South Washington Watershed District Lake Management Plan

Modeling and Assessment

Prepared for
South Washington Watershed District

Prepared by
Barr Engineering Co.

December 2025

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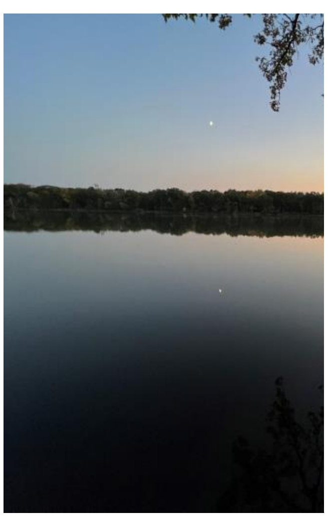

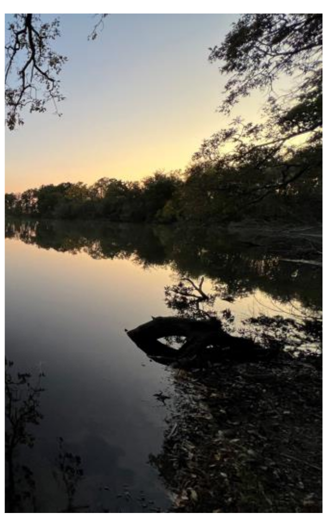

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December 2025

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Abbreviations

CAMP	Community Aided Lake Monitoring Program
cfs	cubic feet per second
CPUE	Catch per Unit Effort
EPA	Environmental Protection Agency
FiN	Fishing in the Neighborhood
FOO	Frequency of Occurrence
FQI	Floristic Quality Index
IBI	Index of Biological Integrity
lbs	pounds
MNDNR	Minnesota Department of Natural Resources
Mobile-P	Mobile Phosphorus Fraction (Sediment)
mg/L	milligrams per liter
MPCA	Minnesota Pollution Control Agency
NCHF	North Central Hardwood Forest
NLCD	National Land Cover Database
Organic-P	Organically-bound Phosphorus Fraction (Sediment)
P8	Program for Predicting Polluting Particle Passage through Pits, Puddles and Ponds
SWWD	South Washington Watershed District
TKN	Total Kjeldahl Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids
USGS	United States Geological Survey
µg/L	micrograms per liter
WCBP	Western Corn Belt Plains

1 Introduction

The South Washington Watershed District (SWWD) is located in the eastern metropolitan area of the Twin Cities and covers two major drainage areas. Water in the western and central portions of the SWWD (or District) flows south and west, ultimately discharging to the Mississippi River; while waters in the eastern portion flow east and south and are part of the Lower Saint Croix River Watershed. Central and western portions of the District are drained via storm sewer and dotted with often interconnected lakes and wetlands. The eastern portion of the District is more rural in nature, and waters flow toward the St. Croix River via creeks, streams, and groundwater.

The SWWD actively monitors and has developed lake management plans for several waterbodies within the District. As part of their on-going efforts to continually improve understanding of water quality conditions and to help inform future management strategies to improve and protect water quality within the District's waters, the SWWD performed a study to evaluate existing conditions in ten lakes and 2 creeks within the watershed. This study involved developing updates to existing watershed and lake water quality models, plus the creation of new models for areas previously unstudied, for nine suburban lakes within the Cities of Oakdale, Woodbury, Cottage Grove, and Newport – Armstrong Lake, Markgrafs Lake, Wilmes Lake (North and South), Colby Lake, Powers Lake, La Lake, Ravine Lake, and Bailey Lake. Five of the waterbodies are currently listed as impaired for excess nutrients by the MN Pollution Control Agency (MPCA). Monitoring data for three of the other waterbodies shows nutrient concentrations that exceed state standards. The MPCA has also recently listed the stream that outlets from Ravine Lake as impaired for ammonia.

The study also involved the analysis of available water quality and flow data for two creeks and one lake in the eastern portion of the watershed within the City of Afton and Denmark Township – Trout Brook, O'Conners Creek, and O'Conners Lake. These waterbodies are located within the Lower Saint Croix River Watershed. The O'Conners subwatershed is considered a landlocked system. Trout Brook discharges into Lake Saint Croix, which is listed as impaired for excess nutrients.

2 SWWD Goals for Lake Management

2.1 Minnesota State Standards and Thresholds

As part of their water resource management programs, the SWWD recognizes and implements strategies to pursue the attainment of state standards for water quality and ecological conditions within the waterbodies that it manages. Table 2-1 summarizes the lake water quality standards and ecological thresholds used by the State of Minnesota to assess lake health. These standards and thresholds are referenced throughout the report and shown on summary plots and figures.

- **Minnesota Lake Eutrophication Standards** -- The MPCA has developed deep and shallow lake eutrophication standards for waterbodies within the state, based on ecoregion. The SWWD is located within the North Central Hardwood Forest (NCHF) and Western Corn Belt Plains (WCBP) ecoregions. However, the MPCA has decided that the NCHF water quality standards will apply to lakes within the WCBP ecoregion, given the proximity to the NCHF ecoregion and the primary characteristics of the watershed (e.g., Ravine Lake). As such, the SWWD has adopted the relevant lake eutrophication standards (phosphorus, chlorophyll-*a*, Secchi disk transparency) for the NCHF ecoregion for all lakes that are managed by the district.

- **Minnesota Chloride Standards** -- Because high concentrations of chloride can harm fish and plant life, the MPCA has established acute and chronic exposure chloride standards. A lake is considered impaired if two or more exceedances of chronic criterion (230 mg/L or less) occur within a three-year period or one exceedance of acute criterion (860 mg/L) is measured.

- **Minnesota Aquatic Plant Thresholds** – The Minnesota Department of Natural Resources (MNDNR) has developed Lake Plant Eutrophication Index of Biological Integrity (IBI) thresholds for assessing the health of aquatic plant communities. The Lake Plant Eutrophication IBI includes two metrics to measure the response of a lake plant community to eutrophication. The first metric is species richness—the estimated number of species in a lake. The second metric is floristic quality index (FQI), which distinguishes the quality of the plant community and can be a reflection of the quantity of nutrients in the lake. Lakes that score below the thresholds contain degraded plant communities and are likely stressed from cultural eutrophication. The IBI thresholds are designated by ecoregion. Ravine Lake is the only lake in this study that falls under the WCBP ecoregion thresholds. All other lakes fall under the NCHF ecoregion thresholds.

Table 2-1 State of Minnesota water quality standards and aquatic plant thresholds used to assess lake health

Type	Parameter	North Central Hardwood Forest Ecoregion[1]		Western Corn Belt Plains Ecoregion[2]
		Shallow Lakes[3]	**Deep Lakes**	**Shallow Lakes**
Water Quality[4]	Total Phosphorus (summer average, µg/L)	≤ 60	≤ 40	N/A
	Chlorophyll-*a* (summer average, µg/L)	≤ 20	≤ 14	N/A
	Secchi Disk Transparency (summer average, µg/L)	≥ 1.0	≥ 1.4	N/A
	Chloride (mg/L)	≤ 230 (chronic) ≤ 860 (acute)		
Aquatic Plants (macrophytes)	Species richness (number of species)	≥ 11	≥ 12	≥ 4
	Floristic Quality Index (FQI)	≥ 17.8	≥ 18.6	≥ 7.7

[1] All lakes in the District are compared to the NCHF ecoregion standards regardless of location.
[2] Ravine Lake is the only lake in the District that falls in the Western Corn Belt Plains ecoregion for MNDNR Plant Eutrophication Index of Biological Integrity (IBI) thresholds.
[3] Shallow lakes have a maximum depth less than 15 feet or littoral area greater than 80% of the total lake surface area.
[4] The summer average is the mean of the concentrations observed between June 1 – September 30.

3 Lake and Watershed Characteristics

Ten lakes – Armstrong, Markgrafs, Powers, Wilmes (North and South), Colby, Bailey, Ravine, La, and O'Conners Lakes – were included in this study. Armstrong Lake is located within the Cities of Lake Elmo and Oakdale. O'Conners Lake is located within Denmark Township, and Ravine Lake is located in Cottage Grove. The other six lakes are located within the City of Woodbury.

3.1 Lakes Summary

Table 3-1 summarizes the lake classification, total watershed area, lake surface area, watershed:surface area ratio, impairment status, and downstream waterbody for each of the lakes included within the study. Five of the lakes are on the Minnesota 303(d) list of impaired waters for excess nutrients.

Most of the lakes included in this study are classified as shallow lakes. The State of Minnesota defines a shallow lake as any lake having a maximum depth less than 15 feet or a littoral area greater than 80% of the total lake surface area. The littoral area of a waterbody is the zone where sunlight can penetrate to the bottom of the lake, allowing aquatic plants to grow. Powers Lake is the only lake that doesn't fit this definition and, as such, is classified as a deep lake.

Healthy shallow lakes are expected to support abundant submerged and floating plant communities throughout most of the lake area. Aquatic plants provide habitat for aquatic insects, zooplankton, fish, waterfowl, and other wildlife. Aquatic plants also compete for nutrients and can shade portions of the waterbody reducing habitable environments for algae.

Table 3-1 Lake characteristics summary for lakes included in study

Lake	Shallow/ Deep	Total Watershed Area (acres)	Lake Surface Area (acres)	Watershed: Surface Area Ratio	Impairment Status	Downstream Waterbody
Armstrong	Shallow	572	10	57:1	Not listed on impaired waters list	North Wilmes
Markgrafs	Shallow	425	44	10:1	Impaired for nutrients since 2006	South Wilmes
Powers	Deep	1,263	61	21:1	Not listed on impaired waters list	South Wilmes
Wilmes Lake (North)	Shallow	2,985	19	157:1	Impaired for nutrients since 2006	South Wilmes
Wilmes Lake (South)	Shallow	5,288	20	264:1		Colby
Colby	Shallow	8,212	73	112:1	Impaired for nutrients since 2006	Bailey
Bailey	Shallow	14,243	62	230:1	Impaired for nutrients since 2024	Central Draw Storage Facility
Ravine	Shallow	1,698	25	68:1	Impaired for nutrients since 2006	Mississippi River
La	Shallow	133	50	3:1	Delisted for nutrients in 2024	Landlocked under typical hydrologic conditions
O'Conners	Shallow	6,305	59	107:1	Not listed on impaired waters list	Landlocked

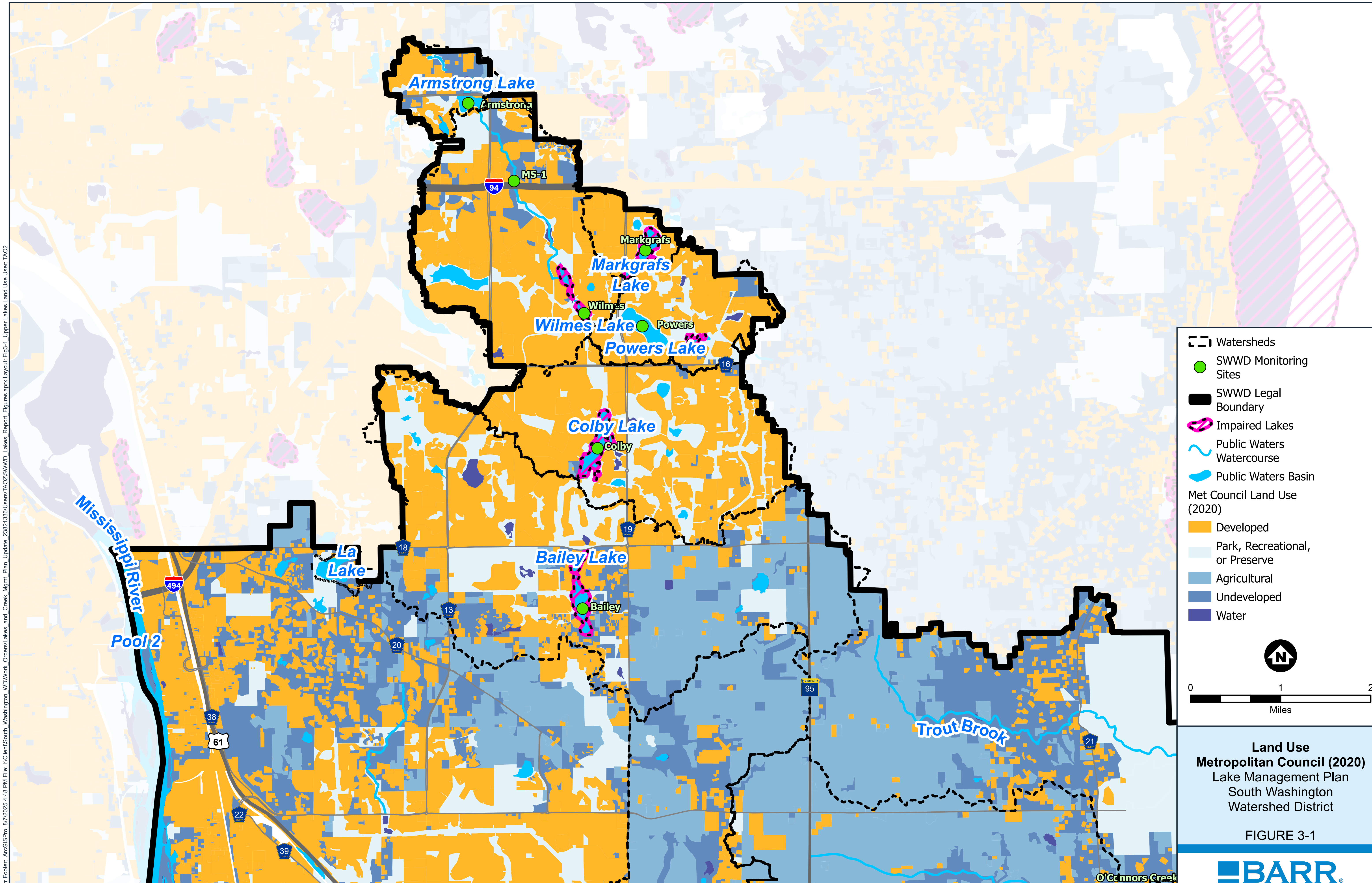
3.2 Watershed Summary

Table 3-2 summarizes the total watershed area, direct watershed area, and primary land use(s) for each of the waterbodies. The predominant land uses for many of the lakes in this study are residential (single-family and multi-family), agricultural, undeveloped, and open space (e.g., park, recreational, or preserve).

Figure 3-1 and Figure 3-2 show the various land use classifications within the major watersheds of the District, summarized into major categories of developed, park, recreational, or preserve, agricultural, undeveloped, and water.

Table 3-2 Watershed summary for lakes included in study

Lake	Total Watershed Area (acres)	Direct Watershed Area (acres)	Primary Land Uses
Armstrong	572	572	Undeveloped, Single Family Residential, Parks
Markgrafs	425	425	Retail/Commercial, Single Family Residential, Parks
Powers	1,263	1,263	Single Family Residential, Parks
Wilmes Lake (North)	2,985	2,413	Single Family Residential, Retail/Commercial, Parks
Wilmes Lake (South)	5,288	615	Single Family Residential, Parks
Colby	8,212	2,924	Single Family Residential, Parks, Golf Course
Bailey	14,243	6,031	Agricultural, Single Family Residential, Parks
Ravine	1,698	1,698	Agricultural, Undeveloped, Parks
La	133	133	Open Water, Undeveloped, Parks
O'Conners	6,305	6,305	Agricultural, Undeveloped, Single Family Residential



Land Use
Metropolitan Council (2020)
Lake Management Plan
South Washington
Watershed District

FIGURE 3-1

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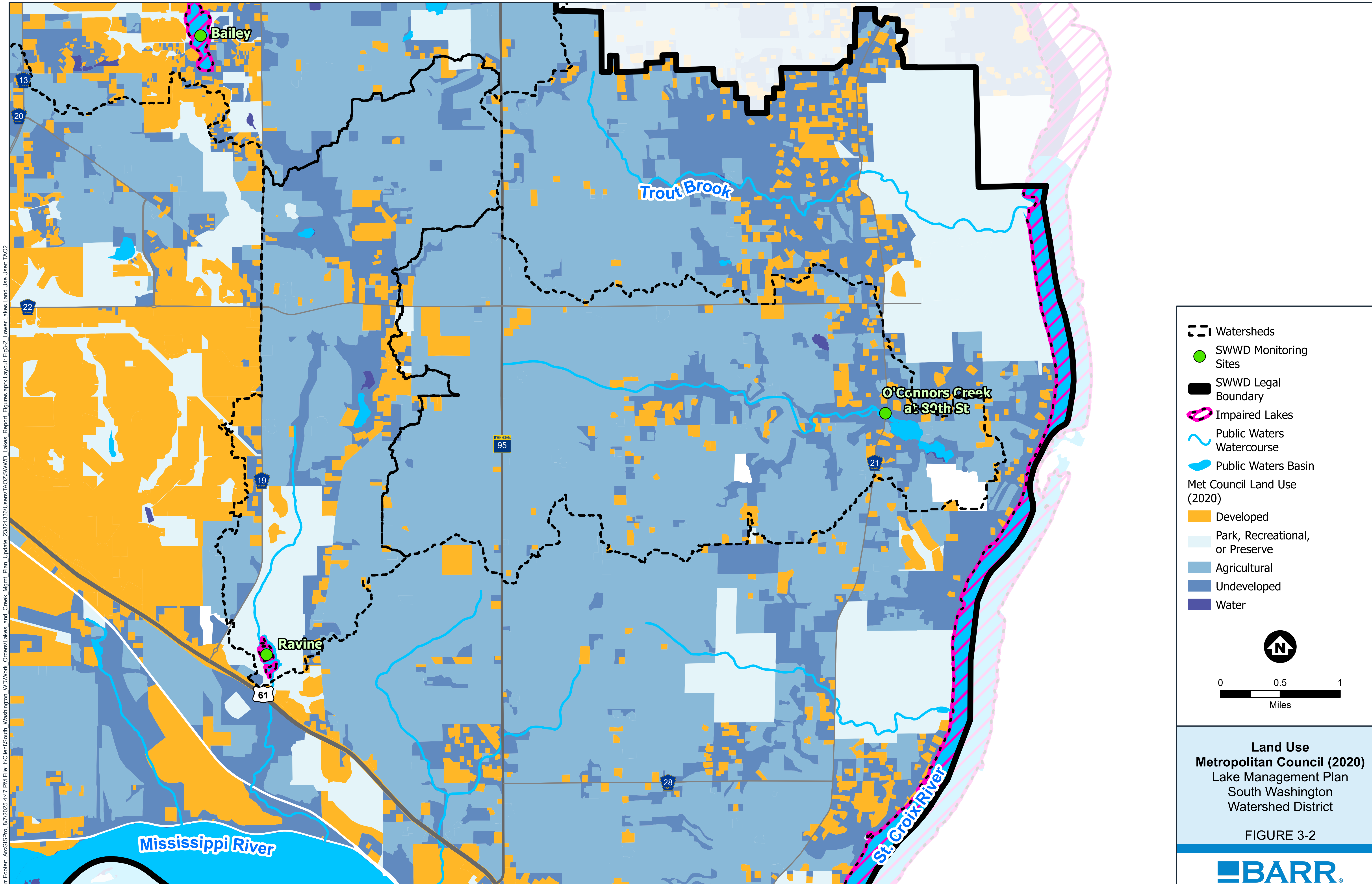


Bailey
Trout Brook
O'Connors Creek at 30th St
Ravine
Mississippi River
St. Croix River
13
20
22
19
95
21
61
28
Watersheds
SWWD Monitoring Sites
SWWD Legal Boundary
Impaired Lakes
Public Waters Watercourse
Public Waters Basin
Met Council Land Use (2020)
Developed
Park, Recreational, or Preserve
Agricultural
Undeveloped
Water
0
0.5
1
Miles
Land Use
Metropolitan Council (2020)
Lake Management Plan
South Washington
Watershed District
FIGURE 3-2
BARR

4 Lake Monitoring Data Background

The physical (water quantity, sediment), chemical (water quality, sediment), and biological (plants, fish, algae, zooplankton) processes within a lake are all linked. Having an understanding of each of these components can give a holistic overview of lake health. The subsections below summarize the monitoring data that was available for use within this study, and the sources for the various datasets.

4.1 Water Quality

Water quality monitoring data was provided for this study by the SWWD. The SWWD collects water quality data for lakes within the District through (1) partnership with the Washington Conservation District, and (2) the Metropolitan Council's Community Aided Lake Monitoring Program (CAMP). CAMP uses volunteers to collect surface water quality data. The data collected through the CAMP program includes total Kjeldahl nitrogen, total phosphorus, chlorophyll-*a*, Secchi transparency depth, and temperature. Qualitative perceptions of a lake's physical and recreational condition at the time of sampling are also noted. Monitoring performed by the Washington Conservation District includes all the parameters monitored through CAMP as well as the collection of chloride concentrations and profile monitoring data (e.g., dissolved oxygen, temperature, and pH at various depths throughout the lake water column).

4.2 Water Quantity

Lake water surface elevations are also monitored by the Washington Conservation District, typically on a bi-weekly to monthly basis between ice-off and late October.

4.3 Aquatic Plants

In 2021 and 2024, the SWWD subcontracted with Stantec to perform aquatic plant community assessments of eight lakes in the district (Stantec, 2021, 2025). To assess the presence, abundance, and health of each lake's aquatic vegetation, Stantec performed point-intercept surveys in June and August of each year, where all submerged, floating leaf, and emergent species were identified at each survey point. The point-intercept survey data were used to calculate various metrics and indices to assess the health of the plant communities. A selection of these plant metrics is summarized in each lake section. Other plant health metrics can be referenced in the 2021 and 2024 Aquatic Vegetation Survey Results reports (Stantec, 2021, 2025).

4.4 Fisheries

Powers, Ravine, and Colby Lakes are included in the Fishing in the Neighborhood (FiN) program run by the MNDNR, which is aimed at increasing angling opportunities, public awareness, and environmental stewardship within the seven-county Twin Cities metro area. The MNDNR FiN program is actively involved in managing the sport and recreational fish populations of these three lakes and, as such, the fish communities are surveyed at a regular frequency, and fishing stocking reports are available. A summary of the most recent fish surveys in each lake, as well as available stocking reports within the last decade, is provided in each lake section.

Markgrafs and La Lakes have historically been used as walleye rearing waterbodies; however, recent fish surveys are unavailable to assess current population abundance.

4.5 Phytoplankton

Phytoplankton (algae) data has not been historically collected on SWWD lakes. Phytoplankton can be single cell, filamentous, or community-based organisms. Understanding the type and abundance of phytoplankton present in a lake can be important for understanding ecological health. An inadequate phytoplankton population can limit a lake's zooplankton population and indirectly limit fish production. However, excess phytoplankton from a high amount of nutrients can reduce water clarity, impact aquatic plant growth, and possibly cause human health concerns. Cyanobacteria, also known as blue-green algae, often outcompete other phytoplankton groups due to ecological advantages. Identifying the specific species present can improve estimates of the potential benefits or limitations of management efforts, support informed decision-making, and help set realistic expectations.

4.6 Zooplankton

Zooplankton data has also not been historically collected on SWWD lakes. Understanding the type and abundance of zooplankton present in a lake can be important for understanding ecological health. Zooplankton are microscopic aquatic animals that drift and move throughout the lake water column. They play major roles in the aquatic food web by consuming algae and are primary food sources for larger organisms such as fish.

4.7 Groundwater

Several of the lakes within the SWWD are known to receive groundwater inflow or lose water into the surrounding groundwater system. Understanding the potential for groundwater interactions and the water quality of shallow groundwater within the vicinity of these lakes is necessary to predict potential in-lake water quality impacts. Unfortunately, available data on groundwater elevations and water quality within the SWWD is limited. This study used publicly available data from the National Water Quality Monitoring Council's Water Quality Portal (National Water Quality Monitoring Council, 2025) to assess groundwater quality. This portal summarizes available data from the United States Geological Survey (USGS), the Environmental Protection Agency (EPA), and over 400 state, federal, tribal, and local agencies. Shallow groundwater wells included within this database, and located within the SWWD, were typically monitored annually. The groundwater total phosphorus concentrations used in the applicable in-lake models were derived from this data. The calculated groundwater concentration for each applicable lake is detailed in its respective section.

4.8 Sediment

Phosphorus release from lake bottom sediments can be a significant contributor to excess nutrient concentrations in lakes. Phosphorus in lake bottom sediment is often bound to a range of different elements such as iron and manganese (often referred to as mobile phosphorus (mobile-P)), aluminum, or calcium. The mobile-P fraction can be released from sediment during low oxygen conditions caused by microbial activity. Phosphorus can also be found incorporated into organic matter in the sediment (organically bound phosphorus (organic-P)). A portion of the organic-P is released into the water column from lake sediments through mineralization by microbes, but typically at a slower rate than mobile-P. The mineralization release rate is controlled by lake water temperature and can occur under aerobic or anaerobic conditions. Phosphorus release from sediment is typically termed as "internal phosphorus loading" or "internal loading".

Sediment cores were collected from eight of the SWWD lakes in October 2024 and used to evaluate the internal phosphorus loading potential of the mobile-P and organic-P fractions. The sediment coring locations are shown on maps in Appendix A. In the top 6 cm of each core, 2 cm slices were cut to analyze the variation in sediment parameters. Table 4-1 summarizes the average concentration and ranges of concentrations of mobile-P and organic-P in the top 6 centimeters of the sediment cores. Table 4-1 also summarizes the average concentration and range of concentrations of extractable iron in the top 6 centimeters of the cores. A higher concentration of extractable iron indicates more iron is available to bind phosphorus in the sediment under aerobic conditions. Table 4-2 provides the calculated average maximum potential mobile-P internal loading rates for the sediment cores collected in 2024 and compares the values with other lakes in the metro area. The calculated average maximum potential mobile-P internal loading rates use a methodology described in Pilgrim et. al. (Pilgrim, Huser, & Brezonik, 2007).

Similar to the observations from lake sediment studies completed in District lakes in 2016 and 2017 (WENCK, 2018), the 2024 sediment coring data indicates that the SWWD lakes with high potential mobile-P loading rates include North and South Wilmes, Ravine, Powers, Colby, and Bailey. The SWWD lakes with moderate mobile-P loading signatures include Armstrong and Markgrafs. All of the SWWD lakes sampled in 2024 had moderate organic phosphorus fractions in their sediment cores, except Ravine (Table 4-1). The sediment core data collected from Ravine Lake indicates that internal phosphorus loading from organic-P could be a considerable source of phosphorus. Previously, research has focused heavily on mobile-P acting as the main mechanism of internal phosphorus loading. However, recent research and monitoring data indicate that organic-P, especially organic-P fractions that are susceptible to biological or chemical decomposition (e.g, phosphate esters, phospholipids), can be a significant source of phosphorus and can maintain high productivity in lakes (Wei, et al., 2022).

Sediment cores were not collected from La or O'Conners Lakes in October 2024. Sediment cores were previously collected in La Lake in 2017. Laboratory release rate experiments conducted on the La Lake sediment cores indicated relatively low anaerobic phosphorus release rates with an average of approximately 1.7 mg/m^2/day (WENCK, 2018). However, even a low internal phosphorus loading rate can notably increase phosphorus concentrations in the lake water column, especially in shallow lakes with small water volumes per surface area of lake bottom.

Table 4-1 Average, Maximum, and Minimum mobile-P, organic-P, and extractable iron concentrations in the top 6 cm of sediment cores collected in 2024

Lake	Number of Sediment Cores	Mobile Phosphorus Concentrations per Volume (µg P/cm^3) Average (Range)	Organic Phosphorus Concentrations per Volume (µg P/cm^3) Average (Range)	Extractable Iron (µg Fe/ cm^3) Average (Range)
Armstrong	1	35 (14 – 65)	25 (19 – 32)	482 (295 – 761)
Markgrafs	2	25 (16 – 34)	68 (59 – 76)	513 (360 - 653)
Powers	1	73 (20 – 120)	42 (36 – 51)	515 (253 - 645)
North Wilmes	1	229 (173 – 260)	61 (30 – 112)	1,754 (1,080 – 2,141)
South Wilmes	1	117 (95 – 160)	56 (46 – 62)	936 (601 – 1,244)
Colby	3	41 (12 – 94)	47 (31 – 66)	853 (417 – 1,346)
Bailey	2	139 (39 – 268)	65 (42 – 93)	824 (445 – 1,348)
Ravine	1	149 (95 – 203)	328 (209 – 498)	1,165 (709 – 1,792)

Table 4-2 Maximum potential mobile-p internal loading rates compared to other Twin Cities Metro Area lakes

Lake	Maximum Potential Internal Phosphorus Load from Mobile-P ($mg/m^2/d$)[8]
North Wilmes	**>12.0**
Ravine	**>12.0**
Bailey	**>12.0**
Kohlman[1]	>12.0
South Wilmes	**>12.0**
Isles (pre-alum, deep hole)[2]	>12.0
Harriett (pre-alum, deep hole)[2]	11.1
Calhoun/Bde Maka Ska (pre-alum, deep)[2]	10.8
Fish E[3]	10.5
Powers	**10.3**
Cedar (pre-alum)[2]	9.3
Fish W[3]	8.1
Como[3]	7.6
North Cornelia (pre-alum)[4]	7.6
Calhoun/Bde Maka Ska (pre-alum, shallow)[3]	5.6
Colby	**5.5 (1.6 – 9.2)**
Armstrong	**4.5**
Keller[1]	3.5
Parkers[3]	3.5
Markgrafs	**3.1 (2.2 – 4.0)**
Phalen[3]	2.3
McCarrons[3]	2.0
La Lake[7]	**1.7**
Bryant[3]	1.5
South Cornelia (pre-alum)[4]	1.3
Mirror Lake[6]	1.0
Smetana[5]	0.7
Minnewashta[3]	0.2

Sources:
[1] (Barr Engineering Co., 2007)
[2] (Huser & Pilgrim, 2014)
[3] (Pilgrim, Huser, & Brezonik, 2007)
[4] (Barr Engineering Co., 2018)
[5] (Barr Engineering Co., 2020)
[6] (Barr Engineering Co., 2023)
[7] Average measured anaerobic release rate (WENCK, 2018)
[8] (Pilgrim, Huser, & Brezonik, 2007)

5 P8 Watershed Modeling Methodology – Suburban Lakes

5.1 P8 Model Runoff and Phosphorus Loading Background

Central to a lake water quality analysis is the use of a water quality model that has the capacity to predict the amount of water and pollutants that reach a lake via stormwater runoff (i.e., watershed or external loading). The P8 (Program for Predicting Polluting Particle Passage through Pits, Puddles, and Ponds) modeling software was used to estimate watershed loads to nine of the lakes within this study (I.E.P, Inc., 1990). P8 incorporates hourly precipitation and daily temperature data. Barr utilized P8 to calculate the daily water volume and nutrient loads introduced from each tributary subwatershed in each of the suburban lakes modeled.

The following subsections describe the methodology Barr utilized for the P8 models.

5.2 Previous Watershed P8 Modeling Efforts

Watershed water quality models were previously developed for the following lakes within the SWWD (WENCK, 2018):

- Armstrong
- North Wilmes
- South Wilmes
- Markgrafs
- Powers
- Colby
- La
- Ravine

Barr reviewed each of these models to understand how they compare to current conditions in the watershed and identify any updates necessary to accurately represent loading to each lake. The updates made to these existing models are described in Section 5.3.

5.3 P8 Model Updates

5.3.1 Data Collection

At the initiation of the study, Barr requested data from the SWWD and its member cities within the study area to assist in updating the P8 models with information on new developments since 2016, when the existing P8 models were developed. This information request included:

- Storm sewer GIS data within areas draining to the modeled lakes, including dimension information (e.g., structure inverts, pipe diameters, upstream/downstream inverts, etc.)

- Supplementary record drawings detailing complex features (e.g., lake/pond outlet structures, pump and lift station information)
- BMP inventory information:
  - Spatial data indicating where BMPs exist within the study area
  - BMP design information, such as what was submitted for permit applications, including drainage area, routing, inlets/outlets, storage volumes, infiltration/filtration rates, etc.
  - Estimated BMP pollutant reduction benefits
  - Record drawings of BMPs
  - Operations and maintenance information for actively managed BMPs
- Bathymetric data for lakes, stormwater ponds, and wetlands
- Hydrologic and climatic monitoring data (e.g., precipitation gauge data)
- Development data:
  - Spatial data showing site stormwater permit locations from 2016-2024
  - Site plans within the study area for new developments since 2016 showing proposed watershed divides, storm sewer, and proposed BMPs
  - Development models (e.g., HydroCAD, XPSWMM, P8) if available for new developments since 2016
  - Recent land use/land cover data

A variety of data was received and reviewed by Barr to help inform the necessary P8 model updates. Those updates are described further in Section 5.3.2.

5.3.2 Model Updates

Generally, the P8 modeling updates made by Barr fall within the following categories:

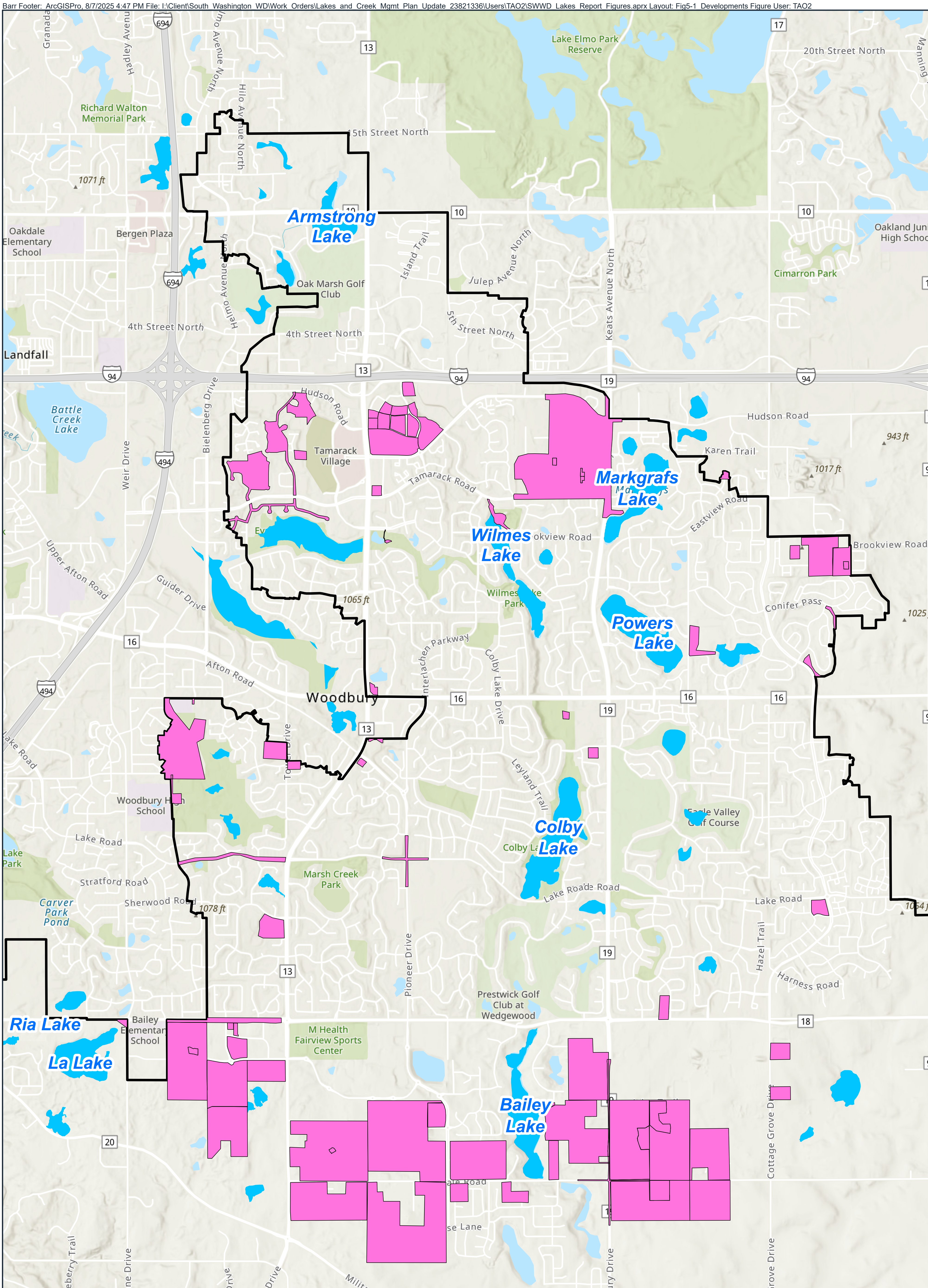
- Climate data
- Watershed hydrology
- Conveyance/routing
- Best management practices

Table 5-1 summarizes specific model updates that were made to each of the study area lakes' contributing P8 model(s), as well as the data sources or assumptions for each of the model updates. Additionally, Figure 5-1 shows the developments throughout the watershed district that were reviewed for updating the previously developed P8 models.

Table 5-1 Water balance outlet rating curve and groundwater flow assumptions

Lake	P8 Model Updates	Data Source / Assumption
Armstrong	Added additional ponds	Oakdale HydroCAD model, GIS storm sewer, LiDAR, assumed depths (if unavailable)
	Updated precipitation files	Oakdale CoCoRaHS Station (2019) and Woodbury M Health Sports Center Gauge (2012-2018, 2020-2022)
	Updated TP and TSS scaling factors	Calibration (see Section 5.3.3)
Markgrafs	Added additional ponds around shoreline of Markgrafs Lake	Woodbury pond inventory, LiDAR, assumed depths (if unavailable), GIS storm sewer
	Updated imperviousness assumptions	Metropolitan Council 2020 Generalized Land Use
	Updated precipitation files	Woodbury M Health Sports Center Gauge
Powers	Added new development information	Woodbury development plans, GIS storm sewer, LiDAR
	Updated precipitation files	Woodbury M Health Sports Center Gauge
North Wilmes	Added new development information	Woodbury development plans, GIS storm sewer, LiDAR
	Updated imperviousness assumptions	Metropolitan Council 2020 Generalized Land Use
	Updated precipitation files	Woodbury M Health Sports Center Gauge
South Wilmes	Added Seasons Park BMP	SWWD Seasons Park final design P8 model
	Updated precipitation files	Woodbury M Health Sports Center Gauge
Colby	Updated imperviousness assumptions	Metropolitan Council 2020 Generalized Land Use
	Added stormwater ponds on west side of Colby Lake	Woodbury pond inventory, LiDAR, assumed depths (if unavailable), GIS storm sewer
	Updated precipitation files	Woodbury M Health Sports Center Gauge
La	Added new development information	Woodbury development plans, GIS storm sewer, LiDAR
	Updated pervious curve number and imperviousness assumptions	Metropolitan Council 2020 Generalized Land Use
	Updated precipitation files	Woodbury M Health Sports Center Gauge
Ravine	Added new development information	Cottage Grove development plans, GIS storm sewer, LiDAR
	Updated precipitation files	Woodbury M Health Sports Center Gauge

In the existing P8 models, Barr did not make updates to the BMPs that were previously modeled, nor were updates made to naming conventions from the source models. It was also noted that the existing P8 models did not remove open water area from subwatershed or impervious areas. Where Barr updated hydrologic inputs to the P8 models, we utilized the same methodology to remain consistent with the previous modeling approach.



Developments and Capital Projects Reviewed for Model Incorporation*

Lakes and Ponds

SWWD Legal Boundary

*Not shown is Cottage Grove HERO Center, City Hall, and Washington County Service Center in the Ravine Lake Subwatershed

0 2,000 4,000
Feet

Developments and Capital Projects Reviewed for Model Incorporation
Lakes Management Plan
South Washington Watershed District
FIGURE 5-1

BARR

5.3.3 P8 Model Calibration

As part of the previous P8 modeling efforts, four of the P8 models were calibrated to the District's monitoring stations. These models were calibrated to observed annual TP loads and total annual volume at the monitoring stations (Wenck, 2018).

Barr verified the volume calibration using more recently collected monitoring data at MS-1, which is located near the intersection of Interstate 94 and Radio Drive and upstream of North Wilmes. For 2019, the modeled total annual volume at MS-1 was within 8% of the observed volume. For 2022, the modeled total volume was within 11% of the observed volume. Updated monitoring data was not available for the other stations used in the original calibration, so Barr could not confirm whether those calibrations remain valid for recent years.

Barr also calibrated the P8 models during the water balance and water quality calibration of the in-lake models (see Sections 6.2.1 and 6.2.2, respectively). P8 uses scaling factors to adjust the amount of particulate pollutant loading being washed off of a watershed and contributing to runoff loading. Barr updated the TP scaling factor for Armstrong Lake to reduce watershed TP loading to the lake during calibration of the in-lake models (see Section 6.4) and to match the monitored in-lake TP concentrations. P8 calculates runoff and associated pollutant loading from directly and indirectly connected impervious surfaces separately from pervious surfaces. Prior modeling efforts assigned a generalized ratio of indirectly connected impervious area to directly connected impervious area across each of the contributing drainage areas to the lakes. Using the Metropolitan Council's 2020 Generalized Land Use layer to calculate imperviousness, Barr updated this assumption within several of the models by calculating specific indirectly vs. directly connected impervious fractions for each individual catchment within the P8 models.

The existing and updated calibration factors are summarized in Table 5-2 below.

Table 5-2 P8 calibration factors

Lake	Existing TP Scale Factor	Updated TP Scale Factor	Existing Indirectly Connected / Directly Connected Imperviousness Ratio	Updated Indirectly Connected / Directly Connected Imperviousness Ratio
Armstrong	2.25	1	45/55	Land use based by subwatershed
Markgrafs	0.9	0.9	75/25	Land use based by subwatershed
Powers	0.89	0.89	24/76	24/76
North Wilmes	0.9	0.9	75/25	Land use based by subwatershed
South Wilmes	0.9	0.9	75/25	75/25
Colby	1.5	1.5	50/50	Land use based by subwatershed
La	1	1	100/0	Land use based by subwatershed
Ravine	1	1	30/70	30/70

5.4 P8 Model Creation – Bailey Lake

Barr developed the Bailey Lake watershed loading model using P8, version 3.5. Utilizing information from SWWD's XPSWMM model (Houston Engineering Inc., 2017) and development data provided by the City of Woodbury, Barr developed the Bailey Lake P8 model using methodology and assumptions outlined in the following subsections.

5.4.1 Hydrology

P8, unlike XPSWMM, is designed for long-term simulation of watershed runoff and pollutant transport; hydrologic modeling in P8 uses a more simplistic methodology than that used in XPSWMM. Subwatersheds developed for XPSWMM modeling were also used for P8 modeling. For the majority of hydrologic parameters required in P8, XPSWMM modeled hydrologic parameters were used directly to ensure consistency between the two models (e.g., imperviousness). However, P8 utilizes both directly and indirectly connected imperviousness for generating pollutant loading. To estimate the indirectly connected imperviousness, Barr utilized the assumed ratio of directly connected imperviousness to indirectly connected imperviousness shown in Table 5-3, which are based on land use categories from the Metropolitan Council's 2020 Generalized Land Use dataset.

Table 5-3 Directly connected and indirectly connected imperviousness ratio by land use

Land Use	Directly Connected Imperviousness Fraction	Indirectly Connected Imperviousness Fraction
Agricultural	0.01	0.99
Farmstead	0.12	0.88
Golf Course	0.05	0.95
Industrial and Utility	0.72	0.28
Institutional	0.4	0.6
Major Highway	0.5	0.5
Mixed Use Residential	0.37	0.63
Multifamily	0.37	0.63
Office	0.72	0.28
Open Water	1	0
Park, Recreational, or Preserve	0.05	0.95
Retail and Other Commercial	0.85	0.15
Single Family Attached	0.30	0.7
Single Family Detached	0.2	0.8
Undeveloped	0	1

Barr also updated the imperviousness for watersheds where new development has occurred since the creation of the SWWD XPSWMM model. These developments are shown in Figure 5-1. For these developments, Barr utilized the impervious assumptions outlined in the NWS and CDSF modeling report (Houston Engineering Inc., 2017), which are included in Table 5-4 below.

Table 5-4 Development imperviousness assumptions for Bailey Lake P8 model

Land Use	Percent Impervious
Medium Density Residential	50.7%
Low Density Residential	35.6%
Park	8.6%

The only parameter required for P8 modeling not included in the hydrologic parameters generated for the XPSWMM model is the pervious curve number. Barr developed the pervious curve number utilizing a weighted approach for various curve numbers based on the hydrologic soil group for underlying soils within each subwatershed. The assumed curve numbers for each hydrologic soil group are included in Table 5-5.

Table 5-5 Pervious Curve Numbers for hydrologic soil groups

Hydrologic Soil Group	Pervious Curve Number
A	39
B	61
C	74
D	80
A/D	80
B/D	80
C/D	80
<Null>	61

5.4.2 Water Quality Device Hydraulics

In P8, runoff generated from subwatersheds, along with associated sediment and pollutant loads, is routed to "devices" which determine how and where flow is hydraulically routed throughout the model. Pipe devices are used for routing and do not provide any runoff volume reduction or water quality treatment, while devices representing water quality treatment BMPs (e.g., ponds, infiltration basins, etc.) have the potential to remove runoff volume and pollutants via particle settling and filtration.

The inputs for each P8 device in the existing XPSWMM model (e.g., pond, infiltration device, pipe, etc.) were developed using storage and outlet hydraulic information directly from the XPSWMM model, including:

- Device outlet types were assigned from the XPSWMM storm sewer utility network

- Storage volume for all above ground features (including the permanent pool and live storage volume for wet ponds) was calculated from stage-area data from the XPSWMM model
- Infiltration rates for infiltrating BMP devices were assumed based on underlying SSURGO soil data or best available information

Barr also included device hydraulics information provided by the City of Woodbury for developments that were constructed after 2017 when the XPSWMM model was built (see Figure 5-1).

Pipe devices were modeled at key locations throughout the watershed so that pollutant loading results could be summarized at these locations. Bailey Lake was also modeled as a pipe device to summarize the watershed inflows for the in-lake modeling.

5.4.3 Precipitation and Temperature

The P8 model was developed using hourly precipitation and daily average temperature data recorded at the Woodbury MnHealth Sports Center Gauge.

5.4.4 Pollutant Loading

Barr utilized the default NURP50 particle scale file to assign particle characteristics and water quality components assumptions used to define particle loading and associated pollutant loading within the Bailey Lake watershed.

6 In-lake Modeling Methodology – Suburban Lakes

6.1 In-lake Model Background

The purpose of in-lake modeling is to establish a relationship between the amount of nutrients that enter a lake and the concentration of these nutrients within the lake water column. Generally, for freshwater lakes, phosphorus is the main nutrient of concern and is typically the most limiting nutrient for algal growth. As such, phosphorus modeling is discussed in greater detail in this report. However, nitrogen also plays a role in limiting algal growth in lakes and is touched on in individual lake sections as appropriate.

There are several processes that dynamically increase or decrease the concentration of phosphorus in the lake water column, including the following listed below (the "-" or "+" indicates that the mechanism generally either reduces or increases phosphorus). The extent to which each of the processes can be modeled is dependent on which model is being used and the monitoring data available to accurately calibrate the process.

- **Watershed Runoff (+):** Phosphorus enters the lake through natural channels, stormwater runoff from surrounding properties, and discharge from storm sewer pipes following precipitation or snow melt events.
- **Upstream Lakes (+):** Outflow from upstream lakes introduces phosphorus into the downstream lakes.
- **Atmospheric Deposition (+):** Phosphorus deposits into the water body from the atmosphere.
- **Settling (-)**: Phosphorus in phytoplankton and attached to particles settles out of the lake water column to the sediments.
- **Flushing (-)**: Typically represents the phosphorus that is discharged through an outlet structure. For landlocked lakes and/or lakes with a high connection to groundwater, flushing includes phosphorus that is discharged to groundwater.
- **Lake Bottom Sediment Loading (+)**: Mobile phosphorus from lake bottom sediments may release into the water column during low oxygen conditions. Organic phosphorus will release as bacteria breakdown debris in the lake sediment that contains phosphorus (e.g., decaying leaves, plants, and algae). Phosphorus release from lake bottom sediments is also known as internal loading.
- **Benthivorous (bottom-feeding) fish (+):** Although not always modeled as a separate internal load, benthivorous fish are presumed to cause additional internal phosphorus loading during certain periods due to stirring of the bottom sediments. Fish defecation can also create nutrient loads.
- **Phytoplankton and macrophyte growth (-):** Phosphorus will be removed from the water column and the sediment through uptake by phytoplankton and macrophytes during the growth phase.
- **Phytoplankton and macrophyte die-off and decay (+):** Phosphorus in the phytoplankton and plant tissues is released into the water column when the species die and decay.

- **Curly-leaf pondweed die-off and decay (+)**: Phosphorus in the plant tissue is released into the water column when curly-leaf pondweed dies and decays. Curly-leaf pondweed die-off and decay occurs much earlier than other native plant species (typically in late June and July), so this species is typically modeled separately when included in a nutrient mass balance.

Two different in-lake models were used within this study to model a selection of the in-lake processes listed above. Model selection and the selected processes incorporated in the models were generally based on the monitoring data available for calibration as well as the desired level of model detail (e.g., better understanding of algal growth and nitrogen dynamics).

6.1.1 Finite Difference Spreadsheet Model

The finite difference spreadsheet model was used for La, Armstrong, Powers, and Bailey Lakes. This phosphorus mass balance model, which was developed by Barr Engineering Co., is an Excel spreadsheet model that calculates daily phosphorus concentrations in the lake water column based upon phosphorus inputs (internal and watershed loading) and phosphorus losses (settling and outflows). The model operates on a daily time step and assumes that every input to the model is completely mixed both vertically and horizontally in the lake water volume. The spreadsheet inputs for the SWWD lakes simulated with this model included:

- Climatic Inputs
  - Air temperature
  - Wind
  - Humidity
- Water Balance
  - Surface water inflows and outflows
  - Direct precipitation
  - Evaporation (estimated from climatic inputs)
  - Groundwater inflows and outflows
  - Water level and volume
- In-lake Physical Parameters
  - Water temperature
  - Dissolved oxygen concentration
- Nutrient Balance
  - Phosphorus (total phosphorus)
- Lake Bottom Sediment Processes
  - Phosphorus release as a function of dissolved oxygen in the lake water column

6.1.2 Barr Shallow Lake Model

The Barr Shallow Lake Model was used for North Wilmes, South Wilmes, Markgrafs, Colby, and Ravine Lakes. This mass balance and ecological model, which was developed by Barr Engineering Co., is a Python programming language-based model that uses inputs from multiple Excel spreadsheets.

The Barr Shallow Lake Model provides a more accurate representation of lake function as it includes phytoplankton growth, nutrient limitation resulting from nitrogen and phosphorus, transformation and losses of nutrients as a function of phytoplankton growth and die-off, simulation of internal phosphorus loading based upon phosphorus composition and concentration in the lake bottom sediments, and calculation of internal loading based upon dissolved oxygen and temperature conditions that vary hourly in the model. The model incorporates the effects of climate on in-lake nutrients and phytoplankton, and hence, an improved understanding of the effect of climatic variability on the expected range of management outcomes can be estimated. This model is considered to be zero-dimensional, meaning it is assumed that every input to the model is completely mixed both vertically and horizontally in the lake water column. The model inputs for the SWWD lakes simulated with this model included:

- Climatic Inputs
    - Air temperature
    - Wind
    - Sunlight
    - Humidity
- Water Balance
    - Surface water inflows and outflows
    - Direct precipitation
    - Evaporation (estimated from climatic inputs)
    - Groundwater inflows and outflows
    - Water level and volume
- In-lake Physical Parameters
    - Water temperature
    - Dissolved oxygen concentration
- Nutrient Balance
    - Phosphorus (ortho-phosphate, dissolved and particulate organic phosphorus, particulate inorganic phosphorus)
    - Nitrogen (nitrate, nitrite, ammonia, dissolved and particulate organic nitrogen)
- Lake Bottom Sediment Processes

- Phosphorus release from iron-bound phosphorus as a function of dissolved oxygen in the lake water column
- Phosphorus release from organically-bound phosphorus as a function of lake water temperature

- Phytoplankton Processes
  - Dissolved phosphorus and nitrogen uptake with growth
  - Phytoplankton settling
  - Particulate phosphorus and nitrogen release with mortality
  - Growth can be phosphorus, nitrogen, temperature, or light limited

Each of the processes listed above occurs at different magnitudes throughout the model duration and hence the processes are quantified (e.g., calibrated) by matching model results with the field-measured parameters. Macrophyte (plant) growth, die-off, and decay processes, as well as rough fish activities, are not simulated in the model.

6.2 In-lake Model Calibration

6.2.1 Water Balance

The first step in model calibration was to simulate the growing season (May 1 – September 30) water balances around each of the lakes. Two years of observed data – one representing a wet year and one representing a dry year – were selected to allow for assessment of in-lake conditions under different climatic conditions. Model year 2019 was selected for modeling a wet year condition, and model year 2022 was selected for modeling a dry year condition. The growing season precipitation totals for model years 2019 and 2022 are summarized in Table 6-1 (sources: Woodbury MnHealth Sports Center Gauge and Oakdale, MN CoCoRaHS Station (Community Collaborative Rain, Hail & Snow Network)). Watershed contributions under existing land use conditions were estimated for model years 2019 and 2022 from P8 modeling as described in Section 5.

Table 6-1 Modeled precipitation amounts for 2019 and 2022 growing seasons

Model Year	Growing Season (May 1 through Sept 30) Precipitation (inches)	
	Armstrong Lake	All other suburban lakes
2019	24.8	24.8
2022	12.3[1]	10.3

[1] Climate data from various precipitation gauges in the SWWD indicated that storm events in May 2022 had notably variable precipitation depths dependent on location in the District. To better match the observed water level conditions on Armstrong Lake during model year 2022, precipitation data from the CoCoRaHS station in Oakdale was used for model calibration.

Water balances were developed for all nine suburban lakes using either the finite difference spreadsheet or Barr Shallow Lake models. The models use estimated daily or bi-hourly watershed runoff inflows (predicted by P8 models), daily precipitation, daily evaporation, estimated groundwater inflow or outflow, outflow rating curves (e.g., pumps, outlet control structure), and observed lake levels to estimate changes in the water level of the lake. Changes in lake water volumes over time were calibrated by matching the modeled lake surface elevations to observed elevations.

Figure 6-1 and Figure 6-2 show example water balance calibrations that were completed for South Wilmes for the 2019 and 2022 growing seasons. The model predicted water levels, shown by the orange line on the plot, were calibrated to match as closely as possible to observed water levels collected bi-weekly from May through September, indicated by the blue circles. Model calibration indicated groundwater outflow in both modeled years for South Wilmes. Figure 6-3 shows the 2019 and 2022 water balance volume comparisons for South Wilmes for all modeled inflows (direct watershed, precipitation, upstream lakes) and outflows (evaporation, discharge through outlet, groundwater discharge), where the total volume is the amount of water that moved through the lake in each year within the May through September time period.

The water balance calibrations and summary pie charts for all nine suburban lakes can be reviewed in Appendix B. Table 6-2 and Table 6-3 provide a summary of the inflow and outflow water balance quantities for each modeled lake for 2019 and 2022, respectively.

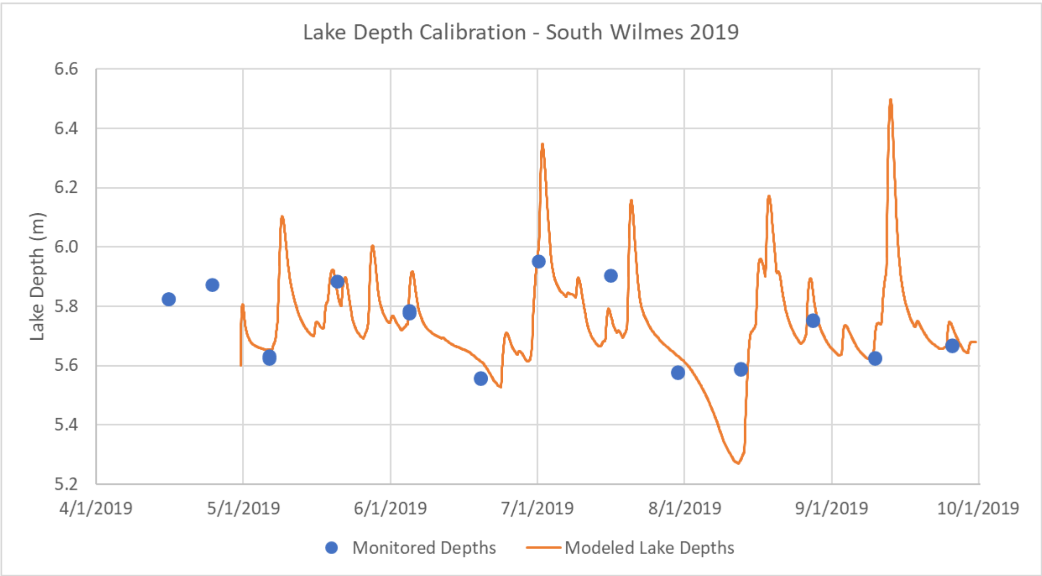


Figure 6-1 South Wilmes Lake 2019 water balance calibration

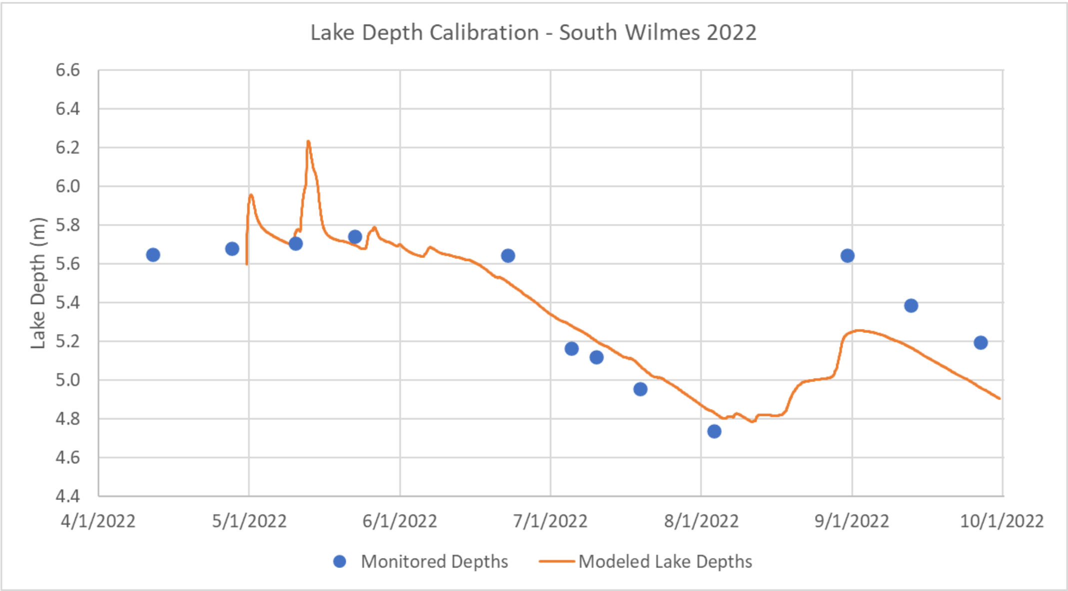
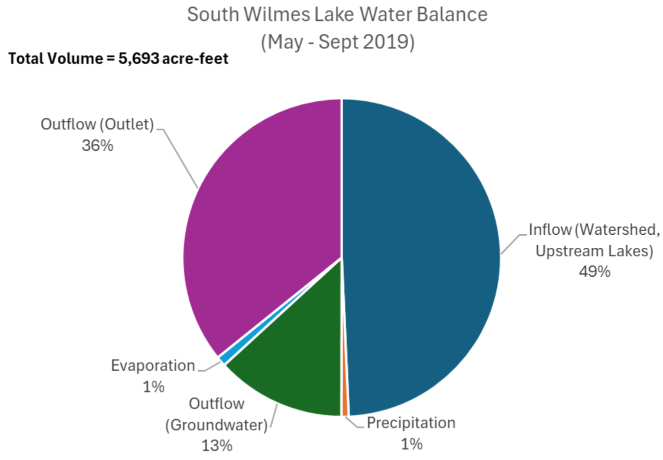


Figure 6-2 South Wilmes Lake 2022 water balance calibration



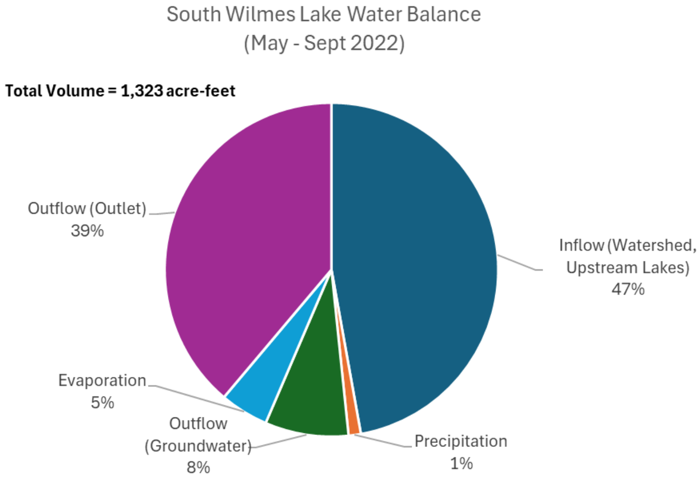


Figure 6-3 South Wilmes Lake water balance summaries

Table 6-2 Suburban lakes 2019 water balance summary

Lake	Total Volume (ac-ft)	% Inflow		% Outflow		% Inflow (I) or Outflow (O)
		Watershed, Upstream Lake	Precipitation	Outlet	Evaporation	Groundwater
Armstrong	409	41%	9%	46%	4%	-
Markgrafs	1,785	13%	5%	45%	7%	29% (I)
Powers	1,990	44%	6%	44%	6%	-
North Wilmes	2,104	48%	2%	48%	2%	-
South Wilmes	5,693	49%	1%	36%	1%	13% (O)
Colby	6,166	47%	3%	47%	3%	-
Bailey	6,663	48%	2%	46%	2%	2% (O)
La	222	2%	52%	-	29%	17% (I)
Ravine	838	18%	6%	42%	8%	26% (I)

Table 6-3 Suburban lakes 2022 water balance summary

Lake	Total Volume (ac-ft)	% Inflow		% Outflow		% Inflow (I) or Outflow (O)
		Watershed, Upstream Lake	Precipitation	Outlet	Evaporation	Groundwater
Armstrong	188	37%	8%	29%	7%	19% (O)
Markgrafs	604	12%	6%	31%	24%	28% (I)
Powers	819	37%	6%	24%	19%	14% (O)
North Wilmes	723	45%	2%	31%	8%	14% (O)
South Wilmes	1,323	47%	1%	39%	5%	8% (O)
Colby	1,793	44%	3%	40%	13%	-
Bailey	1,693	47%	3%	35%	8%	7% (O)
La	143	3%	28%	-	46%	24% (I)
Ravine	504	12%	4%	35%	17%	32% (I)

6.2.1.1 Water Balance Assumptions

Table 6-4 summarizes the outlet rating curve and groundwater assumptions for each of the nine urban lake water balances. In general, groundwater flow is the main water balance calibration parameter for each of the lake models; as such, results for estimates of groundwater flow also include any error that is resultant within the water balance equation from the other parameters. Simulated groundwater inflow/outflow rates within the models were varied on a multi-week to seasonal basis. Groundwater flows were not adjusted daily to precisely match monitored water surface elevations, as doing so would likely not represent actual groundwater flow conditions.

Table 6-4 Water balance outlet rating curve and groundwater flow assumptions

Lake	Outlet Rating Curve Assumptions	Groundwater Flow Assumptions
Armstrong	• The outlet rating curve from the SWWD XPSWMM model was adjusted up to match the elevation of a highpoint surveyed in the channel upstream of the outlet pipes • Between May 13–23, 2022 an additional constant discharge of 0.65 cfs was needed to calibrate the water balance. It is unclear how or why this additional outflow occurred as a portion of the outflow occurred below the outlet invert elevation.	• Model year 2019 required no groundwater inflow or outflow to match the monitored water surface elevations • Model year 2022 groundwater outflows ranged from 0 – 0.15 cfs
Markgrafs	• The outlet rating curve from the SWWD XPSWMM model was referenced	• The monitored water surface elevations remain above the outlet control elevation for extended durations. High volumes of groundwater inflow were required for water balance calibration. • Model year 2019 groundwater inflows ranged from 1.6 – 1.8 cfs • Model year 2022 groundwater inflows ranged from 0.1 – 1.6 cfs
Powers	• Two pumps control the water surface elevations on Powers Lake, each with “pump on” and “pump off” elevations • The reported “pump on” and “pump off” elevations were used when practical; however, there appeared to be periods when pump operations varied from typical operating procedures. Pump operations were adjusted in the model to match monitored water surface elevations when applicable.	• Model year 2019 required no groundwater inflow or outflow to match the monitored water surface elevations • Model year 2022 groundwater outflows ranged from 0.2 – 1.0 cfs
North Wilmes	• The outlet rating curve from the SWWD XPSWMM model was referenced	• Model year 2019 required no groundwater inflow or outflow to match the monitored water surface elevations. • Model year 2022 groundwater outflows ranged from 0.1 – 0.5 cfs

Lake	Outlet Rating Curve Assumptions	Groundwater Flow Assumptions
South Wilmes	• The outlet rating curve from the SWWD XPSWMM model was referenced	• Model year 2019 groundwater outflow was 2.5 cfs • Model year 2022 groundwater outflow was 0.3 cfs
Colby	• The outlet rating curve from the SWWD XPSWMM model was referenced	• Model years 2019 and 2022 required no groundwater inflow or outflow to match the monitored water surface elevations
Bailey	• Up to five pumps control the water surface elevations on Bailey Lake, each with operating procedures for “pump on” and “pump off” elevations • The reported “pump on” and “pump off” elevations were used when practical; however, there appeared to be periods when pump operations varied from typical operating procedures. Pump operations were adjusted in the model to match monitored water surface elevations when applicable. • Available historical pump records were provided by City of Woodbury staff for reference, but no records after 2018 are available. The pump records demonstrated that pump operating procedures may vary.	• Model years 2019 and 2022 groundwater outflow was estimated at 0.4 cfs based on dry climatic conditions in 2022. • There is uncertainty associated with groundwater outflow assumptions in 2019 given that no pump records were available to confirm the pump outflow assumptions used in the model.
La	• Land-locked, no outlet rating curve modeled	• Model year 2019 groundwater inflows ranged from 0.1 – 0.2 cfs • Model year 2022 groundwater inflows ranged from 0.1 – 0.15 cfs
Ravine	• High uncertainty in the Ravine Lake water balance due to beaver dams frequently observed at the Ravine Lake outlet. While field notes provide some information on beaver dam observations, there are no detailed records available for model years 2019 and 2022. • The outlet rating curve from the SWWD XPSWMM model was adjusted up to the estimated top of a beaver dam based on the monitored water surface elevations	• Ravine Lake is known to have sizable groundwater inflow; however, there are no detailed monitoring records available on groundwater inflow rates. The observed water surface elevations and water quality monitoring data were critically assessed to make the best possible assumptions about groundwater inflow rates. • Model year 2019 groundwater inflows were estimated at 0.7 cfs • Model year 2022 groundwater inflows ranged from 0.3 – 0.7 cfs

6.2.2 Water Quality Parameters

Calibration of water quality parameters requires a process in which model parameters and coefficients are reasonably adjusted such that the model predictions are similar to in-lake measurements. The lakes that were modeled using the finite difference spreadsheet were calibrated to match monitored total phosphorus concentrations (Armstrong, Powers, Bailey, La). The lakes that were modeled using the Barr Shallow Lake model were calibrated to the following parameters when monitoring data was available (Markgrafs, South Wilmes, Colby, Ravine).

- Total Phosphorus (TP)
- Chlorophyll-*a*
- Total Kjeldahl Nitrogen (TKN)

Other parameters that are typically calibrated when using the Barr Shallow Model include orthophosphate and nitrate + nitrite concentrations. While these parameters were estimated in the models, calibrations could not be performed because no monitoring data were available.

Additionally, in-lake modeling performed for North Wilmes Lake as a part of this study is high level and has a high amount of relative uncertainty, given no monitoring data is currently available for the lake. Model results were verified for relative accuracy as part of calibrating downstream lake models. Water quality monitoring in North Wilmes will be necessary to develop accurate nutrient loading predictions.

Example in-lake model calibrations for South Wilmes Lake are provided below. The orange line in each plot represents the modeled in-lake concentrations, and the blue circles represent the monitored concentrations.

Plots showing all calibrated parameters for each suburban lake can be found in Appendix B.

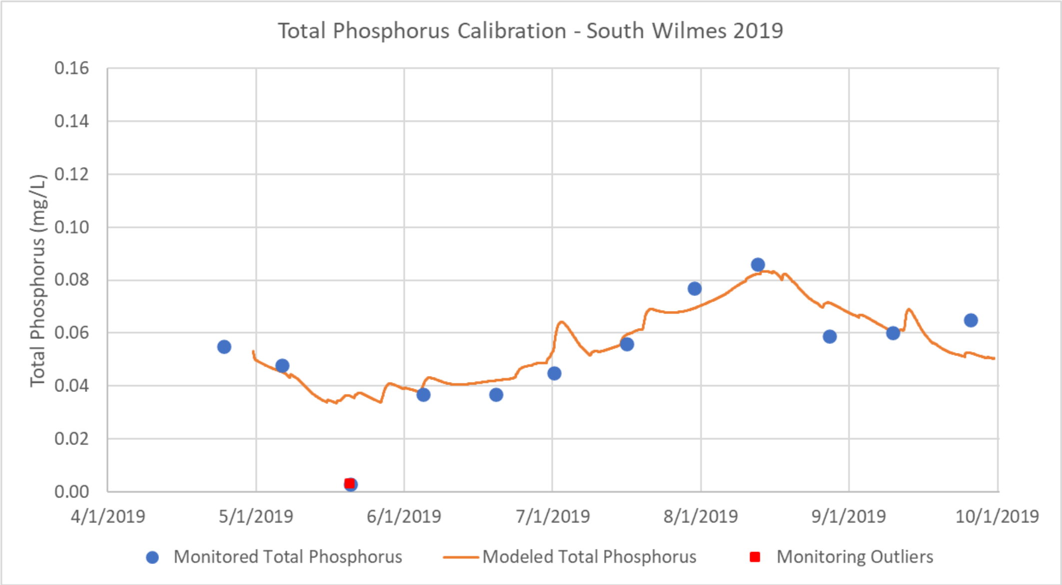


Figure 6-4 South Wilmes 2019 in-lake total phosphorus calibration

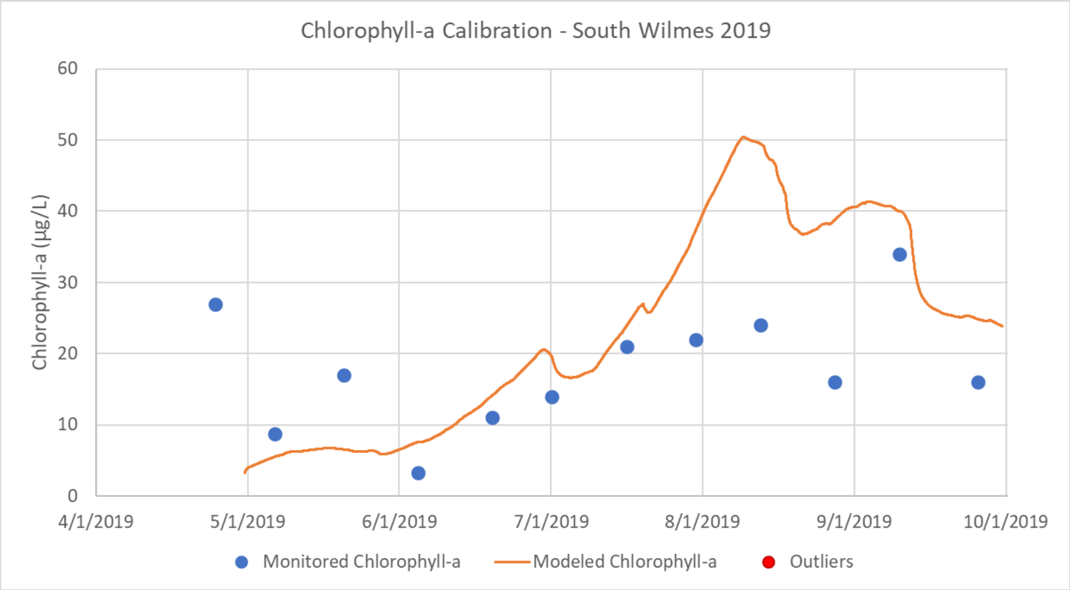


Figure 6-5 South Wilmes 2019 in-lake chlorophyll-*a* calibration

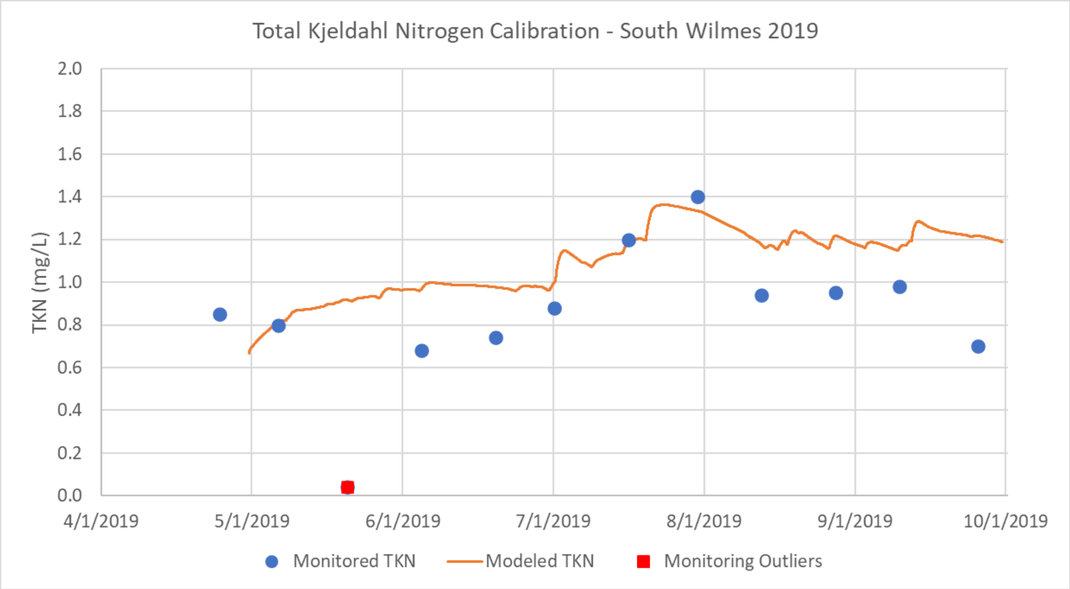


Figure 6-6 South Wilmes 2019 in-lake Total Kjeldahl Nitrogen calibration

6.3 Phosphorus Loading Summaries

After the in-lake water quality model calibrations were finalized, phosphorus loading summaries were developed. The following lake sections summarize the estimated 2019 and 2022 total phosphorus loads to each of the nine suburban lakes between June 1 – September 30 from the watershed (external loads), internal loading from lake bottom sediment (internal loads), and loading from upstream lakes and groundwater inflow, if applicable. Model assumptions and observations are also discussed. Table 6-5 provides a summary of the total phosphorus load quantities for each modeled lake for 2019 and 2022.

Table 6-5 Total phosphorus loading estimates for SWWD suburban lakes: June 1 – September 30

Lake	2019 Total Phosphorus Loads (lbs, June – Sept)				2022 Total Phosphorus Loads (lbs, June – Sept)			
	Watershed	Upstream Lake(s)	Internal Loading from Lake Bottom Sediment	Groundwater	Watershed	Upstream Lake(s)	Internal Loading from Lake Bottom Sediment	Groundwater
Armstrong	65	-	14	-	17	-	35	-
Markgrafs	58	-	135	26	14	-	50	4
Powers	259	-	171	-	73	-	70	-
North Wilmes[1]	209	41	36	-	49	0	33	-
South Wilmes	34	405	55	-	6	12	32	-
Colby	383	263	377	-	102	3	261	-
Bailey	78	603	110	-	17	15	146	-
La	3	-	34	7	<1	-	57	6
Ravine	80	-	67	12	37	-	127	9

[1] Water quality monitoring data has not historically been collected for North Wilmes Lake. As such, water quality model calibration could not be completed. Water quality modeling was based on high level assumptions, approximations from similar lake models, and efforts to calibrate downstream South Wilmes Lake. Water quality monitoring is recommended to better estimate the total phosphorus loads to North Wilmes Lake.

6.4 Proposed Management Practices Modeling

To help inform the District's planning for future management activities, the calibrated in-lake models were used to predict the effects of implementing best management practices on in-lake water quality. For the purpose of this exercise, the District was interested in understanding the expected impact of implementing sediment phosphorus inactivation projects to address internal loading. To do this, the calibrated models were modified as described below, as applicable, for each waterbody.

For each of the lakes, two separate sediment inactivation scenarios were created: one to represent a high efficacy treatment and the second to represent low efficacy. For lakes that the modeling showed additional nutrient reductions (beyond internal treatment) would be needed to decrease simulated phosphorus and chlorophyll-a concentrations to below the state standard, a reduction in watershed load was also simulated.

The results of the conceptual treatment strategy models can be viewed in each individual lake section (Sections 8 - 14).

<u>For lakes modeled with the Barr Shallow Lake Model, the following assumptions were made:</u>

- Scenario: Sediment Inactivation (low and high efficacy)
    - In the Barr shallow lake model, the two primary sediment P fractions are modeled separately: mobile-P and organic-P
    - For both the high and low efficacy scenarios, the mobile-P loading rate was decreased by 85% to represent impacts from a sediment treatment. This reduction is based on monitoring data from sediment inactivation projects that Barr has completed previously, one year after project completion.
    - The organic-P loading rate was decreased by 50% (low efficacy scenario) and 70% (high efficacy scenario) based on best professional judgement and results from the sediment core sampling. A range of efficacy was utilized for simulating impacts to the organic-P loading rate due to additional uncertainty regarding the effectiveness of sediment treatments in reducing organic driven internal loading, depending on project type (alum, alum/iron, or aeration).
- Scenario: Sediment Inactivation (low) + Watershed Load Reduction
    - For this scenario, the mobile-P and organic-P loading rates were decreased by 85% and 50% respectively.
    - A percent reduction was applied to the calibrated watershed loads in each model until the simulated in-lake summer average total phosphorus and chlorophyll-*a* concentrations met or were lower than state standards.

<u>For lakes modeled with the Finite Difference Spreadsheet Model, the following adjustments were made:</u>

- Scenario: Sediment Inactivation (low and high efficacy)
    - In the finite difference spreadsheet model, the sediment P fractions are not modeled separately. Because of this, for these scenarios the overall internal loading rate was

decreased by 60% (low efficacy) and 80% (high efficacy) based on monitoring data from sediment inactivation projects that Barr's completed previously one year after project completion. A range of efficacies was utilized to reflect some uncertainty in the effectiveness of sediment treatments in reducing organic driven internal loading, depending on project type (alum, alum/iron, or aeration).

The models developed for this study represent in-lake conditions and expected impacts from management strategies for a single growing season. Therefore, the estimated in-lake response to each of the conceptual treatment scenarios represent the predicted in-lake water quality improvements for that period only and do not account for longer-term water quality impacts. For example, once implemented sediment inactivation efficacy is known to vary over time. Monitoring data from Twin Cities metro lakes indicate that efficacy declines in the years following initial treatment due to product aging, new watershed inflows, and burial from rough fish activity, which can again increase internal loading over time. Conversely, if water quality management projects improve water clarity, some lakes may experience a greater abundance and extent of submerged plants. Increased plant growth can reduce sediment resuspension, enhance nutrient uptake, and create increased competition with algae, leading to improved water quality. If the increased plant growth is associated with aquatic invasive species, however, this can cause additional concerns for water quality and may warrant the consideration of additional management strategies related to vegetation management.

While modeling provides useful estimates of the expected scale of impacts of water quality management strategies, post-project monitoring is essential to track changes in lake water quality and adjust management plans as needed to continue benefits over time.

7 Armstrong Lake (South Basin)

7.1 Water Quality

Shallow/Deep	Shallow
Location	Oakdale
Surface Area	10 acres
Average/Maximum Depth	1 foot / 5.5 feet
Watershed Area	572 acres
Watershed:Lake Surface Area	57:1
Impairment Status	Not listed on impaired waters list
Downstream Waterbody	North Wilmes Lake

Armstrong Lake is the most upstream of the multi-lake system that was included in this study. The lake is divided into two separate basins, connected by a 36" culvert extending under County Highway 10. The north basin is located in the City of Lake Elmo, and the south basin is in the City of Oakdale. The lake is used primarily for wildlife viewing. When water levels are high enough, water discharges via gravity flow from Armstrong Lake through a storm sewer system, which ultimately discharges to North Wilmes Lake. Only the south basin of Armstrong Lake was considered for this study. This basin is where the lake's monitoring data is being collected and more field data is available for this area of the lake. The north basin is represented in the watershed p8 model, but an in-lake model was not developed for the north basin.

The south basin of Armstrong Lake has a water surface area of approximately 10 acres, a maximum depth of 5.5 feet, and a mean depth of approximately 1 foot. Armstrong Lake is shallow enough for aquatic plants to grow over the entire waterbody and for the lake to mix many times per year (polymictic lake).

The State of Minnesota uses three water quality parameters to assess eutrophication standards within lakes—total phosphorus, chlorophyll-*a*, and Secchi disk transparency—to assess waterbody health and track water quality changes. These three parameters were measured in Armstrong Lake by the SWWD between 1998-2023 (Figure 7-1). Monitoring data collected in 2014 and between 2021 – 2023 was collected at a reduced frequency from other monitored years. Not enough measurements were collected to calculate summer averages for those years. During the monitored years with adequate data, summer average total phosphorus concentrations were better than the state standard between 2004 – 2005, 2010 - 2011, and in 2016. All other monitored years were worse than the state standard summer average concentration. Summer average chlorophyll-*a* concentrations were better than the state standard between 2001 – 2006, 2008 – 2018, and in 2020. Summer average Secchi depths met or were slightly better than the state standard in 2002, 2004-2005, 2008-2009, and 2011-2013. Since 2014, summer average Secchi depths have been worse than the state standard. At the time of this study, Armstrong Lake was not listed as impaired on the Minnesota impaired waters list.

Chloride concentrations were measured by the SWWD between 2002 - 2007 and 2010-2023 (generally between April and September). In the historical record, one observed chloride concentration in October 2023 exceeded the MPCA chronic standard of 230 mg/L. All other observed chloride concentrations were below the chronic standard. Chloride concentrations within Armstrong Lake have been on a steady rise over the past 21 years; with average annual observed concentrations increasing from 54 mg/L to 202 mg/L between 2002 and 2023.

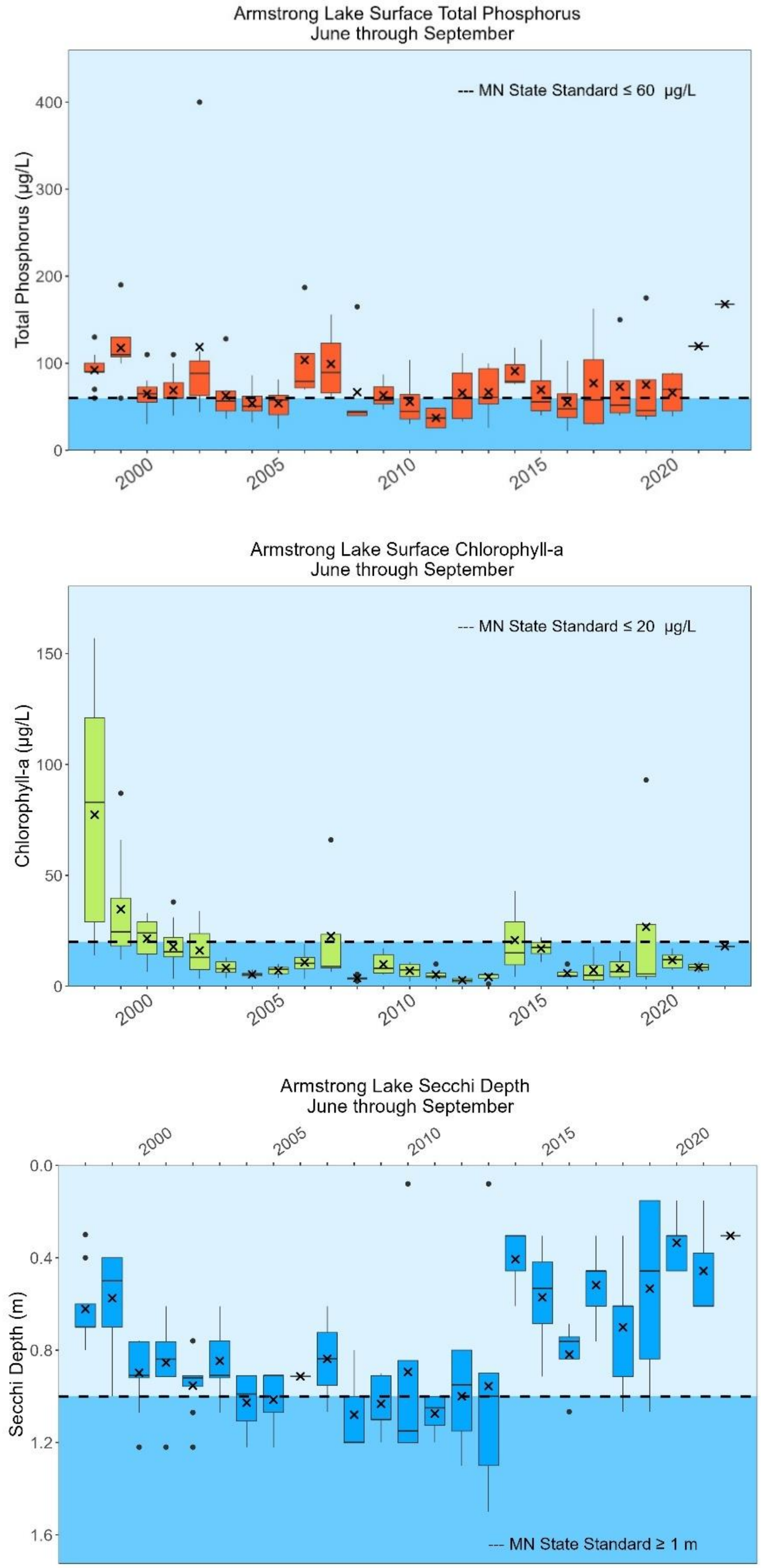


Figure 7-1 Armstrong Lake eutrophication monitoring data (June – September)
Summer averages are shown by x's in the box plots
**Notably high total phosphorus and chlorophyll-a outliers in 2000 are not shown on the plots*

7.2 Ecological Health

7.2.1 Aquatic Plants

Table 7-1 summarizes the calculated Lake Plant Eutrophication IBI values for Armstrong Lake based on point-intercept plant surveys completed in 2021 and 2024. Armstrong Lake scored above both plant IBI metrics in June 2021, June 2024, and August 2024. Water levels were too low in August 2021 to complete a survey. Table 7-1 also summarizes the percentage of the littoral area where aquatic plants were found and lists observed aquatic invasive species, their frequency of occurrence (FOO), and current management practices as applicable. Other plant health metrics can be referenced in the 2021 and 2024 Aquatic Vegetation Survey Results reports (Stantec, 2021, 2025).

Table 7-1 Armstrong Lake aquatic plants overview

Lake	Parameter	June 2021[1]	August 2021[1]	June 2024[2]	August 2024[2]	MnDNR Threshold	Invasive Species Management
Armstrong	Species Richness	14	-	14	14	>11	
	Floristic Quality Index (FQI)	18.2	-	20	19.4	>17.8	
	% Littoral with Vegetation	100%	-	100%	100%		
	Curly-leaf Pondweed FOO	6%	-	2%	-		No management to date
	Purple Loosestrife FOO	P	-	-	-		Managed naturally by leaf eating beetles

[1] (Stantec, 2021 Aquatic Vegetation Survey Results, 2021)
[2] (Stantec, 2025)

7.2.2 Fisheries

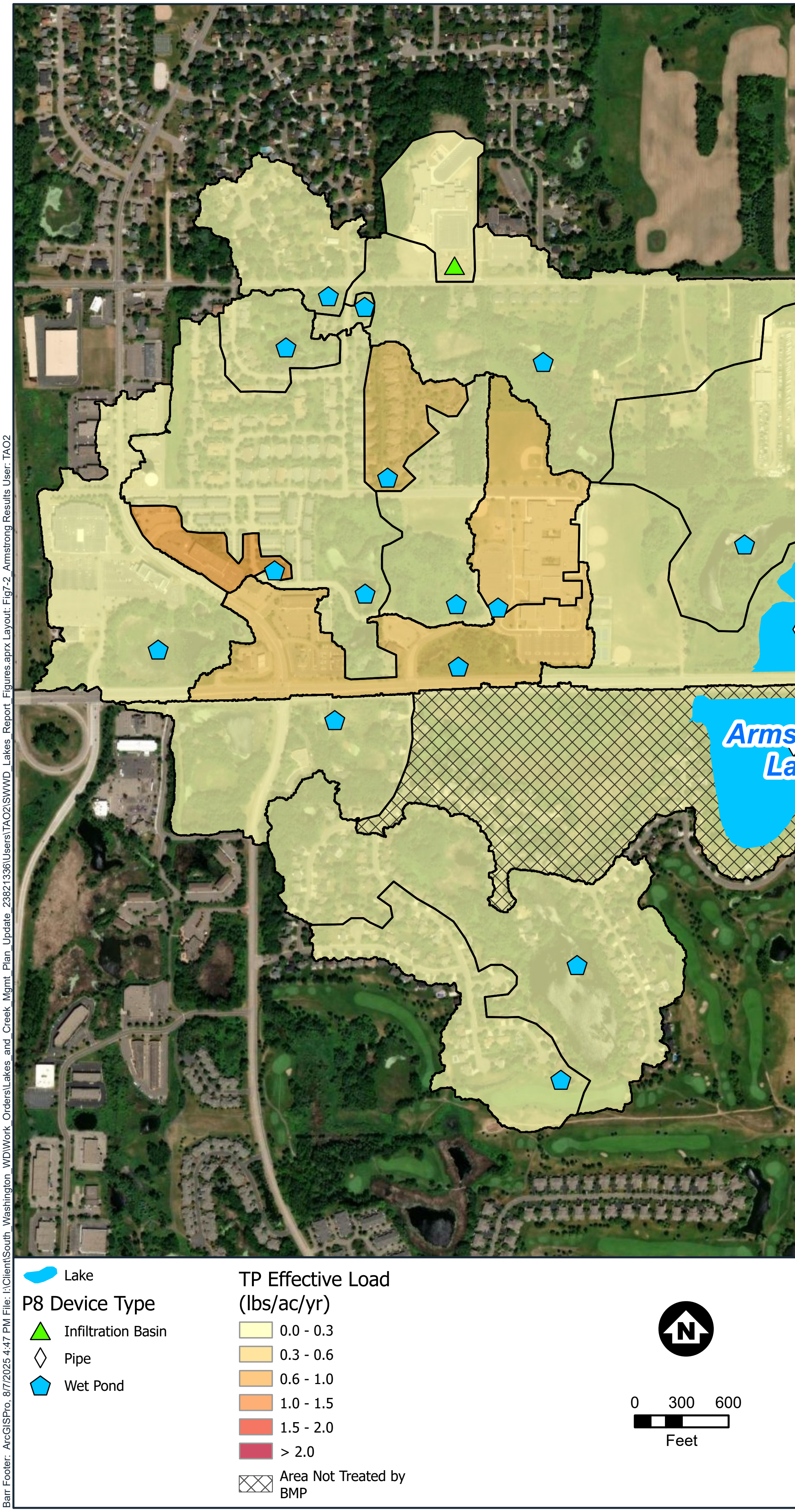
There are no MNDNR fish survey or stocking data available for Armstrong Lake.

7.3 Watershed Total Phosphorus Loads

The P8 model was run for the water years 2019 and 2022 for the purposes of developing the in-lake models. To estimate 10-year average total phosphorus loads to each lake, the P8 models were run from 2012 – 2022.

P8 modeling shows low to moderate watershed total phosphorus loads to Armstrong Lake, with higher loads coming from denser residential areas (e.g., multi-family housing), streets, and commercial areas within the watershed. The map in Figure 7-2 shows the effective areal phosphorus loading by subwatershed, modeled BMPs, and untreated subwatersheds in the Armstrong Lake contributing area. The effective phosphorus load represents the loading rate after pollutant removal by BMPs within the watershed and is reflective of the loading that actually makes it into the receiving waterbody from a given location. The effective phosphorus loads for Armstrong Lake subwatersheds range from 0.0 pounds per acre per year to 0.7 pounds per acre per year, based on the 10-year modeling period (2012-2022).

The direct drainage area to the southern basin of Armstrong Lake does not receive treatment before entering the lake. For the purposes of the P8 modeling, the large wetland west of the southern basin was not modeled as a pond device and is therefore not simulated as providing pollutant removal.



Armstrong Lake
Effective TP Loading
Lake Management Plan
South Washington
Watershed District

FIGURE 7-2

BARR

7.4 In-lake Total Phosphorus Loads

Results of the in-lake modeling for Armstrong Lake showed that during the summer of 2019 (representing a wet year), phosphorus loading into the lake was dominated by loading from the tributary watershed, representing 82% (65 lbs) of the total phosphorus loading from major sources (Figure 7-3). Internal loading that same year was estimated to be 18% (14 lbs) of the total phosphorus load. During the drier summer of 2022, the estimated phosphorus loading into the lake from watershed runoff was notably lower, representing 33% (17 lbs) of total phosphorus loading to the lake. Internal loading in 2022 was notably higher than 2019, representing 67% (35 lbs) of the phosphorus load.

Unfortunately, water quality monitoring data in Armstrong Lake was limited in 2022 (i.e., monitoring data only available April – June and October). As such, Barr can only postulate potential reasons why internal loading was markedly different between 2019 and 2022. Possible reasons for higher internal loading from lake bottom sediment in 2022 could include:

- Longer periods of stronger lake stratification may have led to lower dissolved oxygen concentrations over larger portions of the lake bottom, resulting in phosphorus release from the mobile-phosphorus fraction from larger areas of sediment.
- Changes in the extent and depth of lake mixing.
- Warmer lake temperatures may have led to increased microbial activity, which could have resulted in lower dissolved oxygen levels at the lake bottom and/or enhanced mineralization of organic phosphorus in the sediment.

Phosphorus fractionation data gathered from sediment cores collected in 2024 indicated a moderate potential for internal phosphorus loading. Additional details on the sediment core data can be found in Section 4.8.

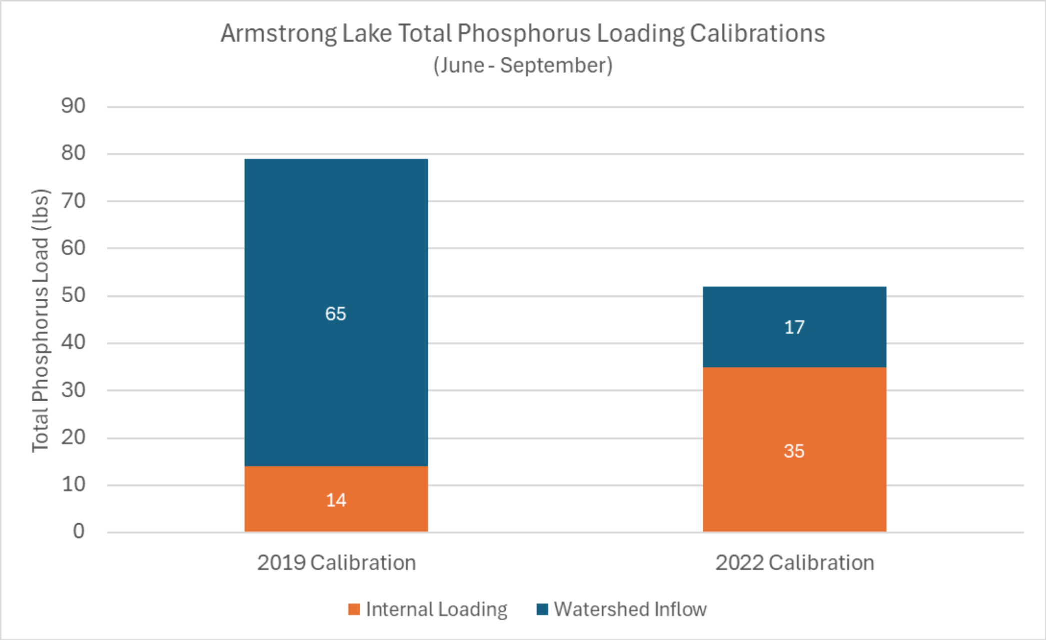


Figure 7-3 Armstrong Lake total phosphorus load estimates – 2019 & 2022

7.5 Management Recommendations

Monitoring of Armstrong Lake between 2017 - 2020 showed that total phosphorus and Secchi depth were worse than state standards. Limited water quality data was collected between 2021 – 2023. The aquatic plant community in June 2021 was relatively diverse with high quality. However, the aquatic invasive species, curly-leaf pondweed, was observed and could be a threat to the native plant community. Given this, future management efforts should focus on improving lake water quality and ecosystem health, monitoring for changes, and continuing water quality and ecosystem health protection measures as improvements are achieved. Table 7-2 summarizes the recommended management strategies that could be considered for Armstrong Lake to help improve water quality conditions and ecological health.

Table 7-2 Armstrong Lake management recommendations

Management/Protection Action		Basis
Address external watershed loads	Enhanced street sweeping program	Consider partnering with the cities of Lake Elmo and Oakdale to implement enhanced street sweeping programs to reduce pollutant loading to stormwater runoff
	Stormwater BMPs	Consider retrofitting or installing new stormwater BMPs in subwatersheds that are currently untreated or undertreated Implement site scale BMPs as opportunities arise
	Chloride	Consider applying chloride reduction strategies such as education and implementation assistance to member cities and other stakeholders
Aquatic Plants	Invasive species management	Continue to monitor invasive species growth (e.g., curly-leaf pondweed) and consider management if the invasive species starts to negatively impact water quality or native plant densities and distribution
	Promote native aquatic plant growth	Encourage native plant reestablishment to promote clear water conditions and competition with algae
Fisheries	Fisheries Monitoring	Consider collecting fish community data
Phytoplankton and Zooplankton	Phytoplankton/Zooplankton Monitoring	Consider monitoring phytoplankton and zooplankton as part of routine monitoring
Water Quality	Water Quality Monitoring	Continue monitoring in-lake water quality and assessing for eutrophication Continue to identify/track chloride levels from winter salt use

8 Markgrafs Lake

8.1 Water Quality

Shallow/Deep	Shallow
Location	Woodbury
Surface Area	44 acres
Average/Maximum Depth	3 feet / 7 feet
Watershed Area	425 acres
Watershed:Lake Surface Area	10:1
Impairment Status	Impaired for nutrients since 2006
Downstream Waterbody	South Wilmes Lake

Markgrafs Lake is located in the City of Woodbury and is used for active and passive recreation and wildlife viewing. Markgrafs Lake is one of the upstream lakes in the multi-lake system included in this study. When water levels are high enough, water discharges via gravity flow from Markgrafs Lake through a storm sewer system, which ultimately discharges to South Wilmes Lake. During extreme storm events (e.g., 100-year flow events), water can be pumped to Powers Lake. Lake level monitoring data indicates that water levels on Markgrafs Lake can sit above the control outlet elevation for extended periods of time due to high volumes of groundwater inflow.

Markgrafs Lake has a water surface area of approximately 44 acres, a maximum depth of 7 feet, and a mean depth of approximately 3 feet. Markgrafs Lake is shallow enough for aquatic plants to grow over the entire waterbody and for the lake to mix many times per year (polymictic lake).

The State of Minnesota uses three water quality parameters to assess eutrophication standards within lakes—total phosphorus, chlorophyll-*a*, and Secchi disk transparency—to assess waterbody health and track water quality changes. These three parameters were measured in Markgrafs Lake by the SWWD between 1994-2023 (Figure 8-1). During the monitored years, summer average total phosphorus concentrations have consistently exceeded the state standard, ranging between 69 – 241 µg/L. Summer average chlorophyll-*a* concentrations were better than the state standard in 1995 and 1996. Otherwise, summer average chlorophyll-*a* concentrations have exceeded the state standard, ranging between 23 - 173 µg/L. Summer average Secchi depths met or were slightly better than the state standard between 1994 – 1996, 2000, 2014, 2019, and 2022-2023. During all other monitored years, summer average Secchi depths have been worse than the state standard. Markgrafs Lake was added to the Minnesota impaired waters list in 2006 as impaired for nutrients.

Chloride concentrations were measured by the SWWD between 2013-2023 (generally between April and September). In the historical record, three observed chloride concentrations in monitoring year 2023 exceeded the MPCA chronic standard of 230 mg/L ranging between 235-248 mg/L. All other observed chloride concentrations were below the chronic standard.

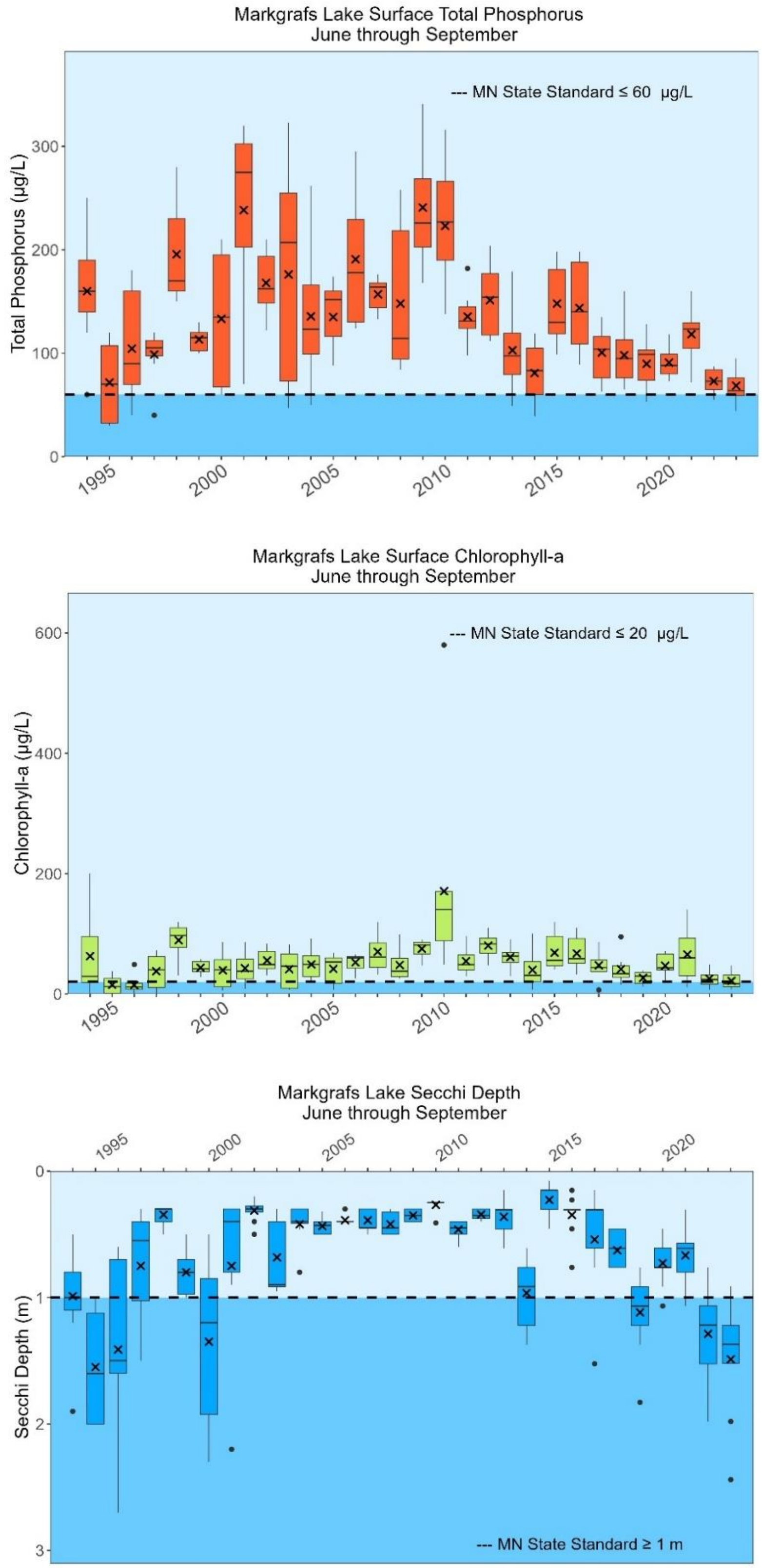


Figure 8-1 **Markgrafs Lake eutrophication monitoring data (June – September)**
Summer averages are shown by x's in the box plots

8.2 Ecological Health

8.2.1 Aquatic Plants

Table 8-1 summarizes the calculated Lake Plant Eutrophication IBI values for Markgrafs Lake based on point-intercept plant surveys completed in 2021 and 2024. Markgrafs Lake scored below both plant IBI metrics in June and August in 2021 and 2024, indicating a degraded plant community that is likely stressed from cultural eutrophication. Table 8-1 also summarizes the percentage of the littoral area where aquatic plants were found and lists observed aquatic invasive species, their frequency of occurrence (FOO), and current management practices. Other plant health metrics can be referenced in the 2021 and 2024 Aquatic Vegetation Survey Results reports (Stantec, 2021, 2025).

Table 8-1 Markgrafs Lake aquatic plants overview

Lake	Parameter	June 2021[1]	August 2021[1]	June 2024[2]	August 2024[2]	MnDNR Threshold	Invasive Species Management
Markgrafs	Species Richness	7	10	7	8	>11	
	Floristic Quality Index (FQI)	11.9	15.4	8.25	10.4	>17.8	
	% Littoral with Vegetation	94%	73%	94%	86%		
	Curly-leaf Pondweed FOO	10%	2%	14%	-		Herbicide applications in 2023 & 2025

[1] (Stantec, 2021 Aquatic Vegetation Survey Results, 2021)
[2] (Stantec, 2025)

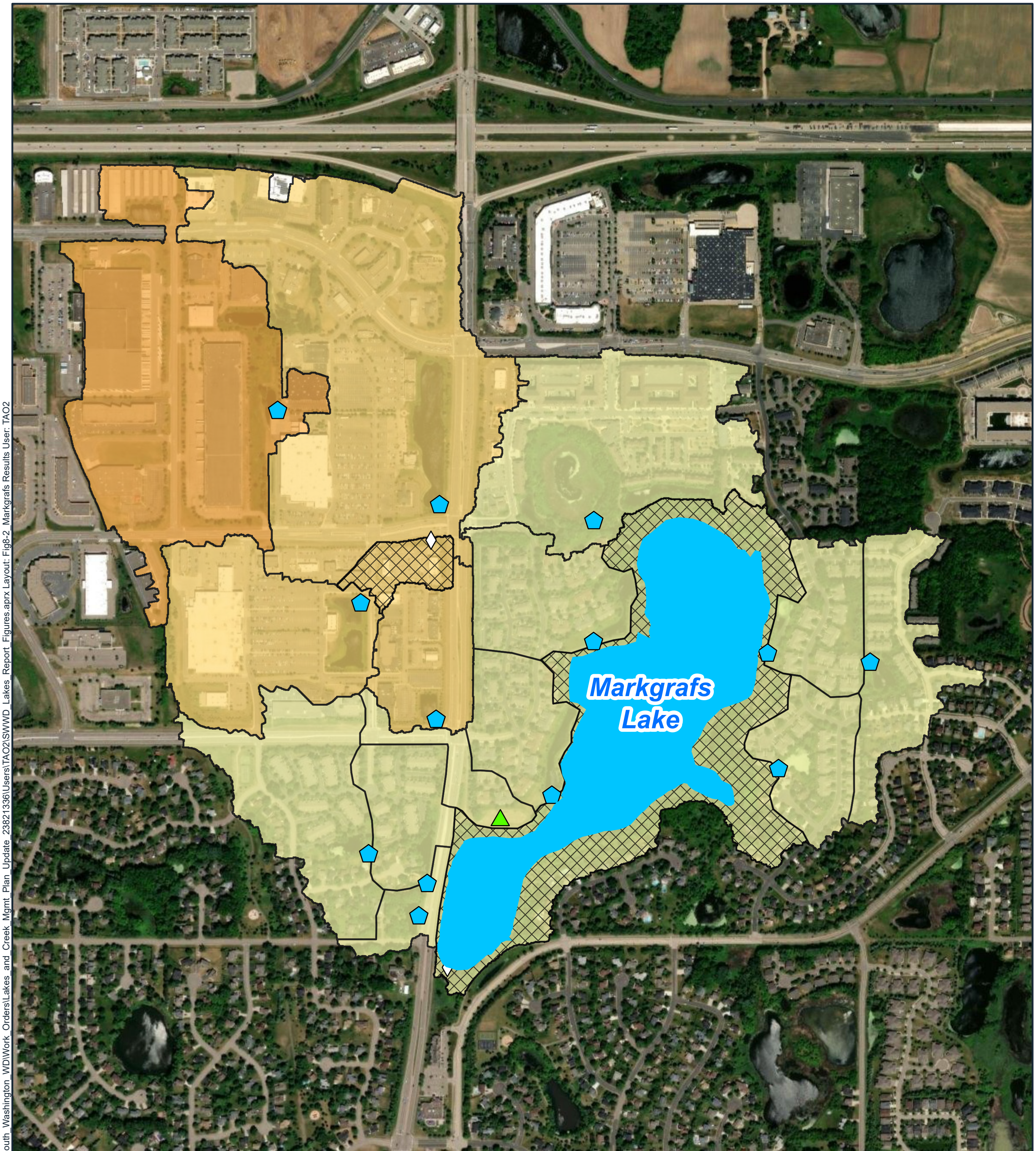
8.2.2 Fisheries

While Markgrafs Lake has historically been used as a walleye rearing waterbody, the absence of recent fish survey data prevents an assessment of current walleye abundance. There are no MNDNR fish survey or stocking data available for Markgrafs Lake.

8.3 Watershed Total Phosphorus Loads

P8 modeling shows low to moderate watershed phosphorus loading to Markgrafs Lake, with higher loads coming from the western portion of the watershed where there are large commercial properties (e.g., Sam's Club, Target), some of which are untreated by BMPs before discharging to Markgrafs Lake. Figure 8-2 shows the effective areal phosphorus load by subwatershed, the modeled BMPs, and untreated subwatersheds. The effective phosphorus load represents the loading rate after pollutant removal by BMPs throughout the watershed and is reflective of the loading that actually makes it into the receiving waterbody from a given location. The effective phosphorus loads for the Markgrafs Lake subwatersheds range from 0.09 pounds per acre per year to 0.6 pounds per acre per year, based on a 10-year modeling period (2012-2022).

The direct drainage area to Markgrafs Lake receives treatment from several small stormwater (sedimentation) ponds around the lake. These basins were added to the P8 model to better represent loading to the lake. However, land directly adjacent to Markgrafs Lake does not receive any treatment before runoff discharges to the lake.



Lake

P8 Device Type
- Pipe
- Dry Pond
- General Device
- Wet Pond
- Infiltration Basin

Effective TP Load (lbs/ac/yr)
- 0.0 - 0.3
- 0.3 - 0.6
- 0.6 - 1.0
- 1.0 - 1.5
- 1.5 - 2.0
- > 2.0
- Area Not Treated by BMP

N

0 300 600
Feet

Markgrafs Lake
Effective TP Loading
Lake Management Plan
South Washington Watershed District

FIGURE 8-2

BARR

8.4 In-lake Total Phosphorus Loads

Results of the in-lake modeling for Markgrafs Lake showed that during both wet (2019) and dry (2022) summers, the phosphorus loading into Markgrafs Lake was dominated by internal loading, representing 62% (135 lbs) of the phosphorus load in 2019 and 74% (50 lbs) in 2022 (Figure 8-3). Model calibration indicates that internal phosphorus loading was notably higher in 2019 than in 2022. Sediment cores collected in 2024 showed that the average concentration of organically bound phosphorus in the top four to six centimeters of sediment was higher than the mobile phosphorus fraction. It is the mobile phosphorus fraction that can be released from sediment during low oxygen conditions. Organically bound phosphorus also releases from lake sediment, but typically at a slower rate than mobile phosphorus, and the release rate is generally controlled by lake water temperature or other environmental parameters that may influence microbial activity. The release rate from the organically bound phosphorus fraction was decreased in the 2022 in-lake model to match monitored in-lake conditions. It's possible that environmental factors influenced the microbial degradation rates of the organic phosphorus fraction in 2022.

The higher estimated groundwater influence on Markgrafs Lake also contributes to uncertainty in the internal loading estimates. Limited groundwater water quality monitoring data is available from wells in the vicinity of Markgrafs Lake. For the purposes of this study, a total phosphorus groundwater concentration of 23 µg/L was assumed based on the 10-year average annual monitoring data available from well 798057 (National Water Quality Monitoring Council, 2025). If the assumed groundwater total phosphorus concentration was too low during the modeled periods, then the internal load is overestimated in the lake calibrations. Conversely, if the assumed groundwater concentration was too high, then the internal load is underestimated in the lake calibrations. Since groundwater inflow is a major component of the Markgrafs Lake water balance, more detailed groundwater monitoring is recommended to confirm the extent of groundwater inflow impacts on in-lake phosphorus concentrations. Assuming a groundwater inflow total phosphorus concentration of 23 µg/L, the total phosphorus load from groundwater in 2019 and 2022 represented 12% (26 lbs) and 6% (4 lbs) of the total phosphorus load, respectively.

The total phosphorus load from watershed runoff was also a notable contribution to Markgrafs Lake, especially during the summer of 2019 (representing a wet year) where watershed runoff represented 26% (58 lbs) of the total phosphorus load. During the drier summer of 2022, the watershed runoff total phosphorus load represented 21% (14 lbs) of the total load.

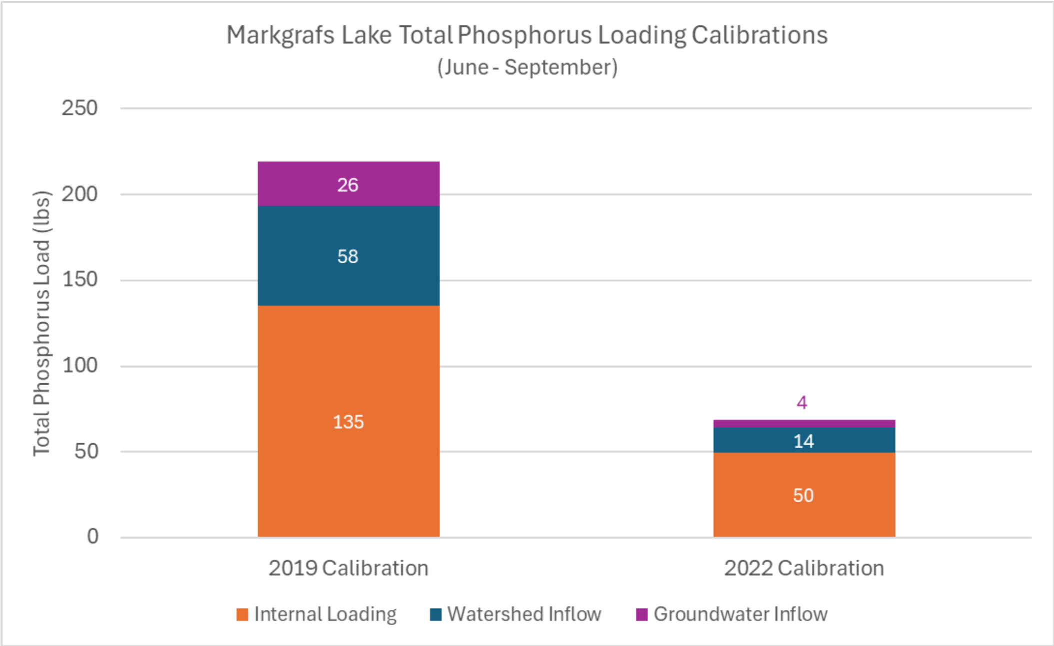


Figure 8-3 Markgrafs Lake total phosphorus load estimates – 2019 & 2022

8.5 Predicted Benefits from Implementing Management Practices

One of the study goals was to estimate the potential impact of lake sediment inactivation projects and how they could be used to improve in-lake water quality. The calibrated in-lake models were used to estimate the impact of implementing a sediment treatment project in Markgrafs Lake.

Table 8-2 summarizes the estimated phosphorus load reduction from two scenarios: sediment inactivation (low efficacy) and sediment inactivation (high efficacy). Results from the in-lake modeling predict a reduction of 32 – 100 pounds of total phosphorus loading into Markgrafs Lake through implementation of sediment inactivation practices for a wet (2019) and dry (2022) year. This reduction in phosphorus loading translates to a reduction in the in-lake total phosphorus concentrations. Model results estimate that the 2019 summer average total phosphorus concentration of 95 µg/L would reduce to 49 - 59 µg/L with the implementation of sediment treatments; and the 2022 summer average total phosphorus concentration of 74 µg/L would reduce to 41 - 48 µg/L (Table 8-2). Figure 8-4 and Figure 8-5 summarize the predicted reductions in in-lake total phosphorus concentrations for the 2019 and 2022 model years, respectively, for each of the sediment inactivation scenarios.

Table 8-2 Summer growing period estimated reductions in phosphorus loads and in-lake concentrations in Markgrafs Lake with the proposed management practices

Scenario	Modeled Summer Total Phosphorus Loading (lbs)					Summer Average Total Phosphorus (µg/L)[1]	Summer Average Chlorophyll-a (µg/L)[1]
	Watershed Inflow	Groundwater Inflow	Internal Loading	Total	Load Reduction		
2019 Existing	58	26	135	219	-	95	26
2019 Sediment Inactivation Low	58	26	56	140	-79	59	19
2019 Sediment Inactivation High	58	26	36	120	-100	49	17
2022 Existing	14	4	50	68	-	74	28
2022 Sediment Inactivation Low	14	4	18	36	-32	48	15
2022 Sediment Inactivation High	14	4	12	31	-38	41	14

[1] Existing conditions summer average concentrations are calculated from calibrated in-lake models

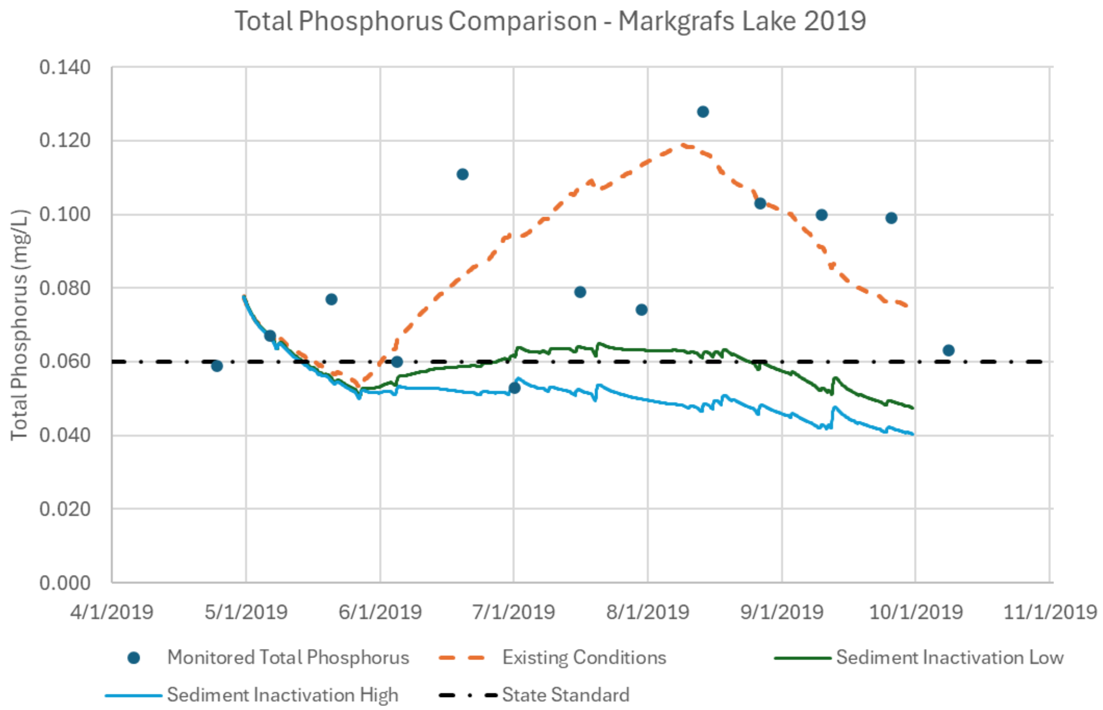


Figure 8-4 **Model-predicted reductions in 2019 total phosphorus concentrations in Markgrafs Lake with the proposed management practices**

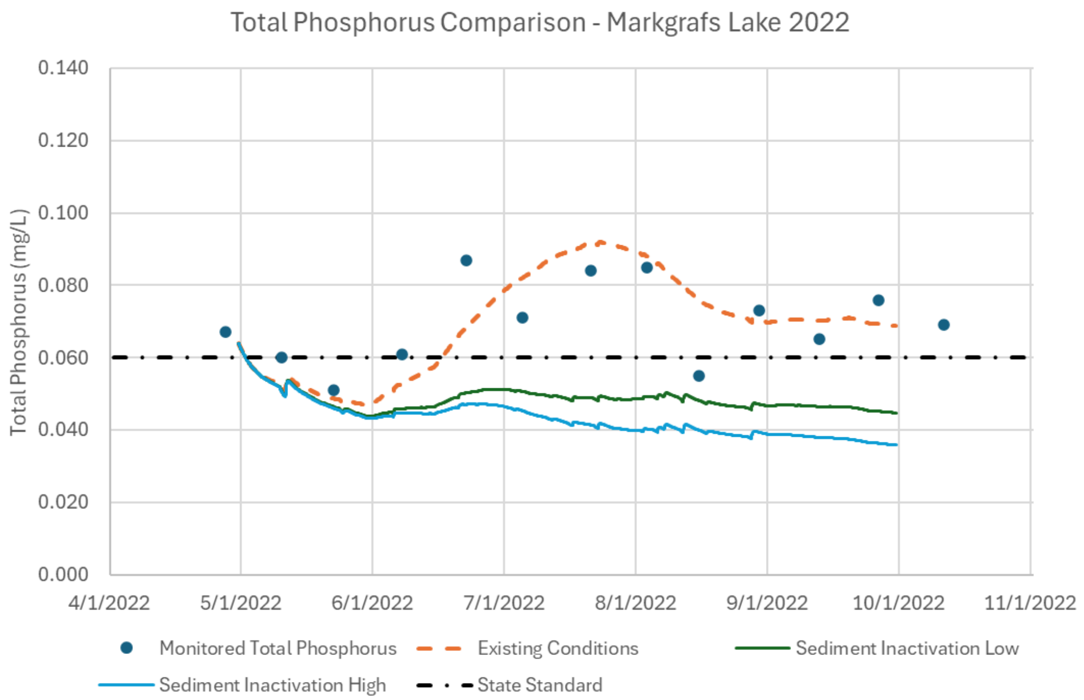


Figure 8-5 **Model-predicted reductions in 2022 total phosphorus concentrations in Markgrafs Lake with the proposed management practices**

Based on these results, it's predicted that implementing sediment inactivation practices within Markgrafs Lake could be sufficient to reduce summer average in-lake concentrations for total phosphorus and chlorophyll-*a* to below the state standards. As noted earlier, the in-lake model results represent the expected lake response to a reduction in internal phosphorus loading from lake bottom sediments during the 2019 and 2022 growing seasons if those management practices had been in place. The various model scenarios incorporate a range of assumed sediment-treatment efficacies to account for application variability and the fact that the specific treatment approach (alum, alum/iron, or aeration) has not yet been selected. Assumed efficacies were informed by monitoring data from comparable Barr sediment-inactivation projects one year post-implementation.

Monitoring of past projects shows that sediment inactivation efficacy can vary and is expected to decline over time due to product aging, new watershed inflows, and/or burial from rough fish activity. As such, results presented in this report provide an estimate of the expected scale of treatment impacts within a given growing season, but do not account for longer-term changes within the system and how that may impact benefits over time. For more information on the sediment inactivation modeling assumptions see Section 6.4.

8.6 Management Recommendations

Monitoring of Markgrafs Lake indicates degraded water quality and shows that the lake has not met water quality standards for most of the historical record. Aquatic plant monitoring from 2021, indicates a degraded plant community that is likely stressed from cultural eutrophication and threatened by the aquatic invasive species, curly-leaf pondweed. Given this, future management efforts should focus on improving lake water quality and ecosystem health, monitoring for changes, and continuing water quality and ecosystem health protection measures as improvements are achieved. Table 8-3 summarizes the recommended management strategies that could be considered for Markgrafs Lake to help improve water quality conditions and ecological health.

Table 8-3 Markgrafs Lake management recommendations

Management/Protection Action		Basis
Address external watershed loads	Enhanced street sweeping program	Continue to work with the City of Woodbury to refine an enhanced street sweeping program to reduce pollutant loading to stormwater runoff
	Stormwater BMPs	Consider retrofitting or installing new stormwater BMPs in subwatersheds that are currently untreated or undertreated Implement site scale BMPs as opportunities arise
	Chloride	Consider applying chloride reduction strategies such as education and implementation assistance to the City of Woodbury and other stakeholders.
Address internal loads	Sediment inactivation treatment	Review and implement a sediment inactivation treatment to reduce lake bottom sediment phosphorus loads
Aquatic Plants	Invasive species management	Continue to monitor invasive species growth (e.g., curly-leaf pondweed) and continue management as needed
	Promote native aquatic plant growth	Encourage native plant reestablishment to promote clear water conditions and competition with algae
Fisheries	Fisheries Monitoring	Consider collecting fish community data
Phytoplankton and Zooplankton	Phytoplankton/Zooplankton Monitoring	Consider monitoring phytoplankton and zooplankton as part of routine monitoring
Water Quality	Water quality monitoring	Continue monitoring in-lake water quality and assessing for eutrophication Continue to identify/track chloride levels from winter salt use Collect additional information on groundwater contributions into the lake (water quality, groundwater-surface water interactions) to confirm impacts

9 Powers Lake

9.1 Water Quality

Shallow/Deep	Deep
Location	Woodbury
Surface Area	61 acres
Average/Maximum Depth	19 feet / 36 feet
Watershed Area	1,263 acres
Watershed:Lake Surface Area	21:1
Impairment Status	Not listed on impaired waters list
Downstream Waterbody	South Wilmes Lake

Powers Lake is located in the city of Woodbury and is used for active and passive recreation and wildlife viewing. Powers Lake is one of the upstream lakes in the multi-lake system included in this study. When water levels are high enough, water discharges via a pumped outlet system from Powers Lake through a storm sewer system, which ultimately discharges to South Wilmes Lake.

Powers Lake has a water surface area of approximately 61 acres, a maximum depth of 36 feet, and a mean depth of approximately 19 feet. Powers Lake is deep enough that aquatic plants will only grow in the photic zone of the lake and the lake will typically only mix to full depth twice per year (dimictic lake).

The State of Minnesota uses three water quality parameters to assess eutrophication standards within lakes—total phosphorus, chlorophyll-*a*, and Secchi disk transparency—to assess waterbody health and track water quality changes. These three parameters were measured in Powers Lake by the SWWD between 1994-2023 (Figure 9-1). Monitoring data collected in 2000 and 2001 was collected at a reduced frequency from other monitored years. Not enough measurements were collected to calculate summer averages for those years. During the monitored years with adequate data, summer average total phosphorus concentrations were better than the state standard between 1994-2004, 2008-2009, 2012-2021, and 2023. Summer average chlorophyll-*a* concentrations were better than the state standard between 1994-1996, 2008, 2012-2013, 2018-2019, and 2021-2023. Otherwise, summer average chlorophyll-*a* concentrations have exceeded the state standard, ranging between 15 - 35 µg/L. Summer average Secchi depths met or were better than the state standard for all monitored years except 2005. At the time of this study, Powers Lake was not listed on the Minnesota impaired waters list.

Total phosphorus concentrations in the hypolimnion (deep waters) of Powers Lake can be notably higher than the concentrations observed in the epilimnion (surface waters) due to strong stratification during the summer months. For example, the 2019 summer average total phosphorus concentrations observed in the epilimnion was 32 µg/L as compared to a summer average of 712 µg/L in the hypolimnion. Phosphorus that is released from lake bottom sediment during this time of the year can remain largely confined in the hypolimnion due to strong lake stratification and reduced lake mixing.

Chloride concentrations were measured by the SWWD between 2003-2023 (generally between April and September) at various depths in Powers Lake. In the historical record, all observed chloride concentrations were below the MPCA chronic standard of 230 mg/L. Observed chloride concentrations in the epilimnion ranged from 11 – 94 mg/L throughout the monitoring period. Observed chloride concentrations in the hypolimnion ranged from 38 – 209 mg/L. However, monitoring year 2019 was an outlier and had notably higher hypolimnion chloride concentrations than any other monitored year, ranging from 55 – 209 mg/L. Most of the observed hypolimnion chloride concentrations are less than 60 mg/L.

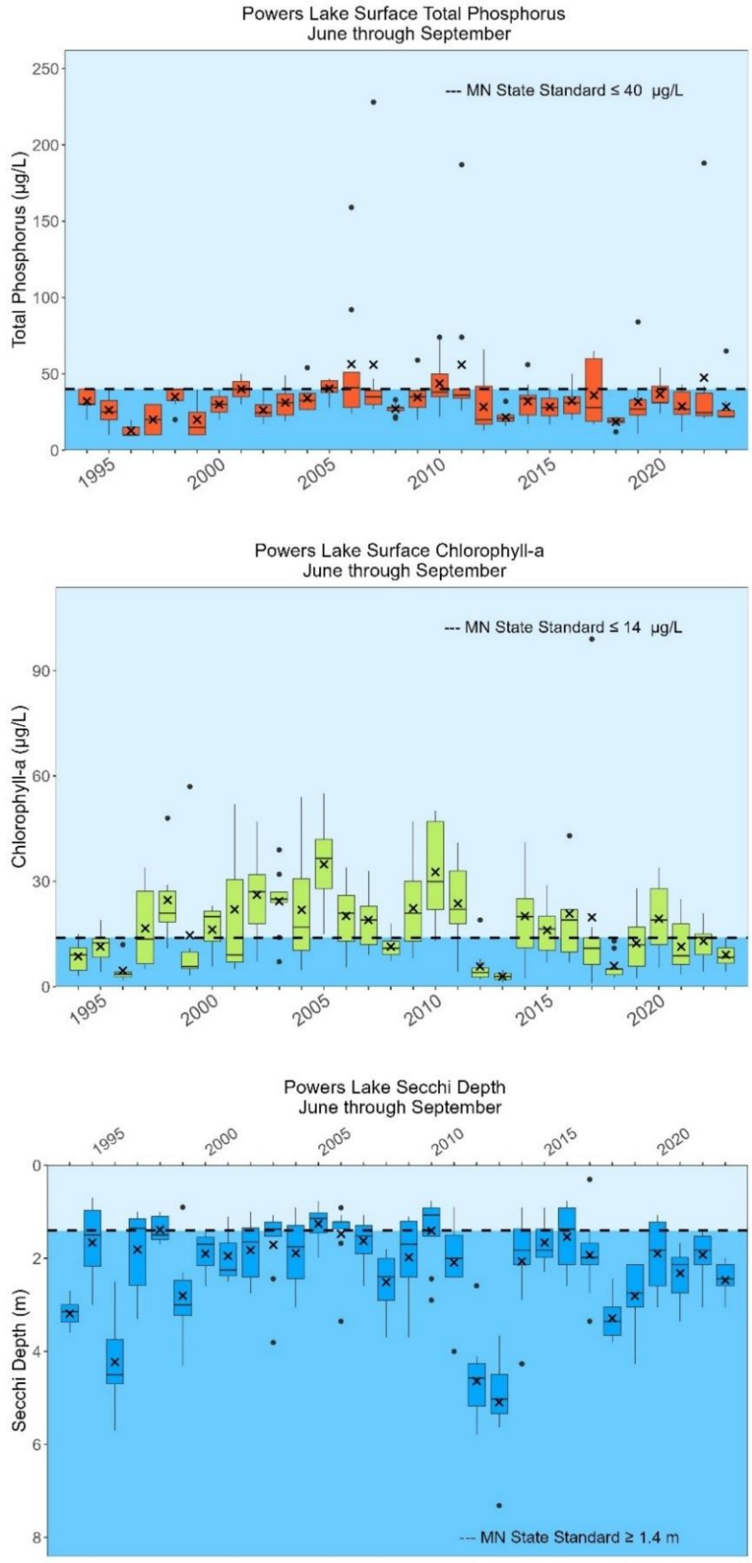


Figure 9-1 **Powers Lake eutrophication monitoring data (June-September)**
Summer averages are shown by x's in the box plots

9.2 Ecological Health

9.2.1 Aquatic Plants

Table 9-1 summarizes the calculated Lake Plant Eutrophication IBI values for Powers Lake based on point-intercept plant surveys completed in 2021 and 2024. Powers Lake scored below both plant IBI metrics in June 2021 and August 2024. However, in August 2021, Powers Lake scored just above both plant IBI metrics. In June 2024, Powers Lake met the species richness threshold, but fell below the FQI threshold. Table 9-1 also summarizes the percentage of the littoral area where aquatic plants were found and lists observed aquatic invasive species, their frequency of occurrence (FOO), and current management practices. At the time of this study, SWWD had not actively managed curly-leaf pondweed or Eurasian watermilfoil. Other plant health metrics can be referenced in the 2021 and 2024 Aquatic Vegetation Survey Results reports (Stantec, 2021, 2025).

Table 9-1 Powers Lake aquatic plants overview

Lake	Parameter	June 2021[1]	August 2021[1]	June 2024[2]	August 2024[2]	MnDNR Threshold	Invasive Species Management
Powers	Species Richness	10	13	12	10	>12	
	Floristic Quality Index (FQI)	16.3	19.2	17	14.2	>18.6	
	% Littoral with Vegetation	86%	85%	88%	77%		
	Curly-leaf Pondweed FOO	46%	18%	42%	-		No management to date
	Eurasian Watermilfoil FOO	3%	9%	36%	53%		No management to date
	Purple Loosestrife FOO	P	P	-	-		Managed naturally by leaf eating beetles

[1] (Stantec, 2021 Aquatic Vegetation Survey Results, 2021)
[2] (Stantec, 2025)

9.2.2 Fisheries

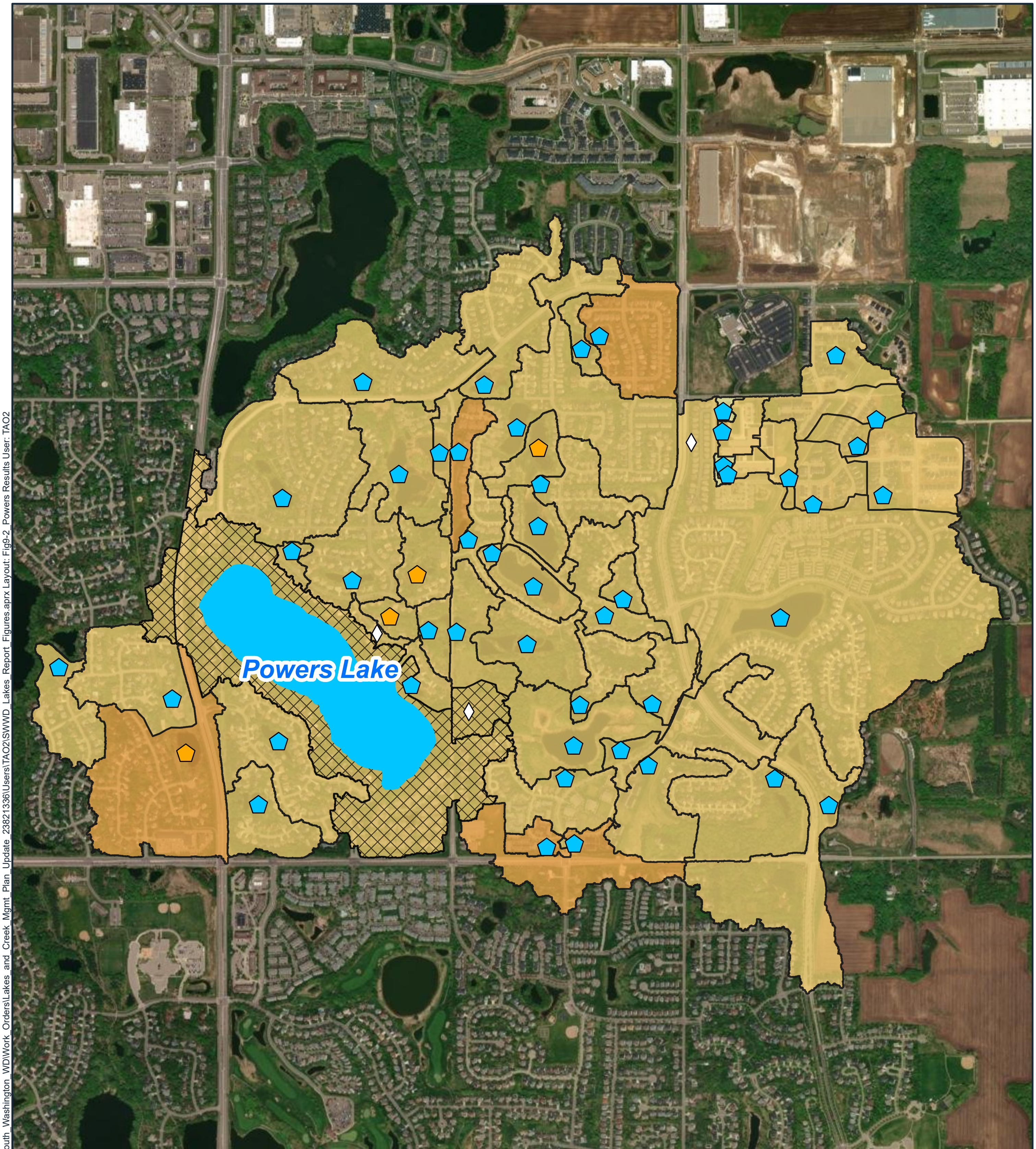
Powers Lake is included in the Fishing in the Neighborhood (FiN) program run by the MNDNR, which is aimed at increasing angling opportunities, public awareness, and environmental stewardship within the seven-county Twin Cities metro area. The MNDNR FiN program is actively involved in managing the sport fish populations of Powers Lake and as such the fish community is surveyed at a regular frequency. Table 9-2 summarizes the fish community data from the most recent survey conducted in July 2022 (MNDNR, LakeFinder, 2022).

Table 9-2 Powers Lake fish survey report - 2022 (MNDNR, LakeFinder, 2022)

Catch Method	Fish Species	Catch per Unit Effort (CPUE)	Normal CPUE Range	Count
Standard trap nets	Black crappie	22.0	1.8 – 18.1	110
	Bluegill	42.2	6.5 – 59.6	211
	Golden shiner	0.2	0.2 – 1.4	1
	Hybrid sunfish	0.4	N/A	2
	Largemouth bass	0.4	0.3 – 0.8	2
	Northern pike	0.2	N/A	1
	Pumpkinseed	0.8	0.8 – 5.3	4
	Walleye	0.4	0.3 – 1.2	2
	Yellow perch	0.4	0.3 – 1.5	2
Standard gill nets	Northern pike	2.0	2.5 – 7.9	2
	Yellow perch	6.0	1.5 – 12.8	6
Standard electrofishing	Largemouth bass	176.8	N/A	193

9.3 Watershed Total Phosphorus Loads

P8 modeling shows low to moderate watershed phosphorus loading to Powers Lake, with higher loads coming from areas of more dense residential development south of Powers Lake and pockets of single-family home neighborhoods that may have undersized treatment. The map in Figure 9-2 shows the estimated effective areal phosphorus loading by subwatershed, the modeled BMPs, and untreated subwatersheds. The effective phosphorus load represents the loading rate after pollutant removal by BMPs within the watershed and is reflective of the loading that actually makes it into the receiving waterbody from a given location. The effective phosphorus loads for the Powers Lake subwatersheds range from 0.3 pounds per acre per year to 0.8 pounds per acre per year, based on a 10-year modeling period (2012-2022). The direct drainage area to Powers Lake does not receive any treatment before runoff discharges to the lake. Additionally, the model results for 2012-2022 do not include treatment from stormwater BMPs at Hasenbank Park, east of Powers Lake across St. Johns Drive. More information on how the BMPs at Hasenbank Park are expected to impact Powers Lake are included in Section 9.5.



Lake

P8 Device Type

Dry Pond

Wet Pond

Pipe

Effective TP Load (lbs/ac/yr)

0.0 - 0.3

0.3 - 0.6

0.6 - 1.0

1.0 - 1.5

1.5 - 2.0

> 2.0

Area Not Treated by BMP

N

0 500 1,000

Feet

Powers Lake
Effective TP Loading
Lake Management Plan
South Washington
Watershed District

FIGURE 9-2

BARR

9.4 In-lake Total Phosphorus Loads

The in-lake model was developed to simulate water quality conditions within the fully mixed epilimnion volume (surface waters) of Powers Lake given that Powers Lake is a deep lake and typically experiences long durations of stratification during the growing season with a distinct epilimnion mixing layer, thermocline, and hypolimnion. The profile monitoring data was used to estimate the volume of the epilimnion throughout the model period. The total phosphorus loading estimates provided in Figure 9-3 represent the phosphorus loads to the mixed, epilimnion layer of Powers Lake. As such, the reported phosphorus load from lake bottom sediment represents the phosphorus load from shallow sediment located in the epilimnion zone as well as phosphorus that migrated to the epilimnion from the hypolimnion via diffusion or mixing events. Phosphorus load estimates to the hypolimnion were not estimated as a part of this study.

Results of the in-lake modeling for Powers Lake show that during the summer of 2019 (representing a wet year), the phosphorus loading to Powers Lake from watershed runoff was greater than the contribution from internal loading. Phosphorus originating from watershed runoff represented 60% (259 lbs) of the phosphorus loading to the lake during summer 2019 (Figure 9-3). Internal loading in summer 2019 was estimated to be 40% (171 lbs) of the phosphorus load to the lake. During the drier summer of 2022, the phosphorus loading to Powers Lake from watershed runoff was notably lower at 73 lbs, representing 51% of the phosphorus loading to the lake. Internal loading in 2022 was also notably lower than 2019 at 70 lbs, representing 49% of the phosphorus load.

Differences in annual lake stratification can influence lake mixing potential as well as the anoxic area over lake bottom sediment, which ultimately influences the potential total phosphorus load that reaches surface waters from lake bottom sediment. Biweekly profile monitoring data is typically collected from Powers Lake between April and October, including the collection of temperature and dissolved oxygen concentrations. The data indicates that Powers Lake can have temperature stratification as early as late April or early May most years, and typically remains stratified until fall. When large storm events occur, there is potential for partial lake mixing, which may result in higher phosphorus concentrations from the metalimnion or hypolimnion reaching the surface waters (epilimnion). Comparing the profile monitoring data between model years 2019 and 2022 indicates that Powers Lake mixed to greater depths in 2019. Intensified lake mixing to deeper depths could be a reason why there was increased internal phosphorus load observed in the 2019 Powers Lake in-lake model. Phosphorus fraction data gathered from sediment cores collected from Powers Lake in 2024 indicated a high potential for internal phosphorus loading. Additional details on the sediment core data can be found in Section 4.8.

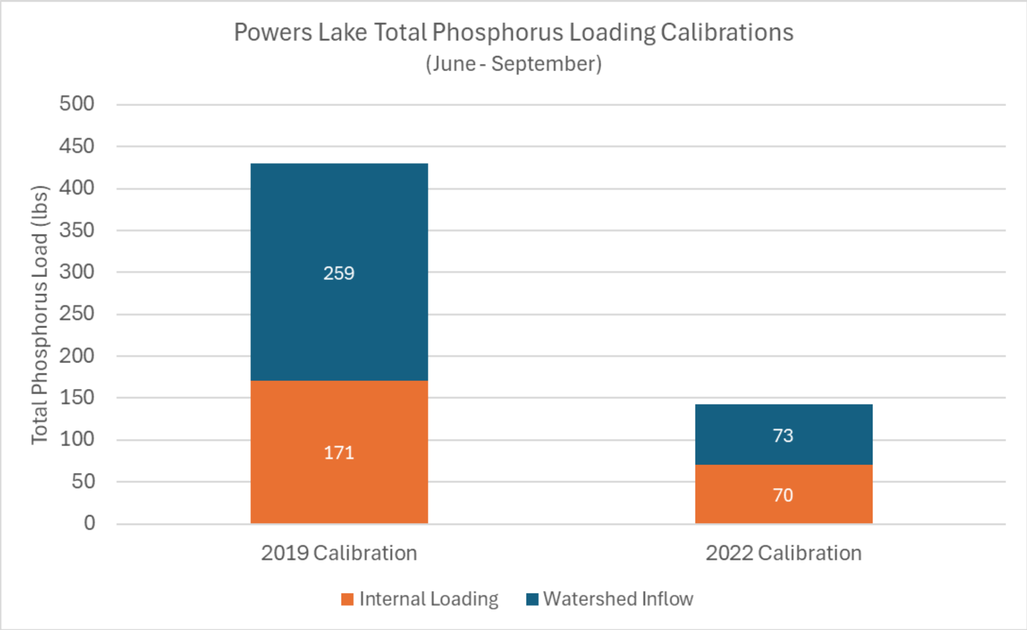


Figure 9-3 Powers Lake total phosphorus load estimates – 2019 & 2022

9.5 Predicted Benefits from Recent Capital Projects

One of the goals of the SWWD with this study was to understand how newly constructed BMPs within the District can be anticipated to impact water quality within downstream receiving waterbodies. The calibrated in-lake models for Powers Lake were used to estimate the impact of the stormwater BMPs within the Hasenbank Park Project on lake water quality. Although the project wasn't operational until 2025, to gain some understanding of how the project can be expected to impact water quality, the BMPs were turned on in the p8 model to simulate watershed nutrient reductions for model years 2019 and 2022.

Table 9-3 summarizes the estimated phosphorus load reduction from watershed inputs during the 2019 and 2022 summer growing periods, assuming the installation of BMPs at Hasenbank Park. A reduction of 14 – 44 pounds of total phosphorus loading to Powers Lake is estimated based on model results for a wet (2019) and dry (2022) year. This reduction in phosphorus load translates to a reduction in the total phosphorus concentration in the lake, with the 2019 summer average total phosphorus concentration of 38 µg/L reduced to 36 µg/L and the 2022 summer average total phosphorus concentration of 26 µg/L reduced to 25 µg/L (Table 9-5). Figure 9-4 and Figure 9-5 summarize the predicted reductions in in-lake total phosphorus concentrations for the 2019 and 2022 model years, respectively.

Evaluating watershed BMPs over a single growing season provides an incomplete picture of their effectiveness. A multi-year perspective better captures the cumulative nutrient reduction benefits that watershed BMPs offer to downstream waterbodies. However, the in-lake modeling completed as a part of this study does not support a long-term benefits analysis.

Although this was not quantified using the in-lake models, it's expected that the Hasenbank Park Project will result in water quality improvements to downstream waterbodies. Reducing total phosphorus concentrations in Powers Lake will decrease phosphorus outflows downstream, creating positive impacts for South Wilmes, Colby, and Bailey Lakes.

Table 9-3 **Summer growing period estimated reductions in phosphorus loads and in-lake concentrations with the construction of the Hasenbank Park Project**

Year	Pounds of Phosphorus Removed (June – Sept)	Summer Average Concentration (µg/L)		
		Existing[1]	With Hasenbank	Reduction
2019 (wet summer)	-44	38	36	-2
2022 (dry summer)	-14	26	25	-1

[1] Existing conditions summer average concentrations are calculated from calibrated in-lake models

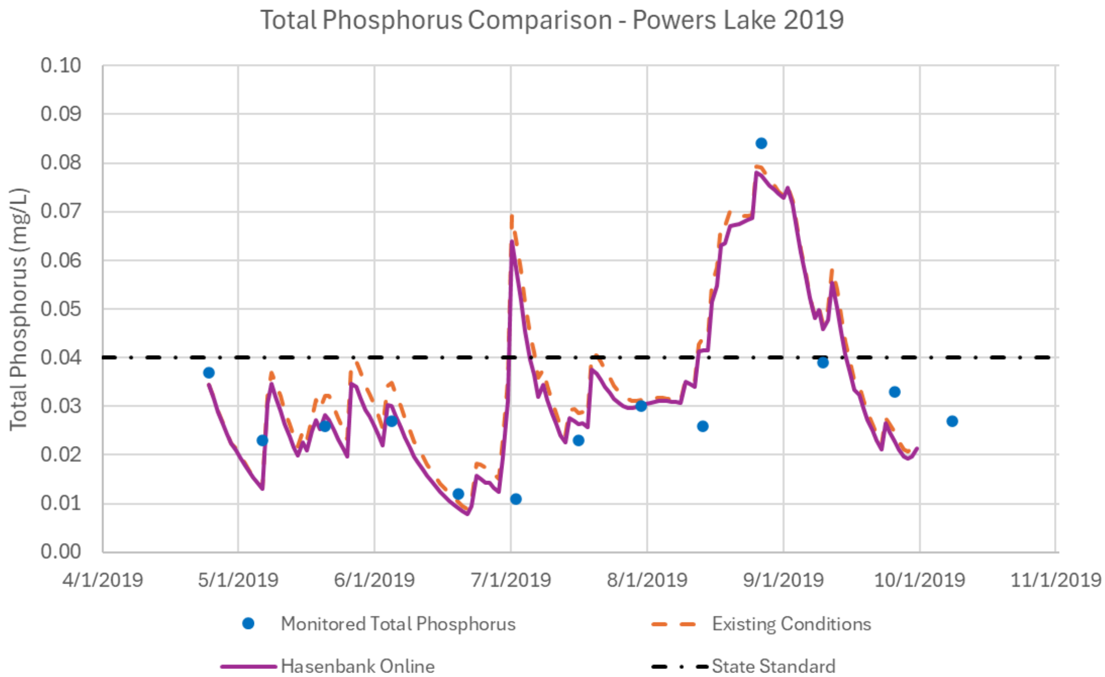


Figure 9-4 **Model predicted reductions in 2019 total phosphorus concentrations in Powers Lake assuming the Hasenbank Park projects were online**

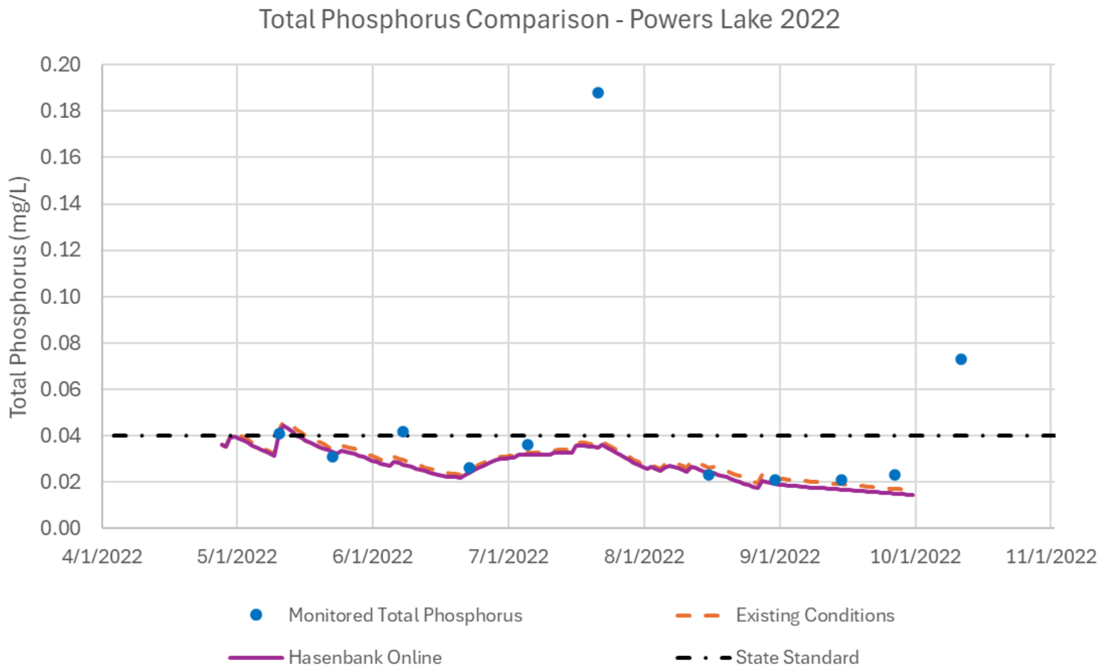


Figure 9-5 Model predicted reductions in 2022 total phosphorus concentrations in Powers Lake assuming the Hasenbank Park projects were online

SWWD has also recently launched a partnership with the City of Woodbury to perform enhanced street sweeping throughout city streets with the goal of reducing phosphorus loads to receiving waterbodies. As of the writing of this report, the City of Woodbury was planning to sweep all city streets once per month with a high efficiency vacuum sweeper. Based on a study completed in 2022, the estimated additional benefit of moving from a typical street sweeping schedule to monthly sweeping within the Powers Lake watershed would result in an additional approximately 7 lbs of TP removal from the estimated watershed loading to Powers Lake each year (Emmons & Olivier Resources (EOR), 2022). The 2022 report notes that estimated load reductions from additional sweeping should not be used for relative comparisons of total loadings due to their planning-level nature. Estimated watershed loading into Powers Lake for 2019 and 2022 is shown in Figure 9-3.

9.6 Predicted Benefits from Implementing In-lake Management Practices

Another study goal was to estimate the potential impact of lake sediment inactivation projects and how they could be used to improve in-lake water quality within the SWWD suburban lakes. The in-lake models that were modified to include the Hasenbank Park Project (Section 9.5) were used to estimate the impact of implementing sediment treatment projects in Powers Lake.

Table 9-4 summarizes the estimated phosphorus load reduction from three scenarios: Hasenbank Park Project only and Hasenbank Park Project coupled with sediment inactivation projects (low and high efficacies). Results from the in-lake modeling predict a reduction of 42 – 137 pounds of total phosphorus loading into Powers Lake through implementation of sediment inactivation practices for a wet (2019) and dry (2022) year. This reduction in phosphorus loading translates to a reduction in the in-lake total phosphorus concentrations. Model results estimate that the 2019 epilimnetic summer average total phosphorus concentration of 38 µg/L would reduce to 23 - 25 µg/L with the implementation of sediment treatments and with the Hasenbank Park Project online; and the 2022 epilimnetic summer average total phosphorus concentration of 26 µg/L would reduce to 18 - 20 µg/L (Table 9-4). The existing summer average total phosphorus concentrations within Powers Lake already meet state standards, but implementing sediment inactivation projects would offer opportunities to further reduce phosphorus concentrations. Figure 9-6 and Figure 9-7 summarize the predicted reductions in in-lake total phosphorus concentrations for the 2019 and 2022 model years, respectively, for each of the proposed BMP scenarios.

Table 9-4 Summer growing period estimated reductions in phosphorus loads and in-lake concentrations in Powers Lake with the proposed management practices

Scenario	Modeled Summer Total Phosphorus Loading (lbs)				Summer Average Total Phosphorus (µg/L)[1]
	Watershed Inflow	Internal Loading	Total	Load Reduction	
2019 Existing	259	171	430	--	38
2019 Hasenbank Online	215	171	386	-44	36
2019 Hasenbank Online + Sediment Inactivation Low	215	68	283	-147	25
2019 Hasenbank Online + Sediment Inactivation High	215	34	249	-181	23
2022 Existing	73	70	143	--	26
2022 Hasenbank Online	59	70	129	-14	25
2022 Hasenbank Online + Sediment Inactivation Low	59	28	87	-56	20
2022 Hasenbank Online + Sediment Inactivation High	59	14	73	-70	18

[1] Existing conditions summer average concentrations are calculated from calibrated in-lake models

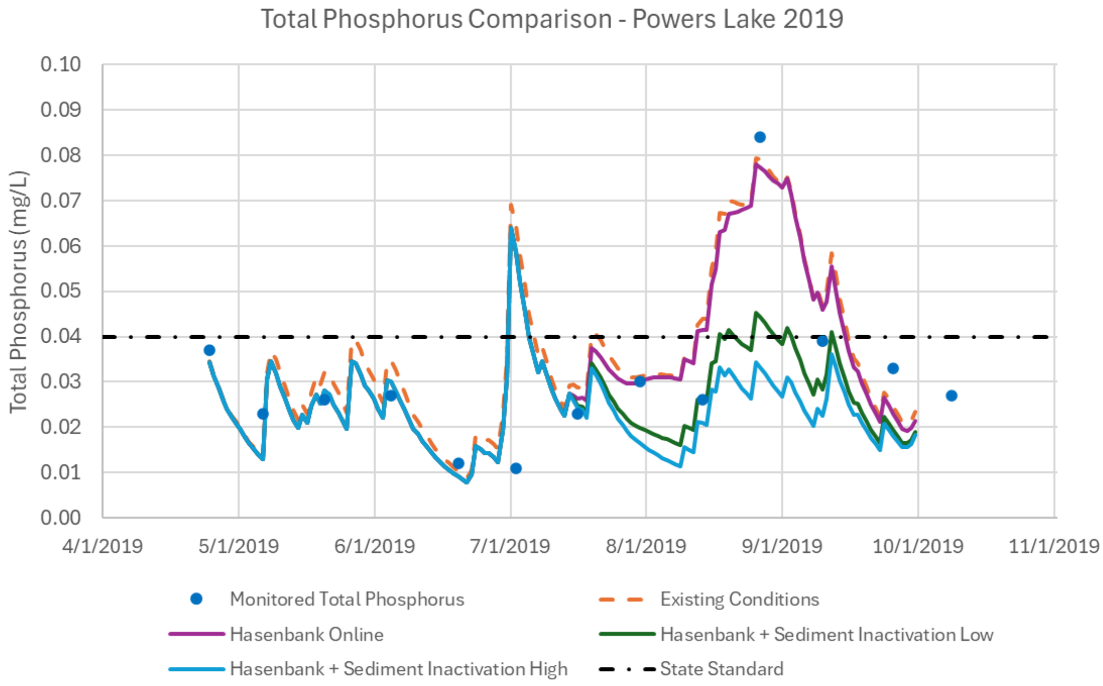


Figure 9-6 **Model-predicted reductions in 2019 (epilimnetic) total phosphorus concentrations in Powers Lake with the proposed management practices**

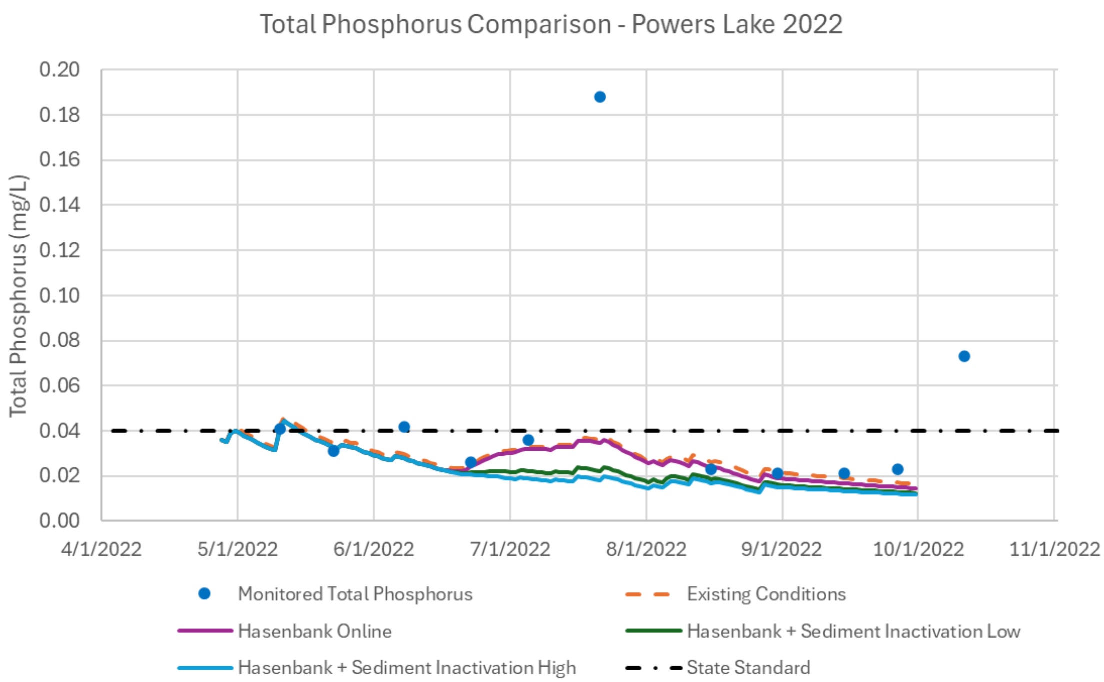


Figure 9-7 **Model-predicted reductions in 2022 (epilimnetic) total phosphorus concentrations in Powers Lake with the proposed management practices**

Model results estimating the in-lake response to reduced phosphorus loading—both from internal treatments and the Hasenbank Park Project—reflect the estimated impacts to in-lake conditions during the 2019 and 2022 growing seasons if those practices had been in place. The modeling incorporates a range of assumed sediment-treatment efficacies to account for application variability and the fact that the specific treatment approach (alum, alum/iron, or aeration) has not yet been selected. Assumed efficacies were informed by monitoring data from comparable Barr sediment-inactivation projects one year post-implementation. Monitoring of past projects shows that sediment inactivation efficacy can vary and is expected to decline over time due to product aging, new watershed inflows, and/or burial from rough fish activity. As such, results presented in this report provide an estimate of the expected scale of treatment impacts within a given growing season, but do not account for longer-term changes within the system and how that may impact benefits over time. For more information on the sediment inactivation modeling assumptions see Section 6.4.

9.7 Management Recommendations

Monitoring of Powers Lake in recent years indicates that the surface waters (epilimnion) are typically meeting state water quality standards. However, monitoring of the deep waters (hypolimnion) shows that total phosphorus concentrations can be more than 50 times the concentrations measured in the surface waters due to high internal loading from lake bottom sediment. Powers Lake also has a threatened native aquatic plant community due to the notable growth of the aquatic invasive species Eurasian water milfoil and curly-leaf pondweed. Given this, future management efforts should focus on protecting lake water quality and improving ecosystem health, monitoring for changes, and continuing water quality and ecosystem health protection measures as improvements are achieved. Table 9-5 summarizes the recommended management strategies that could be considered for Powers Lake to help improve water quality conditions and ecological health.

Table 9-5 Powers Lake management recommendations

Management/Protection Action		Basis
Address external watershed loads	Enhanced street sweeping program	Continue to work with the City of Woodbury to refine an enhanced street sweeping program to reduce pollutant loading to stormwater runoff
	Stormwater BMPs	Consider retrofitting or installing new stormwater BMPs in subwatersheds that are currently untreated or undertreated Implement site scale BMPs as opportunities arise
	Chloride	Consider applying chloride reduction strategies such as education and implementation assistance to the City of Woodbury and other stakeholders
Address internal loads	Sediment inactivation treatment	Review and consider implementing a sediment inactivation treatment to reduce lake bottom sediment phosphorus loads
Aquatic Plants	Invasive species management	Continue to monitor invasive species growth (e.g., curly-leaf pondweed, Eurasian watermilfoil) Consider herbicide management of curly-leaf pondweed and Eurasian watermilfoil to improve native aquatic plant health
	Promote native aquatic plant growth	Encourage native plant reestablishment during and following aquatic invasive species management
Phytoplankton and Zooplankton	Phytoplankton/Zooplankton monitoring	Consider monitoring phytoplankton and zooplankton as part of routine monitoring
Water Quality	Water quality monitoring	Continue monitoring in-lake water quality and assessing for eutrophication Continue to identify/track chloride levels from winter salt use

10 Wilmes Lake

10.1 Lake Water Quality

10.1.1 North Wilmes Lake

Shallow/Deep	Shallow
Location	Woodbury
Surface Area	19 acres
Average/Maximum Depth	7 feet / 21 feet
Direct Watershed Area	2,413 acres
Total Watershed Area	2,985 acres
Watershed:Lake Surface Area	157:1
Impairment Status	Impaired for nutrients since 2006
Downstream Waterbody	South Wilmes Lake

Wilmes Lake is located in the City of Woodbury and consists of two basins, North and South. North Wilmes is used primarily for wildlife viewing and passive recreation. When water levels are high enough, water discharges via gravity flow from North Wilmes Lake through a storm sewer culvert to South Wilmes Lake.

North Wilmes Lake has a water surface area of approximately 19 acres, a maximum depth of 21 feet, and a mean depth of approximately 7 feet. The MNDNR defines a shallow lake as a lake with a maximum depth of 15 feet or with 80% or more of the lake area shallow enough to support emergent and submerged rooted aquatic plants. North Wilmes Lake falls under the MNDNR classification of a "shallow lake" given that the lake's littoral area is greater than 80%. North Wilmes Lake is deep enough that it's feasible that prolonged periods of lake stratification could occur during the growing season. This could not be confirmed as part of this study as no monitoring data is currently being collected in this lake.

In-lake modeling performed for North Wilmes Lake as a part of this study is high level and has a high amount of relative uncertainty, given no monitoring data is currently available for the lake. Model results were verified for relative accuracy as part of calibrating downstream lake models.

10.1.2 South Wilmes Lake

Shallow/Deep	Shallow
Location	Woodbury
Surface Area	20 acres
Average/Maximum Depth	8 feet / 18 feet
Direct Watershed Area	615 acres
Total Watershed Area	5,288 acres
Watershed:Lake Surface Area	264:1
Impairment Status	Impaired for nutrients since 2006
Downstream Waterbody	Colby Lake

Similar to the north basin, South Wilmes Lake is used primarily for wildlife viewing and passive recreation. When water levels are high enough, water discharges via gravity flow from South Wilmes Lake through a storm sewer system, which ultimately discharges to Colby Lake.

South Wilmes Lake has a water surface area of approximately 20 acres, a maximum depth of 18 feet, and a mean depth of approximately 8 feet. South Wilmes Lake falls under the MNDNR classification of a "shallow lake" given that the lake's littoral area is greater than 80% the total lake area. South Wilmes Lake is deep enough that prolonged periods of lake stratification occur during the growing season. Profile monitoring data collected within the lake supports this finding.

The State of Minnesota uses three water quality parameters to assess eutrophication standards within lakes—total phosphorus, chlorophyll-*a*, and Secchi disk transparency—to assess waterbody health and track water quality changes. These three parameters were measured in the south basin of Wilmes Lake by the SWWD between 1994-2023 (Figure 10-1). During the monitored years, summer average total phosphorus concentrations were better than the state standard in 1994, 2004, 2008, 2017 – 2020, and 2022 - 2023. All other monitored years exceeded the state standard summer average concentration. Summer average chlorophyll-*a* concentrations met or were better than the state standard in 1994, 2009 - 2011, 2016, and 2018 – 2019. Summer average Secchi depths met or were better than the state standard between 1994 – 1996, 1998 – 2000, 2003 – 2004, 2006 – 2012, 2014, 2016, and 2018 – 2023. Wilmes Lake was added to the Minnesota impaired waters list for nutrients in 2006 based on the monitoring data collected from the south basin.

Chloride concentrations were measured by the SWWD between 2013 - 2023 (generally between April and September). In the historical record, all observed chloride concentrations were below the MPCA chronic standard of 230 mg/L. The highest observed chloride concentration was 225 mg/L in May 2023.

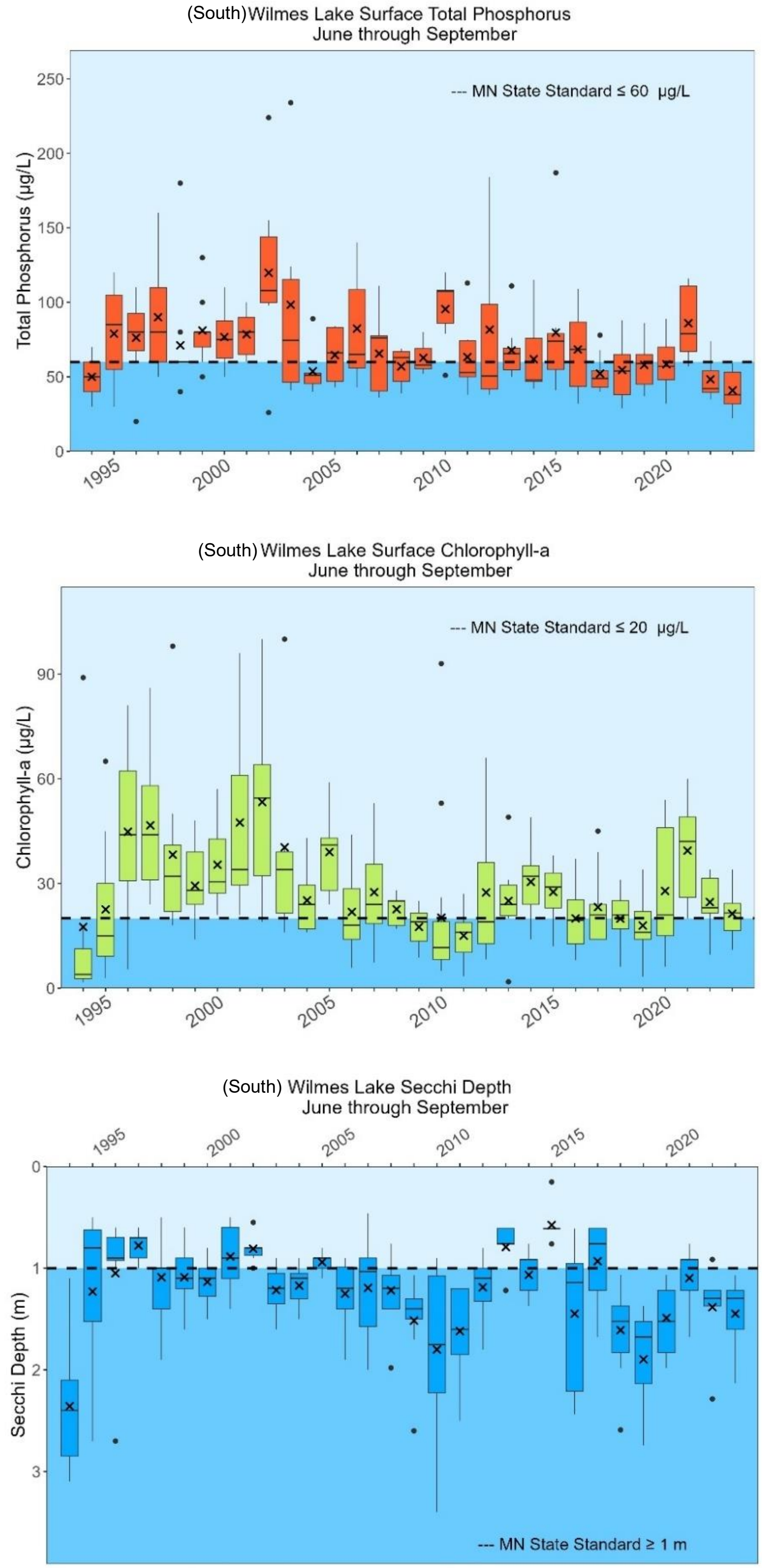


Figure 10-1 **South Wilmes Lake eutrophication monitoring data (June – September)**
Summer averages are shown by x's in the box plots

10.2 Ecological Health

10.2.1 Aquatic Plants

Table 10-1 summarizes the calculated Lake Plant Eutrophication IBI values for Wilmes Lake based on point-intercept plant surveys completed in 2021 and 2024. Wilmes Lake scored below both plant IBI metrics in June and August 2021 and in August 2024 indicating a degraded plant community that is likely stressed from cultural eutrophication. Table 10-1 also summarizes the percentage of the littoral area where aquatic plants were found and lists observed aquatic invasive species, their frequency of occurrence (FOO), and current management practices. Other plant health metrics can be referenced in the 2021 and 2024 Aquatic Vegetation Survey Results reports (Stantec, 2021, 2025).

Table 10-1 Wilmes Lake aquatic plants overview

Lake	Parameter	June 2021[1]	August 2021[1]	June 2024[2]	August 2024[2]	MnDNR Threshold	Invasive Species Management
Wilmes (N & S)	Species Richness	10	6	12	9	>11	
	Floristic Quality Index (FQI)	13.9	10.2	16.4	13.2	>17.8	
	% Littoral with Vegetation	75%	79%	75%	59%		
	Curly-leaf Pondweed FOO	23%	-	10%	-		Treated North Basin for CLP in 2024 & 2025
	Eurasian Watermilfoil FOO	27%	12%	25%	17%		Treated South Basin for EWM in 2024 & 2025

[1] (Stantec, 2021 Aquatic Vegetation Survey Results, 2021)
[2] (Stantec, 2025)

10.2.2 Fisheries

There are no MNDNR fish survey or stocking data available for Wilmes Lake.

10.3 Watershed Total Phosphorus Loads

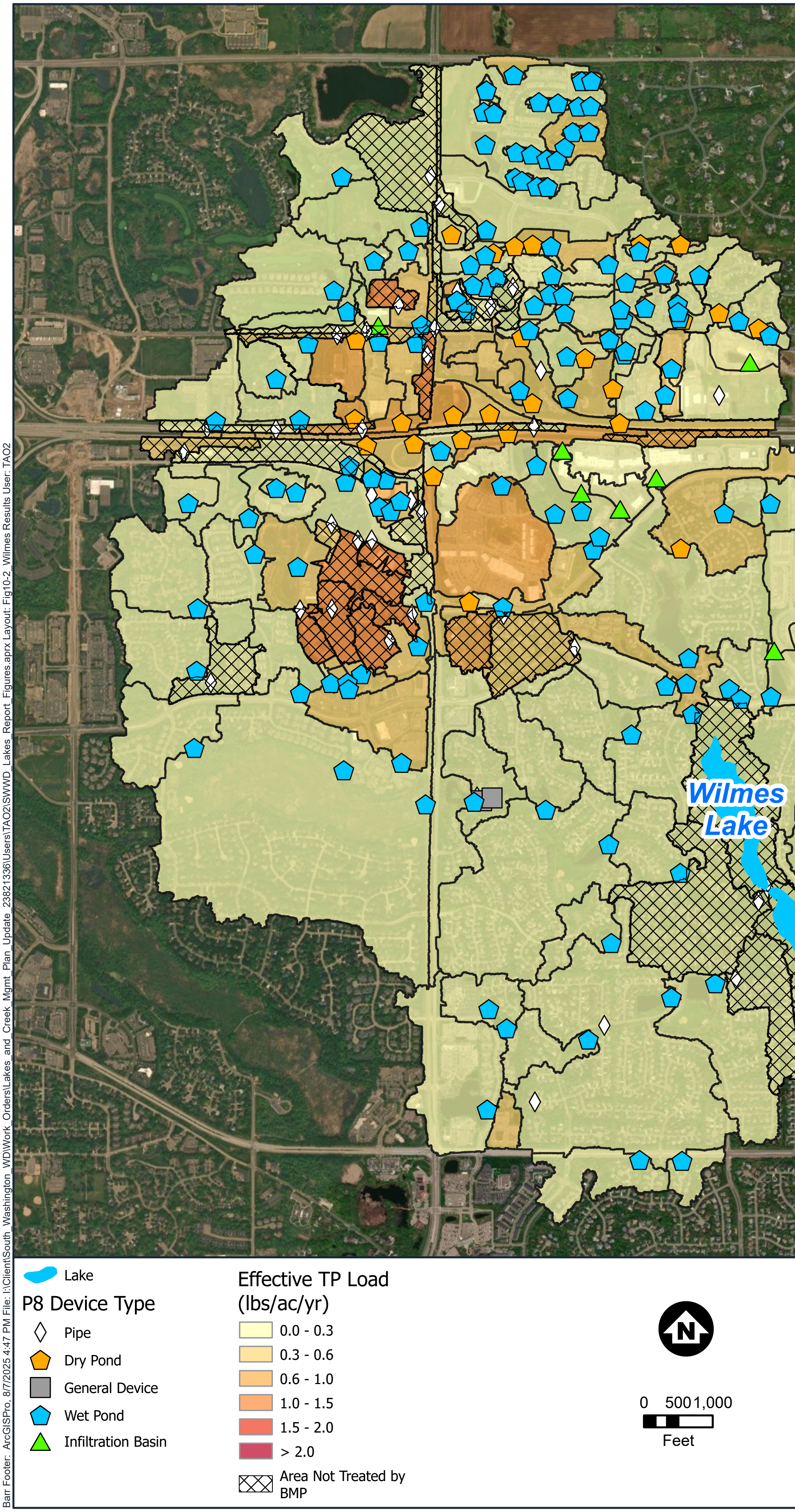
10.3.1 North Wilmes Lake

P8 modeling shows low to high watershed phosphorus loads to North Wilmes Lake, with higher loads coming from commercial areas north of Interstate 94, some of which are untreated by regional stormwater BMPs before entering the storm sewer system and discharging to North Wilmes Lake. The map in Figure 10-2 shows the effective areal phosphorus load by subwatershed, modeled BMPs, and the untreated subwatersheds. The model results include the Seasons Park CC17 filter, which is located within the North Wilmes Lake subwatershed. The effective phosphorus load represents the loading rate after pollutant removal by BMPs within the watershed and is reflective of the loading that actually makes it into the receiving waterbody from a given location. The effective phosphorus loads for the North Wilmes Lake subwatersheds range from 0.01 pounds per acre per year to 1.3 pounds per acre per year, based on a 10-year modeling period (2012-2022).

The model results for 2012-2022 do not include treatment from the Kargel Park Alum Treatment Facility, which is scheduled to go online in 2025. The alum facility will draw water flowing into the north end of North Wilmes Lake and treat it with alum before discharging treated water back into the lake. More information on how the Kargel Park Alum Treatment Facility is expected to impact North Wilmes Lake is included in Section 10.5.

10.3.2 South Wilmes Lake

P8 modeling shows low watershed phosphorus loads to South Wilmes Lake, given that the majority of the drainage area contributing to the lake is single-family residential properties. Some portions of the contributing drainage area are untreated before entering the storm sewer system and reaching the lake or have small rate control or energy dissipation BMPs that were designed to slow flows but not intended to provide significant water quality treatment. The map in Figure 10-2 shows the effective areal phosphorus load by subwatershed, modeled BMPs, and the untreated subwatersheds within this area. The effective phosphorus load represents the loading rate after pollutant removal by stormwater BMPs within the watershed and is reflective of the loading that actually makes it into the receiving waterbody from a given location. The effective phosphorus loads from the South Wilmes Lake subwatersheds range from 0.07 pounds per acre per year to 0.3 pounds per acre per year, based on a 10-year modeling period (2012-2022).



Wilmes Lake
Effective TP Loading
Lake Management Plan
South Washington
Watershed District

FIGURE 10-2

BARR

10.4 In-lake Total Phosphorus Loads

10.4.1 North Wilmes Lake

The in-lake model for North Wilmes Lake could not be calibrated as water quality data has not historically been collected within this lake. The North Wilmes in-lake model was developed using high level assumptions and extrapolations from other similar lake models. Outputs from the North Wilmes model were ultimately used to help calibrate the in-lake model for South Wilmes Lake.

Based on the high-level assumptions developed for the North Wilmes models, during the summer of 2019 (representing a wet year), the phosphorus loading into North Wilmes Lake was dominated by loading from the surrounding watershed, representing 73% (209 lbs) of the phosphorus load from major sources (Figure 10-3). Internal loading in 2019 was estimated to be 13% (36 lbs) of the phosphorus load to the lake. Total phosphorus loads from upstream Armstrong Lake were estimated to represent 14% (41 lbs) of the total phosphorus load to North Wilmes in 2019. During the drier summer of 2022, the phosphorus loading to the lake from watershed runoff was notably lower, representing 60% (49 lbs) of the phosphorus load into the lake. Internal loading in 2022 represented 40% (33 lbs) of the phosphorus load. In the summer of 2022, Armstrong Lake did not discharge to North Wilmes due to low water levels. Therefore, no phosphorus load was attributed to upstream Armstrong Lake in summer 2022 (0%, 0 lbs).

The total phosphorus loading estimates for North Wilmes Lake (presented in Figure 10-3) have a high level of uncertainty associated with them, given that no monitoring data was available for model calibration. If water quality monitoring data is collected in the future, it is recommended that the North Wilmes Lake model be revisited.

Given that the maximum depth of North Wilmes Lake is 21 feet, it’s likely that the lake moderately to strongly stratifies during the summer growing season. The collection of lake profile data is recommended as part of future efforts to better estimate the influence of internal loading on lake water quality conditions. Understanding the role of lake stratification in internal loading dynamics will be important should the District decide to move forward with developing sediment inactivation strategies. The collection of profile water quality data is especially critical since the sediment core phosphorus fractionation data indicated a high potential for internal phosphorus loading from lake bottom sediment within North Wilmes Lake (Section 4.8).

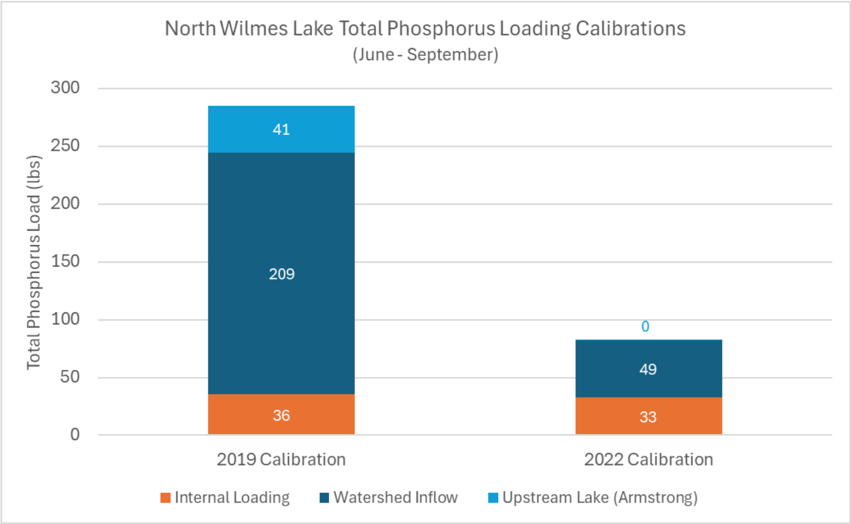


Figure 10-3 North Wilmes Lake total phosphorus load estimates – 2019 & 2022

10.4.2 South Wilmes Lake

Results of the in-lake modeling for South Wilmes Lake showed that during the summer of 2019 (representing a wet year), the phosphorus loading into South Wilmes Lake was dominated by upstream lake inputs, representing 82% (405 lbs) of the phosphorus loading from major sources (Figure 10-4). South Wilmes Lake is unique and is the only lake in this study with multiple upstream lakes. Figure 10-5 summarizes the estimated total phosphorus load from each upstream lake. In 2019, 34% (166 lbs) of the phosphorus load to South Wilmes was from North Wilmes Lake, 31% (156 lbs) from Markgrafs Lake, and 17% (83 lbs) from Powers Lake. Smaller total phosphorus loads were from internal loading (11%, 55 lbs) and watershed runoff (7%, 34 lbs).

During the drier summer of 2022, the phosphorus loading to South Wilmes Lake from upstream lakes was notably lower, representing 23% (12 lbs) of the phosphorus load into the lake. The phosphorus loading from upstream lakes in 2022 was 97% less than the phosphorus load estimated in 2019. In 2022, 16% (8 lbs) of the phosphorus load to South Wilmes was from Markgrafs Lake, 6% (3 lbs) from North Wilmes Lake, and 2% (1 lb) from Powers Lake. Total phosphorus loading from watershed runoff was also notably lower in 2022, representing 13% (6 lbs) of the load to South Wilmes Lake. Internal phosphorus loading estimates in 2022 were similar to 2019, representing 63% (32 lbs) of the phosphorus load.

Biweekly profile monitoring data is typically collected in South Wilmes Lake between April and October of each year, including the collection of temperature and dissolved oxygen concentrations. The data provided for this study indicates that South Wilmes Lake can have temperature stratification as early as late-April or early-May most years and typically remains stratified until fall. This was also observed in the in-lake model. To simulate lake stratification, model calibration parameters were adjusted to suppress the amount of internal phosphorus load reaching the surface waters during the growing season. Given that the monitoring data demonstrates strong lake stratification during the summer, the collection of water quality parameters at multiple depths is recommended to better estimate internal loading influence on surface water conditions (e.g., collecting phosphorus near the surface, at the thermocline, and near the bottom). Understanding how lake stratification influences internal loading will be essential for developing effective sediment inactivation strategies and demonstrating benefits, should the District decide to pursue this type of treatment. The collection of water quality parameters at multiple depths is especially important as sediment core phosphorus fractionation data indicate a high potential for phosphorus release from lake bottom sediment (Section 4.8).

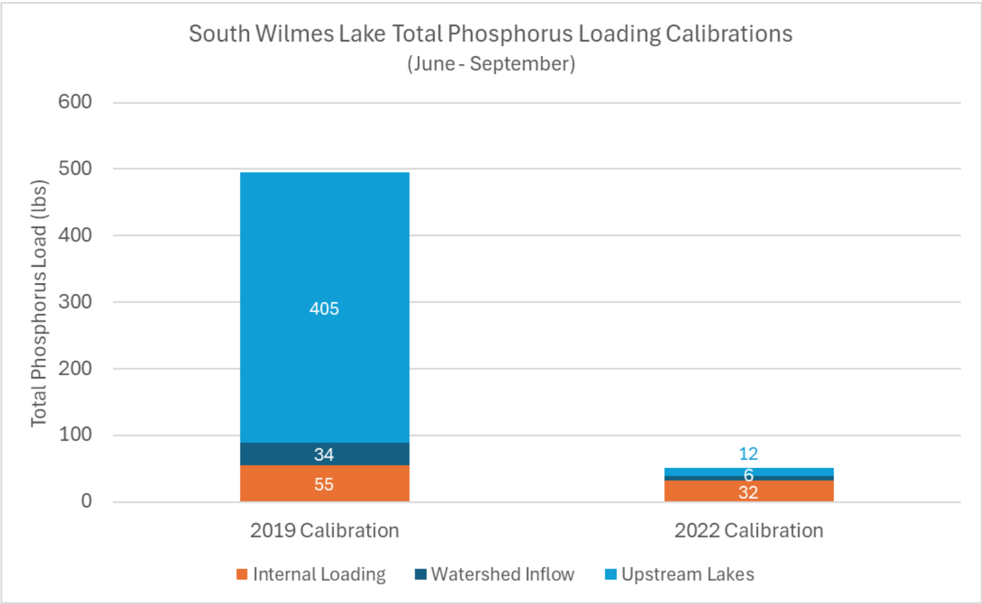


Figure 10-4 South Wilmes 2019 & 2022 estimated total phosphorus loads (June – Sept)

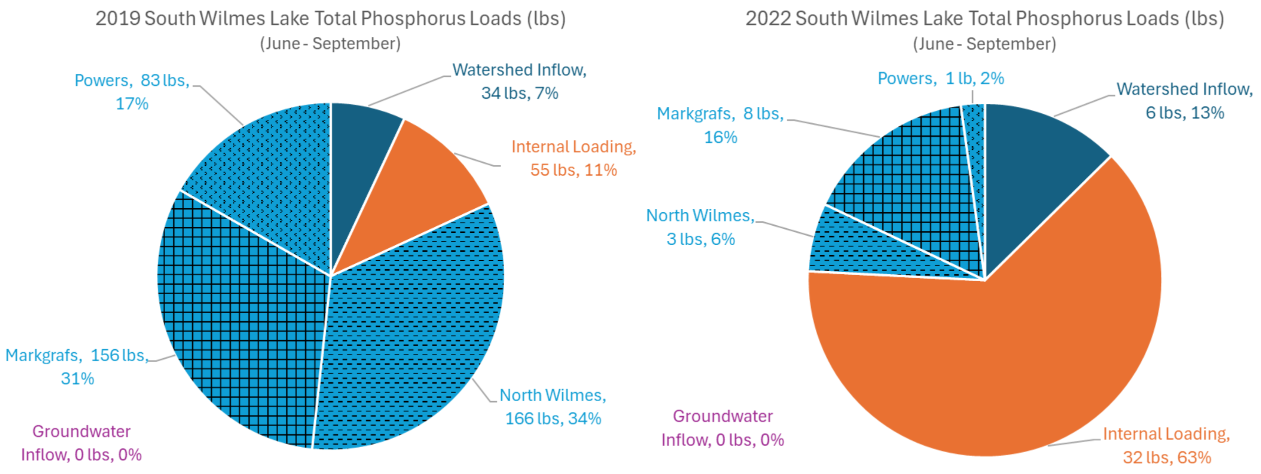


Figure 10-5 South Wilmes estimated total phosphorus loads per source (detailed)

10.5 Predicted Benefits from Recent Capital Projects

One goal of the SWWD with this study was to understand how newly constructed BMPs within the District can be anticipated to impact water quality within downstream receiving waterbodies. The in-lake models for North Wilmes Lake were used to estimate the impact of the Kargel Park alum treatment facility on lake water quality. Although the alum treatment facility was not put online until 2025, the predicted nutrient reductions of the facility (HRGreen, 2020) were used to simulate watershed nutrient reductions for model years 2019 and 2022, as a representation of performance under wet and dry conditions.

Table 10-2 summarizes the estimated phosphorus load reduction from watershed inputs during the 2019 and 2022 summer growing periods using assumptions identified within the 2020 HRGreen design memo for the facility (HRGreen, 2020). A reduction of 21 – 52 pounds of total phosphorus loading to North Wilmes Lake is estimated based on model results for a wet (2019) and dry (2022) year. This reduction in phosphorus load translates to a reduction in the estimated total phosphorus concentration to be expected in the lake, with the 2019 summer average total phosphorus concentration of 71 µg/L reduced to 57 µg/L and the 2022 summer average total phosphorus concentration of 57 µg/L reduced to 45 µg/L (Table 10-2). Figure 10-6 and Figure 10-7 summarize the predicted reductions in in-lake total phosphorus concentrations for the 2019 and 2022 model years, respectively.

Evaluating watershed BMPs over a single growing season provides an incomplete picture of their effectiveness. A multi-year perspective would better capture the cumulative nutrient reduction benefits that watershed BMPs can be expected to offer downstream waterbodies. The in-lake modeling completed as a part of this study does not support a long-term benefits analysis. The District could consider analyzing expected long-term benefits as part of future work.

Although it was not quantified as part of this study, it is expected that the Kargel Park alum treatment facility will also result in water quality improvements to waterbodies downstream of North Wilmes. Reducing total phosphorus concentrations in North Wilmes Lake will decrease phosphorus outflows, creating positive impacts for South Wilmes, Colby, and Bailey Lakes.

Table 10-2 Summer growing period estimated reductions in phosphorus loads and in-lake concentrations with the construction of the alum treatment facility

Year	Pounds of Phosphorus Removed (June – Sept)	In-lake Summer Average TP Concentration (µg/L)		
		Existing[1]	With Alum Treatment Facility	Reduction
2019 (wet summer)	-52	71	57	-14
2022 (dry summer)	-21	57	45	-12

[1] Existing conditions summer average concentrations are calculated from in-lake models

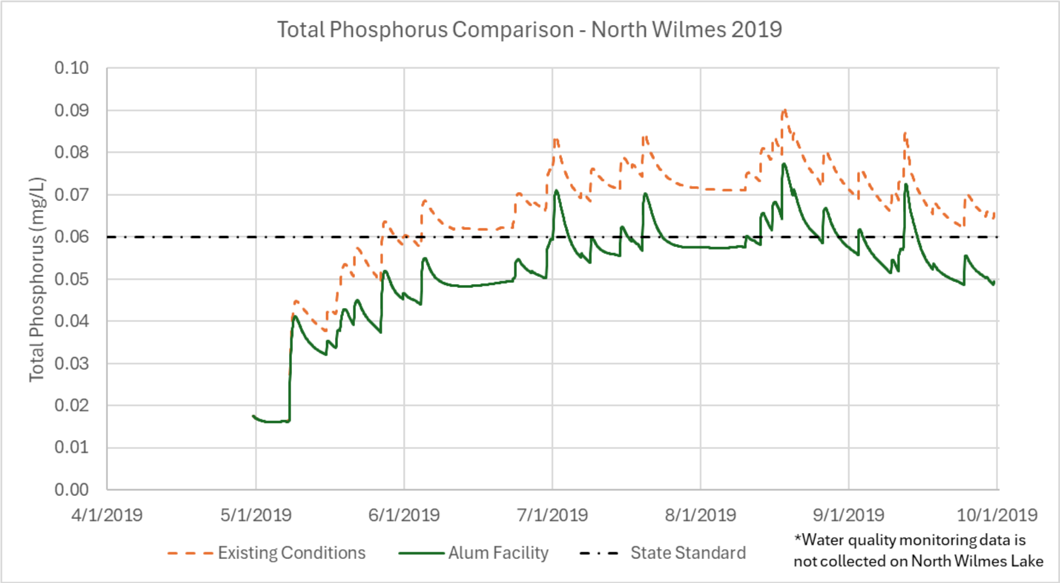


Figure 10-6 **Model predicted reductions in 2019 total phosphorus concentrations in North Wilmes Lake assuming operation of the alum treatment facility**

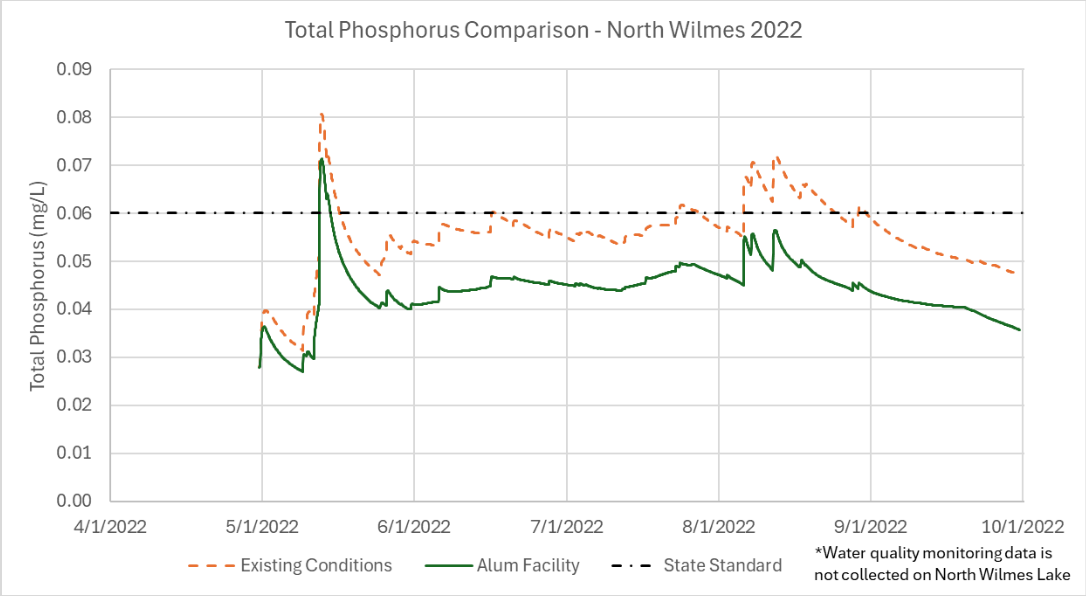


Figure 10-7 **Model predicted reductions in 2022 total phosphorus concentrations in North Wilmes Lake assuming operation of the alum treatment facility**

SWWD has also recently launched a partnership with the City of Woodbury to perform enhanced street sweeping throughout city streets with the goal of reducing phosphorus loads to receiving waterbodies. As of the writing of this report, the City of Woodbury was planning to sweep all city streets once per month with a high efficiency vacuum sweeper. Based on a study completed in 2022 (EOR, 2022) the estimated additional benefit of moving from a typical street sweeping schedule to monthly sweeping within the North Wilmes Lake watershed would result in an additional approximately 9 lbs of TP removal from the estimated watershed loading to North Wilmes Lake each year. The 2022 report notes that estimated load reductions from additional sweeping should not be used for relative comparisons of total loadings due to their planning-level nature. Estimated watershed loading into North Wilmes Lake for 2019 and 2022 is shown in Figure 10-3.

10.6 Predicted Benefits from Implementing Management Practices

10.6.1 North Wilmes Lake

Another of the study goals was to estimate the potential impact of lake sediment inactivation projects and how they could be used to improve in-lake water quality. The in-lake models that were modified to include the Kargel Park alum treatment facility (Section 10.5) were used to estimate the impact of implementing sediment treatment projects in North Wilmes Lake. For North Wilmes Lake, additional watershed load reductions were also analyzed to assess joint impacts on further improving in-lake conditions to meet state standards. Specific watershed BMPs were not modeled; instead, a percent reduction was applied to the watershed loading estimates across the watersheds directly tributary to North Wilmes Lake.

Table 10-3 summarizes the estimated phosphorus load reduction from four scenarios: the Kargel Park alum treatment facility being active; Kargel Park alum treatment facility coupled with sediment inactivation (low and high efficacies); Kargel Park alum treatment facility with sediment inactivation (low) and watershed load reductions. Results from the in-lake modeling predict a reduction of 27 – 30 pounds of total phosphorus loading into North Wilmes Lake through implementation of sediment inactivation practices for a wet (2019) and dry (2022) year. This reduction in phosphorus loading translates to a reduction in the in-lake total phosphorus concentrations. Model results estimate that the 2019 summer average total phosphorus concentration of 71 µg/L would reduce to 44 µg/L with the implementation of sediment treatments and the alum treatment facility; and the 2022 summer average total phosphorus concentration of 57 µg/L would reduce to 27 - 28 µg/L (Table 10-3).

Table 10-3 Summer growing period estimated reductions in phosphorus loads and in-lake concentrations in North Wilmes Lake with the proposed management practices

Scenario	Modeled Summer Total Phosphorus Loading (lbs)					Summer Average Total Phosphorus (µg/L)[1]	Summer Average Chlorophyll-a (µg/L)[1]
	Watershed Inflow	Upstream Lake	Internal Loading	Total	Load Reduction		
2019 Existing	209	41	36	285	--	71	27
2019 Alum Facility	158	41	36	234	-52	57	27
2019 Alum Facility + Sediment Inactiv. Low	158	41	7	205	-80	44	22
2019 Alum Facility + Sediment Inactiv. High	158	41	6	204	-81	44	22
2019 Alum Facility + Sediment Inactiv. Low + Watershed Reductions	129	41	7	176	-109	38	20
2022 Existing	49	0	33	82	--	57	51
2022 Alum Facility	28	0	33	62	-21	45	42
2022 Alum Facility + Sediment Inactiv. Low	28	0	6	35	-48	28	18
2022 Alum Facility + Sediment Inactiv. High	28	0	6	34	-48	27	17

[1] Existing conditions summer average concentrations are calculated from calibrated in-lake models

Based on these results, it's predicted that implementing sediment inactivation practices alone would not be sufficient to reduce summer average concentrations for total phosphorus and chlorophyll-*a* to below the state standards, especially in a wet year. To estimate the additional total phosphorus load reduction needed from watershed management practices to achieve standards for both parameters, we applied a percent reduction to the calibrated watershed loading model results in model year 2019. For this scenario, watershed load reductions were coupled with the Kargel Park alum treatment facility and sediment inactivation (low) assumptions to be conservative. During a wet year (2019), the modeling predicts that an additional 29 pounds of total phosphorus would need to be reduced to meet both total phosphorus and chlorophyll-*a* concentrations in North Wilmes Lake. Figure 10-8 and Figure 10-9 summarize the predicted reductions in in-lake total phosphorus concentrations for the 2019 and 2022 model years, respectively, for each of the proposed BMP scenarios.

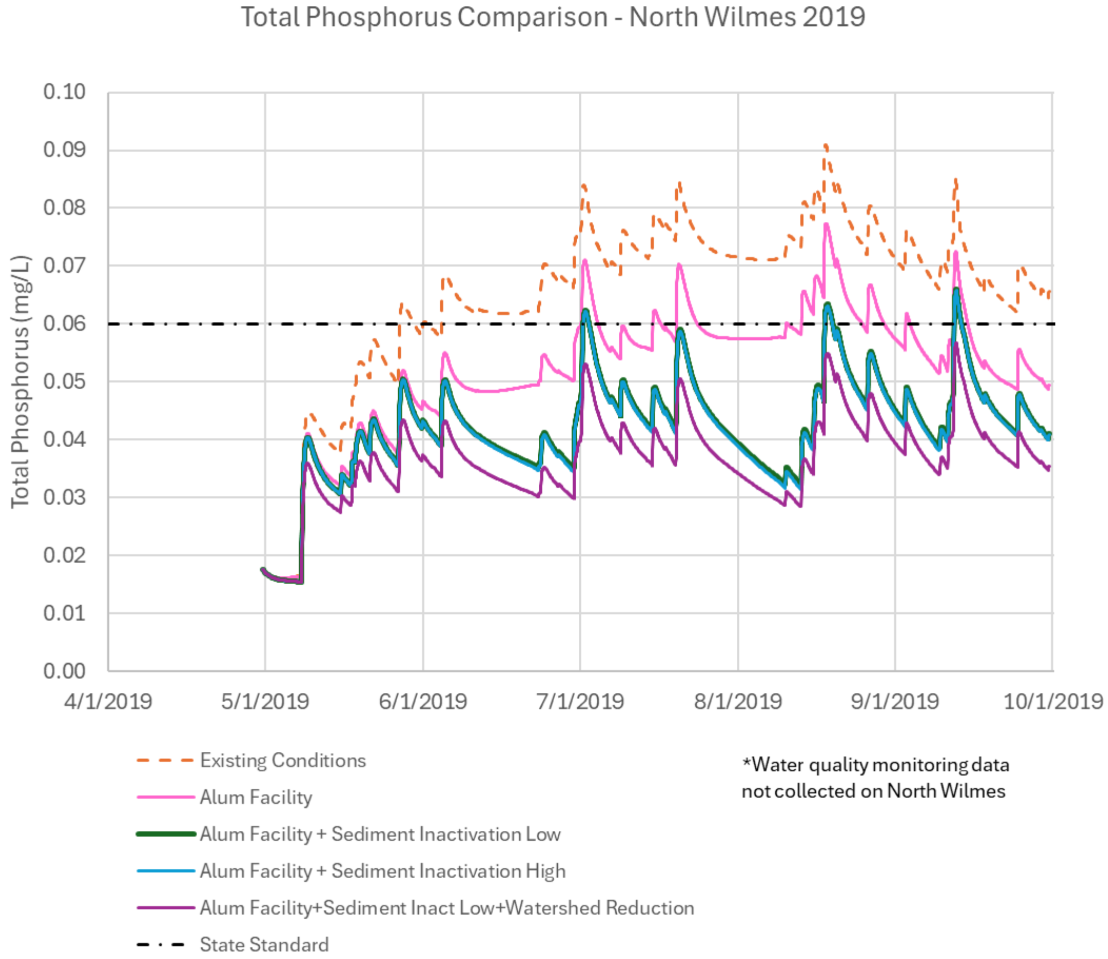


Figure 10-8 Model-predicted reductions in 2019 total phosphorus concentrations in North Wilmes Lake with the proposed management practices

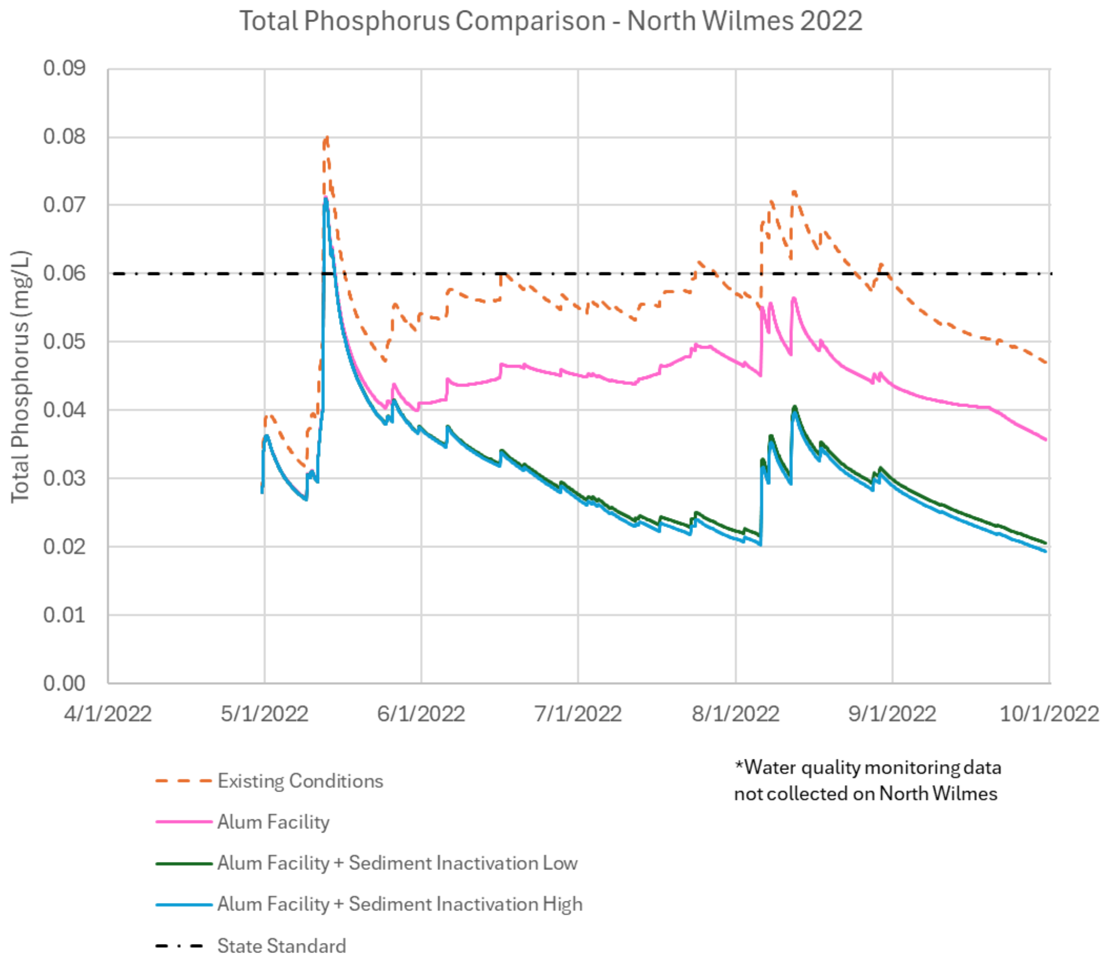


Figure 10-9 Model-predicted reductions in 2022 total phosphorus concentrations in North Wilmes Lake with the proposed management practices

Model results estimating the in-lake response to reduced phosphorus loading—both from internal treatments and the surrounding watershed—reflect the estimated impacts to in-lake conditions during the 2019 and 2022 growing seasons if those practices had been in place at that time. The modeling incorporates a range of assumed sediment-treatment efficacies to account for application variability and the fact that the specific treatment approach (alum, alum/iron, or aeration) has not yet been selected. Assumed efficacies were informed by monitoring data from comparable Barr sediment-inactivation projects one year post-implementation. Monitoring of past projects shows that sediment inactivation efficacy can vary and is expected to decline over time due to product aging, new watershed inflows, and/or burial from rough fish activity. As such, results presented in this report provide an estimate of the expected scale of treatment impacts within a given growing season, but do not account for longer-term changes within the system and how that may impact benefits over time. For more information on the sediment inactivation modeling assumptions see Section 6.4.

10.6.2 South Wilmes Lake

The calibrated in-lake models were used to estimate the impact of implementing sediment treatment projects in South Wilmes Lake. For the South Wilmes Lake scenarios, it was also assumed that all upstream lakes (except for Armstrong Lake) also had internal treatments implemented.

Table 10-4 summarizes the estimated phosphorus load reduction from two scenarios: sediment inactivation (low efficacy) and sediment inactivation (high efficacy). The sediment inactivation scenarios include assumed sediment treatments in both South Wilmes Lake and its upstream contributing waterbodies (except for Armstrong). Results from the in-lake modeling predict a reduction of 28 – 211 pounds of total phosphorus loading into South Wilmes Lake through implementation of sediment inactivation practices for a wet (2019) and dry (2022) year. This reduction in phosphorus loading translates to a reduction in the in-lake total phosphorus concentrations. Model results estimate that the 2019 summer average total phosphorus concentration of 60 µg/L would reduce to 35 - 38 µg/L with the implementation of sediment treatments; and the 2022 summer average total phosphorus concentration of 48 µg/L would reduce to 25 - 27 µg/L (Table 10-4Table 11-5). Figure 10-10 and Figure 10-11summarize the predicted reductions in in-lake total phosphorus concentrations for the 2019 and 2022 model years, respectively, for each of the proposed scenarios.

Table 10-4 Summer growing period estimated reductions in phosphorus loads and in-lake concentrations in South Wilmes Lake with the proposed management practices

Scenario	Modeled Summer Total Phosphorus Loading (lbs)					Summer Average Total Phosphorus (µg/L)[1]	Summer Average Chlorophyll-a (µg/L)[1]
	Watershed Inflow	Upstream Lakes	Internal Loading	Total	Load Reduction		
2019 Existing	35	405	55	495	-	60	28
2019 Sediment Inactivation Low	35	259	14	307	-188	38	20
2019 Sediment Inactivation High	35	239	11	284	-211	35	20
2022 Existing	7	12	32	52	-	48	28
2022 Sediment Inactivation Low	7	8	8	23	-28	27	12
2022 Sediment Inactivation High	7	7	6	21	-31	25	12

[1] Existing conditions summer average concentrations are calculated from calibrated in-lake models

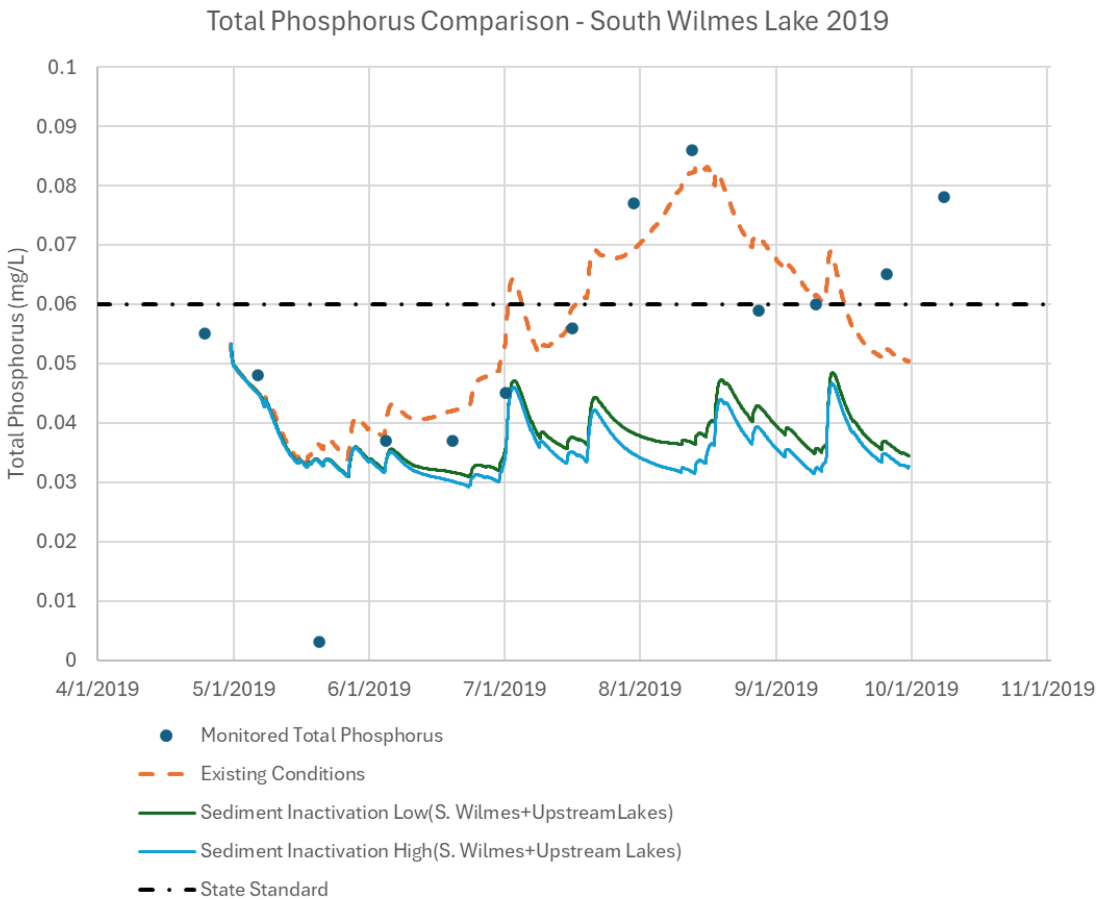


Figure 10-10 Model-predicted reductions in 2019 total phosphorus concentrations in South Wilmes Lake with the proposed management practices

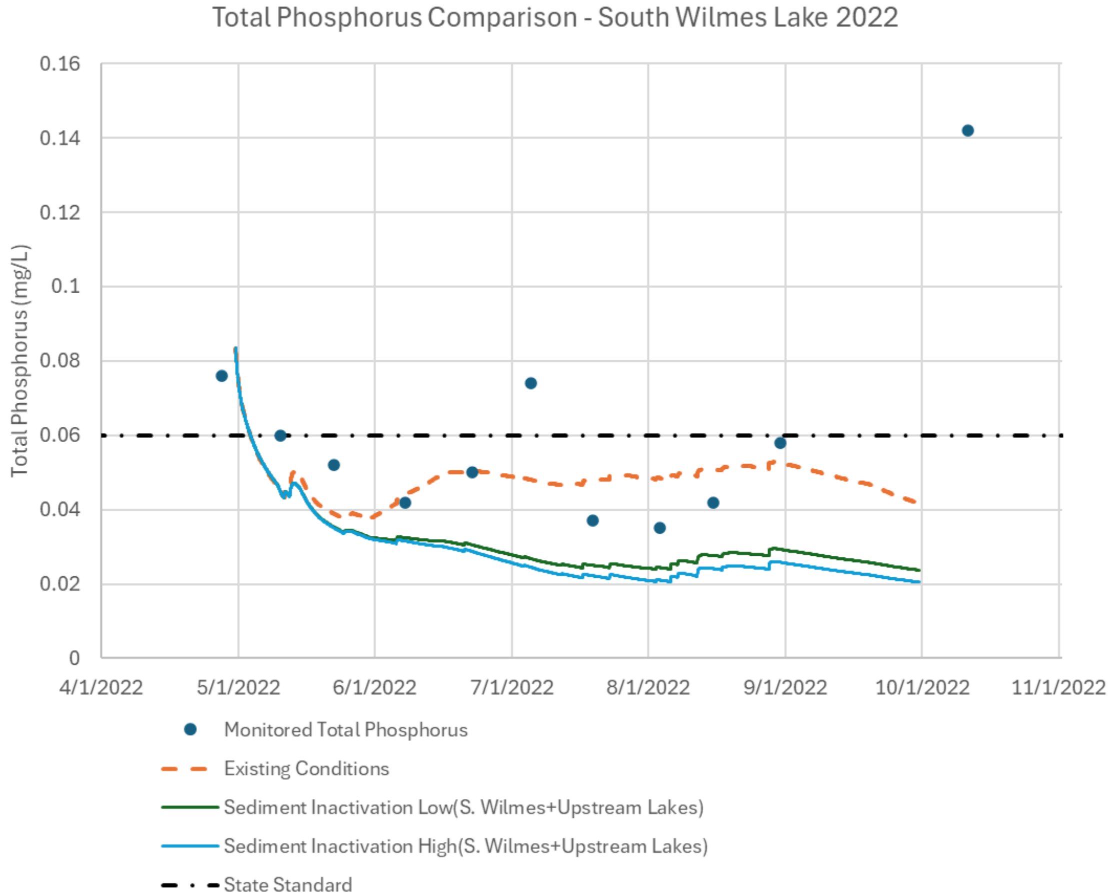


Figure 10-11 Model-predicted reductions in 2022 total phosphorus concentrations in South Wilmes Lake with the proposed management practices

Based on these results, it's predicted that implementing sediment inactivation practices in South Wilmes and its upstream lakes would be sufficient to reduce summer average in-lake concentrations for total phosphorus and chlorophyll-*a* to below the state standards. The in-lake model results represent the expected lake response to a reduction in phosphorus loading—from internal sediments in South Wilmes and its upstream lakes—during the 2019 and 2022 growing seasons if the conceptual management scenarios had been in place. The modeling incorporates a range of assumed sediment-treatment efficacies to account for application variability and the fact that the specific treatment approach (alum, alum/iron, or aeration) has not yet been selected. Assumed efficacies were informed by monitoring data from comparable Barr sediment-inactivation projects one year post-implementation.

Monitoring of past projects shows that sediment inactivation efficacy can vary and is expected to decline over time due to product aging, new watershed inflows, and/or burial from rough fish activity. As such, results presented in this report provide an estimate of the expected scale of treatment impacts within a given growing season, but do not account for longer-term changes within the system and how that may impact benefits over time. For more information on the sediment inactivation modeling assumptions see Section 6.4.

10.7 Management Recommendations

The monitoring data on South Wilmes Lake has been variable in recent years, with some years meeting state water quality standards and other years notably exceeding the standards. No historical water quality monitoring data is available for North Wilmes Lake. Both basins of Wilmes Lake have threatened native aquatic plant communities due to the notable growth of the aquatic invasive species Eurasian water milfoil and curly-leaf pondweed. Given this, future management efforts should focus on improving lake water quality and ecosystem health, monitoring for changes, and continuing water quality and ecosystem health protection measures as improvements are achieved. Table 10-5 summarizes the recommended management strategies that could be considered for Wilmes Lake to help improve water quality conditions and ecological health.

Table 10-5 Wilmes Lake management recommendations

Management/Protection Action		Basis
Address external watershed loads	Enhanced street sweeping program	Continue to work with the City of Woodbury to refine an enhanced street sweeping program to reduce pollutant loading to stormwater runoff
	Stormwater BMPs	Consider retrofitting or installing new stormwater BMPs in subwatersheds that are currently untreated or undertreated Implement site scale BMPs as opportunities arise
	Chloride	Consider applying chloride reduction strategies such as education and implementation assistance to the City of Woodbury and other stakeholders
Address internal loads	Sediment inactivation treatment	Review and consider implementing a sediment inactivation treatment to reduce lake bottom sediment phosphorus loads
Aquatic Plants	Invasive species management	Continue to monitor invasive species growth (e.g., curly-leaf pondweed, Eurasian watermilfoil) Consider continued herbicide management of Eurasian watermilfoil and curly-leaf pondweed to improve native aquatic plant health
	Promote native aquatic plant growth	Encourage native plant reestablishment during and following aquatic invasive species management
Fisheries	Fisheries monitoring	Consider collecting fish community data
Phytoplankton and Zooplankton	Phytoplankton/Zooplankton monitoring	Consider monitoring phytoplankton and zooplankton as part of routine monitoring
Water Quality	Water quality monitoring	Continue monitoring in-lake water quality and assessing for eutrophication within South Wilmes Collect in-lake water quality monitoring data for North Wilmes to assess for eutrophication and validate modeling results Continue to identify/track chloride levels from winter salt use Consider collecting water quality parameters at multiple depths to confirm lake stratification influence on internal phosphorus loading to surface waters

11 Colby Lake

11.1 Water Quality

Colby Lake is located in the City of Woodbury and is used primarily for active and passive recreation and wildlife viewing. When water levels are high enough, water discharges via gravity flow from Colby Lake through a storm sewer system, which ultimately discharges to Bailey Lake.

Colby Lake has a water surface area of approximately 73 acres, a maximum depth of 10 feet, and a mean depth of approximately 5 feet. Colby Lake is shallow enough for aquatic plants to grow over the entire waterbody and for the lake to mix many times per year (polymictic lake).

Shallow/Deep	Shallow
Location	Woodbury
Surface Area	73 acres
Average/Maximum Depth	5 feet / 10 feet
Direct Watershed Area	2,924 acres
Total Watershed Area	8,212 acres
Watershed:Lake Surface Area	112:1
Impairment Status	Impaired for nutrients since 2006
Downstream Waterbody	Bailey Lake

The State of Minnesota uses three water quality parameters to assess eutrophication standards within lakes—total phosphorus, chlorophyll-*a*, and Secchi disk transparency—to assess waterbody health and track water quality changes. These three parameters were measured in Colby Lake by the SWWD between 1994-2023 (Figure 11-1). The summer average total phosphorus concentrations exceeded the state standard for all monitored years. Summer average chlorophyll-*a* concentrations exceeded the state standard for all monitored years except 2023. In 2023, the summer average chlorophyll-*a* concentration was equal to the state standard. Summer average Secchi depths were worse than the state standard for all monitored years except 2019, 2021, and 2023. Colby Lake was added to the Minnesota impaired waters list for nutrients in 2006.

Chloride concentrations were measured by the SWWD between 2013 - 2023 (generally between April and September). In the historical record, all observed chloride concentrations were below the MPCA chronic standard of 230 mg/L. The highest observed chloride concentration was 187 mg/L in May and June 2023.

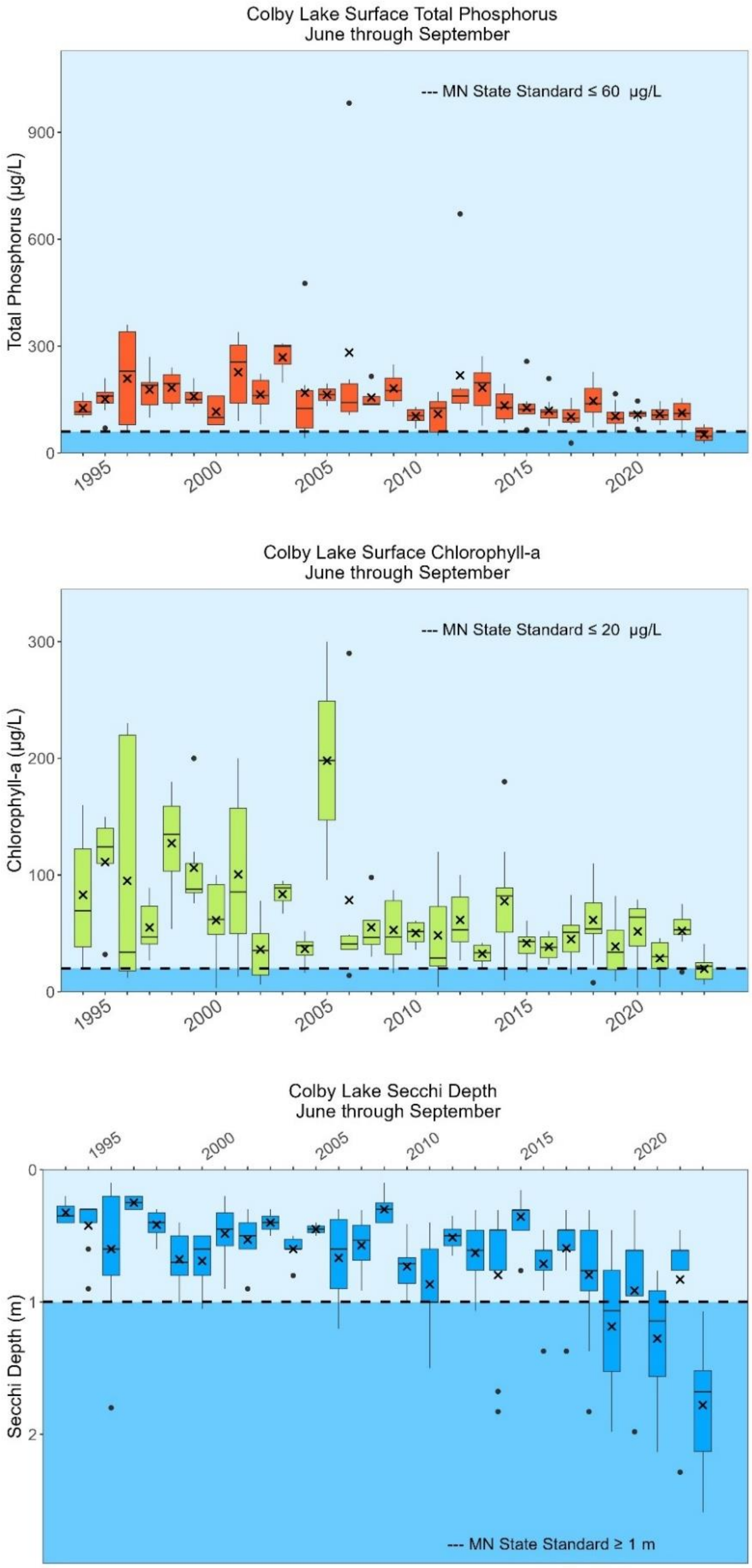


Figure 11-1 Colby Lake eutrophication monitoring data (June – September)
Summer averages are shown by x's in the box plots

11.2 Ecological Health

11.2.1 Aquatic Plants

Table 11-1 summarizes the calculated Lake Plant Eutrophication IBI values for Colby Lake based on point-intercept plant surveys completed in 2021 and 2024. Colby Lake scored below both plant IBI metrics in June and August in 2021 and 2024 indicating a degraded plant community that is likely stressed from cultural eutrophication. Table 11-1 also summarizes the percentage of the littoral area where aquatic plants were found and lists observed aquatic invasive species, their frequency of occurrence (FOO), and current management practices. Other plant health metrics can be referenced in the 2021 and 2024 Aquatic Vegetation Survey Results reports (Stantec, 2021, 2025).

Table 11-1 Colby Lake aquatic plants overview

Lake	Parameter	June 2021[1]	August 2021[1]	June 2024[2]	August 2024[2]	MnDNR Threshold	Invasive Species Management
Colby	Species Richness	10	9	9	10	>11	
	Floristic Quality Index (FQI)	15.2	13.3	11.2	12.3	>17.8	
	% Littoral with Vegetation	99%	82%	71%	91%		
	Curly-leaf Pondweed FOO	89%	P	-	-		Herbicide applications 2022 - 2025
	Eurasian Watermilfoil FOO	40%	29%	3%	2%		Herbicide applications 2022 - 2025

[1] (Stantec, 2021 Aquatic Vegetation Survey Results, 2021)
[2] (Stantec, 2025)

11.2.2 Fisheries

Colby Lake is included in the Fishing in the Neighborhood (FiN) program run by the MNDNR, which is aimed at increasing angling opportunities, public awareness, and environmental stewardship within the seven-county Twin Cities metro area. The MNDNR FiN program has actively been involved in managing the sport fish populations of Colby Lake since 2002, with the goal of providing shore fishing opportunities to the City of Woodbury. As such, the fish community is surveyed at a regular frequency. Fish stocking data from the most recent decade is also available. Table 11-2 summarizes the fish community data from the most recent survey conducted in June 2015 (MNDNR, 2015). Table 11-3 summarizes the fish stocking report from the last decade (MNDNR, 2024). Walleye have been stocked in Colby Lake most recently (i.e., in 2019 and annually between 2022 – 2024).

A winter aeration system was installed in the southeast portion of the lake, near Edgewater Park, in 2012 to provide better winter oxygen conditions and limit the risk of winter fish kills.

Table 11-2 Colby Lake fish survey report – 2015 (MNDNR, LakeFinder, 2015)

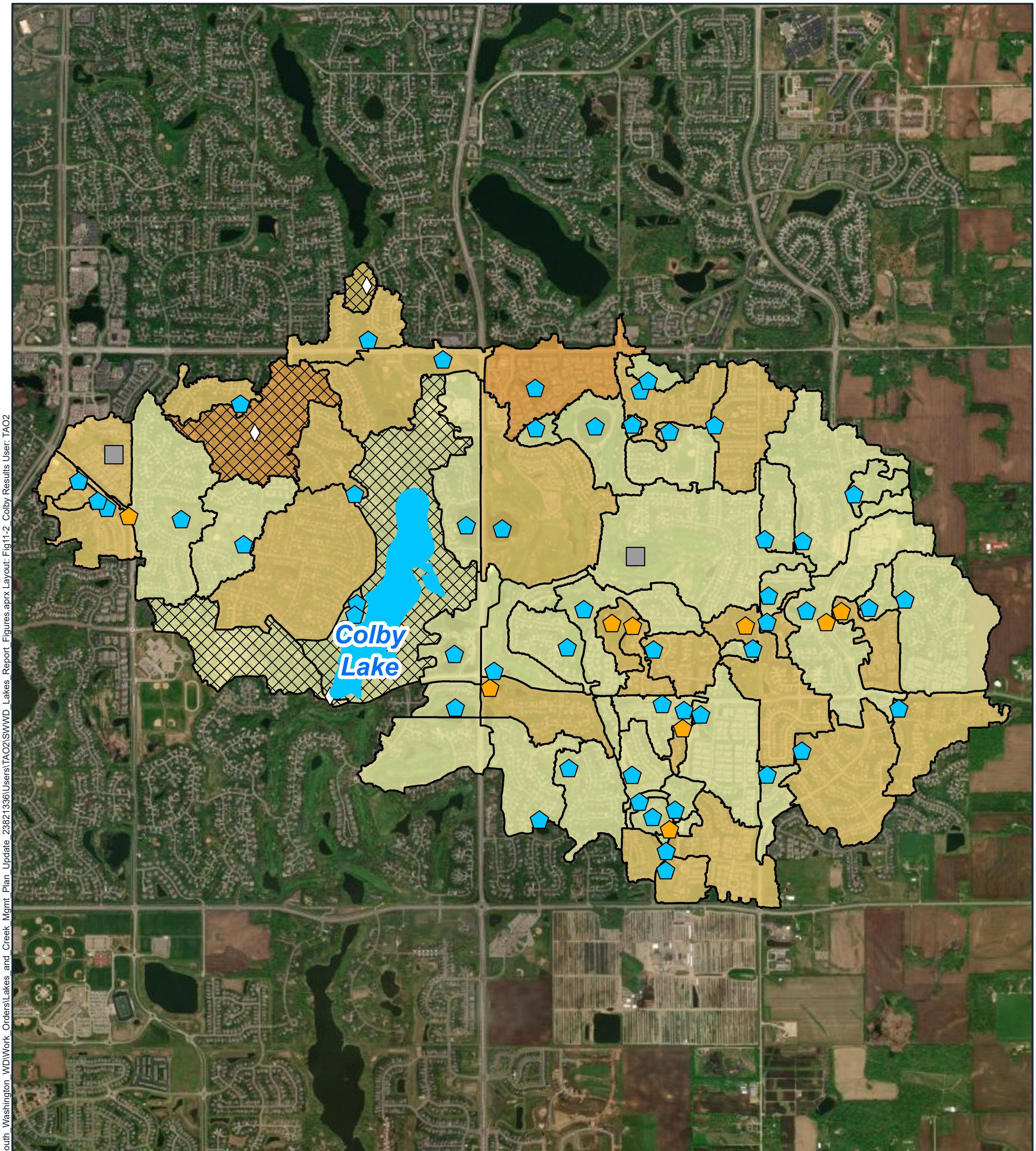
Catch Method	Fish Species	Catch per Unit Effort (CPUE)	Normal CPUE Range	Count
Standard trap nets	Black bullhead	36.3	2.5 – 70.2	254
	Black crappie	1.6	1.3 – 27.7	11
	Bluegill	12.7	2.8 – 43.3	89
	Channel catfish	0.9	N/A	6
	Golden shiner	0.1	0.4 – 3.9	1
	Green sunfish	0.1	0.4 – 3.8	1
	Hybrid sunfish	0.1	N/A	1
	Pumpkinseed	0.7	0.8 – 9.3	5
	Yellow perch	0.4	0.4 – 3.5	3
Standard gill nets	Black bullhead	43.0	8.0 – 90.0	43
	Black crappie	10.0	2.0 – 19.0	10
	Channel catfish	3.0	N/A	3
	Yellow perch	38.0	2.5 – 25.8	38

Table 11-3 Colby Lake fish stocking report – 2015-2024 (MNDNR, LakeFinder, 2024)

Year	Fish Species	Size	Number	Pounds
2024	Walleye	fry	500,000	4.5
2023	Walleye	fry	500,000	4.0
2022	Walleye	fry	400,000	3.3
2019	Walleye	fry	2,000,000	16.2
2016	Channel Catfish		10,000	52.9
	Walleye	fry	420,000	4.3
2015	Channel Catfish	yearlings	2,100	350.0

11.3 Watershed Total Phosphorus Loads

P8 modeling shows low to moderate watershed phosphorus loads to Colby Lake, with higher loads coming from denser residential developments north and east of the lake. Some areas of the drainage area are untreated (e.g., a neighborhood northwest of the lake, and the direct drainage area to the lake) before entering the storm sewer system and reaching the lake. The map in Figure 11-2 shows the effective areal phosphorus load by subwatershed, modeled BMPs, and untreated subwatersheds within the area. The effective phosphorus load represents the loading rate after pollutant removal by stormwater BMPs within the watershed and is reflective of the loading that actually makes it into the receiving waterbody from a given location. The effective phosphorus loads for the Colby Lake subwatersheds range from 0.1 pounds per acre per year to 0.7 pounds per acre per year, based on a 10-year modeling period (2012-2022).



Lake

P8 Device Type

Dry Pond

Wet Pond

General Device

Pipe

Effective TP Load (lbs/ac/yr)

0.0 - 0.3

0.3 - 0.6

0.6 - 1.0

1.0 - 1.5

1.5 - 2.0

> 2.0

Area Not Treated by BMP

N

0 500 1,000

Feet

Colby Lake
Effective TP Loading
Lake Management Plan
South Washington
Watershed District

FIGURE 11-2

BARR

11.4 In-lake Total Phosphorus Loads

Results of the in-lake modeling for Colby Lake showed that during the summer of 2019 (representing a wet year), high total phosphorus loads entered Colby Lake from watershed runoff (37%, 383 lbs) and from the upstream South Wilmes Lake (26%, 263 lbs). A notable internal load was also present in 2019, representing 37% (377 lbs) of the total load (Figure 11-3).

During the drier summer of 2022, the phosphorus loading to Colby Lake from watershed runoff and upstream South Wilmes Lake was notably lower, representing 28% (102 lbs) and 1% (3 lbs) of the phosphorus loading to the lake, respectively. The phosphorus loading from upstream South Wilmes Lake in 2022 was 99% less than the phosphorus load estimated in 2019. Internal loading estimates in 2022 were at a similar magnitude as 2019, representing 71% (261 lbs) of the phosphorus load.

Biweekly profiles collected on Colby Lake indicated that more water column mixing and weaker stratification occurred in late summer 2022 as compared to model year 2019. Dissolved oxygen concentrations were higher at lower depths during this period, possibly re-aerating the sediment and reducing phosphorus loading from the mobile phosphorus fraction in the sediment. Re-aeration of the sediment for a prolonged period could be a reason why there was decreased internal phosphorus load observed in the 2022 Colby Lake model. Phosphorus fractionation data gathered from sediment cores collected in 2024 indicated a high potential for internal phosphorus loading. Additional details on the sediment core data can be found in Section 4.8.

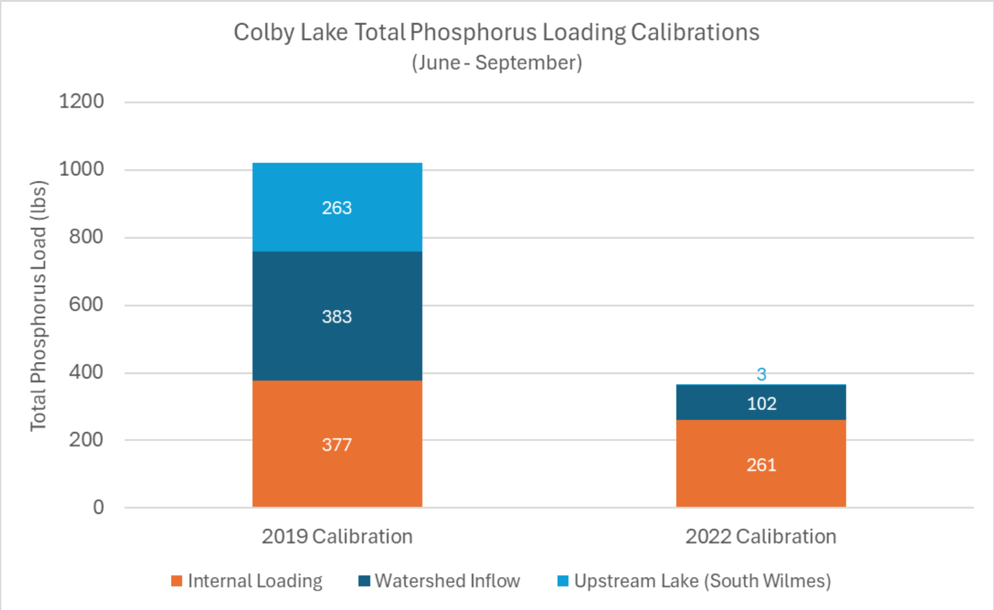


Figure 11-3 Colby Lake 2019 & 2022 estimated total phosphorus loads (June – Sept)

11.5 Nitrogen and Algal Growth

Throughout the growing season, various factors can influence the rate and volume of algae growth within a lake, such as phosphorus, nitrogen, light, and temperature. The Barr Shallow Lake Model was used for simulating water quality within Colby Lake; this model utilizes Michaelis Menten kinetics to determine which factor or combination of factors limit algae growth throughout the modeled time period. Although nitrate, nitrite, ammonia, and ammonium water quality data were not available for model calibration, the effect of nitrogen on chlorophyll-*a* growth can be deduced for Colby Lake using the model. In both the 2019 and 2022 models, accounting for nitrogen limitation as part of the model calibration improved model results with respect to simulating chlorophyll-*a*. Nitrogen limitation was estimated to be most prominent in August 2019 and July – September 2022.

Nitrogen limitation during a portion of the growing season has been seen in numerous other shallow lake modeling efforts performed by Barr. Nitrogen limitation tends to occur because the growth rate of algae can be greater than the degradation rate of nitrogen (i.e., slow decomposition from organic nitrogen forms to ammonia and/or slow nitrification rate changing ammonia to nitrite and nitrate). In other words, algae will continue to grow until they have used up bioavailable forms of nitrogen in the water column, and once depleted, algae growth will be limited. However, nitrogen limitation does not affect all algal species equally. Certain species of cyanobacteria can persist under nitrogen-limited conditions, even as other algal species decline or disappear entirely. This resilience is due to the ability of some cyanobacteria species to fix atmospheric nitrogen by converting N_2 gas into ammonia. The process is energy-intensive, though, and will be used only when necessary. This competitive advantage complicates efforts to predict how nitrogen limitation will influence overall algal growth dynamics in a waterbody. The presence of cyanobacteria with nitrogen fixing abilities could not be confirmed as part of this study, as no phytoplankton monitoring data is currently available for this lake.

The algal growth limitations identified during model calibration reflect which parameters are currently limiting growth. As management is implemented, algal growth limitations will likely change. For example, if phosphorus loading from lake bottom sediment is reduced, phosphorus limitation may become more dominant throughout the growing season. Similarly, a reduced phytoplankton population with phosphorus control could also increase nitrogen concentrations (e.g., reduced nitrogen demand) and alleviate the nitrogen limitation and also eliminate a competitive advantage of cyanobacteria. Outcomes of changes in nitrogen dynamics are more challenging to predict than phosphorus management with the information currently available.

11.6 Predicted Benefits from Implementing Management Practices

One of the study goals was to estimate the potential impact of lake sediment inactivation projects and how they could be used to improve in-lake water quality. The calibrated in-lake models were used to estimate the impact of implementing sediment treatment projects in Colby and upstream lakes (except for Armstrong Lake). For Colby Lake, watershed load reductions were also analyzed to assess joint impacts on further improving in-lake conditions to meet state standards. Specific watershed BMPs were not modeled as a part of this study. Instead, a percent reduction was applied to the calibrated watershed loading estimates across the watersheds directly tributary to Colby Lake.

Table 11-4 summarizes the estimated phosphorus load reduction from three scenarios: sediment inactivation (low and high efficacies) and sediment inactivation (low) coupled with watershed load reductions. The sediment inactivation scenarios include assumed sediment treatments in both Colby Lake and its upstream contributing waterbodies (except for Armstrong). Results from the in-lake modeling predict a reduction of 176 – 399 pounds of total phosphorus loading into Colby Lake through implementation of sediment inactivation practices for a wet (2019) and dry (2022) year. This reduction in phosphorus loading translates to a reduction in the in-lake total phosphorus concentrations. Model results estimate that the 2019 summer average total phosphorus concentration of 101 µg/L would reduce to 52 - 60 µg/L with the implementation of sediment treatments; and the 2022 summer average total phosphorus concentration of 115 µg/L would reduce to 51 - 64 µg/L (Table 11-4).

Table 11-4 Summer growing period estimated reductions in phosphorus loads and in-lake concentrations in Colby Lake with the proposed management practices

Scenario	Modeled Summer Total Phosphorus Loading (lbs)					Summer Average Total Phosphorus (µg/L)[1]	Summer Average Chlorophyll-a (µg/L)[1]
	Watershed Inflow	Upstream Lakes	Internal Loading	Total	Load Reduction		
2019 Existing	383	263	377	1,023	-	101	37
2019 Sediment Inactivation Low	383	155	144	682	-341	60	25
2019 Sediment Inactivation High	383	147	94	624	-399	52	24
2019 Sed Inact. Low + Watershed Reduction	179	155	144	478	-545	47	20
2022 Existing	102	3	261	367	-	115	58
2022 Sediment Inactivation Low	102	2	87	190	-176	64	25
2022 Sediment Inactivation High	102	2	59	163	-204	51	22
2022 Sed Inact. Low + Watershed Reduction	60	2	87	149	-218	58	20

[1] Existing conditions summer average concentrations are calculated from calibrated in-lake models

Based on these results, it's predicted that implementing sediment inactivation practices alone would not be sufficient to reduce Colby Lake summer average concentrations for total phosphorus and chlorophyll-*a* to below the state standards. To estimate the additional total phosphorus reductions needed from watershed management practices to achieve standards for both parameters, we applied a percent reduction to the calibrated watershed loading model results in each model year. For this scenario, watershed load reductions were coupled with the sediment inactivation (low) assumptions to be conservative. During a wet year (2019), modeling predicts that an additional 204 pounds of total phosphorus would need to be reduced from the watershed to meet both total phosphorus and chlorophyll-*a* concentrations in Colby Lake. Modeling indicates that during a dry year (2022), an additional 42 pounds of total phosphorus would be required from the watershed to meet total phosphorus and chlorophyll-*a* concentrations. Figure 11-4 and Figure 11-5 summarize the predicted reductions in in-lake total phosphorus concentrations for the 2019 and 2022 model years, respectively, for each of the proposed BMP scenarios.

Model results estimating the in-lake response to reduced phosphorus loading—both from internal treatments and the surrounding watershed—reflect the estimated impacts to in-lake conditions during the

2019 and 2022 growing seasons if those practices had been in place at that time. The modeling incorporates a range of assumed sediment-treatment efficacies to account for application variability and the fact that the specific treatment approach (alum, alum/iron, or aeration) has not yet been selected. Assumed efficacies were informed by monitoring data from comparable Barr sediment-inactivation projects one year post-implementation. Monitoring of past projects shows that sediment inactivation efficacy can vary and is expected to decline over time due to product aging, new watershed inflows, and/or burial from rough fish activity. As such, results presented in this report provide an estimate of the expected scale of treatment impacts within a given growing season, but do not account for longer-term changes within the system and how that may impact benefits over time. For more information on the sediment inactivation modeling assumptions see Section 6.4.

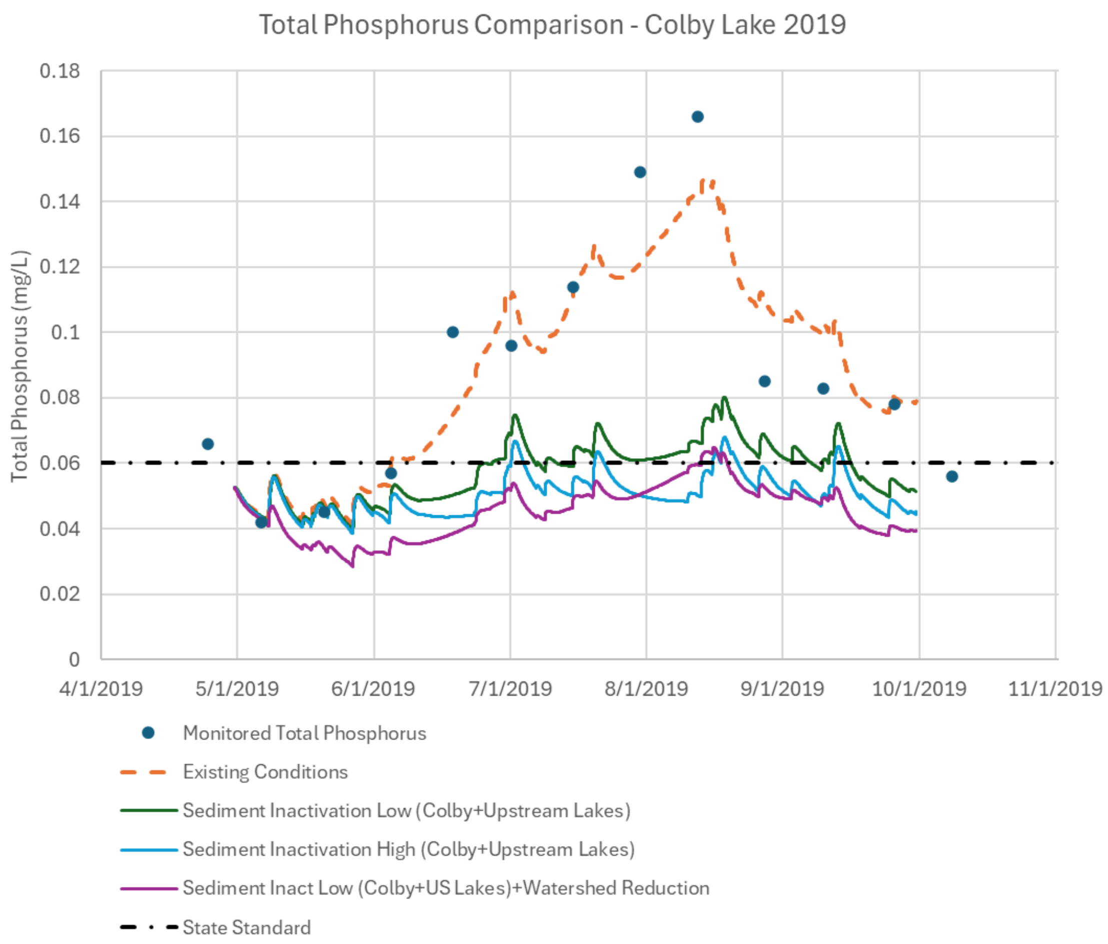


Figure 11-4 **Model-predicted reductions in 2019 total phosphorus concentrations in Colby Lake with the proposed management practices**

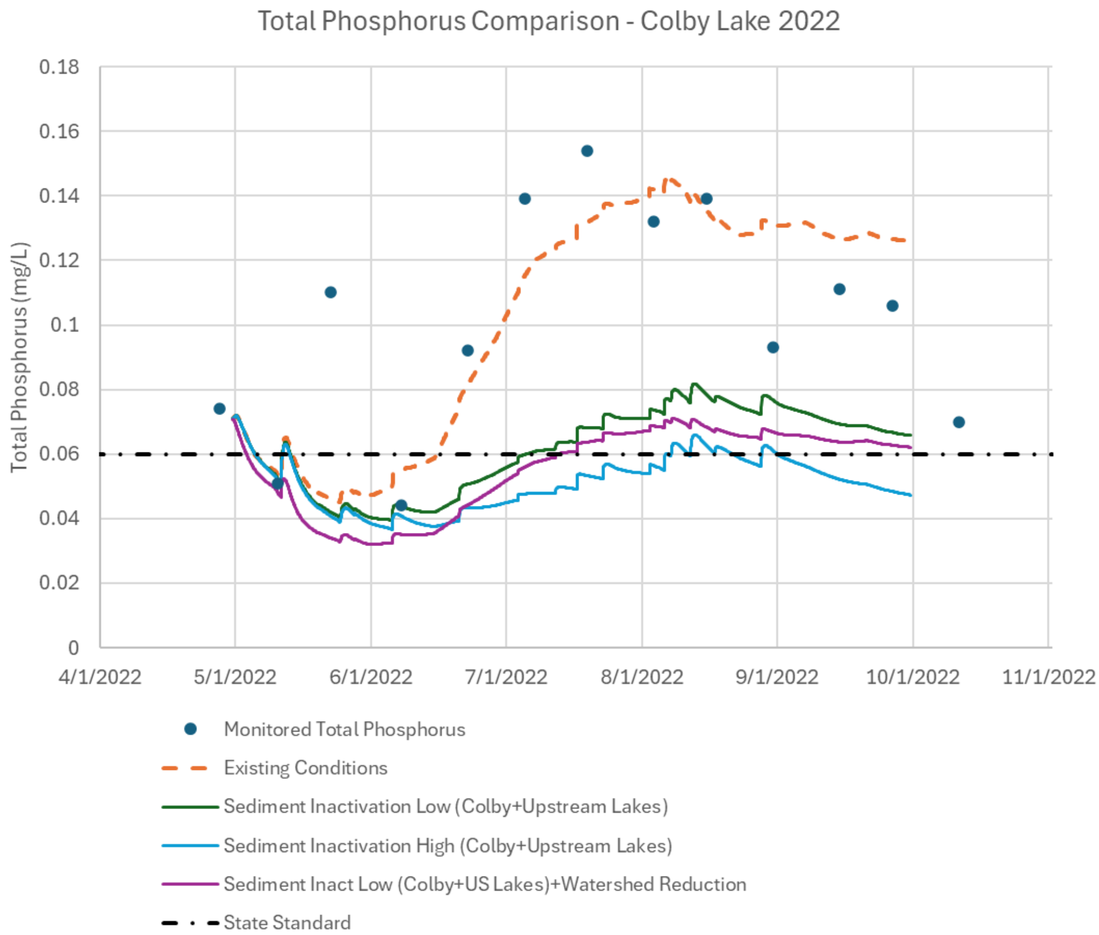


Figure 11-5 **Model-predicted reductions in 2022 total phosphorus concentrations in Colby Lake with the proposed management practices**

11.7 Management Recommendations

Monitoring data from within Colby Lake indicates degraded water quality and shows that the lake has largely not met water quality standards over the course of its monitored history. Colby Lake also has a threatened native aquatic plant community due to the notable growth of the aquatic invasive species Eurasian water milfoil and curly-leaf pondweed. Given this, future management efforts should focus on improving lake water quality and ecosystem health, monitoring for changes, and continuing water quality and ecosystem health protection measures as improvements are achieved. Table 11-5 summarizes the recommended management strategies that could be considered for Colby Lake to help improve water quality conditions and ecological health.

Table 11-5 Colby Lake management recommendations

Management/Protection Action		Basis
Address external watershed loads	Enhanced street sweeping program	Continue to work with the City of Woodbury to refine an enhanced street sweeping program to reduce pollutant loading to stormwater runoff
	Stormwater BMPs	Consider retrofitting or installing new stormwater BMPs in subwatersheds that are currently un-treated or undertreated Implement site scale BMPs as opportunities arise
	Chloride	Consider applying chloride reduction strategies such as education and implementation assistance to the City of Woodbury and other stakeholders
Address internal loads	Sediment inactivation treatment	Review and implement a sediment inactivation treatment to reduce lake bottom sediment phosphorus loads
Aquatic Plants	Invasive species management	Continue to monitor invasive species growth (e.g., curly-leaf pondweed, Eurasian watermilfoil) Consider continued herbicide management of Eurasian watermilfoil and curly-leaf pondweed to improve native aquatic plant health
	Promote native aquatic plant growth	Encourage native plant reestablishment during and following aquatic invasive species management
Phytoplankton and Zooplankton	Phytoplankton/Zooplankton monitoring	Consider monitoring phytoplankton and zooplankton as part of routine monitoring
Water Quality	Water quality monitoring	Continue monitoring in-lake water quality and assessing for eutrophication within Colby Lake Consider supplementing routine monitoring with additional nitrogen parameters for better assessment of phytoplankton nitrogen limitation. Continue to identify/track chloride levels from winter salt use

12 Bailey Lake

12.1 Water Quality

Shallow/Deep	Shallow
Location	Woodbury
Surface Area	62 acres
Average/Maximum Depth	6 feet / 19 feet
Direct Watershed Area	6,031 acres
Total Watershed Area	14,243 acres
Watershed:Lake Surface Area	230:1
Impairment Status	Impaired for nutrients since 2024
Downstream Waterbody	Central Draw Storage Facility

Bailey Lake is located in the City of Woodbury and is used primarily for active and passive recreation and wildlife viewing. When water levels are high enough, water discharges via gravity flow from Bailey Lake to a downstream stormwater pond. Water is then pumped from this pond into the Central Draw Storage Facility, which allows the water to infiltrate into the soil and recharges the groundwater.

Bailey Lake has a water surface area of approximately 62 acres, a maximum depth of 19 feet, and a mean depth of approximately 6 feet. Bailey Lake falls under the MNDNR classification of a "shallow lake" given that the lake's littoral area is greater than 80% the total lake area. Bailey Lake is deep enough that prolonged periods of lake stratification occur during the growing season. Profile monitoring data collected within the lake supports this finding.

The State of Minnesota uses three water quality parameters to assess eutrophication standards within lakes—total phosphorus, chlorophyll-*a*, and Secchi disk transparency—to assess waterbody health and track water quality changes. These three parameters were measured in Bailey Lake by the SWWD between 2017-2023 (Figure 12-1). The summer average total phosphorus concentrations exceeded the state standard for all monitored years except 2023. Summer average chlorophyll-*a* concentrations exceeded the state standard for all monitored years. Summer average Secchi depths were worse than the state standard between 2017 – 2020, but were better than the state standard between 2021 - 2023. Bailey Lake was added to the Minnesota impaired waters list for nutrients in 2024.

Chloride concentrations were measured by the SWWD between 2017 - 2023 (generally between April and September). In the historical record, all observed chloride concentrations were below the MPCA chronic standard of 230 mg/L. The highest observed chloride concentration was 181 mg/L in May 2019.

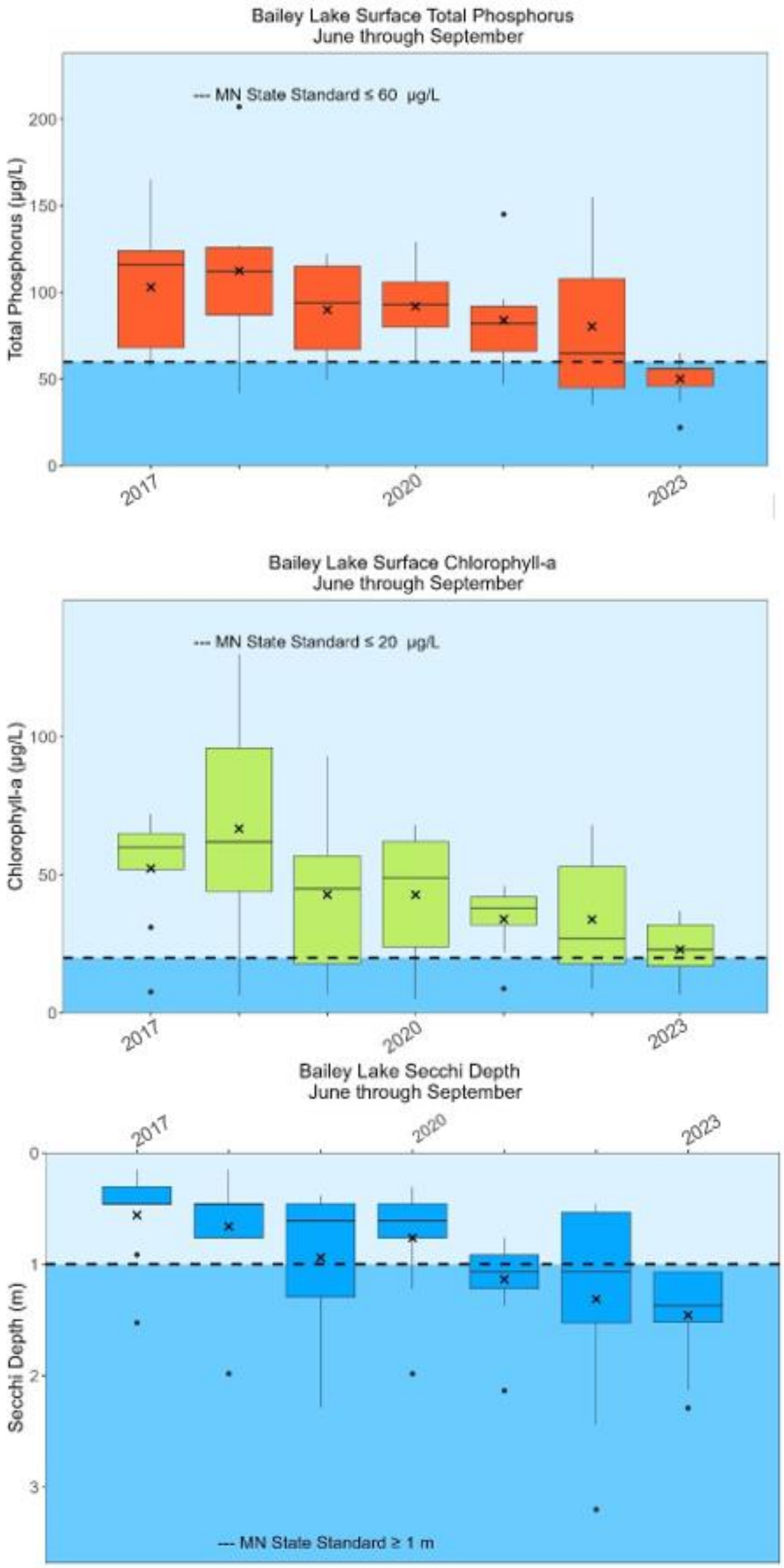


Figure 12-1 Bailey Lake eutrophication monitoring data (June – September)
Summer averages are shown by x's in the box plots

12.2 Ecological Health

12.2.1 Aquatic Plants

Table 12-1 summarizes the calculated Lake Plant Eutrophication IBI values for Bailey Lake based on point-intercept plant surveys completed in 2021 and 2024. Bailey Lake scored above the MNDNR threshold for species richness, but below the threshold for floristic quality in June and August 2021 and June 2024. In August 2024 Bailey Lake scored below both plant IBI metrics indicating a degraded plant community that is likely stressed from cultural eutrophication. Table 12-1 also summarizes the percentage of the littoral area where aquatic plants were found and lists observed aquatic invasive species, their frequency of occurrence (FOO), and current management practices, if applicable. As of this study, SWWD had not actively managed Eurasian watermilfoil, curly-leaf pondweed, or common reed on Bailey Lake. Other plant health metrics can be referenced in the 2021 and 2024 Aquatic Vegetation Survey Results reports (Stantec, 2021, 2025).

Table 12-1 Bailey Lake aquatic plants overview

Lake	Parameter	June 2021[1]	August 2021[1]	June 2024[2]	August 2024[2]	MnDNR Threshold	Invasive Species Management
Bailey	Species Richness	12	12	11	10	>11	
	Floristic Quality Index (FQI)	13.3	14.4	14.3	13.0	>17.8	
	% Littoral with Vegetation	84%	68%	79%	70%		
	Curly-leaf Pondweed FOO	70%	11%	21%	-		No management to date
	Eurasian Watermilfoil FOO	7%	11%	4%	3%		No management to date
	Common Reed FOO	2%	2%	-	-		No management to date

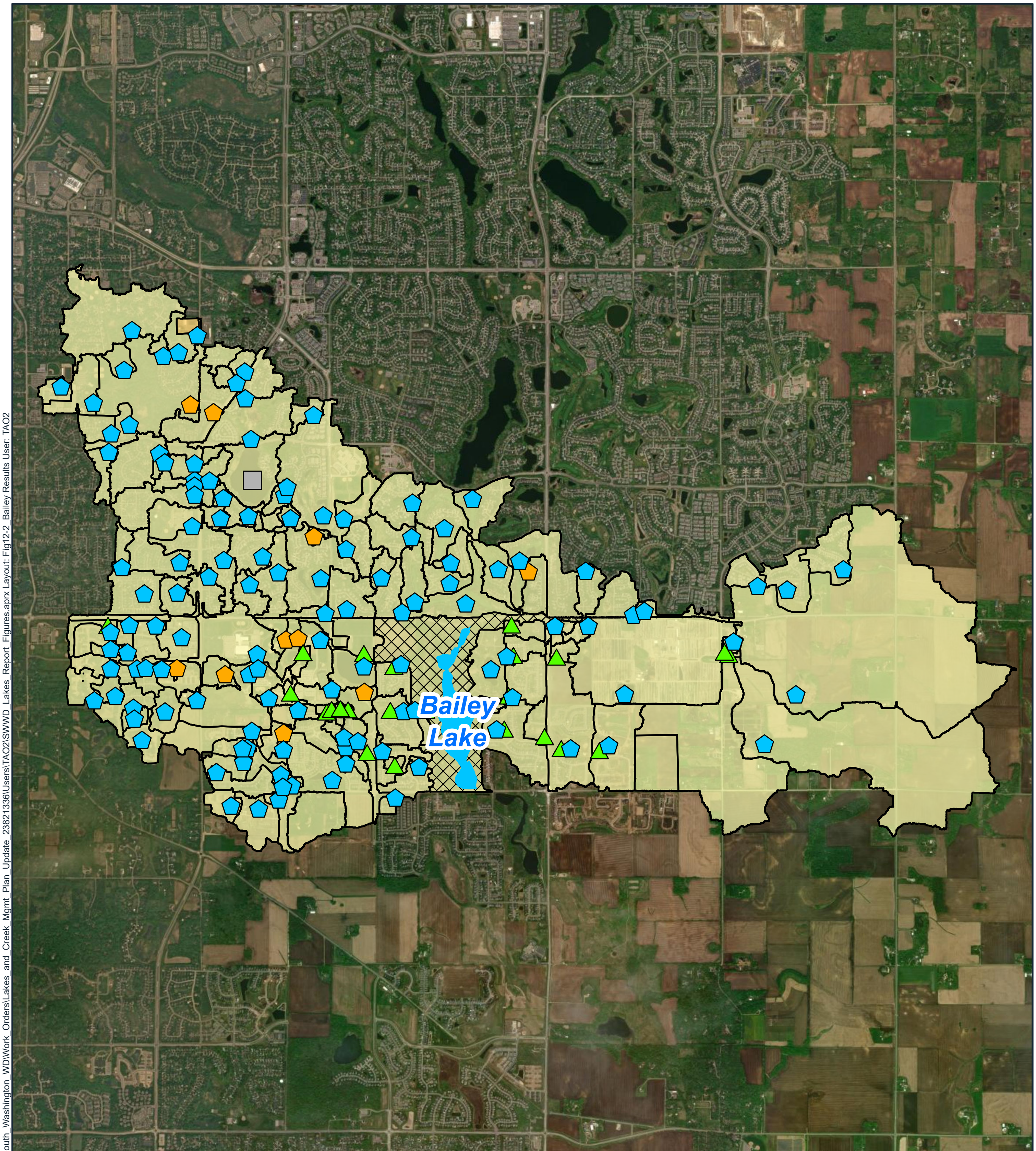
[1] (Stantec, 2021 Aquatic Vegetation Survey Results, 2021)
[2] (Stantec, 2025)

12.2.2 Fisheries

There are no MNDNR fish survey or stocking data available for Bailey Lake.

12.3 Watershed Total Phosphorus Loads

P8 modeling shows low watershed phosphorus loads to Bailey Lake, with the majority of the watershed being devoted to single-family home developments or undeveloped. Many of the developments in this watershed include large infiltration basins that capture and prevent stormwater runoff from reaching Bailey Lake for an average annual rainfall event. The direct drainage area to the lake is currently untreated. The map in Figure 12-2 shows the effective areal phosphorus load by subwatershed, modeled BMPs, and untreated subwatersheds. The effective phosphorus load represents the loading rate after pollutant removal by stormwater BMPs within the watershed and is reflective of the loading that actually makes it into the receiving waterbody from a given location. The effective phosphorus loads for the Bailey Lake subwatersheds range from 0.01 pounds per acre per year to 0.4 pounds per acre per year, based on a 10-year modeling period (2012-2022).



Lake

P8 Device Type

- Dry Pond
- General Device
- Infiltration Basin
- Pipe
- Wet Pond

Effective TP Load (lbs/ac/yr)

- 0.0 - 0.3
- 0.3 - 0.6
- 0.6 - 1.0
- 1.0 - 1.5
- 1.5 - 2.0
- > 2.0
- Area Not Treated by BMP

N

0 2,000 4,000

Feet

Bailey Lake
Effective TP Loading
Lake Management Plan
South Washington Watershed District

FIGURE 12-2

BARR

12.4 In-lake Total Phosphorus Loads

Biweekly profile monitoring data is typically collected in Bailey Lake between April and October of each year, including the collection of temperature and dissolved oxygen concentrations. Lake profile data indicates that Bailey Lake can have temperature stratification as early as late April or early May most years, and typically remains stratified until fall. As such, a stratified in-lake model approach was used for Bailey Lake, and phosphorus inputs to the fully mixed epilimnion volume (surface waters) were estimated for this study (Figure 12-3). The profile monitoring data was used to quantify the volume of the epilimnion throughout the model period. Phosphorus load estimates to the hypolimnion of the lake were not estimated as a part of this study.

Results of the in-lake modeling for Bailey Lake showed that during the summer of 2019 (representing a wet year), phosphorus loading to Bailey Lake was dominated by upstream lake inputs, representing 76% (603 lbs) of the phosphorus loading from major sources (Figure 12-3). Smaller, but still significant, total phosphorus loads were from internal loading (14%, 110 lbs) and watershed runoff (10%, 78 lbs). The reported internal load from lake bottom sediment represents the phosphorus load from shallow sediment located in the epilimnion zone as well as phosphorus that migrated to the epilimnion from the hypolimnion (deep waters) via diffusion across the thermocline or during mixing events.

During the drier summer of 2022, phosphorus loading to Bailey Lake from the upstream Colby Lake was notably lower, representing 8% (15 lbs) of the phosphorus loading to the lake. The phosphorus loading from Colby Lake in 2022 was 98% less than the phosphorus load estimated in 2019. Total phosphorus loading from watershed runoff was also notably lower in 2022, representing 10% (17 lbs) of the load to Bailey Lake. Internal loading estimates in 2022 were similar to 2019, representing 82% (146 lbs) of the phosphorus load.

Differences in annual lake stratification can influence lake mixing potential as well as the anoxic area over lake bottom sediment, which ultimately influences the potential total phosphorus load that reaches surface waters from lake bottom sediment. Given that the monitoring data demonstrates strong lake stratification during the summer, the collection of water quality parameters at multiple depths is recommended to better estimate internal loading influence on surface water conditions (e.g., collecting phosphorus near the surface, at the thermocline, and near the bottom). Understanding how lake stratification influences internal loading will be essential for developing effective sediment inactivation strategies and demonstrating benefits should the District decide to pursue this type of treatment. The collection of water quality parameters at multiple depths is especially important as sediment core phosphorus fractionation data indicate a high potential for phosphorus release from lake bottom sediment (Section 4.8).

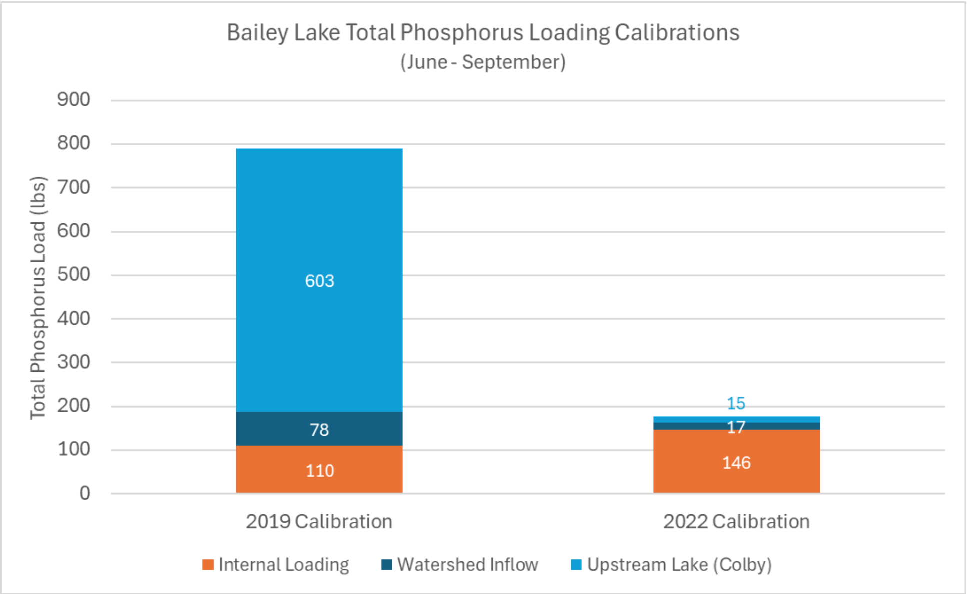


Figure 12-3 Bailey Lake total phosphorus load estimates – 2019 & 2022

12.5 Predicted Benefits from Implementing Management Practices

One of the study goals for Bailey Lake was to estimate the potential in-lake water quality benefits if upstream Colby Lake met state standards. The two Colby Lake proposed model scenarios that were used in the Bailey Lake models were:

- 2019 – Sediment inactivation (low efficacy) for all upstream lakes (except Armstrong) coupled with watershed loading reductions
- 2022 – Sediment inactivation (low efficacy) for all upstream lakes (except Armstrong) coupled with watershed loading reductions

Table 12-2 summarizes the estimated phosphorus load reduction to Bailey Lake if upstream Colby Lake met state standards for both total phosphorus and chlorophyll-a, represented by the scenarios noted above. Results from the in-lake modeling predict a reduction of 331 and 7 pounds of total phosphorus loading into Bailey Lake for a wet (2019) and dry (2022) year, respectively. This reduction in phosphorus loading translates to a reduction in the in-lake total phosphorus concentrations. Model results estimate that the 2019 summer average total phosphorus concentration in Bailey Lake of 84 µg/L would reduce to 51 µg/L with Colby Lake meeting state standards; and the 2022 summer average total phosphorus concentration of 77 µg/L would reduce to 71 µg/L (Table 12-2). Figure 12-4 and Figure 12-5 summarize the predicted reductions in in-lake total phosphorus concentrations for the 2019 and 2022 model years, respectively.

Model results estimating the in-lake response of Bailey Lake to reduced phosphorus loading reflect the estimated impacts that would have occurred during the 2019 and 2022 growing seasons given the assumptions noted above. The results presented provide an estimate of the expected scale of treatment impacts within a given growing season, but do not account for longer-term changes within the system and how that may impact benefits over time. For more information on the sediment inactivation modeling in Colby Lake see Section 11.6 and Section 6.4.

Table 12-2 Summer growing period estimated reductions in phosphorus loads and in-lake concentrations in Bailey Lake with the proposed management practices

Scenario	Modeled Summer Total Phosphorus Loading (lbs)					Summer Average Total Phosphorus (µg/L)[1]
	Watershed Inflow	Upstream Lakes	Internal Loading	Total	Load Reduction	
2019 Existing	78	603	110	791	-	84
2019 Colby Meeting State Standards	78	272	110	460	-331	51
2022 Existing	17	15	146	178	-	77
2022 Colby Meeting State Standards	17	8	146	171	-7	71

[1] Existing conditions summer average concentrations are calculated from calibrated in-lake models

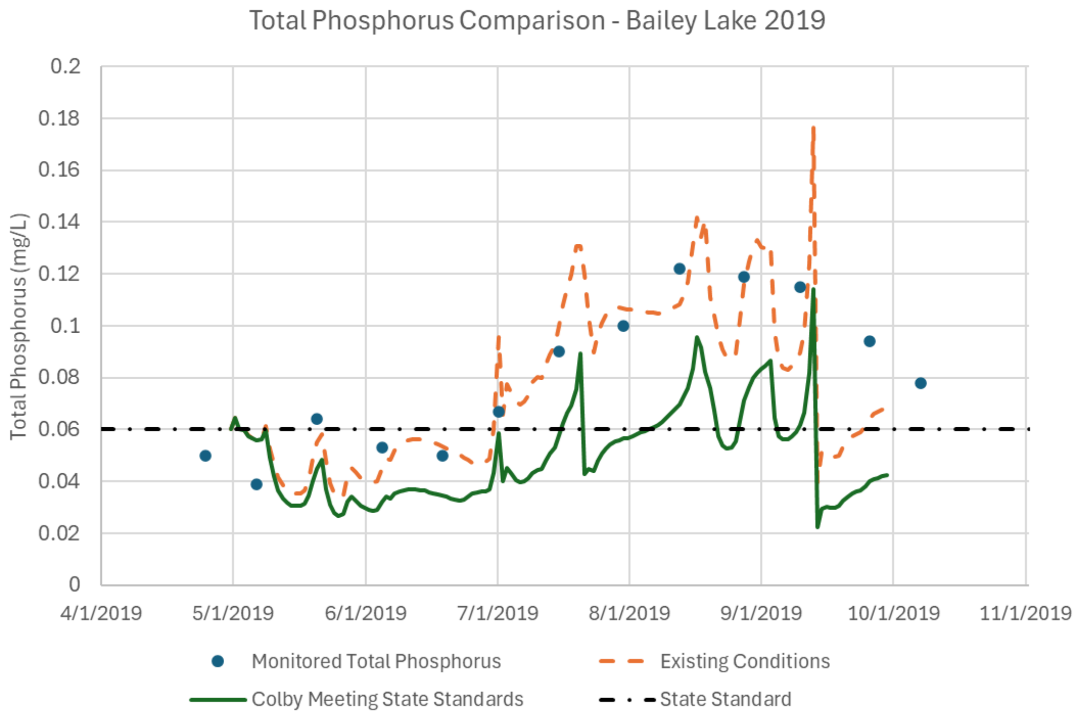


Figure 12-4 **Model-predicted reductions in 2019 total phosphorus concentrations in Bailey Lake with the proposed management practices**

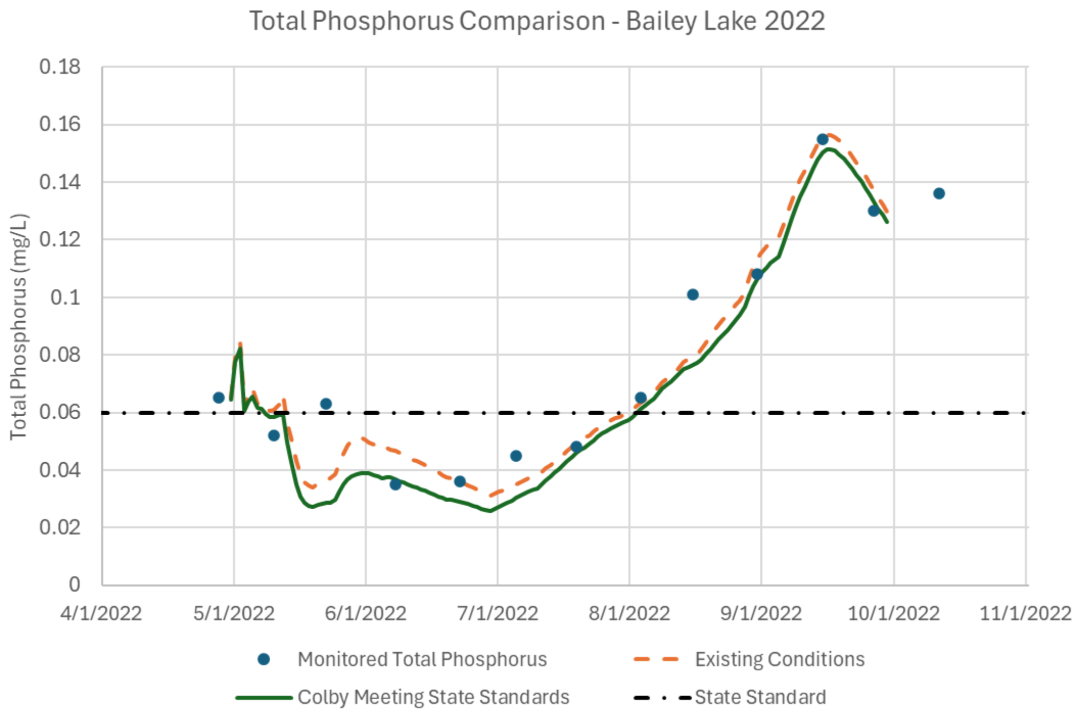


Figure 12-5 **Model-predicted reductions in 2022 total phosphorus concentrations in Bailey Lake with the proposed management practices**

12.6 Management Recommendations

Monitoring of Bailey Lake indicates degraded water quality and shows that throughout the historical record, the lake has largely not met water quality standards. Bailey Lake also has a threatened native aquatic plant community due to the notable growth of the aquatic invasive species Eurasian water milfoil and curly-leaf pondweed. Given this, future management efforts should focus on improving lake water quality and ecosystem health, monitoring for changes, and continuing water quality and ecosystem health protection measures as improvements are achieved. At this time, the SWWD has indicated that they would be most interested in focusing on nutrient reductions into Bailey from upstream lakes and the surrounding watershed before considering the potential for an in-lake sediment treatment. Recommendations in Table 12-3 reflect this strategy. Table 12-3 summarizes the recommended management strategies that could be considered for Bailey Lake to help improve water quality conditions and ecological health.

Table 12-3 Bailey Lake management recommendations

Management/Protection Action		Basis
Address external watershed loads	Enhanced street sweeping program	Continue to work with the City of Woodbury to refine an enhanced street sweeping program to reduce pollutant loading to stormwater runoff
	Stormwater BMPs	As the watershed develops, continue to enforce SWWD rules and consider implementing site scale BMPs as opportunities arise
	Chloride	Consider applying chloride reduction strategies such as education and implementation assistance to the City of Woodbury and other stakeholders
Address upstream lake water quality	Upstream lake water quality	Continue work to improve water quality in the lakes upstream and draining into Bailey Lake.
Aquatic Plants	Invasive species management	Continue to monitor invasive species growth (e.g., curly-leaf pondweed, Eurasian watermilfoil, common reed) Consider herbicide management of Eurasian watermilfoil and curly-leaf pondweed to improve native aquatic plant health
	Promote native aquatic plant growth	Encourage native plant reestablishment during and following aquatic invasive species management
Fisheries	Fisheries Survey	Consider collecting fish community data
Phytoplankton and Zooplankton	Phytoplankton/Zooplankton monitoring	Consider monitoring phytoplankton and zooplankton as part of routine monitoring
Water Quality	Water quality monitoring	Continue monitoring in-lake water quality and assessing for eutrophication within Bailey Lake. Continue to identify/track chloride levels from winter salt use Consider collecting water quality parameters at multiple depths to confirm lake stratification influence on internal phosphorus loading to surface waters

13 La Lake

13.1 Water Quality

Shallow/Deep	Shallow
Location	Woodbury
Surface Area	50 acres
Average/Maximum Depth	5 feet / 10 feet
Watershed Area* **Post BMP Construction*	133 acres
Watershed:Lake Surface Area	3:1
Impairment Status	Delisted for nutrients in 2024
Downstream Waterbody	Landlocked

La Lake is located in the City of Woodbury and is used primarily for wildlife viewing. La Lake doesn't have any upstream waterbodies, has a small watershed area, and is landlocked with no surface outlet.

La Lake has a water surface area of approximately 50 acres, a maximum depth of 10 feet, and a mean depth of approximately 5 feet. La Lake is shallow enough for aquatic plants to grow over the entire waterbody and for the lake to mix many times per year (polymictic lake).

The State of Minnesota uses three water quality parameters to assess eutrophication standards within lakes—total phosphorus, chlorophyll-*a*, and Secchi disk transparency—to assess waterbody health and track water quality changes. These three parameters were measured in La Lake by the SWWD between 1994-2023 (Figure 13-1). Monitoring data collected in 2001 and between 2003 – 2004 was collected at a reduced frequency from other monitored years. Given the low number of samples, computed summer averages have a high level of uncertainty for those years. During the monitored years with adequate data, summer average total phosphorus concentrations were better than the state standard between 2015 – 2023. Prior to 2015, all other monitored years exceeded the state standard. Summer average chlorophyll-*a* concentrations were better than the state standard between 1994 – 1996, 1998, 2002, 2005, 2008, 2011, and 2015 - 2023. Summer average Secchi depths met or were better than the state standard for all monitored years between 1994 – 2023. La Lake was removed from the Minnesota impaired water lists as impaired for nutrients in 2024 due to improved water quality conditions between 2015 – 2023.

Chloride concentrations have not been monitored in La Lake.

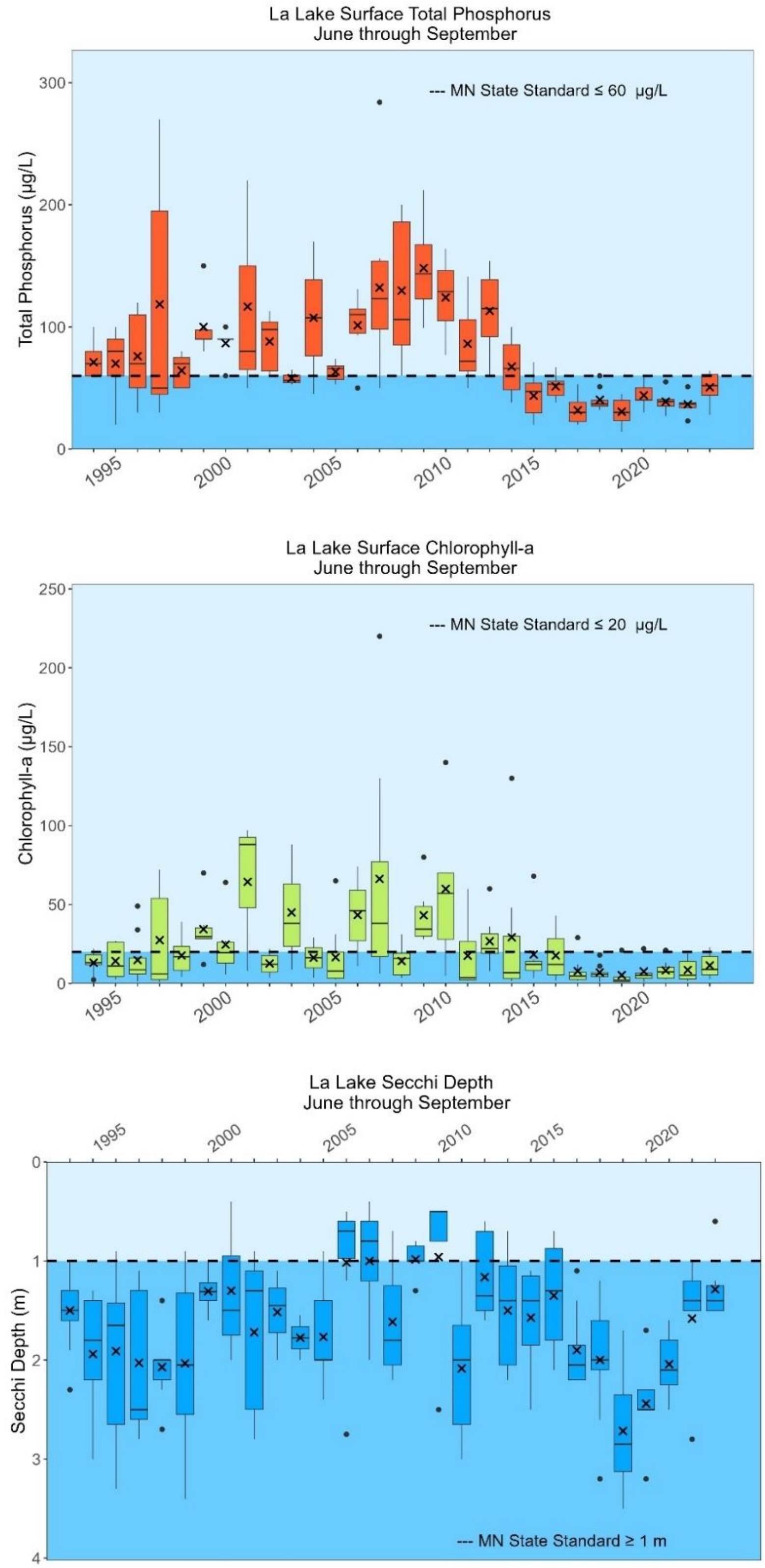


Figure 13-1 La Lake eutrophication monitoring data (June – September)
Summer averages are shown by x's in the box plots

13.2 Ecological Health

13.2.1 Aquatic Plants

Table 13-1 summarizes the calculated Lake Plant Eutrophication IBI values for La Lake based on point-intercept plant surveys completed in 2021 and 2024. La Lake scored above both plant IBI metrics in June 2021, but below both metrics in August 2021 and June and August 2024. The most recent metrics suggest a degraded plant community that is likely stressed from cultural eutrophication. Table 13-1 also summarizes the percentage of the littoral area where aquatic plants were found and lists observed aquatic invasive species, their frequency of occurrence (FOO), and current management practices. Other plant health metrics can be referenced in the 2021 and 2024 Aquatic Vegetation Survey Results reports (Stantec, 2021, 2025).

Table 13-1 La Lake aquatic plants overview

Lake	Parameter	June 2021[1]	August 2021[1]	June 2024[2]	August 2024[2]	MnDNR Threshold	Invasive Species Management
La	Species Richness	12	10	9	8	>11	
	Floristic Quality Index (FQI)	18.7	16	13.4	11.5	>17.8	
	% Littoral with Vegetation	100%	100%	76%	59%		
	Curly-leaf Pondweed FOO	29%	-	-	-		Herbicide applications 2022 - 2025
	Purple Loosestrife FOO	-	P	-	-		Managed naturally by leaf eating beetles

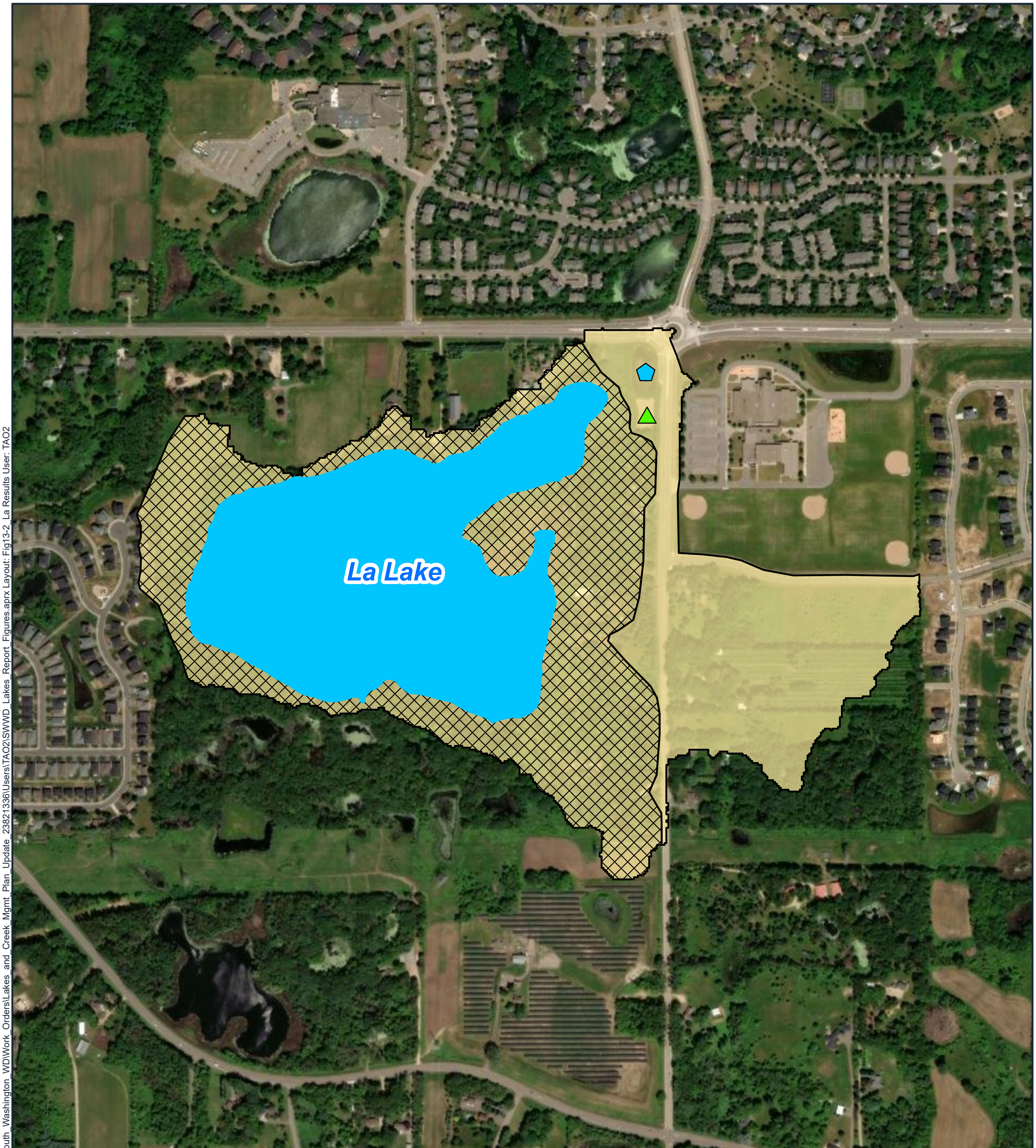
[1] (Stantec, 2021 Aquatic Vegetation Survey Results, 2021)
[2] (Stantec, 2025)

13.2.2 Fisheries

Although La Lake has historically been used as a walleye rearing waterbody, the absence of recent fish survey data prevents an assessment of current walleye abundance. There are no MNDNR fish survey or stocking data available for La Lake.

13.3 Watershed Total Phosphorus Loads

P8 modeling shows low watershed phosphorus loads to La Lake. There are only two watersheds that drain to La Lake under current conditions, totaling approximately 130 acres. The direct drainage area to La Lake is primarily wooded and undeveloped, and runoff does not receive any treatment before entering the lake. The upstream watershed to the lake drains some agricultural land as well as a portion of Woodlane Drive before discharging to a stormwater pond and filtration basin. It should be noted that these BMPs were constructed after 2019, so the modeling results for 2019 only include the direct drainage area to La Lake. The map in Figure 13-2 shows the effective areal phosphorus load by subwatershed, modeled BMPs, and untreated subwatersheds in the area. The effective phosphorus load represents the loading rate after pollutant removal by stormwater BMPs within the watershed and is reflective of the loading that actually makes it into the receiving waterbody from a given location. The effective phosphorus loads for the La Lake subwatersheds range from 0.08 pounds per acre per year to 0.2 pounds per acre per year, based on a 10-year modeling period (2012-2022).



Lake

P8 Device Type

- Infiltration Basin
- Pipe
- Wet Pond

Effective TP Load (lbs/ac/yr)*

- 0.0 - 0.3
- 0.3 - 0.6
- 0.6 - 1.0
- 1.0 - 1.5
- 1.5 - 2.0
- > 2.0
- Area Not Treated by BMP

*Watersheds shown represent post-BMP condition

N

0 300 600
Feet

La Lake
Effective TP Loading
Lake Management Plan
South Washington
Watershed District

FIGURE 13-2

BARR

13.4 In-lake Total Phosphorus Loads

Results of the in-lake modeling for La Lake showed that given the small size of the tributary watershed, phosphorus loading was dominated by internal loading in both the wet summer of 2019 and the dry summer of 2022, representing 77% (34 lbs) and 90% (57 lbs) of the phosphorus loading to the lake, respectively (Figure 13-3). In 2019, smaller total phosphorus loads were estimated to be present from groundwater inflow (16%, 7 lbs) and watershed runoff (7%, 3 lbs). Similarly in 2022, smaller total phosphorus loads were present from groundwater inflow (9%, 6 lbs) and watershed runoff (1%, 0.4 lbs).

There is uncertainty associated with the internal loading estimates for La Lake due to: (1) uncertainty in estimating groundwater interactions and that groundwater inflows were estimated as a major component of the La Lake water balance; and (2) no lake profile monitoring data is available for La Lake (e.g., temperature, dissolved oxygen). Limited groundwater data is available within the vicinity of La Lake, including data on groundwater quality. For the purposes of this study, a groundwater total phosphorus concentration of 88 µg/L was assumed based on the median of the most recent 10 years of annual monitoring data available from well 798060 (National Water Quality Monitoring Council, 2025). The median was used to represent the central tendency of monitoring results for this well, instead of the 10-year average, given the large variability of results within the available monitoring data. If the assumed groundwater total phosphorus concentration was too low during the modeled periods, then the internal phosphorus load is overestimated in the lake calibrations. Conversely, if the assumed groundwater concentration was too high, then the internal phosphorus load is underestimated in the lake calibrations. Since groundwater inflow is a major component of the La Lake water balance, more detailed groundwater monitoring is recommended to confirm the extent of groundwater inflow impacts on in-lake phosphorus concentrations.

Additionally, profile monitoring data, including dissolved oxygen and temperature, has not historically been collected on La Lake. Therefore, no definitive conclusions can be drawn about the effects of lake stratification and mixing on internal loading. Understanding the role of lake stratification on internal loading would be important should the District decide to pursue developing sediment inactivation strategies. The collection of profile water quality data is especially critical since sediment core release rate experiments conducted in 2017 indicated a relatively low potential for internal phosphorus loading from lake bottom sediment (Section 4.8). However, even a low internal phosphorus loading rate can notably increase phosphorus concentrations in the lake water column, especially in shallow lakes with small water volumes per surface area of lake bottom.

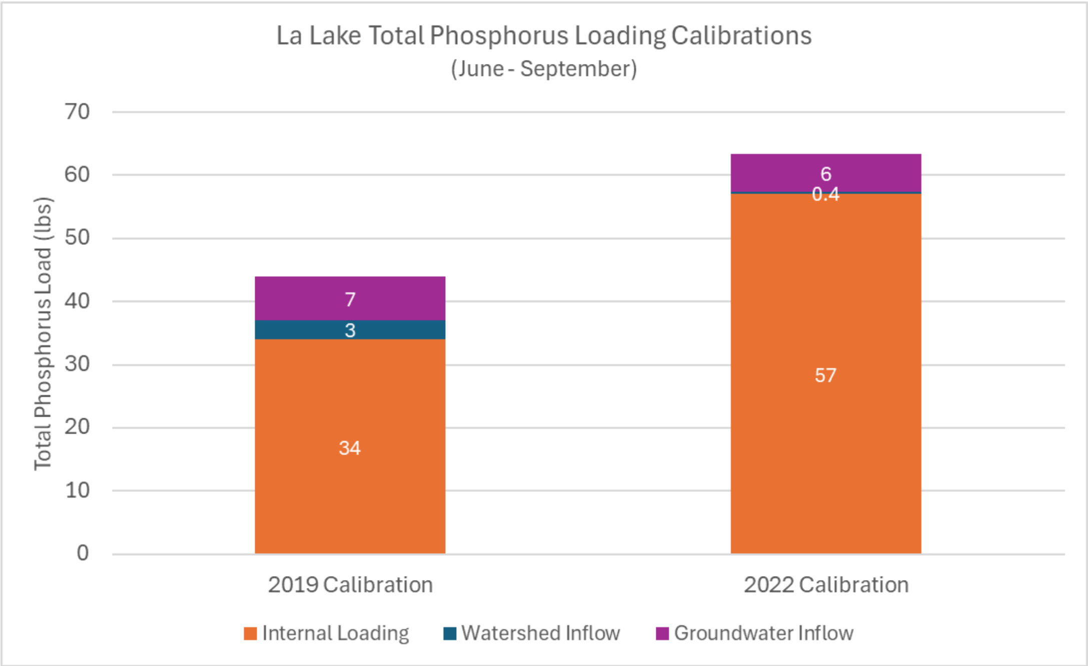


Figure 13-3 La Lake total phosphorus load estimates – 2019 & 2022

13.5 Predicted Benefits from Implementing In-lake Management Practices

One of the study goals was to estimate the potential impact of lake sediment inactivation projects and how they could be used to improve in-lake water quality. The calibrated in-lake models were used to estimate the impact of implementing sediment treatment projects in La Lake.

Table 13-2 summarizes the estimated phosphorus load reduction from two scenarios: sediment inactivation (low efficacy) and sediment inactivation (high efficacy). Results from the in-lake modeling predict a reduction of 21 – 46 pounds of total phosphorus loading into La Lake through implementation of sediment inactivation practices for a wet (2019) and dry (2022) year. This reduction in phosphorus loading translates to a reduction in the in-lake total phosphorus concentrations. Model results estimate that the 2019 summer average total phosphorus concentration of 32 µg/L would reduce to 21 - 24 µg/L with the implementation of sediment treatments; and the 2022 summer average total phosphorus concentration of 39 µg/L would reduce to 23 - 27 µg/L (Table 13-2). The existing summer average total phosphorus concentrations in La Lake already meet state standards, but implementing sediment inactivation projects would offer opportunities to further reduce phosphorus concentrations. Figure 13-4 and Figure 13-5 summarize the predicted reductions in in-lake total phosphorus concentrations for the 2019 and 2022 model years, respectively, for each of the proposed scenarios.

Table 13-2 Summer growing period estimated reductions in phosphorus loads and in-lake concentrations in La Lake with the proposed management practices

Scenario	Modeled Summer Total Phosphorus Loading (lbs)					Summer Average Total Phosphorus (µg/L)[1]
	Watershed Inflow	Groundwater Inflow	Internal Loading	Total	Load Reduction	
2019 Existing	3	7	34	44	-	32
2019 Sediment Inactivation Low	3	7	13	23	-21	24
2019 Sediment Inactivation High	3	7	7	17	-27	21
2022 Existing	0.4	6	57	63	-	39
2022 Sediment Inactivation Low	0.4	6	23	29	-34	27
2022 Sediment Inactivation High	0.4	6	11	17	-46	23

[1] Existing conditions summer average concentrations are calculated from calibrated in-lake models

Model results estimating the in-lake response to reduced internal phosphorus loading reflect the estimated impacts that would have occurred during the 2019 and 2022 growing seasons if a sediment inactivation project had been implemented that year. The modeling incorporates a range of assumed sediment-treatment efficacies to account for application variability and the fact that the specific treatment approach (alum, alum/iron, or aeration) has not yet been selected. Assumed efficacies were informed by monitoring data from comparable Barr sediment-inactivation projects one year post-implementation. Monitoring of past projects shows that sediment inactivation efficacy can vary and is expected to decline over time due to product aging, new watershed inflows, and/or burial from rough fish activity. As such, results presented in this report provide an estimate of the expected scale of treatment impacts within a given growing season, but do not account for longer-term changes within the system and how that may impact benefits over time. For more information on the sediment inactivation modeling assumptions see Section 6.4.

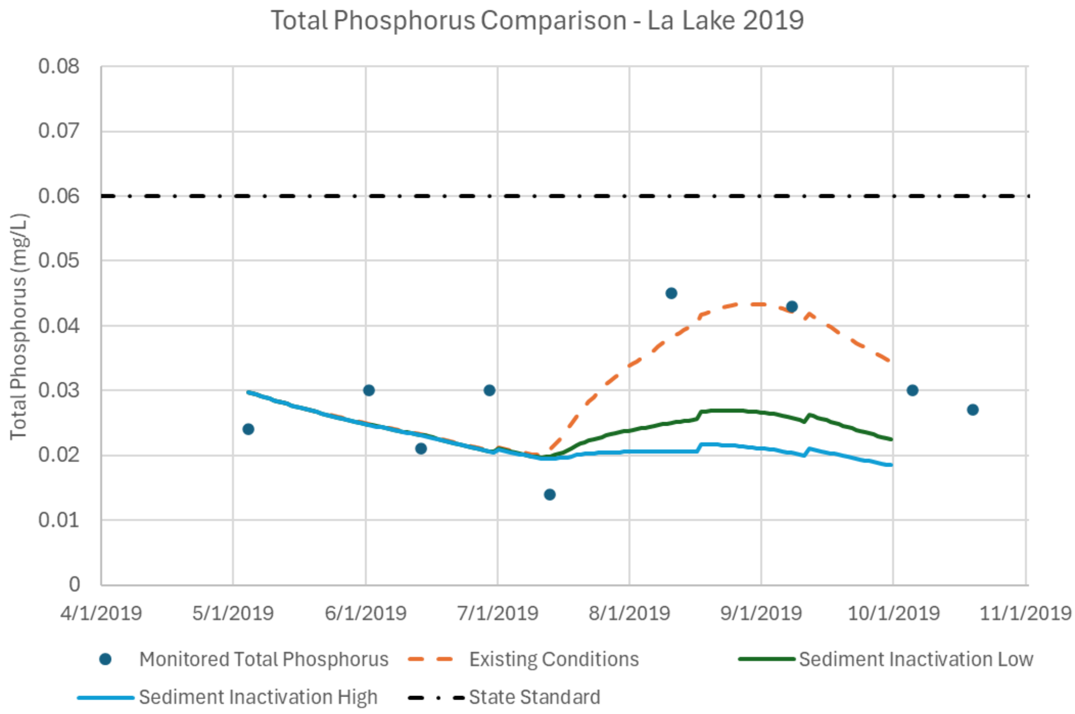


Figure 13-4 Model-predicted reductions in 2019 total phosphorus concentrations in La Lake with the proposed management practices

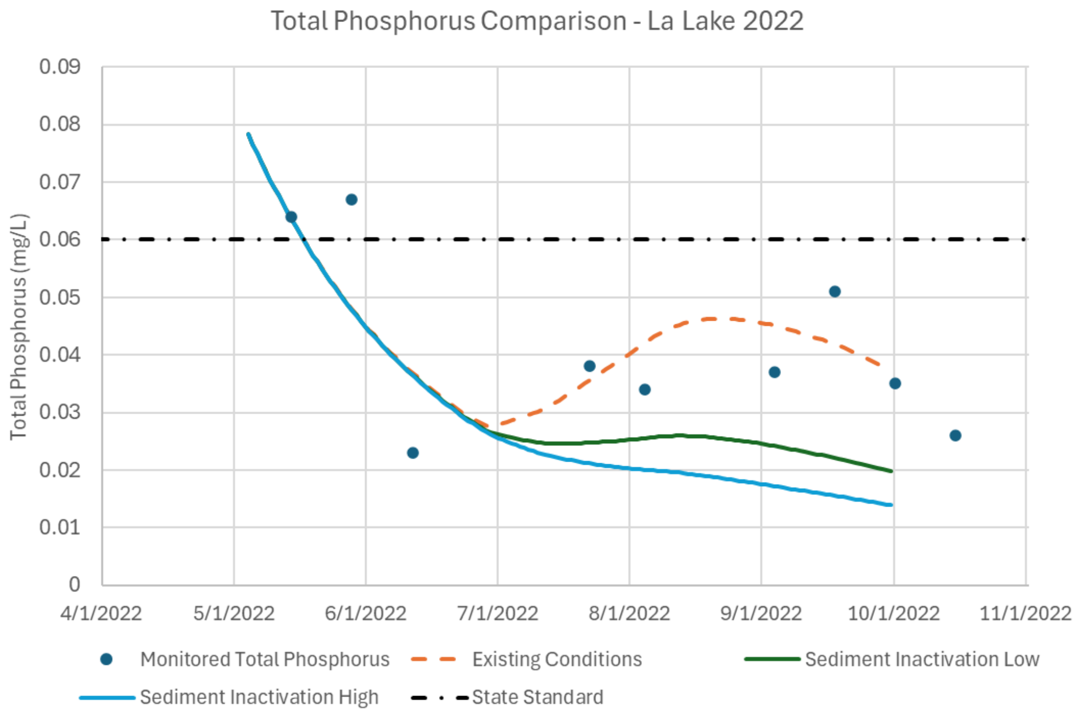


Figure 13-5 Model-predicted reductions in 2022 total phosphorus concentrations in La Lake with the proposed management practices

13.6 Management Recommendations

Monitoring of La Lake indicates improved water quality in recent years; however, the aquatic plant community is threatened by the growth of the aquatic invasive species, curly-leaf pondweed. Given this, future management efforts should focus on protecting lake water quality and improving ecosystem health, monitoring for changes, and continuing water quality and ecosystem health protection measures as improvements are achieved. Table 13-3 summarizes the recommended management strategies that could be considered for La Lake to help protect water quality conditions and improve ecological health.

Table 13-3 La Lake management recommendations

Management/Protection Action		Basis
Address external watershed loads	Enhanced street sweeping program	Continue to work with the City of Woodbury to refine an enhanced street sweeping program to reduce pollutant loading to stormwater runoff
	Stormwater BMPs	If areas of the small watershed develop, continue to enforce SWWD rules and consider implementing site scale BMPs as opportunities arise
Address internal loads	Sediment inactivation treatment	Review and consider implementing a sediment inactivation treatment to reduce lake bottom sediment phosphorus loads should in-lake conditions start to worsen
Aquatic Plants	Invasive species management	Continue to monitor invasive species growth (e.g., curly-leaf pondweed) and continue management as needed
	Promote native aquatic plant growth	Encourage native plant reestablishment during and following aquatic invasive species management
Fisheries	Fisheries Survey	Consider collecting fish community data
Phytoplankton and Zooplankton	Phytoplankton/Zooplankton monitoring	Consider monitoring phytoplankton and zooplankton as part of routine monitoring
Water Quality	Water quality monitoring	Continue monitoring in-lake water quality and assessing trends for eutrophication within La Lake Consider collecting chloride data to monitor in-lake chloride concentrations from winter salt use Consider collecting profile monitoring data (e.g., dissolved oxygen, temperature) Collect additional information on groundwater contributions into the lake (water quality, groundwater-surface water interactions) to confirm impacts

14 Ravine Lake

14.1 Water Quality

Shallow/Deep	Shallow
Location	Cottage Grove
Surface Area	25 acres
Average/Maximum Depth	5 feet/16 feet
Watershed Area	1,698 acres
Watershed:Lake Surface Area	68:1
Impairment Status	Impaired for nutrients since 2006
Downstream Waterbody	Unnamed Stream/Mississippi River

Ravine Lake is located in the City of Cottage Grove and is used primarily for active and passive recreation and wildlife viewing. When water levels are high enough, water discharges via gravity flow from Ravine Lake to a downstream creek which ultimately discharges to the Mississippi River. The total watershed area to Ravine Lake is approximately 4,341 acres; however, a large portion of the watershed only contributes flow to the lake under extreme rainfall events, causing the more typical tributary area to be approximately 1,698 acres.

Ravine Lake has a water surface area of approximately 25 acres, a maximum depth of 16 feet, and a mean depth of approximately 5 feet. Ravine Lake falls under the MNDNR classification of a "shallow lake" given that the lake's littoral area is greater than 80% the total lake area. Ravine Lake is deep enough that prolonged periods of lake stratification occur during the growing season. Profile monitoring data from within the lake supports this finding.

The State of Minnesota uses three water quality parameters to assess eutrophication standards within lakes—total phosphorus, chlorophyll-*a*, and Secchi disk transparency—to assess waterbody health and track water quality changes. These three parameters were measured in Ravine Lake by the SWWD between 1998 – 2023 (Figure 14-1). Monitoring data collected in 2017 was collected at a reduced frequency from other monitored years. Not enough measurements were collected to calculate summer averages for that year. During the monitored years with adequate data, summer average total phosphorus concentrations were better than the state standard in 2005, 2007, 2009 – 2010, 2014 – 2015, and 2019. Summer average chlorophyll-*a* concentrations were better than the state standard in 1998, 2007, and 2009. All other monitored years exceeded the state standard. Summer average Secchi depths met or were better than the state standard in 1999, 2004 – 2020, and 2022 – 2023. Ravine Lake was added to the Minnesota impaired water lists as impaired for nutrients in 2006.

Chloride concentrations were measured by the SWWD between 2005 – 2007 and 2009 – 2023 (generally between April and September). In the historical record, all observed chloride concentrations were below the MPCA chronic standard of 230 mg/L. The highest observed chloride concentration was 84 mg/L in July 2014. Most observed chloride concentrations are below 50 mg/L. While chloride concentrations within Ravine Lake remain well below the chronic water quality standard, concentrations have been on a steady rise within the lake over the past 18 years; with average annual observed concentrations increasing from 10 mg/L to 45 mg/L between 2005 and 2023.

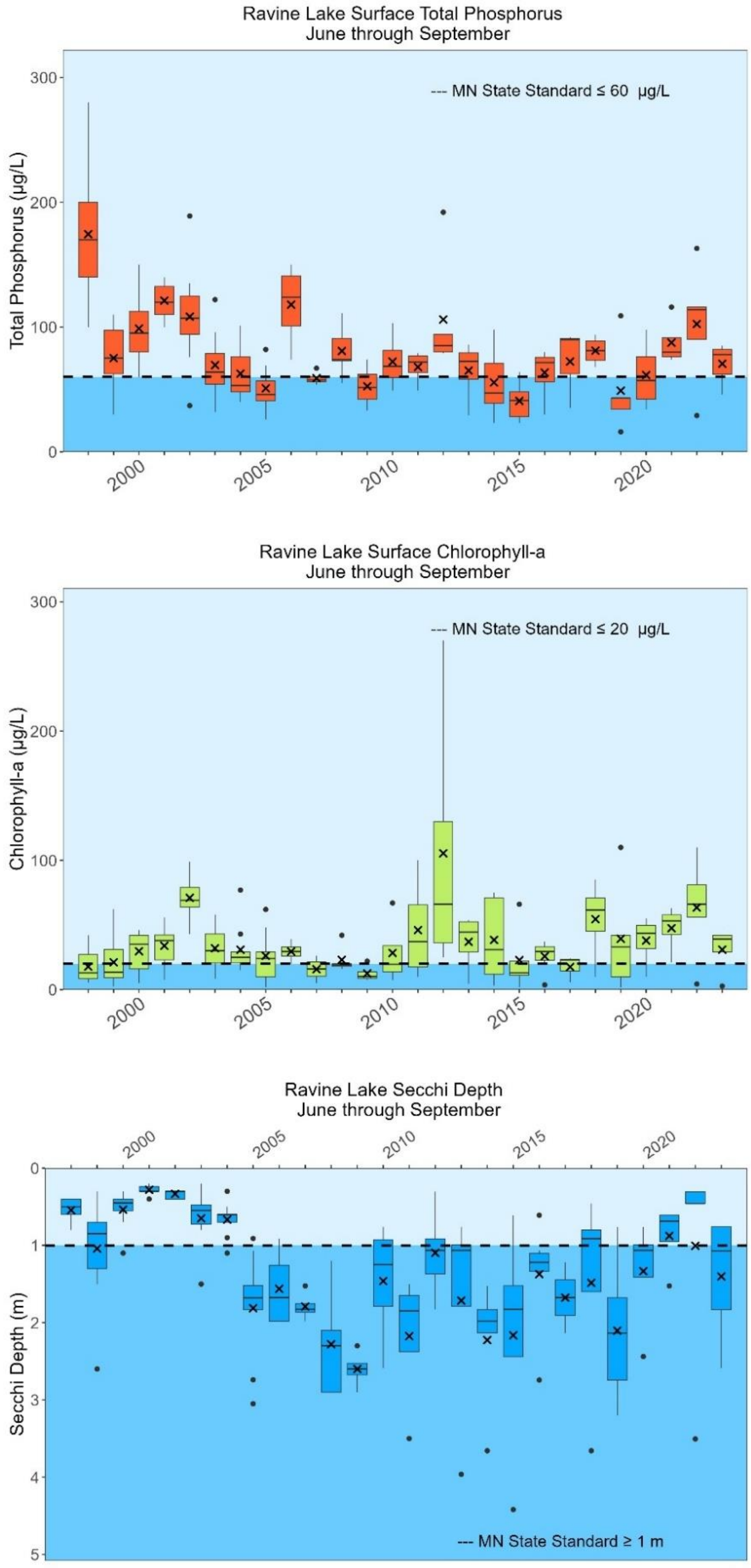


Figure 14-1 Ravine Lake eutrophication monitoring data (June – September)
Summer averages are shown by x's in the box plots

14.2 Ecological Health

14.2.1 Aquatic Plants

Table 14-1 summarizes the calculated Lake Plant Eutrophication IBI values for Ravine Lake based on point-intercept plant surveys completed in 2021 and 2024. Ravine Lake is the only lake in the District that falls in the Western Corn Belt Plains ecoregion. This ecoregion has less stringent MNDNR plant IBI metrics than the other District Lakes, which are located in the North Central Hardwood Forest ecoregion. Comparing Ravine Lake to the Western Corn Belt Plains ecoregion metrics, the lake scored above both plant IBI metrics in June and August in 2021 and 2024. Table 14-1 also summarizes the percentage of the littoral area where aquatic plants were found and lists observed aquatic invasive species, their frequency of occurrence (FOO), and current management practices. Other plant health metrics can be referenced in the 2021 and 2024 Aquatic Vegetation Survey Results reports (Stantec, 2021, 2025).

Table 14-1 Ravine Lake aquatic plants overview

Lake	Parameter	June 2021[1]	August 2021[1]	June 2024[2]	August 2024[2]	MnDNR Threshold	Invasive Species Management
Ravine[3]	Species Richness	11	9	8	7	>4	
	Floristic Quality Index (FQI)	15.5	11.7	11.8	11.7	>7.7	
	% Littoral with Vegetation	90%	88%	78%	75%		
	Curly-leaf Pondweed FOO	86%	27%	39%	-		Herbicide applications 2022 - 2025

[1] (Stantec, 2021 Aquatic Vegetation Survey Results, 2021)
[2] (Stantec, 2025)
[3] Ravine Lake is the only lake in the District that falls in the Western Corn Belt Plains ecoregion, which has different species richness and FQI thresholds than the North Central Hardwood Forest ecoregion.

14.2.2 Fisheries

Ravine Lake is included in the Fishing in the Neighborhood (FiN) program run by the MNDNR, which is aimed at increasing angling opportunities, public awareness, and environmental stewardship within the seven-county Twin Cities metro area. The MNDNR FiN program has been actively involved in managing the sport fish populations of Ravine Lake since 2001. As such, the fish community is surveyed at a regular frequency. Fish stocking data from the most recent decade is also available. Table 14-2 summarizes the fish community data from the most recent survey conducted in August 2021 (MNDNR, 2021). Table 14-3 summarizes the fish stocking report from the last decade (MNDNR, 2024). Walleye have been stocked annually in Ravine Lake since 2002.

Based on the review of aerial imagery and field observations, Ravine Lake does not always completely ice over during the course of winter due to impacts from groundwater inflows. Open water areas in winter can provide better winter oxygen conditions within the lake and limit the risk of winter fish kills.

Table 14-2 Ravine Lake fish survey report – 2021 (MNDNR, LakeFinder, 2021)

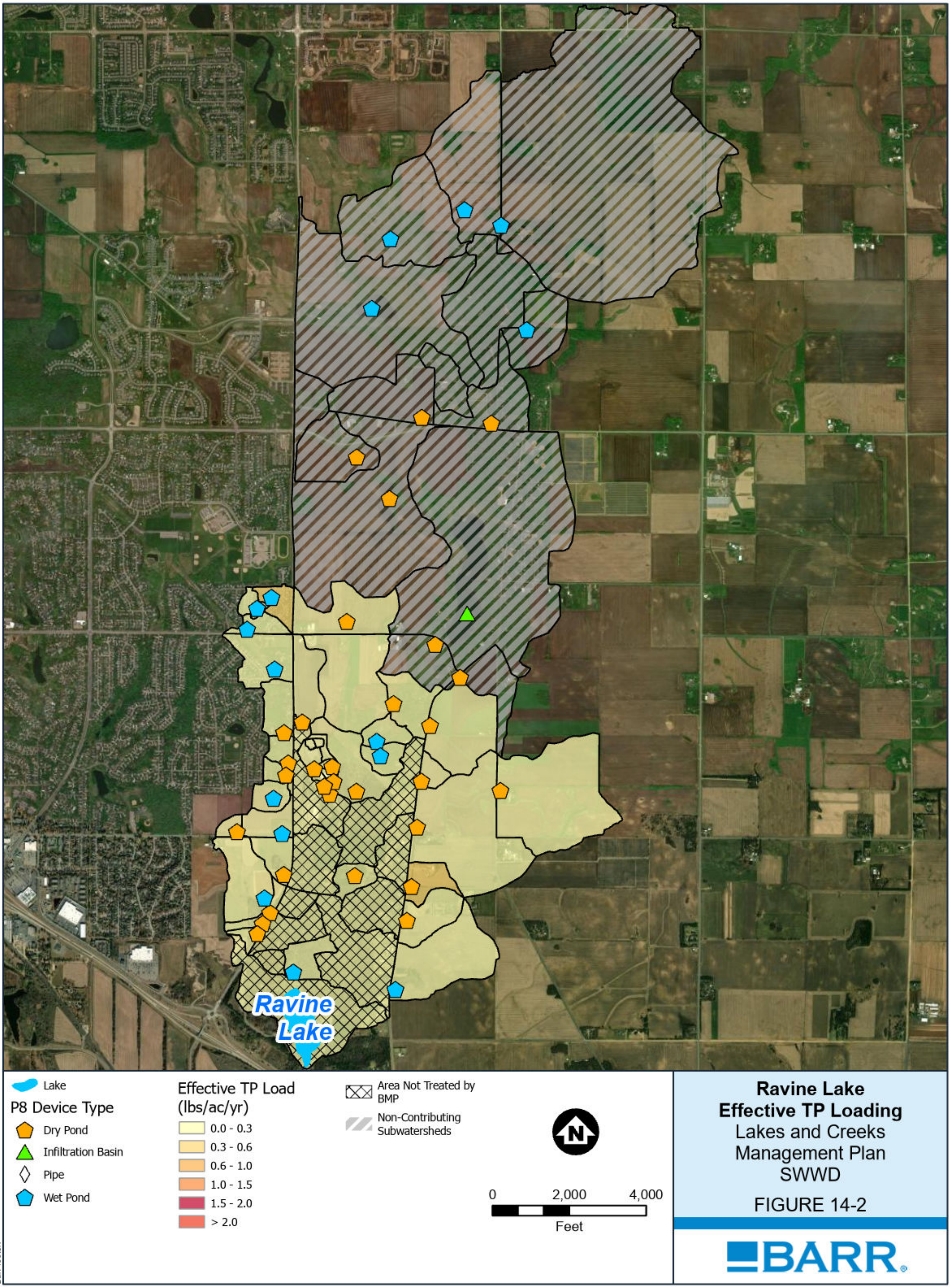
Catch Method	Fish Species	Catch per Unit Effort (CPUE)	Normal CPUE Range	Count
Standard trap nets	Black bullhead	0.2	2.5 – 70.2	1
	Black crappie	0.6	1.3 – 27.7	3
	Bluegill	50.2	2.8 – 43.3	251
	Brown bullhead	0.4	0.2 – 6.2	2
	Hybrid sunfish	1.2	N/A	6
	Pumpkinseed	0.4	0.8 – 9.3	2
	Walleye	0.6	0.3 – 1.3	3
	White sucker	0.4	0.2 – 2.2	2
Standard gill nets	Black bullhead	1.0	8.0 – 90.0	1
	Black crappie	2.0	2.0 – 19.0	2
	Bluegill	7.0	N/A	7
	Northern Pike	1.0	1.5 – 9.0	1
	Yellow Perch	5.0	2.5 – 25.8	5

Table 14-3 Ravine Lake fish stocking report – 2015-2024 (MNDNR, LakeFinder, 2024)

Year	Fish Species	Size	Number	Pounds
2024	Walleye	yearlings	411	75.0
2023	Walleye	adults	43	43.0
2022	Walleye	adults	107	100.0
2021	Walleye	adults	167	233.8
		yearlings	126	18.0
2020	Walleye	adults	60	60.0
		yearlings	252	42.0
2019	Walleye	fingerlings	627	66.0
2018	Walleye	adults	120	120.0
2017	Walleye	adults	75	75.0
2016	Walleye	adults	16	16.0
		yearlings	269	79.0
2015	Walleye	yearlings	1,990	99.5

14.3 Watershed Total Phosphorus Loads

P8 modeling shows low watershed phosphorus loads to Ravine Lake. Approximately half of the area tributary to Ravine Lake does not contribute runoff for events smaller than the 100-year storm event. This area is noted as non-contributing in Figure 14-2. The remainder of the tributary area receives some treatment from stormwater ponds or dry detention areas, but much of the contributing area is regional conveyance through undeveloped land. The map in Figure 14-2 shows the effective areal phosphorus load by subwatershed, modeled BMPs, and untreated subwatersheds within the tributary area. The effective phosphorus load represents the loading rate after pollutant removal by BMPs within the watershed and is reflective of the loading that actually makes it into the receiving waterbody from a given location. The effective phosphorus loads for the Ravine Lake subwatersheds range from 0.0 pounds per acre per year to 0.3 pounds per acre per year, based on a 10-year modeling period (2012-2022).



Ravine Lake
Effective TP Loading
Lakes and Creeks
Management Plan
SWWD

FIGURE 14-2

BARR

Barr footer:

14.4 In-lake Total Phosphorus Loads

Results of the in-lake modeling for Ravine Lake showed that during the summer of 2019 (representing a wet year), phosphorus loading into Ravine Lake from the tributary watershed was greater than the contribution from internal loading or groundwater inflow. Phosphorus load from watershed runoff represented 51% (80 lbs) of the phosphorus loading during the summer of 2019 (Figure 14-3). Internal loading in summer 2019 was estimated to be 42% (67 lbs) and groundwater inflow was estimated to be 7% (12 lbs) of the phosphorus load to the lake. During the drier summer of 2022, the phosphorus loading to Ravine Lake from watershed runoff was notably lower, representing 21% (37 lbs) of the phosphorus loading to the lake. The 2022 total phosphorus load from groundwater inflow was similar to the load in 2019, representing 5% (9 lbs) of the total load. Internal phosphorus loading in 2022 was notably higher than 2019, representing 74% (127 lbs) of the phosphorus load.

As discussed in Section 6.2.1.1 Water Balance Assumptions, there is high level of uncertainty associated with Ravine Lake's water balance calibrations in 2019 and 2022. Beaver dams are frequently observed at the Ravine Lake outlet and, while field notes provided some information on beaver dam observations and occasional removal dates, there are no detailed records available for model years 2019 and 2022 on the exact durations of beaver dam impacts, approximate dimensions of the beaver dams, and/or measured impacts to outflow discharge rates. This led to a high level of uncertainty with simulating hourly outflows from the lake. Secondly, Ravine Lake is known to have sizable groundwater inflow; however, there are no detailed monitoring records available on groundwater inflow rates. Beaver dam impacts to the outlet, coupled with non-monitored groundwater inflow rates, result in two unknowns in the water balance. For the purposes of model development, Barr used the available data on water surface elevations and water quality monitoring to infer the most accurate inflow and outflow rates over the modeled time periods. For example, between mid-June through mid-September 2019 the monitored water surface elevations in Ravine Lake were approximately 0.3 feet above the outlet elevation. During this time period, the model includes an assumption that a beaver dam was installed in the outlet and changed the control elevation of the lake. Similar assumptions were needed in the 2022 model. Given the uncertainty associated with inflows and outflows in the Ravine Lake model, this inherently results in uncertainty in the total phosphorus loading estimates. Improved monitoring on Ravine Lake inflow and outflow rates could support better certainty in phosphorus loading estimates.

The higher estimated groundwater influence on Ravine Lake also contributes to uncertainty in the internal loading estimates. Limited groundwater water quality monitoring data is available from wells in the vicinity of Ravine Lake. For the purposes of this study, a groundwater total phosphorus concentration of 20 µg/L was assumed based on the 10-year average annual monitoring data available from well 778334 (National Water Quality Monitoring Council, 2025). If the assumed groundwater total phosphorus concentration was too low during the modeled periods, then the internal phosphorus load is overestimated in the lake calibrations. Conversely, if the assumed groundwater concentration was too high, then the internal phosphorus load is underestimated in the lake calibrations. Since groundwater inflow is a notable component of the Ravine Lake water balance (Section 6.2.1.1), more detailed groundwater monitoring is recommended to confirm the extent of groundwater inflow impacts on in-lake phosphorus concentrations.

Bi-weekly profile monitoring data is typically collected from Ravine Lake between April and October each year, including the collection of temperature and dissolved oxygen concentrations. The data indicates that Ravine Lake can have temperature stratification as early as late-April or early-May in most years and typically remains stratified until fall. To simulate lake stratification, model calibration parameters were adjusted to suppress the amount of internal phosphorus load reaching the surface waters during the growing season. Given that the monitoring data demonstrates strong lake stratification during the summer, the collection of water quality parameters at multiple depths is recommended to better estimate internal loading influence on surface water conditions (e.g., collecting phosphorus near the surface, at the thermocline, and near the bottom). Understanding how lake stratification influences internal loading will be essential for developing effective sediment inactivation strategies and demonstrating benefits, should the District decide to pursue this type of treatment. The collection of water quality parameters at multiple depths is especially important as sediment core phosphorus fractionation data indicate a high potential for phosphorus release from lake bottom sediment (Section 4.8).

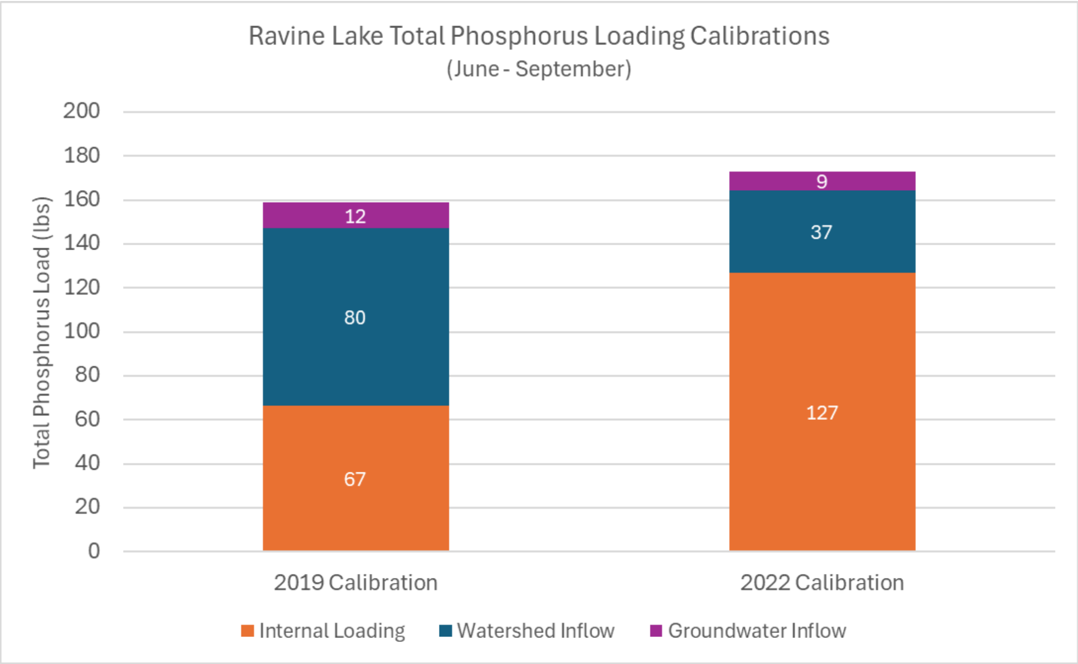


Figure 14-3 Ravine Lake total phosphorus load estimates – 2019 & 2022

14.5 Nitrogen and Algal Growth

14.5.1 Nitrogen Limitation

Throughout the growing season, various factors can influence the rate and volume of algae growth within a lake, such as phosphorus, nitrogen, light, and temperature. The Barr Shallow Lake Model was used for simulating water quality within Ravine Lake; this model utilizes Michaelis Menten kinetics to determine which factor or combination of factors limit algae growth throughout the modeled time period. Although nitrate, nitrite, ammonia, and ammonium water quality data were not available for model calibration, the effect of nitrogen on chlorophyll-*a* growth was deduced for Ravine Lake using the lake model. In both the 2019 and 2022 models, accounting for nitrogen limitation as part of the model calibration improved model results with respect to simulating chlorophyll-*a*. In the 2019 model, nitrogen limitation was initiated in early August to match the drop in chlorophyll-*a* concentrations observed in the monitoring data. In the 2022 model, nitrogen limitation was initiated in early July and sustained throughout portions of August and September to match the drop in chlorophyll-*a* concentrations observed.

Nitrogen limitation during a portion of the growing season has been seen in numerous other shallow lake modeling efforts performed by Barr. Nitrogen limitation tends to occur because the growth rate of algae can be greater than the degradation rate of nitrogen (i.e., slow decomposition from organic nitrogen forms to ammonia and/or slow nitrification rate changing ammonia to nitrite and nitrate). In other words, algae will continue to grow until they have used up bioavailable forms of nitrogen in the water column, and once depleted, algae growth will be limited. However, nitrogen limitation does not affect all algal species equally. Certain species of cyanobacteria can persist under nitrogen-limited conditions, even as other algal species decline or disappear entirely. This resilience is due to the ability of some cyanobacteria species to fix atmospheric nitrogen by converting N_2 gas into ammonia. The process is energy-intensive though and will be used only when necessary. This competitive advantage complicates efforts to predict how nitrogen limitation will influence overall algal growth dynamics in a waterbody. The presence of cyanobacteria with nitrogen fixing abilities could not be confirmed as part of this study, as no phytoplankton monitoring data is currently available for this lake.

The algal growth limitations identified during model calibration reflect which parameters are currently limiting growth. As management is implemented, algal growth limitations will likely change. For example, if phosphorus loading from lake bottom sediment is reduced, phosphorus limitation may become more dominant throughout the growing season. Similarly, a reduced phytoplankton population with phosphorus control could also increase nitrogen concentrations (e.g., reduced nitrogen demand) and alleviate the nitrogen limitation and also eliminate a competitive advantage of cyanobacteria. Outcomes of changes in nitrogen dynamics are more challenging to predict than phosphorus management with the information currently available.

14.5.2 Ammonia Impairment

The unnamed stream downstream of Ravine Lake has been identified as impaired for ammonia (NH_3). When ammonia-based fertilizers are applied to agricultural fields, excess can be washed downstream during storm events. Animal waste from livestock also contains ammonia and can enter downstream waterbodies from runoff. Urban areas can also contribute ammonia to downstream waterbodies, especially from areas where fertilizers or animal waste are present.

Under slightly acidic or neutral pH levels (pH < 7), a large percentage of ammonia that enters a waterbody will be transformed into ammonium (NH_4^+) due to the availability of H^+ ions:

$$NH_3 + H^+ \leftrightarrow NH4^+$$

However, at higher pH levels (pH > 9) when there are less H^+ ions available, a higher percentage will be present as ammonia. This is important because high concentrations of ammonia can be toxic to aquatic organisms. Elevated pH levels in freshwater ecosystems often result from intense photosynthetic activity during algal blooms. During high algal growth conditions, carbon dioxide uptake can be rapid. When carbon dioxide is removed from the water, an equilibrium shift occurs between carbonic acid, bicarbonate, carbonate, and carbon dioxide, which removes H^+ ions, resulting in high pH conditions. pH > 9 is often observed in Ravine Lake during the late summer months (July – September). In September 2017, a pH of 11.2 was recorded. Field notes indicate that there were algal scums along the shoreline and definite algal presence at the monitoring location during this sampling event. It’s possible that intense photosynthetic activity from algal growth may be sustaining elevated pH levels, which in turn increases the proportion of ammonia in the water.

14.6 Predicted Benefits from Implementing In-lake Management Practices

One of the study goals was to estimate the potential impact of lake sediment inactivation projects and how they could be used to improve in-lake water quality. The calibrated in-lake models were used to estimate the impact of implementing sediment treatment projects in Ravine Lake.

Table 14-4 summarizes the estimated phosphorus load reduction from two scenarios: sediment inactivation (low efficacy) and sediment inactivation (high efficacy). Results from the in-lake modeling predict a reduction of 53 – 104 pounds of total phosphorus loading into Ravine Lake through implementation of sediment inactivation practices for a wet (2019) and dry (2022) year. This reduction in phosphorus loading translates to a reduction in the in-lake total phosphorus concentrations. Model results estimate that the 2019 summer average total phosphorus concentration of 71 µg/L would reduce to 35 - 36 µg/L with the implementation of sediment treatments; and the 2022 summer average total phosphorus concentration of 105 µg/L would reduce to 39 - 44 µg/L (Table 14-4). Figure 14-4 and Figure 14-5 summarize the predicted reductions in in-lake total phosphorus concentrations for the 2019 and 2022 model years, respectively, for each of the proposed scenarios.

Based on the model results, it's predicted that implementing sediment inactivation practices alone in Ravine Lake would not be sufficient to reduce summer average concentrations for chlorophyll-*a* below the state standards. However, given the high level of uncertainty in the calibrated models for this lake, the addition of watershed loading reduction scenarios were not summarized for this report. For more background on the model uncertainty associated with the calibration of the Ravine Lake models and recommendations for addressing that as part of next steps, please refer back to Sections 14.4 and 14.5.

Model results estimating the in-lake response to reduced phosphorus internal loading reflect the estimated impacts that would have occurred during the 2019 and 2022 growing seasons if a sediment inactivation project was implemented in those years. The modeling incorporates a range of assumed sediment-treatment efficacies to account for application variability and the fact that the specific treatment approach (alum, alum/iron, or aeration) has not yet been selected. Assumed efficacies were informed by monitoring data from comparable Barr sediment-inactivation projects one year post-implementation. Monitoring of past projects shows that sediment inactivation efficacy can vary and is expected to decline over time due to product aging, new watershed inflows, and/or burial from rough fish activity. As such, results presented in this report provide an estimate of the expected scale of treatment impacts within a given growing season, but do not account for longer-term changes within the system and how that may impact benefits over time. For more information on the sediment inactivation modeling assumptions see Section 6.4.

Table 14-4 Summer growing period estimated reductions in phosphorus loads and in-lake concentrations in Ravine Lake with the proposed management practices

Scenario	Modeled Summer Total Phosphorus Loading (lbs)					Summer Average Total Phosphorus (µg/L)[1]	Summer Average Chlorophyll-a (µg/L)[1]
	Watershed Inflow	Groundwater Inflow	Internal Loading	Total	Load Reduction		
2019 Existing	80	12	67	159	--	71	56
2019 Sediment Inactivation Low	80	12	13	105	-53	36	30
2019 Sediment Inactivation High	80	12	11	103	-55	35	29
2022 Existing	37	9	127	173	--	105	73
2022 Sediment Inactivation Low	37	9	29	74	-98	44	25
2022 Sediment Inactivation High	37	9	23	69	-104	39	24

[1] Existing conditions summer average concentrations are calculated from calibrated in-lake models

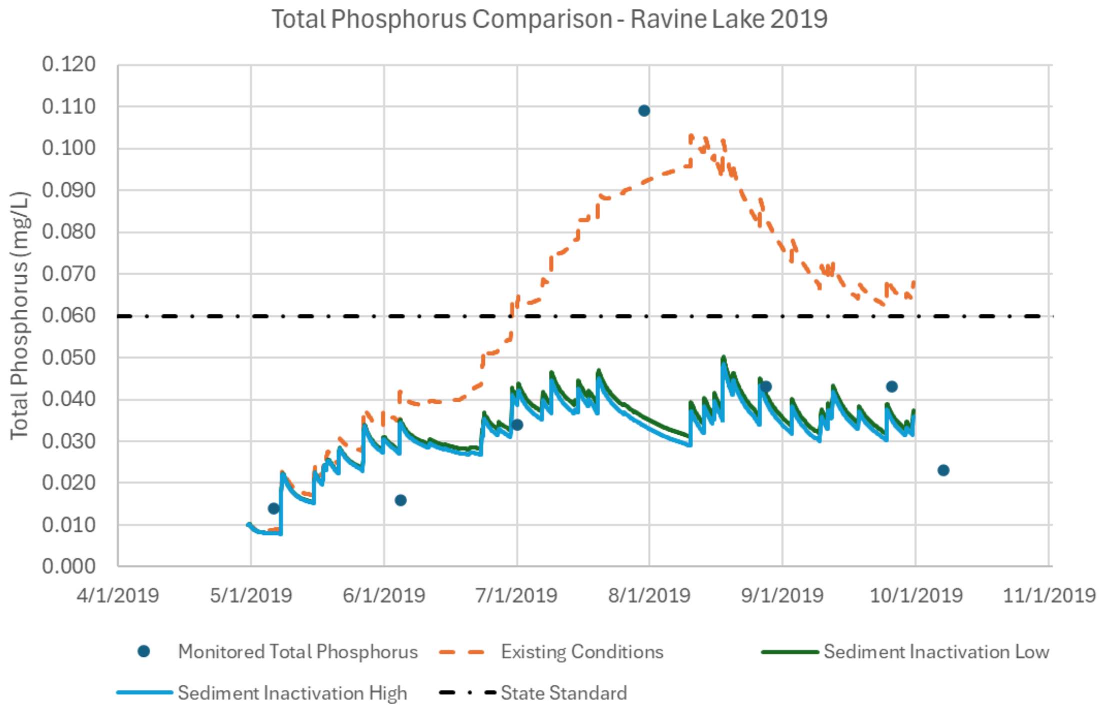


Figure 14-4 **Model-predicted reductions in 2019 total phosphorus concentrations in Ravine Lake with the proposed management practices**

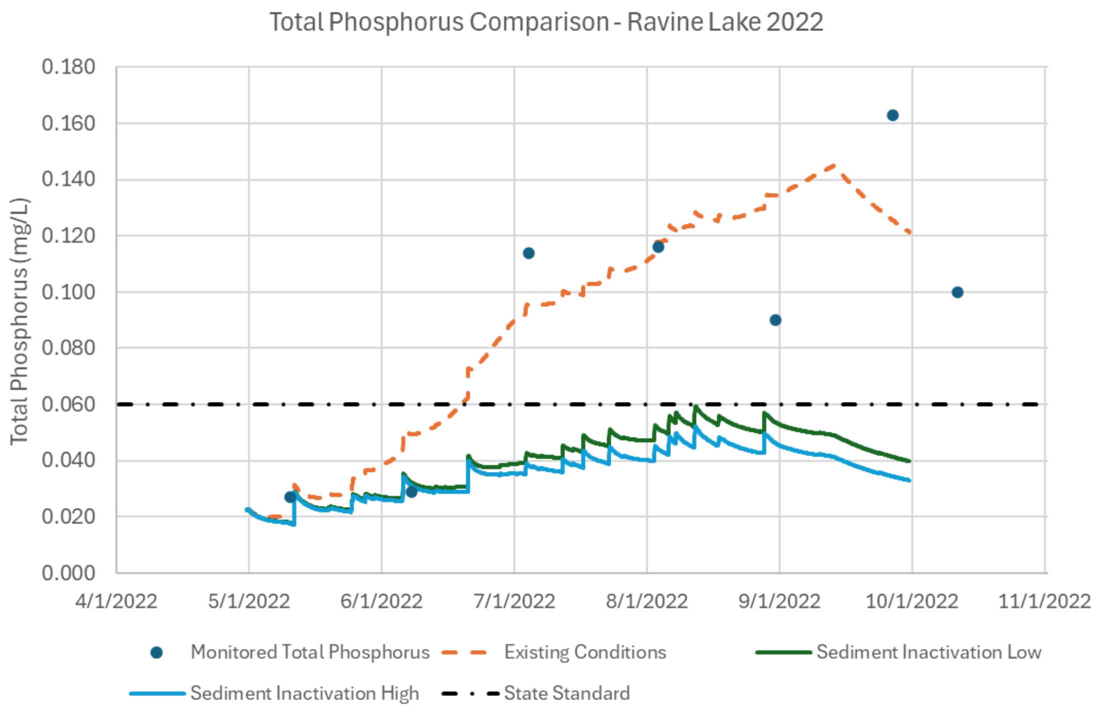


Figure 14-5 **Model-predicted reductions in 2022 total phosphorus concentrations in Ravine Lake with the proposed management practices**

14.7 Management Recommendations

Monitoring of Ravine Lake indicates degraded water quality in recent years and while aquatic plant monitoring indicates the lake is meeting the MNDNR Western Corn Belt Plains ecoregion plant IBI thresholds, the native aquatic plant community is threatened by the growth of the aquatic invasive species, curly-leaf pondweed. Given this, future management efforts should focus on improving lake water quality and ecosystem health, monitoring for changes, and continuing water quality and ecosystem health protection measures as improvements are achieved. Table 14-5 summarizes the recommended management strategies that could be considered for Ravine Lake to help improve water quality conditions and ecological health.

Table 14-5 Ravine Lake management recommendations

Management/Protection Action		Basis
Address external watershed loads	Enhanced street sweeping program	Consider partnering with City of Cottage Grove to implement enhanced street sweeping programs to reduce pollutant loading to stormwater runoff
	Stormwater BMPs	If areas of the watershed develop (e.g., agricultural land to impervious land use types), continue to enforce SWWD rules and consider implementing site scale BMPs as opportunities arise. Protect un-developed natural and park areas tributary to Ravine Lake
Address internal loads	Sediment inactivation treatment	Review and implement a sediment inactivation treatment to reduce lake bottom sediment phosphorus loads
Aquatic Plants	Invasive species management	Continue to monitor invasive species growth (e.g., curly-leaf pondweed) and continue management as needed
	Promote native aquatic plant growth	Encourage native plant reestablishment during and following aquatic invasive species management Monitor success of native vegetation transplant projects
Phytoplankton and Zooplankton	Phytoplankton/Zooplankton monitoring	Consider monitoring phytoplankton and zooplankton as part of routine monitoring
Water Quality	Water quality monitoring	Continue monitoring in-lake water quality and assessing trends for eutrophication within Ravine Lake Continue to identify/track chloride levels from winter salt use Consider collecting water quality parameters at multiple depths to confirm lake stratification influence on internal phosphorus loading to surface waters Consider supplementing routine monitoring with additional nitrogen parameters for better assessment of phytoplankton nitrogen limitation and ammonia impairment Collect additional information on groundwater contributions into the lake (water quality, groundwater-surface water interactions) to confirm impacts

15 Trout Brook, O'Conners Creek, and O'Conners Lake Watershed and In-lake Modeling

15.1 Water Quality

15.1.1 O'Conners Lake

O'Conners Lake	
Shallow/Deep	Shallow
Location	Denmark Township
Surface Area (4-yr average)	59 acres
Average/Maximum Depth	4 feet / 13 feet
Watershed Area	6,305 acres
Watershed:Lake Surface Area	107:1
Impairment Status	Not listed on impaired waters list
Downstream Waterbody	Landlocked

O'Conners Lake is located in Denmark Township and is used primarily for wildlife viewing. O'Conners Creek flows into O'Conners Lake, which is landlocked with no surface outlet. Under typical climatic conditions water leaves O'Conners creek through groundwater discharge or evaporation only.

Because O'Conners Lake is landlocked, the observed historical water levels have been quite variable ranging from 796.9 in May 2009 to 806.5 in October 2020. In the most recent 4 years of monitoring, O'Conners Lake has had an average water surface area of approximately 59 acres, a maximum depth of 13 feet, and a mean depth of approximately 4 feet. O'Conners Lake is typically shallow enough for aquatic plants to grow over the entire waterbody and for the lake to mix many times per year (polymictic lake).

The State of Minnesota uses three water quality parameters to assess eutrophication standards within lakes—total phosphorus, chlorophyll-*a*, and Secchi disk transparency—to assess waterbody health and track water quality changes. These three parameters were measured in O'Conners Lake by the SWWD between 2005 – 2023 (Figure 15-1). Monitoring data collected between 2011 – 2012, 2019, and 2021 were collected at a reduced frequency from other monitored years. Not enough measurements were collected to calculate summer averages for those years. During the monitored years with adequate data, summer average total phosphorus concentrations were better than the state standard between 2005 – 2008, 2013 – 2014, and 2016 – 2018. Summer average chlorophyll-*a* concentrations were better than the state standard between 2005 – 2008 and in 2014, and 2017. Summer average Secchi depths met or were better than the state standard for all monitored years between 2005 – 2023 except 2007 and 2010. At the time of this study, O'Conners Lake was not included on the Minnesota impaired water list.

Chloride concentrations were measured by the SWWD between 2011 – 2018 and 2020 – 2023 (generally between May and September). In the historical record, all observed chloride concentrations were well below the MPCA chronic standard of 230 mg/L. The highest observed chloride concentration was 40 mg/L in May 2020. Most observed chloride concentrations are below 30 mg/L.

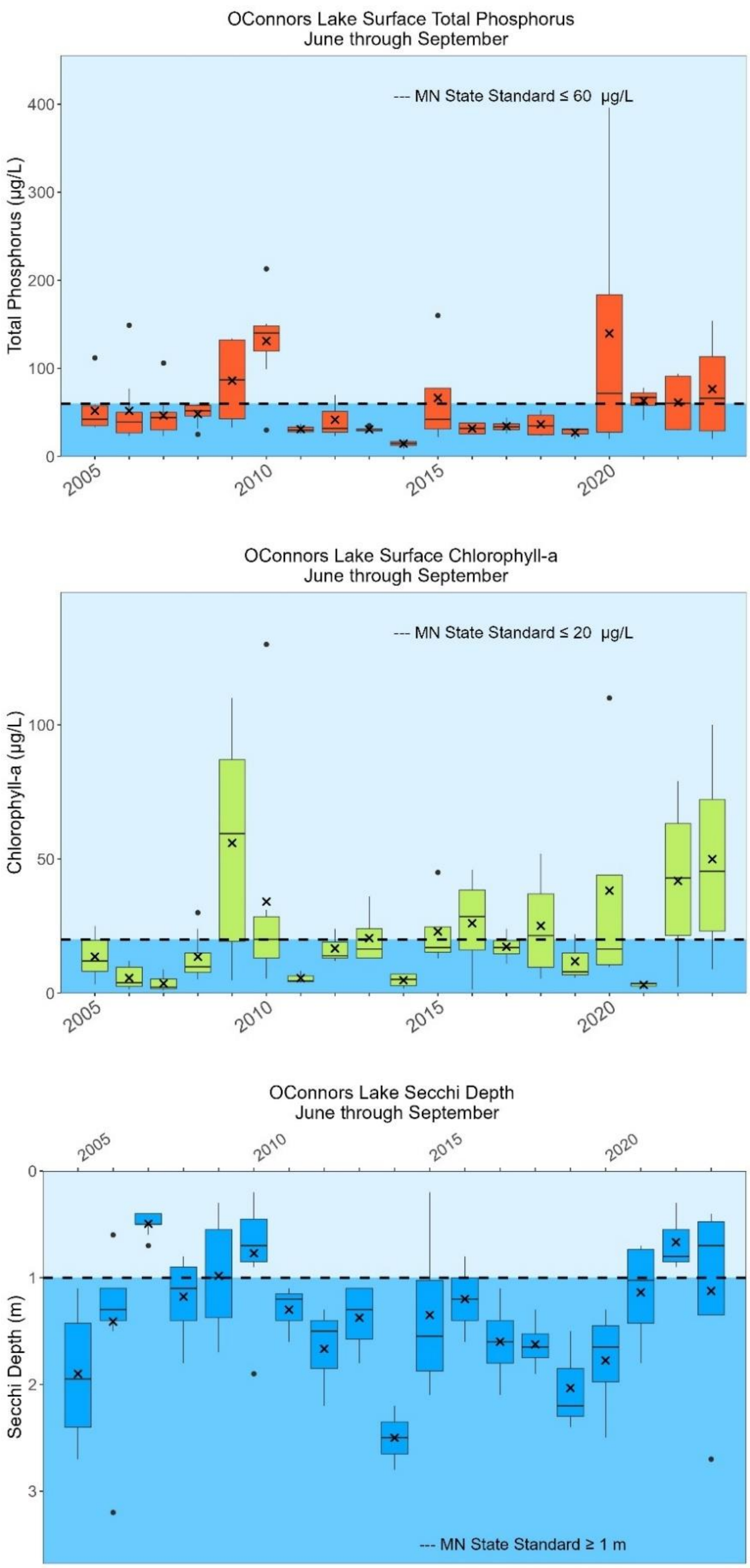


Figure 15-1 O'Conners Lake eutrophication monitoring data (June – September)
Summer averages are shown by x's in the box plots

15.2 Ecological Health

15.2.1 Aquatic Plants

There have been no aquatic plant surveys completed in the last 19 years on O’Conners Lake. Aquatic plant surveys were conducted in 2005 (Emmons & Olivier Resources (EOR), O'Conner's Stream and Lake Management Plan, 2007); however, the data may no longer reflect current lake conditions since water levels between 2020 - 2023 have been 3 – 6 feet higher than what was observed between 2004 – 2005.

15.2.2 Fisheries

There are no MNDNR fish survey or stocking data available for O’Conners Lake.

15.3 Watershed Total Phosphorus Loads

The Trout Brook and O’Conners Creek subwatersheds are located in the eastern portion of the SWWD, and are part of the larger Lower St Croix River Watershed. This portion of the SWWD is more rural in nature than other areas of the District. Primary land cover includes agriculture, forests, and grasslands.

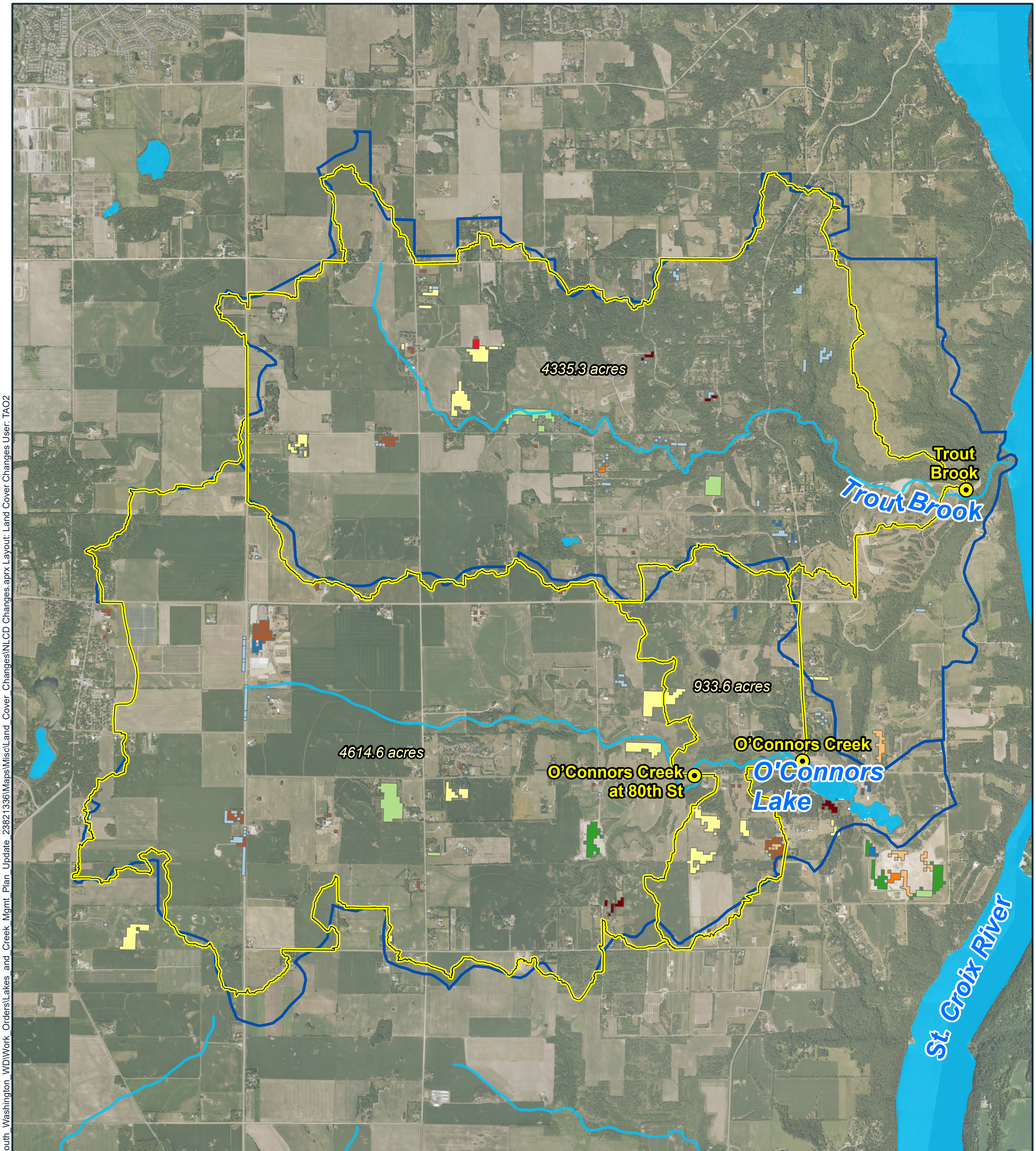
The MPCA completed a TMDL study for the Lake St Croix Watershed in 2012 which includes the Trout Brook and O’Conners Creek subwatersheds. This 2012 study included work to estimate TP areal loading rates for the subwatersheds tributary to Lake St Croix and setting load reduction goals for those areas. Results from the TMDL study indicated an estimated average annual loading of 0.35 lbs TP/acre/year from the Trout Brook and O’Conners Creek subwatersheds. These loads were estimated using information on land uses within these subwatersheds in the early 1990s. The TMDL identified a goal of reducing loads from this area to achieve an average annual loading of 0.22 lbs TP/acre/year to improve water quality conditions within Lake St Croix.

The SWWD has been collecting water quality and flow monitoring data within the Trout Brook and O’Conners Creek subwatersheds since the early 2000s. Water quality sampling for total phosphorus and total suspended solids (TSS) began in 2007, paused for a few years, and then picked up again in the early 2010s. Monitoring data collected by the SWWD was analyzed for this study in order to: 1) assess current water quality conditions within the Trout Brook and O’Conners Creek subwatersheds by estimating current TP loading rates as compared to the goals established within the 2012 TMDL; and 2) to look for trends and correlations within the data that may help to inform future management strategies for improving water quality.

15.3.1 Watershed Characteristics and Monitoring

The monitoring data analyzed for this task was downloaded from the SWWD’s water quality database (wq.swwdmn.org) for the Trout Brook, O’Conners Creek, and O’Conners at 80th monitoring sites.

Figure 15-2 shows the location of these monitoring sites. The original O’Conners Creek monitoring site was located at the crossing with St. Croix Trail; this site was relocated upstream to the O’Conners at 80th location following the 2020 monitoring season, due to on-going challenges with tailwater from O’Conners lake impacting monitoring at the original location. Data collection began at the new monitoring location in 2022.



Monitoring Site
Watershed Boundary
Monitoring Site Subwatershed

2004 to 2021 NLCD Changes

- Agriculture, Developed
- Agriculture, Forest
- Agriculture, Grassland
- Agriculture, Shrubland
- Forest, Agriculture
- Forest, Developed
- Forest, Grassland
- Forest, Shrubland
- Forest, Water
- Grassland, Agriculture
- Grassland, Developed
- Grassland, Forest
- Grassland, Shrubland
- Shrubland, Forest
- Shrubland, Grassland
- Shrubland, Water
- Water, Forest

N

0 2,000 4,000
Feet

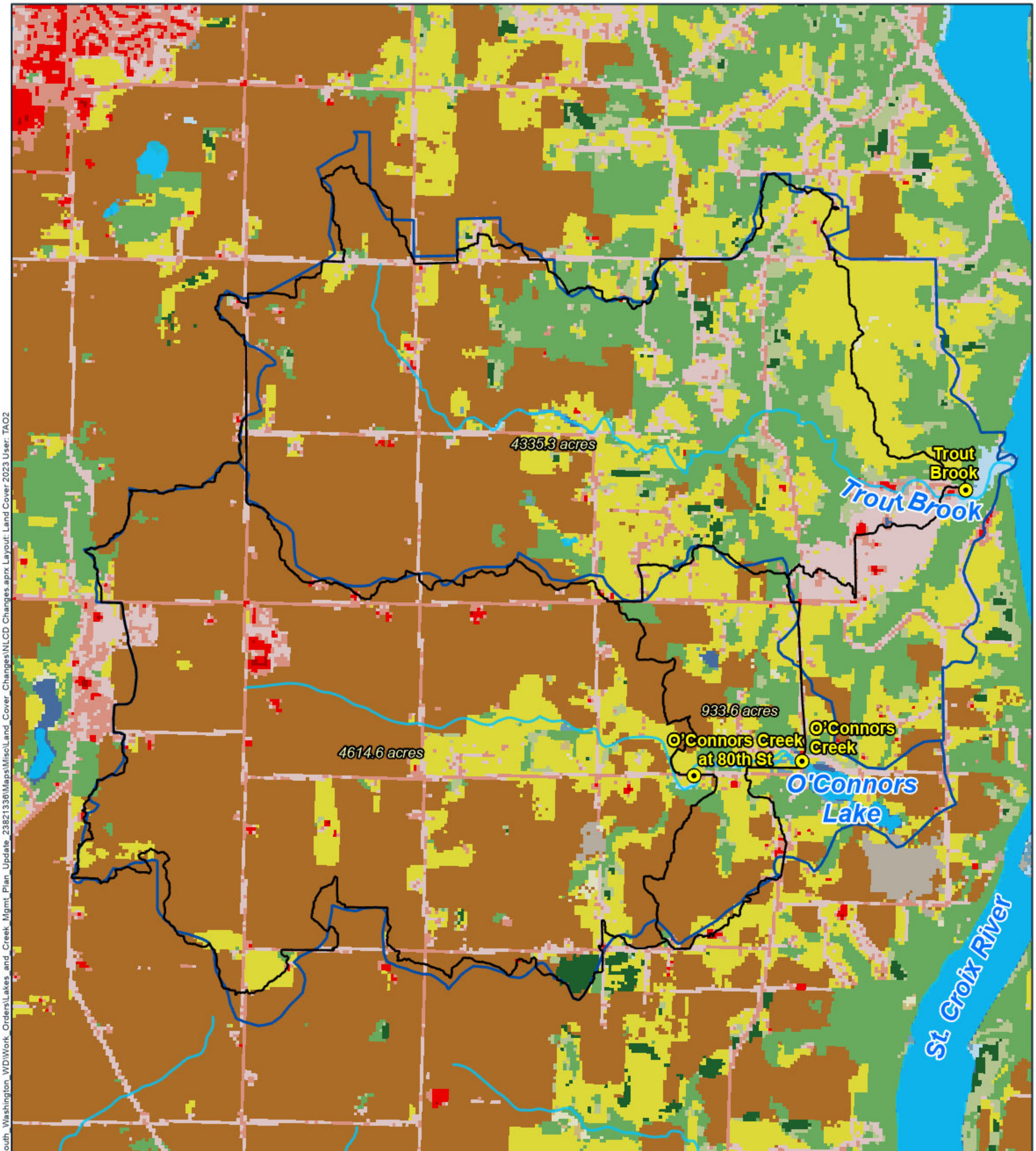
Trout Brook and O'Connors Creek Subwatersheds, Monitoring Locations, and Land Use Changes
Lake Management Plan
South Washington Watershed District
FIGURE 15-2

BARR

The total drainage areas for the Trout Brook and O’Conners Creek subwatersheds and contributing areas to the various monitoring locations are listed in Table 15-1. Landcover within the watersheds as of 2023 is shown in Figure 15-3; the percent landcover type in each watershed from 2023 is shown in Table 15-2. The landcover within these watersheds has been fairly stable over the period of the monitoring record. Changes within landcover type (based on the NLCD) between 2004 and 2021 are presented in Figure 15-2. Differences between landcovers between the Trout Brook and O’Conners Creek watersheds may contribute to different loading mechanisms when considering water quality conditions within the two systems. This is discussed further in review of the monitoring data.

Table 15-1 Total subwatershed area and contributing area to monitoring locations and O’Conners Lake (acres)

Trout Brook		O’Conners Creek			
Watershed	Monitoring Station	Watershed	Lake	Monitoring Station (2004-2020)	Monitoring Station at 80th (2022-present)
5,407	4,335	6,305	6,305	5,548	4,615



Monitoring Site
Watershed Boundary
Monitoring Site Subwatershed

NLCD 2023 Land Cover

- Open Water
- Developed, Open Space
- Developed, Low Intensity
- Developed, Medium Intensity
- Developed, High Intensity
- Barren Land
- Deciduous Forest
- Evergreen Forest
- Mixed Forest
- Shrub/Scrub
- Grassland/Herbaceous
- Pasture/Hay
- Cultivated Crops
- Woody Wetlands
- Emergent Herbaceous Wetlands

N

0 2,000 4,000
Feet

O'Connors and Trout Brook Watershed Land Cover (NLCD 2023)

Lake Management Plan
South Washington Watershed District

FIGURE 15-3

BARR

Table 15-2 Percent landcover type in Trout Brook and O'Conners Creek watersheds (2023)

Landcover Type	Trout Brook	O'Conners Creek
Agriculture	36.9%	67%
Forest	27.5%	6%
Grassland	22.4%	16%
Shrubland	0.1%	1.5%
Urban	13%	9%
Water	0.2%	1%

Raw monitoring data from the three monitoring locations along these creeks are presented in Figure 15-4. Mean values for observed data are presented in Table 15-3. Plots of yearly, monthly, and seasonal concentrations are presented in Appendix C.

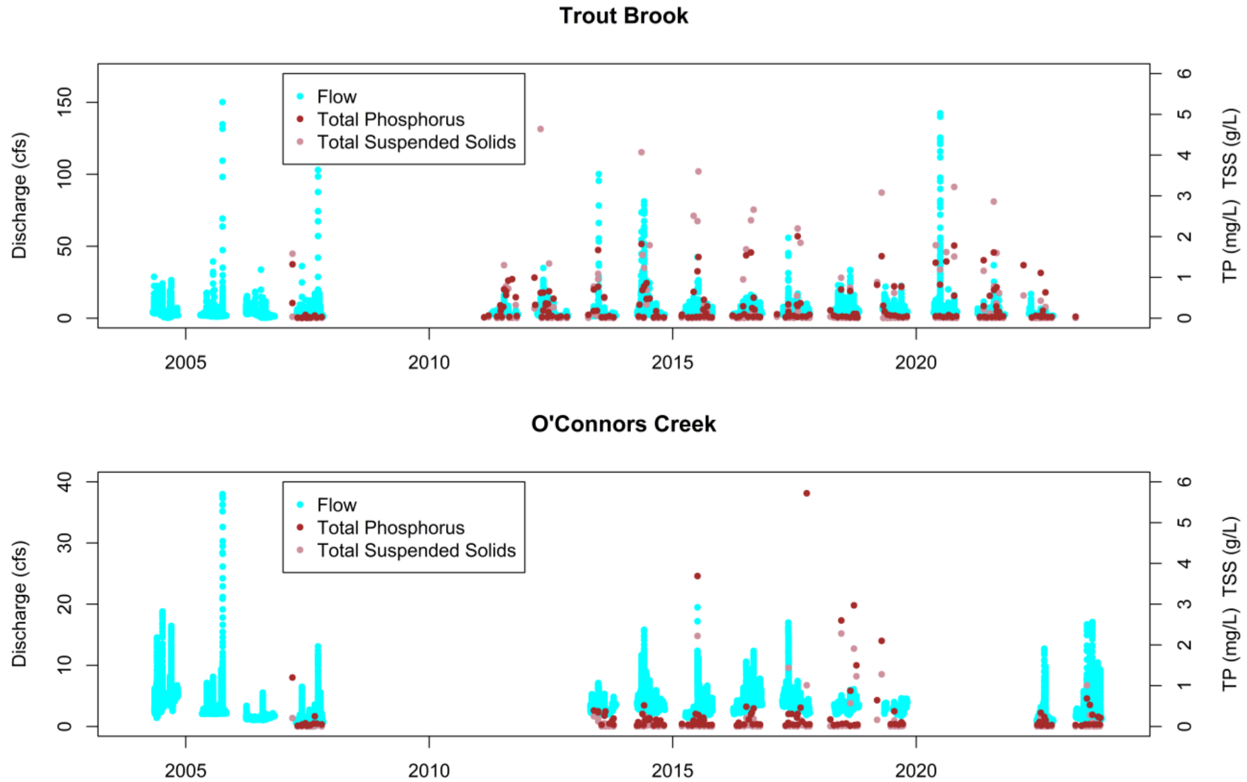


Figure 15-4 Observed flow, TP, and TSS in the Trout Brook and O'Conners Creek[1] watersheds.

*Note that the location of the O'Conners Creek monitoring station changed after the 2020 monitoring season.

Table 15-3 Mean values for observed flow, TP, and TSS for the Trout Brook and O'Conners Creek watersheds over the full historical records

Trout Brook (2005 – 2022)			O'Conners Creek (2005 – 2023)		
Observed Flow (cfs)	TP (mg/L)	TSS (mg/L)	Observed Flow (cfs)	TP (mg/L)	TSS (mg/L)
2.76	0.329	501	2.51	0.294	135

Some simple observations can be made from the information in Figure 15-4 and Table 15-3. Although the mean observed flows are similar, the range of observed flows in Trout Brook is greater than in O'Conners Creek, with peak observed flows being nearly 4 times higher in Trout Brook. Although mean TP concentrations are similar, the mean TSS concentration in Trout Brook is higher than in O'Conners Creek. As a result, the ratio of TSS to TP is higher for Trout Brook, suggesting that the source of phosphorus and the loading mechanism for Trout Brook is different from O'Conners Creek.

15.3.2 Monitoring Data Analysis

Plots of observed streamflow vs. total phosphorus and total suspended solids concentrations were created to observe any apparent correlation between observed flow and pollutant concentrations (Figure 15-5 and Figure 15-6). Based on visual observation, there does not appear to be a strong correlation between observed streamflow and pollutant concentrations over the observed record (i.e., the data are not strongly linear). However, the data show high concentrations of both TP and TSS in Trout Brook under most flow regimes, while high concentrations of both TP and TSS are primarily seen under lower flow events in O'Conners Creek.

Note that some of the concentrations reported within these plots are from averaged flows over a long sample duration (i.e., multiple hours). These datapoints are reflective of composite sample monitoring results where samples are collected and then composited for laboratory analysis. Results are presented as a single (composite, shown in red) pollutant concentration over the entire storm event, such that concentrations are presented as an average over the sampling time period. This is in contrast to grab samples (shown in green), which are reported as an instantaneous result at the time of sampling.

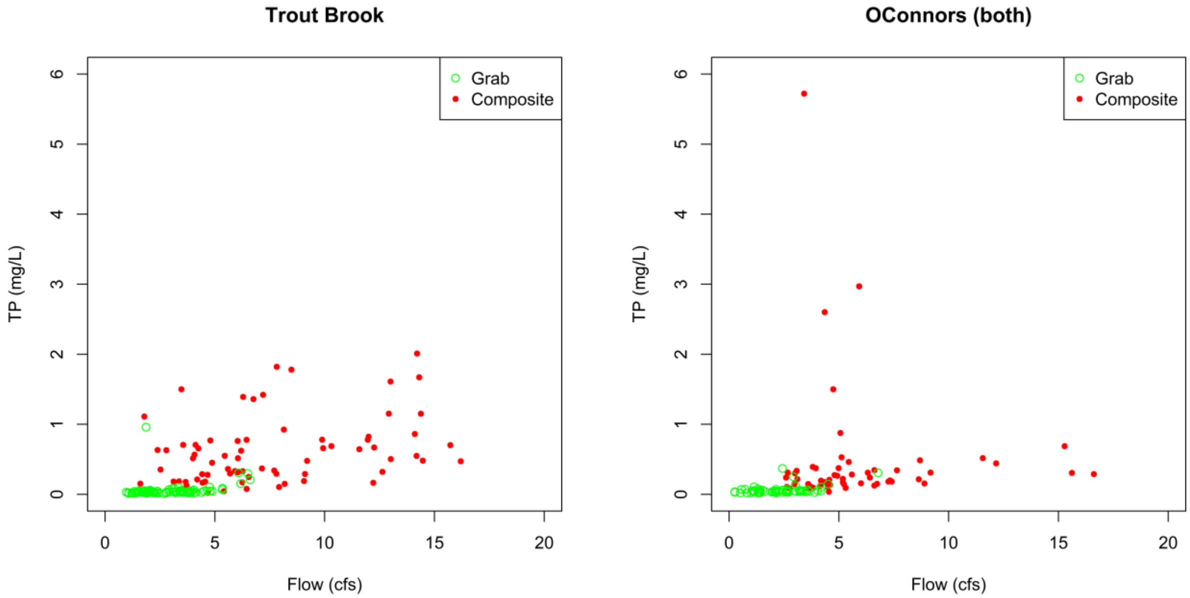


Figure 15-5 Observed flow versus total phosphorus for Trout Brook and O'Conners Creek

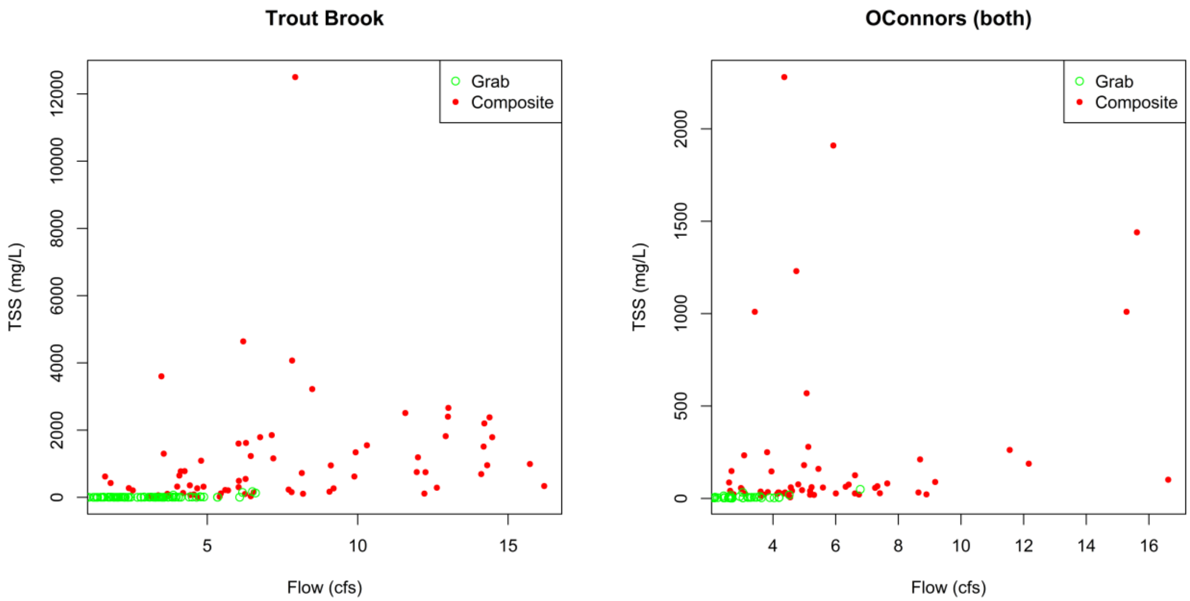


Figure 15-6 Observed Flow versus total suspended solids for Trout Brook and O'Conners Creek

15.3.2.1 Baseflow Separation

As a next step, the separation of baseflow from storm event samples was investigated to consider how the concentration of total phosphorus within the streams varied under conditions when observed flow was primarily a function of baseflow vs. during storm runoff events.

Consistent with the methods used in Janke (2015), baseflow for both Trout Brook and O'Conners Creek was estimated using the sliding interval method as described by Sloto and Crouse (1996). Using this method, storm events are then identified using breakpoints, where one event ended and the next began, which are defined as the point in time where baseflow equals observed flow (i.e., baseflow = total flow). Precipitation is included in Figure 15-7 to help visualize the timing of increased observed flow due to storm events.

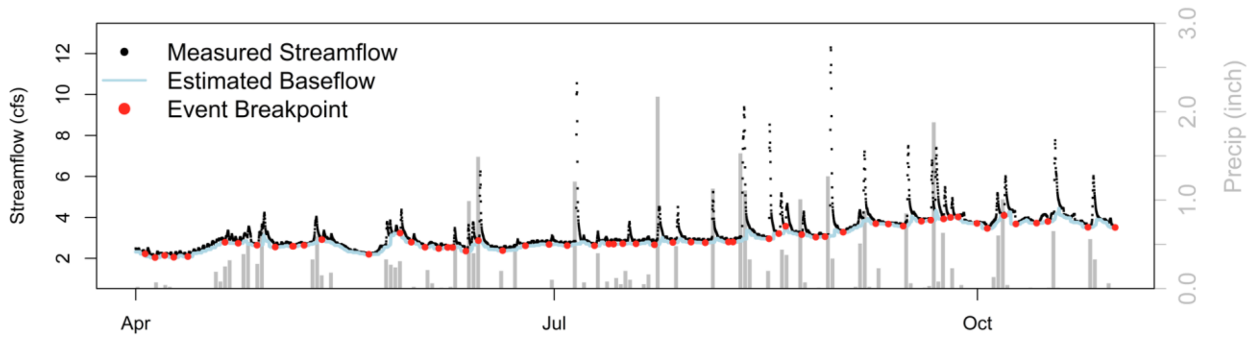
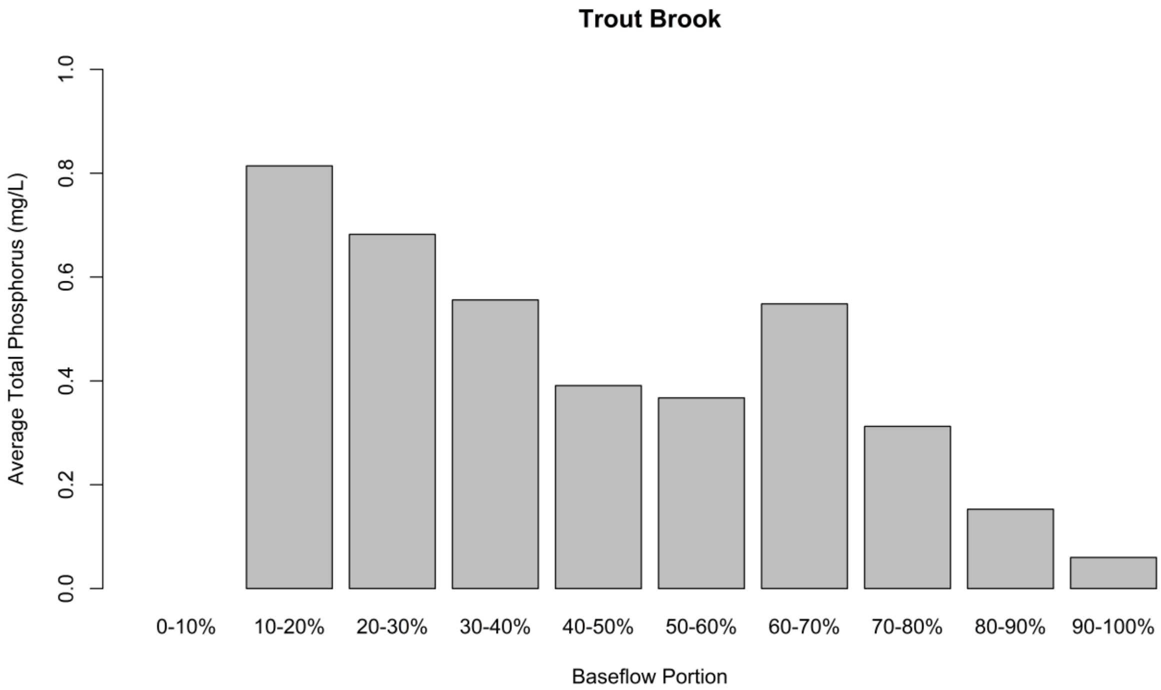


Figure 15-7 Observed flow, estimated baseflow, and precipitation in the O'Conners Creek watershed for 2016

Results from the baseflow separation analysis were then used to compare the average concentration of total phosphorus across the different ranges of percent baseflow present for the period of record for each watershed (Figure 15-8). Using this approach, we see the average total phosphorus concentration observed within the baseflow (i.e., proportions greater than 90%) is estimated to be ~0.06 mg/L for both Trout Brook and O'Conners Creek. However, it is notable that the relationship between baseflow proportion and average total phosphorus is different for Trout Brook and O'Conners Creek. For Trout Brook, lower baseflow proportions (e.g., less baseflow as a fraction of total observed flow) correspond to higher total phosphorus. This suggests, perhaps, that runoff and surface flows across the landscape that generate erosion and solids are primary drivers of loading to Trout Brook. Phosphorus concentrations are more uniform across different flow conditions for O'Conners Creek (e.g., baseflow as a fraction of total observed flow), suggesting that the source and delivery mechanism for O'Conners Creek differs from Trout Brook.



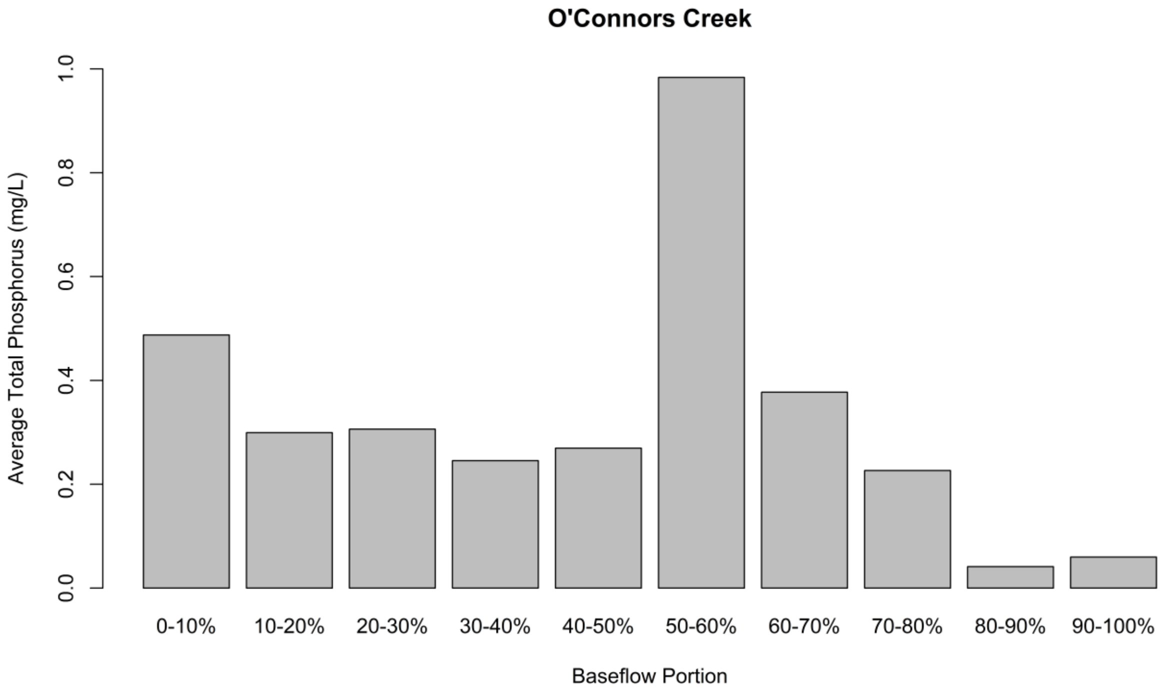


Figure 15-8 Baseflow portion of total observed flow versus average observed total phosphorus concentration

15.3.3 Estimating Phosphorus Loads

LOADEST, a US Geological Survey program for estimating constituent loads in streams and rivers, was used to estimate average annual total phosphorus and total suspended solids loading within the Trout Brook and O'Conners Creek watersheds. LOADEST uses inputs of time series of observed streamflow and point water quality data to develop regression models, which are then used to estimate pollutant loads. The program provides estimates of pollutant loading and reports on the statistical uncertainty associated with these estimates, on a monthly and seasonal basis.

LOADEST requires the time step for inputs on observed concentration and flow data to coincide. It also requires that flow data be presented as a continuous record. Raw monitoring data were pre-processed within R (a computer program for statistical computing) to translate observed flows (from the SWWD data) from a 15-minute interval to a daily average in order to create properly formatted data inputs for LOADEST. Gaps within the observed flow record were filled by using linear regression between observed data points to create a continuous calibration flow dataset. Pollutant concentrations were also averaged daily.

LOADEST was used to estimate TP and TSS loads at the Trout Brook monitoring station for the years 2011-2021. Pollutant loads were estimated at the O'Conners Creek monitoring stations for the years 2013-2019 and 2022-2023. Results from the LOADEST calculations are summarized in Table 15-4. Plots of the estimated annual TP and TSS loading at the two monitoring locations, and the reported 95 percent confidence intervals associated with the estimates are shown in Figure 15-9. As shown in Figure 15-9, the estimated TP and TSS loadings computed at the Trout Brook monitoring location have a high level of uncertainty associated with them. Results from the statistical analysis within LOADEST suggest an estimated load bias of 45% for TP and over 243% for TSS, and Nash Sutcliffe Efficiency values of -5.27 and -36.73 for TP and TSS, respectively. Errors of this magnitude suggest that the statistical relationship resulting from the LOADEST regression analysis is not a good fit for the accurate estimation of loads (i.e., the resultant loading values have a high magnitude of error). In cases such as this, it is suggested that using the mean observed concentration and flow at the monitoring location to compute an observed mean load would be a more accurate representation. For this reason, mean annual flow, concentration, and TP load were calculated for both subwatersheds; and are also included in Table 15-4.

The statistical fit of the regression equation developed from the O'Conners Creek monitoring data is much better, with an estimated bias of -24% for TP and -19% for TSS, and Nash Sutcliffe Efficiency values of 0.24 and 0.45 for TP and TSS, respectively. Results are within what would be considered a 'good fit' for the statistical relationship. Ninety-five percent confidence intervals for estimated TP and TSS loads at the O'Conners Creek monitoring stations are shown in Figure 15-9.

Table 15-4 Load results for Trout Brook and O'Conners Creek

Year	Precipitation	Trout Brook (lb/ac/yr)			O'Conners Creek (lb/ac/yr)[3]		
	(in)	TP (LOADEST[1])	TP (Mean Annual [2])	TSS (LOADEST[1])	TP (LOADEST[1])	TP (Mean Annual [2])	TSS (LOADEST[1])
2011	33	0.442	0.41	939			
2012	28	0.121	0.25	147			
2013	32	0.101	0.33	417	0.104	0.21	17
2014	40	0.332	0.58	1383	0.141	0.16	37
2015	42	0.165	0.38	308	0.087	0.26	28
2016	41	0.128	0.45	205	0.176	0.20	57
2017	36	0.189	0.46	321	0.169	0.64	76
2018	33	0.236	0.29	414	0.140	0.74	55
2019	42	0.405	0.68	729	0.141	0.45	61
2020	36	0.601	1.01	1655			
2021	32	0.165	0.49	155			
2022	27				0.026	0.06	10
2023	32				0.039	0.12	18
MEAN		**0.262**	**0.485**	**607**	**0.114**	**0.316**	**40**

[1] Results from the LOADEST calculations, using AMLE (adjusted maximum likelihood estimation) which provides an estimateof instantaneous load based on daily average flow and concentration.
[2] Mean Annual values computed as a function of mean annual concentration and mean annual flows, based on monitoring data from open water season only.
[3] Change in monitoring location from O'Conners Creek to 80th applied on 1/1/2021.

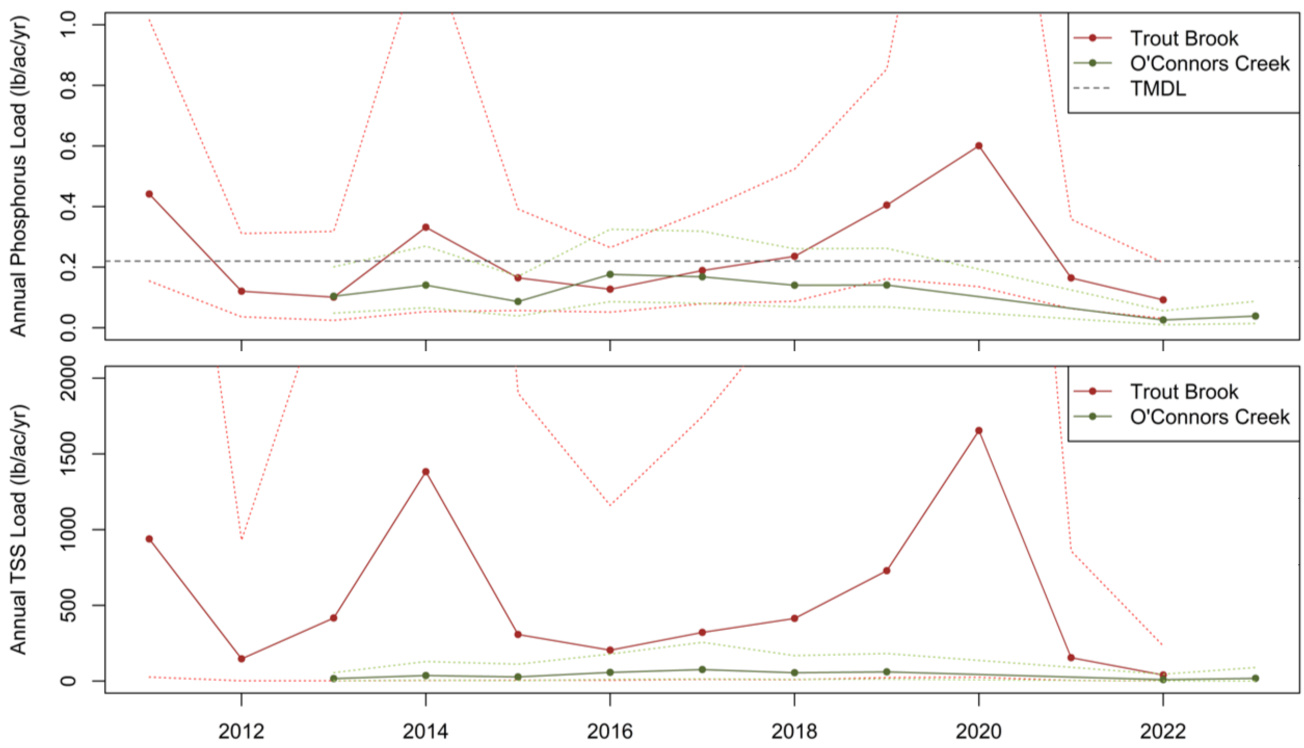


Figure 15-9 LOADEST results for annual loading of total phosphorus and suspended solids in Trout Brook and O'Conners Creek watersheds
95% confidence intervals, based on daily average flow and concentrations, shown as dashed lines. Dashed grey line in top plot is goal for TP loading set by the 2012 TMDL.

As an arithmetic check of the LOADEST results, mean annual TP loading was also calculated for these stations, using annual mean concentrations and annual mean (observed) flows. This method is oversimplified in that it uses noncontinuous data from the flow season to represent the whole year and also is based only on observed flows from the open water season, ignoring low flows during the winter months, however, it can be used to provide a reference. Results from this analysis show that results from the LOADEST TP loading estimates are within the correct range (i.e., LOADEST results are, for the most part, lower than the computed annual means, which would be expected as annual mean flow rates were based upon observed flows during the open water season and do not account for low flows during the winter months).

15.3.4 Conclusions

Both O'Conners Creek and Trout Brook are considered part of the Central Region as related to the State of MN's river eutrophication standards. Current state guidance denotes that rivers classified within Class 2A (i.e., set for a beneficial use of supporting aquatic life) in the Central Region should have an average total phosphorus concentration <100 µg/L. The long-term mean TP concentration observed within both the Trout Brook and O'Conners Creek watersheds are substantially above that standard (Table 15-3). The majority of the landcover within the Trout Brook and O'Conners Creek watersheds is designated as agricultural use, which can be a high source of phosphorus loading and could be contributing to the high TP concentrations. This is particularly true for O'Conners Creek. In addition, an increase in TP concentration can be seen during the summer season (Appendix C), leading to the assumption that prevailing TP concentrations may be due in part to agricultural activities (e.g., fertilization). TP in Trout

Brook may also be a function of erosional processes, judging from the observed relationships between flow rate, TSS, TP, and baseflow.

Based on the average observed flow, concentration, and contributing areas for the Trout Brook and O'Conners Creek watersheds, Trout Brook would be expected to produce slightly higher TP loads to O'Conners Creek and TSS loads 3-4 times that of O'Conners Creek. However, the LOADEST results show Trout Brook has TP loads over twice as high as O'Conners Creek and TSS loads over 10 times higher than O'Conners Creek. During baseflow conditions, both watersheds appear to have similar TP concentrations; observed TP concentrations in Trout Brook are about twice as high as in O'Conners Creek during high flow events.

Results from this analysis show that TP loading within the O'Conners Creek watershed is typically meeting the goals established by the 2012 Lake Saint Croix TMDL. In the Trout Brook watershed, the estimated TP loading is often exceeding the goal established in the TMDL; however, the calculation error and level of uncertainty associated with the loading estimates within the Trout Book watershed are high. Further analysis would be needed to address uncertainties within the data sets and develop a more refined estimate of loading. However, results from this analysis can be used to provide an understanding of the magnitude of nutrient loading within the watershed. High TP concentrations corresponding with observed high flows and low baseflow proportions suggest that erosion may be a contributing factor to nutrient loads within the system.

15.4 In-lake Total Phosphorus Loads

15.4.1 BATHTUB Modeling for O'Conners Lake

The O'Conners Lake response to phosphorus loading was modeled using BATHTUB. BATHTUB is a series of empirical eutrophication models that predict the response to phosphorus inputs for morphologically complex lakes and reservoirs (Walker, 1999). Several models (subroutines) are available for use within the BATHTUB model; in this case, the Canfield-Bachmann model was used to predict the lake response to total phosphorus loads.

The Canfield-Bachmann model estimates the lake phosphorus sedimentation rate, which is needed to predict the relationship between in-lake phosphorus concentrations and phosphorus load inputs (Canfield Jr. & Bachmann, 1981). The phosphorus sedimentation rate is an estimate of net phosphorus loss from the water column through sedimentation to the lake bottom and is used in concert with lake-specific characteristics, such as annual phosphorus loading, mean depth, and hydraulic flushing rate, to predict in-lake phosphorus concentrations. These model predictions are compared to measured data to evaluate how well the model describes the lake system.

To quantify groundwater interactions with O'Conners Lake, Barr developed annual water balances for the years 2023 and 2024, which represent the only years with available overlapping datasets for water surface elevations and inflow monitoring in the contributing creek. Precipitation totals for these years were 31.7 inches (2023) and 34.9 inches (2024). The finite difference spreadsheet model was used to estimate groundwater fluxes around O'Conners Lake, incorporating the observed hydrologic inputs and outputs from the lake. Because O'Conners Lake is landlocked, the only outputs are evaporation. Model results estimated a net groundwater outflow from the lake of 3.3 meters per year in 2023 and 9.7 meters per year in 2024. These values were used in the BATHTUB modeling for O'Conners Lake.

Estimated watershed, internal, and atmospheric phosphorus loads were used to predict in-lake concentrations for years with precipitation conditions similar to those in 2023 and 2024, and where in-lake water quality data were available, to ensure consistency with the variable groundwater flux data. A model representing an average year model was then developed from those results.

Given O'Conners Lake is a landlocked system, the particulate sedimentation term was the primary calibration parameter used in the Canfield-Bachmann model. Calibration was performed by comparing the average annual modeled total phosphorus concentrations to observed values. Additionally, models were calibrated across a range of internal loading scenarios, given that no sediment cores were collected for O'Conners Lake. Barr chose a low, medium, and high internal loading rate based on sediment cores that were collected in lakes with similar watershed to lake area ratios.

15.4.2 Phosphorus Loading Summaries

Model results indicate that the estimated phosphorus budget for O'Conners Lake was dominated by watershed loading across all years, except 2023, which was a dry year. Watershed loads ranged from 140 to 781 lbs, representing 31% to 94% of the phosphorus loading to the lake, depending on assumptions associated with the internal loading rate (see Figure 15-10, Figure 15-11, and Figure 15-12). For dry years with a high assumed internal loading rate, the internal phosphorus load can represent up to 67% (304 lbs).

Estimated nutrient loads for O'Conners Lake have considerable uncertainty given: (1) there is limited monitoring data available for the system; (2) no sediment cores were collected from O'Conners Lake; (3) no water quality profile data is available for the lake; and (4) uncertainty in estimating groundwater interactions within the system, particularly given the understanding that groundwater-surface water interactions are an important consideration within this portion of the watershed.

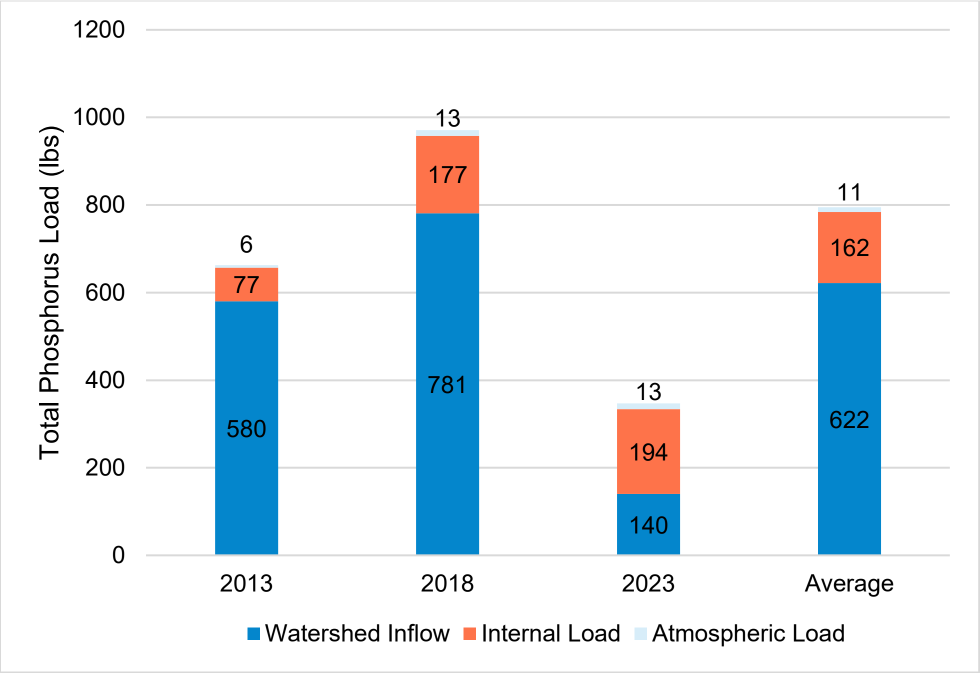


Figure 15-10 O'Conners Lake total phosphorus load estimates for average internal load (7.7 mg/m^2/d)

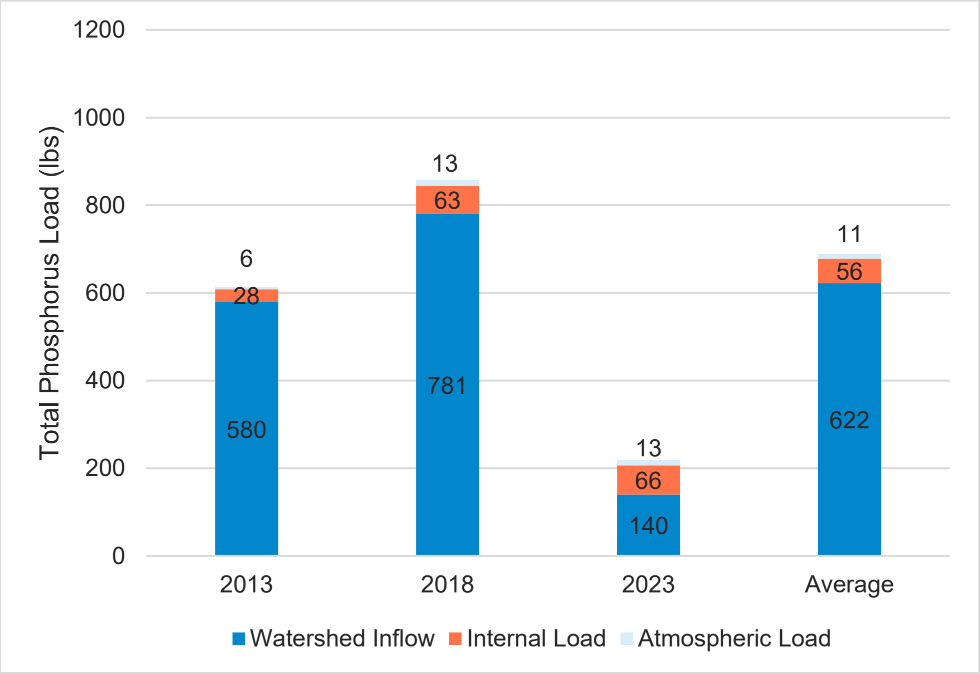


Figure 15-11 O'Conners Lake total phosphorus load estimates for low internal load (1.7 mg/m^2/d)

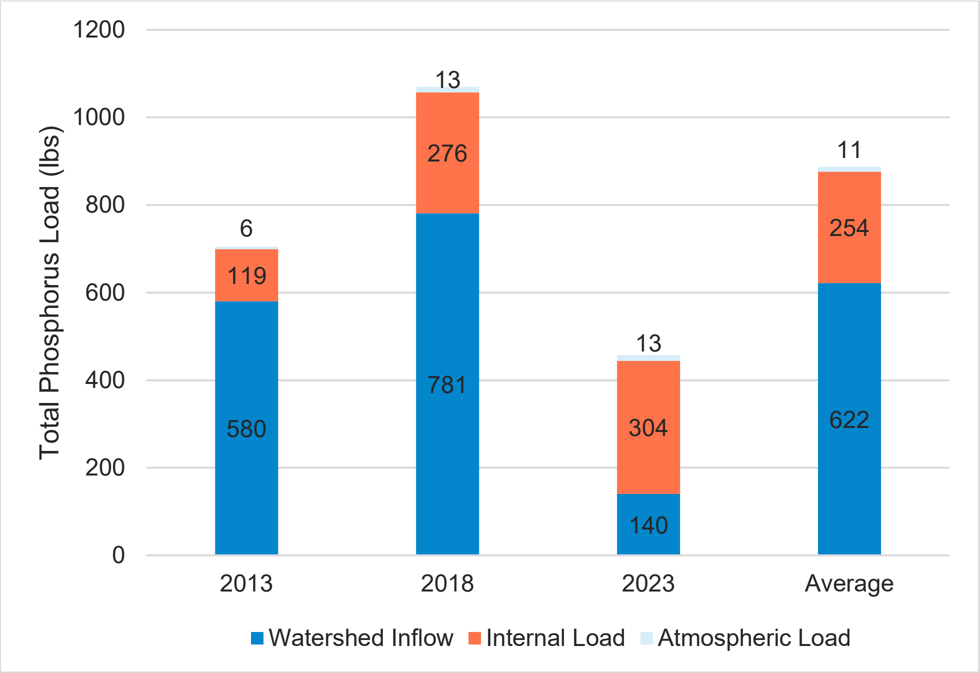


Figure 15-12 O'Conners Lake total phosphorus load estimates for high internal load (12.9 mg/m²/d)

15.4.3 Conclusions

Results from modeling within O'Conners Lake show that watershed runoff tends to dominate nutrient loading into the lake under typical conditions. Under dry conditions, internal loading may become a larger contributor to in-lake phosphorus concentrations. It's recommended that the SWWD continue to collect water quality monitoring data within O'Conners Lake and the contributing watershed. As additional data is available, it is recommended that the O'Conners Lake model(s) be revisited and updated based on additional information.

16 Conclusions

16.1.1 Summary of Phosphorus Loadings

Results of this study show that phosphorus loading to the nine suburban lakes that were included in the study varies widely between waterbodies. For most of the lakes, watershed loading can be a significant source of phosphorus loading during wet conditions. Under dry conditions, internal loading becomes more significant. For those lakes that are located at the downstream end of the chain of interconnected lakes located within the City of Woodbury, phosphorus loading from upstream waterbodies is a dominant source of loading under wet conditions; implying that management strategies to control phosphorus loads in upstream waterbodies could also have benefits for downstream receiving waters. Results from the modeling of a subset of management strategies to reduce phosphorus loading into eight of the nine suburban lakes included in this study are summarized in Sections 8–14.

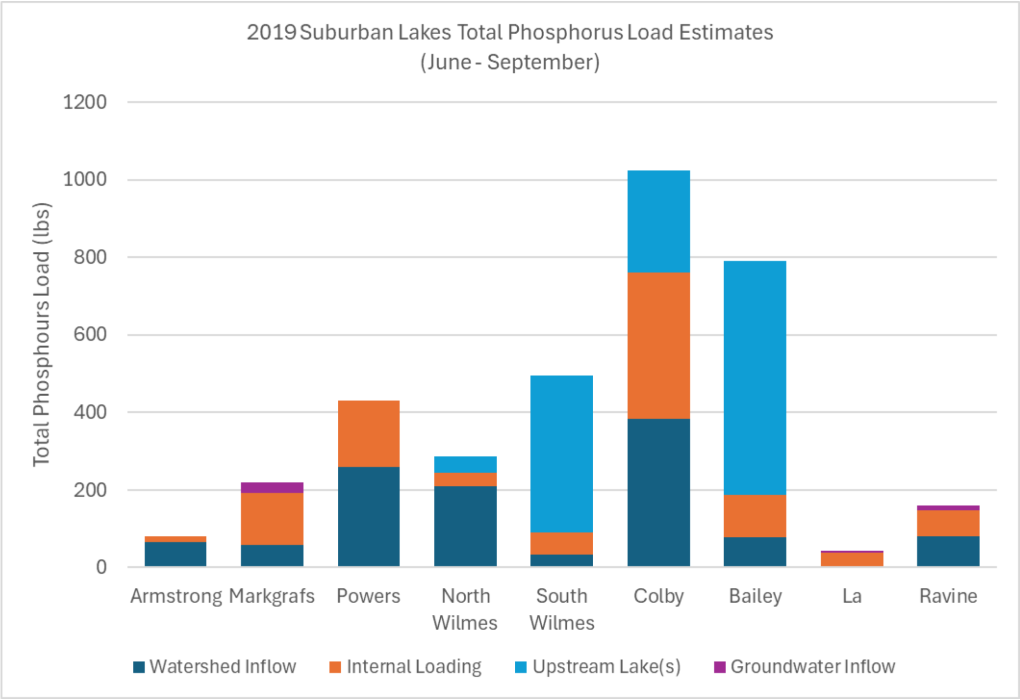


Figure 16-1 Summary of lakes total phosphorus loading for 2019 (wet year)

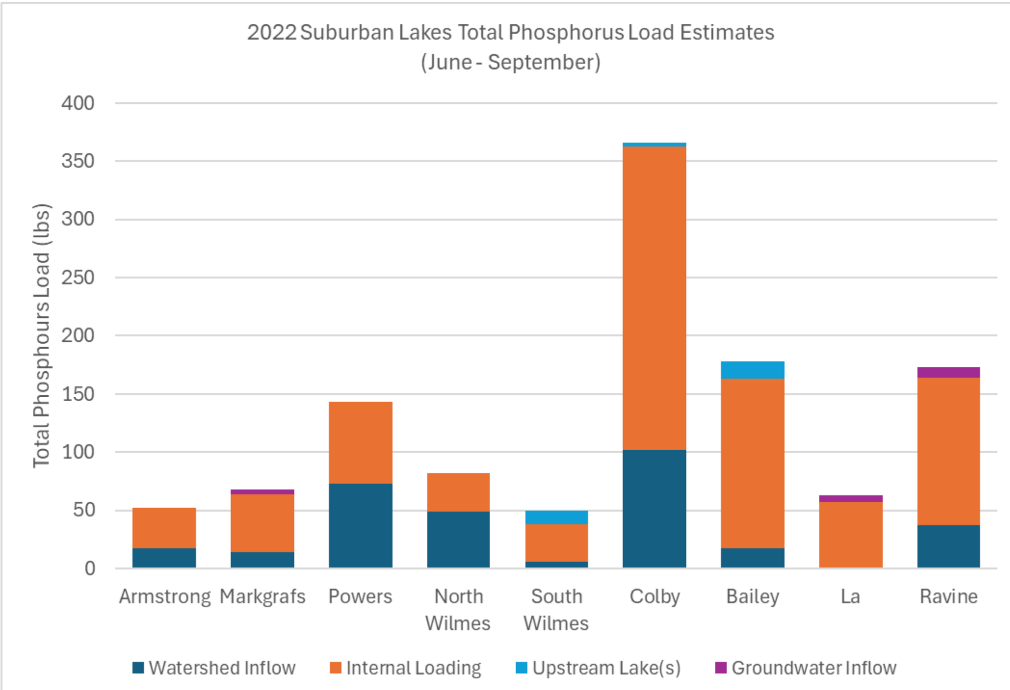


Figure 16-2 Summary of lakes total phosphorus loading for 2022 (dry year)

16.1.2 Considerations for In-lake Management

Results from the sediment coring and laboratory analysis performed in this study show that several of the lakes have high potential for internal phosphorus loading from their lake sediments. To better understand the potential impacts of performing lake sediment phosphorus inactivation projects in these lakes, Barr created planning-level management scenarios which were then applied to each of the calibrated in-lake models. These management scenarios were designed to first consider the potential impacts of performing sediment inactivation projects within the targeted lakes and then (when applicable) considering the scale of watershed load reduction also needed in order to achieve in-lake total phosphorus and chlorophyll-*a* concentrations in line with applicable state eutrophication standards.

Results of these analyses showed that for several of the lakes, sediment inactivation projects could have a considerable impact on in-lake water quality conditions, with the models predicting that in-lake treatment could reduce total phosphorus and chlorophyll-*a* concentrations to meet the state eutrophication standards within a single growing season. Results of the modeling for each of the lakes considered for in-lake treatments are summarized in Sections 8.5, 9.6, 10.6, 11.6, 12.5, 13.5, and 14.6.

The modeling developed for this study represents in-lake conditions and expected impacts from management strategies for a single growing season and does not account for longer-term water quality impacts over time. For example, once implemented, sediment inactivation efficacy is known to vary over time. Monitoring data from Twin Cities metro lakes indicate that efficacy declines in the years following initial treatment due to product aging, new watershed inflows, and burial from rough fish activity, which can again increase internal loading over time. Conversely, if water quality management projects improve water clarity, some lakes may experience a greater abundance and extent of submerged plants. Increased plant growth can reduce sediment resuspension, enhance nutrient uptake, and create increased competition with algae, leading to improved water quality. If the increased plant growth is associated with aquatic invasive species, however, this can cause additional concerns for water quality and may warrant the consideration of additional management strategies related to vegetation management. While modeling provides useful estimates of the expected scale of impacts of water quality management strategies, post-project monitoring is essential to track changes in lake water quality and adjust management plans as needed to continue benefits over time.

16.2 Summary of Findings and Recommendations

Table 16-1 summarizes key findings and takeaways from this study for the lakes that were included. The information within the table includes key considerations related to the lakes' water quality conditions, primary sources of phosphorus loading into the waterbodies, and planning for the management of the lakes in the coming years.

Table 16-1 Summary of findings related to lakes and management planning

Lake	Water Quality Condition	Physical Characteristics	Ranking of Watershed Avg Annual TP Load Relative to Others in the Study	Summary of P Loading into Waterbody by Major Source (Watershed vs. Internal)	Estimated Impact of Loading to Waterbody from Upstream Waterbodies	Notes on Internal Loading	Notes on Watershed Loading, District Priorities, and Other Recently-Completed Projects
Armstrong	Not listed as impaired; Avg TP concentrations typically exceed state standards; chl-a typically meets standard; Secchi disc depth typically not meeting.	Watershed:SA Ratio = 57:1 Very shallow (Avg depth = 1 ft)	Moderate	Wet (2019): 82% Watershed; 18% Internal Dry (2022): 33% Watershed; 67% Internal	N/A	Sediment cores show moderate potential for internal loading depending on lake conditions.	District piloting cattail harvesting in wetland immediately upstream as potential nutrient reduction strategy.
Markgrafs	Impaired; Avg TP concentrations far exceed state standards; chl-a exceeds standard; Secchi disc depth meets some years, does not meet others.	Watershed:SA Ratio = 10:1 Very shallow (Avg depth = 3 ft)	Moderate	Wet (2019): 26% Watershed; 62% Internal Dry (2022): 21% Watershed; 74% Internal	N/A	Sediment cores show moderate potential for internal loading depending on lake conditions. Modeling estimates internal sources >50% of loading in both wet and dry year.	Pockets of higher loading areas within the watershed with opportunity for retrofits; SWWD recently completed a watershed-wide retrofit study and plans to move forward with identified projects in the next few years.
Powers	Not listed as impaired; Avg TP concentrations meet state standards; chl-a meets standards some years, exceeds others; Secchi disc depth meets standard.	Watershed:SA Ratio = 21:1 Deep lake (Avg depth = 19 ft)	High	Wet (2019): 60% Watershed; 40% Internal Dry (2022): 51% Watershed; 49% Internal	N/A	Sediment cores show high potential for internal loading depending on lake conditions.	Pockets of higher loading areas within the watershed with opportunity for retrofits. Hasenbank Park stormwater BMPs constructed (operational in 2025) to treat regional stormwater flow from the eastern portion of Powers Lake watershed. District has received more calls in recent years expressing concerns about the algae.
North Wilmes	Impaired; Avg TP concentrations typically exceed state standards; chl-a typically exceeds standard; Secchi disc depth meets some years, does not meet others.[1]	Watershed:SA Ratio = 157:1 Moderate depth (Avg depth = 7 ft)	Moderate	Wet (2019): 73% Watershed; 13% Internal; 14% Upstream Dry (2022): 60% Watershed; 40% Internal	Some impact from loading from upstream waterbodies.	Sediment cores show high potential for internal loading depending on lake conditions.	Several areas in the watershed with higher loading and/or without stormwater treatment currently and opportunity for retrofits. Kargel Park Alum Treatment Facility began operation in 2025. Facility is expected to have benefits for both North and South Wilmes.
South Wilmes		Watershed:SA Ratio = 264:1 Moderate depth (Avg depth = 8 ft)	Moderate/Low	Wet (2019): 7% Watershed; 11% Internal; 82% Upstream Dry (2022): 13% Watershed; 63% Internal; 23% Upstream	Significant impact from loading from upstream waterbodies.	Sediment cores show high potential for internal loading depending on lake conditions; but relative loading is lower in 2019 / 2022.	
Colby	Impaired; Avg TP concentrations far exceed state standards; chl-a exceeds standard; Secchi disc depth meets some years, does not meet others.	Watershed:SA Ratio = 112:1 Shallow (Avg depth = 5 ft)	Moderate	Wet (2019): 37% Watershed; 37% Internal; 26% Upstream Dry (2022): 28% Watershed; 71% Internal; 1% Upstream	Notable impact from loading from upstream waterbodies.	Sediment cores show high potential for internal loading depending on lake conditions. Modeling estimates high internal loading under both wet and dry years.	Areas of higher loading or without treatment within the watershed with opportunity for retrofits.
Bailey	Impaired; Avg TP concentrations exceed state standards; chl-a exceeds standard; Secchi disc depth meets some years, does not meet others.	Watershed:SA Ratio = 230:1 Moderate depth (Avg depth = 6 ft)	Low	Wet (2019): 10% Watershed; 14% Internal; 76% Upstream Dry (2022): 10% Watershed; 82% Internal; 8% Upstream	Significant impact; particularly in wet years.	Sediment cores show high potential for internal loading depending on lake conditions.	District only recently began collecting water quality data in this waterbody. Considered a lower priority at this time for active management compared to other waterbodies.
La	De-listed in 2024; For recent years, avg TP concentrations, chl-a, and Secchi disc depth meet state standards.	Watershed:SA Ratio = 3:1 Shallow (Avg depth = 5 ft)	Low	Wet (2019): 7% Watershed; 77% Internal Dry (2022): 1% Watershed; 90% Internal	N/A	Modeling estimates high internal load compared to external sources.	Estimated watershed load is low. Small contributing drainage area and not much room for additional treatment.
Ravine	Impaired; Avg TP concentrations typically exceed state standards; chl-a exceeds standard; Secchi disc depth meets some years, does not meet others.	Watershed:SA Ratio = 68:1 Shallow (Avg depth = 5 ft)	Moderate/Low	Wet (2019): 51% Watershed; 42% Internal Dry (2022): 21% Watershed; 74% Internal	N/A	Sediment cores show high potential for internal loading depending on lake conditions. Modeling estimates high internal loading under both wet and dry years.	Contributing watershed is primarily rural and/or undeveloped. Questions about nitrogen impacts within the system and uncertainty in groundwater contributions.

Lake	Water Quality Condition	Physical Characteristics	Ranking of Watershed Avg Annual TP Load Relative to Others in the Study	Summary of P Loading into Waterbody by Major Source (Watershed vs. Internal)	Estimated Impact of Loading to Waterbody from Upstream Waterbodies	Notes on Internal Loading	Notes on Watershed Loading, District Priorities, and Other Recently-Completed Projects
O'Conners	Not listed as impaired: Avg TP concentrations and chl-a below state standards some years and exceed in others; Secchi disc depth typically meets standard.	Watershed:SA Ratio = 107:1 Shallow (Avg depth = 4 ft)	High	Avg: 78% Watershed; 20% Internal	N/A	No sediment cores taken. Modeling estimates internal sediment loading less of a concern than watershed loads.	District has limited data for this waterbody compared to others. Considered a lower priority at this time for active management comparatively.

[1] Water quality data is not currently available for North Wilmes Lake. Summary of water quality conditions is based on data collected in South Wilmes.

The following recommendations are made for next steps for the District to consider to further existing and develop additional management strategies to protect and improve water quality and ecological conditions within the lakes included as a part of this study.

Planning for implementation

- Study results show that phosphorus loading into most of the lakes within this study is heavily influenced by internal loading. Sediment coring and laboratory analysis confirm that a significant portion of this internal loading is expected to come from sediment loading. In particular, Markgrafs, Colby, Bailey, Ravine, and La are strongly influenced by internal loading under both wet and dry conditions.
- As part of next steps, we'd recommend the District to consider moving forward with the planning and implementation of in-lake sediment phosphorus inactivation projects for a selection of these high priority waterbodies. Based on results from this study and understanding of District priorities, we'd recommend Markgrafs, Colby, and Ravine as the first lakes to be considered for sediment inactivation projects. As monitoring continues, the District may then also want to consider treatments in additional waterbodies.
- This would entail performing feasibility studies to consider the different types of sediment phosphorus inactivation strategies that could be utilized and to identify the preferred option for each individual waterbody.
- As part of the effort, we would also recommend collecting additional data, as needed, to inform the feasibility study; and also developing a monitoring plan for assessing and ensuring the long-term success of any implemented treatments.
- A critical component of any long-term lake management strategy includes addressing external watershed loads. We recommend that the District continue their work to find opportunities to reduce and control watershed pollutant loading into their receiving waterbodies.

Monitoring

- Continue existing baseline monitoring and trend analysis for priority waterbodies and pollutants of concern for waterbody health.
- Continue monitoring to track and assess for the presence of aquatic invasive species and the health of the aquatic plant community and to inform management actions that may be needed to improve water quality and protect native aquatic plant health.
- As part of the District's work to develop an updated monitoring plan, consider recommendations noted within each of the individual lake summary sections as related to noted data gaps and/or additional information that would be useful to reduce uncertainty in the lake modeling and assessment of waterbody health and to inform management recommendations.
    - For those waterbodies known, or estimated within this study, to be most impacted by groundwater inflows (i.e., Ravine, O'Conners, Markgrafs, and La), collect additional information to help verify groundwater-surface water interactions and additional groundwater quality data to better estimate loading.

- Study results indicate that South Wilmes, Bailey, and Ravine Lakes would all benefit from additional water quality samples being collected in advance or as part of undertaking a feasibility study to further inform and refine the estimated internal loading influence on the lakes’ surface water quality.
- Consider including North Wilmes within the District’s baseline monitoring program.
- Consider including phytoplankton and zooplankton monitoring as part of the District’s lake monitoring program.

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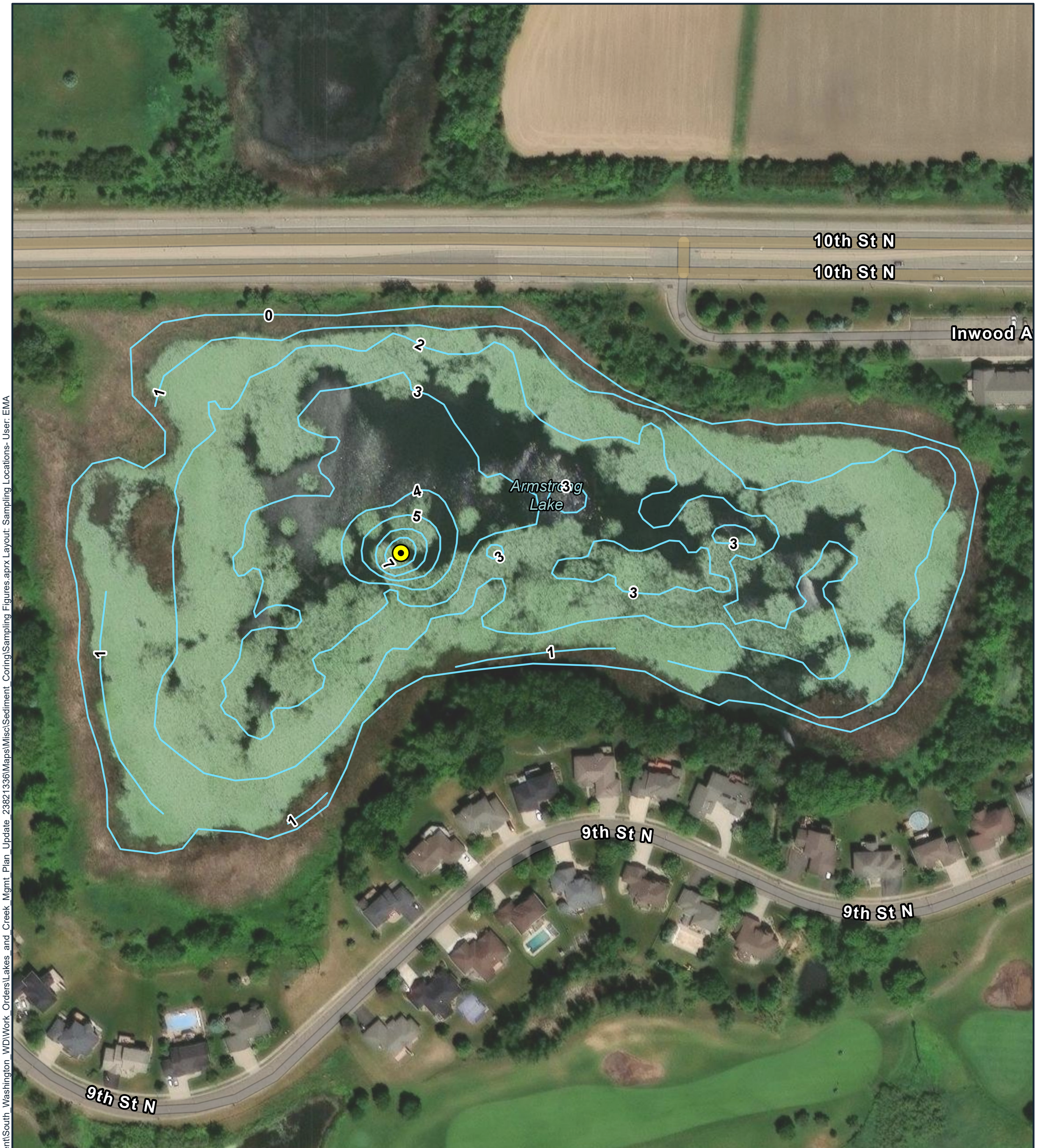
Appendices

Appendix A

Sediment Coring Locations



Coring Location

Bathymetric Contours

0 100 200
Feet

Armstrong
Sediment Coring Locations

SWWD Lakes and Creek Management Plan

BARR

Coring Location

Bathymetric Contours

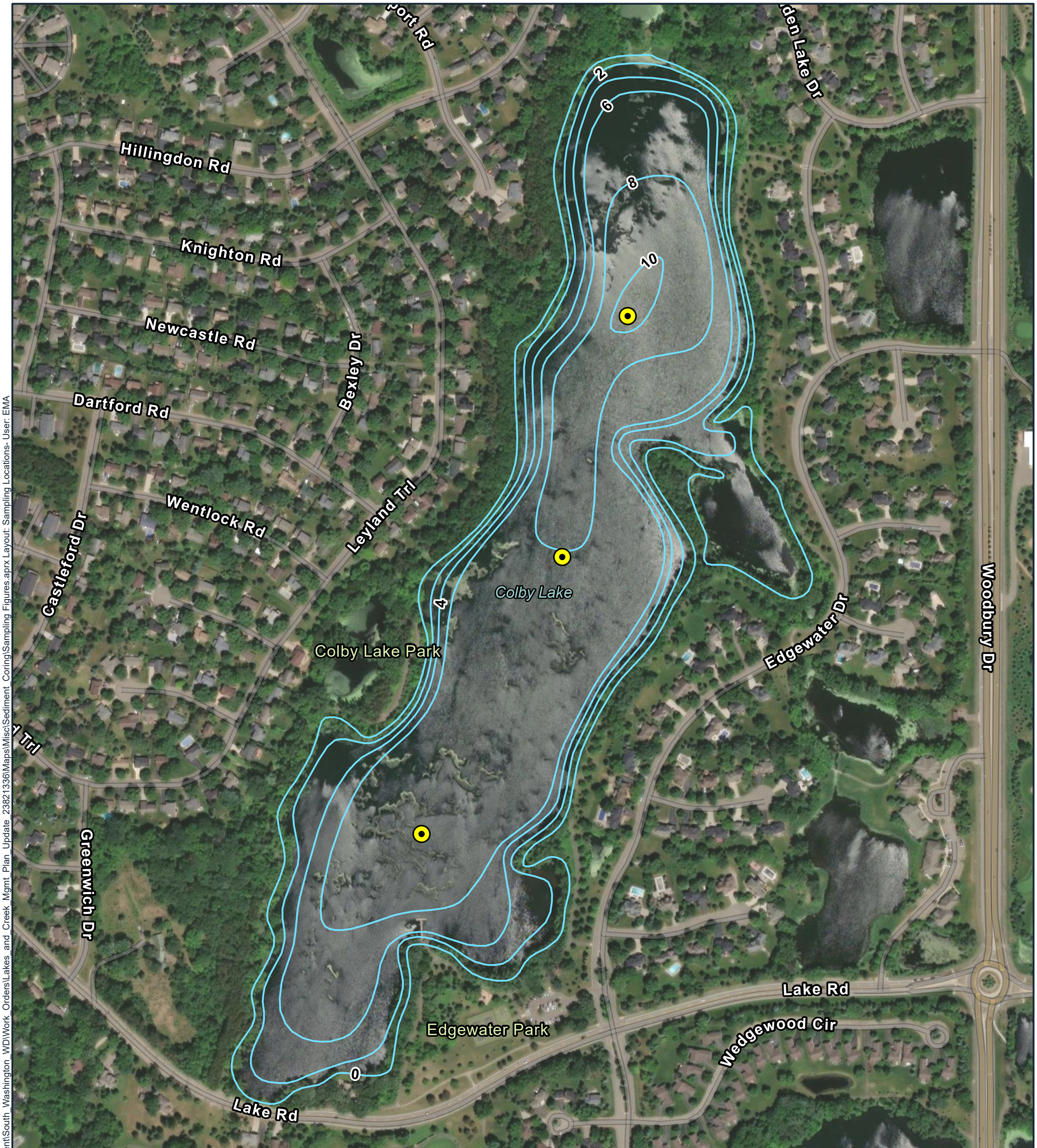
0 250 500

Feet

Bailey
Sediment Coring Locations

SWWD Lakes and Creek Management Plan

BARR



Coring Location

Bathymetric Contours

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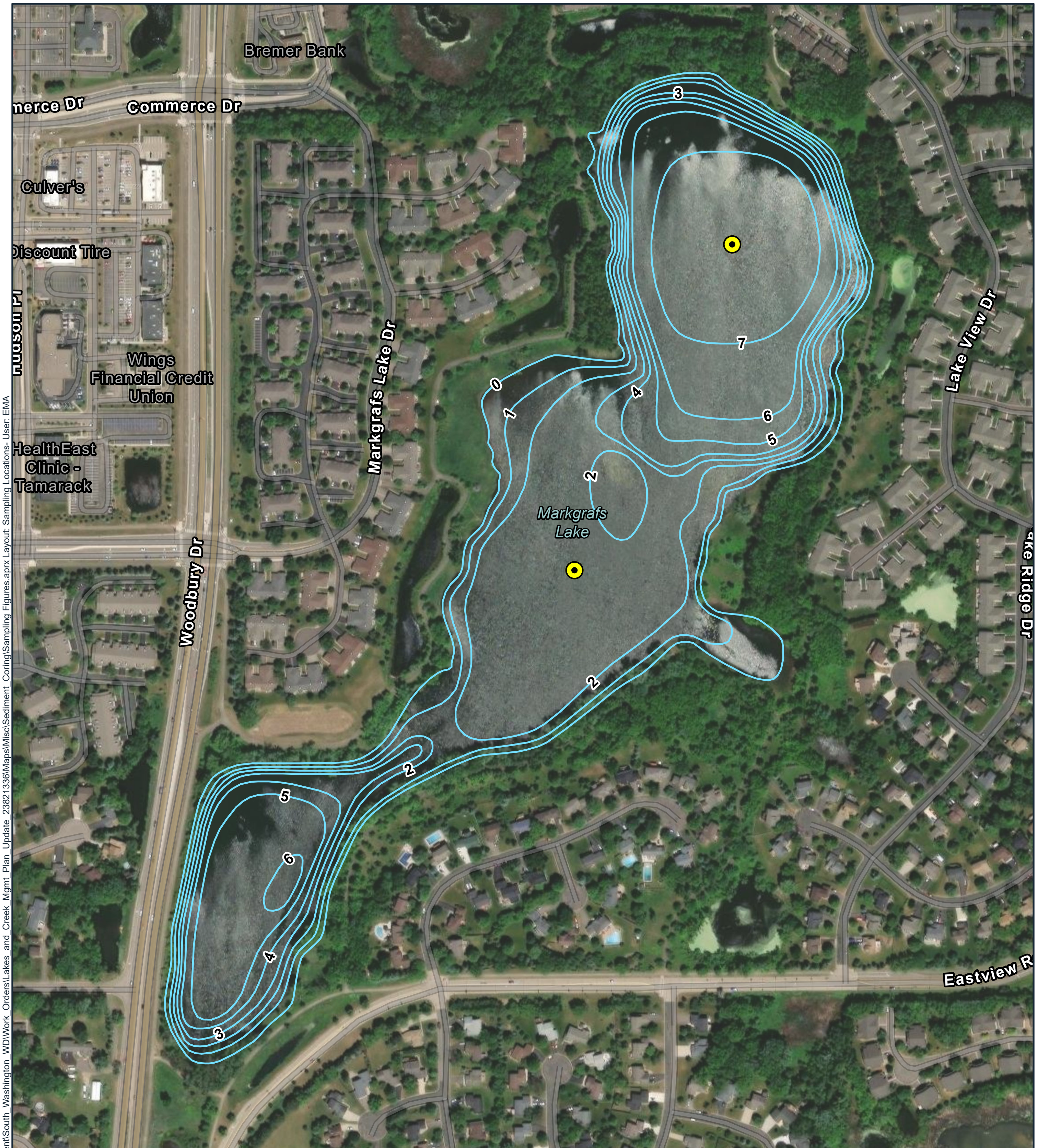
Feet

Colby
Sediment Coring Locations

SWWD Lakes and Creek Management Plan

BARR



Coring Location

Bathymetric Contours

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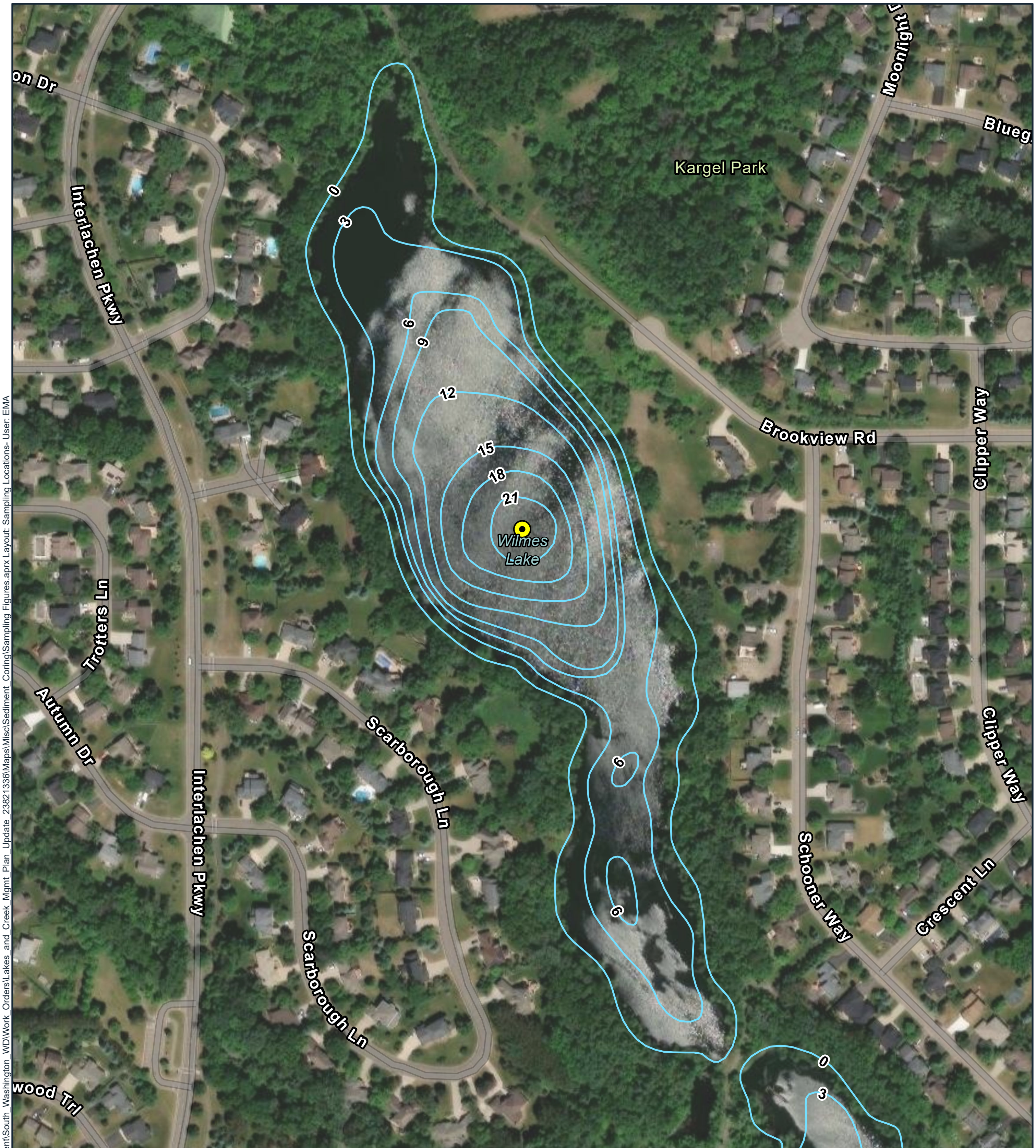
Feet

Markgrafs
Sediment Coring Locations

SWWD Lakes and Creek Management Plan

BARR



Coring Location

Bathymetric Contours

0 150 300

Feet

North Wilmes
Sediment Coring Locations

SWWD Lakes and Creek Management Plan

BARR

Coring Location

Bathymetric Contours

N

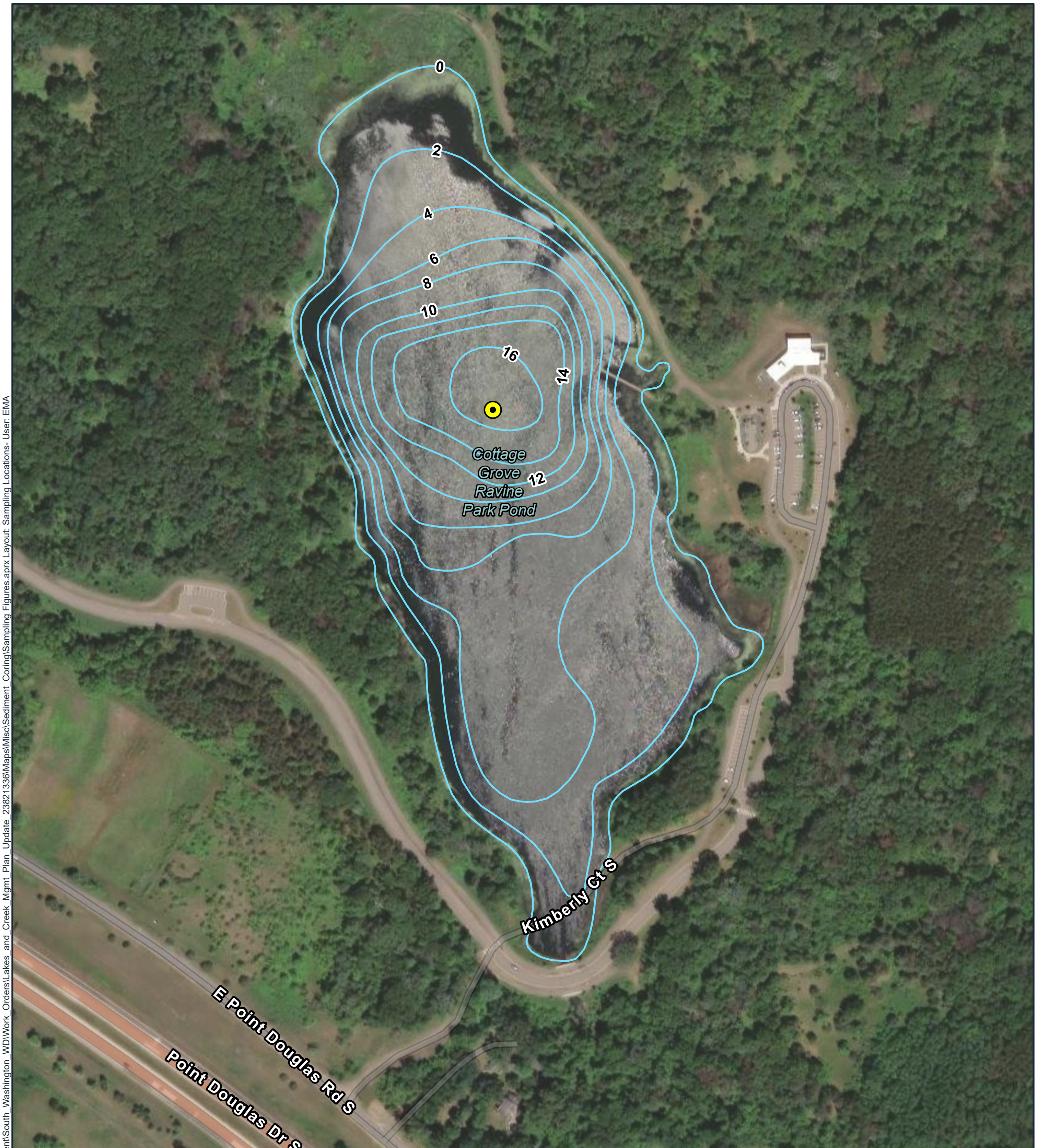
0 200 400

Feet

Powers
Sediment Coring Locations

SWWD Lakes and Creek Management Plan

BARR



Coring Location

Bathymetric Contours

N

0 150 300

Feet

Ravine
Sediment Coring Locations

SWWD Lakes and Creek Management Plan

BARR

Coring Location

Bathymetric Contours

0 150 300

Feet

South Wilmes
Sediment Coring Locations

SWWD Lakes and Creek Management Plan

BARR

Appendix B

In-lake Model Calibration Plots

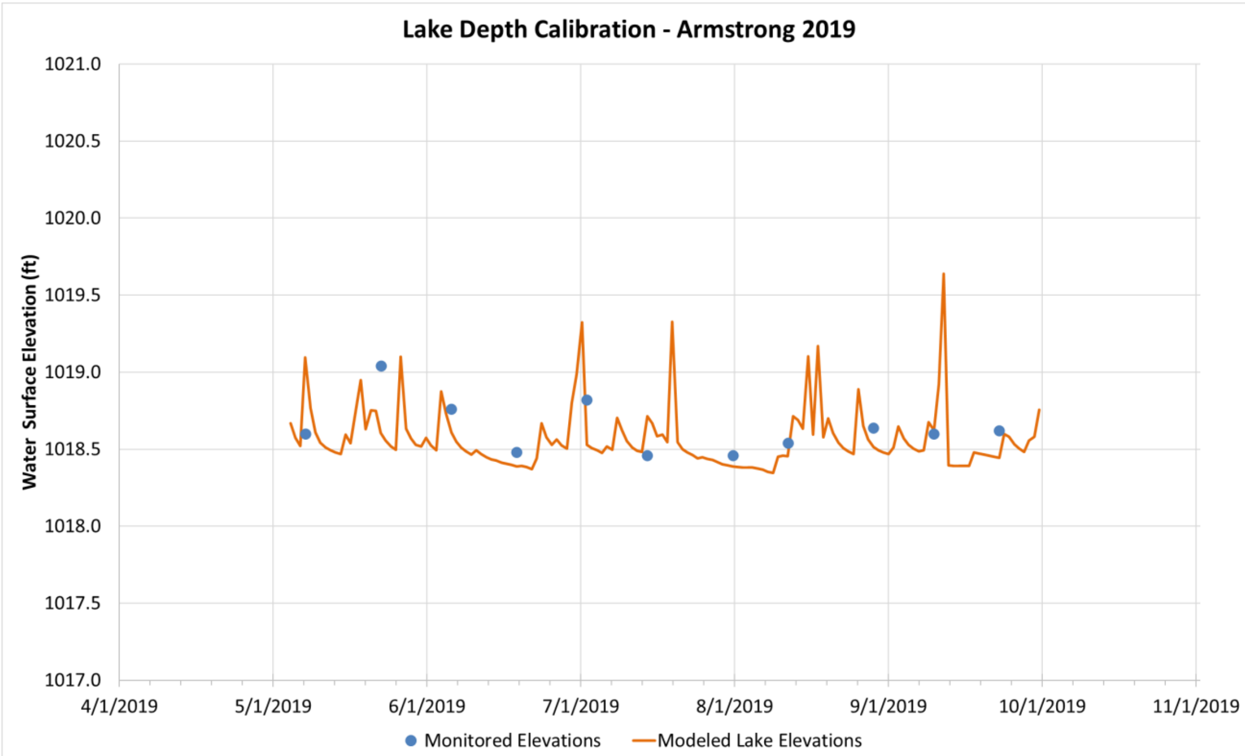


Figure B 1 Armstrong Lake water balance calibration – 2019

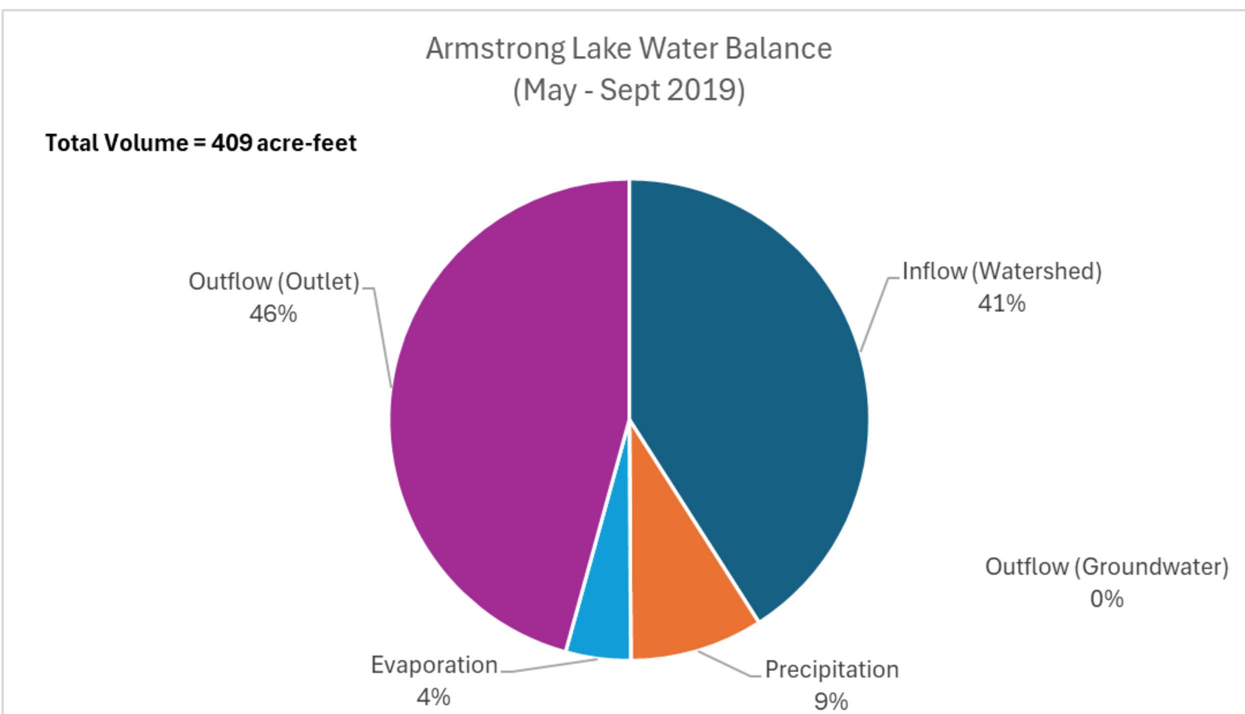


Figure B 2 Armstrong Lake water balance pie chart - 2019

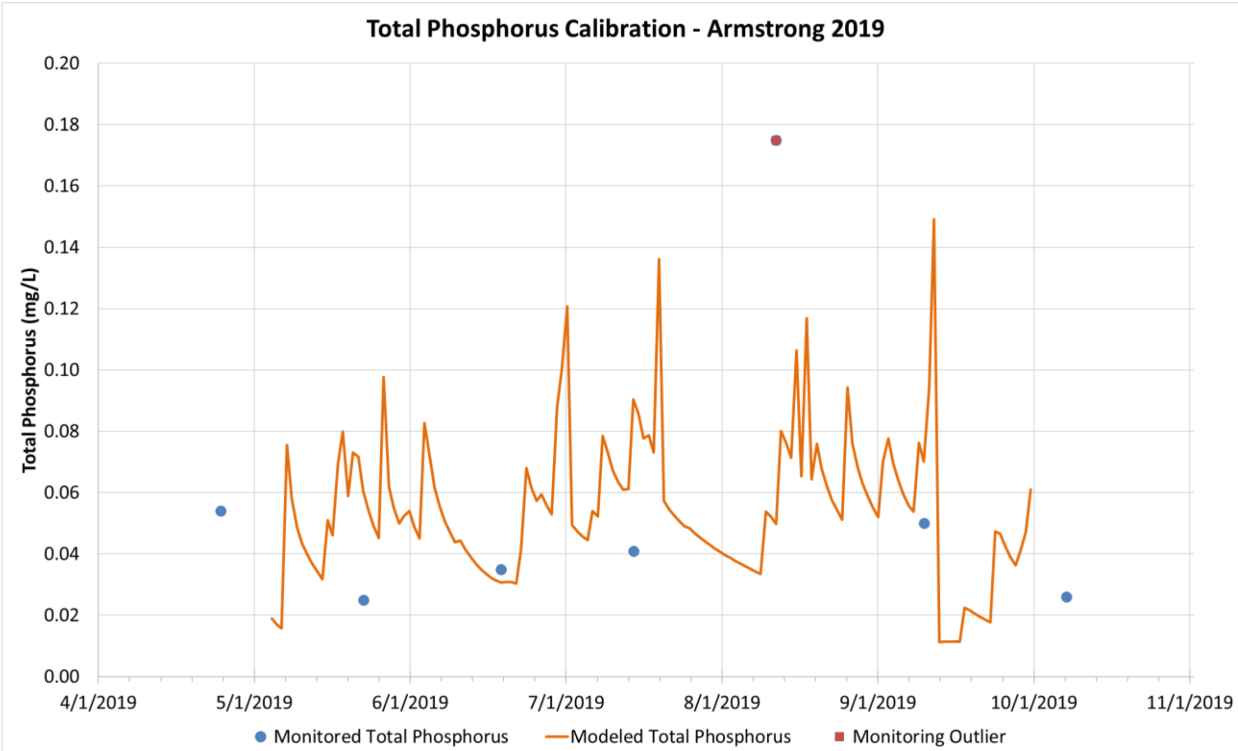


Figure B 3 Armstrong Lake total phosphorus calibration - 2019

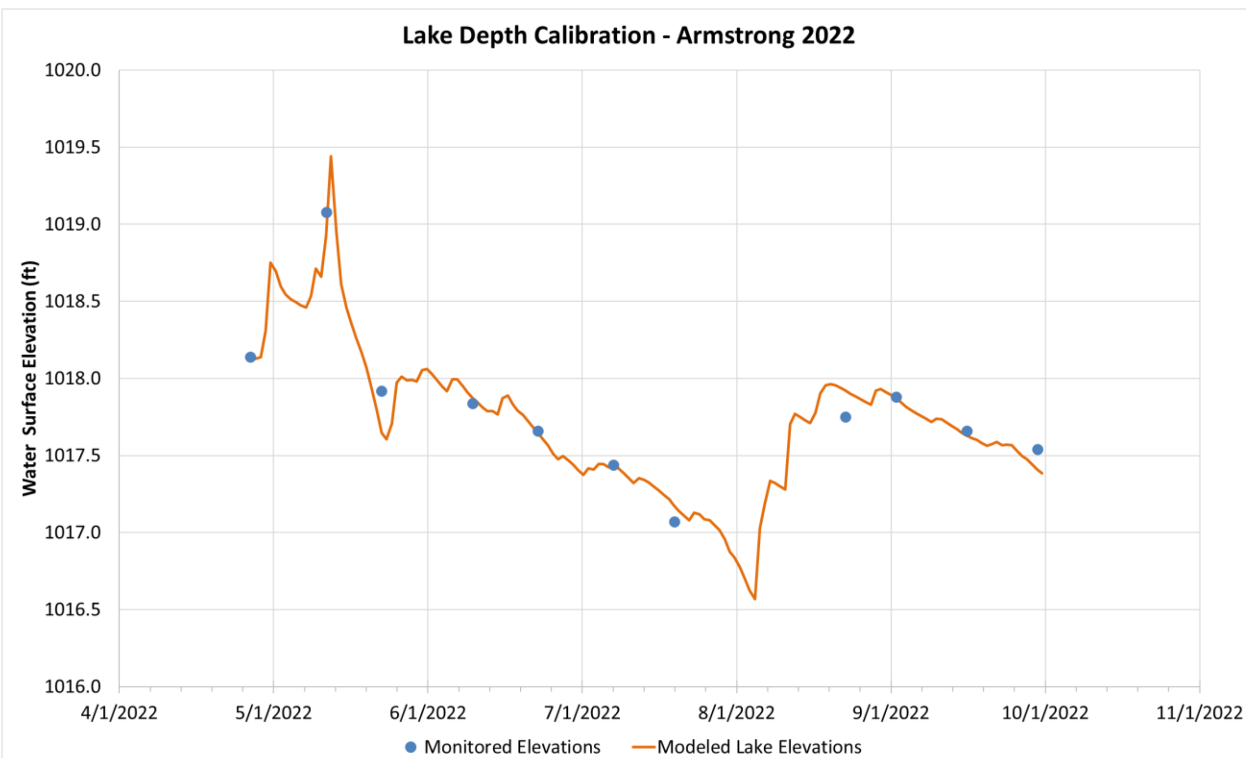
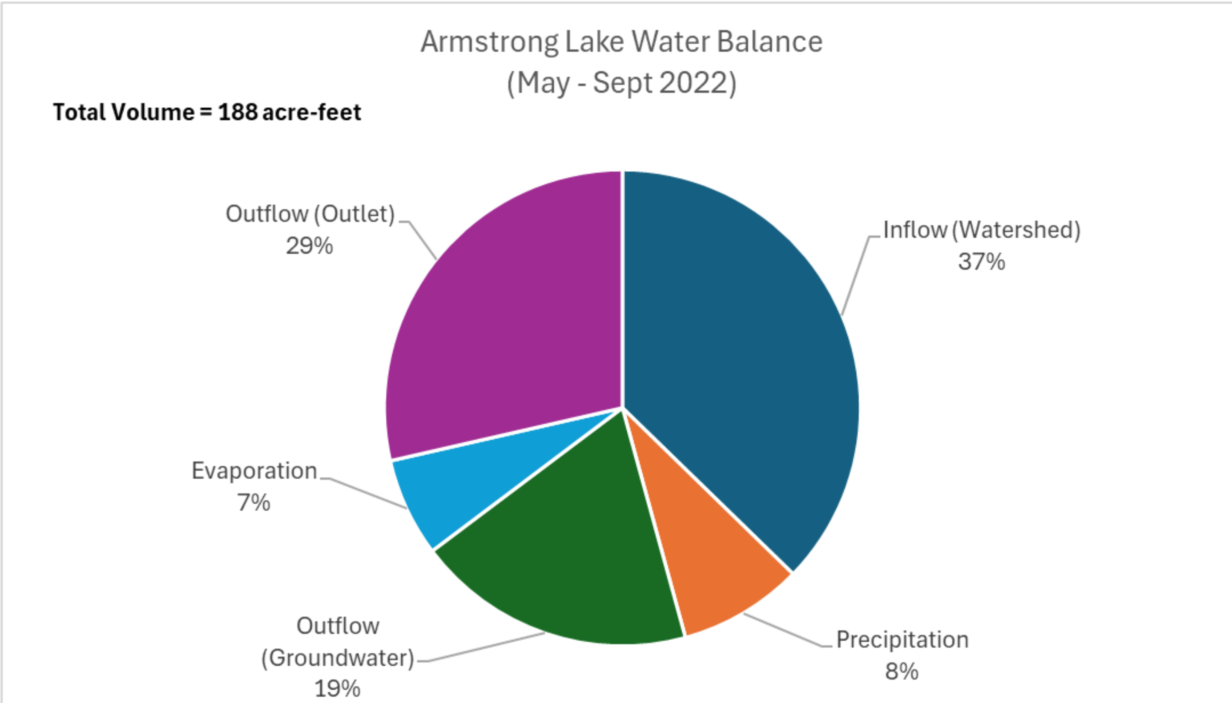


Figure B 4 Armstrong Lake water balance calibration – 2022



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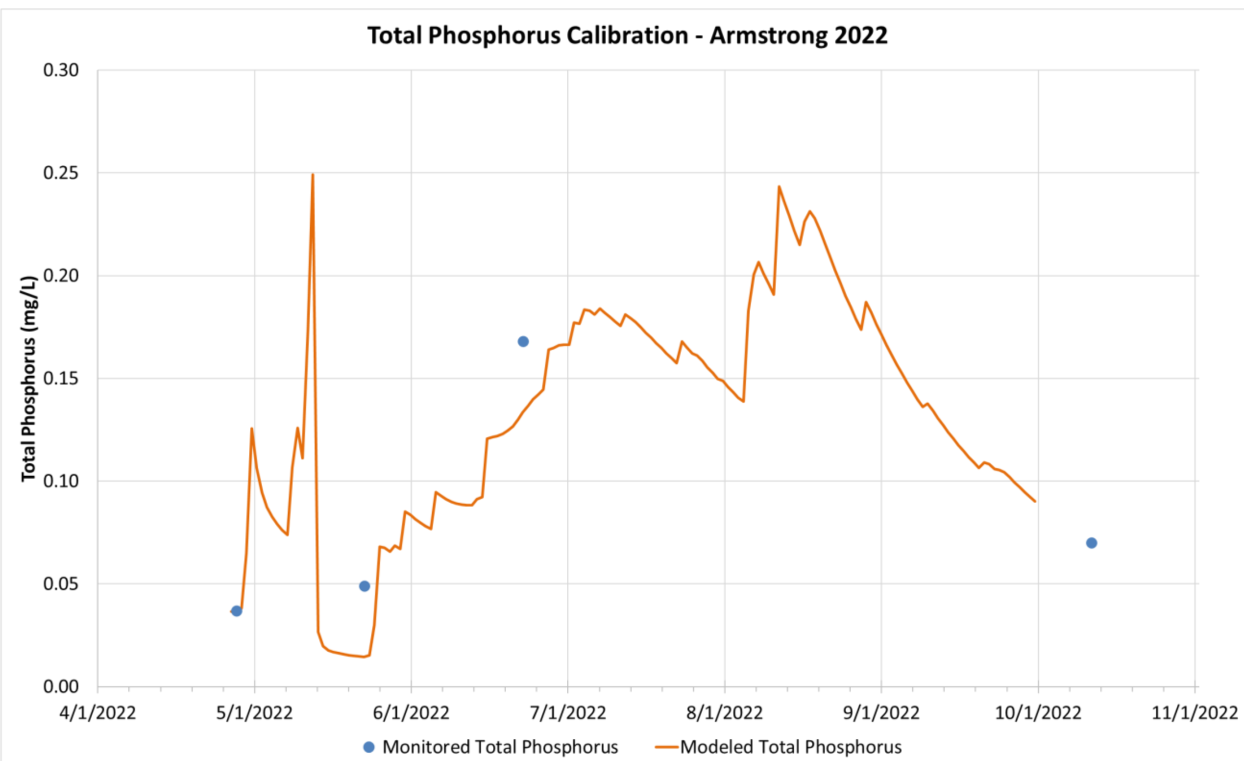


Figure B 6 Armstrong Lake total phosphorus calibration – 2022

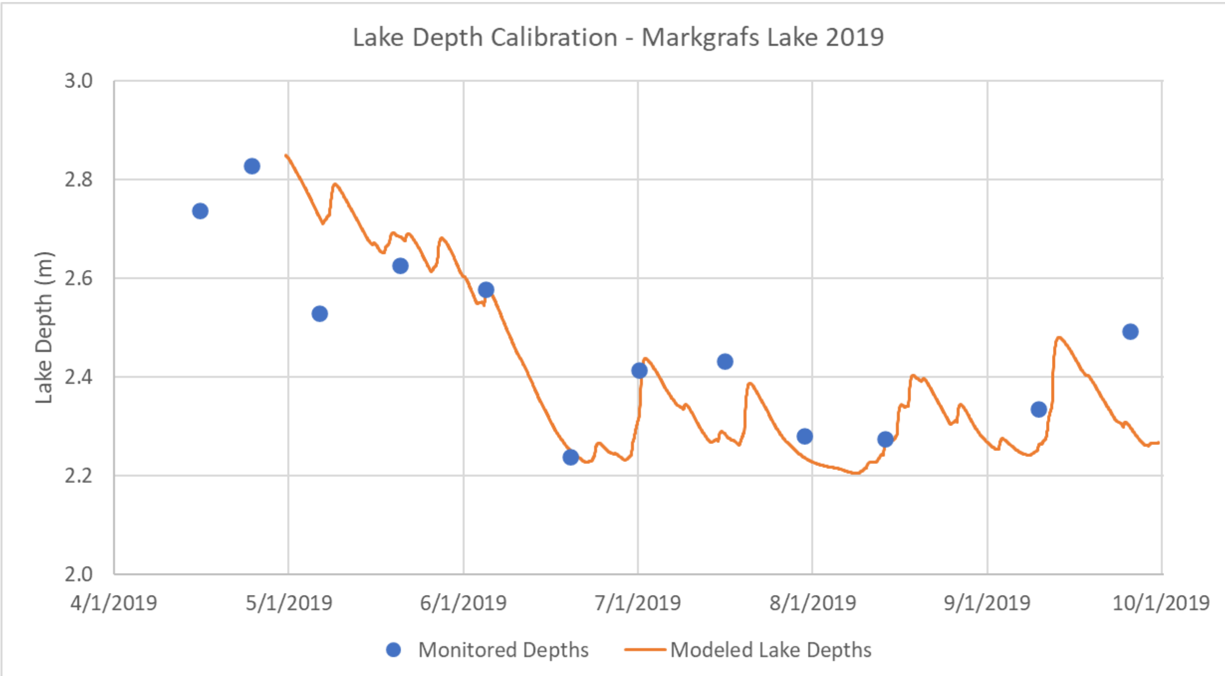
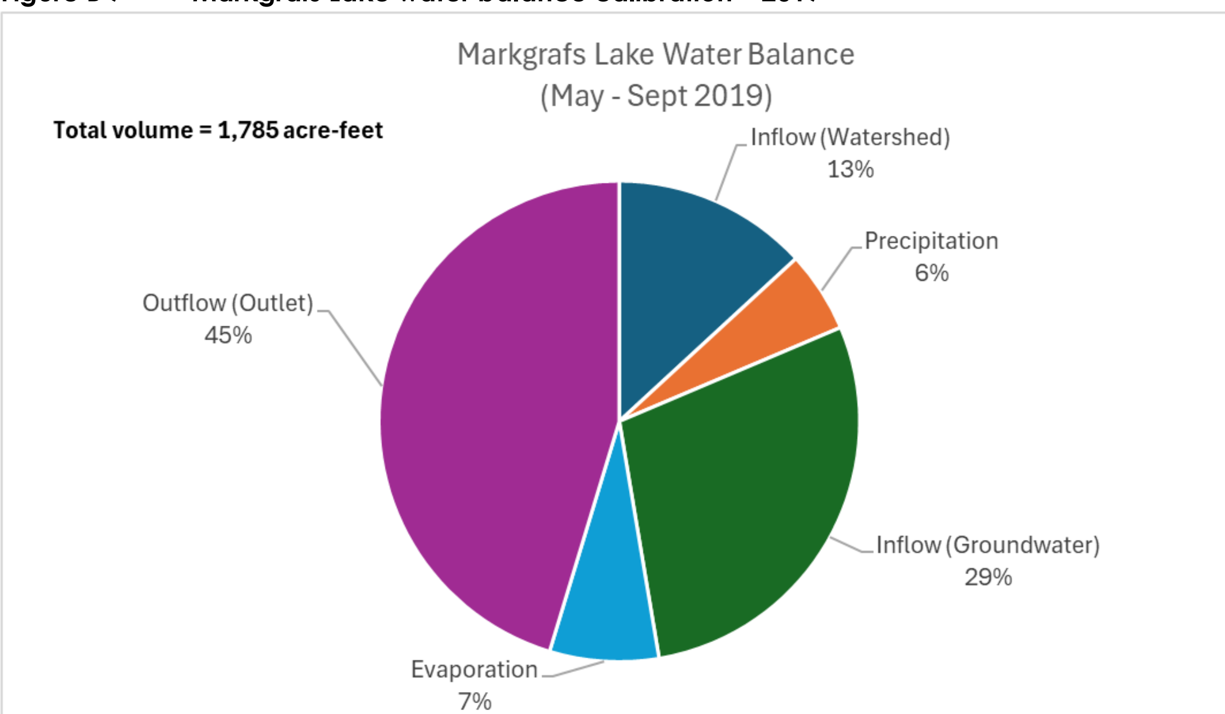


Figure B 7 Markgrafs Lake water balance calibration – 2019



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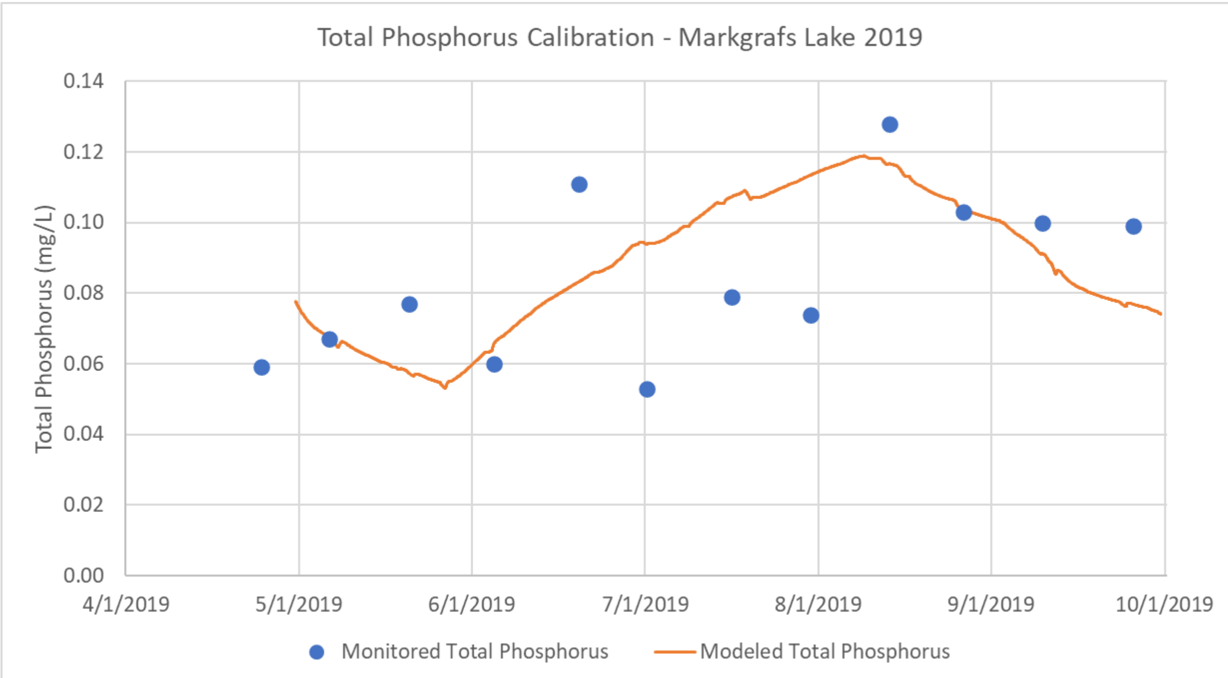

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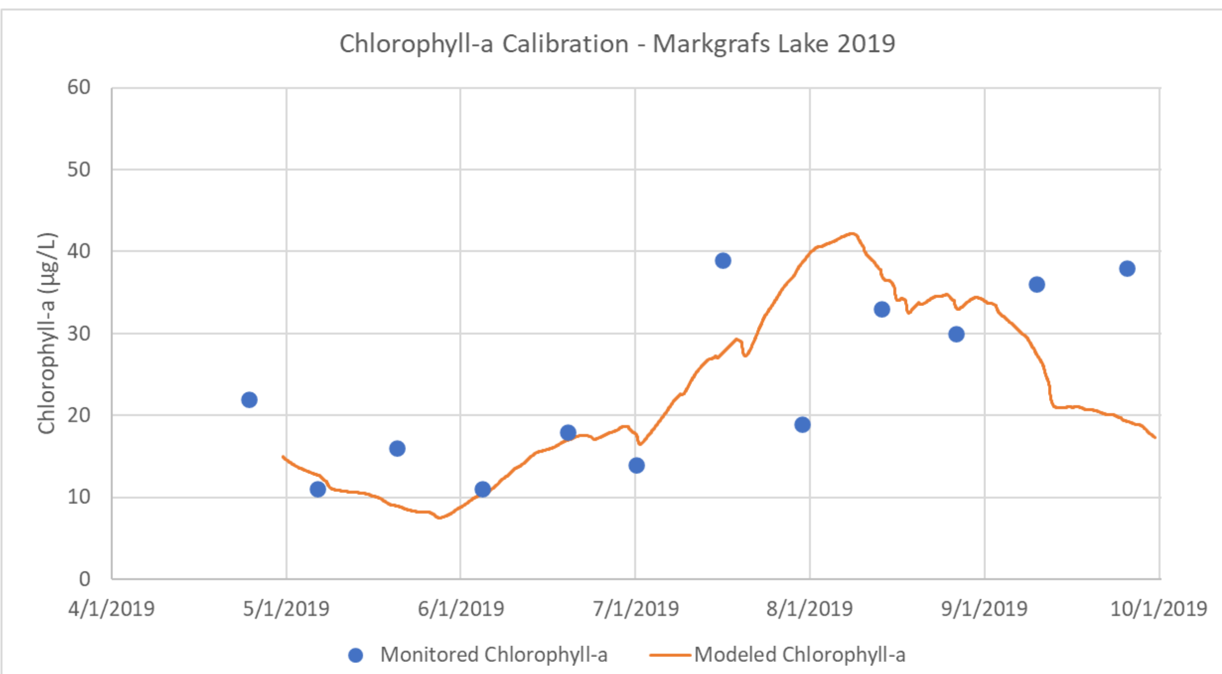


Figure B 10 Markgrafs Lake chlorophyll-a calibration – 2019

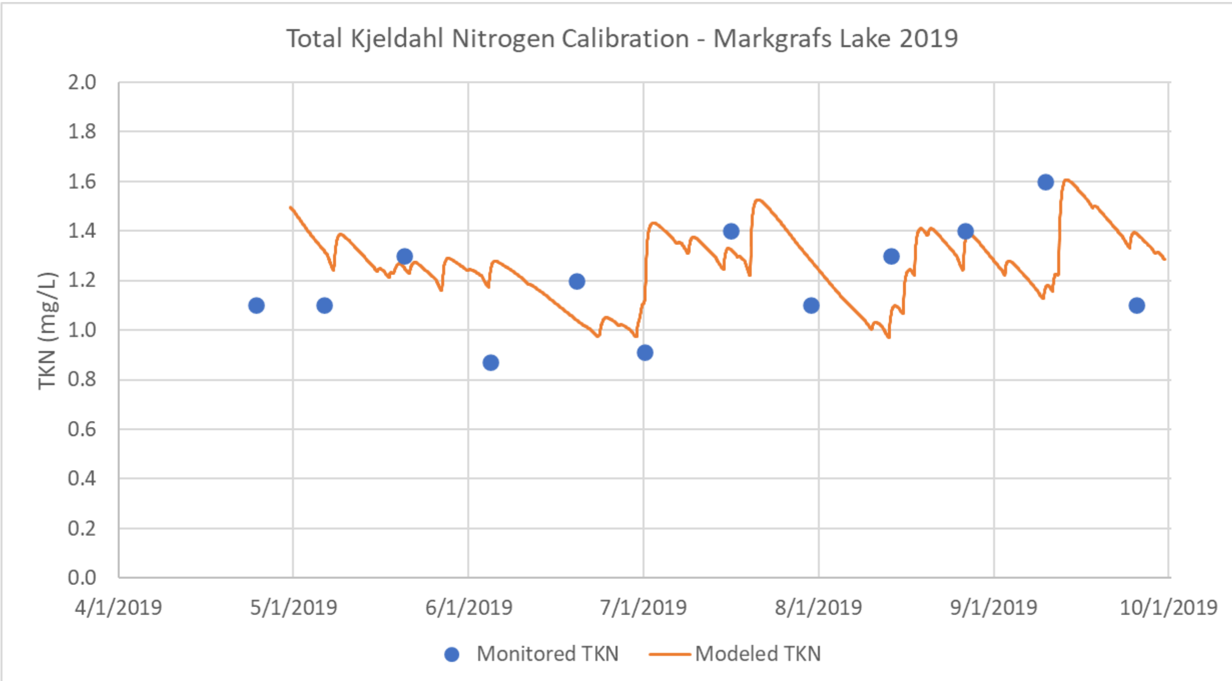


Figure B 11 Markgrafs Lake total Kjeldahl nitrogen calibration – 2019

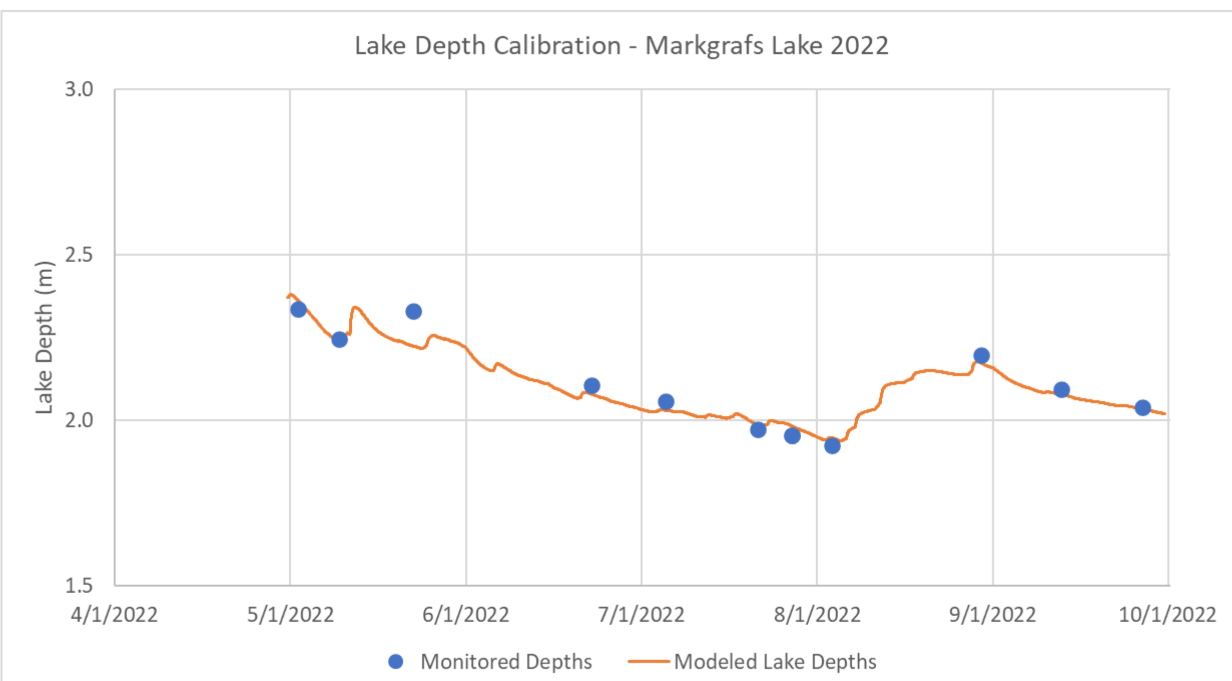


Figure B 12 Markgrafs Lake water balance calibration – 2022

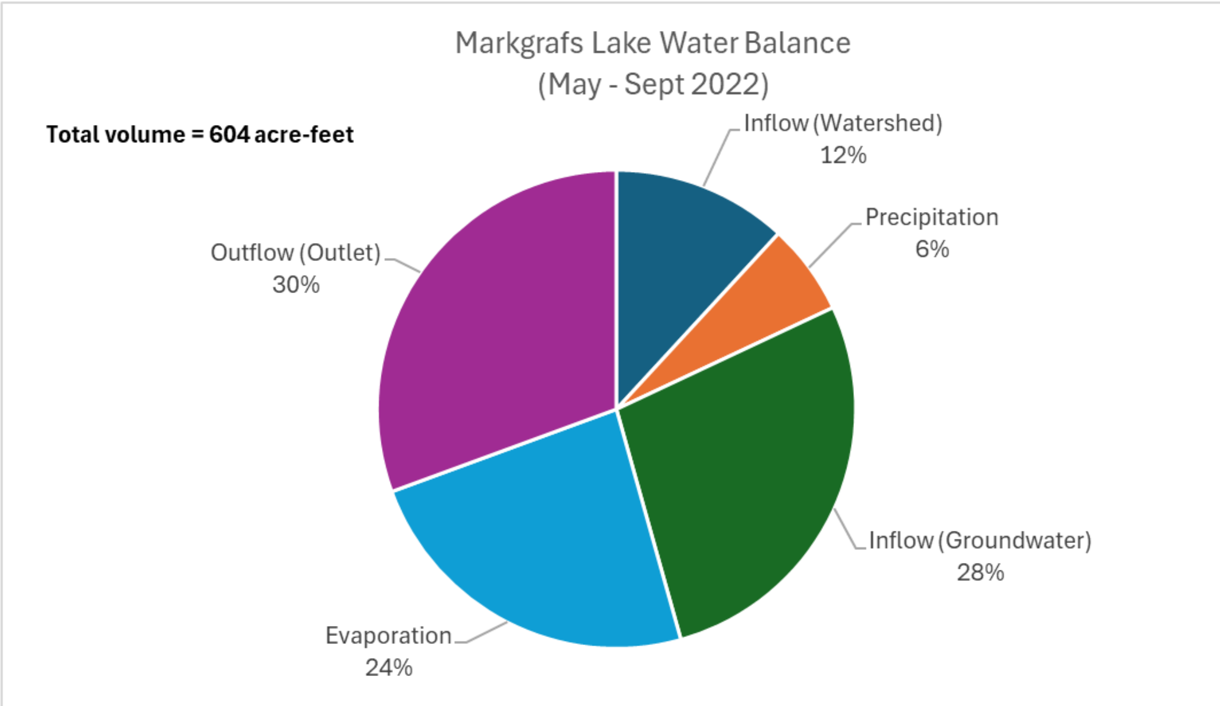


Figure B 13 Markgrafs Lake water balance pie chart - 2022

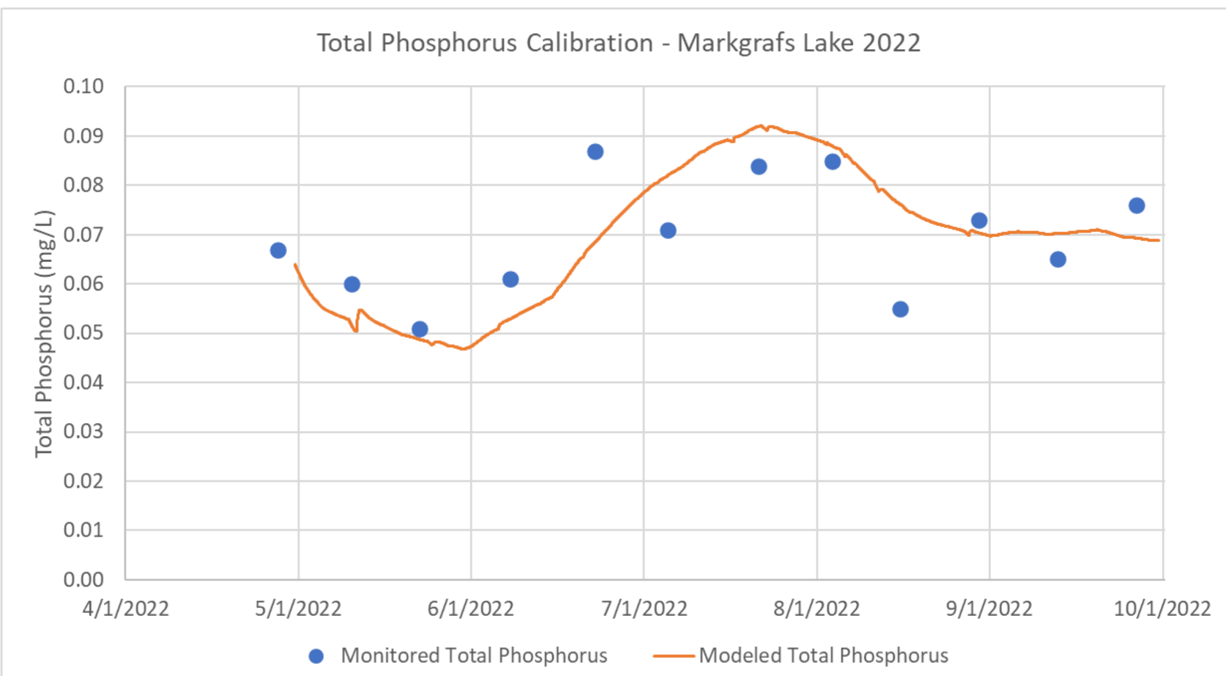


Figure B 14 Markgrafs Lake total phosphorus calibration – 2022

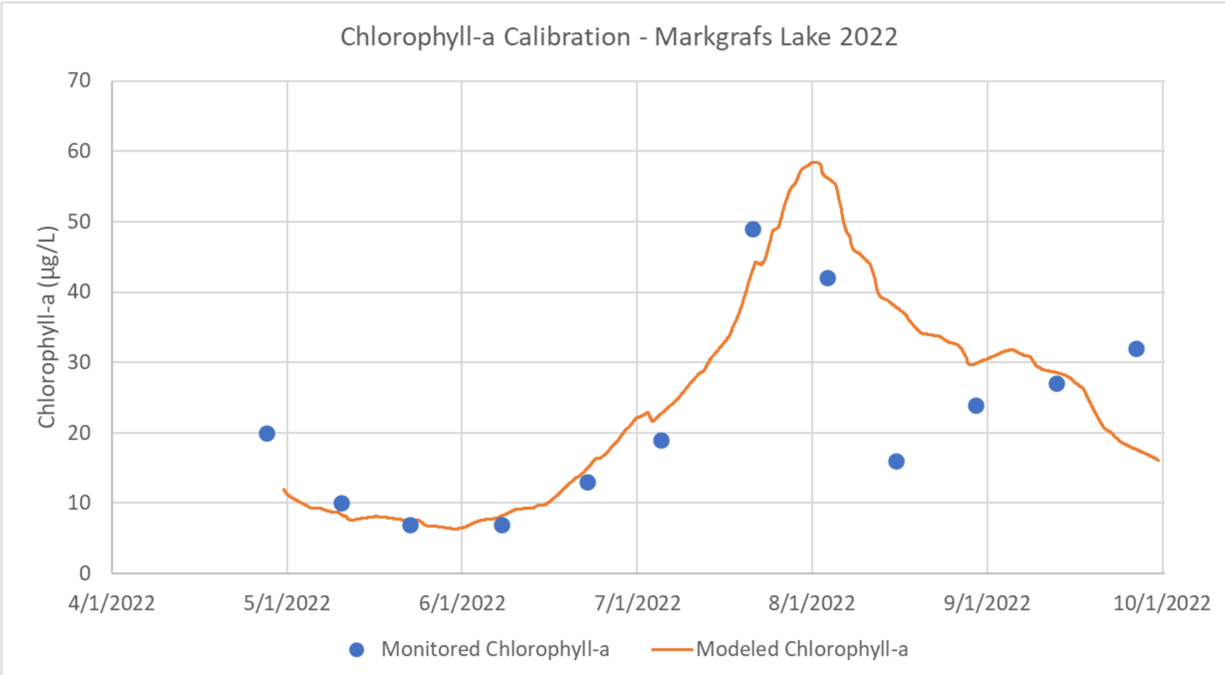


Figure B 15 Markgrafs Lake chlorophyll-a calibration – 2022

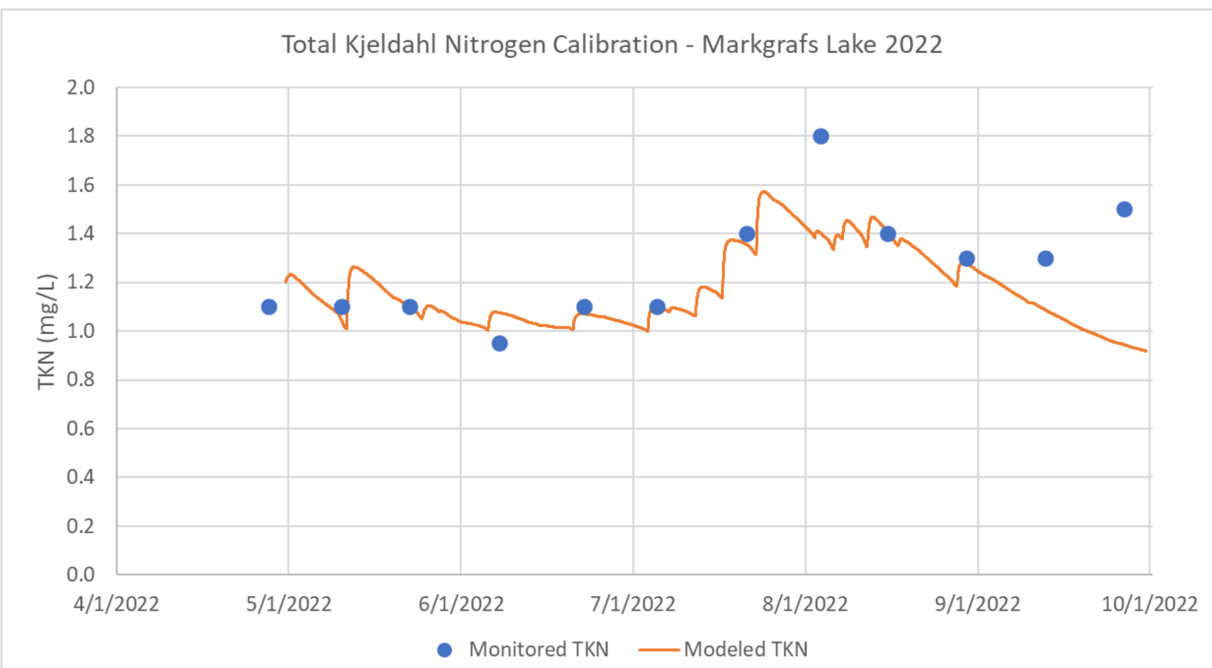


Figure B 16 Markgrafs Lake total Kjeldahl nitrogen calibration – 2022

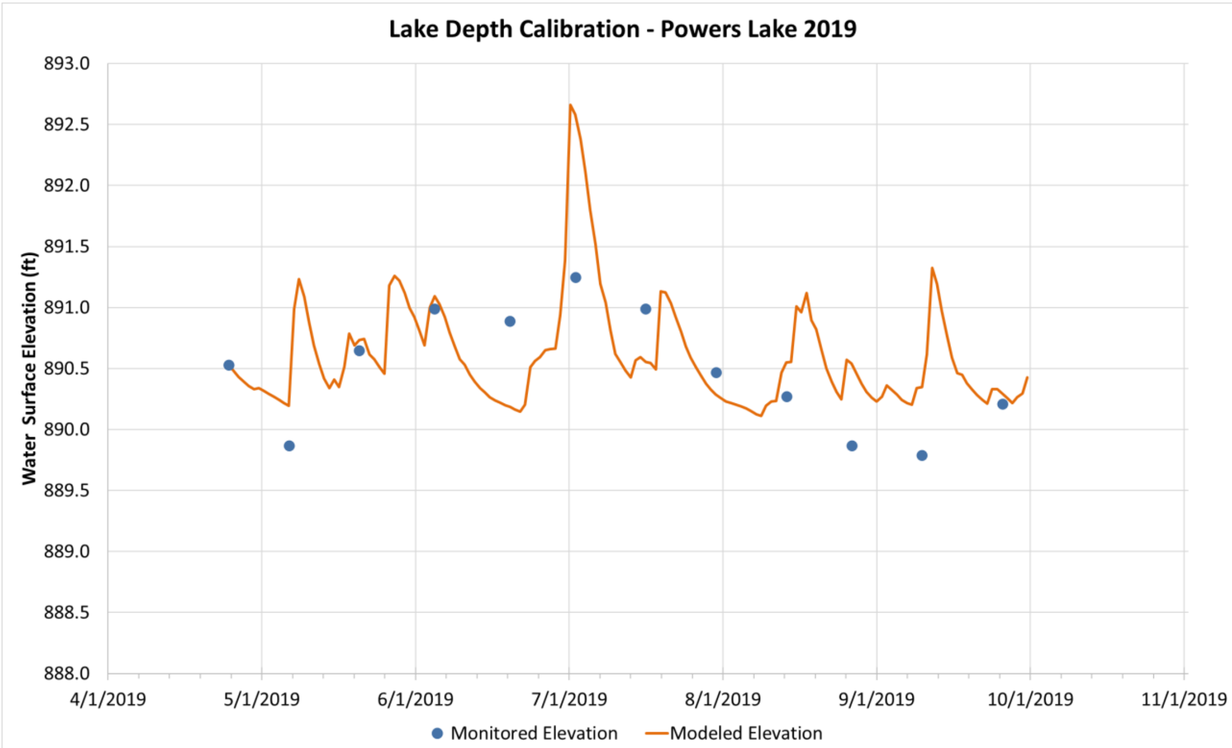
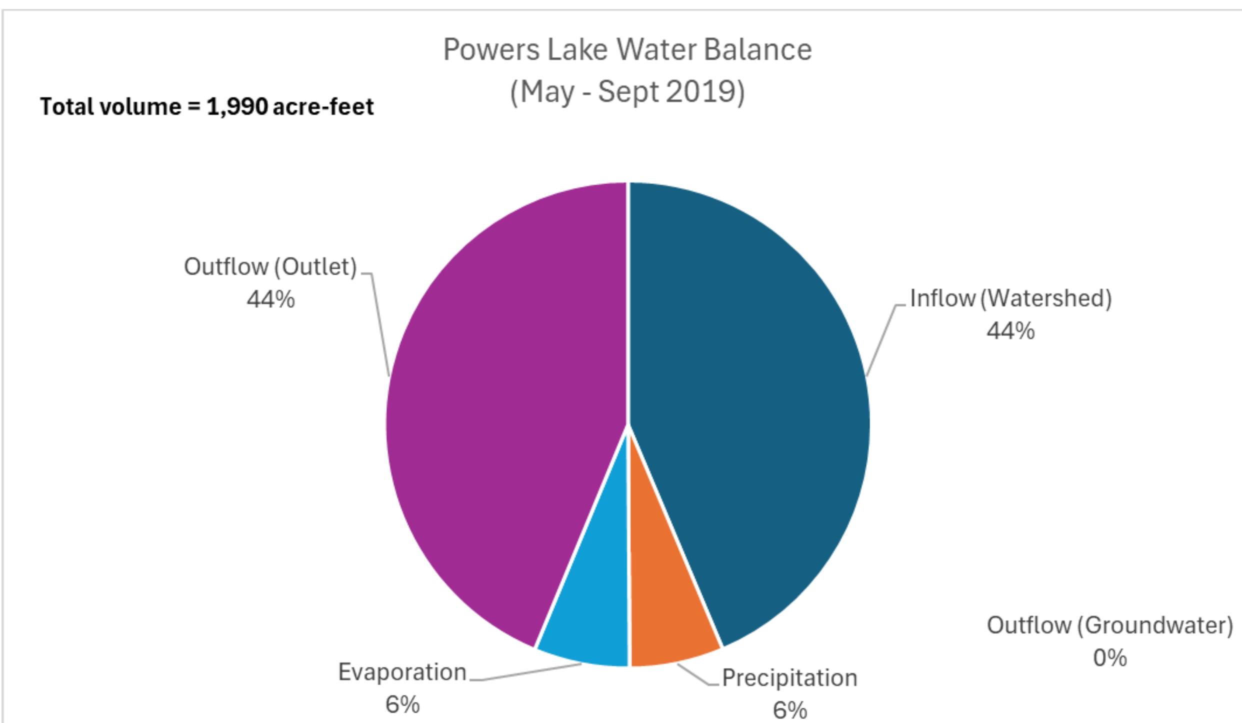


Figure B 17 Powers Lake water balance calibration – 2019



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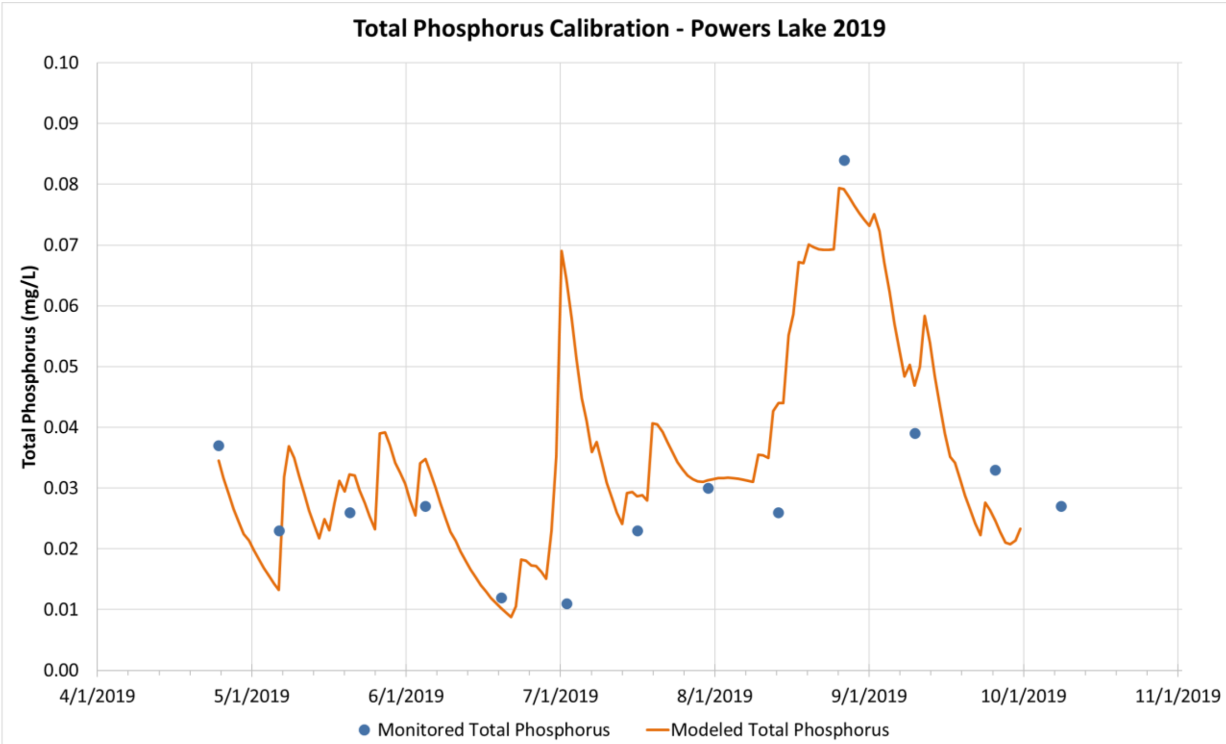

Figure B 19 Powers Lake total phosphorus calibration - 2019

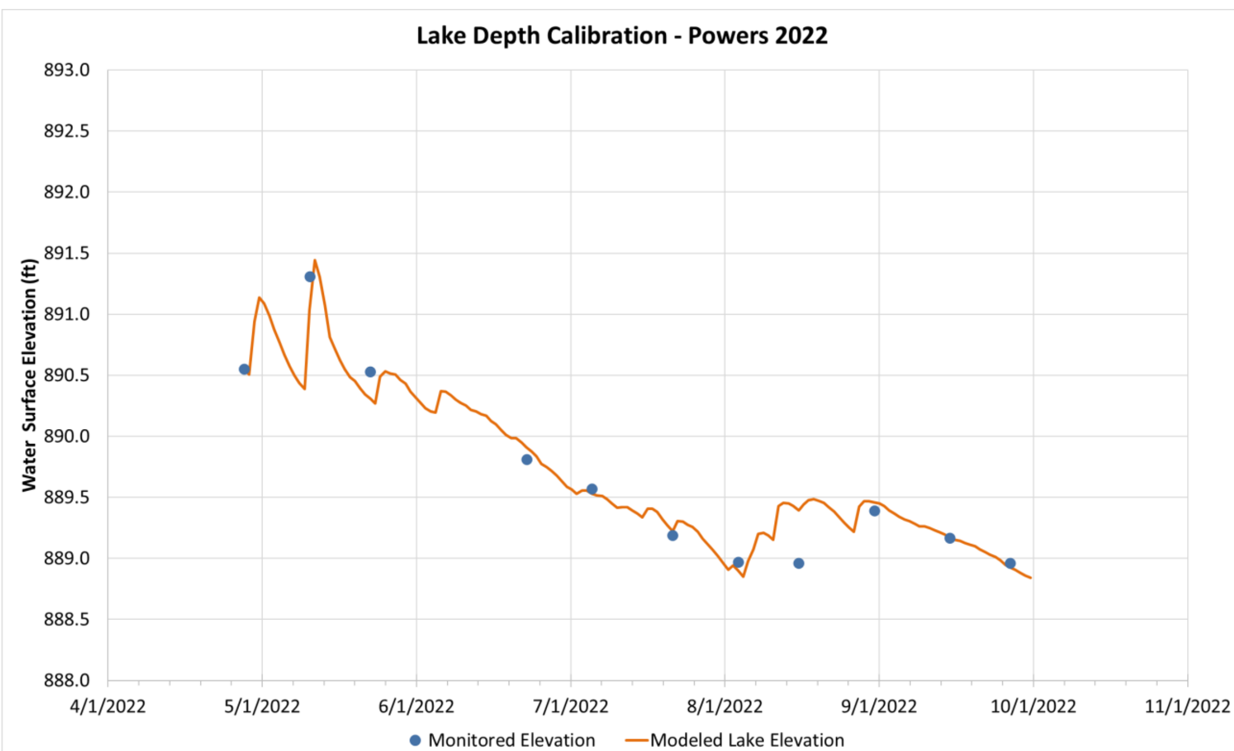

Figure B 20 Powers Lake water balance calibration – 2022

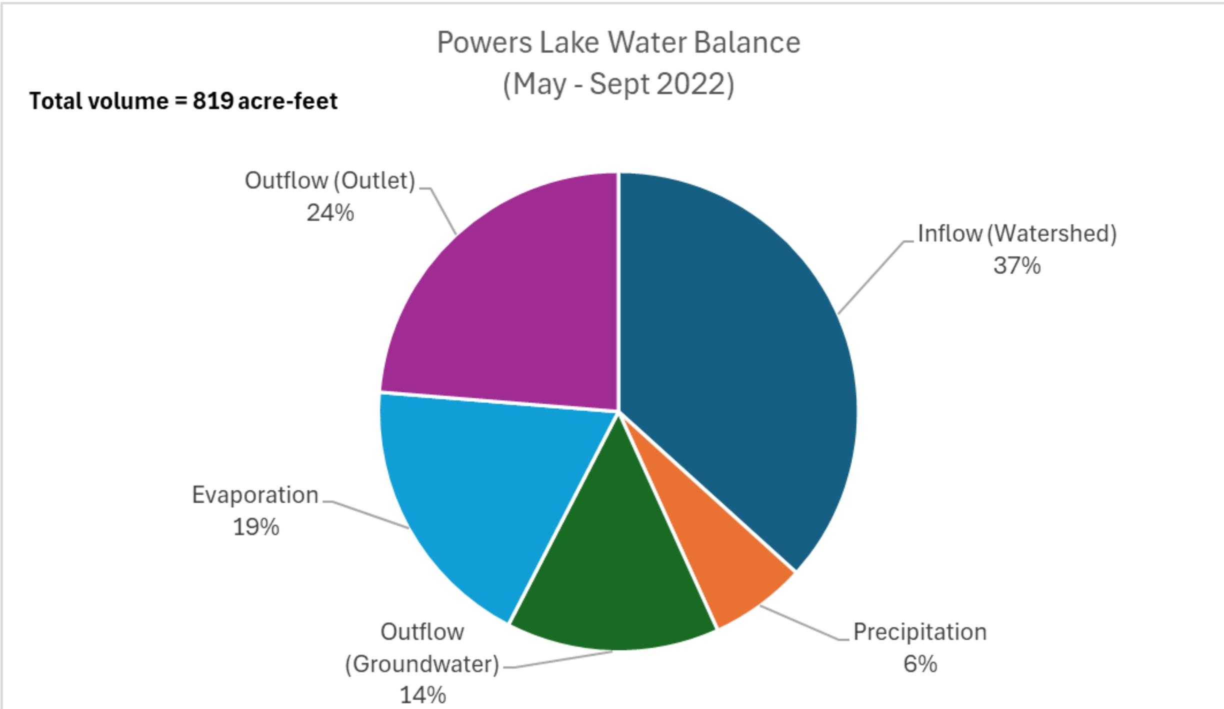

Figure B 21 Power Lake water balance pie chart - 2022

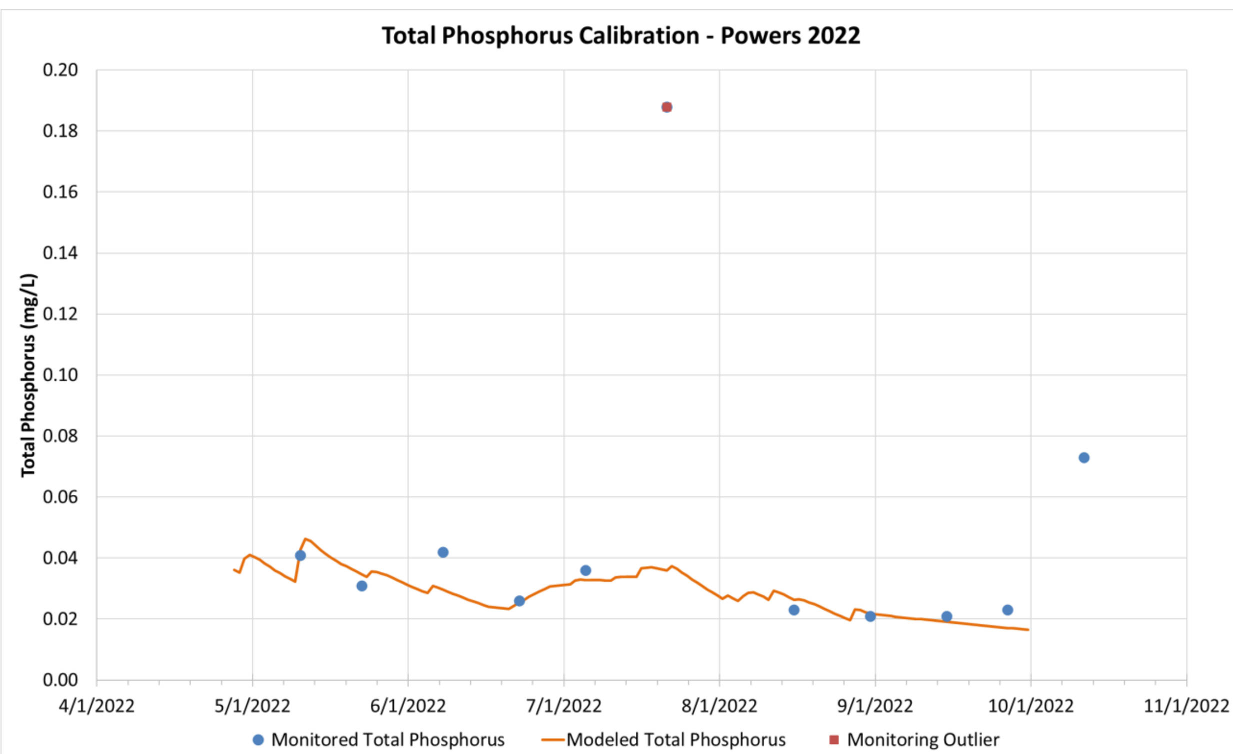

Figure B 22 Powers Lake total phosphorus calibration – 2022

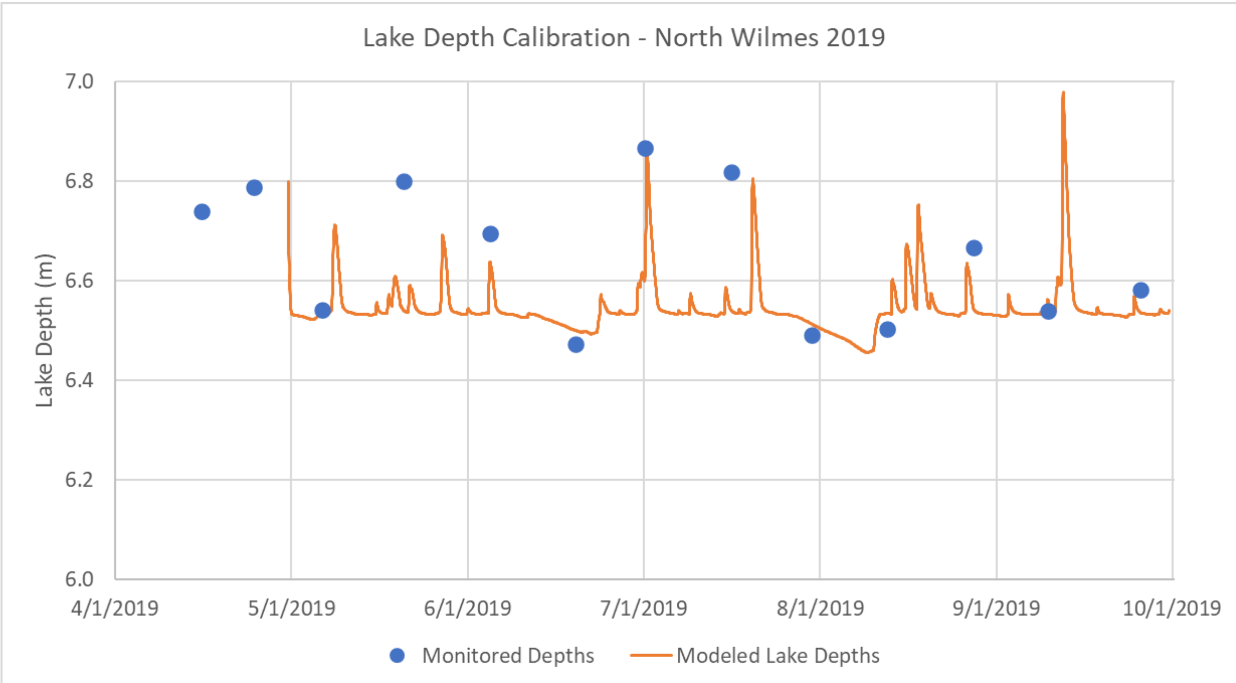

Figure B 23 North Wilmes Lake water balance calibration – 2019

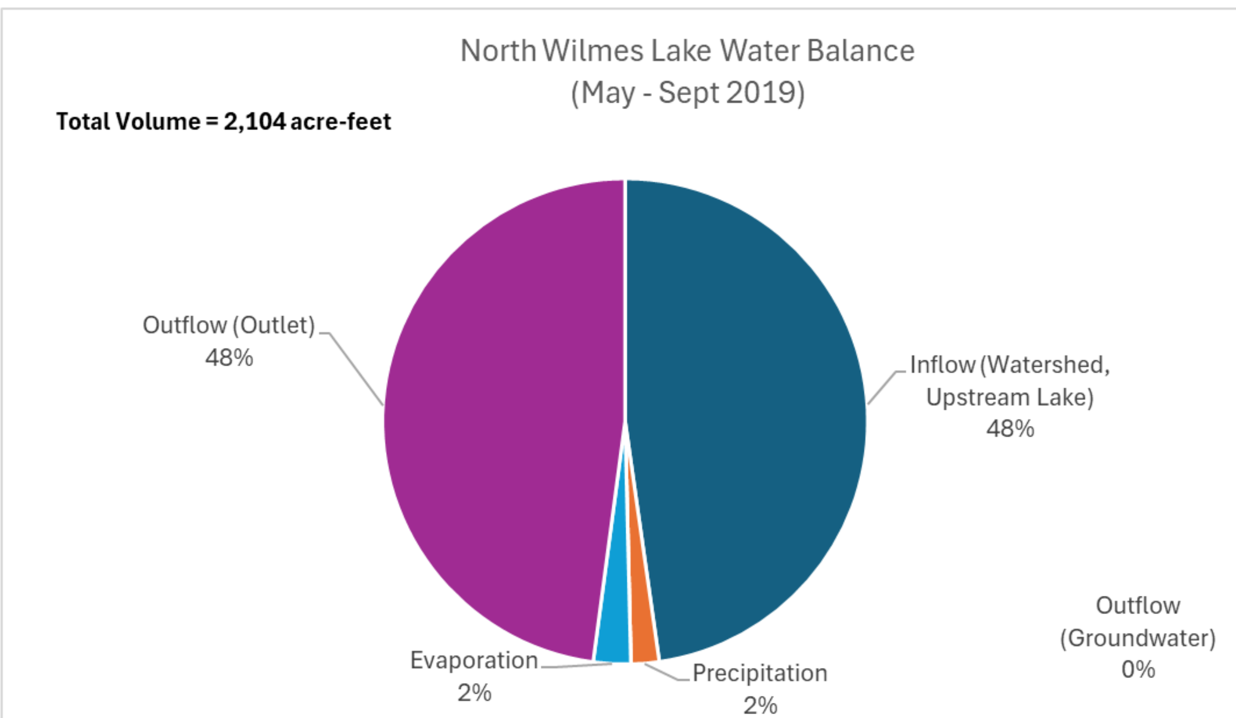

Figure B 24 North Wilmes Lake water balance pie chart - 2019

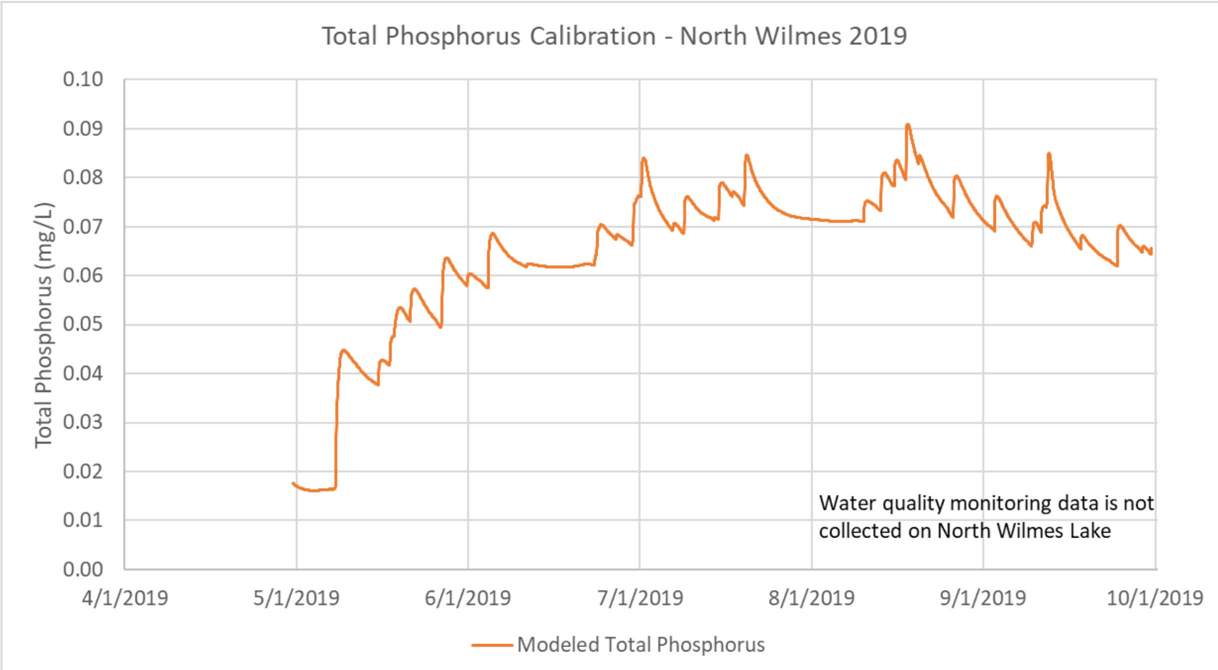

Figure B 25 North Wilmes Lake modeled total phosphorus – 2019

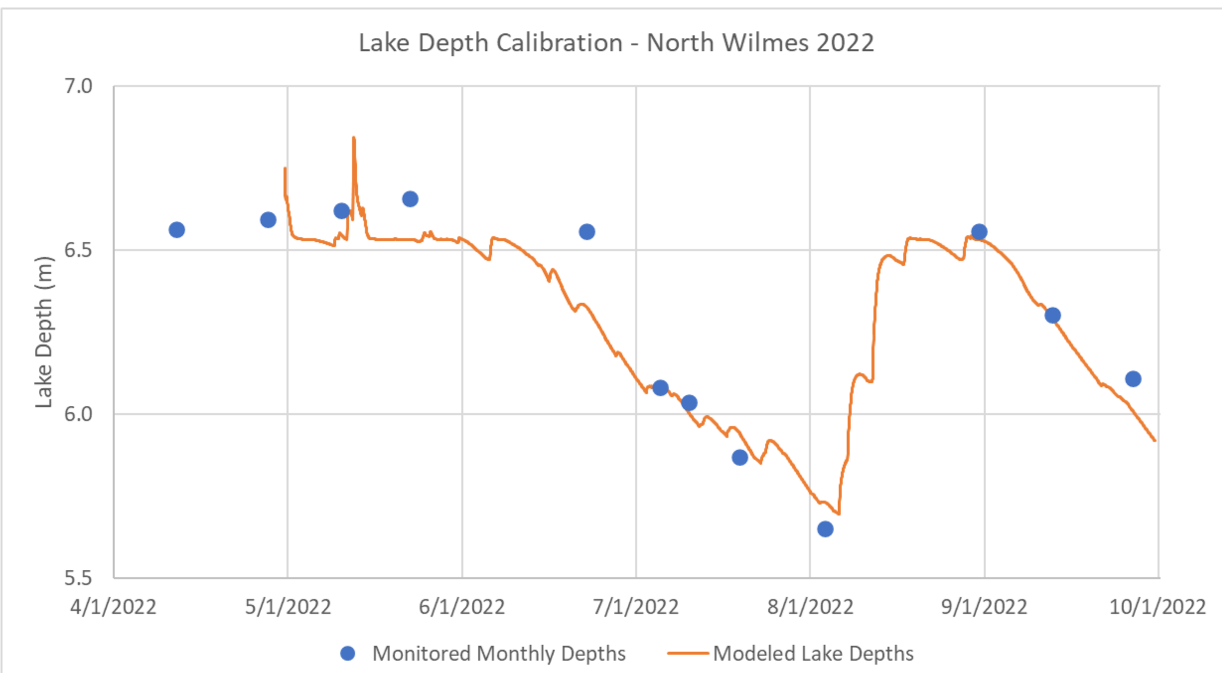

Figure B 26 North Wilmes Lake water balance calibration – 2022

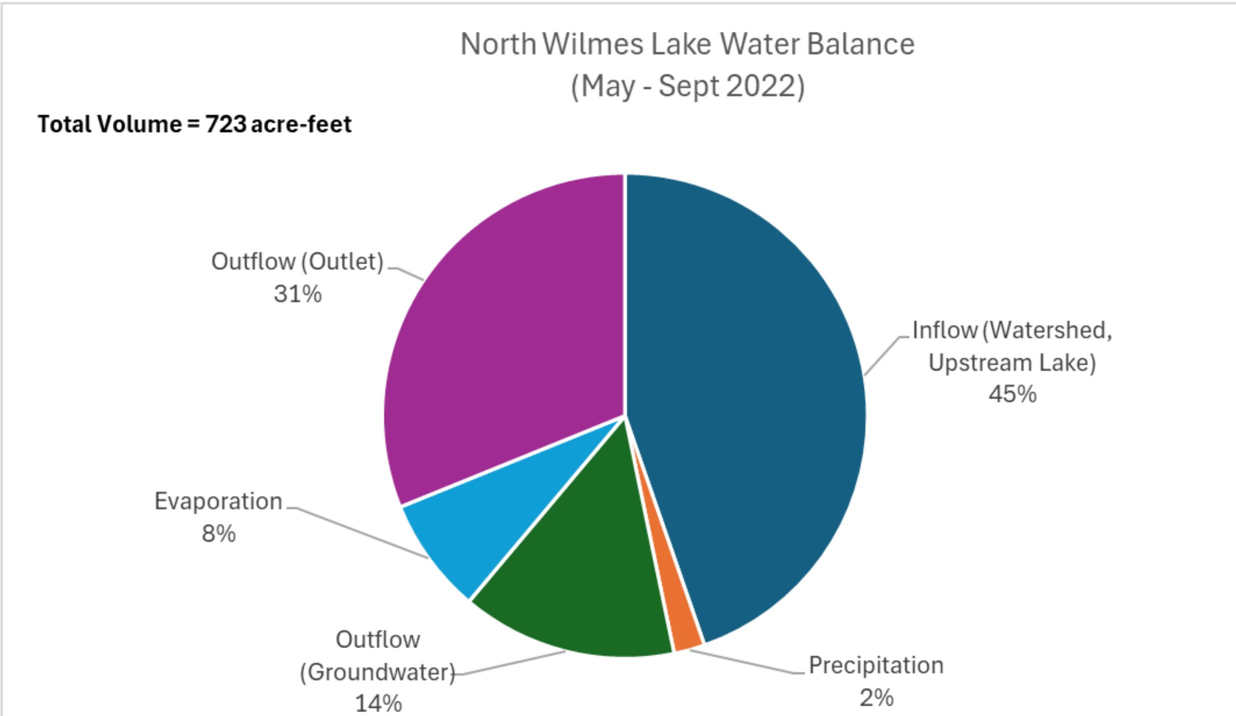

Figure B 27 North Wilmes Lake water balance pie chart - 2022

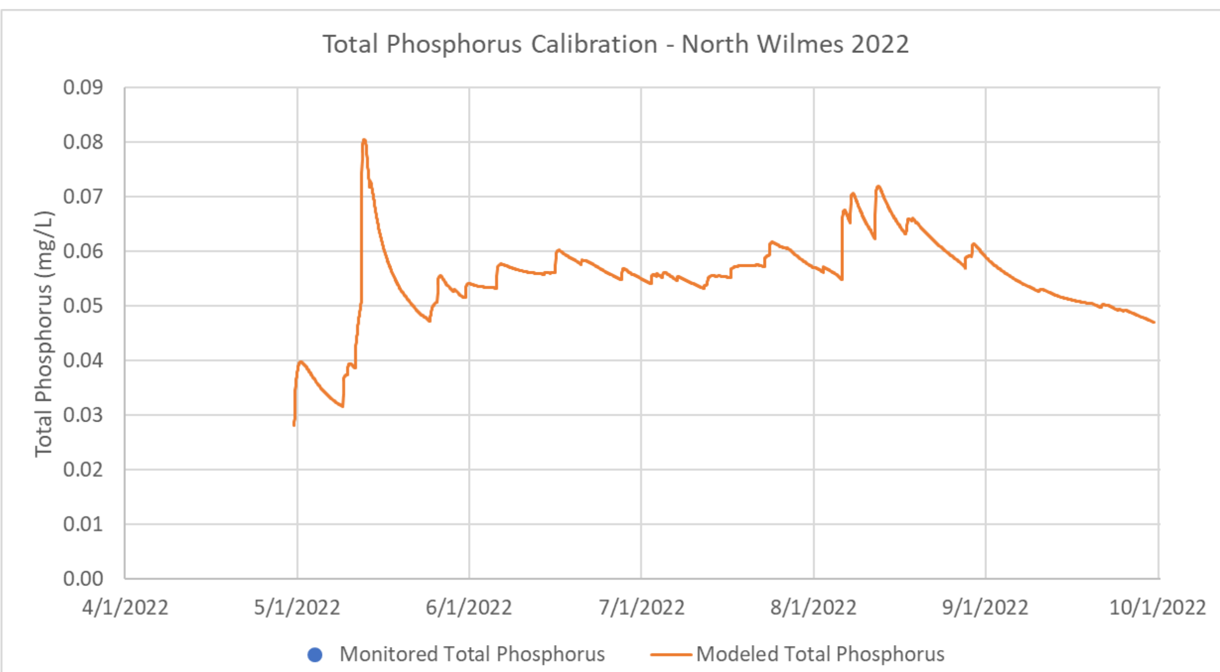

Figure B 28 North Wilmes Lake modeled total phosphorus – 2022

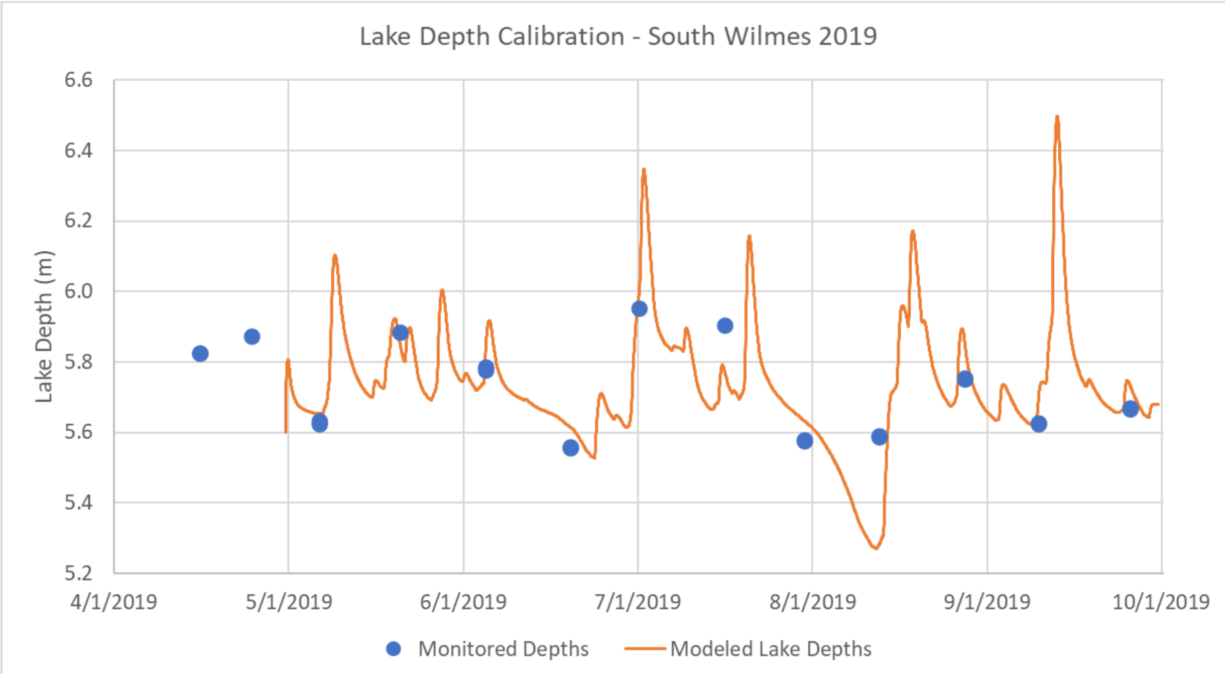

Figure B 29 South Wilmes Lake water balance calibration – 2019

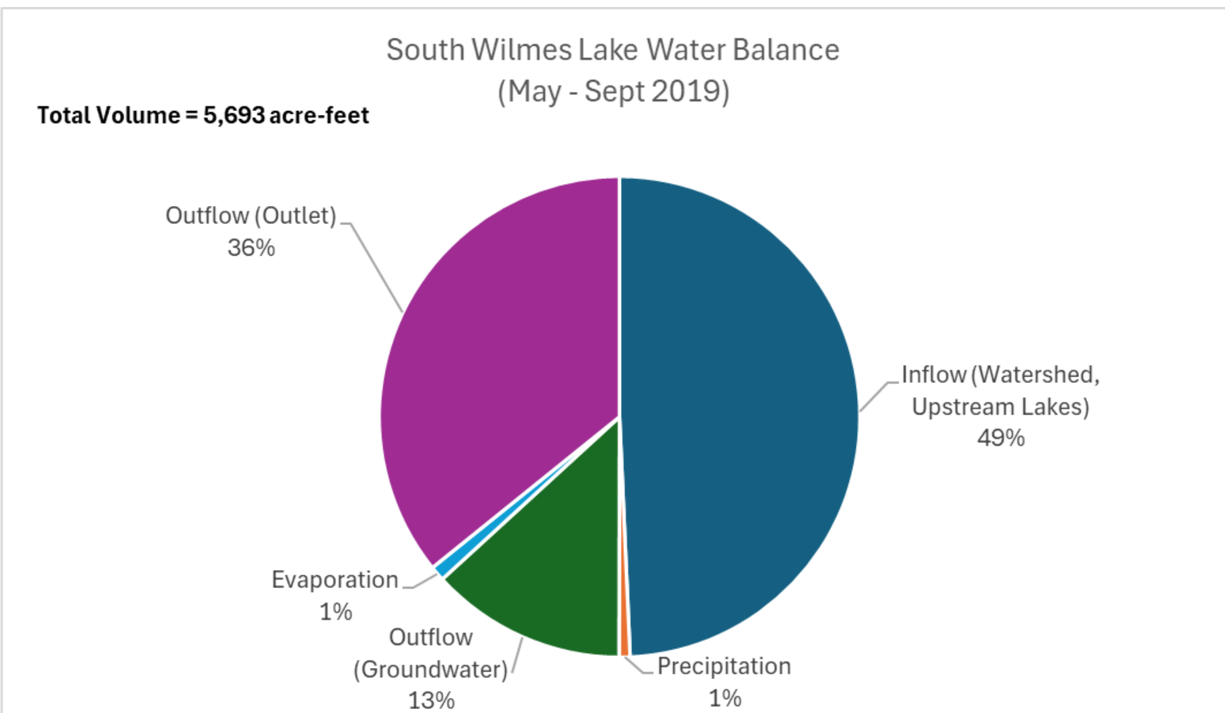

Figure B 30 South Wilmes Lake water balance pie chart – 2019

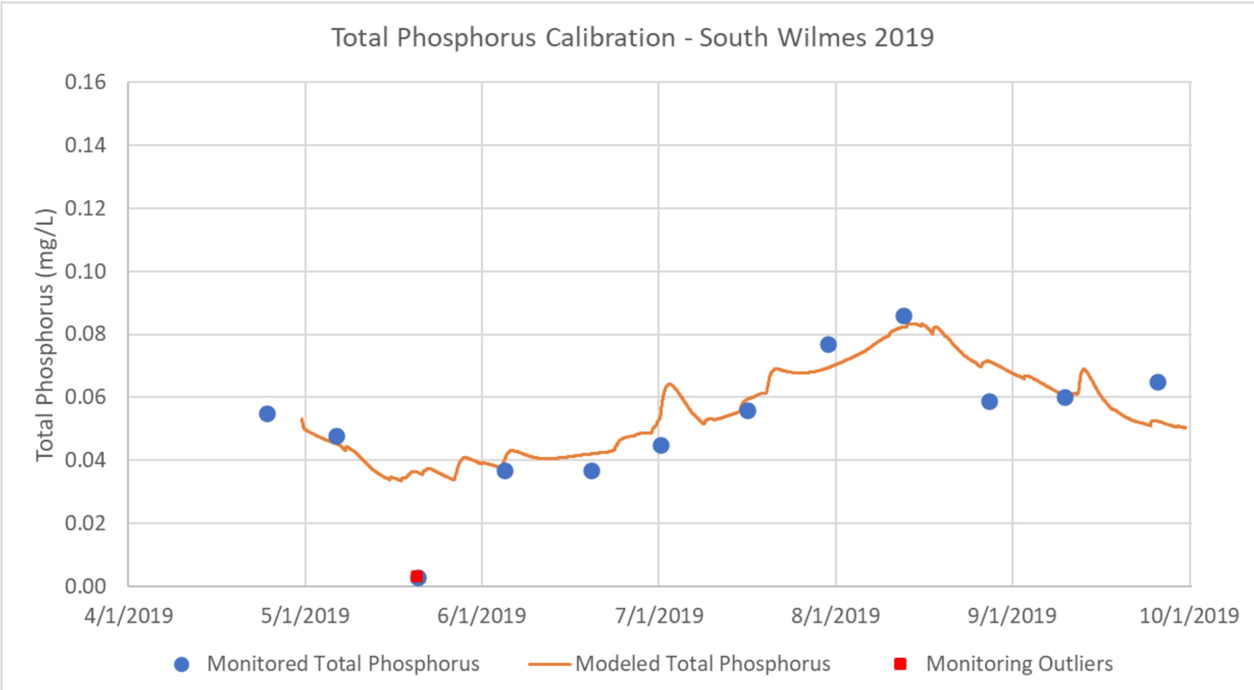

Figure B 31 South Wilmes Lake total phosphorus calibration – 2019

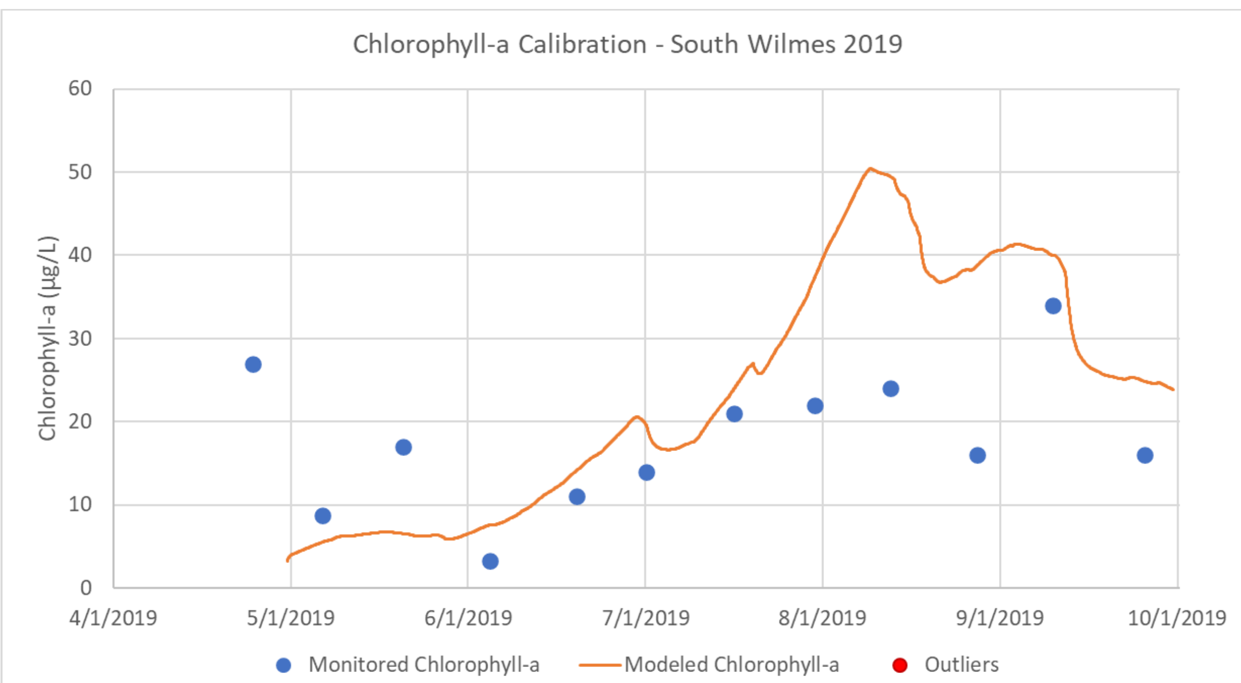

Figure B 32 South Wilmes Lake chlorophyll-a calibration – 2019

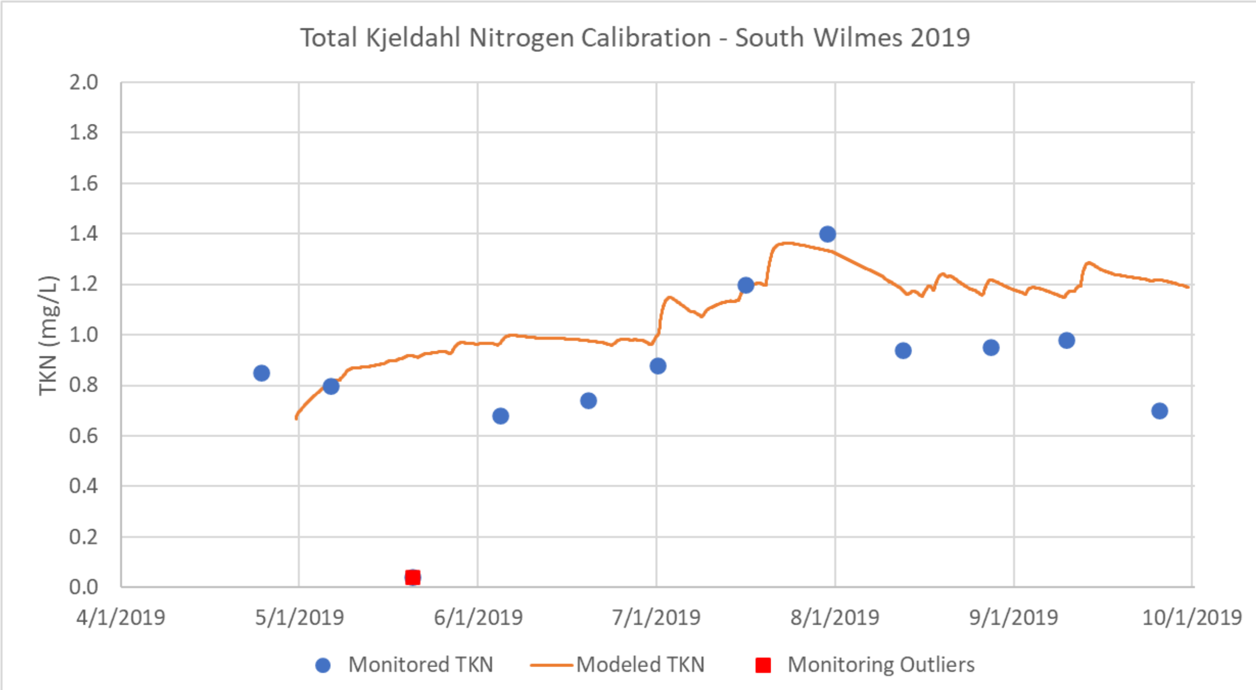

Figure B 33 South Wilmes Lake total Kjeldahl nitrogen calibration – 2019

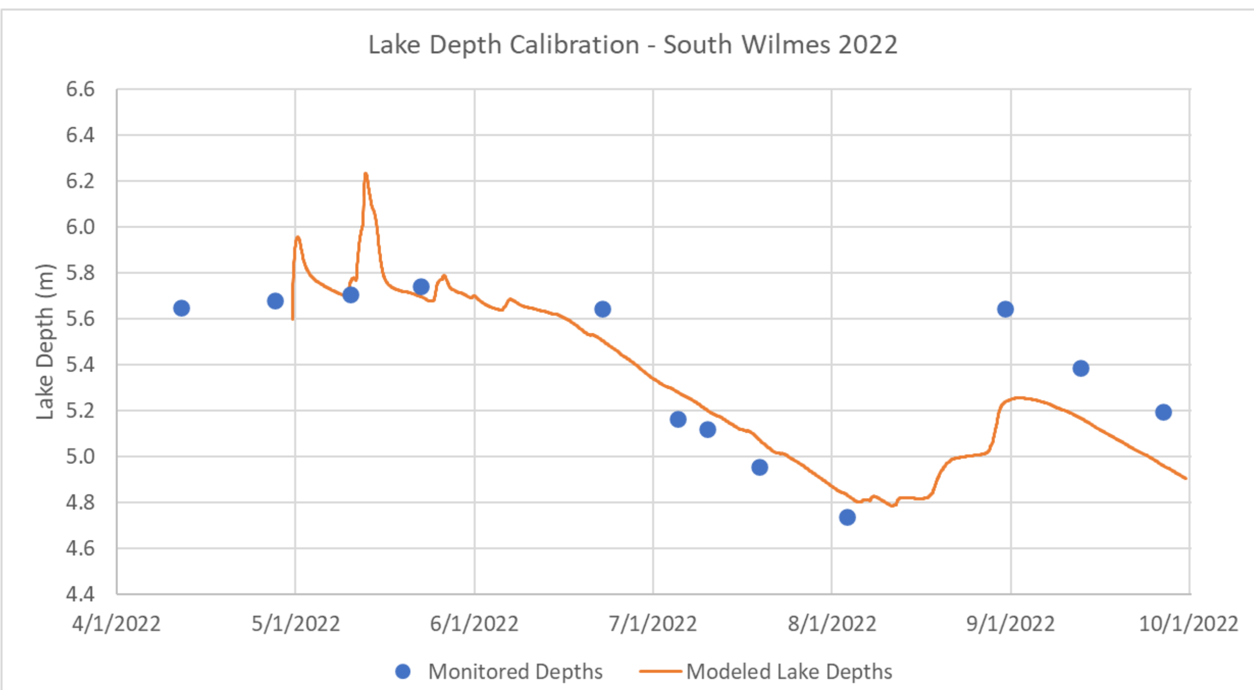

Figure B 34 South Wilmes Lake water balance calibration – 2022

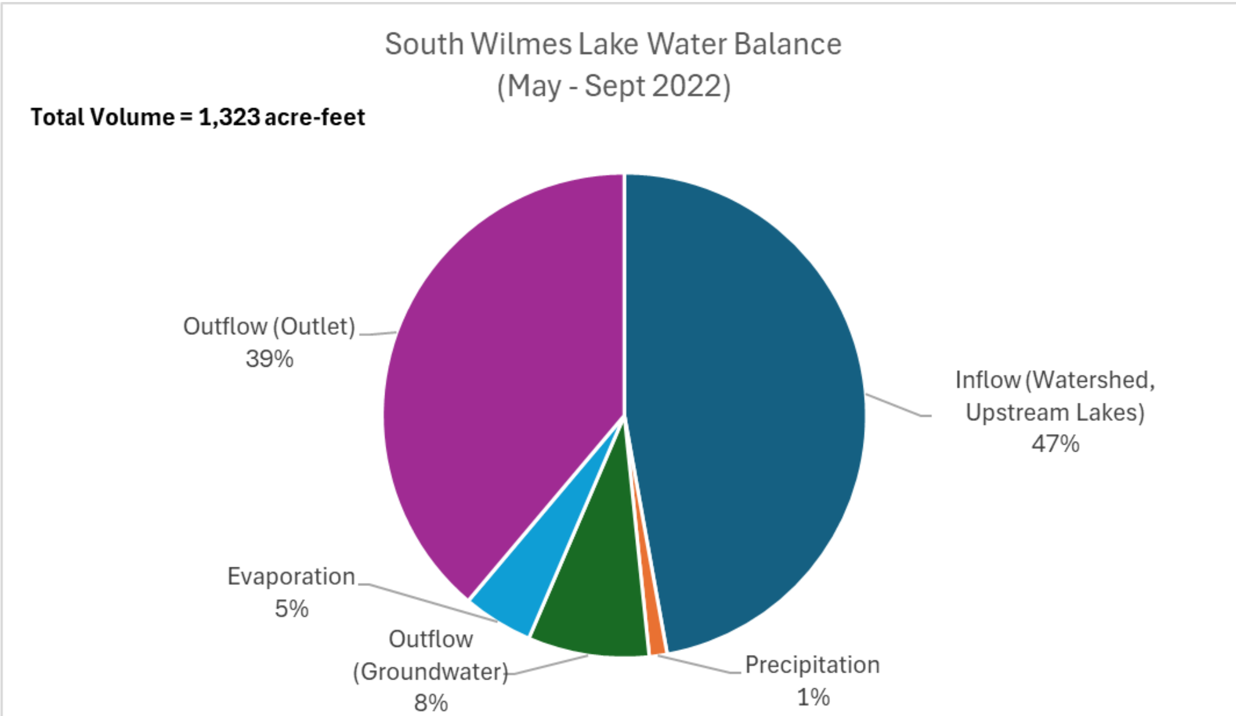

Figure B 35 South Wilmes Lake water balance pie chart – 2022

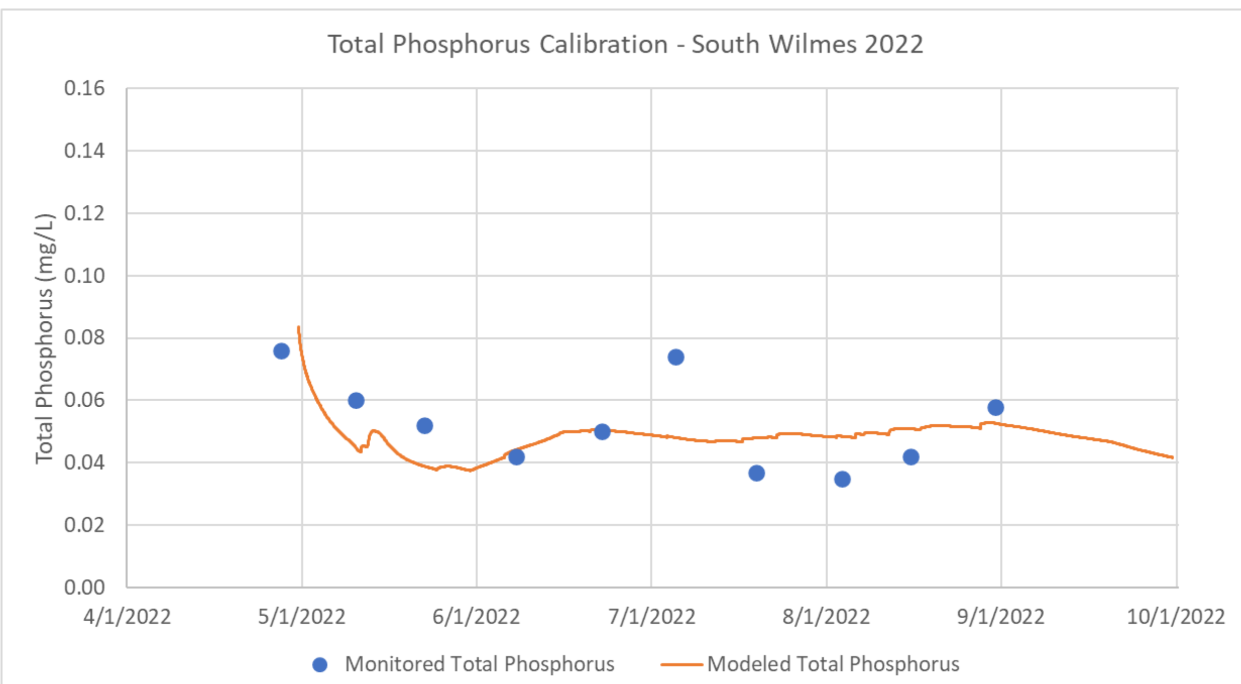

Figure B 36 South Wilmes Lake total phosphorus calibration – 2022

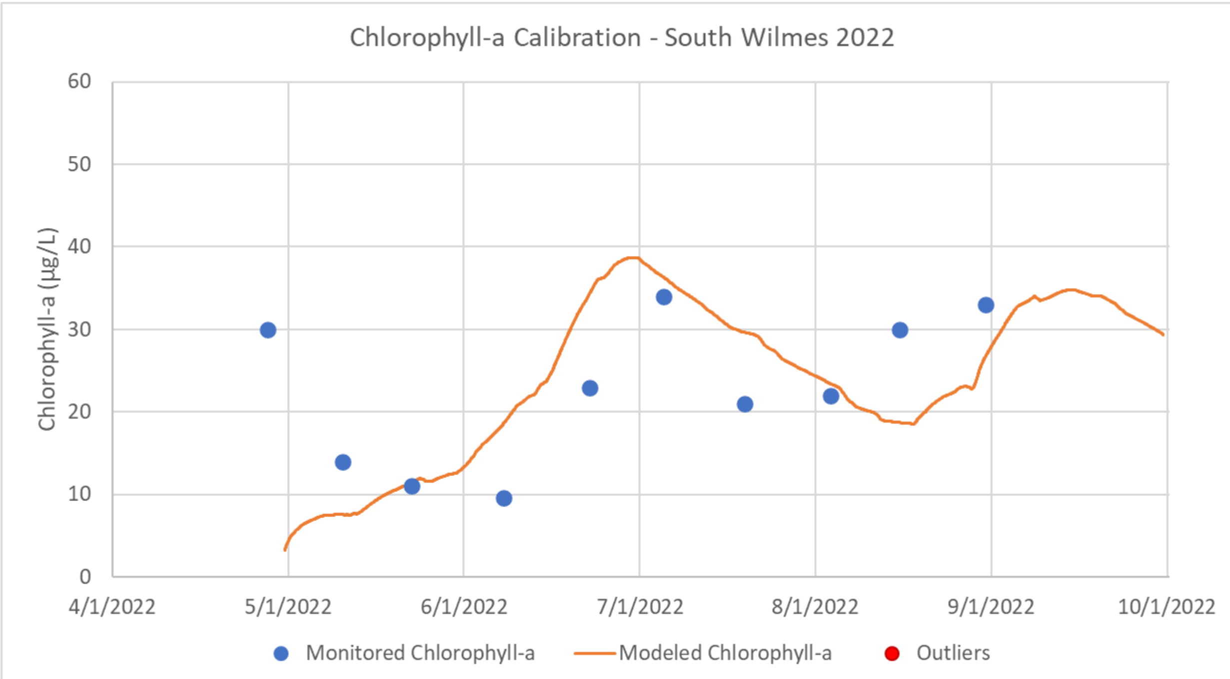

Figure B 37 South Wilmes Lake chlorophyll-a calibration – 2022

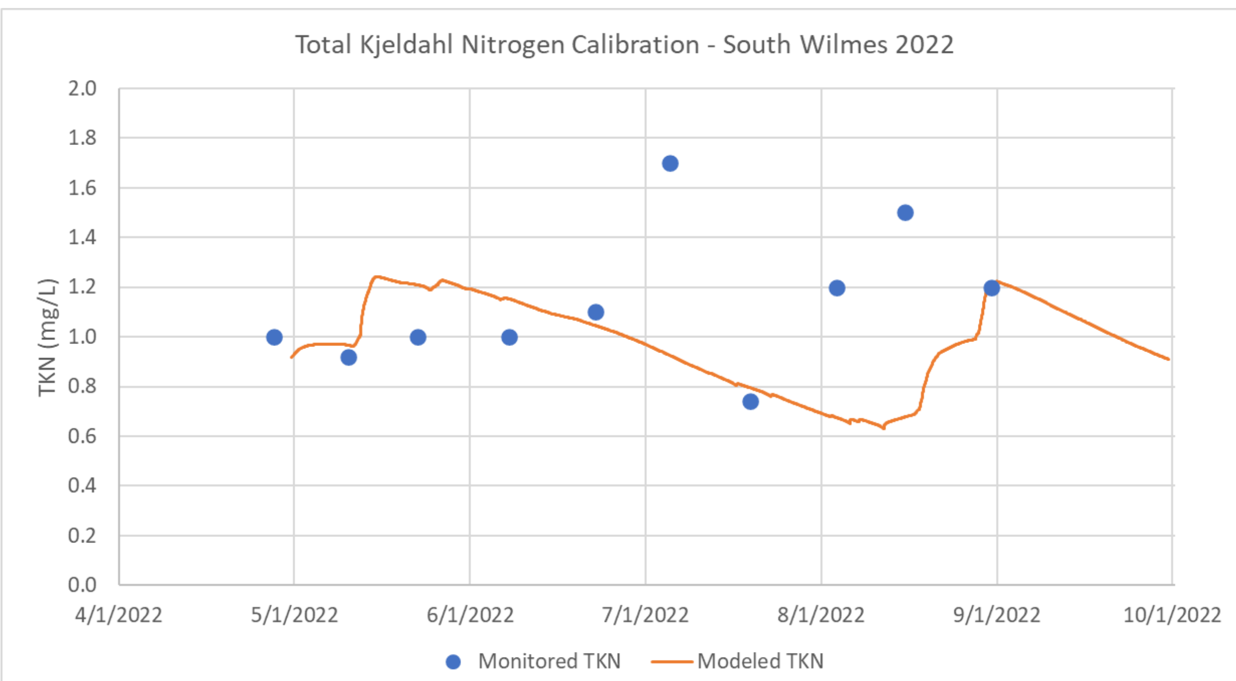

Figure B 38 South Wilmes Lake total Kjeldahl nitrogen calibration – 2022

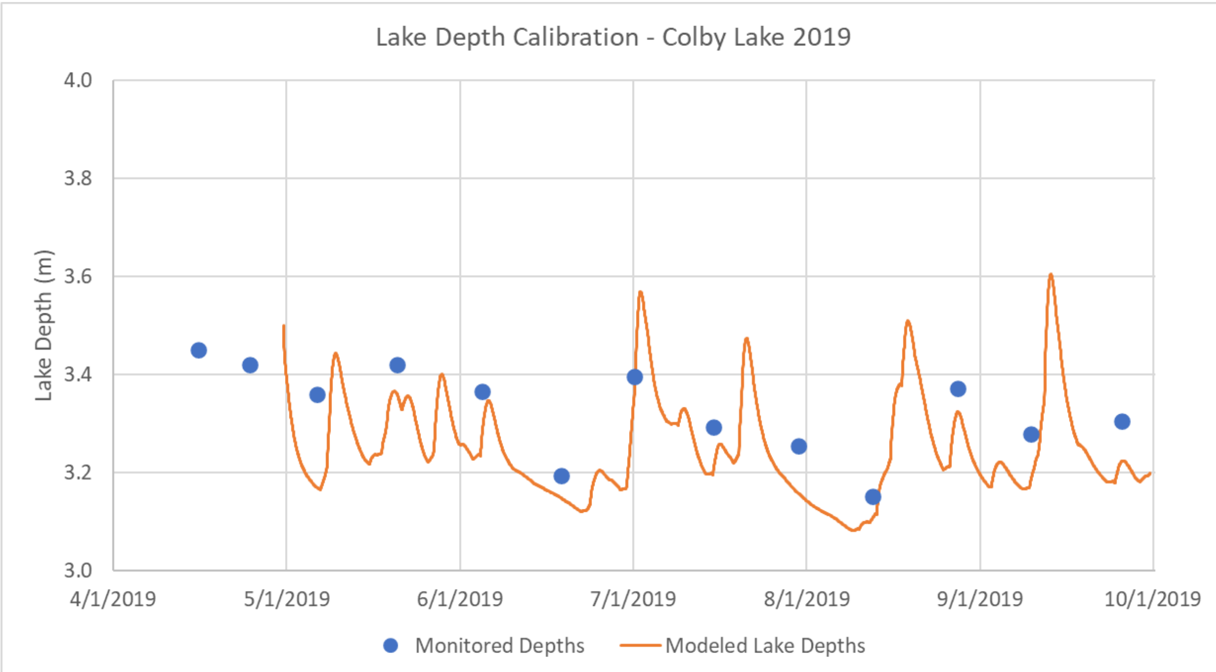

Figure B 39 **Colby Lake water balance calibration – 2019**

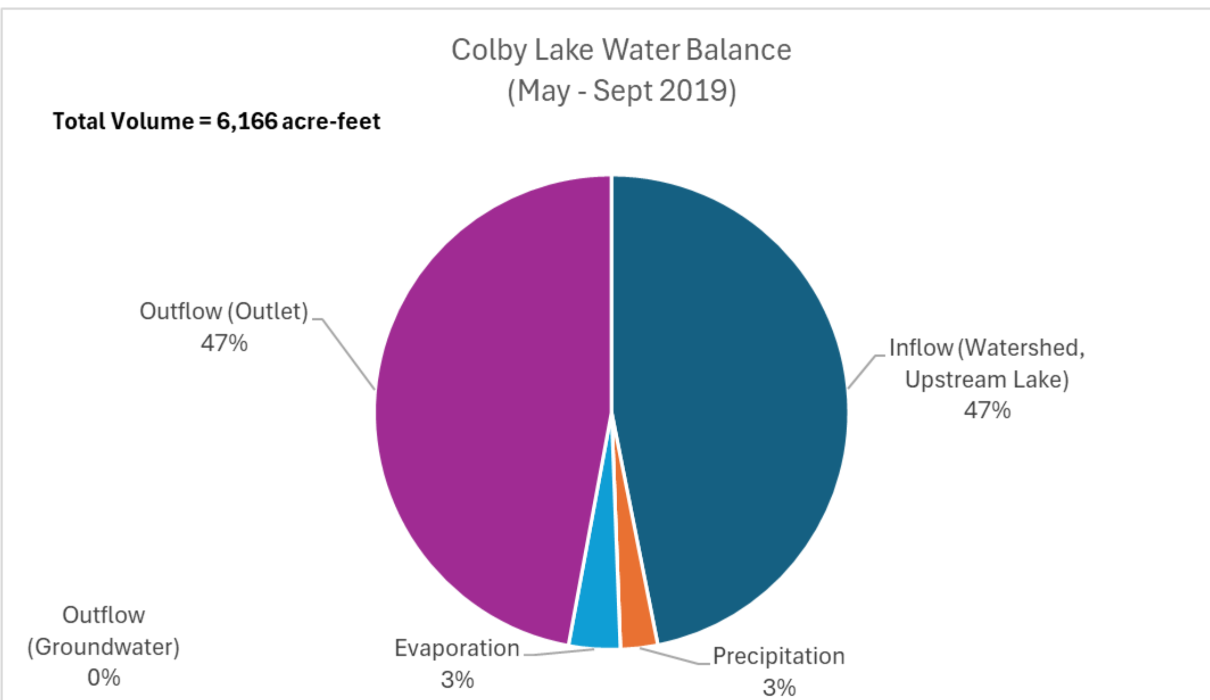

Figure B 40 **Colby Lake water balance pie chart – 2019**

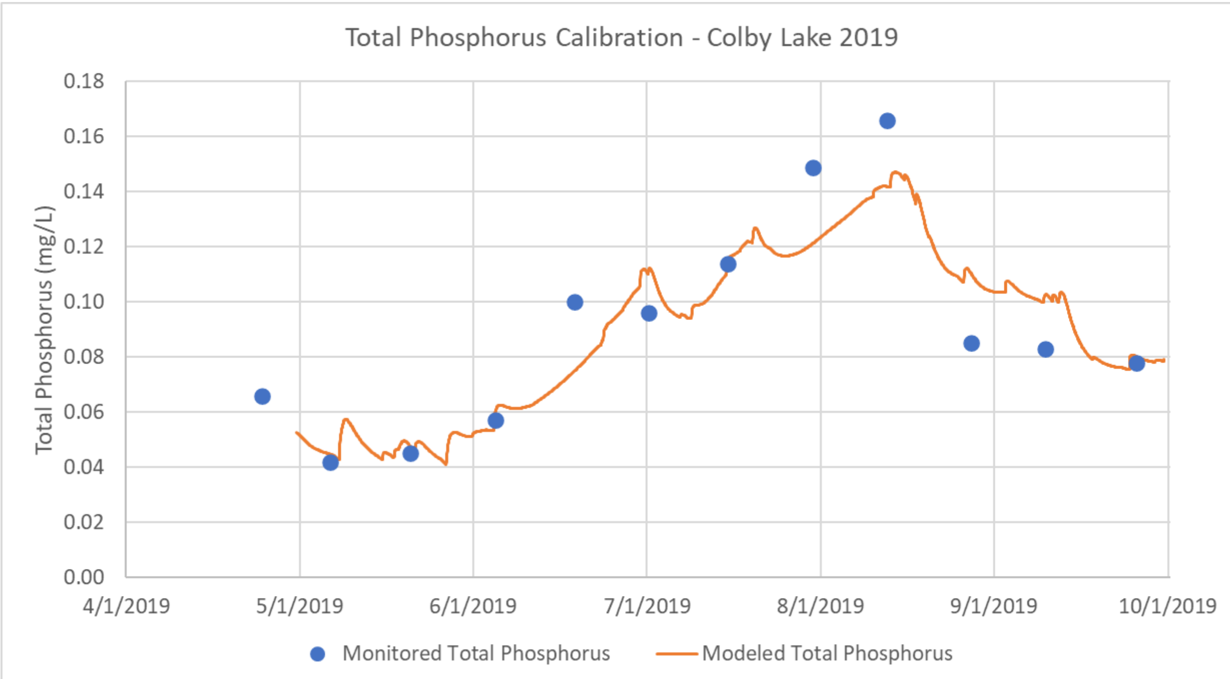

Figure B 41 Colby Lake total phosphorus calibration – 2019

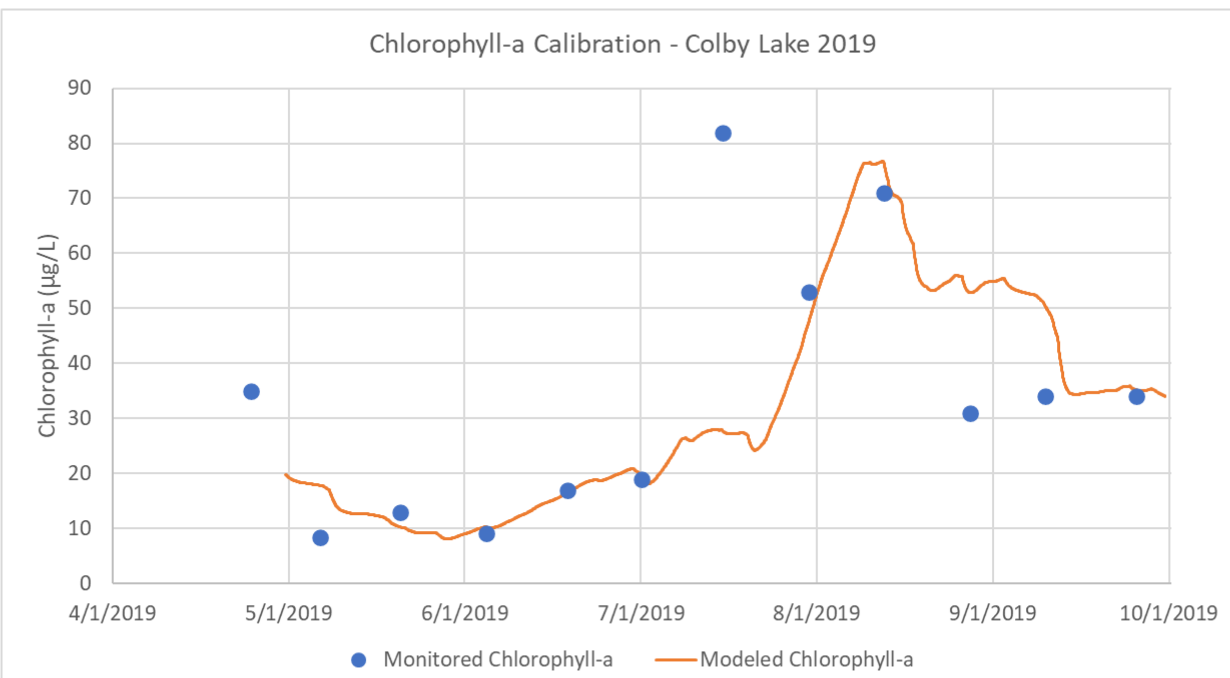

Figure B 42 Colby Lake chlorophyll-a calibration – 2019

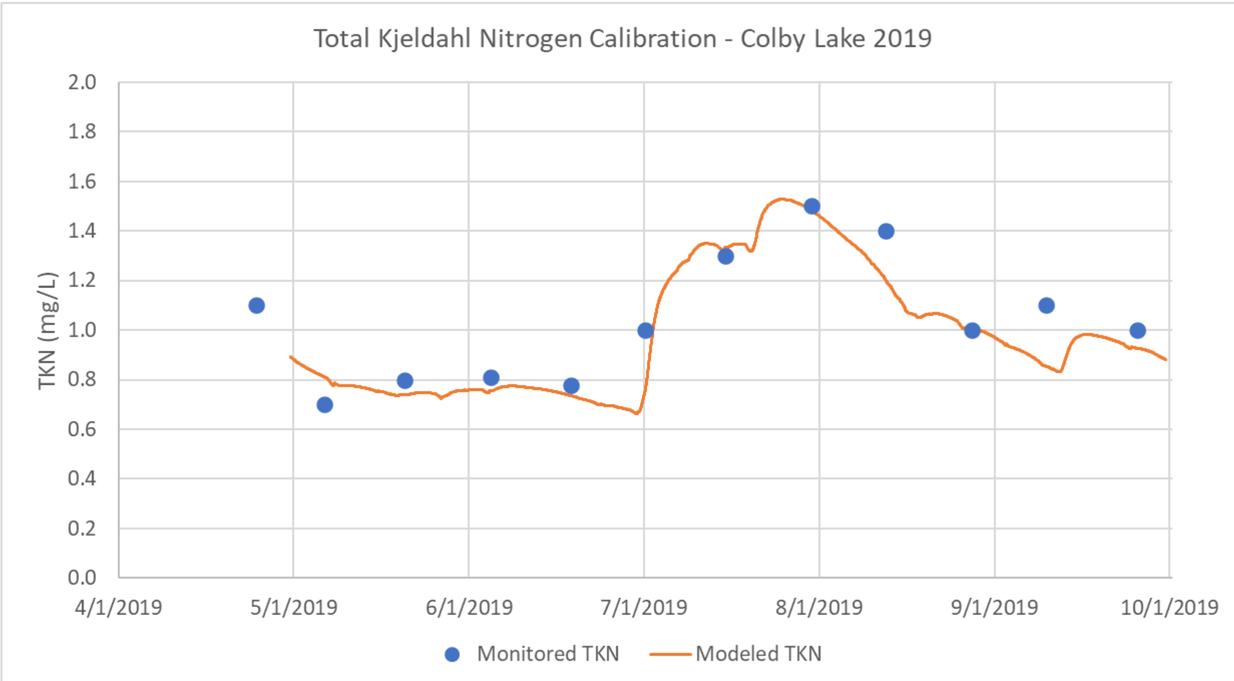

Figure B 43 **Colby Lake total Kjeldahl nitrogen calibration – 2019**

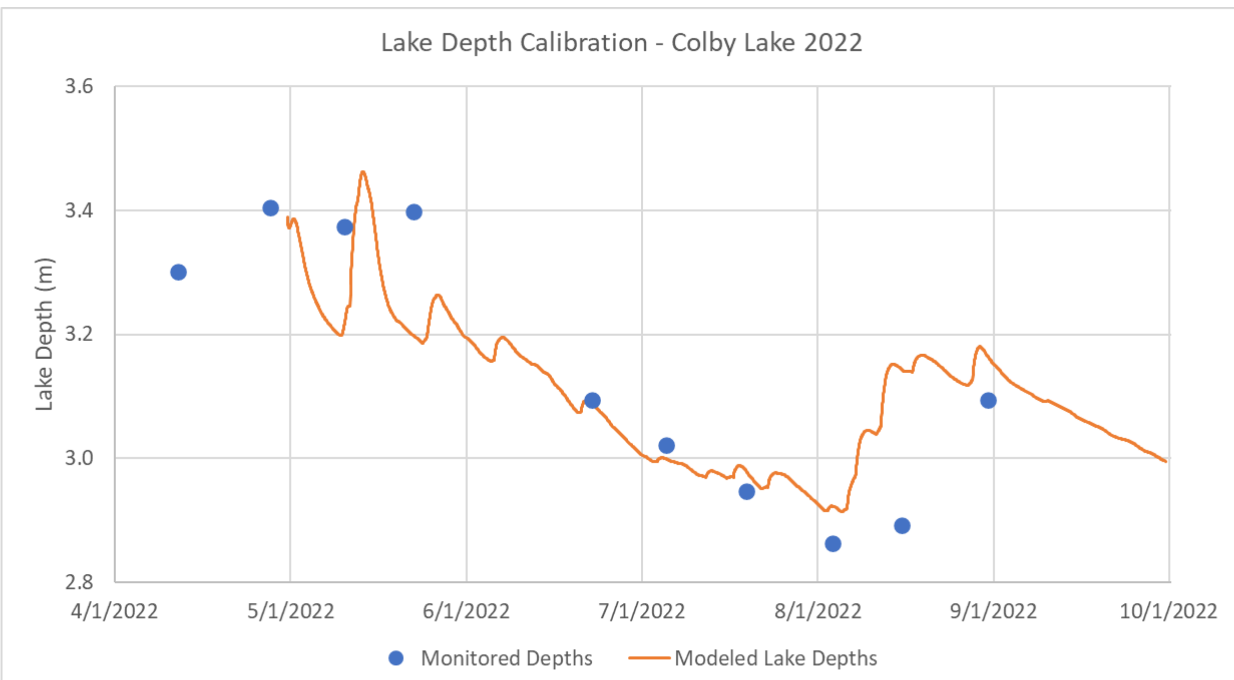

Figure B 44 **Colby Lake water balance calibration – 2022**

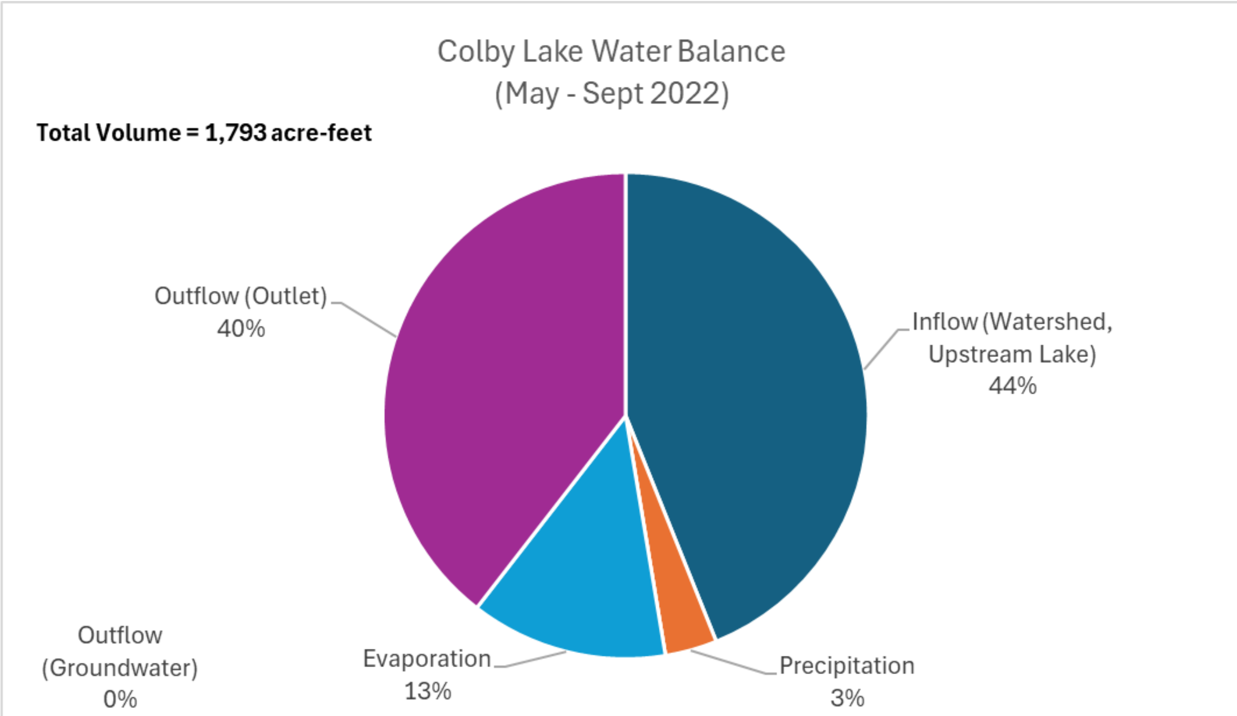

Figure B 45 Colby Lake water balance pie chart – 2022

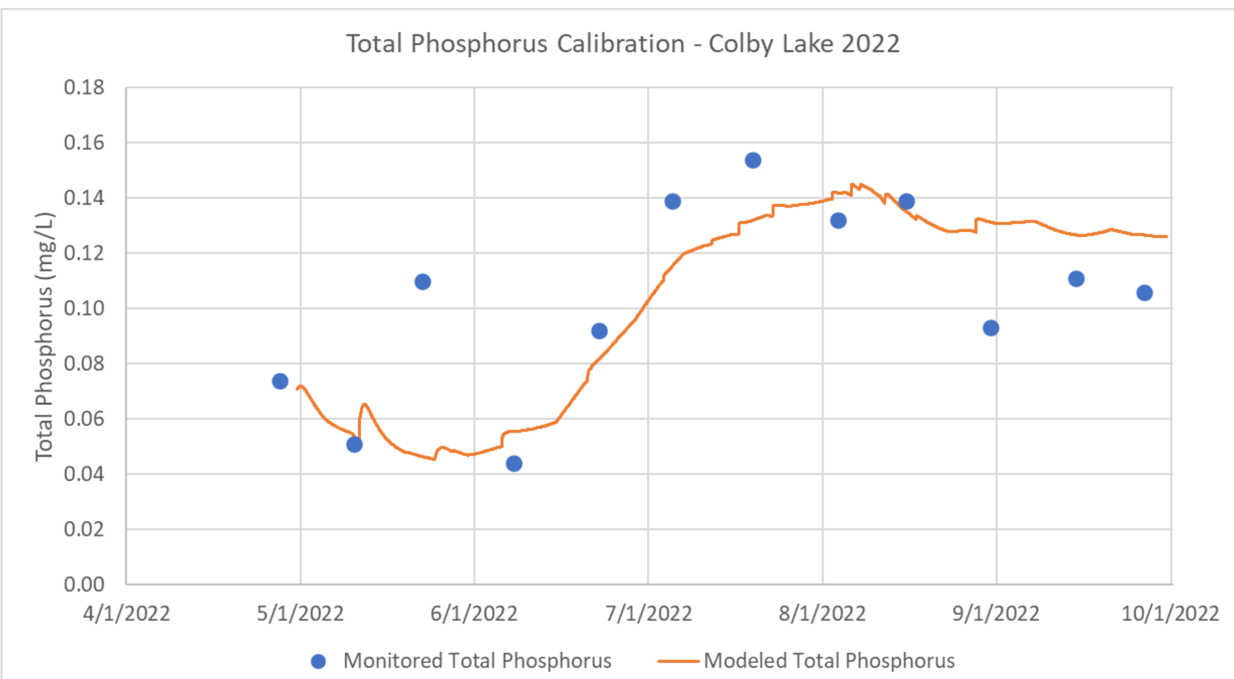

Figure B 46 Colby Lake total phosphorus calibration – 2022

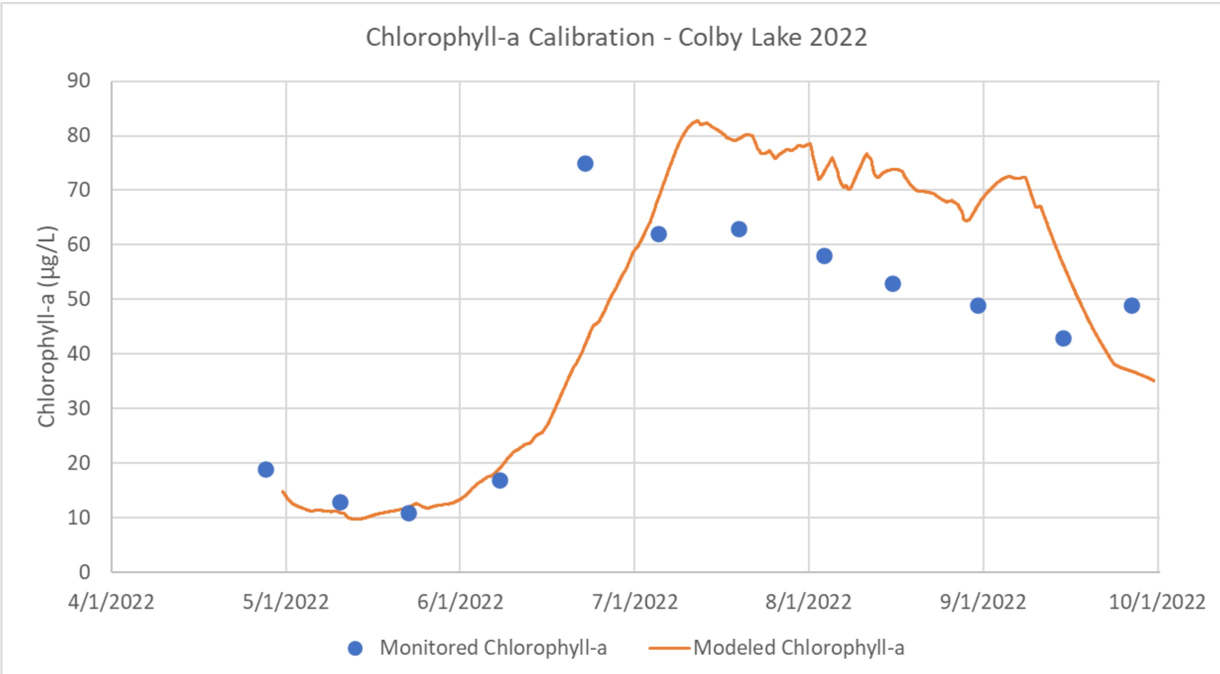

Figure B 47 Colby Lake chlorophyll-a calibration – 2022

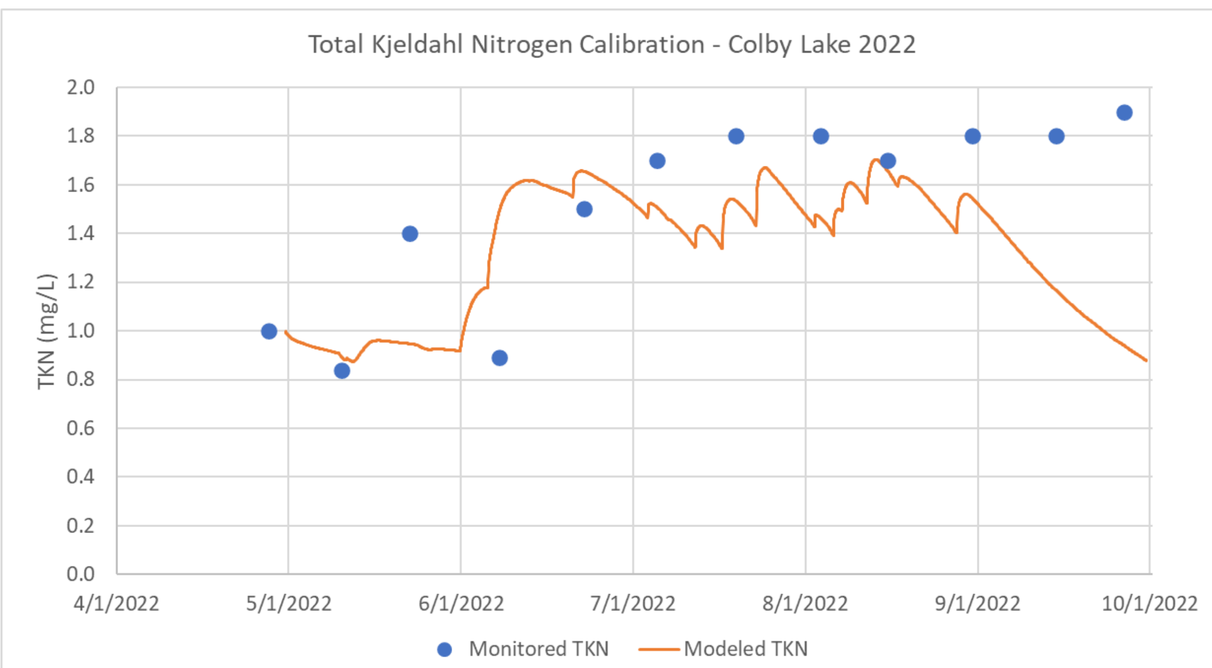

Figure B 48 Colby Lake total Kjeldahl nitrogen calibration – 2022

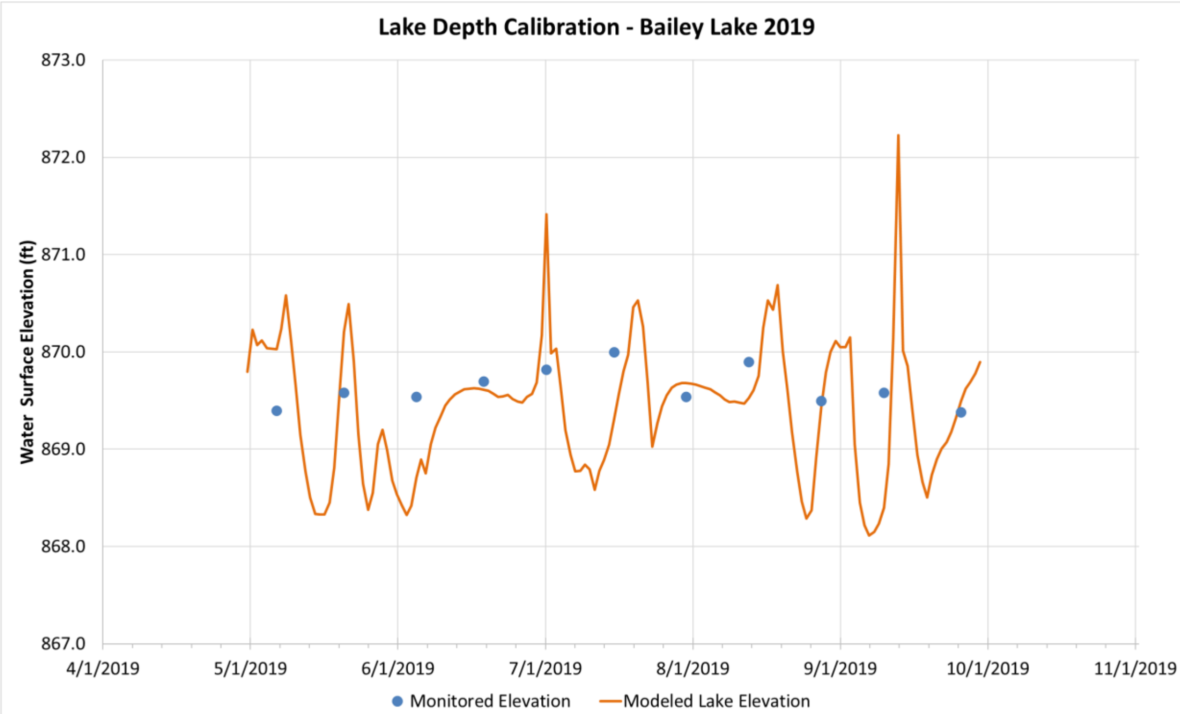

Figure B 49 Bailey Lake water balance calibration – 2019

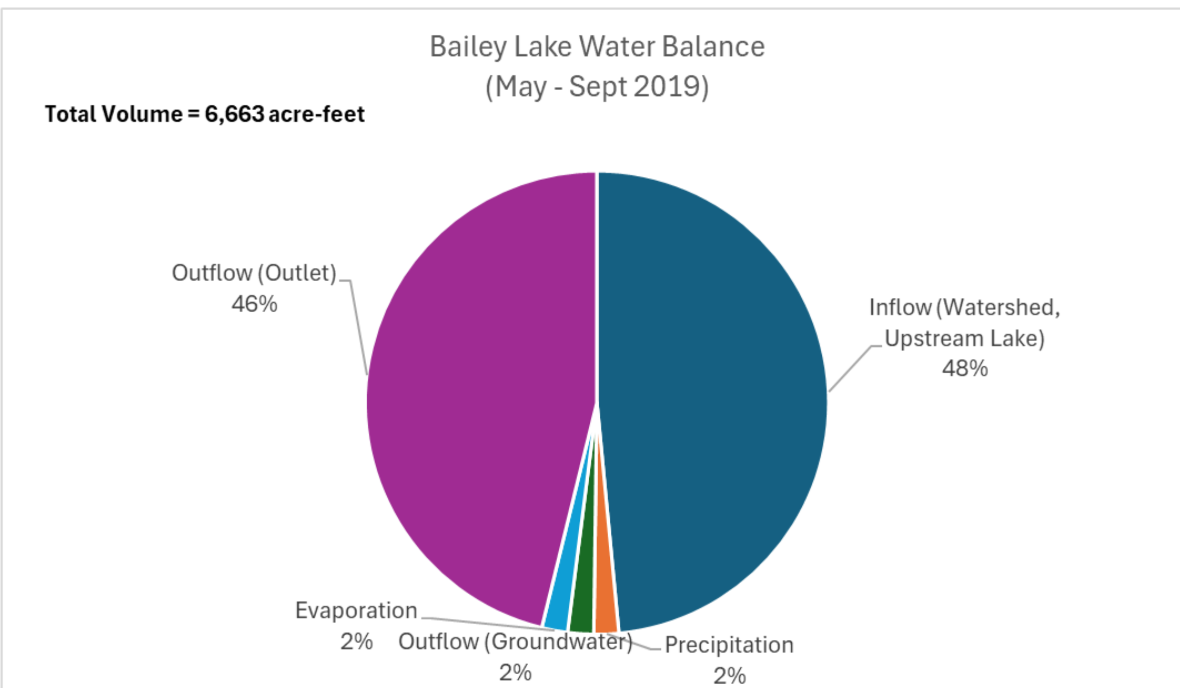

Figure B 50 Bailey Lake water balance pie chart – 2019

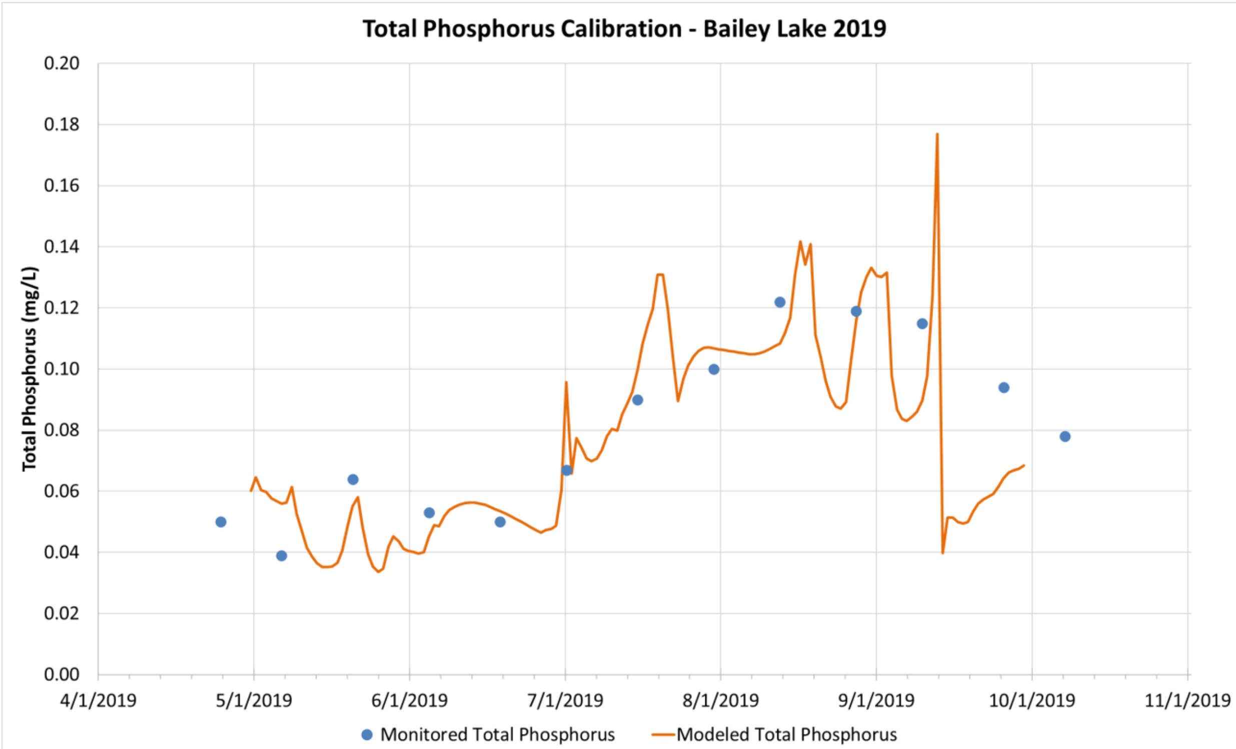

Figure B 51 Bailey Lake total phosphorus calibration – 2019

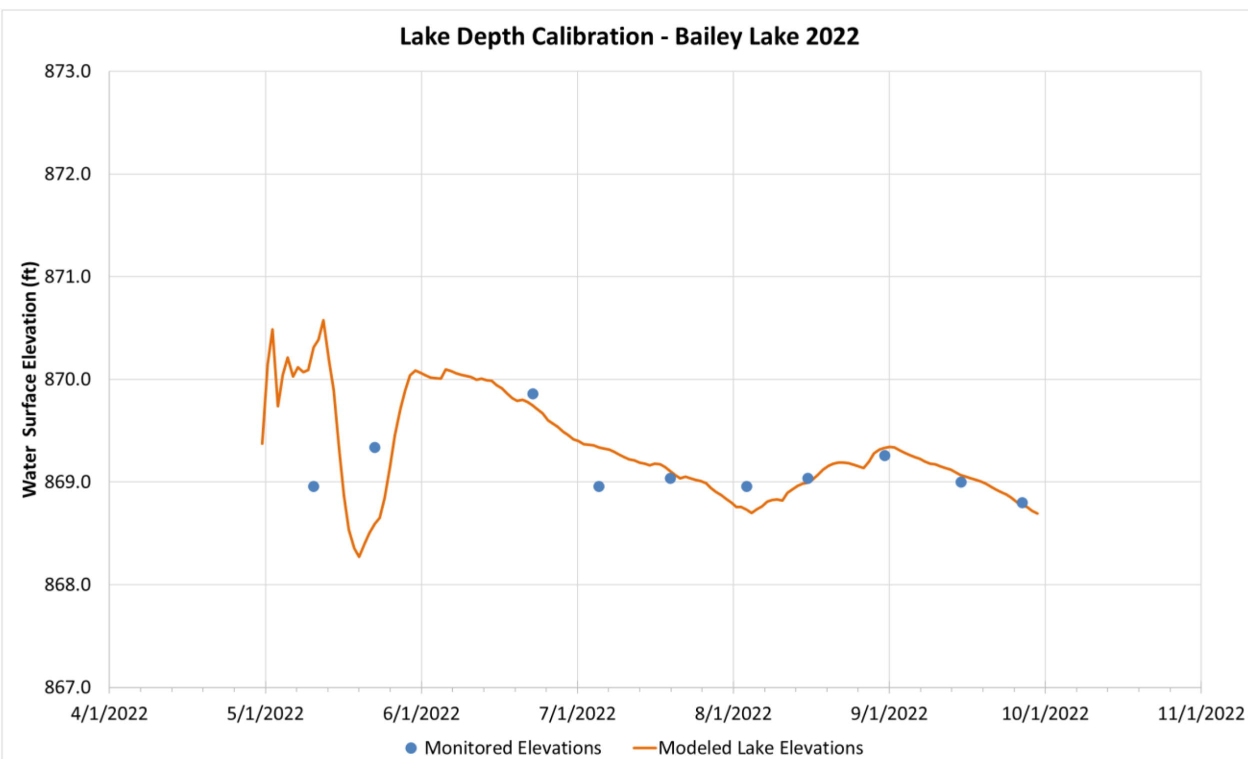

Figure B 52 Bailey Lake water balance calibration – 2022

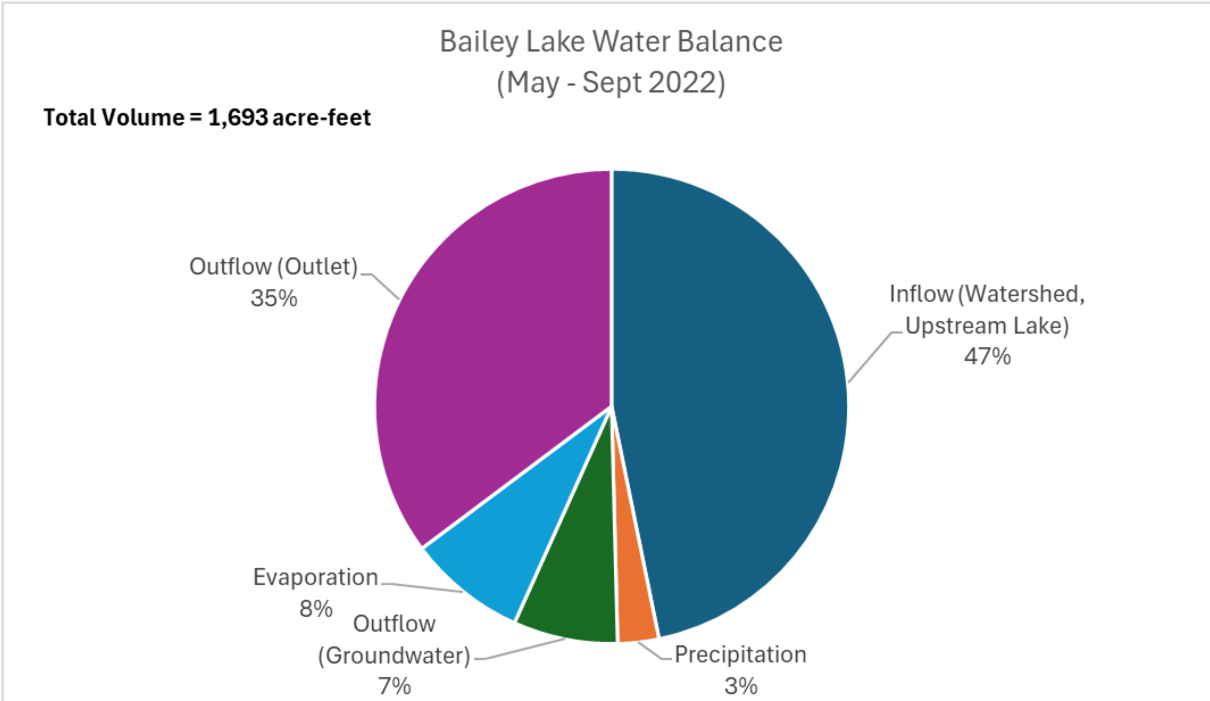

Figure B 53 Bailey Lake water balance pie chart – 2022

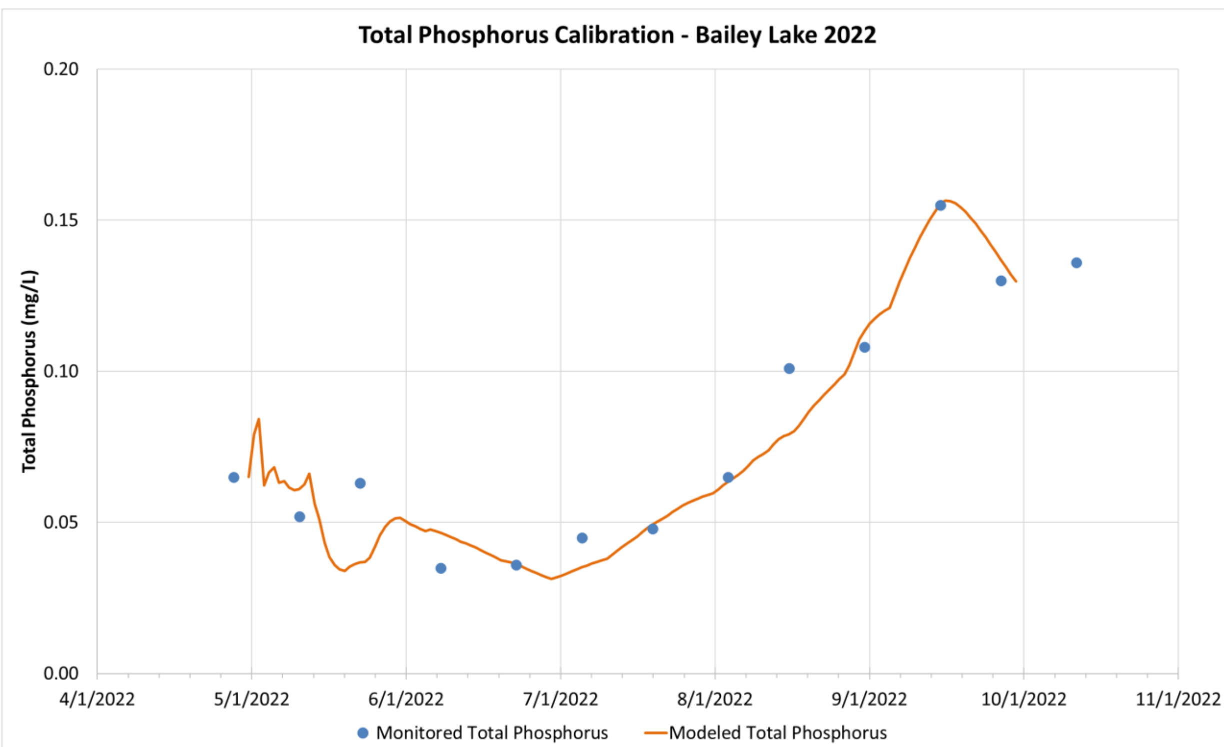

Figure B 54 Bailey Lake total phosphorus calibration – 2022

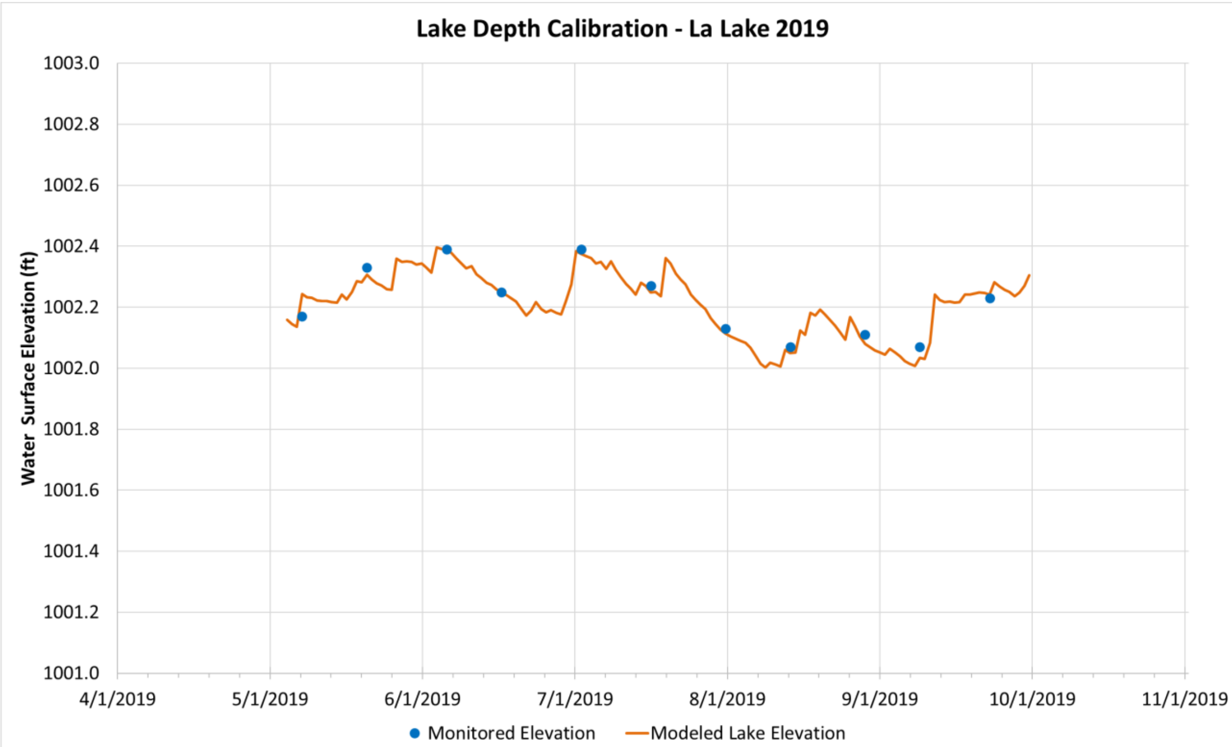

Figure B 55 La Lake water balance calibration – 2019

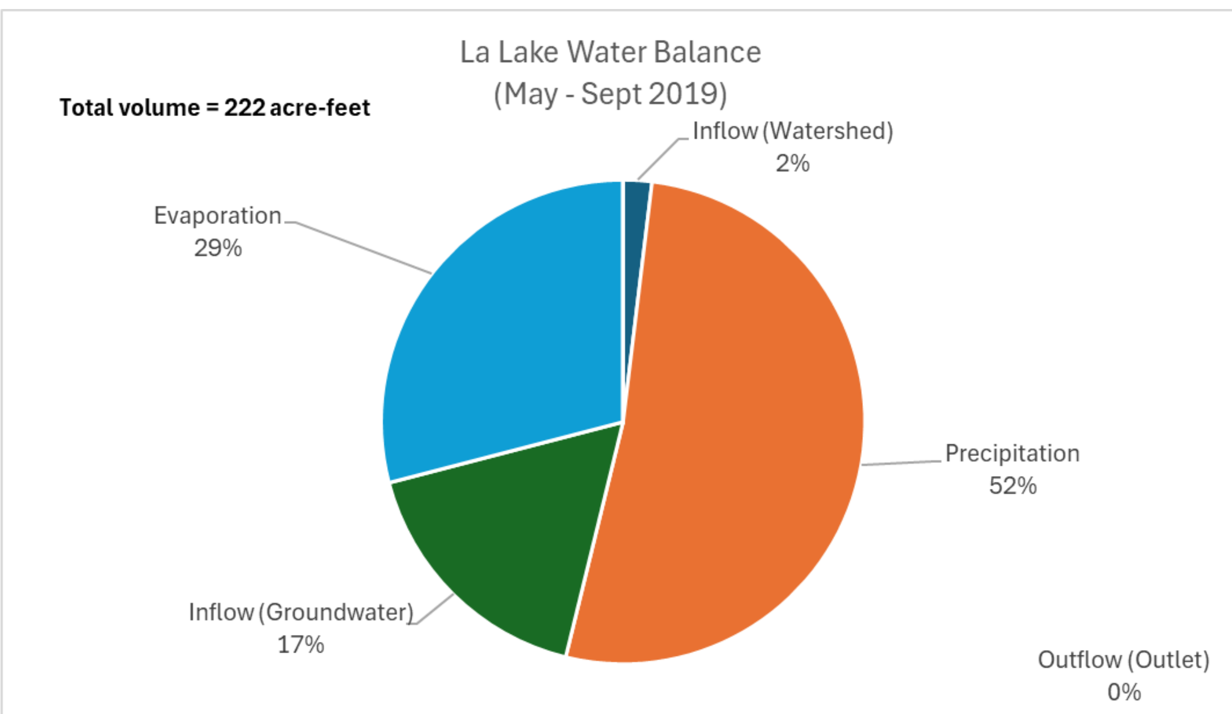

Figure B 56 La Lake water balance pie chart – 2019

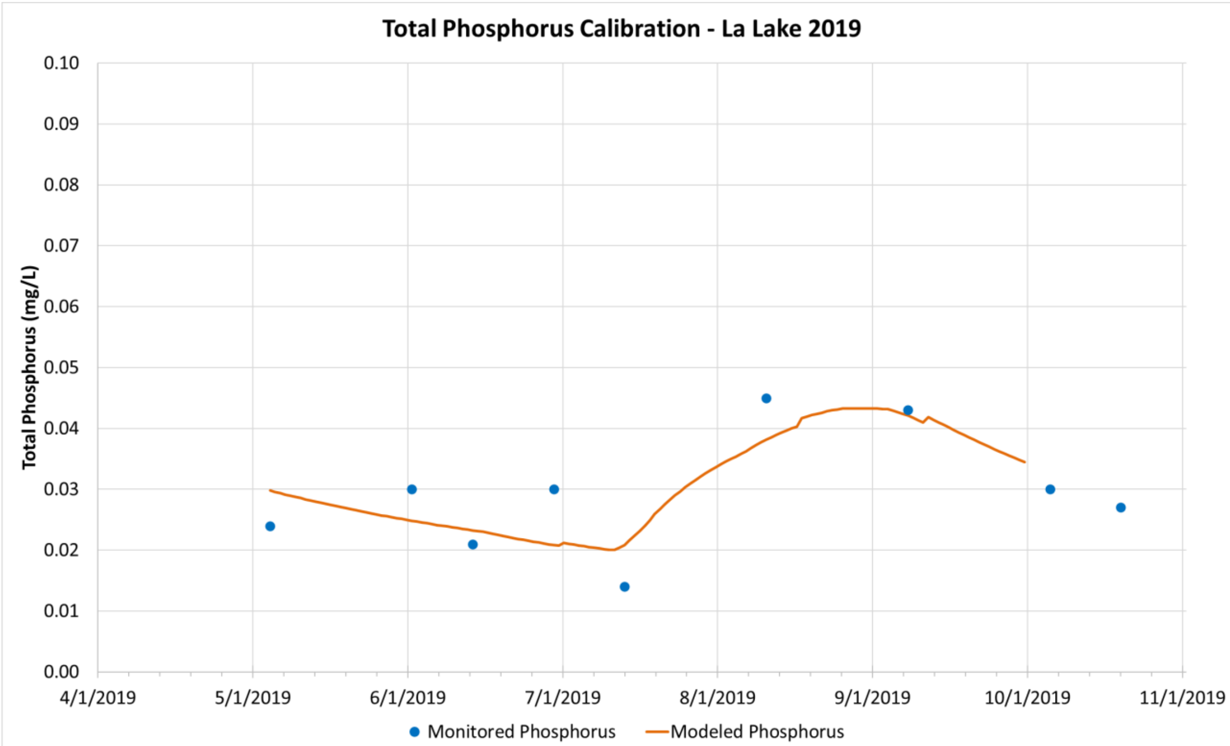

Figure B 57 La Lake total phosphorus calibration – 2019

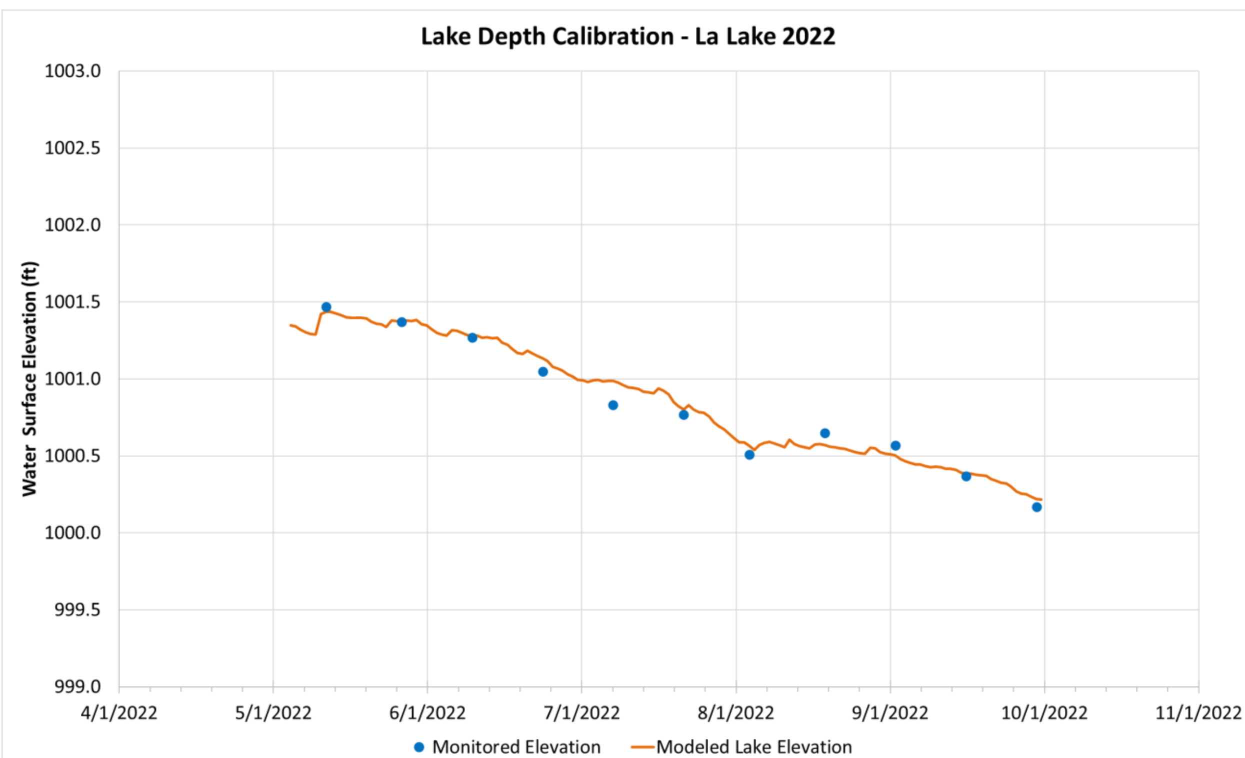

Figure B 58 La Lake water balance calibration – 2022

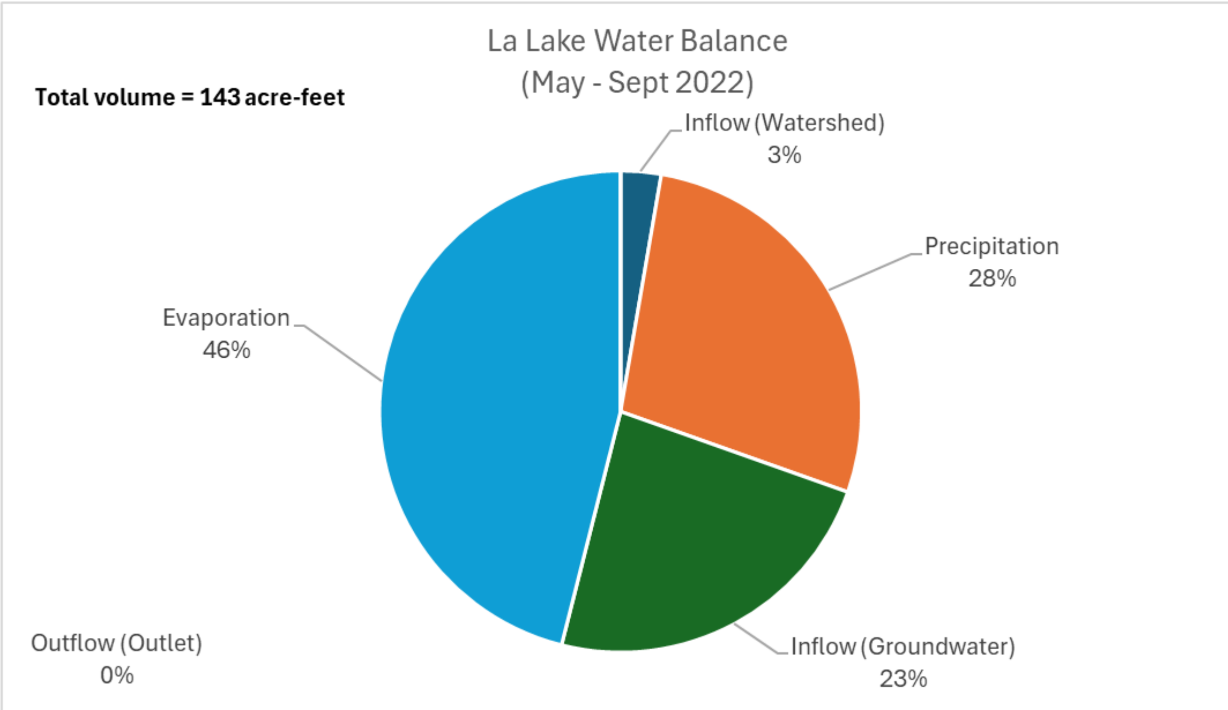

Figure B 59 La Lake water balance pie chart - 2022

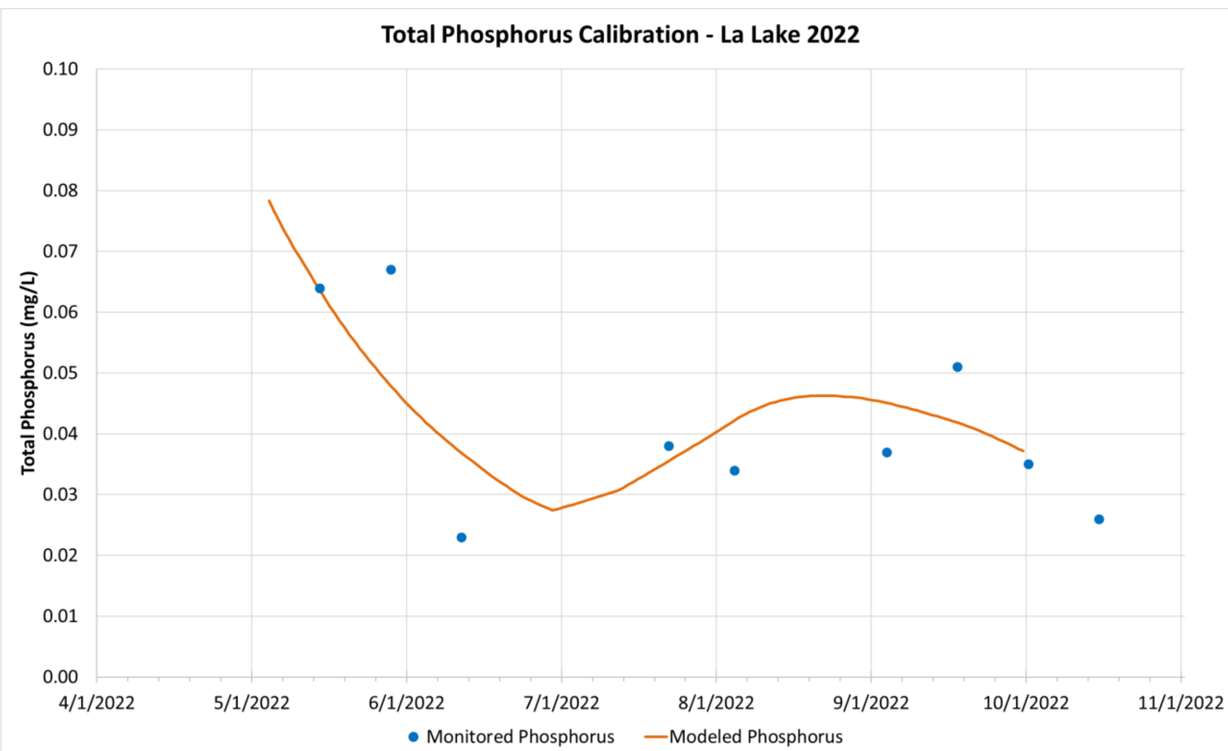

Figure B 60 La Lake total phosphorus calibration - 2022

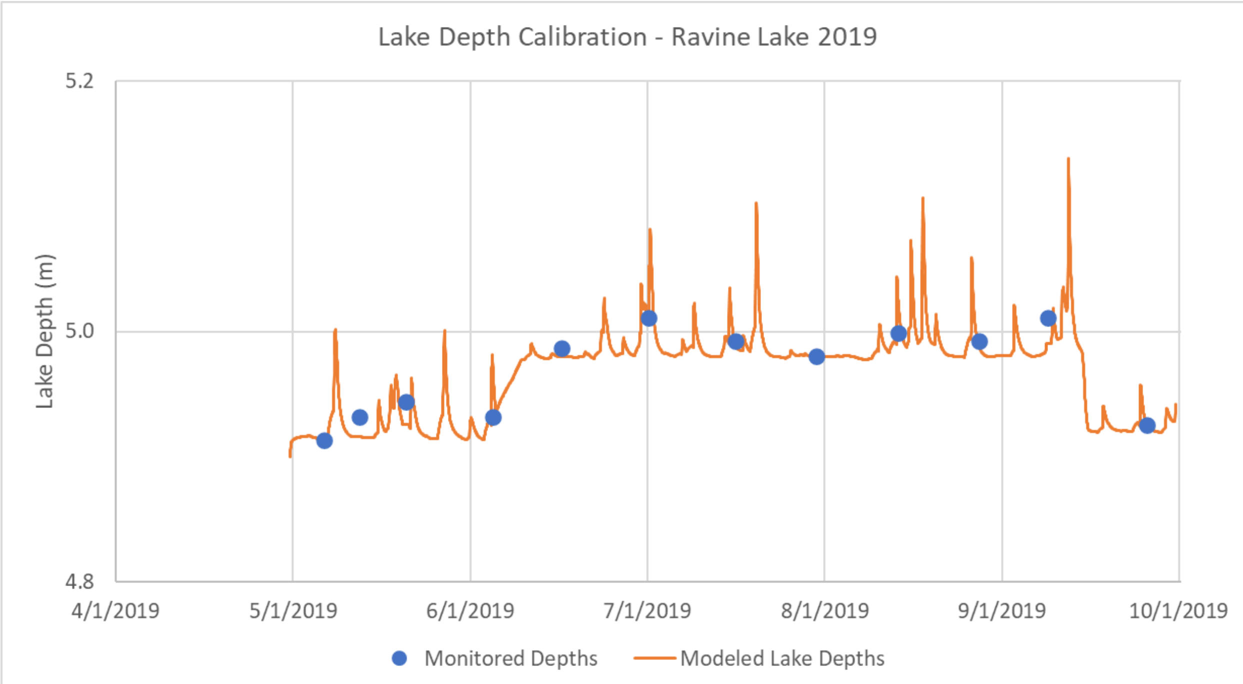

Figure B 61 Ravine Lake water balance calibration – 2019

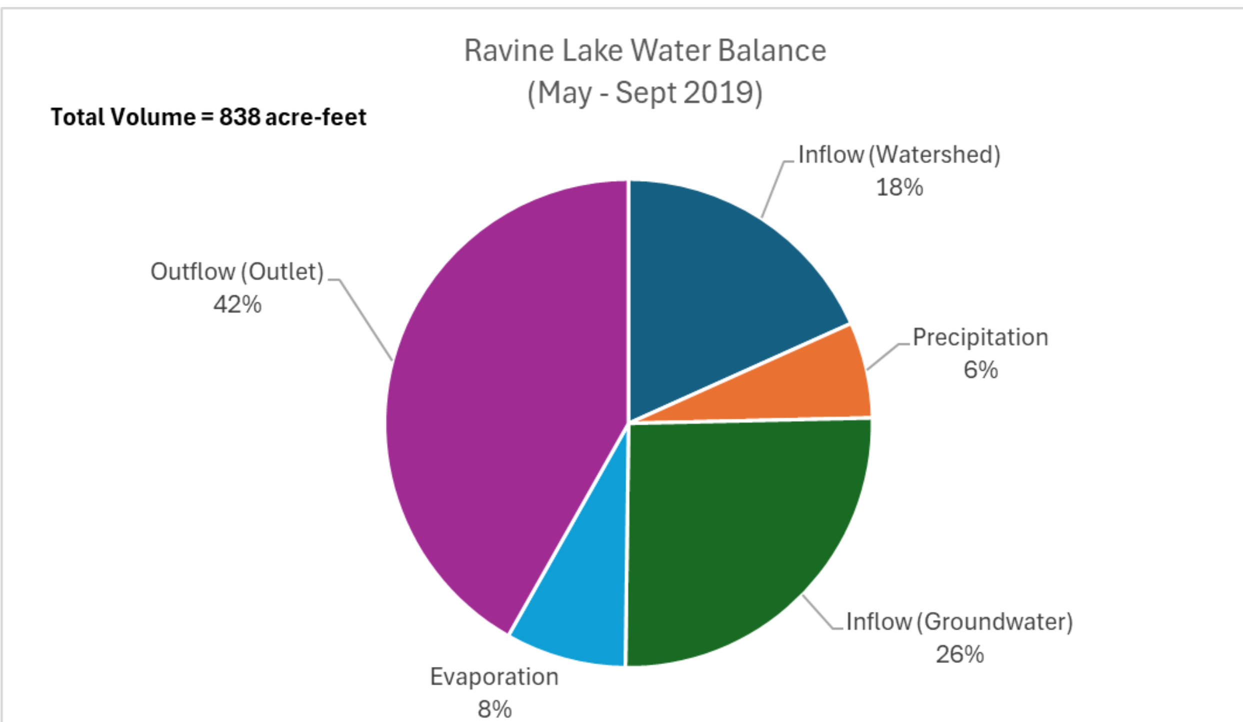

Figure B 62 Ravine Lake water balance pie chart – 2019

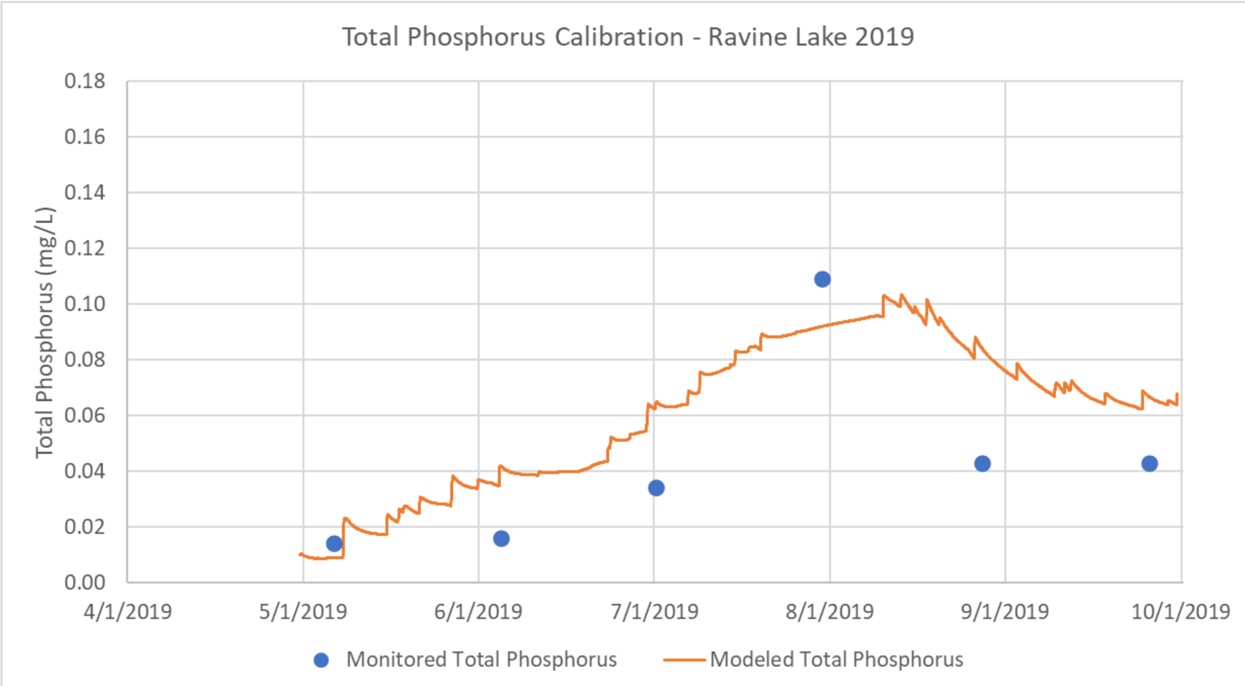

Figure B 63 Ravine Lake total phosphorus calibration – 2019

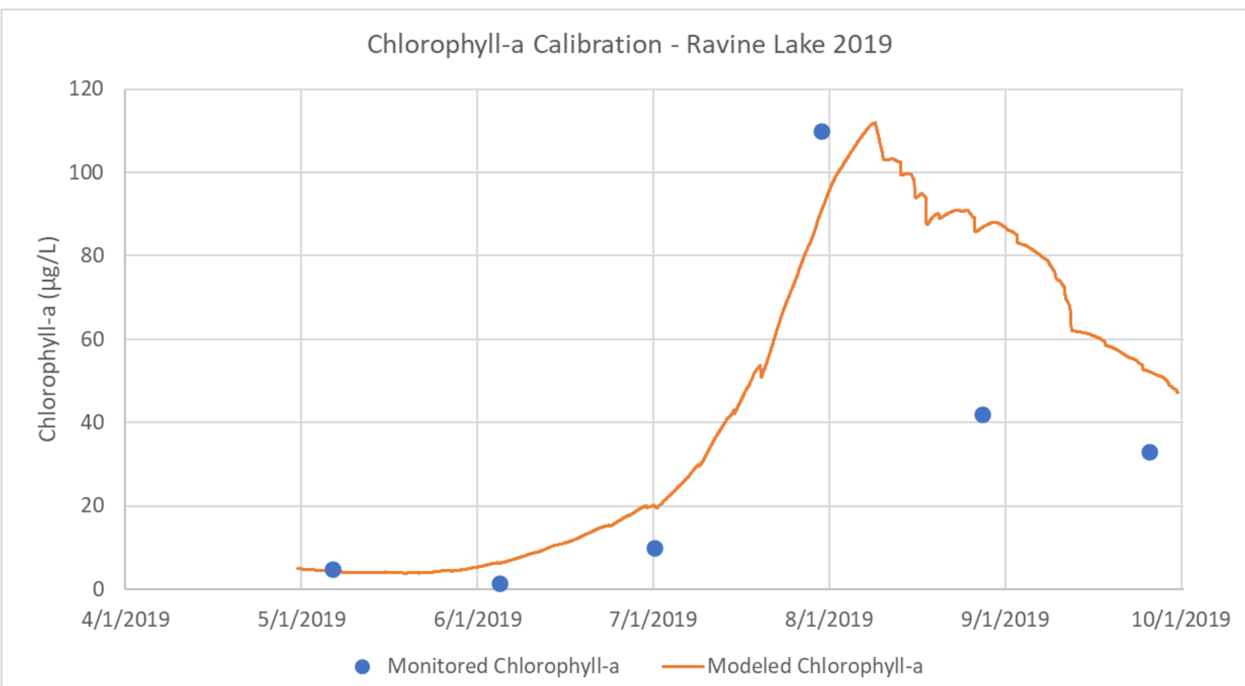

Figure B 64 Ravine Lake chlorophyll-a calibration – 2019

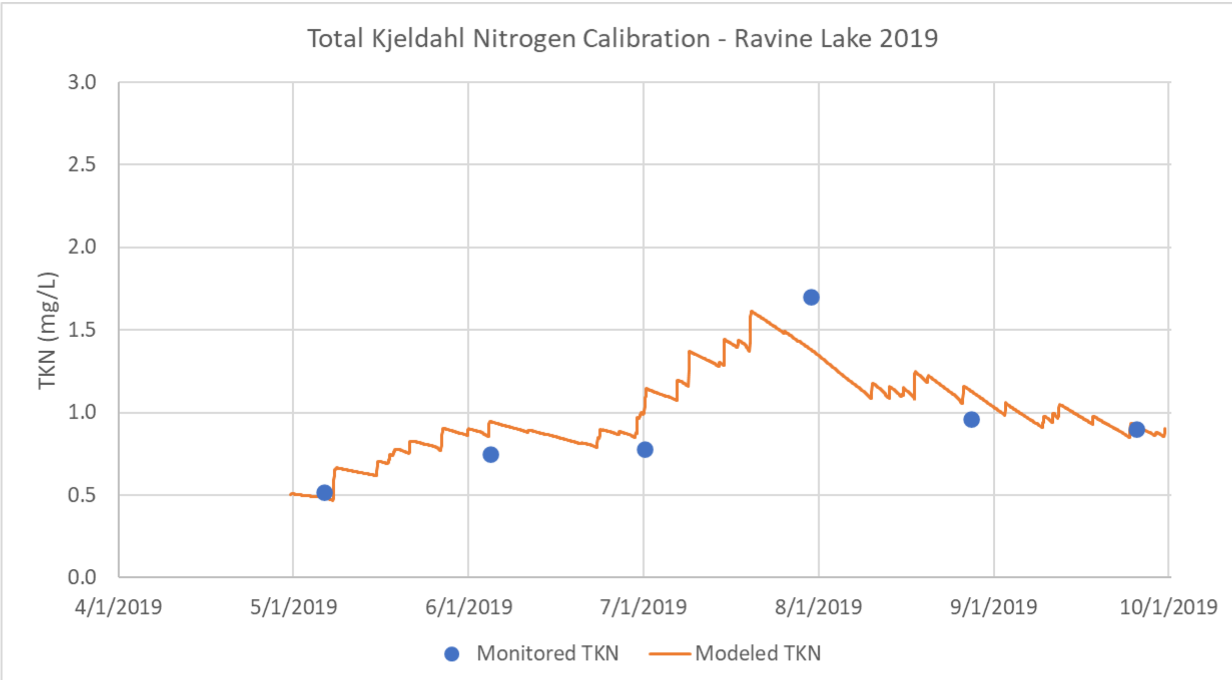

**Figure B 65	Ravine Lake total Kjeldahl nitrogen calibration – 2019**

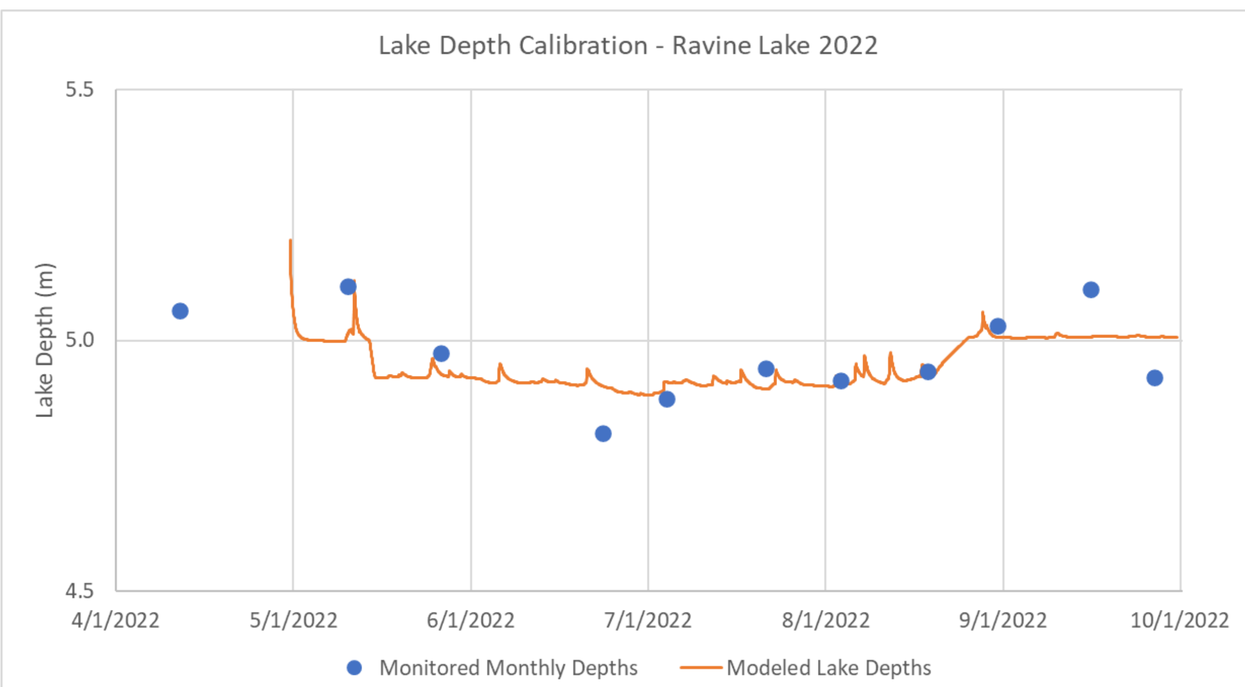

**Figure B 66	Ravine Lake water balance calibration – 2022**

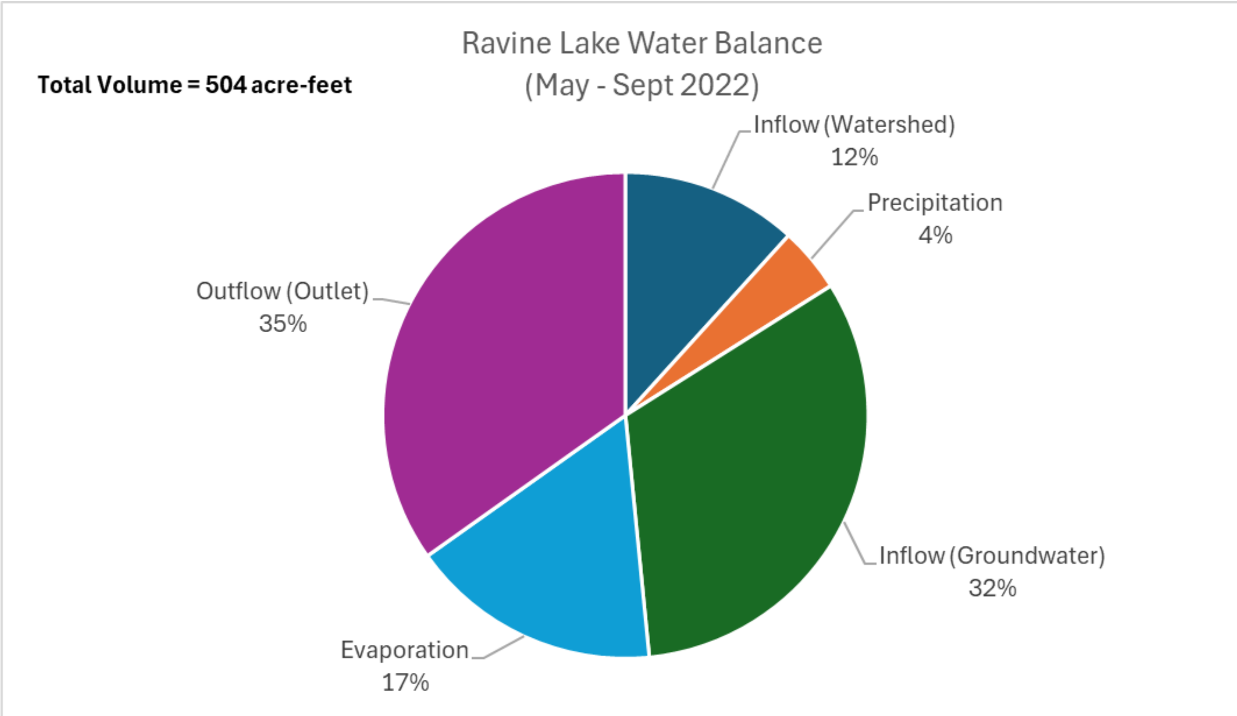

Figure B 67 Ravine Lake water balance pie chart – 2022

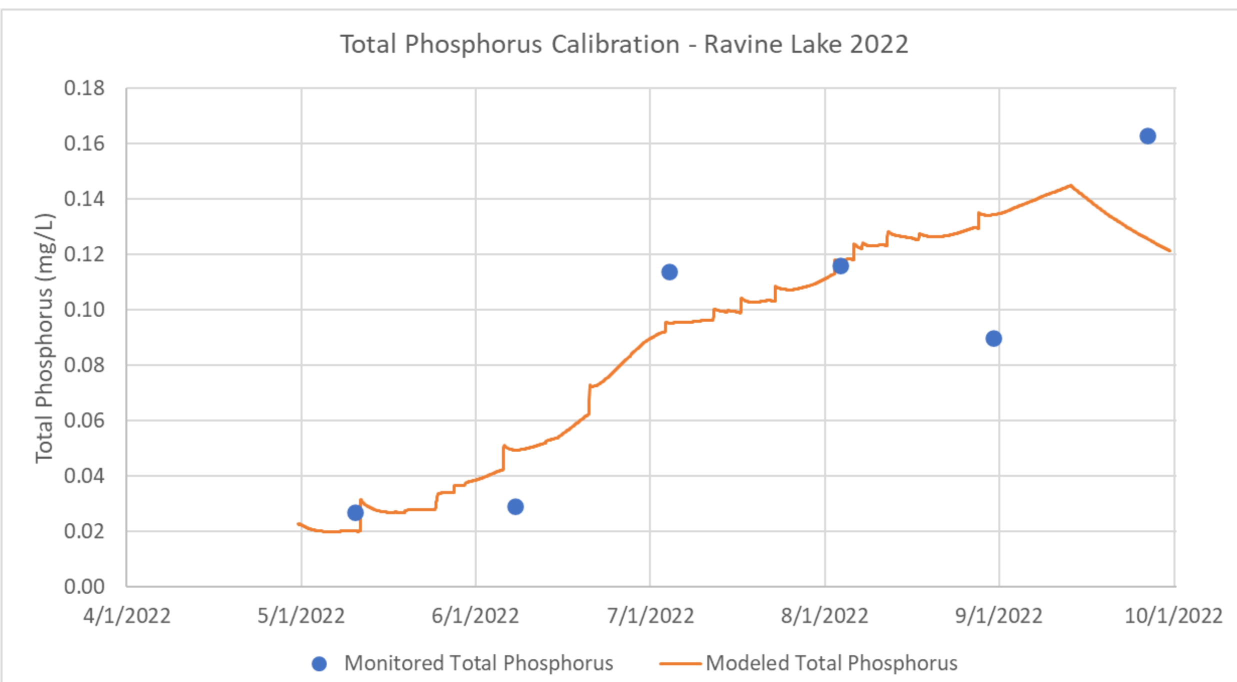

Figure B 68 Ravine Lake total phosphorus calibration – 2022

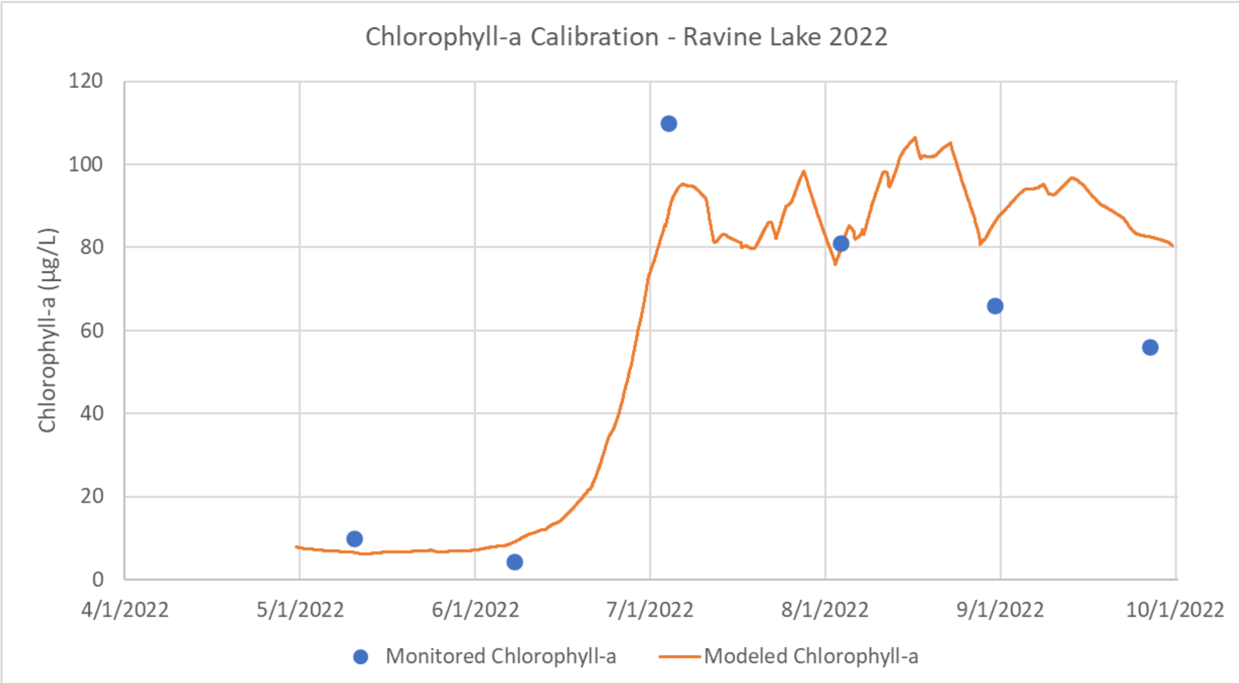

Figure B 69 Ravine Lake chlorophyll-a calibration – 2022

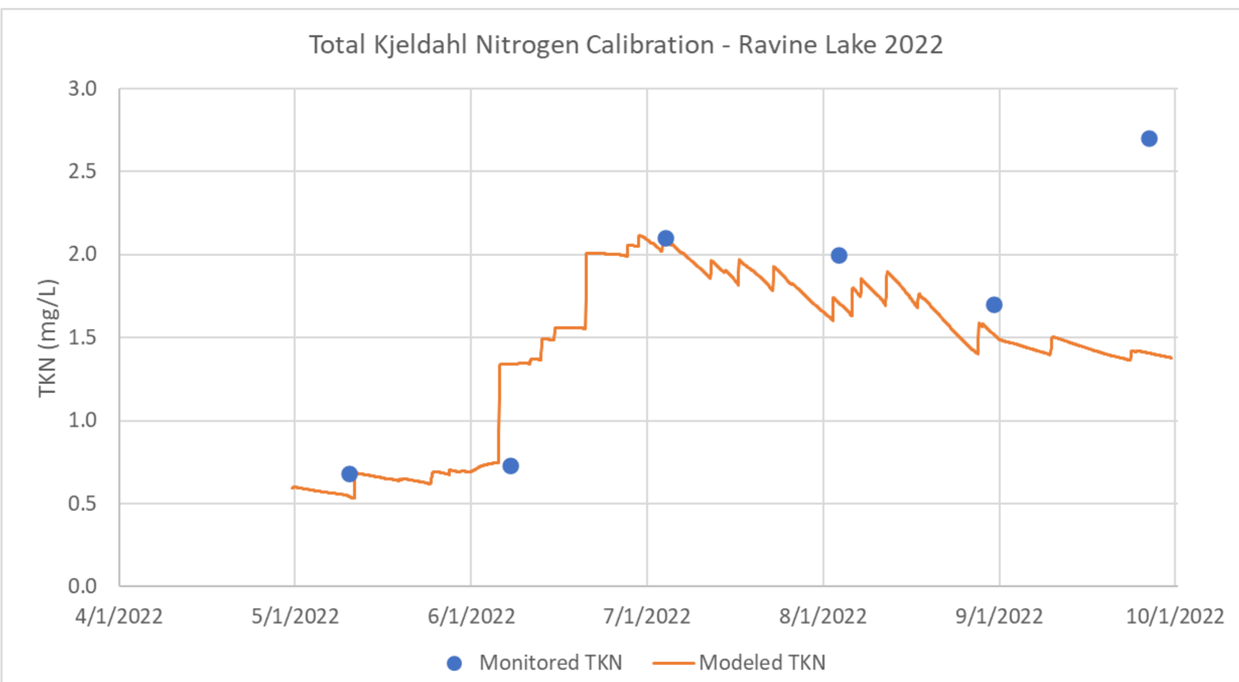

Figure B 70 Ravine Lake total Kjeldahl nitrogen calibration – 2022

Appendix C

Trout Brook and O'Conners Creek WQ Plots

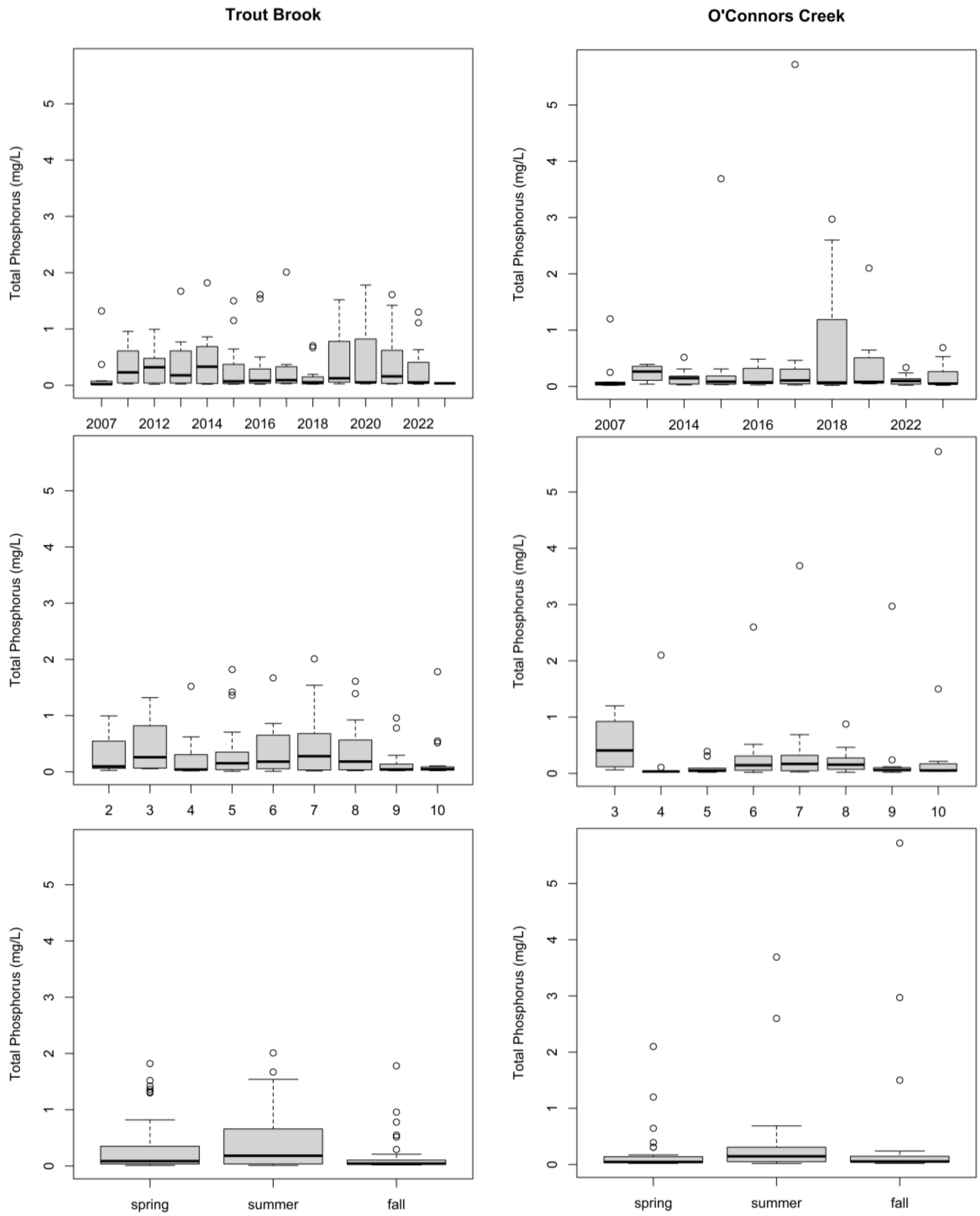

Figure C 1 Yearly, monthly, and seasonal total phosphorus concentrations in the Trout Brook and O'Conners Creek watersheds

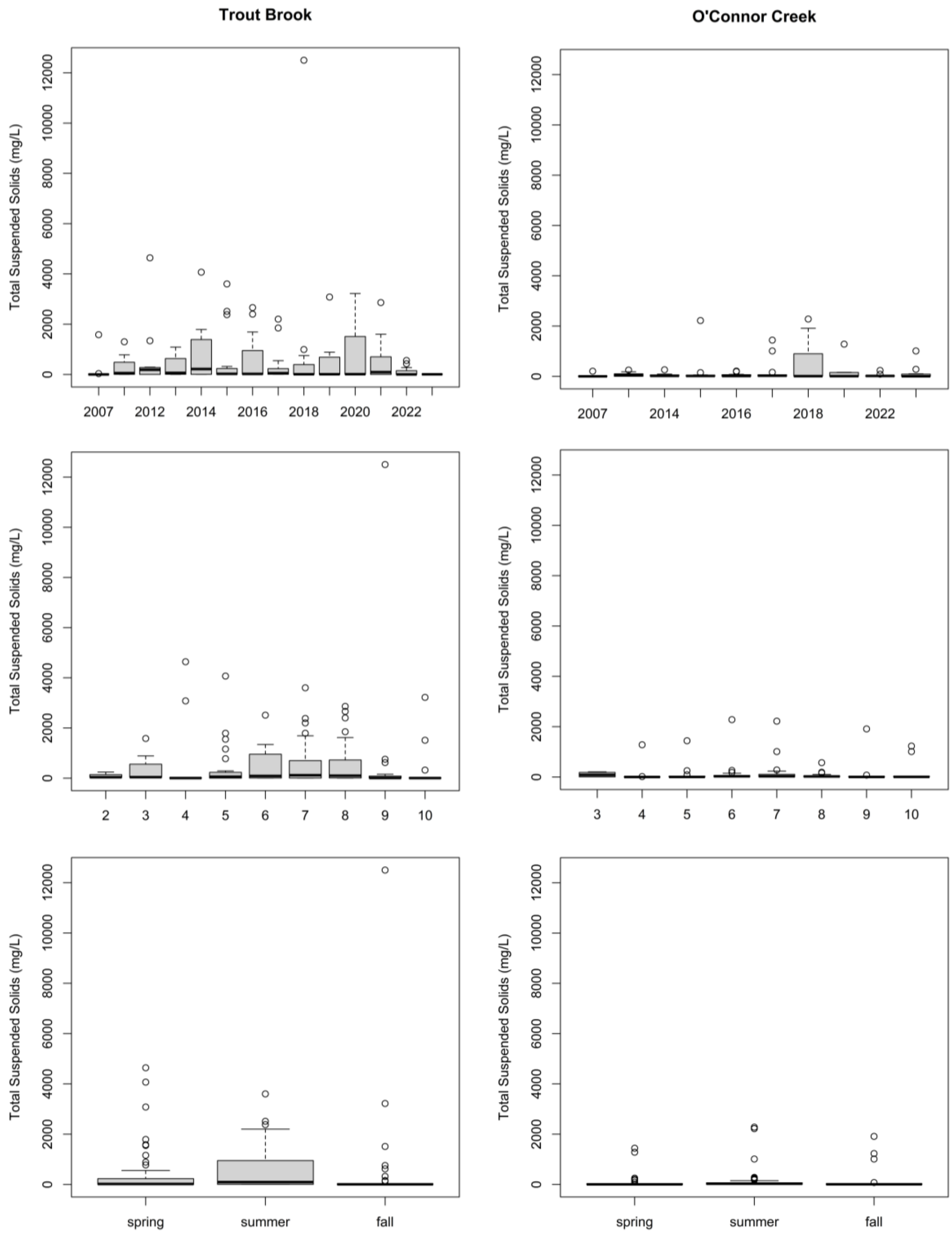

Figure C 2 Yearly, monthly, and seasonal total suspended solids concentrations in the Trout Brook and O'Conners Creek watersheds