Fisheries and Oceans Canada | Pêches et Océans Canada

Rainy River Index of Biotic Integrity – Final Report

December 2014

By

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INTRODUCTION

Freshwater ecosystems are valuable resources, yet they are one of the most highly altered ecosystems on our planet (reviewed in Carpenter et al. 2011). Species extinction rate estimates for recent and future time periods in North America are five times higher for freshwater organisms than terrestrial organisms (Ricciardi and Rasmussen 1999). Demand on freshwater ecosystems continues to increase (Fitzhugh and Richter 2004; Revenga et al. 2005). However, it is recognized that development must proceed in such a way as to balance ecological requirements for long-term sustainability with human demand (Fitzhugh and Richter 2004; Dudgeon et al. 2006). Sampling methods that enable the estimation of biological condition of rivers are needed to quantify changes in river health due to anthropogenic activities.

Initial efforts to assess riverine health focused on chemical and physical properties and then later on benthic invertebrates (reviewed in Karr 1981). While these tools are capable of addressing some aspects of river health, they are indirectly related to fish health, which is often the primary biological concern. The index of biotic integrity (IBI), which was developed as a tool to assess the health of streams (Karr 1981, Karr et al. 1986), is a multi-metric index that evaluates community composition (i.e., feeding guilds, spawning needs, sensitivity to disturbance), species richness, fish abundance and fish health as surrogates of riverine health. Metrics are scored and the sum of the values, or IBI score, is compared to that obtained from regional expectations (Roset et al. 2007). Since its development, the IBI has been modified numerous times to allow for the assessment of riverine health in both large and small rivers around the world (e.g. Aparicio et al. 2011; Raburu and Masese 2012). While the IBI has primarily been used to assess fish populations in lotic ecosystems, it has been adapted for us in a variety of aquatic environments (e.g., lakes, reservoirs, and marine ecosystems) and with other organisms (e.g. macroinvertebrates, diatoms), and has even been applied to terrestrial ecosystems (reviewed in Ruaro and Gubiani 2013). Common concerns of the IBI include a lack of standardized sampling methods, well-defined criteria for selecting reference sites, and methods for determining which, as well as how many, metrics to use (reviewed in Roset et al. 2007). Nevertheless, the IBI remains a useful tool for assessing relative ecological health of aquatic systems and one used by national and state/provincial monitoring programmes.

On January 5, 2000, the International Joint Commission (IJC) issued an order prescribing the method of regulating the levels of the boundary waters of Rainy and Namakan lakes, consolidating and replacing a number of previous orders and supplementary orders (2000 International Joint Commission Order). The new rule curves prescribed in this order (the “2000 Rule Curves”) were in part an attempt to balance concerns related to the environment, hydropower, flooding, and navigation. Under the assumption that a natural hydrograph would benefit the aquatic communities of Rainy Lake and Namakan Reservoir, the 2000 Rule Curves shifted towards a more natural hydrograph. Improvements to water quality in the Rainy River since the previous orders allowed for lower discharges than were previously desirable. These changes resulted in seasonally higher flows to the Rainy River that were expected to have the potential to benefit the aquatic community of the Rainy River. The “Consolidated Order” was effective on February 28, 2001 and contained the provision that monitoring be conducted to allow for a thorough review of the effects of the changing flow prescription by 2015. In 2007, the IJC formed a Rule Curve Assessment Workgroup to develop a plan of study (POS) in which the Workgroup would prioritize the monitoring and analyses required to review the IJC Order in 2015. Specifically, the POS was written to identify priority studies and describe information/data that remained to be collected, identify what entities might collect the data and perform the studies, and to provide an estimate for the cost to accomplish this work by 2015. The Plan of Study (POS) for the Evaluation of the International Joint Commission (IJC) 2000 Order for Rainy and Namakan Lakes and Rainy River was completed in 2009 (Kallemeyn et al. 2009). Several studies were undertaken to assess the impacts of the new rule curve on Rainy and Namakan lakes, however, the focus of the present study was on impacts to the upper Rainy River.

Following the protocol described in Lyons et al. (2001), the Minnesota Department of Natural Resources conducted an IBI Study on the Rainy River in 2002. Using the scoring criteria of Lyons et al. (2001), the results of the 2002 IBI indicated that the health of the Rainy River fish community was good. The 2002 IBI was produced two years following the 2000 Rule Curve changes and thus is not necessarily representative of the pre-2000 rule curve fish community. Roughly one-third of the smallmouth bass, walleye, and northern pike were from year classes 2000-2002 (Eibler and Anderson 2004). Time lags associated with the effects of watershed disturbances are known to occur (Findlay and Bourdages 2000), especially for K-selected species (Rutherford et al. 1992), and thus, it is unlikely that the effects of the 2000 Rule Curve were fully manifested at the time of the 2002 IBI.

The Scope of Work (SOW) for this study was to assess whether Rainy River fish community health has been affected by the 2000 Rule Curves for Rainy and Namakan lakes and to assess the current condition of the fish community. The fish community assessment is identified as a priority in the POS. Fish community data are assessed using a modified version of the index of biotic integrity (IBI) developed by Lyons et al. (2001) and subsequently used by Eibler and Anderson (2004) on the upper Rainy River in 2002. The current health of the upper Rainy River fish community, as it compares to that of 2002, is discussed.

METHODS

Field sampling was conducted between August 08 and August 13, 2013 by Fisheries and Oceans Canada partially as an in-kind addition to the ongoing project assessing the critical spawning and nursery habitat of walleye, sturgeon, and log perch in the upper Rainy River, and with support from the Ontario Ministry of Natural Resources, the Minnesota Department of Natural Resources, and Voyageurs National Park. For the purposes of this study, the upper Rainy River is defined as the section of river from the International Falls dam to Long Sault Rapids, roughly 54 km east of Fort Frances (Figure 1). By focusing on the upper river, the potential influence from backwater or species input from the Lake of the Woods was avoided. Furthermore, this is the section of the river most influenced by the operations of the dam. In addition to the same twenty, 1.6km stations sampled on the upper Rainy River in 2002, four extra 1.6km stations (three Canadian and one American), which were originally intended to be fished in 2002, were also fished in early-August. However, only those stations that were electrofished in both 2002 and 2013 were included in analyses. Furthermore, sites US07 and US10 were excluded from analyses, as they are located in areas that contain side channels between islands and the mainland that were sampled in 2013 but would not have been accessible to sampling in 2002 due to lower flow conditions.

The fish community was sampled during daylight hours by boat electrofishing only, following the methods outlined in Eibler and Anderson (2004). Transects were sampled in 2013 using two Smith-Root SR-14H electrofishing boats, each crewed with three people. Both vessels were equipped with a 5.0 hp generator powered pulsator producing a pulsed DC current and were powered by similar propeller driven outboards. Electrical settings were 3.0 Amps, 60 Hertz, 100-1000 Volts. Vessels moved in a downstream direction as close to the shoreline as possible. Netters used standard Smith-Root 3.175mm mesh nets with 2.44m poles. All observed fish were collected and placed into the live well. When a large northern pike or lake sturgeon was captured, the boat was stopped, the position was marked with a GPS, and the fish was processed at shore. The electrofishing survey resumed immediately after the fish was processed. Immediately following the completion of each transect, all captured fish were identified to species and counted. Game fish (large and small mouth bass, northern pike, muskellunge, walleye, sauger, and lake sturgeon) were individually processed (total and fork length, weight, and a scale sample or fin ray collected), while all other species were counted and bulk-weighed. All fish were externally examined to document the presence of deformities, eroded fins, lesions, or tumours (DELT).

The electrofishing catch data were analyzed using a modified version of the IBI developed by Lyons et al. (2001) for large (non-wadeable), warm water streams of the mid-west. This IBI scores 10 metrics based on regional expectations for minimally degraded sites (see Table 1 for a description of the metrics and the associated scoring criteria). The metrics assess community composition (percent of total catch comprised of riverine specialists, insectivores, and lithophils), species richness (number of native, sucker, riverine, and intolerant species), fish abundance (weight per unit effort (WPUE)), and fish health (occurrence of DELT). Using the IBI scores for individual sites, a paired t-test was performed via Statistica 10 (StatSoft Inc., 2010) to test for overall change in fish-community health between 2002 and 2013. Paired t-tests were also used to test for differences in biomass per unit effort (BPUE) and catch per unit effort (CPUE) between 2002 and 2013.

RESULTS

A total of 51,648 seconds were spent electrofishing 18, 1.6km transects (average of 2869 seconds/transect), resulting in 20,191 captured fish in 2013. In 2002, 43,509 seconds were spent shocking the same 18, 1.6km transects (average of 2417 seconds/transect), resulting in 5,043 captured fish. Despite capturing four times the number of fish in 2013 as were captured in 2002, total biomass of 2013 samples was only 40% greater than total biomass of 2002 samples, an indication that many fish captured in 2013 were either small bodied species or juveniles. Discharge during sampling in 2013 ranged from 370 to 374 $m^3 \cdot s^{-1}$, while in 2002, sampling occurred when discharge was 195-196 $m^3 \cdot s^{-1}$ (Figure 2).

IBI scores ranged from 60 to 85 in 2002 and 55 to 80 in 2013 (Tables 2a-2d, Figure 3). Mean IBI scores were 69.2 and 70.6, in 2002 and 2013, respectively, and were not significantly different (paired $t(17) = 0.68$, $p = 0.51$). IBI scores for some sites were consistent between 2002 and 2013, while other sites increased or decreased slightly (Tables 2a-2d, Figure 3). The largest changes in IBI scores between 2002 and 2013 were at US02, US07, CA05 (decrease of 20) and US10 (decrease of 35). However, as detailed in the Methods, US07 and US10 were dropped from all analyses due to concerns that the location sampled in 2002 was different than that sampled in 2013.

There were several changes in fish-community composition metrics between the 2002 and 2013 IBIs, most of which were driven by an increase in the proportion of yellow perch. Species richness was comparable between the two IBIs, with 39 species captured in 2002 and 44 in 2013; however, 16 species were only found in one IBI dataset (Table 3). An additional four species were captured in 2013 at sites that were not sampled in 2002 (northern brook lamprey, bluntnose minnow, lake chub, and longnose sucker). Major decreases were observed for the metrics percent riverine species (41% decrease) and percent lithophilic spawners (44% decrease). These decreases were offset by increases in the metrics percent insectivores (14 % increase), percent round-bodied suckers (11% increase), and number of intolerant species per site (from 2.5 to 4.9). Yellow perch showed the most dramatic change in abundance, increasing from 1,723 in 2002 to 15,795 in 2013 (Table 3), but most of these individuals were young-of-the-year (YOY) and therefore contributed very little to the biomass of the catch. For most other species, similar numbers were captured in 2002 and 2013, although some relatively large increases (e.g., black crappie, northern pike) and decreases (e.g., emerald shiner and trout perch) were observed (Table 3). In both 2002 and 2013, WPUE was nearly twice as high at Canadian sites than American sites (Figure 4) and was significantly higher overall in 2013 than in 2002 (paired $t(17) = 2.3$, $p = 0.04$). In addition, catch per unit effort (CPUE) was significantly higher in 2013 (paired $t(17) = 4.3$, $p < 0.001$) (Figure 5).

Lyons et al. (2001) considered the occurrence of DELTs above 3% as concerning. Only two Canadian sites in 2002 (CA01 and CA06) and two American sites in 2013 (US09 and US10) had DELT occurrences above 3% (Tables 2b and 2c). Little change occurred in the prevalence of DELTs, which decreased from 1.7% in 2002 to 0.5% in 2013.

DISCUSSION

Based on the results of the IBI in 2002 and 2013, it does not appear that the health of the Rainy River fish community is being greatly impacted, either positively or negatively, by the 2000 Rule Curves. As in 2002, the overall IBI score in 2013 was in the middle of the range of scores considered by Lyons et al. (2002) to be good. For large rivers in Wisconsin, Lyons et al. (2001) considered scores of 80-100 as excellent, 60-79 as good, 40-59 as fair, 20-39 as poor, and less than 20 as very poor (Lyons et al. 2001). Furthermore, variation in IBI score at sites between years was generally within the range observed by Lyons et al. (2001) (5-15 points) for minimally disturbed sites. However, some fish species saw major changes in abundance, which translated to differences in some individual metric scores between 2002 and 2013. Potential mechanisms responsible for these differences are examined below. In addition, we discuss factors that have the potential to mask and or provide misleading results regarding changes in river ecosystem health.

While the overall IBI score in 2013 was very similar to that in 2002, there were some changes in the upper Rainy River fish community that are worth noting, including a large increase in yellow perch abundance and a decrease in emerald shiner. A greater than nine-fold increase in yellow perch caught in the 2013 IBI sampling compared to the 2002 IBI effort resulted in changes to some metrics, including much higher CPUE in 2013. Being insectivorous, but not riverine or lithophilic, the large increase in yellow perch had the effect of increasing the relative abundance of insectivores and decreasing the relative abundance of both riverine and lithophile species, which translated to changes in the scores of the latter two metrics between 2002 and 2013. The additional weight resulting from the increase in mainly YOY yellow perch was not sufficient to change the metric score for percent insectivores (by biomass) at any sites. Being a riverine and lithophilic spawner, the four-fold decrease in emerald shiner abundance further contributed to the decrease in the relative abundance, and thus metric scores, for percent riverine species and percent lithophil spawners in 2013.

The IBI was re-calculated without yellow perch to assess the impact of their exceptionally high abundance of in 2013. However, the influence of perch on individual metrics, and thus overall IBI score, was not as strong as suspected. There was no change in percent insectivores as the calculation for this metric is done using weight and most yellow perch captured were YOY. Percent riverine and lithophil species both increased slightly, with the greatest increase occurring in 2013 for both metrics. While the magnitude of the decrease in emerald shiner was much less than that of the yellow perch increase, the complimentary effects of these changes in abundance appears to be the main driver of the changes in percent riverine and percent lithophil species. The change in relative abundance of these two species greatly affected scores for percent riverine species and percent lithophil spawners at some sites, including CA05, which received scores of 10 for both metrics in 2002 but scores of zero for these metrics in 2013.

The increase in yellow perch abundance, while large, is within the natural range in abundance that has been observed for this species from year to year; yellow perch year-class strength has been reported to fluctuate 8-40 fold (Koonce et al. 1977). Indeed, most of the yellow perch captured in 2013 were YOY. Variation in year-class strength in yellow perch has been attributed to temperature, wind, flow, prey availability, predation, cannibalism, and size of spawning stock (reviewed in Koonce et al. 1977). However, water temperature, which can influence mortality rates directly or indirectly through the above-mentioned variables, is most often cited as being the main driver of year-class strength. More specifically, cold weather appears to be associated with weak year classes while warm weather appears to be associated with strong year classes. While we don't have water temperature data to compare 2002 and 2013, spring freshet occurred two weeks later in 2002 on the lower Rainy River, which suggests that water temperature may have been quicker to warm during the spring of 2013, although monthly average atmospheric temperature was very similar in 2002 and 2013. In addition, the earlier freshet in 2013 may have better coincided with yellow perch spawning activity than in 2002, which could have resulted in an increase in the abundance and quality of preferred spawning habitat. However, we cannot say for certain when yellow perch spawned in 2002 or 2013, nor do we have data regarding the abundance of preferred yellow perch spawning habitat at various river stages.

The large variation in abundance observed in a small number of species between 2002 and 2013, similar to the increase in yellow perch abundance, is probably best explained as the result of natural interannual variability. Emerald shiner abundance is known to fluctuate greatly in other systems (Scott and Crossman 1973) and thus the decrease in abundance observed for this species is not necessarily indicative of the 2000 Rule Curve having a negative effect on emerald shiner. The increased abundance of black crappie in 2013, like yellow perch, was driven by an increase in the abundance of YOY. In particular, CA03 was a major contributor to the increase in black crappy with approximately one third of the black crappie captured in the 2013 IBI coming from this site.

Canadian sites had, on average, higher WPUE than American sites in both 2002 and 2013, which might be related to the orientation of the river. The only major tributaries that enter the Rainy River within the study reach enter on the American side of the river (the Little Fork and Big Fork rivers); however, the sites immediately downstream from these tributaries do not appear to be affected, either positively or negatively, by these tributaries. Land use, based on satellite imagery, appears to be similar for the study reach on both sides of the river, although there may be marginally more agricultural land on the Canadian side of the river, and thus, is it possible that the Canadian side of the river receives slightly higher nutrient inputs. However, as the study reach on the Rainy River runs almost entirely East-West, a more convincing explanation for the consistent differences in WPUE is the difference in the amount of direct sunlight that reaches the north and south margins of this section of the Rainy River. The banks of the Rainy River and the trees along its shores are likely high enough to cast a large shadow along the southern shore for much of the day, particularly in the spring, winter, and fall, while the north shore is exposed to sunlight for most of the day. As primary production is positively related to light exposure in moving waters (Kiffney et al. 2003, 2004), it may be that the Canadian (i.e., north) side of the study reach of the Rainy River experiences higher rates of primary production, which ultimately supports a fish community of greater biomass.

Flow was higher during sampling in 2013 compared to 2002, although it seems unlikely that the difference in flow had a substantial impact on sampling. The discharge/stage relationship is complex. However, the difference in discharge during sampling between the two IBI efforts translates roughly to a one meter increase in water stage elevation (WSE) on the upper Rainy River in 2013 (Jeff Muirhead, pers. comm.). A change in river stage of this scale on the Rainy River would not greatly alter wetted width and thus is unlikely to have substantially altered the habitat sampled by the electrofishing boats, nor the distribution of fishes in the river. Some sites, however, may have been disproportionately influenced by changes in WSE. Unpublished work by Muirhead et al. on the Rainy River suggest that the hydraulics in areas associated with river-bed features such as shoals (e.g., back eddies) are much more sensitive to changes in WSE than more uniform sections of river. As such, fish communities at sites associated with irregular river-bed features may be more sensitive to changes in WSE.

While most sites experienced changes in overall IBI score ranging from 0 to 15, US02, which includes a back-eddy associated with bedrock shoal, increased by 20 points between 2002 and 2013. The increase in IBI score at US02 was mainly driven by an increase in the abundance of suckers (mainly shorthead redhorse), which resulted in increased scores for the metrics WPUE, percent round sucker, and percent insectivores. The geometrics of this site are such that its hydraulics are likely to be more variable across different WSE than sites located in more uniform sections of the river, and thus, the ability of US02 to hold fish might vary greatly even with minor flow changes. Alternatively, as US02 contained relatively few fish in both sampling periods, the difference in IBI score between 2002 and 2013 may be an artifact of sampling error. Using bootstrapping to look at variation in IBI score due to random sampling error, Dolph et al. (2010) found that sampling error resulted in differences in IBI score as high as 40 points at a site (in a scoring system of 0-100). However, sampling error alone rarely (less than 1%) resulted in impairment status change from unimpaired to impaired, or vice versa.

Paller (2002) compared single and multiple pass electrofishing techniques and found that little variability in IBI score was attributed to inefficient sampling. Degraded sites are reported to have higher variability in IBI scores relative to high-quality sites (Fore et al. 1994; Yoder and Rankin 1995; Lyons et al. 2001), while fish assemblages exhibit less variability in IBI scores at undisturbed sites (Steedman, 1988; DeShon 1994; Niemela and Feist 2000; Paller 2002). High quality sites tend to have similar amounts of temporal variability and variability due to random sampling error, whereas poor-to-moderate quality sites have higher levels of temporal variability (Dolph et al. 2010). Lyons et al. (2001) observed that multiple impact sites (impacted by two or more of the following: hydropeaking, point source pollution, non-point source pollution, commercial navigation, or impoundment) had IBI scores that ranged by 10-45 points, whereas least impacted sites varied from 5 to 15 points between years. With the exception of US02 and US05 (20 points higher and lower, respectively, in 2013), variability in IBI scores at sites between years on the Rainy River was within the range observed by Lyons et al. (2001) (5-15 points) for least impacted sites (impacted by no more than one of the following: hydropeaking, point source pollution, non-point source pollution, commercial navigation, or impoundment). Furthermore, average absolute value percent difference between samples on the Rainy River (6.9%) was nearly identical to that observed in the Paller (2002) study (7%).

Lyons et al. (2001) acknowledged the limitations of the IBI to assess ecosystem health accurately, particularly in degraded ecosystems, and suggested that even with multiple samples spanning several years, the IBI will only be capable of detecting major changes in ecosystem health. Niemela and Feist (2000) found IBI scores to be highly variable at sites in large Minnesota rivers and suggested that an inability to catch a representative sample of fish and/or the diversity of disturbances that act on large river systems in comparison to smaller rivers or streams likely contributed to the variability. Indeed, failing to detect rare species, which are more susceptible to random sampling error, has been shown to influence IBI score (Dolph et al. 2010). Metrics based on taxa richness, rather than those based on relative abundance, are more sensitive to the presence/absence of rare taxa (Wan et al. 2010). One solution to this problem is to exclude rare taxa (e.g., those with an expected probability of capture ≤0.05 (Hawkins et al. 2000). In fact, some studies suggest that confining comparisons between sites of concern and reference sites to common species still allows for the detection of human-caused disturbances (Van Sickle et al. 2007; Aroviita et al. 2008). However, taxa richness based metrics showed relatively little variation between 2002 and 2013, and thus as a result, no measures were taken to reduce variability that may have resulted from random sampling error.

The 2002 IBI results may not accurately reflect the fish community prior to the implementation of the 2000 Rule Curve, but it was the only previous IBI conducted on the upper Rainy River assessing the health of the fish community with which to compare. A portion of the fish community surveyed in 2002, including approximately one third of the smallmouth bass, walleye, and northern pike, were from 2000-2002 year classes. However, fish communities are slow to respond to disturbance (Attrill and Depledge 1997), and response time-lags are known to occur (Findlay and Bourdages 2000). Large rivers, in particular, appear to experience long time-lags following restoration activities (Pegg and McClelland 2004). Response to disturbance is slowest for longer lived species, which may take several years to decades to exhibit the full response (Strayer et al. 2014). The life span of fish species in the Rainy River system varies from several years (e.g., cyprinids) to decades (e.g., lake sturgeon), with the majority of the large bodied fish species having expected life spans of 5-10 years. As such, a response time-lag should be expected for most, if not all species occupying the Rainy River. Kanehl et al. (1997) studied the effects of dam removal on a Wisconsin river and found that, while there was a rapid change in habitat, including increased cover for fish, there was a three-year lag between dam removal and smallmouth bass recovery. Thus, it is unlikely that the effects of the 2000 Rule Curve were fully manifested at the time of the 2002 IBI, and likely that the comparison of 2002 data to 2013 data is a valid estimate of change in riverine health resulting from the 2000 Rule Curve.

CONCLUSIONS

When compared to the 2002 IBI data, the 2013 IBI suggests that it is unlikely that the 2000 Rules Curves have had a substantial effect, either positive or negative, on the upper Rainy River fish community. Mean IBI scores were 67.7 and 70.3 in 2002 and 2013, respectively, and were not significantly different. IBI scores ranged from 60 to 85 in 2002 and 55 to 80 in 2013 (60-80 is considered good for large rivers of this region). However, there were large changes in some of the metrics, including percent riverine species (41% decrease), percent lithophil species (44% decrease), percent insectivores (18% increase), and percent round suckers (12% increase). Most of these changes were driven by an increase in the abundance of YOY yellow perch and, to a lesser extent, a decrease in emerald shiner abundance. The changes in the abundance of YOY yellow perch and emerald shiner are thought to be an artifact of natural interannual reproductive variability and not the result of changes caused by the 2000 Rule Curve. However, without multiple sampling efforts between 2002 and 2013, it is difficult to differentiate between natural interannual variability and change resulting from the implementation of the 2002 Rule Curve.

ACKNOWLEDGMENTS

We thank Lisa O'Connor for her significant contribution to the field work. Others who made the field portion of this work possible include Rick Elsner, Kim Mitchell, Tom Mosindy, Ladd Corrin, Tom Burri, Tyler Fellows, Morgan Mathison, James Anderson, and Ryan Maki. We also thank Kevin Peterson, Phil Talmage, and Jeff Eibler for their assistance in coordinating this project. Funding for this work was provided by the Canadian Section of the International Joint Commission.

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Figure 1. The Rainy River between Rainy Lake and Lake of the Woods. Sites for the Index of Biotic Integrity study spanned the section highlighted in red.

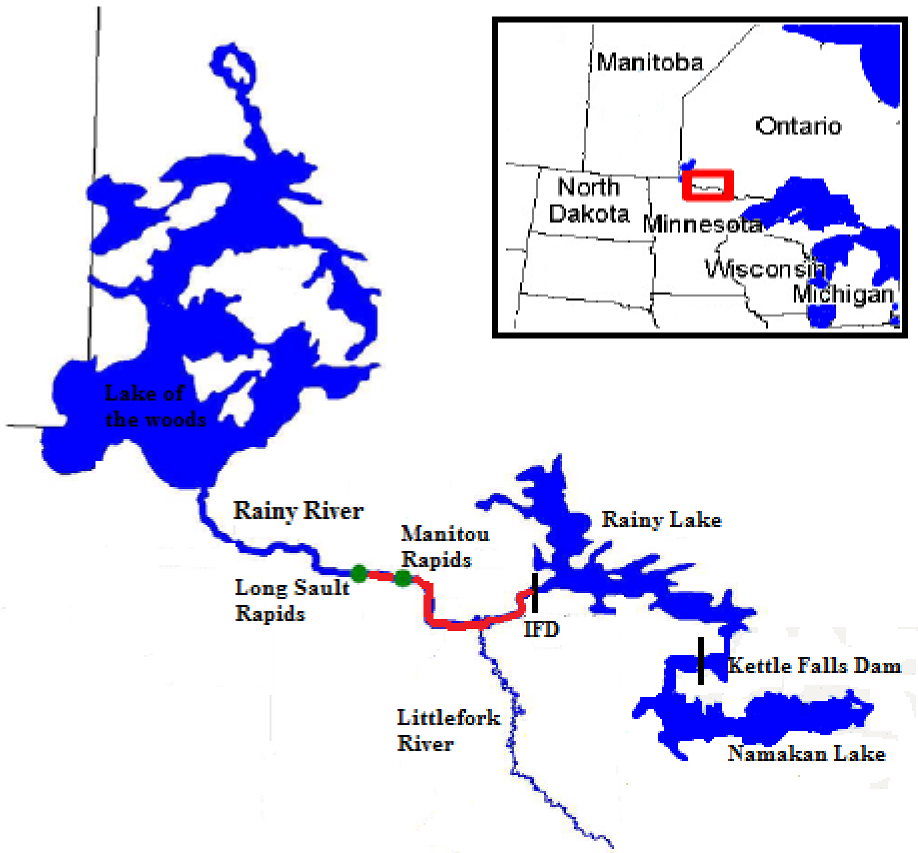

Figure 2. Rainy River hydrograph downstream from Rainy Lake in 2002 and 2013. Arrows identify when sampling occurred for each index of biotic integrity sampling.

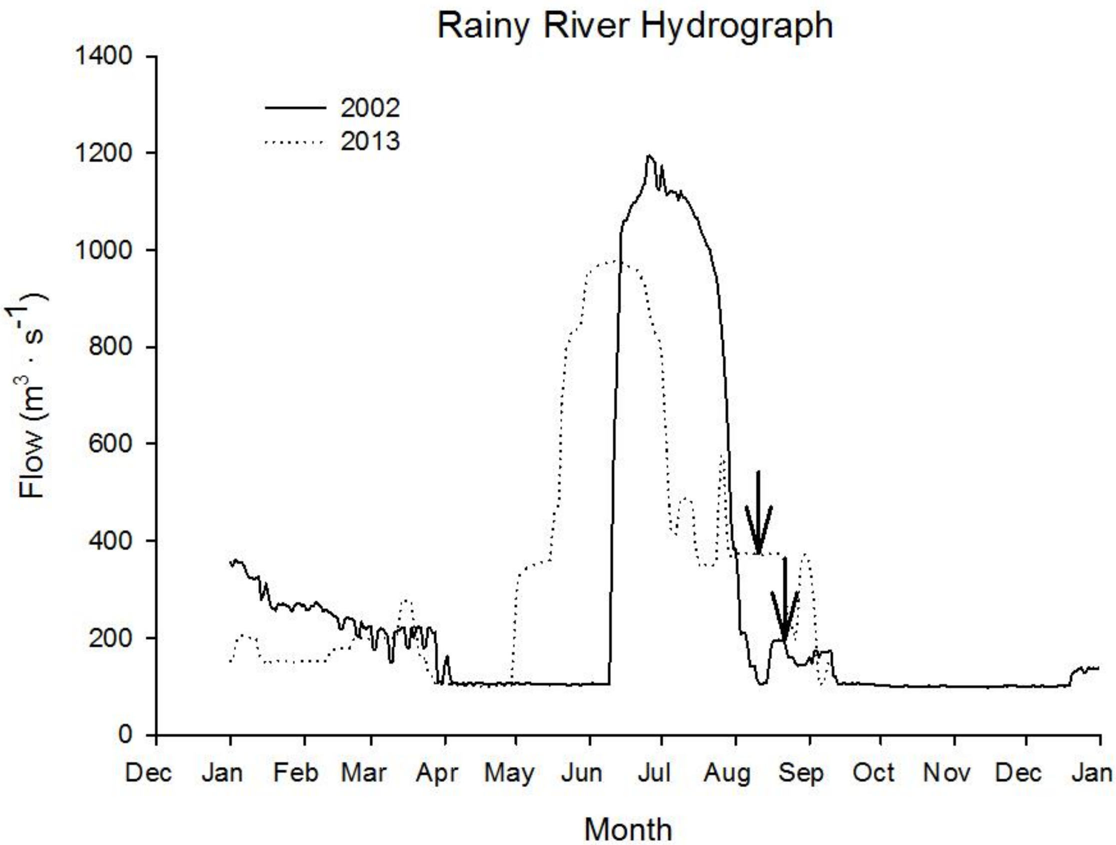

Figure 3. Index of biotic integrity scores for Canadian and American sites on the Rainy River in 2002 and 2013. Note that the figure below includes sites US07 and US10, as well as sites that were only sampled in 2013, all of which were excluded from analyses.

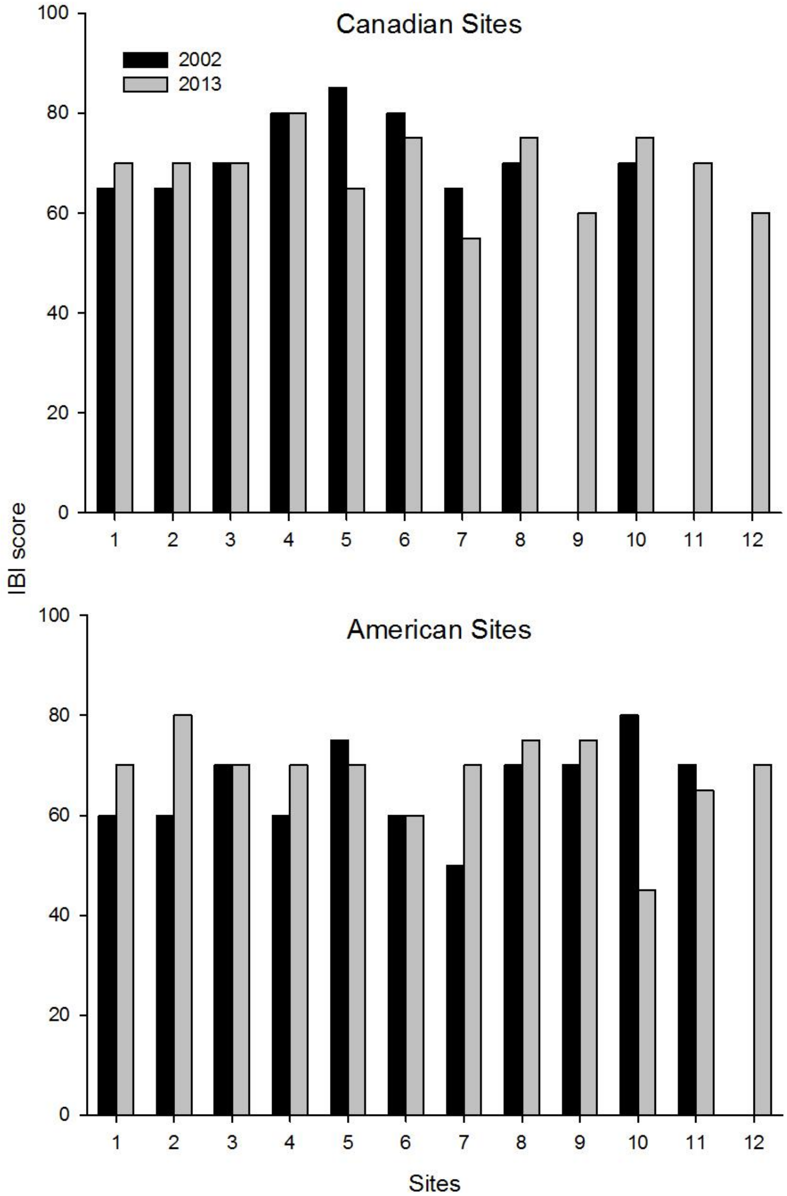

Figure 4. Weight per unit effort for all Canadian and American Sites in 2002 and 2013.

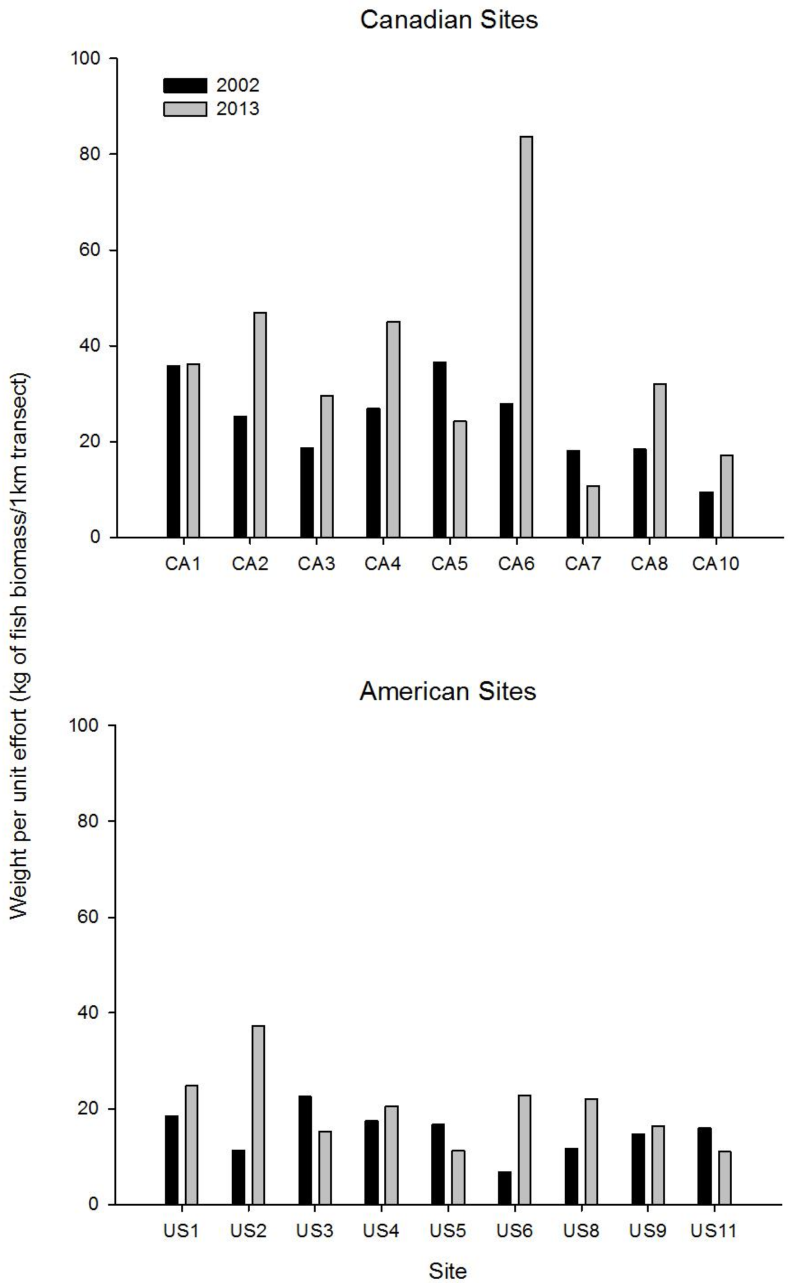

Figure 5. Catch per unit effort for all Canadian and American sites in both 2002 and 2013.

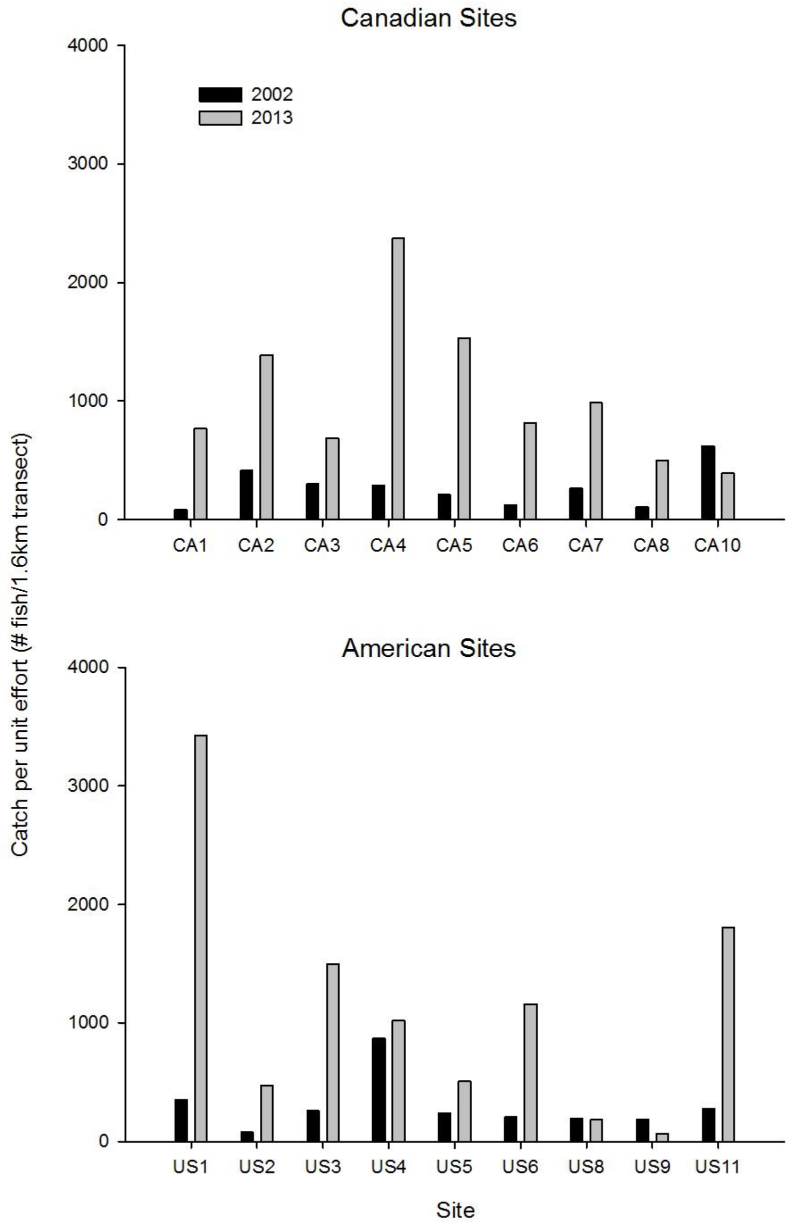

Table 1. Metrics used for the Index of Biotic Integrity (IBI), the scoring criteria, and the metric definitions.

Metric	Poor (0)	Fair (5)	Good (10)	Description
WPUE	0–9.9 kg	10–25 kg	>25 kg	Weight (biomass) to the nearest 0.1 kg of fish collected per 1,600 m of shoreline, excluding tolerant species
Native species	0–7	8–9	>9	Total species minus exotic species
Sucker species	0–2	3–4	>4	Number of species in the sucker family (Catostomidae)
Intolerant species	0–1	2	>2	Number of species considered intolerant of environmental degradation
Riverine species	0–1	2–3	>3	Number of species that are obligate stream or river dwellers not normally found in lentic habitats
% DELT (n)	>3%	3–0.5%	<0.5%	Percentage of total fish captured that were obviously diseased or that had eroded fins, lesions, or tumors
% Riverine (n)	0–10%	11–35%	>35%	Percentage of total fish captured that were obligate stream or river dwellers not normally found in lentic habitats
% Lithophils (n)	0–44%	45–69%	>69%	Percentage of total fish captured that were simple lithophilic spawners (i.e., that spawned on clean rocky surfaces without preparing a nest or guarding their eggs)
% Insectivore (wt)	0–10%	11–60%	>60%	Percentage of total biomass accounted for by insectivores
% Round suckers (wt)	0–10%	11–60%	>60%	Percentage of total biomass accounted for by the genera Cycleptus, Hypentelium, Minytrema, and Moxostoma

Tables 2a-2d. Metric values for each Canadian and American site sampled in 2002 and 2013. Metric scores are presented below each metric value in italics.

a)

2002						Sites					
Site	US01	US02	US03	US04	US05	US06	US07	US08	US09	US10	US11
WPUE	18.4	11.2	22.5	17.4	16.7	6.7	4.7	11.7	14.7	10.2	15.9
	5	*5*	*5*	*5*	*5*	*0*	*0*	*5*	*5*	*5*	*5*
Native species	15	16	16	19	23	14	16	18	21	18	15
	10	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*
% DELT (n)	0.6	2.7	1.9	0.3	0.4	1	0	1.1	1.1	1.8	0.4
	5	*5*	*5*	*10*	*10*	*5*	*10*	*5*	*5*	*5*	*10*
% Round suckers (wt)	66.1	27	75.7	18.8	54.4	73.1	6.8	49.9	48.5	53	41.8
	10	*5*	*10*	*5*	*5*	*10*	*0*	*5*	*5*	*5*	*5*
Sucker species	3	3	3	3	3	4	2	3	3	3	2
	5	*5*	*5*	*5*	*5*	*5*	*0*	*5*	*5*	*5*	*0*
Intolerant species	2	5	3	4	4	2	4	4	6	4	2
	5	*10*	*10*	*10*	*10*	*5*	*10*	*10*	*10*	*10*	*5*
Riverine species	5	4	5	6	9	6	5	6	9	5	7
	10	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*
% Riverine (n)	5.9	30.7	10.9	9.7	60	28.9	40.3	49.7	46.4	67.9	82.7
	0	*5*	*5*	*0*	*10*	*5*	*10*	*10*	*10*	*10*	*10*
% Lithophils (n)	6.2	40	23	15.6	65.8	18.1	42.9	54	44.8	70	78.8
	0	*0*	*0*	*0*	*5*	*0*	*0*	*5*	*5*	*10*	*10*
% Insectivore (wt)	69.5	28.5	78.3	25.3	57.9	76	9.5	50.9	49.9	62.7	12.1
	10	*5*	*10*	*5*	*5*	*10*	*0*	*5*	*5*	*10*	*5*
IBI Score	60	60	70	60	75	60	50	70	70	80	70

Tables 2a-2d continued

b)

2002					Sites				
Site	CA01	CA02	CA03	CA04	CA05	CA06	CA07	CA08	CA10
WPUE	35.8	25.2	18.6	26.9	36.5	27.8	18.1	18.3	9.4
	10	*10*	*5*	*10*	*10*	*10*	*5*	*5*	*0*
Native species	12	16	17	12	14	12	13	12	14
	10	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*
% DELT (n)	12.0	0.7	0.3	1.0	2.4	3.4	1.1	0.0	0.0
	0	*5*	*5*	*5*	*5*	*0*	*5*	*10*	*10*
% Round suckers (wt)	64.8	47.3	72.3	30.7	64.8	74.2	41.1	41.4	75.4
	10	*5*	*10*	*5*	*10*	*10*	*5*	*5*	*10*
Sucker species	4	4	4	3	5	5	4	4	2
	5	*5*	*5*	*5*	*10*	*10*	*5*	*5*	*0*
Intolerant species	1	3	2	3	1	0	1	1	1
	0	*10*	*5*	*10*	*0*	*0*	*0*	*0*	*0*
Riverine species	5	6	7	4	6	6	7	6	6
	10	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*
% Riverine (n)	21.7	6.5	13.7	83.7	83.7	74.8	91.2	75.7	94.5
	5	*0*	*5*	*10*	*10*	*10*	*10*	*10*	*10*
% Lithophils (n)	50.6	17.6	31.4	87.5	83.7	92.4	93.5	78.5	96.8
	5	*5*	*5*	*10*	*10*	*10*	*10*	*10*	*10*
% Insectivore (wt)	64.8	54.6	72.3	30.7	65.3	74.2	41.1	41.4	75.4
	10	*5*	*10*	*5*	*10*	*10*	*5*	*5*	*10*
IBI Score	65	65	70	80	85	80	65	70	70

Tables 2a-2d continued

c)

Year	Sites											
2013	US01	US02	US03	US04	US05	US06	US07	US08	US09	US10	US11	US12
WPUE	24.8	37.3	15.2	20.5	11.2	22.8	8.7	22.0	16.3	8.8	11.0	13.6
	5	*10*	*5*	*5*	*5*	*5*	*5*	*5*	*5*	*0*	*5*	*5*
Native species	19	15	21	23	22	22	14	24	18	15	23	24
	10	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*
% DELT (n)	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.1	4.9	3.8	0.2	0.8
	10	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*0*	*0*	*10*	*5*
% Round suckers (wt)	59.8	79.3	78.9	74.8	68.1	41.8	65.0	91.5	63.8	54.3	33.9	64.3
	10	*10*	*10*	*10*	*10*	*5*	*10*	*10*	*10*	*5*	*5*	*10*
Sucker species	3	5	3	4	4	3	2	4	3	4	5	4
	5	*10*	*5*	*5*	*5*	*5*	*0*	*5*	*5*	*5*	*10*	*5*
Intolerant species	5	3	5	7	5	4	3	6	5	2	6	5
	10	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*5*	*10*	*10*
Riverine species	4	6	7	6	8	8	6	11	12	6	10	12
	10	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*
% Riverine (n)	0.6	1.7	2.8	1.7	4.5	6.4	16.9	13.8	42.6	22.2	3.1	24.3
	0	*0*	*0*	*0*	*0*	*0*	*5*	*5*	*10*	*5*	*0*	*5*
% Lithophils (n)	0.8	3.2	8.2	2.5	7.5	5.2	28.2	22.1	49.2	29.6	2.7	23.0
	0	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*5*	*0*	*0*	*0*
% Insectivore (wt)	67.9	80.4	84.6	79.7	73.2	51.3	70.9	92.9	81.7	54.7	43.3	68.6
	10	*10*	*10*	*10*	*10*	*5*	*10*	*10*	*10*	*5*	*5*	*10*
IBI Score	70	80	70	70	70	60	70	75	75	45	65	70

Tables 2a-2d continued

d)

2013	Sites											
Metrics	CA01	CA02	CA03	CA04	CA05	CA06	CA07	CA08	CA09	CA10	CA11	CA12
WPUE	36.2	47.0	29.6	45.0	24.3	84.2	10.6	32.1	20.0	17.2	33.3	9.9
	10	*10*	*10*	*10*	*5*	*10*	*5*	*10*	*5*	*5*	*10*	*0*
Native species	18	18	23	24	23	21	19	18	20	22	22	22
	10	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*
% DELT (n)	1.0	0.6	0.4	0.0	0.0	0.9	0.0	0.8	0.2	0.3	0.8	0.0
	5	*5*	*10*	*10*	*10*	*5*	*10*	*5*	*10*	*10*	*5*	*10*
% Round suckers (wt)	62.0	67.5	53.2	66.0	59.1	83.7	44.8	64.0	41.1	71.7	90.1	64.5
	10	*10*	*5*	*10*	*5*	*10*	*5*	*10*	*5*	*10*	*10*	*10*
Sucker species	4	4	5	5	4	5	2	4	4	4	4	2
	5	*5*	*10*	*10*	*5*	*10*	*0*	*5*	*5*	*5*	*5*	*0*
Intolerant species	5	4	6	4	4	4	6	4	3	6	4	4
	10	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*
Riverine species	6	5	8	8	7	10	6	10	10	8	10	6
	10	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*	*10*
% Riverine (n)	2.5	1.6	8.4	2.5	3.1	8.8	2.1	11.4	3.4	10.3	8.5	2.8
	0	*0*	*0*	*0*	*0*	*0*	*0*	*5*	*0*	*5*	*0*	*0*
% Lithophils (n)	7.0	5.5	9.8	4.8	4.5	9.8	3.0	13.9	3.9	9.2	12.3	2.9
	0	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*
% Insectivore (wt)	66.4	69.1	55.1	70.5	67.8	88.2	49.5	68.9	54.4	75.3	93.4	74.8
	10	*10*	*5*	*10*	*10*	*10*	*5*	*10*	*5*	*10*	*10*	*10*
IBI Score	70	70	70	80	65	75	55	75	60	75	70	60

Table 3. Species captured and total number of individuals captured from the Rainy River during the 2002 and 2013 Index of Biotic Integrity (IBI).

Common name	Scientific name	2002	2013
American brook lamprey	*Lampetra appendix*	8	1
Silver lamprey	*Ichthyomyzon unicuspis*	8	8
Smallmouth bass	*Micropterus dolomieu*	130	106
Rock bass	*Ambloplites rupestris*	9	25
Pumpkinseed	*Lepomis gibbosus*	0	1
Largemouth bass	*Micropterus salmoides*	3	1
Black crappie	*Pomoxis nigromaculatus*	45	301
Bluegill	*Lepomis macrophirus*	1	0
Common shiner	*Luxilus cornutus*	27	71
Creek chub	*Semotilus atromaculatus*	2	0
Hornyhead chub	*Nocomis biguttatus*	3	8
Golden shiner	*Notemigonus crysoleucas*	0	2
Emerald shiner	*Notropis atherinoides*	1141	288
River shiner	*Notropis Blennius*	32	162
Blacknose shiner	*Notropis heterolepis*	4	3
Spottail shiner	*Notropis hudsonius*	105	244
Rosyface shiner	*Notropis rubellus*	0	1
Mimic shiner	*Notropis volucellus*	0	99
Fathead minnow	*Pimephales promeles*	1	0
Blacknose dace	*Rhinichthys atratulus*	0	2
Longnose dace	*Rhinichthys cataractae*	1	0
Brassy minnow	*Hybognathus hankinsoni*	1	1
Central mudminnow	*Umbra limi*	1	12
Quillback	*Carpiodes cyprinus*	39	25
White sucker	*Catostomus commersoni*	99	210
Silver redhorse	*Moxostoma anisurum*	167	129
Golden redhorse	*Moxostoma erythrurum*	8	46
Shorthead redhorse	*Moxostoma macrolepidotum*	124	366
Rainbow smelt	*Osmerus mordax*	1	1
Cisco	*Coregonus artedi*	1	134
Burbot	*Lota lota*	1	3
Iowa darter	*Etheostoma exile*	0	8
Johnny darter	*Etheostoma nigrum*	42	84
Logperch	*Percina caprodes*	36	9
Blackside darter	*Percina maculata*	96	101
River darter	*Percina shumardi*	47	51
Yellow perch	*Perca flavescens*	1691	15789
Sauger	*Stizostedion canadense*	13	32

Table 3 continued

Common name	Scientific name	2002	2013
Walleye	*Stizostedion vitreum*	178	194
Lake sturgeon	*Acipenser fulvescens*	0	3
Mooneye	*Hiodon tergisus*	8	24
Mottled sculpin	*Cottus bairdi*	2	26
Muskellunge	*Esox masquinongy*	0	2
Northern pike	*Esox lucius*	47	168
Nine-spine stickleback	*Pungitius pungitius*	5	14
Trout Perch	*Percopsis omiscomaycus*	297	1
Tadpole madtom	*Noturus gyrinus*	0	1

Appendix A. Rainy River Index of Biotic Integrity site coordinates (Datum: NAD83).

Station	Start Latitude	Start Longitude	End Latitude	End Longitude
CA01	48.6098	-93.4098	48.6019	-93.4265
CA02	48.5938	-93.4528	48.5888	-93.4689
CA03	48.5422	-93.4999	48.5360	-93.5180
CA04	48.5286	-93.5978	48.5245	-93.6166
CA05	48.5257	-93.6387	48.5180	-93.6543
CA06	48.5194	-93.7294	48.5173	-93.7510
CA07	48.5282	-93.8136	48.5403	-93.8094
CA08	48.5860	-93.8083	48.5994	-93.8157
CA09	48.6233	-93.8304	48.6313	-93.8462
CA10	48.6337	-93.9329	48.6360	-93.9539
CA11	48.6420	-93.9859	48.6445	-94.0074
CA12	48.6448	-94.0907	48.6456	-94.1108
US01	48.6012	-93.4239	48.5923	-93.4406
US02	48.5886	-93.4648	48.5747	-93.4609
US03	48.5515	-93.4598	48.5438	-93.4763
US04	48.5294	-93.5404	48.5283	-93.5619
US05	48.5258	-93.6223	48.5263	-93.6352
US06	48.5154	-93.6794	48.5154	-93.7004
US07	48.5147	-93.7534	48.5139	-93.7753
US08	48.5842	-93.8122	48.5978	-93.8206
US09	48.6119	-93.8258	48.6245	-93.8349
US10	48.6302	-93.8744	48.6311	-93.8965
US11	48.6359	-93.9600	48.6381	-93.9811
US12	48.6397	-94.0359	48.6425	-94.0567