A Fish-based Index of Biological Integrity for Assessing River Condition in Central Alberta

Alberta Conservation Association

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EXECUTIVE SUMMARY

Recent economic growth in Alberta has resulted in the conversion or modification of the majority of prairie-parkland ecosystems for agriculture, energy infrastructure, and urban development. The extent of, and rapid increase in human activities at the landscape scale may pose a serious threat to the integrity of aquatic ecosystems and the fish assemblages they support. We developed a multi-metric, fish-based Index of Biological Integrity (IBI) for assessing the health of aquatic ecosystems in central Alberta. Data on fish assemblages collected via electrofishing by the Alberta Conservation Association were combined with reach and basin-scale environmental variables for 80 river sites on the Battle River. We screened 12 candidate metrics representing attributes of the Battle River fish assemblage for redundancy, as well as their sensitivity to human disturbance variables, using regression and information theory methods. We selected three metrics for the IBI representing two trophic guilds (i.e., percent carnivores and percent omnivores) and one measure of community structure (i.e., species richness) that were unrelated to river size but related to measures of human disturbance. The multi-metric IBI was highly sensitive to changes in cumulative anthropogenic disturbances (statistically indexed as road densities). Regression analysis indicated that cumulative disturbances associated with road densities as low as 7 m/ha (i.e., 0.7 km/km^2) in basins may impair the integrity of fish assemblages. The Battle River IBI provides a single, defensible, easily understood measure of the health of watercourses in the prairie-parkland ecoregion. With the aid of a simple spreadsheet, land managers and researchers can quickly calculate an IBI score using fish data collected from a river section. Additional research on ecological functions and requirements of species in northern systems is recommended to strengthen the basic foundation of guild-based bioassessment methods in Alberta.

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1.0 INTRODUCTION

The landscapes of central Alberta and the prairie-parkland ecosystem in Canada have undergone rapid changes in recent decades. Urban growth combined with similar growth in agriculture and energy sectors have converted or modified the majority of drainage basins for pasture, cropland, roads, suburban housing developments and petroleum pipelines and related infrastructure. Widespread development across the landscape may be a serious threat to the ecological integrity of aquatic ecosystems (Allan 2004). Indeed, there is widespread recognition of the extent and significance of changes in land-use and land cover worldwide, which has led to an increase in studies that seek to establish relationships between land-use and river condition (Hughes et al. 2006). Knowledge of the relationships between land-use and river condition can be used to predict the extent of change in river condition in response to human development and plausible alternative futures.

A common goal in ecosystem management is the preservation and restoration of aquatic resources. One of the key quality elements used to describe the status of aquatic resources is fish assemblage data (Karr and Chu 1999). Direct assessment of fish assemblages is often more relevant than surrogate approaches based on invertebrates or abiotic criteria alone because governmental regulations explicitly call for protection of fish (e.g., U.S. Clean Water Act, Canada's National Parks and Fisheries acts). Further, fish are good indicators of ecological status as they occupy a range of ecological niches and their ecological processes operate over various spatial scales (Karr and Chu 1999). Thus, fish assemblage data are commonly used in bioassessments, linking ecosystem conditions to human-related stressors (e.g., Daniels et al. 2002; Mebane et al. 2003; Bramblett et al. 2005).

Although a variety of biological indices are available for evaluating aquatic habitats, the Index of Biological Integrity (IBI) has been particularly successful for assessing river conditions (Karr and Chu 1999). IBI's are multi-metric indices reflecting important components of ecosystems, including taxonomic richness, habitat and trophic guild composition, and individual health and abundance. However, the characteristics of the IBI change from region to region such that an IBI created for streams in the midwestern U.S., for example, would likely perform poorly in the prairie-parkland ecoregion in

Canada (Fausch et al. 1984; Hughes et al. 1998; Angermeier et al. 2000). Angermeirer et al. (2000) showed that a single IBI metric can vary in its response to disturbance among river basins. Specifically, taxonomic metrics can be constrained by the phylogenetic history of the study basins, while functional metrics can be geographically limited in their response. To our knowledge, there have been few previously published IBI's created for Canadian streams and rivers (but see Steedman 1988; Stevens et al. 2006).

1.1 Study objective

The Government of Alberta requires a bioassesment tool to support the management of aquatic resources. A tool, such as an IBI, would not only aid resource managers to assess current levels of impairment, but also could be used to predict effects of land-use activities in a Geographic Information System (GIS). The primary objective of this study was to develop an IBI (i.e., a tool) for monitoring and evaluating ecological conditions of the Battle River. Resources managers can use this tool (i.e., a formula) with their fish catch data to calculate an IBI score that would be linked to human disturbance in basins and not be confounded by natural environmental gradients (e.g., stream size). Specifically, we tested candidate metrics and validated the IBI through an examination of a spatially-explicit database describing physico-chemical conditions and fish assemblages from 80 river sections.

2.0 STUDY AREA

2.1 Description

The study system was the Battle River in Alberta which flows out of Battle Lake (52˚ 55′ N, 114˚ 10′ W) and eastward through the prairie-parkland ecoregion (Strong and Leggat 1992) for approximately 800 km to the Saskatchewan border (52˚ 51′ N, 109˚ 59′ W), south of Lloydminster (Figure 1). The Battle River then enters the North Saskatchewan River near North Battleford in Saskatchewan. The Battle River drainage is approximately 30,000 km^2 (Alberta Environment 2005). Typical summer flows on the Battle River are between 4 and 8 m^3/s at the Alberta - Saskatchewan border. The river's average gradient is less than 0.4 m/km. Four licensed water control structures occur on

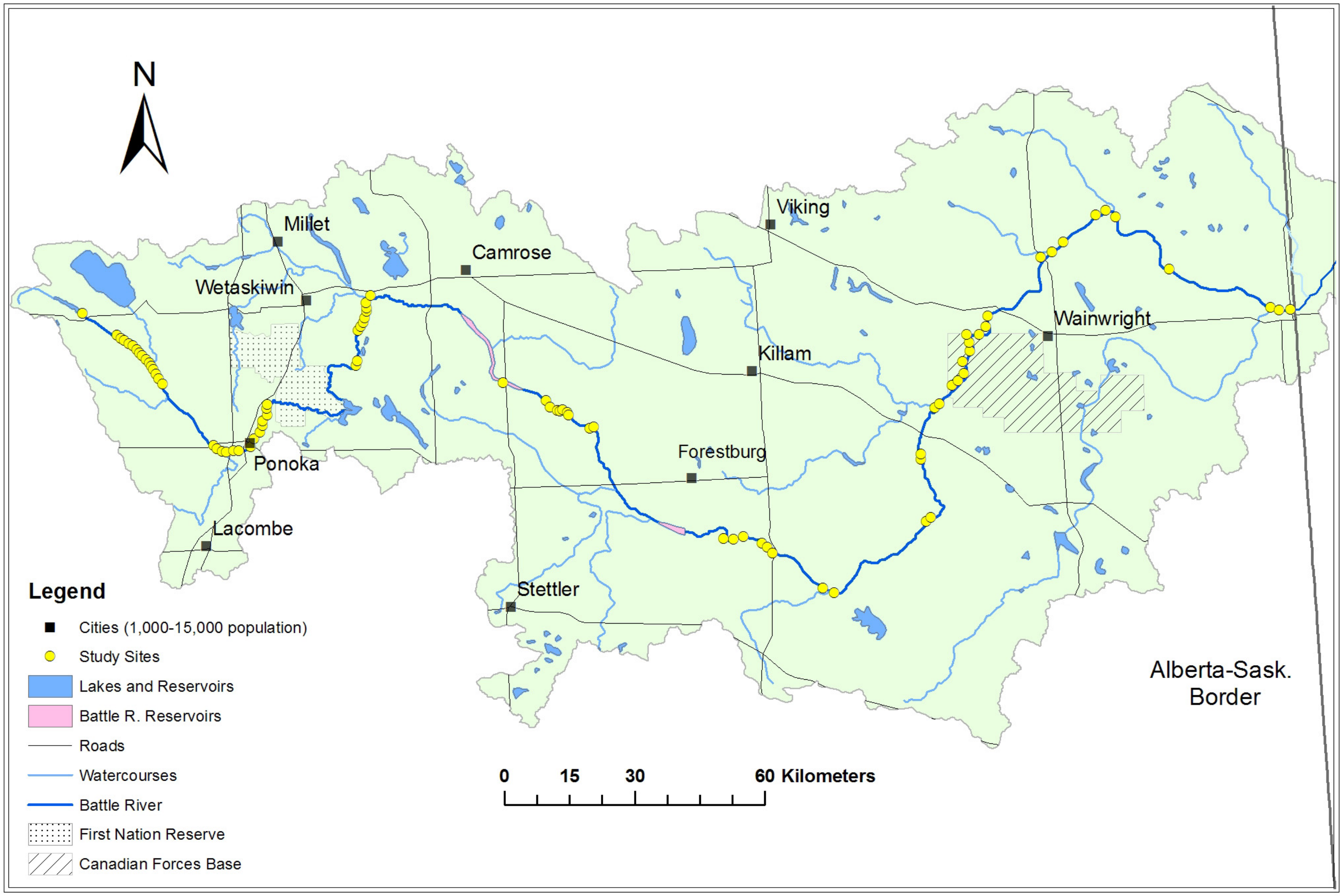

Figure 1. Map of the Battle River and its basin for which the fish-based Index of Biological Integrity (IBI) was developed.

the river for water storage to provide municipal supplies and to supplement downstream flows when the river is low (Buchwald 2001).

2.1 Fish assemblage

The historical fish assemblage of the Battle River comprises approximately 19 coolwater fish species (Nelson and Paetz 1992; Buchwald 2001; Christiansen 2001). Appendix 1 provides a complete list of species known to occur within the Battle River and their ecological characteristics. All species are native to the Battle River in Alberta.

3.0 MATERIALS AND METHODS

3.1 Study design

We selected study sites from reaches of the Battle River to represent the full spatial extent of the river and conditions along it (Figure 1). In the upper section of the Battle River, we selected 40 sites between Battle and Driedmeat lakes (approximately 286 km of river), ten sites along the length of the Battle River from the water control structure on Driedmeat Lake to the Forestburg Reservoir (approximately 116 km), 14 sites along the third reach, defined from the water control structure on Forestburg Reservoir to the western boundary of Canadian Forces Base (CFB) Wainwright (approximately 210 km), and 20 sites on the lowest reach from the western boundary of CFB Wainwright to the Alberta - Saskatchewan border (approximately 200 km). The basin size of study sections varied considerably ranging from 110 km^2 for the smallest study reach at the top of the basin to 24,780 km^2 for the largest study reach located near the Saskatchewan border. River wetted width of the study sites, calculated as the average of five measurements taken along a study section, ranged from 9.3 to 57 m (Table 1). At each site, we identified a 1- or 2-km sample section as a discrete sampling unit. We used 1-km sections for fish sampling in the upper reaches of the Battle River, whereas we used 2-km sections in the lower reaches. Prior to electrofishing and to complement GIS-based data, we collected standard descriptions of habitat and nearby human land-uses at each stream section, including information on stream depth and the condition of the riparian zone (e.g., percent area grazed) (Table 1). Generally, all sites were affected by agricultural and urban-related activities and land-uses, although sites in the vicinity of CFB Wainwright were identified as potential reference sites (i.e., minimally disturbed sites) prior to field work. It is important to note that we excluded four sites that were part of the original study program from analyses due to ineffective electrofishing in water that was deep and of high-velocity.

Table 1. Summary statistics of focal parameters used for describing river sections of the Battle River, Alberta (n = 80).

Variable	Mean	SD	Min	Max	CV(%)
Riparian intactness index (%)[a]	89.6	10.6	55.0	99.3	11.9
% Human-caused bare ground	6.2	8.6	0.0	40.0	138.8
% Vegetative cover	90.0	9.6	60.0	99.0	10.7
% Grazed cover	15.0	23.6	0.0	95.0	157.7
Dissolved oxygen (DO; mg/L)	7.77	0.93	4.13	11.28	11.9
pH	8.29	0.14	7.80	8.60	1.7
Ammonia-nitrogen (mg/L)	0.053	0.033	0.008	0.118	61.1
E. coli (mg/L)	105.0	152.6	5.0	700.0	145.3
Fecal coliforms (mg/L)	110.4	152.7	5.0	700.0	138.3
Total dissolved phosphorous (TDP; mg/L)	0.096	0.083	0.010	0.273	86.9
Total phosphorous (TP; mg/L)	0.177	0.090	0.025	0.382	51.1
Total Kjeldahl nitrogen (TKN; mg/L)	1.49	0.27	0.88	1.98	18.4
Nitrite + nitrate-nitrogen (mg/L)	0.111	0.217	0.003	0.792	194.3
Water Quality Index (WQI)*	68.1	13.1	41.2	89.4	19.3
Cattle density per ha 10-km scale	0.395	0.183	0.158	1.080	46.5
Cattle density per ha in basin	0.453	0.104	0.317	0.655	23.0
% cropland cover 10-km scale	19.5	11.5	2.3	46.5	59.0
% cropland cover in basin	21.0	13.0	0.0	35.4	61.9
% urban cover in basin	0.7	0.7	0.0	2.2	98.8
Population density per ha in basin	0.073	0.036	0.020	0.135	49.2
Road density in basin (m/ha)	12.4	1.0	10.6	14.2	8.3
Manure application in basin (tonnes/ha)	4.336	1.173	2.776	6.654	27.1
Manure application, 10-km scale (tonnes/ha)	3.649	1.910	1.391	10.460	52.3
Upriver dist. to sewage lagoon (m)	29,430	20,750	577	93,134	70.5
Upriver dist. to town (m)	28,134	13,707	0	57,111	48.7
Population density per ha 10-km radii	0.063	0.098	0.001	0.299	155.1
Basin size (ha)	775,165	734,536	11,096	2,478,024	94.8
Basin size (km^2)	7752	7345	111	24,780	94.8
Mean wetted width (m)	19.9	7.4	9.3	57.0	37.3
Mean maximum depth (m)	1.9	0.6	1.0	4.0	29.8

[a]Integrated values of percent human-caused bare ground, percent vegetative cover and percent grazed.
*Integrated values of fecal coliform, *E. coli*, $NO_2 + NO_3$-N, TKN, pH, DO, TDP, and NH_3-N (CCME 2001).

3.2 Fish sampling

We sampled fish at each site through use of a throwing anode and a boat electrofisher (Coffelt VVP-15). Sampling occurred from 13 June to 13 July in 2006 and from 28 May to 21 June in 2007. We recorded electrofishing sampling effort as the number of fish captured per 100 s. Effort ranged from 1,041 to 2,579 s per 1-km site, and from 2,628 to 6,536 s per 2-km site. In the field, we sampled in 500-m sections such that captured fish were held for short periods only and immediately released downstream prior to sampling the remainder of the river section. For all captured fish, we identified species and measured weight (± 1 g) and fork length (± 1 mm). We also examined fish for DELTS: deformity and disease, eroded fins, lesions, and tumors (Daniels et al. 2002; Mebane et al. 2003).

3.3 Water quality

We collected water quality samples from mid to late June 2007 at the location which corresponded to the start point for each sampling reach. We collected 'grab' samples according to Alberta Environment protocols (Alberta Environment, unpublished internal report) and submitted samples to CAEAL (Canadian Association for Environmental Analytical Laboratories) accredited laboratories in Alberta for analysis. Laboratory analyses included tests for nutrients, physical parameters, fecal coliform bacteria, and algal biomass (measured as chlorophyll *a*). We also used field monitoring equipment to record dissolved oxygen (DO, mg/L), pH and conductivity (uS/cm) at the approximate time and location of the grab sample. We used some of the water tests to provide supplementary ecological information only; whereas a select number of tests were used for developing the IBI, for example, fecal coliforms (colony forming unit (CFU)/100 ml), *Escherichia coli* (CFU/100 ml), dissolved nitrite + nitrate as nitrogen (NO_2 + NO_3-N; mg/L), total Kjeldahl nitrogen (TKN; mg/L), DO, pH, total dissolved phosphorous (TDP; mg/L), and total ammonia-nitrogen (NH_3-N; mg/L). We integrated the bacteria and nutrient parameters to create a Water Quality Index adopted from CCME (Canadian Council of Ministers of the Environment; CCME 2001; also see Alberta Environment 2006); scores were based on the number of variables that did not meet water quality guidelines and by how much a measurement exceeded guidelines.

3.4 GIS analyses

Using ArcGIS 9.2, we calculated human disturbance measures to aid with the development and evaluation of the IBI (Table 1). We accessed seven governmental databases (or layers) to create variables that measure various human land-use activities at multiple spatial scales. These layers included the Cropland Insurance Database (source: Agriculture Financial Services Database), Alberta Road Networks (all gravel and paved access, source: Alberta Sustainable Resource Development (ASRD)), Town Sites (i.e., urban cover for municipalities with a population > 1000, with a majority of the buildings on parcels of land smaller than 1,850 m^2; ASRD), Census Block and Population Data (Statistics Canada), Cattle Intensity Census Data (Alberta Agriculture, Food, and Rural Development), Manure Application Data (Alberta Agriculture, Food, and Rural Development) and Wastewater Treatment Sites (Alberta Environment).

We used ArcGIS Hydro tools and a provincial digital elevation model (DEM; 1:20,000) to delineate watersheds and calculate basin size per study location. The upriver starting point for a study section was the drainage point for watershed delineation. We also described the immediate upriver landscape at the spatial scale of 10-km by 5-km wide (i.e., perpendicular from the stream bank). We quantified human land-use activities at various spatial scales, for example, cattle density 10-km upriver as well as percent urban cover in the basins. We also calculated upriver distances to the nearest wastewater treatment site (i.e., sewage lagoon) and the nearest urban centre (i.e., town) to identify potential effects of point sources of pollution on river condition. As possible correlates of fishing pressure and human access to river, we calculated human densities within < 10-km radii of the mid-point of the river sections.

3.5 IBI construction

We considered fish species previously recorded in the Battle River, as well as their habitat requirements and life history in the selection of candidate IBI metrics (Nelson and Paetz 1992; Buchwald 2001; Christiansen 2001; Council 2007; see Appendix 1). Candidate metrics for the IBI reflected various functional and structural guilds that were similar to those successfully used in other IBI programs (e.g., Steedman 1998; Bramblett et al. 2005; Stevens et al. 2006).

We developed the IBI using the Battle River fish assemblage and an initial evaluation of 12 candidate metrics that were either positive or negative scoring (Table 2). Positive scoring metrics (i.e., those that should increase with increasing biological integrity) included number of fish species (all were native), proportion of individuals that are litho-obligates, proportion of carnivores, proportion of insectivorous cyprinids, proportion of benthic invertivores, number of benthic invertivorous species, proportion of older, longer-lived fish, and proportion of intolerant individuals (to human disturbance). Potential negative scoring metrics (i.e., those that should decrease with increasing biological integrity) included the proportion of omnivores, proportion of tolerant individuals (to human disturbance), proportion of individuals with DELTs, and total number of fish.

Many of the above-mentioned candidate metrics were similar to those used in the original IBI by Karr and Dudley (1981) for midwestern U.S. streams, and have been consistently reliable in their performance throughout North America (e.g., native species richness; Karr and Chu 1999). Other metrics were specific to watercourses at northern latitudes in North America, for example in Montana (Bramblett et al. 2005) and southwestern Ontario (Steedman 1988). In these systems, litho-obligates have been shown to be particularly sensitive to human disturbance. Litho-obligate species breed on 'clean' rock and gravel, and have benthic larvae that hide under stones. Their survival is expected to decline in highly turbid waters caused by sedimentation and erosion. We also predicted the relative abundance of long-lived individuals to increase with permanence of suitable habitat, absence of catastrophic anthropogenic disturbance, and connectivity to source populations (Karr and Dudley 1981; Hughes et al. 1998; Bramblett et al. 2005). For guild-based metrics, we selected units based on both an understanding of the fish assemblage and inherent limitations of our dataset (Karr and Chu 1999). For example, if there were less than four species in a guild, the proportion of individuals in a catch for that particular guild was likely a better candidate metric for the IBI (versus number of species). The number of individuals or catch-per-unit-effort (CPUE) was also considered; however, such metrics can be susceptible to sampling biases and may not be reliable (Karr and Chu 1999). For example, high total fish CPUE could suggest either high integrity or indicate nutrient loading from upriver urban and agricultural sources (Steedman 1988; Schleiger 2000).

Table 2. Candidate metrics for the Index of Biological Integrity (IBI) for assessing the ecological condition of the Battle River, Alberta.

Candidate Metric	Description
Positive Scoring	***Values increase with increasing biological integrity***
Number of native species	A decline in taxa richness is generally one of the most reliable indicators of degradation.
Proportion of litho-obligate individuals (LNSC, SHRD, WHSC, LKCH, LNDC, SPSH, BURB, GOLD, MOON, WALL, TRPR)	Expected to decline as human influence increases, such as higher sedimentation that reduces the availability of gravel substrate for spawning.
Proportion of individuals that are top carnivores (Apex predators: WALL, NRPK, BURB)	Viable and healthy populations of top carnivores indicate a relatively healthy, diverse community.
Number of benthic invertivore species (LNSC, SHRD, LNDC, IWDR, TRPR)	Expected to decline with increase in human influence, such as when river habitats become excessively silty or DO is reduced.
Proportion of benthic invertivores (LNSC, SHRD, LNDC, IWDR, TRPR)	Expected to decline with increases in human influence, such as when river habitats become excessively silty or DO is reduced.
Percent older, long-lived fish (NRPK > 600 mm, WALL > 450 mm, WHSC > 400 mm, GOLD > 350 mm)	Older fish indicative of suitable habitat, reduction in anthropogenic disturbance, river connectivity.
Proportion of invertivorous cyprinids (LKCH, LNDC, SPSH, EMSH)	As the invertebrate food source decreases in abundance and diversity due to habitat degradation, there is a shift from insectivorous to omnivorous species.
Proportion of intolerant individuals (LNDC, GOLD, MOON, IWDR)	These species are first to decline with increasing anthropogenic influence.
Negative-scoring	***Values decrease with increasing biological integrity***
Proportion of omnivores (WHSC, FTMN)	As a site declines in quality, the proportion of individuals that are omnivores increases.
Proportion of tolerant individuals (WHSC, SHRD, FTMN)	Expected to increase as habitat, water quality, and watershed conditions are degraded.
Total individuals in sample (catch-per-unit-effort)	Total relative abundance is comparable to the overall ability of the river to support an aquatic community. Generally, sites with lower integrity support fewer individuals.
Proportion of individuals with DELTs (deformities, disease, parasites, fin erosion, lesions or tumours)	These conditions occur frequently below point sources and in areas where toxic chemicals are concentrated; can reflect stress caused by pollution.

Sources: Karr and Chu (1999); Bramblett et al. (2005); Stevens et al. (2006); Noble and Cox (2007). Abbreviations: LNSC = longnose sucker; QUIL = quillback sucker; SHRD = shorthead redhorse; WHSC= white sucker; EMSH = emerald shiner; FTMN = fathead minnow; LKCH = lake chub; LNDC = longnose dace; SPSH = spottail shiner; NRPK = northern pike; BURB = burbot; BRST = brook stickleback; GOLD = goldeye; MON = mooneye; IWDR = Iowa darter; WALL = walleye; YLPR = yellow perch; TRPR = trout-perch; LKWH = lake whitefish; DO = dissolved oxygen.

3.6 Tests for redundancy and sensitivity

Given that the candidate metrics were from IBI programs specific to other ecoregions in lower latitudes, an important step in this study was to verify predicted relationships between measures of human land-use (Table 1) and candidate metrics (Table 2). But first, construction of the IBI began with screening candidate metrics to identify those that were statistically redundant (Minns et al. 1994; Hughes et al. 1998; Lyons et al. 2001). The goal was to generate an IBI with metrics that were only weakly correlated with each other ($r < 0.8$). Decisions to include one metric over another were typically based on whether a metric was better predicted by explanatory variables and whether the metric was part of previously published IBI programs. Unfortunately, some metrics and their definitions were based only on expert opinion due to gaps in ecological knowledge (e.g., percent tolerant individuals).

Next, raw values of candidate metrics were corrected for basin area if they showed a correlation with this variable ($P < 0.10$). We used the residuals from linear regressions to correct raw values. These regressions used data only from reference sites (i.e., sites that were least affected by anthropogenic activities in our study area). References sites were identified as those river sections having a Water Quality Index > 80% (Alberta Environment 2006), urban cover in basin < 1%, cropland cover 10 km upriver < 10% (Degerman et al. 2007), and cattle/ha 10 km upriver of the sampling location < 0.4. The density threshold of 0.4 cattle/ha represents the approximate median value of cattle densities within 5 km perpendicular of the shoreline for 10 km distances upriver of study sections. Local-scale riparian descriptions of the study sections (e.g., riparian width, percent vegetative cover, percent area grazed, percent human-caused bare ground) were not used to identify reference areas because preliminary analyses indicated that these measurements were not strongly related to candidate IBI metrics (also see Stevens et al. 2006).

We screened candidate metrics for responsiveness to disturbance using multivariate regression and an information-theoretic approach that ranked *a priori* models (Burnham and Andersen 2002). Prior to model-building, we assessed Pearson correlations among all focal environmental parameters. If a pair of covariates had an $r > 0.8$, we eliminated one of the two covariates from the regression model to minimize multicollinearity. Of the water quality parameters, TDP was positively correlated with NH_3-N, and negatively correlated with the Water Quality Index. In addition, *E. coli* and fecal coliforms were positively correlated with $NO_2 + NO_3$-N. Of the GIS-based human disturbance variables, percent urban cover in the basin was positively correlated with human population density in the basin, and not surprisingly, the manure application rate variables were positively correlated with cattle density variables. In addition, percent cropland cover in the basin was negatively correlated with cattle density and manure application rate at the 10 km upriver-scale. It is also important to note that basin size was negatively correlated with cattle density and manure application rate in the basin, but positively correlated with percent cropland cover in the basin ($r > 0.8$). Because of the upriver-downriver trends in human activities, basin size may confound potential effects of cattle density and cropland cover variables on IBI scores.

We developed seven *a priori* hypotheses or models considering the human-disturbance covariates and their relationships with one another. The models predicted metric responses using various combinations of select water quality parameters and GIS-based variables. To test for local effects of degraded riparian areas, we also considered a coarse measure of riparian conditions (calculated as the mean value of percent grazed cover, percent non-vegetative cover, and percent human-caused bare ground). The models were structured as follows:

- **Water Quality Index (WQI) model**: the index integrated values of fecal coliform, *E. coli*, $NO_2 + NO_3$-N, TKN, pH, DO, TDP, and NH_3-N;
- **Water Quality Parameters Model**: $NO_2 + NO_3$-N, TKN, TDP, and DO;
- **Agriculture Model**: cattle/ha 10 km upriver, cattle/ha in basin, percent cropland cover 10 km upriver, and riparian intactness index;
- **Urban Model**: percent urban cover in basin, upriver distance to sewage lagoon, upriver distance to town, and road density in basin;
- **Urban+Human Access Model**: human population density within a 10-km radii and road density in basin;
- **GIS-based Urban+Agriculture Model**: cattle/ha 10 km upriver, cattle/ha in basin, percent cropland cover 10 km upriver, percent urban cover in basin, upriver distance to sewage lagoon, upriver distance to town, and road density in basin;

- **Global Model**: above-mentioned covariates, but excluding WQI and population density within 10-km radii because these variables would otherwise contribute to multicollinearity.

We used Akaike's Information Criterion, corrected for small sample sizes (AICc), as a basis to select models (Burnham and Andersen 2002). Primary inferences were drawn from the best model (AICc min) and others within two units of AICc min (Burnham and Andersen 2002). We also calculated Akaike weights (w_i) to assess evidence supporting each model. These weights were necessary to estimate the relative importance of parameters through model averaging, a robust method that reduces model selection bias (Burnham and Andersen 2002). As part of the metric selection approach, we considered a metric for the final IBI if regression showed anticipated responses to measures of anthropogenic disturbance, and if the direction of relationships were consistent; for example, a positive-scoring metric that was negatively correlated with TDP concentrations and percent cropland cover 10 km upriver, but positively correlated with DO concentrations and the riparian intactness index. However, if a moderate level of ambiguity was noted, exceptions were made if the metric under examination had been a proven component of previously published IBI programs. For example, species richness often declines as a result of habitat degradation and pollution associated with agriculture and urbanization (Karr and Chu 1999; Bramblett et al. 2005). In addition to the significance of model coefficients and direction of relationships in a model, R^2 values were considered in screening metrics. A low R^2 value (< 0.2) for the best model suggested that the metric was potentially 'insensitive' to human disturbance.

Upon screening of IBI metrics, we linearly scaled values of positive metrics (i.e., metrics negatively related to the level of human disturbance) from 0 to 1, whereas we inversely scaled negative metrics (i.e., metrics positively related to the level of human disturbance) such that minimum values were assigned 1 and maximum values were assigned 0. The total IBI score per site was a sum of standardized scores of the screened metrics with a perfect score of 3 (for a three-metric IBI) reflecting conditions for maintaining biological integrity. We validated the IBI for responsiveness to disturbance using similar methods for screening metrics (e.g., multivariate regression and an information-theoretic approach). Post-estimation tests included the Cook-Weisberg heteroskedasticity test and multi-collinearity tests through calculation of variance inflation factors. In addition, we plotted basin sizes against IBI residual scores to assess for the presence of strong spatial trends in errors along the Battle River (i.e., pseudo-replication).

4.0 RESULTS

Of the 19 species known to occur in the Battle River (Appendix 1), only 14 species were captured on the 80 sections of the river sampled for fish during 2006 and 2007. Total electrofishing effort was 3,935 min. In total 3,473 fish were captured, of which 49% were white sucker, 15.8% longnose dace, 11.5% lake chub, 9.8% northern pike, 6.6% shorthead redhorse, 3.3% trout-perch, 2.4% walleye, 0.9% fathead minnow, 0.3% burbot, 0.2% spottail shiner, 0.2% goldeye, 0.1% mooneye, 0.06% Iowa darter, and 0.03% longnose sucker.

We used the fish catch data to create 12 candidate metrics, some of which were redundant as determined by correlation analysis (Table 3). We removed percent tolerant individuals and percent intolerant individuals from the IBI program because these metrics were highly correlated with percent omnivores and percent benthic invertivorous individuals, respectively ($r > 0.8$). We also found strong correlations between the percent invertivorous cyprinids metric with the percent benthic invertivorous metric. We chose percent benthic invertivorous individuals for the IBI because a similar metric was successfully used in the IBI for grassland streams in Alberta (Stevens et al. 2006). Correlation analyses also indicated that the number of benthic invertivorous species metric was highly correlated with species richness, and was therefore removed from the final IBI. The remaining metrics ($n = 8$) screened for sensitivity to human disturbance were total catch/100 s, percent long-lived individuals, percent benthic invertivorous individuals, percent omnivores, species richness, percent DELTs, percent litho-obligate individuals, and percent carnivores.

Of the remaining metrics, regression identified percent benthic invertivores, species richness, percent litho-obligates, and percent carnivores as metrics having scores influenced by river size (i.e., the river size coefficient for their regression models had

95% confidence intervals that did not include 0; Table 4). Thus, we calculated standardized residuals from the river size models, versus standardized raw scores, for these metrics.

Table 3. Correlation matrix of candidate metrics for the Battle River Index of Biological Integrity (IBI).

	% Tolerant	**% Intolerant**	**Catch/100 s**	**% Older long-lived**	**# Benthic invert. spp.**	**% Benthic invert.**	**% Omnivores**	**% Invert. cyprinids**	**Species richness**	**% DELTS**	**% Litho-obligates**	**% Carnivores**
% Tolerant	1.00											
% Intolerant	-0.34	1.00										
Catch/100 s	0.23	0.20	1.00									
% Older, long-lived	0.31	-0.12	-0.07	1.00								
# Benthic invert. spp.	-0.17	0.57	0.38	-0.18	1.00							
% Benthic invert.	-0.31	0.92	0.23	-0.14	0.77	1.00						
% Omnivores	0.86	-0.40	0.16	0.28	-0.33	-0.45	1.00					
% Invert. cyprinids	-0.39	0.84	0.32	-0.22	0.73	0.85	-0.47	1.00				
Species richness	-0.03	0.48	0.49	-0.10	0.87	0.67	-0.23	0.62	1.00			
% DELTS	0.54	-0.20	-0.09	0.35	-0.24	-0.24	0.54	-0.29	-0.16	1.00		
% Litho-obligates	0.48	0.35	0.38	0.10	0.55	0.45	0.46	0.45	0.56	0.27	1.00	
% Carnivores	-0.22	-0.32	-0.27	0.01	-0.42	-0.36	-0.25	-0.40	-0.23	-0.04	-0.58	1.00

Note: Highlighted values indicate high correlations (r > 0.80) between metrics.

Table 4. Summary of linear regression models predicting metric values with basin size (ha) for minimally impacted sites (n = 11) in the Battle River, Alberta.

Metric	Y-int.	SE	Coeff.	SE	T	P	R^2
Catch/100 s	1.263	0.592	0.317	0.432	0.73	0.482	0.06
% Older, long-lived fish	0.053	0.110	0.0334	0.0806	0.41	0.689	0.02
% Benthic invertivores*	-0.006	0.088	0.267	0.0644	4.15	0.002	0.66
% Omnivores	0.570	0.102	0.0437	0.0746	0.59	0.573	0.04
Number of species*	4.224	0.858	1.5	0.627	2.39	0.04	0.39
% DELTs	0.274	0.080	-0.069	0.0586	-1.17	0.271	0.13
% Litho-obligates*	0.612	0.100	0.212	0.0728	2.91	0.017	0.48
% Carnivores*	0.453	0.121	-0.212	0.0885	-2.4	0.04	0.39

Note: coefficients (and SEs) were multiplied by 1,000,000 for presentation in table.
*Values were adjusted to remove the effect of river size.

We built, run and ranked models predicting metric responses with measures of human disturbance (Tables 5 and 6). The top models for total fish catch/100 s, percent litho-obligates, percent benthic invertivorous individuals, and percent DELTs were weakly linked to measures of human disturbance ($R^2 < 0.2$; Table 6). Thus, we dropped these metrics from the final IBI. The top model predicting percent long-lived individuals metric was the global model, which had an $R^2 = 0.4$ (Table 5). However, this metric was ambiguous with regards to its relationship with human disturbance based on model-averaged coefficients and related 95% confidence intervals. The percent older, long-lived individuals metric was positively related with DO and negatively related to TDP, but was also positively related to upriver cattle density (10-km scale) and cattle density in basin (Table 7).

Table 5. Model rankings (according to Akaike's Information Criterion Corrected for small sample sizes; AICc), as well as fit of models (R^2), for metrics identified as being unsatisfactory for the Battle River IBI. Codes: WQI = Water Quality Index; GIS = data from geographic information system.

Metric	Model	n	df	LL	AICc	ΔAICc	Akaike weight	R^2
Catch/100 s*	WQI	80	2	4.26	-4.4	8.3	0.01	0.00
	WQI parameters	80	5	8.46	-6.1	6.6	0.03	0.10
	Agriculture	80	5	5.93	-1.1	11.6	0.00	0.04
	Urban (GIS)	80	5	11.8	-12.7	0.0	0.85	0.17
	Urban+access (GIS)	80	3	5.64	-5.0	7.7	0.02	0.04
	Urban+agriculture (GIS)	80	8	11.9	-5.8	6.9	0.03	0.18
	Global model	80	13	19.4	-7.4	5.3	0.06	0.32
% Litho-obligates	WQI	80	2	20.6	-37.1	5.9	0.03	0.04
	WQI parameters	80	5	24.6	-38.3	4.7	0.05	0.13
	Agriculture	80	5	22.6	-34.5	8.5	0.01	0.09
	Urban (GIS)	80	5	26.3	-41.8	1.2	0.30	0.17
	Urban+access (GIS)	80	3	24.7	-43.0	0.0	0.56	0.14
	Urban+agriculture (GIS)	80	8	27.4	-36.7	6.3	0.02	0.19
	Global model	80	13	33.9	-36.3	6.7	0.02	0.31
% Benthic invertivores	WQI	80	2	17.7	-31.3	4.1	0.05	0.01
	WQI parameters	80	5	22.2	-33.6	1.7	0.16	0.11
	Agriculture	80	5	22.3	-33.8	1.5	0.18	0.12
	Urban (GIS)	80	5	20.8	-30.7	4.6	0.04	0.08
	Urban+access (GIS)	80	3	20.8	-35.3	0.0	0.39	0.08
	Urban+agriculture (GIS)	80	8	25.8	-33.5	1.9	0.15	0.19
	Global model	80	13	30.7	-29.9	5.4	0.03	0.28
% DELTs	WQI	80	2	28.7	-53.3	0.0	0.60	0.12
	WQI parameters	80	5	28.1	-45.3	8.0	0.01	0.11
	Agriculture	80	5	28.9	-47.1	6.3	0.03	0.13
	Urban (GIS)	80	5	29.4	-47.9	5.4	0.04	0.14
	Urban+access (GIS)	80	3	29.2	-52.0	1.3	0.31	0.13
	Urban+agriculture (GIS)	80	8	31.7	-45.4	7.9	0.01	0.19
	Global model	80	13	33.6	-35.7	17.6	0.00	0.22
% older, long-lived*	WQI	80	2	-65.6	135	12.8	0.00	0.00
	WQI parameters	80	5	-64.3	139	16.9	0.00	0.03
	Agriculture	80	5	-62.6	136	13.4	0.00	0.07
	Urban (GIS)	80	5	-59.3	129	6.9	0.03	0.15
	Urban+access (GIS)	80	3	-63.3	133	10.5	0.01	0.06
	Urban+agriculture (GIS)	80	8	-56.9	132	9.4	0.01	0.20
	Global model	80	13	-45.5	123	0.0	0.95	0.40

Note: highlighted rows per metric were either models with the lowest AIC value, or models within two units of the minimum value.
*See Table 8 for details on why metric was excluded from IBI.

Table 6. Model rankings (according to Akaike's Information Criterion Corrected for small sample sizes; AICc), as well as fit of models (R^2), predicting scores of IBI metrics for the Battle River IBI.

Metric	**Model**	**n**	**Df**	**LL**	**AICc**	**ΔAICc**	**Akaike weight**	**R^2**
Number of Species	WQI	80	2	5.85	-7.6	10.6	0.00	0.14
	WQI parameters	80	5	14.5	-18.1	0.0	0.98	0.31
	Agriculture	80	5	4.39	2.0	20.2	0.00	0.11
	Urban (GIS)	80	5	6.76	-2.7	15.4	0.00	0.16
	Urban+access (GIS)	80	3	1.79	2.7	20.9	0.00	0.05
	Urban+agriculture (GIS)	80	8	10.2	-2.5	15.7	0.00	0.23
	Global model	80	13	20.8	-10.0	8.1	0.02	0.41
% omnivores	WQI	80	2	1.82	0.5	22.0	0.00	0.19
	WQI parameters	80	5	7.49	-4.2	17.3	0.00	0.29
	Agriculture	80	5	10.5	-10.1	11.4	0.00	0.34
	Urban (GIS)	80	5	14.2	-17.6	3.9	0.12	0.40
	Urban+access (GIS)	80	3	13.9	-21.4	0.0	0.85	0.40
	Urban+agriculture (GIS)	80	8	16.1	-14.2	7.3	0.02	0.43
	Global model	80	13	19.5	-7.5	13.9	0.00	0.48
% carnivores	WQI	80	2	24.7	-45.3	16.0	0.00	0.21
	WQI parameters	80	5	31.1	-51.5	9.8	0.01	0.33
	Agriculture	80	5	24.5	-38.1	23.1	0.00	0.20
	Urban (GIS)	80	5	31.1	-51.3	10.0	0.01	0.33
	Urban+access (GIS)	80	3	31.2	-56.1	5.2	0.07	0.33
	Urban+agriculture (GIS)	80	8	32.9	-47.8	13.5	0.00	0.36
	Global model	80	13	46.4	-61.3	0.0	0.92	0.54
IBI (3 metrics)	WQI	80	2	-42.2	88.5	7.0	0.01	0.34
	WQI parameters	80	5	-37.4	85.6	4.1	0.05	0.41
	Agriculture	80	5	-38.5	87.8	6.3	0.02	0.40
	Urban (GIS)	80	5	-35.7	82.3	0.8	0.26	0.44
	Urban+access (GIS)	80	3	-37.6	81.5	0.0	0.38	0.41
	Urban+agriculture (GIS)	80	8	-32.2	82.5	1.0	0.23	0.49
	Global model	80	13	-26.9	85.3	3.8	0.06	0.55

Note: highlighted rows are models with the lowest AIC value for a particular metric, as well as models within two units of the minimum AIC value.

Table 7. Regression summary of model-averaged coefficients for predicting candidate metrics and the three-metric IBI (number of species, percent carnivores, and percent omnivores) in the Battle River, Alberta.

	Coeff. value	Lower 95% CI	Upper 95% CI	Coeff. value	Lower 95% CI	Upper 95% CI
	Number of species			**Percent carnivores**		
y-intercept	0.265	-0.287	0.816	2.22	1.02	3.42
Water Quality Index	0.00691	0.00308	0.01074	0.00702	0.00399	0.01
Dissolved oxygen	-0.0011	-0.0585	0.0562	-0.0043	-0.0470	0.0384
Total dissolved phosphorous	-2.53	-3.42	-1.64	1.47	0.520	2.43
Total Kjeldahl nitrogen	0.286	0.065	0.507	-0.378	-0.596	-0.160
Nitrite-Nitrate-N	0.442	0.097	0.787	-0.557	-0.881	-0.233
Upriver cattle density	0.239	-0.165	0.643	-0.195	-0.487	0.096
Upriver cropland cover	-0.00467	-0.01128	0.00194	-0.00437	-0.00917	0.00043
Cattle density in basin	-0.203	-0.883	0.478	0.503	0.010	0.997
Riparian Intactness Index	0.00097	-0.00364	0.00558	0.00262	-0.00080	0.00603
Urban cover in basin	0.125	-0.023	0.272	0.099	-0.009	0.207
Upriver distance to 'lagoon'	0.00218	-0.00122	0.00559	0.00214	-0.00036	0.00465
Upriver distance to town	0.00241	-0.00347	0.00828	-0.00343	-0.00779	0.00093
Human density < 10 km radii	-0.017	-0.684	0.651	-0.440	-0.903	0.022
Road density in basin	0.017	-0.112	0.146	-0.141	-0.233	-0.049

	Coeff. value	Lower 95% CI	Upper 95% CI	Coeff. value	Lower 95% CI	Upper 95% CI
	Percent omnivores			**Percent older, long-lived***		
y-intercept	-1.59	-2.32	-0.868	-3.77	-7.38	-0.151
Water Quality Index	-0.00867	-0.01269	-0.00464	-0.00184	-0.0112	0.00751
Dissolved oxygen	-0.0077	-0.0680	0.0526	0.146	0.0118	0.2811
Total dissolved phosphorous	0.679	-1.09	2.45	-4.4	-7.35	-1.46
Total Kjeldahl nitrogen	0.047	-0.249	0.344	0.501	-0.181	1.184
Nitrite-Nitrate-N	-0.271	-0.737	0.195	-0.379	-1.394	0.636
Upriver cattle density	0.042	-0.373	0.457	1.216	0.291	2.142
Upriver cropland cover	0.00375	-0.00269	0.0102	-0.00119	-0.01631	0.01392
Cattle density in basin	-0.276	-1.061	0.510	1.620	0.062	3.179
Riparian Intactness Index	-0.00100	-0.00282	0.00081	0.022	-0.256	0.3
Urban cover in basin	0.031	-0.085	0.146	0.328	-0.012	0.668
Upriver distance to 'lagoon'	-0.00185	-0.00458	0.00087	-0.00283	-0.01080	0.00513
Upriver distance to town	0.00266	-0.00184	0.00716	-0.0111	-0.0252	0.00304
Human Density < 10 km radii	-0.348	-0.922	0.225	-0.162	-1.668	1.344
Road density in basin	0.174	0.114	0.235	0.021	-0.258	0.301

Table 7. Continued.

	Coeff. value	Lower 95% CI	Upper 95% CI	Coeff. value	Lower 95% CI	Upper 95% CI
		IBI				
y-intercept	4.86	2.59	7.12			
Water Quality Index	0.0226	0.0156	0.0296			
Dissolved oxygen	0.0168	-0.0915	0.1251			
Total dissolved phosphorous	-3.16	-7.28	0.961			
Total Kjeldahl nitrogen	-0.037	-0.574	0.500			
Nitrite-Nitrate-N	0.293	-0.678	1.264			
Upriver cattle density	-0.115	-0.865	0.634			
Upriver cropland cover	-0.0112	-0.0222	-0.00016			
Cattle density in basin	0.485	-0.825	1.8			
Riparian Intactness Index	0.00231	-0.00173	0.00635			
Urban cover in basin	0.134	-0.105	0.373			
Upriver distance to 'lagoon'	0.00401	-0.00171	0.00973			
Upriver distance to town	0.00157	-0.00752	0.01066			
Human density < 10km radii	-0.109	-1.201	0.983			
Road density in basin	-0.329	-0.481	-0.177			

Note: highlighted rows indicate confidence intervals that do not overlap with zero.
*Raw scores of percent older, long-lived individuals metric were transformed with log(x + 0.01).

The remaining candidate metrics were species richness, percent omnivores, and percent carnivores. The top model for the species richness metric was the Water Quality model ($R^2 = 0.31$; Table 6). This metric was positively related to the Water Quality Index and negatively related to TDP (Table 7). Although regression indentified species richness as moderately ambiguous (high species richness was also related to high concentrations of TKN and NO_2+NO_3-N), we retained this metric for the final IBI (Table 7). The top model predicting percent omnivores was the urban-access model ($R^2 = 0.4$; Table 6). Percentage of omnivores was negatively related to the Water Quality Index, and positively related to road density in basin (Table 7). Finally, the top model predicting percent carnivores was the global model ($R^2 = 0.54$; Table 6). Percentage of carnivores was positively related to the Water Quality Index, and negatively related to TKN, NO_2 + NO_3-N, as well as road density in basin (Table 7). However, high proportion of carnivores was also related to high concentrations of TDP and high densities of cattle in basin.

Based on the above results, species richness, percent omnivores, and percent carnivores were included in the final IBI (Tables 6 and 7). Top models predicting the three-metric IBI were the urban model, the urban-access model, and the urban-agriculture model (R^2 = 0.41 - 0.49; Table 6). Critical parameters predicting IBI scores were the Water Quality Index, percent upriver cropland cover (< 10-km scale), and road density in basin. We identified significant relations between high IBI scores with high values of the Water Quality Index, but with low percent upriver cropland cover (< 10-km scale) and low densities of roads in basin. Road density alone was an excellent predictor of IBI scores (t_{79} = -7.39, $P < 0.001$; $R^2 = 0.42$; Figure 2). The regression model predicted that a one-unit increase in road density (m/ha) results in a 0.317 unit decrease in IBI scores (i.e., 10.6% decrease in river condition or health). Further, the road density threshold identifying a zero IBI score was approximately ≥ 16.4 m/ha, whereas the threshold identifying a perfect IBI score of three was approximately ≤ 6.9 m/ha. Finally, it is important to note that the residuals (i.e., error) from the univariate road density model do not indicate strong spatial autocorrelation (Figure 3).

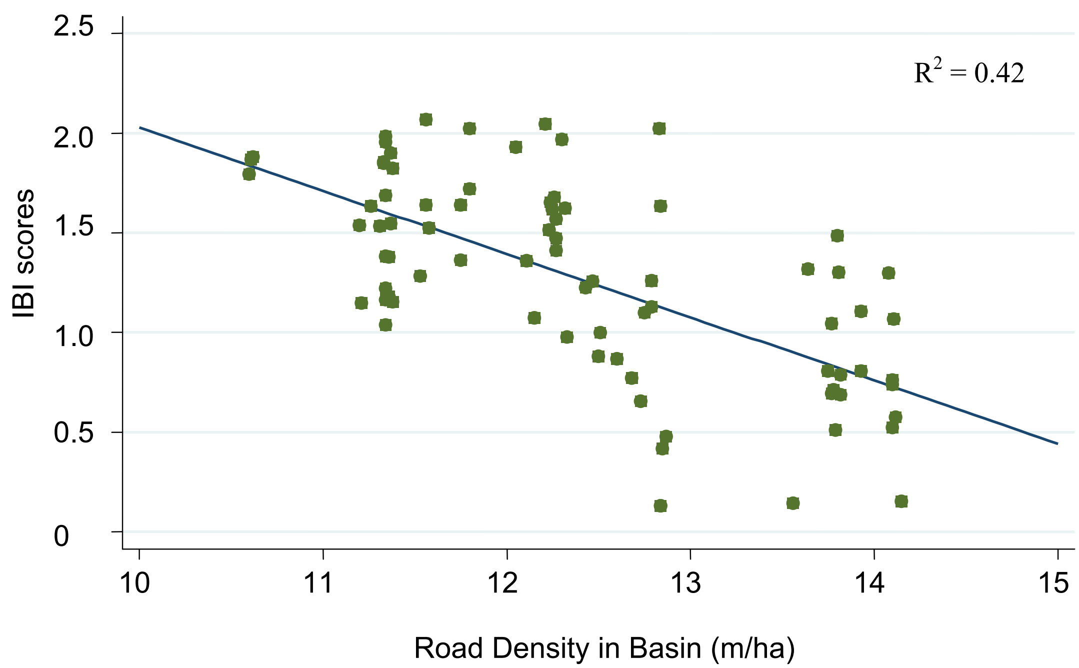

Figure 2. Linear univariate regression predicting scores of a three-metric IBI (percent omnivores, percent carnivores, number of species) with road density (m/ha) in basin: y = 5.196 - (0.3169) * road density.

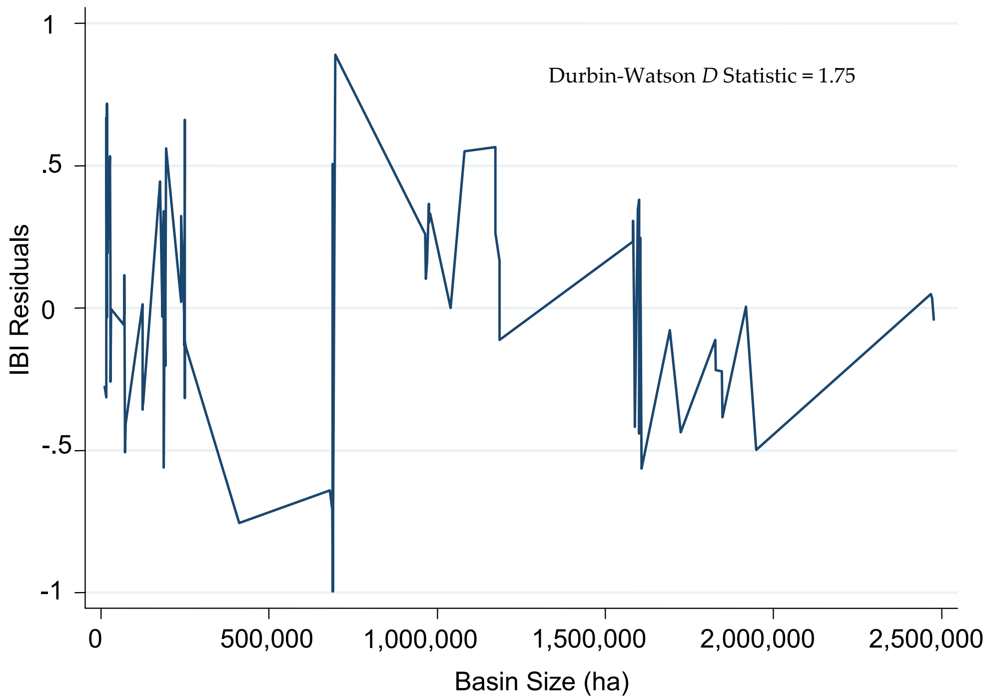

Figure 3. IBI residuals plotted against basin size. Residuals are from a univariate regression model predicting scores of a three-metric IBI (percent omnivores, percent carnivores, number of species) with density (m/ha) of roads in basin.

5.0 DISCUSSION

Few studies conduct a rigorous process of metric selection: *a posteriori* tests of their sensitivity, redundancy, and consistency (Roset et al. 2007), as done for the Battle River assessment. Further, despite the inherent challenges of a fish fauna comprising only 19 species and a limited understanding of the tolerance of the study species to perturbation in northern rivers, we have developed a unique three-metric IBI for assessing the ecological condition of rivers in Alberta's prairie-parkland ecoregion. Other IBI programs may comprise as many as 10 metrics or more, but quite often metrics are redundant and not necessary (Roth et al. 1998; Roset et al. 2007). A three-metric IBI can be just as accurate and precise in classifying degraded water courses (Roth et al. 1998). In this study, metrics described community structure and function, and included species richness and trophic guilds (i.e., percent carnivores and percent omnivores). Integration of the three metrics resulted into an IBI closely linked to human land-use patterns, specifically road networks.

Of the three metrics, species richness may be the most common metric in IBI programs, because it most consistently relates to site quality, and seems to be widely useful for assessing water courses in North America (Angermeier et al. 2000). Similarly, we showed that in the Batter River more species occur in river sections with low concentrations of phosphorus. Trophic guilds, which have been consistently integrated in previous IBI programs, were also used in the Battle River IBI. In general, perturbation to the aquatic environment will impact negatively on species with specialist feeding requirements, but will favour those with flexible or diverse feeding behaviours (Noble et al. 2007). This study found that the relative abundance of carnivores in a fish catch was high when concentrations of nitrogen and road densities in basins were low, whereas high proportion of omnivores occurred in river sections having high densities of roads in basins.

Important to note, the Battle River assessment method did not support use of reproductive guilds, fish condition, and habitat guilds. Although reproductive guild classifications, particularly the lithophilic guild, have been commonly used in previously published IBIs (e.g., Bramblett et al. 2005; Noble et al. 2007), we found lithophils to be unrelated to measures of human disturbance, suggesting that current observed levels of river degradation do not result in further loss of their critical spawning habitat. Further, we recorded a positive relationship between the relative abundance of lithophils and concentrations of nitrogen (TKN; results not shown). However, the lithophil reproductive guild could be useful if re-defined as proportion of lithophils, excluding tolerants, as done in Angermeier et al. (2000). Additional research is required to help identify our study species' tolerance to human disturbance. In addition to the lithophils, the fish condition metric (i.e., the relative abundance of fish with DELTs) was not supported by the Battle River assessment method, even though DELTs were common and the percentages of fish with DELTS were variable among sites (i.e., 75% of sites had fish with DELTs; mean proportion of fish with DELTs across sites = 24%, coefficient of variation, CV = 75%). It is important to note that the criteria employed in other IBI assessments did not support use of DELTs (e.g., Lyons et al. 2001; Mebane et al. 2003), and that European indices do not include fish condition as a metric

(Pont et al. 2006; Roset et al. 2007). A possible explanation as to why some metrics that were rejected by the Battle River assessment method have been successfully integrated into other programs is that our catch consisted predominately of white sucker. For example, although Karr and Dudley (1981) proposed that the status of long-lived species could integrate disturbances to the aquatic environment over multiple years, some species, such as the white sucker, may have great plasticity and may adapt their life histories to survive under different conditions (Noble et al. 2007).

Our study resulted in an IBI closely linked to human land-use patterns, specifically road networks. Road density in basins was a good predictor of IBI scores, and may influence the integrity of fish assemblages through a variety of mechanisms, such as contamination, pollution, hydrologic alteration, fragmentation from improperly used culverts plus stream channelization, and elimination of nursery habitat (Allan 2004; Wheeler et al. 2005). Road density is clearly a surrogate for a variety of anthropogenic effects, and thereby is a simple measure of the cumulative human footprint. The physical habitat and chemical environment of a stream are largely products of land cover types and human activities in its catchment. The relationship of road networks with urban development is intuitive (Pearson r = 0.75 for percent urban cover in basin and road density in basin for sites on the Battle River), and as a road networks, suburban developments and urban cover grows, changes are anticipated to occur in river habitat, water chemistry and in the integrity of fish assemblages. Wheeler et al. (2005) contends that the greatest damage to stream health is inflicted by the cumulative effects of building new roads and highways through relatively undeveloped watersheds, which may become subject to urban and suburban sprawl. Similar to the presence of hydraulically-linked networks of roads, urban development continually affects streams and causes extensive and chronic impacts to chemistry and hydrology, but often at greater magnitudes than other land-use types (Wheeler et al. 2005). For example, previous research has shown that very low levels of urban land cover (8 - 10%) can result in highly altered fish communities (Wheeler et al. 2005). The highest level of urban cover in our study basins in the Battle River drainage was much lower than this threshold (approximately 2%), approaching low intensity development (Wheeler et al. 2005). It is important to note, however, that suburban cover was not quantified, and that measurements of road networks may provide an indirect measurement of both suburban and urban cover. The Battle River IBI identified changes in IBI scores at road densities as low as 7 m/ha.

As river gradients lessen and as the water temperature and river size increases, fish assemblages change and species richness typically increases (Vannote et al. 1980; Mebane et al. 2003). Such changes potentially mask impacts of urban developments and agriculture. Importantly, the Battle River assessment discriminated downriver anthropogenic effects from characteristics of the natural continuum, necessary for providing an effective tool to guide river management and restoration. In the Battle River drainage, livestock densities were highest in the upper reaches, whereas cropland cover was most prevalent downriver toward the Saskatchewan border. It is possible that the effects of cropland and livestock (at the basin-scale) on river condition could not be detected given that they were highly correlated with river size, and that the influence of river size was removed during IBI development (i.e., from metrics). More sampling points from multiple rivers in the region should be considered for a more comprehensive IBI program. In doing so, relationships between river condition with cropland cover and livestock densities can be confirmed. However, for the purposes of monitoring effects of agriculture, the present IBI will be useful as scores were responsive to changes in upriver cropland cover along the Battle River at 10-km scales.

In summary, the Battle River IBI could be applied as a rapid assessment tool to characterize aquatic ecosystem health. It provides a single, defensible, easily understood measure of the health of the river reach in question. The IBI can also be used to evaluate specific management activities to restore river ecosystems (e.g., road density targets). However, multiple IBI samples (i.e., fish catches) in space and time in conjunction with physical and chemical descriptions of reaches are recommended to precisely define the extent of river degradation, and to detect major shifts in river condition. Whether the Battle River can be generalized to other sub-basins in the prairie-parkland ecoregion will require additional sampling in new catchments, verification of scores, and if necessary, recalibration of models (see Hughes et al. 1998; Pont et al. 2006). A possible advantage of a provincial IBI program based on the centrally-located Battle River is that many of the study species are distributed throughout the province (Nelson and Paetz 1992), suggesting that the proposed IBI may be applicable beyond regional watercourses. It is recommended that additional research on ecological functions and requirements of species in northern systems be conducted to strengthen the basic foundation of guild-based bioassessment methods in Alberta.

6.0 LITERATURE CITED

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7.0 APPENDIX

Appendix 1. Fish species in the Battle River and a description of their ecological characteristics (based on Bramblett et al. 2005 and Simon 1999).

Taxon	Species abbrev.	Trophic status[a]	Forage habitat[b]	Repro. Class[c]	Long-lived (>10 y)	Captured this study	Historical obs. only
Catostomidae							
Longnose sucker (*Catostomus catostomus*)	LNSC	IN	BE	LO	X	X	
Quillback sucker (*Carpoides cyprinus*)	QUIL	OM	BE	LO			X
Shorthead Redhorse (*Moxostoma macrolepidotum*)	SHRD	IN	BE	LO	X	X	
White sucker (*Catostomus commersoni*)	WHSC	OM	BE	LO	X	X	
Cyprinidae							
Emerald Shiner (*Notropis atherinoides*)	EMSH	IN	WC	PELA[d]			X
Fathead minnow (*Pimephales promelas*)	FTMN	OM	GE	TR		X	
Lake chub (*Couesius plumbeus*)	LKCH	IN	WC	LO		X	
Longnose dace (*Rhinichthys cataractae*)	LNDC	IN	BE	LO		X	
Spottail shiner (*Notropis hudsonius*)	SPSH	IN	WC	LO		X	
Esocidae							
Northern pike (*Esox lucius*)	NRPK	CA	WC	PHYTO[d]	X	X	
Gadidae							
Burbot (*Lota lota*)	BURB	CA	BE	LO	X	X	
Gasterosteidae							
Brook stickleback (*Culaea inconstans*)	BRST	IN	GE	TR			X
Hiodontidae							
Goldeye (*Hiodon alosoides*)	GOLD	IN	WC	LO		X	
Mooneye (*Hiodon tergisus*)	MON	IN	WC	LO		X	
Percidae							
Iowa darter (*Ethostoma exile*)	IWDR	IN	BE	PHYTO[d]		X	
Walleye (*Sander vitreus*)	WALL	CA	GE	LO	X	X	
Yellow perch (*Perca flavescens*)	YLPR	IC	WC	PHYTOLITH[d]			X

Appendix 1. Continued.

Taxon	Species abbrev.	Trophic status[a]	Forage habitat[b]	Repro. Class[c]	Long-lived (>10 y)	Captured this study	Historical obs. only
Percopsidae							
Trout-perch (*Percopsis omiscomaycus*)	TRPR	IN	BE	LO		X	
Salmonidae							
Lake whitefish (*Coregonus clupeaformis*)	LKWH	IN	BE	LO	X		X

[a]CA = carnivore (>90% fish or other invertebrates); IC = invertivore-carnivore (> 25% both invertebrates and vertebrates); IN = invertivore (> 75% invertebrates); OM = omnivore (25 - 90% plants or detritus); [b]BE = benthic; GE = generalist, WC = water column; [c]LO = litho-obligate; TR = tolerant reproductive strategists; [d]PELA = pelagophils; PHYTO = phytophil; PHYTOLITH = phytolithophil.

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