Bear Lake 2024 Watershed Assessment

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Alan D. Steinman*, Michael C. Hassett, Kathryn J. Tyrrell

Annis Water Resources Institute, Grand Valley State University

740 West Shoreline Drive, Muskegon, MI 49441

*Contact Information
Email: steinmaa@gvsu.edu
Voice: 616-331-3749

Introduction

Bear Lake (Muskegon County, MI) is a small, eutrophic lake located within the Muskegon Lake Area of Concern (AOC). Because of elevated total phosphorus (TP) concentrations and excess algal growth, a Total Maximum Daily Load (TMDL) was issued for Bear Lake in 2008. As noted in last year's report (Steinman et al. 2024), the TMDL erroneously estimated the amount of internal phosphorus loading to Bear Lake. Following the correction (Steinman and Ogdahl 2015), a greater load reduction was needed from the external (watershed) load to meet the TMDL. The restoration of the former celery fields north of Bear Lake resulted in a significant decline in TP concentrations in the formerly flooded celery ponds (Hassett and Steinman 2022), and positive impacts on downstream Bear Lake nutrient concentrations are now starting to show, as noted in this report.

Phosphorus load reduction in Bear Lake also has helped remove the eutrophication and undesirable algae beneficial use impairment (BUI) for the Muskegon Lake AOC (which occurred in 2024), whose geographic boundary includes Bear Lake. The removal of this BUI moves us one step closer to having the Muskegon Lake AOC removed from EPA's list of Great Lakes Areas of Concern.

The Bear Lake - Lake Board has contracted with GVSU's Annis Water Resources Institute to monitor water quality conditions in Bear Lake since 2022. Our 2022 report noted that Bear Lake was still experiencing excess nutrients and chlorophyll *a* (an indicator of algal abundance). We recommended monitoring of tributary nutrient concentrations in order to identify the major sources of nutrients entering Bear Lake, as well as more frequent monitoring of chlorophyll *a* and a survey of lake users to determine their priorities; in addition, the report included our support of the use of Phoslock® to help control phosphorus concentrations in the lake. Our 2023 report (Steinman et al. 2024) built upon lake monitoring efforts by additionally monitoring lake inflows from Bear Creek and Fenner's Ditch in addition to outflowing lake water at the Bear Lake channel. Bear Creek was found to have a diluting effect on phosphorus concentrations in Bear Lake; however, it was found that Fenner's Ditch may be a significant source that resupplies phosphorus to the lake, which is amplified during storm events.

This report contains our third year of findings—including a sediment phosphorus study of Fenner's Ditch—and recommendations for future monitoring activities.

Methods

Water Quality Monitoring

Bear Lake water quality monitoring sites were the same as in previous years to facilitate comparisons of 2024 data with prior results. Due to continued budgetary limits and the desire to sample earlier in the spring season, sampling in 2024 occurred in April, June through August, and in October (skipping May and September 2024); this is an identical sampling regime to 2023, although 2022 occurred monthly from May through October. The four monitoring sites included two sites monitored by Restorative Lake Sciences (RLS) in 2017-2021 (Sites 1 and 3) and two sites previously monitored by AWRI in 2011-2012 and 2022-2023 (Sites 2 and 4). Site locations are specified in Table 1 and Figure 1.

Lake samples were collected once monthly from a jon boat throughout the monitoring period, with sampling occurring usually between 9:00-11:30 AM. Water transparency was measured as Secchi disk depth. General water quality parameters were measured using a YSI EXO2 sonde (YSI, Inc., Yellow Springs, OH), which included sensors for water temperature, dissolved oxygen (DO), pH, specific conductivity (SpCond), and turbidity. Water was collected at surface depth via grab sampling and at middle and near-bottom depths via a Van Dorn water sampler. Samples for water chemistry analysis were collected in 500-mL bottles, stored on ice, and returned to the lab for nutrient analysis, usually within 4 hours.

Separately, an additional 1-L sample was collected in amber bottles at surface and near-bottom depths at each site for chlorophyll a (chl *a*) extraction. One 250-mL sample was collected for phytoplankton identification from the middle depth of each site, which was later composited with subsamples from surface and near-bottom chl *a* sample bottles from each site into a single integrated depth phytoplankton sample per site.

Additionally, we subsampled from surface and near-bottom chl *a* bottles for microcystin analysis. Microcystin is the most common toxin produced by cyanobacteria (blue-green algae). We used the ELISA QuantiPlate kit for Microcystin – High Sensitivity, 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 US EPA. For water bodies that serve as a source of drinking water, the WHO advisory is >1 μg/L and EPA is >1.6 μg/L (0.3 μg/L for infants and pre-schoolers); for water bodies used only for recreation, the WHO advisory is >20 μg/L and EPA is >8 μg/L (WHO 2017; US EPA 2019). Since Bear Lake is used only for recreation, we applied the latter two criteria.

We also collected water samples, using near-surface grabs only, to measure *E. coli* concentrations. One sample was collected from each site in addition to a field duplicate sample each month. These 100-mL aliquots 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.).

After returning to the lab, water from each lake site was gently inverted and subsampled for analysis of 1) phosphorus (P) as both soluble reactive phosphorus (SRP) and total phosphorus (TP); and 2) nitrogen (N) as nitrate (NO_3^-), ammonia (NH_3), and total Kjeldahl nitrogen (TKN) species. Duplicate water quality samples were collected once a month for quality control. Water for SRP and NO_3^- analyses was syringe-filtered through acid-washed 0.45-µm membrane filters into scintillation vials; SRP was refrigerated at 4°C, and NO_3^- was frozen until analysis. TKN was acidified with sulfuric acid; TP and TKN were kept at 4°C until analysis. SRP, TP, NO_3^-, NH_3, and TKN were analyzed on a SEAL AQ2 discrete automated analyzer (US EPA 1993). Any values below detection were reported as ½ of their respective detection limits.

Chl *a* was subsampled for phytoplankton analysis by gently inverting and removing 250 mL from surface and near-bottom samples and combining them with the 250 mL middle depth sample. These integrated depth phytoplankton samples were preserved with 7.5 mL of Lugol's iodine to create a 1% final solution. Phytoplankton taxa were later identified to genus or species, and abundance was estimated via light microscopy as the respective sum of each species' biovolume present at each site on each sampling date.

For a historic comparison of water quality conditions between the current sampling year and recent years of monitoring by Restorative Lake Sciences, AWRI's 2022-2024 data were reformatted to match RLS's data summary methods based on their 2021 Bear Lake water quality report, using only lake sites 1 and 3. AWRI water quality depth profiles (measured at every meter) and nutrient data (near-surface and near-bottom) were averaged into single point values per site, and April 2024 and July 2024 data were compared to historic Spring and Summer data, respectively.

Water quality dashboards for TP, chl *a*, and Secchi depth were created using historic (Steinman and Ogdahl 2013) and current AWRI Bear Lake monitoring data in conjunction with historic RLS data (RLS 2022) and are attached in Appendix A. AWRI 2024 data are presented seasonally by separately averaging surface data into Spring (April and June), Summer (July and August), and Fall (October) seasons. Water quality goals for chl *a* and Secchi depth were established based on thresholds used in AWRI's annual Muskegon Lake water quality dashboard (www.gvsu.edu/wri/dashboard); the TP category's "Meeting Goal" threshold was created from the Bear Lake's TMDL goal of 30 µg/L and the "Desirable" threshold of 24 µg/L from the Muskegon Lake water quality dashboard.

Table 1. Bear Lake and Fenner's Ditch site coordinates and mean max depth across 2024 sampling events.

Site	Latitude (°N)	Longitude (°W)	Max Depth (m)	Avg Secchi (m)
Lake 1	43.24885	86.29026	8.30	0.89
Lake 2	43.25353	86.28691	3.90	0.93
Lake 3	43.25494	86.28431	3.90	0.89
Lake 4	43.26047	86.27346	3.00	0.83
Fenner 1	43.26348	86.27664	1.40	0.87
Fenner 2	43.26721	86.27892	1.20	0.67
Fenner 3	43.26751	86.27906	0.65	0.65

Figure 1. Map of Bear Lake water quality monitoring sites and Fenner's Ditch sediment coring sites.

Sediment Core Internal P Loading Study

Site water samples (surface and near-bottom) and sediment cores were collected on July 23 via jon boat from Fenner's Ditch sites 1 and 2 and on foot from site 3, upstream of the Fenner's Ditch dam (Table 1, Figure 1). Five cores were collected per site, with one core being used for field site characterization and four cores being assigned to a phosphorus internal loading study with n=2 cores per site each being assigned to oxic (bubbled with atmospheric gas) or anoxic treatments (bubbled with N_2 gas buffered with 330 ppm CO_2). Cores were collected using a modified piston coring apparatus described by Smit and Steinman (2015), resulting in intact sediment cores of 20 cm depth and 25 cm of overlying site water. Additional water from each site was collected in a 10-L carboy, sequentially filtered using 1 µm and 0.2 µm filter cartridges, and refrigerated at 4 °C throughout the study. The four cores per site were installed in an environmental growth chamber with air temperature maintained to incubate cores at the ambient water temperature of the hypolimnion of Fenner's Ditch at the time of collection.

Internal load estimates were made using the methods outlined in Moore et al. (1998), with minor modifications (Steinman et al. 2004). Briefly, a 40 mL water sample was removed by syringe through the sampling port of each core tube at days 1, 5, 10, and 20. Immediately after removal, a 20 mL subsample was refrigerated for analysis of TP, and a 20 mL subsample was filtered through a 0.45 μm membrane filter and stored at 4 °C for analysis of SRP. TP and SRP were analyzed on a Seal AQ2 Discrete Analyzer (US EPA 1993). SRP values below detection (5 μg/L) were calculated as one-half of the detection limit. The 40 mL subsample was then replaced with filtered water collected from the corresponding coring site to maintain the original volume and concentration gradient in the core tubes. After concluding the incubation, the top 0-10 cm of sediment was removed from each core, stored in a zip-seal bag, homogenized, and refrigerated at 4 °C until analysis.

Total flux was defined as the total change in mass of P in the water column from day 1 to day 20 divided by the duration of the incubation and normalized by the area of the sediment-water interface, resulting in flux in units of mg P/m^2/d using the following equation (Steinman et al. 2004):

$$P_{rr} = ((C_t - C_0)/d) * (V/A)$$

where, P_{rr} is the net P release rate (positive values) or retention rate (negative values) per unit surface area of sediments, C_t is the TP or SRP concentration in the water column at time t, C_0 is the TP or SRP concentration in the water column at time 0, d is the number of days of incubation, V is the volume of water in the water column, and A is the planar surface area of the sediment cores. Maximum flux was considered to be the greatest rate across three consecutive sampling events; the associated time period is reported with maximum flux (Tables 7 and 8).

Total and maximum flux for both TP and SRP were analyzed for statistically significant effects based on redox state, site, and possible interaction between the two. Comparisons were made using the Scheirer-Ray-Hare (SRH) test, a nonparametric equivalent to 2-way ANOVA (Scheirer et al. 1976) using the rcompanion package in R (Mangiafico 2024, R Core Team 2023). Effects were considered to be significant for p-values < 0.05.

Sediment subsamples were dried and later ashed at 550 °C for analysis of organic matter and sediment TP (as described above). Additional subsamples were dried to a constant mass for metals (Al, Ca, Fe, Mg) and analyzed using EPA method 6010b (USEPA 1996). Other subsamples (2 g) of wet sediment were sequentially fractionated (Psenner and Pucsko 1988, modified by Hupfer et al. 2009 and Dieter et al. 2015) to identify the major P fractions in the sediment. These operational definitions include the 1.0M NH_4Cl extraction, which produces the loosely sorbed P; the 0.11M buffered dithionite (BD) extraction that results in reductant-soluble P (iron oxides and Mn-bound); the 1.0 M NaOH extraction that produces Fe- and Al-bound P, which are mineral associations that can become soluble under hypoxic conditions; and the 0.5 M HCl extraction resulting in Ca- and Mg-bound P, which represents a stable mineral association.

Results

2024 Lake Water Quality

Water temperature and DO observations varied as expected across sampling time, depth, and seasons (Table 2, Figure 2). Bear Lake was isothermal (uniform temperatures from top to bottom, resulting in complex mixing and uniform DO) in April and October (Figure 2). Hypoxic conditions with DO <2 mg/L that were observed at the deepest site (#1) in July and August dissipated by our October sampling date (Figure 2). Site pH values ranged ~7.5-8.5 across all sampling events and tended to be higher (more basic) at surface depths and lower (more neutral) at near-bottom depths (Table 2, Figure 3). Within each sampling event, specific conductivity generally did not vary among sites between surface and ~4 m sampling depths; however, small (~30 µS/cm) increases were seen at near-bottom depths in July and August (Table 2, Figure 4). Turbidity was low throughout the sampling period at all depths and sites, ranging 2-11 FNU (Table 2, Figure 5).

Secchi disk depth measurements of water clarity indicated relatively poor conditions, averaging <1 m across all sites and sampling dates (Table 2). Water clarity was best (i.e., deepest Secchi disk depth) in July, with a maximum depth of 1.30 m at Site 2, while clarity was worst (most shallow disk depth) in August at Site 1 with a depth of only 0.47 m (Figure 6). The high August turbidity corresponded to peaks in chlorophyll (Figure 12), suggesting that algal abundance was at least partly responsible for the low water clarity.

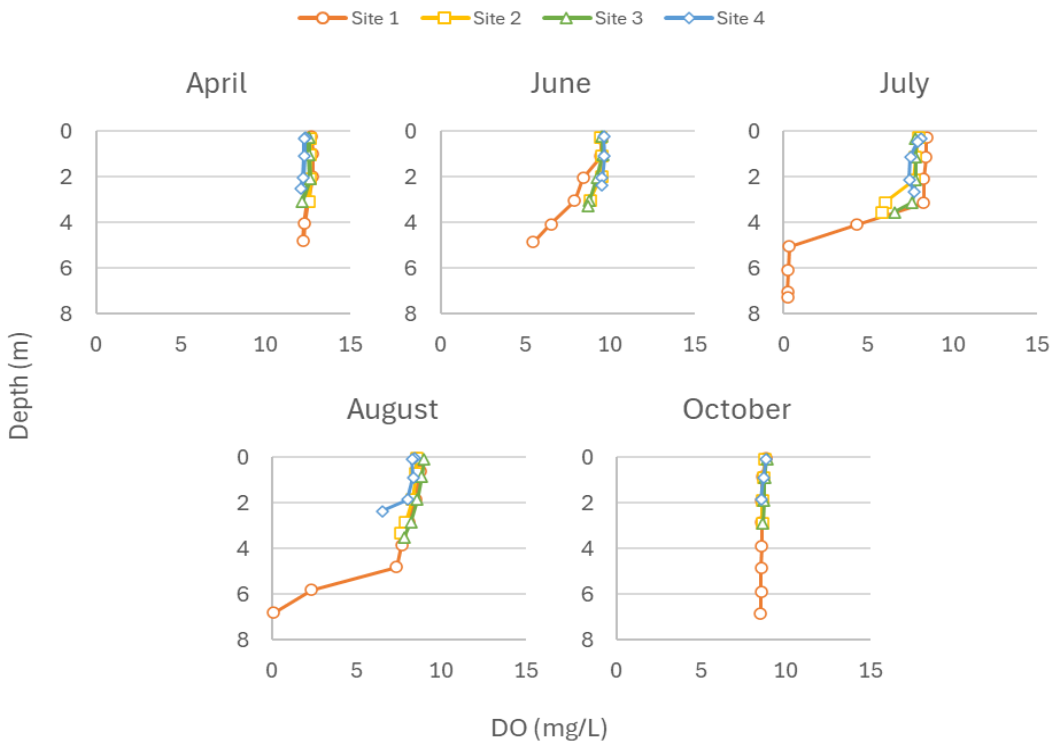

Figure 2. Bear Lake DO concentrations sampled April – October 2024.

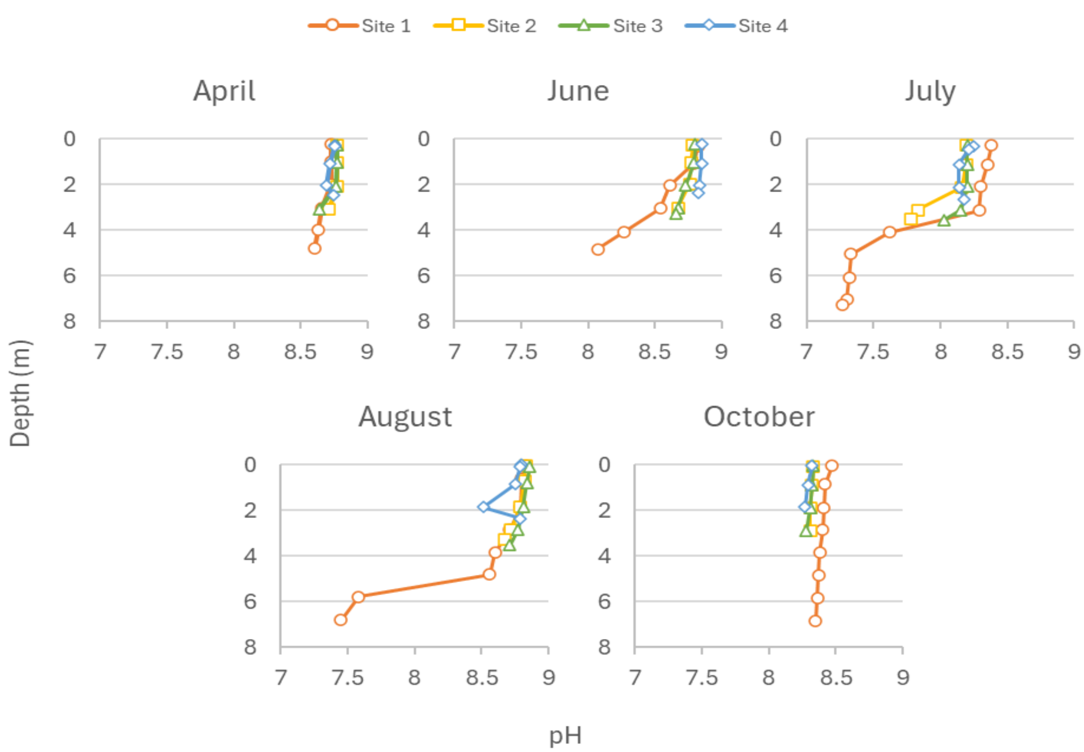

Figure 3. Bear Lake pH sampled April – October 2024.

Table 2. Means (and observed ranges) of Bear Lake general water quality parameters across all sampling months (n=5). Temp = water temperature, DO = dissolved oxygen; SpCond = specific conductivity.

Site	Depth	Temp (°C)	DO (mg/L)	DO (%)	pH	SpCond (µS/cm)	Turbidity (FNU)	Secchi Depth (m)
1	Surface	19.8 (10.2-25.2)	9.6 (8.4-12.7)	104 (9-113)	8.6 (8.4-8.8)	394 (381-415)	4.9 (2.9-10.8)	0.9 (0.5-1.3)
	Middle	19.5 (10.1-24.6)	8.4 (4.3-12.8)	89 (9-114)	8.4 (7.6-8.7)	394 (379-413)	5.2 (3.2-9.5)	
	Bottom	18.5 (9.8-22.5)	5.3 (0.1-12.2)	52 (1-108)	7.9 (7.3-8.6)	405 (379-433)	4.4 (2.9-5.5)	
2	Surface	19.9 (10.5-25.2)	9.5 (8.0-12.6)	102 (9-113)	8.6 (8.2-8.8)	394 (379-411)	5.0 (2.8-8.8)	0.9 (0.6-1.3)
	Middle	19.9 (10.5-25.2)	9.4 (7.7-12.7)	101 (9-113)	8.6 (8.2-8.8)	394 (379-412)	4.7 (3.2-8.6)	
	Bottom	19.6 (10.2-25.0)	8.7 (5.8-12.6)	93 (9-112)	8.4 (7.8-8.7)	396 (379-417)	5.1 (3.5-8.8)	
3	Surface	20.0 (10.6-25.4)	9.6 (7.8-12.6)	104 (9-113)	8.6 (8.2-8.9)	394 (380-412)	4.7 (3-8.6)	0.9 (0.6-1.2)
	Middle	19.8 (10.6-25.3)	9.4 (7.8-12.6)	101 (9-113)	8.6 (8.2-8.8)	394 (380-413)	4.6 (3.3-8.3)	
	Bottom	19.6 (10.2-25.1)	8.8 (6.6-12.2)	94 (9-109)	8.5 (8.0-8.7)	395 (382-414)	5.5 (3.9-8.8)	
4	Surface	20.1 (11.4-25.4)	9.5 (8.2-12.6)	103 (9-113)	8.6 (8.3-8.9)	393 (382-412)	4.9 (3.3-8.6)	0.8 (0.6-1.0)
	Middle	20.0 (11.2-25.3)	9.2 (7.5-12.6)	100 (9-113)	8.6 (8.1-8.8)	394 (382-413)	5.1 (3.7-8.3)	
	Bottom	19.9 (11.4-25.1)	9.3 (7.7-12.2)	100 (9-109)	8.6 (8.2-8.7)	393 (382-414)	5.1 (3.3-8.8)	
All Sites	Surface	19.9 (10.7-25.4)	9.5 (8.1-12.3)	103 (9-113)	8.6 (8.3-8.9)	394 (380-412)	4.9 (3-8.9)	0.9 (0.5-1.3)
	Middle	19.8 (10.6-25.3)	9.1 (6.8-12.2)	98 (9-111)	8.5 (8.0-8.9)	394 (380-412)	4.9 (3.3-8.2)	
	Bottom	19.4 (10.4-25.0)	8.0 (5.0-12.3)	85 (7-113)	8.4 (7.8-8.8)	397 (380-412)	5.0 (3.4-8.3)	

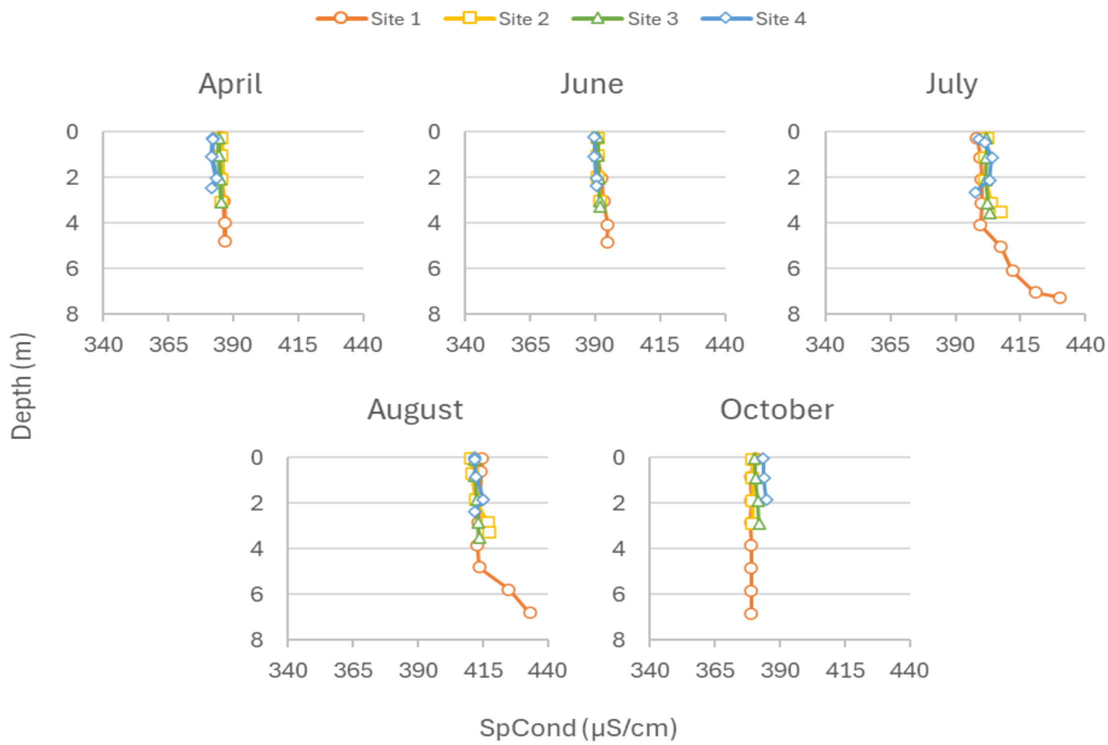

Figure 4. Bear Lake specific conductivity sampled April – October 2024.

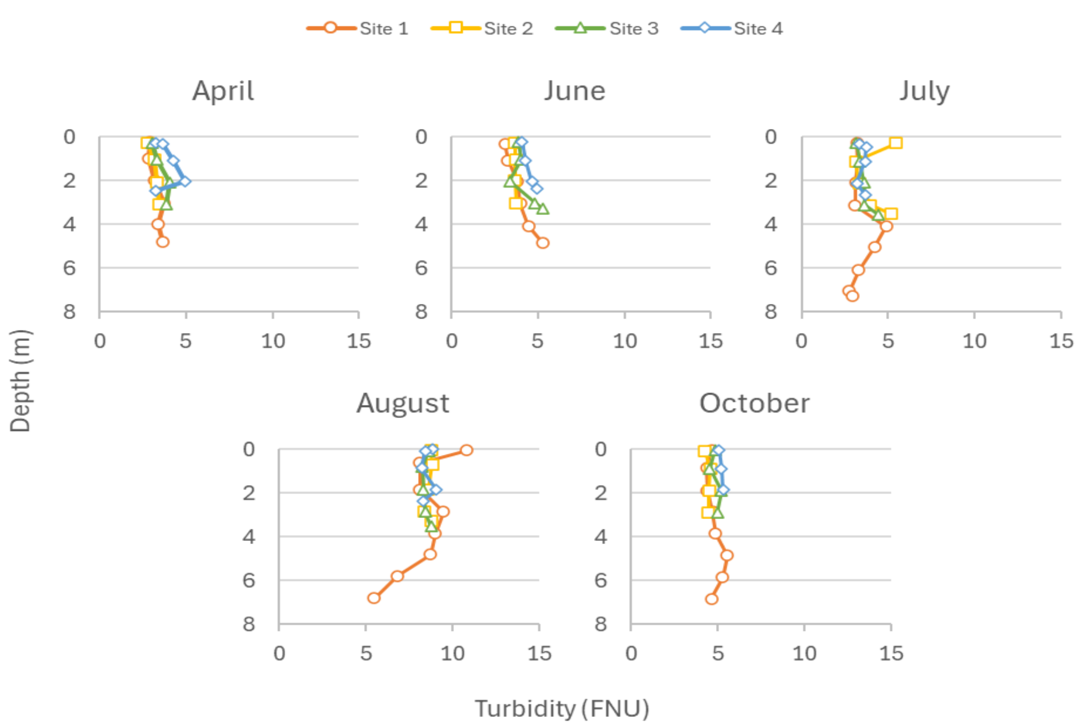

Figure 5. Bear Lake turbidity sampled April – October 2024.

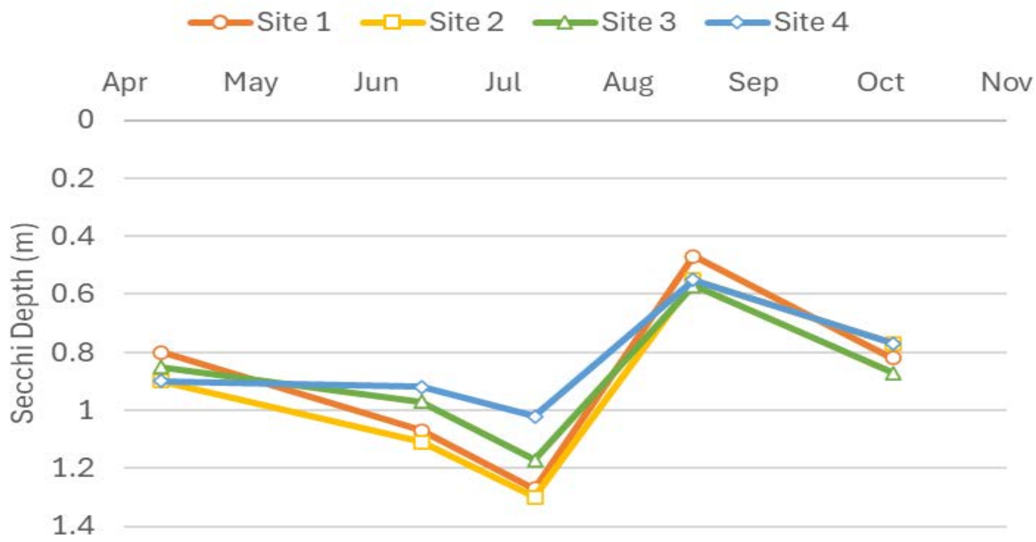

Figure 6. Bear Lake Secchi depth sampled April – October 2024. The greater the Secchi depth, the greater the water clarity.

Lake Nutrients

TP generally ranged from 20 to 60 µg/L across all sites and depths, except for a substantial July spike of 177 µg/L at Site 1's near-bottom sampling depth (Table 3, Figure 7). Likewise, most SRP concentrations ranged from below detection (represented as 2.5 µg/L) to 6 µg/L and reached 111 µg/L during the July spike (Table 3, Figure 8A-B). Across all sampling sites and months, the 2024 grand mean surface TP was 35.1 µg/L (±2.5 SD), somewhat exceeding Bear Lake's TP TMDL target concentration of 30 µg/L but lower than past TP concentrations of 40 to 80 µg/L (Cadmus Group 2007; MDEQ 2008; but see Table 6 below).

Nitrate concentrations decreased throughout spring sampling before rebounding in August, with a surface spike of 0.128 mg/L at Site 1 (Table 3, Figure 9). Ammonia showed strong seasonal trends with all sites and depths ranging <0.2 mg/L (Table 3, Figure 10). TKN concentrations likewise showed similar site groupings over time, gradually increasing throughout the sampling year and ranging from 0.8 to 1.2 mg/L by October (Table 3, Figure 11). Site 1 sediments, the deepest observed site of the lake, and most likely to go hypoxic or anoxic (Steinman and Ogdahl 2015), likely served as the main source of NH_3 and TKN during summer months, with both nutrients spiking in July (respectively 0.8 and 1.7 mg/L), decreasing in August, and reaching concentrations similar to other sites at near-bottom depths in October (Table 3, Figure 11).

Table 3. Means (range) of Bear Lake total phosphorus (TP), soluble reactive phosphorus (SRP), nitrate (NO_3^-), ammonia (NH_3), and total Kjeldahl nitrogen (TKN). BD = below detection.

Site	Depth	TP µg/L	SRP µg/L	NO_3^- mg/L	NH_3 mg/L	TKN mg/L
1	Surface	31.7 (9.3)	3.3 (1.7)	0.05 (0.05)	0.09 (0.09)	0.86 (0.27)
	Bottom	67.0 (62.1)	24.3 (48.7)	0.04 (0.02)	0.29 (0.33)	1.08 (0.48)
2	Surface	35.0 (7.3)	3.2 (1.6)	0.04 (0.02)	0.09 (0.09)	0.94 (0.30)
	Bottom	41.1 (11.4)	3.3 (1.7)	0.04 (0.02)	0.10 (0.07)	0.86 (0.28)
3	Surface	36.0 (11.5)	3.2 (1.5)	0.04 (0.02)	0.08 (0.08)	0.84 (0.22)
	Bottom	46.0 (7.1)	3.1 (1.3)	0.03 (0.02)	0.10 (0.08)	0.87 (0.23)
4	Surface	37.7 (8.0)	3.0 (1.2)	0.04 (0.02)	0.08 (0.08)	0.90 (0.21)
	Bottom	39.1 (10.0)	3.5 (1.4)	0.04 (0.02)	0.10 (0.08)	0.92 (0.18)
Grand Mean	Surface	35.1 (2.5)	3.2 (0.1)	0.04 (0.01)	0.08 (0.01)	0.88 (0.04)
	Bottom	48.3 (12.8)	8.5 (10.5)	0.04 (0)	0.15 (0.09)	0.93 (0.10)

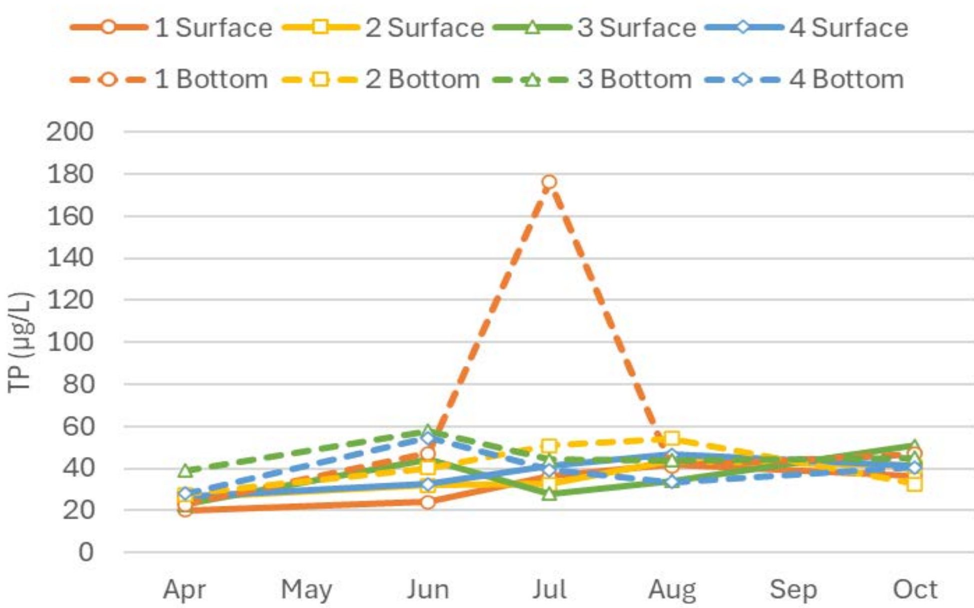

Figure 7. Bear Lake total phosphorus (TP) concentrations sampled April – October 2024 at near-surface (solid lines) and near-bottom depths (dashed lines).

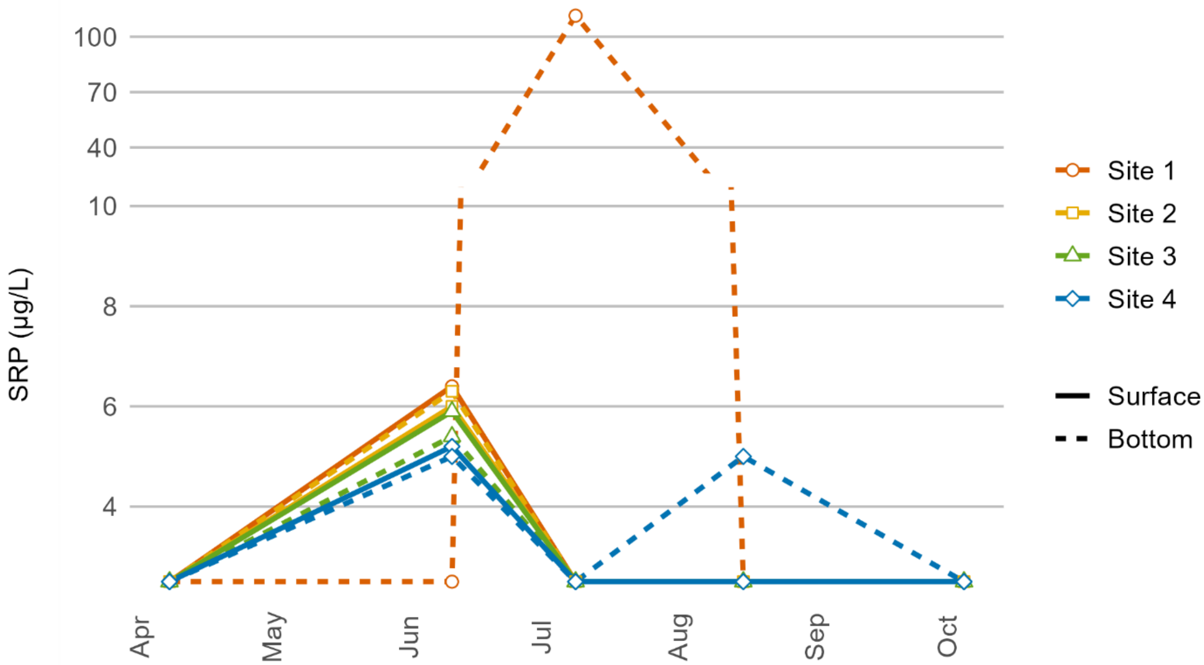

Figure 8. Bear Lake soluble reactive phosphorus (SRP) concentrations sampled April – October 2024 at near-surface (solid lines) and near-bottom depths (dashed lines). Note that samples below analytical detection are reported graphically as 2.5 µg/L. Note the break in the y-axis between 10 and 40 µg/L.

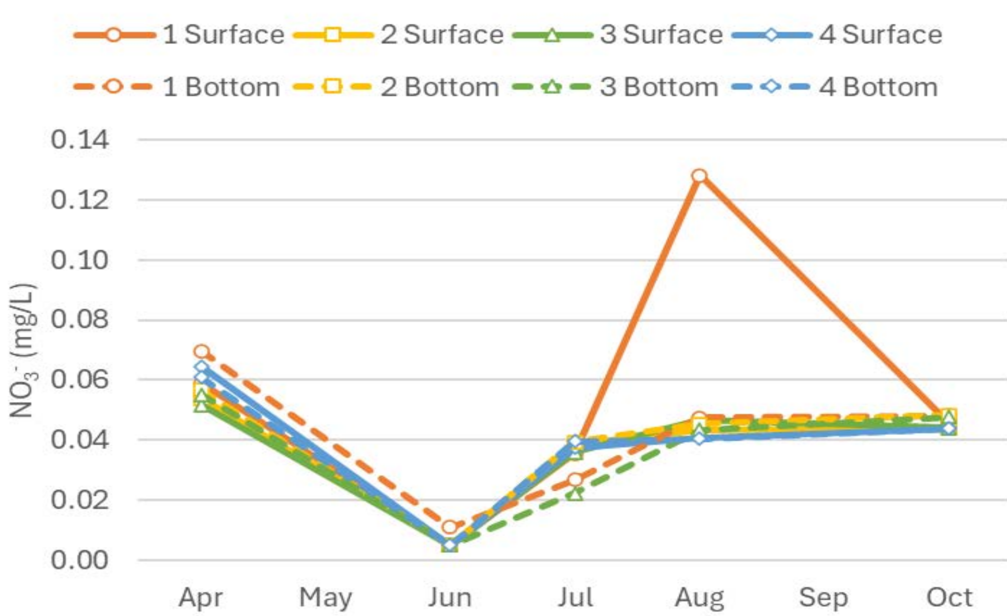

Figure 9. Bear Lake nitrate (NO_3^-) concentrations sampled April – October 2024 at near-surface (solid lines) and near-bottom depths (dashed lines).

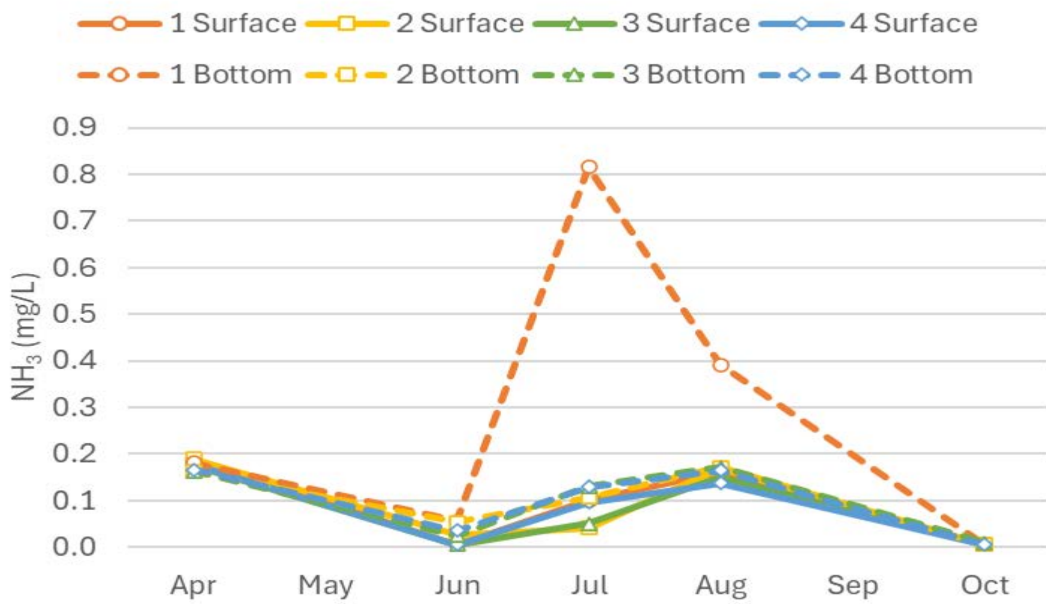

Figure 10. Bear Lake ammonia (NH_3) concentrations sampled April – October 2024 at near-surface (solid lines) and near-bottom depths (dashed lines).

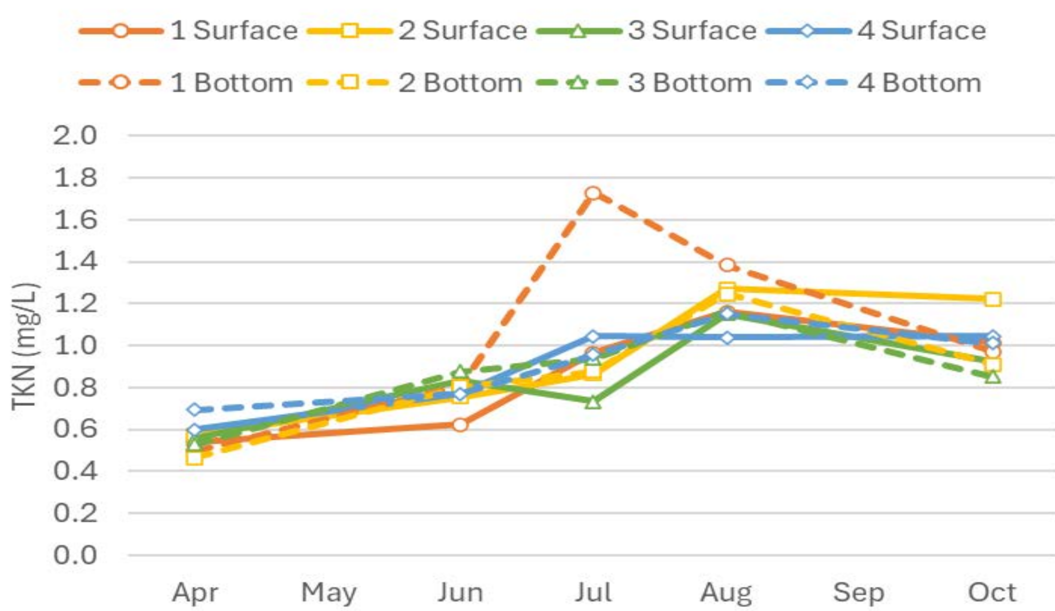

Figure 11. Bear Lake total Kjeldahl nitrogen (TKN) concentrations sampled April – October 2024 at near-surface (solid lines) and near-bottom depths (dashed lines).

Bear Lake chl *a* concentrations ranged from 20 to 30 µg/L through spring before spiking up to 62 µg/L in August and then declining through fall (Table 4, Figure 12). Observed concentrations on all sites, depths, and dates far exceeded Muskegon Lake's chl *a* restoration goal of 10 µg/L.

Although chlorophyll concentrations were higher than desired, the microcystin concentrations across all sites, sampling depths, and dates were <2.5 µg/L, below the recreational guidelines of both WHO and EPA (Table 4, Figure 13). Similar to occurrences in 2023, microcystin concentrations in 2024 began to increase in August, but unlike 2023, they continued to increase into October in surface samples (Figure 13). *E. coli* concentrations also were below detection or ≤6 cfu/100 mL at all sites and sampling events, far below the limit of 300 cfu/100 mL for Michigan recreational waters (Table 4).

Table 4. Means (SD) of Bear Lake biological parameters of water quality. Chl = chlorophyll. NA = not applicable, as *E. coli* samples were not collected at bottom depth.

Site	Depth	Chl *a* µg/L	Microcystin µg/L	*E. coli* (cfu/100 mL)
1	Surface	31.6 (14.0)	0.43 (0.52)	2.7 (3.0)
	Bottom	24.6 (5.0)	0.17 (0.22)	NA
2	Surface	31.1 (17.4)	0.42 (0.64)	2.0 (2.0)
	Bottom	29.8 (14.3)	0.11 (0.14)	NA
3	Surface	28.4 (12.3)	0.72 (1.03)	1.7 (1.4)
	Bottom	30.0 (18.6)	0.12 (0.21)	NA
4	Surface	30.6 (17.3)	0.21 (0.29)	2.5 (2.1)
	Bottom	28.8 (18.7)	0.25 (0.35)	NA
Grand Mean	Surface	30.4 (1.4)	0.44 (0.21)	2.2 (0.5)
	Bottom	28.3 (2.5)	0.16 (0.06)	NA

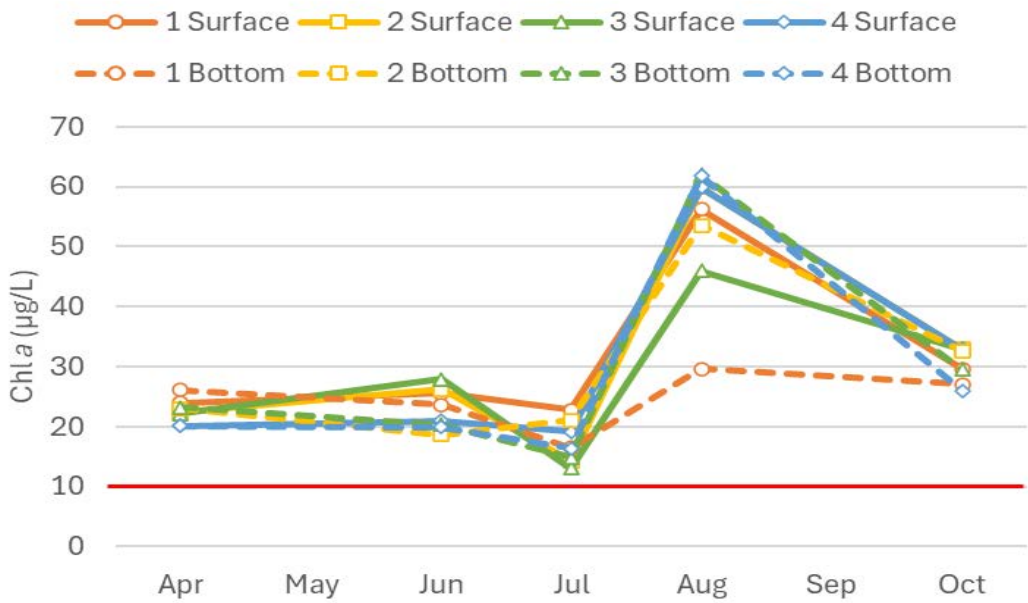

Figure 12. Bear Lake chlorophyll *a* concentrations sampled April – October 2024 at near-surface (solid lines) and near-bottom depths (dashed lines). Red line refers to restoration target of 10 µg/L for Muskegon Lake.

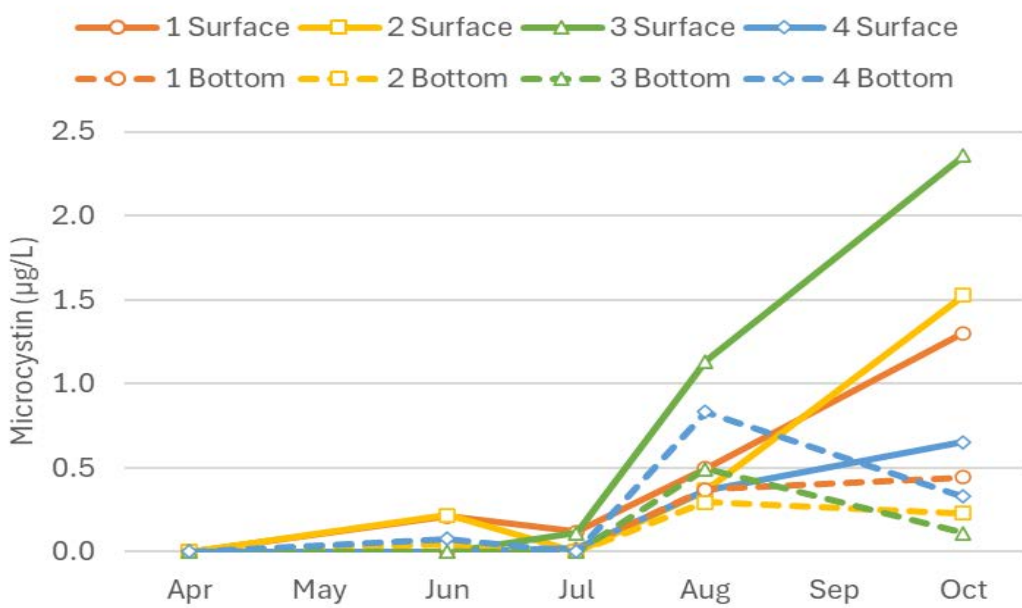

Figure 13. Bear Lake microcystin concentrations sampled April – October 2024 at near-surface (solid lines) and near-bottom depths (dashed lines).

Historical Bear Lake Water Quality

Baseline monitoring is critical to determine trends, but sampling on just one date per month can lead to unrepresentative results, so it is important to look at overall trends, not individual years. In addition, given the different analytical methodologies used by RLS and AWRI, comparisons between those two groupings of years should be done with caution. Ideally, an intercalibration study having both labs test the same water sample for multiple analytes would be conducted to assess for inter-lab differences but that was not done in this case. As a consequence, examining changes over time **within** each grouping (i.e., within 2017-2021 and within 2022-2024) provides the more accurate assessment. Comparisons over the entire period of record (i.e. 2017-2024) are done with caveats given the above limitation. Water quality parameters from spring and summer are summarized in Tables 5 and 6 and Figures 14-20.

Spring Data

Spring data for the 2017-2021 RLS period show few distinct trends; only TP and TKN show relatively clear declines over time (Table 5; Figures 16, 18). Spring data for the 2022-2024 AWRI period reveal a continued decline in TP (Figure 16). The substantial jump in chlorophyll *a* (Figure 19) is consistent with the decline in SRP (Figure 17), suggesting this bioavailable form of phosphorus was being taken up by the algae resulting in more phytoplankton biomass. However, these results are inconsistent with the decline in TP, as the increased algal biomass should also be reflected in increased total phosphorus (Quinlan et al. 2021). This inconsistency points to the problem in evaluating results from two different sampling periods, which used different analytical methods.

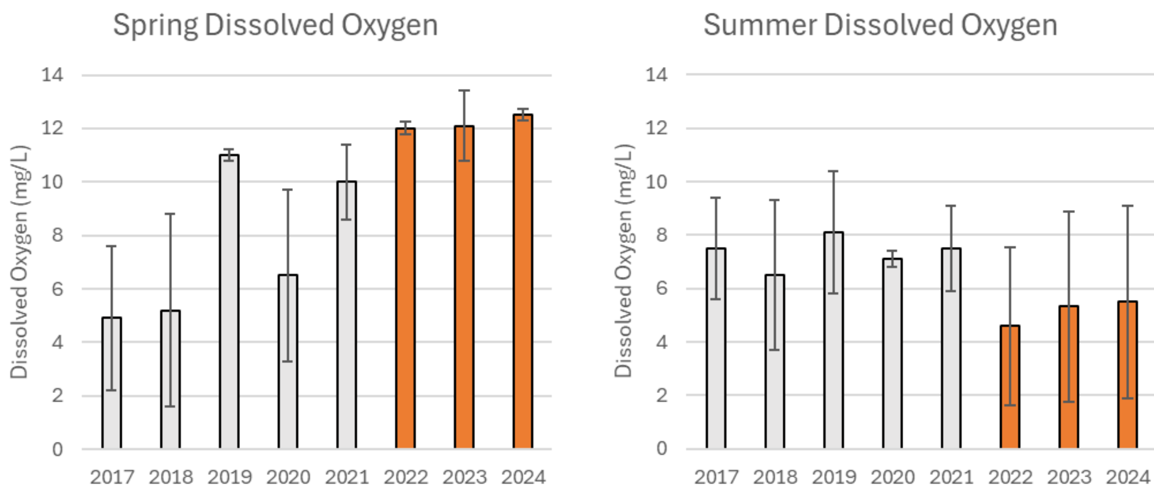

Figure 14. Lake grand mean (±SD) DO across water column depths at the two deep sites during May (left panel) and July (right panel) of each sampling year. Data: 2017-2021 (RLS); 2022-2024 (AWRI).

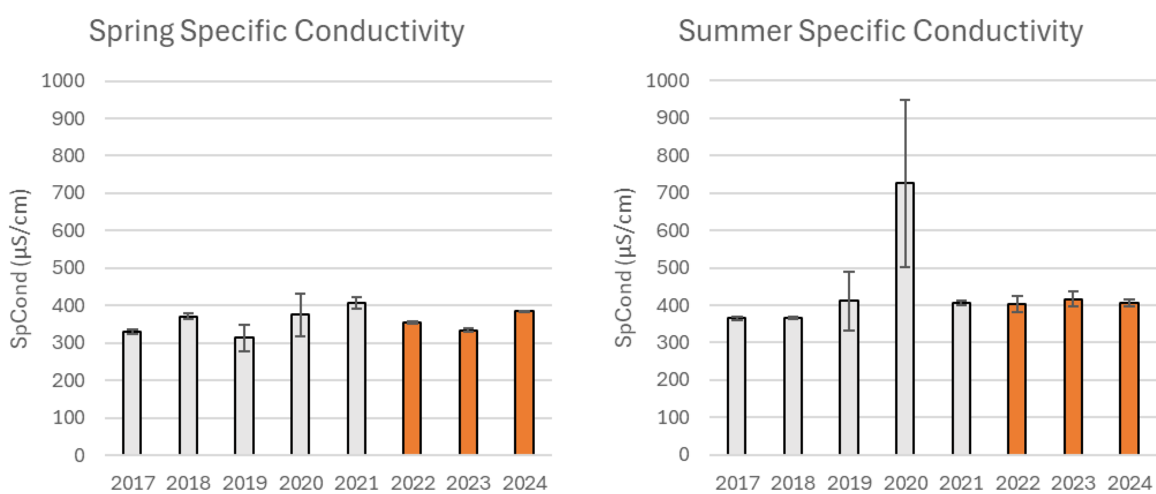

Figure 15. Lake grand mean (±SD) specific conductivity across water column depths at the two deep sites during May (left panel) and July (right panel) of each sampling year. Data: 2017-2021 (RLS); 2022-2024 (AWRI, in orange).

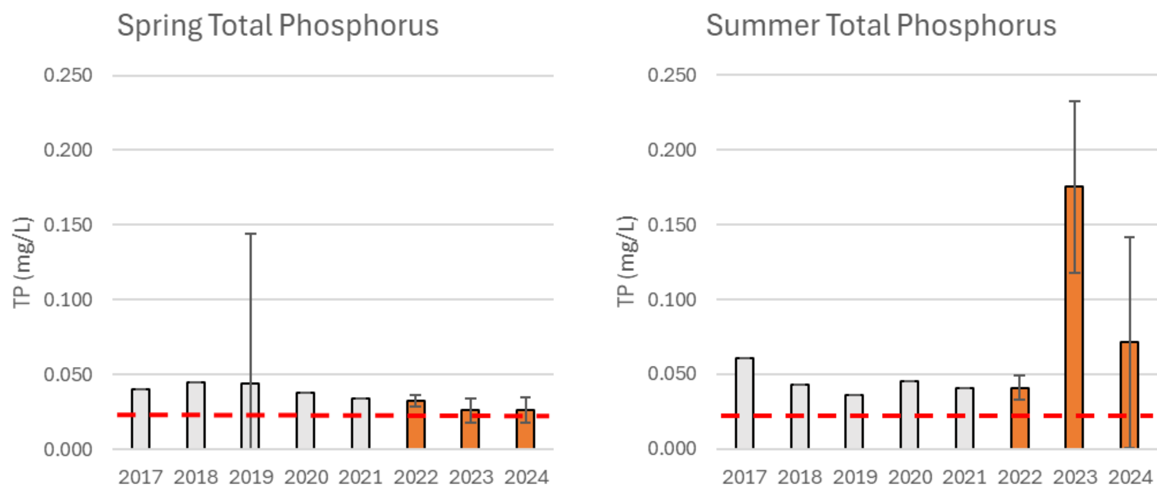

Figure 16. Lake grand mean (±SD) total phosphorus (TP) concentrations across water column depths at the two deep sites during May (left panel) and July (right panel) of each sampling year. Red dashed lines indicate Bear Lake's TMDL for TP: 0.030 mg/L. Data: 2017-2021 (RLS); 2022-2024 (AWRI, in orange).

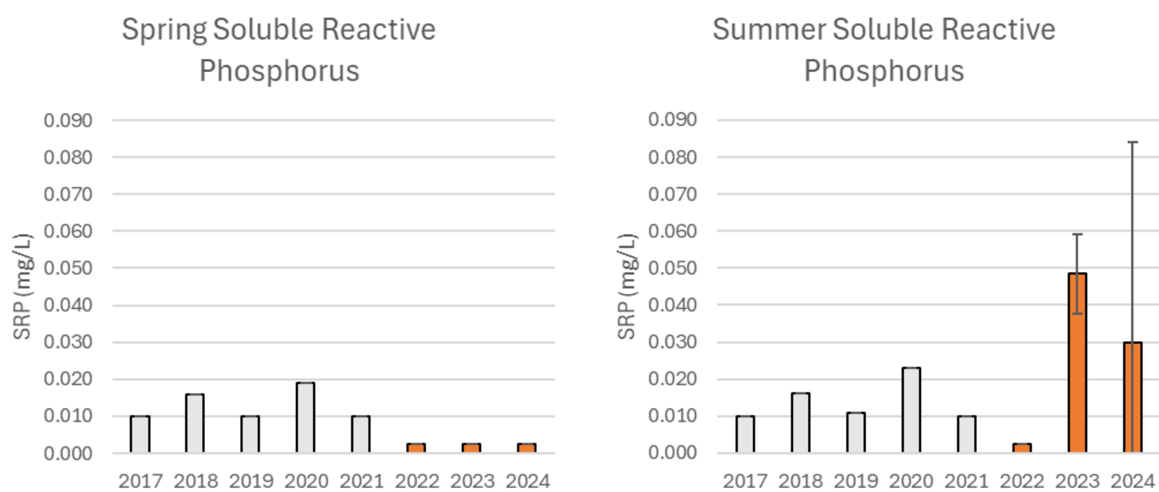

Figure 17. Lake grand mean (±SD) soluble reactive phosphorus (SRP) concentrations across water column depths at the two deep sites during May (left panel) and July (right panel) of each sampling year. Data: 2017-2021 (RLS); 2022-2024 (AWRI, in orange).

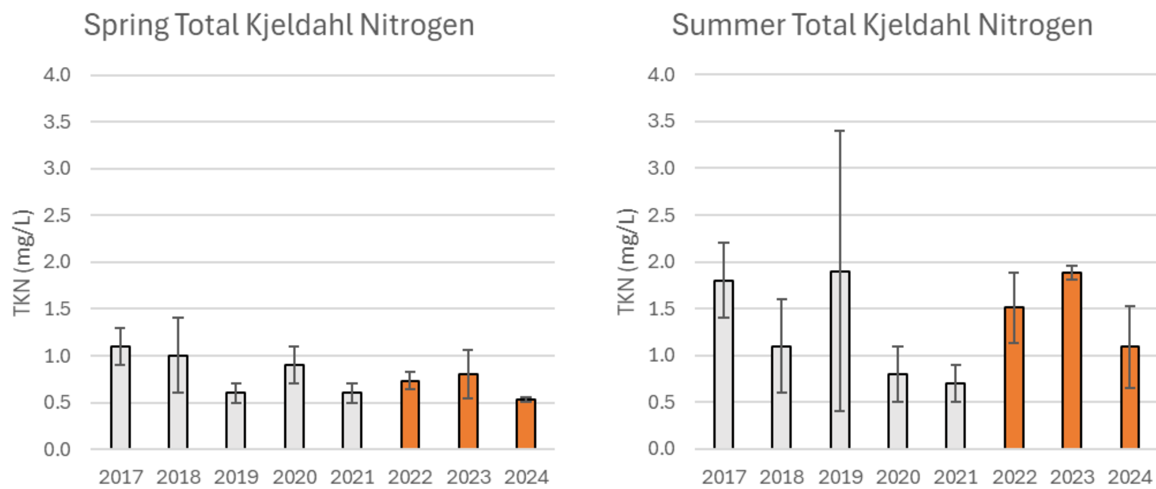

Figure 18. Lake grand mean (±SD) total Kjeldahl nitrogen concentrations across water column depths at the two deep sites during May (left panel) and July (right panel) of each sampling year. Data: 2017-2021 (RLS); 2022-2024 (AWRI, in orange).

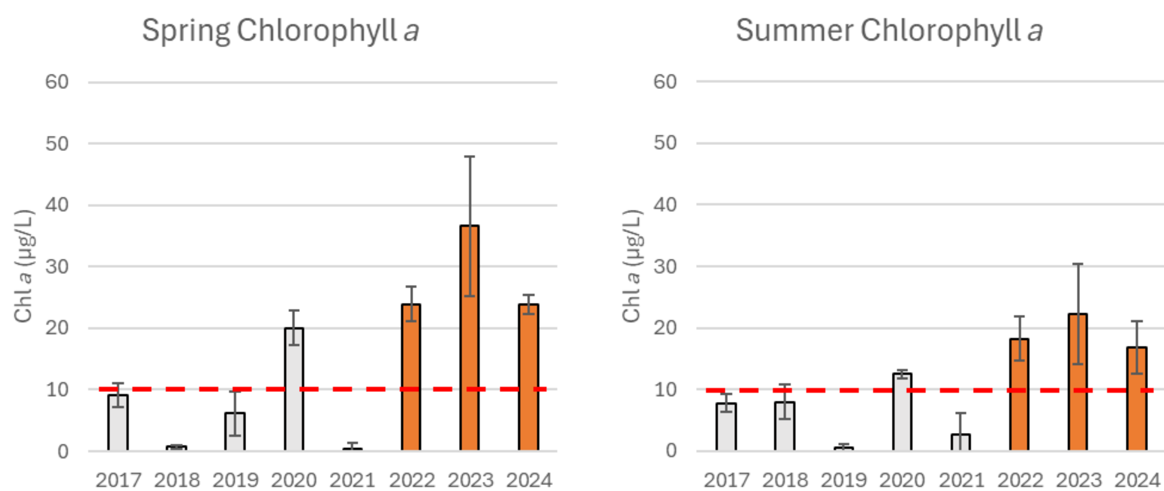

Figure 19. Lake grand mean (±SD) chlorophyll *a* concentrations across water column depths at the two deep sites during May (left panel) and July (right panel) of each sampling year. Red dashed lines indicate Muskegon Lake's restoration goal for chl *a*: 10 µg/L. Data: 2017-2021 (RLS); 2022-2024 (AWRI, in orange).

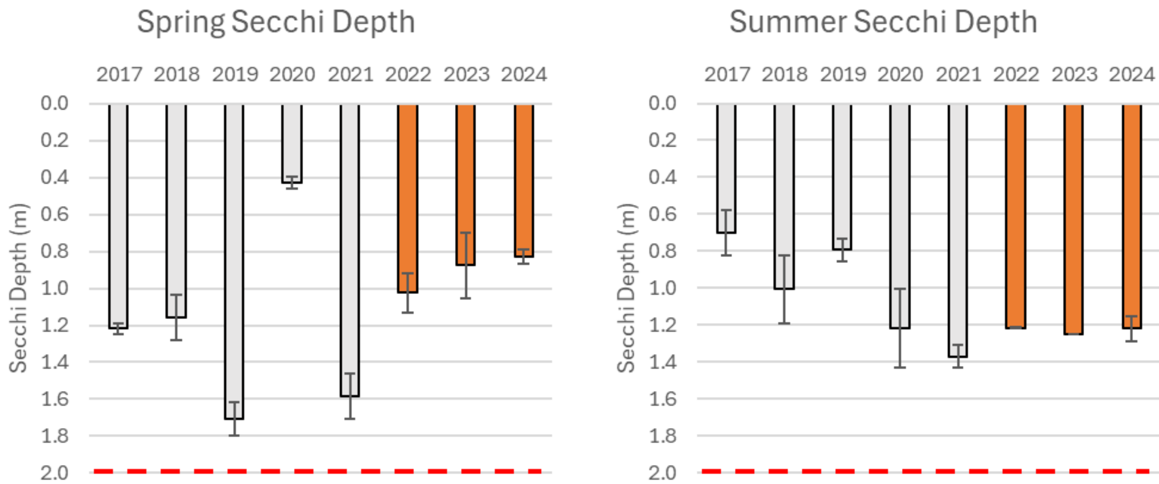

Figure 20. Lake grand mean (±SD) Secchi depth across water column depths at the two deep sites during May (left panel) and July (right panel) of each sampling year. Note that the y-axes are inverted so that data indicate the depth from the lake's surface. Red dashed lines indicate Muskegon Lake's restoration goal for Secchi depth: 2 m (~6.5 ft). Data: 2017-2021 (RLS); 2022-2024 (AWRI, in orange).

Summer Data

Summer data for the 2017-2021 RLS data reveal that 2020 was an anomalous year, with very high specific conductivity, and high SRP and Chl *a* concentrations (Table 6) compared to the other RLS years. Summer data for the 2022-2024 AWRI period reveal substantially higher SRP, TP, and chl *a* concentrations in the two most recent years compared to either 2022 or earlier years (Figures 16, 17, 19). Of interest is that in both 2023 and 2024, mean TP concentrations were heavily biased by summer samples at the deeper sites, which were ~740 µg/L in August 2023 and ~180 µg/L in July 2024. It is likely these high concentrations are a result of internal phosphorus loading from nutrient-rich sediments. Unlike spring, the summer declines in SRP due to phytoplankton are accompanied by increases in both TP and chl *a*. However, the very substantial increase (doubling and tripling) in chlorophyll *a* concentration is troubling; as noted earlier, it is possible this increase may reflect different laboratory methodologies and/or choice of sampling dates that had episodic algal blooms. Continuous monitoring data via *in situ* sensors could help resolve this issue (see recommendations).

Table 5. Long-term trends of Bear Lake deep basin mean (±SD) spring water quality parameters. Shaded data analyzed by RLS; unshaded data analyzed by AWRI.

Year	DO (mg/L)	pH	SpCond (µS/cm)	TP (mg/L)	SRP (mg/L)	TKN (mg/L)	Chl *a* (µg/L)	Secchi Depth (m)
2017	4.9 (2.7)	8.2 (0.1)	329 (6)	0.04 (0)	0.010 (0)	1.1 (0.2)	9.1 (2)	1.2 (0)
2018	5.2 (3.6)	7.9 (0.4)	370 (8)	0.045 (0)	0.016 (0)	1.0 (0.4)	0.7 (0.3)	1.2 (0.1)
2019	11 (0.2)	8.2 (0.1)	314 (35)	0.044 (0.1)	0.010 (0)	0.6 (0.1)	6.2 (3.6)	1.7 (0.1)
2020	6.5 (3.2)	8.4 (0)	376 (57)	0.038 (0)	0.019 (0)	0.9 (0.2)	20.0 (2.8)	0.4 (0)
2021	10 (1.4)	8.4 (0.2)	407 (15)	0.034 (0)	0.010 (0)	0.6 (0.1)	0.5 (0.8)	1.6 (0.1)
2022	12.0 (0.2)	8.7 (0.1)	354 (3)	0.032 (0)	0.003 (0)	0.7 (0.1)	24.0 (2.8)	1.0 (0.1)
2023	12.1 (1.3)	8.5 (0.4)	334 (4)	0.026 (0)	0.003 (0)	0.8 (0.3)	36.6 (11.4)	0.9 (0.2)
2024	12.5 (0.2)	8.7 (0.1)	385 (1)	0.026 (0)	0.003 (0)	0.5 (0)	23.8 (1.6)	0.8 (0)

Table 6. Long-term trends of Bear Lake deep basin mean (±SD) summer water quality parameters. Shaded data analyzed by RLS; unshaded data analyzed by AWRI.

Year	DO (mg/L)	pH	SpCond (µS/cm)	TP (mg/L)	SRP (mg/L)	TKN (mg/L)	Chl *a* (µg/L)	Secchi Depth (m)
2017	7.5 (1.9)	8.4 (0.4)	365 (4)	0.061 (0)	0.01 (0)	1.8 (0.4)	7.8 (1.5)	0.7 (0.1)
2018	6.5 (2.8)	8.2 (0.3)	366 (4)	0.043 (0)	0.016 (0)	1.1 (0.5)	8 (2.8)	1 (0.2)
2019	8.1 (2.3)	8.1 (0.1)	411 (79)	0.036 (0)	0.011 (0)	1.9 (1.5)	0.5 (0.6)	0.8 (0.1)
2020	7.1 (0.3)	8.2 (0.1)	726 (223)	0.045 (0)	0.023 (0)	0.8 (0.3)	12.5 (0.7)	1.2 (0.2)
2021	7.5 (1.6)	8.2 (0.3)	406 (6)	0.041 (0)	0.010 (0)	0.7 (0.2)	2.7 (3.4)	1.4 (0.1)
2022	4.6 (3.0)	7.9 (0.3)	403 (21)	0.041 (0)	0.003 (0)	1.5 (0.4)	18.3 (3.7)	1.2 (0)
2023	5.3 (3.6)	8 (0.4)	417 (21)	0.175 (0.1)	0.048 (0.01)	1.9 (0.1)	22.3 (8.1)	1.3 (0)
2024	5.5 (3.6)	7.9 (0.4)	406 (9)	0.071 (0.1)	0.030 (0.05)	1.1 (0.4)	16.8 (4.2)	1.2 (0.1)

Sediment Core Internal P Loading Study: Fenner's Ditch

Analysis of the water samples collected throughout the incubation period show the change in water column TP and SRP concentrations (Figures 21 and, respectively) as P is released from, or moves into, the sediment. Concentrations of P tended to equilibrate or plateau by day 10 with the exception of a single anoxic core from site 3, wherein P concentrations continued to increase throughout the entire incubation.

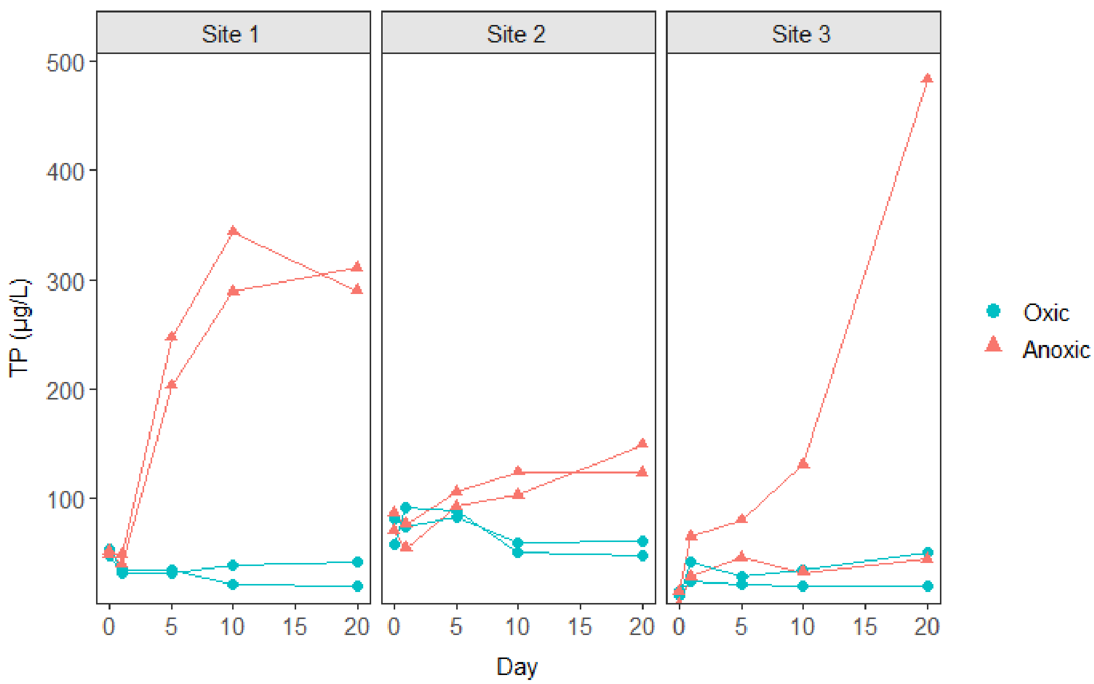

Figure 21. Change in TP concentration (µg/L) over duration of incubation for sediment cores from 3 sampling sites in Fenner's Ditch exposed to either oxic or anoxic conditions.

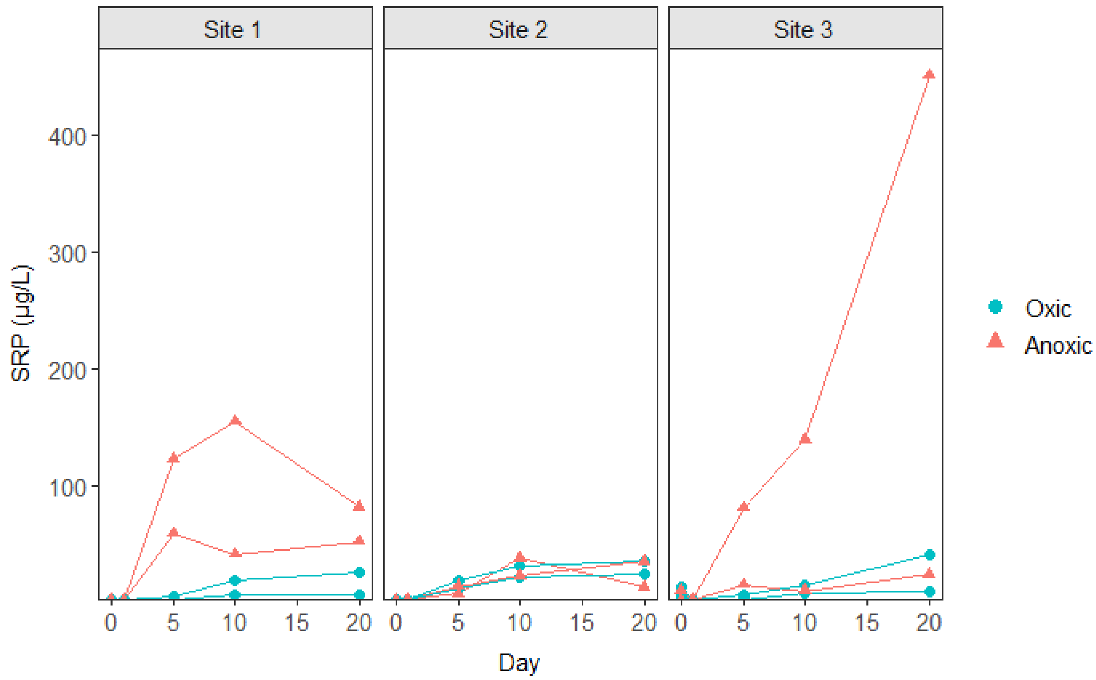

Figure 22. Change in SRP concentration (μg/L) over duration of incubation for sediment cores from 3 sampling sites in Fenner's Ditch exposed to either oxic or anoxic conditions.

Based on the P release rates in Figures 21 and 22, we calculated the apparent maximum TP flux for both TP and SRP. This apparent maximum flux is based on when releases are linear (or close to it) and showed the highest slopes; as a consequence, the 3 consecutive dates vary at each site and for each replicate (see Tables 7 and 8 for dates used to make calculations). The other caveat to this analysis is that these rates will be "worst case" scenarios, since we are calculating maximum rates. However, when planning management actions, it is useful to know worst case situations.

For TP (Table 7), the data reveal two patterns. First, apparent maximum flux was greater in anoxic (no or very low dissolved oxygen levels) than oxic conditions; this contrast was statistically significant (H = 8.3077, df = 1, p-value = 0.004). No significant differences between sites nor interaction between treatment and site were identified. Second, release rates were highest at site 1 (closest to Bear Lake) compared to sites 2 and 3.

For SRP (Table 8), again maximum apparent flux was marginally significantly greater under anoxic than oxic conditions (H = 6.56, df = 1, p-value = 0.010). Qualitatively, this difference was more pronounced at site 1 (closest to Bear Lake) and site 3 (above the dam), though neither sites nor interaction between site and redox state was found to be statistically significant.

Table 7. Replicate values for apparent maximum TP flux (i.e., highest flux from a minimum of three consecutive days) for each combination of site and redox state. Data show the rate and the time span form which release rate was calculated; compare to dates in Figure 21.

	Apparent Max TP Flux (mg/m^2/day)			
	Oxic		Anoxic	
Site 1	-0.25, 0.19	Days 5-20, 1-10	7.70, 9.80	Days 0-5, 0-5
Site 2	0.05, 1.50	Days 0-5, 0-5	1.33, 1.36	Days 1-10, 1-10
Site 3	0.30, 0.90	Days 0-5, 0-5	2.23, 6.72	Days 0-5, 5-20

Table 8. Replicate values for apparent maximum SRP flux (i.e., highest flux from a minimum of three consecutive days) for each combination of site and redox state. Data show the rate and the time span form which release rate was calculated; compare to dates in Figure 22.

	Apparent Max SRP Flux (mg/m^2/day)			
	Oxic		Anoxic	
Site 1	0.10, 0.46	Days 1-10, 1-10	2.83, 6.03	Days 0-5, 0-5
Site 2	0.51, 0.83	Days 1-10, 0-5	0.58, 0.99	Days 0-5, 1-10
Site 3	0.15, 0.57	Days 1-10, 5-20	0.63, 6.17	Days 0-5, 5-20

Organic matter (OM) and sediment TP concentrations in sediment cores from Fenner's Ditch varied by site; we refer to “field” cores as those used to determine conditions at the site from which the cores were extracted, whereas “experimental cores” are those used in our lab manipulations. Both field and experimental cores were taken in the same area at each site. Site 3 (upstream of the dam) sediment had the highest concentrations of both OM and TP, with OM% ranging 24-30% and TP ranging 847-1398 mg/kg (Table 9, Figures 23 and 24). Site 2 (immediately downstream of the dam) sediment had the lowest levels of OM and TP (1-3% and 76-141 mg/kg; Table 9, Figures 23 and 24), likely because most of this material collected behind the dam. Phosphorus-binding metals, including aluminum, calcium, iron, and magnesium were distributed similarly to OM and TP among sites (Figures 25 and 26).

Jensen et al. (1992) in a survey of 15 Danish lakes found that the release of soluble reactive phosphorus was negatively correlated with the surface sediment Fe:P ratio; they found that internal P loading was negatively correlated when oxic sediments had ratios of total Fe:TP > 15 (by weight). In Bear Lake, the field sediments in Fenner’s Ditch exceeded 15:1 at sites 1 and 2 (Table 9), but all of our experimental cores had TFe:TP ratios <15 (Table 9), suggesting that there was insufficient iron to naturally bind phosphorus in the sediment in our core tubes. Site 3 had relatively high Fe in its sediment

(Figure 29) but because its P concentration was so high (Figure 24), its sediment had the lowest iron Fe:P ratio (Table 9), which was likely a major factor contributing to its high P release rates.

Concentrations of the four phosphorus fraction extracts exhibited a similar distribution of P fractions within a site, with relatively little difference between oxic and anoxic treatments (Figure 27). The major difference among sites was in the HCl fraction (the stable P fraction that is bound largely to calcium and magnesium); this fraction was greater in site 2 sediment but lower in the site 3 sediment. In contrast, the BD fraction (reductant-soluble P that includes redox-sensitive iron hydroxide and manganese-bound P) showed the opposite trend to the HCl fraction (Figure 27). The sodium hydroxide (NaOH) extracted P fraction, which is bound to aluminum and iron, also followed the pattern of the BD fraction. Loosely sorbed P, represented by the ammonia chloride (NH_4Cl) fraction, was the smallest phosphorus fraction across all sites and cores and never exceeded 2%.

Table 9. Mean (±SD) sediment characteristics from Fenner's Ditch field sediment cores and incubated oxic and anoxic cores. OM = organic matter, TP = total phosphorus.

		OM%	TP (mg/kg, dry wt)	TFe: TP (by weight)
Site 1	Field core	9.9%	241	25
	Oxic cores	8.5% (8.4-8.5%)	564 (534-594)	9 (8-10)
	Anoxic cores	10.9% (8.4-13.4%)	522 (458-587)	13 (8-19)
Site 2	Field core	1.1%	76	17
	Oxic cores	2.0% (1.5-2.5%)	116 (114-117)	12 (11-12)
	Anoxic cores	2.0% (1.0-3.1%)	141 (68-215)	12 (7-16)
Site 3	Field core	24.1%	1028	7
	Oxic cores	21.6% (15.6-27.6%)	920 (847-992)	8 (6-9)
	Anoxic cores	29.2% (28.2-30.1%)	1211 (1024-1398)	7 (5-9)

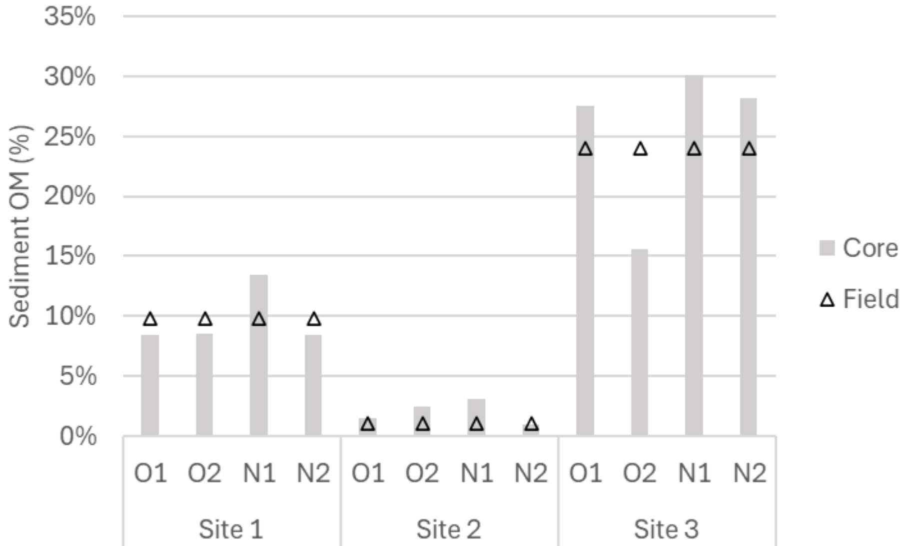

Figure 23. Sediment core (bars) and field site concentrations (symbols) of organic matter (OM) collected for internal phosphorus loading and fractionation study.

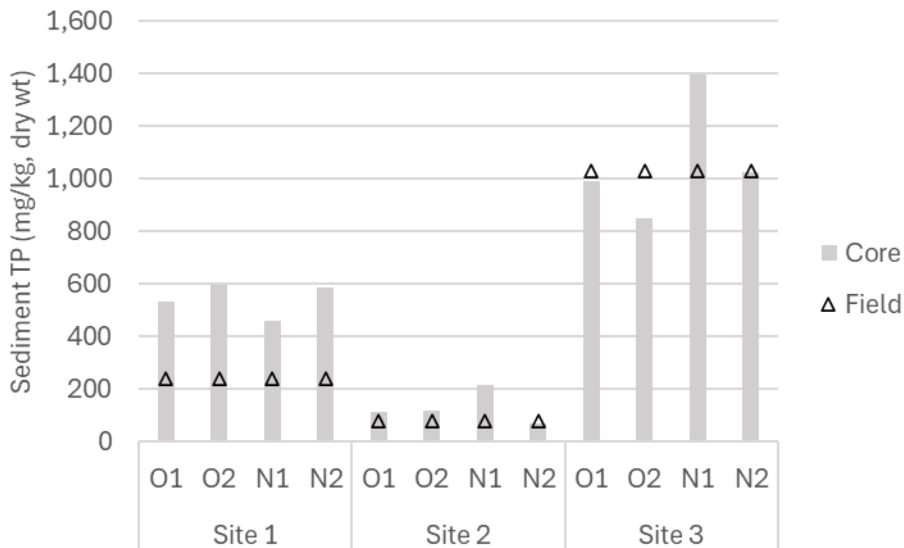

Figure 24. Sediment core (bars) and field site concentrations (symbols) of sediment total phosphorus (TP) collected for internal phosphorus loading and fractionation study.

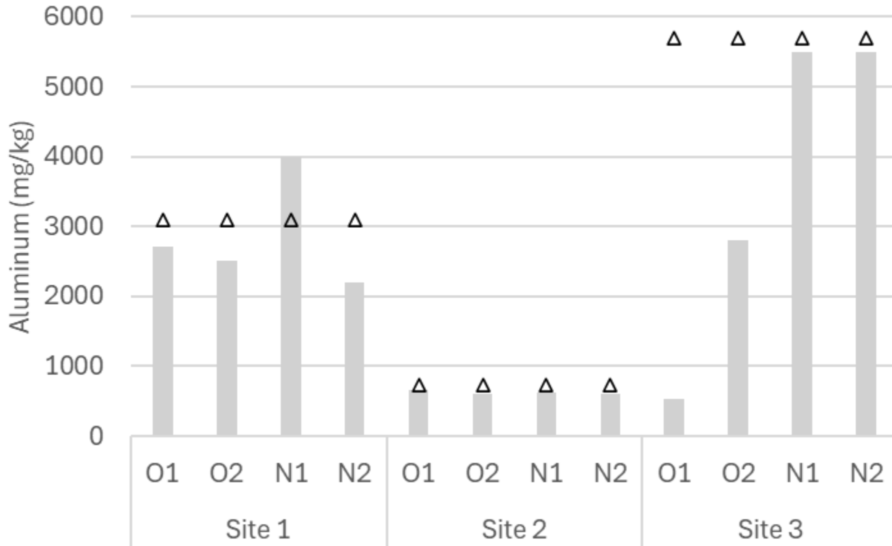

Figure 25. Sediment core (bars) and field site concentrations (symbols) of aluminum (Al) collected for internal phosphorus loading and fractionation study.

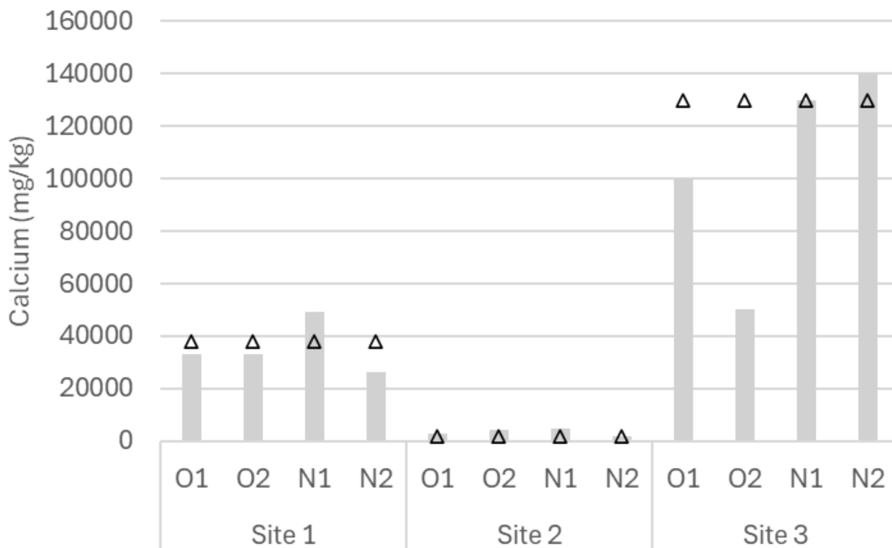

Figure 28. Sediment core (bars) and field site concentrations (symbols) of calcium (Ca) collected for internal phosphorus loading and fractionation study.

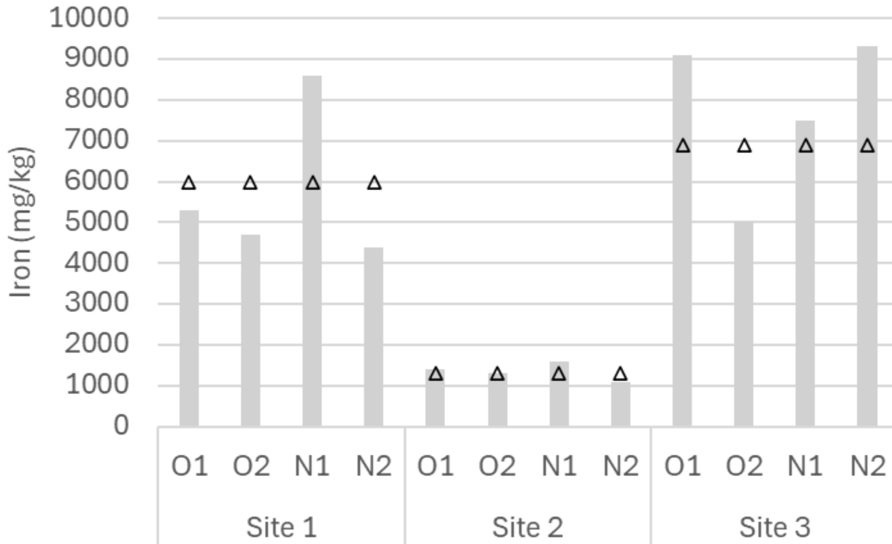

Figure 29. Sediment core (bars) and field site concentrations (symbols) of iron (Fe) collected for internal phosphorus loading and fractionation study.

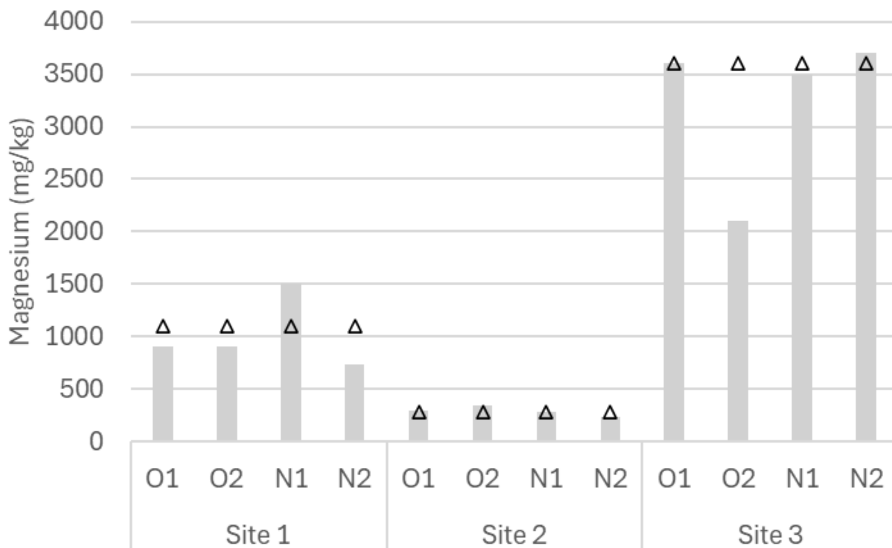

Figure 26. Sediment core (bars) and field site concentrations (symbols) of magnesium (Mg) collected for internal phosphorus loading and fractionation study.

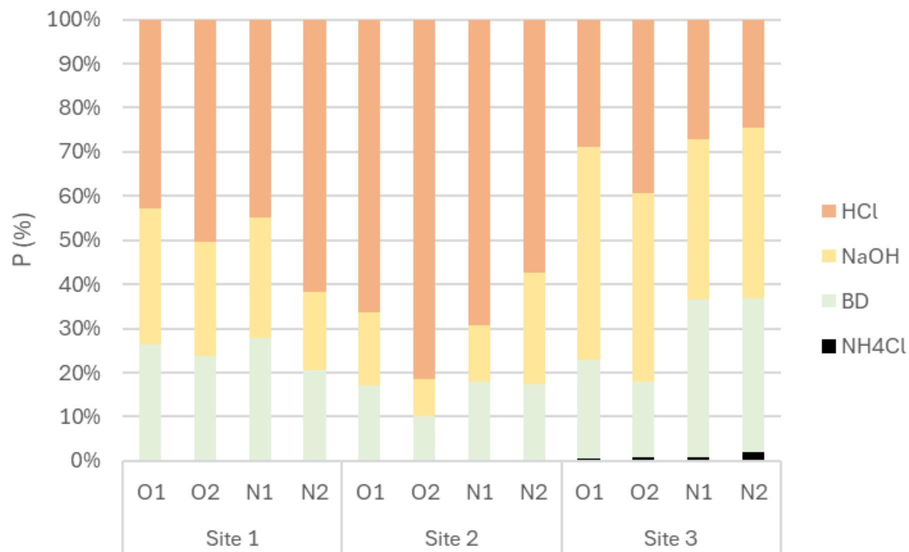

Figure 27. Sediment P fractions (top 0-10 cm) shown as stacked columns by internal loading study core. Note that SRP concentrations from NH_4Cl (loosely sorbed P) are too small to appear in most cores.

Overall, the sediment P release experiment revealed that 1) there is considerable variability both within a site and among sites in Fenner's Ditch regarding sediment P release rates; 2) low oxygen conditions stimulate P release, indicating that redox state (related to iron concentrations) is an important factor in controlling P concentrations in the ditch; 3) the most upstream and downstream sites in the Ditch should receive priority for restoration if the entire Ditch is not to be addressed; and 4) various remediation approaches should be assessed to address P reduction in Fenner's Ditch, as each has potential strengths and weaknesses. These include chemical additions, dredging, and aeration. They are discussed in more detail in the Discussion.

Lake Phytoplankton

Four of the five most abundant phytoplankton taxa observed in Bear Lake are known cyanotoxin producers, and hence, have the potential to be harmful to humans (Table 8, Figure 33). As noted earlier, the microcystin concentrations measured in Bear Lake were below the EPA threshold for recreational waters (see Figure 13); nonetheless, microcystin concentrations (the most common cyanotoxin) increased over the 2024 sampling season, reaching almost 2.5 µg/L at site 3 in October. We did not identify which taxon (or taxa) were responsible for the toxin production, but it likely one or several of the 3 most abundant cyanobacteria, and future monitoring should make microcystin analysis a priority. *Cylindrospermopsis* is typically a subtropical taxon but was first identified in the Great Lakes in

Muskegon Lake (Hong et al. 2006), although studies have shown that it does not produce its toxin, cylindrospermopsin, in the Great Lakes.

Table 8. Mean abundance of cyanobacteria (blue-green algae) biovolume compared to all algae divisions and mean abundances of observed cyanobacteria genera as respective biovolumes compared to all other cyanobacteria. Bold text indicates genera capable of producing cyanotoxins.

Division	Genus	Mean Biovolume (μm^3/mL)	%Total Biovolume
Cyanobacteria	***Aphanizomenon***	31,389,568	17%
Cyanobacteria	***Limnothrix***	27,795,331	15%
Cyanobacteria	***Oscillatoria***	27,431,823	15%
Pyrrhophyta	*Ceratium*	26,799,071	15%
Cyanobacteria	***Cylindrospermopsis***	21,879,244	12%
Bacillariophyta	*Synedra*	12,404,819	7%
Bacillariophyta	*Aulacoseira*	8,274,948	4%
Chlorophyta	*Oocystis*	8,036,933	4%
Cryptophyta	*Cryptomonas*	4,573,287	2%
Cyanobacteria	*Planktolyngbya*	3,609,659	2%

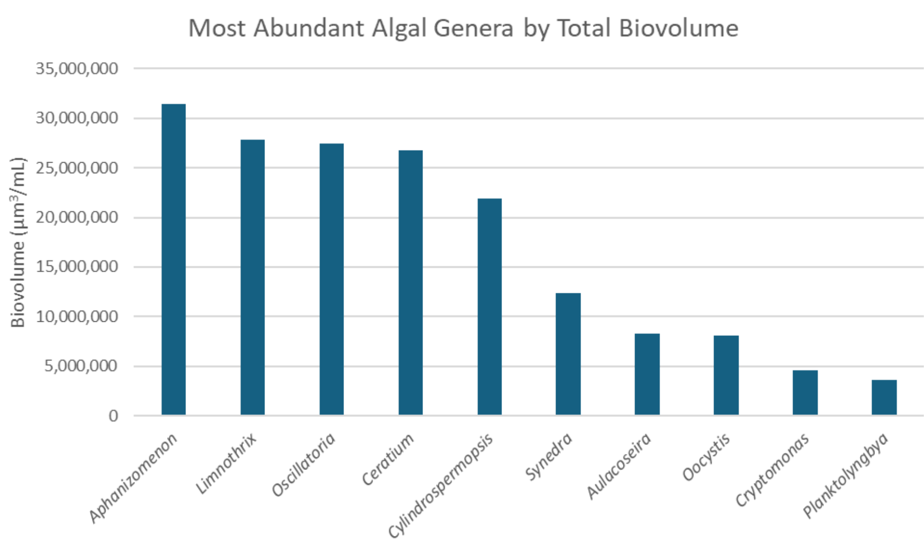

Figure 32. Ten most abundant algal genera based on biovolume.

The spatial variation in community structure among sites was apparent mostly with respect to site 4, where dinoflagellates (mostly *Ceratium*) were most abundant. Elsewhere, cyanobacteria dominated community structure based on biovolume, followed by diatoms, chlorophytes, and dinoflagellates (Figure 33). It is likely the influence of both Bear Creek and the restored celery flats result in the different algal composition at site 4; this influence is diluted by the time the water reaches site 3, leading to dominance by the cyanobacteria.

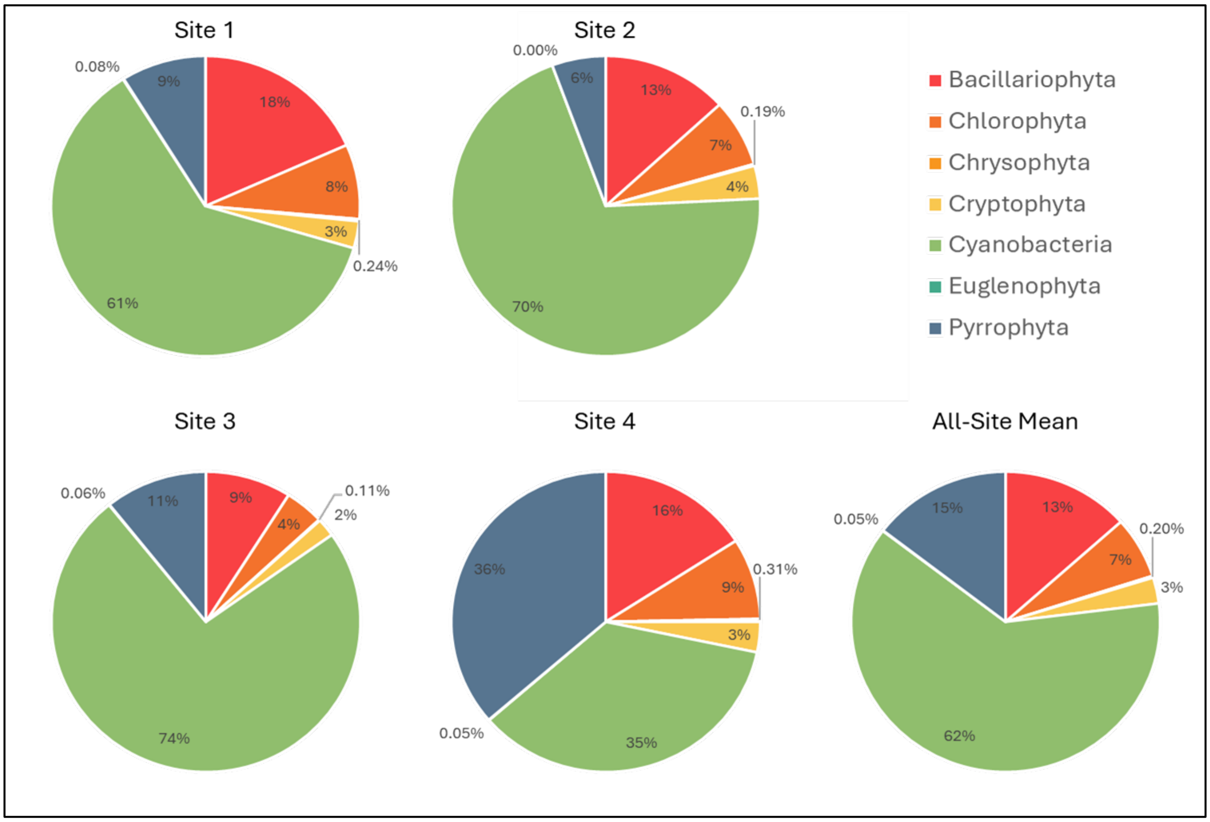

Figure 33. Relative abundance by biovolume of all observed phytoplankton taxonomic divisions.

Summary

Similar to our assessment in 2023, results from our 2024 sampling campaign provide a mixed signal for Bear Lake's spring and summer environmental conditions. From a regulatory perspective, the greatest concern is total phosphorus, given that the lake has an established TMDL of 30 µg/L. Spring TP concentrations barely exceed this threshold, although the summer TP concentrations were well above the TMDL. This exceedance was due to one bottom sample that was extremely high (similar to what occurred in 2023). The TMDL for TP is based on lakewide TP concentrations throughout the growing season of April to September (MDEQ 2008). In 2023, the mean TP concentration of near surface and near bottom depth across all sampling months was 60 µg/L, double the TMDL. In 2024, the mean TP concentration of near surface and near bottom depth across all sampling months was 42 µg/L, largely due to the reduced concentration of summer TP at the near bottom of site 1 (declined from ~740 to 180 µg/L). It is possible that our limited sampling may have skewed the TP concentrations (either high or low) but with additional sampling over time, trend analysis will be more robust. Continued chemical applications to bind P, especially in "hot spots" of high phosphorus, along with the continued maturation of the restored wetland in the former Willbrandt celery fields, should result in a continued decline in TP concentrations.

As was the case in 2023, the algal community, as reflected both in terms of biomass (chlorophyll *a*) and species composition, continues to indicate some degree of water quality impairment. Chlorophyll continues to exceed the target concentration of 10 µg/L, and cyanobacteria (blue-green algae) continue to dominate in the summer months at most sampling sites. The past few summers have seen persistent cyanobacteria blooms in many west Michigan lakes, likely in response to warm temperatures and abundant nutrients, so Bear Lake's condition in 2024 is not unique. The increasing microcystin concentrations in the autumn surface waters are still below federal guidelines but should continue to be monitored given their increases into autumn.

Our study of tributaries in 2023 revealed that the major tributaries flowing into Bear Lake were not major sources of phosphorus, which led us to study Fenner's Ditch. It appears that remaining phosphorus sources of any significance include Fenner's Ditch, as well as a few sediment hot spots at deep holes in the lake. Our sampling in Fenner's Ditch revealed moderately high P release rates at the location sampled closest to Bear Lake, as well as one of two replicates sampled at the furthest upstream site, above the dam. The spatial variation in release rates, even at the same site, is not unusual, as sediment content can vary substantially even within inches of one another. The highest release rates we measured in Fenner's Ditch, which as noted in the text are likely worst-case scenarios, rival rates measured in other eutrophic lakes, both locally and globally (Spears and Steinman 2020). This suggests that management actions should be either continued or accelerated in Fenner's Ditch. These legacy sources may remain problematic for years without intervention, but eventually will deplete themselves even if no additional actions are taken, assuming no new sources of phosphorus are introduced into the Ditch.

We also looked at the trends in water quality going back to 2017 but with an important caveat. The data from 2017 to 2021 were generated by RLS, whereas the data from 2022 to 2024 were generated by AWRI, using different methods. Hence, any comparison over the entire period of record should be done cautiously. It is more appropriate to examine trends from 2017 to 2021 separately from those using 2022 to 2024. The 2017-2021 trends are affected by an anomalous 2020, where the concentrations of most parameters exhibited unusually poor environmental conditions. Things improved in 2021, the final year of RLS' contract. Between 2022 and 2024, there were few changes in the spring data, but summer 2023 data reveal substantial increases in N and P, which dropped back down again in 2024. As noted in prior reports, until a sufficient data set is collected over time, once a month sampling can give an unrepresentative portrayal of true environmental conditions if sampling coincides with an unusual event. Our data set is a good start in examining trends, but additional data are needed to provide a robust picture of lake status.

The Bear Lake Dashboard (Appendix A) provides a quick and intuitive way to review the total phosphorus (TP), chlorophyll a, and water clarity data. The methods for TP and water clarity are fairly standard, so comparing data over time, despite coming from different sources, is reasonable. Although TP is higher than desired, it is evident that concentrations in spring and summer have been declining with time, while summer concentrations continue to vary (Figure A1). In contrast, the substantial increase in chlorophyll concentration since 2022 is most likely attributable to a change in how chlorophyll was measured in the RLS years vs AWRI years (Figure A2). The water clarity data, which is based on Secchi disk depth, and hence relies simply on the human eye, are comparable across time; the data show an increase in clarity up until about 2021, and a slight decrease since then, regardless of season (Figure A3).

Recommendations:

1) **Bear Lake monitoring**: As noted in previous reports, our monthly snapshot monitoring is useful for assessing long-term trends but can miss critical events between sampling dates or potentially give unrepresentative information in the short-term (for example, sampling during a short-lived algal bloom can lead to an overstatement of bloom conditions). Over time, with a sufficient sample size, those anomalous events will have less effect. As a consequence, we recommend continuing the 4 in-lake sites, but suggest the Lake Board consider the purchase of a sonde that would provide continuous water quality data throughout the ice-free seasons. The White Lake Association (WLA) recently received several grants, including one from the Community Foundation for Muskegon County, to help purchase a sonde in 2024, to be deployed in 2025.

These sondes do require that an individual(s) be dedicated to their maintenance and operation, but the benefits can be significant over time. The contact person at the WLA is Jim DeBoer (jdeboer007@hotmail.com) if there is interest. I have inserted below information on the sensor they purchased; more information is available at this website: https://www.nexsens.com/.

2. The Fenner's Ditch data indicate that its sediments are a likely source of phosphorus to Bear Lake. Hence, continued application of chemical inactivants is recommended to control this source. This approach uses a chemical such as EutroSORB, Phoslock, or alum; other approaches include adding iron (to bind phosphorus) or installing aeration, to keep the sediment from going anoxic, which causes the phosphorus to desorb from iron and be released into the water column. All of these approaches have been shown to be effective in other systems, but they do not address the underlying problem and require repeated application over time. Aeration has an upfront infrastructure cost, as well as the need for a power supply. An alternate solution is dredging the high phosphorus sediment out of Fenner's Ditch. This can be costly, requiring permits and disposal costs; the Lake Board may want to consider funding a feasibility study that addresses dredging. If there is interest in such action, we recommend they have a conversation with GEI, Inc. The appropriate contact is Brian Majka: bmajka@geiconsultants.com.

3. The phytoplankton composition indicates the possibility of cyanotoxin production in Bear Lake. This is not a new phenomenon as we identified these toxin-forming taxa in Bear Lake in the past (Xie et al. 2011). Microcystin concentrations in Bear Lake continue to remain below the threshold established for

recreational water use by US EPA, although the levels were higher in October than in prior years. This may be due to climate (longer, warmer summers) and hence beyond any lake management action but it does warrant additional vigilance in the future.

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Appendix A

Bear Lake 2024 Dashboards

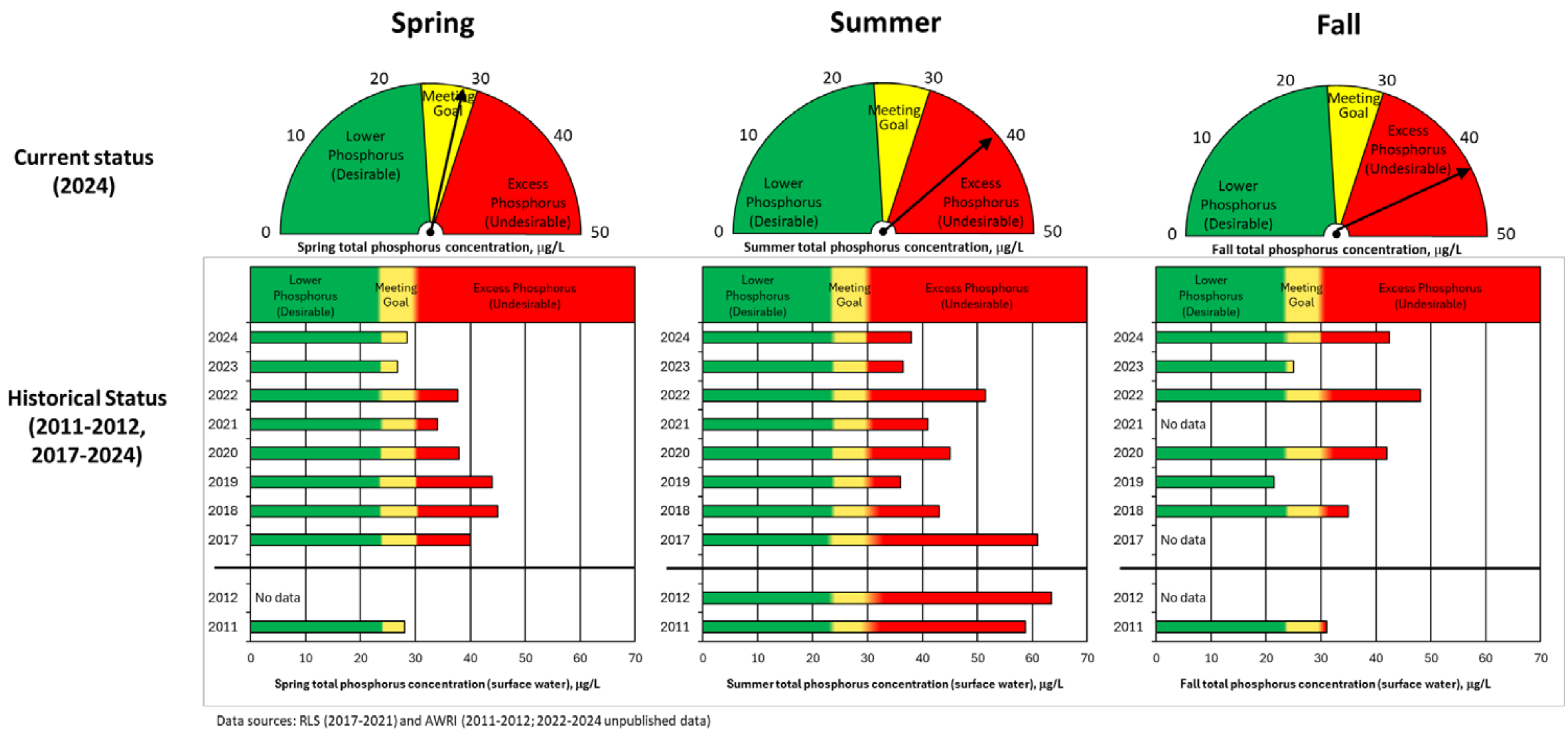

Figure A1. Bear Lake 2024 total phosphorus seasonal dashboard. Classifications are based on >30 µg/L "undesirable" threshold of Bear Lake TMDL, and the <30 µg/L "meeting goal" threshold and <24 µg/L "desirable" threshold of the Muskegon Lake long-term monitoring dashboard.

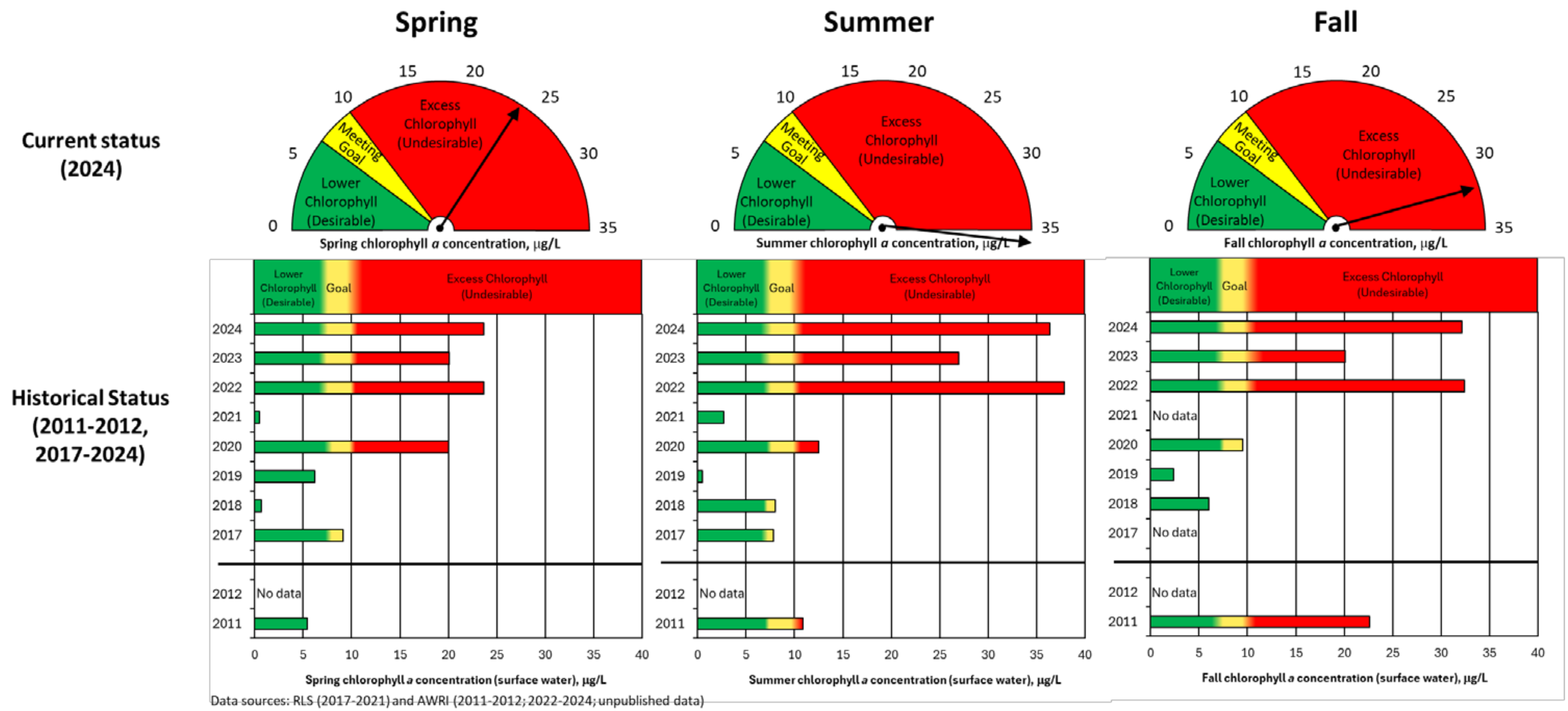

Figure A2. Bear Lake 2024 chlorophyll *a* seasonal dashboard. Classifications are based on >10 µg/L “undesirable” threshold, <10 µg/L “meeting goal” threshold, and <7.3 µg/L “desirable” threshold of the Muskegon Lake long-term monitoring dashboard.

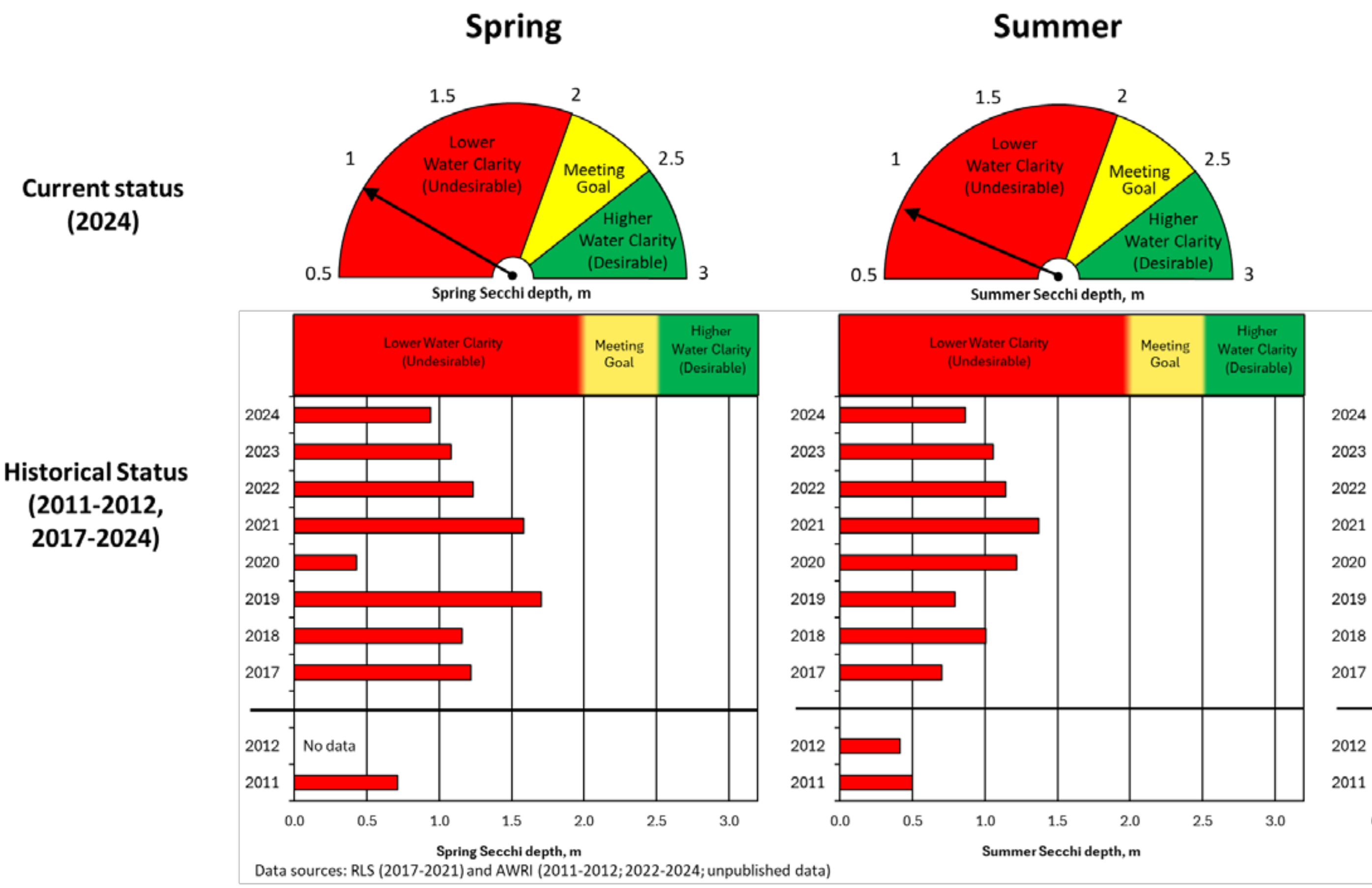

Figure A3. Bear Lake 2024 Secchi disk depth seasonal dashboard. Classifications are based on >2 m "undesirable" threshold, <2 m "meeting goal" threshold, and <2.5 m "desirable" threshold of the Muskegon Lake long-term monitoring dashboard.