Water Qual. Res. J. Can. 2010 · Volume 45, No. 1, 35–46

Influences of Human Stressors on Fish-Based Metrics for Assessing River Condition in Central Alberta

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Economic developments in Alberta have resulted in widespread changes in land use that may deteriorate river conditions for fish. Fish assemblages were characterized with index of biological integrity metrics for the heavily-developed watershed of the Battle River, Alberta. Metric relationships with human stressors were quantified using regression and information theory methods. Although the fauna comprised 14 native species, 50% of the catch was white sucker (Catostomus commersoni Lacepede, 1803). Five statistically unrelated metrics were identified as being responsive to stressors: two trophic guilds, one habitat guild, one reproductive guild, and one measure of community structure. Regression showed that the cumulative effect of human developments, indexed as road density in the basin, was negatively linked to the relative abundance of lithophils and positively linked to the relative abundance of omnivores. Agriculture also threatened the integrity of fish assemblages. Stream sections with higher cattle densities in their basins had fewer lithophils and benthic invertivores; whereas stream sections with higher nutrient concentrations contained fewer species, as well as fewer top carnivores, but more true omnivores. Understanding effects of human footprints that are expanding in western Canada will be critical to the successful management of aquatic resources.

Key words: **fish assemblages, biological integrity, land-use stressors, agriculture, urban development, modelling**

Introduction

The majority of landscapes in Alberta have undergone rapid changes over the last century due to growths in agriculture and petroleum-based economic activities. In the prairie-parkland region of central Alberta, as much as 98% of the landscape may have been converted to agriculture (Timoney and Lee 2001). These changes combined with recent growths in urban and suburban developments have led to cumulative land-use modifications in the majority of drainage basins (Timoney and Lee 2001). Existing, as well as proposed road networks, suburban developments, petroleum pipelines, and related infrastructure across the landscape pose serious threats to the biological integrity of aquatic ecosystems in the region. Indeed, there is widespread recognition of the extent and significance of changes in land use and cover worldwide (Allan 2004), which has led to an increase in studies that seek to develop tools for monitoring ecosystems and to establish relationships between land use and aquatic condition (e.g., Hughes et al. 1998; Daniels et al. 2002; Mebane et al. 2003; Bramblett et al. 2005; Pont et al. 2009). Knowledge of the relationships between land use and aquatic conditions can be used to predict the extent of change in river ecosystems in response to human development and plausible alternative futures (Allan 2004; Pont et al. 2009).

Although a variety of biological indices are available for evaluating aquatic health and ecological conditions, multimetric indices such as the "index of biological integrity" (IBI) have been particularly successful as a monitoring tool (e.g., Hughes et al. 1998; Karr and Chu 1999; Lyons et al. 2001; Daniels et al. 2002; Bramblett et al. 2005; Pont et al. 2009). Multimetric indices reflect various components of biological assemblages, including taxonomic richness, habitat and trophic guild composition, and individual health and abundance. The process of selecting measurable attributes that provide reliable and relevant signals about the ecological impacts of human activities is central to making multimetric indices effective (Karr and Chu 1999). Importantly, the characteristics of biotic assemblages can change from region to region such that metrics used for assessing streams in the midwestern United States, for example, may not be applicable to the prairie-parkland region in Canada (Hughes et al. 1998; Angermeier et al. 2000). One of the obvious challenges with fish-based assessments of northern streams and rivers in western Canada is the naturally depauperate fauna. For example, Alberta includes portions of three major drainage basins (Arctic, Hudson Bay, and Mississippi River drainages), yet has only 52 native species of fishes (Nelson and Paetz 1992). The presence of fewer species reduces the number of potential candidate metrics to characterize assemblages. Further, the biota of northern rivers is thought to be dominated by habitat, trophic, and reproductive generalists adapted to unstable flow regimes with harsh, fluctuating environmental conditions (Dodds et al. 2004; Bramblett et al. 2005).

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The primary objective of our study was to better understand influences of human-related stressors, operating at various spatial scales, on functional and structural attributes of the fish assemblage (i.e., metrics) for a river in central Alberta. Fish assemblage data was used to describe the biotic assemblage, a common approach in monitoring conditions of aquatic ecosystems, because fish are responsive indicators of ecological status (Karr and Chu 1999; Daniels et al. 2002; Mebane et al. 2003; Bramblett et al. 2005; Pont et al. 2009). Further, national and provincial regulations call specifically for the protection of fish and fish habitat. The selection of fish assemblage metrics was based on our catch data, published information on species habitat requirements and life history, and a review of the applicability of metrics that were successfully used in previously published IBIs (e.g., Karr and Chu 1999; Bramblett et al. 2005; Noble et al. 2007). Using regression and information theory methods, multiple hypotheses on fish assemblage responses to human stressors were examined, including effects of nutrient loading, effects of conversion of the landscape in adjacent valleys to agricultural cover, and the effects of cumulative impacts of road networks, petroleum access, and urban developments. A secondary objective was to determine which of the examined fish assemblage metrics may be useful for assessing river conditions as part of a regional IBI monitoring program in Alberta.

Methods

Study Area and Design

The study system was the Battle River in central Alberta (Fig. 1). Unusual for Alberta, this river does not originate in the Rocky Mountains, but rather has its entire watershed contained within the heavily-developed agricultural prairie-parkland ecozone of Alberta (Strong and Leggat 1992). The headwaters begin near Battle Lake (52°55'N, 114°10'W) and the river flows eastward through the prairie-parkland region for approximately 880 fluvial km to the Saskatchewan border (52°51'N, 109°59'W) (Fig. 1). The Battle River then joins the North Saskatchewan River in Saskatchewan. 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. The river flows across the Edmonton-Red Deer-Calgary development corridor. This area (Census Regions 6, 8, and 11) comprised 72% of the Alberta population in the 2001 census and was one of the four densest concentrations of people in Canada (Statistics Canada 2003). Also, two water control structures occur on the river for municipal water storage and to supplement downstream flows when the river is low. The historical fish assemblage of the Battle River is comprised of 19 native fish species (Nelson and Paetz 1992).

Study sites were selected to represent the full spatial extent of the river and conditions along it (Fig. 1). However, sampling locations and the final sample size were influenced by logistical constraints and the availability of safe launches for the electrofishing boat. In the upper section of the Battle River, 40 sites were selected between Battle and Driedmeat lakes (286 fluvial km) to represent possible influences of the Edmonton-Calgary development corridor and related urban features. Downstream of this region 10 sites were selected along the length of the Battle River, from the water control structure on Driedmeat Lake to the Forestburg Reservoir (116 fluvial km). The upper subbasins of the Battle River typically support more livestock compared with lower subbasins where annually cultivated cropland is more prevalent on the landscape (Stevens and Council 2008; this study). In the lower sections, 14 sites were selected along the third reach, defined from the water control structure on the Forestburg Reservoir to the western boundary of Canadian Forces Base (CFB) Wainwright (210 fluvial km), and 20 sites from the western boundary of CFB Wainwright to the Alberta-Saskatchewan border (200 fluvial km). In general, most sites were affected by agricultural activities, although sites in the vicinity of CFB Wainwright were identified as being minimally-disturbed sites prior to field work (Fig. 1).

At each site, a 1- or 2-km sample section (nonwadeable) was identified as a discrete sampling unit. One kilometre sections were used for sampling the upper reaches of the Battle River, whereas 2-km sections were used in the lower reaches to adequately estimate species richness and relative abundance in those waters. 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 (Table 1). 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).

Sampling Methods

Fish were captured by electrofishing using 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. Electrofishing sampling effort was recorded and was defined as seconds (s) of time the anode was "alive" with electricity while in the water (plus for short periods in the air when the anode was being thrown from the boat). 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. At each site, sampling was conducted in a downstream direction and in 500-m subsections, such that captured individuals were held for relatively short periods, and were released approximately 100 m upstream prior to sampling the remainder of the study site. All captured individuals were identified to species and were measured for weight and fork length. All

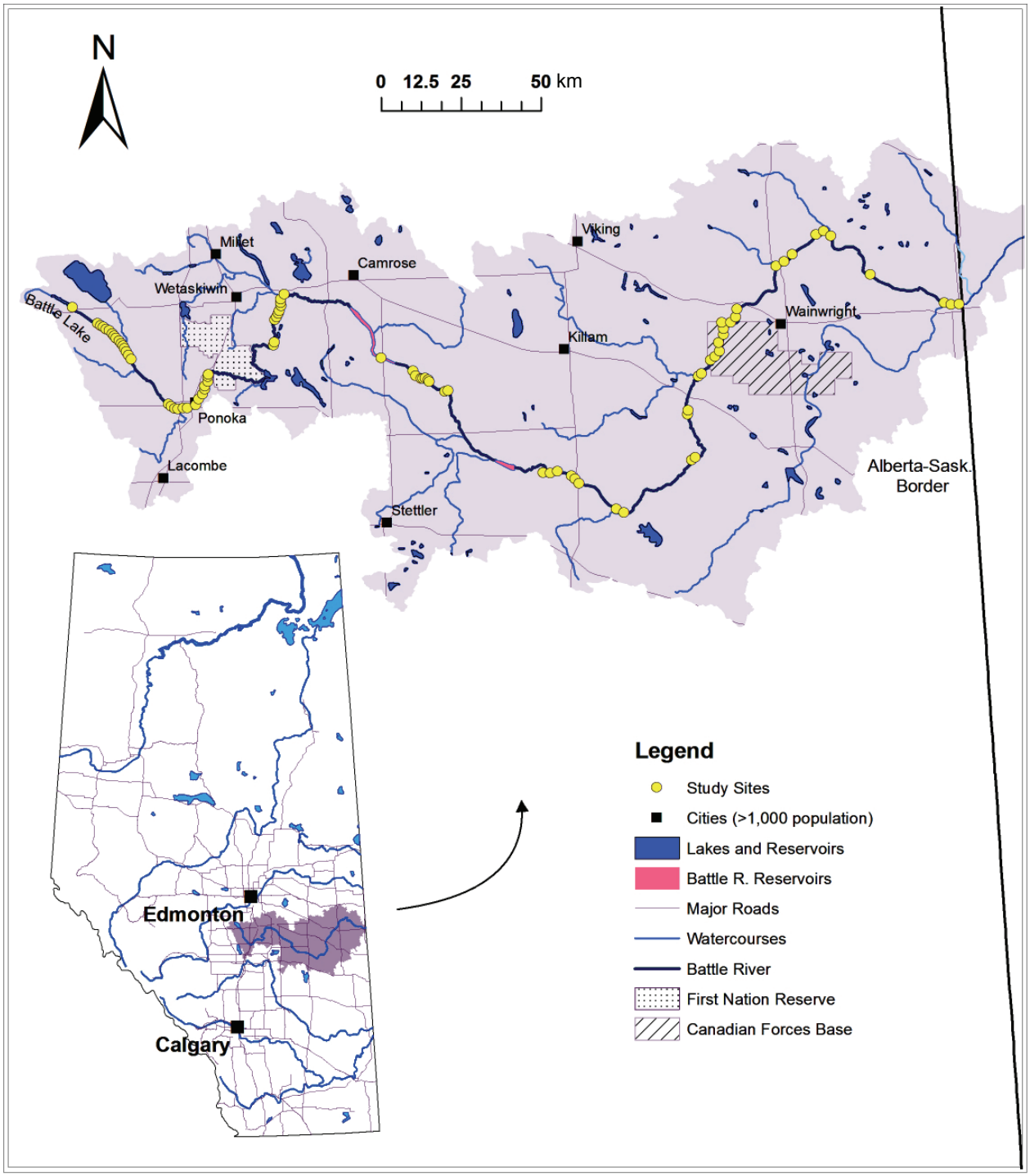

Fig. 1. Map of Alberta and the Battle River, its basin and locations where data on fish assemblage metrics were collected during 2006 and 2007.

individuals were examined for deformity and disease, eroded fins, lesions, and tumors, termed "DELTs" (Daniels et al. 2002; Mebane et al. 2003).

At each electrofishing site, standardized descriptions of instream habitat (e.g., water depth, dominant substrate type) and riparian areas were made (see Stevens and Council 2008). Riparian conditions were described by integrating measurements of the percentage of cover having deep-rooted vegetation (such as tree and shrub species), the percentage of cover with multiple age classes of woody plants, including young classes, and the percentage of shorelines that were incised and

TABLE 1. Summary statistics of focal variables used for describing Battle River sections sampled for fish during 2006 and 2007 (n = 80)

Variables	*Mean*	*SD* [a]	*Min*	*Max*	*CV* [a] *(%)*
RIPARIAN VEGETATION INDEX (%) [b]	67.1	21.3	8.3	93.3	31.7
%cover holding deep-rooted vegetation	72.6	22	5	100	30.3
%cover with tree establishment and regeneration	59.5	31.5	0	96	52.9
%unstable shoreline	30.7	22.1	0	85	71.9
WATER QUALITY VARIABLES					
Dissolved oxygen (DO; mg/L)	7.77	0.93	4.13	11.28	11.9
pH	8.29	0.14	7.8	8.6	1.7
Total phosphorous (TP; mg/L)	0.177	0.09	0.025	0.382	51.1
Total Kjeldahl nitrogen (TKN; mg/L)	1.49	0.27	0.88	1.98	18.4
Ammonia-nitrogen (NH_3-N mg/L)	0.053	0.033	0.008	0.118	61.1
Nitrite+nitrate nitrogen (NO_2+NO_3-N; mg/L)	0.111	0.217	0.003	0.792	194.3
LANDSCAPE VARIABLES					
Cattle density per ha in basin	0.453	0.104	0.317	0.655	23
Manure application rate in basin (t per ha)[c]	4.34	1.17	2.78	6.65	27.1
%adjacent agricultural cover, 10-km upriver	28.3	13.1	6.2	52.4	46.5
%agricultural cover in basin	26.2	15.4	0.5	44.7	58.6
%adjacent cropland cover, 10-km upriver	19.5	11.5	2.3	46.5	59
%cropland cover in basin	21	13	0	35.4	61.9
%urban cover in basin	0.7	0.7	0	2.2	98.8
Road density in basin (m/ha)	12.4	1	10.6	14.2	8.3
Basin size (ha)	775,165	734,536	11,096	2,478,024	94.8
Mean wetted width (m)	19.9	7.4	9.3	57	37.3
Mean maximum depth (m)	1.9	0.6	1	4	29.8

[a] SD = standard deviation; CV = coefficient of variation.
[b] Integrates measurements of riparian vegetation (i.e., calculated as the average of %cover deep-rooted vegetation, %cover with tree establishment and regeneration, and % stable shoreline).
[c] Derived from cattle density estimates by Knaga and Makowecki (2008).

showing signs of instability (Table 1). Water quality samples were collected during mid- to late-June 2007, after electrofishing, at the start point for each sampling site. Samples were submitted to a CAEAL (Canadian Association for Environmental Analytical Laboratories) accredited laboratory within 24 hrs of collection for analysis. Laboratory tests included those for dissolved nitrite plus nitrate as nitrogen (NO_2+NO_3-N; mg/L), total Kjeldahl nitrogen (TKN; mg/L), pH, total phosphorous (TP; mg/L), and total ammonia as nitrogen (NH_3-N; mg/L). Field meters were also used to record dissolved oxygen (DO) (mg/L), pH, and conductivity (µS/cm) at the approximate time and location of the sampling.

Using ArcGIS 9.2 and a provincial digital elevation model (DEM; 1:20,000), study watersheds were delineated and basin sizes were calculated at each study site. The adjacent landscapes were also characterized at the spatial scale of 10-km upriver by 5-km wide, perpendicular from the stream bank. The upriver starting point for a study site was the drainage point for basin and subbasin delineations. Next, multiple GIS (geographic information system) layers were obtained to create variables describing human land-use activities and stressors. These layers included the provincial cropland and hay pasture insurance database from Alberta Agriculture Financial Services (based on 2007 information), and a livestock (cattle and calves) database from Alberta Agriculture, Food, and Rural Development (based on 2001 census). The majority of agricultural cover in the study basin was in cropland (Table 1). Spatial data on human settlements were obtained from Alberta Sustainable Resource Development (ASRD; based on 2007 coverage), specifically urban cover for municipalities with a population greater than 1,000, with a majority of the buildings on parcels of land less than 1,850 m^2. As a correlate of cumulative effects of urban-related stressors and petroleum infrastructure, spatial data on Alberta road networks of gravel and paved access routes were also obtained, including routes to and from approximately 15,000 oil and gas well sites in the watershed. Using these layers, upriver land-use activities were quantified at two spatial scales per study site: basin-wide measurements for all land-use stressors, plus smaller-scale (i.e., subbasin) measurements of agricultural cover, 10 km upriver from the study site by 5 km, perpendicular to the stream bank (Table 1). Biological responses can be expected to change with varying spatial scales in which the stressor may operate (Allan 2004). Further, land use in riparian areas may not mirror land-use trends throughout the drainage basin. Urban cover, in general, was not in close proximity

to the study river (C. E. Stevens, unpublished data), and therefore urban cover was assessed at the basin-level only. Also, the assessments of cattle densities were limited to the basin-level because of the coarse-scale at which these measurements were obtained.

Metric Evaluation

The composition of the fish catch, as well as the life history requirements of the study species were considered in the selection of candidate IBI metrics (Nelson and Paetz 1992; Simon 1999; Bramblett et al. 2005). Candidate metrics reflected various functional guilds and structural components of ecosystems and were similar to those successfully used in other IBIs (Karr and Chu 1999; Bramblett et al. 2005; Noble et al. 2007). Further, candidate metrics included those adopted in systems that were geographically near to Alberta and the Battle River; for example, proportion of lithophils and long-lived individuals were used in streams in Montana (Bramblett et al. 2005). "Tolerance to disturbance" metrics were based on rankings (1 to 10) of study species in Whittier et al. (2007). For the Battle River, study species with ranks >8 were identified as being tolerant species (i.e., fathead minnow [*Pimephales promelas* Rafinesque, 1820], mooneye [*Hiodon tergisus* Lesueur, 1818], and goldeye [*Hiodon alosoides* Rafinesque, 1819]), and species with ranks <6 were identified as being intolerant species (i.e., longnose sucker [*Catostomus catostomus* Forster, 1773], lake chub [*Couesius plumbeus* Agassiz, 1850], and burbot [*Lota lota* Linnaeus, 1758]). For tolerance and guild-based metrics, units (i.e., percent relative abundance, number of species) were selected upon consideration of the catch data and inherent limitations of our dataset. In total, 12 candidate metrics were hypothesized as being either positive- or negative-scoring IBI metrics (Table 2). Positive-scoring metrics were those that increase in value as biological integrity increases, whereas negative-scoring metrics were those that decrease in value as biological integrity decreases (Karr and Chu 1999).

Statistical analyses began with identifying metrics that were statistically redundant (Hughes et al. 1998; Lyons et al. 2001). In other words, with a future multimetric index in mind, recommended metrics were those that were only weakly correlated with each other (Pearson $r < 0.8$). Next, candidate metrics were screened for responsiveness to human stressors using multivariate regression and an information-theoretic approach that ranked a priori models (Burnham and Anderson 2002). Importantly, if a pair of covariates had an $r > 0.8$, one of the two covariates were considered for removal from the proposed model to minimize multicollinearity. Of the water quality variables, NH_3-N was positively correlated with TP, and therefore was removed from models (Pearson $r = 0.83$). It is important to note that basin size was negatively correlated with cattle density in the basin (Pearson $r = -0.82$), but positively correlated with percent agricultural cover in the basin (Pearson $r = 0.91$). Thus, for each candidate metric, six a priori hypotheses or models were developed considering the stressor covariates and their relationships with one another. To minimize multicollinearity, agricultural and global models included either basin area or variables measuring agricultural stressors in basins. All models with the exception of one agricultural model (Model 3) and one global model (Model 6) comprised basin area as a proxy of both stream size and position (see below). Stream size can be an important factor structuring fish assemblages (e.g., Karr and Chu 1999; Fischer and Paukert 2008); whereas stream position can be an important determinant of fish assemblages on systems fragmented by dams (Hitt and Angermeier 2008). The following six models were tested:

Model 1. Water quality model = Basin area, NO_2+NO_3-N, TKN, TP, and DO.

Model 2. Agricultural model I = Basin area, percent adjacent agricultural cover, and riparian vegetation index.

Model 3. Agricultural model II = Livestock density in basin, percent agricultural cover in basin, percent adjacent agricultural cover, and riparian vegetation index.

Model 4. Road model = Basin area and road density in basin.

Model 5. Global model I = Basin area, NO_2+NO_3-N, TKN, TP, DO, percent adjacent agricultural cover (10-km upriver), riparian vegetation index, and road density in basin.

Model 6. Global model II = NO_2+NO_3-N, TKN, TP, DO, livestock density in basin, percent agricultural cover in basin, percent adjacent agricultural cover, riparian vegetation index, and road density in basin.

Postestimation procedures for all regressions included the Cook-Weisberg heteroscedasticity test. If errors were heteroscedastic, robust regression was used, which is an iterative procedure that reweights the observations so that highly influential ones are down-weighted (Davidson and MacKinnon 1985; Hoffman 2004). Also, variance inflation factors (VIFs) were reviewed as a postestimation procedure for ensuring that multicollinearity problems were avoided (Hoffman 2004). These tests resulted in percent urban cover being removed from the road model and subsequent analyses. Importantly, this does not imply that urban cover is unimportant as a stressor, but rather that road density may be an adequate descriptor of urban cover (Pearson $r = 0.75$ for percent urban cover and road density in subbasins of the Battle River).

Akaike's information criterion, corrected for small sample sizes (AIC*c*), was used as a basis to select models (Burnham and Anderson 2002). Primary inferences were drawn from the best model ($AIC_{c_{min}}$) and others within

TABLE 2. Fish assemblage (IBI) metrics examined, including means and ranges from 2006 and 2007 fish catch data collected on the Battle River, Alberta (n = 80)[a,b]

Candidate Metric	*Description*	*Mean (range)*
POSITIVE-SCORING	*VALUES INCREASE WITH INCREASING BIOLOGICAL INTEGRITY*	
Species richness	A decline in taxa richness is generally one of the most reliable indicators of degradation or disturbance	4.2 (0–8)
Percent lithophils (TRPR, SHRD)	Relative abundance expected to decline with higher sedimentation, reducing availability of gravel substrate for spawning	5.9 (0–28%)
Percent top carnivore (Apex predator: wall, NRPK, BURB)	High relative abundance of top carnivores indicates a relatively healthy, productive, and diverse community	21 (0–100%)
Number of benthic invertivorous species (LNSC, SHRD, LNDC, IWDR, TRPR)	Number species expected to decline when river habitats become excessively silty and eutrophic and dissolved oxygen is reduced	1.1 (0–4)
Percent of benthic invertivorous individuals (LNSC, SHRD, LNDC, IWDR, TRPR)	As above	16 (0–82%)
Percent older, long-lived fishes[c] (NRPK >600 mm, WALL >450 mm, WHSC >400 mm, GOLD >350 mm)	Older fish indicative of stable habitat, reduction in anthropogenic disturbance, and river connectivity	7.3 (0–60%)
Percent invertivorous cyprinids (LKCH, LNDC, SPSH)	Relative abundance expected to decline when invertebrates decrease in abundance and diversity due to habitat degradation	16 (0–80%)
Percent intolerant (LNSC, LKCH, BURB)	Includes individuals of species that have tolerance to disturbance values < 6; expected to decrease as river conditions are degraded	6.1 (0–72%)
Total individuals in sample (catch per 100 s)	Total abundance is comparable to the overall ability of the river to support a community; sites in poor condition are expected to support few individuals	1.5 (0–4)
NEGATIVE-SCORING	*VALUES DECREASE WITH INCREASING BIOLOGICAL INTEGRITY*	
Percent true omnivore (WHSC, FTMN)	As habitat declines in quality, the proportion of individuals that are omnivores is expected to increase	55 (0–100%)
Percent tolerant (MOON, GOLD, FTMN)	Individuals of species that have tolerance to disturbance values >8; expected to increase in abundance as river conditions are degraded.	4.8 (0–55%)
Percent DELT (deformities, disease, parasites, fin erosion, lesions or tumours)	These body conditions occur frequently where harmful chemicals are concentrated and can reflect stress caused by highly polluted waters.	25 (0–100%)

[a] Sources: Karr and Chu (1999); Bramblett et al. (2005); Stevens and Council (2008); Noble et al. (2007); Whittier et al. (2007).
[b] Abbreviations: LNSC = Longnose Sucker; SHRD = Shorthead Redhorse; WHSC = White Sucker; FTMN = Fathead Minnow; LKCH = Lake Chub; LNDC = Longnose Dace; SPSH = Spottail Shiner; NRPK = Northern Pike; BURB = Burbot; GOLD = Goldeye; MOON = Mooneye; IWDR = Iowa Darter; WALL = Walleye; TRPR = Trout-perch.
[c] Length classification based on unpublished, provincial data collected by M. Sullivan.

two units of $AICc_{min}$ (Burnham and Anderson 2002). However, if the top model had a low R^2 value (<0.2), we concluded that the metric may be "insensitive" to human stressors and that further research may be required to confirm metric-stressor relationships. Akaike weights (w_i) were also calculated to assess evidence supporting each model, and to estimate model-averaged coefficients. Model averaging is a robust method that reduces model selection bias (Burnham and Anderson 2002). A metric was recommended for IBI development if model averaging confirmed anticipated responses to measures of anthropogenic disturbance, and if the direction of relationships were relatively consistent. For example, a recommended metric could be one that was identified a priori as being a positive-scoring metric (i.e., as having values that increase with biological integrity) and was negatively correlated with TP concentrations, but was positively correlated with the riparian vegetation index. A metric was not recommended for evaluating river condition if the number of predicted relationships was equal to or lesser than the number of nonexpected relationships.

Results

Of the 19 species known to occur in the Battle River, only 14 species were captured on the 80 sections sampled for fish. Catch-per-unit-effort was 0.88 fish per minute of electrofishing. In total, 3,473 fish were captured, of which the most abundant species was white sucker (49% of catch; *Catostomus commersonii* Lacepede, 1803). The remainder of the catch was comprised of 15.8% longnose dace (*Rhinichthys cataractae* Valenciennes, 1842), 11.5% lake chub, 9.8% northern pike (*Esox lucius* Linnaeus, 1758), 6.6% shorthead redhorse (*Moxostoma macrolepidotum* Lesueur, 1817), 3.3% trout-perch (*Percopsis omiscomaycus* Walbaum, 1792), 2.4% walleye (*Sander vitreus* Mitchell, 1818), 0.9% fathead minnow, 0.3% burbot, 0.2% spottail shiner (*Notropis hudsonius* Clinton, 1824), 0.2% goldeye, 0.1% mooneye, 0.06% Iowa darter (*Etheostoma exile* Girard, 1859), and 0.03% longnose sucker. The latter four species were also relatively rare occurring on less than 5% of the study sites. The five species known to occur in the Battle River, but not captured during our study included the quillback sucker (*Carpoides cyprinus* Lesueur, 1817), emerald shiner (*Notropis atherinoides* Rafinesque, 1818), brook stickleback (*Culaea inconstans* Kirtland, 1840), yellow perch (*Perca flavescens* Mitchill, 1814), and lake whitefish (*Coregonus clupeaformis* Mitchill, 1818).

The fish catch data were used to create 12 candidate metrics, some of which were redundant as determined by correlation analysis (Pearson $r > 0.8$). For example, the "percent invertivorous cyprinids" metric was positively correlated with the "percent benthic invertivorous individuals" metric. Also, the "number of benthic invertivorous species" metric was positively correlated with the "native species richness" metric.

Insufficient variation was explained for 5 of the 12 metrics using the suite of parameters and models that were constructed. In other words, for five metrics, the R^2 value was less than 0.2 for the respective top model identified by AICc_{min}. These metrics were: percent older, long-lived individuals; catch per 100 s of electrofishing; percent DELTs; percent tolerants; and percent intolerants (Table 3). For the remaining seven metrics, approximately 39 to 57% of the variation in values was explained. Confidence intervals (95%) of model-averaged coefficients indicated that all seven metrics, with the exception of percent true omnivores, were influenced by basin area (Table 4). Five metrics behaved as expected and were clearly linked to human stressors, based on the 95% confidence intervals (Table 2 and 4). Metrics that were useful for evaluating river condition included: species richness, which was negatively linked to TP concentrations; percent top carnivores, which was negatively related with TKN; percent true omnivores, which was positively related to TP and road density in the basin (Fig. 2); percent benthic invertivorous individuals, which was negatively related to TP and cattle density in the basin; and percent lithophils, which was negatively linked to TP, cattle density in the basin, and road density in the basin (Table 4). Two metrics responded unexpectedly to human stressors. High numbers of benthic invertivorous species occurred in river sections with high concentrations of TKN and high levels of agricultural cover in the basins. Also, the relative abundance of invertivorous cyprinids was higher in river sections with elevated concentrations of TKN and nitrate-nitrite-N (Table 1 and 4).

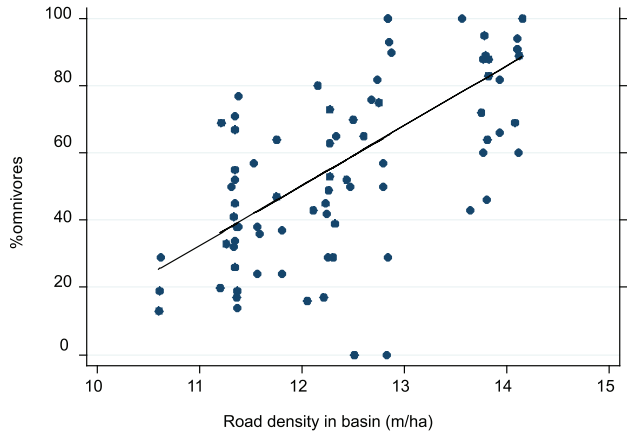

Fig. 2. A stressor-metric relationship, specifically a model-averaged regression line predicting %omnivore metric values with road densities in the basin (m/ha) overlaid with a scatter plot for the 80 sites sampled in the Battle River, Alta. (also see Table 4).

Plausible stressor models, as determined by AIC*c*, for the five metrics highlighted above were typically the water quality model (Model 1) and the road model (Model 4; also see Table 3). The water quality model was selected for predicting species richness, percent benthic invertivorous individuals, percent top carnivores, and percent lithophils; whereas the road model was selected for predicting species richness, percent true omnivores, and percent lithophils. The agricultural models (Models 2 and 3) were chosen for modelling changes in species richness (Table 3).

Discussion

This study demonstrated strong linkages between fish assemblage attributes and human stressors related to urban and agricultural activities that may impair river conditions. Relatively wide gradients of disturbance were identified along the Battle River (see variability of parameters in Table 1) where simple, northern fish assemblage attributes responded as strong indicators to the effects of human activities. Despite the inherent challenges of an assemblage dominated by white sucker and containing a total of only 14 species, five metrics were identified as being sensitive to nutrient concentrations, likely from runoff from agriculture related to cattle manure, and from urban-related influences statistically indexed as road density in basins. Importantly, the five fish assemblage metrics were statistically unrelated and conceptually different and may prove useful if integrated as a biological index for assessing river

TABLE 3. Model rankings (according to Akaike's information criterion corrected for small sample sizes; AICc), as well as fit of models (R^2), predicting metric scores for the Battle River [a,b]

Metric	*Model*	*n*	*df*	*LL*	*AICc*	*ΔAICc*	*Weight*	R^2
Species richness	Water quality	80	6	-143.89	300.9	1.8	0.13	0.50
	Agriculture I	80	4	-145.28	299.1	0.0	0.34	0.48
	Agriculture II	80	5	-144.31	299.4	0.3	0.28	0.50
	Road	80	3	-146.92	300.2	1.1	0.20	0.46
	Global model I	80	9	-141.67	303.9	4.8	0.03	0.53
	Global model II	80	10	-140.86	304.9	5.8	0.02	0.54
No. of benthic invert. spp.	Water quality	80	6	-92.58	198.3	6.2	0.04	0.61
	Agriculture I	80	4	-101.77	212.1	20.0	0.00	0.51
	Agriculture II	80	5	-96.14	203.1	11.0	0.00	0.57
	Road	80	3	-101.90	210.1	18.0	0.00	0.50
	Global model I	80	9	-90.17	200.9	8.8	0.01	0.63
	Global model II	80	10	-84.47	192.1	0.0	0.94	0.68
% benthic invertivores	Water quality	80	6	42.42	-71.7	0.0	0.68	0.57
	Agriculture I	80	4	36.58	-64.6	7.1	0.02	0.50
	Agriculture II	80	5	32.75	-54.7	17.0	0.00	0.45
	Road	80	3	36.74	-67.2	4.5	0.07	0.50
	Global model I	80	9	44.93	-69.3	2.4	0.20	0.59
	Global model II	80	10	44.02	-64.9	6.8	0.02	0.58
% invertivorous cyprinids	Water quality	80	6	29.75	-46.4	0.0	0.82	0.45
	Agriculture I	80	4	23.70	-38.9	7.5	0.02	0.36
	Agriculture II	80	5	22.72	-34.6	11.7	0.00	0.35
	Road	80	3	23.41	-40.5	5.9	0.04	0.36
	Global model I	80	9	31.24	-41.9	4.4	0.09	0.47
	Global model II	80	10	31.15	-39.1	7.2	0.02	0.47
% true omnivores	Water quality	80	6	10.12	-7.1	13.8	0.00	0.34
	Agriculture I	80	4	9.46	-10.4	10.5	0.00	0.33
	Agriculture II	80	5	7.62	-4.4	16.4	0.00	0.30
	Road	80	3	13.58	-20.8	0.0	0.47	0.39
	Global model I	80	9	20.55	-20.5	0.3	0.40	0.49
	Global model II	80	10	20.63	-18.1	2.8	0.12	0.49
% top carnivores	Water quality	80	6	26.15	-39.2	0.0	0.59	0.41
	Agriculture I	80	4	15.86	-23.2	16.0	0.00	0.24
	Agriculture II	80	5	16.98	-23.1	16.0	0.00	0.26
	Road	80	3	21.10	-35.9	3.3	0.11	0.33
	Global model I	80	9	28.48	-36.4	2.8	0.15	0.45
	Global model II	80	10	29.78	-36.4	2.8	0.15	0.46
% lithophils	Water quality	80	6	-251.27	515.7	0.8	0.33	0.59
	Agriculture I	80	4	-254.45	517.4	2.5	0.14	0.55
	Agriculture II	80	5	-256.42	523.6	8.7	0.01	0.53
	Road	80	3	-254.30	514.9	0.0	0.49	0.55
	Global model I	80	9	-250.14	520.9	5.9	0.03	0.60
	Global model II	80	10	-250.62	524.4	9.5	0.00	0.59
% older, long-lived	Water quality	80	6	58.76	-104.4	4.3	0.06	0.06
	Agriculture I	80	4	57.66	-106.8	1.8	0.22	0.04
	Agriculture II	80	5	58.59	-106.4	2.3	0.17	0.06
	Road	80	3	57.47	-108.6	0.0	0.54	0.03
	Global model I	80	9	59.28	-98.0	10.6	0.00	0.08
	Global model II	80	10	60.71	-98.2	10.4	0.00	0.11

TABLE 3 continued on next page

TABLE 3 continued

Metric	*Model*	*n*	*df*	*LL*	*AICc*	*ΔAICc*	*Weight*	*R2*
Catch/100 s	Water quality	80	6	-112.57	238.3	2.3	0.22	0.13
	Agriculture I	80	4	-117.15	242.8	6.9	0.02	0.02
	Agriculture II	80	5	-116.49	243.8	7.8	0.01	0.03
	Road	80	3	-114.82	236.0	0.0	0.71	0.07
	Global model I	80	9	-111.03	242.6	6.7	0.03	0.16
	Global model II	80	10	-111.11	245.4	9.4	0.01	0.16
% DELTs (deformities, disease, parasites, fin erosion, lesions, or tumours)	Water quality	80	6	29.65	-46.2	1.9	0.16	0.14
	Agriculture I	80	4	28.32	-48.1	0.0	0.41	0.11
	Agriculture II	80	5	28.87	-46.9	1.2	0.23	0.13
	Road	80	3	26.39	-46.5	1.6	0.18	0.07
	Global model I	80	9	31.03	-41.5	6.6	0.02	0.17
	Global model II	80	10	31.93	-40.7	7.4	0.01	0.19
% tolerants	Water quality	80	6	-272.06	557.3	3.9	0.08	0.15
	Agriculture I	80	4	-273.23	555.0	1.6	0.25	0.13
	Agriculture II	80	5	-272.82	556.5	3.1	0.12	0.14
	Road	80	3	-273.53	553.4	0.0	0.55	0.12
	Global model I	80	9	-271.73	564.0	10.7	0.00	0.16
	Global model II	80	10	-271.27	565.7	12.3	0.00	0.17
% intolerants	Water quality	80	6	-302.78	618.7	0.0	0.36	0.19
	Agriculture I	80	4	-305.82	620.2	1.5	0.17	0.13
	Agriculture II	80	5	-304.64	620.1	1.4	0.18	0.16
	Road	80	3	-306.55	619.4	0.7	0.25	0.11
	Global model I	80	9	-302.12	624.8	6.1	0.02	0.21
	Global model II	80	10	-300.95	625.1	6.4	0.01	0.23

[a] Highlighted rows are top models (i.e., models with $AICc_{min}$).
[b] df = degrees of freedom; LL = Log-likelihood.

conditions in central Alberta. The five fish assemblage metrics represented two trophic guilds (i.e., percent top carnivores and true omnivores), one habitat/trophic guild (i.e., percent benthic invertivorous individuals), one reproductive guild (i.e., percent lithophils), and one measure of community structure (i.e., species richness).

The abundances of omnivores and lithophils in the Battle River fish catches were highly sensitive to changes in road density in basins. Road density may affect fish assemblages through a variety of mechanisms, such as pollution, hydrologic alteration, stream channelization, fragmentation from improperly maintained culverts, 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 relationship of road networks with urban development is intuitive, and as road networks and urban development grows, changes in river habitat, water chemistry, and in the integrity of fish assemblages are anticipated. Similar to the presence of networks of roads, urban development continually affects streams and causes extensive and chronic impacts to natural hydrology and chemistry (Grapentine et al. 2004), often at greater magnitudes than other land-use types (reviewed in Wheeler et al. 2005). Previous research has shown that even low levels of urban land cover in a basin (8 to 10%) can result in highly altered fish communities (Wheeler et al. 2005). The highest level of urban cover in our study basins was much lower than this threshold (ca. 2%), approaching low intensity development. However, urban cover levels in the Battle River basin may be higher if considering both urban and suburban (e.g., acreage and intensive rural subdivision) developments. Importantly, our measurements of road network densities may suffice to encompass all residential development aspects, as well as industry, such as the petrochemical sector.

The current study provided evidence that current agricultural practices may also threaten the integrity of fish assemblages. A possible mechanism for the observed effects (i.e., reductions in benthic invertivores and lithophils) may be increased sediment and nutrient runoff, causing changes in the composition of basal algal resources and a reduction in available spawning substrates (Berkman and Rabeni 1987; Carpenter et al. 1998; Little et al. 2003). A major nonpoint source of nutrients in agricultural landscapes is often manure from livestock (Carpenter et al. 1998). Although the relationship between fish assemblage metrics and cattle densities may be partially confounded by influences of stream size, we identified that both TP and TKN were important variables related to fish assemblage metrics. This trend was noted despite sampling water and fishes at different times and the associated seasonal variability that may have been introduced by this approach. Previously, researchers have suggested that the effects of

TABLE 4. Regression summary of model-averaged coefficients (and 95% confidence intervals) for predicting fish assemblage metric scores for the Battle River[a]

	Coeff. and 95% CI			*Coeff. and 95% CI*			*Coeff. and 95% CI*		
	# of spp.			***# of benthic invert. spp.***			***%benthic invertivores***		
y-intercept	3.20	-1.28	7.68	4.21	-0.74	9.16	-4.19	-88.68	80.30
Basin area (ha)[b]	1.84	1.23	2.45	1.13	0.67	1.59	19.22	9.97	28.48
DO (mg/L)	0.05	-0.38	0.48	-0.07	-0.33	0.20	1.97	-2.16	6.09
TP (mg/L)	-8.59	-15.47	-1.71	-5.80	-10.07	-1.53	-85.09	-141.87	-28.31
TKN (mg/L)	1.75	-0.50	4.01	1.81	0.36	3.26	20.27	-2.92	43.46
Nitrate+nitrite-N (mg/L)	1.57	-1.24	4.39	1.72	-0.29	3.74	32.63	8.66	56.61
Cattle in basin (per ha)	-3.02	-8.74	2.71	-1.81	-3.75	0.13	-49.62	-86.18	-13.06
Agriculture in basin (%)	0.0018	-0.0037	0.0073	0.0363	0.0127	0.0599	0.0075	-0.0447	0.0598
Agriculture nearby (<10km, %)	-0.0337	-0.0776	0.0102	-0.0125	-0.0373	0.0124	0.0438	-0.4453	0.5328
Riparian condition at site (%)	0.0154	-0.0013	0.0321	0.0017	-0.0067	0.0101	-0.0490	-0.2048	0.1068
Road density in basin (m/ha)	-0.22	-0.77	0.32	-0.35	-0.65	-0.06	-6.57	-15.20	2.06
	%invertivorous cyprinids			***%true omnivores***			***%top carnivores***		
y-intercept	-20.16	-79.12	38.80	-163.14	-273.24	-53.04	115.23	16.13	214.34
Basin area (ha)[b]	17.85	9.75	25.95	3.06	-8.27	14.40	-14.77	-25.42	-4.12
DO (mg/L)	1.66	-2.42	5.73	1.08	-4.56	6.71	-1.65	-6.84	3.54
TP (mg/L)	-115.34	-195.10	-35.58	114.18	22.75	205.62	31.83	-48.46	112.11
TKN (mg/L)	25.76	1.54	49.97	-13.39	-47.49	20.72	-35.74	-64.83	-6.65
Nitrate+nitrite-N (mg/L)	36.71	11.85	61.56	-45.90	-84.65	-7.15	-27.91	-63.08	7.26
Cattle in basin (per ha)	-39.95	-77.81	-2.09	16.75	-86.50	120.01	47.19	-61.29	155.67
Agriculture in basin (%)	0.0288	-0.0849	0.1424	-0.0122	-0.1830	0.1585	-0.0713	-0.5763	0.4336
Agriculture nearby (<10km, %)	-0.0463	-0.5815	0.4888	0.0894	-0.4229	0.6017	-0.1636	-0.5971	0.2700
Riparian condition at site (%)	0.0340	-0.1367	0.2048	-0.1813	-0.4166	0.0540	0.0648	-0.1200	0.2495
Road density in basin (m/ha)	-5.03	-13.31	3.25	17.80	9.53	26.07	-6.49	-19.60	6.61
	%lithophils			***%older, long-lived***			***Catch per 100 s***		
y-intercept	7.61	-6.40	21.63	-12.42	-60.08	35.24	-3.28	-8.03	1.47
Basin area (ha)[b]	8.29	5.82	10.76	-0.39	-5.35	4.56	0.21	-0.40	0.81
DO (mg/L)	-0.67	-1.62	0.29	0.87	-2.48	4.22	0.16	-0.12	0.45
TP (mg/L)	-33.34	-55.93	-10.75	-15.71	-57.90	26.48	-3.39	-8.00	1.21
TKN (mg/L)	5.10	-2.74	12.93	14.55	2.05	27.05	1.57	0.20	2.93
Nitrate+nitrite-N (mg/L)	5.03	-1.57	11.63	-3.74	-20.56	13.08	0.97	-0.82	2.77
Cattle in basin (per ha)	-15.85	-28.92	-2.77	12.83	-22.33	47.98	1.30	-3.36	5.95
Agriculture in basin (%)	0.0060	-0.0159	0.0279	-0.0022	-0.0097	0.0052	0.0010	-0.0037	0.0058
Agriculture nearby (<10km, %)	-0.0569	-0.1949	0.0810	0.1490	-0.0022	0.3002	-0.0019	-0.0313	0.0274
Riparian condition at site (%)	0.0029	-0.0445	0.0502	-0.0534	-0.2244	0.1177	0.0043	-0.0078	0.0163
Road density in basin (m/ha)	-1.06	-1.69	-0.42	2.43	-1.88	6.74	0.41	0.07	0.75
	%DELTs			***%tolerants***			***%intolerants***		
y-intercept	29.41	-20.27	79.09	1.26	-21.94	24.46	5.08	-13.45	23.61
Basin area (ha)[b]	-4.22	-10.71	2.27	3.62	0.53	6.71	5.32	1.19	9.45
DO (mg/L)	-1.11	-5.98	3.76	0.28	-1.83	2.39	-0.84	-2.55	0.88
TP (mg/L)	74.50	-3.66	152.67	-14.75	-48.50	18.99	-66.68	-114.67	-18.69
TKN (mg/L)	-5.74	-30.11	18.62	3.34	-6.40	13.08	9.13	-3.31	21.56
Nitrate+nitrite-N (mg/L)	-4.18	-34.98	26.62	-3.40	-16.45	9.65	15.21	0.68	29.74
Cattle in basin (per ha)	9.24	-61.59	80.08	-6.36	-35.58	22.87	-3.24	-27.86	21.38
Agriculture in basin (%)	-0.0069	-0.0317	0.0180	0.0006	-0.0018	0.0031	0.0134	-0.0262	0.0530
Agriculture nearby (<10km, %)	0.0179	-0.3201	0.3559	0.0075	-0.1429	0.1579	-0.1504	-0.4470	0.1462
Riparian condition at site (%)	-0.2024	-0.3855	-0.0193	0.0257	-0.0534	0.1047	0.0688	-0.0232	0.1607
Road density in basin (m/ha)	2.31	-4.29	8.91	0.01	-2.43	2.44	-0.75	-2.46	0.97

[a] Intervals that do not contain zero are highlighted.
[b] Coefficients multiplied by 1,000,000.

agricultural practices on aquatic ecosystems in Alberta are a serious concern (Little et al. 2003; Wuite et al. 2007). For example, water contamination and nutrient loading have been observed in the Little Bow River, where repeated annual and seasonal applications of manure to the landscape may have led to high accumulation of nutrients in the soil, thereby creating a potential for pollution of surface water and groundwater (Little et al. 2003). To minimize contamination of water resources, the *Agricultural Operation Practices Act* (Alberta Agriculture, Food and Rural Development 2001) has laid out standards pertaining to the containment and application of manure. For example, specific setback distances are required (e.g., 30-m buffers from mechanical spreading of manure), and operators, such as those for confined feeding operations, must demonstrate that there is access to enough land to accommodate manure from their livestock. Given the results from our study and the fact that Alberta has the highest percentage of cattle in the country (Statistics Canada 2006), it is recommended

that current practices be evaluated through biological monitoring as part of an adaptive management strategy for ensuring the conservation of aquatic resources into the future.

This study builds on the bioassessment literature from the United States, and sets a precedent for biological monitoring in western Canada. Although the conclusion of the utility of using fish assemblages as indicators of human activities was similar to Bramblett et al. (2005) in Montana, the details of our study differed from that observed for Great Plains streams. First, the use of fish condition and tolerance guilds, as defined in our study, was not supported. Second, the use of the relative abundance of true omnivores and top carnivores as trophic guild metrics was supported in this study; as opposed to metrics measuring relative abundance of invertivorous cyprinids and number of benthic invertivorous species (see Bramblett et al. 2005). Although discrepancies among studies may be an artifact of different study designs and statistics, outcomes may also vary if the composition of fish assemblages changes from one region to the next. For example, only 14 species were captured in this Battle River study, compared with 37 species described in Bramblett et al. (2005). Further, white sucker was the dominant species in the Battle River fish catch and appeared to be very tolerant to human stressors. Thus, it is not surprising that the percent older, long-lived metric could not be linked to disturbance when white sucker was included in this metric. Further, Noble et al. (2007) contend that long-lived species may have great plasticity and may adapt their life histories to survive under different conditions. Discrepancies among studies may also occur if study regions differ in environmental conditions and individual species differ in their tolerance to disturbance. It is recommended that future research identify species tolerance ranges, as well as nonlinear responses to human disturbances in northern systems, for the advancement of multimetric monitoring indices in western Canada.

In summary, by linking metric scores to descriptions of human activities made in a GIS environment, managers can easily forecast conditions under various landscape scenarios of human development (for example, see Fig. 2). Importantly, the five metrics highlighted in this study could be integrated and used as a rapid assessment tool to characterize aquatic ecosystem health. Determining whether the stressor relationships identified in this study can be extrapolated to other systems outside of the Battle River watershed will require additional sampling in new subbasins and verification of relationships. However, the Battle River is a centrally-located river comprised of species that are distributed throughout most of the province and neighboring Saskatchewan, including the North Saskatchewan River drainage basin to the north and the Red Deer River drainage basin to the south (Nelson and Paetz 1992). Further, the land uses and stressors documented in the Battle River drainage basin are present in and as much a threat to river conditions in other areas of Alberta (Cooke and Prepas 1998; Timoney and Lee 2001; Little et al. 2003). Thus, the stressor relationships and proposed IBI metrics that were identified for the Battle River should be applicable beyond this system. Ecological information generated in this study can be used to protect aquatic resources threatened by increasing pressures from human developments and activities in western Canada.

Acknowledgments

The authors thank the following individuals, agencies, and corporations for their contributions and assistance in delivering this project. The Alberta Conservation Association (ACA), Alberta Fish and Wildlife Division (AFWD), and Alberta Environment (AE) provided staff expertise. Alberta Sustainable Resource Development (ASRD) provided the electrofishing boat. Special thanks to Jason Blackburn (ACA), Brad Hurkett (ACA), Diana Rung (ACA), Chris Delage (ACA), Jason Cooper (AFWD), and Vance Buchwald (AFWD) who collected field data. Chris Teichreb (AE) secured funding for the project through the provincial "Water for Life Strategy," acted as liaison with the Battle River Watershed Alliance, and provided input on various water quality components. Thanks to Lorne Fitch (Cows and Fish organization) for his input on riparian shoreline variables. Thanks also to Alastair Franke who coordinated and chaired several meetings that led to the completion of the Battle River IBI field program. The Department of National Defense (CFB Wainwright) and ATCO (Forestburg) provided financial contributions to the project.

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Received: 29 June 2009; accepted: 4 February 2010.