Assessing the Effectiveness of Instream Structures for Restoring Salmonid Streams

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ABSTRACT

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Stream restoration is a billion dollar industry in North America. Despite this expenditure there remain questions regarding the effectiveness of current techniques, such as instream structures. The objectives of this research were to assess the impact of instream structures on physical habitat in the Nicolet River (Quebec) and to analyze physical habitat and fish abundance data from a large number of restoration projects using meta-analysis. Results of intensive surveys of the Nicolet River suggest that the installation of weirs and deflectors resulted in a greater frequency of pools. These pools have significantly greater depths, lower velocities and larger sediment size than those without structures. Compilation of data from 211 stream restoration projects showed a significant increase in pool area, average depth, large woody debris and percent cover as well as a decrease in riffle area following the installation of instream structures. The physical changes observed in the Nicolet River resulted in improved trout habitat, as measured by applying habitat preference curves, but uneven stocking practices and fishing pressure confounded attempts to verify differences in trout density among pool types. The meta-analysis, however, showed a significant increase in salmonid density and biomass following the installation of structures, although the relationship with physical habitat variables is not strong. Large differences in density response were observed between species. This compilation highlights the potential of instream structures to create

better habitat for and increase the abundance of salmonids, but the scarcity of long-term monitoring of the effectiveness of instream structures is problematic.

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1. Introduction

It is widely acknowledged that humans are negatively affecting the aquatic systems on which our survival depends (Richter et al., 1997; Ricciardi et al., 1999; Fausch et al., 2002; Lake et al., 2007). Since the 1980s the number of stream restoration projects has grown exponentially (Kondolf and Micheli, 1995; Bash and Ryan, 2002) and it is estimated that spending on restoration in the United States alone exceeds U.S.$1 billion per year (Bernhardt et al., 2005; Roni et al., 2008).

Instream structures, such as weirs, deflectors, cover structures, boulder placements and large woody debris (LWD), are a common method of restoring habitat in rivers (Wesche, 1985; Hey, 1996; Roni et al., 2008). These structures act to alter flow and scour patterns, resulting in a more diversified physical habitat of the stream (Champoux et al., 2003; Thompson, 2006). The installation of instream structures is typically carried out with the expectation that improved physical habitat will result in increases in the abundance and biomass of economically and culturally important salmonids (Roni et al., 2008). However, there remain many unknowns regarding the effectiveness of instream structures at producing appropriate habitat as well as uncertainties regarding the population response to changes in physical habitat (Klein et al., 2007). Though the success of instream structures is continually debated, all agree that further research is necessary (e.g. Huusko and Yrjana, 1997; Roni et al., 2002; Cowx and Van Zyll De Jong, 2004; Klein et al, 2007).

The goal of this research is to assess the success of instream restoration structures for improving salmonid habitat. This is achieved in a first step through a detailed study of the effectiveness of the Nicolet River (Quebec) restoration project. In order to generalize these findings, results obtained through meta-analysis of a large number of restoration projects are also presented.

2. Literature Review

2.1 History of trout stream restoration

Streams and rivers in many parts of the world have become degraded through various human activities such as urbanization, pollution, dam creation and water withdrawal, among others (Gleick, 2003; Bernhardt et al., 2005). One of the most obvious symptoms of this degradation has been a decrease in fish stocks, which is considered particularly problematic for sport fish species such as salmonids (Fausch et al., 2002). The first human interventions that attempted to increase trout abundance in streams were made in the late 1800s by wealthy individuals and angling clubs on private land (White, 1996; Thompson and Stull, 2002). Van Cleef (1885) produced the first published guidelines for trout stream restoration. In his report, he stressed that deforestation had reduced fish habitat and was therefore contributing to the decline in trout abundance. He suggested that reforestation and installing logs and branches in pools would provide the trout with adequate cover.

The restoration completed during this early time period usually consisted of building small dams, which increased river depth and provided aeration of the water as it spilled over the dam (White, 2002). Restoration would also include killing predators such as otters, herons and predatory fish (White, 2002). As well, stocking hatchery fish was common practice to ensure that club members remained fully satisfied with their catch (White, 2002). All the restoration attempts during this period consisted of attempting to increase trout yield in small sections of streams by private owners, no attempts were made to improve an entire stream, let alone an entire watershed (Thompson and Stull, 2002).

During the late 1920s and 1930s North American government offices became players in trout stream restoration for the first time (Hunter, 1991; White, 2002). The type of restoration that was carried out at this time period consisted primarily of installing instream structures (Hunter, 1991; White, 2002). There was a lot of experimentation, resulting in the installation of many different types of structures. The overall goal of the restoration was improvement, and the thinking of the time was that any stream could be improved through the installation of structures (White, 2002). The Institute for Fisheries Research at the University of Michigan, the first professional organization focused on research in stream improvement, stated in 1932 "the stream can be modified almost to any degree desired" (as quoted by Thompson and Stull, 2002). In this way, restoration was accomplished on streams regardless of whether they were thought to have diminished trout yields. Improvement during this era also consisted of stocking hatchery trout, controlling competitors and predators, and removing beavers and beaver dams (White, 2002). Re-vegetation and fencing out livestock are thought to have been the most successful improvements of the time (White, 2002).

Following this period of massive instream structure construction there was a lull in restoration attempts coinciding with the Second World War (White, 1996). Following the war, restoration continued, using the structures that had proved their durability if not their effectiveness during the 1930s (Thompson and Stull, 2002). During this period, evaluation of stream improvement received a boost through the use of electro-fishing (White, 2002). For the first time, there was a reasonably inexpensive method to determine trout abundance before and after restoration. However, post-restoration monitoring consisted of only one or two years (White, 2002). Though there was little advancement in the science of stream restoration during this time, scientific knowledge was accumulating in disciplines such as hydrology, geomorphology and ecology; these discoveries would pave the way for the next period of stream restoration (Reuss, 2005).

The surge in environmentalism of the 1970s and the concept of the ecosystem changed the way in which restoration was perceived. The research focus changed from improving natural channels to restoring channels that were negatively impacted by human activity (Hunter 1991). However, the methods to complete this restoration remained strikingly similar to the designs implemented in the 1950s. Before the 1980s, there was a general consensus that the installation of instream structures resulted in increased fish populations (Thompson, 2006). However, re-analysis of these data has shown that many of the studies were flawed, often by not accounting for changes in fishing pressure (Thompson, 2006). Decades of instream structure construction with little proof of successful outcomes make proper evaluation of current stream restoration practices particularly important.

2.2 *Current trout stream restoration projects*

Today, trout stream restoration research suggests that restoration should consist of a combination of watershed management and installation of instream structures (Roper et al., 1997). Watershed management will often consist of reforestation, guidelines to improve water quality and riparian zone protection (Roni et al., 2002). Instream structures are installed to increase specific habitat in areas where it is determined to be lacking. Common goals of instream structure installation include increase in pool habitat and cover, re-initiation of meanders and increase in habitat complexity (Roni et al., 2002). Though it is widely accepted that restoration will be ineffective if watershed management is ignored and only instream structures are installed, this occurs far too often (Roper et al., 1998; Roni et al., 2002; Wohl et al., 2005). While the installation of instream structures is costly, it usually does not require the cooperation of local residents and industry, as do most watershed management goals (Shields et al., 2003).

As the success of instream structures in restoration projects has yet to be agreed upon, there is a need to evaluate individual projects. To this day, projects that conduct post-restoration evaluations to determine their success remain few. Bernhardt et al., (2005) synthesized information for over 3700 restoration projects in the United States and found that only 10% of projects reported any type of monitoring or assessment. Furthermore, most of those that did report monitoring or assessment were not designed in a way to evaluate the effects of restoration or to disseminate their results. Numerous authors have pointed out that without published evaluations of projects, for both those that have succeeded and those that have failed, the science of stream restoration will not evolve (e.g. Kondolf and Micheli, 1995; Downs and Kondolf, 2002; Bernhardt et al., 2005).

Lack of measurable objectives and lack of post-project evaluation and monitoring are listed as two of the greatest problems in stream restoration (Downs and Kondolf, 2002; Bernhardt et al., 2007). Though only a small percentage of stream restoration projects publish results evaluating success, there remain a number of studies that have reviewed the outcome of individual restoration projects. The remainder of this chapter will be a review of this work.

2.3 Assessing restoration projects

There are a number of ways in which restoration success has been measured. While some have stressed that changes to channel geomorphology have been overlooked in favor of biological success criteria (e.g. Kondolf and Micheli, 1995), others have surmised that biota, including fish, have been inadequately assessed as most monitoring focuses of physical responses (Roni et al., 2002; Bash and Ryan 2002). These contradicting viewpoints suggest that further research is necessary on both the physical and biological impacts of restoration structures.

2.3.1 Structural stability

The most obvious measure of success in stream restoration projects is the durability of the structures installed. Many installed structures fail before they are able to alter the channel morphology in any way (Kondolf and Micheli, 1995). Structures have a much higher failure rate following a large magnitude flood (Roper at al. 1998). Schmetterling and Pierce (1999) found that 15% of structures installed on Gold Creek, Montana failed during a 50-year flood in the spring following installation. However, even low magnitude floods have been shown to damage structures. In a survey of over 3000 instream structures, Roper et al. (1998) found that 18% were removed or shifted following floods of 5- to 15-year frequency. A study of 161 instream restoration structures in Oregon and Washington showed that 18% of them failed completely while 60% were either damaged or ineffective (Frissell and Nawa, 1992).

The durability of instream structures is strongly influenced by the slope of the streams in which they are installed; structural failure is much more common in steep, high-energy streams than in lower gradient streams (White, 1996; Champoux et al., 2003). Roper et al. (1998) however, found that structural stability is greater on smaller (low order) streams than on larger (high order) streams and attributes this to differences in stream energy. Roni et al. (2002) conclude that artificial structures have moderate to high failure rates and that their benefit to fish are therefore likely to be temporary.

2.3.2 Physical habitat assessment

In order to evaluate physical effectiveness of structures rather than merely stability, planning, foresight and follow-up assessments are needed (Champoux et al., 2003). Evaluating the physical effect of instream structures requires pre and post-restoration surveys. A number of studies have been conducted that show that physical habitat can be improved through the addition of instream structures during the first few years following construction (White, 1996). The intended effects of instream restoration structures vary, but often include increases in habitat complexity, pool depth and frequency and either sediment retention or scouring (Roni et al., 2002). The chosen structure will depend on the characteristics of the stream and the habitat type believed to be lacking.

Schmetterling and Pierce (1999) studied the effect of 66 instream structures on Gold Creek, Montana. They found that the structures were responsible for the creation of 61 new pools, which were badly needed in the degraded river reach following riparian logging. Similarly Crispin et al. (1993) determined that the installation of 200 instream structures in Elk Creek, Oregon, resulted in a five-fold increase in suitable summer Coho salmon habitat and a six-fold increase in suitable winter habitat. The control reach that had no instream structures saw a decrease in suitable summer habitat and contained no winter habitat. Carré et al. (2007) made detailed surveys of bed topography around two sets of paired deflectors in the Nicolet River, Québec. Surveys were conducted for six years and showed that the downstream pool depth and volume remained relatively constant whereas the upstream pool showed greater fluctuations in size due to changes in annual flow. The authors determined that the difference observed in pool stability was a result of different geomorphic contexts; the upstream pool is located in a meandering section of river whereas the downstream pool is found in a much straighter reach. Both studied deflector pairs have succeeded in increasing the depth of pools.

As with structural stability, a positive effect of instream structures on physical habitat is more consistently observed on low gradient streams than on steeper, mountain streams (White, 1996; Champoux et al., 2003). When failure to create the desired physical habitat occurs, it is often the result of poor design or construction (White, 1996).

While many studies have shown habitat improvement immediately following the installation of restoration structures, there are few studies that have determined the long-term impact of instream restoration structures (Thompson, 2002). The Lawrence Creek, Wisconsin, restoration project was evaluated immediately prior (1963) and following (1966) restoration by Hunt (1969) as well as 36 years later in 1999 by Champoux et al., (2003). The study found that the long-term success of the bank-cover deflectors was largely dependent on the geomorphic context of the river (Champoux et al, 2003). The majority of the structures that had been installed in the steeper upstream morainic section of the river failed or were ineffective in the long term. In contrast, structures installed in the downstream outwash plain section of the river had a much lower failure rate and many, particularly in the section furthest downstream, were maintaining or even improving on the physical habitat observed in 1966. Thompson (2002) reports on the long-term impact of 40 restoration structures on channel morphology. He found that after several major floods, the restoration structures were in various states of operation and determined that some structures were even having a detrimental effect on channel morphology due to bank erosion, channel widening and reduced vegetation. Failure was much more common on the larger Salmon River than on the smaller Blackledge River lending credence to the Roper et al.'s (1998) study showing that failure of instream structures is more likely on larger order streams, due to increased stream energy.

It is clear that when well planned and constructed, instream structures can have a short-term beneficial impact on river morphology and trout habitat (White, 1996). The research on the long-term effectiveness of these structures is less certain and it is becoming clear that instream structures should be utilized to achieve immediate results while watershed management programs are needed to achieve long term improvements in trout habitat (House, 1996; Cowx and Van Zyll De Jong, 2004).

The use of physical habitat assessment to monitor the success of stream restoration structures is widespread (Roni et al., 2002). Though the assessment of restoration success through physical habitat is complicated by changing hydrological conditions it has the advantage over biological assessment of not being affected by disease, harvesting or interspecific competition (Kondolf and Micheli, 1995; Van Zyll De Jong et al., 1997). However, Pretty et al. (2003) caution against the use of physical responses to restoration as a predictor of ecological response. They assessed fish populations and physical parameters in thirteen rivers in the U.K. and found few significant relationships between the two. This indicates that increasing physical habitat does not necessarily lead to changes in fish populations. It is likely for this reason that a number of studies make use of biological measurements in the assessment of restoration success.

2.3.3 Biological effects

The majority of stream restoration projects aim to increase sport fish numbers or size (Roni et al., 2002) and, as such, a number of studies have examined the effect of instream structures on these two factors. Electro-fishing, under water observation and creel samples have all been employed to measure fish populations, and each method has both advantages and disadvantages over the others (Peterson et al., 2004). In order to determine the effect of restoration structures, either pre- and post-restoration or treatment and control data are needed. Ideally, all of these data would be collected since fish populations vary temporally, and even the most carefully chosen control reach never provides a perfect replicate (Shields et al., 2003). Further complications in assessing restoration success using population measurements occur when the installation of structures is combined with changes in fishing pressure or fish stocking (Thompson, 2006).

Increased abundance of salmonids following the installation of instream structures has been documented in a large number of case studies (e.g. Hunt, 1976; Hunt 1988; Keeley at al., 1996; McCubbing and Ward, 1997). Success has been documented using a wide variety of instream structures (Roni et al., 2008). The length of monitoring needed to detect a change in salmonid populations varies, however, Hunt (1976) observed that both number of fish and production (fish growth) increased following treatment, and that the majority of this response was observed three or more years after treatment.

In contrast to case studies that show successful biological results, Cowx and Van Zyll De Jong (2004) report on the failure of a salmonid habitat restoration attempt. In Joe Farrell's Brook, Newfoundland, the installation of ten instream structures did not increase the brook trout density. However, they did find that the Atlantic salmon (*Salmo salar*) aged 1+ years did increase significantly following restoration. They conclude that, in this case, the change in fish density did not warrant the cost of restoration. The authors suspect that the failure of this restoration project is a result, not of the structures not functioning as expected, but of widespread changes to the watershed, which render it inhospitable to salmonids. Determining the causes of failure of projects to increase salmonid abundance is important in increasing understanding of the best way to rehabilitate streams (Kondolf and Micheli, 1995).

2.3.4 Combining physical and biological evaluation criteria

Many studies that attempt to determine the result of restoration look at both changes to physical habitat and changes to fish density. Kondolf and Micheli (1995) suggest that opportunities to relate biological factors to physical form should be used whenever possible in order to increase our understanding of the links between the two. If both physical habitat and trout abundance have improved, it seems likely that the installation of structures is responsible for both. Once again results have been mixed, as some studies have found that following restoration both physical habitat and trout abundance have improved (e.g. Gowan and Fausch, 1996; Roni et al., 2006), whereas other studies have been unable to show increases in trout abundance despite success at improving physical habitat (e.g. Klein et al., 2007). Gowan and Fausch (1996) found that the addition of ten log drop structures to each of six streams resulted in increased pool volume, cover, mean depth and percentage of fine substrates. These changes are reported as substantial and beneficial to the stream studied. Furthermore, the authors found an increase in adult trout abundance and biomass in all treated sections relative to control sections. Similarly, Roni et al (2006) found that pool area, number of boulders and LWD all increased following the installation of boulder weirs. These physical changes were associated with increased abundance of juvenile Coho salmon (*Oncorhynchus kisutch)* and trout. However, Klein et al., (2007) examined 17-performance criteria before and after restoration, or between restored and control sites, and observed improvements in physical habitat criteria but were unable to show any significant improvement in salmonid density, which they attributed to a combination of small sample size, high inter-annual variability and short time since restoration.

Understanding the way in which instream structures act to increase trout abundance is extremely important in predicting the likely effects of restoration. Some studies suggest that increased growth or survival of fish results in higher densities and biomass (Hunt, 1976). Others suggest that the physical habitat changes result in preferred habitat, which larger fish utilize, forcing smaller fish into non-restored areas of the stream, and causing increased biomass in the restored reach (Rosi-Marshall et al., 2006). However, it is also possible that the installation of structures, while increasing the preferred habitat of large trout, decreases the preferred habitat of small sized trout, as has been suggested by Huusko and Yrjana (1997) in their modeling study. Others have determined that the increased abundance observed was due to immigration to the newly restored river sections (Gowan and Fausch, 1996; Roni and Quinn, 2001). If instream restoration structures increase trout abundance through immigration from degraded river reaches, then abundance will only increase if there are no barriers to movement (Gowan and Fausch, 1996). If the amount of preferred habitat for smaller fish decreases, it is possible that adult salmonid abundance will suffer a few years after restoration, and that this effect could be missed in short studies.

In order to investigate how the physical habitat changes initiated by instream structures could affect trout, Shuler et al. (1994) conducted a microhabitat study in the Rio Grande, Colorado. They found that brown trout selected micro-habitat sites based on water velocity and available cover and that the instream structures provided more locations that were preferred by large trout. Similar results were obtained in a modeled simulation of the impact of instream structure installation on quantity of preferred trout habitat (Huusko and Yrjana, 1997).

The link between amount of preferred habitat and density, however, is not clear. Urabe and Nakano (1999) conducted underwater trout observations to determine habitat use. They were then able to determine the preferred habitat of the fish by comparing the used habitat with the locally available habitat. They concluded that areas with high percentages of preferred habitat coincided with higher trout densities. However, Maki-Petays et al. (1999) found that the ability of habitat preference curves to predict density varied considerably. They measured fish densities and habitat characteristics in two rivers to determine if there was a correlation between density at a site and the amount of preferred habitat. The authors propose that the variability in the prediction ability of habitat preference curves is caused by biological factors and environmental variability, forcing fish to use less preferred habitat and therefore masking the habitat-to-fish density relationships.

2.3.5 Difficulties in restoration evaluation

The uncertainty that exists regarding the effect of instream structures on trout abundance is likely due to a number of factors. In order to succeed, the type of structure must be correctly chosen. This is largely dependent on fisheries managers or biologists having a good understanding of the population bottlenecks for the given population (Roni et al., 2002). For example, there is little point in increasing the over-wintering habitat for a population in which spawning habitat is non-existent. As the understanding of fish habitat requirements in all life stages has improved, so has our ability to correctly predict the instream structure most likely to increase abundance. The choice of structure is also largely dependent on the stream characteristics (e.g. slope, bed load movement, discharge), thus an increased understanding of fluvial geomorphology is essential in improving our decision making regarding instream structure installation (Shields, 1983; Roni et al., 2002).

Once the appropriate instream structure has been chosen, its design and construction also dictate how successful the restoration will be. White (1996) stated that failed habitat projects are ones that were badly designed or built. Structures should be built to withstand flood flows, however there is a trade off between the ability to withstand infrequent floods and increased cost. Some projects are simply unlucky and have little chance to succeed due to the occurrence of a large flood with a long recurrence interval soon after installation (e.g. Schmetterling and Pierce, 1999). In areas where winter ice cover occurs, it is also essential that this be taken into account during structure design, as ice has been shown to cause damage to instream structures (Shields, 1983).

A well chosen, designed and built structure is likely to succeed in improving the physical trout habitat (White, 1996). However, increasing trout abundance is a more complicated pursuit (Roni et al., 2002). As previously stated, biological populations fluctuate temporally due to many factors, only one being physical habitat. For salmonids, water quality, in particular water temperature, is often a problem in watersheds that have undergone land use changes (Baird and Krueger, 2003). The increased water temperatures caused by deforestation make the streams less hospitable for salmonids and increase competition with warm-water fish. An adequate food supply is also necessary for salmonid populations to grow. As well, populations will not increase if fishing pressure is too great (Barnhart, 1989).

Finally, even if all of these conditions are met, the stream habitat has improved and trout populations have increased, a poorly designed evaluation procedure can fail to capture the increase in population (Kondolf, 1995). The most likely cause of this is a post-project evaluation that samples fish populations for only one or two years after the completion of restoration. Indeed, an increase in trout abundance requires time. Hunt (1976) found that trout populations did not reach new post-restoration levels until three years after the installation of instream structures. Since trout populations can vary greatly from year to year due to environmental conditions it is necessary to collect many years of data to show conclusively that abundance has changed (Roni et al., 2002).

The difficulty in having a restoration project succeed, coupled with the additional difficulty of proving that it has indeed succeeded, is likely why there are so few projects that claim complete success. Those that are able to show at least partial success, though, provide an encouraging sign that stream restoration is possible and that the installation of instream structures can be a valuable tool in undoing some of the human damage to trout populations and streams.

2.4 Meta-analysis as an evaluation tool[1]

Despite over a century of restoration activity, many unanswered questions remain regarding the effectiveness of various restoration approaches, which is in part due to the inconsistent results among restoration projects (Bernhardt et al., 2005). Literature reviews attempt to bring together the results of a large number of case studies to draw overall conclusions. These generally conclude that there is increased salmonid abundance following restoration (Bayley, 2002; Roni et al., 2002; 2008), even if some case studies did not observe any significant impact (e.g. Johnson et al., 2005; Rosi-Marshall et al., 2006; Klein et al., 2007).

However, traditional literature reviews, while qualitatively describing the results of many individual case studies, do not allow statistical testing of overall trends (Roberts et al., 2006). Meta-analysis overcomes this problem by allowing the formal combination of results from a large number of case studies (Gates, 2002). This method offers the opportunity to uncover trends in restoration success across large geographical areas and to provide a much larger pool of data from which to determine statistical significance. In

[1] Part of Whiteway et al. (submitted for review)

a recent meta-analysis of instream structures, Stewart et al. (2009) found only equivocal evidence of their effectiveness at increasing salmonid abundance and significant variability in success among projects. Their commendable use of strict inclusion criteria required that all projects include some inherent replication or pseudoreplication, which resulted in only 17 studies and 38 data points in their analysis. Their small sample size prevented a comparison between structure types or fish species and limits the conclusions that can be drawn from their study. Meta-analysis provides an opportunity for important questions regarding instream structures to be addressed, however all meta-analyses rely on the availability of quality case studies (Roberts et al., 2006).

3 Methodology

3.1 Nicolet River study site

The Nicolet River, Québec, flows 140km from its headwaters in the Appalachian Mountains to its mouth at the Saint-Laurent seaway. The Nicolet sub-basin of the Arthabaska watershed covers an area of 265 km^2 (Carré et al, 2007). Founded in 1988, the Corporation de Gestion des Rivières des Bois-Francs (CGRBF) restored a 14km stretch of the Nicolet River located to the south east of Victoriaville, Québec (Fig.1). The goal of this restoration was to improve the quality of trout habitat in order to improve sport fishing and increase tourism in the area (CGRBF, 1993).

The restored section of the river flows through forested and agricultural land and has an average high-flow width of 30m. Due to human activity, primarily land use changes, the section chosen for restoration had become degraded and the trout population had dwindled. The primary problems identified by the CGRBF (1993) before restoration activities were commenced were the lack of pools and cover for trout as well as elevated summer water temperatures. The restoration included bank stabilization, fish shelter construction, tree planting, instream structure construction and fish stocking. The instream structures consist of paired deflectors, single deflectors and weirs, all of which were designed to increase pool depth and volume.

Three species of trout, brook (*Salvelinus fontenalis*), brown (*Salmo trutta*) and rainbow (*Oncorhynchus mykiss*) are stocked throughout the fishing season with an average of over two hundred stocked per week. Data on the number and species of fish stocked at each stocking event were obtained from CGRBF representatives. The release location was also recorded for each stocking event. During the 2008 fishing season, 3084 trout were stocked. The vast majority of these were rainbow trout (2599, or 84%); 220 brook trout and 265 brown trout made up the remainder of the fish stocked. Fish were stocked on 14 dates between April 26th and September 6th. Between 40 and 432 trout were stocked on each date. Fish were stocked in two locations that were easily accessible, pools 25, 26, 27 located directly in front of the CGRBF welcome centre and pool 41, located at a CGRBF parking area.

Anglers who wish to fish in the restored section of the Nicolet River must purchase a daily fishing permit from the CGRBF. The restored section of the river contains 70 pools that have been numbered and marked. There are marked trails for anglers to follow from designated parking areas to the numbered pools.

3.2 Restored reach survey

A survey was conducted of the 14km stretch of the Nicolet River to map the location of instream structures and pools and to record water depth. The survey was accomplished by rafting down the restored section of the river in an inflatable raft. Every 100 metres location and water depth were recorded. In addition, GPS coordinates and maximum water depth were taken at each marked pool. It was also noted whether or not a restoration structure was present at each pool, and, if so, the type of structure. Location was recorded using a Garmin Etrex GPS. Depth was measured using a 3-metre graduated pole to an accuracy of approximately 5cm (accuracy was slightly reduced due to movement of the raft).

A small section of restored river was not mapped at the upstream end as there was no easy access to enter the river. It was also not possible to map a small middle section of the river due to the presence of falls.

The GPS coordinates taken along the river survey were imported into the GIS software ArcGIS (ESRI, version 9.2) and overlaid on a map of the hydrologic network (Fig. 2). Surveying was done on two days a week apart. Discharge was slightly higher on the second day of surveying. To correct for this, depth at the same location was measured both days and all depths were corrected for this change. As there is very little change in discharge throughout the restored reach of the river, depth corrections from a single repeated depth measurement in the middle section of the reach is likely sufficient to ensure that there is not a bias in depth measurements based on the discharge at which they were taken.

In order to assess whether there was a difference in the degree of meandering between the areas surrounding the restored pools and those surrounding the non-restored pools, the distances – both straight line and along the curve of the river – between pools were measured in ArcGIS (ESRI, version 9.2). The ratio of the curved line distance between the two nearest pools and the straight line distance between the two nearest pools gave the degree of meandering value. These values were calculated for restored and non-restored pools and compared using a t-test.

3.3 *Physical habitat of pools*

Twenty six pools were chosen for more detailed analysis. At these pools the water depth, water velocity, sediment size and presence or absence of cover was determined at between 63 and 15 point locations, depending on the size of the pool. Measurements were taken at a total of 771 point locations.

Depth was measured to the nearest cm using a 3m measuring rod. Velocity was determined using a vertically mounted axis propeller current meter (Swoffer-2100). When the water depth was less than 0.5m the average water column velocity was approximated by measuring the velocity at a depth of 60% of total depth, whereas the average from velocity measured at 20% and 80% of the flow depth was used for deeper flows (Moyle and Baltz, 1985; Facey and Grossman, 1992). Sediment size was approximated by selecting the rock directly beneath the measuring rod and removing it from the water, when possible, to visually estimate its size. In cases where the rock was too large to lift or the water too deep, a visual estimate was made underwater. Cover was determined to be present if there was a velocity refuge large enough for a 15cm fish within 1m of the point location. Cover was classified as: boulder cover, woody debris cover, undercut banks, or overhanging vegetation cover when overhanging branches disrupted the water flow (Heggenes et al., 1991).

An effort was made to take all physical habitat measurements at a similar discharge, however some variation in discharge was unavoidable. All measurements were taken during summer low flows and discharge ranged between $0.65m^3/s$ and $2.5m^3/s$. In order to correct for slight changes in discharge between measurements all depth and velocity measurements were adjusted to represent the depths and velocities at a discharge of $2.5m^3/s$, as explained below.

At the river cross-section where a pressure transducer is located, depth and velocity measurements were made at the same point locations at a variety of low flow discharges. This allowed the development of a rating curve relating depth to discharge and average velocity. The pressure transducer recorded depth every 15 minutes at this cross section, allowing us to calibrate measurements made on different days. Depth and velocity measurements were adjusted to represent the depth and velocities expected at a $2.5m^3/s$ discharge. To test the validity of this approach, pool #51 was surveyed at two discharges. The first survey occurred on the lowest discharge day ($0.65m^3/s$), and the other was done when discharge was $1.96m^3/s$, the second highest discharge of our survey. The average depth of pool #51 at a discharge of $0.65m^3/s$ was 0.73m, whereas at the higher discharge the average depth was 0.82m, which is significantly greater (Student's t-test, $T_0=2.2$, $p=0.03$) (where the subscript represents the degrees of freedom). Once the depths had been adjusted to correspond to a discharge of $2.5m^3/s$, there was no significant difference between the average depths of pool #51 on the two survey days ($T_{60}=0.25$, $p=0.79$). Average velocity was also compared at this pool for the same two discharges. At $0.65m^3/s$, average water column velocity was 0.07m/s. This was significantly lower than the average velocity at $1.96m^3/s$ of 0.15m/s ($T_{49}=2.95$, $p=0.005$). When velocities were adjusted for a $2.5m^{3/}/s$ discharge there was no difference between the two ($T_{49}=0.39$, $p=0.69$). Thus we feel that these adjustments to the depth and velocity data make comparisons among pool types possible even when data were collected on days with different discharges.

Of the 26 pools that were selected for the habitat availability study, 10 pools were not associated with any restoration structure, and 16 were located downstream of either a deflector structure or a weir structure. Of these restored pools, 10 were deflector pools and 6 were weir pools. Depth, average water column velocity and sediment size were averaged for all non-restored pools, all restored pools, all deflector pools and all weir pools. Percent cover was determined for each pool or non-pool area, and this was then averaged for each pool type.

Student's t-tests were used to determine whether the differences in depth, velocity, sediment size and percent cover were significantly different (α=0.05) for the different pool types.

3.4 Trout habitat use

Two methods were used to determine trout habitat use. The first was snorkeling, where the snorkeler would approach a pool from downstream using ropes in order to minimize splashing. If a trout was observed, and did not appear to be disturbed at the moment of observation, the snorkeler would record trout species, the approximate size of the fish and its height above the bed, and would then drop a weighted marker at the trout's location. The marker would be used to relocate the trout's exact position once snorkeling of the pool was completed. At this point, depth, average water column velocity, sediment size and the presence or absence of cover were recorded in the same manner as for the pool surveys. Snout velocity was also recorded using the same propeller current meter at the height above the bed that the snorkeler observed the fish. If no trout was observed in the pool, the snorkeler would hold their position for 30 minutes in case a trout was disturbed by their approach. This method was used for 7 pools, and two pools were checked twice.

The second method was in a pool that was very deep and from which a snorkeler could not see the bottom of the entire pool. An underwater video camera connected to a laptop and attached to a long rod was lowered towards the bottom of the pool. The camera video taped the underwater images and the connection to the computer allowed us to see the images in real time from an inflatable raft. This method was used for one pool (#41).

3.5 Trout habitat suitability

Trout habitat preference curves have been developed for a number of rivers. These curves are developed by comparing the used habitat to the available habitat in a number of categories. Typically depth, velocity, sediment size, temperature and often the degree of cover are used (Lewis, 1969; Raleigh et al., 1984, 1986; Hapton, 1988; Heggenes et al., 1991; Vismara et al., 2001; Strakosh et al., 2003). While there remains skepticism regarding the transferability of these curves to rivers from which the data were not gathered (Moyle and Baltz, 1985; Rosenfeld, 2003), general suitability curves have been established for a number of species (Strakosh et al., 2003). Using the suitability curves from Raleigh et al., (1984) for rainbow trout and Raleigh et al., (1984), Vismara et al., (2001) and Strakosh et al., (2003) for brown trout the predicted suitability of pools with and without restoration structures, and of weir and deflector pools, were determined. This was achieved by assigning a suitability score from 0 to 1 for each of the point locations to each of the following variables: depth, velocity, substrate and percent cover. These 4 suitability scores were equally weighted and multiplied together in order to determine the total suitability score at each point in each studied pool. Though brook trout are also stocked in the Nicolet, the lack of a general habitat preference curve model prevented brook trout suitability comparisons among pool types.

The suitability scores for each pool type (restored, non-restored, deflector and weir) were averaged and Student's t-tests were used to determine if significant differences (α=0.05) existed between pool types.

3.6 Angler surveys and trout stocking

To determine relative abundance of trout in different reaches of the Nicolet River, angler surveys were used (Appendix 1 and 2). Surveys were handed out at the CGBRF welcome centre to all anglers purchasing fishing licenses. Anglers were asked to enter catch information including species, fish length, date and the pool number where the catch was made. They were also asked to approximate the length of time spent fishing at different pools and to return the surveys to the welcome centre.

Anglers completed 147 surveys during summer 2008. They reported catching 636 trout, for an average of just over 4 trout per angler per trip. Of these, 493 were rainbow trout, 60 were brook trout and 51 were brown trout; there was no species specified for the remaining 32 fish caught. Reported catch represented 19% of the fish stocked. Brook and brown trout tended to be caught earlier in the summer, while rainbow trout were the predominant species caught from early June to the end of the fishing season.

Details from the angler surveys were input into a database. The number of fish caught per pool was calculated, as well as the number of hours spent fishing at each pool and the number of fish caught in different size classes. Catch per unit effort (CPUE) was calculated as the number of fish caught per hour of fishing. Average CPUE was compared between pool types using a t-test. Linear regression was used to test the relationship between fishing pressure and catch.

3.7 *Meta analysis literature search*[2]

In order to analyze the effect of instream structures from a large number of case studies a literature search was conducted by performing key word searches on major biological and environmental science catalogues. ISI web of knowledge, Scopus and JSTOR were searched using keywords "trout OR salmo* AND river OR stream AND restor* OR enhance* OR improve* AND habitat" (where * represents a wildcard). The abstracts and references of articles that appeared relevant were examined. Searching through the reference lists of these articles turned up additional articles and reports. Only studies that provided salmonid density of at least a treated reach and a control reach were included in the meta-analysis. Time series studies, site comparisons and Before-After, Control-Intervention (BACI) studies were also included. Projects needed to have installed one or more of the following: weirs, deflectors, cover structures, boulder placements, and LWD. A total of 51 reports met our criteria (see Appendix 2). Some reports were compilations of many different projects, thus providing a total of 211 stream projects for our analysis.

For each project, we recorded information about the restoration project (year of completion, type of structure installed, cost, length of the restored reach), project monitoring (number of years and type of monitoring - pre-and post restoration and/or treatment and control), and on the species and size classes of salmonids. When available, biomass data and physical habitat data were recorded for the pre- and post-restoration and/or the treatment and control sections. Physical habitat data consisted of the percent pool and riffle areas, mean stream width, number of pieces of LWD, percent cover and

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[2] Part of Whiteway et al. (submitted for review)

mean stream depth. The values for these physical variables were all recorded directly from the reports and no attempt was made to verify the methodology used to obtain each value. For each species and size class of fish the density (no.m^{-2} or no.m^{-1}) and biomass (g m^{-2}) were recorded, or calculated, for the pre- and post-restoration and/or the treatment and control sections. No distinction was made between projects that collected density data via electro-fishing versus snorkelling. Although there is evidence that each method of estimating fish abundance has limitations (Peterson et al., 2004), the method used was consistent within each project and should not bias our results.

3.8 *Compiled data analysis*[3]

Effect size (L) was calculated for each study using the log response ratio

$$L = \ln(x_{tr} / x_c) \qquad (1)$$

where x_{tr} is the treatment mean and x_c the control mean (Hedges et al., 1999). The log response ratio was chosen because it measures the proportional change of important ecological variables caused by the treatment (Janetski et al., 2009). The log response ratio is more useful than the arithmetic ratio because it is affected equally by changes in the numerator and denominator and it is more normally distributed in small sample sizes (Hedges et al., 1999). We did not use Cohen's D effect size (Stewart et al., 2009), because it requires a measure of the standard deviation of the response, which is not available for many single-site restoration projects. For BACI data the change in the treated reach served as the treatment value and the change in the reference reach served as the control. When BACI data were unavailable, the mean difference was used for the

[3] Part of Whiteway et al. (submitted for review)

control and treatment sites, or for before and after restoration. Out of the total of 211 analysed projects, 148 (70%) came from the grey literature. In 113 projects (54%), at least one physical habitat characteristic was monitored in addition to salmonid density and 78 (37%) projects reported biomass data as well as density data.

Data were available for 8 species of salmonids: brook trout (*Salvelinus fontinalis*), brown trout (*Salmo trutta*), rainbow and steelhead trout (*Oncorhynchus mykiss*), cutthroat trout (*Oncorhynchus clarki*), Coho salmon (*Oncorhynchus kisutch*), Atlantic salmon (*Salmo salar*), Chinook salmon (*Oncorhynchus tshawytscha*) and arctic grayling (*Thymallus arcticus*). However, fewer than 10 studies monitored densities of Chinook salmon or arctic grayling, so these were not included in the individual species comparison. Because steelhead trout are anadromous, whereas rainbow trout remain in fresh water throughout their lives, these two subspecies were analysed separately.

Three size classes of salmonids were created based on the most common size classification used in the analysed reports: (1) <10cm in length, which included fish aged 0+ and those classified as fry; (2) 10-15 cm in length, which included fish aged 1+ and those classified as parr; and (3) >15cm, which included age 2+ and 3+ fish and all fish classified as smolts or adults.

Effect size (eq.1) was calculated for salmonid density in all cases, and for each of the following variables when available: salmonid biomass, pool area (%), riffle area (%), width, depth, cover (%), and the number of pieces of LWD. The L values for each project were averaged to obtain a mean effect size for each variable. For each project the density effect size was calculated for each species, size class and year of monitoring. The average density effect size was calculated for each year of monitoring in order to assess the effect of project age. Finally, in order to assess overall project effectiveness, data for the last monitored year were used.

One-sample t-tests were used to determine if the mean effect sizes were significantly different from 0 at α=0.05. ANOVAs were used to test whether there were significant differences (α=0.05) between changes in density based on fish species, fish size class, the use of one structure type or multiple structure types, project age and publication type. Multiple regression analysis was used to determine the effect of changes in physical habitat factors on changes in salmonid density and biomass. Differences among structure types, on both biotic and abiotic variables, were also investigated through ANOVAs: these tests only included projects that used a single structure type.

Figures

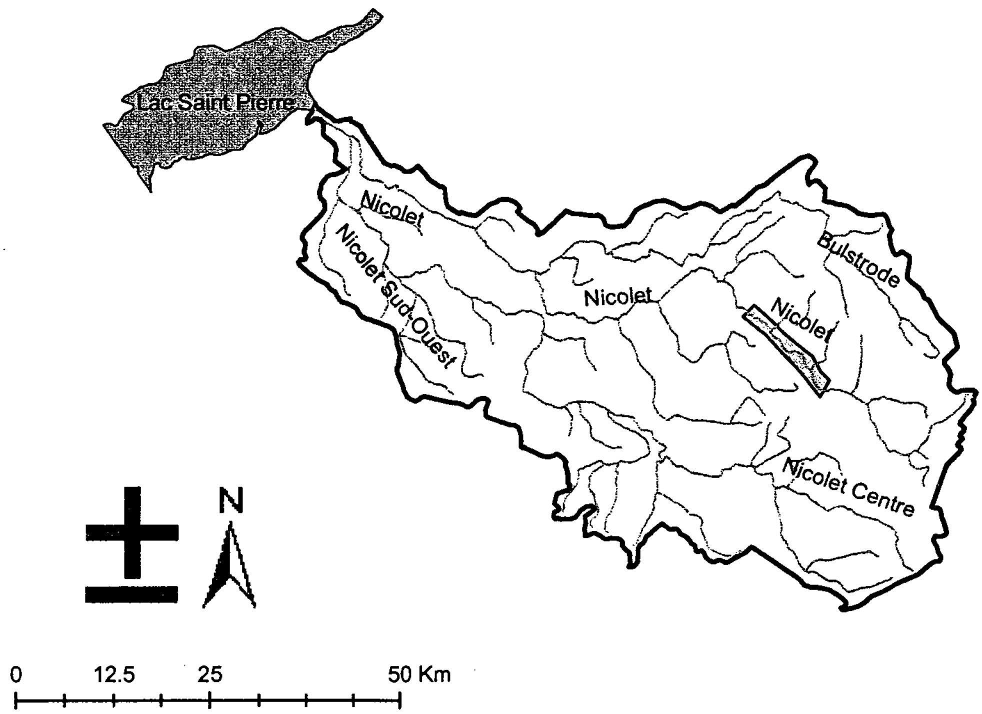

Figure 1. The Nicolet River watershed with the restored section of the river shown within the grey area.

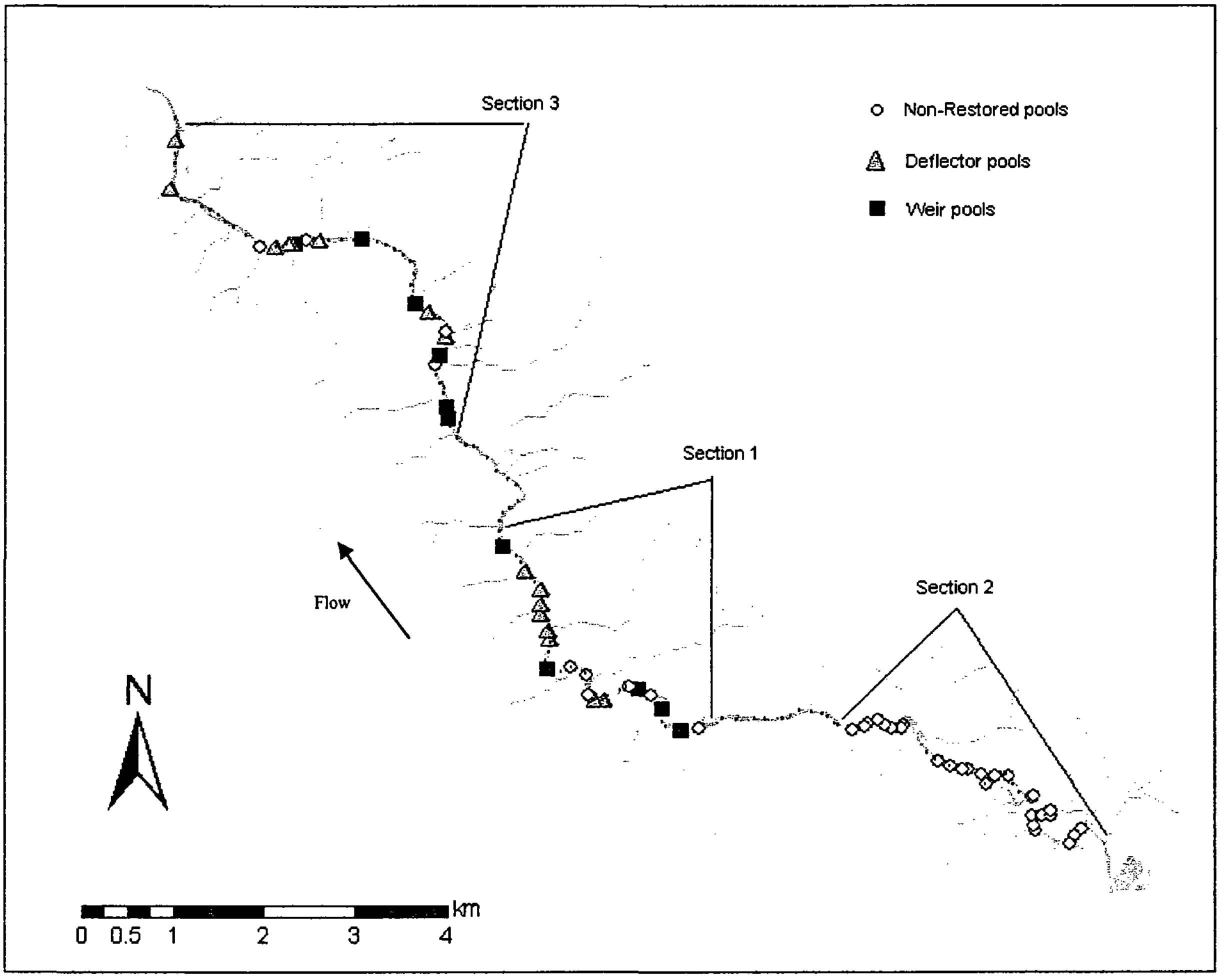

Figure 2. The restored section of the Nicolet River showing the locations of the marked pools.

4. The Effects of Weirs and Deflectors in the Nicolet River

4.1 Physical effects of instream structures

The restored section of the Nicolet River includes 70 pools. Of these, 40 pools have no restoration structure, 17 pools are located downstream of deflectors and 13 pools are located downstream of weirs. The restored section of the river is made up of three sub-sections that are separated by small reaches of the river that are not managed by the CGRBF (Fig. 2). Section 2 is the furthest upstream and there are no restoration structures in it. The restoration structures are spread fairly evenly across the remaining two sections of the river. There was no difference in the degree of meandering between the areas surrounding the restored pools and the non-restored pools (T_{50}=0.76, p=0.44).

Water depth was measured at 173 thalweg locations for which GPS coordinates were obtained, 109 of these were non-pool depths. From this sample the non-pool average thalweg depth of the restored section of the Nicolet River at low flow ($2.5m^3/s$) was 0.34m. The maximum depth of 64 of the 70 pools was measured. The average maximum pool depth of the Nicolet River at the same discharge was 1.30m.

The installation of deflectors and weirs had a significant impact on the physical habitat characteristics of the Nicolet River pools. There was no significant difference in the maximum depth of pools with structures and those without structures (T_{62}=0.26, p=0.39; Fig 3a), however restored pools had larger average depths than non-restored pools (T_{24}=4.52, p<0.0001; Fig. 3b). Restored pools also had significantly lower average water column velocity than non-restored pools (T_{24}=4.67, p<0.0001; Fig 3c) and the average sediment size was larger in restored pools (T_{24}=5.03, p<0.0001; Fig. 3d). Finally, there was more cover in restored pools compared to non-restored pools but the difference was not significant (T_{18}=1.7, p=0.11; Fig. 3e). In the Nicolet, boulders provided a large amount of the available cover. There were, however, differences in the type of cover in restored pools and non-restored pools (Fig. 4). Boulder cover made up 94% of the cover in restored pools, whereas, significantly less cover was created by boulders in non-restored pools (T_{24}=4.19, p=0.0003). Cover from undercut banks and woody debris was more prevalent in non-restored pools than in restored pools (T_{24}=2.54, p=0.018 and T_{24}=2.15, p=0.042 respectively). Overhanging vegetation provided equal cover in both restored and non-restored pools (T_{24}=0.19, p=0.84).

Comparisons were also made between the effects of deflectors and weirs. The average maximum depth of weir pools (1.5m) was greater than that of deflector pools (1.1m) (T_{28}=2.61, p=0.007; Fig. 5a). Weir pools had larger average pool depths than deflector pools (T_{14}=4.69, p=0.0003; Fig. 5b) while deflector pools had higher velocities (T_{14}=5.83, p<0.0001; Fig. 5c). Weir pools also had larger sized sediment than deflector pools (T_{14}=2.17, p=0.047; Fig. 5d) and there is no difference between the cover available in weir pool and deflector pools (T_{12}=0.02, p=0.98; Fig. 5e).

4.2 *The impact of instream structures on habitat suitability*

Using the habitat suitability index developed by Raleigh et al. (1984) for rainbow trout, the mean suitability score for restored pools was significantly greater than that of non-restored pools (T_{24}=6.83, p<0.0001; Fig. 6a). However, mean depth suitability scores were not different between restored and non-restored pools (T_{24}=0.81, p=0.43; Fig.

6b), nor were cover suitability scores (T_9=0.93, p=0.37; Fig. 6e). On the other hand, velocity and sediment size suitability were markedly higher in restored pools than non-restored pools (T_{24}=2.92, p=0.007 and T_{24}=5.44, p<0.0001 respectively; Fig. 6c,d).

Comparisons between deflector pool suitability scores and weir pool suitability scores showed that weir pools were more suitable for rainbow trout than deflector pools (T_{14}=5.63, p<0.0001; Fig. 7a). Both depth and velocity suitability scores were significantly higher for weir pools than deflector pools (T_{14}=5.4, p<0.0001 and T_{14}=6.0, p<0.0001 respectively; Fig. 7b,c). There were no significant differences between the sediment size or cover suitability scores of deflectors and weirs (T_{14}=0.59, p=0.56 and T_9=1.0, p=0.34 respectively; Fig. 7d,e).

Similar results were obtained using brown trout habitat suitability indices (Raleigh et al., 1986, Vismara et al., 2001, Strakosh et al., 2003). Restored pools had higher overall suitability scores than non-restored pools using all three indices. Two of the indices gave significant differences (T_{24}=2.39, p=0.025 and T_{24}=4.37, p=0.0002); however, the results using Vismara et al.'s (2001) preference curve were not significant (T_{24}=1.48, p=0.15) (Fig. 8a). Weir pools also showed higher suitability scores than deflector pools using brown trout indices (T_{14}=2.6, p=0.02 and T_{14}=2.36, p=0.03) in two cases. However, there was no difference when using Strakosh et al.'s (2003) preference curve (T_{14}=0.43, p=0.67) (Fig. 8b).

4.3 Instream structures, habitat use and fish stocking

Only three rainbow trout were observed by snorkeling. All were observed on different days. Two were located in pool #38 and one in pool #34. Pool #38 has a pair of wooden deflectors that are in need of repair and pool #34 has a boulder weir. All trout were observed in deep water, 0.97m, 1.08m and 2.0m. All fish were within 15cm of the bed and the bed was composed of cobbles ranging from approximately 8cm to 15cm. Water column velocities ranged from 0.25m/s to 0.63m/s. Snout velocities varied between 0.04 and 0.46m/s. There was no cover located within 1m of the observed trout, however the percent cover was 36% in pool #34 and 23% in pool #38. Raleigh et al. (1984) show that habitat is most suitable for adult rainbow trout when depth is over 0.45m, average water column velocity is between 0.15m/s and 0.60m/s, sediment size is over 7cm and there is at least 25% cover. Though the low sample size of observed trout in the Nicolet River prevented any statistical test of the applicability of these suitability curves in the river, the location of observed trout does correspond well with the utilized suitability curves.

There were no catches reported from the majority of the pools in the Nicolet (Fig. 9). Of the 451 catches in which location was reported, 121 were caught at pool #25 and 111 at pool #26. No trout were reported caught downstream of pool 46. The number of reported catches is dramatically different between the 3 sections of the Nicolet River: 88% were made in section 2, 12% in section 1 and none were made in section 3. Only 3% of fish were caught at restored pools, the rest were caught at pools which had no restoration structure and were located closer to the stocking sites.

Fish length information was reported for 226 trout catches. The mean length of caught trout was 31 cm. The majority of trout caught were between 20 and 40 cm (Fig. 10).

Fishing pressure is also unevenly distributed across the pools of the Nicolet River. Of the 328 fishing hours reported by anglers, pool #26 had the highest fishing pressure with 88 reported hours fishing at that pool. Conversely, most pools had no reported fishing pressure (Fig. 11). Not surprisingly, there is a strong relationship between fishing pressure and catch (R^2=0.97; Fig. 12). The catch per unit effort (CPUE) as measured by the number of fish caught per hour fished was 0.85 for all managed pools of the Nicolet River. Of the pools with at least one fish reported caught, CPUE did not differ significantly among pool types: the average CPUE for restored pools was 0.54 and 1.1 for non-restored pools (T_8=1.48, p=0.17) (Fig. 13).

4.4 Discussion

As expected, the installation of instream structures appears to have increased the pool frequency in the Nicolet River. Restored pools had similar maximum depth, and greater average depths than pools that are not associated with instream structures, suggesting that deep pool area had increased. Deep pool area is important for moderating elevated water temperature (Matthews et al., 1994) and providing shelter from overhead predators (Wesche 1985). Other studies have similarly documented increases in pool frequency and depth following the installation of instream structures (Crispin et al., 1993; Shields et al., 1995; Klein et al., 2007, Baldigo et al., 2008) and Roni et al., (1996) found that the installation of boulder weirs increased pool area, due primarily to the creation of large pools. Prior to restoration, the Nicolet River was judged to have insufficient pool area (CGBRF, 1993). Therefore, these changes have provided needed habitat for the trout population.

Low velocity in pools is beneficial for trout, as it provides a resting place (Wesche, 1985). A potential problem of weirs and deflectors for creating pools is that they may increase velocity as they either constrict the flow or increase the local bed slope (Wesche, 1985). This has not been the case in the Nicolet restoration project at summer flows as velocities, on average, were lower in the restored pools than in those that are not associated with a structure. Other studies have similarly concluded that instream velocities are lower around instream structures than in similar areas not impacted by restoration (Shuler et al., 1994; Shields et al., 1995). However, Pretty et al. (2003) documented increased velocity following the installation of instream structures.

Changes in sediment size occur following the installation of structures because of the scouring effect of structures at high flow, which removes the finer particles (Swales, 1989). Larger average sediment size was observed in the restored pools of the Nicolet River, however a portion of this increase was likely due to the higher presence of boulders, as the structures were constructed of boulders. Over time some of the structure boulders have migrated into the pools, which inflated the average sediment size in those pools. Boulders made up 15% of the sediment in restored pools compared to only 6% in non-restored pools. Previous studies have documented both increased (Baldigo et al., 2008) and decreased (Jungwirth et al., 1995) mean particle size following the installation of weirs and deflectors.

Increased cover is a frequent goal of instream structures (Roni et al., 2008) and previous studies have documented the ability of weirs and deflectors to increase fish cover (Hunt, 1988; Avery, 2004; Baldigo et al., 2008). The results from the Nicolet River showed that the average percent cover was 8% higher in the restored pool than the non-restored pools, however this difference was not significant. Non-restored pools showed large variation in cover (between 3% and 45%) whereas the restored pools had between 23% and 55% cover. Prior to restoration, cover was deemed lacking (CGRBF, 1993) and it appears that the structures do provide cover, though perhaps not more than is available in non-restored pools. The difference in the type of cover available in restored and non-restored pools was to be expected. That boulder structures provide a large amount of boulder cover is not unusual, and since the restored pools are located mid channel as opposed to near the banks, they cannot provide cover from undercut banks. The absence of woody debris in the restored pools is more interesting. It is possible that high velocities during flood events remove woody debris from these mid channel pools, however the causes were not investigated in this study.

There exist differences between the location of the restored pools and the non-restored pools within the river channel. The majority of the restored pools were dug in the middle of the channel with the structures designed to ensure that the pools retain this location. In contrast the natural pools tend to be located on the outer edge of meander bends. This influences the typical cross section shape of the different pool types (Fig. 14). Though there was no significant difference observed in the degree of meandering between the restored and non-restored pools, it is likely that there do exist differences in the geomorphic context of the restored and non-restored pools due to their different placement within the channel.

Wesche (1985) lists similar objectives for weirs and deflectors, these include: creating pool habitat or deepening existing pools and altering flow patterns. Few studies have explicitly compared the physical habitat effects of deflectors and weirs. Olsen et al.

(1984) did compare physical effects of rock weirs to log weirs and log deflectors and found that log weirs resulted in the greatest pool area, cover and depth, however the values were very similar to those for log deflectors. Weirs had larger maximum pool depth than deflectors and overall deeper pools in the Nicolet whereas flow velocities were higher in deflector than in weir pools. The latter is expected as weir pools included both a scour pool downstream of the structure, in which velocities would be expected to be relatively high, and a backwater pool upstream of the structure, in which velocities are lower. In comparison, deflector pools were located continuously from slightly upstream of the structure to downstream and velocities were high throughout the pool area, except within the recirculation zones.

Weir pools had, on average, larger sediments than deflector pools. It is unlikely that the removal of fine particles due to scouring resulting from weirs would be greater than that from deflectors, as sand and silt made up 22% of the sediment in weir pools and 18% of the sediment in deflector pools. However, it is possible that the boulders forming the weirs were more likely to migrate into the pools than those of the deflectors – weir pool sediment was 20% boulder, compared to 10% for deflector pools. This may have contributed to the larger average sediment size in weir pools.

The restored pools of the Nicolet River scored higher than the non-restored pools in trout habitat suitability. This indicates that the physical habitat changes created by the instream structures have the potential to improve trout summer habitat. However, there are major limitations to using habitat preference curves developed in other rivers (Moyle and Baltz, 1985; Rosenfeld, 2003). Differences in intraspecific competition, habitat availability and food abundance are all listed as potential difficulties for transferability of suitability curves (Strakosh et al., 2003). For that reason, the recommended protocol is to develop, or modify, suitability curves on site (Raleigh et al., 1986; Glozier et al., 1997; Strakosh et al., 2003).

A modification of suitability curves was attempted in the Nicolet. However, low visibility was a problem and only 3 trout were observed undisturbed in over 20 hours of snorkeling. There were few days when visibility exceeded 1.5m, the minimum suggested for making underwater observations (Goldstein, 1978), and visibility never exceeded 2m. Low visibility made it impossible to see the entire width of the river and, in some of the deeper pools, the bottom was not visible. This is problematic as Peterson et al. (2005) found that salmonids responded to the presence of a snorkeler at 10 to 20m; thus fish may react to the presence of a snorkeler well before the snorkeler could actually see them in the Nicolet River. While many studies have used underwater observation to determine fish abundance and habitat (e.g. Goldstein, 1978; Shuler et al., 1994; Thurow and Schill, 1996; Mullner et al., 1998), the majority were conducted on streams much smaller than the Nicolet River and with better visibility.

It was hoped that the use of an underwater video camera would allow trout observations in pool #41. This pool was stocked and had a high catch rate (see Fig. 8) – making the presence of trout likely – but was over 3m deep, which prevented snorkelling observations. The video method, however, also failed to locate any undisturbed trout. If trout in the Nicolet River are as easily disturbed as these experiences suggest, permanent underwater cameras may be the only way to determine micro-habitat use. Electro-fishing can not be done, both because of the presence of anglers and the depths of the pools, and radio tagging fish in a river with such high fishing pressure would prove expensive and likely unwelcome in fish that are consumed.

One of the complications in determining the habitat preference of trout in the Nicolet River is the patchiness of stocking and fishing pressure. A survey of anglers during the summer of 2008 showed that the majority of trout catches occur in pools that either are stocked or are adjacent to stocked pools. It has been found that stocked trout are unlikely to move more than a few kilometres from their stocking site (Cresswell, 1981; Helfrich and Kendall, 1982; Hesthagen et al., 1989; Aarestrup, 2005). This distance is further reduced when they are stocked in warm water, in the same season in which they are caught, into larger streams or rivers (Cresswell, 1981) and into pools rather than riffles (Helfrich and Kendall, 1982). All of these factors make it unlikely that the stocked fish in the Nicolet River will fully use the 14km of restored habitat. Since the stocking sites are known to local anglers, fishing pressure is also highest around the stocked pools. After analysing the location of 434 trout catches, there was no evidence that fish were more likely to be caught in restored pools. However, the uneven stocking and fishing effort made drawing any conclusions about habitat preference impossible.

All anglers that stopped by the CGRBF welcome centre in 2008 were asked to complete the questionnaire regarding their fish catch and hours spent fishing. However, returning the questionnaire was not obligatory. All anglers who fish in the section of the Nicolet River managed by the CGRBF must purchase a fishing license from the CGRBF, the majority of these are obtained at the welcome centre on the day that they fish. However, if the welcome centre is closed, it is possible to purchase a permit at a local store. It is also possible to purchase a multi-day pass, or a yearly pass. In these cases, anglers would not have been asked to complete the questionnaire. The CGRBF estimated that 700 anglers fished in the restored section of the Nicolet in 2008 (Alain Ramsey, personal communication), however only 147 anglers completed the questionnaire. The low response rate may have biased results; the low catch rate and fishing pressure from section 3 could be due to the distance of that section from the welcome centre, discouraging people from returning the questionnaire. However, data from previous years, where reporting catch to the CGRBF was obligatory, also show that the majority of catches were made in section 2: 70% in 2004 and 65% in 2003 (CGRBF, 2004).

The analysis of the effectiveness of weirs and deflectors for increasing trout habitat in the Nicolet River was completed only during the summer at low flow, as is often the case. However, this approach has been criticized because it does not take into account physical habitat available or fish habitat preferences during high flow events or during winter (Gore and Nestler, 1988; Maki-Petays et al., 1997). Instream structures provide different flow conditions during high flow than they do at low flow (Biron et al., 2009), and winter ice cover affects the flow around instream structures (Huusko and Yrjana, 1997). Trout habitat usage also varies when discharge increases (Shirvell 1994; Pert and Erman, 1994; Holm et al., 2001) and when water temperatures drop (Heggenes et al., 1993; Huusko and Yrjana 1997; Maki-Petays et al., 1997). As such, this research only explains a portion of the effect of instream structure installation on trout habitat in the Nicolet River.

There were significant physical differences between pools that have been restored and the natural pools of the Nicolet. The instream structures were maintaining pools where previously there were none, and these pools had more cover than the nearby non-restored pools. The structures have, therefore, accomplished the physical habitat goals for which they were installed. There is some evidence that the physical habitat in the restored pools provided more preferred trout habitat than the non-restored pools, however the inability to observe trout utilizing the habitat makes drawing conclusions difficult. Over a decade has passed since the restoration of the Nicolet was completed and weekly trout stocking is still required to provide anglers with fish. This suggests that physical habitat limitations were not the only factor preventing self-sustaining trout populations. High summer water temperature was listed as one of the factors preventing high trout abundance prior to restoration (CGBRF, 1993). Despite riparian and upstream tree planting in the mid 1990s, water temperature during the summer remains high. Research in a small area of the restored reach of the Nicolet River in 2001 showed that water temperatures were above 27°C in most of that area, including in two moderately deep pools (Dolinsek and Biron, 2001). Deep pools have been shown to increase trout survival during high heat days (Matthews et al., 1994), however this likely remains a stress for trout in the Nicolet. More problematic for creating self-sustaining trout populations was the high fishing pressure. If the average number of fish caught per angler reported in the questionnaires (4.3) was representative of the anglers who did not complete the survey, then approximately 3030 trout were caught during the 2008 fishing season. As only 3084 trout were stocked, the potential number of fish surviving to reproduce was extremely low.

The Nicolet River restoration project has been a success in terms of altering physical habitat characteristics of the river that were deemed inhospitable to trout and the goal of improving angling opportunities has been met. However, a put and take fishery is the final result of an expensive restoration project.

Figures

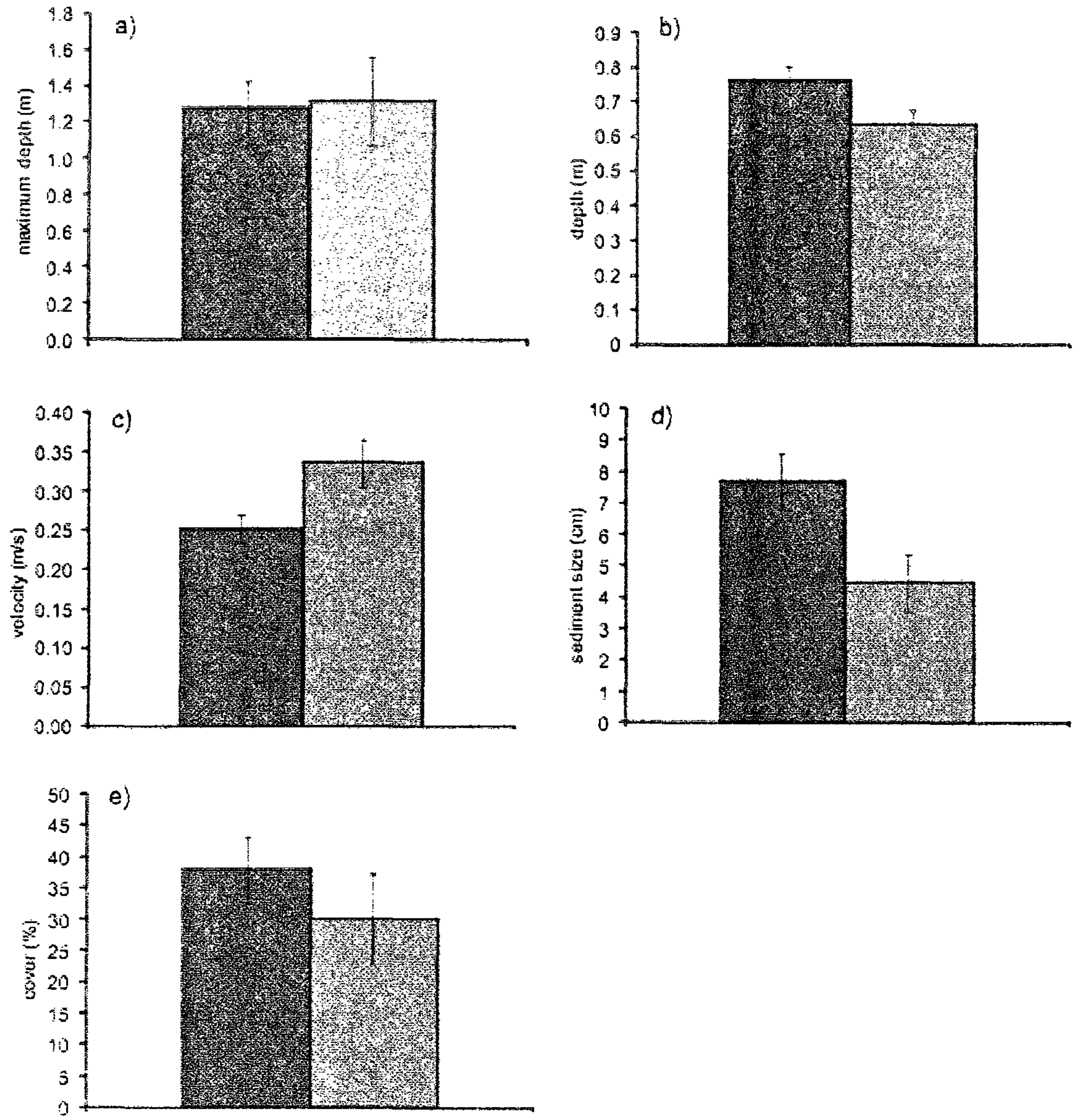

Figure 3. Mean and 95% confidence intervals of the mean of a) maximum pool depth, b) average pool depth, c) average water column velocity, d) sediment size and e) % cover for restored pools (dark grey) and non-restored pools (pale grey).

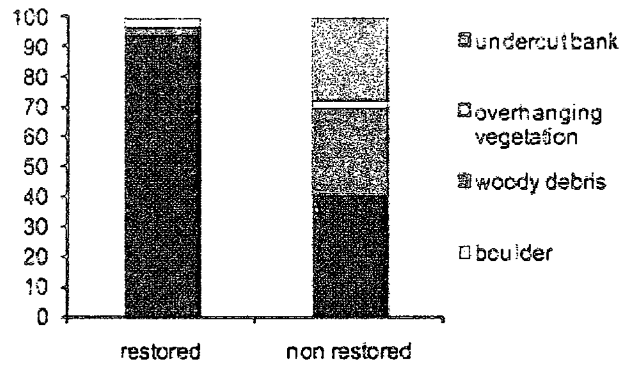

Figure 4. The percent of cover available from various sources in restored and non-restored pools.

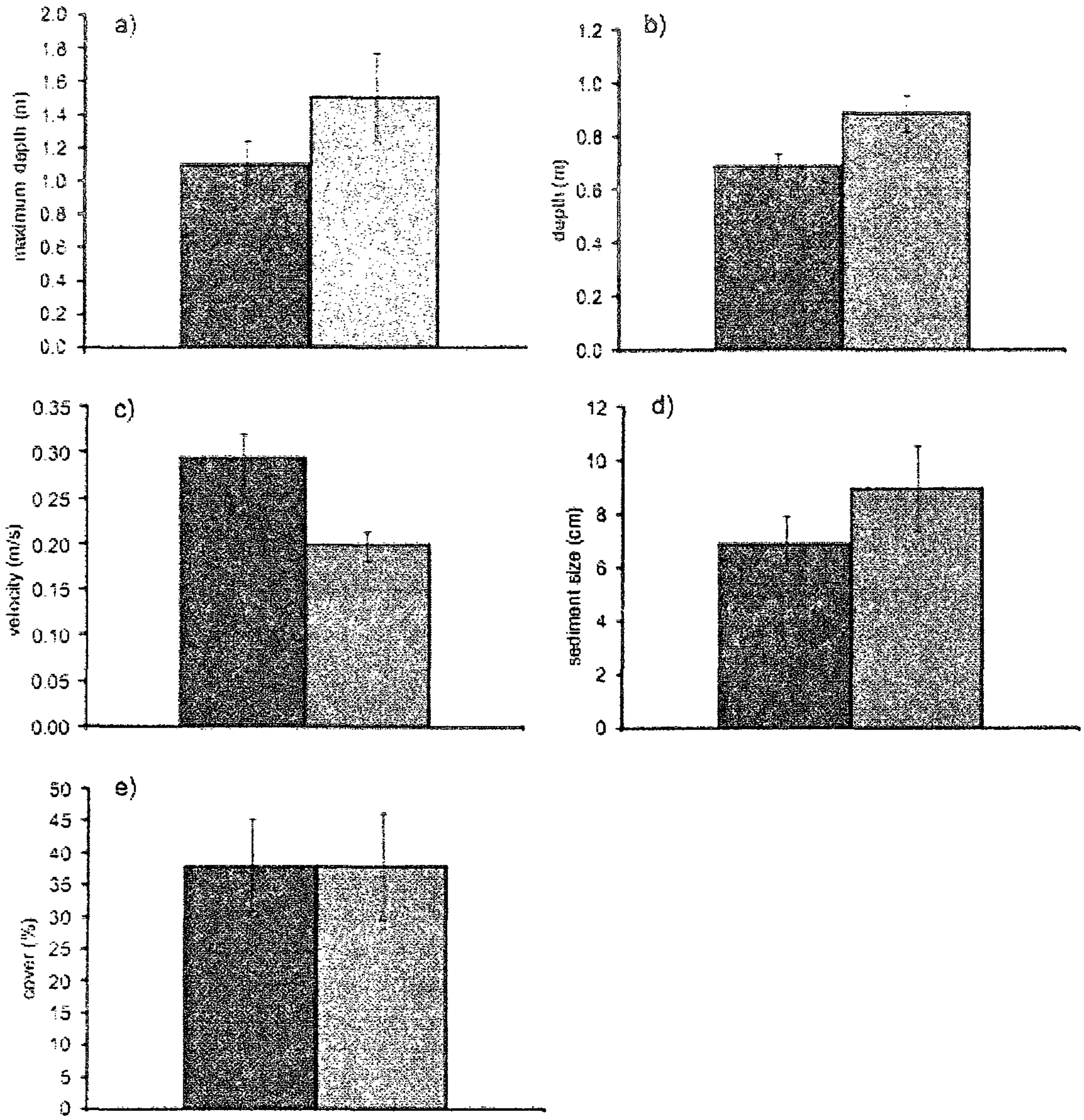

Figure 5. Mean and 95% confidence intervals of the mean of a) maximum pool depth, b) average pool depth, c) average water column velocity, d) sediment size and e) % cover for deflector pools (dark grey) and weir pools (pale grey).

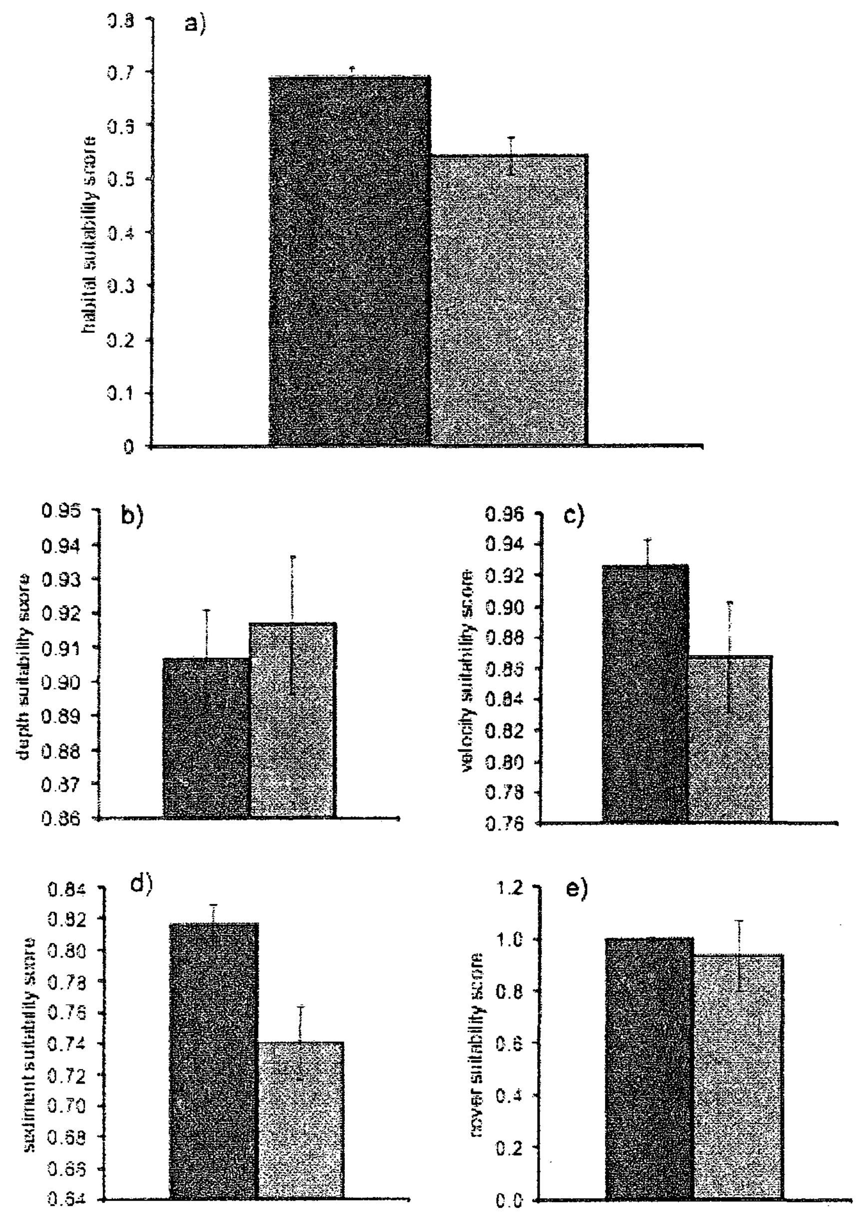

Figure 6. Mean and 95% confidence intervals of the mean from Raleigh et al. (1984) rainbow trout suitability scores, a) total suitability, b) depth suitability, c) velocity suitability, d) sediment size suitability and e) cover suitability. Restored pools are shown in dark grey and non-restored pools in pale grey.

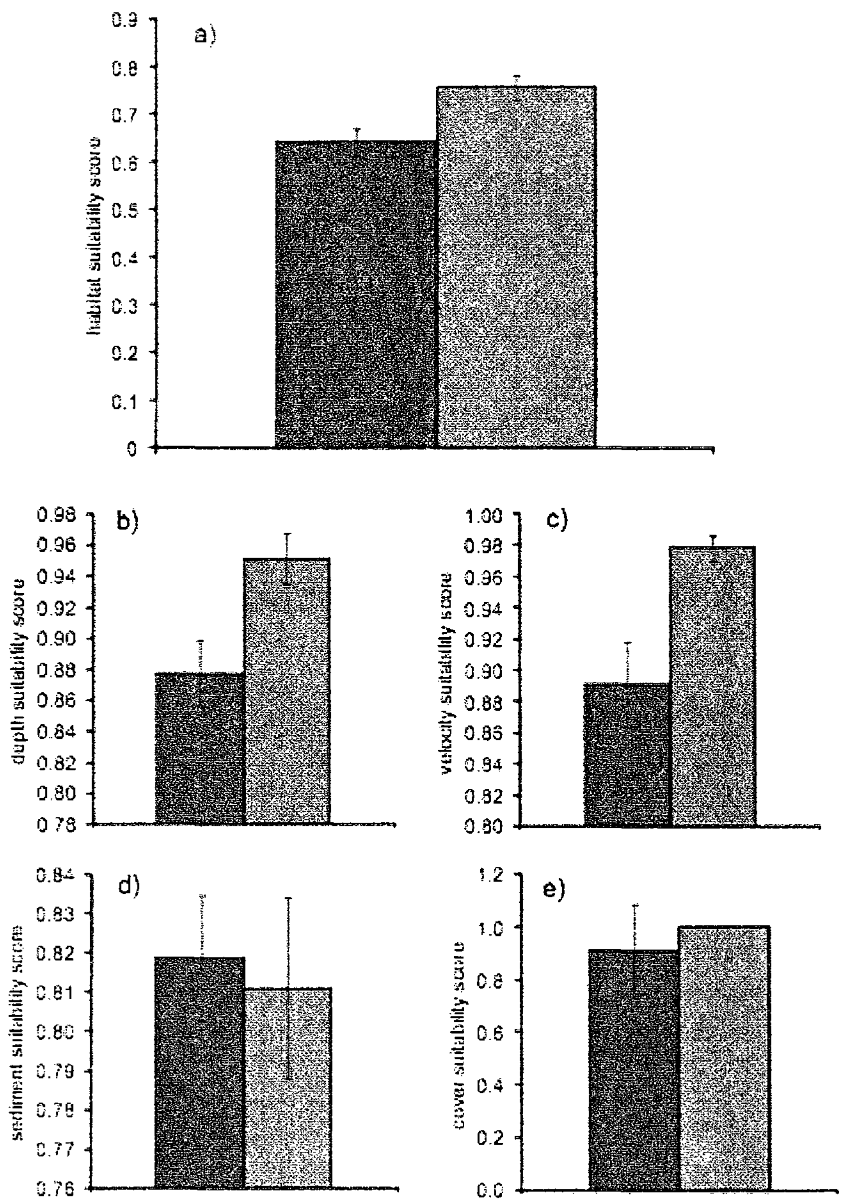

Figure 7. Mean and 95% confidence intervals of the mean from Raleigh et al. (1984) rainbow trout suitability scores, a) total suitability, b) depth suitability, c) velocity suitability, d) sediment size suitability and e) cover suitability. Deflector pools are shown in dark grey and weir pools in pale grey.

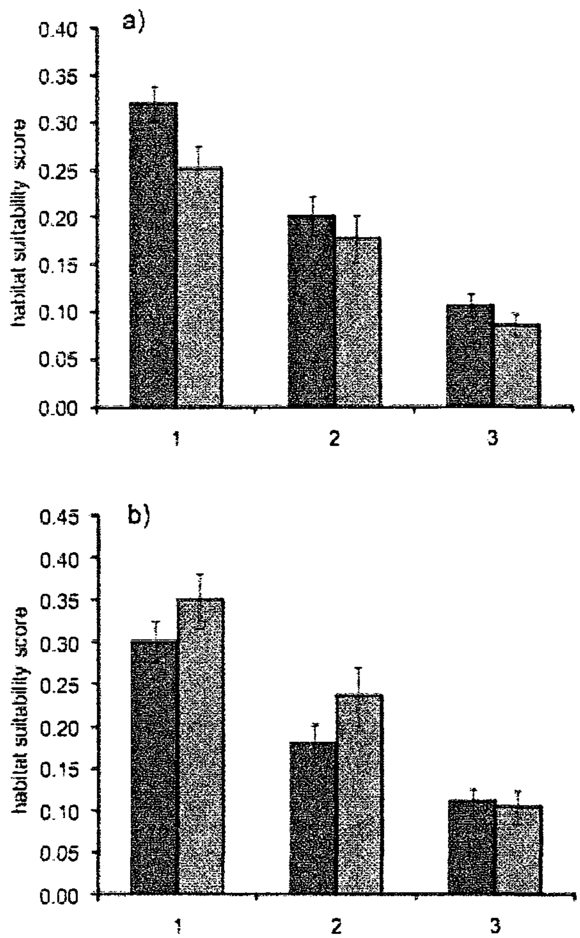

Figure 8. Mean and 95 % confidence intervals of the mean of brown trout total habitat suitability scores of a) restored pools (dark grey) and non-restored pools (pale grey) and b) deflector pools (dark grey) and weir pools (pale grey) using 3 different suitability curves. 1) Raleigh et al. 1986, 2) Vismara et al. 2001 and 3) Strakosh et al. 2003.

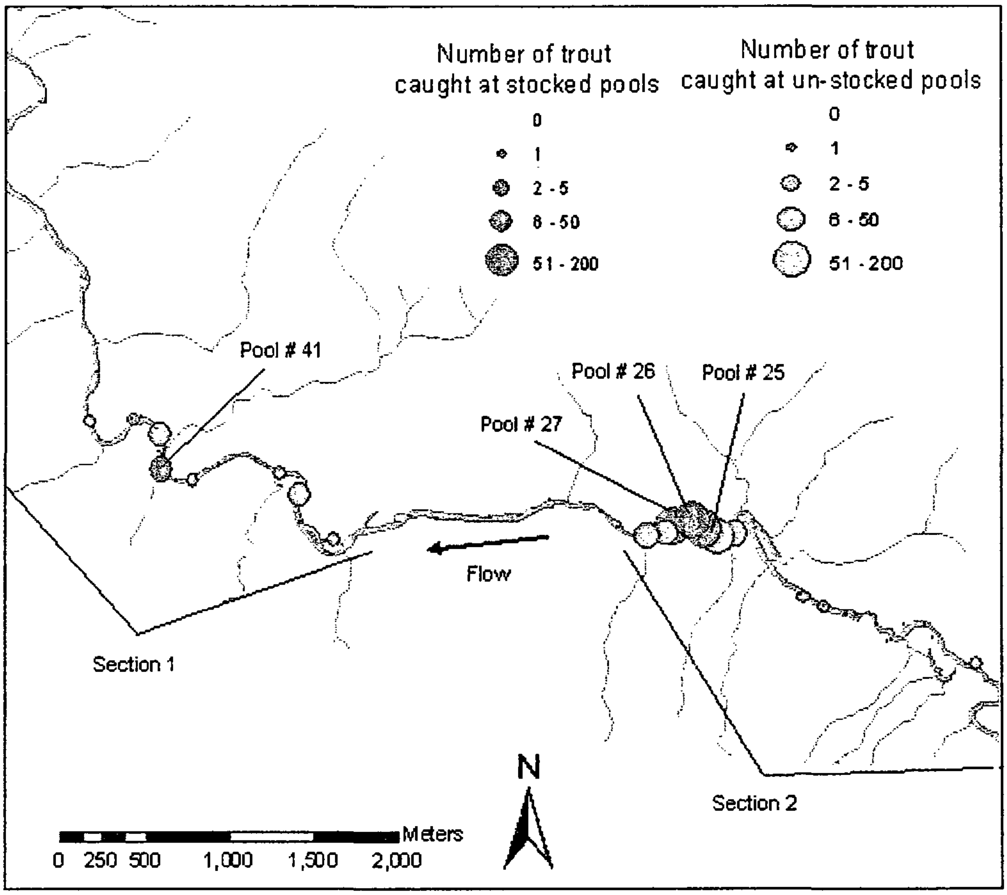

Figure 9. Sections 1 and 2 of the Nicolet River showing the number of trout caught in different pools.

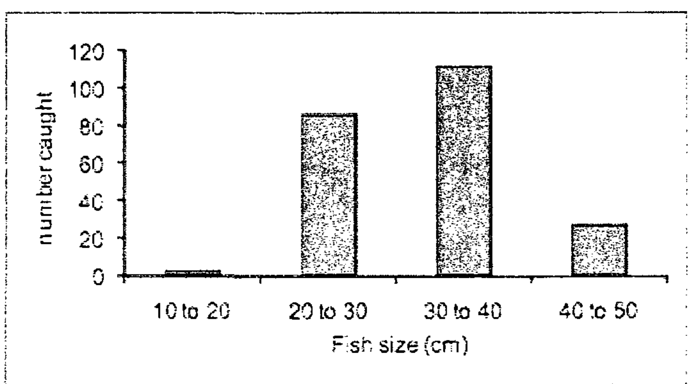

Figure 10. The number of trout caught belonging to different length categories.

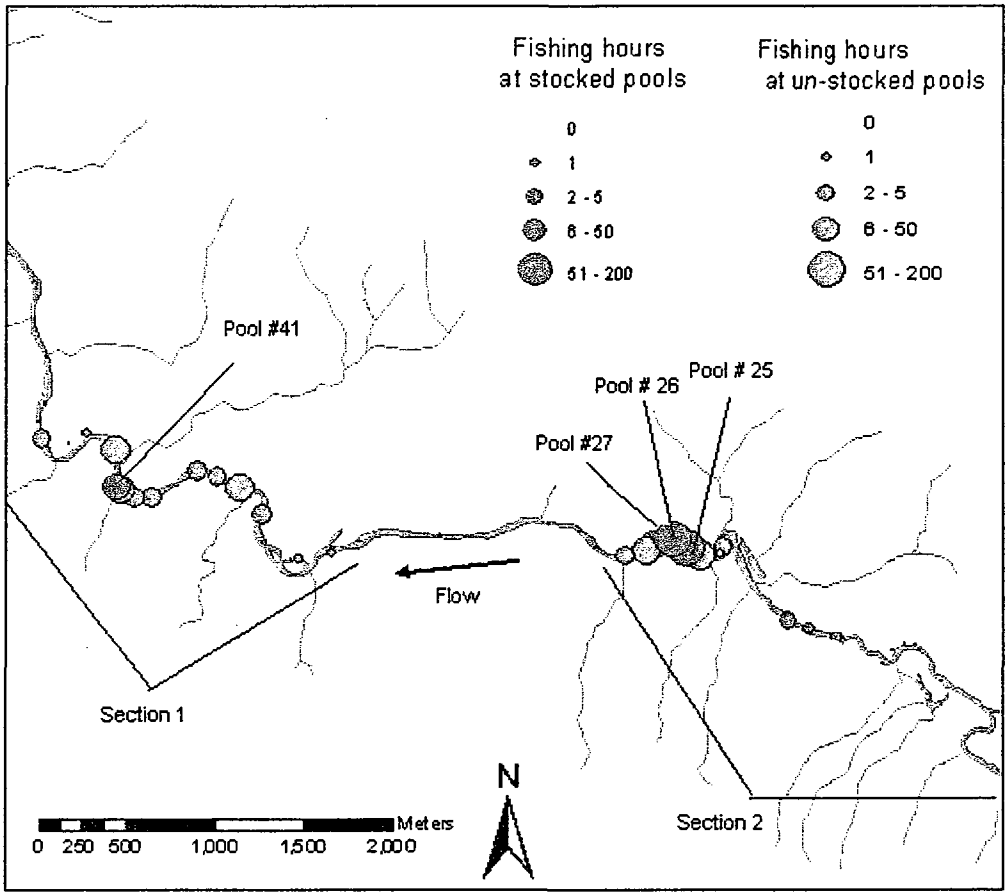

Figure 11. Sections 1 and 2 of the Nicolet River showing the number of angler hours spent at the different pools.

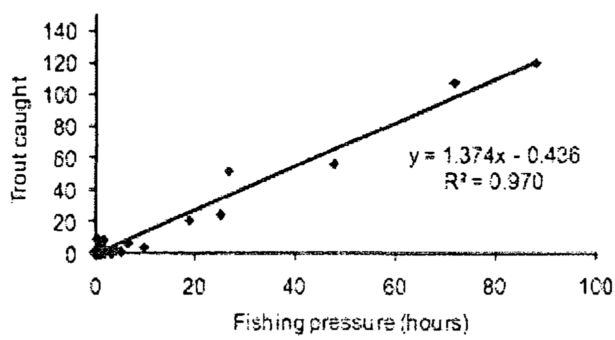

Figure 12. The relationship between hours spent fishing at a pool and the number of trout caught.

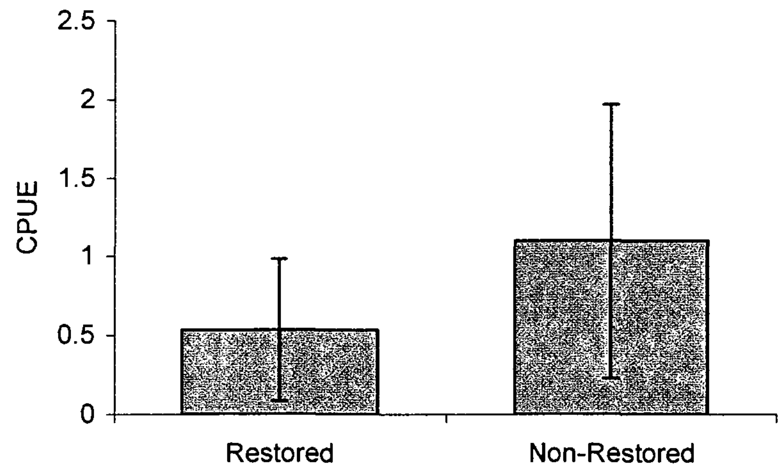

Figure 13. Mean catch per unit effort (CPUE) for restored and non-restored pools and 95% confidence intervals of the mean.

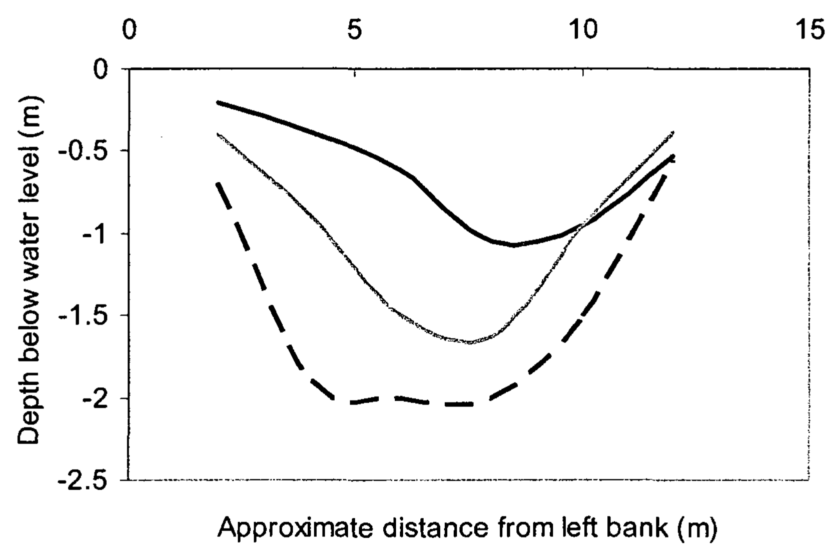

Figure 14. Typical cross section of different pool types. Non restored pools are shown with the black solid line, deflector pools with the grey line and weir pools with the dashed black line.

5. Meta-analysis of Salmonid Stream Restoration Projects[4]

5.1 Physical effects of instream structures

Fifty-three percent of studies installed only one type of structure, 28% used a combination of two structures, 13% combined three structures, 1% combined all 5 structures and 4% did not specify the type of structure(s) installed. The most common instream structures used were cover structures (88), followed by deflectors (87), weirs (69), LWD (46) and boulder placements (41).

The installation of instream structures had significant effects on the physical habitat characteristics of the streams. Overall, there was a significant increase in pool area (mean effect size (L) = 0.65; $T_{72} = 5.56$, $p<0.0001$; Fig. 15a), a decrease in riffle area (mean L = -0.52; $T_{38} = -4.87$, $p<0.0001$; Fig.15b), an increase in the number of pieces of LWD in the river (mean L = 0.73; $T_{46}=6.61$, $p<0.0001$; Fig. 15d), an increase in channel depth (mean L= 0.29; $T_{37}=2.93$, $p=0.006$; Fig. 15e), and an increase in percent cover (mean L = 1.14; $T_{25}=4.67$, $p<0.0001$; Fig. 15f). However, the presence of instream structures had no significant effect on stream width (mean L = -0.01; $T_{75} = -0.11$, p = 0.91; Fig.15c).

Projects with multiple structures increased pool area more than projects with only one type of structure (ANOVA, $F_{[1,73]}=38.5$, $p<0.0001$; Fig. 15a). For all other physical variables, however, there were no significant differences between the effect sizes for projects with multiple and single structures (ANOVA, all p-values > 0.08).

To investigate whether the five structure types had different effects on the physical habitat of streams, we compared the effect sizes only for single-structure

[4] Part of Whiteway et al. (submitted for review)

projects (i.e. the light grey bars in Fig. 15). Effect size did not differ significantly between structure types for any of the six abiotic variables (ANOVA, all p values >0.4) (Fig.15). For visual purposes, we also plotted the mean effect size with 95% confidence intervals for all structure types, regardless of whether they were used alone or in combination (i.e. dark grey bars in Fig. 15).

5.2 *Effects on salmonids*

Overall, average salmonid density and biomass increased following instream structure restoration, with mean effect sizes of 0.51 (T_{210}=6.86, p<0.0001) and 0.48 (T_{77}=5.85, p<0.0001) respectively (Fig. 16 a and b). However, 56 projects (27%) showed a decrease in density following restoration and 10 showed a decrease in biomass (13% of those that monitored biomass). There was no significant difference between density or biomass effect size for projects that installed only one type of structure compared to those that installed multiple structure types (ANOVA, $F_{[1,199]}$=2.34, p=0.128 and $F_{[1,32]}$=2.73, p=0.11), nor was there a significant difference in density or biomass effect among structure types (ANOVA, $F_{[4,108]}$=0.64, p=0.63 and $F_{[4,17]}$=1.10, p=0.39 respectively).

The density effect size varied significantly between species of salmonid (ANOVA, $F_{[6,327]}$=5.20, p<0.0001) (Fig.17). Based on a Tukey-Kramer post-hoc test, the effect size was largest for rainbow trout (1.48, n = 11), and smallest for steelhead trout (0.15, n = 50) (Fig. 17). Ninety-five percent confidence intervals indicate that all species except brook trout and steelhead trout responded positively to the restoration efforts. Size classes responded differently to restoration, with an increasing linear trend among the three salmonid size classes (ANOVA, $F_{[2,319]}$ = 2.93, P = 0.055) (Fig. 18).

Backward stepwise regression was used to investigate the relationship between change in the 6 abiotic variables (pool area, riffle area, width, LWD, depth and cover) and biotic variables (density and biomass). Depth effect size was the only significant predictor of density effect size, although the R^2 value was low (0.11, n = 38, p=0.037; Fig. 19a). Similarly, pool area effect size was determined to be the only significant predictor of biomass effect size (R^2=0.51, n=8, p=0.046; Fig. 19b).

5.3 *Monitoring programs*

The number of projects monitored decreased with increasing project age: 86 projects were monitored 1-year post construction while fewer than five projects were monitored 10 years post construction (Fig. 20). None of the projects were monitored for over 20 years and 45% of all projects were only monitored once. The results for projects over 5 years post construction were combined due to small sample sizes. There was a significant difference in density effect size based on project age (ANOVA, $F_{[4,188]}$= 2.59 p=0.04). The density effect size was greatest in projects monitored 2 years after completion.

Project cost was only reported in 24% of studies (51 out of 211). From this sample the mean project cost, indexed to the dollar value in 2000, is USD $127 490 with a median cost of $36 295. The average cost per metre of restored river length is $34.85 with some projects spending less than $5 per metre of stream restored and others upwards of $100. There was no relationship between total project cost, or project cost per metre of stream restored, and change in salmonid density (R^2=0.008, n =54, p=0.52 and R^2=0.003, n =49, p=0.74 respectively).

A comparison of results published in the primary literature (n = 63) and in the grey literature (n = 148) revealed a slightly larger mean effect size of instream structures on salmonid density in the primary literature (0.55 compared to 0.49), but this difference was not significant (ANOVA, $F_{[1,209]}=0.06$, $p=0.81$).

5.4 *Discussion*

Meta-analysis of a large number of restoration projects showed that 73% of projects resulted in increased local salmonid densities and 87% in increased biomass, with an average effect size of 0.51 and 0.48, respectively. These findings are in agreement with the qualitative findings of previous studies (e.g. Hunt, 1988; Keeley et al. 1996; McCubbing and Ward, 1997; Roni et al. 2008). The 27 % of projects that showed a decrease in overall salmonid density and 13% of projects that recorded a decrease in biomass following restoration did so for a number of reasons. Poor study design (e.g. badly chosen reference reach, short monitoring program), unexpected physical changes (e.g. decreased depth, decreased spawning gravel), structural failure, and unexpected events (e.g. 100 year flood, fish kill, settling pond blowout) were listed as potential reasons for decreased density (Olsen et al., 1984; Hunt, 1986; Hunt, 1988; House et al., 1989; Linløkken, 1997; Reeves et al., 1997; Thorn and Anderson, 2001; Johnson et al., 2005). Increased fishing pressure in the restored reaches was occasionally considered the cause of poor study outcomes (Hunt, 1988; Avery, 2004), but was usually not been measured. A number of studies reported that though overall salmonid density decreased, the density of large fish had increased and that the larger decrease in fish under 10cm was responsible for the overall trend (Avery, 2004; Rosi-Marshall et al., 2006). However, the majority of studies that showed decreased salmonid densities following restoration provide no reason for this outcome. The large variation in how salmonids responded to stream restoration is in agreement with previous observations (Roni et al., 2008; Stewart et al., 2009).

In contrast to our results, Stewart et al. (2009) concluded that the "widespread use of instream structures for restoration is not supported by the current scientific evidence base" (p. 939). Stewart et al. (2009) also conclude that instream structures are more effective on small streams (<8m in width), whereas our analysis showed no difference in density effect size between streams of different widths; in fact streams over 8m in width had a larger mean density increase following restoration than smaller streams (L=0.59, 95% C.I.= 0.28 – 0.90, n=56 compared to L=0.41, 95% C.I.=0.24 – 0.58, n=108). A re-analysis of Stewart et al.'s (2009) data using L (eq. (1)) as the measure of effect size was conducted to reconcile these different findings. Note that we have removed from the dataset the four projects in which either engineered instream structures were not used or no measures of abundance was reported (Mesick, 1995; Scruton et al., 1998; Wu et al., 2000; Wang et al., 2002). We have also corrected a few errors in their data set: the treatment and control sections were reversed in Binns (2004); the n value listed corresponded to fish counted rather than river reaches in Linløkken (1997); not all data from Gargan et al. 2002 were used. The results of our reanalysis show a clear positive effect size of 1.1 for instream structures (T_{28}= 4.90 p<0.0001), markedly larger than the average effect size in this study (0.51).

It is difficult to distinguish between increased fish abundance due to increased recruitment, survival or growth and increases caused by immigration and redistribution within the reach (Gowan and Fausch, 1996). In order to measure changes in population size, the spatial and temporal scale of the study must be fairly large (Stewart et al., 2009). Unfortunately, many studies that attempt to determine the effect of instream structures on salmonid abundance are of short duration and at the reach rather than watershed scale. We excluded studies that specifically measured habitat preference, but did include studies measuring changes in abundance at the reach scale or for only a year following restoration. It is likely, therefore, that some of the studies reporting an increase in salmonid density are due to redistribution of fish. However, as Gowan and Fausch (1996) point out, immigration to preferred habitat is likely to increase the watershed-wide trout population.

As expected, the installation of instream structures resulted in significant changes to the physical stream habitat. An increase in pool area, volume or frequency is a typical goal in instream structure installation (Roni et al, 2008). Our analysis indicated that all types of instream structures have the potential to increase pool area in a stream. Cover, which is a key salmonid habitat variable (Lewis, 1969), can obviously be improved by cover structures but also by weirs and deflectors (the increase for boulder structures was not significant). Surprisingly, none of the projects analysed in this study measured the change in cover following the installation of LWD structures, despite the fact they are often installed to increase cover (Cederholm et al., 1997). Increased mean channel depth is another common restoration goal; deflectors, cover structures and boulder placements were all found to significantly increase depth while weirs showed a non-significant increase in depth.

We found no significant effect of structure type on the observed change in salmonid density. Other studies that have directly compared different structure types have obtained conflicting results. Some studies suggest that deflectors outperform other structure types (e.g; Ward and Slaney, 1981; Hunt, 1988), others that boulder placements improve salmonid densities more than deflectors or weirs (e.g. Olsen et al., 1985), and some have concluded that weirs are preferable (e.g. Van-Zyll-De-Jong et al, 1997). As different structures target different aspects of habitat quality, the best structure for increasing salmonid densities will be the one that best ameliorates the physical habitat deficiencies in an individual stream. It is therefore difficult to provide general recommendations without thorough knowledge of the specific problem. Our results imply that stream restoration practitioners are adept at picking the correct restoration technique, to create the correct habitat for the particular stream, but no one approach will work for all streams.

Surprisingly, despite the clear effect of instream structures on both physical habitat variables (see Fig. 15) and salmonid density (see Fig. 16a), change in habitat variables are not good predictors of changes in salmonid density. This corroborates the results of Pretty et al. (2003) who found few significant relationships between physical and biological variables in their study of thirteen restored streams and raises the question: "what causes changes in salmonid density"? It is likely that multiple habitat changes are required to provide adequate salmonid habitat. As for structure type, habitat variables that contribute to increased salmonid density likely vary from project to project, making it very difficult to establish a causal relationship from a large database which includes rivers in diverse environments.

The observation that larger salmonids respond most strongly to instream structures suggests that they provide habitat that is particularly suited to adult salmonids. Previous studies have similarly documented better responses of larger fish to instream structures (e.g. Hunt, 1988; Gowan and Fausch, 1996) and many studies specifically seek to increase legal (often over 15cm) size trout (Burgess, 1985; Hunt, 1988). Smaller sized trout do not show a strong preference for pool habitat (Bisson et al. 1988), which is likely why density increases are lower for these size classes. That change in pool area and biomass were more strongly correlated than pool area and density also suggests that increased pool area results in preferable habitat for larger salmonids.

Instream structures are typically designed to last at least 20 years (Frissell and Nawa, 1992) though different structures have varying rates of structural failure (Roni et al., 2002). While there is a consensus that more long-term monitoring on the effect of instream structures is needed (Frissell and Nawa, 1992; Kondolf and Micheli, 1995; Bernhardt et al., 2005; Roni et al., 2008; Stewart et al., 2009), the duration of monitoring projects remains short, averaging only 3 years. There are significant problems with determining project effectiveness when monitoring is done for only 1 or 2 years post-restoration as it may take up to 5 years after restoration work is completed before the full effect on salmonids can be seen (Hunt, 1976, Kondolf, 1995). Surprisingly our results show that the mean density effect size is largest for projects that have been in place for 2 years, and that the projects that monitor for 5 years or longer show a significantly lower density increase. Kondolf and Micheli (1995) recommend at least 10 years of post-restoration monitoring to measure physical changes in the river channel, since low recurrence floods are likely to alter the channel and because geomorphological adjustments following the installation of instream structures may take some time. The length of monitoring should also be determined based on the size and dynamic nature of the channel since it takes longer for geomorphological adjustments to take place on large rivers.

The median cost of the projects in our analysis was \$36 295, close to the \$20 000 median cost of over 6000 instream habitat improvement projects compiled by Bernhardt et al. (2005). Costs were lower for projects that were able to use volunteer labour or readily available construction material. Higher costs can be expected for projects on inaccessible river reaches and projects that require the use of heavy machinery. There is, however, no evidence to suggest that higher spending leads to higher project success, as measured by increased salmonid density.

There is often a concern that successful restoration projects are more likely to be reported in the primary literature than unsuccessful projects (Kondolf and Micheli, 1995). While it is impossible to analyze projects that have not been reported in any literature, comparing results that were published in the grey literature with those published in the primary literature allowed us to discount this potential bias.

This quantitative review shows that instream structures are capable of altering physical habitat and increasing salmonid abundance. Following the installation of instream structures, streams tend to get deeper (mean effect size (L) = 0.31), have more cover (L = 1.2), more pieces of LWD (L = 0.73) and to have more pool (L = 0.65) and less riffle area (L = -0.53). The mean effect size of instream structures on salmonid density is 0.51, corresponding to a density increase of 164%. However, the high variability in salmonid response highlights the importance of a thorough understanding of the stream system and of the habitat requirements of the target species. Change in depth is the best predictor of change in salmonid density, but R^2 values remain very low (0.11), while change in pool area best predicts change in biomass. Density response to restoration structures varies between species, with the largest effect size observed for rainbow trout. Concern remains regarding the long-term impact of instream structures on both the physical characteristics of rivers and the populations of salmonids. This concern will not be addressed until a greater number of projects conduct long term monitoring, as currently only 10% of the projects are monitored 10 or more years after installation.

Figures.

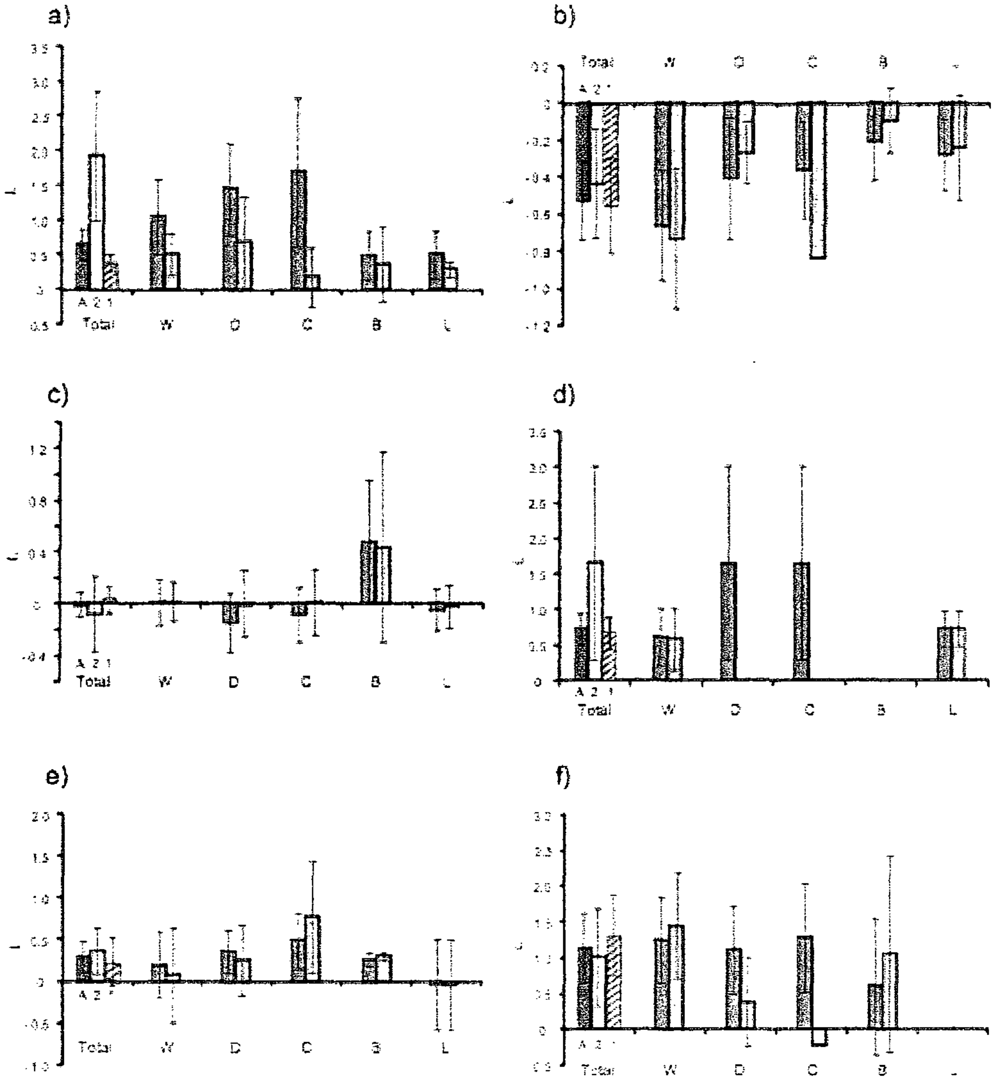

Figure 15. Effect of different types of instream structures (A= all types, 2= projects that used 2 or more structure types, 1= projects that used only 1 structure type, W=weir, D=deflector, C=cover structure, B=boulder placement, L= LWD on the mean (+/- 95% confidence interval) effect size ($L = \ln(x_{tr} / x_c)$) of a) pool area, b) riffle area, c) stream width, d) pieces of LWD, e) stream depth and f) cover. Within each individual structure type the dark grey bar represents the mean for all projects that used that structure

(whether or not another type of structure was used) and the light grey represents the mean for projects that only used that type of structure.

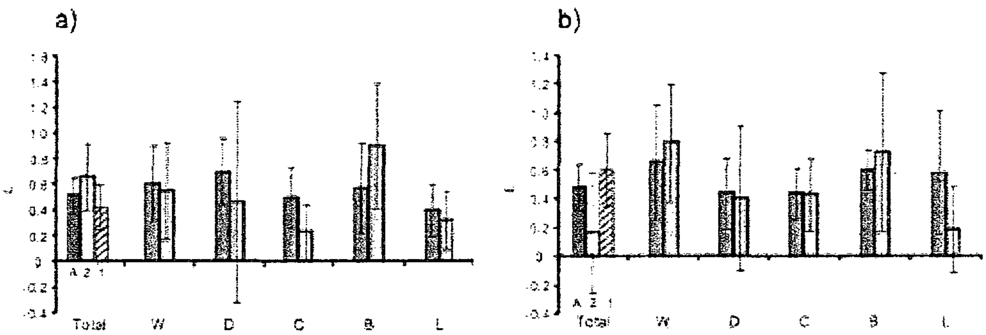

Figure 16. The effect (mean L, 95% C.I.) of structure type on a) salmonid density and b) biomass. A= all types of instream structures combined, 2= projects that used 2 or more structure types, 1= projects that used only 1 structure type, W=weir, D=deflector, C=cover structure, B=boulder placement, L= LWD. Within each individual structure type the dark grey bar represents the mean for all projects that used that structure (whether or not another type of structure was used) and the light grey represents the mean for projects that only used that type of structure.

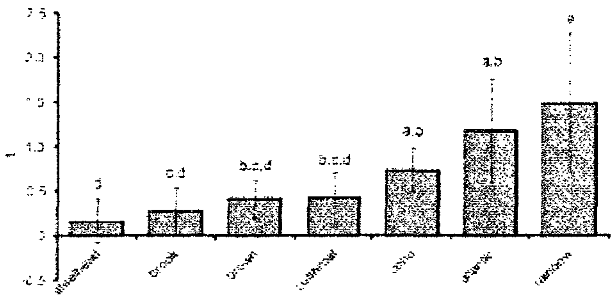

Figure 17. The effect (mean L, 95% C.I.) of instream structures on density of different salmonid species. Similar letters indicate that the mean L does not differ significantly between species.

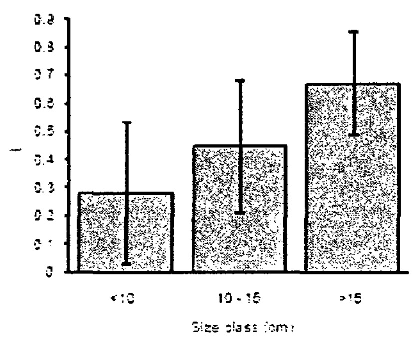

Figure 18. The effect (mean L, 95% C.I.) of instream structures on density of different salmonid lengths (<10, 10 to 15, and >15cm).

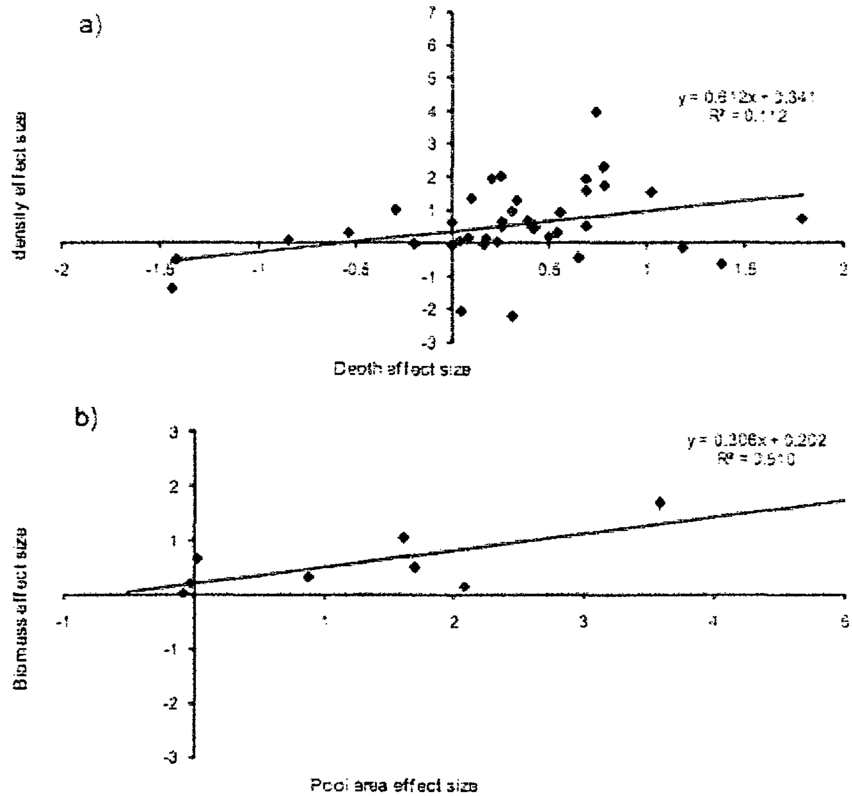

Figure 19. Linear regression of a) salmonid density effect size against depth effect size and b) salmonid biomass effect size against pool area effect size.

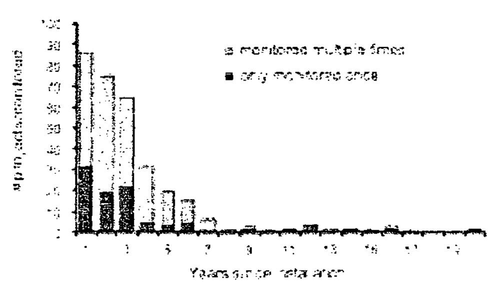

Figure 20. Number of projects monitored in each year following restoration, separated into projects monitored only once and those monitored more than once.

6. Conclusion

Data from the Nicolet River Restoration project as well as that obtained though meta-analysis of 211 other case studies show that instream structures can alter the physical characteristics of streams. There is conclusive evidence that pool area and depth increase, and that riffle area decreases. Average stream depth also increases following the installation of instream structures. Cover, in general, increases following restoration, however there was no more cover in restored pools in the Nicolet than in nearby non-restored pools. LWD was also shown to increase significantly in meta-analysis, though evidence form the Nicolet River suggests that there is less LWD in the restored pools than in nearby pools. Decreased pool velocity and increased sediment size were observed in the Nicolet restoration project, however these variables were rarely measured in other projects and so were not included in the meta-analysis. Width is unaffected by the installation of instream structures. Despite differences observed between the habitat created by weirs and deflectors in the Nicolet River, there is no evidence, when a large number of studies are considered, that structure type has a significant impact on the physical characteristics of a stream.

Though we were unable to determine whether trout habitat use differed in restored and non-restored pools of the Nicolet River, there is evidence that preferred habitat is more prevalent in the restored pools. Meta-analysis shows that both trout density and biomass increase following restoration. Linking physical habitat changes to biological changes proved difficult, suggesting that the required physical changes to increase fish populations vary among rivers and that the ideal structure type differs between projects.

The Nicolet Restoration project highlights some of the difficulties in determining the effectiveness of instream structures when rivers are stocked and have high fishing pressure. It is also a case in which the instream structures appear to have been successful, however there may be other factors within the watershed that are preventing the ability of local trout populations to persist without frequent stocking. The usefulness of instream structures is in creating required habitat quickly to prevent the loss of species or populations. They are not designed as a permanent solution; watershed management to correct wide reaching problems should remain a central part of any restoration project.

In order to adequately assess the usefulness of instream structures, data on the physical and biological effects in the long term are required. The length of time required for complete monitoring will vary greatly depending on the size and stream power of the river. Guidelines are needed to advise project managers on creating good post-management assessment plans. As well the results of a larger number of case studies need to be made available so that future studies can examine the effectiveness of restoration structures in greater detail. Of particularly use would be detailed information on the geomorphic context of the rivers in which restoration took place. Future research should also examine the cost to benefit ratio of restoration projects: a large amount of money has been spent installing structures and it would be interesting to know how much of this cost has been recouped through angling fees, improved ecosystem services and increased tourism.

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Appendix 1 - Nicolet River Angling Survey

Nicolet River Angling Survey

The following short survey is designed to investigate the effect of the restoration structures (deflectors and weirs) on trout habitat. Additionally the information will be useful for assessing the current stocking effectiveness.
We would very much appreciate you taking the time to fill it out and submitting it to the CGRBF.

Date: ______________________

Please fill in one line per trout caught

Species (circle one)	Length	Pool #	Released?
brook / brown / rainbow			yes / no
brook / brown / rainbow			yes / no
brook / brown / rainbow			yes / no
brook / brown / rainbow			yes / no
brook / brown / rainbow			yes / no
brook / brown / rainbow			yes / no
brook / brown / rainbow			yes / no
brook / brown / rainbow			yes / no
brook / brown / rainbow			yes / no
brook / brown / rainbow			yes / no

Please fill in one line for each pool that you fished at (even if you did not catch any trout)

Pool #	Hours spent fishing

Any additional comments:

____

____

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Please note that by filling out and submitting this survey you consent to have this information used in research conducted by the CGRBF and Sarah Whiteway, applicant for Masters of Science from Concordia University.
For further information please contact Sarah Whiteway at 514-848-2424 ext.2507 or sarahwhiteway@gmail.com

Questionnaire pour Pêcheurs - Rivière Nicolet

Le questionnaire qui suit a pour but d'examiner l'effet des structures de restauration (déflecteurs et seuils) sur l'habitat des truites. L'information recueillie sera aussi utile pour analyser les pratiques d'ensemencement dans la riviere Nicolet.
Nous apprécierions beaucoup si vous pouviez prendre le temps de remplir ce questionnaire et de le soumettre à la CGRBF.

Date: ______________________

S.V.P., remplissez une ligne par truite

Espèce (encercler une seule espèce)	Longueur	# fosse	Relâchée?
mouchetée / brune / arc-en-ciel			oui / non
mouchetée / brune / arc-en-ciel			oui / non
mouchetée / brune / arc-en-ciel			oui / non
mouchetée / brune / arc-en-ciel			oui / non
mouchetée / brune / arc-en-ciel			oui / non
mouchetée / brune / arc-en-ciel			oui / non
mouchetée / brune / arc-en-ciel			oui / non
mouchetée / brune / arc-en-ciel			oui / non
mouchetée / brune / arc-en-ciel			oui / non
mouchetée / brune / arc-en-ciel			oui / non

S.V.P., remplissez une ligne pour chaque fosse pêchée (même si vous n'avez rien attrapé)

# fosse	Heures passées à pêcher

Commentaires additionnels:

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S.V.P., notez qu'en remplissant ce questionnaire, vous consentez à ce que cette information soit utilisée dans les recherches du CGRBF et de Sarah Whiteway, candidate à la maîtrise géographie, urbanisme et environment à l'Université Concordia.
Pour plus d'information, contacter Sarah Whiteway au 514-848-2424 ext.2507 ou sarahwhiteway@gmail.com

Appendix 2 - Studies included in meta-analysis

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