Muskegon Lake Post-Restoration Ecological Monitoring

Final Project Report

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Introduction

Muskegon Lake is a 4,150-acre drowned river mouth lake that connects directly to Lake Michigan through a navigation channel. It was designated an Area of Concern (AOC) in 1985 due to ecological problems caused by industrial discharges, shoreline alterations, and the filling of open water and coastal wetlands. Historic sawmill debris, foundry sand, and slag filled 798 acres of open water and emergent wetlands in the AOC. As of 2004, approximately 65% of the shoreline was hardened with wood pilings, sheet metal, and concrete (Steinman et al. 2008), although restoration activities in the past decade (see below) have reduced that percentage to 62%. The shoreline hardening resulted in the loss and degradation of shallow water benthic communities, isolation and fragmentation of coastal wetlands, and the associated degradation of water quality and fish and wildlife populations. Although the benthos has improved since the end of lake-filling practices and wastewater diversion in 1973, shallow water benthic communities remain degraded. Fish and wildlife populations, including lake sturgeon, walleye, white bass, and various species of reptiles, amphibians, and water birds, have been significantly impaired by the loss of habitat. In addition, the introduction of aquatic invasive species has contributed to habitat impairment.

The goal of this project is to evaluate the current status of littoral habitat regions in Muskegon Lake that underwent restoration in 2009-2010 with funding from NOAA through the American Recovery and Reinvestment Act (ARRA), as well as additional sites funded through the Great Lakes Restoration Initiative and NOAA's Restoration Center. We are assessing the condition of macrophyte communities, macroinvertebrate populations, fish populations, and benthic sediments at the sites of former restoration projects at multiple sites throughout the Muskegon Lake AOC.

Methods

Post-restoration ecological monitoring was conducted at two reference and several previous restoration sites to assess the success of fish and wildlife habitat restoration and to provide documentation in support of BUI removals. The sampling sites and the key attributes measured, are listed below:

- NW Reference Site – macrophytes, benthos (i.e., macroinvertebrates), fishes, sediment
- NE Reference Site – macrophytes, benthos, fishes, sediment
- Amoco – macrophytes, benthos, fishes, sediment
- Grand Trunk – macrophytes, benthos, fishes, sediment
- Heritage Landing Circle Bay – macrophytes, benthos, fishes, sediment
- Heritage Landing Scrap Bay – macrophytes, benthos, fishes, sediment
- Kirksey – macrophytes, benthos, fishes, sediment
- Muskegon River Hydrologic Reconnection at Veterans Memorial Park – fishes, water quality
- Bear Creek Hydrologic Reconnection and Wetland Restoration – water quality, benthos, sediment, fishes

Table 1. Selected restoration and monitoring activities associated with Muskegon Lake.

Name	Year(s)	Description
Redirect wastewater discharge into lake	1973-1974	Diversion of municipal and industrial wastewater away from the lake to waste water management system as part of Clean Water Act requirements
Groundwater and soil remediation activities	On-going	Various projects throughout the AOC dealing with chemical contamination
Listing of Muskegon Lake as an AOC	1985	Established Muskegon Lake Public Advisory Council (now Muskegon Lake Watershed Partnership) to identify targets and indicators for BUI removal; coordinate with local, state, and federal partners to develop and implement plans to achieve targets
Remediation of Ruddiman Creek	2006	Great Lakes Legacy Act (GLLA) funding to remove 204,000 lbs of Cr; 126,000 lbs of Pb; 2,800 lbs of Cd, 320 lbs of PCBs; and 260 lbs of benzo-(a)-pyrene
Remediation of Division Street Outfall	2012	GLLA funding to remove 41,000 yd3 of sediment containing mercury and PAHs and restore habitat
Shoreline restoration	2009-2013	NOAA ARRA funding resulted in removal of 208,620 metric tons of unnatural fill, and restoration of 50.2 acres of habitat and 13,073 linear feet of shoreline.
Reconnection/restoration of Bear Lake muck fields	2013-2017	NOAA GLRI funding to reconnect a 39-acre, former celery fields to adjacent Bear Creek to restore habitat, fish passage, remove P-rich sediment and improve water quality
Veterans Memorial Park Habitat Restoration	2015-2019	NOAA funding to improve habitat adjacent to the Muskegon River
Reconnection of lower Muskegon River	2016-2020	NOAA GLRI funding to reconnect 53.5 acres of formerly farmed floodplain to adjacent Muskegon River to restore habitat
Muskegon Lake monitoring program	2003-present	AWRI-GVSU initiated program, funded through an endowment fund at the local Community Foundation, to assess long-term health of lake
Muskegon Lake Observatory	2011-present	GLRI-funded buoy that monitors near real-time water quality in Muskegon Lake

Muskegon Lake Area of Concern Habitat Restoration Project sites

Macrophyte sampling was conducted as consistently as possible with methods previously employed in 2009-2012 (Ogdahl and Steinman 2015). Briefly, transects around Muskegon Lake extend perpendicularly from shore in a lakeward direction at two previously established unrestored reference sites (Northwest [NW] and Northeast [NE] References), four previously established restoration areas

(Amoco, Grand Trunk, Heritage Landing Scrap Bay [hereafter Heritage Landing], and Kirksey), and one newly established transect in a restoration area (Heritage Landing Circle Bay [hereafter Circle Bay]) (Table 2, Fig. 1). Sampling points along a transect were established at the shoreline (0 m), 5 m from shore, every 10 m from 10-100 m, every 25 m from 100-300 m, and every 50 m from the 300 m point until reaching the farthest point of macrophyte growth, as determined by (1) two consecutive sites with no macrophytes present; or (2) the absence of macrophytes at a site greater than 4.5 m deep (5.0 m in 2019), indicating the depth beyond which macrophyte growth becomes light-limited. Double-headed rake tosses allowed us to determine the approximate extent of plant growth and helped discern between extensive bare patches and the actual farthest extent of growth. A transect width of 10 m was chosen to reflect our ability to visually assess the macrophyte community within approximately 5 m of the boat in any direction. Water depth was measured at each site.

At each point along transects, overall plant cover was assigned one of the following ranks: 0 = Bare; 1 = 1–25%; 2 = 26–50%; 3 = 51–75%; or 4 = 76–100%. In addition, all plants within ~5 m radius of each sampling point were identified to species, and species percent abundance (0-100%) was estimated. When plants were too deep to be easily identified from the water surface, a double-headed weighted rake was tossed three times to recover plants for species identification, assigning cover rank, and estimating relative abundance. Voucher specimens for plants that could not be identified were brought back to the lab for identification.

At each transect point, we identified local emergent, submerged, and floating macrophytes, as well as measured relative abundance, overall % cover, and water depth. At randomly-selected points (3-4 per transect), we measured total plant biomass. Biomass samples were collected using a scissor-rake (two rakes attached via chain) for a fixed sampling area of 0.6 m^2; a total of three rake grabs (total area = 1.8 m^2) per point were used unless biomass was very high. Biomass samples were dried at 85˚C for a minimum of 96 hours or until a constant dry weight was achieved. Sampling occurred in mid-to-late summer, when macrophyte growth is most robust.

Sediment was collected at the randomly-selected macrophyte biomass sampling points described above. Organic matter (OM) was analyzed using a well-mixed 5 g subsample, in triplicate, which was dried for 24 hr at 105°C to determine dry mass, then combusted for 1 hr at 550°C to determine the ash-free loss of carbon from the sample. Particle size distributions were measured from the remaining sample by dry sieving (USEPA 2003) and separated into size classes of: >2 mm (gravel/cobble), 1 mm (very coarse sand), 0.5 mm (coarse sand), 0.25 mm (medium sand), 0.125 mm (fine sand), 0.063 mm (very fine sand), and <0.063 mm (coarse silt and smaller). Sediment was sieved for a minimum of 10 min on medium intensity on an Octagon 200 Sieve Shaker and reported as percent sediment dry weight in each size category (% size fraction of total sample).

For macroinvertebrate sampling, three sampling points along each transect were selected where macrophytes and sediment were sampled, representing near-shore, mid-transect, and near the terminus of each transect. At each point, triplicate 1-meter sweeps through the macrophytes were conducted with a D-net. D-net sweeps were not conducted at sites lacking macrophyte coverage. Triplicate D-net samples were kept separate on 1-inch gridded trays, picked for 30-person minutes (i.e., 1 person for 30-min, 2 people for 15 min., etc.), and then the picked macroinvertebrates were combined in the field into one composite sample per point and preserved on site with 95% ethanol (modified from Uzarski et al. 2017). Using taxonomic keys (Merritt et al. 2008) and a Nikon SMZ1270 scope (0.63-8x zoom), macroinvertebrates were identified no farther than family level, except for the following -- Acari (subclass), Diptera adults (order), Oligochaeta (subclass), Chironomids (various tribes), Rissooidea (superfamily), Tanypodinae (subfamily), Turbellaria (class) - and then counted.

Fish surveys were conducted (13-14 and 15-16 August 2019) at five restoration sites along the south shoreline (Circle Bay, Scrap Bay, Grand Trunk, Amoco, and Kirksey) and two reference sites (NW Reference and NE Reference) (Table 2, Fig. 1). Due to difficulty accessing the interior of Circle Bay, sampling locations at Circle Bay (in Heritage Landing) in 2009 and 2010 were approximately 100 m outside Circle Bay. Sampling locations in 2019 were directly inside Circle Bay due to an improvement in boating access to the interior of Circle Bay. The exact locations of fish sampling are reported in Table 3.

Three fyke nets (4-mm mesh) were fished at each site following the protocol of Janetski and Ruetz (2015). Briefly, two fyke nets were set parallel to the shoreline with mouths facing each other and connected at the leads. The third fyke net was placed about 30-50 m from the parallel nets, perpendicular to the shoreline, with the net's mouth facing the shoreline. A detailed description of the fyke nets is provided in Breen and Ruetz (2006), and the type of fyke nets we used select for small-bodied fish (Ruetz et al. 2007). Fyke nets were set during daylight hours with a mean soak time of 23.9 hr at a mean depth of 100 cm (Table 3). Each fish captured was identified to species, measured (total length), and released in the field; however, some specimens were preserved to confirm identifications in the laboratory.

Environmental conditions were measured at each site concurrently with fish sampling. We measured water temperature (°C), dissolved oxygen (mg/L and % saturation), specific conductivity (µS/cm), total dissolved solids (g/L), turbidity (NTU), pH, and chlorophyll-*a* (µg/L) in the middle of the water column using a YSI 6600 multi-parameter data sonde near the mouth of each fyke net. We measured water depth at the mouth of each fyke net and visually estimated the percent cover of submerged aquatic vegetation (SAV) for the length of the lead between the wings of each fyke net.

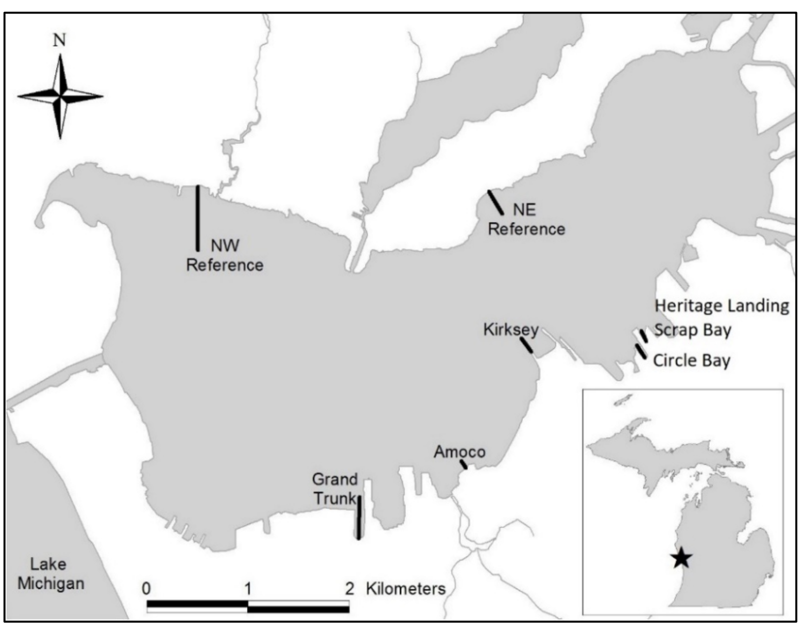

Figure 1. A map of Muskegon Lake Ecological Monitoring transect sites. Veteran's Memorial Park and Bear Creek sampling sites are not shown on this map.

Table 2. Location information for the origin of each sampling transect and restoration details.

Site	Latitude (N)	Longitude (W)	Scheduled Date of Restoration	Actual Date of Restoration	Restoration Type
Northwest Reference	43˚ 14' 50.09"	86˚ 18' 56.67"	N/A	N/A	N/A
Northeast Reference	43˚ 14' 47.96"	86˚ 16' 51.41"	N/A	N/A	N/A
Amoco	43˚ 13' 18.57"	86˚ 17' 04.25"	September 2009	April 2011	Shoreline and underwater fill removal*
Circle Bay	43° 13' 54.00"	86° 15' 43.40"	?	Ongoing	Shoreline and underwater fill removal*
Grand Trunk	43˚ 12' 57.44"	86˚ 17' 49.19"	July 2009	June 2010	Shoreline and underwater fill removal*
Heritage Landing	43˚ 13' 58.33"	86˚ 15' 42.49"	August 2009	April 2011	Shoreline and underwater fill removal*
Kirksey	43˚ 13' 57.58"	86˚ 16' 36.02"	August 2009	October 2010	Shoreline and underwater fill removal*
Muskegon River at Veterans Memorial Park	43° 15' 45.60"	86° 14' 46.10"	2016	2017	Shoreline modification, installation of lunker structures, and removal of water control structure at South Pond**
Bear Creek	43° 16' 04.20"	86° 15' 42.70"	?	April 2017	Dredge and berm removal***

N/A = not applicable
*Fill removal refers to the removal of unnatural fill (i.e., sawmill waste; industrial and/or commercial demolition material, such as broken concrete) at (shoreline) or below (underwater) the ordinary high-water mark.
**Note that due to high Great Lakes water levels, which caused flooding in the park, a temporary barrier was installed to lower water levels in the South Pond in May 2019, which also served as a barrier to fish movement and water exchange with the Muskegon River. A permanent water control structure was installed in December 2019.
***Dredge and berm removal refers to removing phosphorus-rich former farm field sediments and removing Bear Creek berm to hydrologically reconnect restored wetlands to watershed.

Table 3. Date, location (latitude and longitude), soak time, and depth of fish sampling sites in Muskegon Lake. Date is when fyke nets were retrieved. Coordinates are the mean of the three fyke nets. Means are reported ±1 standard error (*n* = 3 fyke nets) for soak time and depth. Site locations are depicted in Fig. 1.

Site	Date	Lat (N)	Long (W)	Soak Time (hr)	Depth (cm)
NW Reference	8/14/2019	43° 14' 49.4"	86° 18' 54.7"	23.6 ± 0.1	96 ± 8
NE Reference	8/16/2019	43° 14' 47.8"	86° 16' 50.8"	23.6 ± 0.2	102 ± 5
Amoco	8/14/2019	43° 13' 14.9"	86° 17' 05.4"	23.9 ± 0.1	97 ± 3
Circle Bay	8/16/2019	43° 13' 54.9"	86° 15' 45.2"	24.1 ± 0.1	92 ± 13
Grand Trunk	8/14/2019	43° 13' 07.2"	86° 17' 46.6"	23.7 ± 0.1	113 ± 12
Heritage Landing	8/14/2019	43° 14' 00.5"	86° 15' 43.2"	24.1 ± 0.1	101 ± 14
Kirksey	8/16/2019	43° 13' 50.7"	86° 16' 30.0"	24.5 ± 0.1	98 ± 6

Veterans Memorial Park sites

Veterans Memorial Park is located on the North Branch of the Muskegon River (Muskegon County, Michigan), which flows into Muskegon Lake and then Lake Michigan, and is located within the Muskegon Lake Area of Concern boundary (Steinman et al. 2008). The park was created on property that was historically wetlands and contains a north pond and south pond (Fig. 2). Fish and water quality sampling were conducted at 11 littoral sites in 2015, 2018, and 2019 (Table 4). Six sites were sampled at the south pond, three sites were sampled at the north pond, and two sites were sampled in Muskegon Lake near where the North Branch of the Muskegon River enters Muskegon Lake (Fig. 2). A stratified random sampling approach was used on the south pond, where the south pond was broken into three main strata (strata #1, 2, and 3 in Fig. 2), and a sampling site was randomly selected (among two approximately equal segments) on each side of the south pond in each stratum. In the north pond, three sampling sites were randomly selected (i.e., among four shoreline segments). The site locations in Muskegon Lake (Fig. 2) were selected in relatively close proximity to Veterans Memorial Park but in areas that would not experience any habitat restoration. The 2015 monitoring served as to evaluate pre-restoration conditions and 2018 and 2019 monitoring served to evaluate post-restoration conditions.

We sampled fish via fyke netting at each site during 7-10 October 2019 for post-restoration monitoring for this project. However, we previously sampled fish via fyke netting at each study site during 5-8 October 2015 (pre-restoration monitoring) and 7-10 August 2018 (post-restoration monitoring). The 2015 and 2018 monitoring was previously reported (see Ruetz and Ellens 2018), but we report some of those data when assessing trends over time. Fyke nets were set during daylight hours and fished an average of 23.75 h (range = 22.20-25.18 h) in 2015, 25.09 h (range = 21.87-26.70 h) in 2018, and 24.79 hr (range = 23.75-26.87) in 2019. Two fyke nets (4-mm mesh) were fished at each site; fyke nets were set with the mouths facing each other and parallel to the shoreline. A description of the design of the fyke nets is reported in Breen and Ruetz (2006), and the type of fyke nets we used tend to select for small-bodied fish (Ruetz et al. 2007). Each fish captured was identified to species, measured (total length), and released in the field; however, some specimens were preserved to confirm identifications in the laboratory.

Environmental conditions were measured at each fish sampling site. We measured water temperature (°C), dissolved oxygen (mg/L and % saturation), specific conductivity (µS/cm), total dissolved solids (g/L), turbidity (NTU), pH, and chlorophyll *a* (µg/L) in the middle of the water column using a YSI 6600 multi-parameter data sonde near the mouth of each fyke net. We measured water depth at the mouth of each fyke net and visually estimated the percent cover of submerged aquatic vegetation (SAV) and emergent aquatic vegetation (EAV) for the length of the lead between the wings of each fyke net. At each site, water was collected by a 1-L grab-sample at mid depth using an acid-washed polyethylene bottle following the protocol of Janetski and Ruetz (2015). Bottles for specific analytes were rinsed with sample water before collection. All samples were stored in the dark, on ice in the field and then processed further upon return to the laboratory. One 250-mL poly bottle was filled with raw water and stored frozen for analysis of total phosphorus (TP). Additionally, 500 mL of water was filtered using a 0.45-µm nitrocellulose filter and analyzed for chloride (Cl), nitrate (NO_3), and soluble reactive phosphorus (SRP). Chloride and nitrate concentrations were determined by ion chromatography on a Dionex ICS-2100. SRP and TP concentrations were determined using a SEAL Analytical AQ2 discrete analyzer.

Figure 2. A map of Veterans Memorial Park (north and south ponds), including the mouth of the north branch of the Muskegon River as well as a portion of Muskegon Lake. The letters around the ponds represent shoreline segments that were selected at random for sampling. Note that the elongated south pond is broken into three strata for sampling. The numbered red points represent approximate sampling locations in Muskegon Lake. The latitude and longitude for each site is reported in Table 4.

Table 4. Latitude (N) and longitude (W) for each sampling site as part of three years of monitoring at Veterans Memorial Park. Coordinates are the mean of two fyke nets at each site. Site locations are depicted in Fig. 2.

		2015		2018		2019	
Location	Site	Lat (°)	Long (°)	Lat (°)	Long (°)	Lat (°)	Long (°)
Muskegon Lake	1	43.25666	86.25440	43.25686	86.25425	43.25637	86.25437
Muskegon Lake	2	43.25478	86.25063	43.25509	86.25033	43.25509	86.25018
North pond	B	43.26365	86.24769	43.26361	86.24771	43.26364	86.24776
North pond	C	43.26426	86.24669	43.26427	86.24743	43.26425	86.24738
North pond	D	43.26362	86.24648	43.26369	86.24651	43.26372	86.24657
South pond	1-B	43.26162	86.24448	43.26176	86.24452	43.26179	86.24458
South pond	2-B	43.26000	86.24223	43.25971	86.24204	43.25963	86.24208
South pond	1-D	43.26194	86.24522	43.26165	86.24503	43.26168	86.24504
South pond	2-D	43.26037	86.24376	43.26047	86.24399	43.26044	86.25218
South pond	3-A	43.25913	86.24117	43.25905	86.24092	43.25907	86.24102
South pond	3-D	43.25887	86.24184	43.25892	86.24196	43.25891	86.24194

Bear Creek Hydrologic Reconnection and Wetland Restoration sites

Sediment organic matter, particle size distribution, and macroinvertebrates were measured at two sites from each of two (East and West) ponds converted to flow-through wetlands, at the downstream end of Bear Creek before flowing into Bear Lake. We previously monitored these sites during pre- and post-restoration studies (Table 1, Fig. 3; Steinman and Ogdahl 2016, Hassett and Steinman 2018). Triplicate ponar grabs were collected and composited into one sample per site. All collected material was returned to the lab and rinsed through a 0.5 mm mesh steel sieve. Retained materials and organisms were preserved in 95% ethanol with rose bengal stain. Macroinvertebrates were sorted from debris in a white enamel pan, aided by bright lights and a 3× magnifying glass, and then stored in 70% ethanol until identification.

Water quality was assessed using the methods for physico-chemistry, TP, and SRP as described above at each pond site prior to sediment and fish sampling with an additional surface water grab sample per site to collect 1 L for chlorophyll *a* (chl) analysis. Chl *a* was vacuum-filtered on a GF/F membrane, frozen at 4°C until extracted, and analyzed on a Shimadzu UV-1601 spectrophotometer (APHA 1992, Steinman et al. 2017).

Fish surveys in the Bear Creek wetlands (21-22 August 2019) were stratified into West and East ponds (Fig. 3; Table 5). Two sites were sampled in each pond for a total of four sites (Fig. 3). Fish were sampled using modified fyke nets following the protocol of Uzarski et al. (2017). Three fyke nets were set at each site. Fyke nets were set individually, spaced at least 20 m apart, and placed in water depths between 20 cm and 100 cm (occasionally fyke nets were set in water slightly deeper than 100 cm due to high Great Lakes water levels). Fyke nets were set with the lead perpendicular to shore if the site was shallow enough for this particular net orientation; at some sites, the lead of each fyke net was set parallel to the shoreline due to a steep water depth gradient (Table 5). Fyke nets were fished overnight with a mean soak time of 22.9 hr (Table 5). Fish were identified, enumerated, measured for total length, and released in the field. A few individuals of certain species (i.e., ones difficult to identify in the field) were euthanized and taken to the laboratory for identification with a dissecting microscope. One fyke net at site A did not fish properly due to a large hole below the water line (likely from a heron or muskrat); therefore, the catch from this net was excluded from analysis (meaning catch results for site A were based on two fyke nets).

Environmental conditions were measured at each site concurrently with fish sampling. Depth was measured at the mouth of each fyke net. Submerged aquatic vegetation (SAV) was visually estimated for the length of the fyke-net lead between the two net wings. At each net, a YSI 6600 V2 multi-parameter data sonde was used to measure water temperature, dissolved oxygen, specific conductivity, total dissolved solids, turbidity, pH, and chlorophyll-*a*. Additionally, a HOBO dissolved oxygen data logger (Onset Company, model U26-001) was installed with one fyke net at each site for a total of four data loggers deployed at the Bear Creek wetlands. The dissolved oxygen loggers were mounted on a fyke net wing, mid water column, and set to measure and log dissolved oxygen concentration and water temperature every 15 min for the duration a fyke net was fished.

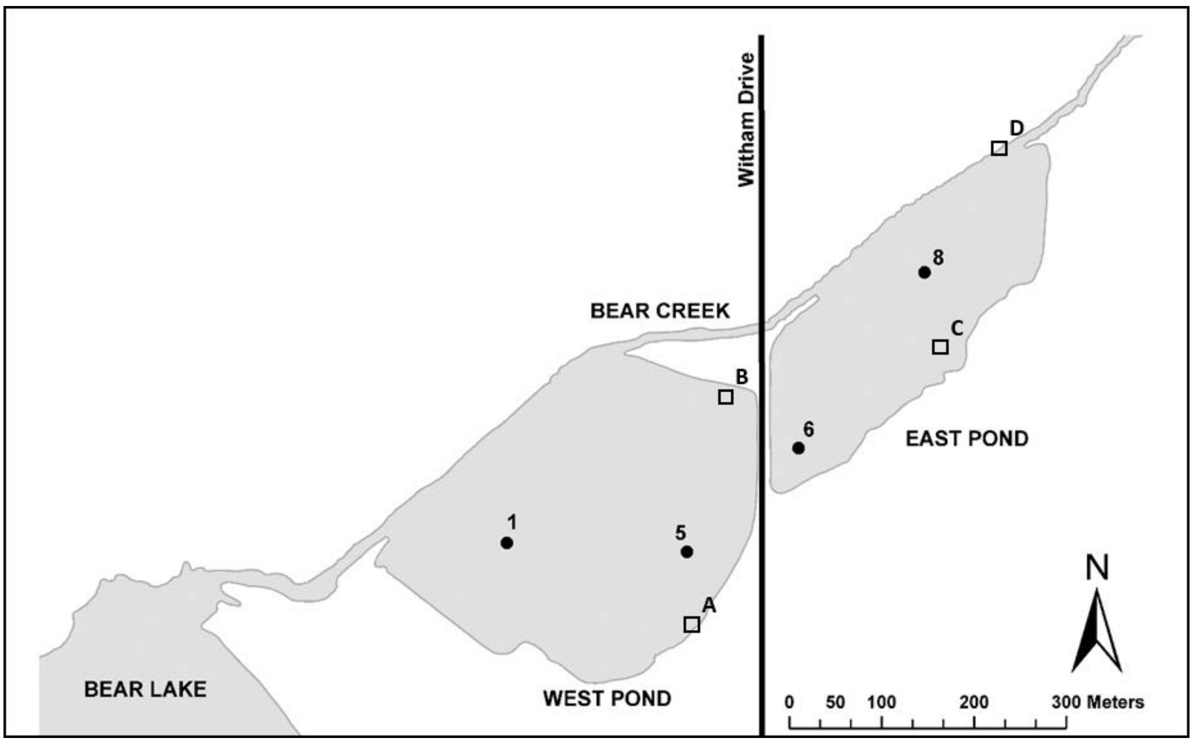

Figure 3. A representative map of Bear Creek west and east pond monitoring sites. Sites 1, 5, 6, 8 (circles) are previously sampled wetland restoration water quality monitoring sites. Sites A-D (squares) are fyke netting locations.

Table 5. Date, location (latitude and longitude), soak time, depth, and the orientation of fyke nets to the shoreline of each fish sampling site in the Bear Creek wetlands. Date is when fyke nets were retrieved. Coordinates are the mean of the three fyke nets. Means are reported ±1 standard error (*n* = 3 fyke nets except at A where *n=2*) for soak time and depth. Site locations are depicted in Fig. 3.

Site	Date	Lat (°)	Long (°)	Soak time (hr)	Depth (cm)	Orientation
A	8/22/2019	43.26473	-86.26297	22.1 ± 0.0	87 ± 4	Parallel
B	8/22/2019	43.26692	-86.26239	22.1 ± 0.1	99 ± 7	Perpendicular
C	8/22/2019	43.26731	-86.25954	23.9 ± 0.1	96 ± 1	Parallel
D	8/22/2019	43.26915	-86.25895	23.5 ± 0.1	88 ± 4	Perpendicular

Data Analysis

Macrophyte biomass within each transect was analyzed by first separately summing the dry mass (g) of all plants collected at each point along a transect and dividing the total dry mass by the area sampled at the point (area sampled = total number of scissor rake grabs × 0.6 m^2) to calculate biomass density at each point as g/m^2. Transect biomass density was then calculated by summing biomass from all sampling points within each transect and dividing by the sum of the sampled points' area. Transect total biomass was calculated by multiplying biomass density by the total area of each transect (total area = transect length × 10 m transect width).

Macrophyte taxon relative abundance was calculated for each transect using a weighted mean as done in previous study years, which incorporated both percent abundance and cover rank in describing the importance of individual taxa in a given transect. In this calculation, the percent abundance of a taxon at a given sampling location was weighted by the cover rank at that location. Consequently, taxa at transect points with high cover ranks contribute more to overall transect mean relative abundance than taxa at points with lower cover ranks. The calculation process for weighted mean taxon relative abundance ($\overline{\mathrm{A}}_W$) is to (1) multiply the percent abundance of a taxon (0-100%) at each sampling location by the cover rank (from 1 to 4) at that location to calculate weighted relative abundance, (2) calculate the sum of weighted relative abundance values for a taxon along the transect, and (3) divide by the sum of cover values for the transect using the formula:

$$\overline{\mathrm{A}}_W = \frac{\Sigma \mathrm{AC}}{\Sigma \mathrm{C}}$$

where *w* stands for weighted, A = taxon percent relative abundance, and C=cover rank. Table 6 illustrates the difference between calculating unweighted vs. weighted mean relative abundance using example values. The use of weighting reduces the abundance of *Vallisneria* but increases the abundance of *Najas*.

Table 6. Hypothetical data from a macrophyte sampling transect for the purpose of explaining weighted means. Mean percent abundance (unweighted) is calculated for each species and compared to weighted mean relative abundance, which accounts for cover rank. Cover ranks: 0 = Bare; 1 = 1–25%; 2 = 26–50%; 3 = 51–75%; or 4 = 76–100%.

Distance from shore	Cover Rank	Species 1	% Abundance	Species 2	% Abundance
5 m	1	*Vallisneria americana*	100	*Najas flexilis*	0
10 m	1	*Vallisneria americana*	100	*Najas flexilis*	0
20 m	3	*Vallisneria americana*	20	*Najas flexilis*	80
30 m	4	*Vallisneria americana*	5	*Najas flexilis*	95
40 m	2	*Vallisneria americana*	0	*Najas flexilis*	100
Mean			45		55
vs.					
Weighted mean			25		75

V. americana = ((100*1)+(100*1)+(20*3)+(5*4)+(0*2))/(1+1+3+4+2) = 25

N. flexilis = ((0*1)+(0*1)+(80*3)+(95*4)+(100*2))/(1+1+3+4+2) = 75

The Coefficient of Conservatism (C) for each species, as determined by the State of Michigan, was applied to each macrophyte species identified at transect sites. C-values range from 0 to 10 and represent the probability that a species will occur within an undisturbed landscape. For example, a species with a C-value of 0 is more likely to be found in highly degraded areas, while a species with C-value of 10 is usually found in higher quality undisturbed areas (Herman et al. 2001). All non-native species were assigned a C-value of 0 (Bourdaghs et al. 2006). A mean C-value was calculated for each transect.

Differences in macrophyte cover, biomass, and sediment OM among sites and years were tested using a Kruskal-Wallis One-Way Analysis of Variance (ANOVA) on Ranks with post-hoc multiple pairwise comparisons using Dunn's Test. Normality was tested using the Kolmogorov-Smirnov test. Two multivariate Principal Component Analyses (PCA) were conducted using multiyear environmental (air temperature [T], organic matter [OM], precipitation [precip], slope, water level [WL], and wind index [WI]) and biological data (biomass cover, biomass total density, & species richness) using whole-transect means from each transect sampled in 2019. PCA input % data were square root-transformed while other input data were log-transformed. Statistical analyses were conducted using SigmaPlot (v.14.0; Systat) except for PCAs, which were conducted using R (v.3.6.2; R Core Team 2019) with the following packages: vegan (Okansen et al. 2019), readr (Wickham et al. 2018), and readxl (Wickham and Bryan 2019).

Macroinvertebrate community composition was measured using Shannon's Diversity Index, separately calculated for each collection method at each collection site as well as for entire transect lengths using the formula:

$$H' = -\sum_{i=1}^{R} p_i * \ln p_i$$

where H' is Shannon's diversity value, richness (R) is the total number of taxa in the dataset, and p_i is the proportion of a given taxon from all taxa in a sample. H'_{max} was calculated as the natural log of H'. Evenness, the relative abundance of all taxa in a given location, was calculated by dividing H'/H'_{max}.

For the Muskegon Lake Area of Concern Habitat Restoration Project sites, we calculated a fish-based index of biotic integrity (IBI) score for each site using an IBI developed by Uzarski et al. (2005) for Great Lakes coastal wetlands that was modified to better represent anthropogenic disturbance (based on land use and water quality) across a gradient of drowned river mouths (Appendix B). A high score suggests a "healthier" ecosystem, whereas a low score suggests a "degraded" ecosystem.

Results – Muskegon Lake

Macrophytes: Environmental factors

All macrophyte transects were characterized by relatively shallow (≤3 m) and consistent water depth followed by a steep drop off, except for the NW Reference site (Fig. 2). This site had a much more gradual slope downward (i.e., gradual increase in water depth) over the length of the transect, and was also substantially longer (650-800 m) than the other sites (100-400 m) (Fig. 2). Transects in 2019 were influenced by record-high water levels in nearby Lake Michigan (and the Great Lakes at large), as NOAA-GLERL's Great Lakes Dashboard (2019) reports that Lakes Michigan-Huron annual depth averaged 177.14 m in 2019, exceeding the long-term average of 176.44 m since 1918, as well as recent averages in 2009-2012 (176.26 m, 176.11 m, 176.04 m, and 175.92 m, respectively). Mean water depth at transect sites increased to different degrees at all sites, with differences between 2012 and 2019 ranging from 0.91-1.64 m and relative increases of 57-186% (Table 7). The maximum depth at which macrophytes were found in our transects increased lake-wide but varied greatly among sites, with NW Reference and Heritage Landing increasing only 0.08-0.29 m (2-7%), whereas the other sites increased from 0.7-1.63 m (22-89%) (Table 7).

Transect length (defined as the last site with macrophytes before two consecutive sites with plant absence or a site greater than 4.5 m deep [5 m in 2019]) at all sites fell within range of previous sampling years and tended to be shorter overall, presumably due to high water level. The new restoration transect at Circle Bay was 175 m, which is longer than the adjacent and previously sampled Heritage Landing site (100 m) and comparable to Kirksey (175 m) (Fig. 2).

Table 7. Mean water depth for transects in Muskegon Lake and the maximum depth at which macrophytes were found along each transect. "--" denotes no sampling in the given year.

	Mean Water Depth					**Max Macrophyte Depth**				
Site	**2009**	**2010**	**2011**	**2012**	**2019**	**2009**	**2010**	**2011**	**2012**	**2019**
NW Ref.	1.09	1.38	1.28	0.93	1.84	3.65	5.30	4.69	4.10	4.39
NE Ref.	1.22	1.28	1.21	1.01	2.65	2.25	2.51	2.86	2.14	3.77
Amoco	--	1.25	1.27	0.76	1.71	--	1.96	2.78	1.52	2.29
Circle Bay	--	--	--	--	2.28	--	--	--	--	4.09
G. Trunk	0.82	0.81	1.06	0.59	1.69	1.80	2.68	4.02	3.24	3.32
H. Landing	2.05	2.14	2.27	1.93	3.03	2.40	2.65	3.50	3.15	3.85
Kirksey	1.07	1.03	1.23	0.77	1.70	1.82	1.33	1.84	1.42	2.64

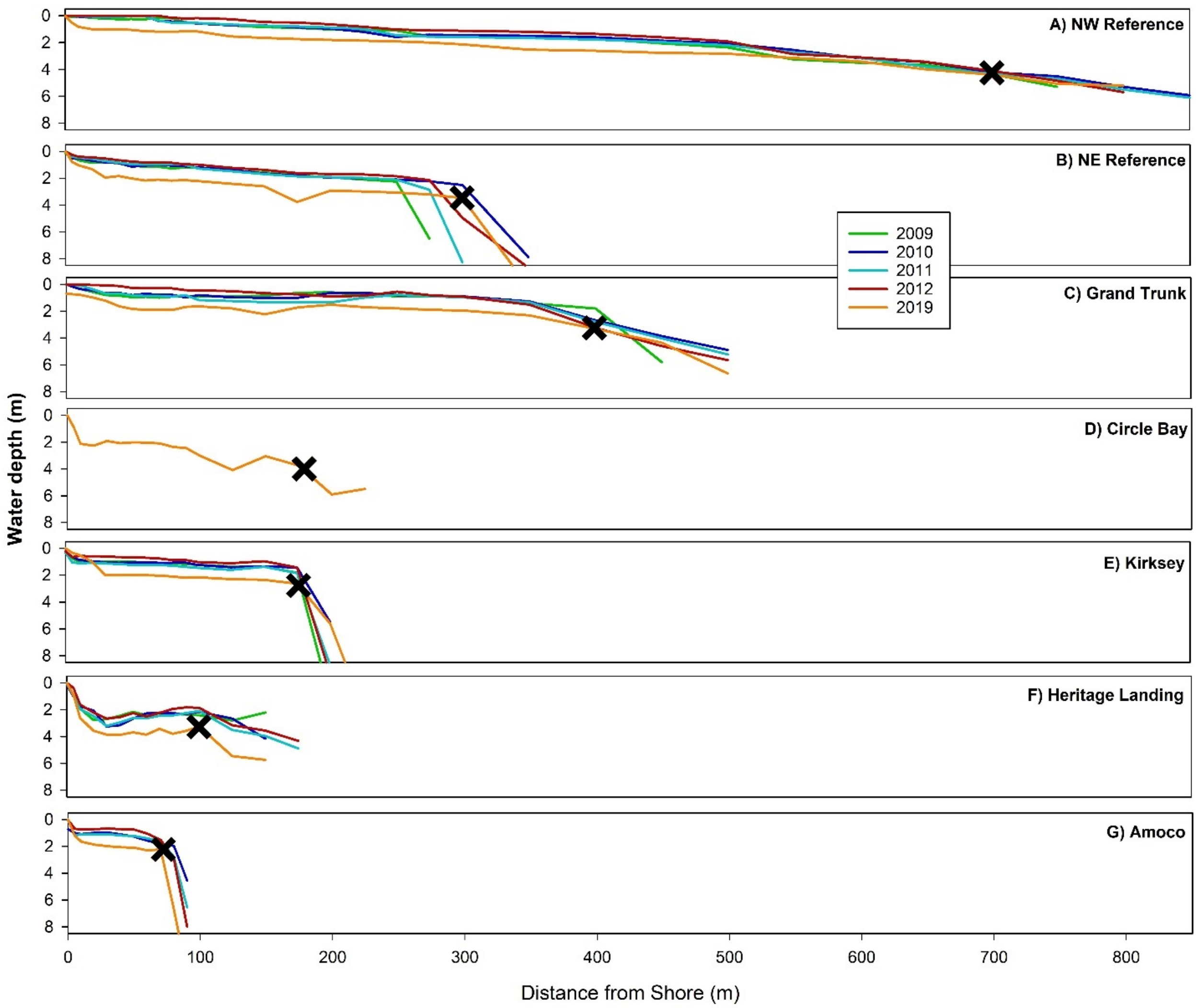

Figure 4. Depth contours at each transect sampled from 2009-2019. "X" indicates the approximate farthest extent of macrophyte growth in the most recent year sampled.

Macrophytes: Biological factors

Mean macrophyte cover rank generally increased from 2012 to 2019, except for Heritage Landing, which decreased from 3.00 to 2.00 and was significantly less than its highest mean rank of 3.58 in 2009 ($p=0.035$; Table 8); the decline may be associated with increased recreational activities in this area since 2012. The largest mean cover increase across all sites in 2019 occurred at Amoco, from 1.64 in 2012 to 3.78 in 2019, and this high 2019 mean value was significantly greater than its lower 2010 and 2011 means ($p=0.038$ and $p<0.001$, respectively; Table 8). The 2019 mean cover rank of 3.40 at Kirksey was statistically higher than its 2009 and 2011 means ($p=0.001$ and $p<0.001$, respectively; Table 8). When comparing all transects in 2019, mean cover rank at Grand Trunk ($p=0.002$), Amoco ($p=0.042$), and Circle Bay ($p=0.012$) were each statistically greater than Heritage Landing. Full summaries of transect cover ranks per sampling site are provided in Appendix Figs. A1-A7.

Mean macrophyte density and total biomass trends were related to each other, so they varied in similar fashions across transects in 2019 (Fig. 5). NW and NE Reference transects showed mixed results, as NW declined from 2012 while NE increased for both macrophyte density and biomass (Fig. 5). Amoco and Kirksey both increased while Grand Trunk and Heritage Landing each decreased from 2012 values (Fig. 5). Rates of change from 2012 to 2019 were proportionately largest at Kirksey (increasing) and Heritage Landing (decreasing).

Table 8. Mean macrophyte cover rank for transects in Muskegon Lake. Cover ranks: 0=Bare, 1=1–25%, 2=26–50%, 3=51–75%, 4=76–100%.

	Mean Cover Rank				
Site	**2009**	**2010**	**2011**	**2012**	**2019**
NW Reference	2.67	2.42	2.33	2.93	3.14
NE Reference	1.56	2.50	2.47	2.68	3.20
Amoco	--	1.70	0.90	1.64	3.78
Circle Bay	--	--	--	--	3.67
Grand Trunk	2.77	3.64	3.57	3.27	3.77
H. Landing	3.58	3.23	2.77	3.00	2.00
Kirksey	1.20	2.82	1.00	2.47	3.40

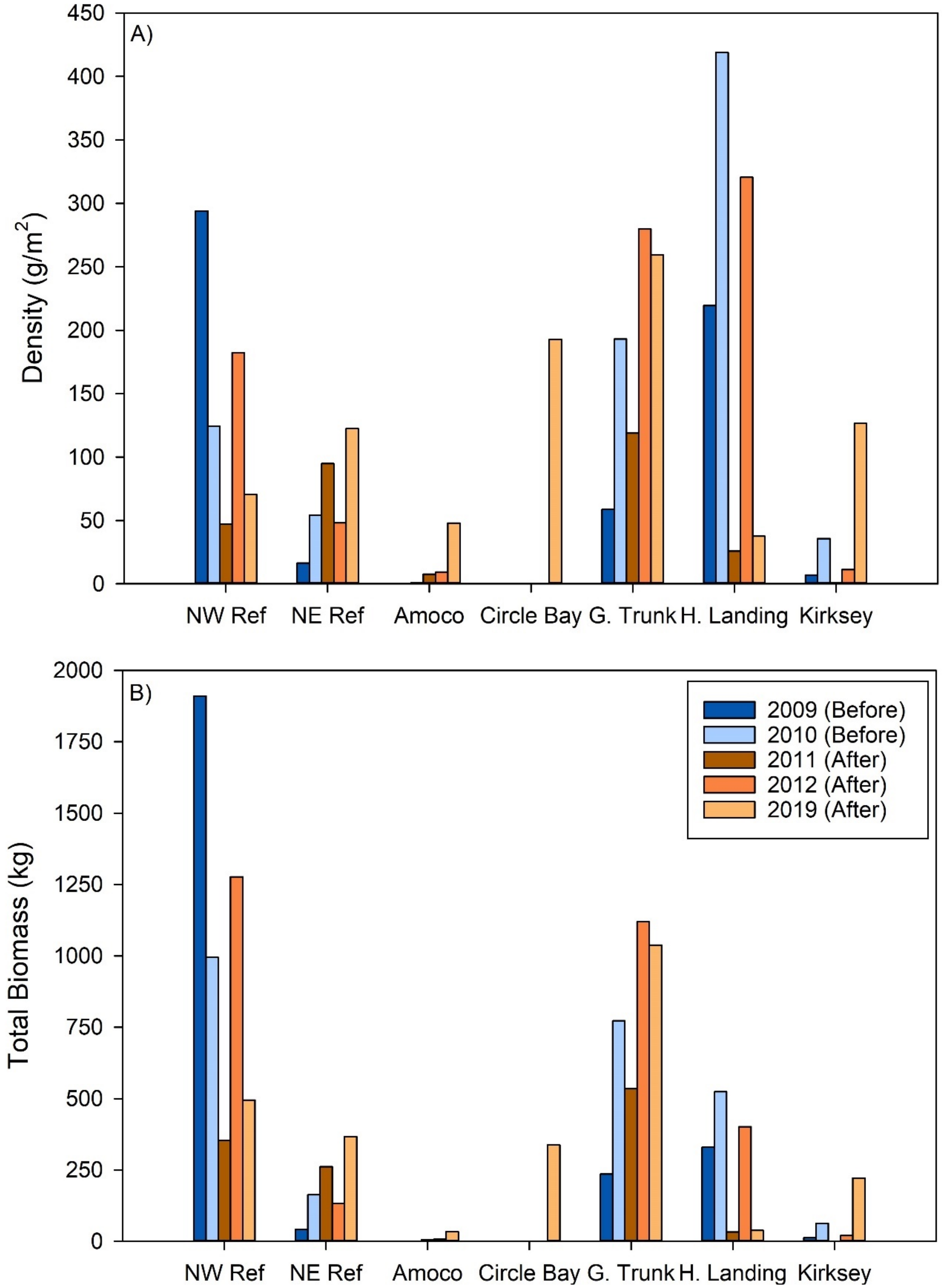

Figure 5. Macrophyte biomass density (g/m^2) (A) and total biomass (kg) (B) at each survey transect before (2009 and 2010) and after (2011, 2012, and 2019) restoration.

Macrophytes: Community composition

Although some populations of larger nearshore emergent macrophytes such as cattails (*Typha* spp.), willows (*Salix* spp.), and sedges (*Schoenoplectus* spp.) observed in previous study years continue to be found at transect points in 2019, emergent vegetation populations appear to have declined overall and given rise to submergent and floating macrophytes, including coontail (*Ceratophyllum demersum*), water celery (*Vallisneria americana*), and pondweed (*Potamogeton* spp.) (Tables 9-13). Macroalgae, including filamentous green algae and charophytes such as *Nitella* spp., accounted for much of the non-emergent vegetation coverage (Tables 9-13). The aquatic invasive charophyte starry stonewort (*Nitellopsis obtusa*) was reported for the first time in AWRI's monitoring for this project, having been found at the NE Ref, Grand Trunk, and Kirksey transects, and at times occurred as a dominant species (Table 13). However, this macroalga was previously documented, first in 2015 (Progressive AE 2018).

Overall, transect communities in 2019 have fewer species and have shifted from emergent macrophytes in 2009-2012 to other submergent and non-macrophyte species (Tables 14-19, Fig. 6). As previously noted, Great Lakes water levels were at record highs in 2019, which is likely connected to the species and community shift. In terms of weighted mean relative abundance, coontail (*Ceratophyllum demersum*; at NW Ref, G. Trunk, Circle Bay, H. Landing) and water celery (*Vallisneria americana*; at NW Ref, Amoco, Kirksey) co-dominated the 7 transects (Table 18). The NE Ref transect was strongly dominated by non-macrophyte algae species and the charophyte *Nitella* spp. (Table 18).

Transect Coefficient of Conservatism (C) values in 2019 were comparable to or slightly higher than previous years, ranging from 3.5 at Kirksey up to 4.8 at Circle Bay (Tables 14-19). Several high-quality species indicated by C-values of 10 (requiring high quality conditions for growth) were found at transects during previous 2009-2012 sampling (Tables 14-17); however, none were observed in 2019 (Table 18). Non-native species are indicated by C-values of 0 and in 2019 included hybrid cattails (*Typha x glauca*), curly-leaf pondweed (*Potamogeton crispus*), Eurasian watermilfoil (*Myriophyllum spicatum*), and purple loosestrife (*Lythrum salicaria*) (Table 18).

Total species richness in 2019 declined substantially from 52 to 26 species at the NW reference transect, due to a decrease of observed emergent macrophyte species, while total richness at other transects and submergent richness overall remained generally consistent with previous years (Table 19, Fig. 7). Across all 2019 transects, emergent species richness ranged from only 1-5 taxa; although we have never observed emergent species at Amoco or Kirksey prior to 2019, GEI did monitor vegetation at these sites immediately following restoration. Emergent wetland plants (bulrushes, sedges, pickerelweed, arrow arum) were planted at the Amoco site, which is now mostly submerged. Our study showed that emergent species ranged from 3-15 at the other transects during pre-restoration sampling in 2009 and 2010 (Tables 14-19, Fig. 7).

Table 9. Dominant taxa based on relative abundance along each of the macrophyte transects in 2009. Loc. = distance from shore in meters. Taxa in bold have Coefficients of Conservation values of zero, indicating non-native or most likely to be found in degraded habitat.

<table>
<tr><th>Loc.</th><th>NW Reference</th><th>NE Reference</th><th>Grand Trunk</th><th>Heritage Landing</th><th>Kirksey</th></tr>
<tr><td>0</td><td rowspan="11">• ***Typha angustifolia***
• ***Phragmites australis***
• *Scirpus americanus*
• *Utricularia vulgaris*</td><td rowspan="2">• *Salix* sp.
• *Scirpus americanus*</td><td rowspan="4">• ***Typha angustifolia***
• ***Lythrum salicaria***
• *Utricularia vulgaris*</td><td rowspan="4">• *Salix exigua*
• *Vallisneria americana*
• *Ceratophyllum demersum*
• *Elodea canadensis*</td><td rowspan="13">• *Vallisneria americana*
• *Potamogeton pectinatus*
• *Potamogeton perfoliatus*
• Macroalgae</td></tr>
<tr><td>5</td></tr>
<tr><td>10</td><td rowspan="3">BARE</td></tr>
<tr><td>20</td></tr>
<tr><td>30</td><td>• *Ceratophyllum demersum*</td><td rowspan="8">• *Ceratophyllum demersum*</td></tr>
<tr><td>40</td><td rowspan="5">• *Chara* sp.
• Macroalgae</td><td rowspan="12">• *Ceratophyllum demersum*
• ***Myriophyllum spicatum***
• *Nymphaea odorata*
• *Potamogeton pusillus*
• ***Potamogeton crispus***
• *Potamogeton perfoliatus*
• *Elodea canadensis*
• *Vallisneria americana*</td></tr>
<tr><td>50</td></tr>
<tr><td>60</td></tr>
<tr><td>70</td></tr>
<tr><td>80</td></tr>
<tr><td>90</td><td rowspan="8">• *Chara* sp.
• *Najas flexilis*
• *Vallisneria americana*</td></tr>
<tr><td>100</td><td rowspan="7">BARE</td></tr>
<tr><td>125</td><td></td></tr>
<tr><td>150</td><td></td><td>BARE</td></tr>
<tr><td>175</td><td></td><td>• *Ceratophyllum demersum*</td></tr>
<tr><td>200</td><td></td><td></td></tr>
<tr><td>225</td><td></td><td></td></tr>
<tr><td>250</td><td rowspan="5">• *Vallisneria americana*</td><td></td><td></td></tr>
<tr><td>275</td><td rowspan="9">• *Vallisneria americana*
• Macroalgae
• *Ceratophyllum demersum*</td><td></td><td></td><td></td></tr>
<tr><td>300</td><td></td><td></td><td></td></tr>
<tr><td>350</td><td></td><td></td><td></td></tr>
<tr><td>400</td><td></td><td></td><td></td></tr>
<tr><td>450</td><td></td><td></td><td></td><td></td></tr>
<tr><td>500</td><td></td><td></td><td></td><td></td></tr>
<tr><td>550</td><td></td><td></td><td></td><td></td></tr>
<tr><td>600</td><td></td><td></td><td></td><td></td></tr>
<tr><td>650</td><td></td><td></td><td></td><td></td></tr>
</table>

Table 10. Dominant taxa based on relative abundance along each of the macrophyte transects in 2010. Loc. = distance from shore in meters. Taxa in bold have Coefficients of Conservation values of zero, indicating non-native or most likely to be found in degraded habitat.

<table>
<tr><th>Loc.</th><th>NW Reference</th><th>NE Reference</th><th>Amoco</th><th>Grand Trunk</th><th>Heritage Landing</th><th>Kirksey</th></tr>
<tr><td>0</td><td rowspan="9">• Typha angustifolia
• Phragmites australis
• Scirpus americanus</td><td>• Typha angustifolia
• Salix sp.</td><td rowspan="3">• Filamentous green algae</td><td rowspan="2">• Typha x glauca
• Lythrum salicaria
• Nasturtium microphyllum
• Sparganium eurycarpum</td><td>• Salix exigua
• Schoenoplectus tabernaemontani</td><td rowspan="12">• Vallisneria americana
• Potamogeton pectinatus
• Potamogeton perfoliatus
• Potomogeton pusillis
• Filamentous green algae</td></tr>
<tr><td>5</td><td rowspan="2">• Najas flexilis
• Filamentous green algae</td><td>• Nymphaea odorata</td></tr>
<tr><td>10</td><td rowspan="14">• Ceratophyllum demersum
• Myriophyllum spicatum
• Nymphaea odorata</td><td rowspan="2">• Ceratophyllum demersum
• Elodea canadensis</td></tr>
<tr><td>20</td><td rowspan="9">• Chara sp.</td><td rowspan="7">• Vallisneria americana
• Filamentous green algae</td></tr>
<tr><td>30</td><td>BARE</td></tr>
<tr><td>40</td><td rowspan="8">• Ceratophyllum demersum
• Elodea canadensis</td></tr>
<tr><td>50</td></tr>
<tr><td>60</td></tr>
<tr><td>70</td></tr>
<tr><td>80</td><td rowspan="8">BARE</td></tr>
<tr><td>90</td><td rowspan="20"></td></tr>
<tr><td>100</td></tr>
<tr><td>125</td><td rowspan="8">• Myriophyllum spicatum
• Heteranthera dubia
• Vallisneria americana</td><td>BARE</td></tr>
<tr><td>150</td><td rowspan="17"></td><td>• Vallisneria americana</td></tr>
<tr><td>175</td><td>BARE</td></tr>
<tr><td>200</td><td rowspan="15"></td></tr>
<tr><td>225</td><td rowspan="6">• Vallisneria americana</td></tr>
<tr><td>250</td><td rowspan="13">• Vallisneria americana
• Ceratophyllum demersum
• Myriophyllum spicatum
• Najas flexilis</td></tr>
<tr><td>275</td></tr>
<tr><td>300</td></tr>
<tr><td>350</td><td rowspan="10"></td></tr>
<tr><td>400</td></tr>
<tr><td>450</td><td rowspan="2">BARE</td></tr>
<tr><td>500</td></tr>
<tr><td>550</td><td rowspan="6"></td></tr>
<tr><td>600</td></tr>
<tr><td>650</td></tr>
<tr><td>700</td></tr>
<tr><td>750</td></tr>
<tr><td>800</td></tr>
</table>

Table 11. Dominant taxa based on relative abundance along each of the macrophyte transects in 2011. Loc. = distance from shore in meters. Taxa in bold have Coefficients of Conservation values of zero, indicating non-native or most likely to be found in degraded habitat.

<table>
<tr><th>Loc.</th><th>NW Reference</th><th>NE Reference</th><th>Amoco</th><th>Grand Trunk</th><th>Heritage Landing</th><th>Kirksey</th></tr>
<tr><td>0</td><td rowspan="9">• Typha angustifolia
• Phragmites australis
• Schoenoplectus pungens
• Utricularia intermedia</td><td rowspan="15">• Filamentous green algae
• Najas flexilis
• Vallisneria americana
• Chara sp.</td><td rowspan="6">BARE</td><td rowspan="3">• Typha angustifolia
• Lythrum salicaria</td><td>• Salix exigua
• Impatiens capensis</td><td>• Filamentous green algae</td></tr>
<tr><td>5</td><td rowspan="6">• Nymphaea odorata
• Ceratophyllum demersum
• Vallisneria americana</td><td rowspan="2">BARE</td></tr>
<tr><td>10</td></tr>
<tr><td>20</td><td rowspan="13">• Ceratophyllum demersum
• Myriophyllum spicatum
• Nymphaea odorata
• Elodea canadensis</td><td>• Vallisneria americana</td></tr>
<tr><td>30</td><td>BARE</td></tr>
<tr><td>40</td><td rowspan="7">• Vallisneria americana
• Potamogeton pectinatus
• Potamogeton perfoliatus</td></tr>
<tr><td>50</td><td rowspan="4">• Vallisneria americana
• Potamogeton perfoliatus
• Potamogeton pectinatus</td></tr>
<tr><td>60</td><td rowspan="5">• Elodea canadensis
• Ceratophyllum demersum
• Vallisneria americana</td></tr>
<tr><td>70</td></tr>
<tr><td>80</td><td rowspan="7">• Najas flexilis
• Chara sp.
• Filamentous green algae</td></tr>
<tr><td>90</td><td rowspan="19"></td></tr>
<tr><td>100</td></tr>
<tr><td>125</td><td>• Ceratophyllum demersum</td><td>BARE</td></tr>
<tr><td>150</td><td rowspan="16"></td><td rowspan="2">• Vallisneria americana
• Potamogeton perfoliatus</td></tr>
<tr><td>175</td></tr>
<tr><td>200</td><td rowspan="4">• Myriophyllum spicatum
• Ceratophyllum demersum
• Vallisneria americana
• Najas flexilis
• Potamogeton pusillus</td><td rowspan="14"></td></tr>
<tr><td>225</td><td rowspan="13">• Vallisneria americana
• Ceratophyllum demersum
• Myriophyllum spicatum
• Najas flexilis
• Potamogeton pusillus
• Potamogeton perfoliatus</td><td rowspan="7">• Vallisneria americana
• Ceratophyllum demersum
• Potamogeton perfoliatus</td></tr>
<tr><td>250</td></tr>
<tr><td>275</td></tr>
<tr><td>300</td><td rowspan="10"></td></tr>
<tr><td>350</td></tr>
<tr><td>400</td></tr>
<tr><td>450</td></tr>
<tr><td>500</td><td rowspan="6"></td></tr>
<tr><td>550</td></tr>
<tr><td>600</td></tr>
<tr><td>650</td></tr>
<tr><td>700</td></tr>
<tr><td>750</td></tr>
</table>

Table 12. Dominant taxa based on relative abundance along each of the macrophyte transects in 2012. Loc. = distance from shore in meters. Taxa in bold have Coefficients of Conservation values of zero, indicating non-native or most likely to be found in degraded habitat.

<table>
<tr><th>Loc.</th><th>NW Reference</th><th>NE Reference</th><th>Amoco</th><th>Grand Trunk</th><th>Heritage Landing</th><th>Kirksey</th></tr>
<tr><td>0</td><td rowspan="10">• Typha angustifolia
• Phragmites australis
• Schoenoplectus pungens</td><td>• Typha x glauca
• Schoenoplectus pungens
• Salix exigua</td><td rowspan="10">• Filamentous green algae
• Vallisneria americana
• Potamogeton pectinatus
• Potamogeton perfoliatus</td><td rowspan="3">• Typha angustifolia
• Typha x glauca
• Lythrum salicaria
• Impatiens capensis</td><td>• Salix exigua
• Impatiens capensis</td><td>• Filamentous green algae
• Vallisneria americana</td></tr>
<tr><td>5</td><td rowspan="9">• Filamentous green algae
• Vallisneria americana
• Chara sp.</td><td>• Filamentous green algae</td><td rowspan="11">• Vallisneria americana
• Potamogeton perfoliatus
• Potamogeton perfoliatus</td></tr>
<tr><td>10</td><td rowspan="10">• Ceratophyllum demersum
• Elodea nutallii</td></tr>
<tr><td>20</td><td rowspan="15">• Ceratophyllum demersum
• Nymphaea odorata
• Elodea nuttallii
• Myriophyllum spicatum</td></tr>
<tr><td>30</td></tr>
<tr><td>40</td></tr>
<tr><td>50</td></tr>
<tr><td>60</td></tr>
<tr><td>70</td></tr>
<tr><td>80</td></tr>
<tr><td>90</td><td rowspan="6">• Najas flexilis
• Chara sp.
• Filamentous green algae
• Potamogeton pectinatus</td><td rowspan="8">• Myriophyllum spicatum
• Vallisneria americana
• Najas flexilis
• Heteranthera dubia</td><td rowspan="18"></td></tr>
<tr><td>100</td></tr>
<tr><td>125</td><td>• Ceratophyllum demersum</td><td>BARE</td></tr>
<tr><td>150</td><td rowspan="15"></td><td>• Potamogeton pectinatus</td></tr>
<tr><td>175</td><td>• Vallisneria americana</td></tr>
<tr><td>200</td><td rowspan="13"></td></tr>
<tr><td>225</td><td rowspan="8">• Vallisneria americana
• Myriophyllum spicatum
• Najas flexilis
• Potamogeton perfoliatus
• Filamentous green algae</td></tr>
<tr><td>250</td></tr>
<tr><td>275</td><td>• Ceratophyllum demersum</td><td rowspan="4">• Ceratophyllum demersum
• Elodea nuttallii
• Myriophyllum spicatum
• Vallisneria americana
• Najas guadalupensis
• Potamogeton pusillus</td></tr>
<tr><td>300</td><td rowspan="9"></td></tr>
<tr><td>350</td></tr>
<tr><td>400</td></tr>
<tr><td>450</td><td rowspan="6"></td></tr>
<tr><td>500</td></tr>
<tr><td>550</td><td rowspan="4">• Ceratophyllym demersum
• Najas flexilis</td></tr>
<tr><td>600</td></tr>
<tr><td>650</td></tr>
<tr><td>700</td></tr>
</table>

Table 13. Dominant taxa based on relative abundance along each of the macrophyte transects in 2019. Loc. = distance from shore in meters. Taxa in bold have Coefficients of Conservation values of zero, indicating non-native or most likely to be found in degraded habitat.

<table>
<tr><th>Loc.</th><th>NW Reference</th><th>NE Reference</th><th>Amoco</th><th>Circle Bay</th><th>Grand Trunk</th><th>Heritage Landing</th><th>Kirksey</th></tr>
<tr><td>0</td><td rowspan="5">• *Ceratophyllum demersum*
• *Lemna major; L. minor*
• ***Typhya* x *glauca***
• Unknown sedge
• *Vallisneria americana*</td><td rowspan="4">• Macroalgae
• *Vallisneria americana*</td><td rowspan="9">• Macroalgae
• *Potamogeton perfoliatus*
• *Potamogeton richardsonii*
• *Vallisneria americana*</td><td rowspan="6">• *Ceratophyllum demersum*
• *Elodea nutelli*
• Macroalgae
• Unknown sedge
• *Vallisneria americana*</td><td rowspan="4">• *Lemna major; L. minor*
• *Lemna triscula*
• ***Typha* x *glauca***</td><td rowspan="2">• Macroalgae
• *Vallisneria americana*</td><td rowspan="5">• Macroalgae
• *Nitella* sp.
• *Salix* sp.
• *Vallisneria americana*</td></tr>
<tr><td>5</td></tr>
<tr><td>10</td><td rowspan="10">• *Ceratophyllum demersum*
• *Elodea canadensis*
• *Elodea nutelli*
• ***Potamogeton crispus***
• *Potamogeton perfoliatus*</td></tr>
<tr><td>20</td></tr>
<tr><td>30</td><td rowspan="11">• *Nitella* sp.</td><td rowspan="7">• *Ceratophyllum demersum*
• *Potamogeton zosteriformis*</td></tr>
<tr><td>40</td><td rowspan="11">• Macroalgae
• *Myriophyllym spicatum*
• *Najas flexilis*
• ***Typhya* x *glauca***
• *Vallisneria americana*</td><td rowspan="5">• *Nitella* sp.
• *Nitellopsis obtusa*
• *Potamogeton perfoliatus*
• *Vallisneria americana*</td></tr>
<tr><td>50</td><td rowspan="9">• *Ceratophyllum demersum*
• *Vallisneria americana*</td></tr>
<tr><td>60</td></tr>
<tr><td>70</td></tr>
<tr><td>80</td><td rowspan="19"></td></tr>
<tr><td>90</td><td rowspan="5">• *Ceratophyllum demersum*
• *Najas flexilis*
• *Nitella* sp.
• ***Potamogeton crispus***
• *Vallisneria americana*</td></tr>
<tr><td>100</td><td rowspan="4">• *Ceratophyllum demersum*
• ***Myriophyllum spicatum***
• *Najas flexilis*
• *Nitella* sp.</td></tr>
<tr><td>125</td><td rowspan="16"></td></tr>
<tr><td>150</td></tr>
<tr><td>175</td></tr>
<tr><td>200</td><td rowspan="5">• *Ceratophyllum demersum*
• *Nitella* sp.
• *Nitellopsis obtusa*
• *Vallisneria americana*</td><td rowspan="13"></td><td rowspan="4">• *Vallisneria americana*</td><td rowspan="13"></td></tr>
<tr><td>225</td><td rowspan="5">• *Heteranthera dubia*
• Macroalgae
• *Potamogeton perfoliatus*
• *Vallisneria americana*</td></tr>
<tr><td>250</td></tr>
<tr><td>275</td></tr>
<tr><td>300</td><td rowspan="3">• *Ceratophyllum demersum*
• ***Myriophyllum spicatum***
• *Vallisneria americana*</td></tr>
<tr><td>350</td><td rowspan="8"></td></tr>
<tr><td>400</td><td rowspan="7">• *Ceratophyllum demersum*
• *Heteranthera dubia*
• ***Myriophyllum spicatum***
• *Vallisneria americana*</td></tr>
<tr><td>450</td><td rowspan="6"></td></tr>
<tr><td>500</td></tr>
<tr><td>550</td></tr>
<tr><td>600</td></tr>
<tr><td>650</td></tr>
<tr><td>700</td></tr>
</table>

Table 14. Coefficient of Conservatism (C) values and weighted mean relative abundance (%) for taxa found along each macrophyte transect in 2009. E = emergent, S = submergent, F = floating. – indicates that no C-value was available for that taxon.

Species	Type	C	NW Ref	NE Ref	G. Trunk	H. Landing	Kirksey
Carex hystericinia	E	2	1				
Ceratophyllum demersum	S	1	13	4	23	63	3
Chara sp.	S	—		18		<1	
Cicuta bulbifera	E	5	<1				
Elodea canadensis	S	1	<1	<1	6	23	<1
Filamentous green algae	—	—	7	11	<1		18
Heteranthera dubia	S	6	<1	2	1	1	
Impatiens capensis	E	2	<1				
Juncus articulatus	E	3	<1				
Juncus canadensis	E	6	<1				
Juncus sp.	E	—	<1				
Lemna minor	F	5	<1		<1		
Lemna trisulca	F	6			<1		
Lythrum salicaria	E	0	<1	1	4		
Myriophyllum spicatum	S	0	2	2	10	5	1
Najas flexilis	S	5	6	23		<1	
Najas guadalupensis	S	7			1		
Nasturtium microphyllum	E	0			2		
Nymphaea odorata	F	6	<1		10	1	
Peltandra virginica	E	6			<1		
Phragmites australis	E	0	10				
Potamogeton crispus	S	0			3		
Potamogeton illinoensis	S	5					4
Potamogeton nodosus	S	6			<1		
Potamogeton pectinatus	S	3	<1	<1	3	<1	19
Potamogeton perfoliatus	S	6		<1	5	<1	16
Potamogeton pusillus	S	4	3	1	4	<1	1
Potamogeton zosteriformis	S	5			<1	1	<1
Ranunculus flabellaris	S	10	<1				<1
Sagittaria sp.	E	—			<1		
Sagittaria latifolia	E	1	<1				
Salix sp.	E	—			<1		
Salix exigua	E	1		4		2	
Schoenoplectus acutus	E	5	4				
Schoenoplectus pungens	E	5	6	8			
Schoenoplectus tabernaemontani	E	4	<1			1	
Spirodela polyrhiza	F	6			<1		
Typha angustifolia	E	0	27	1	8		
Utricularia vulgaris	S	6	8		5		
Vallisneria americana	S	7	12	24	14	3	39
Mean C			3.6	3	3.9	3.8	4.2
Submergent Richness			10	10	13	11	10
Total Richness			25	14	24	14	10

Table 15. Coefficient of Conservatism (C) values and weighted mean relative abundance (%) for taxa found along each macrophyte transect in 2010. See Table 14 for table explanation.

Species	Type	C	NW Ref	NE Ref	Amoco	G. Trunk	H. Landing	Kirksey
Carex comosa	E	5	<1					
Ceratophyllum demersum	S	1	2	8	<1	33	38	
Chara sp.	S	–	2	23				
Cicuta bulbifera	E	5	<1			<1		
Cladium mariscoides	E	10	<1					
Cuscuta gronovii	E	3	<1					
Elodea canadensis	S	1	<1			3	31	
Elodea nuttallii	S	5	<1	6	<1	2		
Eupatorium perfoliatum	E	4	<1					
Filamentous green algae	–	–	3	3	31	5		8
Galium tinctorium	E	5	<1					
Heteranthera dubia	S	6	1	10		2	6	
Hydrocotyle umbellata	E	10	<1					
Impatiens capensis	E	2	<1			<1	<1	
Juncus articulatus	E	3	<1					
Juncus canadensis	E	6	2					
Juncus debilis	E	–				<1		
Lemna minor	F	5	<1			<1		
Lemna trisulca	F	6	<1			<1		
Lythrum salicaria	E	0	<1	1		2		
Myriophyllum spicatum	S	0	7	9	2	10	9	1
Najas flexilis	S	5	7	6	<1		<1	1
Najas guadalupensis	S	7				2		
Nasturtium microphyllum	E	0	<1			2		
Nymphaea odorata	F	6				9	7	
Phragmites australis	E	0	10					
Polyganum punctatum var. *confertiflorum*	E	5				<1		
Potamogeton crispus	S	0		<1		<1		
Potamogeton pectinatus	S	3	<1		4	2		8
Potamogeton perfoliatus	S	6	2	<1	4	2	<1	8
Potamogeton pusillus	S	4	2	<1	2	1	<1	11
Potamogeton zosteriformis	S	5				<1		<1
Ranunculus flabellaris	S	10				<1		
Salix sp.	E	–				<1		
Salix exigua	E	1		2			1	
Salix petiolaris	E	1	<1					
Schoenoplectus pungens	E	5	6	1				
Schoenoplectus tabernaemontani	E	4	<1			<1	1	
Sparganium eurycarpum	E	5				1		
Typha angustifolia	E	0	19	2				
Typha* x *glauca	E	0				3		
Utricularia sp.	S	–						
Utricularia geminiscarpa	S	8	2					
Utricularia intermedia	S	10	2					
Utricularia minor	S	10	2					
Utricularia vulgaris	S	6				2		
Unknown sedge	E	–					<1	
Vallisneria americana	S	7	28	30	56	16	7	64
Various grasses	E	–	2					
Mean C			4.5	3.1	4.4	4	3.6	4.3
Submergent Richness			14	10	8	13	8	7
Total Richness			34	14	8	27	13	7

Table 16. Coefficient of Conservatism (C) values and weighted mean relative abundance (%) for taxa found along each macrophyte transect in 2011. See Table 14 for table explanation.

Species	Type	C	NW Ref	NE Ref	Amoco	G. Trunk	H. Landing	Kirksey
Carex comosa	E	5	<1					
Carex hystericinia	E	2	<1					
Ceratophyllum demersum	S	1	3	7		47	57	2
Chara sp.	S	–	2	15				
Cicuta bulbifera	E	5	<1					
Cirsium muticum	E	6	<1					
Cladium mariscoides	E	10	<1					
Cuscuta gronovii	E	3	<1					
Eleocharis sp.	E	–	<1					
Elodea canadensis	S	1	<1	<1		4	11	
Elodea nuttallii	S	5	<1			1		
Filamentous green algae	–	–	6	11	2	3	<1	7
Galium tinctorium	E	5	<1					
Heteranthera dubia	S	6	2	2		2	<1	
Hydrocotyle umbellata	E	10	<1					
Impatiens capensis	E	2	<1			<1	1	
Juncus sp. 1	E	–	<1					
Juncus sp. 2	E	–	<1					
Juncus sp. 3	E	–	<1					
Juncus sp. 4	E	–	<1					
Juncus articulatus	E	3	1					
Juncus effusus	E	3	<1					
Lemna minor	F	5	1			1		
Lemna trisulca	F	6				<1		
Lythrum salicaria	E	0	1			3		
Myriophyllum spicatum	S	0	6	6		2	2	1
Myosotis laxa	E	6	<1					
Najas flexilis	S	5	8	14			2	1
Najas guadalupensis	S	7				1		
Nasturtium microphyllum	E	0	<1			<1		
Nymphaea odorata	F	6	<1			4	6	
Peltandra virginica	E	6	<1			<1		
Phragmites australis	E	0	14					
Pilea pumila	E	5	<1					
Polyganum virginianum	E	4	<1					
Potamogeton crispus	S	0			4	2		2
Potamogeton illinoensis	S	5		<1				
Potamogeton pectinatus	S	3	1	1	17	1		8
Potamogeton perfoliatus	S	6	2	1	33	2	<1	13
Potamogeton pusillus	S	4	3	4		2	5	2
Rumex sp.	E	–	<1					
Salix exigua	E	1					1	
Salix petiolaris	E	1	<1					
Sagittaria latifolia	E	1	<1					
Schoenoplectus acutus	E	5	2				<1	
Schoenoplectus pungens	E	5	7					
Schoenoplectus tabernaemontani	E	4	<1					
Scutellaria galericulata	E	5	<1					
Spirodela polyrhiza	F	6				<1		
Typha angustifolia	E	0	18			9		
Typha* x *glauca	E	0	1			<1		
Typha latifolia	E	1	<1					
Utricularia sp.	S	–				1		
Utricularia intermedia	S	10	3					
Vallisneria americana	S	7	16	38	44	13	13	66
Various grasses	E	–	<1					
Mean C			3.9	3.8	4	3.4	3.7	3.3
Submergent Richness			12	11	4	12	8	8
Total Richness			48	11	4	22	12	8

Table 17. Coefficient of Conservatism (C) values and weighted mean relative abundance (%) for taxa found along each macrophyte transect in 2012. See Table 14 for table explanation.

Species	Type	C	NW Ref	NE Ref	Amoco	G. Trunk	H. Landing	Kirksey
Carex comosa	E	5	<1					
Ceratophyllum demersum	S	1	10	10		36	63	
Chara sp.	S	–	3	12				
Cirsium muticum	E	6	<1					
Cladium mariscoides	E	10	<1					
Cuscuta gronovii	E	3	<1					
Eleocharis sp.	E	–	<1					
Elodea canadensis	S	1	<1			1	<1	
Elodea nuttallii	S	5	<1	1		9	22	
Epilobium coloratum	E	3				<1		
Eupatorium perfoliatum	E	4	<1					
Eutrochium maculatum	E	4	<1					
Filamentous green algae	–	–	4	14	16	3	3	3
Galium tinctorium	E	5	<1					
Heteranthera dubia	S	6	1	10		2	1	<1
Hydrocotyle umbellata	E	10	<1					
Impatiens capensis	E	2	<1			4	1	
Juncus sp. 1	E	–	<1					
Juncus sp. 2	E	–	<1					
Juncus sp. 3	E	–	<1					
Juncus sp. 4	E	–	<1					
Juncus sp. 5	E	–	<1					
Juncus acuminatus	E	8	<1					
Juncus articulatus	E	3	<1					
Juncus brachycephalus	E	7	<1					
Juncus effusus	E	3	<1					
Leersia oryzoides	E	3	<1					
Lemna minor	F	5	<1			<1		
Lemna trisulca	F	6				<1		
Lycopus sp.	E	–	<1					
Lythrum salicaria	E	0	<1			4		
Moss	E	–	<1					
Myriophyllum spicatum	S	0	6	9	2	8	4	<1
Myosotis laxa	E	6	<1					
Najas flexilis	S	5		12				2
Najas guadalupensis	S	7	13			2		
Nasturtium microphyllum	E	0	<1			<1		
Nuphar variegata	F	7				<1		
Nymphaea odorata	F	6				11	1	
Peltandra virginica	E	6	<1			<1		
Phragmites australis	E	0	14	<1			<1	
Pilea pumila	E	5	<1					
Polyganum punctatum	E	5	<1			<1		
Pontederia cordata	E	8	<1			<1		
Potamogeton crispus	S	0				<1	<1	
Potamogeton illinoensis	S	5	<1					
Potamogeton pectinatus	S	3	1	1	20	2	<1	34
Potamogeton perfoliatus	S	6	2	1	7	<1	1	20
Potamogeton pusillus	S	4	<1	1	2	1	<1	<1
Potamogeton zosteriformis	S	5				1	<1	
Rumex sp.	E	–	<1					
Salix exigua	E	1		1			1	
Sagittaria latifolia	E	1	<1					
Schoenoplectus acutus	E	5	<1					
Schoenoplectus pungens	E	5	3	1				
Schoenoplectus tabernaemontani	E	4	<1					
Scutellaria lateriflora	E	5	<1					
Sparganium eurycarpium	E	5				<1		
Spirodela polyrhiza	F	6				<1		
Typha angustifolia	E	0	25			4		
Typha x glauca	E	0	<1	1		3		
Unknown emergent 1	E	–	<1					

Unknown emergent 2	E	–	<1					
Utricularia vulgaris	S	6				1		
Vallisneria americana	S	7	13	30	53	4	2	40
Various grasses	E	–	<1					
Verbena hastata	E	4	<1					
Mean C			4.2	3.3	4	3.9	3.1	4.4
Submergent Richness			12	10	5	13	11	7
Total Richness			55	14	5	28	15	7

Table 18. Coefficient of Conservatism (C) values and weighted mean relative abundance (%) for taxa found along each macrophyte transect in 2019. See Table 14 for table explanation.

Species	Type	C	NW Ref	NE Ref	Amoco	Circle Bay	G. Trunk	H. Landing	Kirksey
Carex sp.	E	-	<1	<1		2			
Cephalanthus occidentalis	E	7					<1		
Ceratophyllum demersum	S	1	24	4	<1	47	19	46	1
Chara sp.	-	-	<1						
Elodea canadensis	S	1	<1			3	1	8	
Elodea nutallii	S	5	<1			3	3	12	
Filamentous green algae	-	-	11	22	34	9	7	18	15
Heteranthera dubia	S	6	5	1	2	3	2	<1	
Lemna minor	F	5	1			<1	4		
Lemna trisulca	F	6	1			1	5	<1	1
Lythrum salicaria	E	0	<1				<1		
Myriophyllum spicatum	S	0	9	1	<1	8	6	<1	<1
Najas flexilis	S	5	5	1	1		2		2
Nitella sp.	-	-	2	60	2		7		25
Nitellopsis obtusa	-	-		4			5		15
Nuphar lutea variegata	F	7	<1				<1	<1	
Nymphaea odorata	F	7				2	1	1	
Peltandra virginica	E	6				1			
Pontederia cordata	E	8				<1			
Potamogeton amplifolius	F	6				<1			
Potamogeton crispus	S	0						2	3
Potamogeton natans	F	5	<1					1	
Potamogeton pectinatus	S	3	<1	<1	<1		3		<1
Potamogeton perfoliatus	S	6	11	2	21	1	2	2	8
Potamogeton pusillis	S	4	1	1		<1	<1	<1	<1
Potamogeton richardsonii	S	5			2	2	<1		
Potamogeton zosteriformis	S	5	<1			1	7	5	
Salix exigua	E	1				1		<1	11
Schoenoplectus pungens	E	5	<1						
Spirodela polyrhiza	F	6	1			<1	5		
Typha x glauca	E	0	1				6		
Utricularia vulgaris	S	6	<1		<1	<1	2	<1	<1
Vallisneria americana	S	7	27	3	38	15	11	4	19
Various grasses	E	-	<1						
Wolffia sp.	F	5	<1				1		
Mean C			4.2	4	4.3	4.8	4.4	4.2	3.5
Submergent Richness			12	8	9	11	13	11	9
Total Richness			26	12	11	21	25	17	14

Table 19. Grand means (±SD) of mean coefficient of conservatism (C) values, submergent richness, and total species richness at each transect pre- (2009-2010) and post-restoration (2011, 2012, and 2019). ND = no data; NA = not applicable. The “other richness” parameter summarizes non-vascular plants, such as macroalgae, including filamentous green algae.

Parameter	Time	NW Ref	NE Ref	Amoco	Circle Bay	G. Trunk	H. Landing	Kirksey
Sample Size (n=years)	Pre	2	2	1	0	2	2	2
	Post ('11-'12)	2	2	2	0	2	2	2
	Post ('19)	1	1	1	1	1	1	1
Mean C	Pre	4.0 (0.6)	3.0 (0.1)	4.4 (NA)	ND (NA)	3.9 (0.1)	3.7 (0.1)	4.2 (0.1)
	Post ('11-'12)	4.2 (0.2)	3.3 (0.3)	4.0 (0.0)	ND (NA)	3.9 (0.4)	3.1 (0.4)	4.4 (0.8)
	Post ('19)	4.2 (NA)	4.0 (NA)	4.3 (NA)	4.8 (NA)	4.4 (NA)	4.2 (NA)	3.5 (NA)
Submergent Richness	Pre	12.0 (2.8)	10.0 (0.0)	8.0 (NA)	ND (NA)	13.0 (0.0)	9.5 (2.1)	8.5 (2.1)
	Post ('11-'12)	12.0 (0.0)	10.5 (0.7)	4.5 (0.7)	ND (NA)	12.5 (0.7)	9.5 (2.1)	7.5 (0.7)
	Post ('19)	12.0 (NA)	8.0 (NA)	9.0 (NA)	11.0 (NA)	13.0 (NA)	11.0 (NA)	9.0 (NA)
Emergent Richness	Pre	15.0 (2.8)	4.0 (0.0)	0.0 (NA)	ND (NA)	8.0 (2.8)	3.0 (1.4)	0.0 (0.0)
	Post ('11-'12)	38.0 (5.7)	2.0 (2.8)	0.0 (0.0)	ND (NA)	8.0 (2.8)	3.0 (0.0)	0.0 (0.0)
	Post ('19)	5.0 (NA)	1.0 (NA)	0.0 (NA)	4.0 (NA)	3.0 (NA)	1.0 (NA)	1.0 (NA)
Floating Richness	Pre	2.0 (0.0)	0.0 (0.0)	0.0 (NA)	ND (NA)	3.5 (0.7)	1.0 (0.0)	0.0 (0.0)
	Post ('11-'12)	1.5 (0.7)	0.0 (0.0)	0.0 (0.0)	ND (NA)	4.5 (0.7)	1.0 (0.0)	0.0 (0.0)
	Post ('19)	6.0 (NA)	0.0 (NA)	0.0 (NA)	5.0 (NA)	6.0 (NA)	4.0 (NA)	1.0 (NA)
Other Richness	Pre	1.0 (0.0)	1.0 (0.0)	1.0 (NA)	ND (NA)	1.0 (0.0)	0.0 (0.0)	1.0 (0.0)
	Post ('11-'12)	1.0 (0.0)	1.0 (0.0)	1.0 (0.0)	ND (NA)	1.0 (0.0)	1.0 (0.0)	1.0 (0.0)
	Post ('19)	3.0 (NA)	3.0 (NA)	2.0 (NA)	1.0 (NA)	3.0 (NA)	1.0 (NA)	3.0 (NA)
Total Richness	Pre	30.0 (5.7)	15.0 (0.0)	9.0 (NA)	ND (NA)	26.0 (2.8)	13.5 (0.7)	9.5 (2.1)
	Post ('11-'12)	52.5 (4.9)	13.5 (2.1)	5.5 (0.7)	ND (NA)	26.0 (4.2)	14.5 (2.1)	8.5 (0.7)
	Post ('19)	26.0 (NA)	12.0 (NA)	11.0 (NA)	21.0 (NA)	25.0 (NA)	17.0 (NA)	14.0 (NA)

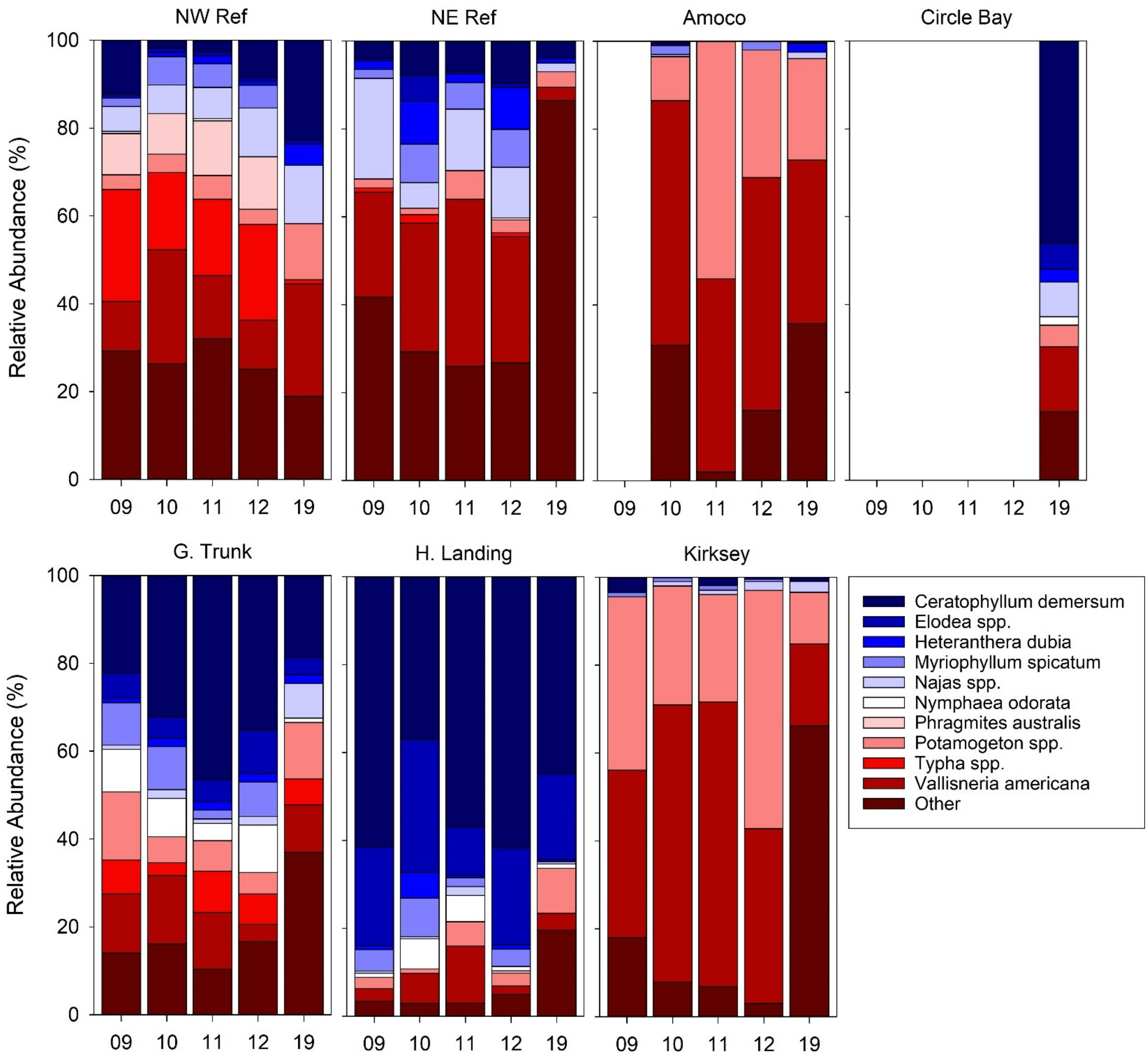

Figure 6. Stacked bar plot of macrophyte taxa weighted relative abundance changes within the five-year (2009-2012, 2019) survey across transects. Each stacked bar represents the average relative abundance of represented macrophyte species at sites along each transect per survey year. Empty columns represent years when a transect was not sampled. Taxa listed are the ten most relatively abundant macrophytes recorded through total five-year study; remaining observed taxa abundances are pooled and listed as “Other”.

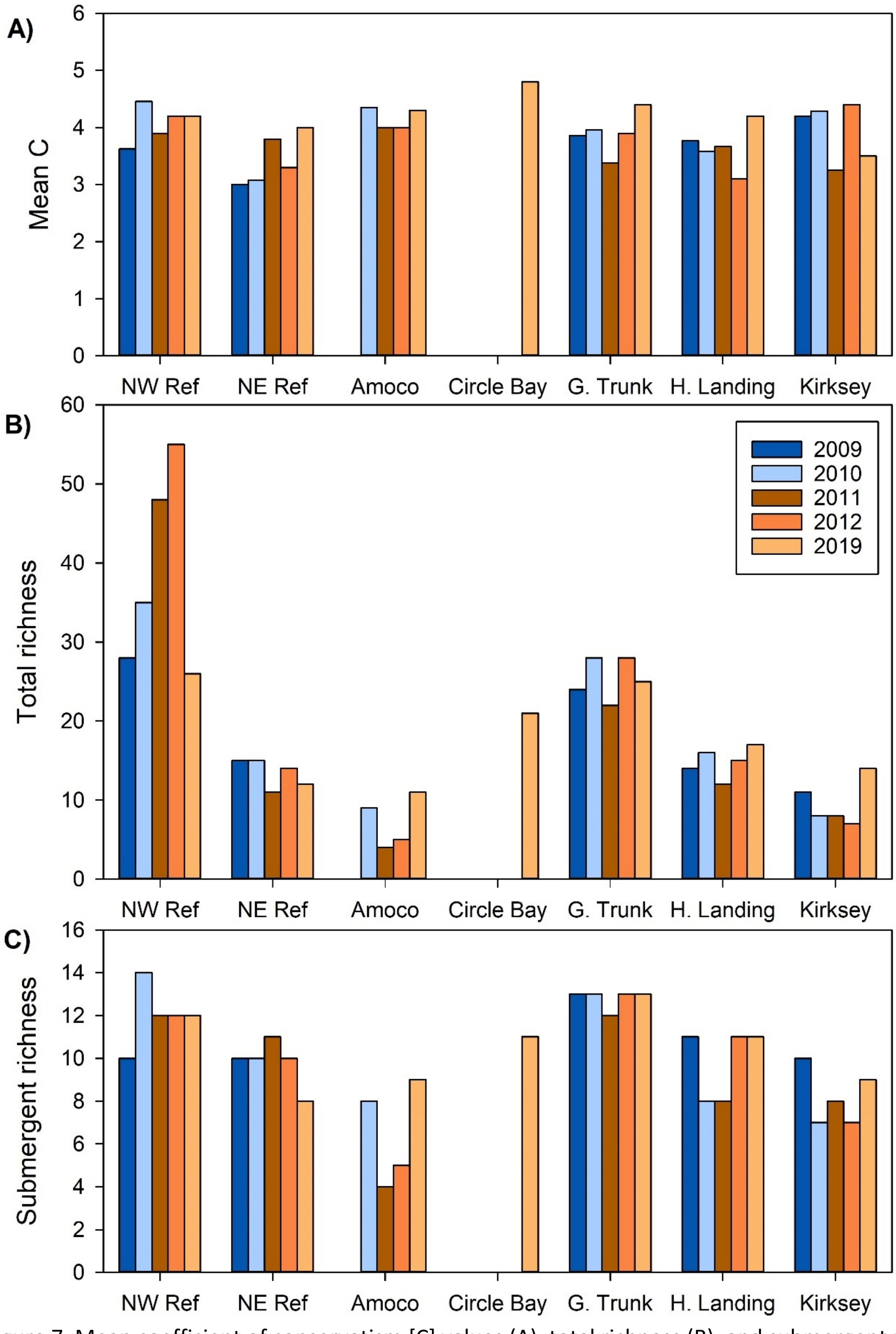

Figure 7. Mean coefficient of conservatism [C] values (A), total richness (B), and submergent richness (C) at each transect before (2009-2010) and after (2011, 2012, and 2019) restoration.

Sediment characterization

As seen in previous years, mean sediment OM percent values were highest at NW Ref, Grand Trunk, and Heritage Landing; all 2019 OM values were within range of previously observed OM (Fig. 8). The increase at NW Ref was driven by the 80 m point (29%, 40%, 49% OM), which was previously dominated in 2012 by large macrophytes (cattails, bulrush, and *Phragmites* spp.) that may have been affected by high water levels in 2019 (Tables 12-13). Sediment OM grand means were significantly different among transects, with the greatest values at Grand Trunk and Heritage Landing and the lowest at Amoco (Fig. 8).

Sediment particle size analysis showed that, with a few exceptions, medium sand (250-500 µm) composed 20-77% of the substrate with the remainder composed of finer rather than coarser substrate, although this varied among transect sites with distance to shore (Figs. 9-15). A full description of sediment data by transect site is provided in Appendix Table 1. The NE Reference and Amoco transects had notably more gravel/cobble (>2 mm), composing 10-15% of nearshore cores (30 and 70 m, and 10 m respectively; Figs. 10 and 11). Grand Trunk and Heritage Landing had the highest ranges of silt/clay (<63 µm) of any 2019 transect, ranging from 42-71% at their mid-to-end transect cores (150 m, and 30and 60 m, respectively; Figs. 13-14).

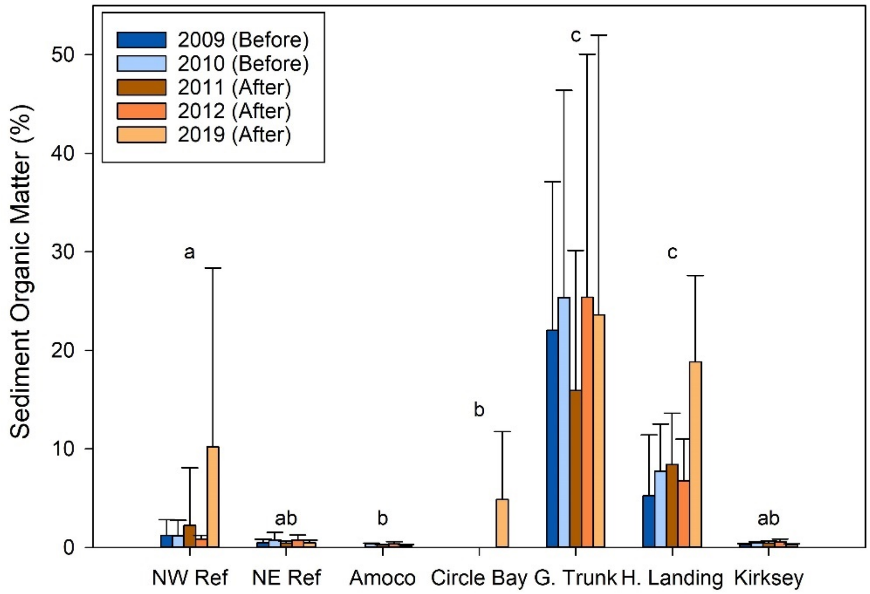

Figure 8. Mean (±SD) sediment organic matter (%) at survey site before (2009 and 2010) and after (2011, 2012, and 2019) restoration. Letters above error bars indicate statistically significant differences between grand means (all years pooled) across sites sampled in 2019 ($p < 0.001$).

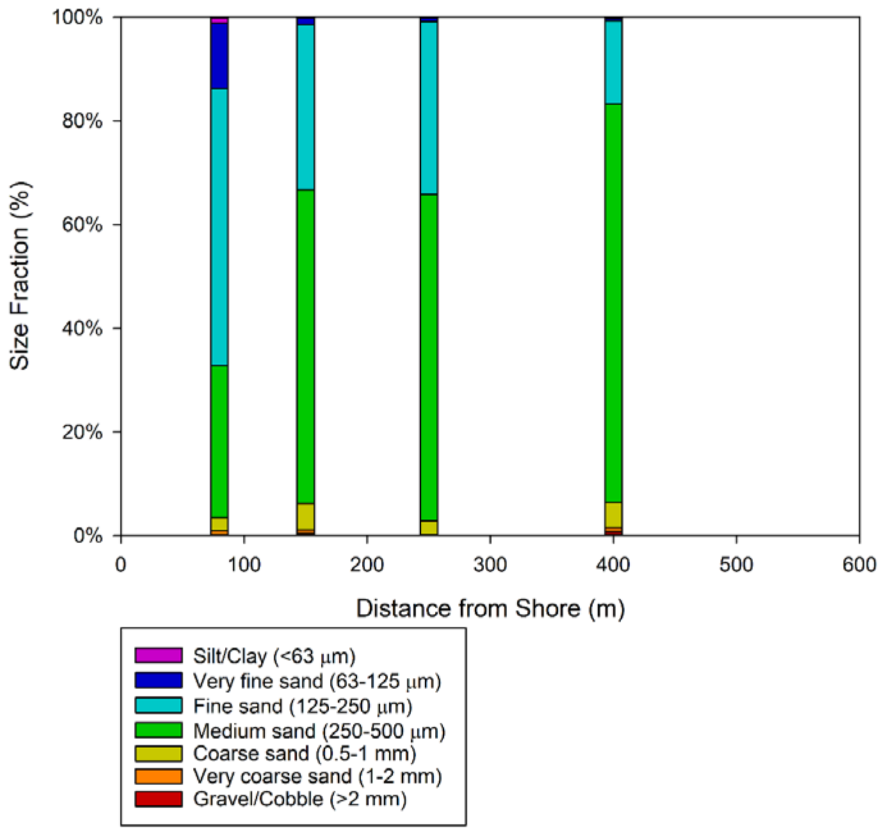

Figure 9. Sediment particle size analysis at NW Reference transect sites in 2019. A full list of sediment fraction values is given in Appendix Table A1.

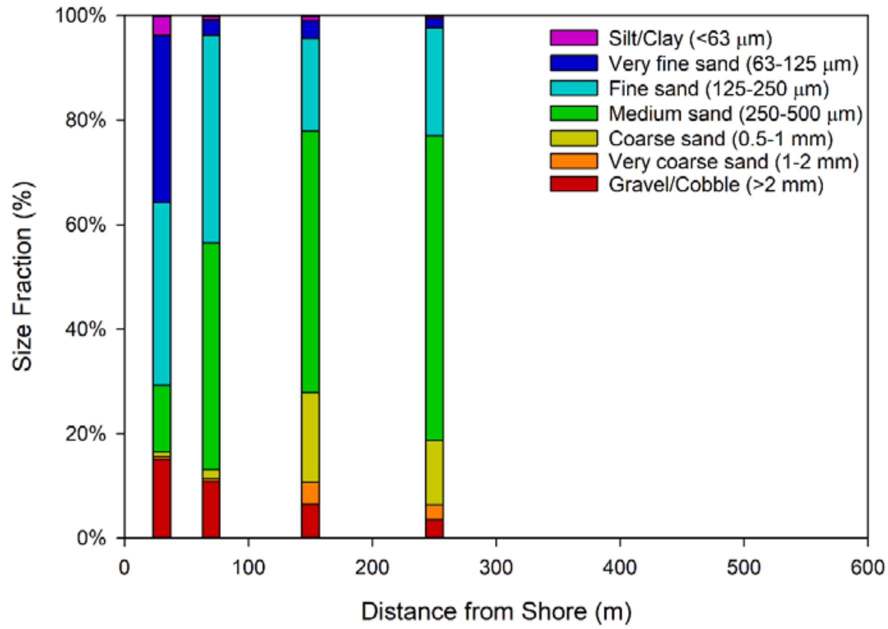

Figure 10. Sediment particle size analysis at NE Reference transect sites in 2019. A full list of sediment fraction values is given in Appendix Table A1.

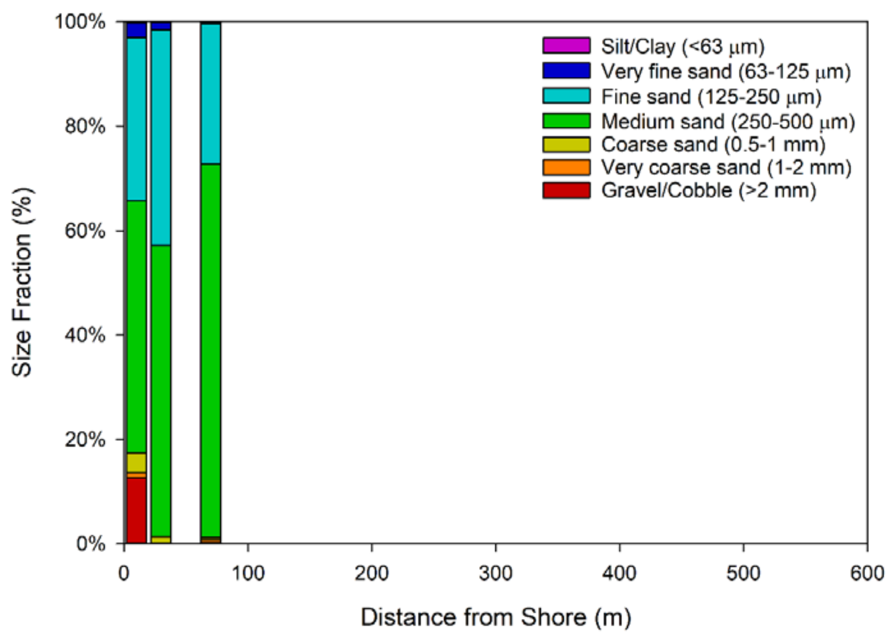

Figure 11. Sediment particle size analysis at Amoco transect sites in 2019. A full list of sediment fraction values is given in Appendix Table A1.

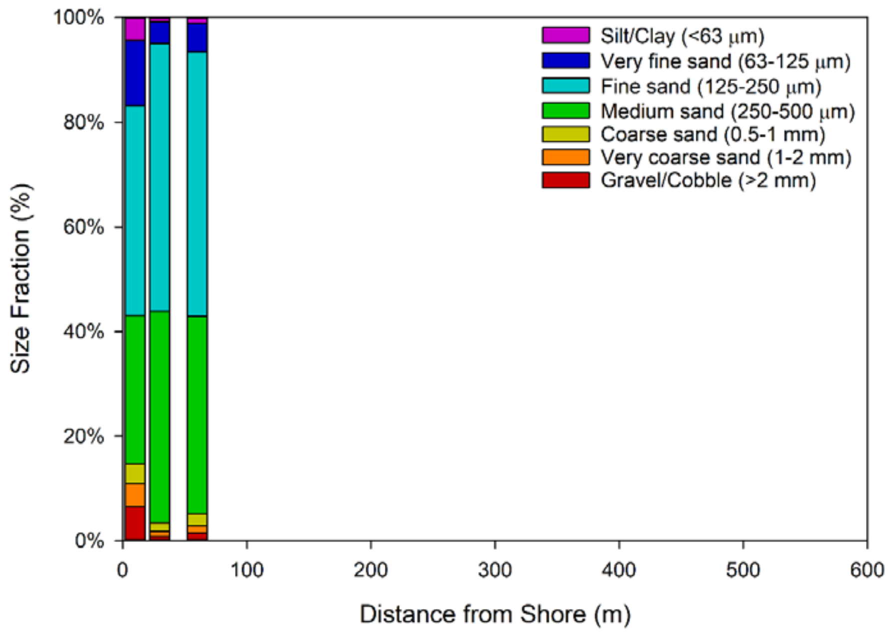

Figure 12. Sediment particle size analysis at Circle Bay transect sites in 2019. A full list of sediment fraction values is given in Appendix Table A1.

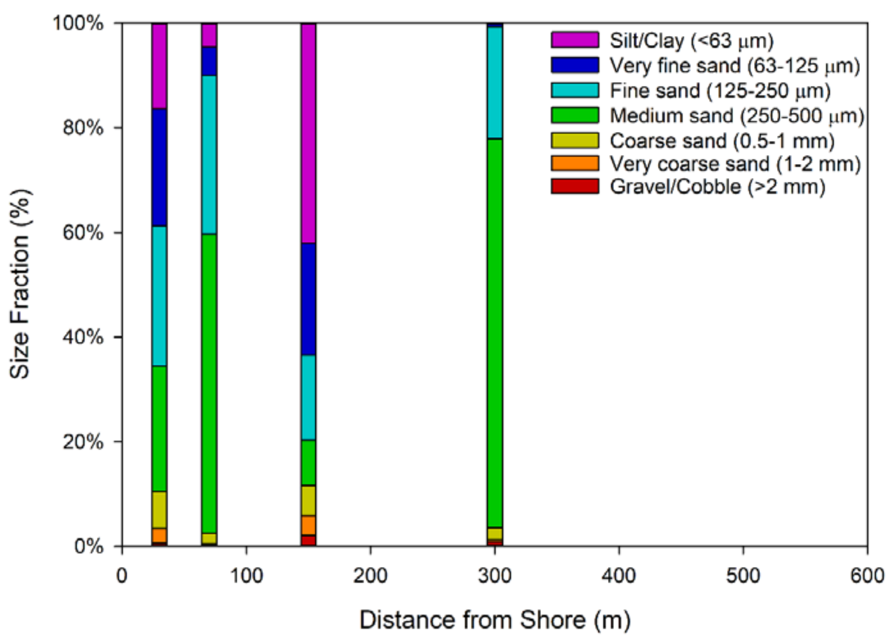

Figure 13. Sediment particle size analysis at Grand Trunk transect sites in 2019. A full list of sediment fraction values is given in Appendix Table A1.

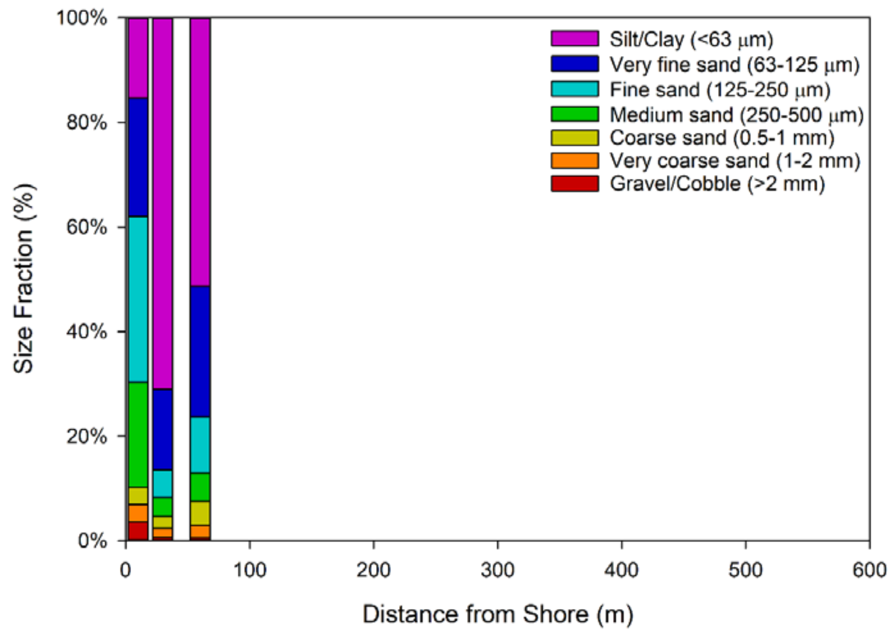

Figure 14. Sediment particle size analysis at Heritage Landing transect sites in 2019. A full list of sediment fraction values is given in Appendix Table A1.

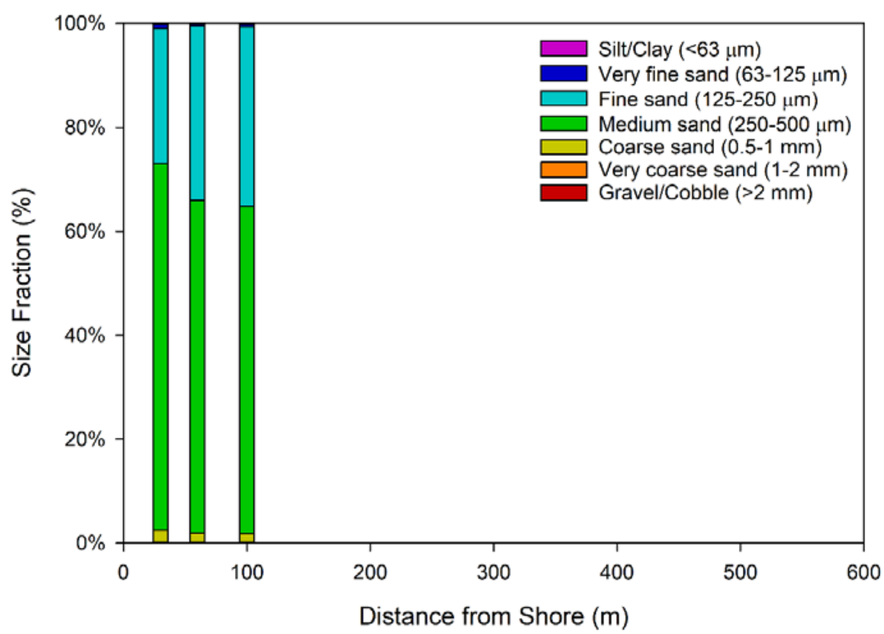

Figure 15. Sediment particle size analysis at Kirksey transect sites in 2019. A full list of sediment fraction values is given in Appendix Table A1.

Muskegon Lake: multivariate analysis

The PCA of environmental transect variables explained 34.69% and 26.42% of variation in the dataset via axes PC1 and PC2, respectively, with WI, T, OM, and WL having the most explanatory power (Fig. 16A). Transects were placed into clusters separated by WI, OM, and slope, with strong overlap between the two reference transects and Kirksey, driven by wind index, while there was more differentiation among the remaining the transects (Fig. 16B). Environmental data were most clearly defined by year, due to differences associated with T, WL, and precip factors (Fig. 16C); high water levels in 2019 helped separate that cluster. Restoration status (reference, pre-, or post-restoration status) clusters largely overlapped, although there was slight separation of the reference sites, driven mostly by wind index (Fig. 16D).

Axes 1 and 2 of the biological variables PCA explained 81.18% and 12.47% of variation in the data, respectively, with each of the four sample variables (richness, biomass, density, cover) having similar explanatory power (Fig. 17A). Transect biological data clustering was less distinct than in the environmental PCA; however, groups of overlapping cluster pairs (NW Ref & G. Trunk; NE Ref & H. Landing; Amoco & Kirksey) were most distinguished from other clusters via richness and biomass (Fig. 17B). Biological data greatly overlapped when organized by year with little separation evident (Fig. 17C). Restoration status again showed overlap among pre- and post-restoration groups; however, the reference cluster, being driven largely by NW Ref data, separated to a degree due to high richness measured in previous sampling years (Fig. 17D).

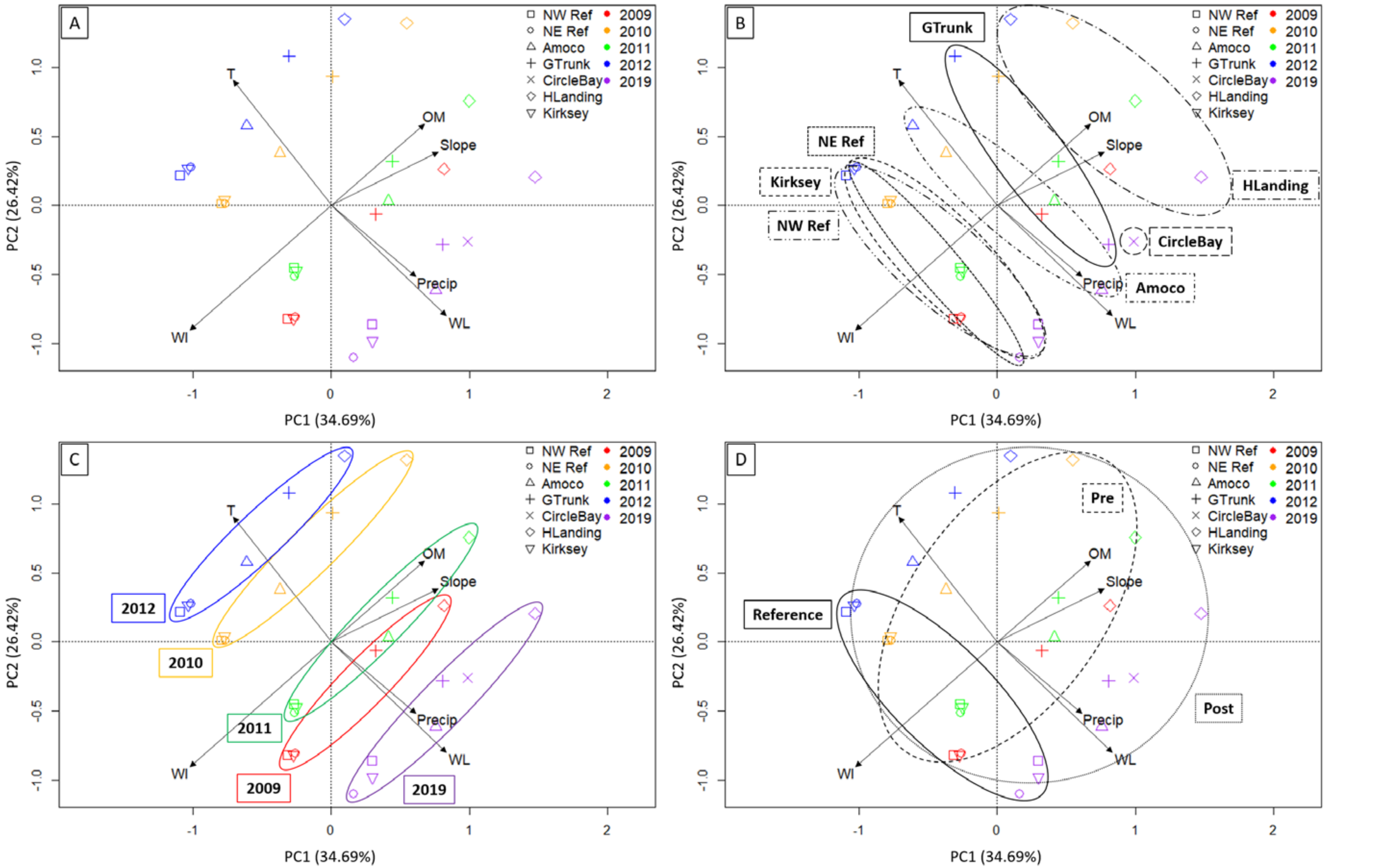

Figure 16. Muskegon Lake transect environmental PCA. (A) Environmental PCA with symbols representing one transect per survey year, symbol shapes representing transects (NW Ref, NE Ref, Amoco, Circle Bay, Grand Trunk, Heritage Landing, and Kirksey), and colors representing survey years (2009-2012 and 2019). Vector length is associated with a variable's explanatory power. (B) Data clustered by transect with site-specific line dash types. (C) Data clustered by survey years (red = 2009, orange = 2010, green = 2011, blue = 2012, and purple = 2019). (D) Data clustered by restoration state (solid line = reference transect, dashed = pre-restoration, and dotted = post-restoration). T = air temperature; OM = organic matter, Precip = precipitation, WL = water level, WI = wind index.

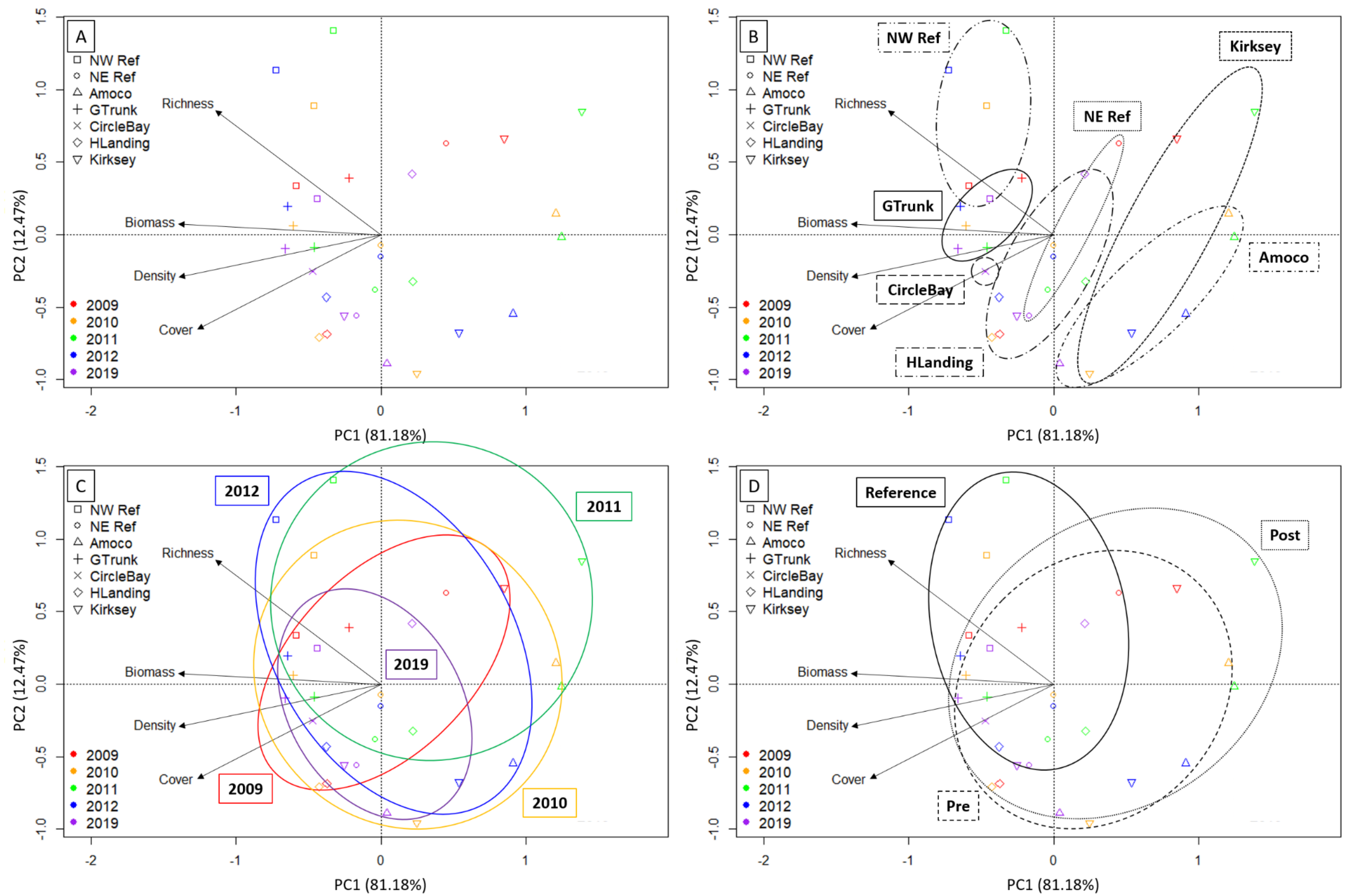

Figure 17. Muskegon Lake transect biological PCA. (A) Biological PCA with symbols representing one transect per survey year, symbol shapes representing transects (NW Ref, NE Ref, Amoco, Circle Bay, Grand Trunk, Heritage Landing, and Kirksey), and colors representing survey years (2009-2012 and 2019). Vector length is associated with a variable's explanatory power. (B) Data clustered by transect with site-specific line dash types. (C) Data clustered by survey years (red = 2009, orange = 2010, green = 2011, blue = 2012, and purple = 2019). (D) Data clustered by restoration state (solid line = reference transect, dashed = pre-restoration, and dotted = post-restoration).

Macroinvertebrate characterization

Macroinvertebrate community composition was not sampled as part of the 2009-2012 habitat restoration assessment, so pre- vs. post-restoration comparisons cannot be made with our 2019 data. No macrophytes were found at the Heritage Landing 60 m site and thus no D-net sweeps were conducted to collect macroinvertebrates at this location. Full summaries of collected taxa and count data per each Muskegon Lake transect point are provided in Appendix Tables A2-A8. Richness within transect points ranged from 4-17 species while whole-transect total species richness ranged from 14-21 (Table 20).

Total abundance of macroinvertebrates was highly variable within transects and was highest at Kirksey due to the abundance of snails (Rissooidea, n=688) across its three collection sites (Table 20). Although insects had the highest species richness of our broad classification groups, snails had the highest total abundance (37%-87%) at all transects except for malacostracans (38%-50%) at Circle Bay and Heritage Landing (Tables 20, A2-A8). Shannon's Diversity index (H') cumulatively for sites was higher at Grand Trunk, Heritage Landing, and Circle Bay restoration transects (2.119, 2.088, 2.007) than at NW and NE reference transects (1.228, 1.798) (Table 20). Circle Bay had the highest maximum diversity (H'_{max}) and Kirksey had the lowest evenness for an entire transect (Table 20).

Table 20. Muskegon Lake 2019 transect macroinvertebrate community composition based on D-net sampling techniques. Shannon's Diversity Index was used to calculate diversity richness (*H'*), maximum diversity (H'_{max}), and evenness within each sampling site as well as totals calculated for each whole transect (gray highlighted text). NA = not applicable; no sample was collected at Heritage Landing 60 m site due to lack of plants to sample for macroinvertebrates. Full abundance count summaries of taxa per site are presented in Appendix Tables A2 – A8.

		D-net				
Transect	Site	Total Species	Total Abundance	H'	H'_{max}	Evenness
NW Ref	80 m	5	56	0.356	1.609	0.221
	250 m	12	238	0.975	2.485	0.392
	400 m	11	119	1.662	2.398	0.693
	total	14	413	1.228	2.639	0.465
NE Ref	30 m	16	170	1.823	2.773	0.658
	150 m	8	146	1.367	2.079	0.657
	250 m	6	56	1.247	1.792	0.696
	total	17	372	1.798	2.833	0.634
Amoco	10 m	10	189	1.208	2.303	0.525
	30 m	11	314	0.786	2.398	0.328
	70 m	4	271	0.616	1.386	0.444
	total	14	774	0.907	2.639	0.344
Circle Bay	10 m	17	200	1.74	2.833	0.614
	30 m	14	118	2.168	2.639	0.822
	60 m	14	143	1.917	2.639	0.726
	total	21	461	2.007	3.045	0.659
Grand Trunk	30 m	12	65	1.885	2.485	0.759
	150 m	10	40	1.95	2.303	0.847
	300 m	13	205	1.618	2.565	0.631
	total	20	310	2.119	2.996	0.707
Heritage Landing	10 m	14	143	1.881	2.639	0.713
	30 m	12	74	2.054	2.485	0.827
	60 m	NA	NA	NA	NA	NA
	total	18	217	2.088	2.89	0.722
Kirksey	30 m	15	306	1.115	2.708	0.412
	60 m	8	385	0.43	2.079	0.207
	100 m	9	145	1.007	2.197	0.458
	total	18	836	0.869	2.89	0.301

Fish

We captured 1440 fish comprising 24 species at seven sites in Muskegon Lake during August 2019 (Table 21). The most abundant fish species across all sites were largemouth bass (39%), bluegill (18%), pumpkinseed (11%), round goby (7%), rock bass (6%), and brook silverside (6%), which accounted for nearly 87% of the total catch (Table 21). Of the 24 fish species captured in 2019, two species were non-native to the Great Lakes basin (Bailey et al. 2004)—round goby (7%) and white perch (<1%) (Table 21).

We observed differences in catch among the seven sites in 2019. Catch was highest at Kirksey followed by Circle Bay and Heritage Landing, whereas catch was lowest at NW Reference (Table 22). The environmental variables we measured during fish sampling (Table 23) did seem to be associated with total catch with the exception that dissolved oxygen concentration was lowest at NW Reference, which also had the lowest total catch. Largemouth bass was the most common species in the catch at every site except Grand Trunk, where gizzard shad was the most common (Table 22). Bluegill was the second most common species in the catch at three sites (Circle Bay, Heritage Landing, and Kirksey), whereas the next most common species at the other sites were rock bass (Grand Trunk), yellow perch (Amoco), pumpkinseed (NE Reference), and brook silverside (NW Reference) (Table 22).

The IBI scores at the seven sites ranged from 42-49 in 2019 (Fig. 18) with a mean IBI score of 45 (Fig. 19). The mean IBI score in 2019 was above the Muskegon Lake Area of Concern target of 36 (Fig. 19) that was established for two fish-related beneficial use impairments: loss of fish habitat and degradation of fish populations. Based on the fish-based IBI, we did not observe a clear pattern that the fish assemblage positively responded to restoration activities at the five restoration sites (Fig. 18).

Table 21. Combined total catch and total length (TL) of fish captured at seven sites in Muskegon Lake using fyke nets (*n* = 21 nets) during August 2019.

Common name	Scientific name	**Catch**	TL (cm) (mean, min, max)		
alewife	*Alosa pseudoharengus*	**1**	7.9	7.9	7.9
rock bass	*Ambloplites rupestris*	**92**	11.51	4.2	21.6
yellow bullhead	*Ameiurus natalis*	**14**	21.59	3.9	31.8
brown bullhead	*Ameiurus nebulosus*	**4**	13.9	6.5	20.7
bowfin	*Amia calva*	**11**	59.62	46.5	68
common carp	*Cyprinus carpio*	**1**	52.1	52.1	52.1
gizzard shad	*Dorosoma cepedianum*	**23**	6.152	5.2	7
banded killifish	*Fundulus diaphanus*	**2**	8.4	8.1	8.7
channel catfish	*Ictalurus punctatus*	**1**	65	65	65
brook silverside	*Labidesthes sicculus*	**89**	5.447	3.8	9.5
pumpkinseed	*Lepomis gibbosus*	**160**	5.467	2.9	17.8
warmouth	*Lepomis gulosus*	**3**	5.967	3.1	11.5
bluegill	*Lepomis macrochirus*	**254**	4.769	2.3	18.5
longnose gar	*Lepisosteus osseus*	**5**	46.22	26.7	61.7
smallmouth bass	*Micropterus dolomieu*	**1**	6.2	6.2	6.2
largemouth bass	*Micropterus salmoides*	**564**	6.012	3.7	10.5
white perch	*Morone americana*	**1**	18.9	18.9	18.9
silver redhorse	*Moxostoma anisurum*	**2**	57.15	56	58.3
golden redhorse	*Moxostoma erythrurum*	**1**	48.1	48.1	48.1
round goby	*Neogobius melanostomus*	**94**	6.871	2.7	12.2
golden shiner	*Notemigonus crysoleucas*	**2**	5.95	5.9	6
yellow perch	*Perca falvescens*	**76**	12.98	9.3	28.9
bluntnose minnow	*Pimephales notatus*	**37**	7.132	4	8.7
black crappie	*Pomoxis nigromaculatus*	**2**	6.05	5.8	6.3
	Total	**1440**			

Table FF. Number and total length (TL; cm) of fish captured by fyke netting (n = 3 nets per site) at seven sites in Muskegon Lake during August 2019. Site locations are depicted in Figure 1.

		Grand Trunk				Amoco				Kirksey				Circle Bay			
Common name	*Scientific name*	**Catch**	TL (mean, min, max)			**Catch**	TL (mean, min, max)			**Catch**	TL (mean, min, max)			**Catch**	TL (mean, min, max)		
alewife	*Alosa pseudoharengus*	**0**	--	--	--	**0**	--	--	--	**1**	7.9	7.9	7.9	**0**	--	--	--
rock bass	*Ambloplites rupestris*	**21**	11.96	7.3	21.6	**19**	9.64	4.2	18.2	**14**	13.7	7.4	18.8	**9**	13.7	7.7	21.5
yellow bullhead	*Ameiurus natalis*	**5**	21.7	11.4	30.7	**1**	21.5	21.5	21.5	**1**	16	16	16	**1**	28.8	28.8	28.8
brown bullhead	*Ameiurus nebulosus*	**0**	--	--	--	**0**	--	--	--	**2**	7.15	6.5	7.8	**0**	--	--	--
bowfin	*Amia calva*	**3**	61.57	60	63.2	**0**	--	--	--	**1**	62.5	62.5	62.5	**3**	58.9	48.5	67.8
common carp	*Cyprinus carpio*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
gizzard shad	*Dorosoma cepedianum*	**23**	6.152	5.2	7	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
banded killifish	*Fundulus diaphanus*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
channel catfish	*Ictalurus punctatus*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
brook silverside	*Labidesthes sicculus*	**0**	--	--	--	**8**	6.3	4.6	9.5	**63**	5.41	4.5	8.9	**0**	--	--	--
pumpkinseed	*Lepomis gibbosus*	**15**	7.947	3.2	15.1	**18**	5.63	3.1	17.2	**56**	4.02	2.9	17.5	**12**	4.85	3.2	8.8
warmouth	*Lepomis gulosus*	**0**	--	--	--	**1**	3.3	3.3	3.3	**1**	3.1	3.1	3.1	**0**	--	--	--
bluegill	*Lepomis macrochirus*	**19**	3.247	2.9	4.2	**19**	3.63	2.7	6.8	**116**	3.77	2.3	17.8	**33**	6.12	2.9	9.9
longnose gar	*Lepisosteus osseus*	**1**	28.1	28.1	28.1	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
smallmouth bass	*Micropterus dolomieu*	**1**	6.2	6.2	6.2	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
largemouth bass	*Micropterus salmoides*	**16**	6.275	3.9	9.8	**24**	5.98	3.7	8.2	**129**	6.04	4.1	10.2	**159**	6.44	4.1	10.5
white perch	*Morone americana*	**1**	18.9	18.9	18.9	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
silver redhorse	*Moxostoma anisurum*	**1**	58.3	58.3	58.3	**1**	56	56	56	**0**	--	--	--	**0**	--	--	--
golden redhorse	*Moxostoma erythrurum*	**0**	--	--	--	**1**	48.1	48.1	48.1	**0**	--	--	--	**0**	--	--	--
round goby	*Neogobius melanostomus*	**10**	9.64	8.2	12.2	**6**	3.63	2.7	5	**27**	8.02	5.1	12.2	**28**	5.69	3.1	9.5
golden shiner	*Notemigonus crysoleucas*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--	**2**	5.95	5.9	6
yellow perch	*Perca falvescens*	**10**	13.83	10.1	19.1	**22**	13.5	9.6	23	**2**	13.2	12.5	13.9	**17**	11.9	10.3	17.5
bluntnose minnow	*Pimephales notatus*	**0**	--	--	--	**9**	6.54	5.8	7.8	**19**	7.19	4	8.6	**9**	7.59	4.4	8.7
black crappie	*Pomoxis nigromaculatus*	**0**	--	--	--	**0**	--	--	--	**1**	6.3	6.3	6.3	**0**	--	--	--
	Total	**126**				**129**				**433**				**273**			

		Scrap Bay				NE Reference				NW Reference			
Common name	*Scientific name*	**Catch**	TL (mean, min, max)			**Catch**	TL (mean, min, max)			**Catch**	TL (mean, min, max)		
alewife	*Alosa pseudoharengus*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
rock bass	*Ambloplites rupestris*	**17**	11.5	6.5	19.6	**11**	9.3	4.6	16.3	**1**	12.3	12.3	12.3
yellow bullhead	*Ameiurus natalis*	**2**	28.2	26.2	30.1	**1**	7.0	7.0	7.0	**3**	21.4	3.9	31.8
brown bullhead	*Ameiurus nebulosus*	**2**	20.7	20.6	20.7	**0**	--	--	--	**0**	--	--	--
bowfin	*Amia calva*	**1**	68.0	68.0	68.0	**1**	58.1	58.1	58.1	**2**	53.0	46.5	59.4
common carp	*Cyprinus carpio*	**0**	--	--	--	**1**	52.1	52.1	52.1	**0**	--	--	--
gizzard shad	*Dorosoma cepedianum*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
banded killifish	*Fundulus diaphanus*	**2**	8.4	8.1	8.7	**0**	--	--	--	**0**	--	--	--
channel catfish	*Ictalurus punctatus*	**0**	--	--	--	**1**	65.0	65.0	65.0	**0**	--	--	--
brook silverside	*Labidesthes sicculus*	**0**	--	--	--	**3**	4.3	3.8	4.9	**15**	5.4	5.0	5.7
pumpkinseed	*Lepomis gibbosus*	**24**	7.5	3.8	13.2	**31**	4.7	3.1	17.8	**4**	11.6	7.3	16.6
warmouth	*Lepomis gulosus*	**0**	--	--	--	**0**	--	--	--	**1**	11.5	11.5	11.5
bluegill	*Lepomis macrochirus*	**54**	6.1	2.7	9.9	**7**	8.9	3.1	18.5	**6**	8.5	3.3	14.2
longnose gar	*Lepisosteus osseus*	**0**	--	--	--	**1**	26.7	26.7	26.7	**3**	58.8	54.4	61.7
smallmouth bass	*Micropterus dolomieu*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
largemouth bass	*Micropterus salmoides*	**135**	5.8	4.2	10.0	**47**	5.5	4.2	7.9	**54**	5.7	4.9	6.8
white perch	*Morone americana*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
silver redhorse	*Moxostoma anisurum*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
golden redhorse	*Moxostoma erythrurum*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
round goby	*Neogobius melanostomus*	**6**	6.7	4.6	12.2	**17**	6.6	3.9	11.1	**0**	--	--	--
golden shiner	*Notemigonus crysoleucas*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
yellow perch	*Perca falvescens*	**15**	13.1	9.3	28.9	**6**	12.9	10.5	18.3	**4**	11.9	10.1	16.1
bluntnose minnow	*Pimephales notatus*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
black crappie	*Pomoxis nigromaculatus*	**0**	--	--	--	**1**	5.8	5.8	5.8	**0**	--	--	--
	Total	**258**				**128**				**93**			

Table 23. Mean ±1 SE (*n* = 3) of environmental conditions at each fish sampling site in Muskegon Lake. Environmental conditions were submerged aquatic vegetation cover (SAV), water temperature (Temp), dissolved oxygen (DO), specific conductivity (SPC), turbidity (Turb), pH, oxidation reduction potential (ORP), and chlorophyll *a* (Chl-*a*). Measurements were made during fyke netting in August 2019 with a YSI sonde (except SAV was estimated visually).

Site	SAV (%)	Temp (°C)	DO (mg/L)	DO (%)	SPC (µS/cm)	TDS (g/L)	Turb (NTU)	pH	ORP (mV)	Chl-*a* (µg/L)
NW Reference	77 ± 2	24.22 ± 0.04	2.47 ± 0.18	29.6 ± 2.1	364 ± 0	0.24 ± 0.00	1.5 ± 0.3	7.40 ± 0.01	329.5 ± 10.1	11.8 ± 3.7
NE Reference	78 ± 17	22.09 ± 0.04	6.37 ± 0.15	73.1 ± 1.8	367 ± 2	0.24 ± 0.00	0.2 ± 0.1	7.79 ± 0.04	336.5 ± 2.3	6.1 ± 0.3
Amoco	82 ± 6	24.67 ± 0.10	6.77 ± 0.16	81.4 ± 2.2	412 ± 4	0.27 ± 0.00	0.1 ± 0.1	7.92 ± 0.02	330.1 ± 1.2	7.4 ± 1.0
Circle Bay	50 ± 13	23.09 ± 0.05	5.93 ± 0.43	69.3 ± 5.2	377 ± 1	0.25 ± 0.00	0.7 ± 0.1	7.77 ± 0.07	329.6 ± 5.6	12.4 ± 1.0
Grand Trunk	42 ± 9	24.77 ± 0.06	5.41 ± 0.07	65.3 ± 1.0	368 ± 1	0.24 ± 0.00	0.5 ± 0.0	7.89 ± 0.01	325.7 ± 1.0	8.3 ± 0.6
Heritage Landing	73 ± 22	24.29 ± 0.12	5.40 ± 0.78	64.6 ± 9.2	381 ± 1	0.25 ± 0.00	0.6 ± 0.3	7.74 ± 0.07	333.1 ± 1.4	9.3 ± 1.3
Kirksey	23 ± 9	22.94 ± 0.37	7.69 ± 0.27	90.2 ± 3.2	367 ± 1	0.24 ± 0.00	0.3 ± 0.2	8.01 ± 0.03	345.5 ± 3.7	7.2 ± 0.4

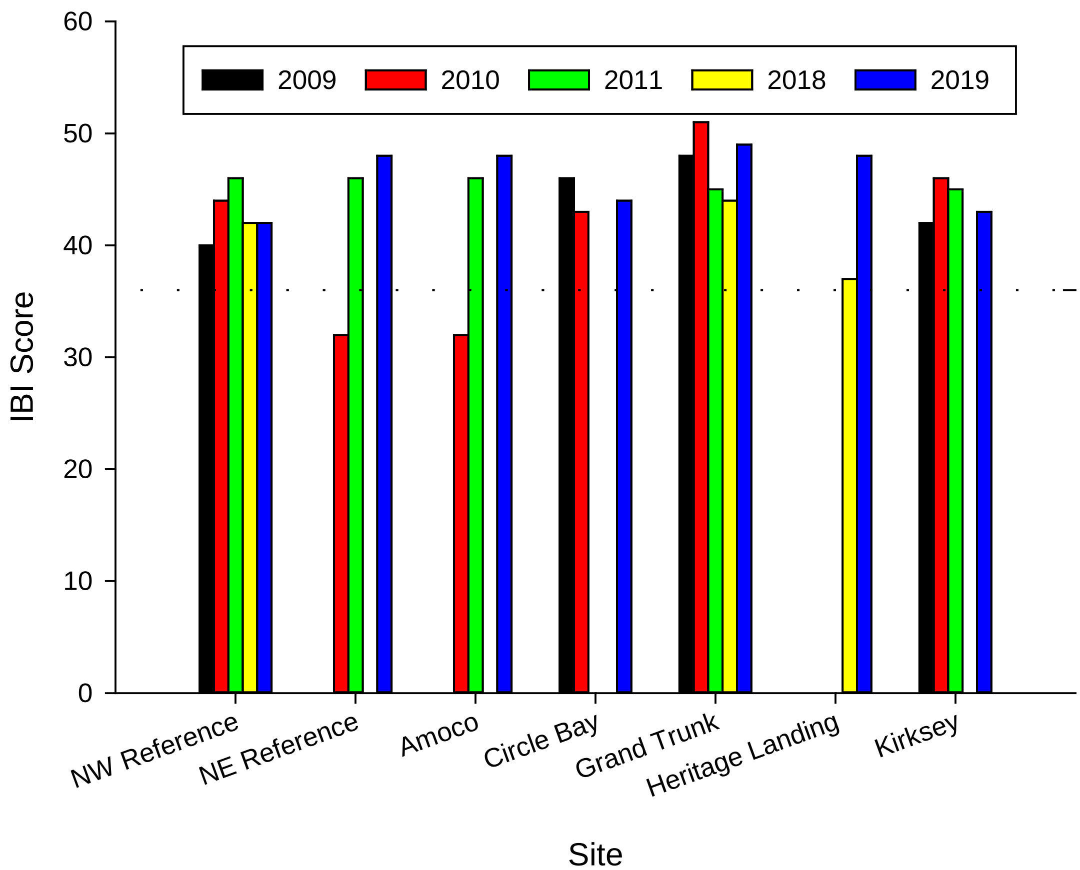

Figure 18. Scores from the fish-based index of biotic integrity (IBI) for three sampling sites in Muskegon Lake. The dashed line represents the numerical delisting target of 36 for the Muskegon Lake Area of Concern (Appendix B). The absence of a bar indicates that site was not sampled during that year.

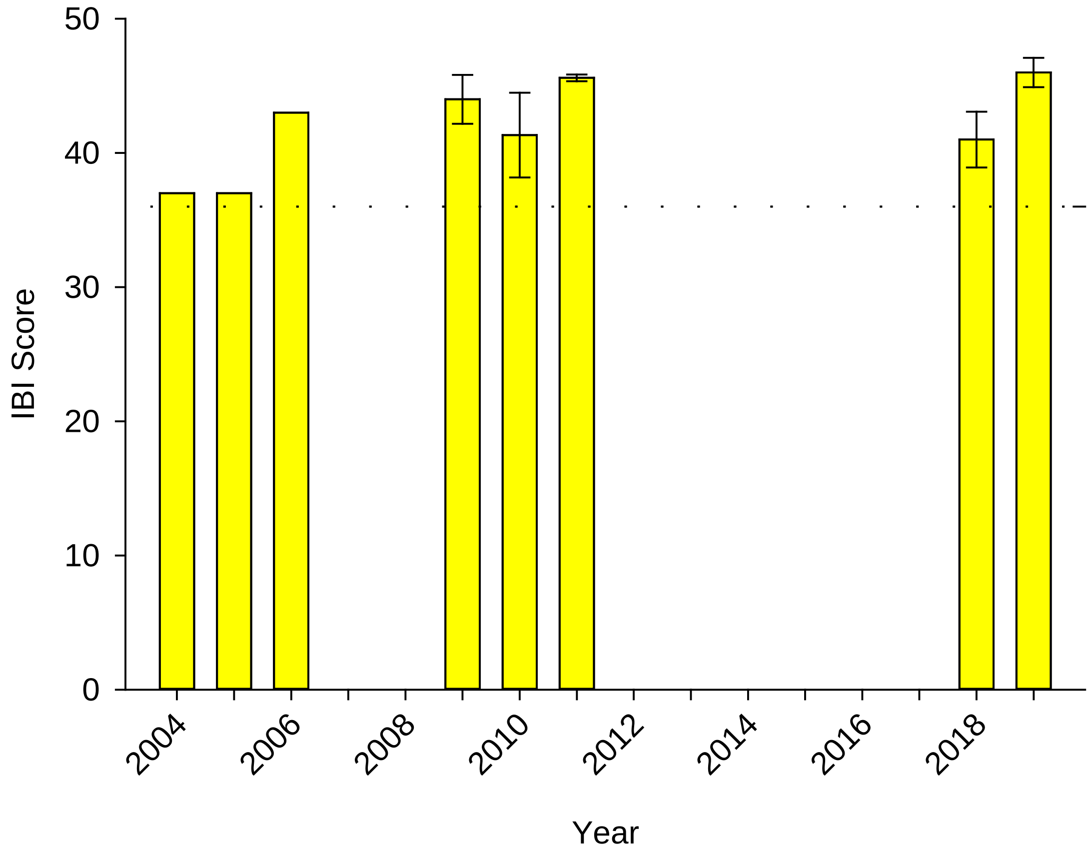

Figure 19. Scores from the fish-based index of biotic integrity (IBI) for Muskegon Lake. The dashed line represents the numerical delisting target of 36 for the Muskegon Lake Area of Concern (Appendix B). Bars are missing for years without fish data. The IBI scores calculated for 2004-2006 were based on sampling site (see Cooper et al. [2007a] for location of sites with submerged aquatic vegetation) that was not part of fish sampling in later years. Mean values (±1 standard error) were reported for 2009 (*n* = 4 sites), 2010 (*n* = 6 sites), 2011 (*n* = 5 sites), 2018 (*n* = 3 sites), and 2019 (*n* = 7 sites).

Results – Veterans Park

Water Quality

Across the 11 fish sampling sites, mean water depth was 88 cm with a mean water temperature of 16.0 °C in 2015, water depth was 81 cm with a mean water temperature of 24.5 °C in 2018, and water depth was 99 cm with a mean water temperature of 15.3 °C in 2019 (Table 24). The marked difference in water temperature between years is because sampling was conducted in August (2018) versus October (2015 and 2019). This difference in water temperature could confound pre- versus post-restoration comparisons of the fish assemblage when comparing 2015 versus 2018.

We found differences in environmental conditions among sampling locations (i.e., north pond, south pond, and Muskegon Lake) and between years (Table 24; Fig. 20). Turbidity and TP markedly declined in the south pond between 2015 and 2018-2019, whereas while there was little difference between years in Muskegon Lake (Figs. 20a and 20c), suggesting the differences were the result of habitat restoration (i.e., the reconnection of the south pond to the Muskegon River in December 2017 before the temporary barrier was installed in May 2019). A similar pattern in the south pond was apparent for SRP, although there were large increases in SRP in the north pond and Muskegon Lake in 2019 (Fig. 20d). Nevertheless, specific conductivity and Cl markedly increased in the south pond in 2019 (Figs. 20b and 20f), which could be due to a lack of water exchange between the south pond and the Muskegon River once the barrier was installed in May 2019. There was not a clear pattern in nitrate concentration among years that was associated with restoration activities, although nitrate concentrations appeared to increase over time in the north and south ponds (Fig. 20e). Comparing nutrient concentrations across years needs to be done with caution given that sampling occurred in during different months. The lower SRP concentrations in 2018, especially in the south pond, may be related, at least in part, to the August sampling date when temperatures were warmer and algal uptake would be stronger.

We observed differences in SAV and EAV among sites and between years (Fig. 21). These differences should be interpreted cautiously because the percentages of SAV and EAV are a function of where fyke nets were set, which was strongly influenced by water depth. Nevertheless, we observed more SAV in the south pond in 2018-2019 than in 2015 (Fig. 21a). This observation could be the result of decreased turbidity in the south pond (Fig. 20a), which likely allowed greater light penetration to the bottom for plant growth. The loss of emergent vegetation in 2018 and 2019 in the Lake and South Pond is consistent with the results at NW Ref in Muskegon Lake, being replaced with submerged vegetation, likely in response to higher water levels.

Fish

We captured 833 fish comprising 17 species at the three (north pond, south pond, and Muskegon Lake) sampling locations in 2019 (Table 25). The pattern in total catch per unit effort (CPUE) among sites was mostly consistent among years (Fig. 22). The reported species richness excludes unknown sunfish, which were likely a hybrid. The most abundant fishes across all sites and years were largemouth bass (21%), bluegill (21%), yellow perch (20%), pumpkinseed (19%), black crappie (8%), warmouth (4%), bullheads (3%), and rock bass (1%), which accounted for nearly 96% of the total catch (Fig. 23). Of the 17 fish species captured in 2019, one species was non-native to the Great Lakes basin (Bailey et al. 2004)—round goby—which composed about 2% of the total catch in 2019 (Table 25).

We observed differences in the fish assemblage among years and among sites (Fig. 23). As noted above, differences among years should be interpreted cautiously because timing of sampling (August versus October) and the installation of the barrier between the south pond and the Muskegon River in May 2019 (prior to 2019 fish sampling). Moreover, the fish assemblage at our sampling sites in Muskegon Lake varied among years, with more rock bass, largemouth bass, and yellow perch (and less pumpkinseed) in 2018 than 2015 and more bluegill in 2019 (Fig. 23c). Given that Muskegon Lake was our "control" site (meaning no habitat restoration was performed at this site), inferring the effects of the restoration efforts at the north and south ponds should be done cautiously. Overall, we did not observe marked differences in the fish assemblage in the north pond between years (Fig. 23a), which is not surprising given that restoration activities were of a lesser scope than in the south pond. However, the south pond showed differences among years, but variation between 2018 and 2019 make interpretation with respect to restoration activities difficult. Compared with 2015, largemouth bass were more common in 2018 and 2019, whereas black crappie (in 2018) and bluegill (in 2019) were only common during one year of post-restoration monitoring. Nevertheless, bullheads were about 10% of the catch in the south pond in 2015 but were absent from the catch in 2018 and nearly absent in 2019 (Fig. 23b; Table 25). Similarly, goldfish and to a lesser degree common carp (both species are non-native) were captured in the south pond in 2015 but were absent from the catch in 2018 (Ruetz and Ellens 2018) and 2019 (Table 25).

Table 24. Mean ± 1 standard error (SE; *n* = 2) of environmental conditions measured during fyke netting for pre-restoration monitoring in October 2015 and post-restoration monitoring in August 2018 and October 2019. SAV is submerged aquatic vegetation, and EAV is emergent aquatic vegetation. Water depth was measured at each fyke net. Water quality variables were measured in situ with a YSI sonde. Negative turbidity measurements should be interpreted as zero. SAV and EAV were estimated visually.

Location	Site	Year	Depth (cm)	Water Temperature (°C)	Dissolved Oxygen (mg/L)	% Dissolved Oxygen	Specific Conductivity (µS/cm)	Total Dissolved Solids (g/L)	Turbidity (NTU)	pH	Chlorophyll *a* (µg/L)	SAV (%)	EAV (%)
Muskegon Lake	1	2015	74±7	17.58±0.24	8.40±0.45	88.1±4.5	432±1	0.285±0.003	1.1±0.9	7.98±0.06	3.4±0.1	50±10	0±0
Muskegon Lake	2	2015	78±5	16.05±0.01	9.14±0.06	92.8±0.6	432±0	0.281±0.000	1.3±0.5	8.15±0.02	3.4±0.1	40±10	0±0
North pond	B	2015	98±4	14.62±0.00	6.57±0.13	65.0±1.5	502±0	0.327±0.001	0.8±1.2	7.50±0.00	3.8±0.4	53±3	45±5
North pond	C	2015	95±6	14.64±0.03	5.63±0.21	55.6±2.1	520±0	0.338±0.000	-0.6±0.2	7.46±0.01	7.0±1.6	20±0	25±0
North pond	D	2015	96±2	14.67±0.00	6.11±0.13	60.3±1.3	496±1	0.323±0.001	-0.7±0.0	7.58±0.01	5.2±0.6	15±5	25±0
South pond	1-B	2015	81±8	16.58±0.60	12.38±0.07	127.3±1.4	544±1	0.354±0.001	26.9±1.6	8.15±0.04	18.3±0.3	0±0	50±0
South pond	1-D	2015	90±4	15.80±0.18	10.12±0.71	102.4±7.6	558±6	0.363±0.004	22.8±0.3	7.75±0.02	16.3±0.9	0±0	20±0
South pond	2-B	2015	92±13	16.57±0.23	13.97±0.06	142.7±1.2	540±0	0.351±0.000	29.7±0.8	8.47±0.02	19.3±0.9	0±0	0±0
South pond	2-D	2015	87±0	16.17±0.03	11.75±0.17	120.0±1.1	547±3	0.356±0.001	28.9±5.1	8.00±0.01	16.9±1.2	0±0	48±3
South pond	3-A	2015	87±3	16.78±0.06	14.58±0.26	149.5±3.4	536±0	0.348±0.000	25.3±1.4	8.54±0.01	14.6±2.0	0±0	0±0
South pond	3-D	2015	93±4	16.60±0.11	15.51±0.65	153.5±0.7	536±0	0.349±0.001	28.9±1.0	8.53±0.00	17.8±1.0	0±0	33±3
Muskegon Lake	1	2018	99±4	24.59±0.01	10.18±0.01	122.3±0.1	388±0	0.252±0.000	0.6±0.2	8.22±0.02	5.8±0.8	8±3	0±0
Muskegon Lake	2	2018	92±1	24.11±0.01	8.03±0.05	95.7±0.7	390±0	0.254±0.000	0.3±0.1	7.85±0.02	4.6±0.1	0±0	0±0
North Pond	B	2018	75±6	24.25±0.01	1.25±0.07	15.1±0.9	401±0	0.261±0.000	2.1±0.5	7.46±0.01	65.4±11.0	80±0	83±8
North Pond	C	2018	77±8	24.29±0.02	1.44±0.68	17.3±8.1	411±3	0.268±0.002	5.7±1.2	7.46±0.04	133.5±16.5	60±0	10±5
North Pond	D	2018	72±0	24.68±0.01	4.67±0.44	56.3±5.2	397±3	0.258±0.002	7.5±0.6	7.75±0.06	39.4±8.9	50±0	0±0
South Pond	1-B	2018	87±8	24.41±0.01	7.75±0.04	92.9±0.5	392±0	0.255±0.000	2.2±0.7	8.13±0.02	16.3±0.1	20±5	0±0
South Pond	1-D	2018	70±2	24.42±0.04	8.01±0.02	96.0±0.4	392±0	0.255±0.000	0.6±0.0	8.16±0.00	15.2±0.6	5±0	0±0
South Pond	2-B	2018	76±2	24.68±0.01	7.77±0.21	93.6±2.5	393±0	0.256±0.000	2.0±0.7	8.12±0.02	18.3±0.2	5±0	0±0
South Pond	2-D	2018	74±6	24.61±0.00	8.02±0.02	96.5±0.3	393±1	0.255±0.000	1.5±0.5	8.12±0.02	15.6±0.1	10±0	0±0
South Pond	3-A	2018	90±3	24.67±0.03	7.50±0.09	90.3±1.1	394±0	0.256±0.000	1.9±0.6	8.10±0.01	17.6±0.5	35±5	0±0
South Pond	3-D	2018	80±3	24.70±0.00	7.47±0.08	89.9±0.9	395±0	0.257±0.001	2.2±0.6	8.10±0.01	16.6±0.1	25±15	0±0
Muskegon Lake	1	2019	134±11	15.12±0.02	9.08±0.06	90.4±0.6	390±0	0.234±0.001	1.2±0.1	7.64±0.04	7.6±0.1	45±15	0±0
Muskegon Lake	2	2019	137±2	14.69±0.04	7.07±0.36	69.8±3.6	391±1	0.254±0.001	1.5±0.3	7.48±0.11	7.4±0.5	30±10	0±0
North Pond	B	2019	89±7	15.18±0.02	6.82±0.61	68.1±6.2	412±1	0.268±0.001	0.4±0.4	7.44±0.04	7.2±0.1	25±0	50±0
North Pond	C	2019	84±1	15.12±0.08	7.55±0.30	75.3±3.0	421±3	0.274±0.002	0.1±0.1	7.52±0.02	10.4±0.2	15±0	0±0
North Pond	D	2019	97±6	15.20±0.02	7.90±0.08	78.8±0.8	414±2	0.270±0.001	0.8±0.2	7.52±0.04	19.2±3.2	25±0	0±0
South Pond	1-B	2019	96±6	15.68±0.06	10.84±0.06	109.2±0.6	688±6	0.447±0.004	4.1±0.1	7.68±0.04	24.6±2.6	32±2	0±0
South Pond	1-D	2019	92±4	14.71±0.02	10.14±0.20	100.2±1.9	679±5	0.442±0.004	5.0±0.8	7.61±0.07	29.0±1.9	70±0	0±0
South Pond	2-B	2019	86±5	15.84±0.01	11.72±0.09	118.6±0.9	732±2	0.476±0.001	5.2±0.4	7.72±0.01	37.6±2.8	50±0	0±0
South Pond	2-D	2019	98±0	14.98±0.03	10.64±0.21	105.7±2.0	720±2	0.468±0.001	5.0±0.4	7.68±0.01	30.3±0.8	70±0	0±0
South Pond	3-A	2019	94±2	16.15±0.13	12.17±0.10	124.1±1.4	742±0	0.482±0.000	6.6±1.3	7.76±0.01	42.2±0.6	15±0	0±0
South Pond	3-D	2019	87±7	15.26±0.08	11.00±0.12	110.0±0.9	732±1	0.476±0.001	5.3±0.3	7.72±0.01	34.3±0.5	70±0	0±0

Table 25. Number and mean total length (TL; ranges reported parenthetically) of fish captured by fyke netting (*n* = 22 nets) at three locations during post-restoration monitoring in October 2019. The sampling locations were Muskegon Lake (*n* = 4 nets), north pond (*n* = 6 nets), and south pond (*n* = 12 nets).

		Total	Lake		North		South	
Common name	Scientific name	Catch	Catch	TL (cm)	Catch	TL (cm)	Catch	TL (cm)
rock bass	*Ambloplites rupestris*	3	3	13.1 (10.7-17.1)	0	--	0	--
yellow bullhead	*Ameiurus natalis*	3	2	17.9 (7.8-28.0)	1	31.4	0	--
brown bullhead	*Ameiurus nebulosus*	1	0	--	0	--	1	11.6
bowfin	*Amia calva*	13	2	70.6 (69.0-72.3)	10	38.9 (34.3-49.3)	1	56.8
spotfin shiner	*Cyprinella spiloptera*	2	0	--	0	--	2	7.5 (6.7-8.3)
brook silverside	*Labidesthes sicculus*	1	0	--	0	--	1	7.9
pumpkinseed	*Lepomis gibbosus*	125	9	12.0 (3.6-19.4)	14	10.5 (5.1-20.1)	102	10.5 (4.5-15.5)
warmouth	*Lepomis gulosus*	17	0	--	9	14.5 (8.8-20.1)	8	9.6 (4.8-18.1)
bluegill	*Lepomis macrochirus*	405	34	7.5 (3.4-19.4)	30	11.2 (5.6-17.7)	341	6.7 (3.3-16.3)
unknown sunfish	*Lepomis* spp.*	12	0	--	2	12.9 (9.7-16.1)	10	15.7 (9.2-18.4)
largemouth bass	*Micropterus salmoides*	177	3	6.7 (5.8-7.3)	22	8.0 (6.9-12.6)	152	6.7 (5.3-27.7)
round goby	*Neogobius melanostomus*	18	0	--	0	--	18	6.9 (3.8-11.9)
yellow perch	*Perca flavescens*	51	5	16.6 (10.3-23.3)	18	14.4 (11.2-19.3)	28	10.9 (7.8-18.6)
black crappie	*Pomoxis nigromaculatus*	3	0	--	0	--	3	21.1 (13.7-25.2)
Central mudminnow	*Umbra limi*	2	0	--	1	7.2	1	7.7
	Total	**833**	**58**		**107**		**668**	

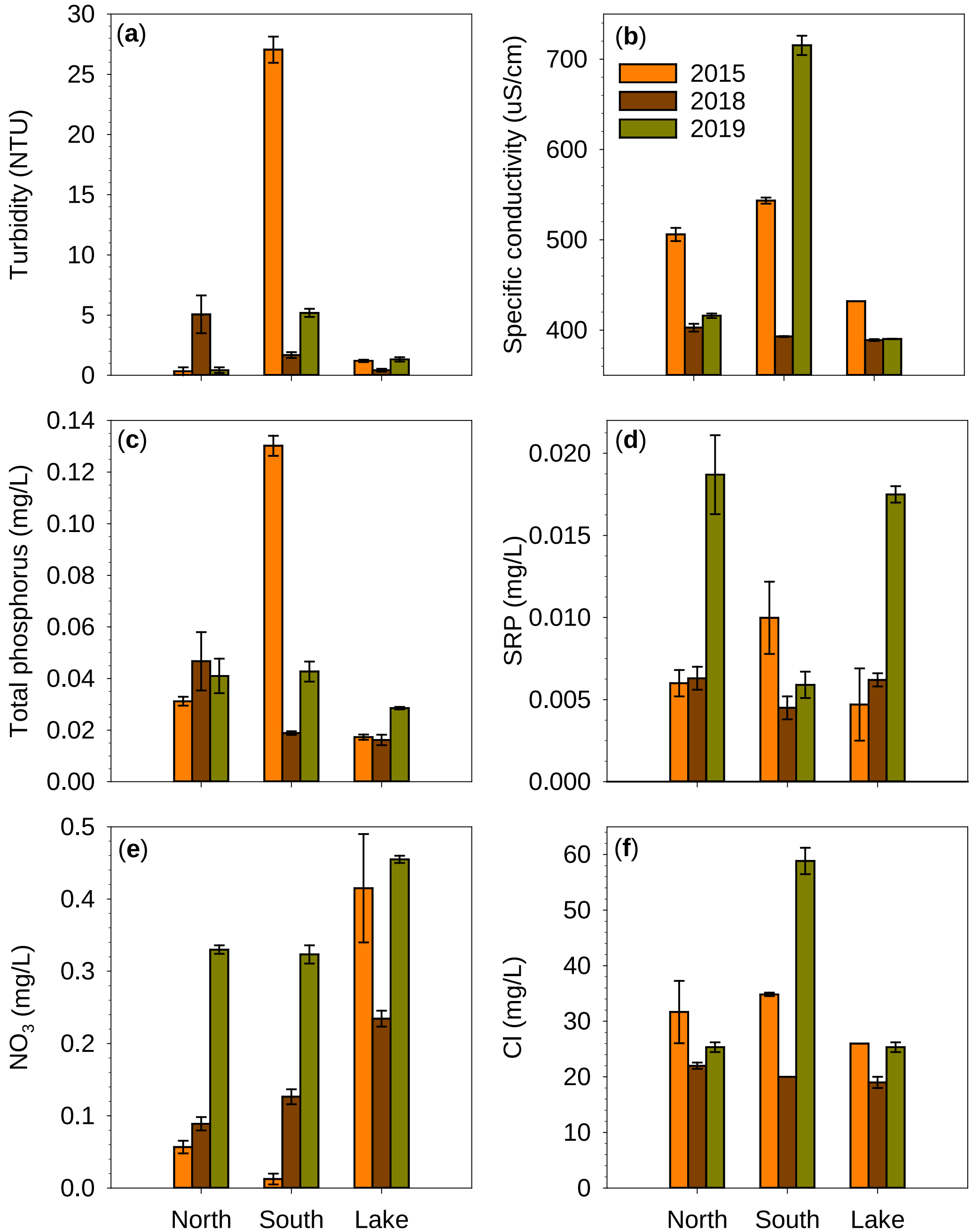

Figure 20. Mean (±1 SE) (a) turbidity, (b) specific conductivity, (c) total phosphorus, (d) soluble reactive phosphorus (SRP), (e) nitrate (NO_3), and (f) chloride (Cl) in the north pond (*n* = 3 sites), south pond (*n* = 6 sites), and Muskegon Lake (*n* = 2 sites). Pre-restoration sampling was conducted in October 2015, and post-restoration sampling was conducted in August 2018 and October 2019. Note that negative values of turbidity were assumed to be zero for calculating mean and SE (see Table 24). If values were less than the detection limit for NO_3 or SRP, then a value of 0.5×detection limit was used to calculate means and SE.

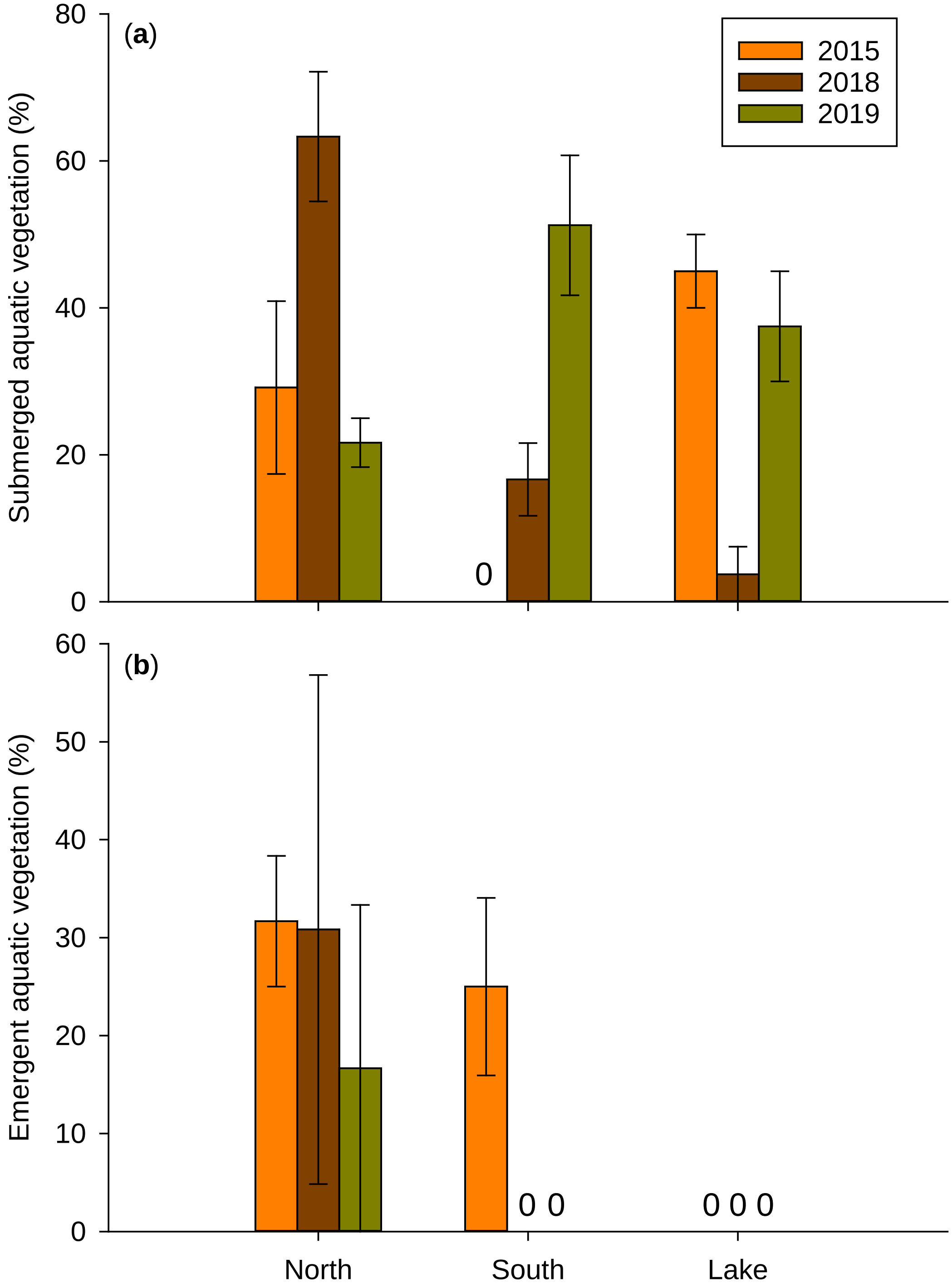

Figure 21. Mean (±1 SE) percent coverage of (a) submerged aquatic vegetation (SAV) and (b) emergent aquatic vegetation (EAV) in the north pond (*n* = 3 sites), south pond (*n* = 6 sites), and Muskegon Lake (*n* = 2 sites) in 2015 (pre-restoration) and 2018-2019 (post-restoration). Zeros indicate the absence of SAV or EAV at a sampling location. Estimates were made at fyke-net locations.

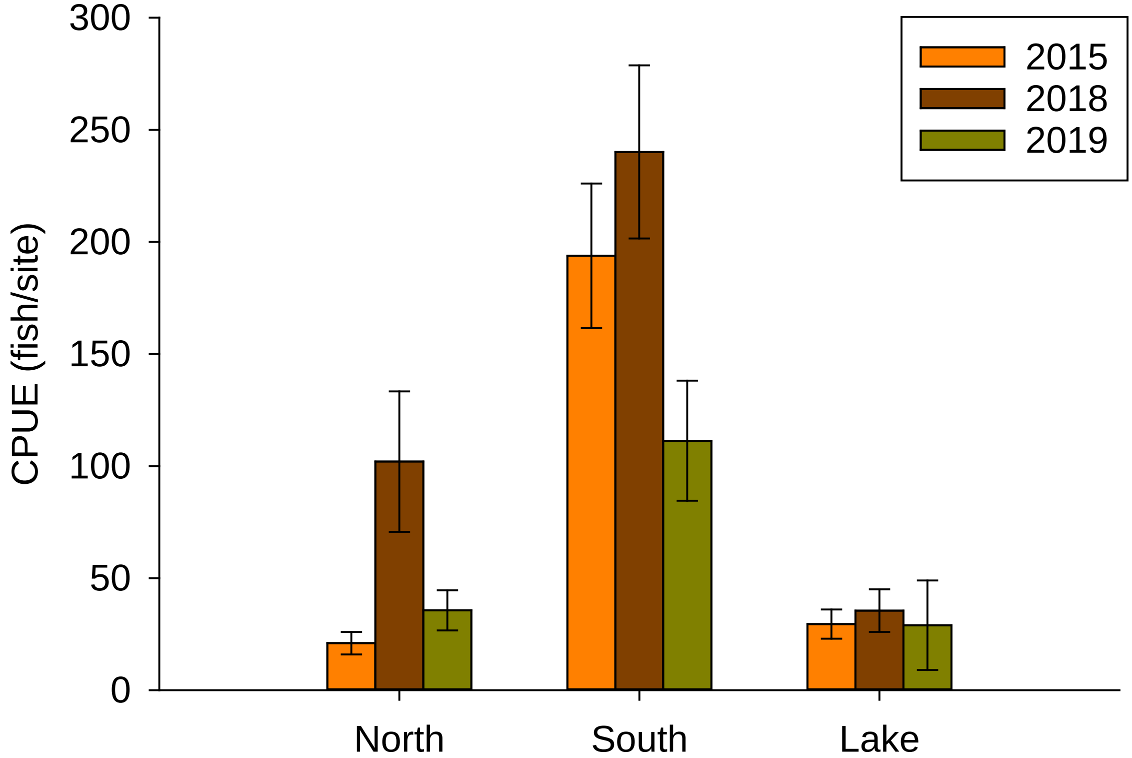

Figure 22. Mean (±1 SE) total catch per unit effort (CPUE) in the north pond (*n* = 3 sites), south pond (*n* = 6 sites), and Muskegon Lake (*n* = 2 sites) in 2015 (pre-restoration) and 2018-2019 (post-restoration). Two fyke nets were fished at each site; thus, CPUE is the total number of fish captured in two fyke nets at a site.

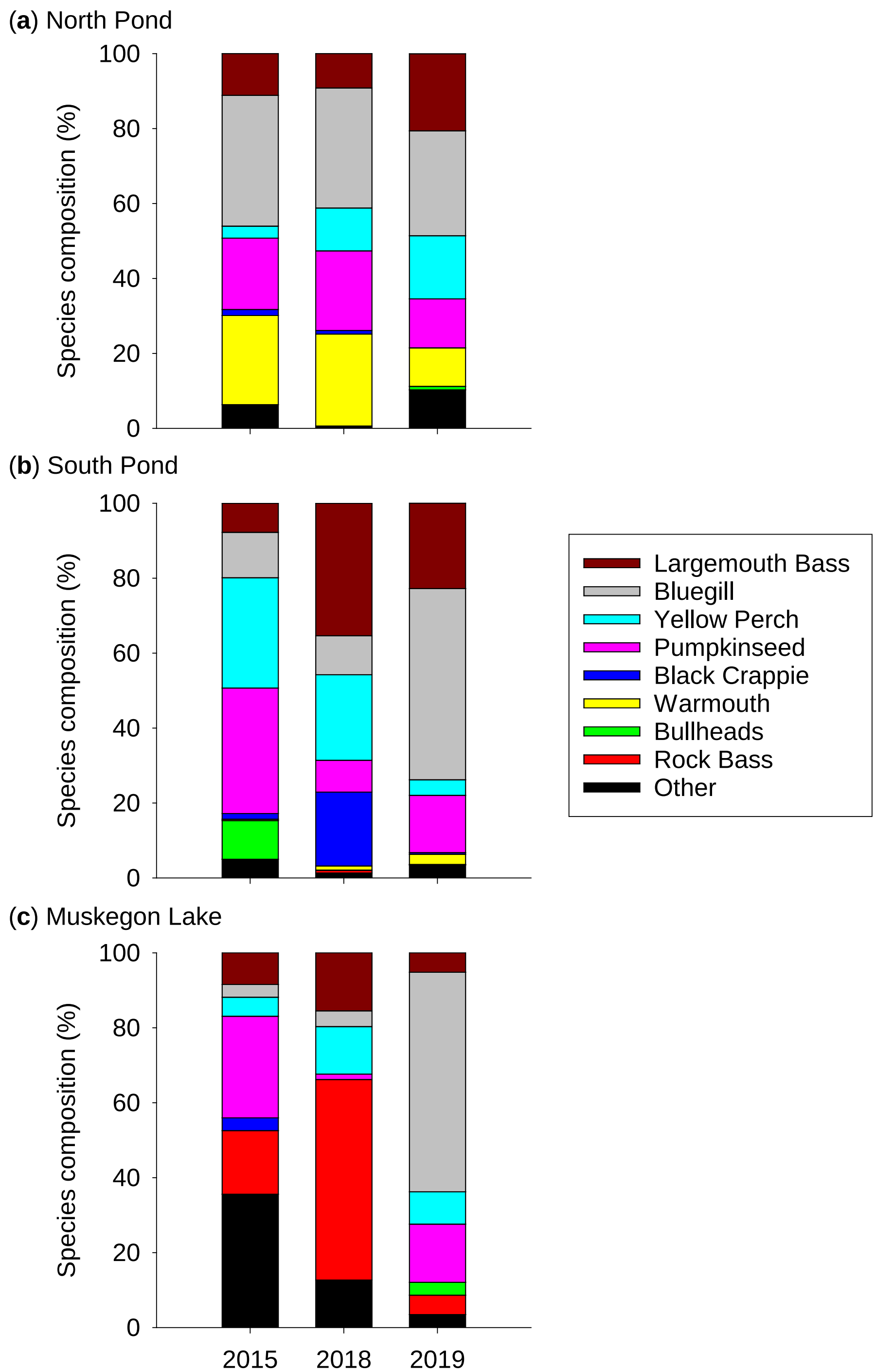

Figure 23. Fish species composition in the catch of the (a) north pond (n = 3 sites), (b) south pond (n = 6 sites), and (c) Muskegon Lake (n = 2 sites) in 2015 (pre-restoration) and 2018-2019 (post-restoration). “Other” includes all species not listed in the legend (see Table 25 for 2019 and Ruetz and Ellens [2018] for 2015 and 2018). Note that warmouth includes hybrid *Lepomis* and bullheads includes three species of *Ameiurus*. Two fyke nets were fished at each site. The number of fish captured varied among the three sampling locations (i.e., north pond, south pond, and Muskegon Lake) and between years.

Results – Bear Creek

Water Quality

Surface water quality at wetland sites (see Fig. 3 for site map) was compared across pre- and post-restoration measurements in Augusts of 2014, 2017, 2018, and 2019 to gauge trends in the years before, after, and following restoration and hydrologic reconnection in spring 2017 (Steinman and Ogdahl 2014, Hassett and Steinman 2018, Hassett and Steinman 2019).

West pond P concentrations remained low in 2019 with ~15 µg TP/L and 2.5-6 µg SRP/L, comparable to 2018 and two ordera of magnitude below 2014 (Table 26). Chl *a* concentrations were low (~5 µg/L) and similar to values observed in previous years (Table 26). West pond water temperature, DO, and pH in 2019 were similar to other post-restoration years (Table 26). SpCond and TDS were slightly higher in 2019 compared to 2017 and 2018, but remained well below pre-restoration 2014 values (Table 26). Turbidity slightly increased in 2019 and ranged only 0.2-0.4 NTU (Table 26).

East pond TP in 2019 increased relative to 2018, and exceeded the recommended 30 µg/L Bear Lake TMDL at site East 6. This may be related to inflows from Bear Creek, which previously averaged about 30 µg/L; SRP remained at low concentrations (Table 27). Chl *a* generally ranged from 5-7 µg/L in 2019 and was comparable to post-restoration 2017-18 measurements, except at site East 6 (35.4 µg/L; Table 27); this high concentration may be a result of backwater pooling resulting in conditions favorable for algal blooms. Physicochemical water quality trends were similar to the West pond (Table 27).

Table 26. Bear Creek West pond post-restoration surface water chemistry comparison between pre- (8/21/14) and post-restoration sampling (8/10/2017, 8/8/2018, 8/21/2019). Chl *a* = lab-extracted chlorophyll *a*; DO = dissolved oxygen; SpCond = specific conductivity; TDS = total dissolved solids; ORP = oxidation-reduction potential; ND = no data.

Parameter	West 1				West 5			
	2014	2017	2018	2019	2014	2017	2018	2019
TP (µg/L)	1261.2	39.5	12.6	14.7	1187.6	45.6	14	15.3
SRP (µg/L)	972.4	2.5	5.5	5.9	903.5	2.5	6.4	2.5
Chl *a* (µg/L)	7.63	6.5	8.81	5.6	8.14	6.7	6.97	4.8
Temp (°C)	23.95	23.79	25	23.5	23.92	23.95	25	23.8
DO (mg/L)	7.1	8.3	9.4	8.0	2.6	8.2	9.4	9.3
DO (%)	84.5	98.4	113.9	93.6	30.8	97.6	114.0	109.7
pH	8.45	8.31	8.73	7.98	7.74	8.30	8.79	8.24
SpCond (µS/cm)	742	399	398	419	750	402	398	418
TDS (g/L)	0.483	0.260	0.259	0.272	0.487	0.262	0.258	0.272
ORP (mV)	377	479	278	229	396	480	280	220
Turbidity (NTU)	3.3	ND	0.0	0.4	4.0	ND	0.0	0.2

Table 27. Bear Creek East pond post-restoration surface water chemistry comparison between pre- (8/21/14) and post-restoration sampling (8/10/2017, 8/8/2018, 8/21/2019). Chl *a* = lab-extracted chlorophyll *a*; DO = dissolved oxygen; SpCond = specific conductivity; TDS = total dissolved solids; ORP = oxidation-reduction potential; ND = no data.

Parameter	East 6				East 8			
	2014	2017	2018	2019	2014	2017	2018	2019
TP (µg/L)	124.4	41.7	30.6	48.8	201.6	37.8	13.1	23.5
SRP (µg/L)	2.5	2.5	6.4	2.5	2.5	2.5	6.6	6.4
Chl *a* (µg/L)	48.06	11.2	10.47	35.4	69.42	11.4	4.49	7.2
Temp (°C)	23.62	21.26	22.5	20.8	24.27	21.23	22	20.9
DO (mg/L)	10.5	10.6	11.3	9.9	11.0	10.3	10.4	9.4
DO (%)	124.4	119.6	130.7	110.3	131.4	115.7	119.4	105.3
pH	9.11	8.58	8.89	8.08	9.17	8.55	8.76	8.06
SpCond (µS/cm)	586	415	394	433	586	413	402	427
TDS (g/L)	0.381	0.270	0.256	0.282	0.381	0.269	0.262	0.278
ORP (mV)	361	470	276	237	359	478	287	233
Turbidity (NTU)	24.1	ND	0.0	2.5	26.9	ND	0.0	1.3

Sediment

Wetland sediment organic matter analyzed during 2019 sampling was compared to previous samples before and after Bear Creek 2016-17 restoration construction. Organic matter in 2017 had decreased from 2012 concentrations, except for site East 6, which is adjacent to the berm supporting Witham Road and was reinforced during 2017 construction (Hassett and Steinman 2018). In 2019, organic matter at sites West 1, West 5, and East 6 all decreased from 2017 values while East 8 slightly increased from 0.3% to 0.9% (Fig. 24). The decline in OM is undoubtedly related to the dredging of the ponds as part of the restoration project.

Similar to Muskegon Lake transect sites, medium sand (250-500 µm) was the majority component of Bear Creek wetland sediment particle size and ranged 49-54% (Fig. 25, Table A9). Sites West 1 and East 6 contained the most silt/clay (<63 µm; 4.8% and 5.2%, respectively) and East 8 contained the most gravel/cobble (>2 mm; 1.7%) (Fig. 25, Table A9).

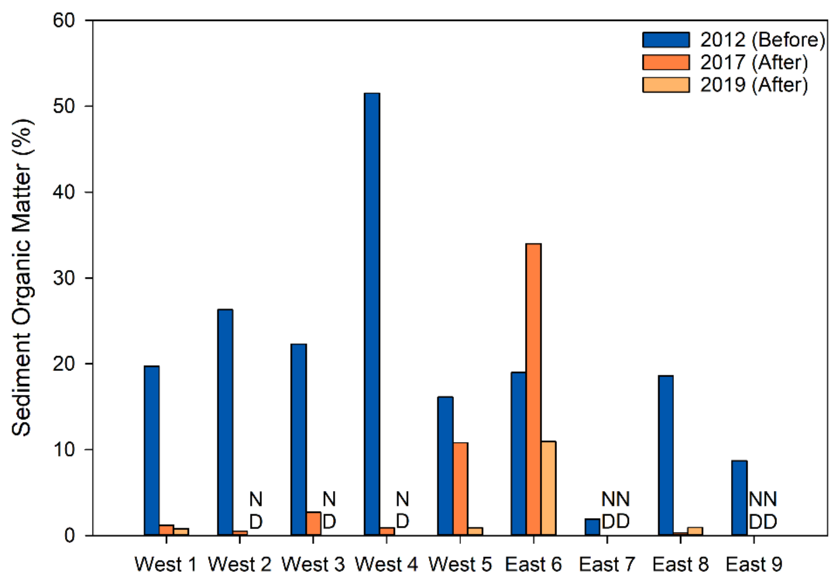

Figure 24. Sediment organic matter (%) at Bear Creek wetland sites before (2012) and after (2017, 2019) restoration. ND = no data; applied to sites not sampled in a given year.

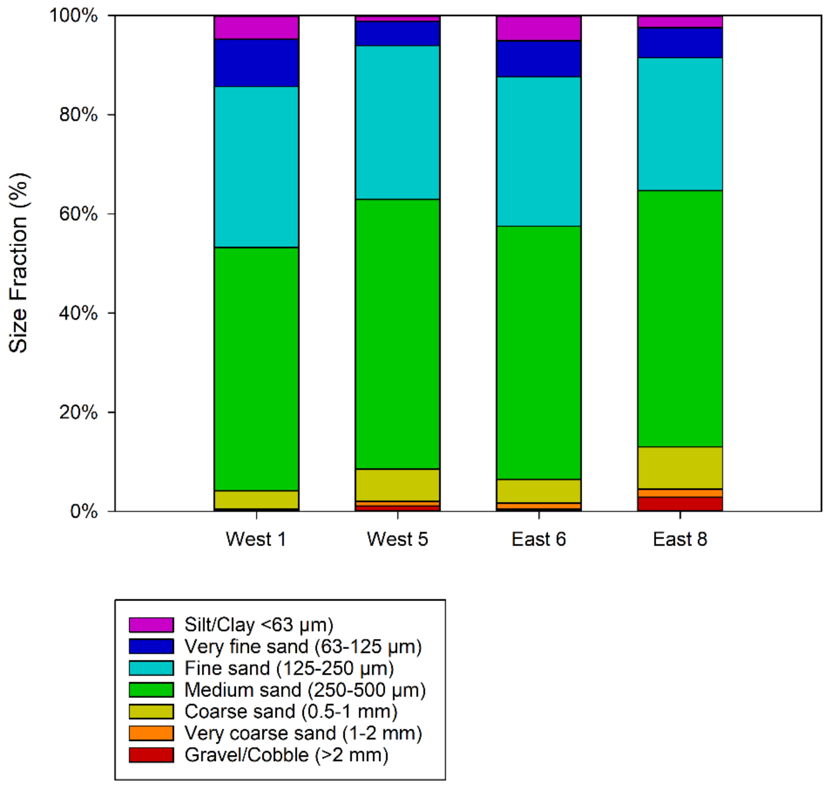

Figure 25. Sediment particle size analysis at Bear Creek wetland sites in 2019. A full list of sediment fraction values is given in Appendix Table A9.

Macroinvertebrates

Full summaries of collected taxa and site count data in the Bear Creek wetland are provided in Appendix Table A10. Species richness was ~3× greater in the former west pond (n=16) than in the east pond (n=5; Table 26). West pond sites were dissimilar in community composition; Site 1 closer to Bear Creek (Fig. 3) was dominated by phantom midges (Chaoboridae: 73%), while Site 5 further away from the creek was more populated with a wider variety of taxa and dominated by fingernail clams (Sphaeriidae: 57%; Table A10). East pond community composition was more similar between sites and both sites were dominated by phantom midges (46-98%; Table A10). East pond Site 6 was co-dominated by Oligochaete worms (45%; Table A10). Cumulatively for pond sites, the west pond had higher diversity (H'), higher maximum diversity (H'_{max}), and higher evenness than the east pond (Table 26).

Table 28. Bear Creek 2019 pond macroinvertebrate community composition based on D-net sampling techniques. Shannon's Diversity Index was used to calculate diversity (*H'*), maximum diversity (H'_{max}), and evenness within each sampling site as well as totals calculated for each whole pond (gray highlighted text). Full abundance count summaries of taxa per site are presented in Appendix Table A10.

		D-net				
Location	Site	Total Species	Total Abundance	H'	H'_{max}	Evenness
West Pond	W1	6	73	0.909	1.792	0.507
	W5	15	316	1.444	2.708	0.533
	total	16	389	1.731	2.773	0.624
East Pond	E6	4	80	0.359	1.386	0.259
	E8	2	50	0.078	0.693	0.113
	total	5	130	0.393	1.609	0.244

Fish

During fish sampling, there were differences in environmental conditions among sites (Table 29). The main differences were in the East Pond, where at Site D, SAV was nearly absent and water temperature was several degrees lower than the other sites, and at Site C, which had a lower DO concentration than the other sites (Table 29). At site D, the only area shallow enough to set fyke nets was a narrow sand flat that was likely the result of deposition from Bear Creek during high flow events. The deposition of the very loose sand was likely a factor preventing the establishment of SAV at site D compared with the other sites. The low DO concentration at site C approached zero for nearly 12 hours (Fig. 26). The low dissolved oxygen concentration at site C was likely related to the large amount of SAV, which should result in high respiration at night; the large bed of SAV also may have decreased water exchange with the open-water area of the east pond.

A total of 331 fish comprising 14 species were captured in 11 fyke nets in the two ponds (Table 30). Largemouth bass (40%), yellow perch (20%), bluegill (18%), and pumpkinseed (12%) accounted for 90% of the catch (Table 30). Round goby was the only non-native fish species to the Great Lakes (Bailey et al. 2004) that we sampled at the Bear Creek wetlands, which accounted for <1% of the catch (Table 30). The majority of the catch from these four species was small (<10 cm TL; Table 30). The mean catch was 30 fish/net, with the highest catch at site C (50 fish/net) and the lowest catch at site D (9 fish/net; Table 31). The low catch at site D may have been associated with the near absence of SAV, although water temperature was lowest at site D. Despite the low dissolved oxygen concentration at site C (Table 29; Fig. 26), CPUE was highest at this site relative to the other sites (Table 30).

Table 29. Mean ±1 SE (n = 3) of environmental conditions at each fish sampling site in the Bear Creek wetland. Environmental conditions were submerged aquatic vegetation (SAV), water temperature (Temp), dissolved oxygen (DO), specific conductivity (SPC), turbidity (Turb), pH, oxidation reduction potential (ORP), and chlorophyll *a* (Chl-*a*). Measurements were made during fyke netting on 22 August 2019 with a YSI sonde (except SAV was estimated visually). See Fig. 3 (p. 10) for site locations.

Site	SAV (%)	Temp (°C)	DO (mg/L)	DO (%)	SPC (µS/cm)	TDS (g/L)	Turb (NTU)	pH	ORP (mV)	Chl-*a* (µg/L)
A	65 ± 10	22.94 ± 0.03	6.76 ± 0.14	78.8 ± 1.6	415 ± 4	0.27 ± 0.00	0.7 ± 0.2	7.73 ± 0.01	311.5 ± 1.7	10.3 ± 0.6
B	63 ± 7	22.71 ± 0.04	6.56 ± 1.28	76.2 ± 14.9	402 ± 1	0.26 ± 0.00	1.2 ± 0.5	7.72 ± 0.13	354.3 ± 1.2	10.1 ± 1.1
C	95 ± 0	21.35 ± 0.23	4.17 ± 0.88	47.1 ± 9.7	425 ± 2	0.28 ± 0.00	2.1 ± 0.4	7.52 ± 0.03	352.1 ± 2.6	25.3 ± 6.4
D	5 ± 0	18.60 ± 0.65	7.91 ± 0.07	85.0 ± 0.8	408 ± 1	0.27 ± 0.00	3.1 ± 0.4	7.50 ± 0.02	353.9 ± 1.2	6.9 ± 1.0

Table 30. Combined catch and total length (TL) of fish captured at four sampling sites in the Bear Creek wetland (n = 11 fyke nets).

Common name	Scientific name	**Catch**	TL (cm) (mean, min, max)		
rock bass	*Ambloplites rupestris*	**3**	10.1	8.8	12.4
black bullhead	*Ameiurus melas*	**1**	23.9	23.9	23.9
yellow bullhead	*Ameiurus natalis*	**3**	28.9	25.5	32.2
brown bullhead	*Ameiurus nebulosus*	**1**	24.8	24.8	24.8
bowfin	*Amia calva*	**7**	52.0	14.1	71.0
gizzard shad	*Dorosoma cepedianum*	**1**	7.5	7.5	7.5
northern pike	*Esox lucius*	**1**	21.6	21.6	21.6
johnny darter	*Etheostoma nigrum*	**1**	3.7	3.7	3.7
pumpkinseed	*Lepomis gibbosus*	**41**	9.7	3.9	16.5
bluegill	*Lepomis macrochirus*	**60**	10.3	3.0	17.0
largemouth bass	*Micropterus salmoides*	**134**	7.5	4.9	29.8
round goby	*Neogobius melanostomus*	**1**	6.8	6.8	6.8
golden shiner	*Notemigonus crysoleucas*	**10**	11.4	10.4	12.4
yellow perch	*Perca falvescens*	**67**	10.2	6.4	17.5
	Total	**331**			

Table 31. Number and TL of fish captured by fyke netting at four sites in the Bear Creek wetland. CPUE is catch per unit effort. Site locations are depicted in Fig. 3.

		Site A				Site B				Site C				Site D			
			TL (cm)				TL (cm)				TL (cm)				TL (cm)		
Common name	*Scientific name*	**Catch**	(mean, min, max)			**Catch**	(mean, min, max)			**Catch**	(mean, min, max)			**Catch**	(mean, min, max)		
rock bass	*Ambloplites rupestris*	**0**	--	--	--	**1**	9.2	9.2	9.2	**0**	--	--	--	**2**	10.6	8.8	12.4
black bullhead	*Ameiurus melas*	**0**	--	--	--	**0**	--	--	--	**1**	23.9	23.9	23.9	**0**	--	--	--
yellow bullhead	*Ameiurus natalis*	**2**	28.9	25.5	32.2	**1**	29	29	29	**0**	--	--	--	**0**	--	--	--
brown bullhead	*Ameiurus nebulosus*	**0**	--	--	--	**1**	24.8	24.8	24.8	**0**	--	--	--	**0**	--	--	--
bowfin	*Amia calva*	**1**	59.0	59.0	59.0	**3**	55	51.2	57.2	**2**	63.1	55.1	71	**1**	14.1	14.1	14.1
gizzard shad	*Dorosoma cepedianum*	**0**	--	--	--	**1**	7.5	7.5	7.5	**0**	--	--	--	**0**	--	--	--
northern pike	*Esox lucius*	**0**	--	--	--	**1**	21.6	21.6	21.6	**0**	--	--	--	**0**	--	--	--
johnny darter	*Etheostoma nigrum*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--	**1**	3.7	3.7	3.7
pumpkinseed	*Lepomis gibbosus*	**10**	8.4	3.9	13.5	**13**	9.73	8.1	16.5	**16**	10.0	6.3	15	**2**	13.6	11.8	15.3
bluegill	*Lepomis macrochirus*	**13**	8.1	6.0	10.2	**16**	9.53	3	17	**28**	11.8	7.7	16.3	**3**	10.8	7.0	15
largemouth bass	*Micropterus salmoides*	**14**	9.3	5.2	29.8	**39**	7.46	5.7	13.3	**65**	7.4	4.9	28	**16**	6.48	5.3	8.5
round goby	*Neogobius melanostomus*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--	**1**	6.8	6.8	6.8
golden shiner	*Notemigonus crysoleucas*	**0**	--	--	--	**10**	11.4	10.4	12.4	**0**	--	--	--	**0**	--	--	--
yellow perch	*Perca falvescens*	**6**	10.3	8.4	15.5	**22**	10.8	6.7	17.5	**37**	9.7	6.4	14.5	**2**	12.8	12.2	13.4
	Total	**46**				**108**				**149**				**28**			
	Fyke Nets (No.)	**2**				**3**				**3**				**3**			
	CPUE (fish/net)	**23**				**36**				**50**				**9**			

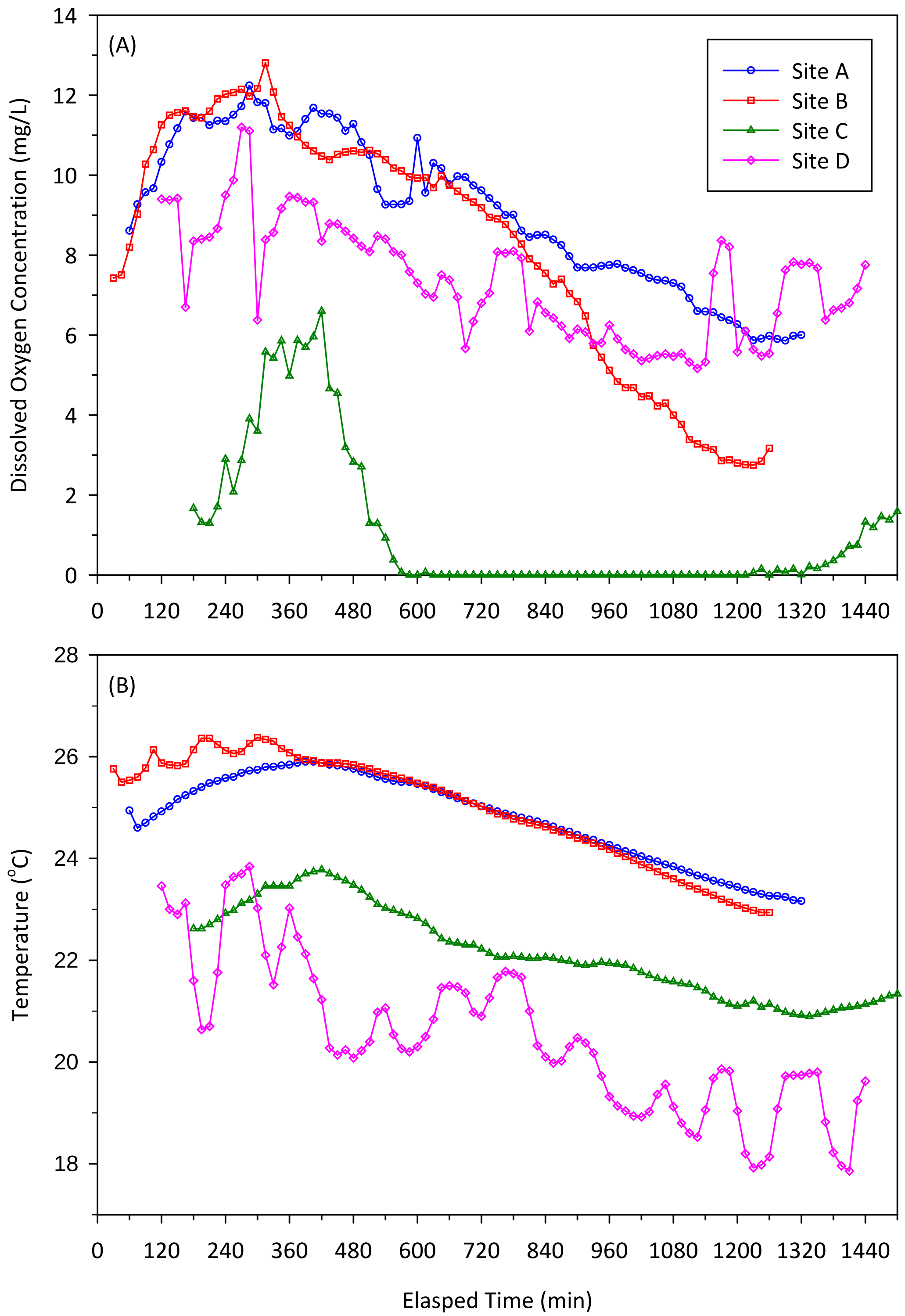

Figure 26. (A) Dissolved oxygen concentration and (B) water temperature measured at 15-min intervals over a diel cycle during fyke netting. The elapsed time of 0 is 12:00 PM on 21 August 2019 and the elapsed time of 1500 min is 2:00 PM on 22 August 2019. Site locations are depicted in Fig. 3.

Discussion

The goal of continued transect monitoring was to determine the effectiveness of restoration in improving habitat quality, which involved separating biological responses associated with restoration from responses due to annual environmental variation. The benefits of long-term monitoring are well-documented in the scientific literature, and include, among others: 1) characterizing how and why nature is changing; 2) providing a way to understand how ecosystems are regulated and function; 3) linking ecological patterns to natural variability; 4) informing management on how ecosystems are responding to human influence; and 5) if restoration has been conducted, are systems responding as expected or is change needed (Likens 1989, Lindenmayer et al. 2012, Hughes et al. 2017).

In this report, we have the ability to evaluate the short-term and longer-term (9-10 years) responses of the macrophyte and fish populations in restored and reference areas of Muskegon Lake. In addition, we are able to provide a limited assessment of biological responses in two other restored areas: Veterans Park and Bear Creek wetlands. Our assessments are conflated by record high water levels in the Great Lakes region, which made the inclusion of reference areas all the more valuable in teasing apart the effects of restoration vs. environmental variability.

Muskegon Lake

Macrophyte Response:

As noted in Ogdahl and Steinman (2015), macrophyte community spatial and temporal trends were associated with both physical habitat (i.e., WI, transect length, and slope) and hydrologic characteristics (i.e., water level and precipitation), making it difficult to assess the effect of restoration alone. By 2019, increases in total macrophyte richness at all restored sites except Grand Trunk, in association with decreases at the two reference sites, suggest a positive impact on macrophytes from restoration activity. Other indicators showed less clear patterns: C-values increased at all restored sites except Kirksey, but they also either remained stable or increased at the reference sites, making it impossible to differentiate the effect of restoration from environmental factors for improved habitat quality. Similar results occurred for density and total biomass, where conflicting patterns were observed at the reference sites (decrease at one site but increase at the other), as well as at restored sites (two sites increasing and two sites decreasing between 2012 and 2019).

After 10 years, we have seen macrophyte community structure improve relative to immediate post-restoration; however, as confirmed by the PCA ordination, the overall trajectory at the restored sites shows variable responses with no clear trend toward improvement relative to the reference sites. It is possible that macrophyte communities at restored sites may never exceed the same attribute values as at the reference sites due to inherent site differences or fundamental changes in sediment influenced by physical disturbance during restoration, but continued monitoring would be necessary to make that determination.

The time elapsed, and nature of, restoration affected the macrophyte response. This is particularly evident in the density and biomass responses at Heritage Landing and Grand Trunk. The April 2011 restoration at Heritage Landing included the physical removal of underwater fill material along the sampling transect; this may account for low macrophyte density and biomass at Heritage Landing in 2011 and the subsequent habitat quality decrease in 2012. Less physically disruptive restoration at

Grand Trunk in June 2010, adjacent to the sampling transect, may account for the less drastic macrophyte density and biomass declines in 2011 and subsequent increases in density, biomass, and habitat quality in 2012 and 2019.

Physical habitat similarities between Grand Trunk and Northwest Reference compared to Heritage Landing may also account for the Grand Trunk macrophyte community biological variables approaching reference-quality levels. Gentle slopes and longer transect lengths at Grand Trunk and Northwest Reference likely increased habitat availability and heterogeneity (i.e., depth and light regimes) for different macrophyte morphologies, promoting increases in macrophyte richness, density, and biomass at those two transects (Duarte and Kalff 1990). Physical habitat similarities also were observed between NE Reference and Kirksey (see overlap in Fig. 16B), which may account for similar biomass and density changes; Kirksey was a shorter transect, and therefore had lower total biomass as well as having significant reductions in biomass and density in 2011 just after restoration.

Although environmental variables naturally shift, extreme precipitation events, temperature changes, and water level fluctuations are predicted to increase in frequency due to climate change (Notaro et al. 2015). Water level increases can reduce light availability for macrophytes (Chow-Fraser et al. 1998) and precipitation can increase dissolved organic carbon loading, reducing light transmittance (Chen et al. 2016). Muskegon Lake macrophytes responded to increased water level or precipitation (2009, 2011, and 2019) with a decrease in macrophyte richness, especially evident during the >1 m water level rise from 2012 to 2019, when the emergent macrophytes *P. australis* and *Typha* spp. were absent at Grand Trunk and Northwest Reference compared to previous survey years. Emergent macrophyte physiological requirements are more easily disrupted by rising water levels than other morphologies (Zohary and Ostrovsky 2011), inhibiting emergent plant growth and seed germination (Coops and Van Der Velde 1995) and decreasing overall habitat richness.

A decrease in Heritage Landing's macrophyte biomass, density, and cover rank in 2019, while these variables increased or remained high at Grand Trunk and Northwest Reference, may again be a product of physical habitat characteristics. Heritage Landing's steep slope, potentially reducing light availability, may have decreased habitat optima for submerged macrophytes, prompting a negative response to rising water levels in 2019. Positive or neutral macrophyte responses at Grand Trunk and Northwest Reference in 2019 relative to prior years may have been influenced by other unmeasured environmental variables (e.g., nutrient concentrations or turbidity) and in part by restoration at Grand Trunk.

Macroinvertebrate Response:

Prior macroinvertebrate studies on Muskegon Lake have focused primarily on sediment habitat, as these organisms were used as indicators of potential sediment toxicity or marine debris habitat (Harris 2017) from past industrial activity (e.g., Carter et al. 2006; Nelson and Steinman 2010). As a consequence, the data from those studies are not comparable to our analyses. Cooper et al. (2007b) sampled the Muskegon River wetland, which is directly upstream of the lake, and found taxa richness in *Peltandra* and water lily habitats ranging from 23 to 38, about double of what we found in our transects. In addition, their dominant taxa included *Gammarus*, *Caecidotea*, *Hyalella azteca*, and Chironomini, in contrast to dominance by gastropods and bivalves that we observed in Muskegon Lake. This difference is likely related to the different macrophyte habitats in which the two groups sampled, as well as the more sheltered environment in the wetland compared to the littoral zones of Muskegon Lake.

Cooper et al. (2007b) also identified organic sediment depth as an important correlate of macroinvertebrate community structure, an observation consistent with our data. Indeed, organic sediment abundance appears to be a critical driver of macrophyte colonization and growth (Ogdahl and Steinman 2015), which in turn influences macroinvertebrate community composition and abundance.

Fish Response:

The fish-based IBI is commonly used to assess habitat in coastal regions of the Great Lakes (Uzarski et al. 2005; Cooper et al. 2018), as it provides an integrated assessment of habitat condition for fish. In Muskegon Lake, the fish composition at the sampling sites was generally consistent with prior sampling efforts (unpubl. data). There was no clear evidence from the fish-based IBI that the fish assemblage positively responded to restoration activities at the five restoration sites. Indeed, fish catch numbers were lowest at the reference sites, but this may be related to the extremely high water levels in 2019 in Muskegon Lake, which negatively impacted macrophyte biomass and density especially at the NW Ref transect.

Veterans Park

Habitat restoration in the south pond likely improved water quality, which corresponded with improvements in the fish assemblage (Ruetz and Ellens 2018). The changes in the south pond were mostly likely caused by the reconnection with the Muskegon River. However, high Great Lakes water levels necessitated the reinstallation of a water control structure in May 2019 at the south pond to control flooding in the park, which may offset ecological improvements in that pond. For instance, specific conductivity and Cl^- concentrations—both indicators of water quality (Uzarski et al. 2005)—were much higher in the south pond in 2019 compared with previous years, suggesting that severing the exchange of water between the Muskegon River and the south pond (with the installation of the water control structure) degraded water quality. Thus, when Great Lakes water levels recede and flooding is no longer a concern for the park, the water control structure at the south pond should be opened to allow for water exchange and fish movements between the Muskegon River and the south pond.

Bear Creek Wetlands

The restored wetland water quality remains high compared to the pre-restoration state (cf. Steinman and Ogdahl 2016, Oldenborg and Steinman 2019), although there was some slight backsliding in TP concentrations in 2019 compared to 2018. Additional sampling in 2020 will determine if this is a trend to be concerned about. Certainly, high water levels have prevented the flow-through marshes from being as efficient at trapping nutrients as they can be.

We have no prior data on the macroinvertebrate or fish communities in these wetlands, so the information generated as part of this study provides a baseline for assessing future recovery trends. As expected, dredging significantly reduced the sediment organic matter, which may limit the recovery of macrophyte (and associated macroinvertebrate) communities in the short term. Indeed, the overall species richness and diversity of the macroinvertebrate communities in these restored ponds was lower than in the restored sites in Muskegon Lake (compare Tables 20 vs. 28), and dominance by fingernail clams and chaoborids is consistent with the overall absence of macrophytes as substrate, with sediment serving that role.

The fish assemblage in the Bear Creek wetland was dominated by sport-fish species that are often prized by anglers (Becker 1983). Moreover, invasive fish species were rarely encountered in sampling, which is another positive indicator. The fish species captured in the Bear Creek wetland are common in nearby Muskegon Lake (Bhagat and Ruetz 2011). Although observations on the fish assemblage prior to restoration of the Bear Creek wetland are not available, the post-restoration monitoring suggested a "healthy" fish assemblage compared with other drowned river mouth lakes (Janetski and Ruetz 2015).

It is anticipated that once water levels decline, and macrophytes have an opportunity to take hold, there will be further reductions in nutrient concentrations due to macrophyte uptake and sediment sequestration, as well as increases in invertebrate and fish diversity and abundance. The timeline for this recovery will be dependent on climate conditions.

Synthesis

Restoration of lake littoral zones results in numerous benefits, including sediment stabilization, enhancement of habitat for macroinvertebrate and fish, and improved water quality, thereby restoring littoral species interactions and food web structure (Brauns et al. 2011). Naturalized shorelines also optimize lake esthetic appeal, influencing increases in lakeside property values (Leggett and Bockstael 2000, Isely et al. 2018) and stimulating recreational usage through tourism and sport fishing (Campbell et al. 2015). Enhanced physical allure of restoration alone caused a projected $11.9 million increase in housing values in neighborhoods adjacent to Muskegon Lake's southern shore (Isely et al. 2018). The improvement of shoreline ecological integrity has also begun to stimulate economic growth for the local community and is conservatively estimated to generate a 5.8:1 return on investment (Isely et al. 2018).

Continued long-term monitoring has indicated that restored transects have improved in habitat quality; however, improvement was neither strong nor consistent. We originally assumed a predictable, rapid post-restoration trajectory that favored habitat quality increase and positive macrophyte community changes (Figure 7A, Hobbs and Norton 1996), which in turn would enhance macroinvertebrate and fish communities. However, hydrologic and meteorological fluctuations among survey years and disturbance associated with the original restoration efforts likely altered the anticipated restoration sequence (Figure 7B, Bullock et al. 2011). Given that climate change is expected to exacerbate the magnitude of water level fluctuation in lakes (Wantzen et al. 2008), it is critical that shoreline restoration efforts account for more frequent and more intense high and low water levels, allowing lake levels to migrate both upland and lakeward with minimal hardening of the shoreline; this type of resiliency (cf. Lake 2013) will help accommodate the recovery of macrophyte zones, improving habitat for other trophic levels.

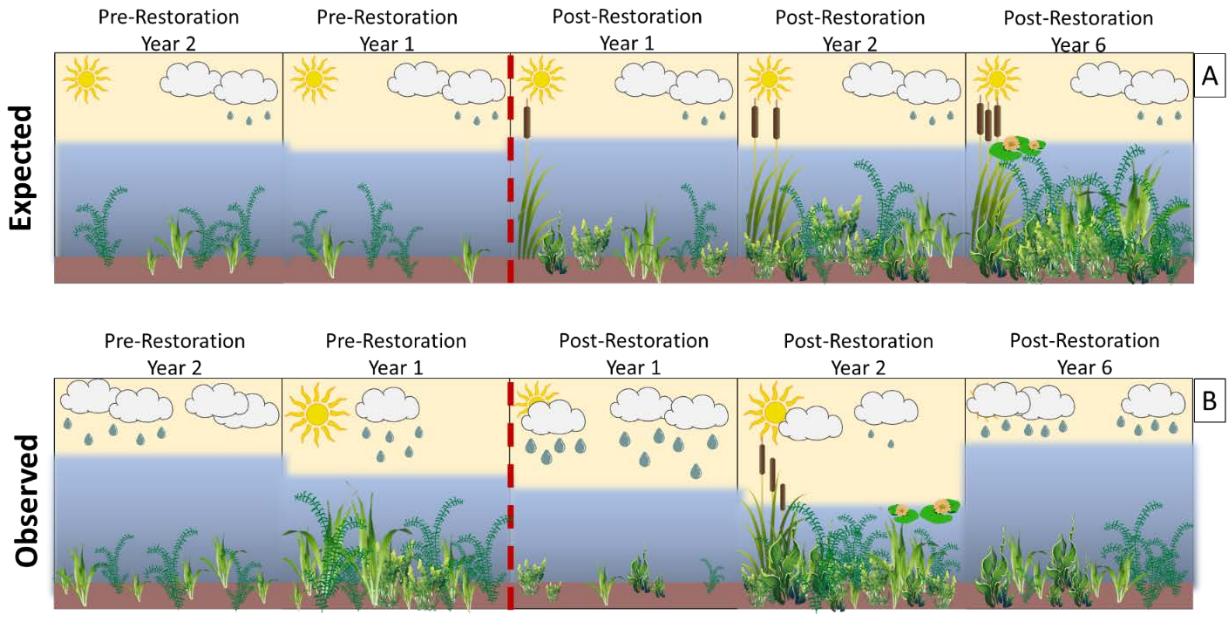

Fig. 27. Muskegon Lake Expected vs. Observed Restoration Trajectory. A conceptual diagram of Muskegon Lake's macrophyte community trajectory; red dashed line represents pre-restoration (left of line) and post-restoration (right of line) conditions. The sun represents warmer air temperatures and clouds represent cooler air temperatures. Raindrop number and size indicates precipitation accumulation during the growing season. Water level is shown in blue. Macrophyte density is represented by macrophyte number, biomass is represented by macrophyte size, and richness is represented by the number of different macrophyte types. A) Expected Muskegon Lake restoration trajectory. B) Observed Muskegon Lake restoration trajectory. Although this figure focuses on macrophytes, it is anticipated that macroinvertebrate and fish populations will follow the vegetative recovery trajectory. Schematic from Kleindl and Steinman (submitted).

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APPENDICES

Appendix A: Macrophyte, Particle Size Fraction, and Macroinvertebrate Data

Appendix B: Fish-Based IBI for Muskegon Lake

Appendix A: Macrophyte, Particle Size Fraction, and Macroinvertebrate Data

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NW Reference

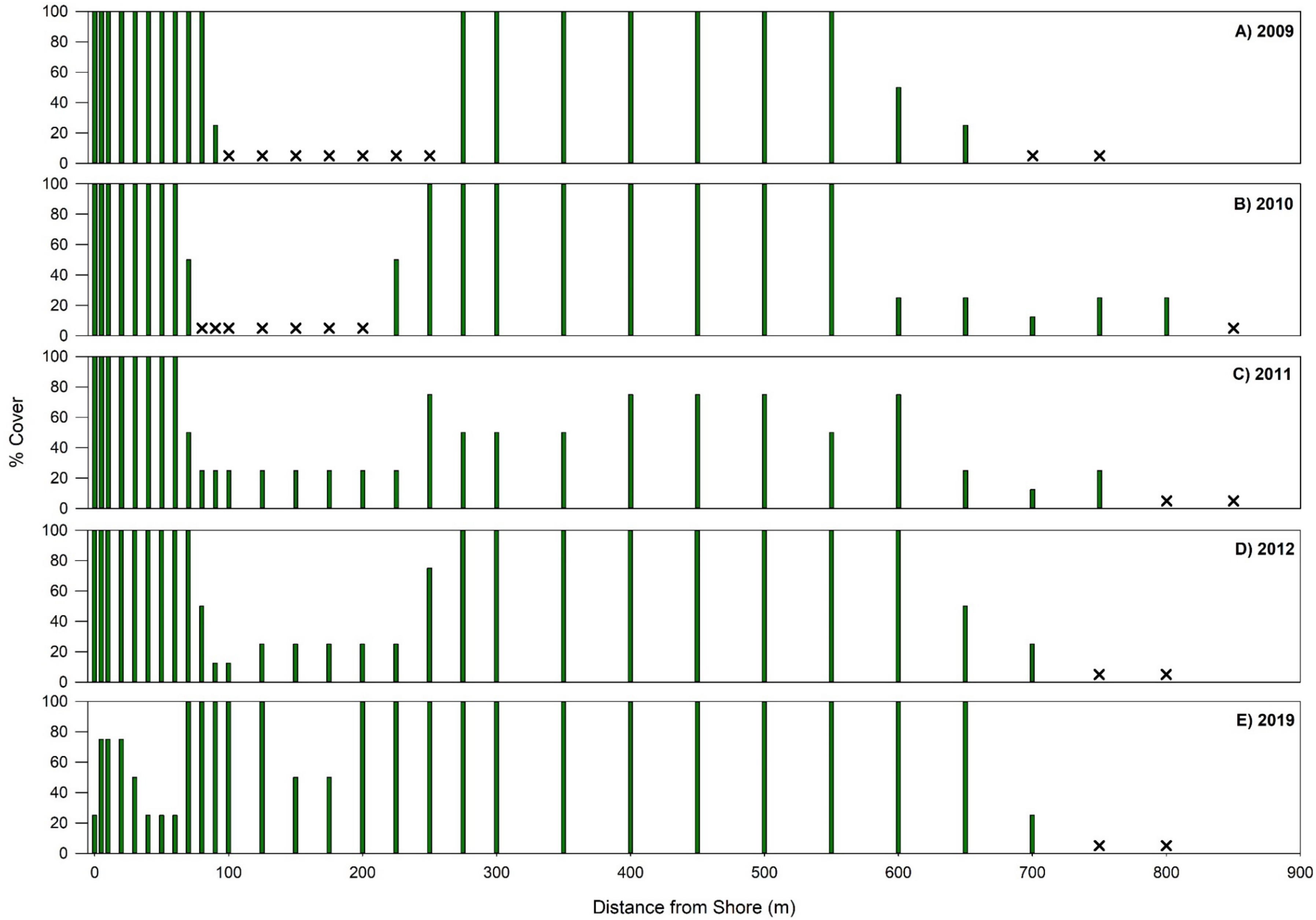

Figure A1. Macrophyte % cover (based on cover ranks) at the NW Reference site from 2009-2019. X indicates 0% cover for a given point along the transect.

NE Reference

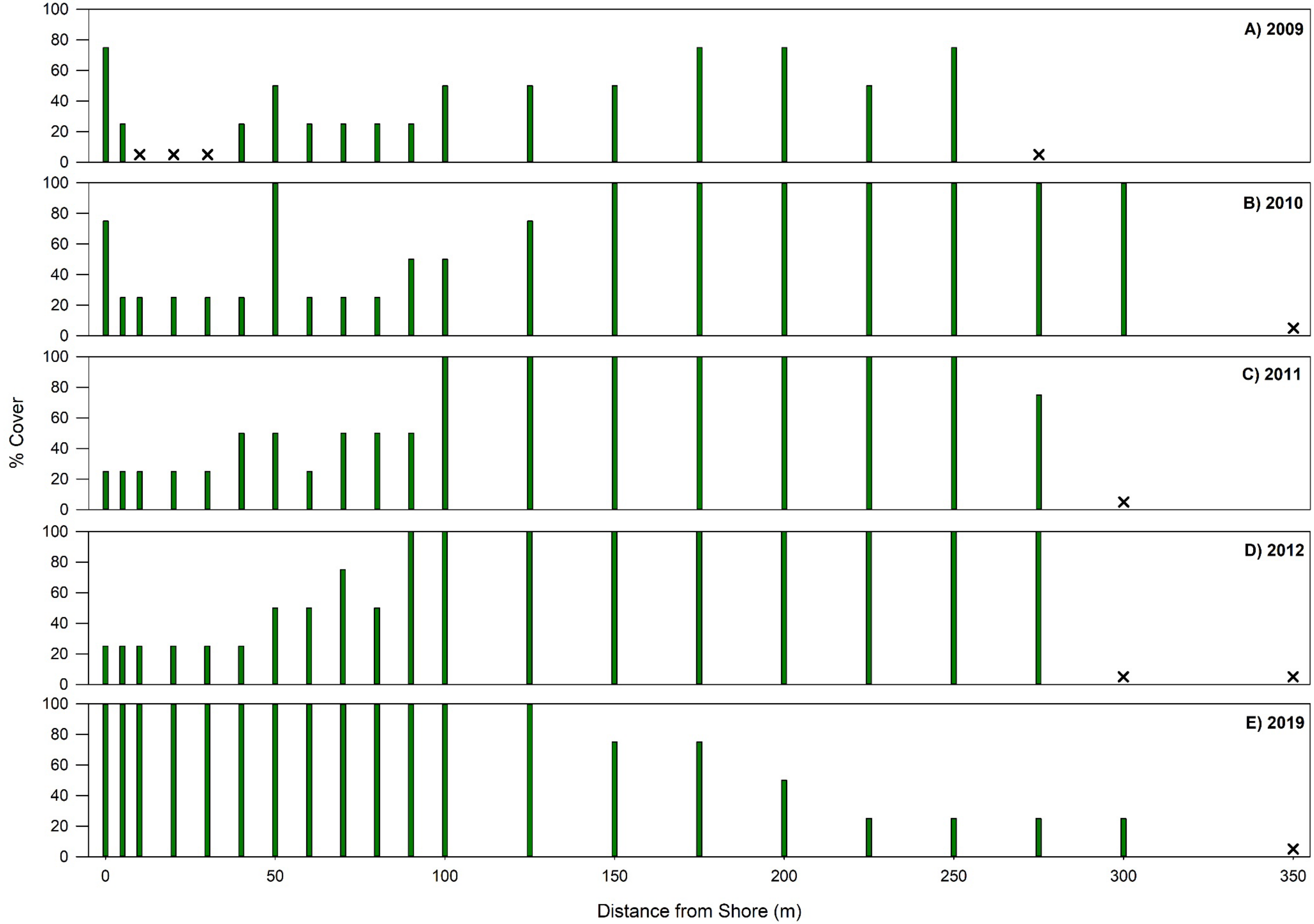

Figure A2. Macrophyte % cover (based on cover ranks) at the NE Reference site from 2009-2019. X indicates 0% cover for a given point along the transect.

Amoco

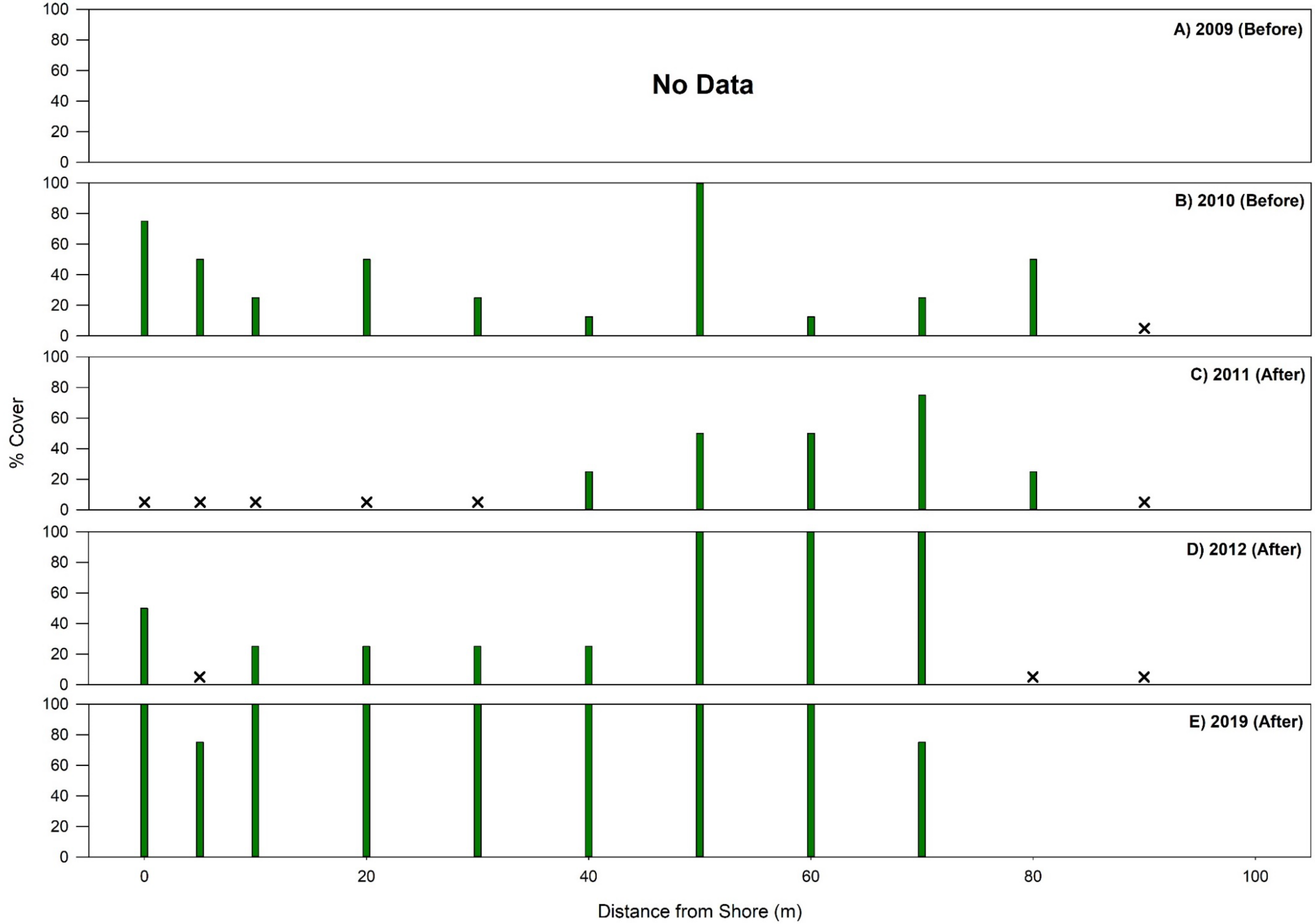

Figure A3. Macrophyte % cover (based on cover ranks) at the Amoco site from 2009-2019. X indicates 0% cover for a given point along the transect.

Circle Bay

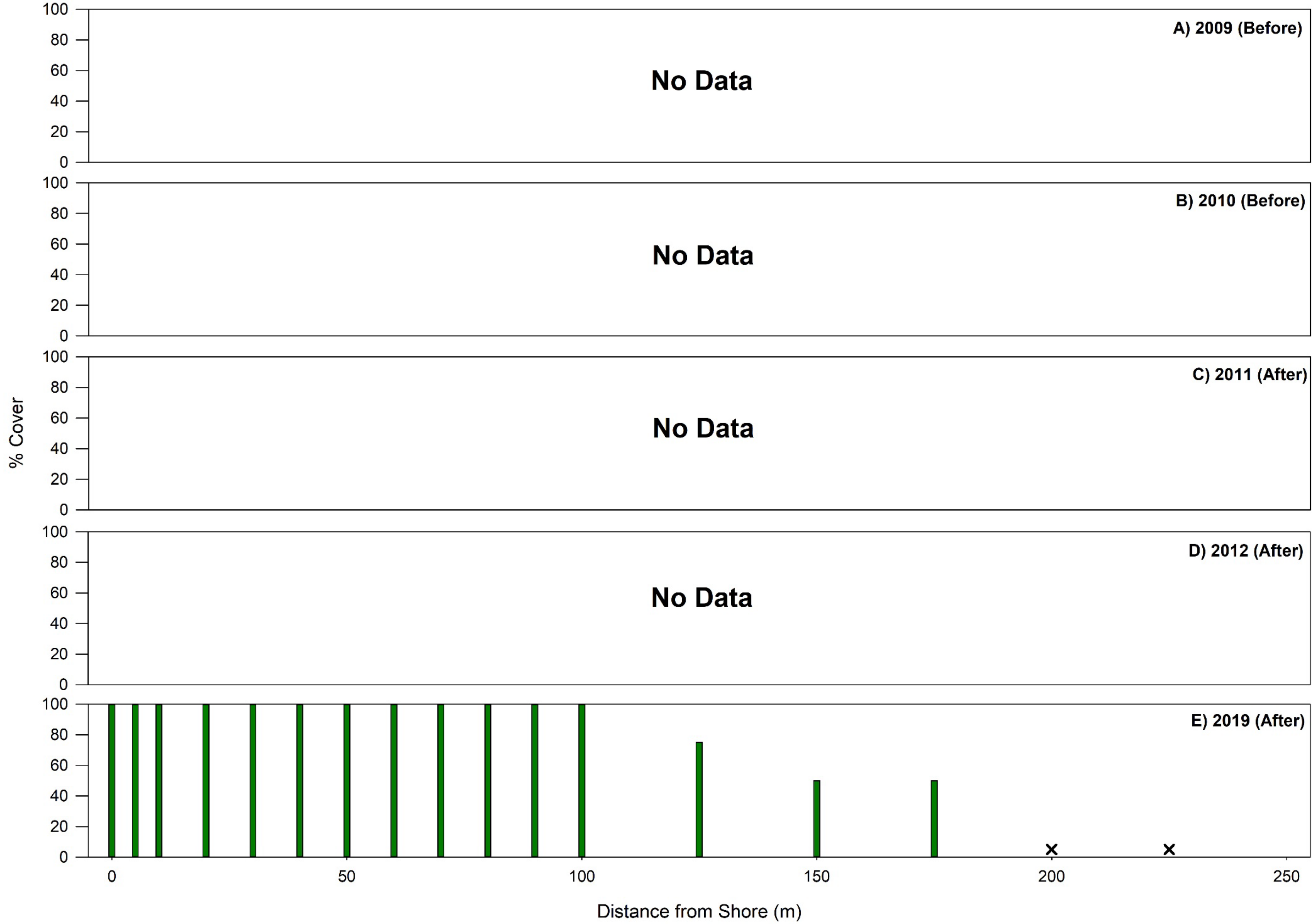

Figure A4. Macrophyte % cover (based on cover ranks) at the Circle Bay site from 2009-2019. X indicates 0% cover for a given point along the transect.

Grand Trunk

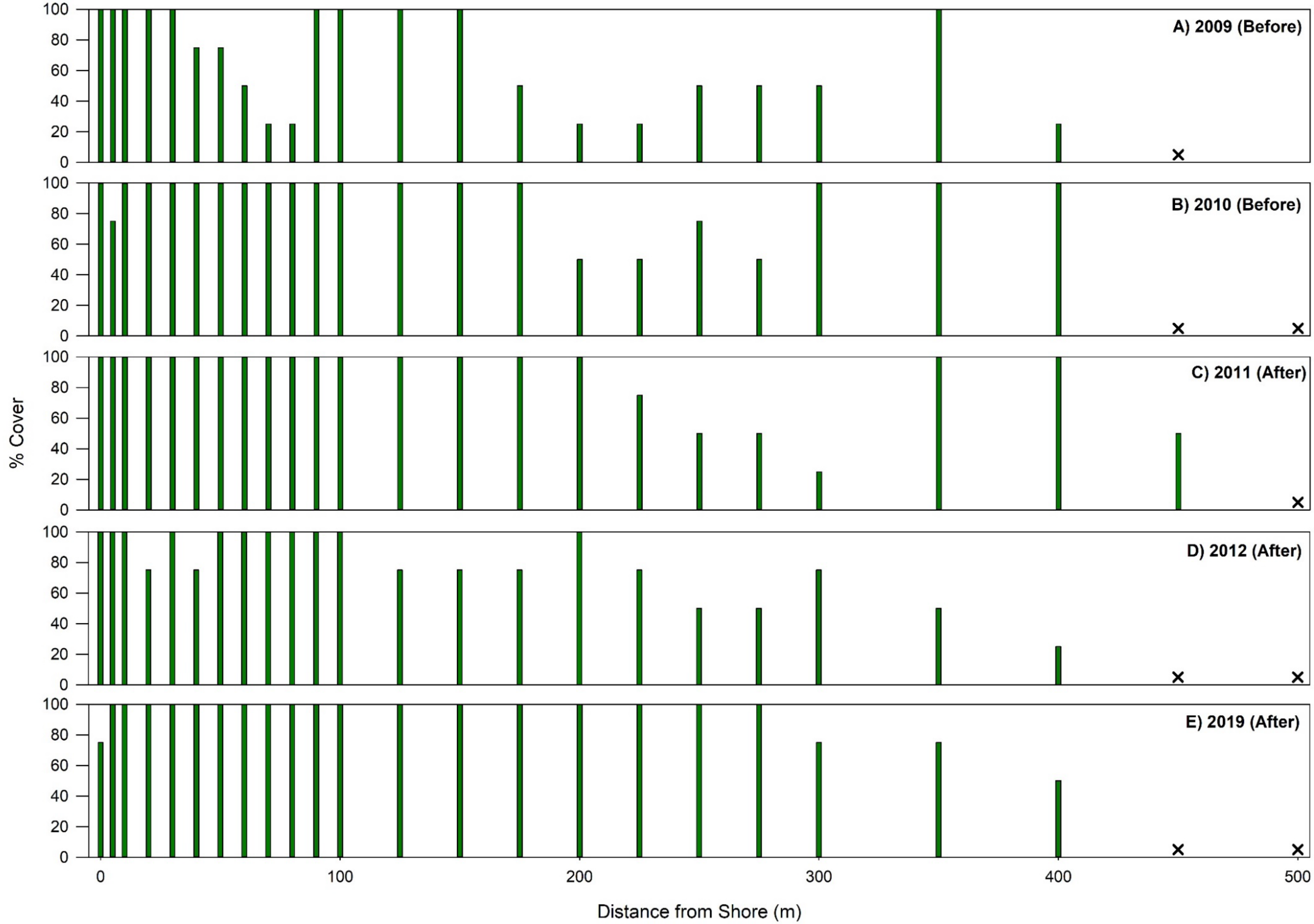

Figure A5. Macrophyte % cover (based on cover ranks) at the Grand Trunk site from 2009-2019. X indicates 0% cover for a given point along the transect.

Heritage Landing

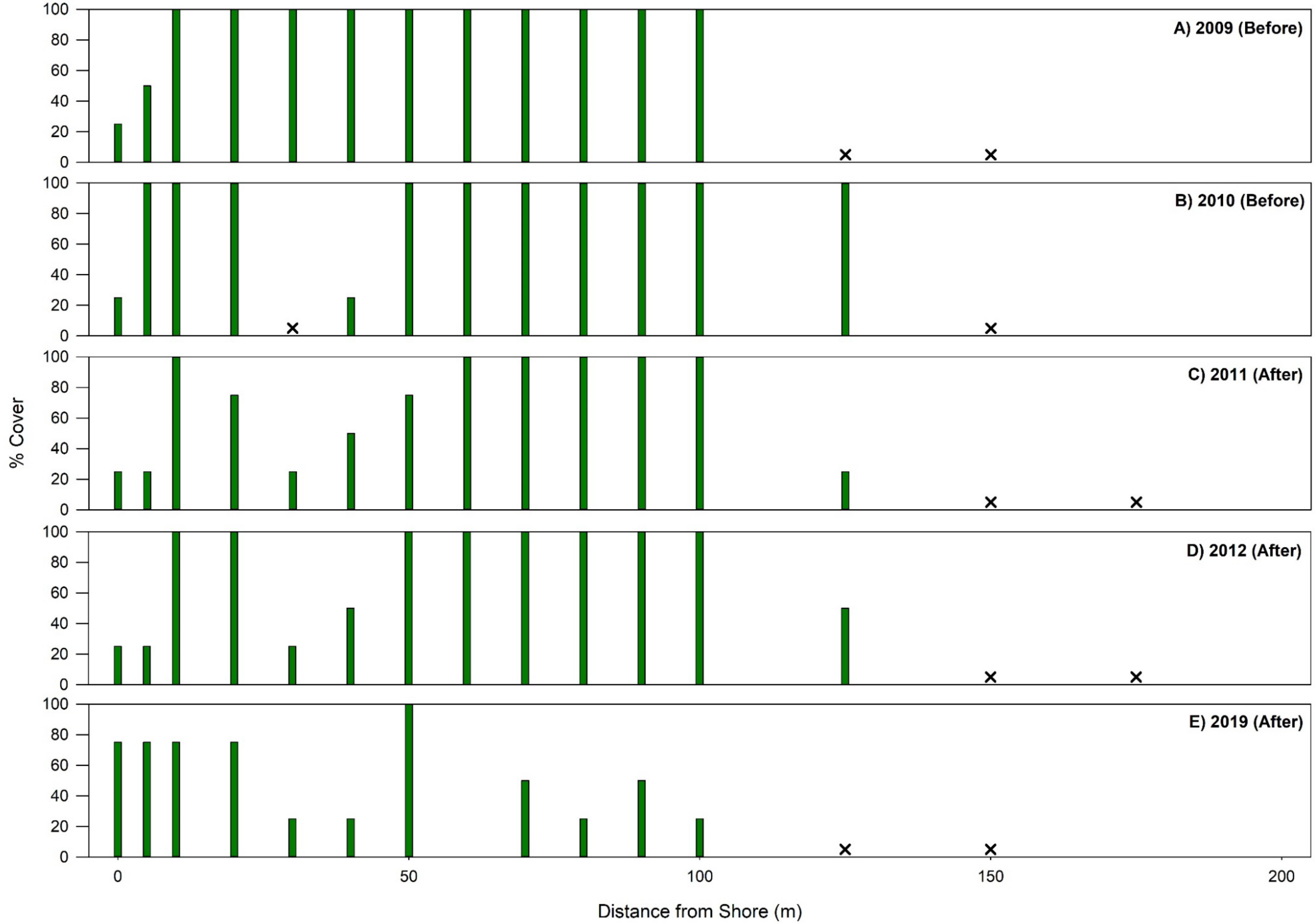

Figure A6. Macrophyte % cover (based on cover ranks) at the Heritage Landing site from 2009-2019. X indicates 0% cover for a given point along the transect.

Kirksey

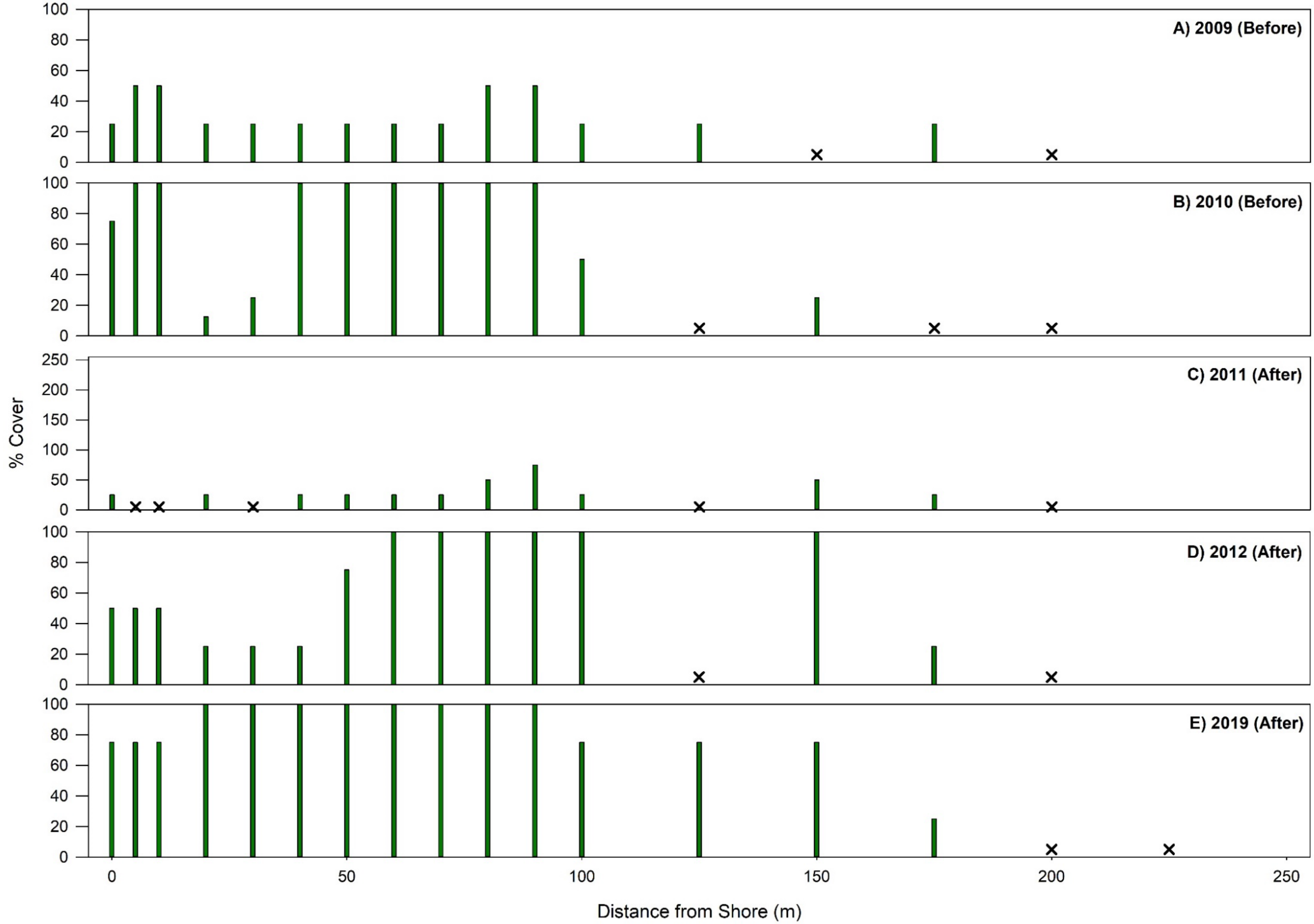

Figure A7. Macrophyte % cover (based on cover ranks) at the Kirksey site from 2009-2019. X indicates 0% cover for a given point along the transect.

Table A1. Particle size fractions from 2019 Muskegon Lake transect sediment, organized by distance from shore.

Transect	Distance (m)	% Size Fraction						
		Gravel/Cobble (>2 mm)	Very coarse sand (1-2 mm)	Coarse sand (0.5-1 mm)	Medium sand (250-500 µm)	Fine sand (125-250 µm)	Very fine sand (63-125 µm)	Silt/Clay (<63 µm)
NW Ref	80	0.1	0.8	2.5	29.4	53.5	12.6	1.1
	150	0.5	0.6	5.1	60.5	31.8	1.4	0.1
	250	0.0	0.2	2.6	63.1	33.3	0.7	0.1
	400	0.8	0.8	4.8	76.9	16.0	0.4	0.2
NE Ref	30	15.0	0.6	0.9	12.7	35.0	31.9	3.8
	70	10.9	0.5	1.7	43.4	39.7	2.9	0.9
	150	6.5	4.2	17.2	50.1	17.6	3.4	1.0
	250	3.5	2.9	12.2	58.4	20.6	1.8	0.5
Amoco	10	12.7	0.9	3.8	48.3	31.3	2.7	0.3
	30	0.0	0.1	1.2	55.9	41.2	1.6	0.0
	70	0.3	0.5	0.4	71.6	26.8	0.3	0.1
Heritage Landing	10	3.6	3.3	3.3	20.0	31.8	22.6	15.4
	30	0.7	1.7	2.2	3.7	5.2	15.5	71.0
	60	0.6	2.4	4.6	5.4	10.7	25.0	51.3
Circle Bay	10	6.5	4.5	3.7	28.4	40.1	12.5	4.4
	30	0.9	0.9	1.7	40.5	51.1	4.1	0.8
	60	1.5	1.3	2.4	37.7	50.5	5.5	1.1
Kirksey	30	0.2	0.1	2.2	70.6	25.9	0.9	0.1
	60	0.0	0.1	1.8	64.1	33.5	0.4	0.1
	100	0.1	0.1	1.6	63.1	34.5	0.5	0.1
Grand Trunk	30	0.6	2.8	6.9	24.0	26.8	22.4	16.4
	70	0.3	0.3	1.9	57.2	30.3	5.5	4.6
	150	2.1	3.8	5.8	8.7	16.2	21.4	42.1
	300	0.8	0.5	2.2	74.4	21.4	0.6	0.2

Table A2. NW Reference 2019 transect macroinvertebrate abundance counts collected via D-net sampling technique.

Class/Subclass	Order/Suborder/Clade	Family	D-net 80 m	250 m	400 m	Total
Acari				1	1	2
Bivalvia	Veneroida	Dreissenidae	1	19	8	28
Gastropoda	Heterobranchia	Valvatidae		1		1
Gastropoda	Littorinimorpha	Rissooidea	52	183	45	280
Gastropoda	Planorboidea	Physidae			1	1
Gastropoda	Planorboidea	Planorbidae	1	3	3	7
Insecta	Diptera	Chironomidae			1	1
Insecta	Diptera	Pseudochironomini/ Chironomini		4	12	16
Insecta	Diptera	Tanypodinae		2	4	6
Insecta	Ephemeroptera	Caenidae		2		2
Insecta	Odonata	Coenagrionidae		1	1	2
Insecta	Trichoptera	Hydroptilidae	1	1		2
Insecta	Trichoptera	Leptoceridae		10		10
Malacostraca	Amphipoda	Hyalellidae	1	11	37	49

Table A3. NE Reference 2019 transect macroinvertebrate abundance counts collected via D-net sampling technique.

Class/Subclass	Order/Suborder/Clade	Family	D-net 30 m	150 m	250 m	Total
Acari			3			3
Bivalvia	Veneroida	Dreissenidae	9		7	16
Gastropoda	Heterobranchia	Valvatidae	2	9	11	22
Gastropoda	Littorinimorpha	Bithyniidae	16	15	2	33
Gastropoda	Littorinimorpha	Rissooidea	74	87	3	164
Gastropoda	Planorboidea	Physidae		2		2
Gastropoda	Planorboidea	Planorbidae	5	11		16
Insecta	Diptera	Orthocladiinae	1			1
Insecta	Diptera	Pseudochironomini/ Chironomini	1		1	2
Insecta	Diptera	Tanypodinae	1	1		2
Insecta	Ephemeroptera	Caenidae	1			1
Insecta	Odonata	Coenagrionidae	5			5
Insecta	Trichoptera	Leptoceridae	16	15		31
Malacostraca	Amphipoda	Gammaridae	1			1
Malacostraca	Amphipoda	Hyalellidae	33	6	32	71
Oligochaeta			1			1

Table A4. Amoco 2019 transect macroinvertebrate abundance counts collected via D-net sampling technique.

			D-net			
Class/Subclass	Order/Suborder/Clade	Family	10 m	30 m	70 m	Total
Bivalvia	Veneroida	Dreissenidae	72	39	58	169
Gastropoda	Heterobranchia	Valvatidae	2	3		5
Gastropoda	Littorinimorpha	Rissooidea	92	250	208	550
Gastropoda	Planorboidea	Physidae	1	1		2
Gastropoda	Planorboidea	Planorbidae	1			1
Insecta	Diptera	Ceratopogonidae	1			1
Insecta	Diptera	Pseudochironomini/Chironomini	10	2		12
Insecta	Diptera	Tanypodinae		1		1
Insecta	Ephemeroptera	Caenidae	1	1		2
Insecta	Odonata	Coenagrionidae		2		2
Malacostraca	Amphipoda	Gammaridae	3	2		5
Malacostraca	Amphipoda	Hyalellidae	6	11		17
Oligochaeta	Haplotaxida	Naididae		2	4	6
Oligochaeta					1	1

Table A5. Circle Bay 2019 transect macroinvertebrate abundance counts collected via D-net sampling technique.

			D-net			
Class/Subclass	Order/Suborder/Clade	Family	10 m	30 m	60 m	Total
Acari			6	3	4	13
Bivalvia	Veneroida	Dreissenidae	9	8	10	27
Gastropoda	Heterobranchia	Valvatidae	5		2	7
Gastropoda	Littorinimorpha	Rissooidea	56	13	39	108
Gastropoda	Planorboidea	Physidae	1	3	1	5
Gastropoda	Planorboidea	Planorbidae	1	3		4
Insecta	Coleoptera	Elmidae	1			1
Insecta	Diptera	Pseudochironomini/Chironomini	1	8	3	12
Insecta	Diptera	Tanypodinae	1	2	3	6
Insecta	Ephemeroptera	Baetidae	1			1
Insecta	Ephemeroptera	Caenidae	1			1
Insecta	Ephemeroptera	Isonychiidae		1		1
Insecta	Odonata	Coenagrionidae	3	4	4	11
Insecta	Odonata	Corduliidae			1	1
Insecta	Trichoptera	Hydroptilidae	1			1
Insecta	Trichoptera	Leptoceridae	14	18	22	54
Malacostraca	Amphipoda	Hyalellidae	83	39	45	167
Malacostraca	Isopoda	Asellidae			1	1
Oligochaeta	Haplotaxida	Naididae		2		2
Oligochaeta			1		3	4
Turbellaria			15	6		21

Table A6. Grand Trunk 2019 transect macroinvertebrate abundance counts collected via D-net sampling technique.

			D-net			
Class/Subclass	Order/Suborder/Clade	Family	30 m	150 m	300 m	Total
Acari			12	6	2	20
Bivalvia	Veneroida	Dreissenidae		1	36	37
Gastropoda	Heterobranchia	Valvatidae		1	1	2
Gastropoda	Littorinimorpha	Rissooidea		3	106	109
Gastropoda	Planorboidea	Physidae	2			2
Gastropoda	Planorboidea	Planorbidae			1	1
Insecta	Coleoptera	Hydrophilidae (larva)	1			1
Insecta	Diptera	Chironomidae			1	1
Insecta	Diptera	Pseudochironomini/Chironomini		2	7	9
Insecta	Diptera	Tanypodinae	1		6	7
Insecta	Hemiptera	Pleidae	1			1
Insecta	Odonata	Coenagrionidae	6	2	1	9
Insecta	Odonata	Corduliidae	1			1
Insecta	Trichoptera	Leptoceridae	18	8	9	35
Malacostraca	Amphipoda	Gammaridae	2			2
Malacostraca	Amphipoda	Hyalellidae	18	13	14	45
Malacostraca	Isopoda	Asellidae	1	2		3
Oligochaeta	Haplotaxida	Naididae			17	17
Oligochaeta			2		4	6
Turbellaria				2		2

Table A7. Heritage Landing 2019 transect macroinvertebrate abundance counts collected via D-net sampling technique.

			D-net		
Class/Subclass	Order/Suborder/Clade	Family	10 m	30 m	Total
Acari			3	2	5
Bivalvia	Veneroida	Dreissenidae	2	5	7
Bivalvia	Veneroida	Sphaeriidae	1		1
Gastropoda	Heterobranchia	Valvatidae	13	4	17
Gastropoda	Littorinimorpha	Rissooidea	24	11	35
Gastropoda	Planorboidea	Physidae	2		2
Gastropoda	Planorboidea	Planorbidae	5		5
Insecta	Diptera	Pseudochironomini/Chironomini	3		3
Insecta	Diptera	Tanypodinae	4	2	6
Insecta	Diptera		1		1
Insecta	Ephemeroptera	Caenidae		1	1
Insecta	Odonata	Coenagrionidae	4	16	20
Insecta	Odonata	Corduliidae		2	2
Insecta	Trichoptera	Hydroptilidae		1	1
Insecta	Trichoptera	Leptoceridae	1		1
Malacostraca	Amphipoda	Hyalellidae	57	21	78
Malacostraca	Isopoda	Asellidae	23	3	26
Oligochaeta				6	6

Table A8. Kirksey 2019 transect macroinvertebrate abundance counts collected via D-net sampling technique.

			D-net			
Class/Subclass	Order/Suborder/Clade	Family	30 m	60 m	100 m	Total
Acari			2			2
Bivalvia	Veneroida	Dreissenidae	12	11	16	39
Gastropoda	Heterobranchia	Valvatidae	4		7	11
Gastropoda	Littorinimorpha	Rissooidea	229	352	107	688
Gastropoda	Planorboidea	Physidae	4	8	1	13
Gastropoda	Planorboidea	Planorbidae	11	2	1	14
Insecta	Diptera	Ceratopogonidae	1	1		2
Insecta	Diptera	Orthocladiinae	2			2
Insecta	Diptera	Pseudochironomini/Chironomini	2	1	6	9
Insecta	Diptera	Tanypodinae			4	4
Insecta	Diptera		1			1
Insecta	Trichoptera	Hydroptilidae	2			2
Insecta	Trichoptera	Leptoceridae	3			3
Malacostraca	Amphipoda	Gammaridae	4			4
Malacostraca	Amphipoda	Hyalellidae	24	8		32
Oligochaeta	Haplotaxida	Naididae		2	2	4
Oligochaeta			5			5
Turbellaria					1	1

Table A9. Sediment particle size fractions from 2019 Bear Creek wetland site sediment, organized by pond.

Pond	Site	% Size Fraction						
		Gravel/Cobble (>2 mm)	Very coarse sand (1-2 mm)	Coarse sand (0.5-1 mm)	Medium sand (250-500 µm)	Fine sand (125-250 µm)	Very fine sand (63-125 µm)	Silt/Clay (<63 µm)
West	1	0.1	0.4	3.8	49.0	32.5	9.5	4.8
	5	1.1	0.9	6.5	54.4	31.0	4.9	1.2
East	6	0.4	1.2	4.9	51.0	30.1	7.2	5.2
	8	2.8	1.7	8.5	51.7	26.8	6.1	2.5

Table A10. Bear Creek 2019 pond site macroinvertebrate abundance counts collected via ponar net sampling technique.

Class/Subclass	Order/Suborder/Clade	Family	West Pond			East Pond		
			Site 1	Site 5	Total	Site 6	Site 8	Total
Acari			1	2	3			
Bivalvia	Veneroida	Dreissenidae		1	1			
Bivalvia	Veneroida	Sphaeriidae		179	179			
Gastropoda	Heterobranchia	Valvatidae	1	3	4			
Gastropoda	Littorinimorpha	Rissooidea		17	17			
Insecta	Diptera	Ceratopogonidae		3	3	6		6
Insecta	Diptera	Chaoboridae	53		53	37	49	86
Insecta	Diptera	Chironomidae		4	4			
Insecta	Diptera	Pseudochironomini/Chironomini	9	67	76			
Insecta	Diptera	Stratiomyidae				1		1
Insecta	Diptera	Tanypodinae	1	3	4			
Insecta	Diptera	Tanytarsini		4	4			
Insecta	Trichoptera	Hydroptilidae		5	5			
Insecta	Trichoptera	Leptoceridae		1	1			
Malacostraca	Amphipoda	Gammaridae		22	22			
Malacostraca	Amphipoda	Hyalellidae		4	4			
Malacostraca	Isopoda	Asellidae					1	1
Oligochaeta			8	1	9	36		36

Appendix B: Fish-Based IBI for Muskegon Lake

We provide additional details regarding the development of a fish-based index of biotic integrity (IBI) used in this report as well as a description of how the IBI was used to set a delisting target for two beneficial use impairments (BUIs; loss of fish habitat and degradation of fish populations) in the Muskegon Lake Area of Concern (see Ruetz [2011] for additional details).

A multi-metric index—termed IBI—was used to set quantitative delisting targets for Muskegon Lake based on annual fish-sampling records collected by the Annis Water Resources Institute (AWRI) in 2004-2006. The IBI approach is widely used across the United States to monitor water quality. Fish are integrators of the overall habitat and water quality; fish also respond to both episodic and cumulative anthropogenic disturbances in an ecosystem. Fish sampling for calculating IBI scores only was required annually because the fish themselves are integrators of time (i.e., the fish assemblage is there continuously). A fish-based IBI can be used to address questions concerning both fish populations and habitat because the IBI is an indicator of both fish community health and overall ecological health of the water body.

A typical IBI includes metrics such as number and composition of species sampled, focuses on indicator species that are particularly sensitive to water quality and habitat alterations, and considers groups of organisms that have similar feeding modes. Once the sampling is complete, a "score" is calculated for each metric in the IBI. The final IBI score is the total of all metrics and is indicative of ecosystem health. A high score suggests a "healthier" ecosystem, whereas a low score is indicative of a "degraded" ecosystem.

The IBI used for setting delisting targets in Muskegon Lake is modified from a fish-based IBI developed for Great Lakes coastal wetlands (Uzarski et al. 2005). The IBI developed by Uzarski et al. (2005) was modified to better represent anthropogenic disturbance (based on land use and water quality) across a gradient of drowned river mouth lakes. The modified, fish-based IBI consisted of 11 metrics (Table B1). A revised fish-based IBI was recently published by Cooper et al. (2018) for Great Lakes coastal wetlands, which could be considered in future assessments.

At least three pieces of evidence suggested that fish populations and, therefore, habitat were no longer severely degraded in Muskegon Lake at the time the target was developed prior to 2009 (Ruetz 2011). First, the fish-based IBI scores calculated based on data collected during 2004-2006 suggested that the ecosystem health of Muskegon Lake was comparable to Pentwater Lake, a drowned river mouth lake that did not suffer the types of severe environmental degradation experienced by Muskegon Lake. Second, the 1987 Remedial Action Plan noted that Muskegon Lake experienced marked improvements in water and habitat quality, including an excellent fishery for numerous fish species, following the construction of a wastewater treatment system. Finally, assessments by the Michigan Department of Natural Resources suggested that Muskegon Lake supported good fishing for several fish species with self-sustaining populations (O'Neal 1997; Hanchin et al. 2007). Therefore, the proposed target for delisting the loss of fish habitat and degradation of fish populations BUIs in Muskegon Lake was to maintain or improve the lake's ecosystem health over a 3-year time span beginning in 2009. The numerical target was set as the average IBI score of ≥36, which was determined based on the mean IBI score during 2004-2006 minus one standard deviation. This target was achieved based on sampling during 2009-2011 (Fig. 18).

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Table B1. Metrics for fish-based index of biotic integrity (IBI) for drowned river mouths. The IBI is modified from Uzarski et al. (2005). Fish sampling should be conducted with fyke nets (Cooper et al. 2007) at shallow (depth ≤1 m) sites with submerged aquatic vegetation. At least three fyke nets should be fished at each site. The catch of fish is then standardized across nets at a site to calculate IBI scores.

Preliminary Drowned River Mouth Lake IBI – SAV habitat only

1. Percent omnivore abundance:
   >70% score = 0 | 50 to 70% score = 3 | <50% score = 5

2. Percent piscivore richness:
   <25% score = 0 | 25 to 35% score = 3 | >35% score = 5

3. Percent carnivore (insectivore+piscivore+zooplanktivore) richness:
   <70% score = 0 | 70-80% score = 3 | >80% score = 5

4. Smallmouth bass (*Micropterus dolomieu*) mean catch per net-night:
   0 score = 0 | >0 to 5 score = 3 | >5 score = 5

5. Insectivorous Cyprinidae richness:
   >3 score = 0 | >1 to 3 score = 3 | 0 to 1 score = 5

6. Percent Centrarchidae abundance:
   0-30 score = 0 | >30 to 60 score = 3 | >60 to 80 score 5 | >80 score = 7

7. Centrarchidae richness:
   0 to 1 score = 0 | >1 to 3 score = 3 | >3 score = 5

8. Mean evenness:
   <0.2 score = 0 | 0.2 to 0.6 score = 3 | >0.6 score = 5

9. Rock Bass (*Ambloplites rupestris*) catch per net-night:
   0 to 1 score = 0 | >1 to 5 score = 3 | >5 score = 5

10. Bluegill (*Lepomis macrochirus*) abundance per net-night:
    0 to 3 score = 0 | >3 to 20 score = 3 | >20 to 30 score = 5 | >30 score = 7

11. *Lepomis* catch per net-night:
    >50 score = 0 | >20 to 50 score= 3 | >5 to 20 score = 5 | 0 to 5 score = 7

Results from the Internal Phosphorus Loading Study in Muskegon Lake: Summer, 2020

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I. Introduction

Muskegon Lake was listed as a Great Lakes Area of Concern (AOC) in 1985 due to a long history of environmental abuse (Steinman et al., 2008). Ultimately, nine beneficial use impairments (BUI) were designated for Muskegon Lake, including eutrophication or undesirable algae. Given the many years of discharge from various industries located on the shoreline directly into the lake, total phosphorus (TP) concentrations became elevated, and were averaging >60 µg/L in 1973 (Freedman et al., 1979), indicative of eutrophic conditions.

In the early 1970s, point source discharges were regulated through the Federal Clean Water Act, and over time the TP concentrations have declined in Muskegon Lake. In the past few decades, TP concentrations have been averaging close to or below 25 µg/L (Steinman et al., 2008), which was established as the restoration target to remove the eutrophication BUI for the lake. However, P levels have increased slightly over the past few years (unpubl. data), and recent results obtained from the Muskegon Lake Observatory (https://www.gvsu.edu/wri/buoy/) suggest that mid-summer hypoxia in the lake may be inducing phosphorus release from the sediments (Weinke and Biddanda, 2018).

Internal phosphorus loading can be defined as all physical, chemical, and biological processes by which phosphorus is mobilized and translocated from the benthic environment (Steinman and Spears, 2020). IPL can delay the recovery of lakes, even after external loading is managed, as diffusion or wind-wave action can promote P release from the sediment into the water column (Jeppesen et al., 2005, Søndergaard and Jeppesen, 2020). This legacy phosphorus in the sediment may be decades old (Carey and Rydin, 2011), and working its way to the sediment-water interface due to concentration gradients within the sediment profile. If IPL is a significant source of phosphorus to the lake, then new management strategies may be needed to maintain TP concentrations below the restoration target.

Our study was designed to determine the importance of internal phosphorus loading as a source of P to Muskegon Lake. We collected sediment cores from three locations using established protocols (Ogdahl et al., 2014; Steinman et al., 2004), measured sediment P release rates and sediment P fractions.

II. Methods

Site Description:

Muskegon Lake has a surface area of ~17 km^2, with mean and maximum depths of 7 and 23 m, respectively. Average hydraulic retention time is ~23 days (Freedman et al., 1979). The major inflow (~95% of discharge) to the lake is the Muskegon River, with a navigation channel connecting to Lake Michigan as the primary outflow (Fig. 1). However, reverse flow can occur depending on prevailing wind direction (Liu et al., 2018), pulling Lake Michigan water inland.

Muskegon Lake has suffered from a long history of environmental abuse. Lumbering activity reached its peak in the mid 1880's, when 47 sawmills surrounded Muskegon Lake. After the crash of the lumber industry, due to the unsustainable harvesting practices of the time, industrial activity was attracted to Muskegon, and the lake's shoreline included foundries, metal finishing plants, a paper mill, and petrochemical storage facilities throughout much of early and mid-1990s. As of 2004, before shoreline restoration began, 64% of the lake's shoreline was hardened in some form (Steinman et al., 2008).

Field Methods:

Three sites were selected for sediment core collections (Fig. 1). Two of the sites were selected to be within the lake's previously established summer hypoxic zone (sites 1 and 2; Biddanda et al., 2018), although water depth limited the locations to depths <8 m. A third site was selected within the littoral zone on the southern shore of Muskegon Lake (site 3; Fig. 1), but physical conditions resulted in coring outside the littoral region. All collections were made on 14 July 2020.

At each site, dissolved oxygen (DO), pH, temperature, and total dissolved solids (TDS) were measured at the surface, middle, and bottom of the water column using a YSI 6600 sonde (only surface and bottom data presented; Table 1). Photosynthetically active radiation (PAR) profiles were measured using a LiCor Li-193SA spherical quantum sensor. Secchi disk depth was measured at each site to estimate water clarity. Water samples for P analysis were collected with a Niskin bottle. Water for soluble reactive phosphorus (SRP) analysis was syringe-filtered immediately through 0.45-µm membrane filters into scintillation vials. Samples were stored on ice until transported to the laboratory, within 5 h of collection. Total P (TP) and SRP samples were stored at 4°C until analysis.

Sediment core sampling and laboratory incubation followed the procedures of Steinman et al. (2004). A total of 18 cores (6 per site x 3 sites) were collected using a piston corer (Fisher et al., 1992, Steinman et al., 2004). After collection, each core was brought to the surface and the bottom was sealed with a rubber stopper prior to removal from the water, resulting in an intact sediment core that was ~20 cm in length, with a 25 cm overlying water column. One core was lost during sediment length adjustment in the laboratory. Core tubes were placed in a vertical rack and kept on ice during transit. An additional core was collected from each site for sediment chemistry analysis; the top 5 cm was removed for the chemical analysis (see below).

Lab Methods:

The 18 sediment cores (6/site) were placed in an environmental growth chamber with the temperature maintained to match the mean ambient condition in the hypolimnion at the time of collection (Table 1). The water column in three of the cores from each site was bubbled with N_2 (with 330 ppm CO_2) to create buffered hypoxic conditions, while the remaining three were bubbled with filtered air to create oxic conditions.

Internal load estimates were made using the methods outlined in Moore et al. (1998), with minor modifications (Steinman et al., 2004). Briefly, a 40 mL water sample was removed by syringe through the sampling port of each core tube at 0 h, 12 h, and days 1, 2, 4, 8, 12, 16, 20, 24, and 28. Immediately after removal, a 20 mL subsample was refrigerated for analysis of TP, and a 20 mL subsample was filtered through a 0.45 µm membrane filter and stored at 4^{o}C for analysis of SRP. TP and SRP were analyzed on a Seal AQ2 Discrete Analyzer (USEPA, 1993). SRP values below detection were calculated as one-half of the 5 µg/L detection limit. The 40 mL subsample was replaced with 1.0 and 0.2 µm sequentially filtered water collected from the corresponding site in the lake; this maintained the original volume and concentration gradient in the core tubes.

Flux (P release rate) calculations were based on the change in water column TP or SRP using the following equation (Steinman et al., 2004):

$$P_{rr} = ((C_t-C_0)/d) * (V/A) \quad [1]$$

where, P_{rr} is the net P release rate (positive values) or retention rate (negative values) per unit surface area of sediments, C_t is the TP or SRP concentration in the water column at time t, C_0 is the TP or SRP concentration in the water column at time 0, d is the number of days of incubation, V is the volume of water in the water column, and A is the planar surface area of the sediment cores. Both maximum and overall P release rates were calculated. Maximum release is based on P accumulation in the water column over the time period that resulted in the maximum apparent release rate, with the caveat that the initial and final samplings cannot be consecutive dates to avoid potential short-term bias. The overall P release rate is based on P accumulation from days 1 to 28.

Following the incubations, the top 5 cm of sediment was removed from each core. The sediment was homogenized, subsampled, and dried and then ashed (550°C for 1 hr) for AFDM (Steinman and Ogdahl, 2016) and TP. The ashed material was analyzed for TP as described previously. The additional sediment core was dried to a constant mass for metals (Fe, Al, Ca, Mg) and analyzed using EPA method 6010b (USEPA, 1996). Other subsamples (2 g) of wet sediment were sequentially fractionated (Psenner and Pucsko, 1988 modified by Hupfer et al., 2009 and Dieter et al., 2015) to identify the major P compounds in the sediment. The 1.0M NH_4Cl extraction produces the loosely sorbed P; the 0.11M buffered dithionite (BD) extraction produces reductant-soluble P (iron oxides and Mn-bound); the 1.0 M NaOH extraction produces Fe- and Al-bound P, which are mineral associations that can become soluble under hypoxic conditions; and the 0.5 M HCl extraction produces Ca-bound P, which represents a stable mineral association.

Statistical analysis

SRP and TP release rates under oxic and anoxic conditions were separately analyzed for both maximum release rate and overall release rate calculations using 2-way Analysis of Variance (ANOVA) tests, incorporating site (3 levels), oxygen state (2 levels: oxic and anoxic), and the interaction of site and oxygen state as factors. Normality assumptions were tested using Shapiro-Wilk tests; maximum release rates of both SRP and TP required log transformation.

Equal variance assumptions were tested using Brown-Forsythe tests. The presence of outliers was tested using Grubb's test. Significant differences were detected using $\alpha = 0.05$. Data analysis was conducted using SigmaPlot (v14.0; Systat Software, Inc.).

III. RESULTS

Field Conditions

Only one of the three sites conformed to our initial expectations regarding stratification and hypoxia. Site 1, at the eastern end of Muskegon Lake and near the edge of the purported hypoxic zone, showed a clear pattern of stratification with temperatures ~7ºC lower at bottom than the surface and DO concentrations ~1.2 mg/L compared to a surface concentration of 9.6 mg/L (Table 1). In contrast, site 2 was in shallower water and had a temperature difference between surface and bottom of only 2ºC with a bottom DO > 5 mg/L (Table 1). Site 3, just outside the littoral zone of the lake's south shore, where the shelf drops off quickly, showed signs of stratification with a 6ºC difference in temperature between surface and bottom, and a bottom DO concentration of just above 2 mg/L (Table 1). TDS concentrations were similar between surface and bottom depths at all sites, whereas pH was consistently 0.7 to 0.8 units lower at the bottom than at the surface.

Surface SRP concentrations were below detection at sites 1 and 2 but were 7 and 13 µg/L at the near bottom, respectively. In contrast, SRP was 13 µg/L at the surface but below detection at the near bottom at site 3 (Table 1). TP was detectable at all sites and followed the same trends seen for SRP, with concentrations higher at the bottom than surface at sites 1 and 2, and the reverse at site 3. TP concentrations ranged from 24 to 41 µg/L (Table 1).

The lake was relatively turbid with similar Secchi depths (1 to 1.1 m) and light extinction coefficients (~1.4 to 1.5) at all three sites (Table 1).

Phosphorus Release Rates

Phosphorus release rates were calculated as both a maximum apparent flux (using maximum release dates) and an overall flux using release from day 1 through day 28. The maximum release rate for SRP was constant and relatively low among the 3 sampling sites under oxic conditions, ranging from 0.71 to 0.77 $mg/m^2/d$ (Table 2, Fig. 2). Rates were much more variable under anoxic conditions due to episodic releases; at site 1, the spike occurred on day 2, and was modest in magnitude but consistent in all 3 replicates (Fig. 2). However, no spikes were measured at site 2, and only 1 replicate at site 3 showed a spike on day 8 under anoxic conditions (Fig. 2). The overall means ranged from 1.25 to 9.71 $mg/m^2/d$ (Fig. 2). Maximum release rates under anoxic conditions were significantly greater than under oxic conditions, but this reversed when measuring overall P release rates (Table 2). Neither site location nor the oxygen × site interaction was statistically significant at the 0.05 level (Table 2).

The maximum release rate for TP also was relatively constant and low among the 3 sites under oxic conditions, ranging from 0.62 to 0.98 $mg/m^2/d$ (Table 3, Fig. 3). Consistent with the SRP data, release rates were much more variable under anoxic conditions, and the TP followed the same general pattern as observed for SRP, although the TP spikes lagged the SRP spikes by a few days at sites 1 and 3 (Fig. 3). SRP accounted for ~45 to 65% of the TP being released from the sediment. Maximum TP release rates under anoxic conditions were significantly greater than under oxic conditions, as observed for SRP, but oxygen status had no significant effect when measuring overall P release rates (Table 3). The Site factor had a marginally significant effect on maximum TP release rates and was highly significant on overall release rate. Interestingly, site 1 had the greatest release rate when assessed with the maximum release rate approach, but the lowest release rate when using the overall release rate approach (Table 3). There was no significant interaction effect, regardless of measurement approach.

Sediment Fractionation

Post-incubation fractionation results were relatively consistent among sites and oxygen content (Table 4, Fig. 4). The greatest percent of SRP in the sediment was measured in the HCl fraction (Ca-bound SRP), ranging from 42-67%, followed by the BD fraction (Fe oxide- and Mn-bound SRP) ranging from 25-42% (Table 4). Lower values were measured in the NaOH fraction (Fe- and Al-bound SRP) at 5-25%, while very low (< 0.5%) amounts of SRP were associated with NH_4Cl fraction (loosely sorbed P; Table 4).

Sediment Metals and Content

Sediment content exhibited several trends (Table 5). First, Ca concentrations were elevated relative to the other measured metals, possibly a result of the lake's prior industrial history, which included concrete disposal in the lake and limestone aggregate stored along the lakeshore (Steinman et al., 2008). Second, the Al, Fe, and Mg concentrations, as well as sediment organic matter and TP content, all were noticeably greater at site 1 than at the other two sites (Table 5). Site 1 is closest to the now decommissioned and demolished coal-fired power plant located at the lake's eastern end, which may account for the elevated metals. Finally, the sediment Fe:P ratio ranged from ~10 to 21, with no clear pattern in oxic vs. anoxic conditions among sites (Table 5).

IV. Discussion:

One of the key stressors facing aquatic ecosystems is excess nutrients, leading to cultural eutrophication (Smith, 2003). This is a global phenomenon (Birk et al., 2020), with a growing recognition that the nutrient sources to lakes can be external, coming from the watershed and atmosphere (cf. Carpenter et al., 1998; Sharpley et al., 2013), as well as internal from the sediment (Olihel et al., 2017; Søndergaard et al., 2012).

Eutrophication or Undesirable Algae is a common BUI impairment for AOCs in the Laurentian Great Lakes (Hartig et al., 2018). The federal Clean Water Act regulated point source discharges, which significantly reduced nutrient loads to many water bodies in the US, but did not address nonpoint source loads. These diffuse inputs are now considered the major cause of harmful algal blooms, with specific regions of the Great Lakes (Green Bay, Saginaw Bay, western Lake Erie) being targeted for remediation through the federally-administered Great Lakes Restoration Initiative.

Muskegon Lake, another designated AOC, has exhibited significant reductions in TP since wastewater was diverted from direct discharge into the lake to the Muskegon County Waste Management System in the early 1970s. TP concentrations have declined from ca. 60 µg/L in 1972 to more recent concentrations of ca. 25 µg/L (Steinman et al., 2008). Weinke and Biddanda (2018) identified a DO threshold for TP concentration in Muskegon Lake; at DO levels <3 mg/L, TP concentrations averaged about 48 µg/L and >3 mg/L, TP concentrations declined to an average of 24 µg/L, providing further support to the putative role of internal loading.

Our laboratory incubations suggest that internal loading can be an important source of P to Muskegon Lake, but the process appears episodic and spatially heterogenous. Spatial heterogeneity of Muskegon Lake sediments is well-established, with both natural features (morphometry, hydrologic regime) and anthropogenic forces (prior lumber and industry on the shoreline) accounting for variation in sediment toxicity, chemistry, and biology (Carter et al., 2006; Liu et al., 2018; Nelson and Steinman, 2013; Rediske et al., 2002, Steinman et al., 2008). In the current study, there was considerable variation in sediment %OM and TP concentrations among the three collection sites. The greater maximum release rates at site 1 are consistent with the higher sediment OM and TP concentration at this site. One replicate sediment core at site 3 released TP up to ~800 µg/L under anoxic conditions, but the other two sediment cores, with similar sediment OM and TP concentrations to the high-release core, released virtually no TP during the same time period. It is plausible that mineralization of organic matter (e.g., phytoplankton) in the high-release core accounted for this spike of TP (cf. Palmer-Felgate et al., 2011). Episodic changes in SRP concentration in Muskegon Lake hypolimnion have been documented previously (Fig. 5; Mancuso, 2020), and have been attributed to changes in DO concentrations (Weinke and Biddanda, 2018).

Post-incubation fractionation data indicated that most of the sediment SRP was associated with the calcium and Fe oxide- and Mn-bound fractions. The relatively high Ca sediment concentrations at all our sampling sites in Muskegon Lake provide stable binding sites for P, but the Fe concentrations were spatially variable. Unfortunately, we did not fractionate sediments prior to incubation, precluding a comparison of how P fractions changed under oxic vs. anoxic conditions, although it make sense that the more mobile fractions would decline following incubations, especially in the anoxic treatments. The Fe:P ratio under oxic conditions at sites 2 and 3 slightly exceeded 15, above which P release is limited (Jensen et al., 1992). Although this ratio's applicability will depend on the dominant fraction to which P is associated, the relatively important role of Fe in Muskegon Lake suggests the ratio has relevance here, and is consistent with the higher maximum release rates measured under anoxic conditions at Site 1, which had a Fe:P ratio <15.

The sediment P flux varied considerably with the measurement approach. Hupfer et al. (2020) provide a comprehensive review of different approaches to measure internal P loading, acknowledging that both theoretical and methodological issues contribute to the complexity of these approaches. While it is recognized that laboratory-based sediment core incubations are not representative of whole lake or annual loads, given the spatial and temporal variation in lakes, the difference in internal loading estimates that derive from how the rate is calculated when employing sediment core incubations has received much less attention. Our study reveals that very dramatic differences in flux can be estimated from the same set of experiments, depending on the approach used.

The problem is compounded if these data are then extrapolated beyond the individual sampling dates to a longer time period, or beyond the sediment core surface area to a larger lake sediment surface area. To demonstrate the influence of different assumptions in assessing the importance of internal phosphorus loading in Muskegon Lake, we provide a set of estimates under various scenarios (Table 6). The main factors that are varied include the length of time of oxic vs. anoxic conditions; the amount of sediment surface area that is experiencing oxic vs. anoxic conditions; and whether the P release rate is based on maximum rate or entire incubation period.

Under the base scenario (1 and 2), we assume: 1) oxic conditions apply for 4 months of the year, anoxic conditions apply for 2 months of the year (Weinke and Biddanda et al., 2018), and there is no discernible release for 6 months of the year (Steinman et al., 2009); and 2) the percent surface area of Muskegon Lake that goes anoxic during those two months is 50% (Fig.

1). After correcting for area, the estimated combined internal load under oxic and anoxic conditions when applying the maximum release rate approach is 6.65 MT per year and when applying the overall incubation time (28 d) approach is 0.57 MT per year (Table 6). The external load entering Muskegon Lake from the Muskegon River (which accounts for 95% of the discharge; Freedman et al., 1979) is estimated at 24 MT (Marko et al., 2013). Hence the estimated internal P loads, based on the maximum and overall release rate calculations, are estimated to be 21.7% and 2.3% of the total load, respectively (Table 6). In scenarios 3 and 4, the duration of oxic, anoxic, and no P release periods in Muskegon Lake is changed to 8 mo, 2 mo, and 2 mo, respectively, while the surface area of oxic vs. anoxic is the same as in scenarios 1 and 2; in this case, the estimated combined internal load is 8.34 MT per year (25.8% of total P load) when applying the maximum release rate approach and 1.01 MT per year (4.1% of total load) when applying the overall incubation time approach (Table 6). Finally, altering the oxic vs. anoxic portions of lake surface area from 50:50 to 75:25 oxic:anoxic (scenarios 5 and 6) result in an estimated combined internal load of 4.59 MT per year (16.1% of total P load) with the maximum release rate approach and 0.62 MT per year (2.5% of total load) with the overall incubation time approach (Table 6).

These percent estimates of internal P loading in Muskegon Lake clearly have huge uncertainties associated with them. First, they don’t account for other external sources, such as other inflows (including eutrophic Bear Lake; Steinman and Ogdahl, 2015) and atmospheric deposition; second, it is unlikely that the anoxic portion of the lake remains entirely anoxic for those two summer months; third, while it is well documented that internal loading is significantly reduced in winter months due to reduction in mineralization and enzymatic reactions, it is unclear whether it drops to zero in Muskegon Lake; and fourth, P release rates are almost certainly not uniform across the entire lake bottom due to differences in bioturbation and sediment geochemistry (cf. Nogaro and Steinman, 2014; Hupfer et al., 2019). However, the data reveal two important points: 1) Only under the most extreme assumptions does internal P loading account for a sizeable (~26%) portion of total P load in Muskegon Lake; and 2) the role of internal loading, at least based on sediment core incubations, can be biased in different directions depending on assumptions.

Our results suggest that under certain conditions, such as low DO, sufficient sediment OM, and low Fe and Ca sediment concentrations, internal loading can contribute to elevated P concentrations in Muskegon Lake. However, this impact will be localized spatially and limited temporally. It is unclear if these pulses of P result in increased algal production and biomass, and if so, how widespread is the effect. Prior laboratory studies showed that algae grown in overlying water from P-release experiments did not significantly stimulate algal production (Cymbola et al., 2008) but there has been no attempt to evaluate this in the field. At this point, it doesn't appear that mitigation measures are needed to address internal P loading in Muskegon Lake.

Finally, one last unresolved question is whether this seasonal low DO state in Muskegon Lake is a natural phenomenon, given the morphometry and hydrodynamics of drowned river mouth lakes (Larson et al., 2013; Liu et al. 2018), or is it due to past water quality impairments. If it is the former, then it is questionable whether restoration efforts are appropriate under the AOC program; if it is the latter, then intervention may be required to achieve the restoration target for TP. A monitoring program that measures both DO and P levels in drowned river mouth lakes would provide the necessary information to determine if seasonal hypoxia/anoxia is a natural or anthropogenic-induced phenomenon and help quantify its ecosystem level impact.

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Table 1. Selected limnological characteristics of sampling sites in Muskegon Lake. BD = below detection (5 µg/L).

Parameter	Site 1		Site 2		Site 3	
	Surface	Bottom	Surface	Bottom	Surface	Bottom
Depth (m)	7.15		4.29		7.10	
Temp (°C)	26.31	19.61	26.23	24.52	26.10	19.92
DO (mg/L)	9.63	1.16	9.51	5.38	9.54	2.1
TDS (g/L)	0.226	0.220	0.222	0.235	0.220	0.214
pH	8.47	7.70	8.52	7.86	8.46	7.66
SRP (µg/L)	BD	13	BD	7	13	BD
TP (µg/L)	30	41	25	34	35	24
Light Extinction K_d	1.42		1.45		1.51	
Secchi depth (m)	1.0		1.0		1.1	

Table 2. Two-way analysis of variance of SRP release rates (mg $P/m^2/d$), with oxygen status and site location as the two factors. Separate analyses were run for the two different release rate calculation approaches. Maximum release rate values were log-transformed for analysis. Bold values indicate $P < 0.05$.

Site	Oxic	Anoxic	Redox P	Site P	Site × Redox
Maximum Release Rate					
1	0.76±0.30	9.71±5.35	**0.009**	0.206	0.282
2	0.71±0.39	1.25±0.37			
3	0.77±0.21	5.98±8.82			
Overall Release Rate					
1	0.27±0.11	-0.23±0.24	**<0.001**	0.061	0.744
2	0.44±0.22	0.08±0.06			
3	0.49±0.16	0.01±0.06			

Table 3. Two-way analysis of variance of TP release rates (mg P/m^2/d), with oxygen status and site location as the two factors. Separate analyses were run for the two different release rate calculation approaches. Maximum release rate values were log-transformed for analysis. Bold values indicate P < 0.05.

Site	**Oxic**	**Anoxic**	**Redox P**	**Site P**	**Site × Redox**
Maximum Release Rate					
1	0.88±0.55	14.90±7.41	**<0.001**	0.066	0.234
2	0.62±0.30	2.76±0.66			
3	0.98±0.13	9.01±10.48			
Overall Release Rate					
1	-0.08±0.10	-0.80±0.43	0.332	**0.004**	0.183
2	0.23±0.38	0.29±0.04			
3	0.50±0.38	0.61±0.54			

Table 4. Sediment fractionation SRP, TP, and organic matter (OM) mean concentrations from post-incubation Muskegon Lake cores, on a dry weight basis.

Site	Oxygen	SRP-bound Fraction								TP	OM
		NH_4Cl		BD		NaOH		HCl			
		µg/g	%	µg/g	%	µg/g	%	µg/g	%	mg/kg	%
1	Oxic	0.16	<0.1	223.4	33.3	165.8	24.7	280.7	41.9	1127	13.9
	Anoxic	0.16	<0.1	189.6	33.5	117.7	20.8	258.9	45.7	980	
2	Oxic	0.25	0.3	27.9	28.6	4.4	4.5	64.8	66.6	120	1.0
	Anoxic	0.16	0.2	45.5	42.0	6.5	6.0	56.2	51.9	198	
3	Oxic	0.05	<0.1	33.6	24.6	26.1	19.1	76.9	56.3	221	3.2
	Anoxic	0.70	0.5	43.2	30.0	16.1	11.2	84.1	58.4	195	

Table 5. Mean (±SD) metal concentrations (g/kg) from Muskegon Lake sediments (dry weight).

Site	Oxygen	Al	Ca	Fe	Mg	Fe:TP (by weight)
1	Oxic	6±5	39±34	14±12	15±13	12.4
	Anoxic	14±3	56±6	21±1	22±2	21.4
2	Oxic	1±0	71±44	2±0	1±0	16.7
	Anoxic	ND	68±18	2±0	1±0	10.1
3	Oxic	1±0	27±12	4±2	1±0	18.1
	Anoxic	1±0	30±13	4±2	1±0	20.5

Table 6. Muskegon Lake total internal P load and % of total P load accounted for by internal P loading release rates for 6 scenarios. See text for further description of scenarios. External load (24 MT/yr) is based on data from Marko et al. (2013) and does not account for all possible external sources, indicating the % internal load values are likely overestimates.

Scenario	Oxygen Status Duration (mo)			Surface Area (%)		P release rate approach		Total Internal Load (MT/yr)	Internal Load (%)
	Oxic	Anoxic	No release	Oxic	Anoxic	Maximum	Overall		
1	4	2	6	50	50	+		6.65	21.7
2	4	2	6	50	50		+	0.57	2.3
3	8	2	2	50	50	+		8.34	25.8
4	8	2	2	50	50		+	1.01	4.1
5	4	2	6	75	25	+		4.59	16.1
6	4	2	6	75	25		+	0.62	2.5

Figure 1. Map of sampling sites (circled numbers) in Muskegon Lake. Star denotes location of Muskegon Lake Observatory (buoy). Dashed contour line represents the 7.5/8.0 m isobath. Inset: Muskegon County outlined within the state of Michigan.

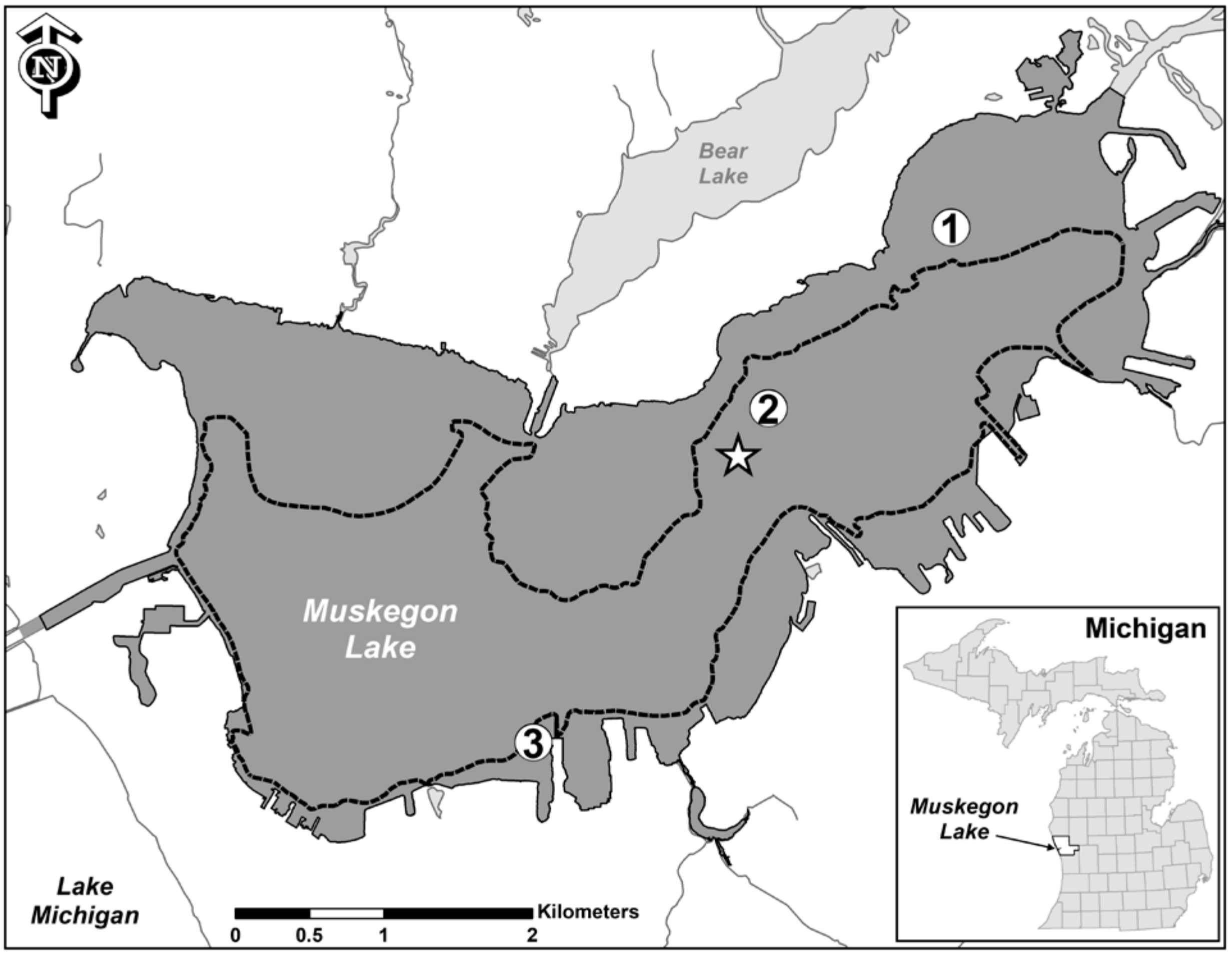

Figure 2. TP release rates from the 3 sampling sites in Muskegon Lake. Legend designations are site number (1-3), aeration treatment (O=oxic; N=anoxic), replicate (1-3).

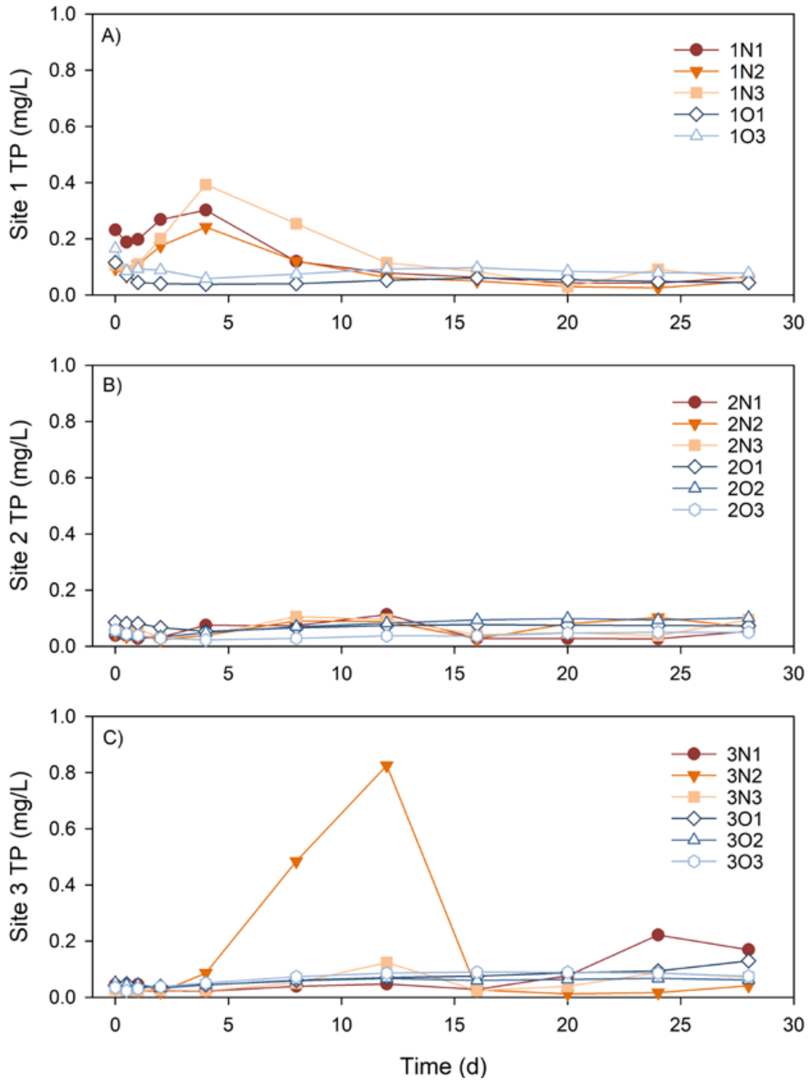

Fig. 3. SRP release rates from the 3 sampling sites in Muskegon Lake. Legend designations are site number (1-3), aeration treatment (O=oxic; N=anoxic), replicate (1-3).

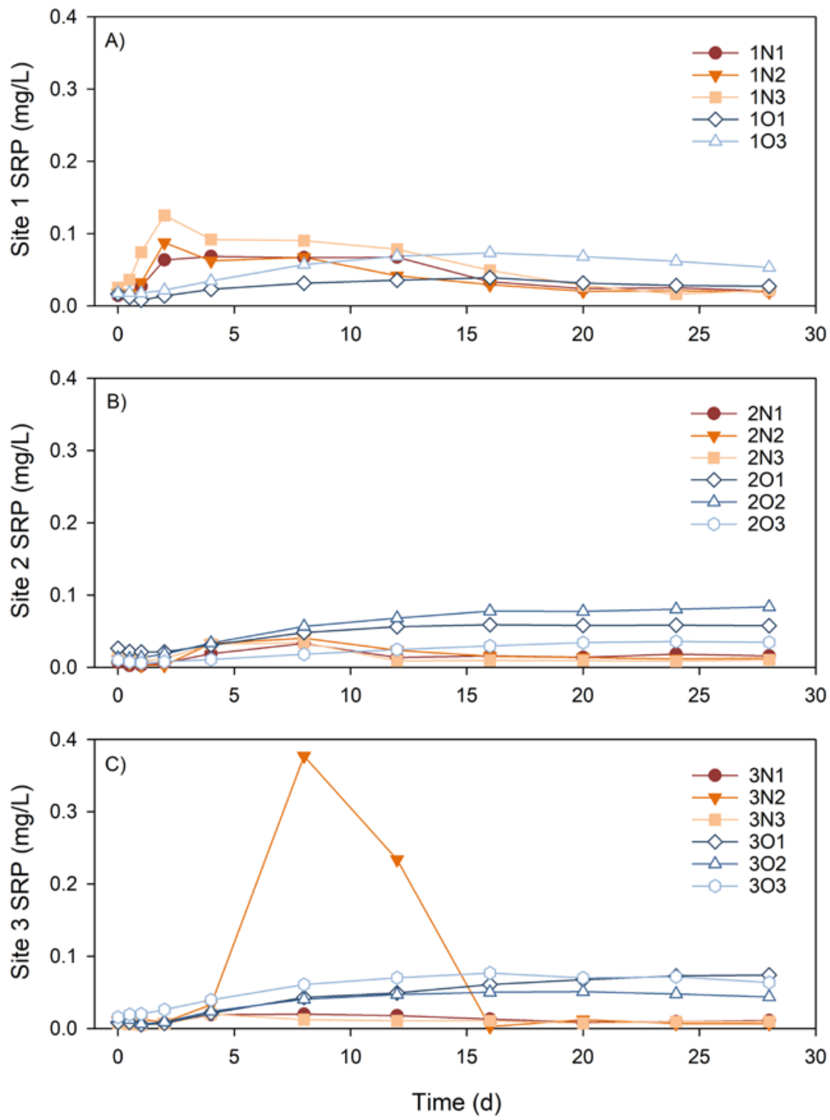

Fig. 4. Mean %SRP from Muskegon Lake sediment fractions. NH_4Cl fractions were less than 0.5% and difficult to see at the top of bars. O = oxic treatment; N = anoxic treatment.

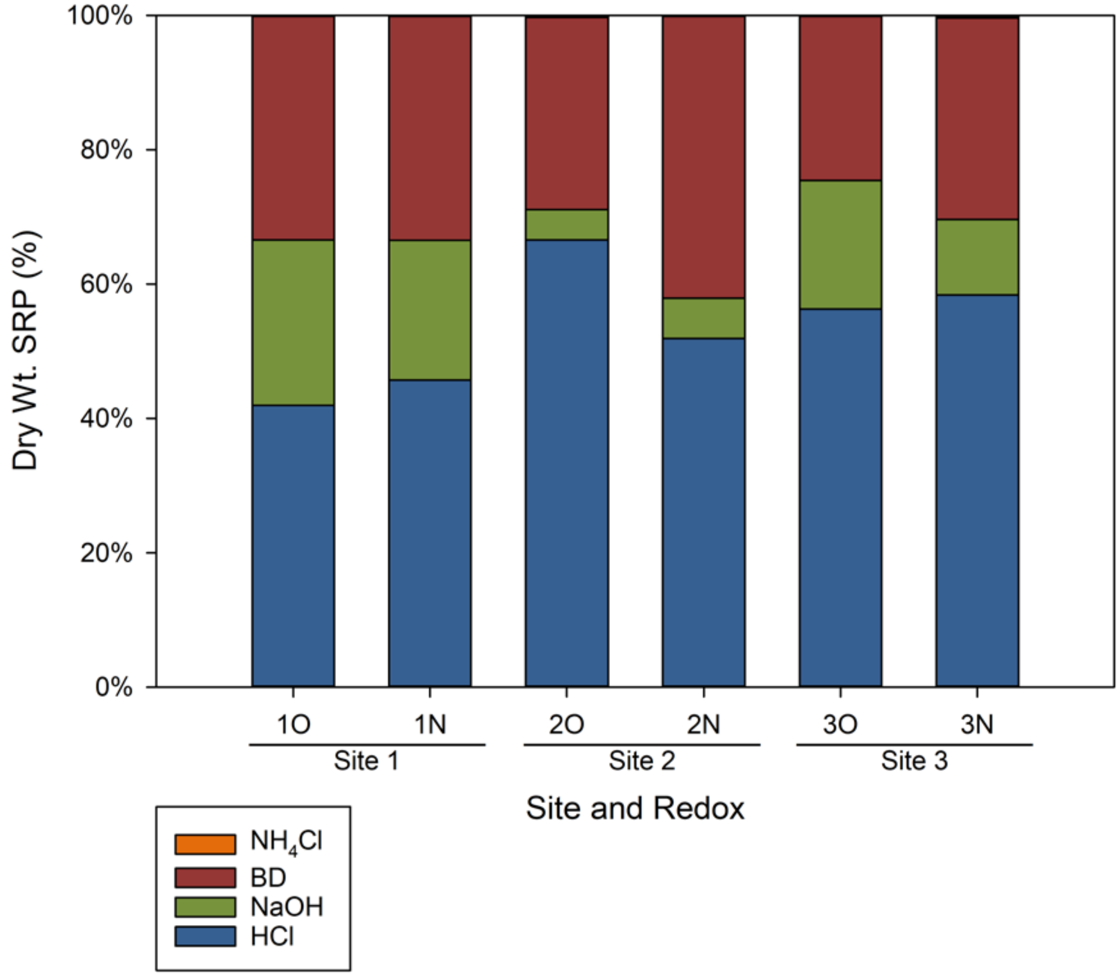

Figure 5. Soluble reactive phosphorus (SRP) concentrations (mg/L) from the Muskegon River inflow (River), near surface and near bottom at the buoy location in Muskegon Lake (MLO Top and MLO Bottom, respectively), and the channel connecting Muskegon Lake and Lake Michigan (Channel). Data from Jasmine Mancuso, M.S. thesis, GVSU.