University of Alberta

Evaluating Water Quality and Biotic Indices in the Lower Little Bow River, Alberta

by

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DEDICATION

I dedicate this work to my family and friends.

To Tyson Gallagher, my greatest supporter, who worked so I could think and write. He is the best field assistant and partner I could ask for and without his support I would be neither where I am nor who I am today.

To my late mother, Victoria Ross, who started me on this long path with her constant encouragement to continue my education. I am sure that her spirit has quickened my word processing skills and I hope that she would be as delighted with the content of my writings as with the fact that I now touch type at over 60 words per minute.

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This work was made possible by your help and encouragement.

ABSTRACT

Environmental indices are useful tools for distilling significant messages out of complex monitoring datasets. A case study of Agriculture and Agri-Food Canada's Watershed Evaluation of Beneficial Management Practices (WEBs) micro-watershed on the lower Little Bow River, Alberta, was used to address two topics: modifications of the Canadian Water Quality Index (CWQI) to provide ecologically relevant monitoring information at the micro-watershed scale, and inclusion of fish species and habitat assessments to support WEBS' broader evaluation of aquatic ecosystem health.

Water quality data collected between 2004-2007 from five study reaches on 5.5 km of the lower Little Bow River were used to calculate CWQI scores under two scenarios: seasonal vs. annual index calculation; and total vs. sub-index (i.e., biological, chemical, and physical) divisions of parameters. Fish diversity and habitat information was collected in a single season in 2009.

Overall, water quality ranged from good to poor. Summer criteria exceedances in fecal coliform, *Escherichia coli*, dissolved oxygen, and total suspended solids produced marginal scores for summer and annual periods, and poor scores for physical and biological sub-indexes. Fish collection found largely generalist warmwater fish species, including minnows, suckers, and northern pike, so index criteria suited to these species were employed. Habitat permanence relied upon maintenance of minimum instream flows, bank stability, and access to overwintering habitats.

Both seasonal and sub-index methods are recommended for use in micro-watershed monitoring as they produced wider score ranges than the standard CWQI and can inform reservoir management of stream flows when related to local fish requisites.

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1 INTRODUCTION

1.1 Background

From ancient times to modern days, the complexity of water processes and interactions has been noted. Sanders et al. (1983) observed that water quality is the result of both human actions and hydrological processes, and so today's water quality management must deal with both. This management of water quality and potential impacts resulting from human activities like agricultural land use is a broad and present concern. Water users and resource managers at local, regional, and national scales invest in what are hoped to be effective practices to maintain water quality. The determination of what is effective, both from an ecological and economical perspective, requires that we have a manageable, rational, and understandable way to measure success. Also important is that decision-makers have access to a valid and scientifically supportable measure of water quality.

Water quality monitoring seeks to describe spatial and temporal patterns in physical, chemical, and biological characteristics of a waterbody. The challenge in this seemingly basic task is reflected by the myriad programs, designs, and models developed to support the tracking of chemical, physical and biological characteristics of aquatic environments (Thomann 1972, Ward et al. 1986, Chapman 1996, USDA 1996). Drawing information out of water quality observations is a fundamental task in environmental monitoring and one that requires the consideration of the whole character of the medium in relation to its function. Water quality indices, in various forms, have been developed as tools for this purpose over the past four decades (Horton 1965, Harkins 1974, Landwehr et al. 1974, Inhaber 1975, Cude 2001, Sarkar and Abbasi 2006, Parparov and Hambright 2007) and are often applied across broad geographical regions to rate degradation of water quality and identify priority areas for management actions. Application of water quality indices to track change at a smaller scale within a small watershed or on one river channel may be challenged by the need for site-specific criteria (Khan et al. 2005), and potential issues of information resolution.

Present water management approaches such as Alberta’s Water for Life strategy (Government of Alberta 2003) establish concurrent goals of supporting both aquatic ecosystem health and the capacity for sustainable human use of water. Integrating physical, chemical, and biological surface water quality analysis with biological monitoring can support a more holistic evaluation of water's many functions, engaging us in the consideration of water quality and quantity needs of all water users, human, and otherwise. Basic information regarding the aquatic community of a watercourse, such as may be obtained through a fish community assessment, is required as a first step in the design of biomonitoring tools. A survey of the receiving environment is also a key component for the establishment of site- or watershed-specific targets for water quality; these targets or objectives support the selection of water quality criteria and index rating scales when indices are applied to evaluate water quality conditions at a local, within-watershed scale.

The complexity of processes means that abstracting the essence of water quality conditions at a reasonable cost is often very difficult (Sanders et al. 1983). Interacting factors suggest we are better off making sure we look at enough parts of the whole system rather than just looking for change in one component (deRosemond et al. 2003, Heathwaite 2010). Information expectations placed on water quality monitoring are often far beyond the ability of the network to supply such information (Sanders et al. 1983). Does a water quality index give us information we can use to understand changes resulting at a local scale from agricultural beneficial management practices (BMPs)? What other indicators might we monitor to detect changes to aquatic and riparian ecosystems? For research in southern Alberta prairie watersheds, can we suggest or reinforce others' suggestions how to better monitor whole systems by integrating monitoring results from the parts (water chemistry + fish + aquatic organisms + riparian plants)? This analysis may suggest changes to a monitoring program that might allow us to capture needed information in different ways, perhaps in alternative ways that better support management decision making (deRosemond et al. 2003, Cimorelli and Stahl 2005).

Given some of these challenges with the structure and design of indices and their frequent use to compare water quality across regional scales, my research aims to examine the value of using indices for site scale water quality monitoring, to compare index outcomes with present single parameter methods for evaluating BMP effectiveness, and to propose mechanisms for introducing broader measures of aquatic ecosystem health to water quality monitoring programs using water quality indices. The Canadian Water Quality Index (CWQI) was selected for use because of its national scope and widespread use within Canada and federal environmental monitoring initiatives. The CWQI also allowed the calculation of index values for the WEBs suite of parameters which omitted parameters required by other regional and provincial water quality indices.

1.2 Research Objectives

Aggregation methods such as indices are used for interpreting water quality at national, provincial, and regional scales, yet their applicability to the measurement of water quality changes associated with agricultural land uses in micro-watershed and reach-scale studies of water quality is uncertain. The information an index contains may usefully generalize water quality over a large area but not at a farm or site, or on an annual basis but not a season. Research objectives for this thesis were:

- To evaluate the influence of index criteria values and frequency of calculation on water quality index ratings generated to compare water quality changes associated with agricultural beneficial management practices; and
- To assess the fish community of the WEBs project area and relate biotic index measures of composition and reach habitat to water quality and aquatic ecosystem health impacts of agricultural BMPs in the Lower Little Bow River Watershed.

Each objective is presented in a thesis chapter, followed by a final synthesis chapter. The synthesis chapter contains a discussion of how the index and fish community information generated might aid water quality managers in effectiveness monitoring for small-scale and local agricultural land-use changes designed to improve water quality and aquatic ecological health.

1.3 Agricultural Beneficial Management Practices

Beneficial management practices are structural or operational techniques designed to be cost-effective and practical means to minimize environmental impacts. Agricultural BMPs are farm management practices that:

- minimize and mitigate impacts and risks to the environment, by maintaining or improving the quality of soil, water, air, and biodiversity;
- ensure the long term health and sustainability of natural resources used for agricultural production; and,
- support the long-term economic and environmental viability of the agriculture

industry (Agriculture and Agri-Food Canada 2008a).

In 2004 Agriculture and Agri-Food Canada (AAFC) began a multi-year national project, the Watershed Evaluation of Beneficial Management Practices (WEBs), in seven watersheds across Canada (including the Lower Little Bow River) to examine the use of agricultural best or beneficial management practices (BMPs) to address surface water quality. BMPs for water quality are structural or operational techniques used to reduce an impact to water quality, and they find growing use in agriculture, forestry, and urban stormwater management as resource managers work to improve the environmental sustainability of their activities.

BMPs to improve surface water quality evaluated by AAFC's WEBs program (AAFC 2008b) include:

- Buffer strips – the combined effect of vegetation type and buffer width on runoff water from irrigated fields evaluated using in-field buried runoff collectors;
- Beef manure management – to compare runoff water quality from plots with no manure to plots with application rates based on the crop's annual nitrogen requirement, its annual phosphorus requirement, and its three-year phosphorus requirements;
- Off-stream watering with fencing – the effect of installing cattle fencing along a 800 m reach to eliminate cattle access to the riparian area with the installation of an off-stream watering system monitored through upstream and downstream water quality;
- Off-stream watering without fencing – the effect of installing an off-stream watering system in a winter and summer pasture used by 500 head of cattle monitored through water quality in the river before and after BMP implementation; and
- Conversion of annual cropland to forages – the effect of crop conversion from barley to forage evaluated by comparing water quality of barley runoff to forage runoff on two irrigated barley fields adjacent to the river.

1.3.1 Assessing BMP Effectiveness

Much of the work in assessing BMPs has focused on success in implementation and adoption of BMPs, but a practice will only be successful if it results in measurable and significant benefits to water quality. Monitoring the effectiveness of these techniques in maintaining or improving water quality is a critical step supporting BMP development and implementation, particularly as much of our present understanding of potential BMP effectiveness relies on assumptions of modeling. The assessment of BMP effectiveness involves evaluating the success of adoption and implementation, as well as the achievement of end-goals the BMP was designed to address. Monitoring effectiveness of a BMP program may be difficult because of the lack of control over exactly what happens and when it happens, the compensation interactions among BMPs, and the gradual water quality responses to changes in practices that may result from stores of the pollutant of concern in the watershed (USDA 1996).

The experimental design used by WEBs water quality sampling employs a modified before-after-control-impact (BACI) design. Water sampling locations are associated with watercourses passing though agricultural areas treated with one or multiple BMPs. Samples are collected from upstream (control) and downstream (impact) locations on the subject watercourse, supporting comparisons between the control and impact samples to describe the effect of the BMP treatment. In some circumstances, water quality sampling completed prior to implementation of the BMP (before) is available and supports a comparison to samples collected post-implementation (after). These comparisons are often made using experimental designs evaluating water quality parameters in two time periods (pre- and post-BMP implementation) and between control and impact (upstream and downstream) sites.

Various researchers have discussed the challenges of statistically valid analysis with BACI designs (Hurlbert 1984, Smith et al. 1993, Underwood 1994), particularly for observational or impact assessment studies (Stewart-Oaten et al. 1986). These include the lack of opportunity for treatment replication or missing pre-impact baseline information in many impact assessments. Upstream-downstream sampling schemes present the potential for upstream loading sources (such as irrigation return flows) to mask the effects of the investigated sources, because individual inputs are often small compared to the cumulative inputs from upstream (Spooner et al. 1985). Monitoring results to date from the lower Little Bow River WEBs watershed have examined percent change in individual parameters between upstream and downstream on-stream sites. In the case of the Lower Little Bow River watershed, Miller et al. (2010) suggested that sediment and contaminants in irrigation return flow upstream of this reach may mask the BMP response.

1.3.2 Adaptive Improvement of Monitoring Programs

Monitoring programs are improved through thoughtful consideration of their effectiveness. The design of a monitoring program is shaped by information about the watershed's behaviour; as results are generated by a monitoring program, more information about the watershed becomes available. This may lead to an opportunity to modify or alter the monitoring program to better achieve its objectives or to realize cost savings in staff time and/or analytical expense. The need for adaptive improvement of water quality monitoring programs has recently been demonstrated by the re-design of water quality monitoring systems in the Lower Athabasca River basin and Alberta Oilsands. Challenges were made to the monitoring program's ability to produce the information needed for management decisions by academics and environmental non-governmental organizations; these prompted review by provincial regulators and the involvement of the federal government and external expert reviewers (Environment Canada 2011). Ultimately, an effective monitoring program should produce information adequate to support resource decision-making.

Work by WEBs (Miller et al. 2008) focuses on the evaluation of BMP water quality effects through water quality surveys completed along with comprehensive study of the watershed form and function within a micro-watershed on the Lower Little Bow River. Taken with earlier work by Little (2001) at the watershed scale and Little et al. (2003) at the sub-basin scale, this research program follows the strategic cyclical scaling (SCS) research paradigm of Root and Schneider (1995). SCS involves continuous cycling between large- and small-scale studies, with large-scale associations used to focus small-scale investigations to ensure that tested causal mechanisms are generating the large-scale relations.

1.4 Understanding the Quality of Water

Water quality is typically understood as a value statement regarding the condition of water and its suitability for use. Quality, in environmental terms, represents those physical, chemical, and biological conditions of the medium that best enable it to meet its functions. Many monitoring programs describe how water quality differs from defined objective values, either established water quality guidelines (e.g., Canadian Council of Ministers of the Environment (CCME) Water Quality Guidelines for Aquatic Life) or site-specific objectives (CCME 2001, CCME 2004, Gartner Lee 2006). A water quality index may also be used as a means to consolidate water quality measurements from multiple parameters, and this value is then used in trend analysis or as a broad watercourse health ranking tool. Some of the ways we presently analyze water quality data include:

- comparisons of instantaneous measured values relative to criteria or objectives, using percent exceedance or some measure of exceedance frequency;
- comparisons of total maximum daily load or dose measures for sampled parameters to water quality criteria or objectives; or
- linking data to qualitative water quality rankings using some type of index method which combines the above analyses.

1.4.1 Parameters: Characterizing the Medium

We examine physical, biological, and chemical characteristics of water as a means to describe the complex nature of water and to capture signals of various processes occurring within the watershed. The aggregation of individual parameters into biological, physical, and chemical divisions is a common first strategy employed to organize what are often complex analytical results generated by water quality sampling programs. By grouping parameters together into broader categories based on their similar genesis or their uses as process indicators (e.g., nutrients, pesticides, etc.), analysts can begin to relate water quality observations to understand watershed processes and hydrologic behaviour. Relationships among parameters should also be considered and discussed as some parameters play a critical role in regulating the activity of other parameters, and a statement of parameter interdependence should be included in water quality analysis. Some parameters may be related to one another more directly: for example, some nitrogen-related parameters, although measured separately, may be grouped together in some analytical methods. What appear to be distinct and individual parameters are actually values dependent on one another, derived as a ratio or remaining fraction from the larger total.

Sanders et al. (1983) suggested the following hierarchical ranking of the multiple water quality-water quantity variables that can be monitored, in part to assign some level of importance:

- first level variables = water quantity (discharge, level, volume);

- second level variables = variables of aggregated effects (temp, DO, pH, turbidity, BOD, cations, anions, conductivity, chlorides, radioactivity);
- third level variables = variables that produce aggregated effects (radioactivity producing compounds, turbidity producing variables); and
- fourth level variables = detailed, specific compounds or species.

1.4.2 Use-Based Determination of Water Quality

> *"In reality, water quality is never considered in all its dimensions - the assessment is driven by the perceived importance of the aquatic environment, the objectives of operation and the human and financial resources".*
>
> *(Meybeck et al. 1996, p.32).*

What is considered to be a good resource varies depending on perception and need. Water is no different – we consider its quality by evaluating its condition against a scoresheet of some kind, assessing how well the water will meet our needed condition for use. To formalize this evaluation, separate water quality guidelines exist for a variety of uses: drinking water, agricultural use (i.e., irrigation water and livestock water), recreation, aesthetics, and for the protection of freshwater aquatic life (Alberta Environment 1999, CCME 2001, CCME 2004). Each guideline presents standards or criteria to gauge the suitability of water for a particular use, drawing on those parameters that have particular relevance or critical influence on that use. A hierarchy of use is also apparent, with greatest precautions afforded to drinking water quality standards and protection of freshwater aquatic life.

1.4.3 Factors Influencing Water Quality

Because water is the universal solvent, it is a medium that reflects its history of geological contact through its chemical character. As a transporter of sediments, nutrients, and pathogens, water moves materials between sources and sinks. Water is also the matrix supporting aquatic organisms from primary producers like algae and macrophytes to higher level animals like invertebrates and fish.

Agricultural effects on water quality have long been studied. Reach level channel morphology is influenced by surrounding landscape features (e.g., valley slope and confinement, bed and bank material, riparian vegetation) interacting with the dynamics of flowing water (Montgomery and MacDonald 2002, Church 2002). Human alterations of the landscape (e.g., road crossings, channel alignment, water withdrawal, and conversion of land cover) can alter water and sediment supply, stabilizing or destabilizing channel shape and manifesting alterations in habitat (Allan 2004). Some of the key modifications of the landscape, water balance, and nutrient regime brought about by agricultural land use have known relationships with water quality (Walling 1980, Kauffman and Kruger 1984, Chambers et al. 2002). These are:

- The development of irrigation, which may disrupt the natural water balance of the soil. Application of water for crop production in semi-arid areas results in increased evapotranspiration, which can cause accumulation of soluble salts within the soil which may be released to surface waters. Local increases in the level of the water table may cause capillary rise of saline groundwater.
- The use of soil fertilizers nitrogen and phosphorus, which may be applied in excess of what is needed by crop growth. Nitrogenous fertilizers, associated with ammonium (NH_4^+) and nitrate (NO_3^-) ions, may be readily leached from soils into groundwater and runoff. Phosphorus forms applied in fertilizer are readily adsorbed by soil particles and their loss from the catchment may be primarily associated with erosion and transport of suspended sediment.
- The acceleration of slope and channel erosion, which are related to high discharge and storm events. Many studies have shown relationships between suspended sediment concentrations and solute concentrations with discharge. Solute concentration may have a variable relationship with discharge, depending on the source of solute-rich waters. Greater discharge results in greater concentrations of a solute in stream waters if the sources of the solute-rich waters are surface runoff or return flows; lower concentrations with increased discharge is explained by dilution of solute-rich baseflows.

1.5 Indicators

As Turnhout (2003) defined, “an ecological indicator stands for a framework of parameters that indicates the current and/or desired ecological or nature quality of a certain area”. Environmental indicators are those factors we have decided convey a simplified measure of quality in a more complex system, to “objectify the concept of quality” (Turnhout et al. 2007). Indicators ideally allow “assessment of both existing and emerging problems, diagnosis of the anthropogenic stressors leading to impairments, establishment of trends in condition for measuring environmental policy and program performance, and ease of communication to the public" (Neimi and MacDonald 2004). While commonly understood indicators of agri-environmental sustainability are most often response indicators (air, soil and water quality, and biodiversity) (Palliser Environmental Services and AARD 2008), indicators may be defined to measure environmental stressors or pressures.

In 2008, Alberta Environment produced *Indicators for Assessing Environmental Performance of Watersheds in Southern Alberta*, a document presenting a framework for evaluating the environmental performance of watersheds in Southern Alberta (Alberta Environment 2008). This document presented indicators/measures to track and suggests that the defined, specific thresholds be made locally valid, stating: "reach-specific thresholds and targets will be needed that represent socially, economically, and scientifically acceptable compromises between the various human uses of water and the protection and restoration of healthy aquatic and riparian ecosystems."

Indicators are positioned at the science - policy interface, and hence they must be flexible and able to follow shifts in this boundary, resulting from advancements in science or changes in policy (Turnhout et al. 2007). Does it not then also follow that indicators should be selected after consideration of available science (natural limits of ecological conditions) and policy questions at the scale of the study? This is one reason why researchers considering the use of indicators and indices established at a broader regional scale need to invest time in the design or consideration of site-specific criteria. As Cimorelli and Stahl (2005) noted, "how these indicators are constructed and how they are used in policy analysis is critical to how the problem is defined, what kinds of solutions will be generated, and how those solutions will be evaluated"(p.51).

1.5.1 Water Quality Indicators

Parameters of water quality are measured or analysed from water samples collected as representative of water within the waterbody or aquifer of interest. Kristensen and Bøgestrand (1996) categorized water quality parameters as follows:

- basic variables used for a general characterization of water quality (e.g., water temperature, pH, conductivity, dissolved oxygen, and discharge);
- suspended particulate matter (e.g., suspended solids, turbidity and organic matter);
- organic pollution indicators (e.g., dissolved oxygen, Biochemical Oxygen Demand (BOD), Chemical Oxygen Demand (COD) and ammonium);

- eutrophication indicators: nutrients (e.g., nitrogen and phosphorus), and various biological effect variables (e.g., chlorophyll a, Secchi disc transparency, phytoplankton and zoobenthos);
- acidification indicators (e.g., pH, alkalinity, conductivity, sulphate, nitrate, aluminum, phytoplankton and diatoms);
- specific major ions (e.g., chloride, sulphate, sodium, potassium, calcium and magnesium) as essential factors in determining the suitability of water for most uses, including public water supply, livestock watering and crop irrigation;
- metals (e.g., cadmium, mercury, copper and zinc);
- organic micropollutants such as pesticides and the numerous chemical substances used in industrial processes (e.g., polychlorinated biphenyl (PCB), hexachlorocyclohexane (HCH) and polycyclic aromatic hydrocarbons (PAHs));
- indicators of radioactivity (e.g., total alpha and beta activity, isotopes of cesium and strontium, ^{137}Cs and ^{90}Sr);
- microbiological indicator organisms (e.g., total coliforms, faecal coliforms and faecal streptococci bacteria); and
- biological indicators of the environmental state of the ecosystem (e.g., phytoplankton, zooplankton, macrophytes, benthic invertebrates and fish).

While a complete inventory of water characteristics would ideally include as many parameters from these categories as possible, each water quality monitoring program selects parameters to include in its sampling based on study goals.

Early water pollution control recognized the need for measures of both the condition of waters receiving pollutants and the polluting effluent being discharged (Dreschler and Nemetz 1978, Minton et al. 1978). Indicators of pollution were identified as signals of shifts to problem states: for example, indicators of eutrophication (nutrient-introduction) might be alterations to a discharging water's nitrogen or phosphorus content or increased algal biomass in receiving waters.

Early monitoring for water quality also largely addressed stressor monitoring as a defined point-source of pollution discharge that posed a clear threat to water quality if permitted to continue. Stressor monitoring focuses on the 'end of pipe' or cause of pollution or change, with a stressor being anything that can induce adverse effects (Roux et al. 1999). Recognition of non-point sources of pollution required the adoption of alternative forms of monitoring, like response monitoring for signs of distress or degradation in biota. Non-point sources are, by their nature, more diffuse, but may have an equal or greater cumulative effect once collected into a receiving waterbody. Response monitoring is designed to capture change occurring within the receiving media and biota (Roux et al. 1999) and such changes in biota may be related to stressors, such as pollutants, acting within the system. However, definitive causal associations between a response and a stressor can rarely be determined in natural systems, and the relationship is often established through a weight of evidence approach (Cormier 2010).

One of the main cautions with any such monitoring is that the cause and effect relationships linking stressors and responses are, in many cases, derived from laboratory experimentation and poor extrapolation to the real environment has been noted (Cormier 2010, Mebane 2010). One of the critical benefits of many monitoring programs is that they endeavour to support or refute such assumptions through field testing. Examples of changed watersheds exist, as Walling (1980) noted: “it must be appreciated that the various processes controlling stream water quality may be in a delicate balance and that a slight modification to the catchment, such as a change in land use, could generate significant changes in water quality"(p.2). It is, however, also likely to be true that the processes controlling stream water quality in another watershed may not be in such a delicate balance that a modification in land use would generate a significant change in water quality. Evaluating response in water quality indicators provides one measure of our understanding of those pollution transport processes associated with land use changes, including the adoption of beneficial management practices (BMPs).

1.5.2 Aquatic Ecosystem Health Indicators

Aquatic ecosystem health indicators have been suggested for both agricultural watersheds (Palliser Environmental Services and AARD 2008) and watersheds in Southern Alberta, in general (Alberta Environment 2008). These include condition indicators that are natural parameters associated with widespread water quality concerns in southern Alberta: total suspended solids, nutrients, dissolved oxygen, temperature, and pathogens.

Because riparian areas represent an ecotone or intermediate zone of interactions between aquatic and terrestrial environments, riparian health indicators are incorporated into aquatic ecosystem health monitoring. Riparian health may be characterized by factors like ground cover, plant community composition, woody stem numbers, among others, yet defining a truly healthy riparian ecosystem remains a challenge. In many watersheds subject to human alteration, particularly in those for which historical, pre-development baseline information is lacking, riparian health is defined against an ideal condition that may not adequately consider site characteristics. Gregersen et al. (2007) noted “it is clear that a watershed in arid regions with steep slopes, naturally sparse vegetative cover and shallow soils needs to be judged differently in its condition than a watershed in more humid regions with forest cover and deep soils” (p.25). Yet, there is often a tendency to judge all riparian habitats against an ideal associated with well-vegetated, treed, humid conditions more typical of upper portions of basins. Healthy riparian ecosystems are those naturally formed by the climate, soils, and flood regime of a river system, and these conditions may be reasonably expected to vary across any given basin. Rating riparian condition, from poor to good, is most directly tackled by comparing if watersheds have “deviated significantly from their natural or undisturbed characteristics” (Gregersen et al. 2007, p.25).

The term ‘habitat’ implies more than simply an environment or physical space – it also describes the ability of an environment to support the occupancy and survival of an organism. The quality of habitat is often evaluated by the suitability of physical, chemical, and biological components of the environment for the growth and survival of taxa of management concern such as fish. Changes to the physical, chemical, or biological character of a watercourse alter habitat function, and so may shift individuals’ patterns of use and rates of growth, reproduction, and mortality. These individual shifts are aggregated to population and community levels, and large-scale observational biodiversity patterns (e.g., species-abundance distributions, species–area curves, body size-diversity distributions) are considered to reflect the underlying processes that structure ecological communities. Using fish community structure to evaluate the influence of non-point source on aquatic ecosystem integrity has noted challenges, as both natural and anthropogenic factors can affect habitat stability of fish communities and alter the normal structural and functional dynamics of fish communities (Ibarra et al. 2005). Observed community patterns are likely the result of both positive and negative interactions within and among species and often interpretation of the limited catch information provided in fish surveys, particularly in single season assessments, requires broad assumptions of species interactions and individual behaviours.

The use of indicators in monitoring to support sustaining aquatic ecosystems was discussed in Roux et al. (1999). In South Africa, a policy shift to adopt national "Water Law Principles" supporting sustainability and ecological reserve objectives required water regulation and management changes. Focus shifted from the assimilative capacity of water (how much of a pollution burden or waste removal service can the watercourse bear) to the ecological capacity of water (how well does the system recover from disturbances). A similar focus may be seen in the progressive inclusion of ecological indicators with water quality surveys in other parts of the world (Karr 1993, Alberta Environment 2008).

1.6 Indices

Scientifically derived environmental indicators are central to environmental decision analysis. Scientists can bring value to decision makers by providing indicator information in a variety of forms to support different perspectives on environmental management choices and questions (Cimorelli and Stahl 2005). One of the varieties of forms used to present indicator information is an aggregated indicator – one number or score that further simplifies the information. Such a tool is also referred to as an index, a single number derived from information provided by more than one environmental indicator (Ott 1978).

Ott (1978) identified six basic uses of environmental indices: to aid managers in resource allocation; to rank environmental conditions across geographical locations; to determine the extent to which standards are being met or exceeded at a specific location; to track changes in condition through time; to inform the public about environmental conditions; and to reduce data to a form that may provide insight to researchers studying some environmental phenomenon. As he stated: "ideally, an index or an indicator is a means devised to reduce a large quantity of data down to its simplest form, retaining essential meanings for the questions that are being asked of the data" (Ott 1978, p.2).

Indices find wide-spread application in today’s society, particularly to enable us to track change in a complex system. From economic evaluators like the Consumer Price Index, to environmental initiatives like the Environmental Sustainability Index, an index can inform the public and policy decision-makers on the magnitude and consistency of differences. Any index can have inherent biases based on subjective weighting or selection of composite metrics; given this, exploration of these potential biases is critical before one index approach is selected over another.

1.6.1 Establishing Criteria

Various terms – objectives, standards, guidelines, criteria – are used to label the desired endpoint against which we compare a given water's condition. Alberta Environment's Surface Water Quality Guidelines for Use in Alberta utilizes the term "guideline", defined as a concentration or narrative statement recommended to support and maintain a designated water use (Alberta Environment 1999). Other jurisdictions may use "criteria" in a similar way, although criteria are also often used to associate a concentration or level with a degree of environmental effect. For guideline or criteria values that are established for a specific site, the term "objective" may be used, for example, to describe a reach-specific guideline. The term "standard" is used for those objectives that are prescribed in environmental regulation (Alberta Environment 1999).

Standards, or criteria, are those values of indicators that represent critical points in quality. Numeric values for minimum acceptable concentrations, maximum allowable concentrations, total maximum daily loads, or numbers of colony forming units are a few examples of commonly applied water quality measures used in standards. Standards may also be in the format of a narrative statement, for example, describing acceptable colour values, suspended sediments, or pH changes (Alberta Environment 1999). Minton et al. (1978) stated that "a basic problem with standards, both receiving-water and effluent as they exist today, is that their use too often implies a deterministic certainty in an uncertain world"(p.1442). The decision regarding what those standard or criteria values will be, and their resulting role in a water quality index, is a complex process involving the contribution of many water quality, ecotoxicology, and human health experts (deRosemond et al. 2003).

In Canada, the CCME provides a national forum for the discussion and establishment of water quality standards, and work continues to evaluate the fit of national standards (through the CCME's Canadian Water Quality Index) to water quality management questions in watersheds across the nation (Gartner Lee 2006). Water quality standards are designed to be protective, but suitability of the standard to the stress-responses in a particular watershed needs to be evaluated before application of a water quality index. Given the uncertainties present in the geographical range of Canadian watersheds alone, consideration of natural variation in a system, potential for exceedances to occur outside of momentary points of monitoring, and timing of quality changes with human uses and needs is critical (Minton et al 1978).

1.6.2 Index Scaling

Scaling, in water quality indices, refers to the establishment of a range of possible quality ratings or scores against a range of possible values of a given parameter. Water quality indices may be scaled across any range; they are frequently established on a scale of 0 to 100, with 0 representing the poorest water quality and 100 representing the best water quality. The range may be further divided into categories and assigned value labels. Alberta Environment's River Water Quality Index uses categories of poor (index score of 0-45), marginal (index score of 46-65), fair (index score of 66-80), good (index score of 81-95), and excellent (index score of 96-100) to represent the concurrence of the observed state of water quality with established guidelines for river water quality health (Alberta Environment 2011c).

An index scaled to evaluate watercourses province-wide should include the range of water quality observed across the spectrum of watercourses found in the province, but using the province-wide index for local-scale monitoring may produce ratings within a range too narrow to enable valid change detection. For this reason, researchers have explored the use of watershed-specific indices and site-specific water quality objectives (Gartner Lee 2006). If the objective of a water quality index used at a local scale is to rate quality changes within that one watercourse, we should consider the intrinsic bounds of water quality set by natural watershed characteristics. A lower gradient, coolwater stream draining through erosive bank materials will produce water of a different quality than a higher gradient, cold water stream draining a forested catchment; scoring water quality schemes that consider the best water quality possible achievable within the natural conditions of the watershed address this. It is important to explicitly discuss and consider what reference conditions should be assigned as "natural" to the watershed. In the case of monitoring change in already altered watersheds shaped by water management, flow regulation, and a long history of grazing and riparian degradation, the determination of natural baseline conditions is challenging.

> *"As for any index, inappropriate use of the AAWQI [Alberta Agricultural Water Quality Index] could lead to misleading or erroneous conclusions about water quality. However, if the index is applied appropriately and if the resulting values are set in the proper*

context, the AAWQI could become a powerful tool for describing water quality in Alberta's agricultural areas."

(Wright et al. 1999, p.ii).

1.7 Study Area: The Lower Little Bow River Micro-watershed

The following sections introduce the lower Little Bow River Micro-watershed and provide background information relating to environmental factors influential to watershed function, water quality and aquatic ecosystem health within the study area.

1.7.1 Riverine Systems

Alberta contains seven major river systems: the Peace, Athabasca, Hay, Beaver, North Saskatchewan, South Saskatchewan, and Milk river basins. Alberta rivers typically represent the upper headwaters of large continental-scale drainage systems. Commencing from points along eastern slope of the North American Continental Divide, Alberta's rivers carry flows formed by snow melt and glacial runoff, rainfall and subsurface flow through a landscape marked by an increasing gradient of human landscape alteration.

1.7.1.1 Regional Setting: South Saskatchewan Basin and Oldman River Sub-basin

The South Saskatchewan River Basin is the largest of the river basins in southern Alberta with a total watershed of 121,095 km^2 (Alberta Environment 2011a). It includes several sub-basins collecting into four rivers, the Red Deer, Oldman, Bow, and South Saskatchewan Rivers, that flow from headwaters in the Rocky Mountains, through the Alberta foothills to prairie and into the adjacent province of Saskatchewan. Waters from the South Saskatchewan River basin drain eastward through Saskatchewan and Manitoba. Mean annual discharge from the Alberta South Saskatchewan River Basin into Saskatchewan is 9,280,000 dam^3 (Alberta Environment 2011a).

Within the South Saskatchewan River Basin, the Oldman River forms the second largest of the four sub-basins of the South Saskatchewan River Basin and covers approximately 23,000 km^2 in southwestern Alberta and 2,100 km^2 in Montana the (Alberta Environment 2011a). Headwaters of the Oldman River and its major tributaries lie in the Rocky Mountains and it flows eastward from forested slopes, through foothill rangelands and dryland and irrigated agricultural plains, to prairie grasslands (Oldman Watershed Council 2010). The lower Little Bow River study area is located on the lower section of the Little Bow River, a prairie tributary of the Oldman River. The Little Bow River, like many other Oldman River tributaries, passes through irrigation reservoirs; the portion of the channel downstream of the Travers Reservoir is known as the Lower Little Bow River.

1.7.1.2 Local Setting: The Little Bow River Watershed

North of the city of Lethbridge, Alberta, and in the northeastern portion of the Oldman River Sub-basin, the Lower Little Bow River Watershed drains an area of 55,664 ha (AAFC 2008a). The Little Bow River is a nonincised, regulated, small river (3rd order stream) that flows within a simple river channel. The channel is approximately 8 to 9 m wide and 0.5 to 1.0 m deep over bottom sediments consisting largely of coarse sand (Miller et al. 2008b).

Flows for the Lower Little Bow River are principally sourced from the Travers Reservoir (Little et al. 2003), and the river discharges into the mainstem Oldman River after traveling a total river length of approximately 65 km through lands largely used for dryland and irrigated agriculture, and native range for livestock. Water yields calculated for gauged watersheds across Alberta demonstrated water deficits for southern watersheds; values calculated for the Little Bow River area ranged from 0 to $-2.0 \cdot 10^3$ m^3 km^{-2} yr^{-1} (Kienzle and Mueller 2010).

The smaller WEBs study area within this watershed contains a 5.5 km section of the Lower Little Bow River and a 2,565 ha micro-watershed including lands used for rotational grazing of cattle, and dryland and irrigated crop production (AAFC 2008a). Sources of water for the micro-watershed include river inflow, precipitation, and irrigation return flows (i.e., runoff resulting from water applied to fields during the growing season and returned to the river mainstem through small ephemeral surface channels) (Little et al. 2003).

For the work completed as part of this research program, the study area on the Lower Little Bow River is divided into five study reaches by the location of WEBs water quality sampling stations (AAFC 2008a).

- Reach 1 (LBW3 to LB4 sampling stations) The WEBs BMP applied incorporates buffer strips to study the combined effect of vegetation type and buffer width on runoff water from irrigated fields evaluated using in-field buried runoff collectors.
- Reach 2 (LBW2 to LBW3 sampling stations). The WEBs BMP applied

incorporates conversion of annual cropland to forages on two irrigated barley fields adjacent to the river. WEBs evaluates the effect of crop conversion from barley to forage comparing water quality of barley runoff to forage runoff.

- Reach 3 (LBW4 to LBW2 sampling stations). The WEBs BMP applied incorporates off-stream watering without fencing to evaluate the effect of installing an off-stream watering system in a winter and summer pasture used by 500 head of cattle monitored through water quality in the river before and after BMP implementation.
- Reach 4, downstream of Hwy 845 (LB4-14 to LBW4 sampling stations). Cattle graze with access to the river channel and riparian areas.
- Reach 5, located at the upstream limit of the study area (LBW1 to LB4-14 sampling stations). The WEBs BMP applied incorporates an off-stream watering system with fencing along an 800 m reach to exclude cattle access from the riparian area. Fencing was installed in 2001.

1.7.2 Climate

Southern Alberta is characterized by a dry sub-humid climate regime. The Lower Little Bow River watershed is located within the Prairies Ecozone and Mixed Grass Ecoregion. Regional monthly temperature is -14^{o}C in January and +25^{o}C in July. Average annual precipitation for the watershed is 379 mm, of which approximately one-third falls as snow (AAFC 2008a). Annual precipitation during the study (2004-2007) ranged from 264 to 598 mm and was lower than the long-term average in all years except for 2005, where it was 1.5 times higher than normal (Miller 2008).

1.7.3 Landforms and Soils

Regional geology of the area consists of coarse glaciofluvial deposits, till, and glaciolacustrine silts (Shetson 1980). The Digital Elevation Model of the Lower Little Bow watershed completed by Miller et al. (2008a) shows that the watershed is composed of two flat plateau or upland areas on the north and south sides of the river, steep coulee walls on each side of the river floodplain, and an alluvial floodplain through which the river runs. Upland terrain is undulating with poorly-defined to well-defined knobs and kettles, glacially formed mounds and depressions. Surficial geological deposits are mainly of glacial till. Regional soil erosion risks have been noted. Soils found in the watershed are primarily Orthic Dark Brown Chernozems, with some Orthic Brown Chernozems and Regosols.

Mapping of hydrological facets of the WEBs Lower Little Bow River watershed completed by Miller et al. (2008a) identified major water shedding areas contributing to surface runoff were located in the valley adjacent to the river and along coulee ridges on the south side of the river valley. Most upland plateau areas presented little potential to contribute to runoff.

1.7.4 Regional Land Uses

Land use at a regional scale (i.e., within the Oldman River watershed) includes agriculture, forestry, mining, recreation, and oil and gas extraction on approximately 60% of the area land base. Grasslands of the southern and eastern portions of the Oldman River watershed are used for ranching and farming, with over half of the land cultivated (Oldman Watershed Council 2010). Land use within the Lower Little Bow River Watershed includes a variety of agricultural activities: cow-calf operations on native range, dryland farming, intensive irrigated row-crop farming, and intensive livestock operations (Little et al. 2003, Miller et al. 2010).

1.7.5 Water Management

The Travers Reservoir is an on-stream reservoir constructed between 1951 and 1954 for the Bow River Irrigation District (Charlton 1987). It divides the Little Bow River into upper (i.e., headwaters above the reservoir) and lower (i.e., river channel below the reservoir to the confluence with the Oldman River) sections. Maximum inflow occurs between March and June (Charlton 1987) with most of the water supplied to the reservoir by diversions off the Bow River through McGregor Lake and off the Highwood River into the Little Bow River (Prepas and Mitchell 1990). Relatively constant water elevations have been maintained within the reservoir from winter to summer (Beckstead 1980). Of the water released from the Travers Reservoir, only 3% flows into the Little Bow River, with the majority of water routed via canal to the Little Bow Lake Reservoir and Bow River Irrigation District. Typical operation of the reservoir includes rapid filling in spring, then fairly steady drawdown through summer (Prepas and Mitchell 1990).

Irrigation return flows enter the lower Little Bow River between early May and early October (Miller et al. 2008b). River flows were noted by Little et al. (2003) to be more variable during the summer, as they are affected by inputs of rainfall and irrigation withdrawal and return flows. Daily flow rates as reported by Miller et al. (2008b) for the Lower Little Bow River from 2004 to 2007 ranged from < 1 to 12.7 $m^3 s^{-1}$. Little et al. (2003) reported regulated flows maintain approximate steady-state flows of 0.57 $m^3 s^{-1}$ in the winter and 0.85 $m^3 s^{-1}$ in the summer. Hydrologic modeling of the watershed completed by Rahbeh et al. (2011) noted similar flow rates at upstream and downstream WEBs sites. Only approximately 10% of stream flow originated within the watershed area.

Information about water allocations for the Lower Little Bow River study area was drawn from the South Saskatchewan River Basin Water Licence Viewer (Alberta Environment 2011b). A total of 43 licences were recorded for the portion of the Lower Little Bow River watershed downstream of the Travers Reservoir, including five groundwater wells and 38 surface water withdrawals. The earliest of these licences date to 1915 (Arrowsmith Coulee, Lethbridge Northern Irrigation District), 1917 (Sorgaard Ranches Limited), and 1931 (Larry Lehto). Of the listed water uses, 24 are for agricultural use (three wells, 21 surface water), 14 are for irrigation (all surface water), and five (two wells, three surface water) are for municipal use. Total allocated annual surface withdrawal volumes for these licences is 1,460 550 $m^3 yr^{-1}$, of which irrigation use represents 91%, agricultural use is 6%, and municipal use is 3%.

All of the province's 13 irrigation districts are found within the South Saskatchewan River Basin (Alberta Environment 2011a). Approximately 20% of the cultivated land in the Oldman River Basin is irrigated (Oldman Watershed Council 2010). A moratorium on new water licenses in the Bow, Oldman, and South Saskatchewan sub-basins was enacted by the South Saskatchewan River Basin Water Management Plan (Alberta Environment 2006); this further stresses the need for water quality protection and thoughtful watershed management.

In southern Alberta, high water temperature, low dissolved oxygen, and point and non-point sources of nutrients and sediments are the primary water quality concerns for the protection of aquatic life (Cross et al. 1986, Koning et al. 2006). The Oldman watershed varies greatly, both in the status of the land and water resources and impacts from human activities. In headwater sub-basins, water quantity is adequate, quality is fair to good, and riparian ecosystems are generally considered to be healthy. However, as the Oldman

River flows eastward, water quality deteriorates, available water supplies diminish, and there are several issues of concern (Oldman Watershed Council 2010). Byrne et al. (2006), in their study of current and future waters issues in the Oldman River Basin, noted concerns regarding climate-driven decreases in flow that resulted in changes in water quality, including enhanced microbial populations and increased water-borne pathogen occurrence. The prevalence of waters artificially checked by dams was also noted as posing a potential for contaminant build-up and risks to human and animal health (Byrne et al. 2006).

Water quality concerns within the Lower Little Bow River watershed are thought to be primarily related to manure and fertilizer use, with nutrient loading and bacteria introduction resulting from agricultural activities (AAFC 2008b). Work by the Oldman River Basin Water Quality Initiative and Alberta Agriculture, Food and Development suggests that water quality within the watershed is strongly influenced by climatic variation in precipitation, with poor water conditions occurring during periods of low flow (AAFRD 2003, Little et al. 2003, Oldman Watershed Council 2010).

1.7.6 Riparian and Aquatic Ecosystems

Riparian and wetland areas are essential landscape features at the interface between the land and surface waterbodies. Plant communities found along rivers, lakes, and ponds are shaped by their proximity to these waterbodies. Access to reliable water sources support the growth of hydrophilic plant species such as cottonwood (*Populus deltoides, P. balsamifera, P. angustifolia* in southern Alberta) or willow (*Salix spp.*)along channel banks or emergent wetland species like rushes and sedges in shallow waters. Dynamic processes like annual flooding transport nutrients and create moist seedbed conditions essential for reproduction by seed of species like cottonwoods along rivers of southern Alberta (Mahoney and Rood 1998, Bigelow 2003). These connections between a waterbody and its near-water environments define a riparian area, a region of distinct growing conditions, community composition and structure.

In the prairie region of southern Alberta, increased soil water resulting from an elevated riparian water table produces unique riparian plant communities that are dramatically different from the surrounding crop and pasture land (Alberta Environment 2008). As a result, riparian zones support much higher levels of terrestrial biodiversity than the surrounding land. Estimates by Chaney et al. (1993) suggested that although riparian areas make up only 2% of the total land base, they support approximately 80% of the fish and wildlife species in all or part of their life cycle.

The Oldman River Watershed Council characterizes riparian areas of the regional Oldman River watershed as less healthy than riparian areas in Alberta as a whole, with those riparian areas found on waterbodies in the Prairie Sub-basins and the Oldman River mainstem showing the greatest degradation (Oldman Watershed Council 2010). As part of WEBs evaluation of a riparian fencing BMP on the Lower Little Bow River (Miller et al. 2010), riparian health was assessed by Alberta Sustainable Resource Development's Cows and Fish program. Prior to the installation of riparian fencing in 2001, riparian habitat on the upstream reach of the study area was given a grade of 65% (healthy but with problems).

Aquatic ecosystems within the Little Bow River watershed include small ephemeral streams and rivers, narrow riverine wetlands along channel margins, and standing waters held within lakes and ponds, both natural and artificial (i.e., dugouts and irrigation reservoirs). Inventory of aquatic biota within the Lower Little Bow River was limited to few catch records related to upstream construction monitoring and benthic invertebrate collections in progress (pers. comm. J.J. Miller). Fish community information is a key component of the evaluation of the watershed's aquatic ecosystem health (Karr 1993, Alberta Environment 2008, Blann et al. 2009).

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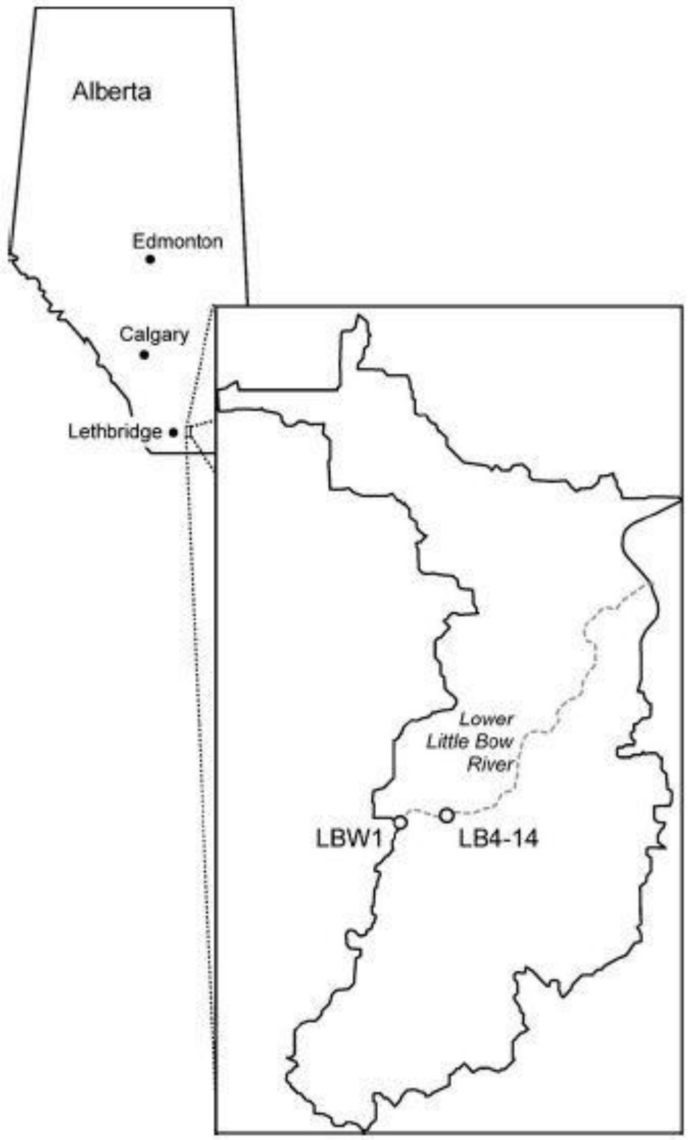

Figure 1-1. Regional Project Setting: Lower Little Bow River Watershed, Southern Alberta (Miller et al. 2010).

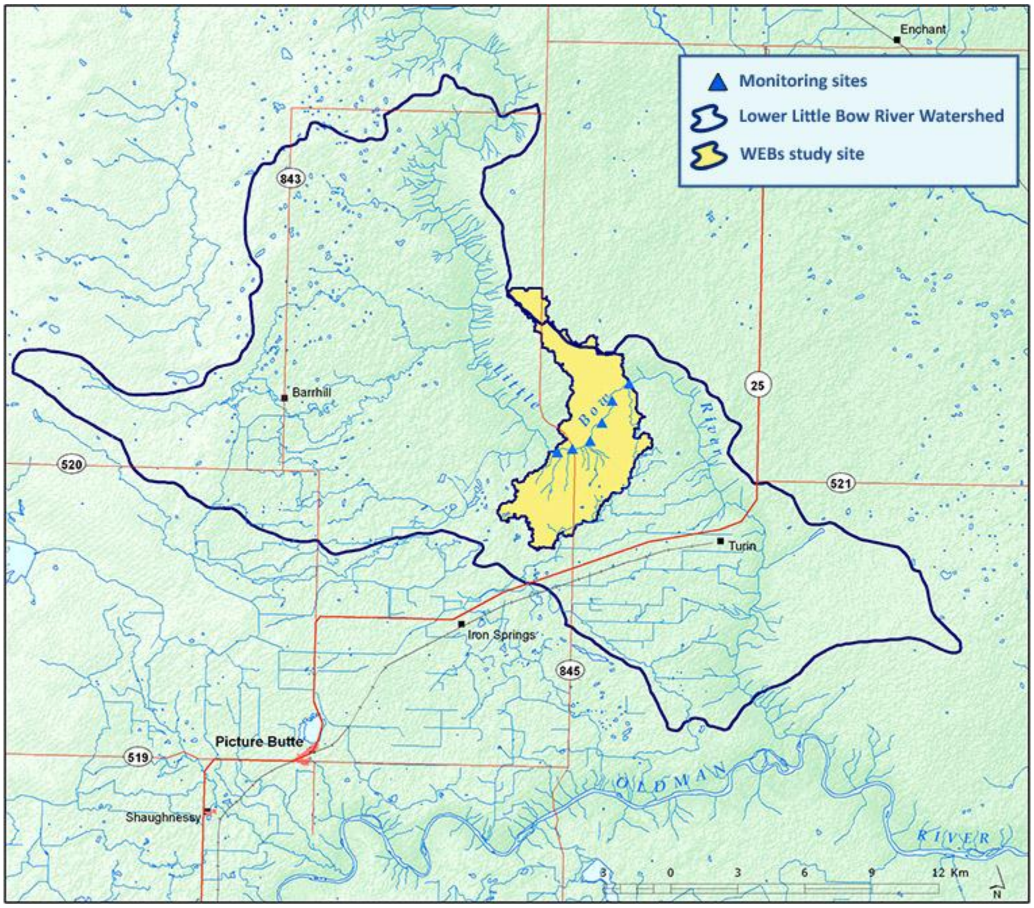

Figure 1-2. Project Study Area: Lower Little Bow River, Alberta (AAFC 2008a).

Figure 1-3. Typical Landscape of the Lower Little Bow River, October 2009.

2 USING WATER QUALITY INDICES IN SITE MONITORING OF AGRICULTURAL BMPS ON THE LOWER LITTLE BOW RIVER, ALBERTA

2.1 Introduction

Water quality studies are important to help us understand the state of a waterbody or watershed with respect to its health and ability to support our various uses. Water quality degradation is seen as an indicator of environmental impact and a trigger for the need for changes to land management practices. Scientifically derived environmental indicators bring value to decision makers by providing information in a variety of forms to support different perspectives on these environmental management choices and questions (Cimorelli and Stahl 2005). One of the forms used to present indicator information is an aggregated indicator – one number or score that simplifies the information. Such a tool is also referred to as an index, a single number derived from information provided by more than one environmental indicator (Ott 1978).

The use of environmental indices, including those for water quality, has become a common approach to enable the distillation of significant messages out of complex and sometimes cumbersome data sets. These indices characterize water quality by considering a spectrum of water quality parameters rather than one parameter alone. In this way, they integrate land and water information and support environmental effects monitoring within complex aquatic systems. Water quality indices (WQI) are in use both locally and internationally; however, spatial scale of application has largely focused on broad regional comparisons of water quality condition or ranking. Indices have been less commonly applied to monitor water quality at site- or farm-scales.

2.1.1 Water Quality Monitoring – Structure and Design

Water quality studies are designed to match the scale at which management questions are being asked. These scales, both temporal and spatial, vary and the studies may, as examples of some of the types of questions:

- compare watersheds at a national or regional scale;
- compare watercourses within a watershed, perhaps on a regional scale, subject to differing land uses and management regimes; or
- compare changes in water quality at local or site scale.

A watershed's hydrologic behaviour varies throughout the year (e.g., changes in contributing water sources and resulting discharge with snowmelt, spring freshet, precipitation events, peak water use and withdrawal). Samples drawn from a river then represent water quality at the moment of sampling. To reduce the variation across samples collected through time, river monitoring programs may stratify the river seasonally – effectively identifying 'sub-populations' of the river's water (Sanders et al. 1983, Spooner et al. 1985, Ward and Loftis 1986).

To identify water quality characteristics that are of concern for water users, many monitoring programs observe how water quality differs from defined objective values, either established water quality guidelines or site-specific objectives (CCME 2001). Analysis of water quality data may involve:

- comparisons of instantaneous measured values relative to criteria or objectives for concentrations, using percent exceedance or some measure of exceedance frequency;
- comparisons of total maximum daily load or dose measures for sampled parameters to water quality criteria or objectives for loads; and
- linkages to qualitative water quality rankings using some type of index method which combines the above comparisons

Guidelines such as the Canadian Water Quality Guidelines for the Protection of Aquatic Life address the need for ecological protection. These are based on toxicity data for the most sensitive species of plants and animals found in Canadian waters and act as "science-based benchmarks for the protection of 100% of the aquatic life species in Canada, 100% of the time" (Environment Canada 2004). Guidelines for human health include the Canadian Drinking Water Quality Guidelines developed by Health Canada and these address more than 85 physical, chemical, and biological attributes of water quality. These guidelines establish maximum acceptable concentrations (MAC) for substances found in water used for drinking and apply to all public and private drinking water supplies and to treated or finished water as it emerges from the tap.

Human health protection is also the driver for the Canadian Recreational Water Quality Guidelines. These guidelines for recreational activities like swimming, sailing, and fishing were developed by Health Canada to address potential health hazards of infections transmitted by disease-causing micro-organisms, and aesthetics and nuisance conditions. Guidelines for agricultural waters use include the Canadian Water Quality Guidelines for the Protection of Agricultural Water Uses, which is based on maximum irrigation rates and the sensitivity of crops to pollutants, and Canadian Water Quality Guidelines for Livestock Water, which is based on the potential for bioaccumulation of substances like toxic chemicals in livestock as a result of drinking water quality (Environment Canada 2004).

Established national, standardized guidelines do not exist for all water quality parameters, for example, those parameters that are not considered critical factors for water uses (e.g., electrical conductivity) or those parameters that exhibit a wide range of natural variation depending on waterbody and watershed characteristics (e.g., temperature). For some variables like these, other regional, provincial, or North American criteria exist. Guideline values are present for many of the measured variables within the WEBs dataset, with the exception of most of the measured or derived nutrient fractions.

2.1.2 Index Development

Water quality indices were initially suggested by Horton (1965) who proposed the WQI as a means to provide the public with information enabling the comparison of water quality changes through time and among locations. In designing his index, he sought to limit the number of variables, use available data, and choose variables of significance to most parts of the United States. Variables were weighted to indicate the variable's influence in degrading water quality. Dissolved oxygen, pH, and a sewage treatment factor were given the highest weighting to show that their role in water quality was considered fundamental, while variables like alkalinity and specific conductance were weighted lower. Ranges of possible values were assigned to each variable and then ratings were scaled using breakpoints. Two dichotomous variables were also used: temperature and "obvious pollution", with values simply assigned as acceptable or unacceptable (Ott 1978).

In 1970, the National Sanitation Foundation developed a Water Quality Index through a Delphi technique that relied on polled results from a large group of water experts. Experts were asked which of 35 starting parameters should be included in an index and were to rank these in significance (on a scale from 1 to 5) as a contributor to water quality. The results were consolidated and returned to the experts with requests to review and revise if desired, in an effort to obtain greater consistency across the participants' ratings.

Variables of greatest importance to this expert panel were: dissolved oxygen, fecal coliform, pH, 5-day biochemical oxygen demand, nitrates, phosphates, temperature, turbidity, total solids, and group variables of toxic substances and pesticides. A further questionnaire asked experts to develop rating curves for each variable; responses were averaged. Use of empirical curves created sub-indices that were implicit non-linear functions (Ott 1978).

The further development of an objective, easily applied, nonparametric statistical procedure for combining a number of water quality parameters in a water quality index included efforts of Harkins (1974), Landwehr et al. (1974) and Inhaber (1975), among others. Smith (1990) suggested improvements to water quality indexing in New Zealand through the use of a minimum operator scoring function for rating water quality. Cude (2001) described the Oregon Water Quality Index (OWQI) to provide a simple and concise expression of ambient stream water quality for general recreational use, including fishing and swimming. The OWQI is a single number that expresses water quality by integrating measurements of eight water quality variables (temperature, dissolved oxygen, biochemical oxygen demand, pH, ammonia+nitrate nitrogen, total phosphorus, total solids, and fecal coliform).

In 2006, Paparov et al. reviewed existing water quality indices and identified correspondence between approaches to index calculation and definitions of general water quality deterioration:

- a WQI calculated as the arithmetic average corresponds to a management strategy where general WQ deterioration is indicated by substantial deterioration of more than half of the separate sub-indices forming the WQIs;
- a WQI estimated using a minimum operator (Smith1990) corresponds to a management strategy for which general WQ deterioration would be indicated by deterioration of any single WQI sub-index – that with the lowest score, or the "minimum operator";
- a WQI estimated as the weighted average corresponds to a management strategy for which general WQ deterioration would be indicated by a combination of individual WQIs and the severity of their deterioration (e.g., the lower or ''worse'' the rating value, the higher its relative weight).

Alberta Agriculture, Food and Rural Development evaluated the function of water quality indices for agriculture and supported the creation of an Alberta Agricultural Water Quality Index (AAWQI) (Wright et al. 1999). This, and the development of other provincial water quality indices through the 1990s, led to development of a national Canadian WQI by the Canadian Council of Ministers of the Environment (CCME 2001). The CWQI was designed as a tool to provide consistent procedures for Canadian jurisdictions to report water quality information to management and the public on a national scale, and is the dominant water quality index in use within Canada (CCME 2011). It is customizable, in that it can be calculated on a set of parameters selected by the user. The CWQI is based on three attributes of water quality information as it relates to violation of water quality objectives during a given time period of interest:

- scope (how many?): the number of water quality variables that do not meet objectives in at least one sample, relative to the total number of variables measured (e.g., tests comparing data to criteria resulted in failure for half of the variables included in the index)
- frequency (how often?): the number of individual measurements that do not meet objectives, relative to the total number of measurements made in all samples (e.g., tests comparing data to criteria resulted in failure for 10% of all measurements); and
- amplitude (how much?): the amount by which measurements which do not meet the objectives depart from those objectives (e.g., data failed criteria by three times the criteria value).

These attributes are combined to form a unitless number, scaled from 0 to 100, with a higher number associated with better water quality. Score number ranges are assigned to water quality rankings which fall within one of five categories: excellent, good, fair, marginal, and poor (deRosemond et al. 1995, CCME 2001).

All indices require a somewhat subjective assignment of relative importance or weight to each of the parameters informing the index. At the most basic level, an index assigns value or weight to a parameter by including it within the index. The Alberta River Water Quality Index (ARWQI) summarizes physical, chemical, and biological data from Alberta’s rivers into a simple descriptor of water quality for each site (Alberta Environment 2008). The ARWQI is based on the unweighted average of four sub-indices calculated annually: metals (may include up to 22 variables measured quarterly);

nutrients (6 variables measured monthly); bacteria (2 variables measured monthly); and pesticides (17 variables measured 4 times during open-water season). Variables in the first three groups are compared to Alberta and federal water quality guidelines while pesticide variables are evaluated when they fall within the laboratory detection limits. The formula used to calculate the individual sub-indices is the same as that used for the CWQI. However, the sub-indices are aggregated using an unweighted average, unlike the aggregation method used by the CWQI (Alberta Environment 2008). The Oldman River Basin Water Quality Index (ORBWQI), a regional index employed in southern Alberta, uses fewer variables than the province-wide measure (no metals are included). It consists of a general index and a separate pesticide index (Alberta Environment 2008). Reported ORBWQI index values for sites within the Lower Little Bow River from 1999 to 2002 range from poor to excellent, with water quality showing a strong relationship with droughts and rainfall events and corresponding flow volumes (AAFRD 2003, Little et al. 2003). All of these indices share similarities, yet differ to some degree in their internal weighting mechanisms and in the parameters they evaluate.

2.1.3 Alternative Approaches

Water quality indices represent one approach to consolidating information about water quality and water use in a watershed system, existing on a middle ground between reductionist and holistic philosophies of watershed science. Other more systems-based approaches involve the integration of information about watershed physical, chemical, and biological processes through modeling. Simplified structural equations may be assembled to identify assumed linkages between stream water quality and environmental conditions. These describe the anticipated pathways of effect of BMPs and identify key parameters that may represent useful indicators of agricultural impacts to water quality, which include indicators of erosion, sedimentation, and transportation of particulate-bound nutrients (e.g., total suspended solids), indicators of nutrient enrichment (e.g., total nitrogen, total phosphorus), and indicators of livestock-introduced pathogenic microorganism (e.g, *Escherichia coli*).

On the reductionist end of the spectrum, monitoring programs may track individual parameters as a means to evaluate change in water quality. This has been the approach to WEBs Lower Little Bow River monitoring to date (Miller et al. 2009, 2010). Work by Little et al. (2003) and by the ORBWQI and AAFRD (2003) also indicated that physical parameters like temperature and flow, and regional climatic variables are likely to be critical variables affecting water quality within the Lower Little Bow River.

Problems of interpretation are encountered when positive change in one season is followed by negative change in another season, or when an improvement in one parameter is paired with a decline in another. Karlen et al. (1994) stated that reductionist approaches, while important for dividing scientific problems into discrete and manageable pieces, may result in information that appears inconsistent, and occasionally conflicting, when combined to address complex agricultural problems.

Comparisons of percentage change, from upstream to downstream sites, rely on several assumptions; namely, that the site treatment acts at the scale of the distance between sites. One of the problems regarding the use of percent change measures is imposed by pollutant loading from upstream. Consider two streams. In one, water enters the upstream site with an already total suspended solids value of 1000 mg L^{-1}. In the other, water enters the upstream site with half the total suspended solids, a value of 500 mg L^{-1}. If both sites receive treatments that reduce sediment loading to the streams equally and reduce total suspended solids by 100 mg L^{-1}, the first site shows a percentage change of 100 mg L^{-1} /1000 mg L^{-1} or 10% while the second site shows a percentage change of 100 mg L^{-1} / 500 mg L^{-1} or 20%. This difference in effect is not a result of the treatment effectiveness, but rather an effect of the pollutant level prior to treatment. If percent change is used to describe treatment effect for sites located sequentially on one river, as each reach's upstream pollutant levels cascade to lower values as a result of the cumulative upstream treatments, the relative effect of each treatment may artificially appear to increase.

Component information from reductionist approaches should be ultimately integrated into holistic solutions to monitoring questions (Karlen et al. 1994). Aggregated indices consider water quality information for management of environmental risk at the level of broader groups (nutrients, sediment, biological activity) and this can be helpful for managers communicating with decision-makers and the public. Using nationally uniform aggregated indices does pose challenges, which include: addressing geographical variation in water quality because of natural conditions; addressing differing water uses and water suitability; and recognizing that constituents outside of the index may impact water quality and yet not be captured in monitoring with a WQI (Ott 1978). The ability of indices to capture small changes, such as criteria exceedances in only one parameter or minor exceedances in multiple parameters, also remains a concern (Gartner Lee 2006) as well as an area of research interest, given a growing range of environmental monitoring questions to which water quality indices are being applied.

2.2 Research Objectives

The local spatial scale of the WEBs study, a 5.5 km length of river within a 2,565 ha microwatershed, directed our general research questions. Agricultural BMPs have been implemented throughout the study area, and describing water quality of the study area as a whole, and within river reaches at scales associated with BMP treatments, was a key objective of this work. Further objectives were developed to evaluate the water quality index method and its application to the WEBs Lower Little Bow River study area, namely:

- How does the frequency of calculation, for seasonal or annual periods, affect generated index values?
- How does the division of water quality parameters into biological, physical, and chemical parameter sub-groups affect generated index values?
- When multiple parameters are combined into a water quality index, do the index outputs for assessing water quality differ from those provided by physical, chemical, and biological, sub-indices or individual water quality values?

2.3 Materials and Methods

2.3.1 WEBs Study

The Lower Little Bow River watershed is located north of the city of Lethbridge, in the northeastern portion of the Oldman River Sub-basin, in southern Alberta, Canada. The region is semi-arid and lies within the Mixedgrass Ecoregion of the Prairie Ecozone. The Lower Little Bow River watershed drains an area of 55,664 ha (AAFC 2008) and a micro-watershed of 2,565 ha forms Agriculture and Agri-Food Canada's WEBs study area. The Lower Little Bow River is a non-incised, regulated small river (3rd order stream) that flows within a simple river channel from 8 to 9 m wide and 0.5 to 1.0 m deep over bottom sediments consisting largely of coarse sand (Miller et al. 2008). Sources of water for the micro-watershed include river inflow, precipitation, and irrigation return flows (runoff resulting from water applied to fields during the growing season and returned to the river mainstem through small ephemeral surface channels) (Little et al.

2003). Lands within the watershed are used for rotational grazing of cattle, and dryland and irrigated crop production (AAFC 2008).

The WEBs water quality dataset used for this research program consists of results for four years of water quality sampling completed by Agriculture and Agri-Food Canada's WEBs Lower Little Bow River study between May 2004 and December 2007. The WEBs dataset included those parameters that describe nutrient, bacteriological, and sediment characteristics. A total of 760 discrete samples were collected over 188 weeks. Grab samples of river water were collected weekly (for chemical analyses) and every two weeks (for bacterial analyses) from April until October, and then monthly during the winter. Monitoring stations were located on the river mainstem and are associated with stilling wells installed by Alberta Agriculture and Rural Development as part of their watershed monitoring since 1999 (AAFC 2008).

Water quality sampling included field measurement of specific conductance (electrical conductivity (EC) with an automatic calibration to 25°C), dissolved oxygen (DO), temperature, and pH using a portable multimeter (MultiLine P4, Wissenschaftlich-Technische, Werkstätten, Germany). Turbidity, a measure of light transmission related to suspended solids present in the water column, was measured in the field using a portable turbidity meter (Hach Model 2100p, Loveland, CO). Water samples in 2006 and 2007 were analysed for chlorophyll a, a measure of primary productivity, using the fluorometric method (APHA 1998) with a portable fluorometer (Aquafluor, Turner Designs, Sunnyvale, CA).

Laboratory analysis of collected water samples determined elemental concentrations (mg L^{-1}) of nitrogen and phosphorus in various fractions. Nitrogen fractions included: nitrogen in ammonia (NH_3-N), nitrogen in nitrate (NO_3-N), nitrogen in nitrite (NO_2-N), total Kjeldahl nitrogen (TKN), total nitrogen (TN), total dissolved nitrogen (TDN), and total particulate nitrogen (TPN). Phosphorus fractions included: phosphorus in phosphate (PO_4-P), total phosphorus (TP), total dissolved phosphorus (TDP), and total particulate phosphorus (TPP). Collected samples were also submitted for laboratory analysis of total suspended solids (TSS). Biweekly, throughout April to October, water samples were submitted for analysis of pathogenic bacteria including counts of Colony Forming Units per 100 ml sample volume of fecal coliforms (FC) and *Escherichia coli* (Ecoli). Flow (cms, or $m^3 s^{-1}$) was calculated using stage-discharge relationships established by Alberta

Agriculture and Rural Development, which was also responsible for the installation of the stilling wells used as the WEBs water quality sampling stations. Detailed WEBs sampling methods are discussed in Miller et al. (2008) and summarized in Tables 2-1 and A2-1.

2.3.2 Experimental Approach and Design

The research program included water within a 5.5 km section of the Lower Little Bow River. Data extended from spring and summer 2004 through to the end of 2007, and included paired upstream and downstream water samples collected on five river reaches (Table 2-1). Multiple variables, including physical, chemical, and biological constituents were measured for each discrete sample. Sampled locations were not changed through the study period. Most parameters were measured or analysed throughout the study period; however, chlorophyll a was added to the program in 2006 and so was measured over two years rather than the complete study duration. Reach water quality was described as the difference between paired upstream and downstream sites.

2.3.3 Dataset Preparation and Exploratory Analysis

A preliminary step for any water quality analysis involves a review of the dataset for potential errors including errors in data entry and errors in precision (e.g., number of significant figures used in data table and their relationship to detection limits and equipment/method sensitivity). Potential errors were identified using manual review of the dataset and descriptive statistical methods applied with the R statistical software package (R Development Core Team 2011). Dataset formatting was also standardized with dataset importing requirements of R software and the CCME Water Quality Index generator (CCME 2011), and checks were completed for errors in text. A new start row and start date for the data set was inserted and the date assigned to Sample # 20040196 was corrected to May to match samples with adjacent sample identification numbers. Columns were reordered to place a sample ID column as the first in the data table.

Physical limits exist for some parameters in surface water bodies and comparison against these plausible ranges served as one means for checking for data error. Samples were ranked by temperature and those values outside of the range of -4°C to +30°C were reviewed for potential error. Scatterplots were also graphed for two parameters with known relationships (e.g., dissolved oxygen and temperature; turbidity and total suspended solids) and points were labeled with sample ID numbers (Appended Figure

A2-1.). Those samples lying outside of the main point cluster were further reviewed to examine the potential for data entry error. In one case, nine sequential samples contained values for dissolved oxygen that were ten times greater than surrounding values. The clear repetition of the pattern indicated an error in placement of the decimal during data entry. Sampling realities such as the program's daytime collection of samples also provided a tool to check for errors. Samples were ranked by sample time on a 24 hour clock, and those collected in 2007 were found to show an error with sampling times recorded in overnight periods.

Negative values appeared within the dataset as a result of the methods used to derive values for some parameters. For some of the sample years, total Kjeldahl nitrogen (TKN) and total nitrogen (TN) resulted not from direct field measurement or analytical evaluation but instead were derived from other measured parameters. A change in analytical methods for TKN and TN occurred between 2006 and 2007 and is described in Miller et al. (2008). Values for TKN were recorded in the data table for samples from the years 2004 and 2005, while a formula was used in 2006 and 2007 to derive TKN from the difference between measured total nitrogen and the sum of measured nitrate-nitrogen and nitrite nitrogen (TN – [(NO_3-N) + (NO_2-N)]). This was reversed with TN values recorded for 2006 and 2007 samples and a formula used to calculate TN (TKN + NO_3-N + NO_2-N) in 2004 and 2005.

Methods of measurement impose limits on the precision of a measured value and the precision of reported values enabled another point of review. Samples contained within the dataset exhibited a range of precision, for example, 2004 entries for chloride (mmol cL^{-1}) were reported to anywhere from one to thirteen significant figures. Where the number of significant figures of reported values varied for a parameter, values were rounded to the limits of precision of the particular analytical method or measurement device used.

Because of the potential importance of maximum values as indicators for water quality concerns, care was needed in the treatment of extreme values or outliers. Maximum values may convey critical information for the determination of water quality conditions, particularly when data are used to generate index ratings using a measure of the number of exceedances of a given standard. Little et al. (2003) noted that within the Lower Little Bow River watershed, maximum water chemistry values generally had stronger correlations with soil type and land use classes than median values. For this work, potential outliers were evaluated for indications of data entry errors; where none were evident, extreme values were retained.

2.3.4 Reduction of the Dataset

Before bringing the data into index calculation, correlation analysis was used to examine strong relationships between pairs of parameters, as bringing only one of two strongly related parameters into the calculation of an index in order can reduce colinearity between parameters that enter in the index calculation. A problem may arise in some cases when water quality variables may appear independent yet are actually duplicating information already provided by other variables within the index. Relationships may be created when a value for one variable is derived from the analysis of other variables; for example, total nitrogen values may be analyzed independently or derived as a sum of total Kjeldahl nitrogen + nitrate-nitrogen + nitrite-nitrogen. This type of correlation may result in a "closed number system", where variables within an index are not independent and the information provided into the index is reduced (Myatt 2007).

The following relationships were known to exist between derived variables:

- Total N = TKN (unfiltered) + NO2-N + NO3-N;
- Total Dissolved N = TKN (filtered) + NO2-N + NO3-N;
- Total Particulate N = TN - TDN; and
- Total Particulate P = Total P - Total Dissolved P.

In 2004 and 2005, TN was calculated based on TKN, NO2-N, and NO3-N; therefore, Total N is not independent of these parameters in these years (in effect, this group of N-fraction variables is a closed number system, and correlation may be overstated as the variables are not independently measured). Similarly, in all years from 2004 to 2007, Total Particulate N is related to TN and TDN since it is derived from those values (not independently measured), as TPP is related to TP and TDP. In 2006-2007, Total N (unfiltered) and Total Dissolved N (filtered) were measured directly, rather than being calculated indirectly, and TKN was derived from TN, NO2-N and NO3-N. Total Particulate N was still derived indirectly, as was Total Particulate P.

Multiplot matrices were also constructed using R to examine relationships between other water quality variables included in the WEBs program. Strong linear correlations were identified using *pairscor.fnc* in R (Baayen 2008), which presents both Pearson's correlation (r) and Spearman Product Moment correlation (r_s) coefficients calculated for pair-wise comparison of variables in combination with a scatterplot, lowess (i.e., locally weighted scatterplot smoothing) line, and histogram (appended Figures A2-3, A2-4). Pearson correlation is used to evaluate linear associations in normally distributed data while Spearman correlation is used for non-normal distributions, is not constrained to linear relationship, and is calculated using observation rank. The value of the correlation coefficient is a number between -1 and +1. If the two compared parameters increase together, the correlation is positive. The correlation is negative if one parameter increases while the other decreases. The closer the correlation value is to 1 or -1, the stronger the relationship between the two parameters. Both methods of correlation analysis were used given the apparent non-normality of some of the water quality variable distributions and the tied ranks anticipated for variables that showed repeated values near or at the analytical detection limit.

This correlation analysis was completed as part of the initial exploration of the dataset, and so greater reliance was placed on the shape of the supporting scatterplot in evaluating correlation significance than on reported p values for correlations generated by the R graphical function. Evaluated histograms presented skewed distributions as a result of the extreme values found in some variables as well as what appeared to be censoring of nutrient data to analytical limits. Non-normality was apparent by visual inspection, with some variables presenting distributions that approximated log-normal or Weibull distributions. Non-normal distributions are commonly observed in water quality measures, particularly in those variables which are counted (e.g., bacterial colony forming units); such distributions require the data to be transformed to normality to enable the use of parametric tests otherwise researchers must employ nonparametric statistical methods (Chapman 1996). As the index method required the inclusion of multiple variables, transformations were not employed.

Strong positive correlations were noted for total suspended solids and turbidity ($r=0.86$, $r_s=0.85$); between fecal coliform and *E.coli* colony counts ($r=0.96$, $r_s=0.99$); and between electrical conductivity and chloride concentration ($r=0.85$, $r_s=0.91$). The concentration of total suspended solids showed positive correlation with both fecal coliform and *E.coli* ($r=0.56$, $r_s=0.56$). Correlation coefficients calculated for nitrogen fractions TKN, TN, TDN, and TPN suggested positive correlations between these nutrient parameters. TKN, TN, and TPN showed strong positive correlations with each other, with slightly weaker correlations with TDN. Calculated correlation coefficients also suggested strong correlations between concentrations of phosphate-phosphorus and total dissolved phosphorus and between total phosphorus and total particulate phosphorus; however, the apparent censoring of phosphorus concentrations to 0.1 mg L^{-1} in total phosphorus and total particulate phosphorus in the WEBs dataset presented challenges to further correlation analysis. As a result, the total number of variables brought forward to the calculation of indices was reduced to 11 of 22 variables measured by WEBs.

2.3.5 Partitioning the Dataset

Following work by Miller et al. (2010), annual median differences in measures for each variable were calculated for each reach (i.e., upstream/downstream pair of sites) to represent information provided by the dataset before the calculation of index scores. Samples were examined for seasonality using visual inspection of plotted water quality measures over time, as annual hydrologic patterns are a common source of variability within water quality datasets (Sanders et al. 1983, Ward and Loftis 1989).

Seasonal median differences were also calculated, following the seasonal stratification of the WEBs dataset into four periods, based on differences in WEBs sampling frequency (i.e., weekly sampling in summer vs. monthly sampling in winter) and important water use periods within the watershed (i.e., in irrigation season). These seasonal periods were:

- Spring (approximately Apr. 15-June 15),
- Summer (approximately June 16-Sept. 15),
- Fall (approximately Sept. 16-Dec. 15), and
- Winter (approximately Dec. 16-Apr. 14).

Partitioning the data by season results in a loss in the number of degrees of freedom (effective sample number), which decreases the sensitivity of subsequent statistical tests (Spooner et al. 1985). However, such partitioning has the potential to capture watershed processes and resulting changes in water quality that may be driven by activities within the watershed that are associated with a particular season, for example, irrigation associated with the summer sampling period, and therefore to produce more meaningful information for responsive management of land uses for water quality protection. Partitioning by season is also a means to remove natural sources of variaibility in water quality due to seasonal and stream discharge variations and this may improve the detection and estimation of trends (Hirsh et al. 1991).

2.3.6 Criteria Selection

Criteria values used in the calculation of exceedances are found in Table 2-4 and were derived from national CCME surface water quality guidelines for aquatic life as well as provincial surface water quality guidelines. For TSS and turbidity criteria values, observed annual and seasonal distributions of TSS and turbidity within the 2004-2007 dataset were used to define natural background levels for the study area.

2.3.7 Index Generation and Water Quality Scoring

Once the dataset evaluation was completed, water quality observations were used to calculate water quality index scores. The CCME Water Quality Index uses the form:

$$WQI = 100 - \left(\frac{\sqrt{F_1^{\,2} + F_2^{\,2} + F_3^{\,2}}}{1.732} \right) \tag{2-1}$$

Where:

F_1 represents the percentage of variables that depart from their objectives at least once, relative to the total number of variables measured:

$$F_1 = \left(\frac{\text{Number of failed variables}}{\text{Total number of variables}} \right) \times 100 \tag{2-2}$$

F_2 represents the percentage of failed individual tests:

$$F_2 = \left(\frac{\text{Number of failed tests}}{\text{Total number of tests}} \right) \times 100 \tag{2-3}$$

F_3 is an asymptotic capping function that scales the normalized sum of the excursions from objectives (*nse*) to yield a range between 0 and 100:

$$F_3 = \left(\frac{nse}{0.01nse + 0.01} \right) \tag{2-4}$$

The collective amount by which individual tests are out of compliance is calculated by summing the departures of individual tests from their objectives and dividing by the total number of tests (both those meeting objectives and those departing from objectives). Departures are equivalent to the number of times by which a concentration is greater than (or less than) the objective. The *nse* variable is expressed as the sum of each test's departure from the criteria relative to the number of tests:

$$nse = \frac{\sum_{i=1}^{n} departure_i}{\#\ of\ tests} \qquad (2\text{-}5)$$

The final water quality index score is then finally calculated using the following formula:

$$\text{CWQI} = 100 - \left(\frac{\sqrt{F_1^2 + F_2^2 + F_3^2}}{1.732} \right) \qquad (2\text{-}6)$$

The factor 1.732 is derived from the greatest possible score achievable by the three composite index factors F_1, F_2, and F_3 (square root of the sum of squares of the maximum scores) and scales the score from 0 to 100, with zero representing the poorest water quality and 100 indicating excellent water quality. Scores are related to qualitative ratings of poor to excellent, typically organized into five groups (Rocchini and Swain 1995, CCME 2001). The groups contain increasingly narrow score ranges as they move from 'poor', with a wide score range of 0 to 44.9, to 'excellent', with a score range of only 95 to 100 (Table 2-2).

Water quality scores were calculated with this index format for each site and for each of the five study reaches using CCME's WQI generator, a macro-supported Excel spreadsheet using Visual Basic (CCME 2011). The WQI generator enables the selection of parameters of interest and the definition of criteria against which a sample is to be judged, and so it is customizable to individual monitoring programs.

Alterations were made to the WQI generator to evaluate two scenarios, each representing a modification to the standard index that relates to our research questions:

Scenario 1: Seasonal vs. Annual Index Scoring

- Water Quality Index scores were calculated for each site, using an annual time frame (resulting in four annual scores calculated for 2004 to 2007) and using a seasonal time frame (resulting in four seasonal scores calculated for each of these years: winter, spring, summer and fall as delineated above).

Scenario 2: Sub-index Scoring vs. Total Scoring

- Three sub-index scores were calculated for each site at an annual scale.

- These sub-indices were developed using select WEBs-measured variables, categorized into biological, physical, and chemical groups, and included:
  - **BI:** biological index of variables (indicators of biological activity: dissolved oxygen, bacterial counts);
  - **PI:** physical index of variables (sediment-related parameters: turbidity, TSS); and
  - **CI:** chemical index of variables (EC, ammonium, nitrate, nitrite, total dissolved phosphorus (orthophosphate), total phosphate, chloride).

Reach scores were based on the difference between upstream and downstream scores at the sites bounding the reach.

2.4 Statistical Analyses

Water quality information generated in this work for the Lower Little Bow River study area included:

- annual and seasonal median values for water quality variables;
- annual and seasonal water quality scores using the CWQI; and
- total and sub-index annual water quality scores.

Outside of any indexing structure, annual and seasonal descriptive statistics for each water quality variable were generated for the WEBs study area as a whole, as an initial representation of water quality variability within the watershed. The median was selected as a better measure of central tendency than the mean, given its robustness against the presence of extreme values (Helsel and Hirsch 1992). The robust equivalent of the standard deviation, the median absolute deviation, MAD, was used as a measure of spread. MAD, like the standard deviation, is a measure of average deviation from the central value which is, in this case, the median rather than the mean. Interquartile range (IQR) was also calculated as another measure of the spread of data about the measure of central tendency. Variability, which has been seen to translate through CWQI calculations by affecting the third term of the index (Statistics Canada 2007), was used to rank parameters.

The effect of index method was evaluated by comparing the reliability of site scores produced by annual and seasonal index methods. A two way mixed model intraclass correlation (ICC) was used to measure inter-rater reliability (Shrout and Fleiss 1979) for the annual, seasonal, and sub-index index methods applied to each site in this work. ICC is a measure of the ratio of between-groups variance to total variance, similar to ANOVA, and produces a coefficient which approaches 1.0 when the between-groups effect (site) is very large relative to the within-groups effect (index). In this way, ICC is a measure of homogeneity: it approaches 1.0 when any given row (site) tends to have the same values for all columns (index). ICC was used here to describe consistency (and concurrently, inconsistency) in the scoring of yearly water quality at WEBs sites by the various index methods developed. Water quality indices presenting a high ICC value with an accepted level of confidence suggest that the indices evaluated site water quality in a similar way, that they could be interchanged, and that the effort needed to calculate seasonal or sub-index scores instead of simply an annual WQI may not be warranted. Values were interpreted as follows: >0.90 = high agreement of scores; 0.75 to 0.90 = moderate agreement of scores; <0.75 = poor agreement of scores (Portney and Watkins 2000). Similarities between annual and seasonal scores was also evaluated using Spearman rank correlation, with rho calculated for paired annual and spring, annual and summer, annual and fall, and annual and winter scores.

Differences in WQI scores by reach were evaluated by comparing score differences between paired upstream and downstream sites bounding each reach, using the Wilcoxon Signed-Rank Test on paired annual and seasonal scores. An accepted Type I error rate of 5%, $\alpha=0.05$, was used in determining statistical significance of any difference. Miller et al. (2010) utilized differences in mean concentration and load to compare water quality at upstream and downstream sites on the Lower Little Bow River as part of WEBs evaluation of agricultural beneficial management practices. The use of a comparison of differences was identified by Spooner et al. (1985) in their discussion of appropriate designs for documenting water quality improvements from agricultural BMPs.

Potential trends in WQI scores were also evaluated at each site using tests for monotonic (one direction) trend. Although Berryman et al. (1988) recommended the use of $n>9$ (e.g., 9 sampling periods) for monotonic trends analysis with the Kendall test, trends in scores for each site were evaluated (annual WQI and sub-indices, $n=4$, seasonal WQI, $n=16$) for monotonic trends using the Mann-Kendall and Seasonal Kendall trend tests in R. As recommended by Ward et al. (1990), a p value of 0.10 was used to assess the statistical significance of results of tests for trend.

2.5 Results and Discussion

One of the challenges faced by those who attempt to study the local effectiveness of small scale environmental modifications is that any one environmental factor may not show a change large enough to confidently assign it as significantly different from the background variation naturally observed. Because of this, change in individual parameters alone may not be enough of a signal to demonstrate an effect. Of interest was whether aggregating the change in time (across a year versus seasons) or in multiple parameters (a total index versus sub-indices) would capture the cumulative effect of small, beneficial changes in individual parameters or simply result in the loss of individual signals of positive and negative changes.

2.5.1 Individual Water Quality Variables

Variability observed in the dataset across the study area as a whole is described in the annual and seasonal median concentrations (or unit measures) (Tables 2-5 and 2-6) with MAD and IQR presented as measures of spread. The coefficient of MAD (e.g., MAD/median expressed as a percent similar to the coefficient of variability calculated for standard deviations about a mean) calculated for annual and seasonal periods is presented in Figures 2-1 and 2-2 as an expression of variability. These figures present a sorted order of variables, with the largest bars indicating those variables with the greatest variability. Variables that evidenced a constant or near constant value through all samples are represented by a very small or absent bar.

At the annual scale, MAD was greatest for fecal coliform and *E.coli* counts, and this variability appeared relatively constant across years. Higher variability in annual medians was also noted in total particulate nitrogen, in 2004, 2005, and 2007, while lower variability was noted in ammonium, nitrate, nitrite, phosphate, and total dissolved phosphorus measures. A collection of variables presented their highest variability in 2005: flow, TSS, Turbidity, nitrate, total nitrogen, total dissolved nitrogen, phosphate, total phosphorus, and total particulate phosphorus. Such variability has been seen to translate through CWQI calculations by affecting the third term of the index (Statistics Canada 2007), and the range in concentrations measured throughout a year further supports the partitioning of the dataset into seasonal periods.

2.5.2 Scenario 1: Water Quality Index Using Season Stratification

In the first scenario developed with the Water Quality Index, both annual and seasonal WQI scores were generated for each site/reach (Table 2-7, Figure 2-3).

2.5.2.1 Annual Scores

All sites produced annual scores that were close in value and considered “poor”, “marginal” and “fair”. The minimum annual score (44.9) within the four years evaluated was observed in 2004 at site LB4 (the downstream limit of the WEBs study area), as was the maximum annual score (69.3), observed in 2004 in the upper half of the watershed at site LBW4. Interestingly, WQI scores in 2004 presented the greatest range in scores, spanning a 24 point range, while scores in all other years differed only from 1-5 points (Figure 2-4).

Reach-based information was generated by a comparison of upstream and downstream sites at each reach, with the difference between site water quality observations characterizing the reach water quality. In work by Miller et al (2010), differences were based on concentrations and loads; here, differences were examined in WQI scores. Positive differences were produced by an increased score downstream, relative to upstream. No reaches demonstrated uniform increase or decrease of water quality scores through 2004-2007 (Figure 2-5).

- In 2004, annual score was improved in two of five reaches: Reach 2 and Reach 4.
- In 2005 and 2006, annual score increase was noted in all reaches with the exception of Reach 1, at the downstream limit of the study area.
- In 2007, this effect was reversed and annual score improvement was noted in Reach 1, as well as in Reaches 4 and 5.

Differences in score were small, with most representing a change of less than 5% of the upstream score. As a result of this similarity, annual reach scores based on a simple average of upstream and downstream site scores, produced ratings that all fell within the “marginal” category (scores between 45.0 and 64.9). Consistent reach score change was not seen at the annual scale across the four years of the dataset and no significant score trends with time were observed in annual WQI scores at any site.

WQI scores for 2004 showed a marked difference from the patterns of scores in other years. The construction of the WQI Index Calculator enabled a review of composite score components; in this case, it supported a review of contributing factors to the high annual score observed in 2004 at LBW4 and the relatively low scores at the other sites in 2004. Samples that were collected in the 2004 period were not uniformly distributed throughout the year and across all sample sites. This was apparent in the calculation of seasonal scores. Sampling began at different start dates from site to site, with sampling started at LBW4 later in the year than at other sites. This variable sampling start date produced the unusually high annual score in 2004 at LBW4 because the index score at this site was generated from only 7 samples, compared to 27 samples used to generated seasonal scores at LB4, LB4-14, and LBW1; and 12 samples used to generated seasonal scores at LBW3 and LBW2. One benefit of the use of seasonal scores for the WEBs WQI score calculation was the easier review of potential issues with annual index analysis such as unbalanced seasonal sampling inputs.

2.5.2.2 Seasonal Scores

The index was not able to produce scores for all seasons in all years as samples in spring and summer periods were collected from some sites in 2004 but sampling was not completed uniformly across all sites until fall 2004. Seasonal scores were produced for all sites for fall 2004 onward, with the exception of winter 2005 which lacked samples to support seasonal score evaluation.

Sites produced seasonal scores from “poor” to “good” – a greater range in quality rating than observed in annual scores. The maximum seasonal score (83.7) within the four years evaluated was produced in spring 2005 at site LB4-14, the downstream limit of the fenced riparian reach evaluated by Miller et al. (2010). The minimum score (35.8) was produced in spring 2004 at the same site. Winter scores ranged from 58.8 to 79.3 (mean=73.2), spring scores ranged from 35.8 to 83.7 (mean=66.3), summer scores ranged from 41.9 to 68.3 (mean=56.9), and fall scores ranged from 49.3 to 74.0 (mean= 64.9).

Reach responses (i.e., the difference in seasonal scores between upstream and downstream sites) were, for the most part, less than 5 points (Figure 2-5):

- Winter scores differences were only generated for winter in 2006 and 2007. Score reach differences in winter were larger than for other seasons (-16.8 to +17.1). Reach 2 and 4 showed winter score increases in 2006 and 2007, while Reach 5 showed score declines in both years. Reach 3 scores increased in 2006 but declined in 2007. Reach 1 scores showed the opposite response.

- Spring scores in 2005 and 2006 showed reach declines in Reach 1, 3 and 4; in 2007, scores increased in Reaches 3 and 4.
- Summer score differences were largely positive in 2004 and 2005, then negative for all reaches except Reach 3 in 2006, and small in 2007 (-0.8 to +0.5).
- Fall score differences were also small from 2005 to 2006, but greater increases were observed in Reach 5 and Reach 1 in Fall 2007.

No seasonal reach scores showed consistent increase or decrease across the four years of the dataset, and no significant score trends with time were observed in seasonal WQI scores at any site.

2.5.2.3 Annual versus Seasonal Scores

Annual and seasonal scores for each of the Lower Little Bow study sites are contrasted in Figure 2-6 and Table 2-7. Differences between the annual and seasonal index scoring methods were evaluated using intraclass correlation coefficients (ICCs) and 95% confidence intervals (CIs) calculated across index methods. Results are presented in Table 2-8. Comparing site scores by year across all 5 indices (one annual and four seasonal) required the omission of years where the WQI score was not available for all seasons. In effect, this reduced the number of comparable WQI scores by half (i.e., subjects=12). Comparing scores across all index methods, ICC was low and negative: -0.137 ($p=0.95$) with a 95%-confidence interval for ICC population values between $-0.198 < ICC < 0.04$. This negative ICC indicated more score variation was present within sites than between indexes, and the ICC also did not support the rejection of the null hypothesis of no correlation, given the wide confidence intervals and p-values >0.05.

When seasonal scores for winter were excluded and ICC computed across 4 indices, the number of subjects included increased to 21 because fewer periods contained missing scores for one or more index. ICC remained negative: -0.0826 ($p = 0.816$) with a 95%-confidence interval between $-0.191 < ICC < 0.12$, and again the ICC did not support the rejection of the null hypothesis (i.e., no correlation), given the wide confidence intervals and p-values >0.05. When both winter and spring were excluded and ICC was calculated across the remaining three indices, 23 subjects were retained. ICC was 0.658 ($p = 4.08e-08$) with a 95%-confidence interval between $0.443 < ICC < 0.822$. This ICC, while positive, still fell below 0.75 (as did the lower bound of the 95% confidence interval) and this suggests a less than unified agreement. It does, however, support rejection of the null hypothesis of ‘no consistency’ and suggests some consistency exists in the water quality scores produced by annual, summer, and fall index methods.

Similarities between paired annual and seasonal scores were also supported, although not strongly, by correlation analysis (Table 2-9). Consistency between annual and summer index scores in the Lower Little Bow River was anticipated, given the strong signal into both indexes provided by summer exceedances of biological criteria, but consistency in produced scores was not found across all seasonal indices. It was more apparent when winter and then both winter and spring indices were excluded in the ICC analysis. ICC value can increase by increasing within-group homogeneity or by increasing between group differences. In this case, the exclusion of winter and spring periods removed groups that contained a wider score range (i.e., lower within-group homogeneity) (Figures 2-6 and 2-7). This suggests some discriminatory value is available from the use of seasonal indices over an annual index alone, particularly for programs monitoring to discriminate a local effect.

Rank analysis of paired upstream and downstream seasonal scores for each reach from 2004-2007 using the Wilcoxon signed-rank test (Table 2-10) supported the rejection of the null hypothesis (i.e., the median difference observed between annual and seasonal upstream and downstream site scores was not greater than would be expected by chance, given an accepted Type I error rate of α=0.05) for Reaches 2 and 4. However, the difference from zero expressed for both of these reaches was small, with the pseudomedian estimated for Reach 2 being only -0.48. Seasonal score for Reaches 1, 3, and 5 were not significantly different from zero.

2.5.3 Scenario 2: Water Quality Index Using Sub-indices

In the second scenario developed with the Water Quality Index, three annual sub-index WQI scores were generated for each site/reach to test for difference between the information provided by a composite WQI and the information provided by looking at smaller sub-index groupings of parameters. Sub-index and annual (i.e., total) WQI scores are presented in Table 2-11 and Figure 2-8 to 2-10.

2.5.3.1 Total Index Scores

Total index scores were the same as the annual index scores calculated for Scenario 1.

2.5.3.2 Sub-index Scores

Biological sub-index (BI) scores produced using a subset of the WEBs data consisting of dissolved oxygen concentrations, and counts of fecal coliform and *E.coli* colony forming units per 100ml (CFU $100ml^{-1}$), ranged from 23.5 to 77.6 (mean=31.9). The maximum score was produced in 2004 at LBW4. The minimum score was produced at the upstream end of the study area in LBW1 and LB4-14 in 2004. Lower BI scores were produced by exceedances in dissolved oxygen, fecal coliform, and *E. coli* variables.

Physical sub-index (PI) scores produced using turbidity and total suspended solids concentration ranged from 34.5 to 42.2 (mean=40.1). The maximum score was produced in 2007 and was found almost uniformly across all sites (i.e., all sites scored 42.1 to 42.2). The minimum score was produced at the upstream end of the study area in LBW1 in 2004. Low PI scores were produced by exceedances in both TSS and turbidity in Spring, and in turbidity alone in Fall.

Chemical sub-index (CI) scores produced by comparing against criteria values measured concentrations of ammonia, nitrate, nitrite, chloride, total dissolved phosphorus, and total phosphorus; and recorded values for electrical conductivity (EC), ranged from 45.3 to 61.8 (mean=55.3). The maximum score was produced in 2004 at LBW4, while the minimum score was noted in 2004 at LB4.

The PI consisted of two variables (TSS and turbidity) and the BI consisted of three variables (dissolved oxygen, fecal coliform, and *E. coli* variables), while the CI represented seven variables. BI and PI scores fell below CI scores in all cases but for one BI score at LBW4 in 2004. Because the BI and PI relied on fewer variables, each variable had greater bearing on the resulting score. CI scores were less affected by exceedances in one or two variables within the CI sub-index, despite the near uniform exceedances in one or more variables apparent in almost all samples. This resulted from nitrite values that were listed within the dataset as 0.1 mg L^{-1} (99% of all values within the dataset were 0.1 mg L^{-1}) – an apparent result of data censored to the analytical detection limit which happened to exceed the criteria value.

Reach score responses (i.e., the difference in subindex scores between upstream and downstream sites) were, for the most part small with only 14% of score differences exceeding ±5 points (Figure 2-9):

- Biological index scores differences were greatest in 2004, with the maximum score increase in Reach 4 followed by the maximum score decline in Reach 3. BI score differences were small in the remaining reaches over the 2004-2007 period, with alternation between score increase and decrease noted in all but Reach 1, which presented consistent decreases in score.
- Physical index score difference was greatest in 2004 at Reach 3. Reach differences in PI scores in 2004 and 2005 were positive in Reaches 5, 3, and 2 while Reach 1 showed decreasing PI scores. In 2006 and 2007, PI score differences were less than 2 points.
- Chemical index score differences were also greatest in 2004, with reaches alternating between score increase and decrease and tracking a similar pattern to BI scores. CI scores increased across Reach 4 from 2004-2007, and Reach 5 from 2005-2007. Aside from a score decrease at Reach 1 in 2004, CI scores at Reaches 1, 2, and 3 appeared relatively stable.

No sub-index reach scores showed consistent increase or decrease across the four years of the dataset, and no significant score trends with time were observed in sub-index WQI.

2.5.3.3 Sub-index Scores vs. Total WQI Score

Total and sub-index scores for each of the Lower Little Bow study sites are contrasted in Figure 2-10 and Table 2-11. Concordance or agreement of sub-indices with the standard WQI was evaluated using ICCs calculated across index methods. Results are presented in Table 2-12. Comparing scores across all sub-index methods, ICC was low (ICC = 0.338), suggesting poor score agreement (ICC<0.75). Poor agreement was also noted between BI, PI, and CI scores (ICC = 0.132), although lower confidence was expressed in the ICC value as a result of wider confidence intervals that included zero and p-value >0.05. Moderate agreement was seen between the total and CI scores (ICC =0.832), with data supporting the rejection of the null hypothesis of no correlation between index methods, given p=1.25e-07.

Similarities between paired total and CI sub-index scores were also supported by correlation analysis (Table 2-13), with both CI and BI sub-index scores strongly correlated with total WQI score. Consistency between total and chemical sub-index scores in the Lower Little Bow River was anticipated, given the proportion of variables within the total index represented in the CI. Summer exceedances of biological criteria were also expected to have a strong influence on total scores.

Statistics Canada (2007) demonstrated that WQI is sensitive to the number of parameters that enter in its calculation for a given number of samples. The larger the number of parameters, the lower the intensity of extreme categories ("poor" and "excellent") in comparison with the "marginal" and "fair" categories. Using sub-indices to supplement an aggregating index addresses several concerns regarding index use: namely, that the value or weight of key indicators in a suite of water quality variables may be reduced when they are combined with a number of other variables, and that specific information about pollutants or processes helpful in guiding water management might be lost. With WEBs data, sub-index scores further distinguish those factors or categories of variables responsible for degradation of system water quality.

Rank analysis of paired upstream and downstream sub-index scores for each reach over the 2004-2007 period using the Wilcoxon signed-rank test (Table 2-14) supported the rejection of the null hypothesis (i.e., the median difference observed between annual and seasonal upstream and downstream site scores was not greater than would be expected by chance, given an accepted Type I error rate of α=0.05) for Reach 1 (p value=0.015, pseudomedian=1.72) and Reach 2 (p value=0.05, pseudomedian=-0.40), although the confidence intervals for Reach 2 are close to zero and present weak evidence of a difference. Sub-index score for Reaches 3, 4, and 5 were not significantly different from zero.

2.5.4 Trend detection in WQI scores

Analysis of trends present in WQI scores using a seasonal Kendall test (MannKendall (RE Development Team 2011) found no significant score trends with time in annual, seasonal or sub-index scores at any site (Table A2-2). Monitoring within a program limited to a single watershed/single treatment observation relies on the ability of the monitoring tool to detect change occurring within the system throughout time. Limitations to this work include the relatively short duration of the dataset. Datasets used to support water quality trend analysis, particularly in impact assessments or observational studies of non-point source pollution, may extend over decades.

For the period including this work, flow within the Oldman River basin was identified as under stress by the Annual Alberta River Flow Quantity Index's long term (10 year) assessment of river flow. In 2004, summer (May-Sept) flow of the Oldman River was diminished outside of its normal range. In 2005 and 2006, flow was within its normal natural range, and in 2007, summer flow was again much below its normal natural range (Alberta Environment 2009). Several studies note the influence of climate on regional water quality within the Oldman River basin and suggest a linkage between higher flows and poorer water quality ratings. In 1995 - 96, a high flow year, Alberta Environment's River Water Quality Index ranked the Oldman River upstream of Lethbridge as marginal, while in a drier period of 2001- 02, the site was ranked as excellent. Drier conditions in the region appear to lead to less runoff and lower potential for the movement of contaminants from land to river. In a wet year or during a runoff event, conditions may return to fair or marginal conditions, as was seen in the 2005-06 Index rating, which is largely related to significant summer rainfall events (AESRD 2012).

As the 2004-2007 period evaluated here includes normal and below normal summer flows, the index responses evaluated may not adequately reflect the range of behaviours of the Lower Little Bow River watershed (i.e., untested with higher summer flows). This is important as several pathways of effect for water quality pollutants depend on precipitation – runoff generation. Trend detection in water quality scoring may be improved with the addition of further sampling periods through time (i.e., with the addition of data from the WEBs 2008-2012 sampling period).

2.6 Conclusions

Within the Lower Little Bow River study area from 2004 to 2007 overall annual water quality varied from poor to fair, while seasonal water quality varied from poor to good. Total water quality, including all parameters, was marginal, as was water quality considering chemical factors. Water quality considering physical factors and biological factors was poor.

In the case of the Lower Little Bow River WEBs study area, comparing annual and seasonal index methods, some level of agreement was evidenced between annual and summer water quality index scores. Summer criteria exceedances in fecal coliform, *E.coli*, dissolved oxygen, and total suspended solids provided a strong signal that was reflected in both Summer and Annual water quality index scores. Differences among annual and spring and winter scores suggests that water quality in these periods is not reflected as well as summer and fall seasons by annual WQI scores. Sub-index methods produced biological and physical index scores that were dissimilar to the scores produced by traditionally used total WQI. Both seasonal and sub-index variations of an aggregating water quality index (CCME WQI) produced a wider range of site scores than the standard annual/total forms of the index.

Many other uses of the CWQI rely on a comparison of data collected on a monthly (n=12) or quarterly (n=4) basis; when WQI is used with samples collected at a greater frequency (i.e., weekly or biweekly vs. quarterly) the index itself can be better tuned to criteria changes that should naturally shift with both natural background variations (TSS, turbidity) and seasonal biotic needs (DO). Such use requires programming improvements to the WQI calculator, to support seasonal functions within the calculator and allow variation of DO criteria with biotic needs and operations that compare measured values to previously observed values for identifying exceedances of change-based criteria

Despite small score differences between index methods, reach water quality scores (score differences between paired upstream and downstream sites) within the Lower Little Bow River WEBs watershed did not support the finding of any trends of significance across the period 2004 to 2007. Detailed examination of the collected dataset for errors prior to analysis also suggested that analytical methods which censor to a detection limit matching a criteria value used in index calculation should be refined to produce more useful information for change detection and future monitoring before data are brought into an index.

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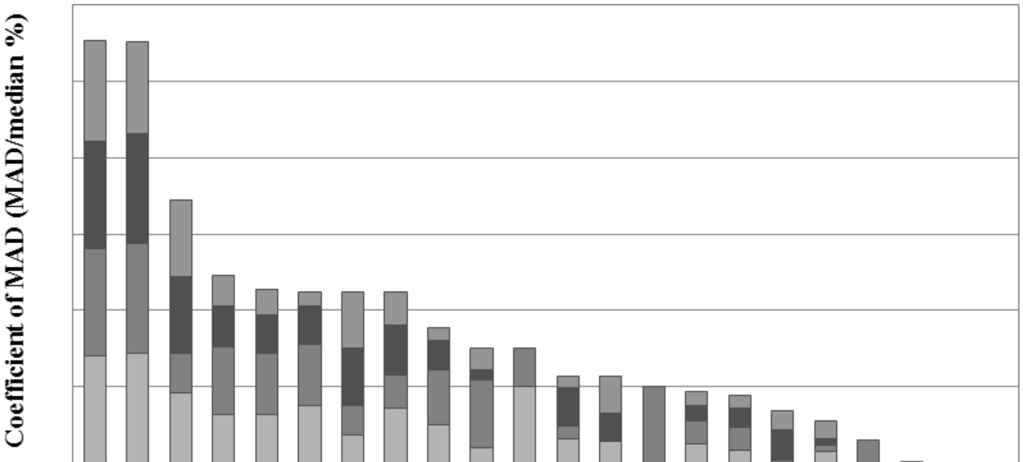
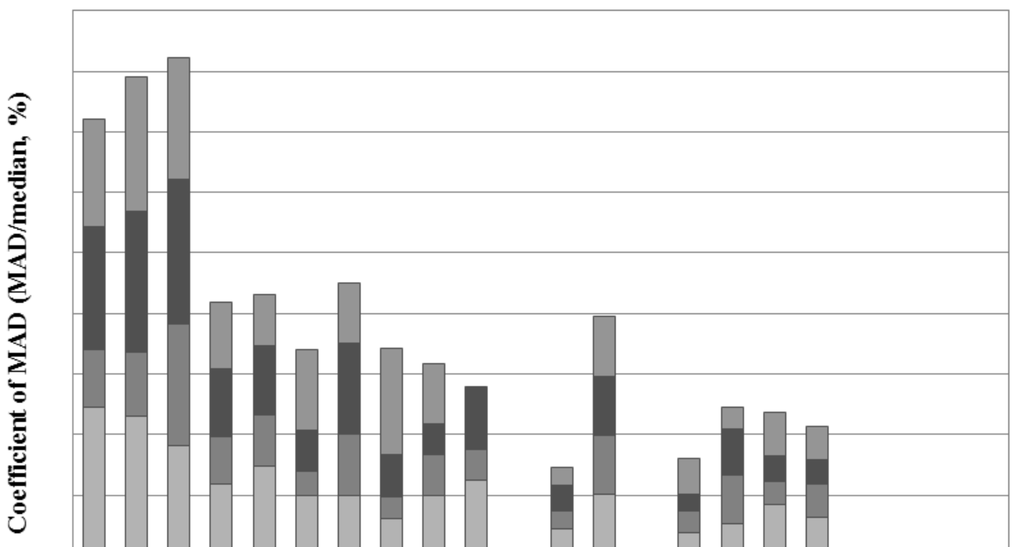


	Fcoli	Ecoli	TPN	TSS	Turb	TPP	TKN	Temp	TP	Flow	TDP	DO	ClVol	PO4.P	TDN	TN	Chla	ECfield	NO3.N	pH	NH3	NO2.N
2007	132	120	100	41	32	20	75	43	17	29	0	15	49	0	19	17	24	23	0	0	0	0
2006	141	144	100	53	50	50	75	66	40	13	0	51	37	0	20	24	41	9	0	0	0	0
2005	140	143	52	89	81	80	38	43	71	89	50	16	0	100	30	30	4	9	30	2	0	0
2004	141	145	92	63	63	75	38	71	50	19	100	32	29		25	17		15	0	0	0	0

Figure 2-1. Annual variability in median values for Lower Little Bow River Watershed Evaluation of Beneficial Management Practices (WEBs) water quality variables, as represented by the median absolute deviation (MAD) coefficient of variability, 2004-2007.



	Fcoli	Ecoli	TPN	TSS	Turb	TPP	TKN	Temp	TP	Flow	TDP	DO	ClVol	PO4.P	TDN	TN	Chla	EC	NO3.N	pH	NH3	NO2.N
Winter	89	111	100	55	42	67	50	87	50	0	0	14	49	0	29	18	36	27	0	0	0	0
Fall	102	116	120	56	58	33	75	36	25	52	0	21	49	0	14	38	21	20	0	1	0	0
Summer	48	54	100	39	42	20	50	18	33	25	0	15	49	0	19	40	19	27	0	0	0	0
Spring	122	115	91	59	74	50	50	31	50	63	0	22	50	0	18	26	42	32	0	2	0	0

Figure 2-2. Seasonal variability in median values for Lower Little Bow River WEBs water quality variables, as represented by the MAD coeffiecient of variability, from 2004-2007.

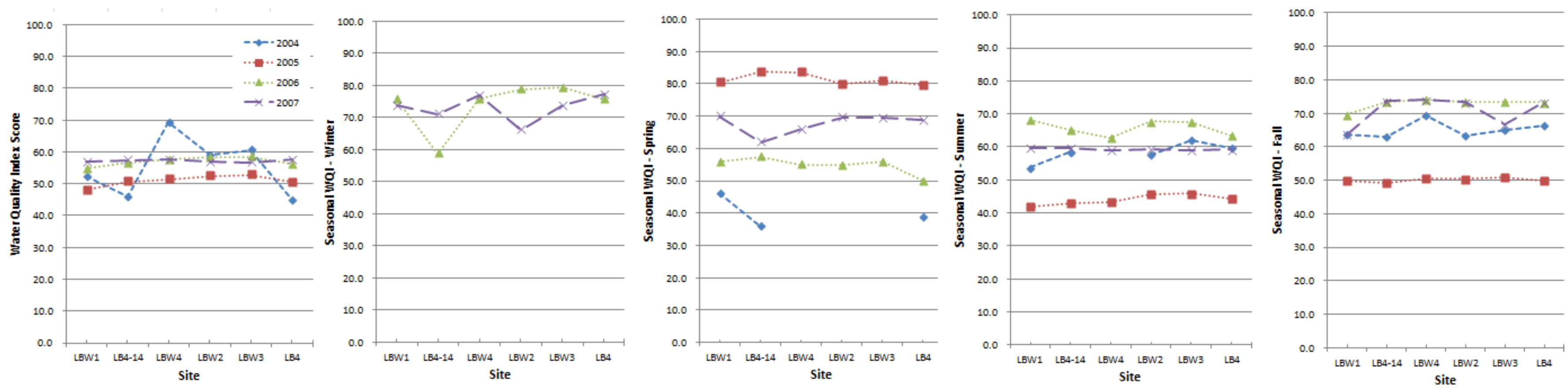


Figure 2-3. Water Quality Index (WQI) scores for Watershed Evaluation of Beneficial Management Practices (WEBs) sites on the Lower Little Bow River, for annual and seasonal timeframes from 2004-2007.

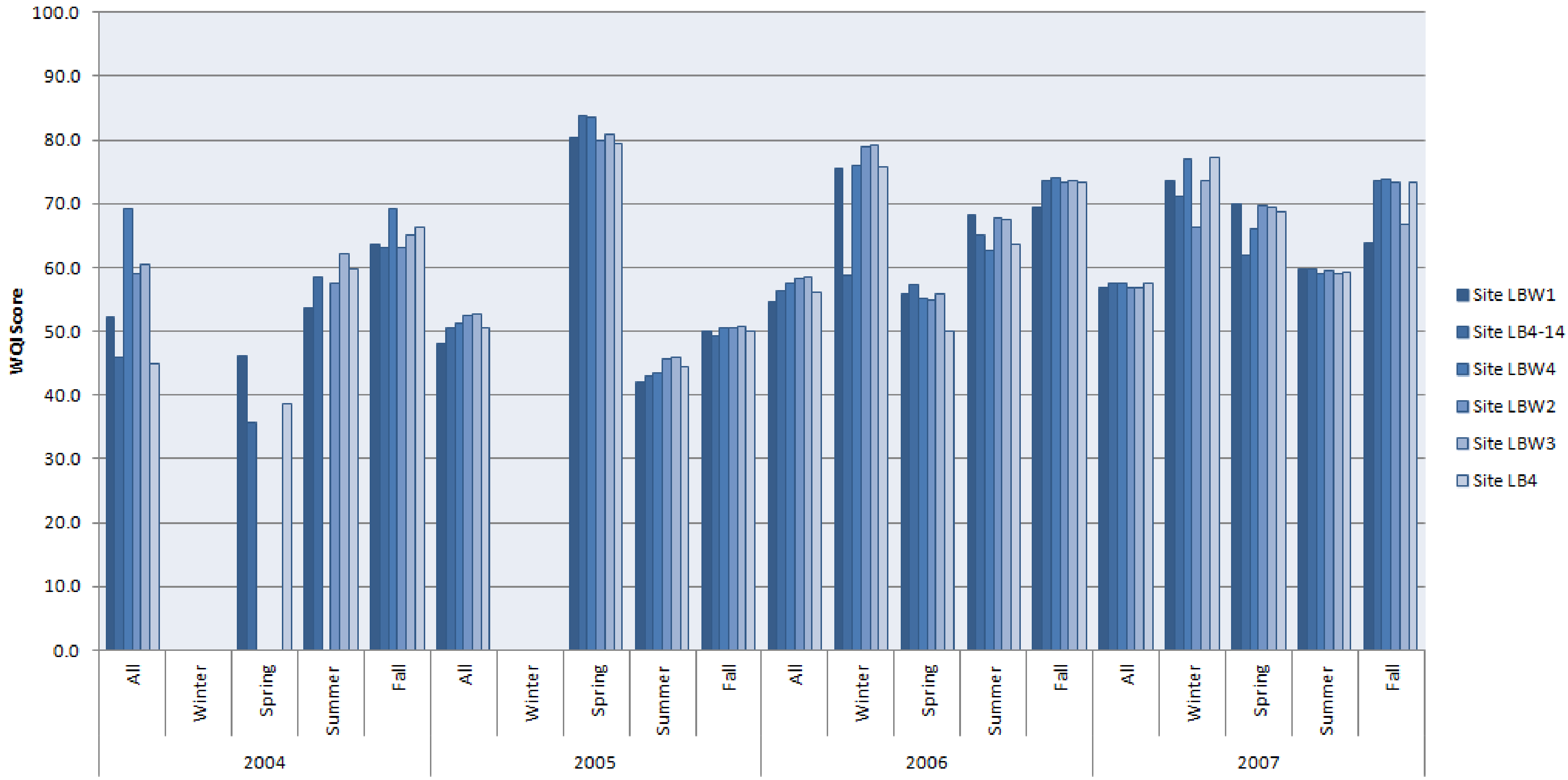


Figure 2-4. Water Quality Index (WQI) scores for Watershed Evaluation of Beneficial Management Practices (WEBs) sites on the Lower Little Bow River, for annual and seasonal timeframes from 2004-2007.

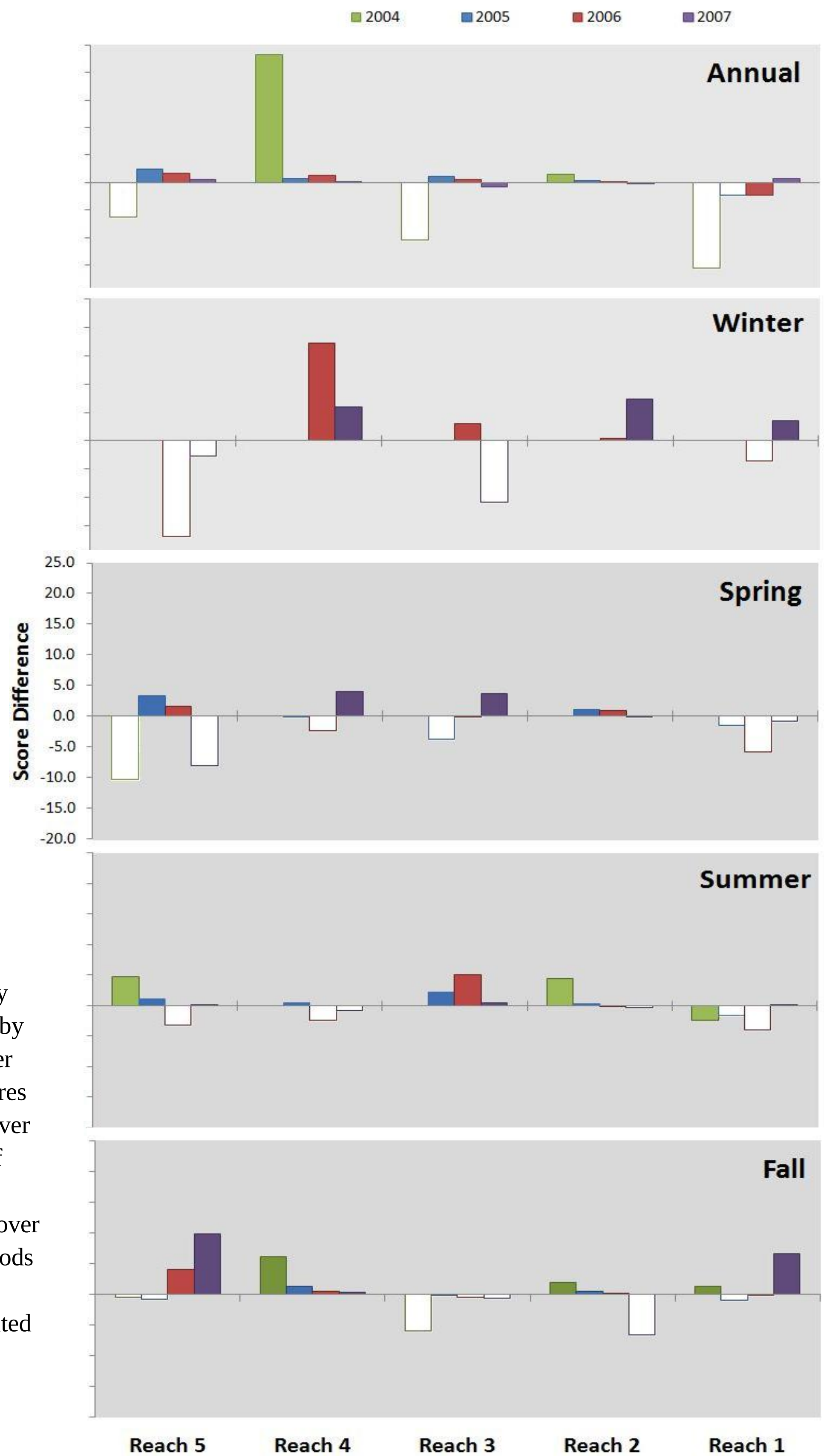


Figure 2-5. Water quality change as demonstrated by reach differences in water quality index (WQI) scores for Lower Little Bow River Watershed Evaluation of Beneficial Management Practices (WEBs) sites, over annual and seasonal periods from 2004-2007. Score improvement is represented by a solid bar and score decline by a hollow bar.

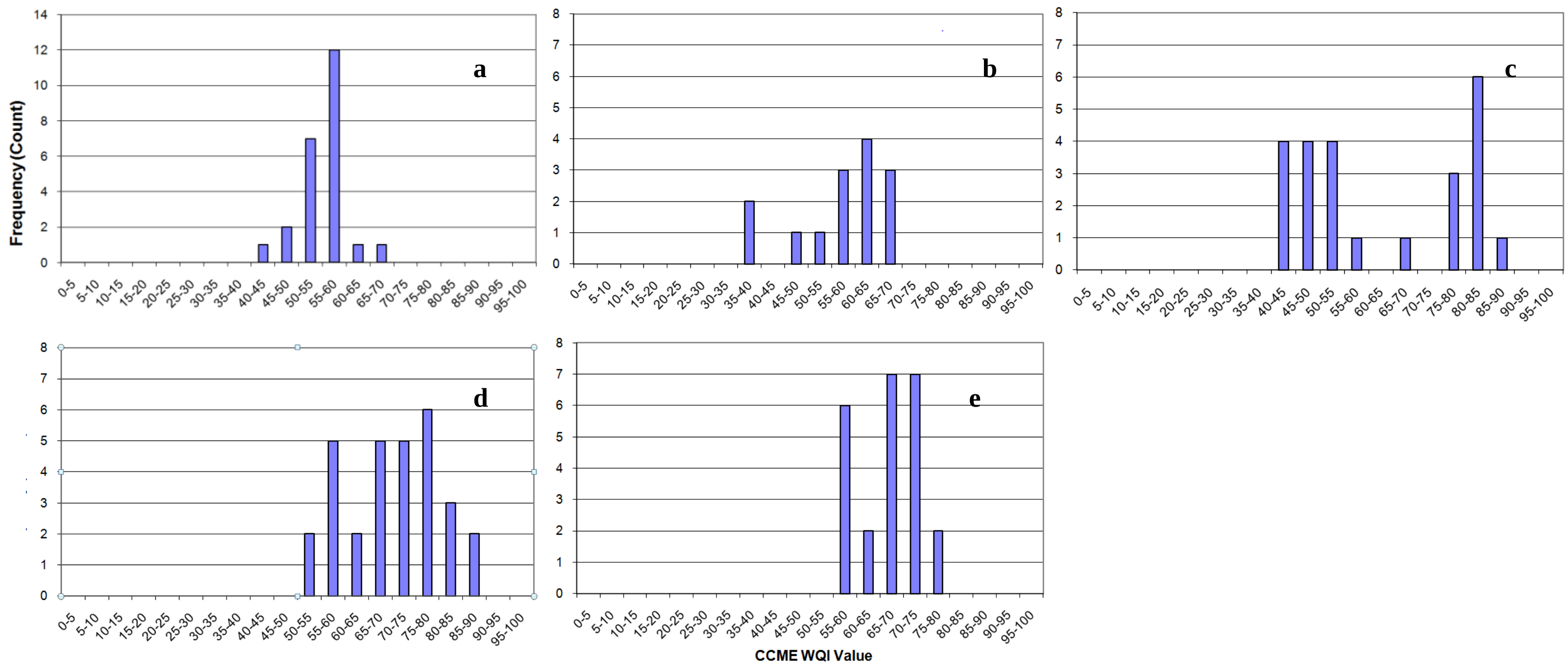


Figure 2-6. Distribution of Water Quality Index (WQI) scores calculated for Watershed Evaluation of Beneficial Management Practices (WEBs) Lower Little Bow River sites (2004-2007), with annual WQI scores over all years (a), and seasonal WQI scores for 2004 (b), 2005 (c), 2006 (d), and 2007 (e).

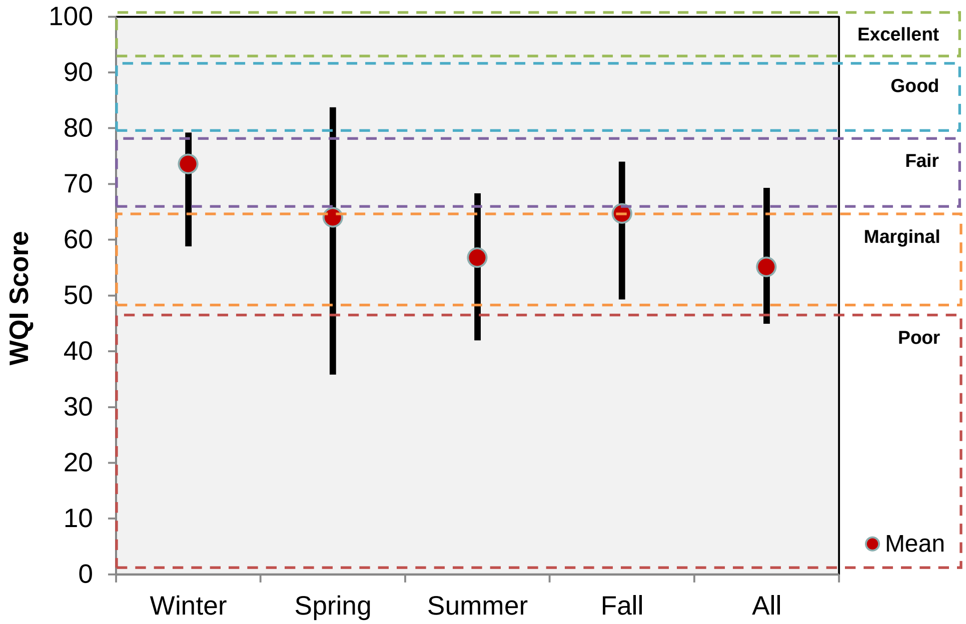


Figure 2-7. Ranges of Lower Little Bow River seasonal and annual water quality index scores against Canadian Water Quality Index (CWQI) qualitative ratings, with mean score over the 2004-2007 sampling periods.

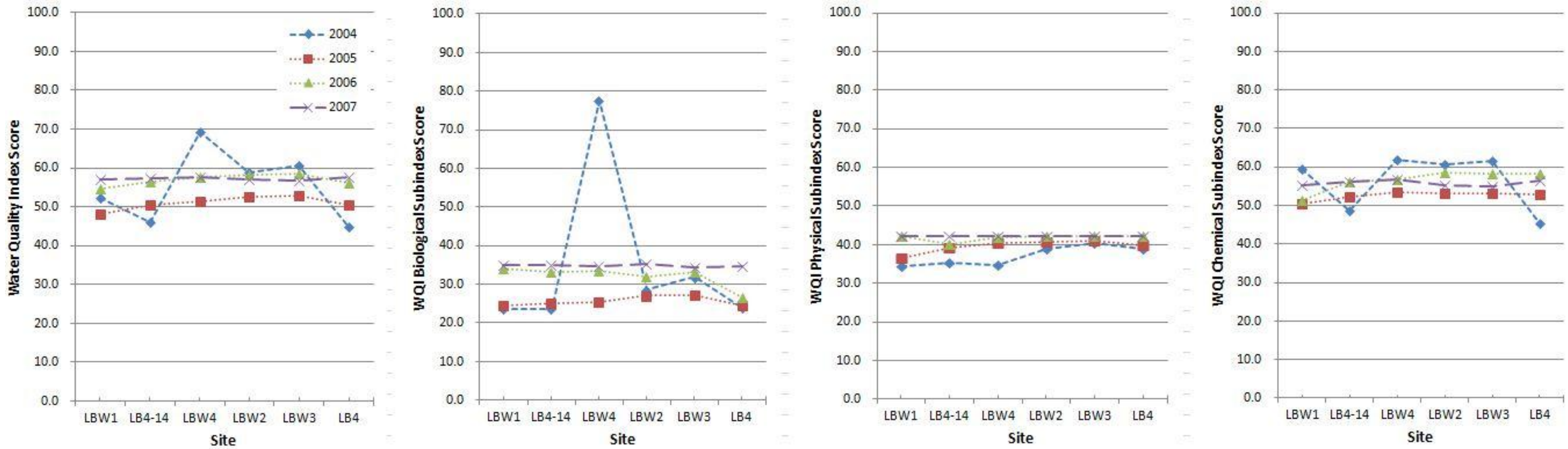


Figure 2-8. Annual water quality index (WQI) and biological, physical, and chemical WQI sub-index scores for Lower Little Bow River Watershed Evaluation of Benefical Management Practices (WEBs) sites, from 2004-2007.

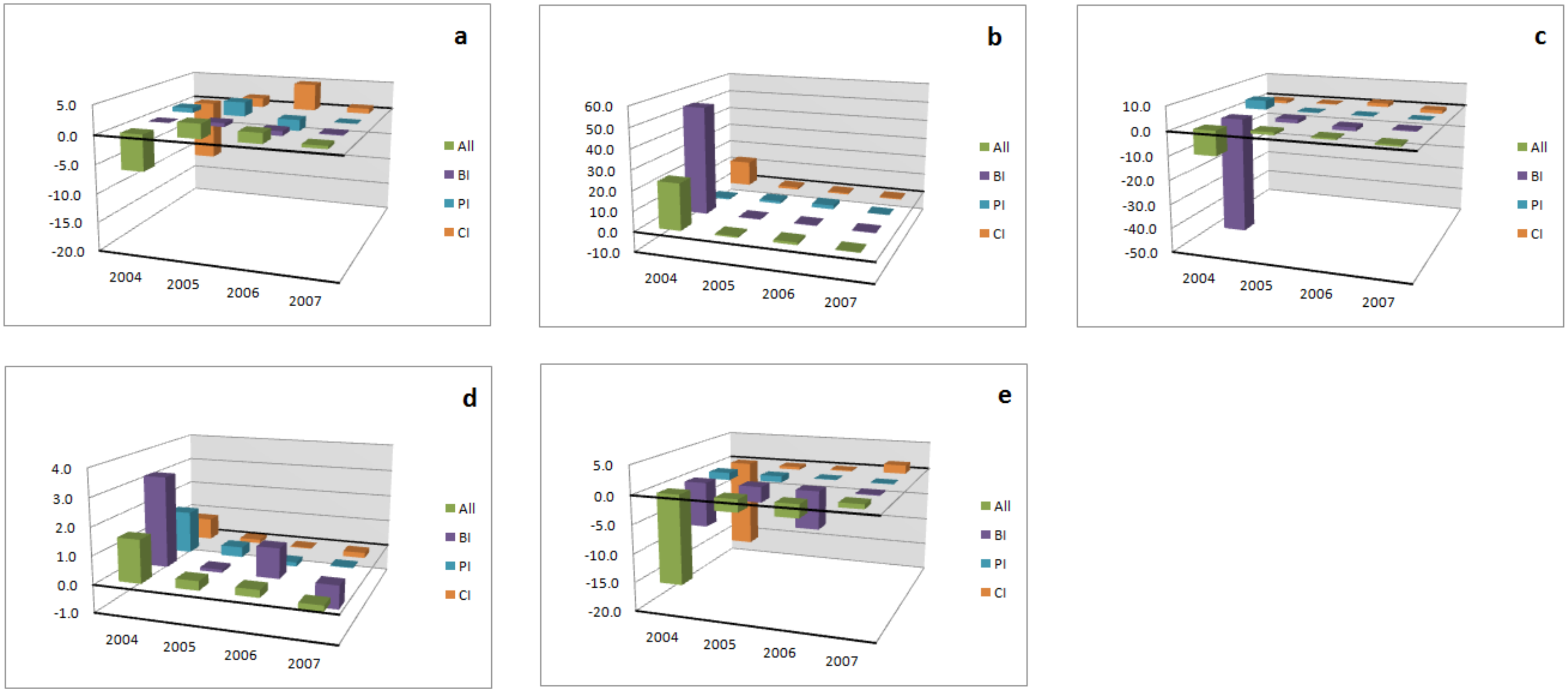


Figure 2-9. Reach differences in total and biological, physical, and chemical Water Quality Index (WQI) sub-index scores for Lower Little Bow River Watershed Evaluation of Beneficial Management Practices (WEBs) sites from 2004-2007 for Reach 5 (a), Reach 4 (b), Reach 3 (c), Reach 2 (d), and Reach 1 (e).

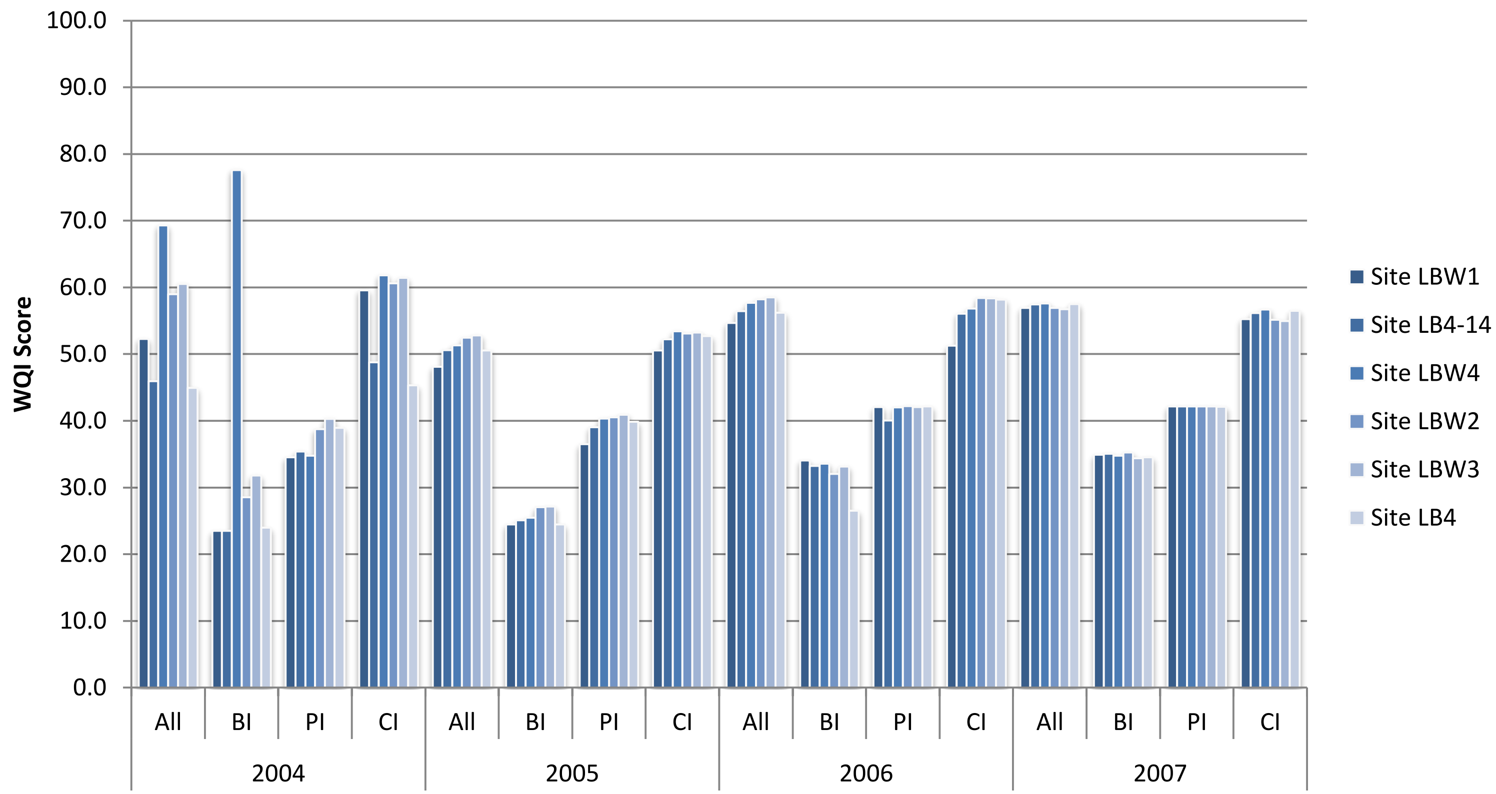


Figure 2-10. Annual water quality index (WQI) and biological, physical, and chemical WQI sub-index scores for Lower Little Bow River Watershed Evaluation of Beneficial Management Practices (WEBs) sites, from 2004-2007.

Table 2-1. Watershed Evaluation of Beneficial Management Practices (WEBs) study reaches, Lower Little Bow River.

Location[1]	**Reach**	**Water Quality Sites (Upstream/Downstream)**	**WEBs Land Use**	**Sampling Period**
0 km	5	LBW1 to LB4-14	off-stream watering system with riparian exclusion fencing (installed 2001) along an 800 m reach	May 2004 to December 2007
0.8 km	4	LB4-14 to LBW4	cattle grazing with access to the river channel and riparian areas	September 2004 to December 2007
1.8 km	3	LBW4 to LBW2	off-stream watering without fencing	September 2004 to December 2007
2.9 km	2	LBW2 to LBW3	conversion of annual cropland to forages on two irrigated barley fields	August 2004 to December 2007
4.2 km	1	LBW3 to LB4	buffer strips on irrigated fields	August 2004 to December 2007

[1] Locations are presented as river distance downstream from the upstream limit of the WEBs study area at site LBW1. Site LB4 represents the downstream limit of the WEBs study area.

Table 2-2. Watershed Evaluation of Beneficial Management Practices (WEBs) water quality parameters sampled from the Lower Little Bow River sub-watershed, Alberta, 2004-2007[1].

Class	Parameter	Unit	Expected Range	General Comments	Collection/Analysis
Physical	Temperature	°C	0 – 40°C	Fundamental water quality parameter, influences solubility, metabolic oxygen demand.	Field - portable water quality meter and associated probes (MultiLine P4, Wissenschaftlich-Technische, Werkstätten, Germany).
Physical	pH		4 – 10	Log[H+], fundamental water quality parameter, influences reactivity of chemical parameters.	Field - portable water quality meter and associated probes (MultiLine P4, WTW, Germany).
Physical	Dissolved oxygen	mg L^{-1}	5 – 10mg L^{-1}	Linked to temperature, (decreases with increasing temperature); atmospheric pressure increases DO; greater flow usually related to increased DO; important driver of aquatic ecosystem health.	Field - portable water quality meter and associated probes (MultiLine P4, WTW, Germany).
Physical	Turbidity	NTU	referenced to distilled water = 0NTU	Light transmission through media (cloudiness, colour); criteria often related to background levels; referenced to distilled water = 0NTU; can be influenced by colour, particle size and shape; individual watershed curves can be described to relate Turbidity and TSS.	Field - using the nephelometric method (APHA 1998) and portable turbidity meter (Hach Model 2100p, Loveland, CO).
Physical	Total Suspended Solids	mg L^{-1}		Measure of suspended particulate within the water column (e.g., silt and clay particles, plankton, algae, fine organic debris, and other particulate matter).	
Physical/ Chemical	Electrical conductivity	µS cm^{-1}	50 – 1500 µS cm^{-1}	Ionic content of water, also referred to as specific conductance (EC at 25°C).	Field - portable water quality meter and associated probes (MultiLine P4, WTW, Germany).
Chemical	Chloride	mmolc L^{-1} or mg L^{-1}		Found in most natural waters and not generally a concern at low levels. Toxicity at higher levels.	
Chemical	Total Ammonia	mg L^{-1}	< 0.1 mg L^{-1}	Criteria related to temperature and pH. Most reduced inorganic N (NH3 + NH4+). A product of microbiological decay of plant and animal protein and used in commercial fertilizers.	Lab - autoanalyzer (TRAACS 800, Bran and Luebbe Inc., Buffalo Grove, Illinois).

1 Details on analytical/field measurement protocols and equipment taken from Chapter 1 of Miller *et al.* 2008.

Table 2-2. Watershed Evaluation of Beneficial Management Practices (WEBs) water quality parameters sampled from the Lower Little Bow River sub-watershed, Alberta, 2004-2007[1].

Class	Parameter	Unit	Expected Range	General Comments	Collection/Analysis
Chemical	Nitrite	mg L^{1} or µg L^{-1}	<0.001 mg L^{-1}		Lab - autoanalyzer (TRAACS 800, Bran and Luebbe Inc., Buffalo Grove, Illinois).
Chemical	Nitrate	mg L^{-1}	< 0.3 mg L^{-1}	most oxidized and stable.	Lab - autoanalyzer (TRAACS 800, Bran and Luebbe Inc., Buffalo Grove, Illinois).
Chemical	Total Kjeldahl Nitrogen	mg L^{-1}		Nitrogen in ammonia and organic nitrogen, but not nitrate/nitrite.	Lab - autoanalyzer (TRAACS 800, Bran and Luebbe Inc., Buffalo Grove, Illinois).
Chemical	Total Nitrogen	mg L^{-1}		Sum of organic nitrogen, nitrate (NO3-), nitrite (NO2-), and ammonia (NH4+).	2004-2005: Total N = sum of TKN (unfiltered sample), NO_2-N and NO_3-N, with values below detection limits not included. 2006-2007: Lab - autoanalyzer Shimadzu TOC-V instrument with TNM-1 unit (Shimazdu, Kyoto, Japan).
Chemical	Total Dissolved Nitrogen	mg L^{-1}		Nitrogen found in sampled water passed through a 0.45 µm filter.	2004-2005: Total dissolved N (TDN) = sum of TKN (filtered sample), NO_2-N and NO_3-N, with values below detection limits not included. 2006-2007: Lab - autoanalyzer Shimadzu TOC-V with TNM-1 unit (Shimazdu, Kyoto, Japan).
Chemical	Total Particulate Nitrogen	mg L^{-1}		Differentiates between element bound to particles and that not bound.	The total particulate N (TPN) fraction was calculated as the difference between TN and TDN.
Chemical	Ortho-Phosphate (Dissolved Reactive P)	mg L^{-1}		Phosphates enter water bodies from surface runoff, natural decay of plants and animals, cleaners, sewage disposal, and fertilizers.	Lab - autoanalyzer (TRAACS 800, Bran and Luebbe Inc., Buffalo Grove, Illinois).
Chemical	Total Phosphate	mg L^{-1}		Phosphorus often results in excessive growth of aquatic plants and eutrophication of fresh water.	Lab - autoanalyzer (TRAACS 800, Bran and Luebbe Inc., Buffalo Grove, Illinois).
Chemical	Total Dissolved Phosphate	mg L^{-1}		Phosphate found in sampled water passed through a 0.45 µm filter.	Lab - autoanalyzer (TRAACS 800, Bran and Luebbe Inc., Buffalo Grove, Illinois).
Chemical	Total Particulate Phosphate	mg L^{-1}		Differentiates between element bound to particles and that not bound.	TPP fraction calculated as difference between TP and TDP.
Biological	Chlorophyll a	relative fluorescent		A measure of primary productivity.	Field - Chlorophyll-a measured during 2006 and 2007 using the fluorometric method (APHA 1998)

Table 2-2. Watershed Evaluation of Beneficial Management Practices (WEBs) water quality parameters sampled from the Lower Little Bow River sub-watershed, Alberta, 2004-2007[1].

Class	Parameter	Unit	Expected Range	General Comments	Collection/Analysis
		units (RFU)			with a portable fluorometer (Aquafluor, Turner Designs, Sunnyvale, CA).
Biological	Fecal coliforms	Colony forming units (CFU) per 100 ml)		For surface water used for irrigation, water should contain less than 100 CFU per 100ml (AEnv 1999).	Lab samples were filtered (0.45 µm), placed on mFC medium with Rosolic acid, incubated in the dark at 44.5°C for 24 hours. Fecal coliform colonies were identified by their blue color, counted under a stereoscopic microscope, and verified by single lauryl, brilliant green bile and *E. coli* broths.
Biological	E.coli	CFU		200 CFU per 100 ml are Recreational Water Quality Guidelines (Health Canada 1992, Alberta Environment 1999).	For *E. coli* analyses, membrane filters were placed on Lactose Gluconronide Agar medium, incubated and counted as for fecal coliform analysis. Colonies confirmed by addition of methyl red, use of citrate, and indole produced by tryphophane.

Table 2-3. Qualitative ratings associated with Water Quality Index score ranges.

Rating	WQI Score	Interpretation
Excellent	95.0 to 100.0	Water quality measurements never or very rarely exceed water quality guidelines.
Good	80.0 to 94.9	Water quality measurements rarely exceed water quality guidelines and, usually, by a narrow margin.
Fair	65.0 to 79.9	Water quality measurements sometimes exceed water quality guidelines and, possibly, by a wide margin.
Marginal	45.0 to 64.9	Water quality measurements often exceed water quality guidelines and/or exceed the guidelines by a considerable margin.
Poor	0 to 44.9	Water quality measurements usually exceed water quality guidelines and/or exceed the guidelines by a considerable margin.

Table 2-4. Water variables and associated quality criteria available for use in index calculation.

Water Quality Variable	Unit	Criteria	Comments[1]
Temperature	change in °C	<3	Alberta surface water guideline: not to be increased by more than 3°C above ambient water temperature. CEQG Temperature guideline: Thermal additions should not alter thermal stratification or turnover dates, exceed maximum weekly average temperatures, nor exceed maximum short-term temperatures. Temperature guidelines are focused more on point discharges which may alter thermal characteristics of receiving waters (e.g., process waters from hydroelectric plants, pulp mills that are heated by their mechanical or cooling use; warm discharge from upper waters in storage reservoirs, cool discharge from deep waters in storage reservoirs).
pH	no units	6.5 to 9	Alberta River Water Quality Index, CEQG: 6.5 to 9. Alberta surface water guideline: to be in the range of 6.5 to 8.5 but not altered by more than 0.5 pH units from background values. Values below 6.5 are considered acidic, while values greater than 9 are considered basic. Some natural water conditions are acidic (e.g., peat draining waters) and others are made acidic (e.g., acid mine drainage, acidification of waters contacting unweathered sulphide bearing rock). Some water is naturally more basic (e.g., groundwater contacting limestone formations) while others become basic as a result of pollutants (e.g., concrete wash water).
Dissolved Oxygen (DO)	mg L^{-1}	>5.5 and >6.5; >8.3 (May-June)	Alberta River Water Quality Index: non-compliant, if less than 6.5 mg L^{-1}. Alberta surface water quality: chronic guideline should be increased to 8.3 from mid May to the end of June to protect emergence of mayfly species into adults, and increased to 9.5 mg L^{-1} for those areas and times where embryonic and larval stages (from spawning to 30 days after hatching) develop within gravel beds (some salmonids). Where natural conditions alone create dissolved oxygen concentrations less than 110% of the applicable criteria means or minima or both, the minimum acceptable concentration is 90% of the natural concentrations (AEnv 1999). CEQG Dissolved Oxygen guideline: warm water biota at early life stages = 6 mg L^{-1} and other life stages = 5.5 mg L^{-1}; cold water biota at early life stages = 9.5 mg L^{-1}, and other life stages = 6.5 mg L^{-1}. USEPA Dissolved Oxygen guidelines: warm water biota: early life stages = 6 mg L^{-1} (7 day mean), 5.0 mg L^{-1} (1 day minimum), other life stages = 5.5 mg L^{-1} (30 day mean), 4.0 mg L^{-1} (7 day mean minimum), 3.0 mg L^{-1} (1 day minimum); cold water biota: early life stages = 9.5 mg L^{-1} (7 day mean intergravel), 6.5 mg L^{-1} (7 day mean water column), 8.0 mg L^{-1} (1 day minimum mean intergravel), 5.0 mg L^{-1} (1 day mean water column), other life stages = 6.5 mg L^{-1} (30 day mean), 5.0 mg L^{-1} (7 day mean), 4.0 mg L^{-1} (1 day minimum).

Table 2-4. Water variables and associated quality criteria available for use in index calculation.

Water Quality Variable	**Unit**	**Criteria**	**Comments**[1]
Total Suspended Solids (TSS)	change from background mg L^{-1}	<10 and <25	Alberta Surface Water Guideline: not to be increased by more than 10 mg L^{-1} over background value. CEQG Suspended Solids Guideline: for clear flow - maximum increase of 25 mg L^{-1} from background levels for any short-term exposure (e.g., 24-h period). Maximum increase of 5 mg L^{-1} from background levels for any long-term exposure (e.g., inputs lasting between 24 h and 30 d). For high flow - maximum increase of 25 mg L^{-1} from background levels at any time when background levels are between 25 and 250 mg L^{-1}. Should not increase more than 10% of background levels when background is >250 mg L^{-1}. USEPA: Settleable and suspended solids should not reduce the depth of the compensation point for photosynthetic activity by more than 10% from the seasonally established norm for aquatic life (for solids – suspended, settleable, and turbidity).
Turbidity	NTU change from background	<2 and <8	CEQG Turbidity Guideline: for clear flow - Maximum increase of 8 NTU from background levels for any short-term exposure (e.g., 24-h period). Maximum increase of 2 NTU from background levels for any long-term exposure (e.g., inputs lasting between 24-h and 30-d). For high flow or turbid waters - maximum increase of 8 NTU from background levels at any one time when background levels are between 8 and 80 NTU. Should not increase more than 10% of background levels when background is >80 NTU.
F.coli	CFU per 100ml	<100	Alberta Agricultural Use Guideline: 100 CFU per 100ml (AEnv 1999, from CCME 1999)
E.coli	CFU per 100ml	<200	CEQG Guideline: the geometric mean of at least five samples taken during a period not to exceed 30 d should not exceed 200 *E. coli* per 100 ml. Resampling should be performed when any sample exceeds 400 E.coli per 100 ml (CCME 1999). Alberta: Alberta River WQI uses 400 CFU/100ml
Chloride	mg L^{-1}	<150 and <350	Alberta Agricultural Use Guideline: 100-700 mg L^{-1} (AEnv 1999 from CCME 1999). CCME WQI Calculator suggests a limit of 150 mg L^{-1}. Range for protection from foliar damage: 355 - 710 mg L^{-1}: for alfalfa, barley, corn and cucumbers. USEPA uses a limit of 860 mg L^{-1} maximum concentration, 230 mg L^{-1} continuous concentration.

Table 2-4. Water variables and associated quality criteria available for use in index calculation.

Water Quality Variable	**Unit**	**Criteria**	**Comments**[1]
Ammonia	mg L^{-1} (un-ionized ammonia)	0.019	CEQG Ammonia Guideline: 19ug L^{-1} expressed as un-ionized ammonia per L. This would be equivalent to 16 µg ammonia-N per L (=19*14.0067 / 17.35052, rounded to two significant figures). The value calculation uses NH3-N values and considers them with temp and pH to calculate an ammonia value to compare to the guideline (e.g., CEQG guideline for ammonia: 1.37 mg L^{-1} at pH 8.0, 10ºC; 2.20 mg L^{-1} at pH 6.5, 10ºC). The toxicity of ammonia relates primarily to the unionized form (NH3) and the concentration of unionized ammonia present in water increases with pH and temperature. The above values represent total ammonia-nitrogen concentrations (at various temperatures and pH levels) above which accompanying NH3 concentrations may harm aquatic life.
Nitrate (NO_3-N)	mg L^{-1}	<2.983	CEQG Guideline for Nitrate (an interim guideline for protection from direct toxic effects as the guidelines do not consider indirect effects due to eutrophication): 13000 µg L^{-1} for nitrate (NO3), 2935 µg L^{-1} for values in Nitrate-nitrogen (NO3-N). Alberta: Agricultural Use for Livestock water is NO3+NO2<100 mg L^{-1}. Although not directly toxic to freshwater aquatic life, these values are included due to their broader influence on conditions that affect aquatic life. Concentrations that stimulate weed growth should be avoided (AEnv 1999).
Nitrite (NO_2-N)	mg L^{-1}	<0.06	CEQG Surface Water Quality Guideline for Aquatic Life and Alberta River WQI: 0.06 mg L^{-1}. Alberta Agricultural Use Guideline: 10mg/L for livestock water
Nitrogen	mg L^{-1}	<1.0	CEQG Surface Water Quality Guideline for Aquatic Life and Alberta Surface Water Guidelines: 1.0 mg L^{-1} chronic guideline (total inorganic + organic N)
Phosphorus	mg L^{-1}	<0.01, <0.02, <0.05	Alberta Surface Water Guideline: Alberta chronic 0.05 mg L^{-1} for phosphorus as P (total inorganic and organic). Although not directly toxic to freshwater aquatic life, these values are included due to their broader influence on conditions that affect aquatic life (AEnv 1999). CEQG: CCME WQI Calculator identifies seasonal variation in P, with criteria set at 0.01 mg L^{-1} for winter and 0.02 mg L^{-1} for the growing season. CCME Canadian Guidance Framework for Phosphorus is for developing phosphorus guidelines. It provides Trigger Ranges for Total Phosphorus (ug/L):ultra-oligotrophic; oligotrophic 4-10; mesotrophic 10-20; meso-eutrophic 20-35; eutrophic 35-100; hyper-eutrophic >100.
Chlorophyll a	RFU (relative fluorescent units)	<0.010 mg/L	Alberta Environment uses chlorophyll a concentrations as a measure of productivity with a range between 2.5 to 25 µgL^{-1} matched to narrative descriptions of eutrophic through oligotrophic waters. Comparison requires the conversion of RFU to concentration; this depends on many factors operating at the time of measurement. AAFC's Prairie Farm Program also identifies an objective of <0.010mgL^{-1} for chlorophyll in surface waters (AAFC 2007)

Table 2-4. Water variables and associated quality criteria available for use in index calculation.

Water Quality Variable	Unit	Criteria	Comments[1]
Electrical Conductivity (EC)	$\mu S\ cm^{-1}$	<750	Colorado State identifies an EC threshold of 0.75 dSm^{-1} associated with salinity risks to crops (http://www.ext.colostate.edu/pubs/crops/00506.html)
TKN	$mg\ L^{-1}$	NA	Total Kjeldahl nitrogen
TDN	$mg\ L^{-1}$	NA	Total dissolved nitrogen
TPN	$mg\ L^{-1}$	NA	Total particulate nitrogen
PO4-P	$mg\ L^{-1}$	NA	Phosphate – phosphorus
TDP	$mg\ L^{-1}$	NA	Total dissolved phosphorus
TPP	$mg\ L^{-1}$	NA	Total particulate phosphorus

[1] Developed from the Surface Water Quality Guidelines for use in Alberta, Alberta Environment 1999, CEQG from CCME 2011.

Table 2-5. Annual median, median absolute deviation (MAD), interquartile range (IQR) and number of samples (n) for water quality variables sampled from the across the Lower Little Bow River WEBs study area, 2004-2007.

		2004				**2005**				**2006**				**2007**			
Variable - Unit		**n**	**Median**	**MAD**	**IQR**	**n**	**Median**	**MAD**	**IQR**	**n**	**Median**	**MAD**	**IQR**	**n**	**Median**	**MAD**	**IQR**
Flow	m^3s^{-1}	112	1.34	0.26	0.333	192	2.67	2.37	5.325	168	2.3	0.31	0.5325	150	2.54	0.74	1.0525
Temp	°C	142	11.1	7.93	10.63	222	13	5.63	6.9	204	13.05	8.6	11.55	185	13.7	5.93	8.2
pH	na	142	8.49	0.04	0.05	222	8.5	0.15	0.1	210	8.6	0	0.1	186	8.5	0	0
DO	mgL^{-1}	134	7.65	2.41	3.33	200	7.11	1.17	1.638	191	6.62	3.4	5.295	186	8.68	1.33	2012
EC	μScm^{-1}	141	396	57.82	84	222	452	39.29	69.75	200	637	57.08	80.75	186	597	138.18	170.75
TSS	mgL^{-1}	142	52.2	32.91	45.8	222	58	51.59	78.4	210	53.8	28.76	42.6	186	67.2	27.28	36
Turb.	NTU	142	36.3	22.98	35.93	222	28.95	23.5	54.08	198	24.6	12.31	17.1	186	33.4	10.82	14.55
F.coli	CFU	88	220	310.6	399.3	111	140	195.7	426.5	114	85	120.09	297	89	210	277.25	402
E.coli	CFU	88	165	238.7	361.5	111	110	157.2	389.5	110	66	94.89	275.25	89	210	252.04	336
Chl a	RFU	0				6	124	4.45	5	177	51	20.76	42	186	55	13.34	22.75
Cl	mgL^{-1}	142	5.85	1.7	2.65	222	7.1	0	0	210	14.2	5.19	3.6	186	10.6	5.19	3.6
NH3	mgL^{-1}	142	0.1	0	0	222	0.1	0	0	210	0.1	0	0	186	0.1	0	0
NO3-N	mgL^{-1}	142	0.1	0	0	222	0.1	0.03	0.03	210	0.1	0	0	186	0.1	0	0
NO2-N	mgL^{-1}	142	0.1	0	0	222	0.1	0	0	210	0.1	0	0	186	0.1	0	0
TKN	mgL^{-1}	142	0.4	0.15	0.2	222	0.4	0.15	0.3	210	0.2	0.15	0.1	186	0.2	0.15	0.1
TN	mgL^{-1}	142	0.58	0.1	0.168	222	0.6	0.18	0.295	210	0.41	0.1	0.1375	186	0.35	0.06	0.08
TDN	mgL^{-1}	142	0.28	0.07	0.1	222	0.33	0.1	0.128	210	0.35	0.07	0.1	186	0.32	0.06	0.08
TPN	mgL^{-1}	142	0.13	0.12	0.16	222	0.27	0.14	0.228	210	0.04	0.04	0.05	186	0.03	0.03	0.03
PO4-P	mgL^{-1}	142	0	0	0.004	222	0.01	0.01	0.014	210	0.01	0	0	186	0.01	0	0
TDP	mgL^{-1}	142	0.01	0.01	0.011	222	0.02	0.01	0.031	210	0.01	0	0	186	0.01	0	0
TP	mgL^{-1}	142	0.06	0.03	0.038	222	0.07	0.05	0.078	210	0.05	0.02	0.03	186	0.06	0.01	0.02
TPP	mgL^{-1}	142	0.04	0.03	0.037	222	0.05	0.04	0.052	210	0.04	0.02	0.031	186	0.05	0.01	0.02

Table 2-6. Seasonal medians , median absolute deviation (MAD), interquartile range (IQR) and number of samples (n) for water quality variables sampled at the Lower Little Bow River Watershed Evaluation of Beneficial Management Practices (WEBs) study area, 2004-2007.

		Spring				Summer				Fall				Winter			
Variable - Unit		**n**	**Median**	**MAD**	**IQR**	**n**	**Median**	**MAD**	**IQR**	**n**	**Median**	**MAD**	**IQR**	**n**	**Median**	**MAD**	**IQR**
Flow	m^3s^{-1}	177	2	1.25	1.64	289	2.42	0.61	1.06	156	1.84	0.96	1.56	0	NA	NA	NA
Temp	°C	189	13.1	4	6	282	18.6	3.26	4.3	186	8.95	3.19	5.33	96	4.15	3.63	4.7
pH	na	189	8.5	0.15	0.1	289	8.5	0.01	0.03	186	8.5	0.07	0.14	96	8.5	0	0
DO	mgL^{-1}	173	7.51	1.65	2.37	275	6.2	0.93	1.25	185	9.27	1.99	2.63	78	9.79	1.38	1.92
EC	μScm^{-1}	189	612	196	276	283	533	142	168	181	549	111	134	96	602	165	217
TSS	mgL^{-1}	189	92	54.6	81.6	289	66.4	26.1	36.4	186	42.2	23.4	31.8	96	30	16.6	27.9
Turb.	NTU	189	42.9	31.9	62.5	283	35.1	14.7	21.1	180	23.7	13.6	19.4	96	17.9	7.49	10.1
F.coli	CFU	87	210	257	443	141	400	193	260	96	46	46.7	109	78	7.5	6.67	8
E.coli	CFU	87	200	230	401	137	360	193	300	96	41	47.4	90.5	78	4	4.45	9.75
Chl a	RFU	108	82.5	34.8	49.3	133	48	8.9	11	91	49	10.4	16.5	37	119	43	56
Cl	mgL^{-1}	189	10.6	5.34	7.1	289	10.6	5.19	7.1	186	10.6	5.19	4.45	96	10.6	5.19	3.6
NH3	mgL^{-1}	189	0.1	0	0	289	0.1	0	0	186	0.1	0	0	96	0.1	0	0
NO3-N	mgL^{-1}	189	0.1	0	0	289	0.1	0	0	186	0.1	0	0	96	0.1	0	0
NO2-N	mgL^{-1}	189	0.1	0	0	289	0.1	0	0	186	0.1	0	0	96	0.1	0	0
TKN	mgL^{-1}	189	0.3	0.15	0.3	289	0.3	0.15	0.2	186	0.2	0.15	0.3	96	0.3	0.15	0.2
TN	mgL^{-1}	189	0.5	0.13	0.27	289	0.45	0.18	0.27	186	0.42	0.16	0.23	96	0.49	0.09	0.133
TDN	mgL^{-1}	189	0.38	0.07	0.1	289	0.32	0.06	0.07	186	0.29	0.04	0.06	96	0.38	0.11	0.145
TPN	mgL^{-1}	189	0.11	0.1	0.2	289	0.07	0.07	0.22	186	0.05	0.06	0.17	96	0.1	0.1	0.19
PO4-P	mgL^{-1}	189	0.01	0	0	289	0.01	0	0	186	0.01	0	0.01	96	0.01	0	0.004
TDP	mgL^{-1}	189	0.01	0	0.01	289	0.01	0	0.01	186	0.01	0	0.01	96	0.01	0	0
TP	mgL^{-1}	189	0.08	0.04	0.07	289	0.06	0.02	0.03	186	0.04	0.01	0.03	96	0.04	0.02	0.030
TPP	mgL^{-1}	189	0.06	0.03	0.06	289	0.05	0.01	0.02	186	0.03	0.01	0.02	96	0.03	0.02	0.021

Table 2-7. Water Quality Index scores (differences[1]) calculated for annual and seasonal terms, Lower Little Bow River, Alberta (2004-2007).

Year	Season	Reach					Site					
		5	**4**	**3**	**2**	**1**	**LBW1**	**LB4-14**	**LBW4**	**LBW2**	**LBW3**	**LB4**
2004		49.1 (-6.3)	57.6 (23.3)	64.1 (-10.3)	59.7 (1.6)	52.7 (-15.6)	52.3	46.0	69.3	59.0	60.5	44.9
	Winter											
	Spring	41.0 (-10.3)					46.1	35.8				38.6
	Summer	56.2 (4.8)			59.8 (4.5)	60.9 (-2.3)	53.8	58.6		57.6	62.1	59.8
	Fall	63.4 (-0.5)	66.2 (6.2)	66.3 (-6.0)	64.2 (1.9)	65.8 (1.3)	63.6	63.1	69.3	63.2	65.1	66.4
2005		49.4 (2.5)	51.0 (0.7)	51.9 (1.1)	52.6 (0.3)	51.7 (-2.2)	48.1	50.6	51.3	52.4	52.8	50.6
	Winter											
	Spring	82.1 (3.2)	83.7 (-0.1)	81.7 (-3.7)	80.4 (1.1)	80.2 (-1.5)	80.5	83.7	83.6	79.9	81.0	79.5
	Summer	42.5 (1.1)	43.3 (0.4)	44.5 (2.2)	45.8 (0.3)	45.1 (-1.5)	41.9	43.1	43.4	45.6	45.9	44.4
	Fall	49.7 (-0.7)	49.9 (1.3)	50.5 (-0.1)	50.6 (0.4)	50.4 (-0.9)	50.0	49.3	50.5	50.4	50.9	49.9
2006		55.6 (1.8)	57.1 (1.2)	57.9 (0.6)	58.4 (0.3)	57.3 (-2.3)	54.7	56.4	57.7	58.2	58.5	56.2
	Winter	67.2 (-16.8)	67.4 (17.1)	77.4 (2.9)	79.1 (0.4)	77.5 (-3.6)	75.6	58.8	75.9	78.9	79.3	75.7
	Spring	56.6 (1.5)	56.2 (-2.4)	54.9 (-0.2)	55.3 (0.9)	52.9 (-5.8)	55.9	57.4	55.0	54.8	55.8	50.0
	Summer	66.7 (-3.2)	63.9 (-2.4)	65.2 (5.1)	67.7 (-0.2)	65.6 (-4.0)	68.3	65.1	62.7	67.8	67.6	63.6
	Fall	71.6 (4.0)	73.8 (0.5)	73.7 (-0.5)	73.5 (0.0)	73.4 (-0.2)	69.6	73.6	74.0	73.5	73.5	73.3
2007		57.2 (0.5)	57.5 (0.2)	57.3 (-0.7)	56.8 (-0.2)	57.1 (0.8)	56.9	57.4	57.6	56.9	56.7	57.5
	Winter	72.4 (-2.6)	74.0 (5.9)	71.6 (-10.7)	69.9 (7.4)	75.4 (3.6)	73.7	71.1	77.0	66.2	73.6	77.2
	Spring	66.0 (-8.0)	64.0 (4.0)	67.8 (3.7)	69.6 (-0.1)	69.1 (-0.8)	70.0	62.0	65.9	69.7	69.5	68.8
	Summer	59.7 (0.2)	59.4 (-0.8)	59.2 (0.5)	59.3 (-0.4)	59.2 (0.2)	59.6	59.8	59.0	59.5	59.1	59.2
	Fall	68.7 (9.9)	73.8 (0.3)	73.6 (-0.7)	70.0 (-6.6)	70.0 (6.6)	63.8	73.6	73.9	73.2	66.7	73.3

[1] Difference values represent change between upstream and downstream scores. Negative values indicate a degraded water quality score.

Table 2-8. Reliability of annual and seasonal indices scoring water quality at Watershed Evaluation of Beneficial Management Practices (WEBs) sites on the Lower Little Bow River using Intraclass Correlation (ICC).

Raters (Indices)	**Subjects (Site-Years)**	**ICC**	**p value**	**95% CI**
All Indices	12	-0.137	(p=0.95)	-0.198 < ICC < 0.04
Spring, Summer, Fall, Annual	21	-0.0826	(p = 0.816)	-0.191 < ICC < 0.12
Summer, Fall, Annual	23	0.658	(p = 4.08e-08)	0.443 < ICC < 0.822

Table 2-9. Spearman rank correlation coefficients for paired annual and seasonal water quality index (WQI) scores at WEBs sites on the Lower Little Bow River.

Annual vs.	**S**	**r_s**	**p-value**
Winter	90.658	0.683	0.01436
Spring	1788.161	-0.161	0.4853
Summer	927.187	0.542	0.007563
Fall	898.477	0.609	0.001574

Table 2-10. Wilcoxon signed rank test of reach differences in annual and seasonal water quality index scores from paired upstream and downstream sites on the Lower Little Bow River, 2004-2007.

			Confidence Interval[1]		
	V	**p-value**	**Lower**	**Upper**	**Sample Estimates: (pseudo)median**
Reach 5	90	0.8617	-2.049956	3.899968	0.2499475
Reach 4	29	0.04431	-7.35	-0.05	-1.425
Reach 3	76	0.6979	-1.549961	3.649974	0.2774446
Reach 2	28.5	0.04363	-1.89994365	-0.04999783	-0.4795014
Reach 1	113	0.0883	-0.3000469	2.9500679	1.250012

[1] 95% confidence interval

Table 2-11. Water Quality Index scores (differences[1]) calculated at annual terms for aggregate and sub-index collections of variables, Lower Little Bow River, Alberta (2004-2007).

		Reach					**Site**					
Year	**Sub-index**	**5**	**4**	**3**	**2**	**1**	**LBW1**	**LB4-14**	**LBW4**	**LBW2**	**LBW3**	**LB4**
2004		49.1 (-6.3)	57.6 (23.3)	64.1 (-10.3)	59.7 (1.6)	52.7 (-15.6)	52.3	46.0	69.3	59.0	60.5	44.9
	BI	23.5 (0.0)	50.5 (54.1)	53.1 (-49.1)	30.2 (3.3)	27.9 (-7.9)	23.5	23.5	77.6	28.5	31.8	24.0
	PI	35.0 (0.9)	35.1 (-0.6)	36.8 (4.0)	39.5 (1.5)	39.6 (-1.3)	34.5	35.4	34.8	38.8	40.3	39.0
	CI	54.2 (-10.8)	55.3 (13.0)	61.2 (-1.2)	61.0 (0.8)	53.4 (-16.1)	59.5	48.8	61.8	60.6	61.4	45.3
2005		49.4 (2.5)	51.0 (0.7)	51.9 (1.1)	52.6 (0.3)	51.7 (-2.2)	48.1	50.6	51.3	52.4	52.8	50.6
	BI	24.8 (0.7)	25.3 (0.4)	26.3 (1.6)	27.1 (0.1)	25.8 (-2.7)	24.5	25.1	25.5	27.0	27.2	24.4
	PI	37.8 (2.6)	39.7 (1.3)	40.4 (0.2)	40.7 (0.4)	40.4 (-1.1)	36.5	39.0	40.3	40.5	40.9	39.8
	CI	51.4 (1.7)	52.8 (1.2)	53.2 (-0.3)	53.1 (0.1)	53.0 (-0.5)	50.5	52.2	53.4	53.1	53.2	52.7
2006		55.6 (1.8)	57.1 (1.2)	57.9 (0.6)	58.4 (0.3)	57.3 (-2.3)	54.7	56.4	57.7	58.2	58.5	56.2
	BI	33.7 (-0.8)	33.4 (0.3)	32.8 (-1.5)	32.6 (1.1)	29.8 (-6.6)	34.1	33.3	33.6	32.1	33.2	26.5
	PI	41.1 (-0.2)	41.0 (1.9)	42.1 (0.2)	42.1 (-0.2)	42.1 (0.1)	42.1	40.1	42.0	42.2	42.0	42.1
	CI	53.7 (4.8)	56.5 (0.8)	57.6 (1.6)	58.4 (0.0)	58.3 (-0.2)	51.3	56.1	56.8	58.4	58.4	58.2
2007		57.2 (0.5)	57.5 (0.2)	57.3 (-0.7)	56.8 (-0.2)	57.1 (0.8)	56.9	57.4	57.6	56.9	56.7	57.5
	BI	35.0 (0.2)	34.9 (-0.3)	35.0 (0.5)	34.8 (-0.9)	34.5 (0.2)	34.9	35.1	34.8	35.3	34.4	34.6
	PI	42.2 (0.0)	42.2 (0.0)	42.1 (0.0)	42.1 (0.0)	42.1 (0.0)	42.2	42.2	42.2	42.1	42.2	42.1
	CI	55.7 (0.9)	56.4 (0.5)	55.9 (-1.5)	55.1 (-0.2)	55.7 (1.5)	55.2	56.1	56.7	55.2	55.0	56.5

[1] Difference values represent change between upstream and downstream scores. Negative values indicate a degraded water quality score.

Table 2-12. Reliability of total and sub-index indices scoring water quality at Watershed Evaluation of Beneficial Management Practices (WEBs) sites on the Lower Little Bow River using Intraclass Correlation (ICC).

Raters (Indices)	Subjects (Site-Years)	ICC	p value	95% CI
All Indices	24	0.338	0.000194	0.136 < ICC < 0.573
BI, PI, CI	24	0.132	0.137	-0.095 < ICC < 0.137
Total and CI	24	0.832	1.25e-07	0.651 < ICC < 0.924

Table 2-13. Spearman rank correlation coefficients for paired total and subindex water quality index (WQI) scores at WEBs sites on the Lower Little Bow River.

Total vs.	S	r_s	p value
BI	643.92	0.720	7.277e-05
PI	1449.51	0.370	0.07532
CI	344.87	0.850	1.459e-07

Table 2-14. Wilcoxon signed rank test of reach total and sub-index water quality index scores from paired upstream and downstream sites on the Lower Little Bow River, 2004-2007.

			Confidence Interval[1]		
	V	p-value	Lower	Upper	Sample Estimates: (pseudo)median
Reach 5	40	0.451	-1.700014	2.700056	-0.5500612
Reach 4	8	0.003425	-11.949933	0.4000337	-0.999725
Reach 3	74	0.7759	-0.7999763	4.3499575	0.1000245
Reach 2	25	0.04987	-9.50E-01	-4.86E-05	-0.3999382
Reach 1	115.5	0.01507	0.3000736	7.3999658	1.726766

Appendix

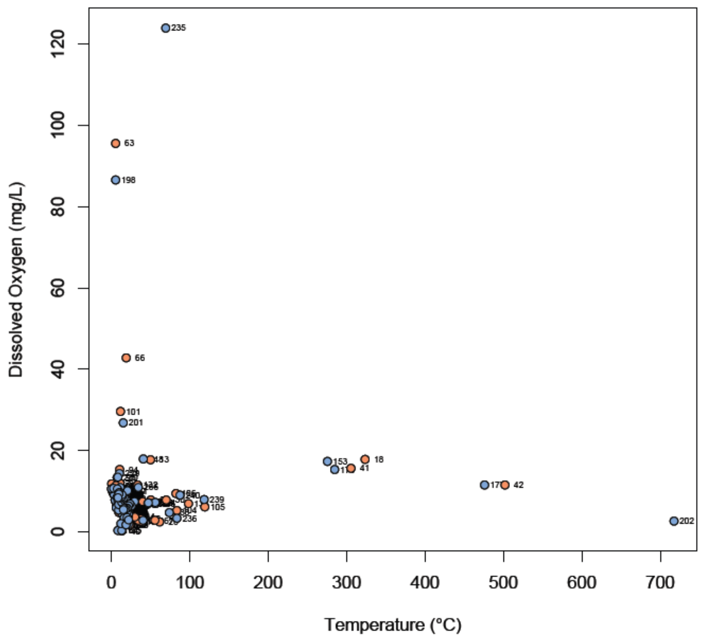


Figure A2-1. Scatterplot used in outlier/error checking, with sample ID labels plotted.

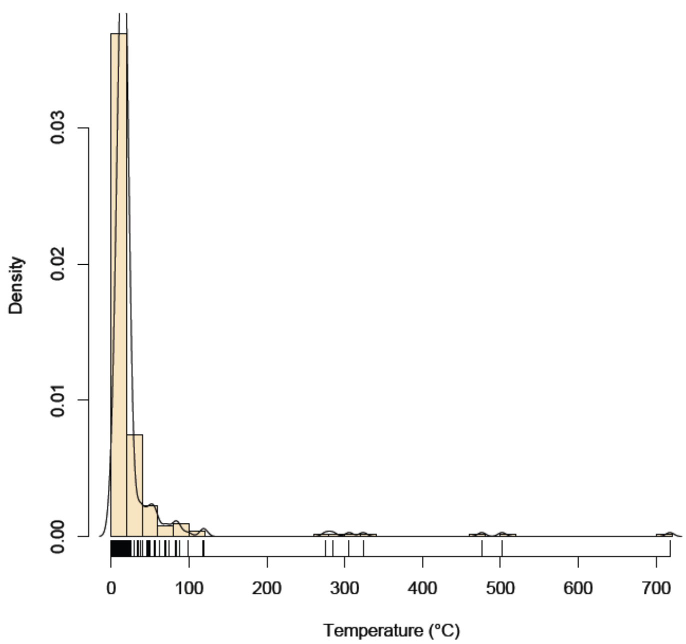


Figure A2-2. Histogram of observed temperatures during 2004 to 2007 Watershed Evaluation of Beneficial Management Practices (WEBs) sampling of the Lower Little Bow River, with potential data errors visible as extreme values.

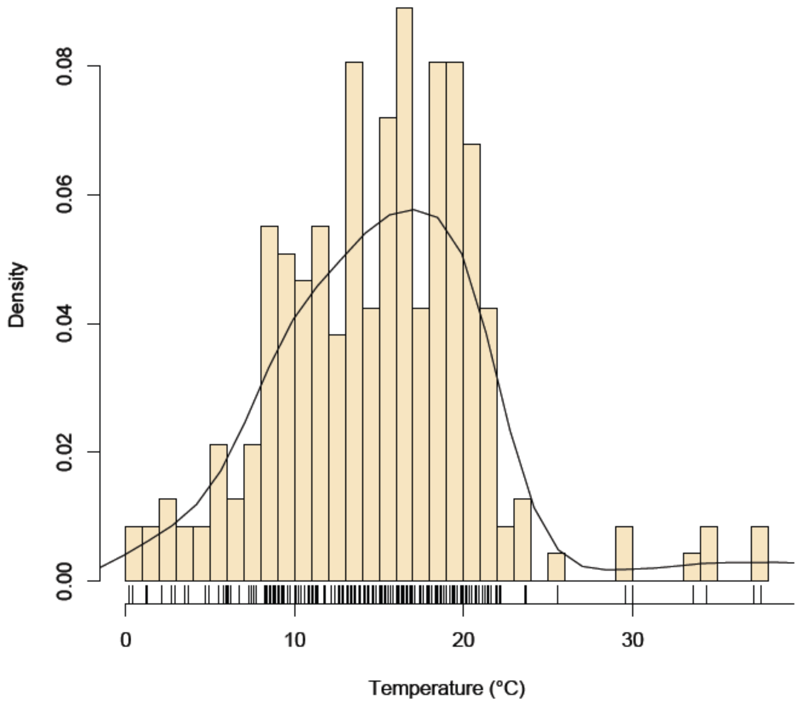


Figure A2-3. Replotted histogram following correction of data entry errors for observed temperatures during 2004 to 2007 Watershed Evaluation of Beneficial Management Practices (WEBs) sampling of the Lower Little Bow River.

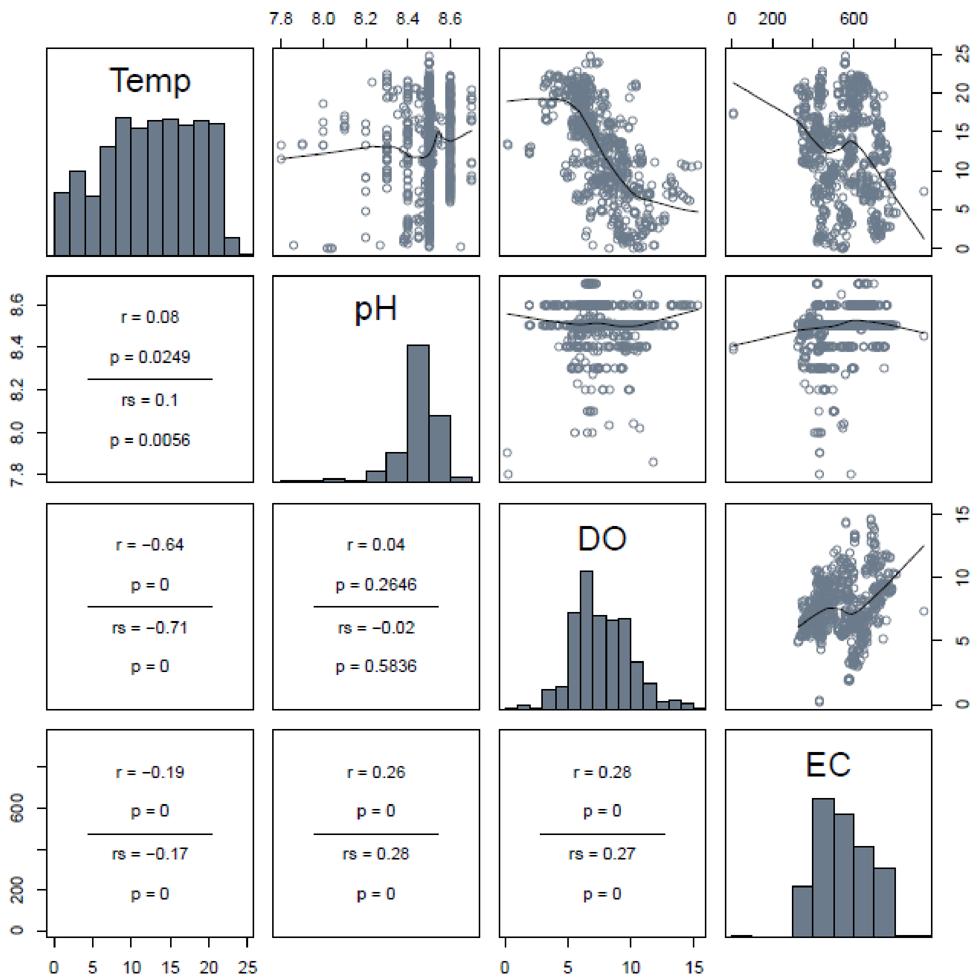


Figure A2-4a. Multiplot matrices generated in R, showing histograms along the diagonal axis, X:Y scatterplots and results of both Pearson (r) and Spearman (rs) correlation analysis.

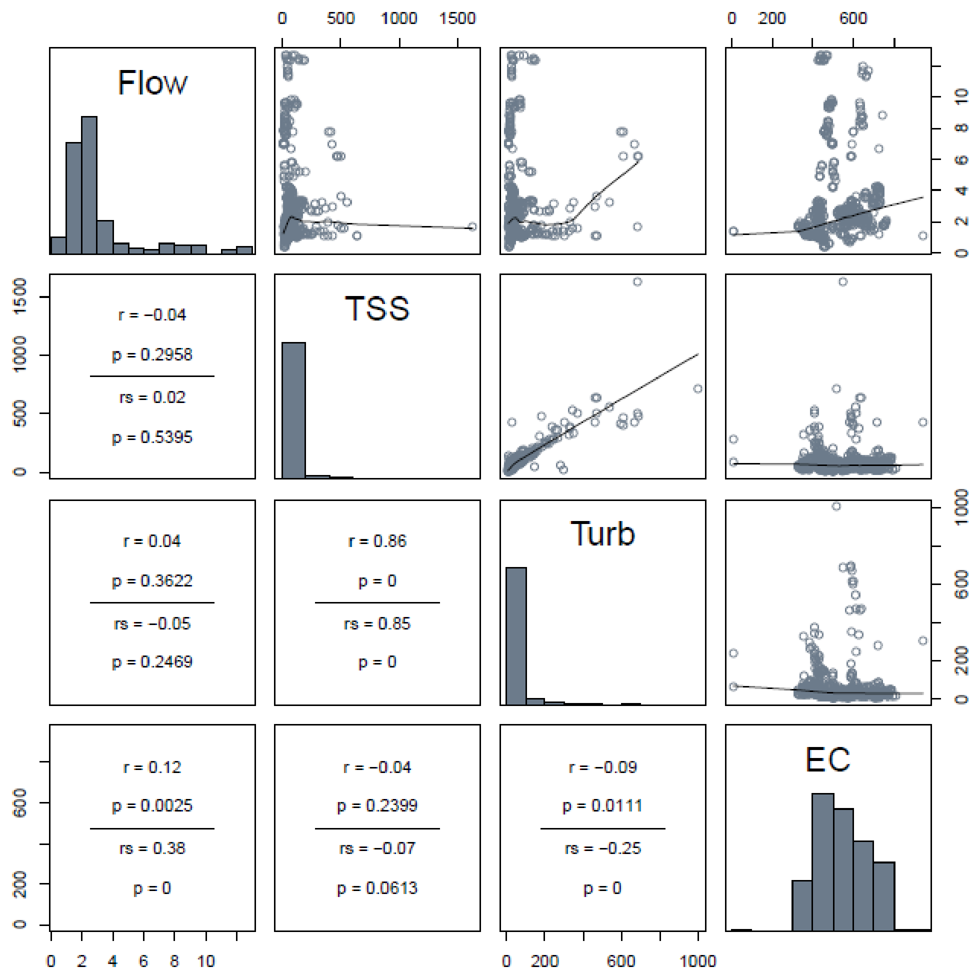


Figure A2-4b. Multiplot matrices generated in R, showing histograms along the diagonal axis, X:Y scatterplots and results of both Pearson (r) and Spearman (rs) correlation analysis.

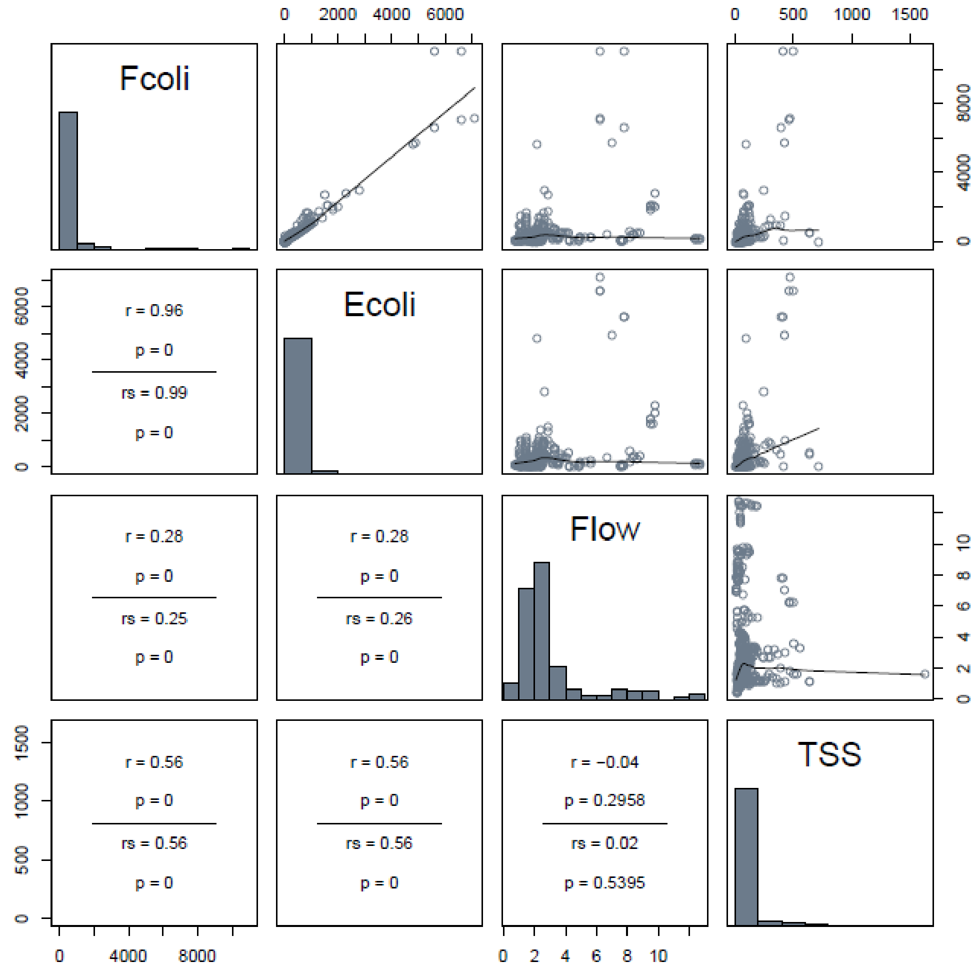


Figure A2-4c. Multiplot matrices generated in R, showing histograms along the diagonal axis, X:Y scatterplots and results of both Pearson (r) and Spearman (rs) correlation analysis.

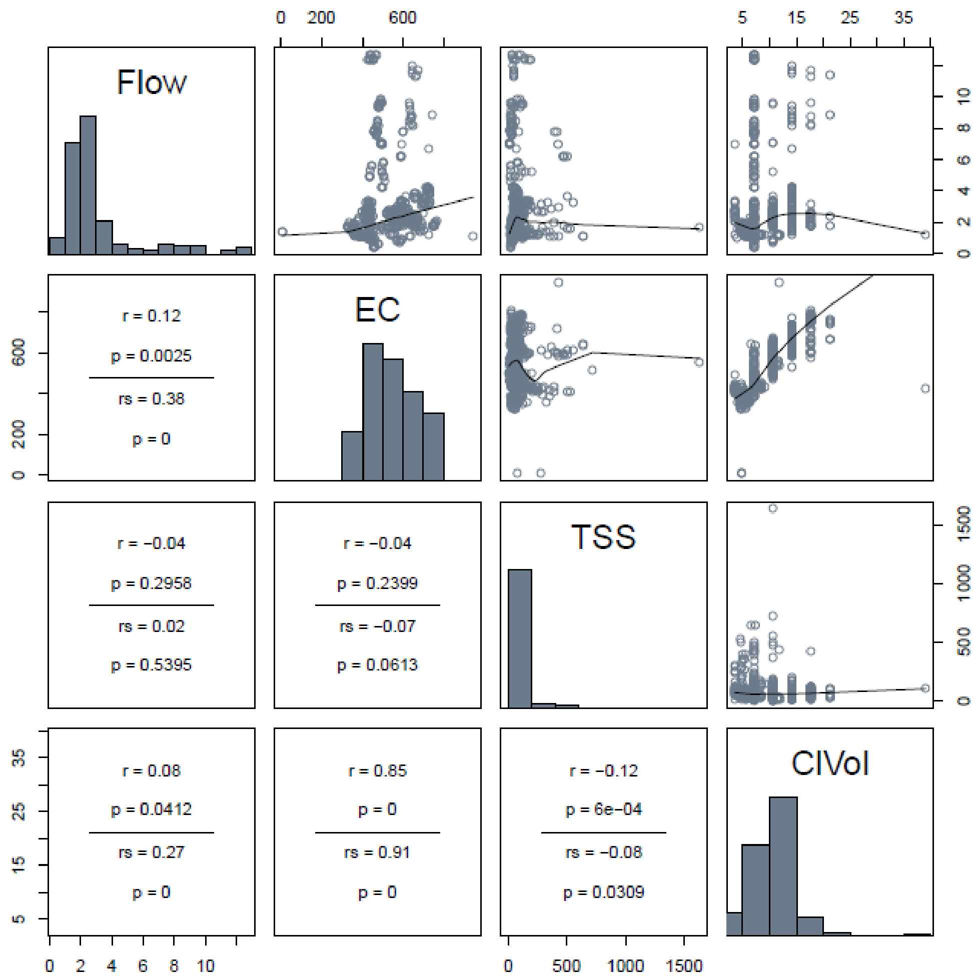


Figure A2-4d. Multiplot matrices generated in R, showing histograms along the diagonal axis, X:Y scatterplots and results of both Pearson (r) and Spearman (rs) correlation analysis.

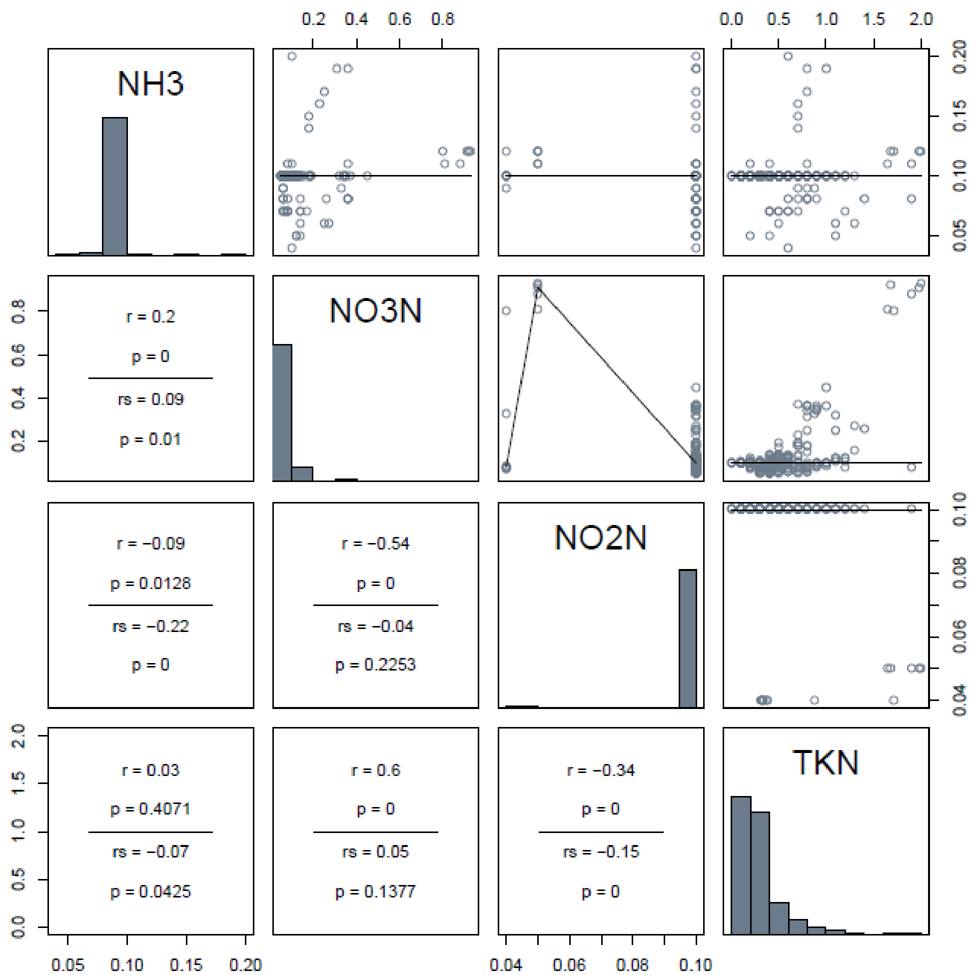


Figure A2-4e. Multiplot matrices generated in R, showing histograms along the diagonal axis, X:Y scatterplots and results of both Pearson (r) and Spearman (rs) correlation analysis.

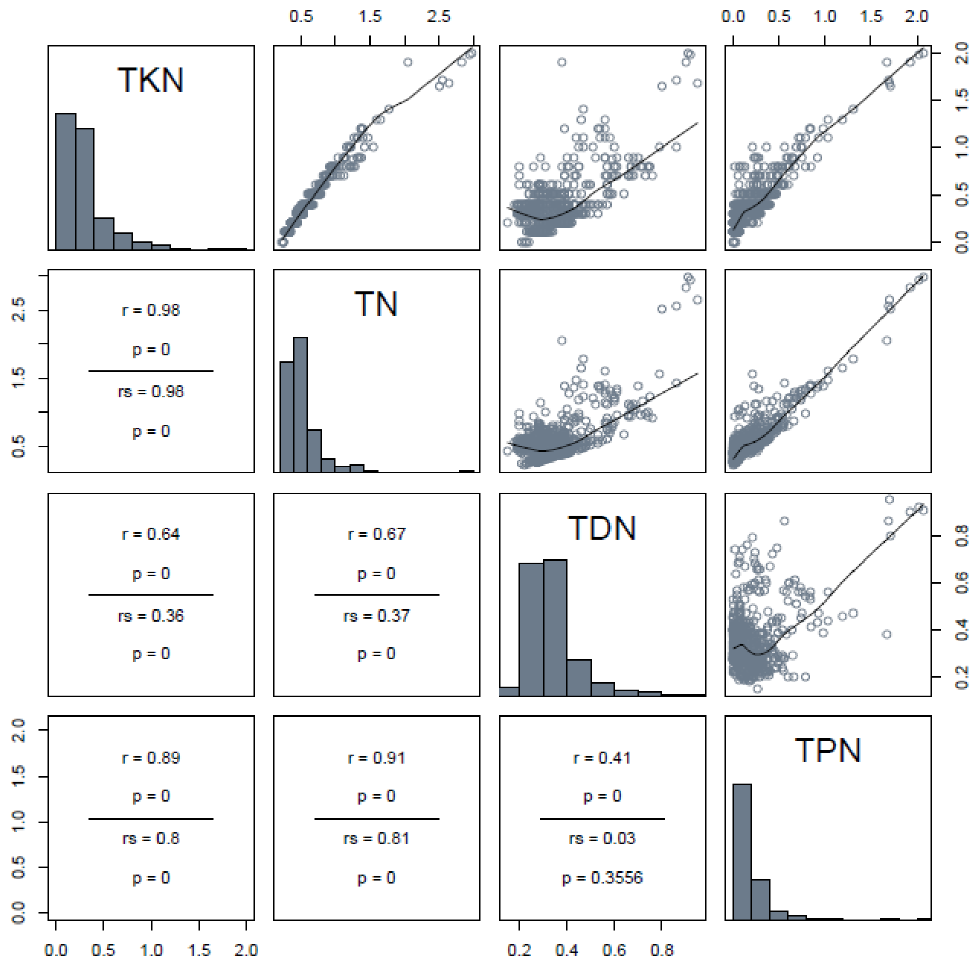


Figure A2-4f. Multiplot matrices generated in R, showing histograms along the diagonal axis, X:Y scatterplots and results of both Pearson (r) and Spearman (rs) correlation analysis.

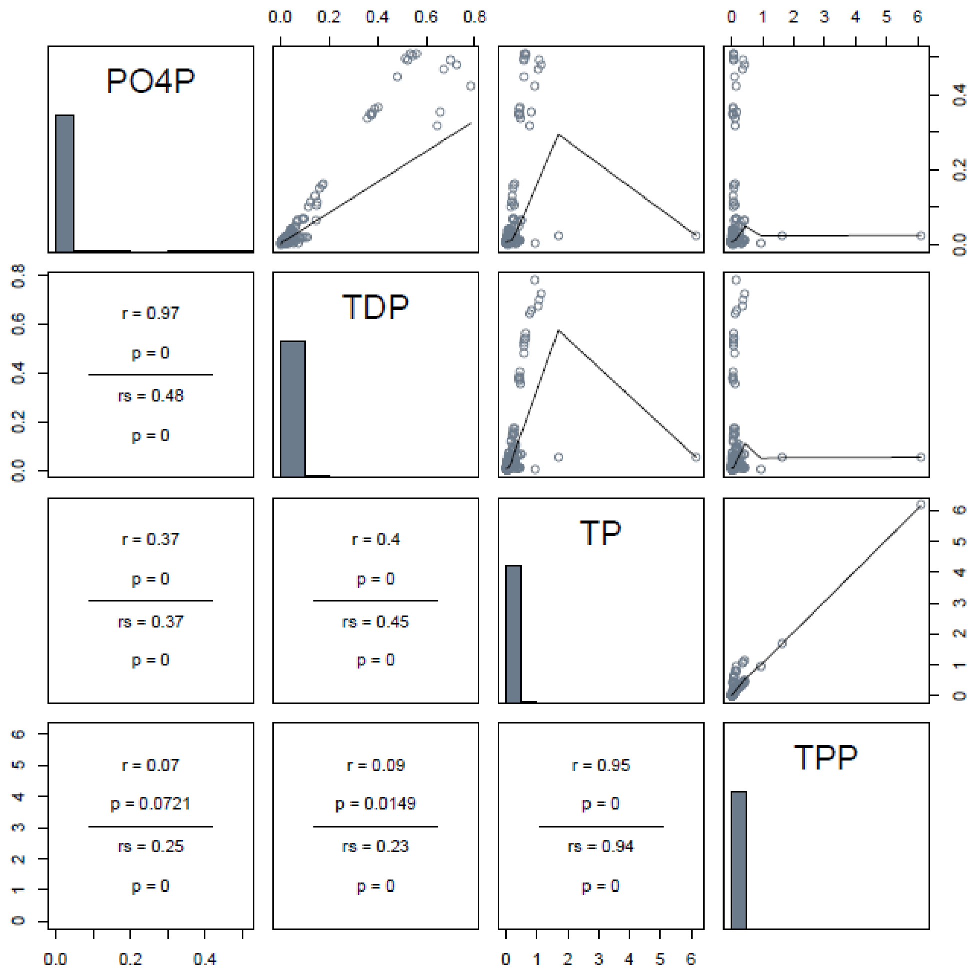


Figure A2-4g. Multiplot matrices generated in R, showing histograms along the diagonal axis, X:Y scatterplots and results of both Pearson (r) and Spearman (rs) correlation analysis.

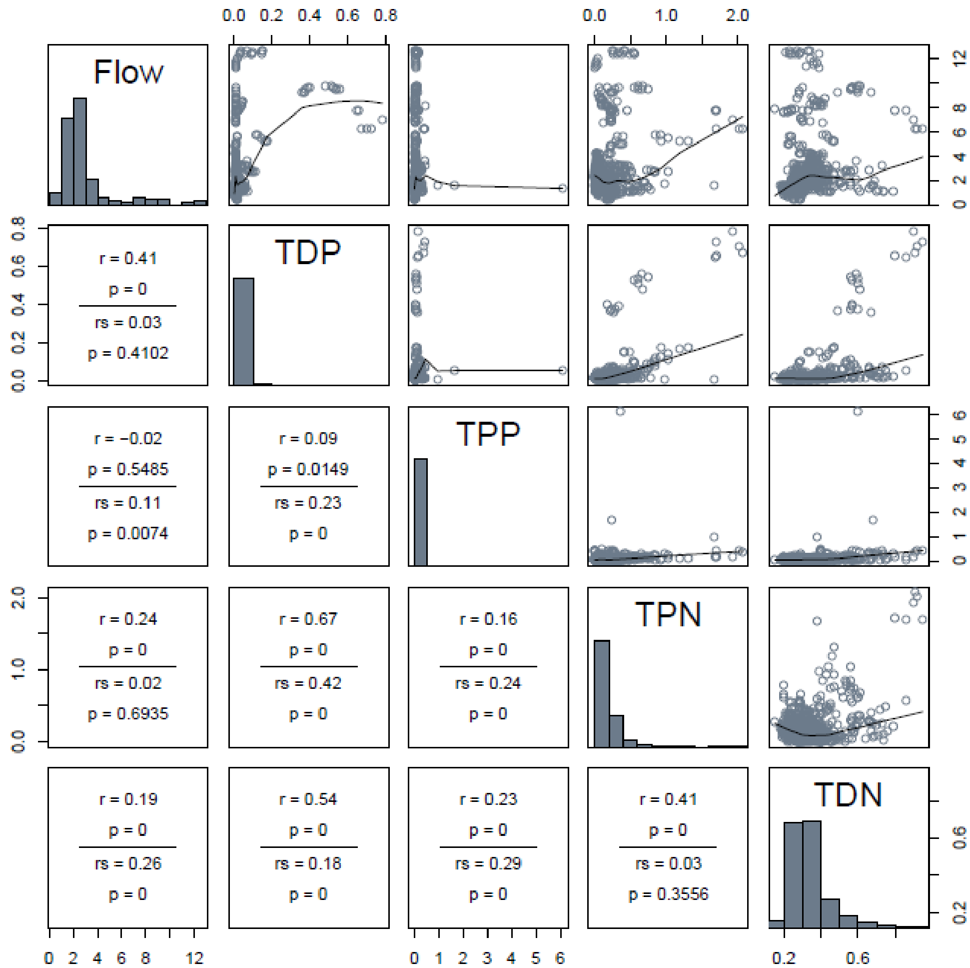


Figure A2-4h. Multiplot matrices generated in R, showing histograms along the diagonal axis, X:Y scatterplots and results of both Pearson (r) and Spearman (rs) correlation analysis.

Table A2-1. Method of chemical analyses and detection limits used for water samples (from Miller et al 2008, Chapter 1).

Chemical	**Method**	**Year** (Apr-Mar)	**Instrument**	**Technicon Method**	**ASTM Method**	**ENVIRODAT Method Code**	**ENVIRODAT VMV Code**	**Detection limit** (mg L^{-1})
Cl	Mercuric thiocyanate	2004-07	TRAACS 800	783-86T				4.0
NH_4-N	Phenate	2004-07	TRAACS 800	780-86T				0.1
NO_2-N	Diazotization	2004-07	TRAACS 800	784-86T				0.1
NO_3-N	Hydrazine Reduction	2004-07	TRAACS 800	782-86T				0.1
DRP	Ascorbic acid reduction	2004-07	TRAACS 800	781-86T				0.002 (2004-05) 0.01 (2006-07)
TKN	Kjeldahl digestion with K_2SO_4, sodium salicyate method	2004-05	TRAACS 800	786-86T				0.04
TN	Pyrolysis and chemiluminescence detection	2006-07	Shimadzu TOC-V with TNM-1		D5176			0.2
TDN	Pyrolysis and chemiluminescence detection	2006-07	Shimadzu TOC-V with TNM-1		D5176			0.2
TP	Kjeldahl digestion, ascorbic acid reduction	2004-05	TRAACS 800	781-86T (A)		582 (D)	15421 (D)	0.002
TDP	Kjeldahl digestion, ascorbic acid reduction	2004-05	TRAACS 800	781-86T (A)		582 (D)	15421 (D)	0.002
TP	Persulfate and autoclave digestion, ascorbic acid reduction	2006-07	TRAACS 800	787-86T (A)		2331 (D)	15423 (D)	0.01
TDP	Persulfate and autoclave digestion, ascorbic acid reduction	2006-07	TRAACS 800	787-86T (A)		2332 (D)	15465 (D)	0.01

A=analysis method, D=digestion method

Table A2-2. Site score components for Water Quality Index Scores generated for sites by year, Lower Little Bow River, Alberta (2004-2007).

Station	Index Period	CCME WQI	F_1	F_2	F_3	Sum of Failed Tests	Norm. Sum of Excursions (nse)	Number of Samples	Total Number Variables	Number of Tests	Number of Failed tests	Number of Passed Tests
LB4	2004	44.9	76.9	32.6	46.0	276.55	0.85	27	13	325	106	219
	2005	50.6	61.5	28.2	52.4	484.81	1.10	37	13	440	124	316
	2006	56.2	61.5	25.7	36.1	235.46	0.57	35	13	416	107	309
	2007	57.5	61.5	24.4	32.2	204.09	0.47	36	13	430	105	325
LB4-14	2004	46.0	61.5	34.5	61.5	519.70	1.60	27	13	325	112	213
	2005	50.6	61.5	28.0	52.4	487.72	1.10	37	13	443	124	319
	2006	56.4	61.5	26.4	34.8	222.28	0.53	35	13	417	110	307
	2007	57.4	61.5	24.5	32.3	206.18	0.48	36	13	432	106	326
LBW1	2004	52.3	61.5	36.1	41.7	231.92	0.72	27	13	324	117	207
	2005	48.1	61.5	27.3	59.5	652.18	1.47	37	13	443	121	322
	2006	54.7	69.2	25.7	26.6	151.04	0.36	35	13	416	107	309
	2007	56.9	61.5	24.5	34.3	225.11	0.52	36	13	432	106	326
LBW2	2004	59.0	53.8	29.7	35.7	80.34	0.55	12	13	145	43	102
	2005	52.4	61.5	27.0	47.6	399.49	0.91	37	13	440	119	321
	2006	58.2	61.5	25.0	28.7	167.24	0.40	35	13	416	104	312
	2007	56.9	61.5	24.5	34.3	225.96	0.52	36	13	432	106	326
LBW3	2004	60.5	53.8	28.3	31.2	65.90	0.45	12	13	145	41	104
	2005	52.8	61.5	26.6	46.9	387.86	0.88	37	13	440	117	323
	2006	58.5	61.5	25.7	26.8	152.38	0.37	35	13	416	107	309
	2007	56.7	61.5	25.0	34.8	230.20	0.53	36	13	432	108	324
LBW4	2004	69.3	38.5	27.6	24.3	27.91	0.32	7	13	87	24	63
	2005	51.3	61.5	26.7	51.0	460.55	1.04	37	13	442	118	324
	2006	57.7	61.5	25.4	30.7	185.52	0.44	35	13	418	106	312
	2007	57.6	61.5	24.3	31.9	202.04	0.47	36	13	432	105	327

Table A2-3. Trend analysis of annual and seasonal water quality index (WQI) scores from the Lower Little Bow River, 2004-2007.

WQI	Site	Mann-Kendall Score	Var(score)	Tau	p-value
Annual	LB4	6	8.667	1	0.0894
	LB4-14	6	8.667	1	0.0894
	LBW1	4	8.667	0.667	0.3082
	LBW2	-2	8.667	-0.333	0.7341
	LBW3	-2	8.667	-0.333	0.7341
	LBW4	-2	8.667	-0.333	0.7341
	Mean	4	8.667	0.667	0.3082
Spring	LB4	2	8.667	0.333	0.7341
	LB4-14	2	8.667	0.333	0.7341
	LBW1	2	8.667	0.333	0.7341
	LBW2	-1	3.667	-0.333	1
	LBW3	-1	3.667	-0.333	1
	LBW4	-1	3.667	-0.333	1
	Mean	2	8.667	0.333	0.7341
Summer	LB4	0	8.667	0	1
	LB4-14	2	8.667	0.333	0.7341
	LBW1	2	8.667	0.333	0.7341
	LBW2	2	8.667	0.333	0.7341
	LBW3	0	8.667	0	1
	LBW4	1	3.667	0.333	1
	Mean	2	8.667	0.333	0.7341
Fall	LB4	3	7.667	0.548	0.4701
	LB4-14	3	7.667	0.548	0.4701
	LBW1	2	8.667	0.333	0.7341
	LBW2	2	8.667	0.333	0.7341
	LBW3	2	8.667	0.333	0.7341
	LBW4	2	8.667	0.333	0.7341
	Mean	2	8.667	0.333	0.7341
Winter[1]	-	-	-	-	-

[1] Winter excluded from trend analysis as number of seasonal scores was <3

Table A2-4.Wilcoxon signed-rank test results for Lower Little Bow River water quality index (WQI) scores by reaches, for seasonal and sub-index scores from 2004-2007.

				Confidence Interval[1]			
Index	**Location**	**V**	**p-value**	**Lower**	**Upper**	**Sample Estimates: (pseudo)median**	**% CL**
ANNUAL	Reach 5	4	0.875	-2.5	6.3	-0.8	88%
	Reach 4	0	0.125	-22.3	-0.2	-1.15	88%
	Reach 3	6	0.875	-1.1	10.3	0.4	88%
	Reach 2	1	0.25	-1.5	0.2	-0.375	88%
	Reach 1	9	0.25	-0.8	15.6	2.275	88%
SPRING	Reach 5	3	1	-3.2	8	0.45	75%
	Reach 4	3	1	-3.9	2.4	-0.325	75%
	Reach 3	3	1	-3.8	3.7	0.075	75%
	Reach 2	1	0.5	-1.1	0.2	-0.725	75%
	Reach 1	6	0.25	0.7	5.8	2.375	75%
SUMMER	Reach 5	3	1	-1.2	3.2	0.4	75%
	Reach 4	5	0.5	-0.3	2.4	0.925	75%
	Reach 3	0	0.25	-5.1	-0.5	-2.5	75%
	Reach 2	4	0.75	-0.3	0.4	0.125	75%
	Reach 1	5	0.5	-0.1	4	1.725	75%
FALL	Reach 5	3	0.625	-9.8	0.7	-2.875	88%
	Reach 4	0	0.125	-6.2	-0.3	-1	88%
	Reach 3	10	0.125	0.1	6.1	0.65	88%
	Reach 2	3	1	-1.9	6.5	1.411	60%
	Reach 1	3	0.625	-6.6	1	-0.925	88%
BI	Reach 5	4	0.875	-4.8	10.7	-1.1	88%
	Reach 4	0	0.125	-13	-0.6	-1.075	88%
	Reach 3	6	0.875	-1.6	1.5	0.525	88%
	Reach 2	2	0.789	-0.8	0.2	-0.216	60%
	Reach 1	7	0.625	-1.5	16.1	0.425	88%
PI	Reach 5	2	0.789	-2.5	2.0.	-0.726	60%
	Reach 4	1	0.423	-1.9	0.6	-0.914	60%
	Reach 3	1	0.197	-2.1	-0.05	-0.200	80%
	Reach 2	2	0.375	-1.5	0.2	-0.325	88%
	Reach 1	8.5	0.269	-0.1	1.3	0.6	80%
CI	Reach 5	4	0.875	-4.8	10.7	-1.1	88%
	Reach 4	0	0.125	-13	-0.6	-1.075	88%
	Reach 3	6	0.875	-1.6	1.5	0.525	88%
	Reach 2	2	0.789	-0.8	0.2	-0.216	60%
	Reach 1	7	0.625	-1.5	16.1	0.425	88%

3 FISH COMMUNITY ASSESSMENT OF THE LOWER LITTLE BOW RIVER

3.1 Introduction

Evaluation of the health of aquatic ecosystems requires an understanding of the structure and function of floral and faunal aquatic communities. The concept of biological integrity of ecosystems was defined by Karr and Dudley (1981) as “the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity and functional organization comparable to that of natural habitat of the region”(p.56). In practice, ecological health studies of riverine systems commonly use measures of community or population level indicators and evaluate riparian vegetation, aquatic invertebrate, and algal and fish assemblages. Multimetric fish-based indices of aquatic ecosystem health, such as the Index of Biotic Integrity (IBI), first formulated by Karr (1981), have been developed for a wide array of lotic systems (Fausch et al. 1984, Simon 1991, Hughes et al. 1998, Angermeier et al. 2000, Bramblett et al. 2005) and recently for regions in Alberta (Stevens et al. 2006, Stevens and Council 2008).

Many river and stream ecosystems in Alberta have been altered through anthropogenic activities that divert or alter channel structure, adjust flow regimes, and/or modify biotic interactions between upland, riparian, and aquatic environments. Bioassessment utilizes relationships between land use changes and biological characteristics of the aquatic ecosystems. Fish species within a study watercourse, specifically their relative abundance and guild composition, for example, can tell us about the watershed’s biological function. As top-level consumers within stream and river food webs, fish can be useful indicators of cumulative changes that affect ecosystem function at lower trophic levels, such as alterations to nutrient cycling (Karr et al. 1986). The use of the fish community as a group of indicators responsive to local water quality and land use changes can, however, be confounded by fish characteristics such as their patterns of movement, often patchy distribution associated with niche habitats, and species interactions (Ricker 1975). Fish assemblages in small lakes and streams in many regions of Alberta are also comprised of few species (<5 species) (Nelson and Paetz 1992, Scrimgeour et al. 2002, 2003, Tonn et al. 2003). This low species richness (i.e., number of species) in Alberta prairie and grassland systems may limit the use of fish as indicator taxa and reduce the number of metrics available to characterize watershed biological integrity (Stevens et al. 2006). Despite these challenges, fish remain fundamental to many studies of watershed health.

Fish provide helpful information for the evaluation of water quality. Fish species differ in their physical tolerances for water quality; tolerance ranges form the basis for many aquatic health criteria used in water quality indices (CCME 2001). Fish present in prairie and grasslands watercourses appear to be dominated by species that are habitat generalists adapted to variable flow regimes, warm summer waters, and high turbidity, and that are more tolerant to changes in flow, temperature, and sediment than assemblages found in more stable systems (Dodds et al. 2004, Bramblett et al. 2005). Even though these fish may lack the sensitivities typically sought in indicators of water quality, their habitat requirements can aid in the review of appropriate reference conditions and objective values in questions of site suitability of a particular water quality index.

The primary objective of this study was to describe the fish community within the lower Little Bow River as a complementary data set for ongoing water quality, riparian health, and aquatic invertebrate studies conducted as part of Agriculture and Agri-food Canada’s Watershed Evaluation of Beneficial Management Practices (WEBs) study. Potential associations between the fish community observed and cattle use of riparian zones within the WEBs study watershed were also evaluated.

3.1.1 Study Area

The Little Bow River drains an overall 55,664-ha watershed in southwestern Alberta. Headwaters are located around High River, Alberta, where flows are diverted into the Little Bow River from the Highwood River. The Little Bow River then flows through two reservoirs (Twin Valley Dam and Reservoir, Travers Reservoir) to join the Oldman River north of Lethbridge (Figure 3-1). WEBs studies a micro-watershed on the lower Little Bow River (50°00’03”N, 112°37’03”W) extending 5.5 km along the mainstem and draining an area of 2,565 hectares (Figure 3-2).

The Little Bow River is located within the Prairie ecozone, with the study area falling within the Mixedgrass natural subregion, on dominant dark brown Chernozemic soils. The region is semi-arid (evaporation > precipitation) and exposed to strong chinook winds (dry down-slope winds off the eastern slopes of the Rocky Mountains). Average annual precipitation is about 386 mm, with approximately one-third falling as snow (AAFC 2008).

Water management throughout the basin has focused historically on flow diversion and storage to provide water for domestic, municipal, and agricultural water users. The Travers Dam was constructed in 1954 for the purpose of water storage for the Bow River Irrigation District (Prepas and Mitchell 1990). In the 1970s, irrigated farming increased within the Little Bow River basin; this combined with periods of low flow and regional drought increased water withdrawals and resulted in critical low flows, fish kills, and water quality concerns. To address these issues, additional reservoirs and diversions were constructed on the Highwood River and the upper Little Bow River watershed (CEAA 2003). Flows in the lower Little Bow River are managed discharges from these irrigation reservoirs.

Present land use within the study area is agricultural, with activities including cow-calf operations on native range, dryland farming, intensive irrigated row crop farming, and intensive livestock operations. Land use varies along the channel, with reaches of the watershed subject to management regimes designed to test the effectiveness of agricultural beneficial management practices, including cattle exclusion fencing. Water quality within these study reaches has been monitored under AAFC's WEBs program since 2004 (AAFC 2008). Previous water quality studies were completed within the watershed under the Oldman River Basin Water Quality Initiative and through monitoring by Alberta Agriculture and Rural Development (AARD). Irrigation return runoff from fall irrigation is a particular concern to water quality within the watershed, and nutrients from manure and fertilizers and bacteria from manure are believed to be impacting water quality (AAFC 2008). Fisheries information for the lower Little Bow River is limited to surveys of reservoirs and upper portions of the watershed (Townsend Environmental Consulting 2003; Terry Clayton, personal communication, June 12, 2009).

3.2 Materials and Methods

3.2.1 Fish Sampling

We sampled fishes and documented channel habitat characteristics within the study area during field surveys on October 5-7, 2009 and October 15-17, 2009. Sampling occurred in the fall to avoid impact to spawning periods of expected fish species and when lower flow conditions were anticipated to increase sampling effectiveness. Our sampling design utilized reach divisions previously established through the WEBs water quality monitoring stations (Miller et al. 2010), with the 5.5 km section of the Little Bow River passing through the study watershed subdivided into five reaches, Reach 1 or R1 (at the downstream limit of the study area) to Reach 5 or R5 (at the upstream limit of the study area) (Figure 3-2). Each reach differed in upland and riparian land use (Table 3-1) and reach breaks (i.e., the division points between reaches) included culvert crossings at four of five reach divisions.

We began sampling at the downstream end of the study area (R1) and worked sequentially upstream. Within each reach, we subsampled at three stations. Station selection was not fully randomized due to requirements for safety: prior to surveying, we used a study area map of the river (1:400 scale) to locate proposed sample locations along the channel at an approximate spacing of 300 m. At each proposed location, wetted channel depth and width were assessed in the field and those sites with depth >1 m or width >10 m were rejected because of safe gear use limitations. The channel was surveyed upstream of the proposed location and the station relocated at the first point where depth and width permitted sampling. The length of channel sampled was adjusted, given the channel width, to standardize the area sampled in each net plot to approximately 300 m^2.

We noted physical habitat characteristics (e.g., dominant substrate type, bank shape, range of water depth, instream cover, temperature), visual indicators of livestock use (e.g., hoof prints, manure, grazed vegetation) and collected fishes at each station. Enclosed net plots spanning the channel were constructed at each station with two seine nets installed across the channel at downstream and upstream ends of the station. Seines used were white nylon panels of 47 mm mesh, 1.83 m tall and 9.14 m long, anchored to the channel bed with lead line and to channel banks with 2.27 kg weights and wooden stakes. A float line held the upper edge of the net at the water line.

Gear selectivity for size or age classes of fish can bias the sample and environmental conditions (e.g., turbidity, flow, depth) can affect gear efficiencies (Ricker 1975). To address these issues, we used several kinds of sampling apparatus. We used seining and electrofishing methods at every station and a third method, minnow trapping, at one station per reach. Once the net plot was installed, we sampled fish within the enclosure by pulling a seine net (47 mm mesh, 1.83 x 9.14 m) through the plot, by electrofishing in one pass using a backpack electrofisher (Model LR-24, Smith Root Inc., standard pulse DC at 90-130 V, 30-40 Hz, 12% duty cycle) and by recovering fish captured on the downstream seine stop net. We also collected, identified, and separately enumerated additional fish from outside the enclosure that were captured on the upstream seine net. At one station per reach, we set six minnow traps baited with fish-based cat food overnight within the open channel.

Fish collected were maintained in temporary holding tanks filled with native water and oxygenated using a battery-powered portable air pump. Fish were identified, enumerated, measured, and released. The length measurement used was fork length, or the distance from the tip of a fish's snout to the fork of its tail fin. For confirmation of field identification, fish were photographed in the field. Detailed meristic characters (e.g., fin ray and lateral line scale counts) were extracted from photographs and identifications were confirmed using keys for Catostomidae and Cyprinidae (Nelson and Paetz 1992) and using reference samples from the University of Alberta Museum of Zoology's Ichthyology Collection. Field collections were completed under regulatory approvals from Alberta Sustainable Resource Development (Fish Research Licence 09-2426 FL) and the University of Alberta Animal Care & Use Committee for Bioscience (Animal Use Protocol 701907).

3.3 Data Analyses

Our description of fish community composition used both structural (taxonomic) and functional (ecological) measures of diversity. Counts of individual fish (abundance) were generated for each station and relative abundance (percentage of total catch represented by the species) was calculated for each study reach. To estimate species richness, species counts were tallied for each station and aggregated to a total species count for each reach. The frequency of site occurrence for each species (V, %) was calculated according to the formula:

$$V = a / A \times 100\%, \qquad (3\text{-}1)$$

where a is the number of stations when some particular species was caught; and A is the total number of all stations sampled during the study period.

While many studies of stream fish communities rely on time-based or linear distance measures to standardize sampling effort, we utilized area-based catch per unit effort values based on our fishing to depletion within our stream net enclosures. Fish density was calculated by dividing the total number of fish collected within each net enclosure (using seining and electrofishing techniques) by the wetted stream area within the enclosure. To evaluate effectiveness of the different sampling methods used, electrofishing catches were converted to catch-per-unit-effort values of the number of fish collected per 100 seconds of fishing effort to standardize for differences in fishing time between stations. Minnow trapping effort was equal across reaches and catches were presented as catch per set of 6 traps set overnight.

We used several common diversity indices to assess the pattern of species abundance. Structure of the fish community was expressed through estimated species richness (S) and species diversity indices including Shannon-Weiner's species diversity index, H' (Shannon and Weaver 1949); and Simpson's evenness measure ($E_{1/D}$), an evenness index using the reciprocal variation of Simpson's diversity index that is less influenced by species richness (Smith and Wilson 1996):

$$H' = - \Sigma p_i \ln(p_i) \, , \qquad (3\text{-}2)$$

where p_i is the share of *i*-species in the abundance of all the caught species; and

$$E_{1/D} = \frac{1/D}{S} \qquad (3\text{-}3)$$

where D is the sum of squared species proportions ($D = \Sigma p_i^2$) and S is the total number of all caught species.

Species richness, the number of species within an assemblage, is considered to be one of the most powerful parameters in determining stream condition because a direct correlation exists between high-quality resources and the number of fish species in warm water assemblages (Simon 1991). It is a common metric used in bioassessment as it consistently relates to site quality (Angermeier et al. 2000). An assemblage with greater species richness is assumed to indicate a lower level of impact as a richer species assemblage is more likely to include those specialized species that are more sensitive to degradation or change in water quality. Evenness, another dimension of diversity, describes the distribution and abundance of individuals among species in an area.

Evenness measures compare observed diversity to a theoretical maximum diversity (Pielou 1975). When all species are equally abundant, evenness equals 1; the greater the differences in abundance across species, the smaller the evenness value, which approaches zero. In degraded environments, tolerant species dominate; as their relative abundance increases, evenness is reduced and total diversity declines even though species remain present in the community.

Proportional abundance indices, like the Shannon-Wiener index or the Simpson's index of diversity, relate both richness and evenness components of diversity within one value. The Shannon-Wiener index can be interpreted as describing uncertainty, and therefore diversity: if an individual is picked at random from an infinite population, H' is a measure of how uncertain one is that the individual picked will be of a particular species – and this uncertainty will increase with greater diversity (Pielou 1969, DeJong 1975). Simpson's index of diversity represents the probability that two individuals randomly selected from a sample will belong to different species.

We compared diversity across reaches using Morisita's similarity index (CqN). Calculations of index values utilized SPADE (Chao and Shen 2009), a software tool that estimates a class of generalized Morisita similarity/dissimilarity indices for more than two communities (in our case, reaches) and also presents pair-wise comparisons between reaches. The Morisita similarity index used by SPADE, CqN, is a similarity measure comparing N communities based on species information shared by q communities (Chao et al. 2008). Approximate variances were obtained by bootstrapping using 1000 repetitions.

Using natural breaks in histograms of fork length distributions of captured fish and length-age information from the literature, we assigned individual fish to either young-of-year (YOY) or ≥ 1 yr categories. Lake chub < 75 mm, white sucker < 100 mm (Stevens and Council 2008), spottail shiner < 60 mm (OME 2005), river shiner < 50 mm (Becker 1983; Nelson and Paetz 1992), emerald shiner < 47 mm (Fuchs 1967; Campbell and MacCrimmon 2006), and longnose dace < 50 mm (Jefferies *et al.* 2008) were considered YOY. Larger individuals were considered ≥ 1 yr and additionally were identified as adult if length corresponded with length at maturity information available (Nelson and Paetz 1992; Froese and Pauly 2008). Once identified by length, we counted the number of individuals >1 yr for each reach. We also characterized collected individuals by species traits including trophic class, substrate preference, reproductive strategy and locomotion morphology (Goldstein and Simon 1999, Simon 1999, Goldstein and Meador 2005), following work done by Stevens et al. (2006) and Stevens and Council (2008) in their evaluation and design of IBIs for use in Alberta.

Stevens *et al.* (2006) developed an IBI for small grassland streams in the Red Deer River basin using five metrics for bioassessment: 1) % ≥ 1 yr fish, 2) occurrence of white sucker, 3) number of ≥ 1 yr fathead minnows standardized to catch effort, 4) % DELTs (deformities, sign of disease, eroded fins, lesions, and tumors), and 5) number of young of the year fish standardized to catch effort. In their development of a fish IBI for use in the Battle River watershed in Central Alberta, Stevens and Council (2008) selected three metrics only: species richness and two trophic guild metrics (percent omnivores and percent carnivores) through a process designed to reduce metric redundancy and select an IBI closely linked to land use patterns. Our work evaluated species richness, and relative abundance of omnivores, carnivores, benthic invertivores, insectivorous cyprinids, and ≥ 1 yr fish for potential use in future IBIs designed for the lower Little Bow River, and excluded any measure of fish health as we did not make observations of deformities or disease in our sampling.

Field characteristics recorded at the time of sampling were compiled into a spreadsheet listing physical measures of habitat (e.g., wetted channel depth, channel width, percent coverage of dominant substrate types). Qualitative observations were transformed into simple numeric index values. For example, at each station, we qualitatively estimated livestock intensities on the landscape per stream-side (within 50 m of the channel) using a 4-point scale as i) 0 = ungrazed, ii) 1 = minimally grazed, iii) 2 = moderately grazed, and iv) 3 = intensively grazed, and summed the two stream-side scores from each bank for each site (Stevens et al. 2006).

3.4 Results and Discussion

3.4.1 Total Catch and Relative Abundance of Species

A total of seven fish species were represented by the 971 individuals we collected within the study area (Table 3-2). The number of species collected at a station ranged from 1 to 6 (mean = 3.4), with catches from 10 to 119 fish (mean = 62). When catches were aggregated to the reach level, species richness ranged from 5 to 7, with accumulated species reaching 7 after sampling at 13 stations, totaling 3,609 m^2 of river habitat (Figure 3-3).

Relative abundance values for each species were calculated for each reach using pooled station catches. Spottail shiner (*Notropis hudsonius)* was the most abundant species found at all sampled stations. River shiner ranked as the second-most abundant species at the downstream end of the study area in Reach 1, while white sucker and lake chub shared similar ranks at the upstream end of the study area in Reach 5. Emerald shiner maintained low relative abundance throughout all reaches. Rarer species included northern pike (*Esox lucius*) and longnose dace (*Rhinichthys cataractae*), with one and two individuals collected, respectively. Ranked log-abundance plots generated for each reach are presented in Figure 3-5.

Fish density, a value chosen to standardize catch numbers for the difference in sampled area, was calculated using all individuals sampled from within net enclosures (by seining and electrofishing) and ranged from 1.01 to 27.13 fish per 100 m^2 through the study area (Table 3-3, Figure 3-6). To evaluate the relative contribution of electrofishing to these catches, specific catch per unit effort values representing standardized values for electrofishing alone were also calculated and ranged between 0.00 to 4.80 fish per 100 seconds through the sampled stations. Catch per unit effort values for minnow trapping ranged from 1 fish to 17 fish caught per 6-trap-set.

Fish density is a partial measure of a habitat's productivity, and productivity, most typically expressed as biomass, is often used as an indicator of changing habitat quality and function (Minns et al. 1994). In our case, we were unable to calculate species biomass as data collected were limited to size and not mass; however, density combined with the narrow size ranges observed for several species allows a rough estimate of productivity from density.

Of the five study reaches, Reach 5, the study section of the channel where cattle were excluded with fencing, contained the greatest numbers of these fishes and also individual representatives of two less common species (the longnose dace; and the northern pike, which was only collected in Reach 5 during this sampling period). This resulted in a ranked log abundance curve (a means for visually representing species richness and species evenness) for Reach 5 that appeared to differ from the curves for Reaches 1 through 4. Fish density values, reflecting standardized values for seining sampling effort, were greatest in Reaches 2 and 5, suggesting that these reaches may support more individuals. Density values for minnow trapping reflect similar patterns.

3.4.2 Community Diversity Measures

Species diversity measures including richness, dominance, and evenness indices of pooled reach-level fish community data are presented in Table 3-4. Shannon-Wiener index values varied from a low of 0.50 (in Reaches 3 and 4) to a high of 1.17 (in Reach 5); a similar pattern repeated in the modified Simpson's Index of Diversity and evenness measures. Estimation of Morisita's similarity index across fish communities sampled from the five study reaches, following Chao et al. (2008), was 0.928 (estimated SE=0.012, 95%CI=0.904, 0.953). Similarity index values for pair-wise comparisons between two reaches, rather than between all reaches, are presented in Table 3-5.

Estimated fish species richness values found throughout the study area were similar, with values from 5 to 7 species. Fish communities of the Canadian prairies have similarly low measures of diversity (Chu et al. 2003). Compared to regional watercourses where species assemblages range from < 5 species (Nelson and Paetz 1992, Scrimgeour et al. 2002, 2003; Tonn et al. 2003) to 14 species (Stevens and Council 2008), the lower Little Bow presents an intermediate species richness. Estimated species richness was greatest within Reach 5; however, this difference from other reaches was due to the presence of two species, each represented by one individual only.

Evenness values for reaches of the study area, represented by Simpson's evenness measure (E1/D) range from 0.21 to 0.40. Margalef (1972) provided a general range of index values for the Shannon-Weiner index, based on empirical data, of between 1.5 and 3.5. Observed values for the fish community of the lower Little Bow River (a minimum of 0.50 in Reaches 3 and 4 to a maximum of 1.17 in Reach 5) fall at the lower end of this spectrum. Simpson's index of diversity presents a similar pattern, with highest diversity seen in Reach 5 (0.65) and lowest in Reach 3 (0.20).

Significance of these apparent differences can be interpreted through a comparison of diversities across multiple communities using similarity indices – we utilized a variation of Morisita's index as described by Chao et al. (2008). While the overall estimated similarity of diversity in reaches is high (i.e., the calculated value of 0.928 approaches the maximum similarity of 1), pair-wise comparisons demonstrate a greater dissimilarity (1-

similarity) between Reach 5, the reach treated with riparian fencing and cattle exclusion, and other reaches. Calculated community diversity measures presented a clear pattern, suggesting high diversity of fish in Reach 5 and low diversity values in Reach 3. Of the four diversity indices used, three indicated the highest species diversity in Reach 5.

Modification of physical habitat can lead to temporary or enduring changes in the composition of stream fish assemblages depending on the severity of the disturbance (Reice et al. 1990). Maximum diversity is likely to occur in sites where habitat diversity is enhanced and strong biotic interactions (e.g., predation, competition between species) are mediated by intermediate environmental disturbance (Resh et al. 1988). Greater species diversity in a reach protected from cattle access, therefore, may relate to the maintenance of greater habitat heterogeneity in a river reach protected from bank, streambed, and riparian vegetation disturbance associated with livestock using river channels for drinking water and cooling.

3.4.3 Bioassessment Metrics

Fish within the lower Little Bow River represent a collection of largely generalist minnows and suckers and mesohabitat features appear relatively uniform in their availability throughout the watershed. Within reaches, however, microhabitat availability, species traits, individual strategies, and interactions between members of the larger aquatic community shape the fish assemblage through ecological processes including niche partitioning, predation, and competition. Bioassessment measures calculated for the lower Little Bow River present one tool to evaluate changes in these ecological processes.

Several parameters describing the fish assemblage were selected for their potential future use as metrics or measures of watershed condition, following work by Stevens et al. (2006) and Stevens and Council (2008) in their development of fish index of biological integrity (IBI) approaches for watercourses in south-central Alberta. Raw values generated for indicator parameters (i.e., metrics) to characterize reaches follow in Table 3-6. Species traits (Goldstein and Meador 2005) identified for the fish species found in the lower Little Bow River are presented in Table 3-6.

Possible age class distinctions were apparent in the histograms of fork lengths for river shiner, emerald shiner, and white sucker; histograms for all species were interpreted with age-length relationships drawn from the literature (Figure 3-7). All lake chub (n=73, mean fork length=34.8 mm, SE=1.15) were considered YOY, and all but three measured spottail shiner (n=692, mean fork length=37.8 mm, SE=0.16) were YOY. White suckers, river shiners, and emerald shiners included both YOY and > 1 yr age classes. Of the white suckers caught (n=107, median fork length =56 mm), 6% were 1 yr or greater in age; of the river shiners caught (n=24, median fork length =30 mm), 17% were 1 yr or greater; and of the emerald shiners caught, (n=11, median fork length =57 mm), 64% were 1 yr or greater. The two longnose dace (fork lengths=57 and 75 mm) and the only northern pike (fork length=650 mm) captured were also adult.

Presence of young of the year fish indicates the function of habitat for reproduction of cyprinids and white suckers, while the occurrence of older individuals (based on observed length) in six of the seven species demonstrates that habitat conditions support the persistence and growth of longer-lived species. The metric of longer-lived individuals (i.e., ≥ 1 yr fish) included members of six species and has the potential to serve as a measure of community resilience to changes in system flow regimes as it is expected to increase with the permanence of suitable habitat, connectivity to other populations and decreased levels of anthropogenic disturbance causing mortality (Bramblett et al. 2005, Stevens et al. 2006). Within the lower Little Bow, permanence of habitat is reliant upon the maintenance of minimum instream flows and access to overwintering habitats (e.g., deep pools unlikely to freeze entirely).

The metric for relative abundance of insectivorous cyprinids also seems well suited as an indicator of impacts to the aquatic invertebrate community, given its inclusion of four species found throughout our study area and their range in invertebrate prey preferences and feeding habits. In original work to develop the IBI (Karr et al. 1986; Miller et al. 1988), index scores for the percent individuals as insectivorous cyprinids metric were 5 (> 45%), 3 (20–45), and 1 (< 20%). Scores of 5 represented concordance with reference conditions representing an intact, non-degraded watershed, while scores of 1 indicated impairment. Applying this general scoring method, all of our study reaches would receive a score of 5.

Use of the relative abundance of white suckers metric used by Stevens and Council (2008) in their Alberta IBI was redundant as we had already chosen to use a percent omnivore measure and white sucker formed the only species within that category.

Goldstein et al. (1994) defined an omnivore as a species that consumes significant quantities of both plant and animal materials and has the physiological ability to utilize both. Karr’s IBI used index scores for percent individuals as omnivores of 5 (< 20%), 3 (20–45%), and 1 (> 45%). Applying this scoring method, Reaches 1 through 4 would be scored high (5) while Reach 5 would be scored intermediate (3).

Issues relating to the IBI measures proposed in our work include challenges with dealing with low relative abundance values. Species richness varied from reach to reach due to the presence of single individuals of two species, longnose dace, and northern pike. Many fish (e.g., northern pike and longnose dace) are solitary swimmers while some species (e.g., spottail shiner) are more commonly found in aggregations. Fish surveys may produce 'rare' species as an artifact of sampling techniques and as a result of fish behavior in swimming (Russo 1982). Larger schools are more easily captured in seine netting while solitary swimmers dependent on cover, not from surrounding fish but from habitat structures, are more difficult to both net and electrofish. In our work, the relative abundance of fish species considered carnivores or top predators was limited to northern pike while percent omnivores included only white sucker.

3.4.4 Habitat Characteristics of Stations

Stations exhibited similarities in gross habitat structure, with average channel wetted width ranging from 5.9 to 9.5 m along the moderately meandering, incised channel occupying the coulee valley bottom. Areas of greater width were noted, associated with wide meander bends and modified irrigation withdrawal points at reach breaks between R2-R3 and R3-R4. All reaches were dominated by run morphology, with small areas of deep riffles observed only in stations within the uppermost (R5) and lowermost reaches (R1).

Average water depth at sampling stations varied from 0.45 to 0.85 m. Fluctuation in reach water depth was also evident throughout sampling events. On October 6, 2009, we observed visible changes to water levels while sampling at stations, with water level dropping 0.15 m over a two-hour period at Station 2-3 and wetted depth decreasing 0.47 m overnight on October 6, 2009. During the course of our field sampling, construction projects and infrastructure upgrades on upstream reservoirs resulted in alterations of daily discharges to the lower Little Bow River.

Precipitation events during our field program within the watershed included snowfall on October 7 and 14, 2009. The later snowfall was followed by daytime melting temperatures the next day. While air temperature varied greatly within a typical sampling day (morning temperatures in the range of -4 to -1°C shifting to afternoon temperatures from 3 to 14°C), water temperature varied only slightly, from 3 to 6°C.

While mesohabitat characteristics of the study area (channel width, geomorphology, instream structural complexity) appeared relatively uniform through the five study reaches on our initial reconnaissance surveys, variation was noted during fish sampling (Table 3-8). Substrate characteristics showed a slight change in dominant sediment size through the study reaches, from sands at downstream reaches to small gravels and sands in mid- and upper reaches. Instream habitat features (i.e., boulder coverage, percent cutbank, macrophyte presence) also varied across the stations sampled. Boulder presence was greatest in the upstream end of R4 (downstream of a large highway culvert crossing) and at the upper end of R5, although the lowest reach (R1) also possessed channel sections with abundant boulders and contained the greatest proportion of cutbanks. While aquatic vegetation was noted in seined samples in R2-R5, macrophyte growth was only visible within the channel in R5. Riparian vegetation condition, rated by observed impacts from livestock grazing, ranged from 0 (ungrazed within 50 m of the channel banks) to 5.3 (heavy grazing on both banks), with poorest condition noted in R4.

3.4.5 Reach Comparisons

Observations of variation in fish abundance and density suggest the presence of some difference in the instream habitat function of reaches, particularly Reach 5 and Reach 2; while depth, width and sediment characteristics appeared largely uniform across the study area, riparian and aquatic vegetation, channel banks, and streambed did visually appear less disturbed in Reach 5 compared to the other sampled reaches where cattle access was present. Riparian vegetation along Reach 2 also contained several wetland patches along the stream margins. In stream ecosystems, the major form of environmental variability is fluctuation in stream flow (Jackson et al. 2001) and the presence of flow refugia provided by floodplain wetlands, not explicitly documented in our study, may be an important habitat feature within the study area.

Interestingly, Reaches 5 and 2 similarly ranked as the two lowest impacted reaches in our ranking of observed cattle impacts to riparian vegetation and banks. In Reach 4, the reach presenting the greatest riparian cattle impacts, the range of fish density observed across the three sampled stations was lower than that observed in other reaches. This may be a reflection of reduced habitat heterogeneity throughout Reach 4 compared to the other study reaches, or a result of physical disturbance reducing the densities of fish populations (Jackson et al. 1992). Field observations of turbid and fast-flowing water uniformly through all reaches – conditions limiting effective dipnetting for fish capture with electrofishing – raise some concern with the sampling efficacy of our electrofishing efforts. CPUE values for electrofishing were also high for Reach 2, but were highest in Reach 1, the section of the study area with the greatest proportion of cutbanks, or banks overhanging the wetted channel.

Caution is warranted in defining a causal relationship between the fish community and cattle exclosure; our observations represent a single season sampling program with an unreplicated treatment and we lack information regarding the occurrence of fish movement between reaches. Additional environmental gradients may also have influence on this pattern: for example, microhabitat features may be present within the channel at a resolution lower than that detectible by our habitat assessment protocol. Such challenges in linking fish community responses to alterations of livestock access to riparian and stream habitats have been well discussed by others (Rinne 1988, Sarr 2002, Fisher et al. 2010), and may be partially addressed through more detailed future surveys of reach habitat features (e.g., pool coverage, gravel bars, riparian wetlands), additional surveys of the fish community in both fall and other seasons, and evaluation of fish movement within reaches and across reach breaks.

Recent work by Miller et al. (2010) shows a clear improvement in detailed riparian health measures following streambank fencing on the lower Little Bow River and, given the close functional and structural linkages between riparian and aquatic riverine habitats, it seems reasonable to contemplate a concurrent improvement in aquatic ecosystem health may be seen through increases in fish densities, abundance, and ultimately diversity.

3.4.6 Connecting Biotic and Water Quality Indices

Work to link changes observed in the fish community to environmental stressors, and to link stressors to changes in land use practices, is challenging. The development of IBIs is one approach to describe such linkages between the biotic and abiotic components of a watershed based, in part, on assumptions of interactions and biotic responses that are often drawn from watershed or regional scales. Researchers evaluating agricultural practices in riparian zones have achieved varying results in finding connection between land use change and improvement in the fish community (see Roth et al. 1996, Wichert and Rapport 1998, Stewart et al. 2001, Sarr 2002, Argent and Lenig 2005; Teels et al. 2006, Wang et al. 2006, Blann et al. 2009, and Fisher et al. 2010). In one case, Wang et al. (2006) completed a 13 year study of riparian best management practices and observed physical changes to habitat (i.e., increased substrate size; reduced sediment depth, embeddedness, and bank erosion) and improved overall habitat quality at stations where a natural vegetative buffer existed or streambank fencing was installed on a small warm water stream in Wisconsin, yet could not detect significant improvements in fish communities. The authors speculated that long-standing historical alterations to watershed function and connectivity might limit the ability of a stream fish community to achieve targeted outcomes of improved health (i.e., diversity improvements requiring the addition of fish species that are no longer found in the watershed).

Teels et al. (2006) used IBIs to evaluate the effectiveness of agricultural riparian buffer treatments at 36 treatment and 12 reference sites in Virginia over a shorter time span (2000 to 2003) and found mixed results but positive biotic integrity response at sites treated with riparian buffers when sites were previously highly disturbed and below small, relatively undisturbed watersheds. With their study occurring at the early stage of a recovery trajectory (i.e., riparian vegetation was newly established at the study commencement), the authors comment that additional time may be necessary before restoration improvements can manifest themselves in a way that can be measured by IBIs. Recent work by Fisher et al. (2010) evaluated the effects of riparian buffers on instream habitat, fish assemblage structure, and population characteristics (e.g., growth) on three streams in central Iowa, at both reach and microhabitat scales. The authors found that the fish assemblage structure of the study streams was not highly influenced by the presence of riparian buffers but rather related to instream habitat, which varied greatly across their sampled sites. Insensitivity to riparian buffers, they suggest, may relate to prairie fish species demonstrating innate tolerances for extremes in physico-chemical characteristics of streams (i.e., in sediment delivery, temperature and flow). A functional response was noted, however, with differential growth responses of two fish species with the presence of riparian buffers, indicating that there is some influence of riparian buffers on instream features (e.g., food availability) that affect fish population dynamics. These few examples provide a hint of the complexity and challenge in tracing the pathways of effect from agricultural riparian protection to aquatic communities.

Fish assemblage response to larger landscape processes can also be challenging to interpret. In their work on a central Alberta IBI, Stevens and Council (2008) found a positive relationship between species richness response and the WQI, yet also found high species richness to be related to high concentrations of total Kjedahl nitrogen (TKN) and nitrates/nitrites (NO2+NO3-N) – nutrient components of the WQI that are typically associated with degradation of water quality. Proportion of carnivores was positively related to the WQI, and interestingly, also related to high concentrations of TDP and basin cattle densities, factors also typically associated with degraded water quality. This suggests that the response of species richness and percent carnivore metrics may not be clearly indicative of water quality 'improvements', but influenced by nutrient enrichment.

Stevens and Council (2008) in their work in central Alberta found that percent omnivores was negatively related to the Water Quality Index (WQI), with greater relative abundance of omnivores related to decreased water quality. Dominance of omnivores suggests specific components of the food base are less reliable, increasing the success of more opportunistic species. However, as our omnivore measure is effectively a measure of white sucker abundance, it may also relate positively to habitat conditions because white sucker is a litho-obligate breeder and previous research has demonstrated that this reproductive guild is particularly sensitive to human disturbance (Steedman 1988, Bramblett et al. 2005). Litho-obligate species breed on rock and gravel and have benthic larvae which require interstitial spaces; in highly turbid waters, sediment deposition may limit successful reproduction. This is just one example of challenges posed in the interpretation of a species' sometimes contradictory single traits.

3.5 Conclusions

While we have suggested comparisons between reaches, the proximity of sample stations to one another and the likelihood of uncontrolled biological processes acting across reach boundaries is acknowledged. Longitudinal processes within rivers, by their nature, create environmental gradients and link reach characteristics, including fish species assemblages, resulting in samples that are not explicitly independent. As a result, this works serves primarily as a baseline description of the present fish community assemblage within the lower Little Bow River and provides a point of comparison to future study monitoring habitat alterations through changes in the fish assemblage over time. It is interesting to note, however, that our observed spatial variation in the fish community of the lower Little Bow River indicates that greater fish diversity may be supported by Reach 5, the riparian-fenced river reach.

The development of a complete and locally calibrated IBI for use in monitoring fish community response to environmental impact or, alternatively, to environmental improvement resulting from the application of beneficial management practices, remains a future objective. In many, if not most, bioassessments, calculated raw metric values are indexed to habitat conditions at reference sites and condition scores generated to rank one study watercourse against another and relative to the reference condition. In our preliminary work here, we have generated values for community diversity and function that could form future metrics. Reference condition selection poses a challenge: do we utilize the ‘best’ of our sites as our reference condition, or do we define a reference condition based upon regional fisheries data? Conceptually, metrics applied to degraded sites should score lower than at less impacted sites. Given that the intent of the WEBs program is to evaluate the observed effectiveness of agricultural beneficial land use practices, within the study area it seems flawed to assign a gradient of habitat condition across the study area as would be needed to create metric scores from our measured values. Present understanding of watershed processes, regional drivers of watershed degradation, and land-use impacts to fish habitat suggest several possible impact-response couples between habitat changes and the fish community within the lower Little Bow River.

Observations of future trends in the fish community metrics presented here can form a complementary monitoring tool for the lower Little Bow WEBs program. By evaluating changes in the life history attributes of the fish assemblage through metrics tracking the relative abundance of fish species traits, we can include a ‘fish-eye view” of studied watershed land-use changes. Perhaps we may find natural variation within our system that reduces our power to detect change related to agricultural practices applied within the watershed such as riparian recovery with cattle exclusion, or we may document a robust “prairie fish” community tolerant to changes documented in instream and riparian habitats. By evaluating a key taxonomic receptor of watershed and water quality impacts directly, however, we can position ourselves to make ecologically significant observations of aquatic ecosystem responses to the watershed's environmental changes through time.

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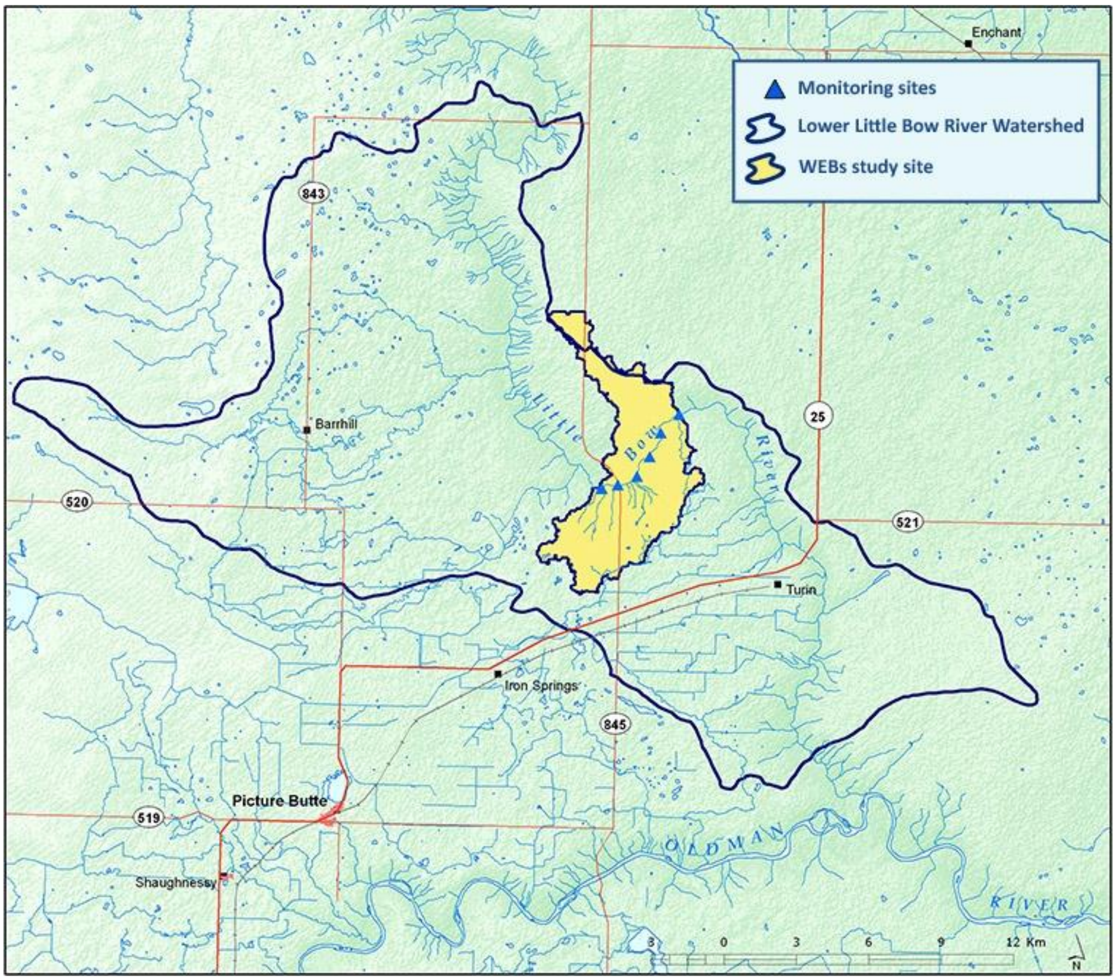

Figure 3-1. Little Bow River watershed, southwestern Alberta (AAFC 2008).

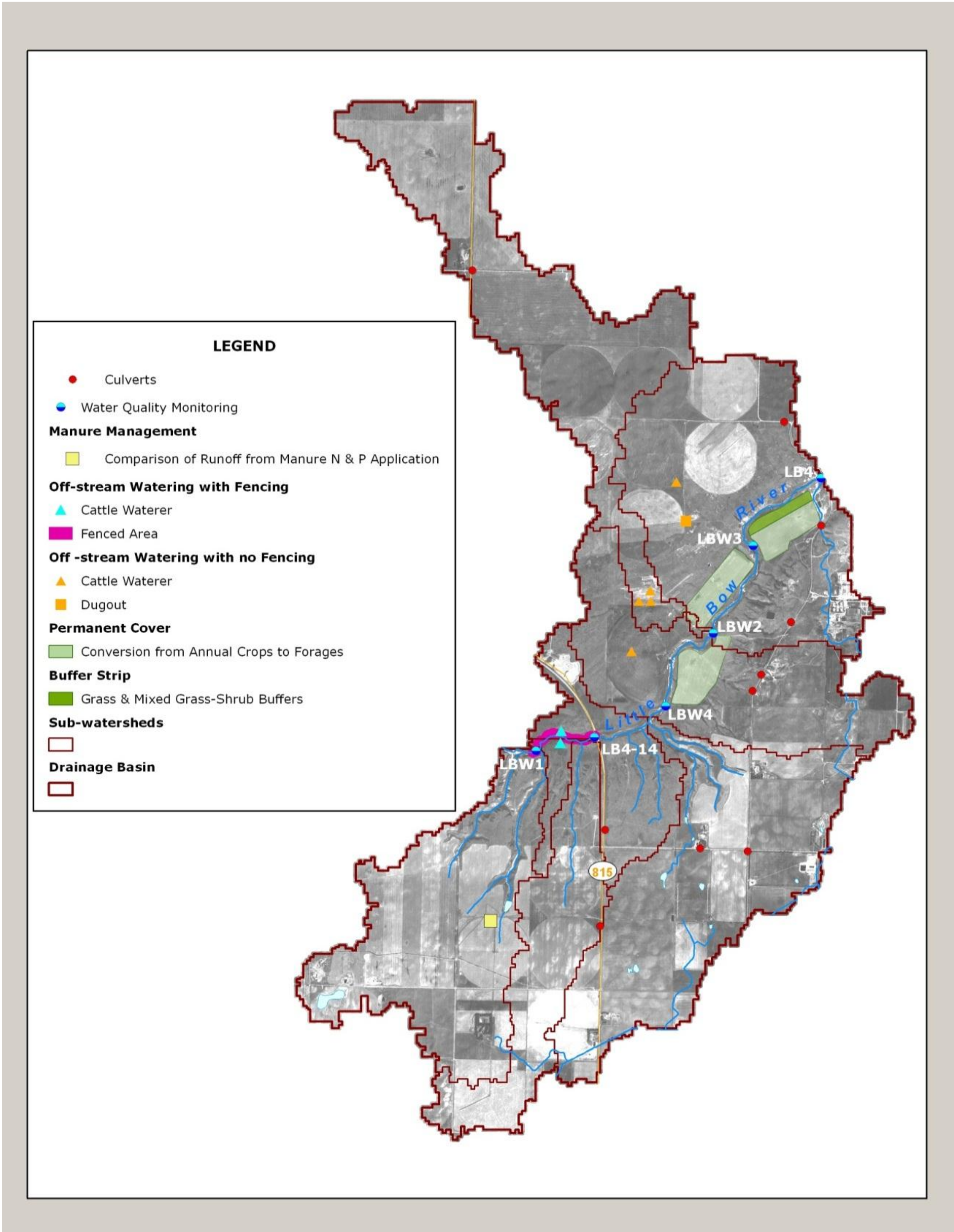

Figure 3-2. Lower Little Bow River study area, with identified monitoring locations and studied riparian BMPs (from Miller et al. 2008). Study reaches extend from LBW1 (Reach 5) downstream to LB4 (Reach 1).

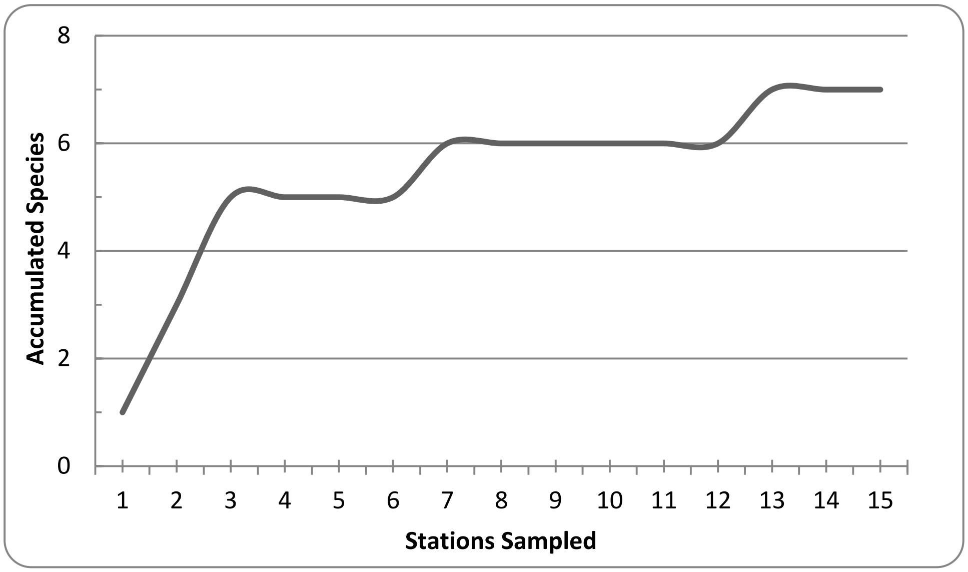

Figure 3-3. Observed species accumulation with sampled area (number of stations) by fish collection on the lower Little Bow River, Alberta, Fall 2009.

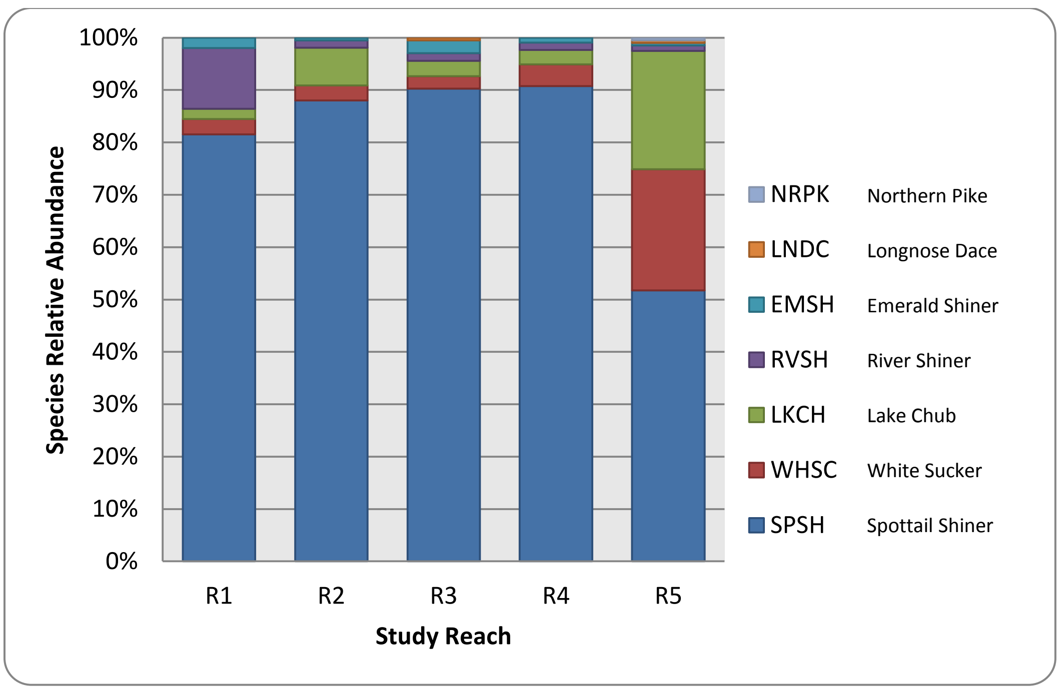

Figure 3-4. Relative abundance of species by reach on the lower Little Bow River, Alberta, Fall 2009.

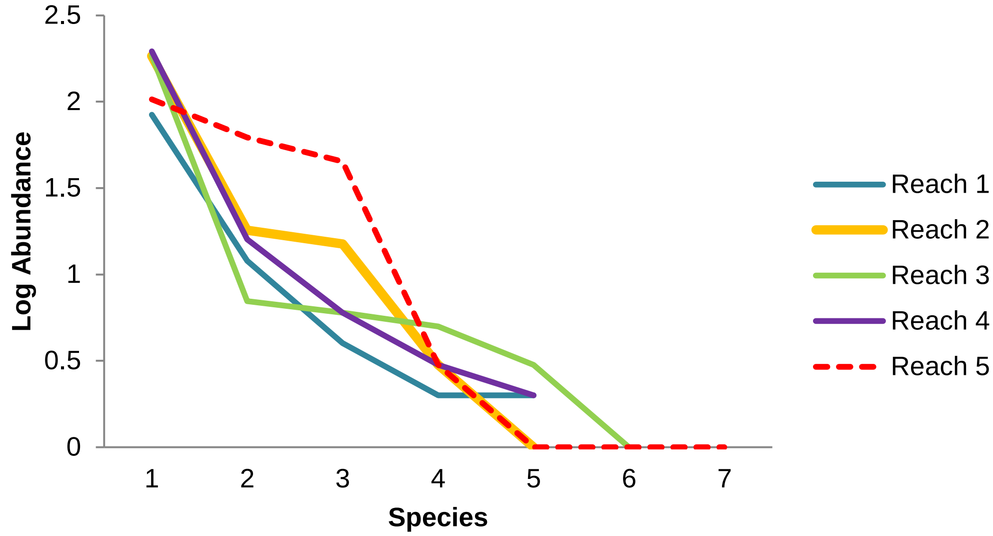

Figure 3-5. Ranked log-abundance curves showing the number of individuals of the different fish species for each reach within the lower Little Bow River study area, Fall 2009.

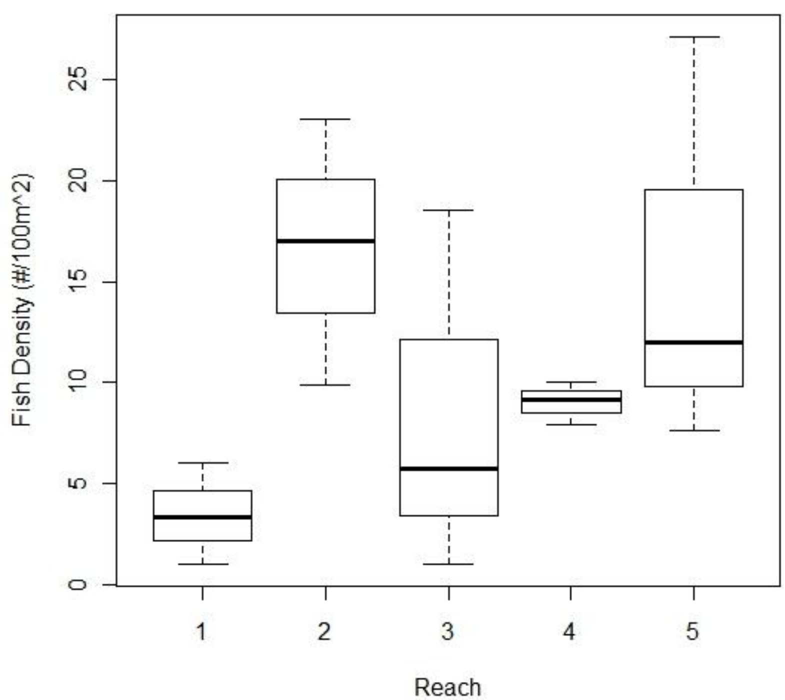

Figure 3-6. Station fish density (total number of individuals per 100m^2 collected within net enclosure) by reach in the lower Little Bow River, Fall 2009.

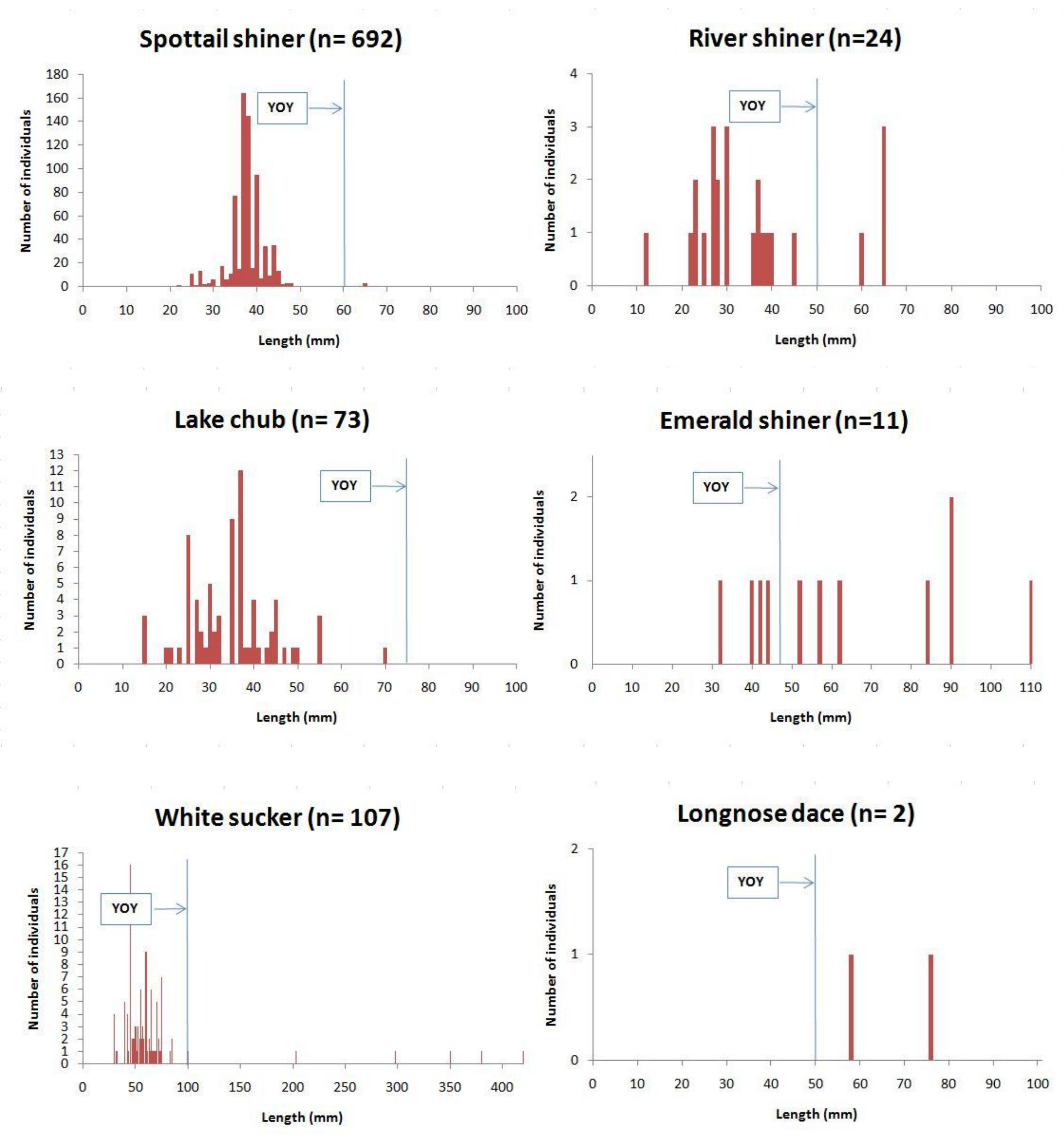

Figure 3-7. Histograms of fork lengths for six of seven fish species caught in the lower Little Bow River, Alberta, across all reaches. No graph is presented for northern pike (n=1, individual fork length = 650 mm).

Table 3-1. Study reaches relative to WEBs water quality stations and land use treatments.

Reach	**Location**	**Distance**[a]	**Water Quality Monitoring Station**	**Riparian Land Use**
R1	Downstream limit of study area	4.2 km	LBW3 to LB4	Field, with buffer strip
R2	-	2.9 km	LBW2 to LBW3	Field, using permanent cover
R3	-	1.8 km	LBW4 to LBW2	Field, with buffer strip
R4	Downstream of Hwy. 845	0.8 km	LB4-14 to LBW4	Livestock access, cattle use
R5	Upstream limit of study area	0 km	LBW1 to LB4-14	Fenced riparian area, livestock exclusion

[a] Distance from the upstream end of the study area at R5 to the upstream start of each reach

Table 3-2. Total number of individuals caught, frequency of site occurrence, and relative abundance within each reach for fish species found during sampling at 15 stations within 5 reaches (R1-R5) on the lower Little Bow River, Alberta.

Family	Species	Common name & Abbreviation		Catch	% Sites Occupied	Relative (%) Abundance R1	R2	R3	R4	R5
Cyprinidae	*Notropis hudsonius*	Spottail shiner	SPSH	752	100	81.6	88.0	90.2	90.7	51.8
	Couesius plumbeus	Lake chub	LKCH	74	60	1.9	7.2	2.9	2.8	22.6
	Notropis blennius	River shiner	RVSH	24	47	11.7	1.4	1.5	1.4	1.0
	Notropis atherinoides	Emerald shiner	EMSH	11	47	1.9	0.5	2.4	0.9	0.5
	Rhinichthys cataractae	Longnose dace	LNDC	2	13	0.0	0.0	0.5	0.0	0.5
Catostomidae	*Catostomus commersonii*	White sucker	WHSC	107	73	2.9	2.9	2.4	4.2	23.1
Esocidae	*Esox lucius*	Northern pike	NRPK	1	7	0.0	0.0	0.0	0.0	0.5

Table 3-3. Standardized catch-per-unit-effort means (maximum, minimum) by reach for fish density, electrofishing, and minnow trapping.

Reach	Fish Density[a]	Electrofishing[b]	Minnow Trapping[c]
R1	3.45 (6.00, 1.01)	1.87 (4.80,0.00)	1
R2	16.65 (23.04, 9.90)	1.71 (3.18,0.00)	12
R3	8.44 (18.53, 1.04)	0.08 (0.23,0.00)	2
R4	9.01 (10.00, 7.91)	0.35 (1.05,0.00)	7
R5	15.60 (27.13, 7.66)	0.90 (1.80, 0.11)	17

[a] Fish density (total number of individuals from net enclosures, per 100 m^2; [b] electrofishing catch per 100 seconds; [c] minnow trapping catch per six-trap-set, duration= 16.5 hr +2.2/-1.0 hr

Table 3-4. Calculated species diversity indices for fish collected within five reaches on the lower Little Bow River, Fall 2009 (S=estimated species richness, H'=Shannon-Weiner diversity index, 1-D=Simpson's Index of Diversity, $E_{1/D}$=Simpson's evenness measure).

Reach	**S**	**H'**	**1-D**	**$E_{1/D}$**
1	5	0.70	0.88	0.30
2	5	0.62	0.30	0.28
3	6	0.50	0.20	0.21
4	5	0.50	0.22	0.26
5	7	1.17	0.65	0.40

Table 3-5. Pair-wise comparisons of diversity between reaches using Morisita similarity values, following Chao et al. (2008) and Chao and Shen (2009).

Similarity Matrix, $C22_{(i,j)}$	**1**	**2**	**3**	**4**	**5**
1	1.000	0.992	0.991	0.991	0.792
2		1.000	0.996	0.999	0.825
3			1.000	0.999	0.768
4				1.000	0.790
5					1.000

Table 3-6. Raw values for fish community parameters of the lower Little Bow River, Fall 2009.

Parameter	**Reach**				
	R1	**R2**	**R3**	**R4**	**R5**
Species richness	5	5	6	5	7
% omnivores	3.8	8.1	3.4	7.2	28.7
% carnivores	0.0	0.0	0.0	0.0	0.5
% benthic invertivores	0.0	0.0	0.5	0.0	0.5
% insectivorous cyprinids	84.6	90.5	95.2	91.5	69.4
% ≥1 yr fish	2.9	0.6	2.9	1.8	4.2

Table 3-7. Species traits for fish found within the lower Little Bow River.

Species	**Trophic Class** [a]	**Substrate Preference** [b]	**Reproductive Guild** [c]	**Geomorphology Preference** [d]	**Locomotion** [d]
White sucker	invertivore-detritivore	rock, gravel	lithophils, broadcast spawner, migratory	riffle, pool	hugger
Spottail shiner	planktivore, invertivore	cobble, sand	broadcast spawner, migratory	pool, run or main channel	cruiser
Lake chub	invertivore-planktivore	gravel	lithopelagophils	pool, run or main channel	cruiser
Longnose dace	invertivore	boulders, gravel	broadcast spawner	riffle	hugger
Emerald shiner	planktivore	sand	broadcast spawner	pool, run or main channel	cruiser
River shiner	invertivore	gravel and sand	broadcast spawner	pool, run or main channel	cruiser
Northern pike	carnivore	variable	broadcast spawner	pool and backwater	accelerator

[a] Goldstein and Simon (1999); [b] Page and Burr (1991); [c] Simon (1999); [d] Scott and Crossman (1973), Page and Burr (1991).

Table 3-8. Habitat characteristics of the five study reaches of the lower Little Bow River, Alberta from observations at three stations within each reach, Fall 2009.

Parameter	**Reach**				
	1	**2**	**3**	**4**	**5**
distance from upstream limit of study area (km)	4.2	2.9	1.8	0.8	0
total sampled area (m^2)	895.5	696.2	842.8	873.5	890.6
mean sampled length (m)	38.3	29.2	40.7	38.0	40.0
mean wetted width (m)	7.8	8.5	7.1	7.7	7.4
mean wetted depth (m)	0.74	0.58	0.60	0.51	0.57
dominant (secondary) substrate type[a]	s (sg)	s (sg)	s (s, sg)	s (sg, mg)	s (sg, c)
mean boulder coverage (%)	1	1	0	2	2
mean cutbank coverage (%)	40	17	12	7	12
scored riparian impact	2	0.3	1.7	5.3	0

[a] s=sand (<2 mm), sg=small gravel (<4 mm), mg=medium gravel (<25mm), c=cobble (>64 mm <256mm)

Table 3-9. Anticipated responses of the lower Little Bow River fish community metrics to habitat change (in Stevens and Council 2008 from Karr and Chu 1999, Bramblett et al. 2005, Stevens et al. 2006, Noble et al. 2007).

Measure	Local Species	Metric Response and Relationship to Habitat Quality/Indicated Impact
Proportion of litho-obligate individuals	White Sucker, Lake Chub, Longnose Dace, Spottail Shiner	Declines as higher sedimentation reduces the availability of gravel substrates suitable for spawning
Proportion of individuals that are top predators	Northern Pike	Increases as viable populations of top predators indicate a relatively healthy, diverse community
Number of benthic invertivore species	Longnose Dace	Declines as river habitats become excessively silty or DO is reduced, resulting in altered benthic invertebrate prey community
Percent older, long-lived fish	Northern Pike >600 mm	Increases as river connectivity is improved and suitable habitat conditions support fish species over lifetime
Proportion of invertivorous cyprinids	Lake Chub, Longnose Dace, Spottail Shiner, Emerald Shiner	Declines may indicate decreased invertebrate food source due to habitat degradation
Proportion of intolerant individuals	Longnose Dace	First to decline with habitat degradation as these species are less tolerant to habitat changes
Proportion omnivores	White Sucker	Expected to increase as site declines in quality
Proportion tolerant individuals	White Sucker	Expected to increase as site declines in quality

Table 3-10. Habitat measurements and observations for sampled stations on five reaches of the lower Little Bow River, Alberta, October 2009.

Reach		**1**			**2**			**3**			**4**			**5**	
Station	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**
sampled area (m^2)	298	298	300	300	204	192	295	289	259	278	300	296	300	274	317
sampled length (m)	40.0	35.0	40.0	40.0	21.5	26.0	50.0	35.0	37.0	37.0	40.0	37.0	38.0	38.0	44.0
wetted width (m)	7.5	8.5	7.5	7.5	9.5	8.5	5.9	8.3	7.0	7.5	7.5	8.0	7.9	7.2	7.2
maximum width[a] (m)	-	-	-	-	-	9.1	-	9.0	8.0	-	-	-	-	-	-
minimum width[a] (m)	-	-	-	-	-	5.6	-	7.5	6.0	-	-	-	-	-	-
wetted depth (m)	0.75	0.76	0.70	0.60	0.70	0.45	0.45	0.51	0.85	0.50	0.53	0.50	0.60	0.50	0.60
maximum depth[a] (m)	-	-	-	-	0.90	1.05	-	-	0.95	-	0.60	0.50	0.75	0.55	0.80
minimum depth[a] (m)	-	-	-	-	0.50	0.15	-	-	0.65	-	0.45	0.30	0.45	0.40	0.35
dominant substrate type	sand	sand	sand	sand	sand	sand	sand	sand	sand	sand	sand	sand	sand	sand	sand
secondary substrate type[b]	small gravel	small gravel	small gravel	small gravel	small gravel	small gravel	-	small gravel	small gravel	small gravel	small gravel	med. gravel	small gravel	small gravel	cobble
boulder coverage (%)	1	1	2	1	0	2	0	0	0	0	0	5	0	0	5
cutbank coverage (%)	50	40	30	50	0	0	10	5	20	5	10	5	20	15	0
scored riparian impact	2	2	2	0	0	1	2	2	1	6	5	5	0	0	0

[a] maximum and minimum width and depth values are presented where field measurements varied by >0.5 m from the mean value recorded for the station

[b] secondary substrate type is listed where observed and represents the second-most dominant substrate type after the primary or dominant substrate size

4 SYNTHESIS

> *"Everything that can be counted does not necessarily count; everything that counts cannot necessarily be counted."*
>
> (*attributed to Albert Einstein)*

4.1 Research Summary

This research described two components of the Lower Little Bow River aquatic ecosystem: water quality of the river, through versions of a Water Quality Index, and fish community, through diversity and habitat measures. Despite the multiple relationships between these components, telling a cohesive story based on their separate study can be challenging. In attempting to bring these two sides of the Little Bow River story into my thesis, I hope that my research will encourage others who study purely water quality or purely fisheries to reach across their disciplines.

4.1.1 Water Quality Index Information

Water quality is an often monitored environmental component of the aquatic ecosystem. This research restructured the standard CCME Water Quality Index to address index function with seasonal frequency of calculation and sub-index divisions of parameters into biological, chemical, and physical groups. Each index scenario produced site/reach scores; index scores were compared using rank order and rater reliability analyses.

Overall, water quality within the lower Little Bow River ranged from good to poor between 2004 and 2007. Water quality was best in spring 2005 downstream of the fenced riparian reach evaluated by Miller et al. (2010) and worst in the year previous at the same site. Chemical sub-index scores were almost uniformly higher than physical and biological sub-index scores, due largely to common exceedances of dissolved oxygen, fecal coliform, and *E. coli*, criteria, as well as exccedance of criteria for parameters measuring suspended sediment.

Water quality indices, on the whole, captured similar overall patterns in water quality condition as noted in single parameter analysis (Miller et al. 2010), particularly in the reversal of direction in most year to year score improvements and declines. Seasonal and sub-index partitioning, however, resulted in a wider score range than was initially seen in the standard WQI scores (which included all parameters and was calculated annually).

Seasonal scores allowed review of water quality at times of year associated with major watershed events: spring freshet, irrigation withdrawals, return flows, and ice coverage. Sub-index scores also amplified small changes within the watershed and provided a signal of exceedances in key processes; for example, physical index scores were low in spring because of total suspended solids and turbidity exceedances associated with spring melt. These features are important because they improve options for linking WQI score information with information on watershed behaviour, a key task for water and land managers aiming to control and improve water quality. Trend analysis, which would allow the review of scores for evidence of upwards or downwards trends in water quality, was hindered by the relatively short time frame represented by the data.

4.1.2 Fish Community Assessment Information

Baseline information on the fish fauna of the study area was collected in a single-season survey of the five WEBs study area reaches. Fish within the lower Little Bow River represented a collection of largely generalist minnows and suckers, although one sportfish, an individual northern pike (*Esox lucius*), was also collected within the riparian fenced reach of the study area. Presence of young of the year fish indicates habitat function for reproduction of cyprinids and white suckers, while the occurrence of older individuals (based on observed length) in six of the seven species demonstrates that habitat conditions support the persistence and growth of longer-lived species.

Habitat complexity within the lower Little Bow River study area was provided by cutbanks, riparian wetlands along the channel margins, deeper pools, and deep riffles (i.e., a habitat unit shallower than a pool or a run, where water movement over gravel or cobble substrate results in surface turbulence). Wetted depth within the channel was noticeably variable day to day during the field collection program, likely as a result of controlled discharge at upstream water management facilities. Within the lower Little Bow, permanence of habitat is reliant upon the maintenance of minimum instream flows and access to overwintering habitats (e.g., deep pools unlikely to freeze entirely). Bank stabilization, through a combination of riparian vegetation retention and control of bank-eroding high flows, is also important for retention of key instream habitat features such as cutbanks.

Mesohabitat features appeared relatively uniform in their availability throughout the watershed. Within reaches, however, microhabitat availability, species traits, individual strategies, and interactions between organisms were likely drivers of the fish assemblage through ecological processes such as niche partitioning, predation, and competition. These features were only peripherally described by the study methods.

4.2 Research Applications

4.2.1 Aquatic Ecosystem Monitoring

Because Alberta watersheds are altered by land and water uses, and as numerous modifications have been made to the natural hydrologic regime of many river systems, establishing reasonable natural water quality guidelines and criteria is challenging. In our streams and rivers, a WQI score alone does not present a complete picture. Flow alterations have resulted in modified patterns of water and sediment movement within many systems like the Lower Little Bow River; such modifications are known to alter habitat features for aquatic and riparian species. Given these physical changes to the hydrologic regime of the watershed, it also seems fair to ask: what watershed characteristics now form the ‘new’ baseline for the river and how might we evaluate recovery of degraded systems following the application of beneficial management practices?

Relying on one component of the environment to be representative of all of the complex processes and interactions underway is a sure recipe for creating an overly-simplified picture of a watershed’s aquatic ecosystem. In this work, combining the study of water quality and fish community provided better opportunities to understand limits to aquatic ecosystem function in the watershed.

We found that undertaking a fish and fish habitat inventory prior to the selection of criteria to be used in the WQI supported a review of those criteria against life requisites of species actually found within the watershed. Often, criteria selected are designed to be the most protective and so are based on needs of the most sensitive species, such as cold water salmonids, regardless of whether these species have been historically or are currently present within the watershed. For our use of WQI scores to monitor within one river or watershed, fish community information allowed us to select criteria ranges suited to the physiological requirements of resident species.

In return, knowledge of water quality can inform fisheries management. WQI, when collected at a seasonal scale, can identify times of the year when watershed conditions may challenge fish growth and survival. The seasonal WQI for the Lower Little Bow River produced low summer WQI values, driven by dissolved oxygen levels exceeding (i.e., falling below) criteria in summer samples in 2004, 2005, and 2007. Low dissolved oxygen, in combination with higher summer water temperatures, acts as a stressor to fish and other aquatic organisms reliant on water as their media for respiration (Schlosser 1991). This knowledge could also be used in reservoir management of stream flows. Sub-index level WQIs can similarly identify specific parameters or processes (e.g., dissolved oxygen, or sediment-transport related parameters) that may limit health of the aquatic ecosystem. Having WQI information at the sub-index level enables management actions targeted at those specific parameters of concern.

The CCME WQI uses criteria values that define the ideal for water quality by looking at a national suite of watercourses and establishing a high standard for water quality protection. Using this national index for local-scale monitoring within a historically modified, flow controlled agricultural watershed produced ratings within a narrow score range that limited detection of potential score change due to the application of watershed beneficial management practices designed to improve water quality. In our study of the Lower Little Bow River micro-watershed, shifting the temporal scale of the index from an annual level to a seasonal level improved the range of index ratings.

4.2.2 Agricultural Beneficial Management Practices

Evaluating the effectiveness of agricultural watershed beneficial management practices requires an understanding of both the watershed processes involved in the effect and the receiving environment the practices themselves are designed to protect. Practical application of this work includes support for the WEBs research on-going on the Lower Little Bow River. Using a revised water quality index at a tighter spatial and temporal scale allowed us to rate WEBs sites by overall water quality and seasonal performance, while sub-index scores provided detailed indication of variables driving quality performance. Baseline information on fish assemblages and instream habitats across the study area also suggested a gradient of reaches that could be interpreted by the BMPs applied by WEBs. For example, Reach 5, an approximately 800 m section of the lower Little Bow River treated with cattle exclusion fencing, presented the greatest fish diversity and instream habitat heterogeneity of the study area reaches.

BMP effectiveness monitoring within one watershed or watercourse using BACI has been challenged by factors including delayed response times while riparian vegetation reestablishes, masking of treatment improvements by upstream loading or pollutant discharges, and the limited instream treatment function of many employed BMP practices. Effectiveness monitoring using paired watersheds has, in many cases, provided stronger evidence of BMP efficacy for water quality improvement, yet presents its own challenges.

4.3 Opportunities for Future Research

4.3.1 Developing Site-Specific Water Quality Guidelines for WEBs

Criteria or guideline values we associate with acceptable water quality are selected to provide the highest level of protection for water resources, in line with the precautionary principle. While the interest in protection is shared by this researcher, it seems unfair to impose what might be an unreachable standard without consideration of natural physical constraints to site water quality and the needs of resident species. To use a water quality index for monitoring at a local scale (i.e., to rate quality changes over time within one watercourse), it is important to consider the intrinsic bounds of water quality set by natural watershed characteristics and explicitly discuss what conditions should be assigned as "natural" to the watershed. WEBs objectives in the Lower Little Bow River include monitoring for change in response to riparian exclusion fencing and other BMPs within an already altered watershed (and one that is shaped by on-going water management, flow regulation, and upstream agricultural modifications). What should be considered the natural baseline conditions, and hence, the criteria values to be used in generating quality ratings within a water quality index designed for monitoring change? We needed to first consider how our set of criteria fit the characteristics of our watershed.

CCME directives for the establishment of site-specific water quality criteria recognize the benefit of such an approach; however, they also specify that anthropogenically altered river characteristics should not be used as a basis for site-specific conditions. Given the history of flow regime modification and agricultural land use change within the Lower Little Bow River, the identification of acceptable or comparable watershed reference conditions remains a challenge to the design of defensible site-specific criteria.

In the absence of historical datasets or more detailed historical conditions, using a combination of national/regional criteria and observed local ranges for a given variable provides a starting point for environmental monitoring with indices. Development of such a modified WQI using site-specific criteria would enable the design of an index centred on the score variability seen in the Lower Little Bow River (i.e., scores within the fair to good range) and this may allow us better resolution when looking for change from reach to reach in response to applied BMPs. The modified criteria would also be justified against the habitat characteristics required to support the known aquatic community within the study area.

In watersheds for which we have some understanding of water quality, particularly in areas where an existing impact has already drawn our interest and concern, constructive information is more likely to be generated by tools that can describe not simply that our water may have changed from “good” to “fair” but which can tell us more about the changes we might see within that “fair” or “good” category. For this reason, future research opportunities exist in the application of modified WQI criteria for water quality monitoring in the lower Little Bow River – an example I developed, following work by Lumb et al. (2006) is presented here along with the standard CWQI.

Table 4-1. Potential criteria values to be considered in the design of a modified site-based WQI.

Variable	Standard WQI Criteria	Modified WQI Criteria
DO	6.5	5.5
TSS	seasonal median + MAD + 25 mg L^{-1}	seasonal median + 2MAD + 25 mg L^{-1}
Turbidity	seasonal median + MAD + 8 NTU	seasonal median + 2MAD + 8 NTU
Chloride	150 mg L^{-1}	350 mg L^{-1}
Phosphorus	0.01 – 0.02 mg L^{-1}	0.05 mg L^{-1}

4.3.2 Developing a Fish Index of Biological Integrity for WEBs

Physical changes within the watershed as a result of applying agricultural beneficial management practices have the potential to alter stresses and habitat features that shape the ecological processes that drive community structure. Measures of the ecological community, therefore, are potential tools for monitoring environmental change resulting from BMP application.

Fish indices of biotic integrity can now be developed, now using the known species within the watershed and their habitat requirements. This can provide a biotic index corollary to the WQI that would support on-going monitoring of the system response to riparian and agricultural land-use changes. As with water quality indices, criteria or reference condition selection poses a challenge: for modified systems, do we utilize the 'best' of our local sites as our reference condition, or do we more broadly define a reference condition based upon regional information? In the case of the Lower Little Bow River, I suggest value would be gained by the definition of a reference condition using fisheries and aquatic ecosystem information from state fish and wildlife agencies located south of Alberta and Saskatchewan borders, also within terrestrial grassland ecosystems subject to historic hydrologic modifications.

Using fish community structure to evaluate the influence of non-point source pollutants on aquatic ecosystem integrity has noted challenges, as both natural and anthropogenic factors can affect habitat stability of fish communities and alter the normal structural and functional dynamics of fish communities. Changes to the physical, chemical, or biological character of a watercourse alter habitat function, and so may shift individuals' patterns of use and rates of growth, reproduction, and mortality. These individual shifts are aggregated to population and community levels, and large-scale observational biodiversity patterns (e.g., species-abundance distributions, species–area curves, body size-diversity distributions) are considered to reflect the underlying processes that structure ecological communities. Some caution is warranted in this ecological 'scaling up'; observed community patterns are likely the result of both positive and negative interactions within and among species and often interpretation of the limited catch information provided in fish surveys, particularly in single season assessments, requires broad assumptions of species interactions and individual behaviours.

Researchers evaluating agricultural practices in riparian zones have achieved varying results in finding connection between land use change and improvement in the fish community. Improvements to diversity measures, for example, may require the addition of fish species no longer found in the watershed as a result of historical alterations to watershed function and connectivity. Positive relationships between species richness response and the WQI may be matched with high species richness and nutrient correlations typically associated with degradation of water quality.

Despite these challenges, living components of the aquatic ecosystem (both flora and fauna) should continue to be measured in conjunction with water quality surveys. Support for their health ultimately forms one of the primary reasons for our efforts in monitoring and managing water quality. Programs to evaluate watershed response to land use practices benefit from the different perspectives of watershed and aquatic ecosystem health that are provided by water quality and fish community information.

4.3.3 Monitoring the Right Variables

A watercourse monitoring program may be of limited value because it does not capture key parameters that limit water uses or pose the greatest environmental risks. Residual pesticides, toxic substances, and personal pharmaceutical products may pose great risks of harm to aquatic ecosystems and both human and livestock water uses (Sumpter 2010; AESRD 2012a, 2012b), but these parameters are not sampled as part of the WEBs monitoring program. Future research should include at least basic sampling for these additional parameters to assess the potential for their harm and influence to overtake any improvements occurring within the WEBs micro-watershed.

Additionally, for water quality data used in WQI scoring, analytical limits should be reviewed to ensure parameters are measured to levels of precision that reduce censoring of the data to criteria values. This censoring happens when a laboratory test for a particular parameter can only measure to a value above the criteria. This results in uniform exceedances for that parameter and not only produces lower WQI scores but reduces the information the variable brings into a monitoring program designed to detect change.

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