PROJECT CLARITY
2020 Annual Monitoring Report
(Dec. 2019 – Nov. 2020)

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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 has made 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 recently, monthly average TP concentrations often exceeded 125 µg/L, and at times exceeded 200 µg/L (Holden 2014). However, annual mean TP concentrations of less than 100 µg/L have been observed in both 2018 and 2019 (Hassett et al. 2020). Nonetheless, even these concentrations are excessive, and meeting the TMDL target remains a major challenge in the Macatawa watershed (Steinman et al. 2018). The TMDL estimated that a 72% reduction in phosphorus loads from the watershed would be required to meet the TP concentration target (Walterhouse 1999). Remediation projects and BMPs are 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, initiated a long-term monitoring program in the Lake Macatawa watershed in 2013. 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 2020, in combination with data reported previously from 2013-2019. As noted in last year's report, we terminated sampling upstream and downstream of the restored wetlands in April 2019 given the limited value of the information provided, and we replaced that monitoring with sampling sediment within the two restored wetlands to determine how much, and in what form, the 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). This report also contains several Appendices, covering additional information generated by AWRI that is related to the monitoring program (A: fish monitoring in Lake Macatawa; B: iron slag filter study; C: the Lake Macatawa dashboard; and D: wetland monitoring).

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.

2.2 Lake Macatawa: Long-Term Monitoring

Water quality monitoring in the lake was conducted at 5 sites during spring, summer, and fall 2020 (Table 1, Fig. 1). Although spring sampling historically occurs in early to mid-May, 2020 spring sampling was delayed until mid-June due to COVID-19. The sampling sites correspond with Michigan Department of Environment, Great Lakes & Energy (EGLE formerly MDEQ) 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.

Water for SRP and NO_3^- analyses was syringe-filtered through 0.45-µm membrane filters into scintillation vials; SRP was refrigerated at 4°C and NO_3^- frozen until analysis. NH_3 and TKN were acidified with sulfuric acid and kept at 4°C until analysis. SRP, TP, NH_3, NO_3^-, and TKN were analyzed on a SEAL AQ2 discrete automated analyzer (U.S. EPA 1993). 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.

Mann-Whitney rank sum tests were used to detect significant differences in pre- and post-restoration distributions of SRP, TP, and chl *a*. An equal number (n=40) of seasonally corresponding data points from all pre-restoration (summer 2013 – fall 2015) and the most recent post-restoration (summer 2018 – fall 2020) sampling events were incorporated in the rank sum test, pooling data from all sites (1-5). Statistical significance was set with $\alpha = 0.05$ and testing was performed in SigmaPlot v.14.0 (Systat Software, Inc.).

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 criteria.

Table 1. Location and 2020 water column seasonal mean depth at Lake Macatawa long-term monitoring locations.

Site	Latitude	Longitude	Depth (m)
1	42.7913	-86.1194	8.8
2	42.7788	-86.1525	5.9
3	42.7872	-86.1474	4.4
4	42.7755	-86.1822	10.9
5	42.7875	-86.1820	5.2

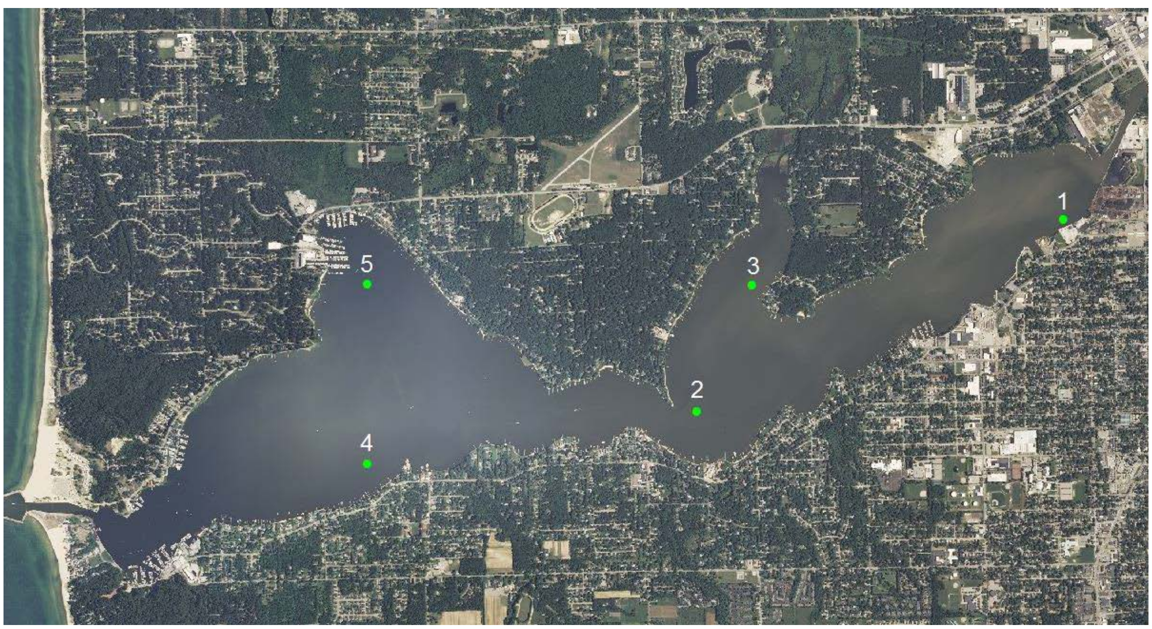

Figure 1. Map of Lake Macatawa showing the 5 sampling locations (green dots) for long-term water quality monitoring.

2.3 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. In addition, atmospheric deposition 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 with 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 P and precipitation amount were conducted in Microsoft Excel.

3. Results and Discussion

3.1 Sampling Year 2020

Spring sampling was delayed from May to June 2020 due to COVID-19. Perhaps related to that delay, Lake Macatawa's water column temperature and DO showed evidence of incipient stratification in spring, with higher DO and temperature values at the surface than at the bottom (Table 2). As expected, there was evidence of strong stratification in summer but the fall sampling indicated the lake had turned over based on the relatively uniform DO and temperature values throughout the water column (Table 2). Summer DO at the near bottom sampling depths at both Site 1 (nearest to the Macatawa River mouth: 0.81 mg/L) and site 4 (nearest to the Lake Michigan channel: 0.86 mg/L) were among the lowest measured by AWRI in Lake Macatawa (Fig. 2). A LOWESS (locally weighted scatterplot smoothing) trend analysis of summer bottom DO at sites throughout our monitoring period of record shows the 2016-2019 increase of bottom DO concentrations ended and decreased in 2020; the fit of this analysis improves when considering only sites 1, 2, and 4 (3-site R^2 = 0.61; Fig. 2A) in the main flow of Lake Macatawa and not also including sites 3 and 5 in the bays (5-site R^2 = 0.18; Fig. 2B).

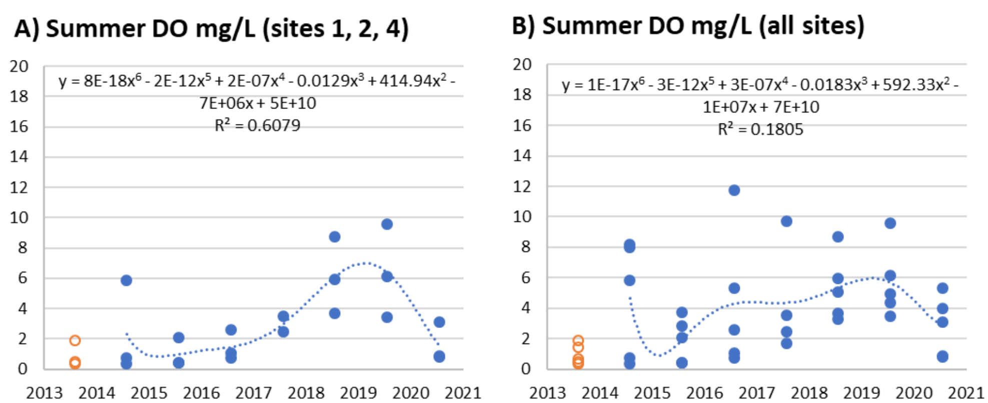

Figure 2. Locally weighted scatterplot smoothing (LOWESS) trend analyses of summer DO site data from Lake Macatawa. Only the most recent 7 years of data (2014-2020) are used for each seasonal LOWESS due to regression-fitting software limitations; excluded data (2013) are noted as hollow circle symbols. Sites 1, 2, and 4 (panel A) represent the main flow of Lake Macatawa via the Macatawa River watershed and additional sites (panel B) incorporate conditions in the lake's northern Big Bay and Pine Creek Bay.

Surface water SRP was highest in spring at site 1 (21 µg/L) and at near-bottom depth in summer at sites 1 and 4 (28 and 33 µg/L; Fig. 3A,B). Measured TP concentrations often exceeded the 50 µg/L TMDL recommendation at surface depth and showed a slightly increasing seasonal trend, generally ranging 47-112 µg/L (Table 3, Fig. 3C). The seasonal increase was stronger for Bottom TP, with spring ranging 15-42 μg/L and fall ranging 55-127 μg/L (Fig. 3D). Notably, spring bottom TP concentrations were generally below the TMDL recommendation, averaging 38 μg/L (Table 3, Fig. 3D). For both SRP and TP, site 1 (closest to Macatawa River mouth) had the highest concentrations among all sites, depths, and seasons (Fig. 3A-D), revealing: 1) watershed runoff remains the major source of P to Lake Macatawa; 2) the lake itself is responsible for at least some P retention (presumably due to sedimentation of both biotic and biotic material) as water flows to Lake Michigan; and 3) inflow from Lake Michigan helps dilute the P concentrations in Lake Macatawa, especially at the westernmost sampling sites, similar to observations in Muskegon Lake (Liu et al. 2018).

Chl *a* concentrations exceeded EGLE's hypereutrophic boundary of 22 μg/L and June 2020 was the highest "spring" measurement by AWRI in Project Clarity history with 202 μg/L at site 1 (Table 3, Fig. 3E). While chlorophyll bloom concentrations of this magnitude have occurred previously in project sampling history, they have been rare and generally occurred in summer (July) sampling. Again, the late "spring" sampling in 2020 due to COVID-19 resulted in warmer temperatures (Table 2) than during our usual spring sampling, which in combination with the high P and N concentrations, contributed to this bloom. Photosynthetic activity from the algae undoubtedly accounted for the high surface DO concentrations (Table 2), which were well above DO saturation.

Table 2. Lake-wide means (1 SD) of select general water quality parameters recorded during 2020 monitoring year. Within 2020, "n" is the number of lake sites composing the seasonal mean at each depth. Data are shaded for readability. Dates of sampling events: 6/19/2020; 7/23/2020; 10/20/2020. Total dissolved solids (TDS) sensor was inoperative during Spring 2020 sampling.

Season	Depth	n	Temp. (°C)	DO (mg/L)	SpCond (μS/cm)	TDS (g/L)	Turbidity (NTU)
Spring	Surface	5	24.46 (0.76)	16.51 (1.62)	451 (26)	ND (NA)	14.2 (9.8)
	Middle	5	20.64 (2.69)	8.80 (4.23)	445 (40)	ND (NA)	8.3 (2.8)
	Bottom	5	16.97 (3.35)	5.27 (2.61)	392 (58)	ND (NA)	13.3 (5.1)
Summer	Surface	5	25.79 (0.56)	10.53 (0.88)	462 (53)	0.300 (0.035)	8.7 (2.7)
	Middle	5	24.38 (1.07)	7.42 (2.80)	440 (47)	0.286 (0.031)	8.1 (3.3)
	Bottom	5	20.43 (3.98)	2.81 (1.97)	401 (43)	0.261 (0.027)	10.6 (2.0)
Fall	Surface	5	11.60 (0.24)	9.56 (0.30)	500 (77)	0.325 (0.050)	101.3 (60.9)
	Middle	5	11.47 (0.18)	8.81 (0.72)	502 (78)	0.326 (0.050)	147.5 (95.4)
	Bottom	5	11.44 (0.18)	8.42 (0.83)	505 (80)	0.328 (0.052)	173.5 (83.1)

NO_3^- and TKN started high in spring and generally decreased throughout 2020, likely associated with spring fertilizer application of nitrogen (Neilsen et al. 1982). NH_3 concentrations had the opposite increasing trend and were highest in Fall (Table 3, Fig. 4).

Microcystin concentrations were tested at all seasons, sites, and depths and were two or more degrees of magnitude below World Health Organization and Environmental Protection Agency guidelines for recreational waters (respectively 20 μg/L and 2 μg/L). Microcystin concentrations remained <0.1 μg/L throughout 2020 sampling and were seasonally lower in 2020 than in 2019 except for the Fall 2020, especially site 4 (near Lake Michigan Channel) at the near bottom depth, which peaked at 0.09 μg/L compared to 0.02 ug/L in 2019. However, these concentrations remain well below regulatory guidelines.

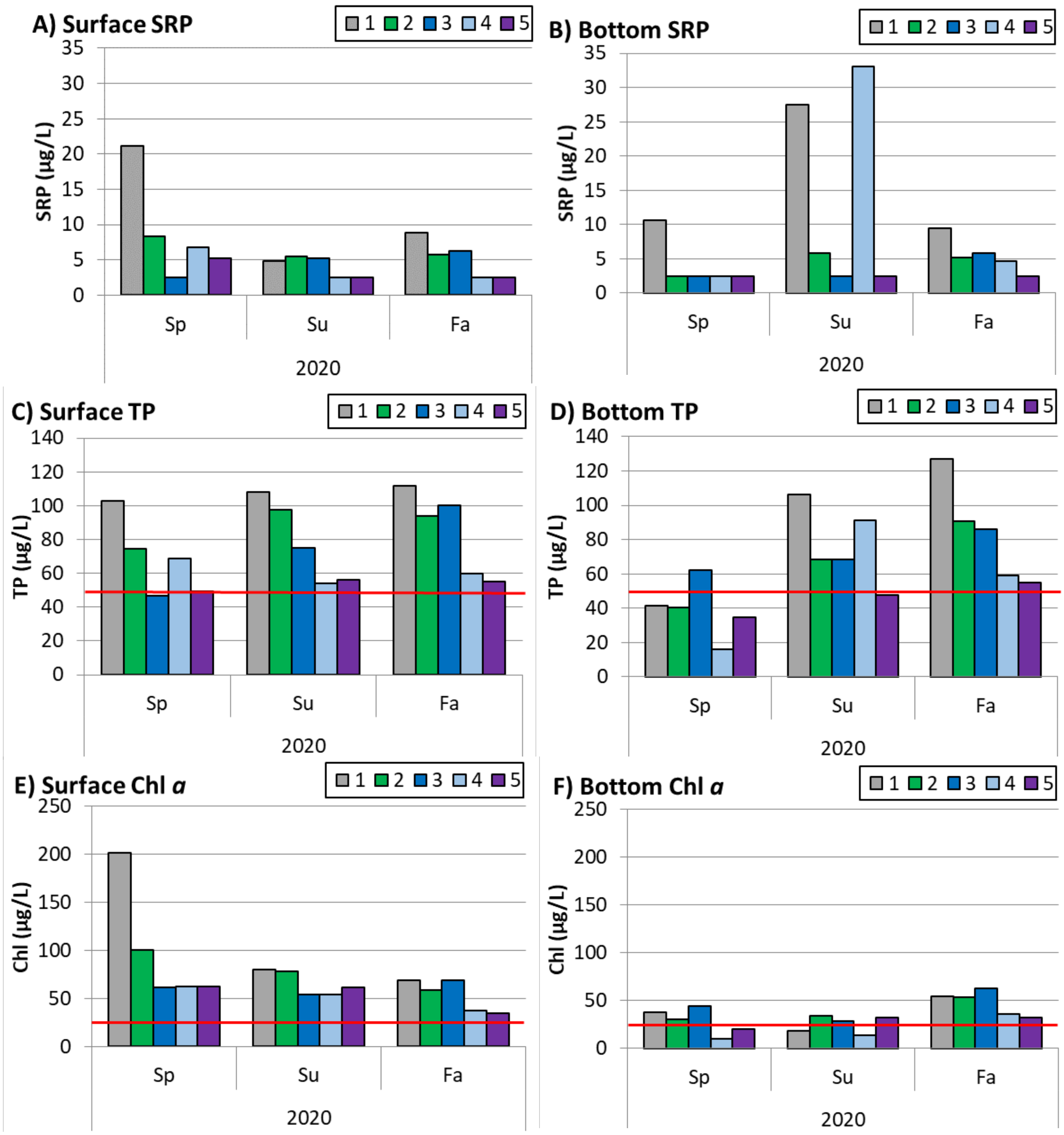

Figure 3. Soluble reactive phosphorus ([SRP]: A, B); total phosphorus ([TP]: C, D); and chlorophyll *a* ([chl *a*]: E, F) concentrations measured at the 5 monitoring stations in Lake Macatawa during 2020. The red horizontal lines on TP figures (C, D) indicate the interim total maximum daily load (TMDL) 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 EGLE for assessing chl *a* in Lake Macatawa (Holden 2014). Note scales change on y-axes. COVID-19 delayed Spring 2020 sampling from May until June.

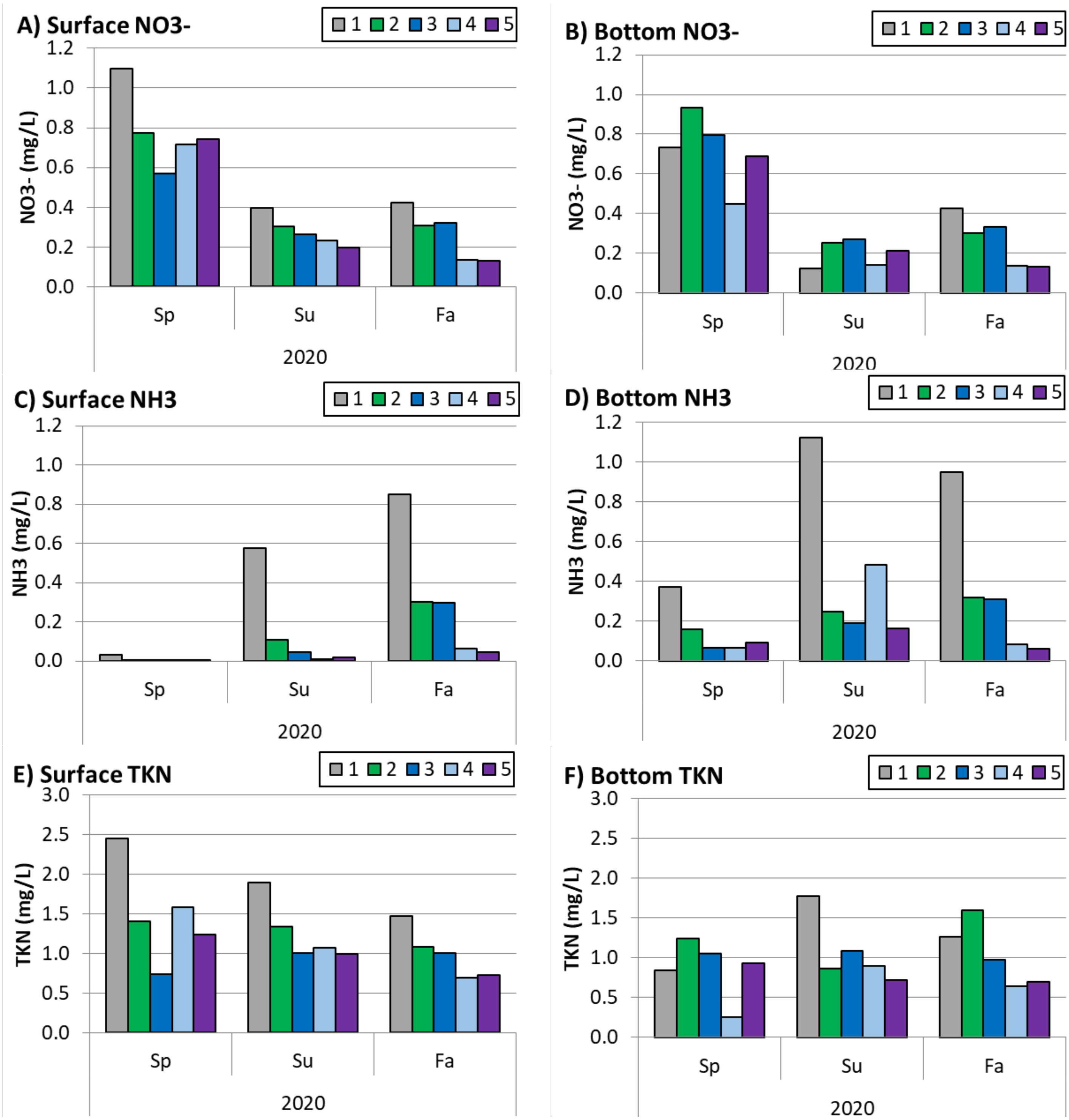

Figure 4. Nitrate ([NO_3^-]: A, B); ammonia ([NH_3]: C, D); and Total Kjeldahl Nitrogen ([TKN]: E, F) concentrations measured at the 5 monitoring stations in Lake Macatawa during 2020. Note scales change on y-axes. COVID-19 delayed Spring 2020 sampling from May until June.

Table 3. 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 2020 monitoring year. Within 2020, "n" is the number of lake sites composing the seasonal mean at each depth. Data are shaded for readability. See Table 2 for dates of sampling events. Note different units for the analytes.

Season	Depth	n	SRP (µg/L)	TP (µg/L)	NO_3^- (mg/L)	NH_3 (mg/L)	TKN (mg/L)	ext. Chl (µg/L)	Secchi Depth (m)
Spring	Surface	5	9 (7)	68 (23)	0.78 (0.19)	0.01 (0.01)	1.49 (0.63)	98 (61)	0.7 (0.1)
	Bottom	5	4 (4)	39 (17)	0.72 (0.18)	0.15 (0.13)	0.86 (0.37)	29 (13)	
Summer	Surface	5	4 (1)	78 (24)	0.28 (0.08)	0.15 (0.24)	1.26 (0.38)	66 (13)	0.6 (0.1)
	Bottom	5	14 (15)	76 (23)	0.20 (0.07)	0.44 (0.40)	1.07 (0.41)	25 (9)	
Fall	Surface	5	5 (3)	84 (25)	0.26 (0.13)	0.31 (0.32)	1.00 (0.32)	54 (17)	0.6 (0.2)
	Bottom	5	6 (3)	84 (29)	0.27 (0.13)	0.34 (0.36)	1.03 (0.40)	48 (13)	

3.2 Pre- vs. Post-Restoration Comparison

As noted in prior reports, it is likely that it will take a considerable period of time before lake water quality responds on a consistent basis to actions taken in the watershed. This is because lakes have a built-in resistance to change (cf. Abell et al. 2020), which is influenced by: 1) the lake's hydraulic residence time (those with shorter residence times respond faster); 2) the quantity, quality, and location of implemented management interventions in the watershed; and 3) the importance of internal nutrient loading in the lake. Given that watershed-based management changes are both recent and at a relatively small scale in the Macatawa watershed, it is not expected that Lake Macatawa water quality will respond quickly. Nonetheless, it is important to begin measuring change to establish baselines and evaluate trends.

Unexpectedly, both surface and bottom SRP concentrations in Lake Macatawa were significantly higher post-restoration compared to pre-restoration (Fig. 7). This analysis excludes 2016 and 2017, the years immediately following major restoration construction activities (Fig. 5A,B), which may have resulted in greater release of SRP. Research in the western basin of Lake Erie has shown that implementation of modified tillage practices to reduce sediment erosion has resulted in increased loading of soluble phosphorus (Jarvie et al. 2017), so a similar phenomenon may be occurring in the Macatawa watershed. However, given the limited sample size of this analysis (n=5 per seasonal sampling event and n=40 total per restoration period), results should be viewed with caution.

Post-restoration summer and fall TP concentrations follow a trend seen in recent years (2018-present) of being lower than during pre-restoration (Table 4, Fig. 5C,D). Unlike SRP, post-restoration TP concentrations had a significantly lower range of values than pre-restoration samples and this was seen at both sampling depths ($P=0.003$ and $P<0.001$; Fig. 7C,D); this suggests that sediment transport may be declining in recent years, as much of the TP is associated with sediment. A decline in sediment would result in increased water clarity and a concomitant increase in Secchi depth, as was observed ($P < 0.001$; Figs. 4G, 6G).

We started measuring nitrogen concentrations in Lake Macatawa in 2017, after experiments revealed that the algae in the lake were co-limited by phosphorus and nitrogen (Steinman et al. 2016). As a consequence, we do not have pre- vs. post-restoration comparison data for N. However, since we began measuring different forms of N in 2017, several trends are apparent: 1) nitrate concentrations are always lowest in summer, irrespective of site (Fig. 6A,B); this is likely due to rapid uptake by phytoplankton, given their co-limitation with nitrogen during this high-productivity period; and 2) both forms of inorganic N (nitrate and ammonia), as well as TKN, exhibit a strong spatial distribution, with highest concentrations at site 1 and lowest concentrations at site 5 (Fig. 6), reflecting the strong inputs from the watershed.

Chl *a* at the surface has increased during the post-restoration period based on our limited sampling (P=0.05; Table 4, Figs. 5E, 7E), although no differences were detected in the near bottom samples (P=0.573; Figs. 5F, 7F). The increase in surface chlorophyll is consistent with a reduction in sediment loading, as greater light transmittance in the water column will allow more photosynthesis and algal growth. Of course, as algae bloom, they eventually will reduce water clarity due to the abundance of their own cells in the water column, but until that point, water transparency will improve.

Table 4. 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 2020). Data are color coded for readability. ND = no data.

Season	Depth	Period	n	SRP (ug/L)	TP (ug/L)	NO_3^- (mg/L)	NH_3 (mg/L)	TKN (mg/L)	ext. Chl (ug/L)	Secchi Depth (m)
Spring	Surface	Pre	2	3 (0)	66 (4)	ND	ND	ND	25 (4)	0.6 (0.1)
		Post	5	16 (23)	109 (64)	1.43 (0.52)	0.21 (0.27)	1.66 (0.35)	63 (29)	0.6 (0.3)
	Bottom	Pre	2	3 (1)	98 (30)	ND	ND	ND	24 (3)	
		Post	5	15 (22)	108 (67)	1.40 (0.49)	0.37 (0.17)	1.52 (0.59)	41 (16)	
Summer	Surface	Pre	3	6 (3)	110 (66)	ND	ND	ND	67 (39)	0.4 (0.1)
		Post	5	6 (3)	73 (22)	0.25 (0.05)	0.22 (0.09)	1.33 (0.11)	68 (26)	0.7 (0.1)
	Bottom	Pre	3	17 (18)	107 (49)	ND	ND	ND	32 (13)	
		Post	5	11 (4)	85 (23)	0.28 (0.10)	0.44 (0.10)	1.28 (0.16)	34 (8)	
Fall	Surface	Pre	3	10 (12)	134 (23)	ND	ND	ND	63 (43)	0.4 (0.1)
		Post	5	8 (6)	78 (8)	0.93 (0.84)	0.40 (0.26)	1.35 (0.31)	57 (27)	0.5 (0.1)
	Bottom	Pre	3	11 (13)	158 (19)	ND	ND	ND	61 (35)	
		Post	5	10 (6)	84 (3)	1.03 (0.95)	0.41 (0.24)	1.34 (0.32)	47 (12)	

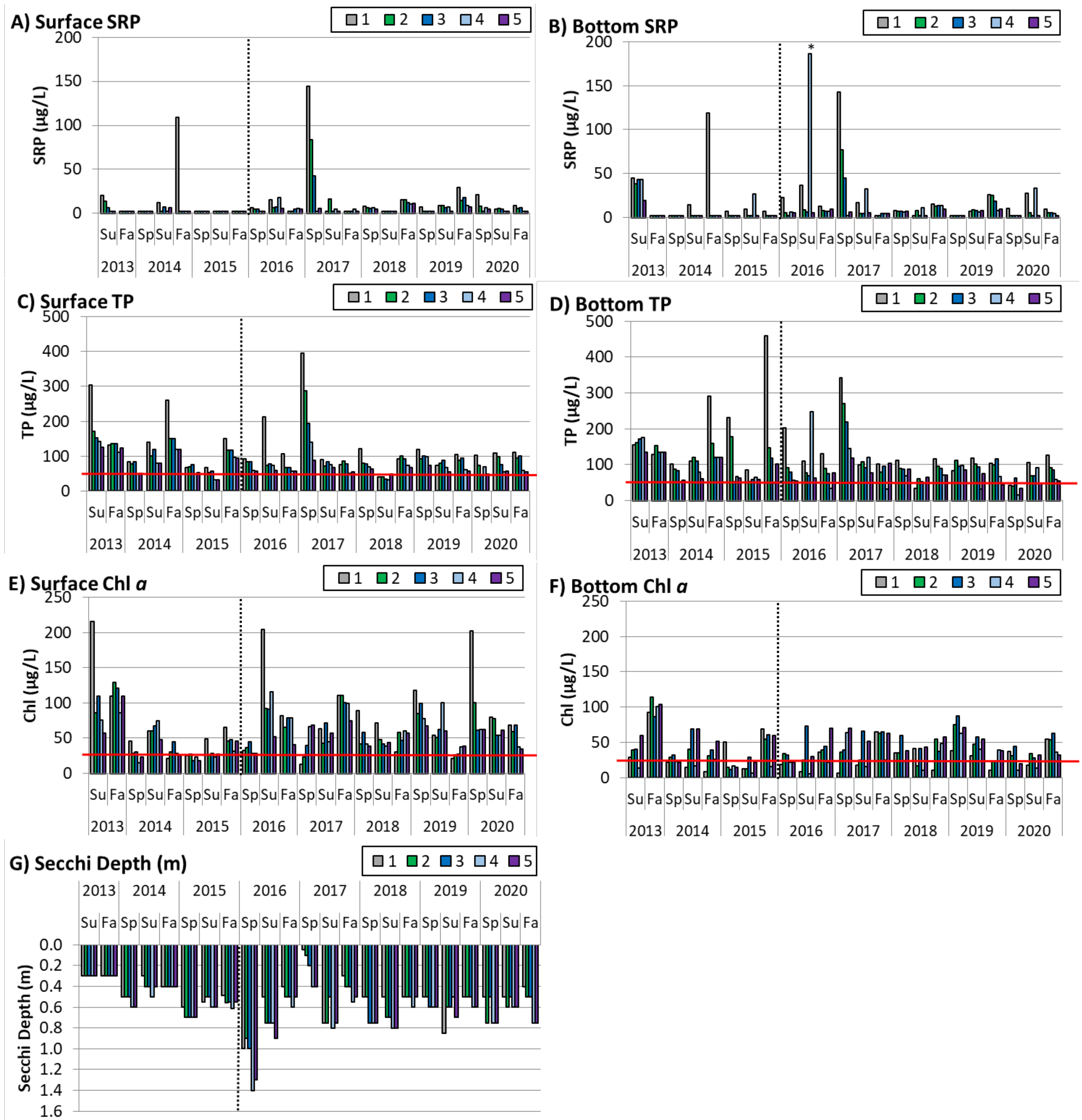

Figure 5. Soluble reactive phosphorus ([SRP]: A, B); total phosphorus ([TP]: C, D); chlorophyll *a* ([chl *a*]: E, F); and Secchi disk depth: G) levels measured at the 5 monitoring stations in Lake Macatawa from 2013 through 2020. The red horizontal lines on TP figures (C, D) indicate the interim total 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 EGLE 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. COVID-19 delayed Spring 2020 sampling from May until June.

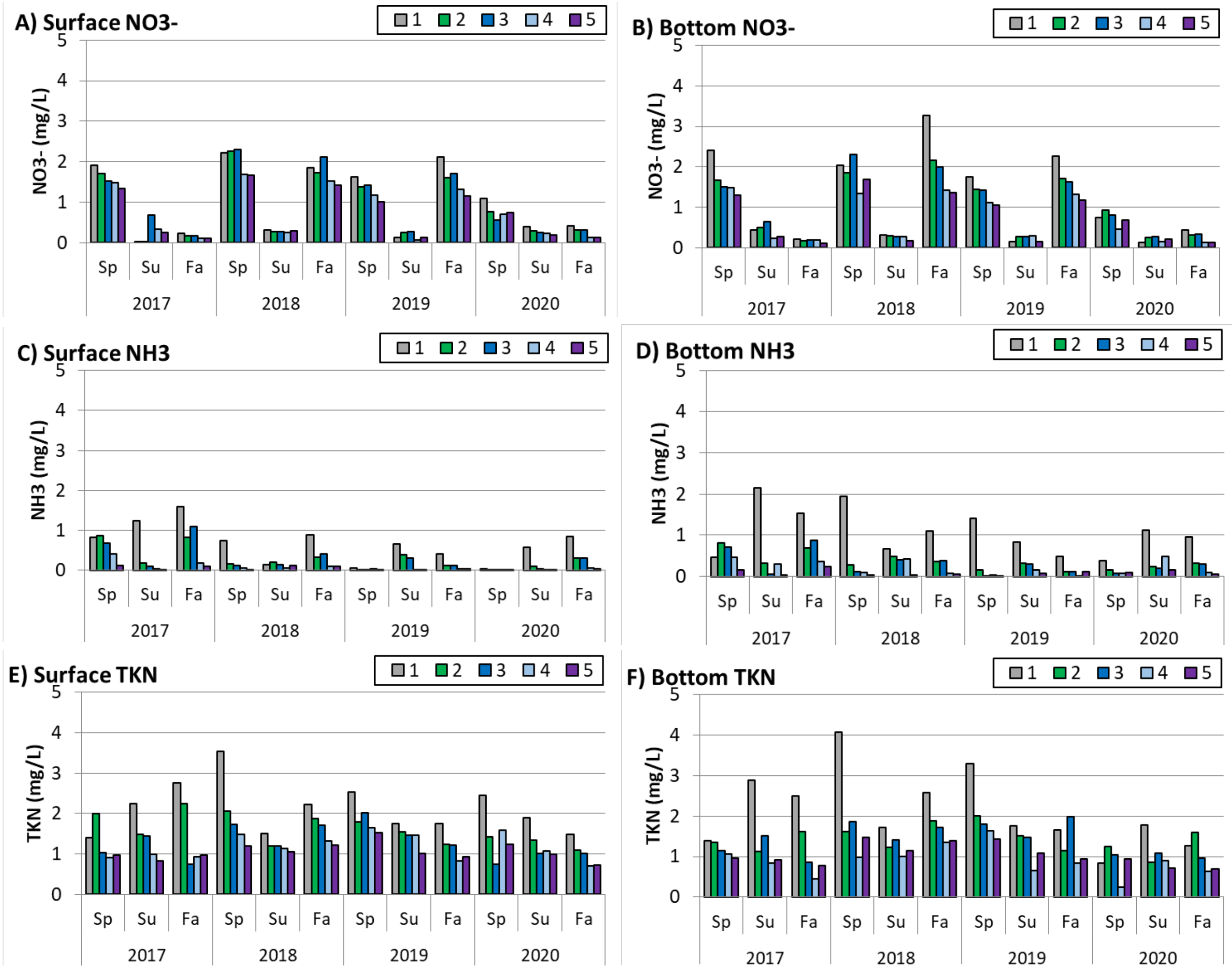

Figure 6. Nitrate ([NO_3^-]: A, B); ammonia ([NH_3]: C, D); and Total Kjeldahl Nitrogen ([TKN]: E, F) concentrations measured at the 5 monitoring stations in Lake Macatawa from 2017 through 2020. Note scales change on y-axes. COVID-19 delayed Spring 2020 sampling from May until June.

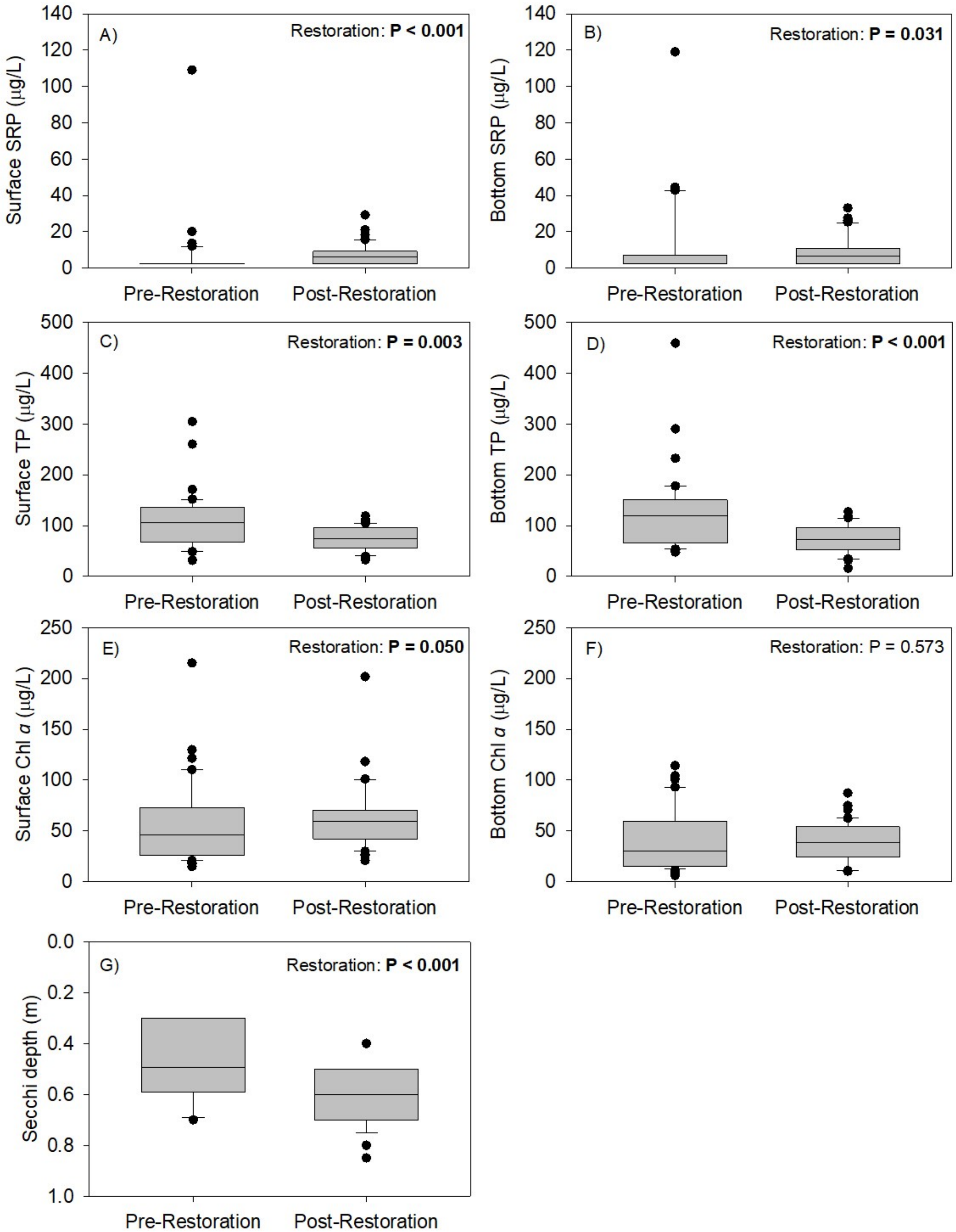

Figure 7. Box plots of soluble reactive phosphorus ([SRP]: A, B); total phosphorus ([TP]: C, D); chlorophyll *a* ([chl *a*]: E, F; and Secchi disk depth: G) levels measured at the 5 monitoring stations in Lake Macatawa during all pre-restoration sampling dates (summer 2013 – fall 2015) and an equal and seasonally corresponding number of post-restoration sampling dates (summer 2018 – fall 2020). Boxes represent the middle 50% of data; the horizontal line crossing the box is the median data value; whiskers represent the upper 25% and lower 25% of data, excluding outliers; points outside of the box and whiskers are considered outliers. P-values are results of Mann-Whitney rank sum tests of pre- vs. post-restoration data. Note scales change on y-axes.

3.3 Lake Macatawa Phosphorus-Precipitation Relationship

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 2020, the relationship between precipitation and TP concentration in the lake was not statistically significant (Fig. 8; $R^2 = 0.002$; 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. 8; $R^2 = 0.382$; $p = 0.102$) but is still not statistically significant and is driven largely by one data point (2017). We view these data as appropriate for screening purposes only, as the TP concentrations are single seasonal 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. Extreme events have become more common over the past 50 years (Dethier et al. 2020) and are predicted to continue increasing with climate warming (Kunkel et al. 2013).

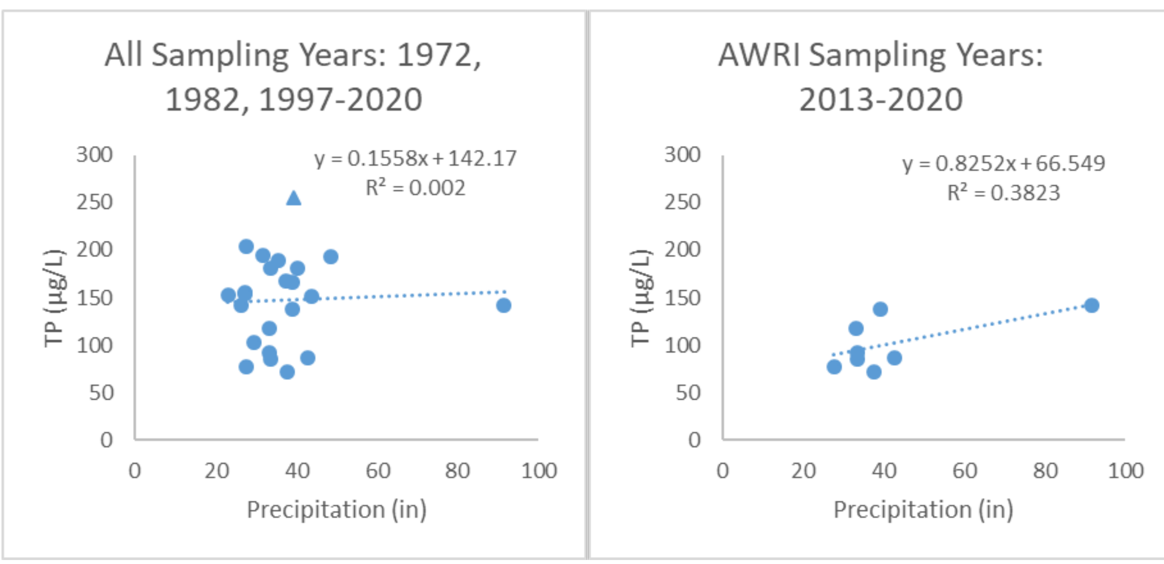

Figure 8. Linear regressions plotting annual precipitation vs. mean total phosphorus (TP) concentration in Lake Macatawa. Historical TP data sources include U.S. EPA (1972; STORET), MDEQ/EGLE (1982, 1997-2012; S. Holden, personal communication), and AWRI (since 2013). Precipitation data sources include the National Climatic Data Center/National Centers for Environmental Information (2005-2020; NOAA) and Weather Underground (1972-2004; The Weather Channel Company).

In order to gauge whether smaller and more closely associated spans of time would have a stronger relationship with seasonally-sampled Lake Macatawa phosphorus data, daily precipitation from 2013-

2020 was summarized via Excel PivotTable into weeks and months of total precipitation prior to each specific sampling date as follows: 1 week, 2w, 3w, 4w (1 month), 2mo, 3mo, 4mo, 5mo, 6mo, 7mo, 8mo, 9mo, 10mo, 11mo, 12mo. Separately for each weekly or monthly summary, total precipitation was analyzed against surface and bottom lake-wide average SRP and TP using linear regression; R^2 values were summarized (Table 5). R^2 values in the regressions were stronger for SRP than for TP, in general (Table 5). Using the more recent antecedent precipitation data (Table 5, cf. Fig. 9), instead of the full antecedent calendar year (cf. Fig. 8), may be a better indicator of the precipitation-P relationship for Lake Macatawa.

Table 5. Summaries of linear regressions comparing total precipitation against average (dashboard average: sites 1, 2, 4 only) Lake Macatawa soluble reactive phosphorus (SRP) and total phosphorus (TP); increasing periods of time summing total precipitation reflect the days immediately prior to each AWRI Lake Macatawa sampling day. Regression R^2 values are color coded to indicate the strongest relationships in green and peak values are indicated with bold text.

Linear Regression R^2 values				
#days	SRP		TP	
total PPTN	Surface	Bottom	Surface	Bottom
1-day	0.04	0.09	**0.36**	**0.55**
7-day	0.04	0.01	0.00	0.00
14-day	0.21	0.18	0.00	0.08
21-day	0.11	0.05	0.00	0.08
28-day	0.38	0.28	0.00	0.01
2-month	**0.47**	0.32	0.02	0.15
3-month	0.36	0.39	0.06	0.25
4-month	0.28	0.48	0.08	0.26
5-month	0.39	**0.54**	0.00	0.09
6-month	0.46	0.39	0.00	0.03
7-month	0.36	0.28	0.03	0.01
8-month	0.41	0.25	0.04	0.01
9-month	0.45	0.15	0.01	0.07
10-month	0.23	0.12	0.02	0.27
11-month	0.15	0.14	0.04	0.29
12-month	0.14	0.14	0.05	0.30

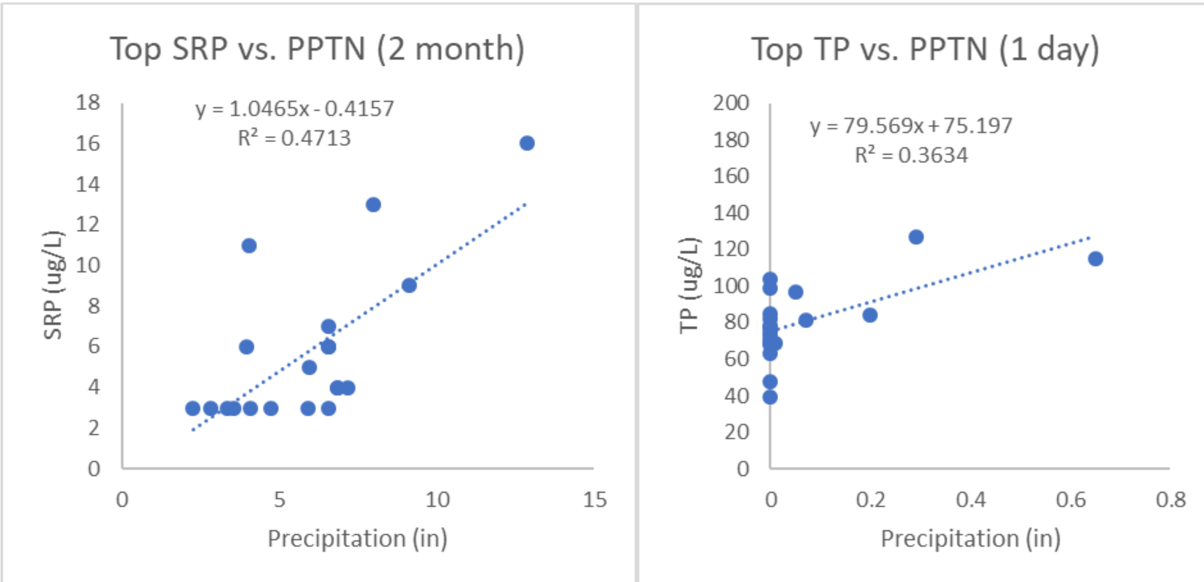

Figure 9. Linear regressions plotting annual precipitation vs. mean soluble reactive phosphorus (SRP) and total phosphorus (TP) concentrations in Lake Macatawa. Surface SRP and TP data are lake-wide means of seasonal 2013-2020 AWRI sampling events. Precipitation data sources were provided by the National Climatic Data Center/National Centers for Environmental Information (2013-2020; NOAA).

4. Summary

The 2020 water quality in Lake Macatawa was either consistent or slightly improved from the past few years. Although TP concentrations remain substantially above the 50 µg/L TMDL target, post-restoration concentrations have declined significantly compared to pre-restoration values. Of some concern is that dissolved phosphorus values have increased significantly over time; it is unclear if this is related to soil tillage practices in the watershed or changes in plankton usage of dissolved P.

Prior year concerns over high nitrate levels in spring and fall are somewhat allayed based on the 2020 concentrations, which were considerably lower than in the past few years. This may be a reflection of best management practices in the watershed having a beneficial effect with respect to nitrate application and retention. Future monitoring will help us make that determination.

Chlorophyll concentration (indicative of algal biomass) and Secchi disk depth (indicative of water clarity) in 2020 were little changed from the past few years, and have either remained similar (chlorophyll), or have improved (secchi depth), compared to pre-restoration values.

Overall, water quality trends are generally positive, although as noted previously, the current phosphorus and chlorophyll concentrations are well above what should be observed in a "healthy" lake. There will continue to be 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.

The appendices address additional studies AWRI is conducting in the lake and watershed. Fish monitoring (Appendix A), funded through Project Clarity, is in its 7th year, and the current data indicated that the littoral fish assemblage showed both positive and negative indicators of the lake's ecological health. Yellow perch, bluegill, and pumpkinseed were common species captured in our surveys, and they are indicators of good water quality, whereas commonly found gizzard shad and spotfin shiner are often associated with poor water quality.

The iron slag filter project (Appendix B), funded through MDARD, has been effective at reducing P concentrations, but their optimal effectiveness is in regions with very high SRP concentrations in the tile drain effluent. There appears to be no concern over their leaching toxic substances. Given their cost, their impact in reducing P will be localized at high P "hot spots".

The Lake Macatawa Dashboard (Appendix C) provides a visual option for quickly surveying how critical water quality parameters (Total Phosphorus, Chlorophyll *a*, and Water Clarity) are changing over time and responding to restoration efforts in the watershed. As noted in this report, all parameters remain in the undesirable range.

We are also analyzing sediment P retention in the Middle Mac and Haworth restored wetlands (Appendix D), with internal AWRI funding through the Allen and Helen Hunting Innovation Research Fund. Our results suggest that the undisturbed sediments of the Haworth and Middle Macatawa wetlands are retaining their capacity to serve as P sinks. However, there are indications that some areas are close to reaching P saturation, which may convert into P sources to overlying water. Continued monitoring will help determine the P status of these sediments, and whether or not management intervention is recommended to replenish their P binding ability.

Finally, although not included in this report, our work on a computational SWAT model for the Macatawa watershed has restarted with the hiring of Dr. Sean Woznicki, a geospatial ecologist.

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7. Appendices

Appendix A. Long-Term Fish Monitoring of Lake Macatawa.

Appendix B. Iron Slag Filters.

Appendix C. Lake Macatawa Dashboard.

Appendix D. Wetland Sediments.

Appendix A. Long-Term Fish Monitoring of Lake Macatawa.

Long-Term Fish Monitoring of Lake Macatawa: Results from Year 7 (2020)

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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 seventh year (2020) 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). We also begin to assess patterns in the data over time. However, the true value of this fish monitoring effort will come as we continue to accumulate more data so that we can 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). 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. In 2020, high water levels in the Great Lakes made fish sampling challenging; as a result, fyke nets were not fished at site #3 (but all other sampling was completed).

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 14 September 2020 during daylight hours (i.e., between 0900 and 1400) and fished for about 23.2 h (range = 22.6-24.0 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 10 September 2020. 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 104 cm (Table 2). Mean water temperature was similar during fyke netting (18.3 °C; Table 2) and boat electrofishing (19.2 °C; Table 3). At fyke nets, mean percent cover of macrophytes was low at all sites, with values <5% at sites #1, #3, and #4, and 8% at site #2. At boat electrofishing transects, percent macrophyte cover increased across sites from the river mouth (site #1 = 10%, site #2 = 30%) toward Lake Michigan (site #3 = 80%, site #4 = 75%). There may be a trend of increasing percent macrophyte cover over time (Figure 2B). 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 turbidity (i.e., lower turbidity is associated with more macrophytes), with turbidity in 2020 intermediate compared with lower turbidity in 2016-2019 and higher turbidity in 2014-2015 (Figure 3B). 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; Kleindl and Steinman 2021).

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 Muskegon Lake, the drowned river mouth lake for which we have the longest time series of water quality observations (Bhagat and Ruetz 2011; see also https://www.gvsu.edu/wri/director/muskegon-lake-water-quality-dashboard-78.htm). Although there has been a pattern of turbidity and specific conductivity decreasing over time, values in 2020 were greater than 2019 in most instances (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 an overall trend towards improvement (Figure 3). Within the lake itself, there was typically a gradient in specific conductivity and turbidity in most years, 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, while backflow from oligotrophic Lake Michigan during westerly winds can bring cleaner, colder water into Lake Macatawa, similar to what has been observed in Muskegon Lake (Liu et al. 2018).

We captured 1,182 fish comprising 23 species in Lake Macatawa during 2020 sampling surveys (Table 4). The total catch was similar to previous years, although the number of fish species captured was at the lower bound of what we have previously encountered, keeping in mind that we were able to sample fish with fyke nets at only three of four sites in 2020 because of high water levels (Figure 4). The most abundant fishes in the combined catch (i.e., fyke netting and boat electrofishing) were gizzard shad (37%), bluegill (19%), and spotfin shiner (16%), which composed 72% of the total catch (Figure 5A). Four of the 23 species captured during 2020 were non-native to the Great Lakes basin (Bailey et al. 2004)—alewife, common carp, white perch, and round goby—which composed 4% of the total catch (Table 4).

In fyke netting, bluegill (29%), spotfin shiner (28%), gizzard shad (19%), round goby (5%), and brook silverside (4%) were the most abundant fishes in the catch, composing 85% of all fish captured (Figure 5B). Due to high water levels (Table 2), we were unable to set fyke nets at site #3. The two most abundant species at each site were gizzard shad and bluegill at site #1, spotfin shiner and gizzard shad at site #2, and bluegill and spotfin shiner at site #4 (Table 5). The number of fish captured also varied among sites, with the most fish captured at site #4 and the least at site #2 (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 difference in the relative abundance (i.e., percentage of a fish species in the total catch for a given year) was that we captured more spotfin shiner in 2020 compared with most previous 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 (61%), largemouth bass (8%), white sucker (5%), emerald shiner (5%), bluegill (4%), and golden shiner (4%), which composed 87% of the total catch (Figure 5C). Gizzard shad was most abundant in the catch at all sites (Table 6). The second most abundant species in the catch was white sucker at site #4, emerald shiner at site #1, and largemouth bass at sites #2 and #3 (Table 6). Total catch also varied among sites in 2020, with the higher catch at sites #3 and #4 and lower catch at sites #1 and #2 (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 2020 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 seventh year of an effort to characterize the littoral fish assemblage of Lake Macatawa. This monitoring effort is providing a baseline to assess how the fish assemblage responds to restoration activities in the Lake Macatawa watershed. After 7 years of fish monitoring, there are both positive and negative indicators of the ecological health in Lake Macatawa. Yellow perch, bluegill, and pumpkinseed were common species captured in our surveys, and they are indicators of good water quality (Janetski and Ruetz 2015; Cooper et al. 2018). Nevertheless, other common fish species in our surveys, such as gizzard shad and spotfin shiner, are often associated with poor water quality (Janetski and Ruetz 2015). The absence of rock bass (*Ambloplites rupestris*) in the catch likely indicates poor 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 2020 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 nets could not be set at Site 3 in 2020 due to high water levels; however, water quality measurements were recorded.

	Fyke netting		Electrofishing			
			Start		End	
Site	Lat (°)	Long (°)	Lat (°)	Long (°)	Lat (°)	Long (°)
1	42.79604	-86.12117	42.79576	-86.12035	42.79572	-86.12292
2	42.79035	-86.14381	42.78988	-86.14397	42.79155	-86.14378
3	42.78614	-86.17440	42.78669	-86.17564	42.78542	-86.17375
4	42.77952	-86.19733	42.77888	-86.19800	42.78079	-86.19563

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 15 September 2020 with a YSI sonde. Chl-*a* values were not reported in 2020 due to a sensor malfunction.

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
1	98 ± 9	19.14 ± 0.05	9.97 ± 0.01	108.0 ± 0.2	564 ± 1	0.37 ± 0.00	17.3 ± 1.0	7.72 ± 0.02	352.4 ± 2.1
2	113 ± 7	18.51 ± 0.02	9.86 ± 0.10	105.4 ± 1.1	468 ± 0	0.30 ± 0.00	21.4 ± 1.7	8.16 ± 0.02	356.1 ± 0.6
3	116 ± 3	18.16 ± 0.03	11.28 ± 0.06	119.7 ± 0.6	384 ± 0	0.25 ± 0.00	9.0 ± 0.0	8.52 ± 0.13	356.0 ± 8.8
4	90 ± 6	17.23 ± 0.02	9.98 ± 0.03	103.9 ± 0.4	381 ± 0	0.25 ± 0.00	24.8 ± 4.2	7.32 ± 0.03	403.9 ± 2.6

Table 3. Water quality variables at fish sampling sites in Lake Macatawa. Measurements were made during nighttime boat electrofishing on 10 September 2020 with a YSI sonde. Chl-a values were not reported in 2020 due to a sensor malfunction.

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)
1	19.53	7.85	85.7	536	0.348	23.9	7.79	365.2
2	19.22	6.62	71.7	479	0.311	18.5	7.8	361.9
3	19.18	10.11	109.5	376	0.245	9.5	8.17	392.0
4	18.69	11.18	119.9	376	0.244	8.7	8.48	357.2

Table 4. Number and total length (TL; mean, minimum, and maximum) of fish captured by fyke netting (*n* = 9 nets) on 15 September 2020 at three sites and boat electrofishing (*n* = 4 transects) on 10 September 2020 at four sites in Lake Macatawa. Total is the total catch combined for both gears. Fyke nets were not set at Site 3 due to high water levels.

			Fyke netting				Electrofishing			
				TL (cm)				TL (cm)		
Common name	Scientific name	Total	Catch	Mean	Min	Max	Catch	Mean	Min	Max
alewife	*Alosa pseudoharengus*	1	0	--	--	--	1	7.1	7.1	7.1
yellow bullhead	*Ameiurus natalis*	1	1	22.6	22.6	22.6	0	--	--	--
bowfin	*Amia calva*	6	4	53.5	44.1	65.9	2	50.1	37.1	63.0
white sucker	*Catostomus commersonii*	38	12	38.8	23.4	47.8	26	41.1	28.1	47.1
common carp	*Cyprinus carpio*	7	1	63.8	63.8	63.8	6	49.8	39.3	74.5
spotfin shiner	*Cyprinella spiloptera*	185	185	6.0	3.2	9.9	0	--	--	--
gizzard shad	*Dorosoma cepedianum*	443	126	9.3	7.1	16.5	317	10.9	6.9	19.6
northern pike	*Esox lucius*	1	1	49.8	49.8	49.8	0	--	--	--
channel catfish	*Ictalurus punctatus*	4	4	45.5	15.5	61.5	0	--	--	--
brook silverside	*Labidesthes sicculus*	41	29	7.0	5.5	8.8	12	8.1	6.9	8.9
green sunfish	*Lepomis cyanellus*	1	1	11.7	11.7	11.7	0	--	--	--
pumpkinseed	*Lepomis gibbosus*	19	10	13.3	6.9	19.8	9	11.4	9.8	13.5
bluegill	*Lepomis macrochirus*	219	196	5.4	4.0	19.0	23	11.3	4.6	17.6
bluegill or pumpkinseed	*Lepomis* spp.[1]	18	18	3.5	2.7	3.9	0	--	--	--
largemouth bass	*Micropterus salmoides*	49	7	10.9	8.9	13.8	42	21.5	8.1	43.0
white perch	*Morone americana*	6	1	9.4	9.4	9.4	5	14.6	8.5	24.2
round goby	*Neogobius melanostomus*	38	32	6.3	3.6	7.9	6	8.9	7.5	11.0
emerald shiner	*Notropis atherinoides*	26	2	7.3	5.4	9.1	24	9.3	8.5	10.6
golden shiner	*Notemigonus crysoleucas*	31	12	8.7	5.4	14.5	19	9.3	7.8	14.0
spottail shiner	*Notropis hudsonius*	4	3	10.1	8.0	11.6	1	10.5	10.5	10.5
yellow perch	*Perca falvescens*	23	5	17.1	9.2	27.3	18	14.5	8.6	24.8
bluntnose minnow	*Pimephales notatus*	15	14	6.3	5.1	9.0	1	9.0	9.0	9.0
black crappie	*Pomoxis nigromaculatus*	6	3	8.0	7.7	8.5	3	8.9	7.8	9.9
walleye	*Sander vitreus*	6	0	--	--	--	6	58.6	45.0	66.7
	Total	1182	667				521			

[1]Individuals were too small to easily identify to species but were either bluegill or pumpkinseed

Table 5. Number and total length (TL; mean, minimum, and maximum) of fish captured by fyke netting (*n* = 3 nets per site) at three sites in Lake Macatawa on 15 September 2020. Site locations are depicted in Figure 1. Fyke nets were not set at Site 3 due to high water levels.

		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
yellow bullhead	*Ameiurus natalis*	**1**	22.6	22.6	22.6	**0**	--	--	--	--	--	--	--	**0**	--	--	--
bowfin	*Amia calva*	**2**	52.0	51.6	52.4	**0**	--	--	--	--	--	--	--	**2**	55.0	44.1	65.9
white sucker	*Catostomus commersonii*	**6**	36.5	23.4	42.9	**3**	45.1	40.6	47.8	--	--	--	--	**3**	37.1	24.7	44.1
common carp	*Cyprinus carpio*	**0**	--	--	--	**0**	--	--	--	--	--	--	--	**1**	63.8	63.8	63.8
spotfin shiner	*Cyprinella spiloptera*	**13**	5.5	3.8	8.7	**32**	6.9	4.2	9.9	--	--	--	--	**140**	5.8	3.2	8.6
gizzard shad	*Dorosoma cepedianum*	**96**	9.2	7.1	16.5	**29**	9.6	8.2	12.1	--	--	--	--	**1**	14.0	14.0	14.0
northern pike	*Esox lucius*	**0**	--	--	--	**0**	--	--	--	--	--	--	--	**1**	49.8	49.8	49.8
channel catfish	*Ictalurus punctatus*	**2**	34.6	15.5	53.7	**1**	51.4	51.4	51.4	--	--	--	--	**1**	61.5	61.5	61.5
brook silverside	*Labidesthes sicculus*	**26**	6.9	5.5	8.6	**2**	8.1	7.4	8.8	--	--	--	--	**1**	8.2	8.2	8.2
green sunfish	*Lepomis cyanellus*	**0**	--	--	--	**0**	--	--	--	--	--	--	--	**1**	11.7	11.7	11.7
pumpkinseed	*Lepomis gibbosus*	**3**	14.1	11.9	16.6	**6**	14.0	10.6	19.8	--	--	--	--	**1**	6.9	6.9	6.9
bluegill	*Lepomis macrochirus*	**45**	6.6	4.1	19.0	**10**	5.6	4.2	9.1	--	--	--	--	**141**	5.1	4.0	7.0
bluegill or pumpkinseed	*Lepomis* spp.[1]	**7**	3.2	2.7	3.9	**0**	--	--	--	--	--	--	--	**11**	3.8	3.6	3.9
largemouth bass	*Micropterus salmoides*	**6**	11.0	8.9	13.8	**0**	--	--	--	--	--	--	--	**1**	10.6	10.6	10.6
white perch	*Morone americana*	**0**	--	--	--	**0**	--	--	--	--	--	--	--	**1**	9.4	9.4	9.4
round goby	*Neogobius melanostomus*	**1**	7.5	7.5	7.5	**0**	--	--	--	--	--	--	--	**31**	6.3	3.6	7.9
emerald shiner	*Notropis atherinoides*	**0**	--	--	--	**0**	--	--	--	--	--	--	--	**2**	7.3	5.4	9.1
golden shiner	*Notemigonus crysoleucas*	**9**	8.7	5.4	14.5	**0**	--	--	--	--	--	--	--	**3**	8.6	7.3	9.3
spottail shiner	*Notropis hudsonius*	**1**	8.0	8.0	8.0	**1**	10.7	10.7	10.7	--	--	--	--	**1**	11.6	11.6	11.6
yellow perch	*Perca falvescens*	**1**	22.4	22.4	22.4	**1**	16.6	16.6	16.6	--	--	--	--	**3**	15.5	9.2	27.3
bluntnose minnow	*Pimephales notatus*	**0**	--	--	--	**10**	6.3	5.7	7.4	--	--	--	--	**4**	6.4	5.1	9.0
black crappie	*Pomoxis nigromaculatus*	**0**	--	--	--	**0**	--	--	--	--	--	--	--	**3**	8.0	7.7	8.5
	Total	219				95				--				353			

[1]Individuals were too small to easily identify to species but were either bluegill or pumpkinseed

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 10 September 2020. 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**	7.1	7.1	7.1	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--
bowfin	*Amia calva*	**0**	--	--	--	**1**	63.0	63.0	63.0	**0**	--	--	--	**1**	37.1	37.1	37.1
white sucker	*Catostomus commersonii*	**1**	28.1	28.1	28.1	**2**	41.3	39.6	42.9	**1**	41.2	41.2	41.2	**22**	41.7	34.8	47.1
common carp	*Cyprinus carpio*	**3**	48.0	39.3	59.5	**1**	39.6	39.6	39.6	**0**	--	--	--	**2**	57.5	40.5	74.5
gizzard shad	*Dorosoma cepedianum*	**67**	10.4	6.9	18.2	**61**	11.9	7.4	19.6	**128**	10.9	7.9	18.0	**61**	10.6	7.5	15.6
brook silverside	*Labidesthes sicculus*	**3**	7.3	6.9	8.0	**4**	8.4	7.8	8.9	**1**	8.2	8.2	8.2	**4**	8.3	8.0	8.5
pumpkinseed	*Lepomis gibbosus*	**1**	11.4	11.4	11.4	**8**	11.4	9.8	13.5	**0**	--	--	--	**0**	--	--	--
bluegill	*Lepomis macrochirus*	**8**	12.9	10.3	16.8	**12**	10.0	4.6	17.6	**2**	10.2	7.9	12.4	**1**	16.7	16.7	16.7
largemouth bass	*Micropterus salmoides*	**9**	27.3	9.8	40.0	**16**	22.8	10.2	42.2	**6**	17.7	11.5	43.0	**11**	16.7	8.1	34.5
white perch	*Morone americana*	**1**	21.2	21.2	21.2	**2**	9.2	8.5	9.8	**2**	16.7	9.2	24.2	**0**	--	--	--
round goby	*Neogobius melanostomus*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--	**6**	8.9	7.5	11.0
emerald shiner	*Notropis atherinoides*	**21**	9.3	8.5	10.6	**3**	9.5	8.6	10.1	**0**	--	--	--	**0**	--	--	--
golden shiner	*Notemigonus crysoleucas*	**1**	7.8	7.8	7.8	**3**	11.2	9.7	14.0	**3**	9.2	8.6	9.9	**12**	9.0	7.8	10.8
spottail shiner	*Notropis hudsonius*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--	**1**	10.5	10.5	10.5
yellow perch	*Perca falvescens*	**1**	18.9	18.9	18.9	**5**	16.9	9.1	24.8	**1**	9.4	9.4	9.4	**11**	13.5	8.6	20.7
bluntnose minnov	*Pimephales notatus*	**0**	--	--	--	**0**	--	--	--	**0**	--	--	--	**1**	9.0	9.0	9.0
black crappie	*Pomoxis nigromaculatus*	**1**	9.0	9.0	9.0	**0**	--	--	--	**0**	--	--	--	**2**	8.9	7.8	9.9
walleye	*Sander vitreus*	**1**	45.0	45.0	45.0	**1**	66.7	66.7	66.7	**2**	53.9	48.7	59.1	**2**	66.1	65.5	66.7
	Total	119				119				146				137			

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.

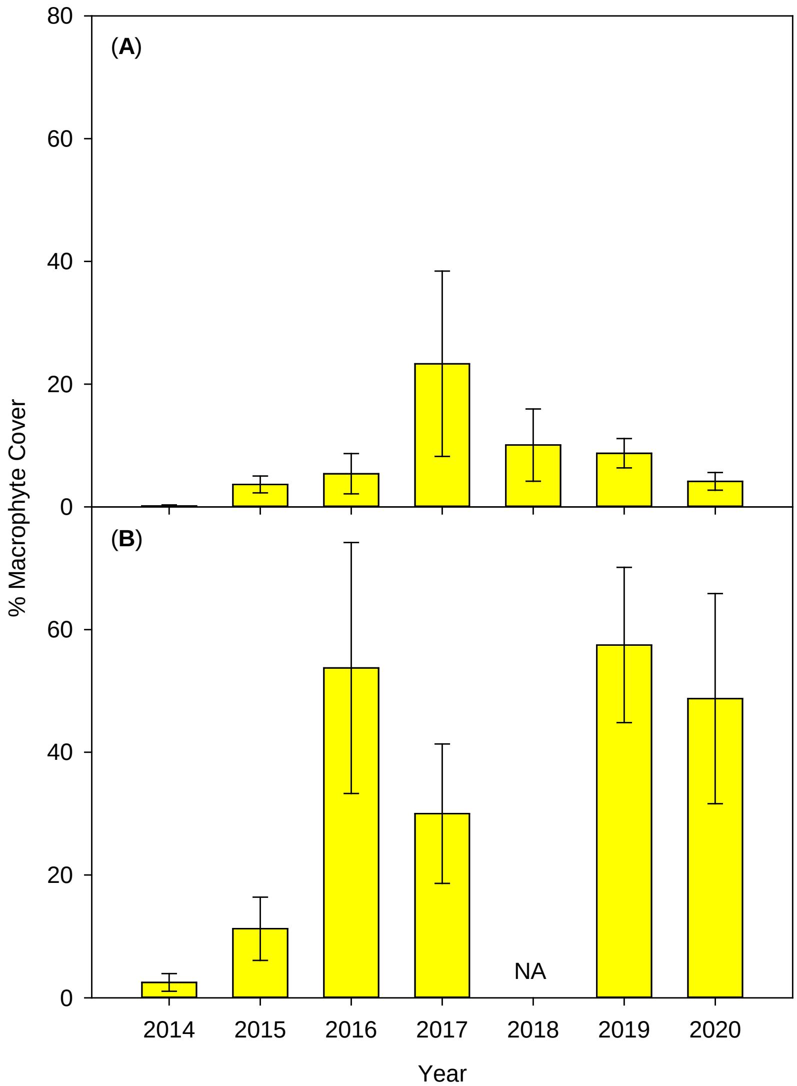

Figure 2. Mean (±1 standard error) percent macrophyte cover visually estimated at (**A**) fyke net locations and (**B**) boat electrofishing transects in Lake Macatawa (*n* = 4 sites per year). Note that the area where macrophyte cover was assessed during fyke netting is much less compared with a boat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

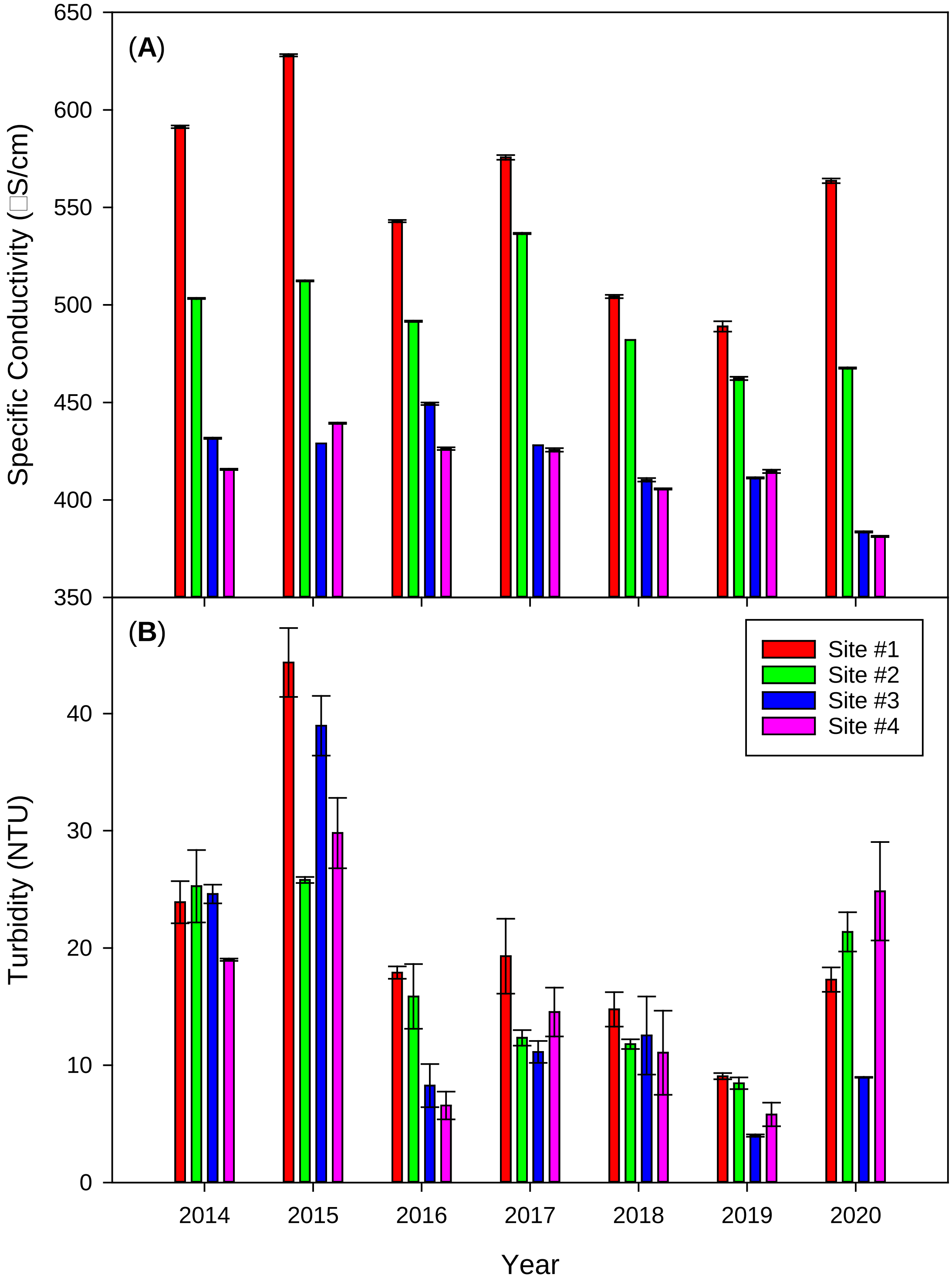

Figure 3. Mean (**A**) specific conductivity and (**B**) turbidity measured during fyke netting in Lake Macatawa. Error bars represent ±1 standard error (n = 3 nets per site), although they may be too small to be visible for some means.

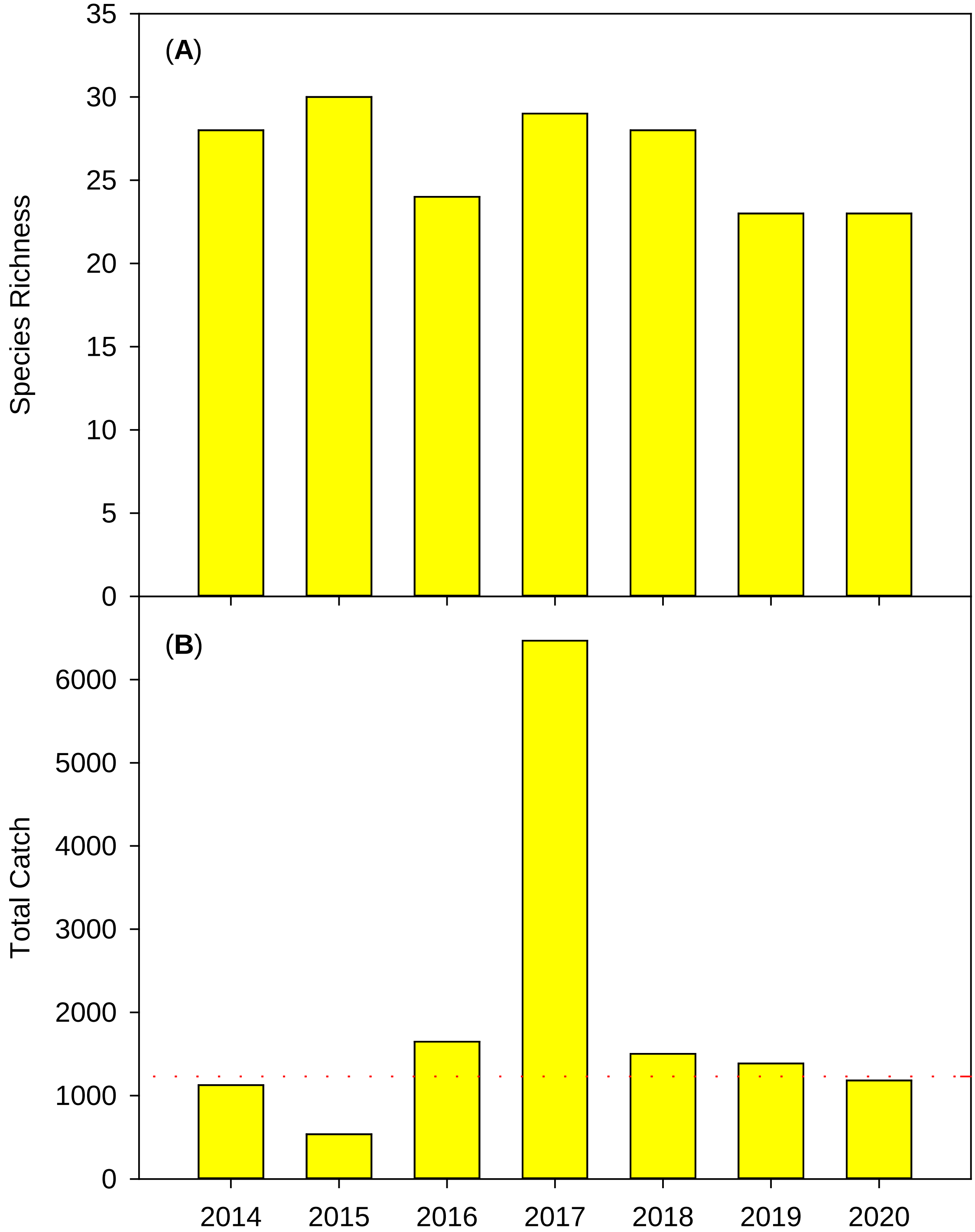

Figure 4. (**A**) Number of fish species captured and (**B**) total number of fish captured using both fyke netting and boat electrofishing each year in Lake Macatawa (dashed red line is average over all years excluding 2017). *Note*: the high catch in 2017 was due to 5,288 brook silversides

captured from a single fyke net at site #4, and fyke netting in 2020 only was completed at three sites.

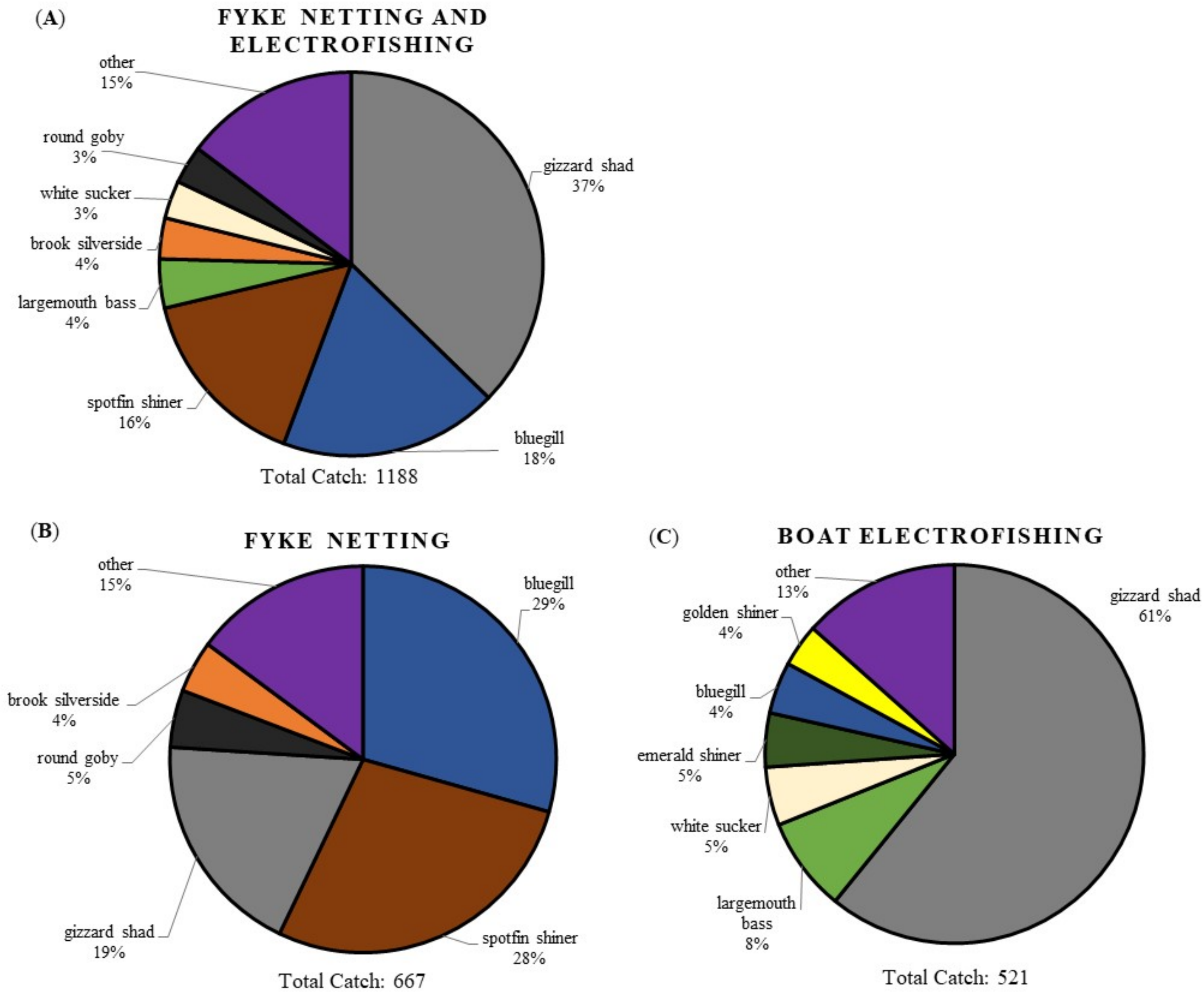

Figure 5. Fish species captured in littoral habitats of Lake Macatawa by (**A**) fyke netting and boat electrofishing (i.e., combined catch), (**B**) fyke netting (n = 9 nets), and (**C**) boat electrofishing (n = 4 transects) during September 2020. Catch data, including the species pooled in the "other" category, are reported in Table 4.

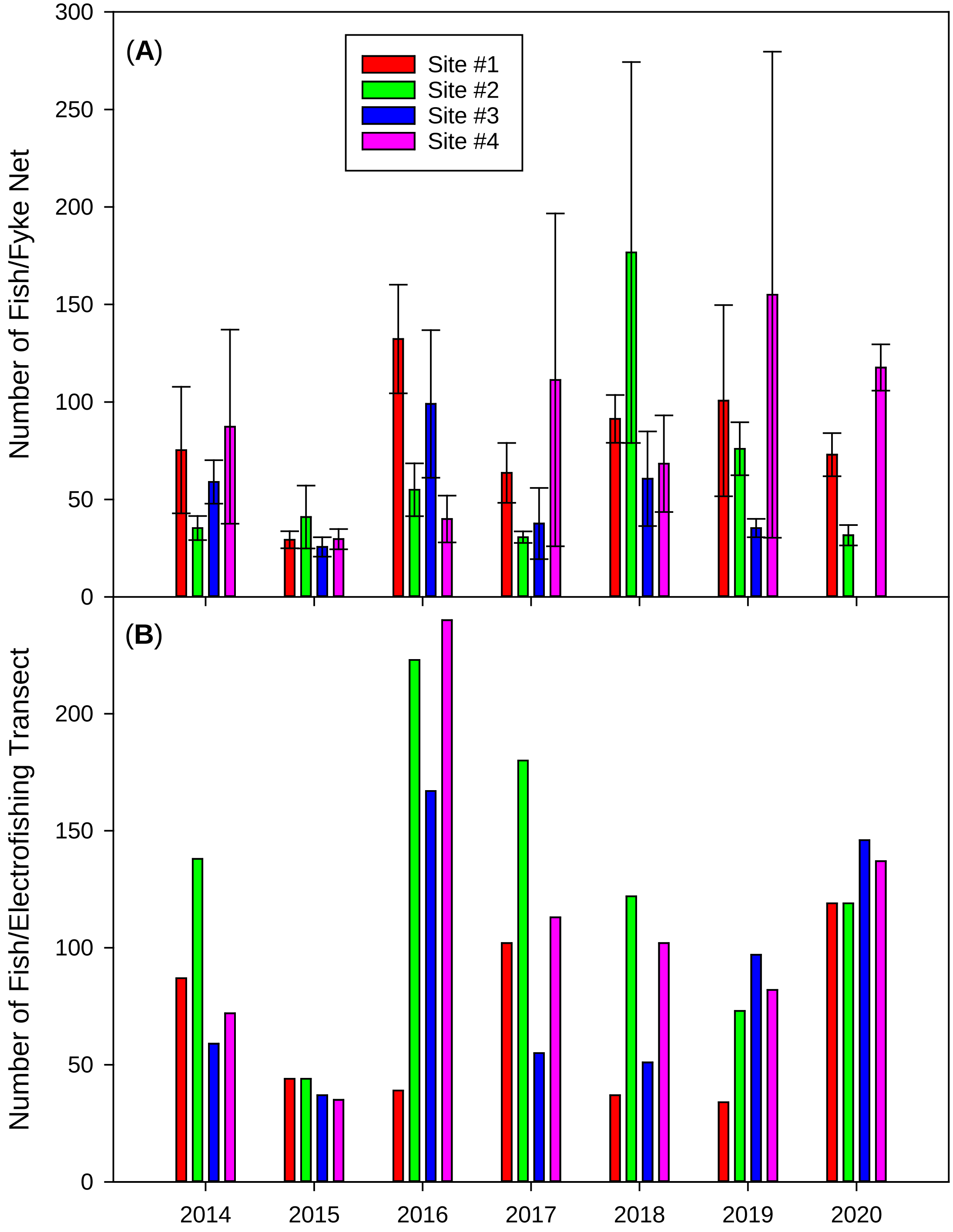

Figure 6. (**A**) Mean number (±1 standard error) of fish captured in fyke nets (*n* = 3 nets per site) and (**B**) number of fish captured during a boat electrofishing transect (*n* = 1 transect per site) in Lake Macatawa. *Note*: 5,288 brook silversides captured in a single fyke net at site #4 in 2017 were excluded when calculating means (and SE) for fyke netting. Fyke nets were not set at site #3 in 2020.

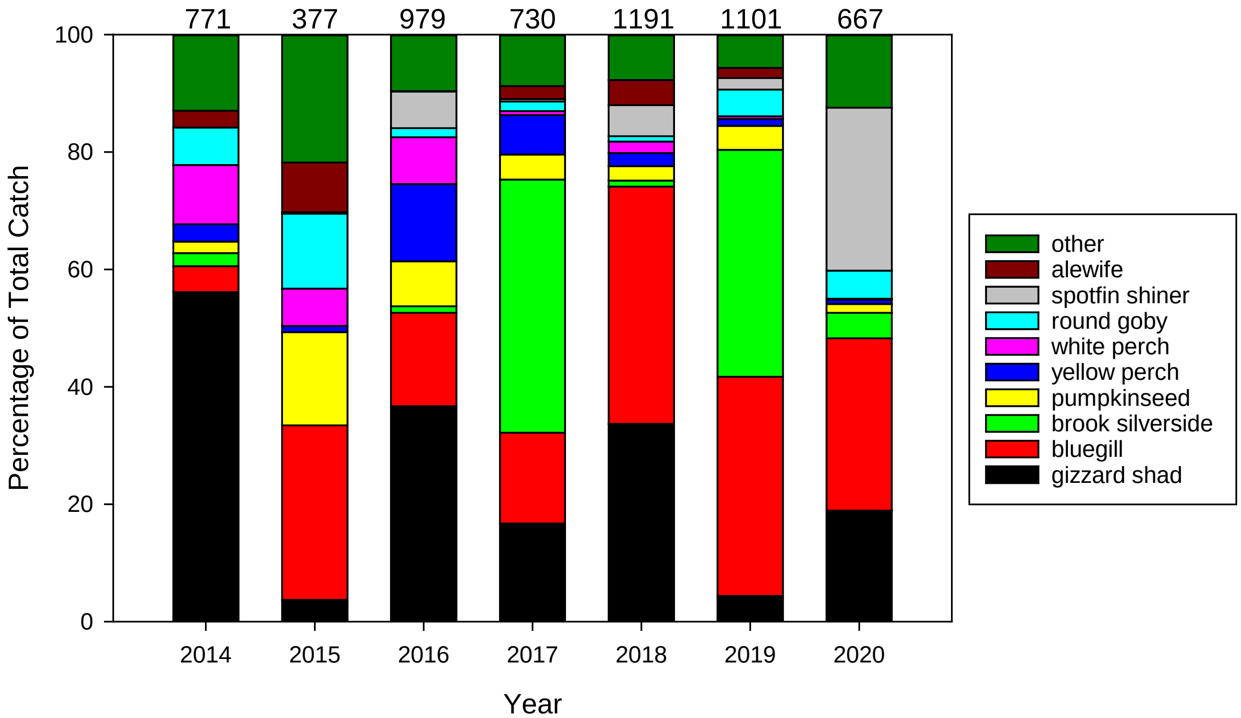

Figure 7. Fish species composition (pooled across sites) in fyke netting surveys for each sampling year. The number of fish captured differed among years, which is reported at the top of each bar. *Note*: 5,288 brook silversides captured in a single fyke net at site #4 in 2017 were excluded from the percentage of total catch, and fyke nets were not set at site #3 in 2020.

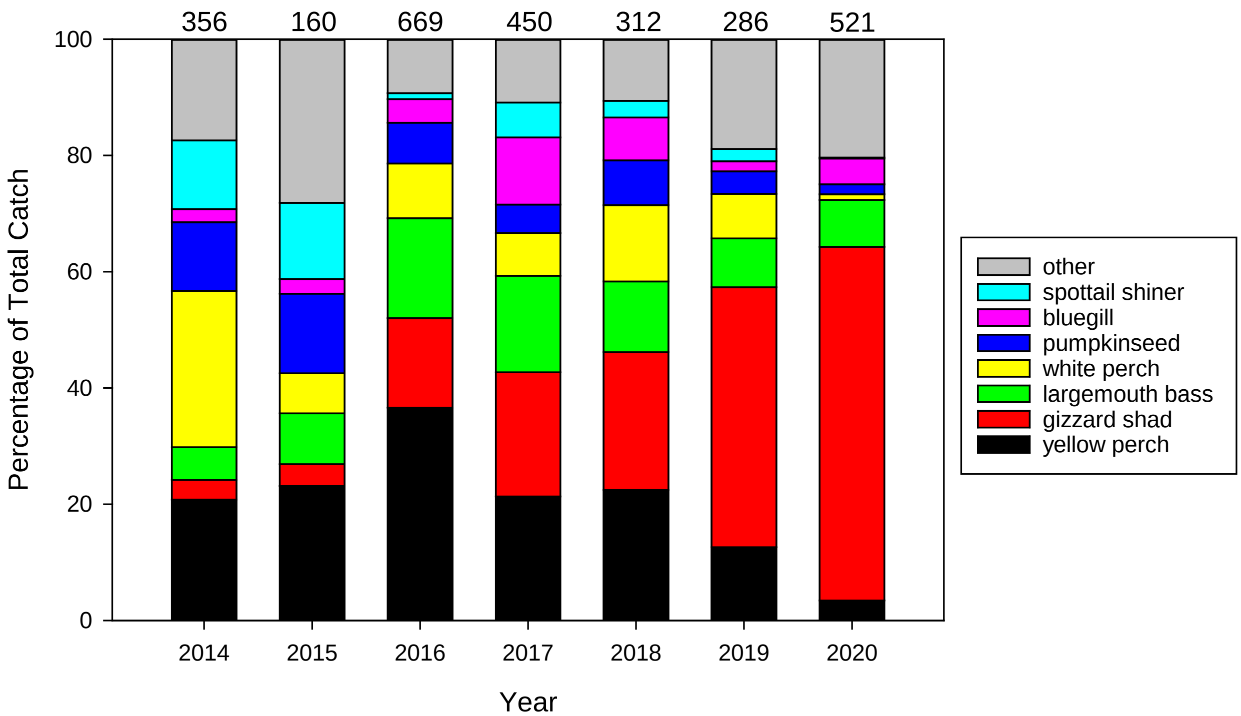

Figure 8. Fish species composition (pooled across sites) in nighttime boat electrofishing surveys for each sampling year. The number of fish captured differed among years, which is reported at the top of each bar.

Appendix B. Iron Slag Filters

Slag Filter BMP Performance in the Macatawa Watershed

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December, 2020

1. Overview

Lake Macatawa and its watershed suffer from excess phosphorus (P) (Walterhouse 1999). The TMDL (total maximum daily load) for the lake calls for a P reduction of 55,000 lb/yr from the estimated load of 138,500 lb/yr (1997), with the majority of that load reduction coming from nonpoint sources (Walterhouse 1999). If fully implemented, the model results indicate that the water column total phosphorus (TP) concentration will be reduced to 50 µg/L, thereby meeting the TMDL goal.

Previous research has demonstrated that agriculture is a major source of nonpoint P in this watershed (Steinman et al. 2018). Although overland flow is generally considered the major transport mechanism for P, there are situations when significant P transport occurs through agricultural tile drainage (King et al. 2015). It is currently uncertain how much P enters Lake Macatawa from surface runoff vs. subsurface runoff, but very high concentrations of both TP and bioavailable P were measured in tile drain effluent from agricultural fields in the watershed (Clement and Steinman 2017). Similar findings have been reported for agricultural areas in the western basin of Lake Erie (Calhoun et al. 2002; Smith et al. 2015), which has been subject to devastating harmful algal blooms the past few decades (Michalak et al. 2013; Wynne and Stumpf 2015).

Various best management practices (BMPs) have been implemented in the Macatawa watershed to reduce P loading, including among others, grassed waterways, cover crop plantings, gypsum application, and wetland restoration, although their effectiveness has been questioned (Steinman et al. 2018). 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). Interest was expressed as to whether iron slag filters may be an effective management practice in the Macatawa watershed. To that end, the Outdoor Discovery Center (ODC) Network, along with Dykhuis Farms and Plant Tuff, Inc., committed to install up to six iron slag filters at the end of agricultural tile lines in the watershed. To date, three of the six have been constructed. Our goal was to evaluate the efficiency of these systems in removing P, while also monitoring for the presence of potentially toxic chemicals leaching from the slag, which may be released into surface waterways. It is our understanding that these filters are the first application of their kind within Michigan.

2. Methods

2.1 Overall site description

The deployment of iron slag filters (Figs. 1, 2) was proposed in the Macatawa watershed in 2018 and three sites were selected as of 12/31/19 for installation. Several additional sites were considered for installation in 2020. Multiple sites throughout the watershed were sampled prior to installation to determine which locations were best suited for iron slag filter installation. Of the three currently constructed filters, two were completed in April 2019 (Behind Mill 1 and Oak Grove 2) and the third (Fillmore Flex; Fig. 3) in September 2019. We present the data related only to the completed iron slag filters. Due to the COVID-19 pandemic, sampling was not permitted for several months in spring 2020 due to the state of Michigan's "stay-at-home" executive order. Sampling resumed once the order was lifted.

Filters are designed to work passively, receiving water after it infiltrates through soils into subsurface tiled drains (Fig. 1). Water moves up and through the iron slag gravel in large concrete tank(s), where the slag binds with and removes the P from the water before it passively releases to adjacent surface waterways (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 moving through the tile drains (indicated by standing water on the farm field), and serves as the inflow access point for water collection for most sites. Outflow water can be sampled either from the outflow pipe, or via access points in the top of the tank with a hose and hand pump (Fig. 4).

2.2 Field and Laboratory Processes

Prior to installation, grab samples were taken monthly at only the outflow pipe, which at that time was a direct open connection from the tile drain pipes to adjacent surface waters. Pre-installation sampling dates are provided in Table 1. Post-installation samples were collected bimonthly; sampling approaches varied among sites due to differences in filter design and implementation (Fig. 4). Post-installation sampling occurred at the inflow using a hand pump and hose to siphon water accessed through an inflow pipe at Behind Mill (Fig. 4A) and through the control box at Oak Grove 2 and Fillmore Flex (Figs. 1, 4B). Outflow was sampled at the original outflow pipe (Fig. 4E; which remained after installation) 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 without disturbing the calcium carbonate top layer (Figs. 4C-D).

General water quality was monitored 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 soluble reactive P (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.

2.3 Metals and PAHs

Chemical analysis sampling for metals (mercury, arsenic, barium, cadmium, chromium, cobalt, copper, lead, molybdenum, nickel, selenium, silver and zinc), low-level mercury, Polycyclic Aromatic Hydrocarbons ([PAHs] 2-Methylonaphthalene, 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, and Pyrene) and available cyanide was conducted three times post-installation (1-week, 6-months and 1-year). Analyses were conducted at TRACE Analytical Laboratories, Inc. (Muskegon, MI) using Standard Methods (US EPA 1993). Any values below analytical detection methods were calculated as ½ their detection limits.

2.4 Data analysis

The water residence time in the reservoir depends on the rate at which the water enters the slag filter, determined by rain, irrigation, or a control box. This means that water during baseflow conditions can spend much longer in the reservoir than water during stormflow conditions, especially in dry summer months. Therefore, for statistical purposes, the inflow and outflow values were paired for each sampling event rather than using grand means. SRP, TP, and turbidity samples were analyzed using paired t-tests or Wilcoxon tests, depending on normality. Normality assumptions were tested using Shapiro-Wilk tests. Significance was set at $\alpha = 0.05$. Data analysis was conducted using R (version 3.4.2 R core Team, Vienna, Austria). Percent reductions were calculated for the paired mean inflow and mean outflow measurements post-installation for SRP and TP by calculating the difference between mean inflow and mean outflow concentrations, dividing that number by the mean inflow concentration, and multiplying it by 100. Negative percent reduction values indicated an increase in P concentration.

3. Results and Discussion

3.1 Pre-Installation Sampling

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

Mean pre-installation SRP concentrations ranged from 135 to 283 µg/L (Table 2); these particular sites were selected for installation due to their relatively high initial concentrations. Mean TP concentrations ranged from 167 to 721 µg/L (Table 2). Among site variance was high, reflecting differences in field management, drainage, and time of year sampled. The highest mean P concentrations were measured at Fillmore Flex (Table 2).

3.2 Post-Installation Sampling

Following installation, mean DO, conductivity, TDS, and ORP showed modest declines in the outflow compared to the inflow (Table 1). In contrast, pH increased in the outflow, presumably due to buffering from the calcium carbonate layer. Whereas turbidity declined significantly at Behind Mill 1 and Oak Grove 2, it increased slightly at Fillmore Flex (Table 1). Reduction in turbidity between inflow and outflow was anticipated as water velocity decreases when water enters the iron slag filter, allowing sediment particles to settle out of the solution before the water reaches the surface waters. The Fillmore Flex outflow pipe is angled and below grade (not pictured). It has been observed to experience back flow from the drainage ditch into the outflow pipe if water levels are sufficiently high. This could result in back flow into the slag filter, which may explain the higher levels of turbidity in the outflow than inflow and general variability (Table 1). We did not measure sediment load within the pipes pre-or post-installation. It is unclear if the sediment will impact the longevity of the slag filter's P trapping effectiveness.

After installation of the slag filters, both SRP and TP decreased in the tile drain effluent: percent reductions of SRP ranged from 57.3% (Oak Grove 2) to 7.4% (Fillmore Flex); percent reductions of TP ranged from 76.5% (Oak Grove) to 59.5% (Behind Mill 1) (Table 2). Analysis of paired inflow and outflow samples at each of the sampling sites revealed significant reductions of TP at Behind Mill 1 and Oak Grove 2 ($p = 0.0011$ and $p = 0.0001$, respectively) and of SRP ($p = 0.0028$ and $p = 0.0004$, respectively; Table 3). Fillmore Flex had a marginally significant reduction of SRP ($p = 0.0825$) and a statistically significant reduction of TP ($p = 0.0002$) (Table 3).

These slag filters function by the binding of phosphorus to iron, similar to what happens under oxic conditions in lake sediments (Mortimer 1941; Orihel et al. 2017). Therefore, their ability to reduce P is not surprising, and they should remain effective as long as conditions within the reservoir remain oxic and the binding sites do not become saturated. Prior studies have confirmed the ability of iron slag to bind phosphorus, with phosphate percent removals of 95% in metal-free and mixed-metal solutions of synthetic stormwater (Okochi and McMartin 2011) and a TP removal percentage of 77% (Shilton et al. 2006) at a treatment plant, although P removal effectiveness dropped off after 5 years.

Although TP reductions of ~60 to 75% are certainly helpful, mean outflowing concentrations of 100 to 329 µg/L are still far above the 50 µg/L goal for Lake Macatawa. There will likely be additional removal of P due to biotic uptake and abiotic adsorption as the nutrient flows downstream to the lake, although it is uncertain to what degree this removal will be offset by additional P sources entering the drainage system. Hence, while iron slag filters clearly can reduce P, their installation is only part of the solution, and will need to be complemented by other BMPs, and perhaps ultimately, a more holistic public works project.

3.3 Metals and PAHs

Metals were measured during pre-installation and 1-week, 6-months and 1-year post installation at all sites. Behind Mill 1 and Oak Grove 2 were sampled a few months past the 1-year mark because of COVID-19 restrictions. All metals, PAH compounds, and available cyanide were below U.S. Environmental Protection Agency (EPA; source: https://www.epa.gov/dwstandardsregulations/secondary-drinking-water-standards-guidance-nuisance-chemicals) and World Health Organization (WHO; source: https://www.wqa.org/learn-about-water/common-contaminants) standards for drinking water. Recreational water standards are unavailable.

Mercury was below 12 ng/L for all sites, which is orders of magnitude below the drinking water standards for 2,000 ng/L (US EPA) and 6,000 ng/L (WHO). Behind Mill 1 showed a slight increase in mercury between pre-installation and 6-month post-installation, although 1-year post installation was below pre-installation. Oak Grove and Fillmore Flex remained relatively steady with Fillmore Flex having the highest levels (Fig. 5A). Arsenic was found in detectable amounts only at Fillmore Flex pre-installation, and was below both the EPA and WHO standard of 0.010 mg/L (Fig. 5B). Barium was found below detection 1-week post installation at Behind Mill 1 and Oak Grove 2 and at 1 year at Fillmore Flex, 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 had similar although reduced values, where Fillmore Flex saw a reduction from the highest value to below detection (Fig. 5C). Chromium was detected 1-week post installation at Oak Grove 2 and Fillmore Flex, 6-week post installation at Behind Mill 1 and Fillmore Flex and 1-year post installation only at Fillmore Flex, but all were below both the EPA (0.10 mg/L) and WHO (0.05 mg/L) standards (Fig. 5D). Copper increased at Behind Mill 1 and Oak Grove 2 from pre-installation to 1-year post installation, while Fillmore Flex Cu levels declined from pre- to 1-year post installation (Fig. 5E, 5F). Zinc was above the detection level only at Oak Grove at 6-month post installation (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, and Terphenyl-d14, were all below detection limits for all sites and sampling dates (data not shown). Cyanide (available) was detectable only at Fillmore Flex at 6-month post installation (0.0027 mg/L).

Prior work has shown that the presence of cadmium, lead, and zinc had minimal effect on P removal when using electrical arc furnace slag media; however, copper did significantly inhibit P removal (Okochi and McMillan 2011). Although metal concentrations were not measured in the slag media used in the present study, the copper levels in the leachate from our slag filters were extremely low (Fig. 5), and presumably, had minimal impact on P removal. Moreover, studies have shown that there are no detrimental effects to human health or to the environment as a result of leaching from steel slags, and that due to the very high temperatures applied in their formation, heavy metals are bound tightly together within the slag matrix (Proctor et al. 2002; Johansson Westholm 2010; Shilton et al. 2006), which is consistent with our data.

4. Summary

The iron slag filters have been installed for a minimum of 1 year and the current data indicate they are effective at removing P from tile drain effluent. There is considerable variation in the percent reduction among the 3 sites, but this is not unexpected. The percent reductions in P are significant but still well above the 50 µg/L goal. Encouragingly, there is no indication that the iron slag is releasing toxic metals, PAH compounds, or cyanide at levels that would cause concerns for drinking water standards.

We conclude that the installation of these filters should be targeted to areas where tile drain effluent P levels are very high (SRP > 250 µg/L) to obtain an optimal cost/benefit ratio. While they are not a panacea, when installed in combination with other best management practices, iron slag filters can play an important localized role in reducing P to Lake Macatawa.

5. Acknowledgements

Funding was provided through Project Clarity funds; our thanks to Travis Williams, Dan Callam, Rob Vink, and David Nyitray of ODC for all of their help and knowledge of the area, as well as the other partners of Project Clarity including Kelly Goward and Steve Bulthuis of the MACC, Todd Losee and Steve Niswander of Niswander Environmental, Dr. Aaron Best, Sarah Brokus, and Randy Wade of Hope College, and the Dykhuis Farms and their family for sampling access and participation in this project.

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Table 1. Mean (1 standard deviation [SD]) values of selected water quality parameters for tile drain in/outflow iron slag pre- and post-installation monitoring. Date of first sampling is provided below each Pre/Post. 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 (mV)	Turbidity (NTU)
Behind Mill 1	Pre (9/20/2018 thru 4/16/2019)		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 (4/25/2019 thru 5/27/2020)	Inflow	22	13.56 (5.97)	8.34 (1.73)	7.18 (0.38)	905 (506)	0.588 (0.329)	321.5 (66.7)	46.9 (62.4)
		Outflow	22	13.36 (5.21)	7.23 (1.97)	7.91 (0.82)	820 (169)	0.533 (0.110)	301.2 (80.5)	9.3 (11.6)
Oak Grove 2	Pre (9/20/2018 thru 3/19/2019)		7	9.13 (7.20)	11.30 (2.64)	8.19 (0.25)	676 (132)	0.439 (0.086)	284.3 (72.6)	0.0 (5.4)
	Post (5/7/2019 thru 5/27/2020)	Inflow	18	13.15 (7.10)	8.68 (2.16)	7.58 (1.07)	724 (175)	0.471 (0.114)	325.2 (63.6)	26.9 (30.5)
		Outflow	17	12.62 (7.14)	8.49 (1.98)	8.08 (1.23)	711 (139)	0.462 (0.090)	305.0 (64.6)	6.2 (9.0)
Fillmore Flex	Pre (3/26/2019 thru 8/28/2019)		9	13.89 (5.69)	7.86 (1.55)	7.65 (1.81)	894 (1187)	0.581 (0.772)	221.0 (113.5)	20.0 (45.3)
	Post (9/18/2019 thru 9/24/2020)	Inflow	19	14.23 (7.77)	7.34 (2.43)	7.81 (0.73)	537 (164)	0.349 (0.106)	315.5 (87.5)	28.2 (66.8)
		Outflow	19	14.65 (7.95)	7.13 (2.68)	8.33 (1.54)	453 (158)	0.294 (0.102)	288.7 (101.7)	33.7 (68.1)

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-installation monitoring. Percent reduction was calculated by finding the difference between inflow and outflow values divided by inflow x100. 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		8	135 (131)	167 (161)	N/A	
	Post	Inflow	25	365 (357)	598 (652)	52.3	59.5
		Outflow	25	174 (122)	242 (199)		
Oak Grove 2	Pre		7	229 (328)	254 (328)	N/A	
	Post	Inflow	23	157 (167)	426 (634)	57.3	76.5
		Outflow	22	67 (57)	100 (87)		
Fillmore Flex	Pre		9	267 (201)	721 (699)	N/A	
	Post	Inflow	23	283 (192)	907 (1368)	7.4	63.7
		Outflow	22	262 (170)	329 (148)		

Table 3. Mean water quality parameters (1 SD) for paired inflow and outflow at each site for TP, SRP and Turbidity. Data are shaded to improve readability. n= number of successful sampling events per site. α = 0.05. V = Wilcoxon test statistic; t = test statistic for the t-test.

Site	Analysis	n	Mean Inflow		Mean Outflow	Stats
Behind Mill 1	TP (µg/L)	25	595 (653)	>	242 (199)	p = 0.0011 V = 278
	SRP (µg/L)	25	364 (355)	>	174 (122)	p = 0.0028 V = 270
	Turbidity (NTU)	19	49 (61)	>	9 (12)	p = 0.0002, t = 4.6
Oak Grove 2	TP (µg/L)	21	438 (664)	>	104 (86)	p = 0.0001 V = 217
	SRP (µg/L)	21	164 (188)	>	69 (57)	p = 0.0004 V = 210
	Turbidity (NTU)	10	27 (29)	≈	5 (8)	p = 0.0063 t = 3.3
Fillmore Flex	TP (µg/L)	17	997 (1451)	>	304 (126)	p = 0.0002 V = 210
	SRP (µg/L)	17	285 (208)	≈	234 (149)	p = 0.0825 V = 152
	Turbidity (NTU)	15	26 (66)	≈	12 (19)	p = 0.6819 t = 0.4

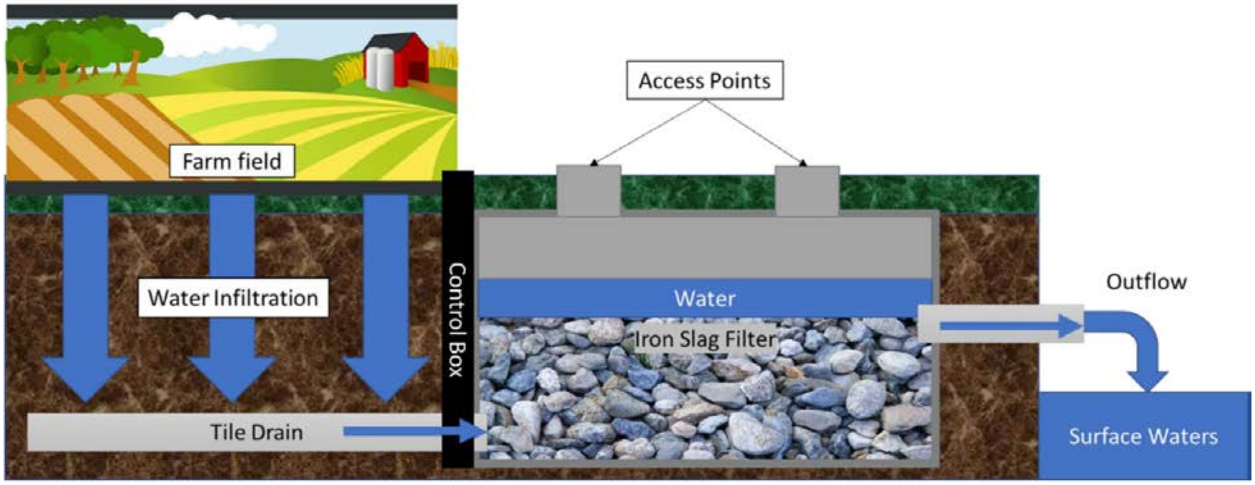

Figure 1. Stylized cross-section of iron slag filters design/function. Figure is not to scale as iron slag filter size is dependent on multiple factors (e.g. size of the tiled field, water velocity from the tile drains, soil type). See text for more detail on how the filters function. Image credit: Maggie 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 out a pipe near the top (not pictured) that leads to a nearby surface drain (not pictured). Photo credit: Macatawa Area Coordinating Council.

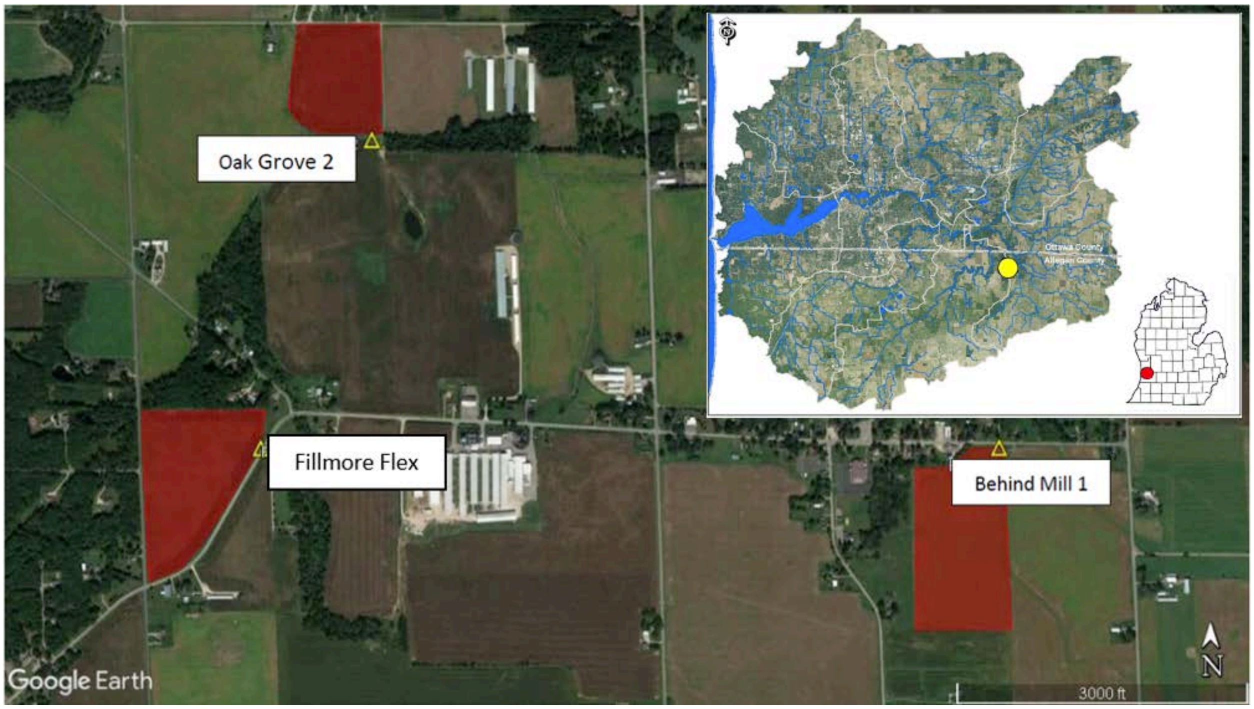

Figure 3. Completed iron slag filter sites are indicated by yellow triangles, the red polygons represent the adjoining fields being drained, each approximately 30 acres. Fillmore Flex and Oak Grove 2 sites flow into the south branch of the Macatawa River, which later joins the main branch of the Macatawa River. Behind Mill 1 flows into Peter's Creek. The insert shows the location within the Macatawa watershed by the yellow circle, and within the lower peninsula of Michigan by the red circle.

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

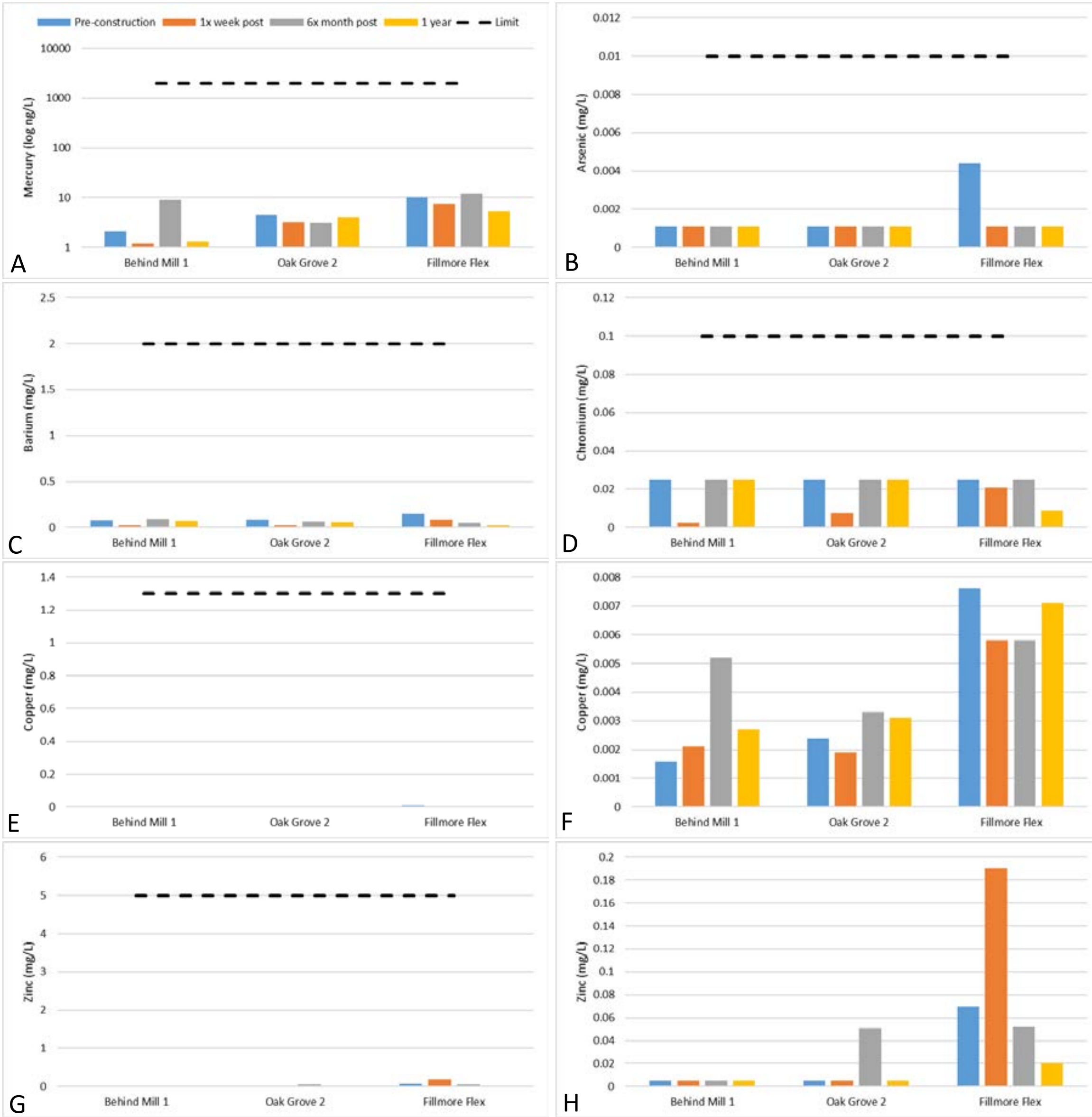

Figure 5. Metals for pre-installation, 1-week, 6-month and 1-year post installation. Blue is pre-installation, orange is 1-week post installation, grey is 6-month post installation, and yellow is for 1-year post installation. The black dotted line represents the drinking water standard from either the 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 G respectively. Note: y-axis for panel A is logarithmic scale.

Appendix C. Lake Macatawa Dashboard.

Lake Macatawa Water Quality Dashboard

2020

Prepared: January 2021

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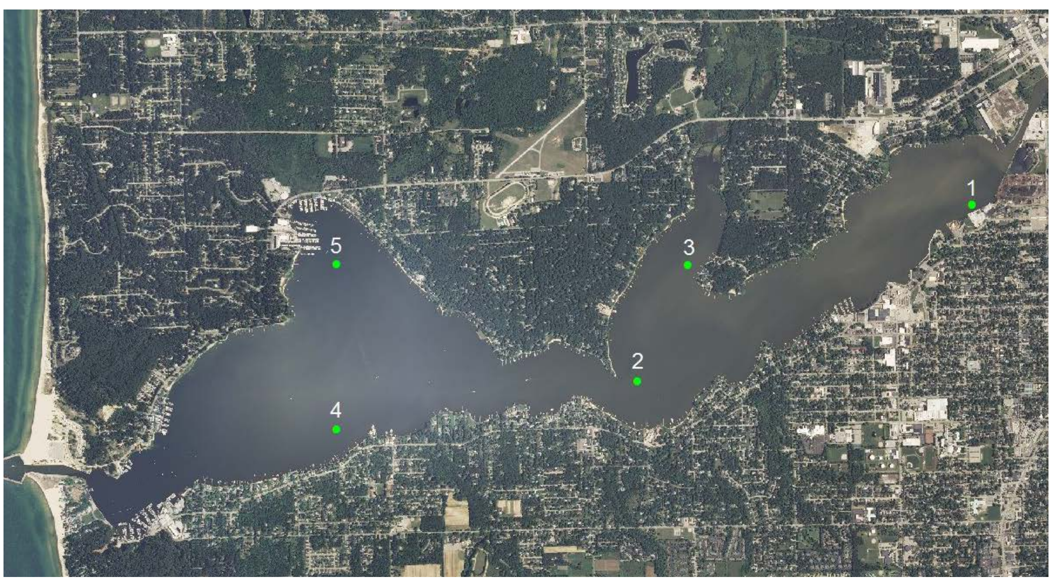
Introduction

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 collected continuously and may either capture or miss episodic, short-term conditions. The value of the dashboard is an assessment of long-term trends, not of 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.

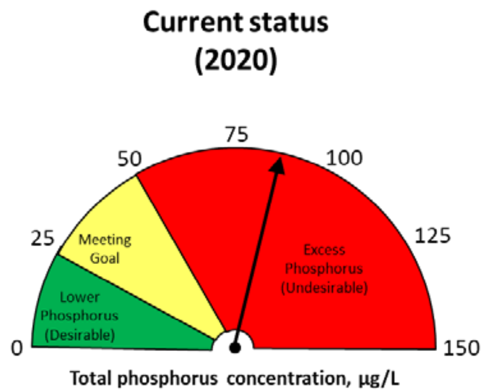
Historical data are included in the dashboard to facilitate comparison of current findings with past status of the selected water quality indicators. Sources for historical data include U.S. EPA (1972; STORET), Michigan Department of Environment, Great Lakes, and Energy (formerly MDEQ; 1982-2012; S. 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).

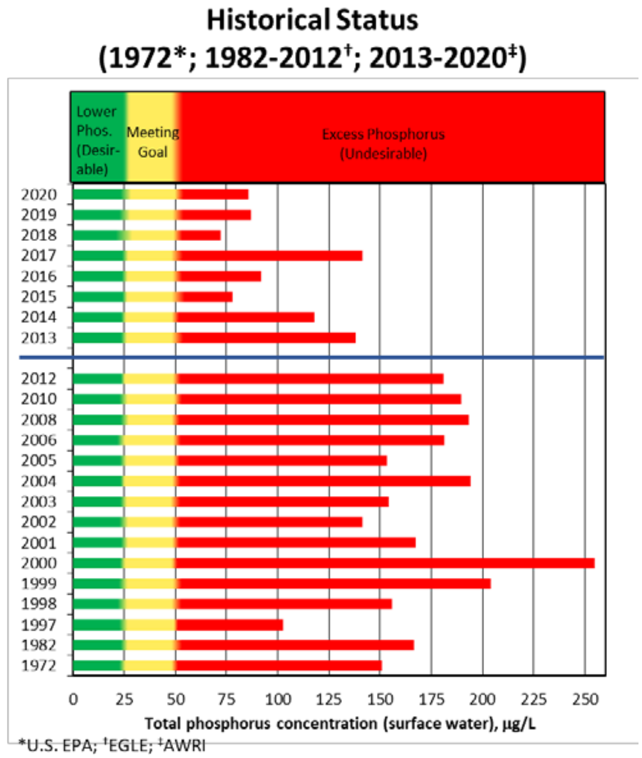


Map of Lake Macatawa showing the 5 sampling locations (green dots) for long-term water quality monitoring. Dashboard indicators were calculated based on data from Sites 1, 2, and 4.

Total Phosphorus

2020 Mean Concentration: 86 µg/L
Target Concentration: 50 µg/L





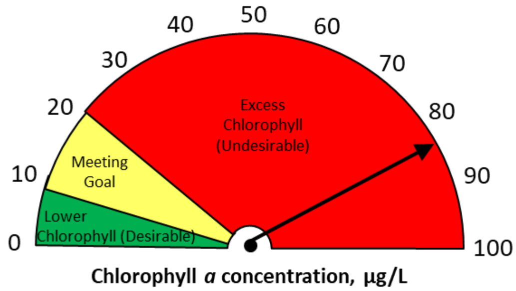
Phosphorus (P) is an essential element for living organisms. In many freshwater systems, P is the element that limits algal growth. However, when it becomes too abundant, it can help stimulate undesirable algal blooms. Phosphorus comes in many forms; we selected Total Phosphorus (TP) as the dashboard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

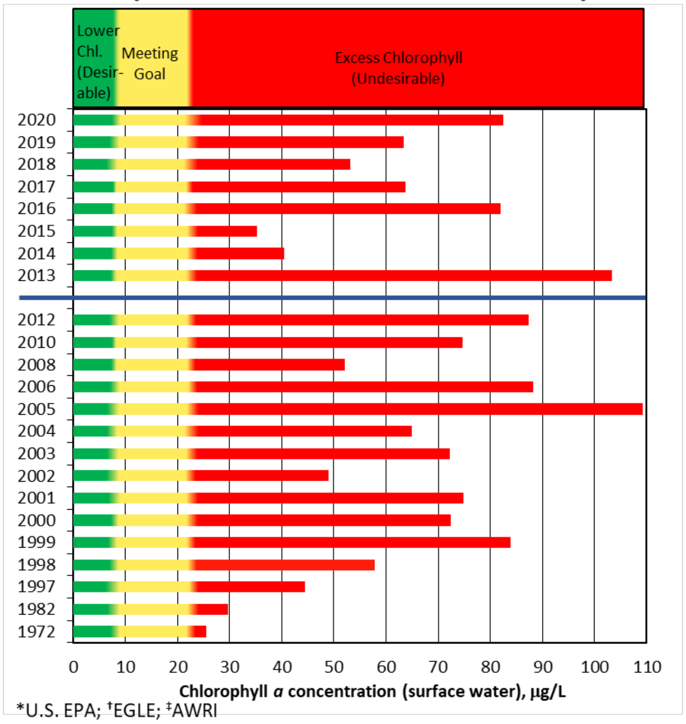
Lake Macatawa has a history of extremely high TP concentrations (i.e., > 100 µg/L), placing it in the past in the "hypereutrophic" trophic state. As a result of this nutrient enrichment, the State of Michigan established an interim target TP concentration of 50 µg/L in Lake Macatawa. Thus, the TP dashboard 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 2020 exceeded the water quality goal. Although the average TP concentration in recent years remains greater than the target of 50 µg/L, the overall trend shows a decline since the start of Project Clarity.

Chlorophyll *a*

2020 Mean Concentration: 83 µg/L
Target Concentration: 22 µg/L





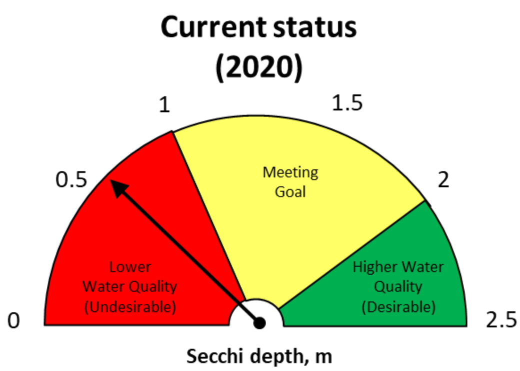
Chlorophyll *a* is the green pigment found in photosynthetic plants and algae. Measuring chlorophyll *a* is a relatively simple way to estimate the amount of algal biomass present in lake water, although it has some limitations. First, chlorophyll *a* does not provide information on whether or not the 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.

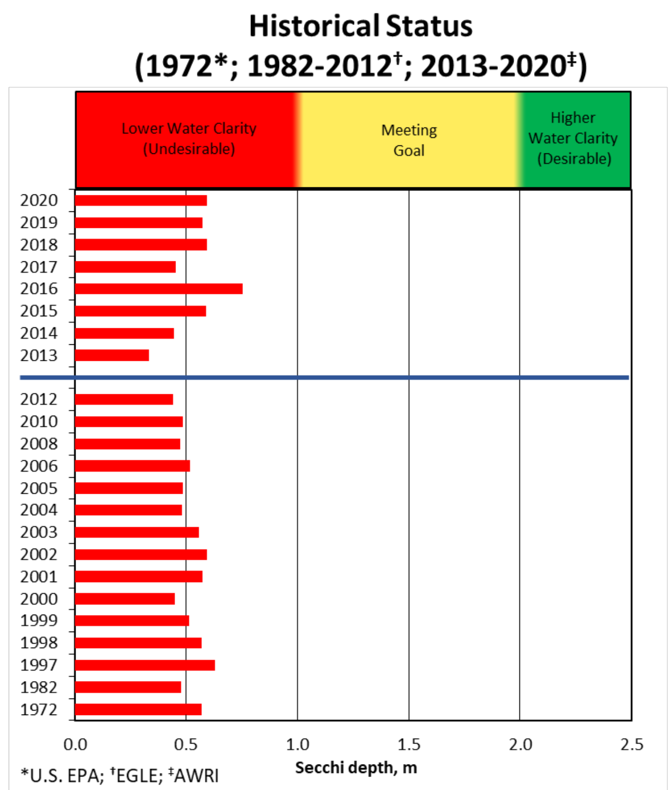
Lake Macatawa has a history of excess algal biomass and high chlorophyll *a* concentrations, typically exceeding the "hypereutrophic" threshold commonly used by EGLE (22 µg/L) in its assessments of the lake. The chlorophyll *a* dashboard shows that the concentration will meet the water quality goal once 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.

Although the current status for the chlorophyll *a* indicator is **Undesirable**, the high concentration is biased by an unusually high Spring concentration at one site. Removing that site would still result in an undesirable rating, but the increase from 2019 would be less severe. The 2020 mean chlorophyll concentration continues a trend of relatively high chlorophyll since the lower values in 2014 and 2015.

Secchi Disk Depth (Water Clarity)

2020 Mean Depth: 0.59 m (~1.9 ft)
Target Depth: 1 m (~3.3 ft)



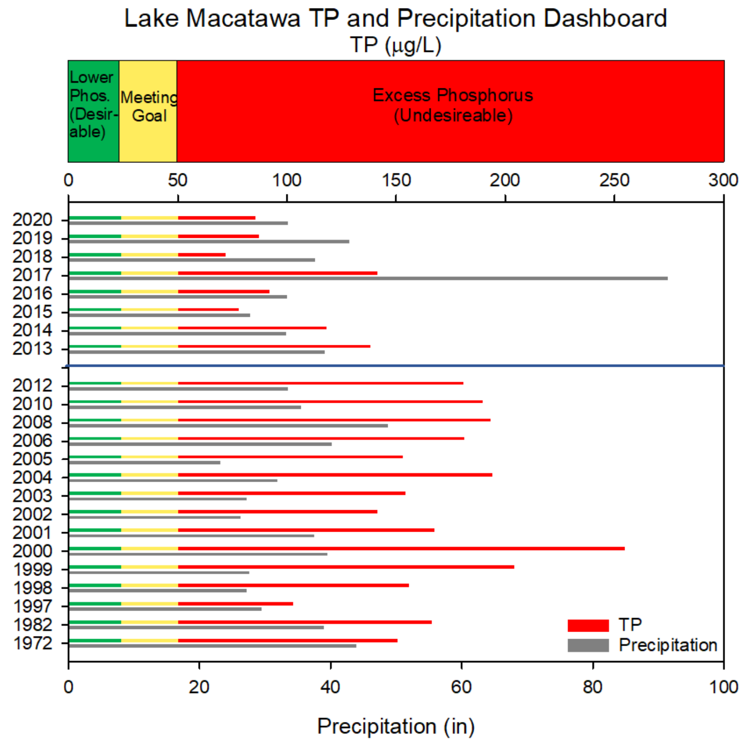


Secchi disk depth is an estimate of water clarity. It is measured using a standard black and white disk, named after the Italian priest Angelo Secchi, who first used an all-white disk for marine waters in 1865. Lake ecologists modified it to black and white in the late 1800s. The Secchi disk is a simple 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 2020 was shallower (i.e., less clear) than the water quality goal. Secchi depth in 2020 remains very similar to the past few 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 EGLE 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 2020, 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 not significant (R^2 = 0.002; p = 0.838).

Interestingly, the relationship between TP and precipitation is much improved since 2013 (R^2 = 0.382; p = 0.102) but is still not statistically significant. This relationship is based on only 8 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.

Appendix D. Wetland Sediments.
Appendix D.

PROJECT CLARITY
2020 Wetland Sediment Study

January 2021

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1. Overview

One of the key elements of the restoration plan in Project Clarity was obtaining two properties in the Macatawa watershed to create/restore wetlands. Created wetlands are often used to help detain flow and remove both nutrients and sediments before they reach a receiving water body. They can range in size from very small to very large, and depending on their age, design, sediment type, and biotic structure, can be very effective in retaining pollutants (Guardo et al. 1995, Fink and Mitsch 2004, Kadlec and Wallace 2008).

The two properties acquired for wetland creation/restoration are referred to as the Middle Macatawa and Haworth properties. Project 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) started soon thereafter, but was terminated in April 2019 given the limited value of the information provided. The monitoring effort was replaced with sampling sediment within the two restored wetlands to determine how much phosphorus (P), and what form of P, is accumulating in each system. This information is a way to assess the effectiveness of these restored wetlands at trapping and retaining P. This appendix provides an overview of the P dynamics in the wetlands to date.

2. Methods

2.1 Overall watershed 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.

2.1.1 Monitoring & Data Collection

Sediment Analysis: Sediment cores were collected from the restored Haworth and Middle Macatawa wetlands on 1 and 8 October 2020, 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 1 and Figs. 1, 2. When sufficient overlying water was present at coring sites, surface water was sampled with minimal disturbance prior to sediment coring in order to assess general water quality parameters, as well as measure total phosphorus (TP) and soluble reactive phosphorus (SRP). 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. Sediments were collected in triplicate using a modified piston coring apparatus (Fisher et al. 1992; Davis and Steinman 1998). 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. Additional water was collected via grab sample from either a pool of standing water within the wetland (Middle Macatawa, near site 6) or the river upstream of the wetland (Haworth) for SRP analysis.

Cores were transported to the laboratory on the same day, extruded to isolate the top 10 cm of sediment, and refrigerated at 4°C. Water samples were stored in coolers on ice and were also transported to the laboratory on the same day and refrigerated at 4°C until analysis. 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 experiments, and P fractionation analysis. Water for SRP analysis was syringe-filtered through 0.45-µm membrane filters into scintillation vials; TP and SRP samples were both 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.

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. 2017). 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 1. Site coordinates of post-restoration Haworth (10/1/2020, North Branch) and Middle Mac (10/8/2020, Macatawa River) wetland sediment coring.

Wetland	Site	Latitude (°N)	Longitude (°W)
Haworth	1	42.74591	86.08409
	2	42.74394	86.08308
	3	42.74405	86.07838
	4	42.74545	86.07757
Middle Mac	5	42.77902	86.00584
	6	42.77810	86.00515
	7	42.77702	86.00864
	8	42.77771	86.01033

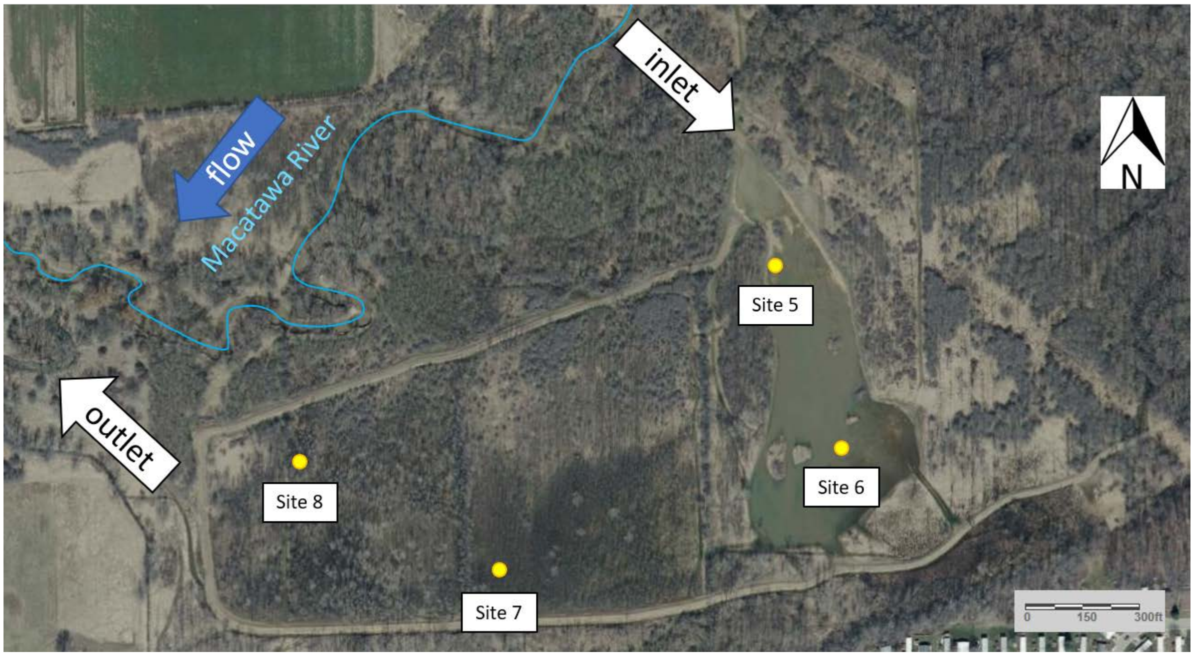

Figure 1. Middle Macatawa wetland sediment coring sites (n=4 total; sites 5, 6 are in the excavated restoration area and sites 7, 8 are in the unexcavated area).

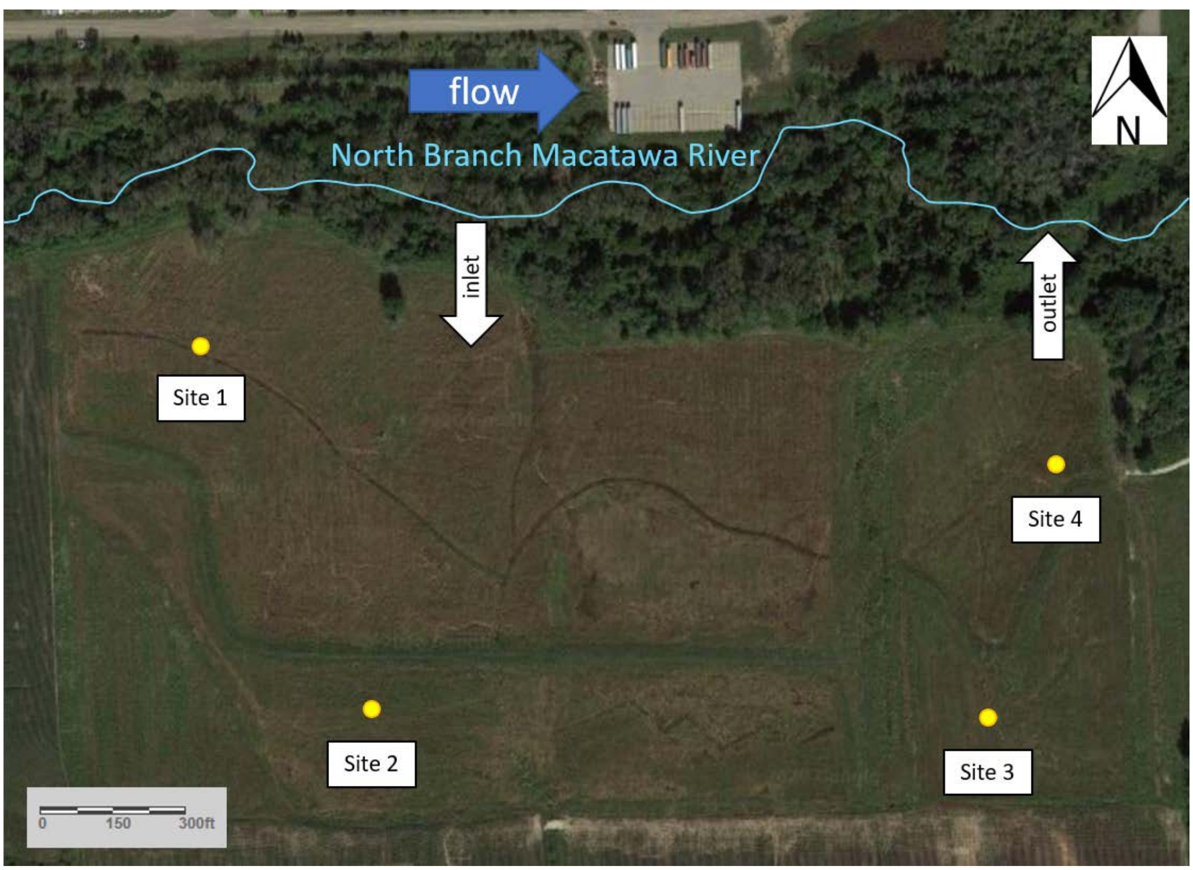

Figure 2. Haworth wetland sediment coring sites (n=4 sites total, 1 site per wetland cell).

2.1.2 Data Analysis

Monitoring of wetland sediment started in 2019; as a consequence, there are no prior data to assess annual trends. Rather, sediment data analysis for 2020 focuses 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.

3. Results and Discussion

3.1 Middle Macatawa Property

Overlying water was present in insufficient quantity throughout the four Middle Macatawa wetland sites for water quality analyses. Sediment TP and OM in collected cores varied depending on where they were sampled in throughout the restored wetland. As seen in 2019, surface (0-10 cm) sediments at sites 5 and 6 in the excavated restoration area had lower TP (386-402 mg/kg dry weight) and OM (3-4%) compared to sites 7 and 8 in the unexcavated area, which were much higher (1383-1402 mg/kg dry wt.; 10-12%; Table 2, Fig. 3A,B). These trends were expected due to the removal of sediment-laden P and associated organic matter at the excavated restored sites, and were consistent with other wetland restoration sites in west Michigan that have been dredged (Oldenborg and Steinman 2019).

Comparison of soil TP and maximum P adsorption values (S_{max}) yields an indicator of whether the soils are saturated with P or have the capacity to adsorb more. Although S_{max} values were much greater at the unexcavated sites than the excavated sites (Table 2), values were still below soil TP, although not by much, suggesting these soils are close to P saturation. Among the unexcavated sites, site 5 has considerable additional sorption capacity, but one of the three replicate cores at site 6 indicated P oversaturation, whereas the other two replicates were still undersaturated (Table 2).

EPC_0 was lower at excavated sites 5 and 6 (mean = 0.002 and 0.014 mg/L) than at sites 7 and 8 (mean = 0.032 and 0.026 mg/L; Table 2). Normally, EPC_0 would be compared to the overlying water to determine if the sediments acted as a P sink (EPC_0 < water SRP) or a P source (EPC_0 > water SRP). However, there was no overlying water present in 2020. Instead we compared EPC_0 with the measured stormflow SRP concentrations collected at the upstream site during post-restoration monitoring and on the 2020 sediment coring day. In both cases, EPC_0 was lower than all measured SRP concentrations (Fig. 4A,B). Although EPC_0 was lower than water SRP at all sites, EPC_0 values and SRP concentrations were much closer to each other and much higher overall at the unexcavated sites (7 and 8), suggesting greater potential for P release from those sediments in the future.

Fractionation of surface sediments showed that the loosely-sorbed NH_4Cl-P fraction, which is easily released from sediments, was found in minimal (0.04-0.18 µg/g) concentrations at all sites (Fig. 5). In contrast, the NaOH-P fraction (Fe- and Al-bound P) was consistently the largest fraction (145-479 µg/g) in the Middle Macatawa wetland sediment at all sites (Fig. 5); this fraction is redox sensitive so if the water column were to become anoxic, the iron-bound P could be released into the water column (Orihel et al. 2017).

Overall, the trend observed in sediment P studies from 2019 to 2020 indicates that Middle Macatawa wetlands contain high levels of P and these wetland sediments may have limited capacity or binding sites to uptake additional P (Hassett et. al 2020). Sediment TP concentrations generally remain similar to those seen in previous years and are close to calculated maximum sorption values (Table 2). This could be potentially problematic if continued P loading, from either excess runoff or mineralization of decomposing vegetation, results in the saturation of P binding sites in the soils. Continued monitoring will allow us to make that determination, and whether management practices (such as vegetation or sediment removal in unexcavated areas) would be justified to maintain efficient functioning of this wetland system.

Table 2. Middle Macatawa wetland site and mean (1 SD) sediment characteristics from cores collected 10/8/2020. All sediment is homogenized from near surface depth (0-10 cm, except Site 8 = 0-9 cm). TP = total phosphorus, OM = organic matter, EPC_0 = equilibrium phosphorus concentration, S_{max} = phosphorus sorption maximum. Site 5 core replicate C data were so small (i.e., sediment sample adsorbed so much P during isotherm lab processing) that sediment adsorption vs. added P concentration relationship could not be estimated; EPC_0 and S_{max} results are listed as NA = not applicable and are not included in wetland mean.

Site	Standing Water, cm	Core	Sediment TP (dry), mg/kg	Sediment OM, %	$EPC_{0,}$ mg/L	$S_{max,}$ mg/kg
5	0	A	402	3%	0.002	711.2
		B			0.002	763.7
		C			NA	NA
6	0	A	386	4%	0.013	322.2
		B			0.018	466.3
		C			0.012	474.3
7	0	A	1383	10%	0.032	1561.2
		B			0.032	1620.4
		C			0.032	1617.9
8	0	A	1402	12%	0.026	1455.4
		B			0.026	1562.1
		C			0.026	1560.3
Wetland Mean			893	7%	0.020	1101.4
SD			577	5%	0.013	583.2

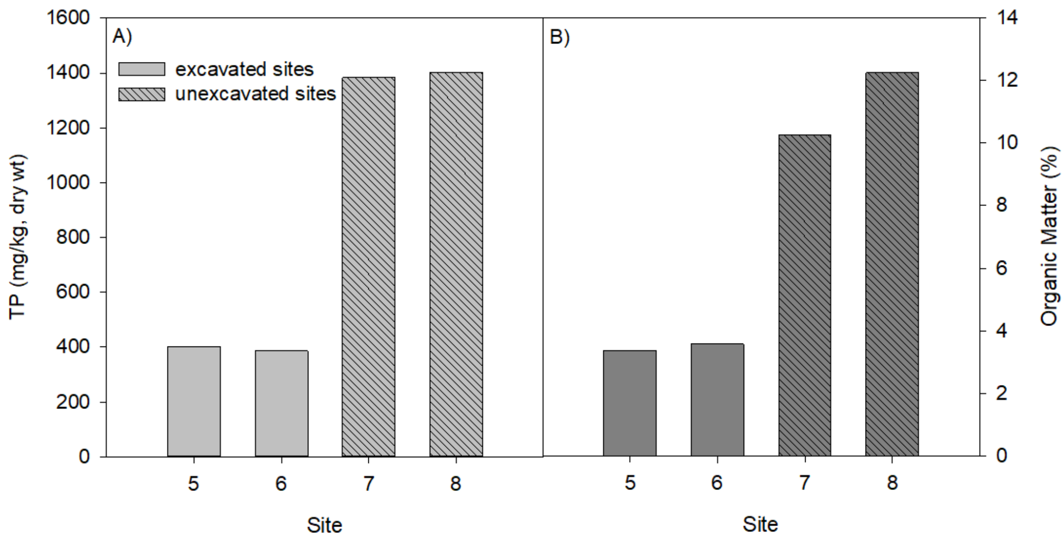

Figure 3. Middle Macatawa post-restoration wetland core surface sediment TP (A) and organic matter (B).

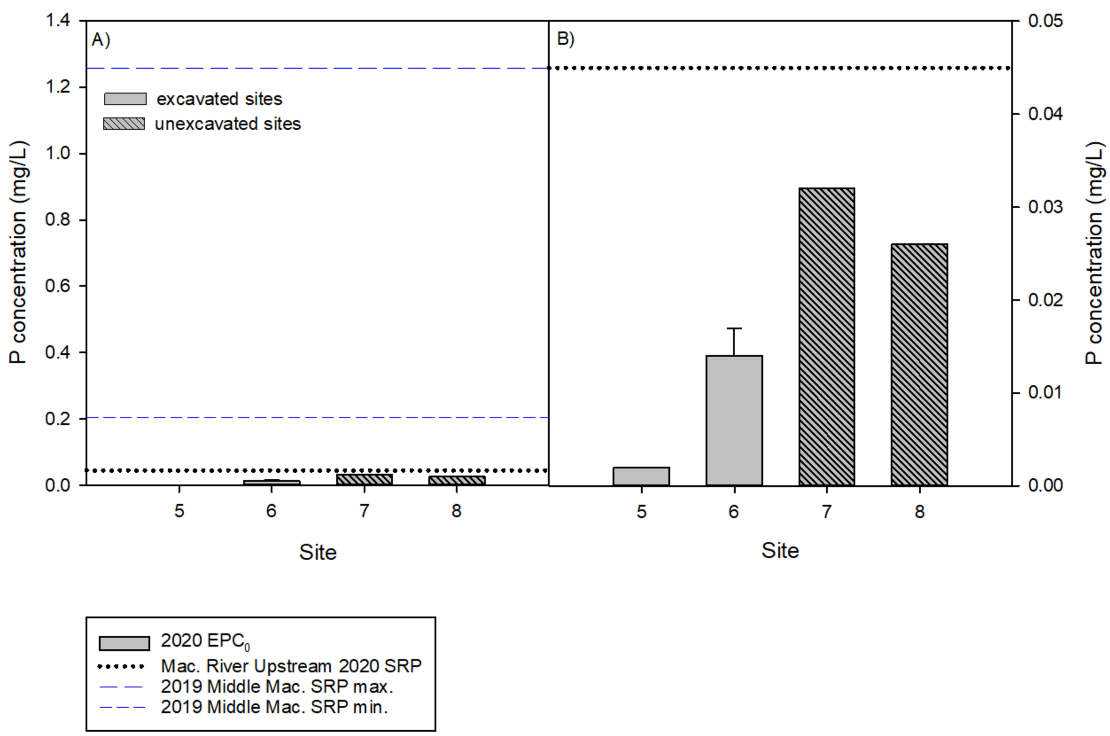

Figure 4. Middle Macatawa post-restoration wetland mean (1 SD) equilibrium P concentrations (EPC_0) in cores (n=3). Dotted reference lines represent grab sample at the restored Middle Macatawa wetland pond on 10/8/2020 sediment coring day (0.045 mg/L). Dashed reference lines represent maximum and minimum (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 is a blow-up of panel A to show greater detail; note the change of y-axis scale and that the minimum Macatawa Upstream stormflow SRP are above the displayed scale. No error bars for sites 5, 6, and 8 due to lack of variation in replicates (see Table 2).

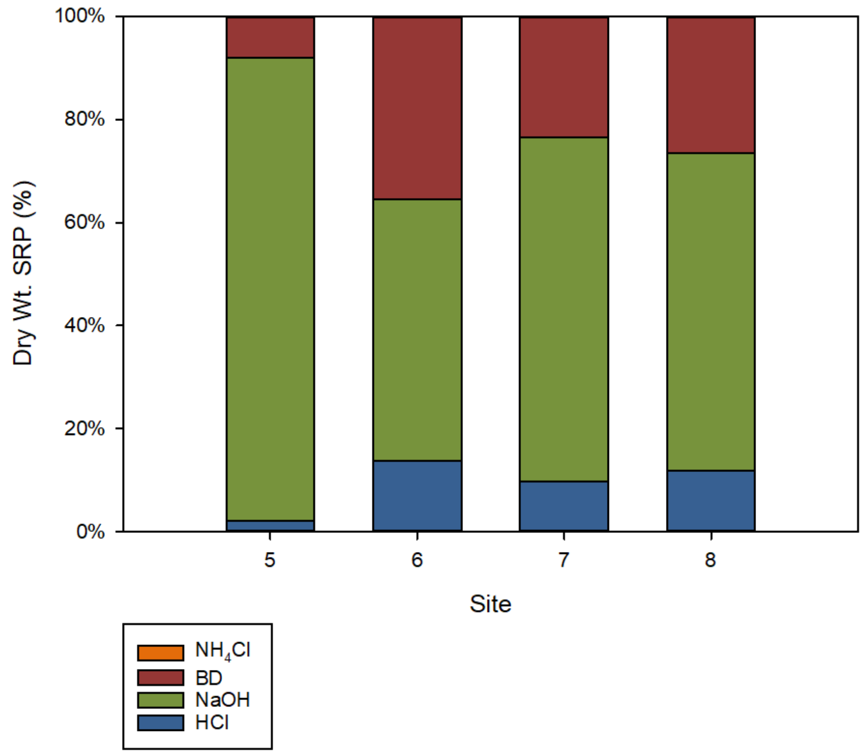

Figure 5. Middle Macatawa post-restoration wetland surface (top 10 cm) sediment P fractions shown as stacked columns by site. Note that SRP concentrations from NH_4Cl (loosely sorbed P) are too small to appear in this figure.

3.2 Haworth Property

Similar to the Middle Macatawa wetland, there was insufficient overlying water throughout the four Haworth wetland sites for water quality analyses. Sediment TP at Haworth was about one-third of that at the unexcavated sites in the Middle Macatawa wetland, ranging 199-440 mg/kg (Table 3, Fig. 6A). Haworth sediment OM ranged 2-4%, very similar to the excavated restoration area at the Middle Macatawa wetland (Table 3, Fig. 6B).

EPC_0 ranged 0.006-0.038 mg/L among all Haworth cores, with an average of 0.022 mg/L (Table 3). Since no standing water was present, EPC_0 values were compared to 2019 mean base and storm flow SRP data (0.021 and 0.077 mg/L, respectively), as well as a single grab sample collected during the 10/1/2020 sediment coring day (0.015 mg/L), all collected at the North Branch tributary monitoring site (i.e., the water that will flood the wetland during stormflow). EPC_0 values exceeded the mean 2019 SRP and the single 2020 SRP sample at sites 1, 2, and 4 (Fig. 7). However, these samples were taken at baseflow, when river water is unlikely to enter the wetland. During stormflow, when SRP values increase 4 to 10× that of baseflow (Fig. 7; Steinman et al. 2018), the sediment is likely to serve as a P sink.

S_{max} values tended to be similar to or lower than soil TP concentrations, suggesting these cells, with the exception of cell 3, are either close to or already saturated with P. Based on fractionation results, 2 of 4 sites were dominated (53-77%) by the NaOH-P fraction (Fe- and Al-bound P), continuing the trend observed in 2019. The remaining Haworth sites were dominated by the less redox-sensitive (but more pH sensitive) HCl-P fraction (45-51%; Fig. 8).

Overall, Haworth sediments show an interesting contrast, with EPC values suggesting the capacity to adsorb more P but Smax values suggesting P saturation at most sites. Site mean sorption maximum values appear to have decreased at all sites from 2019 by ~47-62% (data not shown). Continued monitoring of these restored wetland sediments will provide more information with which to assess whether this is a trend or an indication of interannual variability.

Table 3. Haworth wetland site and mean (1 SD) sediment characteristics from cores collected 10/1/2020. 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**	**Sediment TP (dry), mg/kg**	**Sediment OM, %**	**EPC_0, mg/L**	**S_{max}, mg/kg**
1	0	A	440	4%	0.038	465.2
		B			0.035	329.9
		C			0.034	323.2
2	0	A	385	3%	0.018	212.4
		B			0.022	295.9
		C			0.025	255.5
3	0	A	199	2%	0.007	226.4
		B			0.006	272.7
		C			0.009	251.5
4	0	A	272	3%	0.026	236.5
		B			0.023	202.7
		C			0.017	275.2
Wetland Mean			324	3%	0.022	278.9
SD			109	1%	0.012	63.0

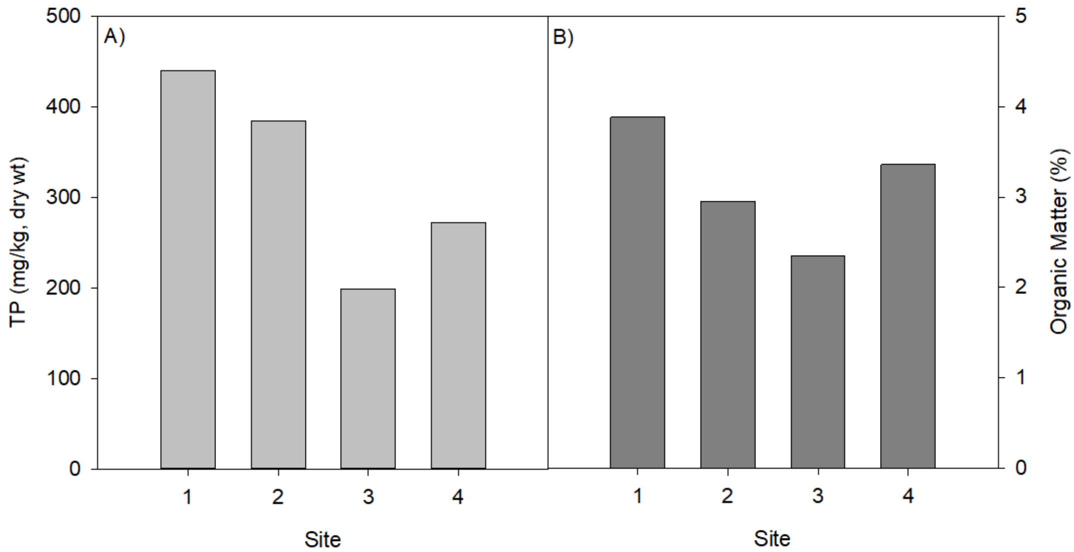

Figure 6. Haworth post-restoration wetland sediment core surface sediment TP (A) and organic matter (B).

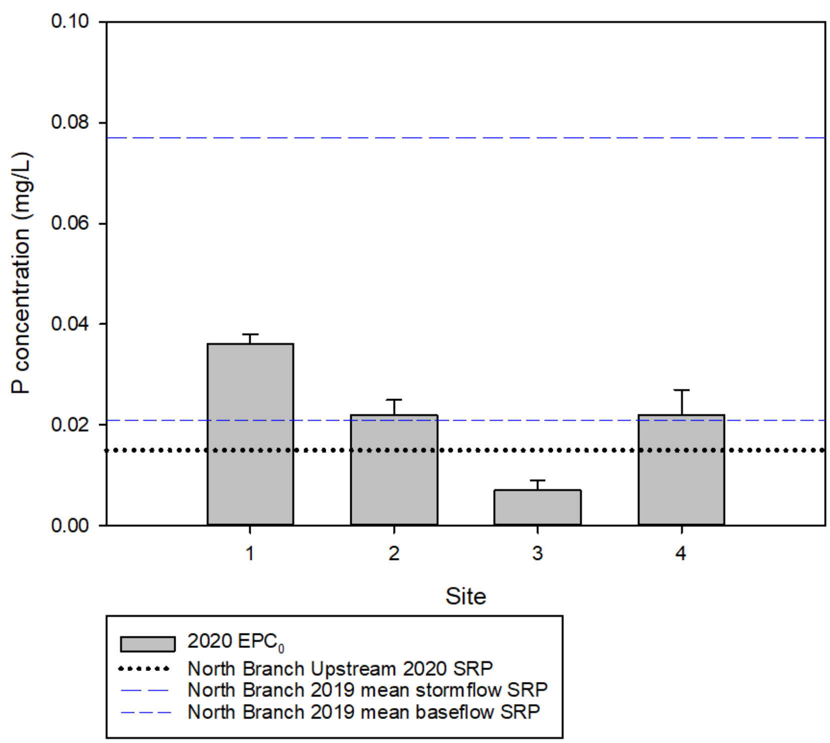

Figure 7. Haworth post-restoration wetland mean (1 SD) equilibrium P concentrations (EPC_0) in non-sterilized live cores (n=3). Dotted reference line represents grab sample at North Branch Upstream tributary monitoring site on 10/1/2020 sediment coring day (0.015 mg/L). Dashed reference lines represent mean baseflow (0.021 mg/L) and stormflow (0.077 mg/L) SRP concentration collected at the North Branch Upstream tributary monitoring site in 2019, the most recent 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.

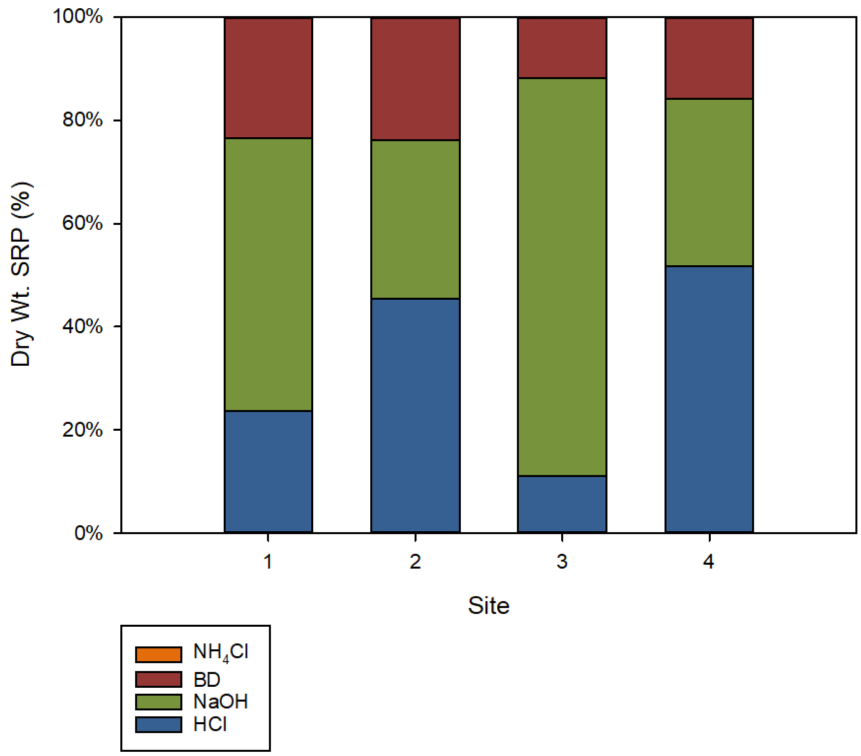

Figure 8. Haworth post-restoration wetland surface (top 10 cm) sediment P fractions shown as stacked columns by site. Note that SRP concentrations from NH_4Cl (loosely sorbed P) are too low to appear in this figure.

4. Summary

Analyses of the P biogeochemistry in the undisturbed sediments of the Haworth and Middle Macatawa wetlands indicate that they retain the capacity to serve as P sinks. However, there are indications that some areas are close to reaching P saturation, which may convert into P sources to overlying water. Continued monitoring will help determine the P status of these sediments, and whether or not management intervention is recommended to replenish their P binding ability.

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