

A COMPARISON OF ELECTROFISHING AND VISUAL SURVEYING METHODS FOR ESTIMATING FISH COMMUNITY STRUCTURE IN TEMPERATE RIVERS

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ABSTRACT

Studies attempting to describe fish community structure in shallow riverine environments typically rely on electrofishing and/or visual (snorkelling) surveys, but few have addressed the relative efficiencies of these two methods at estimating fish density and biomass across wide ranges of geography, taxonomy and life history stages. Multiple paired electrofishing and visual surveys were conducted in 18 temperate Canadian rivers in order to obtain community-wide density and biomass estimates from both methods. Partial canonical multivariate analyses were applied to the paired fish community matrices comparing the results of both surveying methods at the taxonomic levels of family, genus and species, as well as size classes within families and species, to assess the particular effectiveness of each sampling method. Although electrofishing estimates of family and species richness were generally greater, snorkelling surveys tended to generate higher density and biomass estimates for different size classes of many salmonid and cyprinid species. Moreover, mean river biomass estimates derived from visual surveying matched those obtained from our best mean river biomass estimates arising from the two methods combined. This study provides empirical evidence that electrofishing and visual survey methods generate different types of information when assessing fish community structure at the family level or by size classes. Our results provide ample background information for determining the most accurate sampling method for a particular fish community assemblage, which is fundamental to fisheries management and research. Copyright © 2014 John Wiley & Sons, Ltd.

KEY WORDS: sampling methods; snorkelling; fish density; biomass; species richness

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INTRODUCTION

Determining how best to quantify fish populations, be it with measures of species richness, abundance or production, is the first requirement and often the most difficult step to developing scientifically sound monitoring and management programmes or models of population dynamics (Meador *et al.*, 2003; Peterson *et al.*, 2004; Korman *et al.*, 2009). Various methods are used to sample fish in rivers, but all ultimately provide biased representations of the true species assemblage and size composition. For example, by altering the sampling mode (e.g. passive versus active capture) or targeting species that differ in their ecology and life history (e.g. size class and behaviour) or preferred habitats (e.g. sandy versus boulder substrate, moderate versus slow water velocities and low versus high water turbidity), relative sampling efficiency will change and generate conflicting estimates of fish abundance (Miranda and Shramm, 2008). It is therefore difficult to decide what sampling method or combination thereof should be used

to conduct censuses of riverine fish populations. With anthropogenic changes occurring in freshwater systems (Vörösmarty *et al.*, 2010), it is essential to the maintenance and/or improvement of fish populations that efforts are targeted at refining our knowledge of the advantages and limitations of different sampling methods over a wide range of river types (Korman *et al.*, 2009).

Many ecological studies have used electrofishing and snorkelling surveys for estimating population density, species richness, growth or production of freshwater fishes in small and shallow riverine areas (i.e. <2-m depth; Griffith, 1981; Hankin and Reeves, 1988; Mullner *et al.*, 1998; Joyce and Hubert, 2003). Electrofishing is particularly useful for the precise identification and measurement of individuals within a population, including cryptic species that may not be readily detected visually (Willis, 2001). However, this sampling method may have low capture efficiency, particularly for small (Reynolds, 1996) and shoaling fish like some cyprinids and catostomids (Kimmel and Argent, 2006), potentially leading to underestimates of overall community densities. By contrast, visual surveys of freshwater fish populations appear to be fairly reliable when compared with electrofishing provided that certain conditions (i.e. few

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macrophytes or emerging plants, homogeneous substrate and high water visibility) occur at the time of sampling (Helfman, 1983). However, snorkelling surveys also have several shortcomings, including problems with accurate species identification, counting and estimating the size of fishes (Brock, 1982). These inherent sampling biases have been quantified for some species, notably salmonids, but to our knowledge, no studies have yet attempted to assess sampling differences for entire fish communities across a wide range of temperate rivers.

Cunjak *et al.* (1988) compared the abundance of juvenile Atlantic salmon (*Salmo salar*) as estimated by electrofishing and snorkelling in three rivers in eastern Canada and revealed that snorkelling counts consistently underestimated density, especially for younger and/or smaller fish that frequented shallow stream margins where underwater detection and enumeration was difficult. Visual surveying has also been found to underestimate several other salmonids [1+ coho salmon (*Oncorhynchus kisutch*), steelhead trout (*Oncorhynchus mykiss*), bull (*Salvelinus malma*), cutthroat (*Oncorhynchus clarkii*), rainbow (*Oncorhynchus mykiss*), brown (*Salmo trutta*) and brook trout (*Salvelinus fontinalis*)] relative to electrofishing, although estimates from both methods appear to be generally well correlated ($r > 0.90$; Hankin and Reeves, 1988; Wildman and Neumann, 2003; Thurow *et al.*, 2006). Despite the differences in abundance estimates, length–frequency distributions obtained by either method have been fairly similar (Wildman and Neumann, 2003). In addition to demonstrating sampling differences among various salmonid species across selected rivers, the studies cited above, like many others, generally compare fish abundance and/or diversity counts, rather than deriving biomass estimates between surveying methods.

The apparent consensus in the literature is that sampling biases are influenced by a number of factors, from the composition of the fish community (species, individual size and behaviour) to environmental descriptors that characterize the sampling site and time (Reynolds, 1996; Meador *et al.*, 2003; Korman *et al.*, 2009). Most authors suggested the need for more comprehensive evaluations of the relative performance of these sampling methods for a wider range of species, river and habitat type. In an attempt to address these knowledge gaps, this study aims to (i) evaluate the respective performances of electrofishing and snorkelling methods for estimating relative fish community richness, density and biomass, in selected systems; and (2) establish background information guiding the choice of the most suitable sampling method for the inventory of fish populations to inform fisheries management and research. This study is intended to provide information that will facilitate standardized comparisons among studies employing similar methodologies.

MATERIALS AND METHODS

Study sites

Eighteen small to intermediate wadeable rivers (wetted river width ranging from 17 to 116 m) comprising 4 in Alberta, 5 in Ontario, 6 in Québec and 3 in New Brunswick were selected based on sampling feasibility and river accessibility (Figure 1). Between 25 and 50 relatively homogeneous sampling sites, measuring approximately 300 m² ($5.08 \pm 0.44 \text{ m} \times 59.19 \pm 3.59 \text{ m}$, mean \pm SD; width \times length), were surveyed per river, for a total of 663 sites. Sampling sites broadly represented a uniform habitat type, which was visually assessed and categorized into run, riffle or pool according to Jowett (1993). Sampling sites were spread along 10–15-km river segments, with a 60-m minimum buffer between sites to ensure that fish would not migrate between sites as a result of our sampling efforts, thus avoiding pseudoreplication in the data. Across river widths, sites were placed sequentially along the left shore, followed by the middle stretch, determined by taking the mid-point of the wetted river width and then along the right shore, with each completed sequence looping back to the left shore. The first site of the sequence was placed at random, either on the left or right shores or the middle stretch.

Sampling protocol

Field work was carried out during summer months (late June, July, and August) in 2011 and 2012. Paired single-pass electrofishing and snorkelling surveys were conducted at each site, for a total of 1326 surveys (two surveys per site). Paired surveys were conducted in random order at the same time period between 08:30 and 18:00 on consecutive days, with a minimum 24-h recovery period intended to allow fish to re-establish themselves after a sampling event. To increase the accuracy of species identification and length estimation under water for electrofishing and snorkelling methods, surveyors were trained continuously for a month prior to data collection.

Electrofishing surveys were administered in accordance to the Ontario Ministry of Natural Resources (Jones, 2011) policy standards using a Smith-Root LR-24 backpack electrofishing unit (Smith-Root®, Vancouver, WA, USA). An electrofishing operator, flanked by two assistants collected stunned fish into Smith-Root® trapezoid dipnets (4" wide back, 14" wide \times 17" long and 8" deep and of 1/4" mesh size), while moving upstream at a rate of 3 s m⁻² in a zigzag fashion. Electrofishing parameters such as voltage, frequency for sampling whole fish communities and shocking seconds were adjusted in response to water conductivity to produce a constant average power of 200 W and of 60 Hz and over a mean time of

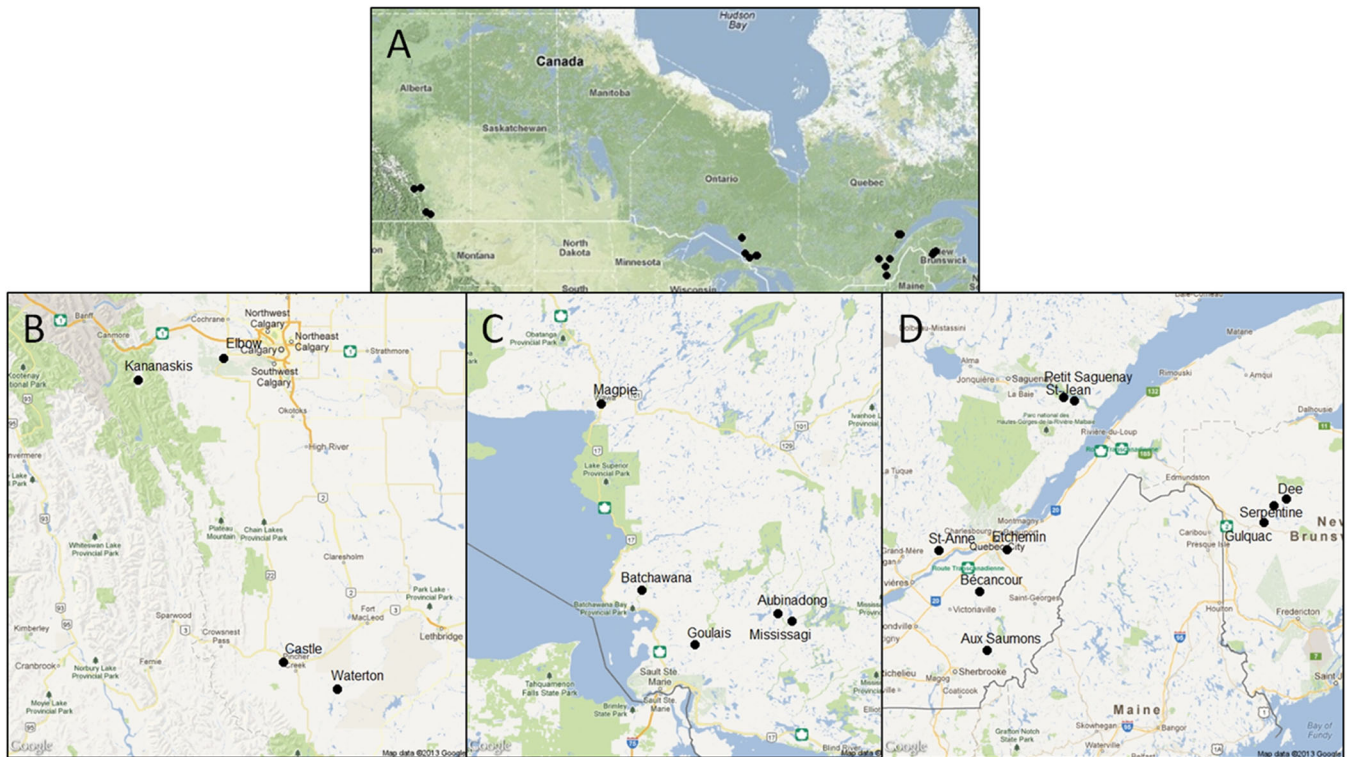


Figure 1. Map of Canada displaying the 18 rivers studied, grouped by province (A). Alberta (B): Castle, Waterton, Elbow and Kananaskis Rivers; Ontario (C): Magpie, Goulais, Batchawana, Aubinadong and Mississagi Rivers; Québec (D): Sainte-Anne, Etchemin, Bécancour, aux Saumons, Saint-Jean and Petit Saguenay Rivers; New Brunswick (D): Dee, Gulquac and Serpentine Rivers

913.26 ± 74.61 s. Setting constant parameters ensured that sampling effort was standardized across electrofishing sampling events as well as between electrofishing teams. Captured fish were identified to species, and their size (total body length, ±0.1 cm) and mass (wet blotted weight, ±0.1 g) were measured after completion of electrofishing at a given site. Fish were then allowed to recover from handling stress and released back to their place of capture. Visual surveys were conducted using two divers swimming in a slow upstream fashion (approximately 6 s m⁻²), covering an average surface area of 299 ± 7.49 m² (mean ± SD), over an average of 1920 ± 314.4 s. Fish species identification and total length were recorded *in situ* in 5-cm class increments.

Physical variables

At the end of individual visual and electrofishing surveys, the physical variables of each site were estimated: (i) flow velocity (m s⁻¹), taken at 40% of the total water depth; (ii) depth was measured with a Marsh-McBirney Flo-Mate 2000 flow meter and wading rod (Hach Company, Loveland, CO, USA); (iii) water temperature; and (iv) conductivity were measured with a YSI Model 30 handheld conductivity meter (YSI inc., Yellow Springs, OH,

USA); and (iv) the proportion cloud cover was assessed visually. These physical descriptors may differ between paired electrofishing and visual surveys because the surveys were conducted on different days. Water temperature and cloud cover were measured once by site and survey, whereas mean flow velocities and depths were determined 10 times by site and survey (randomly dispersed throughout each site), all of which were retained for analysis for the purpose of removing their confounding effect on the observed differences between surveying methods. Other site variables were measured for each site but were not included in this study because of the following: (i) habitat-specific variables (e.g. vegetation and substrate cover) did not differ within our 24-h sampling period; and (ii) our study aims at comparing fish community structure between sampling methods across a range of rivers rather than assessing which environmental variables best predict overall fish communities.

Fish matrices

Species richness was determined by counting the number of different species represented in a site. Density estimates were taken as the total number of each species observed for each site of 300 m² multiplied by 100 (i.e. the number

of a given species 100 m^{-2}), while species biomass estimates were calculated by adding individually measured fish masses for species at any given site multiplied by 100 (i.e. the total mass in grams for a given species/ 100 m^2). Mass–length relationships derived from electrofishing data were calculated for each species per river and used to estimate mass from length for all fish recorded during visual surveys (Le Cren, 1951). The sampling design was not suitable for ascertaining absolute species density and biomass counts for each site sampled and both surveying methods; consequently, all values presented herein represent relative estimates per site. Additionally, taxon occurrence (N) refers to the number of sites where a given family, species or species-by-size-class was observed with either sampling method, removing the cases of non-occurrences. Total densities and biomasses for a given species were categorized by 5-cm size classes (i.e. 1, 0–5 cm; 2, 5–10 cm; 3, 10–15 cm; 4, 15–20 cm and 5, 20–25 cm), thus creating species-by-size-class density and biomass matrices, for both surveying methods. Densities and biomasses for each species were therefore merged into 5-cm size class increments to provide the information required for assessing sampling differences, from small to larger fish, interspecifically and intraspecifically.

Statistical analyses

Wilcoxon matched-pairs signed-rank tests were computed on tabulated species richness to test whether these scores differed significantly between sampling methods across sites. The richness scores were calculated using all 55 species observed in the 663 sites.

Fish density and biomass data from paired surveys were compiled for each of 15 families, the 27 most prevalent species, and 101 species-by-size-class combinations across all sites, resulting in 12 fish community matrices (e.g. three taxonomic levels for densities and three for biomasses; six matrices per surveying method). The fish data were Hellinger transformed, which expresses the density or biomass data as relative values per site and takes their square root (Legendre and Gallagher, 2001). This data transformation meant that subsequent analyses were not affected by double zeros, and a more meaningful analysis is obtained because ‘no ecological conclusion can be drawn from the simultaneous absence of a species at two sites’ or in our case at one site using two surveying methods (Legendre and Legendre, 2012, section 7.2.2). Forward selection of physical variables (standardized survey means for flow velocity, water depth, water temperature and cloud cover) was conducted using redundancy analysis (RDA) to select the best explanatory variables for the model describing each fish

community matrix (*packfor* package in R, Blanchet *et al.*, 2008; Dray *et al.*, 2011).

Partial canonical multivariate analyses of variance, conducted for related samples using partial RDAs (pRDAs: Legendre and Legendre, 2012, section 11.1.10), were performed on each fish community matrix (e.g. family biomass estimates for electrofishing versus visual surveying) to assess the effect of sampling method while controlling for the effects of selected physical variables and paired surveys per site (i.e. the fact that each site had been sampled twice). For each significant multivariate pRDA, univariate family-specific, species-specific and species-by-size-class-specific pRDA were conducted and corrected for multiple testing using the Hochberg correction (Hochberg, 1988). Univariate pRDA enabled the identification of which densities or biomasses significantly differed between sampling methods for the various taxa. Cumulative fit R^2 was also calculated using the *goodness* function from the multivariate pRDA for each taxon: families, species and species-by-size-classes (*vegan* package, Oksanen *et al.*, 2011; Legendre and Legendre, 2012, section 9.3). Taxa with cumulative fit R^2 values above community average were considered as being well explained by the multivariate model and therefore as being differently sampled by the two sampling methods. Taxa with cumulative fit values above community average are displayed as the proportion of the cumulative fit value of a particular taxon over the sum total of all cumulative fit values for all taxa. Odds ratios were also calculated for significantly different density and biomass estimates from each surveying method. All tests of significance for pRDAs were conducted using permutation tests involving 9999 permutations of the residuals under the reduced model using the function *anova.cca* in the *vegan* package (Oksanen *et al.*, 2011).

Our ‘best species biomass matrix’ was derived by selecting the largest estimate obtained when comparing electrofishing and snorkelling surveys at each site; these estimates were then averaged by river (number of rivers = 18). Similarly, total fish biomasses averaged by river were calculated for electrofishing and visual surveys respectively, and a linear relationship between each of our sampling methods and our best possible outcome was drawn. Model II regressions were computed to compare our ‘best mean river biomass estimates’ to those obtained from electrofishing and visual surveys (Legendre and Legendre, 2012, section 10.3.2). We used function *lmodel2* to compute model II simple linear regressions using the major axis method, as well as parametric 95% confidence intervals for the slope and intercept parameters, which were used to determine whether model slopes and intercepts differed significantly from a 1:1 slope and a 0 intercept respectively (Legendre, 2011).

Table I. Environmental attributes and mean physical variables by river and sampling method

River (no. of sites)	Latitude	Longitude	Mean wetted width (m)	Mean adjusted conductivity (µS)	Mean flow velocities (m/s)		Mean water depth (cm)		Mean water temperature (°C)		Mean cloud cover (% coverage)	
					E	V	E	V	E	V	E	V
Castle (42)	49.53	-114.04	46.39	237.71	0.56	0.52	40.51	38.58	14.62	17.39	32.00	27.75
Waterton (47)	49.40	-113.63	41.29	188.02	0.37	0.37	40.71	40.31	13.47	15.79	29.22	19.84
Elbow (46)	51.00	-114.50	35.30	377.50	0.59	0.57	36.91	37.21	9.08	11.22	33.98	24.51
Kananaskis (47)	50.90	-115.15	25.47	333.08	0.38	0.37	33.32	31.91	8.72	10.53	27.50	25.33
Magpie (30)	47.99	-84.79	45.50	118.75	0.31	0.33	41.17	39.81	18.84	18.47	57.73	51.30
Batchawana (43)	47.03	-84.48	50.34	145.25	0.37	0.37	45.77	45.43	21.02	21.28	53.93	49.64
Goulais (30)	46.75	-84.07	35.80	65.40	0.12	0.12	41.68	40.48	21.71	22.50	54.84	50.63
Aubinadong (40)	46.91	-83.44	40.80	208.57	0.34	0.33	46.48	47.83	22.50	21.69	48.10	53.50
Mississagi (36)	46.87	-83.33	89.89	170.52	0.22	0.22	40.99	43.55	21.07	20.96	44.24	43.68
Sainte-Anne (30)	46.67	-72.12	116.12	55.41	0.23	0.26	40.13	39.94	22.47	23.35	60.83	51.00
Saint-Jean (49)	48.23	-70.22	31.08	152.24	0.54	0.57	44.88	48.24	18.48	18.51	56.53	43.57
Petit Saguenay (25)	48.20	-70.06	33.54	129.74	0.43	0.43	49.86	50.81	19.99	19.95	33.13	65.21
Etchemin (40)	46.68	-71.08	81.51	139.86	0.29	0.26	37.68	35.77	23.67	23.26	35.75	18.75
Bécancour (50)	46.24	-71.50	37.24	156.00	0.27	0.26	48.06	45.54	22.60	21.78	44.24	38.10
aux Saumons (50)	45.62	-71.39	54.95	91.25	0.25	0.24	33.11	34.34	21.40	21.39	53.80	54.10
Dee (30)	47.13	-67.00	14.33	25.93	0.50	0.45	33.08	31.80	18.89	19.89	51.67	44.83
Serpentine (30)	47.20	-66.82	20.43	26.60	0.37	0.37	28.36	28.03	17.02	18.10	38.00	24.52
Gulquac (30)	46.96	-67.16	16.80	44.66	0.43	0.36	32.40	33.22	16.41	17.80	35.17	50.67

Rivers are grouped by province, and numbers in brackets refer to the number of sites sampled in each river. Adjusted conductivity is corrected for water temperature. E, electrofishing; V, visual

RESULTS

On average, our rivers were 45.38 m wide, of low water velocity (0.36 m s^{-1}), shallow depth (39.70 cm), warm water temperature (18.5°C), moderate cloud cover (42%) and elevated temperature-adjusted conductivity ($148.14 \mu\text{S m}^{-1}$; Table 1) during sampling efforts. None of the physical variables met the assumptions of normality before or after transformation; therefore, careful consideration of each of these variables and their impact on predicting the distribution of fishes for each of the models was conducted prior to carrying out the analyses described in the next paragraph.

Surveying differences for estimating community structure

Species richness differed significantly between sampling methods (Wilcoxon signed-rank test: $p < 0.001$), with average species richness for electrofishing sampling greater (mean = 3.67, range = 0–12 species per site) than for visual surveying (mean = 3.37, range = 0–12 species per site). Of the 55 species recorded, 48 were observed using snorkelling, while all but *H. regius* were counted during electrofishing. Species nearly or completely absent from visual surveys included *Anguilla rostrata*, *Ameiurus nebulosus*, *Umbra limi*, *Esox masquinongy*, *Ichthyomyzon fossor*, *Petromyzon marinus*, *Moxostoma macrolepidotum*, *Lota lota* and *Lethenteron appendix*.

All physical variables (water velocity, depth, water temperature and cloud cover; Table 1) were deemed significant (in all tests, $p \leq 0.005$) and retained by forward selection for all 12 fish community matrices. There was a significant difference in family densities and biomasses between electrofishing and visual surveys (multivariate pRDA tests, $p < 0.005$). Density and biomass estimates for

Percopsidae, Cottidae, Umbridae, Lottidae, Anguillidae and Petromyzontidae were significantly greater for electrofishing surveys, while the reverse was true for Salmonidae (univariate pRDA tests with Hochberg correction, $p \leq 0.05$, Figure 2). Gasterosteid snorkelling density and biomass were between 3.0 and 6.5 times greater than electrofishing estimates (univariate pRDA tests with Hochberg correction, $p = 0.06$ and $p = 0.058$ respectively). Cumulative fit values for density or biomass estimates echoed these results, with Cottidae (29.25%), Umbridae (7.09%), Lottidae (13.01%), Anguillidae (8.80%) and Petromyzontidae (20.25%), each contributing more than the community average to the global model (cumulative fit values for biomass estimates displayed and illustrated by shaded bars in Figure 2). Density and biomass differences occurring for these families therefore explained the greatest cumulative proportion of variation between the two sampling methods.

For 16 of the 27 more prevalent species listed in Table 2, significant differences for species densities and/or biomasses were found between electrofishing and visual surveys (multivariate pRDA tests, $p < 0.005$, and univariate pRDA with Hochberg correction, $p < 0.05$, Figure 3). Salmonids *S. salar*, *Prosopium williamsoni*, *Oncorhynchus* spp. and several cyprinid shoaling species, namely *Luxilus cornutus*, *Semotilus corporalis*, *Rhinichthys atratulus* and *Exoglossum maxillingua*, were found to have over 1.5 times greater visual density estimates for salmonids or a 2 to 10-fold increase in snorkelling density estimates for cyprinids. Electrofishing density estimates were at least 1.5 times greater than visual estimates for cryptic species like *L. lota*, *Etheostoma* spp., *Cottus* spp. and *Percopsis omiscomaycus* in addition to *S. trutta*, *Ambloplites rupestris* and *Rhinichthys cataractae*. Of these species, *P. williamsoni*

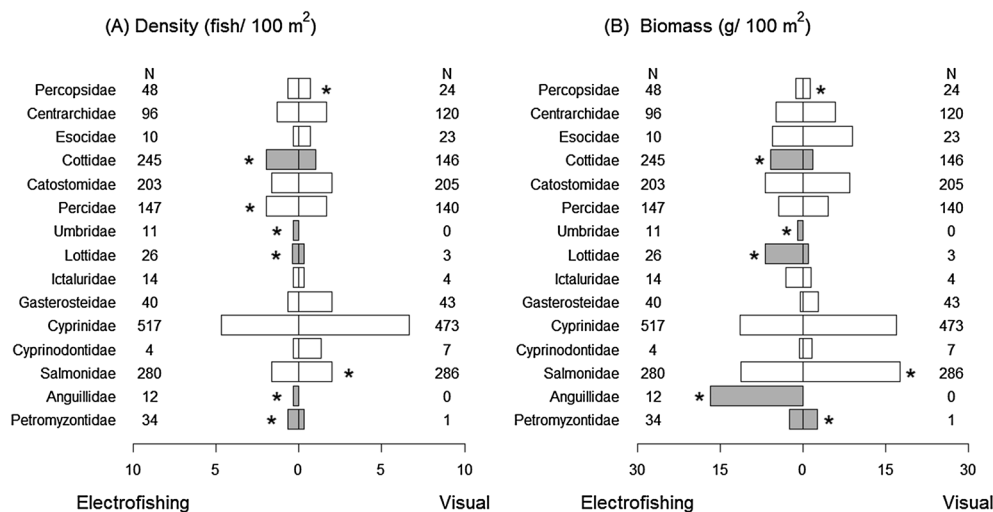


Figure 2. Median relative family density (A) and biomass (B) estimates per 100 m^2 (abscissa), for electrofishing and visual sampling methods. Shaded bars indicate the families that are well explained by the global model and differ notably between the two sampling methods (cumulative fit for families). *N*, taxon occurrence; *pRDA tests with Hochberg correction, $p \leq 0.05$ between sampling methods

Table II. Prevalent species listed by province and river of provenance

Species	Provenance (province and rivers)
Atlantic salmon <i>Salmo salar</i> L., 1756	St-Jean, Petit Saguenay, Dee, Serpentine, Gulquac
Black nose dace <i>Rhinichthys atratulus</i> (Hermann, 1804)	aux Saumons, Bécancour, Etchemin, Petit Saguenay, St-Jean, Batchawana, Goulais, Magpie, Aubinadong, Serpentine, Dee, Gulquac
Blunt nose minnow <i>Pimephales notatus</i> (Rafinesque, 1820)	aux Saumons, Bécancour, Goulais, Mississagi, Aubinadong
Brook stickleback <i>Culaea inconstans</i> (Kirtland, 1840)	Etchemin, Batchawana, Goulais, Magpie, Gulquac
Brook trout <i>Salvelinus fontinalis</i> (Mitchill, 1814)	aux Saumons, Bécancour, Etchemin, Petit Saguenay, St-Jean, Batchawana, Goulais, Magpie, Mississagi, Aubinadong, Serpentine, Dee, Gulquac, Elbow, Kananaskis Castle, Waterton, Kananaskis, Elbow
Brown trout <i>Salmo trutta</i> L., 1758	Castle, Waterton, Kananaskis, Elbow
Bull trout <i>Salvelinus malma</i> (Walbaum, 1792)	aux Saumons, Goulais, Magpie, Mississagi, Aubinadong, Waterton, Castle
Burbot <i>Lota lota</i> (Linnaeus, 1758)	aux Saumons, Bécancour, Etchemin, St-Anne, Goulais, Mississagi, Aubinadong, Batchawana, Dee, Gulquac
Common shiner <i>Lucilus cornutus</i> (Mitchill, 1817)	aux Saumons, Bécancour, Etchemin, Petit Saguenay, St-Jean, St-Anne, Aubinadong, Batchawana, Mississagi, Goulais, Dee, Serpentine, Gulquac
Creek chub <i>Semolilus atromaculatus</i> (Mitchill, 1818)	Bécancour, Etchemin, St-Anne
Cutlips minnow <i>Exoglossum maxillingua</i> (Lesueur, 1817)	aux Saumons, St-Anne, Goulais
Darter spp. (Tessellated, Johnny darters), <i>Etheostoma</i> spp. (Rafinesque, 1820)	QC, QC, ON
Fallfish <i>Semotilus corporalis</i> (Mitchill, 1817)	QC, QC, ON, NB
Fantail darter <i>Etheostoma flabellare</i> Rafinesque, 1819	QC
Lake chub <i>Couesius plumbeus</i> (Agassiz, 1850)	QC, ON, NB, AB
Logperch <i>Percina caprodes</i> (Rafinesque, 1818)	QC, ON
Long nose dace <i>Rhinichthys cataractae</i> (Valenciennes, 1842)	QC, ON, AB
Mountain whitefish <i>Prosopium williamsoni</i> (Girard, 1856)	AB
Northern pike <i>Esox lucius</i> L., 1758	QC, ON
Rainbow trout <i>Oncorhynchus mykiss</i> (Walbaum, 1792)	ON, AB
Rainbow-cuthroat hybrid (raincut) <i>Oncorhynchus</i> spp. (Richardson, 1836)	AB
Rock bass <i>Ambloplites rupestris</i> (Rafinesque, 1817)	QC, ON
Sculpin spp. (Slimy and Mottled sculpins) <i>Cottus</i> spp. (Girard, 1850)	QC, ON, NB
Smallmouth bass <i>Micropterus dolomieu</i> Lacepède, 1802	QC, ON
Sucker spp. (Long nose, White and Mountain suckers) <i>Catostomus</i> spp. Lesueur, 1817	QC, ON, NB, AB
Three spine stickleback <i>Gasterosteus aculeatus</i> L., 1758	QC, NB
Trout-perch <i>Percopsis omiscomaycus</i> (Walbaum, 1792)	QC, ON, AB

QC, Québec; ON, Ontario; NB, New Brunswick; AB, Alberta

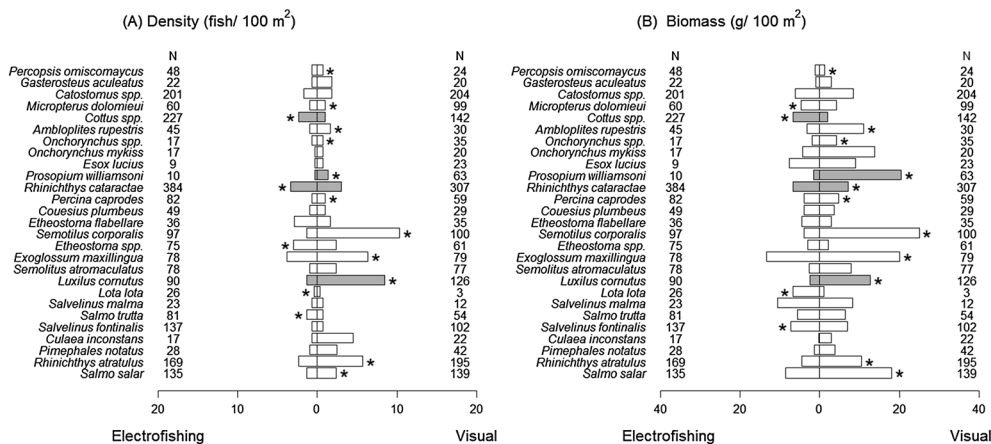


Figure 3. Median relative species density (A) and biomass (B) estimates per 100 m², for electrofishing and visual sampling methods. Refer to caption of Figure 2

(9.29%), *R. cataractae* (15.05%), *Cottus* spp. (18.59%) and *L. cornutus* (8.46%) had the greatest cumulative fit values derived from both density and biomass species matrices (cumulative fit values for density estimates displayed and illustrated by shaded bars in Figure 3). Likewise, biomass estimates were significantly different between sampling methods for all the same species in addition to *S. fontinalis* (univariate pRDA tests with Hochberg correction, $p < 0.05$), and *S. trutta* ($p = 0.075$).

Thus far, all size classes were merged to give total family and species averages for electrofishing and visual surveys. Consistent with previous family-level and species-level results, there was a significant difference in species-by-size-class densities and biomasses between electrofishing and visual surveys for sculpins, several salmonid and cyprinid species by 5-cm size increments (multivariate pRDA tests, $p < 0.005$). For example, juvenile *S. salar* (1, 0–5 cm; 2, 5–10 cm) had approximately 1.5 to 4 times greater visual density and biomass estimates than

electrofishing surveys (univariate pRDA tests, $p < 0.008$), while the reverse was true for density estimates for juvenile *S. trutta* of 5–10 cm (univariate pRDA tests, $p = 0.03$, Figure 4). Likewise, smaller *R. cataractae* (2: 5–10 cm) had between two and three times higher density and biomass estimates for electrofishing surveys (univariate pRDA tests, $p = 0.008$), while juvenile *R. atratulus* (1, 0–5 cm and 2, 5–10 cm), *L. cornutus* (1, 0–5 cm; 2, 5–10 cm; and 3, 10–15 cm), *Nocomis biguttatus* (1: 0–5 cm) and *S. corporalis* (1, 0–5 cm, and 3, 10–15 cm) all had greater density and biomass estimates for visual sampling (univariate pRDA tests, $p < 0.008$, Figure 5). Snorkelling yielded overall greater density and biomass estimates for larger salmonid species and for adults, save for trout species *Salvelinus malma* and *S. trutta*, which revealed greater densities and biomasses across size classes for electrofishing surveys. Larger *S. salar* and *S. fontinalis* individuals also showed similar sampling outcomes. Not only were snorkelling biomass and density estimates at least 1.5 times greater for

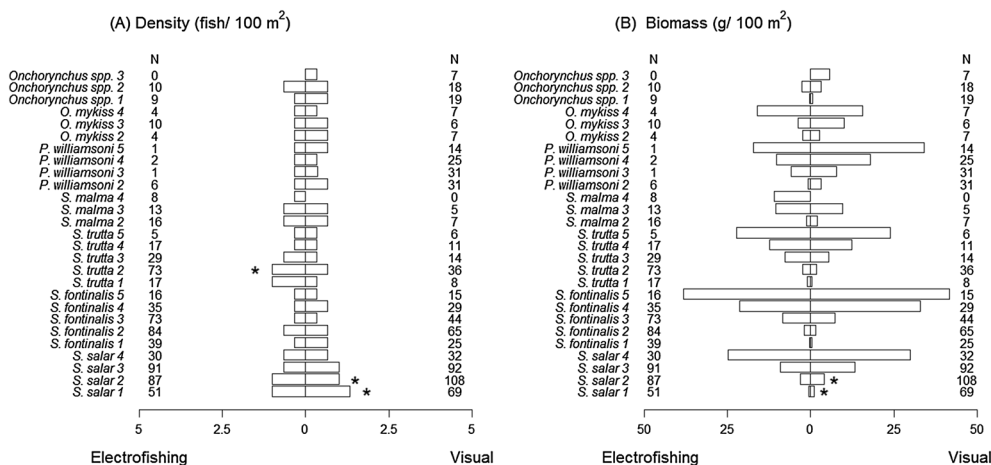


Figure 4. Median salmonid (size class) relative density (A) and biomass (B) estimates per 100 m², for electrofishing and visual sampling methods. Refer to caption of Figure 2 (1, 0–5 cm; 2, 5–10 cm; 3, 10–15 cm; and 4, 15–20 cm)

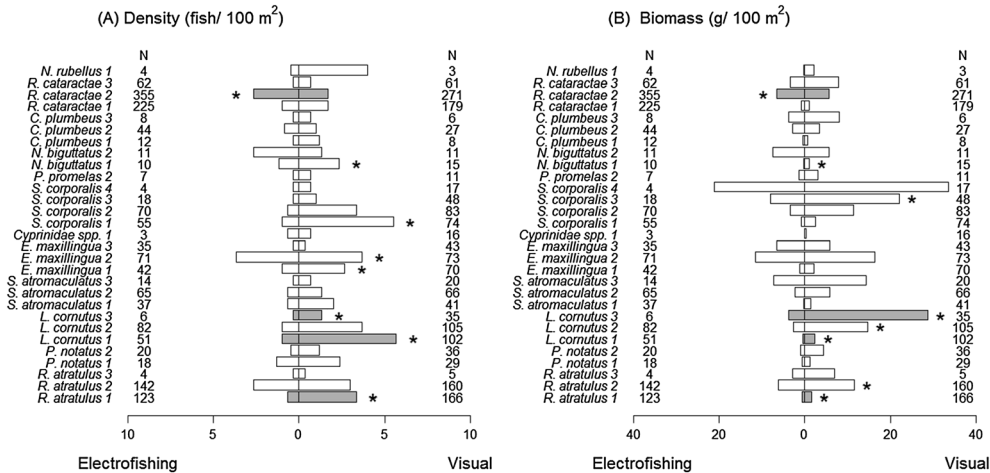


Figure 5. Median cyprinid (size class) relative density (A) and biomass (B) estimates per 100 m², for electrofishing and visual sampling methods. Refer to caption of Figure 2 (1, 0–5 cm; 2, 5–10 cm; 3, 10–15 cm; 4, 15–20 cm; and 5, 20–25 cm)

larger cyprinids (*S. corporalis*, *L. cornutus*, *E. maxillingua* and *S. atromaculatus* of 3, 10–15 cm, and 4, 15–20 cm size classes) but also certain size classes (*S. corporalis* of 10–15 and 15–20 cm and *L. cornutus* of 10–15 cm) were totally unrepresented in electrofishing surveys. Cumulative fit values greater than the community average for both biomasses and densities were seen for *Cottus* spp. measuring between 5 and 10 cm (30.41%), *L. cornutus* of 0–5 cm (16.79%) and 10–15 cm (9.26%), *R. cataractae* of 5–10 cm (25.25%) and *R. atratulus* of 0–5 cm (19.72%); cumulative fit values for density estimates displayed and illustrated by shaded bars in Figure 5).

Interriver sampling differences

The ‘best mean river biomass estimates’ obtained from combining the snorkelling and electrofishing survey data were regressed against mean biomass estimates calculated

from visual or electrofishing surveys. Type II regressions quantifying the relationship between our ‘best mean river biomass estimates’ and those derived from visual ($R^2=0.98$, slope = 1.09, intercept = 0.27, $p < 0.005$; Figure 6A) or electrofishing surveys ($R^2=0.46$, slope = 4.69, intercept = -1.11, $p < 0.001$; Figure 6B) were significant. The slope derived from the regression between our ‘best mean river biomass estimates’ in relation to those from visual sampling was not significantly different from a slope of 1, but the intercept significantly differed from the origin of 0, indicating that mean estimates obtained through snorkelling closely approximated ‘best mean river biomass estimates’. The slope and intercept obtained when regressing ‘best mean river biomass estimates’ against mean river biomass estimates for electrofishing were significantly different from the 1:1 relationship. Biomass estimates derived from electrofishing were more variable than those from the snorkelling surveys, as indicated by the lower R^2 .

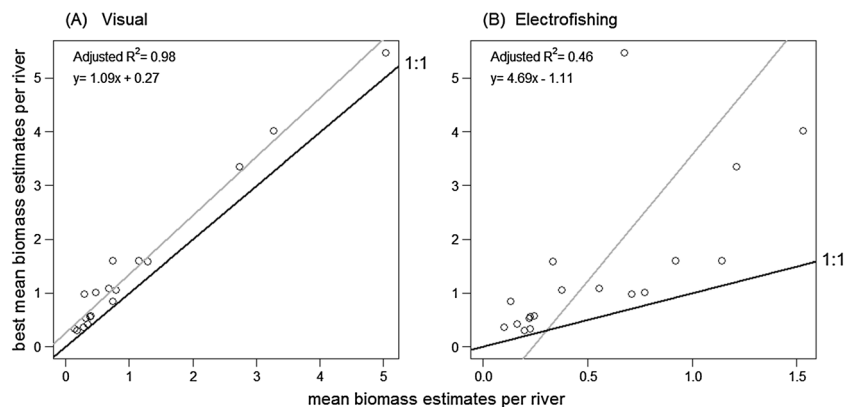


Figure 6. Major axis model II regressions displaying mean relative biomass per river for the ‘best mean biomass estimates’ in relation to mean relative biomass estimates derived from visual sampling (A) and electrofishing (B) methods. Grey lines indicate the major axis regression line for the model, and black lines represent the 1:1 relationships

DISCUSSION

Our results quantify the relative performances of electrofishing and snorkelling methods for estimating fish species richness, as well as population densities and biomasses for each family, species, and selected species and size classes sampled across several Canadian rivers. When attempting to maximize species richness of sampling sites, electrofishing yields superior results relative to visual surveys. In particular, all species but *Hybognathus regius* were detected in electrofishing surveys, while species characterized by behavioural or cryptic colouration were nearly or totally absent from visual surveys. In fact, significantly different estimates of densities and biomasses between sampling methods were found for known cryptic families, including the Cottidae, Lottidae, Anguillidae and Petromyzontidae. These results support earlier studies that found visual surveying significantly underestimated cryptic fish densities (Willis, 2001). Species like *A. rostrata*, *L. lota* and all lamprey species, especially larval or ammocete life history stages, typically shelter under boulders, inhabit crevices or burrow into the soft muddy or sandy stream bottoms (Scott and Crossman, 1973; Bernatchez and Giroux, 2000). *A. nebulosus*, *U. limi* and Cottidae species are relatively small species, coloured to blend in well with surrounding substrate and vegetation, ostensibly negatively biasing visual surveys. Likewise, *R. cataractae* had significantly higher density and biomass estimates in electrofishing surveys, likely resulting from this species' tendency to shelter under gravel and cobble substrate, thereby limiting their detection when snorkelling.

While sampling via electrofishing may be favoured for assessing overall fish community richness and/or detecting the presence of elusive or incidental species, visual surveying is significantly more efficient at estimating densities and biomasses of salmonid and gasterosteid and, to a lesser extent, centrarchid and esocid families. Salmonid species such as *S. salar*, *P. williamsoni*, *O. mykiss* and *Oncorhynchus* hybrids (rainbow and cutthroat trout hybrids) were best assessed in snorkelling surveys, despite earlier findings suggesting that fish with fine scales, as is the case with salmonids, tend to be more vulnerable to detection by electrofishing than fish with coarse scales (e.g. cyprinids; Meador *et al.*, 2003). These authors suggested that coarser scales offer more protection from galvanotaxis than finer scales, which implies that electrofishing surveys are inefficient at collecting cyprinid species and better suited for salmonids within a site. However, in our study, salmonid species such as *S. salar*, *P. williamsoni*, *O. mykiss* and *Oncorhynchus* hybrids were best sampled with snorkelling surveys. Our results are thus more aligned with the findings of Heimbuch *et al.* (1997), who showed that salmonids, centrarchids and some cyprinids may be able to detect and

avoid the electrical field outside of the effective shocking radius. Such was also the case in the present dataset for *P. williamsoni*, *M. dolomieu*, *A. rupestris* ranging 0–10 cm, *E. lucius*, *S. corporalis*, *E. maxillingua*, *L. cornutus* and *R. atratulus*. All these species save for *P. williamsoni*, who occasionally school in deeper runs, are solitary and conspicuously occupy the water column, also facilitating their enumeration in underwater visual surveying. Furthermore, schooling fishes like juvenile gasterosteids and shoals of small to intermediate-sized catostomids and cyprinids like *R. atratulus* (0–5 and 5–10-cm size classes), *L. cornutus* (0–5, 5–10 and 10–15 cm) and *S. corporalis* (0–5 and 5–10 cm) are often found in vegetated habitats along the shallow banks of rivers, which would normally suggest that detection while snorkelling would be impaired and lower density estimates generated. However, when combined with the fact that smaller fish have a higher chance of escaping electrofishing dipnets and appropriate time was accorded to visually assess high densities of fishes, it is not surprising that visual density estimates for these smaller size classes are close to double those obtained via electrofishing.

When evaluating differences between electrofishing and visual surveys for certain species over a range of size classes, sampling method outcomes are more variable. In this study, density and biomass estimates of juvenile *S. salar* life stages ranging approximately from 0 to 10 cm in total length are significantly greater for visual surveys than for electrofishing estimates, but when *S. salar* measuring between 10 and 20 cm are sampled, both sampling methods yield similar density and biomass estimates. These findings differ from those of Cunjak *et al.* (1988), who found that for high juvenile fish densities, snorkelling provided significantly lower salmonid densities. Although not supported for *S. salar* fry and parr life stages in this study, higher estimates were generated via electrofishing relative to snorkelling surveys of *S. trutta*, *S. fontinalis*, *S. malma*, in keeping with earlier studies (Wildman and Neumann, 2003; Thurow *et al.*, 2006). Our results suggest that *P. williamsoni*, *O. mykiss* and *Oncorhynchus* hybrids are best sampled via snorkelling surveys and this observation holds especially true for the largest individuals sampled in this dataset (size range from 10 to 25 cm). These observations oppose previous assertions that electrofishing often yields samples that overrepresent large fish and underrepresent small fish (Mullner *et al.*, 1998). Rather, it is our opinion that discrepancies in estimates for larger individuals between snorkelling and electrofishing surveys arise from: (i) avoidance of the electrical field from outside the shock radius and (ii) specific habitat preferences for deeper pools or fast flowing runs, which limit the range of application of backpack electrofishing.

When looking at the relationship between mean relative biomass variations between rivers for our 'best mean river

biomass estimates' and those obtained from visual or electrofishing surveys, it is apparent that snorkelling estimates most closely resemble our 'best mean river biomass estimates', while those derived from electrofishing surveys are more variable. Electrofishing is likely more variable because it is dependent on site attributes that vary greatly between rivers (i.e. water conductivity) and limit electrofishing sampling efficiency, especially for large individuals. However, because large shoals of juvenile fish and bigger individuals account for the greatest biomass portion for any given site or river, we suggest that snorkelling surveys may best capture river biomass estimates and should be the preferred choice, should one want to estimate fish production within a particular river.

These results provide the data and analysis needed for informing future research and management practices. For example, our study reveals that for larger salmonids, *P. williamsoni*, *O. mykiss* and *Oncorhynchus* hybrids, snorkelling surveys yield higher estimates, while certain families and/or species surveyed exhibiting cryptic colouration or behaviour (e.g. Cottidae species, *A. rostrata*, *L. lota*, *A. nebulosus* and *U. limi*) are best sampled via electrofishing methods. These results suggest that any future study focussed on assessing abundance or biomass estimates for rivers containing any of these species should use visual and electrofishing surveys respectively to do so. With the information generated from each surveying method, recommendations may be inferred; however, prescribing a comprehensive guiding framework is more complex as it is contingent on the particular objectives and limitations set out by the study.

Certain training and/or operational costs may limit the scope of a study. In particular, substantial training to meet certification requirements and greater resources (i.e. personalized gear and time and money spent transporting gear) allocated for personnel conducting electrofishing surveys will often result in a fewer number of sites or rivers covered during the survey. Conversely, visual surveying is generally considered low cost and does not harm fish but requires that snorkelers be properly trained for correct species identification and accurate size estimation for a given river. Although challenging, our experience is that proper training is possible, thus providing us with a reliable, non-invasive census of the fish community sampled per river.

This study is unique in that it allows for the direct comparison of electrofishing and snorkelling surveying methods for the estimation of several biotic indices, such as measures of fish quantities (i.e. species richness, density and biomass), across a wide variety of temperate rivers in Canada. By controlling for the effects of experimental and abiotic factors, the performance of electrofishing and snorkelling methods for estimating fish community richness, density and biomass was directly compared and clearly displayed

in such a way that one may select the most suitable sampling method for estimating specific families, species and size classes in selected systems. This study provides ample evidence that electrofishing and visual surveying methods generate different types of information; whether assessing fish community structure at the family level or by size classes, our results provide insight as to the most effective sampling method for a given fish community composition, which is fundamental to fisheries management and research.

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