- **Safe Campus Community – Preventing Harassment, Violence and Domestic Violence Certificate of Completion;** Issued by the University of Western Ontario on June 24, 2014

- **Heartsaver AED (C);** Issued by Instructor Robert B. Cotey in Waterloo, Ontario on December 21, 2010

Coaching and Athletics

Can-Fit-Pro Personal Trainer Specialist
- Issued on August 14, 2009

3M National Coaching Certification Program Coaching Card
- Issued on April 4, 2001

Vehicle Operation

- **Class "G" Ontario Driver's Licence;** Originally issued – 1994; Updated and Issued on July 13, 2015; Expires on September 1, 2020

- **Pleasure Craft Operator Card;** Issued on May 29, 2011

Past Work Experience:

Can-Fit-Pro Personal Trainer Specialist: October 2010 to August 2011; Vitalogy Fitness – Kitchener, Ontario

Can-Fit-Pro Personal Trainer Specialist: January 2009 to October 2010; Achieve Fitness – Guelph, Ontario

Store Manager: October 1999 to November 2008; Foot Locker Canada – North Bay, Ontario (Northgate Square) and St. Catharines, Ontario (Pen Centre)

Long-term Occasional Contract Teacher: October 2004 to September 2006; Nipissing-Parry Sound Catholic District School Board (St. Joseph-Scollard Hall Catholic Secondary School); North Bay, Ontario
- Grade 9 Science (Academic); Grade 10 Science (Applied)
- Grade 9 Geography (Academic and Applied); Grade 11 Physical Geography (University/College)
- Grade 10 Canadian History in the Twentieth Century (Academic); Grade 11 History to the Sixteenth Century (University/College)
- Grade 9 Religion (Open); Grade 10 Religion (Open); Grade 11 World Religions (Open)
- Grade 9 Learning Strategies (Open)
- Grade 9 Integrated Technologies (Open)

Occasional Contract Teacher: September 2004 to October 2004; Near North District School Board (Chippewa Secondary School); North Bay, Ontario

Occasional Contract Teacher: June 2003; Simcoe Muskoka Catholic District School Board (St. Nicholas Catholic School); Barrie, Ontario

Watershed Field Technician: June 1995 to September 1996; Ontario Ministry of Natural Resources – North Bay, Ontario