Best Practices Around Online (Formerly Offline) Ponds in Urban Stream Restoration: A Waterloo, Ontario Case Study

by

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AUTHOR'S DECLARATION

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

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Abstract

Monitoring successful urban stream restorations can provide guidance for best practices for restoration design. My case study was located at Critter Creek, a tributary to the Grand River, in northeast Waterloo, Ontario, Canada, where due to high flow and tight meanders, six constructed offline ponds have become connected to the main stream and are now online ponds. This project aimed to evaluate how these online (formerly offline) ponds are affecting the restoration of the stream. The majority of research on this topic has concentrated on ponds specifically constructed for stormwater management or on urban restored streams without ponds. In many restoration plans, offline ponds are proposed to compensate for cut-fill balances and/or for habitat diversity. The relationship of these offline ponds to the function and ecology of the channel has not often been assessed.

Benthic macroinvertebrates were used as indicators of restoration as a proxy for water quality. Using a Surber sampler, samples were collected in the reaches of the stream upstream of the inlet and downstream of the outlet of each pond. Comparative samples were taken from waterbodies that provided a restored stream without offline ponds and a reference stream. In the laboratory, all benthic macroinvertebrates were identified to the Family level.

Ecology-based metrics (EPT, functional feeding groups, etc.) and an index (Hilsenhoff FBI) were used to characterize the assemblages. The Percent Model Affinity (PMA) Method was used to determine the impairment of the streams and Mann-Whitney tests were conducted to determine if differences existed between the samples taken close to the ponds and those not close to the ponds. Those tests were also conducted to determine differences between Critter Creek and the reference and other restored stream.

PMA results from monitoring from previous years indicated that Critter Creek was an impaired stream, and this research shows that the stream is still impaired with PMA values less than 33.23% for 83 of 89 samples. Mann-Whitney tests showed that the location in Critter Creek, whether it be adjacent to an online pond or not, does not have an effect on the benthic assemblages. They also indicate that the composition of the benthic assemblages in Critter Creek has not reached the same stage as those in Laurel Creek, the reference stream, or Clair Creek, the other restored stream. While

the habitat and functional requirements of organisms between all three streams is similar, the water quality present in Critter Creek is much lower than in the other two streams.

Further monitoring could be completed to determine the trajectory of the benthic macroinvertebrate assemblages in this restoration. However, given that monitoring for the past 10 years has shown that the ecosystem integrity of Critter Creek is not improving, it is advised that action be taken now to improve the stream restoration. The concepts of urban ecology were addressed in this study and a wider-scope monitoring program could be completed to determine the impact of urbanization on the restoration of Critter Creek.

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Chapter 1: Introduction

There are many important reasons for restoring urban streams. They offer habitats for flora and fauna, can help improve water quality and flood control, and have social and cultural effects for people living in the watershed (Walsh *et al*., 2005). They also offer important ecological and social services, and monetary values (Bernhardt and Palmer, 2007).

Given this, there has been an increase in urban stream restoration as part of urban planning (Lake *et al.*, 2007). A by-product of this push for restorations comes in the form of rushed restorations that are not based on sound science and do not necessarily include adequate monitoring programs (Bernhardt *et al*., 2005; Lake *et al*., 2007). Habitat creation is not necessarily a guarantee that aquatic communities will be supported (the “Field of Dreams” theory, Palmer *et al*., 1997). This thesis addresses theories behind urban stream restoration using a specific case study in Waterloo, Ontario, Canada.

1.1 Critter Creek restoration

Critter Creek restoration was undertaken as part of the Millennium Recreation Project (now RIM Park and Grey Silo Golf Course) that also included the construction of other recreational facilities in the adjacent land area (golf course, soccer fields, ball diamonds, parking areas). The purpose of the restoration was to provide an aesthetically pleasing area while increasing habitat within the existing ecosystem, a purpose that involves both human and nature (Stantec Consulting Ltd., 2000). The restoration occurred from March to July 2001 (Stantec Consulting Ltd., 2002). Stantec Consulting Ltd. completed the Environmental Planning Report while the restoration design was done by Schollen & Co. (Stantec Consulting Ltd., 2000). The original objectives from the City of Waterloo for the restoration were:

- “To restore the health and sustainability of the watercourse through the application of rehabilitation techniques which are focused on natural channel design principles”

- “To enhance the aquatic habitat potential of the creek through the removal of migration barriers and the diversification of in-stream conditions”

- “To establish a diverse vegetated riparian corridor along the length of [Critter Creek] to enhance aquatic habitat and achieve terrestrial habitat objectives”

- “To integrate into the design initiatives to enhance water quality within [Critter Creek] including linear wetlands and other features” (Stantec Consulting Ltd., 2000, 5.1)

While these restoration goals aim to improve habitat, they do not explicitly state goals with regard to aquatic organisms. As will be explored further, physical habitat improvements are often the main goals of restoration due to the ease of achieving that goal rather than reaching certain biological populations (Kondolf and Micheli, 1995). While habitat improvements are often major restoration goals, increasing habitat does not guarantee an improvement in all aspects of the ecosystem. It is important to consider the landscape processes throughout the watershed, not simply those within the stream (Selvakumar *et al*., 2010). Many abiotic and biotic factors other than habitat improvements play an important part in improving biological populations (Kondolf and Micheli, 1995).

Before restoration, Critter Creek had been moved and altered many times to accommodate agricultural practices in the area. Immediately before restoration it was a channelized agricultural drainage ditch (Stantec Consulting Ltd., 2000). Flow had been restricted as the creek had been used as a dumping ground for building materials and other debris. The culvert present at Woolwich Street and a drop structure at the cattle crossing path both impeded fish passage (Schollen, 2000). At the point where the creek met the floodplain there was also an old sugar shack that used old tires for fuel. From this burning contaminated waste was present and needed to be cleaned up (Stantec Consulting Ltd., 2000).

The other streams in this research are Clair Creek located in the City of Waterloo, and Laurel Creek, located on the outskirts of the city of Waterloo. Clair Creek is another restored stream however since it does not contain online ponds it is used as a reference with which to determine how the restoration of Critter Creek has been impacted by the online ponds. Laurel Creek is a stream that runs from rural Waterloo through urban Waterloo, with this study being conducted on the portion before it flows through a major urban area. It will be used as a reference stream since it is a natural stream, and therefore may be considered an ideal target for the restoration of Critter Creek. All of the study streams are located in the Grand River watershed.

The success of the Critter Creek restoration and the impacts of the online ponds were assessed based on the Theoretical Framework that follows. Since specific goals or ecological endpoints were not stated in the original restoration objectives, a reference condition approach, as advised by the Society for Ecological Restoration International (SERI, 2004), was taken. As was explained earlier Clair Creek and Laurel Creek are the reference streams in this research. This research will help to advance the knowledge of urban stream restoration since no other studies to date have evaluated the presence of online ponds on stream restoration. It will also add to the body of knowledge on benthic macroinvertebrate assemblages found within urban restored streams and the overall usefulness of using benthic macroinvertebrates in stream restoration evaluations. Finally, it will help to formulate ideas on the appropriate trajectory that restored streams should take and what the goals of restoration should be.

1.2 Objectives and hypotheses of research

The long-term objective of my study was to determine the impact of online ponds on restoration. Short-term objectives were used to achieve long-term objectives. My objectives were:

1. To use the Percent Model Affinity method to determine the state of impairment of Critter Creek, Laurel Creek and Critter Creek.
2. To determine how the online ponds are affecting the restoration of Critter Creek.
3. To determine the state of restoration of Critter Creek as compared to Laurel Creek and Clair Creek.

Null hypotheses

Benthic macroinvertebrates were used as indicators of restoration in this research. The main hypotheses, based on the objectives, are:

1. That there has been no change in the state of restoration in Critter Creek over the past 10 years.
2. That the PMA method will conclude that Critter Creek, Clair Creek and Laurel Creek are impaired streams.
3. That the macroinvertebrate communities collected in areas surrounding the ponds are from the same population as those collected from areas not adjacent to online ponds in Critter Creek.

4. That the macroinvertebrate communities collected at the pond inlets are from the same population as those collected at pond outlets.
5. That the macroinvertebrate communities collected in Critter Creek are from the same population as those collected in Laurel Creek.
6. That the macroinvetebrate communities collected in Critter Creek are from the same population as those collected in Clair Lake.

Alternate hypotheses

1. That there has been a change in the state of restoration in Critter Creek over the past 10 years.
2. That the PMA method will conclude that Critter Creek, Clair Creek and Laurel Creek are impaired streams.
3. That the macroinvertebrate communities collected in areas surrounding the ponds are not from the same population as those collected from areas not adjacent to online ponds in Critter Creek.
4. That the macroinvertebrate communities collected at the pond inlets are not from the same population as those collected at pond outlets.
5. That the macroinvertebrate communities collected in Critter Creek are not from the same population as those collected in Laurel Creek.
6. That the macroinvetebrate communities collected in Critter Creek are not from the same population as those collected in Clair Lake.

Chapter 2: Theoretical Framework

The theoretical framework for this study begins with a definition of ecological restoration and then examines stream restoration. Next the concepts of urban ecology and rural-urban gradients are examined. Finally, my research is set into context by these theories with an examination of urban stream restorations, their measures of success, and monitoring techniques.

2.1 Ecological restoration

Ecological restoration is “the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed” (SERI, 2004, p3). As per this definition, the Society for Ecological Restoration International (SERI) defines a recovered ecosystem as one that “contains sufficient biotic and abiotic resources to continue its development without further assistance or subsidy” (SERI, 2004, p3). While this is the ideal definition (currently under revision by SERI, SD Murphy pers. comm.), the difficulty in this process is nevertheless implicit. Further definitions of ecological restoration have advised that the goal of restoration should be to return the ecosystem to a less damaged state within the confines of some acceptable ecological limits (Falk *et al*., 2006). The problems associated with the “historical ecosystem” concept have been addressed by the idea of a “novel” ecosystem. A novel ecosystem is one that has responded to biotic and abiotic changes, resulting in an assemblage of species that has not been historically found. Based on the trajectory, resilience, resistance, thermodynamic efficiency, the provision of ecosystem goods and services, and social services of an ecosystem, the goal of a restoration might be better suited to hit the novel ecosystem rather than an historical ecosystem (Hobbs *et al*, 2009). It has been suggested that in highly disturbed environments, such as urban habitats, restoration projects might be more successful if the goal was to establish a novel ecosystem (Schaefer, 2009).

2.2 Stream restoration

As per the definition, a recovered ecosystem is evaluated based on its biotic and abiotic resources. The main factors affecting biotic communities affected by stream restorations are channel geomorphology, floodplain connectivity, riparian habitat, water temperature and chemistry, the availability of nutrients and energy resources, and biotic interactions (Konrad *et al.*, 2008).

The simplest stream restoration is a passive restoration; this involves removing the perturbation and subsequently allowing the ecosystem to recover naturally once the removal is complete. However, considering that there may be disturbances after the removal but before completed restoration, restoration might need to be continually implemented. An example would be continual dredging after a dam removal (Falk *et al*., 2006). This would reject the definition of a recovered ecosystem which includes that the ecosystem develop "without further assistance or subsidy" (SERI, 2004, p4).

Active restorations are more often a result of a deliberate method that followed from explicit experimental designs using natural and physical sciences as the framework. They typically involve habitat creation, creation of meander patterns, stream grading and substrate changes. They are generally more time and money consuming and may be more invasive to the surrounding landscape than passive restorations. Given that there is limited evidence that active restorations result in any more significant changes in stream benthic invertebrate colonization than passive restorations, the type of restoration project should be thoughtfully chosen by the restoration ecologist (Jahnig *et al*., 2010).

Of all these techniques, re-establishing aquatic habitats has become a large part of stream restoration due to the dependence of diverse communities on functioning habitat (Tullos *et al*. 2006). In Ontario many of these habitat restorations are triggered by the Fisheries Act and the Policy for the Management of Fish Habitat administered by Fisheries and Oceans Canada (Fisheries Act, 1985; DFO, 1986). The Policy has the three main goals of conserving, restoring, and developing fish habitat (DFO, 1986). It works under the "no net loss" guiding principle which means that it may be possible to "balance unavoidable habitat losses with habitat replacement" which would result in a stream restoration project (DFO, 1986, p7). The goal of restoring physical habitats may also be more feasible than restoring biological communities (Kondolf and Micheli, 1995). Furthermore habitat traits such as meander pattern establishment and substrate may respond faster than biological traits, therefore they would be easier to examine in a shorter monitoring program (Tullos *et al*., 2009).

Even though the majority of stream restorations focus on improving habitat, this will affect, but not necessarily restore, biotic communities (Selvakumar *et al*., 2010). In examining the literature, there seems to be an unwritten assumption that increases in habitat heterogeneity by channel reconfiguration will increase species diversity by increasing habitat, surface area, physical refugia and food availability (Palmer *et al*., 2010). As this sentence presages, recent meta-analyses have found conflicting evidence about whether this relationship is consistent. Palmer *et al*. (2010) reviewed 78 studies that increased habitat heterogeneity and found that only two of them reported increases in invertebrate density, the main test of success in those studies. The attributing factors to this failure were the lack of factual knowledge surrounding the theory of habitat heterogeneity and biodiversity and the fact that if the degradation (high pollutant loads, altered hydrological regimes) is sufficient enough the improvements of habitat heterogeneity may not have the opportunity to influence macroinvertebrate communities. Conversely, another meta-analysis completed by Miller *et al*. (2010) found that there were positive correlations between habitat restoration and macroinvertebrate richness, but had no significant effect on macroinvertebrate density. The main difference between these studies lies in the definition of "success". Palmer *et al*. (2010) based their results on the researchers deeming the increase in invertebrate density a biological success whereas Miller *et al*. (2010) simply examined invertebrate density and richness, regardless of the definition of success. While it is important to ensure that macroinvertebrate communities can thrive in the restored stream, the type of organisms present (ie. those that indicate good or bad water quality, or indicator species) are often a better measure of success. If organisms that are not characteristic of the reference stream are not present in the restored stream that the restoration should not be considered a success and changes should be made.

2.3 Urban ecology

Urban ecology is the study of the "distribution and abundance of organisms in and around cities, and on the biogeochemical budgets of urban areas" (Pickett *et al*., 2001, p129). Urban ecology is necessary as a subset of general ecological theory as it has been acknowledged that the ways that humans are altering ecosystems is not completely understood (Pickett *et al*., 2001). Studying these human-nature interactions may also lead to better ecosystem models, ultimately resulting in better solutions to environmental problems. Reducing environmental problems and destruction in urban areas is ultimately important to sustain cities as they are the basis for human quality of life. The two variables affecting ecosystems studied by urban ecology are the patterns and processes of ecosystems and the patterns of human activities, generally termed disturbances (Grimm *et al*., 2000). In the past the main area of study has been of Ecology in the City, with Ecology of the City becoming an emerging field (Pickett *et al*., 2001).

Ecology in the City is focused on the biophysical environment, soils, plants and vegetation, and animals and wildlife in urban areas (Pickett *et al*., 2001). Research in this area tries to determine if processes and ecological patterns differ in urban areas as compared to other environments. For example, researchers might investigate if forest succession varies between an urban tree stand and a rural one (Grimm *et al*., 2000). Some examples of the way that urban areas affect the natural ecosystems is through the heat island effect, pollution, physical disturbances to the soil, forest structure alterations and habitat fragmentation (Pickett *et al*., 2001).

Ecology of the City takes a systems-oriented approach and evaluates urban systems as a dynamic, connected, open system. It examines ecology as it relates to urban characteristics, such as species richness as it is related to human population size or biogeochemical budgets (Pickett *et al*., 2001).

The level of human influence on an ecosystem is not always equal based on the degree of urbanization, termed the "urban gradient". In general the most intense effects of urbanization on ecosystems is found in city centres and the anthropogenic influence decreases as distance from urban centres increases (McDonnell and Hahs, 2008). This concept is useful in urban stream restoration because it acknowledges that one type of restoration will not react the same way in all situations. It also is important in stream evaluations since the individual stream location must be taken into account when comparing streams or choosing restoration goals.

2.4 Urban stream restoration

Given the influence of urban ecology, streams restoration should be examined thorough that lens. It is important to find ways to maximize restoration benefits in an urban environment. Furthermore the original degradation of the restoration site must be examined in order to determine if there are times when an ecosystem cannot recover or there are thresholds beyond which a small disturbance will have a great effect on the system (Palmer and Bernhardt, 2006; Suding and Gross, 2006). There may be constraints to restorations in urban systems, such as fixed points like bridges and buried services, which may put restrains on the project design.

Reasons for restoring urban streams include: they offer habitats for flora and fauna, they can help improve water quality and other hydraulic characteristics, they may reduce erosion and they may have social and cultural effects, such as increases in outdoor education and physical activity, for the people living in the watershed (Walsh *et al*., 2005). They can also provide monetary value, or non-monetary ecological and ecosystem services (Bernhardt and Palmer, 2007). Often the restorations stem from the need to undo previous human modifications such as damming, channelization, channel modification or river diversions (Gregory, 2006). As these reasons are acknowledged by urban planners, urban restorations are becoming more useful and integrated into urban planning. In order to continue this trend and their effectiveness an interdisciplinary approach must be taken (Palmer and Bernhardt, 2006; Bernhardt and Palmer, 2007).

The characteristics of an urban stream differ from rural streams because of increases in impervious land cover, urban pollutants, and less natural habitat (Walsh *et al*., 2005). Instream barriers and human created obstructions may prevent water macroinvertebrate dispersion while land use changes may restrict land dispersion of adult aquatic invertebrates (Urban *et al*., 2006). Walsh *et al*. (2005) have described the *urban stream syndrome* where there is a hydrograph with steeper peaks, elevated concentrations of nutrients and contaminants, fewer small streams in the network, altered channel morphology and stability, and reduced biotic richness with the dominance and presence of more tolerant species and a decrease in sensitive invertebrates (Paul and Meyer, 2001; Meyer *et al*., 2005, Walsh *et al*., 2005). In terms of benthic macroinvertebrates, as compared to rural streams, urban streams will have less or no sensitive species, disturbance-tolerant taxa and a dominance of oligochaetes and chironomids (Walsh *et al*. 2005). These conditions, as well as the available land, political situation, for example the funding available or the value that the government places on stream restoration, and a lack of technical knowledge when applying urban restoration techniques to urban streams, present complicated issues in urban stream restoration (Bernhardt and Palmer, 2007).

2.5 Evaluating stream restorations

The elements of urban ecology influence the evaluation of the success of an urban stream restoration. Grimm *et al*. (2000) described the three fundamental drivers of human elements in an urban ecosystem as the flow of information and knowledge, the incorporation of culturally based values and perceptions, and the creation and maintenance of institutions and organizations. They show that ecosystem evaluation in an urban setting is constrained by the coarse scale environmental context and societal patterns and processes. Based on the interaction between land use and ecological patterns associated with this land use there will be changes in ecological conditions and changes in human perception and attitudes stemming from a project.

While the definition of restoration includes the measure of success as a recovered ecosystem, evaluating this can be problematic. This definition solely relies on ecosystem processes without involving the human element for urban stream restorations. The human measures of success could be based on many factors including cost, stakeholder satisfaction, aesthetics, infrastructure, recreational opportunities/community involvement, or helping to advance the science (Palmer *et al*., 2005). According to SERI, a restored ecosystem (SERI, 2004):

1. Has the same characteristic assemblage of species and community structure as those found in the reference stream
2. Contains indigenous species
3. Has all functional groups present, of if they are not present the groups have the potential to colonize naturally
4. Has a physical environment that can sustain reproducing populations necessary for the ecosystem's continued stability or development along the desired trajectory
5. Apparently functions normally and signs of dysfunction are absent
6. Is not a closed system – biotic and abiotic flows and exchange from the ecosystem to a larger ecological matrix or landscape
7. Is protected from threats to its health and integrity from the surrounding landscape
8. Is a resilient system - is as self-sustaining as the reference ecosystem

Considering that ecological endpoints are not always the basis to begin a restoration, restoration as a whole process should also be evaluated. Table 1 describes ways of measuring restoration success based on the whole restoration process rather than simply the state of the ecosystem. These measures mainly evaluate changes in ecological condition and do not incorporate human interactions.

Table 1. Criteria for a successful river restoration

<table>
<tr><td>Palmer et al., 2005</td><td>Jansson et al., 2005</td><td>Falk et al., 2006</td></tr>
<tr><td>A successful river restoration should ensure that:
An image is developed and guided by dynamic ecological endpoint
Endpoint not necessarily natural historical state, but should acknowledge the regional context
Ecosystems are improved
Needs to be measurable therefore requires monitoring
Acknowledges that ecological conditions will follow different paths and trajectories and that the measures may not recover at the same rate
Resiliency is increased
Ecosystems are dynamic, a steady state might not always be a resilient state
Important to reduce the amount of post-project maintenance that needs to be completed
No lasting harm is done
Need to think about impact of restoration on current and downstream populations
Ecological assessment is completed
Funding and time for pre- and post-project assessments must be included in the restoration plan</td><td>In addition to those criteria specified by Palmer et al., 2005, a successful river restoration should have a prediction or hypothesis of the intended effect of the restoration goal</td><td>A successful river restoration should:
1. Have explicitly stated goals
2. Should be informed by ecological knowledge
3. Should include pre- and post-restoration assessments - Results of these assessments being used to inform and guide other restoration efforts.</td></tr>
</table>

In order to fully understand the SERI criteria and Table 1, certain definitions are required. Resilience is the "ability of an ecosystem to regain structural and functional attributes that have suffered harm from stress or disturbance" (SERI, 2004, p7). Compared to a rural stream, disturbance could play a larger role in an urban stream due to the urban stream syndrome and direct anthropogenic influences (Walsh *et al*., 2005). Ecosystem integrity is "the state or condition of an ecosystem that displays the biodiversity characteristic of the reference, such as species composition and community, and is fully capable of sustaining normal ecosystem functioning" (SERI, 2004, p7). Ecosystem health is "the state or condition of an ecosystem in which its dynamic attributes are expressed within 'normal' ranges of activity relative to its ecological stage of development" (SERI, 2004, p7). Given that the definition of ecosystem health can often be fuzzy, for the purposes of this research it will be replaced with the term ecosystem integrity. While the science of ecological restoration advises certain measures of success, often in practice these cannot be achieved. Logistical and monetary constraints often restrict the way that the restoration is completed (Gillilan *et al*., 2005). Gillilan *et al*. (2005) critique the five criteria proposed by Palmer *et al*., (2005) and from a practitioner's point of view they feel that the criteria may not always be suitable. The guiding image may not always be completed without bias and may be constrained by budget or time. They stress that pre-restoration baseline data is required in order to assess whether the ecosystem is improved and to determine if no lasting harm has been done, however this data is rarely available. When it is available it is usually only a snapshot in time therefore it does not take into account natural fluctuations in the system.

2.6 Monitoring: Its importance and its relation to restoration goals and success

Regardless of the measure of success being used (see Table 1), success is generally evaluated from an ecological perspective based on such keywords as "increased resiliency", "improved ecosystems", "characteristic assemblage of species", and "all functional groups present". A characteristic assemblage of species and the functional group requirements could be determined from an historical or reference ecosystem. In order to evaluate the concepts behind these keywords, monitoring must be completed. Unfortunately, monitoring is not always included in the restoration plan. The lack of monitoring could be due to difficulties in the monitoring process, lack of foresight on the part of the parties initiating the restoration or lack of funding. Furthermore, when monitoring is included there is generally a greater influence on the physical rather than biological conditions of the stream (Kondolf and Micheli, 1995). Bernhardt *et al*. (2005) found in a study of restored streams in the United States that only 10 % of the projects received any monitoring. This can have an effect on future projects as there is no communication of ideas to help inform other restorations.

The information gathered from monitoring can be used for direct comparison where specific parameters from the restored ecosystem are compared to a reference ecosystem, as advised by SERI. Problems can arise when using too many parameters and some have improved while others have not The information could also be used in attribute analysis where the attributes are analyzed with respect to the characteristics of a restored stream. Trajectory analysis is when the monitoring data collected periodically at the restoration site is plotted to establish trends (SERI, 2004).

The monitoring data can also be used to assess specific end-point restoration goals, criteria for a successful river restoration (Palmer *et al*., 2004; Falk *et al*., 2006). The endpoints are either theoretical if setting targets, or empirical if using a reference ecosystem. Since restored streams are part of a dynamic ecosystem natural fluctuations could impede the ecosystem from reaching these targets, thus incorrectly deeming the restoration a “failure”. End-point restoration goals do not help future restoration projects because they do not show the mechanisms by which the ecosystem parameters did not meet the objectives (Falk *et al*., 2006). Adaptive management, where a project is designed and if once evaluated is found be under-performing the approach is changed, also requires monitoring data (Palmer *et al*., 2005). Unfortunately this data is not always publicly available, restricting the flow of knowledge and information regarding restoration failures to other scientists and practitioners (Palmer *et al*., 2005; O’Donnell and Galat, 2008).

Often restoration project evaluations are conducted without allowing sufficient time for ecological recovery (Palmer *et al*., 2010). Short-term recovery could be between 1 and 10 years of restoration with long-term recovery occurring around 25 or 30 years after restoration. When analyzing monitoring information and finally assessing the success of the restoration it is important to consider the levels of natural variation, the limitations imposed by activities within the watershed (i.e. urbanization, deforestation), the timeline of the restoration processes and the impact of the restoration processes on the biological communities (Tullos *et al*., 2006).

The timeframe of monitoring will also contribute to the definition of success and the ability to monitor success. The recovery rate following a restoration or disturbance may be influenced by the initial state of the ecosystem, with the idea that more degraded ecosystems (often urban) will show greater improvement after restoration (Miller *et al*. 2010). In their criteria, Palmer *et al*., (2005) point out that different metrics will have different trajectories and timelines for recovery. For example, a one year post-restoration study determined that in-stream habitat restoration had a significant positive effect on macroinvertebrate richness but no significant effect on macroinvertebrate density (Miller *et al*. 2010). In a constructed rock weir, benthic macroinvertebrate assemblages colonized after two months have been shown to be the same as those found in a similarly constructed weir two years old (Walther and Whiles, 2008). The time after restoration will affect the community composition, with reference streams and older restored streams having less variable community compositions. Typically there is also a drop in abundance and diversity of benthic macroinvertebrates immediately following the restoration, followed by a relatively rapid recovery to pre-restoration levels (Muotka and Syrjanen, 2007). Invertebrate densities have been shown to be greater in older restored and natural streams. The compositions of macroinvertebrate communities will change differently in restored and natural streams over a given time. For example the communities in a natural stream changed little over 2 years, however the community structure in a restored stream changed considerably in two years (Muotka *et al*., 2002).

In practice, while biological monitoring may occur, it is not always found to be the basis of what is considered a “successful” restoration. In a survey of 39 practitioners in the Mid-Western United States, Alexander and Allan (2006) found that success was based on the response of a specific indicator in only 11% of the cases. Furthermore, the most common response of success was based on the “positive effects on the human community”. They concluded that there is often a disconnect between what respondents felt was a successful restoration and what was considered an ecologically successful restoration.

Chapter 3: Methodological Framework

3.1 Using benthic macroinvertebrates

Benthic macroinvertebrates have many attributes which make them ideal parameters to compare between reference and restored streams. They do not typically move in the ecosystem and therefore are good indicators of the biotic state of the stream close to where they are collected. They also play an important role in the food web meaning that they are a part of larger ecosystem processes (Tullos *et al*., 2006). They have a broad range of trophic levels and pollution tolerances which enables analysis of cumulative effects (Barbour *et al*., 1999). They have a typical species specific response to environmental pollution, even if the source does not discharge pollutants (Bonada *et al*., 2006).

Restoration activities aim to increase resiliency and as per the definition of resiliency, the stream's response to disturbance. A single, natural hydraulic disturbance will generally reduce benthic density and richness (Konrad *et al*. 2008) therefore restoration will impact benthic density and richness. They also have short life cycles which enable short-term environmental variation to come to light. Logistically, they are easy to identify, are inexpensive to sample and are abundant in most streams (Barbour *et al*., 1999). Macroinvertebrates are well studied in the literature since results can be compared to other urban-gradient studies (Walsh *et al*., 2005).

One potential drawback of using benthic macroinvertebrates is the chance that the organisms will flee quickly when disturbed by the placement of the sampler on the stream bed. As well, some organisms are easier to collect than others (i.e. larger organisms) and there may be an uneven spatial distribution pattern of the organisms. However, since this uneven distribution is based on the interactions between habitat, habit and food availability, and therefore the impact of the restoration, it can be seen as a positive attribute. Macroinvertebrates may not respond to all types of impacts targeted at different trophic levels (e.g. herbicides and aquatic plants), and may also respond to stream processes not related to pollution, such as current velocity and substrate (Merritt *et al*., 2008). Since this study relates to *restoration* and not water pollution per se, and I use a reference condition, this non-response to pollution is not a contributing factor in the analysis. Since abundance and distribution may vary seasonally the organisms were collected in early and late summer (end of June to early July, and end of September to early October, respectively) in 2010 (Merritt *et al*., 2008).

Other parameters could be used for comparison between restored and reference streams. Kondolf and Micheli (1995) suggest that post project geomorphic evaluations are the best way to examine the restoration of the project. Chemical and physical water quality indicators could also be used however these are only representative of the conditions at the time of sampling whereas the presence of individual macroinvertebrates is influenced by past and present conditions, which brings in a temporal aspect to the analysis. Bacteria could also be used but they cannot distinguish between less impacted streams as well as macroinvertebrates can (Lear *et al*., 2009).

3.2 Using a Surber sampler

There are many different methods found in the literature that have been used for macroinvertebrate sampling and the type of sampling often does not have an effect on the results (Miller *et al*, 2010). Given this fact, there is no one “correct” method for sampling, and the sampling method may not be as important as other factors such as the site sampled, the season sampled or the year sampled (Borisko *et al*., 2007). The sampling approaches have changed over the years due to the results rendered by certain methods, the priorities of those completing the sampling, the complex spatial and temporal nature of freshwater systems and the required precision of results (Bonada *et al*. 2006).

A Surber sampler is a basic quadrat sampling method that requires that the area of sampling is known, that the sampling frame of the Surber sampling does not change, and that the organisms being sampled are relatively immobile, a characteristic of benthic macroinvertebrates (Krebs, 2009). It has a 930 cm^2 sampling frame and a 500 μm mesh net. As with all quadrat counts, the Surber sampling method could result in a positive bias since the people completing the sampling would rather count an organism than ignore it, meaning that organisms that might not be completely within the sampling frame are sampled (Krebs, 2009). Once the large rocks have been removed from the sampling area a stick is used to agitate the bed and dislodge benthic macroinvertebrates found there. There could be a variation with the depth of penetration using the stick within the Surber sampler frame (Merritt *et al*., 2008).

Other often-employed methods in macroinvertebrate sampling are the kick-and-sweep method, D-net sampling and electrofishing (eg. Gibson *et al*. 1996; Lepori *et al*. 2005; Jones *et al*., 2007). In this study, a Surber sampler was used mainly for logistical reasons. Field work was completed individually, therefore it was necessary to choose a method that facilitated one person sampling. Given that it is a quadrat sampling technique, the Surber sampler can give an accurate density of organisms present in the stream, whereas other methods may simply give a general inventory of organisms present. Furthermore since organisms were removed individually from the rocks organisms that are tightly attached to the rocks were collected. Those organisms may not have been collected had the travelling kick-and-sweep method been employed since foot kicks may not have dislodged them from the rocks.

While many studies suggest taking a subset of organisms from the sample (usually 100) (e.g. Hilsenhoff, 1982; Vinson and Hawkins, 1996; Barbour and Gerritsen, 1996; Jones *et al*., 2007; Beche and Statzner, 2009) during identification, I identified all of the organisms collected in one sample. This was necessary since not all of the samples contained 100 organisms, which is the result of using a Surber sampler. Compared to other methods where sampling could continue until 100 organisms were found, the Surber sampler only permits sampling of the number of organisms present in a specific area of streambed, and this may be less than 100 organisms. Gibson *et al*. (1996) developed three criteria for sampling method selection:

1. Minimize year-to-year randomization resulting from natural events
2. Maximize gear (equipment) efficiency
3. Maximize accessibility to targeted assemblage

My sampling method meets these three criteria. Since this was only a one-year study it may be possible that my results are the product of an anomalous year. By sampling twice a year, however, it was hoped that this would reduce the impact of a natural event. Gear efficiency was maximized in this case since the Surber sampler was inexpensive and the sampling could be completed by one person and was used at each site. Finally, since riffle and pool habitats were chosen this sampling method maximized the accessibility of targeted macroinvertebrate assemblages.

3.3 Sampling different habitat types

In this study, both riffle and pool habitats were sampled. Sampling multiple habitats gave the opportunity to gather a comprehensive macroinvertebrate inventory. Other habitats could have been sampled (ie. woody debris, banks) however in invertebrate analysis the type of habitat sampled rarely has an impact on the outcome (Gerth and Herlihy, 2006). Riffle organisms are often most sensitive to physical and chemical changes resulting from land cover changes, such as those seen in urban watersheds. Bank habitats can become refuges for organisms in time of stress or when riffle habitat is of low quality (Roy *et al*. 2003), however considering that there was high quality riffle habitat its sampling was not required. Since the plans included restoring riffles and pools through creation of meander patters, it was thought to be important to sample both these habitats.

There are conflicting results with regards to the number and diversity of organisms found in pool and riffle habitats. When little difference has been found between pool and riffle organisms, the study area is generally spread over a large spatial extent (Gerth and Herlihy, 2006; Chessman *et* al., 2007). This is probably due to the fact that local circumstances will influence riffles and pools in specific areas but this will average out over larger scales (Chessman *et al*., 2007). Bloksam *et al*., (2008) found that total and Ephemerptera, Plecoptera, and Tricoptera (EPT) taxa family richness, and Hilsenhoff's Family Biotic Index (FBI) were higher in multiple habitat rather than single habitat samples. Overall they found that richness metrics were higher and dominance metrics were lower in multiple habitat samples. Roy *et al*. (2003) found that non-chironomid taxa richness was higher in riffles but total insect density did not significantly differ throughout habitats.

3.4 Rationale for choosing this type of study

This was a Control-Impact (CI) study where samples are taken from a control, or reference, site and from an impact or test, in this case restored, site. The effectiveness of the restoration was considered the impact, with less restoration inferring more impact. Theoretically, as the restoration takes place and evolves the invertebrate community moves closer to the reference community. The samples from each site were compared statistically to determine if a difference was present. If yes, then an impact is present and the site has not yet been fully restored, when fully restored is defined as a return to a reference condition. This type of study is often seen in the literature due to the lack of monitoring planning and therefore pre-restoration data. While Control-Impact studies are valid, one difficulty lies in the process of identifying a suitable reference site (Osenberg *et al*., 2006). Since the reference site represents the anticipated end state of the test site, if an inadequate reference site is chosen the test is ineffective.

An ideal situation would have been to complete a Before-After-Control-Impact (BACI) study which does not require a reference site or to have portions of the test stream that act as the control. This type of study is useful because it incorporates specific biotic and abiotic influences at the study site that may not be present at the reference site. A BACI study was not used throughout the whole research because there was no “before” and “after” data available for Clair Creek, Laurel Creek, or Critter Creek coincident with the restoration. A BACI study was used to describe the impact that the restoration has had on Critter Creek (Osenberg *et al*., 2006). Samples were collected by Natural Resources Solutions Inc. from 2000 to 2004 as part of the monitoring program originally set up for the Critter Creek restoration plan and this data was used in addition to my sample collection.

3.5 Taxonomic identification of organisms

All organisms were identified to the Family level other than Hirudinea, Nematoda and Oligochaeta. Finer identification was not completed due to limitations in identification skills. Since Hirudinea, Nematoda and Oligochaeta were identified to a coarser level they were removed from the analysis. It was also determined that a finer identification level would offer limited improvement in the results based on the increase in time and effort required. Cao *et al*. (2003) and Yuan (2007) point out that some types and magnitudes of error have little effect on ultimate biological assessments and, furthermore, that attention to issues of data quality should be based on the ultimate uses. Chessman *et al*. (2007) found that identifying samples to genus only offered marginal improvement to results, however others (e.g. Hilsenhoff, 1988; Lenat and Resh, 2001) have found that greater taxonomic scale is more capable of detecting impact and the degree of impact on an ecosystem. While tolerance values may vary within a family, considering that relatively few organisms were collected in this study, the variation would have a minor impact (Chessman *et al*., 2007).

3.6 Reference site selection

Three streams were chosen for this study. All three study streams are located within the Grand River watershed. The control site in this study is Laurel Creek. In this case, regional references were chosen rather than site-specific references since there is no ecosystem upstream of Critter Creek or Clair Creek that could be considered “natural”. Clair Creek and Laurel Creek were chosen after speaking with local resources managers who identified them as potential sites in the area and then site visits were made to ensure that they would be good reference sites.

In general, one limitation with choosing a reference site is the difficulty of finding a site that is either an historical representation of the restored site, or situated along the ecosystem trajectory of the restored site. Furthermore, many sites are chosen by professional opinion and not on sound scientific basis (Bonada *et al*. 2006). While ideally a restoration will return a stream to the conditions found in a reference site, this may not always be possible, therefore comparison to a reference site may not always be ideal. This thinking was implied in the Percent Model Affinity analysis which did not require a reference site as it uses an ideal reference condition to compare the impairment of the restored stream (Barton, 1996). Another technique could be to use a continuum or gradient approach to choosing a reference stream where the goals for recovery are catered to the degree of urbanization of the impaired stream's watershed (Purcell *et al*. 2009).

Chapter 4: Methodology

4.1 Study Sites

Samples were collected from three streams in Waterloo, Ontario, Canada (43.28 N, 80.32 W, 334.4 m above sea level) (Figure 1). The soil in this area is typically sandy loam and the area received 899.1 mm of precipitation in 2009 (Presant and Wicklund, 1971; University of Waterloo, 2009). The following streams were sampled:

- Critter Creek (43.52 N, 80.50 W), a restored stream that contains online ponds. Samples were taken at the
    - Inlets to online ponds (5 reaches)
    - Outlets from online ponds (5 reaches)
    - Reaches not affected by online ponds (4 reaches)
- Clair Creek (43.46 N, 80.58 W), a restored stream with no online ponds (5 reaches)
- Laurel Creek (43.48 N, 80.60 W), a reference stream (4 reaches)

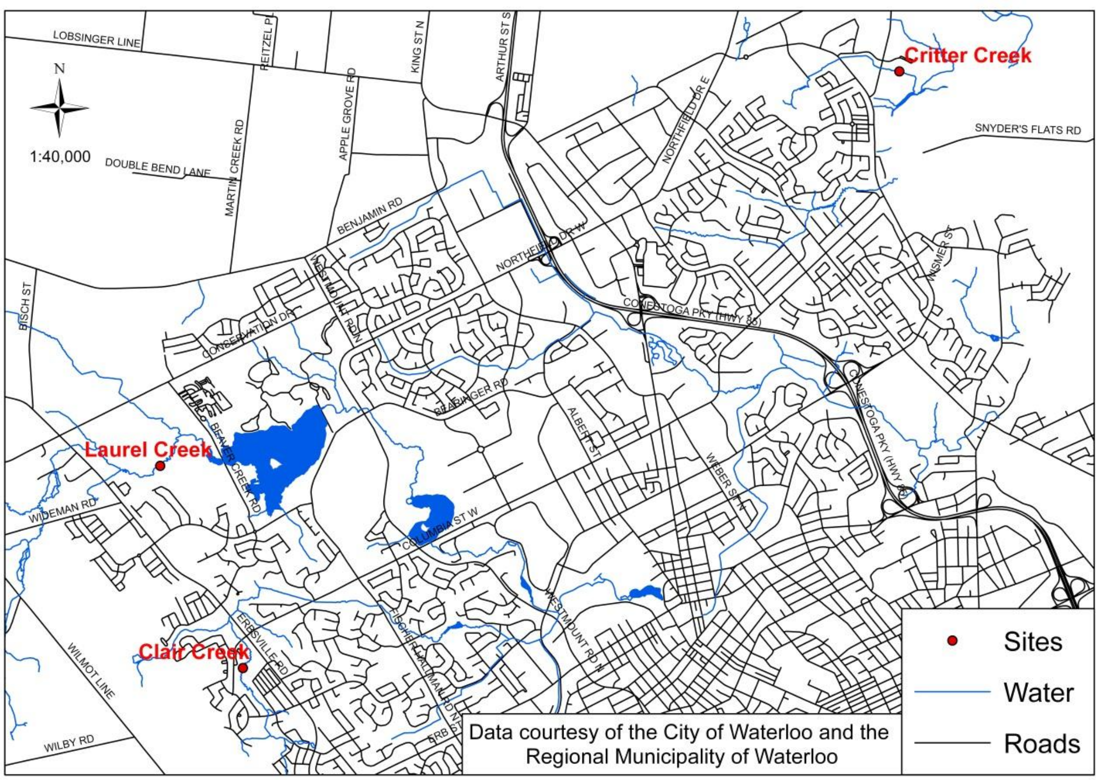

Figure 1. Location of study streams in Waterloo, Ontario, Canada.

Critter Creek, previously called Creek ‘C’ is located in northeastern Waterloo, Ontario. It begins west of University Avenue and flows into the Grand River, with this research taking place on the stretch from University Avenue to the Grand River floodplain (about 1 km). The stream has been restored as a type ‘C’ channel upstream of Woolwich Street and a type ‘B’ channel downstream of Woolwich Street with an average 1 m bankfull width throughout (Stantec Consulting Ltd., 2000).

Clair Creek begins in the Clair Hills subdivision in western Waterloo, Ontario and flows through Clair Lake eventually outletting into Silver Lake. Sampling was conducted in the restored portion of the stream located in the Clair Hills subdivision. The objectives of this restoration were to provide conveyance of low flows, enhancement of habitat, flood storage and to create and sustain a natural environmental feature which is an integral part of the future for the residents of Clair Hills (Stanley Consulting, 1998).

Laurel Creek is a natural stream that begins in rural Waterloo, Ontario and outlets into the Laurel Creek reservoir in northwestern Waterloo. This research was conducted in a 1 km stretch that runs through a deciduous forest beginning east of Erbsville Road which is located before it flows through a major urban area.

4.2 Original intent of online ponds in Critter Creek

This research is focused on one main part of Critter Creek. During restoration construction, offline ponds were placed throughout Critter Creek and over time these ponds have become permanently connected to the main stream channel (Figure 2).

The original design of Critter Creek explains the pond placement as part of the restoration. Pond 1 was designed to enhance fish passage and refuge and to plan for the fact that the culvert currently located at Woolwich Street is to be removed in the future. Ponds 2 and 3 (now connected), and 6 were meant to provide additional diversity to the stream structure. Ponds 4 and 5 were to provide fish refuge and spawning areas. All of the ponds other than Pond 1 were placed in areas where current groundwater inlets to the stream were already present (Schollen, 2000).

In the restoration plans the focus of the offline ponds was clearly on the biological and resilience factors of the stream (Schollen, 2000). Conversely, in the stormwater management report prepared by Stantec Consulting Ltd. (2000) Ponds 5 and 6 are referred to as “stormwater management ponds” and their uses for water attenuation and quality is acknowledged. Before construction was completed the ponds became online. In response, the City of Waterloo continually changed the creek to remove the ponds. It was after the City of Waterloo decided to stop this upkeep that the ponds became permanently online (D Stephenson, pers. comm.). Sometime between summer 2003 and summer 2004 Pond 6 became continuously part of the main stream channel, rather than simply during overflow from storm events. It is unknown when Ponds 1, 2, 3, and 4 became connected to the main stream channel, however based on monitoring reports it must have occurred sometime between 2005 and 2009 (NSRI, 2004).

Figure 2. Location of online ponds in Critter Creek. This orthophoto was taken before the ponds became permanently online therefore that is why they are not connected in the photo.

4.3 Sampling

Samples were taken based on study reaches, with one reach being composed of a riffle-pool-riffle sequence of approximately 5m. If a conventional riffle-pool-riffle sequence was not available locations were chosen based on apparent water velocity with faster areas being riffle habitats and slower areas being pool habitats. Samples were taken downstream-to-upstream, therefore ensuring that there were no drift organisms from previous samples being collected in another sample. The locations not adjacent to the ponds were chosen randomly, however some bias could have occurred as I tried to choose locations that were spread evenly throughout the stream. Three replicates were taken in each reach with four non-adjacent pond reaches sampled in Critter Creek, five reaches sampled in Clair Creek and four reaches sampled in Laurel Creek. The reaches that inlet and outlet from each online pond in Critter Creek were also sampled.

Overall, 164 samples were collected from three streams which resulted in the collection and identification of 19,264 organisms from 42 Families plus Hirudinea, Oligochaeta and Nematoda. In the early summer season, 88 samples were collected on 15 days between June 8, 2009 and July 8, 2009. In the late summer season 76 samples were collected on 10 days between September 13, 2009 and September 30, 2009. These sampling periods were chosen to include the changes of season to the biological community (Gibson *et al*., 1996).

In-stream sampling was completed using a Surber sampler. The sampling frame was placed on the stream bed with the net facing downstream. The large rocks located within the sampling frame were picked up and macroinvertebrates located on the rocks were removed with forceps and placed in a nalgene bottle filled with water and approximately 10 ml of 10% formalin. After all the rocks had been removed the ground within the sampling frame was agitated with a large stick to a depth of approximately 10 cm for approximately two minutes. The stream flow swept the agitated sediment through the net and the benthic macroinvertebrates dislodged from the sediment became trapped in the net. The sampler was then removed from the streambed and the macroinvertebrates were removed from the net and placed in the same nalgene container. Containers were transported to the laboratory and stored in a refrigerator for less than 48 hours. Samples were then poured in a metal container where the organisms were removed and placed in a vial with 10% formalin.

If the stream bed was composed of clay or sand, a shovel was used to remove a sample of the bed and then placed in a plastic bag. Within four hours the bags were returned to the laboratory for processing. The mud sample was placed in a metal container and water was added to dissolve the mud. Next, enough salt was added to reach the saturation point of the water. Once saturated living organisms within the sample floated to the surface of the water, they were removed using forceps and placed in a sealed vial containing 10% formalin.

In the laboratory all of the macroinvertebrates in each sample were identified and counted using a microscope. The organisms were identified down to the Family level, other than Hirudinea and Oligochaeta (subclasses) and Nematode (phylum). Following the Ontario Benthos Biomonitoring Network (OBBN) protocol, an organism was counted if it had a head, with larval exuvia and empty shells not counted (Jones *et al*., 2007). Samples were then returned to the glass vials for future reference if necessary.

Several quality control measures were taken. In the field, the jars were labelled with the sample number to prevent misidentification of samples. Field notes were taken on the collection date, such as habitat (riffle/pool), stream bed sediment characteristics and vegetation. These field notes helped to determine confounding factors with regards to benthic macroinvertebrate assemblages in the analysis. The stick and the Surber sampler were rinsed after each sample collection to prevent cross-contamination. One person completed the sampling to ensure that the stream bed sampling depth was consistent.

In the laboratory I ensured that all samples were transferred from the jars to vials by rinsing the jars into white trays. Since the students were self-trained in the identification process the organisms were identified to the lowest possible level given their abilities. It must be acknowledged, however, that misidentifications could have occurred. It has been shown that the hardest organisms to identify are Chironomidae, Epheroptera, and Tricoptera. Problems could be created during transport to the lab and preservation in the lab. However the variation and magnitude that these problems present varies from lab to lab and therefore it is not possible to quantify the error (Stribling *et al*., 2008). For future studies it is suggested that there be a quality control protocol implemented. This could include confirmation of identification by the other students or by an expert in the field. However, given that adequate time and care was given by the students during the identification process, there is a high level of confidence in the identification results.

4.4 Metrics Used

A variety of metrics were used in this study to evaluate the macrovinertebrate communities collected and they are explained in this section. The metrics are a useful way of condensing the individual organism data collected to evaluate the assemblages as a whole. The metrics are divided into the following categories: composition measures which can be used as a proxy for water quality, diversity measures, richness measures which indicated how many organisms are able to live in the stream (regardless of the water quality present), functional feeding groups and function habit groups. One index, Hilsenhoff's Family Biotic Index, which can be used as a proxy for water quality, was also used.

The metrics used in this study, with all organisms included other than Hirudinea, Oligochaeta, and Nematoda are found in Table 2. These metrics were chosen based on the prevalent metrics used in the literature.

Table 2. Metrics used in this study.

Diversity Measures
Simpson's Diversity
Shannon-Wiener Diversity
Richness Measures
Number of Organisms
Family Richness
Composition Measures
% Ephemeroptera
% Plecoptera
% Tricoptera
% Ephemeroptera, Plecoptera, Tricoptera
% Chironomidae
% Diptera
% Coleoptera
Functional Feeding Group
% Filterer-Collector
% Scrapper
% Gatherer-Collector
% Omnivore
Functional Habit Group
% Clinger
Index
FBI

4.4.1 Diversity Measures

Diversity measures are based on the assumption that more organisms occur at unimpacted sites and that the distribution at unimpacted sites is more even than at impacted sites (Cao *et al*., 1996). They also assume that all individuals assigned to a particular class are identical, even though they may vary in size, health, etc. Furthermore, classes are compared equality, as opposed to the techniques employed in trait-based metrics (Krebs, 2009).

4.4.1.1 Simpson's Diversity

Simpson's diversity is calculated using the following formula:

$1 – D = 1 - \Sigma(p_i)^2$

where $1 – D$ = Simpson's index of diversity

p_i = Proportion of individual species in the ith community

This index ranges from 0, indicating low diversity, to almost 1. It is most useful when describing the dominance of certain species. Since p_i is squared, when p_i is large Simpson's diversity will decrease less proportionally as compared to the decrease when p_i is small (Krebs, 2009).

4.4.1.2 Shannon-Wiener Diversity

This metric answers the question "how difficult would it be to predict correctly the species of the next individual collected?" (Krebs, 2009, p 444). The uncertainty in the answer to this question is measured by the Shannon-Wiener index of diversity using the following formula:

$H' = \Sigma(p_i)(\log p_i)$

where H' = Shannon-Wiener index of diversity

p_i = Proportion of individual species in the ith community

The more negative H', the more diversity is present in the sample. Adams and McCune (1979) determined that the estimates of H' from field data are usually biased so that sampled H' is less than the true H'. Considering that this study is always comparing sampled data and is not interested in the actual values of H' as measures of restoration, this bias does not matter.

4.4.2 Trait-based approach (Functional Habit and Feeding Groups)

Benthic macroinvertebrate traits have been shown to change with hydrological disturbances and anthropogenic pollution. Trait analysis can be used because they change with environmental stress, they can be used at many scales (as compared to species composition analysis which is more restricted based on geographical areas), they can be linked to food web dynamics, and they may not require an extremely coarse level of identification. Biological traits are based on the physiology, morphological adaptations, and life histories of an organism. Ecological traits "reflect an organism's environmental preferences and behaviours associated with the preferences". When choosing traits for the analysis it was important to consider the response rate of the trait (Vieira *et al*., 2006).

4.4.2.1 Functional feeding groups

This method of analysis is based on the River Continuum Concept developed by *Vannote et al.* (1980). This theory describes the communities that are found along a natural river system based on the functional feeding groups of organisms present. The functional feeding groups are characterized by the energy and organic matter transport, storage, and use by the organism which is regulated by geomorphic processes and therefore changes throughout a river system. It has characterized the distribution of collectors, shredders, grazers, and predators that theoretically would be present in headwaters, medium-sized streams, and large rivers. The theory utilizes the idea of equilibrium between the organic and energy inputs into the stream and the organisms present.

The River Continuum Concept classifies organisms based on their functional feeding group therefore organisms from different taxa can occupy the same functional feeding group. In terms of the food web, this means that they may ingest different organisms but the method of ingestion is the same. This type of analysis cannot determine the overall human impact, but can discriminate the type of impact that will have an effect on food availability (Bonada *et al*., 2006).

Theoretically, based on the River Continuum Concept, the maximum taxonomic richness of macroinvertebrates will be present in mid-order streams. However, since this theory fits best for streams in forested, temperate zone with appreciable slopes, it does not always hold true. Paller *et al*. (2006) found that functional group composition was unrelated to stream size. As well, restoration may only affect/improve those organisms whose functional traits are directly affected by restoration (Tullos *et al*., 2009).

4.4.3 Hilsenhoff's Family Biotic Index (FBI)

This index uses the pollution tolerance of organisms to determine the level of stream impairment. Each organism is assigned a tolerance value of 0 to 10, with a value of 0 indicating that the organism has a very low tolerance to pollution and a value of 10 indicating that the organisms has a very high tolerance to pollution. The original design of the index indicates that pools should not be used in the analysis, however they were included here due to reasons stated earlier (Hilsenhoff, 1987). The Index is calculated using the following formula:

$FBI = \Sigma(x_i)(t_i)/n$

where FBI = Hilsenhoff's Family Biotic Index

x_i = number of individuals within a taxon

t_i = tolerance value of a taxon

n = total number of organisms in the sample

4.4.4 Utility of - and relationship between - metrics

Not all metrics may be considered equally useful in all situations. Purcell *et al*., (2009) developed a screening process for choosing the metrics most suitable for analysis of urban streams. The screening process was based on 5 criteria:

- the range of the metric, to determine if it could detect change;
- the area-based effects of the metric, based on the fact that there will be faunal shifts as the catchment area increases;
- the quantile regression of the metric, which was used to determine the relationship between the metrics and the urban gradient;
- the redundancy of the metric where metrics were plotted against each other and determined to be redundant if r<0.7; and
- the ecological category that the metric fell into, with two metrics selected from each ecological category (composition and metrics, functional feeding group, and habit).

In their study, after going through the analysis, the metrics chosen were the percent of EPT individuals, the EPT richness, the percent of filterer individuals, the filterer richness, the percent of clinger individuals and the clinger richness.

Resh and Jackson (1993) tested the ability to detect impairment of 20 metrics and found that the most effective were species richness, Margalef's and Hilsenhoff's Biotic Index and percent scrapers, a functional feeding group metric. Family richness and EPT have been found to be correlated, however trait richness and EPT are only weakly correlated. The single trait presence of an organism, unless it is a keystone species, will not incur the same importance as the presence of that trait by many organisms (Beche and Statzner, 2009). The type of stream can also affect which metric to use with one study examining filter-feeders or scrapers in channelized streams and shredders in restored streams (Muotka and Syrjanen, 2007). In some cases diversity measures did not change in a restoration but the abundance of certain taxa did (Walther and Wiles, 2008).

Shannon's Diversity was found to be higher in restored than reference reaches in an urban catchment (Tullos *et al*. 2009). Compared to rural and agricultural reaches, urban reaches had a higher abundance of opportunistic taxa such as Chironomidae, Baetidae, Hydroptilidae and Simuliidae in both restored and control reaches. Furthermore, the functional traits of macroinvertebrates did not significantly differ between control and restored reaches in the urban catchment but did significantly differ in the rural and agricultural catchments (Tullos *et al*. 2009).

4.5 Analyses

Two main analyses were used in this study: the Percent Model Affinity (PMA) Method and the Mann-Whitney statistical test. The PMA Method used the raw data, that is the actual numbers of organisms collected in each sample. The Mann-Whitney statistical test evaluated the differences in the metrics found between two samples (ie. the difference in Shannon's Diversity measures between Critter Creek and Laurel Creek) and rejected or accepted the null hypothesis.

4.5.1 Percent Model Affinity (PMA)

The Percent Model Affinity (PMA) method compares an expected community with the one that is sampled and based on the difference between these two infers statements about the effect that pollutants are having on that ecosystem. The original theory was developed by sampling streams in New York State and developing a model of expected community composition of 7 major groups: Oligochaeta, Ephemeroptera, Plecoptera, Coleoptera, Tricoptera, Chironomidae, and Other (usually composed of Simuliidae, Gammaridae, Asellidae, Physidae and Empididae). The percent similarity of the sample community is then compared to the ideal community using the following formula:

$PSC = 100 - 0.5\Sigma \left| a - b \right|$

where PSC is the Percent Similarity of the Community

a is the percent of individuals of a taxon in sample A (model value)

b is the percent of individuals of a taxon in sample B

A PSC equal to 100% means that the sampled community has exactly the same organism community composition as the ideal community. Shifts in dominance of less tolerant to more tolerant groups decrease the PSC in order to detect the existence of impaired habitats (Novak and Bode, 1992).

Barton (1996) modified the PMA method to an agricultural catchment in southern Ontario. Based on his research, he concluded that in this area the organism compositions should be 20.49521% Ephemeroptera, 5.19188% Plecoptera, 14.15905% Tricoptera, 12.08047% Coleoptera, 31.34344% Chironomidae, 5.23257% Oligochaeta, 11.49739% Other (A Schiedel, pers. comm.). He considered that a PSC value less than 33.32% was considered impaired. He found that this method was more reliable than using simple community descriptors like richness, EPT and FBI (Barton, 1996).

The PMA method was used in order to compare pre- and post-community composition monitoring that had already been completed. This method is a specific case of the Bray-Curtis distance as related to the distance of each community from the median of all reference communities. Therefore in this method it is necessary that the community descriptor adequately represents the community at the reference site and will respond to degradation (Bailey *et al*., 1998).

4.5.2 Mann-Whitney and Kruskal-Wallis statistical tests

The majority of literature has used analyses of variance (ANOVA) to test differences between reference and restored streams (e.g. Muotka *et al*., 2002; Roy *et al*. 2003; Lepori *et al*., 2005; Muotka and Syrjanen, 2007; Walther and Whiles, 2008). ANOVA assumptions include that the data be normally distributed. Shapiro-Wilk and Anderson-Darling normality tests were completed on the data collected in this study and it was determined that neither the original nor a log transformation of the data were normal. Therefore, the non parametric Mann-Whitney and Kruskal-Wallis tests were used to analyze the data.

Non parametric tests are used either for nominal or ordinal level data or when the nature of the data is unknown. Nominal and ordinal scale variables are both qualitative variables, with normal scale having no order to its categories and ordinal having order to its categories (Burt and Barber, 1996). This study collected ratio level data, but since the nature of the data is unknown (and may therefore not be normally distributed) a non parametric test was used. While a non parametric test may be less powerful and less robust than its parametric alternative, when the data is not normally distributed it is the statistical test chosen (Burt and Barber, 1996).

Mann-Whitney and Kruskal-Wallis tests operate on the same principles. A Mann-Whitney test compares two populations; a Kruskal-Wallis test compares more than two populations. Both require that the populations are composed of independent random samples. The values from each sample are combined and then ranked from lowest to highest. Each value is then assigned a new value based on its rank and the sums of the ranks are computed for each sample. The assumption is that two, or more than two, independent samples from the same population will have mean ranks that are almost equal. If the average ranks are the same than this supports the null hypothesis that the samples are from the same population. If the average ranks are different, than this supports the alternate hypothesis that the samples are taken from different populations (Burt and Barber, 1996).

Chapter 5: Results

5.1 A comment on the results collected using the Surber sampler and the shovel method

Samples were collected using the Surber sampler (n=142) and the shovel method (n=22). In general the Surber sampler was used when the stream bed sediment was larger than fine sand and the shovel method was used for smaller bed grain sizes. The two methods were used because riffle-pool-riffle samples were collected in each reach however sometimes substrates varied within these habitats. The Mann-Whitney tests revealed that there was a significant difference (p=0.000 or p=0.007) between the two sampling methods in regards to metrics other than Family Biotic Index (FBI) (p=0.336) and percent Ephemeroptera (p=0.246). Given that there was such a difference in assemblages collected using the two methods, only samples collected using the Surber sampler were included in the subsequent analysis. I removed samples collected using the shovel method because it increased habitat homogeneity in the analysis. By attempting to keep bed sediment size homogeneous, the possibility that bed sediment size could be a confounding factor is removed.

5.2 A comment on the results of percent Ephemeroptera, percent Plecoptera, percent Tricoptera, and percent EPT.

Ephemeroptera were only found in 9 samples, totalling 16 organisms out of over 19,000 collected in the study and there were no Plecoptera collected. NRSI collected 344, 6, 36, 166 and 62 Ephemeroptera in 2000, 2001, 2002, 2003 and 2004, respectively. They also found no Plecoptera in their five years of monitoring. Considering that so few organisms were collected, the utility of this metric has been greatly diminished when examining the three objectives in the study. Percent EPT encompasses the total of the Ephemeroptera, Plecoptera and Tricoptera taxa selected. In this study there were no Plecoptera taxa collected and very few Ephemeroptera taxa collected. Given this, only the percent Tricoptera metric was measured in this analysis.

5.3 Results from the Percent Model Affinity (PMA) method

Monitoring was completed at Critter Creek before restoration construction in May 2000 and for four years after construction. This monitoring was completed in accordance with recommendations from the Laurel Creek Watershed Monitoring Program, the Eastbridge District North Master Drainage Plan (Eastbridge MDP) and the Terms of Reference from the Millennium Recreation Project Environmental Planning Study with the organism collection being completed by Natural Resource Solutions Inc. (NSRI) (Stantec Consulting Ltd, 2001). In 2000 and 2001 the samples were collected using a Surber sampler with the streambed being disturbed for 20 minutes at each site. Samples were preserved in 10% formalin in the field and were identified to genus level, or species level when possible (NSRI, 2000; NSRI, 2001) In 2002, 2003, and 2004 Critter Creek was sampled at the same four locations using a "D" frame kick net and the "travelling kick and sweep" method. The net was placed in the stream and the area in front of the net, in all different habitats present, and the habitat was disturbed for three to five minutes. Samples were preserved in 10% formalin in the field and were identified to genus level, or species level when possible (NSRI, 2002). NSRI used the Percent Model Affinity (PMA) method. They completed 2 replicates of 4 riffle areas. The sampling completed in this research was matched up to their sampling location and those results along with the results from NSRI are summarized in Table 3 (NSRI, 2004).

Table 3. Percent Model Affinity (%) results from Critter Creek in 2000, 2001, 2002, 2003, 2004 and 2009. Data was collected by NRSI from 2000-2004 with data collected in 2009 being completed in this research. Bolded values indicate when the PMA value is less than 33.32% which indicates an impaired stream.

Station	Replicate	2000	2001	2002	2003	2004	2009 Early	2009 Late
1	1	46.94	39.51	71.04	71.70	**29.50**	41.54	36.98
1	2	N/A	**32.33**	51.57	63.56	37.12	**27.71**	**31.25**
2	1	39.81	**26.50**	58.84	53.28	**32.71**	**14.83**	**24.03**
2	2	N/A	34.15	63.64	39.65	**32.50**	**12.78**	**24.19**
3	1	**27.16**	45.15	51.96	45.16	**30.50**	**14.43**	**12.18**
3	2	N/A	45.46	53.02	39.55	**28.50**	**11.93**	**14.31**
4	1	**18.94**	44.11	**32.87**	23.44	**33.23**	**26.99**	**11.99**
4	2	N/A	48.07	**32.11**	**16.92**	**26.50**	**17.21**	**15.20**

The PMA method was also completed on data collected in other reaches in Critter Creek, and in Clair Creek and Laurel Creek. In 2009 the number of times that the PMA method indicated that the stream was unimpaired in Critter Creek was 3 out of 40 samples in the early season and 3 out of 49 samples in the late season. In Clair Creek the stream was deemed unimpaired in 14 of 14 samples in the early season and 8 of 15 samples in the late season. In Laurel Creek the stream was deemed unimpaired in 10 of 12 samples in the early season and 5 of 12 samples in the late season.

5.4 Within-stream restoration of Critter Creek

The first test to determine the impact of the online ponds was to examine the difference between the inlet and outlet samples. However, this method could cause a problem because it may be that the inlet samples are simply outlet samples from the previous pond (ie. the inlet to Pond 6 is simply downstream outlet from Pond 5). Mann-Whitney tests were completed which compared the outlet samples from the upstream pond to the inlet samples from the next downstream pond. The null hypothesis was accepted ($p \leq 0.05$) for all metrics other than the percent Gatherer-Collectors ($p=0.021$) and FBI ($p=0.021$) between Ponds 4 ($n=4$) and 5 ($n=4$) and the percent Coleoptera ($p=0.047$) and percent Omnivore ($p=0.047$) between Ponds 5 ($n=4$) and 6 ($n=6$). The inlet to Pond 1 ($n=4$) was compared to the outlet of Pond 6 ($n=10$) since this inlet sample was not influenced by any upstream ponds. In this case the Mann-Whitney tests accepted the null hypothesis ($p \leq 0.05$) for every metric other than Number of Organisms ($p=0.007$).

Regardless of the fact that the inlet and outlet samples may not be completely independent (see Chapter 6 for further discussion), Mann-Whitney tests were completed which compared the samples surrounding the ponds to those not surrounding the ponds (Table 4), inlet and outlet samples individually to the non-adjacent pond samples (Table 5), and inlet to outlet samples (Table 6).

Table 4. Significance values from Mann-Whitney tests comparing metrics from the samples surrounding the online ponds (both season n=65; early n=31; late n=34) and the other samples in Critter Creek (both season m=20; early n=8; late n=12). Bolded values indicate when the null hypothesis, that the samples are from the same population, has been rejected ($p \leq 0.05$).

	Season		
	Both	Early	Late
Diversity Measures			
Simpson's Diversity	**0.006**	0.636	**0.001**
Shannon-Wiener Diversity	**0.003**	0.543	**0.000**
Richness Measures			
Number of Organisms	0.504	0.723	0.255
Family Richness	0.442	0.708	**0.037**
Composition Measures			
% Tricoptera	**0.010**	0.183	**0.016**
% Chironomidae	0.272	0.528	0.416
% Diptera	0.510	0.826	0.568
% Coleoptera	**0.000**	0.199	**0.000**
Functional Feeding Group			
% Filterer-Collector	0.297	0.892	0.173
% Scrapper	0.673	0.599	0.111
% Gatherer-Collector	0.109	0.098	0.227
% Omnivore	0.103	0.104	0.104
Functional Habit Group			
% Clinger	0.806	0.237	0.377
Index			
FBI	0.112	0.543	0.209

Table 5. Significance values from Mann-Whitney tests comparing metrics from the online pond inlet samples (both seasons n=34; early n=17; late n=17) and outlet samples (both season n=31; early n=14; late n=17) to the other samples in Critter Creek (both seasons n=20; early n=8; late n=12). Bolded values indicate when the null hypothesis, that the samples are from the same population, has been rejected (p≤0.05).

	Inlet Samples			Oulet Samples		
	Season			Season		
	Both	Early	Late	Both	Early	Late
Diversity Measures						
Simpson's Diversity	**0.017**	0.641	**0.007**	**0.013**	0.733	**0.003**
Shannon-Wiener Diversity	**0.019**	0.683	**0.002**	**0.005**	0.539	**0.001**
Richness Measures						
Number of Organisms	0.603	0.580	0.506	0.500	0.891	0.199
Family Richness	0.691	0.638	0.060	0.469	0.890	0.143
Composition Measures						
% Tricoptera	0.088	0.323	0.139	**0.009**	0.168	**0.019**
% Chironomidae	0.116	0.363	0.227	0.623	0.835	1.000
% Diptera	0.397	0.705	0.706	0.583	0.891	0.417
% Coleoptera	**0.001**	0.278	**0.002**	**0.000**	0.267	**0.001**
Functional Feeding Group						
% Filterer-Collector	0.471	0.769	0.520	0.394	0.838	0.141
% Scrapper	0.727	0.345	0.140	0.731	0.646	0.329
% Gatherer-Collector	0.307	0.322	0.199	0.056	0.056	0.199
% Omnivore	0.616	0.502	0.259	**0.022**	**0.026**	**0.049**
Functional Habit Group						
% Clinger	0.329	0.727	0.288	0.582	0.088	0.859
Index						
FBI	0.068	0.294	0.232	0.446	0.946	0.535

Table 6. Significance values from Mann-Whitney tests comparing the metrics from inlet samples (all season n=34; early n=17; late n=17) to outlet samples (all seasons n=31; early n=14; late n=17). Bolded values indicate when the null hypothesis, that the samples are from the sample population, has been rejected ($p \leq 0.05$).

	Season		
	Both	Early	Late
Diversity Measures			
Simpson's Diversity	0.738	0.781	0.558
Shannon-Wiener Diversity	0.541	0.905	0.449
Richness Measures			
Number of Organisms	0.890	0.937	0.743
Family Richness	0.756	0.795	0.634
Composition Measures			
% Tricoptera	**0.010**	0.183	**0.016**
% Chironomidae	0.272	0.528	0.416
% Diptera	0.510	0.826	0.568
% Coleoptera	0.936	0.626	0.860
Functional Feeding Group			
% Filterer-Collector	0.984	0.382	0.262
% Scrapper	0.903	1.000	0.600
% Gatherer-Collector	0.166	0.081	0.890
% Omnivore	**0.016**	**0.002**	0.301
Functional Habit Group			
% Clinger	0.111	0.074	0.582
Index			
FBI	0.120	0.095	0.730

In general there is little in-stream variation with respect to the online ponds in Critter Creek given that the null hypothesis was accepted for the majority of metrics examined (Tables 4, 5, and 6). Exceptions to this rule are found in the diversity measures, percent Coleoptera and percent Tricoptera in the late season when comparing the inlet and outlet samples to the other samples (Tables 4 and 5). The null hypothesis was also rejected for the percent Tricoptera and percent Omnivore metric when comparing the outlet samples to other samples (Table 5) and inlet to outlet samples (Table 6).

5.5 Overall restoration of Critter Creek

To determine the overall restoration of Critter Creek, Mann-Whiney tests comparing Critter Creek to Laurel Creek (Tables 7 and 8) and to Clair Creek (Table 9 and 10) were completed.

Table 7. Significance values from Mann-Whitney tests comparing the metrics from Critter Creek to Laurel Creek (both seasons n=24, early n=12, late n=12). Critter Creek was examined by samples not adjacent to the ponds (both seasons n=20; early n=8; late n=12) and by inlet and outlet samples (both seasons n=65; early n=34; late n=31). Bolded values indicate when the null hypothesis, that the samples are from the sample population, has been rejected (p≤0.05).

	Inlet and Outlet Samples			Samples Non Adjacent to Ponds		
	Season			Season		
	Both	Early	Late	Both	Early	Late
Diversity Measures						
Simpson's Diversity	0.592	0.544	0.944	0.346	0.643	0.073
Shannon-Wiener Diversity	0.276	0.330	0.443	0.346	0.817	0.065
Richness Measures						
Number of Organisms	**0.000**	**0.002**	**0.001**	**0.000**	**0.037**	**0.000**
Family Richness	0.710	0.524	1.000	0.378	0.726	0.186
Composition Measures						
% Tricoptera	**0.000**	**0.000**	**0.000**	**0.000**	**0.002**	**0.010**
% Chironomidae	**0.000**	**0.000**	**0.000**	**0.000**	**0.008**	**0.006**
% Diptera	**0.000**	**0.022**	**0.000**	**0.001**	**0.049**	**0.002**
% Coleoptera	**0.003**	**0.013**	0.097	0.696	0.333	0.643
Functional Feeding Group						
% Filterer-Collector	**0.000**	**0.000**	**0.000**	**0.000**	**0.009**	**0.006**
% Scrapper	**0.002**	**0.002**	**0.086**	**0.030**	**0.012**	0.733
% Gatherer-Collector	**0.009**	**0.031**	**0.043**	**0.003**	**0.009**	**0.028**
% Omnivore	**0.000**	**0.000**	**0.000**	**0.000**	**0.007**	**0.000**
Functional Habit Group						
% Clinger	**0.037**	0.330	**0.015**	0.109	0.097	0.100
Index						
FBI	**0.000**	**0.000**	**0.000**	**0.000**	**0.001**	**0.000**

Table 8. Significance values from Mann-Whitney tests comparing the metrics from Critter Creek to Laurel Creek (both seasons n=24, early n=12, late n=12). Critter Creek was examined as inlet samples (both seasons n=34; early n=17, late n=17) and outlet samples from the online ponds (both seasons n=31; early n=14; late n=17). Bolded values indicate when the null hypothesis, that the samples are from the sample population, has been rejected (p≤0.05).

	Inlet Samples			Outlet Samples		
	Season			Season		
	Both	Early	Late	Both	Early	Late
Diversity Measures						
Simpson's Diversity	0.586	0.425	0.947	0.581	0.797	0.808
Shannon-Wiener Diversity	0.425	0.352	0.642	0.281	0.440	0.388
Richness Measures						
Number of Organisms	**0.000**	**0.012**	**0.004**	**0.000**	**0.007**	**0.004**
Family Richness	0.594	0.518	0.964	0.765	0.678	0.839
Composition Measures						
% Tricoptera	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**
% Chironomidae	**0.001**	**0.002**	**0.013**	**0.000**	**0.001**	**0.001**
% Diptera	**0.003**	0.084	**0.001**	**0.001**	**0.027**	**0.002**
% Coleoptera	**0.013**	**0.037**	0.158	**0.007**	**0.024**	0.092
Functional Feeding Group						
% Filterer-Collector	**0.000**	**0.003**	**0.001**	**0.000**	0.002	**0.000**
% Scrapper	**0.010**	**0.014**	**0.121**	**0.007**	0.005	0.267
% Gatherer-Collector	**0.008**	**0.010**	**0.080**	0.071	0.258	0.116
% Omnivore	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**
Functional Habit Group						
% Clinger	**0.024**	0.121	**0.028**	0.183	0.877	0.054
Index						
FBI	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**

Table 9. Significance values from Mann-Whitney tests comparing the metrics from Critter Creek (both seasons n=89, early n=40, late n=49) to Clair Creek (both seasons n=29, early n=14, late n=15). Bolded values indicate when the null hypothesis, that the samples are from the sample population, has been rejected (p≤0.05).

	Season		
	Both	Early	Late
Diversity Measures			
Simpson's Diversity	0.316	**0.026**	0.423
Shannon-Wiener Diversity	0.466	0.106	0.542
Richness Measures			
Number of Organisms	**0.000**	**0.000**	**0.001**
Family Richness	**0.030**	**0.032**	0.103
Composition Measures			
% Tricoptera	**0.000**	**0.001**	**0.005**
% Chironomidae	**0.001**	**0.000**	0.174
% Diptera	**0.003**	**0.000**	**0.011**
% Coleoptera	**0.000**	**0.000**	**0.000**
Functional Feeding Group			
% Filterer-Collector	**0.001**	**0.010**	**0.011**
% Scrapper	0.246	**0.030**	0.409
% Gatherer-Collector	0.886	0.240	0.451
% Omnivore	**0.000**	**0.000**	**0.000**
Functional Habit Group			
% Clinger	**0.011**	**0.003**	0.213
Index			
FBI	**0.000**	**0.000**	**0.001**

Table 10. Significance values from Mann-Whitney tests comparing the metrics from Critter Creek to Clair Creek (all seasons n=29, early n=14, late n=15). Critter Creek was examined by inlet samples to the online ponds (all seasons n=34; early n=17, late n=17) and outlet samples from the online ponds (all seasons n=31; early n=14; late n=17). Bolded values indicate when the null hypothesis, that the samples are from the sample population, has been rejected ($p \leq 0.05$).

	No Pond Samples			Pond Samples		
	Season			Season		
	Both	Early	Late	Both	Early	Late
Diversity Measures						
Simpson's Diversity	0.542	0.339	**0.036**	0.138	**0.018**	0.880
Shannon-Wiener Diversity	0.349	0.838	**0.017**	0.206	0.059	0.832
Richness Measures						
Number of Organisms	**0.000**	**0.003**	**0.002**	**0.000**	**0.000**	**0.002**
Family Richness	**0.035**	0.204	**0.025**	0.057	**0.031**	0.254
Composition Measures						
% Tricoptera	**0.049**	**0.036**	0.337	**0.000**	**0.001**	**0.001**
% Chironomidae	**0.009**	**0.001**	0.197	**0.003**	**0.000**	0.294
% Diptera	**0.013**	**0.006**	0.053	**0.006**	**0.000**	**0.000**
% Coleoptera	**0.002**	**0.011**	**0.043**	**0.000**	**0.000**	**0.000**
Functional Feeding Group						
% Filterer-Collector	**0.046**	0.051	0.261	**0.001**	**0.014**	**0.026**
% Scrapper	0.236	0.116	0.841	0.300	**0.027**	**0.006**
% Gatherer-Collector	0.528	0.116	0.883	0.371	0.384	0.209
% Omnivore	**0.000**	0.068	**0.003**	**0.000**	**0.000**	0.368
Functional Habit Group						
% Clinger	0.058	**0.009**	0.464	**0.015**	**0.009**	0.200
Index						
FBI	**0.000**	**0.001**	**0.045**	**0.000**	**0.000**	**0.001**

In general, when comparing Critter Creek to Laurel Creek there is no difference in diversity measures, but a significant difference in almost all other metrics (Tables 7 and 8). The Clingers metric is the one exception as it does not always follow this trend, with the samples not adjacent to the ponds and outlet samples accepting the null hypothesis, and inlet and outlet samples together, and the inlet samples on their own rejecting the null hypothesis. As compared to Clair Creek, the other restored stream, a significant difference seen with the composition measures but not always in the functional feeding group or diversity measures (Tables 9 and 10).

Chapter 6: Discussion and Conclusions

6.1 Setting the stage: using the PMA method

According to the PMA analysis the ecosystem integrity of Critter Creek has not improved and may even be declining (Table 3). NRSI attributed the decline in stream health seen in 2004 to the fact that upstream of Pond 6 there was another stormwater management pond that had elevated phosphorus levels and its impact is believed to have travelled throughout the system. As well, Pond 6 had become online between 2003 and 2004 and they believed that this negatively affected the benthic assemblages since it would have an influence on water temperature, flow regime and sediment transport (NSRI, 2004). This was also seen in the number of Ephemeroptera collected. This metric drastically decreased after original construction in 2001 but had continued to increase until 2004 when it decreased again, presumably due to the same reasons as the decrease in PMA. Given that there were no Plecoptera collected by either NRSI, or in this research either the restoration did not improve the Plecoptera habitat or improve water quality compared to pre-restoration conditions. This health of the stream has continued to be poor, even nine years after monitoring had originally been completed.

The PMA method results can be used to determine whether Laurel Creek and Clair Creek are adequate reference streams. As was explained earlier, finding a representative reference site can be challenging (Bonada *et al*., 2006). The PMA results indicate that Critter Creek is an impaired stream in both seasons while Clair Creek and Laurel Creek are unimpaired in the early season and sometimes impaired, sometimes not, in the late season. The habitat sampled did not affect these PMA results. The difference between seasons can be attributed to two categories in the analysis, “Chironomidae” and “Other”, with the number of Chironomidae decreasing from early to late season and the number of Other organisms increasing from early to late. The samples that were found to be impaired were dominated by a larger number of Other organisms, mainly Asellidae, Simuliidae and Gammaridae. However, the overall results from this preliminary analysis show that Clair Creek and Laurel Creek are more restored or natural in comparison to Critter Creek and can therefore serve as goals along the restoration trajectory for Critter Creek.

6.2 Within-stream restoration of Critter Creek

Overall the results indicate that the function and habit requirements of benthic macroinvertebrates in Critter Creek are similar, without regard for their placement in relation to the online ponds. The result was seen in the comparison between assemblages before and after the ponds compared to the rest of the stream (Table 4), and in the comparison between inlet and outlet samples (Table 5) where the null hypothesis was accepted the majority of the time. The only exception to this result was when examining the Omnivore metric where it was concluded that a difference was found between the outlet samples and the rest of the stream and between the inlet and outlet samples. The only omnivore found in these samples was Planariidae and the mean number of Planariidae found in the outlet samples in the early and late seasons was greater than in the inlet and non-adjacent pond samples.

Benthic macroinvertebrates were used as a proxy for water quality, since some organisms are more tolerant of pollution than others (Hilsenhoff, 1988). The composition metrics and FBI evaluated pollution intolerant species. The composition measures show that in general there is no in-stream variation in Critter Creek, with a rejection of the null hypothesis only occurring in the late season with percent Tricoptera and percent Coleoptera (Tables 4, 5 and 6). Both of these metrics had greater values in non-adjacent pond samples than adjacent pond samples, and greater values in inlet samples than outlet samples. Tricoptera have been shown to be more sensitive to changes in water quality than changes in habitat, therefore the online ponds may be negatively affecting the water quality in the stream (Walsh, 2006).

No in-stream changes were seen in relation to the richness measures, a result seen in other restoration projects (Selvakumar *et al*., 2010). In terms of diversity, no change was seen in the inlet-outlet comparison (Table 6) however the null hypothesis was rejected in the late season in relation to Simpson’s and Shannon-Weiner Diversity when examining inlet and outlet samples in relation to non-adjacent pond samples (Table 5). The diversity was lower in samples associated with the ponds mainly due to the large numbers of Asellidae, Planariidae and Simuliidae that were collected. These results indicate that while the online ponds may not be affecting the biotic restoration of the stream they may have an effect on the water quality in the stream, given that those organisms can be used as a proxy for low water quality (Hilsenhoff, 1988).

Overall these finding indicate that that the impact of the ponds on the stream is large enough to affect further downstream assemblages other than directly at the outlet, effectively creating a stream system that is "one big pond". Another possibility may be that the benthic assemblages in the stream are influenced by larger, watershed-scale impacts and not in-stream changes or disturbances. Given these results, other types of analysis such as water quality testing or geomorphic surveys could be completed that address the impact of the online ponds since those assessment tools will have a different scale of impact than benthic macroinvertebrates.

6.3 Overall restoration of Critter Creek

Given that the preliminary results using the PMA method concluded that Laurel Creek was an adequate reference stream, Critter Creek was compared to Laurel Creek (Table 6). The differences between the two streams could be attributed to the natural and restoration states, or to the urban-gradient theory (Walsh *et al*., 2005). While Laurel Creek is a natural stream is does differ from Critter Creek in that it is wider, has finer bed sediment and it has forest riparian cover compared to Critter Creek which has meadow and scrubland riparian cover. When Critter Creek was broken down into inlet and outlet samples and other stream samples the results were similar (Table 7), which follows the previous finding that there is virtually no in-stream variation with respect to placement of the online ponds. Overall the diversity metrics indicate that there is no difference between Critter Creek and Laurel Creek. However, almost every other metric rejected the null hypothesis, meaning that the state of restoration of Critter Creek has not reached the reference condition.

While both streams have the capability of supporting a similar diversity and number of families, the function that those organisms are completing is different between them. This goes against a previous correlation where as human density increases, and therefore urbanization, taxa richness decreases (Ourso and Frenzel, 2003; Urban *et al*., 2006; Gresens *et al*., 2007). Exceptions to this rule come by examining the percent Clinger metric where a difference is not seen in the early season. In both seasons there are more clingers in Laurel Creek than Critter Creek however the difference between the two is less in the early season. This is mainly due to the number of Simuliidae collected in Critter Creek in the early season that are absent in the late season as they are the mainly clinging organisms. Simuliidae have been shown to be greater in urban restored streams (i.e. Critter Creek) as compared to rural reference streams (i.e. Laurel Creek) (Tullos *et al*., 2009). This explains the seasonality of the clinger results, being that the null hypothesis is accepted in all early tests. Clingers may prefer stable and sediment-free substrates, resulting in decreases in clingers in urban sites as greater erosion in the watershed increases the amount of sediment in urban restored streams (Purcell *et al*., 2009), a result found in this study.

The null hypothesis was also accepted in terms of percent Coleoptera when looking at those samples not affected by the ponds and the late season for both inlet and outlet samples. In the examination of outlet samples the percent Gatherer-Collectors accepted the null hypothesis in both seasons. This could be due to water quality changes as seen in the in-stream analysis of Critter Creek. This inference is backed up by the null hypothesis being rejecting with regards to the FBI metric, and similar to another study where an increase in FBI was associated with an increase in percent impervious cover (Ourso and Frenzel, 2003). Critter Creek is located adjacent to a parking lot and to many roads where urban stormwater runoff could be leading to a decrease in water quality (Gresens *et al*., 2007).

The basis of this research was the Reference Condition Approach, where the restored stream is compared to a reference stream, and it is thought that a successful restoration will produce a restored stream similar to a reference stream. Clearly, that has not been achieved in the restoration of Critter Creek. However, it is important to determine if that was the original goal of this restoration. The original objectives of the restoration were "[t]o restore the health and sustainability of the watercourse" and "[t]o enhance the aquatic habitat potential of the creek" (Stantec Consulting Ltd., 2000 5.1). Given those objectives, the results in this research do not support or disprove that the restoration has met its original goals.

6.4 What might be the trajectory of restoration of Critter Creek?

Given that the PMA method determined that Clair Creek was a relatively unimpaired stream, the metrics were compared between that stream and Critter Creek (Tables 9 and 10). A significant difference was seen with the composition measures but not always in the functional feeding group or diversity measures. There were also minimal differences in the significant results of the samples surrounding the ponds and those not surrounding the ponds. This could mean that while the function of the two streams is similar the water quality of the streams (determined from the composition metrics) is not, further confirmed by the rejection of the null hypothesis for the FBI metrics.

The monitoring completed 10 years ago concluded that the restoration trajectory of Critter Creek was not on the right path however nothing was done to remedy the situation. Considering that this current study was completed ten years after restoration and showed little improvement in the state of the stream, it can be argued that monitoring should have been completed for longer than the original four years. Either monitoring should have continued to chart the stream trajectory further to see if the stream would reach the state of the reference condition, or modifications should have been made (following the adaptive management ideology) (Palmer *et al*., 2005). Given that the results of this study indicate that the invertebrates in the stream have not reached the reference condition, modifications to increase water quality (for example reducing direct urban runoff into the stream) should be completed now.

It would also be advised to complete more years of monitoring to see if the macroinvertebrate assemblages change over time, similar to results found in Selvakumar *et al*. (2010) where Ephemeroptera, Plecoptera and Tricoptera were replaced by Chironomidae, Tubificidae, Amphipoda and Gastropoda after restoration. While restoration succession in terms of plant growth and geomorphic properties is well studied, benthic macroinvertebrate succession is not. Often restoration goals are established with an assumed trajectory, however it must be kept in mind that restorations are part of dynamic systems and that the trajectory may change over the course of time. This is summed up by Lake *et al*., (2007), when they state that “in many situations, such as in urban and degraded rural landscapes, restoration to an historical natural state may be impossible and restoration to any target may be unpredictable without knowledge of the species in the regional pool” (pg 603). Furthermore the trajectory will be influenced by the present and future conditions, meaning that maybe the theories in restoration should shift from a “historic” stance to a “futuristic” stance (Choi, 2004). This will be even more important when including climate change in the science (Schaefer, 2009).

The invertebrate succession present in this stream could either be deterministic or stochastic. A deterministic succession will comprise an invertebrate composition trajectory that is steady in one direction and will reach a single equilibrium endpoint. A stochastic succession is one where the change among states is discontinuous or abrupt and there may be multiple paths that the invertebrate composition trajectory state could take (Suding and Gross, 2006). The type of succession and trajectory that is present during the restoration of Critter Creek should be examined further through more years of monitoring.

The trajectory and succession of the benthic macroinvertebrates in the stream is mainly dependent on the way that they colonize. Based on the PMA results (Table 3), there has been no real change in the characteristic assemblages that were found in Critter Creek in the past 10 years. After the initial disturbance of the restoration construction, macroinvertebrates will typically recolonize rapidly from refuges that were not impaired during the construction. However immigration, whether it is from aerial dispersal or fluvial drift, will take much more time. Furthermore once the organisms have reached the stream their survival will depend on adequate abiotic and biotic conditions, such as habitat, streamflow, and predator-prey interactions (Spänhoff and Arle, 2007). Assembly rules, which are how the communities come together given dispersal and environmental constraints, will also come into effect (Lake *et al*, 2007).

The colonization capacity of the stream must also be taken into account. Given that the area surrounding Critter Creek is continually being urbanized the colonization capacity of the stream will decrease (Malmqvist, 2002; Urban *et al*., 2006). If the organisms are not currently present it will be even more difficult for them to colonize and move the assemblages towards those found in the reference stream. Schaefer (2009) refers to the ecological memory of an area being the biotic characteristics of an area that will affect the trajectory of the ecosystem into the future. In urban areas this memory is less due to the decrease in habitat. They state that an ecosystem with a greater ecological memory will have greater resilience. Further research could include a whole watershed study since connectivity is needed for species movement (Lake *et al*., 2007). This would also help to pinpoint large watershed-scale impacts on the stream, how they differ from stream-scale impacts, and what has the biggest influence on benthic macroinvertebrate assemblages. Given that this connectivity is going to change with urbanization, many years of monitoring will help to address this question. Given this, further monitoring could be used to expand upon the macroinvertebrate trajectory knowledge in restoration and determine where Critter Creek falls along this path.

Benthic macroinvertebrates are not the only way to examine the ecosystem state or trajectory of a restored stream (Kondolf and Micheli, 1995; Lear *et al*., 2009). Further research could include streamflow monitoring, habitat assessments and water quality testing which could confirm the results from this study. A habitat assessment would also cater to one of the original goals of restoration (improving aquatic habitat potential) (Stantec Consulting Ltd., 2000).

6.5 Restoration success: examining opportunities for further monitoring with regards to ecosystem resiliency and integrity.

The Critter Creek restoration adhered to a few of the criteria for a successful restoration (Table 1): it had objectives and goals, it included pre- and post-restoration assessments, and it is starting to develop a habitat to sustain and colonize functional groups. However, the original restoration plan lacked explicitly stated goals and therefore in this study the restoration was compared to an adequate reference, and has not yet achieved that state.

The measures of success outlined in Table 1 include the improvement in ecosystem resiliency and integrity. Ecosystem resiliency refers to the ability of the ecosystem to rebound and recover from a stress or disturbance (SERI, 2004). A natural disturbance could be a flood or drought during which the macroinvertebrates would require refugia to survive. A human disturbance could include riparian clearing, nutrient enrichment, or the introduction of a non-native species which affects macroinvertebrates by changing food webs and therefore community structure (Lake *et al*., 2007).

In this study the response of a long-term disturbance was investigated: the process of the offline ponds becoming online. This would have happened due to a larger streamflow than was originally planned in the construction. The restoration was not able to rebound from this disturbance (given the PMA results which show increasing impairment after the ponds came online), meaning that it could not be considered resilient. It also has not rebounded to its reference state. Further investigations into the effect of other disturbances, such as pollution or nutrient inputs from the surrounding urban landscape, would be useful to further examine the resiliency of the ecosystem. For example, NSRI (2004) attributed the decline in PMA values in 2004 to elevated phosphorus levels. Specific monitoring of phosphorus levels in the stream in future monitoring would help to determine if this disturbance is still present.

The scale of investigation could also be examined in terms of disturbance. Only some populations may be affected by a certain disturbance and will therefore only react at a certain scale (Tabacchi *et al*., 2009), an idea brought up earlier when examining the impact of the ponds. Large-scale disturbances may affect small-scale processes therefore a common question posed is whether restorations should occur, and whether they will be successful, if the large-scale disturbances are not addressed (Lake *et al*., 2007). Lake *et al.* (2007) feel that restoration projects may only be successful when done at larger scales, however this is not always feasible. The low water quality present in Critter Creek may not improve with more restoration modifications since it is mainly influence by watershed-scale pollution, such as agricultural runoff. However, this small-scale study of Critter Creek was necessary to determine if there was a problem with the restoration of Critter Creek.

Ecosystem trajectory is important when examining disturbance since there may be non-linear responses to disturbances along the restoration trajectory. The result would be multiple equilibrium states, a factor not included in one year of data in comparison to a reference condition (Suding and Gross, 2006). This means that the ecosystem will not necessarily need to return to a pre-disturbance state after disturbance (Stanley *et al*., 2010).

The number of sampling units, and therefore the scale of investigation, has always been an issue in ecological monitoring. Often the number of samples needed to take is smaller than the number that is actually taken due to logistical and budgetary constraints (Resh and McElravy, 1993), as was the case in this study. Ideally the sample size in this study could increase but this was restricted by the number of restored streams in the area of investigations, as well as the number of reference streams available, especially ones with natural online ponds. Further studies could expand the sampling frame, for example to all of Ontario, or even Canada. This expansion could bring in other complicating issues, such as homogeneity in sampling streams, which would require a change in sampling design.

Ecosystem integrity involves the restored stream containing an assemblage of organisms characteristic of the reference stream (SERI, 2004). Limited redundancy in streams for some ecosystem functions but not others has been shown therefore the full complement of species originally present within particular functional groups may be required to restore particular ecosystem processes (Lake *et al*., 2007). In this study, Critter Creek is not improving its ecosystem integrity based on its PMA results (compared to an ideal situation for the area) or in relation to the assemblages found in the reference stream. Pinpointing the cause of this loss of ecosystem integrity involves looking at the ecosystem as a whole: from the disturbances affecting it, to habitat, to abiotic and biotic differences between Critter Creek and Laurel Creek.

The impacts of the restoration as a part of urban ecology must also be acknowledged when determining the restoration success. This research did not include any elements of human influence on the stream. Natural and anthropogenic disturbances should be considered all together since anthropogenic disturbances interact with natural disturbances and sometimes it is hard to separate the two (Stanley *et al*., 2010). Further research could examine pollutants or energy and nutrient flows into and out of the system as they are affected by humans. Critter Creek is located in an urban environment however it was compared to the more rural Laurel Creek. Another study could try to find an urban restoration for comparison, however this may be impossible.

6.6 Conclusions

This study highlights a rejection of the thinking that increases in biodiversity will lead to ecosystem functioning (Lake *et al*., 2007). While restoration increased habitat and the stream structure, Critter Creek is still not a successful restoration, even 10 years after the original restoration. Schaefer (2009) summed up this problem by stating that "[g]etting from ornamental to fundamental, to a system rather than a collection of ornaments, is one of the biggest challenges facing restoration projects within cities" (Schaefer, 2009, p172).

It also shows that online ponds, while not originally part of the restoration plan, are affecting the whole stream in Critter Creek, not simply those areas directly downstream of the ponds. Further monitoring could occur at Critter Creek to plot the trajectory of macroinvertebrates and eventually suggest restoration changes to improve the characteristic assemblages found there. A larger area monitoring program could also occur which would help to determine influence of large-scale disturbance and the impact of the urban-rural gradient on the restoration.

While further monitoring may help to add to the relevant literature regarding benthic macroinvertebrate succession in restored streams, it will not help to determine the state of Critter

Creek. It is a stream that has poor water quality that has not improved 10 years after restoration and changes should be made now to improve the ecosystem integrity. Constructing small dams to return the ponds offline again may be a short term solution, but given that this has already occurred and did not work (D Stephenson, pers. comm.), this would not be an adequate long term solution. Either the meanders of the stream should be lengthened and the stream widened to be able to contain all the regular streamflow, or the stream will continue to basically be one large stormwater management pond. On a larger scale, improvements to water quality may need to be addressed, for example reducing agricultural runoff, in order to ever permanently improve water quality in Critter Creek.

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