# DEVELOPMENT, EVALUATION, AND IMPLEMENTATION OF A STANDARDIZED FISH COMMUNITY-BASED INDEX OF BIOTIC INTEGRITY FOR EVALUATING THE ECOLOGICAL HEALTH OF BOREAL PLAINS STREAMS AND RIVERS IN SASKATCHEWAN, CANADA 

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By
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#### Abstract

Freshwater ecosystems face increasing threats from anthropogenic influences and multiple stressors, necessitating effective management techniques to assess, conserve, and restore aquatic health. Fish-based Index of Biotic Integrity (IBI) tools play a crucial role in assessing and monitoring the health of freshwater ecosystems. Despite a prosperous, significant fishery and ample aquatic habitats, Saskatchewan (SK), and much of Canada's boreal region, currently lack a fish-based IBI framework, and the development and evaluation of such a tool could complement existing monitoring programs and provide a novel approach to fisheries and aquatic resource management within SK, and more broadly, northern Canada. This study developed and evaluated a fish-based IBI framework for streams and rivers of the Beaver River watershed in the Boreal Plain ecozone of SK. This watershed exhibits a gradient of human disturbance, ranging from agriculture in the south to relatively unimpacted forest landscapes in the north, making it an ideal location to study the potential effects of human stressors on fish and aquatic ecosystems and evaluate the IBI in a relatively homogenous area with multiple land-use stressors.

By assessing various measures of land use and fish habitat, I classified minimally disturbed (or low-stress) conditions, established a gradient of stream and river health throughout the Beaver River watershed at 18 sites, and then determined fish community response to known stressors. A potential limitation of fish-biomonitoring studies is the effect of seasonality and timing of sampling on the interpretation of results, especially in northern regions where temperature extremes likely influence fish reproduction and mobility. Therefore, I revisited five of the sites annually over a three-year period to test the sensitivity of the IBI to interannual variability. I identified nine metrics, selected across the major metric categories, that showed the highest responsiveness to human disturbance. As expected, IBI scores decreased with increasing stress, but a depauperate and tolerant fish community, confounded by high interannual variability in environmental conditions and the fish community, created difficulties in developing the IBI and limited my ability to attribute variations to natural trends through time or anthropogenic influence. My results reinforce the importance of long-term monitoring to decipher trends in natural variation of fish communities from variation created by anthropological stressors and can inform fisheries and aquatic ecosystem health management and decision making in SK as well as other Boreal Plains' watersheds throughout Canada.


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[^0]
## LIST OF ABBREVIATIONS

| AAFC | Agriculture and Agri-food Canada |
| :--- | :--- |
| AB | Alberta |
| ALCT | Alcott Creek |
| ANOVA | Analysis of variance |
| AUSRIVAS | Australian River Assessment System |
| BEAST | BEnthic Assessment of SedimenT |
| BKWT | Backwater Creek |
| BMI | Benthic macroinvertebrate |
| BRST | Brook stickleback |
| CA | Carnivore |
| CCME | Canadian Council of Ministers of the Environment |
| CDED | Canadian Digital Elevation Data |
| CESI | Canadian environmental sustainability indicators |
| CI | Confidence interval |
| CPAWS | Canadian Parks and Wilderness Society |
| CPOM | Coarse particulate organic matter |
| CPUE | Catch per unit effort |
| CW | Coolwater |
| DELT | Anomalies, including deformities, eroded fins, lesions, and |
| DEM | tumors |
| Digital elevation model |  |
| DFO | Department of Fisheries and Oceans Canada |
| DLRD | DeLaRonde Creek |
| DN | Dissolved nitrogen |
| DNA | Deoxyribonucleic acid |
| DNNS | Dennis Creek |
| DO | Dissolved oxygen |
| DOC | Dissolved organic carbon |
| DP | Dissolved Phosphorus |
| ECCC | Environment and Climate Change Canada |
| EEM | Environmental Effects Monitoring |
| EPA | Environmental Protection Agency |
| EUWFD | European Water Framework Directive |
| FL | Florida |
| FLTN | Flotten River |
| FNU | Formazin nephelometric units |
| FPOM | Fine particulate organic matter |
| FTMN | Fathead minnow |
| GCS | Geographic Coordinate System |
| GE | Generalist |


| GIS | Geographic Information System |
| :---: | :---: |
| IBI | Index of biotic integrity |
| IC | Invertivore-carnivore |
| ICI | Invertebrate community index |
| IN | Invertivore |
| INIT | Initial |
| INT | Intolerant |
| IQR | Interquartile range |
| LNDY | Landry Creek |
| LO | lithophil |
| MA | Massachusetts |
| MB | Manitoba |
| MD | Missing data |
| MF | Mature forest |
| MIST | Mistohay Creek |
| MMI | Multi metric index |
| MOD | Moderate tolerance |
| MPCA | Minnesota Pollution Control Agency |
| NA | Not applicable/ not assessed |
| NAD | North American Datum |
| NESS | Nesslin Creek |
| NOLN | Nolin Creek |
| NPS | Nonpoint source |
| NTS | National Topographic System |
| NTU | Nephelometric turbidity unit |
| NY | New York |
| OM | Omnivore |
| ON | Ontario |
| OSB | Oriented strand board |
| OTTR | Otter Creek |
| PL | Phytolithophil |
| PO | Phytophil |
| PPWB | Prairie Provinces Water Board |
| PS | Psammophil |
| QAQC | Quality assurance quality control |
| RAMP | Regional Aquatics Monitoring Program |
| RIVPACS | River Invertebrate Prediction and Classification System |
| RL | Reach length |
| ROBN | Robin Creek |
| SAT | Saturation |
| SD | Standard deviation |


| SHR | Shrub |
| :--- | :--- |
| SK | Saskatchewan |
| SLBY | Sulby Creek |
| SPC | Specific conductivity |
| SRC | Saskatchewan Research Council |
| SUKW | Sukaw Creek |
| SWA | Saskatchewan Watershed Authority |
| TDS | Total dissolved solids |
| TEA | Tea Creek |
| TL | Total length |
| TN | Total Nitrogen |
| TOL | Tolerant |
| TP | Total Phosphorus |
| TR | Tolerant reproductive strategies |
| TSA | Test Site Analysis |
| TSS | Total suspended solids |
| UNGS | Unknown Creek near Goodsoil |
| UNMT | Unknown Creek Near Makwa River |
| UNPG | Unknown Creek Near Pagan Lake |
| UNSW | Unknown Creek near Spiritwood |
| US | United States |
| USA | United States of America |
| USEPA | United States Environmental Protection Agency |
| USGS | United States Geological Survey |
| UTM | Universal Transverse Mercator |
| VIBI | Vegetation Index of Biotic Integrity |
| WC | Water column |
| WHSC | White sucker |
| WSA | Water Security Agency |
| WW | wetted width |
| WWF | World Wildlife Fund Canada |
| YF | Young forest |
| YOY | Young of year |
|  |  |

## Chapter 1: GENERAL INTRODUCTION

### 1.1 Introduction

Freshwater ecosystems are vulnerable to anthropogenic influences and multiple stressors (Birk et al., 2022; Ormerod et al., 2010; Reid et al., 2019) and proper management techniques are required to assess, conserve, or restore aquatic health. Freshwater is a key component for industry and agriculture, and provides many other provisioning, regulating, and cultural services, encompassing aspects such as the provision of freshwater, quality management of water resources, habitat support, erosion mitigation, climate modulation, sustenance production, and opportunities for leisure and tourism (Kaval, 2019; Tomscha et al., 2017; Vari et al., 2022). Freshwater ecosystems also provide habitat for many species (Dudgeon et al., 2006) and have an outsized contribution to biodiversity (Vega \& Wiens, 2012). Freshwater ecosystems are listed as having the highest per-hectare monetary value of all inland ecosystems (Costanza et al., 2014) yet they are considered more imperiled and at risk than marine and terrestrial environments (Dudgeon et al., 2006; McAllister et al., 1997; Ricciardi \& Rasmussen, 1999). Now more than ever, there is a continuing and growing need for research and conservation of freshwater ecosystems (Abell, 2002, Dudgeon et al., 2006; Ormerod et al., 2010; Reid et al., 2019; Strayer \& Dudgeon, 2010).

Over the past few decades, much literature has been dedicated to identifying the major stressors contributing to freshwater ecosystem degradation and biodiversity loss (Borgwardt et al., 2019; Collen et al., 2014; Craig et al., 2017). Major anthropogenic stressors on freshwater ecosystems include overexploitation of resources, landscape and hydrological alterations, invasive species, and pollution resulting from chemical spills, fertilizer, and pesticide inputs, sedimentation, and effluent/sewage discharge (Collen et al., 2014; Environment \& Climate Change Canada (ECCC), 2016; Mateo-Sagasta et al., 2017; WWF, 2018). Conventional agriculture has been listed as one of the most extensive and environmentally disruptive land use practices affecting freshwater ecosystems (Brauns et al., 2022; Mateo-Sagasta et al., 2017; United States Environmental Protection Agency, 2007). Large-scale agricultural practices common today rely on heavy use of synthetic N-P-K fertilizers and pesticides (Ali et al., 2020; Stanley \& Preetha, 2016), and significant landscape modifications and degradation (ECCC, 2016; Gliessman, 2014). This can lead to increased nutrient and chemical run-off, cultural eutrophication and depleted oxygen levels (Carpenter et al., 1998; US Geological Survey, 2010),
contaminated waterbodies (Main et al., 2015), increased erosion, turbidity, and sediment deposition (Burdon et al., 2013; Ross \& McKenna, 2023), stream thermal increases (Quinn \& Wright-Stow, 2008), freshwater habitat loss, alteration, and fragmentation (Thorpe et al., 1999), among other effects (Brauns et al., 2022; ECCC, 2016; Mateo-Sagasta et al., 2017). Fish are particularly vulnerable to these anthropogenic impacts since they spend their entire life cycle in water, and as such, are continuously exposed to aquatic contaminants and habitat degradation (Couture \& Biron, 2023; Mamun \& An, 2020; Stanley \& Preetha, 2016).

One approach to assess the effects of multiple stressors on aquatic ecosystems, known as the Index of Biotic Integrity (IBI), is a commonly used ecosystem health assessment tool that uses the fish community to characterize aquatic health (Karr, 1981; Karr et al., 1986). The IBI, initially created by $\operatorname{Karr}$ (1981), is a multimetric approach that acts as a quantitative measure of ecosystem health by incorporating data across multiple levels of organization to obtain a single index score reflecting the extent of human disturbance on natural communities and ecosystems (Barbour et al., 1999; Karr, 1981; Mack, 2004; Schoolmaster et al., 2012). The IBI synthesizes measures, known as metrics, of the biological community that change in a predictable manner with physical and chemical alterations from anthropogenic activity and disturbance (Barbour et al., 1995; Alford \& Gotwald, 2019). Ecological assessment and monitoring methods, such as the IBI, have become an integral part of aquatic ecosystem management throughout many regions of the world.

Multimetric indices, such as the IBI, are essential tools for making objective assessments of complex systems experiencing multiple stressors where the underlying causal processes are poorly understood (Schoolmaster et al., 2012), such as diffuse land use from agriculture, industry, and urbanization. Despite this, many areas still lack regulated biological assessment and monitoring. In Canada, the use of fish in ecological monitoring and assessment has often been focused on chemical tissue analysis or sentinel species (Environment Canada, 2012a), with less research involving the use of fish community assemblages to interpret ecosystem health. Furthermore, aquatic ecosystems in the prairie provinces (including Alberta, SK, and Manitoba), have an increased risk of effects due to intensive agriculture (Donald et al., 2001; Malaj et al., 2020a; Malaj et al., 2020b) among other stressors on the landscape (Davies \& Hanley, 2010; Fitzsimmons, 2001; Thorpe et al., 1999).

Saskatchewan (SK) comprises various landscapes from south to north (Pomeroy et al., 2005), with the Boreal Plain ecozone, located across central SK, comprising $\sim 1 / 3$ of the provincial area. The Boreal Plain ecozone is a transitional region, where prairie meets forest (Massie, 2014) and provides an intersection of multiple and prevalent stressors, including agriculture, forestry, and other industry. SK's boreal region is large and contains minimal, localized, human-related impacts compared to the southern populated area of the province (Davies \& Hanley, 2010; Phillips et al., 2023). However, future industrial development in this region is likely. Despite containing more than 100,000 waterbodies (Ashcroft et al., 2006; Pomeroy et al., 2005), limited research has focused on assessing the various aquatic environments in this area. SK's aquatic systems and fisheries have significant social, cultural, economical, and ecological benefits (Ashcroft et al., 2006), although the use of fish in monitoring in the province has been negligible (Davies \& Hanley, 2010; Phillips et al., 2023; Environment Canada, 2012b; Knackstedt, 2015).

For this research, various physical, chemical, and biological components of streams and rivers were sampled throughout the Beaver River watershed located in the Boreal Plain ecozone of SK, Canada, to develop a fish-based IBI. This watershed is dominated by agricultural activity in the south and a relatively unimpacted forest landscape in the north. Forestry operations occur in a patchy distribution throughout the watershed and may have more localized effects. This gradient of human disturbance across the landscape provides a unique opportunity to study the potential effects of human stressors on fish and aquatic ecosystems and evaluate the IBI in a relatively homogenous area with multiple land-use stressors. Saskatchewan, and much of Canada's boreal region, currently lack a fish-based IBI framework, and the development and evaluation of such a tool could complement existing monitoring programs and may provide a novel approach to fisheries and aquatic resource management within SK, and more broadly, northern Canada. Developing and evaluating approaches to assess broad-scale non-point source or localized point source impacts and cumulative effects from multiple stressors will contribute to the sustainability of aquatic ecosystems.

## Chapter 2: LITERATURE REVIEW

### 2.1 Biological Assessment and Monitoring

The concept of maintaining a healthy ecosystem (Costanza \& Mageau, 1999; Schaeffer et al., 1988) and achieving ecological integrity (Canada National Parks ACT, 2000) were initially founded on the increasing awareness of environmental degradation beginning in the early 1970s (e.g., Clean Water Act, 1972, 1977; Stockholm Declaration on the Human Environment, 1972; Rio Declaration on Environment and Development, 1992). Society increasingly became aware that to preserve a healthy ecosystem it is integral that not only should the physical and chemical integrity of a system be intact, but also the biotic integrity (Canada National Parks ACT, 2000, US Environmental Protection Agency (USEPA), 2011). This advanced approach led to the introduction of biological (biotic) integrity as an objective of the 1977 U.S. Clean Water Act and Canada's National Park Act (2000) and broke ground for biological monitoring and assessment. Biotic integrity is defined as "the ability to support and maintain a balanced, integrated, adaptive community of organisms, having a species composition, diversity, and functional organization comparable to that of natural habitats in a region" (Frey, 1977; Karr \& Dudley, 1981; USEPA, 2011).

Declining aquatic resource health resulted in the initial development and implementation of methods to manage and conserve the biotic integrity of aquatic ecosystems (Karr, 1981; Lazorchak et al., 2002). Traditionally, nonbiological measures, such as physical and chemical water quality monitoring, were used in environmental monitoring and assessments to manage aquatic ecosystem health (Karr \& Dudley, 1981). Physicochemical assessment and monitoring often relied on point source pollution and toxic chemical concentrations, overlooking significant anthropogenic perturbations such as cumulative and nonpoint source impacts, alteration of watershed hydrology and energy sources, and degradation of physical habitat structure (Karr et al., 1986). As a result, physicochemical monitoring can fail to detect degradation, manage, and protect aquatic resources, and improve biotic integrity, since it is an indirect measure of biological and ecological condition (Herricks \& Schaeffer, 1985; Karr et al., 1986). Understanding that healthy aquatic ecosystems can support balanced and diverse biological
communities led to the incorporation of biological monitoring (biomonitoring) into aquatic management and monitoring programs (Karr, 1981; Karr et al., 1986). Biological assessment, often referred to as bioassessment, involves evaluating the state of a waterbody by conducting biological surveys and utilizing direct measurements of biological indicators (also known as bioindicators) within aquatic ecosystems (Barbour et al., 1999; Lazorchak et al., 2002; USEPA, 2011). Assessments of physical (structure and flow regime) and chemical (water quality) habitat conditions, including instream and riparian areas, are typically conducted alongside bioassessments (Barbour et al., 1999; USEPA, 2011). Bioassessments are not intended to replace traditional physicochemical monitoring, but instead are meant to complement these measures and provide a comprehensive, accurate, and rapid assessment of ecological health, prior to more indepth ecological monitoring to determine the cause of any observed impairment (Barbour et al., 1999, USEPA, 2011).

### 2.1.1 Multimetric Indices

Stewardship of ecological integrity requires the ability to make integrative assessments of biological and ecological resources. This led to the development of multimetric indices (MMIs) which incorporate data across multiple levels of organization to obtain a single index reflecting the extent of human disturbance (Schoolmaster et al., 2012). In ecology, the MMI is a common monitoring and assessment tool that acts as a quantitative measure of ecosystem health and is used to assess and indicate the effect of human disturbance on natural communities (Alford \& Gotwald, 2019; Barbour et al., 1999; Lee et al., 2018). The MMI synthesizes measures, known as metrics, of the biological community that are responsive to physical and chemical alterations brought on by human activity and disturbance. Metrics are characteristics of the fish, or other taxonomic, assemblage that have known predictable responses with increasing environmental disturbance (Barbour et al., 1995). Multimetric indices are useful tools for assessing and indicating overall ecosystem health for complex systems where the underlying causal processes are poorly understood (Alford \& Gotwald, 2019; Schoolmaster et al., 2012).

Another biomonitoring tool, often contrasted with the multimetric approach (Bowman \& Somers, 2006; Gerritsen, 1995; Reynoldson et al., 1997), and used to make integrative assessments of biological and ecological resources, is the multivariate approach to data analysis (Gerritsen, 1995; Norris, 1995). Initially, MMIs were quite popular among water management
authorities in the United States (Davis et al., 1996), while the multivariate approach to data analysis was commonly advocated for use in the United Kingdom (e.g., the River Invertebrate Prediction and Classification System (RIVPACS)), (Wright et al., 1983; Wright, 1995), Australia (Nichols et al., 2010; Smith et al., 1999), and Canada (e.g., the BEnthic Assessment of SedimenT (BEAST, Resh et al., 2000). To date, both approaches have become well-established and provide assessments of ecosystem health all over the developed and developing world. Similarities and differences between the multimetric and multivariate approaches have been discussed in detail in the literature (Barbour et al., 1999; Gerritsen, 1995; Norris, 1995).

Various types of multimetric indices have been developed, including the Invertebrate Community Index (DeShon, 1995), the Family Biotic Index (Hilsenhoff, 1988), the Fish Index of Biotic Integrity (Karr, 1981), the Benthic Index of Biotic Integrity (Fore et al., 1996; Kerans \& Karr, 1994), Lake Macroinvertebrate Integrity Index (Blocksom et al., 2002), the Vegetation Index of Biotic Integrity (Mack, 2001), the Multimetric Macro-Algal Index (Stevens et al., 2022), the Amphibian Index of Biotic Integrity (Micacchion, 2002), and the River Macrophyte Index (Aguiar et al., 2014), to name a few. Multimetric indices are well established for stream and riverine habitats (Barbour et al., 1999; US Environmental Protection Agency (USEPA), 2002; Mamun \& An, 2018); however, through time, the scope and application of MMIs have expanded to include various ecosystem types such as lakes/reservoirs (Blocksom et al., 2002; Bacigalupi et al., 2021), wetlands (Archer et al., 2010; Rooney \& Bayley, 2012; Cooper, 2018), estuaries (Chainho et al., 2008; Giri et al., 2023), marine and coastal habitats (Andersen et al., 2016; Souza \& Vianna, 2020), and terrestrial habitats (Andreasen et al. 2001), as well as different biological assemblages, including periphyton (MacDougall, 2014), algae (Stevens et al., 2022), microbes (Niu et al., 2018), macrophytes (Aguiar et al., 2014), invertebrates (DeShon, 1995; Barbour et al., 1999), amphibians (Micacchion, 2002; Stapanian et al., 2015), and fishes (Karr, 1981; Souza \& Vianna, 2020). They also can cover various spatial scales, from watersheds (Cantin \& John, 2012), to regional (Baker et al., 2005) and national (Stoddard et al., 2008) level assessments. This integrated assessment approach provides direct measures of the effects of preexisting and present stressors on the biotic integrity of waterbodies while reflecting the overall ecological integrity of aquatic ecosystems (USEPA, 2011). Multimetric indices have many applications: they can be used for environmental and cumulative effects assessments for both point and nonpoint source evaluations; they can help to establish and characterize reference
conditions of the biotic community; they can be used in initial screening contexts or long-term monitoring to characterize the existence and severity, as well as sources and causes of impairment; and, they enable evaluation of the effectiveness of management and restoration actions (Barbour et al., 1999; Karr, 2006).

### 2.1.2 The Index of Biotic Integrity

The Index of Biotic Integrity (IBI), initially created by Karr (1981), is a multimetric approach that provides a framework for assessing fish assemblage data to evaluate environmental quality. The initial IBI included 12 biological metrics that are based on fish community parameters (species richness, composition, tolerance, trophic and habitat measures, and condition) and six biotic integrity (condition) classes (very good, good, fair, poor, very poor, and no fish) (Barbour et al., 1999; Karr, 1981; Karr et al., 1986). An assumption of the IBI is that the collected fish sample represents the entire fish community, and much literature has been devoted to the required stream reach length and sampling procedures that allow a representative sample to be collected (Hughes et al., 2002; Klemm \& Lazorchak, 1995; Lyons, 1992).

Metrics of species richness and composition evaluate fish species richness, fish habitat quality, fish species tolerances, and taxonomic guild structure (Barbour et al., 1999). Trophic composition metrics assess trophic dynamics and the quality of energy sources of the fish assemblage (Barbour et al., 1999). Fish abundance and condition metrics indirectly assess fish population recruitment, mortality, abundance, and condition (Barbour et al., 1999). Multimetric indices, such as the IBI, aim to quantitatively capture the biologist's expert assessment of the fish assemblage's condition through quantitative standards (Barbour et al., 1999). Best professional judgment is required to choose the appropriate fish community parameter to represent each metric and to create scoring criteria (Barbour et al., 1999; Karr, 1981). For the original index, each metric is assigned a score of 1,3 , or 5 , based on the extent of deviation from reference condition values, with 1 indicating significant deviation, 3 indicating moderate deviation, and 5 indicating approximates reference condition values. The metric scores are summed to get a final index score per site ranging from a maximum value of 60 (considered in excellent condition) to a minimum value of 12 (considered in very poor condition) (Barbour et al., 1999; Karr, 1981). This allows comparison between impact and reference sites. It is important to choose appropriate
metrics based on biogeographic and stream size considerations (Barbour et al., 1999; Karr, 1981). Original IBI metrics and a list of alternatives can be found throughout the literature. The use of an IBI for ecological assessment and monitoring purposes is well-established throughout many regions of the world. The IBI was initially established to evaluate the condition of small, warmwater streams of the Midwestern U.S., but the original IBI metrics, scoring system, and condition categories have been modified to create region (e.g., watershed, ecoregion) and ecosystem (e.g., rivers, lakes, impoundments, estuaries) specific indices and to include other assemblage types (e.g., algae, vegetation, benthic macroinvertebrates, amphibians). After assessment of site condition using the fish community, impacted areas should receive further evaluation of individual metrics and assessments to determine the cause of degradation. Careful examination of specific metrics can indicate reasons for impairment (Karr, 1981); however, the indices provide a more robust, integrated assessment of site health when all the metrics (i.e., multiple parameters of the fish community) are taken together (Barbour et al., 1999; Karr, 1981). Multimetric indices, such as the IBI, are essential for making objective assessments of entire ecosystems where the cause-and-effect relationships are complex and poorly understood (Barbour et al., 1999; Schoolmaster et al., 2012).

### 2.1.3 Fish as Bioindicators

Biological indicators (bioindicators) include biota, or biological responses that have varying sensitivities to environmental disturbance, making them valuable indicators of ecosystem health (Everard et al., 2011; Naigaga et al., 2011; Parmar et al., 2016). Within Canada, benthic macroinvertebrates and fish are two commonly used bioindicator taxa used by governing agencies (Environment Canada, 2012a; Environment Canada, 2012b). Much literature exists on the use of different aquatic assemblages as bioindicators; however, for the scope of this research, only the use of fish as bioindicators will be discussed.

Fish are frequently used as bioindicator organisms in bioassay experiments, tissue chemical concentrations, as single sentinel species, and in community-level approaches. Fish are effective indicators of the condition of aquatic systems for a variety of reasons: extensive lifehistory information usually exists; fish communities occupy a range of trophic, reproductive, and habitat guilds as well as tolerance levels to environmental degradation; fish are long-lived species, incorporating cumulative environmental changes at the individual and community level;
acute toxicity (survival) and stress effects (condition, growth, reproduction) can and have been evaluated extensively; and fish are ubiquitous, as well as important, and familiar to the public (Ashcroft et al., 2006; Minns et al., 1994; Simon \& Lyons, 1995). Using fish as indicators directly relates to the objective of Canada's Fisheries Act (1985), to protect fisheries resources. Some disadvantages of using fish as bioindicators include the selectivity of sampling methods, the number of personnel required for sampling, and seasonal variability of fish reproduction, migration, and behaviour (Barbour et al., 1999; Karr, 1981; Pope \& Willis, 1996).

### 2.1.4 Spatial and Temporal Variation in the IBI and Fish Communities

Multimetric indices, such as the IBI, are useful for synthesizing and comparing ecological data across spatial and temporal gradients. If methods are comparable, a major application of the IBI is its ability to evaluate community and ecosystem health over various levels of spatial organization (Pont et al., 2006; Pyron et al., 2008; Stoddard et al., 2008). Subsequently, MMIs can be used to assess ecological health across streams, watersheds, ecoregions, or greater spatial scales (Meador et al., 2008). For example, the sampling of sites along a gradient of disturbance has been used to monitor ecological changes downstream of known point source pollution (Kim et al., 2013), or, more commonly, cumulative effects from diffuse pollution across a landscape (Alford \& Gotwald, 2010; Cantin \& Johns, 2012; Stevens et al., 2010). Similarly, the IBI can be used to assess temporal variation of ecological integrity (Sharif et al., 2021; Pyron et al., 2008; Zhu \& Chang, 2008) across days, weeks, seasons, or even years. The index is designed to integrate spatial-temporal variation from anthropogenic disturbances. Changes in index scores and individual metrics across space and time can directly reflect changes in aquatic ecosystem health.

A significant limitation of MMIs comes from discrepancies in the development of the index and not accounting for natural variation (Pyron et al., 2008). There are several ways natural variation on the landscape can be modeled into the index (Bailey et al., 2004; Baker et al., 2005; Cao et al., 2007). Similarly, variations in index values due to field and laboratory sampling or analytical approaches (e.g., sampling period, reach length, sampling procedure, gear and operator bias, species identification, etc.) can be minimized through the application of proper QAQC techniques (e.g., standard operating procedures, checklists, quality checks, etc.) (Canadian Council
of Ministers of the Environment, 2016; Carter \& Resh, 2013). However, some forms of variation in metric and index scores are much more difficult to control.

Variations in IBI scores due to inherent irregularity in the biological assemblage being assessed (such as seasonal variations in fish life cycles, movement, and reproduction (e.g., Bronmark et al., 2013)), seasonal or inter-annual differences in environmental conditions, or stochastic events (e.g., episodic drought or flood conditions) can be more difficult to control, and when not accounted for, can lead to index scores reflecting alternate factors besides biotic integrity. Seasonality and timing of sampling are of particular importance in northern regions where temperature extremes influence fish reproduction and mobility (Sutela et al., 2017). In northern boreal regions, spawning migrations, migrations to/from winter refuge habitat (spring and fall migrations), and various temperature-related responses can cause changes in the resident fish community over time (Sutela et al. 2017). Habitat shifts in freshwater resident fish species may be most significant in boreal regions where habitat suitability can strongly vary with season (Erkinaro et al., 1998). Pope \& Willis (1996) documented how sampling between seasons can affect not only the number and relative abundance of fish but the size and age structure, as well as the growth and condition of the fish. To reduce temporal bias in the fish assemblage and increase comparability between studies, it is imperative to restrict timing of sampling to a narrow window, especially for multiyear research (Pyron et al., 2008; Reece et al., 2001; Sutela et al., 2017). Sampling at base flow conditions is also recommended since extreme high or low flow conditions can create misleading fish community results (Barbour et al., 1999; Sutela et al., 2017).

Seasonal variability in the IBI can be minimized by avoiding sampling during times of migration or reproduction, sampling across multiple years, or by sampling across seasons and developing season-specific indices (Smokorowski et al., 1998). For instance, it has been suggested that sampling during the summer can lead to higher index scores compared to spring and early summer sampling (Karr et al., 1987). Contrary to this, a study conducted on Lake Ontario's littoral zone found a seasonal pattern where early spring and fall provided the highest index scores compared to sites sampled in summer (Smokorowski et al., 1998). This can likely be attributed to adult fish migrating to shallower water to spawn and highlights the importance of the study region and ecosystem type, as well as applying expert ecological knowledge of the local fish community when determining sampling procedures. Similarly, comparability between
spring/early summer and fall catches may not be adequate as fish biomass and abundance can increase over the growing season. In SK, spawning season for resident native stream fishes is generally early spring (with a few exceptions, see below), and fish movement and spawning for lotic migratory species (e.g., white sucker, longnose sucker, northern pike) also is typically completed by late spring/early summer (Scott \& Crossman, 1973). Sampling in times of spawning and fish migration can lead to bias in the relative abundance and species compositions. To minimize the influence of spawning and reproduction, young of the year (YOY) should be excluded from samples, and time of sampling should be restricted to later in summer/early fall, especially for multiyear studies. Random, episodic events (e.g., flood or drought) are much more difficult to control and add more complexity and uncertainty into bioassessment and monitoring (Kilgour et al., 2013). It is essential that any IBI be evaluated through time to capture, evaluate, and eliminate sources of variability, including those from chance events.

### 2.2 Reference Condition

An integral component of stream biomonitoring is establishing a basis for comparison between sites (Bailey et al., 2004; Stoddard et al., 2006). Natural variability in the landscape, due to differences in climatic and physiographic variation, creates distinct biogeographical realms (Ecological Stratification Working Group, 1996; Gallant et al., 1989; Omernik, 1987). Therefore, it is essential to categorize streams based on relatively homogenous regional frameworks, including climatic or physiographic features (e.g., ecoregions, physiographic zones, climate zones, etc.), that may account for some of the natural habitat and biological variation between sites (Hughes et al., 1992; Jha \& Diplas, 2017; Waite et al., 2000). To capture as much variation as possible, sites can be further classified using another finer-scale variable, such as stream order or other hydrologic features, that will account for a different type of variation and further refine ecological similarity (Omernik \& Griffith, 1991; Phillips et al., 2023; Waite et al., 2000). Developing homogenous regions of ecological similarity minimizes spatial variability of biological, physical, and chemical components within a region while maximizing variability between regions; this serves as a foundation for the creation of regional reference conditions (Bailey et al., 2004; Stoddard et al., 2006; Waite et al., 2000).

Reference condition represents the natural variation in relatively undisturbed or minimally disturbed ecological conditions (Bailey et al., 2004; Hughes et al., 1986; Stoddard et
al., 2006). Establishment of appropriate reference conditions is essential for interpretation of biological surveys. Although reference condition can be established from the use of actual reference sites, historical data, modeling or extrapolation, expert opinion/consensus, or a combination of each (Barbour et al., 1999), it has been argued that systematic monitoring of real sites that represent the natural range of minimally disturbed conditions is most appropriate (Bailey et al., 2004; Barbour et al., 1999). There are two types of reference condition: sitespecific and regional reference (e.g., reference condition approach) (Bailey et al., 2004; Barbour et al., 1999). Site-specific reference conditions generally involve evaluating conditions upstream of a confirmed point source discharge or comparing them with conditions from a similar, but uncontaminated site (e.g., a paired site approach) (Barbour et al., 1999). Following the reference condition approach (Bailey et al., 2004; Phillips et al., 2023), regional reference conditions are developed by measuring conditions at multiple, relatively unimpaired sites within an ecologically similar region.

Choosing an adequate sample reach length is also essential for bioassessment since it directly impacts assessments of fish community health and diversity at both local and regional levels (Angermeier \& Karr, 1986; Barbour et al., 1999; Klemm \& Lazorchak, 1995). Several studies have focused on this issue and both fixed-distance (standard length of stream) and proportional-distance (standard number of stream channel widths) designations have been used (Barbour et al., 1999; Hughes et al., 2002). Proportional-distance approaches allow for variation in reach length depending on stream size; however, sampling programs are encouraged to establish maximum sampling time and/or distance when using this approach (Barbour et al., 1999). Comparable to selecting sites for reference condition, it is also essential that the selected stream reach is representative of the principal physical habitat characteristics of the stream/region (e.g., riffles, pools, runs) (Bailey et al., 2004; Lyons, 1992; Stoddard et al., 2006).

### 2.3 Watershed Stressors in Boreal Plain Environments

### 2.3.1 Agriculture and Effects on Aquatic Ecosystems and Fish Community Assemblages

Agriculture is considered one of the most extensive and environmentally disruptive land use practices (Brauns et al., 2022; Mateo-Sagasta et al., 2017; USEPA, 2007). Changes in farming practices since the end of World War 2, brought on by new technologies and increasing market demand, have led to fewer and larger farms across Canada, with resulting increases in
monoculture cropland and livestock (ECCC, 2016). The increased intensification of agricultural practices has led to an increased awareness of the stresses agricultural production places on natural ecosystems. The prairie provinces (Alberta, SK, and Manitoba) are at a heightened risk of these effects due to intensive agriculture on the landscape (Main et al., 2016; Malaj et al., 2020a; Malaj \& Morrissey, 2022). Reduced-till or no-till farming practices, and increasing summer fallow in the western provinces, have improved soil quality since the 1980s (ECCC, 2016). However, water quality as indicated by ECCC's Agri-Environmental Performance Index has shown a decline over the past five decades. This indicator looks at common agricultural stressors on aquatic ecosystems, including nitrogen, phosphorus, bacteria, and pesticides (ECCC, 2006). Recent declines in livestock populations since the early 2000s have reduced perennial crops and increased annual crop areas, bringing with it an increased use of pesticides, herbicides, and phosphorus fertilizer, and a subsequent higher risk of aquatic ecosystem contamination (Gliessman, 2014; Malaj et al., 2020; Malaj \& Morrissey, 2022). Furthermore, increases in fertilizer and manure application on farms have led to increased inputs of nitrogen, phosphorus, and agricultural bacteria (e.g., coliforms) into ground and surface water (Gliessman 2014; US Geological Survey, 2010).

### 2.3.1.1 Nutrient Run-off and Eutrophication

A well-known effect of large-scale agricultural practices common today is the increased nutrient loads into ground and surface waters. The overapplication of synthetic fertilizers is the most extensive contribution to nonpoint source water pollution in North America (Birk et al., 2020; Gliessman, 2014; US Geological Survey, 2010). Inorganic nutrients, such as nitrogen (N) and phosphorus $(\mathrm{P})$, are two significant and limiting nutrients that occur in aquatic ecosystems in various forms as ions or dissolved in solution and are essential for aquatic plant and algae growth (Carpenter et al., 1998; Miller \& Spoolman, 2012; US Geological Survey, 2010). Aquatic plants transform dissolved inorganic forms of nitrogen (e.g., nitrate, nitrite, and ammonium) and phosphorus (e.g., orthophosphate) into organic, bioavailable forms that can be used in higher trophic production (Chislock et al., 2013). Nitrogen and phosphorus, from agricultural synthetic N-P-K fertilizers and livestock manure, can leach from agricultural fields and farms into surrounding water sources. The resulting excess nutrient supply to waterbodies overstimulates aquatic plant and algae growth, which, following death and decomposition, can use up all the
available oxygen and reduce dissolved oxygen levels to well-below-normal concentrations (Carpenter et al., 1998; Miller \& Spoolman, 2012; US Geological Survey, 2010). Eutrophication can happen naturally in some systems; however, high nutrient enrichment from human activities, including agriculture, causes a semi-natural process to intensify. Cultural eutrophication can lead to various effects on aquatic ecosystems, including an increase in plant and animal biomass, increased turbidity and sedimentation, reduced species diversity and shifts in the dominant biota (Bacigalupi et al., 2021), and the potential development of anoxic conditions (Chislock et al., 2013). Water contamination can also occur as a result of livestock manure/sewage application onto fields or direct release (intentional or not) into waterways (Gurian-Sherman, 2008; Mallin et al., 1997; Sakadevan \& Nguyen, 2017). This can lead to the introduction of pathogens and further organic enrichment/depleted oxygen in aquatic ecosystems (Keena et al., 2022; USEPA, 2007).

### 2.3.1.2 Pesticides and Aquatic Ecosystems

Conventional agriculture relies heavily on synthetic pesticides (e.g., herbicides, insecticides, rodenticides, and fungicides) for production yields, which, like the use of fertilizers, can lead to overapplication and contaminated waterbodies (ECCC, 2016; Malaj et al., 2020b; Malaj \& Morrissey, 2022). Pesticides most often end up in aquatic ecosystems via runoff and leaching from agricultural fields (Ali et al., 2020; Stanley \& Preetha, 2016), but other sources, such as atmospheric deposition (Messing et al., 2011) have been identified as well. Fish spend their entire life cycle in water, and as such, they are continuously exposed to aquatic contaminants, including pesticides, via direct contact, respiration through the gills, and consumption of contaminated prey (Stanley \& Preetha, 2016).

Direct and indirect effects of pesticides on fish and fish communities have been studied. The majority of research on pesticide exposure to fish involves acute toxicity assessments in the form of in vivo or in vitro bioassays (e.g., static, static-renewal, flow-through systems, and injection, and dietary exposures) (Stanley \& Preetha, 2016). Various lethal and sublethal effects of pesticides on fish have been verified, including altered behaviour, physiology, biochemistry, disruption of endocrine, cardiovascular, nervous, digestive, and reproductive systems, and genotoxicity (Ali et al., 2020; Stanley \& Preetha, 2016, Gibbons et al., 2015). Far fewer studies have researched the effects of pesticides on fish chronically, especially through semi-
controlled/mesocosm, and field toxicity assessments (Murthy et al., 2013). This is especially important since pesticide exposure in aquatic ecosystems occurs in the presence of many other aquatic stressors (Chará-Serna et al., 2019). Pesticide contamination in aquatic ecosystems can also lead to shifts in fish prey, including benthic macroinvertebrate community diversity and abundance (Cavallaro et al., 2019) which can indirectly impact fish growth and survival (Ewing, 1999). Reductions in plankton and aquatic plant abundances from pesticides (Sweilum, 2006) can also reduce fish habitat and increase predation risk. Traditional, fat-soluble, organic pesticides led to the bioaccumulation and biomagnification of contaminants in fish tissue, especially higher trophic-level piscivores, and may still remain in the environment to date (Amoatey \& Baawain, 2019).

Albeit less of a concern than the other agricultural-related aquatic stressors discussed previously, metal contamination of trace elements, such as arsenic, cadmium, copper, lead, selenium, and zinc, from the use of pesticides, fertilizers, manures, and municipal biosolids, can be leached and volatilized, and end up in aquatic ecosystems (ECCC, 2016; Gupta et al., 2019; Shentu et al., 2015). Maintaining proper vegetative buffer zones and wetland structure on the landscape can help to mitigate the effects of nutrients, pesticides, and metals on aquatic ecosystems (Cavallaro et al., 2019; Main et al., 2017; Ruso et al., 2019).

### 2.3.1.3 Habitat Loss and Alteration

Fish habitat can be destroyed or altered through direct landscape modification during agricultural activities. With the increasing food demands of our society and subsequent agricultural technology advancements, farming has trended towards intensification of cropland and livestock, which requires large-scale input costs, machinery, and production (ECCC, 2016). These advancements in agriculture may come at a cost to the environment, however, as additional acres of cropland and livestock are required to meet the current food demands and the cost of sustaining conventional agriculture (ECCC, 2016; Gliessman, 2014). Removal, conversion, and degradation of grassland, woodland, and low-lying wetland ecosystems to increase crop yields have become common practice in conventional agriculture (ECCC, 2016; Gliessman, 2014; Malaj \& Morrissey, 2022). Removal of vegetation buffers, ditching and draining of wetlands, and diverting streams (Main et al., 2017; Ruso et al., 2019), can modify and reduce aquatic habitat (Brauns et al., 2022; Burdon et al., 2013; Newcombe \& Jensen, 1996).

It can lead to increased erosion and sediment deposition as discussed above, increased nutrient and chemical run-off due to reduced vegetative buffer zones, but also loss of allochthonous inputs from removed riparian zones, and stream thermal increases, among other effects. Additionally, the prairie provinces, including SK, have an increased risk of salinization of soils and water bodies as a result of climate and agricultural irrigation (ECCC, 2016).

Agricultural practices, including tilling and removal of riparian vegetation and wetlands, can lead to indirect habitat loss from increased silt, sand, and mineral erosion from surface runoff and deposition into nearby water sources (Gliessman, 2014). High suspended solids and turbidity can minimize sunlight penetration into the water, reducing the photosynthesis of bottomdwelling plants and stream algae (Carpenter et al., 1998; US Geological Survey, 2010). Furthermore, increased erosion from riparian vegetation and wetland reduction/removal can increase erosive processes and cause further turbidity and stream siltation/sedimentation, subsequently impacting fish habitat and populations by suffocating aquatic plants and invertebrates, as well as fish spawning habitat and eggs, and/or eventually altering aquatic community diversity and reproduction (Newcombe \& Jensen, 1996). Effects of suspended sediment and stream deposition can range from behavioural and sublethal effects (e.g., reduced growth rate, reduced fish density, reduced population size), to lethal effects (Newcombe \& Jensen, 1996). The decrease in natural vegetation buffers and increase in soil disruption has led to soil loss and siltation of waterways through wind and water erosion. Deposited organic solids will experience extensive decomposition and the formation of anaerobic environments (Mallin et al., 1997). Fine-grained suspended solids, such as silt, can even injure fish gills or cause asphyxiation (Newcombe \& Jensen, 1996).

### 2.3.2 Forestry Operations and Effects on Aquatic Ecosystems and Fish Community

## Assemblages

Over half of SK's land area is forested, covering approximately 34 million hectares (Government of SK, 2023). Saskatchewan's commercial forest makes up 11.7 million hectares ( 5.3 million hectares are classified as productive forest land and used for commercial harvesting), an area entirely within the Boreal Plain Ecozone (Government of SK, 2023). Forestry is SK's second largest industry (after mining), generating over $\$ 1$ billion in forest product sales annually, and providing many socio-economic benefits to the province, including
eight large forest products manufacturing facilities (seven are in operation), greater than 210 small forest businesses, and over 230 supply chain forestry-related businesses (Government of SK, 2019). The Beaver River watershed alone contains four of the seven operating timber manufacturing facilities, including two sawmills, one oriented strand board (OSB) mill, and one pulp and paper mill. The SK Ministry of Environment regulates all forest use on crown lands, providing licenses and permits to forest companies that must enact various rights and responsibilities for long-term sustainable forest management (Government of SK, 2023). Forestry operations can have various potential effects on aquatic ecosystems and communities, depending on the type of forest management practices in place.

### 2.3.2.1 Effects on Stream Hydrology

Forest harvesting can have impacts on stream hydrology by affecting the quantity and timing of stream flows (Canadian Parks \& Wilderness Society (CPAWS), 2020; Regional Aquatics Monitoring Program (RAMP), 1997). Forests are an important part of the hydrologic cycle, playing a significant role in the movement of water. Trees are constantly exchanging water with the surrounding environment via absorption through soil and transpiration through leaves (Aron et al., 2019). Forests also provide shade which aids in slowing the snowmelt and resulting runoff (Pomeroy et al., 2012). Tree harvesting can lead to less uptake of precipitation and faster melting snowpack, causing increases in runoff and higher peak flows which occur over a shorter period (Pomeroy et al., 2012, RAMP, 1997). Soil can also become compacted by logging equipment and lead to increased runoff (Cambi et al., 2015). Changes in the timing and quantity of stream flows can have a direct impact on the fish community (Bunn \& Athington, 2002; Poff et al., 1997; Rytwinski et al., 2017). Forestry's effects on stream hydrology depend on several factors, including harvesting practices, topography, vegetation, soil structure, and climate (RAMP, 1997). For example, impacts to stream hydrology can be minimized through appropriate forest management, such as understanding the climate and soil type of an area, restricting timing of operations (e.g., logging only in winter months when the soil is frozen), avoiding clear-cutting, and prompt site reforestation following disturbance (RAMP, 1997).

### 2.3.2.2 Effects on Water Quality

Forest harvesting can lead to increased levels of sediment and turbidity in nearby aquatic ecosystems and negatively impact aquatic habitat (Government of British Columbia, 2023a; Hauer et al., 2018). Operating logging equipment nearby or within waterbodies can cause increased sediment loads (Cambi et al., 2015; Government of British Columbia, 2023b). Additionally, the removal of riparian vegetation can cause stability issues in stream banks and increased erosion and sediment loads (Natural Resources Canada, 2012). Vegetation removal further from the stream also causes higher soil erosion by wind and water as the trees are no longer able to intercept precipitation, stabilize the forest floor, and slow surface runoff (Government of British Columbia, 2023a). There are various known negative effects of higher sediment loads on receiving aquatic ecosystems and fish habitat (Department of Fisheries \& Oceans (DFO), 2000; Hauer et al., 2018). Forest harvesting can also affect stream (and other waterbodies) chemistry levels, including nutrients, temperature, and dissolved oxygen levels (Martin et al., 2000; Steedman \& Kushneriuk, 2000; Steedman et al., 2001).

### 2.3.2.3 Effects on In-Stream Habitat

Forest harvesting can affect in-stream habitat in several ways. In-stream habitat can be directly altered or disturbed through equipment operation near or within the waterway (RAMP, 1997). Increases in peak flows and sediment loads can cause disruption to fish habitat (e.g., smothering of aquatic plants or gravel beds needed for forage/cover and spawning) (Government of British Columbia, 2023a). Furthermore, the removal of riparian vegetation can alter allochthonous inputs (e.g., changing nutrient inputs and woody debris for habitat cover) to the stream (Bambi et al., 2023; Studinski \& Hartman, 2015), reduce shading, increase water temperature (Macdonald et al., 2003), and increase primary and secondary productivity (Fuchs et al., 2003; Planas et al., 2000).

Much of the negative impacts described above have been reduced in recent years from the introduction of government regulations and application of appropriate forestry management techniques (Government of SK, 1996, 2023). For example, in SK, the forest industry is required by law to ensure all timber harvest areas are reforested within a few years of logging (either through planting seedlings, equipment disturbance to encourage regrowth, or natural regeneration; Government of SK, 2023). Although SK does not have specific targets set,
provincial and federal regulations require potential effects on riparian and other sensitive areas (e.g., significant fish spawning habitat), as well as soil compaction, to be minimized (Government of SK, 1996).

### 2.3.3 Fish Assemblage and Metric Response to Environmental Stressors

This section outlines how detailed knowledge of local fish assemblage traits is essential for IBI development and interpretation. Environmental degradation spurred by land use, habitat alteration, and declining water quality can cause measurable shifts in the fish assemblage (Karr, 1981; Alford \& Gotwald, 2019; Bacigalupi et al., 2021). These alterations can occur for various reasons, including direct habitat destruction (e.g., loss of spawning, nursery, or adult foraging habitat) or indirect trophic interactions (e.g., declines and alterations in benthic macroinvertebrate prey from agricultural runoff) (Cavallaro et al., 2019; Couture \& Biron, 2023). Fish assemblage metrics are one useful way to monitor these changes through time and space (Alford \& Gotwald, 2019; Griffith \& Mcmanus, 2020).

Species richness and the number of individuals can change with increasing environmental degradation. More specifically, environmental degradation of fish habitat can lead to a decrease in the relative abundance or percent composition of intolerant species, and a subsequent increase in tolerant fish capable of withstanding stressors (Bramblett et al., 2005; McCormick et al., 2001; Miller et al., 1988). Productivity can be reduced in degraded ecosystems which can alter trophic structure. Degraded environments often lead to shifts in the availability of food resources and the trophic structure of the fish and benthic invertebrate communities can provide insight into the production and consumption dynamics in an ecosystem (Barbour et al., 1999, Karr, 1981). Omnivorous fish will generally increase following disturbance due to their opportunistic foraging ecology (Karr et al., 1986; Mamun \& An, 2020; McCormick et al., 2001). Dominance of omnivores often arises from degradation in the food base (e.g., invertebrates). The proportion of the fish community that are specialized foragers (e.g., insectivorous cyprinids, benthic insectivores) and top carnivores will reduce with declines in stream quality (Cantin \& John, 2012; Mamun \& An, 2020). Although changes in the abundance of specific species (percent compositions) will vary depending on the species and disturbance in question, fish total abundance may show an increase in response to environmental perturbations due to the opportunistic nature, shorter life cycle, and broader tolerance of certain fish (Haine et al., 2012;

Vile \& Henning, 2018). Any species of stream fish that has specific or unique habitat requirements can often be favourable as an indicator species (Lee et al., 2018). For example, lithophilic or phytophilic spawners that require coarse substrate or macrophytes for spawning will be disproportionally impacted by increased watershed erosion and stream sediment deposition (Hughes et al., 1998; Krause et al., 2013; McCormick et al., 2001). Similarly, the frequency of abnormalities (deformities, eroded fins, lesions, and tumors, (DELTs)) and parasites in the community can be higher in areas experiencing point source and/or cumulative stressors (Cantin \& John, 2012; Sanders et al., 1999). Additionally, changes in condition-related indicators, such as Fulton's body condition factor (Ricker, 1975), can diminish in response to declining stream health and pollution. Changes in community composition, species abundance, and morphology can indicate environmental conditions (e.g., physical, chemical, biological) that are outside a species' preferred limits. Poor water quality conditions can often be identified by the abundance of highly tolerant families and species (Alford \& Gotwald, 2019; Lee et al., 2018). It is expected that IBI scores reflecting the entire fish assemblage will decrease with increasing environmental degradation (Vile \& Henning, 2018).

### 2.4 Saskatchewan's Fishery

Saskatchewan is home to an abundance of lakes, rivers, and streams that contain fish, the majority of which occur in the northern part of the province (Ashcroft et al., 2006). Saskatchewan's northern region is dominated by less productive ecosystems, characterized by low water temperatures, short open-water growing seasons (due to climatic conditions), and low dissolved nutrients (due to geological conditions) (Ashcroft et al., 2006). Aquatic ecosystem productivity is influenced by a variety of factors, and the primary influences affecting fish assemblages include nutrient availability, water temperatures, growing season length, and the available habitat (Ashcroft et al., 2006).

Saskatchewan has a relatively low species richness compared to regions in lower latitudes and milder climates. Sixty-nine species of fish occur within the province, 58 of which are native, and 11 are exotic (Ashcroft et al., 2006). The 69 species in the province belong to 15 different families and over half of the species are small fish that belong to the minnow (Cyprinidae) family (Ashcroft et al., 2006). The commercial and recreational fisheries focus most of their effort on only five species: northern pike (Esox Lucius), walleye (Sander vitreus), yellow perch
(Perca flavescens), lake trout (Salvelinus namaycush), and lake whitefish (Coregonus clupeaformis) (Ashcroft et al., 2006). Saskatchewan fish species, their distribution throughout the province, water class type occupied, thermal requirements, and their trophic, forage, reproduction, and tolerance guilds are summarized in Prestie (2014, unpublished).

Both the federal and provincial governments share management of the SK fishery resource. The Federal Fisheries Act (1985) was the major governing legislation concerning the management of all inland fisheries in Canada; however, SK established its' own comprehensive Act (the SK Fisheries Act of 1994) and Regulations (the SK Fisheries Regulations of 1995) dealing with fisheries management, the first Canadian province to do so. Currently, the SK government is responsible for fisheries management and fish marketing within the province, and the federal government has authority concerning the protection of fish (e.g., Species at Risk Act, 2002) and fish habitats, as well as fish sale/trade outside the province (e.g., Freshwater Fish Marketing Act) (Ashcroft et al., 2006).

### 2.5 Significance of Research

Saskatchewan has an abundance of freshwater ecosystems and proper management techniques are needed to conserve fisheries and other aquatic resources in the province. Saskatchewan's fishery is of ecological, economic, cultural, and recreational importance (Ashcroft et al., 2006); however, incorporation of fish into monitoring programs has been limited (Davies \& Hanley, 2010; Knackstedt, 2015; Phillips et al., 2023), despite being useful indicators of aquatic health (Cantin \& John, 2012; Krause et al., 2013; Minnesota Pollution Control Agency (MPCA), 2014). In SK, Environment Canada's Environmental Effects Monitoring (EEM) is currently the only program used to monitor fish health, but its application is restricted (Environment Canada, 2010, 2012a) and it may overlook common cumulative effects.
Developing and evaluating approaches to assess broad-scale non-point source or localized point source impacts and cumulative effects from multiple stressors (e.g., agricultural-related effects) will contribute to the sustainability of the province's aquatic ecosystems.

### 2.6 Research Objectives and Hypotheses

The overall objective of this research is to adapt and critically evaluate a fish-based IBI for the Beaver River watershed located in the Boreal Plain ecozone in SK. This will determine if
fish and fish communities in wadable streams and rivers in SK are responsive to environmental stressors. The responsiveness and sensitivity of SK fish to common environmental stressors will determine their capability for use as indicators of aquatic ecosystem health.

There are three primary objectives of this research:

1) Determine expected fish community structure and fish condition in minimally disturbed streams in northern SK based on natural physical (stream size, substrate, temperature) and chemical (nutrients, metals, dissolved oxygen) gradients.

For my first objective, I hypothesize that fish community structure and fish condition will have limited variation among minimally disturbed streams and rivers. I hypothesize that fish condition will be of the highest quality and will remain relatively consistent across natural environmental gradients in minimally disturbed conditions. Additionally, I expect to see fish species richness and abundance increase from headwater to higher-order streams.
2) Determine if fish community structure and fish condition vary with a gradient of human disturbance (agriculture, forestry, municipal waste, etc.) by applying and evaluating a fish-based IBI. This will ultimately determine if fish communities in northern SK can be used as indicators of stream health.

For my second objective, I hypothesize that fish communities will be responsive to a gradient of disturbance within SK, with the IBI revealing lower scores in areas of impairment. Additionally, I hypothesize that impacted sites will have lower species richness and abundance, a higher percentage of tolerant species, and fish with a higher frequency of abnormalities.
3) Determine the sensitivity of the IBI (including fish communities, water quality, and habitat variables) to inter-annual variability.

For the third objective, I hypothesize that fish communities, fish condition, water quality, and habitat variables will show some interannual variance within sites, due to differences in environmental conditions and fish residency and mobility between years; however, these
differences will not be reflected in IBI scores, with greater variance among sites than within sites.

## Chapter 3: MATERIALS \& METHODS

### 3.1 Study Area

Sampling for this study was carried out in the Beaver River watershed located in SK, Canada. The Beaver River Watershed is in west-central SK, along the SK-Alberta border, and is within the Boreal Plain ecozone (Figure 3.1). This ecozone comprises rolling plain founded on sedimentary rock, thick glacial deposits, scattered lakes and glacial kettles, and boreal forest and aspen parkland consisting of mixed hardwood and coniferous species (Acton et al., 1998) and includes the Mid-Boreal Upland, Mid-Boreal Lowland, and Boreal Transition ecoregions. This region is strongly influenced by continental climatic conditions with mean annual temperature between $-2^{\circ} \mathrm{C}$ to $2^{\circ} \mathrm{C}$, mean summer temperatures of $13^{\circ} \mathrm{C}$ to $15.5^{\circ} \mathrm{C}$, and mean winter temperatures between $-17.5^{\circ} \mathrm{C}$ to $-11^{\circ} \mathrm{C}$; the mean annual precipitation varies between 300 mm to 625 mm throughout the Boreal Plain (Ecological Stratification Working Group, 1996). The thermal region of this area is considered cold water and/or cool water. The Beaver River is the major drainage basin of the area (total watershed drainage area: $50,005 \mathrm{~km}^{2}$ ), originating at Beaver Lake, Alberta, and flowing east across the SK-Alberta border before eventually draining northward into the Churchill River system (Figure 3.1) which is part of the Hudson Bay drainage of the Atlantic Ocean. The SK portion of the Beaver River watershed comprises $33,104 \mathrm{~km}^{2}$.

The major stressors in the Beaver River watershed are agriculture (e.g., livestock, manure production, rangeland, cropland, fertilizer, and pesticide inputs) and forestry operations (Davies \& Hanley, 2010), mixed with some oil and gas exploration. The watershed is dominated by agricultural activity in the south and a relatively unimpacted forest landscape in the north (Figure 3.1). Forestry operations occur in a patchy distribution throughout the watershed and may have more localized effects (Figure 3.1). This gradient of human disturbance across the landscape provides a unique opportunity to assess potential effects on fish and aquatic ecosystems and evaluate the IBI in a relatively homogenous area with multiple land-use stressors (Figure 3.1).


Figure 3. 1| Land use map showing disturbed and undisturbed land cover for the Beaver River watershed relative to upstream watershed areas for sites. The spatial and temporal distribution of study sites across the watershed, the major drainage basin (the Beaver River), and other major waterbodies of the region are also shown. Forest harvest areas are given for the previous twenty years prior to sampling (1998-2018).

### 3.2 Fish Assemblage

To date, no known extensive studies have been conducted on the Beaver River watershed fish assemblage. However, 40 species of fish across 10 families, are known to occur within the larger Churchill River drainage basin which contains the Beaver River watershed (Appendix A). This includes two imperiled species (lake sturgeon (Acipenser fulvescens) and shortjaw cisco (Coregonus zenithicus)), eight species of commercial and recreational importance (northern pike (Esox lucius), walleye (Sander vitreus), sauger (Sander canadense), yellow perch (Perca flavescens), burbot (Lota lota), arctic grayling (Thymallus arcticus), lake trout (Salvelinus namaycush), and lake whitefish (Coregonus clupeaformis), and four introduced sport fish species (brook trout (Salvelinus fontinalis), rainbow trout (Oncorhynchus mykiss), splake (Salvelinus
fontinalis x Salvelinus namaycush), and tiger trout (Salmo trutta x Salvelinus fontinalis) (Ashcroft et al., 2006). Seventeen species of fish (six families) are reported from the Alberta portion of the Beaver River watershed (Nelson \& Paetz, 1992).

### 3.3 Study Site Selection

More than 100 potential sites were initially identified using 1:250,000 scale topographic maps, Google Earth satellite imagery, and local expert knowledge ( 150 sites including 57 named and 70 unnamed streams and tributaries). Sites were then classified by watershed (Beaver River watershed and respective drainage basins) and stream order (wadable streams, order 1-4, following Strahler, 1957), and potential reference and impact sites were identified a priori based on surrounding land use (intact forest, agriculture, and/or forestry operations and disturbance). Stream order was calculated using 1:50,000 scale National Topographic Survey of Canada series maps and was verified via ArcMap (version 10.6.1) and the National Hydrological Network of Canada 1:50,000 topographic map layer. Sampling was restricted to perennial streams. Since maps often do not give accurate representations of stream networks, exploratory sampling was required to further verify and refine previously identified sample sites (Table 3.1). Prior to field surveys, Rural Municipality maps were consulted to obtain landowner permission. Of the 150 sites initially identified, only 32 sites met the necessary criteria (see Table 3.1) and were retained for further sampling consideration. Of the potential sample sites, 28 sites were sampled (18 independent streams, five of which were resampled twice each) due to high rainfall events and corresponding flood water throughout the watershed during the summer months of 2017 and 2018. As best as practical, sampling was avoided during early spring runoff from snow melt and extreme high-flow episodic events (e.g., high rainfall), which can create very different biological and chemical conditions compared to baseflow (Klemm \& Lazorchak, 1995). The 18 sample sites were chosen systematically throughout the watershed along a gradient of agricultural and forestry disturbance (Figure 3.1, Table 3.2). All sites had a similar degree of connectivity (and therefore colonization potential) and lack of human (e.g., dams, weirs, etc.) and natural (e.g., waterfalls, beaver dams, etc.) barriers to fish movement. Sites were sampled throughout JulyAugust 2016, June-October 2017, and August 2018. To assess the IBI's ability to handle potential annual variation in fish communities, fish conditions, sampling conditions, and site
habitat characteristics, five of the streams sampled in the 2016 field season were reassessed in 2017 and 2018 (Figure 3.1; Table 3.2).
Table 3. 1| Initial site selection criteria used to refine potential sample sites within the Boreal Plain ecoregion.

## Initial Site Selection Criteria

| Watershed | Beaver River and sub-catchments |
| :---: | :---: |
| Ecoregion | Mid-Boreal Upland, Boreal Transition |
| Stream Subsystem | perennial |
| Stream Order | $1-4$ |
| Accessibility | road access or <1km hike |
| Surrounding Land Use | intact forest, forestry operations, agriculture <br> Stream Attributes Considered, deep pools, velocity/swift current, <br> unstable substrate, metal objects, beaver <br> impoundments, log jams, etc. |
|  |  |

### 3.4 Stream Reach Designations

An appropriate stream reach length must be used to obtain adequate estimates of fish species richness in streams for ecological and fish community-level assessments (Barbour et al., 1999; Klemm \& Lazorchak, 1995; Lyons, 1992). Forty times the average wetted stream width is thought to be an adequate reach length (Barbour et al., 1999; Lyons, 1992) to capture greater than $90 \%$ of the species in the stream reach (Klemm \& Lazorchak, 1995) and was selected for this research. Maximum and minimum reach lengths of 300 m and 100 m were chosen to ensure the representative stream fish assemblage is still captured (Angermeier \& Karr, 1986; Barbour et al., 1999; Karr, 1981), while respecting time constraints. The average wetted width was estimated through visual observation and quantifying stream width at three transects along $\sim 300 \mathrm{~m}$ of the stream. To reduce influence on habitat quality and fish communities, all sample sites were located a minimum distance of 100 m from the nearest tributary or road/bridge crossing (Barbour et al., 1999).

Table 3. 2| Site Characteristics for the 18 streams and rivers sampled in the Beaver River Watershed. See Appendix B for further details on site characteristics.

|  | Stream Name | Site Code | Sample Date | Revisited Site? | UTM Zone | Easting | Northing |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Alcott Ck | ALCT-01 | 2016-07-30 | Yes | 12U | 673017 | 5960036 |
|  | Alcott Ck | ACLT-02 | 2017-08-25 | Yes | 12 U | 673017 | 5960036 |
|  | Alcott Ck | ACLT-03 | 2018-08-08 | Yes | 12 U | 673017 | 5960036 |
|  | Nolin Ck | NOLN-01 | 2016-08-17 | Yes | 12 U | 668201 | 5995148 |
|  | Nolin Ck | NOLN-02 | 2017-08-26 | Yes | 12 U | 668201 | 5995148 |
|  | Nolin Ck | NOLN-03 | 2018-08-10 | Yes | 12 U | 668201 | 5995148 |
|  | Sukaw Ck | UNBH-01 | 2016-08-20 | Yes | 12 U | 608644 | 6037966 |
|  | Sukaw Ck | UNBH-02 | 2017-09-10 | Yes | 12 U | 608644 | 6037966 |
|  | Sukaw Ck | UNBH-03 | 2018-08-12 | Yes | 12 U | 608644 | 6037966 |
|  | Flotten R | FLTN-01 | 2016-08-23 | Yes | 12 U | 659840 | 6051158 |
|  | Flotten R | FLTN-02 | 2017-08-30 | Yes | 12 U | 659840 | 6051158 |
|  | Flotten R | FLTN-03 | 2018-08-10 | Yes | 12 U | 659840 | 6051158 |
|  | DeLaRonde Ck | DLRD-01 | 2016-08-25 | Yes | 12 U | 651198 | 6052904 |
|  | DeLaRonde Ck | DLRD-02 | 2017-08-28 | Yes | 12 U | 651198 | 6052904 |
|  | DeLaRonde Ck | DLRD-03 | 2018-08-09 | Yes | 12 U | 651198 | 6052904 |
|  | Unknown Ck Goodsoil | UNGS-01 | 2017-06-30 | No | 12 U | 624573 | 6026986 |
|  | Unknown Ck Spiritwood | UNSW-01 | 2017-07-10 | No | 13 U | 331828 | 5915824 |
|  | Unknown Ck (Backwater Ck) | BKWT-01 | 2017-07-22 | No | 12 U | 661760 | 5998449 |
|  | Otter Ck | OTTR-01 | 2017-07-26 | No | 13 U | 350486 | 5970623 |
|  | Dennis Ck | DNNS-01 | 2017-07-27 | No | 12 U | 640110 | 6049245 |
|  | Unknown Ck Pagan Lake | UNPG-01 | 2017-07-30 | No | 12 U | 675350 | 6021786 |
|  | Nesslin Ck | NESS-01 | 2017-08-12 | No | 13 U | 379777 | 5972709 |
|  | Landry Ck | LNDY-01 | 2017-10-14 | No | 12 U | 649518 | 6052403 |
|  | Mistohay Ck | MIST-01 | 2017-10-15 | No | 12 U | 618360 | 6034672 |
|  | Unknown Ck Makwa Tributary | UNMT-01 | 2017-10-16 | No | 12 U | 638445 | 6002535 |
| $\pm$ | Tea Ck | TEA-01 | 2017-10-17 | No | 13 U | 331293 | 5977289 |


| Robinson Ck | ROBN-01 | $2017-10-17$ | No | 13 U | 315604 | 5952313 |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Sulby Ck | SLBY-01 | $2017-10-19$ | No | 13 U | 308134 | 5981072 |

### 3.5 Fish Community Surveys (Biological Data)

Fish were collected from designated streams using a Smith-Root LR-24 backpack electrofisher along the estimated reach length. Electrofishing has proven to be the most effective and comprehensive collection method to obtain a representative sample of the fish assemblage in wadable streams (Barbour et al., 1999). The electrofishing crew consisted of a minimum of 2-3 personnel trained with the sampling techniques prior to surveys. Surveys began at the downstream portion of the reach, moving in bank to bank transects in an upstream fashion to ensure all habitat types were sampled (Figure 3.2). To reduce sampling bias, the same individual operated the electrofisher, while the others helped net and collect the fish. Captured fish were placed in 50L Rubbermaid ${ }^{\mathrm{TM}}$ tubs situated at approximately three locations along the reach to allow fish to be released near the point of capture. Fish holding tubs were aerated and placed in the shade, away from the water's edge to reduce stress to fish. Additionally, the following precautions were taken to minimize fish injury and stress: a single pass electrofishing survey; use of the lowest effective power setting; in all cases, fish response during capture and handling was assessed prior to increasing and in determining the appropriate output settings; avoiding turning the power on and off when fish are near the anode; and, minimizing fish exposure time through efficient hand netting. Fish were kept in holding tanks until the entire reach was electrofished, thus ensuring the same fish were not recaptured.

Data on fish community composition was collected at each site. Once electrofishing ceased, all fish were processed for total length $(0.1 \mathrm{~cm})$, weight $(0.001 \mathrm{~g})$, external abnormalities (deformities, eroded fins, lesions, tumors, parasites, fungus, emaciation, and other anomalies) and identified to the species level prior to being released. Any fish that were not identified in the field were brought back to the lab for identification under a microscope. In cases where greater than 25 fish of the same species were collected, a subsample $(\mathrm{n}=25)$ of that species was processed for length and weight measurements (Barbour et al., 1999). Subsamples included fish from a range of lengths. Young of the year fish (less than 20 mm total length) were not included in the study and were released on site (Barbour et al., 1999). Electrofishing seconds were recorded to calculate catch per unit effort as the number of fish/100s. A standardized fish collection sheet was completed at each site (Appendix C).


Figure 3. 2| Image showing reach designation and electrofishing survey design for collection of environment and biological parameters.

### 3.6 Site Assessments (Environmental Data)

Site assessments included physical characterization of the stream as well as an evaluation of stream water quality and habitat. Various physicochemical and habitat observations and measurements were taken at the stream reach or watershed scale (depending on the characteristic being assessed) to evaluate fish habitat quality, calculate additional stream hydrologic parameters, and get a final habitat assessment score at each site. The definition of "habitat" for this research refers to the quality of the instream and riparian features that affects the structure and function of the fish community. Habitat evaluations are an integral component of assessments of ecological integrity and should be completed alongside biological sampling (Barbour et al., 1999). Chosen parameters, along with the standard operating procedures to collect each parameter for the site assessment, are based on methods outlined in Barbour et al. (1999), Carter (2012), MoE \& SWA (2012), and Cantin \& John (2012). The field site assessment, including stream site physical characterization, water quality and habitat assessment,
follows the standardized site assessment field sheet template in Appendix D and were completed at each site.

### 3.6.1 Stream Physical Characterization

The physical characterization portion of the site assessment (Appendix D) included a general description of the site, recording previous and past weather conditions and watershed features (e.g., land use, pollution, erosion) and measurement of various instream features. Percent composition of stream habitat types (e.g., riffle, run, pool) were characterized through visual assessment along the entire reach. Instream measurements were taken at three transects along the stream reach length (starting point, middle and end of the reach) and the average value was used in the final assessment. Stream wetted width, the high-water mark, bankfull width, and bankfullwetted depth were all measured. Depth measurements were taken at three or six points along each transect depending on stream wetted width (for smaller streams $\leq 5 \mathrm{~m}$ : three measurements; for larger streams > 5m: six measurements). Velocity was recorded near the downstream transect using either the velocity head rod (ruler) method (following the methods in Carter, 2012), a flow-velocity meter, or a semi-buoyant object and measuring tape (following methods described in MoE \& SWA, 2012). Slope was measured using a hand level, measuring tape (covering $\sim 20 \mathrm{~m}$ ), and a survey measuring rod at a representative point along the stream reach. Stream substrate and substrate embeddedness were characterized by visually estimating percent composition of each substrate type (bedrock, boulder (> 25.6 cm ), cobble ( $6.4-25.6 \mathrm{~cm}$ ), gravel ( $0.2-6.4 \mathrm{~cm}$ ), sand/silt/clay ( $<0.2 \mathrm{~cm}$ ) and organic material) and amount of substrate embeddedness along the entire reach. Presence of any sediment odors, oils and deposits were noted. Photographs, taken at each stream transect, were used to aid in characterizing habitat features.

### 3.6.2 Stream Water Quality

Various water quality parameters (temperature $\left({ }^{\circ} \mathrm{C}\right)$, conductivity ( $\mu \mathrm{s} / \mathrm{cm}$ ), chlorophyll $a(\mathrm{Chl} a)$ $(\mu \mathrm{g} / \mathrm{L})$, dissolved oxygen (DO) (mg/L), pH , and turbidity ( FNU ) see Appendix D) were measured and recorded in situ at the upstream portion of the stream reach via a YSI EXO2 Multiparameter Sonde. All water quality instruments were calibrated in the laboratory prior to field use. Water samples were syringe filtered ( 60 mL Luer-Lok Tip syringe, $0.45 \mu \mathrm{~m}$ filter) into
collection bottles for dissolved organic carbon (DOC), dissolved nitrogen (DN), and dissolved phosphorus (DP) and frozen until laboratory analysis. Three samples of benthic Chl $a$ were collected from available substrate (e.g., rock, wood, sand/silt/clay) by scrubbing algal biomass from a standardized surface area and processing the sample through glass microfiber, GF/F filters ( $0.45 \mu \mathrm{~m}$ porosity, 47 mm diameter), via a filtration apparatus and hand pump. Additionally, $\sim 1 \mathrm{~L}$ of stream water (taken from just below the water surface) was filtered for sestonic $\mathrm{Chl} a$ and $\sim 1 \mathrm{~L}$ for total suspended solids (TSS) (pre-weighed, glass, microfiber, GF/F filters). Further water samples were collected for general chemistry, total nutrients, dissolved metals, and ultra low-level mercury.

Dissolved N, DP, Chl $a$ and TSS samples were analyzed at the U of S Aquatic Food Webs Laboratory (University of SK Toxicology Department, Saskatoon, SK, Canada). Nitrate, ammonia, and phosphate were analyzed using a YSI 9500 photometer. Chl $a$ (benthic, sestonic) samples were analysed with a Turner Designs fluorometer following hot ethanol extraction and all TSS filters were dried via a drying oven and reweighed. The SK Research Council (SRC) Environmental Analytical Laboratory performed water chemistry, total nutrient, and dissolved metal analysis on collected water samples.

### 3.6.3 Stream and Fish Habitat Assessment

A habitat assessment was performed at each stream site and consisted of a visual-based qualitative description of the physical habitat for both instream and riparian areas of the stream reach (Barbour et al., 1999). Sites were classified into high or low gradient streams based on U.S. Environmental Protection Agency (EPA) and Australian River Assessment System (AUSRIVAS) stream classification methods (Barbour \& Stribling, 1991; Barbour \& Stribling, 1994). The assessment included characterization of the stream riparian vegetation, canopy cover, woody debris, aquatic vegetation, bank stability, and channel pattern. Habitat assessment methods are based on the EPA standard protocol for a visual-based habitat assessment approach. The EPA's habitat assessment condition categories (optimal, suboptimal, marginal, and poor) and scoring criteria ( $0-20$ ) were used to allow consistency throughout the assessment and integration of each parameter into a final habitat assessment score per site reflecting the quantity and quality of stream fish habitat (/100\%).

### 3.7 Statistical Analysis, Normality, and Heteroscedasticity Assumptions

To ensure data followed assumptions of normality and homoscedasticity, data was analyzed by assessing skewness, kurtosis, histograms, Q-Q plots, as well as Shapiro-Wilk, and Levene's tests. Variables and fish metrics that did not meet assumptions of normality were $\log _{10}$ transformed prior to using the corresponding parametric statistical tests. Variables and metrics that were made less normal via $\log _{10}$ transformations were analyzed using the raw, closer to normal distribution, data. Where raw data values $=0, \log _{10}(x+1)$ was used. For all sites, the raw, non transformed, data was used to establish criteria thresholds (see below). For sites that had repeat site visits, the average of the three years were used to calculate low, moderate, and high stress criteria. Fish metrics were calibrated using data from the average of the three years as well, with the exception of the body condition (Fulton's $K$ ) of white sucker. Since white sucker were only collected in 12 of the 18 total sites, I chose to use data from all sites where white sucker were collected, including revisits to the same site. Fish metrics and the IBI were scored using all sites to obtain site scores for each year the site was sampled. All statistical analyses were done in SPSS (IBM Corp. released 2021. IBM SPSS Statistics for Windows, Version 28.0. Armonk, NY: IBM Corp.) unless otherwise stated.

### 3.8 Development of Reference Conditions

Sites were classified by geographical and stream/river type environmental descriptors and by determining the minimally disturbed or reference conditions (Stoddard et al., 2006) that are expected for physical habitat structure, water chemistry, and nutrients of the region. The simplified steps to develop reference conditions for the purpose of this IBI are as follows:
A) Site classification and minimizing natural variability
B) Assess and characterize collected environmental data
C) Develop and select environmental metrics for setting stressor classes
D) Establish reference criteria and site stress classes

### 3.8.1 Site Classification

The initial step to develop an IBI and reference conditions involves classifying streams into relatively homogenous units to organize and interpret natural variability amongst waterbodies and to minimize spatial complexity in the study region (Barbour et al., 1999). Geographical
(climatic, geomorphological, hydrological, biogeographic conditions) and waterbody type (size, catchment area, elevation, flow, ecological characteristics) differences may result in biological differences and biological communities which are not truly comparable (Bailey et al. 2004). To minimize this variability, environmental data needs to be organized into a narrow spatial scale. Stream sites in the Beaver River watershed were initially classified by watershed, ecoregion, and stream order (refer to section 2.3.3 Study Site Selection for further details). Sampling sites were restricted to the Beaver River watershed, the Mid-Boreal Upland and Boreal Transition ecoregions, and stream orders one through four (Table 3.1, Table 3.2). Due to the high flow conditions and resulting relatively low number of streams with conditions appropriate to sample, all 18 streams were retained as one grouping for analysis (Mid-Boreal Upland and Boreal Transition ecoregions and stream orders 1-4) and site stress classification.

Drainage area, stream order, and wetted width were assessed between the low (reference), moderate, and high stress classes (once developed) to test any effect of natural gradients on site stress groupings.

### 3.8.2 Environmental Metric Development and Selection

Environmental data (water quality, habitat, and land use) collected during the site bioassessment surveys or created using ArcGIS analyses techniques were used to further define and verify the sample sites into low, moderate, and high stress classes. When defining reference conditions, it is important to consider various biological, physical, chemical, and hydrologic conditions in the study region while avoiding influence of the biotic community (Bailey et al., 2004; Barbour et al., 1999). The following section outlines the methods undertaken to select environmental variables and develop quantitative criteria for defining the low (reference), moderate, and high stress classes.

### 3.8.2.1 Water Quality Measures

More than 60 variables related to physical, chemical, and biological properties of the stream water were collected during field sampling. Ideally, when developing reference condition, water quality data should be collected over long time frames, multiple seasons, and years. This allows mean values to be established for each parameter, rather than using one grab sample which may not accurately reflect the average natural conditions. Due to logistical constraints, single grab
samples were used to represent water quality conditions at each site. A literature search was initially used to gather information on established water quality - related reference criteria for SK, additional provinces in Canada, the United States, Europe, and other areas. Water quality variables were initially narrowed by focusing attention on those that were most important to lotic environments, fish, fish habitat, and local stressors. Then, variables with too many missing values among sites ( $>1 / 4$ of all sites), or an insufficient range across all sites (potential reference and test sites) established via best professional judgement (e.g., whether the data was well below established reference conditions for a similar region or had a very narrow range consistent with natural background concentrations or established reference conditions) were removed. Next, using the five revisited sites, signal to noise ratios were analysed to remove variables with a higher variance between years (within sites) than among sites (S:N ratio < 1). This step eliminated water quality data that was too variable over time to see a difference among sites. Although these steps never removed many variables, they can eliminate a few relatively unimportant or highly variable parameters from the model.

### 3.8.2.2 Fish (Physical) Habitat Measures

Approximately 60 variables related to stream physical habitat and the fish habitat assessment were recorded during the site assessment (41 parameters were used to describe and characterize physical habitat, 12 variables were related to habitat assessment, and 10 parameters were collected to calculate a single habitat assessment score). I retained only 10 of the habitat assessment variables to come up with a single site-specific score that represents the fish habitat quality (fish habitat indicator) in the development of reference conditions. The other variables were either redundant with the visual-based habitat assessment parameters or were used to further aid the habitat assessment scores and investigation of site stress classes.

### 3.8.2.3 GIS Analysis and Land Use Measures

Human-related disturbance measures were created by evaluating the physical characteristics and land use for the upstream contributing watersheds to each of the study sites using ArcMap version 10.6.1. Initial steps involved creating a default file geodatabase through ArcCatalog and then uploading all site coordinates into ArcMap to be used as reference points for delineating the upstream watershed area of each site using the ArcHydro Analysis Tool extension and Canadian

Digital Elevation Data (CDED). Digital Elevation Models (DEMs) for various National Topographic System (NTS) zones in the study area were accessed and downloaded through the government of Canada Open Data portal (https://open.canada.ca/data/en/dataset/7f245e4d-76c2-4caa-951a-45d1d2051333) provided by Natural Resources Canada. Hydrologic, hydrographic, topographic, and land use data were collected using various data extraction sources (see Appendix E for a full list of ArcGIS Data files assessed). Once obtained, data file layers were added into ArcMap, projected to a consistent Geographic Coordinate System (GCS) (NAD 1983 UTM 13N), and clipped to the Beaver River watershed polygon (created from selecting and merging various sub watershed polygons within the Canadian Watershed data layer and creating a new Beaver River watershed layer for all subsequent analyses). NAD 1983 was chosen as the GCS as it is the current reference system adopted as a national georeferencing standard by most federal and provincial agencies in Canada and endorsed by the Canadian Council on Geomatics. UTM zone 13 N was chosen (even though majority of the study sites fell in UTM zone 12U) since that is the SK standard and the major UTM zone used by government agencies in the province. Prior to land use criteria development, the upstream watershed area for each study site was delineated using the ArcHydro Analysis Tool extension. Refer to Appendix F for a simplified version of the watershed delineation steps used to create the site upstream contributing watersheds.

### 3.8.2.4 Land Cover and Site Physiography

To determine land cover in each site's upstream watershed, all data layers were merged into one layer using data management tools. Then the tabulate area function (in spatial analyst tools) was used to calculate various land cover measures for each of the upstream contributing watersheds to the 18 study sites, using the previously created polygon site watershed layer as the input raster/ feature zone data and the Agriculture and Agri-food Canada Land Use Layer (Annual Crop Inventory 2017; Appendix E). The resulting land cover data was then exported from the existing ArcMap file geodatabase to an excel spreadsheet where the percent composition of the total landcover in each site's upstream watershed was calculated for various land cover types (water, exposed land/barren, urban and developed land, shrubland, wetland, grassland, pasture, and forage land, cultivated lands (including 13 crop types), and forest land (coniferous, broadleaf, and mixed wood)). From this, four land cover measures, standardized to watershed
area, were developed: 1) \% agriculture (cultivated, pasture and forage land), 2) \% urban cover (urban and developed land), 3) \% human-related disturbance in watershed (\% agriculture and \% urban cover), and 4) \% natural landscape (including \% water and wetlands, forest, shrubland, and grassland). Site physiographical data (stream gradient, stream order, elevation, ecoregion, etc.) were also determined using various ArcMap analysis tools and existing data files (Appendix E). Stream order was determined from the National Hydrological Network of Canada layer (1:50,000 scale) and 1:50,000 topographic maps. I chose not to use the DEM to determine stream order, as it was obviously incorrect and overestimated stream order.

### 3.8.2.5 Road Data

The total length of roads was calculated within each site watershed (polygon) to develop road disturbance related measures using the National Road Network Data layer for the province of SK (Appendix E). First, the road shapefile was intersected/clipped to fit within each site watershed boundary, then spatial join was used to merge/sum the total length of roads in each polygon to create a new polygon feature class with the corresponding road data for each site's upstream watershed. The data was exported to excel, and the road density measure was calculated. The number of upstream road crossings was manually counted for each site's watershed within ArcMap.

### 3.8.2.6 Mining and Oil and Gas Wells

Mining and oil and gas well data for the province was used to determine the amount of land use in each site's watershed from natural resource extraction using the Saskatchewan Ministry of Energy and Resources data layer. No mines were located in the Beaver River watershed (approximately 140 mines are located in SK); however, there are some active and abandoned oil and gas operations present (>1500 wells). The number of oil and gas wells per watershed area (wells/ $\mathrm{km}^{2}$ ) was calculated for each site by using the spatial join feature in ArcMap, allowing the number of point features (vertical and non vertical wells) within each site watershed (polygon) to be tallied. The data was extracted from the attribute table in ArcMap and exported to an excel spreadsheet to calculate the final natural resource extraction related measures.

### 3.8.2.7 Forestry Harvest Operations

Logging operations are quite common in the Beaver River Watershed and data covering the past 50 years was collected from four local logging companies (Appendix E) to determine the total area logged per site watershed. In ArcMap, the total area of polygons (harvest blocks) within a site watershed was calculated by initially merging all the harvest data from the different companies into one data frame and then using the select by attribute feature to select harvest data from the last 20 years (1998-2018) to create a new layer. Twenty years was chosen for forest harvest operations data since this is a moderate timeline for succession/recovery of riparian and upstream vegetation for small and medium sized streams (Newaz, 2009; Quinn \& Wright-Stow, 2008). The spatial join feature was then used to determine the total harvest area in each site watershed by defining the merge feature as sum and match option as intersect for the harvest area data. These steps combined gave a new polygon feature class (layer) showing each site watershed and the corresponding harvest data over the past 20 years. The data was then exported to an excel spreadsheet and the percent area logged (standardized by watershed area) between the years 1998-2018 was calculated for each site.

### 3.8.2.8 Human Population Census Data

Population census data for SK (Appendix E) was used to determine the number of people per upstream watershed area for each of the study sites. In ArcMap, census blocks existing within the Beaver River watershed were clipped to include only those that fell in the upstream watershed area for the study sites. This created a new layer to allow estimation of watershed population size. Population within each site watershed polygon was calculated by estimating the percent area the clipped census block (clipped to each site watershed) makes up of the original census block (census area prior to clipping). Then this percentage was multiplied by the population of the original census block to get the number of people per $\mathrm{km}^{2}$ that fall within the new clipped census blocks. The spatial join tool (with site watersheds set as the target feature and census polygons as the join feature) was used to get the total summed population in each individual site watershed. The resulting output attribute table was exported to an excel spreadsheet to estimate the human population density (people/ $\mathrm{km}^{2}$ ) for each site watershed. It should be noted that there is an inherent amount of error in this calculation as the actual population is not evenly distributed across a census block as the calculation assumes. For the
purpose of this study, this estimation is considered approximate and adequate. Where obvious errors occurred (e.g., there were no apparent homesteads or communities within a site watershed, as determined from google earth and ground truthing) the human population size was adjusted to more accurately reflect the actual population of zero.

### 3.8.2.9 Landfill Data

Data for recorded landfills in SK was uploaded into ArcMap and used to determine the total number of landfills per site watershed from the Water Security Agency recorded landfills in SK data layer. Once the landfill layer was overlayed and clipped to the size of the site watershed boundary, the number of landfills per watershed was totalled.

### 3.8.2.10 Livestock Data

The number of heads of cattle was calculated for each site watershed (Appendix E) by following similar steps as were taken for the human population census data. However, this metric is not very accurate as I had to use the entire Southern SK cattle layer and multiply this by the percent area each watershed occupies of the larger cattle layer. This measure is assuming even distribution of cattle across the landscape, much like the human population layer, except this measure is less accurate due to the coarse scale at which this information layer was obtained. Therefore, I decided not to use this measure for calculating reference conditions and stress classes.

### 3.8.3 Minimizing Multicollinearity for Environmental Metrics

Lastly, water quality, land use, and habitat measures were run through a correlation analysis (Table 3.3) to eliminate any redundant variables, minimize multicollinearity, and select final environmental metrics. A correlation coefficient (r) greater than 0.7 was used as a cut-off value to determine whether variables are autocorrelated and the more biologically relevant variable or variable with the highest range among sites was retained for further analysis. This ensured that the selected metrics for development of reference conditions and stress classes for the index were nonredundant environmental measures.

Table 3.3| Correlation matrix used as a final step to eliminate redundant environmental measures (water quality, land use, and habitat; $\mathrm{n}=56$ variables total) prior to development of the low, moderate, and high stress site watershed ratings. The heat map represents correlations ranging from dark green (positive 1) to dark red (negative 1). Variables that have correlations close to 0 are indicated by white or light green and red colouration. The heat map is split between page 57 and 58.

|  |  |  |  |  |  |  |  |  |  |  |  | $\overline{0}$ 0 0 0 0 0 0 0 |  | $\begin{aligned} & \bar{u} \\ & \tilde{N} \\ & \tilde{N} \\ & E \\ & \tilde{u} \\ & \vdots \\ & \ddot{u} \end{aligned}$ | $\begin{aligned} & \text { İ } \\ & \underline{E} \\ & \hat{K} \end{aligned}$ |  |  | $\begin{aligned} & \text { E } \\ & \text { E } \\ & \frac{\text { E}}{\circ} \end{aligned}$ |  |  |  |  |  |  |  |  |  |  |  | $\begin{aligned} & \text { E } \\ & \text { 을 } \\ & \text { 을 } \\ & \text { 흗 } \end{aligned}$ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Habitat Assessment (\%) |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| \% Human-Related Disturbance in Watershed | -0.7 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| \% Area Logged Per Watershed | 0.0 | -0.4 | 1 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| No. of Oil and Gas Wells Per Watershed |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Area (wells/km2) | -0.1 | 0.3 | -0.2 | 1 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Road Density ( $\mathrm{m} / \mathrm{km} 2$ ) | -0.3 | 0.6 | 0.1 | 0.4 | 1 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| \% Agriculture | -0.7 | 1.0 | -0.4 | 0.3 | 0.6 | 1 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| \% Urban Cover | -0.3 | 0.8 | -0.4 | 0.5 | 0.7 | 0.8 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| \% Natural Landscape | 0.5 | -0.9 | 0.5 | -0.4 | -0.6 | -0.9 | -0.9 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| No. of Upstream Road Crossings | -0.5 | 0.7 | 0.1 | 0.4 | 0.7 | 0.7 | 0.7 | -0.6 | 1 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Human Population Density (people/km2) | -0.6 | 0.8 | -0.4 | 0.3 | 0.5 | 0.8 | 0.9 | -0.8 | 0.7 | 1 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| No. of Landfills Per Watershed | 0.1 | 0.2 | -0.3 | 0.7 | 0.3 | 0.2 | 0.6 | -0.4 | 0.3 | 0.3 | 1 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Water Temp ( ${ }^{\circ} \mathrm{C}$ ) | -0.2 | 0.1 | -0.2 | 0.1 | 0.1 | 0.1 | 0.2 | -0.1 | 0.2 | 0.2 | 0.3 | 1 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| TDS ( $\mathrm{mg} / \mathrm{L}$ ) | -0.4 | 0.7 | -0.2 | 0.3 | 0.7 | 0.7 | 0.8 | -0.7 | 0.8 | 0.8 | 0.2 | 0.1 | 1 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| SPC (us/cm at 25C) | -0.4 | 0.7 | -0.2 | 0.3 | 0.7 | 0.7 | 0.8 | -0.7 | 0.8 | 0.8 | 0.2 | 0.0 | 1.0 | 1 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| TSS (mg/L) | -0.5 | 0.2 | 0.1 | 0.2 | -0.1 | 0.2 | -0.1 | -0.1 | 0.1 | 0.0 | -0.3 | 0.0 | 0.2 | 0.2 | 1 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Turbidity (FNU) | -0.4 | -0.1 | 0.3 | -0.1 | -0.1 | -0.1 | -0.3 | 0.3 | 0.0 | 0.0 | -0.4 | 0.2 | 0.1 | 0.1 | 0.6 | 1 |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| DO (\% SAT) | -0.1 | 0.0 | -0.1 | 0.2 | 0.0 | 0.0 | 0.3 | -0.2 | 0.3 | 0.4 | 0.3 | 0.0 | 0.3 | 0.3 | -0.1 | 0.2 | 1 |  |  |  |  |  |  |  |  |  |  |  |  |  |
| DO ( $\mathrm{mg} / \mathrm{L}$ ) | 0.0 | -0.1 | 0.0 | 0.0 | -0.1 | -0.1 | 0.1 | -0.1 | 0.1 | 0.1 | 0.0 | -0.7 | 0.1 | 0.2 | -0.1 | 0.0 | 0.7 | 1 |  |  |  |  |  |  |  |  |  |  |  |  |
| Nitrate (mg/L ${ }^{\text {) }}$ | 0.1 | -0.1 | -0.2 | 0.0 | -0.3 | -0.1 | -0.2 | 0.0 | -0.5 | -0.3 | 0.0 | 0.0 | -0.3 | -0.4 | 0.3 | -0.1 | -0.2 | -0.2 | 1 |  |  |  |  |  |  |  |  |  |  |  |
| Total Nitrogen ( $\mathrm{mg} / \mathrm{L}$ ) | -0.1 | 0.0 | 0.5 | 0.2 | 0.4 | 0.0 | 0.2 | 0.0 | 0.4 | 0.0 | -0.1 | -0.4 | 0.3 | 0.4 | 0.2 | 0.0 | 0.1 | 0.3 | -0.1 | - |  |  |  |  |  |  |  |  |  |  |
| Ammonia ( $\mathrm{mg} / \mathrm{LN}$ ) | -0.4 | 0.7 | -0.3 | -0.1 | 0.5 | 0.7 | 0.5 | -0.6 | 0.4 | 0.6 | -0.2 | -0.2 | 0.7 | 0.7 | 0.2 | 0.0 | 0.0 | 0.2 | 0.0 | 0.3 |  |  |  |  |  |  |  |  |  |  |
| Phosphate ( $\mathrm{mg} / \mathrm{LP}$ ) | -0.3 | 0.5 | 0.1 | 0.4 | 0.8 | 0.5 | 0.5 | -0.4 | 0.6 | 0.4 | 0.3 | 0.4 | 0.6 | 0.6 | 0.2 | 0.1 | -0.1 | -0.3 | -0.2 | 0.2 | 0.2 | 1 |  |  |  |  |  |  |  |  |
| Total Phosphorus ( $\mathrm{mg} / \mathrm{L}$ ) | 0.2 | -0.3 | 0.1 | -0.2 | -0.1 | -0.3 | -0.4 | 0.4 | -0.3 | -0.3 | -0.3 | -0.5 | -0.2 | -0.1 | 0.1 | 0.0 | -0.2 | 0.2 | 0.0 | 0.5 | 0.1 | -0.3 | 1 |  |  |  |  |  |  |  |
| Benthic Chla ( $\mathrm{mg} / \mathrm{m} 2)$ | 0.1 | 0.4 | -0.3 | 0.3 | 0.4 | 0.4 | 0.6 | -0.5 | 0.4 | 0.4 | 0.2 | -0.1 | 0.6 | 0.6 | 0.0 | -0.2 | 0.1 | 0.2 | -0.5 | 0.1 | 0.4 | 0.2 | -0.1 | 1 |  |  |  |  |  |  |
| Suspended Chla (ug/L) | -0.3 | 0.5 | -0.2 | 0.2 | 0.4 | 0.5 | 0.3 | -0.4 | 0.3 | 0.3 | -0.2 | -0.2 | 0.6 | 0.6 | 0.5 | 0.3 | 0.0 | 0.2 | -0.2 | 0.1 | 0.5 | 0.3 | 0.1 | 0.6 | 1 |  |  |  |  |  |
| P. Alkalinity ( $\mathrm{mg} / \mathrm{L}$ ) | 0.0 | 0.0 | 0.3 | 0.1 | 0.3 | -0.1 | 0.0 | 0.1 | 0.1 | -0.2 | -0.1 | -0.6 | 0.1 | 0.2 | 0.2 | 0.0 | 0.0 | 0.4 | 0.0 | 0.6 | 0.2 | 0.1 | 0.5 | 0.1 | 0.3 | 1 |  |  |  |  |
| Fluoride (mg/L) | -0.3 | 0.4 | 0.2 | 0.2 | 0.6 | 0.3 | 0.4 | -0.3 | 0.7 | 0.5 | 0.1 | 0.0 | 0.7 | 0.7 | 0.1 | 0.1 | 0.3 | 0.2 | -0.3 | 0.6 | 0.5 | 0.5 | 0.1 | 0.3 | 0.4 | 0.5 | 1 |  |  |  |
| Bicarbonate ( $\mathrm{mg} / \mathrm{L}$ ) | -0.4 | 0.7 | -0.1 | 0.4 | 0.8 | 0.7 | 0.8 | -0.7 | 0.8 | 0.8 | 0.3 | 0.0 | 1.0 | 1.0 | 0.2 | 0.1 | 0.3 | 0.2 | -0.3 | 0.4 | 0.6 | 0.6 | -0.2 | 0.6 | 0.6 | 0.3 | 0.8 | 1 |  |  |
| Carbonate ( $\mathrm{mg} / \mathrm{L}$ ) | 0.0 | 0.0 | 0.3 | 0.1 | 0.3 | -0.1 | -0.1 | 0.1 | 0.1 | -0.2 | 0.0 | -0.6 | 0.1 | 0.2 | 0.2 | -0.1 | 0.0 | 0.4 | -0.1 | 0.6 | 0.2 | 0.1 | 0.5 | 0.1 | 0.3 | 1.0 | 0.5 | 0.2 |  |  |
| Chloride ( $\mathrm{mg} / \mathrm{L}$ ) | -0.3 | 0.7 | 0.0 | 0.3 | 0.9 | 0.7 | 0.8 | -0.7 | 0.8 | 0.7 | 0.3 | 0.1 | 0.8 | 0.8 | -0.1 | -0.2 | 0.1 | 0.0 | -0.3 | 0.3 | 0.5 | 0.7 | -0.3 | 0.5 | 0.3 | 0.2 | 0.6 | 0.8 | 0.2 |  |
| Total Alkalinity ( $\mathrm{mg} / \mathrm{L}$ ) | -0.4 | 0.7 | -0.1 | 0.4 | 0.8 | 0.6 | 0.8 | -0.7 | 0.8 | 0.7 | 0.3 | 0.0 | 1.0 | 1.0 | 0.2 | 0.1 | 0.3 | 0.2 | -0.3 | 0.5 | 0.6 | 0.6 | -0.1 | 0.6 | 0.6 | 0.3 | 0.8 | 1.0 | 0.3 | 0.8 |
| Sum of lons ( $\mathrm{mg} / \mathrm{L}$ ) | -0.4 | 0.7 | -0.2 | 0.4 | 0.7 | 0.7 | 0.8 | -0.7 | 0.8 | 0.8 | 0.2 | 0.0 | 1.0 | 1.0 | 0.2 | 0.1 | 0.3 | 0.2 | -0.3 | 0.4 | 0.7 | 0.6 | -0.2 | 0.6 | 0.6 | 0.2 | 0.8 | 1.0 | 0.2 | 0.8 |
| Total Hardness ( $\mathrm{mg} / \mathrm{L}$ ) | -0.4 | 0.7 | -0.1 | 0.3 | 0.7 | 0.7 | 0.8 | -0.7 | 0.8 | 0.8 | 0.2 | 0.0 | 1.0 | 1.0 | 0.2 | 0.2 | 0.3 | 0.2 | -0.4 | 0.4 | 0.7 | 0.6 | -0.1 | 0.6 | 0.6 | 0.2 | 0.7 | 1.0 | 0.2 | 0.8 |
| Sulfate ( $\mathrm{mg} / \mathrm{L}$ ) | -0.4 | 0.6 | -0.2 | 0.2 | 0.5 | 0.6 | 0.6 | -0.6 | 0.5 | 0.7 | 0.0 | -0.1 | 0.9 | 0.9 | 0.3 | 0.2 | 0.2 | 0.2 | -0.2 | 0.3 | 0.8 | 0.4 | 0.0 | 0.6 | 0.7 | 0.0 | 0.5 | 0.8 | 0.0 | 0.6 |
| Calcium (mg/L) | -0.5 | 0.7 | -0.1 | 0.5 | 0.7 | 0.6 | 0.8 | -0.6 | 0.8 | 0.8 | 0.3 | 0.1 | 0.9 | 0.9 | 0.2 | 0.2 | 0.3 | 0.1 | -0.3 | 0.4 | 0.6 | 0.6 | -0.1 | 0.5 | 0.5 | 0.1 | 0.7 | 1.0 | 0.1 | 0.7 |
| Magnesium (mg/L) | -0.4 | 0.7 | -0.2 | 0.2 | 0.7 | 0.7 | 0.7 | -0.6 | 0.7 | 0.7 | 0.1 | 0.0 | 1.0 | 1.0 | 0.1 | 0.2 | 0.3 | 0.2 | -0.4 | 0.3 | 0.7 | 0.5 | -0.2 | 0.6 | 0.6 | 0.2 | 0.7 | 0.9 | 0.2 | 0.8 |
| Potassium ( $\mathrm{mg} / \mathrm{L}$ ) | -0.3 | 0.7 | -0.4 | 0.4 | 0.7 | 0.7 | 0.8 | -0.8 | 0.6 | 0.6 | 0.4 | -0.2 | 0.7 | 0.8 | 0.0 | -0.2 | 0.2 | 0.3 | -0.1 | 0.2 | 0.7 | 0.4 | -0.2 | 0.6 | 0.6 | 0.3 | 0.4 | 0.8 | 0.3 | 0.7 |
| Sodium (mg/L) | -0.2 | 0.5 | -0.2 | 0.2 | 0.6 | 0.5 | 0.6 | -0.5 | 0.6 | 0.6 | 0.1 | 0.1 | 0.9 | 0.9 | 0.2 | 0.1 | 0.2 | 0.0 | -0.3 | 0.4 | 0.7 | 0.5 | -0.1 | 0.7 | 0.6 | 0.2 | 0.8 | 0.9 | 0.2 | 0.7 |
| Mercury (ug/L) | -0.1 | 0.1 | -0.1 | 0.1 | 0.0 | 0.1 | 0.2 | -0.1 | 0.0 | 0.3 | 0.2 | 0.7 | 0.1 | 0.0 | 0.1 | 0.1 | -0.1 | -0.6 | 0.2 | -0.2 | 0.0 | 0.2 | -0.3 | -0.1 | -0.1 | -0.5 | 0.1 | 0.1 | -0.5 | 0.0 |
| Aluminum (ug/L) | -0.2 | 0.0 | -0.2 | -0.2 | -0.6 | 0.0 | -0.1 | 0.0 | -0.2 | 0.0 | -0.2 | 0.1 | -0.2 | -0.3 | 0.2 | 0.0 | 0.1 | 0.0 | 0.3 | -0.2 | 0.0 | -0.6 | -0.2 | -0.3 | -0.4 | -0.5 | -0.5 | -0.4 | -0.5 | -0.4 |
| Arsenic (ug/L) | -0.2 | 0.1 | -0.2 | 0.5 | -0.1 | 0.1 | 0.2 | -0.2 | 0.1 | 0.3 | 0.3 | 0.3 | 0.4 | 0.3 | 0.4 | 0.4 | 0.1 | -0.1 | 0.1 | -0.1 | 0.1 | 0.1 | -0.2 | 0.2 | 0.2 | -0.3 | 0.1 | 0.3 | -0.3 | -0.1 |
| Iron (ug/L) | 0.0 | -0.1 | 0.0 | -0.1 | -0.5 | 0.0 | -0.2 | 0.0 | -0.2 | -0.2 | -0.1 | -0.3 | -0.1 | -0.2 | 0.1 | 0.0 | -0.2 | 0.0 | 0.3 | 0.0 | 0.1 | -0.3 | 0.0 | -0.2 | -0.2 | -0.1 | -0.4 | -0.2 | -0.1 | -0.2 |
| Selenium ( $u \mathrm{~g} / \mathrm{L}$ ) | 0.0 | 0.0 | 0.1 | -0.1 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | -0.1 | -0.1 | 0.9 | -0.1 | 0.0 | -0.2 | -0.3 | 0.2 | 0.8 | -0.1 | 0.4 | 0.2 | -0.2 | 0.4 | 0.0 | 0.1 | 0.6 | 0.1 | 0.0 | 0.6 | 0.1 |
| Copper (ug/L) | -0.6 | 0.4 | 0.0 | 0.0 | 0.0 | 0.4 | 0.2 | -0.3 | 0.2 | 0.4 | -0.2 | 0.1 | 0.1 | 0.1 | 0.2 | 0.2 | -0.1 | -0.1 | 0.2 | -0.1 | 0.2 | -0.2 | -0.2 | -0.2 | -0.1 | -0.3 | 0.0 | 0.1 | -0.3 | 0.1 |
| Uranium (ug/L) | -0.6 | 0.6 | -0.3 | 0.3 | 0.2 | 0.7 | 0.7 | -0.8 | 0.4 | 0.7 | 0.1 | -0.1 | 0.6 | 0.6 | 0.4 | 0.2 | 0.5 | 0.5 | 0.0 | 0.1 | 0.5 | 0.1 | -0.2 | 0.4 | 0.5 | -0.1 | 0.2 | 0.6 | -0.1 | 0.4 |
| Zinc (ug/L) | -0.1 | 0.2 | 0.0 | 0.0 | 0.2 | 0.1 | 0.3 | -0.1 | 0.3 | 0.5 | 0.1 | 0.7 | 0.4 | 0.3 | -0.2 | 0.3 | 0.1 | -0.4 | -0.4 | -0.2 | 0.0 | 0.2 | -0.4 | 0.2 | -0.1 | -0.7 | 0.1 | 0.3 | -0.7 | 0.3 |
| Titanium (ug/L) | 0.0 | -0.1 | 0.3 | 0.0 | 0.0 | -0.1 | -0.1 | 0.1 | 0.0 | -0.2 | -0.3 | -0.7 | 0.1 | 0.1 | 0.3 | -0.1 | 0.0 | 0.5 | 0.1 | 0.7 | 0.3 | -0.1 | 0.4 | 0.0 | 0.2 | 0.8 | 0.4 | 0.2 | 0.8 | 0.0 |
| Boron (mg/L) | 0.0 | 0.3 | -0.2 | 0.2 | 0.5 | 0.2 | 0.4 | -0.3 | 0.3 | 0.3 | 0.1 | 0.2 | 0.7 | 0.7 | 0.3 | 0.2 | 0.0 | -0.1 | -0.2 | 0.2 | 0.5 | 0.4 | 0.0 | 0.6 | 0.6 | 0.1 | 0.5 | 0.7 | 0.1 | 0.4 |
| Manganese ( $\mathrm{ug} / \mathrm{L}$ ) | -0.2 | 0.0 | 0.1 | -0.1 | -0.3 | 0.0 | -0.3 | 0.1 | -0.3 | -0.2 | -0.5 | -0.6 | 0.0 | 0.0 | 0.5 | 0.3 | -0.3 | 0.2 | 0.2 | 0.1 | 0.3 | -0.3 | 0.2 | -0.1 | 0.3 | 0.3 | -0.1 | -0.1 | 0.3 | -0.2 |
| Nickel ( $\mathrm{ug} / \mathrm{L}$ ) | -0.6 | 0.4 | -0.1 | 0.2 | 0.0 | 0.4 | 0.3 | -0.4 | 0.4 | 0.6 | 0.0 | -0.3 | 0.4 | 0.4 | 0.3 | 0.1 | 0.5 | 0.6 | -0.2 | 0.3 | 0.4 | -0.1 | 0.1 | 0.0 | 0.1 | 0.2 | 0.4 | 0.4 | 0.2 | 0.1 |
| Strontium (mg/L) | -0.3 | 0.6 | -0.1 | 0.5 | 0.7 | 0.5 | 0.7 | -0.5 | 0.7 | 0.6 | 0.2 | 0.0 | 0.9 | 0.9 | 0.3 | 0.2 | 0.2 | 0.1 | -0.4 | 0.4 | 0.6 | 0.6 | -0.1 | 0.7 | 0.7 | 0.2 | 0.7 | 0.9 | 0.2 | 0.7 |
| Vanadium ( $u \mathrm{~g} / \mathrm{L}$ ) | -0.7 | 0.5 | -0.2 | 0.1 | 0.0 | 0.6 | 0.4 | -0.5 | 0.4 | 0.6 | 0.0 | 0.3 | 0.4 | 0.3 | 0.3 | 0.3 | 0.3 | 0.0 | 0.2 | 0.0 | 0.3 | 0.0 | -0.3 | -0.1 | 0.0 | -0.4 | 0.1 | 0.3 | -0.4 | 0.1 |
| Barium (mg/L) | -0.1 | 0.1 | 0.1 | 0.3 | 0.4 | 0.1 | 0.4 | -0.3 | 0.3 | 0.3 | 0.4 | 0.2 | 0.5 | 0.5 | 0.1 | 0.4 | 0.2 | 0.0 | -0.2 | 0.2 | 0.1 | 0.4 | 0.1 | 0.3 | 0.2 | 0.0 | 0.2 | 0.5 | 0.0 | 0.3 |
| Chromium ( $\mathrm{ug} / \mathrm{L}$ ) | -0.3 | 0.4 | -0.2 | 0.1 | 0.0 | 0.4 | 0.3 | -0.4 | 0.3 | 0.5 | 0.1 | 0.6 | 0.4 | 0.2 | 0.1 | 0.2 | 0.0 | -0.4 | -0.1 | -0.3 | 0.1 | 0.1 | -0.4 | 0.2 | 0.0 | -0.7 | -0.1 | 0.2 | -0.7 | 0.2 |
| Cobalt (ug/L) | -0.7 | 0.6 | 0.0 | 0.2 | 0.1 | 0.6 | 0.4 | -0.6 | 0.5 | 0.6 | -0.1 | -0.2 | 0.5 | 0.5 | 0.3 | 0.1 | 0.1 | 0.2 | -0.1 | 0.2 | 0.4 | 0.0 | -0.3 | 0.2 | 0.2 | -0.1 | 0.1 | 0.5 | -0.2 | 0.4 |
| Molybdenum ( $\mathrm{ug} / \mathrm{L}$ ) | -0.2 | 0.0 | 0.0 | 0.5 | 0.0 | 0.0 | 0.3 | -0.1 | 0.2 | 0.3 | 0.3 | 0.2 | 0.4 | 0.4 | 0.5 | 0.4 | 0.3 | 0.0 | -0.1 | 0.2 | 0.0 | 0.2 | 0.0 | 0.2 | 0.2 | -0.2 | 0.2 | 0.4 | -0.2 | 0.0 |



### 3.8.4 Setting Reference Criteria and Establishing Stress Classes

As best as practical, sites were selected across a gradient of disturbance to reflect potential low (reference sites), moderate (intermediate sites), and high (test sites) stressor classes (Table 3.2, Figure 3.1). Test sites are those sites with known exposure to a stressor(s). However, this does not necessarily imply a priori if these sites are environmentally damaged. Information on environmental damage can only be established a posteriori once the index is developed and site stress classifications are verified via the biological community (Bailey et al., 2004; Barbour et al., 1999). Google Earth imagery, topographic maps, forestry operations maps, local knowledge, literature search, and initial site visits were used to determine dominant land use, road access, terrain (forest, bog, agriculture, etc.), proximity to point and non-point source pollution in the watershed, and to initially select streams across a gradient of local disturbances. It should be noted that stressor importance and type used in the analysis will vary with the specific region under study and for the purposes of this research were restricted to the Beaver River Watershed. First, all streams with minimal surrounding disturbances were identified as potential reference sites. Additionally, reference sites need to be representative of the study area. Various criteria were used to initially identify the potential reference sites (Table 3.4 , a priori criteria). Second, stream locations that were considered to be the most disturbed were identified as potential high stress sites. Lastly, any streams that fell into moderately disturbed categories (greater disturbance than potential reference sites but less than the most disturbed sites) were identified.

Various approaches were analyzed and compared to determine the most appropriate method for developing reference condition and stress classification using the environmental variables (mean and standard deviation, median and interquartile range, equal interval or ranked approach, specific cut-off criteria, and established guidelines). The equal interval or ranked approach separates sites evenly in a ranked order into low, moderate, and high stress categories (e.g., 6 sites in low, 6 sites in moderate, and 6 sites in high stress). Specific cut-off criteria are selected based on established literature, protocols, observing descriptive statistics of the data, and professional judgement (knowledge of the watershed and land use in the area). For water quality variables where established criteria or guidelines existed, the established criteria were also used to obtain stress classifications (e.g., high stress $=$ sites exceeding the guidelines, moderate stress $=$ sites below and down to $10 \%$ of the guideline, and low stress $=$ sites below $10 \%$ of the established guideline). Depending on the distribution of the data, some of the above
classification methods worked better than others (see section 5.4 for further discussion on stress criteria selection methods). To maintain consistency in defining stress classes across all the data, I decided to use the percentile approach, where the median and interquartile range were used to define low (reference), moderate, and high stress classes for each stream watershed. For example, when looking at a positive scoring variable, $\geq 75^{\text {th }}$ percentile $=$ high stress, $50^{\text {th }}-75^{\text {th }}$ percentile $=$ moderate stress, and $\leq 25^{\text {th }}$ percentile $=$ low stress. For negative scoring variables, the opposite is true (e.g., $\geq 75^{\text {th }}$ percentile $=$ low stress, $50^{\text {th }}-75^{\text {th }}$ percentile $=$ moderate stress, and $\leq 25^{\text {th }}$ percentile $=$ high stress $)$.

To calculate the final stress classifications for each site, all the variable stress ratings were tallied. Two approaches were compared for this step (the total summed rating and using the mode of the dataset). For the total summed rating, the first step was to assign numeric values of 1,2 , and 3 to the stress categories (low, moderate, and high). The total summed rating was then calculated by taking the total sum of the values assigned to each stress category to get the final stress rating for each site (e.g., for the 19 remaining variables used, 19-31.66 $=$ low stress, 31.67$44.33=$ moderate stress, and 44.34-57 $=$ high stress). For the mode approach, the stressor rating that most often occurs is simply chosen as the final stressor rating for each site. The mode approach was rejected as this method does not take all the data values into account, just the most common one. Therefore, this approach can be misleading (e.g., final rating is low when really there were four low, three high and one moderate scores). For ease of use and consistency, the total summed rating was chosen as the most appropriate and easily interpreted method to determine final stress ratings.

Table 3. $4 \mid$ Qualitative and quantitative reference site criteria used in the development of reference conditions for the Index of Biotic Integrity in the Beaver River Watershed. A priori criteria are those used to initially identify and select potential reference sites. A posteriori criteria were calculated when verifying reference sites.

## Reference Site Criteria

## A Priori Criteria

## Sites Should:

be representative of the major characteristics of the streams (substrate, discharge, stream type, habitats, etc.)
be accessible and safe during sampling
cover a wide range of physical and chemical conditions (e.g., stream discharge, drainage area, elevation, habitat) within the study area
be $>10 \mathrm{~km}$ from any known point source of pollution
be $>50 \mathrm{~m}$ upstream or $>300 \mathrm{~m}$ downstream of a bridge, low level crossing, water impoundment, extraction, diversion, livestock watering area, significant confluence, discharge, or lake inflow, areas subject to channel modification, dredging or shoreline/ riparian disturbance be > than 100 m from nearest road or road crossing
have no, or as little as possible, human-related disturbance (including mining, logging, agriculture, flow modifications or urbanization) be truly perennial (as indicated by the presence of fish, univoltine insects and riparian vegetation)
have a riparian vegetative zone width $>18 \mathrm{~m}$ and no riparian zone modification

## A Posteriori Criteria

## Sites Should:

have a habitat assessment score $\geq 94 \%$
have human-related disturbance in watershed $\leq 0.2 \%$
have zero oil and gas wells within watershed
have $\leq 2.6 \%$ area harvested within watershed
have a human population density of zero within watershed
have a road density $\leq 77 \mathrm{~m} / \mathrm{km}^{2}$ within watershed
have nitrate and phosphate concentrations $\leq 0.04 \mathrm{mg} / \mathrm{L}$
~ have TN and TP concentrations $\leq 0.68 \mathrm{mg} / \mathrm{L} \mathrm{N}$ and $\leq 0.03 \mathrm{mg} / \mathrm{L}$ P, respectively
have DO concentrations $\geq 9.5 \mathrm{mg} / \mathrm{L}$
have TSS concentrations $\leq 1.22 \mathrm{mg} / \mathrm{L}$
Some of the qualitative criteria in this table have been modified from Carter, 2012, Bailey et al., 2004, Davies, 1994, and USEPA, 2002.
Quantitative criteria are those which were calculated using percentiles of the total site distribution.
Watershed refers to the upstream contributing watershed area to that site as calculated from ArcGIS.

### 3.9 Development of an Index of Biotic Integrity (IBI)

The IBI development is an iterative process, and site stress classification, metric selection, and metric calibration were revisited throughout the analysis. The initial step to develop an IBI involves classifying streams into relatively homogenous units to organize and interpret natural variability among streams (Barbour et al., 1999; see section 3.8.1 Site Classification). This is followed by characterizing the distribution of reference and impaired conditions (low stress to high stress; see section 3.8.2 Environmental Metric Development and Selection and section 3.8.4 Setting Reference Criteria and Establishing Stress Classes). The following section will describe metric selection and calibration for the index, metric scoring, and final index development. Metrics sensitive to anthropogenic stress are chosen and scored relative to undisturbed, or low stress, fish communities. An assessment of ecological status can then be made based on individual metric and final index scores (European Water Framework Directive (EUWFD), 2002).

The IBI development process can be summarized in five main steps:

1. Organize and interpret natural variability among sites/streams (described above)
2. Develop an understanding of reference and impaired conditions (described above)
3. Metric selection
4. Metric calibration
5. Metric scoring and index development

### 3.9.1 Metric Selection

The next step in the IBI development process involves identifying candidate metrics for the index. Candidate metrics were selected based on characteristics of the resident fish community (trophic, tolerance and reproductive guilds, forage habitat and other life history traits) and by reviewing metrics used in previous IBIs (Barbour et al., 1999; Bramblett et al., 2005; Cantin \& John, 2012; Karr, 1981; Karr et al., 1986; Long \& Walker, 2005; Mebane et al., 2003; Prestie, 2014, unpublished; Stevens et al., 2006; Stevens \& Council, 2008). Preference was given to metrics used in biogeographically similar areas. Information on fish characteristics was gathered from Nelson \& Paetz (1992), Joynt \& Sullivan (2003), Stewart \& Watkinson (2004), Scott \& Crossman (1973), Prestie (unpublished, 2014), Atton \& Merkowsky (1983), Bramblett et al. (2005), Cantin \& John (2012), Stevens \& Council (2008) and Stevens et al. (2006). The first step
involves creating or choosing appropriate metrics within each metric category. Representative metrics were selected from each of four primary categories:

1) richness and composition measures for diversity and dominance of the assemblage;
2) tolerance measures that represent fish assemblage sensitivity to perturbation;
3) trophic, habitat, and reproductive measures for information on fish community ecology and guilds; and
4) Abundance and condition measures on individual fish health

### 3.9.2 Metric Calibration

The next step involves assessing and calibrating the candidate metrics to determine core metrics for final integration in the index. Following the statistical methods and procedures in Stoddard et al. (2008) and Barbour et al. (1999) appropriate metrics will have adequate variability in data values among sites (metric range), temporal stability (metric reproducibility), responsiveness to stressor gradients (ability to discriminate between reference and impaired conditions) (metric responsiveness) and independence from other metrics (metric redundancy). The metric calibration process involves six critical steps (modified from Stoddard et al., 2008):

1. Classification of metrics into representative metric categories
2. Assess metric range
3. Assess metric reproducibility (temporal stability)
4. Check metric redundancy
5. Check metric responsiveness to stressors
6. Adjust final metrics for natural gradients

### 3.9.2.1 Metric Range

This step involves assessing individual metrics for insufficient data values and range. Descriptive statistics were analyzed to help characterize metric performance. If a metric had a high proportion of zero values at sites or minimal variability between sites, it was removed. This eliminates metrics with very narrow ranges (e.g., richness metrics including only one or two taxa) or comparable values across sites.

### 3.9.2.2 Metric Reproducibility

Temporal variability of each metric was analyzed using signal: noise ratios ( $\mathrm{S} / \mathrm{N}$; ratio of the variance among all sites (signal) to the variance of repeated visits to the same sites (noise)) (Kaufmann et al., 1999) at the five revisited sample sites. This step helps eliminate highly variable metrics from the index.

### 3.9.2.3 Metric Redundancy

Correlation analysis was used to assess metric collinearity and minimize metric redundancy. For this study, metric values were considered redundant when the Pearson correlation coefficient (r) was $\geq 0.7$, which is equivalent to metrics that share approximately half their information content ( $r>0.71, R^{2}=0.5$, Stoddard et al., 2008). Increasing the cut-off value to $r \geq 0.8$ made minimal difference in the quantity of metrics being retained. Only reference site data was used in the correlation analysis to avoid misinterpreting metric correlation from metric response to similar stressor gradients. When two or more metrics had $\mathrm{r} \geq 0.7$, the decision to include one metric over another was based on biological relevance, metric relationship to the stressors (consistent anticipated response to anthropogenic disturbance), and whether the metric was successfully used in a previously published IBI program. Preference was given to metrics used in biogeographically similar IBIs.

### 3.9.2.4 Metric Responsiveness

Besides being ecologically relevant to the resident fish assemblage and local stressors, the metrics chosen for IBI development also need to be responsive to disturbance. Characterization of metric response to a disturbance scale additionally allows metrics to be used as a diagnostic tool (Barbour et al., 1999). To assess metric responsiveness, the distribution of metrics was compared between the previously classified reference and high stress sites (section 2.3.8). If there is minimal overlap, the metric is useful at discriminating between reference and impaired conditions. Metric values were plotted against stress categories (determined from physicochemical parameters, habitat assessments and land use) to reveal metric responsiveness. T-tests were used to compare mean metric values between least disturbed and most disturbed sites. Additionally, t scores were used as a measure of metric responsiveness, where higher scores indicate greater response to the stressor gradient (Stoddard et al., 2008).

### 3.9.2.5 Variability with Natural Gradients

Metrics can also vary with natural gradients (e.g., stream order, catchment area, wetted stream width, stream gradient). Therefore, it is important to differentiate between variability caused by anthropogenic (stressor-based gradients) and naturally driven gradients. This step involves adjusting metrics that show a moderate or greater correlation ( $r \geq 0.4$ ) with natural gradients. Stressors themselves can vary with those same natural gradients (Stoddard et al., 2008), and as a result, fish community data should only be assessed at reference sites, removing any potential anthropogenic stressors. Correlation and multiple linear regression analysis was used to assess metric relationships to various natural gradients (stream order, $\log _{10}$ upstream drainage area $\left(\mathrm{km}^{2}\right)$, and $\log _{10}$ average wetted stream width (m)) using only data from our low stress sites and if necessary, predict and adjust metric scores.

### 3.9.3 Metric Scoring and Index Development

### 3.9.3.1 Setting Metric Scores

The purpose of an index is to provide an easily interpreted integrated measure of biological condition (Barbour et al., 1999; Karr, 1981). Metrics vary in their scale (e.g., percentages, integers, dimensionless numbers, etc.); therefore, it is important to standardize core metrics to a common scale to allow equal weighting (e.g., a $50 \%$ change on one metric provides an equal value to the assessment as a $50 \%$ change in another metric). Discrete scoring (e.g., $1,3,5$ ), common to IBIs developed in the past, has been criticized for increasing the variability of the final IBI (Blocksom, 2003). Therefore, I developed a common scoring scale by applying linear interpolation to metric values between upper and lower thresholds to obtain unitless, continuous scores between 0 and 10 (Hughes et al., 1998; Minns et al., 1994; Stoddard et al., 2008). The $95^{\text {th }}$ and $5^{\text {th }}$ percentiles of the entire site distribution were used to set the upper (ceiling, i.e., 10) and lower (floor, i.e., 0) thresholds, respectively, and to remove outliers (Hughes et al., 1998; Minns et al., 1994; Stoddard et al., 2008). Metric values above or below the upper and lower thresholds were assigned a value of 10 or 0 , respectively. For negative scoring metrics (metrics that increase in response to perturbation) the lower threshold is set at the 95th percentile and the upper threshold will be the 5th percentile. This approach is known to produce an MMI with the lowest variability and highest responsiveness (Stoddard et al., 2008). This procedure allows all metrics
to be weighted equally and establishes single site-specific scores for simplified management. The formula for linear interpolation is as follows:
$\mathrm{Y}=\mathrm{Y} 1+((\mathrm{Y} 2-\mathrm{Y} 1) /(\mathrm{X} 2-\mathrm{X} 1)) *(\mathrm{X}-\mathrm{X} 1)$
where, Y 1 and Y 2 are the minimum and maximum scores possible for the equation (e.g., 0 or 10 ), X 1 and X 2 are the minimum and maximum observed metric values (e.g., the $5^{\text {th }}$ and $95^{\text {th }}$ percentile values), X is the observed metric value to be interpolated between 0 and 10 , and Y is the interpolated value.

For metric scoring and index development, the distribution of data at all sites (including the five revisited sites) was used to set expectations. This allowed all sites, including those sampled across different years, to be compared.

### 3.9.3.2 Scoring the Index

For each site, the metric scores (between 0 and 10) were summed, multiplied by 10 , and divided by the total number of metrics to get a site-specific score between 0 and 100 (Cao et al., 2007; Vander Laan \& Hawking, 2014). Biological condition of the waterbody was then assessed via the summed index score. Additionally, we can loosely interpret index scores based on biocriteria set from threshold values to establish health ratings (e.g., very poor, poor, fair, good, and very good condition).

### 3.10 Assessing Interannual Variability in the IBI and Environmental Data

Five of the sites were sampled over a three-year period (2016, 2017, and 2018, Table 3.2) to evaluate environmental data and the IBI for potential interannual variability of fish habitat, water quality, fish metrics and site index scores. Due to time constraints, I was only able to sample five sites in 2016; therefore, these sites were revisited in the 2017 and 2018 field season to allow assessment of potential annual variation in fish assemblages and the IBI. Ideally, two or more sites should be sampled within the low, moderate, and high stress groupings. The final human disturbance gradient was not established until after the initial sampling, and therefore, I was not able to randomly select the revisited sites from the low, moderate, and high stress groups. Instead, two sites fell in the low stress category (Alcott Creek and Flotten River) and three in the
moderate stress group (Nolin Creek, Sukaw Creek, and DeLaRonde Creek). This analysis allows a determination of the stability of the index and fish metrics through time, and ultimately assess the practicality of the IBI as an ecological assessment tool for lotic environments in northern regions with environmental extremes and immense seasonal variability. Repeated measures ANOVAs were performed using R Statistical Language v 4.3.1 (R Core Team, 2023) to compare the effect of site, sampled over a three-year period, on various water quality and habitat parameters used in the development of reference conditions, as well as fish species richness, fish relative abundance, the IBI metrics and scores. Post Hoc Bonferroni correction method was used to determine which sites were significantly different.

## Chapter 4: RESULTS

### 4.1 The Fish Assemblage

A total of 3210 fish were collected from streams in the Beaver River watershed, comprising 16 species across six different families (including one hybrid species (Chrosomus neogaeus x eos)) (Table 4.1). White sucker was the most commonly found species, occurring in over half of the sites sampled ( $57 \%$ of sites), followed by brook stickleback and fathead minnow (occurring at $54 \%$ of sites). Lake chub was the fourth most common species collected, occurring in half of the streams sampled. Spottail shiner was the least commonly found species, occurring at only one $4^{\text {th }}$-order stream site. Longnose sucker was the second least occurring fish species, found in only $14 \%$ of the sites sampled. The stream with the highest species richness was Sukaw Creek in 2018 (10 species) and Flotten River in 2016 ( 10 species). Sukaw Creek, Flotten River, and Alcott Creek were the most species-rich overall, while four sites (Nolin Creek, Backwater Creek, Nesslin Creek, and Tea Creek) contained only one species of fish. Five species of fish dominated the Beaver River stream fish assemblage, making up $85 \%$ of all the fish collected in this study. Fathead minnow comprised $32 \%$ of the fish collected, followed by brook stickleback ( $20 \%$ ), finescale dace ( $16 \%$ ), white sucker ( $8 \%$ ), and lake chub ( $8 \%$ ). Although 1035 of the 3210 individual fish collected were fathead minnow, the majority of these were collected at only one site ( $n=738$ at Spiritwood Creek). Spiritwood Creek had the highest relative abundance of fish ( $\mathrm{n}=806$ ), most of which were fathead minnows. Nesslin Creek had the lowest abundance of fish, with only one northern pike being collected at the sampling site, despite 37 minutes of electrofishing time.

### 4.2 Development of Reference Conditions

### 4.2.1 Environmental Metric Development and Selection

Of the 59 water quality, habitat, and land use variables selected and/or created during environmental metric development and selection (see Appendix G and H), 19 variables (one habitat, three land use, and 15 water quality related variables) were chosen for use in developing reference condition and stress classes for the IBI (Table 4.2). Only one measure of habitat (fish habitat assessment score) was chosen to represent fish habitat quality (Table 4.2). The habitat assessment score was not highly correlated ( $\mathrm{r}>0.7$ ) with any other environmental variables. Of the 11 land use related measures at various scales (Appendix G), only three were selected for use
in reference condition development (the number of oil and gas wells per watershed area, percent area harvested per watershed, and the percent human-related disturbance in each watershed (Table 4.2). Fifteen of the 19 variables retained are measures of water quality, many of which $(\mathrm{n}=13)$ exceeded established guidelines at some of the sites (Table 4.3).

The percent area harvested per watershed, habitat quality (assessment score), TSS, turbidity, and concentrations of nitrate, total nitrogen, total phosphorus, aluminum, iron, manganese, nickel, and barium were not highly correlated (r>0.7) with any other variables. Copper (ug/L), vanadium (ug/L), and cobalt (ug/L) were all highly correlated; vanadium and cobalt were removed as they both had a very low range, were consistent with natural background criteria, and were well below established guidelines. Copper was also removed from the model due to variations in testing accuracy between sites. Dissolved oxygen ( $\mathrm{mg} / \mathrm{L}$ ) was highly correlated with water temperature $\left({ }^{\circ} \mathrm{C}\right)$, selenium ( $\mathrm{ug} / \mathrm{L}$ ), and dissolved oxygen (\% SAT). Water temperatures were highly variable throughout the sampling period (June-October) due to significant fluctuations in air temperature. Water temperature was a significant predictor of dissolved oxygen concentrations $\left(\mathrm{R}^{2}=0.50, \mathrm{~F}(1,26)=26.04, \mathrm{p}<0.001\right)$; therefore, I removed $\mathrm{DO}(\mathrm{mg} / \mathrm{L})$ from the model to avoid seasonal fluctuations in water temperature confounding DO values. Specific conductivity and percent human-related disturbance in the watershed were highly correlated with each other, as well as many other water quality and land use variables. However, I chose to keep both as separate measures of disturbance since they were both highly correlated with many other, and distinct, environmental stressors.

Table 4. 1| Ecological characteristics of fish species collected in the Beaver River Watershed and used in IBI metric development.

| Common Name | Scientific Name | Thermal Regime | Trophic Guild | Forage <br> Habitat | Reproductive Guild | General <br> Tolerance | Total <br> Abundance | Adult Length (TL, cm) | Mean <br> Lengt <br> h <br> (cm) | Mean Weight (g) |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| brook | Culaea | CW | IN | GE | TR | MOD | 646 | 3.8-6.9 | 4.29 | 0.75 |
| stickleback | inconstans |  |  |  |  |  |  |  |  |  |
| burbot | Lota lota | C | IC | BE | LO | MOD | 63 | 38.1-83.8 | 14.46 | 35.13 |
| finescale dace | Chrosomus | C | OM | BE | PL | MOD | 524 | 5.0-8.3 | 4.58 | 0.9 |
|  | neogaeus |  |  |  |  |  |  |  |  |  |
| finescale dace x | Chrosomus | C | OM | BE | PL | MOD | 34* | MD | 4.69 | 0.93 |
| northern | neogaeus $x$ |  |  |  |  |  |  |  |  |  |
| redbelly dace | Chrosomus eos |  |  |  |  |  |  |  |  |  |
| fathead minnow | Pimephales promelas | W | OM | GE | TR | TOL | 1034 | 4.5-7.1 | 4.33 | 0.72 |
| iowa darter | Etheostoma exile | W | IN | BE | PL | INT | 30 | 4.6-6.8 | 5.54 | 0.72 |
| lake chub | Couesius plumbeus | C | IN | WC | LO | MOD | 244 | 12.7-18.3 | 6.7 | 3.43 |
| longnose dace | Rhinichthys cataractae | CW | IN | BE | LO | INT | 63 | 6.4-11.4 | 6.9 | 3.15 |
| longnose sucker | Catostomus catostomus | C | IN | BE | LO | MOD | 7 | 30.5-45.7 | 8.73 | 6.25 |
| logperch | Percina caprodes | W | IN | BE | LO | INT | 34 | 7.6-14.7 | 8 | 4.64 |
| northern | Chrosomus eos | C | OM | BE | PL | MOD | 33 | 3.1-6.8 | 4.66 | 0.97 |
| redbelly dace northern pearl dace | Margariscus margarita | C | IC | WC | LO | MOD | 95 | 6.5-12.0 | 7.49 | 4.06 |
| northern pike | Esox lucius | CW | CA | WC | PO | MOD | 49 | 45.7-100.2 | 34.69 | 17.75 |
| spottail shiner | Notropis hudsonius | W | IN | WC | LO | MOD | 6 | 5.8-12.7 | 3.85 | 0.49 |
| white sucker | Catostomus commersonii | CW | OM | BE | LO | TOL | 261 | 25.4-50.8 | 8.79 | 11.61 |
| yellow perch | Perca flavescens | CW | IC | WC | PL | MOD | 86 | 11.4-30.5 | 12.57 | 18.5 |

$\sim$ Sources: Eakins (2022); Prestie (unpublished, 2014); Cantin and John (2012); Bramblett et al. (2005); Stewart and Watkinson
$N_{\text {(2004); Joynt and Sullivan (2003); Nelson and Paetz (1992); Scott and Crossman (1973) }}$

Thermal Regime: $\mathrm{C}=$ cold water; $\mathrm{W}=$ warm water; $\mathrm{CW}=$ coolwater (inhabits both types)
Trophic Guild: CA = carnivore; $\mathrm{OM}=$ omnivore; $\mathrm{IN}=$ invertivore; $\mathrm{IC}=$ invertivore-carnivore
Forage Habitat: $\mathrm{BE}=$ benthic; $\mathrm{WC}=$ water column; $\mathrm{GE}=$ generalist
Reproductive Guild: PS = psammophil; $\mathrm{PO}=$ phytophil; $\mathrm{LO}=$ lithophil; $\mathrm{PL}=$ phytolithophil; $\mathrm{TR}=$ tolerant reproductive strategies
Tolerance: INT = intolerant; MOD = moderate; TOL = toelrant
*Abundance counts may not be accurate due to difficulty in identifying hybrids
MD: Missing Data

Table 4. 2| Final list of stressors ( $\mathrm{n}=19$ ) used to develop reference condition and site watershed stress classes for wadable streams and rivers in the Beaver River Watershed.

Stressor Category
Environmental Variable

## Habitat Indicators

Fish Habitat Habitat Assessment Score (\%)

## Land Use Indicators

Land Cover \% Human-Related Disturbance in Watershed
Natural Resource Extraction No. of Oil and Gas Wells Per Watershed Area (wells/km²) Forestry Operations \% Area Logged Per Watershed (1998-2018)

## Water Quality Indicators

| Nutrients | Benthic Chl-a $\left(\mathrm{mg} / \mathrm{m}^{2}\right)$ |
| :---: | :---: |
|  | Suspended Chl-a $(\mu \mathrm{g} / \mathrm{L})$ |
|  | Nitrate (mg/L N) |
|  | Total Nitrogen (mg/L) |
| Physical | Phosphate (mg/L P) |
|  | Total Phosphorus (mg/L) |
|  | SPC (us/cm at $\left.25^{\circ} \mathrm{C}\right)$ |
|  | TSS (mg/L) |
| Ions | Turbidity (FNU) |
| Metals | P. Alkalinity (mg/L) |
|  | Aluminum $(\mu \mathrm{g} / \mathrm{L})$ |
|  | Arsenic $(\mu \mathrm{g} / \mathrm{L})$ |
|  | Iron $(\mu \mathrm{g} / \mathrm{L})$ |
|  | Manganese $(\mu \mathrm{g} / \mathrm{L})$ |
|  | Mercury $(\mu \mathrm{g} / \mathrm{L})$ |
|  |  |

Table 4. 3| Water quality variables from the Beaver River watershed that exceeded established guidelines for the protection of aquatic life in Canada or other nearby regions. The data shown in this table only include those variables that have exceeded the existing criteria for the protection of aquatic life.

| Water Quality Variable |  | Established Criteria | Units | No. of Sites Not Meeting Criteria (/18) | Applicable Location | Citation |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Nutrients |  |  |  |  | Cold River SK; |  |
|  | Total Nitrogen | 0.453; <0.7 (oligotrophic) | $\mathrm{mg} / \mathrm{L}$ | 18; 13 | Canada | PPWB 2015; Government of Canada 2008 |
|  |  |  | $\mathrm{mg} / \mathrm{L}$ |  |  | PPWB 2015 \& SK Environmental Quality |
|  | Ammonia | 0.0156 | N | 18 | SK | Guidelines 2006 |
|  |  |  | $\mathrm{mg} / \mathrm{L}$ |  | Surface waters, | Environmental Quality Guidelines for |
|  | Phosphate | 0.015 | P | 16 | Alberta | Alberta Surface Waters 2014 |
|  | Total Phosphorus | 0.035 | $\mathrm{mg} / \mathrm{L}$ | 12 | SK northern sites | SK Environmental Quality Guidelines 2006 |
| Physical | Sestonic Chl a | <10 (oligotrophic) | $\mu \mathrm{g} / \mathrm{L}$ | 3 | Canada | Government of Canada 2008 |
|  | SPC | 150-500 acceptable | $\mu \mathrm{s} / \mathrm{cm}$ | 5 | US | USEPA 2012 |
|  | Dissolved Oxygen | $>5$ | $\mathrm{mg} / \mathrm{L}$ | 1 | Prairie Provinces | PPWB 2015 |
| Ions | *Sulfate | 250; Equation | $\mathrm{mg} / \mathrm{L}$ | 1 | Canada | Elphick et al. 2011 |
|  | Fluoride | 0.12 | $\mathrm{mg} / \mathrm{L}$ | 5 | Prairies; Canada | PPWB 2015 \& CCME 2002 |
| Metals | Arsenic | 5 | $\mu \mathrm{g} / \mathrm{L}$ | 1 | SK; Canada | SK Environmental Quality Guidelines 2006 \& CCME 1997 |
|  | Iron | 300 | $\mu \mathrm{g} / \mathrm{L}$ | 4 | Canada | CCME 1987 |
|  | Selenium | 1 | $\mu \mathrm{g} / \mathrm{L}$ | 6 | Canada | CCME 1987 |

[^1]
### 4.2.2 Setting Reference Criteria and Establishing Stress Classes

The qualitative and quantitative criteria used to develop reference sites (represented here as low stress) are shown in Table 3.4 of the methods section. Table 4.4 shows the final criteria chosen for the low, moderate, and high stress classes, the number of sites that fell in each stress class, the mean values $( \pm \mathrm{SD})$, and variable range for each of the 19 stressors selected for development of the watershed stress classes and the index. Using the median and IQR for the stress criteria development gave four sites in the low (reference) class, 11 sites in the moderate stress class, and three sites in the high stress group (Table 4.5).

Land use and habitat quality varied across the 18 sites sampled (Figure 4.1). Land use changed as expected between the low, moderate, and high stress sites. The percent natural landscape and fish habitat quality (as indicated by habitat assessment scores) decreased with increasing stress (Figure 4.1). Water quality data showed a slightly more convoluted relationship with the stressor gradient (Figures 4.2-4.4). Total suspended solids (TSS), specific conductivity (SPC), sestonic chlorophyll a, total nitrogen, phosphate, nitrate, p. alkalinity, and manganese concentrations all increased with increasing stress. Total phosphorus, benthic chlorophyll a, turbidity, aluminum, arsenic, iron, and mercury displayed non-monotonic relationships with increasing stress and had the highest concentrations in the moderate stress sites.

Although there is a distinction between the low and high stress groupings for the land use stressors assessed in this study, out of the 10 variables, only the number of oil and gas wells per watershed area showed a significant difference between the low and high stress groupings $(\mathrm{t}(5)=$ $-5.298, p=0.003$; Table 4.4). Similarly, mean fish habitat quality decreased by $\sim 43 \%$ between the low to high stress sites, but no significant difference was detected $(\mathrm{t}(2.022)=2.438, \mathrm{p}=$ 0.134; Table 4.4). I found a significant difference between the low and high stress groupings for 18 out of the 46 water quality variables assessed in this study, six of which were retained for use in developing the stress gradient (Appendix H).

### 4.2.3 Effect of Natural Gradients between Stress Classes

I tested for differences in various natural gradients, including drainage area, stream order, stream wetted width, reach area, and elevation, between the low, moderate, and high stress groupings to ensure the differences I observed cannot be attributed to natural variations on the landscape. I
found no significant differences for any of the variables between any of the stress classes (twosample independent t -test, $\mathrm{p}>0.05$ ).

Table 4. $4 \mid$ Stressors ( $n=19$ ) used in the development of reference conditions (represented here as low stress) and watershed stress classes for the Index of Biotic Integrity (IBI). The stress criteria used to establish stress class thresholds, the initial number of sites (out of 18) within each threshold, and recommended guidelines for comparison are also given. The mean stressor values and range prior to the development of the low (reference), moderate, and high stress classes are also shown. Water quality variables that exceeded established guidelines for the protection of aquatic life at some of the sites are given in bold.

| Stressor | Stress Criteria Used (\# of Sites) |  |  | Mean Value ( $\pm 2 \mathrm{SD}$ ) |  |  | Range | Recommended Guidelines | Citation |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Low | Moderate | High | Low | Moderate | High |  |  |  |
| Habitat Assessment Score (\%) | $\geq 94$ (6) | 77-94 (7) | $\leq 77$ (5) | $\begin{gathered} 94.1 \\ ( \pm 3.6) \end{gathered}$ | $\begin{gathered} 82.5 \\ ( \pm 17.9) \end{gathered}$ | $\begin{gathered} 58.0 \\ ( \pm 26.2) \end{gathered}$ | 36.0-98.3 | NA | NA |
| Human-Related |  |  |  |  |  |  |  |  |  |
| Disturbance in Watershed (\%) | $\leq 0.2$ (5) | 0.2-13.2 (8) | $\geq 13.2$ (5) | $\begin{gathered} 0.21 \\ ( \pm 0.15) \end{gathered}$ | $\begin{gathered} 13.5 \\ ( \pm 23.0) \end{gathered}$ | $\begin{gathered} 20.7 \\ ( \pm 24.2) \end{gathered}$ | 0-71.4 | *NA | NA |
| No. of Oil and Gas |  |  |  |  |  |  |  |  |  |
| Wells Per Watershed Area | $\leq 0$ (8) | $\underset{(5)}{>0-0.015}$ | $\begin{gathered} \geq 0.0151 \\ (5) \end{gathered}$ | $0( \pm 0)$ | $\begin{gathered} 0.01 \\ ( \pm 0.03) \end{gathered}$ | $\begin{gathered} 0.03 \\ ( \pm 0.01) \end{gathered}$ | 0-0.07 | *NA | NA |
| Area Harvested Per |  |  |  | 8.2 | 10.1 | 11.8 |  |  |  |
| Watershed (\%) | $\leq 2.6$ (5) | 2.6-17.2 (8) | $\geq 17.2$ (5) | $( \pm 7.0)$ | $( \pm 8.1)$ | ( $\pm 9.6$ ) | 0-21.8 | NA | NA |


| Total Nitrogen (mg/L) | $\leq 0.7$ (5) | 0.7-1.5 (8) | $\geq 1.7$ (5) | $\begin{gathered} 0.74 \\ ( \pm 0.26) \end{gathered}$ | $\begin{gathered} 1.28 \\ ( \pm 0.76) \end{gathered}$ | $\begin{gathered} 1.29 \\ ( \pm 0.32) \end{gathered}$ | 0.5-3.2 | River), $1.14 \mathrm{mg} / \mathrm{L}$ (Beaver River) | $\begin{gathered} \text { 2008, PPWB, } \\ 2015 \end{gathered}$ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  |  |  |  |  |  |  | $<0.025$ <br> (oligotrophic), | Government of Canada, |
| Total Phosphorus (mg/L) | $\begin{gathered} \leq 0.03 \\ (5) \end{gathered}$ | $0.03-0.10$ <br> (7) | $\geq 0.10$ (6) | $\begin{gathered} 0.058 \\ ( \pm 0.037) \end{gathered}$ | $\begin{gathered} 0.095 \\ ( \pm 0.126) \end{gathered}$ | $\begin{gathered} 0.061 \\ ( \pm 0.051) \end{gathered}$ | 0.01-0.45 | $0.025-0.075$ (mesotrophic), | $\begin{gathered} \text { 2008, PPWB, } \\ 2015 \end{gathered}$ |


|  |  |  |  |  |  |  |  | 0.023 (Cold River), 0.171 (Beaver River), 0.035 (SK northern sites) |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Nitrate ( $\mathrm{mg} / \mathrm{L} \mathrm{N}$ ) | $\begin{gathered} \leq 0.04 \\ \text { (6) } \end{gathered}$ | $0.04-0.08$ <br> (6) | $\geq 0.08$ (6) | $\begin{gathered} 0.057 \\ ( \pm 0.008) \end{gathered}$ | $\begin{gathered} 0.062 \\ ( \pm 0.040) \end{gathered}$ | $\begin{gathered} 0.074 \\ ( \pm 0.070) \end{gathered}$ | 0.01-0.15 | $\begin{gathered} 2.9 \text { (Canada), } 3 \\ (\mathrm{SK}) \end{gathered}$ | Government of Canada, 2008, PPWB, 2015 <br> Government |
| Phosphate (mg/L P) | $\begin{gathered} \leq 0.04 \\ (5) \end{gathered}$ | $0.04-0.08$ <br> (8) | $\geq 0.08$ (5) | $\begin{gathered} 0.04 \\ ( \pm 0.03) \end{gathered}$ | $\begin{gathered} 0.07 \\ ( \pm 0.05) \end{gathered}$ | $\begin{gathered} 0.10 \\ ( \pm 0.06) \end{gathered}$ | 0.003-0.19 | 0.015 (AB) | of Alberta, 2018 |
| Planktonic Chl-a ( $\mu \mathrm{g} / \mathrm{L}$ ) | $\leq 1.2$ (5) | 1.2-3.0 (8) | $\geq 3.0$ (5) | $\begin{gathered} 1.3 \\ ( \pm 1.7) \end{gathered}$ | 2.6 ( $\pm 2.6)$ | $\begin{gathered} 14.4 \\ ( \pm 13.6) \end{gathered}$ | 0.15-28.4 | $\begin{gathered} <10 \text { (oligotrophic), } \\ 10-30 \\ \text { (mesotrophic) } \end{gathered}$ | Government of Canada, 2008 |
| Benthic Chl-a ( $\mathrm{mg} / \mathrm{m}^{2}$ ) | <0.8 (5) | 0.8-4.6 (8) | $\geq 4.6$ (5) | $\begin{gathered} 1.5 \\ ( \pm 1.2) \end{gathered}$ | $\begin{gathered} 3.9 \\ ( \pm 3.4) \end{gathered}$ | $\begin{gathered} 2.7 \\ ( \pm 2.3) \end{gathered}$ | 0.31-11.2 | $<20$ (oligotrophic), $20-70$ <br> (mesotrophic) | Government of Canada, 2008 |
|  |  |  |  | 2.7 | 12.5 | 9.8 |  | max increase of 8 (acute), 2 (chronic) from background levels, max 10 | CCME, 1999, |
| Turbidity (FNU) | $\leq 2.9$ (5) | 2.9-9.5 (8) | $\geq 9.5$ (5) | $( \pm 1.8)$ | ( $\pm 14.5$ ) | $( \pm 3.6)$ | 0.5-43.2 | NTU <br> increase of 25 (acute), 5 (chronic) (Canada); 3-48.8 (Beaver River), | ECCC, 2020 |
| TSS (mg/L) | $\leq 1.2$ (5) | 1.2-5.6 (8) | $\geq 5.6$ (5) | $\begin{gathered} 1.8 \\ ( \pm 2.2) \end{gathered}$ | 5.3 ( $\pm 5.6$ ) | $\begin{gathered} 12.0 \\ ( \pm 5.1) \end{gathered}$ | 0.2-16.5 | 1.2-4.8 (Cold River) | CCME, 1999, PPWB, 2015 |
| SPC ( $\mu \mathrm{S} / \mathrm{cm}$ at | $\leq 327.5$ | 327.5-490.5 | $\geq 490.5$ | 288.1 | 530.8 | 536.7 |  |  |  |
| $\left.\mathbf{2 5}^{\circ} \mathrm{C}\right)$ | (5) | (8) | (5) |  |  | ( $\pm 80.9$ ) | 207-1460 | 150-500 (USA) | USEPA |
| P. alkalinity (mg/L) | $\leq 6$ (12) | 6-10 (2) | $\geq 10$ (4) | $\begin{gathered} 2.2 \\ ( \pm 1.5) \end{gathered}$ | 4.4 ( $\pm 4.1$ ) | $\begin{gathered} 8.0 \\ ( \pm 6.6) \end{gathered}$ | 1-14 | NA | NA |


| Aluminum ( $\mu \mathrm{g} / \mathrm{L}$ ) | $\leq 1.8$ (5) | 1.8-4.1 (8) | $\geq 4.1$ (5) | $\begin{gathered} 4.1 \\ ( \pm 3.5) \end{gathered}$ | $4.2( \pm 5.3)$ | $\begin{gathered} 3.1 \\ ( \pm 2.2) \end{gathered}$ | 0.9-20 | $\begin{gathered} 5(\mathrm{pH}<6.5), 100 \\ (\mathrm{pH} \geq 6.5) \end{gathered}$ | 99 |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Arsenic ( $\mu \mathrm{g} / \mathrm{L}$ ) | $\leq 0.7$ (5) | 0.7-1.8 (8) | $\geq 1.8$ (5) | 0.6 |  | 1.4 |  | 5 | CCME, 1999, |
|  |  |  |  | $( \pm 0.2)$ | 2.3 ( $\pm 2.2)$ | $( \pm 0.5)$ | 0.4-7.9 |  | PPWB, 2015 |
|  |  |  |  | 190.1 | 267.8 | 177.3 |  |  |  |
| Iron ( $\mu \mathrm{g} / \mathrm{L}$ ) | $\leq 92$ (6) | 92-270.8 (7) | $\geq 270.8$ (5) | ( $\pm 248.9)$ | ( $\pm 271.8$ ) | ( $\pm 61.3$ ) | 7.9-950 | 300 | CCME, 1999 |
|  |  |  |  |  |  |  |  | use equation (for hardness of 25-670 |  |
|  | $\leq 24.5$ | 24.5-92.8 |  | 35.2 | 58.1 | 131.7 |  | $\mathrm{mg} / \mathrm{L}$ and pH 5.8 - |  |
| Manganese ( $\mu \mathrm{g} / \mathrm{L}$ ) | (5) | (8) | $\geq 92.8$ (5) | $( \pm 47.9)$ | ( $\pm 37.8$ ) | ( $\pm 69.5$ ) | 4.9-200 | 8.4) | CCME, 1999 |
|  | $\leq 0.003$ | 0.003-0.008 |  | 0.005 | 0.005 | 0.002 |  |  |  |
| Mercury ( $\mu \mathrm{g} / \mathrm{L}$ ) | (8) | (6) | $\geq 0.008$ (4) | $( \pm 0.004)$ | $( \pm 0.004)$ | $( \pm 0.001)$ | 0.001-0.012 | 0.026 | CCME, 1999 |

NA (not applicable) is used where recommended quality guidelines did not exist for a nearby region.
PPWB: Prairie Provinces Water Board
CCME: Canadian Council of Ministers of the Environment
ECCC: Environment \& Climate Change Canada
AB: Alberta
SK: Saskatchewan
*See Phillips et al., 2023 for recommended guidelines in neighbouring southern ecoregions.

Table 4. $5 \mid$ Established site watershed stress classes (low, moderate, and high stress) for the 18 streams and rivers sampled in the Beaver River Watershed.

Watershed Stress

| Low | Moderate | High |
| :---: | :---: | :---: |
| Alcott Ck | DelaRonde Ck | Backwater Ck |
| Flotten R | Dennis Ck | Makwa Ck |
| Landry Ck | Goodsoil Ck | Mistohay Ck |
| Nesslin Ck | Nolin Ck |  |
|  | Otter Ck |  |
|  | Pagan Ck |  |
|  | Robin Ck |  |
|  | Sulby Ck |  |
|  | Sukaw Ck |  |
|  | Spiritwood Ck |  |
|  | Tea Ck |  |



Figure 4. 1| Variations in land use and habitat assessment scores after the final classification of sites into the respective stress classes (low, moderate, and high) for the 18 streams and rivers sampled in the Beaver River watershed. Only the variables retained for use in the stress classification and IBI development are shown. Land use was calculated for the upstream catchment area contributing to each site. The open circle is an indication that an outlier is present in the data. The asterisk (*) is an indication that an extreme outlier is present in the data. Refer to methods section 2.3.8.2 Environmental Metric Development and Selection for further description of land use and habitat metrics and their development.


Figure 4. 2| Variations in physicochemical and ion (p. alkalinity) concentrations after the final classification of sites into the respective stress classes (low, moderate, and high) for the 18 streams and rivers sampled in the Beaver River watershed. Only the variables retained for use in the stress classification and IBI development are shown. The open circle is an indication that an outlier is present in the data. The asterisk $\left(^{*}\right)$ is an indication that an extreme outlier is present in the data. Refer to methods section 2.3.8.2 Environmental Metric Development and Selection for further description of water quality metric development.


Figure 4. 3| Variations in nutrient data after the final classification of sites into the respective stress classes (low, moderate, and high) for the 18 streams and rivers sampled in the Beaver River watershed. Only the variables retained for use in the stress classification and IBI development are shown. The open circle is an indication that an outlier is present in the data. The asterisk (*) is an indication that an extreme outlier is present in the data. Refer to methods section 2.3.8.2 Environmental Metric Development and Selection for further description of water quality metric development.


Figure 4. 4| Variations in metal concentrations after the final classification of sites into the respective stress classes (low, moderate, and high) for the 18 streams and rivers sampled in the Beaver River watershed. Only the variables retained for use in the stress classification and IBI development are shown. The open circle is an indication that an outlier is present in the data. The asterisk (*) is an indication that an extreme outlier is present in the data. Refer to methods section 2.3.8.2 Environmental Metric Development and Selection for further description of water quality metric development.

### 4.3 The Index of Biotic Integrity

### 4.3.1 Metric Selection and Calibration

Initially, greater than 100 metrics were compiled from the literature. From this list, 42 metrics were identified that better represented the fish assemblage, the biogeographic area, the water body type, and local stressors (see Appendix I).

### 4.3.1.1 Metric Range

Six metrics had a narrow range (<4) (number of water column species, number of subterminal mouth minnow species, number of insectivorous cyprinid species, number of tolerant species, number of benthic invertivorous species, and number of sensitive (or intolerant) species). Seventeen of the 42 metrics that were assessed had zero values at approximately a third or greater sites (number of cyprinid species excluding tolerants, percent Cyprinidae species excluding tolerants, percent of tolerant reproductive guild, percent generalists, percent individuals with DELTs, number of insectivorous cyprinid species, percent insectivorous cyprinids, percent white sucker, percent brook stickleback, percent fathead minnow, condition of sentinel species, percent of individuals with parasites, number of benthic invertivorous species, percent of benthic invertivorous fish, percent of benthic invertivorous species, number of sensitive (or intolerant) species, and percent intolerant individuals). Following Stoddard et al.'s (2008) methods, metrics were initially eliminated if their range was $<4$ or if $>1 / 3$ of samples had values $=0$. I found it difficult to find enough metrics with $<1 / 3$ of the values $=0$; therefore, the threshold for removal was any metric with $>1 / 2$ values $=0$ (see Section 2.5.3.2 for further details). Similarly, the threshold for too small of a range was set to $<3$.

### 4.3.1.2. Metric Reproducibility

I found that the fish assemblage being assessed had overall very low $\mathrm{S} / \mathrm{N}$ ratios (range $=0.15$ to 2.68; mean $\pm \mathrm{SD}=0.81 \pm 0.64$ ) and thus, the threshold for rejection had to be lower than suggested elsewhere. The top five metrics with the highest $\mathrm{S} / \mathrm{N}$ ratios, and therefore the best stability through time, were the number of sensitive (or intolerant) species ( $\mathrm{S} / \mathrm{N}=2.68$ ), the number of insectivorous cyprinid species ( $\mathrm{S} / \mathrm{N}=2.65$ ), the number of cyprinid species ( $\mathrm{S} / \mathrm{N}=$ 2.19), the number of cyprinid species excluding tolerants ( $\mathrm{S} / \mathrm{N}=1.91$ ), and species richness ( $\mathrm{S} / \mathrm{N}$ $=1.63$ ). The metrics with the lowest $\mathrm{S} / \mathrm{N}$ ratios, and therefore the highest variability through
time, were the percent intolerant fish ( $\mathrm{S} / \mathrm{N}=0.15$ ), the percent of benthic invertivorous fish ( $\mathrm{S} / \mathrm{N}$ $=0.15)$, the percent benthivores excluding white sucker $(\mathrm{S} / \mathrm{N}=0.18)$, the relative abundance of coldwater fish $(\mathrm{S} / \mathrm{N}=0.21)$, and the percent individuals with DELTs $(\mathrm{S} / \mathrm{N}=0.25)$.

Initially, I chose a cut-off of $\mathrm{S} / \mathrm{N}<1$ below which to eliminate metrics with too high variability through time to discriminate among sites of different conditions. This left 14 of the 42 metrics (number of sensitive (or intolerant) species, number of benthic invertivorous species, condition of sentinel species, percent brook stickleback, number of insectivorous cyprinid species, relative abundance of coolwater fish, species richness, number of benthic species, number of cyprinids and catastomid species excluding fathead minnow, number of cyprinid species, number of benthic and water column fish species, number of water column species, percent of water column fishes, and number of cyprinid species excluding tolerants), most of which were highly correlated to one another. After selecting a lower threshold of $\mathrm{S} / \mathrm{N}<0.5$ to eliminate metrics, I was still only able to obtain very few metrics that were non-redundant for the final index.

### 4.3.1.3 Metric Redundancy

Majority of the metrics were highly correlated, and even after restricting the removal criteria to $r$ $\geq 0.8$, I was only able to obtain at most three metrics that were not highly correlated. Most metrics were positively correlated with all other metrics except the percent coolwater fish, percent coldwater/coolwater fish, percent top carnivores and piscivores, percent of water column fishes, and the $\log _{10}$ condition of sentinel species (Table 4.6). These five metrics tended to have negative correlations with all other metrics. The $\log _{10}$ condition of sentinel species metric was excluded from the correlation analysis as the sentinel species, white sucker, was only found in two of the low stress/reference sites. To obtain enough metrics for the final index, this step, along with the metric range and reproducibility steps, were given less importance in the metric selection process.

Table 4. 6| Pearson correlation coefficients ( r ) of candidate fish metrics $(\mathrm{N}=42)$ for use in the Index of Biotic Integrity. Pearson correlation coefficients $\geq 0.7$ were used as a cut-off for removal. The heat map represents correlations ranging from dark green (positive 1 ) to dark red (negative 1). Variables that have correlations close to 0 are indicated by white or light green and red colouration. Metrics are represented by numbers, ranging from 1-42. The list of metrics corresponding to the numbers is given below the table. The metrics retained for use in the IBI are given in bold.








 Generalists, and 42: $\log _{10}$ Condition of Sentinel Species

### 4.3.1.4 Metric Responsiveness

It was difficult to obtain enough metrics to create an ecologically balanced index following all the Stoddard et al. (2008) metric selection and calibration steps. Even after relaxing criteria for the metric range, metric reproducibility, and metric redundancy steps I was only able to achieve a maximum of three nonredundant metrics that did not necessarily have high metric responsiveness. To increase the number of metrics, I decided to favour metric responsiveness over all other selection criteria. It has been suggested that metric responsiveness holds a higher importance in determining metric success and discrimination ability between sites of different condition in an index (Barbour et al., 1999; Stoddard et al., 2008). None of the raw metrics assessed had a significant difference between the low and high stress classes (independent samples t-test, $p>0.05$ ). However, I still chose to sort metrics by their responsiveness to the stressor gradient. I chose to assess the top 20 most responsive metrics (highest t-scores) and select a metric from each of the representative metric categories. Metrics were selected based on responsiveness to stressors while still trying to maximize the independence among metrics and the diversity of metric types (i.e., selecting from different metric categories). Priority was placed on selecting metrics with the highest t scores until all metric categories were full. Metric range, reproducibility, and redundancy were also considered in this selection process but given less priority. For example, when two or more metrics were highly correlated, I selected the one with the highest t -score to be representative of the metric category. Additionally, when metrics shared similar t-scores, the one with the largest range, the lowest number of zero values, and the highest $\mathrm{S} / \mathrm{N}$ ratio would be chosen. Considering these additional selection criteria helped to remove any potential poorly functioning metrics. From this I was able to create an index with nine metrics that showed the greatest responsiveness to the established stress gradient (Table 4.7).

### 4.3.1.5 Variability with Natural Gradients

Simple linear regression and correlation analyses were used to test if stream order, upstream drainage area, or wetted stream width significantly predicted the fish metric values retained for use in the IBI ( $\mathrm{n}=9$ ), as well as additional measures of relative abundance and species richness. Plots of the data points and calculated correlation coefficients were also considered. Despite the low number of reference sites in this study, to avoid misinterpreting potential anthropogenic influence on fish communities, only data from low stress sites ( $n=4$ ) was used for this
assessment. Stream order and $\log _{10}$ average wetted stream width had the highest and strongest correlations with fish metrics ( $\mathrm{r} \geq 0.4$ ), followed by $\log _{10}$ upstream drainage area. The $\log _{10}$ relative abundance metric showed no correlations greater than 0.4 with any of the stream sizerelated variables; however, the $\log _{10}$ relative abundance excluding tolerants metric, the $\log _{10}$ relative abundance of coldwater fish metric, and the $\log _{10}$ relative abundance of coolwater fish metric all had correlations $>0.4$ with stream order. Species richness showed a strong relationship with wetted width $(r=0.71)$, but the correlations with stream order $(r=0.02)$ and drainage area $(\mathrm{r}=0.22)$ were $<0.4$.

As expected for metrics of stream size, stream order was highly correlated with wetted width $(r=0.81)$ and upstream drainage area $(r=0.82)$, and upstream drainage area was also highly correlated with wetted stream width $(\mathrm{m})(\mathrm{r}=0.77)$; since all three natural gradients were highly correlated, I chose one to be representative of the natural gradient with stream size. I decided to remove the upstream drainage area as it was the least correlated with any of the potential metrics and requires access to ArcGIS or similar software for calculation (see Godwin \& Martin 1975 for further details outlining issues surrounding calculating the upstream catchment area in the prairie provinces region). Stream order is easy to calculate with limited resources, is relatively unaffected by human disturbance, and has been used in a similar region and waterbody type (SK Prairie and Boreal Plain ecozones wadable rivers and streams) in the development of a multivariate tool for assessing lotic ecosystem health using benthic macroinvertebrates (Phillips et al., 2023). Stream order and wetted width were both highly correlated with a similar number of metrics; however, regression analysis revealed that the average wetted stream width had a much stronger relationship with the most responsive metrics in this study. Like stream order, the wetted stream width is easy to obtain but does require data collection in the field, rather than merely using a 1:50,000 topographic map (Strahler, 1957). I chose to use the wetted stream width to be representative of the natural gradient in stream size for the purposes of this study since the data is easy to obtain and has been shown to have a strong relationship predicting fish assemblage structure and species richness elsewhere (Merkowsky, 1998; Zhu et al., 2016).

When only looking at the most responsive metrics' $(\mathrm{n}=9)$ relationships with the average wetted stream width (Figure 4.5C), three metrics had a coefficient of determination ( $\mathrm{R}^{2}$ ) value greater than 0.4 ; however, only one relationship was significant $\left(\log _{10}\right.$ percent individuals with
parasites $\left(R^{2}=0.92, \mathrm{p}<0.05\right), \log _{10}$ percent white sucker $\left(\mathrm{R}^{2}=0.78, \mathrm{p}>0.05\right)$, and the number of water column species $\left(\mathrm{R}^{2}=0.79, \mathrm{p}>0.05\right)$. None of the additional abundance-related metrics or fish species richness metrics were significantly predicted by stream wetted width ( $\mathrm{p}>0.05$, Figure 4.5A\&B).

Since wetted stream width was associated with greater than $40 \%$ of variation in metric values at my reference sites, I used the linear regression equation to calculate residual values (e.g., observed value - predicted value). These residual values represent a measure of naturalgradient corrected metric variability after adjusting for stream size (Vander Laan \& Hawking, 2014) and were used in metric scoring and index development. Due to the low number of reference/low stress sites in this study, I considered the relationship between fish species richness and stream wetted width found by Merkowsky (1998) for a much larger sample size of SK Boreal Plains streams and rivers to adjust my metric scores. However, despite the low number of reference sites in this study, I decided to use my sites to assess the relationship between fish metrics and stream size since the relationship found by Merkowsky was not a great fit for my data (Figure 4.5A).


Figure 4. 5| Fish species richness (A), $\log _{10}$ fish relative abundance (B) and the retained IBI metrics' (C) variability with stream size (represented by wetted stream width (m)) for low stress, reference sites. The best fit line for the equation of the regression model $\left(=-0.157+3.618\left(\log _{10}\right.\right.$ wet width $)$ ) used to predict fish species richness for Boreal Plains streams and rivers in Merkowsky (1998) is given in figure A for comparison.

### 4.3.2 Metric Scoring and Index Development

### 4.3.2.1 Metric Scores

Metric scores generally followed the expected trend with increasing stress (e.g., high stress equals lower metric scores for positive scoring metrics) (Figure 4.6, Table 4.8). Of the nine metrics assessed, the $\log _{10}$ relative abundance of coldwater fish metric had the lowest mean score across all sites $(1.6 \pm 2.6)$, while the $\log _{10}$ percent fish with parasites metric had the highest mean score $(7.8 \pm 3.5)$. I found the percent top carnivores and piscivores metric had the highest variability around the mean $(3.4 \pm 4.1)$, while the $\log _{10}$ relative abundance of coldwater fish had the lowest variability $(1.6 \pm 2.6)$ (Table 4.8). The highest mean metric score found in any site was 7.3 ( $\pm 3.4$, range of 0.3-10; Landry Creek). Landry Creek has the smallest drainage area of all sites sampled (stream order 2, drainage area of $53 \mathrm{~km}^{2}$ ) and is surrounded by young intact mixedwood forest, and has high instream canopy coverage, relatively low aquatic vegetation replaced by attached algae and moss, large amounts of woody debris, large cobble/boulder substrate, and high bank erosion characteristic of low order, higher elevation streams. The lowest mean metric score was 2.3 ( $\pm 4.3$, range of 0-10; Tea Creek). Tea Creek is a relatively small stream (stream order 3, drainage area of $103 \mathrm{~km}^{2}$ ) surrounded by young mixed wood forest with patchy logging activity and is characterized by low instream canopy coverage and woody debris, moderate aquatic vegetation coverage, finer substrate with some cobble and gravel, and low watershed erosion. Thirteen of the 18 sites had large dispersion of metric scores (Flotten Creek, Landry Creek, Nesslin Creek, DeLaRonde Creek, Dennis Creek, Nolin Creek 2017, Otter Creek, Robinson Creek, Sulby Creek, Sukaw Creek, Tea Creek, Backwater Creek, and Mistohay Creek) ranging from a minimum metric score of 0 to a high of 10 . The minimum variation in metric scores was found in Alcott Creek ( $\mathrm{SD}=2.5$ ) while Nesslin Creek had the highest variation $(\mathrm{SD}=$ 5.0). Once scored, none of the nine metrics were significantly different between the low and high stress classes (independent samples $t$-test, $\mathrm{p}>0.05$ ). The percent top carnivores metric had a significantly different mean between the low and moderate stress groupings $(\mathrm{t}(13)=2.262, \mathrm{p}<$ 0.05 ) while the $\log _{10}$ percent of individuals with parasites metric had significantly different means between the moderate and high stress groupings $(\mathrm{t}(10.04)=-2.497, \mathrm{p}<0.05)$.

I used Levene's Test for equality of variance to assess potential differences in variability between the low, moderate, and high stress groupings. Out of the nine metrics used to develop the IBI, only two metrics, the $\log _{10}$ percent white sucker $(\mathrm{F}=12.961, \mathrm{p}<0.05)$ and the percent
brook stickleback ( $\mathrm{F}=8.676, \mathrm{p}<0.05$ ), had significantly different variances between the low $(\mathrm{n}=4)$ and high $(\mathrm{n}=3)$ stress sites ( $9.0 \pm 1.1$ vs $6.4 \pm 5.6$ and $9.1 \pm 1.8$ vs $6.7 \pm 5.8$, respectively). All metrics shared similar variance $(\mathrm{p}>0.05)$ between the low and moderate stress groupings. The $\log _{10}$ percent of individuals with parasites metric had significantly different variances between the moderate $(\mathrm{n}=11,6.9 \pm 4.1)$ and high $(\mathrm{n}=3,9.9 \pm 0.10)$ stress groupings $(\mathrm{F}=7.653, \mathrm{p}<0.05)$.

Table 4. 7| Metrics $(\mathrm{n}=9)$ used in the development of the Index of Biotic Integrity. Mean raw metric values $( \pm$ SD $)$ are given for each stress class as well as metric range.

| Metric | Low Stress | Mean Value $( \pm$ SD $)$ <br> Moderate Stress | High Stress | Range |
| :---: | :---: | :---: | :---: | :---: |
| Positive Scoring (decrease with stress) |  |  |  |  |
| Relative Abundance of Coldwater Fish | $0.85( \pm 1.28)$ | $0.92( \pm 1.71)$ | $0.13( \pm 0.18)$ | $0-5.4 \mathrm{fish} / 100 \mathrm{~s}$ |
| Percent Top Carnivores and Piscivores | $68.07( \pm 38.14)$ | $19.32( \pm 31.46)$ | $33.33( \pm 57.74)$ | $0-100 \%$ |
| Percent Invertivores | $48.41( \pm 33.89)$ | $39.63( \pm 29.66)$ | $80.09( \pm 26.51)$ | $0-100 \%$ |
| Percent Benthivores Excluding White Sucker | $25.11( \pm 21.02)$ | $23.34( \pm 22.65)$ | $8.33( \pm 14.43)$ | $0-59.91 \%$ |
| Number of Water Column Species | $1.83( \pm 1.11)$ | $0.94( \pm 0.83)$ | $1.0( \pm 0.0)$ | $0-4$ species |
| Negative Scoring (increase with stress) |  |  |  |  |
| Percent of Individuals with Parasites | $2.38( \pm 4.35)$ | $7.36( \pm 9.08)$ | $0.0( \pm 0.0)$ | $0-24 \%$ |
| Percent Coolwater Fish | $53.42( \pm 32.22)$ | $46.65( \pm 33.34)$ | $83.60( \pm 22.21)$ | $0-100 \%$ |
| Percent White Sucker | $1.11( \pm 1.39)$ | $15.47( \pm 28.85)$ | $14.72( \pm 23.37)$ | $0-100 \%$ |
| Percent Brook Stickleback | $7.83( \pm 15.55)$ | $16.33( \pm 22.47)$ | $29.99( \pm 51.94)$ | $0-89.97 \%$ |

Table 4. 8| Mean metric and index scores ( $\pm$ SD) for each stress class as well as the range of scores for the nine metrics retained in the IBI. Scores are scaled to range between $0-10$ for metrics and $0-100$ for the IBI. Negative scoring metrics were corrected to be on a positive scale (e.g., a score of 2 is of lower quality compared to a score of 8 ).

|  | Mean Score $( \pm$ SD $)[$ Range $]$ <br> Metric |  |  |  |  | Low Stress |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |


| Percent Brook Stickleback | $9.13( \pm 1.75)$ | $[6.5-10]$ | $7.36( \pm 3.82)$ | $[0-10]$ | $6.67( \pm 5.77)$ | $[0-10]$ | $0-10$ |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Index of Biotic Integrity | $66.3( \pm 11.9)$ | $[49.7-76.5]$ | $48.5( \pm 13.1)$ | $[23.2-67.9]$ | $49.3( \pm 9.6)$ | $[40.7-59.7]$ | $23-77 \%$ |



Figure 4. 6| Shifts in fish metric scores with the stressor gradient (low, moderate, and high stress). Negative scoring metrics (metrics that increase in response to human disturbance) are corrected so that a lower score (closer to 0 ) represents lower quality and higher scores (closer to 10) represent higher quality. Metric abbreviations are as follows: $\log _{10}$ RelAbundCold = relative abundance of coldwater fish, PercentTopCarn = percent of top carnivore and piscivore fish, PercentInvertivores = percent of invertivorous fish, $\log _{10}$ PercentBenthivores-exclWHSC $=\log _{10}$ percent of benthivorous fish excluding white sucker, WaterColumnSpp = number of water column species, $\log _{10}$ PercentParasites = percent of individual fish
with parasites, PercentCoolwater $=$ percent coolwater fish, $\log _{10}$ PercentWhiteSucker $=\log _{10}$ percent of fish that are white sucker, and PercentBrookStickleback = percent of fish that are brook stickleback. The open circle is an indication that an outlier is present in the data. The asterisk $(*)$ is an indication that an extreme outlier is present in the data. The larger bold asterisk indicates a significant difference.

### 4.3.2.2 Index of Biotic Integrity Scores

Index scores ranged between a low of 23.2 (Tea Creek) to a maximum value of 76.5 (Alcott Creek). I calculated a mean index score of 52.6 ( $\pm 13.9$ ). The index scores were highest in the low stress sites $(66.3 \pm 11.9)$, lower in the moderate stress sites $(48.5 \pm 13.1)$ and showed a slight increase in the sites demonstrating high stress (49.3 $\pm 9.6$ ) (Figure 4.7, Table 4.8). A two-sample independent $t$-test was used to test if fish community structure and condition (represented by IBI scores) were significantly different between the low and high stress, low and moderate stress, and the moderate and high stress streams and rivers. No significant differences were found between the low $(\mathrm{n}=4)$ and high $(\mathrm{n}=3)$ stress sites $(\mathrm{t}(5)=2.01, \mathrm{p}>0.05)($ Cohen's $\mathrm{D} 95 \% \mathrm{CI}=-$ $0.28,3.24)$ and the moderate and high stress sites $(\mathrm{t}(12)=-0.096, \mathrm{p}>0.05)$ (Cohen's $\mathrm{D} 95 \% \mathrm{CI}=-$ $0.06,-1.34)$. There was a significant difference between the low and moderate ( $\mathrm{n}=11$ ) stress sites $(\mathrm{t}(13)=2.369, \mathrm{p}<0.05)$ (Cohen's D $95 \% \mathrm{CI}=1.38,0.10$ ). Levene's Test indicated that IBI scores showed an equal variance between the low and high ( $\mathrm{F}=0.133$, $\mathrm{p}>0.05$ ), the low and moderate ( $\mathrm{F}=0.118, \mathrm{p}>0.05$ ), and the moderate and high $(\mathrm{F}=0.440, \mathrm{p}>0.05)$ stress groupings.

To further assess fish condition, I compared raw values for three condition-related metrics 1) the percent of fish with parasites, 2) the percent of fish with abnormalities, and 3) the body condition of white sucker between the low and high stress streams and rivers. The condition (Fulton's $K$ ) of a sentinel species (white sucker in this study) is a commonly-used condition endpoint (Environment Canada 2010, 2012a). I found no significant difference between any of the low and high stress sites for either of the three condition-related metrics (twosample independent t -test, $\mathrm{p}>0.05$ ) and, although the $\log _{10}$ percent of fish with parasites and the $\log 10$ precent of fish with abnormalities metrics show an increasing trend from low to moderate stress (Figure 4.8), no significant difference were found between the low and moderate stress sites (two-sample independent t -test, $\mathrm{p}>0.05$ ). However, I did find a significant difference between the moderate and high stress sites for the $\log _{10}$ percent of individuals with parasites metric $(\mathrm{t}(10.04)=-2.497, \mathrm{p}<0.05)$, but not the other two condition-related metrics.


Figure 4. 7| Index of Biotic Integrity scores for the low, moderate, and high stress streams and rivers sampled in 2017 from the Beaver River watershed. The asterisk (*) indicates a significant difference.


Figure $4.8 \mid$ Variability in the condition of white sucker (C. commersonii) (A), percent of individuals in the fish community with abnormalities (disease, lesions, tumors, parasites, etc.) (B), and percent of individuals in the fish community with parasites (C) for the low, moderate,
and high stress streams and rivers in the Beaver River watershed. The open circle is an indication that an outlier is present in the data. The asterisk (*) indicates a significant difference.

### 4.4 Interannual Variability in Environmental Data and the IBI

I used repeated measures ANOVAs to make between year comparisons of fish habitat quality, including habitat assessment scores and various water chemistry parameters used in the development of reference/low stress conditions and the human disturbance gradient. I also assessed fish species richness, fish relative abundance, the nine fish community metrics used to develop the IBI, and the final IBI scores. This analysis allowed me to determine if there were any statistically significant differences in the environmental and fish community data among the five sites (i.e., whether spatial variability was large enough to overcome expected temporal variability). Two of the sites are categorized as low stress (Alcott Creek and Flotten River) while the other three sites are considered moderate stress (Nolin Creek, Sukaw Creek, and DeLaRonde Creek); therefore, I expected to see potential significant differences between the low stress sites (Alcott Creek and/or Flotten River) and the three moderate stress sites. However, a high amount of variability in the data between years may mask potential differences between the low and moderate stress sites.

The ANOVA was significant for 11 of the 16 fish habitat-related variables assessed. Of the 16 environmental variables, only four of those had significant Bonferroni post hoc pairwise contrasts (habitat assessment scores, total phosphorus, specific conductivity, and arsenic; Table 4.9; Figures 4.9-4.12). Fish habitat quality differed between one moderate stress site, Nolin Creek, and all the other four sites (moderate and low stress sites). Nolin creek had a much lower mean habitat assessment score ( $38 \pm 1.8$ ) compared to the other four sites, with the next closest site scoring 54 points higher ( $91.5 \pm 1.8$, DeLaRonde Creek, moderate stress). Total phosphorus was significantly different between one moderate and one low stress site (Sukaw Creek and Flotten River, respectively). Specific conductivity showed significant differences between two moderate stress sites (Sukaw Creek and DeLaRonde Creek) and arsenic showed significant differences between low stress and moderate stress sites (Alcott Creek and Nolin Creek, Alcott Creek and Sukaw Creek, Flotten Creek and Nolin Creek, Flotten Creek and DeLaRonde Creek) and two moderate stress sites (Sukaw Creek and Nolin Creek). I found a significant difference in fish relative abundance between Alcott creek and Nolin and DeLaRonde creeks (Table 4.10,

Figure 4.13). Species richness was significantly different between Flotten River and DeLaRonde Creek (Table 4.10, Figure 4.13). The ANOVA was significantly different for three (relative abundance of coldwater fish, number of water column species, and percent brook stickleback) of the nine metrics used to develop the IBI (Table 4.10); however, only two metrics, the number of water column species and percent white sucker, had significant Bonferroni post hoc pairwise contrasts between some of the sites (Table 4.10, Figure 4.14). The number of water column species was significantly different between both low stress sites and one moderate stress site (Alcott Creek and DeLaRonde Creek, Flotten River and DeLaRonde Creek). The Bonferroni post hoc pairwise contrast showed a significant difference in the percent white sucker between the two low stress sites (Alcott Creek and Flotten River). The repeated measures ANOVA showed no significant differences in index scores between the five sites (Table 4.10, Figure 4.15).

Table 4. 9| Repeated measures ANOVA results for the water quality and habitat variables for the five revisited sites over the threeyear sampling period. Significant p-values are in bold. Significant differences between sites as indicated by Bonferroni post hoc tests are also given.

N DF (1,2) F-value P-value Geta Squared

|  | N | DF (1,2) | F-value | P-value | Geta Squared | Post Hoc Significance |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Habitat |  |  |  |  |  |  |
| Habitat assessment score (\%) | 5 | 4, 8 | 632.2 | <0.0001 | 1.00 | Nolin ck*Alcott ck, Nolin ck*Sukaw ck, Nolin ck*Flotten river, Nolin ck*DeLaRonde ck |
| Nutrients |  |  |  |  |  |  |
| Log 10 total nitrogen (mg/L) | 5 | 4, 8 | 1.6 | 0.265 | 0.38 | none |
| Log 10 total phosphorus (mg/L) | 5 | 4, 8 | 4.1 | 0.042 | 0.64 | Sukaw ck*Flotten river |
| Nitrate ( $\mathrm{mg} / \mathrm{L} \mathrm{N}$ ) | 5 | 4, 8 | 3.5 | 0.062 | 0.62 | none |
| Phosphate (mg/L P) | 5 | 4, 8 | 5.3 | 0.022 | 0.69 | none |
| Log 10 planktonic Chl-a (ug/L) | 5 | 4, 8 | 9.6 | 0.004 | 0.80 | none |
| Log 10 benthic Chl-a (mg/m2) | 5 | 4, 8 | 2.8 | 0.099 | 0.41 | none |
| Physical |  |  |  |  |  |  |
| Log 10 turbidity (FNU) | 5 | 4, 8 | 14.7 | 0.0009 | 0.73 | none |
| Log 10 TSS (mg/L) | 5 | 4, 8 | 3.1 | 0.079 | 0.35 | none |
| Log 10 SPC (us/cm at 25C) | 5 | 4, 8 | 11.2 | 0.002 | 0.80 | Sukaw ck*DeLaRonde ck |
| Log 10 p . alkalinity ( $\mathrm{mg} / \mathrm{L}$ ) | 5 | 4, 8 | 5.3 | 0.022 | 0.55 | none |
| Metals |  |  |  |  |  |  |
| Log 10 aluminum (ug/L) | 5 | 4, 8 | 5.4 | 0.021 | 0.55 | none |
| Log 10 arsenic (ug/L) | 5 | 4, 8 | 155.7 | <0.0001 | 0.95 | Alcott ck*Nolin ck, Alcott ck*Sukaw ck, Nolin ck*Sukaw ck, Sukaw ck*Flotten ck, Flotten ck*DeLaRonde ck |
| Log 10 iron (ug/L) | 5 | 4, 8 | 49.1 | <0.0001 | 0.94 | none |
| Log 10 manganese (ug/L) | 5 | 4, 8 | 12.3 | 0.002 | 0.80 | none |
| Log 10 mercury (ug/L) | 5 | 4,8 | 1.5 | 0.286 | 0.05 | none |

Abbreviations are as follows: TSS $=$ total suspended solids, $\mathrm{SPC}=$ specific conductivity
significance at $\alpha=0.05$


Figure 4. 9| Variability in stream and river habitat assessment scores over the three-year sampling period for the five revisited sites in the Beaver River watershed. Alcott Creek and Flotten River are classified as low stress sites and Nolin Creek, DeLaRonde Creek, and Sukaw Creek are moderate stress. Asterisks (*) represent significant differences between sites.


Figure 4. 10| Variability in raw nutrient water quality data over the three-year sampling period for the five revisited sites in the Beaver River watershed. Alcott Creek and Flotten River are classified as low stress sites and Nolin Creek, DeLaRonde Creek, and Sukaw Creek are moderate stress. Each panel represents one of six nutrient parameters used to develop the Index of Biotic Integrity. Asterisks (*) represent significant differences between sites.


Figure 4. 11| Variability in raw physicochemical and ion (p. alkalinity) water quality data over the three-year sampling period for the five revisited sites in the Beaver River watershed. Alcott Creek and Flotten River are classified as low stress sites and Nolin Creek, DeLaRonde Creek, and Sukaw Creek are moderate stress. Each panel represents one of four physico-chemical or ion parameters used to develop the Index of Biotic Integrity. Asterisks (*) represent significant differences between sites.


Figure 4. 12| Variability in raw metal water quality data over the three-year sampling period for the five revisited sites in the Beaver River watershed. Alcott Creek and Flotten River are classified as low stress sites and Nolin Creek, DeLaRonde Creek, and Sukaw Creek are moderate stress. Each panel represents one of five metal parameters used in the development of the Index of Biotic Integrity. Asterisks (*) represent significant differences between sites.

Table 4. 10| Results of the repeated measures ANOVA for species richness, relative abundance, the metrics used to develop the IBI, and the Index of Biotic Integrity for the five revisited sites over the three-year sampling period. Significant p-values are in bold. Significant differences between sites as indicated by Bonferroni post hoc tests are also given.

Geta
N DF $(1,2)$ F-value P-value Squared

Post Hoc Significance

| Metric |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| $\log _{10}$ Relative abundance | 5 | 4, 8 | 13.43 | 0.001 | 0.73 | Alcott ck*Nolin ck, Alcott ck*DeLaRonde ck |
| Species richness | 5 | 4, 8 | 25.29 | <0.001 | 0.92 | Flotten river*DeLaRonde ck |
| $\mathrm{Log}_{10}$ Relative abundance of coldwater fish | 5 | 4, 8 | 3.83 | 0.050 | 0.59 | none |
| Percent top carnivores and piscivores | 5 | 4, 8 | 1.77 | 0.228 | 0.42 | none |
| Percent invertivores | 5 | 4, 8 | 0.92 | 0.497 | 0.29 | none |
| $\mathrm{Log}_{10}$ Percent benthivores excluding WHSC | 5 | 4, 8 | 2.16 | 0.165 | 0.50 | none |
| No. of water column species | 5 | 4, 8 | 23.77 | <0.001 | 0.91 | Alcott ck*DeLaRonde ck, Flotten river*DeLaRonde ck |
| $\log _{10}$ Percent of fish with parasites | 5 | 4, 8 | 2.91 | 0.093 | 0.55 | none |
| Percent coolwater fish | 5 | 4, 8 | 0.65 | 0.644 | 0.19 | none |
| $\log _{10}$ Percent white sucker | 5 | 4, 8 | 0.76 | 0.582 | 0.23 | Alcott ck*Flotten river |
| Percent brook stickleback | 5 | 4, 8 | 8.38 | 0.006 | 0.75 | none |
| Index of Biotic Integrity | 5 | 4, 8 | 0.83 | 0.542 | 0.25 | none |

significance at $\alpha=0.05$


Figure 4. 13| Between and within site variability in fish species richness and relative abundance of the five revisited sites over the three-year sampling period. Significant differences between sites are shown.


Figure 4. 14| $\log _{10}$ variability in fish metric scores over the three-year sampling period for each of the five revisit sites. Each panel represents one of the nine metrics used to develop the Index of Biotic Integrity. Asterisks $\left(^{*}\right.$ ) represent significant differences in metric scores between sites. Metric abbreviations are as follows: PercentLitho = percent of litho-obligate individuals, PercentParasites = percent of individual fish with parasites, PercentTolReprod = percent of tolerant reproductive guild, PercentTopCarn = percent of top carnivore and piscivore fish, RelAbund = relative abundance (fish/100s), RelAbundCold = relative abundance of cold-water fish (cold-water fish/100s), SpeciesRich = total number of species. Alcott ck, DeLaRonde ck, and Flotten river are classified as low stress sites and Nolin ck and Sukaw ck are moderate stress.


Figure 4. 15| Annual variability in Index of Biotic Integrity scores for each of the five revisited sites over the three-year sampling period.

## Chapter 5: DISCUSSION

The goal of this research was to adapt, apply, and critically evaluate a fish communitybased ecological assessment tool, known as the Index of Biotic Integrity, for Boreal Plain lotic environments that experience a range of stressors. The Boreal Plain ecozone is influenced by agriculture, forestry, mining, oil and gas operations, and urbanization, among other disturbances, and provides a unique opportunity to assess and evaluate a fish-based IBI in a relatively homogenous area with multiple land-use stressors. The primary objectives of this research were to determine the expected fish community structure and fish condition in minimally disturbed streams in northern SK, to determine if fish community structure and fish condition vary with a gradient of human disturbance by applying and evaluating a fish-based IBI, and to determine the sensitivity of IBI methods and results to inter-annual variability.

I was able to develop and adapt a fish-based IBI for streams and rivers in the Beaver River watershed located in the Boreal Plain ecozone of SK, Canada. In doing so, I determined minimally disturbed (or low-stress) conditions and developed a gradient of stream and river health (e.g., low, moderate, and high stress) throughout the Beaver River watershed. By assessing various measures of land use and physical and chemical components of habitat, I identified streams in low stress to be surrounded by forested environments that overall were characterized by intact riparian vegetation, relatively minimal bank erosion and sediment deposition, and ample fish habitat, including such instream structure as submerged vegetation and downed woody debris, undercut banks, varying types of substrate and or riffle/pool/run habitat suitable for a diverse fish assemblage. High stress streams were located in highly modified landscapes that lacked intact riparian habitat, surrounded by cultivated cropland and generally were characterized by moderate to heavy erosion and sediment deposition, and less optimal, highly homogenous instream habitat. The moderate stress sites tended to have a mix of characteristics and fell somewhere in between the low and high stress sites as expected. However, the fish community's response to such disturbances was more convoluted.

This study identified nine metrics that showed consistent relationships with and the highest responsiveness to human disturbance. These nine metrics were selected across the major metric categories, including species richness and composition, trophic, habitat, reproductive, and tolerance guilds, and fish abundance and condition measures, to create an IBI that, as best as possible, describes the fish community composition and structure of the region. Although the
selection of these responsive metrics meant that there were general relationships between fish community structure/condition and stream health, I was unable to detect a statistically significant difference in IBI scores between low and high stress sites owing to large variability within groups. However, there was a statistically significant difference in the mean IBI scores between the low stress and moderate stress sites.

### 5.1 Fish Community Structure and Condition in Low Stress Boreal Plains' Streams

### 5.1.1 Assessing Fish Species Richness and Abundance in Low Stress Sites

Assessing the natural physical and chemical environmental variables of the region allowed me to determine the expected fish community structure and fish condition for minimally disturbed (low-stress) streams and rivers in the Beaver River watershed in SK. Within low stress sites, I hypothesized that fish species richness and abundance would increase from headwater to higher-order streams and rivers. Research has shown that habitat diversity (Angermeier \& Karr, 1984; Langeani et al., 2005) and stream position in the drainage network (Ohio EPA, 1987; Vannote et al., 1989) can influence stream fish community richness and abundance and MMIs, such as the IBI, should account for this natural variation. In my study, fish species richness showed no relationship with stream order, a very weak relationship with drainage area, and a strong relationship with stream wetted width. I was only able to detect one significant relationship between a fish metric ( $\log _{10}$ percent individuals with parasites) and stream wetted width; however, three metrics had correlation coefficients greater than 0.4 and required adjustment. Contrary to what I found, it has been suggested that calibrating metrics for aquatic vertebrate multi-metric indices (MMIs) may be more important than for macroinvertebrate MMIs (Whittier et al., 2007; Stoddard et al., 2008). Interestingly, I did find that measures of relative abundance were more strongly correlated to stream order than wetted width or drainage area; however, these results were also not significant.

Due to the lack of statistically significant relationships between fish species richness and fish relative abundance and stream order, drainage area, and stream wetted width, I can only speculate on the correlations I found. Other research conducted on fish communities in streams and rivers in the SK Boreal Plain region (Merkowsky, 1998) also found fish species richness was best predicted by a channel morphometry parameter, stream wetted width, over drainage basinrelated parameters, such as drainage area, and stream order. Contrary to this, earlier research
conducted on boreal SK streams (Larson, 1976) found stream order to be a significant predictor of fish species richness. Unlike the research conducted by Merkowsky (1998) and Larson (1976) none of the three-stream size-related parameters (stream order, upstream drainage area, and wetted width) I assessed significantly predicted species richness or relative abundance in my study. Despite the small number of sites classified as low stress in this study, I decided to use the relationship between species richness and stream wetted width found in my study since my sites appear to have a higher species richness compared to that found by Merkowsky (1998) (Figure 4.5A). Furthermore, Merkowsky (1998) did not consider human influence and it is unknown if the sites sampled in that study were comparable to low stress or reference conditions. If sites were not all in reference condition, this could explain the lower fish species richness found in the Merkowsky (1998) study for similar streams. It is also possible that there was a difference in the amount of sampling effort between our studies that could help explain my higher predicted fish species richness.

The non-significant relationships between fish species richness and fish abundance and stream size can likely be explained by the relatively few reference or low stress sites $(\mathrm{n}=4)$ in my study, creating a low statistical power. Although this study provides some insights into stream fish relationships with stream size, a larger number of least disturbed or low stress sample sites is needed to further evaluate the relationships between fish metrics and natural gradients in the Boreal Plain ecozone. Additionally, it has been suggested that analyzing biological data at small geographical scales, such as that observed in this study, may remove the effects of a natural gradient (Ode et al., 2008). This may help to explain the nonsignificant relationships between fish metrics and stream size observed here. To date, relatively few studies have researched predictors of fish communities in SK boreal streams (Larson, 1976; Merkowsky, 1998) and I suggest that a more in-depth analysis be conducted on a larger dataset during baseflow conditions to compare with my results. Including measures of habitat complexity in further studies would also be beneficial (Gratwicke \& Speight, 2005; Kovalenko et al., 2012). Additional comparison with alternative community assessment methods, such as environmental DNA analysis (eDNA), should also be considered to rule out sampling bias during extreme flow events (Doi et al., 2017; Evan et al., 2017; Olds et al., 2016).

### 5.1.2 Assessing Variability in Fish Community Structure and Condition in Low Stress Sites

I hypothesized that the overall fish community structure and fish condition would have the least amount of variation among minimally disturbed or low stress streams and rivers and that the fish community will show greater variability with higher stress. Within the low stress sites, there was considerable variability in metric and index scores, indicating variability in the fish community structure and condition of my low stress streams and rivers. However, I found a similar amount of variability between the low, moderate, and high stress sites, indicating that the fish community showed similar variance within each of the stress classes. Of the nine metrics used to develop the IBI, only two negative scoring metrics, the $\log _{10}$ percent white sucker metric, and the percent brook stickleback metric, had significantly different variances between the low and high stress sites. Biological communities, including fish communities, are known to have higher variance in areas of suboptimal habitat or degraded habitat quality due to anthropogenic stress (Cote et al., 2013). This could explain the significantly higher variance for these two metrics in the high stress sites.

The high amount of variability in the fish community, as indicated by IBI and metric scores within each stress class, including the low stress sites, may be an indication that the stress classification in my study is not the most appropriate, or that there are other stressors not accounted for affecting the fish community (see section 5.4 for further discussion). For example, four of my sites were assigned to the low stress class, but three of these sites scored within eight points of each other, while one site, Nesslin Creek, scored much lower (24 points lower). This suggests that the fish community at Nesslin Creek did not reflect the relatively minimally disturbed environmental conditions and lack of human influence found at that site. There are a few possible explanations for site anomalies such as this. Further evaluating the site reveals that only one fish, a northern pike, was collected from this site. Factors such as sampling bias, variability in environmental and hydrological conditions, and/or seasonal effects may be attributed to the low catch at such sites. Nesslin Creek had a mean stream velocity in the top $90^{\text {th }}$ percentile of all sites and contained relatively high flow conditions at the time of sampling. The swift current and relatively deep pools may have impacted our ability to effectively see and capture a representative sample of the fish community at this site. Additionally, seasonality could contribute to the high variability in index scores within stress classes. However, using the low stress sites as an example, this appears to not be the case for the low index score at Nesslin

Creek. For example, within the low stress group, three sites were sampled throughout August of 2017 (including Nesslin Creek), and one site (Landry Creek) was sampled in mid-October when water levels and temperatures were much lower ( $\sim 3^{\circ} \mathrm{C}$ vs $\geq 17^{\circ} \mathrm{C}$ ), with ice beginning to form on the water's surface. These temperature/seasonal differences could affect the fish community present and therefore the resulting IBI scores, but Nesslin creek was sampled during August when the majority of the other sites were sampled. Further, Landry Creek was sampled late during October, but the IBI score at this site was within the expected range, ruling out seasonality as an issue for the unusually low index score of Nesslin Creek. Although sampling during suboptimal conditions was avoided as best as practical, for the purpose of this research it was impossible to completely avoid flow extremes due to the fluctuating environmental conditions during the open water season in the Beaver River watershed in 2017. This could have contributed to the high variability in index scores within site classes, including the low stress sites, since stream fish are known to take refuge in times of flow and temperature extremes (Freeman et al., 2022; Sutela et al., 2017).

It is important to note that establishing minimally disturbed reference conditions from a range of sites will always lead to some inherent variability in the community being assessed (Bailey et al., 2004) due to natural variations in environmental conditions. The Reference Condition Approach and similar methods used to model reference conditions from a range of minimally disturbed sites help to minimize and account for this natural variation. It may simply be that the variance found in the fish community in my study is an acceptable amount for this region, and more research in SK and other Boreal Plain watersheds will need to be conducted to fully answer this question. It is possible that the fish community in this region is naturally more variable regardless of the stress classification or reference condition methods chosen. A similar study conducted on the Beaver River watershed in Alberta appeared to also have high variability in fish metric values, but a lower variability overall in the final IBI scores (Cantin \& John, 2012). Analogous research has suggested a higher tolerance among northern biotic communities (Phillips et al., 2023; Bramblett et al., 2005; Cantin \& John, 2012; Stevens et al., 2006, 2010) and may explain the high and similar variability between our low and high stress sites. Northern biotic communities require adaptation to unstable flow regimes and harsh fluctuating environmental conditions characteristic of northern regions, making them highly tolerant trophic, reproductive, and habitat generalists (Dodds et al. 2004).

### 5.2 Fish Community Structure and Condition Across a Disturbance Gradient in Boreal Plains' Streams

### 5.2.1 Variation in Fish Community Structure Across a Disturbance Gradient

I hypothesized that fish community structure and fish condition, as indicated by index scores, would change with gradients of human disturbance. Additionally, I hypothesized that impacted sites would have lower species richness and abundance, a higher percentage of tolerant species, and fish with a higher frequency of abnormalities. A gradient of stream disturbance from human influence was established using various environmental parameters and surrounding land use conditions for each stream and river sampled. Human influence, determined by environmental conditions, increased as expected from low to high stress sites. I was able to show that fish community structure and condition, as indicated by IBI scores, somewhat follow the expected trend with human disturbance; although these differences were significant between the low and moderate stress sites, no significant difference was found between the low and high or moderate and high stress sites. None of the nine fish community metrics assessed showed a significant difference between low and high stress sites; however, the percent of top carnivores and piscivores metric and the percent fish with parasites metric were significantly different between low and moderate, and the moderate and high stress sites, respectively. The percent top carnivores metric was highest in the low stress sites as expected and although the percent individuals with parasites metric increased between the low and moderate stress groupings, the high stress sites actually had the lowest mean score. IBI scores were highest in the low stress sites (indicating higher biotic integrity), decreased by an average of 18 points in the moderate stress group, and showed a slight increase between the moderate and high stress sites ( $<1$ point). Finding the highest mean metric and index scores in the low stress (minimally disturbed sites) indicates the presence of a healthier, more diverse fish community, or higher biotic integrity (Karr, 1981; Li et al, 2018; Mamun \& An, 2020).

The relatively small number of sample sites in my low and high stress sites (four low stress vs three high stress sites) led to low power and may have contributed to the lack of a significant difference in IBI scores. It is likely that the larger sample size of the moderate stress sites $(\mathrm{n}=11)$ led to a higher power and statistically significant difference between the low and moderate stress groupings, even though the moderate and high stress sites share a similarly low mean index score. Interestingly, all of the sites had relatively low IBI scores. The overall mean

IBI score was 52.6 out of 100 and the mean index score for the sites in the low stress category was 66.3 , with the highest site scoring only 76.5 points. This is lower than what was expected for sites considered minimally disturbed or the best available conditions of this region, considering the negligible anthropogenic influence found on these streams. One explanation is that the sites may have been incorrectly assigned during stress classification. Standard protocols and quantitative criteria were used during the site classification steps to try to minimize variance within groups and maximize the variance between groups. However, depending on the environmental data chosen to develop stress classes, I found some variation in the sites assigned to each stress class (see section 5.4 for further discussion). Additionally, due to my relatively low number of sample sites, I had to maintain all the sites as one grouping for stress classification and further analyses. Although the streams sampled in this study were contained in one basin, they comprised two separate ecoregions and four stream orders. Interestingly, all the sites in the low stress grouping were found in the Mid-Boreal Upland ecoregion, while sites in both the moderate and high stress groups were spread between the Mid-Boreal Upland and Boreal Transition ecoregions. Two out of three sites in the high stress grouping were located in the Boreal Transition ecoregion. This may be explained by the relatively higher presence of agriculture in the Boreal Transition region compared to the Mid-Boreal Upland. More sites would have allowed me to further refine site groupings by a single ecoregion and smaller-scale descriptors, and perhaps narrow spatial heterogeneity and minimize some of the biological variation I observed. Descriptors, such as stream order or ecoregional (Krause et al., 2013; Phillips et al., 2023) and watershed-based (Shearer \& Berry, 2002; Cantin \& John, 2012) delineations have been found adequate at partitioning the natural variability amongst minimally disturbed sites in similar research.

The low IBI scores found in this study may also be an indication of a fish community that is not overly responsive to anthropogenic stressors (Dodds et al., 2004; Poff \& Allan, 1995). Interestingly, the study conducted on the Alberta portion of the Beaver River watershed found a similarly low, albeit slightly lower, IBI score (63 out of 100) at their lowest stress site. As discussed above, analogous research in biogeographically similar areas has found northern biotic communities to be depauperate and highly tolerant (Bramblett et al., 2005; Cantin \& John, 2012; Phillips et al., 2023; Stevens et al., 2006, 2010). The adaptation of northern biota to highly variable physicochemical conditions may help to explain the low index scores found overall in
my study sites and may potentially mask the effects of diffuse nonpoint source pollution and habitat degradation making it difficult to detect measurable differences between our stress classes (Dodds et al., 2004; Poff \& Allan, 1995). The Boreal Plain ecozone comprises characteristics of both the highly variable northern great plains and Boreal ecosystems (Massie, 2014). The higher tolerance of biotic communities in northern regions complicates the adaptation and use of an IBI in these areas (Cantin \& John, 2012; Stevens \& Council, 2008; Stevens et al., 2006, 2010). It is also possible, however, that there is some unaccounted-for stressor that is affecting the fish community at such sites, and therefore leading to lower than expected site IBI scores.

Additionally, the depauperate nature of the study region, characteristic of northern regions, further complicates the use of community-based bioassessment tools, such as the IBI. Saskatchewan has a relatively low species richness compared to regions in lower latitudes and milder climates. Sixty-nine species of fish occur within the province, 58 of which are native, and 11 are exotic (Ashcroft et al., 2006). Approximately 40 of these species are known to occur within the Churchill River drainage basin, which contains the Beaver River watershed, including two imperiled species and four introduced sport fish trout species (Morris \& Somers, 2015; Scott \& Crossman, 1973). In this study I found only 16 species of fish, all of which are native to the region. The IBI was initially developed for relatively species-rich midwestern streams and rivers (Karr, 1981); however, IBIs have been adapted and validated for use in areas of low fish community richness (Bramblett et al., 2005; Cantin \& John, 2012; Long \& Walker, 2005; Shearer \& Berry, 2002; Stevens \& Council, 2008; Stevens et al., 2006, 2010). Many of these studies have at least briefly discussed the difficulties associated with IBI adaptation to speciesdepauperate regions. A study conducted on the James River basin in North and South Dakota found poor performance of most metrics for the upper portion (North Dakota portion) of the basin due to the low species richness of the area $(\mathrm{n}=11)$. However, they were able to develop a responsive IBI for the lower South Dakota region of the basin, which only contained 22 species of fish. Additionally, despite the low species richness of the region, Stevens et al. (2010) were able to successfully develop an IBI for the Battle River in Alberta and link human disturbance to fish metrics for only 14 species of fish. Similarly, Stevens et al. (2006) were able to successfully relate fish metrics to human disturbance for a mere five species of tolerant fish found in grassland streams in Alberta. However, both studies contained a much higher percentage of
agriculture in the respective basins (mean of $26 \%$ and $72 \%$, respectively) compared to this study (mean of $11 \%$ ) and much larger sample sizes ( 69 and 80 sites, respectively). The high level of agricultural disturbance in these studies may have tremendous effects on nutrient and sediment loads in these waterbodies, increasing the likelihood of detecting differences in site quality (Allan, 2004; Johnson et al., 1997; Stevens et al., 2006, 2010).

Similarly, Cantin \& John (2012) developed an IBI for the Beaver River watershed in Alberta and found consistent relationships between five fish metrics and human influence for a similar number of fish species $(\mathrm{n}=17)$ to what I found on the SK side of the watershed. The amount of agricultural disturbance was also comparable, although slightly higher, than that in the current study (e.g., $17 \%$ average cropland for Alberta study, $5 \%$ average cropland in this study); however, the Alberta study had a much larger sample size ( 50 sites). Out of the five metrics chosen for the Alberta portion of the watershed, only the percent top carnivores metric was in the top 10 most responsive metrics to human disturbance in this study. This difference in fish metric responsiveness to human influence on the landscape could arise from differences in how human influence is determined; however, methods comparable to that of Cantin \& John (2012) were used in this study.

Another potential factor may be the size of streams assessed in each of these studies. Interestingly, Stevens et al. (2010) found that the failure rate for small stream test sites occurred less often compared to larger stream test sites, potentially indicating a higher resilience to disturbance in the small stream fish assemblage. Streams in my study ranged between small and moderately sized first- and fourth-order streams, while the sites sampled on the Alberta side of the watershed were much larger rivers in comparison, including the Beaver River itself and two of its major tributaries, the Sand and Amisk rivers. The fish species collected were similar between the two studies; however, the Alberta study collected more invertivorous cyprinid species (e.g., spottail shiner, river shiner, and emerald shiner) and large-bodied river species (e.g., longnose sucker, white sucker, yellow perch, walleye, and lake whitefish). This may explain why they were able to find a stronger consistent relationship for metrics such as percent benthic invertivores and percent invertivorous cyprinids. Additionally, larger rivers, such as the Battle (Stevens \& Council, 2008; Stevens et al., 2010) and Beaver (Cantin \& John, 2012) Rivers in Alberta, likely experience a disproportionate amount of human influence (Vorosmarty et al., 2010) compared to the small streams assessed in the present study.

### 5.2.2 Variation in Fish Condition Across a Disturbance Gradient

I hypothesized that fish condition would be the highest and show little variation in minimally disturbed conditions. I found that fish condition, as assessed through IBI scores, had the highest quality among low-stress streams and rivers as expected, but the index scores were only significantly different between the low and moderate stress sites. To further assess fish community health, I evaluated three condition-related metrics. Assessing additional measures of fish condition (Fulton's $K$ of a sentinel species (white sucker), the percent of the fish community with abnormalities, and the percent of the fish community with parasites) showed no significant differences between the minimally disturbed (low stress) and high stress sites or moderate stress sites. However, one metric, the percent of the fish community with parasites, was significantly different between the moderate and high stress sites. I found no obvious trend in the body condition of white sucker, using Fulton's $K$ factor, between any of the stress classes, which is likely a result of the small number of sample sites (two low and two high). There was a $25 \%$ and $39 \%$ difference between the low and moderate stress sites for the percent individuals with parasites and the percent individuals with abnormalities metrics, respectively. For the percent individuals with parasites metric, none of the sites classified as high stress based on land use and environmental data contained fish with obvious parasites, and for the percent individuals with abnormalities metric only one site contained fish with noted abnormalities within the high stress grouping. This created a very low average for these metrics in the high stress sites. This could have resulted from the misclassification of sites (discussed above), sampling bias or error, variational in environmental conditions (e.g., temperature), or it could be that condition metrics related to abnormalities and parasites are not highly responsive to the types of human disturbance in this region. It is also possible that additional unaccounted-for stressors may be affecting the fish community and contributing to the lack of significant differences in fish condition.

Sampling bias was minimized as much as possible by following standard operating procedures. In some sites, especially those sampled in higher temperatures, some stress to the fish was inevitable. This may have led to a higher frequency of such abnormalities as bruising, hemorrhaging, fin fraying, etc. To avoid misinterpreting potential injury accrued from collection, storage, and handling of the fish, all bruising, hemorrhaging, fraying, and other potential handling or electrofishing related fish injuries (e.g., spinal curvature) were not included in metric
calculation. It is possible, however, that some fish hosting parasites went undetected, especially if the abnormality never showed subcutaneously (i.e., due to nonlethal sampling, only external abnormalities and parasites were noted) although this bias should have been similar between all stress classes.

Water temperature is known to influence fish parasite outbreaks (Hakalahti \& Valtonen, 2005; Schaaf et al., 2017), including black spot cysts caused by a parasitic trematode commonly found on fish in this study. Two of the three high stress sites were sampled in mid-October when water temperatures on average are much lower compared to July and August. This could have contributed to the reduced presence of obvious external parasites in these sites. It should also be noted that the percent abnormalities and percent parasite metrics were likely highly correlated as the percent abnormalities also included parasites.

Finally, it is possible that metrics assessing abnormalities and parasites are not a good indicator of stream and fish community health in SK watersheds. If the fish community is highly tolerant to rapidly changing environmental conditions this may also explain the lack of differences in fish condition between low and high stress sites, due to a relatively tolerant fish community regardless of the presence of anthropogenic stressors. If this is the case for SK boreal streams, fish community response to human influence may be less noticeable than other regions with less tolerant taxa (e.g., salmonids). Furthermore, it has been suggested that parasite occurrence can be independent of habitat and water quality, leading to misinterpretation of site quality (Mebane et al., 2003) and analogous research has avoided such metrics for this reason (Cantin \& John, 2012). Additionally, as discussed above, a major caveat for this analysis is the small number of sample sites in the low and high stress groupings. The small sample size makes it difficult to detect measurable differences between sites of different health ratings and therefore these results should be regarded only as a preliminary investigation to inform more in-depth research.

### 5.3 Interannual Variation in Environmental Data and the Index of Biotic Integrity

I hypothesized that fish communities, water chemistry, and fish habitat variables will show interannual variance within sites, due to differences in environmental conditions and fish residency and mobility between years; however, these differences will not be reflected in IBI scores, with greater variance among sites than within sites. This test of annual variance in fish
communities and environmental variables allow me to determine the stability of the index and fish metrics through time and help critically evaluate the reliability of IBI-type approaches as an ecological assessment tool for lotic environments in northern regions. An essential part of any biological monitoring tool is its ability to differentiate between biotic changes brought on by anthropogenic disturbances and those that happen as a result of natural variations in environmental conditions through time.

Using a repeated measures ANOVA, I detected a significant difference among sites for some of the environmental variables related to water chemistry and fish habitat, fish relative abundance and species richness, as well as two of the fish community metrics (the number of water column species and the percent brook stickleback metrics). When looking at only the 2017 data, two metrics were significantly different between the low and moderate stress sites (the percent top carnivore metric) and the moderate and high stress sites (the percent parasites metric). I was not able to detect a significant difference between either of these metrics for any of the sites when assessing the mean scores over three years. This may indicate that the variability between the years was too high and masked potential differences between sites. For those environmental measures and fish metrics that I detected significant differences among sites, significant differences occurred between low stress and moderate stress sites as expected. This indicates that differences in some of the environmental data were larger among sites than within sites over the three-year sampling period, a necessary component of the IBI. However, differences were also seen between sites of the same stress grouping. In an ideal situation I would only detect significant differences between sites of different stress ratings; since this was not always the case, a significant difference between the low stress sites, or any of the moderate stress sites, may indicate that one or more of the sites was not assigned to the correct stress grouping. However, some range within the individual stress groupings is inevitable and the ability to detect these differences, even among years, could be seen as a strength of the IBI.

Although I was able to detect significant differences among sites for fish species richness and relative abundance, post hoc comparisons revealed only two of the nine metrics were significantly different among sites when comparing across years. These two metrics also differed from the two metrics that were significantly different between stress groups using only the 2017 dataset. This likely can be attributed to a large amount of variability in the fish community assemblage between years in the study sites. Analyzing signal-to-noise ratios for fish community
metrics during the index development process also revealed a high amount of variability between years in the fish community. Even though a significant difference in IBI scores was detected between the low and moderate stress sites in 2017, I was not able to detect any significant difference when looking at the IBI scores across all three years. This also suggests that like seven of the nine fish community metrics, the IBI score variability within a site and among years was higher than desired. I tried to sample revisited sites in a similar time frame each year of sampling; however, there were differences in environmental conditions, due to environmental stochasticity, such as flow regime, temperature, etc., that may have contributed to the variability in the fish community between years. Temporal variability in the fish community (and IBI) can be minimized by avoiding sampling during times of migration or reproduction, removing YOY from catch (Barbour et al., 1999), or by sampling across seasons and years and developing season-specific indices (Smokorowski et al., 1998; Zhu et al., 2021). Although I removed YOY fish less than 20 mm total length from the dataset and restricted the timing of sampling as best as practical to be within a similar time frame each year, for the purpose of this research, it was impossible to only sample in baseflow conditions and avoid all fish reproduction and migration. For example, in this study, species such as brook stickleback, fathead minnow, longnose dace, and northern redbelly dace are known to reproduce into late July and/or August and many undergo intermittent, multiple spawning events (Scott \& Crossman, 1973). Revisited sites were sampled within 30 days of the initial sample date annually, but it is possible that changing flow and temperatures may have affected fish reproduction and mobility (Albanese et al., 2004) and therefore capture rate. Random, episodic events, such as flood or drought, affect species distribution and abundance, are difficult to control and add complexity and uncertainty into bioassessment tools such as the IBI (Kilgour et al., 2013). Furthermore, increasing stream flow has led to higher connectivity and mobility of fish assemblages and could alter fish species composition and abundance (Franssen et al., 2006; Ngor et al., 2018; Pander et al., 2019).

For this analysis, two sites fell in the low stress category (Alcott Creek and Flotten River) and three in the moderate stress group (Nolin Creek, Sukaw Creek, and DeLaRonde Creek). Therefore, an obvious shortcoming of this analysis was the lack of high stress sites in my revisited sites group and any further research should sample a larger number of sites across the entire disturbance gradient (e.g., low, moderate, and high stress). Additionally, a larger sample size of streams and rivers would also help to offset potential outliers.

This analysis aids in determining the stability of the index and fish metrics through time, an essential component of any ecological assessment tool. Despite the large fluctuations in environmental conditions in northern boreal regions with environmental extremes and immense seasonal variability, I was able to detect differences in some of the environmental and fish community data between some of the sites when analyzing data over three years with highly variable flow regimes. However, I was only able to detect significant differences for two of the nine metrics used to develop the index, indicating seven of the most responsive metrics in this study were highly variable through time. Additionally, no significant differences in index scores were detected between sites when analyzing data across multiple years, suggesting greater temporal variability among years than spatial variability across sites. My results reinforce the importance of long-term monitoring to decipher trends in natural variation of fish communities from variation created by anthropogenic stressors.

### 5.4 Index of Biotic Integrity Development Methods

### 5.4.1 Development of Reference Conditions

### 5.4.1.1 Site Classification

Sites were classified by geographical and stream or river type environmental descriptors and by determining the minimally disturbed or reference conditions (Stoddard et al., 2006) that are expected for the fish community assemblage, physical habitat structure, water chemistry, and nutrients of the region. Standard protocols and quantitative criteria were used during the site classification steps. However, depending on the environmental variables chosen to classify the sites into stress groupings, the site assignment to each stress class varied slightly. For example, when sites were classified initially using various redundant land use-related measures, it seemed that there was an improved separation of sites between stress classes, and those differences were found to be significant. However, this classification scheme was not chosen as it contained many redundant variables.

Another potential issue arising during site stress classification was the relatively low number of sample sites in this study. Due to the high flow conditions in the watershed during the 2017 season, I was only able to sample 18 separate streams. Ideally, when defining reference conditions, a large number of streams (20+) would be sampled and used to characterize variability amongst waterbodies (Bailey et al., 2004; Bowman \& Somers, 2005; Reynoldson \&

Wright, 2000). Due to the high flow conditions and resulting relatively low number of streams with conditions appropriate to sample, I had to maintain all 18 streams as one grouping for site classification (Mid-Boreal Upland and Boreal Transition ecoregions and stream orders 1-4). Much bioassessment research has highlighted the importance of classifying study sites into analogous groupings to effectively partition natural variability among reference or minimally disturbed sites (Bailey et al. 2004; Bowman \& Somers, 2005; Phillips et al. 2023). Sites are commonly classified based on large-scale geographic or geomorphological environmental descriptors, such as ecoregion, biogeographic zones, surficial geology, climate, etc., as well as smaller-scale river and stream environmental descriptors, including stream size, discharge, drainage basin, elevation, water temperature, and so forth. Although the streams sampled in this study were contained in one basin, they comprised two separate ecoregions and four stream orders. Additional sites would have allowed me to further refine site groupings by a single ecoregion and smaller-scale descriptors, and perhaps further minimize spatial heterogeneity.

### 5.4.1.2 Establishing Stress Classes

Various approaches were analyzed and compared to determine the most appropriate method for developing reference condition and stressor classification (mean and SD, median and interquartile range, equal interval/ranked approach, specific cut-off criteria, and established guidelines). Changing the stress classification method led to differences in site assignment to stress classes, thereby affecting my disturbance gradient. Depending on the distribution of the data, some of the above classification methods worked better than others. For example, where data were heavily skewed to one direction (e.g., sites were more heavily weighted towards high habitat assessment scores), selecting a specific cut-off value seemed to be more appropriate to determine stressor classes (e.g., $\geq 80 \%=$ low stress; $60 \%-80 \%=$ moderate stress; $\leq 60 \%=$ high stress). This is because even choosing the $75^{\text {th }}$ percentile as a cut-off could lead to underrepresenting the number of low stress sites and over representing the amount of moderate and high stress sites (e.g., $75^{\text {th }}$ percentile $=$ habitat assessment score of $95 \% ; 25^{\text {th }}$ percentile $=$ score of $76 \%$ ). Using the previous example, all sites scoring below $76 \%$ on the habitat assessment portion would fall into the high stress category, which may not accurately reflect a "high stress" site in other areas. Where there was a lot of spread/variance in the data, using the mean $\pm$ SD did not work well and often led to stressor values going below 0 or above 100 when
there was a large SD and/or highly skewed data. Using an equal interval or ranked approach (e.g., six sites in low, six sites in moderate, and six sites in high) often created a very large range within a single stress grouping or a very narrow separation between stress categories (e.g., $0.1 \%$ difference in urban cover between low and moderate sites). This approach would also separate sites that share the same stressor value into different categories (e.g., two sites have the same number of oil and gas wells per watershed, but one site is placed in the low stress category while the other falls in moderate stress because they are in the $6^{\text {th }}$ and $7^{\text {th }}$ positions, respectively).

For water quality variables where established criteria or guidelines existed, the established criteria were used to obtain stress classifications for comparison (e.g., high stress $=$ sites exceeding the guidelines, moderate stress = sites below and down to $10 \%$ of the guideline, and low stress $=$ sites below $10 \%$ of the established guideline). However, not all water quality variables have established criteria or guidelines, and therefore, this approach would not work for all variables. Additionally, this approach often left inconsistent site numbers in each stress category, including, few or no sites in one or two of the stress categories (e.g., zero sites in low stress, two in moderate stress, and the rest in high stress).

Using the median and interquartile range and specific cut-off criteria approaches allowed for a more distinct separation between stress categories. Selecting a specific cut-off criterion worked quite well since it often allowed for separation between the stress categories and a slightly smaller range within categories; this approach is comparable to the Jenks Natural Breaks method where cut-off values are chosen based on the natural gaps in the data (Davies \& Hanley, 2010). However, this method can be ambiguous depending on who is choosing the criteria and will vary with each region and stressor. For example, the specific cut-off criteria used in this study were very low due to low watershed disturbance, and likely not attainable in other watersheds with more populated and developed areas. Therefore, this approach is not a standardized method that can be applied to other watersheds and was avoided for the purposes of this study. The median and IQR does not work well when the data is heavily skewed (as discussed above) or when there are many zero values in the dataset, as this could lead to all stress categories (low, moderate, and high stress) equalling $0 \%$ (e.g., for the number of landfills, the low ( $<25^{\text {th }}$ percentile), moderate ( $25^{\text {th }}-75^{\text {th }}$ percentiles), and high ( $>75^{\text {th }}$ percentile) stress values were all equal to zero). In fact, when the dataset had a high number of zero values, choosing a specific cut-off criterion worked best.

The percentiles and specific cut-off criteria approaches worked best overall; however, the specific cut-off criteria approach is difficult to standardize, it is ambiguous, and may vary with watersheds. To maintain consistency in defining stressor classes across all the data, I decided to use the percentile approach, where the median and interquartile range were used to define low (reference), moderate, and high stressor classes for each stream. Using the median and interquartile range also allowed for a consistent number of sites to fall in each stress category, contrary to the mean and SD or choosing a specific cut-off value, where the number of sites in each stress category could vary with stressor type. Establishing quantitative criteria for reference sites aids in a consistent framework for selection and can be easily adapted to other watersheds and regions (Stoddard et al., 2006, Bailey et al., 2004).

### 5.4.2 Development of an Index of Biotic Integrity (IBI) for Boreal Plains'Streams and Rivers

Metric selection and calibration for the index, metric scoring, and final index development are important steps to create a successful IBI that can discriminate biotic condition between sites of varying stress. Due to the low fish species richness in my study region, I had difficulty finding metrics that passed the calibration steps (metric range, reproducibility, redundancy, and responsiveness) discussed in Stoddard et al. (2008), despite assessing 42 individual metrics.

### 5.4.2.1 Metric Range

The first step in the metric calibration process was to assess individual metrics for insufficient data values and range. Even though I selected candidate metrics from IBIs created for similar biogeographic regions and nearby areas it was difficult to find enough metrics with a large range between sites (e.g., a range $>4$ or $<1 / 3$ of their values $=0$ ) due to the low species richness and abundance in some of the sites. Therefore, I had to increase the threshold for removal to be any metric with $>1 / 2$ values $=0$ or a range of $<3$. This initially allowed enough sites to move on to the next steps in the calibration process. A very narrow range between sites (e.g., only one or two taxa collected) or missing taxa (many zero values) can lead to insufficient variability between sites to allow discrimination among sites in different conditions (Stoddard et al., 2008).

### 5.4.2.2 Metric Reproducibility

For a metric to discriminate between sites in good and poor condition, low sampling variation (noise) is crucial. The sampling variation should be minimal compared to the amongsite differences (signal) and metrics with high $\mathrm{S} / \mathrm{N}$ values tend to have higher consistency in metric response to stressors than those with low S/N ratios (Stoddard et al., 2008). Although there are no fixed thresholds for eliminating metrics based on $\mathrm{S} / \mathrm{N}$ ratios, an $\mathrm{S} / \mathrm{N}$ value $<1$ suggests that visiting a single site twice returns as much metric variability as visiting two different sites (Stoddard et al., 2008; Stoddard et al., 2005). The threshold chosen should depend on the inherent variability in the assemblage being assessed (Stoddard et al., 2008; Stoddard et al., 2005). Stoddard and others (2008) have suggested that fish metrics tend to have high S/N values (e.g., 4 or 5) and, as such, a higher threshold for rejection. This is contrary to assemblages such as periphyton, which tend to have low S/N values (e.g., 1 or 1.5 ) and a low threshold for rejection. Macroinvertebrate metrics tend to be more intermediate (e.g., $\mathrm{S} / \mathrm{N}$ values $=2.0$ ) (Stoddard et al., 2008).

Ideally, metric temporal variability would be analyzed at only the reference site population; however, due to the relatively low number of sites classified as low stress in this study, I had to look at the distribution of metrics at all sites. I found the fish metrics to be highly variable through time (very low $\mathrm{S} / \mathrm{N}$ ratios), an indication that there was high sampling variation in sites between years. To get enough metrics for the final IBI I had to choose a threshold for removal of $\mathrm{S} / \mathrm{N}<0.5$ (extremely low), and therefore even the selected metrics had high variability. The most likely explanation for this is the fluctuating and extreme differences in water levels experienced between 2016, 2017, and 2018 field seasons, yielding differences in the fish stream assemblage. Sampling had to be delayed nearly a month for two of the sites (Alcott Creek and Sukaw Creek) due to high water levels. While I was able to sample the other three sites in similar time frames between the three years, there was a difference in water levels and flow (highest in 2017). The flooding in 2017 likely can explain some of this variability in fish metrics between years.

### 5.4.2.3 Metric Redundancy

The metric redundancy step is used to avoid inclusion of metrics that share highly similar biological or taxonomic information and/or are highly correlated (Hughes et al. 1998; Lyons et
al. 2001; Stevens \& Council 2008). I initially followed Stoddard et al. (2008) and considered metrics redundant when the Pearson correlation coefficient (r) was $\geq 0.7$, as this equates to metrics sharing approximately half their information content ( $\mathrm{r}>0.71, \mathrm{R} 2=0.5$, Stoddard et al., 2008). However, correlation analysis indicated strong levels of multicollinearity among all except two metrics and even restricting the removal criteria to $r \geq 0.8$, I was only able to obtain at most three metrics that were not highly correlated. The low species richness found in the region may explain the high levels of multicollinearity between metrics, as only a few fish species made up many individual metrics. Interestingly, although multicollinearity of metrics was an issue for other IBIs in nearby depauperate regions, other studies were able to obtain 5+ metrics that were not highly redundant for their final IBI (Stevens et al., 2006, 2010; Cantin \& John, 2012). To obtain enough metrics for the final index, I had to lessen the importance of the metric redundancy, metric range, and metric reproducibility steps during the metric selection process. Although many studies report multicollinearity of metrics as a limitation of the IBI, others (Fausch et al., 1990; Minns et al., 1994) discussed the importance and potential strength that redundancy can add to an IBI, as although metrics may appear redundant, they may provide unique information to the index (Minns et al., 1994).

### 5.4.2.4 Metric Responsiveness

None of the candidate metrics in this study were significantly different between the low and high stress sites, indicating a lack of responsiveness to the human stressors evaluated in this study. This could be due to several factors. As discussed above, it is possible that sites were inaccurately assigned to stress classes, or additional unaccounted-for stressors may be affecting the fish community in my study and contributing to the lack of significant differences between the low and high stress sites. Other factors, including the low species richness and relatively tolerant assemblages of the region, may be confounding the effects of human influence on stream fish communities. As discussed previously, the current study lacked a large sample size of low and high stress sites, even after maintaining all the sites as one grouping. A higher number of sample sites should be used to further assess fish community metric response to human disturbance in Boreal Plains environments.

## Chapter 6: SYNTHESIS AND RECOMMENDATIONS

### 6.1 Conclusions

The overall goal of this research was to adapt, apply, and critically evaluate a fish communitybased ecological assessment tool, known as the Index of Biotic Integrity, for Boreal Plain lotic environments that undergo a range of stressors. The Boreal Plain ecozone is a transition zone, between prairie and boreal forest, and as such, is influenced by anthropogenic disturbances distinct to both prairie and boreal forest ecosystems, providing a unique opportunity to assess and evaluate a fish-based IBI in a relatively homogenous area with multiple, and prevalent, landuse stressors. Through this research, I was able to develop and apply a fish-based IBI for streams and rivers in the Beaver River watershed located in the Boreal Plain ecozone of SK, Canada. By assessing various measures of land use and fish habitat, I determined minimally disturbed (or low-stress) conditions and developed a gradient of stream and river health (e.g., low, moderate, and high stress) throughout the Beaver River watershed. The primary objectives of this research were to determine the expected fish community structure and fish condition in minimally disturbed streams in northern SK, to determine if fish community structure and fish condition vary with a gradient of human disturbance by applying and evaluating a fish-based IBI, and to determine the sensitivity of IBI methods and results to inter-annual variability.

For objective one, I expected to see fish species richness and abundance increase from headwater to higher-order streams. I was able to identify general relationships between natural gradients (stream order, drainage area, and wetted stream width) and fish species richness, abundance, and the nine IBI metrics. Using the wetted stream width to be representative of stream size, I was only able to detect a significant relationship between one fish metric, the $\log _{10}$ percent individuals with parasites, and wetted width, likely a result of the low sample size for minimally disturbed/low stress streams. I also hypothesized that fish community structure and fish condition will have the least amount of variation among minimally disturbed streams and rivers and that the fish community will show greater variability with higher stress. Within the low stress sites, there was considerable variability in metric and index scores, indicating high variability in the fish community structure and condition of low stress streams and rivers. Contrary to my expectations, I found a similar amount of variability between the low, moderate, and high stress sites, indicating that the fish community was not more variable with higher stress. However, there were significantly different variances between the low and high stress sites for
two of the nine fish community metrics used to develop the IBI, the $\log _{10}$ percent white sucker and the percent brook stickleback metrics. Both metrics had a higher variability in the high stress sites. Additionally, I also hypothesized that fish condition will be of the highest quality in minimally disturbed conditions. I found that fish condition, as assessed through IBI scores, had the highest scores among low-stress streams and rivers as expected, but the index scores were only significantly different between the low and moderate stress sites. Assessing additional measures of fish condition, including the body condition of white sucker, the percent of the fish community with abnormalities, and the percent of the fish community with parasites metrics, showed no significant differences in fish condition between the minimally disturbed (low stress) and high stress sites or moderate stress sites; however, the percent of the fish community with parasites was significantly different between the moderate and high stress sites.

The second objective of this study was to determine if fish community structure and fish condition vary with a gradient of human disturbance by applying and evaluating a fish-based IBI. I hypothesized that fish community structure and fish condition will change with gradients of human disturbance. Additionally, I hypothesized that impacted sites would have lower species richness and abundance, a higher percentage of tolerant species, and fish with a higher frequency of abnormalities. I was able to establish a gradient of stream disturbance using various environmental parameters and surrounding land use conditions for each stream and river sampled. Human influence, determined by environmental conditions, increased as expected from low to high stress sites. This study identified nine fish assemblage metrics, selected across the major metric categories, that showed consistent relationships with and the highest responsiveness to human disturbance. Using these nine metrics, I was able to create an IBI that, as best as possible, describes the fish community composition and structure of the region. I was able to show that fish community structure and condition, as indicated by IBI scores, do follow the expected trend with human disturbance. Although I was unable to detect a statistically significant difference in IBI scores between low and high stress sites, due to large variability within groups and small samples sizes, I was able to find a statistically significant difference in the mean IBI scores between the low stress and moderate stress sites, indicating a change in fish community structure and condition with increasing anthropogenic stress. When assessing fish metrics individually, none of the nine fish community metrics assessed showed a significant difference between low and high stress sites; however, the percent of top carnivores and piscivores metric
and the percent fish with parasites metric were significantly different between low and moderate, and the moderate and high stress sites, respectively.

The third objective of this research was to determine the sensitivity of the IBI to interannual variability. Any biological monitoring method must have the ability to distinguish between biotic changes brought on by anthropogenic disturbances and those brought on by cyclical, natural changes in the environment. For this objective, I revisited five of the sample sites annually over the course of three years and resampled fish habitat and water chemistry variables, as well as the fish community present. I used this data to develop fish metric and IBI scores for each year to assess potential variability in Boreal Plain stream fish communities and habitat. I hypothesized that fish communities, fish condition, water quality, and habitat variables will show interannual variance within sites, due to differences in environmental conditions and fish residency and mobility between years; however, these differences will not be reflected in IBI scores, with greater variance among sites than within sites. As expected, I found considerable variability in environmental data and fish communities between years. Despite the large fluctuations in environmental conditions in northern boreal regions with environmental extremes and immense seasonal variability, I was still able to detect significant differences among some of the sites, indicating that variability among sites was higher than within sites for some of the environmental and fish community data. I was able to detect significant differences among sites for fish species richness and abundance metrics, but only two fish community metrics used to develop the IBI, and no significant differences among sites were found when assessing IBI scores. Therefore, majority of the most responsive metrics in this study were highly variable through time and unlike our expectations, the overall index displayed greater temporal variability across years than spatial variability among sites when assessing data across the three sampling dates.

### 6.2 Limitations and Future Recommendations

Some of the shortcomings of this research were the small sample size of the low and high stress streams and rivers, the lack of an independent dataset to further evaluate the IBI across space, and sampling across different seasons (summer and autumn). Due to logistical and time constraints of this project, I was unable to sample more sites within each ecoregion of the Boreal Plain ecozone and across all levels of disturbance. More sites would have allowed me to better
minimize and evaluate the spatial heterogeneity of the region. Future research should consider using a larger sample set to develop and model relationships between boreal plains stream fish communities and environmental data, including natural gradients such as stream size. A better understanding of the important drivers of stream fish communities in this understudied region would help inform future management of aquatic resources in the area. A larger sample size would have also allowed me to develop an independent dataset (different from the revisited sites) to further evaluate the IBI over larger spatial regions and to verify and ensure the accuracy and consistency of the IBI's results independent of the dataset used in its development. Additionally, I suggest that future studies be restricted to an even shorter time frame than that used here (e.g., only sampling in later summer) or to develop season specific indices to avoid seasonal bias and subsequent changes in environmental and fish community data. In addition to the above, this study demonstrates the importance of longer-term studies (e.g., over multiple years) to analyze the cyclical and natural trends in boreal stream fish communities and habitat through time. Furthermore, I suggest comparing this SK-based IBI with other commonly used ecosystem monitoring assemblages and methods in the region, such as benthic macroinvertebrates and environmental DNA analysis techniques.

Despite the depauperate, generalist fish assemblage and naturally harsh nature of the region, I was able to develop, adapt, and evaluate a fish community-based IBI for the Beaver River watershed; yet, due to the small sample size and high intragroup variability, I was unable to detect a significant difference in index scores between reference and high stress sites. The fish community displayed substantial variability amongst all stress classes and analysis of the index over a three-year period revealed more temporal variation than spatial variability among locations, making effects from anthropogenic disturbance indistinguishable from natural variability through time. The province of SK, as well as Canada's boreal region, currently do not have a fish-based Index of Biotic Integrity (IBI) framework. Canada is one of the most freshwater-rich places in the world and, despite its limitations, the creation and assessment of this tool should complement existing monitoring programs and provide insight into alternative approaches managing fisheries and aquatic resources in SK and northern Canada as a whole.

## Literature Cited

Abell, R. (2002). Conservation Biology for the Biodiversity Crisis: A Freshwater Follow-up. Conservation Biology, 16(5), 1435-1437. http://www.jstor.org/stable/3095340

Acton, D. F., Padbury, G. A. and Stushnoff, C. T. (1998). The Ecoregions of Saskatchewan. Regina: Canadian Plains Research Center.

Aguiar, F. C., Segurado, P., Urbanič, G., Cambra, J., Chauvin, C., Ciadamidaro, S., Dörflinger, G., Ferreira, J., Germ, M., Manolaki, P., Minciardi, M. R., Munné, A., Papastergiadou, E., \& Ferreira, M. T. (2014). Comparability of river quality assessment using macrophytes: a multi-step procedure to overcome biogeographical differences. The Science of the Total Environment, 476-477, 757-767.
https://doi.org/10.1016/J.SCITOTENV.2013.10.021
Albanese, B., Angermeier, P. L., \& Dorai-Raj, S. (2004). Ecological correlates of fish movement in a network of Virginia streams. Canadian Journal of Fisheries and Aquatic Sciences. 61, 857-869.

Allan, J. D. (2004). Landscapes and riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology and Evolutionary Systematics. 35, 257- 284.
Ali, S., Wali, A. F., Yatoo, A. M., Majid, S., Rasool, S., Khan, R., Ali, M. N., Wani, J. A., Farooq, S., Rasool, S., Wani, H. A., \& Rehman, M. U. (2020). Effect of pesticides on fish fauna: Threats, challenges, and possible remedies. Bioremediation and Biotechnology: Sustainable Approaches to Pollution Degradation, 27-54. https://doi.org/10.1007/978-3-030-35691-0_2/FIGURES/4

Alford, J. B., \& Gotwald, H. S. (2019). Associations Between Fish and Benthic
Macroinvertebrate Biotic Integrity and Non-Point Source Pollution Estimates in the Nolichucky River Watershed. Journal of the Tennessee Academy of Science, 94(1-2), 5671. https://go-galecom.cyber.usask.ca/ps/i.do?p=AONE\&sw=w\&issn=0040313X\&v=2.1\&it=r\&id=GALE\%7 CA598537945\&sid=googleScholar\&linkaccess=fulltext

Amoatey, P., \& Baawain, M. S. (2019). Effects of pollution on freshwater aquatic organisms. Water Environment Research, 91(10), 1272-1287. https://doi.org/10.1002/WER. 1221

Andersen, J. H., Aroviita, J., Carstensen, J., Friberg, N., Johnson, R. K., Kauppila, P., Lindegarth, M., Murray, C., \& Norling, K. (2016). Approaches for integrated assessment
of ecological and eutrophication status of surface waters in Nordic Countries. Ambio, 45(6), 681-691. https://doi.org/10.1007/S13280-016-0767-8/TABLES/3

Andreasen, J. K., O’Neill, R. V., Noss, R., and Slosser, N. C. (2001). Considerations for the development of a terrestrial index of ecological integrity. Ecological Indicators. 1,21-35.

Angermeier, P. L. \& Karr, J. R. (1984). Relationships between woody debris and fish habitat in a small warmwater stream. Transactions of the American Fisheries Society. 113, 716-726.
Angermeier, P. L., and Karr J. R. (1986). Applying an index of biotic integrity based on streamfish communities: considerations in sampling and interpretation. North American Journal of Fisheries Management. 6, 418-429.

Archer, R. W., Christopher, P., Lorenz, J., \& Jones, K. E. (2010). Monitoring and Assessing Marsh Habitat Health in the Niagara River Area of Concern. Final Project Report. Environment Canada-Great Lakes Sustainability Fund. www.birdscanada.org
Aron, P. G., Poulsen, C. J., Fiorella, R. P., \& Matheny, A. M. (2019). Stable Water Isotopes Reveal Effects of Intermediate Disturbance and Canopy Structure on Forest Water Cycling. Journal of Geophysical Research: Biogeosciences, 124(10), 2958-2975. https://doi.org/10.1029/2019JG005118

Ashcroft, P., Duffy, M., Dunn, C., Johnston, T., Koob, M., Merkowsky, J., Murphy, K., Scott, K., \& Senik, B. (2006). The SK Fishery: History and Current Status. Technical report 2006-2. Saskatchewan Environment.

Atton, F. M. and J. J. Merkowsky. (1983). Atlas of Saskatchewan Fish. Saskatchewan Dept. of Parks and Renewable Resources, Regina, Sask. Fish Tech. Rep. 83-2. 281pp.
Bacigalupi, J., Staples, D. F., Treml, M. T., and Bahr, D. L. (2021). Development of fish-based indices of biological integrity for Minnesota lakes. Ecological Indicators, 125, 107512.

Bailey, R. C., Norris, R. H., \& Reynoldson, T. B. (2004). Bioassessment of freshwater ecosystems: Using the reference condition approach. Kluwer Academic Publishers. DOI:10.1007/978-1-4419-8885-0

Baker, E. A., Wehrly, K. E., Seelbach, P. W., Wang, L., Wiley, M. J., \& Simon, T. (2005a). A Multimetric Assessment of Stream Condition in the Northern Lakes and Forests Ecoregion Using Spatially Explicit Statistical Modeling and Regional Normalization. Transactions of the American Fisheries Society, 134(3), 697-710. https://doi.org/10.1577/T03-205.1

Bambi, P., Tonin, A. M., Rezende, R. de S., Vieira, F. C., Graciano Miranda, F. G., Boyero, L., \& Gonçalves Júnior, J. F. (2023). The legacy of forest logging on organic matter inputs and storage in tropical streams. Biotropica, 55(1), 40-52.
https://doi.org/10.1111/BTP. 13155
Barbour, M. T., and Stribling, J. B. (1991). Use of habitat assessment in evaluating the biological integrity of stream communities. Proceedings of symposium, Biological Criteria: Research and Regulation. Hyatt Regency Crystal City, Arlington, Virginia.

Barbour, M. T., and Stribling, J. B. (1994). A technique for assessing stream habitat structure. 156-178, In Proceedings of "Riparian ecosystems of the humid U.S. and management, function and values." National Association of Conservation Districts. Washington, DC.
Barbour, M. T., Stribling, J. B. and Karr J. R. (1995). Multimetric approach for establishing biocriteria and measuring biological condition. Pages 63-77 in W.S. Davis and T.P. Simon (eds.). Biological assessment and criteria. Tools for water resource planning and decision making. Lewis Publishers, Boca Raton, Florida.

Barbour, M. T., Faulkner, C., \& Gerritsen, J. (1999). Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates, and Fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency, Office of Water, Washington, D.C. http://www.epa.gov/OWOW/monitoring/techmon.html

Birk, S., Chapman, D., Carvalho, L., Spears, B. M., Andersen, H. E., Argillier, C., Auer, S., Baattrup-Pedersen, A., Banin, L., Beklioğlu, M., Bondar-Kunze, E., Borja, A., Branco, P., Bucak, T., Buijse, A. D., Cardoso, A. C., Couture, R. M., Cremona, F., de Zwart, D., and Hering, D. (2020). Impacts of multiple stressors on freshwater biota across spatial scales and ecosystems. Nature Ecology \& Evolution, 4(8), 1060-1068.
https://doi.org/10.1038/s41559-020-1216-4
Blocksom, K. A., Kurtenbach, J. P., Klemm, D. J., Fulk, F. A., \& Cormier, S. M. (2002). Development and evaluation of the lake macroinvertebrate integrity index (LMII) for New Jersey lakes and reservoirs. Environmental Monitoring and Assessment, 77(3), 311333. https://doi.org/10.1023/A:1016096925401/METRIC

Blocksom, K. A. (2003). A performance comparison of metric scoring methods for a multimetric index for Mid-Atlantic Highlands streams. Environmental Management, 31, 670-682.

Borgwardt, F., L. Robinson, D. Trauner, H. Teixeira, A.J.A. Nogueira, A.I. Lillebø, G. Piet, M. Kuemmerlen, et al. (2019). Exploring variability in environmental impact risk from human activities across aquatic ecosystems. Science of the Total Environment. 652, 1396-1408. https://doi.org/10.1016/j.scitotenv.2018.10.339.

Bowman, M. F., \& Somers, K. M. (2006). Evaluating a novel Test Site Analysis (TSA) bioassessment approach. Am. Benthol. Soc, 25(3), 712-727. https://doi.org/10.1899/08873593

Bramblett, R. G., Johnson, T. R., Zale, A. v., \& Heggem, D. G. (2005). Development and Evaluation of a Fish Assemblage Index of Biotic Integrity for Northwestern Great Plains Streams. Transactions of the American Fisheries Society, 134(3), 624-640. https://doi.org/10.1577/t04-051.1

Brauns, M., Allen, D. C., Boëchat, I. G., Cross, W. F., Ferreira, V., Graeber, D., Patrick, C. J., Peipoch, M., von Schiller, D., \& Gücker, B. (2022). A global synthesis of human impacts on the multifunctionality of streams and rivers. Global Change Biology, 28(16), 47834793. https://doi.org/10.1111/GCB. 16210

Bronmark, C., Hulthen, K., Nilsson, A., Skov, C., Hansson, L., Broderson, J., Chapman, B. (2013). There and back again: Migration in freshwater fishes. Canadian Journal of Zoology. 92(6), 1-13. DOI:10.1139/cjz-2012-0277

Bunn S.E., and Arthington, A.H. (2002). Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environ Manag., 30. 492-507.

Burdon, F., McIntosh, A., \& Harding, J. (2013). Habitat loss drives threshold response of benthic invertebrate communities to deposited sediment in agricultural streams. Ecological Applications: A Publication of the Ecological Society of America., 23, 1036-1047.

Canadian Council of Ministers of the Environment. (2016). Guidance Manual for Environmental Site Characterization in Support of Environmental and Human Health Risk Assessment. Volume 1 Guidance Manual.

Canadian Parks and Wilderness Society (CPAWS) Southern Alberta Chapter. (2020). The Effects of Forest Harvest on Hydrology: Examining the Effects of Logging on Hydrological Processes, Streamflow, and Flood Risk in Alberta's Southern Eastern Slopes. Available at: https://cpaws-southernalberta.org/wp-content/uploads/2021/06/The-effects-of-forest-harvest-on-hydrology.pdf

Cambi, M., Certini, G., Neri, F., Marchi, E. (2015). The impact of heavy traffic on forest soils: A review. Forest Ecology and Management, 338, 124-138.
Cantin, A., and John, T. (201)2. A fish-based index of biological integrity for assessing ecological condition of the Beaver River watershed. Technical Report, T-2012-001, produced by Alberta Conservation Association, Sherwood Park, Alberta, Canada. 49pp.

Cao, Y., Hawkins, C. P., Olson, J., \& Kosterman, M. A. (2007). Modeling natural environmental gradients improves the accuracy and precision of diatom-based indicators. Journal of the North American Benthological Society, 26(3), 566-585. https://doi.org/10.1899/06-078.1
Carpenter, S., N.F., Caraco, D.L., Correll, R.W., Howarth, A.N., Sharpley, and V.H., Smith. (1998). Nonpoint Pollution of Surface Waters with Phosphorus and Nitrogen. Ecological Applications, 8(3). DOI:10.2307/2641247

Carter, L. (2012). Canadian Aquatic Biomonitoring Network, field manual - wadeable streams. Dartmouth, NS: Environment Canada.

Carter, J. L., \& Resh, V. H. (2013). Analytical Approaches Used in Stream Benthic Macroinvertebrate Biomonitoring Programs of State Agencies in the United States. U.S. Geological Survey Open-File Report 2013-1129, 50 p. DOI: 10.3133/ofr20131129
Cavallaro, M. C., Main, A. R., Liber, K., Phillips, I. D., Headley, J. V, Peru, K. M., Morrissey, C. A. (2019). Neonicotinoids and other agricultural stressors collectively modify aquatic insect communities. Chemosphere, 226, 945-955.
https://doi.org/10.1016/j.chemosphere.2019.03.176
Chainho, P., Chaves, M. L., Costa, J. L., Costa, M. J., \& Dauer, D. M. (2008). Use of multimetric indices to classify estuaries with different hydromorphological characteristics and different levels of human pressure. Marine Pollution Bulletin, 56(6), 1128-1137.

Chará-Serna, A. M., Epele, L. B., Morrissey, C. A., \& Richardson, J. S. (2019). Nutrients and sediment modify the impacts of a neonicotinoid insecticide on freshwater community structure and ecosystem functioning. The Science of the Total Environment, 692, 12911303. https://doi.org/10.1016/J.SCITOTENV.2019.06.301

Chislock, M. F., Doster, E., Zitomer, R. A., \& Wilson, A. E. (2013). Eutrophication: Causes, Consequences, and Controls in Aquatic Ecosystems. Nature Education Knowledge, 4(4), 0-10. https://www-nature-com.cyber.usask.ca/scitable/knowledge/library/eutrophication-causes-consequences-and-controls-in-aquatic-102364466/

Collen, B., Whitton, F., Dyer, E. E., Baillie, J. E. M., Cumberlidge, N., Darwall, W. R. T., Pollock, C., Richman, N. I., Soulsby, A. M., and Böhm, M. (2014). Global patterns of freshwater species diversity, threat and endemism. Global Ecology and Biogeography, 23, 40-51.

Cooper, M.J., Lamberti, G.A., Moerke, A.H., Ruetz, C.R., Wilcox, D.A., Brady, V.J., Brown, T.N., Ciborowski, J.J.H., Gathman, J.P., Grabas, G.P., Johnson, L.B., and Uzarski, D.G. (2018). An expanded fish-based index of biotic integrity for Great Lakes coastal wetlands. Environ. Monit. Assess., 190, 580https://doi.org/10.1007/s10661-018-6950-6

Côté, I. M., Darling, E. S., and Brown, C. J. (2016). Interactions among ecosystem stressors and their importance in conservation. Proc R Soc B, 283(1824). DOI:
https://doi.org/10.1098/rspb.2015.2592
Costanza, R., and Mageau, M. (1999). What is a healthy ecosystem? Aquatic Ecology, 33, 105115.

Costanza, R., R. de Groot, P. Sutton, S. van der Ploeg, S.J. Anderson, I. Kubiszewski, S. Farber, and R.K. Turner. (2014). Changes in the global value of ecosystem services. Global Environmental Change, 26, 152-158. https://doi.org/10.1016/j.gloenvcha.2014.04.002.

Couture, T., \& Biron, P. M. (2023). Morphological Quality Index (Mqi), Fish Communities and Biotic Integrity in Agricultural Streams. https://doi.org/10.2139/SSRN. 4509011

Craig, L. S., Olden, J. D., Arthington, A. H., Entrekin, S., Hawkins, C. P., Kelly, J. J., Kennedy, T. A., Maitland, B.M., Rosi, E. J., Roy, A. H., Strayer, D. L., Tank, J. L., West, A. O. \& Wooten, M. S. (2017). Meeting the challenge of interacting threats in freshwater ecosystems: a call to scientists and managers. Elementa Science of the Anthropocene. 5, 1-15.

Davies, P. E. (1994). Monitoring river health initiative, river bioassessment manual. National River Processes and Management Program (Freshwater Systems: Tasmania).
Davis, W. S., Snyder, B. D., Stribling, J.B., and Stoughton, C. (1996). Summary of State Biological Assessment Programs for Streams and Wadeable Rivers. EPA 230-R-96-007. U.S. Environmental Protection Agency; Office of Policy, Planning, and Evaluation; Washington, DC.

Davies, H., and Hanley, P.T. (2010). State of the Watershed Report. Saskatchewan Watershed Authority. 39 pp .

Department of Fisheries and Oceans (DFO). (2000). Effects of sediment on fish and their habitat. DFO Pacific Region Habitat Status Report 2000/01. Available at: https://waves-vagues.dfo-mpo.gc.ca/library-bibliotheque/255660.pdf
DeShon, J. E. (1995). Development and application of the invertebrate community index (ICI). In Davis W.S. \& and Simon T.P. (Eds.), Biological assessment and criteria: Tools for water resource planning and decision making. (pp. 217-243). Lewis Publishers.

Dodds, W. K., Gido, K., Whiles, M. R., Fritz, K. M., and Matthews, W. J. (2004). Life on the edge: the ecology of Great Plains prairie streams. Bioscience. 54, 205-216.

Doi, H., Inui, R., Akamatsu, Y., Kanno, K., Yamanaka, H., Takahara, T., Minamoto, T. (2017). Environmental DNA analysis for estimating the abundance and biomass of stream fish. Freshwater Biology. 62, 30-39.

Donald, D. B., Gurprasad, N. P., Quinnett-Abbott, L., \& Cash, K. (2001). Diffuse geographic distribution of herbicides in northern prairie wetlands. In Environmental Toxicology and Chemistry, 20(2).
Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z. I., Knowler, D. J., Lévêque, C., Naiman, R. J., Prieur-Richard, A. H., Soto, D., Stiassny, M. L. J., \& Sullivan, C. A. (2006). Freshwater biodiversity: importance, threats, status and conservation challenges. Biological Reviews, 81(2), 163-182. https://doi.org/10.1017/S1464793105006950

Eakins, R. J. (2022). Ontario Freshwater Fishes Life History Database. Version 5.28. Online database. (https://www.ontariofishes.ca), accessed 20 July 2023. https://ontariofishes.ca/home.htm

Ecological Stratification Working Group (Canada). (1996). A national ecological framework for Canada. Agriculture and Agri-Food Canada, Research Branch, Centre for Land and Biological Resources Research and Environment Canada, State of the Environment Directorate, Ecozone Analysis Branch, Ottawa, ON.
Environment and Climate Change Canada (ECCC). (2016). Canadian environmental sustainability indicators: risk to soil and water quality from agriculture. Consulted on February 12, 2023. Available at: www.ec.gc.ca/indicateurs-indicators/default.asp?lang=en\&n=30607EED-1.

Environment and Climate Change Canada (ECCC). (2020). Canadian Environmental Sustainability Indicators: Water quality in Canadian rivers. Consulted on Feb 02, 2019.

Available at: www.canada.ca/en/environment-climate-change/services/environmental-indicators/water-quality-canadian-rivers.html.

Environment and Climate Change Canada (ECCC). (2022). Canadian Environmental Sustainability Indicators: Pulp and paper effluent quality. Consulted on March 30th, 2023. Available at: www.canada.ca/en/environment-climate-change/services/environmental-indicators/pulp-papereffluent-quality.html.

Environment Canada. (2010). Pulp and Paper Environmental Effects Monitoring (EEM) Technical Guidance Document. Government of Canada, Ottawa. 490 pp.

Environment Canada. (2012). Metal mining technical guidance for environmental effects monitoring. Government of Canada, Ottawa. 550 pp.

Environment Canada. (2012). Canadian Aquatic Biomonitoring Network: Field Manual for Wadeable Streams, 57 p.

Erkinaro, J., Julkunen, M., and Niemela, E. (1998). Migration of juvenile Atlantic salmon Salmo salar in small tributaries of the subarctic River Teno, northern Finland. Aquaculture, 168, 105-119.

Evans, N.T., Shirey, P.D., Wieringa, J.G., Mahon, A.R. (2017). Comparative Cost and Effort of Fish Distribution Detection via Environmental DNA Analysis and Electrofishing. Fisheries, 42(2), 90-99. DOI:10.1080/03632415.2017.1276329

Everard, M., Fletcher, M. S., Powell, A., \& Dobson, M. K. (2011). The Feasibility of Developing Multi-Taxa Indicators for Landscape Scale Assessment of Freshwater Systems. Freshwater Reviews, 4(1), 1-19. https://doi.org/10.1608/frj-4.1.129

Economou, A. N. (2002) Development, Evaluation \& Implementation of a Standardised Fishbased Assessment Method for the Ecological Status of European Rivers - A Contribution to the Water Framework Directive (FAME). Defining Reference Conditions (D3), Final Report.

Ewing, R.D. (1999). Diminishing Returns: Salmon Decline and Pesticides. (Biotech Research and Consulting, Inc., Corvallis, 1999), 55 p. Retrieved from http://www.krisweb.com/biblio/gen_open_ewing_1999_diminishing.pdf

Fausch, K., Lyons, J., Karr, J., and Angermeier, P. (1990). Fish communities as indicators of environmental degradation. American Fisheries Society Symposium, 8, 123-144.

Fitzsimmons, M. (2001). Effects of deforestation and reforestation on landscape spatial structure in boreal Saskatchewan, Canada. Forest Ecology and Management, 174(2003), 577-592.

Fore, L. S., Karr, J. R., \& Wisseman, R. W. (1996). Assessing invertebrate responses to human activities: Evaluating alternative approaches. Journal of the North American Benthological Society, 15(2), 212-231. https://doi.org/10.2307/1467949
Franssen, N.R., Gido, K.B., Guy, C.S., Tripe, J.A., Shrank, S.J., Strakosh, T.R., Bertrand, K.N., Franssen, C.M., Pitts, K.L., and Paukert C.P. (2006). Effects of floods on fish assemblages in an intermittent prairie stream. Freshwater Biology, 51(11), 2072-2086. DOI:10.1111/j.1365-2427.2006.01640.x

Frey, D. (1977). Biological integrity of water: an historical approach: RK Ballentine and LJ Guarraia (editors) - The Integrity of Water. Proceedings of a Symposium, 1975. U.S. Environmental Protection Agency. Washington, DC. pp 127-140.
Fuchs SA., Hinch, SG., and Mellin, E. (2003). Effects of Streamside Logging on Stream Macroinvertebrate Communities and Habitat in the Sub-Boreal Forests of British Columbia, Canada. Canadian Journal of Forest Research, 33(8), 1408-1415.
Gallant, A.L., Whittier, T.R., Larsen, D.P., Omernik, J.M., and Hughes, R.M. (1989). Regionalization as a Tool for Managing Environmental Resources. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR.

Gerritsen, J. (1995). Additive biological indices for resource management. Journal of the North American Benthological Society, 14(3), 451-457. https://doi.org/10.2307/1467211

Gibbons, D., Morrissey, C., Mineau, P. (2015). A review of the direct and indirect effects of neonicotinoids and fipronil on vertebrate wildlife. Environmental Science and Pollution Research International. 22(1), 103-118. DOI: 10.1007/s11356-014-3180-5. PMID: 24938819; PMCID: PMC4284370.

Giri, S., Sajiya Mujawar, B., Haragi, S., Rathod, J., Baban, Trivesh, I., Dhanya, M., Lal, M., Sri, Murugesan, H., Kiranya,, Bijukumar, B., Pramila, A., Ranjeet Kutty, S., Kumar, P., Bhavan, S. G., Mujawar, S., Mayekar, T, Kumar, P, and Appukuttannair, B. (2023). A multi-metric fish index to measure the ecological quality of tropical predominantly open estuaries along the western coast of India. Environmental Monitoring and Assessment, 195(3), 1-21. https://doi.org/10.1007/S10661-023-11013-2

Gliessman, S. R. (2014). Agroecology: The Ecology of Sustainable Food Systems, Third Edition. Agroecology: The Ecology of Sustainable Food Systems, Third Edition, 1-366. https://doi.org/10.1201/B17881/AGROECOLOGY-STEPHEN-GLIESSMAN

Government of Alberta. (2018). Environmental Quality Guidelines for Alberta Surface Waters. Water Policy Branch, Alberta Environment and Parks. Edmonton, Alberta.

Government of British Columbia. (2023a). Fish-Forestry Interaction Research. Available at: https://www2.gov.bc.ca/gov/content/environment/plants-animals-ecosystems/fish/aquatic-habitat-management/fish-forestry

Government of British Columbia. (2023b). Fish-Forestry Interaction Research. Slim-Tumuch Fish-Forestry Study, 1971-1975. Available at: https://www2.gov.bc.ca/assets/gov/environment/plants-animals-and-ecosystems/fish-fish-habitat/fish-forestry/slim-tumuch_fish-forestry_study.pdf

Government of Canada. (1985). The Fisheries Act, 1985. Retrieved from the Government of Canada Justice Laws website: http://laws-lois.justice.gc.ca/eng/acts/F-14/index.html.

Government of Canada. (2000). Canada National Parks Act, 2000. Retrieved from the Government of Canada Justice Laws website: http://laws-lois.justice.gc.ca/eng/acts/N-14.01/page-13.html.

Government of Canada. (2008). Technical guidance document for Water Quality Indicator practitioners reporting under the Canadian Environmental Sustainability Indicators (CESI) initiative 2008. Retrieved from the Government of Canada Publications webpage at https://publications.gc.ca/site/archivee-archived.html?url=https://publications.gc.ca/collections/collection_2011/ec/En4-138-2010-eng.pdf

Government of Saskatchewan. (1994). The Fisheries Act (Saskatchewan), 1994. Retrieved from the Government of Saskatchewan website: http://www.publications.gov.sk.ca/details.cfm?p=523.

Government of Saskatchewan. (1996). The Forest Resources Management Act. Chapter F-19.1. https://www.canlii.org/en/sk/laws/stat/ss-1996-c-f-19.1/latest/ss-1996-c-f-19.1.html
Government of Saskatchewan. (2019). Forestry in Saskatchewan. Forestry Development Branch. https://www.saskatchewan.ca/business/investment-and-economic-development/key-economic-sectors/forestry-development

Government of Saskatchewan. (2023). Saskatchewan Forestry Sector Overview. Saskatchewan Forestry Development. https://www.saskatchewan.ca/business/investment-and-economic-development/key-economic-sectors/forestry-development

Gratwicke, B., and Speight, M.R. (2004). The relationship between fish species richness, abundance and habitat complexity in a range of shallow tropical marine habitats. Journal of Fish Biology, 66, 650-667.

Griffith, M. B., \& Mcmanus, M. G. (2020). Consideration of spatial and temporal scales in stream restorations and biotic monitoring to assess restoration outcomes: A literature review, part 1. https://doi.org/10.1002/rra. 3692

Gupta, S. K., Roy, S., Chabukdhara, M., Hussain, J., \& Kumar, M. (2019). Risk of metal contamination in agriculture crops by reuse of wastewater: An ecological and human health risk perspective. In Water Conservation, Recycling and Reuse: Issues and Challenges (pp. 55-79). Springer Singapore. https://doi.org/10.1007/978-981-13-31794_3

Gurian-Sherman, D. (2008). CAFOs Uncovered: The Untold Costs of Confined Animal Feeding Operations. Cambridge, MA: Union of Concerned Scientists. www.ucsusa.org/assets/documents/food_and_agriculture/cafos-uncovered.pdf

Hakalahti, T., Bandilla, M., and Valtonen. E.T. (2005). Delayed transmission of a parasite is compensated by accelerated growth. Parasitology. 131, 647-656.

Hain, E. F., Nelson, S. A. C., Tracy, B. H., and Cakir, H. I. (2012). Application of GIS Techniques for Developing a Fish Index of Biotic Integrity for an Ecoregion with Low Species Richness. Southeastern Naturalist, 11(4), 711-732.

Hauer, C., Leitner, P., Unfer, G., Pulg, U., Habersack, H., Graf, W. (2018). The Role of Sediment and Sediment Dynamics in the Aquatic Environment. In: Schmutz, S., Sendzimir, J. (eds) Riverine Ecosystem Management. Aquatic Ecology Series, vol 8. Springer, Cham. https://doi.org/10.1007/978-3-319-73250-3_8

Herricks, E.E., and Schaeffer, D.J. (1985). Can We Optimize Biomonitoring? Environmental Management, 9(6), 487-492.

Hughes, R.M., Heiskary, S.A., Mathews, W.J., and Yoder, C.O. (1992). Use of Ecoregions in Biological Monitoring. Loeb, S.L. (ed), Biological monitoring of freshwater ecosystems. Chelsea, MI: Lewis Publishers.

Hughes, R.M., Kaufmann, P.K., Herlihy, A.T., Intelmann, S.S, Corbett, S.C., Arbogast, M.C., and Hjort, R.C. (2002). Electrofishing distance needed to estimate fish species richness in raftable Oregon rivers. North American Journal of Fisheries Management, 22(4), 12291240.

Hughes, R.M., Kaufmann, P.R., Herlihy, A.T., Kincaid, T.M., Reynolds, L. and Larsen, D.P. (1998). A process for developing and evaluating indices of fish assemblage integrity. Canadian Journal of Fisheries and Aquatic Sciences, 55, 1618-1631.

Hughes, R.M., Larsen, D.P. \& Omernik, J.M. (1986). Regional Reference Sites: A Method for Assessing Stream Potentials. Environmental Management, 10(5), 629-635.

Hilsenhoff, W. L. (1988). Rapid Field Assessment of Organic Pollution with a Family-Level Biotic Index. Journal of the North American Benthological Society, 7(1), 65-68.

Jha, R., and Diplas, P. (2017). Elevation: a consistent and physically-based framework for classifying streams. Journal of Hydraulic Research, 56(3), 299-312.

Johnson, L. B., Richards, C., Host, G. E., and J. W. Arthur. (1997). Landscape influences on water chemistry in Midwestern stream ecosystems. Freshwater Biology. 37, 193-208.

Joynt, A. and Sullivan, M. J. (2003). Fish of Alberta. Lone Pine Publishing, Edmonton, Alberta, Canada.

Karr, J. R. (1981). Assessment of biotic integrity using fish communities. Fisheries, 6(6), 21-27.
Karr, J. R., (2006). Seven foundations of biological monitoring and assessment. Biologia Ambientale, 20, 7-18.

Karr, J.R., and Dudley, D.R. (1981). Ecological Perspective on water quality goals. Environmental Management, 5, 55-68.

Karr, J.R., Fausch, K.D., Angermeier, P.L., Yant, P.R., Schlosser, I.J. (1986). Assessing biological integrity in running waters a method and its rationale. Illinois Natural History Survey Special Publications 5.

Karr, J. R., Yant, P. R., Fausch, K. D., \& Schlosser, I. J. (1987). Spatial and temporal variability of the index of biotic integrity in three Midwestern streams. Transactions of the American Fisheries Society, 116, 1-11.

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

Kaval, P. (2019). Integrated catchment management and ecosystem services: A twenty-five year overview. Ecosystem Services, 37: https://doi.org/10.1016/j.ecoser.2019.100912.
Keena, M., Meehan, M., \& Scherer, T. (2022). Environmental Implications of Excess Fertilizer and Manure on Water Quality. North Dakota State University.

Kerans, B. L., \& and Karr J.R. (1994). benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. Ecological Applications, 4, 768-785.

Kilgour, B. W., Francis, A. P., \& Mercier, V. (2013). Reducing the sensitivity of the water quality index to episodic events. Water Quality Research Journal of Canada, 48(1), 113. https://doi.org/10.2166/wqrjc.2013.008

Kim, J. H., Oh, H. M., Kim, I. S., Lim, B. J., \& An. (2013). Ecological Health Assessments of an Urban Lotic Ecosystem Using a Multi-metric Model along with Physical Habitat and Chemical Water Quality Assessments. Int. J. Environ. Res, 7(3), 659-668.

Klemm, D.J., and Lazorchak, J.M. (1995). Environmental Monitoring and Assessment Program Surface Waters: Field operations and methods for measuring the ecological condition of wadable streams. United States Environmental Protection Agency, Ecological Monitoring Research Division. Washington, DC.
Knackstedt, T. (2015). Boreal Watershed Management Strategy: Evaluation of the Lake Athabasca

Watershed. Final report for Master of Sustainable Environmental Management. University of Saskatchewan, Saskatoon, SK. 54 pp.

Kovalenko, K.E., Thomaz, S.M. \& Warfe, D.M. (2012). Habitat complexity: approaches and future directions. Hydrobiologia. 685, 1-17. https://doi.org/10.1007/s10750-011-0974-z

Krause, J.R., Bertrand, N., Kafle, A., Troelstrup, N. H. Jr. (2013). A fish index of biotic integrity for South Dakota's Northern Glaciated Plains Ecoregion. Ecological Indicators, 3(2013), 313-322.

Langeani, F., L., Casatti, H.S., Gameiro, A.B., do Carmo. (2005). Riffle and pool fish communities in a large stream of southeastern Brazil. Neotropical Ichthyology. 3(2), 305311. DOI:10.1590/S1679-62252005000200009

Larson, D.J. (1976). Stream habitat and forest harvesting. Sask. Dep. Tour. Renew. Resour., Fish. Branch. Tach. Rep. 76-3.

Lazorchak, J. M., Hill, B. H., Brown, B. S., Mccormick, F. H., Engle, V., Lattier, D. J., Bagley, M. J., Griffith, M. B., Maciorowski, A. F., \& Toth, G. P. (2002). Biomonitoring and bioindicator concepts needed to evaluate the biological integrity of aquatic systems. Markert, B.A., Breure, A.M., Zechmeister, H.G. (Eds.), Bioindicators and biomonitors (831-873). Boston: Elsevier.

Lee, S.-J., Lee, E.-H., \& An, K.-G. (2018). Lotic Ecosystem Health Assessments Using an Integrated Analytical Approach of Physical Habitat, Chemical Water Quality, and Fish Multi-Metric Health Metrics. Pol. J. Environ. Stud, 27(5), 2113-2131. https://doi.org/10.15244/pjoes/78044

Li, T., Huang, X., Jiang, X., \& Wang, X. (2018). Assessment of ecosystem health of the Yellow River with fish index of biotic integrity. Hydrobiologia, 814, 31-43. https://doi-org.cyber.usask.ca/10.1007/s10750-015-2541-5

Long, J. M. and Walker, D. J. (2005). Small scale application and assessment of a Index of Biotic Integrity for a large boreal river. Hydrobiologia. 544, 177-187.

Lyons, J. (1992). Using the index of biotic integrity (IBI) to measure environmental quality in warmwater streams of Wisconsin. General Technical Report NC-149. St. Paul, MN: U.S. Dept. of Agriculture, Forest Service, North Central Forest Experiment Station.
Macdonald, J.S., MacIsaac, E.A., and H.E. Herunter. (2003). The effect of variable-retention riparian buffer zones on water temperatures in small headwater streams in sub-boreal forest ecosystems of British Columbia, Can. J. For. Res, 33, 1371-1382.

MacDougall, M. J. (2014). Development and Evaluation of a Periphytic Diatom Biomonitoring Platform of the Assessment of Cumulative Effects in Lakes of the Muskoka River Watershed, Ontario, Canada. M.Sc. Thesis. University of Waterloo, Waterloo, Ontario, Canada. https://uwspace.uwaterloo.ca/handle/10012/8834

Mack, J. J. (2001). Vegetation Index of Biotic Integrity (VIBI) for Wetlands: ecoregional, hydrogeomorphic, and plant community comparisons with preliminary wetland aquatic life use designations. Ohio EPA Final Report to U.S. EPA Grant No. CD985875-01. Ohio Environmental Protection Agency, Wetland Ecology Group, Division of Surface. Testing Biological Metrics and Development of Wetland Assessment Techniques using Reference Sites. Ohio Environmental Protection Agency, Wetland Ecology Group, Division of Surface Water, Columbus, Ohio. Volume 1.

Mack, J. J. (2004). Integrated Wetland Assessment Program. Part 4: Vegetation Index of Biotic Integrity (VIBI) and Tiered Aquatic Life Uses (TALUs) for Ohio wetlands. Ohio EPA Technical Report WET / 2004-4. Ohio Environmental Protection Agency, Wetland Ecology Group, Division of Surface Water, Columbus, Ohio.

Main, A. R., Fehr, J., Liber, K., Headley, J. V., Peru, K. M., \& Morrissey, C. A. (2017). Reduction of neonicotinoid insecticide residues in Prairie wetlands by common wetland plants. Science of The Total Environment, 579, 1193-1202. https://doi.org/10.1016/J.SCITOTENV.2016.11.102
Main, A. R., Michel, N. L., Cavallaro, M. C., Headley, J. V., Peru, K. M., \& Morrissey, C. A. (2016). Snowmelt transport of neonicotinoid insecticides to Canadian Prairie wetlands. Agriculture, Ecosystems and Environment, 215, 76-84. https://doi.org/10.1016/J.AGEE.2015.09.011

Main, A. R., Michel, N. L., Headley, J. V., Peru, K. M., \& Morrissey, C. A. (2015). Ecological and Landscape Drivers of Neonicotinoid Insecticide Detections and Concentrations in Canada's Prairie Wetlands. Environmental Science \& Technology, 49(14), 8367-8376. https://doi.org/10.1021/ACS.EST.5B01287

Malaj, E., Freistadt, L., \& Morrissey, C. A. (2020a). Spatio-Temporal Patterns of Crops and Agrochemicals in Canada Over 35 Years. Front. Environ. Sci., 8(556452).

Malaj, E., Liber, K., \& Morrissey, C. A. (2020b). Spatial distribution of agricultural pesticide use and predicted wetland exposure in the Canadian Prairie Pothole Region. Science of the Total Environment, 718(134765). https://doi.org/10.1016/j.scitotenv.2019.134765

Malaj, E., \& Morrissey, C. A. (2022). Increased reliance on insecticide applications in Canada linked to simplified agricultural landscapes. Ecological Applications, 32(3). https://doi.org/10.1002/EAP. 2533

Mallin, M. A., Burkholder, J. M., McIver, M. R., Shank, G. C., Glasgow, H. B., Touchette, B. W., \& Springer, J. (1997). Comparative Effects of Poultry and Swine Waste Lagoon Spills on the Quality of Receiving Streamwaters. Journal of Environmental Quality, 26(6), 1622-1631. https://doi.org/10.2134/JEQ1997.00472425002600060023X
Mamun, M., \& An, K. G. (2020). Stream health assessment using chemical and biological multimetric models and their relationships with fish trophic and tolerance indicators. Ecological Indicators, 111, 106055. https://doi.org/10.1016/J.ECOLIND.2019.106055

Martin, C.W., Hornbeck, J.W., Likens, G.E., and Buso, D.C. (2000). Impacts of intensive harvesting on hydrology and nutrient dynamics of northern hardwood forests. Can. J. Fish. Aquat. Sci., 57 (Suppl. 2), 19-29.

Massie, M. (2014). Forest Prairie Edge: Place History in Saskatchewan. University of Manitoba Press. 344 pp.
Mateo-Sagasta J, Zadeh SM, \& Turral H. (2017). Water pollution from agriculture: a global review. In the Food and Agriculture Organization of the United Nations Rome, 2017 and the International Water Management Institute on behalf of the Water Land and Ecosystems research program Colombo, 2017.

McAllister, D. E., Hamilton, A. L., and Harvey, B. (1997). Global freshwater biodiversity: striving for the integrity of freshwater systems. Sea Wind., 11, 1-140.

McCormick, F. H., Hughes, R. M., Kaufmann, P. R., Peck, D. V., and Stoddard, J. L. (2001). Development of an Index of Biotic Integrity for the Mid-Atlantic Highlands Region. Transactions of the American Fisheries Society, 130, 857-877.

Meador, M. R., Whittier, T. R., Goldstein, R. M., Hughes, R. M., \& Peck, D. V. (2008). Evaluation of an Index of Biotic Integrity Approach Used to Assess Biological Condition in Western U.S. Streams and Rivers at Varying Spatial Scales. Transactions of the American Fisheries Society, 137(1), 13-22. https://doi.org/10.1577/T07-054.1

Mebane, C.A., T.R., Maret, \& R.M., Hughes. (2003). An Index of Biological Integrity (IBI) for Pacific Northwest Rivers. Transactions of the American Fisheries Society. 132(2), 239261. DOI: 10.1577/1548 8659(2003)132<0239:AIOBII>2.0.CO;2

Merkowsky, A. (1998). Predicting the importance of boreal streams as fish habitat. Pages 169176 in M.K. Brewin and D.M.A. Monita, tech. cords. Forest-fish conference. Land management practices affecting aquatic eecosystems Proc. Forest-Fish Conf., May 1-4, 1996, Calgary, Alberta. Nat. Resour. Can. For. Serv., North. For. Cent., Edmonton, Alberta. Inf. Rep. NOR-X-356.

Messing, P. G., Farenhorst, A., Waite, D. T., McQueen, D. A. R., Sproull, J. F., Humphries, D. A., \& Thompson, L. L. (2011). Predicting wetland contamination from atmospheric deposition measurements of pesticides in the Canadian Prairie Pothole region.
Atmospheric Environment, 45(39), 7227-7234.
https://doi.org/10.1016/j.atmosenv.2011.08.074

Micacchion, M. (2002). Amphibian Index of Biotic Integrity (AmphIBI) for Wetlands. State of Ohio Environmental Protection Agency, Wetland Ecology Group Division of Surface Water. Columbus, OH.

Miller, G.T., and Spoolman, S.E. (2012). Living in the Environment, 17th edition. Belmont, CA: Brooks/Cole Cengage Learning.
Miller et al., 1988 Regional Applications of an Index of Biotic Integrity for Use in Water Resource Management. Fisheries, 13(5),12-20.

Miller, D. L., Hughes, R. M., Karr, J. r., Leonard, P. M., Moyle, P. B., Schrader, L. H., Thompson, B. A., Daniels, R. A., Fausch, K, Fitzhugh, G. A., Gammon, J. R., and Halliwell, D. (1988). Regional Applications of an Index of Biotic Integrity for Use in Water Resource Management. Fisheries, 13, 12-20. 10.1577/15488446(1988)013<0012:RAOAIO>2.0.CO;2.

Saskatchewan Ministry of Environment (MoE) and Saskatchewan Watershed Authority (SWA). (2012). Saskatchewan Northern Great Plains Ecosystem Health Assessment Manual 2012, Version 1.0. Saskatchewan Ministry of Environment, Regina, Saskatchewan, Canada.

Mamun, M., \& An, K. (2018). Ecological Health Assessments of 72 Streams and Rivers in Relation to Water Chemistry and Land-Use Patterns in South Korea. Turkish Journal of Fisheries and Aquatic Sciences, 18, 871-880. http://doi.org/10.4194/1303-2712v18_7_05

Minns, C.K., Cairns, V.W., Randall, R.G. and Moore, J.E. 1994. An index of biotic integrity (IBI) for fish assemblages in the littoral zone of Great Lakes areas of concern. Canadian Journal of Fisheries and Aquatic Sciences. 51: 1804-1822.

Minnesota Pollution Control Agency (MPCA). (2014). Development of a Fish-Based Index of Biological Integrity for Minnesota's Rivers and Streams. Document number wq-bsm203. Minnesota Pollution Control Agency, Environmental Analysis and Outcomes Division, St. Paul, MN.

Morris, M., and Somers, C. (2015). Saskatchewan Fishes: A Folding Pocket Guide to All Known Native and Introduced Species. Waterford Press. Florida.

Murthy, K.S., Kiran, B.R., and Venkateshwarlu, M. (2013). A review on toxicity of pesticides in fish. Int. J. Open Sci. Res. 1(1), 15-36.

Naigaga, I., Kaiser, H., Muller, W. J., Ojok, L., Mbabazi, D., Magezi, G., \& Muhumuza, E. (2011). Fish as bioindicators in aquatic environmental pollution assessment: A case study in Lake Victoria wetlands, Uganda. Physics and Chemistry of the Earth, 36(14-15), 918928. https://doi.org/10.1016/j.pce.2011.07.066

Natural Resources Canada. (2012). The effects of logging in riparian areas. Frontline Express, Canadian Forest Service - Great Lakes Forestry Center. Available at: https://cfs.nrcan.gc.ca/pubwarehouse/pdfs/33363.pdf

Nelson, J.S., and M.J. Paetz. 1992. The fishes of Alberta. The University of Alberta Press, University of Alberta, Edmonton, Alberta, Canada. 437 pp.

Newaz, S. (2009). Riparian habitat and vegetation recovery of headwater streams after clearcut harvesting. [Unpublished thesis submitted in partial fulfillment of the requirements for the Master of Science degree in Biology]. Lakehead University, Thunder Bay, Ontario.

Newcombe, C. P., \& Jensen, J. O. T. (1996). Channel Suspended Sediment and Fisheries: A Synthesis for Quantitative Assessment of Risk and Impact. North American Journal of Fisheries Management, 16, 693-727. https://afspubs-onlinelibrary-wiley-com.cyber.usask.ca/doi/epdf/10.1577/15488675\(1996\)016\<0693\%3ACSSAFA\>2.3.CO\%3B2

Ngor, P. B., Grenouillet, G., Phem, S., So, N., \& Lek, S. (2018). Spatial and temporal variation in fish community structure and diversity in the largest tropical flood-pulse system of South-East Asia. Ecology of Freshwater Fish, 27(4), 1087-1100. https://doi.org/10.1111/EFF. 12417

Nichols, S. J., Robinson, W. A., and Norris, R. H. (2010). Using the reference condition maintains the integrity of a bioassessment program in a changing climate. Journal of the North American Benthological Society., 29, 1459-1471. https://www.jstor.org/stable/45128433

Niu, L., Li, Y., Wang, P., Zhang, W., Wang, C., Li, J., \& Wu, H. (2018). Development of a microbial community-based index of biotic integrity (MC-IBI) for the assessment of ecological status of rivers in the Taihu Basin, China. Ecological Indicators, 85, 204-213. https://doi.org/10.1016/J.ECOLIND.2017.10.051

Norris, R. H. (1995). Biological monitoring: The dilemma of data analysis. Journal of the North American Benthological Society, 14(3), 440-450. https://doi.org/10.2307/1467210

Ode, P. R., Hawkins, C. P, and Mazor, R. D. (2008). Comparability of biological assessments derived from predictive models and multimetric indices of increasing geographic scope. Journal of the North American Benthological Society, 27, 967-985.

Ohio Environmental Protection Agency (Ohio EPA). (1987). Biological criteria for the protection of aquatic life: volumes I-III. Ohio Environmental Protection Agency, Columbus, Ohio.

Olds B.P., Jerde, C.L., Renshaw, M.A., Li, Y., Evans, N.T., Turner, C.R., Deiner, K., Mahon, A.R., Brueseke, M.A., Shirey, P.D., Pfrender, M.E., Lodge, D.M., and Lamberti, G.A. (2016). Estimating species richness using environmental DNA. Ecology and Evolution. 6(12), 4214-4226. doi: 10.1002/ece3.2186

Omernik, J.M. 1987. Ecoregions of the conterminous United States: Annals of the Association of American Geographers. 77(1): 118-25.

Omernik, J.M., and Griffith, G.E. 1991. Ecological regions versus hydrologic units--Frameworks for managing water quality: Journal of Soil and Water Conservation, 46(5): 334-340.

Ormerod, S. J., Dobson, M., Hildrew, A. G., \& Townsend, C. R. (2010). Multiple stressors in freshwater ecosystems. Freshwater Biology, 55(SUPPL. 1), 1-4. https://doi.org/10.1111/j.1365-2427.2009.02395.x

Pander, J., Knott, J., Mueller, M., \& Geist, J. (2019). Effects of environmental flows in a restored floodplain system on the community composition of fish, macroinvertebrates and macrophytes. Ecological Engineering, 132, 75-86.
https://doi.org/10.1016/J.ECOLENG.2019.04.003
Parmar, T. K., Rawtani, D., \& Agrawal, Y. K. (2016). Bioindicators: the natural indicator of environmental pollution. Frontiers in Life Science, 9(2), 110-118. https://doi.org/10.1080/21553769.2016.1162753

Phillips, I. D., McMaster, G., Chivers, D. P., and Bowman, F. M. (2023) Saskatchewan Condition Assessment of Lotic Ecosystems (SCALE): a multivariate tool for assessing the integrity of Northern Great Plains wadeable rivers and streams. FACETS, 8, 1-31. DOI: 10.1139/facets-2022-0158

Planas, D., Desrosiers, M., Groulx, S.-R., Paquet, S. and R., Carignan. (2000). Pelagic and benthic algal responses in eastern Canadian Boreal Shield lakes following harvesting and wildfires, Can. J. Fish. Aquat. Sci, 57 (Suppl. 2), 136-145.

Poff, L. N., and Allan, J. D. (1995). Functional organization of stream fish assemblages in relation to hydrological variability. Ecology. 76, 606-627.

Poff, N.L., Allan, J.D., Bain, M.B., Karr, J.R., Prestegaard, K.L., Richter, B.D., Sparks, R.E., Stromberg, J.C. (1997). The natural flow regime: a paradigm for river conservation and restoration. Bioscience, 47, 769-84.

Pomeroy, J., Fang, X., and Ellis, C. (2012). Sensitivity of snowmelt hydrology in Marmot Creek, Alberta, to forest cover disturbance. Hydrol. Process., 26, 1891-1904.

Pomeroy, J. W., De Boer, D., Martz, L. W., Pomeroy, J., De Boer, D., \& Martz, L. (2005). Hydrology and Water Resources of Saskatchewan Centre for Hydrology Report \#1.

Pont, D., Hugueny, B., Beier, U., Goffaux, D., Melcher, A., Noble, R., Rogers, C., Roset, N., \& Schmutz, S. (2006). Assessing river biotic condition at a continental scale: A European approach using functional metrics and fish assemblages. Journal of Applied Ecology, 43(1), 70-80. https://doi.org/10.1111/j.1365-2664.2005.01126.x

Pope, K.L., and Willis, D.W. (1996). Seasonal Influences on Freshwater Fisheries Sampling Data. Reviews in Fisheries Science, 4(1), 57-73. DOI:10.1080/10641269609388578

Prestie, N. (2014). Development of a framework for a fish assemblage-based index of biotic integrity in Saskatchewan. Honours Thesis, Unpublished. University of Saskatchewan, Saskatoon, Saskatchewan, Canada.

Pyron, M., Lauer, T. E., LeBlanc, D., Weitzel, D., \& Gammon, J. R. (2008). Temporal and spatial variation in an index of biological integrity for the middle Wabash River, Indiana. Hydrobiologia, 600(1), 205-214. https://doi.org/10.1007/s10750-007-9232-9

Quinn, J. M., \& Wright-Stow, A. E. (2008). Stream size influences stream temperature impacts and recovery rates after clearfell logging. Forest Ecology and Management, 256(12), 2101-2109. https://doi.org/10.1016/j.foreco.2008.07.041

Reece, P.F., Reynoldson, T.B., Richardson, J.S., Rosenberg, D.M. (2001). Implications of Seasonal Variation for Biomonitoring with Predictive Models in the Fraser River Catchment, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences, 58(7), 1411-1417. DOI:10.1139/cjfas-58-7-1411

Regional Aquatics Monitoring Program (RAMP). 1997. Potential Effects of Forestry on Aquatic Ecosystems. Available at: http://www.rampalberta.org/resources/forestry/potential+effects.aspx

Reid, A. J., Carlson, A. K., Creed, I. F., Eliason, E. J., Gell, P. A., Johnson, P. T.J., Kidd, K. A., MacCormack, T. J., Olden, J. D., Ormerod, S. J., Smol, J. P., Taylor, W. W., Tockner, K., Vermaire, J.C., Dudgeon, D., Cooke, S. J. (2019). Emerging threats and persistent conservation challenges for freshwater biodiversity. Biological Reviews, 94, 849-873.

Resh, V. H., D.M. Rosenberg, \& and Reynoldson T.B. (2000). Establishing reference conditions in the Fraser River catchment, British Columbia, Canada, using the BEAST (BEnthic Assessment of SedimenT) predictive model. Assessing the Biological Quality of Fresh Waters. RIVPACS and Other Techniques. Edited by J.F. Wright, D.M. Sutcliffe, and M.T. Furse. Freshwater Biological Association, Ambleside, UK. https://www.researchgate.net/publication/265871250_Establishing_Reference_Condition _for_Benthic_Invertebrate_Monitoring_in_the_Fraser_River_Catchment_British_Colum bia_Canada

Reynoldson, T. B., and Wright, J. F. (2000). The reference condition: problems and solutions, p. 293-304. In Wright JF, Sutcliffe DW, Furse MT (ed.), Assessing the biological quality of fresh water: RIVPACS and other techniques. Freshwater Biological Association, Cumbria, UK.

Reynoldson, T. B., Norris, R. H., Resh, V. H., Day, K. E., \& Rosenberg, D. M. (1997). The reference condition: A comparison of multimetric and multivariate approaches to assess water-quality impairment using benthic macroinvertebrates. Journal of the North American Benthological Society, 16(4), 833-852. https://doi.org/10.2307/1468175

Ricker, W. E. (1975). Computation and inter-predation of biological statistics of fish populations. The Quarterly Review of Biology, 51(2).
Ricciardi, A., \& Rasmussen, J. B. (1999). Extinction Rates of North American Freshwater Fauna. Conservation Biology, 13(5), 1220-1222.

Rooney, R. C., \& Bayley, S. E. (2012). Development and testing of an index of biotic integrity based on submersed and floating vegetation and its application to assess reclamation wetlands in Alberta's oil sands area, Canada. Environmental Monitoring and Assessment, 184(2), 749-761. https://doi.org/10.1007/s 10661-011-1999-5
Ross, C. D., \& McKenna, O. P. (2023). The Potential of Prairie Pothole Wetlands as an Agricultural Conservation Practice: A Synthesis of Empirical Data. Wetlands, 43(1), 111. https://doi.org/10.1007/S13157-022-01638-3/TABLES/1

Ruso, G. E., Morrissey, C. A., Hogan, N. S., Sheedy, C., Gallant, M. J., \& Jardine, T. D. (2019). Detecting Amphibians in Agricultural Landscapes Using Environmental DNA Reveals the Importance of Wetland Condition. Environmental Toxicology and Chemistry, 38(12), 2750-2763. https://doi.org/10.1002/ETC. 4598

Rytwinski, T., Taylor, J.J., Bennett, J.R. K.E., Smokorowski \& S.J., Cooke. (2017). What are the impacts of flow regime changes on fish productivity in temperate regions? A systematic map protocol. Environ Evid., 6, 13. https://doi.org/10.1186/s13750-017-0093-z

Sakadevan, K., \& Nguyen, M. L. (2017). Livestock Production and Its Impact on Nutrient Pollution and Greenhouse Gas Emissions. Advances in Agronomy, 14. http://dx.doi.org/10.1016/bs.agron.2016.10.002147
Sanders, R. E., Miller, R. J., Yoder, C. O., and Rankin, E. T. (1999). The use of external deformities, erosion, lesions, and tumors (DELT anomalies) in fish assemblages for characterizing aquatic resources: A case study of seven Ohio streams. Pages 225-248 in T. P. Simon, editor. Assessing the sustainability and biological integrity of water resources using fish communities. Lewis Press, Boca Raton, Florida.

Schaaf, C.J., Kelson, S.J., Nusslé, S.C., Carlson, S.M. (2017). Black spot infection in juvenile steelhead trout increases with stream temperature in northern California. Environ Biol Fish. 100, 733-744. https://doi.org/10.1007/s10641-017-0599-9

Schaeffer, D.J., Herricks, E.E., and Kerster, H.W. (1988). Ecosystem health: I. Measuring ecosystem health. Environmental Management. 12(4): 445-455.Schoolmaster, D. R., Grace, J. B., \& Schweiger, E. W. (2012). A general theory of multimetric indices and their properties. Methods in Ecology and Evolution, 3(4), 773-781.
https://doi.org/10.1111/j.2041-210X.2012.00200.x
Scott, W.B., and Crossman, E.J. (1973). Freshwater fishes of Canada. Bulletin of Fisheries Research Board of Canada 184. Government of Canada. 966pp.

Sharif, A. E., Yahyavi, B., Bayrami, A., Rahim Pouran, S., Atazadeh, E., Singh, R., \& Abdul Raman, A. A. (2021). Physicochemical and biological status of Aghlagan river, Iran: effects of seasonal changes and point source pollution. Environmental Science and Pollution Research. 28(12), 15339-15349. https://doi.org/10.1007/S11356-020-116609/TABLES/7

Shearer, J. S., \& Berry, C. R. (2002). Index of biotic integrity utility for the fishery of the James river of the dakotas. Journal of Freshwater Ecology, 17(4), 575-588. https://doi.org/10.1080/02705060.2002.9663935

Shentu, H., Yang, X., Baligar, V. C., Zhang, T., \& Stoffella, \&. (2015). Soil pollution in the world (EEC. In Journal of Environmental Indicators (Vol. 9). USEPA. http://www.eea.europa.eu/data-and-maps/indicators

Simon, T.P., and Lyons, J. (1995). Application of the index of biotic integrity to evaluate water resource integrity in freshwater ecosystems. In Davis, W. S. \& T. P. Simon (eds), Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making. Lewis Publishers, Boca Raton, FL. pp 245-262.

Smith, M. J., Kay, W. R., Edward, D. H. D., Papas, P. J., Richardson, K. S. J., Simpson, J. C., Pinder, A. M., Cale, D. J., Horwitz, P. H. J., Davis, J. A., Yung, F. H., Norris, R. H., \& Halse, S. A. (1999). AusRivAS: Using macroinvertebrates to assess ecological condition of rivers in Western Australia. Freshwater Biology, 41(2), 269-282. https://doi.org/10.1046/J.1365-2427.1999.00430.X

Smokorowski, K. E., Stoneman, M. G., Cairns, V. W., Minns, C. K., Randall, R. G., \& Valere, B. (1998). Trends in the nearshore fish community of Hamilton harbour, 1988 to 1997, as measured using an index of biotic integrity. Canadian Technical Report of Fisheries and Aquatic Sciences No. 2230. Great Lakes Laboratory for Fisheries and Aquatic Sciences, Department of Fisheries and Oceans, Burlington, Ontario.

Souza, G. B. G., \& Vianna, M. (2020). Fish-based indices for assessing ecological quality and biotic integrity in transitional waters: A systematic review. Ecological Indicators, 109, 105665. https://doi.org/10.1016/J.ECOLIND.2019.105665

Stanley, J., \& Preetha, G. (2016). Pesticide Toxicity to Fishes: Exposure, Toxicity and Risk Assessment Methodologies. Pesticide Toxicity to Non-Target Organisms, 411-497. https://doi.org/10.1007/978-94-017-7752-0_7

Stapanian, M. A., Micacchion, M., \& Adams, J. v. (2015). Wetland habitat disturbance best predicts metrics of an amphibian index of biotic integrity. Ecological Indicators, 56, 237-242. https://doi.org/10.1016/j.ecolind.2015.04.005

Steedman, R. J. (1988). Modification and Assessment of an Index of Biotic Integrity to Quantify Stream Quality in Southern Ontario. Can. I. Fish. Aquat. Sci., 45, 492-501.

Steedman, R.J., and R.S., Kushneriuk. (2000). Effects of experimental clearcut logging on thermal stratification, dissolved oxygen, and lake trout (Salvelinus namaycush) habitat volume in three small boreal forest lakes. Can. J. Fish. Aquat. Sci., 57 (suppl. 2), 82-91.

Steedman, R.J., Kushneriuk, R.S., and R.L., France. (2001). Littoral water temperature response to experimental shoreline logging around small boreal forest lakes. Can. J. Fish. Aquat. Sci., 58, 1638-1647.

Stevens, C., and Council, T. (2008). A fish-based index of biological integrity for assessing river condition in central Alberta. Technical Report, T-2008-001, produced by the Alberta Conservation Association, Sherwood Park and Lethbridge, Alberta, Canada. 29 pp. + App.

Stevens, C. E., Council, T., \& Sullivan, M. G. (2010). Influences of Human Stressors on FishBased Metrics for Assessing River Condition in Central Alberta. Water Qual. Res. J. Can. 45(1), 35-46.

Stevens, C., Scrimgeour, G., Tonn, W., Paszkowski, C., Sullivan, M., and Millar, S. (2006). Development and testing of a fish-based index of biological integrity to quantify the health of grassland streams in Alberta. Technical report (T-2006-001) produced by Alberta Conservation Association, Edmonton, Alberta, Canada. 50 pp + App.

Stevens, L. M., Forrest, B. M., Dudley, B. D., Plew, D. R., Zeldis, J. R., Shankar, U., Haddadchi, A., \& Roberts, K. L. (2022). Use of a multi-metric macroalgal index to document severe eutrophication in a New Zealand estuary. New Zealand Journal of Marine and Freshwater Research, 56(3), 410-429. Https://Doi.Org/10.1080/00288330.2022.2093226

Stewart, K.W. and Watkinson, D.A. 2004. The Freshwater Fishes of Manitoba. University of Manitoba Press, Winnipeg, Manitoba, Canada.

Stoddard, J.L., D.V. Peck, A.R. Olsen, D.P. Larsen, J. Van Sickle, C.P. Hawkins, R.M. Hughes, T.R. Whittier, G. Lomnicky, A.T. Herlihy, P.R. Kaufmann, S.A. Peterson, P.L. Ringold, S.G. Paulsen, and R. Blair. (2005). Environmental Monitoring and Assessment Program (EMAP) Western Streams and Rivers Statistical Summary. Report No. EPA 620/R05/006United States Environmental Protection Agency, Office of Research and Development, Washington, DC.

Stoddard, J. L., Larsen, D. P., Hawkins, C. P., Johnson, R. K., \& Norris, R. H. (2006). Setting Expectations for the Ecological Condition of Streams: The Concept of Reference

Condition. In Source: Ecological Applications, 16(4). http://www.jstor.orgURL:http://www.jstor.org/stable/40062000

Stoddard, J. L., Herlihy, A. T., Peck, D. v., Hughes, R. M., Whittier, T. R., \& Tarquinio, E. (2008). A process for creating multimetric indices for large-scale aquatic surveys. Journal of the North American Benthological Society, 27(4), 878-891. https://doi.org/10.1899/08-053.1

Strahler, A.N. (1952). Dynamic basis of geomorphology. Geological Society of America Bulletin, 63, 923-938.

Strayer, D.L., and Dudgeon, D. (2010). Freshwater biodiversity conservation: recent progress and future challenges. Journal of the North American Benthological Society. 29(1): 344358.

Studinski, J. M., \& Hartman, K. J. (2015). The effects of riparian logging on terrestrial invertebrate inputs into forested headwater streams. Hydrobiologia, 743(1), 189-198. https://doi.org/10.1007/S10750-014-2036-9

Sutela, T., Vehanen, T., Huusko, A., Maki-Petays, A. (2017). Seasonal shift in boreal riverine fish assemblages and associated bias in bioassessment. Hydrobiologia, 787, 193-203. https://doi-org.cyber.usask.ca/10.1007/s10750-016-2959-4

Sweilum, M.A., (2006). Effect of sublethal toxicity of some pesticides on growth parameters, haematological properties and total production of Nile tilapia (Oreochromis niloticus L.) and water quality of ponds. Aquacult. Res., 37(11), 1079-1089.

Tomscha, S.A., S.E. Gergel, and M.J. Tomlinson. (2017). The spatial organization of ecosystem services in river-floodplains. Ecosphere, 8(3), e01728. https://doi.org/10.1002/ecs2.1728.

Thorpe, J., and Godwin, B. (1999). Threats to Biodiversity in Saskatchewan. Plant Ecology Section, Environment Branch, SRC Publication No. 11158-1C99. Saskatchewan Research Council SRC, Saskatoon, SK.

United States Environmental Protection Agency. (2007). National Water Quality Inventory: Report to Congress. http://www.epa.gov/305b

United States Environmental Protection Agency (USEPA). (2002). Summary of Biological Assessment Programs and Biocriteria Development for States, Tribes, Territories, and Interstate Commissions: Streams and Wadeable Rivers. EPA-822-R-02-048. U.S.

Environmental Protection Agency, Office of Environmental Information and Office of Water, Washington, D.C.
United States Environmental Protection Agency (USEPA). 2011. A primer on using biological assessments to support water quality management. Environmental Protection Agency, Office of Science and Technology and Office of Water, Washington, D.C.
United States Environmental Protection Agency. (2011). A Primer on Using Biological Assessments to Support Water Quality Management.
U.S. Geological Survey. (2010). Nutrients in the Nation's Streams and Groundwater, 1992-2004. Reston, Virginia: USGS Circular 1350.
Vander Laan, J. J., and Hawking, C. P. (2014). Enhancing the performance and interpretation of freshwater biological indices: An application in arid zone streams. Ecological Indicators, 36(2014), 470-482.

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

Vári, Á., Podschun, S.A., Erős, T., Hein, T., Pataki, B., Loja, L., Adamescu, C.M., Gerhardt, A., Gruber, T., Dedic, A., Ciric, M., Gavrilovic, B., Baldi, A. (2022). Freshwater systems and ecosystem services: Challenges and chances for cross-fertilization of disciplines. Ambio., 51, 135-151. https://doi-org.cyber.usask.ca/10.1007/s13280-021-01556-4
Vega, G., \& Wiens, J.J. (2012). Why are there so few fish in the sea? Proceedings of the Royal Society of London B: Biological Sciences, 279, rspb20120075.

Vile, J. S., \& Henning, B. F. (2018). Original Articles Development of indices of biotic integrity for high-gradient wadeable rivers and headwater streams in New Jersey. https://doi.org/10.1016/j.ecolind.2018.03.027

Vorosmarty, C. J., McIntyre, P. B., Gessner, M. O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S. E., Sullivan, C. A., Liermann, C. R. \& Davies, P. M. (2010). Global threats to human water security and river biodiversity. Nature, 467, 555-561.
Waite, I. R., Herlihy, A. T., Larsen, D. P., Klemm, D. J. (2000). Comparing strengths of geographic and nongeographic classifications of stream benthic macroinvertebrates in the Mid-Atlantic Highlands, USA. Journal of the North American Benthological Society, 19, 429-441.

Whittier, T. R., Hughes, R. M., Stoddard, J. L., Lomnicky, G. A., Peck, D. V., and Herlithy, A. L. (2007). A structured approach for developing indices of biotic integrity: Three examples from streams and rivers in the western USA. Transactions of the American Fisheries Society, 136, 718-735. DOI: 10.1577/T06-128.1

World Wildlife Fund Canada (WWF). (2018). Living Planet Report - 2018: Aiming Higher. Grooten, M. and Almond, R.E.A. (Eds). WWF, Gland, Switzerland.

Wright, J. F. (1995). Development and use of a system for predicting the macroinvertebrate fauna in flowing waters. Australian Journal of Ecology, 20(1), 181-197. https://doi.org/10.1111/J.1442-9993.1995.TB00531.X

Wright, J. F., M.T. Furse, \& and P.D. Armitage. (1983). RIVPACS: A technique for evaluating the biological quality of rivers in the U.K. European Water Pollution Control., 3, 15-25.

Zhu, D., \& Chang, J. (2008). Annual variations of biotic integrity in the upper Yangtze River using an adapted index of biotic integrity (IBI). Ecological Indicators, 8(5), 564-572. https://doi.org/10.1016/j.ecolind.2007.07.004

Zhu, H., Hu, X. D., Wu, P. P., Chen, W. M., Wu, S. S., Li, Z. Q., Zhu, L., Xi, Y. L., \& Huang, R. (2021). Development and testing of the phytoplankton biological integrity index (P-IBI) in dry and wet seasons for Lake Gehu. Ecological Indicators, 129, 107882.
https://doi.org/10.1016/J.ECOLIND.2021.107882

Zhu, R., Qing, L., Wanf, W., Chu, L., and Yan Y. (2016). Effects of local, river-network and catchment factors on fish assemblages in the headwater streams of the Xin'an basin, China. Journal of Freshwater Ecology, 32(1). https://doi.org/10.1080/02705060.2016.1278408

## Appendix

Appendix $\mathbf{A} \mid$ List of fish species known from the Churchill River basin in Saskatchewan.

| Common Name | Scientific Name | Family | Species Status |
| :---: | :---: | :---: | :---: |
| Lake Sturgeon | Acipenser fulvescens | Acipenseridae | S2 |
| Longnose Sucker | Catostomus catostomus | Catostomidae | S5 |
| Shorthead Redhorse | Moxostoma macrolepidotum | Catostomidae | S4, S5 |
| White Sucker | Catostomus commersoni | Catostomidae | S5 |
| Slimy Sculpin | Cottus cognatus | Cottidae | S5 |
| Deepwater Sculpin | Myoxocephalus thompsoni | Cottidae | S5 |
| Spoonhead Sculpin | Cottus ricei | Cottidae | S5 |
| Sand Shiner | Notropis stramineus | Cyprinidae | S3, S4 |
| Asexual Hybrid Dace | Chrosomus eos-neogaeus | Cyprinidae | S5 |
| Blacknose Shiner | Notropis heterolepis | Cyprinidae | S4, S5 |
| Creek Chub | Semotilus atromaculatus | Cyprinidae | S3, S4 |
| Emerald Shiner | Notropis atherinoides | Cyprinidae | S5 |
| Fathead Minnow | Pimephales promelas | Cyprinidae | S5 |
| Finescale Dace | Chrosomus neogaeus | Cyprinidae | S5 |
| Lake Chub | Couesius plumbeus | Cyprinidae | S5 |
| Longnose Dace | Rhinichthys cataractae | Cyprinidae | S5 |
| Northern Redbelly Dace | Chrosomus eos | Cyprinidae | S3, S4 |
| Northern Pearl Dace | Margariscus margarita | Cyprinidae | S5 |
| Spottail Shiner | Notropis hudsonius | Cyprinidae | S5 |
| Northern Pike | Esox lucius | Esocidae | S5, PS |
| Burbot | Lota lota | Gadidae | S5 |
| Brook Stickleback Ninespine | Culaea inconstans | Gasterosteidae | S5 |
| Stickleback | Pungitius pungitius | Gasterosteidae | S5 |
| Iowa Darter | Etheostoma exile | Percidae | S5 |
| Johnny Darter | Etheostoma nigrum | Percidae | S5 |
| Common Logperch | Percina caprodes | Percidae | S5 |
| Sauger | Sander canadense | Percidae | S5 |
| Walleye | Sander vitreus | Percidae | S5, PS |
| Yellow Perch | Perca flavescens | Percidae | S5 |
| Trout-perch | Percopsis omiscomaycus | Percopsidae | S5 |
| Lake Trout | Salvelinus namaycush | Salmonidae | S5 |
| Round Whitefish | Prosopium cylindraceum | Salmonidae | S5 |
| Arctic Grayling | Thymallus arcticus | Salmonidae | S5 |
| Cisco (lake herring) | Coregonus artedi | Salmonidae | S5 |


| Lake Whitefish | Coregonus clupeaformis | Salmonidae | S5 |
| :---: | :---: | :--- | :---: |
| Shortjaw Cisco | Coregonus zenithicus | Salmonidae | S1 |
| Brook Trout | Salvelinus fontinalis | Salmonidae | I, NA, PS |
| Rainbow Trout | Oncorhynchus mykiss | Salmonidae | I, NA, PS |
| Splake | Salvelinus fontinalis x Salvelinus |  | I, NA, PS; |
|  | namaycush | Salmonidae | H |
| Tiger Trout | Salmo trutta x Salvelinus fontinalis | Salmonidae | I, NA, PS; |
| H |  |  |  |

Sources: Cantin \& Johns (2012); Joynt \& Sullivan (2003); Morris \& Somers (2015); Prestie (unpublished, 2014); Scott \& Crossman (1973), and Saskatchewan Conservation Data Center (updated 2023) (found at http://biodiversity.sk.ca/TaxaList/sk-taxa-vertebrate-all.pdf). *Species status represents provincial species rankings (NatureServe conservation status within SK): S1: Critically Imperilled; S2: Imperilled; S3: Vulnerable; S4: Apparently Secure; S5:
Secure; NA: Not Assessed; PS: Provincially Stocked; I: Introduced; H: Hybrid

Appendix B|Additional Site Characteristics for the 18 streams and rivers sampled in the Beaver River Watershed


| Elevation $(\mathbf{m})$ | $\underset{\left(\mathbf{k m}^{2}\right)}{\text { Drainage Area }}$ | Stream Orier | Reach Area $\left(m^{2}\right)$ | $\underset{\left(\mathrm{m}^{3} / \mathbf{s}\right)^{*}}{\text { Stream Discharge }}$ | Discharge ( $\mathrm{m}^{3} / \mathrm{s}$ )* <br> Estimated High Flow |
| :---: | :---: | :---: | :---: | :---: | :---: |
| 658.5 | 80.7 | 1 | 500.8 | 1.06 | ${ }^{1.53}$ |
| 688.5 | 80.7 | 1 | 558.0 | ${ }_{0} .88$ | 0.99 |
| 668.5 | 80.7 | 1 | 591.8 | ${ }^{1.06}$ | 1.12 |
| 476 | 1392 | 4 | ${ }_{34,0}$ | 0.03 | 0.05 |
| 476 | 1392 | 4 | 375.8 | 0.19 | ${ }_{0}^{035}$ |
| 476 | 139.2 | 4 | 3897 | 0.25 | ${ }_{0} .46$ |
| 536 | 176.4 | 3 | 16096 | ${ }_{0} .30$ | 0.61 |
| 536 | 176.4 | 3 | 26056 | 1.15 | ${ }^{1.31}$ |
| ${ }_{536}$ | ${ }^{176.4}$ | 3 | 27098 | ${ }^{1.06}$ | ${ }^{1.19}$ |
| 483 | 3447 | 4 | 42560 | ${ }_{0} .84$ | 1.22 |
| 483 | 344.7 | 4 | 5499.0 | 3.26 | 3.46 |
| 483 | 344.7 | 4 | 57620 | ${ }_{3.84}$ | ${ }^{4.25}$ |
| 546 | ${ }^{70.8}$ | 2 | 11427 | ${ }_{0} .39$ | 0.56 |
| 546 | 20.8 | 2 | ${ }_{1364}{ }^{1}$ | 0.44 | 0.51 |
| 546 | 20.8 | 2 | ${ }^{13032}$ | 0.75 | ${ }^{0.90}$ |
| 497 | 197. | 3 | 4097 | ${ }^{1.13}$ | 1.86 |
| 587 | 264.2 | 3 | 94.5 | ${ }^{124}$ | 1.55 |
| 477 | ${ }^{317.0}$ | 4 | 561.7 | ${ }_{0} .87$ | ${ }^{1.05}$ |
| ${ }_{513}$ | 261.5 | 3 | 968.3 | 2.62 | 2.68 |
| 599 | ${ }^{61.8}$ | 2 | 515.3 | 1.12 | ${ }^{1.28}$ |
| 468 | 111.7 | 3 | 9953 | 1.58 | 1.66 |
| 525 | 311.8 | 3 | 430.0 | 0.89 | 1.02 |
| 563 | 52.5 | 2 | 371.5 | ${ }^{0.13}$ | ${ }^{0.19}$ |
| 512 | 187.4 | 4 | 719.0 | 0.25 | ${ }_{0} .32$ |
| 511 | 115.9 | 2 | 345.5 | ${ }^{020}$ | ${ }^{0.54}$ |
| ${ }_{53}$ | 1028 | 3 | ${ }_{644}^{64}$ | ${ }^{0.26}$ | ${ }^{0.31}$ |
| 572 | 348.9 | 4 | ${ }_{1136.5}$ | 0.55 | 0.56 |
| 542 | 430.6 | 4 | 14550.0 | 1.32 | 1.48 |


| UTM Zone | Easting | Northing | Ecoregion ${ }^{+}$ | Dominant Surrounding Land Use ${ }^{+}$ | Local Watershed Erosion |
| :---: | :---: | :---: | :---: | :---: | :---: |
| 12 U | 673017 | 5960036 | MBU | Forest | None |
| 12 U | 673017 | 5960036 | MBU | Forest | Low |
| 12 U | 673017 | 5960036 | MBU | Forest | None |
| 12 U | 668201 | 5995148 | MBU, BT | Cultivated | Heavy |
| 12 U | 668201 | 5995148 | MBU, BT | Cultivated | Heavy |
| 12 U | 668201 | 5995148 | MBU, BT | Cultivated | Heavy |
| 12 U | 608644 | 6037966 | MBU | Forest | Low |
| 12 U | 608644 | 6037966 | MBU | Forest | Low |
| 12 U | 608644 | 6037966 | MBU | Forest | Low |
| 12 U | 659840 | 6051158 | MBU | Forest | Low |
| 12 U | 659840 | 6051158 | MBU | Forest | Low |
| 12 U | 659840 | 6051158 | MBU | Forest | None |
| 12 U | 651198 | 6052904 | MBU | Forest | Low |
| 12 U | 651198 | 6052904 | MBU | Forest | Low |
| 12 U | 651198 | 6052904 | MBU | Forest | Low |
| 12 U | 624573 | 6026986 | BT | Cultivated | Moderate |
| 13 U | 331828 | 5915824 | BT | Cultivated, Urban | Moderate |
| 12 U | 661760 | 5998449 | MBU, BT | Cultivated | Moderate |
| 13 U | 350486 | 5970623 | MBU, BT | Forest | None |
| 12 U | 640110 | 6049245 | MBU | Forest | Moderate |
| 12 U | 675350 | 6021786 | BT, MBU | Forest | Low |
| 13 U | 379777 | 5972709 | MBU | Forest | Low |
| 12 U | 649518 | 6052403 | MBU | Forest | Heavy |
| 12 U | 618360 | 6034672 | MBU | Forest | Moderate |
| 12 U | 638445 | 6002535 | BT, MBU | Cultivated | Heavy |
| 13 U | 331293 | 5977289 | BT, MBU | Forest, Logging | Low |
| 13 U | 315604 | 5952313 | MBU, BT | Forest, Cultivated | Low |
| 13U | 308134 | 5981072 | MBU, BT | Forest | Moderate |

+Ecoregion and surrounding land use are listed in descending order of area occupied for the upstream contributing watershed.
*Determined based on the conditions at time of sampling.

BT: Boreal Transition

Appendix C| Fish collection sheet adapted from Barbour et al. 1999.

## Fish Collection Field Sheet

Test Site Reference Site
Date: (Y/M/D) $\qquad$ Location: DS (start): E $\qquad$ N

Time: Start: $\qquad$ End: $\qquad$ US (end): E $\qquad$ N

Field Crew: $\qquad$ Elevation $\qquad$ Stream Name/Code:

Site Description:

## Electrofishing Information

Sampling Duration: Start Time $\qquad$ End Time $\qquad$ Duration $\qquad$
Water Conductivity ( $\mu \mathrm{s} / \mathrm{cm}$ ) $\qquad$ Water Temperature $\left({ }^{\circ} \mathrm{C}\right)$ $\qquad$
Electrofishing Output Parameters
Voltage (volts) $\qquad$ Duty Cycle (\%) $\qquad$ Frequency (Hertz) Other

| Species | Fish ID Code | Total Length (cm) | Weight (g) | *Anomalies |
| :---: | :---: | :---: | :---: | :---: |
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| :--- | :--- | :--- | :--- | :--- |
| 20 |  |  |  |  |
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| Species | Fish ID Code | Total Length (cm) | Weight (g) | *Anomalies |
| :---: | :---: | :---: | :---: | :---: |
| 30 |  |  |  |  |
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| 60 |  |  |  |  |
| :--- | :--- | :--- | :--- | :--- |
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*Anomaly Codes: $\mathrm{D}=$ deformities, $\mathrm{E}=$ eroded fins, $\mathrm{L}=$ lesions, $\mathrm{T}=$ tumors, $\mathrm{P}=$ parasites, $\mathrm{F}=$ fungus, $\mathrm{S}=$ emaciated, $Z=$ other

Electrofishing On Time (sec) (for CPUE in fish/s)
Mortality/ Injury (\# of indivs.)
Fish collected for: Hg Yes No Species __ \# of specimens
Voucher Specimens Yes No Species __ \# of specimens
Pictures Taken Yes No

Fish Collection Field Sheet Additional Notes
Stream Name/Site Code: $\qquad$
Date: (YYYY/MM/DD) $\qquad$
$\qquad$
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Appendix D| Standardized stream site assessment sheet adapted from Barbour et al. 1999.
WADEABLE STREAM SITE ASSESSMENT
Test Site Reference Site
Stream Name/Code:

Date: (Y/M/D) $\qquad$ Location: DS (start): E $\qquad$ N

Time: Start: $\qquad$ End: $\qquad$ US (end): E $\qquad$
N $\qquad$
Field Crew: $\qquad$ Elevation (masl): $\qquad$ Site Distance from Road: $\qquad$
Site Description:
$\qquad$
$\qquad$

PHYSICAL CHARACTERIZATION


## Watershed Features


\(\left.\begin{array}{|l|}\hline Instream Canopy Coverage (pg 7) <br>
- Determine the category for terrestrial canopy cover along the entire reach <br>

\square 0 \% \quad \square 1-25 \% \quad \square 26-50 \% \quad \square 51-75 \% \quad \square 76-100 \%\end{array}\right]\)| Large Woody Debris (pg 8 \& 9) |
| :---: |
| - Determine the category for coarse woody material along the entire reach |
| $\square$ None $\quad \square$ Low $\quad \square$ Moderate $\quad \square$ High |

Aquatic Vegetation (pg 9)

- Determine the portion of the reach with aquatic vegetation and indicate the dominant type and the dominant species present
$\square 0 \% \quad \square$
1-25\%
26-50\%)
51-75\%
76-100\%

Dominant type: $\square$ Emergent
$\square$ Submergent
Rooted floating
$\square$ Free floating
$\square$ Floating Algae
$\square$ Attached Algae
Dominant Species Present $\qquad$
Riparian Vegetation (pg 10 \& 11)

- Indicate the dominant community type, the dominant species present, and the vegetation stage along the reach


## Riparian Vegetative Community

$\square$ None
Grass/Sedge/Bog
Shrub
$\square$ Deciduous
$\square$ Coniferous
$\square$
Mixedwood
Cultivated (or cropland)
$\square$ Pasture

## Dominant Species Present

## Riparian Vegetation Stage

- Indicate the dominant type along the reach
INIT
$\square$ SHRYFMF
NA


## Sediment/Substrate Characteristics (pg 12)

- Visually estimate \% composition along entire reach (total must equal 100)
- Indicate presence of any odors, oils, or deposits along reach


## Substrate Composition

| Substrate | \% <br> Composition |
| :--- | :---: |
| Bedrock |  |
| Boulder (> 25.6 <br> cm) |  |
| Cobble (6.4-25.6 <br> cm) |  |
| Gravel (0.2-6.4 <br> cm) |  |

## Odors

$\square$ Normal/None
$\square$ Anaerobic
$\square$ Chemical
$\square$ Sewage
Petroleum
Other $\qquad$

## Oils

$\square$ None
$\square$ Present

Deposits
$\square$ None
$\square$ Present

| Sand/silt/clay $(<$ <br> $0.2 \mathrm{~cm})$ |  |
| :--- | :--- |
| $*$ Organic |  |

*Organic = Detritus (sticks, wood, coarse plant materials (CPOM)), Muck-Mud (black, very fine organic (FPOM)) and Marl (grey, shell fragments)

| Photographs | Transect 1 | Transect 2 | Transect 3 |
| :--- | :--- | :--- | :--- |
| Field sheet |  |  |  |
| Across Site |  |  |  |
| Upstream |  |  |  |
| Downstream |  |  |  |
| Substrate (exposed/ aquatic) |  |  |  |
| Other |  |  |  |


| Channel Measurements |  |  | Transect 1 | Transect 2 |
| :--- | :--- | :--- | :--- | :--- |
|  | Transect 3 | Avg |  |  |
| Habitat Type (at each transect) |  |  |  |  |
| Bankfull Width (m) |  |  |  |  |
| Bankfull-wetted Depth (m) |  |  |  |  |
| Wetted Stream Width (m) |  |  |  |  |
| Reach Area (avg WW x RL) <br> $\left(\mathrm{m}^{2}\right)$ |  |  |  |  |

## Depth and Velocity

|  | Transect 1 (US) |  |  |  |  |  |  | Transect 2 |  |  |  |  |  |  | Transect 3 (DS) |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | 1 | 2 | 3 | 4 | 5 | 6 | A | 1 | 2 | 3 | 4 | 5 | 6 | A | 1 | 2 | 3 | 4 | 5 | 6 | A |
| Distance from Shore (m) |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Stream <br> Depth (cm) |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Velocity Head Rod (ruler) |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Flowing water |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |


| $\begin{array}{\|l} \begin{array}{l} \text { depth } \\ \left(\mathrm{D}_{1}\right) \\ (\mathrm{cm}) \end{array} \\ \hline \end{array}$ |  |  |  |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| $\begin{array}{\|l} \hline \text { D of } \\ \text { stagnation } \\ \left(\mathrm{D}_{2}\right)(\mathrm{cm}) \end{array}$ |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Change in depth ( $\Delta \mathrm{D}=$ $\mathrm{D}_{2}$-D $\mathrm{D}_{1}$ ) (cm) |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Velocity <br> Meter |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Revolutions |  |  |  |  |  |  |  |  |  |  |  |  |  |
| $\begin{array}{\|l\|} \hline \text { Time } \\ \text { (min of } 40 \\ \text { sec) } \end{array}$ |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Orange/ Golf Ball (at thawleg) |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Distance <br> (m) |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Time (sec) |  |  |  |  |  |  |  |  |  |  |  |  |  |

## Slope (Hand Level and Measuring Tape Method)

| Measurements | Upstream (US) | Downstream (DS) | Calculation |
| :--- | :--- | :--- | :--- |
| Height of Rod <br> $(\mathrm{m} / \mathrm{cm})$ |  |  |  |
| Distance (m) |  |  | $\mathbf{U S}_{\text {dis }}+\mathbf{D S}_{\text {dis }}=$ |
| Change in Height <br> (Dht) |  |  | $\mathbf{D S S}_{\mathbf{h t}}-$ USS $_{\mathbf{h t}}=$ |
| Slope (Dht/total dis) |  |  |  |

## WATER QUALITY (pg 12)

|  |  |  |
| :--- | :---: | :---: |
| Air temp: $\left({ }^{\circ} \mathrm{C}\right)$ |  |  |
| Water temp: $\left({ }^{\circ} \mathrm{C}\right)$ | $\mathrm{Chl} \mathrm{a:}(\mu \mathrm{~g} / \mathrm{L})$ |  |
| Turbidity: (FNU) | $\mathrm{DO}:(\mathrm{mg} / \mathrm{L})$ | $\mathrm{pH}:$ |
| Conductivity: $(\mu \mathrm{s} / \mathrm{cm})$ |  |  |



## Collect:

$\square$ eDNA Volume filtered (mL) $\qquad$
BMI \# of samples $\qquad$

Notes:
$\qquad$
$\qquad$
$\qquad$
$\qquad$
$\qquad$

$\qquad$

Appendix E|ArcGIS Data files accessed and used to develop the upstream contributing watershed area, watershed land use, and additional physiographic information for the 18 study sites in the Beaver River watershed.

| Data File | Description | Spatial <br> Representation | Spatial <br> Resolution | Provided By |
| :---: | :---: | :---: | :---: | :---: |

Annual Crop
Logging/Harvest

Crop and land cover inventory for central and southern Saskatchewan (2013)

## Logging and harvest data for the Beaver River

 Watershed areaVector Unknown

Human
Population Census Data
Landfill Data

National Road
Grid $\quad 30 \mathrm{~m}$

Census data for SK (2016)
Vector

Vector

Vector
Road data for SK
1: 5000

Mine Data

|  |  |  |  | Government of Saskatchewan |  |
| :---: | :---: | :---: | :---: | :---: | :---: |
| Oil and Gas Well Data | Oil and gas well data for SK | Vector | Unknown | Saskatchewan Ministry of Energy and Resources, Government of Saskatchewan | Geographic Information for the Province of Saskatchewan: <br> https://www.saskatchewan.ca/government/notarized-documents-legislation-maps/maps |
| Livestock Data | Cattle (and all other livestock) inventory for southern SK census blocks (2016) | Vector | Unknown | Statistics Canada, <br> Census of Agriculture | Statistics Canada Census of Agriculture: https://www150.statcan.gc.ca/n1/daily-quotidien/170510/dq170510a-cansim-eng.htm |

Appendix $\mathbf{F} \mid$ Steps taken to complete watershed delineation using Arcmap 10.8.1.

## Watershed Delineation Steps:

a) Converted UTM coordinates into Latitude and Longitude, then uploaded excel file via ArcCatalog into ArcMap table of contents.
b) Mosaic raster data (all individual DEMs) into one dataset using Mosaic to New Raster Tool.
c) Project raster to consistent geographic coordinate system (NAD 1983 13N) using project raster tool. Chose NAD 1983 because it is the major coordinate system used in Canada. Used UTM zone 13 N following the Saskatchewan standard.
d) Clipped all layers being used to fit into the Beaver River watershed using Clip Raster Tool (raster data) or the Geoprocessing Tool (vector data)
e) Fill sinks in DEMs via terrain processing tab, DEM manipulation and fill sinks option.
f) Create initial flow direction for data using terrain processing tab.
g) Define flow accumulation (stream channel initialization) via terrain processing to determine water accumulation and flow across the watershed. Then compared output to the CanVec Watercourse and Waterbodies topographic map 1:50,000 scale layer to ensure the flow accumulation from the DEM accurately represents streams and rivers within the Beaver River watershed.
h) Define the stream network using terrain processing and the stream definition option. It is important to choose an appropriate threshold value to be used to create a stream. A typical cell value to use with a national elevation dataset is 5,000 or using the default percentage of $1 \%$ of the maximum flow accumulation established in step g above (Maidment and Morehouse 2002). Using the $1 \%$ of the max flow accumulation established in step $g$ was not a small enough scale for the stream sizes of interest for this study as only the larger, higher order streams and rivers were defined. Therefore, a smaller threshold value had to be chosen.
i) Define stream segmentation using terrain processing and stream definition tool. The result is only used as a basis for deciding the lowest point in each catchment and assigning unique hydro ID's/ identifiers for each stream segment. The highest value on the stream link grid should be consistent with the number of catchments created (14,943 in the Beaver River Watershed).
j) Catchment grid delineation via terrain processing. The output produces watersheds/catchments using DEM in grid format (raster).
k) Catchment polygon processing via terrain processing to convert rater catchments from above step into vector polygon catchments. This creates multiple small catchments within the Beaver River watershed. If catchment area is too small, can merge multiple catchments to get a specific catchment size (e.g., $10 \mathrm{~km}^{2}$ )

1) Drainage line processing via terrain processing used to convert the input stream link grid into a drainage line feature class. Each line in the feature class carries the identifier of the catchment in
which it resides. Drainage lines/ streams were visually assessed with the 1:50,000 scale Canvec topographic map layer to verify accuracy. There were clearly discrepancies between the CanVec 1:50,000 streams and the ArcMap drainage lines produced (approximately $75 \%$ consistency). ArcMap seemed to be a lot more sensitive, overall creating more tributaries and higher stream orders. Therefore, inconsistent watershed boundaries needed to be delineated by hand in a later step (alternatively, can burn streams into the DEM using the Canvec topographic map layer.
m) Adjoint catchment processing function using terrain processing. The Adjoint Catchment tool is used for calculating the distance upstream. The three functions, Catchment Polygon Processing, Drainage Line Processing and Adjoint Catchment Processing convert the raster data developed so far to vector format.
n) Drainage point processing tool is then used to generate the drainage point feature class associated to the input catchment feature class and flow accumulation grid. The drainage point feature represents the location of the cell with the maximum flow accumulation value within each catchment.
o) Point delineation (using the site coordinates as reference/ pour points) was used to create individual polygon sub watersheds/catchments for each site of interest. This last step allowed the creation of the upstream contributing watershed area/ catchments for each of the 18 study sites. Due to differences in drainage lines created via step 1 above and the streams in the Canvec topographic map layer, approximately $50 \%$ of the watersheds were not perfect and one watershed was missing altogether. Therefore, using the edit tool and reshape polygon function, various watersheds had to be manually recreated and digitized to reflect the topology and hydrology of the area more accurately.

Appendix $\mathbf{G} \mid$ Mean values $( \pm \mathrm{SD})$ and range for the land use variables $(\mathrm{n}=11)$ created during the development of reference conditions and site watershed stress classes. Whether or not the variable was retained for watershed stress class selection and the Index of Biotic Integrity development is also shown.

| Land Use Stressor Category | Stressor | Description | Mean ( $\pm$ SD) | Range | Retained? |
| :---: | :---: | :---: | :---: | :---: | :---: |
| Land Cover | \% Agriculture | Percent area that is cultivated, pasture and forage landscape in watershed | $11.1( \pm 19.8)$ | 0-67.3 | NO |
|  | \% Urban Cover | Percent area that is urban landscape or developed in watershed | 0.6 ( $\pm 1.0)$ | 0-4.1 | NO |
|  | \% Natural Landscape | Percent area that is forest, shrubland, grassland, and water (including wetlands) in watershed | 86.2 ( $\pm 21.2)$ | 25.5-99.6 | NO |
|  | \% Human-Related Disturbance | Percent area that is agricultural or urban cover in watershed | $11.7( \pm 20.6)$ | 0-71.4 | YES |
| Natural Resource Extraction | No. of Oil and Gas Wells | Number of vertical and nonvertical wells in each watershed | $2.9( \pm 4.4)$ | 0-13 | YES |
| Municipal Waste | No. of Landfills, Lagoons and Wastewater | Number of landfills, lagoons, and wastewater treatment plants in each watershed | $0.2( \pm 0.5)$ | 0-2 | NO |
| Forestry Operations | \% Area Harvested | Percent area that is harvested (standardized by watershed area) in each watershed | $10.0( \pm 7.7)$ | 0-21.8 | YES |
| Road Corridors | No. of Upstream Road Crossings | Number of stream and river road crossings per watershed | $5.7( \pm 6.9)$ | 0-22 | NO |
|  | Road Density (m/km2) | Density of roads (standardized by watershed area) in each watershed | $217.9( \pm 225)$ | 0-890.6 | NO |


| Population | Human Population Density <br> (people/km2) | Population density (standardized <br> by watershed area) for each <br> watershed | $0.5( \pm 1.0)$ | $0-3.8$ | NO |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: |
| Livestock | No. of Cattle | Number of Cattle in each <br> watershed | $1735.5( \pm 1018.2)$ | $458-3762$ | NO |
|  |  | wat |  |  |  |

Appendix H| Mean values ( $\pm$ SD) and range for the water quality and habitat variables $(\mathrm{n}=48)$ assessed during the development of reference conditions and site watershed stress classes. Variables that were significantly different between the low and high stress classes are given in bold. Whether or not the variable was retained for watershed stress class selection and the Index of Biotic Integrity development is also shown.

| Stressor Category | Stressor | Mean ( $\pm$ SD) | Range | Retained? |
| :---: | :---: | :---: | :---: | :---: |
| Fish Habitat | Habitat Assessment Score (\%) | 81.0 ( $\pm 20.2$ ) | 36.0-98.3 | YES |
|  | *Local Watershed Erosion (\%) | 64.3 ( $\pm 24.0)$ | 25.0-100.0 | NO |
| Nutrients | Nitrate ( $\mathrm{mg} / \mathrm{L} \mathrm{N}$ ) | 0.06 ( $\pm 0.04)$ | 0.01-0.15 | YES |
|  | Total nitrogen (mg/L) | $1.2( \pm 0.6)$ | 0.5-3.2 | YES |
|  | Phosphate (mg/L P) | $0.07( \pm 0.05)$ | 0.0-0.19 | YES |
|  | Total Phosphorus (mg/L) | $0.1( \pm 0.1)$ | 0-0.5 | YES |
|  | AVG Benthic Chl a ( $\mathrm{mg} / \mathrm{m}^{2}$ ) | 3.1 ( $\pm 2.9)$ | 0.3-11.2 | YES |
|  | AVG Suspended Chl a (ug/L) | $4.3( \pm 7.0)$ | 0.2-28.4 | YES |
|  | Ammonia (mg/L N) | $0.1( \pm 0.2)$ | 0.0-0.7 | NO |
| Physical | SPC (us/cm at $25^{\circ} \mathrm{C}$ ) | 477.9 ( $\pm 276.0)$ | 207-1460 | YES |
|  | TSS (mg/L) | 5.6 ( $\pm 5.7$ ) | 0.2-16.5 | YES |
|  | Turbidity (FNU) | $9.9( \pm 12.0)$ | 0.5-43.2 | YES |
|  | Air Temp ( ${ }^{\circ} \mathrm{C}$ ) | $16.7( \pm 9.3)$ | -3.0-26.0 | NO |


| Ions | Water Temp ( ${ }^{\circ} \mathrm{C}$ ) | 14.8 ( $\pm 7.9)$ | 2.4-24.1 | NO |
| :---: | :---: | :---: | :---: | :---: |
|  | TDS (mg/L) | $324.1( \pm 212.2)$ | 123.0-1090.0 | NO |
|  | DO (mg/L) | $9.2( \pm 2.4)$ | 2.4-12.7 | NO |
|  | DO \% SAT | 94.6 ( $\pm 19.5)$ | 28.4-122.0 | NO |
|  | P. alkalinity (mg/L) | $4.5( \pm 4.3)$ | 1.0-14.0 | YES |
|  | Bicarbonate (mg/L) | $299.7( \pm 118.9)$ | 145.0-653.0 | NO |
|  | Carbonate (mg/L) | $5.3( \pm 5.3)$ | 1.0-17.0 | NO |
|  | Chloride (mg/L) | 3.4 ( $\pm 4.5$ ) | 0.1-18.0 | NO |
|  | Total alkalinity (mg/L) | $253.7( \pm 99.3)$ | 119.0-535.0 | NO |
|  | Sum of ions (mg/L) | 434.6 ( $\pm 248.9$ ) | 188.0-1310.0 | NO |
|  | Total hardness (mg/L) | 242.6 ( $\pm 136.0)$ | 108.0-723.0 | NO |
|  | Sulfate (mg/L) | 27.1 ( $\pm 78.5$ ) | 1.5-340.0 | NO |
|  | Fluoride (mg/L) | $0.1( \pm 0.0)$ | 0.0-0.2 | NO |
|  | Calcium (mg/L) | $52.7( \pm 20.9)$ | 27.0-112.0 | NO |
|  | Magnesium (mg/L) | 27.1 ( $\pm 21.8)$ | 10.0-108.0 | NO |
|  | Potassium (mg/L) | $4.3( \pm 4.4)$ | 1.5-18.0 | NO |
|  | Sodium (mg/L) | $14.3( \pm 16.1)$ | 2.0-75.0 | NO |
| Metals | Mercury (ug/L) | $0.005( \pm 0.004)$ | 0.001-0.012 | YES |
|  | Aluminum (ug/L) | 4.0 ( $\pm 4.4)$ | 0.9-20.0 | YES |
|  | Arsenic (ug/L) | $1.8( \pm 1.8)$ | 0.4-7.9 | YES |
|  | Iron (ug/L) | $235.5( \pm 237.9)$ | 7.9-950.0 | YES |
|  | Manganese (ug/L) | $65.3( \pm 53.3)$ | 4.9-200.0 | YES |
|  | Selenium (ug/L) | $0.9( \pm 0.9)$ | 0.1-3.5 | NO |
|  | Titanium (ug/L) | $0.6( \pm 0.3)$ | 0.1-1.0 | NO |
|  | Boron (mg/L) | $0.04( \pm 0.02)$ | 0.01-0.11 | NO |
|  | Copper (ug/L) | $0.2( \pm 0.2)$ | 0.0-0.9 | NO |
|  | Nickel (ug/L) | $0.7( \pm 0.5)$ | 0.1-2.1 | NO |
|  | Strontium (mg/L) | $0.2( \pm 0.1)$ | 0.1-0.5 | NO |
|  | Uranium (ug/L) | $0.6( \pm 0.7)$ | 0.0-2.8 | NO |


| Vanadium $(\mathrm{ug} / \mathrm{L})$ | $0.3( \pm 0.3)$ | $0.0-0.9$ | NO |
| :---: | :---: | :---: | :---: |
| Zinc $(\mathrm{ug} / \mathrm{L})$ | $1.0( \pm 1.4)$ | $0.0-5.9$ | NO |
| Barium $(\mathrm{mg} / \mathrm{L})$ | $0.05( \pm 0.01)$ | $0.03-0.07$ | NO |
| Chromium $(\mathrm{ug} / \mathrm{L})$ | $0.3( \pm 0.2)$ | $0.0-0.5$ | NO |
| Cobalt $(\mathrm{ug} / \mathrm{L})$ | $0.1( \pm 0.1)$ | $0.0-0.2$ | NO |
| Molybdenum $(\mathrm{ug} / \mathrm{L})$ | $0.6( \pm 0.7)$ | $0.1-2.7$ | NO |

* Includes three erosion-related measures (local watershed/bank erosion, water clarity, and sediment/substrate deposits) combined into one measure of erosion

Appendix I| Mean raw values $( \pm$ SD $)$ and range for the fish metrics $(\mathrm{n}=42)$ that were assessed for potential use in the IBI. Whether or not the metric was retained for use in the Index of Biotic Integrity is also shown. Only the most responsive metrics were retained for use in the IBI.

| Metric Category | Metric | Mean $( \pm$ SD $)$ | Range | Retained? |
| :---: | :---: | :---: | :---: | :---: |
| Richness and Composition |  |  |  |  |
| Measures | Species Richness* | $4.9( \pm 3.0)$ | $1-10.0$ | NO |
|  | Number of Benthic Species | $2.4( \pm 1.9)$ | $0-7.0$ | NO |
|  | Number of Benthic Invertivorous Species** | $0.8( \pm 1.1)$ | $0-3.0$ | NO |
|  | Percent of Benthic Invertivorous Fish | $6.5( \pm 13.0)$ | $0-60.7$ | NO |
|  | Percent of Benthic Invertivorous Species | $12.0( \pm 15.5)$ | $0-50.0$ | NO |
|  | No. of Subterm Mouth Minnow Species | $1.5( \pm 1.3)$ | $0-4.0$ | NO |
|  | Percent Subterm Mouth Minnows | $30.9( \pm 31.7)$ | $0-100.0$ | NO |
|  | Number of Cyprinids and Catastomid Species (excluding | $2.3( \pm 1.9)$ | $0-5.0$ | NO |
|  | FTMN) | $0.9( \pm 0.9)$ | $0-3.0$ | NO |
|  | No. of Insectivorous Cyprinid Species | $19.5( \pm 26.9)$ | $0-100.0$ | NO |


|  | No. of Cyprinid Species | $2.1( \pm 1.9)$ | 0-6.0 | NO |
| :---: | :---: | :---: | :---: | :---: |
|  | No. of Cyprinid Species (Excluding Tolerants (FTMN)) | 1.6 ( $\pm 1.6)$ | 0-5.0 | NO |
|  | Percent Cyprinidae Species (Excluding Tolerants) | 27.0 ( $\pm 26.6)$ | 0-100.0 | NO |
|  | Number of Benthic and Water Column Fishes | 3.8 ( $\pm 2.6$ ) | 1-9.0 | NO |
|  | No. of Water Column Species | $1.4( \pm 1.0)$ | 0-4.0 | YES |
|  | Percent of Water Column Fishes | 34.5 ( $\pm 35.5$ ) | 0-100.0 | NO |
|  | Percent Coldwater Fish | 37.1 ( $\pm 32.6)$ | 0-100.0 | NO |
|  | Percent Coolwater Fish | 47.2 ( $\pm 32.8$ ) | 0-100.0 | YES |
|  | Percent Coldwater/Coolwater Fish | 84.3 ( $\pm 27.0)$ | 0-100.0 | NO |
| Tolerance Measures | Percent Brook Stickleback | $15.9( \pm 23.8)$ | 0-90.0 | YES |
|  | Number of Sensitive (or Intolerant) Species* | $0.7( \pm 0.9)$ | 0-3.0 | NO |
|  | Percent Intolerant Individuals | $5.4( \pm 12.2)$ | 0-59.0 | NO |
|  | Number of Tolerant Species | $1.1( \pm 0.8)$ | 0-2.0 | NO |
|  | Percent Tolerant Individuals | 23.6 ( $\pm 29.4$ ) | 0-100.0 | NO |
|  | Percent Fathead Minnow | 12.6 ( $\pm 24.3)$ | 0-91.4 | NO |
|  | Percent of Tolerant Reproductive Guild | 28.5 ( $\pm 33.8)$ | 0-98.3 | NO |
|  | Percent White Sucker | $11.1( \pm 21.8)$ | 0-100.0 | YES |
| Trophic/Habitat/Reproductive |  |  |  |  |
| Measures | Percent Omnivores*** | $32.7( \pm 32.5)$ | 0-100.0 | NO |
|  | Percent Top Carnivores and Piscivores*** | 30.8 ( $\pm 37.8$ ) | 0-100.0 | YES |
|  | Percent Invertivores | 56.3 ( $\pm 33.5$ ) | 0-100.0 | YES |
|  | Percent Benthivores | 37.1 ( $\pm 32.0)$ | 0-100.0 | NO |
|  | Percent Benthivores (Excluding WHSC) | 26.0 ( $\pm 28.3)$ | 0-93.3 | YES |
|  | Percent Generalists | 28.5 ( $\pm 33.8$ ) | 0-98.3 | NO |
|  | Percent Litho-Obligate Fish** | 43.2 ( $\pm 35.4)$ | 0-100.0 | NO |
| $\stackrel{\text { Fish Abundance and }}{ } \quad \begin{gathered}\text { Condition }\end{gathered}$ | Relative Abundance (CPUE, fish/100s)* | 3.8 ( $\pm 5.8)$ | 0-22.6 | NO |


| Relative Abundance (Excluding Tolerants (CPUE, |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: |
| fish/100s)) | $2.4( \pm 3.8)$ | $0-18.6$ | NO |  |
| Relative Abundance of Coldwater Fish (CPUE, Fish/100s) | $1.0( \pm 1.7)$ | $0-6.6$ | YES |  |
| Relative Abundance of Coolwater Fish (CPUE, Fish/100s) | $1.7( \pm 3.7)$ | $0-19.0$ | NO |  |
| Relative Abundance of Coldwater/Coolwater Fish (CPUE, | $2.7( \pm 4.0)$ | $0-19.1$ | NO |  |
| Fish/100s) | $7.2( \pm 8.8)$ | $0-30.0$ | NO | Ner |
| Percent Individuals with DELT | $5.5( \pm 8.8)$ | $0-30.0$ | YES |  |
| Percent of Individuals with Parasites | $0.9( \pm 0.1)$ | $0.6-1.2$ | NO |  |
| Condition of Sentinel Species (WHSC) |  |  |  |  |

[^2]
[^0]:    Figure 4. 15| Annual variability in Index of Biotic Integrity scores for each of the five revisited sites over the three-year sampling period. 113

[^1]:    *Denotes a livestock/agricultural guideline.
    Guidelines developed for a similar region or waterbody type were used. Where a biogeographically similar guideline did not exist, the most sensitive/restrictive criteria existing in the literature was consulted
    For sites that had repeat site visits, if $\geq 1$ year exceeded the guideline it was included only once.
    All criteria thresholds are based on chronic values.

[^2]:    * Indicates the metric was used in Karr's original Index of Biotic Integrity (1981)
    ** Indicates the metric was used in the Index of Biotic Integrity created for the Alberta portion of the Beaver River Watershed (Cantin \& Johns 2012)

