Green Lake Watershed Assessment

December 2022

Alan D. Steinman* and Michael C. Hassett

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

The ecological health of lakes is currently under threat from a changing climate, land use, and invasive species, among other stressors (Allan et al. 2013). Increasing temperatures can result in stronger and longer periods of water column stratification, exacerbating tendencies of hypolimnetic hypoxia or anoxia (Jane et al. 2022). Land use change, from either more intensive agricultural practices or urban development, can result in more nutrient runoff leading to the proliferation of harmful algal blooms (Ho et al. 2019). Finally, aquatic invasive species have the potential to dominate lake community structure, outcompeting native species, some of which have high cultural and recreational value (Hanley and Roberts 2019). For those lakes that are not yet impacted, preserving their health makes both economic and ecological sense.

This report is focused on the ecological health of Green Lake (Allegan County, Michigan), which currently supports a strong cisco (*Coregonus artedi*) population. However, there is concern that the surrounding land use, which is largely agricultural (52% of the watershed) and urban developed (42% of the shoreland) as of 2006, may be contributing pollutants that may threaten the cisco population due to changing lake conditions. The Michigan Department of Natural Resources has identified Green Lake as a conservation priority because of its high water quality and its vulnerability to anthropogenic disturbances. Specifically, the major stressors of concern in Green Lake that we examined were 1) increasing water temperatures leading to longer periods of stratification and potential hypoxia/anoxia and 2) increasing nutrient concentrations due to watershed runoff. Although phosphorus is usually considered the limiting nutrient in freshwater systems (Schindler 1977), there is a growing realization that both phosphorus (P) and nitrogen (N) can co-limit algal growth in lakes, so we measured both P and N (Conley et al. 2009; Maberly et al. 2020).

Given the paucity of information on nutrient inputs to Green Lake, this study assessed key sources of nutrients in the watershed and constructed a phosphorus budget for the lake. We also measured the dissolved oxygen status in the lake with oxygen and temperature profiles to assess cisco habitat. This information was used to determine the overall trophic status of Green Lake; which nutrient, if either, is currently limiting primary productivity in Green Lake; relative and absolute contributions of N and P from the major inputs to Green Lake; and the type of location of the Best Management Practices that would be the most effective in the Green Lake watershed.

Methods

Lake Water Quality Sampling

Green Lake water quality was measured via canoe 1-2 times monthly by Allegan Conservation District (ACD) from May 2021 – May 2022; specific dates are provided in Table 1. Three sites were established based on their geographic spread throughout the lake and were sampled at surface depth via grab sampling and at middle and near-bottom depths via a Van Dorn water sampler (Table 2, Fig. 1). Water samples were collected in 500-mL bottles, stored on ice, and returned to the lab for nutrient analysis, usually within 4 hours. On several seasonal sampling dates (Table 1), staff from Grand Valley State University's Annis Water Resources Institute (AWRI) joined the ACD field crew and additionally measured general water quality parameters including dissolved oxygen (DO), temperature, pH, specific conductivity (SpCond), total dissolved solids (TDS), turbidity, and chlorophyll *a* using a YSI 6600 multiparameter sonde (YSI, Inc., Yellow Springs, OH). Lake sampling was halted during dangerous winter conditions when ice was too thick to canoe through or too thin to walk on. Some winter lake sampling occurred through the ice via an auger (Table 1).

After returning to the lab, water from each 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 both nitrate (NO_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^-, and TKN were analyzed on a SEAL AQ2 discrete automated analyzer (U.S. EPA 1993). Any values below detection were reported as ½ of their respective detection limits.

Table 1. Dates and locations of field sampling events for water quality monitoring.

Year	Month	Day	Sampling Event			
			Lake	Tributaries	Wells	Notes
2021	May	14	X			Lake YSI
		25	X	base		
		26		storm		
	June	11	X			
		28	X	storm		
	July	9	X		X	
		23		storm		
		27	X	base	X	Lake YSI
	August	16	X			
		30	X	base	X	
	September	10	X			Lake YSI
		27	X	base	X	
	October	22	X	base	X	Lake YSI
	November	5	X	base	X	
		23	X			
	December	10	X	base	X	
2022	January	14		base	X	
	February	14	X	base	X	
	March	9		base	X	
	April	12	X	base	X	
		13		storm		
	May	12	X	base		Tributary *E. coli* & MST*
		25		storm		Tributary *E. coli* & MST
	June	10			X	

*Molecular Source Tracking

Table 2. Lake site coordinates.

Site ID	Name	Latitude	Longitude
Lake1	S Lobe	42.7493	-85.5970
Lake2	Narrows	42.7551	-85.5906
Lake3	N Lobe	42.7549	-85.5565

Figure 1. Map of lake and tributary water quality monitoring sites. Lake sites are marked as red squares and tributary sites are marked as yellow stars on the map.

Tributary Water Quality Sampling

Water quality monitoring sites were established at Green Lake's southern outflow and two inflowing tributaries on the northwest and northeast sides of the lake (Fig. 1) and sampled monthly at baseflow conditions (sampling days preceded by 72 hours of dry conditions) by the Allegan Conservation District (Tables 2, 3). Additional storm flow samples were collected when local weather gauges indicated at least 0.25" of precipitation had fallen during a single storm event. TP and SRP were measured during the whole monitoring period from May 2021 – May 2022 and NO_3^- and TKN were additionally measured from October 2021 onward. Water samples were collected and analyzed as described above.

Table 3. Tributary site coordinates.

Site ID	Name	Latitude	Longitude
Trib1	S Outlet	42.7463	-85.5941
Trib2	NW Inlet	42.7585	-85.5945
Trib3	NE Inlet	42.7558	-85.5827

Homeowner Well Sampling

Homeowner well water samples were monitored as a proxy for groundwater and were collected by first identifying willing homeowners around Green Lake whose homes had wells and that would grant permission to ACD staff to sample from their hose spigots (Fig. 2), which are generally not hooked up to water softener or home filtration systems, which would bias our samples. Groundwater was collected monthly from July 2021 – June 2022 (Table 1). Spigots were sanitized with bleach wipes and allowed to air dry for several minutes prior to sampling. Homeowner well flow rate and pipe volume were retrieved from the State of Michigan's Wellogic website and were used to estimate the time needed to completely flush well water lines prior to water collection. Water samples were collected in 500-mL plastic bottles, then handled and processed as described above for TP, SRP, and NO_3^- analyses.

Figure 2. Map of Green Lake homeowner well water quality monitoring sites. Sites are marked as yellow pushpins on the map.

E. coli and Microbial Source Tracking

We also collected water samples from tributaries on an opportunistic basis to measure *E. coli* concentrations and potential sources in tributary inlets during base and storm flow events. Water was collected in sterile bottles at the left, center, and right areas of transects at tributary sites described above and a 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. If the geometric mean of transect replicates was determined to exceed a threshold of 300 colony-forming units (cfu) per 100 mL, then the remaining site sample water was composited and analyzed for microbial source tracking to determine the origin of the sampled *E. coli* (Flood et al. 2022). Source organism genetic markers examined include human (HF183), dog (DG3), ruminants (Rum2Bac), and porcine (Pig2Bac) with Sketa as a sample processing control.

Lake Sediment Phosphorus Fractionation

Two sediment cores were collected in August 2022 from the alluvial fans of the NW and NE tributary inlets in Green Lake from a jonboat using a piston coring apparatus (Fisher et al. 1992; Davis and Steinman 1998). Corers were constructed of a 0.6-m long polycarbonate core tube (7-cm inner diameter) marked in 1-cm increments and a polyvinyl chloride (PVC) attachment assembly for coupling to aluminum drive rods. The piston was advanced 20 to 25 cm prior to deployment to maintain a water layer on top of the core during collection. The corer was positioned vertically at the sediment–water interface and pushed downward with the piston cable remaining stationary. After collection, the core was brought to the surface and the bottom was sealed with a rubber stopper prior to removal from the water, resulting in an intact sediment core that was ~20 cm in length, with a 25-cm overlying water column. The piston was then bolted to the top of the core tube to keep it stationary during transit. Core tubes were placed in a vertical rack and maintained at ambient temperature during transit.

In the lab, the top 10 cm of sediments were extruded into a zip-seal bag and refrigerated at 4°C until laboratory analysis of sediment TP, organic matter (OM), ash-free dry mass (AFDM), and % solids. Sediments were homogenized by hand and then subsampled for subsequent analysis. Sediment OM, AFDM, and % solids were determined using gravimetric procedures (i.e., dry for 24 hours at 105˚C, weigh, ash at 550˚C for 1 hour, re-weigh; Steinman et al. 2017a). The resultant ashed material was used for analysis of sediment TP on a Seal AQ2 Discrete Analyzer (U.S. EPA 1993).

A separate subsample of 2 g of wet sediment was placed in centrifuge tubes for sequential phosphorus fractionation (Psenner et al. 1998; modified from Hupfer et al. 2009). Extracts were analyzed for the following sediment P fractions: 1) NH_4Cl-extracted labile P (loosely sorbed); 2) BD-extracted reductant-soluble P (iron hydroxides, Mn-bound); 3) NaOH-extracted Fe- and Al-bound P; and 4) HCl-extracted Ca- and Mg-bound P.

Table 4. Green Lake sediment coring coordinates.

Site	Latitude	Longitude
NW core	42.75789	-85.5938
NE core	42.75518	-85.5834

Phosphorus Source Budget

It is possible to estimate internal phosphorus loading (P release from the sediment) using the TP content in the sediment (Nürnberg 1988). This is a relatively crude estimate compared to measuring P flux directly or calculating hypolimnetic phosphorus accumulation (cf. Hupfer et al. 2020), it nonetheless provides a first order estimate. We calculated estimates based on both the TP and BD-P content in the sediment (Fig. 3). We then multiplied that flux (mg m^{-2} d^{-1}) by the surface area of the lake with DO concentrations of less than 2 mg/L, which is the nominal concentration when P is released from its bound form to ferric iron (Mortimor 1941). This value was then normalized by the period of time the lake was hypoxic. This internal load was compared to the external load based on our observed data (see above) to evaluate the relative sources of P to Green Lake.

Figure 3. Map of Green Lake sediment coring sites. Sites are marked as green circles on the map.

Hydrologic Characteristics

Green Lake bathymetry lake contours were retrieved from GIS Open Data for the State of Michigan (https://gis-michigan.opendata.arcgis.com/), vertices of line data were extracted, and depth values of the vertices were interpolated using Triangulated Irregular Network (TIN) analysis in QGIS (v3.12, QGIS Development Team). Interpolated depths were summarized as a raster volume to estimate the total volume of Green Lake and converted from 42,224,636.7 cubic feet to 969 acre-feet for reporting.

The drainage area of the whole Green Lake watershed, including the lake itself, was estimated using the Long Term Hydrologic Impact Analysis (LTHIA) model website (https://engineering.purdue.edu/~lthia/) at Green Lake's southern outflow and reported as 2478.81 acres (Fig. 4). Additional watershed drainage areas were determined for the northwest and northeast tributaries (1097.7 and 321.81 acres, or 44.28% and 12.98% of the total lake drainage area, respectively; Fig. 4).

The GIS Open Data website's Base Flow of Michigan Streams dataset lists baseflow of Green Lake discharge as an annual average of 2.014 m^3/s at its southern outflow (https://gis-michigan.opendata.arcgis.com/). As flow was not directly measured during this study, the average baseflow discharges of the NW and NE tributaries were estimated as a fraction of the whole lake drainage area and upstream tributary drainage areas. We did this by multiplying the subwatershed acreage (as a percent of total lake drainage) by the lake's annual average baseflow discharge to calculate tributary baseflow as 0.892 m^3/s (NW tributary) and 0.261 m^3/s (NE tributary). Although baseflow is known to generally vary seasonally depending on the amount of water in an aquifer, we acknowledge that this variance is not incorporated into this model and seasonality is reflected only by changes in measured P and N concentrations and by changes in precipitation values.

Precipitation data for July 2008 – October 2022 were downloaded from the National Oceanic and Atmospheric Administration's National Centers for Environmental Information (NCEI). Precipitation data logged from directly within Green Lake's watershed were not located, so local precipitation was estimated by averaging data from two nearby regional weather stations located ~5 miles southwest and ~ 11 miles north, respectively (Wayland 2.0 W, MI, US [US1MIAN0001] and Caledonia 4.4 WNW, MI, US [US1MIKN0043]). Daily precipitation average data were converted from inches to feet and multiplied by the LTHIA watershed acreage (2478.81 acres) to calculate daily precipitation as a volume (in acre-feet), which was then summarized into monthly precipitation volumes.

Hydraulic retention time (HRT) was calculated using a method from Lauber (1999) by incorporating TIN-derived lake volume, precipitation volumes derived from LTHIA acreage and NCEI daily precipitation, and stream discharge at baseflow from the State of Michigan as described above. As this calculation approach relies on an average annual discharge rate that is applied year-round, the resulting HRT has little precision regarding seasonal or monthly groundwater fluctuations in the water table but is still accurate as a first order approximation of annual HRT. HRT was calculated as follows, where Y = year and AF = acre-feet:

$$HRT\ (Y) = \frac{lake\ volume\ (AF)}{[stream\ discharge\ (AF/Y) + precipitation\ (AF/Y)]}$$

P baseflow loads were calculated by multiplying the monthly average baseflow volume to each respective monthly baseflow nutrient concentration. Concentrations from months that were not sampled were estimated as the average of the preceding and anteceding concentrations of a given site's nutrient timeseries.

$$Load\ \left(\frac{mg}{s}\right) = Concentration\ \left(\frac{mg}{L}\right) * Flow\ \left(\frac{m^3}{s}\right) * \left(\frac{1000\ L}{m^3}\right)$$

Finally, a comparison of Green Lake P loads from inflows and outflows was created for the May 2021 – April 2022 monitoring year. Tributary loading rates were determined using data and methods described above and multiplying the seasonal TP mean or annual TP mean with the corresponding seasonal or annual discharge and applying conversion rates as needed.

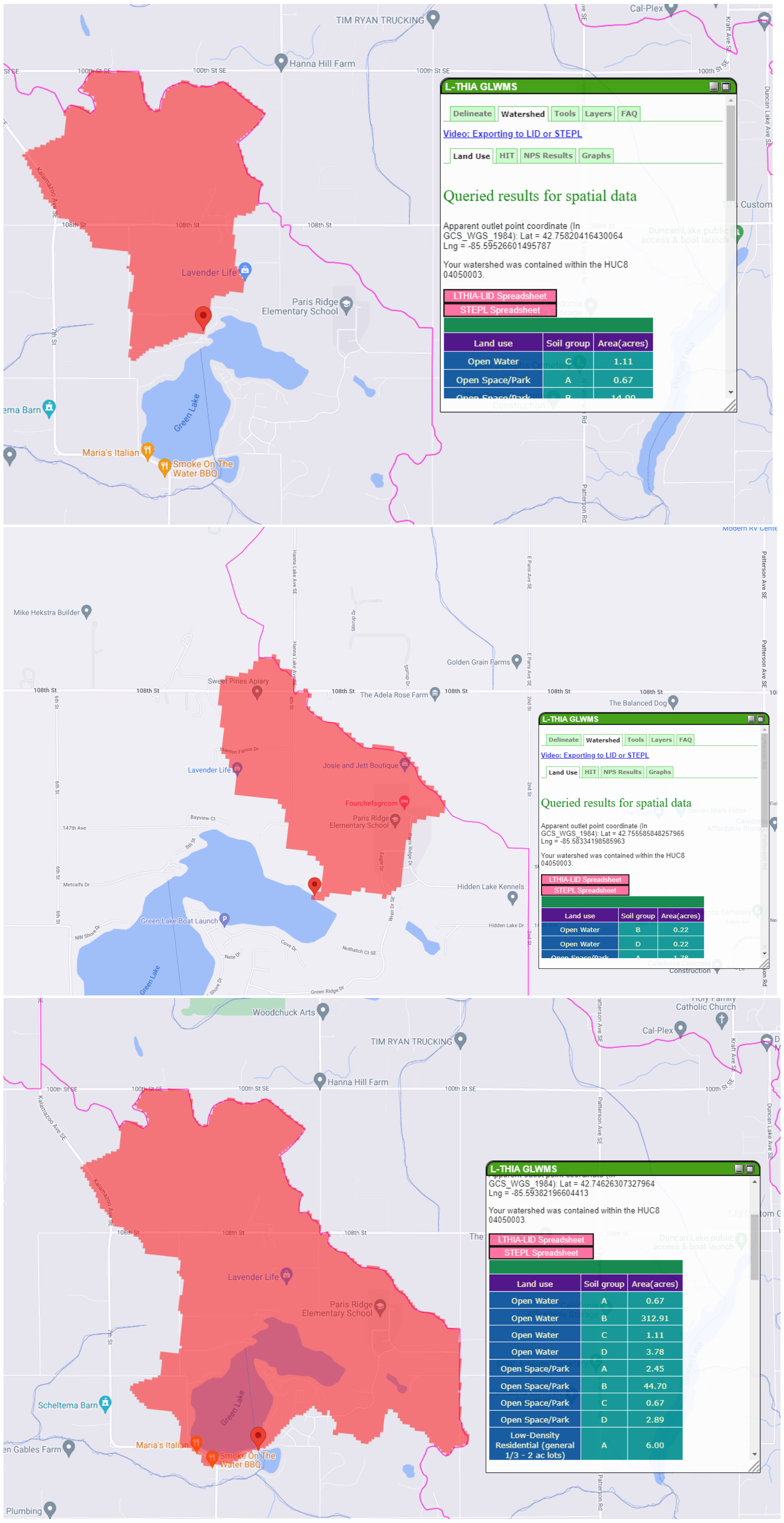


Figure 4. Screenshots of the LTHIA model website (https://engineering.purdue.edu/~lthia/) highlighting the drainage areas of Green Lakes NW tributary (top image), NE tributary (middle image), and S outlet, which encompasses the entire watershed (bottom image).

Results

Lake Water Quality Sampling

Green Lake's water is quite clear in the colder parts of the year with Secchi disk depths reaching down to >6 m (>33% of total depth) at measured sites (Table 5). As expected, this clarity declines (i.e., Secchi depths become shallower) in the warmer months, ranging as low as 0.6 m - 1.1 m in August 2021 (Table 5). Given that mean TDS did not increase in the summer and fall (Table 6), it is likely that algal growth is responsible for declining water clarity, which is consistent with the greater chlorophyll *a* concentrations during the warmer months (Table 6).

On average, seasonal means of site water quality parameters followed expected trends for west Michigan lakes (Table 6). Water temperature was highest at the surface and decreased with depth, with warmer temperatures in summer and fall. Dissolved oxygen (DO) concentrations declined with depth, and in summer and fall dropped below 1 mg/L at the lake bottom, indicating hypoxic conditions (Table 6). SpCond ranged from 360-470 µS/cm and was higher in deeper samples (Table 6), likely due to settling of denser ions (e.g., chloride), which do not get mixed into the epilimnion. Mean pH ranged 7.5-8.5, reflecting somewhat alkaline water conditions (Table 6). A three-season average of lake water quality parameters is provided in Table 7.

Table 5. Green Lake monthly averages of Secchi depth and total site depth

		Secchi Depth (m)			Total Depth (m)			% Secchi of Total Depth		
Year	Month	Lake1 S Lobe	Lake2 Narrows	Lake3 N Lobe	Lake1 S Lobe	Lake2 Narrows	Lake3 N Lobe	Lake1 S Lobe	Lake2 Narrows	Lake3 N Lobe
2021	May	5.5	4.4	4.3	20.0	15.0	15.8	28%	29%	27%
	Jun	3.8	2.2	2.5	19.0	14.5	15.5	20%	15%	16%
	Jul	2.1	2.1	2.1	19.7	15.8	16.6	11%	13%	12%
	Aug	1.1	0.8	0.6	19.0	14.5	16.0	6%	5%	4%
	Sep	2.5	2.4	2.2	19.3	15.3	16.3	13%	16%	14%
	Oct	4.9	4.9	5.2	19.0	14.5	16.0	26%	34%	32%
	Nov	4.6	5.0	4.6	19.9	15.5	16.0	23%	32%	29%
	Dec	6.6	6.7	6.2	20.0	15.5	17.0	33%	43%	37%
2022	Jan	ND	ND	ND	ND	ND	ND	ND	ND	ND
	Feb	ND	ND	ND	20.2	15.4	17.0	ND	ND	ND
	Mar	ND	ND	ND	ND	ND	ND	ND	ND	ND
	Apr	2.4	2.4	2.1	20.0	15.5	16.5	12%	16%	13%
	May	2.7	2.7	2.1	19.5	14.0	16.0	14%	20%	13%

Table 6. Seasonal means of select general water quality parameters (±SD) recorded seasonally on 5/14/2021, 7/27/2021, and 9/10/2021. DO = dissolved oxygen; SpCond = specific conductance; TDS = total dissolved solids; Chl *a* = sonde-measured chlorophyll *a*.

Parameter	Depth	Spring	Summer	Fall
Temperature (°C)	Surface	15.5 (0.2)	26.6 (0.3)	23 (0.1)
	Middle	10.6 (2)	13.7 (1.8)	15.7 (1.5)
	Bottom	6.8 (0.5)	7.2 (0.6)	7.3 (0.9)
DO (mg/L)	Surface	11.1 (0.1)	9.3 (0.1)	8.8 (0.1)
	Middle	11.9 (0.7)	7.1 (4.9)	4 (5.8)
	Bottom	4.3 (0.5)	0.9 (0.5)	0.3 (0)
DO (%)	Surface	111.6 (0.9)	115.5 (2.4)	102 (1.9)
	Middle	106.9 (3.3)	67.2 (43.9)	38.7 (55.9)
	Bottom	35.3 (4.2)	7.8 (3.9)	2.9 (0.3)
SpCond (µS/cm)	Surface	419.3 (5.7)	362.3 (0.6)	365.7 (7.6)
	Middle	422.3 (6)	417.0 (11.8)	416.3 (11.7)
	Bottom	435 (6.9)	445 (10.6)	466.7 (12.2)
pH	Surface	8.54 (0.04)	8.56 (0.03)	8.57 (0.02)
	Middle	8.35 (0.23)	7.92 (0.34)	7.79 (0.34)
	Bottom	7.51 (0.19)	7.47 (0.05)	7.43 (0.02)
TDS (g/L)	Surface	0.273 (0.004)	0.235 (0.001)	0.238 (0.005)
	Middle	0.274 (0.004)	0.271 (0.008)	0.271 (0.008)
	Bottom	0.283 (0.004)	0.290 (0.007)	0.303 (0.008)
Turbidity (NTU)	Surface	4.3 (3.5)	3.4 (1.4)	4.7 (0.1)
	Middle	2.8 (0.1)	1.3 (3.4)	5.8 (1.3)
	Bottom	5.1 (3.2)	22.9 (41.2)	8.2 (3.1)
Chl *a* (µg/L)	Surface	0.5 (0.4)	2.1 (1.6)	3.1 (1.3)
	Middle	4.9 (0.7)	5.9 (3.3)	5.4 (4.4)
	Bottom	5.5 (1.6)	12.9 (2.3)	11.7 (9.5)

Table 7. Whole lake means (±SD) of seasonally measured general water quality parameters.

Parameter	Surface	Middle	Bottom
Temp (°C)	21.7 (5.7)	13.3 (2.6)	7.1 (0.3)
DO (mg/L)	9.7 (1.2)	7.7 (4.0)	1.9 (2.1)
DO (%)	109.7 (6.9)	70.9 (34.2)	15.3 (17.5)
SpCond (µS/cm)	382.4 (32.0)	418.6 (3.3)	448.9 (16.2)
pH	8.56 (0.02)	8.02 (0.3)	7.47 (0.04)
TDS (g/L)	0.249 (0.021)	0.272 (0.002)	0.292 (0.010)
Turbidity (NTU)	4.1 (0.7)	3.3 (2.3)	12.1 (9.5)
Chl *a* (µg/L)	1.9 (1.3)	5.4 (0.5)	10 (4)

TP concentrations peaked in the summer months at all sites and increased substantially with depth and to a lesser degree from Lake1 to Lake 3 sites, with a maximum value of 43.5 µg/L (Table 8, Fig. 5). SRP concentrations were below detection at almost every site, depth, and sampling event until April 2022 (Table 9, Fig. 6). When SRP was measurable, it ranged ~8-20 ug/L and was largest at bottom sampling depths (Table 9, Fig. 6).

Table 8. Green Lake total phosphorus (TP) concentrations across all sites and depths from May 2021 – May 2022. Color coding is a gradient of TP concentration from lowest (3.5 µg/L, white) to highest (43.5 µg/L, green).

	Lake1			Lake2			Lake3		
Date	surface	middle	bottom	surface	middle	bottom	surface	middle	bottom
5/14/2021	3.5	12	15.9	3.5	3.5	16	3.5	16.9	19
5/25/2021	3.5	7.6	16.1	3.5	9.3	19.5	3.5	12.1	30.8
6/11/2021	3.5	3.5	3.5	3.5	3.5	3.5	3.5	3.5	6.5
6/28/2021	3.5	3.5	26.3	12.7	14.2	23.9	19.4	17.7	29
7/9/2021	10.8	18.9	31.8	12.2	27.5	27.9	13	30.3	30.7
7/27/2021	7.5	8.2	20.5	8.9	16.8	17.7	12.3	18.4	24.3
8/16/2021	11.4	17	15.4	11.6	22.8	19.6	11.7	25.9	28.1
8/30/2021	10.3	20.5	21.1	11.8	21.1	43.5	7.5	15.1	27.6
9/10/2021	11.5	17.2	22.1	17	30.8	26.9	13.4	18.6	22.7
9/27/2021	10.8	10	17.3	9.1	11.1	30	10	15.8	23.3
10/22/2021	6.9	7.9	19.9	8.1	12.6	37.5	6.8	8	28.8
11/5/2021	3.5	3.5	11	3.5	3.5	14.5	3.5	3.5	35.8
11/23/2021	8.5	17.7	8.3	10.3	10.9	14.5	8.1	3.5	3.5
12/10/2021	8.8	15	12.7	8	7.6	9.1	10.8	7.5	9.7
2/14/2022	11.7	9.2	20.6	3.5	3.5	3.5	9	6.9	23.8
4/12/2022	9.1	7.9	3.5	3.5	3.5	8.3	8.8	8.7	7.6
5/12/2022	13.3	11.8	19.6	3.5	12.3	12.7	8.8	16.3	3.5
Grand Mean	**8.1**	**11.3**	**16.8**	**7.9**	**12.6**	**19.3**	**9.0**	**13.5**	**20.9**

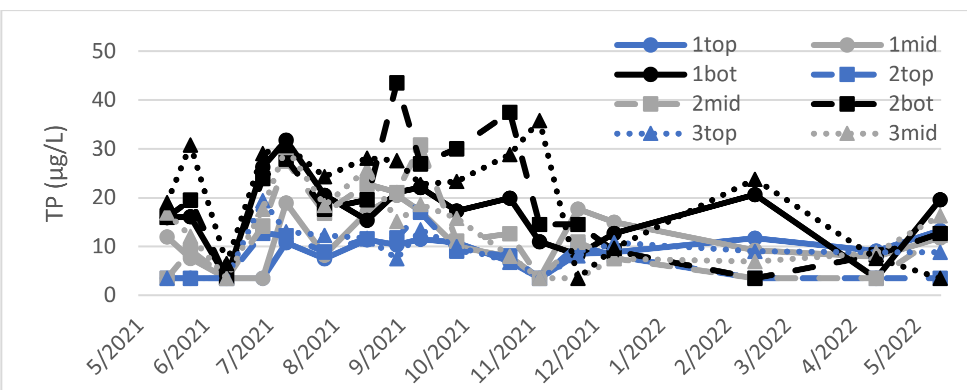


Figure 5. Green Lake total phosphorus (TP) concentrations sampled May 2021 – May 2022 at near-surface (top), middle (mid), and near-bottom (bot) depths.

Table 9. Green Lake soluble reactive phosphorus (SRP) concentrations across all sites and depths from May 2021 – May 2022. Color coding is a gradient of SRP concentration from lowest (2.5 µg/L, white) to highest (19.6 µg/L, green).

Date	Lake1			Lake2			Lake3		
	surface	middle	bottom	surface	middle	bottom	surface	middle	bottom
5/14/2021	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5
5/25/2021	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5
6/11/2021	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5
6/28/2021	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5
7/9/2021	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5
7/27/2021	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5
8/16/2021	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	13.2
8/30/2021	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	14.8
9/10/2021	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5
9/27/2021	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5
10/22/2021	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5
11/5/2021	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5
11/23/2021	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5
12/10/2021	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5
2/14/2022	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5
4/12/2022	9.1	7.9	3.5	3.5	3.5	8.3	8.8	8.7	7.6
5/12/2022	13.3	11.8	19.6	3.5	12.3	12.7	8.8	16.3	3.5
Grand Mean	**3.52**	**3.36**	**3.56**	**2.62**	**3.14**	**3.44**	**3.24**	**3.68**	**4.21**

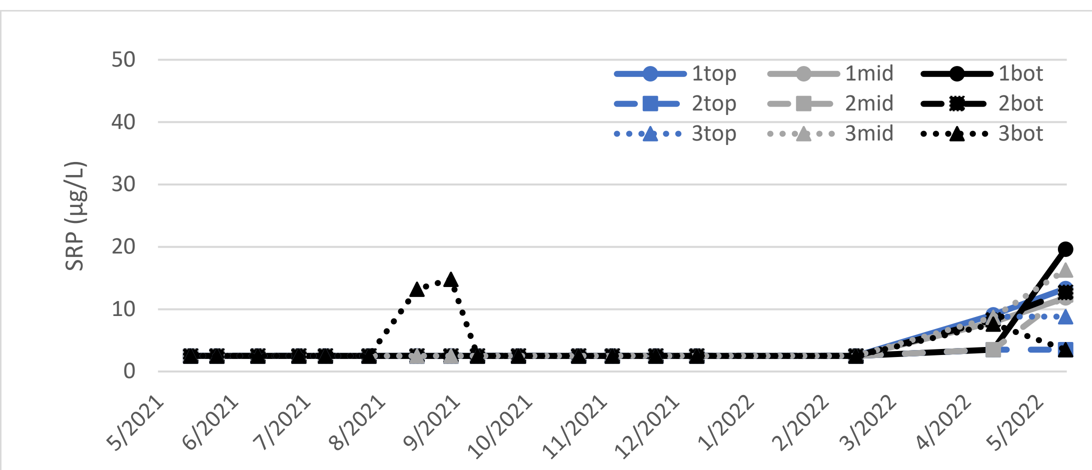


Figure 6. Green Lake soluble reactive phosphorus (SRP) concentrations sampled May 2021 – May 2022 at near-surface (top), middle (mid), and near-bottom (bot) depths.

NO_3^- concentrations were relatively low overall, never exceeding 0.5 mg/L. They were highest in late winter and spring (up to 0.46 mg/L) and lowest in fall (0.01 mg/L; Table 10, Fig. 7). NO_3^- concentrations were slightly higher at Lake sites 2 and 3 than site 1 (Table 10, Fig. 7). TKN accumulated in the bottom of the lake throughout 2021 and reached maximum values of 3.08-4.35 mg/L on November 5th, suggesting internal nitrogen loading of ammonia. Concentrations decreased two weeks later, suggesting the lake had experienced fall turnover (Table 11, Fig. 8). Sampling was less frequent in Spring 2022, but TKN accumulated to 3.1 mg/L in the narrows (Site 2) in April, while the rest of the lake was more homogenous at that time, suggesting a spring turnover event had occurred (Table 11, Fig. 8). Overall, the grand means for each season were very similar at all 3 sites (Table 11).

Table 10. Green Lake nitrate (NO_3^-) concentrations across all sites and depths from May 2021 – May 2022. Color coding is a gradient of NO_3^- concentration from lowest (0.01 mg/L, white) to highest (0.46 mg/L, green).

Date	Lake1			Lake2			Lake3		
	surface	middle	bottom	surface	middle	bottom	surface	middle	bottom
5/14/2021	0.29	0.30	0.27	0.35	0.34	0.34	0.35	0.35	0.34
7/27/2021	0.06	0.18	0.16	0.05	0.06	0.27	0.06	0.22	0.15
9/10/2021	0.04	0.12	0.20	0.11	0.15	0.07	0.07	0.13	0.06
10/22/2021	0.05	0.07	0.01	0.12	0.10	0.09	0.10	0.11	0.09
11/5/2021	0.11	0.13	0.10	0.12	0.14	0.01	0.13	0.15	0.01
11/23/2021	0.10	0.12	0.06	0.16	0.15	0.16	0.16	0.13	0.17
12/10/2021	0.09	0.11	0.12	0.16	0.15	0.19	0.18	0.14	0.20
2/14/2022	0.14	0.14	0.16	0.25	0.27	0.26	0.25	0.23	0.46
4/12/2022	0.12	0.12	0.11	0.24	0.23	0.23	0.21	0.25	0.20
5/12/2022	0.22	0.22	0.20	0.30	0.29	0.28	0.31	0.28	0.25
Grand Mean	**0.12**	**0.15**	**0.14**	**0.19**	**0.19**	**0.19**	**0.18**	**0.20**	**0.19**

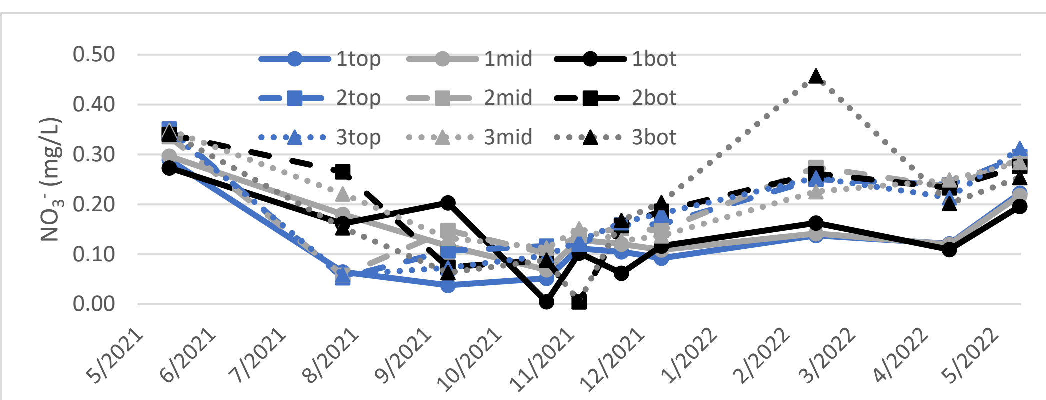


Figure 7. Green Lake nitrate (NO_3^-) concentrations sampled May 2021 – May 2022 at near-surface (top), middle (mid), and near-bottom (bot) depths.

Table 11. Green Lake total Kjeldahl nitrogen (TKN) concentrations across all sites and depths from May 2021 – May 2022. Color coding is a gradient of TKN concentration from lowest (0.38 mg/L, white) to highest (4.35 mg/L, green).

Date	Lake1			Lake2			Lake3		
	surface	middle	bottom	surface	middle	bottom	surface	middle	bottom
5/14/2021	0.49	0.63	1.14	0.56	0.54	0.91	0.48	0.48	0.62
7/27/2021	0.51	0.66	1.47	0.53	0.55	0.55	0.38	0.67	1.60
9/10/2021	0.60	0.65	1.47	0.69	0.85	1.97	0.67	0.78	2.29
10/22/2021	0.47	0.53	2.47	0.46	0.44	2.36	0.52	0.43	2.40
11/5/2021	0.66	0.52	4.35	0.69	0.60	3.08	0.57	0.64	3.88
11/23/2021	0.67	0.68	0.81	0.84	0.82	0.82	0.78	0.83	0.87
12/10/2021	0.95	0.67	0.87	0.90	0.77	0.85	0.87	0.73	0.90
2/14/2022	0.96	0.71	1.04	0.75	0.87	1.00	0.80	0.85	1.14
4/12/2022	0.63	0.77	0.64	0.61	0.61	3.10	0.55	0.47	0.52
5/12/2022	0.50	0.63	0.71	0.51	0.53	0.77	0.45	0.58	0.82
Grand Mean	**0.64**	**0.65**	**1.50**	**0.65**	**0.66**	**1.54**	**0.61**	**0.65**	**1.50**

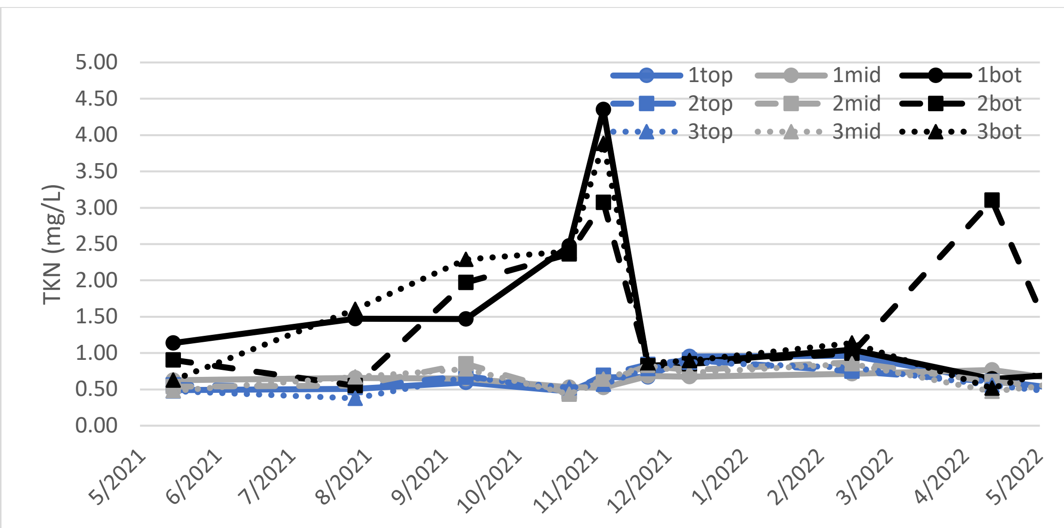


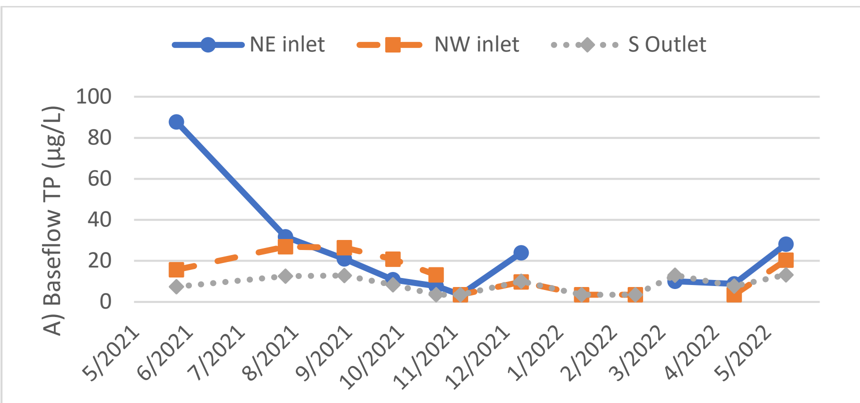
Figure 8. Green Lake total Kjeldahl nitrogen (TKN) concentrations sampled May 2021 – May 2022 at near-surface (top), middle (mid), and near-bottom (bot) depths.

Using the grand mean values for TN (combining nitrate and TKN) and TP in Green Lake, the mean TN:TP ratio by mass in the water column on the annual basis was 11.26 (TN:TP = 12.49:1.11).

Tributary Water Quality Sampling. To briefly summarize the sampling locations of the lake relative to tributaries (Fig. 1), the NE tributary is upstream of the northern lobe of the lake, the NW tributary is closer to the narrows, and the S outlet is downstream of the southern lobe of the lake.

TP concentrations at baseflow in the inflowing tributaries reached up to 88 µg/L in the NE tributary, but both streams generally ranged ~10-30 ug/L and were below detection limits in winter months (Fig. 9A). The S outlet was generally lower than the two inlet streams and reached a maximum of 13 µg/L (Fig. 9A), reflecting that Green Lake was retaining P. Storm flow concentrations of TP were highest in both inlets and reached 375 µg/L in the NE inlet (Fig. 9B). Storm flow TP in the S outlet was almost always below detection and recorded as 3.5 µg/L, again indicative of Green Lake's TP retention ability (Fig. 9B).

Baseflow SRP concentrations were below detection and reported as 2.5 µg/L for much of the sampling year at all three stream sites (Fig. 10A). When SRP was measurable during spring through fall 2021, it was highest in the NW inlet and ranged 5-13 µg/L (Fig. 10A). On these dates, SRP composed up to 33%-71% of TP (Figs. 9A, 10A). Baseflow SRP was not detectable on any sampling date at the S outlet (2.5 µg/L; Fig. 10A). Storm flow concentrations at the tributaries exceeded baseflow and ranged 12-241 µg/L, representing 20%-86% of storm flow TP (Fig. 10B). Storm SRP at the S outlet remained below detection on all sampling dates (2.5 µg/L; Figure 10B).



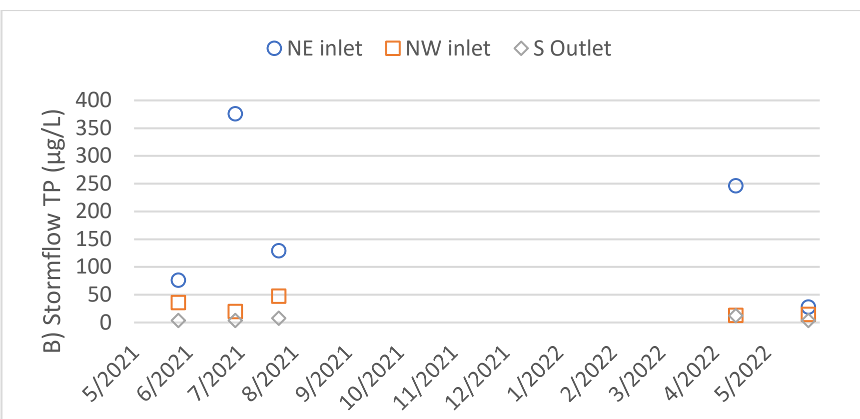
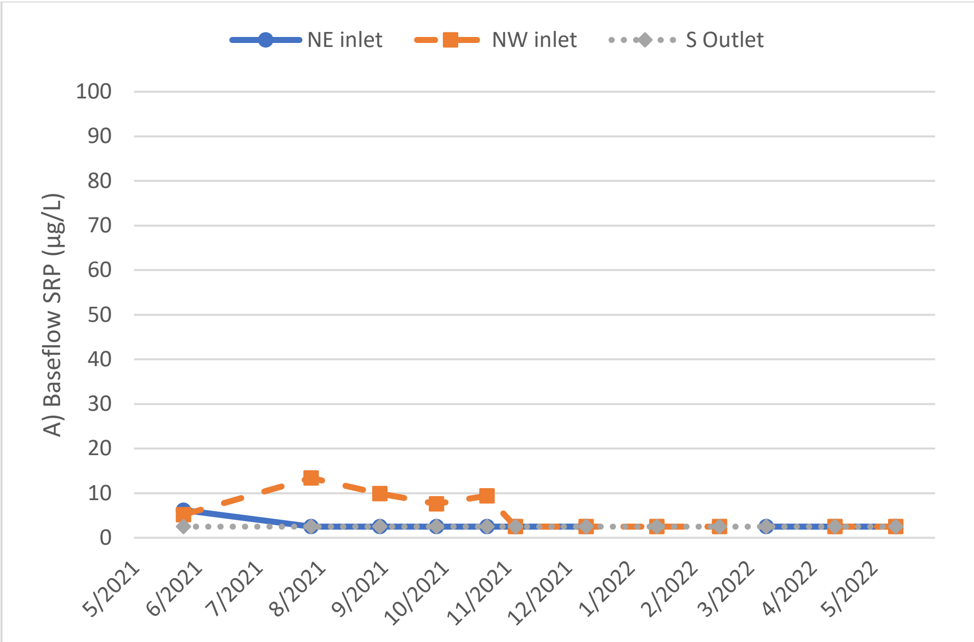


Figure 9. Green Lake tributary and outflow total phosphorus (TP) concentrations during baseflow (A) and stormflow (B). Note that y-axis scales change between panels.



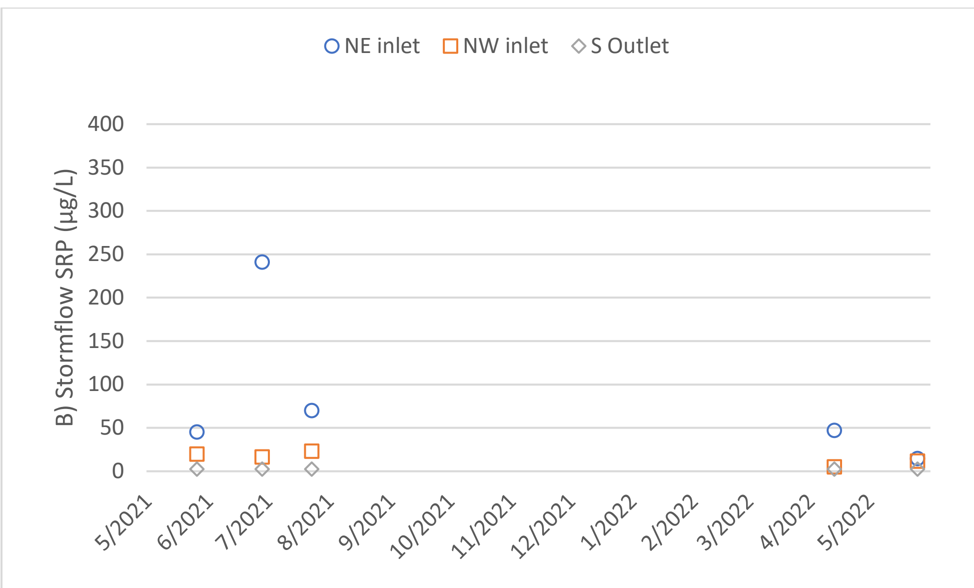


Figure 10. Green Lake tributary and outflow soluble reactive phosphorus (SRP) concentrations during baseflow (A) and stormflow (B). Note that y-axis scales change between panels.

NO_3^- and TKN were later additions to the sampling regime and collection began in October 2021. Baseflow NO_3^- concentrations ranged 0.13-1.9 mg/L and showed similar trends in both tributaries (Fig. 11A). NO_3^- concentrations in the S outlet were detected every month but remained low in the range of 0.10-0.23 mg/L (Fig. 11A). Only two storms were recorded with NO_3^- data in April and May 2022. Storm concentrations in the stream (0.11-0.84 mg/L) appeared to be similar or slightly less than baseflow concentrations while S outlet values stayed basically the same (0.12-0.18 mg/L; Fig. 11B).

Baseflow TKN concentrations were generally lower than NO_3^- concentrations in both tributaries and ranged 0.27-0.94 mg/L (Fig. 12A). TKN in the S outflow was notably higher than in the tributaries at baseflow for much of the sampling period and ranged 0.41-0.82 mg/L (Fig. 12A). Storm flow TKN ranged 0.49-0.95 mg/L in the tributaries, representing a slight (~0.04-0.1 mg/L) increase at most sites and times and a larger 0.69 mg/L increase in the NE inlet from April baseflow (Fig. 12B). Storm flow decreased TKN concentrations in the S outlet and ranged 0.45-0.58 mg/L (Fig. 12B).

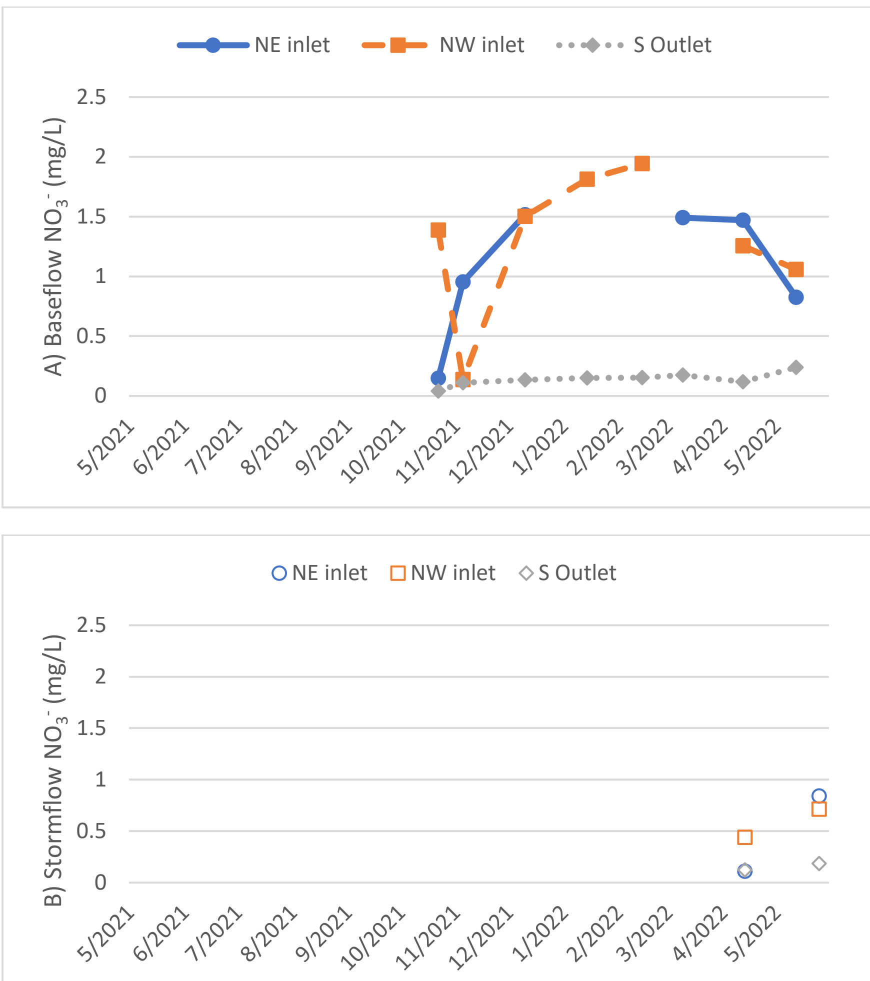


Figure 11. Green Lake tributary and outflow nitrate (NO_3^-) concentrations during baseflow (A) and stormflow (B).

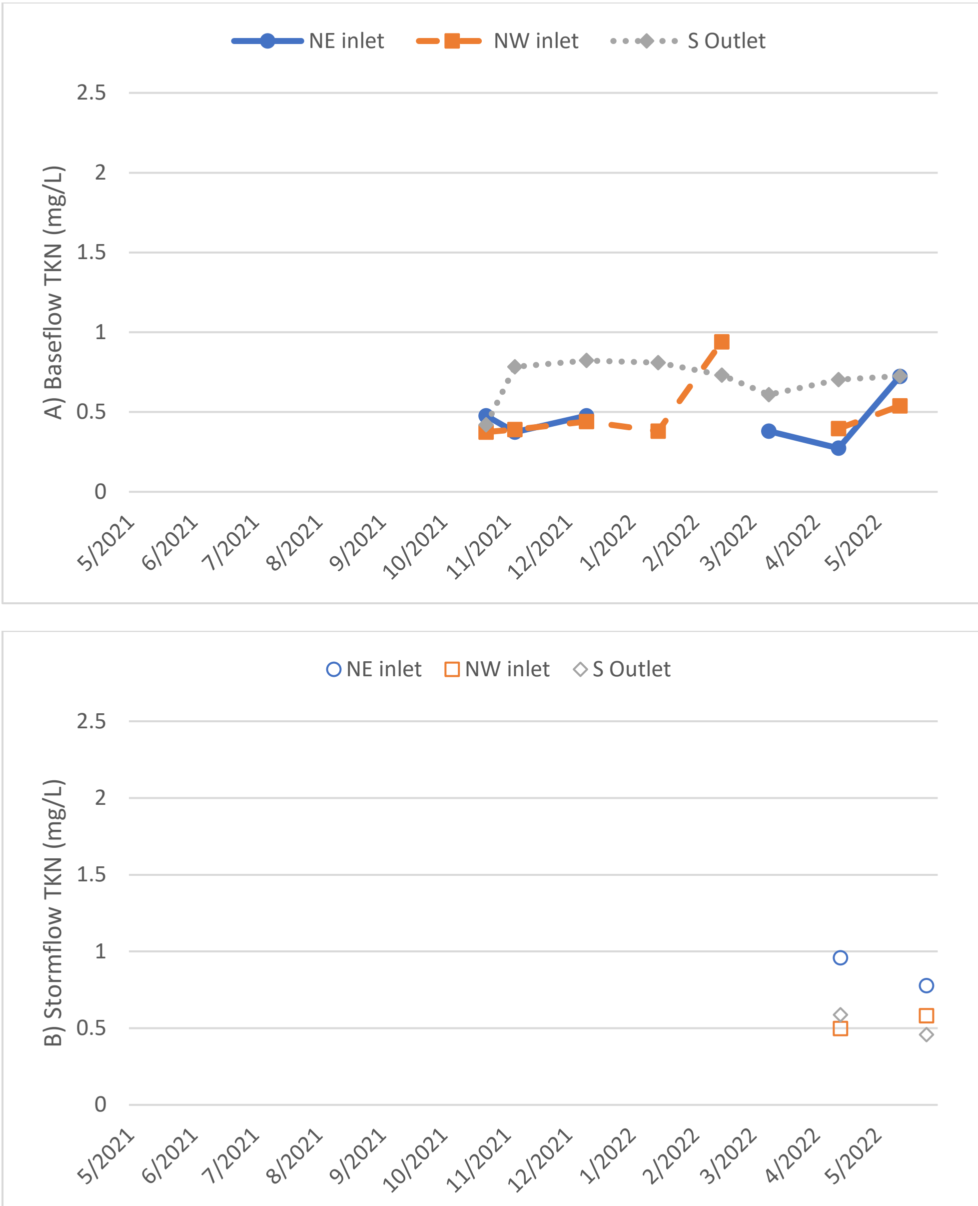


Figure 12. Green Lake tributary and outflow total Kjeldahl nitrate (TKN) concentrations during baseflow (A) and stormflow (B).

Well Sampling

To recap the locations of our 5 groundwater sampling sites via homeowner wells, three sites (#495, #511, and #4657) are located on the north shore of Green Lake near the narrows and of these sites, well #4657 is closest to the NW tributary (Figure 3). Well #4572 is on the eastern shore of the south lobe of the lake and incorporates groundwater flow from Green Lake's peninsula (Fig. 2). Well #567 is on the southern shore of Green Lake (Fig. 2). Groundwater was able to be sampled consistently throughout the warmer months but sampling opportunities became less available as homeowners traveled in winter and/or winterized their homes.

Well water TP concentrations were either below detection (3.5 µg/L) or measured at low concentrations (6.6 µg/L) for sites #495, #511, #567, and #4572 (Table 12). Site #4657 TP concentrations were higher by a degree of magnitude and ranged 22-52 µg/L (Table 12). SRP followed a similar pattern with detectable concentrations only being found at site #4657 and ranging 15-24 ug/L, representing 60%-80% of the TP on each sampling date (Table 13). This difference between sites potentially indicates either erroneous methodological assumptions regarding home and well plumbing, as the use of polyphosphates is a known method of water softening, or else could indicate a groundwater source of phosphorus somewhere south of the NW tributary that may not have been detectable through well sites #495 and #511.

Table 12. Green Lake well water TP concentrations by date. Color coding is a gradient of SRP concentration from lowest (3.5 µg/L, white) to highest (51.5 µg/L, green). ND = no data.

Date	#495	#511	#567	#4572	#4657
7/9/2021	6.7	3.5	3.5	3.5	30.3
8/30/2021	3.5	3.5	3.5	3.5	36.1
9/27/2021	3.5	3.5	3.5	3.5	26.6
10/22/2021	3.5	3.5	3.5	3.5	51.5
11/5/2021	3.5	3.5	3.5	3.5	ND
12/10/2021	ND	3.5	6.5	3.5	ND
1/14/2022	ND	3.5	ND	3.5	ND
2/14/2022	ND	3.5	3.5	ND	ND
3/9/2022	ND	3.5	ND	3.5	ND
4/12/2022	ND	3.5	ND	3.5	ND
5/12/2022	ND	3.5	3.5	3.5	26.9
6/10/2022	ND	3.5	3.5	3.5	21.8

Table 13. Green Lake well water SRP concentrations by date. Color coding is a gradient of SRP concentration from lowest (2.5 µg/L, white) to highest (33.7 µg/L, green). ND = no data.

Date	**#495**	**#511**	**#567**	**#4572**	**#4657**
7/9/2021	2.5	2.5	2.5	2.5	24.1
8/30/2021	2.5	2.5	2.5	2.5	23.3
9/27/2021	2.5	2.5	2.5	2.5	15.8
10/22/2021	2.5	2.5	2.5	2.5	33.7
11/5/2021	2.5	2.5	2.5	2.5	ND
12/10/2021	ND	2.5	2.5	2.5	ND
1/14/2022	ND	2.5	ND	2.5	ND
2/14/2022	ND	2.5	2.5	ND	ND
3/9/2022	ND	2.5	ND	2.5	ND
4/12/2022	ND	2.5	ND	2.5	ND
5/12/2022	ND	2.5	2.5	2.5	17.9
6/10/2022	ND	2.5	2.5	2.5	15.3

Detectable NO_3^- concentrations were more common among sites and consistently lowest at sites #511 and #4657, ranging 0.02-0.13 mg/L (Table 14). Site #495 and #4572 NO_3^- concentrations ranged 2-5 mg/L (Table 14). Site #567 concentrations skewed even higher and commonly ranged 5-9.7 mg/L (Table 14).

Grand means averaging TP, SRP, or NO_3^- concentrations individually for each well across the entire sampling period are provided for homeowner reference in Table 15 and Fig. 13. We note that the State of Michigan's legal limit for NO_3^- in drinking water is 10 mg/L and while that threshold has not been exceeded in this study, we encourage the homeowners of site #567 to pursue additional NO_3^- well water quality monitoring and for watershed partners to consider future groundwater monitoring around the southern lobe of Green Lake.

Table 14. Green Lake well water NO_3^- concentrations by date. Color coding is a gradient of NO_3^- concentration from lowest (0.02 mg/L, white) to highest (9.74 mg/L, green). ND = no data.

Date	**#495**	**#511**	**#567**	**#4572**	**#4657**
7/9/2021	4.70	0.06	8.47	4.82	0.08
8/30/2021	5.38	0.06	7.92	3.72	0.06
9/27/2021	ND	ND	ND	ND	ND
10/22/2021	5.00	0.12	7.61	3.02	0.13
11/5/2021	2.18	0.12	4.77	2.47	ND
12/10/2021	ND	0.13	8.73	2.25	ND
1/14/2022	ND	ND	ND	ND	ND
2/14/2022	ND	0.14	1.67	ND	ND
3/9/2022	ND	0.13	ND	1.74	ND
4/12/2022	ND	0.02	ND	1.64	ND
5/12/2022	ND	0.14	9.74	1.72	0.17
6/10/2022	ND	0.15	8.64	2.80	0.17

Table 15. Green Lake well water mean (±SD) TP, SRP, and NO_3^- concentrations.

Well	**TP (µg/L)**	**SRP (µg/L)**	**NO_3^- (mg/L)**
#495	4.1 (1.4)	2.5 (0)	4.3 (1.5)
#511	3.5 (0)	2.5 (0)	0.1 (0)
#567	3.8 (1)	2.5 (0)	7.2 (2.7)
#4572	3.5 (0)	2.5 (0)	2.7 (1.1)
#4657	32.2 (10.6)	21.7 (7)	0.1 (0)
Grand Mean	6.7 (3.4)	4.5 (2.3)	2.5 (1.2)

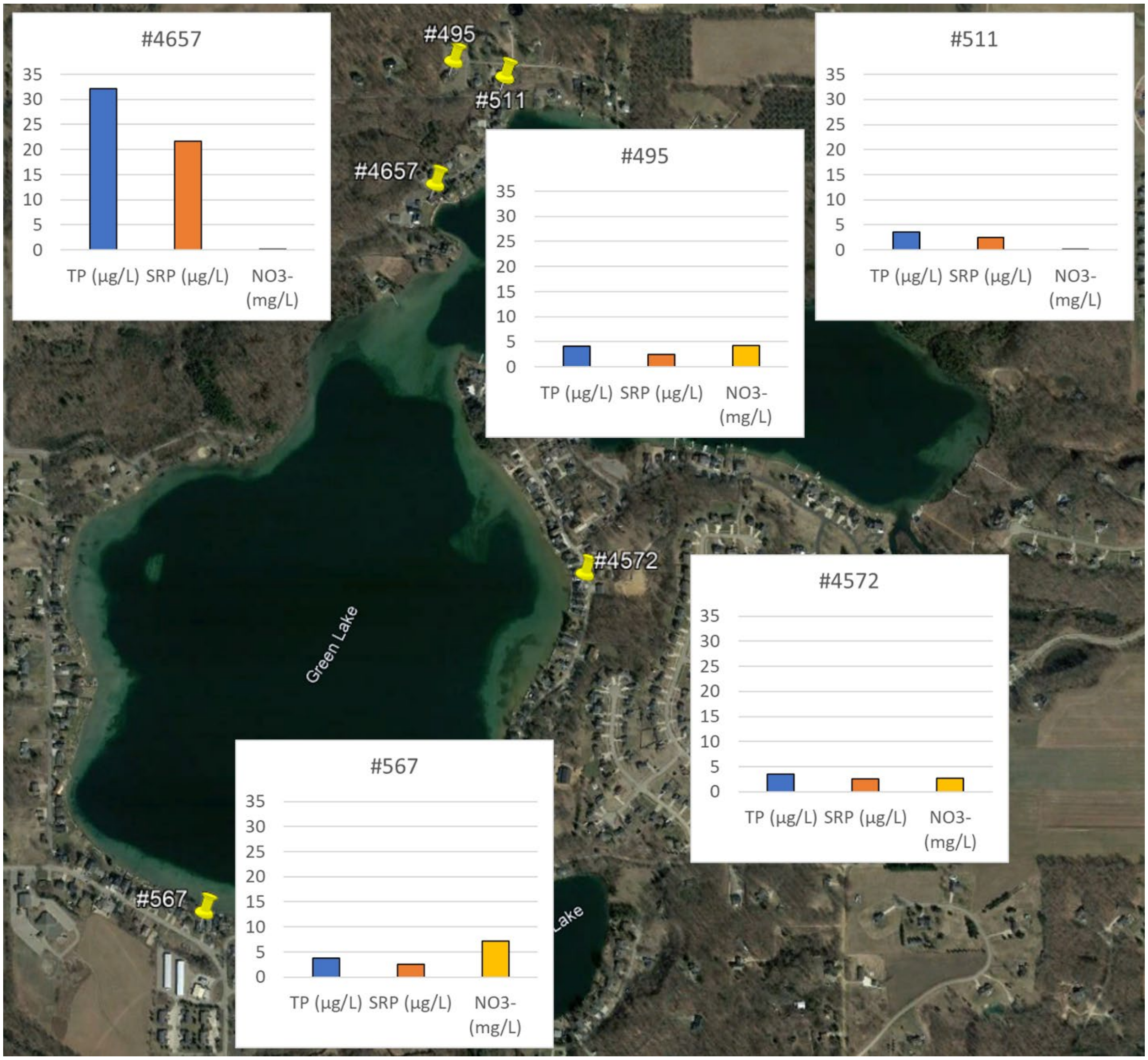


Figure 13. Green Lake well water mean nutrient concentrations relative to sampling locations.

Lake Sediment Phosphorus Fractionation

Sediment characteristics were similar at both coring sites. Both the organic matter content and the sediment TP were slightly higher in the NW core (15.3% and 454 mg/kg dry wt.) relative to the NE core (12.8% and 425 mg/kg dry wt.; Table 16).

The loosely sorbed NH_4Cl-P fraction was found in minimal quantities in both cores (1.3-4.4 µg/g; Table 16, Fig. 14). The BD-P fraction consisting of reductant-soluble P (Fe and Mn oxides and hydroxides) was 71 µg/g in both cores (Table 16, Figure 15). The NaOH-P fraction which binds to iron and aluminum was higher in the NE core (74 µg/g) than the NW core (51 µg/g; Table 16, Fig. 14). This trend reversed for the HCl-P fraction, which binds to Ca and Mg, and was higher in the NW core (118 µg/g) than the NE core (52 µg/g; Table 16, Fig. 14). In general, about two-thirds of the fractioned SRP was in the stable form (HCl-P and NaOH-P), and less likely to be released from the sediment, with the remaining one-third in the mobile form (NH_4Cl-P and BD-P).

When comparing sediment TP concentrations in Green Lake to other west Michigan lakes and wetlands examined by AWRI for prior studies, Green Lake sediment TP means rank at the lower and less eutrophic end of the spectrum (Fig. 15). Indeed, Green Lake concentrations were below several other lakes and were more similar to values measured in a freshly restored wetland along the lower Muskegon River (Fig. 15).

Table 16. Sediment phosphorus fractionation results from Green Lake cores collected downstream of the NW and NE tributary inflows. Means (±SD) are the result of n=2 pseudoreplicate subsamples per each core. Note that µg/g is an equivalent ratio to mg/kg.

Fractionation Parameters		**NW Core**		**NE Core**	
		Mean	**SD**	**Mean**	**SD**
OM%		15.3%	0.3%	12.8%	0.2%
Sediment TP (mg/kg, dry wt.)		454	43	425	28
P fractions as SRP-P (µg/g, dry wt.)	NH_4Cl	1.27	0.04	4.44	0.34
	BD	71.7	0.2	71.3	11.5
	NaOH	51.3	1.7	74.0	0.9
	HCl	118.4	17.7	52.3	1.5

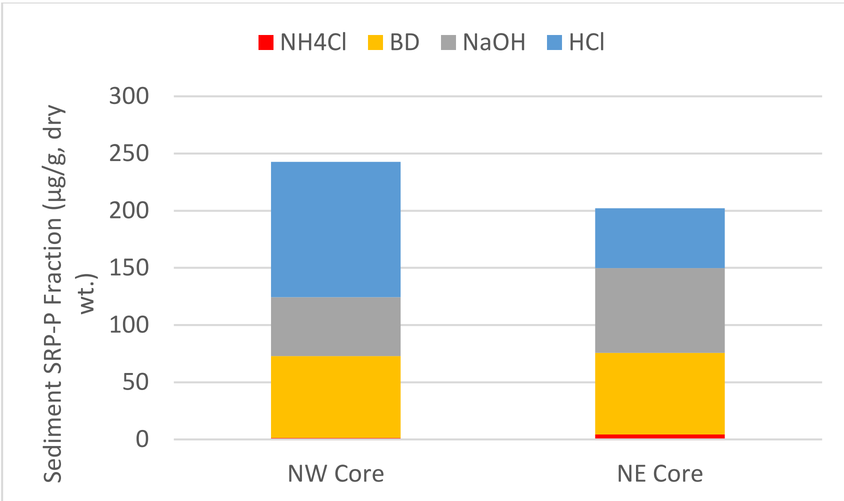


Figure 14. Sediment phosphorus fractionation results from Green Lake cores collected downstream of the NW and NE tributary inflows. Note that the NH_4Cl-P fraction is present in the NW core, but in a low quantity that may be difficult to view in this format (~1.3 µg/L).

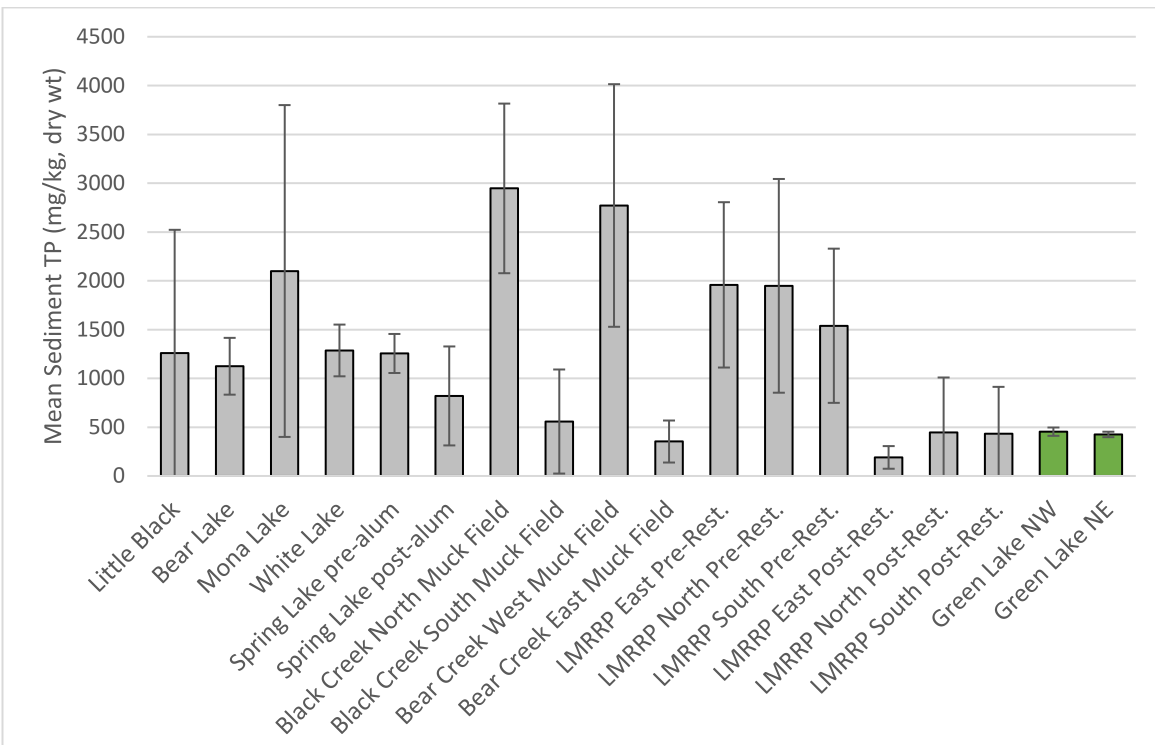


Figure 15. Comparison of mean (±SD) sediment TP measured in Green Lake and other west Michigan waterbodies. Sources: Little Black Lake: Steinman et al. 2011; Mona Lake: Steinman et al. 2009; White Lake: Steinman et al. 2008; Spring Lake pre-alum: Steinman et al. 2004; Spring Lake post-alum: Steinman and Ogdahl 2008; Bear Lake: unpublished data; Black Creek muck fields: Steinman and Ogdahl 2011; Bear Creek: Steinman and Ogdahl 2013; LMRRP pre-restoration: Steinman et al. 2017b; LMRRP post-restoration: unpublished data.

E. coli and Microbial Source Tracking

Of the two tributary sampling sites, *E. coli* was detected in lower quantities at the NW inlet (baseflow = 118 cfu/100 mL; storm flow = 180 cfu/100 mL) and did not exceed the 300 cfu/100 mL threshold to warrant further analysis (Table 17). Concentrations at the NE inlet were 548 cfu/100 mL during baseflow and 408 cfu/100 mL during storm flow (Table 17). Ruminant-sourced *E. coli* was detected during both base and storm flow regimes (424 and 584 cfu/100 mL, respectively) and human-sourced *E. coli* was additionally detected during storm flow only (408 cfu/100 mL; Table 17).

Table 17. NE inlet water sampling for *E. coli* concentrations and microbial source tracking. All samples and controls above the minimum detection limits (360 GC/100 mL) are indicated with bold text and microbial sources of *E. coli* above minimum detection are highlighted in red bold text.

	5/12/2022 baseflow	5/25/2022 storm
E. coli (cfu/100 mL)	**548**	**488**
Human - HF183 (GC/100 mL)	360	**408**
Dog - DG3 (GC/100 mL)	360	360
Ruminant - Rum2Bac (GC/100 mL)	**424**	**584**
Porcine - Pig2Bac (GC/100 mL)	360	360
Sample Processing Control - Sketa (GC/100 mL)	**784**	**1000**

Hydrologic Characteristics

Triangulated Irregular Network (TIN) interpolation reported an estimated total lake area of 42,224,636.7 ft^3, or 969 ac ft (Fig. 16). This is a conservative estimate of lake volume because the state's GIS shapefiles for bathymetry do not include (1) points for the deepest points within each subbasin, and (2) the excavated channel extending southeast from the northeastern subbasin of the lake to otherwise landlocked properties on Cove Drive, Green Ridge Drive, and Nuthatch Court SE. Each of these points would add volume for estimating the true volume of Green Lake; however, they would likely contribute relatively small additional volumes compared to this first-order approximation.

Precipitation varied seasonally across the measurement period, generally ranging 1.3 - 10 inches per month (Table 18), and was within the range of precipitation observed over the past 15 years (Fig. 17). Across the 2478.81-acre watershed drainage area, these precipitation totals are equivalent to monthly volumes of 264 - 2127 ac ft (Table 18). Relative to the estimated lake volume of 969 ac ft, these precipitation volumes represent 33 - 219% of total Green Lake volume (Table 18). Although some land covers and land uses throughout the watershed would intercept water and prevent it from entering the lake itself (Appendix Table 1), estimates show that enough precipitation fell in the watershed during this measurement period to fill Green Lake many times over.

We applied our hydraulic residence time (HRT) calculation to summer (May - September) and winter seasons (October - April) in addition to an annual HRT summary (Table 19). As one might expect, the HRT was somewhat shorter during the summer season when precipitation is greater; in contrast, HRT was shorter during the winter season compared to summer (Table 19). This estimated difference between seasons may have been larger if monthly baseflow data was incorporated into this calculation instead of applying an annual average. Overall, the HRT of Green Lake (both seasonally and annually) is on the scale of days, rather than months or years.

TP load calculations revealed several trends. First, baseflow loads varied seasonally in inflowing tributaries, being higher in summer (145-260 kg) and lower in winter (60-95 kg; Table 20). TP loads in waters leaving Green Lake at the S outflow were more consistent year-round (237-270 kg) than the tributaries (Table 20). Storm flow TP loads generally increased across the board at all sites and times compared to baseflow, except for the S outflow which instead decreased during the summer (Table 20). Finally, the NW tributary has a higher TP load during baseflow than the NE tributary in both seasons but this trend reverses during storm events (Table 20).

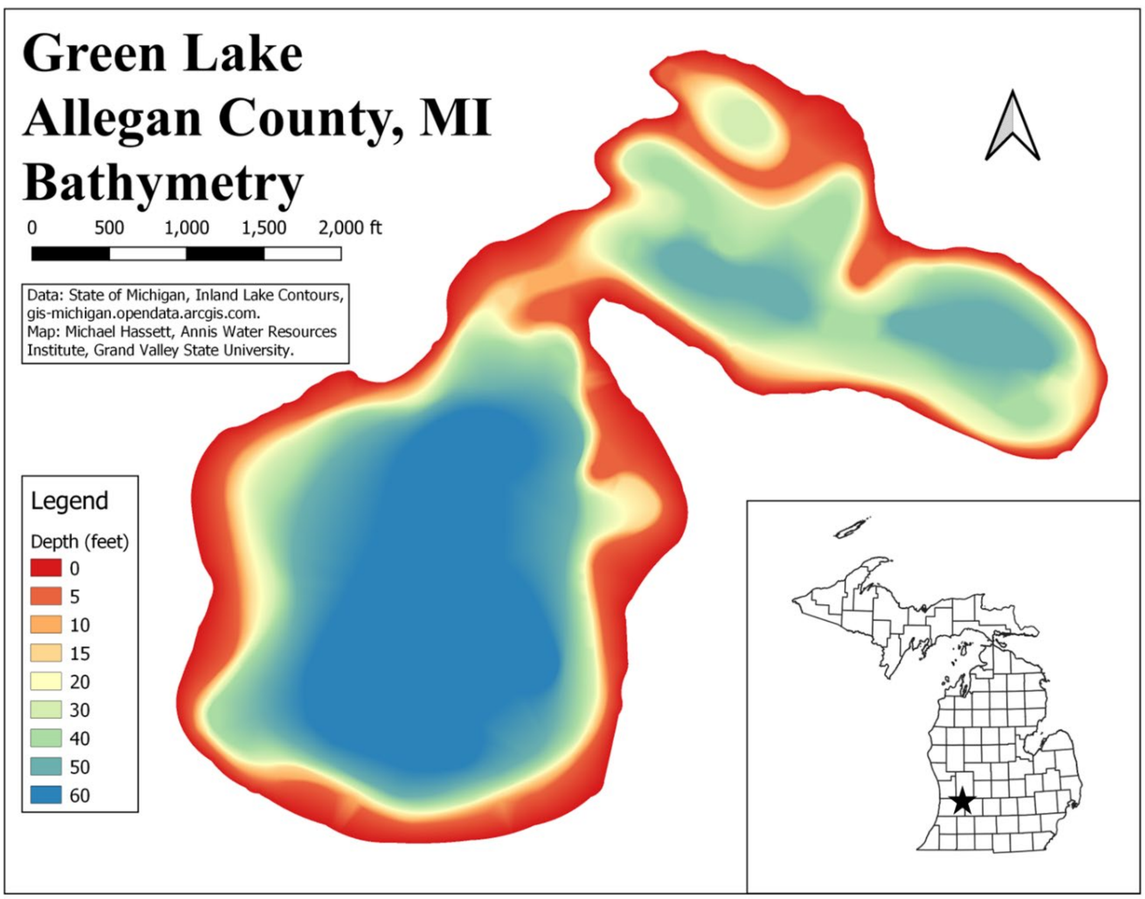


Figure 16. Green Lake bathymetry interpolated via Triangulated Irregular Network (TIN) analysis. Calculated total lake volume was 969 ac ft. Green Lake's location within Allegan County is indicated with a star on the inset mini-map.

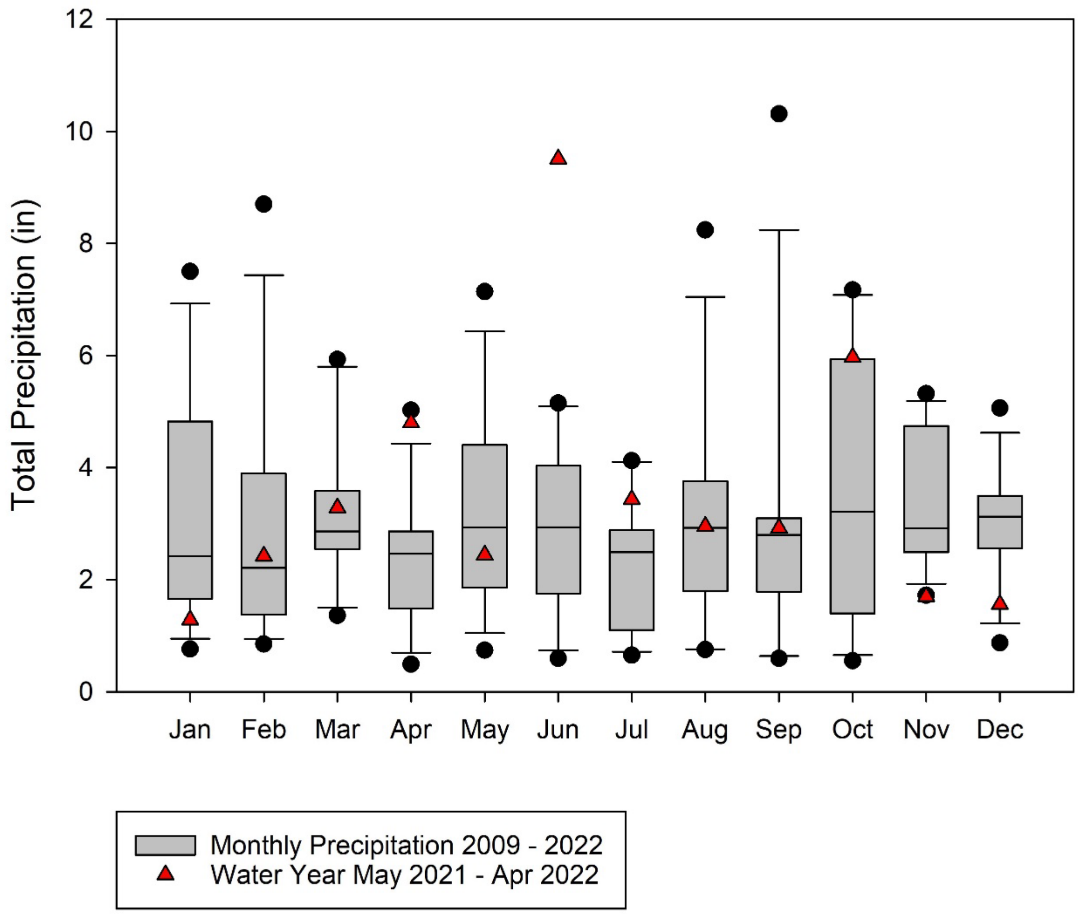


Figure 17. Distributions of Green Lake monthly precipitation totals from July 2008 – November 2022. Boxes represent upper 25%, median 50%, and lower 50% of values, whisker represent upper and lower extreme values, and points beyond whiskers are estimated to be outliers (i.e. >1.5x of the value of the upper quartile, or less than the lower quartile).

Table 18. Monthly total precipitation from January 2021 – October 2022 summed as inches, calculated as volumes, and as a percent of the TIN-derived Green Lake volume.

	Monthly Precipitation (in)	Monthly Precipitation (ac ft)	% Total Lake Volume
2021	**38.9**	**8025**	**828%**
Jan	1.4	290	30%
Feb	1.5	309	32%
Mar	1.8	369	38%
Apr	1.6	332	34%
May	2.4	503	52%
Jun	10.3	2127	219%
Jul	3.4	709	73%
Aug	3.0	617	64%
Sep	2.9	602	62%
Oct	6.0	1236	128%
Nov	2.3	471	49%
Dec	2.2	462	48%
2022	**34.0**	**7018**	**724%**
Jan	1.3	264	27%
Feb	4.0	829	86%
Mar	3.5	723	75%
Apr	5.0	1023	106%
May	5.0	1030	106%
Jun	1.9	387	40%
Jul	3.8	791	82%
Aug	4.0	827	85%
Sep	1.6	324	33%
Oct	4.0	819	85%
Grand Total	**72.8**	**15043**	**1552%**

Table 19. Calculation of Green Lake seasonal and annual hydraulic retention time (HRT). Discharge (Q) is the sum of precipitation and average seasonal or annual baseflow in acre-feet volumes as described in each HRT calculation approach. Seasonal HRT calculations incorporate the total Q from all months within each seasonal boundary; HRT (months) multiply that result based on the number of months per season (summer n=5; winter n=7); HRT (days) multiply that result based on a 30-day month.

Season	Parameter	Value
Summer (May-Sep)	Q (af/season)	26,020
	HRT (season)	0.04
	HRT (months)	0.19
	HRT (days)	5.59
Winter (Oct-Apr)	Q (af/season)	35,057
	HRT (season)	0.04
	HRT (months)	0.26
	HRT (days)	7.82
Annual (12 months)	Q (af/season)	51,516
	Mean HRT (year)	0.02
	Mean HRT (months)	0.19
	Mean HRT (days)	5.83

Table 20. Budget of seasonal and annual mean inflowing (tributary) and outflowing TP loading rates.

	Base TP Load (kg)			
	NW Inflow	NE Inflow	NW+NE	S Outflow
summer (May-Sept)	260	145	405	270
winter (Oct-Apr)	95	60	155	237
Annual	**355**	**205**	**560**	**507**
	Storm TP Load (kg)			
	NW Inflow	NE Inflow	NW+NE	S Outflow
summer (May-Sept)	399	665	1064	128
winter (Oct-Apr)	205	1184	1389	452
Annual	**604**	**1849**	**2453**	**580**
	Base + Storm TP Load (kg)			
	NW Inflow	NE Inflow	NW+NE	S Outflow
summer (May-Sept)	659	810	1469	398
winter (Oct-Apr)	300	1245	1544	689
Annual	**959**	**2055**	**3014**	**1087**

Phosphorus Source Budget

To estimate internal phosphorus loading (IPL) in Green Lake, we used equations developed by Nürnberg (1988):

$$TP\ release\ rate\ \frac{mg}{m^2 * d} = -5.59 + 0.079 * \left(sediment\ TP\ \frac{\mu g}{g}, wet\ weight\right)$$

$$BD - P\ release\ rate\ \frac{mg}{m^2 * d} = -1.38 + 0.285 * \left(sediment\ BD - P\ \frac{\mu g}{g}, wet\ weight\right)$$

These are based on the P content from the top 10 cm of wet sediment. However, they are based on only 7 lakes, five from Ontario and two from Connecticut.

Our estimated release rates (Table 21) appear high given the relatively low TP concentrations measured in the hypolimnion of Green Lake. These release rates would be categorized as eutrophic to hypereutrophic in Nurnberg's classification scheme, which is clearly not the case. Hence, the IPL budget is likely an overestimate relative to the external P budget. These values can be converted to tons per year (Table 21) with the following assumptions: the lake has a sediment surface area of ~1.25 km^2 (~309 acres), the entire sediment surface releases P at the same rate, and using DO values provided by sensors deployed by MDNR in the south lobe of the lake (Appendix Fig. 1) and assuming release occurs only when DO is less than 2 mg/L (~150 d/yr). Estimated IPL ranges from 0.4 to 0.65 metric tons/yr (Table 21) depending on which site and which P fraction is used. This compares to an observed external load of ~3.01 tons/yr.

Table 21. Estimated internal phosphorus release rates ($mg/m^2/d$) based on Nürnberg (1988) and values converted to tons/yr, with external load based on observed data (tons/yr).

		Release Rates		
Fraction	Site	($mg/m^2/d$)	MT/yr	External Load (MT/yr)
TP	NW Site	2.94	0.55	3.01
	NE Site	2.15	0.40	
BD-P (SRP)	NW Site	3.46	0.65	
	NE Site	3.29	0.62	

To estimate the relative importance of IPL to external loading, we use the BD-P fraction, as that represents the more mobile form of P. Averaging the two sites results in an estimated annual internal phosphorus load of 0.635 MT/yr to Green Lake, compared to an observed external load of 3.01 MT/yr, resulting in internal P loading accounting for 21% of the total P load. This is very likely an overestimate given that 1) it is unlikely that the entire sediment surface in Green Lake will release P since the shallower areas do not go hypoxic/anoxic (but without bathymetry data, we could not determine what percent of surface area to exclude); and 2) our external load is based only on the two tributaries, whose sub-basin area covers about half of the entire watershed; any inputs from groundwater or surface runoff from around the lake could not be estimated.

Using the same approach for the calculation of external P loading to Green Lake, we calculated the external N load (Table 22). The inflow TN load is slightly greater than the outflow but given the uncertainties in our measurements, they are fairly equivalent. Hence, unlike P, Green Lake does not appear to retain N overall in its system, although there may be differential retention and release among the different N species (e.g., nitrate vs. ammonia).

We could not calculate internal N loading, so there is no comparison of internal vs. external loads in Green Lake. However, the lake appears to be in a quasi-equilibrium state with respect to inflow vs. outflow of total nitrogen.

Table 22. Budget of seasonal and annual mean inflowing (tributary) and outflowing TP loading rates.

	Base NO_3^-+TKN Load (MT)			
	NW Inflow	NE Inflow	NW+NE	S Outflow
summer (May-Sept)	18.7	5.3	24.0	25.5
winter (Oct-Apr)	31.0	7.8	38.8	30.5
annual	**52.1**	**13.4**	**65.5**	**53.3**
	Storm NO_3^-+TKN Load (MT)			
	NW Inflow	NE Inflow	NW+NE	S Outflow
summer (May-Sept)	15.2	5.6	20.7	17.0
winter (Oct-Apr)	15.4	5.2	20.6	26.4
annual	**31.4**	**11.1**	**42.5**	**43.0**
	Base+Storm NO_3^-+TKN Load (MT)			
	NW Inflow	NE Inflow	NW+NE	S Outflow
summer (May-Sept)	33.9	10.9	44.8	42.5
winter (Oct-Apr)	46.4	13.0	59.4	56.9
annual	**83.5**	**24.5**	**108.0**	**96.4**

Discussion

We conducted a comprehensive survey of Green Lake's water quality, with a special emphasis on its phosphorus and nitrogen sources and loads, to assess if the lake is under current or imminent stress. Lakes can serve as sentinels of stress given their depressed location on the landscape, where potential threats move downhill or downstream and accumulate in lake systems (Williamson et al. 2008).

Lake Status

In the case of Green Lake, the overall water quality is good, as evidenced by the high water clarity, relatively high dissolved oxygen concentrations except at the bottom during warmer months, and low to moderate phosphorus and nitrogen concentrations in the water column. Chlorophyll *a* concentration, which serves as a proxy for algal biomass, was low to moderate. We did not examine phytoplankton community structure or cyanotoxin concentrations to determine if there are potentially harmful algal species present, but this is something the lake association may want to consider in the future.

Cisco require cold, well-oxygenated water to survive and reproduce (Frey 1955; Jacobson et al. 2008); they are susceptible to temperature and oxygen stress, which makes cisco a good indicator of reductions in cold-water habitat under climate warming and eutrophication (Jacobson et al. 2008). Indeed, increased nutrients can lead to greater primary production, and the subsequent mineralization of the increased organic matter (in the form of settling phytoplankton) leads to greater respiration and DO depletion (Magee et al. 2019). Another potential driver of increased respiration is dissolved organic matter (DOM), which in sufficient quantity can enhance lake stratification due to its absorption of light in surface waters, which increases epilimnetic temperatures. DOM also serves as a substrate for microbial respiration, which can deplete DO (Read and Rose 2013). However, the high water clarity in Green Lake suggests DOM loading is not currently an issue, although may be a future consideration.

The DO data from this study are fairly typical for deep lakes in west Michigan. DO concentrations in Green Lake become problematic for cisco below ~13.5 m in July, when they drop below 5 mg/L. Shortly thereafter, DO concentrations drop to near zero into October at all depths greater than 13.5 m, suggesting cisco move to shallower areas, and possibly suboptimal habitat (cf. Spoor 1990). Lacking data from prior years, it is impossible to know if DO conditions have remained the same or have deteriorated over time; longitudinal studies are recommended to examine longer-term DO trends in the lake.

The absolute concentrations of N and P in Green Lake were modest. Although TP concentrations increased at the bottom depths in summer at sites 2 and 3, the elevated levels of 30 to 40 µg/L are still low relative to other west Michigan lakes that we have surveyed at similar depths during the same time of year, where concentrations can often be close to 1 mg/L (1,000 µg/L) (Steinman et al. 2004, 2009). The increase in TKN bottom depth concentrations to 3-4 mg/L in late fall are noteworthy. This may be related to ammonia release from the sediments during low DO conditions (Yang et al. 2020). Another possible mechanism for the TKN increase may be nitrate reduction activity; the tributary nitrate concentrations in fall were relatively high (~1.5 mg/L) yet the nitrate concentrations leaving Green Lake were very low (0.1-0.2 mg/L). Nitrate-reducing bacteria may be converting this incoming nitrate to ammonia before it leaves the lake.

The TN:TP mass ratio, when averaged on an annual basis, was 11.26. This ratio is often used as an indicator to assess which nutrient is limiting algal growth, that in turn provides lake managers with important information as to which nutrient is most critical to control. Bergström (2010) suggested that TN:TP is not as good a predictor of nutrient limitation as the ratio of dissolved inorganic nitrogen (i.e., nitrite, nitrate, and ammonia) and TP, but others have used TN:TP effectively. Downing and McCauley

(1992) reported that phytoplankton in lakes are significantly more frequently N than P limited when the TN:TP mass ratio is below 14. However, many of the lakes they considered had TP concentrations exceeding 30 µg/L, which was rarely the case in Green Lake. Others have used TN:TP ratios of 9 (Guildford and Hecky 2000) and 19 (Bergström 2010) as proxies for identifying N-limited conditions for phytoplankton. Given that Green Lake's TN:TP ratio falls within previously established ratios for N vs. P limitation, it is likely that both N and P co-limit algal growth, which is a common situation for lakes throughout the world (Bratt et al. 2020). What this means for Green Lake is that to maintain its high water quality, both nitrogen and phosphorus must be controlled in the future. Bioassays, whereby water samples are spiked with varying concentrations of N and P to determine which nutrient, if any, limit algal growth would provide a greater understanding of limiting nutrients in Green Lake.

Tributary Status

Although the lake is in reasonably good condition, one of the study's goals was to assess potential trouble areas so proactive management decisions can be made before water quality problems manifest themselves. The tributary monitoring data revealed relatively modest P concentrations coming into Green Lake from the north, with one exception in May 2021. Episodic events, coincident with storms, often can account for the majority of annual loads to a lake; in this case, the TP concentration spiked to ~375 µg/L with a concomitant SRP concentration of ~250 µg/L. We did not conduct molecular source tracking on that sampling date but we did the following May, when a separate storm event resulted in a much smaller spike of P, and the sources were identified primarily as ruminants and secondarily as humans. Additional field investigations may be able to pinpoint the farm(s) and septic system(s) responsible for this P source.

Both TP and SRP concentrations entering Green Lake were either marginally greater (baseflow) or substantially higher (stormflow) than the P concentrations leaving the lake. This indicates that P entering Green Lake is being retained in the system, either via biotic uptake (phytoplankton and/or aquatic vegetation) or adsorption to other substrates. The ultimate fate of this phosphorus cannot be stated with certainty, but it is likely that it finds its way to the sediments at some point. This sediment P can eventually be a source of internal phosphorus loading (see below).

Inflow nitrate concentrations were relatively high during baseflow (plateauing at ~ 2 mg/L) but lower during storm events, suggesting dilution from the precipitation. As noted above, the nitrate concentrations leaving Green Lake are much lower, presumably due to a combination of biotic uptake, denitrification, and/or dissimilatory reduction.

Well Status

We did not measure groundwater directly due to resource limitations. In theory, sampling the home spigots that draw from well water should provide us with a proxy of groundwater concentrations. Overall, P concentrations were relatively modest with one exception, but even site #4657 did not have unusually high concentrations. As a consequence, we do not believe that groundwater is a major source of P to Green Lake. The nitrate concentrations, on the other hand, were highly variable among the 5 wells, with sites #495 and #567, in particular, reaching levels of concern. Given this site's location on the south end of Green Lake, the high nitrate concentrations may be associated with failing septic systems. If there is not a septic management plan in place for Green Lake, with regular maintenance, we strongly encourage one to be implemented as soon as possible.

Internal vs External Phosphorus Loads

We acknowledge that there is uncertainty in the phosphorus load calculations, as described in the results section. Nonetheless, we believe they provide a useful first-order approximation of loads into Green Lake given the resource limitations associated with the project.

The annual external P load estimate of 3.01 MT is very likely an underestimate given that we did not capture all storm events and the tributary loading captures only about half of the total watershed, so surface runoff from the remaining area, plus groundwater inputs, are not included. A more robust budget would require autosamplers on all inflows and outflows, nested wells to address groundwater concentrations and flows more rigorously, and better estimates of surface runoff from the watershed not captured by the tributary flow; the resources required to generate these data were far beyond the project's budget. However, our combination of monthly baseflow and episodic storm event sampling provides a first order estimate of external loading.

Conversely, our annual internal P load estimate of ~0.5 MT is almost certainly an overestimate, given that we assumed the entire lake surface area was capable of releasing P and our estimates are based on previously established equations using sediment P content from a limited number of lakes outside our region. However, absent sediment core incubations or more comprehensive data to calculate hypolimnetic phosphorus accumulation, it was the only remaining way to estimate internal loading. There were clear issues with this approach, as the calculated P release rates were much higher than what we would expect given the relatively low P concentrations in the hypolimnion. In addition, our lack of detailed bathymetric data in Green Lake to assess what percentage of lake surface area remains oxic throughout the year, precluded us from normalizing P release by the portion of lake surface area that was susceptible to internal loading. Indeed, estimated annual internal P loadings from Mona Lake in west Michigan, with known internal P release problems and a surface area twice the size of Green Lake, were estimated at only 0.1 to 0.5 MT/yr, confirming that our internal P loading estimates for Green Lake are likely too high.

Conclusions and Recommendations

Green Lake water quality is in reasonably good condition, especially considering the high density of housing on its southern shoreline. Based on our analyses, there are no identifiable "hot spots" that require immediate attention to minimize eutrophication or hypoxia.

Nonetheless, our analyses did identify several issues where additional information may fill knowledge gaps that could help inform future management decisions. As resources become available, we recommend the following studies be considered:

1) Determine whether nitrogen or phosphorus is limiting algal growth: conduct short-term bioassay experiments that add nutrients in enclosed containers, which are submerged in Green Lake and evaluate algal growth (cf. Biddanda et al. 2008)
2) Analyze the algal community structure and cyanotoxin concentrations in Green Lake to determine if there are possible algal toxin producers
3) Measure internal nutrient loading (N and P): conduct sediment core incubations under controlled conditions in the laboratory under oxic and anoxic conditions to directly measure sediment release of N and P (cf. Steinman et al. 2009)

4) Examine in more detail the high nitrate concentrations in the well water at the south end of Green Lake: conduct additional sampling of well water and groundwater (install piezometers) to determine if this is a localized issue or more pervasive (cf. Brennan et al. 2016)

While none of these studies need to be conducted imminently, we recommend that they be conducted sooner rather than later. The information generated will allow for a more proactive stance in protecting the health and integrity of Green Lake.

Acknowledgements

Funding for this project was provided through the Allegan Conservation District by a grant from the Michigan Department of Natural Resources – Fisheries Habitat Grant Program.

We would like to thank Brian Talsma and the technicians and volunteers of the Allegan Conservation District for their work in their help with planning and conducting water quality sampling. Thank you to Brian Scull at AWRI for analyzing water and soil chemistry. Additional thanks to Maggie Oudsema, Emily Kindervater, Travis Ellens, Ellen Foley, Paris Velasquez, Kate Lucas, and Jacquie Molloseau in the Steinman Lab at AWRI for their assistance in the lab and field. Finally, we thank Kurt Thompson (deceased) for sharing his knowledge and expertise during the planning phases of this project and in GIS analysis.

References

Allan, J.D., P.B. McIntyre, S.D.P. Smith, B.S. Halpern, G. Boyer, A. Buchsbaum, A. Burton, L. Campbell, L. Chadderton, J. Ciborowski, P. Doran, T. Eder, D.M. Infante, L.B. Johnson, C.G. Joseph, A.L. Marino, A. Prusevich, J. Read, J. Rose, E. Rutherford, S. Sowa, and A.D. Steinman. 2013. Joint analysis of stressors and ecosystems services to enhance restoration effectiveness. Proceedings of the National Academy of Sciences 110 (1): 372-377. doi:10.1073/pnas.1213841110.

Bergström, A.K. 2010. The use of TN: TP and DIN: TP ratios as indicators for phytoplankton nutrient limitation in oligotrophic lakes affected by N deposition. Aquatic Sciences 72(3): 277-281.

Biddanda, B., Steinman, A., Nemeth, L., Rediske, R., Hong, Y. and Kendall, S. 2008. Nutrient bioassays of plankton biomass and metabolism in an urbanized drowned river-mouth lake (Mona Lake, Michigan). Journal of Freshwater Ecology 23: 41-53.

Bratt, A.R., Finlay, J.C., Welter, J.R., Vculek, B.A. and Van Allen, R.E. 2020. Co-limitation by N and P characterizes phytoplankton communities across nutrient availability and land use. Ecosystems 23(6): 1121-1137.

Brennan, A.K., Hoard, C.J., Duris, J.W., Ogdahl, M.E., and Steinman, A.D. 2016. Water quality and hydrology of Silver Lake, Oceana County, Michigan, with emphasis on lake response to nutrient loading, 2012–14. U.S. Geological Survey Scientific Investigations Report 2015–5158, 75 p. http://dx.doi.org/10.3133/sir20155158.

Conley, D.J., Paerl, H.W., Howarth, R.W., Boesch, D.F., Seitzinger, S.P., Havens, K.E., Lancelot, C. and Likens, G.E. 2009. Controlling eutrophication: nitrogen and phosphorus. Science 323(5917): 1014-1015.

Davis, W.P., and A.D. Steinman. 1998. A lightweight, inexpensive benthic core sampler for use in shallow water. J. Freshwat. Ecol. 13: 475-479.

Downing, J.A. and McCauley, E. 1992. The nitrogen: phosphorus relationship in lakes. Limnology and Oceanography 37(5): 936-945.

Fisher, M.M., M. Brenner, and K.R. Reddy. 1992. A simple, inexpensive piston corer for collecting undisturbed sediment/water interface profiles. J. Paleolimnol. 7: 157–161.

Flood, M.T., Hernandez-Suarez, J.S., Nejadhashemi, A.P., Martin, S.L., Hyndman, D. and Rose, J.B. 2022. Connecting microbial, nutrient, physiochemical, and land use variables for the evaluation of water quality within mixed use watersheds. Water Research 219, p.118526.

Frey DG. 1955. Distributional ecology of the cisco (*Coregonus artedii*) in Indiana. Invest Indiana Lakes Streams 4: 177–227.

Guildford, S.J. and Hecky, R.E. 2000. Total nitrogen, total phosphorus, and nutrient limitation in lakes and oceans: Is there a common relationship?. Limnology and Oceanography 45(6): 1213-1223.

Hanley, N. and Roberts, M. 2019. The economic benefits of invasive species management. People and Nature 1(2): 124-137.

Ho, J.C., Michalak, A.M. and Pahlevan, N. 2019. Widespread global increase in intense lake phytoplankton blooms since the 1980s. Nature 574(7780): 667-670.

Hupfer, M., D. Zak, R. Roβberg, C. Herzog, and R. Pöthig. 2009. Evaluation of a well-established sequential phosphorus fractionation technique for use in calcite-rich lake sediments: identification and prevention of artifacts due to apatite formation. Limnology and Oceanography: Methods 7: 399-410.

Hupfer, M., Reitzel, K., and Gruneberg, B. 2020. Methods for measuring internal phosphorus loading. Pages 15-44. In: Internal Phosphorus Loading in Lakes. A.D. Steinman and B.M. Spears (editors). J. Ross Publishing.

Jacobson, P.C., Jones, T.S., Rivers, P., and Pereira, D.L. 2008. Field estimation of a lethal oxythermal niche boundary for adult ciscoes in Minnesota lakes. Trans. Am. Fish. Soc. 137: 1464–1474. https://doi.org/10.1577/T07-148.1

Jane, S.F., Mincer, J.L., Lau, M.P., Lewis, A.S., Stetler, J.T. and Rose, K.C. 2022. Longer duration of seasonal stratification contributes to widespread increases in lake hypoxia and anoxia. Global Change Biology. https://doi.org/10.1111/gcb.16525

Lauber, T. 1999. Can the big bayou be saved? Water quality assessment and management recommendations for Spring Lake watershed, Ottawa and Muskegon Counties, Michigan. Thesis for the Degree of M.S. Michigan State University.

Maberly, S.C., Pitt, J.A., Davies, P.S. and Carvalho, L. 2020. Nitrogen and phosphorus limitation and the management of small productive lakes. Inland Waters 10(2): 159-172.

Magee, M.R., McIntyre, P.B., Hanson, P.C. and Wu, C.H. 2019. Drivers and management implications of long-term Cisco oxythermal habitat decline in Lake Mendota, WI. Environmental Management, 63(3): 396-407.

Mortimor, C.H 1941. The exchange of dissolved substances between mud and water in lakes. Journal of Ecology 29: 280-329.

Nürnberg, G.K. 1988. Prediction of phosphorus release rates from total and reductant-soluble phosphorus in anoxic lake sediments. Canadian Journal of Fisheries and Aquatic Sciences 45(3): 453-462.

Psenner, R., N. Bostrom, M. Dinka, et al. 1988. Phosphorus fractionation: advantages and limits of the method for the study of sediment P origins and interactions. Arch. Hydrobiol. Beih. Ergebn. Limnol. 30: 98-110.

QGIS Development Team (2022). QGIS Geographic Information System. Open Source Geospatial Foundation Project. http://qgis.osgeo.org.

Read, J.S. and Rose, K.C. 2013. Physical responses of small temperate lakes to variation in dissolved organic carbon concentrations. Limnology and Oceanography 58(3): 921-931.

Schindler, D.W. 1977. Evolution of phosphorus limitation in lakes: natural mechanisms compensate for deficiencies of nitrogen and carbon in eutrophied lakes. Science 195(4275): 260-262.

Spoor, W.A. 1990. Distribution of fingerling brook trout, *Salvelinus fontinalis* (Mitchill), in dissolved oxygen concentration gradients. Journal of Fish Biology 36(3): 363-373.

Steinman, A.D., X. Chu, and M. Ogdahl. 2009. Spatial and temporal variability of internal and external phosphorus loads in an urbanizing watershed. Aquatic Ecology 43: 1-18.

Steinman, A.D. and M. Ogdahl. 2008. Ecological effects after an alum treatment in Spring Lake, Michigan. Journal of Environmental Quality 37:22-29.

Steinman, A.D., and M.E. Ogdahl. 2011. Does converting agricultural fields to wetlands retain or release phosphorus? J. No. Am. Benthol. Soc. 30: 820-830.

Steinman, A.D. and M.E. Ogdahl. 2013. Muskegon Lake AOC habitat restoration design: Bear Lake hydrologic reconnection/wetland restoration. Final Project Report. National Oceanic and Atmospheric Administration. http://www.gvsu.edu/cms4/asset/DFC9A03B-95B4-19D5-F96AB46C60F3F345/bear_muck_final_report_final.pdf

Steinman, A.D., R. Rediske, and K.R. Reddy. 2004. The reduction of internal phosphorus loading using alum in Spring Lake, Michigan. Journal of Environmental Quality 33: 2040-2048.

Steinman, A.D., M. Ogdahl, and M. Luttenton. 2008. An analysis of internal phosphorus loading in White Lake, Michigan. Pages 311-325. F.R. Miranda and L.M. Bernard (editors). Lake Pollution Research Progress. Nova Science Publishers, NY.

Steinman, A.D., M.E. Ogdahl, and C.R. Ruetz III. 2011. An environmental assessment of a small shallow lake (Little Black Lake, MI) threatened by urbanization. Environmental Monitoring and Assessment 173: 193-209.

Steinman, A.D., Lamberti, G.A., Leavitt, P., and Uzarski, D.G. 2017a. Biomass and pigments of benthic algae. Pages 223-241. In: Methods in Stream Ecology. Vol 1 (3rd Ed.) R. Hauer and G. Lamberti (editors). Elsevier Press.

Steinman, A.D., M.C. Hassett, M. Oudsema, and S.K. Hamilton. 2017b. Lower Muskegon River Reconnection Project Pre-Restoration Monitoring Report. National Oceanic and Atmospheric Administration. https://www.gvsu.edu/cms4/asset/DFC9A03B-95B4-19D5-F96AB46C60F3F345/bosma_pre-restoration_report_may2017.pdf

U.S. EPA. 1993. Methods for Chemical Analysis of Inorganic Substances in Environmental Samples. Method 365.31; EPA-600/4-79R-93-020/100.

Williamson, C.E., Dodds, W., Kratz, T.K. and Palmer, M.A. 2008. Lakes and streams as sentinels of environmental change in terrestrial and atmospheric processes. Frontiers in Ecology and the Environment 6(5): 247-254.

Yang, C., Yang, P., Geng, J., Yin, H. and Chen, K. 2020. Sediment internal nutrient loading in the most polluted area of a shallow eutrophic lake (Lake Chaohu, China) and its contribution to lake eutrophication. Environmental Pollution 262, p.114292.

Appendix

Table 1. LTHIA model output of the Green Lake watershed area by soil group and 2006 land use/land cover.

Land use	Soil group	Area(acres)
Open Water	A	0.67
Open Water	B	312.91
Open Water	C	1.11
Open Water	D	3.78
Open Space/Park	A	2.45
Open Space/Park	B	44.7
Open Space/Park	C	0.67
Open Space/Park	D	2.89
Low-Density Residential (general 1/3 - 2 ac lots)	A	6
Low-Density Residential (general 1/3 - 2 ac lots)	B	71.17
Low-Density Residential (general 1/3 - 2 ac lots)	D	3.34
High-density Residential (townhomes to 1/4 ac lots)	A	2.45
High-density Residential (townhomes to 1/4 ac lots)	B	16.01
Deciduous Forest	A	12.23
Deciduous Forest	B	254.2
Deciduous Forest	C	72.95
Deciduous Forest	D	47.37
Evergreen Forest	B	1.56
Mixed Forest	A	1.11
Mixed Forest	B	4.23
Mixed Forest	D	0.22
Shrub; Scrub	B	8.45
Shrub; Scrub	D	5.34
Grassland; Herbaceous	A	0.89
Grassland; Herbaceous	B	31.8
Grassland; Herbaceous	D	2.22
Pasture/Hay	A	40.25
Pasture/Hay	B	442.79
Pasture/Hay	C	13.12
Pasture/Hay	D	17.57
Cropland generalized agriculture	A	52.04
Cropland generalized agriculture	B	711.89
Cropland generalized agriculture	C	28.91
Cropland generalized agriculture	D	40.48
Woody Wetlands (swamp)	A	4.45
Woody Wetlands (swamp)	B	59.16
Woody Wetlands (swamp)	C	38.7
Woody Wetlands (swamp)	D	115.87
Emergent Wetlands (marsh)	B	2.67
Emergent Wetlands (marsh)	D	0.22
Total		**2478.81**

Figure 1. MDNR buoy data on dissolved oxygen concentrations.