PROJECT CLARITY
2019 Annual Monitoring Report
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1. Overview

Project Clarity is a large-scale, multidisciplinary, collaborative watershed remediation project aimed at improving water quality in Lake Macatawa. A holistic approach that includes wetland restoration, in-stream remediation, Best Management Practices (BMPs), and community education is being implemented as part of a multimillion dollar public-private partnership. The project is expected to have many economic, social, and ecological benefits – while achieving the ultimate goal of improved water quality in Lake Macatawa.

Lake Macatawa is the terminus of a highly degraded watershed and has exhibited the symptoms of a hypereutrophic lake for more than 40 years (MWP 2012, Holden 2014). Extremely high nutrient and chlorophyll concentrations, excessive turbidity, low dissolved oxygen, and a high rate of sediment deposition make it one of the most hypereutrophic lakes in Michigan (MWP 2012, Holden 2014). Nonpoint source pollution from the watershed, particularly agricultural areas, is recognized as the primary source of the excess nutrients and sediment that fuel the hypereutrophic conditions in Lake Macatawa (MWP 2012).

Because of this nutrient enrichment, Lake Macatawa and all of its tributaries are included on Michigan's 303(d) list of impaired water bodies, prompting the issuance of a phosphorus (P) Total Maximum Daily Load (TMDL) for Lake Macatawa in 2000. The TMDL set an interim target total phosphorus (TP) concentration of 50 µg/L in Lake Macatawa (Walterhouse 1999). Up until two years ago, monthly average TP concentrations often exceeded 125 µg/L, and at times exceeded 200 µg/L (Holden 2014). However, the past two years have resulted in annual mean TP concentrations of less than 100 µg/L. Nonetheless, even these concentrations are excessive, and meeting the TMDL target remains a major challenge in the Macatawa watershed. The TMDL estimated that a 72% reduction in phosphorus loads from the watershed would be required to meet the TP concentration target (Walterhouse 1999). Though remediation projects and BMPs focused on key areas in the watershed, Project Clarity is focused on reducing sediment and phosphorus loads, and working to meet the TMDL target for Lake Macatawa.

The Annis Water Resources Institute (AWRI) of Grand Valley State University, in cooperation with the Outdoor Discovery Center Macatawa Greenway (hereafter, ODC), the Macatawa Area Coordinating Council, and Niswander Environmental, has initiated a long-term monitoring program in the Lake Macatawa watershed. This effort provides critical information on the performance of restoration projects that are part of Project Clarity, as well as the ecological status of Lake Macatawa. The goal of the monitoring effort is to measure pre- and post-restoration conditions in the watershed, including Lake Macatawa. This report documents AWRI's monitoring activities in 2019, in combination with data reported previously from 2013-2018. Several changes in monitoring activities have occurred since the 2018 report: 1) we terminated sampling up- and downstream of the restored wetlands in April 2019 given the limited value of the information provided; and 2) we replaced that monitoring with sampling sediment within the two restored wetlands to determine how much P and what form of P is being accumulated (i.e., assessing the effectiveness of these restored wetlands in trapping and retaining phosphorus).

Although it will likely take many years before the benefits of restoration actions in the watershed are expressed in the lake, these initial results help establish the baseline conditions against which we can assess future changes, similar to what is being done in Muskegon Lake (cf. Steinman et al. 2008; Bhagat and Ruetz 2011; Ogdahl and Steinman 2015). We also include several Appendices (A: fish monitoring in Lake Macatawa; B: iron slag filter study; and C: the Lake Macatawa dashboard.

2. Methods

2.1 Overall site description

The Macatawa watershed (464 km^2/114,000 acres), located in Ottawa and Allegan Counties, includes Lake Macatawa, the Macatawa River, and many tributaries. It is dominated by agricultural (46%) and urban (33%) land uses, which have contributed to the loss of 86% of the watershed's natural wetlands (MWP 2012). The watershed includes the Cities of Holland and Zeeland and parts of 13 townships (MWP 2012). Lake Macatawa is a 7.2 km^2/1,780 acre drowned river mouth lake. It is relatively shallow, with an average depth of 3.6 m/12 ft and a maximum depth of 12 m/40 ft in the western basin. The Macatawa River, the main tributary to the lake, flows into the lake's shallow eastern basin. A navigation channel in the western end of the lake connects Lake Macatawa with Lake Michigan. AWRI's monitoring initiative is focused on 1) two key wetland restoration areas in the Macatawa watershed (Figs. 1, 2) and 2) Lake Macatawa. Details on these two efforts are provided below.

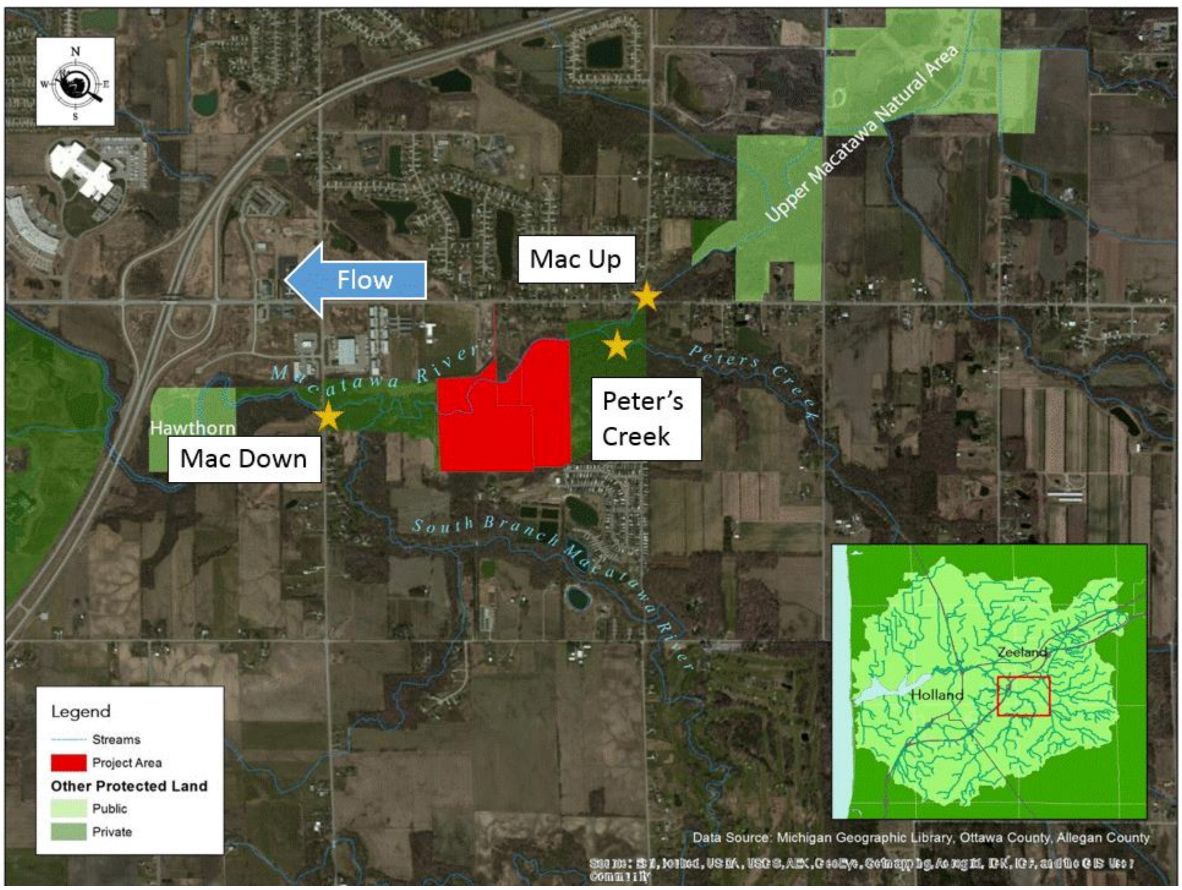

Figure 1. The Middle Macatawa wetland restoration study area, map provided by ODC. Sampling locations (n = 3), located on Peter's Creek and the Macatawa River, are indicated with gold stars. Insert shows were the property is located (red rectangle) within the Macatawa Watershed.

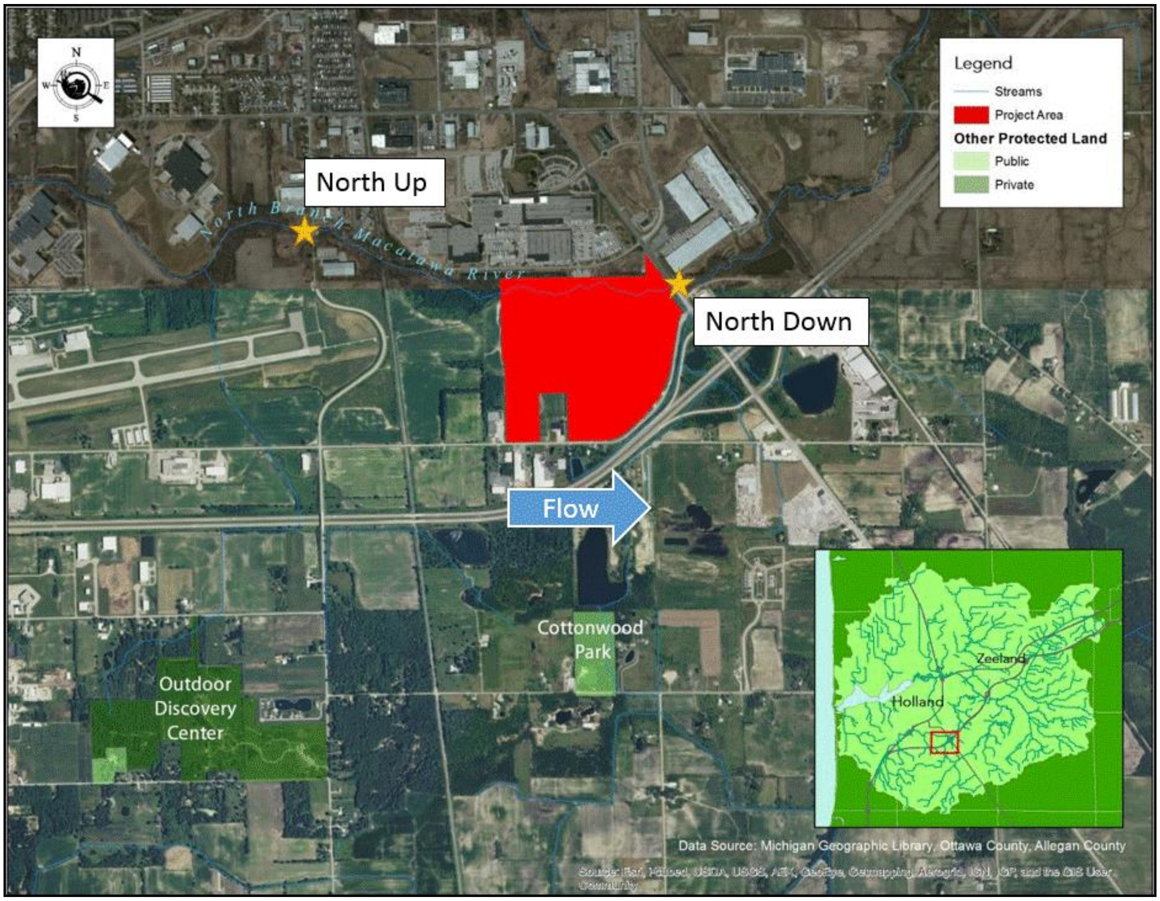

Figure 2. The Haworth wetland restoration study area, map provided by ODC. Sampling locations (n = 2), located on the North Branch of the Macatawa River, are indicated with gold stars. Insert shows were the property is located (red rectangle) within the Macatawa Watershed.

2.2 Wetland Restoration: Middle Macatawa & Haworth Properties

<u>*2.2.1 Monitoring & Data Collection*</u>

The Middle Macatawa and Haworth properties were acquired as part of Project Clarity and designated for wetland restoration. Restoration goals included slowing the flow of water in the Macatawa River and its tributaries, particularly during high flow events, thus trapping and retaining suspended sediments and nutrients. Restoration construction at Middle Macatawa and Haworth was completed in late September and early October 2015, respectively. Tributary monitoring (see below) was terminated in April 2019 given the limited value of the information provided, and replaced with sampling sediment within the two restored wetlands to determine how much P, and what form of P, is being accumulated (i.e., to assess the effectiveness of these restored wetlands at trapping and retaining phosphorus).

Tributary Monitoring: AWRI established monitoring sites upstream and downstream of each restoration area (Figs. 1 and 2). The Middle Macatawa study area (Fig. 1) has two upstream sites (Macatawa River [Macatawa Up] and Peter's Creek, which flow into the Macatawa River) and one downstream site (Macatawa River at the USGS gauging station [Macatawa Down]). The Haworth study area (Fig. 2) consists of monitoring locations upstream and downstream of the restoration area on the North Branch of the Macatawa River.

Water quality monitoring concluded in April 2019 and hydrologic monitoring concluded in Fall 2018. This report includes new data from December 2018 through April 2019; sampling occurred monthly during baseflow conditions and 1 storm event (~≥ 0.5 inches of rain proceeded by 72 hours of dry weather; Table 1). During each monitoring event, general water quality parameters (dissolved oxygen [DO], temperature, pH, specific conductivity, total dissolved solids [TDS], redox potential [ORP: oxidation-reduction potential – the degree to which a substance is capable of oxidizing or reducing another substance], and turbidity) were measured using a YSI 6600 sonde. Grab samples were collected for analysis of P (both soluble reactive phosphorus [SRP] and total phosphorus [TP]) and nitrogen (N: ammonia [NH_3], nitrate [NO_3^-], and total Kjeldahl nitrogen [TKN]) species. All water quality measurements and sample collection took place in the thalweg of the channel at permanently-established transects. Duplicate water quality samples and sonde measurements were taken every other month during baseflow conditions and all storm events. All samples were placed in a cooler on ice until received by the AWRI lab, usually within 4 hours, where they were stored and processed appropriately (see below).

Water for SRP and NO_3^- analyses was syringe-filtered through 0.45-µm membrane filters into scintillation vials; SRP was refrigerated and NO_3^- frozen until analysis. NH_3 and TKN were acidified with sulfuric acid and kept at 20°C until analysis. SRP, TP, NH_3, NO_3^-, and TKN were analyzed on a SEAL AQ2 discrete automated analyzer (U.S. EPA 1993). Any values below detection were calculated as ½ the detection limit for the purposes of analysis.

Stream hydrographs were generated at each monitoring location using water level loggers and staff gauges that were installed at permanently established transects at 4 of the monitoring locations (the Macatawa Down site did not require one because of the already established USGS gauge). Manual water velocity (using a Marsh McBirney Flow-mate 2000) and stage measurements were taken at each transect during each baseflow sampling event and over a range of high flow conditions to develop stage-pressure, stage-discharge, and pressure-discharge relationships. The best-fit model at each site was applied to the high-frequency pressure data recorded by the water level loggers to develop a stream hydrograph at each location (Chu and Steinman 2009).

Table 1. Precipitation summary for the 2019 storm event sampled by AWRI.

	3/15/2019
Rainfall (in)	1.07
Duration (h)	63
Intensity (in/h)	0.02

Sediment Analysis: Sediment cores were collected from the restored Haworth and Middle Macatawa wetlands on 8 and 15 October 2019, respectively, for sediment P analyses. Four coring site locations per wetland were determined using stratified random selection techniques to ensure that two cores were collected in each of the unexcavated and excavated restoration areas (Middle Macatawa) and that one core was collected from each of four wetland cells (Haworth). Coring site coordinates and maps are provided in Table 2 and Figs. 3, 4. When sufficient overlying water was present at coring sites, surface water was sampled with minimal disturbance prior to sediment coring; general water quality parameters, TP, and SRP were analyzed using the methods described above. Sediments were collected in triplicate using a modified piston coring apparatus (Fisher et al. 1992; Steinman et al. 2004). The modified piston corer was constructed of a 0.6-m long, 7-cm inner diameter, 7.6-cm outer diameter polycarbonate tube that was driven into the ground to collect cores of at least 10-15 cm depth of surface sediment. Cores were transported to the laboratory on the same day, extruded to isolate the top 10 cm of sediment, and refrigerated at 4°C. Within 24 hours of refrigeration, sediments were composited per site, homogenized and subsampled for sediment organic matter (OM), ash-free dry mass (AFDM), sediment TP concentration, isotherm, and fractionation analyses.

Sediment OM and AFDM were determined using gravimetric procedures (i.e., dry for 24 hr at 105˚C, weigh, ash at 550˚C for 1 hr, re-weigh; Steinman et al. 2006). The resultant ashed material was used for analysis of sediment TP on a Seal AQ2 Discrete Analyzer (U.S. EPA 1993).

Isotherm analysis, which provides an indication of the propensity of sediments to release or take up P from overlying water, were conducted in triplicate for each of the homogenized composite sediment cores. The procedure is a modification of Mozaffari and Sims (1994) and Novak et al. (2004). Briefly, tubes containing 3 g of sediment with 10 mL of inorganic P solutions (KH_2PO_4 dissolved in 0.01 M $CaCl_2$) containing 0, 0.01, 0.1, 1, 10, 50, 100, and 500 µg P/L were shaken for 24 h. After centrifugation and filtration (0.45 µm), the inorganic P in the supernatant was analyzed as described above. P sorption is calculated as the difference between the amount of P initially added to the tube and that in the solution at equilibrium.

Calculations were made as follows (after Olila and Reddy 1993):

- P lost after the 24-hr equilibration is considered sorbed (S_1):

$$S_1 = (V/m)(C_0 - C_{24})$$

  where C_0 = the concentration of P added (µg/L); V = total volume (mL); C_{24} = solution P concentration after 24-h equilibration (µg/L); and m = mass of dry sediment (g).

- Native sorbed P (S_0) was estimated using the least squares fit of the plot of S_1 vs. C_{24} at low P concentrations (i.e., during linear relationship):

$$S_1 = S_0 + bC_{24}$$

- The constant (y-intercept) was considered as the initial sediment P present in the adsorbed phase. The values for S_O and S_1 were added to obtain the corrected P sorption (S):

$$S = S_1 + S_0$$

- The equilibrium P concentration (EPC_0) of the sediments, defined as the solution P concentration at which S_1 = 0, was calculated from the equation:

$$EPC_0 = S_0/b$$

- The P sorption isotherm was constructed by plotting the mean quantity of P sorbed (mg/kg) against the mean P equilibrium concentration (mg/L) using the linear version of a Langmuir equation:

$$c/(x/m) = (1/S_{max})c + 1/(k)(S_{max})$$

  where x/m (mg/kg) is the quantity of P sorbed by the sediment, S_{max} (mg/kg) is the P sorption maxima, k (L/mg) is a sorption constant relative to P binding energy, and c (mg/L) is the P equilibrium concentration.

Sediment cores were separately subsampled and sequentially fractionated (based on Psenner and Puscko 1988 with modifications from Hupfer et al. 2009) at room temperature, resulting in four fractions: 1) Labile P (loosely sorbed) using 1M NH_4Cl; 2) Reductant-soluble P (iron hydroxides, Mn-bound) using 0.11 M Buffered Dithionite ($NaHCO_3/Na_2S_2O_4$); 3) Fe- and Al-bound P using 1M NaOH; and 4) Ca- and Mg-bound P using 0.5 M HCl.

Table 2. Site coordinates of post-restoration Haworth (10/8/2019, North Branch) and Middle Mac (10/15/2019, Macatawa River) wetland sediment coring.

Wetland	Site	Latitude	Longitude
Haworth	1	42.74596	-86.08406
	2	42.74385	-86.08310
	3	42.74400	-86.07831
	4	42.74548	-86.07771
Middle Mac	5	42.77905	-86.00589
	6	42.77777	-86.00518
	7	42.77703	-86.00856
	8	42.77772	-86.01037

Figure 3. Middle Macatawa wetland sediment coring sites (n=4 total; sites 5, 6 are in the excavated restoration area and sites 7, 8 are in the unexcavated area).

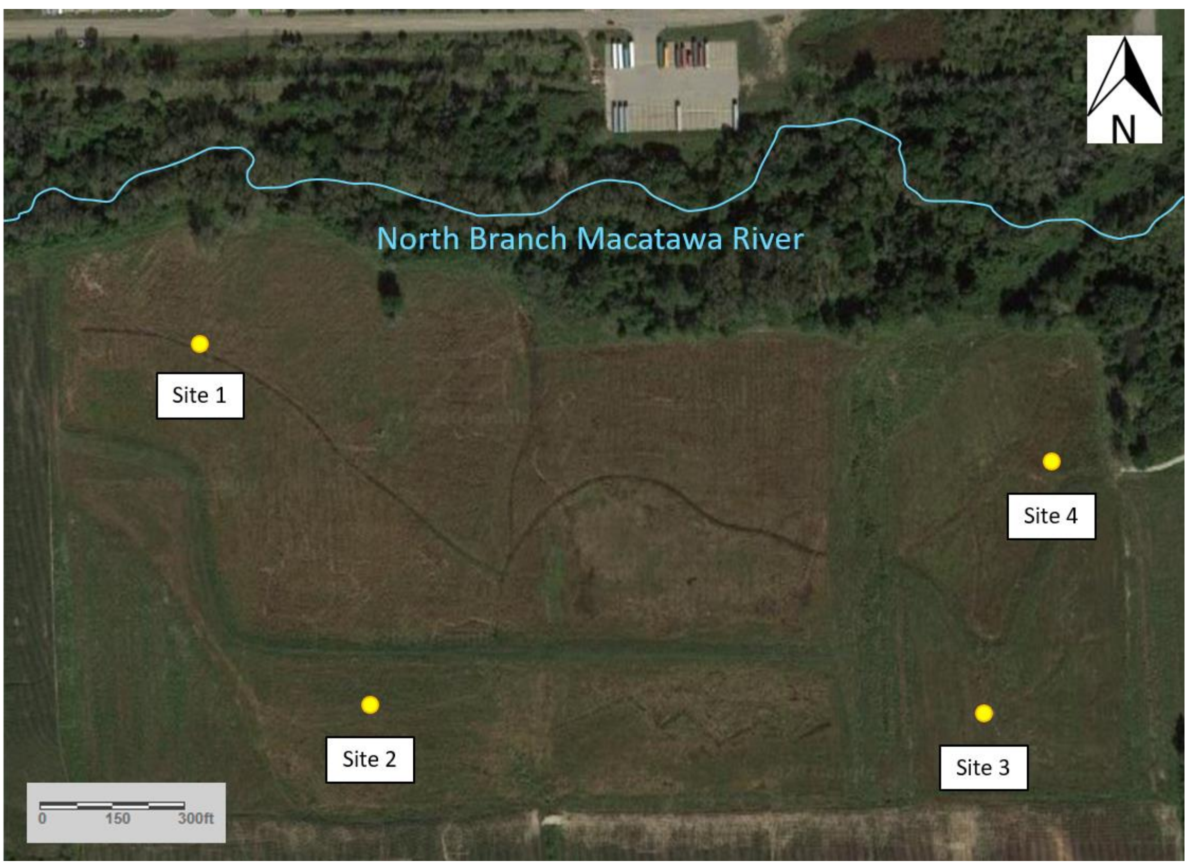

Figure 4. Haworth wetland sediment coring sites (n=4 sites total, 1 site per wetland cell).

2.2.2 Data Analysis

Our analysis focuses on characterizing water quality in the tributaries connecting to the two restored wetlands, and identifying 1) upstream vs. downstream differences during baseflow and stormflow conditions, and 2) pre- vs. post-restoration differences in nutrients and turbidity.

Tributary Monitoring - Upstream vs. Downstream:

Upstream-downstream differences between site pairs (e.g., North Up vs. North Down) within 2019 at baseflow and at stormflow were statistically tested using either a two-tailed paired t-test (normally-distributed data) or Mann-Whitney rank sum test (non-normally distributed data). A one-way analysis of variance test (ANOVA; normally distributed data) or Kruskal-Wallis test (one-way ANOVA on ranks; non-normally distributed data) were used to compare data from the three Middle Macatawa sites simultaneously. ANOVAs that detected significant differences were followed by post-hoc Tukey pairwise multiple comparison tests. Baseflow conditions were evaluated separately for each wetland site group; however, stormflow was not statistically analyzed in 2019 due to low sample size (n=1; 3/15/2019).

Tributary Monitoring - Pre- vs. Post-Restoration:

Past years of Project Clarity reporting have included statistical analysis of pre-restoration vs. post-restoration data, with minimal statistically significant differences detected overall at both wetlands. This is likely due to 1) the wetlands still being young and having suboptimal abilities to retain nutrients; and 2) high interannual variance. Due to a lack of detected effect thus far and the truncated sampling timeframe this year, tributary data in 2019 were not analyzed for pre- vs. post-restoration differences. Instead, we focused on characterizing restoration period trends as mean values.

Post-Restoration Wetland Sediment:

Wetland sediments were not collected prior to restoration activities, so a statistical comparison of pre- and post-restoration sediment condition is not possible. Sediment data analysis in 2019 focused on assessing the current P retention capacity of the restored wetlands. These datasets will serve as baseline data in future years, allowing us to rigorously measure P retention changes over time.

2.3 Lake Macatawa: Long-Term Monitoring

Water quality monitoring in the lake was conducted at 5 sites during spring, summer, and fall 2019 (Table 3, Fig. 5). The sampling sites correspond with Michigan Department of Environmental Quality (MDEQ, now EGLE) monitoring locations to facilitate comparisons with recent and historical data. At each sampling location, general water quality measurements (DO, temperature, pH, specific conductivity, TDS, ORP, turbidity, chlorophyll *a* [chl *a*], and phycocyanin [cyanobacterial pigment]) were taken using a YSI 6600 sonde at the surface, middle, and near bottom of the water column. Water transparency was measured as Secchi disk depth. Water samples were collected from the surface and near-bottom of the water column using a Van Dorn bottle and analyzed for SRP, TP, NO_3^-, NH_3, TKN, and chl *a*. Samples also were taken for phytoplankton community composition and archived for possible future analysis.

Table 3. Location and water column depth at Lake Macatawa long-term monitoring locations.

Site	Latitude	Longitude	Depth (m)
1	42.7913	-86.1194	8.5
2	42.7788	-86.1525	5.3
3	42.7872	-86.1474	3.7
4	42.7755	-86.1822	10.2
5	42.7875	-86.1820	4.4

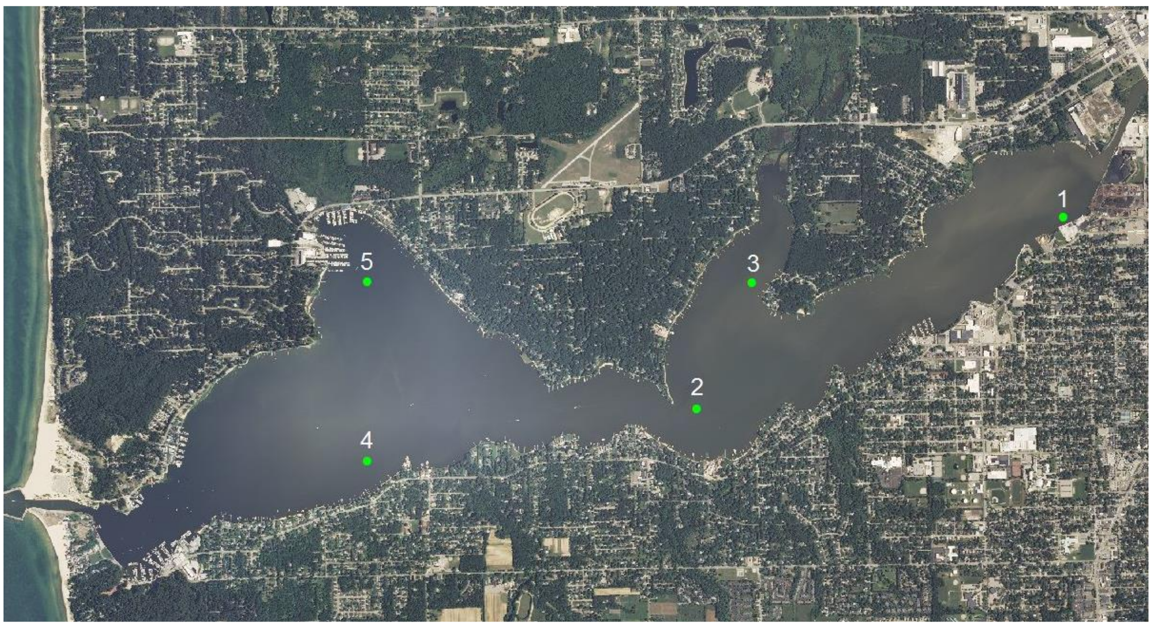

Figure 5. Map of Lake Macatawa showing the 5 sampling locations (green dots) for long-term water quality monitoring.

Water for SRP and NO_3^- analysis was syringe-filtered through 0.45-µm membrane filters into scintillation vials; SRP was refrigerated and NO_3^- frozen until analysis. NH_3 and TKN were acidified with sulfuric acid and kept at 20°C until analysis. SRP, TP, NH_3, NO_3^-, and TKN were analyzed as previously described. Chl *a* samples were filtered through GF/F filters and frozen until analysis on a Shimadzu UV-1601 spectrophotometer (APHA 1992). Any values below detection were calculated as ½ the detection limit for the purposes of analysis.

In addition, we continued testing for microcystin, which began in 2017. Microcystin is the most common toxin produced by cyanobacteria (blue-green algae). We used the ELISA QuantiPlate kit for Microcystins High Sensitivity, which is not as sensitive an assay as using High-Performance Liquid Chromatography (HPLC) but serves as a useful screening tool if microcystin is present in the lake. This kit has a greater detection limit than the QuantiTubes that were used in 2017 but still ranks below the HPLC for sensitivity. Advisories for microcystin consumption have been developed by the World Health Organization (WHO) and US EPA. For drinking water, the WHO advisory is triggered when microcystin concentrations >1 µg/L and the EPA advisory is >1.6 µg/L; for recreational use, WHO is >20 µg/L and EPA is >2 µg/L. Since Lake Macatawa is used only for recreation, we applied the latter two thresholds.

2.4 Macatawa Watershed Phosphorus – Precipitation Analysis

P concentrations in Lake Macatawa are influenced by many variables, but one of the most significant is precipitation because rain and snow events create surface and subsurface runoff from farms and developed areas, as well as result in atmospheric deposition, which can contain significant amounts of P (cf. Brennan et al. 2016). Consequently, it is of interest to know if changes in lake P concentrations are related to precipitation, land use changes, or a combination of the two.

Sophisticated (i.e., computationally intensive) watershed models are often used for this kind of analysis, but developing those models was outside our scope of work. Rather, we took a coarse-level approach to look at how TP concentrations near the Middle Macatawa restored wetland and in Lake Macatawa compared to precipitation amounts from the Tulip Airport in Holland using data from NOAA's National Centers for Environmental Information (NCEI, formerly the National Climatic Data Center), and Weather Underground. Linear regressions on TP and precipitation amount were conducted in SigmaPlot 14.0.

Additional value-added projects in 2019 include: Lake Macatawa fish community sampling and analysis (Appendix A), preliminary results of the iron slag filter study (Appendix B), and the Lake Macatawa Dashboard (Appendix C).

3. Results and Discussion

3.1 Wetland Restoration: Middle Macatawa Property

<u>3.1.1 Sampling Year 2019</u>

Baseflow:

Mean temperature (4.30-4.49°C) and DO concentrations (10.57-12.92 mg/L) remained similar among the three Middle Macatawa sites and were indicative of non-impaired water quality (Table 4). Mean specific conductivity met or exceeded the 600 µS/cm threshold of human induced stress in aquatic ecosystems (cf. Steinman et al. 2011). Conductivity, TDS, and turbidity were highest in Macatawa River sites and lowest in Peter's Creek (Table 4). Note that TDS is a calculation-based value that is automated from the YSI sensor based on conductivity, where SpCond*0.001*0.65 = TDS. Hence, when conductivity decreases during stormflow, so too does TDS.

Mean SRP concentrations ranged 23-40 µg/L among Middle Macatawa sites (Table 5, Fig. 6A). Mean TP concentrations were ~2-4x higher than SRP and ranged 65-131 µg/L (Table 5, Fig. 6C). SRP concentration was highest at the downstream site, but this difference was not statistically significant (Table 6; Fig. 8). TP was highest at the Macatawa upstream site and both the upstream and downstream sites were each significantly greater than Peter's Creek ($p < 0.001$; Table 6; Fig. 8).

Mean NO_3^- concentrations ranged 3.06-8.98 mg/L and made up a larger component of measured N at baseflow compared to both NH_3 (0.30-0.45 mg/L) and TKN (1.25 - 1.67 mg/L; Table 5; Fig. 7A, C, E). Peter's Creek was a significantly greater source of NO_3^- than other sites ($p < 0.001$; Table 6; Fig. 8). Although relatively lower in concentration, NH_3 concentrations of 0.1 mg/L usually indicate polluted surface waters, and >0.2 mg/L can be toxic for some aquatic animals (Cech 2003). TKN is composed of NH_3, ammonium (NH_4^+), and organic N compounds; ~21-36% of TKN was represented by NH_3, suggesting much of the reduced N in the Middle Macatawa tributaries is in the form of organic N (Table 5). Although NH_3 was not significantly different among sites ($p = 0.558$; Table 6; Fig. 8), TKN was significantly greater at the Mac Up site than at Peter's Creek ($p = 0.024$; Table 6; Fig. 8).

Stormflow:

Stormflow during the 3/15/19 sampling event decreased water temperature from the Dec.-Apr. baseflow mean of 4.4°C to ~2.5°C (Table 4). DO concentrations were similar between base- and stormflow (11.8 and 12.2 mg/L), which is less of a difference than has been seen in previous years (Table 4). However, due to the shortened 2019 sampling period monitoring only in colder months of the year this is an expected difference as this colder water has higher capacity to contain dissolved oxygen and was not averaged with data from warmer months. Mean specific conductivity and TDS decreased by ~50% per site during stormflow while turbidity increased from 6.9-16.6 NTU at baseflow to 61.6-114.4 NTU at stormflow (Table 4).

Nutrient data during the 3/15/2019 storm event generally followed expected trends based on previous sampling years in this watershed but may have been affected by the truncated seasonal sampling. Mean SRP and TP concentrations each increased by about 7- and 5-fold, respectively, during the storm, ranging 206-274 µg SRP/L and 413-618 µg TP/L (Table 5). Mean NO_3^- decreased 2-5 mg/L (~64%) at all sites while mean NH_3 and TKN increased (~0.17 mg/L and ~0.5 mg/L, respectively) from mean baseflow values during the 3/15/19 storm event, except for Mac Up TKN which slightly decreased during stormflow (Table 5).

The nutrient concentrations at baseflow following wetland restoration fluctuated over time (lines in Fig. 6 B, D and Fig. 7 B, D, F) but show no obvious trends post-restoration. In contrast, all post-restoration nutrient concentrations during stormflow, with the exception of nitrate, showed an overall decline since 2017 (symbols in Fig. 6 B, D and Fig. 7 B, D, F). **However, for almost all nutrients in 2019, the downstream concentrations during stormflow were greater than at least one of the two upstream sites, suggesting that the wetlands were likely serving as nutrient sources and not sinks.** This may be related to the time of year we sampled but at least for these data it appears that any nutrient reductions observed were likely related to changes in agricultural practices and not due to wetland filtration.

Hydrographs:

The stage-discharge relationship produced the best fit for models at the Macatawa Upstream ($y=0.0002x^2+0.0167x-0.1582$; $R^2=0.760$; Fig. 9) and Peter's Creek ($y=0.074e^{0.0277x}$; $R^2=0.723$; Fig. 10) sites. Although the model and observed data are reasonable fits based on the R^2 results, personal observations at these sites suggest these models are likely underestimating flow during high flow conditions; hence, additional data need to be collected during storm and high flow conditions to capture the extreme high flows at these sites.

3.1.2 Pre- vs. Post-Restoration Comparison

Baseflow:

Sampling in 2019 revealed little change in long-term post-restoration general water quality trends and continued trends previously seen in 2018. Temperature and turbidity were lower in post-restoration sampling but both parameters continue to have high variance, likely due to post-restoration having more winter sampling than the pre-restoration era. Specific conductance, TDS, and turbidity post-restoration means all decreased in 2019 from 2018 (Table 7).

Post-restoration mean P concentrations remained similar to pre-restoration values. Mean NO_3^- is higher in post-restoration sampling, but both NH_3 and TKN are lower during post-restoration (Table 8, Fig. 11); because these reductions occurred both upstream and downstream, we attribute the lower NH_3 and TKN concentrations to changes in agricultural practices and not to wetland filtration.

Stormflow:

Post-restoration stormflows changed only slightly from before, as might be expected, since we added only one 2019 storm; trends observed in 2018 remain true. Stormflow data once again have more complex, and sometimes site-dependent, trends in post-restoration than were seen in pre-restoration. Stormflow decreased temperature by 4-7°C, likely due to averaging more winter sampling events; DO correspondingly increased by ~3 mg/L (Table 7). Specific conductivity, TDS, and turbidity decreased in post-restoration, but due to high variances these are unlikely to be significant differences (Table 7, Fig. 12F). Mean SRP and TP concentrations increased from pre- to post-restoration, while NO_3^- and NH_3 decreased (Table 8, Figs. 12A-D). TKN decreased over time at Macatawa Up and increased over time at Peter’s Creek and Macatawa Down, suggesting changing ratios of organic N between restoration periods (Table 8, Fig. 12E). Overall, the increased mean P concentrations, and the reduced NO_3 concentrations, during post-restoration vs. pre-restoration at both upstream and downstream sites, suggest that these changes are not due to wetland restoration but likely associated with changing land use practices in the watershed.

The pre- vs. post-restoration analyses (both baseflow and stormflow) are conflated by unequal numbers of sampling events per season; although this analysis will be discontinued with the cessation of tributary monitoring, we recommend that any future analyses of these data should stratify by season when comparing pre- vs. post-restoration time periods.

Table 4. Mean (1 SD) values of selected water quality parameters at the Middle Macatawa wetland restoration site during the 2019 period of record (Dec. 2018 – Apr. 2019). Note that the number of observations (n) changes between baseflow and stormflow regimes. Stormflow water quality data sampling occurred 3/15/2019. NA = not applicable.

Flow	Site	n	Temp. (°C)	DO (mg/L)	SpCond (µS/cm)	TDS (g/L)	Turbidity (NTU)
Base	Mac. Up	5	4.49 (6.87)	10.57 (1.16)	701 (191)	0.456 (0.124)	16.6 (8.0)
	Peter's Creek	5	4.43 (5.70)	12.92 (0.58)	589 (55)	0.383 (0.036)	6.9 (3.5)
	Mac. Down	5	4.30 (6.02)	11.87 (1.15)	630 (115)	0.410 (0.075)	12.7 (8.7)
Storm	Mac. Up	1	1.64 (NA)	12.21 (NA)	316 (NA)	0.206 (NA)	61.6 (NA)
	Peter's Creek	1	3.03 (NA)	12.29 (NA)	312 (NA)	0.203 (NA)	85.7 (NA)
	Mac. Down	1	2.94 (NA)	12.13 (NA)	284 (NA)	0.184 (NA)	114.4 (NA)

Table 5. Mean (1 SD) values of selected water chemistry parameters for phosphorus (total phosphorus [TP] and soluble reactive phosphorus [SRP]) and nitrogen (nitrate [NO_3^-], ammonia [NH_3], and total Kjeldahl nitrogen [TKN]) at the Middle Macatawa wetland restoration site during the 2019 period of record (Dec. 2018 – Apr. 2019). Data are divided into baseflow and stormflow conditions. Stormflow water quality data sampling occurred 3/15/2019.

Flow	Site	n	SRP (µg/L)	TP (µg/L)	NO_3^- (mg/L)	NH_3 (mg/L)	TKN (mg/L)
Base	Mac. Up	5	32 (27)	131 (36)	3.06 (0.61)	0.38 (0.32)	1.67 (0.43)
	Peter's Creek	5	23 (17)	65 (32)	8.98 (1.31)	0.45 (0.10)	1.25 (0.08)
	Mac. Down	5	40 (24)	117 (53)	5.29 (1.19)	0.30 (0.18)	1.41 (0.12)
Storm	Mac. Up	1	206 (NA)	413 (NA)	1.03 (NA)	0.52 (NA)	1.56 (NA)
	Peter's Creek	1	241 (NA)	495 (NA)	3.16 (NA)	0.63 (NA)	1.70 (NA)
	Mac. Down	1	274 (NA)	618 (NA)	2.02 (NA)	0.49 (NA)	1.95 (NA)

Table 6. Statistical analysis results comparing 2019 upstream vs. downstream water quality parameters at Middle Macatawa tributary sampling sites at baseflow. Stormflow 2019 data were not statistically analyzed due to low sample size (n=1). Parameter column indicates water quality parameter and transformation used to meet assumptions of normality and variance. Data were analyzed using either 1-way ANOVA (1WA) or Kruskal-Wallis 1-way ANOVA on ranks (r). Significant differences (p-values < 0.050) between sites are indicated with bold text and not significantly (NS) different data are in plain text.

Flow	Parameter	Test	p-value	Notes
Base	SRP	1WA	0.504	NS
	sqrt TP	**r**	**<0.001**	**Mac. Up > Peter's Creek; Mac. Down > Peter's Creek**
	NO_3^-	**1WA**	**<0.001**	**Peter's Creek > Mac. Down > Mac. Up**
	NH_3	1WA	0.558	NS
	TKN	**r**	**0.024**	**Mac. Up > Peter's Creek**
	Turbidity	1WA	0.135	NS

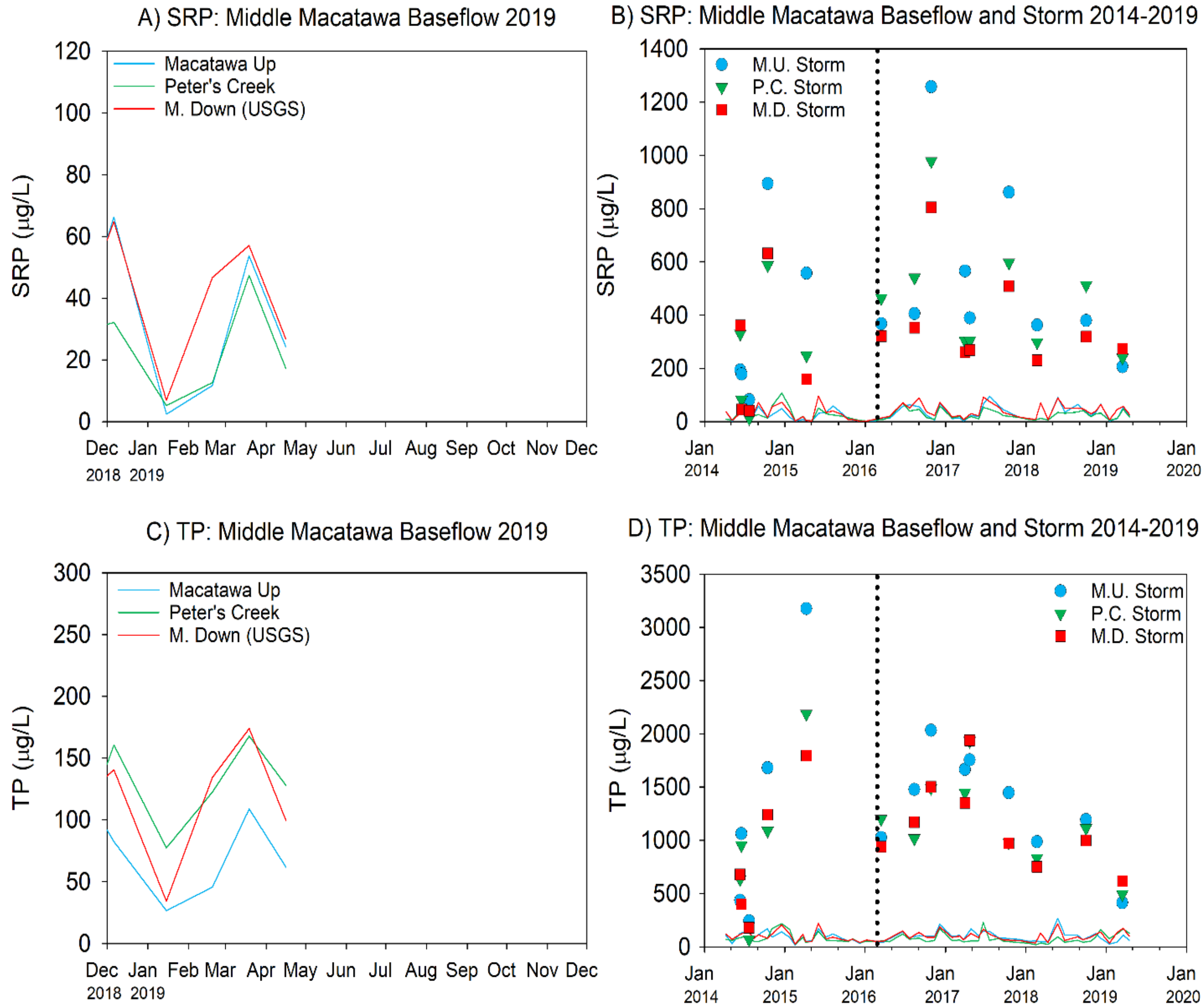

Figure 6. Soluble reactive phosphorus (SRP) (A, B) and total phosphorus (TP) (C, D) concentrations measured at Middle Macatawa restoration site in 2019 (A, C) and over total project history (B, D). Colored data lines in A and C magnify the 2019 baseflow data shown in B and D, which allow us to include both baseflow and storm event concentrations in same graph; Symbols represent storm events. Note changes to scales of y-axes. Vertical dotted lines represent approximate completion date of wetland restoration construction. Legend in A, C also applies to B, D. Vertical dotted line represents approximate completion date of wetland restoration construction.

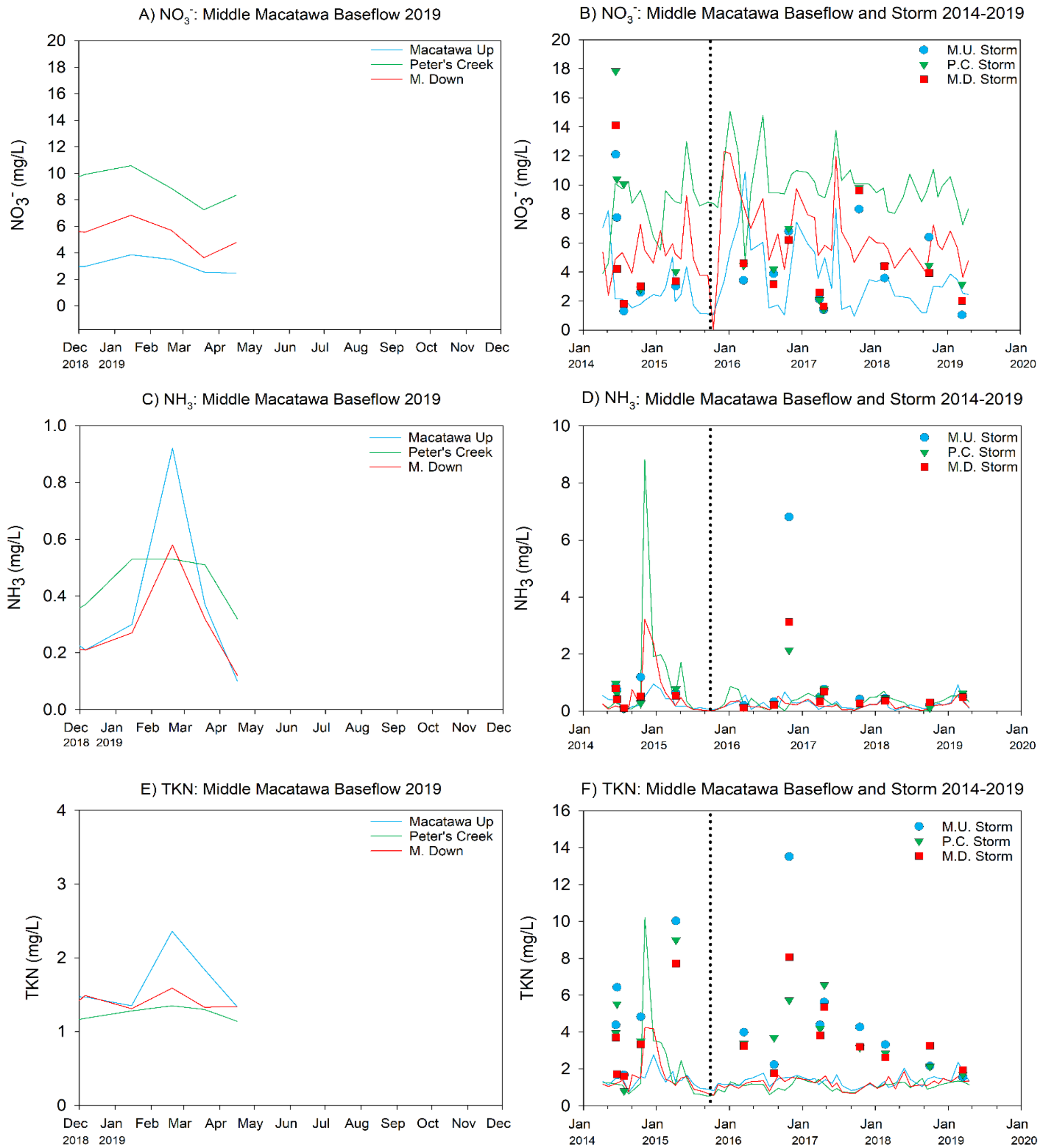

Figure 7. Nitrate (NO_3^-) (A, B), ammonia (NH_3) (C, D), and total Kjeldahl nitrogen (TKN) (E, F) concentrations measured at the Middle Macatawa restoration site in 2019 (A, C, E) and over total project history (B, D, E). Colored data lines in A, C, and E magnify 2019 baseflow data shown in B, D, and F, which allow us to include both baseflow and storm event concentrations in same graph; symbols represent storm events. Vertical dotted lines represent approximate completion date of wetland restoration construction. Note changes to scales of y-axes. Legend in A, C, E also applies to B, D, F.

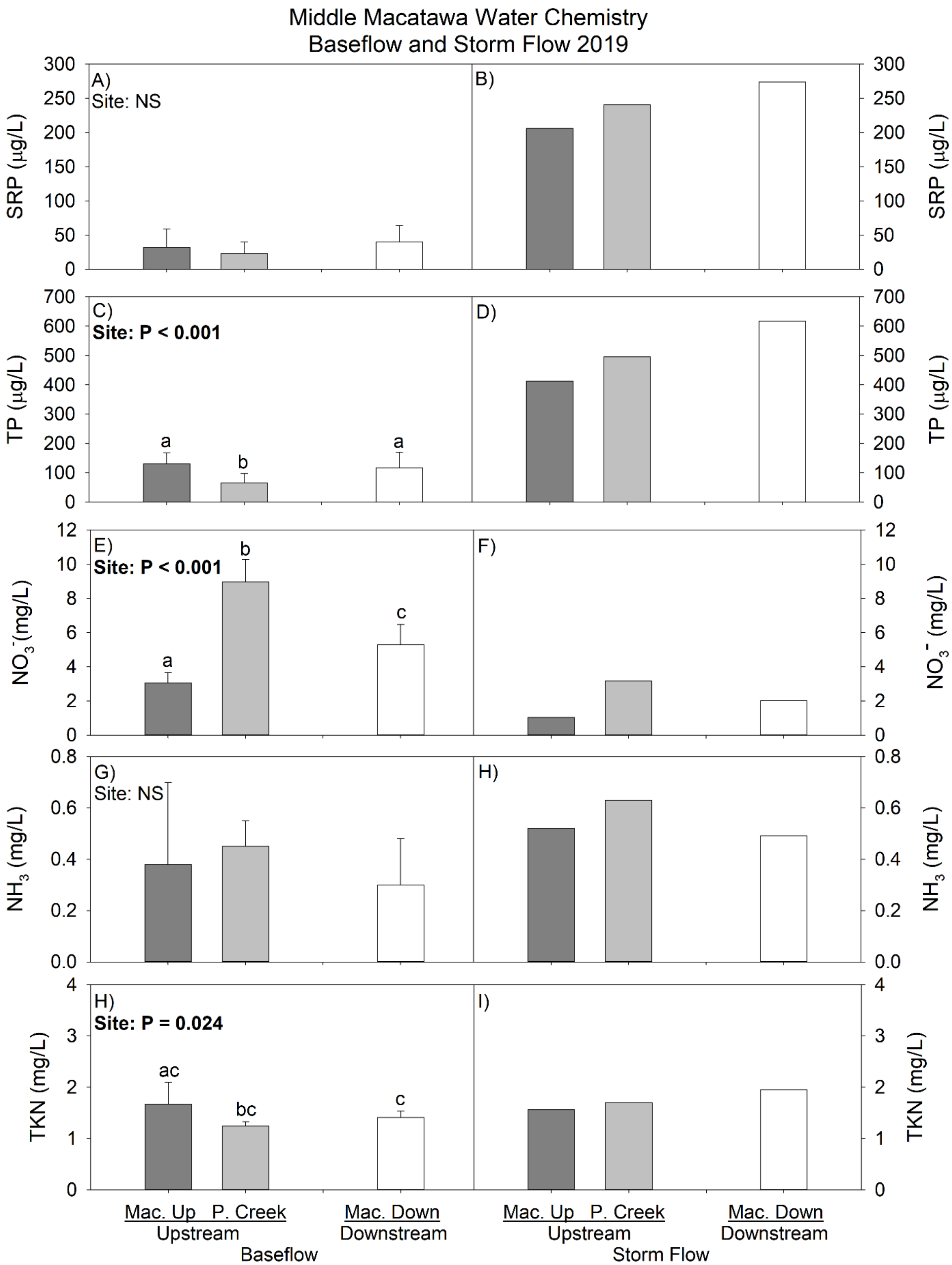

Figure 8. Middle Macatawa mean (1 SD) water chemistry at baseflow (A, C, E, G, I) and stormflow (B, D, F, H, J) in the 2019 sampling year (Dec. 2018 – April 2019). River water from Macatawa Up and Peter's Creek sites flow together and combine before reaching Macatawa Downstream site. Lower case letters indicate significant differences between sites. NS = not significant.

Mac Up Hydrograph

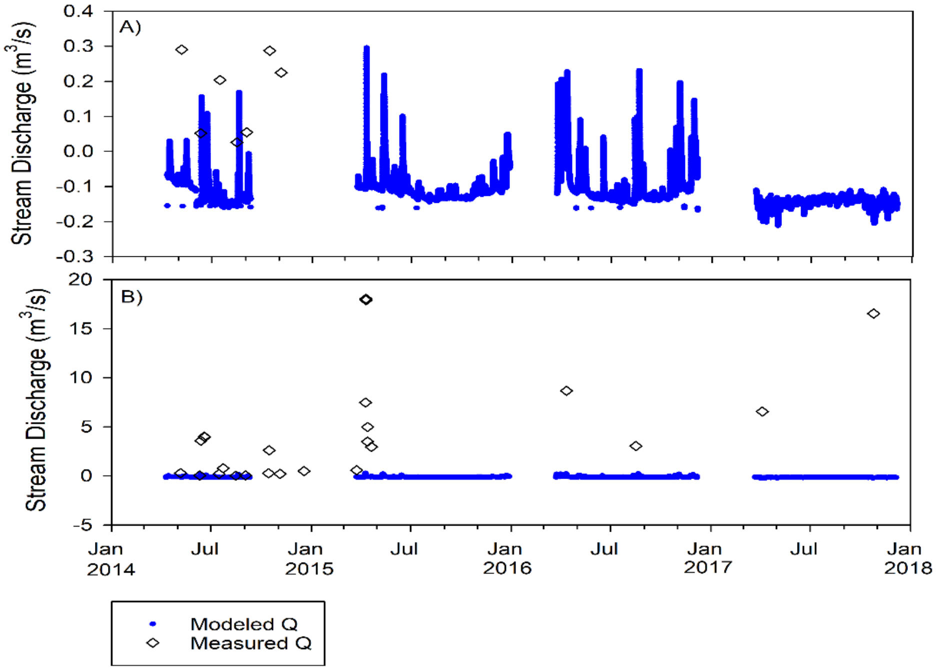

Figure 9. Macatawa Upstream site hydrologic modeling. Panel A displays detail of modeled stream discharge (Q) and panel B displays full range of in situ measured Q. Note the different y-axis scales.

Peter's Creek hydrograph

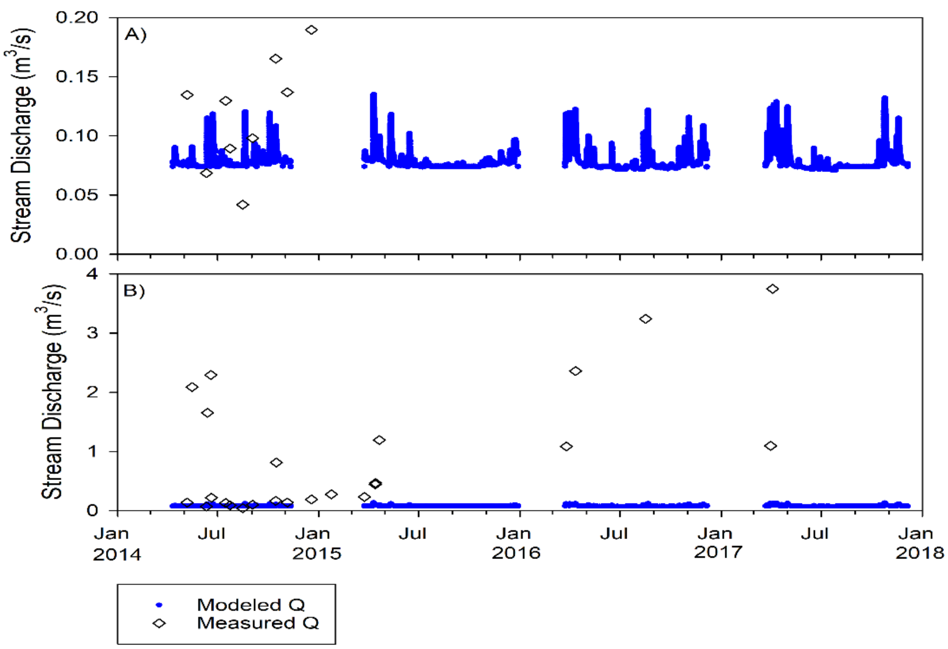

Figure 10. Peter's Creek site hydrologic modeling. Panel A displays detail of modeled stream discharge (Q) and panel B displays full range of in situ measured Q. Note the different y-axis scales.

Table 7. Grand means (1 SD) of selected water quality parameters at the Middle Macatawa wetland restoration site. Each cell has two rows per column: data in the top row represent entire pre-restoration period of record (Apr. 2014 – Sept. 2015); data in the bottom row represent entire post-restoration period of record (Oct. 2015 – Apr. 2019). Note that the number of observations (n) changes between flow regimes and restoration periods. Date of storm sampling events: Pre - 6/12/14; 6/18/14; 7/23/14; 10/15/14; 4/9/15. Post - 3/14/16; 8/12/16; 10/27/16; 3/30/17; 4/20/17; 10/15/17; 2/20/18; 3/15/2019.

Flow	Site	Period	n	Temp. (°C)	DO (mg/L)	SpCond (μS/cm)	TDS (g/L)	Turbidity (NTU)
Base	Mac. Up	Pre	18	12.34 (7.50)	10.26 (2.23)	711 (184)	0.462 (0.119)	14.7 (12.4)
		Post	38	11.13 (8.46)	10.10 (2.46)	751 (130)	0.488 (0.084)	11.6 (7.9)
	Peter's Creek	Pre	18	12.35 (7.38)	10.45 (2.39)	665 (163)	0.432 (0.106)	11.3 (6.6)
		Post	38	10.70 (7.54)	10.92 (2.20)	654 (89)	0.425 (0.058)	7.3 (4.9)
	Mac. Down	Pre	18	12.17 (7.40)	10.53 (2.39)	765 (240)	0.497 (0.156)	10.5 (6.9)
		Post	38	10.49 (7.97)	10.75 (2.29)	706 (86)	0.459 (0.056)	8.5 (6.0)
Storm	Mac. Up	Pre	3	14.26 (6.78)	7.43 (2.68)	444 (207)	0.288 (0.135)	581.7 (697.8)
		Post	8	9.54 (7.49)	10.46 (2.76)	370 (113)	0.241 (0.073)	314.0 (206.0)
	Peter's Creek	Pre	2	17.00 (3.75)	7.49 (0.81)	460 (201)	0.299 (0.130)	141.6 (182.5)
		Post	8	9.72 (6.65)	10.76 (2.67)	318 (110)	0.207 (0.072)	326.9 (270.5)
	Mac. Down	Pre	3	14.00 (6.66)	7.88 (2.42)	481 (201)	0.313 (0.130)	462.2 (475.9)
		Post	8	9.78 (7.09)	10.55 (2.68)	348 (114)	0.226 (0.074)	293.6 (208.2)

Table 8. Grand means (1 SD) of selected water chemistry parameters at the Middle Macatawa wetland restoration site. Each cell has two rows per column: data in the top row represent pre-restoration period of record (Apr. 2014 – Sept. 2015); data in the bottom row represent post-restoration period of record (Oct. 2015 – Apr. 2019). Data are divided into baseflow and stormflow conditions. Data are divided by baseflow and stormflow conditions and by pre- and post-restoration periods, respectively. Note that the number of observations (n) changes between flow regimes and restoration periods. See Table 7 for dates of storm sampling events.

Flow	Site	Period	n	SRP (μg/L)	TP (μg/L)	NO_3^- (mg/L)	NH_3 (mg/L)	TKN (mg/L)
Base	Mac. Up	Pre	18	27 (19)	101 (44)	2.90 (2.00)	0.32 (0.25)	1.41 (0.46)
		Post	38	31 (27)	101 (50)	3.63 (2.27)	0.24 (0.18)	1.36 (0.33)
	Peter's Creek	Pre	18	30 (26)	88 (53)	8.54 (2.19)	1.05 (2.06)	1.98 (2.26)
		Post	38	23 (17)	66 (40)	9.99 (1.89)	0.32 (0.24)	1.08 (0.25)
	Mac. Down	Pre	18	37 (27)	104 (51)	5.20 (1.51)	0.56 (0.87)	1.59 (1.02)
		Post	38	37 (27)	95 (45)	6.41 (2.49)	0.21 (0.14)	1.21 (0.30)
Storm	Mac. Up	Pre	5	381 (339)	1319 (1181)	5.35 (4.49)	0.71 (0.41)	5.47 (3.07)
		Post	9	533 (328)	1333 (486)	4.11 (2.55)	1.13 (2.13)	4.56 (3.60)
	Peter's Creek	Pre	4	254 (261)	687 (454)	10.28 (6.14)	0.49 (0.39)	3.45 (1.96)
		Post	9	471 (229)	1169 (418)	4.56 (2.54)	0.59 (0.62)	3.71 (1.58)
	Mac. Down	Pre	5	248 (251)	860 (657)	5.31 (4.99)	0.48 (0.25)	3.62 (2.48)
		Post	9	372 (182)	1138 (407)	4.24 (2.47)	0.66 (0.94)	3.70 (1.95)

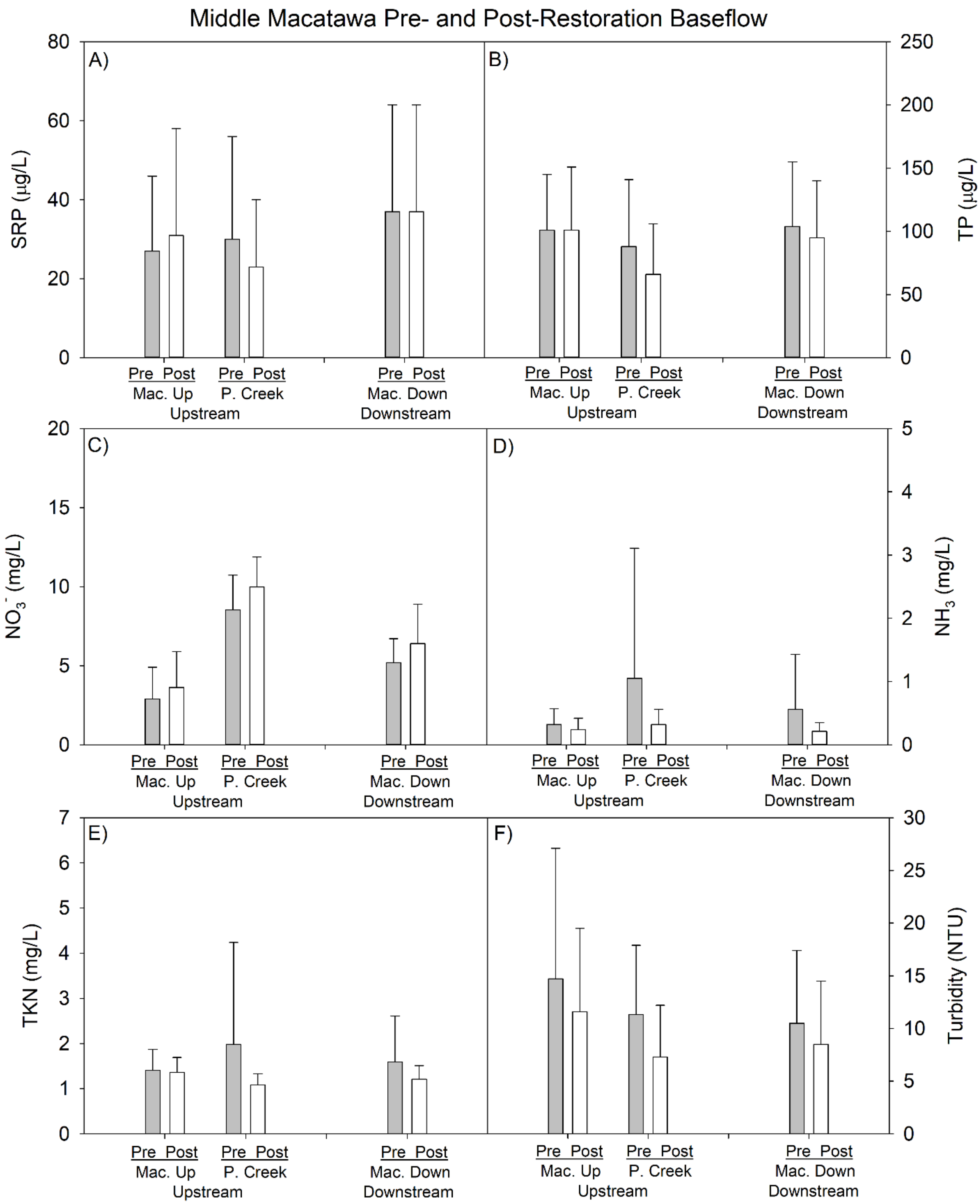

Figure 11. Middle Macatawa pre- and post-restoration water chemistry comparison at baseflow as of 2019 sampling year. Error bars represent 1 SD.

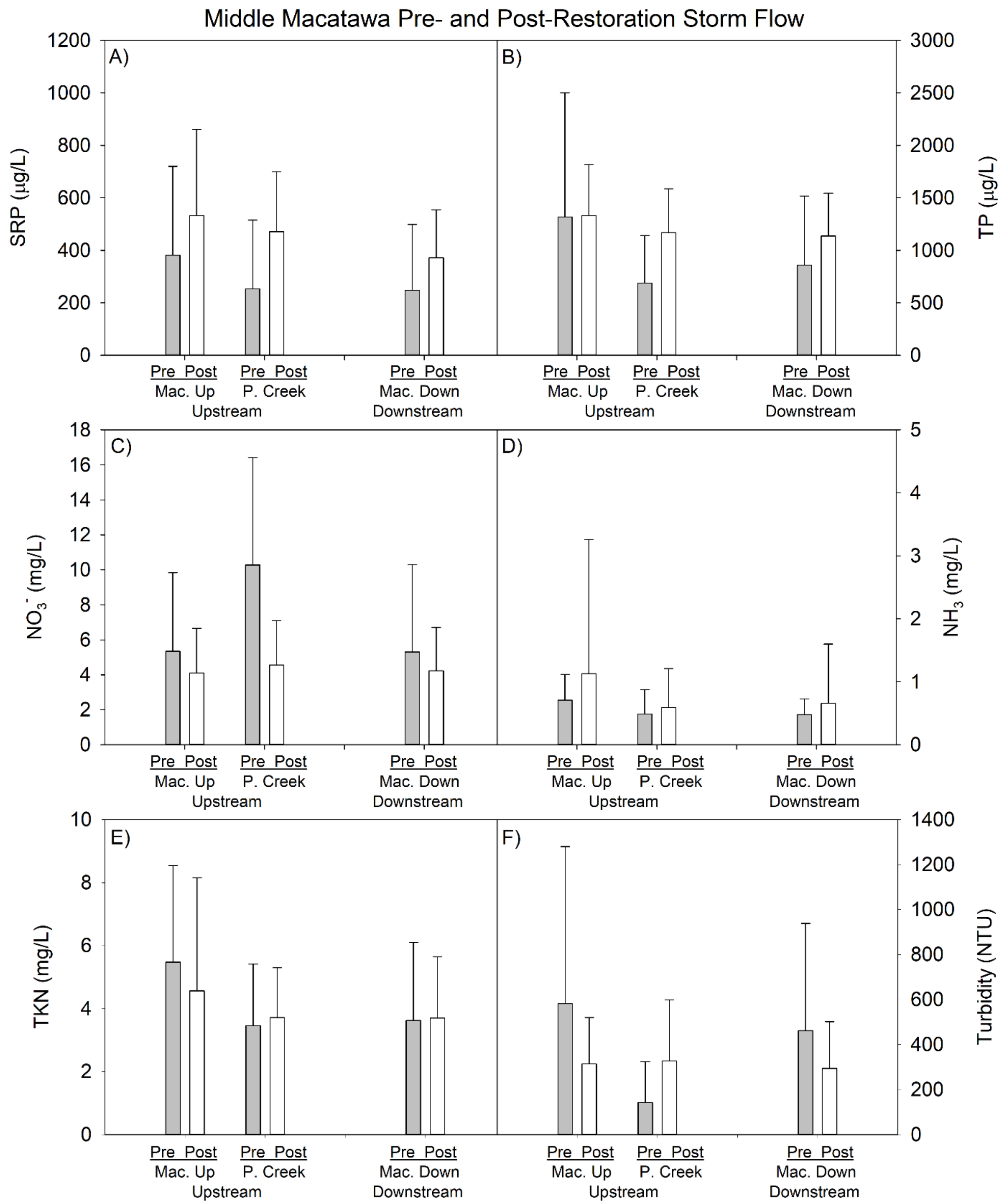

Figure 12. Middle Macatawa pre- and post-restoration water chemistry comparison at stormflow as of 2019 sampling year. Error bars represent 1 SD.

3.1.3 Wetland Sediment Analyses

Overlying water was present at all Middle Macatawa wetland sites allowing surface water quality and P chemistry to be collected and analyzed. Sites 5 and 6 are in the excavated restoration wetland area, which is more frequently inundated with water due to their closer proximity to the inlet area connecting this wetland to the Macatawa River; sites 7 and 8 are in the more densely vegetated, non-excavated wetland area closer to the wetland's diffuse outlet area and berm, and tend to be drier.

Water temperature was consistent among sites, ranging 7.7-9.2°C (Table 10). DO concentration was higher at excavated sites 5 and 6 than unexcavated sites 7 and 8 (respectively 9.9 and 5.4 mg/L; Table 10), which is expected due to the higher leaf litter, decomposition, and biological oxygen demand in non-excavated sites. Specific conductivity ranged 265-336 µS/cm at all sites and TDS ranged from 0.172-0.218 g/L (Table 10). Turbidity varied within the restoration area and with proximity to inlet and outlet areas; pipe-proximate sites 5 and 8 (21.9 and 10.3 NTU) were higher than sites 6 and 7 (2.1 and 6.7 NTU; Table 10). Overlying site water column SRP and TP concentrations were much greater at the unexcavated sites (7 and 8) than the excavated sites (5 and 6), suggesting that removal of sediment-laden P (and associated organic matter) during excavation removed major sources of P to the overlying water (cf. Oldenborg and Steinman 2019). Indeed, the SRP and TP concentrations in the overlying water at the unexcavated sites were extremely high (SRP: 442 and 1577 µg/L; TP: 1185 and 2087 µg/L; Table 10; Fig. 13). These high concentrations may be attributable to diel hypoxia, due to increased nighttime respiration associated with mineralization of organic matter; this would result in P release from Fe-bound P in the sediments (Steinman and Spears 2020).

Surface (top 0-10 cm) sediment TP within cores ranged 468-1473 mg/kg dry weight (Table 11, Fig. 14A). Unexcavated site 7 and 8 cores had more organic matter than excavated sites 5 and 6 (~11% vs. ~4%; Table 11, Fig. 14B), which is consistent with above conjecture that mineralization of organic matter may also be a major source of P to the overlying water.

EPC_0 was lower in excavated sites 5 and 6 (mean = 0.001 and 0.016 mg/L) than at sites 7 and 8 (mean = 0.023 and 0.032 mg/L; Table 11, Fig. 15); regardless of site, EPC_0 was lower than the overlying water SRP (and all measured stormflow SRP concentrations collected at the upstream site during post-restoration monitoring; Fig. 15). These EPC values suggest the P concentration gradient should result in the sediments serving as a sink for the P in the overlying water.

Fractionation of the top 10 cm of sediments showed that at all sites, the NaOH-P fraction (Fe- and Al-bound P) was consistently the largest fraction (54-140 µg/g) of Middle Macatawa wetland sediment; this fraction is redox sensitive so if the water column went anoxic, the iron-bound P could be released into the water column, which may be accounting, at least in part, for the very high P concentrations in the overlying water. The loosely sorbed NH_4Cl-P fraction, which is easily released from the soils, was present only in minimal quantities (0.01-0.02 µg/g; Fig. 16). At all sites except 5, the sediment P concentrations were generally equivalent to the P sorption maxima, suggesting that the soils had little capacity to adsorb more P.

Overall, the sediment P studies indicate that despite the EPC_0 values, the P-saturated sediments have very limited capacity or binding sites, if any, to take up additional P. This may be responsible for the very high P concentrations in the overlying water; alternatively, the sampling date corresponded with the downward slope of a recent high flow event. There may not have been sufficient time for the high P water entering the wetland to be absorbed/adsorbed by the biota/sediment before we sampled. Future sampling and analyses will help us determine the reason for the high P in the overlying water.

Table 10. Middle Macatawa wetland site mean general water quality parameters and P chemistry of overlying site water on 10/15/2019. Sites 5 and 6 are in the excavated wetland area; sites 7 and 8 are in the unexcavated wetland area. SpCond= specific conductivity.

Parameter	Site				Wetland Mean (1 SD)
	5	6	7	8	
Temperature (°C)	8.87	7.72	9.21	7.84	8.41 (0.74)
DO (mg/L)	9.92	9.81	6.54	4.28	7.64 (2.73)
SpCond (µS/cm)	325	265	335	336	315 (34)
TDS (g/L)	0.211	0.172	0.218	0.218	0.205 (0.022)
Turbidity (NTU)	21.9	2.1	6.7	10.3	10.3 (8.5)
SRP (µg/L)	40.5	208.8	442.6	1576.6	567 (693)
TP (µg/L)	183.3	356.4	1185.4	2087.3	953 (874)

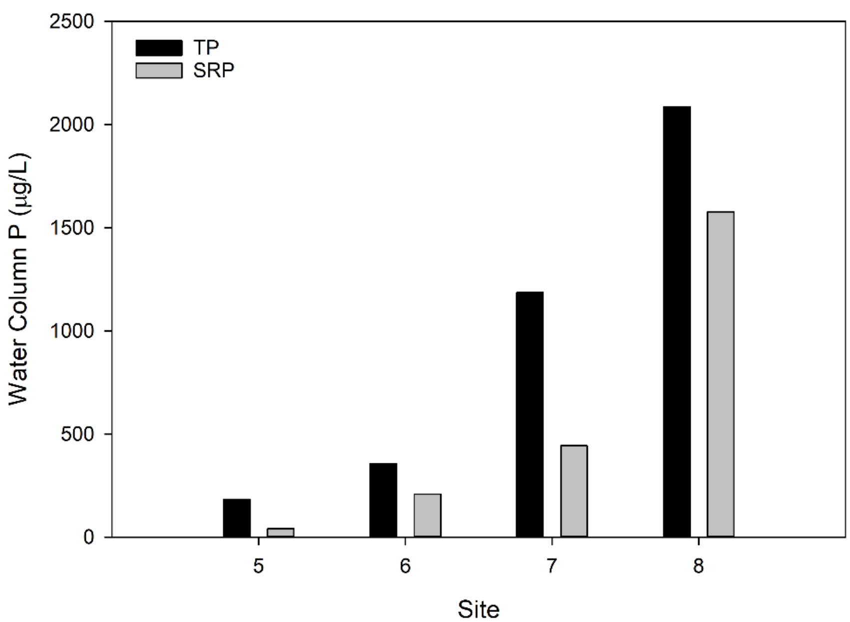

Figure 13. Middle Macatawa post-restoration wetland overlying water column P chemistry during 10/15/2019 sediment coring. Sites 5 and 6 are in the excavated wetland area; sites 7 and 8 are in the unexcavated wetland area.

Table 11. Middle Macatawa wetland site and mean (1 SD) sediment characteristics from cores collected 10/15/2019. All sediment is homogenized from near surface depth (0-10 cm). TP = total phosphorus, OM = organic matter, EPC_0 = equilibrium phosphorus concentration, S_{max} = phosphorus sorption maximum.

Site	Standing Water (cm)	Core Replicate	Sediment TP (dry, mg/kg)	Sediment OM (%)	EPC_0 (mg/L)	S_{max} (mg/kg)
5	17	A	496	4%	0.002	876.75
		B			0.001	931.62
		C			0.001	896.22
6	19	A	681	4%	0.017	670.90
		B			0.013	426.06
		C			0.018	778.29
7	18	A	1159	9%	0.023	1486.20
		B			0.023	1236.77
		C			0.022	1488.56
8	21	A	1473	11%	0.034	1533.98
		B			0.033	1524.75
		C			0.030	1443.03
Wetland Mean			962	7%	0.018	1107.76
SD					0.013	415.33

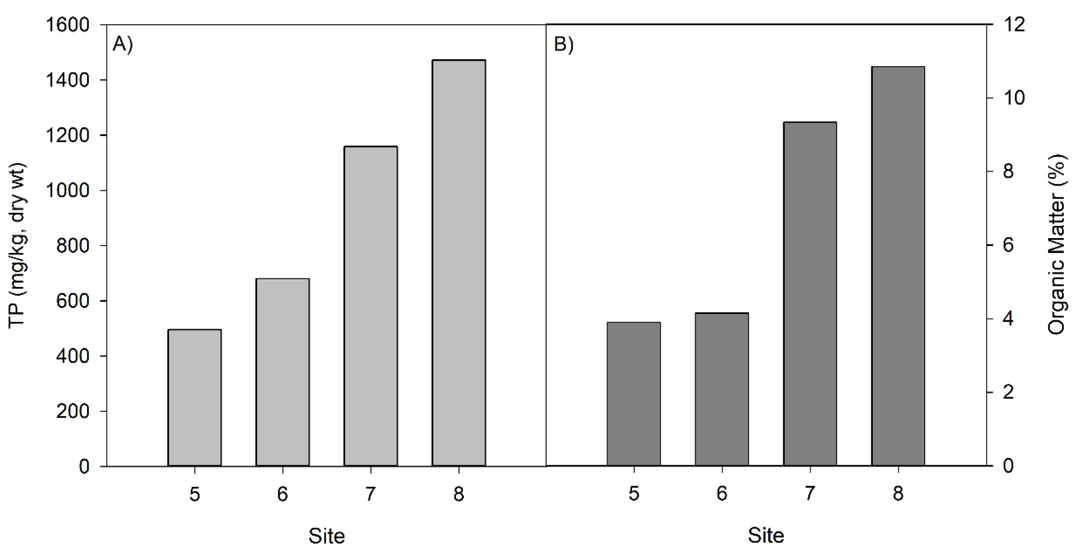

Figure 14. Middle Macatawa post-restoration wetland core surface sediment TP (A) and organic matter (B).

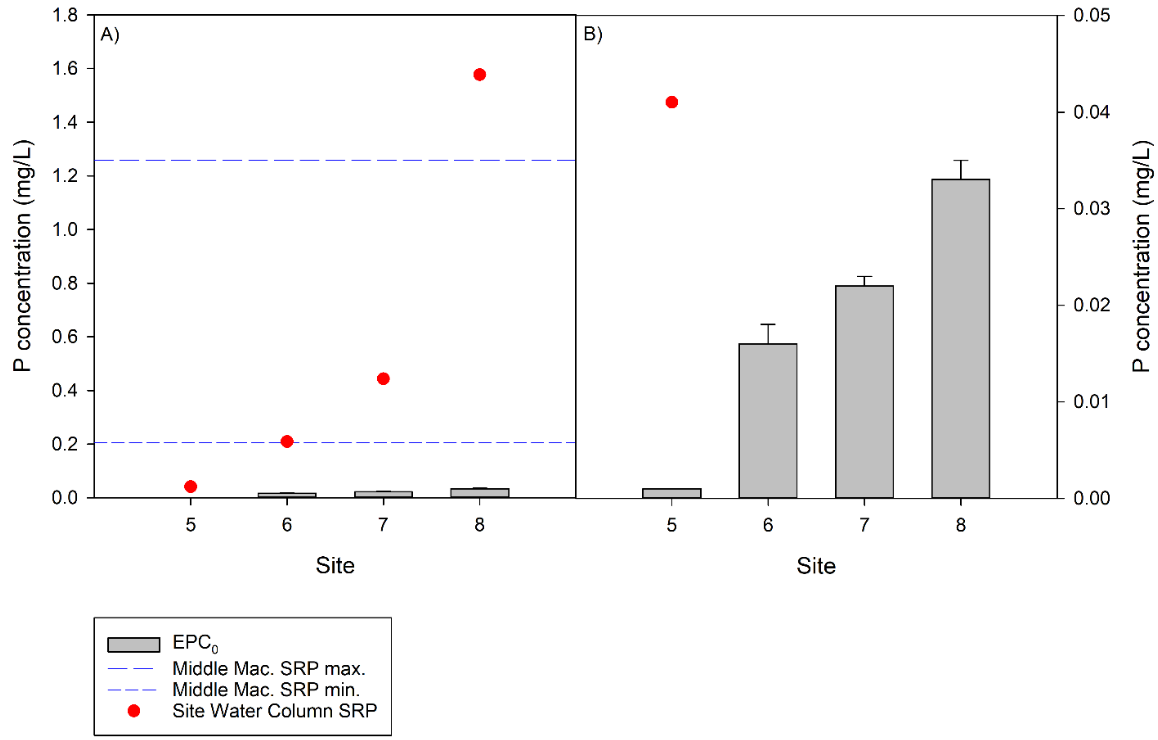

Figure 15. Middle Macatawa post-restoration wetland mean (1 SD) equilibrium P concentrations (EPC_0) in cores (n=3) and water column SRP concentrations at coring sites (dots). Dashed reference lines represent maximum and minimum (1.258 and 0.206 mg/L) observed SRP concentrations collected at the Macatawa Upstream tributary monitoring site from all post-restoration stormflow sampling (2016-2019). Panel B provides more detail of panel A; note the change of y-axis scale and that water column SRP from sites 6-8 and the minimum Macatawa Upstream stormflow SRP are above the displayed scale.

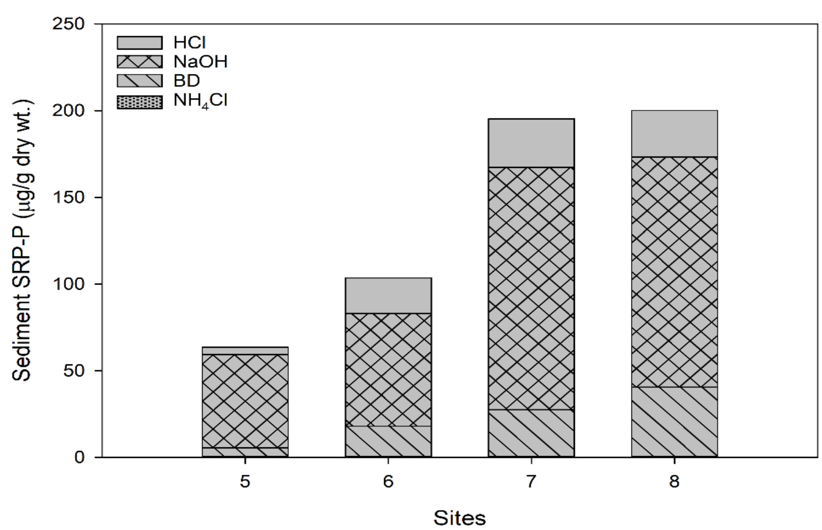

Figure 16. Middle Macatawa post-restoration wetland surface (top 10 cm) sediment P fractions shown as stacked columns by site. Note that SRP concentrations from NH_4Cl (loosely sorbed P) are too small to appear in this figure.

3.2 Wetland Restoration: Haworth Property

3.2.1 Sampling Year 2019

Baseflow:

Water quality trends at Haworth were similar to values seen in the Middle Macatawa during the shortened 2019 sampling year. Mean temperature (4.46-4.55°C) and DO (14.28-14.76 mg/L) were both slightly higher at the upstream site (Table 12). Mean specific conductivity, TDS, and turbidity all increased between the upstream and downstream Haworth sites (Table 12). Mean SRP and TP each increased 2 µg/L at the downstream site, but this difference was not large enough to be statistically significant (Tables 13, 14; Figs. 17A, C). Mean NO_3^- slightly decreased at the downstream site while NH_3 and TKN slightly increased, but these trends were also not significant (Tables 13, 14; Figs. 18A, C, E, 19).

Stormflow:

Contrary to other wetland sites, water temperature slightly increased at the downstream site and had a corresponding decrease in DO during stormflow (Table 12). Overall values for specific conductivity and TDS each decreased by ~50% during stormflow compared to baseflow, but the relative difference between upstream and downstream sites in each flow regime didn't change (Table 12). Turbidity increased by an order of magnitude during stormflow (Table 12).

SRP and TP concentrations increased by 3-fold and 5-fold, respectively, during stormflow, ranging 64-77 µg SRP/L and 196-213 ug TP/L and exceeding the proposed Lake Macatawa total maximum daily load (TMDL) of 50 µg/L for TP (Table 13, Fig. 17). NH_3 and TKN each increased during stormflow, while NO_3^- decreased (Table 13, Fig. 17).

Hydrographs:

The pressure-discharge relationship produced good fits for models at the North Upstream site ($y=0.025x^2+0.0898x-0.0446$; $R^2=0.985$; Fig. 20) and the North Downstream site ($y=0.0239x^2-0.1485x+0.3166$; $R^2=0.982$; Fig. 21). Multiple storm and high flow events on given dates correspond between the two figures; although some variation is expected due to differences between sites, slightly higher discharge values at the downstream site may be explained by the larger drainage area for surface runoff compared to the upstream site (Figs. 20, 21).

3.2.2 Pre- vs. Post-Restoration Comparison

Baseflow:

Overall patterns of pre- and post-restoration water quality at the Haworth wetland tributary sites in 2019 followed similar seasonal trends seen at the Middle Macatawa sites. Mean temperatures declined post-restoration but to a lesser degree at the downstream site (Table 15). Dissolved oxygen declined post-restoration at the upstream site but increased slightly at the downstream site (Table 15). Conductivity declined following restoration at both sites, while TDS and turbidity also declined post-restoration, but at both upstream and downstream locations (Table 15). This is not necessarily surprising, as wetland restoration is not expected to have a noticeable water quality benefit at baseflow when most flow bypasses the wetland.

Mean nutrient concentrations were slightly higher following restoration for all analytes at both the upstream and downstream sites (Table 16, Fig. 22). Unlike at Middle Mac, it does not appear that agricultural management practices are improving water quality at baseflow in the Haworth sub-basin (North Branch).

Stormflow:

Overall, mean temperature and turbidity values increased during stormflow relative to baseflow, whereas mean DO, specific conductivity, and TDS declined during stormflow (Table 15). Stormflow increased SRP and TP concentrations by up to an order of magnitude, and TKN values 3-fold (Table 16). Nitrate and ammonia values were generally similar during stormflow and baseflow periods (Table 16).

Following restoration, the relative declines in temperature, conductivity, and TDS were of the same magnitude at both the upstream and downstream locations (Table 15). This was also the case for the increase in DO at both the upstream and downstream locations; restoration had little effect on turbidity values at either location (Table 15). Nutrient responses to restoration were mixed during stormflow: NO_3^- and NH_3 concentrations increased to a similar degree post-restoration at both the upstream and downstream sites, whereas SRP, TP, and TKN values either declined or increased modestly at the upstream site but increased substantially at the downstream site, indicating the wetland was likely releasing nutrients, not retaining them.

Table 12. Mean (1 SD) values of selected water quality parameters at the Haworth wetland restoration site for the 2019 period of record (Dec. 2018 – Apr. 2019). Data are divided into baseflow and stormflow conditions. Stormflow water quality data sampling occurred 3/15/2019. SpCond= specific conductivity; NA = not applicable.

Flow	Site	n	Temp. (°C)	DO (mg/L)	SpCond (µS/cm)	TDS (g/L)	Turbidity (NTU)
Base	North Up	5	4.55 (6.96)	14.76 (2.90)	652 (106)	0.424 (0.069)	3.7 (1.8)
	North Down	5	4.46 (7.03)	14.28 (1.69)	684 (122)	0.444 (0.080)	5.3 (1.9)
Storm	North Up	1	3.45 (NA)	12.20 (NA)	326 (NA)	0.212 (NA)	53.4 (NA)
	North Down	1	4.89 (NA)	11.66 (NA)	354 (NA)	0.230 (NA)	56.3 (NA)

Table 13. Mean (1 SD) values of selected nutrient concentrations at the Haworth restoration site for the 2019 period of record (Dec. 2018 – Apr. 2019). Data are divided into baseflow and stormflow conditions. Stormflow water quality data sampling occurred 3/15/2019. NA = not applicable.

Flow	Site	n	SRP (µg/L)	TP (µg/L)	NO_3^- (mg/L)	NH_3 (mg/L)	TKN (mg/L)
Base	North Up	5	21 (17)	45 (25)	2.59 (0.82)	0.12 (0.14)	0.92 (0.13)
	North Down	5	22 (21)	47 (30)	2.47 (0.82)	0.14 (0.19)	1.00 (0.24)
Storm	North Up	1	77 (NA)	213 (NA)	0.99 (NA)	0.21 (NA)	1.36 (NA)
	North Down	1	64 (NA)	196 (NA)	0.83 (NA)	0.19 (NA)	1.42 (NA)

Table 14. Statistical analysis results of 2019 sampling at Haworth sites comparing upstream vs. downstream parameters at baseflow. Stormflow 2019 data were not statistically analyzed due to low sample size (n=1). Parameter column indicates water quality parameter and transformation used to meet assumptions of normality and variance. All data were analyzed using 2-tailed paired t-tests (T) and NH_3 data were log transformed prior to testing to meet assumptions of normality. All parameters were not significantly (NS) different.

Parameter	Test	p-value	Notes
SRP	T	0.932	NS
TP	T	0.930	NS
NO_3^-	T	0.826	NS
log NH_3	T	0.994	NS
TKN	T	0.493	NS
Turbidity	T	0.202	NS

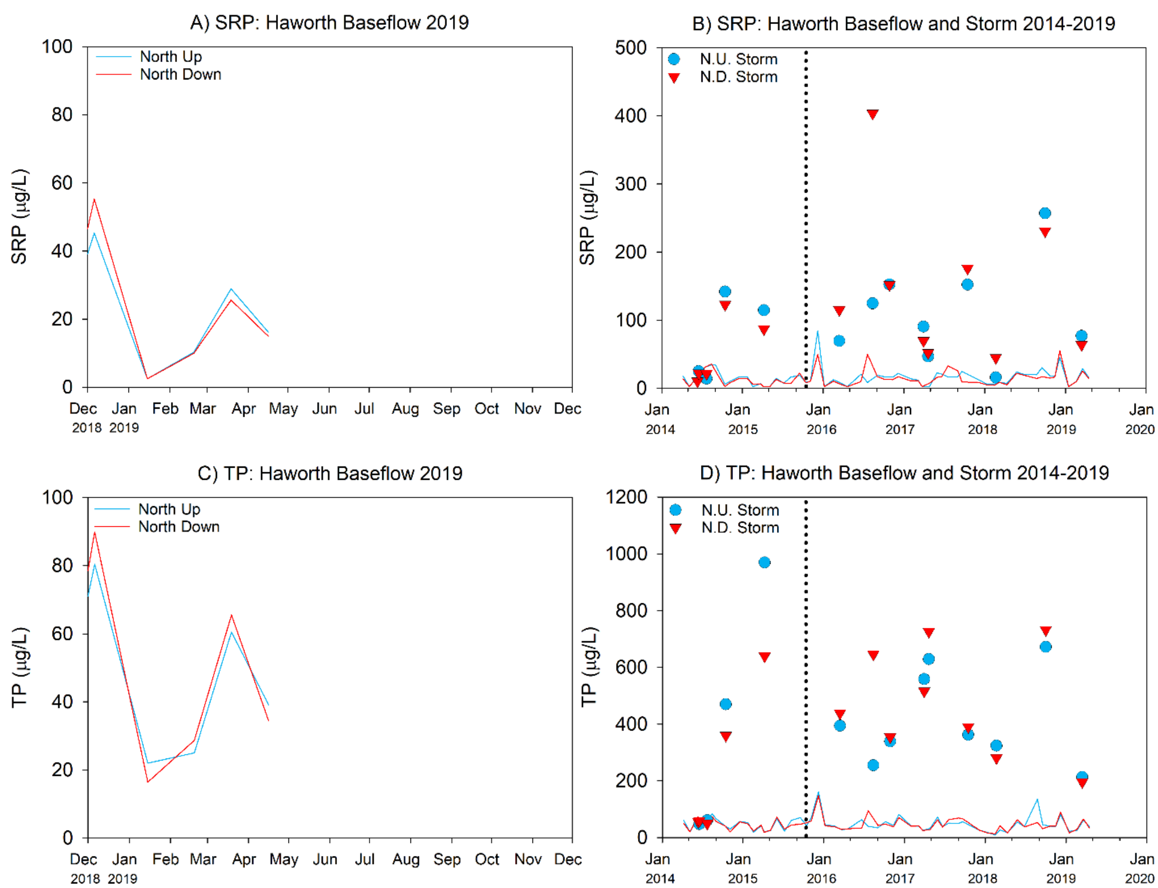

Figure 17. Soluble reactive phosphorus (SRP) (A, B) and total phosphorus (TP) (C, D) concentrations measured at Haworth wetland for 2019 (A, C) and total project history (B, D). Colored data lines in A and C magnify 2019 baseflow data shown in B and D, which allow us to include both baseflow and storm event concentrations in same graph; symbols represent storm events. Vertical dotted lines represent approximate completion date of wetland restoration construction. Note changes to scales of y-axes. Legend in A, C also applies to B, D.

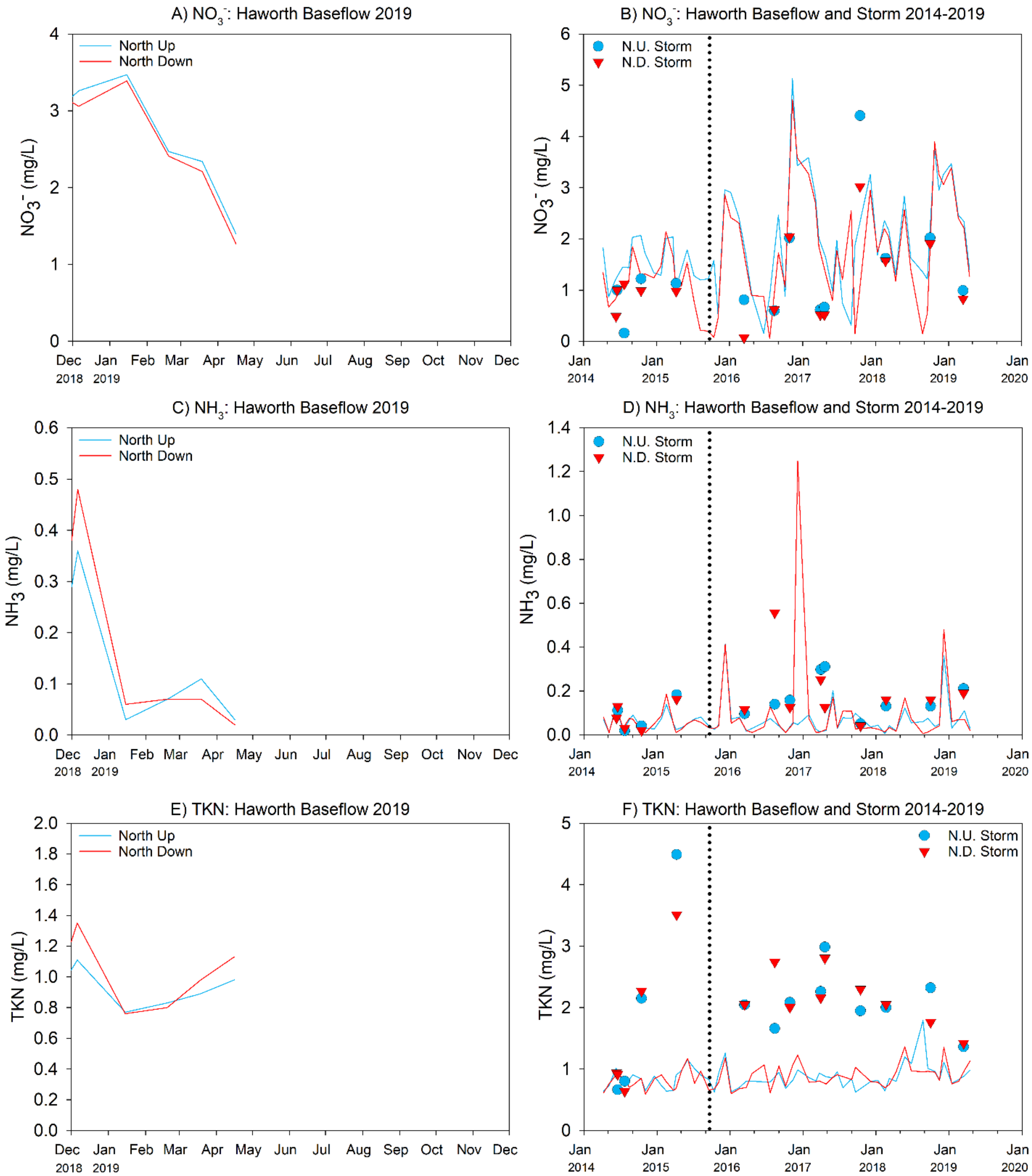

Figure 18. Nitrate (NO_3^-) (A, B), ammonia (NH_3) (C, D), and total Kjeldahl nitrogen (TKN) (E, F) concentrations measured at the Haworth wetland for 2019 (A, C, E) and total project history (B, D, E). Colored data lines in A, C, E magnify 2019 baseflow data shown in B, D, F, which allow us to include both baseflow and storm event concentrations in same graph; symbols represent storm events. Vertical dotted lines represent approximate completion date of wetland restoration construction. Note changes to scales of y-axes; and that y-axis scales are lower than at Middle Macatawa sites (Fig. 7). Legend in A, C, E also applies to B, D, F.

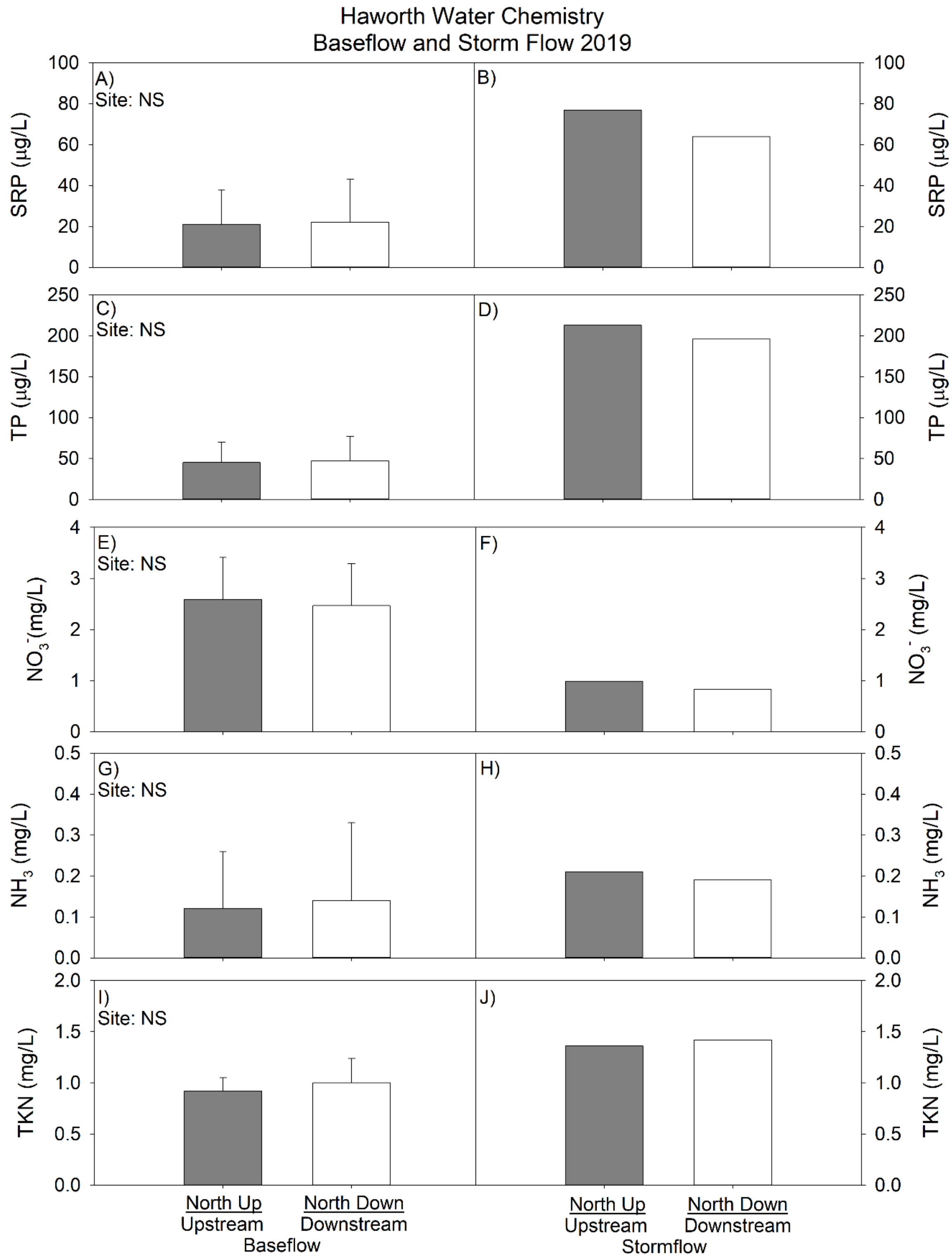

Figure 19. Mean (1 SD) water quality values at Haworth sites for 2019 sampling year (Dec. 2018 – Apr. 2019) at baseflow (A, C, E, G, I) and stormflow (B, D, F, H, J). Note that scales change in y-axes among water quality parameters.

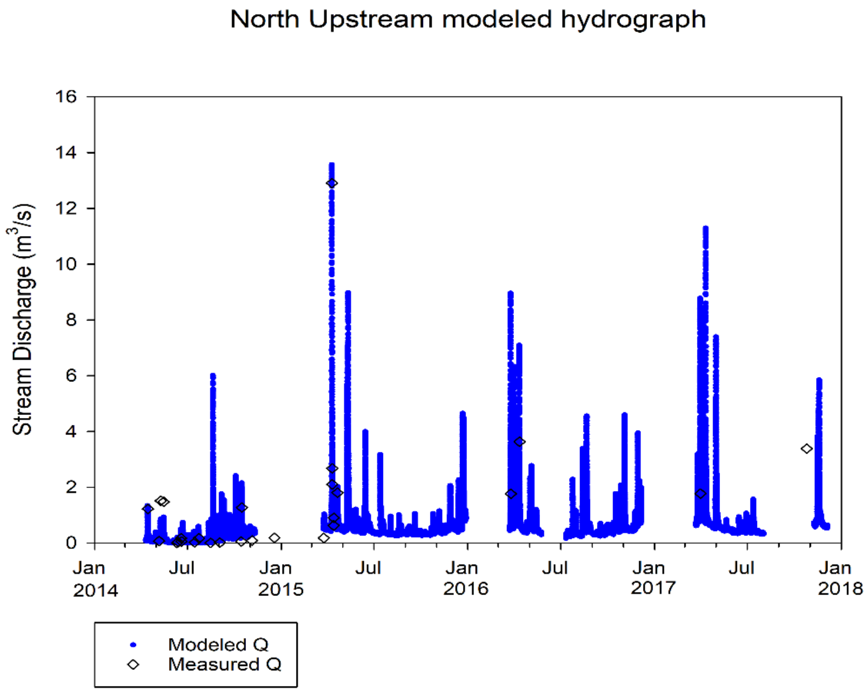

Figure 20. North Upstream site hydrologic modeling.

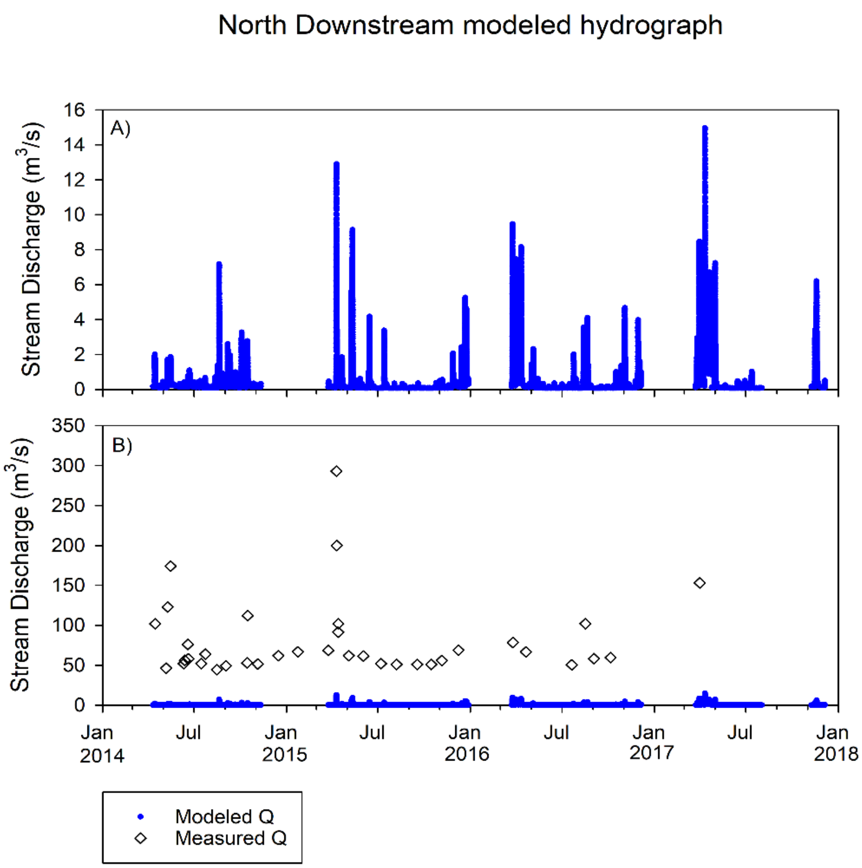

Figure 21. North Downstream site hydrologic modeling. Panel A displays detail of modeled stream discharge (Q) and panel B displays full range of in situ measured Q. Note the different y-axis scales.

Table 15. Grand mean (1 SD) values of selected water quality parameters at the Haworth wetland restoration site in pre- and post-restoration sampling periods. Grand mean cells have two rows per column: data in the top row represent pre-restoration sampling (Apr. 2014 – Sept. 2015) and data in bottom row represent post-restoration sampling (Oct. 2015 –Apr. 2019). Data are divided into baseflow and stormflow conditions. Note that the number of observations (n) changes between flow regimes and restoration periods. Date of storm sampling events: Pre - 6/12/14; 6/18/14; 7/23/14; 10/15/14; 4/9/15. Post - 3/14/16; 8/12/16; 10/27/16; 3/30/17; 4/20/17; 10/15/17; 2/20/18; 3/15/2019.

Flow	Site	Period	n	Temp. (°C)	DO (mg/L)	SpCond (µS/cm)	TDS (g/L)	Turbidity (NTU)
Base	North Up	Pre	18	12.38 (7.11)	11.02 (3.89)	843 (144)	0.548 (0.093)	6.4 (3.6)
		Post	38	9.95 (7.88)	10.74 (3.13)	773 (183)	0.502 (0.119)	5.9 (5.5)
	North Down	Pre	18	11.93 (6.96)	10.32 (3.36)	844 (194)	0.549 (0.126)	5.6 (3.0)
		Post	38	10.04 (7.83)	10.45 (3.09)	808 (167)	0.525 (0.109)	5.3 (4.9)
Storm	North Up	Pre	3	13.80 (5.92)	7.77 (2.29)	432 (283)	0.281 (0.184)	200.7 (223.6)
		Post	8	10.28 (6.85)	10.03 (2.70)	375 (94)	0.244 (0.061)	201.2 (301.0)
	North Down	Pre	3	13.80 (6.06)	7.84 (2.32)	478 (150)	0.310 (0.098)	143.6 (146.0)
		Post	8	11.60 (6.81)	9.51 (2.96)	412 (90)	0.268 (0.058)	146.3 (109.5)

Table 16. Grand mean (1 SD) values of selected nutrient concentrations at the Haworth restoration site in pre- and post-restoration sampling periods. Grand mean cells have two rows per column: data in the top row represent pre-restoration sampling (Apr. 2014 – Sept. 2015) and data in bottom row represent post-restoration sampling (Oct. 2015 – Apr. 2019). Data are divided into baseflow and stormflow conditions. Note that the number of observations (n) changes between flow regimes and restoration periods. See Table 15 for dates of storm sampling events.

Flow	Site	Period	n	SRP (µg/L)	TP (µg/L)	NO_3^- (mg/L)	NH_3 (mg/L)	TKN (mg/L)
Base	North Up	Pre	18	14 (11)	48 (21)	1.51 (0.38)	0.06 (0.04)	0.84 (0.15)
		Post	38	17 (14)	48 (29)	2.09 (1.10)	0.07 (0.08)	0.88 (0.21)
	North Down	Pre	18	13 (10)	44 (19)	1.17 (0.50)	0.06 (0.04)	0.80 (0.15)
		Post	38	16 (13)	47 (26)	1.91 (1.16)	0.10 (0.22)	0.89 (0.19)
Storm	North Up	Pre	3	90 (67)	500 (455)	0.84 (0.59)	0.08 (0.09)	2.48 (1.87)
		Post	9	110 (72)	416 (164)	1.53 (1.23)	0.17 (0.09)	2.07 (0.45)
	North Down	Pre	5	53 (49)	233 (263)	0.92 (0.24)	0.08 (0.06)	1.65 (1.22)
		Post	9	156 (119)	511 (173)	1.28 (1.01)	0.19 (0.16)	2.23 (0.37)

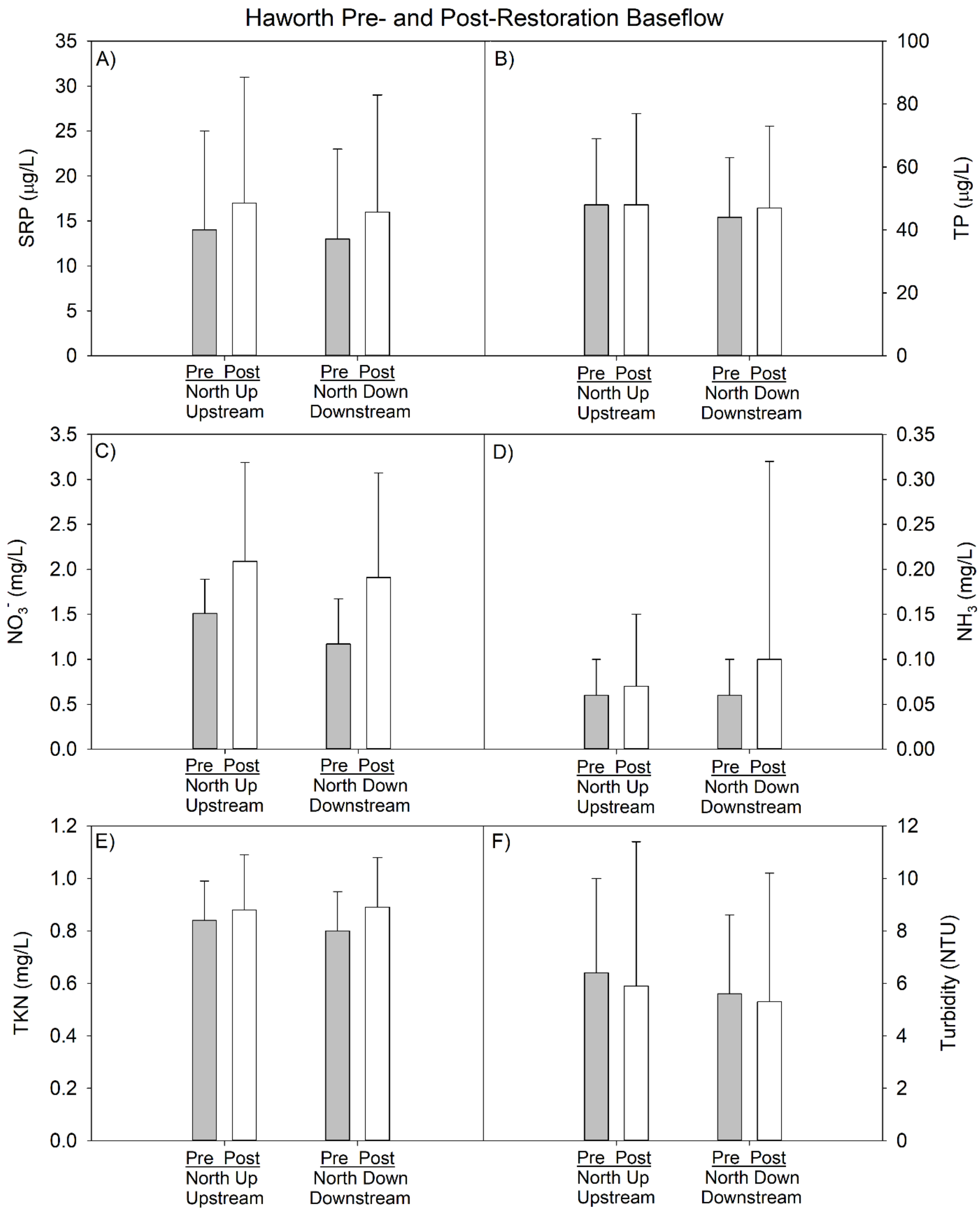

Figure 22. Haworth pre- and post-restoration water chemistry comparison at baseflow as of 2019 sampling year (Dec. 2018 – Apr. 2019). Error bars represent 1 SD.

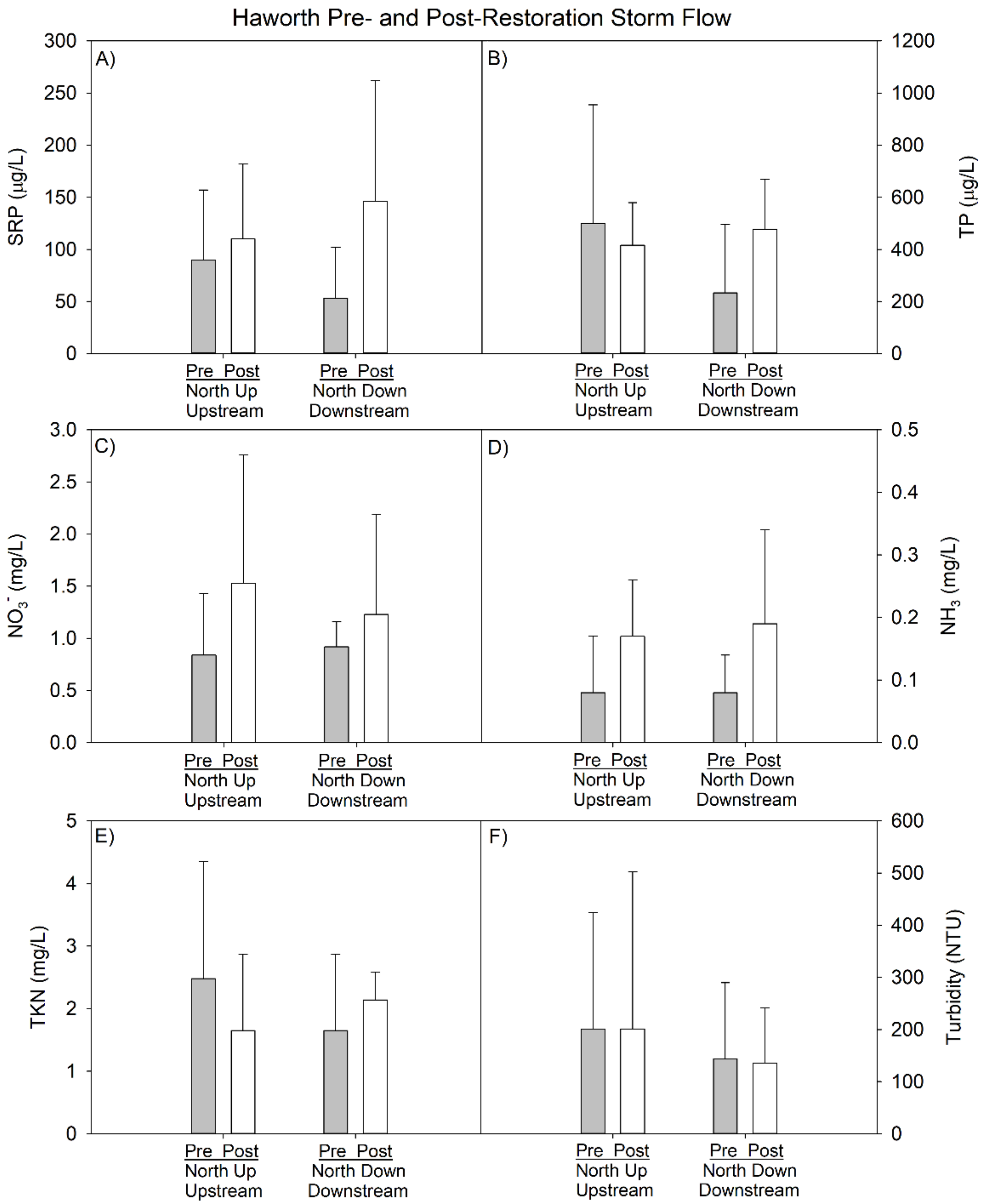

Figure 23. Haworth pre- and post-restoration water chemistry comparison at stormflow as of 2019 sampling year (Dec. 2018 – Apr. 2019). Error bars represent 1 SD.

3.2.3 Wetland Sediment Analyses

Overlying water depth at Haworth wetland coring sites was insufficient for sampling for water quality and chemistry analyses. Anecdotally, field technicians noted that sites 2 & 3 were relatively open and grass covered, site 1 had denser and taller grasses, and site 4 was dominated by cattails.

Surface (top 0-10 cm) sediment TP and organic matter content from Haworth cores generally had lower amounts than Middle Mac wetland cores. TP values ranged 241-372 mg/kg dry weight (Table 17, Fig. 24A) and organic matter ranged 3-8% among sites with the highest OM at site 4 likely due to thick cattail cover at this coring location (Table 17, Fig. 24B).

EPC_0 ranged 0.007-0.062 mg/L among all site cores, averaging 0.024 mg/L for the entire Haworth wetland (Table 17, Fig. 25). No overlying water data was available for EPC_0 comparison; instead, sites were compared to the observed SRP concentrations at the upstream site of the North Branch of the Macatawa River (i.e., the water that will flood the wetland during stormflow) during the current year's post-restoration monitoring during both baseflow and stormflow. EPC_0 at site 1 was more than twice the observed river SRP at baseflow and site 4 was just above the EPC_0 (Fig. 25). However, since the wetland is not designed to accept tributary water during baseflow, these two exceedances are likely not consequential. During stormflow conditions, tributary SRP concentrations exceed EPC_0 at all sites (Fig. 25), suggesting the sediments serve as P sink when flooded by river water.

Fractionation results of Haworth wetland sediments varied somewhat from the Middle Macatawa wetland sediments. While the NaOH-P fraction (Fe- and Al-bound P) dominated at all the Middle Mac sites, it was the largest fraction at only 2 of the 4 sites at Haworth (Fig. 26). At the other two sites, the HCl-P fraction was either dominant (site 4) or equivalent (site 2) to the NaOH-P fraction; Fig. 26). The loosely sorbed NH_4Cl-P fraction was present only in minimal quantities at all Haworth sites (0.01-0.03 µg/g; Fig. 26).

Based on these data, the Haworth sediments appear to capable of assimilating P and holding it in relatively stable fractions. Only site 4, with a low P sorption maximum, may act as a P source. Sediment core incubations would be able to assess whether there are differences in sediment P retention capacity among regions in the Haworth wetland.

Table 17. Haworth wetland site and mean (1 SD) sediment characteristics from cores collected 10/15/2019. All sediment is homogenized from near surface depth (0-10 cm). TP = total phosphorus, OM = organic matter, EPC_0 = equilibrium phosphorus concentration, S_{max} = phosphorus sorption maximum.

Site	Standing Water (cm)	Core Replicate	Sediment TP (dry, mg/kg)	Sediment OM (%)	EPC_0 (mg/L)	S_{max} (mg/kg)
1	0.5	A	272	4%	0.062	655.15
		B			0.049	706.74
		C			0.054	765.84
2	1.5-5	A	371	3%	0.008	655.32
		B			0.007	677.29
		C			0.009	659.91
3	0	A	241	3%	0.010	643.72
		B			0.011	546.25
		C			0.014	573.05
4	10	A	372	8%	0.028	127.30
		B			0.018	225.60
		C			0.018	209.26
Wetland Mean			314	4%	0.024	537.12
SD					0.021	238.49

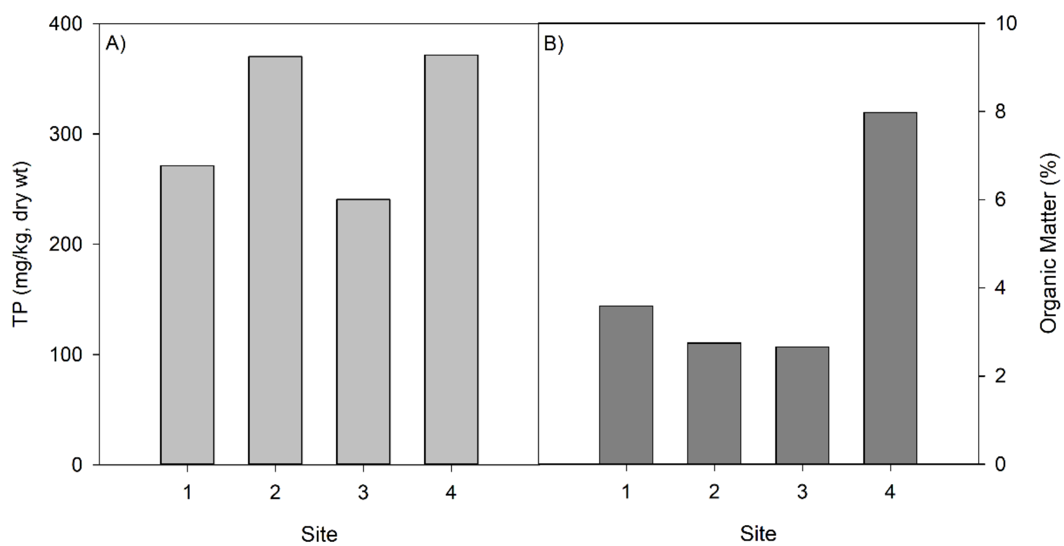

Figure 24. Haworth post-restoration wetland sediment core surface sediment TP (A) and organic matter (B).

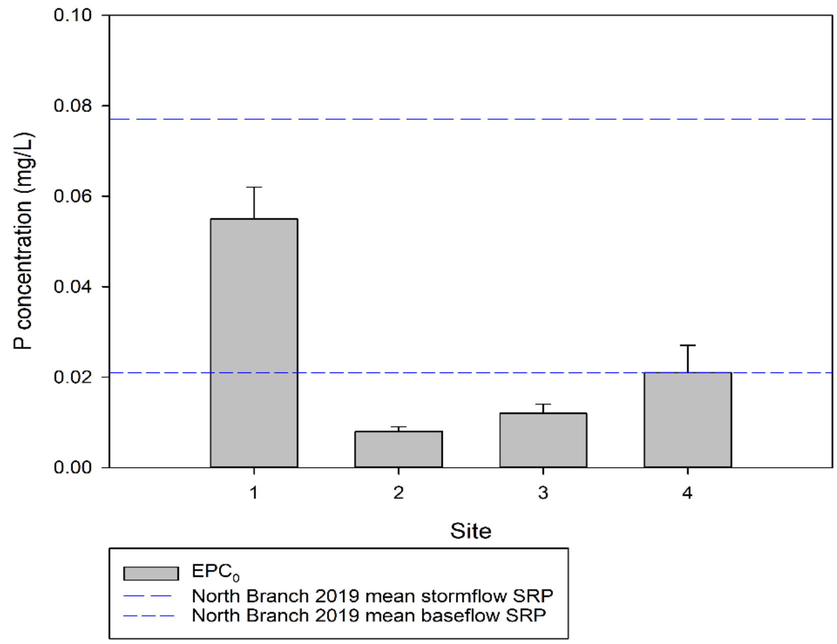

Figure 25. Haworth post-restoration wetland mean (1 SD) equilibrium P concentrations (EPC_0) in non-sterilized live cores (n=3). Dashed reference lines represent mean baseflow (0.021 mg/L) and stormflow (0.077 mg/L) SRP concentration collected at the North Branch Upstream tributary monitoring site in the current sampling year. Insufficient overlying water was present during sediment coring for collection and analysis to determine SRP concentration so we used concentrations from adjacent tributary.

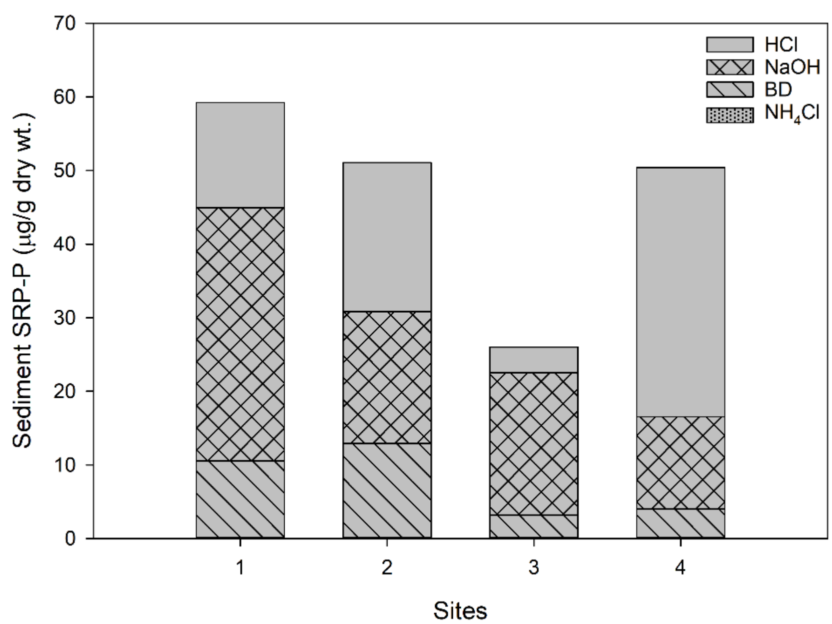

Figure 26. Haworth post-restoration wetland surface (top 10 cm) sediment P fractions shown as stacked columns by site. Note that SRP concentrations from NH_4Cl (loosely sorbed P) are too low to appear in this figure.

3.3 Lake Macatawa: Long-Term Monitoring

<u>3.3.1. Sampling Year 2019</u>

Lake Macatawa's water column was well-mixed in spring and fall 2019 as temperature and DO were generally consistent across sampling depths (Table 18). Summer 2019 stratification was evident in mean DO concentrations with surface waters averaging 9.16 mg/L and near bottom water at 5.70 mg/L. Low DO at site 1 (nearest to the Macatawa River mouth; 3.46 mg/L) in summer builds upon evidence of previous years of seasonal hypoxia in Lake Macatawa. Notably, the deepest Lake Macatawa sampling location (site 4; ~9.1 m) had higher DO (9.6 mg/L) compared to its nearest neighboring sites (2 and 5) at bottom depths (4.35 and 6.12 mg/L, respectively). Site 4's proximity to the Lake Michigan channel suggests intrusion and mixing of DO-rich cold water from Lake Michigan into Lake Macatawa may be occurring, as has been modeled in another nearby drowned river mouth, Muskegon Lake (Liu et al. 2018). A trend analysis (LOWESS) of summer DO concentrations at bottom depths shows what appears to be an increase over time, although the fit (R^2 = 0.29) is poor.

Lake-wide seasonal mean concentrations of SRP gradually increased throughout the spring to fall 2019 monitoring period, while mean TP and NO_3^- started high in spring before decreasing in summer and recovering in fall, similar to trends of previous years (Table 19, Figs. 27, 28, 29). When 2019 SRP concentrations were highest in fall 2019, concentrations showed a spatial trend of sites farther away from Lake Michigan (sites 1, 2, 3) having the highest SRP concentrations (Fig. 27). Site TP concentrations exceeded TMDL recommendations (50 µg/L) in 2019 in each season at all surface depths (Figs. 27C, D). Mean NO_3^- concentrations bottomed out in summer, while both NH_3 and TKN gradually decreased from spring to fall (Table 19, Fig. 28). Across all measured N species and depths, higher N concentrations were generally reported at Site 1 than at other lake sites (Table 19, Fig. 29).

The chlorophyll data at both near surface and near bottom generally declined over time (Fig. 27). When compared to the nutrient data, several patterns emerge: 1) the high chlorophyll levels in spring are strongly and inversely correlated to SRP, suggesting that the bioavailable P is being actively taken up by the phytoplankton at the start of the growing season, and that P is likely the limiting nutrient for algal growth at this time of year; 2) However, by summer, nitrate (and to a lesser degree, ammonia) levels decline (Table 20), while chlorophyll is still relatively high, indicating N is now the primary limiting nutrient although the relatively low SRP concentrations suggest there may be co-limitation of algal growth as shown by Steinman et al. (2016) in Pine Creek Bay; and 3) By fall, chlorophyll levels drop and the SRP and nitrate levels rebound, as there is reduced algal demand for the nutrients. **These patterns are similar to what we observed in 2018 but are not consistent over earlier time periods (Figs. 28 and 29), which again emphasizes the importance of not assuming short-term observations are predictive of long-term results.**

Although ELISA testing reported varying microcystin presence across sampling seasons, sites, and depths, concentrations remained degrees of magnitude lower than advisory levels (EPA: 2 µg/L for recreational use) lake-wide in 2019 (data now shown). Microcystin was found at all Spring 2019 samples at <0.05 µg/L, improving slightly from Spring 2018 (<0.06 µg/L), although no microcystin was detected in Spring 2017. In summer 2019, microcystin was detected only at sites 1 Bottom and 3 Top (0.01 and 0.06 µg/L, respectively). Fall 2019 continued a decreasing concentration trend seen in previous years, with all sites <0.03 µg/L.

Table 18. Lake-wide means (1 SD) of select general water quality parameters recorded during 2019 monitoring year. Within 2019, “n” is the number of lake sites composing the seasonal mean at each depth. Data are shaded for readability. Dates of sampling events: 5/8/2019; 7/17/2019; 9/25/2019.

Season	Depth	n	Temp. (°C)	DO (mg/L)	SpCond (µS/cm)	TDS (g/L)	Turbidity (NTU)
Spring	Top	5	14.20 (0.67)	14.69 (1.18)	508 (15)	0.330 (0.010)	19.3 (11.7)
	Middle	5	14.04 (0.78)	15.01 (0.87)	515 (23)	0.335 (0.015)	14.4 (1.9)
	Bottom	5	13.15 (1.37)	12.48 (3.41)	542 (113)	0.352 (0.073)	22.2 (6.1)
Summer	Top	5	24.46 (0.66)	9.16 (2.24)	514 (69)	0.335 (0.044)	10.0 (1.5)
	Middle	5	23.15 (2.31)	6.98 (1.59)	501 (78)	0.326 (0.051)	10.8 (1.6)
	Bottom	5	18.42 (5.23)	5.70 (2.39)	411 (74)	0.267 (0.048)	14.8 (6.2)
Fall	Top	5	10.49 (0.09)	10.46 (0.47)	526 (64)	0.341 (0.042)	12.8 (2.8)
	Middle	5	10.43 (0.07)	10.20 (0.49)	527 (67)	0.343 (0.044)	13.4 (3.0)
	Bottom	5	10.44 (0.14)	9.84 (0.76)	541 (91)	0.352 (0.059)	16.4 (5.7)

Table 19. Lake-wide means (1 SD) of phosphorus (soluble reactive phosphorus [SRP] and total phosphorus [TP]), nitrogen (nitrate [NO_3^-], ammonia [NH_3] and total Kendal nitrogen [TKN]), laboratory extracted chlorophyll *a* (chl *a*), and Secchi disk depths measured during 2019 monitoring year. Within 2019, “n” is the number of lake sites composing the seasonal mean at each depth. Data are shaded for readability. See Table 18 for dates of sampling events.

Season	Depth	n	SRP (µg/L)	TP (µg/L)	NO_3^- (mg/L)	NH_3 (mg/L)	TKN (mg/L)	ext. Chl *a* (µg/L)	Secchi Depth (m)
Spring	Top	5	3 (2)	97 (16)	1.33 (0.24)	0.03 (0.02)	1.90 (0.40)	89 (20)	0.6 (0.4)
	Bottom	5	3 (0)	95 (12)	1.36 (0.28)	0.32 (0.61)	2.03 (0.73)	67 (18)	
Summer	Top	5	7 (3)	73 (13)	0.17 (0.09)	0.27 (0.28)	1.44 (0.27)	66 (20)	0.7 (0.0)
	Bottom	5	8 (1)	84 (33)	0.23 (0.07)	0.34 (0.29)	1.30 (0.44)	46 (11)	
Fall	Top	5	16 (9)	81 (21)	1.59 (0.37)	0.14 (0.16)	1.19 (0.36)	30 (8)	0.5 (0.0)
	Bottom	5	18 (8)	86 (30)	1.62 (0.42)	0.16 (0.18)	1.32 (0.49)	27 (12)	

Table 20. Lake-wide grand means (1 SD) of phosphorus concentrations (soluble reactive phosphorus [SRP] and total phosphorus [TP]), laboratory extracted chlorophyll *a* (chl *a*), and Secchi disk depths measured during multi-year project history. Grand mean cells have two rows per cell: data in the top row represent pre-restoration sampling (Summer 2013 – Fall 2015) and data in bottom row represent post-restoration sampling (Spring 2016 – Fall 2019). Data are color coded for readability. ND = no data.

Season	Depth	Period	n	SRP (µg/L)	TP (µg/L)	NO_3^- (mg/L)	NH_3 (mg/L)	TKN (mg/L)	ext. Chl *a* (µg/L)	Secchi Depth (m)
Spring	Top	Pre	2	3 (0)	66 (4)	ND	ND	ND	25 (4)	0.6 (0.1)
		Post	4	17 (26)	119 (69)	1.65 (0.35)	0.28 (0.28)	1.72 (0.40)	54 (24)	0.6 (0.4)
	Bottom	Pre	2	3 (1)	98 (30)	ND	ND	ND	24 (3)	
		Post	4	18 (25)	125 (63)	1.62 (0.25)	0.45 (0.11)	1.74 (0.48)	44 (17)	
Summer	Top	Pre	3	6 (3)	110 (66)	ND	ND	ND	67 (39)	0.4 (0.1)
		Post	4	6 (3)	72 (25)	0.24 (0.06)	0.24 (0.10)	1.35 (0.12)	69 (29)	0.7 (0.0)
	Bottom	Pre	3	17 (18)	107 (49)	ND	ND	ND	32 (13)	
		Post	4	10 (4)	87 (26)	0.30 (0.10)	0.44 (0.12)	1.35 (0.09)	36 (8)	
Fall	Top	Pre	3	10 (12)	134 (23)	ND	ND	ND	63 (43)	0.4 (0.1)
		Post	4	9 (6)	77 (8)	1.16 (0.87)	0.42 (0.31)	1.46 (0.24)	57 (31)	0.5 (0.0)
	Bottom	Pre	3	11 (13)	158 (19)	ND	ND	ND	61 (35)	
		Post	4	11 (6)	85 (3)	1.28 (0.98)	0.43 (0.29)	1.45 (0.30)	47 (13)	

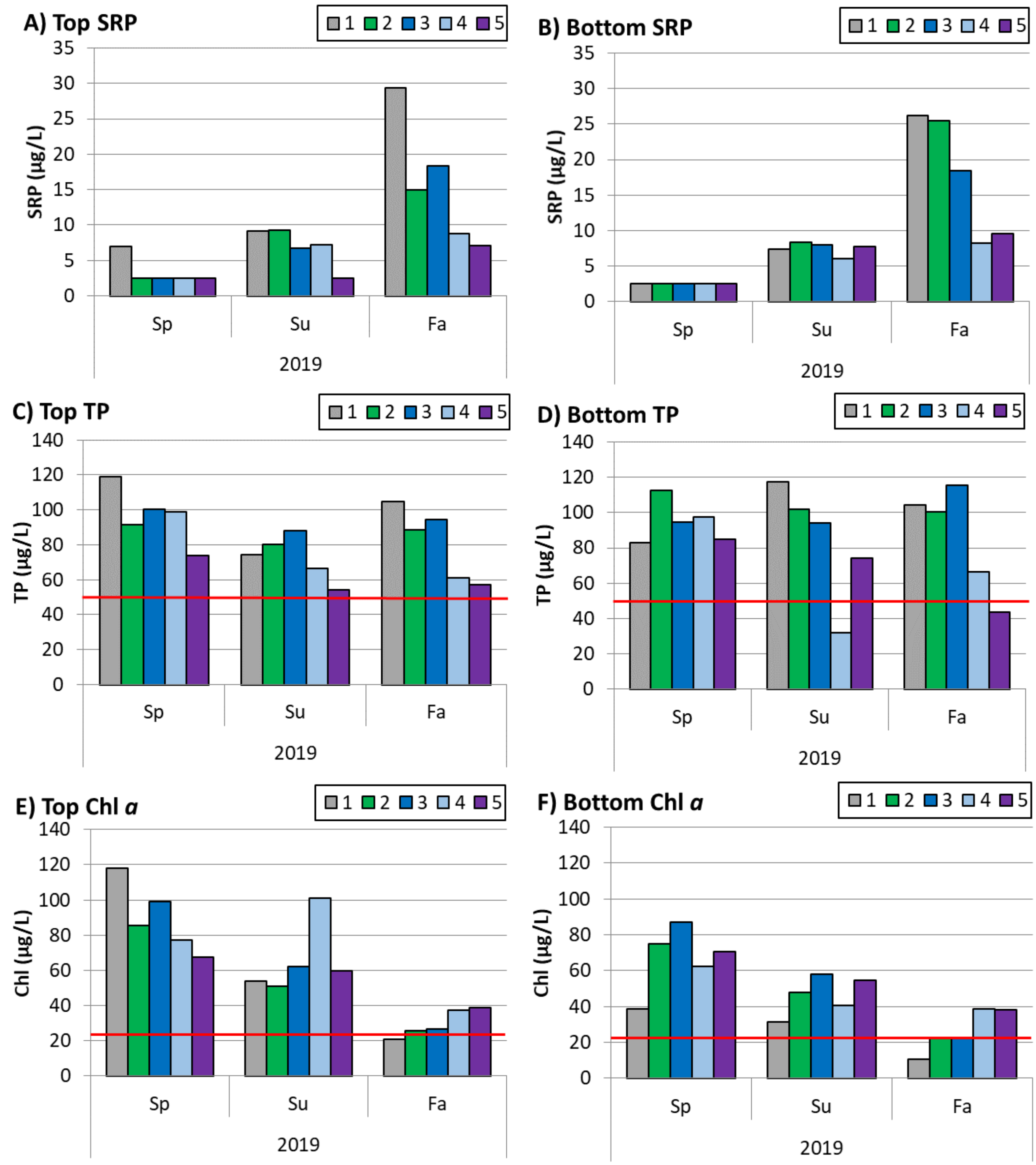

Figure 27. Soluble reactive phosphorus ([SRP]: A, B); total phosphorus ([TP]: C, D); and chlorophyll *a* ([chl *a*]: E, F) concentrations measured at the 5 monitoring stations in Lake Macatawa during 2019. The red horizontal lines on TP figures (C, D) indicate the interim total maximum daily load (TMDL) goal of 50 µg/L (Walterhouse 1999). The red horizontal lines on chl *a* figures (E, F) indicate the hypereutrophic boundary of 22 µg/L used by MDEQ for assessing chl *a* in Lake Macatawa (Holden 2014). Note scales change on y-axes.

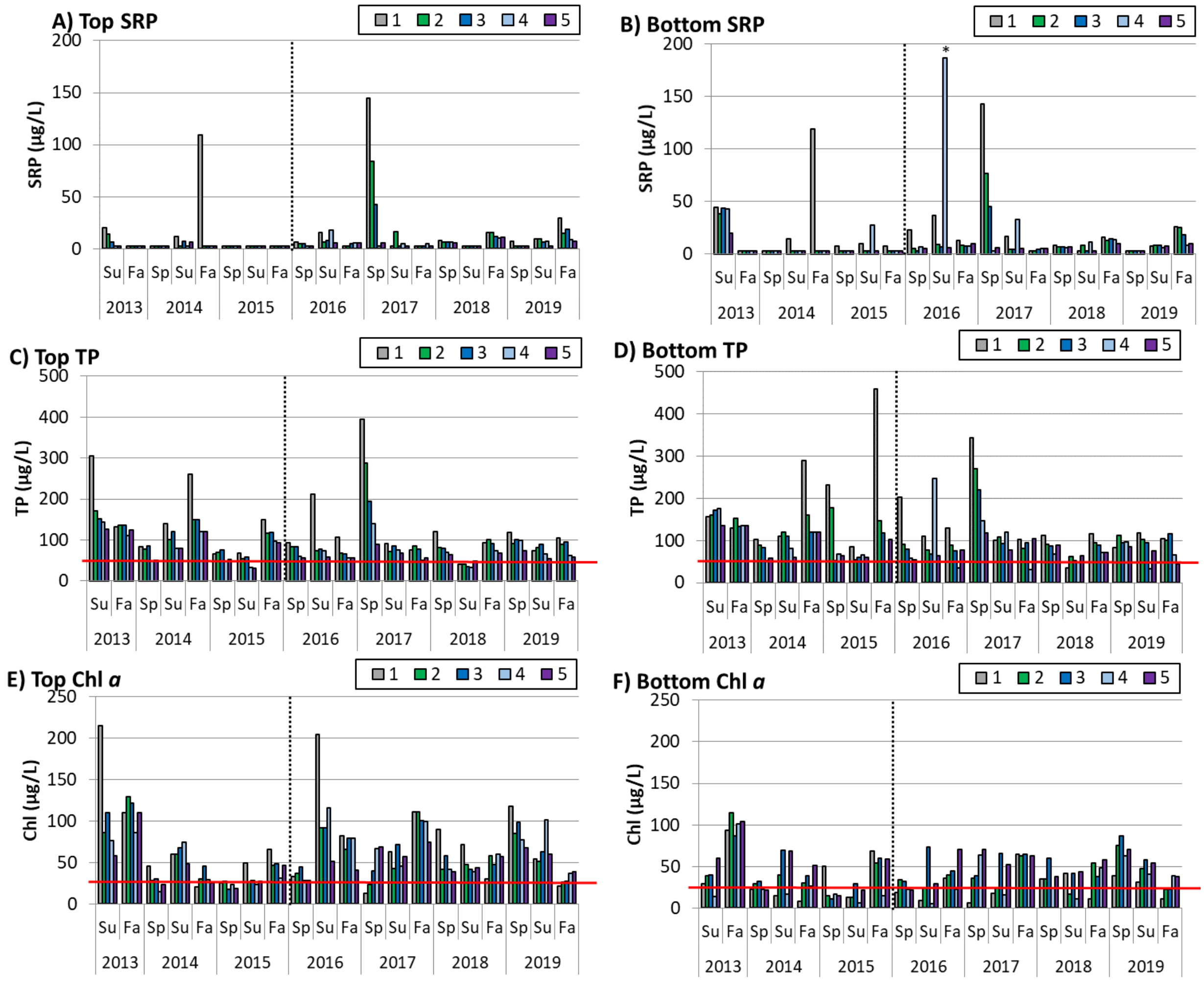

Figure 28. Soluble reactive phosphorus ([SRP]: A, B); total phosphorus ([TP]: C, D); and chlorophyll *a* ([chl *a*]*:* E, F) concentrations measured at the 5 monitoring stations in Lake Macatawa from 2013 through 2019. The red horizontal lines on TP figures (C, D) indicate the interim total daily maximum load (TMDL) goal of 50 µg/L (Walterhouse 1999). The red horizontal lines on chl *a* figures (E, F) indicate the hypereutrophic boundary of 22 µg/L used by MDEQ for assessing chl *a* in Lake Macatawa (Holden 2014). Summer 2016 site 4 SRP bottom depth sample (B, asterisked) is a likely outlier due to sediment disturbance. Note scales change on y-axes. Vertical dotted lines represent approximate restoration construction completion dates for Middle Macatawa and Haworth wetlands.

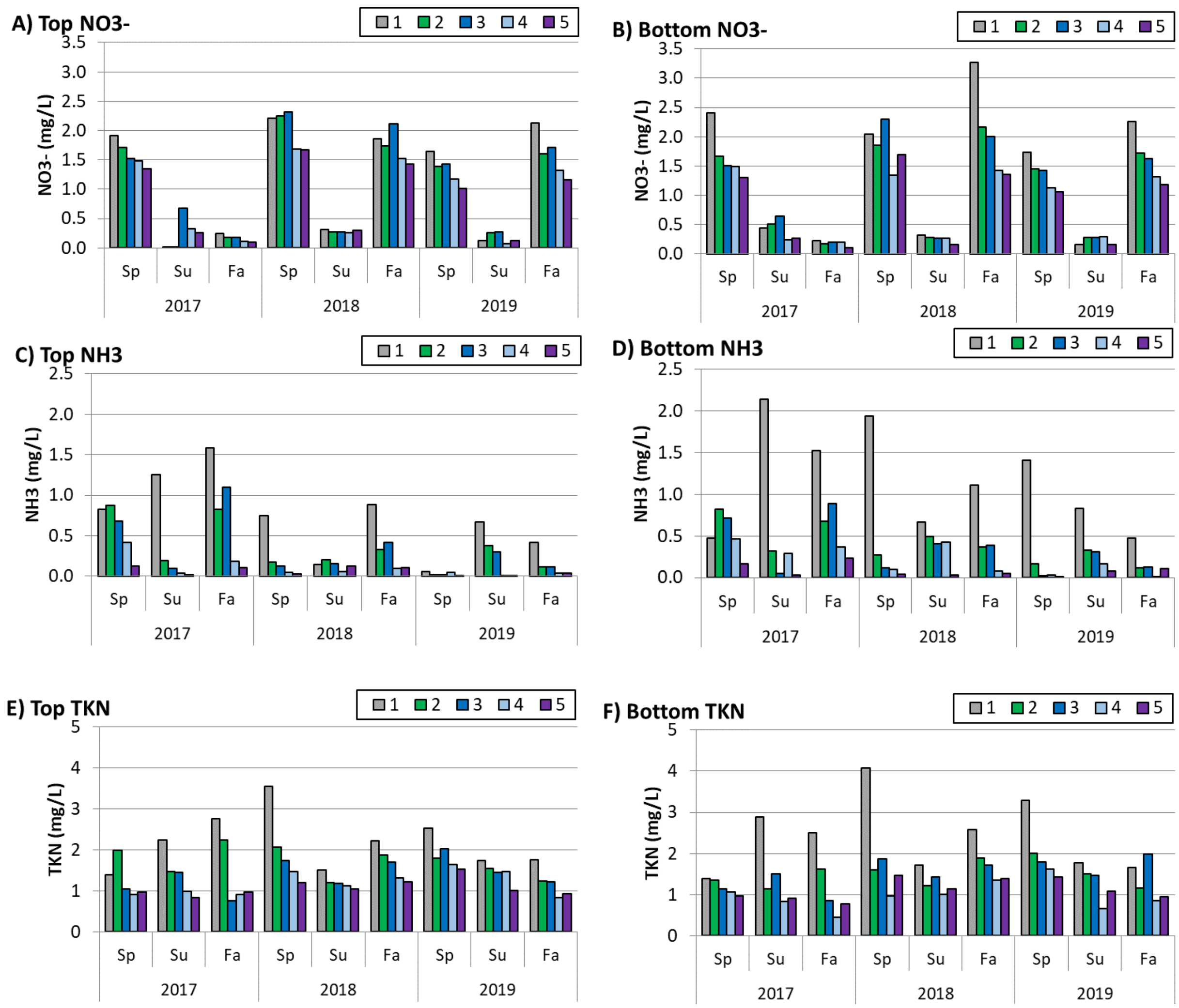

Figure 29. Nitrate ($[NO_3^-]$: A, B); ammonia ($[NH_3]$: C, D); and Total Kjeldahl Nitrogen ([TKN]: E, F) concentrations measured at the 5 monitoring stations in Lake Macatawa from 2017 through 2019. Note scales change on y-axes.

3.3.2 Pre- vs. Post Restoration Comparison

A qualitative comparison of pre- vs. post restoration water quality reveals two contrasting patterns. First, and perhaps of greatest management concern, is the increasing mean SRP concentrations at both top and bottom depths (Table 20). This comparison is based on very limited sample sizes so this increase should be viewed with caution, but extensive work in the western basin of Lake Erie, an area plagued by harmful algal blooms, has shown the spring runoff of SRP is an excellent predictor of how severe the toxic harmful algal blooms will be in the lake that summer (Michalak et al. 2013). Hence, when there is a wet spring, and dissolved P enters the agricultural drains of the Macatawa watershed either by surface runoff or tile drainage, we would expect increased chlorophyll concentrations, as observed in the spring and to a lesser degree in the summer chlorophyll data in Lake Macatawa (Table 20). While the precipitation cannot be controlled, and is actually expected to increase as a function of climate change (Kim et al. 2016), management of P runoff is controllable through detention practices, and we recommend particular attention be paid to controls in the spring season.

The second pattern is the overall reduction of TP in summer and fall seasons at both top and bottom depths (Table 20). For the most part, this does not appear to be related to reduced algal biomass (as seen in chlorophyll concentration), so presumably it is due to reduced sediment-attached TP. This may be due to improved management practices in the watershed; while reduced precipitation could also account for less sediment runoff, the precipitation data indicate an increase over time (see Dashboard; Appendix C).

3.4 Lake Macatawa Dashboard

It is well known that precipitation will influence lake condition because runoff carries nutrients and sediment, which ultimately reach the downstream receiving water bodies (Baker et al. 2019). Hence, when examining lake condition in a particular year, it makes sense to compare the lake health to the precipitation regime in that year. This has been shown in the western basin of Lake Erie, where heavy spring rains transported recently applied P fertilizer into the Maumee River, and eventually Lake Erie, triggering massive harmful algal blooms (Michalak et al. 2013). Hence, years with anomalously good or bad lake condition may be driven largely by the timing of fertilizer application and precipitation.

In Lake Macatawa, the relationship between lake TP and precipitation has not been clear-cut. Between 1972 and 2019, the relationship between precipitation and TP concentration in the lake was not statistically significant (Figs. 30; $R^2 = 0.001$; $p = 0.893$; Appendix C). For example, some years have very high TP concentrations and relatively low precipitation (e.g., 2000 and 2004), whereas other years have modest levels of TP and relatively high precipitation (e.g., 2017). Interestingly, the relationship between TP and precipitation is much improved since 2013 (Fig. 30; $R^2 = 0.364$; $p = 0.152$) but is still not statistically significant. This relationship is based on only 7 data points, so it should be viewed cautiously. We view these data as appropriate for screening purposes only, as the TP concentrations are single sampling events, which may miss pulses of high P concentrations after storm events. In addition, relatively light steady rainfall, which has a chance to soak into the ground, is less likely to result in surface runoff and erode sediment particles than intense, episodic events. These latter events are predicted to become more common with climate warming (Kunkel et al. 2013).

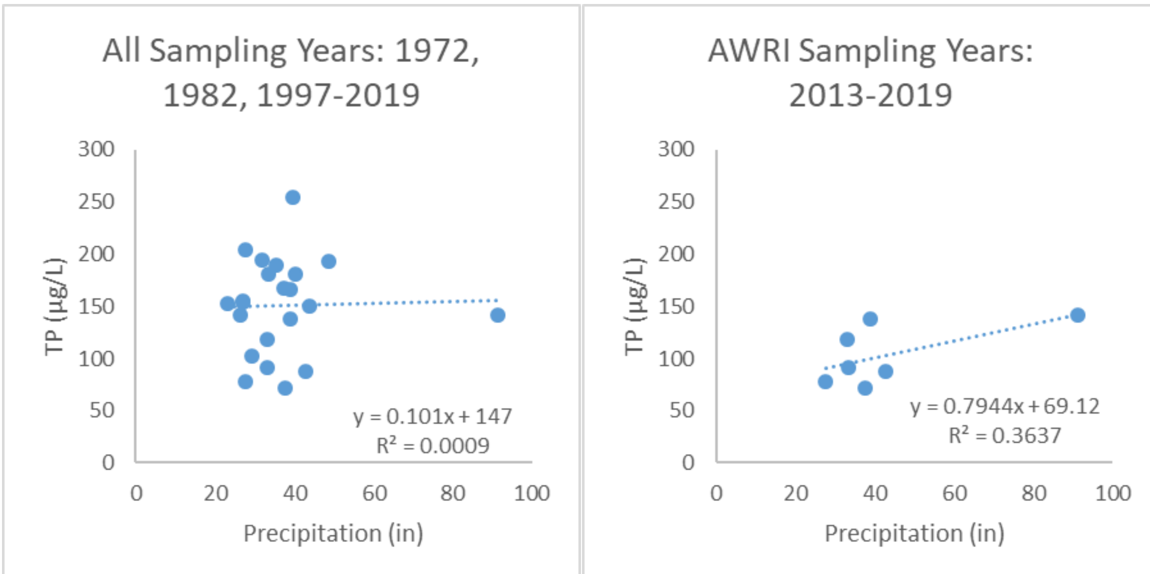

Figure 30. Linear regressions plotting annual precipitation vs. mean total phosphorus (TP) concentration in Lake Macatawa. Historical TP data sources include U.S. EPA (1972; STORET), Michigan Department of Environmental Quality (1982, 1997-2012; S. Holden, personal communication), and AWRI (since 2013). Precipitation data sources include the National Climatic Data Center/National Centers for Environmental Information (2005-2019; NOAA) and Weather Underground (1972-2004; The Weather Channel Company).

4. Summary

The results of the truncated 2019 tributary monitoring revealed that the main indicators of water quality in the watershed: SRP, TP, and turbidity showed some indications of improvement at baseflow in the Middle Macatawa but not the North Branch of the Macatawa River (Haworth site) when comparing the pre- vs. post-wetland restoration periods of record. More importantly, the absolute concentrations of SRP and TP remain at undesirable levels, especially in the Middle Mac. During stormflow, post-wetland restoration P concentrations are considerably higher than pre-wetland restoration, irrespective of site, while turbidity remains unchanged between the two periods. Absolute concentrations of SRP and TP are far in excess of desirable conditions, and clearly indicate more time, and likely additional management practices, are needed to improve Lake Macatawa water quality. We stopped collecting tributary water quality samples in April 2019, so future water quality trends will be based on Lake Macatawa and on a localized, project-specific basis (e.g., see below).

The iron slag filters have been effective at reducing P concentrations, but their optimal effectiveness is in regions with very high SRP concentrations in the tile drain effluent. Given their cost, it is difficult to justify their implementation in regions with "relatively" low SRP concentrations, so their impact in reducing P will be localized at high P “hot spots”. The current land management practices, including grassed waterways, winter cover crops, gypsum application, two-stage ditches, and restored wetlands, are all likely to improve water quality, but it will take time to see any benefits in Lake Macatawa given the amount of P stored in the watershed soils (legacy P; Jarvie et al. 2013, Sharpley et al. 2013) and the relatively modest footprint of some management practices. In addition, these restoration activities are still very recent and the natural environment varies over time, so detecting change against a variable reference state can be difficult.

We started our analyses of sediment P retention in the Middle Mac and Haworth restored wetlands in 2019. Our results suggest that the sediment in these wetlands presently have limited potential to adsorb additional P. Biotic uptake may end being a larger sink for P than abiotic adsorption. Of the P that is currently bound in the sediment, most of it is in either the Fe- and Al-bound P (Middle Mac) or both Fe-/Al-bound P and Ca-bound fractions (Haworth), indicating that P release from the sediment is either sensitive to DO concentrations (Fe-P) or relatively stable (Ca/Mg-P). Continued monitoring will allow us to determine the dominant sinks for P in the wetlands, allowing for optimal management decisions.

Lake data show the water quality in Lake Macatawa was relatively stable from the past few years, with a few caveats. Post-Project Clarity (PC) chlorophyll concentration was greater than prior to restoration in spring but was lower in fall; similarly, post-restoration TP was greater in spring but lower in summer and fall compared to pre-restoration values. **While phosphorus and chlorophyll reductions are certainly good news, the absolute concentrations of both indicators remain much greater than desirable.** All post-restoration mean TP values exceed the 50 µg/L TMDL, some by 2-3×, while mean chlorophyll values exceed the 22 µg/L threshold. And note that these thresholds, which were set for pragmatic, not ecological reasons, are much higher than what would truly be protective of Lake Macatawa (e.g., the thresholds for TP and chlorophyll *a* in Muskegon Lake, set for purposes of being delisted as a Great Lakes Area of Concern, are fully half of those for Lake Macatawa: 25 and 10 µg/L, respectively; Steinman et al. 2008). Hence, our data indicate that water quality in Lake Macatawa is still severely impaired. We expect to see year-to-year variation in these indicators, and it will take time to see overall trends. We caution once again that it can take decades for actions in the watershed to result in improvements in a lake.

As noted in last year’s report, high nitrate concentrations continue to be a concern in the lake. We have previously found that at least some algae in Lake Macatawa are co-limited by phosphorus and nitrogen (Steinman et al. 2016), and the summer drawdown of nitrate in the lake, presumably due to phytoplankton uptake, is another indicator that nitrogen is a critical element for algal growth in Lake Macatawa. We strongly recommend the agricultural community focus on both nitrogen (cf. Dinnes et al. 2002) and phosphorus controls.

Also noted last year, our work on a computational SWAT model for the Macatawa watershed has been delayed because the postdoctoral research associate working on this project took a job at Penn State University. We plan to begin SWAT work again in 2020 with the hiring of a geospatial ecologist at AWRI, and run scenarios that show the effectiveness of different management practices in different regions of the watershed.

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Appendices

Appendix A. Long-Term Fish Monitoring of Lake Macatawa: Results from Year 5.

Appendix B. Iron Slag Filters.

Appendix C. Lake Macatawa Dashboard

Appendix A.

Long-Term Fish Monitoring of Lake Macatawa: Results from Year 6 (2019)

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Introduction

This study was initiated to provide critical information on littoral fish populations that will be used to evaluate the performance of watershed restoration activities that are part of Project Clarity. In autumn 2014, we initiated a long-term monitoring effort of the littoral fish assemblage of Lake Macatawa to establish baseline ecological conditions and evaluate ecological change over time. Our fish sampling plan for Lake Macatawa is similar to our ongoing, long-term (since 2003) monitoring effort in Muskegon Lake (Bhagat and Ruetz 2011). By using the same monitoring protocols in each water body, Muskegon Lake can serve as a "control" to evaluate temporal changes in Lake Macatawa in an effort to assess how the lake is responding to watershed restoration activities.

Our primary objective in the sixth year (2019) of sampling was to continue to characterize the pre-restoration (baseline) littoral fish assemblage. We made preliminary comparisons with previous work in Muskegon Lake (see Bhagat and Ruetz 2011) as well as with six Lake Michigan drowned river mouths for which we have data (see Janetski and Ruetz 2015). However, the true value of this fish monitoring effort will come in future years as we examine how the littoral fish assemblage responds to restoration activities in the watershed.

Methods

Study sites.—Lake Macatawa is a drowned river mouth lake in Holland, Michigan that is located on the eastern shore of Lake Michigan in Ottawa County. Lake Macatawa has an area of 7.20 km^2, mean depth of 3.66 m, and maximum depth of 12.19 m (MDNR 2011). The shoreline has high residential and commercial development, and the watershed consists mainly of agricultural land (MDNR 2011). Fish sampling was conducted at four littoral sites in Lake

Macatawa that represented a gradient from the mouth of the Macatawa River to the connecting channel with Lake Michigan (Figure 1; Table 1). In 2016, much of the riparian vegetation was removed at site #2 for a construction project (Figure 1), which substantially changed littoral habitat.

Fish sampling.—At each study site, we sampled fish via fyke netting and boat electrofishing. Using both sampling gears should better characterize the littoral fish assemblage than either gear by itself because small-bodied fishes are better represented in fyke netting and large-bodied fishes are better represented in nighttime boat electrofishing (Ruetz et al. 2007). Fyke nets were set on 16 September 2019 during daylight hours (i.e., between 0900 and 1400) and fished for about 25.2 h (range = 23.8-26.2 h). Three fyke nets (4-mm mesh) were fished at each site; two fyke nets were set facing each other and parallel to the shoreline, whereas a third fyke net was set perpendicular to the shoreline following the protocol used by Bhagat and Ruetz (2011). A description of the design of the fyke nets is reported in Breen and Ruetz (2006). We conducted nighttime boat electrofishing at each site on 5 September 2019. A 10-min (pedal time) electrofishing transect was conducted parallel to the shoreline at each site with two people at the front of the boat to net fish. The electrofishing boat was equipped with a Smith-Root 5.0 generator-powered pulsator control box (pulsed DC, 220 volts, ~7 amp). For both sampling methods, all fish captured were identified to species, measured (total length), and released in the field; however, some specimens were preserved to confirm identifications in the laboratory.

We measured water quality variables (i.e., temperature, dissolved oxygen, specific conductivity, total dissolved solids, turbidity, pH, oxidation-reduction potential [ORP], and chlorophyll *a*) in the middle of the water column using a YSI 6600 multi-parameter data sonde. We made one measurement at each fyke net (n = 12) and one measurement at the beginning of each electrofishing transect ($n = 4$). We measured the water depth at the mouth of each fyke net and visually estimated the percent macrophyte cover for the length of the lead between the wings of each fyke net (see Bhagat and Ruetz 2011). We also visually estimated the percent macrophyte cover for the length of each electrofishing transect during nighttime fish sampling.

Results and Discussion

We characterized water quality variables at each site during fish sampling (Tables 2 and 3). The mean water depth at fyke nets was 99 cm (Table 2). Mean water temperature was similar during fyke netting (20.9 °C; Table 2) and boat electrofishing (21.6 °C; Table 3). At fyke nets, mean percent cover of macrophytes was 5% at sites #1 and #3, whereas mean percent cover of macrophytes was slightly greater at sites #2 (15%) and #4 (10%). At boat electrofishing transects, percent macrophyte cover increased across sites from the river mouth (site #1 = 25%, site #2 = 50%) toward Lake Michigan (site #3 = 75%, site #4 = 80%). There may be a trend of increasing percent macrophyte cover over time (Figure 2). We hypothesized that low densities of macrophytes in Lake Macatawa during 2014 and 2015 were caused by insufficient light penetrating the water column to allow submersed plants to grow; both turbidity from inflowing sediment and abundant phytoplankton growth in the lake water column can reduce light penetration.

As stated in past reports, aquatic macrophytes are important habitat for fish (e.g., Radomski and Goeman 2001), and their return is an important goal for the restoration of the fish community in Lake Macatawa. The presence of macrophyte beds in the vicinity of our fish sampling sites were likely related to the lower turbidity that we observed in the lake (2016-2019) compared with 2014-2015 (Figure 3B; see also Secchi disk depth dashboard in Appendix C). A detailed macrophyte survey, conducted on a 3-5 year interval, would provide useful information for Lake Macatawa's ecological status (see Ogdahl and Steinman 2014).

Compared to six Lake Michigan drowned river mouths, water quality in Lake Macatawa (measured during autumn fish sampling) has been most similar to Kalamazoo Lake, especially with respect to high turbidity and specific conductivity (Janetski and Ruetz 2015). Turbidity and specific conductivity were higher in Lake Macatawa than in Muskegon Lake, the drowned river mouth lake for which we have the longest time series of water quality observations (Bhagat and Ruetz 2011), although turbidity and specific conductivity in 2019 were among the lowest we have recorded in Lake Macatawa since we began sampling in 2014 (Figure 3). High levels of turbidity and specific conductivity often are associated with relatively high anthropogenic disturbance in Great Lakes coastal wetlands (Uzarski et al. 2005). Thus, the water quality we measured in Lake Macatawa appears on the degraded side of the spectrum among Lake Michigan drowned river mouths (see Uzarski et al. 2005, Janetski and Ruetz 2015), but there may be a trend towards improvement (Figure 3). Within the lake itself, there was a gradient in specific conductivity and turbidity, with higher levels at the east end (i.e., near river mouth) and lower levels closer to Lake Michigan (Tables 2 and 3; Figure 3). This is to be expected given that most of the sediment entering the lake comes from the Macatawa River, which runs off largely agricultural land and through urbanized Holland.

We captured 1,387 fish comprising 23 species in Lake Macatawa during 2019 sampling surveys (Table 4). The total catch was similar to previous years, although the number of fish species captured was slightly lower (Figure 4). The most abundant fishes in the combined catch (i.e., fyke netting and boat electrofishing) were brook silverside (31%), bluegill (30%), and gizzard shad (13%), which composed 74% of the total catch (Figure 5A). Four of the 23 species captured during 2019 were non-native to the Great Lakes basin (Bailey et al. 2004)—alewife, common carp, white perch, and round goby—which composed 7% of the total catch (Table 4).

In fyke netting, brook silverside (39%), bluegill (37%), round goby (5%), gizzard shad (4%), pumpkinseed (4%), and spotfin shiner (2%) were the most abundant fishes in the catch, composing 91% of all fish captured (Figure 5B). Bluegill was the most abundant species in the catch at all but site #4, where brook silverside was most abundant (Table 5). The second most abundant fish species in the catch (listed in order abundance in the catch) was gizzard shad at site #1, round goby at site #4, pumpkinseed at site #2, and brook silverside at site #3 (Table 5). The number of fish captured also varied among sites, with the most fish captured at site #4 (Table 5; Figure 6A). Compared with the previous fyke netting surveys, the most abundant species in the catch varied among years (Figure 7) as did the patterns in total catch among sites (Figure 6A). The main differences in the relative abundance (i.e., percentage of a fish species in the total catch for a given year) were that we captured more brook silverside and less gizzard shad in 2019 compared with most previous years (Figure 7). The relative abundance of bluegill in 2019 also was high compared with most other years (Figure 7). As we continue monitoring Lake Macatawa, we will be better able to assess spatiotemporal patterns and whether these observed patterns are associated with other environmental variables.

In boat electrofishing, the most abundant fishes captured were gizzard shad (45%), yellow perch (12%), white sucker (10%), largemouth bass (8%), white perch (8%), and pumpkinseed (4%), which composed 87% of the total catch (Figure 5C). Gizzard shad was most abundant in the catch at sites #1, #2, and #3, whereas yellow perch was most abundant in the catch at site #4 (Table 6). The second most abundant species in the catch was gizzard shad at site #4, largemouth bass at site #2, and white sucker at sites #1 and #3 (Table 6). Total catch also varied among sites in 2019, with the higher catch at sites #2, #3, and #4 and lower catch at site #1 (Figure 6B). Compared with previous boat electrofishing surveys, the most abundant species in the catch varied among years, although the pattern was more similar among recent years (i.e., 2016-2019; Figure 8). The main difference in 2019 was that yellow perch were less common and gizzard shad were more common in the catch than previous years (Figure 8). Overall, there appears to be less variability in species composition based on boat electrofishing surveys compared with fyke netting surveys (see Figure 8 vs. Figure 7).

In conclusion, the observations reported here are the sixth year of an effort to characterize the littoral fish assemblage of Lake Macatawa. This monitoring effort will provide a baseline to assess how the fish assemblage responds to restoration activities in the Lake Macatawa watershed. After 6 years of fish monitoring, there are both positive and negative indicators of the ecological health of Lake Macatawa. Common fish species in our surveys included gizzard shad and spotfin shiner, which are often positively associated with poor water quality (Janetski and Ruetz 2015). Alternatively, yellow perch, bluegill, and pumpkinseed also were common, and these species are often positively associated with good water quality (Janetski and Ruetz 2015; Cooper et al. 2018). As we continue to build our time series of observations, we will be able to make more robust inferences about the littoral fish assemblage of Lake Macatawa (in terms of assessing the baseline, evaluating change over time, and comparing abiotic and biotic variables with other drowned river mouth lakes in the region) and better identify likely underlying mechanisms driving spatiotemporal patterns.

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Table 1. Locations (latitude and longitude) for each 2019 fish sampling site; coordinates are the mean of the three fyke nets and the start and end of each boat electrofishing transect. Approximate site locations are depicted in Figure 1.

	Fyke netting		Electrofishing Start		Electrofishing End	
Site	Lat (°)	Long (°)	Lat (°)	Long (°)	Lat (°)	Long (°)
1	42.79593	-86.12138	42.79581	-86.12054	42.79561	-86.12321
2	42.79001	-86.14375	42.78826	-86.14490	42.79030	-86.14385
3	42.78634	-86.17449	42.78479	-86.17305	42.78672	-86.17504
4	42.77988	-86.19684	42.77935	-86.19714	42.78074	-86.19550

Table 2. Mean ± 1 standard error (*n* = 3) of water quality variables at fish sampling sites in Lake Macatawa. Measurements were made during fyke netting on 17 September 2019 with a YSI sonde.

Site	Depth (cm)	Water Temperature (°C)	Dissolved Oxygen (mg/L)	Dissolved Oxygen (%)	Specific Conductivity (µS/cm)	Total Dissolved Solids (g/L)	Turbidity (NTU)	pH	Oxidation Reduction Potential	Chlorophyll *a* (µg/L)
1	103 ± 8	22.83 ± 0.22	13.68 ± 1.12	158.6 ± 12.1	489 ± 3	0.32 ± 0.00	9.1 ± 0.3	8.30 ± 0.12	320.3 ± 2.4	53.0 ± 4.9
2	91 ± 13	20.89 ± 0.10	11.02 ± 0.02	123.5 ± 0.4	462 ± 1	0.30 ± 0.00	8.5 ± 0.5	8.26 ± 0.04	330.4 ± 1.6	54.0 ± 2.5
3	116 ± 6	20.41 ± 0.06	11.89 ± 0.03	131.8 ± 0.3	411 ± 0	0.27 ± 0.00	4.0 ± 0.1	8.66 ± 0.03	340.2 ± 2.0	31.4 ± 2.0
4	84 ± 10	19.50 ± 0.17	10.58 ± 0.33	115.4 ± 4.0	415 ± 1	0.27 ± 0.00	5.8 ± 1.0	8.45 ± 0.05	338.3 ± 2.2	34.9 ± 1.6

Table 3. Water quality variables at fish sampling sites in Lake Macatawa. Measurements were made during nighttime boat electrofishing on 5 September 2019 with a YSI sonde.

Site	Water Temperature (°C)	Dissolved Oxygen (mg/L)	Dissolved Oxygen (%)	Specific Conductivity (µS/cm)	Total Dissolved Solids (g/L)	Turbidity (NTU)	pH	Oxidation Reduction Potential (mV)	Chlorophyll *a* (µg/L)
1	21.95	12.72	145.8	517	0.336	12.0	8.53	323.5	64.5
2	22.23	14.80	169.6	473	0.308	9.7	8.89	319.8	68.3
3	21.98	13.00	148.8	418	0.272	6.6	8.95	329.0	38.9
4	20.16	13.12	144.6	401	0.261	6.5	8.89	322.1	44.3

Table 4. Number and total length (TL; mean, minimum, and maximum) of fish captured by fyke netting (*n* = 12 nets) on 17 September 2019 and boat electrofishing (*n* = 4 transects) on 5 September 2019 at four sites in Lake Macatawa. Total is the total catch combined for both gears.

			Fyke netting				Electrofishing			
				TL (cm)				TL (cm)		
Common name	Scientific name	Total	Catch	Mean	Min	Max	Catch	Mean	Min	Max
alewife	*Alosa pseudoharengus*	19	19	7.9	4.0	9.8	0	--	--	--
bowfin	*Amia calva*	7	3	51.1	37.5	66.0	4	43.8	36.0	50.5
freshwater drum	*Aplodinotus grunniens*	1	0	--	--	--	1	20.2	20.2	20.2
white sucker	*Catostomus commersonii*	35	6	41.9	37.9	46.2	29	40.4	23.6	51.2
common carp	*Cyprinus carpio*	1	0	--	--	--	1	67.4	67.4	67.4
unknown minnow	Cyprinidae[1]	2	2	3.2	3.0	3.4	0	--	--	--
spotfin shiner	*Cyprinella spiloptera*	22	22	6.2	3.8	9.1	0	--	--	--
gizzard shad	*Dorosoma cepedianum*	176	48	8.8	4.6	16.6	128	12.4	6.1	31.5
northern pike	*Esox lucius*	2	1	80.4	80.4	80.4	1	73.2	73.2	73.2
brook silverside	*Labidesthes sicculus*	429	426	7.0	4.3	10.5	3	7.4	6.6	8.2
green sunfish	*Lepomis cyanellus*	2	2	7.0	6.9	7.1	0	--	--	--
pumpkinseed	*Lepomis gibbosus*	56	45	7.1	4.1	15.2	11	14.0	6.5	19.0
bluegill	*Lepomis macrochirus*	416	411	5.1	2.2	19.3	5	18.3	17.2	19.0
largemouth bass	*Micropterus salmoides*	36	12	13.4	8.0	39.0	24	28.7	9.2	41.1
white perch	*Morone americana*	27	5	10.6	6.3	17.5	22	13.3	7.8	22.0
shorthead redhorse	*Moxostoma macrolepidotum*	2	2	48.6	47.7	49.5	0	--	--	--
round goby	*Neogobius melanostomus*	50	50	6.2	3.5	10.8	0	--	--	--
golden shiner	*Notemigonus crysoleucas*	11	7	6.9	3.5	8.7	4	8.0	6.9	8.9
emerald shiner	*Notropis atherinoides*	7	5	7.8	4.5	10.3	2	9.5	8.7	10.2
spottail shiner	*Notropis hudsonius*	21	15	8.1	5.2	12.3	6	11.1	10.7	12.0
yellow perch	*Perca falvescens*	49	13	17.1	9.6	22.0	36	16.1	7.6	20.7
bluntnose minnow	*Pimephales notatus*	5	5	6.3	4.5	9.3	0	--	--	--
black crappie	*Pomoxis nigromaculatus*	3	2	7.5	7.0	8.0	1	28.1	28.1	28.1
walleye	*Sander vitreus*	8	0	--	--	--	8	33.7	18.4	62.5
	Total	1387	1101				286			

[1]Likely emerald shiner

Table 5. Number and total length (TL; mean, minimum, and maximum) of fish captured by fyke netting (n = 3 nets per site) at four sites in Lake Macatawa on 17 September 2019. Site locations are depicted in Figure 1.

		Site #1				Site #2				Site #3				Site #4			
			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
alewife	*Alosa pseudoharengus*	**1**	8.7	8.7	8.7	**0**	--	--	--	**17**	8.0	6.0	9.8	**1**	4.0	4.0	4.0
bowfin	*Amia calva*	**2**	51.8	37.5	66.0	**1**	49.8	49.8	49.8	**0**	--	--	--	**0**	--	--	--
white sucker	*Catostomus commersonii*	**1**	41.0	41.0	41.0	**3**	42.1	37.9	46.2	**1**	40.0	40.0	40.0	**1**	43.7	43.7	43.7
unknown minnow	Cyprinidae[1]	**0**	--	--	--	**2**	3.2	3.0	3.4	**0**	--	--	--	**0**	--	--	--
spotfin shiner	*Cyprinella spiloptera*	**3**	6.1	5.1	7.0	**10**	5.4	3.8	7.8	**0**	--	--	--	**9**	7.0	5.8	9.1
gizzard shad	*Dorosoma cepedianum*	**38**	8.3	6.7	9.5	**1**	16.6	16.6	16.6	**5**	11.5	4.6	16.5	**4**	8.3	4.6	13.2
northern pike	*Esox lucius*	**1**	80.4	80.4	80.4	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
brook silverside	*Labidesthes sicculus*	**37**	7.3	4.8	8.5	**2**	5.9	5.2	6.6	**20**	6.6	4.8	8.0	**367**	7.0	4.3	10.5
green sunfish	*Lepomis cyanellus*	**0**	--	--	--	**0**	--	--	--	**2**	7.0	6.9	7.1	**0**	--	--	--
pumpkinseed	*Lepomis gibbosus*	**13**	8.7	4.1	15.2	**27**	6.7	4.8	13.4	**0**	--	--	--	**5**	5.8	4.4	6.3
bluegill	*Lepomis macrochirus*	**191**	5.0	2.5	19.3	**160**	4.8	2.2	7.2	**46**	6.0	3.4	19.0	**14**	6.0	3.9	19.1
largemouth bass	*Micropterus salmoides*	**2**	21.9	8.9	34.9	**2**	8.4	8.3	8.5	**0**	--	--	--	**8**	12.5	8.0	39.0
white perch	*Morone americana*	**0**	--	--	--	**3**	11.5	6.3	17.5	**0**	--	--	--	**2**	9.3	9.3	9.3
shorthead redhorse	*Moxostoma macrolepidotum*	**1**	47.7	47.7	47.7	**0**	--	--	--	**0**	--	--	--	**1**	49.5	49.5	49.5
round goby	*Neogobius melanostomus*	**0**	--	--	--	**5**	8.5	6.1	10.8	**7**	6.0	4.3	9.1	**38**	5.9	3.5	8.5
golden shiner	*Notemigonus crysoleucas*	**3**	5.0	3.5	6.1	**2**	8.2	7.7	8.7	**0**	--	--	--	**2**	8.5	8.5	8.5
emerald shiner	*Notropis atherinoides*	**1**	4.5	4.5	4.5	**0**	--	--	--	**3**	9.9	9.5	10.3	**1**	4.6	4.6	4.6
spottail shiner	*Notropis hudsonius*	**1**	11.2	11.2	11.2	**2**	11.6	10.8	12.3	**3**	5.7	5.4	5.9	**9**	7.8	5.2	11.4
yellow perch	*Perca falvescens*	**7**	15.6	9.6	20.0	**3**	17.1	16.3	17.6	**2**	20.7	19.3	22.0	**1**	20.1	20.1	20.1
bluntnose minnow	*Pimephales notatus*	**0**	--	--	--	**4**	5.6	4.5	8.5	**0**	--	--	--	**1**	9.3	9.3	9.3
black crappie	*Pomoxis nigromaculatus*	**0**	--	--	--	**1**	7.0	7.0	7.0	**0**	--	--	--	**1**	8.0	8.0	8.0
	Total	302				228				106				465			

[1]Likely emerald shiner

Table 6. Number and total length (TL; mean, minimum, and maximum) of fish captured by nighttime boat electrofishing (n = 1 transect per site) at four sites in Lake Macatawa on 5 September 2019. Site locations are depicted in Figure 1.

		Site #1				Site #2				Site #3				Site #4			
			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
bowfin	*Amia calva*	**0**	--	--	--	**1**	36.0	36.0	36.0	**0**	--	--	--	**3**	46.4	41.3	50.5
freshwater drum	*Aplodinotus grunniens*	**0**	--	--	--	**1**	20.2	20.2	20.2	**0**	--	--	--	**0**	--	--	--
white sucker	*Catostomus commersonii*	**5**	32.3	23.6	40.5	**6**	40.2	37.1	47.2	**7**	40.4	35.2	51.2	**11**	44.2	37.6	48.5
common carp	*Cyprinus carpio*	**1**	67.4	67.4	67.4	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
gizzard shad	*Dorosoma cepedianum*	**20**	11.1	8.0	14.0	**21**	11.3	7.9	14.2	**71**	13.0	6.1	31.5	**16**	12.6	9.0	16.5
northern pike	*Esox lucius*	**0**	--	--	--	**0**	--	--	--	**1**	73.2	73.2	73.2	**0**	--	--	--
brook silverside	*Labidesthes sicculus*	**0**	--	--	--	**1**	8.2	8.2	8.2	**0**	--	--	--	**2**	7.1	6.6	7.5
pumpkinseed	*Lepomis gibbosus*	**0**	--	--	--	**5**	12.4	6.5	16.0	**2**	12.3	11.7	12.8	**4**	17.0	12.2	19.0
bluegill	*Lepomis macrochirus*	**0**	--	--	--	**4**	18.4	17.2	19.0	**1**	18.0	18.0	18.0	**0**	--	--	--
largemouth bass	*Micropterus salmoides*	**4**	35.5	30.6	41.1	**10**	27.2	21.1	34.0	**4**	34.1	30.0	36.5	**6**	23.0	9.2	40.3
white perch	*Morone americana*	**0**	--	--	--	**8**	14.7	7.8	22.0	**6**	14.8	8.7	21.9	**8**	10.8	8.6	16.4
golden shiner	*Notemigonus crysoleucas*	**0**	--	--	--	**2**	7.9	6.9	8.9	**0**	--	--	--	**2**	8.1	7.7	8.5
emerald shiner	*Notropis atherinoides*	**2**	9.5	8.7	10.2	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
spottail shiner	*Notropis hudsonius*	**1**	10.9	10.9	10.9	**0**	--	--	--	**1**	11.0	11.0	11.0	**4**	11.2	10.7	12.0
yellow perch	*Perca flavescens*	**0**	--	--	--	**9**	17.8	15.6	20.5	**2**	16.1	15.7	16.5	**25**	15.5	7.6	20.7
black crappie	*Pomoxis nigromaculatus*	**1**	28.1	28.1	28.1	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
walleye	*Sander vitreus*	**0**	--	--	--	**5**	26.2	18.4	36.5	**2**	59.4	56.3	62.5	**1**	19.8	19.8	19.8
	Total	34				73				97				82			

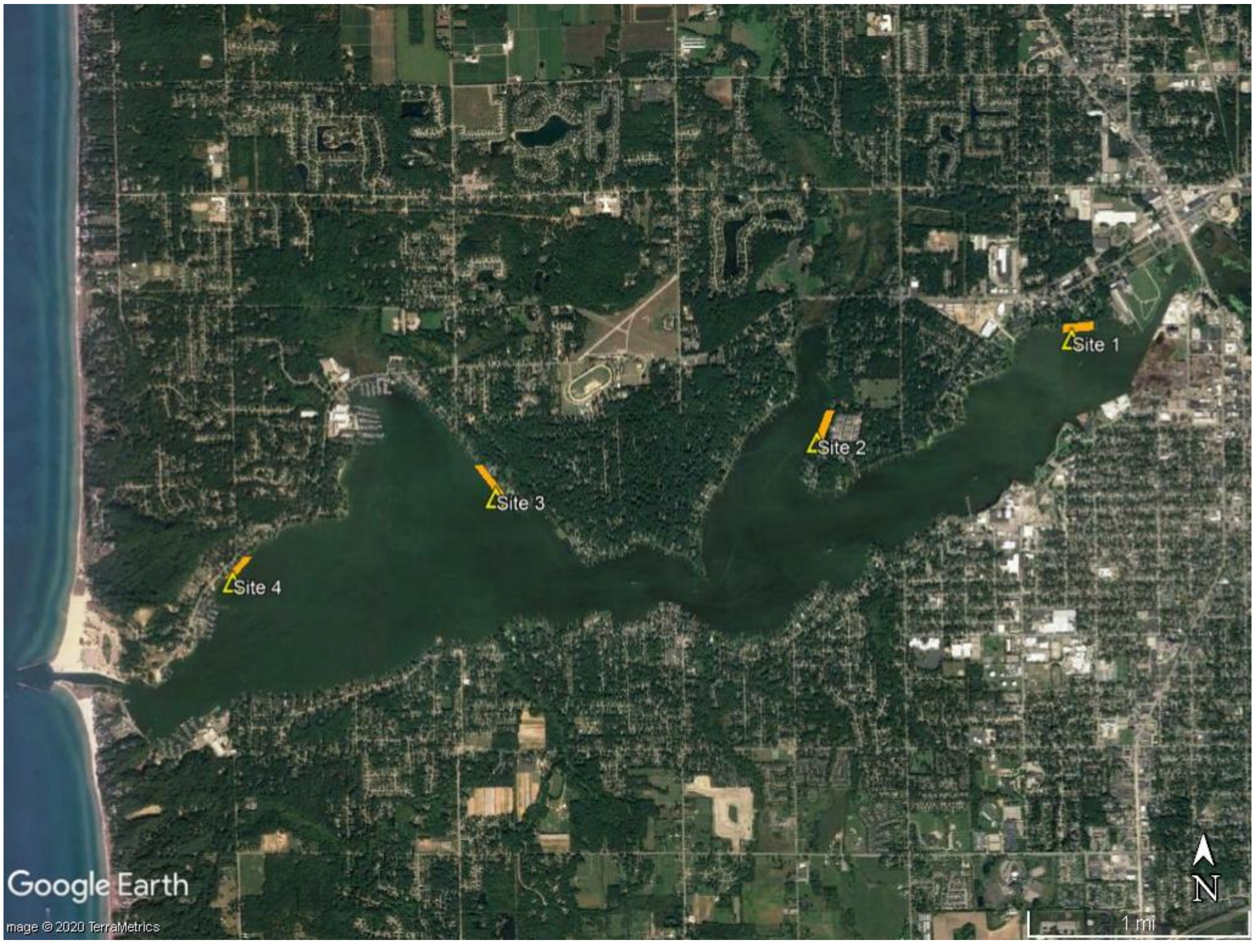

Figure 1. Map of Lake Macatawa (Ottawa County, Michigan) showing fish sampling sites (triangles). The orange transects depict approximately where boat electrofishing was conducted at each site. Site #1 is closest to the Macatawa River and site #4 is closest to Lake Michigan. Note that riparian vegetation was cleared at site #2 in 2016.

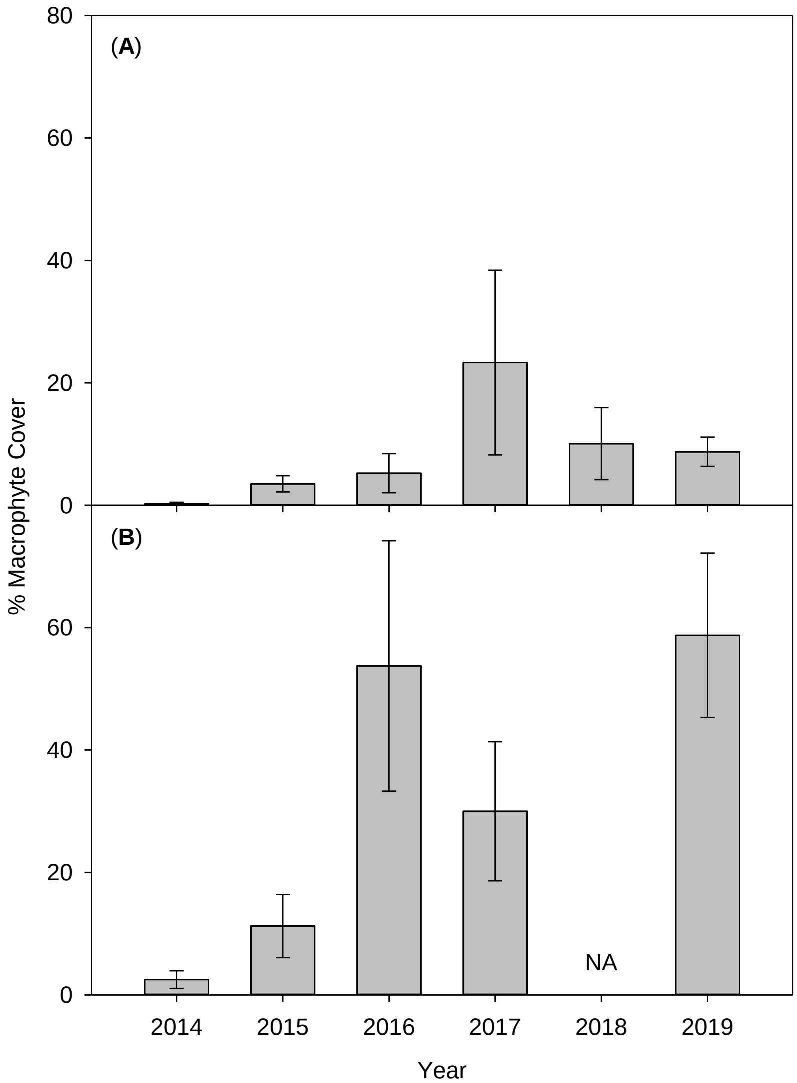

Figure 2. Mean (±1 standard error) percent macrophyte cover visually estimated at (**A**) fyke net locations and (**B**) boat electrofishing transects in Lake Macatawa (*n* = 4 sites per year). Note that the area where macrophyte cover was assessed during fyke netting is much less compared with a boat electrofishing transect. NA means data were not available (i.e., water clarity during boat electrofishing prevented visual estimation).

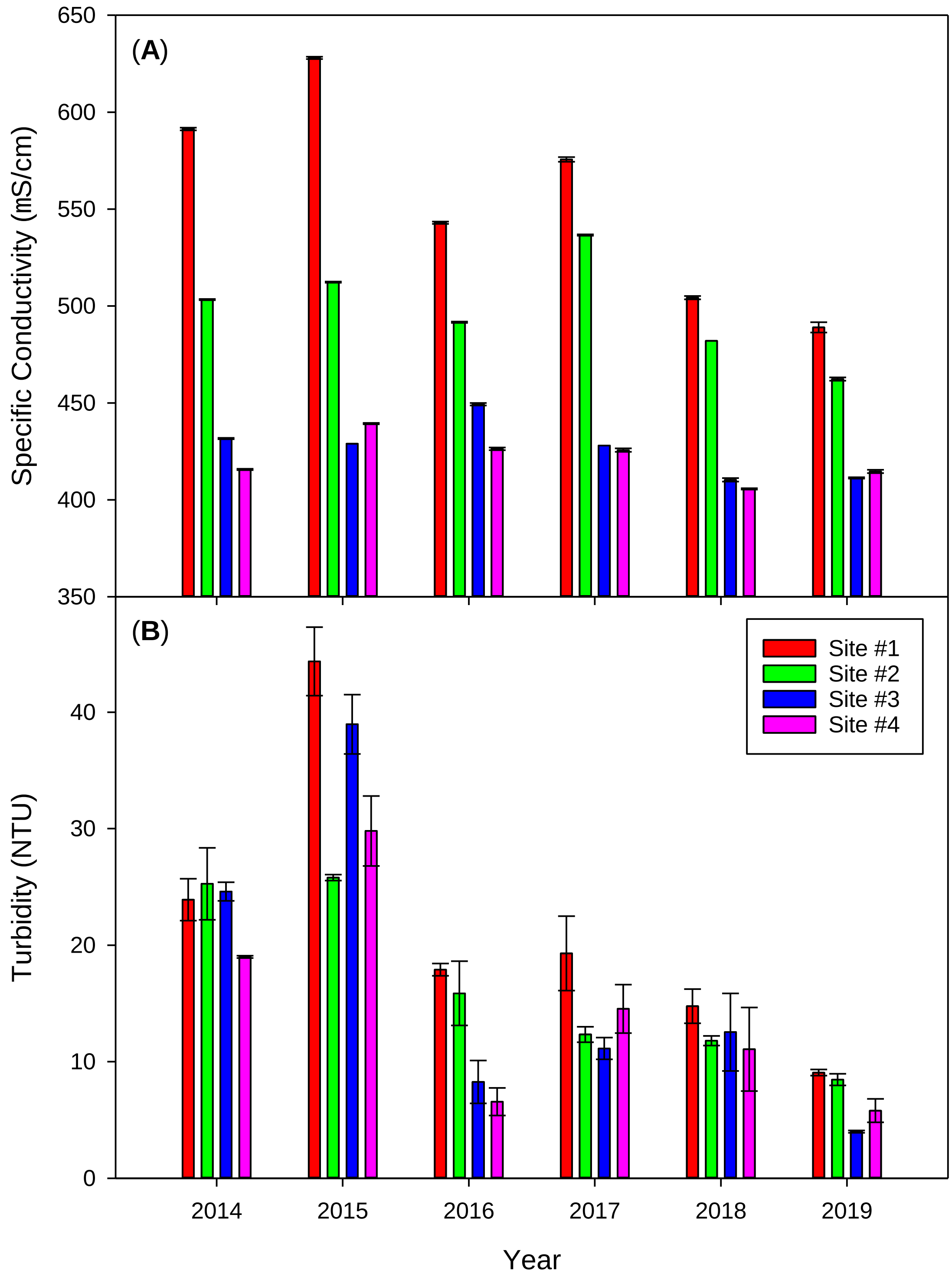

Figure 3. Mean (**A**) specific conductivity and (**B**) turbidity measured during fyke netting in Lake Macatawa. Error bars represent ±1 standard error (n = 3 nets per site), although they may be too small to be visible for some means.

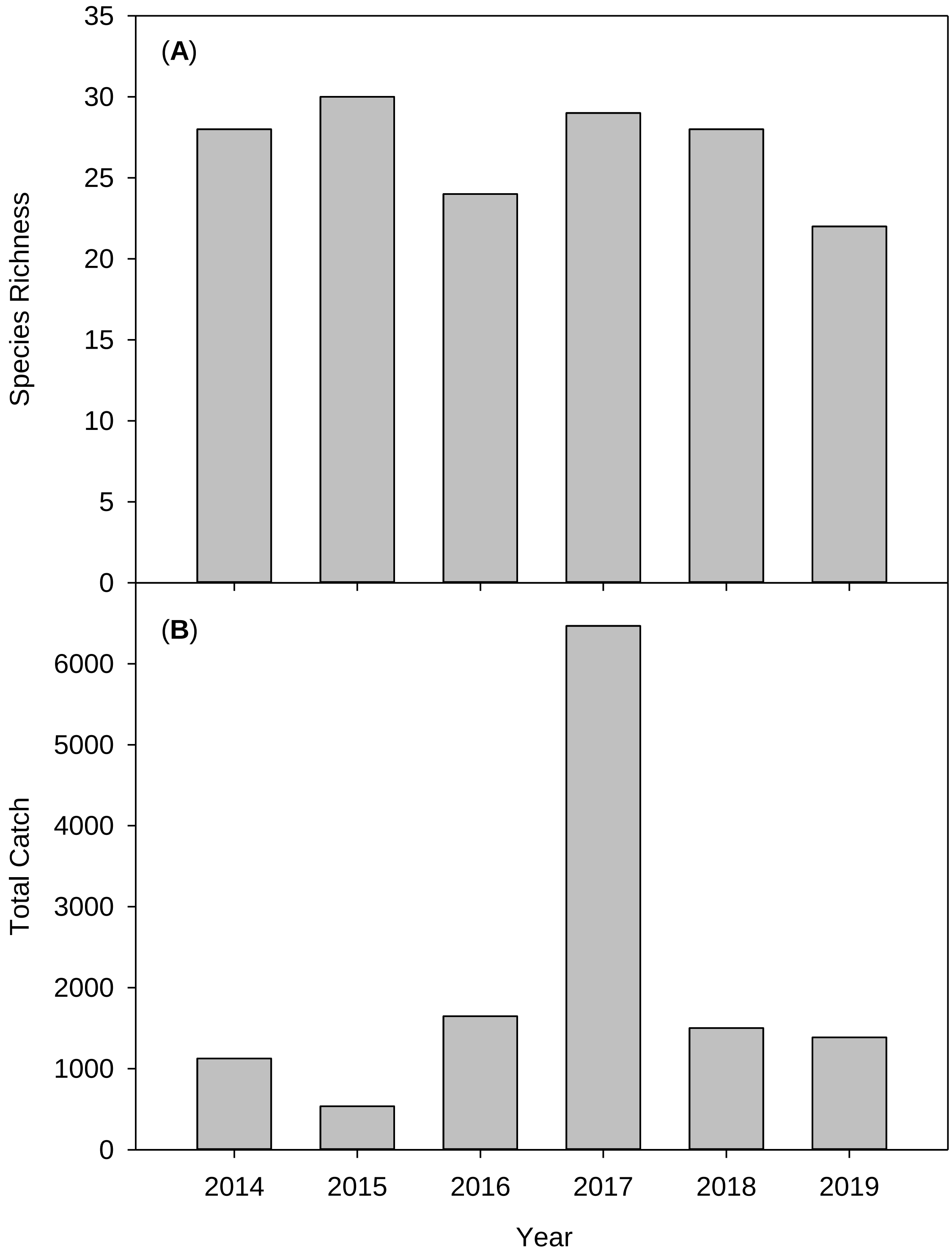

Figure 4. (**A**) Number of fish species captured and (**B**) total number of fish captured using both fyke netting and boat electrofishing each year in Lake Macatawa. *Note*: the high catch in 2017 was due to 5,288 brook silversides captured from a single fyke net at site #4.

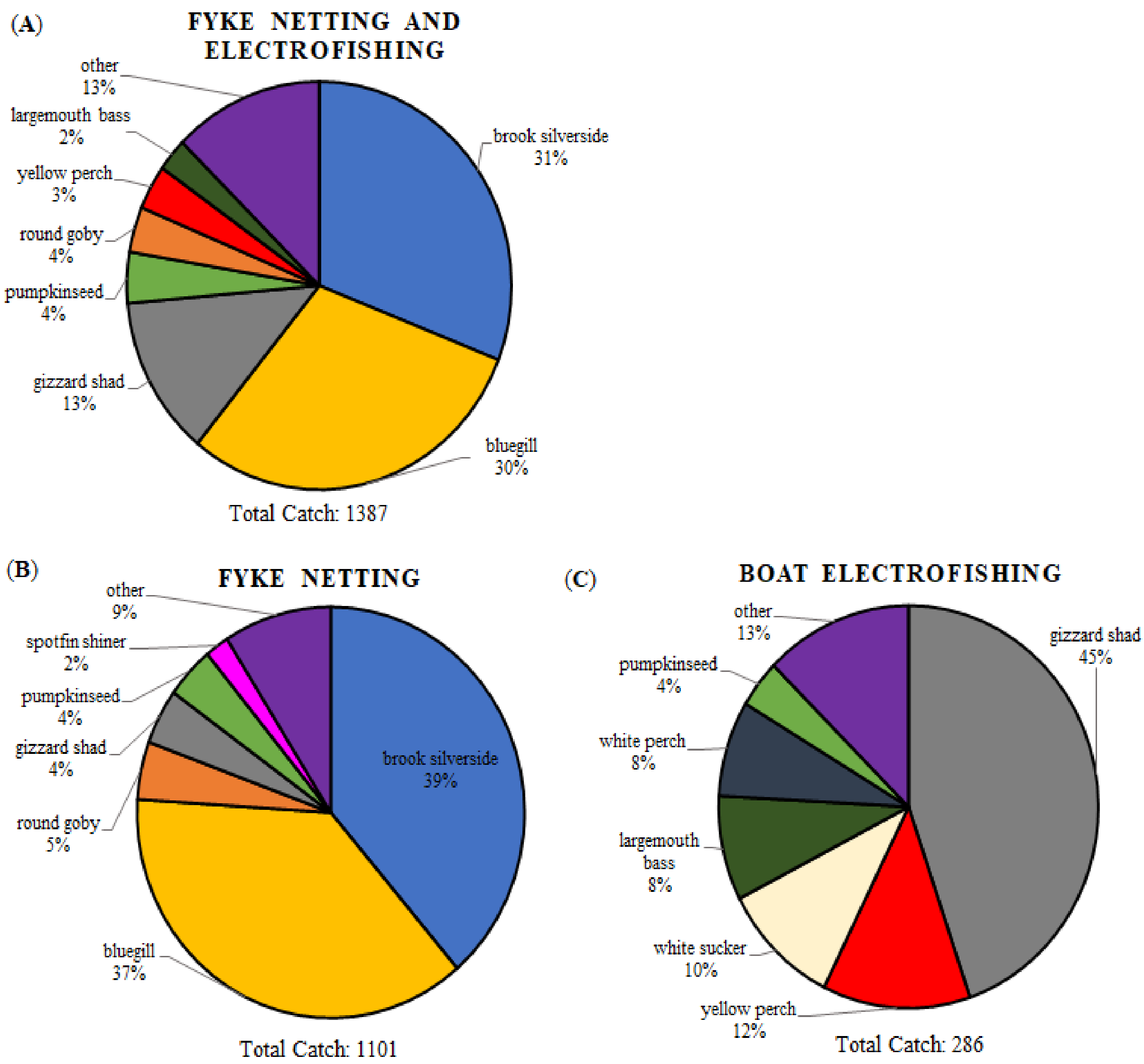

Figure 5. Fish species captured in littoral habitats of Lake Macatawa by (**A**) fyke netting and boat electrofishing (i.e., combined catch), (**B**) fyke netting (n = 12 nets), and (**C**) boat electrofishing (n = 4 transects) during September 2019. Catch data, including the species pooled in the "other" category, are reported in Table 4.

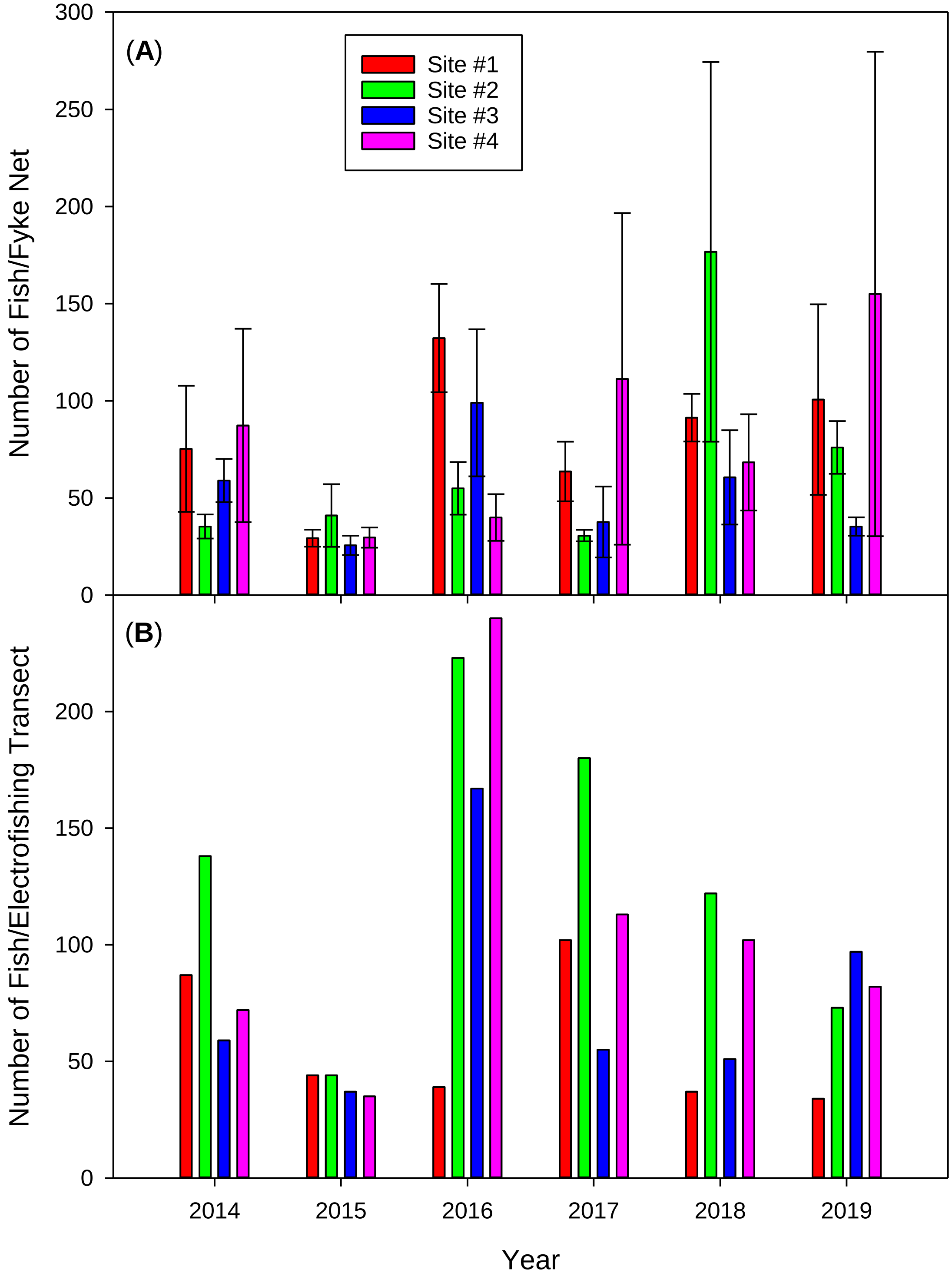

Figure 6. (**A**) Mean number (±1 standard error) of fish captured in fyke nets (*n* = 3 nets per site) and (**B**) number of fish captured during a boat electrofishing transect (*n* = 1 transect per site) in Lake Macatawa. *Note*: 5,288 brook silversides captured in a single fyke net at site #4 in 2017 were excluded when calculating means for fyke netting.

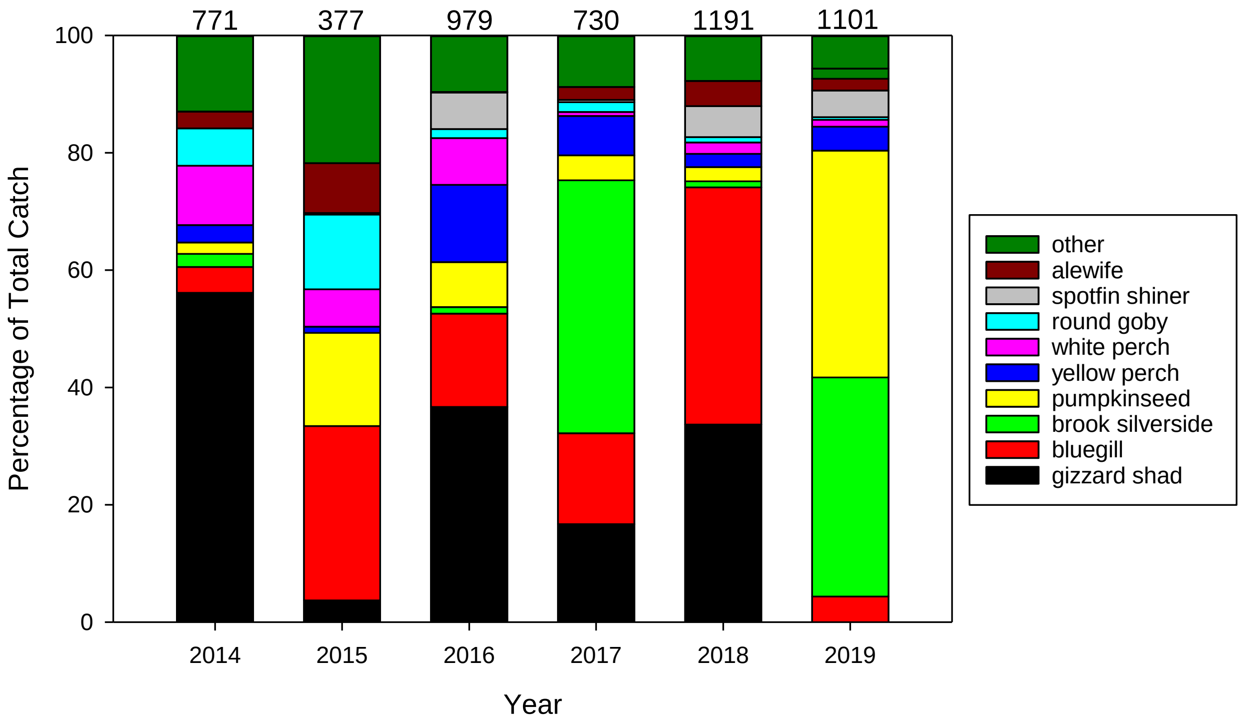

Figure 7. Fish species composition (pooled across sites) in fyke netting surveys for each sampling year. The number of fish captured differed among years, which is reported at the top of each bar. *Note*: 5,288 brook silversides captured in a single fyke net at site #4 in 2017 were excluded from the percentage of total catch.

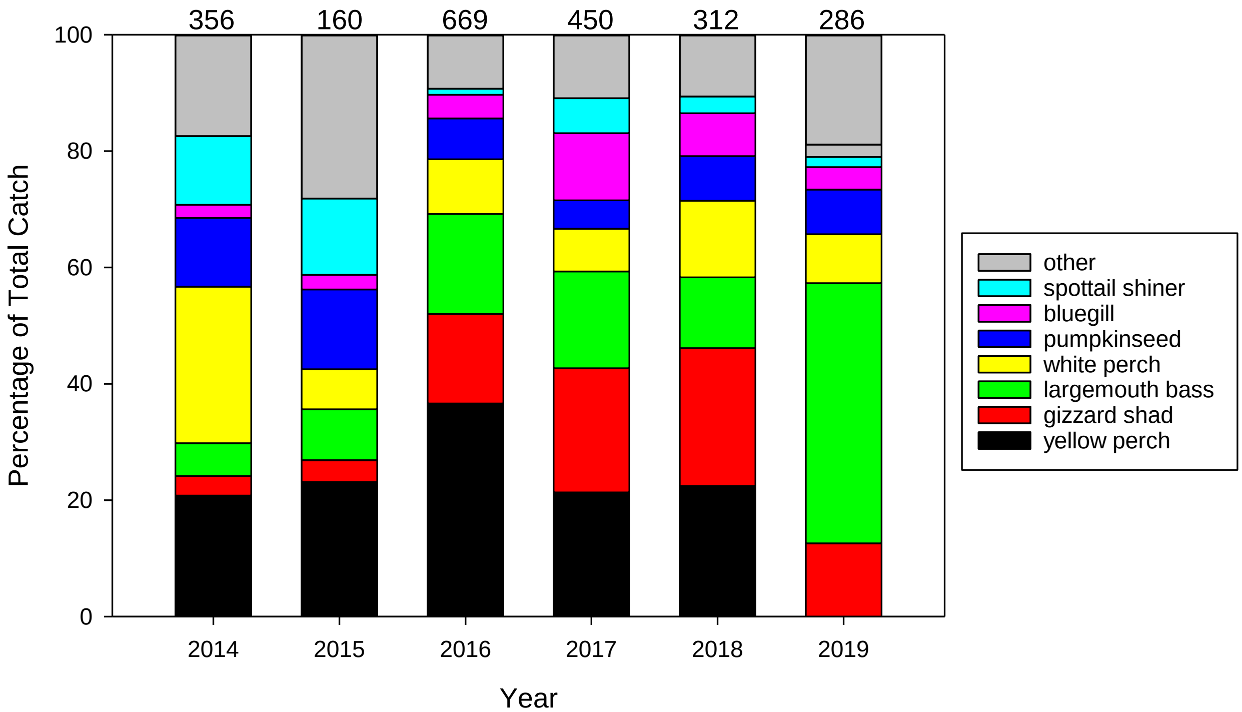

Figure 8. Fish species composition (pooled across sites) in nighttime boat electrofishing surveys for each sampling year. The number of fish captured differed among years, which is reported at the top of each bar.

Appendix B.

Slag Filter BMP Preliminary Performance in the Macatawa Watershed

January 2020

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1. Overview

The Macatawa Watershed suffers from excess phosphorus within the watershed with previous work demonstrating high amounts of both total and bioavailable phosphorus (P) present in tile drain effluent from agricultural fields (Clement and Steinman 2017). Iron slag, a waste product from the steel industry, can chemically bind P and has been implemented previously in agricultural settings (Roychand et al. 2020; Hua et al. 2016). The Outdoor Discovery Center (ODC) Network, along with Dykhuis Farms and Plant Tuff, Inc., has committed to install up to six iron slag filters at the end of agricultural tile lines in the Macatawa watershed. The goal of Grand Valley State University's (GVSU) Annis Water Resources Institute (AWRI) is to evaluate the efficiency of these systems in removing P, while also monitoring for the presence of unintended chemicals leaching from the slag, which could potentially be released into surface waterways. It is our current understanding that these filters are the first application of their kind within Michigan.

2. Methods

2.1 Overall site description

The use of large-scale iron slag filters (Figs. 1, 2) to remove phosphorus from agricultural tile drain runoff was proposed in the Macatawa watershed in 2018 and three sites have been selected as of 12/31/19 for construction. Several additional sites are being considered for installation in 2020.

Filters are designed to work passively, receiving water after it infiltrates through soils into subsurface farm field drains (Fig. 1). Water moves up and through the iron slag gravel in a large concrete tank, where slag binds with and removes P from the water before it is passively released to adjoining surface waters (Figs. 1, 2). A layer of calcium carbonate particulate was applied within the treatment tank to help balance tile drain water pH. A control box allows for the slag filter to be bypassed if too much water is in the tile drains (indicated by standing water on the farm field), and serves as the inlet access point for water collection for most sites. Outflow water can be sampled either from the outflow pipe, or the access points in the top of the tank.

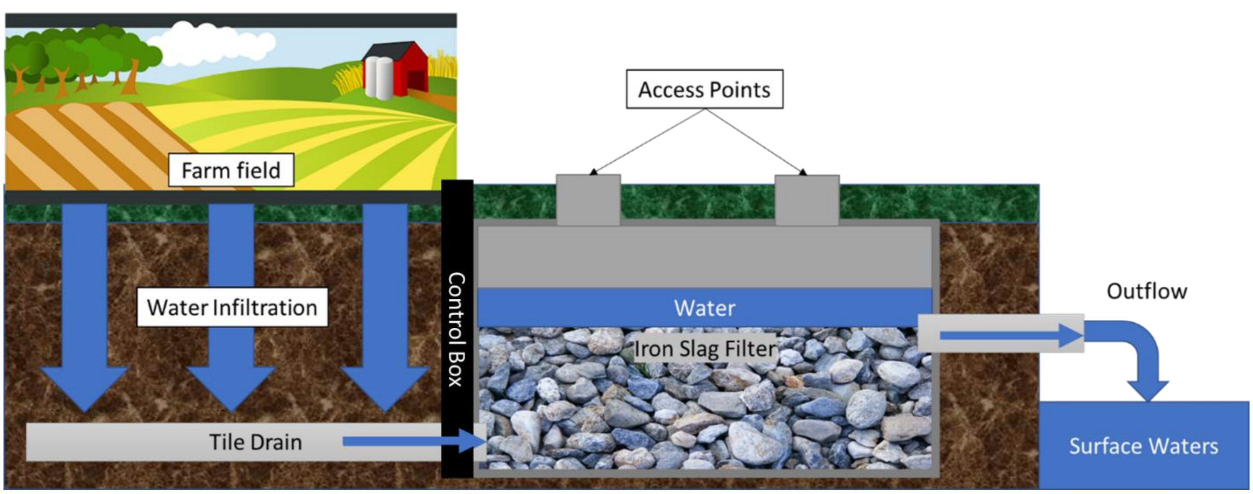

Figure 1. Stylized cross-section of iron slag filter design/function. Figure is not to scale as iron slag filter size is dependent upon multiple factors (e.g., size of the tiled field, water velocity from tile drains, soil type). Image credit: Maggie Oudsema.

Figure 2. Inlet pipes being laid in the bottom of an iron slag filter. Drainage water enters at the bottom, moves up through the iron slag material (not yet installed) and leaves in a pipe near the top (not pictured) that leads to a nearby surface drain (not pictured). Photo credit: Macatawa Area Coordinating Council.

Figure 3. Completed iron slag filter sites are indicated by the yellow triangles, the red polygons represent the adjoining fields being drained, each approximately 30 acres. Filmore Flex and Oak Grove 2 sites flow into the South Branch of the Macatawa River, which later joins the main branch of the Macatawa River. Behind Mill 1 flows into Peter's Creek. Map was created using Google Earth.

2.2 Field and Laboratory Processes

Prior to construction, grab samples were taken monthly at only the outflow pipe, which at that time was a direct connection from the tile drain pipes to adjoining surface waters. First sampling dates are provided in Table 1. Post-construction samples were taken two times per month and sampling methods varied among sites due to variation in filter design and implementation (Fig. 4). Post-construction sampling occurred at the inflow using a hand pump and hose to siphon water accessed through inflow pipe at Behind Mill 1 (Fig. 4A) and through the control box at Oak Grove 2 and Filmore Flex (Fig. 1, 4B). Outflow was sampled at the original outflow pipe (Fig. 4E; which was undisturbed during construction) for all sites, however, when the outflow pipe was inaccessible due to being underwater, samples were taken via one of the access points on top of the tank using a hand pump and hose (Figs. 4C-D).

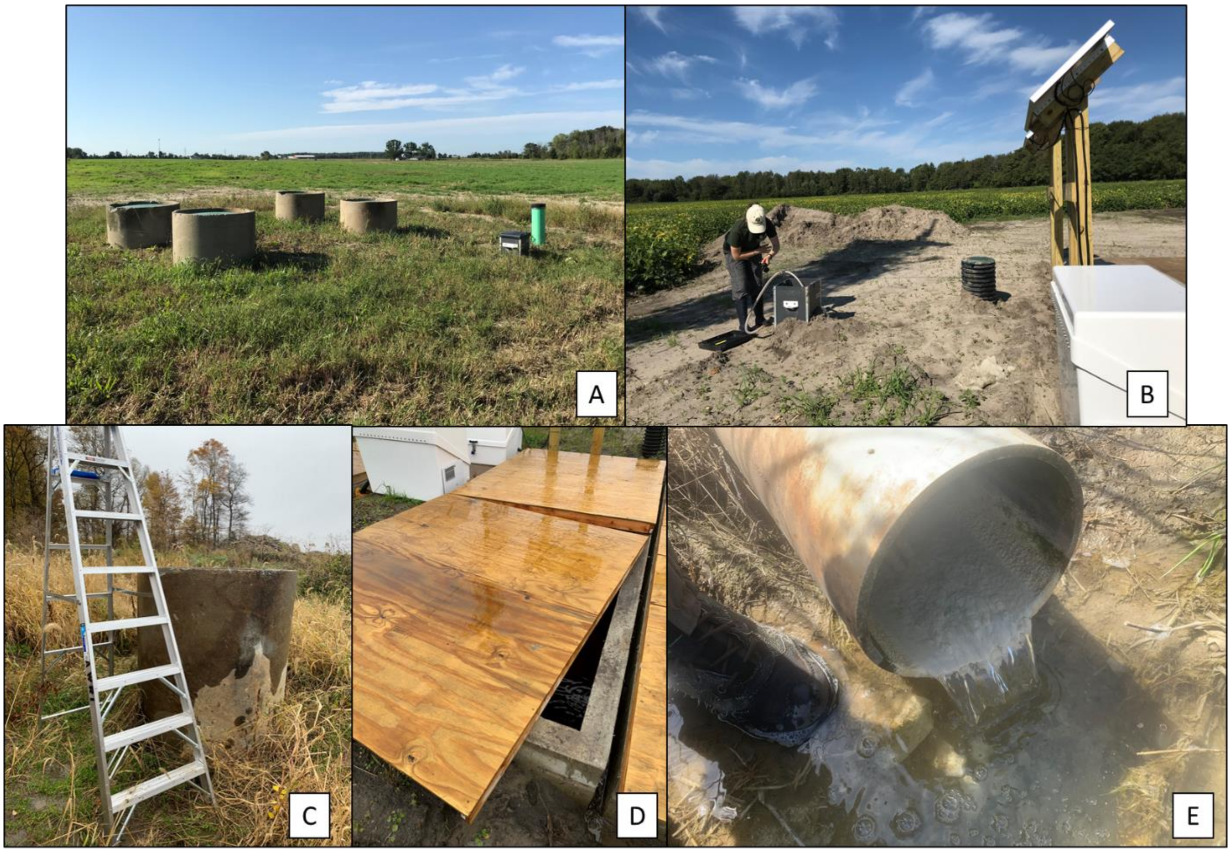

Figure 4. Photos of different sampling locations. A) Behind Mill 1 with completed iron slag filter in place. The green tube (far right) is an inflow sampling port that was installed only at this location. Access ports (four cement upright tubes with green caps) are for two slag filters which receive tile drain water from the adjoining field (in background) and used to sample outflow. B) Research Assistant, Emily Kindervater, using a hand pump siphon to sample from a below-ground control box (inflow) at Filmore Flex; Oak Grove 2 has a similarly constructed inflow (not shown). C) Slag filter outflow access point for Oak Grove 2, which required a ladder to removing the large plastic green cap and accessing filters. D) Slag filter outflow access point from Filmore Flex in which is covered with large plywood lids. E) Outflow pipe at Behind Mill 1; the white particulate residue inside the tube is from a calcium carbonate layer placed on top of the iron slag inside the tanks to balance pH.

AWRI monitored general water quality with a YSI 6600 data sonde (temperature, dissolved oxygen [DO], pH, specific conductivity [SpCond], total dissolved solids [TDS], redox potential [ORP: oxidation-reduction potential – the degree to which a substance is capable of oxidizing or reducing another substance], and turbidity). Grab samples were collected for analysis of TP and SRP. All samples were placed in a cooler on ice until received by the AWRI lab, usually within 4 hours, where they were stored and processed appropriately. Water for SRP analyses was syringe-filtered through 0.45-µm membrane filters into scintillation vials and refrigerated until analysis. TP and SRP were analyzed on a SEAL AQ2 discrete automated analyzer (U.S. EPA 1993). Any values below detection were calculated as ½ the detection limit.

Multiple slag sites were sampled before construction for project partners to determine which locations were best suited for iron slag filter installation. We present only preliminary data here, as a full year of post-construction sampling has not yet occurred. This report only has general quantitative comparisons of the results and statistical analysis will not be attempted until one full year of sampling has been completed. Of the three currently constructed filters, two were completed in April 2019 (Behind Mill 1 and Oak Grove 2; Fig. 3) and the third (Filmore Flex; Fig. 3) in September 2019.

2.3 *Metals and Inorganic Compounds*

Chemical analysis sampling for metals (Mercury, Arsenic, Barium, Cadmium, Chromium, Cobalt, Copper, Lead, Molybdenum, Nickel, Selenium, Silver, and Zinc), inorganic compounds (2-Methylnaphthalene, Acenaphthene, Acenaphthylene, Anthracene, Benzo (a) anthracene, Benzo (a) pyrene, Benzo (b) fluoranthene, Benzo (g,h,i) perylene, Benzo (k) fluoranthene, Chrysene, Dibenz (a,h) anthracene, Fluoranthene, Fluorene, Indeno (1,2,3-cd) pyrene, Naphthalene, Phenanthrene, Pyrene), and available cyanide in water were conducted prior to construction (for any background concentrations), and are scheduled to be conducted three times post-construction (1-week, 6-months and 1-year). Sampling for chemical analysis was conducted by AWRI staff using the procedures, sampling bottles, and analyses provided by TRACE Analytical Laboratories, Inc. (Muskegon, MI). Any values below TRACE's analytical detection methods were calculated as ½ their respective detection limits.

3. Results and Discussion

3.1 Pre-Construction Sampling

Pre-construction mean water temperatures varied between sites and ranged 8.61 - 13.89 °C, with lower mean temperatures associated with Behind Mill 1 and Oak Grove 2 likely due to the sampling occurring mostly in the winter months, whereas Filmore Flex occurred during warmer months (Table 1). Mean DO ranged 7.86 - 11.30 mg/L with the lowest mean DO found at Filmore Flex (Table 1). All sites had similarly neutral mean pH (Table 1). Mean SpCond ranged 676 - 894 µS/cm, the lowest being at Oak Grove 2, while Behind Mill 1 and Filmore Flex were similar (Table 1). Mean ORP was similar among all sites (Table 1). Mean turbidity was lower at Behind Mill 1 and Oak Grove 2 than at Filmore Flex with all sites <25 NTU (Table 1).

Mean P concentrations varied widely among sites and are likely reflective of field management, drainage, and time of year sampled. The highest mean TP and SRP values were measured at Filmore Flex at 721 and 267 µg/L, respectively (Table 2).

3.2 Post-Construction Sampling

Mean post-construction water temperature increased from pre-construction at Behind Mill 1 and Oak Grove 2, with a slight decrease at Filmore Flex at both inflow and outflow (Table 1). Post-construction temperatures were similar between sites (Table 1), as sampling occurred at a similar time of year. Post-construction mean DO was ~7 mg/L for all sites at inflow and outflow, representing a decrease at Behind Mill 1 and Oak Grove 2 from pre-construction (Table 1). All sites maintained neutral mean pH post-construction, which was reinforced by the calcium carbonate application during construction (Table 1). Mean SpCond and TDS increased in Behind Mill 1 and Oak Grove 2 inflow while outflow remained similar to pre-construction conditions (Table 1). Filmore Flex showed reductions in both mean SpCond and TDS inflow and outflow post construction (Table 1). Mean ORP showed little variation between inflow and outflow or pre- and post-construction (Table 1).

A large difference was measured in mean turbidity at Behind Mill 1 and Oak Grove 2 between post-construction inflow and outflow. Outflow turbidity was similar to pre-construction levels at these sites. Filmore Flex showed a decrease from pre- to post-construction while inflow and outflow were similar to the other two sites (Table 1). Reduction in turbidity between inflow and outflow is anticipated as water velocity slows down as it flows through the iron slag filter, allowing sediment particles to settle out of solution before the water reaches the surface waters. We did not measure sediment load within the pipes pre- or post-construction. It is unclear if the sediment will impact the longevity of the slag filter's P trapping effectiveness.

Mean P concentrations continued to vary widely among sites and likely reflect differences in field management practices, drainage, and time of year sampled. If conditions were constant among all fields, we'd expect the pre-construction outflow concentrations to be equivalent to the post-construction inflow concentrations but that clearly was not the case (Table 2). **Nonetheless, after installation of the slag filters, the results clearly show these devices remove both SRP and TP from the tile drain effluent**: % reductions of SRP ranged from 76% at Oak Grove 2 to 26.3% at Filmore Flex (Table 2). Percent reductions of TP ranged from 87.1% at Oak Grove 2 to 20.4% at Filmore Flex (Table 2).

3.3 Metals and Inorganic Compounds

Metals were measured during pre-construction and 1-week post-construction sampling at all sites while 6-month post-construction measurement have been completed to date only at Behind Mill 1 and Oak Grove 2 (Fig. 5). All metals, inorganic compounds, and available cyanide were below Environmental Protection Agency (EPA; source: https://www.epa.gov/dwstandardsregulations/secondary-drinking-water-standards-guidance-nuisance-chemicals) and World Health Organizations (WHO; source: https://www.wqa.org/learn-about-water/common-contaminants) standards for drinking water.

Mercury was below 10 ng/L for all sites, which is well below drinking water standards 2,000 ng/L (EPA) and 6,000 ng/L (WHO). Behind Mill 1 is showing a slight increase in Mercury between pre-construction and 6-months post-construction, while Oak Grove 2 is showing a slight decrease during the same time (Fig. 5A). Arsenic was found in detectable amounts only at Filmore Flex pre-construction, and was below both the EPA and WHO standard of 0.010 mg/L (Fig. 5B). Barium was found below detection 1-week post construction at Behind Mill 1 and Oak Grove 2, although all samples were below drinking standards of 2.0 mg/L (EPA) and 0.7 mg/L (WHO). Behind Mill 1 and Oak Grove 2 were similar pre- and 6-month post-construction, while Filmore Flex still showed a reduced but still measurable 1-week post-construction concentration (Fig. 5C). Chromium was detected 1-week post-construction at only Oak Grove 2 and Filmore Flex, still below both the EPA (0.10 mg/L) and WHO (0.05 mg/L) standards (Fig. 5D). Copper at Behind Mill 1 and Oak Grove 2 increased from pre-construction to 6-month post-construction. While both Oak Grove 2 and Filmore Flex decreased pre- to 1-week post construction (Fig. 5E, 5F). Zinc was below detection at Behind Mill 1 in both pre- and post-construction. Zinc was detected only at Oak

Grove 2 during 6-month post construction sampling. Filmore Flex increased from pre- to 1-week post-construction although all values were below the EPA (5 mg/L) standard (Fig. 5G, 5H).

Cadmium, Cobalt, Lead, Molybdenum, Nickel, Selenium, Silver, Naphthalene, 2-Methylnaphthalene, Acenaphthylene, Acenaphthene, Fluorene, Phenanthrene, Anthracene, Fluoranthene, Pyrene, Benzo (a) anthracene, Chrysene, Benzo (b) fluoranthene, Benzo (k) fluoranthene, Benzo (a) pyrene, Indeno (1,2,3-cd) pyrene, Dibenz (a,h) anthracene, Benzo (g,h,i) perylene, Nitrobenzene-d5, 2-Fluorobiphenyl, Terphenyl-d14, and Cyanide (available) were all below detection limits for all sites and sampling dates (data not shown).

4. Summary

The iron slag filters have been installed for less than one year so long-term performance is impossible to predict but the current data indicate they are effective at removing P from tile drain effluent. The variation in % reduction is high among the 3 sites, but the lowest performing site (Filmore Flex) is also the most recently installed, so it may still be functioning within an acclimation period. Future sampling will determine if its effectiveness improves over time.

Encouragingly, there is no indication that the tile slag is releasing toxic metals, inorganic chemicals, or cyanide at levels that would cause concerns for drinking water standards. If these chemicals were to be released, we'd expect them to flush with their initial use, which has not been the case. However, we will continue to monitor these chemicals.

Table 1. Mean (1 standard deviation [SD]) values of selected water quality parameters for tile drain in/outflow iron slag pre- and post-construction monitoring. Date of first sampling is provided below each site name. Data are shaded to improve readability. n = number of successful sampling events per site; abbreviations in main text.

Site	Pre/ Post	Outflow/ Inflow	n	Temp. (°C)	DO (mg/L)	pH	SpCond (µS/cm)	TDS (g/L)	ORP	Turbidity (NTU)
Behind Mill 1 (4/25/2019)	Pre	Outflow	8	8.61 (5.54)	10.93 (2.00)	7.91 (0.46)	872 (130)	0.567 (0.085)	253.2 (103.0)	7.9 (16.8)
	Post	Inflow	17	15.69 (4.51)	8.09 (1.68)	7.08 (0.28)	1039 (496)	0.675 (0.322)	313.0 (72.8)	47.0 (67.5)
		Outflow	17	15.14 (3.99)	6.84 (1.73)	7.77 (0.73)	867 (157)	0.564 (0.102)	289.1 (87.1)	6.8 (8.5)
Oak Grove 2 (5/7/2019)	Pre	Outflow	7	9.13 (7.20)	11.30 (2.64)	8.19 (0.25)	676 (132)	0.439 (0.086)	284.3 (72.6)	11.8 (NA)
	Post	Inflow	12	16.67 (4.70)	7.72 (1.72)	7.59 (1.31)	757 (199)	0.492 (0.129)	306.7 (71.9)	28.3 (29.5)
		Outflow	10	16.66 (5.28)	7.54 (1.54)	7.95 (1.05)	667 (184)	0.434 (0.119)	291.8 (54.2)	6.9 (8.8)
Filmore Flex (9/18/2019)	Pre	Outflow	9	13.89 (5.69)	7.86 (1.55)	7.65 (1.81)	894 (1187)	0.581 (0.772)	221.0 (113.5)	22.5 (47.8)
	Post	Inflow	4	11.86 (4.89)	7.79 (1.42)	7.15 (0.10)	561 (90)	0.365 (0.058)	343.3 (40.9)	4.0 (2.4)
		Outflow	4	11.74 (4.48)	7.89 (1.27)	6.54 (0.28)	475 (71)	0.309 (0.046)	349.7 (31.7)	4.3 (1.5)

Table 2. Mean (1 SD) values of soluble reactive phosphorus (SRP) and total phosphorus (TP) for tile drain in/outflow iron slag pre- and post-construction monitoring. Data are shaded to improve readability. n= number of successful sampling events per site. N/A = not applicable.

Site	Pre/ Post	Outflow/ Inflow	n	SRP (µg/L)	TP (µg/L)	% Reduction SRP	% Reduction TP
Behind Mill 1	Pre	Outflow	8	113 (105)	147 (131)	N/A	
	Post	Inflow	12	401 (446)	745 (826)	57.6	68.7
		Outflow	12	170 (130)	233 (182)		
Oak Grove 2	Pre	Outflow	7	229 (328)	254 (328)	N/A	
	Post	Inflow	13	179 (65)	582 (87)	76.0	87.1
		Outflow	13	43 (60)	75 (88)		
Filmore Flex	Pre	Outflow	9	267 (201)	721 (699)	N/A	
	Post	Inflow	5	452 (104)	496 (90)	26.3	20.4
		Outflow	5	333 (116)	395 (76)		

Figure 5. Metals for pre- and 1-week and 6-month post construction. Blue is pre-construction, orange 1-week post construction, yellow 6-month post construction. The grey dotted line represents the drinking water standard for either EPA or WHO which ever was the smaller of the two standards for the given chemical. The legend in A applies to B-H. F and H are enlarged versions of E and H respectively.

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Appendix C.

Lake Macatawa Water Quality Dashboard

2019

Prepared: January 2020

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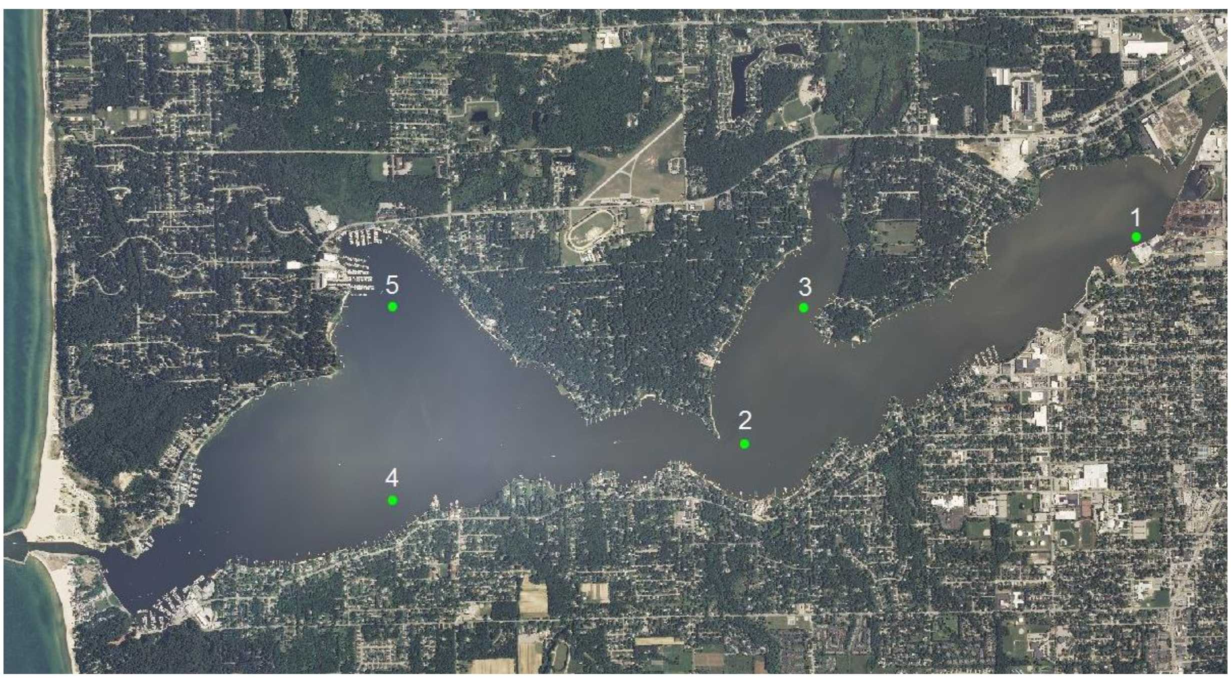
Introduction

As part of Project Clarity, Grand Valley State University's Annis Water Resources Institute (AWRI) established a monitoring program on Lake Macatawa in 2013. The goal of the monitoring program is to evaluate and document the progress toward achieving Project Clarity's goal of improved water quality in Lake Macatawa. The monitoring program involves sampling the lake 3 times per year for a suite of biological, physical, and chemical parameters. Hence, information is not continuous and may reflect episodic, short-term conditions. Focus should be on long-term trends, not short-term events.

Key water quality indicators were selected from the many parameters that are monitored to create a water quality dashboard for Lake Macatawa (see full annual report for all parameters). The goal of the dashboard is to provide a visual representation of the current status and historical trends in Lake Macatawa water quality, by rating each indicator along a scale from desirable (green) to undesirable (red) conditions. Each scale also includes a category that indicates the water quality goal for the lake is being met (yellow). The indicators that were chosen are commonly used to assess lake health: total phosphorus concentration, chlorophyll *a* concentration, and Secchi disk depth (water clarity). Each indicator is described in more detail below.

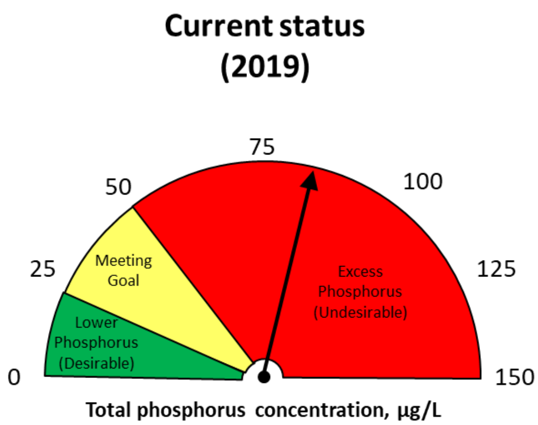
Historical data are included in the dashboard to facilitate comparison of current findings with past status of the selected water quality indicators. Sources for historical data include U.S. EPA (1972; STORET), Michigan Department of Environmental Quality (1982-2012; S. Holden, personal communication), and AWRI (since 2013). All current and historical data shown represent the annual average value of an indicator across Sites 1 (east basin), 2 (central basin), and 4 (west basin; see map below).

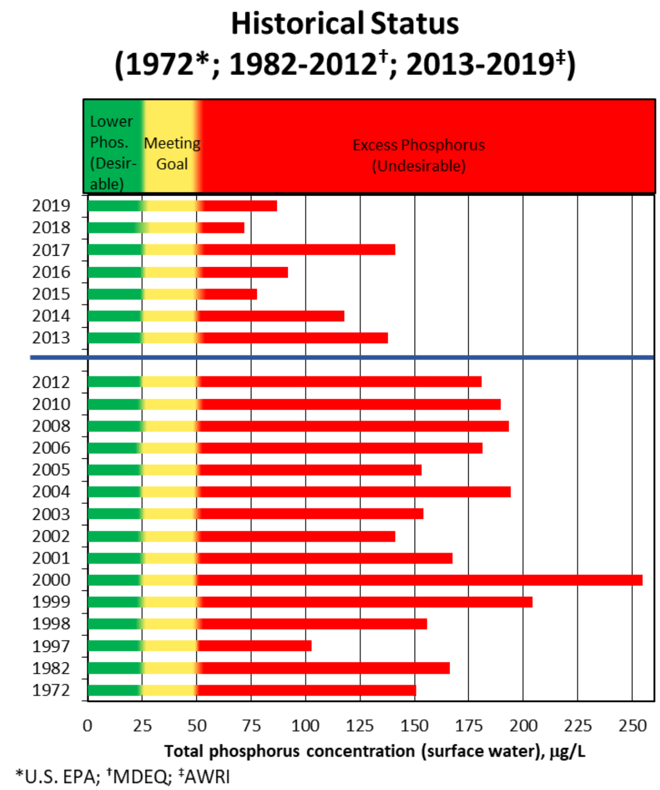


Map of Lake Macatawa showing the 5 sampling locations (green dots) for long-term water quality monitoring. Dashboard indicators were calculated based on data from Sites 1, 2, and 4.

Total Phosphorus

2019 Mean Concentration: 87 µg/L
Target Concentration: 50 µg/L





Phosphorus (P) is an essential element for living organisms. In many freshwater systems, P is the element that limits algal growth. However, when it becomes too abundant, it can help stimulate undesirable algal blooms. Phosphorus comes in many forms; we selected Total Phosphorus (TP) as the dashboard indicator because it includes all the forms of P in the lake (i.e., particulate and dissolved).

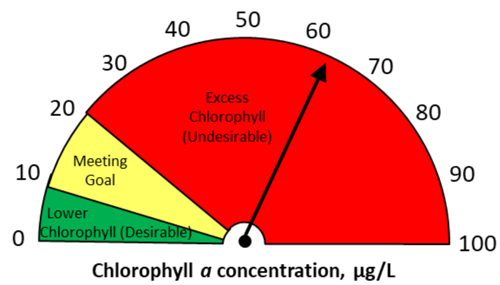
Lake Macatawa has a history of extremely high TP concentrations (i.e., > 100 µg/L), placing it in the "hypereutrophic" trophic state. As a result of this nutrient enrichment, the State of Michigan has established an interim target TP concentration of 50 µg/L in Lake Macatawa. Thus, the TP dashboard shows the water quality goal as being met when TP concentrations are < 50 µg/L. While attaining this goal would be a significant improvement in water quality from current conditions, Lake Macatawa would still be in an impaired "eutrophic" state, which we define as TP concentration > 24 µg/L. Therefore, the TP dashboard shows the ultimate desired TP concentration as < 24 µg/L.

The current status for the total phosphorus indicator is **Undesirable**, meaning that the average TP concentration in 2019 exceeded the water quality goal. Some annual variation in TP concentration should be expected and although the mean 2019 TP concentration is greater than that measured in 2018, the overall trend shows a decline since the start of Project Clarity.

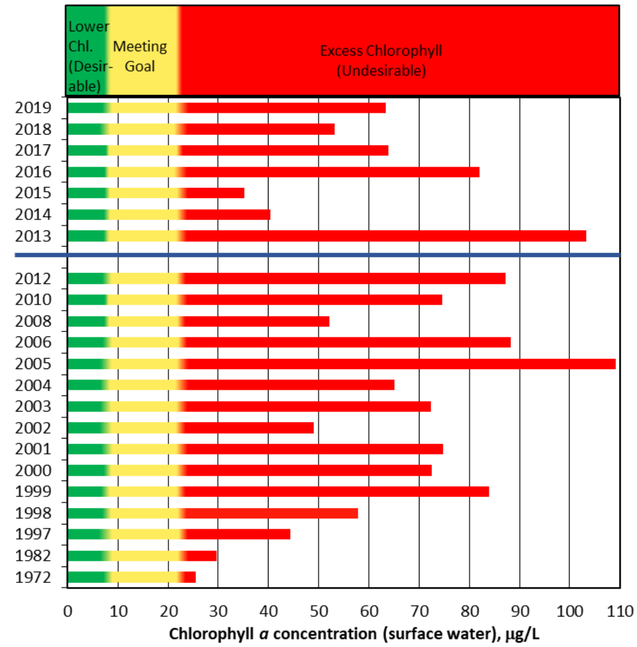
Chlorophyll *a*

2019 Mean Concentration: 63 µg/L
Target Concentration: 22 µg/L

Current status (2019)



Historical Status (1972*; 1982-2012†; 2013-2019‡)



*U.S. EPA; †MDEQ; ‡AWRI

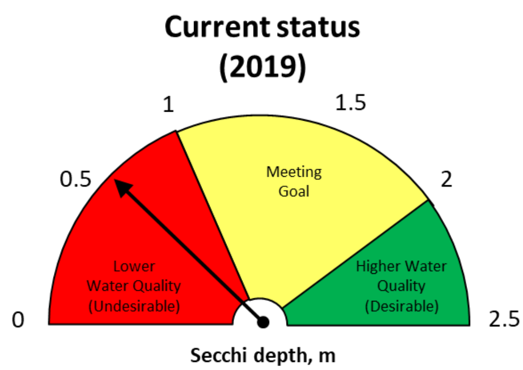
Chlorophyll *a* is the green pigment found in photosynthetic plants and algae. Measuring chlorophyll *a* is a relatively simple way to estimate the amount of algal biomass present in lake water, although it has some limitations. First, chlorophyll *a* does not provide information on whether or not the algae present produce toxins. Second, chlorophyll concentrations can change depending upon environmental conditions, such as light or nutrient level. Finally, chlorophyll *a* concentrations may be low due to very active predation by grazers (zooplankton), so the measurement may give an underestimate of how much algal biomass would otherwise be present.

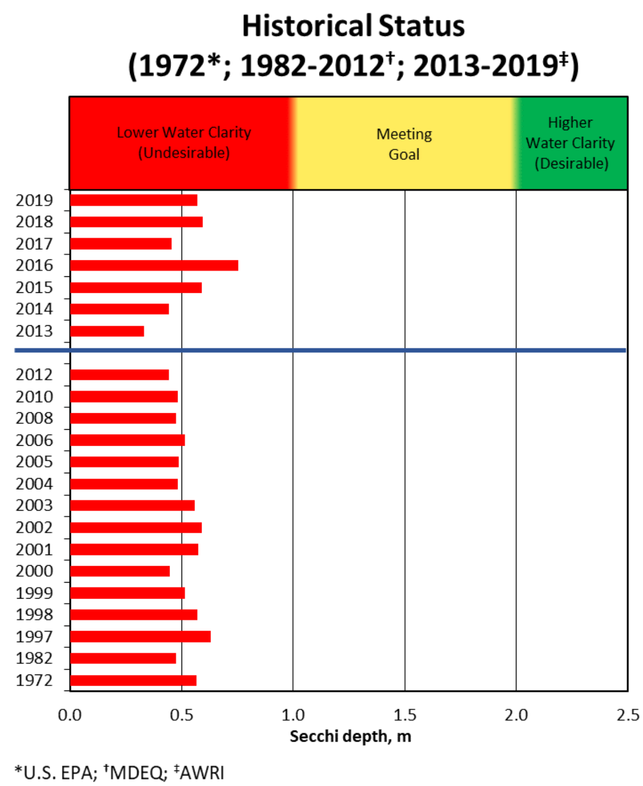
Lake Macatawa has a history of excess algal biomass and high chlorophyll *a* concentrations, typically exceeding the "hypereutrophic" threshold commonly used by MDEQ (22 µg/L) in its assessments of the lake. The chlorophyll *a* dashboard shows that the concentration will meet the water quality goal once it is < 22 µg/L. Although meeting the chlorophyll *a* goal would be a significant improvement in water quality, Lake Macatawa would still be categorized as "eutrophic" (i.e., > 7 µg/L chlorophyll *a*). Thus, the chlorophyll *a* dashboard shows that the ultimate desired chlorophyll *a* concentration is < 7 µg/L.

The current status for the chlorophyll *a* indicator is **Undesirable**, meaning that the average chlorophyll *a* concentration in 2019 exceeded the water quality goal. The 2019 mean chlorophyll concentration continues a trend of relatively high chlorophyll since the lower values in 2014 and 2015.

Secchi Disk Depth (Water Clarity)

2019 Mean Depth: 0.57 m (~1.9 ft)
Target Depth: 1 m (~3.3 ft)



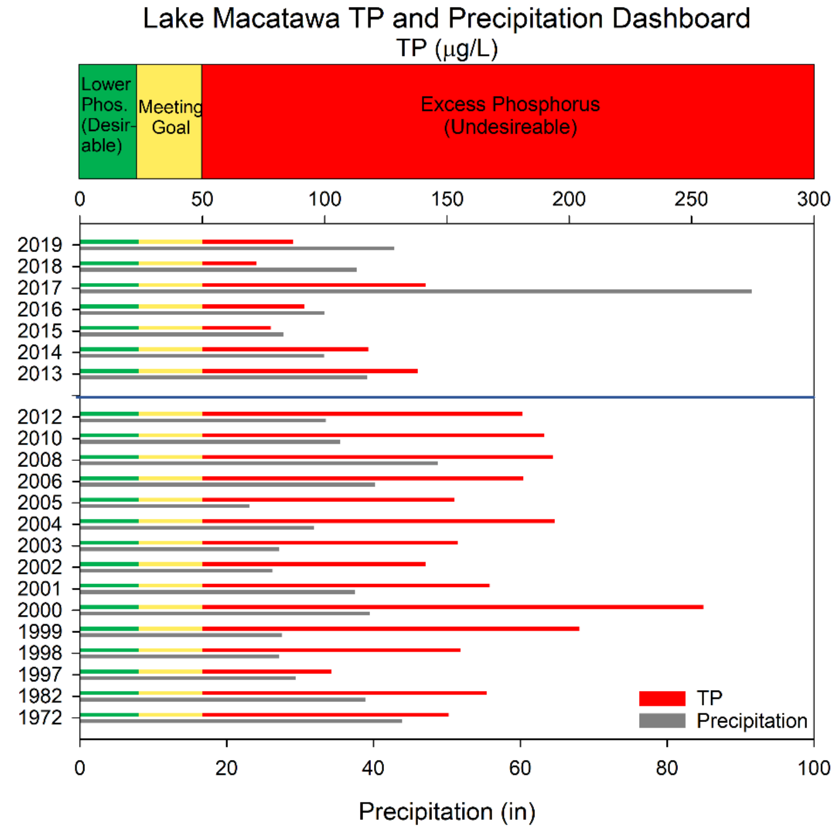


Secchi disk depth is an estimate of water clarity. It is measured using a standard black and white disk, named after Angelo Secchi, who first used an all-white disk for marine waters in 1865. Lake ecologists modified it to black and white in the late 1800s. The Secchi disk is a simple and easy way to measure water clarity, although if waters are cloudy, the disk depth tells you nothing about why the lake is turbid (e.g., is it due to suspended algae or suspended sediment?).

Along with excess phosphorus and chlorophyll *a* concentrations, Secchi depths have historically reflected extremely impaired conditions in Lake Macatawa. Oligotrophic lakes, such as Lake Tahoe, have Secchi disk depths down to 21 m (~70 ft) or deeper. Conversely, hypereutrophic lakes, such as Lake Macatawa, typically have Secchi depths shallower than 1 m (~3 ft). The water clarity goal for Lake Macatawa is modest, with a Secchi depth > 1 m. Because Secchi depths between 1 and 2 m are indicative of a eutrophic state, a desirable Secchi depth is > 2 m.

The current status for the Secchi depth indicator is **Undesirable**, meaning that the average Secchi depth in 2019 was shallower (i.e., less clear) than the water quality goal. Secchi depth in 2019 remains well within the variability seen in recent years.

Total Phosphorus and Precipitation



Phosphorus concentrations in Lake Macatawa are influenced by many variables, but one of the most significant is precipitation because rain and snow events create runoff from farms and urban areas, when phosphorus can be transported to Lake Macatawa either in the dissolved form or as attached to sediment particles; precipitation also results in atmospheric deposition, which can contribute phosphorus directly to the lake and landscape. As a consequence, it is of interest to know if annual changes in lake phosphorus concentrations are related to precipitation.

To answer this question, we examined total phosphorus (TP) concentrations in the lake, based on data from MDEQ and AWRI (sampled 3× per year at 3 sites), and compared them to precipitation data from the Tulip City Airport in Holland. As seen above, between 1972 and 2019, the relationship between precipitation and TP concentration in the lake is not directly related; for example, some years have very high TP concentrations but relatively low precipitation (e.g., 2000 and 2004), whereas other years have modest levels of TP but relatively high precipitation (e.g., 2017). Indeed, the statistical relationship between the two is significant (R^2 = 0.001; p = 0.893).

Interestingly, the relationship between TP and precipitation is much improved since 2013 (R^2 = 0.364; p = 0.152) but is still not statistically significant. This relationship is based on only 7 data points, so it should be viewed cautiously. We view these data as appropriate only for screening purposes, as the TP concentrations are means of seasonal lake sampling events, which likely miss pulses of high P concentrations after storm events.