Fisheries Research 242 (2021) 106016

Contents lists available at ScienceDirect

Fisheries Research

journal homepage: www.elsevier.com/locate/fishres

A comparison of passive and active gear in fish community assessments in summer versus winter

Hossein Mehdi [a,*], Samantha C. Lau [b], Caitlyn Synyshyn [a], Matthew G. Salena [a], Markelle E. Morphet [a], Jonathan Hamilton [a], Melissa N. Muzzatti [a], Erin S. McCallum [c], Jonathan D. Midwood [d], Sigal Balshine [a]

[a] Department of Psychology, Neuroscience & Behaviour, McMaster University, 1280 Main Street West, Hamilton, Ontario, L8S 4K1, Canada
[b] Department of Biology, McMaster University, 1280 Main Street West, Hamilton, Ontario, L8S 4K1, Canada
[c] Department of Wildlife Fish and Environmental Studies, Swedish University of Agriculture Sciences, SE-90183, Umeå, Sweden
[d] Great Lakes Laboratory for Fisheries and Aquatic Sciences, Fisheries and Oceans Canada, 867 Lakeshore Road, Burlington, ON, L7S 1A1, Canada

A R T I C L E I N F O

Handled by Niels Madsen

Keywords:
Hamilton Harbour
Gear selectivity
Seasonality
Fish community
Gear type
Winter

A B S T R A C T

Fish populations and communities are monitored using a variety of sampling gears, each with their own inherent biases. Gear biases can arise from a number of factors, such as fish species characteristics (e.g., body shape/size, physiology, and behaviour), species habitat requirements, as well as the abiotic characteristics of sites sampled. Such factors and their effects on gear selectivity are also heavily influenced by seaso FEEDBACK derstanding the effects of seasonal changes on gear selectivity is of vital importan winter—a season seldom studied in freshwater systems. Here, we compared the selectivity, efficiency, and degree of biodiversity in fish communities sampled using three gear types: minnow traps, Windermere traps, and electrofishing during summer and winter in Hamilton Harbour, ON, Canada. Catch per unit effort was similar among gear types in the summer, whereas in the winter, minnow traps captured the most fish. Electrofishing samples were the most species rich and species diverse, but only during the summer. Additionally, sampling efficiency and the number of different species encountered was highest when all gear types were used in combination, followed by electrofishing alone, Windermere traps alone, and minnow traps alone in both seasons. Each gear type differed in its selectivity for certain species, which was further influenced by seasonality. This resulted in the fish communities caught within each gear type being dissimilar from one another. Our study highlights the importance of understanding gear type selectivity, particularly under different climatic conditions, and outlines the importance of incorporating multiple gear types in ecological assessments of fish populations and communities.

1. Introduction

Fish population monitoring and community assessment are a cornerstone for fisheries management, conservation, and ecology. The accurate assessment of fish populations and communities is an essential yet extremely challenging task, as fish can occupy large and diverse habitats, move long distances, and be over- or underrepresented when monitored using certain sampling gears (MacKenzie et al., 2002; Elphick, 2008; Dextrase et al., 2014). Selecting the appropriate sampling gear, or gear type, is one of the most crucial decisions fish biologists and resource managers consider when conducting fish community and population assessments, as there is a great variety of gear types available, each with their own advantages and disadvantages (Portt et al., 2006; Bernhardt and Palmer, 2011; Jähnig et al., 2011). Gear types are often classified into two broad categories: (i) *active gear* and (ii) *passive gear*. Active gear has to be moved or activated by the sampler in order to catch fish (e.g., electrofishing, seine netting, and trawling; Portt et al., 2006; Winger et al., 2010). In contrast, passive gear is left out for a period of time before being retrieved, relying solely on the animal's movement and interaction towards it for capture to occur (e.g., minnow

* Corresponding author.
E-mail addresses: mehdih1@mcmaster.ca (H. Mehdi), lausc@mcmaster.ca (S.C. Lau), synyshyc@mcmaster.ca (C. Synyshyn), salenamg@mcmaster.ca (M.G. Salena), morphetm@mcmaster.ca (M.E. Morphet), hamilj11@mcmaster.ca (J. Hamilton), muzzattm@mcmaster.ca (M.N. Muzzatti), erin.s.mccallum@gmail.com (E.S. McCallum), jon.midwood@dfo-mpo.gc.ca (J.D. Midwood), sigal@mcmaster.ca (S. Balshine).

https://doi.org/10.1016/j.fishres.2021.106016

traps, Windermere traps, fyke nets, and gill nets; Hamley, 1975; Lagler, 1978; Portt et al., 2006; He and Pol, 2010). The usage of different gear types can lead to vast differences in fish population and community estimates, as each gear type has inherent biases and selectivity towards certain species, sexes, sizes, and habitats (Murphy and Willis, 1996; Ruetz III et al., 2007).

Fish sampling gear varies in a number of ways, including size, shape, period of deployment/activation, and usage of bait (Murphy and Willis, 1996; Portt et al., 2006). Such variety allows researchers to choose from a wide selection of gear types depending on targeted species of interest, habitat characteristics, and labour and gear cost considerations (Murphy and Willis, 1996; Diana et al., 2006). For example, fyke nets are well-suited for the capture of small-bodied mobile fishes, whereas electrofishing is well suited for the capture of sedentary fishes and is less size-selective (Hubert, 1996; Reynolds, 1996; Chick et al., 1999; Dolan and Miranda, 2003; Breen and Ruetz, 2006; Ruetz et al., 2007). Consequently, gear type selection can yield unforeseen biases in population and community assessments, especially if only a single sampling gear type is employed in a study, resulting in partial representation of the true community or population (Murphy and Willis, 1996). Gear type biases can influence our confidence in critical studies of fish assemblages, including those utilised in conservation efforts of endangered species, control and management of invasive species, and assessments of fish communities in response to anthropogenic disturbances (e.g., pollution, habitat modification, and climate change; Brandner et al., 2013; McCallum et al., 2019; Mehdi et al., 2021). For instance, in a study comparing the selectivity of electrofishing, trawling, seining, and drift netting in several large rivers draining into the North Sea and Baltic Sea, researchers found that electrofishing on average yielded higher biodiversity metrics, while species composition differed significantly across each sampling method (Zajicek and Wolter, 2018). Similarly, a study comparing the effectiveness of six different gear types (seine nets, hoop nets, trap nets, Windermere traps, minnow traps, and electrofishing) found staggering differences in the abundance, species richness, and species composition of fish caught between each gear type (Lapointe et al., 2006). Numerous additional studies have demonstrated the differences in fish population and community estimates that can arise from inherent gear type biases. However, despite such biases, single gear type sampling techniques continue to be employed in fish population and community studies, resulting in certain species being over- or underestimated; thereby hampering accurate population and community estimates.

Although gear type selectivity is widely recognised as a hindrance in fish community research, one issue that has received little attention is how seasonal changes in catch are influenced by gear selectivity. As mentioned before, species characteristics, species life history traits, species habitat requirements, sampling site characteristics (e.g., water depth, flow, clarity, temperature, and substrate type), and even inter- and intraspecies interactions can all contribute to gear type selectivity (Penczak and Jakubowski, 1990; Hubert and Fabrizio., 2007; Hubert et al., 2012). However, fish capture and sampling gear encounter rates are largely dependent on fish activity, which in turn is strongly modulated by seasonality (Rudstam et al., 1984; Wilson et al., 2011; Olsen et al., 2012). Most species have distinct seasonal movements and behaviours as their habitat requirements, spawning activity, food availability, and physiology change considerably across seasons (Hurst, 2007; McMeans et al., 2020). At higher latitudes, winters bring cold temperatures, shortened photoperiods, ice cover, hypoxia, and limited food availability (Shuter et al., 2012). Most fish species respond to the onset of such winter conditions with pronounced reductions in movement, foraging, growth, reproduction, as well as constriction of their home range (Hurst, 2007; McMeans et al., 2020). This is largely driven by reductions in metabolic rate, typical of ectothermic organisms would presumably decline (Hurst, 2007; Shuter et al., 2012; McMeans et al., 2020). In contrast, during the spring and summer when environments are warmer and more productive, fish are generally more active, explorative, and have higher energy demands met by greater metabolic scopes (Hasley et al., 2015). As such, fish are more likely to encounter and be caught by sampling gear, specifically passive gear types. Understanding the effects of seasonal changes on gear selectivity is of critical importance, especially in temperate and polar regions where winter is a dominant season, yet remains vastly understudied (McMeans et al., 2020). Research of this kind can further enhance and guide fisheries management, especially when most decisions have historically been based on research conducted during warmer periods of the year.

Our objective was to compare the selectivity and efficiency of three gear types on fish communities during summer and winter in shallow streams flowing into Hamilton Harbour, ON, Canada. Using minnow traps (passive gear), Windermere traps (passive gear) and boat electrofishing (active gear), we compared the abundance, richness, diversity, community composition, and the selectivity and species discovery rate (i.e., efficiency) of each gear type in the summer and winter. We predicted that overall abundance, richness, and diversity of fish samples would be lower during the winter, given lower fish mobility. Based on previous studies, we also predicted that these metrics would differ across gear types, with electrofishing being the most successful and efficient, especially during the winter, when fish activity is subdued, making fish less likely to encounter passive gear types. We further hypothesised that our gear types would capture distinct fish communities, as each gear type is inherently biased towards certain fish species characteristics (e.g., body shape/size, habitat preference, physiology, and behaviour). Furthermore, we predicted that the species makeup of communities captured within each gear type would differ between summer and winter, as gear type species selectivity is likely to differ across seasons.

2. Materials and methods

2.1. Study area

We sampled fish communities in Hamilton Harbour (ON, Canada), a large freshwater embayment situated at the western end of Lake Ontario. Due to historical degradation caused by anthropogenic development, Hamilton Harbour is listed as one of 43 Areas of Concern under the Great Lakes Water Quality Agreement (2012). The present study was focused on shallow (<2 m) streams and wetlands that flow into the harbour. Sampling took place at two different areas of the harbour, with five sampling sites in each area. First, we sampled along Red Hill Creek, which flows from Albion Falls on the Niagara Escarpment and discharges into the eastern end of Hamilton Harbour at the Windermere Basin. This area is heavily degraded and industrialised, with clear signs of anthropogenic modifications and shoreline alterations (McCallum et al., 2019; Mehdi et al., 2021). Second, we sampled sites located on the western end of the harbour, along Desjardins Canal, West Pond, and Spencer Creek. These sites are within Cootes Paradise Marsh, the largest wetland in western Lake Ontario. The marsh is a protected nature sanctuary, known for its rich biodiversity, use as an important migratory waterfowl stopover site, and fish nursery habitats (Leslie and Timmins, 1992; Smith and Chow-Fraser, 2010). Despite the considerable biodiversity found within Cootes Paradise Marsh, it is among the most degraded wetlands in Lake Ontario due to poor water quality and its hypereutrophic state (Chow-Fraser, 2006; Thomasen and Chow-Fraser, 2012). These particular sampling sites were initially targeted for accessibility and are part of a long-term research program (McCallum et al., 2019; Mehdi et al., 2021; Nikel al., 2021). See Supplementary Fig. 1 for a map of the sampling sites.

2.2. Sampling techniques

Fish communities were sampled during the summer (July and August) and winter (November, December, and March) of 2018 and 2019. Fish were sampled using a combination of passive (minnow traps and Windermere traps) and active (electrofishing from a boat) gears. These sampling gear types were chosen as they have been commonly used in shallow systems with different habitat types and were also used in previous research conducted at our study sites (McCallum et al., 2019; Mehdi et al., 2021; Nikel et al., 2021). Additionally, the gear types selected in our study offer novel insight into gear selectivity in the shallow zones of Hamilton Harbour. Ongoing monitoring efforts in Hamilton Harbour have largely been performed using electrofishing from large vessels, a method that cannot be utilised in shallow systems (Boston et al., 2016). At each site, and on each sampling event, 10 black minnow traps (wall height = 16.7 cm; entry hole diameter = 2.10 cm, trap length = 40.5 cm; mesh diameter = 0.76 cm), each baited with ~20 g of corn, were deployed from land ~10 m apart from one another. Two meshed Windermere traps (wall height = 66.0 cm; entry hole diameter = 17.5 cm, trap length = 96.0 cm; mesh diameter = 0.30 cm), each baited with ~100 g of corn, were also deployed from land ~10 m away from the first and last minnow traps. Minnow and Windermere traps were deployed on ropes extending ~5 m from shore. Traps were retrieved 24 h post-deployment. Additionally, at each site, two 50 m transects (within 5 m from shore) were sampled from a boat using a portable electrofishing unit (1.5-KVA Electrofisher, Smith-Root Inc.). All gear types were deployed at the same depth (see Supplementary Table 3). Sites on the eastern end of Hamilton Harbour were sampled five times in the summer and three times in the winter. Sites on the western end were sampled four times in the summer and three times in the winter. Sampling was always performed during daytime, between 0800 and 1400 h on weather permitting days. On each field date, all five sites on either end of Hamilton Harbour were sampled using all three techniques, with the exception of one of the summer dates, when electrofishing could not be performed due to heavy rainfall during sampling on the east end of the harbour. See Supplementary Table 1 for additional field sampling information.

During each sampling event, we measured the following water quality parameters: temperature, dissolved oxygen (YSI ProODO), pH, salinity, conductivity, and total dissolved solids (Oakton multiparameter Testr) at each site (see Supplementary Table 2). Habitat characteristics were also assessed at each site based on a subset of the metrics used in the Qualitative Habitat Evaluation Index (QHEI; Taft and Koncelik, 2006; Strickland et al., 2010) and following a previously described protocol by McCallum et al. (2019). Habitat metrics taken included: total water depth, water clarity (Secchi depth), substrate type, sediment particle size, shoreline slope, degree of sinuosity, degree of anthropogenic modifications (i.e., physical modifications of the shore-water interface), riparian zone width, degree of estimated bank erosion, and the presence of any aquatic plants (see Supplementary Table 3).

Fish collected at each sampling site were transported in dark-coloured, aerated bins to shore, where they were counted and identified to species level. At each site, fish caught using minnow traps or Windermere traps or electrofishing were pooled and measured with other fish caught with the same respective gear type. The standard and total lengths (mm) and body mass (g) of the first 15 individuals of a given species caught at each site were individually measured, while the remaining fish were only counted and batch-weighed. This was done to reduce processing time and handling stress. Native fishes were immediately returned to their site of collection, while invasive fish were euthanised with an overdose of benzocaine (small fishes; < 10 cm) or by a lethal cephalic blow (large fishes; > 10 cm), as required by the Ontario Ministry of Natural Resources and Forestry (2015). All fish were handled

2.3. Statistical analysis

All statistical analyses were performed using R (version 3.6.2, R Core Team, 2019). Prior to any analysis of fish communities, fish count data were mean-standardised for each gear type (multi-gear mean standardisation; following Gibson-Reinemer et al., 2017) to allow the different gear types to be compared to one another. Catch per unit effort (CPUE) was calculated per trap deployed for minnow traps and Windermere traps and per shock-second for electrofishing. Fish abundance (in total number and biomass), species richness (number of identified species), species diversity (Shannon-Weiner Index), proportion of benthic species, proportion of invasive species (*invasive*: not native to the Great Lakes), proportion of tolerant species (*tolerant*: able to respond and adapt to disturbances and perturbations in its environment as defined by Eakins (2018)), and proportion of resilient species (*resilient*: able to recover and double its population within <1.4 years following exploitation as defined by Eakins (2018)) were analysed using permutation linear mixed effects models (PLMM; n = 5000 iterations) with gear type and season as main effects, and sampling period within each season and sampling site as random effects (lme4 and predictmeans packages; Bates et al., 2015; Luo et al., 2020). Overall morphological differences (total length and body mass) between fish caught in each gear type were analysed in a similar manner, while including fish species as a random effect. Tukey's HSD post-hoc tests were used to identify significant pairwise differences between each gear type within a season. Fish community compositional differences among gear types and between seasons were visualised using unconstrained principal coordinate analysis (PCoA) biplots performed on Bray-Curtis dissimilarity matrix, with 80 % confidence ellipses overlaid on top of each gear type (Oksanen et al., 2019). Differences in gear type and seasonal community compositions were further analysed using permutation ANOVAs with 5000 permutations (Vegan package; Oksanen et al., 2019). Fish communities were further examined using similarity percentages (SIMPER) analysis to determine which species were driving compositional differences within each gear type. Permutation tests (n = 5000) were used to identify gear type significant differences in CPUE of these characteristic species within each season. Although all species were included in the model, only those that contributed > 5% to the total abundance were analysed. Indicator species analysis (ISA) was used to examine which species were indicative of each gear type within each season. Indicator species values (ISVs), ranging from 0 (absent from all samples) to 1 (present in all samples within gear type), were generated to determine which species are considered "true" indicators and are consistently present within a certain gear type. Permutation tests (n = 5000) were also carried out to determine which species were significant indicators. Species accumulation curves (SACs) were used to determine gear type efficiency and measured if the sample size (number of sampling sites) was large enough to adequately characterise the communities caught within each gear type during each season (McCune and Grace, 2002). Each sampling effort (site) in our SACs was conducted on the species richness found in 10 minnow traps, two Windermere traps, or two electrofishing transects. SACs were generated for each gear type within each season from random permutations of the data (n = 5000) to determine the average number of new species found and standard deviations for each increase in sampling effort (specaccum function, Vegan package; Oksanen et al., 2019). SACs, for each individual gear type and for all gear types combined, were compared by calculating the initial slope (averaged across efforts 1 through 5). We also compared the amount of effort needed for each gear type when used individually versus in combination to reach a cut-off of >1% of new species discovered per effort standardised by the maximum amount of potential species that can be caught using each method. SACs were based only on sampling events when all three gear types were employed. Data are s characteristics and abundances from all sampling events. The native vs. non-native status of each species in Ontario (native/non-native) is based on the Ontario Ministry of Natural Resources and Forestry eria. Tolerance describes a species' ability to respond and adapt to disturbances and perturbations in its environment following Eakins (2018). Resilience describes a species recovering capacity and its doubling ving exploitation (low > 4 years, medium 1.4–4 years, high < 1.4 years; Froese and Pauly, 2020). Habitat is based on where in the water column each species is typically found (Eakins, 2018). Abundance catch number of individuals caught of each species using each of the three gear types (minnow traps, Windermere traps, and electrofishing) in summer and winter (summer | winter). Catch abundance data are ely represented from all sampling events within each season.

<table>
<tr><th></th><th></th><th>Species Characteristics</th><th></th><th></th><th></th><th>Catch Abundance</th><th></th><th></th></tr>
<tr><th></th><th>Family</th><th>Native/Non-native</th><th>Tolerance</th><th>Resilience</th><th>Habitat</th><th>Minnow Traps</th><th>Windermere Traps</th><th>Electrofishing</th></tr>
<tr><td>s rupestris (Rock bass)</td><td>Centrarchidae</td><td>Native</td><td>Intermediate</td><td>Medium</td><td>Benthopelagic</td><td>0 | 0</td><td>0 | 6</td><td>0 | 0</td></tr>
<tr><td>nebulosus (Brown bullhead)</td><td>Ictaluridae</td><td>Native</td><td>Intermediate</td><td>Medium</td><td>Benthic</td><td>34 | 37</td><td>71 | 9</td><td>192 | 13</td></tr>
<tr><td>a (Bowfin)</td><td>Amiidae</td><td>Native</td><td>Intermediate</td><td>Low</td><td>Benthopelagic</td><td>1 | 0</td><td>0 | 0</td><td>1 | 13</td></tr>
<tr><td>uratus (Goldfish)</td><td>Cyprinidae</td><td>Non-native</td><td>Tolerant</td><td>Medium</td><td>Benthopelagic</td><td>25 | 0</td><td>42 | 1</td><td>280 | 5</td></tr>
<tr><td>s commersonii (White sucker)</td><td>Catostomidae</td><td>Native</td><td>Tolerant</td><td>Low</td><td>Benthic</td><td>5 | 2</td><td>27 | 0</td><td>146 | 13</td></tr>
<tr><td>onstans (Brook stickleback)</td><td>Gasterosteidae</td><td>Native</td><td>Intermediate</td><td>High</td><td>Benthopelagic</td><td>6 | 6</td><td>40 | 0</td><td>3 | 5</td></tr>
<tr><td>arpio (Common carp)</td><td>Cyprinidae</td><td>Non-native</td><td>Tolerant</td><td>Medium</td><td>Benthopelagic</td><td>1 | 0</td><td>1 | 0</td><td>8 | 16</td></tr>
<tr><td>cepedianum (Gizzard shad)</td><td>Clupeidae</td><td>Non-native</td><td>Tolerant</td><td>Medium</td><td>Pelagic</td><td>2 | 3</td><td>0 | 19</td><td>166 | 6</td></tr>
<tr><td>s (Northern pike)</td><td>Esocidae</td><td>Native</td><td>Intermediate</td><td>Low</td><td>Benthopelagic</td><td>0 | 0</td><td>3 | 0</td><td>14 | 3</td></tr>
<tr><td>s sicculus (Brook silverside)</td><td>Atherinopsidae</td><td>Native</td><td>Intermediate</td><td>High</td><td>Pelagic</td><td>0 | 0</td><td>0 | 0</td><td>1 | 2</td></tr>
<tr><td>s osseus (Longnose gar)</td><td>Lepisosteidae</td><td>Native</td><td>Tolerant</td><td>Low</td><td>Benthopelagic</td><td>0 | 0</td><td>0 | 0</td><td>1 | 0</td></tr>
<tr><td>anellus (Green sunfish)</td><td>Centrarchidae</td><td>Native</td><td>Tolerant</td><td>Medium</td><td>Benthopelagic</td><td>19 | 26</td><td>77 | 18</td><td>161 | 75</td></tr>
<tr><td>bbosus (Pumpkinseed sunfish)</td><td>Centrarchidae</td><td>Native</td><td>Intermediate</td><td>Medium</td><td>Benthopelagic</td><td>5 | 6</td><td>33 | 9</td><td>67 | 5</td></tr>
<tr><td>acrochirus (Bluegill sunfish)</td><td>Centrarchidae</td><td>Native</td><td>Intermediate</td><td>Medium</td><td>Benthopelagic</td><td>50 | 15</td><td>330 | 19</td><td>283 | 10</td></tr>
<tr><td>nutus (Common shiner)</td><td>Cyprinidae</td><td>Native</td><td>Intermediate</td><td>Medium</td><td>Benthopelagic</td><td>0 | 0</td><td>8 | 2</td><td>38 | 0</td></tr>
<tr><td>s salmoides (Largemouth bass)</td><td>Centrarchidae</td><td>Native</td><td>Tolerant</td><td>Low</td><td>Benthopelagic</td><td>3 | 0</td><td>32 | 0</td><td>105 | 15</td></tr>
<tr><td>nericana (White perch)</td><td>Moronidae</td><td>Non-native</td><td>Intermediate</td><td>Low</td><td>Benthopelagic</td><td>38 | 0</td><td>82 | 2</td><td>4 | 0</td></tr>
<tr><td>melanostomus (Round goby)</td><td>Gobiidae</td><td>Non-native</td><td>Intermediate</td><td>Medium</td><td>Benthic</td><td>283 | 84</td><td>61 | 20</td><td>46 | 2</td></tr>
<tr><td>therinoides (Emerald shiner)</td><td>Cyprinidae</td><td>Native</td><td>Intermediate</td><td>High</td><td>Benthopelagic</td><td>0 | 2</td><td>9 | 2</td><td>10 | 5</td></tr>
<tr><td>udsonius (Spottail shiner)</td><td>Cyprinidae</td><td>Native</td><td>Intermediate</td><td>Medium</td><td>Benthopelagic</td><td>3 | 5</td><td>148 | 2</td><td>18 | 3</td></tr>
<tr><td>escens (Yellow perch)</td><td>Percidae</td><td>Native</td><td>Intermediate</td><td>Medium</td><td>Benthopelagic</td><td>14 | 3</td><td>8 | 7</td><td>65 | 13</td></tr>
<tr><td>prodes (Common logperch)</td><td>Percidae</td><td>Native</td><td>Intolerant</td><td>Medium</td><td>Benthic</td><td>14 | 1</td><td>10 | 1</td><td>9 | 12</td></tr>
<tr><td>s notatus (Bluntnose minnow)</td><td>Cyprinidae</td><td>Native</td><td>Intermediate</td><td>Medium</td><td>Benthopelagic</td><td>1 | 1</td><td>8 | 1</td><td>56 | 3</td></tr>
<tr><td>s promelas (Fathead minnow)</td><td>Cyprinidae</td><td>Native</td><td>Tolerant</td><td>High</td><td>Benthopelagic</td><td>2 | 8</td><td>74 | 7</td><td>282 | 15</td></tr>
<tr><td>igromaculatus (Black crappie)</td><td>Centrarchidae</td><td>Native</td><td>Tolerant</td><td>Medium</td><td>Benthopelagic</td><td>0 | 0</td><td>2 | 0</td><td>10 | 1</td></tr>
<tr><td>erythrophthalmus (Rudd)</td><td>Cyprinidae</td><td>Non-native</td><td>Tolerant</td><td>Low</td><td>Benthopelagic</td><td>0 | 1</td><td>62 | 5</td><td>53 | 9</td></tr>
<tr><td>atromaculatus (Creek chub)</td><td>Cyprinidae</td><td>Native</td><td>Intermediate</td><td>Medium</td><td>Benthopelagic</td><td>0 | 0</td><td>0 | 0</td><td>5 | 0</td></tr>
</table>

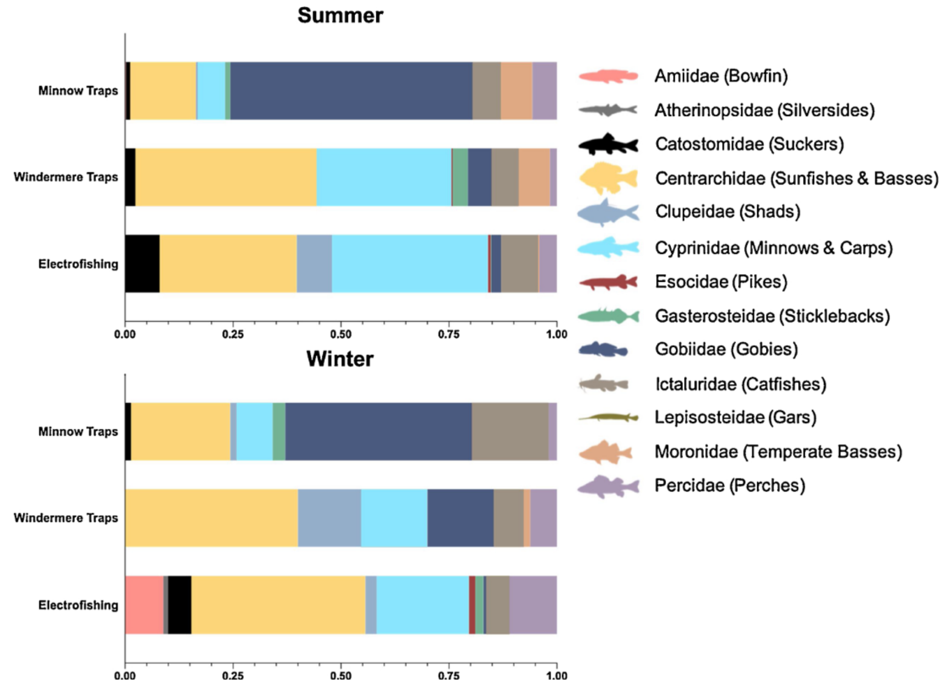

Fig. 1. Fish family composition broken down by season and gear type. Proportions based on gear-standardised catch per unit effort of all sampling events within each season. See Table 1 for species within each family group and see Supplementary Fig. 2 for fish species composition within each gear type and season.

3. Results

3.1. Abundance, richness, and diversity

Across all sites and sampling events, we caught 4226 fish (3658 in the summer and 568 in the winter) composed of 27 unique species (Table 1; Fig. 1). Minnow traps captured 706 fish (506 in the summer and 200 in winter) composed of 20 species. Windermere traps captured 1258 fish (1128 in the summer and 130 in winter) composed of 23 species. Electrofishing captured 2268 fish (2024 in the summer and 244 in winter) composed of 26 species.

Overall, fish abundance (number of fish) was greatly reduced in the winter compared to the summer (PLMM, $t = -3.28$, $p < 0.01$; Fig. 2A). In the summer, fish abundance was not significantly different across gear types (CPUE, PLMM, $F = 0.04$, $p = 0.41$; Fig. 2A). However, during the winter, gear type played a critical role in determining the number of fish caught (PLMM, $F = 12.35$, $p < 0.001$), with minnow traps more successful at capturing fish than either Windermere traps ($t = 3.84$, $p < 0.001$; Fig. 2A) or electrofishing ($t = 4.78$, $p < 0.001$; Fig. 2A). Windermere traps and electrofishing were equally successful at capturing fish during the winter ($t = 0.36$, $p = 0.93$). Biomass did not differ between seasons (PLMM, $t = 1.47$, $p = 0.12$) nor across gear types (PLMM, $F = 0.08$, $p = 0.27$).

Similar to abundance, species richness and species diversity (Shannon-Weiner Index) were greatly reduced during the winter compared to the summer (PLMM, $t_{(\text{Richness})} = -4.44$, $p < 0.001$; $t_{(\text{Diversity})} = -3.99$, $p < 0.01$; Fig. 2B; Fig. 2C). In the summer, species richness and species diversity varied significantly by gear type (PLMM, $F_{(\text{Richness})} = 14.89$, $p < 0.001$; $F_{(\text{Diversity})} = 12.16$, $p < 0.001$; Fig. 2B; Fig. 2C), with catches by minnow traps being considerably less species rich and less species diverse than catches from Windermere traps ($t_{(\text{Richness})} = -3.72$, $p < 0.001$; $t_{(\text{Diversity})} = -3.58$, $p < 0.01$; Fig. 2B; Fig. 2C) and electrofishing ($t_{(\text{Richness})} = -5.39$, $p < 0.001$; $t_{(\text{Diversity})} = -4.81$, $p < 0.001$; Fig. 2B; Fig. 2C). During the summer, the catch yielded by Windermere diversity did not differ across gear types during the winter (PLMM, $F_{(\text{Richness})} = 0.71$, $p = 0.46$; $F_{(\text{Diversity})} = 0.42$, $p = 0.63$; Fig. 2B; Fig. 2C).

3.2. Gear type species discovery rate

As sampling effort (number of sites) increased, the species accumulation curves for all gear types individually and cumulatively (all three gear types together) began to reach the asymptote in both summer and winter. This indicated that all gear types were able to adequately sample their respective fish communities, while also demonstrating significant differences in the total number of species that can potentially be caught by each gear type (Fig. 3). The species encounter rate as sampling effort increased was highest when all gear types were used in combination, followed by electrofishing alone, Windermere traps alone, and then minnow traps alone. This pattern was observed during both the summer and winter. Additionally, the initial species discovery slope (averaged across efforts 1 through 5) was greatest when all gear types were used in combination ($2.22_{(\text{Summer})}$, $2.30_{(\text{Winter})}$), followed by electrofishing alone ($2.12_{(\text{Summer})}$, $1.87_{(\text{Winter})}$), Windermere traps alone ($1.91_{(\text{Summer})}$, $1.26_{(\text{Winter})}$), and then minnow traps alone ($1.19_{(\text{Summer})}$, $1.07_{(\text{Winter})}$) in both seasons. In the summer, the effort needed to reach an increase of <1% in species richness was overall higher in the winter than in the summer, with an average effort of ~26 sites needed in the winter compared to ~22 sites in the summer. In the summer, the effort needed to reach an increase of <1% in species richness was 26 sites for minnow traps alone, 18 sites for Windermere traps alone, 28 sites for electrofishing alone, and 14 sites for when all gear types were used in combination. In the winter, the effort needed to reach an increase of <1% in species richness was >30 sites for minnow traps alone, 29 sites for Windermere traps alone, 23 sites for electrofishing alone, and 19 sites for when all gear types were used in combination.

3.3. Fish species characteristics and morphology

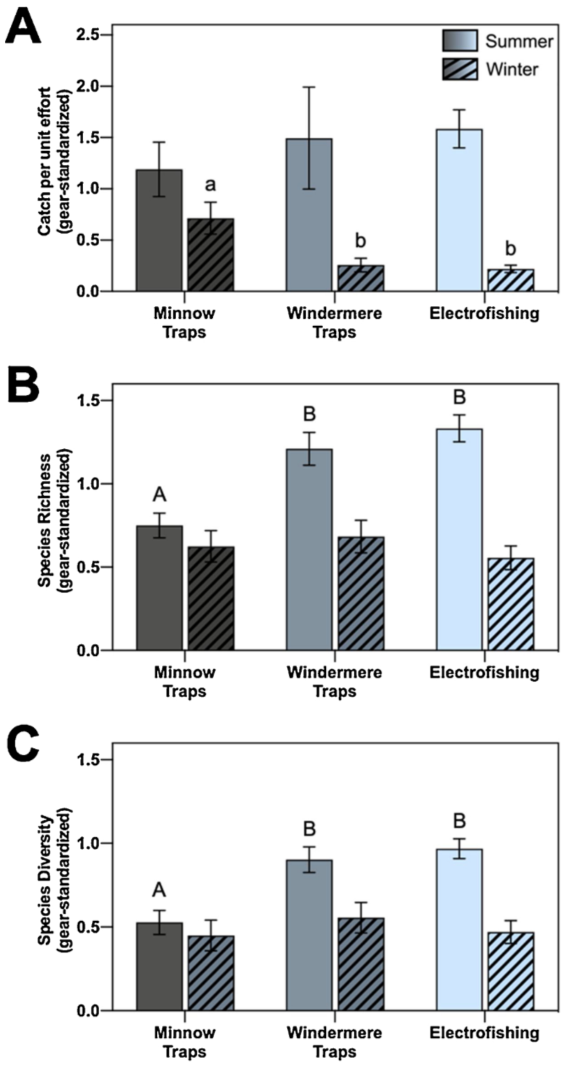

Fig. 2. Mean (± SE) gear-standardised **(A)** abundance, **(B)** species richness, and **(C)** species diversity of fish caught using minnow traps, Windermere traps, and electrofishing in the summer (solid) and winter (hatched). Different uppercase letters indicate significant pairwise differences between gear types in the summer, while different lowercase letters indicate significant pairwise differences between gear types in the winter.

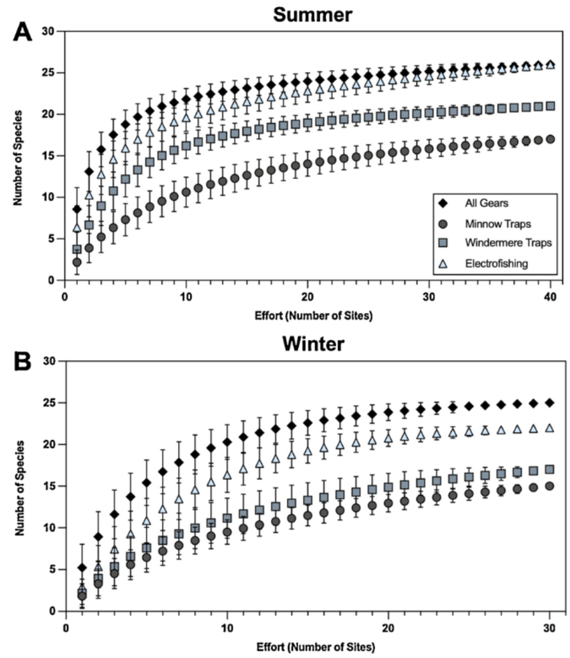

Fig. 3. Species accumulation curves for all gear types combined and each gear type individually in the **(A)** summer and **(B)** winter in all sampling sites across different events. Each data point represents the average richness achieved for each level of effort ± standard deviation. The efforts needed to reach <1% change in species richness in each gear type within each season are: minnow traps ($26_{(Summer)}$, $>30_{(Winter)}$); Windermere traps ($18_{(Summer)}$, $29_{(Winter)}$); electrofishing ($28_{(Summer)}$, $23_{(Winter)}$); all gears ($14_{(Summer)}$, $19_{(Winter)}$).

benthic fishes caught (PLMM, $F_{(Summer)} = 19.63$, $p < 0.001$); $F_{(Winter)} = 15.10$, $p < 0.001$; Fig. 4A). In both seasons, minnow trap catches consisted of a greater proportion of benthic fishes compared to catches using Windermere traps ($t_{(Summer)} = 5.01$, $p < 0.001$; $t_{(Winter)} = 3.02$, $p < 0.01$) and electrofishing ($t_{(Summer)} = 5.91$, $p < 0.001$; $t_{(Winter)} = 5.48$, $p < 0.001$). In the summer, the proportion of benthic fishes caught using Windermere traps did not differ from that of electrofishing ($t = 0.20$, $p = 0.98$), while in the winter, Windermere traps tended to catch more benthic fishes than electrofishing ($t = 2.25$, $p = 0.07$).

The proportion of non-native fishes did not differ between seasons (PLMM, $t = 1.03$, $p = 0.14$; Fig. 4B). During the summer, the proportion of non-native fishes was influenced by gear type (PLMM, $F = 13.35$, $p < 0.001$; Fig. 4B), with a higher proportion of non-native fishes caught by minnow traps compared to those caught using both Windermere traps ($t = 3.68$, $p < 0.01$) and electrofishing ($t = 5.05$, $p < 0.001$). all three gear types (PLMM, $F = 2.01$, $p = 0.14$; Fig. 4B).

The proportion of tolerant to intermediate and intolerant fishes caught did not differ between seasons (PLMM, $t = 0.10$, $p = 0.93$; Fig. 4C). However, gear type significantly influenced the proportion of tolerant species caught during both seasons (PLMM, $F_{(Summer)} = 21.94$, $p < 0.001$; $F_{(Winter)} = 16.09$, $p < 0.001$; Fig. 4C). Fishes caught using electrofishing were generally more tolerant than those caught in Windermere traps ($t_{(Summer)} = 3.73, p < 0.001$; $t_{(Winter)} = 3.70, p = 0.001$) and minnow traps ($t_{(Summer)} = 6.41, p < 0.001$; $t_{(Winter)} = 5.37, p < 0.001$) in both seasons. Windermere traps caught more tolerant fishes than minnow traps, but only significantly so during the summer ($t_{(Summer)} = 2.37$, $p = 0.049$; $t_{(Winter)} = 1.64$, $p = 0.24$).

The proportion of highly resilient fishes to those of medium or low resilience was not significantly affected by season (PLMM, $t = 0.23$, $p = 0.63$; Fig. 4D) nor gear type (PLMM, $F = 0.72$, $p = 0.49$; Fig. 4D); however, a significant interaction was detected, indicating gear type differences in the degree of resilient species caught were seasonally-dependent (PLMM, $F = 6.49$, $p = 0.002$; Fig. 4D). During the summer, more resilient species were caught using Windermere traps than via electrofishing ($t = 3.75$, $p = 0.001$). Similarly, Windermere traps tended to catch more resilient fishes than minnow traps ($t = 2.28$, $p = 0.06$). The proportion of resilient fishes caught using either minnow traps or electrofishing was similar ($t = 1.14$, $p = 0.49$). In the winter, the proportion of resilient fishes caught did not differ across gear types (PLMM, $F = 2.03$, $p = 0.14$; Fig. 4D).

To assess gear type size selectivity, we compared the differences in body size (estimated by total length in mm) of the most commonly caught species across our three gear types. These commonly caught species included: brown bullhead, goldfish, white sucker, green sunfish, pumpkinseed sunfish, bluegill sunfish, white perch, round goby, yellow

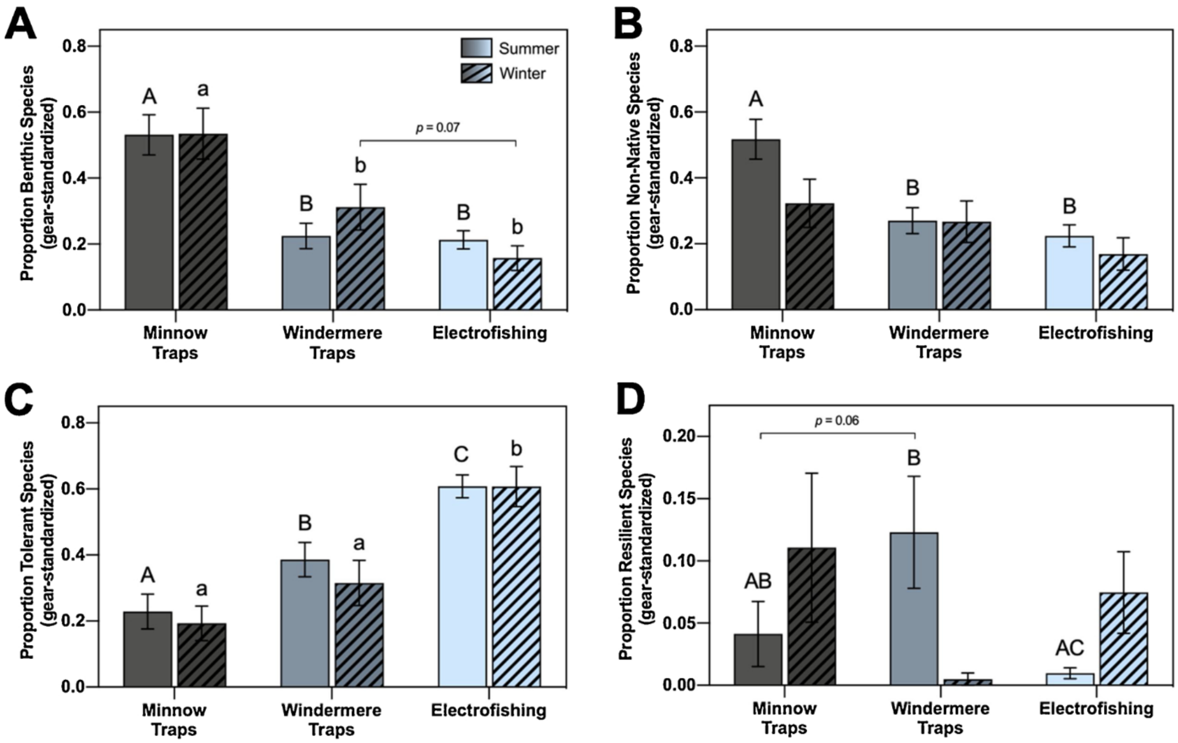

Fig. 4. Mean (± SE) gear-standardised **(A)** proportion of benthic species, **(B)** proportion of non-native species, **(C)** proportion of tolerant species, and **(D)** proportion of resilient species of fish caught using minnow traps, Windermere traps, and electrofishing in the summer and winter. Different uppercase letters indicate significant pairwise differences between gear types in the summer. Different lowercase letters indicate significant pairwise differences between gear types in the winter.

size of fish caught depended on which gear type was utilised (PLMM, $F = 10.86$, $p < 0.001$). During the summer, electrofishing caught larger fish than both minnow traps ($t = 2.33$, $p = 0.05$) and Windermere traps ($t = 3.78$, $p < 0.001$). In contrast, fish caught in minnow traps and Windermere traps were of similar sizes ($t = 0.74$, $p = 0.74$). During the winter, electrofishing caught larger fish than minnow traps ($t = 4.89$, $p < 0.001$) but not Windermere traps ($t = 1.82$, $p = 0.17$). Furthermore, Windermere traps caught fish of larger sizes than minnow traps ($t = 3.11$, $p < 0.01$). See Supplementary Table 7 for a detailed breakdown of body size by species. Similarly, body mass of the most commonly caught fishes was greater in the winter than in the summer (PLMM, $t = 5.25$, $p < 0.001$). Additionally, body mass of fish varied significantly based on which gear type they were caught in (PLMM, $F = 15.2$, $p < 0.001$). During both seasons, minnow traps selected for fishes with smaller body mass than electrofishing ($t_{(Summer)}$ = -2.41, $p = 0.04$; $t_{(Winter)}$ = -2.41, $p = 0.04$) but not Windermere traps ($t_{(Summer)}$ = -1.81, $p = 0.17$; $t_{(Winter)}$ = -1.75, $p = 0.19$). However, fish caught in Windermere traps and electrofishing did not differ significantly in their body mass in either season ($t_{(Summer)} = 0{,}62$ $p = 0.81$; $t_{(Winter)} = 1.48$, $p = 0.31$).

3.4. Community composition

The composition of fish communities differed significantly between seasons (PERMANOVA, $F_{(Season)}$ = 4.33, $p < 0.001$; Fig. 5) and across gear types (PERMANOVA, $F_{(Gear\ type)} = 6.94$, $p < 0.001$; Fig. 5). During both seasons, fish communities caught in minnow traps appeared to be most dissimilar from fish communities caught using electrofishing (PCOA; Fig. 5). Similarity percentage analysis revealed that differences in key species contributing to the overall dissimilarity across gear types were as follows during the summer: minnow traps were more successful

During the winter: minnow traps caught more round goby and brown bullhead than Windermere traps and electrofishing (Table 2).

In both seasons, fish communities caught within each gear type were identifiable by the presence of a number of indicator species (see Supplementary Table 5). In the summer, fish communities caught using minnow traps were identifiable by only round goby; Windermere trap fish communities were identifiable by brook stickleback, rudd, and spottail shiner; electrofishing fish communities were only identifiable by gizzard shad. In the winter, minnow trap fish communities were once again identifiable by round goby; electrofishing fish communities were identifiable by smallmouth bass; no indicator species were detected for fish communities caught using Windermere traps.

4. Discussion

Our study explored how certain gear types (minnow traps, Windermere traps, and electrofishing), when used in shallow aquatic systems, differed in their ability to catch fish, their efficiency, and their selectivity, with a specific focus on how these parameters are further modulated by seasonality. We found that minnow traps were more successful at catching fish (per unit effort) in comparison to Windermere traps and electrofishing, but only during the winter. This was contrary to our predictions, as we expected active gear type (e.g., electrofishing) to be more successful at catching fish, especially during the winter, when fish are generally less active and therefore the rate at which traps are encountered would presumably be lowest (Hurst, 2007; McMeans et al., 2020). The increased capture success of minnow traps during the winter may be due to more fish occupying lower depths (i.e., beyond the range of electrofishing from a boat), perhaps even burrowing under substrate during colder months of the year. As such, fish during the winter may encounter benthic-oriented sampling gear types (e.g., minnow traps)

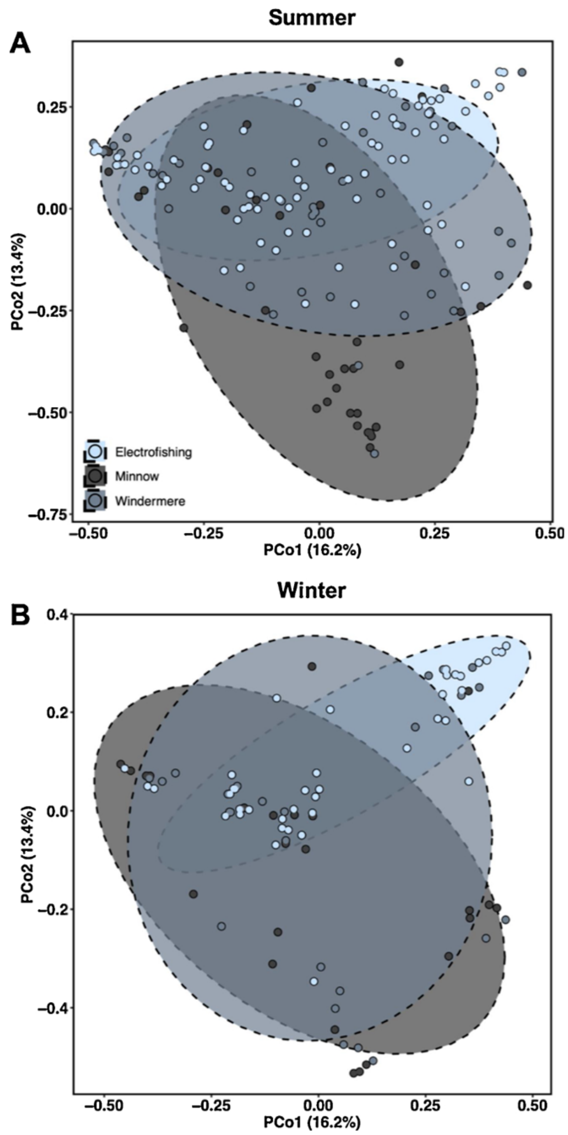

Fig. 5. Principal coordinate analysis (PCoA) ordination output of fish community compositions with 80 % confidence ellipses overlaid on each gear type in the **(A)** summer and **(B)** winter.

get more dispersed—possibly not reaching fish that concentrate at lower depths during the winter (SFCC, 2007; Larocque et al., 2020). Likewise, benthic fishes may be more likely to escape an electrofishing shock by burrowing into the substrate, perhaps making their relatively low abundance in our electrofishing samples less surprising. Moreover, responses of fish to an electrofishing shock are modified by temperature, where fish are less responsive in the cold, making their capture by electrofishing more difficult than during the summer (SFCC, 2007). Furthermore, the addition of bait (food) in our passive gear types may have contributed to the increased capture success of fish in minnow traps during the winter, as food is often scarce and limited during the winter (Shuter et al., 2012; Speers-Roesch et al., 2018).

Unlike abundance, the richness and diversity of species we collected were highest in electrofishing hauls, although this was only the case during the summer. Minnow traps and Windermere traps are typically considered to be more selective than electrofishing, as they are both and fish with a wider range of body sizes than when using either minnow traps or Windermere traps (see Supplementary Table 4). The entry hole size in minnow traps and Windermere traps results in size-selectivity, thereby limiting fishes of larger body sizes (e.g., common carp, northern pike, and longnose gar) from being caught. In support of our results, previous studies have also demonstrated that species richness and species diversity (Shannon-Weiner Index) are often higher in active gear types (e.g., electrofishing and seining) as they are less selective and are therefore able to catch a wider variety of species compared to passive gear types (e.g., minnow traps and Windermere traps; Lapointe et al., 2006).

As predicted, fish community assemblages differed significantly across gear types and between seasons. In both seasons, fish communities caught using minnow traps were most dissimilar to those caught using electrofishing. This was expected as minnow traps are often highly selective for benthic and benthopelagic small-bodied fishes, while electrofishing is less selective overall (Weaver et al., 1993). Minnow trap catches were dominated by benthic species (~53 %) during both seasons in comparison to Windermere trap and electrofishing catches. Additionally, minnow traps were more successful at catching invasive species (a category mostly dominated by round goby). The higher proportion of invasive fishes caught in minnow traps could be because our study sites are highly abundant with round goby (McCallum et al., 2019; Mehdi et al., 2021), a benthic invasive species that was rarely caught in our electrofishing samples. A number of previous studies have used minnow traps to track and monitor the spread and persistence of invasive species, such as round goby (Young et al., 2010; McCallum et al., 2014; Bose et al., 2018; McCallum et al., 2018). Conversely, a study investigating the effects of sampling techniques on round goby population assessments found improved round goby catchability using electrofishing compared to minnow traps (Brandner et al., 2013). It is however important to note that the minnow trap deployment period used by Brandner et al. (2013) was only 20 min long, while in our study, traps were left out for 24 h, suggesting that deployment time length plays a significant role in catch success. Additionally, round goby are nocturnal feeders; hence, overnight deployment is likely to yield higher catch success than daytime deployment (Johnson et al., 2008). Moreover, deeper waters (>1.5 m) and impaired water clarity will both act to reduce the efficacy of electrofishing in catching benthic fishes (e.g., round goby and brown bullhead) since these individuals cannot be seen by the netter. Differences in species characteristics, study design, and overall composition reported here highlight the importance of using multiple gear types in fish community and population assessments, as usage of a single gear type might result in under- or overestimation of certain species, possibly leading to inaccurate community and/or population estimates (Portt et al., 2006; Pope et al., 2010). This may be especially critical in studies monitoring species of concern (e.g., invasive species or endangered species), where inaccurate population estimates can be consequential for the management decisions they are meant to inform.

While using multiple gear types seems ideal when conducting fish community and population assessments, it is worth noting that employing multiple gear types can be time consuming and labour- and cost-intensive, especially during the winter, when working conditions are often sub-optimal. In our study, we found that when all three gear types were combined, the species discovery rate was highest, implying that more unique species are captured and a more holistic and accurate view of the community can be achieved when multiple gear types are used simultaneously. This was further highlighted as each gear type demonstrated targeted selectivity for certain species. In addition, gear type species selectivity differed between seasons, indicating that the species selectivity of different gear types can be modulated by seasonal changes in biotic and abiotic factors. We found that a number of species percentages (SIMPER) analysis showing the contribution of key species to the overall dissimilarity between different gear types. Average A and Average B represent the gear-standardised catch per unit effort e) for each species of the pair of gear types being compared. Only species that contributed > 5% to the overall abundance are shown. Bolded averages indicate significant differences ($p < 0.05$).

	Summer					Winter				
on	Total dissimilarity	Species	Average A	Average B	Contribution	Total dissimilarity	Species	Average A	Average B	Contribution
		Lepomis macrochirus (Bluegill sunfish)	0.13	0.48	10.8		*Neogobius melanostomus* (Round goby)	**0.37**	**0.04**	**23.47**
		Culaea inconstans (Brook stickleback)	**0.02**	**0.06**	**6.93**		*Ameiurus nebulosus* (Brown bullhead)	**0.15**	**0.02**	**14.07**
v Traps B. Windermere	90.16 %	*Lepomis cyanellus* (Green sunfish)	0.05	0.11	6.71	90.45%	*Lepomis cyanellus* (Green sunfish)	0.11	0.04	10.59
							Lepomis macrochirus (Bluegill sunfish)	0.06	0.04	9.04
							Pimephales promelas (Fathead minnow)	0.03	0.02	5.26
		Neogobius melanostomus (Round goby)	**0.73**	**0.04**	**19.39**		*Neogobius melanostomus* (Round goby)	**0.37**	**0.002**	**20.12**
		Lepomis macrochirus (Bluegill sunfish)	0.13	0.22	10.85		*Lepomis cyanellus* (Green sunfish)	0.1	0.09	14.07
		Carassius auratus (Goldfish)	0.06	0.17	8.1		*Ameiurus nebulosus* (Brown bullhead)	**0.15**	**0.02**	**13.37**
v Traps B. ishing	91.93 %	*Lepomis cyanellus* (Green sunfish)	0.05	0.13	7.6	94.73%	*Lepomis macrochirus* (Bluegill sunfish)	0.06	0.02	6.41
		Ameiurus nebulosus (Brown bullhead)	0.06	0.14	7.49		*Pimephales promelas* (Fathead minnow)	0.03	0.02	5.42
		Catostomus commersonii (White sucker)	0.01	0.13	6.15					
		Pimephales promelas (Fathead minnow)	0.005	0.25	5.9					
		Lepomis macrochirus (Bluegill sunfish)	0.48	0.22	12.03		*Lepomis cyanellus* (Green sunfish)	0.04	0.09	17.35
		Lepomis cyanellus (Green sunfish)	0.11	0.13	9.02		*Neogobius melanostomus* (Round goby)	0.04	0.002	9.09
		Carassius auratus (Goldfish)	0.06	0.17	7.59		*Lepomis macrochirus* (Bluegill sunfish)	0.04	0.02	8
mere Traps B. ishing	89.51 %	*Ameiurus nebulosus* (Brown bullhead)	0.1	0.14	7.55	91.63%	*Perca flavescens* (Yellow perch)	0.02	0.02	6.71
		Pimephales promelas (Fathead minnow)	0.11	0.25	7.03		*Pimephales promelas* (Fathead minnow)	0.02	0.02	6
		Catostomus commersonii (White sucker)	0.04	0.13	6.35		*Ameiurus nebulosus* (Brown bullhead)	0.02	0.02	5.83
		Scardinius erythrophthalmus (Rudd)	**0.09**	**0.05**	**5.7**					
		Micropterus salmoides (Largemouth bass)	0.05	0.09	5.1					

capturing spottail shiner, brook stickleback, and rudd during the summer; in the winter, no species were exclusively captured by Windermere traps. Electrofishing was the optimal gear type for capturing gizzard shad during the summer and largemouth bass during the winter. Such differences in gear type selectivity further highlight the importance of using multiple gear types in fish community assessments, as several species in our study were caught almost exclusively by one gear type. Additionally, the seasonal changes in selectivity shed light on the importance of incorporating fish life history traits and the interactions between biotic and abiotic factors when considering which gear type(s) to use in fish population and community surveys conducted across seasons.

Overall, our study demonstrated that minnow traps, Windermere traps, and electrofishing differ considerably in their catchability, selectivity, and efficiency. We clearly demonstrate how these parameters may be modulated by seasonality, a variable rarely considered in gear type selection research. In the summer, abundance did not vary among gear types, whereas in the winter, minnow traps captured the most fish per unit effort. Fish communities caught using electrofishing were the most species rich and species diverse, but this pattern was only apparent during summer sampling. Furthermore, we observed a high degree of species selectivity within each gear type, where fish communities differed significantly from one another, depending on which gear type was used. In addition, gear types differed in key species caught within each season. Understanding and accurately assessing fish populations and communities during winter is an important yet challenging task. Our findings contribute to the recent focus on characterising seasonal differences (Larocque et al., 2020; McMeans et al., 2020; Mehdi et al., 2021), which is especially important since our current knowledge on winter ecology in higher latitudinal regions is lacking, partly due to the challenges associated with field sampling during that time of year. In conclusion, we recommend the usage of multiple gear types; specifically, a combination of active and passive gears. A combination gear type approach would allow researchers to gain a more holistic and accurate view of the fish community or population surveyed, especially if surveys are conducted across seasons, where gear type selectivity can change drastically.

CRediT authorship contribution statement

Hossein Mehdi: Conceptualization, Methodology, Formal analysis, Investigation, Writing - original draft, Writing - review & editing, Visualization. **Samantha C. Lau:** Investigation, Writing - review & editing. **Caitlyn Synyshyn:** Investigation, Writing - review & editing. **Matthew G. Salena:** Investigation, Writing - review & editing. **Markelle E. Morphet:** Investigation, Writing - review & editing. **Jonathan Hamilton:** Investigation, Writing - review & editing. **Melissa N. Muzzatti:** Investigation, Writing - review & editing. **Erin S. McCallum:** Investigation, Formal analysis, Writing - review & editing. **Jonathan D. Midwood:** Conceptualization, Methodology, Writing - review & editing. **Sigal Balshine:** Conceptualization, Methodology, Writing - review & editing, Supervision, Project administration, Funding acquisition.

Declaration of Competing Interest

The authors report no declarations of interest.

Acknowledgments

We are extremely grateful to the many volunteers, students, technicians, and others who have helped make this project possible. The following, is a list of people (in alphabetical order) that we especially want to thank: Chelsea Airstone, Mark Bainbridge, Shagufta Bi, Jac-

Kalbfleisch, Taylor Luu, Kyle Mataya, Samuel Matthew, Elise Millar, Jessica Miller, Tabitha Mirza, Wes Moir, Ghida Mourtada, Kirsten Nikel, Avani Pathak, Bert Posedowski, Maria Pricop, Emilie Rayner, Victoria Restivo, Hailey Schultz, Nivetha Srikanthan, Tys Theysmeyer, Andy Turko, Ellie Weir, and Eliza Wu. This study was supported by grants from Natural Sciences and Engineering Research Council of Canada (NSERC), Royal Bank of Canada Bluewater, and City of Hamilton Water to S. Balshine, and an NSERC PGS-D to H. Mehdi.

Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.fishres.2021.106016.

References

Bates, D., Mächler, M., Bolker, B., Walker, S., 2015. Fitting linear mixed-effects models using lme4. J. Stat. Softw. 67. https://doi.org/10.18637/jss.v067.i01.

Bernhardt, E.S., Palmer, M.A., 2011. River restoration: the fuzzy logic of repairing reaches to reverse catchment scale degradation. Ecol. Appl. 21, 1926–1931. https://doi.org/10.1890/10-1574.1.

Bose, A.P.H., McCallum, E.S., Raymond, K., Marentette, J.R., Balshine, S., 2018. Growth and otolith morphology vary with alternative reproductive tactics and contaminant exposure in the round goby *Neogobius melanostomus*. J. Fish Biol. 93, 674–684. https://doi.org/10.1111/jfb.13756.

Boston, C.M., Randall, R.G., Hoyle, J.A., Mossman, J.L., Bowlby, J.N., 2016. The fish community of Hamilton Harbour, Lake Ontario: status, stressors, and remediation over 25 years. Aquat. Ecosyst. Health Manage. 19, 206–218. https://doi.org/10.1080/14634988.2015.1106290.

Brandner, J., Pander, J., Mueller, M., Cerwenka, A.F., Geist, J., 2013. Effects of sampling techniques on population assessment of invasive round goby *Neogobius melanostomus*. J. Fish Biol. 82, 2063–2079. https://doi.org/10.1111/jfb.12137.

Breen, M.J., Ruetz, C.R., 2006. Gear Bias in Fyke netting: evaluating soak time, fish density, and predators. North Am. J. Fish. Manage. 26, 32–41. https://doi.org/10.1577/m05-013.1.

Chick, J.H., Coyne, S., Trexler, J.C., 1999. Effectiveness of airboat electrofishing for sampling fishes in shallow, vegetated habitats. North Am. J. Fish. Manage. 19, 957–967. https://doi.org/10.1577/1548-8675(1999)019<0957:eoaefs>2.0.co;2.

Chow-Fraser, P., 2006. Development of the Water Quality Index (WQI) to assess effects of basin-wide land-use alteration on coastal marshes of the Laurentian Great Lakes. Coastal Wetlands of the Laurentian Great Lakes: Health, Habitat and Indicators, pp. 137–185.

Dextrase, A.J., Mandrak, N.E., Barnucz, J., Bouvier, L.D., Gaspardy, R., Reid, S.M., 2014. Sampling effort required to detect fishes at risk in Ontario. Can. Manuscr. Rep. Fish. Aquat. Sci. 3024, 50 p v +.

Diana, C.M., Jonas, J.L., Claramunt, R.M., Fitzsimons, J.D., Marsden, J.E., 2006. A comparison of methods for sampling round goby in rocky littoral areas. North Am. J. Fish. Manage. 26, 514–522. https://doi.org/10.1577/m05-049.1.

Dolan, C.R., Miranda, L.E., 2003. Immobilization thresholds of electrofishing relative to fish size. Trans. Am. Fish. Soc. 132, 969–976. https://doi.org/10.1577/t02-055.

Eakins, R.J., 2018. Ontario Freshwater Fishes Life History Database. Online database, Version 4.84. http://www.ontariofishes.ca.

Elphick, C.S., 2008. How you count counts: the importance of methods research in applied ecology. J. Appl. Ecol. 45, 1313–1320. https://doi.org/10.1111/j.1365-2664.2008.01545.x.

Froese, R., Pauly, D., 2020. FishBase. www.fishbase.org.

Gibson-Reinemer, D.K., Ickes, B.S., Chick, J.H., 2017. Development and assessment of a new method for combining catch per unit effort data from different fish sampling gears: multigear mean standardization (MGMS). Can. J. Fish. Aquat. Sci. 74, 8–14. https://doi.org/10.1139/cjfas-2016-0003.

Great Lakes Water Quality Agreement, 2012. Great Lakes Water Quality Agreement. https://binational.net/2012/09/05/%202012-glwqa-aqegl/.

Hamley, J.M., 1975. Review of gillnet selectivity. J. Fish Res. Board Can. 32, 1943–1969.

He, P., Pol, M., 2010. Fish behaviour near gillnets: capture processes and influencing factors. In: He, P. (Ed.), Behaviour of Marine Fishes. Wiley-Blackwell, Ames, Iowa, USA, pp. 205–236.

Hubert, W.A., 1996. Passive capture techniques. Fisheries Techniques, 2nd edition. American Fisheries Society, Bethesda, Maryland, pp. 157–192.

Hubert, W.H., Fabrizio, M.C., 2007. Relative abundance and catch per unit effort. In: Guy, C., Brown, M. (Eds.), Analysis and Interpretation of Freshwater Fisheries Data. American Fisheries Society, Bethesda, Maryland, pp. 279–325.

Hubert, W.A., Pope, K.L., Dettmers, J.M., 2012. Passive Capture Techniques. Nebraska Cooperative Fish & Wildlife Research Unit – Staff Publications, p. 111.

Hurst, T.P., 2007. Causes and consequences of winter mortality in fishes. J. Fish Biol. 71, 315–345. https://doi.org/10.1111/j.1095-8649.2007.01596.x.

Jähnig, S.C., Lorenz, A.W., Hering, D., Antons, C., Sundermann, A., Jedicke, E., Haase, P., 2011. River restoration success: a question of perception. Ecol. Appl. 21, 2007–2015.

Lagler, K.F., 1978. Capture, sampling and examination of fishes. In: Bagenal, T. (Ed.), Methods for Assessment of Fish Production in Fresh Waters. Blackwell Scientific Publications, Oxford, UK, pp. 7–47.

Lapointe, N.W.R., Corkum, L.D., Mandrak, N.E., 2006. A comparison of methods for sampling fish diversity in shallow offshore waters of large rivers. North Am. J. Fish. Manage. 26, 503–513. https://doi.org/10.1577/m05-091.1.

Larocque, S., Boston, C.M., Midwood, J.D., 2020. Seasonal daily depth use patterns of acoustically tagged freshwater fishes informs nearshore fish community sampling protocols. Can. Tech. Rep. Fish. Aquat. Sci. 3409, 38 p viii +.

Leslie, J.K., Timmins, C.A., 1992. Distribution and abundance of larval fish in Hamilton harbour, a severely degraded embayment of Lake Ontario. J. Great Lakes Res. 18, 700e708 https://doi.org/10.1016/S0380-1330(92)71330-6.

Luo, D., Ganesh, S., Koolaard, K., 2020. Predictmeans: Calculate Predicted Means for Linear Models. R Package Version 1.0.4. https://CRAN.R-project.org/package=predictmeans.

MacKenzie, D.I., Nichols, J.D., Lachman, G.B., Droege, S., Royle, A.A., Langtimm, C.A., 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology 83, 2248–2255. https://doi.org/10.1890/0012-9658(2002)083[2248:ESORWD]2.0.CO;2.

McCallum, E.S., Charney, R.E., Marenette, J.R., Young, J.A.M., Koops, M.A., Earn, D.J.D., Bolker, B.M., Balshine, S., 2014. Persistence of an invasive fish (*Neogobius melanostomus*) in a contaminated ecosystem. Biol. Invasions 16, 2449–2461. https://doi.org/10.1007/s10530-014-0677-2.

McCallum, E.S., Bose, A.P.H., Lobban, N., Marentette, J.R., Pettitt-Wade, H., Koops, M.A., Fisk, A.T., Balshine, S., 2018. Alternative reproductive tactics, an overlooked source of life history variation in the invasive Round Goby. Can. J. Fish. Aquat. Sci. 76, 1562–1570. https://doi.org/10.1139/cjfas-2018-0340.

McCallum, E.S., Nikel, K.E., Mehdi, H., Du, S.N.N., Bowman, J.E., Midwood, J.D., Kidd, K.A., Scott, G.R., Balshine, S., 2019. Municipal wastewater effluent affects fish communities: a multi-year study involving two wastewater treatment plants. Environ. Pollut. 252, 1730–1741. https://doi.org/10.1016/j.envpol.2019.06.075.

McCune, B., Grace, J.B., 2002. Analysis of Ecological Communities. MjM Software Design, Gleneden Beach, Oregon.

McMeans, B.C., McCann, K.S., Guzzo, M.M., Bartley, T.J., Bieg, C., Blanchfield, P.J., Fernandes, T., Giacomini, H.C., Middel, T., Rennie, M.D., Ridgway, M.S., Shuter, B.J., 2020. Winter in water: differential responses and the maintenance of biodiversity. Ecol. Lett. 23, 922–938. https://doi.org/10.1111/ele.13504.

Mehdi, H., Lau, S.C., Synyshyn, C., Salena, M.G., McCallum, E.S., Muzzatti, M.N., Bowman, J.E., Mataya, K., Bragg, L.M., Servos, M.R., Kidd, K.A., Scott, G.R., Balshine, S., 2021. Municipal wastewater as an ecological trap: effects on fish communities across seasons. Sci. Total Environ. 759, 143430 https://doi.org/10.1016/j.scitotenv.2020.143430.

Murphy, B.R., Willis, D.W. (Eds.), 1996. Fisheries Techniques, 2nd edition. American Fisheries Society, Bethesda, Maryland.

Nikel, K.E., McCallum, E.S., Mehdi, H., Du, S.N.N., Bowman, J.E., Midwood, J.D., Scott, G.R., Balshine, S., 2021. Fish living near two wastewater treatment plants have unaltered thermal tolerance but show changes in organ and tissue traits. J. Great Lakes Res. 47, 522–533. https://doi.org/10.1016/j.jglr.2021.01.017.

Oksanen, J.F., Blanchet, G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson, G.L., Solymos, P., Stevens, M.H.H., Szoecs, E., Wagner, H., 2019. Vegan: Community Ecology Package. R Package Version 2.5-6. https://CRAN.R-project.org/package=vegan.

Olsen, E.M., Heupel, M.R., Simpfendorfer, C.A., Moland, E., 2012. Harvest selection on Atlantic cod behavioral traits: implications for spatial management. Ecol. Evol. 2, 1549–1562. https://doi.org/10.1002/ece3.244.

Ontario Minstry of Natural Resources, 2015. Invasive Species Act. https://www.ontario.ca/laws/statute/s15022.

Penczak, T., Jakubowski, H., 1990. Drawbacks of electric fishing in rivers. In: Cowx, I.G. (Ed.), Developments in Electric Fishing. Blackwell, Oxford, pp. 115–122.

Pope, K.L., Lochmann, S.E., Young, M.K., 2010. Methods for assessing fish populations. Inl. Fish. Manag. North Am. 28.

Portt, C.B., Coker, G.A., Ming, D.L., Randall, R.G., 2006. A Review of Fish Sampling Methods Commonly Used in Canadian Freshwater Habitats.

R Core Team, 2019. R: A Language and Environment for Statistical Computing. Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

Reynolds, J.B., 1996. Electrofishing. In: Murphy, B.R., Willis, D.W. (Eds.), Fisheries Techniques, 2nd edition. American Fisheries Society, Bethesda, Maryland, pp. 221–253.

Rudstam, L.G., Magnuson, J.J., Tonn, W.M., 1984. Size selectivity of passive fishing gear: a correction for encounter probability applied to gill nets. Can. J. Fish. Aquat. Sci. 41, 1252–1255.

Ruetz, C.R., Uzarski, D.G., Krueger, D.M., Rutherford, E.S., 2007. Sampling a littoral fish assemblage: comparison of small-mesh fyke netting and boat electrofishing. North Am. J. Fish. Manage. 27, 825–831. https://doi.org/10.1577/m06-147.1.

SFCC, 2007. Fisheries Management SVQ Level 2: Catch Fish Using Electrofishing Techniques, p. 44.

Shuter, B.J., Finstad, A.G., Helland, I.P., Zweimüller, I., Hölker, F., 2012. The role of winter phenology in shaping the ecology of freshwater fish and their sensitivities to climate change. Aquat. Sci. 74, 637–657. https://doi.org/10.1007/s00027-012-0274-3.

Smith, L.A., Chow-Fraser, P., 2010. Impacts of adjacent land use and isolation on marsh bird communities. Environ. Manage. 45, 1040e1051 https://doi.org/10.1007/s00267-010-9475-5.

Speers-Roesch, B., Norin, T., Driedzic, W.R., 2018. The benefit of being still: energy savings during winter dormancy in fish come from inactivity and the cold, not from metabolic rate depression. Proc. R. Soc. B Biol. Sci. 285 https://doi.org/10.1098/rspb.2018.1593.

Strickland, T., Fisher, L., Korleski, C., 2010. Methods for Assessing Habitat in Lake Erie Shoreline Waters Using the Qualitative Habitat Evaluation Index.

Taft, B., Koncelik, J.P., 2006. Methods for Assessing Habitat in Flowing Waters: Using the Qualitative Habitat Evaluation Index (QHEI). Ohio EPA technical bulletin.

Thomasen, S., Chow-Fraser, P., 2012. Detecting changes in ecosystem quality following long-term restoration efforts in Cootes Paradise Marsh. Ecol. Indic. 13, 82–92. https://doi.org/10.1016/j.ecolind.2011.04.036.

Weaver, M.J., Magnuson, J.J., Clayton, M.K., 1993. Analyses for differentiating littoral fish assemblages with catch data from multiple sampling gears. Trans. Am. Fish. Soc. 122, 1111–1119.

Wilson, A.D.M., Binder, T.R., McGrath, K.P., Cooke, S.J., Godin, J.J., 2011. Capture technique and fish personality: angling targets timid bluegill sunfish, *Lepomis macrochirus*. Can. J. Fish. Aquat. Sci. 68, 749–757.

Winger, P.D., Eayrs, S., Glass, C.W., 2010. Fish behaviour near bottom trawls. In: He, P. (Ed.), Behaviour of Marine Fishes. Wiley-Blackwell, Ames, Iowa, USA, pp. 205–236.

Young, J.A.M., Marentette, J.R., Gross, C., McDonald, J.I., Verma, A., Marsh-Rollo, S.E., Macdonald, P.D.M., Earn, D.J.D., Balshine, S., 2010. Demography and substrate affinity of the round goby (*Neogobius melanostomus*) in Hamilton Harbour. J. Great Lakes Res. 36, 115–122. https://doi.org/10.1016/j.jglr.2009.11.001.

Zajicek, P., Wolter, C., 2018. The gain of additional sampling methods for the fish-based assessment of large rivers. Fish. Res. 197, 15–24. https://doi.org/10.1016/j.fishres.2017.09.018.