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Effectiveness of shallow water habitat remediation for improving fish habitat in a large temperate river

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ABSTRACT

Loss of shallow water riparian zones in the St. Clair River has reduced availability of nursery areas and refuge for fishes. To remediate habitat losses and provide fish nursery areas, five remediation projects were carried out along the river's United States bank from 2012 to 2014, replacing seawalls with sloping banks and adding in-stream structure (e.g., root wads and boulders). Project evaluation is necessary to determine success, however there is no standard sampling protocol for shallow habitat in large rivers, especially when both adults and juvenile fishes should be targeted. Therefore, to assess remediation effectiveness and suggest appropriate sampling techniques for large river shorelines, we employed a multi-gear sampling strategy targeting multiple fish species and life history stages at five shoreline remediation and four control sites. We collected juvenile fishes with minnow traps and backpack electrofishing and adult fishes with gillnets. Poisson models were used to evaluate catch per unit effort (CPUE) differences between remediation and control sites for species of management priority (e.g., game fishes and rare species) and taxonomic groups. Model estimates were then used to calculate proportional abundances and compare species composition between site types. Results indicated that electrofishing CPUEs of Darters, mottled sculpin *Cottus bairdi*, rare threatened and endangered species, and juvenile and adult Centrarchidae were higher at remediation sites than at control sites. Additionally, juvenile Centrarchidae and mottled sculpin had a higher proportional abundance in electrofishing collections at remediation sites than at control sites. In contrast, CPUEs and proportional abundances were similar for all taxonomic and management priority groups of fish collected in minnow traps and gillnets. Electrofishing captured more species and more individuals and is therefore a valuable sampling technique for large river shorelines. Nevertheless, addition of minnow traps and gillnets allowed for a more comprehensive assessment of fish assemblages. Overall, this multi-faceted survey approach demonstrates that shoreline remediation projects were beneficial to recreational and ecologically important species in the St. Clair River.

1. Introduction

Shallow water riparian zones are important habitats within large rivers, used by most riverine fishes during at least one life history stage. These areas function as nurseries for larval fish (Goodyear, 1982; Love et al., 2017; Schiemer et al., 2001, 2002; McDonald et al., 2014), provide juvenile and prey fishes refuge from predators (Copp, 1997; Grenouillet et al., 2000; Lapointe et al., 2007, 2010; Pander et al., 2015), and support primary and secondary production (Junk et al., 1989; Thorp and Delong, 1994). However, river engineering and other anthropogenic influences have degraded these areas in many rivers. Therefore, remediation may be required to re-establish a functional riparian zone (Hughes et al., 2005; Sparks, 1995).

Globally, riparian zones are threatened by a number of factors including shoreline hardening, development, simplification (e.g., straightening), and increased wave energy from boat traffic (Hartig et al., 2011; Liedermann et al., 2014; Schludermann et al., 2014; Strayer and Findlay, 2010). These alterations reduce the area of riparian zone available to fishes through increasing depths, in-filling floodplains and shallow water habitat, and removing natural structure along river banks. Similar to other large rivers, anthropogenic activity has degraded riparian habitat in the St. Clair River, Michigan. Specifically, shallow water areas of the river's riparian zone have been artificially deepened and armored with seawalls to improve waterfront access and navigation (Edsall et al., 1988; Herdendorf et al., 1986). These habitat alterations and corresponding declines of fish populations led to the designation of the St. Clair River as an Area of Concern (AOC) in 1987, with the decline of fish and wildlife populations and loss of fish and wildlife habitat beneficial use impairments (BUIs 3 and 14) listed for the system

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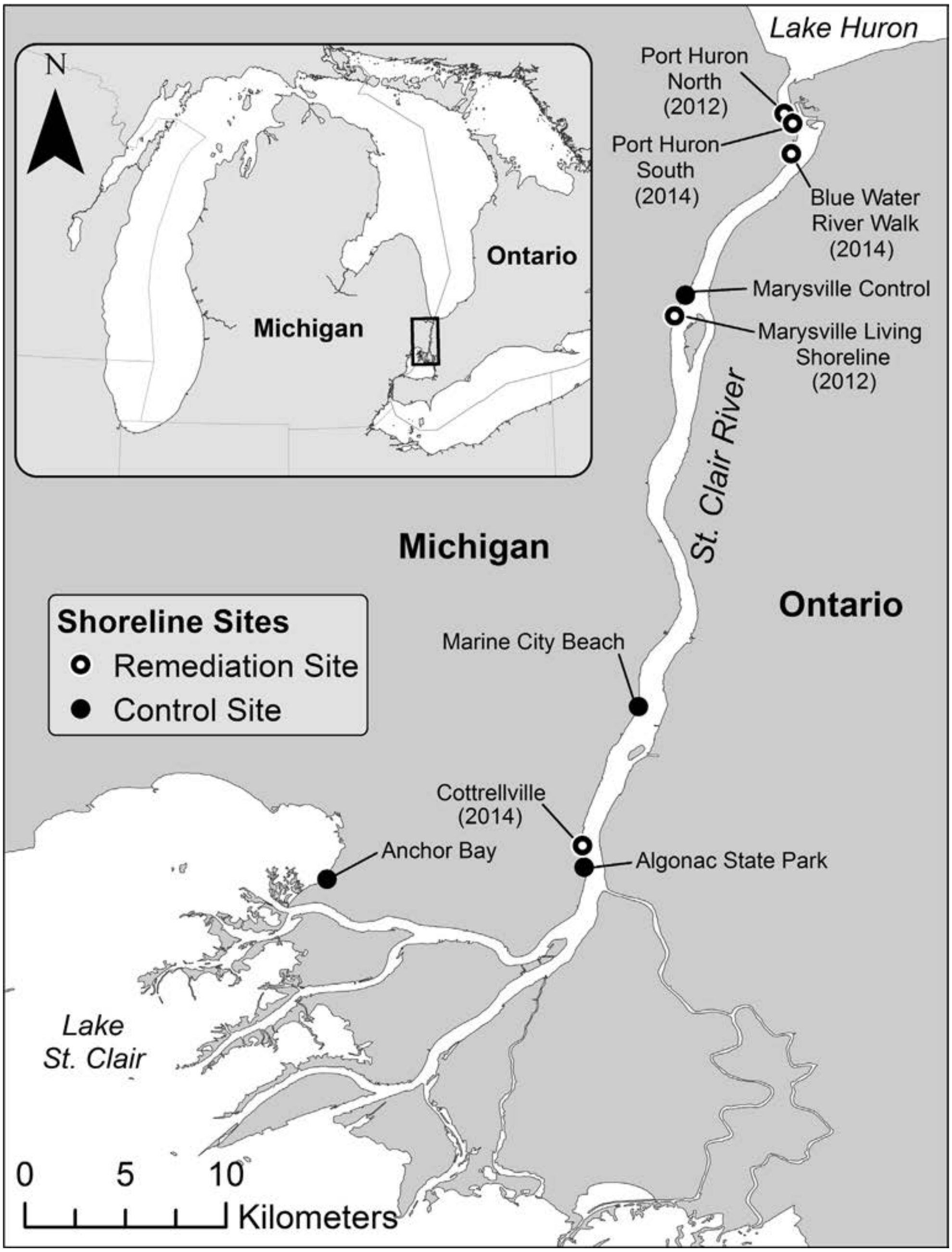

Fig. 1. Locations shoreline remediation and monitoring sites in the St. Clair River. The year remediation projects were completed are included in parentheses below site names.

(GLWQA, 2013; Manny, 2003). Spawning habitat remediation has increased availability of spawning substrates for lithophilic spawning species (Fischer et al., 2018; Prichard et al., 2017) and species from other reproductive guilds have been documented spawning throughout the St. Clair River (Goodyear, 1982). Contemporary monitoring programs have documented the presence of fish larvae for at least 21 genera throughout the river (Pritt et al., 2015), indicating larvae are available to be recruited to shallow water areas within the system, leaving the availability of shallow water nursery habitat as an impediment to the river's fisheries.

To increase the availability and complexity of shallow water riparian habitat, five shoreline remediation projects were completed in the St. Clair River between 2012 and 2014 totaling over 2.1 km of shoreline (Fig. 1). All five remediation projects used shoreline softening techniques, replacing existing vertical seawalls with sloped banks and decreased water depths to mitigate historic losses of shallow water habitat (Detroit River Public Advisory Council, 2014; St. Clair River Bi-National Public Advisory Council, 2013). Additionally, boulders, root wads, and other in-stream structures were added to increase habitat heterogeneity (Fig. 2). A main objective of these remediation projects was to provide nursery areas for larval and juvenile fishes and refuge for small-bodied fishes (St. Clair River Bi-National Public Advisory Council, 2013).

To evaluate the effectiveness of shoreline remediation projects in the St. Clair River and to assess the applicability of these methods for fish habitat remediation in other large waterways, we compared fish abundances at five shoreline remediation projects to four other shallow water riparian areas of the St. Clair River that had not been restored and served as control sites. Our first objective was to quantify differences in catch per unit effort (CPUE) of fishes at remediation and control sites. Our second objective was to then quantify differences in species composition at remediation and control sites. Because the remediation

Fig. 2. Shoreline sampling sites in the St. Clair River. The Port Huron North (A), Port Huron South (B), Blue Water River Walk (C and D), Marysville Living Shoreline (E), and Cottrellville (F) remediation sites included a mixture of shoreline softening (e.g., removal of seawall), native vegetation, and placement of cobble, root wads, and boulders to provide in-stream structure. The top of wave breaks at the Blue Water River Walk (C and D) and Cottrellville (F) sites are highlighted with an arrow in the upper portion of each panel and a bed of emergent vegetation planted at the Marysville Living Shoreline is shown in panel E. In contrast, in-stream structure at the Marysville (G), Marine City Beach (H), and Algonac State Park (J) control sites was primarily limited rip-rap along the banks, although terrestrial vegetation over-hung portions of the bank at the Algonac State Park site and aquatic vegetation was prevalent at the Anchor Bay site (I).

projects were directed towards providing fish habitat, we hypothesized that species CPUEs would be greater at remediation sites than control sites and that species composition at remediation sites would have higher proportional abundances of rare, threatened, and endangered species. Lastly, our third objective was to assess the function of remediation sites as nursery habitats. We hypothesized that remediation sites would provide better nursery habitat than non-remediated sites, predicting higher CPUE of juvenile fishes at remediation sites.

2. Methods

2.1. Study area

The St. Clair River comprises the upper 65 km of the St. Clair-Detroit Rivers System (SCDRS), which connects lakes Huron to the downstream Erie (Fig. 1). The SCDRS forms the southernmost portion of the Michigan, USA – Ontario, Canada border and is the only Great Lakes connecting channel with unregulated flow (Roseman et al., 2014). Lake Huron is the primary water supply to the St. Clair River and acts as a buffer to precipitation events, minimizing variability in water levels, discharge (Edsall et al., 1988; Anderson et al., 2010), and temperature (Hatcher et al., 1991; Hondorp et al., 2014). Subsequently, discharge through the St. Clair River remains relatively constant with a mean of 5150 m^3/s (Liu et al., 2012) and the river naturally lacks a floodplain. Water depths within the river are variable, ranging up to 27 m and shipping channels are maintained at a 10 m depth minimum (Bennion and Manny, 2011; Edsall et al., 1988). The river maintains a single channel unit until it transitions into Lake St. Clair, where the river delta forms the largest wetland complex in the Laurentian Great Lakes (Albert, 2003).

2.2. Shoreline remediation projects

The five remediation projects we evaluated were all funded through the Great Lakes Restoration Initiative and carried out by local townships. The Blue Water River Walk was the longest remediation site, with 1300 m of shoreline softened, followed by the Marysville Living Shoreline (488 m), Cottrellville (152 m), Port Huron South (146 m), and Port Huron North (92 m). Seawalls were removed at all sites, except Port Huron North, where existing infrastructure (e.g., roads and buried electric cable) prevented grading of the bank and removal of the seawall (Fig. 2). Instead, riprap was added in front of the seawall to create a sloping bank and reduce reflectance of wave energy. The banks of the remediation sites were reinforced with cobble and broken limestone (64–500 mm). The Blue Water River Walk and Cottrellville shorelines were further protected by stacked boulders extending above the water surface and placed along the main channel to deflect and reduce wave energy. Coarse substrates (64–500 mm) were added at most sites to raise bed elevation and decrease water depths. With the exception of Port Huron South, sites were wadeable and the bottom gradually transitioned to the deeper main channel. Root wads and large boulders (> 1 m diameter) provided additional complexity in the near shore areas of all sites, except Port Huron North. All of the sites incorporated plantings of native trees, shrubs, and grasses, but only the Marysville Living Shoreline included patches of emergent vegetation (e.g., *Scirpus* spp.), which were planted at the up- and downstream ends of the site and broken limestone was stacked above the water surface to provide wave breaks to protect the emergent vegetation.

In addition to the five shoreline remediation sites, four control sites were evaluated for comparison with the remediation projects. Control sites were selected based on availability of shallow water areas with public access. This limited our control sites to the Marysville Control site, Marine City Beach, Algonac State Park, and a location near Anchor Bay in the St. Clair River delta (Figs. 1 and 2). The Marysville Control site, Marine City Beach, and Algonac State Park were all a mixture of riprapped bank and sandy beach. Little aquatic or terrestrial vegetation was present at the Marysville Control and Marine City Beach sites, but vegetation at Algonac State Park included a mixture of grasses and shrubs along the bank and some submerged aquatic vegetation. Sand was the dominant substrate at these sites and in-stream structure was limited to riprap. The Anchor Bay site was the most distinct, but as part of a historically important fish nursery area of St. Clair River (Goodyear, 1982), it added to our assessment on the effectiveness of the remediation sites as nursery areas. Aquatic vegetation was prevalent at the Anchor Bay site and a seawall created an abrupt transition from land to water, with a water depth of 1.5 m at the bank.

2.3. Data collection

A two-year evaluation of shoreline remediation projects began in spring of 2015 and was completed in December 2016. Because remediation sites were physically distinct and deployment of many standard gears was unfeasible at some sites, different suites of gears were used at different sites. A non-standard multifaceted sampling strategy was used to target multiple species and life stages of fish. Sampling included collections of juvenile and adult fishes with backpack electrofishing, minnow traps, and gillnets. In August 2015 a single backpack electrofishing survey was conducted at four of the shoreline remediation sites and three additional control sites (Marysville Control, Marine City Beach, and Algonac State Park; Table 1). In 2016 the survey was expanded, to include monthly sampling from May-October. The Port Huron South site was not sampled because depths were too great to safely sample with backpack electrofishing. Backpack electrofishing was chosen over boat electrofishing, because shallow water and a high density of obstructions (e.g., large boulders and woody debris) prevented motorboat access to the shoreline remediation sites. Sites were sampled following the Michigan Department of Environmental Quality (MIDEQ) Qualitative Biological and Habitat Survey Protocols for Wadeable Streams and Rivers (MDEQ, 2000). Sampling began at the downstream end of a site and proceeded upstream until 30 min had elapsed or the end of the site was reached, if fewer than 100 fish were collected and the entire site was not sampled, electrofishing occurred for an additional 15 min. A complete unit of effort was considered to be an hour of electrofishing. Two netters were used to collect fish, with a netter positioned on the left and right of the electrofisher. Fish were identified to species, measured to the nearest mm, and released. If over 30 individuals of a single species were collected, the first 30 were measured and the remainder was enumerated to obtain a total count. Bulk counts were only made for small-bodied species (e.g., Atherinopsidae, Cyprinidae, and darters).

Table 1
Gears used to collect fish in remediation and control sites in riparian zones along the St. Clair River.

Site	Type	Minnow Trap	Backpack Electrofishing	Gillnet
Port Huron North	Remediation	x	x	
Port Huron South	Remediation	x		
Blue Water River Walk	Remediation	x	x	x
Marysville Control	Control	x	x	
Marysville Living Shoreline	Remediation	x	x	x
Marine City Beach	Control		x	
Cottrellville	Remediation	x	x	x
Algonac State Park	Control	x	x	x
Anchor Bay	Control	x		

Small bodied fish, juvenile fish, and mudpuppies (see Appendix A for scientific names) were collected with minnow traps (0.6 cm wire mesh traps 42 cm long and 23 cm diameter with a 2.5 cm opening at each end) at the five shoreline remediation sites. Algonac State Park and Anchor Bay served as control sites and a third control site (Marysville Riprap) was added in 2016. Every other week from 6 May to 16 December 2015 and from 16 March to 1 November 2016 minnow traps were fished overnight in gangs of five, spaced 3.75 m apart on a 15 m setline (Conard, 2015; Manny et al., 2014). Traps were baited with cloth bags containing about 28 g of cheese. Each bi-weekly collection was considered a sample unit and one 12 h set of a five trap gang was considered to be a single unit of effort. Occasionally, traps were tampered with (e.g., moved from a sample location) or opened upon retrieval and data from these traps were excluded from enumeration of fish and calculation of effort. Fish were identified to species, measured to the nearest mm, and released.

Adult fishes were also targeted monthly from April to August and October 2016 with monofilament experimental gillnets, with 2.5, 3.8, 5.1, 6.4, 7.6, and 10.2 cm mesh panels 7.6 m long and 1.8 m high. Sampling in 2016 occurred at three remediation sites and one control site (Algonac State Park) where a row boat could be deployed from shore (Table 1). Pilot surveys were also conducted at three sites in September 2015 and two sites in November 2015. Nets were deployed by boat at sunset and retrieved the following morning and all fish were processed following the previously described protocol. Because the same gillnets were used for the study, a single unit of effort was considered to be 12 h of soak time. Similar to minnow trap and backpack electrofishing collections, fish were identified to species, measured to the nearest mm, and released.

2.4. Data analysis

Differences in the number of individuals of a species or group of species ($N_{i,j}$) between remediation and control sites were assessed with generalized linear models using the Poisson distribution. The Poisson distribution offered two advantages. First, it is the most appropriate distribution for counts and matched the distribution of our fish collection data. Second, the Poisson distribution is mathematically connected to the multinomial distribution (Agresti, 2012; McElreath, 2016; Qian, 2016), which allowed inference on species composition differences between site types. To account for differences in effort across sampling events, sampling effort was included as an offset in the models, which allowed inference to focus on CPUE. Because effectiveness of capturing a given species varied by gear and standardizing effort across gear types is not practical (Quist et al., 2009), each gear type was analyzed separately with the following model:

$$N_{i,j} \sim Poisson(\lambda_{i,j} * effort_{i,j}), \tag{1}$$

$$log_e(\lambda_{i,j}) \sim normal(\mu_{i,j}, \sigma), \tag{2}$$

$$\mu_{i,j} = \alpha_i + \beta_i * \text{Remediation}, \tag{3}$$

where $\lambda_{i,j}$ is the CPUE of species or group i for sampling event j, $\mu_{i,j}$ is mean $\log_e$ CPUE of species or group i and sampling event j, α_i is the mean $\log_e$ CPUE of species or group i at control sites, and β_i is the effect of remediation on species or group i on the $\log_e$ scale. The sum of α_i and β_i is the mean $\log_e$ CPUE of species group i at remediation sites.

The generalized linear models assumed that all sites were randomly selected, that remediation was the only difference between remediation and control sites, and that all sites were constant through time. Remediation sites were chosen based on availability of public land without existing infrastructure that would hinder construction (e.g., buried pipelines) and control sites were selected based on availability of public access to wadeable shorelines, instead of on existing habitat conditions. Although site selection was not truly random, site selection was based on non-biological attributes (i.e., public land) and minimized bias that may have been introduced had sites been selected using biological criteria. Selection of control sites with wadeable shorelines allowed the second assumption to be met, because control sites had similar depth and flow characteristics as remediation sites. The third assumption was met, because the relative stability of flows through the St. Clair River ensured physical structure of sites remained constant through the study period. Lastly, because the remediation projects were conducted by local townships prior to our evaluation and minimal preconstruction evaluations were conducted, our evaluation pertains specifically to the post-construction state.

Species were grouped to mirror fisheries objectives for the SCDRS established by the SCDRS Initiative (scdrs.org; Appendix A). The initiative lists 1) smallmouth bass, 2) yellow perch, 3) Suckers (Catostomidae), 4) rare, threatened, and endangered (RTE) species, and 5) Non-Native species as indicators of system health and function. Higher abundances and proportional abundances of the first four species and groups and declines in Non-Native species were considered beneficial to the system. Species listed as state or federally imperiled, threatened, or endangered were considered RTE species. We also considered burbot an RTE species because it is globally impaired and an important indicator of ecosystem health of the Laurentian Great Lakes and connecting channels (McCullough et al., 2015; Stapanian et al., 2010). Non-Native fishes were considered fishes listed as introduced to the Laurentian Great Lakes (accidently or intentionally) by the U.S. Geological Survey nonindigenous aquatic species database (https://nas.er.usgs.gov/). Other species were grouped by genera if the genera was observed frequently enough to be modeled separately, otherwise, genera were then grouped by family or sub-family, and lastly as Other, if species from a particular family were rarely observed (Appendix A). Where possible, indicator species were included as unique groups in the models and were not pooled with other species within the same genus or family.

To account for ontogenetic shifts in habitat use and response to remediation, species and groups were split into adult and juvenile life stages, using lengths at maturity for regional populations of each species, published by Scott and Crossman (1998) and Trautman (1981). We limited this distinction to larger-bodied species (Appendix A). Small-bodied species (e.g., Cyprinidae) were not split into adult and juvenile life stages, due to their rapid growth and early age of maturity. However in many instances, either adults or juveniles comprised an overwhelming majority (> 90%) of the catch of a species or group, preventing abundance of both life-stages from being accurately modeled. In these instances, only the dominant life-stage was included in the analysis. This resulted in eleven species groups for backpack electrofishing collections, five for minnow trap collections, and three for gillnet collections.

Bayesian generalized linear models (Appendix B) for each species group and gear type were fitted using the jagsUI package in R 3.3.2 (Kellner, 2016). Models were run with three chains for 20,000 iterations with the first 5000 iterations discarded as the burn-in phase and 10% of the remaining 15,000 iterations kept for inference. Non-informative priors were used for all parameters to reflect our naïveté about model parameters. Model performance was assessed using trace plots to ensure model convergence and the Bayesian p-value to verify model fit (Gelman et al., 1996). Because models are less likely to fit the tail ends of the data distribution as well as they fit the mean, we calculated a Bayesian p-value for the 5th and 95th percentiles following Qian (2016). Values near 0.5 indicate good fit and values near 0 or 1 indicate poor fit. Poor model fit generally arose when a species or group was observed too infrequently to be accurately modeled, therefore if model fit was poor for a species or group captured with a given gear, that species or group was pooled with other similar species following the methods previously described. After verifying model fit, the 95% credible intervals of β_j were used to determine if CPUE of a species or group differed between remediation and control sites for a given gear. Parameter estimates with 95% credible intervals that did not overlap zero were considered to be significant.

Using the connection between the Poisson and multinomial distributions, CPUE of species or group i for site type k estimated from the Poisson models was converted to proportional abundance (PA) through:

$$PA_{i,k} = \lambda_{i,k} / \sum_{i=i}^{n} \lambda_{i,k} \tag{4}$$

Estimates of proportional abundance of species or group i at site type k were obtained by dividing $\lambda_{i,k}$ for each model iteration by the summation of all $\lambda_{i,k}$ from that iteration. For control sites, λ_i was equal to $\exp(\alpha_i)$ and λ_i at remediation sites was equal to $\exp(\alpha_i + \beta_i)$. Lastly, if α_i and β_i are correlated (e.g., high estimates of α_i correspond to high estimates of β_i) the difference between the proportional abundances derived from the estimates may be significant, but the 95% credible intervals of the posterior estimates may still overlap (McElreath, 2016; Qian, 2016). Therefore, to focus inference specifically on the difference between proportional abundance at remediation and control sites, we calculated the difference between the respective estimates for each iteration and calculated the median and 95% credible interval for that difference. Differences with 95% credible intervals that excluded zero were considered to be significant.

To determine if juvenile fishes were more abundant at remediation sites, we subset our data to include only native juvenile fishes (Appendix A) and summed juvenile counts across species for each sampling event. This approach placed focus on the total abundance of juvenile fishes and accounted for juvenile fishes that could not be included in the species or group specific models. Generalized linear models using the Poisson distribution (equations (1)–(3)) were used to determine if CPUE of juvenile fishes differed between site types, for each gear. The same model conditions (e.g., number and length of chains) used to evaluate species CPUEs were used for the models focused on juvenile fish abundance.

3. Results

3.1. Fish collections

A total of 6625 fish and 54 species were collected during the course of the study (Appendix A). Backpack electrofishing yielded the largest collections of fish (3660 individuals) and species (41). Species unique to backpack electrofishing samples included longnose dace, pugnose minnow, stonecat, and troutperch. Four species considered as RTE (burbot, pugnose minnow, mudpuppy, and river redhorse) were collected with backpack electrofishing. Collections of Non-Native species were dominated by round and tubenose gobies, although these species can be difficult to collect with backpack electrofishing (Jude and DeBoe, 1996), they composed over 16% (587 individuals) of the total catch and were therefore retained for analysis. Adult and juvenile CPUE were only able to be analyzed for Centrarchidae (excluding smallmouth bass) collected with backpack electrofishing and either adults or juveniles of other species or groups were observed too infrequently to accurately model. However, all smallmouth bass and 98% of Catostomidae collected were juveniles and 97% of yellow perch collected were adults. Therefore, analysis and inference on these fishes pertained to the dominant life stage observed.

Minnow trap collections yielded 2706 fish and 28 species between 2015 and 2016. Unique species collected by minnow trapping included central mudminnow, longear sunfish, and pumpkinseed sunfish. Mudpuppy was the only RTE species collected and over 83% of the mudpuppies observed with minnow traps were collected at remediation sites. However, mudpuppies were observed in fewer than 10% of samples and mudpuppy CPUE could not be accurately modeled. Similarly, juvenile yellow perch were observed in fewer than 10% of the samples and were therefore pooled with adults (which comprised 76% of the yellow perch collected with minnow traps) for analysis. Lastly, 91% of the Centrarchidae collected were juveniles, therefore analysis and inference of Centrarchidae CPUE was focused on juveniles.

Gillnet collections yielded 259 individual fish and 32 species. Unique fish species collected by gill-netting included brindled madtom, common carp, freshwater drum, muskellunge, northern madtom, and walleye. Rare, threatened, or endangered species collected in gillnets were brindled madtom, river redhorse, and northern madtom. However, RTE species were rarely observed and were pooled into the Other class for analysis. Although adult and juvenile Catostomidae were evenly represented in gillnet collections, neither life stage was observed frequently enough to warrant separate analysis of CPUE for each life stage, thus both life stages were pooled for analysis. Similar to backpack electrofishing collections, 97% of yellow perch collected were adults and analysis of yellow perch CPUE from gillnet collections focused on the adult life stage.

3.2. CPUE and proportional abundance

Comparing model estimates of electrofishing CPUE at remediation and control sites indicated CPUE for mottled sculpin, Darters, juvenile smallmouth bass, other Centrarchidae adults, Centrarchidae juveniles, and RTE was respectively, 7.2, 4.0, 3.4, 3.7, 4.5, and 12.5 times greater at remediation sites (Fig. 3; Table 2). The β estimates for these species and groups were all greater than zero and had 95% credible intervals that did not contain zero. This pattern in Darters was likely driven by logperch, which composed 82% of the Darters collected, with rainbow darters and Iowa darters making up the remaining 12% and 6% of the Darters collected, respectively. Similarly, higher CPUE of non-smallmouth bass Centrarchidae at remediation sites may have been driven by rockbass, which composed 93% and 72% of the adult and juveniles collected, respectively. Bluegill, green sunfish, and largemouth bass composed the remainder of the non-smallmouth bass Centrarchidae collected with backpack electrofishing. The remaining five species groups (Cyprinidae, Non-Native species, juvenile Catostomidae, adult yellow perch, and Other) had β estimates with 95% credible intervals that overlapped zero, indicating CPUE for these species groups were similar at remediation and control sites.

Differences in proportional abundance between remediation and control sites indicated that mottled sculpin and juvenile Centrarchidae

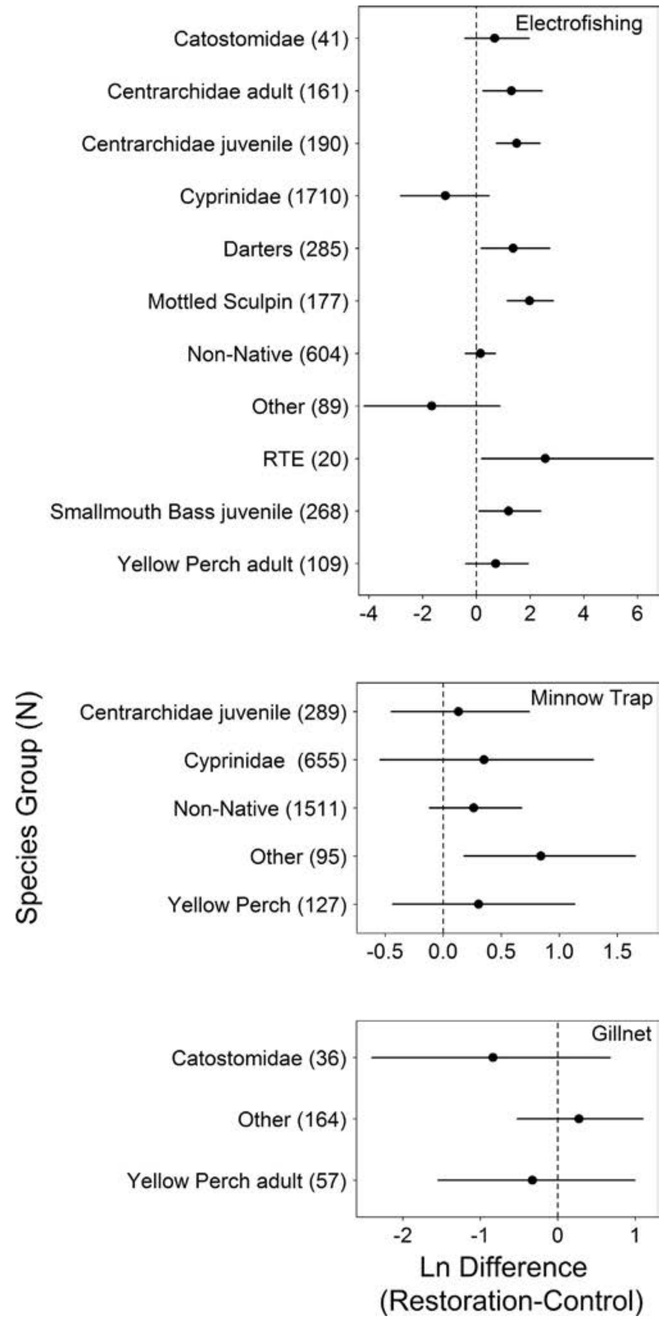

Fig. 3. Log$_e$ difference in CPUE between remediation and control sites for species and groups of species collected with electrofishing (top panel), minnow traps (middle panel), and gillnets (bottom panel). Points show the median estimates and lines represent the 95% credible intervals. Values greater than zero indicate higher CPUE at remediation sites and 95% credible intervals that do not contain zero indicate a significant difference between remediation and control sites. The number of individuals collected for each species or group is shown in parentheses.

(excluding smallmouth bass) composed a larger proportion of the species composition of backpack electrofishing collections at remediation sites compared to control sites (Fig. 4; Table 2), whereas Cyprinidae composed a smaller proportion of the species composition of electrofishing collections at remediation sites. Cyprinidae composed a median of 21% less of the species composition and mottled sculpin and juvenile non-smallmouth bass Centrarchidae both composed a median of 10% more of the species composition at remediation sites, compared to control sites. The 95% credible interval of the difference between RTE

Table 2
Parameter estimates (back-transformed to the normal scale) from Poisson models estimating the CPUE and differences in proportional abundances between shoreline remediation and control sites for species and species groups collected with backpack electrofishing, minnow traps, and gillnets. The median CPUE of species and species groups at control sites and its 95% credible interval are represented by *α*. The median difference in CPUE between remediation and control sites and the 95% credible interval is represented by β, estimates greater than one indicate higher CPUE at remediation sites and 95% credible intervals that exclude one indicate significance differences. The sum of *α* and β is the median CPUE of a species group at remediation sites.

Gear	Species Group	CPUE						Proportional Abundance		
		α	α 95% CI		β	β 95% CI		Difference	95% CI	
Electrofishing	Catostomidae	0.49	0.10	1.27	2.02	0.65	7.46	0.004	−0.02	0.03
	Centrarchidae adult	0.91	0.32	2.12	3.70	1.29	11.14	0.04	−0.01	0.10
	Centrarchidae juvenile	1.63	0.74	3.01	4.52	2.12	10.67	0.10	0.03	0.18
	Cyprinidae	8.33	2.29	27.99	0.31	0.06	1.64	−0.21	−0.51	−0.02
	Darters	0.7	0.20	1.85	3.96	1.19	16.11	0.03	−0.01	0.09
	Mottled Sculpin	0.87	0.38	1.76	7.21	3.18	17.65	0.10	0.04	0.17
	Non-Native	14.14	9.18	21.71	1.19	0.68	2.05	−0.12	−0.31	0.10
	Other	0.18	0.01	0.93	0.20	0.01	2.48	−0.004	−0.03	0.001
	RTE	0.02	0.0004	0.19	12.45	1.34	453.59	0.005	−0.001	0.02
	Smallmouth Bass juvenile	1.36	0.47	3.28	3.36	1.05	11.31	0.05	−0.03	0.13
	Yellow Perch adult	0.98	0.32	2.41	2.03	0.64	6.88	0.01	−0.04	0.05
Minnow Trap	Centrarchidae juvenile	0.28	0.15	0.47	1.14	0.64	2.09	−0.02	−0.10	0.05
	Cyprinidae	0.09	0.03	0.22	1.43	0.56	3.74	0.003	−0.05	0.04
	Non-Native	1.71	1.20	2.38	1.30	0.88	1.94	−0.01	−0.11	0.10
	Other	0.08	0.03	0.16	2.33	1.18	5.24	0.03	−0.01	0.06
	Yellow Perch	0.09	0.03	0.19	1.38	0.63	3.14	0.002	−0.04	0.03
Gillnet	Catostomidae	0.73	0.13	2.24	0.43	0.10	2.04	−0.09	−0.36	0.05
	Other	2.43	1.13	4.87	1.32	0.58	3.05	0.18	−0.08	0.46
	Yellow Perch adult	0.99	0.27	2.66	0.73	0.21	2.83	−0.06	−0.34	0.14

species proportional abundance at remediation and control sites overlapped zero, however, the 90% credible interval did not overlap zero (this was the only instance where the 95% CI included zero and the 90% CI did not), suggesting that RTE species may compose a larger proportion of the species composition of backpack electrofishing collections at remediation sites than at control sites, but with less certainty than what was observed for mottled sculpin and juvenile non-smallmouth Bass Centrarchidae. Regardless, the rarity of these species is reflected in the proportional abundance estimates, which were 0.6% and 0.07% of the composition at remediation and control sites, respectively.

Models estimating CPUE for minnow trap collections indicated that Cyprinidae, Non-Native species, Centrarchidae, and yellow perch CPUEs were similar at remediation and control sites (Fig. 3; Table 2). Only fishes included in the Other category had a β estimate with a 95% credible interval that did not include zero; this estimate indicated that a median of 2.3 more individuals classified as Other species were collected per unit effort of minnow trapping at remediation sites. None of the species groups composed a significantly different proportion of the composition of species collected with minnow traps at remediation or control sites (Fig. 4; Table 2).

Few fish were collected with gillnets and only three groups were able to be considered in the analysis: Catostomidae, yellow perch, and all Other fishes. The β estimates for all three groups had 95% credible intervals that overlapped with zero, indicating their CPUE was similar at remediation and control sites (Fig. 3; Table 2). Similarly, the 95% credible intervals of differences in proportional abundance between remediation and control sites contained zero for all three groups collected with gillnets, indicating species composition of gillnet collections was similar between site types (Fig. 4; Table 2).

3.3. Juvenile fish abundance

A total of 510 native juvenile fish were collected with backpack electrofishing, 320 juvenile fish collected with minnow traps, and 59 juvenile fish collected with gillnets. The β estimates for juvenile fish CPUE were greater than zero for all gear types. However, backpack electrofishing was the only gear where the 95% credible interval of β estimate did not include zero (Fig. 5), indicating juvenile fish CPUE in backpack electrofishing collections was 3.4 times greater at remediation sites, than at control sites.

4. Discussion

Our results indicate that shoreline remediation was beneficial to a subset of species, given Centrarchidae (including juvenile smallmouth bass), Darters, mottled sculpin, and RTE species were more abundant in backpack electrofishing collections at shoreline remediation sites. Moreover, CPUE was not significantly lower at remediation sites for any species group, regardless of gear used, suggesting the remediation sites were not detrimental to any of the species groups evaluated. The remediation projects increased habitat complexity by creating shallow, low velocity habitats with shoreline softening and the addition of cobble substrates and large woody debris. The increased habitat heterogeneity provided some of the components (e.g., cover, current breaks) of large river nursery habitats required by developing fishes, as indicated by greater backpack electrofishing CPUE of juvenile fishes at the restored sites.

4.1. Species responses

Similar to our results, Shirey et al. (2016) documented increased abundance of mottled sculpin and Centrarchidae following remediation of Juday Creek in northern Indiana. Higher CPUE of mottled sculpin and juvenile smallmouth bass at shoreline remediation sites may pertain to the addition of cobble and boulders at remediation sites, since these species prefer rock substrates (Lapointe et al., 2007; Matheson and Brooks, 1983; Orth and Newcomb, 2002; Trautman, 1981). Macroinvertebrate densities and abundances may also increase following in-stream remediation with gravel substrates, which can lead to higher abundances of insectivorous fishes (Mueller et al., 2014). Furthermore, cobble substrates provide spawning substrates for lithophilic spawning fishes, which coupled with shallow, low velocity habitats, can provide nursery areas for developing fish (Lorenz et al., 2013).

Higher CPUE of RTE species in backpack electrofishing collections at remediation sites is encouraging. Two species, burbot and pugnose

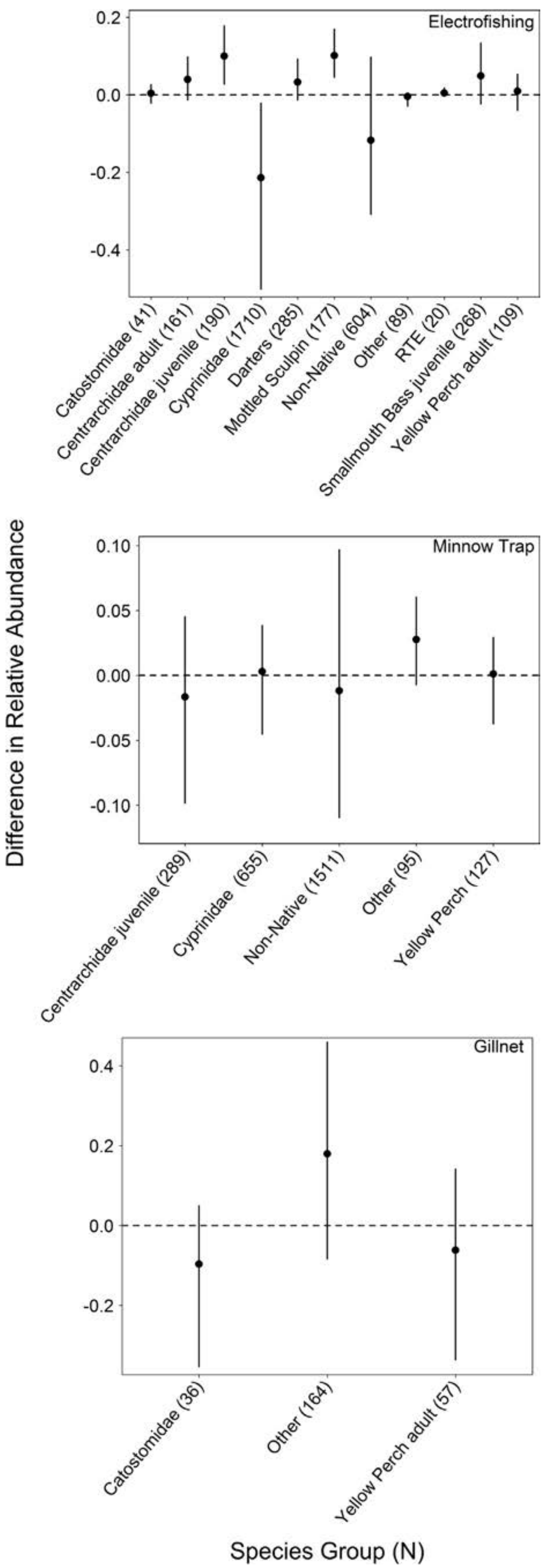

Fig. 4. Proportional differences in proportional abundances between shoreline remediation and control sites for species and species groups collected with electrofishing (top panel), minnow traps (middle panel), and gillnets (bottom panel). Points show the median difference and vertical lines represent the 95% credible interval. Values greater than zero indicate higher proportional abundance at remediation sites and 95% credible intervals that do not contain zero indicate a significant difference in proportional abundance between site types. The number of individuals collected for each species or group is shown in parentheses.

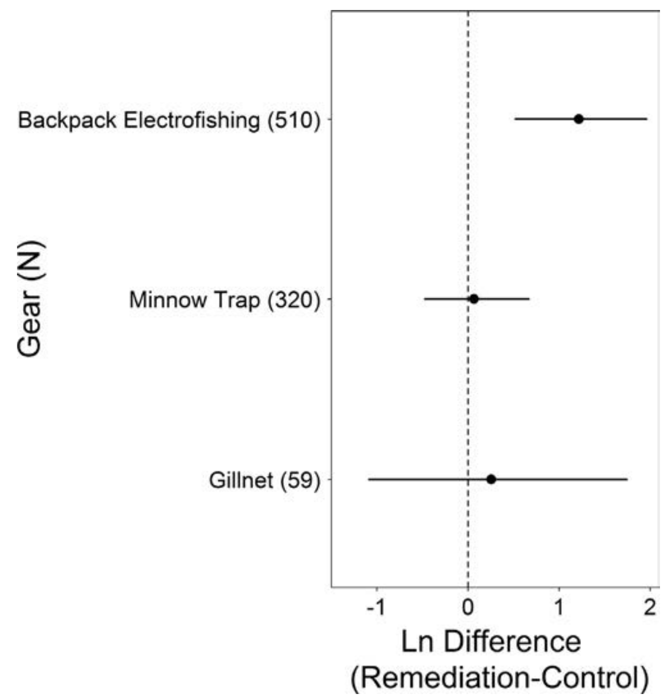

Fig. 5. Log$_e$ difference in juvenile fish CPUE between remediation and control sites by gear type. Points show the median estimates and lines represent the 95% credible interval. Values greater than zero indicate higher CPUE at remediation sites and 95% credible intervals that do not contain zero indicate a significant difference between remediation and control sites. The number of individuals collected with each gear is shown in parentheses.

minnow were only collected at shoreline remediation sites. However, it is important to note that RTE species were infrequently observed during our evaluation and there was a high degree of uncertainty in the backpack electrofishing CPUE estimates of this group. More extensive monitoring would be required to reduce the amount of uncertainty in CPUE estimates of RTE species and determine if catches of RTE species remain higher at shoreline remediation sites.

Other indicator species considered in our analysis (Catostomidae, yellow perch, and Non-Native species) had similar CPUE and proportional abundances at remediation and control sites. Since the Catostomidae group was composed of multiple species and genera with different habitat preferences (Poff and Allan, 1995; Scott and Crossman, 1998; Trautman, 1981), species specific responses to remediation may have been masked by the aggregation of multiple species. Additionally, few Catostomidae were collected, which may have reduced our statistical power to detect differences in CPUE and/or proportional abundance.

Yellow perch and Non-Native species were more commonly collected, however, these species may have been less responsive to habitat remediation. Yellow perch presence at seining sites in the Detroit River was only weakly correlated with habitat metrics (Lapointe et al., 2007), which is supported by our results of similar CPUE and proportional abundance at remediation and control sites. Similar abundances of Non-Native species collected at remediation and control sites with backpack electrofishing could arise from round and tubenose gobies (which comprised 81% and 16%, respectively, of the Non-Native species collected with the backpack electrofishing) being more difficult to detect over the coarse substrates at remediation sites (Jude and DeBoe, 1996). However, minnow trap collections also indicated that round goby abundances (round goby composed 99% of Non-Native species collected with minnow traps) were not significantly different between site types, corroborating our observations with backpack electrofishing. Although round goby may benefit from remediation projects that include the addition of rock substrates which provide cover and spawning substrate (Jude and DeBoe, 1996), round gobies occupy a wide range of habitat types and rapidly colonize new areas (Janssen and Jude, 2001; Jude et al., 1992), thus our study sites may have provided similar functional habitat for this species, regardless of remediation activities.

Groups of species that were more abundant at remediation sites did not always comprise a larger portion of the fish assemblage. Only juvenile Centrarchidae (excluding smallmouth bass), mottled sculpin, and RTE species (at the 90% credible interval level) had significantly higher proportional abundances at remediation sites, suggesting shoreline remediation projects may have been more beneficial for these species than for the rest of the fish community. Jude and DeBoe (1996) cautioned that habitat enhancement projects may disproportionally benefit round goby and other invasive species, however, we did not detect higher CPUE or proportional abundances of Non-Native fishes at remediation sites. Therefore, the shoreline remediation projects we evaluated likely had net positive impacts for native species.

4.2. Gear effectiveness

Differences in CPUE and proportional abundance were only observed from our backpack electrofishing collections, suggesting that the gear and methods used to evaluate remediation projects can also be expected to influence conclusions about fish assemblage responses to remediation. Because capture probability of a given species varies with gear type (Steffensen et al., 2015; Wildhaber et al., 2016), the composition of species collected with a single gear may not accurately represent the true species composition at a site. In this study, several species were observed most often or only observed with a single gear type. For instance, juvenile smallmouth bass were most effectively captured with backpack electrofishing and inference on this species' use of remediation sites would have been inconclusive if only gillnets or minnow traps were used to sample fishes. Likewise, 86% of the mudpuppies collected during our study were collected with minnow traps. Although we were unable to analyze mudpuppy abundances separately from Other species, our collections with minnow traps provide some indication of mudpuppy use of remediation sites. The type of gear may also influence the suite of species collected. Small-bodied and juvenile fishes are more responsive to remediation and habitat changes and gears such as backpack electrofishing, seines, and trawls that can effectively capture small and juvenile fishes may be more effective at detecting trends in species composition and recruitment (Copp, 2010; Fetzer et al., 2017; Goforth and Carman, 2005, 2009; Lorenz et al., 2013). Thus, gears that target species or life-history stages responsive to habitat changes may be more effective for evaluating remediation projects.

Although the suite of species effectively captured with gillnets and minnow traps may have been less responsive to remediation, different results across gear types may also stem from how species were grouped. Differences in CPUE and proportional abundances were most notable for electrofishing collections, where species groupings offered the most resolution of the three gear types. As the number of groups decreased and number of species within a group increased, the CPUE and proportional abundance differences between site types decreased and the amount of uncertainty increased. This is expected, because aggregating species into categories yields a "coarser" response variable that assumes each species within a group responds to the environment similarly (Poff and Allan, 1995; Pyron et al., 2011), an assumption that is more likely to be violated as the number of species within a group increases. Consequently, groups that violate this assumption and contain species that respond to the environment in opposite ways will show little response to differences in site types or ecological gradients. In this sense, the backpack electrofishing surveys were most effective at addressing our objectives of quantifying differences in CPUE and proportional abundance between shoreline remediation sites and control sites, because more species and smaller groups of species could be analyzed, than with collections from minnow traps and gillnets. However, more adult and game fishes were collected in gillnets and minnow traps were more effective at collecting mudpuppies and gobies. Therefore, inclusion of data from all gear types provides a more comprehensive assessment of the fish assemblages using shoreline habitats.

4.3. Project viability and continued value to fish

Although our evaluations indicate these remediation projects may be beneficial to native fishes, long-term viability is influenced by the physical and biological processes acting on these habitats. Changing water levels, ice scour, periphyton growth, and decay of root wads can impact use by fishes and reduce long-term performance of remediation projects (Johnson et al., 2006; Liedermann et al., 2014; Pander and Geist 2010, 2016; Roni et al., 2008). Assessments over a greater temporal scale would provide an opportunity to evaluate long-term performance of remediation projects. However, some of the projects we assessed had been established for a few years prior to being evaluated. Given that the first remediation projects we examined were completed four years prior to our assessment, our results indicate that improvements in shoreline habitat provided functional fish habitat for at least that long.

The timing of our study also provides insight into the longevity of the shoreline remediation projects in the St. Clair River. Preceding our study, two severe winters resulted in near complete ice coverage of Lake Huron in 2013–2014 and 2014–2015 (https://www.glerl.noaa.gov/data/ice/) producing high ice flows in the St. Clair River during the spring thaws. Ice flow through the river did not noticeably alter the structure of the remediation sites and project designs were likely suitable to withstand ice scour. However, high water levels in 2015 and 2016 (https://tidesandcurrents.noaa.gov/stationhome.html?id = 9014087) submerged many protective structures (e.g., wave breaks and shoreline boulder strips) at the remediation sites, allowing waves from passing vessels to reach the bank unabated. Waves produced from passing vessels can temporarily increase lateral water velocities and erosive forces acting on shorelines (Gharbi et al., 2010; Liedermann et al., 2014). Erosion was noticeable at some of the shoreline remediation sites and was severe enough at one location that bank reinforcement with broken limestone was required. Bank reinforcement extending from the high to low water levels and establishment of riparian vegetation could stabilize the river bank and reduce erosion. Additionally, wave breaks such as those installed at the Blue Water River Walk and Cottrellville sites, could help reduce wave energy produced by vessel wakes in large navigable rivers (Liedermann et al., 2014; Schludermann et al., 2014). Despite the potential of ice scour and erosion to degrade shoreline remediation projects, the structure of projects we evaluated remained largely intact and higher CPUE of juvenile fish at remediation sites indicates these projects continued to provide benefits over control sites to developing fishes.

5. Conclusions

Our multi-faceted fish community approach to evaluating shoreline remediation projects allowed us to detect abundance differences between remediation and control sites for multiple species and life history stages. This approach also provided insight about the characteristics of three sampling gears we used in these shallow wadeable habitats. In particular, the greatest number of fish and species were collected with backpack electrofishing in wadeable areas close to shore. Backpack electrofishing was also effective at collecting small and juvenile fishes which are less mobile and more sensitive to habitat changes than larger adult fishes (Goforth and Carman, 2005, 2009), making these fishes good indicators of remediation effectiveness (Lorenz et al., 2013). Juvenile fishes were more abundant in backpack electrofishing collections at shoreline remediation sites and, considering these collections showed multiple species (including two indicators of ecosystem health in the SCDRS, smallmouth bass and RTE species) were more abundant at remediation sites, our evaluation indicates shoreline softening can provide nursery habitat in large rivers and has improved the beneficial use of the St. Clair River shoreline.

Declarations of interest

None.

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Appendices A and B. Supplementary data

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.ecoleng.2018.07.022.

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