White Lake 2024 Monitoring Report

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Introduction

The White Lake Association retained the Steinman Lab at Grand Valley State University's Annis Water Resources Institute (AWRI) to monitor White Lake during the 2024 sampling season. White Lake was delisted as a Great Lakes Area of Concern in 2014, and community residents in the Whitehall/Montague (MI) area are interested in how the lake is performing over the past decade. A modest monitoring program was recommended to generate a baseline on White Lake's water quality status.

This report provides details on the 2024 sampling campaign and compares the results to prior water quality data.

Methods

Four White Lake water quality monitoring sites were sampled by AWRI in 2024: two sites in the main basin of the lake as previously established for historic monitoring by White Lake Association volunteers and reported on the Michigan Clean Water Corps website (MiCorps; GLC-HRWC 2024a, 2024b), as well as two additional sites near the Lake Michigan channel and the west side of the causeway connecting Whitehall to Montague (Table 1; Figure 1).

Lake sites were monitored monthly via jonboat from May through October 2024 and were visited in sequence from downstream to upstream, beginning at Site 1 (channel) and ending at Site 4 (causeway). After completing the first sampling event in May, technicians noticed that GPS coordinates for Site 2 (White Lake West on MiCorps) had resulted in a shallower sampling location than what was reported in previous sampling years; Site 2 was adjusted from June onward, and the updated site coordinates are reported in Table 1 and Figure 1.

Water transparency was measured as Secchi disk depth, followed by water collections via grab sampling at surface depth and via VanDorn sampler at near-bottom lake depth. A mid-depth water sample was collected to be later composited along with surface and bottom depth subsamples as a whole water column measurement of the phytoplankton community. Water samples were collected in acid-washed bottles and stored on ice until they were subsampled, processed, and preserved in the laboratory as detailed below, usually within 4 hours. Duplicate water quality samples were collected from 1 site per month. Additional near-surface water at each site was separately collected for *E. coli* measurements using unused, sterilized plastic specimen containers with 100 mL fill lines and hinged lids. Duplicate *E. coli* samples were collected from 1 site per month; additionally, a DI (deionized water) field blank and DI laboratory blank were measured monthly for quality control. Physicochemical water quality parameters were measured at the near-surface and at each meter of depth per site using an EXO2 multiparameter sonde (YSI, Inc.; Yellow Springs, OH) equipped with probes for measuring water temperature, dissolved oxygen, pH, specific conductivity, and turbidity.

Upon return to the laboratory, bulk water samples were gently inverted for homogenization and subsampled for the analysis of total phosphorus (TP), soluble reactive phosphorus (SRP), nitrate (NO_3^-), ammonia (NH_3), total Kjeldahl nitrogen (TKN), chlorophyll *a* (chl *a*), microcystin, and phytoplankton community composition. SRP and NO_3^- samples were filtered using acid-washed 0.45 µm nylon syringe filters. NH_3 and TKN samples were preserved via sulfuric acid acidification. TP, SRP, NH_3, and TKN samples were refrigerated at 4°C while NO_3^- was frozen until analysis via Seal AQ2 discrete autoanalyzer (U.S. EPA 1993). Any values below detection were reported as ½ of their respective method detection limits.

Chl *a* samples were vacuum filtered through GF/F glass fiber filters. Chl *a* filters and residue were frozen until analysis via acetone extraction and measurement via spectrophotometer. A 250 mL subsample of water from surface, middle, and bottom depths per each site were composited as a 750 mL total volume and preserved with 7.5 mL of Lugol's iodine solution to create a final 1% solution to preserve phytoplankton communities. Phytoplankton was identified to genus or species light microscopy; biovolume was estimated using standard methods (Hillebrand et al. 1999).

A separate 1 mL subsample was collected from each sample and frozen until analysis using a high-sensitivity enzyme-linked immunosorbent assay (ELISA) kit for microcystin (Envirologix; Portland, ME), which serves as a useful screening tool if microcystin is present in the lake. Advisories for microcystin exposure have been developed by the World Health Organization (WHO) and U.S. EPA. For drinking water, the WHO advisory is >1 µg/L and EPA is >1.6 µg/L for adults and 0.3 µg/L for infants and pre-schoolers; for recreational use, WHO is >20 µg/L and EPA is >8 µg/L (WHO 2017; U.S. EPA 2019). Since White Lake is not a drinking water source, we applied only the recreational criteria.

E. coli samples were analyzed via the IDEXX Colilert-18® method. Briefly, substrate powder was added to aliquots and incubated in Colilert Quanti-Tray®/2000 at 35°C for 18 hours, then trays were exposed to long-wave ultraviolet light and blue tray wells were counted as positive. The number of positive wells was the most probable number (MPN) per 100 mL, and 300 colony-forming units (cfu) per 100 mL is a recognized upper limit as being safe for total body contact in the state of Michigan (MCL 323.1062 of 2006 et seq.).

Historic water quality data were retrieved from the MiCorps website from White Lake (East = AWRI site 3) and White Lake (West = AWRI site 2) sites for sampling years 2014-2023 (GLC-HRWC 2024a, 2024b). Summary and analysis of water quality data were performed using R statistical software (v4.4.1; R Core Team 2023), including the dplyr and ggplot2 packages (Wickham et al. 2023, Wickham 2016).

Table 1. White Lake site coordinates and maximum depth across 2024 sampling events.

Site	Latitude (°N)	Longitude (°W)	Depth (m)
1	43.37542	86.42149	9.5
2	43.37619	86.39542	15.7
3	43.38446	86.37606	15.7
4	43.41340	86.35270	4.5

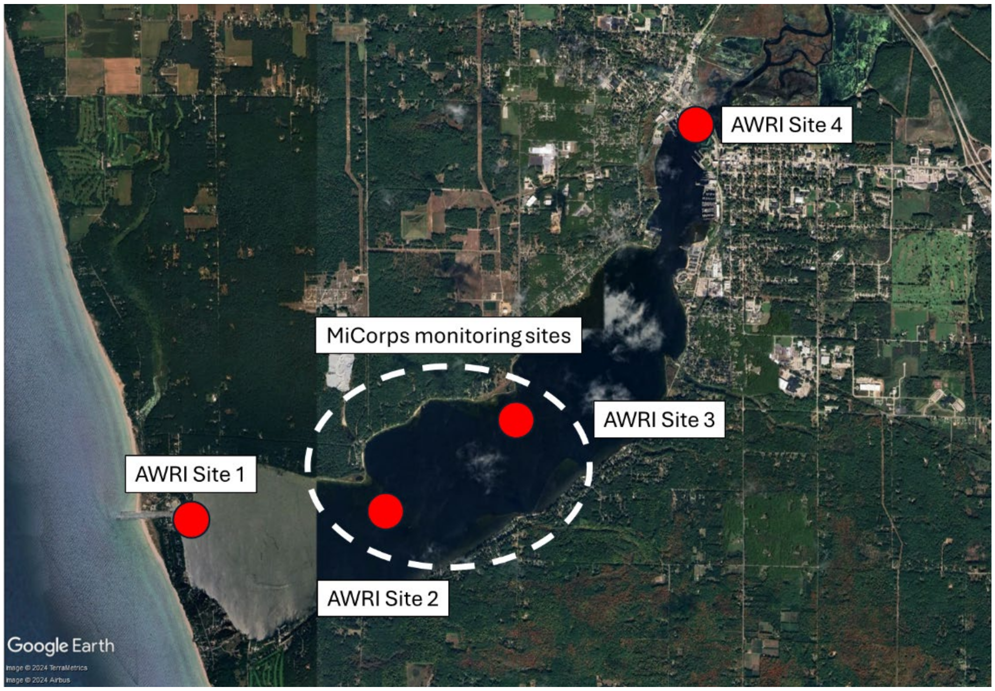

Figure 1. Map of 2024 White Lake monitoring sites. Site 1 = near Lake Michigan channel. Sites 2 & 3 = historic monitoring sites. Site 4 = near White River.

Results

2024 Water Quality

Physical Parameters

Temperature and dissolved oxygen (DO) varied as expected with respect to seasonality and depth. Thermal stratification was apparent from July through September at approximately the 5 to 7 m depth before kinetic mixing resulted in the breakdown of the thermocline by October (Figure 2). Site 4, which was closest to the mouth of the White River, was consistently ~2 °C cooler than the other sites. During the stratified period (summer to early fall), the center of the lake (sites 2 & 3) experienced near hypoxic conditions with DO <2-3 mg/L (Figure 3). These conditions persisted at depths of 7 m and below, representing a substantial fraction of the water column. During the spring and early summer, DO tended to be higher at site 1, where Lake Michigan water was likely mixing with White Lake water. By October, however, this trend reversed and DO was lowest at site 1 and highest at site 4. Across all depths, pH in the lake ranged from 7.13 to 8.79, with surface waters having generally higher pH (i.e., more basic) and deeper waters being more neutral. This pattern is likely related to photosynthesis by phytoplankton in the photic zone (depths with sufficient light penetration to support photosynthesis), as the process consumes hydrogen ions (H^+).

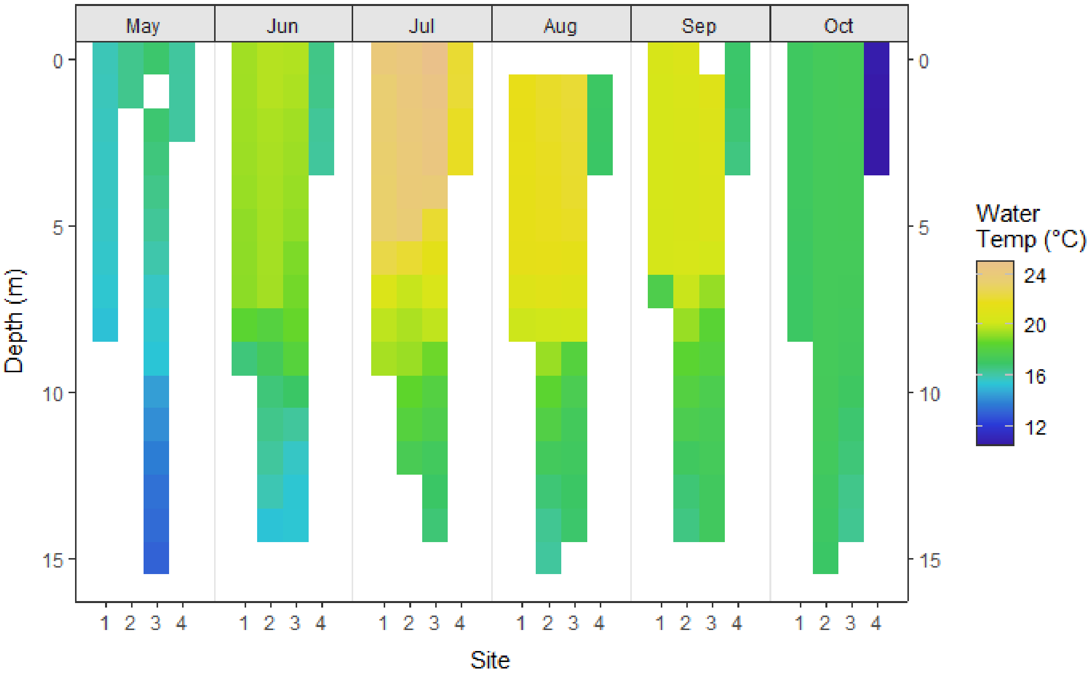

Figure 2. White Lake 2024 water temperature (°C) at ~1-meter depth intervals.

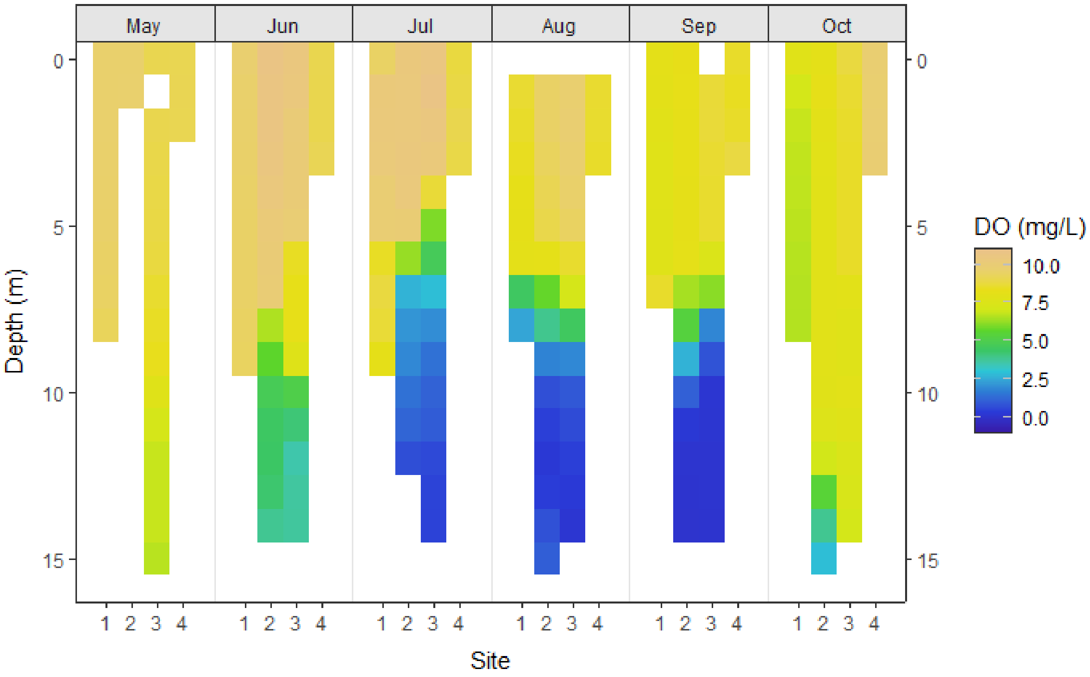

Figure 3. White Lake 2024 water column dissolved oxygen (mg/L) at ~1-meter depth intervals.

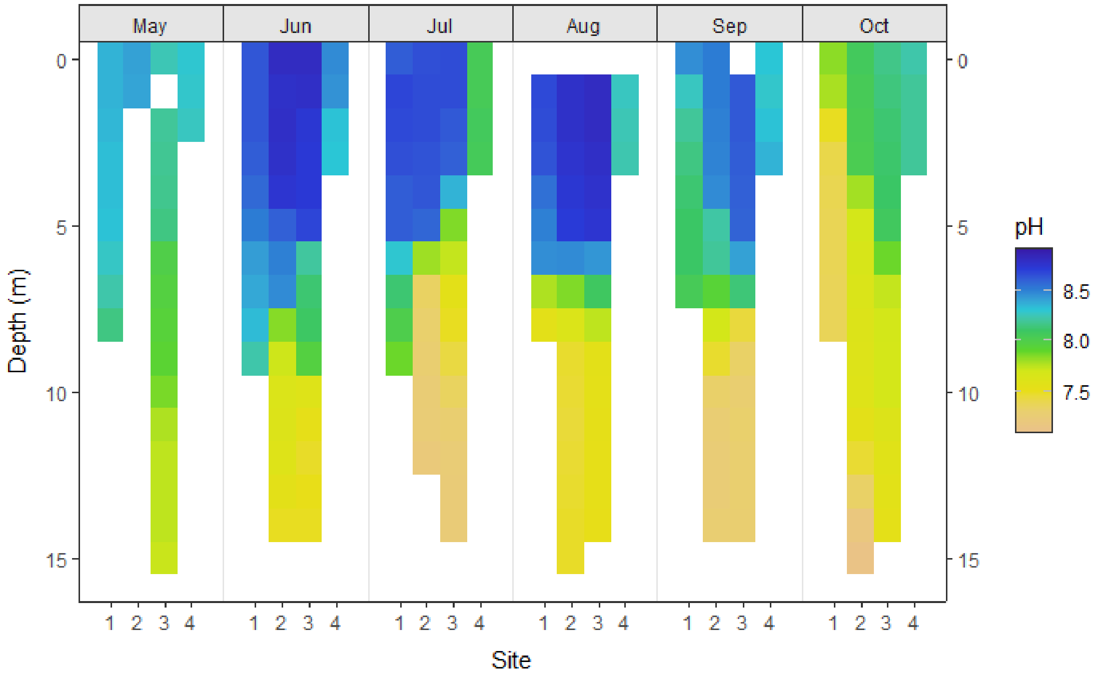

Figure 4. White Lake 2024 water column pH at ~1-meter depth intervals.

Specific conductivity ranged from 331.8 to 429.6 µS/cm (Figure 5), which is typical for West Michigan drowned river mouth lakes (Mader et al. 2023). Conductivity was consistently highest at site 4, indicating inflow of dissolved ions from upstream in the watershed. At site 1 (closest to the Lake Michigan channel), the decrease in conductivity at near-bottom depth in June, July, and September is likely due to low ion-water intrusion from Lake Michigan into White Lake.

White Lake showed low turbidity throughout the sampling season, with values below 5 FNU (Formazin Nephelometric Units) across all sampling sites, dates, and depths (Figure 6). We did observe limited spatial and temporal variation across all sites; the highest turbidity values were generally observed at site 4 (near the river mouth) and in August. Secchi disk depths typically correlate inversely with turbidity (as turbidity increases, Secchi depths decline), and that is exactly what we found in White Lake, as May measurements with very low turbidity had Secchi depths that reached up to 4.15 m depths, and during August, when turbidity was higher, Secchi depths declined to 1.72 m (Figure 7).

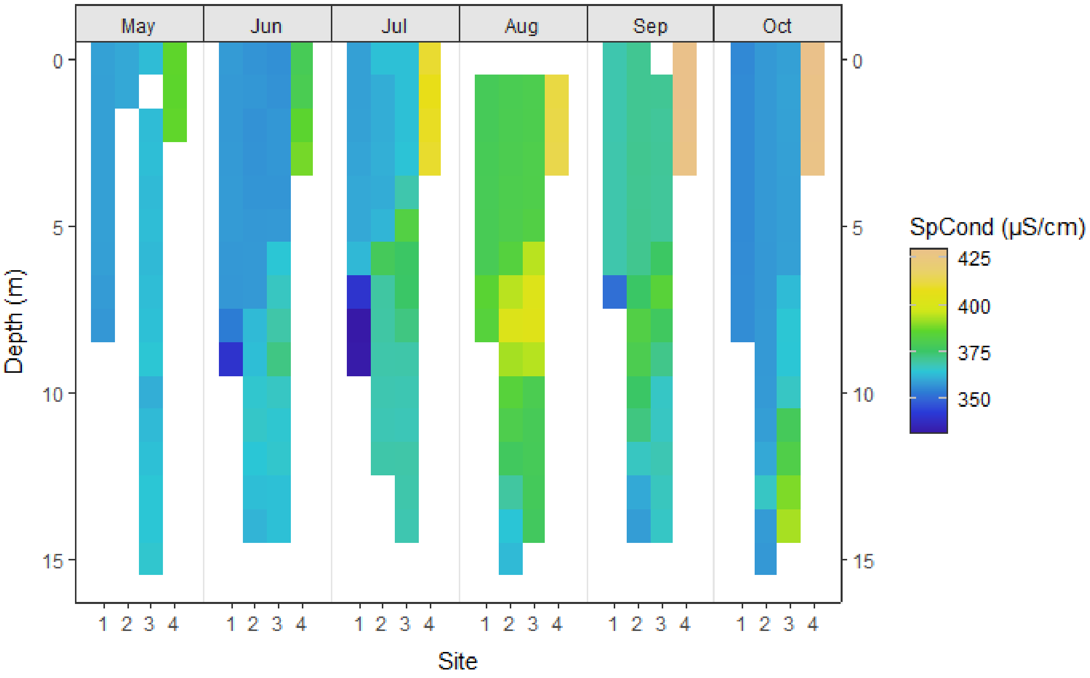

Figure 5. White Lake 2024 water column specific conductivity (µS/cm) at ~1-meter depth intervals.

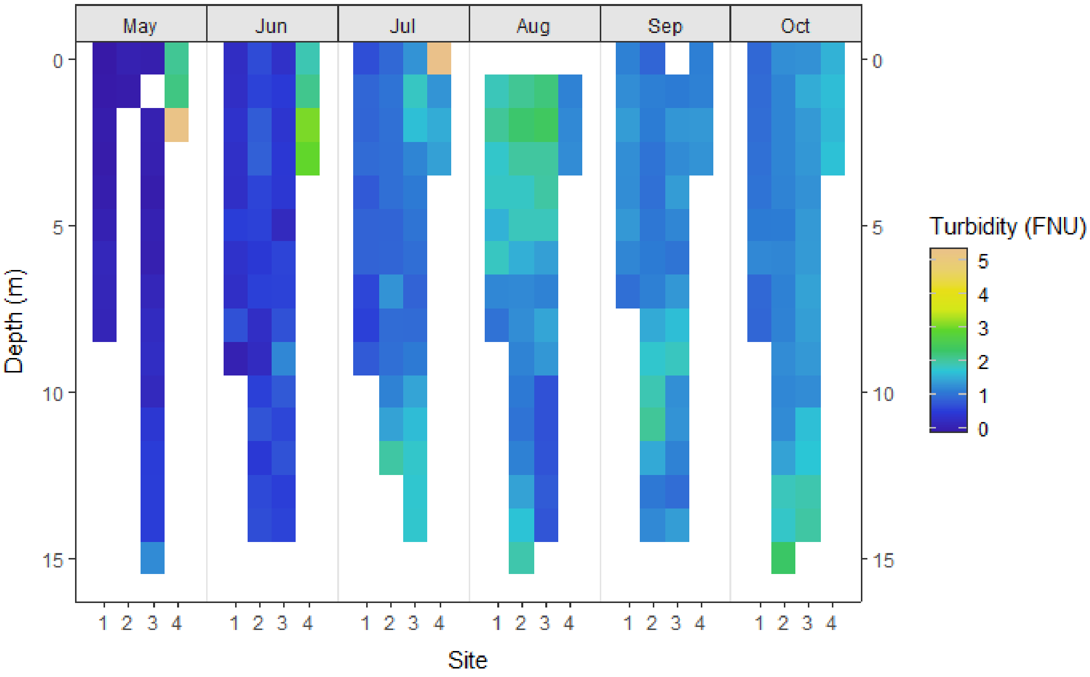

Figure 6. White Lake 2024 water column turbidity (FNU) at ~1-meter depth intervals.

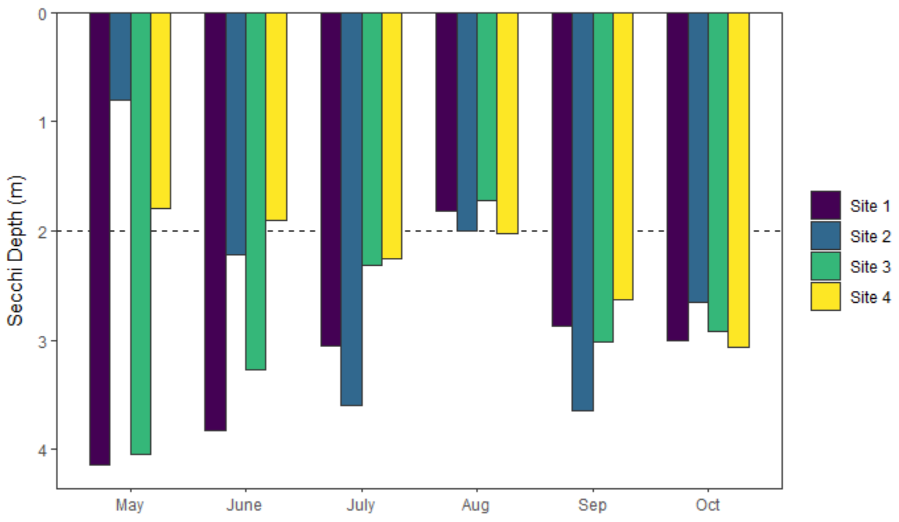

Figure 7. **White Lake 2024 Secchi depth (m). Larger (deeper) values indicate greater water clarity. Dashed line represents ~2 m annual average target for delisting (MDEQ 2014).**

Chemical Parameters

Total phosphorus (TP) concentrations in 2024 ranged from 10 to 51 µg/L at the surface and 10 to 215 µg/L at near-bottom (Figure 8). August and September saw an increase in TP concentrations in the deeper parts of the lake, with Sites 2 and 3 reaching concentrations >150 µg/L during those two months. Observed lake-wide mean concentrations at the surface remained near or below the AOC restoration goal (30 µg/L; MDEQ 2014) from May to September but increased to ~45 µg/L in October.

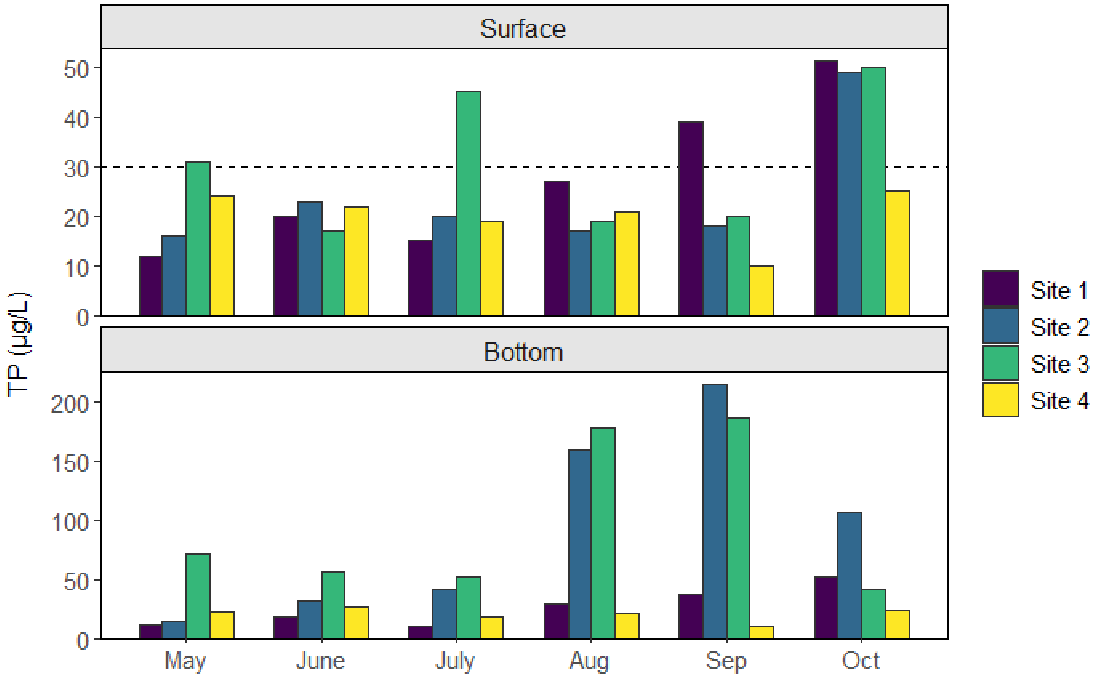

Figure 8. White Lake 2024 water column TP (µg/L) at surface and near-bottom depth. Dashed line denotes 30 µg/L average annual surface concentration threshold for delisting (MDEQ 2014). Note difference in y-axis scales between surface and bottom.

Soluble reactive phosphorus (SRP) varied greatly across seasons, ranging from below detection to 34 µg/L at the surface and to 146 µg/L at near-bottom (Figure 9). At the surface, the lowest concentrations lake-wide occurred in summer (July – August), and highest concentrations occurred during the fall (September – October). Similarly to TP, near-bottom concentrations peaked in late summer/early fall (August – September) and decreased later in the fall. Mean SRP:TP ratios were variable (Table 2), ranging from 10% to 90%, and were noticeably higher in the bottom water of sites 2 and 3 in summer, consistent with internal phosphorus loading from the sediment.

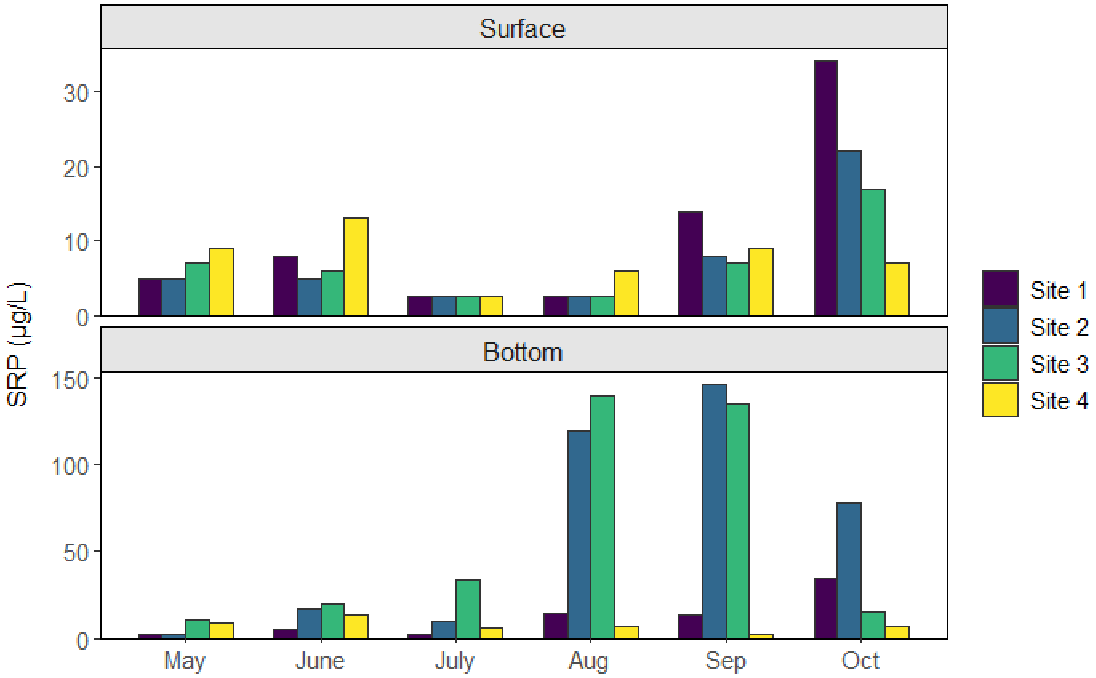

Figure 9. White Lake 2024 water column SRP (µg/L) at surface and near-bottom depth. Note difference in y-axis scales between surface and bottom.

Table 2. Mean SRP:TP ratios from all sampling sites on all dates.

Date (Month)	Depth	Site 1	Site 2	Site 3	Site 4
May	Top	0.417	0.313	0.226	0.277
	Bottom	0.208	0.179	0.155	0.391
Jun	Top	0.400	0.217	0.391	0.591
	Bottom	0.263	0.531	0.351	0.500
Jul	Top	0.167	0.125	0.067	0.132
	Bottom	0.227	0.244	0.654	0.316
Aug	Top	0.093	0.147	0.123	0.286
	Bottom	0.483	0.744	0.787	0.333
Sep	Top	0.359	0.444	0.235	0.900
	Bottom	0.342	0.679	0.722	0.250
Oct	Top	0.667	0.449	0.363	0.280
	Bottom	0.660	0.736	0.366	0.292

Nitrate (NO_3^-) concentrations were consistently highest at site 4 at the mouth of the White River, reaching a maximum of 460 µg/L in October (Figure 10). Agricultural fertilizer applications near the river are a likely source of the elevated nitrate levels. Across the remaining lake sites, nitrate concentrations varied seasonally with the lowest concentrations observed from summer into late fall.

Ammonia (NH_3) concentrations ranged from below detection to 160 µg/L at the surface (May) to 300 µg/L near the bottom (August; Figure 11), with both spikes occurring at site 3. The elevated ammonia levels in the bottom water of sites 2 and 3 in summer/fall are consistent with release from sediments under low DO conditions. Generally, TKN (sum of ammonia and organic N) ranged from 250 – 640 µg/L and did not vary greatly between surface and near-bottom depths (Figure 12). Spikes outside this range were observed at the surface at site 1 in August (1710 µg/L) and site 3 in October (3260 µg/L) and at near-bottom at site 3 in May (930 µg/L). There was little correspondence between ammonia and TKN, despite ammonia being a component of TKN, suggesting organic nitrogen was an N source in White Lake. It is unknown if the producers in the watershed are adding urea to their fields or if there may be industrial or residential sources of organic nitrogen to White Lake but more investigation may be appropriate.

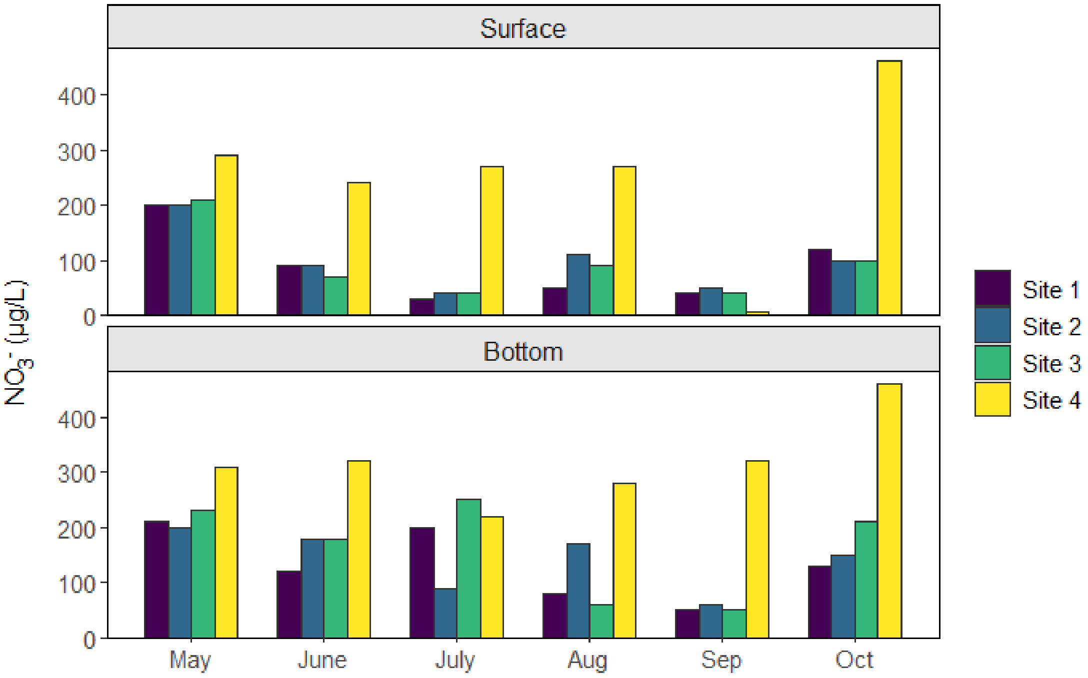

Figure 10. White Lake 2024 water column NO_3^- concentrations (µg/L) at surface and near-bottom depth.

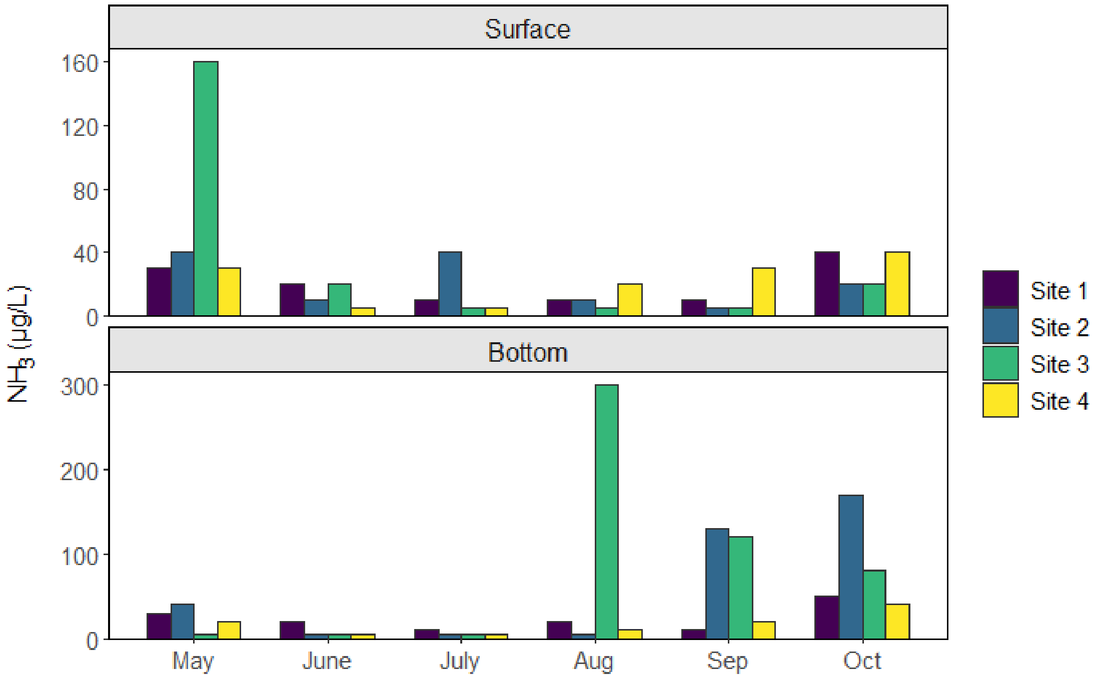

Figure 11. White Lake 2024 water column NH_3 concentrations (µg/L) at surface and near-bottom depth. Note difference in y-axis scales between surface and bottom.

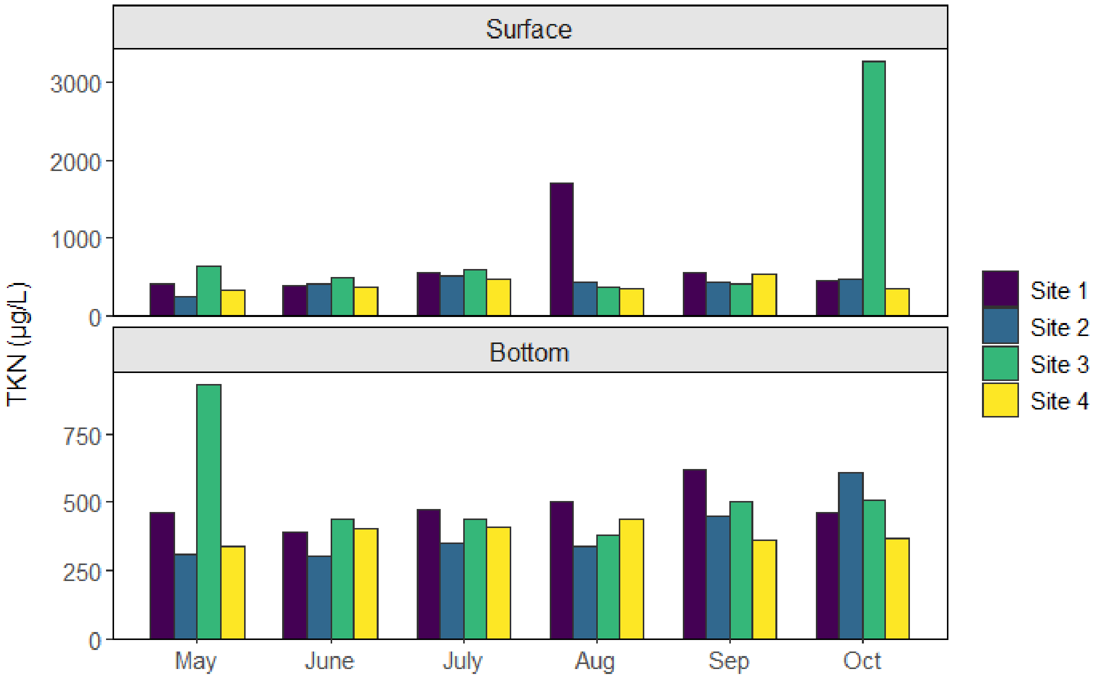

Figure 12. White Lake 2024 water column TKN (µg/L) at surface and near-bottom depth. Note difference in y-axis scales between surface and bottom.

Biological Parameters

Chlorophyll *a* concentrations ranged from 1.01 to 23.81 µg/L at the surface and from 0 to 11.46 µg/L near bottom (Figure 13). At the surface, concentrations were low in May, and spikes were observed from June onward variably across sites 1, 2, and 3. Concentrations remained below 4 µg/L at site 4 throughout the duration of the sampling period. At near-bottom depths, spikes were observed at site 1 (nearest to Lake Michigan channel) in August (8.67 µg/L) and September (11.46 µg/L). Averaged across all sampling events (May – October), mean chlorophyll levels remained below the AOC delisting target of 10 µg/L at all sites and depths with the exception of site 3 surface, which had an observed mean value of 12.10 µg/L over the duration of the sampling period (Table 3).

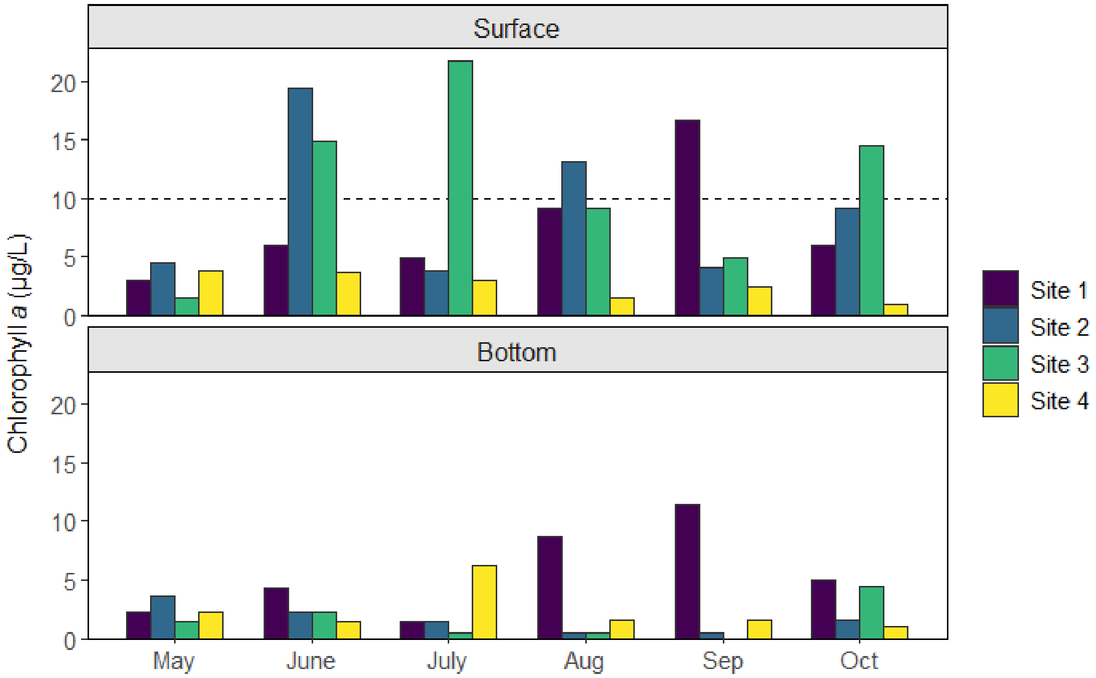

Figure 13. White Lake 2024 water column chlorophyll *a* (µg/L) at surface and near-bottom depth. Dashed line denotes 10 µg/L average annual surface concentration threshold for delisting (MDEQ 2014).

Table 3. Mean chlorophyll *a* concentrations (µg/L) across all sampling events (May – October).

	Annual Mean Chl *a* (µg/L)			
Depth	Site 1	Site 2	Site 3	Site 4
Surface	7.62	9.02	12.10	2.55
Bottom	5.57	1.66	1.53	2.34

Microcystin concentrations ranged from below detection to low throughout the duration of the sampling period, with the maximum concentration of 0.88 µg/L being observed in July at site 3 (Figure 14). All observed concentrations were an order of magnitude less than the EPA recommended threshold for recreation waters of 8 µg/L (U.S. EPA 2019).

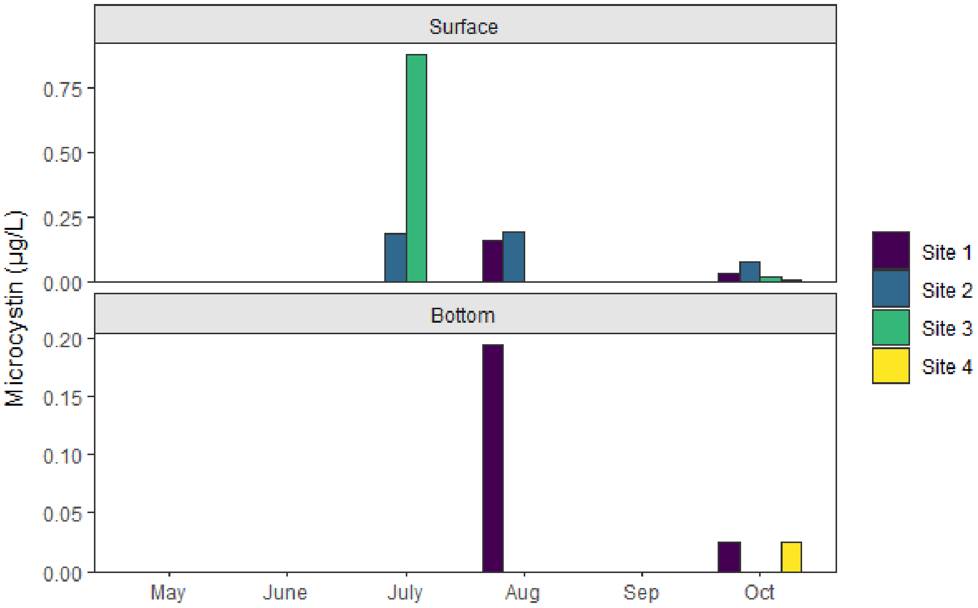

Figure 14. White Lake 2024 water column microcystin concentrations (µg/L) at surface and near-bottom depth. Note difference in y-axis scales between surface and bottom.

Overall, the White Lake phytoplankton community biovolume in 2024 was dominated by diatoms although there was variation in community structure among sites (Figure 15). The Pyrrophyta (dinoflagellates) dominated at Site 1, closest to the Lake Michigan channel, whereas Bacillariophyta (diatoms) dominated at Sites 2-4 in the main lake basin and closest to the White River (Figure 15).

The 10 most abundant phytoplankton taxa based on biovolume composed 91% of all algal biovolume observed in this study (Table 4). The most abundant taxon was the dinoflagellate *Ceratium* (Table 4); cyanobacteria taxa that were observed in this study but not in the top 10 included *Lyngbya*, *Microcystis*, and *Anabaena* – all of which were observed at <1% of total biovolume (data not shown). None of the most abundant taxa (i.e., accounting for >5% of total community biovolume) are known to produce or release cyanotoxins.

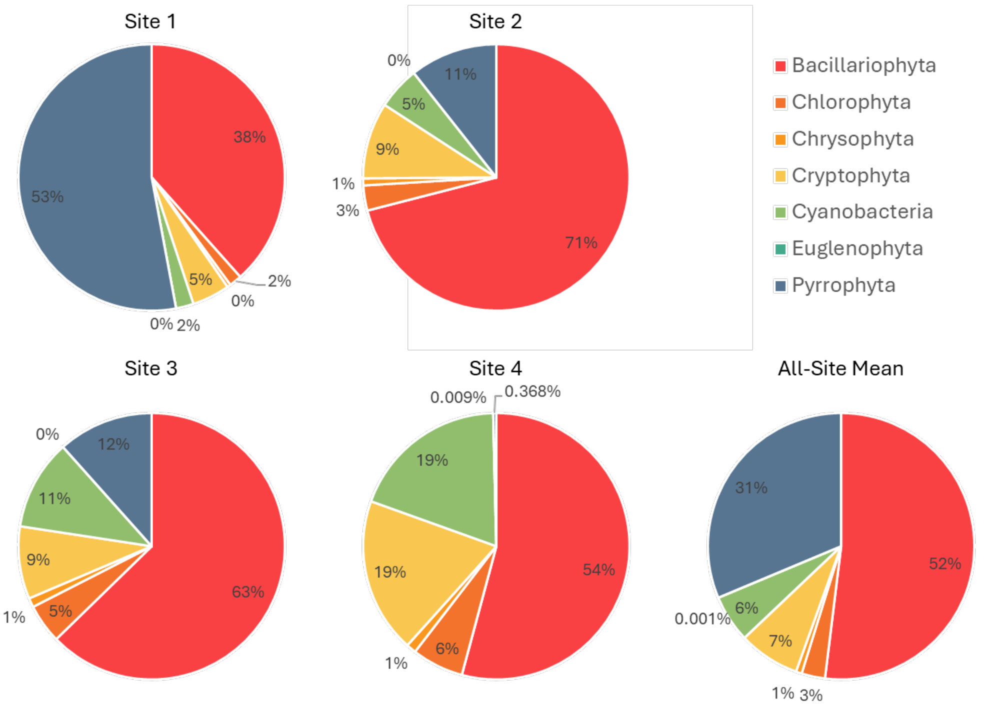

Figure 15. Phytoplankton biovolume (µm³/mL) by Division from the 4 sampling sites and the grand mean. Euglenophyta biovolume is 0% at Sites 1, 2, & 3.

Table 4. Biovolume of the 10 most abundant taxa observed in White Lake.

Division	**Genus**	**Total Biovolume (µm³/mL)**	**Biovolume (%)**
Pyrrophyta	*Ceratium*	3,040,083	28%
Bacillariophyta	*Aulacoseira*	2,268,183	21%
Bacillariophyta	*Fragilaria*	1,761,530	16%
Bacillariophyta	*Asterionella*	738,162	7%
Cryptophyta	*Cryptomonas*	698,830	6%
Bacillariophyta	*Cyclotella*	585,474	5%
Pyrrophyta	*Peridinium*	325,215	3%
Chlorophyta	green coccoids	152,196	1%
Bacillariophyta	*Melosira*	141,712	1%
Cyanobacteria	*Limnothrix*	140,313	1%

E. coli concentrations were consistently highest at site 4, near the causeway, ranging from 45 to 93 colony-forming units (cfu)/100 mL (Figure 15). These levels were well below the Michigan recreation standard of 300 cfu/100 mL (MCL 2006). Concentrations at all other sites were less than or equal to 5 cfu/100 mL across all sampling events.

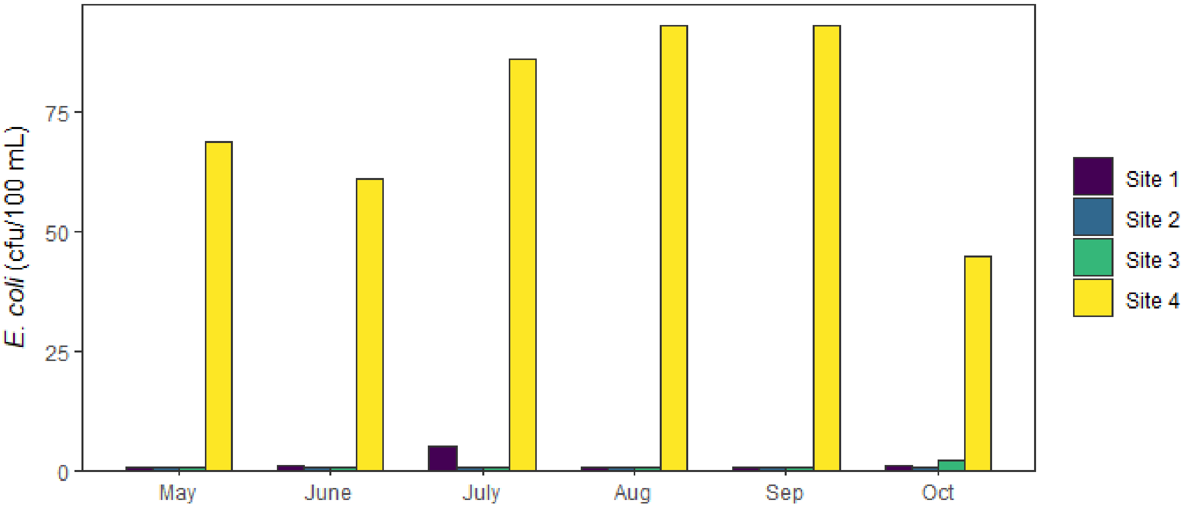

Figure 15. White Lake 2024 surface *E. coli* concentrations (cfu/100 mL).

Historical Data Comparison

Over the past ten years, surface total phosphorus concentrations at sites 2 and 3—measured during spring turnover (April/May) and/or late summer (September)—have consistently remained below 35 µg/L with the notable exception of spring 2015 (Figure 16). Surface TP data from May through September 2024 are consistent with the historical data. However, extended sampling in 2024 (Figure 8) showed that some increases in P may be detectable later into the fall.

Chlorophyll concentrations over the past ten years, measured as an integrated composite sample from twice the Secchi depth to the surface, ranged from 0.5 to 37 µg/L (Figure 17). All measured concentrations in 2024 fell within this range and followed the same general trend of higher levels nearer to the surface and lower levels at greater depths, almost certainly related to light levels.

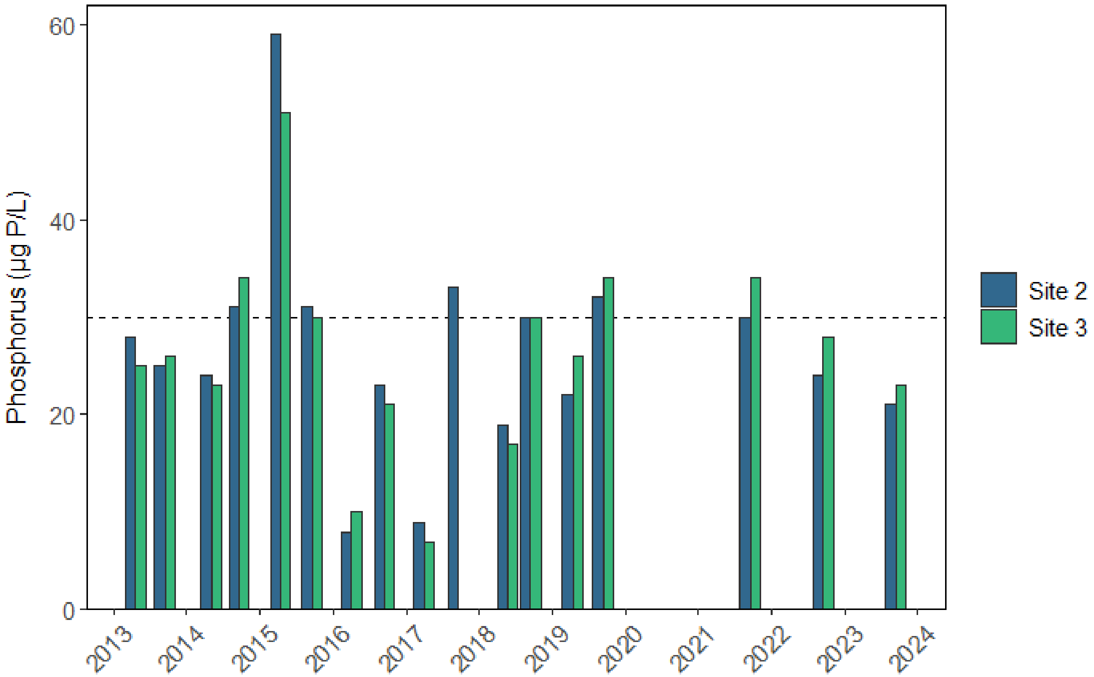

Figure 16. White Lake surface P levels (µg/L) from 2013 through 2023. Data obtained from the MiCorps website. Dashed line denotes 30 µg/L average annual surface concentration threshold for delisting (MDEQ 2014). Long-term data available only at sites 2 and 3.

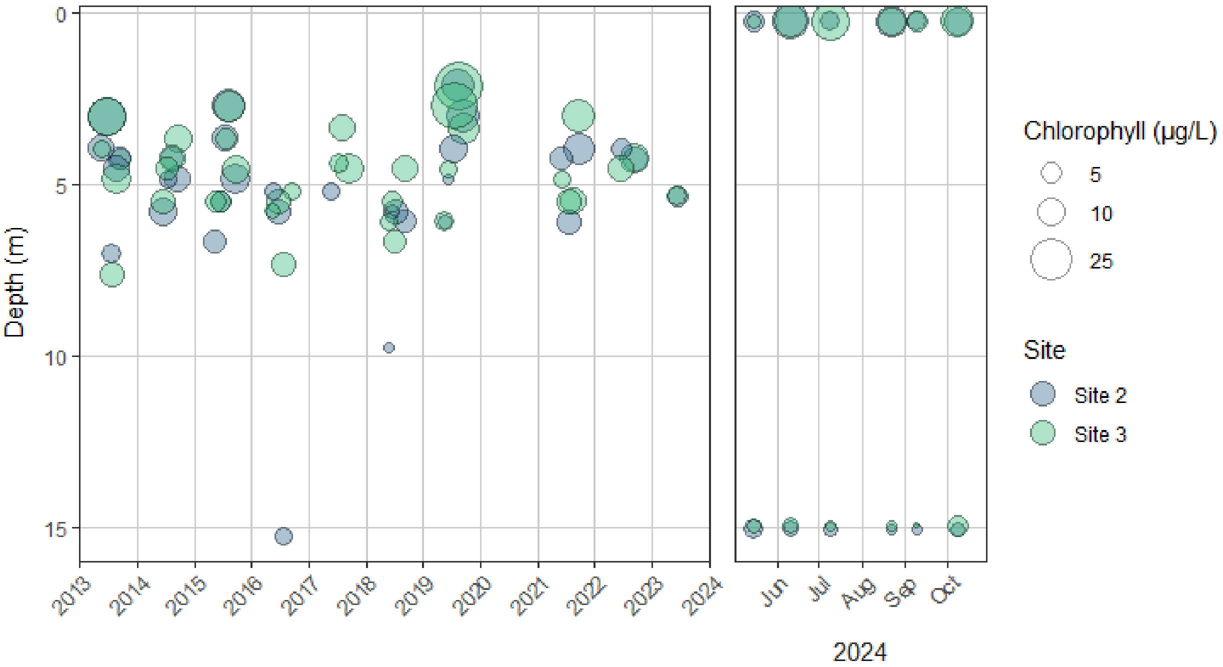

Figure 17. White Lake chlorophyll *a* concentration (µg/L) measured as an integrated composite sample from 2x Secchi depth to surface (2013 – 2023; MiCorps website data) and at surface/near bottom (2024; AWRI data). Long-term data available only at sites 2 and 3.

Discussion

White Lake was officially delisted by the US EPA in 2014 as a Great Lakes Area of Concern (AOC). Its eight beneficial use impairments (BUIs) were determined to have been addressed:

- Restrictions on Dredging Activities- Removed September 2011
- Eutrophication or Undesirable Algae - Removed April 2012
- Degradation of Benthos - Removed June 2012
- Restrictions on Fish and Wildlife Consumption - Removed February 2013
- Loss of Fish and Wildlife Habitat - Removed April 2014
- Degradation of Fish and Wildlife Populations - Removed April 2014
- Restrictions on Drinking Water Consumption or Taste and Odor Problems - Removed March 2014
- Degradation of Aesthetics - Removed March 2014

Because the AOC program addresses only the BUIs present at the time of listing, new impairments such as PFAS or climate change are not covered. Hence, it is important to assess the Lake's health to ensure lake health is not backsliding. The goal of this study was to examine the lake's ecological health in the ice-free period of 2024.

In general, White Lake continues to meet most of its restoration targets and is in good ecological health. The three water quality parameters that are most frequently measured to assess lake condition are water clarity, total phosphorus, and chlorophyll *a*.

Water clarity in 2024, based on Secchi disk depth, usually exceeded 2 m, which is similar to values measured in Muskegon Lake and is typical of drowned river mouth lakes at this latitude in Michigan (Steinman et al. 2008; Mader et al. 2023). Water clarity can be influenced by dissolved organic matter that "stains" the water, as well as particulate matter (usually algae), both of which, when present in high concentrations, can reduce light transmission through the water column. It is unusual for drowned river mouths to have very high light transmission (> 5 m) because of natural staining of waters, usually tannins leaching from vegetation; indeed, many lakes in this region were once (or still) named Black Lake because of this staining effect. Hence, any recent significant decline in light transmission would most likely be due to excessive algal growth, which has not been observed in White Lake (see below).

Total phosphorus is an indicator which on average remained near or below the 30 µg/L restoration target for White Lake. Nonetheless, there is evidence of potential internal P loading (P release from sediments) at the middle, deeper sites in White Lake. Internal loading often occurs in lakes when dissolved oxygen concentrations drop below ~2 mg/L (Steinman and Spears 2020). This usually happens in summer months after lakes stratify and the bottom layer of the lake (hypolimnion) becomes depleted of oxygen because of ongoing respiration by microbes and the inability of atmospheric oxygen to penetrate the temperature barrier between the top and bottom layers. Once DO drops below ~2 mg/L, the microbes seek out new sources of electrons for respiration, and iron is a preferred substrate. Electrons from the oxidized form of iron (Fe^{3+}) are accepted, resulting in the chemical reduction of iron (Fe^{2+}), which in turn results in the release of phosphorus that had been bound to Fe^{3+}. This P can then diffuse out of the sediment (internal P loading) and stimulate algal growth, depending on whether the P is bioavailable or not.

In White Lake, it is evident (Table 2) that a substantial fraction of released P is indeed in the bioavailable form. Although SRP concentrations in the hypolimnion are high (150 µg/L) compared to the surface water, compared to concentrations measured in other drowned river mouth lakes in the region, they are not excessive (e.g., Mona Lake: >800 µg/L; Spring Lake: >1,000 µg/L; Steinman et al. 2004; 2009). Hence, this is not an imminent problem but one that is worthy of continued monitoring.

Chlorophyll *a* is a plant pigment that is used as a proxy for algal abundance in lakes, as all photosynthetic algae have chl *a*. On average, chlorophyll *a* concentrations remained below the target of 10 µg/L but averages can be deceptive. There were multiple sampling dates when surface chlorophyll *a* concentrations exceeded the 10 µg/L threshold, especially at the two middle sites during the summer. These exceedances never topped 20 µg/L. Algal blooms were common in other West Michigan drowned river mouth lakes in summer and fall 2024, often at much higher levels than we measured. It is also possible that we missed bloom events in White Lake given our monthly sampling. The new observatory being installed in White Lake will provide chlorophyll data at a much greater temporal resolution, providing a more robust dataset.

The phytoplankton community structure revealed a paucity of potentially toxic cyanobacteria (blue-green algae) taxa. Instead, dinoflagellates and diatoms dominated the phytoplankton in White Lake. Although dinoflagellates can form red tides in marine ecosystems (specifically, the taxon *Karenia brevis*), they are not known to be toxic in freshwater systems. Similarly, some marine diatoms can form toxins, but not the ones living in freshwaters. The phytoplankton taxonomic structure was consistent with the low microcystin concentrations measured in White Lake. It is unclear why *Microcystis*, the cyanobacterium that was so persistent in Muskegon Lake blooms during summer/fall of 2024, did not exhibit a similar pattern in White Lake. Experiments would be necessary, such as transplant studies where White Lake plankton were incubated in Muskegon Lake water and vice versa, to tease out possible reasons for this difference.

The other water quality measurements made in this study provide important baseline information for future comparisons and do not raise any special concerns. The nitrate levels at site 4 do suggest there is agricultural runoff reaching the lake, but the concentrations quickly become diluted by site 3. Nonetheless, restoration of the fringing riparian wetlands along upstream reaches of the White River will help reduce inflowing nutrients.

Conclusion

In summary, the water quality remains good in White Lake and relatively unchanged since the 2014 de-listing status. There has been no consistent improvement or degradation for TP and Chl *a* but the high P concentrations and low DO levels in the hypolimnion of the middle sites should be monitored for any further changes. The White Lake Observatory will provide more robust data, although from a limited spatial area; as shown in the current study, data from one site in White Lake cannot be generalized for the entire lake so appropriate caveats are required. Finally, as the riparian wetlands continue to be restored, monitoring for changes in the nitrate and *E. coli* levels is recommended.

Acknowledgements

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