Assessing the impact of riprap bank stabilization on fish habitat:

A study of Lowland and Appalachian streams in Southern Québec

William Massey

A thesis

in the Department

of

Geography, Planning and Environment

Presented in Partial Fulfillment of the Requirements

For the Degree of Masters of Science

(Geography, Urban and Environmental Studies) at

Concordia University

Montreal, Quebec, Canada

April 2017

CONCORDIA UNIVERSITY

School of Graduate Studies

This is to certify that the thesis prepared

By: William Massey

Entitled: Assessing the impact of riprap bank stabilization on fish habitat: A study of Lowland and Appalachian streams in Southern Québec

and submitted in partial fulfillment of the requirements for the degree of

Master of Science (Geography, Urban and Environmental Studies)

complies with the regulations of the University and meets the accepted standards with respect to originality and quality.

Signed by the final Examining Committee:

Dr. Monica Mulrennan	Chair
Dr. Jan Franssen	External Examiner
Dr. Michel Lapointe	Examiner
Dr. Pascale Biron	Supervisor

Approved by ______________________________

Chair of Department or Graduate Program Director

__________ 2017 ______________________________

Date Dean of Faculty

ABSTRACT

Assessing the impact of riprap bank stabilization on fish habitat: a study of Lowland and Appalachian streams in Southern Québec

William Massey

There is a growing concern over the potential environmental impacts of riverbank stabilization using rock riprap as the occurrence of these structures continues to increase in river networks. Habitat diversity and quality are often used as a proxy for fish community health. Habitat assessments, however, frequently yield contrasting results between studies and it remains unclear how non-salmonid species in small streams may be affected by bank stabilization. The aim of this thesis was to evaluate how riprap structures impact fish habitat in small Lowland and Appalachian streams by combining quantitative and qualitative approaches. Metrics measured were: mesohabitat and in-stream cover proportions, Hydro-Morphological Index of Diversity (HMID), and a modified Qualitative Habitat Evaluation Index (QHEI). Results show that in more pristine Appalachian streams, QHEI scores are lower at stabilized reaches due to loss of in-stream cover and riparian vegetation. However, riprap stabilization had less impact on already altered, straightened Lowland streams. In this latter context, some possibly beneficial alterations of fish habitat were observed in riprapped reaches due to the coarsening of the substrate and an induced increase of slope. These positive effects are, however, limited to short stabilized reaches, and extensive (> 100 m) riprapping of the bed should be avoided as it can result in the drying of the bed during summer months, as was observed in this study in some tributaries of the Salvail River. Both metrics (HMID and QHEI) revealed the positive or neutral effect of riprap on increasing flow diversity and heterogeneity for Lowlands sites with a correlation

of 0.72 ($p < 0.01$). However their effect scores are inconsistent in the Appalachian streams as only QHEI showed a negative effect of riprap, suggesting caution when interpreting habitat quality results based on a single metric.

Acknowledgements

I would like to express my gratitude to my supervisor Dr. Pascale Biron, whose continuous support, patience and guidance has made the learning process of this thesis a positive one. I would also like to thank Guénolé Choné for always being there to help with his seemingly limitless knowledge and problem solving skills. I am forever grateful to Maxim Bulat, Véronique Parent and Bertrand Ehly who worked so hard in the field with me to get those measurements. Also, for their expertise in fish sampling and identification, I would like to thank Gabriel Lanthier, Ariane Bouffard and Sarah Chabot. Thanks to NSERC and FRQNT for the financial support which allowed me to focus more on this work. The constructive comments of four anonymous reviewers on a previous version of the chapter 3 manuscript are greatly appreciated. To my family and friends who never gave up on me, and to Elliott Vezina who taught me to never give up on myself, I thank you.

Contribution of Authors

Chapter 3: Co-Authors: Pascale M. Biron and Guénolé Choné

Table of Contents

List of Figures

List of Tables

Chapter 1. Introduction

1.1 Fish habitat in Southern Québec: a cause for concern

Natural river ecosystems provide a variety of habitats which in turn may support diverse fish communities (Breschta and Platts, 1986). However, deterioration of physical habitat and water quality is noted in most densely populated areas, with growing concern for the impact on fish (Comité de concertation et de valorisation du basin de la Rivière Richelieu [COVABAR], 2013b). Although current Canadian legislation does offer some protection for fish habitat through section 35 of The Fisheries Act, it has been severely weakened by the passing of Omnibus Bill C-38 under the Conservative government. The previous prohibition against harmful alteration, disruption or destruction of fish habitat (HADD) has now been changed to apply only to habitats that support commercial, recreational or Aboriginal fisheries in cases where permanent alterations may result in "serious harm to fish" (Government of Canada Justice Laws Website [GC], January 30, 2014). Despite these legislative changes at the federal level, there remains concern about fish habitat from the various ministries involved in river interventions at the provincial level. For example, riprap bank stabilization is often used in infrastructure projects realized by the Ministry of Transport of Québec, who would like to mitigate any potential negative impacts for fish habitat.

Riprap bank stabilization is the most commonly used erosional control measure in Canada and the United States, being applied as a default for protecting nearby road works and bridge pier infrastructures (U.S. Department of Transportation Federal Highway Administration [US DoT], 1989). The sheer volume of riprapped banks can be illustrated in regions such as Montérégie (Québec) which possesses an extensive network of small streams intersecting roads. While these riprap interventions can often be small relative to the overall stream length, their increasing application in Québec streams is cause for concern regarding any cumulative environmental impacts. Furthermore, many riprap structures become destabilized over time, necessitating frequent and costly maintenance (Federal Emergency Management Agency [FEMA], 2009). As a large number of streams in Québec are already highly disturbed due to agricultural practices which tend to increase fine sediment loads and remove riparian vegetation (Berryman, 2008; Simoneau and Thibault, 2009), these locations may be more sensitive to disturbances related to riprap construction and/or maintenance.

Despite its widespread application in river systems, riprap is still predominantly an engineering concern with little effort made to collect biological data or information regarding habitat conditions (Fishenich, 2003). This situation is highlighted in several literature reviews which indicate that most studies regarding riprap impacts on the lotic environment are done post riprap treatment while before and after comparisons remain extremely rare (Craig and Zale, 2001; Schmetterling et al., 2001; Reid and Church, 2015). Furthermore, a large variability in experimental results and observations have been reported regarding the impacts of riprapping on fish and fish habitats. Due to these inconsistencies and highly empirical findings, a complete understanding of how riprap influences fluvial processes over time, including potential impacts on physical fish habitat and other ecological implications, has yet to be achieved. Clearly the gaps in knowledge regarding riprap limit the possibility for environmental managers to make informed decisions.

1.2 Research objectives

In order to better inform decision makers as to how the current design and application of riprap bank stabilization that is used in Southern Québec may be affecting fish wildlife, it is important to first understand potential impacts on fish habitat. Indeed, it is a sound ecological principle that the diversity of habitats in an area is positively related to the diversity of biota, theoretically allowing predictions to be made from one metric to another (Newson and Newson, 2000). Therefore, the quantification and qualification of the physical environment of fish habitat at riprap stabilized sites is an essential step in understanding potential impacts on fish communities, one which requires comparatively less resources. The main objective of this study is thus to examine how riprap structures affects fish habitat in small streams. Fish habitat is evaluated through the combination of quantitative and qualitative methods, an approach that is expected to improve the accuracy of overall habitat assessment (Fernandez et al., 2011).

In chapter 3 a paired comparison experimental design of riprapped vs. non riprapped reaches is presented to answer research questions such as: (A) do localized small extents of riprap (40 meters on average) such as those found at bridge/stream crossings provide the same quantity and quality of fish habitat as reference reaches?; (B) does the physiographic setting (Appalachian vs. Lowland) affect habitat assessment usage and interpretation of results?; (C) what are the most important explanatory variables for understanding differences in fish habitat metrics between stabilized and non-stabilized sites?; and (D) to what extent do different fish habitat assessment protocols differ in their robustness (ability to consistently describe fish habitat both temporally and spatially)?

In chapter 4 a Before After Control Impact (BACI) experimental design is used to evaluate riprap impacts on both fish and fish habitat in a Lowland river to answer the following research questions: (1) how do fish and fish habitat metrics of quality and diversity respond to riprap treatment?; and (2) do differences in fish and fish habitat diversity agree with each other?

In most circumstances the use of riprapping is expected to alter the fluvial environment as well as the processes involved that create dynamic fish habitats in ways that may be detrimental for overall fish habitat quality. From an environmental management stand point, the continued use of riprap as a general erosional control measure must be weighed against other potentially negative impacts in order to meet society needs while minimizing ecological harm. The results from this study will therefore be used to inform management decisions regarding future riprap designs by providing quantitative and qualitative information, and explaining some of the variability concerning impacts on fish habitat across a range of fluvial environments.

Chapter 2. Literature Review

The following literature review will demonstrate the pertinence of continuing to establish connections between static riprap structures and their impacts on dynamic fish habitats, despite the extreme diversity of lotic environments. It will be shown that previous studies have reported variable results leading to uncertainty in the scientific community as to whether or not riprap poses a significant threat to the health of fish communities. The pros and cons of different types of fish habitat assessments will also be addressed, highlighting the tradeoff between resources available and accuracy of results.

2.1. Riprap bank stabilization

2.1.1. An environmental management dilemma

Bank stabilization is a general term used to describe the structural modification of the bank/banks or bed of a watercourse channel, which is either presently or is predicted to pose a problem for society or the natural environment (Fischenich, 2003). This need for human intervention usually stems from situations where the force of water flow is causing excessive erosion, potentially damaging riparian zones, water quality, private property or public infrastructures such as roads and bridges. Depending on the scale, nature of the problem and the characteristics of the watercourse, different stabilization structures may be employed, ranging from less invasive deflection techniques to heavier modifications such as armoring/revetment (Fischenich, 2003). For example, armoring techniques may be used at large scales such as in the protection of highway embankments which often run parallel to rivers or to reduce erosion of contaminated soils into waterways through the stabilization of kilometers of riverbank (Price and Birge, 2005; Gidley et al., 2012). Other smaller scale applications of armoring include the stabilization of bridge piers, culverts and surrounding banks (Bouska et al., 2010). These stabilization structures can be constructed using various materials such as vegetation, gigantic concrete slabs or rock riprap, each with their own list of pros and cons from engineering and environmental conservation perspectives. Given such a wide range of options, determining the best structure and material to use for a given stabilization project can prove to be a difficult management decision when faced with the task of balancing societal and environmental requirements.

The utilization of vegetation as an alternative to anthropogenic bank stabilization materials (soft versus hard structures respectively) is often argued as having minimal, if not positive impacts on the environment while fulfilling the same basic function of erosional control. Indeed river management guidelines published by drainage basin organizations often promote the maintenance of healthy riparian zones and the rehabilitation of native vegetation to areas experiencing erosion since root systems naturally stabilize the soil and do not perturb normal ecosystem functions (Organisme de bassin versant de la baie Missisquoi [OBVBM], 2010). For example, the OBV Richelieu/Saint-Laurent COVABAR has documented several recent stabilization projects using native vegetation to solve local erosion problems at two small streams in the Saint-Jean-Baptiste area ([COVABAR], 2013a). Other than the aesthetic and ecosystems benefits of planting native vegetation as a form of bank stabilization, this process is thought to be less costly and require less maintenance than other forms of stabilization such as rock riprap (OBVBM, 2010). The stabilizing effect of bank vegetation on the geometry of a channel has also been well quantified (Millar, 2000). Despite this, rock riprap is still the most commonly employed bank stabilization structure and has become a default erosional control measure (U.S. Department of Transportation Federal Highway Administration [US DoT], 1989).

Rock riprap is essentially graded stone (rocks of variable sizes) of angular shape that can be lined along the banks or bed of a channel as a form of hard structure stabilization (Jafarnejad et al. 2017). Generally the rocks are loosely placed either by hand or heavy machinery, although they can also be placed in mesh wire cages known as gabions in situations where the slope of the bank is too steep or the velocity of the water is too great to permit a less stable structure (US DoT, 1989). The popularity of the use of riprap in stabilization projects is in part due to its positive portrayal in many engineering guidebooks. For example, from a structural and economic perspective, riprap is considered to be a more internally flexible and cheaper alternative relative to other bank armoring techniques such as concrete lining. Furthermore, riprap is highlighted as being easily maintained by adding rocks to damaged areas and the overall construction process is extremely low-tech requiring a minimum of heavy machinery (US DoT, 1989, Fischenich, 2003). Such use of hard structures are especially recommended when then bank to be stabilized cannot easily support vegetation due to a lack of topsoil, steep slopes or poor lighting (Duke Energy, 2014). These views, coupled with the effects of immediate erosion control offered by riprap make it an appealing choice given societies tendency towards quick fixes.

From a management perspective hard revetments may seem an easy choice, as evidenced by the predominance of rock riprap in our river networks. However, the riprap design, that is the grade, positioning and amount of rocks used has historically been solely an engineering consideration, based on the hydraulic and geomorphological conditions of the construction site, with little consideration for long term physical and/or ecological changes (Fishenich, 2003). This is problematic because natural river systems are dynamic and local channel characteristics may change over time, whereas stabilization structures such as riprapping are in comparison permanent and rigid (Breschta and Platts, 1986). Hard stabilization structures therefore may pose restrictions on normal fluvial functions, the full extent of which remains largely unknown on a quantitative level (Fischenich, 2003). Further complicating the issue is the variability in results of studies that have been done concerning the positive and major to minor negative impacts of riprap on the environment and biota (Shields et al., 1995; Schmetterling et al., 2001). Given the presence of scientific uncertainty a precautionary stance is advisable regarding future management decisions surrounding the use of riprap stabilization (Breschta and Platts, 1986).

It may also be argued that instances of structural failure and the overall frequency of maintenance often required by riprap is reason enough to question the continuing use of this method as a solution to problems of erosion (Breschta and Platts, 1986; Federal Emergency Management Agency [FEMA], 2009). Consequently, recent studies have begun focusing on the mechanisms and predictability of riprap failure. For instance Jafarnejad et al., 2017 used experimentation to develop an empirical relationship, which may be used to forecast time-to-failure. They defined time-to-failure as the duration until complete structural collapse, which depended on block size, channel slope, and specific discharge. Interestingly, the number of blocks that eroded was found to be minimal until failure occurred. This indicates that maintenance procedures only involving the replacement of eroded blocks, without addressing the complete failure that could result from sliding or slumping may not be efficient.

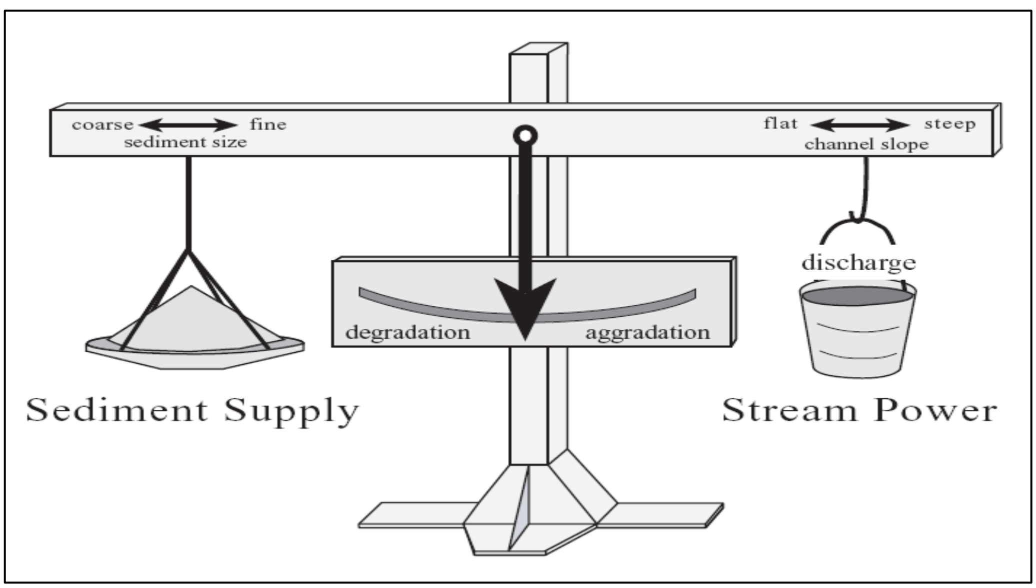
New techniques for bank stabilization are emerging, however, which in many cases consist of a compromise between soft and hard stabilization structures. For example, biotechnical approaches to bank stabilization, which incorporate vegetation as well as human-made structural components, are increasingly discussed in the literature (Shields et al., 1995; FEMA, 2009). Unfortunately, alternative structures have not yet been able to replace traditional riprapping as a standard erosional control measure due in part to a relatively low amount of documentation and high uncertainty regarding long-term project performances (Shields et al., 1995). Therefore in order for environmental managers to make a movement towards 'greener riprapping' more scientific evidence is needed concerning the full environmental impacts of riprap as well as a greater confidence in the ecological and societal benefits offered by alternative stabilization techniques. The remainder of this literature review will depart from the discussion of alternatives to riprap and instead address the potential impacts of riprap on local channel morphological conditions as well as natural fluvial processes with an emphasis on the dynamic nature of fish habitats.

2.1.2. Impacts on local channel morphology and fluvial processes

Natural ecosystems are complex, encompassing a variety of interactions between biotic and abiotic components across a range of spatial scales. The lotic environment presents a prime example of this complexity as it provides important bio-geochemical linkages between the entire watershed, adjacent riparian zones, the channel corridor and the downstream body of water that it eventually joins (Allan & Castillo, 2007). The addition of rock riprapping to stabilize a portion of stream channel will ultimately result in changes to local channel morphology, the riparian zone as well as altering the interactions between the stream banks and stream bed, affecting the overall sediment supply to the stream network and disrupting its downstream transfer (Pitlick and Wilcock, 2001). Indeed, stream evolution, riparian succession, sedimentation processes and biological community processes are the most likely parameters to be affected by riprap bank stabilization (Fischenich 2003). A discussion on the impacts of a structural change to the stream channel therefore must consider not only changes in local morphology but also potential alterations of the natural fluvial processes that are essential for conserving these linkages.

In very general terms rivers and their tributaries act as 'jerky conveyor belts' transporting sediments, along with water and nutrients, from upstream to downstream and from source to sink. However, the relative contribution of sediments from different sources (hill slopes, channel bank, upstream channel bed) as well as the overall sediment load is not constant and will depend on the characteristics of the watershed and the volume of water flowing (Ferguson, 1981). A river responds to the temporal and spatial variability of water and sediment inputs through constant adjustments in its morphology (width, depth, slope, and channel pattern) (Pitlick and Wilcock, 2001). The processes by which these adjustments occur are governed by the principle of mass conservation whereby if the supply of water and sediment is not balanced, the result will be changes in channel geomorphology (Blum and Tornquist, 2000). Natural river systems therefore maintain a dynamic equilibrium between fluid and solid discharge as demonstrated by the balance model (Figure 1). If the overall sediment input from the watershed is low, the maintenance of equilibrium will depend strongly on the sediment supply from the riverbanks and bed (Sear, 1996). The extensive riprapping of a stream channel, which would significantly reduce sediment inputs from bank erosion, may therefore not only disrupt this equilibrium but also prevent local morphological adjustments to be made (Sear, 1996; Piegay et al. 2005). Furthermore, the stabilization of banks which naturally experience high levels of erosion may simply increase erosion elsewhere along the channel as the river attempts to compensate, exacerbating the disturbance (Breschta & Platts, 1986). In such situations, a high level of scour may develop along the riprap base, potentially leading to structure destabilization and the need for frequent riprap maintenance in an endless and costly cycle (Smith and Dragovich, 2008). By altering patterns of erosion and sediment supply and stabilizing the stream banks, riprapping has the potential to interrupt the processes of natural channel mobility and morphological evolution that occur over long timescales.

Figure 2.1 - Lane's Balance Model (Blum and Tornquist, 2000)



The characteristics of natural riverbanks are in part determined by the processes of morphological evolution and sedimentation but also by the process of riparian succession. Bank vegetation also influences fluvial processes and morphology through mechanisms such as flow resistance, bank strength, and the formation of log jams, which, although difficult to quantify, must be considered in any discussion of fluvial geomorphology (Hicken 1984). For example, the types of riparian vegetation present may range from a mix of bare ground and grasses to trees and shrubs each offering different degrees of erosion control and effective bank stability which is assumed to be in equilibrium with the hydraulic forces within the channel, due to the processes discussed above (Breschta and Platts, 1986). Indeed, streams with forested riparian zones may have significant morphological differences from streams with non-forested riparian zones (Hession et al., 2003). Therefore, streams with different bank vegetation successional stages may respond differently to replacement by rock riprap or other forms of bank stabilization. The traditional riprap construction process often involves the complete removal of local bank vegetation, essentially reverting it to an earlier successional stage that not only affects the balance of hydraulic forces but also results in changes to the roughness elements of the bank, altering flow resistance (Hicken, 1984). Although riprap design does have the potential for vegetation to regrow within the rock interstices, it remains unclear whether or not this will translate into the full restoration of bank vegetation function or if it will eventually reach the same successional stage as before the construction occurred (Fischenich, 2003).

Depending on the size and location of the project riprap impacts may extend well beyond the local channel corridor, altering downstream dynamics of erosion and deposition as well as patterns of riparian succession and the mechanisms by which vegetation in turn affects fluvial form and processes. However, the degree to which riprapping will have either a positive or negative impact on stream hydrology and morphology will also depend strongly on the spatial scale, in addition to the geomorphological context in which the particular stream network is set (Shields et al., 1995). For example, at the microscale (approximately the diameter of the mean stone size), the addition of riprap larger than the natural substrate can create fine scale changes in flow conditions differing greatly from the main channel (Shields et al., 1995; Pitlick and Wilcock, 2001). Empirical evidence suggests that adding structural elements such as large rocks to a stream can alter rates of scour and deposition, potentially increasing the heterogeneity (variability) of the physical environment (Yarnell et al., 2006). The potential positive impacts of this added heterogeneity are especially evident for streams in agricultural watersheds that experience a large influx of fine sediments, which can cover the natural bed material, creating a homogeneous environment and an overall loss of habitat (Pitlick and Wilcock, 2001; Yarnell et al., 2006). At the meso- (approximately 10 – 15 times the channel width) and macro-scale (the stream network and surrounding floodplain), riprap can alter the planform geometry and cross-sectional shape of the channel as well as connections with the floodplain and riparian vegetation, potentially leading to habitat losses or decreased habitat quality for biota (Shields et al., 1995; Gidley et al., 2012). Thus while at the micro-scale riprap may lead to local changes that positively influence the fluvial environment, some potentially negative impacts may become apparent at the meso- and macro-scale.

Further complicating our ability to identify riprap impacts on fluvial morphology and processes and whether they are positive or negative towards the environment is the level of degradation of the channel prior to the riprap construction and the extent of the bank which has been stabilized (Gidley et al., 2012). Other anthropogenic sources of disturbance such as agricultural land use practices or the damming of a tributary may already be perturbing natural fluvial conditions through, for example, an increase input of fine sediments (Pitlick and Wilcock, 2001; Matthaei et al., 2010). Logically, riprap is more often needed in degraded areas rather than in pristine environments, as it is an erosional control measure. In such cases it is quite possible that any structural change to the channel could result in an improvement, although it may not necessarily be an improvement over natural conditions. Care must therefore be taken when attempting to generalize riprap impacts on the fluvial environment due to the highly individualized nature of each river and the variability of processes, which may be effected differently across a range of spatial scales.

2.2. Fish habitats

2.2.1. Temporal and Spatial Dynamics

One of the many ecosystem functions/services of rivers is the provision of habitat for various types of biota. Fish habitats in particular are dynamic in both time and space, being sensitive to changes in the fluvial processes discussed in the previous section (sedimentation, riparian succession and geomorphology) that directly influence local channel characteristics (Knighton, 1998; Florsheim et al., 2008). Indeed natural lotic systems, including smaller streams, provide diverse habitats supporting a range of species with variable life histories (Breschta and Platts, 1986). For example, small-scale differences in hydraulic conditions can create discrete morphological units within the channel which are characterized by different flow velocities, depths and substrate sizes, producing a unique set of habitat parameters that biota can utilize (Armantrout, 1998). These 'habitat units' may be of direct importance for certain fish species throughout the year or at specific stages during their life cycles such as seasonal spawning in fast flowing well aerated riffles or the growing out of fry in deep, slow flowing pools (Bain and Haifang, 2012). In addition to water depth and velocity, the availability of specific sediment types and sizes may affect the persistence of fish and invertebrates in an area (Gordon, 1992). Salmon for example, require a mix of sediment types such as gravel, sand and cobble as spawning substrate, and depending on what life cycle stage they are in, these needs may change (Breschta and Platts, 1986; Schmetterling et al., 2001). Thus a diversity of physical parameters within a channel is often required in order to satisfy a variety of fish habitat needs.

Other factors may influence the quality of habitat in a given area, affecting its overall functional significance to different species. For example, the resultant shade produced by overhanging vegetation from the riparian zone can moderate peak summer temperatures, providing refuge for fish species such as many types of salmonids that are sensitive to extended exposure to high water temperatures (Ruesch et al., 2012). Riparian vegetation may also provide an input of woody debris, which can serve as cover from predation while simultaneously altering local hydraulic conditions creating additional habitats that differ from the main channel (Angradi et al., 2004). Indeed, hydraulic connections between the channel, its banks and the floodplain are essential for maintaining quality fish habitat in many lotic ecosystems by allowing fish access to backwater habitats and other zones of refuge such as undercut banks (Shields et al., 1995; Florsheim et al., 2008).

The availability of good quality habitats within the fluvial environment may also have indirect importance due to interconnections between species in the form of predator prey relationships. For instance, while a certain channel may have habitats that are acceptable for fish species, if the habitat requirements of prey species, such as various insect species, are not met, the decrease in food sources may reduce the ability for fish to persist in that area (Meyer et al., 2007). Healthy fish habitat therefore typically involves diverse physical conditions both in the channel and along the riparian zone, leading to a heterogeneous environment that can satisfy the needs of all local biota (Brown, 2007). It is a sound ecological principle that the complexity of physical habitat components is positively correlated to biological complexity (Newson and Newson, 2000). Thus, losses in habitat complexity and heterogeneity are often associated with negative impacts towards species diversity and community structure, potentially reducing local biodiversity through both direct and indirect effects (Cardinale et al., 2002; Benton, et al., 2003; Brown, 2007).

The types of habitats available and the habitat requirements of different fish communities will, however, depend largely on the overall characteristics of the stream and the ecosystems it supports. Although no two streams are exactly the same, they may be broadly classified according to similarities in physical characteristics, which will be strongly influenced by the geological context of the watershed. For example, areas of mountainous terrain are characterized by rivers with high gradients, fast flows and coarse bed materials such as gravels, cobbles and boulders (Church et al., 2012). Watersheds of Lowland geology will however, often produce sand-bed rivers that typically have low gradients, slower flows and finer sediments consisting mostly of coarse and fine sands that may develop into bedforms such as ripples and dunes (Hassan et al., 1995). The hydraulic and geomorphic differences exhibited by these types of rivers will lead to fundamental differences in physical fish habitat and ecosystem characteristics and consequently, the types of fish communities living there. Warm-water sand-bed streams, for example, often support extremely diverse biological communities consisting mostly of fish species from the Cyprinids (minnows) and Centrarchids (sunfish) family (Shields et al., 2003). Gravel-bed rivers on the other hand tend to support cold-water species of the Salmonidae family such as varieties of trout and salmon (Avery, 1995; Schmetterling, 2001). Differences also exist in terms of biological diversity, often measured as species richness, which is generally considered to be much higher in warm-water streams compared to cold-water fish communities (Gorney et al., 2012). Clearly there are strong links between the influence of geology on fluvial processes, the characteristics of the streams produced and the types of fish habitat and biological communities supported. Any assessment of fish habitat diversity, quantity or quality must therefore consider not only the habitat needs of the relevant species but differences in watershed geology and geomorphology as well.

2.2.2. Habitat parameters and types of assessments.

Due to the dynamic nature of fish habitats and the types of communities they support, many different methodologies exist for assessing habitat parameters at multiple scales, both quantitatively and qualitatively. For example, biologically meaningful habitat parameters that are typically measured quantitatively include: flow velocity, flow depth, bed substrate composition, temperature and cover (Avery, 1995; Cardinale et al., 2002; Gidley et al., 2012; Ruesch et al., 2012). Although these parameters can be highly variable at the microscale (<1m) they may also be grouped into distinct hydraulic units, which at the mesoscale (~10m) exhibit relatively homogeneous conditions compared to the overall diversity within the channel (Frissell et al., 1986). In the context of fish habitat, these discrete hydraulic units, often termed 'mesohabitats', are generally classified into four main categories: pools, riffles, runs and glides (Breschta and Platts, 1986). While the theory behind this classification is well established (Breschta and Platts, 1986), mesohabitat identification in the field can be quite difficult depending on the water level and the degree of anthropogenic influences within the channel, requiring extensive training (Rankin, 1989; Rankin, 2006). Several studies focus on measuring mesohabitat dimensions (width, length, maximum depth) as well as other characteristics such as velocity and substrate composition in order to identify their sensitivity to anthropogenic influences (Moerke et al., 2003; Lau et al., 2006; Gorney et al., 2012). The importance of mesohabitats can be illustrated by the fact that the rehabilitation of deep pools and riffles is the subject of many stream and fish habitat restoration projects (Newbury, 2013)

The variability of fish habitat parameters as well as the sheer complexity of some environments also makes measuring it difficult and time consuming. In order to circumvent this problem, researchers have attempted to balance available resources with the accuracy of habitat assessment by either focusing on the overall habitat conditions of a stream section or on specific parameters which are important for the particular species of interest. For instance, many types of Rapid Bioassessment Protocols (RBP's) and Qualitative Habitat Evaluation Indexes (QHEI's) exist to rapidly measure or visually estimate all habitat components that are significant for native fish species (Barbour et al., 1999; Rankin, 1989; Rankin, 2006). These types of assessments are geared towards making more general statements about habitat availability, quality and hence suitability of a particular stream to support diverse fish communities by including components such as channel morphology, riparian zone characteristics and landuse practices within the surrounding watershed (Rankin, 1989; Rankin, 2006). However, while more holistic approaches do help to save time and resources, being relatively quick to complete, they lack the predictive powers offered by more quantitative methods and also introduce biases related to the subjectivity of visual assessments (Gostner et al., 2012).

Successful fish habitat characterization may be found in studies that focus on a few parameters (i.e. habitat hydraulics) that tend to account for the majority of habitat variation (Rhoads et al., 2003). For example, intense sampling of depth and velocity within a stream section has been shown to be an accurate measure for fish habitat heterogeneity without the need for more data intensive methods such as habitat modeling (Gostner et al., 2012). Indeed, habitat models are often used in order to predict changes in available habitat following anthropogenic alterations to natural flow regimes. However, due to the large amounts of data they require to run, such models are generally highly empirical and therefore focus on only one or a few species. Furthermore, models and other methods that focus only on the hydraulic regime (i.e. microscale habitat characteristics, depth and velocity) may not capture habitat dynamics at larger scales which require the consideration of the riparian zone and associated habitats (Fernandez et al., 2011).

Clearly, variations in fish habitat assessments, whether quantitative or qualitative, mirror the variability in lotic ecosystems as well as the research question or goals of the study. As a consequence, two studies assessing fish habitat for the same stream section but using different protocols may ultimately produce different results. In order to address this problem, a recent study compared 50 different fish habitat assessment methodologies and their ability to monitor fish habitat across a range of spatial scales over long periods of time (Fernandez et al., 2011). They argue that the development of regional standards such as the European Guidance Standard for Assessing the Hydromorphological Characteristics of Rivers is necessary in order to improve the systematic assessment of fish habitat despite potential differences in management goals (CEN, 2002). It was found that combining methods that involve both quantitative and qualitative measurements could maximize the accuracy of future fish habitat assessments (Fernandez et al., 2011). Thus, assessments such as the QHEI, developed by the state of Ohio, which combines visual and quantitative measurements, may be a good trade off between accuracy of habitat assessment and low resource requirements.

2.3. Variability of riprap impacts on habitat and biota

From the previous discussion it becomes apparent that riprap bank stabilization has the potential to significantly alter fish habitat either through direct structural changes or by affecting the fluvial processes that form them. Indeed, by creating static environments, riprapping may remove natural heterogeneous conditions with negative implications for biota (Breschta and Platts, 1986; Florsheim et al., 2008). For this reason several studies have attempted to determine the impacts (positive, negative or neutral) that riprap may have on different fish species, with a few focusing on fish habitat. However, for the most part results are variable and inconsistent between studies, making it difficult to move forward with regards to modifying riprap designs. Although unfortunate, others have noted that this variation in findings is not surprising due to significant differences between studies in terms of experimental design, species and life cycle stages considered, the degree of perturbation and physical characteristics of the sites (Craig and Zale, 2001; Schmetterling et al., 2001; Fischenich, 2003). These differences will be highlighted in the following sections.

2.3.1. Gravel-bed rivers

Positive findings of riprap on fish and physical habitats have been noted by Avery (1995) who found that the number of deep pools significantly increased after the placement of riprap along two reaches of Millville Creek (~ 4km of a gravel-bed stream), Wisconsin. Deep pools are important habitats for fish fry that cannot tolerate fast flows, as well as for refuge during times of low flow in the summer when water levels may decrease connectivity along the stream channel (Rankin, 1989; Rankin, 2006). A significant increase in the standing stocks of brown trout was also observed following riprap treatment, however this was the only species sampled. Although the sampling of fish and the physical characteristics of the study reach was conducted both prior to and two years following the riprapping construction, Avery (1995) acknowledges that no reference zone was established which limited the "before" and "after" comparison. It is also clear that the stream reach was highly degraded prior to riprapping with high levels of erosion and no bank vegetation.

Mixed results have been noted by Knudsen and Dilley (1987) who observed an increase in both yearling steelhead and cutthroat trout and a reduction of juvenile Coho salmon and young-of-the-year trout in two large rivers, Lower Deschutes River and Decker Creek, Washington, after the placement of riprap. However, adverse impacts were observed for all species sampled for three smaller streams which had experienced bed as well as bank modifications, suggesting a relationship between the magnitude of the impact and the degree of stream alteration. This study was, however, limited due to the fish sampling which only took place before riprap treatment and again within three weeks after stabilization. Therefore, their results may be more related to the disturbance caused by the construction process (i.e. a temporary displacement of the fish) rather than to the long-term impact of the riprapping.

Negative impacts on fish habitat parameters have been documented by Peters et al. (1998) who found that riprapped reaches had significantly less overhanging vegetation and lower large woody debris densities in comparison to the reference reaches. Impacts on fish densities however, were less clear, with sub-yearling trout, Coho and Chinook salmon densities being lower at riprap stabilized banks while older trout densities were either unaffected or increased. Similar variability in results was noted by Li et al. (1984) who reported the presence of juvenile cutthroat while also noting the absence of Chinook salmon and lower larval fish densities along continuous revetments.

2.3.2 Fine-sediment rivers (sand/silt/clay)

Fewer studies have documented the impacts of riprap within the context of sand-bed rivers, however the variability in results is similar to that found for gravel-bed rivers. For example, a higher overall relative fish abundance was reported for stabilized sites in Coeur d'Alene River, Idaho USA (Gidley et al., 2012). In terms of impacts on fish habitat Gidley et al. (2012) noted a positive correlation between the average rock diameter of the stabilization and relative fish abundance. In this case, few overall differences were seen in habitat metrics between stabilized and non-stabilized reaches due to the consistently low habitat quality along all reaches of this stream (Gidley et al., 2012). In contrast, negative riprap impacts on fish habitat have been reported by Gorney et al. (2012) who found that riprapped reaches had pools which were significantly shallower and riffles significantly longer compared to non-stabilized reference reaches. Although both of these studies worked with larger sample sizes, results are limited in each case to one particular river.

2.3.3. Breaking down the variability

The above variability in results can be explained in part by differences in experimental designs ranging from before/after riprapping (Knudson and Dilley, 1987; Avery, 1995) to pairwise comparisons of stabilized vs. non-stabilized reaches (Peters et al., 1998; Gidley et al., 2012; Gorney et al., 2012) to studies that focus on only one river (Avery, 1995; Gidley et al., 2012; Gorney et al., 2012) and those spanning a diversity of rivers (Knudson and Dilley, 1987; Peters et al., 1998). Furthermore, few studies sampled multiple species across various seasons, and those that did (Peters et al., 1998; Gorney et al., 2012) had difficulty obtaining significant differences due to small sample sizes and a large amount of variability in parameters measured.

Several studies demonstrated variability in riprap impacts related to the life-stage of the species. For instance young-of-the-year salmon tended to be negatively affected by riprap stabilization while older cohorts did not seem disturbed (Li et al., 1984; Knudson and Dilley, 1987; Peters et al., 1998). This is not surprising as salmon species in particular express different habitat needs depending on their life cycle stage and are often found to be sensitive to bed heterogeneity (Schmetterling et al., 2001).

Differences in terms of the level of habitat degradation and geomorphic characteristics of the river (i.e. sand-bed vs. gravel-bed) also lead to inconsistent results between studies. For example, positive and no-difference results were noted for impacts of riprap on physical fish habitat when the river was already highly degraded due to anthropogenic activity prior to riprapping (Avery, 1995; Gidley et al., 2012) whereas negative results were reported when riprap removed natural bank vegetation, significantly altering the stream from reference conditions (Knudson and Dilley, 1987; Peters, 1998). The overall lack of results concerning sand-bed rivers also makes it difficult to understand riprap impacts on fish and fish habitats in these environments.

2.4. Conclusion

River ecosystems are complex, showing a high level of diversity both within and between watersheds, providing dynamic habitats through time and space (Allan & Castillo, 2007). Since theoretically no two rivers are the same, it is difficult to formulate generalizations concerning anthropogenic impacts on the lotic environment. For this reason, although riprap clearly has the potential to significantly alter natural fluvial process and the creation of fish habitats by imposing a static structure into a dynamic system, there remains a lack of scientific consensus as to whether the overall impacts are negative, neutral or even positive (Craig and Zale, 2001). Given the abundance of riprap stabilization structures in our streams and rivers, and the scientific uncertainties regarding impacts, more compelling scientific evidence is required in order to ensure that riprap designs represent the best possible compromise between societal and ecological needs.

The research presented in this thesis will therefore attempt to evaluate quantitative and qualitative changes in biologically significant fish habitat parameters by comparing stream segments that have been stabilized by riprap with reference non-stabilized segments. The research will consider various aspects of the riparian zone, local hydraulic conditions and the overall geomorphological context (Lowland versus Appalachian) in order to explain the potential variability in riprap impacts between environments.

Chapter 3. Impacts of river bank stabilization using riprap on fish habitat in two contrasting environments

This chapter was written in collaboration with my supervisor Dr. Pascale Biron and Research professional Guénolé Choné. The manuscript is currently published with the following citation:

Massey, W., Biron, P. M., Choné, G. (2017). Impacts of river bank stabilization using riprap on fish habitat in two contrasting environments. *Earth Surf. Process. Landforms 42*, 635–646.

As first author I was responsible for the development of research objectives, methodology, site selection, experimental design, collecting the data, presentation of results, statistical analyses and writing of the manuscript. Dr. Pascale Biron provided guidance for all stages of the research and writing process as well as contributing significant revisions in response to the constructive criticism from anonymous reviewers of earlier versions of the manuscript. Guénolé Choné contributed major revisions to statistical analysis, which greatly improved overall transparency and the limitations of our analysis. It should also be noted that there are some changes in chapter 3 of this thesis compared to the publication due to clarifications requested by my evaluation committee.

Abstract

Riverbank stabilization using rock riprap is commonly used for protecting road and bridge structures from fluvial erosion. However, little is known about how streams adjust to such perturbation or how this can affect fish habitat in different fluvial environments, particularly for non-salmonid species in small streams. The objective of this study is to assess impacts of riprap on fish habitat quantity and quality through a pairwise comparison of 27 stabilized and non-stabilized stream reaches in two physiographic regions, the St. Lawrence Lowlands and the Appalachian highlands of Montérégie-Est (Quebec, Canada). Both quantitative (hydro-morphological index of diversity, HMID) and qualitative (Qualitative Habitat Evaluation Index , QHEI) fish habitat assessment techniques are applied in order to compare results between methods. For each stream reach depth and velocity were measured to calculate HMID. In-stream cover (woody debris, overhanging vegetation, undercut banks, aquatic macrophytes) and habitat units (pools, riffles, runs, glides) were also documented and used to determine QHEI. Results show that overall bank stabilization using riprap at bridge and stream crossings alters fish habitat characteristics. Loss of in-stream covers and riparian vegetation lower QHEI scores at stabilized reaches, especially in more pristine Appalachian streams, but has less impact on already altered straightened Lowlands streams. In this latter context, some positive alterations of fish habitat were observed in riprapped reaches due to the coarsening of the substrate and an induced increase of slope. The two metrics (HMID and QHEI) revealed similar differences between stabilized and non-stabilized sites for Lowlands sites, but their level of agreement was much less in the Appalachian streams, suggesting caution when interpreting habitat quality results based on a single metric.

Key Words: fish habitat, bank stabilization, habitat metrics, qualitative habitat assessment, river straightening.

3.1 Introduction

Bank stabilization using rock riprap is commonly applied at road stream crossings to protect bridge structures from fluvial erosion (Biedenharn et al., 1997; Richardson et al., 2001; L'Association des transports du Canada (ATC), 2005; Chou et al., 2011). The design characteristics (grade, slope and thickness) have been predominantly based on engineering concepts, with little consideration for long-term effects on physical and/or ecological processes (Fishenich, 2003).

There have been several attempts over the past 35 years to quantify riprap impacts on fish communities, abundance and habitat, many of which have already been compiled into existing literature reviews (Craig and Zale, 2001; Schmetterling et al., 2001; Reid and Church, 2015). However, in part because of differences between investigations and the purpose of riprap stabilization, highly variable results have been reported and it remains unclear to what extent riprap influences the majority of fish habitat characteristics. For example, some studies listed in the summary table of Reid and Church (2015) (their Table 2) as having a positive effect used riprap for fish habitat restoration projects (e.g., Hunt, 1988; Binns, 1994); thus, the positive outcome in these cases is not surprising. Also, focusing on the true effects of riprap that are separate from other confounding variables such as channelization or diking (e.g., Chapman and Knudsen, 1980) is essential (Schmetterling et al. 2001), but very difficult to do in practice.

A review of studies that have attempted to isolate the effect of riprap on fish habitat and fish population (Table 3.1) highlights that cold water fish (mainly salmonids) were in general targeted, as also noted by Reid and Church (2015). Positive and negative impacts were noted, with more negative effects in small streams (Knudsen and Dilley, 1987; Gidley et al., 2012) and where the pre-stabilization stream condition was good (Table 3.1).

Armouring of both banks or of the channel bed is expected to have more severe temporal and spatial effects (Reid and Church, 2015), however these cases are seldom studied – most of the cases presented in Table 3.1 only had one bank stabilized, and none had riprap on the bed.

Also contributing to the variation in results reported is the large number of fish habitat assessments available. Table 3.1 reveals that few studies on riprap impact have used a standardized protocol for fish habitat. Biologically meaningful habitat parameters that are typically measured quantitatively at the micro-scale include: flow velocity, flow depth, bed substrate composition, temperature and cover (Bisson et al., 1982; Avery, 1995; Parasiewicz, 2001; Cardinale et al., 2002; Hauer et al., 2011; Gidley et al., 2012; Ruesch et al., 2012). For example, the Hydro-Morphological Index of Diversity (HMID), with intense sampling of depth and velocity within a stream section, has been shown to be an accurate measure for fish habitat hydraulic heterogeneity without the need for more data intensive methods such as habitat modeling (Gostner et al., 2012). However, such microscale measurements may not capture habitat dynamics which requires the consideration of the riparian zone and associated habitats (Fernandez et al., 2011).

When grouped into distinct hydraulic units at the mesoscale (~10m), microscale (<1m) hydraulic variables are expected to exhibit relatively homogeneous conditions compared to the overall diversity within the channel (Frissell et al., 1986; Parasiewicz, 2001). In fact, many fish restoration projects aim

Table 3.1. Studies focusing on the impact of river bank riprap on fish habitat and fish communities.

Study	Site	Size and type of streams	Type of fish	Physical habitat assessment method	# of sites	Experimental design	Reason for riprap	Stream condition	Identified effect
Chapman & Knudsen (1980)	Puget Sound area, Washington	Small, gravel and sand	Salmonids	Velocity, cover, substrate	36	Paired (altered and control)	Channelization	Degraded	Negative
Pennington et al. (1983)	Lower Mississippi River, Missouri	Large, sand	Gizzard shad, catfish	None	4	Not paired (2 concrete revetments, 2 unaltered	Unstable banks	Confined by levees	Positive
Li et al. (1984)	Willamette River, Oregon	Large, gravel	Salmonids	Velocity, temperature, substrate, cover	12	Paired (spur dikes, continuous revetment, natural)	Unstable banks	Degraded	Negative
Farabee (1986)	Mississippi River, Missouri	Large, sand	Gizzard shad, catfish	None	4	Not paired (2 with riprap, 2 natural banks)	Unstable banks	Degraded	Positive
Knudsen and Dilley (1987)	Central western Washington	Small and medium, gravel	Salmonids (trout)	None	5	BACI (paired - upstream control/riprap treatment)	Repair damage from flood	Good	Negative
Hunt (1988)	Willow Creek and Doc Smith Branch, Wisconsin	Small, sand and gravel	Salmonids (trout)	None	2	Before/after (no control)	Habitat improvement	Degraded	One negative, one positive
Avery (1995)	Milville Creek, Wisconsin	Small, gravel	Salmonids (trout)	Depth and bank cover	2	Before/after (no control)	Unstable banks	Highly degraded	Positive on pool habitat,
Dardeau et al. (1995)	Yazoo River, Mississippi	Large, sand, silt and clay	Catfish	Habitat Evaluation Procedure (US Fish and Wildlife Service, 1980)	Unknown	Not paired (compared banks with/without cover, main channel, riprap and sandbars)	Habitat improvement	Degraded	Positive
Peters et al. (1998)	Western Washington	Medium to large, sand and gravel	Salmonids	Mesohabitat identification, depth, velocity, substrate, cover, vegetation, woody debris	67	Paired (riprap/control) for each stream	Flood protection	Good	Negative
Garland et al. (2002)	Lake Wallula (impounded Columbia River), Washington	Large, gravel	Salmonids	Velocity, depth, substrate	277	Not paired - distributed sites (21% riprap, 79 % natural)	Unstable banks	Good	Negative

Study	Site	Size and type of streams	Type of fish	Physical habitat assessment method	# of sites	Experimental design	Reason for riprap	Stream condition	Identified effect
Kimball and Kondolf (2002)	Redwood Creek, California	Small, gravel	Salmonids	Depth, mesohabitat units, woody debris	5	Not paired – comparison with non-riprapped adjacent streams	Unstable banks	Good	Negative
Quigley and Harper (2004)	North Thompson River (British Columbia, Canada)	Large, gravel	Salmonids	Depth, velocity, overhanging vegetation, woody debris	93	Not paired - distributed sites (58% riprap, 42 % natural)	Railway protection	Good	Negative in summer, neutral in winter
Quigley and Harper (2004)	Thompson River (British Columbia, Canada)	Large, gravel	Salmonids	Depth, velocity, overhanging vegetation, woody debris	101	Paired sites with similar velocity	Railway protection	Good	Positive for large riprap
Quigley and Harper (2004)	Coldwater River (British Columbia, Canada)	Medium, gravel	Salmonids	Depth, velocity, overhanging vegetation, woody debris	6	Paired sites with similar velocity and water depth	Habitat improvement	Good	Negative and positive in summer; mostly negative in winter
Quigley and Harper (2004)	Lower Fraser River (British Columbia, Canada)	Large, gravel and sand	Salmonids	Depth, velocity, overhanging vegetation, woody debris	147	Not paired	Railway protection	Good	Positive in winter, negative at high flow, neutral otherwise
White et al. (2010)	Kansas River, Kansas	Large, fine sediments	Catfish and carpsucker	None	48	Random sampling of riprap, log jam or mud bank	Unstable banks	Degraded	Positive
Gidley et al. (2012)	Coeur d'Alene, Idaho	Medium and large, fine sediments	Cold (salmonids) and warmwater fish (cyprinids)	Rapid Bioassessment Protocol (Barbour et al., 1999)	24	Paired (stabilized, unstabilized)	Reduce erosion to limit entry of contaminated soil into river system	Degraded	Positive
Gorney et al. (2012)	Mac-o-chee Creek, Ohio	Small, sand	Cyprinids, centrachids	QHEI	6	Paired (constrained, unconstrained, recovering)	Road protection	Degraded	Negative

at rehabilitating these mesohabitat features, which are generally classified as pools, riffles, runs and glides (Bisson et al., 1982; Beschta and Platts, 1986; Kershner and Snider, 1992; Maddock, 1999; Hauer et al., 2011; Newbury, 2013).

Other fish habitat protocols have focused on a qualitative assessment of the overall habitat conditions of a stream section or on specific parameters that are important for the particular species of interest. For instance, many types of Rapid Bio assessment Protocols (RBP's) and Qualitative Habitat Evaluation Indexes (QHEI) exist, allowing all habitat components that are significant for native fish species to be rapidly measured or visually estimated (Rankin, 1989; 2006; Barbour et al., 1999). These types of assessment include components such as channel morphology, riparian zone characteristics and landuse practices within the surrounding watershed (Rankin, 1989; 2006). While these holistic approaches save time and resources, they lack the predictive power offered by more quantitative methods and also introduce biases related to the subjectivity of visual assessments (Gostner et al., 2012).

Combining methods that involve both quantitative and qualitative measurements could maximize the accuracy of fish habitat assessments (Fernández et al., 2011), but is seldom done in practice. In addition, in the context of riprap bank stabilization, very little information is available to determine whether quantitative microscale and qualitative mesoscale habitat measurement methods generate similar results, and if these metrics are affected by factors such as reach length or flow conditions.

The objective of this study is to assess the impact of riprap bank stabilization on fish habitat for small streams in two physiographic regions in Quebec (Canada): the St. Lawrence Lowlands and the Appalachian highlands. Both quantitative (HMID) and qualitative (QHEI) fish habitat assessment techniques are applied in order to compare results between methods. It is hypothesized that riprap contributes to a loss of in-stream cover and habitat complexity for sites in regions with overall better fish habitat quality such as the Appalachian highlands while potentially adding fish habitat diversity in the highly perturbed St. Lawrence Lowlands.

3.2 Methodology

Study Area

Riprap bank stabilization in this study was always used to protect bridges or culverts. Fish habitat data were collected from 27 sites located in the Montérégie-Est region (Quebec, Canada, Figure 3.1). These sites are small streams, with an average wetted width of 7.8 m (Table 3.2). In all cases both banks were stabilized, and so was the bed in 14 cases. These sites were stabilized with riprap between 2000 and 2012, representing construction techniques currently used in Quebec. These techniques consist of a 1.5:1 bank slope, with a horizontal encroachment at the bed, angular riprap ranging in diameter between 300 and 800 mm and no vegetation (Figure 3.2).

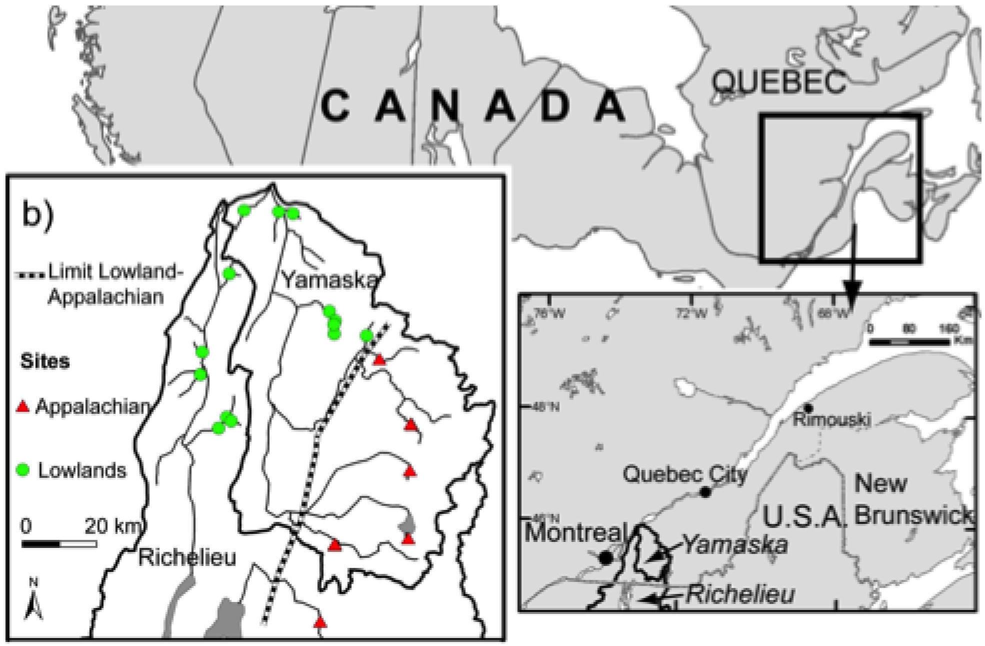


Figure 3.1: a) Location of the study sites in the Montérégie-Est region, south-east of Montreal, Quebec (Canada); b) study sites located either in the Lowlands region (circles), or Appalachian region (triangles).

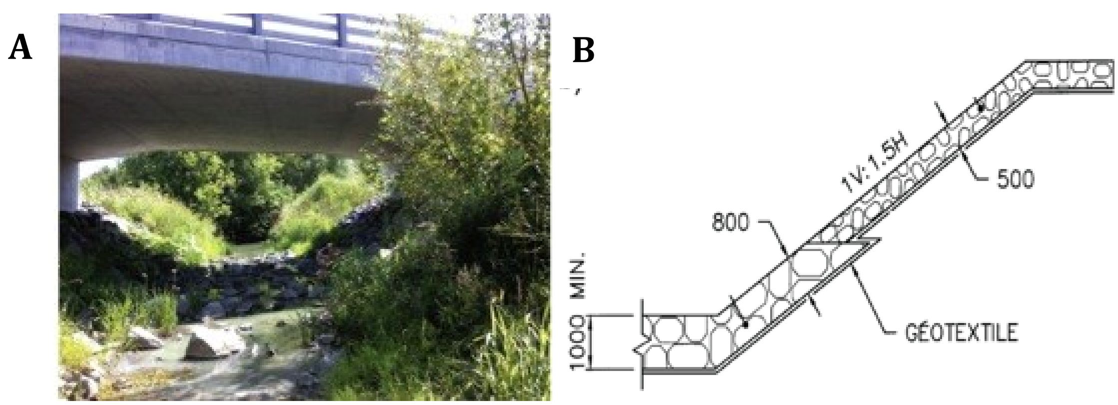


Figure 3.2: a) Photograph of the Lowlands site 17399 (Grande Décharge) showing both banks and bed stabilized with riprap; b) Engineer plans for the right bank and part of the bed for the same site.

Table 3.2: Main characteristics of the Lowlands (fine sediment) and Appalachian (gravel) field sites. Stream order is evaluated from 1 : 20 000 maps. "Bankfull discharge" is the 2yr discharge calculated with : Q = 0.597A^0.869 (A in km2, Q in m3/s) This formula come from Vermont:

[**USGS Scientific Investigations Report 2014–5078: Estimation of Flood Discharges at Selected Annual Exceedance Probabilities for Unregulated, Rural Streams in Vermont**]

*** Sites where repeated measurements were taken in the non-stabilized reaches**

Site ID	Name of stream	Drainage area (km^2)	Bankfull discharge (m^3/s)	Stream order	Water surface slope stabilized /non-stabilized (%)	Wetted width (m)	Length of riprap (m)	Bed riprapped?	Straightened stream?
Lowlands									
17536*	Le Ruisseau	10.2	4.5	3	0.640 / 0.250	5.5	59	No	No
17399*	La Grande Décharge	15.4	6.4	3	0.125 / 0.047	5.0	20	Yes	Yes
17320*	Des Hurons	160.2	49.2	5	0.036 / 0.134	8.0	32	No	No
17533*	Décharge des Onze	32.3	12.2	2	1.747 / 0.024	5.3	55	Yes	Yes
16602*	Des Hurons	138.7	43.4	5	0.704 / 0.008	10.0	75	Yes	Yes
11659	Duncan	78.6	26.5	4	0.042 / 0.047	7.0	56	Yes	No
17791*	des Aulnages	12.5	5.4	3	0.030 / 0.038	3.0	44	Yes	Yes
17537*	Beloeil	58.4	20.5	3	0.134 / 0.004	8.0	33	Yes	Yes
17737*	Richer	14.2	6.0	3	0.285 / 0.066	3.0	27	Yes	Yes
10917*	Pot au Beurre	76.1	25.8	4	1.077 / 0.024	5.0	42	Yes	No
11943*	Saint-Louis	47.1	17.0	3	0.247 / 0.014	4.0	32	Yes	No
12458*	Fagnant	13.8	5.9	3	0.949 / 0.022	5.0	27	Yes	No
17571	Bras-de-Vis	24.8	9.7	4	0.066 / 0.274	8.4	49	No	No
10997	Chibouet	58.8	20.6	4	0.037 / 0.109	6.2	26	No	Yes

17276	Chibouet	68.2	23.4	4	0.284 / 0.027	8.5	125	Yes	Yes
17733	Chibouet	71.0	24.3	4	0.113 / 0.025	5.8	40	Yes	No
17201	Chibouet	72.0	24.5	4	0.002 / 0.086	5.3	55	Yes	No
Appalachian									
16818	Gear	70.2	24.0	4	0.508 / 0.507	10.5	48	No	No
17132	Gear	70.3	24.0	4	0.580 / 0.507	10.5	36	No	No
16549	Aux Brochets	98.4	32.2	5	1.060 / 1.352	11.2	59	No	No
17631	Aux Brochets Nord	58.1	20.4	4	0.031 / 0.564	11.3	26	No	No
17507	No name	10.7	4.7	3	0.813 / 0.585	2.9	22	Yes	No
10777	Castagne	46.5	16.8	4	0.255 / 0.625	8.9	26	No	No
14789	Castagne	47.8	17.2	4	0.100 / 0.625	8.9	23	No	No
10766	Yamaska Nord	58.0	20.3	4	0.098 / 0.520	9.9	22	No	No
17155	Cold	43.9	16.0	4	0.985 / 0.558	7.7	32	No	No
15891	Le Renne	202.8	60.4	4	0.034 / 0.075	26.8	33	No	No

The study sites are located in two major drainage basins: the Richelieu (drainage area of 23 720 km^2, 3 874 km^2 in Quebec), dominated by the St. Lawrence Lowlands, and the Yamaska (drainage area of 4 784 km^2), which extends from the St. Lawrence Lowlands in its downstream portion upinto the Appalachian Highlands (Figure 3.1). None of the flows are regulated by upstream reservoirs for the watersheds in which these sites are located. The St. Lawrence Lowlands are characterized by low gradient rivers as well as a large portion of land devoted to agriculture (70% Richelieu, 50% Yamaska). Streams in the Lowlands are often highly perturbed because of an influx of fine sediments, loss of riparian vegetation and channel straightening resulting from extensive row crop and pasture land use (Berryman, 2008; Simoneau and Thibault, 2009, Table 3.2). In contrast, the Appalachian Highlands are characterized by relatively higher gradient gravel and cobble-bed streams which are much less perturbed than those of the Lowlands, and are thus often viewed as better quality habitats (Berryman, 2008). In order to account for differences in physical characteristics and fish habitat quality, study sites were classified as either Lowlands (17 sites) or Appalachian (10 sites) based on a map of surface geology and evaluation of gradient (< 0.2% for the Lowlands, > 0.5% for the Appalachian sites) (Figure 1b). Gradient was evaluated based on field surveys of the slope of the water surface for the extent of the study sites (20 – 125m)

Site selection and experimental design

A paired comparison of riprap stabilized and non-stabilized reaches was chosen for this study as no data were collected prior to riprap construction. Non-stabilized control reaches were selected using aerial photography and extensive field observations in order to find a location within 500m of the stabilized reach that was morphologically similar, with the same planform geometry, and that also represented the average depth and velocity of the stream.

Habitat measurements

Fish habitat data were collected during mid to late summer between 2013 and 2015, under low flow conditions. At the microscale, measured fish habitat variables were depth, velocity and substrate size. Cross-sections (transects orthogonal to the channel flow direction) were established along the sampled reach with spacing between cross-sections equal to roughly 1/3 the channel bankfull width, following Kaufmann et al. (1999). This resulted in 4 to 17 transects at each stabilized or non-stabilized site. Water depth measurements were taken at 0.25m intervals along each cross-section, with extra measurements added at conspicuous lateral breaks in slope. Depth was measured using a total station (Leica TC805L). Average current velocity measurements were collected with a Swoffer (model 2100) propeller current meter at 4 to 6 positions along each cross-section. Microscale habitat variables were interpolated using spline in the GIS software ArcGIS 10.1 to determine the proportions of area coverage for shallow, deep and slow velocity zones. High and low depth zones were defined as the top and bottom 25% of the entire dataset, which corresponds to > 0.43 m and < 0.17 m, respectively. For velocity, as all the measurements in this study were collected at low flow, faster measured currents are in a range of 0.25 m/s, and are thus unlikely to have a marked impact on fish. Therefore, only slow velocity associated with backwaters were quantified, as these zones are known to provide refuges for early life-stages fishes (Grift et al., 2003; Nunn et al., 2007; Ridenour et al. 2009). Sluggish zones were determined as the bottom range of the propeller meter resolution (< 0.02 m/s).

At the mesoscale, habitat units (pools, riffles, runs and glides) and in-stream cover (woody debris, undercut banks, overhanging vegetation, shallows and aquatic macrophytes) were sampled following protocols established by Kaufmann et al. (1999) and Rankin (2006). Note that there is potential subjectivity in delimiting these units precisely, so only clearly identifiable mesohabitat units were included in this study. This explains why the total value of mesohabitats does not amount to 100% of the reach area. Undercut banks were quantified as a length of stream bank while surface length and width were measured for other cover types. Woody debris was measured as the number of pieces >1m length and 15cm diameter, per meter of stream bank. The proportion of each mesoscale habitat (pool, riffle, run and glide) and in-stream cover type was calculated by dividing the total cumulative surface area of the respective habitat by the surface area of the corresponding reach with the exception of woody debris, which was treated as a density.

Two habitat quality metrics were used in this study for both stabilized and non-stabilized reaches: QHEI and HMID. The QHEI is a multimetric, visual assessment method which provides a holistic assessment by combining important fish habitat variables (substrate, in-stream cover, channel morphology, bank erosion and riparian zone, pool/glide and riffle/run quality, gradient and drainage area) to give a final score indicating the overall suitability of a stream reach to support fish communities. In this study, we have modified the one developed by the State of Ohio EPA to remove inherent biases towards riprapping by treating riprap as a natural substrate and removing categories that automatically assigned riprapped reaches a lower score. The use of the QHEI within a Southern Quebec context is justified by the large commonality between study sites fish and Ohio warmwater fish species (ex: troutperch, logperch, stonecat, variants of suckers, dances, darters, crappies, redhorses) which were included in the IBI scoring, as well as similarities in geology and previous applications in the Richelieu watershed (COVABAR, 2013). The QHEI is relevant for assessing fish habitat in both sand and gravel-bed rivers as in the elaboration of this method the scoring of each metric was calibrated using an Index of Biological Integrity (IBI) indicative of fish communities living in both environments (Rankin, 1989; 2006). In rural streams in Ontario, Gazendam et al. (2016) also found good correlations between QHEI components and benthic data.

The HMID was developed by Gostner et al. (2012), who demonstrated its relevance as a metric for evaluating physical habitat heterogeneity through the use of correlation analysis against geomorphic diversity. It combines the coefficient of variation (CV) for depth and velocity measurements into a single metric for each reach with higher values representing greater heterogeneity:

$$HMID_{site} = V_{(v)} * V_{(d)} \qquad (1)$$

where $V_{(v)}$ is the partial diversity of flow velocity v and $V_{(d)}$ is the partial diversity of water depth d. Partial diversity $V_{(i)}$ of a variable i is calculated by:

$$V_{(i)} = (1 + CV_i)^2 = (1 + \sigma_i/\mu_i)^2 \qquad (2)$$

where CV is the coefficient of variation, σ is the standard deviation and μ is the mean.

Both QHEI and HMID measurements were repeated on two separate years for 11 non-stabilized Lowlands sites (identified in Table 2) to assess the impact of year-to-year temporal variability (during low flow conditions). The impact of the length of the surveyed reach was also assessed at these 11 sites, by computing HMID and QHEI scores for non-stabilized reaches three times longer than in the original experimental design.

Other physical characteristics of the studied reaches

In addition to data describing fish habitats, some characteristics of the studied reach were also assessed in order to explain observed variations of the fish habitats. Firstly, the length of the stabilized reach, as well as the slope and average width of both reaches, were computed from the data acquired with the total station. Median grain size (D_{50}) was also estimated using the standard Wolman pebble count technique. At the stabilized reach, in an attempt to evaluate re-naturalization, the percentage of rocks from the riprap, as opposed to the percentage of fluvial sediments, was visually assessed using a 1m-diameter hoop, following Platts et al. (1983) and Bain and Stevenson (1999) protocols of embeddedness assessment. The protocol consisted of two situations: 1) if bed and banks were stabilized then 4 evenly spaced measurements were taken along the right bank, the left bank and the middle of the channel respectively (12 total) and 2) If only the banks were stabilized then 5 measurements were taken along the right and left banks (10 total). Measurements were only taken were riprap was present. These estimations, which originally aimed to assess riprap embeddedness, were solely done on the bed of the stabilized reach. In cases where the whole bed was not stabilized, the remaining non-riprapped central portion of the channel is small enough to be considered negligible (with the exception of site 15891, see Table 2).

As shown in Table 3.2, several Lowlands streams had previously been straightened to various extents. In order to quantify this effect, a "degree of straightening" index of the non-stabilized reach was determined through a visual characterization of the sites, using aerial photography, based on the amount of human-induced channel straightening over a length of 20 times the bankfull channel width. Four categories were identified: low, medium, high and very high, where "very high" is entirely channelized (i.e., sinuosity of 1.0). The degree of straightening was computed for the Lowlands sites since none of the Appalachian sites were straightened. The sinuosity of both the stabilized and the non-stabilized reaches was also computed from the same data, This method has, however, some limitations because of uncertainties and subjectivity surrounding the establishment of a true valley centerline on flat Lowlands terrain.

Finally, an extended longitudinal profile of the water surface was taken with a DGPS (model Spectra Precision 80) for 10 stabilized sites of the Lowlands area, from approximately 700m upstream to 200m downstream of the stabilized reach.

Data analysis and statistical treatment

Differences between mean values of habitat and physical metrics were tested using t-tests by permutation, with a paired version to compare stabilized and non-stabilized reaches, and a non-paired version to compare Lowlands and Appalachian sites, following Legendre and Legendre (2012). These non-parametric versions of the paired and independent samples student t-test were chosen as there was no evidence that the dataset followed a normal distribution. In theory, t-tests by permutation require homoscedasticity (Legendre and Legendre, 2012). However, Good (2006) indicates that these tests are not very sensitive to a difference of variance: a ratio of variance of up to 5 produced errors in calculated p-values less than 1.5%. Accordingly, t-tests by permutation were conducted in all cases where the ratio of variance was less than 5. In order to reduce heteroscedasticity, a logarithmic transformation was also applied to the slope measurements as well as to habitat unit proportions (pools, riffles, runs and glides) and in-stream cover proportions (woody debris, undercut banks, overhanging vegetation, shallows and aquatic macrophytes).

Multiple linear regression was used to explain variation in differences between the HMID and QHEI scores of stabilized and non-stabilized reaches using R software. The final variables included in the model were chosen by applying an improved forward selection method, as outlined by Blanchet et al. (2008), on 7 potential explanatory variables: 1) degree of straightening, differences between the stabilized and non-stabilized reaches in 2) slope, 3) sinuosity, 4) D_{50} and 5) average width, 6) length of stabilization, and 7) degree of embeddedness of the riprap. As the sample size for Appalachian sites was too low to permit successful multiple linear regression, correlation was used when attempting to explain differences between HMID and QHEI scores for stabilized and non-stabilized reaches (Pearson correlation with Student tests by permutation). Correlations between the explanatory variables were also computed and tested (Student tests by permutation).

Finally, it has to be reminded that the small number of sites assessed (17 in the Lowlands and 10 in the Appalachian zone) do not provide a high statistical power. Following Peters et al. (1998), a "conservative alpha level" of 10% was used for statistical comparison because of the high variability in the data and resulting low power. In addition, despite carefully choosing the sites to be as representative as possible of these two contrasted environments, because some sites are located on the same river or in geographically close contexts, there is potentially minor violation of observations independency.

3.3 Results

Differences between stabilized and non-stabilized reaches

Box-and-whisker diagrams of the proportion of shallow, deep, slow and fast flow are presented for the Lowlands and Appalachian sites in Figure 3.3. In the Lowlands context, the proportion of shallow depth zones was 12% higher in stabilized reaches than in non-stabilized reaches ($p < 0.05$) while the proportion of zones with a sluggish velocity was 27% lower ($p < 0.01$) (Figure 3.3a). Stabilized reaches also had a water surface slope 5.4 times greater ($p < 0.05$) and a narrower wetted channel width (21%, $p < 0.01$) than non-stabilized reaches.

In the Appalachian context, deep zones were found to be 12% more frequent in stabilized reaches (Figure 3.3b), with no significant changes in channel width. Slopes were found to be shallower at the stabilized reach ($p<0.10$).

A comparison of stabilized against non-stabilized reaches for sites in the Lowlands context revealed clear differences for all mesohabitat categories (Figure 3.4). Stabilized reaches are characterized by a lower proportion of pools (6%, $p < 0.05$) and glides (8%, $p < 0.01$) and a higher proportion of riffles (9%, $p < 0.01$). Runs were only observed in stabilized reaches. Little difference was found between the proportions of pool, glide and run mesohabitats between stabilized and non-stabilized reaches in the Appalachian context; however the proportion of riffles was significantly lower for stabilized reaches (17%, $p < 0.05$).

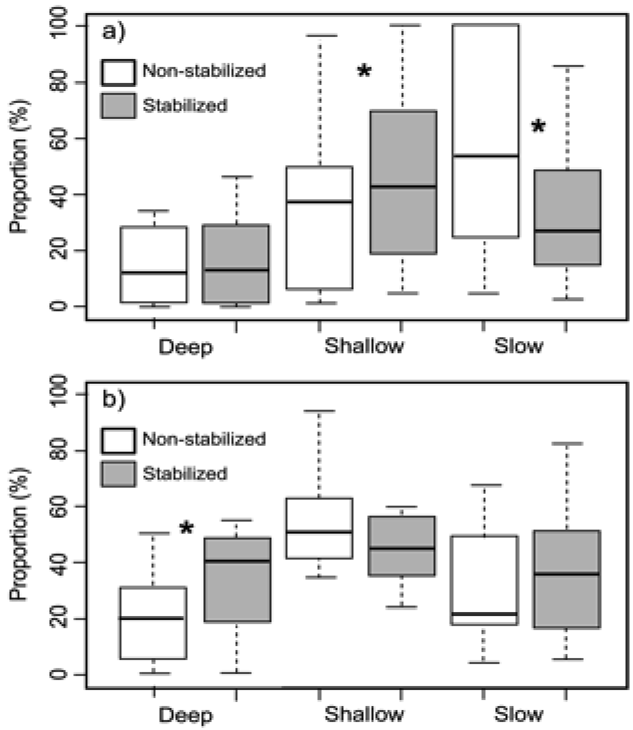


Figure 3.3: Distribution of study reach microhabitat zones for Lowlands sites a) and Appalachian sites b). Note: * denotes significant difference (p<0.05). Proportions are of reach wetted surface area.

Proportions of instream cover types reveal, as expected, that only non-stabilized reaches from both the Lowlands and the Appalachian sites displayed undercut banks (Figure 3.5). Similarly, stabilized reach sites had lower proportions of overhanging vegetation in the Appalachian context ($p < 0.05$) and in the Lowlands ($p < 0.01$) (Figure 3.5). More shallows were also observed at the stabilized reach for the Appalachian sites ($p < 0.1$), and more woody debris density was also found in non-stabilized reach in both Lowlands and Appalachian sites, but the low number of observations of woody debris prevents any statistical inference.

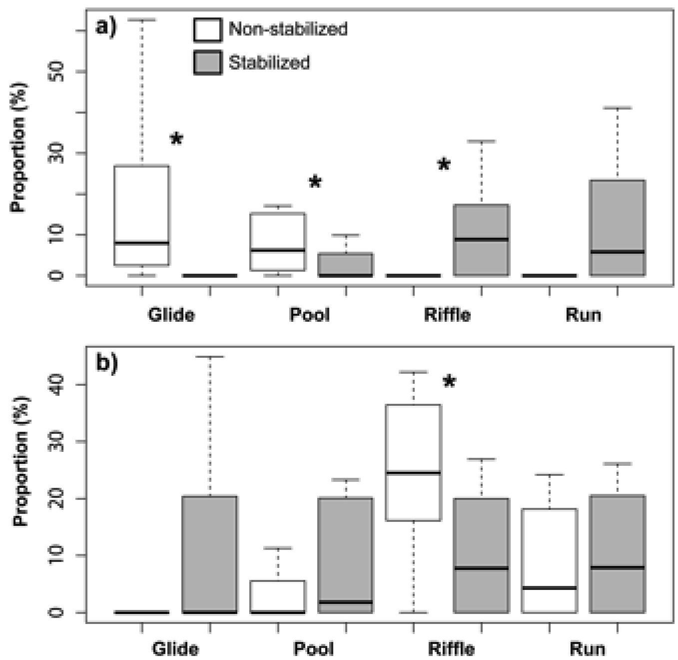


Figure 3.4: Distribution of study reach mesohabitats for Lowlands sites a) and Appalachian sites b). Note: * denotes significant difference (p<0.05). Proportions are of reach wetted surface area. Since the proportion of runs in the Lowlands non-stabilized sites was 0%, it was not possible to run a statistical test in this case.

HMID mean values are higher in stabilized reaches compared to non-stabilized reaches, with a 53% increase for Lowlands sites ($p < 0.05$) and a 21% increase for Appalachian sites ($p < 0.10$) (Figure 3.6). The Lowlands sites in particular demonstrate a large amount of variability in HMID values, ranging from 4.9 to 23.2 for stabilized reaches and 1.9 to 17.5 for non-stabilized reaches.

Values of QHEI ranged from 28 to 71 for the Lowlands sites and 42 to 80 for the Appalachian sites. The value of QHEI scores was found to be markedly higher (45%, p <

0.01) for non-stabilized Appalachian reaches compared to non-stabilized Lowlands reaches, highlighting their superior habitat quality.

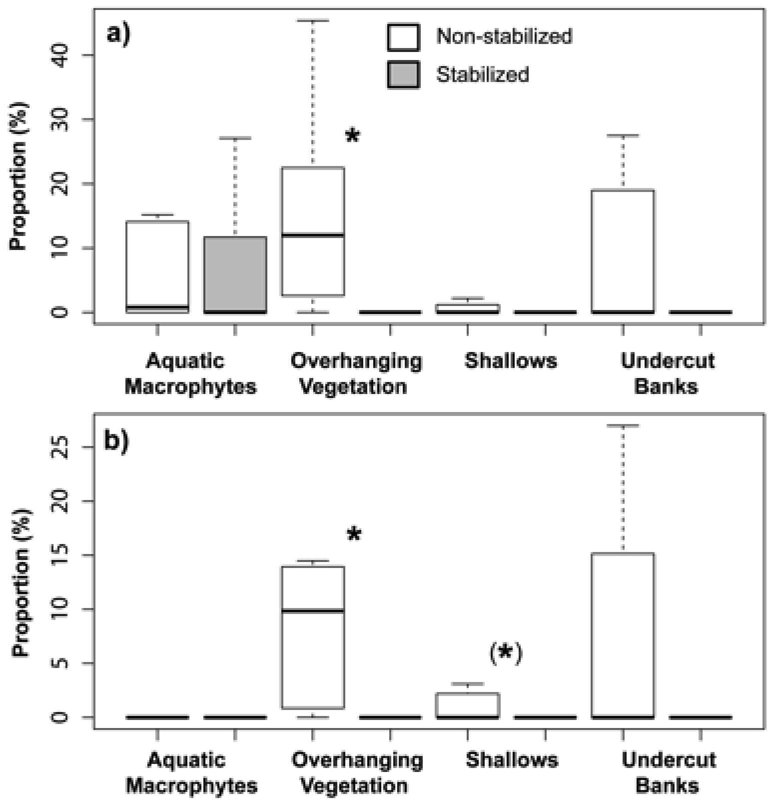


Figure 3.5: Distribution of study reach in-stream cover for Lowlands sites a) and Appalachian sites b). Note: * and (*) denote significant difference with $p<0.05$ and $0.05<p<0.10$, respectively. Proportions are of reach wetted surface area except undercut banks which are expressed as a proportion of bank length. Since the proportion was 0% for undercut banks at stabilized reaches, as well as for macrophytes at non-stabilized reaches in the Appalachian sites, it was not possible to run a statistical test in these cases.

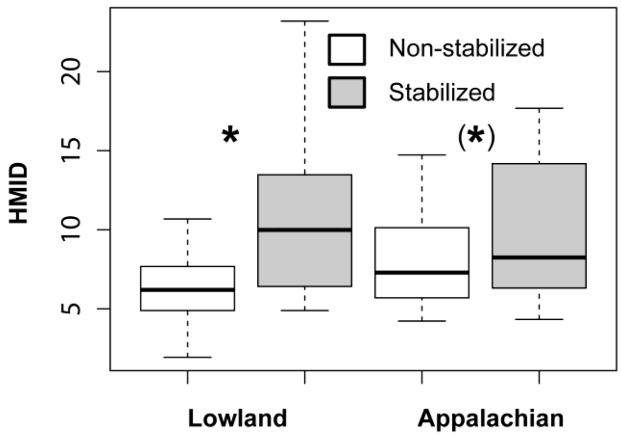


Figure 3.6: Distribution of study reach Hydro-morphological Indices of Diversity (HMID) scores for Lowlands and Appalachian sites. Note: * and (*) denote significant difference with p<0.05 and 0.05<p<0.10, respectively. Higher HMID values represent greater heterogeneity.

A comparison of stabilized and non-stabilized reaches revealed clear differences in QHEI for Appalachian streams (14% lower, $p < 0.01$) but not for Lowlands streams (Figure 3.7). In the Lowlands, there is a high variation in habitat quality, with 7 sites having higher scores in the non-stabilized reaches and 10 where the highest score is in the stabilized reach, resulting in no statistically significant differences (Figure 3.7).

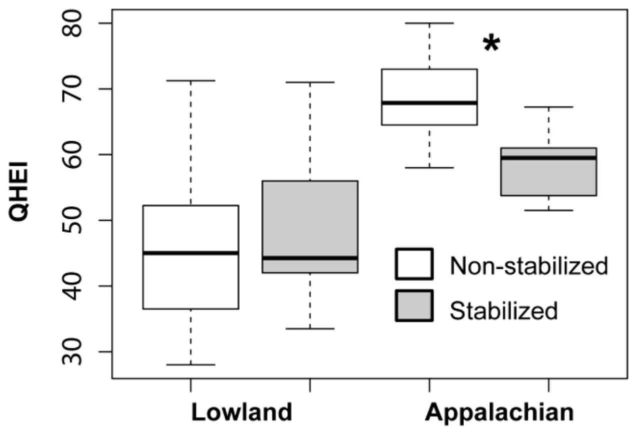


Figure 3.7: Distribution of study reach Qualitative Habitat Evaluation Indices (QHEI) scores for Lowlands and Appalachian sites. Note: * denotes significant difference ($p<0.05$). Higher QHEI values represent greater potential for a reach to act as suitable fish habitat for a range of species.

QHEI and HMID variability

Multiple linear regression revealed that 67% of the variation in the magnitude of the difference between HMID scores of the stabilized and non-stabilized reaches (HMID stabilized – HMID non-stabilized) for the Lowlands sites can be explained by two independent variables: the degree of embeddedness in the stabilized reach (negative relationship, $p<0.01$) and the degree of straightening of the non-stabilized reach (positive relationship, $p<0.01$) (Table 3.3).A multiple linear regression was also performed for the difference and QHEI scores between the stabilized and the non-stabilized reach and it was found that 69% of the variation of the QHEI difference can be explained by two independent variables: the difference of slope between stabilized and non-stabilized reaches (positive relationship, $p<0.01$) and the degree of straightening of the non-stabilized reach (positive relationship, $p<0.01$) (Table 3.3). Correlations between explanatory variables can limit the confidence in the variable selections process of the regression models (Zuur et al. 2007). A strong correlation was indeed observed between the degree of embeddedness and the difference of slope between the stabilized and non-stabilized reaches for the Lowlands sites ($r=-0.63$, $p<0.01$). This correlation may be explained by the change in stream power at the stabilized site associated with the change of slope, which in some cases promotes sediment deposition (hence embeddedness). When excluding one of the correlated variables from the model, as suggested by Zuur et al. (2007), the selected variables for the two models (HMID and QHEI) remain the same, with the addition of the length of the stabilization in the QHEI model (Table 3.3). For the Appalachian sites, HMID and QHEI differences were not found to be correlated with any of the tested explanatory variables.

Repeated measurements of QHEI and HMID at 11 non-stabilized reaches in the Lowlands did not indicate any statistically significant differences ($p = 0.831$ and 0.153, respectively) and are highly correlated ($r=0.88$ and 0.90, $p<0.01$ in both cases). Similarly, no statistical differences were observed between QHEI or HMID scores measured over a longer reach on the same day ($p = 0.226$ and 0.524, respectively) with highly correlated QHEI scores ($r=0.84$, $p<0.01$). HMID scores measured over different lengths, however, were not correlated.

Table 3.3 – Results of forward selection for HMID and QHEI multiple linear regressions for Lowlands sites.

Differences are always score of stabilized reach - score of non-stabilized reach.

	Explanatory Variable	**Cumulative adjusted R^2**	**Direction of correlation**	**Coefficient**	**Significance**
HMID Difference	(Intercept)			6.3183	
	Degree of embeddedness (correlated with the difference of slope)	0.434	negative	-4.2173	p<0.01
	Degree of straightening	0.668	positive	3.7928	p<0.01
QHEI Difference	(Intercept)			-26.955	
	Difference of slope (correlated with the degree of embeddedness)	0.470	positive	9.101	p<0.01
	Degree of straightening	0.685	positive	8.627	p<0.01
QHEI Difference (with the difference of slope excluded)	(Intercept)			-16.45994	
	Degree of straightening	0.411	positive	11.15813	p<0.01
	Degree of embeddedness (correlated with the difference of slope)	0.627	negative	-7.33244	p<0.01
	Length of stabilization	0.762	positive	0.23018	p<0.05

3.4 Discussion

Stabilization impacts in the Saint Lawrence Lowlands

Results show that several aspects of a river are impacted by stabilization in the Saint Lawrence Lowlands. In particular, width is narrower and water surface slope is steeper at the stabilized reach. It is hypothesized from field observations that the narrowing of the channel at the stabilized reach is due to stabilization design. While this narrowing is likely specific to regional riprap design practices, the steepening of the longitudinal slope appears related to a geomorphological effect. Indeed, the detailed DGPS profiles revealed that breaks-in-slopes are present at the stabilized reaches, with a gentle slope often observed upstream of the stabilized reaches, a sharp increase in slope at the stabilized reach, and gentler slope downstream of stabilization. Comparison of these detailed longitudinal profiles with data available prior to stabilization (obtained from the design blueprints) revealed that these breaks-in-slopes were present before the reconstruction (as most stabilizations result from maintenance of previously stabilized sites). The breaks-in-slopes could be inherited from older under-sized stream crossing structures, or could have been created by stream incision just downstream from the stabilization with the older stream crossing structures acting as a grade-control point. This particular feature of the longitudinal profile is responsible for the observed higher proportion of shallow depth zones, lower proportion of zones with a sluggish velocity, lower proportion of pools and higher proportion of riffles and runs. While these changes could be considered as habitat loss for fish species that prefer or require a mild gradient and slow flows typical of the Lowlands context, they actually create more flow diversity, in association with the coarser riprap bed material, thus increasing the HMID scores.

A deeper look at the breakdown of the differences of the QHEI scores shows that among the 7 components of the QHEI, the three with the largest impact are in order 1) differences in substrate (positive), 2) differences in in-stream cover (negative) and 3) difference in riparian zones (negative). The role of riprap on the bed on fish in an overall clay-substrate stream such as those found in the Lowlands should be further investigated (see below). Here, riprap on the bed is considered a better substrate than clay because it creates a coarser substrate, with more heterogeneity in the particles sizes, two criteria considered positive by the QHEI assessment. Thus, negative biases towards riprapping in the original QHEI index developed by the State of Ohio EPA were removed in this study. With regards to the riparian zones, the removal of riparian vegetation at the stabilized reach is, not surprisingly, considered as having a negative impact on the aquatic habitat. The observed decrease in in-stream cover at stabilized reaches is consistent with other observations made (Figure 3.3) and with the findings of other studies. For instance, Peters et al. (1998) and Thompson (2002) found that riprapped reaches had significantly less overhanging vegetation and lower large woody debris densities in comparison to reference reaches.

QHEI and HMID regression models are very similar. Firstly, both include the degree of straightening, indicating that a straightened non-stabilized reach offers a poorer-quality habitat. This implies that the straightening process lowers what is measured by these two indices as a good habitat. This is not a consequence of the stabilization itself, but it highlights the importance of the reference condition of the river: in a heavily impacted river, such as a highly-straightened one, QHEI and HMID scores can become higher at the stabilized reach than in the reference one. However, it must be mentioned that we are not advocating the use of riprap as a form of fish habitat improvement as any modifications in such disturbed streams are likely to result in better fish habitat. It may also be worthwhile to consider some modifications to the riprap design to incorporate vegetation (e.g., Shields et al., 1995), lateral rock or wood outcrops (e.g., spur dikes, Shields et al., 2000; Chou and Chuang, 2011).

Secondly, the two models include either the degree of embeddedness or the difference of slopes. As shown by removing the difference of slope as an explanatory variable (Table 3.3), the effect of one or the other cannot be distinguished, and the regression models should be interpreted accordingly. Consequently, QHEI and HMID regression models can be considered equivalent and highlight the positive effect on fish habitats of the presence of rocks or the steepening of slope, without the possibility to distinguish the separate effects of these two variables. This positive effect of stabilization is confirmed by fishing data from Asselin (2016) that show that, with a sampling based on 9 of the 17 Lowlands sites from this study, stabilization has either a positive effect or no effect on fish diversity, density and biomass. Regression models explaining these fishing results show the preponderant importance of the presence of rocks amongst all measured explanatory variables (Asselin et al., 2016). This subset of our Lowlands sample is however characterized by more straightened rivers ($p<0.05$). This overall positive effect of stabilization is captured by the HMID scores, which are higher on average at the stabilized reaches, but not by the QHEI scores. Our revised QHEI (removing the riprap bias) does indeed a take into account the positive effect of the presence of rocks, but seems to underestimate its importance for fish. The original QHEI from Ohio EPA (with a negative bias against riprap) would have therefore clearly underestimated the quality of fish habitat in highly disturbed, straightened Lowlands streams. Other studies have noted similar positive correlations between habitat heterogeneity and biological diversity (Pedersen et al., 2014) supporting the ecological hypothesis that rivers with more diverse habitats tend to support more diverse species communities (Newson and Newson, 2000).

Stabilization impacts in the Appalachian context

Modifications of the river observed at stabilized reaches in the Appalachians contrasts in some ways with what was observed in the Lowlands. The gentle slope, with fewer riffles and more zones of high depth indicate that the flow at stabilized reaches is modified, but differently. These modifications contribute to increase flow diversity, as for the Lowlands, which is translated into slightly higher HMID scores. In contrast with the modification of slopes observed in the Lowlands, these modifications at the stabilized reaches are not believed to be a consequence of bank stabilization per se, but rather of road position in the landscape: in such a mountainous environment, roads are likely placed at the valley toe, resulting in gentle slope at the river crossing.

Despite higher flow diversity, QHEI scores indicate that the habitat quality in the Appalachians is more severely altered than in the Lowlands. This is likely due to the more pristine state of the non-stabilized reach, and thus reflects more the impact of stabilization itself. The analysis of the different components of the QHEI scores reveals that the three most affected components are the same as in the Lowlands: in-stream cover, riparian area and substrate. In-stream cover and riparian area QHEI sub-scores are lower at the stabilized reaches, for similar reasons than in the Lowlands. However, in the Appalachian context, the substrate quality sub-scores are lower at the stabilized reach than at the non-stabilized one, due to a decrease in the substrate heterogeneity. Consequently, all changes measured by QHEI scores are negative for fish habitat. These changes are, however, not captured by HMID.

Habitat metrics robustness and agreement

Correlation between inter-annual measurements of QHEI and HMID scores indicates that these metrics are relatively robust to quantify fish habitat, at least under low flow conditions. This is reinforced by the small spatial variability displayed by the QHEI score. However, when using longer reaches in the Lowlands where slopes are generally very shallow, small steps with a steeper slope create drastic changes in flow conditions, resulting in unrealistically high HMID scores when one of these steps is included in a given reach.

Despite our attempt to only focus on the impact of riprap and to work on stream reaches of similar size, confounding factors such as the removal of wood in straightened streams, or differences in slope and width between stabilized and non-stabilized reaches, make it difficult to ensure that only riprap effects on fish habitat is assessed. Previous studies have revealed more negative impact of riprap in small streams with only one stabilized bank (Knudsen and Dilley, 1987; Schmetterling et al., 2001). It was thus expected that stabilizing both banks, as well as the bed in some cases, would result in marked negative effects, particularly in the more pristine Appalachian streams. This was the case for the QHEI method (Figure 7), but not for the HMID (Figure 6). In fact, even if the two metrics used in this study were deemed robust they resulted in contrasted trends between stabilized and non-stabilized reaches in the 10 studied Appalachian streams. In contrast, both metrics are well correlated for the Lowlands sites with a correlation of 0.72 ($p < 0.01$) between the differences in QHEI scores per site and those of the HMID scores. For the

Lowlands, the positive or neutral effect of riprapped bed on fish observed in some of the most degraded streams (Asselin et al., 2016) highlights the very large impact of flow heterogeneity provided by the presence of rocks on the bed, which is well picked up by HMID, but not so by QHEI.

Finally, it should be noted that the small samples sizes (17 sites in the Lowlands, 10 in the Appalachians) limit the conclusions that can be drawn from the study. Despite the conservative significance level of 10%, these small samples imply a low power of the statistical tests. Consequently, in addition to the possibly reported false positive results (due to a high significant threshold), some effects of stabilization may have not been detected. Furthermore, the reported effects may be exaggerated due to the so-called "winner's curse" (Button et al., 2013).

3.5 Conclusion

This study has shown that bank stabilization using riprap at bridge and stream crossings alters fish habitat characteristics in Lowlands and Appalachian streams in ways that may have negative or positive implications for local fish communities. More marked geomorphological changes were observed in the Lowlands, with the creation of a break-in-slope at the stabilized site. This induces an increase of slope at the stabilized reaches which can be seen as a positive alteration of fish habitat, although the specific effects of heterogeneity brought by the presence of coarser (riprap) substrate on the bed and the increase in slope cannot be distinguished. These positive effects are counterbalanced by the decrease in in-stream cover and riparian vegetation, resulting in a global positive effect limited to rivers that already had poor fish habitat quality. The effect of stabilization on in-stream cover and riparian vegetation is also observed in the Appalachian context. In this geomorphological context however, the relative homogeneity of the (riprap) substrate at the stabilized site compared to natural gravel-bed heterogeneity is seen as negative for fish habitat.

The two fish habitat metrics QHEI and HMID also revealed similar trends for the Lowlands streams in terms of differences between stabilized and non-stabilized sites. However, the level of agreement between QHEI and HMID was much less in the Appalachian streams, which may be problematic for studies assessing fish habitat based on a single metric. In addition, comparison with biological data showed that QHEI underestimates the positive effect of the presence of rocks in Lowlands sites.

The novelty of this study was to show that in small streams, relatively small extents of riprapping can have different impacts on fish habitat. The measured impacts depend greatly on the geomorphological context (Lowlands versus Appalachian, undisturbed versus straightened stream), but also on the chosen metrics for habitat assessment. While more research is still needed to confirm the results due to the low statistical power linked with small samples sizes, this study contributes to our understanding of the large variability in reported results from previous studies. Future studies should also focus on examining the effect of greater extents of riprap (150m +), which are often used in small streams in Quebec and elsewhere, on both fish habitat quality and biological communities.

In chapter 3 a paired comparison of stabilized versus non-stabilized reaches yielded interesting results regarding differences in fish habitat, the importance of considering the geomorphological context of the study stream, and the level of agreement between different metrics used for fish habitat assessment. However, the high amount of variability in fish habitat conditions when comparing between rivers as well as confounding variables such as prior modifications at the bridge sites and positioning of roads makes drawing conclusions about riprap impacts difficult. Furthermore it remains unclear to what extent positive or negative trends in fish habitat data in relation to riprap, are biologically significant. In chapter 4 I attempt to address these concerns through a study focusing on one river (Salvail) where there was an opportunity to assess fish habitat conditions before and after riprap construction. Biological sampling was also conducted in an effort to draw comparisons with fish habitat trends.

Chapter 4. Pre- and post-assessment of fish habitat and fish communities in a highly disturbed Lowland river

4.1 Introduction

In 2010 the municipality of St. Jude (Montérégie) experienced a massive landslide where a large portion of the Salvail River bank and adjacent hillslope slumped into the main channel. Since a family of four people died during this tragic event, it prompted an environmental investigation by the Ministry of Transport to determine the causes of the landslide and ways to protect against future occurrences (Transports Québec, 2011). Because bank erosion of the Salvail River was considered an aggravating factor in the geotechnical analysis, several sections of the Salvail channel were stabilized by riprapping, protecting areas deemed most at risk for future landslides. As these works were responses to environmental emergencies they provided only a small window for environmental assessment. The proposed stabilization was considered to pose only minimal risk to fish habitat with minor impacts related to projected increases in turbidity during the construction process (Ministère du Développement Durable, de L'Environnement et des Parcs, 2013). Precautionary measures taken during construction were to use sediment barriers, working outside the reproductive period of native species and avoiding excessive contact between the channel and heavy machinery (Les Services exp inc, 2013).

There is currently a lack of fish habitat assessments or biological data acquired before riprap application, as most studies regarding riprap impacts on fish habitat consist predominantly of pair-wise comparisons, with the inherent challenge of establishing proper control sites (chapter 3, tab. 2). Furthermore, Before After Control Impact studies (BACI) are quite rare in the literature and those that have been performed focused on salmonids (trout) (Knudsen and Diley, 1987; Hunt, 1988; Avery, 1995). Acquiring robust before/after information with a focus on non-salmonid species is therefore essential in further isolating true riprap impacts, which are often influenced by confounding variables, related to anthropogenic disturbances such as channelization and livestock impacts (Chapman and Knudsen, 1980).

Understanding riprap impacts on fish habitat in Lowland streams such as the Salvail River is particularly important in Southern Québec where warm-water streams are less studied than cold-water ones, even if they are known to support diverse community structures (Shields et al., 2003) with a high species richness (Gorney et al., 2012), consisting mostly of non-salmonid species from the Cyprinid (minnow), Centrachid (Sunfish) and Percidae (Perch) families. Furthermore, most fish species of small streams tend to be habitat specialists (Gorman and Carr, 1978). For instance, several species native to the Salvail River either prefer or require pool/glide habitats (deep water, sluggish zones), such as redfin shiner (Breder and Rosen, 1966) and trout-perch (Page and Burr, 1991). If such habitats are altered or removed during the riprap construction, these fish species may be negatively affected. Clearly more factors than simply turbidity during the construction work should be considered.

The main objectives of this study were to (A) quantitatively and qualitatively assess fish habitat before and after riprap application at two sites along the Salvail River and (B) compare biological sampling of fish before and after riprap application at the same sites. Research questions to be addressed are (1) how do fish and fish habitat metrics of quality and diversity respond to riprap treatment?; and (2) do potential differences in fish biodiversity (species richness and Simpsons Index of Diversity) and fish habitat diversity agree with each other?

4.2 Methodology

Study area and riprap design

The Salvail River is a medium sized (bankfull width of about 15m) tributary of the Yamaska river situated in the Lowlands of Montérégie-est Québec within the Richelieu watershed. The predominant land use near sites to be stabilized consists of agricultural land with a lack of well-developed riparian buffer. This area was evaluated as having uncharacteristically high artesian water pressure at the base of the Salvail's banks which also showed evidence of high levels of erosion (Transports Québec, 2011). The bed material of the Salvail River consists mainly of clay deposits from the former Champlain Sea, which may be fairly resistant to erosion under low flow conditions but may exhibit severe bank instability under higher flow conditions.

The riprap stabilization project of the most actively eroded reaches of the Salvail River was based on two main procedures. The first step involved removing the massive buildup of clay material on the upper bank. The goal here was to reduce pressure by removing these heavy loads and thus reducing future risk of landslides. The slope of the upper bank was also adjusted to a gentler ratio (Figure 4.1, top). The second step aimed at protecting the base of the bank to prevent any imbalances due to fluvial erosion. Two layers of riprap were applied and embedded into the bank in an attempt to cement them into place. The first layer placed directly on the clay bank was comprised of riprap rocks (D_{50} =200mm) at a minimum thickness of 500mm. A second layer of larger rocks (D_{50} =400mm) was placed on top with a minimum thickness of 800mm (Figure 4.1, (bottom)).

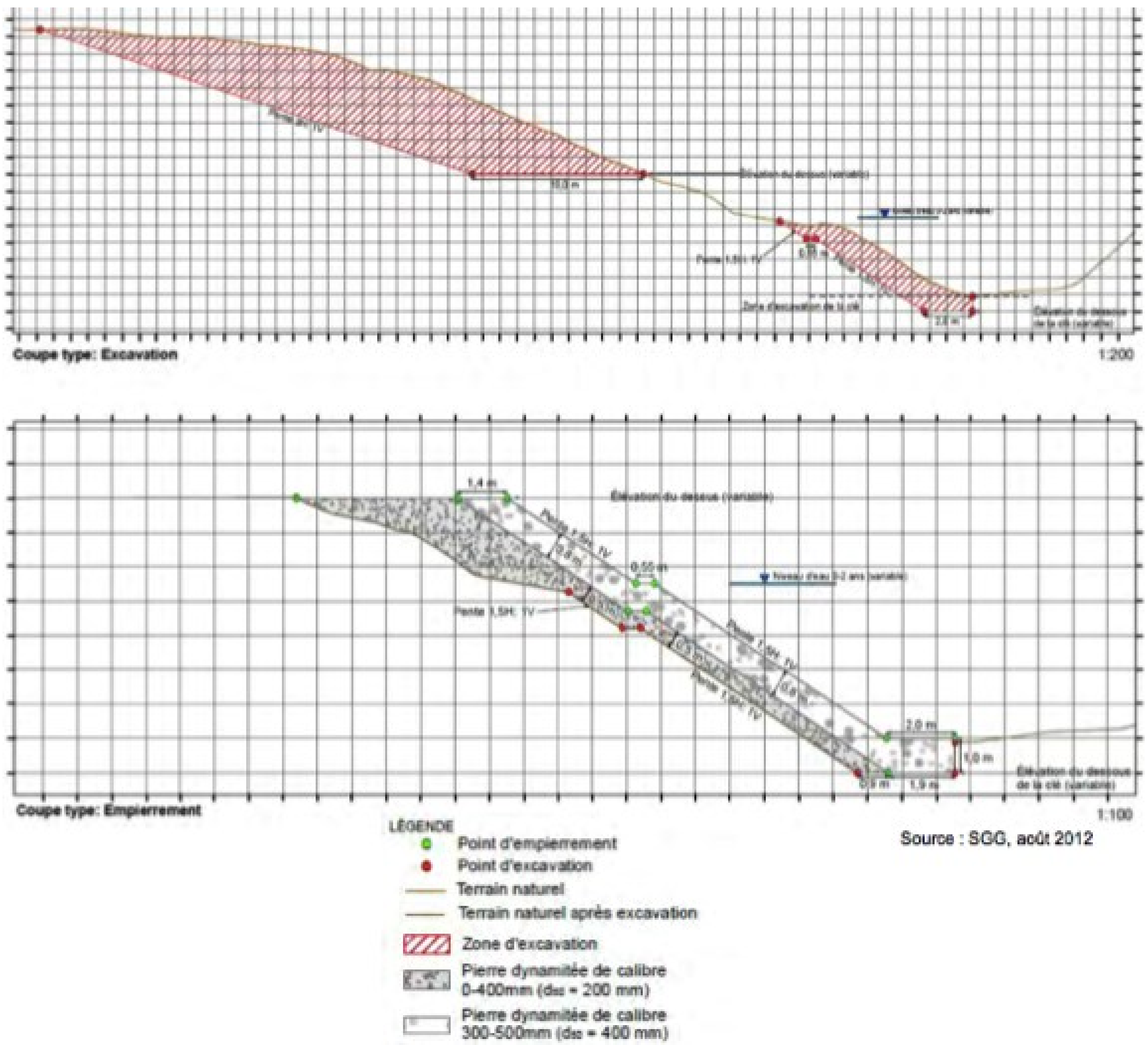


Figure 4.1: Construction plans of bank stabilization on the Salvail River as preventative measures for landslides. The upper portion shows the area at the top of the bank to be removed while the lower part shows the riprap bank stabilization. This figure is an excerpt from Ministère du Développement Durable, de L'Environnement et des Parcs, 2013.

Site selection and experimental design

Two study sites that were scheduled for riprap stabilization in 2013 were selected along the Salvail River (Figure 4.2). In order to minimize the potential influence of anthropogenic disturbances these sites were chosen because they were over a kilometer apart and even further from earlier riprap interventions. Although extensive stabilization projects were planned for several tributaries, only locations to be stabilized at the main channel were selected to maintain consistency of riprap design. Riprap construction at these locations was completed in October 2013, on one bank only. Site 1 received ~90m of bank stabilization while site 2 received ~60m.

The experimental design for this study consists of a comparison of fish and fish habitat metrics before and after riprap stabilization. Three control sites, each ~90m long, were established both upstream, control A and downstream control B and C, from the projected stabilization at site 1 (Figure 4.2). While the impacts of the riprap itself are hypothesized to be localized, due to minimal changes in flow conditions near the structure, it was uncertain if the initial disturbance caused by the construction process would have downstream effects.

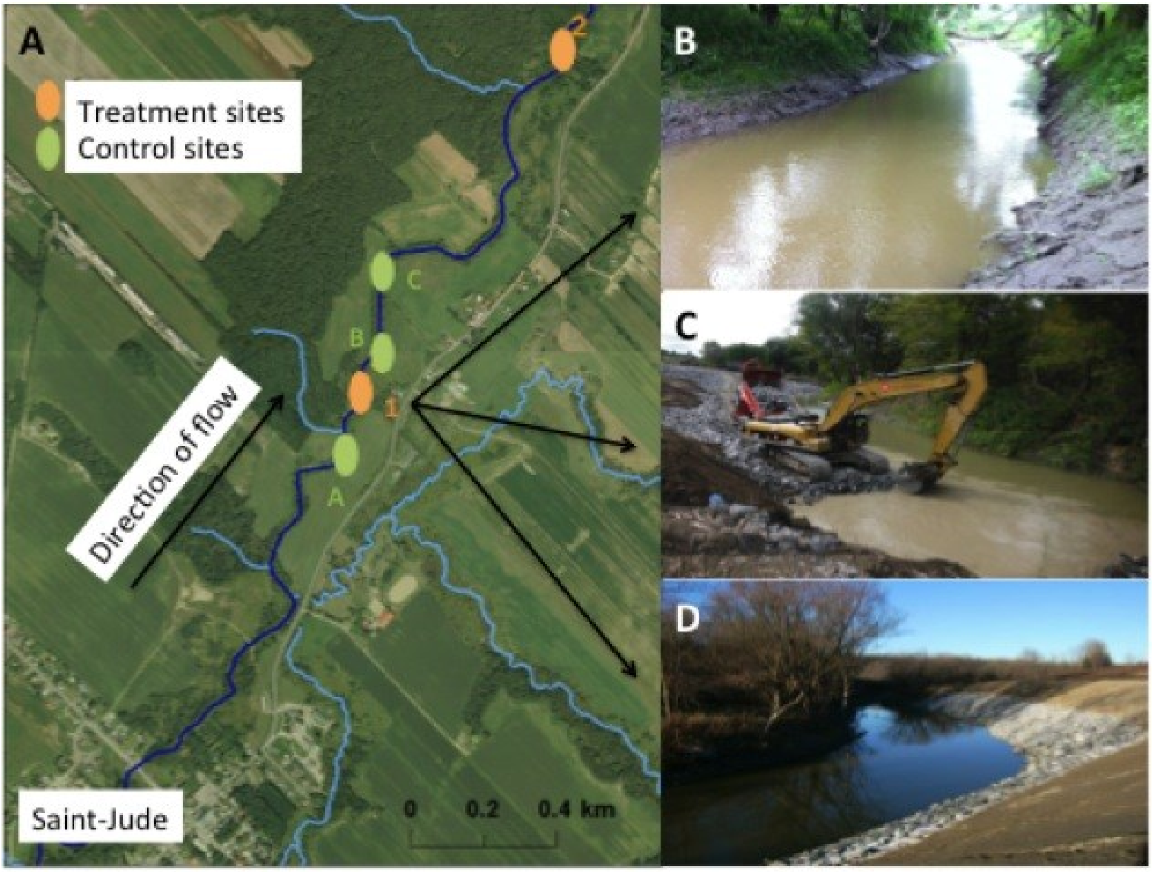


Figure 4.2: A) Location of study sites along the Salvail river. B) Site 1 in July 2013 prior to riprap treatment. C) Site 1 during riprap construction in August 2013, showing how natural bank is removed and replaced with graded stone. D) Site 1 in October 2013 post riprap treatment.

Habitat measurements

Sampling of fish habitat variables took place during July and August 2013 under low-flow conditions, prior to riprap construction and again during the same months and flow conditions in 2014, approximately 10 months after riprap construction. . Low flow discharges were estimated from the velocity measurements for each sample year, yielding similar values (0.531m^3/s for 2013 and 0.427m^3/s for 2014).

Fish habitat was assessed at the microscale for sites 1 and 2 through extensive measurements of depth and velocity. Following the same methodology as outlined in chapter 3, 10 cross-sections (transects orthogonal to the channel flow direction) were established along which depth was measured at 0.25m intervals using a total station (Leica TC805L) and average current velocity measurements were taken at 3 to 5 positions (where differences in flow were obvious). The greater water depth and lack of obstructions at these sites allowed for the use of an ADV (Acoustic Doppler Velocimeter), an instrument which measures velocity in three dimensions at a frequency of 25 Hz. The accuracy and resolution of data collected by the ADV offers a theoretical improvement to the Swoffer (model 2100) propeller current meter used in Chapter 3, which samples only in one direction. These microscale measurements were then used to calculate HMID, a habitat quality measurement developed by Gostner et al. (2012) that evaluates the diversity depth and velocity. Habitat quality was also evaluated using the QHEI metric as detailed in chapter 3

HMID was only sampled at sites 1 and 2 before and after riprap due to time constraints, however QHEI and all mesoscale data, presented as either proportions of wetted surface area, proportion of bank length or density, were sampled for the 3 control sections as well as sites 1 and 2. Mesoscale data included habitat units (pools, riffles, runs and glides) and in-stream cover (woody debris, undercut banks, overhanging vegetation and aquatic macrophytes) and were sampled following protocols established by Kaufmann et al. (1999) and Rankin (2006) as outlined in chapter 3. Because woody debris in the Salvail River was often observed as small to large accumulations, for the purposes of this study it was measured as an area and evaluated as a proportion of the wetted area of the respective study reach. Additionally, the thalweg was measured at regular 5m intervals with additional measurements taken when changes in slope were observed. This was done over a 580m distance starting just upstream from the first control site to slightly downstream from the final control site.

Biological sampling

Fish sampling was conducted on August 2 and 20, 2013, before the riprap disturbance and again on August 26, 27 and 28, 2014, 11 months after construction was completed. Unfortunately, some environmental conditions varied significantly between sampling times which may have affected the presence or absence of fish as well as their capture. For instance, water level was characterized as knee high in 2014 compared to waste high in 2013. Furthermore, heavy rains due to a storm occurred the day before and the morning of August 2, 2013 sampling, leading to a relatively higher discharge and greater turbidity. Daily maximum air temperature was also quite variable with 24°C recorded on August 2 and 27°C on August 20 for 2013, and 28°C, 27°C and 21°C for August 26, 27 and 28, 2014 respectively.

Fish data was collected for treatment sites 1 and 2 as well as for control C located roughly halfway between the two treatment sites (Figure 4.2). The reference location was selected for similarities in physical fish habitat conditions to sites 1 and 2 (i.e. presence of pools and low amount of woody debris), although access to the river and wadeability did influence choice.

Fish were caught using a seine measuring 18m long, 1.4m high with a mesh size of 0.3cm. A seine was chosen as the ideal method for fish capture due to the high level of turbidity, characteristic of the Salvail River, which prevented any visual estimation or electric fishing techniques that involve temporarily stunning the fish. Fish capture using a seine has been shown to produce good yields, even for collection efforts as low as 15% of the wetted area (Cianfrani, 2009, Sullivan et al., 2006). Using such a large net did however lead to some practical difficulties due to the presence of obstacles such as boulders and accumulations of woody debris, which although scarce, had to be circumvented or untangled from the bottom of the seine resulting in the potential for escapees. The riprap itself also proved to be problematic for sampling as some fish may have been able to hide between the rocks, leading to a potentially lower capture rate. The seine was cast from the right bank (facing downstream) at the downstream end of the site and dragged out orthogonally to the channel flow direction to a distance of halfway between banks. It was then pulled upstream and around to the same bank, creating a half circle. Fish sampling was done starting downstream in order to prevent the disturbance of upstream sampling sites (Matthews and Hill, 1979). Sampling was repeated 3 to 5 times consecutively per site in order to maintain a consistent collection effort of about 30% of wetted area.

Collected fish were weighed, measured for length, and identified at the species level. Fish measuring less than 5cm long were identified only at the family level. Due to the high turbidity of this study area, which tended to wash out fish coloration, certain species that are similar such as Mimic/Sand shiners and Tessellated/Johnny darters were not always distinguishable or identified.

Measures of biodiversity

Species richness

Commonly used as a substitute for biodiversity, species richness (S) was evaluated because it is easy to interpret, measure and compare between sites where community structures are the same (Mendes, 2008). Species richness was calculated simply as the number of species per site for each year. This metric is limited however, as it gives each species equal weight, providing no information about potential evenness, rarity or dominance. For a better, more accurate understanding of species diversity, it is often recommended to consider the relative abundance of species in addition to S (Mendes, 2008).

Simpson's Index of Diversity

The Simpson's Index of Diversity is a multivariate dominance index often used as a metric for evaluating fish diversity which takes into account both richness and evenness to measure the probability that two individuals randomly selected from a sample will belong to different species (Cianfrani, 2009). For this study the Simpson's Index of Diversity was calculated as 1-D where,

$$D = (\Sigma\ n(n-1))/(N(N-1))$$

In this case (n) is the total number of individuals of a given species and (N) is the total number of species collected. Values range from 0-1 with 1 being the maximum possible diversity. It is important to note that while the Simpson's Index of Diversity is quite useful for demonstrating dominance, it remains a poor indicator of richness due to the lack of sensitivity to the addition of rare species (Mendes, 2008).

4.3 Results

Changes in fish habitat before and after riprap stabilization

Several differences in physical fish habitat metrics were observed after riprap stabilization for experimental sites 1 and 2 as well as all 3 control reaches, summarized in

Table 4.1. For mesohabitats, the proportion of pool/glide habitat units (zones of deeper, sluggish flow) decreased by 15.6% after riprap stabilization for site 1 and 20.6% for site 2 while the proportion of riffle/run habitat units (zones of shallower, faster flow) increased by 10.7% and 15.9% respectively. An opposite trend was observed for the proportion of pools/glides at control sites, which increased in all cases from 2013 to 2014. These changes are not believed to be due to a backwater effect as control A is located 30m upstream from treatment site 1 and controls B and C are located roughly 1km upstream from treatment site 2. Anthropogenically induced channel narrowing was not observed as only one bank was stabilized. The proportion of riffles/runs, however, remained low at control sites for both sample years.

Due to severe channel incision, frequent slumping and high turbidity observed in the Salvail River, most in-stream cover types were scarce. The only notable exception is the proportion of woody debris which increased drastically from 2013 to 2014 for control site B (26.3%) and C (16.3%) with minimal to no difference between sample years for experimental sites 1 and 2, as well as upstream control site A.

Both metrics for fish habitat quality scored higher post stabilization (tab. 4.1). HMID values increased by 225% at site 1 and 13% at site 2. While QHEI scores in 2013 (pre-stabilization) were low, as expected in lowland rivers, they demonstrate a post stabilization increase by 4.5/91 and 4/91 points for sites 1 and 2 respectively. Interestingly QHEI scores also increased by 5/91 points for control B and 6/91 points for control C, which were located downstream from riprap stabilized site 1, however there was no change in the QHEI score for control A located upstream from the construction.

Table 4.1 Summary of mesohabitat and in-stream cover proportions, HMID and QHEI habitat quality metrics before riprap (2013) and after riprap (2014). To better demonstrate differences in habitat conditions, pool/glide (deeper, slow flowing zones) and riffle/run (shallow, faster flowing zones) units were combined due to their similarities in microhabitat conditions (Beschta and Platts, 1986).

Site	Sampled year	Mean wetted width (m)	Mean thalweg depth (m)	Proportions as % area			QHEI 0 -91	HMID
				Pool/Glide	Riffle/Run	Woody debris		
Control A	2013	6.05	0.47	48.8	9.9	29.0	48.0	-
	2014	6.00	0.45	60.3	13.3	23.7	48.0	-
Site 1	2013	7.10	1.15	31.5	0.0	3.1	44.0	4.63
	2014	4.50	0.44	15.9	10.7	6.0	48.5	10.4
Control B	2013	7.00	0.64	28.5	0.0	6.9	45.0	-
	2014	6.80	0.87	54.4	0.0	33.2	50.0	-
Control C	2013	7.50	0.62	41.7	0.0	2.2	45.0	-
	2014	6.20	0.80	78.0	4.3	18.5	51.0	-
Site 2	2013	7.85	0.73	31.8	0.0	4.9	43.5	7.01
	2014	5.20	0.72	11.2	15.9	3.2	47.5	8.8

A closer look at scores of the individual fish habitat variables evaluated in the QHEI's for sites 1 and 2 revealed marked differences before and after treatment (Table 4.2). Experimental sites responded similarly to riprap treatment with higher scores noted in 2014 for the substrate and mesohabitat quality categories and consistently lower scores for the channel morphology and riparian zone components. A higher score was also noted for in-stream cover post stabilization at site 1 only.

Table 4.2 Summary of QHEI sub-categories for experimental sites.

***Differences here denote the change in score from 2013 to 2014 where blue indicates improved habitat quality and red showing decreases.**

Habitat component	Site 1			Site 2		
	2013	**2014**	**Δ***	**2013**	**2014**	**Δ***
Substrate max 20	4	9	+5	4	9	+5
In-stream cover max 20	6	8	+2	7	7	0
Channel Morphology max 14	9	7	-2	7	6	-1
Riparian zone max 7	5	2.5	-2.5	5.5	3.5	-2
Gradient max 10	8	8	0	8	8	0
Mesohabitat quality max 20	12	14	+2	12	14	+2
Total QHEI max 91	44	48.5	+4.5	43.5	47.5	+4

Riprap impacts

Results for the analysis of riprap impacts on fish habitat characteristics (Figure 4.3) show that riprap likely is having an impact on the reduction of pool/glide proportions with an average decrease for riprap treated sites of 18.1% while showing an average increase of 24.6% at controls (although the increase at controls is not due to the riprap but possibly the construction process, see discussion below). Riprap also appears to have an impact on the presence of riffles as there are much larger average proportions observed at treatment sites (13.3%), compared to controls (2.6%) (Figure 4.3 b). There does not seem to be an impact of riprap on woody debris or QHEI for these study sites as the control sites exhibited larger average increases in woody debris proportions (12.4%) compared to treatment sites (0.6%) (Figure 4.3 c) while controls and treatments also showed similar differences from 2013 to 2014 in terms of average QHEI scores, 4.25% and 3.7% increases respectively.

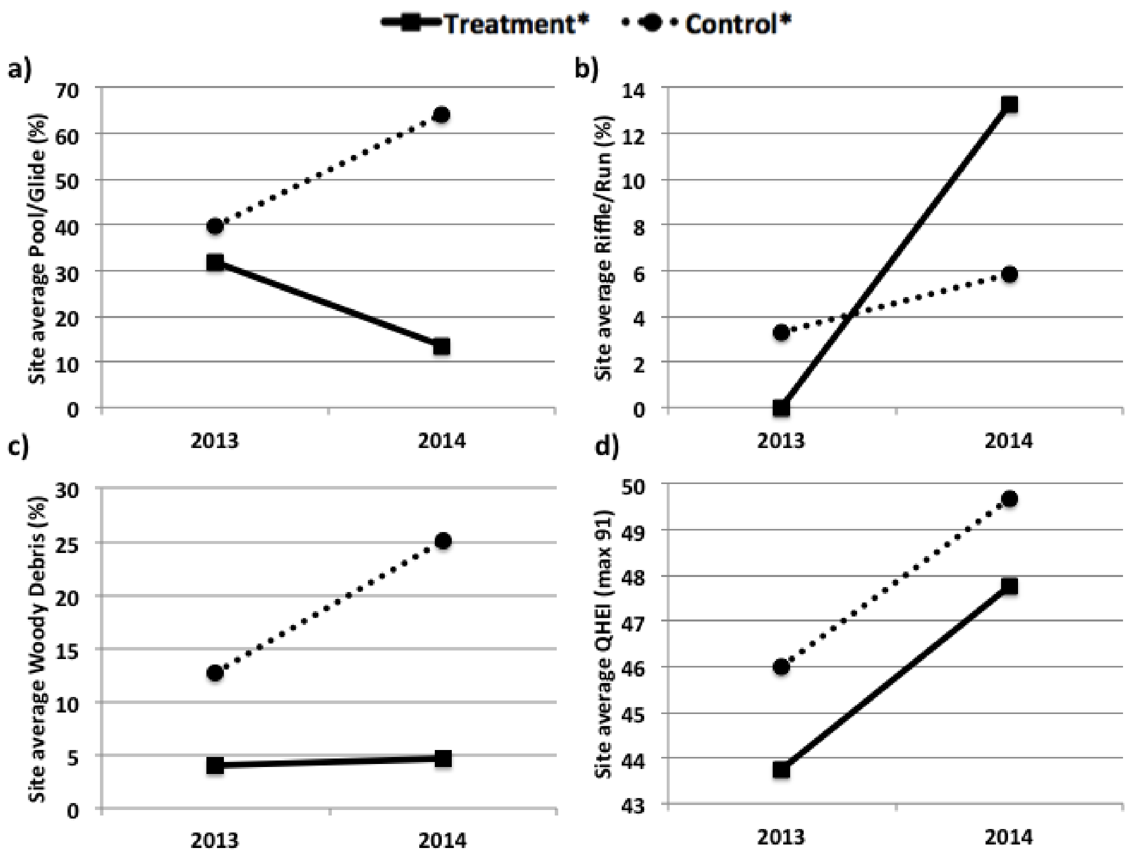


Figure 4.3 Comparison of treatment vs. control for changes pre and post stabilization for fish habitat characteristics a) pool/glide b) riffle/run c) woody debris and fish habitat quality metric d) QHEI note * are site averages to allow observational comparisons of responses.

Changes in fish diversity following riprap stabilization

A total of 637 fish were captured among 16 species identified during the 2013 sampling period, compared to 2803 fish captured among 16 species in 2014 (Table 4.3). Fish sampling revealed 6 rare species, identified as a total of 5 or less for the entire dataset.

The 2 most abundant species were the silvery minnow for 2013 and mimic/sand shiner for 2014

Table 4.3 Summary of fish sampling for 3 sites at the Salvail River.

Note: for the purposes of this study rare species were identified as a total of 5 or less occurrences for the entire dataset, highlighted in red, while the 2 most abundant species were highlighted in blue. Fish less than 5 cm in length were not identified. Johnny and tesselated darters as well as mimic and sand shiners were grouped together due to difficulties in distinguishing between species.

Scientific name	Common name	Number of individuals identified (n)							
		Site 1		Site 2		Control		Total	
		2013	2014	2013	2014	2013	2014	2013	2014
Catostomus commersoni	White sucker	6	-	25	33	8	-	39	33
Cyprinella spiloptera	Spotfin shiner	-	44	7	47	-	81	7	172
Etheostoma olmstedi/nigrum	Johnny/Tesselated darter	6	82	12	34	6	150	24	266
Luxilus cornutus	Common shiner	3	264	19	18	7	88	29	370
Notemigonus crysoleucas	Golden shiner	-	14	4	4	-	7	4	25
N/D	Mimic/Sand shiner	-	272	60	47	-	215	60	534
Perca flavescens	Yellow perch	7	-	5	-	3	-	15	0
Percepsis omiscomaycus	Trout-perch	7	6	38	71	32	38	77	115
Pimephales notatus	Bluntnose minnow	29	184	45	46	35	54	109	284
Semotilus atromaculatus	Creek chub	-	137	3	30	1	47	4	214
Ambloplites rupestris	Rock bass	6	18	-	23	4	12	10	53
Sander vitreus	Walleye	-	-	-	-	2	-	2	0
Percina caprodes	Logperch	-	1	-	7	1	11	1	19
Ameiurus nebulosus	Brown bullhead	-	-	-	-	1	-	1	0
Moxostoma macrolepidotum	Shorthead redhorse	-	-	-	-	1	-	1	0
Micropterus dolomieu	Smallmouth bass	-	-	-	5	-	-	0	5
Moxostoma anisurum	Silver redhorse	-	66	-	73	-	150	0	289
Semotilus corporalis	Fallfish	-	109	-	6	-	60	0	175
Notorus flavus	Stonecat	-	1	-	-	-	1	0	2
Notropis hudsonius	Spottail shiner	-	9	-	1	-	2	0	12
Rhinichthys cataractae	Longnose dance	-	1	-	3	-	-	0	4
Hybognathus nuchalis	Silvery minnow	65	112	87	5	102	114	254	231
Not identified	< 5cm in length	37	666	132	348	148	303	317	1317
	Total ID'ed	**129**	**1320**	**305**	**453**	**203**	**1030**	**637**	**2803**

Species richness was found to be higher in 2014 for all sampled sites with 6 new species identified, however a much greater difference was observed at treatment sites (+7 species), which received stabilization, compared to control C (+2 species) (Figure 4.3), indicating that riprap may have an impact on species richness. The Simpson's Index of Diversity also increased from 2013 to 2014 for all sites (Figure 4.3) although the differences were similar for controls (+ 0.19) and treatments (+0.13) and as such riprap does not appear to be having an impact on this parameter for these sites .

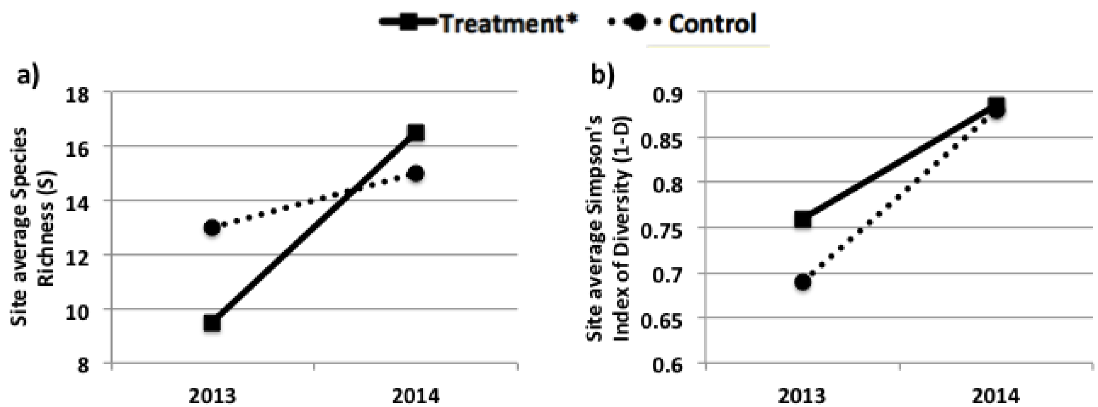


Figure 4.4 Comparison of treatment vs. control for changes pre and post stabilization for fish diversity characteristics a) species richness (S) b) Simpson's Index of Diversity (1-D) note * are site averages to allow observational comparisons of responses. In both cases higher values represent greater diversity.

4.4 Discussion

The state of habitat in the Salvail River and the observed impacts of riprap construction

Overall fish habitat quality metrics QHEI and HMID scored fairly low in 2013 (before stabilization) as expected for a Lowland river in an agricultural setting. Indeed, rivers with high levels of disturbance, influx of fine sediments and the removal of riparian vegetation associated with agricultural practices, typically exhibit poor habitat quality and low diversity (Pitlick and Wilcock, 2001; Yarnell et al., 2006). This point is further illustrated by the results of physical habitat measurements, which indicate that only two main mesohabitat types, pools and glides (both characterized as deeper zones of sluggish flow) dominated the Salvail.

Riffle and run mesohabitats (shallow zones of fast flow) on the contrary, were rare, and those identified were often quite small. Field observations noted that riffle and run formations usually resulted from woody debris accumulations that spanned half of the wetted width or more, forcing the river to adjust around it.

Field assessments revealed that the only type of in-stream cover that was consistently observed at all study sites was woody debris. Accumulations, however, were generally small with the exception of control A, which had a log jam spanning half the wetted width. Low woody debris recruitment is often associated with poor quality, recovering riparian zones (Andrus et al., 1988), which is typical of agricultural areas where farmers attempt to gain more space for their crops by removing mature woody vegetation.

Comparison of 2013 and 2014 results indicate that some important changes in fish habitat metrics have occurred due to riprap bank stabilization in the Salvail River. While there was an overall marked improvement in both habitat quality and diversity at treatment sites 1 and 2 riprap appears to have an impact on pool/glide habitats, which decreased at treatment sites while increasing at controls, and riffle/run habitats, which increased more at treatment sites compared to controls. The changes in woody debris proportions and QHEI scores cannot be attributed with any certainty to riprap as controls showed either similar or greater differences between study years. As was highlighted in chapter 3, observed riprap impacts depend largely on the quality of fish habitat prior to riprap installation (Gidley et al., 2012). The Salvail River being located in the Lowlands with predominantly agricultural land use, it is not surprising that the addition of roughness improved habitat and diversity. The narrowing of the average wetted width as well as the introduction of a new, larger substrate size, associated with riprap treatment at sites 1 and 2 has produced new diversity which is evidenced by the increase in proportion of riffle/run habitat types at experimental sites while remaining low at control sites. The addition of new habitat diversity following riprap was also highlighted by the HMID results, which were sensitive to flow variations around the riprap as well as the faster velocities noted at runs that differed greatly from the rest of the measurements. The new diversity added through riprap stabilization was further captured in the QHEI results, which showed improved overall habitat quality in 2014 compared to 2013 due mostly to increases in sub-category scores related to substrate heterogeneity and mesohabitat diversity. However, categories that evaluated channel morphology and riparian zone decreased after riprapping due to evidence of new erosion on the opposite banks of both experimental sites as well as the complete removal of riparian vegetation during the construction process (Figure 4.2-D). This is particularly alarming, as much riparian vegetation has already been removed in this area. It should be noted that QHEI scores also increased from 2013 to 2014 at two control sites b and c due to higher scores in sub metrics related to mesohabitat diversity and in-stream cover (woody debris) showing that habitat quality can be improved without adding riprap.

Arguably the most interesting change in fish habitat characteristics from 2013 to 2014 is related to the in-stream cover type woody debris, which was hypothesized to decrease as other studies have noted significantly lower large woody debris densities at riprapped reaches when compared to reference reaches (Schmetterling et al., 2001;

Thompson, 2002). It was therefore initially surprising to observe large increases in the proportion of woody debris for control sites B and C with only small differences at stabilized sites 1 and 2. However, the large quantity of woody debris measured at control sites B and C is likely due to increased input during the construction disturbance that completely removed woody riparian vegetation (Figure 4.2). As these sites are located downstream from site 2 they likely received most of that input in wood. Further supporting this theory is the observation that control A, located upstream from site 2, actually showed a decrease in woody debris proportion and was the only site where QHEI score remained unchanged from 2013 to 2014. This was a situation that was not anticipated in the experimental design and site selection, thus control sites B and C cannot be considered true controls, as they are not independent of treatment related to riprapping.

The increase in woody debris proportions also explains the large increase in the proportion of pool habitats at control B and C. Indeed, significant positive correlations between pool area and woody debris volume have been documented (Beechie and Sibley, 1997). However, as the authors note, low slope channels already have pool-forming mechanisms and are consequently less sensitive to large woody debris abundance. Therefore, while the increased woody debris proportions at control sites B and C and resultant pool formation have improved both QHEI and fish habitat quality scores, the improvement may only be a temporary response to the construction disturbance.

Measures of fish diversity and comparison to habitat changes

Fish sampling revealed a surprisingly high level of fish diversity and abundance for such a disturbed river given that most anecdotal evidence collected from local landowners pointed to a disbelief that "anything could be living in that river" because it was largely viewed as a receptacle for agricultural runoff. The post riprap treatment of 2014 samples yielded over 4 times the number of fish compared to 2013. However, species identified varied quite a bit between sampling periods, many of which were rare, totaling less than 5 individuals identified over the entire dataset. While a high number of rare species is expected for fish community structures, especially in warm-water lowland streams that typically exhibit more diverse communities compared to cold-water streams (Gorney et al., 2012), the large differences between species sampled in this study may be attributed to catch method and schooling behavior, leading to hit or miss sampling, rather than changes in habitat choices related to the introduction of riprap.

Both metrics for evaluating biodiversity, that is, Species Richness (S) and Shannon's Index of Diversity (1-D) have increased between pre-stabilization (2013) and post-stabilization (2014) sampling periods for all study sites. Riprap appears to be having a positive impact on (S) in particular as post stabilization measurements increased much more at treatment sites compared to controls. Shannon's Index of Diversity however, does not seem to be affected by riprap for these sites as both treatment and control sites showed similar differences. These results are in agreement with those of fish habitat metrics, which revealed increases in habitat diversity and quality. Results also indicate that not only did the number of species increase post stabilization, but so did their abundance. As Cianfrani (2009) notes, species evenness is a key factor to consider when evaluating the health and stability of overall fish communities. Other studies have noted positive effects of riprap in already degraded streams, on metrics such as fish abundance (Lister et al., 1995), which is likely due to the stable habitat it provides (Gidley et al., 2012). However, while some studies have noted negative impacts of riprap on fish, mainly in terms of decreased abundance at riprapped sites (Chapman and Knudsen, 1980; Knudsen and Dilley, 1987; Garland et al., 2002), such results were attributed to losses in habitat, which were not observed in the Salvail sampling sites. Indeed, several studies have already documented the positive correlation between habitat heterogeneity/diversity and fish species diversity (Schlosser, 1982; Cianfrani, 2009; Pendersen et al., 2014), further supporting the ecological hypothesis that rivers with more diverse habitats often support a greater biological diversity (Newson and Newson, 2000). It is therefore quite possible that the increased heterogeneity of habitat measurements found at riprapped sites 1 and 2 as well as the increase of more rare habitat patches (riffle/runs) is positively affecting local fish diversity.

It remains unclear as to whether the observed increase in fish diversity metrics at the Salvail River is due to riprapping or simply the inter-annual variability of fish assemblages, since the control site responded with similar increases. The causes may also be different, for instance, evidence exists which directly highlights the positive effects of flow heterogeneity caused by riprap stabilization on fish metrics (Asselin, 2016). Alternatively, other studies have shown the significant positive effect woody debris can have on fish habitat choices and increasing their abundance (Gatz, 2008). Therefore, as the control site was over 500m downstream from treatment site 1 it is hypothesized that the large woody debris input due to the construction disturbance, as found in increased woody debris proportions at nearby control B and C for habitat sampling, likely affected habitat conditions and therefore fish diversity. Since treatment sites 1 and 2 experienced increases in fish diversity metrics without large changes in woody debris proportions compared to controls B and C, such increases may be attributed, at least partially, to the riprap.

Unfortunately, as no control for fish sampling was established upstream from the construction, this remains speculative.

Extensive riprap stabilization for Salvail River tributaries

While impacts of riprap on fish habitat remain somewhat uncertain in the Salvail's main channel, some dramatic effects can be seen in several of its tributaries. At least three of the Salvail's tributaries received complete riprap stabilization (both banks and bed) over lengths of more than 600m. The streambeds in these cases were also raised by over a meter, resulting in large sections being dry over several months in the year as shown in a side study (Figure 4.4). This constitutes an extreme loss of fish habitat, as these are not naturally ephemeral streams, but the infiltration of water through interstitial space between rocks led to water moving under the surface over significant portion of the stabilized streams. Clearly a better understanding of how streams respond to riprap over long distances is required as fish habitat should be available all year round. It is also important to note that the environmental assessment of the impacts of the stabilization on fish habitat in these tributaries only highlighted potential risk associated with high turbidity during construction work, but did not raise the highly problematic situation of the streams becoming dry part of the year. As shown above, riprap can have positive effects in highly disturbed environments, but not when used to raise bed level over hundreds of meters.

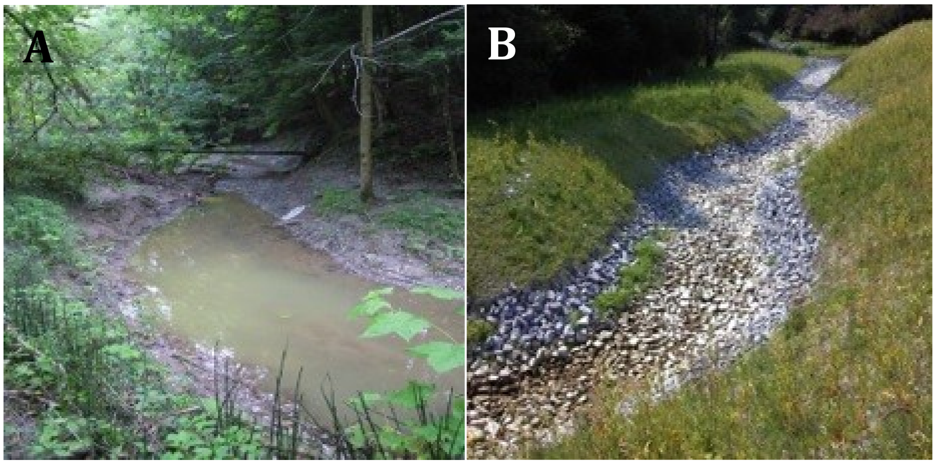


Figure 4.5 Tributary 5 of the Salvail River – A) upstream natural state B) downstream post riprap work. Both photos were taken in July 2014.

4.5 Conclusion

Fish habitat assessments revealed several clear changes after riprapping, which may constitute an improvement in terms of quality and diversity. However, as the Salvail was already fairly disturbed and evaluated as having overall poor quality habitat, it is unlikely for riprap to have similar positive impacts in less perturbed environments. It is also unclear as to whether the increase in riffle/run habitats observed post-riprapping is actually important for fish species living in this area or simply represents an alternative habitat choice. Furthermore, while the different fish habitat assessments were in agreement overall, they tended to lack precision, especially when applied in areas of frequent disturbance as the consistent identification of degraded habitats can be difficult. It is therefore recommended to adapt assessments to the regional conditions of the study area as well as to incorporate the specifics habitat needs of local fish species.

Although fish sampling did reveal increases in species richness and diversity after riprapping, the lack of control upstream makes it difficult to conclude whether differences are due to riprap impacts or the disturbances created from the construction process. This situation does highlight, however, that riprap construction may have important downstream effects (500m+), particularly when large amounts of woody debris are removed from the riparian zone.

Regardless of the impact of riprap it is noteworthy that there was consistent agreement between increased fish habitat diversity/quality and species diversity found at all study sites. This may be especially important from an environmental management perspective, as fish habitat assessments are considerably more time/cost efficient than fish sampling. Thus, when responding to environmental emergencies such as the 2010 landslides at the Salvail River, where time is extremely limited, quick assessments like the QHEI may prove to be a robust measure of fish habitat quality.

Finally, observations of the effects of complete riprapping for 100m+ of several Salvail tributaries demonstrate an urgent need to better understand how streams respond to riprap over long distances. Unlike the stabilization in the main Salvail River where riprap was installed at the former bed elevation, the approach used in the tributaries involved adding riprap on top of the natural bed. This should clearly be avoided in the future as it is very likely that the problem of drying during the summer months would occur in other contexts than the Salvail region. Future studies should investigate the year-round availability of fish habitat in heavily stabilized reaches.

Chapter 5. General conclusions

Through the combination of quantitative and qualitative approaches to fish habitat assessment this study has identified significant differences in fish habitat parameters between stabilized and non-stabilized reaches, as well as important changes in fish and fish habitat diversity after riprap treatment. Findings indicated that small extents of riprap such as those found at bridge/stream crossings tended to offer habitats with less in-stream cover and riparian vegetation, which may be considered as negative for fish habitat. Positive habitat trends were also noted, associated with riprapped reaches through the combination of increased slope and coarser substrates leading to increased flow heterogeneity. However, positive impacts were only observed in streams with already poor habitat and the use of riprap is not recommended as a habitat rehabilitation strategy. Indeed,results of habitat assessments varied greatly between Appalachian and Lowland sites due to their fundamentally different baseline habitat quality, demonstrating the importance of considering the reference state of fish habitat quality when attempting to interpret riprap impacts. Variability in riprap impacts can also be explained by other geomorphological factors such as whether the stream is straightened or not as well as the habitat assessment chosen (HMID versus QHEI), each of which have their limits in terms of precision.

BACI experimentation for two sites at the Salvail River revealed that riprap may be contributing to increased habitat heterogeneity and quality. However, reference habitat quality was low and it is unclear if these changes are biologically important. Furthermore, while there was observed increase in fish diversity which agrees with fish habitat trends, it is difficult to conclude whether the increase is due to riprap or instead to the increased woody debris proportions which are likely related to the construction disturbance.

Accurately measuring fish habitat is a complex endeavor due primarily to the large variability in habitat conditions that often differ between regions. Furthermore, the frequent anthropogenic disturbances in many Lowland watersheds can make the establishment of control sites quite difficult, as was evident in the Salvail study. As a result, fish habitat conditions are very site specific, therefore habitat assessment protocols which measure quality in a holistic way (QHEI) or measure few parameters (HMID) often lack precision. While some of the results discussed in this thesis have found such measures to be fairly robust for evaluating fish habitats at sites in the same stream, these assessments may yield conflicting results when the morphological context changes. This explains why attempts to draw general conclusions about highly empirical data which incorporates potentially low precision fish habitat assessments often end in conflicting information concerning the impacts of riprap on fish habitat.

It is clear however that riprap does significantly alter local channel flow conditions, which may have implications for its long-term maintenance and biological life. It is therefore important that more care is taken to measure fish habitat data in order to evaluate conditions prior to riprap stabilization. This is particularly relevant when many extensive projects are scheduled in the same river network in order to better understand how these systems respond to potential cumulative impacts.

References

Allan, J. D., & Castillo, M. M. (2007). *Stream ecology: Structure and function of running waters*. Dordrecht: Springer.

Andrus, C. W., Long, B. A., Froehlich, H. A. (1988). Woody debris and its contribution to pool formation in a coastal stream 50 years after logging. *Canadian Journal of Fisheries and Aquatic Sciences, 45(12)*, 2080-2086.

Angradi, T. R., Schweiger, E. W., Bolgrien, D. W., Ismert, P., Selle, T. (2004). Bank stabilization, riparian land use and the distribution of large woody debris in a regulated reach of the upper Missouri river, North Dakota, USA. *River Research and Applications*, *20*, 829-846.

Armantrout, N. B. (1998). *Glossary of Aquatic Habitat Inventory Terminology.* Bethesda, Md, USA: American Fisheries Society.

Asselin J. (2016). L'effet des enrochements sur l'utilisation de l'espace par les communautés de poissons dans les Basses-Terres du St-Laurent. Unpublished MSc thesis, Département des Sciences biologiques, Université de Montréal, 83 p.

Avery, E. L. (1995). Effects of streambank riprapping on physical features and brown trout standing stocks in Millville Creek. Wisconsin Department of Natural Resources Research Report 167.

Bain, M. B., Haifang, J. (2012). A habitat model for fish communities in large streams and small rivers. *International Journal of Ecology*, 2012, 1-8.

Barbour, M.T., Gerritsen, J., Snyder, B. D., Stribling, J. B. (1999). *Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic*

Macroinvertebrates and Fish, Second Edition. EPA 841-B-99-002. Washington, D.C.:U.S. Environmental Protection Agency; Office of Water;

Berryman, D. (2008). État de l'écosystème aquatique du bassin versant de la rivière Yamaska : faits saillants 2004-2006, Québec, ministère du Développement durable, de l'Environnement et des Parcs, Direction du suivi de l'état de l'environnement, ISBN 978-2-550-53592-8 (PDF), 22 pages

Beschta, R. L. and Platts, W. S. (1986). Morphological features of small streams: significance and function. *Water Resources Bulletin: American Water Resources Association*, *22(3)*, 369-379.

Biedenharn, D. S., Elliott, C. M., Watson, C. C. (1997). *The WES stream investigation and streambank stabilization handbook.* Vicksburg, Mississippi: U.S. Army Engineer Waterways Experiment Station (WES).

Binns, N. A., and Eiserman, F. M. (1979). Quantification of fluvial trout habitat in Wyoming. *Transactions of the American Fisheries Society*, *108*, 215-228.

Bisson, P. A., Nielsen, J. L., Palmason, R. A., Grove, L. E. (1982). A system of naming habitat types in small streams, with examples of habitat utilisation by salmonids during low stream flow. In *Acquisition and Utilisation of AquaticHabitat Information Proceedings of a symposium held 28-30 October, 1981, Portland, Oregon,* Armandtrout NB (ed). Western Division of American Fisheries Society, Bethesda, Maryland; 62–73.

Beechie, T. J., Sibley, T. H. (1997). Relationships between channel characteristics, woody debris, and fish habitat in Northwestern Washington streams. *Transactions of the American Fisheries Society*. *126(2),* 217-229.

Blanchet, F. G., Legendre, P., Borcard, D. (2008). Forward selection of explanatory variables. *Ecology, 89*, 2623-2632.

Blum, M. D., and Tornquist, T. E. (2000). Fluvial response to climate and sea-level change: a review and look forward. *Sedimentology*, *47*, 2–48

Bouska, W. W., Keane, T., and Paukert, C. P. (2010). The effects of road crossings on prairie streamhabitat and function. *Journal of Freshwater Ecology*, *25(4),* 499-506.

Breder, C. M. and Rosen, D. E. (1966). *Modes of reproduction in fishes.* Neptune City, New Jersey: T.F.H. Publications. 941p. Web.

Brown, B. L. (2007). Habitat heterogeneity and disturbance influence patterns of community temporal variability in a small temperate stream. *Hydrobiologia*, *586(1)*, 93-106.

Button, K. S., Ionnidis, J., Mokrysz, C., Nosek, B.A., Flint, J., Robinson, E., Munafò, M.R. (2013). Power failure: why small sample size undermines the reliability of neuroscience. *Nature Reviews Neuroscience*, 14**,** 365-376.

Cardinale, B. J., Palmer, M. A., Swan, C. M., Brooks, S., & LeRoy Poff, N. (2002). The influence of substrate heterogeneity on biofilm metabolism in a stream ecosystem. *Ecology*, *83(2),* 412-422.

CEN. (2002). A guidance standard for assessing the hydromorphological features of rivers. CEN TC 230/WG2/TG 5: N32. 21 pp.

Chapman, D. W., Knudsen, E. (1980). Channelization and livestock impacts on salmonid habitat and biomass in Western Washington. *Transactions of the American Fisheries Society*, *109*, 357-363.

Chou, W. C., Chuang, M. D. (2011). Habitat evaluation using suitability index and habitat type diversity: a case study involving a shallow forest stream in central Taiwan. *Environmental Monitoring Assessment, 172*: 689-704.

Church, M., Biron, P. M., Roy, A. G. (Eds.). (2012). *Gravel-bed Rivers: Processes, Tool, Environments*. Hoboken, N.J.: Wiley.

Cianfrani, C. M., Sullivan, S., Hession, W. C., Watzin, M. C. (2009). Mixed stream channel morphologies: implicaitons for fish community diversity. *Aquatic Conservation: Marine and Freshwater Ecosystems, 19*, 147-156.

Comité de concertation et de valorisation du bassin de la Rivière Richelieu (COVABAR). (2013a). Raport Annuel 2012-2013. Retrieved January 1, 2014 from: http://www.covabar.qc.ca/documents/accueil/Rapport%20annuel%20202 2013%20final.pdf

Comité de concertation et de valorisation du bassin de la Rivière Richelieu (COVABAR). (2013b). Rapport de caractérisation des cours d'eau du bassin versant du ruisseau Coderre. 52 pages.

Craig, A. J., and A. V. Zale. (2001). Effects of bank stabilization structures on fish and their habitat - A literature review. US Geological Survey, Washington Department of Ecology, and Montana State University. Bozeman, MT.

Dardeau, T. K., Killgore, J., Miller, A. (1995). Using riprap to create or improve riverine Habitat. In *River, Coastal and Shoreline Protection: Erosion Control Using Riprap and Armourstone*, Thorne CR, Abt SR, Barends FBJ, Maynord ST, Pilarczyk KW (eds). Wiley, New York; 609-620.

Duke Energy. (2014). Stabilization techniques. Retrieved January 1, 2014 from: http://www.duke-energy.com/shoreline-stabilization/stabilization techniques.asp

Farabee, G. B. (1986). Fish species associated with revetted and natural main channel border habitats in pool 24 of the Upper Mississippi River. *North American Journal of Fisheries Management, 6*, 504-508.

Federal Emergency Management Agency (FEMA). (2009). Engineering with nature: alternative techniques to riprap bank stabilization. Retrieved January 1, 2014from: http://www.fema.gov/pdf/about/regions/regionx/Engineering_With_Natu e_Web.pdf

Ferguson, R. I. (1981). Channel forms and channel changes. In Lewin, J. (Ed.) London : British Rivers. Allen and Unwin, pp.90-125

Fernández, D., Barquín, J., Raven, P. J. (2011). A review of river habitat characterisation methods: indices vs. characterisation protocols. *Limnetica, 30(2)*: 217-234.

Fischenich, J. C. (2003). Effects of riprap on riverine and riparian ecosystems, ERDCEL TR03-4, U.S. Army Engineer Research and Development Center, Vicksburg, MS.

Florsheim, J. L., Mount, J. F., Chin, A. (2008). Bank erosion as a desirable attribute of rivers. *Bioscience, 58(6),* 519–529.

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

Garland, D., Tiffan, K. F., Rondorf, D. W., Clark, L. O. (2002). Comparison of subyearling fall Chinook salmon's use of riprap revetments and unaltered habitats in Lake Wallula of the Columbia River. *North American Journal of Fisheries Management*, *22*, 1283 1289.

Gatz, A. J., (2008). The use of floating overhead cover by warmwater stream fishes. *Hydrobiologia, 600*, 307-310.

Gazengam, E., Gharabaghi, B., Jones, F. C., Whiteley, H. (2011). Evaluation of the Qualitative Habitat Evaluation Index as a Planning and Design Tool for Restoration of Rural Ontario Waterways. *Canadian Water Resources Journal / Revue canadienne des ressources hydriques*, *36*, 149-158.

Gidley, C. A., Scarnecchia, D. L., & Hansen, J. A. (2012). Fish community structure associated with stabilized and unstabilized shoreline habitats, Coeur d'Alene River, Idaho, *USA. River Research and Applications*, *28(5),* 554-566.

Good, P. I., Hardin, J. W. (2006). *Common errors in statistics (and how to avoid them)*. Hoboken, New Jersey: John Wiley & Sons.

Gorman, R., Karr, J. (1978). Habitat structure and stream fish communities. *Ecology, 59(3),* 507-515.

Gorney, R. M., Williams, M. G., Ferris, D. R., & Williams, L. R. (2012). The influence of channelization on fish communities in an agricultural Coldwater stream system. *American Midland Naturalist*, *168(1),* 132-143.

Gostner, W., Alp, M., Schleiss, A. J., Robinson, C. T. (2012). The hydro-morphological index of diversity: a tool for describing habitat heterogeneity in river engineering projects. *Hydrobiologia*, *712*, 43-60.

Government of Canada Justice Laws Website. (January 30, 2014). The Fisheries Act. Retrieved from http://laws-lois.justice.gc.ca

Grift, R. E., Buijse, A. D., Van, Densen, W. L. T., Machiels, M. A. M., Dranenbarg, J., Klein, Breteler, J. G. P., Backx, J. J. G. M. (2003). Suitable habitats for 0-group fish in rehabilitated floodplains along the lower River Rhine. *River Research and Applications, 19*: 353–374.

Hassan, M. A., Schick, A. P., Shaw, P. (1995). Movement of pebbles on a sand bed river, Botswanna. Proceedings: Application of Tracers in Arid Zone *Hydrology*. IASH, 232.

Hauer, C., Unfer, G., Tritthart, M., Formann, E., Habersack, H. (2011). Variability of mesohabitat characteristics in riffle-pool reaches: testing an integrative evaluation concept (fgc) for MEM-application. *River Research and Applications, 27*, 403-430.

Hession, W. C., Pizzuto, J. E., Johnson, T. E., Horwitz, R. J. (2003). Influence of bank vegetation on channel morphology in rural and urban watersheds. *Geology*, *31*, 147 151.

Hickford, M. J. H., Schiel, D. R. (2011). Synergistic interactions within disturbed habitats between temperature, relative humidity and UVB radiation on egg survival in a diadromous fish. *PLos ONE*, *6(9),* e24318.

Hickin, E. J. (1984). Vegetation and river channel dynamics. *The Canadian Geographer*, *28*, 111-126.

Hunt, R. L. (1988). *A compendium of 45 trout stream habitat development evaluations in Wisconsin during 1953-1985*. Technical Bulletin 162. Wisconsin Department of Natural Resources, Madison, Wisconsin.

Jafarnejad, M., Mário, F., Pfister, M., Schleiss, A. (2017) Time-based failure analysis of compressed riverbank riprap. *Journal of Hydraulic Research, 55(2)*, 224-235.

Kaufmann, P. R., Levine, P., Robison, E. G., Seeliger, C., Peck, D. V. (1999). Quantifying Physical Habitat in Wadeable Streams. EPA/620/R-99/003. U.S. Environmental Protection Agency, Washington, D.C.

Kershner, J. L., Snider, W. M. (1992). Importance of a habitat-level classification system to design instream flow studies. In *River Conservation and Management,* Boon PJ, Calow P, Petts GE (eds). Chichester: Wiley 179-193.

Kimball, L. C., Kondolf, G. M. (2002). *Analysis of channel geomorphology and habitat forming processes for feasibility assessment of rip-rap removal, Muir Woods National Monument Mill Valley, California. Final Report.* Prepared for The National Park Service, Muir Woods National Monument, and the Golden Gate National Recreation Area. University of California, Berkeley, Department of Landscape Architecture.

Knighton, D. (1998). *Fluvial forms and processes: a new perspective*. New York, NY: Arnold.

Knudsen, E. E., and S. J. Dilley. (1987). Effects of riprap bank reinforcement on juvenile salmonids in four western Washington streams. *North American Journal of Fisheries Management*, *7*, 351-356.

Kondolf, G. M. (2011). Setting goals in river restoration: When and where can the river "Heal itself"? In *Stream restoration in dynamic fluvial systems: Scientific approaches, analyses, and tools. 194*, 29-43. Washington, DC: AGU

L'Association des transports du Canada (ATC). (2005). *Manuel de conception hydraulique des ponts*. Gouvernement du Québec.

Lau, J. K., Lauer, T. E., Weinmann, M. L. (2006). Impacts of channelization on stream habitats and associated fish assemblages in east central Indiana. *American Midland Naturalist*, *156*, 319-330.

Legendre P, Legendre L. (2012). *Numerical Ecology, 3rd English edition*. Amsterdam: Elsevier Science BV.

Les Services exp inc. (2013). Stabilisation du talus de la rivière Salvail, site 2, 3 et 21: demande concernant les travaux urgents à soustraire de la procedure d'évaluation et d'examen des impacts sur l'environnement.

Li, H. W., Schreck, C. B., Tubb, R. A. (1984). *Comparison of habitats near spur dikes, continuous revetments, and natural banks for larval, juvenile, and adult fishes of the Willamette River*. Oregon Cooperative Fishery Research Unit, Oregon State University, Water Resources Research Institute, Project 373905, Contract 14 08-001-G-864, Technical report, Corvallis.

Lister, D. B., Beniston, R. J., Kellerhals, R., Miles, M. (1995). Rock size affects juvenile salmonid use of streambank riprap. In River, Coastal and Shoreline Protection: Erosion Control Using Riprap and Armourstone, Throne CR, Abt SR, Barends FBJ, Maynard ST, Pilarczyk KW (eds). New York: John Wiley and Sons, Ltd. 621–634.

Maddock, I. (1999). The importance of physical habitat assessment for evaluating river health. *Freshwater Biology, 41*: 373-391.

Matthews, W. J., Hill, L. G. (1979). Influence of physico-chemical factors on habitat selection by red shiners, Notropis lutrensis (Pisces: Cyprinidae). Copeia 1979: 70 81.

Mendes, R. S., Evangelista, L. R., Thomaz, S. M., Agostinho, A. A., Gomes, L. C. (2008). A unified index to measure ecological diversity and species rarity, *Ecography 31*, 450 456.

Meyer, J.L., Strayer, D.L., Wallace, J.B., Eggert, S.L., Helfman, G.S., Leonard, N.E. (2007). The contribution of headwater streams to biodiversity in river networks. *Journal of the American water resources association*, 43, 86-103.

Moerke, A. H., Lamberti, G. A. (2003). Responses in fish community structure to restoration of two Indiana streams. *North American Journal of Fisheries Management*, 23, 748-759.

Millar, R .G. (2000). Influence of bank vegetation on alluvial channel patterns. *Water Resource Research*, *39*, 1109-1118.

Ministère du Développement Durable, de L'Environnement et des Parcs. (2013). Rapport d'analyse environnementale de la demande de soustraction à la procédure d'évaluation et d'examen des impacts sur l'environnement du projet de stabilisation du talus de la rivière Salvail, sur le territoire de la municipalité de Saint-Jude par la Municipalité de Saint-Jude.

Newbury, R. W. (2013) Designing fish-passable riffles as gradient controls in Canadian streams. *Canadian Water Resources Journal, 38(3),* 232-250.

Organisme de bassin versant de la baie Missisquoi (OBVBM). (2010). Protecting the quality of life of lakes and watercourses. [Bedford, Québec]: Shoreline Management Guide 2010 edition.

Newson, M. D., and Newson, C. L., (2000). Geomorphology, ecology and river channel habitat: mesoscale approaches to basin-scale challenges. *Progress in Physical Geography,4(2),* 195-217.

Nunn A. D., Harvey, J. P., Cowx, I. G. (2007). Benefits to 0+ fishes of connecting man made waterbodies to the lower River Trent, England. *River Research and Applications, 23*, 361–376.

Page, L. M. and Burr, B. M. (1991). *A field guide to freshwater fishes of North America north of Mexico.* Boston: Houghton Mifflin Company. 432p. Web.

Parasiewicz P. (2001). MesoHABSIM: A concept for application of instream flow models in river restoration planning. *Fisheries, 26*: 6-13.

Pedersen, M. L., Kristensen, K. K., Friberg, N. (2014). Re-Meandering of Lowlands Streams: Will Disobeying the Laws of Geomorphology Have Ecological Consequences? *PLoS ONE* 9: e108558. doi:10.1371/journal.pone.0108558

Pennington, C. H., Baker, J. A., Bond, C. L. (1983). *Fishes of selected aquatic habitats on the Lower Mississippi River.* Technical ReportE -83-2, U.S. ArmyC orps of EngineersW aterwayEs xp. Stn., Vicksburg.

Peters, R. J., Missildines, B. R., and D.L. Low. (1998). Seasonal fish densities near river banks stabilized with varoius stabilization methods; first year report of the flood technical assistance project. USFWS. Lacey, WA; 34 pp.

Pitlick, J., Wilcock, P. (2001). Relations between streamflow, sediment transport, and aquatic habitat in regulated rivers. *Geomorphic processes and riverine habitat: water scienceand application, 4*, 185-198.

Platts, W. S., Megahan, W. F., Minshall, W. G. (1983). *Methods for evaluating stream, riparian, and biotic conditions,* General Technical Report INT-138, USDA, Forest Service, Rocky Mountain Research Station, Ogden, UT.

Price, D. J., Birge, W. J. (2005). Effectiveness of stream restoration following highway reconstruction projects on two freshwater streams in Kentucky. *Ecological Engineering*, *25*, 73-84.

Quigley, J. T., Harper, D. J. (2004). *Streambank protection with rip-rap: an evaluation of the effects on fish and fish habitat*. Canadian Manuscript Report of Fisheries and Aquatic Sciences 2701.

R Development Core Team. (2010). R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3 900051-07-0, http://www.R-project.org

Rankin, E. T. (1989). The quantitative habitat evaluation index [QHEI]: rational, methods, and application. State of Ohio Environmental Protection Agency.

Rankin, E. T. (2006). *Methods for assessing habitat in flowing waters: using the qualitative habitat evaluation index (QHEI)*. OHIO EPA Technical Bulletin EAS. State of Ohio Environmental Protection Agency. Division of Surface Water.

Reid, D., Church, M. (2015). Geomorphic and Ecological Consequences of Riprap Placement in River Systems. *Journal of the American Water Resources Association*, 51, 1043-1059.

Rhoads, B. L., Schwartz, J. S., Porter, S. (2003). Stream geomorphology, bank vegetation, and three-dimensional habitat hydraulics for fish in Midwestern agricultural streams. *Water Resources Research*, 39, 1218-1231.

Richardson, E. V., Simons, D. B., Lagasse, P. F. (2001). *River engineering for highway encroachments – highways in the river environment*. Publication No. FHWA NHI 01-004, U.S. Department of Transportation, Federal Highway Administration, National Highway Institute, Washington, DC.

Ridenour, C. J., Starostka, A. B., Doyle, W. J., Hill, T. D. (2009). Habitat used by Macrhybopsis chubs associated with channel modifying structures in a large regulated river: implications for river modification. *River Research and Applications, 25*: 472-485.

Ruesch, A. S., Torgensen, C. E., Lawler, J. J., Olden, J. D., Peterson, E. E., Volk, C. J. & Lawrence, D. J. (2012). Projected Climate-Induced habitat Loss for Salmonids in the John Day River Network, Oregon, U.S.A. *Conservation Biology*, *26*, 873-882.

Schmetterling, D. A., Clancy, C. G., Brandt, T. M. (2001). Effects of riprap bank reinforcementon stream salmonids in the western United States. *Fisheries*, *26(7),* 1460-1467.

Sear, D. A. (1996). The sediment system and channel stability. In: Brookes, A. and Shields, F.D. Jr. (Eds) River Channel Restoration: Guiding Principles for Sustainable Projects, John Wiley & Sons, 149-177

Shields, F. D., Cooper, C. M. & Testa, S. (1995). Towards greener riprap: environmental considerations from microscale to macroscale. Retrieved January 1, 2014,from

http://or.ducks.org/media/conservation/mtproject/MTData/Workshop_3/

Infomation/scientific_abstracts.pdf

Shields, F. D., Knight, S. S., Morin, N., Blank, J. (2003). Response of fishes and aquatic habitants to sand-bed stream restoration using large woody debris. *Hydrobiologia*, *494*, 251-257.

Simoneau, M. and Thibault, G. (2009). État de l'écosystème aquatique du bassin versant de la rivière Richelieu : faits saillants 2005-2007, Québec, Ministère du Développement durable, de l'Environnement et des Parcs, Direction du suivi de l'état de l'environnement, ISBN 978-2-550-56454-6 (PDF), 23 p.

Smith, H. G. and Dragovich, D. (2008) Sediment budget analysis in slope-channel coupling and in-channel sediment storage in an upland catchment, Southeastern Australia. *Geomorphology*, *101*, 643-654

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

Thompson, D. M. (2002). Long-term effect of instream habitat-improvement structures on channel morphology along the Blackledge and Salmon Rivers, Connecticut, USA. *Environmental Management, 29*: 250–265.

Transports Québec. (2011). Glissement de terrain du 10 mai 2010. Rapport sur les caractéristiques et les causes. Saint-Jude, Montérégie. Rapport MT11-01.

U.S. Department of Transportation Federal Highway Administration. (1989). Design of riprap revetment. Retrieved January 1, 2014 from: http://www.fhwa.dot.gov/engineering/hydraulics/pubs/hec/hec11sI.pdf

White, K., Gerken, J., Paukert, C., Makinster, A. (2010). Fish community structure in natural and engineered habitats in the Kansas river. *River Research and Applications*, *26*, 797-805.

Yan, G. J., He, X. K., Cao, Z. D., Fu, S. J. (2013). An interspecific comparison between morphology and swimming performance in cyprinids. *Journal of Evolutionary Biology*, *26*, 1802-1815.

Yarnell, S. M., Mount, J. F., & Larsen, E. W. (2006). The influence of relative sediment supply on riverine habitat heterogeneity. *Geomorphology*, *80(3-4),* 310-324.

Zuur, F. Z., Ieno, E. N., Smith, G. M. (2007). *Analysing Ecological Data*. New York: Springer.

OhioEPA

Modified version

Qualitative Habitat Evaluation Index And Reach Characterization Field Sheet

QHEI Score: Maximum 91

Stream: ___________________________________ **Site length:** _____ **(m)** **Date:** _ _ / _ _ / 13

Location (proximity to local benchmark)**:** ____

Scorers Name: _________________________ **[GPS] N:** _ _ ° _ _ ' _ _ . _ " **W:** _ _ ° _ _ ' _ _ . _ "

1] Substrate (check only 2 type boxes; estimate % or note every type present in pools and riffles).

Best Types	Pool	Riffle	Best Types	Pool	Riffle	Origin	Quality
□ □ BLDR / SLABS [10]	___	___	□ □ Hardpan [4]	___	___	□ Limestone [1]	□ Heavy [-2] (Silt)
□ □ Boulder [9]	___	___	□ □ Detritus [3]	___	___	□ Tills [1]	□ Moderate [-1] (Silt)
□ □ Cobble [8]	___	___	□ □ Muck [2]	___	___	□ Wetlands [0]	□ Normal [0] (Silt)
□ □ Gravel [7]	___	___	□ □ Silt [2]	___	___	□ Hardpan [0]	□ Free [1] (Silt)
□ □ Sand [6]	___	___	□ □ Artificial [0]	___	___	□ Sandstone [0]	□ Extensive [-2] (Embeddedness)
□ □ Bedrock [5]	___	___				□ Lacusturine [0]	□ Moderate [-1] (Embeddedness)
						□ Shale [-1]	□ Normal [0] (Embeddedness)
						□ Coal Fines [-2]	□ None [1] (Embeddedness)

Number of best types: □ **4 or more [2]** □ **3 or less [0]**

*Treat Riprap as a natural substrate (not artificial); ignore sludge from point-sources.

Comments:

Substrate Maximum 20

2] Instream Cover (check if present)

- □ **Undercut Banks [1]**
- □ **Overhanging Vegetation [1]**
- □ **Shallows (in slow water) [1]**
- □ **Rootmats [1]**
- □ **Logs or Woody Debris [1]**
- □ **Pools > 70 cm [2]**
- □ **Rootwads [1]**
- □ **Boulders [1]**
- □ **Oxbows, Backwaters [1]**
- □ **Aquatic Macrophytes [1]**

Amount (check one or average two)

- □ **Extensive > 75% [11]**
- □ **Moderate 25 - 75% [7]**
- □ **Sparse 5 - < 25% [3]**
- □ **Nearly Absent < 5% [1]**

Comments:

Cover Maximum 20

3] Channel Morphology

(check one in each category or two and average)

Sinuosity	Development	Dynamic Stability
□ High [4]	□ Excellent [7]	□ High [3]
□ Moderate [3]	□ Good [5]	□ Moderate [2]
□ Low [2]	□ Fair [3]	□ Low [1]
□ None [1]	□ Poor [1]	

Comments:

Channel Maximum 14

4] Riparian Zone (check one in each category for each bank or two per bank and average)

Riparian Width (L R)	Flood Plain Quality (L R)	(L R)
□ □ Wide > 50 m [4]	□ □ Forest, Swamp [3]	□ □ Conservation Tillage [1]
□ □ Moderate 10-50 m [3]	□ □ Shrub or Old Field [2]	□ □ Urban or Industrial [0]
□ □ Narrow 5-10 m [2]	□ □ Residential, Park, New Field [1]	□ □ Mining / Construction [0]
□ □ Very Narrow < 5 m [1]	□ □ Fenced Pasture [1]	
□ □ None [0]	□ □ Open Pasture, Rowcrop [0]	

Comments:

Riparian Maximum 7

5] Gradient (%)

Habitat composition ➡ **% Pool:** ___ **% Glide:** ___ **% Run:** ___ **% Riffle:** ___

Stream Width (m)

*See table 2 in manual for scoring

Comments:

Gradient Maximum 10

6] Pool / Glide and Riffle / Run Quality

Maximum Depth
check only one
- □ **> 1 m [6]**
- □ **0.7 - < 1 m [4]**
- □ **0.4 - < 0.7 m [2]**
- □ **0.2 - < 0.4 m [1]**
- □ **< 0.2 m [0]**

Channel Width
check one or 2 and average
- □ **Pool Width > Riffle Width [2]**
- □ **Pool Width = Riffle Width [1]***
- □ **Pool Width < Riffle Width [0]**

Current Velocity
check all that apply
- □ **Torrential [-1]**
- □ **Very Fast [1]**
- □ **Fast [1]**
- □ **Moderate [1]**
- □ **Slow [1]**
- □ **Interstitial [-1]**
- □ **Intermittent [-2]**
- □ **Eddies [1]**

Comments:

*Note: in reaches where no pool or riffle is present pool width is assumed to = riffle width.

Pool / Current Maximum 12

Indicate for functional riffles (depth > 5cm else metric = 0); check one or 2 and average

Riffle Depth
of largest riffle
- □ **> 10 cm [2]**
- □ **5 – 10 cm [1]**
- □ **< 5cm [metric = 0]**

Run Depth
- □ **Maximum > 50 cm [2]**
- □ **Maximum < 50 cm [1]**

Riffle / Run Substrate
- □ **Stable (e.g., Cobble, Boulder) [2]**
- □ **Mod. Stable (e.g., Large Gravel) [1]**
- □ **Unstable (e.g., Fine Gravel, Sand) [0]**

Riffle / Run Embeddedness
- □ **None [2]**
- □ **Low [1]**
- □ **Moderate [0]**
- □ **Extensive [-1]**

Comments:

Riffle / Run Maximum 8

Sampled Reach (check all that apply)

Y N
Is Riprap stabilization present (if yes please complete Riprap assessment field sheet) □ □

Stage
- □ **High**
- □ **Up**
- □ **Normal**
- □ **Low**
- □ **Dry**

Or Mean Depth ____ cm

Clarity
- □ **< 20 cm**
- □ **20 - < 40 cm**
- □ **40 – 70 cm**
- □ **> 70 cm**

Secchi Depth ____ cm

Canopy
- □ **> 85% - Open**
- □ **55% - < 85%**
- □ **30% - < 55%**
- □ **10% - < 30%**
- □ **< 10% - Closed**

Comments (Reach typical of stream?; Access directions, concerns, etc.)**:**

Stream Drawing (show flow direction, landmarks, relative location of mesohabitats, POV and code for photographs, riprap if any, log jams, areas of significant erosion etc.)**:**

Appendix B

Below are the results for the Qualitative Habitat Evaluation Index training day. Seven individuals assessed the same 4 stream reaches and the final scores were compared. In order to limit subjectivity of the assessment a maximum difference of 10% was allowed. Most of the variation in scores is related to difficulties associated with the consistent identification of functional in-stream cover types.

Site	Lowest score	Highest score	Difference	% difference from max score (91)
16602 Reference	36	53	17	18.7
16602 Stabilized	59	67.75	8.75	9.6
17132 Reference	42	50	8	8.8
17132 Stabilized	44	52	8	8.8

Note: scores in red are deemed unacceptable while scores in green demonstrate that scores fell in the acceptable range.

Example of score breakdown per evaluator for sub-category in-stream cover: