Fish biodiversity and morphological quality in small agricultural streams of Montérégie, Québec

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ABSTRACT

Fish biodiversity and morphological quality in small agricultural streams of Montérégie, Québec

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Stream channelization and modification is a widespread practice on agricultural land in Montérégie, Quebec, however, it is well known that channel simplification reduces the variability of instream habitat complexity and affects the biodiversity of these streams. Despite over 30,000 km of streams being subjected to channelization-type development in Quebec, very little is known about the extent of stream degradation and effects it may have on local geomorphology and ecology. The Morphological Quality Index (MQI) is a tool used to measure a stream's hydrogeomorphological quality and has been shown to be a reliable predictor to help assess habitat quality in small headwater streams. The purpose of this research is to determine the health of fish communities in small streams in Montérégie and assess whether there is a significant relationship between biological communities and the MQI. Over 1,220 fish samples and 85 stream reaches with drainage areas less than 130 km^2 were analysed in Montérégie. Results showed that small streams in Montérégie were higher in fish biodiversity than expected, with clear relationships between the fish metrics and proportion of land use in forested or agricultural categories. The MQI was also able to predict biological communities with up to an $R^2 = 0.50$, which shows that the MQI could be used as a reliable tool to efficiently assess streams while providing insight on how stream modification affects overall biodiversity in agricultural watersheds.

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1. General Introduction

Freshwater species are one of the fastest declining groups of biodiversity in the world, with habitat degradation being one of the leading causes (Tickner et al., 2020). Flow modification and degradation of habitat which includes excavations, dams, and river straightening are some of the leading causes of population and range decline in freshwaters species (Dudgeon et al., 2006). This has become increasingly apparent in agricultural landscapes, for example in North America, a vast expanse of land has been turned into modern agricultural systems. Croplands have replaced more than 98% of North American prairies and large swaths of forests (Blann et al., 2009). Stream channelization practices are common in agricultural landscapes which involve the deepening and straightening of natural river channels, usually for the purpose of flood control, drainage, and navigation (Gregory, 2001). However, stream channelization is also known to reduce in-stream heterogeneity such as channel morphology and flow conditions (Frothingham et al., 2001; Heatherly et al., 2007; Lau et al., 2006). For example, stream straightening increases the risk of concentrated and sudden high flow velocities during heavy precipitation within the channel that can cause increased erosion and deposition of sediments and organisms further downstream (Schowalter, 2022). The deepening of channels can also cause the elimination of natural flood pulses which sustain connected habitats such as wetlands (Steiger et al., 1998). Impoundments or dams can disrupt the natural sequence or presence of pool-riffle complexes which provide varied habitat for different organisms (Schowalter, 2022). And finally, the removal of riparian vegetation can expose streams to increased sunlight, higher temperatures, and greater risks of algal blooms which can cause decreased resources and deaths in organisms inhabiting these streams (Schowalter, 2022). Better understanding is needed between geomorphological characteristics in agricultural streams and how they affect biological communities to conserve and restore these habitats.

2. Literature Review

2.1 Fish communities in agricultural streams

Considering how these stream habitats vary greatly from their natural counterparts, we expect important influences and modifications to the aquatic communities that inhabit them. Indeed, fish in agricultural streams were found to have a negative correlation between the health of fish communities and the amount of agricultural land in the watershed (Wang et al., 1997). It was found that recently channelized streams had lower abundances of species due to insufficient habitat availability (Talmage et al., 2002). Piedmont watersheds in the U.S also found that agriculture is a good predictor for species diversity and abundance in mainstream reaches where the former was found to be significantly lower than in forested areas (Walser & Bart, 1999). However, agriculture showed to be a poor predictor of species diversity and abundance in headwater reaches suggesting that headwater reaches are less susceptible to agricultural influences than mainstream reaches (Walser & Bart, 1999).

Although there was a clear decline in biotic integrity when the percentage of agricultural land use exceeded 50% in the watershed, some sites maintain good biotic integrity even when agricultural land use exceeded 80%. These 'good sites' found in highly agricultural areas shared similar qualities in that they maintained rocky substrates and were not channelized (Wang et al., 1997). This suggests that channelization could be a limiting factor to good biotic integrity along with other in channel characteristics such as substrate type. A local study conducted by the Ministry of Environment in Quebec also investigated the state of fish communities in agricultural streams compared to forested areas in the streams of Saint-George, Quebec (Richard & Giroux, 2004). They found that there was a higher presence of pollution tolerant species, omnivores, and 10-20% of fish had physical deformities in agricultural areas. For streams in forested areas, the biotic integrity of fish was found to be much higher (Richard & Giroux, 2004). Channel morphology appears to have a significant influence on fish assemblages, with channelization demonstrating a strong effect on local biotic integrity.

Biological Indices

Ecological communities can be assessed by looking at their relative biodiversity. Biodiversity is the measure of intraspecific and interspecific diversity between species in a given area. It specifically measures both richness and evenness which refers to the number of different species present and the proportion of species present in a given site respectively (Harper & Hawksworth, 1994). Biodiverse communities are considered to be more resilient, have increased stability, and productivity, and is why many scientists use this as an indicator for ecosystem health (Isbell et al., 2015). Biodiversity is most widely measured using two different indices: the Simpson's Index and the Shannon-Wiener Diversity Index (Daly et al., 2018). The Simpson's Index considers the number of species present and the abundance of each species. However, the Simpson's index is considered to be a dominance index because common or dominant species are given more importance. On the other hand, the Shannon-Weiner Index is widely accepted as the index for diversity as it measures the randomness of individual species equally and therefore takes into account even small changes in diversity comparatively better than the Simpson's Index (Dejong, 2021). Although these indices are quite practical for simple analyses of comparing biodiversity relative to other communities, they may be insufficient and even interpreted incorrectly for understanding more complex interactions (Morris et al., 2014). The limitations presented in these indices are that diversity does not necessarily represent community health in all cases, if for example the composition of species is considered invasive for a certain area or how certain ecosystems naturally show low species diversity. These indices do not capture the ecological role of certain species and by extension ecosystem functioning or resilience (Chiarucci et al., 2011). Therefore, these diversity indices may be more indicative of community structure or complexity but less so of community health per se.

2.2 Index of Biological Integrity (IBI)

The limitations highlighted above with some common indices bring into the conversation the use of indicator species. These are species that are usually sensitive to specific types of environmental disturbances and their presence or absence can indicate the stability of an ecosystem (Landres et al., 1988). By measuring the proportion of different indicator species, one can better assess the "health", i.e. the ability of an ecosystem to maintain its ecological functions in the face of disturbance (Karr, 1991). In general, the presence of generalist species which are adapted to many different environments, including poor conditions, tend to indicate a system that is unstable and degraded, whereas the presence of specialist species, which require specific environmental conditions, can indicate that the ecosystem is able to maintain stable habitats with specific niches (Karr, 1991). In an agricultural setting for example, intensive agriculture can have an effect on increased fish biomass likely related to the excess inputs of nutrients such as phosphorus. However, despite an increase in productivity, we find an imbalance in the food chain by an increase in omnivorous species (Richard & Giroux, 2004). Omnivores are generalists who are able to feed on a broad range of food sources, including detritus giving them a high tolerance to pollution. This is to the detriment of other species such as insectivores which prefer less disturbed sites. Therefore, the presence of certain species from different trophic levels and their abundance can be indicative on the level of environmental disturbance and species diversity (Richard & Giroux, 2004).

An index that takes into account biodiversity and indicator species is the Index of Biological Integrity (IBI). The IBI was originally developed as a tool for biological assessment by James Karr in the early 1980s as a way to evaluate the quality of small streams based on fish assemblages in the Midwest of the US but has since been adapted in many other regions and ecosystems (Karr, 1991). The IBI is composed of 12 biological metrics that falls into three categories: Species composition, Trophic composition, and Fish abundance and condition (Table 2.1). These metrics are meant to capture certain biological attributes of the sampled site indicative of underlying biological integrity. Biotic integrity as defined by Karr (1986) are "aquatic systems in which composition, structure, and function have not been adversely impaired by human activities" (p. 2). The IBI scores each metric based on the value expected for a pristine site of similar size and geographical region. A higher score would indicate that there is less perturbation and most reassembles what an undisturbed or natural site of similar type would look like. The lower the score the more it deviates away from what an undisturbed site would look like and would therefore suggest a site of reduced quality. The scores are also classified into rankings from excellent to very poor. The final IBI score is the sum of each variable which has a score of 5, 3 or 1 points, depending on whether the value resembles or differs widely from the expected value for a natural watercourse of similar size and geographic region with the higher total score indicating the best composition quality. In the absence of a reference stream, the limits of the criteria are defined by the subdivision of the distribution of values located below the 95th percentile, with values closer to the 95th percentile receiving a higher score (Barbour et al., 1999). In summary, "the IBI is most appropriately used to interpret large amounts of data from complex fish communities when the objective is to assess biotic integrity" (Karr, 1986, p. 6).

Fish are also an ideal indicator organism to assess biological integrity as they occupy multiple positions in the trophic chain (e.g., omnivores, herbivores, insectivores etc.) and their diets integrate resources from both aquatic and terrestrial origin. Another advantage is the ease in which to identify fish species on site compared to other aquatic taxa such as diatoms and invertebrates, which can require higher specialist identification and can involve tedious lab work and analysis (Karr, 1991).

Table 2.1. James Karr's original IBI metrics for assessing biotic integrity of fish in small streams in the Midwestern U.S (Karr, 1986).

Category	Metric	Scoring Criteria 5	3	1
Species richness and composition	1. Total number of fish species 2. Number and identity of darter species 3. Number and identity of sunfish species 4. Number and identity of sucker species 5. Number and identity of intolerant speices	Expectations for metrics 1-5 vary with stream size and region and are discussed in the text.		
	6. Proportion of individuals as green sunfish	<5%	5-20%	>20%
Trophic composition	7. Proportion of individuals as omnivores	<20%	20-45%	>45%
	8. Proportion of individuals as insectivorous cyprinids	>45%	45-20%	<20%
	9. Proportion of individuals as piscivores (top carnivores)	>5%	5-1%	<1%
Fish abundance and condition	10. Number of infividuals in a sample	Expectations for metric 10 vary with stream size and other factors and are discussed in the text.		
	11. Proportion of individuals as hybrids	0%	>0-1%	>1%
	12. Proportion of individuals with disease, tumors, fin damage, and skeletal anomalies	0-2%	>2-5%	>5%

2.3 IBI Adapted for Quebec

The IBI scoring system must also be adapted to each geographical region. For instance, the criteria for species richness and composition metrics will vary greatly with stream size and region and should be assessed using comparable data. Another metric that would also have to be adjusted is the total number of fish species and number of species which also generally increase with stream size (Karr, 1986). An IBI for Quebec rivers was developed by several researchers mainly by assessing the level of pollution tolerance in trophic species captured in the Saint-Laurence River and recording different variables used for the analysis of fish habitats. They used these fish communities to define each criterion to form the IBI for Quebec from Karr's original IBI (Richard 1994, 1996; La Violette & Richard, 1996; Martel & Richard, 1998; Saint-Jacques, 1998; La Violette, 1999; Rioux & Gagnon, 2001; Saint-Jacques & Richard, 2002).

It was also further adapted by Richard & Giroux (2004) for smaller streams in the Quebec context, as the first IBI developed for Quebec was originally adapted for large rivers in Strahler's 3rd order or greater for the St. Lawrence Lowlands. The indices for the abundance of piscivore species, the number of intolerant species, and the number of Catostomidae, were not adaptable for smaller rivers and streams situated at the head of the basin (Strahler stream orders < 3) and thus were the variables modified. This is because rivers at the head of the basin naturally lack these species, either due to the size of the river or the type of environment of these areas. As a result, six criteria adapted for small streams in Quebec were kept (Table 2.2). Further changes involved the final IBI score being doubled to become more comparable to Karr's original index which had 12 variables in total. As a result, Richard & Giroux (2004) adapted the IBI again for streams and rivers at the head of the basin of first and second order for the streams of Saint George, Quebec.

Table 2.2. IBI indicators for fish communities in the Saint-Georges streams, Quebec (Richard & Giroux., 2004).

Variables	Scoring Criteria		
	Excellent 5	Fair 3	Poor 1
Composition and abundance			
1. Biomass (g)	≤ 165	166-329	≥ 330
2. Number of species	≥ 6	3-5	≤ 2
3. Relative abundance of tolerant species	≤ 33%	34-67%	≥ 68%
Trophic level			
4. Relative abundance of omnivores	≤ 18%	19-36%	≥ 37%
5. Relative abundance of insectivores	≥ 63%	32-62%	≤ 31%
Physcial condition			
6. Proportion of fish with DELT type anomalies	0-2%	2.1-5%	≥ 5.1%

2.4 Geomorphological Assessment

Morphological assessment refers to the broad identification and evaluation of overall river conditions including channel form and pattern, geomorphic conditions, and human alterations. The emphasis is placed on cause-effect relationships of physical processes that affect the overall system rather than singular instream physical habitat assessments. It is typically analyzed at a reach scale meaning that the area can vary in size depending on the boundaries of homogenous morphological characteristics of river sections and the size of the stream. Unlike other hydrogeomorphological indices, morphological assessments utilize remote sensing and maps more extensively in conjunction with field surveying. In doing so, morphological assessments can also take into account temporal changes in channel evolution through historical data which can give insight to future geomorphic changes (Belletti et al., 2015). Some examples of morphological tools used in many states in the U.S. is the Rapid Geomorphic Assessment (RGA) developed by Simon and Downs (1995). This tool was conceived in order to rapidly assess the stability of channels at a large scale to quickly identify critical areas that may put buildings or people at risk. The method uses a combination of site evaluations and GIS-based information to rank channel stability and identify areas of high socio-economic risk. Further fieldwork can also be used to collect more data on the areas at risk to better evaluate the extent and severity of risk at those particular sites (Simon & Downs, 1995). Whereas the RGA was developed for rapid watershed scale assessments, the state of Vermont has also adapted their own stream assessment protocol for reach scale analysis. Their protocol can be broken down into six steps: evaluation of the valley and river corridor; stream channel; stream banks, buffers, and corridors; flow modifiers; channel bed and planform changes; and finally use of Rapid Habitat Assessment (RHA). This protocol relies on a significant amount of field observation, although it also incorporates the use of maps (Vermont Agency of Natural Resources, 2009). In Canada, the Ontario Ministry of the Environment acknowledges that erosion is a normal aspect of river behaviour and provides design criteria aiming to preserve a "stable" sustainable fluvial system and its associated habitat (Ontario Ministry of Environment, 2003). It uses either a 10-step detailed design approach, including geomorphic criteria for habitat protection-restoration, or a simplified design approach for small headwater streams that do not provide habitat for a sensitive aquatic species.

2.5 Morphological Quality Index (MQI)

The Morphological Quality Index (MQI) is a morphological assessment method developed in Europe to address the need to achieve a 'good water status' in rivers in terms of morphological and ecological quality as required by the Water Framework Directive (WFD) in the European Union (EU) (Rinaldi et al., 2013; 2016). According to the WFD, the reference state is defined by undisturbed or minor anthropogenic impact where the river is able to perform its usual morphological functions and artificiality does not significantly affect the river dynamics at the catchment and reach scale. As such, the MQI consists of 28 indicators which determine the level of anthropogenic disturbance on river health in terms of geomorphological functionality, artificiality, and channel adjustments (Table 2.3). The details of each indicator score computation is provided in Appendix A. Each indicator is assigned a score depending on the relative disturbance. The final MQI score is a range between 0 to 1, 1 indicating the most natural river. There are subdivisions between this range that fall into five quality categories: very good, good, moderate, poor, and very poor. Remote MQI can also be assessed using GIS software. In this case, about 18-21 of the indicators could be assessed depending on satellite imagery resolution and available data, as some criteria require field observations (e.g. bed clogging, bed sills, in-channel large wood, and cross-sectional variability). This method was designed to be relatively simple and efficient to use for practitioners, however it still requires a level of geomorphological background in the sense that the selection of variables and their assigned scores are relative to the user's judgment. The overall assessment is carried out using a complementary combination of GIS analysis and field surveys. The indicators can be classified into the three main components evaluated: river continuity, morphology, and vegetation; and their corresponding assessment criteria: functionality, artificiality, and channel adjustments. Furthermore, certain indicators are more suited to be applied in specific classes of channels: confined channels, partially confined channels, and unconfined channels (Rinaldi et al., 2016).

The MQI is a very well-suited tool for this study because of its relative simplicity and efficiency to use, in that it does not require advanced geomorphological expertise nor intensive geomorphological fieldwork. In particular, there is an opportunity to develop the MQI as a remote sensing tool since several of its components can be obtained using GIS analysis. It has already been shown that the MQI can to a certain extent be approximated by remote MQI (rMQI) which could reduce the necessity for field surveys and expand the spatial coverage for preliminary assessments (Lemay, 2020). Furthermore, Lemay (2020) has shown that the MQI is strongly correlated with the Qualitative Habitat Index (QHEI), a well-known indicator for fish habitats (Rankin, 1989; Taft, 2006). However, so far there has not been any attempt to relate MQI to actual fish biodiversity and health in agricultural streams. If successful in establishing a relationship between MQI and fish ecology, this remotely sensed tool could also be used to conduct preliminary fish assessment and provide a better understanding between the relationship of fish assemblages and morphological characteristics.

Table 2.3. Morphology Quality Index (MQI) indicators, assessment methods, and confinement typologies (C= confined, PC= partially confined & U= unconfined) (modified from Rinaldi et al., 2016).

Indicators and assessed parameters	Assessment methods	Ranges of application
F1 – Longitudinal continuity in sediment and wood flux Presence of crossing structures (weirs, check-dams, bridges, etc) that potentially may alter natural flux of sediment and wood along the reach	*Remote sensing* and/or *database of interventions:* identification of crossing structures; *field survey*: visual assessment of partial or complete interception (qualitative)	All river types
F2 – Presence of a modern floodplain Width and longitudinal length of a modern floodplain	*Remote sensing–GIS*: measurement of width and longitudinal length (quantitative); *field survey*: identification/checking of modern floodplain (qualitative)	*PC–U*; not evaluated in the case of mountain streams along steep (>3% slope) alluvial fans
F3 – Hillslope – river corridor connectivity Presence and length of elements of disconnection (e.g., roads) within a buffer 50-m wide for each side of the river	*Remote sensing–GIS*: identification and measurement of length of disconnecting elements (quantitative); *field survey*: checking disconnecting elements (qualitative)	*C*
F4 – Processes of bank retreat Presence/absence of retreating banks	*Remote sensing* and/or *field survey*: identification of eroding banks (qualitative)	*PC–U*; not evaluated in the case of Low-Energy ERT types from 17 to 22
F5 – Presence of a potentially erodible corridor Width and longitudinal length of an erodible corridor, i.e., area without relevant structures (e.g., bank protections, levées) or infrastructure (e.g., houses, roads)	*Remote sensing–GIS*: measurement of width and longitudinal length (quantitative)	*PC–U*
F6 – Bed configuration – valley slope Identification of bed configuration (i.e., cascade, step pool, etc.) in cases where transverse bed structures are present and in comparison with the expected bed configuration based on valley slope	*Topographic maps*: mean valley slope (quantitative); *field survey*: identification of bed configuration (qualitative)	single-thread, alluvial *C* (ERT types from 4 to 7), except the case of deep streams when observation of the bed is not possible
F7 – Planform pattern Percentage of the reach length with altered planform and geomorphic units	*Remote sensing–GIS:* identification and measurement of length of altered portions (quantitative); *field survey*: identification/checking (qualitative)	*PC–U;* Confined ERT types 8, 9, 10, 11, 15, 19, 22
F8 – Presence of typical fluvial landforms in the floodplain Presence/absence of appropriate landforms in the floodplain (e.g., oxbow lakes, secondary channels, etc.)	*Remote sensing* and/or *field survey*: identification and checking of fluvial forms (qualitative)	*PC–U*
F9 – Variability of the cross section Percentage of the reach length with alteration of the natural heterogeneity of the cross section that is expected for that river type and is caused by human factors	*Field survey*: identification/checking (qualitative); *remote sensing–GIS:* identification and measurement of length of altered portions (quantitative)	All types
F10 – Structure of the channel bed Presence/absence of alterations of bed sediment (armouring, clogging, bedrock outcrops, bed revetments)	*Field survey*: visual assessment (qualitative)	All types, except the case of deep channels when observation of the bed is not possible
F11 – Presence of in-channel large wood Presence/absence of large wood	*Field survey*: visual assessment (qualitative)	All types; not evaluated above the tree-line and in streams with a natural absence of woody riparian vegetation
F12 – Width of functional vegetation Mean width (or areal extension) of functional riparian vegetation in the fluvial corridor potentially connected to channel processes	*Remote sensing–GIS:* identification and measurement of mean width of functional vegetation (quantitative)	All types; not evaluated above the tree - line and in streams with a natural absence of riparian vegetation
F13 – Linear extension of functional vegetation Longitudinal length of functional riparian vegetation along the banks with direct connection to the channel	*Remote sensing–GIS:* identification and measurement of longitudinal length of functional vegetation (quantitative)	All types; not evaluated above the tree - line and in streams with a natural absence of riparian vegetation
A1 – Upstream alteration of flows Amount of changes in discharge caused by interventions upstream (dams, diversions, spillways, retention basins, etc.)	*Hydrological data*: evaluation of reduced/increased discharge caused by interventions (quantitative). In the absence of available data, the assessment is based on the presence of flow intervention and its use (qualitative)	All types

Table 2.3. continued

A2 – Upstream alteration of sediment discharges Presence, type, and location (drainage area) of relevant structures responsible for bedload interception (dams, check-dams, weirs)	*Remote sensing–GIS* and/or *database of interventions:* identification of structures and relative drainage area (quantitative)	All types
A3 – Alteration of flows in the reach Amount of alterations of discharge caused by interventions within the reach	See A1	All types
A4 – Alteration of sediment discharge in the reach Type and spatial density of structures intercepting bedload (check dams, weirs) along the reach	*Remote sensing–GIS* and/or *database of interventions:* identification and number of structures (quantitative)	All types
A5 – Crossing structures Spatial density of crossing structures (bridges, fords, culverts)	*Remote sensing–GIS* and/or *database of interventions:* identification and number of structures (quantitative)	All types
A6 – Bank protections Length of protected banks (walls, rip-raps, gabions, groynes, bioengineering measures)	*Remote sensing–GIS* and/or *database of interventions:* length of structures (quantitative)	All types
A7 – Artificial levées Length and distance from the channel of artificial levées	*Remote sensing–GIS* and/or *database of interventions:* length and distance of structures (quantitative)	*PC–U*
A8 – Artificial changes of river course Percentage of the reach length with documented artificial modifications of the river course (meander cutoff, relocation of river channel, etc.)	*Historical /bibliographic information* and/or *database of interventions* (quantitative)	*PC–U*
A9 – Other bed stabilization structures Presence, spatial density and typology of other bed-stabilizing structures (sills, ramps) and revetments	*Remote sensing–GIS* and/or *database of interventions:* identification, number or length of structures (quantitative)	All types
A10 – Sediment removal Existence and relative intensity of past sediment mining activity (over the last 100 years, with a particular focus on the last 20 years)	*Database of interventions* and/or *information available by public agencies; field survey* and/or *remote sensing:* indirect evidence (qualitative)	All types; not evaluated in the case of ERT type 1
A11 – Wood removal Existence and relative intensity (partial or total) of in-channel wood removal during the last 20 years	*Database of interventions* and/or *information available by public agencies; field survey:* additional evidence (qualitative)	All types; not evaluated above the tree - line and in streams with natural absence of riparian vegetation
A12 – Vegetation management Existence and relative intensity (selective or total) of vegetation cuts during the last 20 years	*Database of interventions* and/or *information available by public agencies; field survey:* additional evidence (qualitative)	All types; not evaluated above the tree - line and in streams with natural absence of riparian vegetation
CA1 – Adjustments in channel pattern Changes in channel pattern from 1930s to 1960s based on changes in sinuosity, braiding, and anastomosing indices	*Remote sensing–GIS* (quantitative)	All types; evaluated only for sufficiently large channels
CA2 – Adjustments in channel width Changes in channel width from 1930s to 1960s	*Remote sensing–GIS* (quantitative)	All types; evaluated only for sufficiently large channels
CA3 – Bed-level adjustments Bed-level changes over the last 100 years	*Cross sections / longitudinal profiles* (if available); *field survey*: evidence of incision or aggradation (qualitative/quantitative)	All types; evaluated in case field evidence or information is available

3. Research Purpose

Little is known on the spatial variability of fish communities in agricultural streams in Montérégie, Quebec. This project will therefore provide a well-needed portrait of the fish communities in the Montérégie area, which is where agricultural activities in Quebec are concentrated. Furthermore, the literature has shown that connections between geomorphic qualities of streams and rivers clearly exist with fish assemblages. Although spatial and temporal factors also have a big role to play; predictors at the reach-scale were very indicative with stream bed characteristics and slope being the most important. However, these variables do not tend to be easily collected and require a decent amount of fieldwork effort. Another research venue to explore is the assessment of other morphological characteristics such as those in the MQI, which also includes remote variables that are easily assessed, and determine whether its indicators could also be strong predictors of fish assemblage and health. Therefore, chapter 4 will explore the composition and distribution of fish communities in Montérégie, Quebec whilst evaluating their biotic integrity using the Index of Biotic Integrity (IBI). Chapter 5 will focus on evaluating whether geomorphological variables characterising agricultural stream quality used in the Morphological Quality index (MQI) is able to predict fish assemblages and community health.

4. Fish biodiversity in small streams of Montérégie (Quebec) and potential for restoration

This chapter was written in collaboration with my supervisor Dr. Pascale Biron, as well as Renée Gravel from the Ministère de l'Environnement, de la Lutte contre les changements climatiques, de la Faune et des Parcs du Québec (MELCCFP). A French version of this manuscript will soon be submitted to the journal *Le Naturaliste canadien*.

As first author I was responsible for the development of the methodology, for the analysis and interpretation of the results, and production of maps pertaining to Index of Biotic Integrity (IBI). Dr. Pascale Biron provided guidance throughout the research and writing process and contributed important adjustments and helpful feedback throughout earlier versions of the manuscript. Renée Gravel provided the fish dataset from the ministry and was responsible for the research, results and writing pertaining to the restoration aspect of very small streams. The research methods and results for the restoration of very small streams were not my own work and are derived from Renée Gravel's separate study (Gravel, 2021). They were included in this chapter as they are relevant for the *Naturaliste Canadien publication* and confirm observations made in this study. Renée Gravel also provided valuable insights and comments on this manuscript.

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4.1 Abstract

In Montérégie, Quebec, several human interventions affect watercourses, including stream channelization, which is a common practice in agriculture. Despite more than 30,000 km of watercourses subject to this type of development in Quebec, very little is known about the ecological state of these linearized and often over deepened watercourses. This research aims to document the ecological state of small watercourses in Montérégie using the Index of Biotic Integrity (IBI), evaluated from the provincial fish database of the Ministry of Environment, and the Fight against Climate Change, and Wildlife and Parks (MELCCFP), as well as to determine the restoration potential of small waterways that have been developed. More than 1,228 samples of fish caught between 1930 and 2019 in Montérégie in streams of Strahler orders 1 to 3 (drainage areas less than 130 km^2) were analyzed. Restoration practices were also evaluated in 6 small watercourses in Montérégie where improvements were made (groins, two-level channels, bank planting methods) using biophysical characterization as well as experimental fishing before (2017) and after (2019) these improvements. The results show that the biotic integrity of fish is significantly correlated with the percentage of agricultural land use and forest cover. The most degraded fish communities were found downstream of central Montérégie (in the Richelieu and Yamaska watersheds) which has the highest concentration of agricultural land use and the least forest cover. However, small streams in Montérégie also showed high fish abundance and biodiversity including many vulnerable, threatened, and sport fishes, especially in wetlands near large tributaries. Restoration efforts in very small streams have also shown improved fish community health by improving overall channel morphology. These results demonstrate how small streams in agricultural watersheds in Montérégie play a more important role than was previously thought for fish communities and should therefore be prioritized in restoration projects.

Key words
Fish communities, agricultural drainage, small streams, habitat, Index of Biotic Integrity (IBI), restoration.

4.2 Introduction

Primary headwater streams in watersheds alone comprise up to 85% of the total length of the river system (Gravel 2021, Lane et al. 2022). In agricultural settings, these small streams are generally considered poor habitats with low potential for productivity and little interest for restoration. However, few studies have actually looked at their biodiversity and potential for restoration. Montérégie is considered the breadbasket of Quebec with a flat and fertile territory and crops that occupy nearly 60% of the area and where watercourses have been largely modified to improve agricultural drainage. In 2001, it was estimated that about 30,000 km had been redeveloped for this purpose (Beaulieu, 2001; Rousseau and Biron, 2009). These channelizations typically consist of linearization, deepening and widening of the stream by dredging, as well as removal of wood supply by removing trees and shrubs from the bank and eliminating riparian vegetation (Sanders et al. 2020). Every year, hundreds of kilometers of these small streams are cleaned out to return them into this linear form and ensure rapid drainage. Figure 1 shows the mapping of dredging projects filed with the Ministry of Environment, and the Fight against Climate Change, and Wildlife and Parks (MELCCFP) from 2011 to 2020 (Gravel, 2021).

Fish are considered a good indicator of aquatic ecosystem condition (Karr, 1986). Several studies have shown that stream channelization reduces habitat heterogeneity and alters flow by reducing riffle-pool sequences (Frothingham et al., 2001; Heatherly et al., 2007; Lau et al., 2006), which in turn affects the composition of fish communities (Rhoads et al. 2003; Sanders et al. 2020; Tóth et al. 2019). One might think that poor water quality is the primary issue in agricultural settings, but Wichaert and Rapport (1998) reveal that the distribution of sensitive fish species in rural areas is instead associated with biophysical factors such as the removal of functional riparian vegetation leading to increased stream bank instability, elevated water temperature due to loss of shade from riparian vegetation, and stream erosion and siltation due to intensive tillage practices. Stream morphology thus has a significant impact on habitat diversity and consequently on species biodiversity in streams (Schlosser, 1991). However, despite the extent of anthropogenic interventions in headwater streams in Montérégie, little is known about the region's fish communities and its role in maintaining a good biological state in the aquatic ecosystem (Gravel, 2021).

This article presents the results of two separate but complementary studies that provide a portrait of our knowledge of small streams in Montérégie. The first study aimed to evaluate the biotic integrity of fish in small streams in the region and examined the results of experimental fisheries historically compiled in the fisheries database of the MELCCFP's General Directorate of Wildlife of Estrie, Montreal, Montérégie and Laval (DGFa). It focused on fisheries conducted in small streams in Montérégie from 1930 to 2019. The second study was specifically aimed at better understanding the fish communities in agricultural headwater streams subject to different types of agricultural drainage interventions, as well as documenting their response to conventional drainage interventions vs. interventions integrating physical quality restoration efforts. The overall objective of this paper is to better understand the fish biodiversity and fish community status in this highly agricultural region and the restoration potential of these small streams.

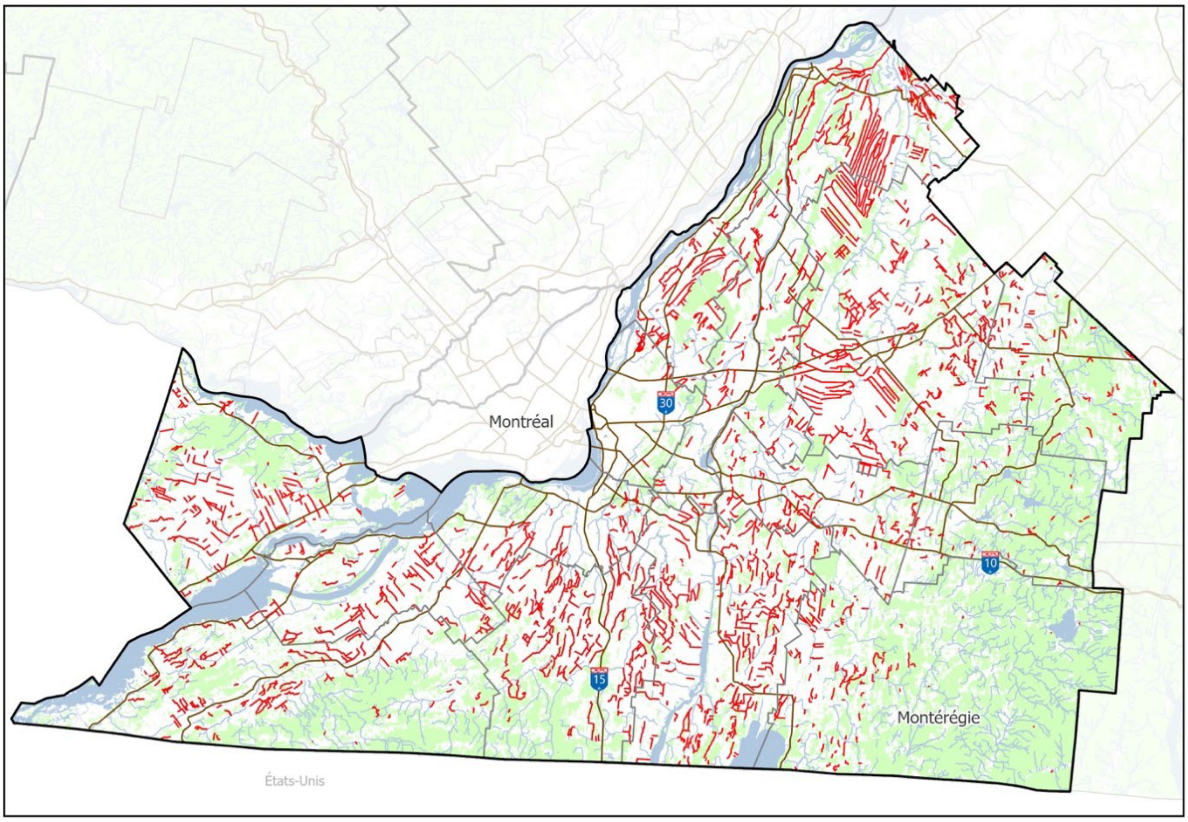


Figure 1. Map of in-stream intervention projects for agricultural drainage purposes submitted to MELCCFP from 2011 to 2020 (Source: Gravel, 2021).

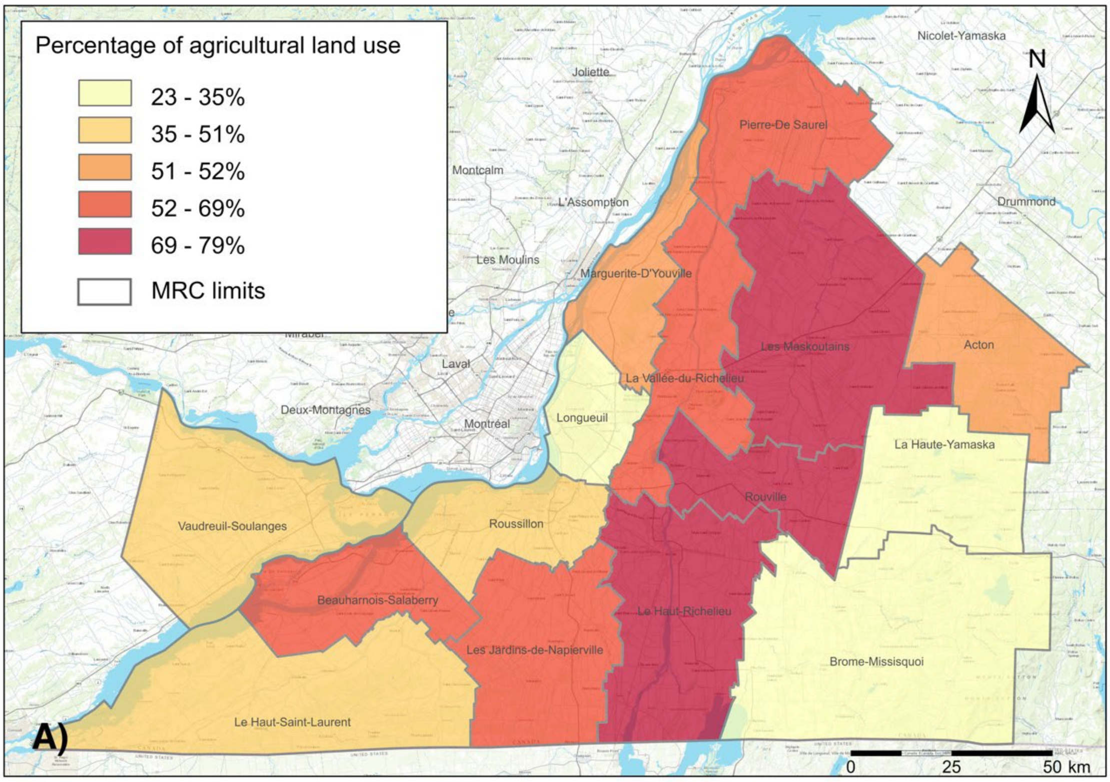
4.3 *Methodology*

4.3.1 Data source

Fish data were obtained from the database of experimental fishing results compiled by the DGF (Direction de la gestion de la Faune) (MFFP 2019) and included various fish inventories conducted in Montérégie between the 1930s and 2019. Data were extracted for small streams, for a total of 1,228 samples - a sample being a record of a fishing effort conducted during a given period of time.

There is no clear definition of what a "small stream" is. For example, the Ohio Environmental Protection Agency (EPA) considers streams with a drainage area of less than 260 ha (2.6 km2 or 1 mi2) to be "Primary Headwater Streams" (Ohio EPA, 2012) while the Environmental Protection Agency (EPA) considers small streams to have a drainage area of less than 130 km2 (or < 50 mi2) (MPCA, 2014; Davis et al., 1996). These definitions are consistent with the majority of cases considered in this study, which primarily includes streams of Strahler order ≤ 3. The average drainage area of small streams extracted from the DGFa database was 24 km^2 (median 10 km^2, ranging from 0.28 km^2 to 250 km^2), with over 98% of streams less than 130 km^2. The drainage area of the sub-basins of each of the small streams was extracted from the CRHQ ("Cadre de reference hydrologique du Québec", Quebec hydrological reference framework), while the Strahler order was determined by specific analysis of the position of the watercourse in the watershed. The monitoring of restoration potential was conducted in a subgroup of small streams, namely very small streams located at the head of the watershed, with a Strahler order of 1 or 2, and a watershed size of less than 5 km^2. They are identified by the term "very small streams" (VSS), and it is generally these VSS that are subject to maintenance (dredging) and intervention (Gravel, 2021).

The maps that follow are presented according to regional county municipality (MRC) because this is how usage data is compiled by the government and watercourse management is a legal responsibility of the MRCs. Since July 28, 2021, the MRCs of Brome-Missisquoi and Haute-Yamaska have moved from the Montérégie region to the Estrie region (see decree 961-2021 in the Gazette officielle du Québec n°30 of 2021). However, the old territorial delimitation is used here because it is representative of the time period covered by the data. Figure 2 shows the proportion of MRCs that are occupied by agricultural activities and forest cover. These land use types are the most dominant in the region and are known to have a strong impact on the condition of biological communities (Petsch et al., 2021; Stewart et al., 2001).



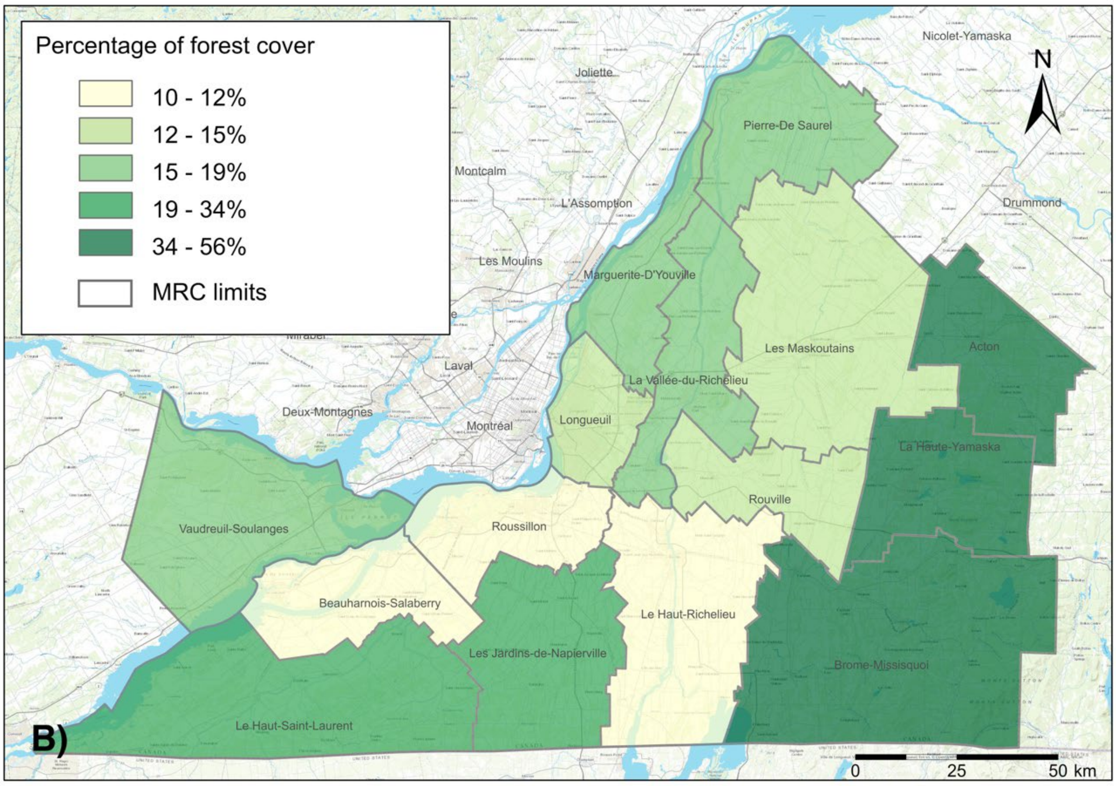


Figure 2. Proportion of A) agricultural land use and B) forest cover in the different MRCs of Montérégie (source: Gouvernement du Québec, 2019).

4.3.2 Treatment of different fishing methods in the DGF database

The database identified several different fishing gears. Traditionally, seine-type gear was used extensively for sampling in small streams, however since around the 2000s, electrofishing has been the most commonly used gear. Thus, it was necessary to report the fishing efforts on a comparable denominator and therefore fishing methods were grouped into distinct categories according to the type of gear and statistical similarity of their catch per unit effort (CPUE). This resulted in five groups: electrofishing, fyke nets, nets, seine, and shore seine. Individual tripartite distributions for each biological metric were calculated within each group so that only similar fishing gears were compared to each other. Some samples did not specify fishing effort; thus, approximately 13% of the data could not be used in this study. An exception was made for the seine and shore seine methods, which were considered to be instantaneous catches and therefore their sample size was based on the average instantaneous catch. This is because the database did not provide consistent details on the net area sampled. Further implications on the use of different methods on the results is discussed in Appendix B.

4.3.3 The Index of Biotic Integrity (IBI)

There are various ways to assess the diversity and condition of biotic communities. For example, Morris et al. (2014) suggest that biological communities can be assessed by examining their relative biodiversity, which is the measure of intraspecific and interspecific diversity in a given area. In addition, various multi-metric indices have been developed, including Karr's (1986) Index of Biotic Integrity (IBI), originally developed for assessing small streams using fish communities in Ohio and composed of 12 metrics. However, the use of such an approach cannot be done without adapting the choice of metrics according to local geographies and communities. After examining the applications of the IBI used in Quebec (Richard & Giroux, 2004; Saint-Jacques & Richard, 2002; Théberge & Côté, 2008), and taking into account the specificities of the region, only five metrics were retained: relative diversity (Shannon-Wiener index), the proportions of benthic insectivorous, omnivorous and pollution-tolerant species, as well as the catch-per-unit-effort (CPUE) (Table 4.1). Indeed, several of Karr’s metrics (1986) did not apply to the small and very small streams of Quebec (Gravel, 2021). For example, the proportion of darters is not a relevant metric here, because the darter family is much less diverse in Quebec than in Ohio, and darters are rarely found in Strahler order 1 or 2 streams in Montérégie. With respect to other metrics, the database did not contain the information that would have been required to refer to them (e.g., proportion of individuals with external anomalies).

To complement the IBI analysis, species at risk, aquatic invasive species, and species of sport fish interest (as defined by provincial regulations) were also examined. Once each metric was calculated for each sample, it was divided into a tripartite distribution using the 95th percentile method (Barbour et al., 1999) to assign IBI unit limits so that it could be distinguished whether the metric for that fish community was poor (1), moderate (3), or good (5) (Table 4.1). Depending on which scoring criterion the metric fits into, each metric is then summed up to give the final IBI value. Following Richard & Giroux (2004), IBI units were doubled in this study to allow comparison with the IBI values obtained with the 12 original metrics in Karr's (1986) IBI (Table 4.2).

Table 4.1. IBI metrics and their respective score boundaries for Poor (1), Moderate (2), Good (3) classes (adapted from Karr, 1991).

		Scoring criteria		
		Good	**Moderate**	**Poor**
Category	Metric	**5**	**3**	**1**
	1. Shannon-Wiener Index	>1.4	>0.87-1.4	<0.87
Species Richness and composition	2. Number of Tolerant Species (%)	<46%	46%-83%	>83%
Trophic composition	3. Number of Omnivores (%)	<37%	>37%-73%	>73%
	4. Number of Benthic Insectivores (%)	>13%	>0%-<13%	<0%
Fish abundance and condition	5. CPUE (catch/s)	>0.13-<134	>0-0.13 or >134-<507	<0.13 or >507

Table 4.2. Total IBI score and corresponding quality category after adding up scoring criteria from each metric and multiplied by two.

IBI	QUALITY CLASS
46-50	*Very Good*
38-42	*Good*
30-34	*Moderate*
22-26	*Poor*
10-18	*Very Poor*

4.3.4 Restoration Potential

In order to assess the restoration potential of very small streams, fishing efforts were conducted during the summers of 2017 and 2019, both before and after in-stream work. Fishing was carried out using electrofishing, in an open station, in three Montérégie watersheds where an enhanced stream (E) and a control stream (C) were sampled. A biophysical characterization was also performed. The reclamation was intended to improve the physical quality of the fish habitat compared to traditional types of intervention (i.e., dredging in a watercourse that had already been channelized in the past). Three types of improvements were tested: deflectors in the Acton MRC, two-stage channel in the Brome-Missisquoi MRC and streambank planting in the Marguerite-d'Youville MRC. Thus, 6 VSS, 3 of each type (E and C), were sampled (watersheds ranging from 0.8 to 5 km^2) in order to evaluate the restoration potential based on the response of the fish community.

4.4 Results

4.4.1 Index of Biotic Integrity (IBI)

Of the 1,228 fish samples collected in small streams in Montérégie, 85 fish species were documented, not including samples where identification was limited to genus. Overall, the IBI values indicate poor to average quality of fish communities in Montérégie, although the range of values covers the entire IBI spectrum (min. 10, max. 50). The distribution of average IBI values by MRC reveals that, in general, better quality fish communities are found upstream (in the southern part of Montérégie) compared to downstream in the northern part of Montérégie (Figure 3).

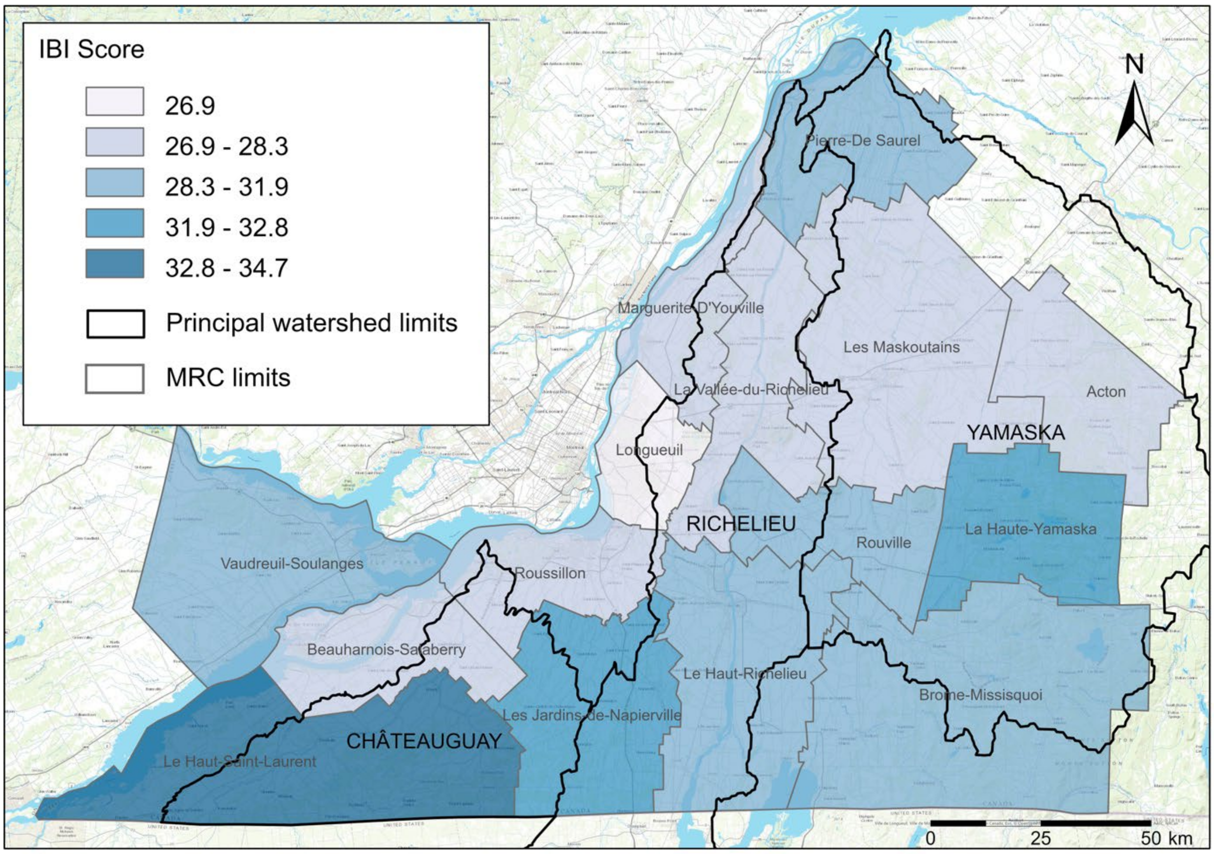


Figure 3. Average values of the IBI of fish communities in small streams in each MRC and limits of the 3 main watersheds (Châteauguay, Richelieu, Yamaska).

In order to analyse the temporal evolution of aquatic biodiversity, fish data in small streams were compared before and after the 1990s (1930 to 1989 versus 1990 to 2019). The data were compared in this way to distribute the samples as evenly as possible across time and space. Before the 1990s, 70 different species of fish were caught, and after the 1990s, there were 75 species, with 60 species in common between the two periods. Prior to the 1990s, IBI was generally of average quality, with poor quality in the Marguerite D'Youville and Acton MRCs (Figure 4A). After the 1990s, the central regions of Montérégie, which are also located further downstream in the watersheds, such as the MRC La Vallée-du-Richelieu and Les Maskoutains, saw their situation deteriorate and became the areas with the poorest IBI. On the contrary, the regions located in the upstream parts of the watersheds (in the south) such as the MRC Le-Haut-Saint-Laurent have improved and have fish communities of higher quality than the rest of the region with high IBI values (figure 4B). Overall, there has been a deterioration over time in the IBIs of the MRCs in the centre of Montérégie, while those in the periphery of Montérégie have generally improved (Figure 4C).

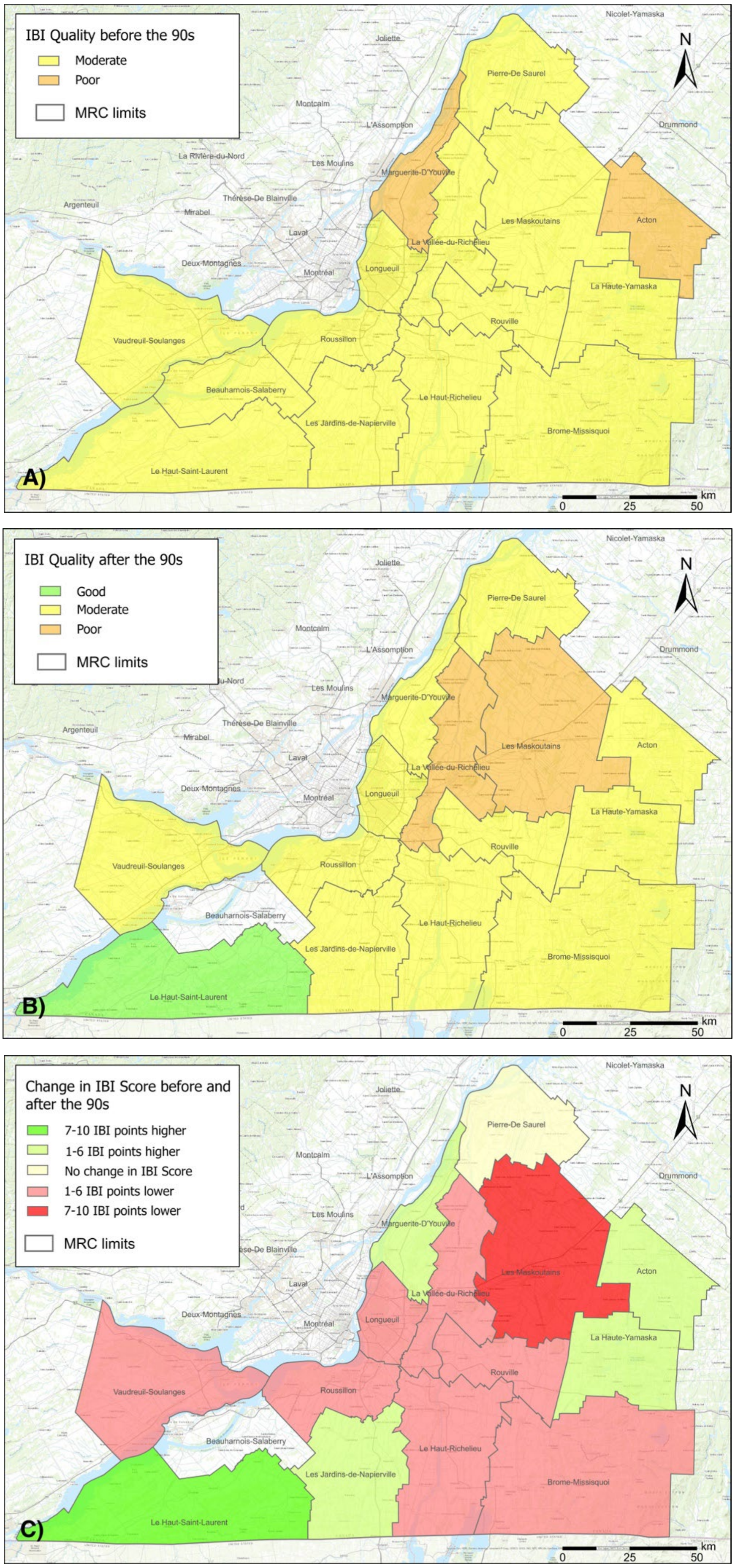

IBI Quality before the 90s
Moderate
Poor
MRC limits
A)
IBI Quality after the 90s
Good
Moderate
Poor
MRC limits
B)
Change in IBI Score before and after the 90s
7-10 IBI points higher
1-6 IBI points higher
No change in IBI Score
1-6 IBI points lower
7-10 IBI points lower
MRC limits
C)
Pierre-De Saurel
Marguerite-D'Youville
Les Maskoutains
Acton
La Vallée-du-Richelieu
Longueuil
La Haute-Yamaska
Rouville
Vaudreuil-Soulanges
Roussillon
Beauharnois-Salaberry
Le Haut-Richelieu
Les Jardins-de-Napierville
Brome-Missisquoi
Le Haut-Saint-Laurent
0 25 50 km

Figure 4. Quality of fish communities in small streams based on the IBI (A) before and (B) after the 1990s in Montérégie, and (C) change in the IBI of each MRC between the two periods (pre- and post-1990). Note: no data available for Beauharnois-Salaberry >1990.

4.4.2 Species of interest in Montérégie

4.4.2.1 Species at risk

A total of 6 different species with special status were identified in small streams in the DGF database. These included 2 of the 3 threatened fish species in Quebec, the sand darter (*Ammocrypta pellucida*), and the northern lamprey (*Ichthyomyzon fossor*). In addition, 4 of the 5 vulnerable fish species in Quebec were found: Bridle Shiner (*Notropis bifrenatus*), Rainbow Smelt (*Osmerus mordax*), River Redhorse (*Moxostoma carinatum*), and Channel Darter (*Percina copelandi*). Figure 5 shows that species of precarious status are concentrated near wetlands and floodplains connected to major rivers. This reflects the richness of these environments, which provide suitable habitat for their basic needs (breeding, feeding, refuge).

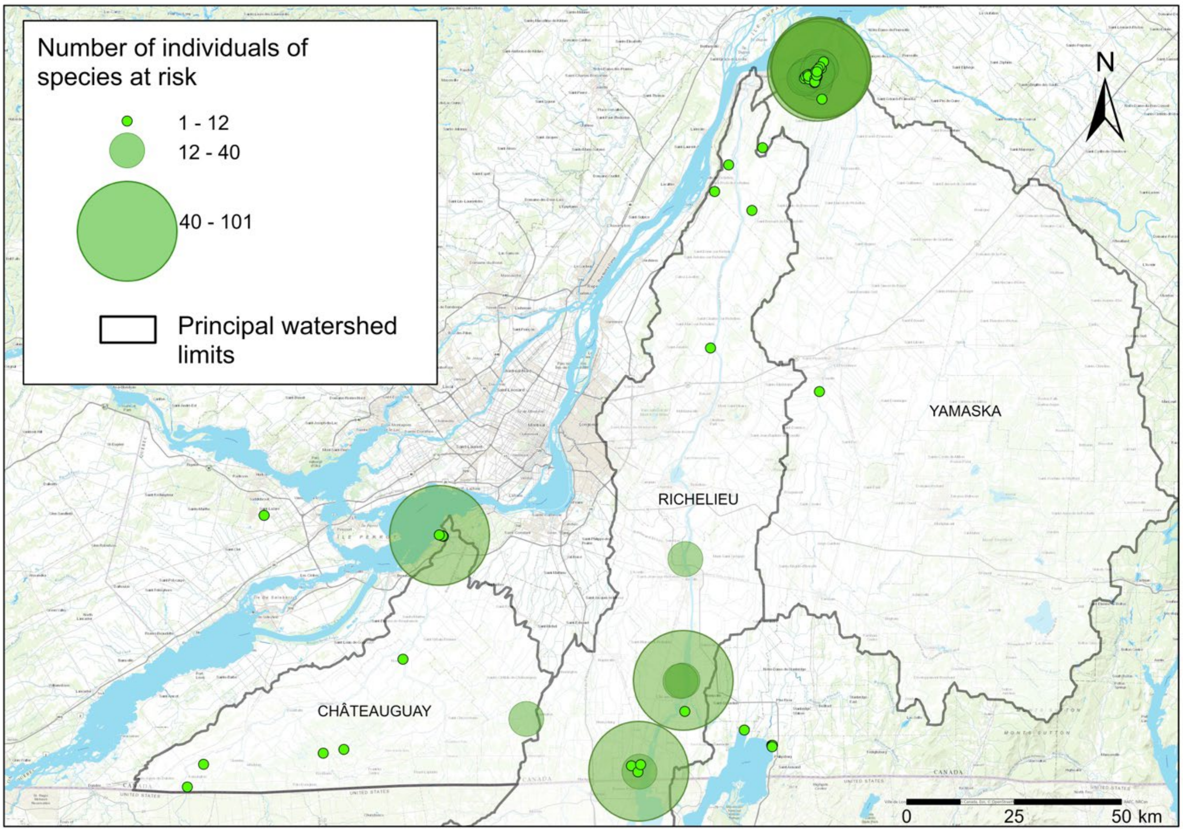


Figure 5. Species at risk under the Quebec Act regarding threatened and vulnerable species in small watercourses in Montérégie.

4.4.2.2 Non-native species

A total of 6 non-native fish species were recorded in small streams in Montérégie: tench (*Tinca tinca*), round goby (*Neogobius melanostomus*), green sunfish (*Lepomis cyanellus*), common carp (*Cyprinus carpio*), gizzard shad (*Dorosoma cepedianum*) and rainbow trout (*Oncorhynchus mykiss*). Based on the fishing data in small streams, these species appeared to be more prevalent in northern (downstream) Montérégie than in the southern (upstream) region, likely due to their respective entry ways through which they arrived in Quebec (Figure 6). The most abundant areas were downstream of the Pot-au-Beurre River, whose mouth is located in a large floodplain known as Baie Lavallière, and which is flooded by the St. Lawrence River every year (MRC de Pierre-De Saurel). These wetlands are habitats with high biodiversity (hotspots), but non-native species represent an additional pressure on these rich environments, with a possible impact on many species of interest (of precarious status or of sporting interest).

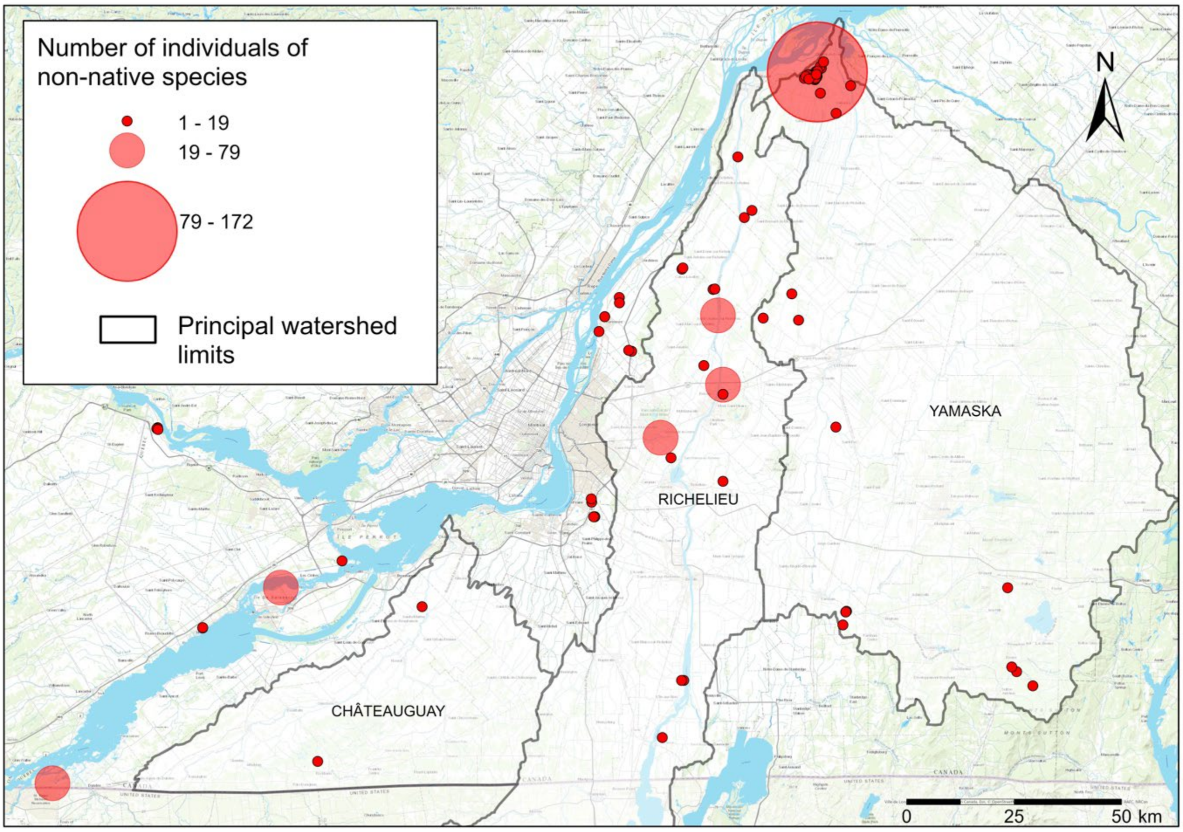


Figure 6. Non-native fish species found in small streams in Montérégie.

4.4.2.3 Sport fish species

Inventories conducted by the ministry confirm the presence of 21 species of sport fish in Montérégie's small streams. These species are well distributed and abundant throughout the region and are found in large numbers in small streams in agricultural areas. Two important hotspots for sport fish species relative to the rest of the region are downstream of the Pot-au-Beurre River (Baie Lavallière sector) in the Pierre-De Saurel MRC; and in the Charrette Brook and Brazeau Bay sector in the Vaudreuil-Soulanges MRC, near the Quebec-Ontario border. Again, these two hotspots are located near wetlands and floodplains connected to major rivers, in this case the St. Lawrence and Ottawa Rivers respectively (Figure 7). The high frequency of individual sport fish in small streams suggest that these habitats are also very important for the maintenance of these populations.

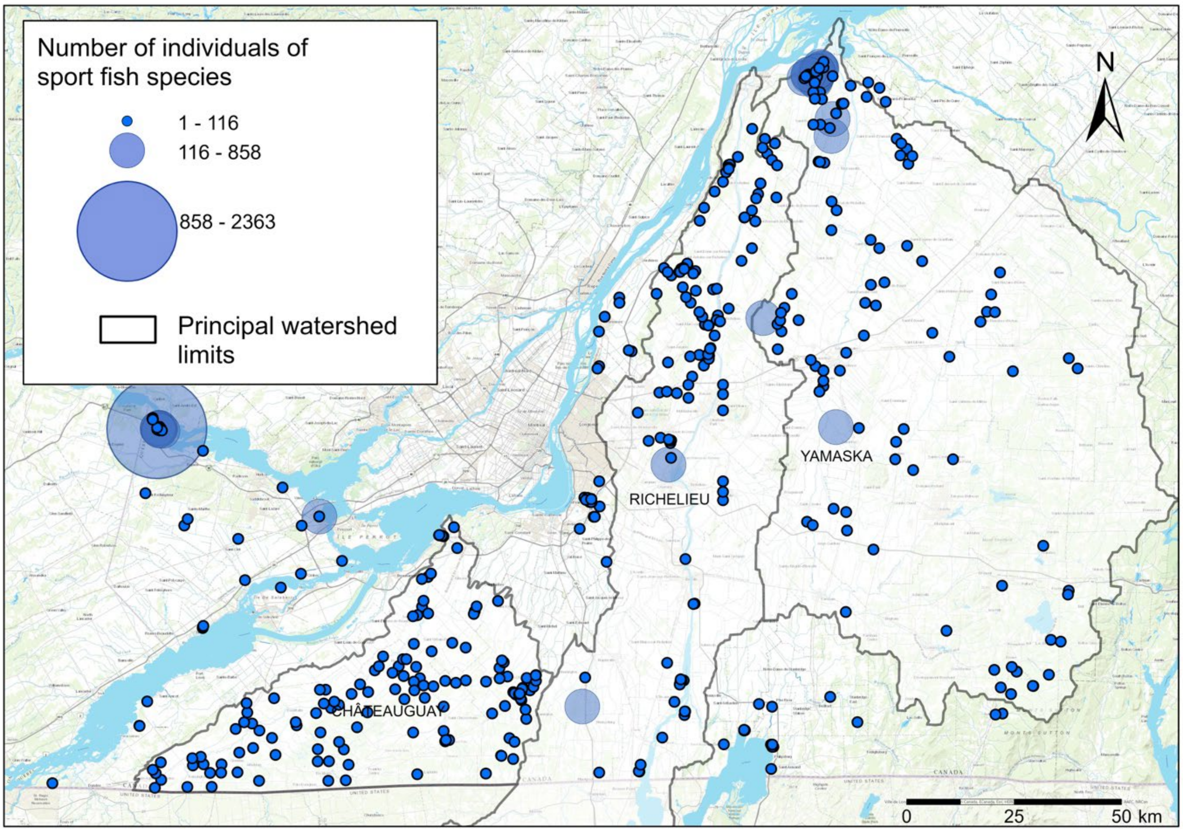


Figure 7. Sport fish species found in small watercourses in Montérégie.

The data suggests that small streams in Montérégie can support healthy communities with high biodiversity, including habitat provision for species of special concern and of sporting interest. Wetlands and floodplains connected to large tributaries are particularly rich in all types of species including species of special concern and species of sport interest, although these areas are also subject to pressures such as the presence of non-native or invasive competing species. These environments are well known to have high fish biodiversity, often supporting rare and endangered species due to their connection to large river systems that serve as migratory pathways (Junk et al., 2006). However, these wetlands and floodplains are also vulnerable to invasion by exotic species due to their high habitat diversity and natural disturbance regime (Junk et al. 2006). Therefore, it is important to preserve wetlands and floodplains connected to major river systems to promote the prevalence of native species and limit the spread of invasive ones. It is also essential to maintain good quality streams as they also support rich and diverse populations that contribute to the maintenance of species sustainability.

4.4.3 Ichthyological diversity and restoration potential of very small streams

The monitoring of the 6 very small streams (VSS) allowed us to draw various conclusions. First, VSS host abundant and diversified fish communities. Indeed, from 2 to 15 species have been recorded in these streams, totalling 19 different species (Gravel, 2021). Within the 6 VSS sampled in this specific study, species of sport interest were present in low diversity and abundance. In addition, no pollution-sensitive species were found, nor were there any species of precarious status. However, it is expected that the communities in very small streams would be more homogeneous. For example, more than 95% of the individuals caught in VSS in this specific monitoring study were in the omnivore and insectivore trophic guilds. Omnivores are considered to be a reflection of water quality and habitat degradation, as they have less specific dietary requirements than fish with more specialized diets such as insectivores (La Violette et al., 2003). Generalist insectivores are expected to dominate in variable environments such as headwater streams or highly modified streams in the U.S. Midwest (D'Ambrosio et al. 2014) that are comparable to Montérégie from a bio geoclimatic perspective.

Even though the structure of fish communities in VSS remain simple, post-monitoring suggests that with tangible improvements to the physical quality of habitat, one can measure an improvement in the fish communities (biodiversity, abundance, condition). For example, in the Brome Missisquoi MRC site (Branch 6 of the Poulin Stream), which had been improved by the creation of a two-stage channel over a distance of 1 km, went from a homogeneous to a heterogeneous habitat. This configuration favoured a concentration of flow which, even during low flow periods, generated a diversity of water depths, velocity, and substrates, offering fish a good diversity of micro-habitats. Post-work fish monitoring showed an increase in abundance and diversity, as well as an improvement in the condition of the fish community based on observed trophic guilds and reproductive types, whereas these variables either decreased or remained the same in the control streams (Gravel, 2021). Such results suggest that VSS simplified by dredging activities have lower biological productivity than their full potential.

In contrast, two of the restorations did not lead to tangible improvements in the short-term physical quality of the VSS with a consequent lack of change in the fish community (Gravel, 2021). These observations suggest that it is not so much the magnitude of the enhancement that matters as the magnitude of the effect on the physical quality of the wetted habitat. For example, even simple, small sections providing shaded cover or occasional pools appear to promote a concentration of fish in these areas.

4.5 Discussion

4.5.1 Fish community health in small streams

There are 118 species of freshwater fish in Quebec (Mingelbier et al. 2016). It is notable that 85 of these 118 species can be found in small streams in Montérégie, including species of precarious status and sporting interest. This testifies to the great richness of small streams in this territory whilst undergoing strong anthropic pressure. To bring this into perspective, according to the fisheries carried out in the St. Lawrence River between 1995 and 2019 within the framework of the Réseau de suivi ichtyologique (“fish monitoring network” - a major ichthyological inventory program conducted by the Ministry of Environment (MELCCFP), 79 species were identified, with all fishing gear combined (seine, net and trawl), between Montreal and Batiscan (Marc Mingelbier, pers. comm.).

It is also interesting to analyse the possible impact of land use changes in Montérégie based on temporal evolution and spatial distribution of fish communities in these small streams. For example, the concentration of fish community quality degradation in the central region (Figure 4) appears to follow the high concentration of agriculture observed in this same region (Figure 2A). In contrast, the relatively higher quality of fish communities in the south (Figure 3) corresponds to a higher proportion of forest cover area (Figure 2B). This analysis does show a significant relationship between fish community quality and land cover at the sub watershed scale. The percentage of area occupied by agricultural activities negatively affects the health of fish communities ($R^2 = 0.76$), while the proportion of forest cover positively affects their health ($R^2 = 0.78$). However, at a more regional scale, i.e. by MRC, forest cover has a greater effect on the average health of the fish community than agricultural cover, although this effect remains less statistically strong than when compared at the sub-watershed scale ($R^2 = 0.22$). Agricultural land use, on the other hand, has a negligible regional effect on average fish community health at the MRC scale ($R^2 \approx 0$). It should be noted that in the 1980s and 1990s, impressive water treatment efforts were carried out in Quebec, mainly in municipal, domestic and industrial wastewater, which may have improved water quality and thus reduced anthropogenic pressure on fish communities. The 1990s also marked a peak in terms of "agricultural modernization", a process that led to an intensification of practices through mechanization and the increasing use of agricultural inputs (Ruiz, 2019). Thus, on the wastewater side, there was an improvement in water quality, while on the agricultural side, there was an intensification of pressures on aquatic environments.

Other studies, however, have concluded that land use type is not always a reliable indicator of fish community structure, and some streams in agricultural settings have even shown potential to maintain fish diversity comparable to that of protected areas (Tóth et al., 2019; Wang et al., 1997). Richard and Giroux (2004) noted that agricultural streams with forested sections showed a significant difference in fish community composition and health. One hypothesis would be that it is more specifically the "naturalness" of the stream and its proximal zone that would be important for maintaining stream quality. A meta-analysis conducted by Environment Canada to guide conservation and restoration efforts also advocated that we preserve: a forest area of at least 2 km^2 per watershed, 30 - 50% of the watershed with wooded area, at least 30 m of natural riparian buffer on each stream bank, and that the vegetation remain natural for 75% of the length of the watercourse (Environment Canada, 2013). In addition, to provide a greater effect on the stream, the riparian buffer should provide a tall, dense, and diverse canopy, providing 60% or more shade cover (Bentrup, 2008). From the samples collected in this study, approximately 51% of the sub watersheds had at least 2 km^2 of forest area per watershed, 33% of the sub watershed had at least 30% or more with wooded area, and about 35% of streams sampled had more than 50% natural riparian buffer along the length of the watercourse. Currently, the majority of the Monteregian landscape does not meet these criteria, and the biological quality indicators reflect this.

4.5.2 Fish community health in larger watersheds

This study focused on small streams, but the results are quite consistent with previous assessments of fish communities in larger rivers in Montérégie, Quebec. In the Châteauguay River, for example, a 1993 sampling campaign revealed that ecosystem integrity as measured by fish and benthic invertebrates was considered good to excellent in the upstream Quebec portion of this watershed, primarily in the Haut-Saint-Laurent MRC, while the quality of fish communities was significantly lower downstream (Simoneau, 2007). The decrease in ecological integrity has been attributed to agricultural and industrial runoff from the Fèves, des Anglais, and Esturgeon Rivers, while the upper portion remains relatively healthy with inputs of clean water coming from the Trout River due to the joint efforts of the surrounding municipalities to treat their wastewater (Simoneau, 2007). Thus, there has been an improvement in the IBI at the headwaters, and a degradation at the downstream end, for which causal data are only available in relation to water quality.

In the Richelieu River, in the downstream sectors (MRC La Vallée du Richelieu), the data collected around 1995 indicated that the degradation of fish communities was acute (Saint-Jacques, 1998). This could be explained in part by the fact that wastewater was still not treated at the time of data collection. It should be noted that there are still regular overflows and other "one-time incidents" of pollutant discharges into Quebec's small streams. Also, since the database of the DGF includes sampling for various purposes, it is possible that it includes sampling carried out following reported fish kills, for example, which would influence the quality of the overall assessment. According to Saint-Jacques (1998), downstream of the Richelieu River, where there is an accumulation of agricultural runoff and pollution, fish biomass has increased (due to overproduction) and the number of species has decreased. But again, due to lack of data, it remains difficult to distinguish the effect of water quality, land use or physical quality of the streams (large or small) in the river system.

Finally, according to the data available for the third largest watershed in Montérégie, 155 km of the Yamaska River length was studied where biotic integrity was considered mostly fair (79 km, 51%) and poor (33 km, 21%) with only 2% of the study area rated as excellent (La Violette, 1999). Not surprisingly, the fish communities downstream of the Yamaska River (located near the mouth of the river) showed the poorest quality, probably due to point source effluent (La

Violette, 1999), but also likely due to the significant occupation of its watershed by agricultural activities, which occupy more than 60% of the area.

An important finding of this study is the importance of wetlands and floodplains along the edges of major river systems. This is consistent with Environment Canada (2013), which states that wetlands are critical components of fish habitat. These environments provide a very important contribution to fish productivity and for biodiversity. They are recognized as critical habitats for supporting productivity and especially fishing activities. Based on these findings, it could be advantageous to prioritize the restoration of small streams located near large rivers or the St. Lawrence River.

4.5.3 Limitations of the study

To date, Quebec has developed several tools to evaluate the integrity of aquatic ecosystems, but most are based on physico-chemical and bacteriological parameters or on small organisms (benthic insects, diatoms), thus lacking a proper monitoring program for fish communities or standardized methods for collecting fish inventories. Furthermore, none of the studies have so far been able to distinguish the relative importance of the physical quality of the streams (including their riparian zones) on the quality of the fish communities as well as on water quality, which is not optimal for making informed decisions on the restoration of small watercourses.

Given these shortcomings, it is likely that the picture presented here underestimates the true diversity of small streams. Furthermore, due to the extended time period of the data, this picture may no longer reflect the current state of the communities. It is therefore clear that the spatial and temporal coverage of the fish communities of small streams could be improved. Currently, there is no standardized method for collecting inventories, and many areas have never been surveyed, or are only covered by aging data. For example, in the Montérégie region, the database of the DGF is populated by fisheries that may have had different experimental objectives. It is therefore necessary to take into account the sampling biases associated with the monitoring objectives, for example: the lack of precision regarding fishing effort, which prevents the calculation of CPUE, the choice of gear, which may omit the capture of certain groups of fish, even though they are present, and the seasonal bias (most of the fishing is done during low flow, i.e. in August and September), which does not reflect the temporal variability of the communities. In addition, very little data on physical habitat quality has been collected. All these factors influence the representativeness of the data presented here. This demonstrates the importance of developing a standardized approach to conducting ichthyological inventories, and of deploying sufficient resources to ensure that such monitoring is integrated into discussions aimed at preserving the aquatic ecosystem.

Finally, although the IBI has been successful in assessing the overall ecological status of small streams in Montérégie, caution must always be exercised in the selection and interpretation of individual biological parameters. For future use, each IBI metric should be interpreted and selected individually for each situation and should not be applied where it is inappropriate, such as in streams where there are not enough fish that can be captured.

4.5.4 Perspectives for restoration

Although the emphasis in Quebec has long been placed on the importance of water quality, it appears from the monitoring of very small streams presented here that the physical quality of the channel should be taken into consideration to improve the aquatic ecosystem. Indeed, fish communities was shown to respond positively to an improvement in physical quality. According to Wasson et al. (1995), traditional interventions that maintain small streams in simplified forms (linear and deep trapezoidal shapes) have reduced fish biomass by 80% or more.

In order to better guide the restoration of small and very small streams and to increase their chances of success, it would be of interest to better identify the key criteria indicating good restoration potential; for example, consideration of fluvial processes and specific unit stream power. Also, it would be interesting to compare the effect of the following variables: (a) good physical quality of the wetted channel (e.g., balance of forms and processes), (b) amount of riparian cover present and in good condition, and (c) sufficient proportion of forest cover and wetlands in the watershed.

Given that small streams make up the vast majority of the hydrographic network and are biodiversity hotspots, it is essential to better understand how the conservation of natural stream processes in agricultural settings can contribute to improving the quality of aquatic ecosystems, especially within the perspective of climate change where resilience should be enhanced. This study certainly indicates that healthy fish populations can be maintained in agricultural areas if adequate forest areas and wetlands are also maintained.

4.6 Conclusion

Contrary to common misconceptions, small streams in agricultural landscapes can have high biodiversity and fish abundance. These waterways can support healthy communities and provide suitable habitat for many species, including species at risk and fish of sporting interest. Although heavily modified in the past for the needs of agricultural drainage, small streams also represent significant potential for restoration.

In Montérégie, the distribution of healthy fish communities is strongly correlated with the proportion of agricultural land and forest cover in the watersheds, with areas of high agricultural intensity and lack of forest having the worst biotic integrity. In very small streams degraded by frequent human intervention, restoration with two-stage channels has proven effective. Maintenance of small patches of forest could also provide significant improvements to aquatic communities. In addition, wetlands near large tributaries and floodplains have been shown to be particularly rich in biodiversity and highlight their importance as habitat to be protected.

Our understanding of aquatic ecosystems is currently limited by the lack of data on fish, particularly in small streams. There is also a need to improve standardization of fish data collection to facilitate future analyses. This study reveals that Montérégie has real potential to maintain an important biodiversity and species hub in the face of increasing biodiversity loss and climate change challenges. Given the ubiquitous presence of small streams in this region heavily impacted by human interventions, conservation plans should focus on restoring the fluvial processes of these streams in order to maintain fish biodiversity in these watersheds.

5. Morphological Quality Index (MQI), fish communities and biotic integrity in agricultural streams

This chapter was written in collaboration with my supervisor Dr. Pascale Biron. As first author I was responsible for development of the methodology, collection of data, presentation of results, production of maps, statistical analysis, and writing of the manuscript. Dr. Pascale Biron provided continuous guidance on the research method and made important contributions in the revision and feedback throughout the writing process of the manuscript. This manuscript will be submitted to the journal *Geomorphology.*

This chapter is linked to the previous chapter in that it continues the examination of fish communities in small streams in Quebec and further investigates the relationship between stream geomorphology and biological communities. The analysis uses the same fish database and utilizes the biotic integrity assessment of fish communities from the first paper to compare with the Morphological Quality Index (MQI).

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5.1 Abstract

Physical degradation of aquatic habitats is one of the leading causes of freshwater species decline. Yet, stream and river evaluation largely lack efficient tools to assess aquatic habitat quality on a large scale, especially in degraded environments such as agricultural streams. The Morphological Quality Index (MQI) is a potential tool that relies primarily on remote sensing to evaluate stream processes and their level of artificiality, thus providing greater spatial coverage at a lower cost than commonly used fish habitat indices. This research aims to evaluate stream morphology, quantified by the MQI, with biological indices such as the Index of Biological Integrity (IBI) in agricultural streams in order to improve our understanding of how stream morphology may affect biological communities and determine if the MQI could provide a holistic tool to evaluate ecological quality for biodiversity recovery. The MQI was estimated in 85 reaches located in agricultural watersheds in the Montérégie region in Quebec (Canada). Each stream was evaluated using remotely sensed data: 1-m LiDAR, historical aerial photos (1960-2010), and orthoimages. Field assessments were also conducted to validate morphological indices for the MQI. Historical fish data obtained through the Ministry of Forests, Wildlife, and Parks (MFFP) of Quebec were used to calculate the Index of Biological Integrity (IBI) in each of the 85 reaches. Results showed a significant relationship ($R^2 = 0.30$) between MQI and IBI. Two multivariate predictors of the MQI, *Continuity* and *Morphology*, were able to reliably predict the IBI considering both spatial and temporal factors ($R^2 = 0.50$). The MQI and IBI appeared to have a weaker relationship in higher quality streams, suggesting that this relationship was only effective in more degraded streams. Finally, invasive species were found to more likely inhabit poor morphological quality streams, whilst vulnerable species were more likely to avoid them. These findings indicate that the MQI could provide a useful tool to evaluate the ecological status of degraded streams at a wider scale, whilst providing insight on how to approach stream restoration for improved geomorphological function and local ecology.

Keywords

Agriculture, hydrogeomorphology, fish, biodiversity, remote sensing.

5.2 Introduction

Rivers and streams have always played an important role in the provision of strategic services to humankind. However, with the oncoming of the industrialized era since the early 19th century, our impact on natural watercourses has expanded significantly (Schmutz & Sendzimir, 2018). Flow modification and degradation of habitat which includes dredging, dams, and river straightening are some of the leading causes of population and range decline in freshwaters species (Dudgeon et al., 2006), with only 14% of the world's river basin area having fish populations that have escaped significant human degradation (Su et al., 2021). Consequently, freshwater habitats are now becoming one of the fastest declining ecosystems for biodiversity in the world (Tickner et al., 2020).

Watercourses in agricultural watersheds are particularly degraded due to channelization, a common practice where streams are artificially straightened and narrowed, usually to redirect and evacuate water more quickly. Channel straightening is known to reduce the variability of instream habitat complexity and therefore, affect the composition and abundance of biological communities (Frothingham et al., 2001; Heatherly et al., 2007; Lau et al., 2006). Higher flow velocities, for example, may force aquatic species to adapt or be replaced by generalist or invasive communities which are more suited to altered hydraulic conditions (Schmutz & Sendzimir, 2018). Due to a deeper riverbed caused by artificial excavation and aggravated incision, channelized streams are also designed for a higher flow capacity than naturally meandering streams eliminating natural flooding and flood pulses (Steiger et al., 1998). This loss of lateral floodplain connectivity can translate to the loss of wetland habitat as well as restricted lateral migration of species and therefore lower resilience to adverse conditions (Schmutz & Sendzimir, 2018). These physical processes are essential to maintaining ecological habitats in-stream, wetlands, and other floodplain habitats.

Despite several thousands of kilometres of streams being subjected to channelization-type developments (for example an estimated 30,000 km in Quebec, Canada; Beaulieu, 2001; Rousseau & Biron, 2009), very little is known about the morphological and ecological state of these streams and our understanding of how physical processes shape ecological habitats and communities is poor (Elosegi et al., 2010). In particular, degraded agricultural watersheds with warm water streams have often been neglected compared to salmonid streams (Talmage et al., 2002). Although there have been several ecological studies that investigate the relationship between physical habitat and biodiversity, many use static unit characterizations (e.g. pool, riffle, runs), without taking into consideration the dynamic nature of streams and rivers. Notably, the limitations arise from the spatiotemporal constraints of these methods where there is a lack of consideration of habitat at a wider scale and of temporal variation (Rinaldi et al., 2013).

The Morphology Quality Index (MQI) is a tool to assess stream quality whilst considering geomorphological processes. The MQI consists of 28 indicators that measure anthropogenic disturbance and is measured using a scoring system that evaluates functionality, artificiality, and historical channel adjustments (Rinaldi et al., 2016). The MQI aims to be relatively simple to use as advanced expertise is not required compared to traditional geomorphological analysis. Most of its metrics can also be assessed remotely, significantly reducing field cost, and expanding spatial coverage (Lemay et al., 2021). There is also evidence that the MQI is a good predictor for fish habitat based on the Qualitative Habitat Evaluation Index (QHEI) (Lemay et al., 2021). Although unit habitat characterizations are useful for ecological assessments, they cannot replace an accurate biological assessment. Therefore, linking the relationship between geomorphology, habitat, as well as biological communities could make the MQI a holistic tool for ecological assessment in small streams.

This research aims to incorporate geomorphological assessment using the MQI with that of fish populations to provide a more comprehensive understanding between the dynamic morphology of streams and biological communities. Understanding how stream morphology affect species biodiversity is crucial to the restoration of streams and conservation of species. Providing further biological significance to the MQI may also facilitate its use as an ecological evaluation tool. Stream evaluation could become more integrated, accessible, and holistic to conservationists and land users, whilst increasing the potential for better ecological monitoring and the conservation of aquatic environments. Agricultural watersheds are an ideal environment to observe the effect of geomorphology on ecological communities as they are a large source of aquatic degradation whilst also having the best restoration potential as it is easier to implement habitat changes on agricultural land than it is in an urbanized environment. The tested hypothesis in this study is that since geomorphology and hence the MQI has a strong influence on fish habitat quality, it is predicted that the MQI will also be a reliable indicator for biological integrity. Given the relative simplicity of using the MQI and its ability to be assessed remotely, the MQI has important implications as a tool for providing holistic, large-scale ecological assessment and a better understanding of how to improve stream processes and ecological well-being.

5.3 Materials and Methods

5.3.1 Study Area

The study area is the region of Montérégie, located in the south of Quebec (Canada) near Montreal. It is an area of relatively low gradient and fertile land considered highly agricultural with nearly 60% of its territory occupied by croplands, earning its title as the breadbasket of Quebec (MAPAQ, 2014) (Figure 8). In the second half of the 20^{th} century, stream channelization was very common in this region where many of these streams were once meandering. Major modifications have thus been made to several streams by straightening and implementation of trapezoidal channels through recurring dredging practices (Lemay et al., 2021). The focus of this study is on small agricultural streams, mainly stream orders 1, 2 and 3, with in the vast majority drainage area less than 130 km^2 (i.e. the small stream definition of the Unites States Environmental Protection Agency; Davis et al., 1996; MPCA, 2014).

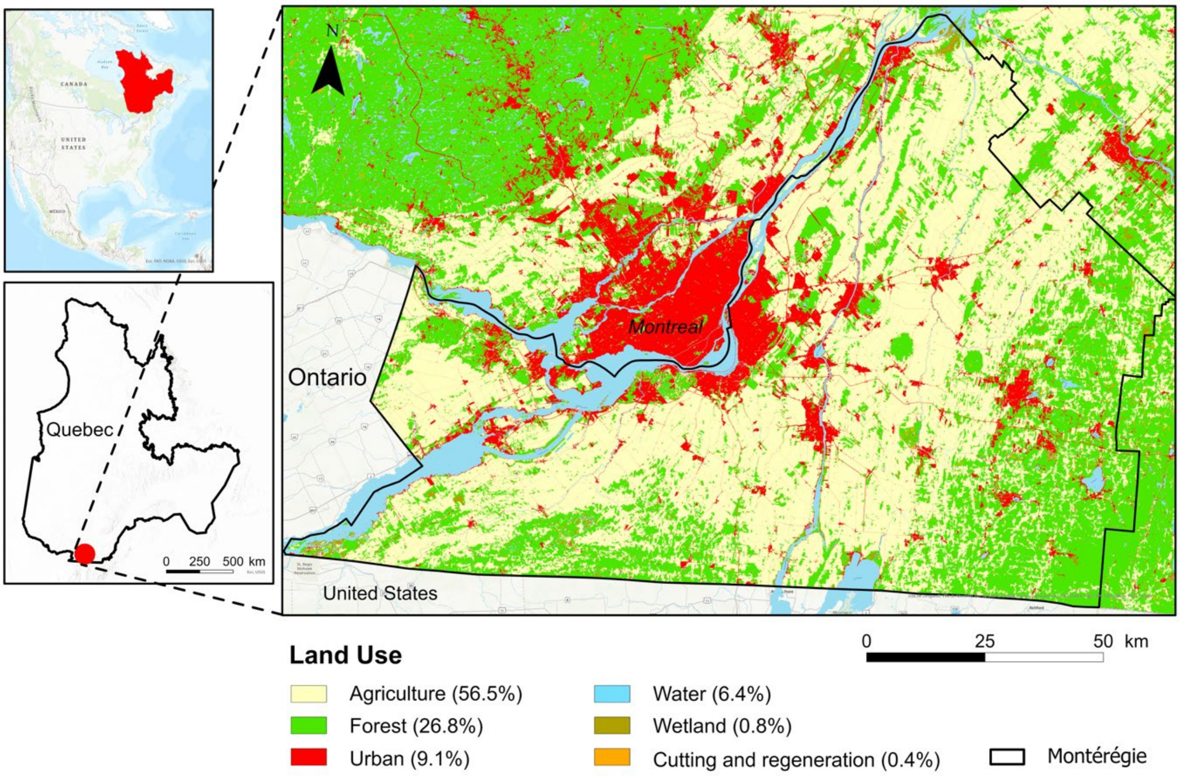


Figure 8. Land use in Montérégie, Quebec and the surrounding area (Land use data from: Gouvernement du Québec, 2019).

5.3.2 Morphological Quality Index (MQI)

MQI data was collected in the region of Montérégie across 20 different areas (Figure 9), in 36 streams and 85 stream reaches during low flow stages (June – August) between 2019 and 2022. The average drainage area of these streams is 36 km^2 (median of 25 km^2, ranging from 0.3 to 153 km^2) indicating a strong dominance of small streams. About 87% of these streams were mostly agricultural (>50% land use in the sub watershed) and the other 13% were mostly forested or natural (e.g. wetlands).

To evaluate MQI, a preliminary assessment was made to acquire general information about each stream using ArcGIS software, high-resolution LiDAR (Light Detection And Ranging) DEMs (Digital Elevation Models), and other soil and land use data. Each stream was then subdivided into homogenous reaches based on similar geomorphic characteristics, such as sinuosity, soil type, and vegetation and were typically between 1-5km long. These streams were evaluated using the partly confined and unconfined MQI sheet which uses 23 out of the 28 available indicators (Rinaldi et al., 2016). Out of the 20 stream sites studied, 15 were clearly channelized. Field validation of the indicators were assessed using observational analysis and validated by local land users for historical accounts and long-term observations, as well as the use of aerial photography using drones. Remote MQI (RMQI) was also assessed in certain areas using only remote sensing tools (Lemay et al., 2021). In this case, about 18-21 of the indicators could be assessed remotely depending on satellite imagery resolution and available data, while a few metrics could only be acquired using field observations (e.g. bed clogging, bed sills, in-channel large wood, and cross-sectional variability) (Lemay et al., 2021). The final MQI score is a range between 0 to 1, 1 indicating the most natural river (Table 5.1). This range is divided into five categories: *Very good*, *Good*, *Moderate*, *Poor*, and *Very poor*. Furthermore, the MQI's indicators can also be divided into different categories of sub metrics which are: *Continuity*, *Morphology*, and *Vegetation*. *Continuity* is a measure of the connection of river processes such as flow and sediment transport along both longitudinal and lateral continuity. *Morphology* refers to the presence of channel patterns such as cross-sectional variability and the presence of floodplain landforms. Finally, *Vegetation* is a measurement of the functional riparian cover both along the width and the length of the channel (Rinaldi et al., 2016).

Table 5.1. MQI score boundaries and corresponding quality category.

MQI	QUALITY CLASS
0.85 ≤ MQI ≤ 1.0	*Very Good or High*
0.7 ≤ MQI < 0.85	*Good*
0.5 ≤ MQI < 0.7	*Moderate*
0.3 ≤ MQI < 0.5	*Poor*
0.0 ≤ MQI < 0.3	*Very Poor or Bad*

5.3.3 Index of Biotic Integrity (IBI)

Fish data were obtained for the Montérégie region from the Quebec Ministry of Forests, Wildlife and Parks (MFFP). This database comprises of fish data collected by the MFFP for their own research purposes from the 1930s to most recently 2019, adding up to a total of 1,228 samples altogether (Chapter 4). A total of 334 samples overlapped with stream reaches where MQI was computed (Figure 9). Since the MQI data was only collected in recent years, the fish data analysis also looked separately at the more recent (> 2009) fishing efforts (35 reaches and 103 fish samples) from this database for the comparison with MQI data.

The Index of Biological Integrity (IBI) was originally developed as a tool for biological assessment by Karr (1986) in the early 1980s to evaluate stream quality based on fish assemblages in the Midwest of the United States. However, it has since been adapted to many other regions and ecosystems (Karr, 1991). The IBI has been adapted for small streams in Quebec (Saint-Jacques & Richard, 2002; Richard & Giroux, 2004; Théberge & Côté, 2008) and metrics were selected based on their appropriateness for the region and data availability. Therefore, the metrics selected for this study were: relative biodiversity (measured by the Shannon-Wiener Index), proportion of benthic insectivores, tolerant species, and omnivores; as well as the catch per unit effort (CPUE) (Table 5.2). Each fish sample was then calculated using the IBI metrics and divided into a tripartite distribution using the 95^{th} percentile method (Barbour et al., 1999), to assign score boundaries between *Good, Moderate,* and *Poor* fish communities for the region (Table 5.2). The scores for each metric were then added up to give the final IBI score (Table 5.3). Following Richard & Giroux (2004), the final scores were also doubled as to closer resemble the original IBI scores with 12 metrics by Karr (1986). Since this fish dataset was taken over many different years and projects, one notable limitation is the multiple use of different methods of fishing. This was addressed by grouping each fishing method into five distinct categories based on type of fishing gear and statistically similar catch per unit efforts (CPUE). The following fishing method groups were determined: electrofishing, fyke nets, small nets, seine nets and shore seines. Only comparable fishing methods were therefore compared by calculating the tripartite distribution within each of these five groups (Chapter 4).

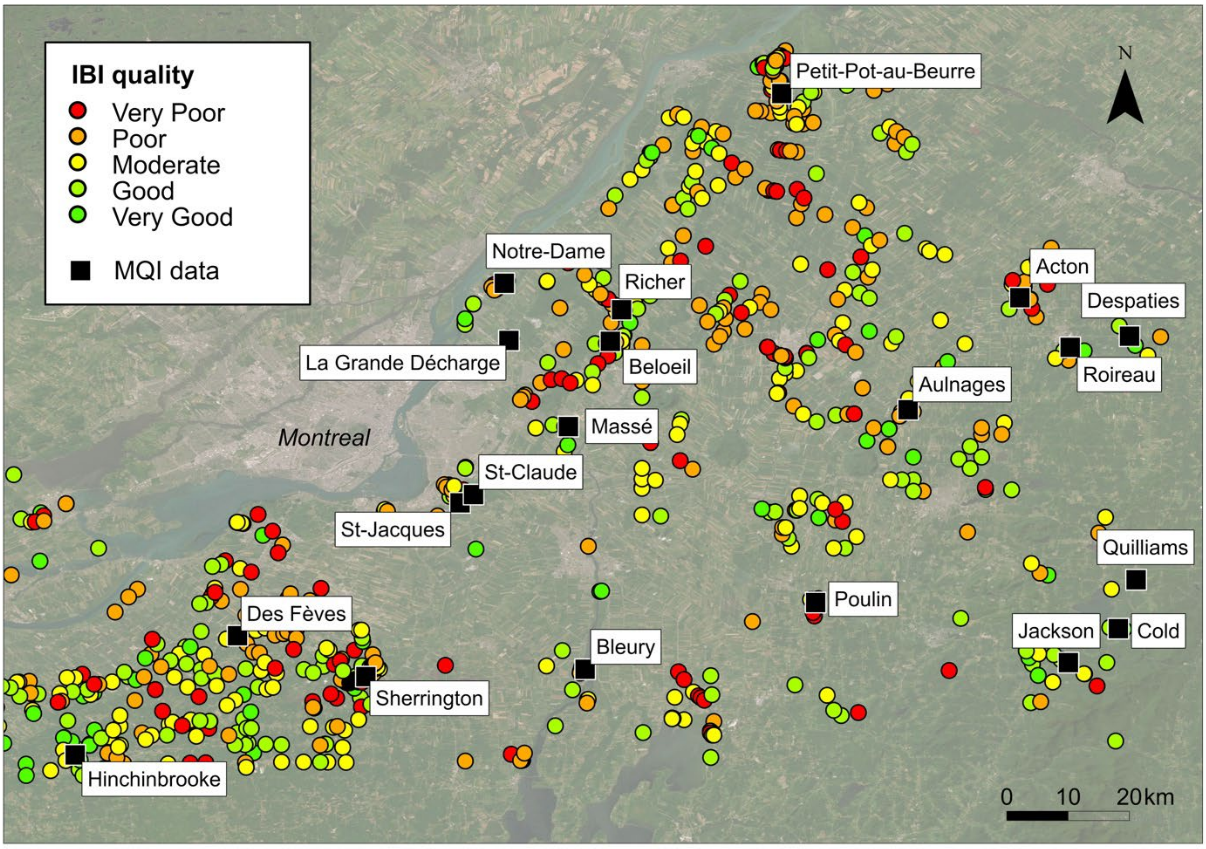


Figure 9. Distribution of MFFP fish data from which IBI quality was computed and MQI data collected in Montérégie.

Table 5.2. IBI metrics and their respective score boundaries for Poor (1), Moderate (2), Good (3) classes (adapted from Karr, 1991).

		Scoring criteria		
Category	Metric	Good 5	Moderate 3	Poor 1
Species Richness and composition	1. Shannon-Wiener Index	>1.4	>0.87-1.4	<0.87
	2. Number of Tolerant Species (%)	<46%	46%-83%	>83%
Trophic composition	3. Number of Omnivores (%)	<37%	>37%-73%	>73%
	4. Number of Benthic Insectivores (%)	>13%	>0%-<13%	<0%
Fish abundance and condition	5. CPUE (catch/s)	>0.13-<134	>0-0.13 or >134-<507	<0.13 or >507

Table 5.3. Total IBI score and corresponding quality category after adding up scoring criteria from each metric and multiplied by two.

IBI	QUALITY CLASS
46-50	*Very Good*
38-42	*Good*
30-34	*Moderate*
22-26	*Poor*
10-18	*Very Poor*

5.3.4 Statistical Analysis

A simple linear regression was conducted to determine whether the independent variable (MQI score), can predict the dependent variable (IBI score). We tested whether there was a significant relationship between MQI and IBI score by verifying whether the intercept and slope were significantly different from zero. Standard regression diagnostics was also analysed to determine the accuracy of the fit. However, since the model did not fit by observing the properties of the residuals, a non-linear model was tested (polynomial regression) and individual multiple predictors (such as MQI sub metrics of continuity, morphology and vegetation, Rinaldi et al., 2016) was considered in a multiple regression model. Finally, a principal components analysis (PCA) was conducted to analyse all biological variables, geomorphological variables, and other variables such as year and location, to determine whether there are underlying relationships with each other.

5.4 Results

5.4.1 The Morphological Quality Index in agricultural streams

All the studied Montérégie streams were found in predominantly agricultural watersheds with sub-watersheds having an agricultural land use >50%. In most of these agricultural watersheds, the morphological quality was on average *Moderate* to *Poor* (MQI score of 0.36 – 0.66) (Figure 10). Although about 75% of these streams were severely channelized, had hard banks on either side, and were stripped of functional vegetation (Figure 11A), others kept considerable vegetation cover, meandering channels, and wider buffers. The absence of dams, the presence of wider functional vegetation, and less channelized reaches led to higher MQI scores and usually higher IBI scores as well (Figure 11B).

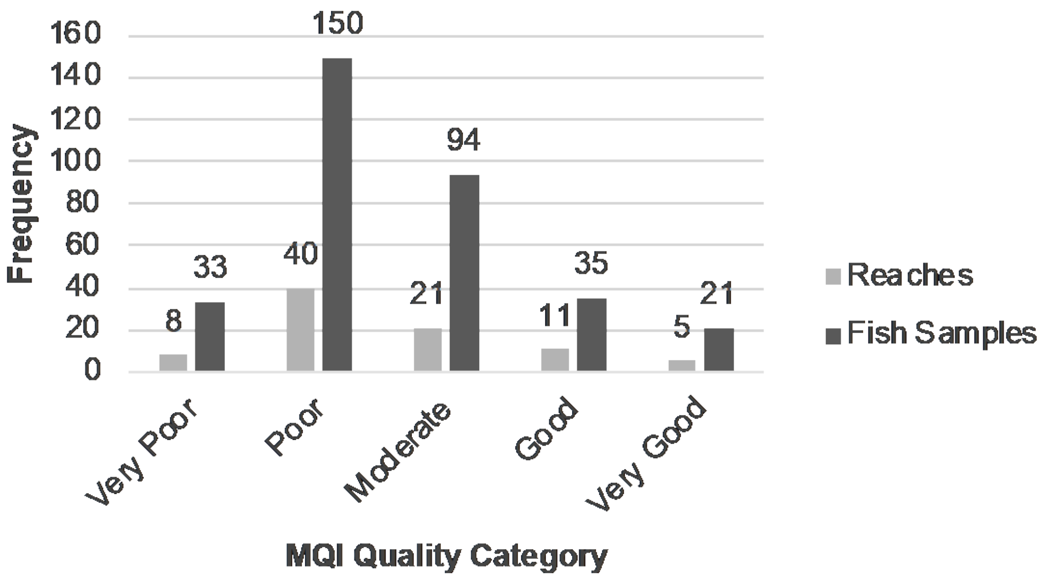


Figure 10. Distribution of number of reaches and fish samples by MQI quality category.

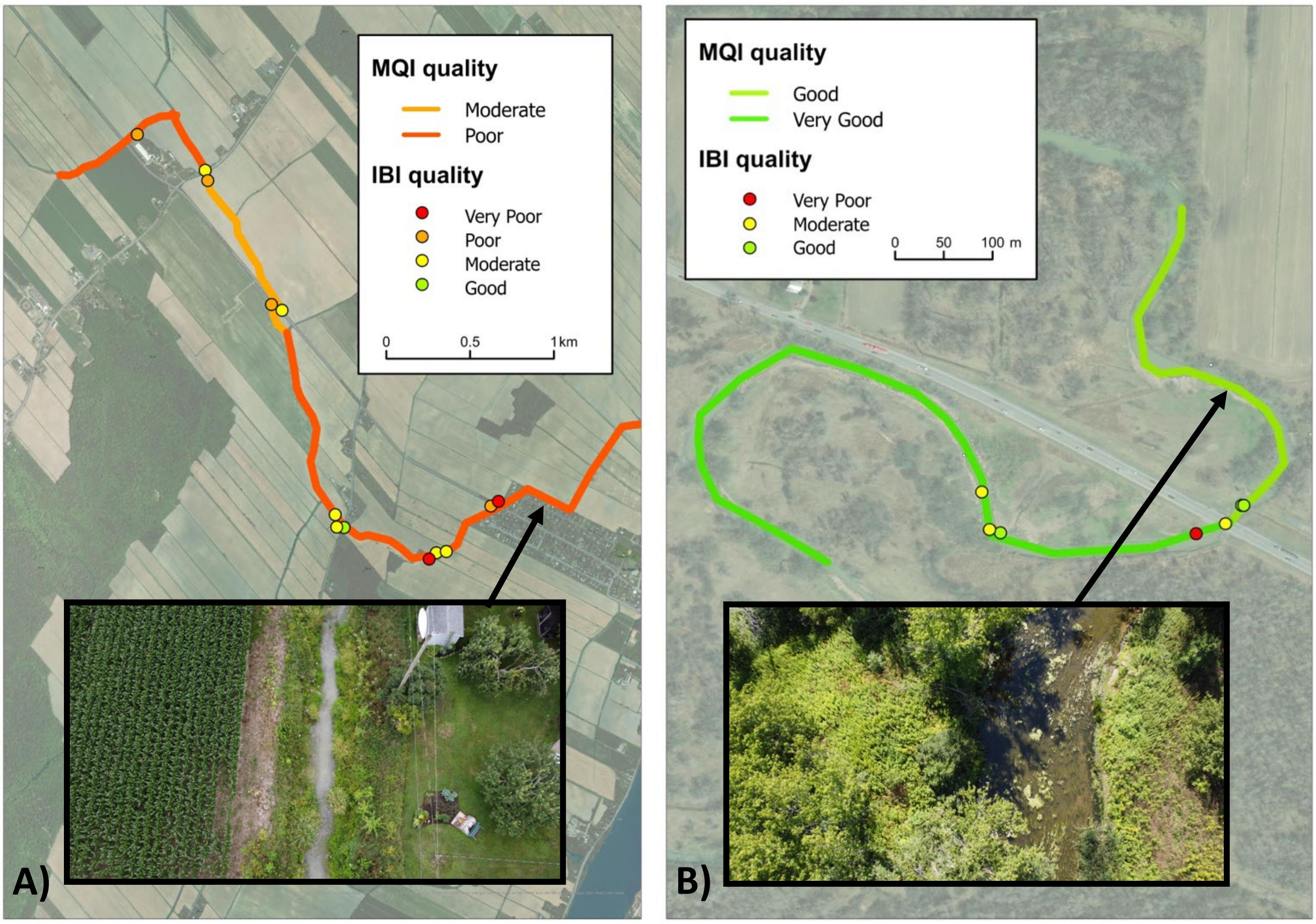


Figure 11. Examples of a poor morphological quality stream (Ruisseau Richer) (A) and good morphological quality stream (Rivière St-Jacques) (B).

5.4.2 Relationship between the IBI and the MQI

Unlike MQI scores which were mainly limited to the *Moderate* and *Poor* categories (Figure 10), the quality in IBI varied greatly within the Montérégie streams (IBI score 10 – 50) (Figure 9). As a result, when comparing between the MQI and the IBI's quality categories (i.e. *Very Good* to *Very Poor*), the two metrics did not correspond well with one another. A quantitative analysis of the MQI and IBI numerical scores was also conducted using only more recent IBI data. This results in a stronger relationship, especially with fish data taken in the last 10 years (>2009). Using a polynomial regression analysis, the MQI explains about 30% of the variance in the IBI (R^2 = 0.30) for fish samples taken in the last 10 years (Figure 12). There is a significant positive relationship between MQI and IBI scores up until an MQI score of around 0.75 (so from the *Poor* to *Good* categories) and above that the relationship is no longer significant, showing a lot of scatter and even a negative trend. Note that these *Good and Very Good* quality streams are located near suburban areas where factors other than the morphological quality may affect fish communities.

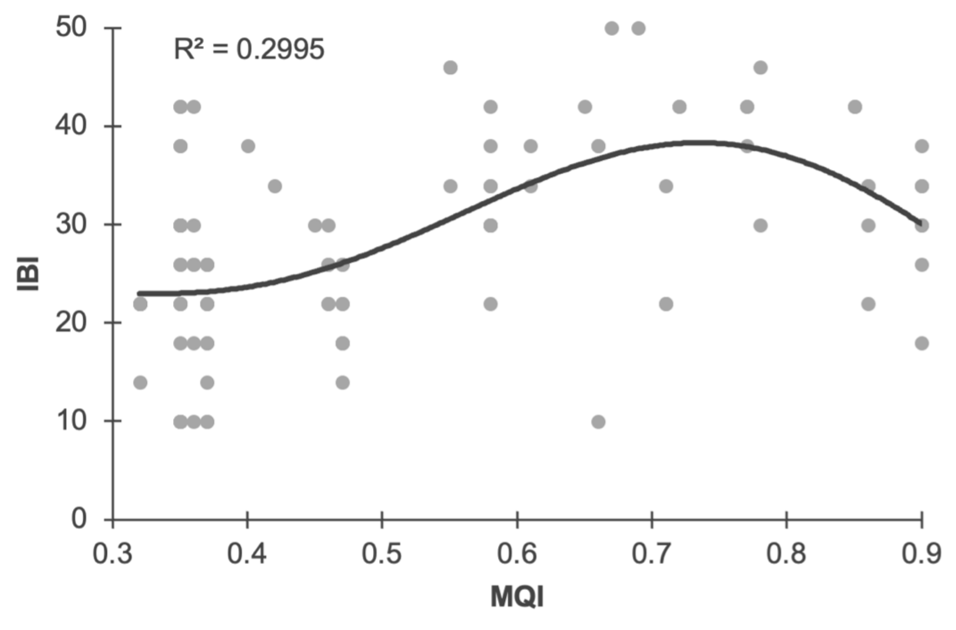


Figure 12. Polynomial regression between MQI and IBI in the Montérégie area.

5.4.3 Relationship between the IBI and the MQI sub metrics

The MQI is also composed of sub metrics which can be further explored in relation to fish communities. When each of the MQI sub metrics were tested to best predict the IBI, *Continuity* was found to be the single best predictor of the IBI with an $R^2 = 0.37$ using a polynomial regression (Figure 13). Spatial factors were also found to improve the overall prediction of the variance in the relationship between MQI and IBI. Using a multivariate regression analysis using recent IBI data (>2009); location (i.e., coordinates), *Morphology*, and *Continuity* were together the best predictors of the IBI with an $R^2 = 0.50$ which is the strongest predictability between the MQI sub metrics and the IBI in this study.

In addition to determining the relationship between the IBI and different MQI sub metrics, the effect of different morphological sub metrics on individual biological indicators within the IBI were also examined. Overall, healthier streams as measured by the MQI had a positive effect on the health of fish communities, especially in terms of biodiversity (as measured by the Shannon-Wiener Index) and the presence of benthic insectivores which are sensitive species. *Continuity*, once again, has the strongest overall influence on these two biological metrics, with a coefficient of correlation (r) of 0.38 and 0.33, respectively. The morphological variables influenced the biological variables as expected where better quality streams increased overall biodiversity and the number of benthic insectivores which are both positive indicators. Meanwhile, worse morphological quality streams appeared to have a negative correlation with tolerant species and catch per unit effort, and finally had a negligible effect on omnivorous species (Figure 14).

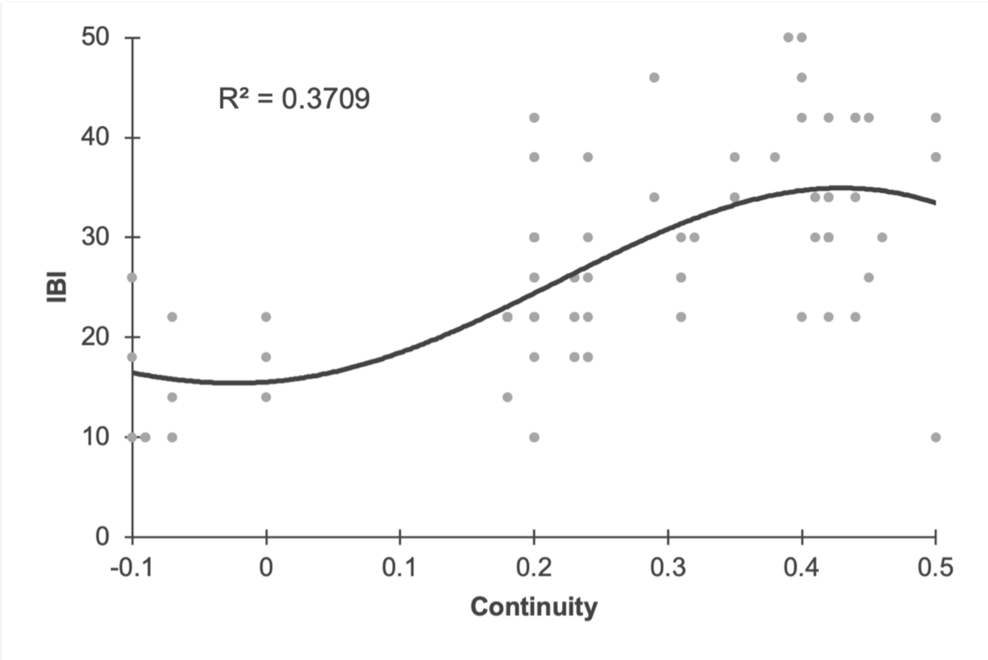


Figure 13. Polynomial regression between Continuity and IBI.

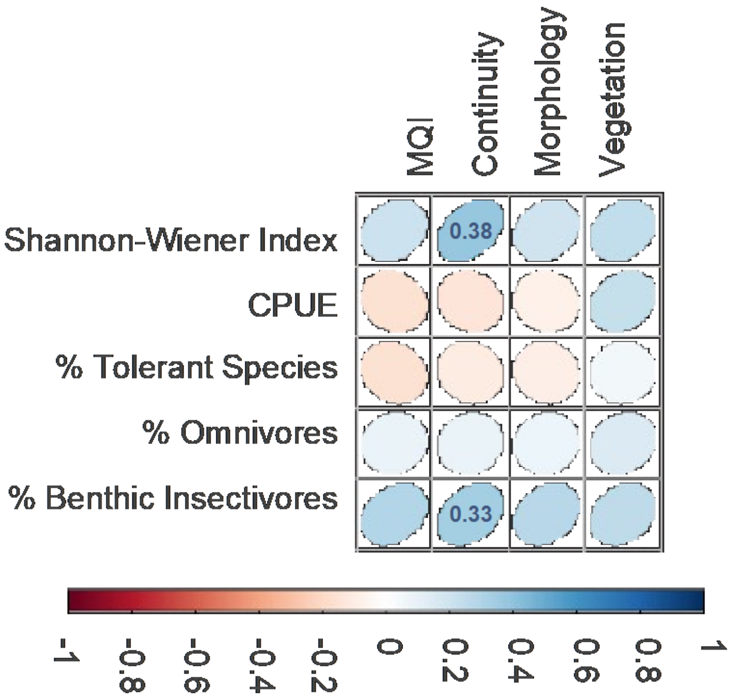


Figure 14. Correlation plot between MQI and IBI sub-metrics. The two highest correlation coefficients (0.38 and 0.33) are for the Continuity MQI sub metric.

5.4.4 Relationship between biological communities and land use

Overall, in all the 1,228 fish samples collected in small agricultural streams, there were at least 85 different species of fish caught, including 4 vulnerable species, 2 threatened species, and 6 invasive species according to Québec's provincial designations (Chapter 4). Invasive species were more likely to inhabit *Poor* quality streams (52%), whilst vulnerable species were completely absent from *Poor* quality streams and instead mostly inhabited *Moderate* (99%) and *Good* (1%) quality streams (Figure 15). Streams in wetlands near large tributaries in particular were found to have high biodiversity and provide important habitat for vulnerable and threatened species (Chapter 4). When looking at how the land use in each sub watershed affected overall fish community health, IBI scores were strongly negatively correlated with agricultural land use ($R^2 = 0.79$), and strongly positively correlated with natural land cover ($R^2 = 0.78$). Streams in sub watersheds with higher agricultural land use tended to have a more important presence of omnivores and tolerant species, while sub watersheds with more presence of natural land use such as forestry and wetlands, tended to have higher biodiversity and more sensitive species such as benthic insectivores.

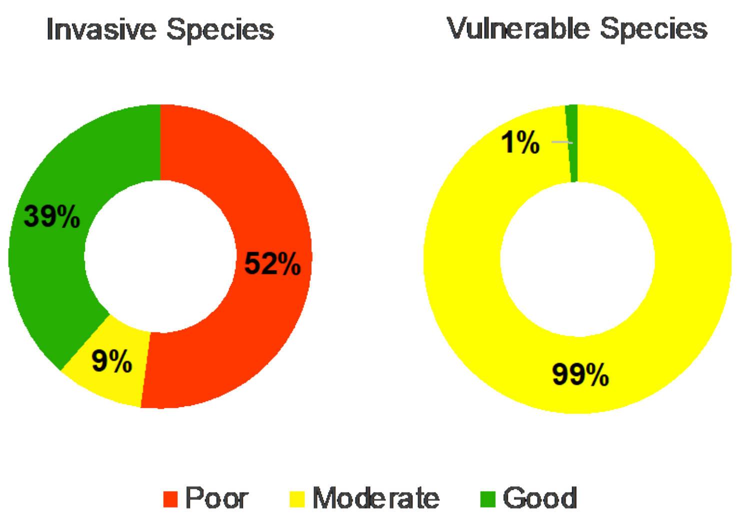


Figure 15. Distribution of vulnerable and invasive species in different morphological quality streams.

5.5 Discussion

In general, it is expected to find very little or poor-quality fish communities in agricultural landscapes, since these areas are more prone to anthropization and stream degradation. Despite agricultural land use being widespread in southern Quebec and stream channelization common, very little is known on the extent of degradation in these types of watersheds and how it affects biological communities. This study aimed to quantify the degradation of these streams using the MQI and investigating the relationship between MQI quality and fish community health present using the IBI.

Agricultural land use at the sub-watershed scale has a strong negative effect on the quality of fish communities. Other studies support this finding in that there was a clear biotic decline in fish communities when the percentage of agricultural land use exceeded 50% in the watershed, which was the case in Montérégie (Wang et al., 1997). Fish in agricultural streams tended to have a higher presence of pollution tolerant species, omnivores, and a high percentage of fish with physical deformities (Richard & Giroux, 2004). Although agricultural land use in the watershed has a clear negative impact on fish communities, many healthy fish communities could still be found in these agricultural streams. For example, some streams that were in sub watersheds with greater than 80% agricultural land use still had *Good* to *Very Good* quality IBI scores and tended to have higher than average MQI scores (0.54). A study by Wang et al. (1997) has also shown that even in watersheds where agricultural land use exceeded 80% some sites still maintained good biotic integrity comparable to protected sites. This suggests that even in highly agricultural landscapes, healthy fish communities can flourish if streams maintain good geomorphological quality. Some of these streams indeed showed high fish biodiversity, significant abundances, and provided habitat for sensitive and vulnerable species.

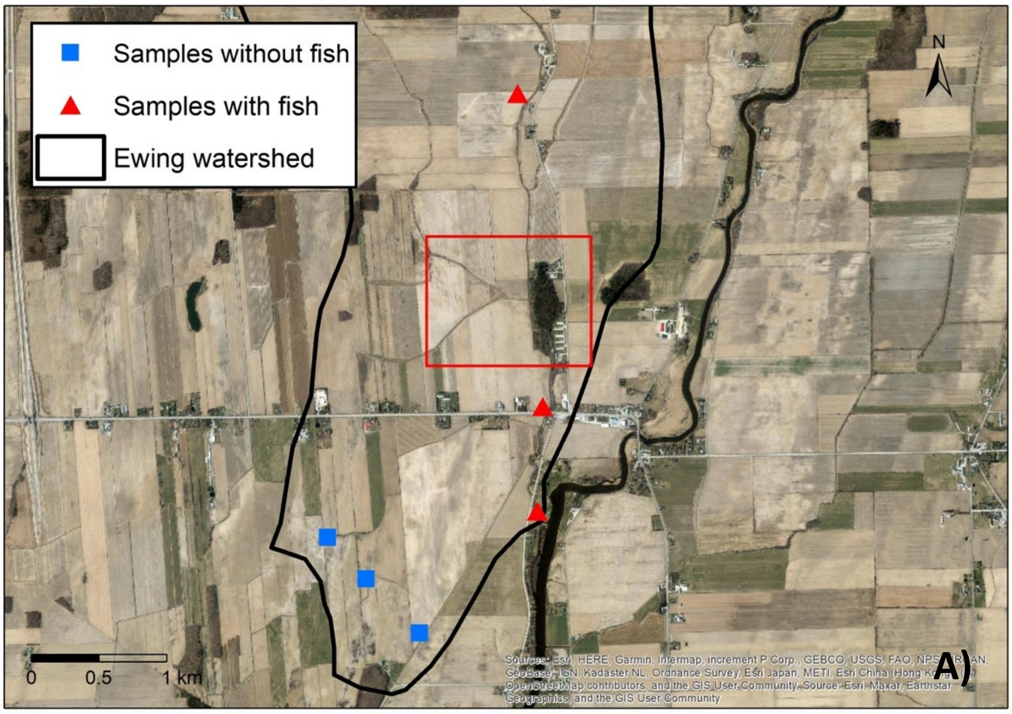
Most of the streams sampled in the region of Montérégie were of *Moderate* to *Poor* quality, which is expected in this primarily agricultural region. However, IBI scores varied widely within these streams from *Very Poor* to *Very Good* quality, with a statistically significant (R^2 = 0.30) relationship between the MQI and IBI for recent fish data, i.e. those that correspond to the period when the MQI was collected (in the last 10 years). This suggests that overall, the geomorphological characteristics of the MQI have an important effect on biological communities. Considering fewer reaches (n = 35) and fish samples (n = 103) could be compared when only including more recent fish data, it is possible that the strength of the relationship is not as strong than if a larger database with more recent fish data was available. Indeed, the reduced (recent) fish sample dataset is less normally distributed than the overall sample, which can affect the relationship between the IBI and MQI.

The MQI/IBI relationship also appears to fade above an MQI score of about 0.75 with even a slight negative correlation. This suggests that after a certain quality of stream is attained, the MQI loses its predictability over the IBI, and fish communities no longer significantly improve in response to better morphological stream conditions. In terms of restoration, this suggests that the MQI can be used most effectively for measuring ecological restoration in relatively more degraded streams. Other factors could have also affected why the IBI was found to be worse than expected in *Good* to *Very good* quality streams in this study. Some of these higher quality streams were found in proximity to residential areas and could have also been affected by water quality issues such as household effluent. However, additional data from similar studies focusing on high quality salmonid streams also revealed a lack of relationship between the MQI and IBI in *Good* to *Very Good* quality MQI streams in New Zealand and Ontario (Canada) (Foote et al., in review). *Continuity* was found to be the single best MQI sub metric in predicting the IBI with an $R^2 = 0.37$. *Continuity* is a measure of the connection of river processes, such as flow and sediment transport, both longitudinally and laterally. Therefore, it appears as though stream modifications such as non-permeable dams and immovable banks which were observed in several of the Montérégie streams, would have a particularly strong influence on biological communities. Finally, a multivariate analysis revealed that location, *Continuity* and *Morphology* were overall the best predictors of the IBI with an $R^2 = 0.50$. Considering that there are many other abiotic variables such as water quality or temperature which were not included in this assessment and other biotic factors that determine the composition of stream communities, an explanation of 50% of the variance in biological integrity by the geomorphological metrics in the MQI is relatively important. Considering how the MQI focuses primarily on reach scale geomorphological indicators, the addition of watershed scale indicators such as land use which has shown to be strongly correlated with IBI (Chapter 4) may even improve this predictability. Indeed, many other biological factors can influence fish population and composition including recruitment, competition, mortality, and exotic species (de la Hoz Franco & Budy, 2005; Gebrekiros, 2016; Jackson et al., 2001). Location was found to be an important variable in this study as fish community health varied depending on their location within Montérégie. For example, the central part of this region which has intensified agricultural land use and fewer forested areas than in the south, is where poorer fish communities were found. Whereas areas further from the centre of Montérégie and in the South where there was a higher abundance of natural land cover, was where the healthiest population of fish communities tended to be. Both *Continuity* and *Morphology* are important indicators of the naturalness of the physical channel, therefore fish communities tended to degrade with more severe forms of channelization. This is supported by the findings by Wang et al. (1997) where sites that still maintained good biotic integrity even when agricultural land use exceeded 80% tended to share similar geomorphological qualities, namely rocky substrates, and were not channelized. Although *Vegetation* did not synergistically give the best outcome for the prediction of the IBI, it still had a slight effect on the overall IBI with an $R^2 = 0.17$. Further studies by the ministry of environment in Québec also determined that reach level presence of agricultural and forested sections in the same stream showed significant difference in fish community composition and health (Richard & Giroux, 2004). This suggests that agricultural land use is not the only determining factor that influences fish community health and composition, but rather that local geomorphological qualities also have an important effect on biotic integrity even in agriculture dominant landscapes. Using the MQI as a practical tool to measure stream quality thus has a strong potential for understanding how we can improve streams for better local ecology.

5.5.1 Temporal evolution of RMQI

The RMQI provides a means to assess how streams' morphological quality has evolved in time and how this temporal evolution can affect fish communities. For example, fish data were collected in the Ewing watershed in Montérégie as early as the 1930s to most recently in 1996 (Figure 16A). This stream used to be meandering and was straightened in the 1970s. Historical photographs from 1964, so prior to channelization (Figure 16B,C), were used to compute a historical RMQI in this watershed. Three fish samples were taken in the Ewing stream before the 70s, i.e., one in 1941 and two in 1965 when the RMQI was estimated as 0.75, which is of good morphological quality. Although only three samples were taken, 422 fish were caught using seine methods and 17 different species were found. The fish communities caught in the Ewing stream had an average Shannon-Wiener Index of 1.62 which is considered excellent when compared to other small streams in Montérégie. Its average trophic composition for omnivores (39%) and benthic insectivores (25%) are also considered excellent, and tolerant species (37%) also excellent. The historical fish data and RMQI assessed through satellite imagery suggests that the Ewing stream before the 1970s was in good ecological and morphological condition.

After the 1970s, streams in the Ewing watershed became heavily channelized and consequently the more recent RMQI score is 0.47, which is of poor morphological quality. Three fish samples were collected after the 1970s in this watershed where no fish were caught (Figure 16A). The samples were collected using electrofishing methods in the Howick-Leduc stream and the Ligne de Noyan stream in 1996, which have been channelized since the 60s. This suggests there was a higher abundance and biodiversity of fish prior to the 1970s and that the straightening of these streams resulted in degradation of both their physical and biological condition. Of course, such conclusions must be made with caution as there are clear limitations in making historical comparisons where fish samples were taken from different streams, with different historical interventions, and fishing methods. Therefore, the difference between the abundance and biodiversity of fish before and after the 70s could be attributed to several factors including channel straightening, the natural occurrence of fish in these streams, and fishing methods used. However, it does appear as though the health of fish communities were better prior to the 1970s in non-channelized streams, compared to more recently channelized streams in the Ewing watershed.



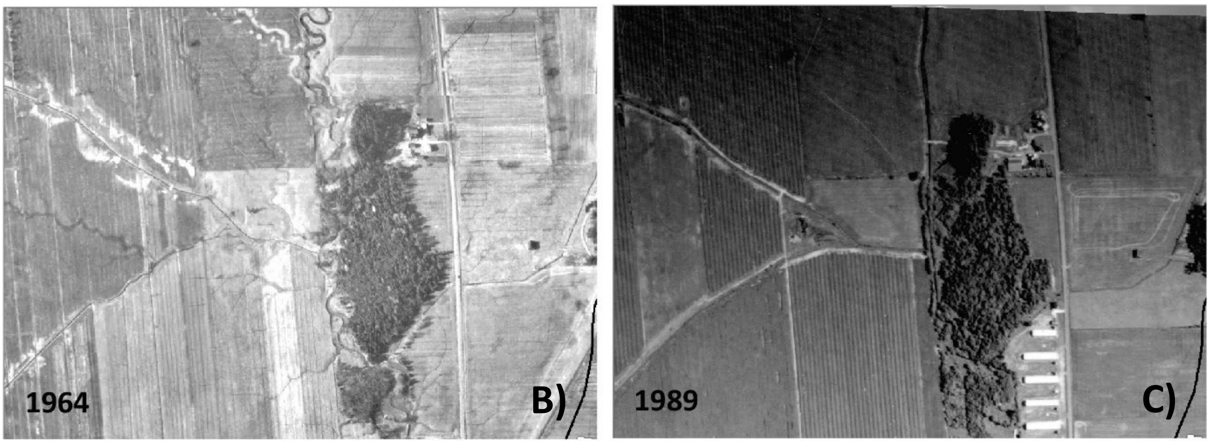


Figure 16. Fish data in the Ewing watershed. Red triangles indicate samples taken before the 1970s, and blue squares represent samples taken after the 1970s. The red rectangle corresponds to the straightened streams illustrated in figure 16B (before straightening) & C (after straightening).

5.5.2 Limitations

Although the MQI and IBI have been effective tools in measuring stream's morphological and biological quality, there are a few limitations worth mentioning. The largest limitation in the biological dataset is the lack of standardization among all the fish samples. Since, the dataset is an agglomeration of samples taken from different projects, with different purposes, and using multiple types of fishing methods; the lack of standardization can affect the biological results from the abundance of fish caught, the type of species captured, to the type of data that was recorded. Although this study attempted to standardize the fishing methods by statistically grouping different fishing methods, some discrepancy can still occur. As for the MQI, since it is largely a qualitative method of assessment, different assessors who contributed to the collection of the MQI may have different sensitivities to certain metrics based on their experience and knowledge and may assess the same reach slightly differently. Although all the assessors who collected the MQI had a basic understanding in geomorphology, it is still possible that some judgments differed and may have affected the consistency of MQI scores. However, overall, the categorization and assessment of streams were similar. Since not all MQI could be field validated, an inherent limitation of the RMQI was the occasional limited resolution or obstruction of view in identifying and assessing certain features. Field validations occasionally revealed unforeseen features that were not possible to see using satellite imagery and could cause discrepancies with the MQI. However overall, the MQI and RMQI remained statistically similar — Lemay et al. (2021) obtained a correlation of coefficient of 0.98 between RMQI and MQI — and could be rectified by gathering additional data from other sources (e.g.

regional dam directories, land use assessments, local municipal authorities etc.). Finally, since the majority of streams assessed in the Montérégie region have had some measure of intervention, many were found to be of *Poor* to *Moderate* quality. Having a wider range of streams in other quality categories with available fish data taken in more recent years would have increased the sample size and possibly improved the relationship trends between the two indices.

5.6 Conclusion

Although agricultural land use strongly affected the geomorphological quality of most streams in Montérégie, some streams with better geomorphological quality were found to have significant presence of healthy biological communities and biodiversity. The MQI was able to quantify the level of degradation in these streams and link a predictable outcome in the aquatic communities that inhabited them. Most importantly, the MQI provides insight on which morphological attributes we can improve to better improve certain biological communities and to measure this progress. Since the MQI tool can largely be used remotely, it is significantly less resource intensive than standard fish habitat indices such as QHEI which require field observations. It is also relatively easy to use for land users or conservation authorities with clear guidelines (Rinaldi et al., 2016). There is thus great potential to better understand streams in degraded environments and prioritize restoration areas, for example by focusing on poor morphological quality zones located close to high MQI reaches. Incorporating an MQI assessment in various watershed restoration projects provides an opportunity for widespread application in a large percentage of the hydrographic network to improve small agricultural streams and slow down habitat and biodiversity loss.

6. General conclusions

Small streams in agricultural settings have long been neglected in terms of ecological recovery, however this study shows that even in intensely modified landscapes, these streams have a surprising wealth of biological diversity and abundance than previously thought. These trends closely follow land use patterns in the region and attest to how significantly land use management affects biodiversity, whilst identifying particularly important habitats such as fluvial wetlands to protect. Applied tools such as the MQI illustrate that stream degradation and its effect on biodiversity can be measured and demonstrates direct links between improved channel morphology and biological communities. The MQI also identifies specific morphological characteristics that can be quantified and improved for better community composition, namely allowing fluvial processes to operate and avoiding straightening of naturally meandering streams. This provides potential for the MQI as an effective tool for land users and researchers to better understand our streams and how to approach improved aquatic management and restoration.

This study focused on small streams in Montérégie (Quebec), but the findings have applications far beyond this territory. Indeed, many countries or regions have severely degraded watercourses in agricultural watersheds which need to be restored. In the future, better monitoring of fish in these small streams could help fill in the gaps of understanding, make better assessments, and monitoring changes of the current ecological state of agricultural streams. The MQI could also be further assessed with other biological groups such as macroinvertebrates or focus on specific morphological metrics such as the role of channel vegetation and watershed coverage on the prevalence of aquatic biodiversity in these landscapes. Small streams make up a significant portion of our hydrographic network and also provide the most potential for restoration and better management. Previously there lacked sufficient conventional tools outside of research and specified professionals to quantifiably address stream degradation and ways to improve aquatic biodiversity. However, this study has shown that the MQI could potentially benefit the need for an efficient and effective method to measure and improve streams, thus enhancing local ecology.

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8. Appendix A

MQI Score Computation

A total score is computed as the sum of scores across all components and aspects.

The Morphological Alteration Index (MAI) is first defined as:

MAI = Stot / Smax

Where Stot is the sum of the scores, and Smax is the maximum score that could be reached when all appropriate indicators are in class C (see tables below). Therefore, MAI ranges from 0 (no alteration) to 1 (maximum alteration).

The Morphological Quality Index (MQI) is then defined as:

MQI = 1 – MAI = 1 – Stot/Smax

GEOMORPHOLOGICAL FUNCTIONALITY

CONTINUITY

F1	Longitudinal continuity in sediment and wood flux	score	selection
A	Absence of alteration in the continuity of sediment and wood	0	
B	Slight alteration (obstacles to the flux but with no interception)	3	
C	Strong alteration (discontinuity of channel forms and interception of sediment and wood)	5	

F2	Presence of a modern floodplain	score	selection
A	Presence of a continuous (>66% of the reach) and wide modern floodplain	0	
B1	Presence of a discontinuous (10÷66%) but wide modern floodplain or >66% but narrow	2	
B2	Presence of a discontinuous (10÷66%) and narrow modern floodplain	3	
C	Absence of a modern floodplain or negligible presence (≤10% of any width)	5	

Not evaluated in the case of mountain streams along steep (>3%) alluvial fans

F4	Processes of bank retreat	score	selection
A	Bank erosion occurs for >10% and is distributed along >33% of the reach	0	

B	Bank erosion occurs for ≤10%, or for >10% but is concentrated along ≤33% of the reach or significant presence (>25%) of eroding banks by mass failures	2	
C	Complete absence (≤2%) or widespread presence (>50%) of eroding banks by mass failures	3	
Not evaluated in the case of Low Energy ERT types from 17 to 22 and Groundwater-Fed streams			

F5	**Presence of a potentially erodible corridor**	score	selection
A	Presence of a wide potentially erodible corridor (EC) for a length >66% of the reach	0	
B	Presence of a potentially EC of any width for 33-66% of the reach or for >66% but narrow	2	
C	Presence of a potentially EC of any width but for ≤33% of the reach	3	

F7	**Planform pattern**	score	selection
A	Absence (≤5%) of alteration of the natural heterogeneity of geomorphic units and channel width	0	
B	Alterations for a limited portion of the reach (≤33%)	3	
C	Consistent alterations for a significant portion of the reach (>33%)	5	

F8	**Presence of typical fluvial landforms in the floodplain**	score	selection
A	Presence of floodplain landforms (oxbow lakes, secondary channels, etc.)	0	
B	Presence of traces of landforms (abandoned during the last decades) but with possible reactivation	2	
C	Complete absence of floodplain landforms	3	

F9	**Variability of the cross-section**	score	selection
A	Absence (≤5%) of alteration of the cross-section natural heterogeneity (channel depth)	0	
B	Presence of alteration (cross-section homogeneity) for a limited portion of the reach (≤33%)	3	
C	Presence of alteration (cross-section homogeneity) for a significant portion of the reach (>33%)	5	
Not evaluated in the case of low energy straight, sinuous, meandering or anabranching channels with natural absence of bars (lowland rivers, low gradients and/or low bedload) and groundwater-fed streams (natural cross-section homogeneity)			

F10	Structure of the channel bed	score	selection
A	Natural heterogeneity of bed sediments and no significant armouring and/or clogging	0	
B	Evident armouring or clogging for ≤50% of the reach	2	
C1	Evident armouring or clogging (>50%), or burial (≤50%), or occasional substrate outcrops	5	
C2	Evident burial (>50%), or substrate outcrops or alteration by bed revetments (>33% of the reach)	6	

Not evaluated for deep rivers when it is not possible to observe the channel bed

F11	Presence of in-channel large wood	score	selection
A	Significant presence of large wood along the whole reach (or "wood transport" reach)	0	
B	Negligible presence of large wood for ≤50% of the reach	2	
C	Negligible presence of large wood for >50% of the reach	3	

Not evaluated above the tree-line and in streams with natural absence of riparian vegetation (e.g. north-European tundra)

F12	Width of functional vegetation	score	selection
A	High width of functional vegetation	0	
B	Medium width of functional vegetation	2	
C	Low width of functional vegetation	3	

Not evaluated above the tree-line and in streams with natural absence of riparian vegetation (e.g. north-European tundra)

F13	Linear extension of functional vegetation and presence of emergent aquatic macrophytes	score	selection
A	Riparian vegetation >90% of maximum length, or riparian vegetation>33% and significant presence of emergent aquatic vegetation (low-energy channels)	0	
B	Riparian vegetation 33÷90%, or riparian vegetation >90% but very limited presence of aquatic vegetation, or riparian vegetation ≤33% but significant presence of aquatic vegetation	3	
C	Riparian vegetation ≤33%, or <90% but very limited presence of aquatic vegetation	5	

Not evaluated above the tree-line and in streams with natural absence of riparian vegetation (e.g. north-European tundra)

ARTIFICIALITY

Upstream alteration of longitudinal continuity

A1	**Upstream alteration of flows**	score	selection
A	No significant alteration (≤10%) of channel-forming discharges and with return interval>10 years	0	
B	Significant alteration (>10%) of discharges with return interval>10 years or release of increased low flows downstream dams during dry seasons	3	
C	Significant alteration (>10%) of channel-forming discharges	6	

A1H	**Upstream alteration of flows without potentially relevant effects on channel morphology**	score	selection
A	Absence of any type of structure altering flow discharges (dams or other abstractions)	0	
B	Presence in the catchment of one or more structures altering flow discharges	11	
C	Reach located between abstraction and restitution section of hydropower plant and/or reach immediately downstream of a hydropower reservoir	22	

Evaluated for the application of the HMQI

A2	**Upstream alteration of sediment discharges**	score	selection
A	Absence or negligible presence of structures for the interception of sediment fluxes (dams for drainage area ≤5% and/or check dams/abstraction weirs for drainage area ≤33%)	0	
B1	Dams (area 5-33%) and/or check dams/weirs with total bedload interception (area 33-66%) and/or check dams/weirs with partial interception (area>66%)	3	
B2	Dams (area 33-66%) and/or check dams/weirs with total bedload interception (drainage area >66% or at the upstream boundary)	6	
C1	Dams for drainage area >66%	9	
C2	Dam at the upstream boundary of the reach	12	

A3	**Alteration of flows in the reach**	score	selection
A	No significant alteration (≤10%) of channel-forming discharges and with return interval>10 years	0	
B	Significant alteration (>10%) of discharges with return interval>10 years	3	

C	Significant alteration (>10%) of channel-forming discharges	6	

A4	**Alteration of sediment discharge in the reach**	score	selection
A	Absence of structures for the interception of sediment fluxes (dams, check dams, abstraction weirs)	0	
B	*Channels with S≤1%:* consolidation check dams and/or abstraction weirs ≤1 every 1000 m *Steep channels (S>1%):* consolidation check dams ≤1 every 200 m and/or open check dams	4	
C	*Channels with S≤1%:* consolidation check dams and/or abstraction weirs >1 every 1000 m *Steep channels (S>1%):* consolidation check dams >1 every 200 m and/or retention check dams or presence of a dam or artificial reservoir at the downstream boundary (any bed slope)	6	
Where transversal structures, including bed sills and ramps (see A9), are >1 every d1, select 'x' next to 6		*6*	
Where transversal structures, including bed sills and ramps (see A9), are >1 every d2, select 'x' next to 12		*12*	
where d1=150 m and d2=100 m in steep channels; d1=750 m and d2=500 m in channels with S≤1%			

A5	**Crossing structures**	score	selection
A	Absence of crossing structures (bridges, fords, culverts)	0	
B	Presence of some crossing structure (≤1 every 1000 m in average in the reach)	2	
C	Presence of many crossing structure (>1 every 1000 m in average in the reach)	3	

A6	**Bank protections**	score	selection
A	Absence or localized presence of bank protections (≤5% total length of the banks)	0	
B	Presence of protections for ≤33% total length of the banks (sum of both banks)	3	
C	Presence of protections for >33% total length of the banks (sum of both banks)	6	
For a high density of bank protection (>50%) select 'x' next to 6		6	
For an extremely high density of bank protection (>80%) select 'x' next to 12		12	

A7	**Artificial levees**	score	selection
A		0	

	Absent or set-back levees, or presence of close and/or bank-edge levees ≤5% bank length		
B	Bank-edge levees ≤50%, or ≤33 in case of total of close and/or bank edge>90%	3	
C	Bank-edge levees >50%, or >33% in case of total of close and/or bank edge>90%	6	
For a high density of bank-edge levees (>66%) select 'x' next to 6		6	
For an extremely high density of bank-edge levees (>80%) select 'x' next to 12		12	

A8	**Artificial changes of river course**	score	selection
A	Absence of artificial changes of river course in the past (meanders cut-off, channel diversions, etc.)	0	
B	Presence of changes of river course for ≤10% of the reach length	2	
C	Presence of changes of river course for >10% of the reach length	3	
In case of historical drainage and dredged works for >50% of the reach select 'x' next to 6 (*)		6	
In case of historical drainage and dredged works for >80% of the reach select 'x' next to 12 (*)		12	
(*) when an additional score for A6 and/or A7 is not already applied			

A9	**Other bed stabilization structures**	score	selection
A	Absence of structures (bed sills/ramps) and revetments absent or localised (≤5%)	0	
B	Sills or ramps (≤1 every d) and/or revetments ≤25% permeable and/or ≤15% impermeable	3	
C1	Sills or ramps (>1 every d) and/or revetments ≤50% permeable and/or ≤33% impermeable	6	
C2	Revetments >50% permeable and/or >33% impermeable	8	
For a high density of bed revetment (impermeable >50% or permeable >80%) select 'x' next to 6		*6*	
For an extremely high density of bed revetment (impermeable >80%) select 'x' next to 12		*12*	
d=200 m in steep channels (S>1%); d= 1000 m in channels with S≤1%			

A10	**Sediment removal**	score	selection
A	Absence of recent (last 20 years) and past (last 100 years) significant sediment removal activities	0	
B1	Sediment removal activity in the past (last 100 years) but absent during last 20 years	3	
B2		4	

	Recent sediment removal activity (last 20 years) but absent in the past (last 100 years)		
C	Sediment removal activity either in the past (last 100 years) and during last 20 years	6	

A11	**Wood removal**	score	selection
A	Absence of removal of woody material at least during the last 20 years	0	
B	Partial removal of woody material during the last 20 years	2	
C	Total removal of woody material during the last 20 years	5	

Not evaluated above the tree-line and in streams with natural absence of riparian vegetation (e.g. north-European tundra)

A12	**Vegetation management**	score	selection
A	No cutting interventions on riparian (last 20 years) and aquatic vegetation (last 5 years)	0	
B	Selective cuts and/or clear cuts of riparian vegetation ≤50% of the reach and partial or no cutting of aquatic vegetation, or no cutting of riparian but partial or total cutting of aquatic vegetation	2	
C	Clear cuts of riparian vegetation >50% of the reach, or selective cuts and/or clear cuts of riparian vegetation ≤50% of the reach but total cutting of aquatic vegetation	5	

Not evaluated above the tree-line and in streams with natural absence of riparian vegetation (e.g. north-European tundra)

9. Appendix B

Limitations of the Index of Biotic Integrity (IBI)

As stated previously, an important limitation of the biological dataset was the use of different fishing methods recorded. To address this, a Tukey test was performed for each fishing method using catch per unit effort (CPUE) to compare sample biases between fishing gear. The test revealed that there were five distinct groups of methods: electrofishing, fyke nets, nets, seine, and shore seine. Each fishing method was assigned to one of these five groups based on the type of method and on statistical averages. Individual tripartite distributions were calculated within each group so that each fishing method was being evaluated with other comparable fishing gear.

Although this method is appropriate for standardizing and comparing the relative health of the region, there are still some limitations when comparing these group methods in absolutes and using different fishing gear within those groups. For example, before the 90s, seine net fishing methods were more commonly used, whereas after the 90s electrofishing became the norm. Electrofishing on average is known to catch a larger variety of species than other fishing methods (Mehdi et al., 2021). Therefore, when directly comparing quantitative values such as species biodiversity and composition, more species may appear prevalent after the 1990s but weren't captured before due to the primary use of seines resulting in an underestimation of the diversity of fish in aquatic inventories before the 90s. However, in a sense, the mixed use of different methods may also be positive in that different methods have different capture successes based on species habitat selection which means some catches may contain species that would not have been otherwise caught with other methods (Oliveira et al., 2014) but makes it difficult to compare when occurring in different periods.

Although five similar groups of fishing methods were differentiated, another limitation is that some of the fishing gear within those groups may still vary slightly from each other which might give less accurate score boundaries for that group method overall. For example, the seine group in particular does not specify net sizes which means a higher variability in catch and less consistent results. Some smaller nets with lower catch sizes may be disproportionately undervalued compared to other bigger nets in its own group. This may affect small sized nets with smaller CPUE to be scored poorly when in reality, their catch may be representative of the population for their size class. Therefore, if we had a more standardized method, there would be greater confidence in the accuracy of the score boundaries, and we would be able to compare individual biological metrics against each other (e.g. electrofishing would have a higher average biodiversity than seine methods therefore harder to compare when looking at absolute values). Depending on the fishing method, certain biological metrics may also lend themselves better to the type of gear. For example, electrofishing is better suited when looking at biodiversity and composition, whereas seine perhaps less so. Therefore, samples using electrofishing and these types of biological metrics may more accurately represent the type of biodiversity caught in this region. There are many biological, physical, and chemical factors that can affect capture efficiency in each method (Peterson & Paukert, 2009). Without standardization, these limitations can be difficult to eliminate. It is also difficult to verify whether the average abundance of each metric is considered “normal” as populations in other studies in different regions can differ greatly or may not use the same biological metrics appropriate to the same study area and dataset. Therefore, the IBI must be used cautiously whilst keeping in mind it is a method to assess relative health in a specific region, and that further studies using standardized methods will be needed to establish expected averages for a specific area.