Stream Restoration Monitoring in Theory and Practice
A Case Study of Restored Streams in the Region of Waterloo, Ontario, Canada

By

Colin Nathan Yates

A thesis
presented to the University of Waterloo
in fulfillment of the
thesis requirements for the degree of
Master of Environmental Studies
in
Environment and Resource Studies

Waterloo, Ontario, Canada, 2008

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

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

Abstract

Recently, the importance of quantifying the success of stream/river restoration projects has become a priority in restoration. The absence of ecological monitoring of stream restoration has been made very evident, resulting in the questioning of the viability of restoration activities that have taken place, the ecological approaches used and of restoration as a field of study as a whole. Priority has been set towards illustrating what a successfully restored stream should consist of with development of conceptual frameworks. My study builds upon that concept, by drawing a methodological framework that illustrates how successful stream restoration projects should be quantified using a stream restoration monitoring protocol; asking the question whether a stream restoration monitoring protocol can be created and whether it can appropriately quantify the success of restored stream reaches; further, what assessment technique(s) are best suited for monitoring; ecological, geomorphic or a hybrid approach. In Waterloo, Ontario 29 restored test stream reaches were assessed using benthic macroinvertebrates. Benthos community composition was described using Family Richness, Simpson's Diversity, % EPT, and % Chironomidae. The same reaches were also assessed using a geomorphic assessment technique I designed for this study, which focused on channel stability measures and substrate type as habitat. The methodology was then used to develop information on disturbed (n=7) and natural (n=5) reference reaches in Waterloo. The reference condition approach was used to quantify the relative placement of the restored test streams to reference condition. The ecological assessment technique was best able to quantify the success of a restored reach, by showing linear relationships between benthic metrics in a PCA analysis (0.657). The geomorphic approach, as analyzed by a Non-metric multidimensional scaling test did not consistently evaluate or significantly distinguish between restored reaches and reference conditions, shown by a stress of 25.31. However, a canonical correspondence analysis showed that there are some relationships, although weak, between the ecological approach and geomorphic approach (0.696; p=0.03). This study showed that it is possible to quantify the success or lack of success of restored stream reaches and it is recommended that a hybrid approach be used when monitoring for stream restoration success.

Acknowledgements

There were many people that deserve recognition and special thanks for their contribution to this research project. All of their contributions, both big and small played an important role in completion and success of the project.

First, I would like to thank my supervisor and mentor, Dr. Stephen Murphy. I wish to thank him for his enthusiasm in my research, and the ability to allow me learn from both successes and failures. Most of all I would like to thank him for his endless commitment to my research, to me as his pupil and as a person.

I wish to thank my committee member, Dr. Maren Oelbermann. Her openness, insight and constructive critiques were greatly appreciated throughout the process as well as in all other matters of discussion.

There are numerous individuals that also deserve special recognition. These people volunteered their time and expertise; without their assistance the project would be still ongoing indefinitely. A special thanks to Megan Ihrig, a trusty assistant, who spent endless hours with me in the streams. I could not have accomplished this project without her assistance and organization in the field. Also, to Matt Scoular, who assisted me the field and in the lab. His insight was a great asset. Finally, to my helpful volunteers who assisted me in the lab with the seemingly endless set of tasks. I would like to specially recognize the dedication of Lauren Cymbaly and her endless patience cleaning benthic samples and helping with sediment sieving among many other chores. Also, I wish to thank Emily Stairs and Emily Chow for their assistance in the lab and helping to organize the data sheets.

Table of Contents

List of Figures

List of Tables

Chapter 1.0 - Introduction

Ecological restoration is primarily concerned with the response and actions taken towards degradation (point-source pollution, loss of habitat) often related to anthropogenic activities. This has been shown through the expanse of research focused primarily on the response to degradation, by seeking novel approaches to solving ecosystem problems (Ehrenfeld and Toth 1997; Holl et al. 2003; Suding et al. 2004; Manning et al. 2006). As a result, monitoring initiatives undertaken to measure the success of restoration activities and its goals have been minimal. The field of restoration is in need of a re-oriented approach that includes the study of the response of a system to restoration as well as developing techniques to enhance the practice of restoration (Choi 2003).

The introduction of the concept of "dynamic systems" (flux of nature and disturbance regimes) to ecology was the transition to the development of new concepts and tools for repairing degraded lands (Chapman 2006). With the creation of new concepts and tools for ecological restoration, ecologists have struggled through the development of a universally consistent terminology in both that field, as well as those coming from contributing fields and theories. Therefore, it has been difficult to determine what classifies as successful restoration. And, it has been exemplified by the conflict whether restoration should seek the use of restoration goals or endpoints or whether restoration should seek to continue working on an upwards path towards a "restored" condition (Bradshaw 1996; Ormerod 2003). Conceptual thinking has largely solved the problem of end-points through the use of goal or target setting, and the creation of desired attributes for a given restored system (Hobbs and Harris 2001; Ormerod 2003). However, critics of Hobbs and Harris (2001) and Omerod (2003) are struggling with the goal-setting concept. These individuals suggest that the success of a restoration project cannot be determined by meeting goals, due to the fact that the goal is simply unknown or not systemically determined (Grayson et al. 1999; Hackney 2000; Wilkins et al. 2003; Ryder and Miller 2005).

Despite conflicts in theory, the Society for Ecological Restoration (SER) journal, the SER Primer for Ecological Restoration, has adopted the concept of goal setting. The SER describes the fundamental goal of a restoration activity as a system which "contains sufficient biotic and abiotic resources to continue its development without further assistance" and to which "potential threats to the health and integrity of the restored ecosystem have been eliminated (SER 2002)." Through the evolution of goal setting and establishment of what constitutes a restored ecosystem, the research began to acknowledge the need to recognize whether or not a restored system has successfully re-obtained its biotic and abiotic resources and has eliminated threats to its integrity (Suding et al. 2004). The use of goal setting is important to be able to assess the success of a restoration activity. Without establishing the direction a restoration activity should take, it is difficult to assess whether the goal(s) have been accomplished and ecosystem function and services are re-established (Hobbs and Harris 2001; Hobbs 2003).

Researchers have been conceptually tackling the dilemma of what constitutes ecologically successful stream restoration in the field, and have been developing theoretical criteria and standards for the various disciplines of restoration (Palmer et al. 2005; Gillian et al. 2005). This has become an important element of restoration because of evidence that restoration projects have tried to re-instate processes not historically known to the landscape, specifically in stream channel form (Kondolf 2006). Analysts are emphasizing the importance of post-restoration monitoring, and working on the development of protocols, which impose rigorous post-restoration assessment strategies. Palmer et al. (2005) describes that the challenge ahead is to determine whether the standards and criteria that have been conceptually devised, and can be implemented as an in-field assessment protocol (Palmer et al. 2005). It also has to be acknowledged which indicators may provide the appropriate information for determining the successful restoration of a given system (Palmer et al. 2005). This involves the trial of numerous indicators and the possible creation of new ones that may satisfy the need to evaluate particular restored systems. Roni et al. (2005) suggests numerous different techniques for measuring stream restoration success and emphasizes the importance of integrating different monitoring approaches (e.g. ecological and geomorphological).

With the call for the design and trial of *in situ* techniques for quantitatively evaluating the success of restoration activities, the assessment protocol must be discipline specific (Bash and Ryan 2002). Qualitative measures have been utilized to access restoration activities, but are difficult to replicate and primarily deal with the social values gained in regards to the activity. The feedbacks of assessments focusing on social values may overlook an absence of increase or decrease in ecosystem services or processes. Qualitative measures for assessing restoration success, as described by Buckley and Haddad (2006), is a whole realm of study in itself and does not fit within the scope of this study. Recently Jones and Hanna (2004) have conducted trials for coastal shoreline restoration, and post-restoration success, utilizing soil stability indicators for a case study on a specific coastal classification type.

This project is focused upon filling a gap in the literature by contributing to the study of fluvial systems, by testing geomorphic and ecological indicators associated with the channel and its aquatic integrity. Similarly, other projects have begun to test various methods of stream channel restoration practices, such as the role of woody debris, boulders and importance of vegetative cover in riparian zones (Angradi et al. 2004; Lepori et al. 2005b; Rios and Bailey 2006).

Researchers from all different disciplines of ecological restoration are struggling to fill the gap in literature of various disciplines for the practitioners in the field. As science continues to work into the gaps of knowledge, it is becoming more evident that indicators have to be developed into the sub-discipline levels, because or since generic evaluation strategies cannot provide the consistency in measuring the success of restoration necessary.

1.1 Problem

The practice of ecological restoration of is in its infancy. Yet, ecological restoration is evolving and rapidly adapting new applications for different environments, including prairies, wetlands as well as riparian areas and many more. Much of the practice of ecological restoration is based upon the activity itself, and has become a catch phrase; where pollution ecologists strive to restore the soil, and fisheries biologists seek to restore collapsed fish stocks (Ormerod 2003). Despite all the action, and novel approaches to the restoration of various habitats, restoration has been subject to criticism over how to measure the success of a restoration activity. Hobbs and Harris (2001) discuss this need of measurement and appropriate goal setting. As they note, many different measurement techniques have been adopted, but these assessments are generally more useful for evaluating conservation status (Hobbs and Harris 2001). More recently, and specific to stream restoration conceptual frameworks have been developed for stream restoration that suggest what an ecologically successful restoration activity should consist (Palmer et al. 2005; Gillilan et al. 2005). However, little effort has been spent on establishing a method of testing success in the field (Bash and Ryan 2002). Therefore, further research is needed to pursue ways of determining a methodological framework which conforms with the accepted concepts of what a successful stream restoration activity consists of. Then, test the methodological framework in the field to determine whether restoration success can be quantified. This has begun in some fields. Jones and Hanna (2004) began to integrate monitoring into the practice of shoreline restoration in order to measure short term goals, as well as others in various restoration disciplines. Despite the use of restoration monitoring in other fields, the current absence of a consistent methodological framework for stream restoration places practitioners at a disadvantage when developing the design of a restoration project. This weakens monitoring initiatives due to the absence of comparison to sites of similar stream channel characteristics, and also due to the overall lack of knowledge transfer between practitioners.

1.2 Primary Question

Can a post-restoration monitoring and evaluation model be devised that effectively assesses the success of stream restoration projects by developing an approach which draws on both fluvial geomorphic and ecological theories?

1.21 Sub-questions

Q1. Is one evaluation method, ecological or geomorphic assessment better able to distinguish the success of a given stream restoration activity?

Q2. Or, would the monitoring and evaluation of restored stream reaches, based on a hybrid approach that uses indicators drawn from both geomorphological and ecological theories, provide a more informed understanding of the relative success of restoration?

1.3 Hypotheses

I hypothesized that stream restoration activities, as measured by the designed evaluation protocol in the proposed study will provide reliable quantitative feedback that will indicate the success, of a given restored stream reach. Therefore, the protocol will be able to demonstrate the effectiveness of the ecological and fluvial geomorphological techniques adopted in the study. I hypothesized that both ecological and geomorphic approaches will perform equally well on their own, as various ecological and geomorphic techniques have been used to assess restoration activities in the past (Kondolf 1995; Ryder and Miller 2005; Lepori et al. 2005b). Further, by testing two distinct approaches in the evaluation of stream restoration, it will be possible to determine whether a hybrid approach to restoration monitoring will provide a more informed decision base for the success of a given restoration activity. A result of apparent relationships between ecological communities and geomorphic function discussed by both geomorphologists and ecologists (see Kondolf et al. 2003; Sullivan et al. 2004, 2006; Lepori et al. 2005a) it was thought that a hybrid approach would provide a thorough indication of the relative success of the restoration activity on a given reach.

Chapter 2.0 – Introduction to the Methodological Framework

2.1 Review of the Literature

As I introduced in the problem section, the field of ecological restoration is in its infancy and has been steadily developing and expanding into numerous different disciplines. Therefore, a great deal of research is still being focused upon theoretical aspects of the field to conceptualize the basic understanding of how restoration should be undertaken (Allison 2007; Temperton 2007). Despite the many questions at the foundation of the science, much time and effort has been extended into finding techniques to restore ecosystem conditions in all types of environments. However, to understand the creation of techniques for restoration, the basic background of restoration must be acknowledged as well. In reference to this study, the role of ecology and fluvial geomorphology in the development of stream restoration and their methodological techniques are of particular interest.

Ecological (or ecosystem) restoration is one of the many applied fields that emerged from ecological theory, as well as from social and economic theories (Choi 2003). Numerous different studies have revealed new ideas for the conceptual approach to ecological restoration and how the systems that are studied should be perceived (Suding 2004; Chapman 2006). Here, I focus on the application of ecological theories, and how they pertain to monitoring for restoration success.

To define restoration and whether a system has been successfully restored we must be able to define an ecosystem and ecology because we must be able to work within its basic framework (Ehrenfeld and Toth 1997). Also, it is extremely important that practitioners be able to satisfactorily define all the components of an ecosystem, its boundaries, and all its flows before it is even possible to attempt a restoration strategy for an ecosystem, let alone delineating a trajectory towards its outcome (Ehrenfeld and Toth 1997). Anand and Desrochers (2004), show that the complexity of ecological restoration itself must not be simplified by using rudimentary definitions of ecosystem. Odum (1969) defined an ecosystem as “a unit of biological organization, with interactions within its system so that a flow of energy leads to characteristic trophic structures and material cycles within the system.” Kay (1993) proposed that the definition for ecology and ecosystems should also encompass the concepts of how conditions are continuously changing throughout space and time. This relates to the idea of the "flux of nature" which was alluded to in the introduction. The concept of disturbance regimes versus states of equilibrium (i.e. climax communities) in ecology plays an important role in ecological restoration (Chapman 2006). Suding et al. (2004) also discussed disturbance, but also the use of succession and the idea of feedbacks in alternative states, which may help to predict system collapse; an important element in monitoring for restoration success.

The initial role of ecological restoration was to be the "acid test" for ecological theories, and test the foundations and knowledge of ecology in the field (Bradshaw 1987; Michener 1997; Young et al. 2005). Ecological restoration has been the catalyst that has brought about greater understanding to various disciplines of ecology (e.g. plant ecology) (Young et al. 2005). Therefore, it can be acknowledged that the relationship between ecological restoration and ecology can be viewed as mutualistic. However, Lake et al. (2007) suggests that this transfer between ecological theory and restoration has not been occurring in stream restoration projects; the focus has rather been on implementation strategies.

The role of geomorphology in ecological restoration is the same as its ecological counterpart, often being used as a tool for restoration. However, geomorphology and, more specifically with reference to my study; fluvial geomorphology does not make up the backbone of restoration in the way that ecological theory does. Geomorphology can be more easily defined as a tool for restoration, and restoration, therefore acts as an "acid test" for geomorphology theory. Newson and Sear (1998) describe fluvial geomorphology as the science that studies the evolution and behaviour of river channels at various scales from cross-sections to catchments. The science also studies the range of processes and responses through a time scale (Newson and Sear 1998). Kondolf et al. (2003) suggest that geomorphology is an appropriate management and monitoring tool for restoration, as it is possible to test the effects of restoration practices, and determine the lifespan of given restored habitats based on aquatic habitat monitoring. It is through aquatic habitat monitoring that the roles of geomorphology and ecology become importantly linked in the design of post-restoration monitoring strategies. Therefore, it can be observed that geomorphic indicators provide feedbacks on the ecological integrity of a watercourse, and similarly the ecological indicators for geomorphic integrity.

2.11 Goal Setting in Ecological Restoration

In the field of ecological restoration and monitoring, the importance of setting goals cannot be understated. Establishing a strong restoration goal(s) and objectives provides benefits for the restoration activity itself, in terms of defining the trajectory of the activity, but it also provides a starting point which monitoring can be conducted from.

Throughout the use of ecological restoration there have been different forms of goal setting. Historical use of goal setting focused on setting specific restoration endpoints, where a particular community of organisms was the expected result. Although this method does provide a basis from which monitoring can start from, it does not account for ecological processes, such as disturbance, and evolution (Lake et al. 2007). By not taking into account ecological processes, the potential for restoration success is minimal if it is compared to the original goal.

The approach that is now receiving growing recognition is the use of "futuristic" approaches, and the use of dynamic goal setting. Choi (2003) described a "futuristic" approach to ecological restoration and acknowledged the unpredictable nature of ecological communities and ecosystems. Palmer et al. (2005) describe the use of establishing a dynamic goal or a "guiding image" in river restoration, suggesting that the successful restoration of system process is greater. For the use of dynamic goal setting, the monitoring strategy must be dynamic as well, but must also work off from a basic monitoring framework, which can easily adopt new parameters as a restored site evolves. This is the reason why the collection of baseline data is crucial before establishing attainable goals for a restoration activity (Lake 2005).

2.12 Ecological Restoration and Monitoring

Scientists have been so devoted to, and interested in, utilizing restoration as the means to test theories in ecology, geomorphology and various other disciplines but the practice of ecological restoration monitoring and its theories has remained untested. This has left vast crevasses in literature related to the post-monitoring of the effectiveness of the techniques and the ecosystems that have been "restored." It has been only until recently that this gap in the literature has been identified, and studies have begun to broaden the restoration framework to encompass post-restoration monitoring strategies and standards (Bash and Ryan 2002; Alexander and Allen 2006; Bernhardt et al. 2007).

Bash and Ryan (2002) provide significant evidence to suggest that post-restoration monitoring has not been conducted, or has been conducted but assessment parameters varied, providing insufficient information and suggest that standardized monitoring guidelines should be established. In *Monitoring Stream and Watershed Restoration*, Roni (2005) re-iterates the findings of Bash and Ryan, stating that only 10% of stream restoration projects were monitored.

Also in 2006, Alexander and Allen tabulated the number of projects in the upper Midwest United States which had undertaken a monitoring strategy and concluded the same result as the earlier studies had done. Most recently, O'Donnell and Galat (2007) found that only 34% of projects in the Upper Mississippi River Basin were using quantifiable methods to evaluate project success; many of these projects failed to collect data before and after restoration. Thus making it impossible to compare future field based assessment results. In another study, it was shown that two-thirds of restoration projects completed in the United States were said to have been completed successfully, even with the absence of measureable objectives (Bernhardt et al. (2007). Even with cost of stream restoration activities in the United States (which included in-stream habitat enhancement, channel stability and improvement of water quality projects) being estimated at $1 billion per year (Bernhardt et al. 2007); there was no still evidence of an increase use of monitoring techniques (Alexander and Allen 2006). As a result of the lack of monitoring in restoration, the learning curve of the field and various techniques employed has been inhibited (Alexander and Allen, 2006; Lake et al. 2007); as well as ecological theory itself.

A significant aspect of addressing the need for monitoring is the observed absence of funding for such endeavors. Long term monitoring is expensive, and funding agencies have not provided incentives to undertake the task (Bash and Ryan, 2002). However, this situation provides a catch-22, such that if restoration is not proven to be successful it will lose public support (Woolsey et al. 2007).

Due to the growing evidence of poor monitoring protocols, research has set about determining appropriate frameworks to evaluate successful restoration. Palmer et al. (2005) developed a set of conceptual standards for river restoration that addresses this need for post-restoration monitoring. Similarly, key issues of poor monitoring regimes were discussed in Roni (2005). However, in both cases the monitoring frameworks developed were theoretically based and formulated on suggestions for evaluating various stream restoration activities. A possible explanation for the slow shift to restoration monitoring is provided by Allen et al. (1997). From a workshop organized by the National Science Foundation (NSF), attendees of a number of different disciplines commented upon the use of ecological restoration. Allen et al. (1997) suggested that the absence of a strong methodological framework to be utilized to perform basic research has been deterring ecologists and conservation biologists from becoming involved in ecological restoration. However, in order to bridge the gap between design and practice, methodological approaches must be undertaken to determine effective means to monitor restoration. Holl et al. (2003) acknowledges that the development of methodological studies for post-restoration monitoring is difficult, primarily due to variation between sites. Generally, empirical studies focus on procedures that rely heavily on replication, or the study of a highly specific method of stream restoration, such as the role of woody debris as described in Angradi et al. (2004). However, a dataset of restored sites maintaining identical characteristics is rare. Holl et al. (2003) suggests that research endeavors should not get discouraged by heterogeneity of restored sites. Therefore, in order to successfully quantify the success of restoration activities, the variations between sites must be taken into account. For example, criteria for site selection must use general classification schemes (e.g. Rosgen's channel classification) on a landscape scale to minimize site specific noise of minor soil variations, or vegetation communities.

The appropriate use and, therefore, potential success of a monitoring strategy is largely dependent upon the type initiative undertaken. Roni (2005) describes the types of monitoring techniques that can be adopted for restoration; baseline, status, trend, implementation, effectiveness, and validation. It is important to recognize that all types of monitoring are equally important, but they are dependent upon the goal of the monitoring exercise. In order to design an appropriate monitoring strategy, careful consideration must be taken into determining which type is best suited for a particular monitoring initiative.

More specifically in regards to the discipline of river/stream restoration, a significant amount of discussion and research has been conducted on a conceptual approach what methodological frameworks should be based upon. Boon (1998) began discussing the importance of river restoration by exploring restoration through a series of dimensions, understanding the conceptual, spatial, temporal, technological and presentational dimensions. A number of other approaches have been taken which have been generally associated with the setting of restoration goals and methodologies that the goals will meet (Kondolf 1995; Ehrenfeld 2000; Pedroli et al. 2001; Palmer et al. 2005). Criteria for measuring success by using goals was proposed by Palmer et al. (2005) and supported by Gillilan et al. (2005). These are among many guidelines that have already been published that deal with how or what parameters should be included into a monitoring (MacDonald et al. 1991; Bauer and Ralph 1999; Kaufmann et al. 1999; Pollock et al. in Roni, editor). Acceptance of methodological standards for monitoring can allow for the design of rigorous and appropriate monitoring strategies. In order to design appropriate restoration monitoring activities, the goals of the initial restoration activity must be clearly understood and accounted for. Therefore the monitoring design must reflect the goals, and choose the appropriate parameters to determine whether the goals have been met. Also, the information gathered must be able to provide feedback to the original management decisions (Palmer et al. 2005; Roni et al 2005; in Roni, editor). This allows for adaptation to the original design, which could prevent the result of restoration failure, or re-evaluating the scale of the original goals, to achieve the next level of restoration success.

In light of the obvious recommendations on the importance of goal setting and feedback to the initial restoration design and management decisions, generally much of the criteria lack emphasis upon specific testable parameters for the post-monitoring of river restoration activities, which provide the necessary insight into successful restoration. This is due to the fact that there are few thorough quantitative evaluations of stream restoration projects have been undertaken (Roni et al. 2005; in Roni, editor). In the existing evaluative methods, the current literature shows conflict between fields of thought in the restoration community and the indicators that should be further developed to monitor restoration of stream/river systems. This conflict concerns how restoration monitoring should be undertaken. Ryder and Miller (2005) suggest that the Hobbs and Harris (2001), Harris (2003), and Lake (2005) perspective of utilizing ecological/stability based techniques to evaluate system structure does not indicate a viable system. Those that support the Ryder and Miller (2005) view, suggest that biological communities in ecosystems provide the indicators necessary to suggest whether a restoration activity can be deemed as successful. Due to ecological restoration's diverse background, conflicting views of how to approach the various dilemmas found within the field are common and are to be expected.

In relation to the conflict in the use of indicators and minor acceptance for standards for successful restoration, restoration and monitoring activities are still not appropriately evaluated and lack systematic approaches (Ryder and Miller 2005). Presently, a number of different indicators have been utilized to pursue accurate assessment of river restoration projects. Water quality, through the use of nutrient indicators, and chlorophyll for biological productivity are two examples of biological and ecological indicators utilized. Lepori et al. (2005b) utilized benthic macroinvertebrate sampling to assess the effectiveness of in-stream structures. Geomorphic indicators of percent moisture, vegetation cover, and substrate have also been recommended

Roni et al. (2005; in Roni, editor). Indices have also been developed that measure the stability of stream channels, most notably were the methods constructed by Rosgen (2001). Another strategy commonly adopted by southwestern Ontario conservation authorities is the Rapid Geomorphic Assessment; a highly qualitative personal judgement based activity. However, due to the conflict discussed, minimal research has been conducted on post-restoration monitoring, that contains both ecological and fluvial geomorphic approaches to evaluation.

2.2 Methodological Framework

As a result of the minimal research conducted on developing a methodological framework that integrates both ecological theory and fluvial geomorphology theory into stream restoration monitoring practice, I have presented following framework.

The task now at hand is to broaden the framework as the field of ecological restoration ages, evolves, and develops stronger associations with other fields of thought. This largely reflects the incorporation of long term monitoring and post-assessment of restoration activities. Suding et al. (2004) has recently acknowledged the need for a broadening of ecological restoration conceptual framework, and has pointed towards the inclusion of predictive tools for monitoring or assessing the success of restoration activities. In this study I developed a methodological framework by re-introducing theories already associated with the general concept of ecological restoration and applying them to a new component of the framework, the practice of stream restoration monitoring (Figure 1).

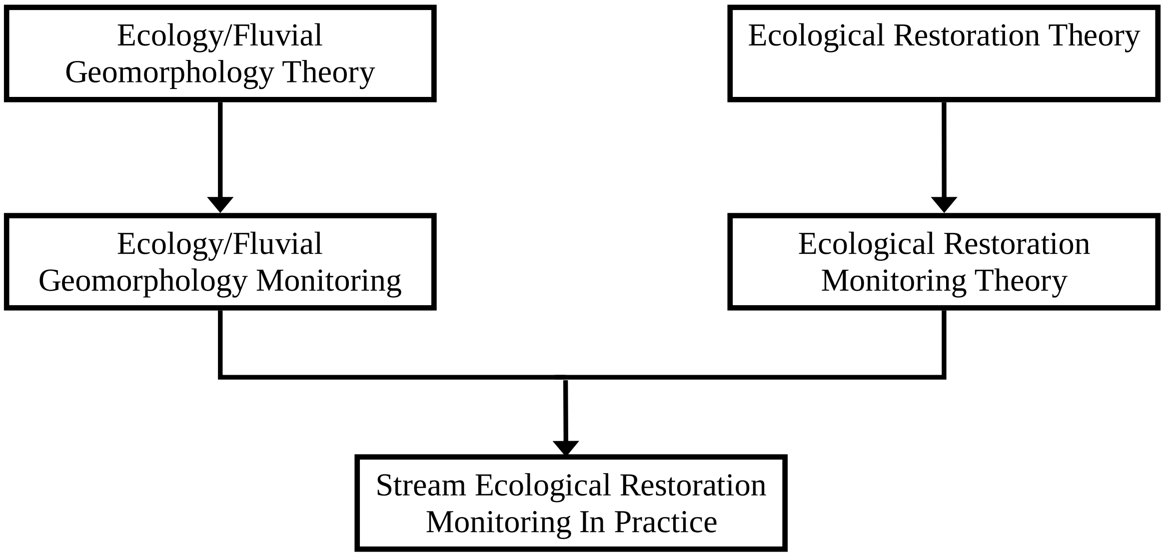


Figure 1. A methodological framework for the development of stream restoration monitoring.

Ecological restoration theory was used as the platform in this approach. I chose the use of ecological and fluvial geomorphology theories as a means to test a restoration project. These particular theories were chosen due to their multidisciplinary nature and wide range of techniques in the field (Roni et al. 2005; in Roni, editor). As Lake (2005) suggested, restoration ecology is already multidisciplinary in nature; and therefore its practice must also reflect that. More, specifically I chose ecological theory over others, focusing on biological characteristics, because the assessment techniques are generally more encompassing and can readily incorporate bio-assessment techniques. Ryder and Miller (2005), Schwartz and Herricks (2007) among others, have shown the ability of bio-assessment techniques to evaluate the ecological success of specific in-stream naturalization techniques. Further, ecological techniques can also incorporate the process of redesigning the physical attributes of the system, re-establishing nutrient and chemical balances etc., and re-introducing indigenous species, or removing exotics are key components to the theoretical practice of ecological restoration (Bradshaw 1996). Also, by pairing various ecological and geomorphic measures into the methodological framework for stream restoration monitoring it coincides with the strong body of literature describing the relationship between ecological and geomorphic variables within overall stream integrity (Kondolf et al. 2003; Sullivan et al. 2004, 2006; Lepori et al. 2005a). A broad framework can be more effectively used when taking a dynamic systems approach to restoration monitoring. Lake

(2005), suggested that stream restoration should be studied at the catchment scale; viewing each catchment as a system and develop restoration goals accordingly. This strongly agrees with the use of Palmer et al. (2005) “guiding image” and Choi (2003) dynamic goal setting previously discussed.

The methodological approach I have described can be easily adopted into various other models specifically developed for monitoring restoration success with the use of goal setting. These models which generally consist of the practice of setting a restoration goal, determining a trajectory, establishing an approach, and evaluating progress have been described by Kondolf (1995), Boon (1998), Palmer et al. (2005) and Roni (2005). Roni (2005) describes a thorough model for monitoring for stream restoration success. I adopted and re-organized his model for this study (Figure 2).

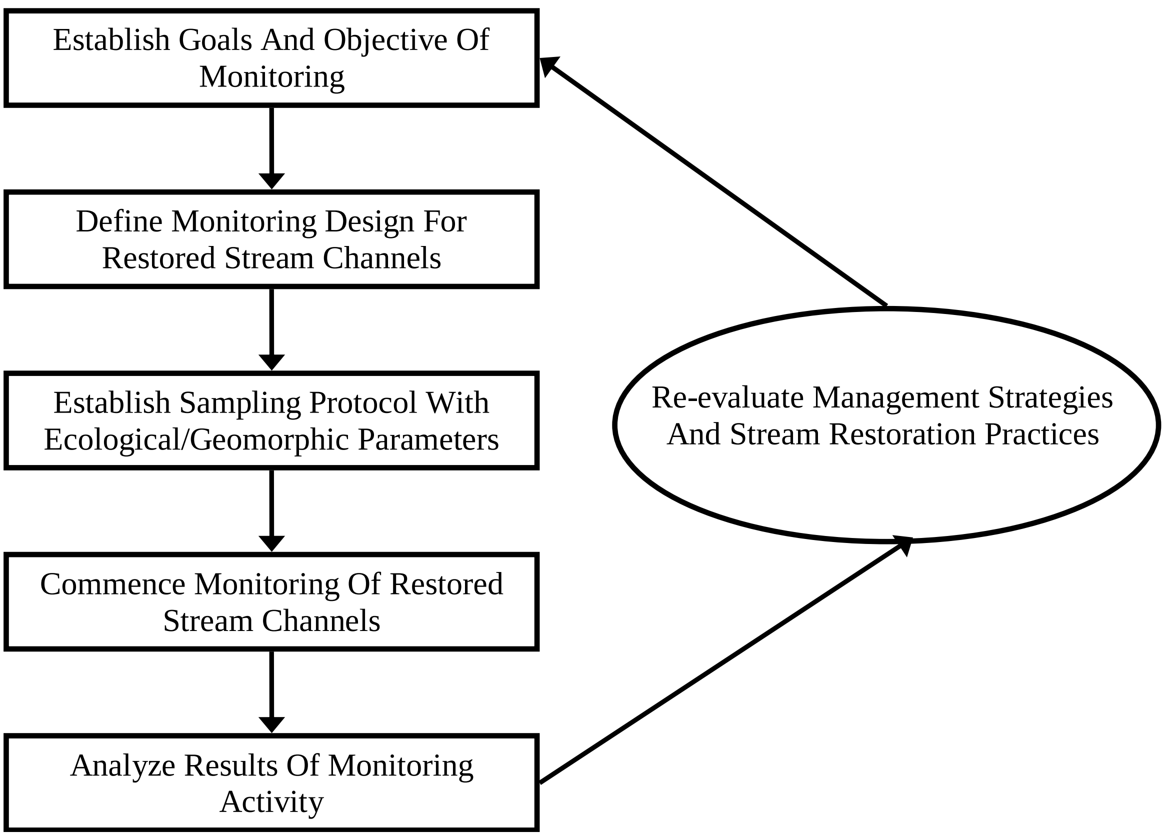


Figure 2. Framework of design for stream restoration monitoring; adapted from Roni et al. (2005; in Roni, editor).

Ecological indicators combined with a fluvial geomorphology index help to determine whether an appropriate post-restoration monitoring criteria can be successfully adopted for a given channel classification or stream order. This methodology uses a framework that allows for feedback between ecological and fluvial geomorphic indicators in monitoring design which reflects the goals of the monitoring activity. Upon completion of the monitoring activity, a positive feedback loop reports the results of the monitoring to inform future stream restoration activities and related management decisions. The second component of the study provided an analysis of the presented ecological indicators and geomorphic techniques that were used to measure the restored sites. This analysis will determine which set of indicators are more appropriate for determining restoration success, or whether the combination of the two approaches, significantly demonstrates the state of direction of the restoration activity.

Chapter 3.0 – Field Study: Assessment of Methodological Approach

3.1 Introduction

The current literature available on restoration monitoring methodology is minimal (Bash and Ryan 2002, Giller 2005); and numerous in-field techniques have been adopted in order test related theories of restoration ecology (Angradi et al. 2004; Ryder and Miller 2005; Lepori et al. 2006; Schwartz and Herrick 2007). Sieving through the mass of techniques to establish a criteria of testable parameters suited to site characteristics and various stream orders is an immense challenge. Appropriate indicators must be chosen, and also the appropriate techniques to most accurately measure the selected indicators. Simplistic, qualitative methods of site assessment of restoration activities do not provide meaningful data in the long term. Therefore, specific quantitative measures must be applied to specific disciplines within restoration monitoring (Roni et al. 2005; in Roni, editor). Appropriate methods of assessment must be adopted that are specific to the discipline and specific to various levels of classification within a particular discipline. As a result, I proposed a methodological framework for testing stream restoration success.

Here I test the applicability of the methodological approach in the field by choosing basic ecological and geomorphic indicators. Ecological and geomorphic indicators were used to compare known in stream relationships (Lepori et al. 2005a; Chessman 2006; Doyle 2006; Rios and Bailey 2006). For this study, I chose benthic macroinvertebrates and geomorphic channel stability indicators to test the success of restored streams in City of Waterloo; these measures and their application will be discussed in further detail in the methodology section. However, it is very important to acknowledge that the indicators and methods chosen for this study are not necessarily the only techniques that could have been applied; such as chemical analysis (e.g. NO_3, PO_4), heavy metals, and chlorophyll. Organic matter, woody debris could have also been adopted in the geomorphic index (Angradi et al. 2004) The techniques and framework developed represent the foundation and integration of two bodies of theory to which all stream restoration monitoring should follow.

3.3 Study Area

Assessment of the streams took place in the Region of Waterloo, primarily located within the limits of the City of Waterloo. The City of Waterloo (43.3N, 83.32W; elevation 334.2 m) has a population of 114,700 in a 64 km^2 area (City of Waterloo 2008). Geographically the area is comprised of kames dominated by sand and gravel till, which are the primary channel forming substrate with overlying luvisolic soil (Chapman and Putnam 1984; Agriculture and Agri-Food Canada 1998; Food and Agriculture Organization of the United Nations 1998). The region also receives approximately 904 mm/yr of precipitation (University of Waterloo 2007).

All streams assessed were located within urban boundaries and maintained common landscape level stressors (i.e. land use designations). Natural reference sites were selected on the headwaters of the Laurel Creek, which is used as the City of Waterloo's reference for water quality. Natural reference sites were chosen based upon minimal exposure to urban land use impacts (e.g. intense residential, commercial or industrial development) and a study conducted on the Laurel Creek watershed in 1996 (University of Waterloo 1997). Disturbed reference sites were also highlighted by the City of Waterloo, as candidate areas for restoration. Both natural and disturbed reference condition sites were selected based on their near-urban influence and use by the City of Waterloo. All restored test reaches and reference reaches were located on sub-basins with the Grand River Watershed (Figure 3). Six previously restored streams (Clair Creek; 1999, Colonial Creek; 1996, Critter Creek; 2000-01, Forwell Creek; 2002, and Lower Laurel Creek; 1993-95) varying in time after restoration (~5-15 years) were selected for assessment (Figure 4). The restored sites were composed of various different restoration techniques, all using a natural channel design as a basic approach, supplemented with various erosion control and riparian habitat planting techniques.

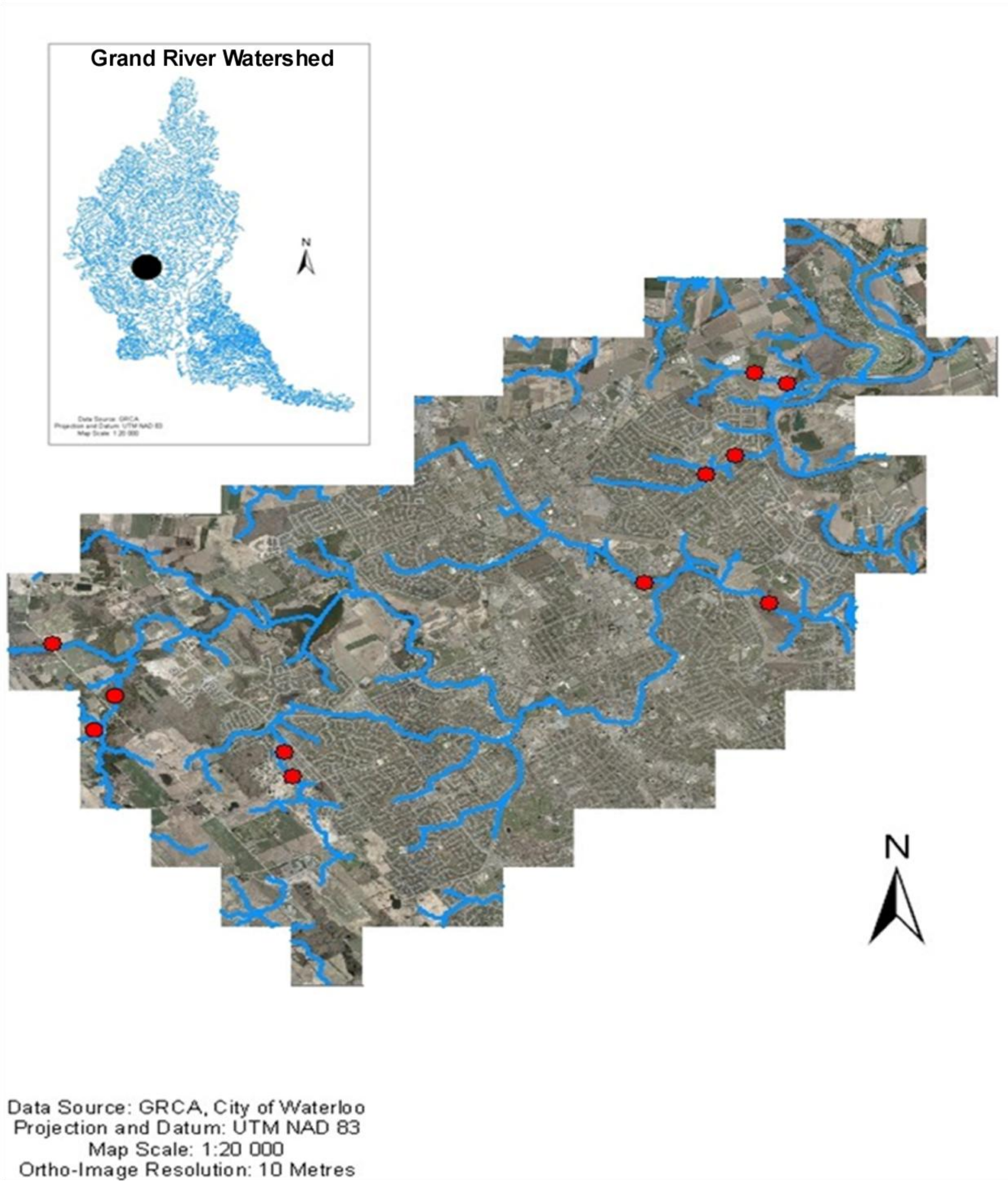


Figure 3. Stream sites sampled in the City of Waterloo, Grand River watershed. Each site contained multiple reaches (City of Waterloo 2001; Grand River Conservation Authority 2008).

Table 1. Description of restored reaches of South Clair Creek.

1. South Clair Creek Restored	Description
Year Restored	1999
Reaches Sampled	6
Biophysical Description	Located within Clair Hills residential development. Several stormwater retention ponds are located on either side of the channel. Channel Width – 1.8m Channel Depth – 0.30m
UTM Coordinates	17 T 0533940, 4812062

Table 2. Description of restored reaches of Colonial Creek.

2. Colonial Creek	Description
Year Restored	1996
Reaches Sampled	5
Biophysical Description	Valley encroached by residential development. Stormwater retention pond on top of valley. Channel Width – 2.5m Channel Depth – 0.40m
UTM Coordinates	17 T 0540343 4817012

Table 3. Description of restored reaches of Upper Critter Creek.

3. Critter Creek Upstream	Description
Year Restored	2000/2001
Reaches Sampled	5
Biophysical Description	Located beside RIM Park. Multiple offline ponds beside channel; channel has purged bank and entered ponds at various locations. Channel Width – 2.5m Channel Depth – 0.4m
UTM Coordinates	17 T 0540545 4814696

Table 4. Description of restored downstream reaches of Critter Creek.

4. Critter Creek Downstream	Description
Year Restored	2000/2001
Reaches Sampled	5
Biophysical Description	Located by the Grey Silo Golf Course; offline ponds located beside channel. Channel Width – 1.5m Channel Depth – 0.25m
UTM Coordinates	17 T 0541042 4818105

Table 5. Description of restored reaches on Forwell Creek.

5. Forwell Creek	Description
Year Restored	2002
Reaches Sampled	4
Biophysical Description	Located between commercial and residential development; log cribbing used on banks. Channel Width – 4.5m Channel Depth – 0.50m
UTM Coordinates	17 T 0539081 481487

Table 6. Description of restored reaches on Laurel Creek.

6. Laurel Creek Restored	Description
Year Restored	1993-1995
Reaches Sampled	4
Biophysical Description	Located in Betchel Park Channel Width – 5.0m Channel Depth – 0.60m
UTM Coordinates	17 T 0540802 4814468 Betchel Park

Table 7. Description of natural reference reaches sampled in Region of Waterloo.

Natural Reference	Description
Reaches Sampled	5
Biophysical Description	Located at various segments along the upper Laurel Creek; near urban residential influence. Channel Width – 4.0m Channel Depth – 0.35m
UTM Coordinates	17 T 0530579 4813767 17 T 0531870 4814185 17 T 0532598 4813987 17 T 0532638 4813942

Table 8. Description of disturbed reference reaches in the Region of Waterloo.

Disturbed Reference	Description
Reaches Sampled	7
Biophysical Description	Clair Creek-located in Clair Hills residential development. Valley enchroached by subdivision Channel Width – 2.0m Channel Depth – 0.55m Colonial Creek-located in Woolner Park; channel incased with gabion. Enchroached by residential development. Channel Width – 2.0m Channel Depth – 0.30m
UTM Coordinates	17 T 0533844,4811838 Clair Creek 17 T 0539893,4816696 Colonial N. Branch (Woolner Park)

3.4 Ecological Sampling Design

The assessment of the aquatic ecosystem integrity in the channel was conducted by utilizing an in-stream monitoring approach. However, only the benthic macroinvertebrate sampling technique was utilized in this study. Benthics are the most commonly used for ecological/biological indicators of freshwater ecosystems (Hawkes, 1979; Wiederholm 1980; Abel 1989; Bailey et al. 2004). A benthic analysis approach is adopted because it is believed that benthos demonstrate site specific relationships with geomorphic indicators, and rapidly respond to non-point source pollutants because they are relatively sedentary and have short lifecycles (Bailey et al. 2004). Although fish are often used as a biotic component in assessment, in this case it was not chosen because the macroinvertebrates were easier to collect and their communities are known to change to a greater degree with different stressors.

The technique sampled a series of 30 m reaches in length that were pre-defined with respect to a reference or test sites (previously restored reaches). The aquatic sampling and the reach length (~30 m) and stream width (2-5 m) was consistent for all the reaches evaluated, despite potentially small differences in the length of restored reaches. A minimum of 4 reaches in the restored basin were sampled.

Following the Ontario Benthic Biomonitoring Network protocol (Jones et al. 2004), benthos was collected walking upstream along the reach in a zig-zag across the channel profile (water's-edge to water's edge), characteristic of most benthos sampling techniques. The D-net

(500 µm mesh) travelling kick-sweep method was used as the collection method. Benthos collected were separated from debris and stored in formalin. From each site, 100 specimens were picked at random from a Marchant box and identified to family level. The Marchant box contained 100, 1 cm^3 boxes.

3.41 Analysis of Ecological Measures

To measure whether samples obtained from the restored reaches have improved following the restoration exercise, the samples of benthos from each reach were run through various statistical measures. Several descriptors were used to depict the communities found in each reach, including family richness, Simpson's Species Diversity, % EPT (*Ephemoptera; Plecotpera; Trichoptera*) and % Chironomidae. The use of the % EPT is a commonly used indicator of high quality streams and the influence of geomorphic condition on the benthic community (Plafkin et al. 1989; Lenat and Crawford 1994; Sullivan et al. 2004). The % Chironomidae is a common measure that indicates presence/absence of high nutrient and heavy metal loads in the water column.

Principal components analysis (PCA) was used to determine whether there were any relationships between the benthic metrics used in the ecological assessment, and determine the overall ability of the ecological assessment to quantify restoration success. The PCA was run with A Pearson correlation coefficients among the benthic metrics was conducted to ordinate the descriptors of the community and highlight relationships found between them. This was conducted using PCORD v.4. A standardized PCA was used so no one of the metrics would be given more weight in the analysis. Principle components analysis is an effective technique for ordinating community data. However, as community data sets become more heterogeneous results become distorted and difficult to interpret (Grace and McCune 2002).

Finally, Bray-Curtis ordination was used to provide a multivariate description of community structure to determine the median distance of each community from the reference communities. The restored sites were tested against benthic communities in disturbed reference and communities in natural reference reaches. PC-ORD was also used in the Bray-Curtis analysis, using the variance-regression method. The original Bray-Curtis method was avoided due to its sensitivity to outliers in the dataset (McCune and Grace 2002). With the use of a relatively small dataset in this study, the variance-regression method was chosen.

The five measures were then used in a set of criteria to determine where each restored test reach aligns in relation to the natural and disturbed reference conditions (Table 1). In this excerise, the criteria were given the term ecocriteria. This method of categorizing test sites was adopted from Bailey et al. (1998). To determine whether a test site was in natural reference the observed metric values must be greater than the first-quartile value (Family Richness, Simpsons Diversity, % EPT and Bray-Curtis), of the expected or natural reference range or less than the third quartile (% Chironomidae). All restored test reaches outside this range do not lie in reference. The same was conducted with the disturbed reference; except a test site would have to lie below the third-quartile (Family Richness, Simpsons Diversity, % EPT and Bray-Curtis) and above the first quartile (% Chironomidae) to lie in disturbed reference.

Table 9. Criteria to label restored test sites as in natural reference or disturbed reference. *Indicates the criterion value; to be within natural or disturbed reference.

		Family Richness	Simpson's Diversity	% EPT	% Chironomidae	Bray-Curtis Variation
	Minimum	21	0.433	3.36	17.5	0.79
	1st Quartile	23*	0.482*	5.86*	21.6	1.38*
	Median	25	0.714	26.9	53.2	1.78
Natural	3rd Quartile	25.5	0.843	28.5	71*	2.05
Reference	Maximum	26	0.850	36.9	71.4	2.53
	Minimum	13	0.675	0.336	10.7	0.89
	1st Quartile	14.5	0.721	0.346	12.4*	1.01
	Median	15	0.749	1.00	17.8	1.1
Disturbed	3rd Quartile	15.5*	0.759*	3.75*	26.6	1.34*
Reference	Maximum	16	0.801	9.00	33.3	1.79

3.5 Fluvial Geomorphic Approach

The geomorphic assessment of the restored and reference sites was undertaken by measuring a set of five parameters designed as indicators for stream bed and bank stability. Stability indicators in both the aquatic and terrestrial portion of a stream channel provide important feedback to habitat quality and overall ecological integrity, as well as basic physical channel sustainability. The index developed and put in practice, was a synthesis of Rosgen's Bank Erosion Hazard Index (BEHI), field techniques used by Annable (1996) which has often been adopted in southwestern Ontario (Newbury Hydraulics 2002). The BEHI has been more widely adopted in the United States by various organizations, such as the Arkansas Department of Environmental Quality. The study chose to remove itself from the Rapid Geomorphic Assessment protocol that has been commonly used in southwestern Ontario (Credit Valley Conservation 2002), as well as variations of EPA rapid habitat assessment (Plafkin et al. 1989; Ciesielka and Bailey 2007).

The rapid geomorphic assessment approach makes uses of a checklist system that is more qualitative and based on personal bias whereas the BEHI approach and measurements used by Annable (1996) lends itself to be a more thorough and replicable quantitative assessment. The BEHI approach has been widely adopted throughout the United States (Vermont Agency of Natural Resources 2004; Arkansas Department of Environmental Quality 2004).

The study utilized two stream bed measures; stream bed shear stress, calculated from stream velocity required to shear particular sediment particle size, and basic substrate heterogeneity (Appendix B). Researchers have linked different thresholds of shear stress with overall channel stability (Brookes 1988; Booth 1990; Bledsoe and Watson 2001; Bledsoe et al. 2007). Substrate heterogeneity also plays an important role in diversity of macro-invertebrates present as well as the geomorphic evolution of the channel itself. The relationship between habitat or substrate heterogeneity and biotic patterns and processes has been widely documented and are well understood (Turner and Gardner 1991) demonstrating the importance of substrate heterogeneity, when determining the relative health of a channel and its relationship to biotic organisms.

Bank stability measures that were adopted for the study included bank height to bank-full ratio, a root depth to bank height ratio and bank angle. These measures were adopted from

Rosgen's BEHI (Rosgen et al. 2001), and integrate the affect of all stream channel erosion processes. The method adopted acknowledges the importance of both stream bank/bed characteristics in relationship with their erodibility potential and the hydraulic/gravitational forces working on them.

These measures were taken a minimum of five times through a 30 m reach, and conducted on both reference and restored test sites. The measures were taken starting at downstream reaches and progressing upstream. It is important to acknowledge here that the geomorphic assessment was conducted following the sampling of benthic macroinvertebrates in order not to disturb the substrate and therefore negatively affecting the results of the ecological portion of the assessment.

The geomorphic assessment measures were taken on both the left and right bank of the stream, always measuring starting from the right bank and working across the profile. The bank height to bank-full ratio was obtained by measuring the distance from water's edge to the top of bank, from water's edge to the bank-full line, followed by calculating the ratio from the distances. Similarly, the rooting depth to bank height ratio was obtained by measuring distance the roots penetrated from the top of bank towards the water's edge. The bank angle was measured by using a hand held clinometer; placed half way up the bank height and then obtaining the degree.

An A. OTT Kempten current meter (50 mm diameter propeller) was used to take velocity measurements. A minimum of three velocity measures were taken through the stream profile, at approximately 75% of the depth. In some cases this was found not to be possible in riffles that had insufficient water levels. One measure was always taken in the thalweg of channel and others at evenly spaced locations between.

Following completion of the bank measures, a substrate sample was taken. The substrate was slowly dried in an oven at ~35 ^{o}C until all the moisture had been removed. The samples were then put through a mechanical sieving machine for seven minutes. Large coarse gravel was hand sieved out prior to placing the sediment in the mechanical sieve. The overall weight of the sample was taken and weight of the various substrate sizes (in mm) (Appendix B). Sieve sizes were chosen in order to capture a range of sediment sizes; sieves at the boundaries of sediment classes (i.e. gravel-sand, sand-silt). Sediment over 38.1mm was measured with a caliper and no sediment over 101.4mm was collected in the field. The largest predominant particle size collected was then used to calculate whether the stream bed was experiencing stream velocity over the limiting shear stress for that particle size. This was calculated by using an interpolation table provided by Vermont Agency of Natural Resources (2004) for particle entrainment and transport (Appendix B).

Each element was used to calculate the potential risk of erosion in the stream channel. This was done by the creation of indices, which represents an extension to Rosgen's BEHI. The higher score received on the indices corresponds with a greater risk of erosion and general overall instability of the stream channel. Therefore, a low score indicating low risk of erosion and greater channel sustainability (Appendix B).

3.51 Analysis of Geomorphic Approach

To test whether the geomorphic index effectively assesses stream reaches geomorphic integrity and habitat potential a non-metric multidimensional scaling (NMS) test was used. The use of NMS ordinations are common when the data is found to be non-normal, discontinuous or there is the use of questionable scales (McCune and Grace 2002). In this case, NMS was used to question the reliability of the scale used in the geomorphic index. To test the scale PC-ORD v.4, autopilot mode was used for the NMS test. The program was calibrated with a correlation distance measure and random starting configuration. Forty runs were computed with real data and 50 runs with randomized data. One dimension was used for the final solution.

The Bray-Curtis ordination was also adopted for testing the distance of the restored test site geomorphic index results to the reference sites median geomorphic index results. Similar to the Bray-Curtis ordination conducted on the benthic community of the ecological assessment, PC-ORD was used with a variance-regression method. Bray-Curtis tests are generally more commonly adopted in ecological studies (Bailey et al. 1998; Ciesielka and Bailey 2007). This was done on a per restored test reach basis. Comparing each restored test reach to the median natural and disturbed reference geomorphic index results. A comparative analysis of restored test sites to natural and disturbed reference allowed the placement of restored test sites on a range between the two reference condition ranges (Table 2)

The geomorphic index results and the Bray-Curtis variation were used to determine where each test reach aligned in relation to natural and disturbed geomorphic reference condition. For a restored test reach to lie within natural reference, the value for the geomorphic index and Bray-

Curtis must be below the third-quartile. The same is true for disturbed reaches. However, the test reach must lie above the first quartile value (Table 2).

Table 10. Geo-criteria for assessing a restored test reaches alignment with natural and disturbed reference condition calculated from geomorphic assessment index and Bray-Curtis Variation. *denotes the criterion value. In natural reference the reach values must be less than the indicated value and disturbed reference the Index Value must be greater.

		Index Value	Bray-Curtis Variation
Natural Reference	Minimum	22.60	0.000
	1st Quartile	22.65	0.0100
	Median	23.90	0.0600
	3rd Quartile	24.70*	0.100*
	Maximum	26.55	0.320
Disturbed Reference	Minimum	17.40	0.000
	1st Quartile	27.48*	0.0200
	Median	32.35	0.0900
	3rd Quartile	33.53	0.210*
	Maximum	37.65	0.310

3.52 Analysis of Combined Assessment

In order to determine the relationship between the ecological and geomorphic indices as measure of whether a hybrid approach would be best suited for stream restoration monitoring protocol a canonical correspondence analysis (CCA) was also undertaken. This method was used to determine whether the benthic community descriptors (Spp.) are related to geomorphic "environmental" (Envt.) variables in the assessment process. However, CCA is limited to number of environmental variables that can be tested. In CCA, if the number of environmental variables is similar to the number of sites, the constraints between the axes becomes weaker and do not discriminate between community and environmental variables (McCune and Grace 2002). The application of the CCA as described by ter Braak and Verdonschot (1995) is a popular multivariate analysis technique among aquatic ecologists for measuring environmental influences on benthic communities (McCune and Bruce 2002; Bailey et al. 2004). The use of the CCA test has also been commonly used to quantify the relationship between macroinvertebrates and various environmental influences, specifically silt and chemical compositions in stream environments (Dodkin et al. 2005). The CCA used all four of the ecological metrics used to describe the benthic community and the five geomorphic index measures. Scores were standardized with the centering and normalizing method in PC-ORD, optimizing the benthic metrics and geomorphic measures using Bi-plot scaling and a Monte Carlo test. The resulting species (Spp.)-environment (Envt.) correlations and a p-value <0.05 was used to determine how strong relationship between the two assessment types used in the study.

A second CCA test was used in the study to test the strength of apparent relationships observed between benthic metrics and geomorphic index results. This was conducted because CCA assumes that relationships pre-exist between species community structure and the environmental variables (McCune and Grace, 2002). By using a pre-described community with benthic metrics (%EPT, Simpson diversity, etc) the CCA may show greater bias and therefore a stronger relationship. The same calibration as was used on the first CCA test was used again; at a p-value <0.05. The standard p-value of <0.05 was used in this study because the purpose of the study was to test whether relationships or effect exists at all. A smaller p-value may have eliminated the chance of observing the presence of weaker relationships (Bross 1971; Shinichi and Cuthill 2007). Therefore the null hypothesis that states there is no relationship, could not be rejected, even if some relationship existed. Future studies may test specific relationships between individual benthic metrics and geomorphic indicators and should explore higher significance levels.

Chapter 4.0 – Analysis of the Restoration Monitoring Methodology

4.1 Results

The assessment of the developed monitoring protocol used twenty-nine (n=29) restored test reaches in the City of Waterloo. A set of natural (n=5) and disturbed reference (n=7) reaches were also tested using the same parameters. Field work was conducted between May 8 and June 4, 2007.

The protocol used both ecological and geomorphic approaches for stream assessment to determine a comprehensive picture of the quality of restored stream reaches in relation to known disturbed and natural stream reaches within the City of Waterloo. Also, the assessment of the restored stream reaches was used to test the applicability of the monitoring protocol to effectively quantify the condition of the stream reaches and to determine whether one monitoring approach (ecological or geomorphic) would be best suited for determining stream restoration success.

4.11 Ecological Assessment Results

The ecological assessment used several measures of the benthic macroinvertebrate samples collected to quantify community structure. These measures were taken for both reference sites (i.e. natural and disturbed reaches) as well as restored test reaches in urban and near-urban reaches of the City of Waterloo. A range of values described by the use of quartiles used to describe the community condition of the reference reaches and to build an ecocriteria to compare restored test sites to (Table 3). The metrics used in the ecocriteria were used a benchmark for comparison of the restored test reaches. The metrics applied (Family richness, Simpson's diversity, % EPT, % Chironomidae and Bray-Curtis variation) were useful as they describe different influences and their affects on the ecological integrity of the water column (Table 3). Bray-Curtis was specifically used for describing a measure of the variation of the benthic communities from the reference.

Table 11. Results of post-restoration monitoring of restored test reaches (n=29) of six restored basins in the Region of Waterloo.

Site Name	Measure	Reach					
		1	2	3	4	5	6
1. S. Clair Creek	Richness	12	15	13	12	13	13
	Diversity	0.612	0.649	0.713	0.583	0.697	0.594
	% EPT	1.35	0.671	0.335	0.337	2.34	0.000
	% Chironomidae	59.8	37.6	45.9	22.2	34.1	21.2
	Bray-Curtis N. R.	1.97	1.19	1.39	1.62	1.58	1.60
	Bray-Curtis D. R	0.340	0.350	0.700	0.620	1.18	0.710
2. Colonial Creek	Richness	15	14	15	20	17	
	Diversity	0.818	0.815	0.630	0.831	0.808	
	% EPT	0.641	3.20	1.32	6.71	1.94	
	% Chironomidae	11.5	8.01	12.2	29.7	13.6	
	Bray-Curtis N. R.	1.66	1.60	1.98	1.68	2.16	
	Bray-Curtis D. R	0.470	0.400	0.750	0.300	0.320	
3. Critter Creek Upstream	Richness	9	14	15	12	10	
	Diversity	0.795	0.795	0.759	0.694	0.802	
	% EPT	0.324	1.66	2.85	0.334	0.00	
	% Chironomidae	2.92	9.27	21.35	15.7	11.0	
	Bray-Curtis N. R.	1.49	1.59	1.69	1.42	1.56	
	Bray-Curtis D. R.	0.490	0.440	0.660	0.650	0.490	
4. Critter Creek Downstream	Richness	24	19	14	12	14	
	Diversity	0.756	0.775	0.706	0.498	0.469	
	% EPT	3.68	4.47	2.35	1.32	1.65	
	% Chironomidae	6.02	7.90	5.70	1.98	2.97	
	Bray-Curtis N. R.	1.32	1.45	1.27	1.53	1.70	
	Bray-Curtis D. R.	0.440	0.750	0.690	0.760	0.700	
5. Forwell Creek	Richness	19	16	16	17		
	Diversity	0.765	0.748	0.672	0.788		
	% EPT	4.45	2.05	2.32	3.27		
	% Chironomidae	29.1	18.5	19.5	33.7		
	Bray-Curtis N. R.	1.13	1.31	1.07	0.760		
	Bray-Curtis D. R.	0.600	0.500	0.490	1.01		
6. Laurel Creek	Richness	18	13	21	17		
	Diversity	0.764	0.793	0.823	0.779		
	% EPT	12.6	14.3	12.2	12.6		
	% Chironomidae	10.3	10.9	16.4	11.2		
	Bray-Curtis N. R.	1.36	1.68	1.29	1.61		
	Bray-Curtis D. R.	0.490	0.780	0.600	0.650		

The principal components analysis of the benthic metrics in the ecological approach of the assessment showed greater variance than expected in the second and fourth axes; this was depicted by a greater actual eigenvalue than the broken-stick eigenvalue. However, the second axis showed the greatest variance (36.4 %), and was used in the analysis (Table 4). The fourth axis only representing approximately 7.00 % of the sample was too weak an association to conduct further analysis. The first axis sampled represented the greatest amount of variance. However, the observed eigenvalue of the random tests was less than the broken-stick eigenvalue, suggesting that the metrics used in the study do not account for a significant number of influences on the benthic community.

Table 12. Extracted variance from randomized tests (n=4) of the principle components analysis (PCA) on the ecological metrics of all reaches tested natural and disturbed reference and restored test reaches (n=41 reaches).

Axis	Eigenvalue	% of Variance	Cum. % of Variance	Broken-Stick Eigenvalue
1	1.792	44.800	44.800	2.0830
2	1.455	36.369	81.168	1.0830
3	0.472	11.794	92.962	0.5830
4	0.282	7.0380	100.00	0.2500

The resulting analysis of the second axis in the PCA depicted that family richness and %EPT were correlated (0.6574), having a moderate linear relationship. The other ecological metrics used in the study had little to no relationship with each other (Table 5).

Table 13. A cross-products matrix from the PCA showing relationship between ecological metrics of all reaches tested natural and disturbed reference and restored test reaches (n=41 reaches).

	%EPT	%Chironomidae	Family Richness	Species Diversity
%EPT	1.000	--	--	--
%Chironomidae	0.06164	1.000	--	--
Family Richness	0.6574	0.3431	1.000	--
Simpson Diversity	0.3083	-0.3981	0.08699	1.000

Following the analysis of the relationship between benthic metrics, the results as described by the benthic metrics were plotted against eco-criteria standards for reference condition. Test reaches observed above the limit denoted by “N” (Family Richness, Simpon’s Diversity and % EPT) and below “N” (% Chironomidae) are stated to be in natural reference.

Those found below the limit denoted by “D” (Family Richness, Simpson’s Diversity and % EPT) and above “D” (% Chironomidae) are stated to be in disturbed reference. Describing the Bray-Curtis variation results, a restored test reach observed below the determined reference limit is stated to be in reference (Figure 4.5 & 4.6).

As established in the ecocriteria in the methodology, for a reach to be considered in natural or disturbed reference it must lie within reference for all benthic metrics used. In this study it was observed that no restored reaches were found to lie within the natural reference range. Also, no restored test sites were observed to be in disturbed reference (Figure 4). The test sites appear to lie in relative position spread between natural reference and disturbed reference condition.

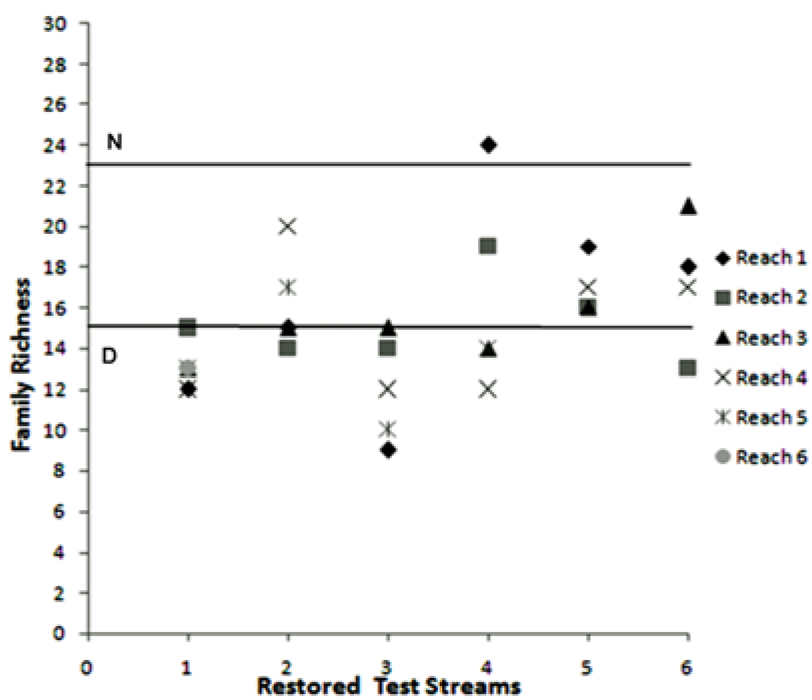


Figure 4.1. Results from reach levels of Family Richness plotted against ecocriteria reference limits for Family Richness.

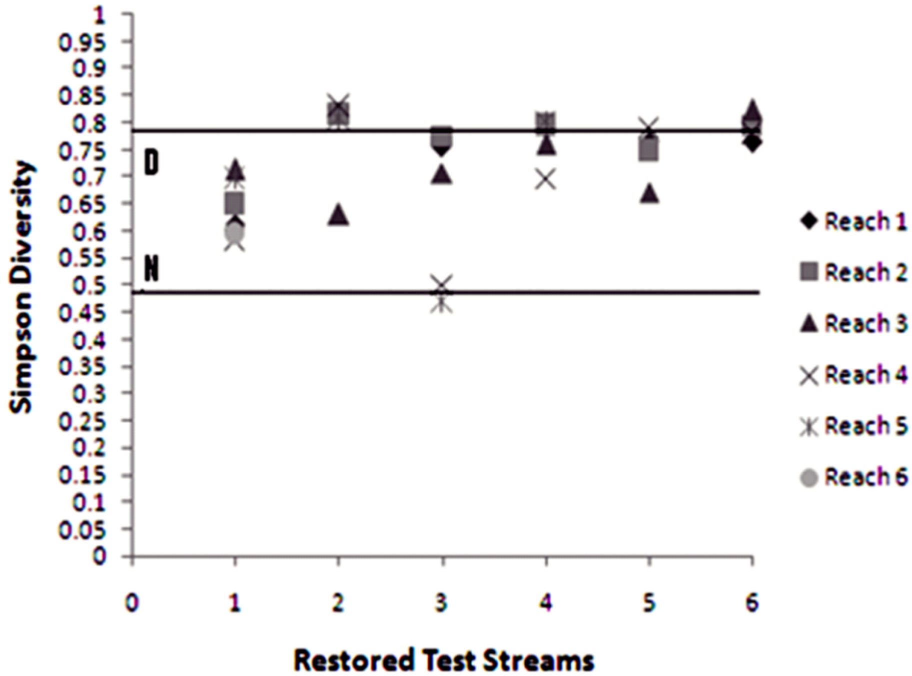


Figure 4.2. Restored reaches Simpson's Diversity results plotted against ecocriteria reference limits of Simpson Diversity.

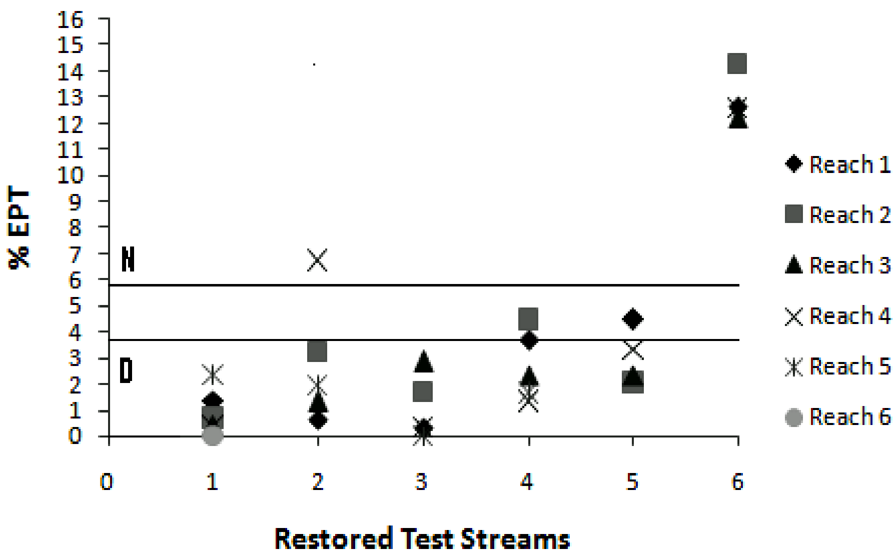


Figure 4.3. Results of restored reaches % EPT plotted against ecocriteria reference limits of % EPT.

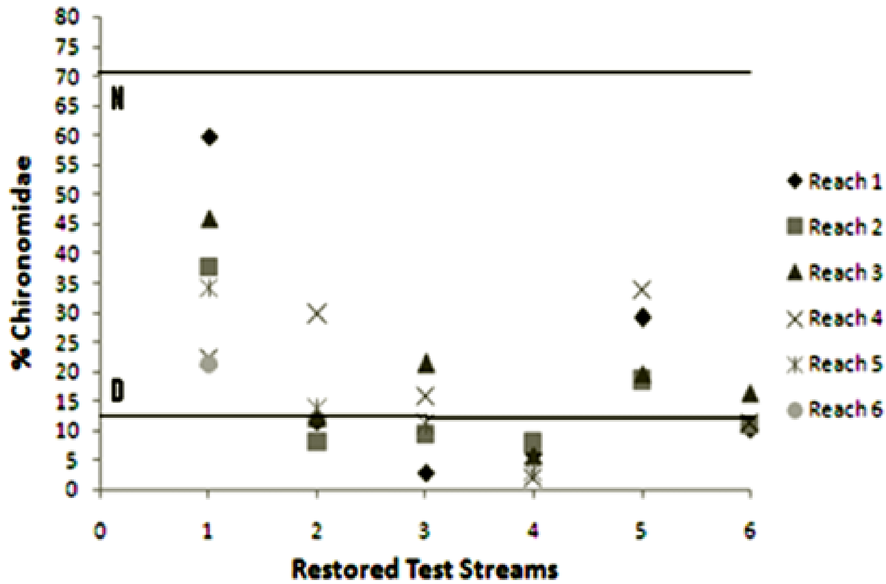


Figure 4.4. Results of restored reaches for % Chiromonidae plotted against ecocriteria reference limits of % Chiromonidae.

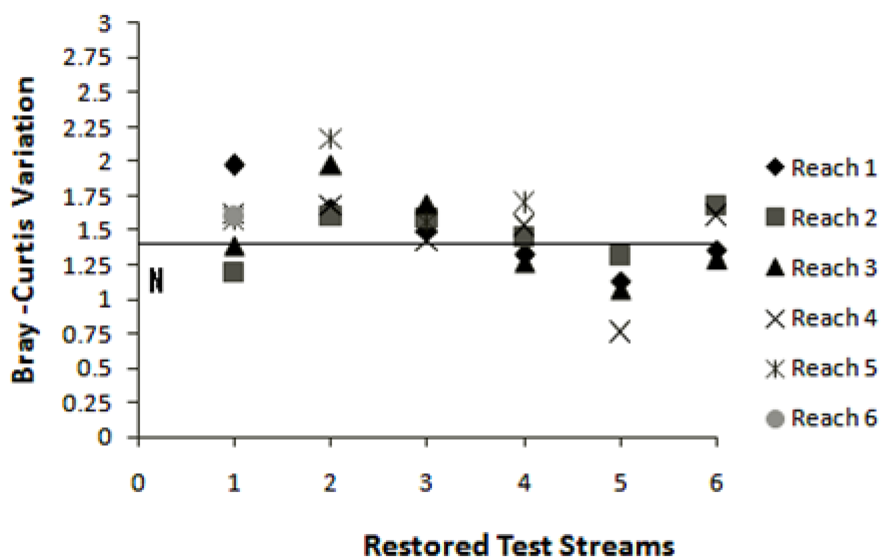


Figure 4.5. Results of restored reaches Bray-Curtis variation plotted against ecocriteria natural reference limit for Bray-Curtis variation denoted by “N”.

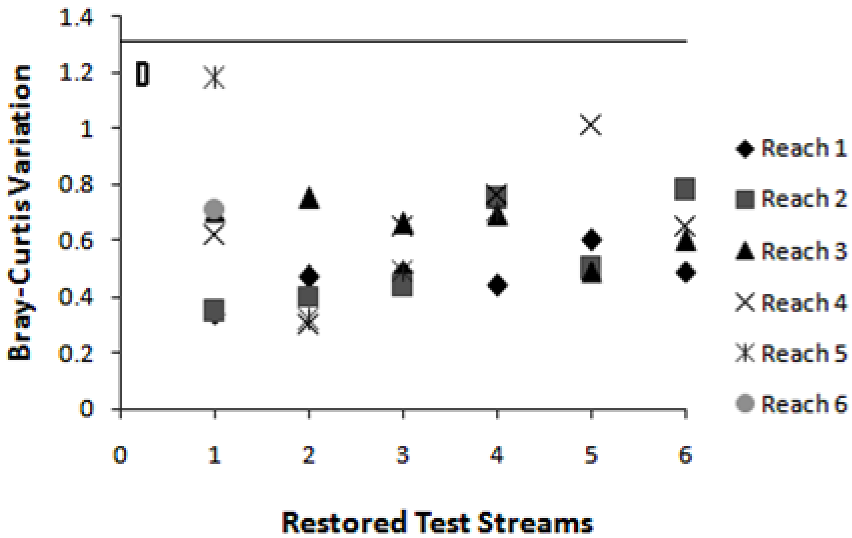


Figure 4.6. Results of restored reaches Bray-Curtis variation plotted against ecocriteria disturbed reference limit for Bray-Curtis variation denoted by "D".

4.12 Geomorphic Assessment Results

The geomorphic assessment of the restored test reaches was used to quantify the overall geomorphic stability and habitat of the stream channel. This analysis used a similar method as the ecological assessment of the aquatic integrity of the streams. A series of measures were used to quantify the channel characteristics (bank angle, bank-full height-bank height ratio, rooting depth-bank height ratio, substrate heterogeneity, and limiting shear stress) which were than used to calculate a geomorphic index value (Table 6). The measurement value was given a corresponding value in the index for each geomorphic measure (e.g. a bank angle of 20^{o} has an index value of 1.9 and would be labeled as very good). This occurred for each measure in a transect and then the total index value was calculated for the transect. The median index value was then determined for a reach (Table 6).

Table 14. Geomorphic sustainability index and subsequent field measurement values used to calculate total reach geomorphic sustainability; adopted from Rosgen (2001).

		Very Good	Good	Moderate	Poor	Very Poor	Extreme
Bank Angle (^{0})	Value	0-20	21-60	61-80	81-90		
	Index	1.0-1.9	2.0-3.9	4.0-5.9	6.0-7.9		
Root depth : Bank Height (m)	Value	1.0-0.9	0.89-0.5	0.49-0.3	0.29-0.15	0.14-0.05	<0.05
	Index	1.0-1.9	2.0-3.9	4.0-5.9	6.0-7.9	8.0-9.0	10
Bank Height : Bank-full (m)	Value	1.0-1.1	1.11-1.19	1.2-1.5	1.6-2.0	2.1-2.8	>2.8
	Index	1.0-1.9	2.0-3.9	4.0-5.9	6.0-7.9	8.0-9.0	10
Substrate Heterogeneity (%)	Value	100-80	79-55	54-30	29-15	14-5	< 5
	Index	1.0-1.9	2.0-3.9	4.0-5.9	6.0-7.9	8.0-9.0	10
Over Shear Stress Limit	Value	YES – R	NO - P	YES - P	NO - R		
	Index	0	0	5	5		
Total Reach Sustainability		4.0-7.9	8-15.9	16-23.9	24-31.9	32-36	36.1-50

The values of the geomorphic indices were described at the reach level. The greater the value observed in the geomorphic index, the lower the perceived stability of the reach was, and therefore the lower the index-value the greater geomorphic reach stability. Reaches were observed to show variable geomorphic index results between streams and within streams (i.e. reach level) (Table 6). Based on the geomorphic index results, reaches also were designated in a range from “Very Good” to “Extreme” (Table 6). The purpose of labeling of the geomorphic index results in a range from “Very Good” to “Extreme” simply allows for ease of interpretation during presentation and not to provide further analysis.

Bray-Curtis variation was used to quantify the difference between restored test reaches and disturbed and natural reference condition. The individual metrics for test reaches were compared to values of the same community but in natural or disturbed reference. Very little variation was observed between restored test reaches and reference conditions (Table 7).

The geomorphic index results and Bray-Curtis variation were used to quantify a placement of a reach near reference condition. In the geomorphic index, reaches had to have been observed below the denoted “N” to be placed in natural reference and above the denoted “D” to be in disturbed reference (Figure 5). Reaches observed between the reference condition limits are stated to be neither in disturbed or natural reference. The Bray-Curtis depicts the variation of the results of the geomorphic metrics. The closer the value is to zero, the closer the reach is to reference. However, for a reach to be within reference it must lie in that condition in both the geomorphic index and Bray-Curtis variation as described in the geocriteria, and not lie in both natural and disturbed reference in the Bray-Curtis variation. It was observed that no reach was found to be in natural reference or disturbed reference condition (Figure 5).

Table 15. Geomorphic assessment results for restored test reaches (n=29) from geomorphic index, its associated rating and Bray-Curtis varication values in the Region of Waterloo.

Site Name	Measure	Reaches					
		1	2	3	4	5	6
S. Clair Creek	Index Value	34.80	20.25	20.30	20.70	17.25	22.35
	Rating	Poor	Good	Good	Good	Good	Good
	Bray-Curtis N.R.	0.0350	0.0400	0.0450	0.0350	0.0300	0.230
	Bray-Curtis D.R.	0.0300	0.130	0.0500	0.0250	0.0200	0.0250
Colonial Creek	Index Value	24.40	21.50	32.75	33.35	33.00	
	Rating	Moderate	Good	Poor	Poor	Poor	
	Bray-Curtis N.R.	0.0400	0.0300	0.0400	0.0450	0.0200	
	Bray-Curtis D.R.	0.0200	0.0200	0.0150	0.0250	0.0100	
Critter Creek Upstream	Index Value	21.10	16.90	22.00	24.15	32.35	
	Rating	Good	Good	Good	Moderate	Poor	
	Bray-Curtis N.R.	0.0450	0.0350	0.0200	0.180	0.0800	
	Bray-Curtis D.R.	0.0250	0.000	0.0600	0.0750	0.0500	
Critter Creek Downstream	Index Value	23.50	20.75	23.70	27.30	27.40	
	Rating	Good	Good	Good	Moderate	Moderate	
	Bray-Curtis N.R.	0.0200	0.0900	0.100	0.0100	0.140	
	Bray-Curtis D.R.	0.0600	0.0400	0.100	0.0100	0.130	
Forwell Creek	Index Value	26.75	30.15	32.35	32.80		
	Rating	Moderate	Moderate	Poor	Poor		
	Bray-Curtis N.R.	0.0200	0.0450	0.0200	0.0800		
	Bray-Curtis D.R.	0.0100	0.0700	0.0100	0.0800		
Laurel Creek	Index Value	19.00	22.92	27.30	24.00		
	Rating	Good	Good	Moderate	Moderate		
	Bray-Curtis N.R.	0.0550	0.0800	0.0300	0.0350		
	Bray-Curtis D.R.	0.145	0.0100	0.0100	0.0100		

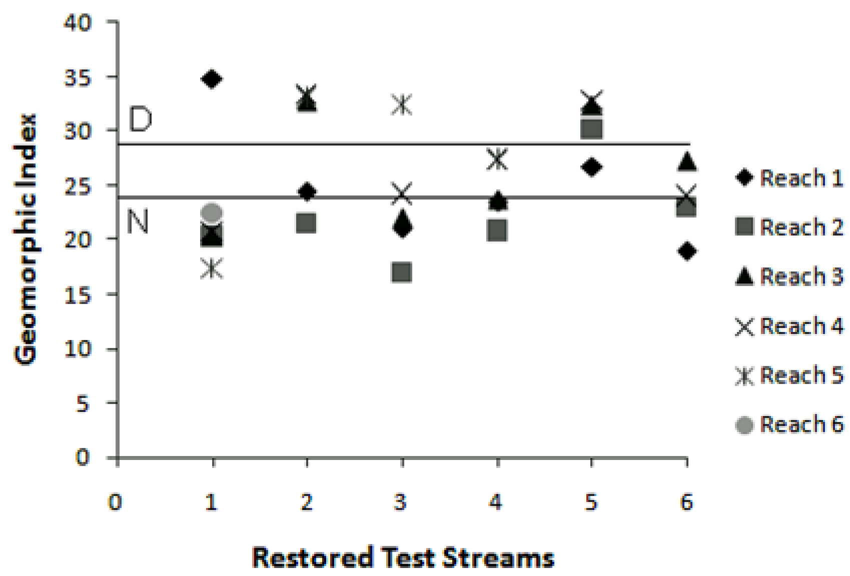


Figure 5.1 Results of restored reaches geomorphic index totals geomorphic plotted against geocriteria reference limits for geomorphic index values.

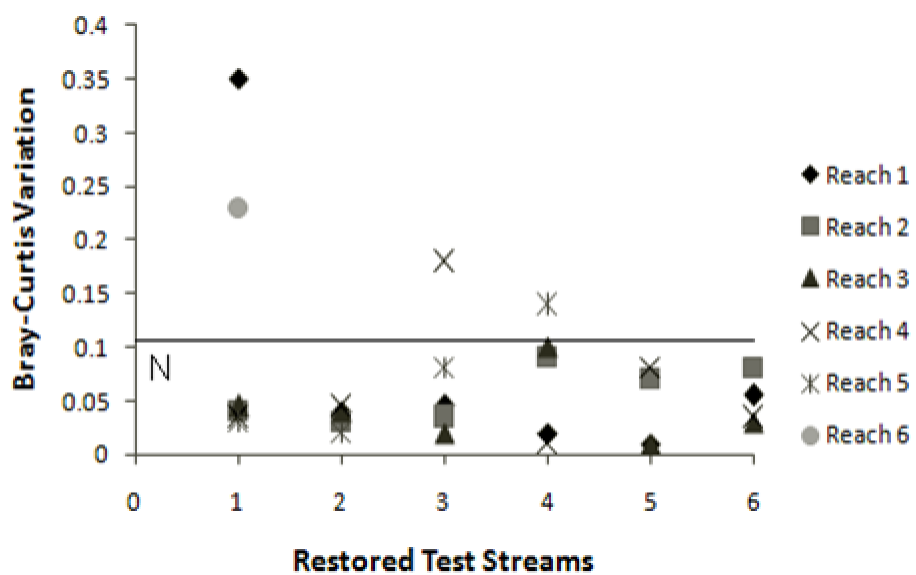


Figure 6.2. Results of restored reaches Bray-Curtis variation plotted against geocriteria reference limits for Bray-Curtis variation.

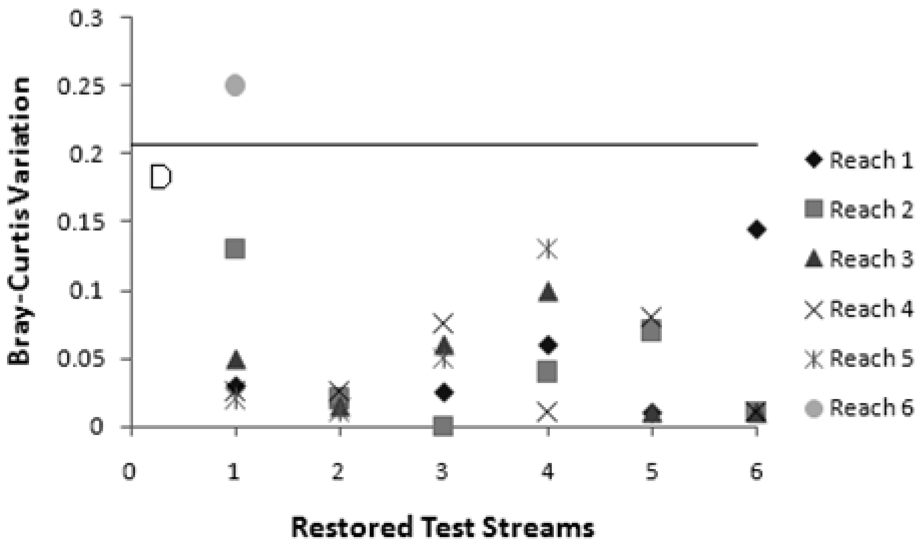


Figure 6.3. Results of restored reaches Bray-Curtis variation plotted against geocriteria disturbed reference limits for Bray-Curtis variation.

In order to test the strength of the geomorphic index a non-metric multi-dimensional scaling (NMS) test was conducted on the geomorphic assessment methodology. This test was used to determine whether the indicators and scale used, were appropriate and accurately measured the reaches sampled. The NMS used all reaches sampled in the study (n=41); the PC-ORD program chose a one-dimensional solution from the 50 randomized runs calculated by the Monte Carlo test.

Table 16. Stress in relation to the dimensionality (number of axis, n=6) comparing real data to randomized data of Monte Carlo test.

Stress in Real Data, 40 runs				Stress in Randomized Data, 50 runs			
Axes	Minimum	Mean	Maximum	Minimum	Mean	Maximum	p
1	25.306	35.645	49.798	25.267	40.055	50.000	0.0392
2	8.515	10.380	30.831	6.243	15.606	28.428	0.0784
3	1.259	2.149	17.297	0.000	5.553	12.148	0.0784
4	0.001	0.206	7.063	0.000	1.675	12.490	0.0980
5	0.000	0.003	0.037	0.000	0.043	1.528	0.0784
6	0.000	0.182	7.243	0.000	0.000	0.004	0.3333

The analysis of the one-dimensional solution (axis 1), accepting a p-value <0.05 resulted in a final stress of 25.31 and instability of 0.00. A stress value is the measure of accuracy. A stress value >20, has been shown by Clarke (1993) to be difficult to interpret. It has been suggested that stress should be generally found between 10 and 20; the closer stress moves towards 20 the more difficult the results are to interpret (McCune and Grace, 2002). Saintilan (2004) showed how low stress values (<5) accurately depict relationship between geomorphic variables. Therefore, the final stress of 25.31 observed from the results from this study is high, demonstrating no relationship between the measures and scale adopted for the geomorphic assessment. Therefore, the geomorphic assessment requires adjustment and re-evaluation to be an accurate assessment protocol in the field.

4.13 Testing a Hybrid Approach

The Canonical Correspondence Analysis (CCA) was used to evaluate whether there were any correlation with the benthic metrics and geomorphic measures. This measure helped to determine whether a hybrid approach for stream restoration monitoring could be used. The CCA made use of the Monte Carlo test with a null hypothesis; $H_{0,}$ that there is no structure in the main matrix and therefore no linear relationship between the matrices tested. The first axis was analyzed because it was observed to have the greatest percent explained variation (18.6%), the other axes explained little variation (Table 9).

Table 17. Canonical correspondence analysis (CCA) with benthic metrics and geomorphic variables; depicting a moderate relationship between the techniques.

	Axis 1	Axis 2	Axis 3
Observed Eigenvalue	0.003	0.001	0.000
Variance of variables % of variance explained Cumulative % explained	 18.6 18.6	 6.2 24.8	 1.1 26.0
Pearson Correlation, Community Metrics-Envt.	0.696	0.382	0.243
Kendall (Rank) Corr., Community Metrics-Envt.	0.520	0.156	0.146
P	0.340	0.600	0.850
Mean Eigenvalue	0.002	0.001	0.000
Minimum Eigenvalue	0.001	0.000	0.000
Maximum Eigenvalue	0.006	0.003	0.001

Having accepted the first axis for analysis, because the observed eigenvalue was found to be greater than the expected, a species metric-environment correlation was conducted using 99 runs of randomized data with the Monte Carlo test. Axis 1 showed the greatest species metric-environment correlation, accepting a p-value <0.05 (Table 10). From the analysis of axis 1 it was concluded that the H_0 must not be accepted, and that there is relationship between benthic community descriptors and geomorphic stability/habitat in the reaches sampled.

Table 18. Monte Carlo test results of Species metric - Geomorphic correlation as determined from axis 1.

	Real data	Randomized data Monte Carlo test, 99 runs			
Axis	Spp-Envt Corr.	Mean	Minimum	Maximum	P
1	0.696	0.529	0.329	0.780	0.0300
2	0.382	0.380	0.206	0.567	0.450
3	0.243	2.93	0.104	0.536	0.6900

A second CCA test was used to clarify the strength of the relationship between the benthic community and the geomorphic variables, using the hypothesis that a CCA test assumes a relationship already exists between the community structure (benthic metrics) and environmental variables (geomorphic index). Following the same methodology and null hypothesis as the first CCA test, it was observed that the first axis explained the most variation (14.5 %), and had an observed eigenvalue greater than expected in the Monte Carlo test (Table 11). The Pearson correlation described a strong relationship of 0.812.

Table 19. Canonical Correspondence Analysis (CCA) second test using benthic community and geomorphic variables.

	Axis 1	Axis 2	Axis 3
Observed Eigenvalue	0.325	0.094	0.089
Variance of variables % of variance explained Cumulative % explained	 14.5 14.5	4.2 18.7	4.0 22.6
Pearson Correlation, Benthic community-Envt.	0.812	0.603	0.497
Kendall (Rank) Corr., Benthic community-Envt.	0.356	0.359	0.344
P	0.0100	0.610	0.150
Mean Eigenvalue	0.144	0.099	0.075
Minimum Eigenvalue	0.076	0.066	0.049

Maximum Eigenvalue	0.198	0.147	0.102

The analysis of the first axis using randomized Monte Carlo test, showed a strong community (Spp.) - geomorphic (Envt.) correlation (0.812). The observed p-value 0.140, <0.05 accepts the null hypothesis that there is no relationship between benthic community structure and geomorphic index results, due to a low confidence level in the correlation (Table 12).

Table 20. Monte Carlo test results of benthic community - geomorphic variables correlation as determined from axis 1.

	Real data	Randomized data Monte Carlo test, 99 runs			
Axis	Spp-Envt Corr.	Mean	Minimum	Maximum	P
1	0.812	0.746	0.583	0.829	0.140
2	0.603	0.699	0.584	0.845	0.960
3	0.497	0.662	0.513	0.824	1.000

As a result, the relationship in the first CCA test is weaker than made apparent. The analysis illustrates as the second test shows no confidence in the correlation. Therefore, if there is a relationship between ecological and geomorphic indices used, it is weak and shows little confidence.

4.2 Discussion

Many studies have discussed what the field of river/stream restoration requires in order to create sustainable fluvial systems (Palmer et al. 2005; Gillian et al. 2005; Lake 2005; Ryder and Miller 2005). In this study the influence of previous conceptual frameworks was used to create a methodological framework specifically designed for stream restoration and the ability to quantify the success of such projects. The conceptual framework was used as a basis to develop an in-field methodology to evaluate stream restoration with two distinct approaches: ecology and fluvial geomorphology. A basic assessment protocol was formulated and tested in the field, posing the question of whether a monitoring and evaluation model could be devised that effectively assesses the success of a stream restoration project. The ecological and geomorphic approaches were tested separately and together to determine whether one methodology possessed stronger evaluative characteristics than the other. In the process, this study introduced a new rapid geomorphic assessment technique that focused on both channel stability and habitat quality.

In order to test the effectiveness of the assessment approaches used in the study, I assumed that ecological and geomorphic integrity are equally important to achieve a successfully restored stream. This assumption was based on the work of Kondolf et al. (2003), Sullivan et al. (2004; 2006) and Lepori et al. (2005a), and their belief in the importance of this relationship between the two disciplines.

4.21 The Approach

In general, the monitoring protocol devised showed confidence in its ability to assess the condition of a series of restored test reaches as well as the designated disturbed and natural reference reaches. This was an expected result of the study, proving the hypothesis that the creation of a basic quantitative technique for measuring stream restoration success was possible.

This result was described by the two CCA tests used, which showed a Pearson correlation of 0.696 (p-value <0.05) and 0.812 (p-value >0.05). The test demonstrated a weak, but present relationship between the benthic community and environmental variables (geomorphic stability/habitat indicators). The ability to quantify the presence/absence of such a relationship in the study was an important element in the demonstration of the effectiveness of conceptual framework in practice. The importance of ecological and geomorphic relationship has been shown in stream assessment research. Sullivan et al. (2004) described the importance of geomorphic condition and habitat quality on the benthic macro-invertebrate community; however, these two relationships have been rarely put into practice in a restoration context. Those that have approached assessment of restoration projects have focused on an ecological or geomorphic context (Kondolf et al. 2003; Sullivan et al. 2004, 2006; Lepori et al. 2005a). These relationships will be discussed further as it relates to the importance of using a hybrid approach to monitoring.

4.22 A Hybrid Approach

The relationship between aquatic integrity and the corresponding geomorphic stability and habitat quality has been well documented (Hall and Killen 2005; Sullivan et al. 2004, 2006). Therefore, it was expected that relationships would be observed between the benthic community and results from the geomorphic index. However, in this study the relationship between the two approaches to monitoring was present, but limited. The canonical correspondence analysis (CCA) showed a moderate species-environment relationship when the benthic metrics were used in the analysis, demonstrating 0.696, with a p-value <0.05. However, a CCA makes the assumption that there is some form of relationship between the species and environmental variables, by ignoring the community structure that is unrelated to the environmental variables (McCune and Grace 2002). In order, to test the strength of the species-environment relationship in this study and its importance in restoration monitoring, a second CCA test was conducted on the community data, without the metrics. This test demonstrated a stronger relationship between species-environment than the first CCA (0.812), however a p-value of 0.140 was observed. Therefore, the null hypothesis was accepted that there is no relationship between species and environment as shown by the benthic community and geomorphic variables. A p-value of <0.05 was used in the CCA because the intent of the analysis was determine whether any relationship existed between the ecological and geomorphic approach. A smaller p-value (e.g. $p= 0.01$) would have shown increased significance of potential relationships or effect at all (Shinichi and Cuthill 2007). However, it would have ignored weaker relationships that may provide future insight.

In the analyses I used in this study show that the relationship observed between ecological and geomorphic measures are not as strong as expected, or as visible as the literature demonstrated. There will have to be further alterations to the monitoring approach proposed, specifically the geomorphic assessment. The geomorphic approach should be re-tested using appropriate field trials and greater attention in general for the development of a strong quantitative based rapid geomorphic assessment. However, it would be short-sighted to state that the geomorphic portion of the study held no value in the final evaluation of the study and its overall importance as an exercise in stream restoration monitoring. The assessment of the 29 restored reaches in this study showed that none of the reaches were found to be within disturbed or natural reference condition, but rather somewhere on a continuum between these two conditions. This was true for both ecological and geomorphic components. It was expected that a restored test reach found to be natural ecological reference would also be observed to be within natural geomorphic reference; and likewise for a test reach in disturbed reference or a test reach observed in the continuum between disturbed and natural reference. Such relationships have been described between geomorphic and ecologically variables in the past (Roni et al. 2005; in Roni, editor).

Further, the intent of the geomorphic aspect of the study was to test a new methodology that builds upon the current geomorphic and habitat assessments currently used in southern Ontario. This result does not suggest that relationships do not exist between ecological and geomorphic variables in a fluvial environment it states that the specific measures used in the geomorphic approach failed demonstrate them. However, this result questions strength of some relationships that have been described in previous studies, and the methods at which were used to quantify those relationships should be carefully scrutinized.

4.23 Ecological vs. Geomorphic Characteristics

The results showed that the ecological evaluation method was more effective at measuring differences in the quality of restored stream reaches than the geomorphic evaluation method. This was an unexpected result; it was expected that both approaches would perform equally well on their own.

The benthic metrics used to evaluate the ecological condition of the water column in restored streams was shown to be an appropriate measure. The PCA conducted on the metrics showed consistency through most measures when working at a family level of taxonomic identification. The Simpson's Diversity metric may have shown less variance from the other three metrics (Family Richness, % EPT, and % Chironomidae) if identification was at a species level. As metrics will show greater accuracy the lower the taxonomic identification proceeds (Jones et al. 2004). However, a family level identification is considered to be an acceptable level of identification by OBBN (Jones et al. 2004), and were found to show acceptable resolution in this study. The greatest relationship observed was found between % EPT and family richness. % EPT and family richness have been observed to show positive relationships in disturbed and undisturbed streams in the past (Compin and Céréghino 2003). Gage et al. (2004) also observed relationships between family richness and % EPT in high and low disturbance streams; low disturbance streams having higher family richness and % EPT. Similarly, I observed that family richness increased with increasing %EPT, and therefore decrease respectively.

When the metrics were applied to the ecocriteria, the results showed clear distinction between test reaches found to be in either natural or disturbed reference, while placing the remaining restored test reaches on a continuum between. Some of the unexplained variation in ecological results observed may be due to the inability to capture all the environmental influences on the ecological integrity of these reaches; as the streams were geographically situated in intense urban landscape they are subject to various stressors. The presence of increased flows due to impervious surfaces, as well as concentrations of road salt, metals and hydrocarbons known to be in urban stormwater runoff are just a few variables that are difficult to account for (Pitt 1995; Duke et al. 1999; Paul and Meyer 2001; Gresens et al. 2007).

The geomorphic protocol developed for the use of this study was found to be inaccurate and not as effective as the ecological evaluation of the stream reaches tested. Observed stress in non-metric multi-dimensional scaling ordination showed that the scale and/or metrics used in the test were not reliable. Further indication of the unreliability of the geomorphic index, was that it was not able to accurately show distinct differences between Bray-Curtis variation from natural to disturbed reference condition. Most restored test sites showed near zero variation from both disturbed and natural reference reaches; whereas, the ecological approach defined clear variation among test and reference sites. However, in both approaches all test sites remained somewhere between disturbed and natural condition. As previous studies have indicated there are strong relationships between benthic communities and the environment they live in. The presence of stable geomorphic conditions has been found to contain twice as many taxa of macroinvertebrate species (Chessman et al. 2006). Therefore it would have been expected that variation in a reach would appear to be similar; the ability to show such relationships is important.

However, due to the minimal variation observed between natural and reference in the geomorphic approach it suggests that further alterations and trials are required to produce an index that can accurately depict the stability and habitat condition of a stream reach. As one objective of the study was to test a new approach to geomorphic assessment, a failure to produce strong results, further suggests that a strong rapid geomorphic assessment technique for southwestern Ontario is needed. Current geomorphic and habitat assessment practices used in Ontario such as the current Rapid Geomorphic Assessment (RGA) are qualitative or highly time intensive field exercises as conducted by Annable (1996). Rapid habitat assessment (RHA) techniques in Ontario follow a similarly to the geomorphic, and have also been adopted from elsewhere (Plafkin et al. 1989; Ciesielka and Bailey, 2007). This study simply illuminates the fact that further questions need to asked in relation to the ability of successfully quantify the geomorphic state of streams. Continued work on this approach should seek to test other geomorphic indicators (e.g. organic material), as well as address specific issues related measuring rooting depth: bank height ratio in the visible absence of roots on the channel and difference between various root types (i.e. grass and woody vegetation). Also, the re-evaluation of the scale adopted in the geomorphic index to determine whether the index adopted was weak compared to the indicators used within it.

4.3 Implications and Future Uses

Today's use of restoration, specifically in and along watercourses as well as other aquatic environments is growing ever more popular (Choi, 2004), both as a field of research and activity undertaken by various organizations. As activities continue to occur in and around watercourses, specific characteristics and monitoring activities are needed. This will help to ensure that the activities being undertaken are substantiated and most importantly implemented with a rigorous monitoring program that is designed pre-restoration (Palmer et al. 2005).

Through the use of this study, I demonstrated a basic approach to monitoring restored stream reaches that can be easily adopted in various landscapes and corresponding land uses. The protocol and its corresponding conceptual framework should be viewed as a method to evaluate baseline conditions of restored stream reaches. The practitioner should identify key stressors and chose appropriate indicators to reflect that will provide measures that are replicable through time. Once again the importance of realistic goal setting is crucial and indicators should be developed to reflect them (Ehrenfeld 2000; Palmer et al. 2005; Giller 2005; Roni et al. 2005; in Roni, editor). Overall, the practitioner should identify realistic goals for the project if they have not been previously established.

The methodological element of the restoration monitoring protocol described can be applied to numerous fields. The importance of acknowledging the relationship between environmental variables in a system, illustrates the quality of the system that much more vividly;

illuminating stressors that may have otherwise gone unnoticed had not a broader approach been taken. Therefore, causing restoration activities to continually fail, while potentially incurring negative results than the inherent good the project was intended to make. In such cases, perhaps restoration of streams should look towards a more passive approach by allowing the streams to re-instate their own form of heterogeneity (Gillilan et al. 2005; Giller et al. 2005), while only actively removing stressors. Whatever the restoration approach adopted the necessity for monitoring remains; not only for quality assurance, but to enhance restoration as a science by gaining knowledge of ecosystem response to alterations made through restoration.

The use of reference conditions in restoration also contributes to the methodological framework, and overall application of the framework to different fields.

In this, study I used the reference condition approach to determine the relative placement of restored test sites to disturbed and natural reference condition in the region. This study expanded on the reference condition approach by also testing "test" reaches against a disturbed range. Reference condition approach in general compares observed stream condition of test reaches to an expected condition natural reference condition for a region. The practice of the reference condition approach can then categorize where a reach is in relation to reference and determining the "severity of the fail (Bailey et al. 2004)." However, the "severity of the fail" does not necessarily inform the practitioner whether the condition of the restored reach is more severe then what is believed to be pre-restoration conditions. Therefore, simply testing restored reaches against natural reference does not provide the practitioner with sufficient information unless there was significant pre-restoration monitoring occurring on the reach. Limited use of pre-restoration data and extensive monitoring (Bash and Ryan 2002; Alexander and Allan 2007), means that developing a dataset of disturbed conditions for a region allows for a point of reference to begin monitoring on stream reaches that have already been restored and those of which that should be restored but lack sufficient data through time. As Bailey et al. (2004) described, reference condition approach takes a step past traditional assessment and a great deal can be done with the approach. As this study has exemplified, it allows restored reaches to be placed on a continuum between disturbed and natural reference states by describing variation in natural and disturbed communities.

However, it is important to acknowledge that this monitoring protocol does not give an exact position of the restored “test” sites in relation along the continuum between disturbed and natural reference. Without pre-restoration data this would be impossible and without this data quantification of improvements from pre-restoration condition may have been made in the reaches which still appear to be in disturbed reference. However, it is possible to say in relation to other un-restored disturbed streams that makeup the disturbed reference dataset that the restored stream’s management strategy requires re-evaluation in order to improve conditions further. For example, if a test reach was observed to maintain a benthic community and geomorphic characteristics similar to disturbed reaches, a practitioner has reasonable evidence to suggest that re-evaluation of the restoration strategy is required. In the case of a restored reach observed to have characteristics similar to a natural reference condition that the restoration strategy employed has been successful.

Several items should be addressed when adopting the monitoring protocol devised in my study. First, a larger sample size of reaches particularly of reference reaches should be developed into the future, and used to establish a database. Studies should focus primarily on the development of a strong quantitative based rapid geomorphic assessment technique; either building on the technique devised in my study or the creation of a new method. Finally, these studies should be conducted by individuals with a high level of understanding (minimum of graduate student level) for fluvial process and aquatic ecology and be able to incorporate concepts of ecological restoration.

4.4 Conclusions

From this study, I propose that through the appropriate use of ecological and geomorphic assessment as depicted in the conceptual framework, a stream restoration monitoring protocol can be established and successfully practiced in the field. The underlying concepts of the framework show that through the use, and adaptation of the reference condition approach that the relative placement of restored test reaches in a disturbed to natural reference condition continuum can be accomplished. This is an important element when considering future management of restored streams, and determining the trajectory that managers should take.

To date in southwestern Ontario streams the ecological measures used in this and other studies showed the greatest consistency in evaluating restored stream condition. Although the geomorphic technique and its corresponding index were observed to be weak it is recommended that a hybrid approach to stream restoration monitoring be used. This will allow for capture of the relationships between aquatic ecological and geomorphic conditions and their affects on overall stream health that may otherwise be missed with a pure ecological assessment. Therefore, I recommend that further studies test the effectiveness of geomorphic assessment techniques in southwestern Ontario. These trials should be conducted outside of the restoration context so it may be used as a more effective tool in the future.

References

Abel, P.D.1989. Water pollution biology. Ellis Horwood Ltd., Chichester, England. P. 231.

Agricultural and Agri-Food Canada. 1998. The Canadian System of Soil Classification, 3rd ed. Agriculture and Agri-Food Canada Publication 1646.

Alexander, G.G., and D.J. Allen. 2006. Stream restoration in the upper Midwest, U.S.A. *Restoration Ecology* **14**: 595-604.

Allen E.B., W.W. Covington, and D.A. Falk. 1997. Developing a conceptual basis for restoration ecology. *Restoration Ecology* **5**: 275-276.

Allison, S.K. 2007. You can't not choose: embracing the role of choice in ecological restoration. *Restoration Ecology* **15**: 601-605.

Anand, M., and R.E. Desrochers. 2004. Quantification of restoration success using complex systems concepts and models. *Restoration Ecology* **12**: 117-124.

Angradi, T. R.. E. W. Schweiger, D.W. Bolgrien, P. Ismert, and T. Selle. 2004. Bank stabilization, riparian land use and distribution of large woody debris in a regulated reach of the Upper Missouri River, North Dakota, USA. River Res. Applic. **20**: 829–846.

Annable, W.K. 1996. Database of morphological characteristics of watercourses in Southern Ontario. Ontario Ministry of Natural Resources ISBN # 0-7778-5112-1.

Arkansas Department of Environmental Quality. 2004. Using a Bank Erosion Hazard Index to Estimate Annual Sediment Loads From Streambank Erosion in the Loads in the West Fork White River Watershed.
<http://www.awag.org/2004%20Conference/Track%203/ADEQ_BEHI_english_ASAE.pdf.> Accessed February 15, 2008.

Bailey, R.C., R.H. Norris, and T.B. Reynoldson. 2004. Bioassessment of freshwater ecosystems: Using the reference condition approach. Kluwer Academic Publishers.

Bailey, R.C., M.G. Kennedy, M.Z. Dervish and R.M. Taylor. 1998. Biological assessment of freshwater ecosystems using a reference condition approach: Comparing predicated and actual benthic invertebrate communities in Yukon streams. *Freshwater Biology* **39**: 765-774.

Bash J.S., and C.M. Ryan. 2002. Stream restoration and enhancement projects: Is anyone monitoring? *Environmental Management* **29**: 877-885.

Bauer, S.B., and S.C. Ralph. 1999. Aquatic habitat indicators and their application to water quality objectives within the Clean Water Act. U.S. Environmental Protection Agency, EPA 910-R-99-014.

Bernhardt, E.S., E.B. Sudduth, M.A. Palmer, J.D. Allan, J.L. Meyer, G. Alexander, J. Follastad-Shah, B. Hassett, R. Jenkinson, R. Lave, J. Rumps, and L. Pagano. 2007. Restoring rivers one reach at a time: Results from a survey of U.S. river restoration practitioners. *Restoration Ecology* **15**: 482-493.

Bledsoe B.P., and C.C. Watson. 2001. Effects of urbanization on channel instability. American Water Resources Association **37**: 255-270.

Bledsoe B.P., M.C. Brown and D. A. Raff. 2007. *American Water Resources Association* **43**: 757-772.

Boon, P.J. 1998. River restoration in five dimensions. *Aquatic Conservation: Marine and Freshwater Ecosystems* **8**: 257-264.

Booth, D.B. 1990. Stream-channel incision following drainage-basin urbanization. Water Resources Bulletin **26**: 407-417.

Bornette, G., C. Amoros, H. Piegay, J. Tachet, and T. Hein. 1998. Ecological complexity of wetlands within a river landscape. *Biological Conservation* **85**: 35-45.

Bradshaw, A.D. 1987. Restoration: the acid test for ecology. In: Restoration Ecology: A Synthetic Approach to Ecological Research (eds Jordan, W.R., Gilpin, M.E. & Aber, J.D.). Cambridge University Press, Cambridge, UK, pp. 23–29.

Bradshaw, A.D. 1996. Underlying principles of restoration. *Fisheries and Aquatic Sciences* **53**: 3-9.

Brookes, A. 1988. Channelized rivers: Perspectives for environmental management. Wiley, Chichester.

Bross, I.D.J. 1971. Critical Levels, Statistical Language and Scientific Inference, in Godambe and Sprott (eds) *Foundations of Statistical Inference*. Toronto: Holt, Rinehart & Winston of Canada, Ltd.

Buckley, M., and B.M. Haddad. 2006. Socially strategic ecological restoration: a game-theoretic analysis shortened: socially strategic restoration. *Environmental Management* **38**: 48-61.

Chapman, R.L. 2006. Ecological restoration restored. *Environmental Values* **15**: 463-478.

Chapman, L.J. and Putnam, D.F.1984. The physiography of southern Ontario. 3rd ed. Toronto: Ontario Ministry of Natural Resources. (Ontario Geological Survey. Special volume 2).

Chessman, B.C., K.A. Fryirs, and G.J. Brierley. 2006. Linking geomorphic character, behaviour and condition to fluvial biodiversity: implications for river management. *Aquatic Conservation: Marine and Freshwater Environments* **16**: 267–288.

Choi, Y.D. 2004. Theories for ecological restoration in changing environment: toward 'futuristic' restoration. *Ecological Research* **19**: 75-81.

Ciesielka, I.K., and R.C. Bailey. 2007. Hierarchical structure of stream ecosystems: consequences for bioassessment. *Hydrobiologia* **586**: 57–67.

City of Waterloo. 2008. Statistical Profile. <http://www.city.waterloo.on.ca/DesktopDefault.aspx?tabid=1454> Accessed March 14, 2008.

Clarke, K.R. 1993. Non-parametric multivariate analyes of changes in community structure. *Australian Journal of Ecology* **18**: 117-143.

Compin, A., and R. Céréghino. 2003. Sensitivity of aquatic insect species richness to disturbance in the Adour–Garonne stream system (France). *Ecological Indicators* **2**: 135-142.

Credit Valley Conservation Authority. 2002. East Credit sub-watershed study -Background report. <http://www.creditvalleycons.com/bulletin/downloads/FINALBRSub13.pdf>. Accessed March 3, 2008.

Dodkin, I., B. Rippey and P. Hale. 2005. An application of canonical correspondence analysis for developing ecological quality assessment metrics for river macrophytes. *Freshwater Biology* **50**: 891–904.

Doyle, M.W. 2006. A heuristic model for potential geomorphic influences on trophic interactions in streams. *Geomorphology* **77**: 235-248.

Duke, D. L., T. S. Lo and M. W. Turner, 1999. Chemical constituents in storm flow vs. dry weather discharges in California storm water conveyances. *Journal American Water Resources Association* **35**: 821–836.

Ehrenfeld, J.G. 2000. Defining the limits of restoration: the need for realistic goals. *Restoration Ecology* **8**: 2-9.

Ehrenfeld, J.G., and L.A. Toth. 1997. Restoration ecology and the ecosystem perspective. *Restoration Ecology* **5**: 307-317.

Food and Agriculture Organization of the United Nations. 1998. World reference base for soil resources. Food and Agriculture Organization of the United Nations, International Society for Soil Science, International Soil Reference and Information Centre, Rome.

Gage, M.S., A. Spivak, and C.J. Paradise. 2004. Effects of Land Use and Disturbance on Benthic Insects in Headwater Streams Draining Small Watersheds North of Charlotte, NC. *Southeastern Naturalist* **3**: 345-358.

Giller, P.S. 2005. River restoration: seeking ecological standards. Editor's introduction. *Applied Ecology* **42**: 201-207.

Gillilan, S., K. Boyd, T. Hoitsma, and M. Kauffman. 2005. Challenges in developing and implementing ecological standards for geomorphic river restoration projects: a practitioner's response to Palmer et al. (2005). *Applied Ecology* **42**: 223-227.

Grand River Conservation Authority (2008). Grand River Watershed Map. <http://www.grandriver.ca/index/document.cfm?Sec=12&Sub1=55&sub2=24.>Accessed February 15, 2008.

Grayson, J. E., M. G. Chapman and A. J. Underwood. 1999. The assessment of restoration of habitat in urban wetlands. *Landscape and Urban Planning* **43**: 227–236.

Gregory, S.V., F.J. Swanson, W.A. McKee, and D.W. Cummins. 1991. An ecosystem perspective of riparian zones: Focus on links between land and water. *Bioscience* **41**: 540-551.

Gresens S.E., K.T. Belt, J.A. Tang, D.C. Gwinn, and P.A. Banks. 2007. Temporal and spatial responses of Chironomidae (Diptera) and other benthic invertebrates to urban stormwater runoff. *Hydrobologia* **575**: 173-190.

Hall, L.W. and W.D. Killen. 2005. Temporal and spatial assessment of water quality, physical habitat, and benthic communities in an impaired agricultural stream in California's San Joaquin Valley. *Environmental Science and Health Part A - Toxic/Hazardous Substances & Environmental Engineering* **40**: 959-989.

Hackney, C. T., 2000. Restoration of coastal habitats: expectation and reality. *Ecological Engineering* **15**: 165–170.

Hawkes, H.A. 1979. Invertebrates as indicators of river water quality. In biological Indicators of Water Quality. Eds. A. James and L. Evison, Chap. 2 John Wiley Chichester, England.

Hobbs, R.J. 2003. Ecological management and restoration: Assessment, setting goals and measuring success. *Ecological Management and Restoration* **4**: S2-S3.

Hobbs, R.J., and J.A. Harris. 2001. Restoration Ecology: Repairing the Earth's Ecosystems in the New Millennium. *Restoration Ecology* **9**: 239-246.

Holl, K.D., E.E. Crone, C.B. Shultz. 2003. Landscape restoration: Moving from generalities to methodologies. *BioScience* **53**: 491-502.

Jones, C., K.M. Somers, B. Craig, and T.B. Reynoldson. 2004. Ontario Benthos Biomonitoring Network Protocol Manual. V 1.0. Ontario Benthos Biomonitoring Network.

Jones, K., and E. Hanna. 2004. Design and implementation of an ecological engineering approach to coastal restoration at Loyola Beach, Kleberg County, Texas. *Ecological Engineering* **22**: 249-261.

Kaufmann, P.R., P. Levine, E.G. Robison, C. Seeliger, and D.V. Peck 1999. Quantifying physical habitat in wadeable streams. U.S. Environmental Protection Agency, EPA/620/R-99/003, Corvallis, Oregon.

Kay, J.J., 1993, "On the Nature of Ecological Integrity: Some Closing Comments" in S. Woodley, J. Kay, G. Francis (Eds.), 1993. *Ecological Integrity and the Management of Ecosystems,* St. Lucie Press, Delray, Florida. pp.210-212.

Kondolf, G.M. 1995. Geomorphological stream channel classification in aquatic habitat restoration – uses and limitations. *Aquatic Conservation - Marine and Freshwater Ecosystems* **5**: 127-141.

Kondolf, G.M., H. Piegay and D. Sear. 2003. Integrating Geomorphological Tools in Ecological and Management Studies. In Kondolf, G.M. and H. Piegay. Editors. 2003. Tools in fluvial geomorphology. Wiley; West Sussex, England.

Kondolf, G.M. and H. Piegay. Editors. 2003. Tools in fluvial geomorphology. Wiley; West Sussex, England.

Kondolf, G.M. 2006. River restoration and meanders. *Ecology and Society* **11**: 42-59.

Lake, P. S., 2005. Perturbation, restoration and seeking ecological sustainability in Australian flowing waters. *Hydrobiologia* **552**: 109–120.

Lake, P.S., N. Bond and P. Reich. 2007. Linking ecological theory with stream restoration. *Freshwater Biology* **52**: 597-615.

Lenat, D. R., and J. K. Crawford. 1994. Effects of land use on water quality and aquatic biota of three North Carolina Piedmont streams. *Hydrobiologia* **294**: 185–199.

Lepori, F., D. Palm, and B. Malmqvist. 2005a. Effects of stream restoration on ecosystem functioning: detritus retentiveness and decomposition. *Applied Ecology* **42**: 228-238.

Lepori, F., D. Palm, E. Brännäs, and B. Malmqvist. 2005b. Does restoration of structural heterogeneity in streams enhance fish and macroinvertebrate diversity? *Ecological Applications* **15**: 2060-2071.

MacDonald, L.H., A.W. Smart, and R.C. Wismer. 1991. Monitoring guidelines to evaluate effects of forestry activities on streams in the Pacific Northwest and Alaska. U.S. Environmental Protection Agency, Region 10, EPA/910/9-91/001, Seattle.

McCune, B., and J. B. Grace. 2002. Analysis of ecological communities. MjM Software Design, Oregon.

Michener, W.K. 1997. Quantitatively evaluating restoration experiments: research, design statistical analysis, and data management considerations. *Restoration Ecology* **5**: 324-337.

Naveh, Z. 2005. Epilogue: Toward a transdisciplinary science of ecological and cultural landscape restoration. *Restoration Ecology* **13**: 228-234.

Newbury Hydraulics. 2002. A stream network inventory, fluvial geomorphic assessment, impact assessment and natural channel design for Red Hill Creek, Volume I and II prepared for City of Hamilton. <https://www.myhamilton.ca/NR/rdonlyres/2B3C92B5-0088-4040-8928-0D557DD1208B/0/rhvp0304.pdf.> Accessed March 3, 2008.

Newson, M.D., and D. Sear. 1998. The role of geomorphology in monitoring and managing river sediment systems. *Chartered Institution of Water and Environmental Management* **12**: 18-24.

O’Donnell, T.K., and D.L. Galat. 2008. Evaluating success criteria and project monitoring in river enhancement within an adaptive management framework. *Environmental Management* **41**: 90–105.

Odum, E. P. 1969. The strategy of ecosystem development. *Science* **164**: 262-270.

Ormerod, S.J. 2003. Restoration in applied ecology: editor’s introduction. *Applied Ecology* **40**: 44-50.

Palmer, M.A., R.F. Ambrose, N. LeRoy Poff. 1997. Ecological theory and community restoration ecology. *Restoration Ecology* **5**: 291-300.

Palmer, M.A., E.S. Bernhardt, J.D. Allan, P.S. Lake, G. Alexander, S. Brooks, J. Carr, S. Clayton, C.N. Dahm, J. Follstad Shah, D.L. Galat, S.G. Loss, P. Goodwin, D.D. Hart, B. Hassett, R. Jenkinson, G.M. Kondolf, R. Lave, J.L. Meyer, T.K. O’Donnell, L. Pagano, and E. Sudduth. 2005. Standards for ecologically successful river restoration. *Applied Ecology* **42**: 208-217.

Paul, M. J. and J. L. Meyer, 2001. Streams in the urban landscape. *Annual Review of Ecology and Systematics* **32**: 333–365.

Pedroli, B., G. de Blust, K. van Looy and S. van Rooij. 2002. Setting targets in strategies for river restoration. *Landscape Ecology* **17**: 5-18.

Pitt, R. E. 1995. Biological effects of urban runoff discharges. In E. E. Herricks, (ed.), Stormwater Runoff and Receiving Systems: Impact, Monitoring and Assessment. Lewis Publ. Boca Raton, Florida, 127–162.

Plafkin, J. L., M. T. Barbour, K. D. Porter, S. K. Gross, and R. M. Hughes. 1989. Rapid bioassessment protocols for use in stream and rivers: Benthic macroinvertebrates and fish. US Environmental Protection Agency, Washington, DC.

Pollock, M.M., T.J. Beechie, S.S. Chan, and R. Bigley. 2005. Monitoring restoration of riparian forests. Pages 67-96 *in* P. Roni, editor. Monitoring stream and watershed restoration. American Fisheries Society, Bethesda, Maryland.

Rios, S.L., and R.C. Bailey. 2006. Relationship between riparian vegetation and stream benthic communities at three spatial scales. *Hydrobiologia* **553**: 153-160.

Roni, P., editor. 2005. Monitoring stream and watershed restoration. American Fisheries Society, Bethesda, Maryland.

Roni, P., M.C. Liermann, C. Jordan, and E.A. Steel. 2005. Steps for designing a monitoring and evaluation program. Page 13-34 *in* P. Roni, editor. Monitoring stream and watershed restoration. American Fisheries Society, Bethesda, Maryland.

Rosgen, D.L. 2001. A practical method for computing streambank erosion rate Pagosa Springs, Colorado: Wildland Hydrology. <http://www.wildlandhydrology.com/assets/Stream**bank_erosion_**paper.pdf.> Accessed March 10, 2007.

Rosgen, D. L. 1996. Applied river morphology. Pagosa Springs, Colorado: Wildland Hydrology.

Rosgen, D.L. 1994. A classification of natural rivers. *Catena* **22**: 169-199.

Ryder, D.S. and W. Miller. 2005. Setting goals and measuring success: linking patterns and processes in stream restoration. *Hydrobiologia* **552**: 147-158.

Saintilan, N. 2004. Relationships between estuarine geomorphology, wetland extent and fish landings in New South Wales estuaries. *Estuarine Coastal and Shelf Science* **61**: 591-601.

Schwartz, J.S., and E.E. Herrick. 2007. Evaluation of pool-riffle naturalization structures on habitat complexity and the fish community in an urban Illinois stream. *River Res. Applic.* **23**: 451-466.

Shinichi, N., and I.C. Cuthill. 2007. Effect size, confidence interval and statistical significance: a practical guide for biologists. *Bio. Rev*. **82**: 591-605.

Society for Ecological Restoration Science and Policy Working Group. 2002. The SER primer on ecologicalal restoration. Retrieved January 18, 2007, from http://www.ser.org/.

Suding, K.N., K.L. Gross, and G.R. Houseman. 2004. Alternative states and positive feedback in restoration ecology. *TRENDS in Ecology and Evolution* **19**: 46-53.

Sullivan, S.M.P., M.C. Watzin, and W.C. Hession. 2004. Understanding stream geomorphic state in relation to ecological integrity: evidence using habitat assessments and macroinvertebrates. *Environmental Management* **34**: 669-683**.**

Sullivan, S.M.P., M.C. Watzin, and W.C. Hession. 2006. Influence of stream geomorphic condition on fish communities in Vermont, U.S.A. *Freshwater Biology* **51**: 1811–1826.

Temperton, V.M. 2007. The recent double paradigm shift in restoration ecology. *Restoration Ecology* **15**: 344-347.

ter Braak, C. J. F. and Verdonschot, P. F. M. 1995. Canonical correspondence analysis and related multivariate methods in aquatic ecology. *Aquatic Sciences* **57**: 255-289.

Turner, M. G. and R. H. Gardner, 1991. Quantitative methods in landscape ecology. Springer-Verlag, New York, 552 pp.

University of Waterloo. 1997. WATgreen: Laurel Creek Symposium Workshop. <http://www.soer.uwaterloo.ca/pdfs/LaurelCreek.pdf.> Accessed March 14, 2008.

University of Waterloo. 2007. University of Waterloo Weather Station Annual Summary. <http://weather.uwaterloo.ca/download/historical/2007/2007_annual.pdf> Accessed March 14, 2008.

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

Vermont Agency of Natural Resources. 2004a. Vermont Stream Geomorphic Assessment. <http://www.anr.state.vt.us/dec//waterq/rivers/docs/assessmenthandbooks/rv_apxnsoilstratbehi.pdf.> Accessed February 15, 2008.

Vermont Agency of Natural Resources. 2004b. Particle entrainment and transport. <http://www.anr.state.vt.us/dec//waterq/rivers/docs/assessmenthandbooks/rv_apxoshearstress.pdf. > Accessed December 07, 2007.

"Watershed Map" [map]. 1:20,000. GRCA GIS Data [computer files]. Cambridge, Ont.: Grand River Conservation Authority, 2008 and City of Waterloo, 2001. Using ArcView GIS [GIS Software]. Version 3.2. Redlands, CA: Environmental Research Institute, Inc., 1992-1993.

Wiederhom, T. 1980. Use of benthos in lake monitoring. *Journal of the Water Pollution Control Federation* **52**: 443-447.

Wilkins, S., D. A. Keith and P. Adam, 2003. Measuring success: evaluating the restoration of a grassy eucalypt woodland on the Cumberland Plain, Sydney, Australia. *Restoration Ecology* **11**: 489–503.

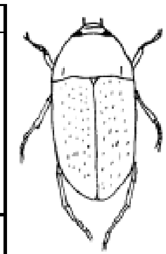
Woolsey, S., F. Capelli, T. Gonser, E. Hoehn, M. Hostmann, B. Junker, A. Paetzold, C. Roulier, S. Schweizer, S.D. Tiegs, K. Tockner, C. Weber and A. Peter. 2007. A strategy to assess river restoration success. *Freshwater Biology* **52**: 752–769.

Young, T.P., D.A. Peterson, and J.J. Clary. 2005. The ecology of restoration: historical links, emerging issues and unexplored realms. *Ecology Letters* **8**: 662-673.

Appendix A – Ecological Sampling Methods

Figure 6. Benthic sampling field sheet (OBBN 2004).

Ontario Benthos Biomonitoring Network Field Sheet: STREAMS



Date:	Stream name:
Time	Site #:
Agency:	Location (Sampling reach centroid, use deg./min./sec. or specify other)
Investigators:	Latitude: Elevation (m asl):
Water Quality	Longitude:
Water Temperature (°C):	Conductivity (uS/cm): pH:
DO (mg/l):	Alkalinity (mg/l as $CaCO_3$):

Site Description and Map

Draw a map of the site (with landmarks) and indicate areas sampled. Attach photograph (optional)
Show north arrow.

Benthos Collection Method (circle one):	**Gear Type** (circle one)
• Traveling Kick & Sweep • Grab Sample	• D-net • Ponar • Other (*specify*):
• Other (specify):	• Ekman • Rock Baskets
	Mesh Size: 500 micron (or specify)

Sub-samples	Sampling distance covered (m)	Time (min.)	Max. Depth (m)	Wetted Width (m)	Max. Hydraulic Head (mm)	# Grabs pooled per sample
Sample 1: Riffle (cross-over)	KICK AND SWEEP ONLY					GRAB SAMPLING ONLY
Sample 2: Pool						
Sample 3: Riffle (cross-over)						

Appendix B – Geomorphic Sampling Methods

Figure 7. Geomorphic assessment field template used in Spring 2007 assessments.

YATES GEOMORPHIC ASSESSMENT

Site Name:		Date:		Time:		UTM:	
	Cross Sections						
	Riffle			Pool			Riffle
Transect	1	2	3	4	5	6	7
Bank Angle							
Height to Top of Bank							
Height to Bank Full							
Rooting Depth							
Velocity TW							
Pt.							
Pt.							
Substrate type							

Table 21. Sieves and corresponding sizes used to determine substrate heterogeniety.

Sediment Type	Sieve Size (mm)						
Gravel	38.1	25.4	15.87	9.52	6.35	4.76	2.00
Sand	0.59	0.30	0.26	0.12			
Silt	0.064	0.015					

Table 22. Limiting shear stress and velocity for non-cohesive sediments (Vermont Agency of Natural Resources 2004b)

	D(mm)	Vc (m/s) for shear
Boulders		
Very Large	2032	1.328928
Large	1016	0.938784
Medium	508	0.67056
Small	254	0.469392
Cobble		
Large	127	0.329184
Small	63.5	0.2286
Gravels		
Very Coarse	33.02	0.158496
Coarse	15.24	0.109728
Medium	7.62	0.073152
Fine	4.064	0.051816
Very Fine	2.032	0.036576
Sands		
Very Coarse	1.016	0.021336
Coarse	0.508	0.016764
Medium	0.254	0.013716
Fine	0.127	0.012192
Very Fine	0.0762	0.010668
Silts		
Coarse	0.0508	0.009144
Medium	0.0254	0.00762

Appendix C – Reach Pictures

Figure 8. Clair Creek restored site photograph of two riffle segments.

Figure 9. Colonial Creek restored reaches.

Figure 10. Critter Creek Upstream restored riffle and pool segment.

Figure 11. Critter Creek downstream restored reaches.

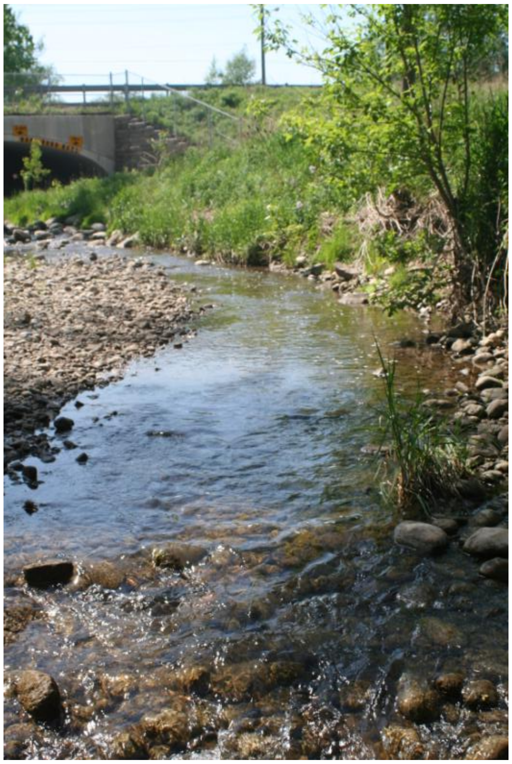


Figure 10. Forwell Creek restored riffle and pool segments.

Figure 11. Laurel Creek restored reach.

Figure 12. Natural reference reaches in Upper Laurel Creek.

Figure 13. Disturbed reference reaches on N. Colonial Creek and S. Clair Creek.