North American Journal of Fisheries Management 32:313–324, 2012

ISSN: 0275-5947 print / 1548-8675 online
DOI: 10.1080/02755947.2012.678560

ARTICLE

Gear-Type Influences on Fish Catch and a Wetland Fish Index in Georgian Bay Wetlands

Maja Cvetkovic,* Kristina Kostuk, and Patricia Chow-Fraser
Biology Department, McMaster University, 1280 Main Street West, Hamilton, Ontario L8S 4K1, Canada

Abstract
The Laurentian Great Lakes are managed by many jurisdictions that use a variety of survey methods and gear types to monitor fish assemblages in coastal marshes. Lack of standardization in these methods makes it difficult for organizations to compare data because of inherent biases in gear types. Of equal concern is the uncertainty of the effect of gear bias on fish-related index scores for ecosystem health. Our first objective was to investigate whether there were differences in catch data between two commonly used sampling gears: fyke nets (FN) and boat electrofishing (EF). Secondly, we investigated whether catch differences in data associated with gear biases can lead to significant differences when these data are used to generate scores for biotic indices such as the published Wetland Fish Index (WFI). We sampled 26 coastal wetlands in Georgian Bay (Lake Huron) in the summers of 2004 and 2005. A majority (73%) of the more than 10,000 fish were caught by FN; this gear also captured a greater number of species and functional taxa and selected for larger piscivores. By comparison, EF captured larger invertivores. Fyke nets were more selective for individuals from the Centrarchidae, Cyprinidae, and Ictaluridae families, while EF was more effective for darters (e.g., the Iowa darter *Etheostoma exile* and johnny darter *E. nigrum*) and white suckers *Catostomus commersonii*. Despite these biases in catch data, we obtained statistically similar WFI scores with both gear types. Therefore, although the fish abundance and species composition information collected from FN and EF are not directly comparable, when necessary they can be used interchangeably to generate a fish-based index of ecosystem health.

Coastal wetlands are primary spawning and nursery grounds for Great Lakes fishes and, as such, are valuable habitats that require consistent monitoring (Jude and Pappas 1992). Although fisheries research is an active field within the Great Lakes, standardization of fish sampling has been identified as an unresolved issue for sampling streams, rivers, wetlands, and lakes (Bonar and Hubert 2002). Sample methodologies can affect the quality of the data collected, and this can affect conclusions regarding trends in fish assemblages over time (Bonar and Hubert 2002).

Many environmental organizations, including provincial and national parks, conservation authorities, and First Nations, routinely sample fish assemblages as part of their monitoring of aquatic ecosystems. This often includes collecting data such as presence or absence, abundance, length, and weight of fishes. Often, more than one agency has the shared responsibility for assessing fish assemblages in large watersheds and water bodies. Data sharing allows different programs, environmental agencies, and researchers with limited funds to build an appropriately large database (e.g., Jude and Pappas 1992; Wei et al. 2004; Cao et al. 2005) to properly manage these large watersheds. This, however, requires a consistent choice of gear by the various organizations to ensure that data can be compared and analyzed in a statistically valid manner (Bonar and Hubert 2002; McGeoch and Gaston 2002).

Currently, fisheries managers across the Great Lakes region employ both active and passive fish sampling gears, using a range of effort. Active gears include methods such as seining and electrofishing, which are methods used to capture and analyze fish immediately (Hayes 1989). Electrofishing generally tends to select for sedentary fish species, such as large predators (Bohlin et al. 1989; Reynolds 1989). Passive gears capture fishes by entanglement or entrapment devices that are set for a certain

*Corresponding author: maja.cvet@gmail.com
Received May 16, 2011; accepted December 8, 2011

period of time, and include hoop nets, minnow traps, and fyke nets (FNs; Hubert 1989). These gears tend to be biased towards mobile fishes, which are usually smaller and school (Bohlin et al. 1989; Hubert 1989). Two commonly employed fish sampling methods are FNs and boat electrofishing (EF). Fyke nets can be set unpaired or paired, with two wings extending out from either side, forcing the fishes in through successively smaller hoops (Brazner and Beals 1997; Seilheimer and Chow-Fraser 2006, 2007; Ruetz et al. 2007), while EF stuns the fishes and forces most species to float to the surface, enabling technicians to retrieve them (Bohlin et al. 1989; Reynolds 1989).

Studies have shown that every gear is selective to a certain degree (e.g., Weaver et al. 1993; Onorato et al. 1998; Chow-Fraser et al. 2006; Lapointe et al. 2006), which is why it is important to identify how different gear types influence catch of particular species for a given habitat. There seems to be a general consensus that accurate assessments of fish assemblages are best achieved by a combination of gears (Paukert 2004; Butcher et al. 2005; Clavero et al. 2006; Ruetz et al. 2007), although no study has compared all gears possible in one habitat. Conversely, studies aimed at researching specific taxa of interest or species at risk tend to find that one method is generally superior to others when a variety of gears are compared (Schwanke and Hubert 2004; Benson and Sutton 2005; Mangan et al. 2005; Hardie et al. 2006; Poos et al. 2007). While many studies have concentrated on the efficacy of gear in assessing fish assemblage or species-at-risk, there is a considerable lack of information on the influence of gear biases on ecological indicators, such as fish-based indices of ecosystem health (e.g., Gammon and Simon 2000; Hughes and Herlihy 2007).

Ecologists are increasingly using fish assemblages as indicators of ecosystem health in addition to conducting routine sampling for population characteristics. Following the introduction of the fish-based index of biotic integrity (IBI; Karr 1981), other indices have been developed for streams and rivers (Simon and Emery 1995; Simon and Lyons 1995; Angermeier and Karr 1986; Emery et al. 2003), Great Lakes Areas of Concern (Minns et al. 1994; Simon and Lyons 1995), lakes (Drake and Pereira 2002), and the coastal wetlands of the Laurentian Great Lakes (IBI, Uzarski et al. 2005; Wetland Fish Index [WFI], Seilheimer and Chow-Fraser 2007). These tools offer a relatively quick and effective way for managers to assess ecosystem condition on a routine basis and to track changes related to human disturbance or remedial actions (Seilheimer and Chow-Fraser 2007; Seilheimer et al. 2009).

Presently, we do not know the extent to which coastal wetland fish indicators are biased by different sampling techniques. As such, comparison of fish index scores derived from different gears, or single versus multiple gears, may be inappropriate since the only research available has shown that no one gear can accurately sample the entire fish community and that sampling biases are inherent to each method (Chow-Fraser et al. 2006; Ruetz et al. 2007). While some indices of biotic integrity have incorporated multiple sampling gears in their metric use (Gammon and Simon 2000; Drake and Pereira 2002), very few studies have tested the effect of multiple sampling methods on indicator response (see Gammon and Simon 2000; Seilheimer and Chow-Fraser 2007). As the focus of fisheries-related research shifts to overall habitat health, studies regarding the reliability of index performance are essential if we are to move forward with the science of ecosystem-based indicators in the Great Lakes.

Previous studies have compared the performance of FNs and EF in coastal wetlands of Lakes Erie and Ontario (Chow-Fraser et al. 2006), and of eastern Lake Michigan (Muskegon Lake; Ruetz et al. 2007). Chow-Fraser et al. (2006) found that gear type significantly affected total catch, biomass, size, and the functional taxa that were caught. Ruetz et al. (2007) found that species composition and size distribution were significantly different between FN and nighttime EF. Consistent with Chow-Fraser et al. (2006), FNs captured more small-bodied fishes (e.g., family Cyprinidae), whereas EF caught more large-bodied fishes such as salmonids, catostomids, walleye *Sander vitreus*, and common carp *Cyprinus carpio*. They concluded that the fish assemblage was best assessed using a combination of both gears (Ruetz et al. 2007).

In this study, we assess the performance of 24-h FNs and daytime EF in coastal wetlands of eastern and northern Georgian Bay (Lake Huron) to determine the selectivities associated with each method. We predict that the fish assemblage will differ according to gear biases associated with each sampling method, and we test this using a variety of characteristics, including richness, total catch, biomass, length, weight, and trophic guild. This empirical comparison will contribute information to the catalog of studies aimed at enhancing knowledge related to fisheries sampling and management. Second, we investigate whether a fish-based index that uses species information to infer water quality, the WFI (Seilheimer and Chow-Fraser 2007), is sufficiently robust to override biases that are associated with each gear. Specifically, we test the assumption that catch information collected from two different survey methods known to select for different fish taxa, trophic guilds, and total catch would generate significantly different WFI scores.

STUDY SITES

We sampled 26 sites, spanning the basin of Georgian Bay (Figure 1), which is situated in the eastern basin of Lake Huron. With over 5,000 km of shoreline, including thousands of islands, coastal marsh habitat is abundant throughout Georgian Bay, especially along the eastern and northern shore (see Cvetkovic and Chow-Fraser 2011; Midwood et al., in press). The bay is composed primarily of pre-Cambrian granitic bedrock, although the southern portion transitions into the softer sedimentary limestone common in the southern Great Lakes (Weiler 1988). As a result of its geology, the coastal waters along the eastern and northern shores of this basin are dystrophic (see Midwood and Chow-Fraser 2010; DeCatanzaro and Chow-Fraser 2011).

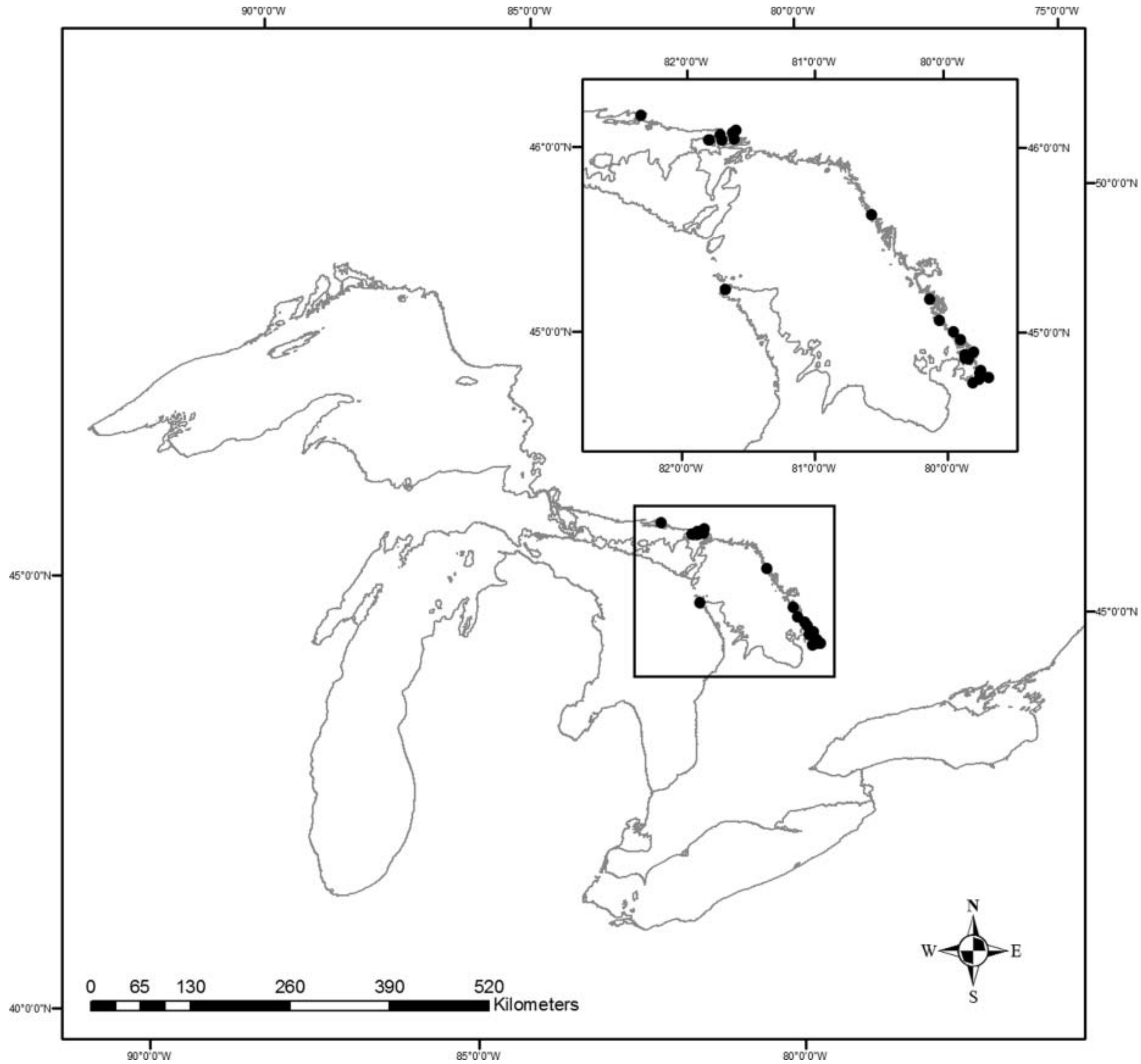

FIGURE 1. Location of 26 coastal wetlands sampled in Georgian Bay, Lake Huron, during 2004 and 2005.

Wetlands in Georgian Bay are primarily lacustrine with direct lake water connectivity (see Albert et al. 2005), and while most marshes are found in protected embayments similar to those in the lower lakes, some are distributed along the exposed shoreline (Cvetkovic 2008). Compared with other areas in the Great Lakes, Georgian Bay has been relatively undersampled, and only recently have studies revealed the low-nutrient, low-turbidity characteristics of these waters (Chow-Fraser et al. 2006; DeCatanzaro et al. 2009; Cvetkovic and Chow-Fraser 2011), with high species diversity and taxa exhibiting low tolerance to disturbance (Croft and Chow-Fraser 2007; Seilheimer and Chow-Fraser 2007). Despite the high percentage of excellent quality sites, a few marshes in the southern portion of Georgian Bay are showing signs of human-induced degradation (Cvetkovic 2008; Cvetkovic and Chow-Fraser 2011).

METHODS

Fyke netting.—Wetlands were visited once per season in the summers of 2004 and 2005. A total of six FNs were set in pairs at each site. Two sets of paired large nets (height × width = 1-m × 1.25-m rectangular opening, 4.25-m length, one net with 13-mm nylon mesh and the other with 4-mm nylon mesh), and one set of paired small nets (1-m × 0.5-m rectangular opening, 2.1-m length, 4-mm nylon mesh for both) were placed in submergent vegetation. If no submergent beds were available, nets were set along the edge of emergent vegetation. Large nets were set at depths from 0.5 to 1 m, and small FNs were set in water from 0.25 to 0.5 m deep. Fyke nets were paired with their mouth openings facing each other and connected by an 8-m lead (4-mm mesh). The nets were positioned parallel to the shoreline, the wings (2.5 m, 4-mm mesh) extending out at

45° angles from the net opening. Nets, wings and lead were secured in place using six steel poles. These nets remained in the water for a 24-h period.

Boat electrofishing.—Electrofishing sampling occurred during the day, between 0900 and 1600 hours. Georgian Bay coastal wetlands are notorious for the prevalence of rocky shoals, and we often required the aid of local residents to help us reach our sites without damaging the boat. As a result, it was unsafe to conduct EF surveys at night, which is a common sampling time. Electrofishing was conducted using a Smith-Root SR-16EB electrofisher (boat length = 5.5 m, draft up to 80–100 mm) with a 7.5 GPP generator. Two round, 1-m diameter LPA-6 anode arrays were extended on a pair of booms mounted on opposite sides of the bow of the boat at 25° from the center. Electrofishing settings were 60 pulses/s DC, and output ranged from 3 to 6 amps and 300–600 V, based on conductivity of the water. Conductivity ranged from 52 to 321 μS/cm (mean ± SE = 136.73 ± 8.26 μS/cm). These settings are in accordance with numerous other studies (Bohlin et al. 1989; Brousseau et al. 2005; Ruetz et al. 2007; Eros et al. 2009). Boat speed was maintained at idle, allowing netters to obtain stunned fishes.

Transects typically occurred within 1 m of shoreline or emergent vegetation (and through submergent vegetation wherever possible) at 0.5- to 1.0-m depths, and always coincided with FN placement. Electrofishing was completed either the day before or the day after FN placement, depending on weather. Normally, three transects were completed at an average of 300–500 shock seconds, for an approximate total of 1,000–1,500 shock seconds; total time varied with area of the wetland. At some sites, less effort (i.e., fewer shock seconds) was expended because of the small size of the wetland (see similar methods in Gammon and Simon 2000; Chow-Fraser et al. 2006; Ruetz et al. 2007; Seilheimer and Chow-Fraser 2007; Eros et al. 2009). Mean transect length ranged from 200 to 500 m, according to the size of the wetland. During sampling, one person retrieved fishes at the bow of the boat, and the boat operator maintained the settings and retrieved any missed fishes.

Sample processing.—Geographic coordinates of all FNs and transects were recorded with a Garmin Etrex Summit handheld global positioning system (Garmin eTrex GPS) at 3–6-m accuracy (see Figure 1). Fishes were all identified to species according to Scott and Crossman (1973) and Holm et al. (2009), except for some age-0 fish (e.g., sunfishes). Fishes were counted and fork length was measured to the nearest millimeter. Biomass was estimated using published length–weight regressions (Schneider et al. 2000) from the measured fishes. If more than 15 individuals of a species were present, only 15 randomly chosen individuals were measured; the remainder was enumerated, and an average length and biomass were obtained. For FN sampling, fishes were removed from the FN and placed into a plastic container filled with water for identification and measurement, then released in the same location. During EF sampling, all netted fishes were placed into a live well on board. Upon completion of each transect, fishes were identified and measured. Processed fish were released unharmed at the capture site.

Ecological indices of wetland quality.—Seilheimer and Chow-Fraser (2007) developed the Wetland Fish Index (WFI_{basin}) to assess the degree of human disturbance associated with water quality impairment in coastal wetlands of the Great Lakes. As such, the WFI_{basin} is used as a surrogate to determine water quality information and not biotic integrity. One hundred wetlands were surveyed in order to obtain an adequate sample size and disturbance gradient in the basin. The WFI_{basin} is based on relationships between fish species and water chemistry variables, as determined empirically by multivariate analysis, and utilizes species-specific values of tolerance based on these relationships. The index can use presence/absence (WFI_{basin}-PA) or abundance (WFI_{basin}-AB) fish data in the equation

$$WFI = \frac{\sum_{i=1}^{n} Y_i T_i U_i}{\sum_{i=1}^{n} Y_i T_i},$$

where Y_i is the presence or $\log_{10}$ abundance ($\log[x + 1]$) of species i, T_i is the value from 1 to 3 (indicating niche breadth), and U_i is the value from 1 to 5 (indicating tolerance of degradation). Seilheimer and Chow-Fraser (2007) published U and T scores for 61 fish species of the Great Lakes, allowing for easy calculation and implementation of the index. The WFI_{basin} (hereafter referred to simply as WFI) ranges in values from 1 to 5, higher scores indicating higher quality.

This fish-based index has been used successfully to distinguish between impacted and nonimpacted regions of a wetland in Southern Ontario (Seilheimer et al. 2007), and is correlated with independent measures of wetland quality such as the Water Quality Index and the Wetland Macrophyte Index (Seilheimer et al. 2009). As part of the development and assessment of the WFI, Seilheimer and Chow-Fraser (2007) sampled 31 sites across the Great Lakes using both FNs and EF in order to determine whether data sets from both methods could be used interchangeably to generate WFI scores. A range of effects was seen in terms of species richness and index scores, depending on lake. Since the effects of lake, geography, and water quality influenced their results, we set out to limit such variables in our exploration of gear bias. Here, we purposely surveyed a range of sites in the relatively undersampled region of Georgian Bay to determine the full effect of gear bias on index scores. In this study, we generated WFI scores for all 26 wetlands to determine whether sampling method affects this biotic index in this region of the Great Lakes (see Table 1).

Statistical analyses: fish assemblage.—All data manipulation and statistical analyses were performed with SAS JMP 8.0 software. A number of summary variables were calculated for each method, including total catch, biomass, species richness, functional taxa (i.e., a species that is classified into two or more trophic guilds based on size), and length of fish. These variables were $\log_{10}(x + 1)$ transformed and tested for normality and homogeneity of variances using the appropriate tests. Any

TABLE 1. Summary of the 26 Georgian Bay wetlands sampled with fyke nets (FN) and boat electrofishing (EF) in this study. Presence/absence (PA) and abundance (AB) Wetland Fish Index (WFI) scores (Seilheimer and Chow-Fraser 2007) are included for each method and site. There were no significant differences between FN and EF with respect to mean WFI-PA scores (3.73 versus 3.65, respectively; paired t-test: $P = 0.31$, df = 25) or mean WFI-AB scores (3.65 versus 3.58, respectively; paired t-test: $P = 0.24$, df = 25).

		WFI-PA		WFI-AB	
Wetland	Date sampled	FN	EF	FN	EF
Boom Camp	10 Aug 2004	3.65	3.09	3.41	3.20
Charles Inlet	7 Jul 2004	3.83	3.67	3.93	3.79
Dead Horse Bay	9 Aug 2005	3.08	3.81	3.08	3.72
Dogfish Bay	10 Aug 2005	3.79	3.81	3.24	3.50
Garden Channel	23 Jun 2004	4.00	4.00	4.05	4.09
Green Island North	2 Jun 2004	3.80	3.00	3.91	3.28
Green Island North	8 Jun 2005	3.70	3.81	3.67	3.68
Green Island South	9 Jun 2005	3.77	3.68	3.75	3.60
Hay Bay 1	5 Jul 2005	3.88	3.44	3.60	3.31
Hog Bay	7 Jun 2004	3.69	3.76	3.68	3.45
Iroquois Bay	5 Aug 2004	3.64	3.96	3.42	3.74
Jumbo Bay	8 Aug 2004	3.91	3.77	3.91	3.85
Jumbo Bay	13 Aug 2005	3.81	3.75	3.91	3.77
Longuissa Bay	28 Jun 2004	3.55	3.60	3.43	3.24
Matchedash Bay	27 Jul 2004	3.69	3.23	3.52	3.06
Moose Bay	14 Jun 2004	3.78	4.07	3.85	4.25
Moreau Bay	16 Jun 2004	3.97	3.71	3.99	3.57
North Bay	15 Jun 2005	3.70	3.67	3.51	3.71
Oak Bay	8 Jun 2004	3.63	3.50	3.53	3.27
Ojibway	16 Jun 2005	3.70	3.82	3.64	3.60
Robert's Bay	1 Jun 2004	3.89	3.47	3.79	3.30
Sturgeon Bay South	26 Jul 2004	3.42	3.76	3.34	3.50
Treasure Bay	13 Jun 2005	3.67	3.89	3.55	3.54
Vincent's Bunk	6 Aug 2004	3.38	3.81	3.08	3.55
Wardrope Island	4 Aug 2004	3.88	3.60	4.13	3.78
Wardrope Island	9 Aug 2005	4.05	3.29	4.10	3.66

variables not meeting the assumptions are reported using nonparametric statistics.

These parameters were compared with a paired t-test to distinguish any significant differences between gears, while controlling for within-site variations. In order to determine the extent of the fish assemblage sampled by each gear, we first determined total species richness by pooling data from both gears on a per-wetland basis. We then calculated the proportion of species caught by each gear relative to this "total" species richness, and conducted a paired t-test to determine whether there were differences associated with gear.

Species were grouped according to trophic guild, which encompassed planktivores, omnivores, invertivores, invertivore–piscivores, and piscivores, as per feeding behaviors described in Scott and Crossman (1973). To determine whether there was a gear bias towards specific trophic guilds, we used a paired t-test to compare frequencies of each group for each wetland by gear type. We then conducted a two-way analysis of variance (ANOVA) to determine whether set type had an effect on the size (length and weight) of a particular trophic guild.

Paired t-tests were used to compare differences in catch, length, and weight for each species by gear type. A minimum of two individuals of a species had to be captured by either gear in order to statistically compare the differences. The following species had to be separated according to length and tested in two separate categories because they occupy different trophic guilds based on their size: black crappie *Pomoxis nigromaculatus*, largemouth bass *Micropterus salmoides*, smallmouth bass *M. dolomieu*, and yellow perch *Perca flavescens* (see Table 2 for size specifications). All species were tested for differences in length and weight between gears using a paired t-test, except for brook stickleback, mottled sculpin, ninespine stickleback, and white bass because their total catch only equaled one.

Wetland Fish Index.—We compared the WFI scores generated from catch data collected by both methods. We used

TABLE 2. Comparison of the total number of species caught by FN and EF. Asterisks indicate significant differences in species abundance between gears (paired t-test; $P < 0.05$).

	Number of specimens			Number of wetlands		
Species	Both	FN	EF	Both	FN	EF
Alewife *Alosa pseudoharengus*	2	1	1	2	1	1
Banded killifish *Fundulus diaphanus*	54	31	23	12	6	8
Black crappie *Pomoxis nigromaculatus*[a]*	109	92	17	9	9	4
Black crappie[b]*	7	5	2	4	4	1
Blackchin shiner *Notropis heterodon*	251	133	118	22	14	13
Blacknose shiner *N. heterolepis**	628	588	40	11	11	6
Bluegill *Lepomis macrochirus**	548	490	58	4	4	4
Bluntnose minnow *Pimephales notatus**	1,520	1,319	201	23	22	22
Bowfin *Amia calva**	76	59	17	18	18	8
Brook silverside *Labidesthes sicculus*	61	10	51	12	7	10
Brook stickleback *Culaea inconstans*	1	1	0	1	1	0
Brown bullhead *Ameiurus nebulosus**	261	183	78	23	22	15
Bullhead (juvenile) *Ameiurus* spp.	2	1	1	2	1	1
Central mudminnow *Umbra limi*	10	1	9	2	1	1
Common carp *Cyprinus carpio*	6	4	2	4	2	2
Common shiner *Luxilus cornutus*	202	71	131	10	5	9
Emerald shiner *Notropis atherinoides*	20	15	5	4	3	3
Golden shiner *Notemigonus crysoleucas*	171	111	60	14	8	9
Iowa darter *Etheostoma exile**	30	3	27	13	2	12
Johnny darter *E. nigrum**	39	8	31	11	4	11
Largemouth bass *Micropterus salmoides*[c]	661	450	211	14	13	12
Largemouth bass[d]	71	27	44	16	11	12
Logperch *Percina caprodes*	26	6	20	8	4	5
Longear sunfish *Lepomis megalotis**	145	126	19	7	7	3
Longnose gar *Lepisosteus osseus*	31	23	8	14	10	7
Mimic shiner *Notropis volucellus**	426	368	58	15	12	8
Mottled sculpin *Cottus bairdii*	2	2	0	1	1	0
Muskellunge *Esox masquinongy**	2	2	0	2	2	0
Ninespine stickleback *Pungitius pungitius*	1	1	0	1	1	0
Northern pike *Esox lucius*	25	16	9	11	8	4
Northern redbelly dace *Phoxinus eos*	8	6	2	3	3	1
Pumpkinseed *Lepomis gibbosus**	2,794	2,105	689	25	25	20
Rock bass *Ambloplites rupestris**	342	329	13	25	24	10
Round goby *Neogobius melanostomus*	3	2	1	2	2	1
Sand shiner *Notropis stramineus*	42	9	33	4	2	2
Shiners (juvenile) *Notropis* spp.	7	0	7	3	0	2
Shorthead redhorse *Moxostoma macrolepidotum*	45	44	1	2	2	1
Smallmouth bass *Micropterus dolomieu*[e]	19	15	4	6	5	3
Smallmouth bass[f]	17	8	9	9	6	6
Spotfin shiner *Cyprinella spiloptera*	7	4	3	3	2	1
Spottail shiner *Notropis hudsonius*	114	22	92	17	8	13
Sunfish (juvenile) *Lepomis* spp.*	458	401	57	17	17	8
Tadpole madtom *Noturus gyrinus*	28	28	0	7	7	0
White bass *Morone chrysops*	1	0	1	1	0	1
White perch *M. americana*	8	1	7	1	1	1
White sucker *Catostomus commersonii**	21	3	18	10	3	9
Yellow perch *Perca flavescens*[g]	964	388	576	26	23	23
Yellow perch[h]	52	23	29	19	11	15

[a]0–160 mm, [b]>160 mm, [c]0–100 mm, [d]>100 mm, [e]20–80 mm, [f]>80 mm, [g]<150 mm, [h]>150 mm.

both the presence and abundance data to generate WFI-PA and WFI-AB scores for each sampling method, and a paired t-test was used to determine whether significant differences existed among scores between gears. We also calculated a WFI-PA score for each site using combined data from both gears (WFI-PAtotal). These total site scores were compared with each of the FN and EF WFI-PA scores using a paired *t*-test in order to determine whether index scores generated using both gears would be comparable to scores generated by a single gear.

RESULTS

Wetlands were sampled between June and August, and water temperatures ranged from 14.38°C to 28.60°C (mean ± SE = 21.79 ± 0.46°C) over both years. Overall, we caught 10,320 fishes (44 species belonging to 16 families) with both sampling methods. Common species included blackchin shiner, bluntnose minnow, largemouth bass, pumpkinseed, and juvenile yellow perch. Fyke nets captured 73% of the fishes caught (7,535), while EF captured the remaining 22% (2,785; Table 2). We saw similar trends in terms of biomass, 69% (431 kg) of the total biomass being attributable to FN.

Of the 44 species identified in total, 42 were caught by FN compared with 39 by EF (Table 2). Mean species richness in each wetland was 11.9 (Table 2). Species that were only caught by FN included brook stickleback, mottled sculpin, muskellunge, ninespine stickleback, and tadpole madtom; species that were only caught by EF were *Notropis* spp. and white bass (Table 3). When both gear types were combined we counted 48 taxa in total (which includes all species and distinguishes between juvenile and adult black crappie, largemouth bass, smallmouth bass, and yellow perch), 46 being captured by FN and 43 by EF. Regardless of sampling method, mean number of functional taxa per wetland was 12.7 (Table 3).

When we analyzed data on a site-by-site basis, FN caught greater number of species, higher total catch, and biomass. Fyke net had significantly greater species richness than EF (mean of 12.9 versus 10.8, respectively; paired *t*-test: $P = 0.02$, df = 25); in addition, when compared with the total species richness, FN caught a significantly greater proportion of species than EF (mean of 0.78 versus 0.64, respectively; paired *t*-test: $P = 0.02$, df = 25). Paired *t*-test ($P < 0.0001$, df = 25) showed that FN generally recovered a greater total catch and biomass (mean of 289.8 and 10.3 kg, respectively) relative to EF (107.1 and 4.6 kg, respectively). There were similar trends when data were sorted according to functional taxa (mean of 13.7 versus 11.7, respectively, for FN and EF; paired *t*-test: $P = 0.03$, df = 25). When the data were pooled for all sites, we found that EF captured significantly longer fish than did FN (*t*-test: $P < 0.0001$, df = 25; Table 2); on a site-by-site basis, however, we found no differences between methods with respect to fish lengths (Wilcoxon's signed rank test: $P = 0.93$, df = 25).

Species were sorted into trophic guilds, and the relative proportion of each category (planktivore, omnivore, invertivore, invertivore–piscivore, piscivore) was calculated per site for each method. Results from paired *t*-tests showed that invertivore–piscivores were more likely to be caught by FN than by EF gear (Wilcoxon's signed rank test: $P = 0.04$, df = 25). There were no statistical differences between methods for any other feeding categories. Results from a fixed two-way ANOVA showed that there was a significant interaction between gear type and

TABLE 3. Summary statistics for 26 Georgian Bay coastal wetlands sampled using FN and EF methods during the summer season in 2004 and 2005 (SEs in parentheses).

	Survey method		
Variable	Both methods	FN only	EF only
Number of fish caught	10,320	7,535	2,785
Percent all fish caught		73	27
Biomass of fish (kg)	386.4	267.6	118.7
Percent all fish biomass		69	31
Number of species caught	44	42	39
Percent total species caught		95	89
Number of functional taxa caught	48	46	43
Percent total functional taxa caught		96	90
Mean fish length (cm)	8.1 (0.07)	7.9 (0.08)	8.6 (0.1)
Mean fish weight (g)	37.4 (2.28)	35.5 (2.71)	42.7 (4.20)
Mean species richness per wetland	11.9 (0.50)	12.9 (0.68)	10.8 (0.70)
Mean number of functional taxa per wetland	12.7 (0.54)	13.7 (0.65)	11.7 (0.82)
Mean number fish per wetland	198.5 (22.96)	289.8 (36.1)	107.1 (13.4)
WFI-PA	3.69 (0.03)	3.73 (0.04)	3.65 (0.052)
WFI-AB	3.62 (0.04)	3.65 (0.058)	3.58 (0.054)

TABLE 4. Analysis of variance for effects of trophic guild and gear type and their interactions on log length ($Log_{10}L$) and log biomass ($Log_{10}B$), for 26 wetlands sampled by 24-h FN and EF in Georgian Bay. For both $Log_{10}L$ and $Log_{10}B$ in ANOVA, corrected total = 10,317; error df = 10,308; and overall model significance is <0.0001.

		$Log_{10}L$		$Log_{10}B$	
Source	df	*F*-ratio	*P*-value	*F*-ratio	*P*-value
Gear type (GT)	1	0.04	0.84	0.17	0.68
Trophic guild (TG)	4	808.53	<0.0001	783.09	<0.0001
GT × TG	4	46.21	<0.0001	55.74	<0.0001

trophic guild with regards to the length and biomass of fish caught (Table 4). Post hoc Tukey–Kramer analyses showed EF caught invertivores that had a significantly greater length and biomass compared with those caught by FN, whereas the reverse was true for piscivores (Figure 2). While there were no significant differences between lengths of invertivore–piscivores captured by each method, FN captured invertivore–piscivores with a significantly higher biomass than those caught by EF (Figure 2).

When we analyzed the data at a species level, there was evidence of bias in catch data between the two gears. Fyke nets captured significantly greater numbers of black crappie, blacknose shiner, bluegill, bluntnose minnow, bowfin, brown bullhead, longear sunfish, mimic shiner, pumpkinseed, rock bass, and age-0 sunfish relative to EF (paired *t*-test: $P < 0.05$; see Table 3). By contrast, EF caught greater numbers of Johnny darter, Iowa darter, and white sucker (paired *t*-test: $P < 0.05$; see Table 3) compared with FN. Length and biomass analyses showed that FN caught larger bowfin than EF (mean length ± SD = 530.9 ± 144.4 mm versus 403.6 ± 190.4 mm, respectively, Wilcoxon's signed rank test: $P = 0.008$; mean biomass ± SD = 1,859.0 ± 866.8 g versus 989.7 ± 1,092.1 g, respectively, Wilcoxon's signed rank test: $P = 0.008$, df = 7) and longer yellow perch (mean length ± SD = 105.7 ± 38.7 mm versus 96.2 ± 33.9 mm, respectively, Wilcoxon's signed rank test: $P = 0.03$, df = 21). There were no significant differences between sizes of fishes captured by the two different gears for any other species; there were also not enough comparable data points to conduct a paired t-test for a majority of species.

Wetland Fish Index PA scores associated with FN ranged from 3.08 to 4.05, while corresponding WFI-AB scores ranged from 3.08 to 4.13 (see Table 1). By comparison, WFI-PA scores for EF ranged from 3.00 to 4.07, while corresponding WFI-AB scores ranged from 3.06 to 4.25. There were no significant differences between FN and EF with respect to mean WFI-PA scores (3.73 versus 3.65 respectively; paired *t*-test: $P = 0.31$, df = 25) nor for mean WFI-AB scores (3.65 versus 3.58, respectively; paired *t*-test: $P = 0.24$, df = 25). Scores for $WFI\text{-}PA_{total}$ ranged from 3.45 to 4.00 (Table 1) and were not significantly different from FN WFI-PA scores (mean of 3.72 versus 3.73, respectively; paired *t*-test: $P = 0.92$, df = 25) or EF WFI-PA scores (mean of 3.72 versus 3.65, respectively; paired *t*-test: $P = 0.10$, df = 25).

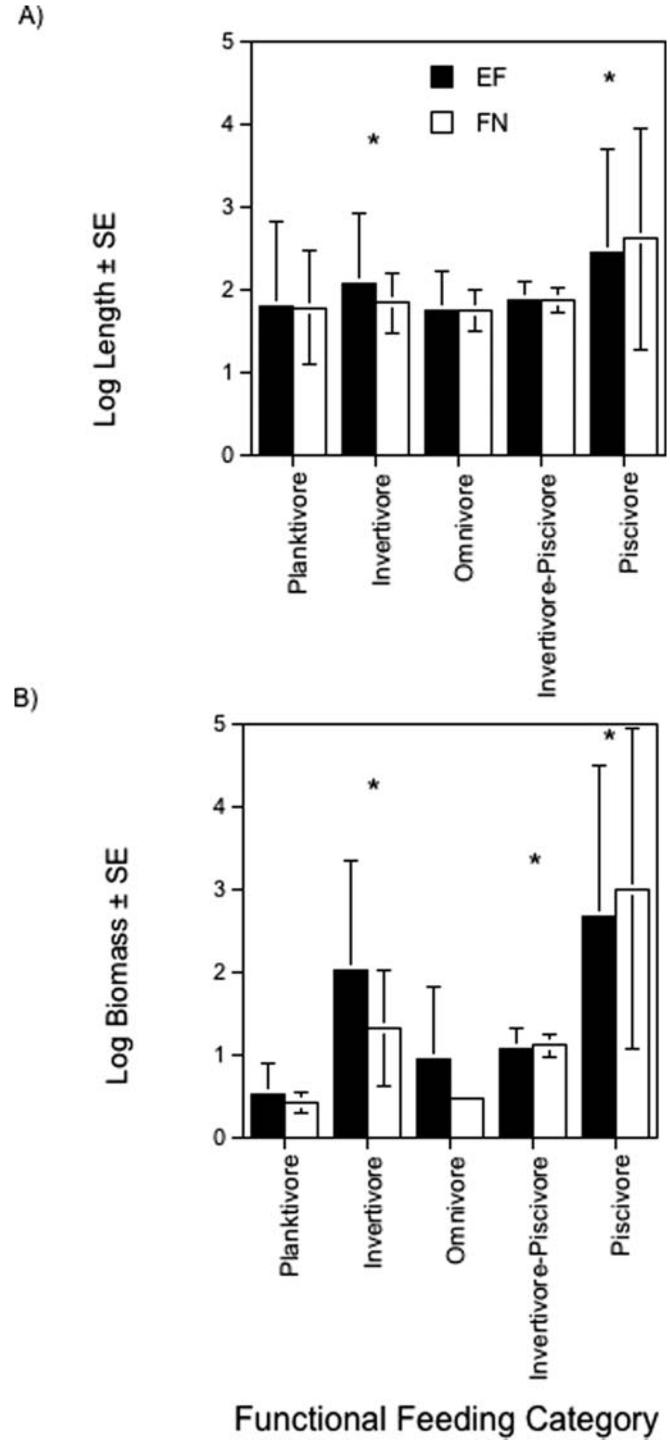

FIGURE 2. Mean (A) log length (±SE) and (B) log biomass (±SE) plotted for trophic guild captured by EF and FN for 26 Georgian Bay coastal wetlands. Asterisks indicate significant differences (where $P < 0.002$), as indicated by a post hoc Tukey–Kramer test.

DISCUSSION

There were clear distinctions between FN and EF sampling methods in Georgian Bay coastal wetlands, FN recovering more species, catch, biomass, and invertivore–piscivores. Fyke nets caught more species in the Ictaluridae, Centrarchidae, and Cyprinidae families, while EF caught more darters and white suckers. Although there was a disparity in the fish assemblage between the two gears, this did not significantly affect the WFI-PA and WFI-AB scores, which remained similar regardless of sampling method.

While many studies have been published that compare gear types, few studies focus specifically on the use of FNs and EF in coastal or littoral habitats, two methods of choice by environmental agencies within the Great Lakes community. Chow-Fraser et al. (2006) assessed these two methods in 11 wetlands in Lakes Erie and Ontario, and Ruetz et al. (2007) investigated the gear biases specific to these methods in three sites sampled over multiple seasons and years in Lake Muskegon, Michigan. Otherwise, the majority of studies focus on a variety of passive (trap nets, gill nets, hoop nets) versus active gears (seining, backpack electrofishing, EF, snorkeling; e.g., Pugh and Schramm 1998; Lapointe et al. 2006; Eros et al. 2009; Eggleton et al. 2010). In general, the species assemblage we collected using these two methods are in accordance with other studies in coastal habitats (Weaver et al. 1993; Brousseau et al. 2005; Chow-Fraser et al. 2006; Ruetz et al. 2007), and our observation that FNs caught the majority of the total catch is consistent with Chow-Fraser et al. (2006) and Ruetz et al. (2007).

Our results showed that EF caught fewer species compared with FNs, which is similar to that reported by Ruetz et al. (2007). Many studies have compared species richness between active and passive gears in a similar manner as we have in our study, with varying results. In flood plain systems, Eggleton et al. (2010) found that EF caught more species compared with other passive methods; however, mini-FNs were associated with the second-highest species richness and captured many unique taxa. Eros et al. (2009) reported the same trend in a comparison with gill nets in river systems. Other studies have reported no difference in species richness between EF and hoop nets (Pugh and Schramm 1998; Lapointe et al. 2006) or FNs, or a combination thereof (Chow-Fraser et al. 2006), although the latter study noted differences between methods when water quality was taken into consideration, where EF captured more fishes than FNs in higher quality sites sampled in Lakes Erie and Ontario. Since Georgian Bay wetlands generally exhibit high water quality (Cvetkovic and Chow-Fraser 2011), this was not a factor in our study.

One factor that may have affected species richness results is that EF effort was not taken into account in this study. Previous work has shown that transect length is correlated with species richness in streams (Reynolds et al. 2003; Kanno et al. 2009; Flotemersch et al. 2011), and it has been recommended that EF sampling continue until 30 times the expected species richness is caught (Flotemersch et al. 2011). This ensures that rare species will be included in the composition. Based on the results of a preliminary study conducted in one site in Georgian Bay, Kostuk (2006) found that total catch, biomass, and richness all increased linearly with total shock seconds expended. These findings suggest that the transect length and total shock seconds electrofished in the 26 sites we sampled may not have been sufficient in terms of effort, and that our catch could have been greater with increased effort. This is important to consider since the significant differences we observed in total catch, biomass, and richness between FN and EF may not have been observed if EF effort had been higher.

A higher proportion of invertivore–piscivores were captured by entrapment, and this is likely because FNs were more efficient at capturing the centrarchids, which included black crappie, bluegill, longear sunfish, pumpkinseed, and rock bass, all of which are carnivorous fishes. A majority of these species are small-bodied and are widely distributed and in high abundance in coastal habitats that serve as nursery grounds during the summer season (see Cvetkovic 2008; Cvetkovic et al. 2010). They also tend to swim in schools and undergo a diel migration, making them more susceptible to capture by passive gear such as FNs (Hubert 1989; Pugh and Schramm 1998; Ruetz et al. 2007). This schooling behavior is also exhibited by many cyprinids, such as blacknose shiner and bluntnose minnow, which we found were more likely to be captured by FNs, a trend consistent with the literature (Hubert 1989; Ruetz et al. 2007). Previous studies have shown that catfishes (family Ictaluridae) in general are more efficiently captured by passive techniques (Pugh and Schramm 1998; Lapointe et al. 2006; Ruetz et al. 2007), and this was supported by our results, where brown bullhead was caught in much higher numbers by FNs than by EF.

Although we found that larger fishes tended to be caught with EF compared with FN, the differences disappeared once site-specific variations were taken into account. There seems to be a disparity in the literature regarding size differences between entrapment and EF methods. Some studies have found that EF tends to catch larger specimens and larger length ranges of species compared with entrapment methods (e.g., fyke and hoop nets; Pugh and Schramm 1998; Sammons et al. 2002; Chow-Fraser et al. 2006; Dauwalter and Fisher 2007; Ruetz et al. 2007). Eros et al. (2009), however, showed that EF tended to catch both large and small specimens in greater proportion to gill nets in a European lake, and Pugh and Schramm (1998) showed that in a large river, hoop nets recovered larger length ranges of flathead catfish *Pylodictis olivaris* and channel catfish *Ictalurus punctatus* than did EF. One factor that may affect our results is that we electrofished during the day, whereas Ruetz et al. (2007) sampled at night, and thus we may have missed the diel migration of large-bodied fish species swimming inshore from deeper waters at night. In addition, we sampled only during the summer season, and it has been shown that certain species (e.g., centrarchids, cyprinids, and percids) are captured disproportionately over the spring, summer, and fall seasons (Bayley and Austen 2002; Brousseau et al. 2005). Sampling over multiple seasons and during both day and night would ensure a much more representative picture of the fish assemblage at these sites.

In this study, EF selected for larger invertivores. In order to determine what was driving this relationship, we systematically excluded invertivore taxa from the EF data and reran the analyses. When brown bullhead were excluded, the relationship was no longer significant, leading us to the conclusion that EF caught larger brown bullhead than FNs when all sites were combined, which was confirmed by a *t*-test ($P < 0.0001$, df $= 25$). As brown bullhead are known to occupy shallow waters (0–2 m) regardless of season (Brousseau et al. 2005), it is likely that adults were inhabiting vegetated areas during the time of sampling. Large-bodied brown bullhead were likely caught in greater proportion by EF because they were more affected than juveniles by the electric current, as a result of the increase in voltage gradient that occurs with size (Bohlin et al. 1989; Reynolds 1989).

Invertivores that were caught in significantly greater numbers by EF included white sucker, Johnny darter, and Iowa darter. White sucker was caught by EF in 9 of the 12 sites where this species was present, and this trend is consistent with the literature (Bayley and Austen 2002; Ruetz et al. 2007). Possible reasons that more darters were caught by EF include the fact that darters are morphologically adapted to prey on bottom-dwelling organisms (Paine et al. 1982; Holm et al. 2009) and therefore spend a majority of their time near the sediment; this attribute, coupled with their small size and the fact that they do not school, would preclude them from being trapped often by sedentary FNs (Bohlin et al. 1989; Hubert 1989), especially when wings are often not stationed completely on the bottom. Brousseau et al. (2005) mentioned in their report on EF protocols that EF methods are more efficient at capturing benthic species in low-turbidity sites, which is very indicative of sites found in Georgian Bay.

Fyke nets captured significantly larger piscivores than did EF. When we excluded bowfin from our analyses, which were caught in significantly higher numbers by FNs, there was no effect of gear on size of piscivores caught by both methods. Although Koch et al. (2009) stated that bowfin are usually caught with trap gear (supporting our findings), bowfin in the upper Mississippi River have been caught with both trap methods and EF methods (Mundahl et al. 1998; Weigel et al. 2006; Koch et al. 2009), and in Lakes Erie and Ontario, Chow-Fraser et al. (2006) did not find bowfin to be favored by either FNs or EF on a consistent basis. We suspect that one of the reasons electrofishing caught fewer and smaller bowfin relative to FNs is due to the sampling time (Brousseau et al. 2005). Large bowfin likely move into deeper and cooler waters during the day to feed, at which time electrofishing surveys were being conducted. They move back into shore in the evening, and it is this diel movement that enhances their chance of being trapped by FNs, which are set parallel to shore. The existence of rocky shoals in eastern and northern Georgian Bay, especially with the current low water levels, precluded nighttime surveys in this study.

We found that survey methods used in this study did not influence WFI scores in Georgian Bay when either presence/absence or abundance data were used. Similarly, WFI-PA scores generated by either FN data or EF data were not significantly different from WFI-PA scores generated by data when these gears were combined. This is empirical evidence that the WFI can infer analogous water quality conditions whether fyke netting or EF methods are used, even when different fish assemblages are collected. We attribute this to the fact that the WFI is based on the tolerance of groups of species to environmental degradation, and as long as there is no systematic bias that eliminates an entire indicator group from the sample, the WFI score associated with either gear will yield an accurate indication of water quality conditions in that wetland. Since Seilheimer and Chow-Fraser (2006, 2007) developed the WFI as an index of water quality impairment (resulting from human activities) rather than as an index of biotic integrity, further studies need to be conducted to determine if data collected by these two methods can be used interchangeably when calculating indices of biotic integrity, which may rely on characterization of the entire fish community.

Our findings have important management implications if conservation agencies are interested in using indices to track the quality of wetlands throughout the Great Lakes. One of the advantages of the WFI is that it requires only species information, and hence managers with access to archival data sets consisting of catch data are able to generate WFI scores for those time periods (see Seilheimer et al. 2011). This allows them to evaluate the historical habitat quality and accordingly compare past and present ecosystem health that is directly related to anthropogenic impact. Since our study only evaluated the differences in WFI scores between fish sampled by FN and EF gears, caution should be used if the aim of a study is to compare WFI scores generated with data that were collected using other sampling methods. In addition, this study did not take effort into account, and, as evidence shows that fish assemblage data and multimetric indices differ with degrees of effort (Dauwalter and Pert 2003: Reynolds et al. 2003; Maret et al. 2007), we recommend that future work be done to assess these effects with regards to gear methods and index scores. We did not expand our evaluation to include other indices, but we hope that future gear comparison studies will be conducted to crosswalk other indicators such as the IBI (e.g., Uzarski et al. 2005).

Many studies have concluded that more than one technique may be required to properly sample fish assemblages with a range of individuals from juveniles to adults (Drake and Pereira 2002; Van Snik Gray et al. 2005; Chow-Fraser et al. 2006). We agree with this response, particularly if the goal is to fully characterize the fish assemblage. While we found differences in total catch between FN and EF methods, these discrepancies may be minimized once effort and time of sampling is taken into account. If conservation agencies are using fish assemblages to infer water quality of coastal habitats, we recommend they compare WFI scores from sites that are based on data that were collected by the same gear whenever possible. This will reduce any unnecessary variation in ecological comparisons. If this is not possible, our results show that WFI scores calculated from FN or EF sampling methods in Georgian Bay may be compared in monitoring and ecological assessments, since fish assemblages in these sites will be representative of the water quality, which has been shown to be high throughout the region (Cvetkovic and Chow-Fraser 2011).

ACKNOWLEDGMENTS

This projected was funded by a grant to P.C-F. from the Ontario Ministry of Natural Resources as part of the Canada–Ontario Agreement. A special thanks to Titus Seilheimer for assistance with this project during all stages, including editing and comments. We thank the following graduate and undergraduates for assisting in the field: T. Seilheimer, M. Croft, L. O'Reilly, H. Pankhurst, T. Docherty, C. Robinson, and R. Razavi. Assistance was also provided from the Ontario Ministry of Natural Resources field staff in Espanola, particularly B. O'Farrell, and T. Kleinboeck and Parks Canada, particularly S. Parker and A. Promaine. We would also like to thank all of the reviewers who contributed comments to this manuscript.

REFERENCES

Albert, D. A., D. A. Wilcox, J. W. Ingram, and T. A. Thompson. 2005. Hydrogeomorphic classification for Great Lakes coastal wetlands. Journal of Great Lakes Research 31:129–146.

Angermeier, P. L., and J. R. Karr. 1986. Applying an index of biotic integrity based on stream-fish communities: considerations in sampling and interpretation. North American Journal of Fisheries Management 6:418–429.

Bayley, P. B., and D. J. Austen. 2002. Capture efficiency of a boat electrofisher. Transactions of the American Fisheries Society 131:435–451.

Benson, A. C., and T. M. Sutton. 2005. Evaluation of sampling techniques for age-0 lake sturgeon in a Lake Michigan tributary. North American Journal of Fisheries Management 25:1378–1385.

Bohlin, T., S. Hamrin, T. G. Heggberget, G. Rasmussen, and S. J. Saltveit. 1989. Electrofishing theory and practice with special emphasis on salmonids. Hydrobiologia 173:9–43.

Bonar, S. A., and W. A. Hubert. 2002. Standard sampling of inland fish: benefits, challenges, and a call for action. Fisheries 27(3):10–16.

Brazner, J. C., and E. W. Beals. 1997. Patterns in fish assemblages from coastal wetland and beach habitats in Green Bay, Lake Michigan: a multivariate analysis of abiotic and biotic forcing factors. Canadian Journal of Fisheries and Aquatic Sciences 54:1743–1761.

Brousseau, C. M., M. G. Clark, and R. G. Randall. 2005. Protocol for boat electrofishing in nearshore areas of the lower Great Lakes: transect and point survey methods for collecting fish and habitat data, 1988 to 2002. Canadian Manuscript Report of Fisheries and Aquatic Sciences 2702.

Butcher, A., D. Mayer, D. Smallwood, and M. Johnston. 2005. A comparison of the relative efficiency of ring, fyke, fence nets and beam trawling for estimating key estuarine fishery populations. Fisheries Research 73:311–321.

Cao, Y., C. P. Hawkins, and A. W. Storey. 2005. A method for measuring the comparability of different sampling methods used in biological surveys: implications for data integration and synthesis. Freshwater Biology 50:1105–1115.

Chow-Fraser, P., K. Kostuk, T. S. Seilheimer, M. Weimer, T. MacDougall, and T. Theÿsmeÿer. 2006. Effect of wetland quality on sampling bias associated with two fish survey methods for coastal wetlands of the lower Great Lakes. Pages 233–256 *in* T. P. Simon and P. M. Stewart, editors. Coastal wetlands of the Laurentian Great Lakes: health, habitat and indicators. Indiana Biological Survey, Bloomington.

Clavero, M., F. Blanco-Garrido, and J. Prenda. 2006. Monitoring small fish populations in streams: a comparison of four passive methods. Fisheries Research 78:243–251.

Croft, M. V., and P. Chow-Fraser. 2007. Use and development of the wetland macrophyte index to detect water quality impairment in fish habitat of Great Lakes coastal marshes. Journal of Great Lakes Research 33:172–197.

Cvetkovic, M. 2008. Factors affecting macrophyte and fish distribution in coastal wetlands of Georgian Bay. Master's thesis. McMaster University, Hamilton, Ontario.

Cvetkovic, M., and P. Chow-Fraser. 2011. Use of ecological indicators to assess the quality of Great Lakes coastal wetlands. Ecological Indicators 11:1609–1622.

Cvetkovic, M., A. Wei, and P. Chow-Fraser. 2010. Relative importance of macrophyte community versus water quality variables for predicting fish assemblages in coastal wetlands of the Laurentian Great Lakes. Journal of Great Lakes Research 36:64–73.

Dauwalter, D. C., and W. L. Fisher. 2007. Electrofishing capture probability of smallmouth bass in streams. North American Journal of Fisheries Management 27:162–171.

Dauwalter, D. C., and E. J. Pert. 2003. Effect of electrofishing effort on an index of biotic integrity. North American Journal of Fisheries Management 23:1247–1252.

DeCatanzaro, R., and P. Chow-Fraser. 2011. Effects of landscape variables and season on reference water chemistry of coastal marshes in eastern Georgian Bay. Canadian Journal of Fisheries and Aquatic Sciences 68:1009–1023.

DeCatanzaro, R., M. Cvetkovic, and P. Chow-Fraser. 2009. The relative importance of road density and physical watershed features in determining coastal marsh water quality in Georgian Bay. Environmental Management 44:456–467.

Drake, M. T., and D. L. Pereira. 2002. Development of a fish-based index of biotic integrity for small inland lakes in central Minnesota. North American Journal of Fisheries Management 22:1105–1123.

Eggleton, M. A., J. R. Jackson, and B. J. Lubinski. 2010. Comparison of gears for sampling littoral-zone fishes in floodplain lakes of the lower White River, Arkansas. North American Journal of Fisheries Management 30:928–939.

Emery, E. B., T. P. Simon, F. H. McCormick, P. L. Angermeier, J. E. Deshon, C. O. Yoder, R. E. Sanders, W. D. Pearson, G. D. Hickman, R. J. Reash, and J. A. Thomas. 2003. Development of a multimetric index for assessing the biological condition of the Ohio River. Transactions of the American Fisheries Society 132:791–808.

Eros, T., A. Specziar, and P. Biro. 2009. Assessing fish assemblages in reed habitats of a large shallow lake: a comparison between gillnetting and electric fishing. Fisheries Research 96:70–76.

Flotemersch, J. E., J. B. Stribling, R. M. Hughes, L. Reynolds, M. J. Paul, and C. Wolter. 2011. Site length for biological assessment of boatable rivers. River Research and Applications 27:520–535.

Gammon, J. R., and T. P. Simon. 2000. Variation in a Great River index of biotic integrity over a 20-year period. Hydrobiologia 422/423:291–304.

Hardie, S. A., L. A. Barmuta, and R. W. G. White. 2006. Comparison of day and night fyke netting, electrofishing and snorkelling for monitoring a population of the threatened golden galaxias (Galaxias auratus). Hydrobiologia 560:145–158.

Hayes, M. L. 1989. Active fish capture methods. Pages 123–145 *in* L. A. Nielsen and D. L. Johnson, editors. Fisheries techniques. American Fisheries Society, Bethesda, Maryland.

Holm, E., N. E. Mandrak, and M. E. Burridge. 2009. The ROM field guide to freshwater fishes of Ontario. Royal Ontario Museum, Toronto.

Hubert, W. A. 1989. Passive capture techniques. Pages 95–122 *in* L. A. Nielsen and D. L. Johnson, editors. Fisheries techniques. American Fisheries Society, Bethesda, Maryland.

Hughes, R. M., and A. T. Herlihy. 2007. Electrofishing distance needed to estimate consistent index of biotic integrity (IBI) scores in raftable Oregon rivers. Transactions of the American Fisheries Society 136:135–141.

Jude, D. J., and J. Pappas. 1992. Fish utilization of Great Lakes coastal wetlands. Journal of Great Lakes Research 18:651–672.

Kanno, V., J. C. Vokoun, D. C. Dauwalter, R. M. Hughes, A. T. Herlihy, T. R. Maret, and T. M. Patton. 2009. Influence of rare species on electrofishing distance when estimating species richness of stream and river reaches. Transactions of the American Fisheries Society 138:1240–1251.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6(6):21–27.

Koch, J. D., M. C. Quist, K. A. Hansen, and G. A. Jones. 2009. Population dynamics and potential management of bowfin (Amia calva) in the upper Mississippi River. Journal of Applied Ichthyology 25:545–550.

Kostuk, K. 2006. Use of zoobenthos to indicate human-induced disturbance and degree of exposure in Great Lakes coastal wetlands. Master's thesis. McMaster University, Hamilton, Ontario.

Lapointe, N. W. R., L. D. Corkum, and N. E. Mandrak. 2006. A comparison of methods for sampling fish diversity in shallow offshore waters of large rivers. North American Journal of Fisheries Management 26:503–513.

Mangan, M. T., M. L. Brown, T. R. St. Sauver. 2005. A comparison of passive gears for selective yellow perch harvest. North American Journal of Fisheries Management 25:1067–1072.

Maret, T. R., D. S. Ott, and A. T. Herlihy. 2007. Electrofishing effort required to estimate biotic condition in southern Idaho rivers. North American Journal of Fisheries Management 27:1041–1052.

McGeoch, M. A., and K. J. Gaston. 2002. Occupancy frequency distributions: patterns, artifact, and mechanisms. Biological Reviews of the Cambridge Philosophical Society 77:311–331.

Midwood, J. D., and P. Chow-Fraser. 2010. Mapping floating and emergent aquatic vegetation in coastal wetlands of eastern Georgian Bay, Lake Huron, Canada. Wetlands 30:1141–1152.

Midwood, J. D., D. Rokitnicki-Wojcik, and P. Chow-Fraser. In press. Development of an inventory of coastal wetlands for eastern Georgian Bay, Lake Huron. ISRN Ecology [online serial].

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

Mundahl, N. D., C. Melnytschuk, D. K. Spielman, J. P. Harkins, K. Funk, and A. M. Bilicki. 1998. Effectiveness of bowfin as a predator on bluegill in a vegetated lake. North American Journal of Fisheries Management 18:286–294.

Onorato, D. P., R. A. Angus, and K. R. Marion. 1998. Comparison of a small-mesh seine and a backpack electroshocker for evaluating fish populations in a north-central Alabama stream. North American Journal of Fisheries Management 18:361–373.

Paine, M. D., J. J. Dodson, and G. Power. 1982. Habitat and food resource partitioning among four species of darters (Percida: Etheostoma) in a southern Ontario stream. Canadian Journal of Zoology 60:1635–1641.

Paukert, C. P. 2004. Comparison of electrofishing and trammel netting variability for sampling native fishes. Journal of Fish Biology 65:1643–1652.

Poos, M. S., N. E. Mandrak, and R. L. McLaughlin. 2007. The effectiveness of two common sampling methods for assessing imperiled freshwater fishes. Journal of Fish Biology 70:691–708.

Pugh, L. L., and H. L. Schramm Jr. 1998. Comparison of electrofishing and hoopnetting in lotic habitats of the lower Mississippi River. North American Journal of Fisheries Management 18:649–656.

Reynolds, J. B. 1989. Electrofishing. Pages 147–163 *in* L. A. Nielsen and D. L. Johnson, editors. Fisheries techniques. American Fisheries Society, Bethesda, Maryland.

Reynolds, L., A. T. Herlihy, P. R. Kaufmann, S. V. Gregory, and R. M. Hughes. 2003. Electrofishing effort requirements for assessing species richness and biotic integrity in western Oregon streams. North American Journal of Fisheries Management 23:450–461.

Ruetz, C. R., D. G. Uzarski, D. M. Krueger, and E. S. Rutherford. 2007. Sampling a littoral fish assemblage: comparison of small-mesh fyke netting and boat electrofishing. North American Journal of Fisheries Management 27:825–831.

Sammons, S. M., D. A. Isermann, and P. W. Bettolli. 2002. Variation in population characteristics and gear selection between black and white crappies in Tennessee reservoirs: potential effects on management decisions. North American Journal of Fisheries Management 22:863–869.

Schneider, J. C., P. W. Laarman, and H. Gowing. 2000. Length-weight relationships. Chapter 17 *in* J. C. Schneider, editor. Manual of fisheries survey methods II: with periodic updates. Michigan Department of Natural Resources, Fisheries Special Report 25, Ann Arbor.

Schwanke, C. J., and W. A. Hubert. 2004. Evaluation of three gears for sampling spawning populations of rainbow trout in a large Alaskan river. North American Journal of Fisheries Management 24:1078–1082.

Scott, W. B., and E. J. Crossman. 1973. Freshwater fishes of Canada. Reprint 1998. Galt House Publishing, Oakville, Ontario.

Seilheimer, T. S., and P. Chow-Fraser. 2006. Development and use of the wetland fish index to assess the quality of coastal wetlands in the Laurentian Great Lakes. Canadian Journal of Fisheries and Aquatic Sciences 63:354–366.

Seilheimer, T. S., and P. Chow-Fraser. 2007. Application of the wetland fish index to northern Great Lakes marshes with emphasis on Georgian Bay coastal wetlands. Journal of Great Lakes Research 33:154–171.

Seilheimer, T. S., T. P. Mahoney, and P. Chow-Fraser. 2009. Comparative study of ecological indices for assessing human-induced disturbance in coastal wetlands of the Laurentian Great Lakes. Ecological Indicators 9:81–91.

Seilheimer, T. S., A. Wei, and P. Chow-Fraser. 2011. Changes in fish communities of Lake Ontario coastal wetlands before and after remedial action plans. ISRN Ecology [online serial] Article 820529. DOI:10.5402/2011/820529.

Seilheimer, T. S., A. Wei, P. Chow-Fraser, and N. Eyles. 2007. Impact of urbanization on the water quality, fish habitat, and fish community of a Lake Ontario marsh, Frenchman's Bay. Urban Ecosystems 10:299–319.

Simon, T. P., and E. B. Emery. 1995. Modification and assessment of an index of biotic integrity to quantify water resource quality in great rivers. Regulated Rivers: Research and Management 11:283–298.

Simon, T. P., and J. Lyons. 1995. Application of the index of biotic integrity to evaluate water resource integrity in freshwater ecosystems. Pages 245–262 *in* W. S. Davis and T. P. Simon, editors. Biological assessment and criteria: tools for water resource planning and decision making. CRC Press, Boca Raton, Florida.

Uzarski, D. G., T. M. Burton, M. J. Cooper, J. W. Ingram, and S. T. A. Timmermans. 2005. Fish habitat use within and across wetland classes in coastal wetlands of the five Great Lakes: development of a fish-based index of biotic integrity. Journal of Great Lakes Research 31:171–187.

van Snik Gray, E., R. M. Ross, and R. M. Bennett. 2005. Bioassessment of fish communities of the upper Delaware River. Northeastern Naturalist 12:203–216.

Weaver, M. J., J. J. Magnuson, and M. K. Clayton. 1993. Analyses for differentiating littoral fish assemblages with catch data from multiple sampling gears. Transactions of the American Fisheries Society 122:1111–1119.

Wei, A., P. Chow-Fraser, and D. Albert. 2004. Influence of shoreline features on fish distribution in the Laurentian Great Lakes. Canadian Journal of Fisheries and Aquatic Sciences 61:1113–1123.

Weigel, B. M., J. Lyons, and P. W. Rasmussen. 2006. Fish assemblages and biotic integrity of a highly modified floodplain river, the upper Mississippi, and a large, relatively unimpacted tributary, the lower Wisconsin. River Research and Applications 22:923–936.

Weiler, R. R. 1988. Chemical limnology of Georgian Bay and the North Channel between 1974 and 1980. Hydrobiologia 163:77–83.