DEPARTMENT OF TRANSPORTATION

Wet Pond Modeling for Contaminant Retention and Maintenance

John S. Gulliver, Principal Investigator

St. Anthony Falls Laboratory
University of Minnesota

May 2026

Research Report
Final Report 2026-18

LOCAL ROAD RESEARCH BOARD

To get this document in an alternative format or language, please call 651-366-4720 (711 or 1-800-627-3529 for MN Relay). You can also email your request to ADArequest.dot@state.mn.us. Please make your request at least two weeks before you need the document.

Technical Report Documentation Page

1. Report No.	2.	3. Recipients Accession No.
MN 2026-18		
4. Title and Subtitle Wet Pond Modeling for Contaminant Retention and Maintenance		5. Report Date May 2026 6.
7. Author(s) William Herb, Poornima Natarajan, Benjamin D. Janke, John S. Gulliver, and Jacques C. Finlay		8. Performing Organization Report No.
9. Performing Organization Name and Address St. Anthony Falls Laboratory University of Minnesota 2 Third Ave., Minneapolis, MN 55414		10. Project/Task/Work Unit No. #2024001 11. Contract (C) or Grant (G) No. (c) 1036342 (wo) 81
12. Sponsoring Organization Name and Address Minnesota Department of Transportation Office of Research & Innovation 395 John Ireland Boulevard, MS 330 St. Paul, Minnesota 55155-1899		13. Type of Report and Period Covered Final Report 14. Sponsoring Agency Code
15. Supplementary Notes http://mdl.mndot.gov/		

16. Abstract (Limit: 250 words)

We have found that many ponds are stratified at 0.3m (1ft) depth, often resulting in a bottom region of low dissolved oxygen water and thereby causing the pond to release phosphorus trapped in the bottom sediments back into the water column. It is critical to maintain treatment capacity to reduce pollution to downstream priority waters. This research investigates maintenance approaches to mitigate phosphorus pollution from ponds. The main objectives and tasks of the project are to collect field measurements and perform data analysis to verify a pond model, MinPond, and model pond management methods that can reduce phosphorus export for six ponds. The major conclusions are: an intense street sweeping regime is the most cost-effective means of reducing phosphorus export from a stormwater pond or historic wetland that serves as a stormwater practice by decreasing the load coming from the watershed; chemical treatment of the sediments is another effective means of reducing phosphorus export but only when the phosphorus release from the sediments is substantial; and Bubble aeration that is successful in destratifying the water column can bring oxygen down to the sediments and reduce a high phosphorus release from the sediments.

17. Document Analysis/Descriptors		18. Availability Statement	
Ponding, Phosphorus, Water quality management, Computer models		No restrictions. Document available from: National Technical Information Services, Alexandria, Virginia 22312	
19. Security Class (this report)	20. Security Class (this page)	21. No. of Pages	22. Price
Unclassified	Unclassified	128	

Wet Pond Modeling for Contaminant Retention and Maintenance

Final Report

Prepared by:

William Herb[2]
Poornima Natarajan[2]
Benjamin D. Janke[2,3]
John S. Gulliver[1,2]
Jacques C. Finlay[3,2]

[1]Department of Civil, Environmental, and Geo-Engineering, University of Minnesota
[2]St. Anthony Falls Laboratory, University of Minnesota
[3]Department of Ecology, Evolution, and Behavior, University of Minnesota

May 2026

Published by:

Minnesota Department of Transportation
Office of Research & Innovation
395 John Ireland Boulevard, MS 330
St. Paul, Minnesota 55155-1899

Acknowledgements

This project benefited from the advice and review of the Technical Advisory Panel (TAP), with Ross Bintner as the technical liaison and Jackie Jiran and Chad Kooistra serving as project managers. Other members of the TAP included Joe Bischoff, Jesse Carlson, Rachael Crabb, Noah Czech, Jack Distel, Dan Edgerton, Terry Jeffery, Shahram Missaghi, Randy Neprash, Tiffany Schaufler, Patrick Sejkora, Travis Thiel, Eric Vogel, Tom Wesolowski, and Josh Wolf. Field and laboratory measurements were undertaken with the assistance of Diego Avila, Sam Brandt, Levi Burrows, Nick Findlay, Grace Fritz, Addie Johnson, Owen MacDonald, Mara McCollor, Madigan Rios, and Emma Squires from the St. Anthony Falls Laboratory (SAFL). Data collected by the city of Bloomington staff at the Canterbury Oaks pond were leveraged for this project.

Table of Contents

List of Figures

List of Tables

Definitions

In this report, we address the water quality in stormwater ponds and historic wetlands, and associated indicators of performance for water quality management, which affects the quality of water released from stormwater ponds and historic wetlands to downstream waters. The following are definitions of terms that will be used throughout this document; see Chapter 2 of Janke et al. (2023) for fuller descriptions of these terms and how they are applied to various water bodies used to manage runoff:

Pond: Here, a pond is classified as a water body of less than 10 acres in surface area which has infrastructure to deliver and release stormwater runoff. Such ponds and historic wetlands typically contain standing water for a majority of the time. This report is primarily concerned with constructed stormwater ponds and wetlands that have been converted to receive stormwater runoff (historic wetlands).

Pond or historic wetland with emergent vegetation: A pond as above, with a majority of cover by emergent vegetation, such as cattails and bulrushes. These are typically less than 0.60 m (~2 ft) deep.

Open water pond: A pond or historic wetland with a majority of open water (without emergent vegetation). These are typically between 0.5 and 4 m (1.6 and 13 ft) in depth.

Duckweed: We use this term to describe free floating (i.e. not rooted) aquatic vegetation in the ponds (regardless of pond type) and includes both duckweed (*Lemna* spp.) and water meal (*Wolffia* spp.). Duckweed cover of low, medium, and high correspond to surface coverage of < 25%, 25-50%, and > 50%, respectively.

Age of pond: For constructed ponds, age is based on the construction year; for natural waterbodies (historic wetlands) that were connected by storm sewers, age is based on the time of development around the pond based on historic aerial imagery.

Executive Summary

Background and Objectives

Many of the 16,658 urban stormwater ponds and wetlands treating stormwater, managed as part of MS4 systems in Minnesota (MPCA 2021), and roughly an equal number of commercial ponds in Minnesota are connected to stormwater pipes and act as a part of the stormwater treatment system by storing runoff and settling solids along with associated pollutants to the bottom of the pond. Older, non-maintained ponds may no longer be providing the water-quality benefits of the original design (Taguchi et al. 2018). We have found that many ponds are stratified at 0.3 m (1 ft) depth and have high organic matter levels, often creating a bottom region of low dissolved oxygen water (Taguchi et al. 2020; Janke et al. 2023). The low dissolved oxygen concentration will cause these ponds to re-release phosphorus trapped in the bottom sediments back into the water column (i.e., internal phosphorus loading). It is critical to maintain both stormwater ponds (infrastructure created to treat pollution) and natural water bodies that receive stormwater, which can be managed to reduce pollution to downstream priority waters (a flooded wetland of varying type), and develop methods to improve their functionality, but specifications are minimal. Through modeling efforts and field measurements this research further investigates maintenance approaches to mitigate phosphorus pollution from ponds.

The main project objectives and tasks were to:

1) Model pond management methods that can reduce internal phosphorus loading for at least four additional ponds beyond those in Taguchi et al. (2022), and with additional maintenance procedures, increase the diversity of conditions represented
2) Collect field measurements and perform data analysis to verify the model, including recent novel pond treatments, toward improvement of pond phosphorus retention performance

By understanding the effectiveness of various pond remediation techniques for reducing phosphorus export from ponds, this project will help manage and apply design retrofits to existing and new ponds to improve pond performance. The information will help stormwater managers who are responsible for evaluating and maintaining ponds that treat stormwater. Ultimately, better management will guide allocation and more efficient use of limited resources to address environmental impacts of stormwater throughout Minnesota and beyond.

Overview of Methods and Activities

Major project methods and activities include the following:

1) Development of a 1-D retention pond phosphorus model by modifying the existing MinLake model for lakes (Riley and Stefan 1987) to develop MinPond
2) Field data collection at ponds and laboratory studies on pond sediments to obtain additional or key data required for MinPond verification

3) Application of the MinPond to evaluate pond remediation techniques (chemical sediment treatment, mechanical aeration, smart water level control, wind sheltering reduction, watershed pollutant load reduction, and lining with clay)
4) Assessment of the cost effectiveness of the pond remediation techniques

Outcomes and Benefits

The outcomes of this project can be seen in the following recommendations:

1) A major water quality goal of stormwater ponds and historic wetlands that receive stormwater runoff is to reduce phosphorus exported to receiving water bodies. To achieve this water-quality goal, the manager needs to reduce inputs of phosphorus to the pond/historic wetland, which comes from watershed runoff and can also come from internal loading from the sediments.
2) An intense street-sweeping regime is the most cost-effective means of reducing phosphorus export from a stormwater pond or historic wetland that serves as a stormwater practice by decreasing the load coming from the watershed.
3) If the stormwater pond or historic wetland is stratified, with high concentrations of phosphorus below the surface layer, an outlet designed to avoid transporting the high phosphorus concentrations to the receiving water body is important.
4) Chemical treatment of the sediments is another effective means of reducing phosphorus export only when the phosphorus release from the sediments is substantial.
5) Oxygen addition to the pond or historic wetland via microbubbles will reduce phosphorus release from the sediments if the oxygen addition is sufficient.
6) Bubble aeration that is successful in destratifying the water column can bring oxygen down to the sediments and reduce a high phosphorus release from the sediments. However, if the aerator does not fully destratify the water column, the result could be a higher phosphorus export, as the vertical transport of phosphorus will be enhanced by the aerators.

The research has resulted in four primary benefits to the state of Minnesota:

1) Reduce Road User Costs – Pond maintenance techniques that minimize sediment phosphorus release will lower the need for expensive secondary treatment technologies required to treat downstream waterbodies like lakes that receive the runoff. This will allow municipalities to reduce taxes to road users.
2) Environmental Aspects – More cost-effective and optimal pond maintenance and design will enable improved treatment of runoff for phosphorus control and enable the allocation of limited resources to be applied to other environmental goals.
3) Operation and Maintenance Savings – Understanding the most cost-effective means of assuring reduced phosphorus release from retention pond sediments will clarify and simplify operation and maintenance activities.
4) Reduce Risks – Knowledge of the potential for sediment phosphorus release and the best techniques to alleviate it will reduce the risk to receiving water bodies and help protect Minnesota's aquatic resources.

Chapter 1: Introduction

Ponds are widely implemented stormwater control measures for runoff quantity and quality control in urban areas and are used to remove pollutants such as solids, nutrients, metals and hydrocarbons from runoff through the settling of particles. Arguably, the most important of these pollutants is phosphorus, because it is the limiting nutrient restraining algae blooms, including the growth of toxic cyanobacteria. Pond sediments typically act as sinks for phosphorus; however, low dissolved oxygen (DO) concentration above the sediments can trigger the release of previously buried phosphorus. The lack of DO in the pond may be due to thermal stratification, chemical stratification due to road salt inputs, poor mixing resulting from sheltering from canopy around the pond, or a combination of all three. Low DO conditions are widely known to induce sediment phosphorus release but the presence of anoxic conditions in shallow ponds (Taguchi et al 2018a, 2018b) is surprising and poorly understood since ponds are assumed to frequently mix. The result is increased phosphorus concentrations in the pond outflows that diminish or even negate their intended function.

According to a Minnesota Pollution Control Agency (MPCA) survey of regulated Municipal Separate Storm Sewer Systems (MS4s), there are 16,658 urban stormwater ponds and wetlands treating stormwater managed as part of MS4 systems in Minnesota (MPCA 2021). This number does not include the countless privately owned stormwater ponds associated with individual property developments. Given the large number and popularity of ponds implemented for stormwater treatment in Minnesota, pond re-design and maintenance measures that can improve the phosphorus retention in ponds and limit the impairment of receiving surface water bodies is necessary.

This project investigated maintenance and re-design measures required to reduce or eliminate phosphorus pollution from ponds and wetlands treating stormwater through modeling and data analysis. It is a follow-up to the Local Road Research Board (LRRB)-funded project, *Wet Pond Maintenance for Phosphorus Retention*, completed in June 2022 (Taguchi et al. 2022). In that project, we successfully modeled the stratification, dissolved oxygen, and phosphorus dynamics in four wet ponds. We also simulated the pond response to six remediation techniques and estimated the cost per pound of phosphorus retained for each remediation technique. These results have been used to provide guidelines and recommendations that assist stormwater practitioners in cost-effectively managing phosphorus loading and discharge from the ponds. We also learned that 1) ponds are highly variable, and the stratification dynamics are complex, 2) pond stratification is horizontally stable, such that two- or three-dimensional models are not required to capture seasonal dynamics. The first point means that more than four ponds are required to develop a more comprehensive assessment of pond response to non-routine maintenance activities. The second point means that it will be easier to model the ponds, because higher order models (two- and three-dimensional) are not required. One result of the modeling effort in this project would be to expand the evaluation of Taguchi et al. (2022) to incorporate additional ponds with select features that will assist in the determination of maintenance activities for individual ponds that have high phosphorus concentration through sediment release of phosphorus and/or high phosphorus loading from the watershed.

1.1 Objectives

The goal of this research project was to further investigate maintenance approaches to mitigate phosphorus pollution from ponds and historic wetlands treating stormwater through modeling efforts and field measurements. The objectives of the project were to: 1) model pond maintenance methods that can reduce internal phosphorus loading on at least four additional ponds and historic wetlands beyond those in Taguchi et al. (2022), and with additional maintenance procedures, increase the diversity of conditions represented; 2) collect field measurements and perform data analysis to verify the model, including on recent novel treatments, toward improvement of phosphorus retention performance in ponds and historic wetlands; and 3) develop a Pond Classification System that will accompany the already-developed Pond Assessment Tool (Janke et al. 2023, Natarajan et al. 2025), providing a method for a more comprehensive perspective on pond phosphorus. The primary benefit of the classification system would be to facilitate assessment of a large number of pond sites. Ponds could then be scored and ranked using the Pond Assessment Tool to prioritize further monitoring or maintenance. The resulting Pond Classification System is provided in Appendix A.

The project's sub-objectives were to investigate four additional ponds (designed and constructed in upland areas) or historic wetlands treating stormwater, the cost-effectiveness of the maintenance techniques that proved successful in the prior project, i.e., chemical treatment of sediments (e.g., alum) to impede sediment release of phosphorus, mechanical aeration sufficient to mix the pond and reduce or eliminate stratification, and watershed reduction of phosphorus input. We also investigated reduced sheltering around ponds and historic wetlands to promote wind mixing and reduce or eliminate stratification, the impact of smart water surface control to reduce phosphorus export, and lined versus unlined upstream ponds to determine if one of them is better for phosphorus export control. Field data were collected to fill key knowledge gaps and to inform modeling analyses. This included sampling of ponds and historic wetlands recently treated with alum to assess how well these ponds responded to this increasingly popular approach to lake management to control phosphorus release from pond sediments.

1.2 Outline of Project Approach

MinLake is a lake water-quality model developed over 47 years at the St. Anthony Falls Laboratory. It has been successfully applied to many lake water-quality assessments. Existing versions of MinLake simulate thermal stratification and wind mixing and vertical profiles of dissolved oxygen and nutrient concentrations in the water column. For application to ponds in this project, existing versions of MinLake were modified to develop a one-dimensional (1-D) MinPond to simulate temperature, dissolved oxygen, and phosphorus conditions in the pond and in the outflow. Field measurements (water quality, water level, dissolved oxygen, temperature, and wind speed) taken on the pond studied were used to verify the performance of the model as applied to these ponds.

Major project activities included:

1) Development of a 1-D retention pond phosphorus model by modifying the existing MinLake model for lakes (Riley and Stefan 1987). This model will simplify modeling efforts over the numerous coefficients required to be fit in the 2-D and 3-D models.
2) Field data collection at ponds and laboratory studies on pond sediments to obtain additional or key data required for MinPond verification
3) Application of the 1-D MinPond to evaluate pond remediation techniques (chemical sediment treatment, mechanical aeration, smart water level control, wind sheltering reduction, watershed pollutant load reduction, and lining with clay)
4) Assessment of the cost-effectiveness of the pond remediation techniques

1.3 Benefits

The research employed modeling studies to investigate the benefits of different types of pond management actions so that phosphorus pollution from ponds and historic wetlands to receiving water bodies will be eliminated, thereby allowing environmental benefits of ponds and historic wetlands to be fully realized. The maintenance recommendations will also aid cost-effective operation of ponds and historic wetlands treating stormwater. Thus, the research will result in:

1) **Reduced Road User Costs –** Pond maintenance techniques that minimize sediment phosphorus release will lower the need for expensive secondary treatment technologies required to treat downstream waterbodies like lakes that receive the runoff. This will allow municipalities to reduce stormwater utility fees and taxes to road users.

2) **Environmental Aspects** – More cost-effective and optimal pond maintenance and design will enable improved treatment of runoff for phosphorus control and enable the municipalities to properly budget for stormwater needs and the allocation of limited resources to be applied to other environmental goals.

3) **Operation and Maintenance Savings** – Understanding the most cost-effective means of assuring reduced phosphorus release from retention pond sediments will clarify and simplify operation and maintenance activities.

4) **Reduced Risks** – Knowledge of the potential for sediment phosphorus release and the best techniques to alleviate it will reduce the risk to receiving water bodies and help protect Minnesota's aquatic resources.

Chapter 2: Field Data Collection and Laboratory Studies

This chapter summarizes the data collected from field monitoring at the selected pond and historic wetland sites over two field seasons, i.e., 2023 and 2024. An intensive monitoring approach was undertaken to sample the environmental conditions and water quality in the ponds/historic wetlands from May through October during each field season. Laboratory studies with sediments from select ponds/historic wetlands sites were conducted to determine the potential for sediment phosphate release under changing redox conditions. The lab and field data were evaluated to determine conditions that may result in high phosphorus concentrations in the water columns. The inclusion of pond and historic wetland sites with a history of chemical treatment (alum and iron) allowed an assessment of the effectiveness of chemical treatment on the phosphorus water quality in the ponds/historic wetlands.

2.1 Site Selection and Description

Five pond and historic wetland sites were selected for field data collection over the 2023 and 2024 field seasons (April – October) and were instrumented for continuous monitoring (see Methods Section 3.2). The sampled ponds and historic wetlands represent a range of characteristics affecting phosphorus retention, as identified in previous work (Janke et al. 2021): age, tree cover/wind sheltering, floating plants, water depth, and hydrogeologic setting. Alameda and Shoreview Commons are highly tree-sheltered, which tend to have high (but variable) floating plant cover in all years. Alameda also serves as a control for climate variability, as it has been monitored intensively since 2017. Canterbury Oaks is a small pond with moderately high sheltering and high (but variable) floating plant cover. Camden Central represents low levels of wind sheltering and LP-53 is a larger historic wetland with moderate sheltering, and both pond/historic wetland sites lack floating plant cover. Land use setting for all five ponds/historic wetlands is primarily residential. Shoreview Commons and LP-53 were previously treated with iron filings and alum, respectively, for controlling sediment phosphorus release. The Briarcroft pond, monitored as part of a related project (Finlay et al. 2026) and supplemented with efforts from this project, was added to provide a new, constructed pond with low tree sheltering. The relevant characteristics of the pond and historic wetland sites are given in Table 2-1; with detailed maps shown in Figure 2-1 and Appendix B.

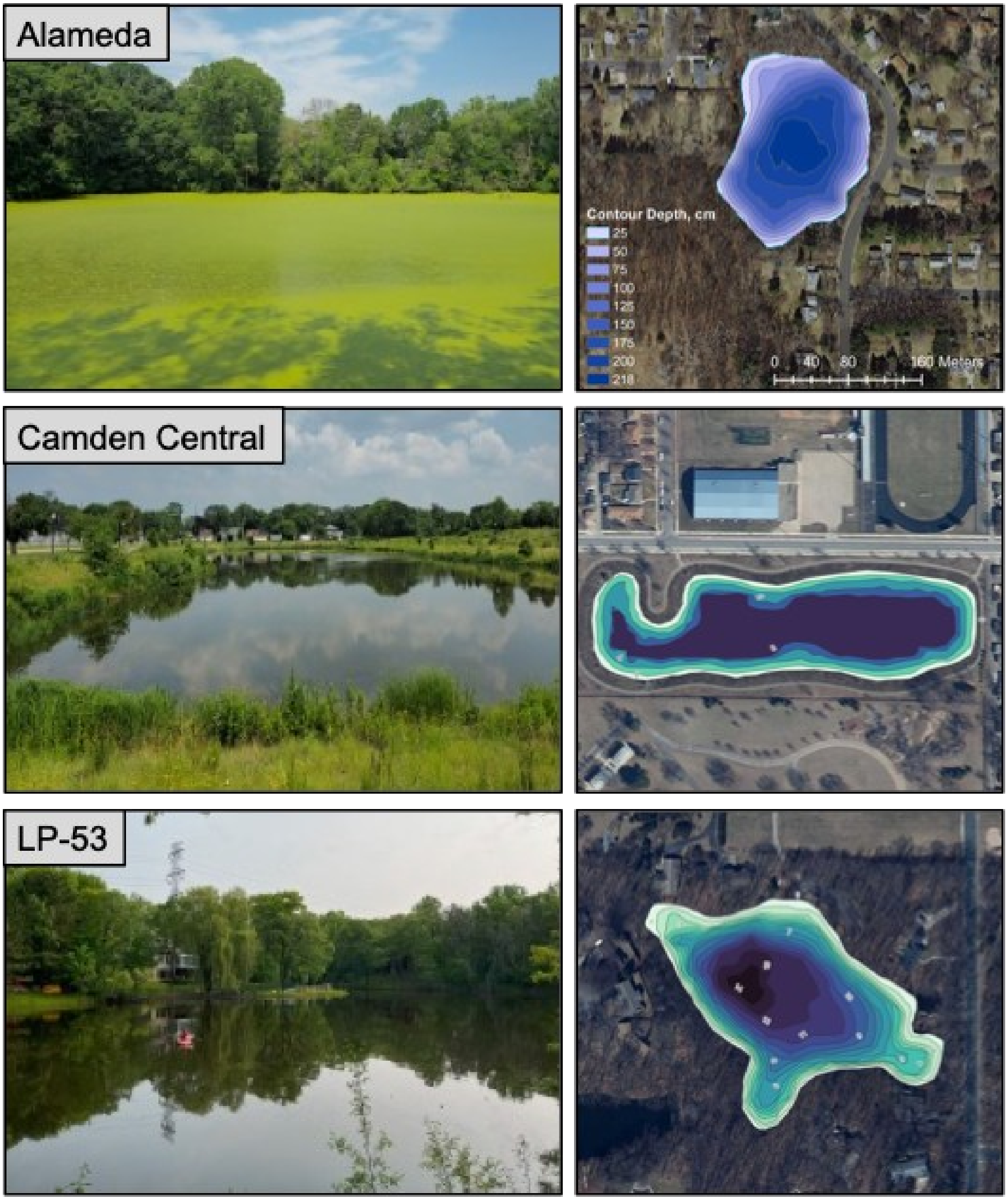
Table 2-1. Characteristics of stormwater ponds and historic wetlands studied as part Field and Laboratory activities in this project. For field monitoring tasks, 'sampling' indicates regular (approximately bi-weekly) collection of water samples and water chemistry profiles, and 'monitoring' indicates installation of continuous monitoring stations for water level, temperature, and wind speed during the May to November period. For sediment P release studies, several pond and historic wetland sites were cored as part of previous work (indicated by years prior to 2023). Briarcroft and Canterbury Oaks were monitored by Finlay et al. (2026)[1], supplemented by efforts from this project.

Pond	Field Season 1 (2023)	Field Season 2 (2024)	Sediment Core Study	Pond Area, ac	Max Depth, m	Drainage Area, ac	Age*, yr	Submerged Vegetation	Floating Plants	Wind Sheltering	Historic Wet-land	Dominant Land Use
Alameda	Sampling+ Monitoring	Sampling+ Monitoring	2017	2.9	2.1	285	74	No	Yes	High	Yes	Residential
Shoreview Commons	Sampling+ Monitoring	Sampling+ Monitoring	2018	2.9	1.2	144	34	Yes (moderate)	Yes	High	Yes	Residential
Camden Central	Sampling+ Monitoring	Sampling+ Monitoring	2021**	4.1	2.0	235	16	Yes (high)	No	Moderate /Low	No	Residential / Institution
Canterbury Oaks	Sampling	Sampling+ Monitoring	2025	0.84	1.2	16	>32*	Yes (moderate)	Yes	Moderate /High	Yes	Residential
LP-53	Sampling	Sampling+ Monitoring	2023	3.9	4.3	46	53	Yes (high)	No	Moderate /High	Yes	Residential
Briarcroft	NA (Sampling + Monitoring in 2025)		NA	1.0	2.75	NA	3	Yes (moderate)	No	Moderate /High	No	Residential

*Ages of pond sites are the year of construction or year connected to the stormwater network, relative to 2023, determined by historic aerial imagery or information provided by the cities.

**Sediment data for Camden Central was obtained from Stantec (2022).

[1] https://wrc.umn.edu/projects/managing-veg


Alameda
Contour Depth, cm
25
50
75
100
125
150
175
200
218
0 40 80 160 Meters
Camden Central
LP-53

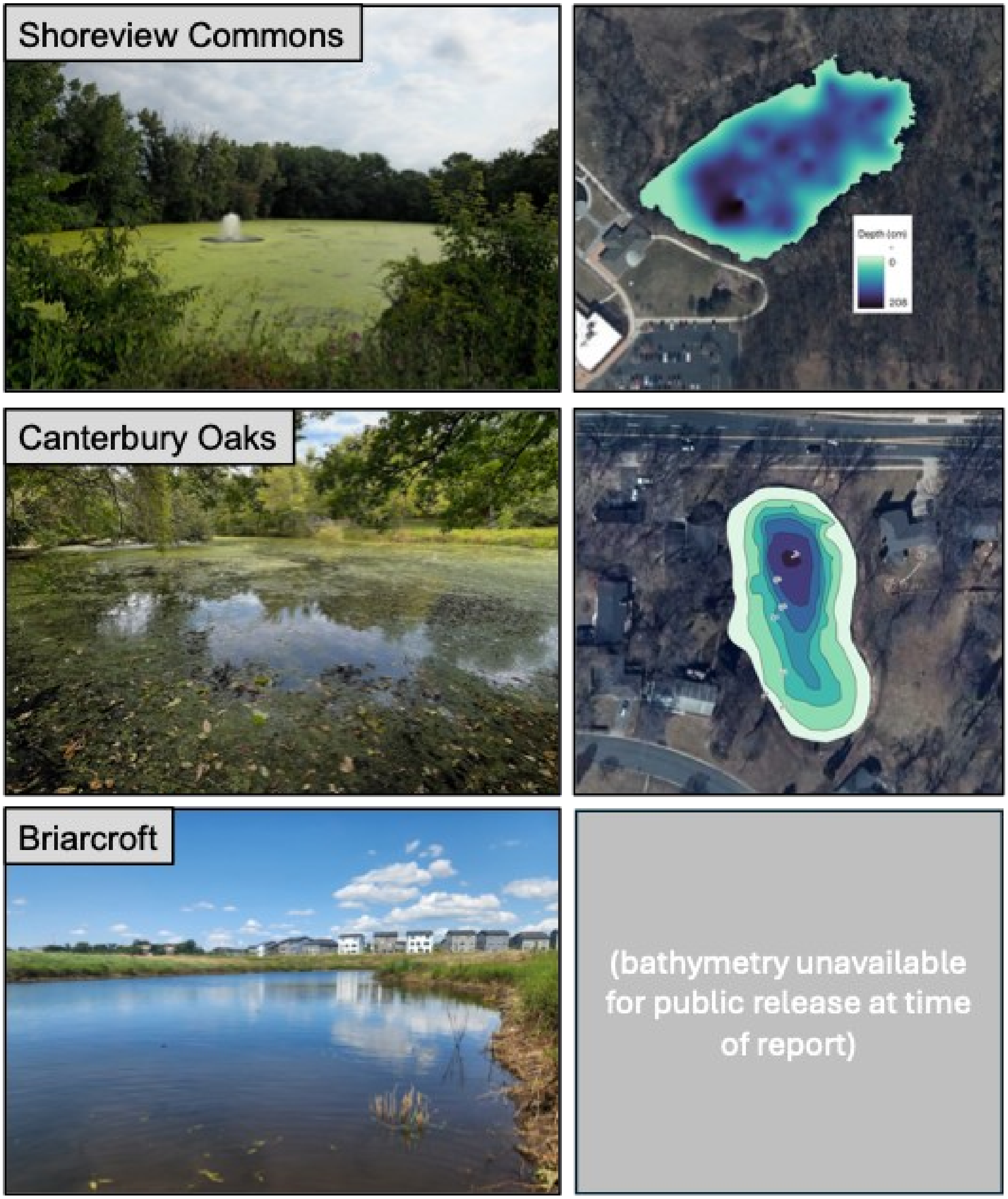


Figure 2-1 Locations of pond monitoring sites and aerial photographs (MN Geospatial Commons). See Appendix B for detailed sampling maps.

2.2 Field Sampling and Monitoring Methods

The goal of field sampling and monitoring efforts in the pond and historic wetland sites was to collect measurements of phosphorus (P; as total, dissolved, and soluble reactive P) as well as of parameters that influence phosphorus cycling in ponds and historic wetlands. These parameters included the following:

- Dissolved oxygen (impacts biological processing of organic matter and can contribute to release of sediment-bound P when levels are low);
- Temperature (impacts biological processing rates, and vertical profiles can identify water column mixing and stratification dynamics);
- Conductivity (affected by dissolved ions, with elevated levels indicative of road salt accumulation and salinity stratification);
- Water level (indicative of pond hydrology and water balance, potential direct mixing of water column by inflows);
- Wind speed (contributes to physical mixing of pond water column);
- Duckweed cover at high levels and coverage (> 50% pond area), can impact stratification, and thus oxygen dynamics and phosphorus levels.

All data collected during field sampling and monitoring activities were intended primarily for use in MinPond development, testing, and calibration (see Chapter 3). Continuous water level and water temperature were used to test model water balance and heat transfer components, with phosphorus concentrations and profiles (DO, temperature, conductivity) used to assess model accuracy. Wind speed measurements were used to help adjust wind coefficients in the model's water surface energy balance.

2.2.1 Sampling Protocol

Five ponds and historic wetlands (Alameda, Shoreview Commons, Camden Central, Canterbury Oaks, and LP-53) were observed and sampled during two field seasons, i.e., from April through October of 2023 and 2024. All ponds/historic wetlands were visited every two weeks for collection of water samples and profiles of water quality (temperature, conductivity, dissolved oxygen) using a Hach multi-parameter water quality meter. A conductivity level above 300 – 400 µS/cm is generally indicative of chloride concentration, because there are no other ion sources in our study systems that typically create a conductivity above this level. A complete water quality profile of dissolved oxygen (DO), temperature, and conductivity was collected near the deepest point in each pond/historic wetland, with measurements taken at 0 cm, 12.5 cm and 25 cm, and then at 25 cm intervals until the maximum depth allowing for a stable measurement. Weather conditions (air temperature, cloud cover, wind speed and direction) were recorded by the field crew at each site visit.

At each pond/historic wetland, five surface (epilimnion) water samples were collected just below the water surface, avoiding as much duckweed as possible, and composited into a single sample in the field. Additionally, a single water sample was collected from near the pond bottom (hypolimnion) at the deepest point, using a Van Dorn sampler. On two sampling dates in 2023, an articulated pole sampler with a Nalgene bottle was used because of a broken Van Dorn sampler. The pole was extended to the greatest depth possible with the bottle in the ‘upside down’ position, before allowing it to flip, releasing air and collecting water at depth. Sampling locations at the ponds and historic wetlands are shown in Appendix B. All water samples were processed, stored, and analyzed at the St. Anthony Falls Laboratory for concentrations of total phosphorus (TP), total dissolved phosphorus (TDP), and soluble reactive phosphorus (ortho-phosphate; SRP) following the APHA (1985) method.

At Canterbury Oaks, profiles and water samples were collected by the SAFL crew as well as the City of Bloomington as part of the Finlay et al. (2026) study[2]. The water samples from the 2023 field season were analyzed at the Tri-city/William Lloyd analytical lab in Bloomington. Briarcroft, also part of the Finlay study, was profiled and sampled by the SAFL crew during the 2025 field season, with samples processed at SAFL.

Field crews also made visual assessment of the extent (fraction of open water) of floating plant cover (duckweed and watermeal) during each site visit at all ponds. Unless floating plants were pushed to a side of the pond due to high winds (which was noted by the field crew), the visual estimates of floating plant cover represent the surface coverage at the time of site visit. Floating plant mass was sampled, when present, on most site visits at 5 to 7 locations in the open water portion of the ponds, using a 28-cm diameter mesh skimmer submerged and carefully pulled upwards out of the water to capture a sample. The plant mass collected was then scraped and rinsed into a plastic bag for storage. Samples were dried at 60 °C and weighed in the lab to obtain biomass. While the floating plant biomass data are not explicitly discussed in this chapter, the biomass data were used in the modeling exercise to simulate the effects of floating plant cover on ponds.

2.2.2 Continuous Monitoring Methods

Four ponds/historic wetlands (Alameda, Shoreview Commons, Camden Central, and Canterbury Oaks; Table 2-1) were monitored continuously during the 2023 and/or 2024 field seasons. At Camden Central (2023, 2024) and Canterbury Oaks (2024), monitoring stations were the same as used in past projects (e.g., Janke et al. 2023), recording water level, wind speed, vertical temperature profile, and DO concentration near the bottom of the ponds. Equipment was mounted to a post driven into the pond sediment near the pond’s deepest point. Equipment consisted of an anemometer (LaCrosse TX-23U) mounted roughly 90 cm above the water surface, light intensity at roughly 60 cm above the pond bottom, an ultrasonic distance gauge for water level measurement, a thermistor chain of 4 to 6 nodes spaced roughly 15 – 46 cm apart in the vertical direction (spacing determined by water depth), and a Luminescent DO probe mounted 30 – 60 cm above the bottom. Data were logged at 10- or 15-minute intervals with an Arduino-type data logger connected to a solar panel. No station was installed at Canterbury Oaks in 2023.

For Shoreview Commons and Alameda, where data have been collected for several years prior to this project, we used simpler setups consisting of discrete Onset Hobo sensors, including temperature

[2] https://wrc.umn.edu/projects/managing-veg

pendants set at 15 – 46 cm intervals on a post, along with a DO probe near the bottom (30 – 60 cm above the sediment) and a specific conductivity probe (Alameda only) at a depth of roughly 15 cm above the bottom to indicate when high chloride concentrations occurred. Pressure sensors were used to record water level at these sites, but no anemometers were installed.

In the 2024 field season, LP-53 was added for continuous monitoring for water level, wind speed, and vertical temperature profile. Due to its greater depth (roughly 450 cm), a buoy was used to float the anemometer, data logger, and the upper portion of a thermistor chain, and a pressure transducer mounted near the anchor of the buoy cable was used to record water depth. Wildlife (likely muskrats) toppled the buoy roughly a week after installation, submerging and destroying the anemometer, data logger, and upper water column thermistors. A replacement buoy consisting of a marine float attached to the original anchor cable was hung with several discrete Onset Hobo pendant temperature loggers to provide temperature profile data, though no further wind speed data collection was possible at the site. The lower water column sensors were unaffected by this modification.

During the 2024 field season, all DO loggers were installed 30 cm above the pond bottom for consistency, except for at the deeper LP-53 historic wetland site, where it was mounted 60 cm above the sediment. No logger was installed at Alameda or Shoreview Commons in 2024, as both sites are persistently anoxic at depth per several years of data collection. For Canterbury Oaks, data collected by the SAFL field crew were leveraged with data collected by the City of Bloomington as part of the Finlay et al. (2026) study[3].

The team similarly leveraged data collection from another pond in the Finlay et al. (2026) study during that project's 2025 field season. This additional site, Briarcroft, was chosen due to being a newer construction with less wind sheltering and no duckweed, which contrasted with the other pond and historic wetland sites used for model development and testing. Briarcroft had a continuous monitoring station identical to those used in the other sites, and the SAFL crew provided sampling and data collection support.

2.3 Data Collection Summary

Data collection efforts for the two field seasons are summarized in Table 2-2. In the 2023 field season, the pond and historic wetland sites were routinely sampled from May through October, with a total of 10 to 12 site visits per pond. Camden Central was first sampled in June 2023 due to a shift in the ponds to be sampled. In 2024, each site was sampled 12 to 13 times by the project team between April and October. Canterbury Oaks was sampled in early April of both years by the city of Bloomington. Briarcroft was sampled between late July and November 2025.

Continuous monitoring equipment was installed at four pond and historic wetland sites (Alameda, Shoreview Commons, Camden Central) in June or July 2023, and removed in early November 2023. For the 2024 field season, monitoring stations were installed at five sites to record data from June or July to

[3]https://wrc.umn.edu/projects/managing-veg

mid-November. Some issues were encountered with the stations at most pond/historic wetland sites in either year, and these issues are summarized in more detail in the next section. Of note, at LP-53, the buoy was over-turned by wildlife, likely muskrats, one week after installation (around 6/29/25) resulting in permanent damage to the datalogger, anemometer, and solar panels. A provisional station consisting of discrete temperature loggers was installed on July 25, with the lower sensors on the original buoy (DO, water level, light, temperature) remaining unchanged. However, the result of this modification was a complete loss of wind data for the season at this site, as there were no spare anemometers and we were unable to construct a more durable buoy within a reasonable time.

Due to difficulties with monitoring and resulting loss of data, Canterbury and Briarcroft were added from a concurrent project (Finlay et al. 2026) to supplement data needs for model development and testing.

Table 2-2. Summary of field data collection at the five pond and historic wetland sites during 2023 and 2024 field seasons. For Canterbury Oaks, data were collected by the SAFL field crew and the City of Bloomington as part of Finlay et al. (2026) study. For LP-53, station was initially installed 6/14/2024 and subsequently damaged by wildlife; the replacement equipment was installed 7/25/2024. *For Turnover, general pattern is given (intermittent or frequent), but if the pond/historic wetland was stratified through most of season then date of fall turnover is given. **Canterbury Oaks had a manual duckweed removal in July of 2023, which reduced duckweed cover briefly before rapidly regrowing (see Finlay et al. 2026 study).

Pond	Site Visits	Site Visits		Monitoring Station		Turnover*	Duckweed Cover
		First	Last	Start	End		
2023 Field Monitoring							
Alameda	12	5/16/2023	10/19/2023	7/21/2023	11/1/2023	10/18/2023	High
Shoreview Commons	12	5/16/2023	10/19/2023	7/21/2023	11/1/2023	Intermittent; 8/9/2023	High
Camden Central	10	6/16/2023	10/19/2023	6/16/2023	11/1/2023	Intermittent	None
Canterbury Oaks	11	4/11/2023	11/7/2023	NA	NA	Intermittent	High**
LP-53	11	5/22/2023	10/31/2023	NA	NA	Frequent	None
2024 Field Monitoring							
Alameda	13	4/25/2024	10/14/2024	6/1/2024	11/9/2024	10/14/2024	High
Shoreview Commons	13	4/30/2024	10/25/2024	7/25/2024	11/5/2024	Intermittent; 9/24/2024	High
Camden Central	12	5/10/2024	10/8/2024	8/9/2024	11/17/2024	Intermittent	None
Canterbury Oaks	12	4/8/2024	9/21/2024	6/14/2024	11/22/2024	Intermittent	High
LP-53	13	5/1/2024	10/15/2024	7/25/2024	11/10/2024	10/15/2024	None
2025 Field Monitoring (Finlay et al. 2026)							
Briarcroft	6	7/25/2025	11/6/2025	7/25/2025	11/6/2025	Frequent	None

2.4 Data Quality and Data Processing

Climate: The average air temperature recorded during summer 2023 (May to October) was 20.1 °C, the highest on record for this period (vs. the 1991-2020 mean of 18.1 ˚C) at the Minneapolis-St. Paul International Airport. The average air temperature recorded during the same period in 2024 was 19.8 °C. The total precipitation depths recorded during the May to October period of 2023 was 45 cm, which was greater than the previous two years, but still remained almost 13 cm below the 1991-2020 average (66 cm vs. 57 cm) and more than half (24 cm) of the total precipitation for this period fell between Sept. 23 and Oct. 31, 2023. The May-October 2024 period recorded 66 cm precipitation, which was more than 8 cm rainfall depth above the 1991-2020 average. All five pond and historic wetland sites were of adequate depth to remain accessible and to allow for satisfactory sample collection in both field seasons, including the driest months in 2023. Antecedent winter conditions were a more exceptional contrast between field seasons. Winter 2022-2023 snowfall total was 2.3 m at the Minneapolis-St. Paul Airport with a mean air temperature of -2 ˚C from November – April, while Winter 2023-24 was the warmest and among the driest winters on record, with a snowfall total of 0.88 m (0.39 m of which fell during March 2024) and a mean November – April air temperature of 1.4 ˚C[4].

Water Depth: For several sites, a pressure sensor mounted above ground near the pond shore recorded the atmospheric pressure, which was used to convert pressure monitored by the water level sensors into hydrostatic pressure. This pressure was then converted to depth using the specific weight of water as a function of mean water column temperature.

Wind Speed: Wind speed was corrected to a height of 10 m using an assumption of a logarithmic vertical wind profile, and assuming an aerodynamic roughness length of 0.01 for a water surface.

Relative Thermal Resistance to Mixing (RTRM): RTRM is a measure of water column stratification strength, calculated as the density difference between two points *i* and *j* in a water column, relative to the density difference between water at 4˚C and at 5˚C (which is considered unstable):

$$RTRM = \frac{\rho_i - \rho_j}{\rho_{4^\circ C} - \rho_{5^\circ C}} \quad (2.1)$$

We make this calculation between the top and bottom temperature nodes in the pond water column using the continuous temperature data, to assess water column stability and illustrate sub-daily mixing, which the model is able to simulate.

Data gaps: Difficulties were encountered in monitoring several sites during the two field seasons. Critical gaps are summarized below:

1. **LP-53 (6/14/24 – 7/25/24, all data):** Wildlife toppled the initial buoy at LP-53 in 2024 and a replacement wasn't installed until July 25, so the useful data record at that site start from July 25 onward in 2024. **No wind data were collected or used from this historic wetland site**.

[4] Retrieved from https://mrcc.purdue.edu/CLIMATE

2. **Camden Central (7/3/24 – 8/9/24, all data):** A memory card failure at Camden Central in 2024 prevented data collection until after August 9.
3. **Briarcroft (8/23/25 – 11/6/25, wind data):** The anemometer malfunctioned after the first data download in August 2025, so a gap in wind speed data exists from August 23 onward at that site.
4. **Alameda (6/1/24 – 11/9/24, temperature data):** several temperature sensors failed so temperature data are unavailable for this site in 2024.

2.5 Field Monitoring Results

Field data were collected primarily for providing input to model development and testing tasks. While collected data have been integrated with previous and current observations in related projects by the project team (e.g., Finlay et al. 2026) to provide more comprehensive understanding of phosphorus cycling in ponds, we leave presentation of those results to later project reports. All field data, including continuous data collected at the stations, vertical profiles from site visits (DO, temperature and specific conductivity), floating plant cover observations, and concentrations of phosphorus in water samples, are provided for each of the six sites in Appendix B. Selected data are presented here within the main report to draw a contrast between a more open pond with no floating plants (Camden Central) and more heavily-sheltered, duckweed-covered sites (Alameda and Canterbury Oaks). Some data included for Canterbury Oaks and Briarcroft were partially collected by the project team under a related project (Finlay et al. 2026).

2.5.1 Stratification and Dissolved Oxygen Dynamics

Contour plots (spatially- and temporally-interpolated vertical profiles) of the temperature, conductivity, and DO data collected during site visits at Alameda and Camden Central during the 2023 and 2024 field seasons are shown below. Time series of monitoring data from the field seasons are presented for Camden Central and Canterbury Oaks in the following section. Data for all pond and historic wetland sites are in Appendix B.

To illustrate contrasting stratification dynamics in the ponds and historic wetlands, consider the contour plots for Camden Central and Alameda (Figure 2-2and Figure 2-3). These ponds are similar in size and depth, but vary considerably in terms of setting: Alameda is round, heavily sheltered by trees and topography, and persistently covered by floating plants, while Camden is long and narrow in shape and lacks sheltering. Due to high sheltering and dense floating plant cover, much of Alameda's water column is temperature stratified and anoxic throughout the season, with generally lower surface temperatures than Camden Central, which shows stratification but generally less anoxia. Alameda's lower surface temperatures may indicate a cooling effect of the dense floating plant cover.

Both Alameda and Camden Central showed high conductivity near the start of the 2023 field season, which contributed to early season stratification. This conductivity was likely caused by high road deicer application during the previous winter, in which near-record snowfall totals were observed in the Twin Cities. The following winter was among the warmest and driest on record, and consequently, conductivity is low in the ponds from the start of the 2024 field season onward. This substantial contrast in conductivity between the two field seasons is observed in the other pond and historic wetland sites as well (Appendix B), especially for the deeper LP-53 historic wetland. This suggests an important role for antecedent winter conditions and salt application on setting up of spring stratification in the deeper (>~2 m) sites.

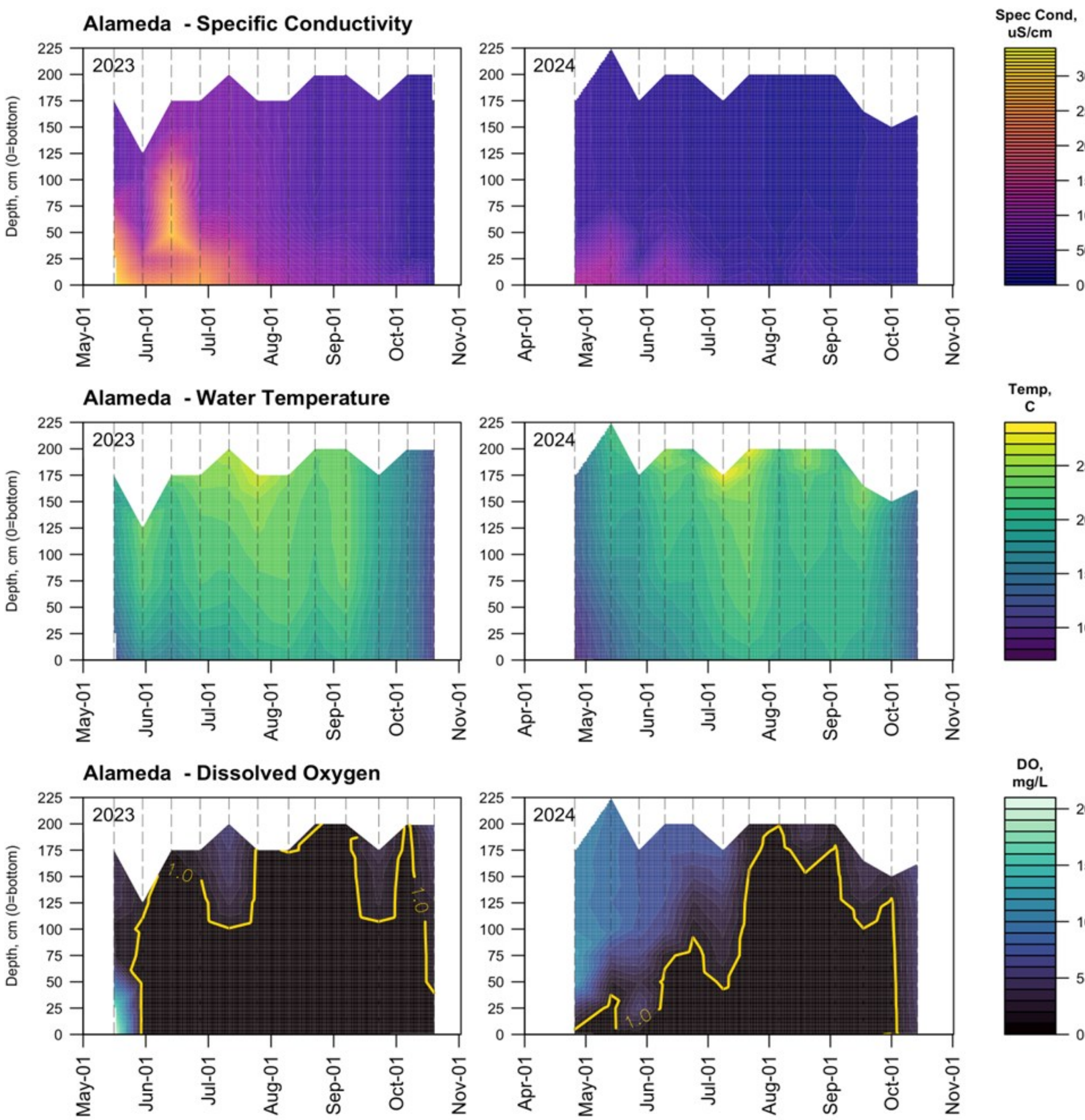


Figure 2-2. Specific conductivity (top row), temperature (middle row), and dissolved oxygen (bottom row) contour plots for Alameda: Color indicates values per the scale at right, with water depth relative to the pond bottom on the y-axis and time along the x-axis (left column is 2023 and right column is 2024). Vertical dashed lines are dates of site visits when profiles were collected; linear interpolation used to fill in the gaps between profile dates. On the dissolved oxygen plot, a contour for 1.0 mg/L indicates levels below which the pond is considered anoxic.

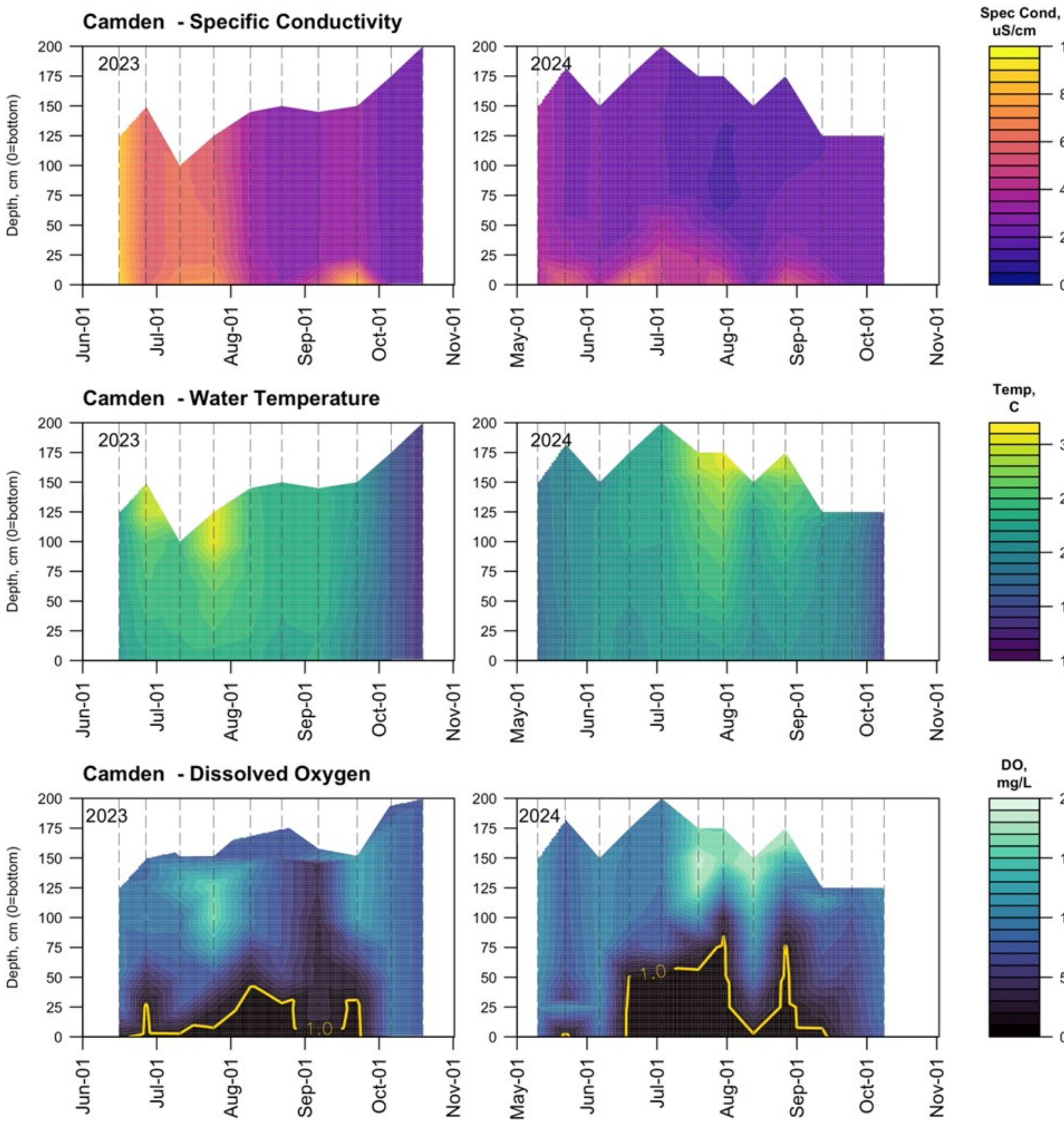


Figure 2-3. Specific conductivity (top row), temperature (middle row), and dissolved oxygen (bottom row) contour plots for Camden Central: Color indicates values per the scale at right, with water depth relative to the pond bottom on the y-axis and time along the x-axis (left column is 2023 and right column is 2024). Vertical dashed lines are dates of site visits when profiles were collected; linear interpolation used to fill in the gaps between profile dates. On the dissolved oxygen plot, a contour for 1.0 mg/L indicates levels below which the pond is considered anoxic.

The continuous time series of data collected (water depth, wind speed, temperature at multiple depths, and near-bottom DO) provides insight on daily dynamics of stratification and mixing, which are not possible to capture with single profiles collected every 2-3 weeks during mid-day. To simplify presentation of the data, quantities have been condensed into daily mean values for water level and wind speed, with DO summarized as daily mean, minimum, and maximum. Temperature data have been condensed into calculations of top-vs-bottom relative thermal resistance to mixing (RTRM), summarized as daily mean, minimum, and maximum. Values of RTRM near 0 (< 12.5 per Richardson et al. in review) can be interpreted as an unstable, mixed (or mixing) water column, while higher values indicate a stratified, stable water column. We present results for Camden Central and Canterbury Oaks here for illustration, while results for all ponds and historic wetlands are shown in Appendix B. Note that the data collected by the continuous monitoring stations have a shorter record than the data collected on sampling trips to the sites due to required set-up time.

Canterbury Oaks shows daily dynamics expected of a smaller, sheltered, duckweed-covered pond in 2024 (Figure 2-4): daily mean wind speeds are generally low (< 1.0 m/s) until fall, when leaves drop and wind speeds can be higher. Water column stability (RTRM) is variable but high throughout the summer, before turning over in late September (and potentially a couple times prior to that in mid-August and early September). Accordingly, the pond is persistently anoxic near the bottom (30 cm) until the pond becomes roughly isothermal in October (Figure 2-5). Isolated mixing events likely due to runoff inputs can be observed from decreases in RTRM coincident with increases in water level, though none of these events are sufficient to mix the pond top-to-bottom. Last, the pond also shows the substantial loss in water level due to drought in late summer 2024, when almost no precipitation was observed between late August and late October.

By contrast, Camden Central shows patterns consistent with a more wind-exposed site that lacks duckweed cover (Figure 2-4). Wind speeds are generally higher and much more variable throughout the season than at Canterbury Oaks in 2024. Stratification is still present, as suggested by the profile data (Figure 2-2) and illustrated by the high daily variability in RTRM (Figure 2-4). However, the Camden Central pond generally mixes (RTRM ≤ 0) at various times throughout the season, likely due to wind, runoff inputs (water level increases), and surface heat transfer. The persistent negative RTRM values observed in early October indicate a highly unstable water column due to rapid cooling as the pond approaches isothermal conditions in early November. Interestingly, DO near the pond bottom remains low throughout much of the season until fall turnover, except for a few events in August, likely driven by runoff events (Figure 2-5). This contrasts slightly with the 2023 patterns at the site (Appendix B), which show more frequent mixing events and variable but more consistently oxic conditions (> 2 mg/L) near the pond bottom. The long drawdown of water level from late August to late October in 2024 due to drought is also present at Camden Central, with the inflows observed in mid-September likely caused by water that is pumped into the pond from an upstream lake. These pumping events could largely be interpreted from water level changes during dry periods.

Overall, these patterns illustrate the impact on pond oxygen dynamics of several interacting factors, including wind access (affected by tree sheltering and perhaps floating plants), direct mixing by stormwater inflows, potential groundwater inflows and shading of the water column by duckweed. For this latter factor, low DO conditions in the pond would be expected to result from the inhibited photosynthesis in the water column during the day, though we acknowledge a lack of information on diurnal patterns of DO within the entire water column (Janke et al. 2023).

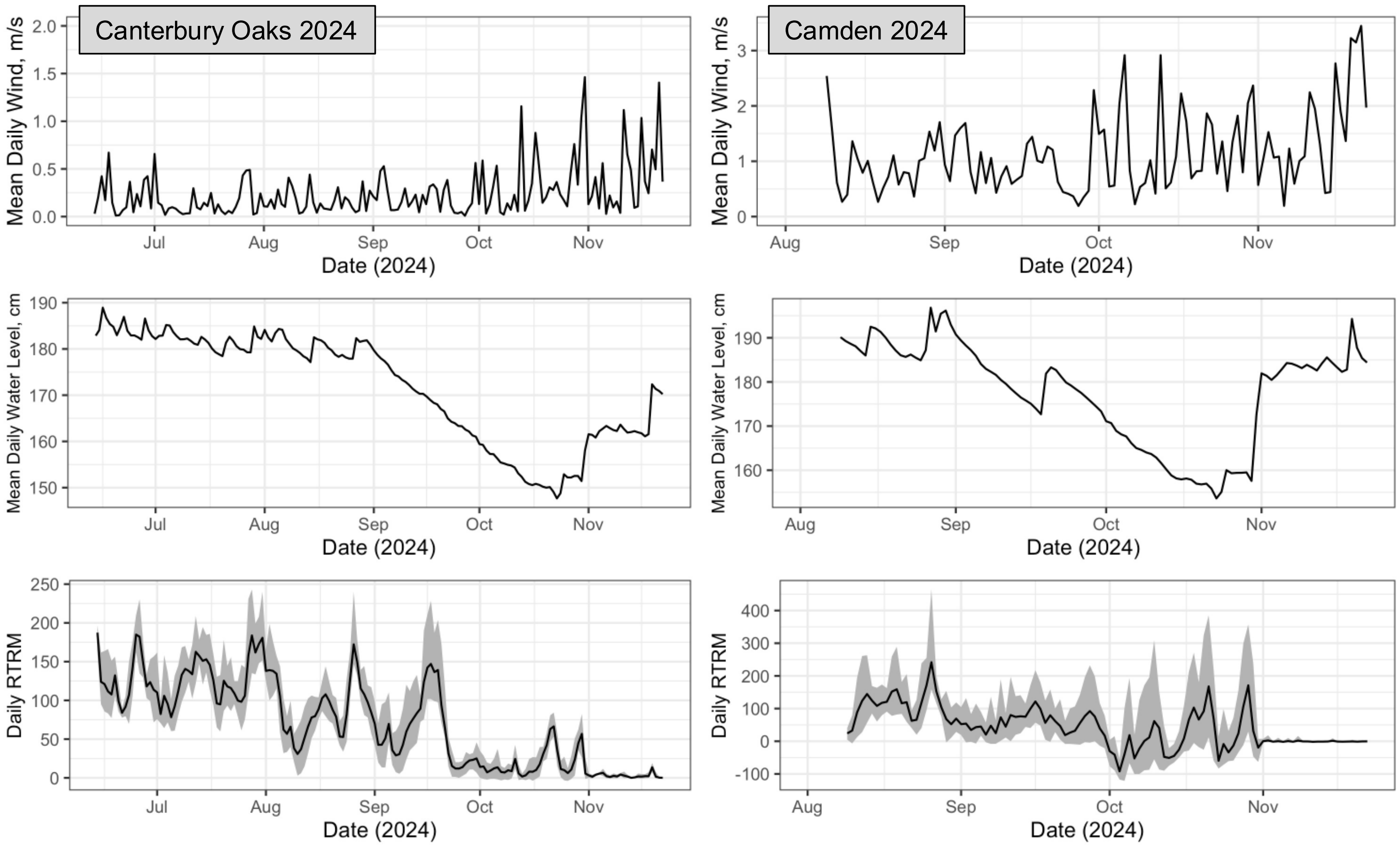
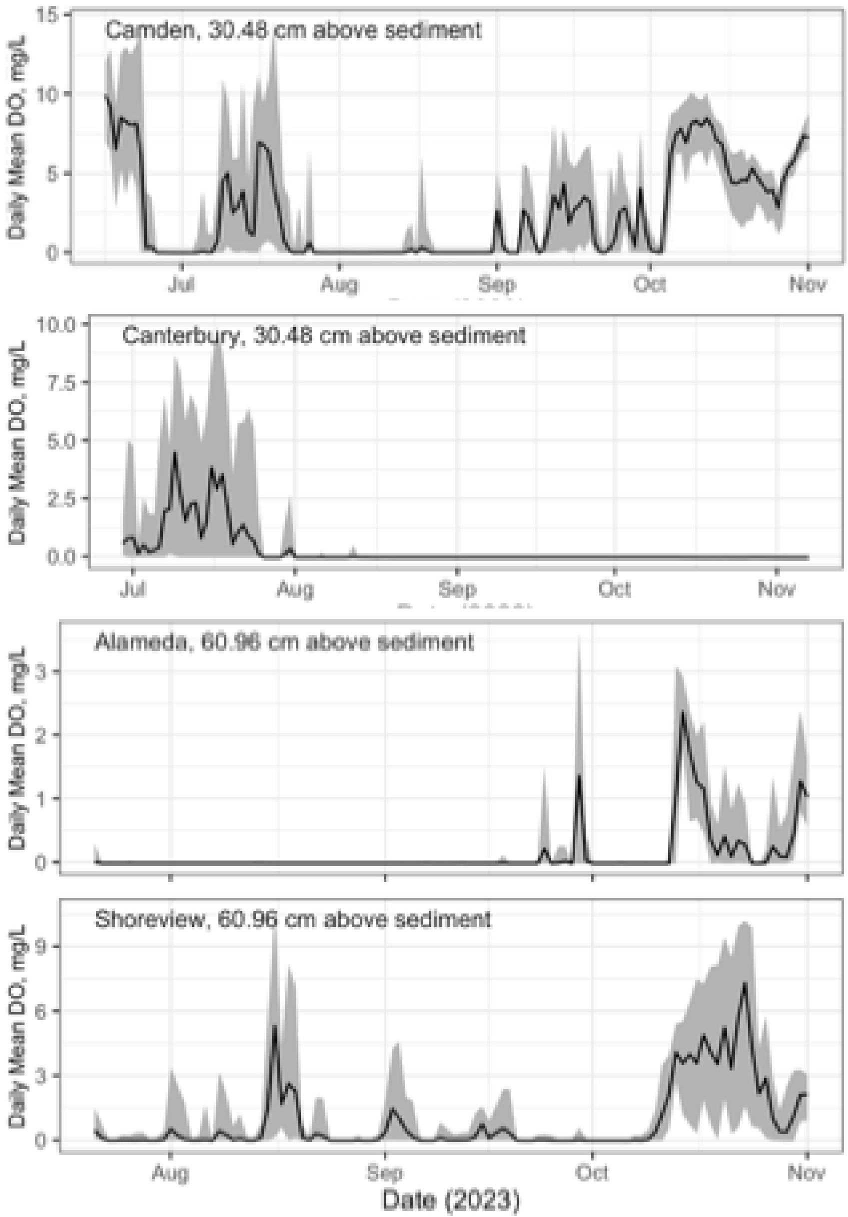


Figure 2-4. Mean daily wind speed (m/s) (top row), mean daily water level (cm) (middle row), and daily relative thermal resistance to mixing (RTRM) as mean (black line) minimum and maximum (dark gray envelope), observed at Canterbury Oaks (left column) and Camden Central (right column) during the 2024 field season.



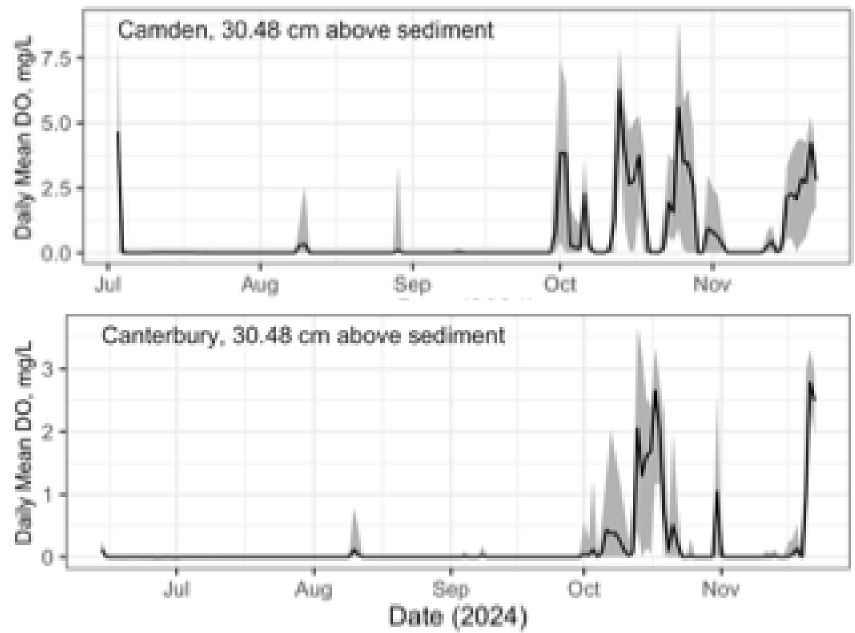


Figure 2-5. Daily mean, minimum, and maximum dissolved oxygen concentration (mg/L) observed near the bottom of several of the monitored pond and historic wetland sites. Daily minimum and maximum illustrated by gray envelope around black line (mean DO). Left column shows DO observed during the 2023 monitoring season, while the right column (two sites) shows DO observed during the 2024 monitoring season. Note that no DO values above zero were observed at LP-53 (depth ~ 100 cm above sediment) during the entire season (July – November 2024), so no plot is included here.

2.5.2 Phosphorus Water Quality

The time series for phosphorus concentrations in the pond and historic wetland sites are shown in Appendix B. The time series for Shoreview Commons and LP-53 are presented as a special case of post-treatment water quality, as phosphorus-binding chemicals (iron filings and alum) were applied to these historic wetland sites.

Phosphorus concentrations measured at the pond and historic wetland sites during the 2023 and 2024 field seasons (Table 2-3) followed the trends of temperature and mixing. Alameda had the highest surface and bottom total phosphorus (TP) concentrations, followed by Canterbury, Camden Central, Shoreview Commons, and LP-53. Except for Alameda and Canterbury Oaks, all other ponds/historic wetlands had low soluble reactive phosphorus (SRP) concentrations in the surface and hypolimnion during the monitoring period. This is often observed, because algae and floating plants quickly convert SRP to organic, particulate phosphorus. In Alameda and Canterbury Oaks with heavy floating plant cover and low DO, the high hypolimnion concentrations of TP and SRP could be due to anoxic sediment phosphorus release throughout the season. The pervasiveness of anoxia in these historic wetland sites is reflected by the anoxic factor (Table 2-3), which is the fraction of a given period that the DO near the sediment is below 1 mg/l. For example, in Alameda, about 89% sediment area was anoxic during June to August 2023 compared to 69% area during the same period in 2024, while 68% of the Canterbury Oaks sediment was exposed to anoxic water during 2024. The site had a physical duckweed removal in July 2023 that likely lowered the anoxic factor in that year. The TP concentrations in Camden Central showed a surface TP that continued to increase after August until concentrations approached the bottom TP at the end of the season (plots shown in Appendix B). This can be explained by the mixing and stratification cycle observed at Camden Central starting in mid-August.

LP-53 with alum-treated sediments and no floating plant cover had the lowest TP in the entire water column. All of the ponds/historic wetlands except LP-53 had relatively high surface TP and SRP concentrations, including Shoreview Commons with iron-treated sediments. Canterbury Oaks had the highest surface TP at 353 µg/L in 2023 and 270 µg/L in 2024. The bottom water TP was higher than the surface in all ponds/historic wetlands, which could either be due to the timing of phosphorus inputs, the settling of particulate phosphorus or sediment phosphate release. It appears that the herbicide treatments in the Camden Central pond, while decreasing duckweed cover, did not do much to change total phosphorus concentrations in either the surface or bottom portions of the pond. Duckweed cover in Camden Central was replaced by high algae (phytoplankton) concentrations. The Camden Central Pond, however, is the first herbicide-treated pond that we have investigated in detail.

The Briarcroft pond was chosen because it was newer and less sheltered than the other sites, with no floating plant cover and some submerged plants present. In the 2025 field season, which was initiated later in the summer at this site, the July-August mean values of TP, SRP, and TDP were 0.076 mg/L, 0.014 mg/L, and 0.029 mg/L, respectively (plots shown in B). Anoxic factor was not calculated but would be close to 0, as the water column was well-oxygenated over the summer and fall (July – November), with mean water column DO of >8 mg/L over the 2025 field season.

Table 2-3. Summary of water quality at Alameda, Shoreview Commons, Camden Central, Canterbury Oaks, and LP-53 during the 2023 and the 2024 field seasons. Mean values of anoxic factor*, floating plant coverage, and phosphorus concentrations for the monitored period (May to October) are provided along with standard deviation of the mean value for each pond.

Parameter	Mean for 2023 Field Season					Mean for 2024 Field Season				
	Alameda	Shoreview Commons	Camden Central	Canterbury Oaks	LP-53	Alameda	Shoreview Commons	Camden Central	Canterbury Oaks	LP-53
Season Anoxic factor (May to Oct)	0.85	0.44	0.50	0.35	0.087	0.56	0.51	0.45	0.62	0.38
Summer Anoxic factor (Jun to Aug)	0.89	0.46	0.63	0.20	0.11	0.62	0.64	0.54	0.68	0.47
Floating plant cover (%)	83 ± 30	65 ± 35	0	40 ± 28	0	52 ± 39	64 ± 33	0	58 ± 27	0
Surface TP (µg/L)	130 ± 40	151 ± 94	154 ± 35	353 ± 163	36 ± 11	165 ± 57	165 ± 80	150 ± 60	270 ± 82	52 ± 25
Bottom TP (µg/L)	602 ± 384	170 ± 110	175 ± 55	n/a	117 ± 55	286 ± 134	218 ± 124	209 ± 51	426 ± 215	100 ± 24
Surface SRP (µg/L)	41 ± 35	19 ± 13	12 ± 8.1	289 ± 197	8.2 ± 3.6	46 ± 25	50 ± 32	8.4 ± 4.1	110 ± 78	7.3 ± 3.2
Bottom SRP (µg/L)	115 ± 99	18 ± 13	11 ± 5.6	n/a	9.7 ± 3.6	124 ± 97	57 ± 54	9.2 ± 4.1	178 ± 111	7.6 ± 3.4

*Anoxic factor = fraction of pond area with bottom DO < 2 mg/L during the monitored period

2.6 Laboratory Phosphorus Release Study Results

The purpose of the laboratory experiments was to determine the sediment phosphorus flux under oxic and anoxic conditions, and to analyze the sediment phosphorus chemistry. The sediment phosphorus release study was conducted on four historic wetland sites during this study: Alameda, Shoreview Commons, LP-53, and Canterbury Oaks. Shoreview Commons was treated with iron filings in 2021 and LP-53 was treated with alum in 2019. Alameda and Shoreview Commons (pre-iron treatment) were sampled in 2017 and 2018 in our previous studies (Taguchi et al. 2020; Janke et al. 2021), following the same protocol as this project (Section 2.6.1). Sediment phosphorus release rate for Camden Central was extracted from the study by Stantec (2022) on 17 ponds in the City of Minneapolis. We have included data on the previously-studied ponds /historic wetlands and Camden Central to support the discussion of the results for the six pond and historic wetland sites studied in this project.

2.6.1 Methods

At each pond/historic wetland site, intact sediment cores with overlying pond water were collected from five locations distributed in the pond. While LP-53 and Canterbury Oaks were cored in June 2023 and April 2025, respectively, other sites were cored as part of previous studies, as indicated in Table 2-1.

The phosphorus (P) released from the sediments is typically in the form of phosphate (or SRP). The experiments were conducted in five-5 cm diameter columns of water and sediment taken from the ponds or historic wetlands. The water was passed through a 1 micron filter to remove most sediment, and then the column experiments were undertaken at ~20 °C and consisted of three phases. First, the water columns over the sediments were kept oxic by aeration (dissolved oxygen or DO > 9 mg/L). Then, air bubbling was turned off and the DO in the unmixed water column was allowed to decrease due to the sediment oxygen demand (SOD). In the third phase, the water column was mixed by bubbling ultrapure nitrogen gas to maintain anoxic conditions (DO < 1 mg/L). The water columns were periodically sampled throughout the experiment to determine the concentrations of orthophosphate (soluble reactive phosphorus; Standard Methods 4500-P, APHA 1995). One water sample was drawn from the approximate center of the mixed water columns during the air bubbling and nitrogen bubbling phases. During the air off phase, sampling was done at multiple points distributed across the water column depth to account for the concentration gradient that can develop under an unmixed state. The average phosphorus concentration in the entire water column was estimated assuming an exponential distribution with height.

The sediment phosphorus flux (mg/m^2/day) was calculated as the linear change in phosphorus mass in the overlying water (where, mass = concentration × water column volume; mg) divided by the phase duration (days) and the sediment area (same as the column area, m^2).

2.6.2 Results

The phosphorus release study results for Alameda, Shoreview Commons, LP-53, and Canterbury Oaks are shown in Figure 2-6, where the rate of change of phosphate concentration in the water column indicates phosphate release rate. Under oxic (aerated) conditions, the water column phosphate concentrations remained low in the water columns of Shoreview Commons, LP-53, and Canterbury Oaks. The relatively small increase in phosphate under oxic conditions in the Alameda water columns can be caused by mineralization of labile organic P in these sediments (Jensen and Andersen 1992). The "air off" phase was used to begin the anoxic sediment release, at the inflection point of each curve, and to determine sediment oxygen demand. Once the air supply was switched off, the water column DO concentrations started decreasing due to the sediment oxygen demand. The rate of water column DO consumption was more rapid in the Alameda and Canterbury Oaks cores, with DO concentration reaching < 1 mg/L within 5 to 9 days in Alameda and 2 to 7 days in Canterbury Oaks, while it took 7 to 12 days in Shoreview Commons and 7 days (one column only) in LP-53. High sediment oxygen demand is indicative of opportunistic aerobic respiration by microbes and is roughly proportional to the microbe population. Further, the sediment microbial activity is related to the phosphorus release from the sediments (Taguchi et al. 2020). This is supported by the observed concomitant increase in water column phosphate due to sediment phosphate release under the reduced DO conditions in all Alameda cores and some of the cores from Canterbury Oaks and Shoreview Commons.

In the last phase, with an anoxic water column maintained by nitrogen gas bubbling, phosphate release continued to occur in all sediment cores from Alameda and Canterbury Oaks (Figure 2-6). Three (out of five) cores from the iron-treated Shoreview Commons pond showed phosphate release, which may indicate regions of low iron addition (SV1) along with regions of high iron deposition (SV3 and SV5). One Shoreview Commons column showed decreasing phosphate concentrations over time, which is likely due to precipitation of phosphate with ferrous iron (Natarajan et al. 2017). There was no measurable phosphate release from LP-53 sediments, which can be attributed to the alum treatment of the sediments in 2019, so the treatment was still successful as of the 2023 season. The higher phosphate release in Shoreview Commons than LP-53 may also indicate a longer effectiveness and/or a more even distribution of alum treatment versus iron filings treatment of sediments.

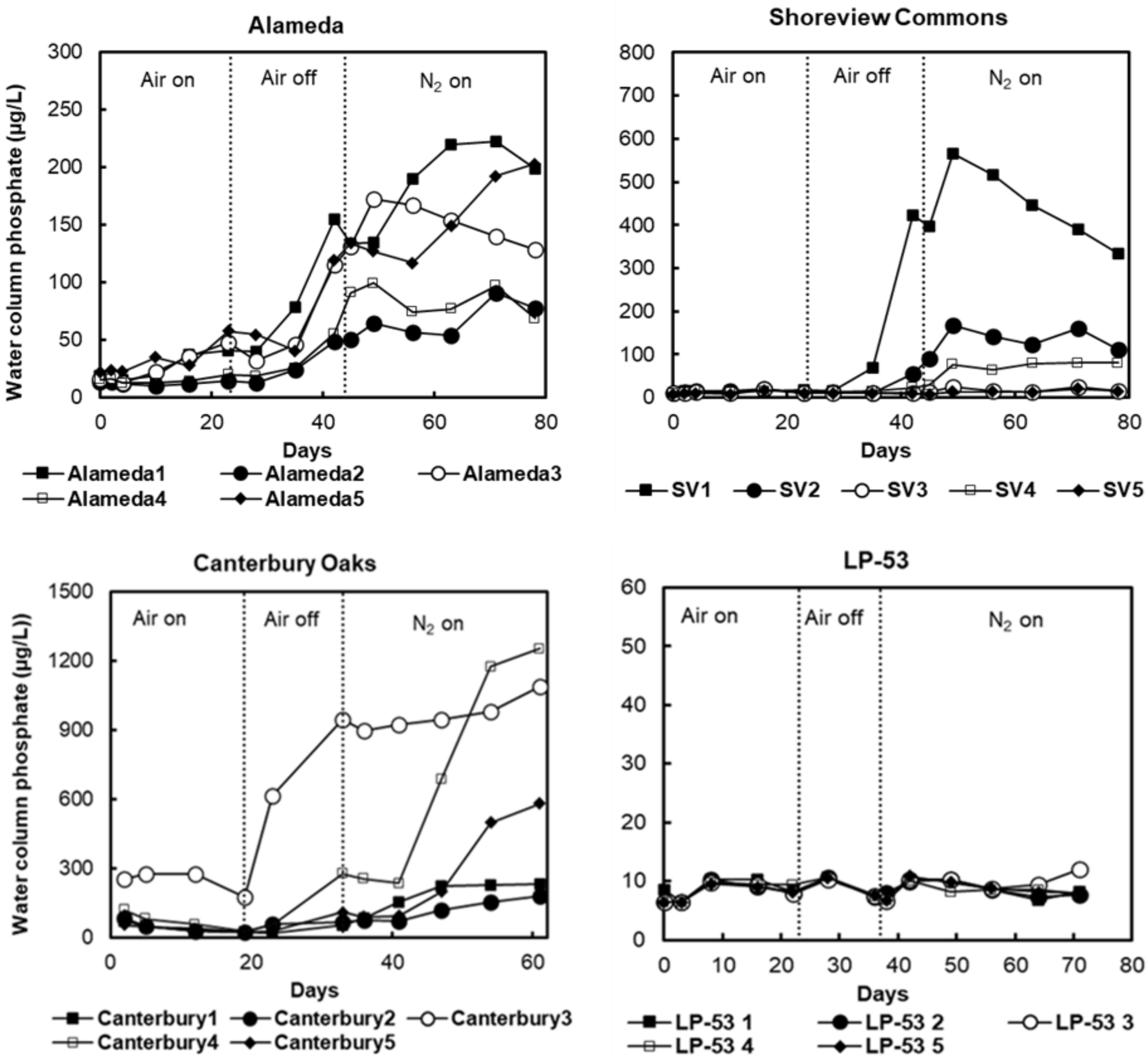


Figure 2-6. Laboratory phosphorus release study results showing the ortho-phosphate concentrations in the water columns of the five sediment cores from each pond/historic wetland site under the air bubbling (oxic; 22 days), air off (22 days), and N_2 bubbling (anoxic; 40 days) phases at 20°C. Notice the difference in y-axis scale in the plots. Data shown for Shoreview Commons and LP-53 are post treatment with iron filings and alum, respectively.

The mean oxic and anoxic sediment phosphate release rates for the six pond and historic wetland sites are summarized in Table 2-4. Camden Central had a previous anoxic sediment phosphorus release study performed by Stantec (2022), which is included in Table 2-4 (oxic flux was not reported). Under oxic conditions, the phosphate flux was minor for Canterbury Oaks. The oxic phosphate flux of 0.5 mg/m2/day for Alameda is likely caused by the mineralization of labile organic P in these sediments (Jensen and Andersen 1992). The anoxic phosphate flux values for the Alameda, Canterbury Oaks and Camden Central are moderately high and indicate the risk for internal phosphorus loading in these ponds under summer anoxia. The phosphate flux for May through October of each year was computed from Equation 2.2:

$$Phosphate\ Flux = (1 - Anoxic\ Factor) * Oxic\ Flux + Anoxic\ Factor * Anoxic\ Flux \quad (2.2)$$

The computed phosphate flux is fairly high for Alameda and Camden Central, medium for Shoreview Commons and Canterbury Oaks and almost zero for LP-53. Again, the herbicide treatments in Camden Central appears to have little effect on phosphate release from the Camden Central sediments.

Table 2-4. Sediment phosphate flux, sediment oxygen demand (S_{max}), and sediment organic content in the sediments of ponds and historic wetlands studied. Values listed are mean ± standard deviation for five sediment cores from each site. Values after chemical treatment of sediments are reported for Shoreview Commons (iron filings) and LP-53 (alum). Average organic matter content in the top 4 cm depth of sediments provided.

Pond	Oxic phosphate flux ($mg/m^2/day$)	Anoxic phosphate flux ($mg/m^2/day$)	Computed phosphate flux (May to October)		S_{max} ($g/m^2/day$)	Sediment organic content (dry g/g)
			2023	2024		
Alameda	0.50 ± 0.44	2.3 ± 1.6	2.0	1.5	1.4 ± 0.62	28%
Shoreview Commons	-0.01 ± 0.04	1.9 ± 2.6	0.8	1.0	0.92 ± 0.34	30%
Camden Central	n/a (~0)*	3.8 ± 1.8	1.9	1.7	n/a	26%
Canterbury Oaks	-1.2 ± 1.1	3.7 ± 3.2	0.5	1.8	0.94 ± 0.67	17%
LP-53	0.02 ± 0.03	0.01 ± 0.03	0.02	0.02	0.65 ± 0.14	25%

*Assumed Camden Central to have zero oxic phosphate flux.

Iron and alum treatments in Shoreview Commons and LP-53, respectively, appeared to be effective for controls of anoxic phosphate release compared to Alameda which has no history of any treatment. This finding is supported by our previous analysis of untreated sediments from Shoreview Commons that showed significant phosphate flux of 4.3 $mg/m^2/day$ under oxic and 5.9 $mg/m^2/day$ under anoxic conditions (Natarajan and Gulliver 2022). The pre-treatment anoxic release flux from LP-53 was 7.1 $mg/m^2/day$ (personal communication, J. Bischoff). The sediment treatment in Shoreview Commons and LP-53 still appear to be effective as these ponds do not exhibit substantial sediment phosphorus release under anoxic conditions in the laboratory.

Chapter 3: Development and Verification of 1-D Retention Pond Phosphorus Model

While field monitoring provides valuable information on the hydrologic and water quality performance of stormwater ponds and wetlands, computer models can greatly extend the usefulness of monitoring data by providing more complete information on parameters such as temperature and dissolved oxygen, by extrapolating the performance of the pond for a variety of climate conditions, and by enabling scenario analysis, where the sensitivity of the pond performance to each design parameter, such as remediation techniques, can be systematically analyzed.

In this project, a hydrodynamic and water quality model was used to replicate pond/wetland conditions observed in the field and predict the impact that remediation scenarios could have on sediment phosphorus release and phosphorus export. Sediment phosphorus release can be affected by a variety of factors, but it is largely understood to depend heavily on the concentration of dissolved oxygen in benthic water adjacent to the sediment surface (Steinman and Spears 2019). Dissolved oxygen, in turn, is affected by surface reaeration, temperature, oxygen demand in the water column and from the sediment, vertical mixing processes that transport the oxygen from the surface to the sediment, and temperature or chemical stratification, that tends to resist vertical mixing. Hence, analyzing the export of phosphorus from ponds requires a rather complete physical-chemical process model.

3.1 Numerical Model Development

3.1.1 Background on Hydrodynamic and Water Quality Model

MinLake is a one-dimensional lake water quality model developed over 47 years at the St. Anthony Falls Laboratory, which has been successfully applied to many lake water quality assessments (e.g. Stefan and Ford, 1975; Ford and Stefan, 1980; Gulliver and Stefan, 1982; Riley and Stefan 1988, West and Stefan, 1998; Fang and Stefan 2009). It was chosen for this study for the following reasons:

1) Prior research has shown that stormwater ponds seem to be relatively homogeneous horizontally, but not vertically. This may not be true during storm flows, but our data taken between storms indicates that it is. This means that a one-dimensional model in the vertical direction is sufficient for pond modeling.
2) MinLake was chosen over other known one-dimensional models, such as the General Lake Model (https://gmd.copernicus.org/articles/12/473/2019/) out of the University of Western Australia, because of familiarity of the PIs. Dr. Gulliver worked with the MinLake model in the late 1970s and Dr. Herb worked with the model in the late 2000s.
3) MinLake and other models will all require adjustments to be able to handle pond stratification and remediation, because of the prevalence of floating plants (duckweed and water meal), the sheltering from the wind and the need to develop subroutines that successfully implement remediation strategies, such as aeration.

Existing versions of MinLake simulate thermal stratification, wind mixing and vertical profiles of temperature, dissolved oxygen, chloride, and nutrient concentrations for a vertical profile in the water column. The processes included in the MinLake model are summarized in Figure 3-1. The lake or pond is assumed to be horizontally uniform with possible vertical stratification. MinLake uses an atmospheric heat exchange model to model surface heat transfer at the water surface due to radiation, convection, and evaporation as a function of the current weather conditions. In addition to the temperature profile in the water column, the sediment temperature is also modeled to a depth of 10 m, to capture the thermal inertia of the sediment on the water column. The vertical dissolved oxygen profile is determined based on surface reaeration, biological oxygen demand in the water column, sediment oxygen demand, and vertical transport of oxygen from diffusion and wind mixing. As a part of the dissolved oxygen and nutrient models, an algae growth model is included that can simulate the seasonal growth, senescence, and settling of up to three algae types. Phosphorus is modeled as three components: dissolved phosphorus, phosphorus bound up in algae, and phosphorus bound up in detritus. Further details on the mathematical formulations used in the MinLake model can be found in Riley and Stefan (1987).

For application to stormwater ponds in this project, the existing version of MinLake (Riley and Stefan 1987) was modified as follows:

1) The simulation time step was reduced from 1 day to 1 hour, to capture diurnal stratification and mixing patterns.
2) A model for typical pond outlet structures was added, including weir and orifice water level controls and the ability for the outlet to draw water from a specified depth.
3) The computer code was updated from Fortran 77 to Fortran 95.

A similar MinLake-based pond model was previously used to simulate temperature profiles and heat export from ponds, but it did not consider dissolved oxygen and nutrients (Herb et al. 2009). Further updates to the model for this project include a sub-model for mechanical aeration and a sub-model to consider the effects of free-floating plants (e.g. duckweed) on stratification, mixing and surface oxygen transfer. The new model is called MinPond.

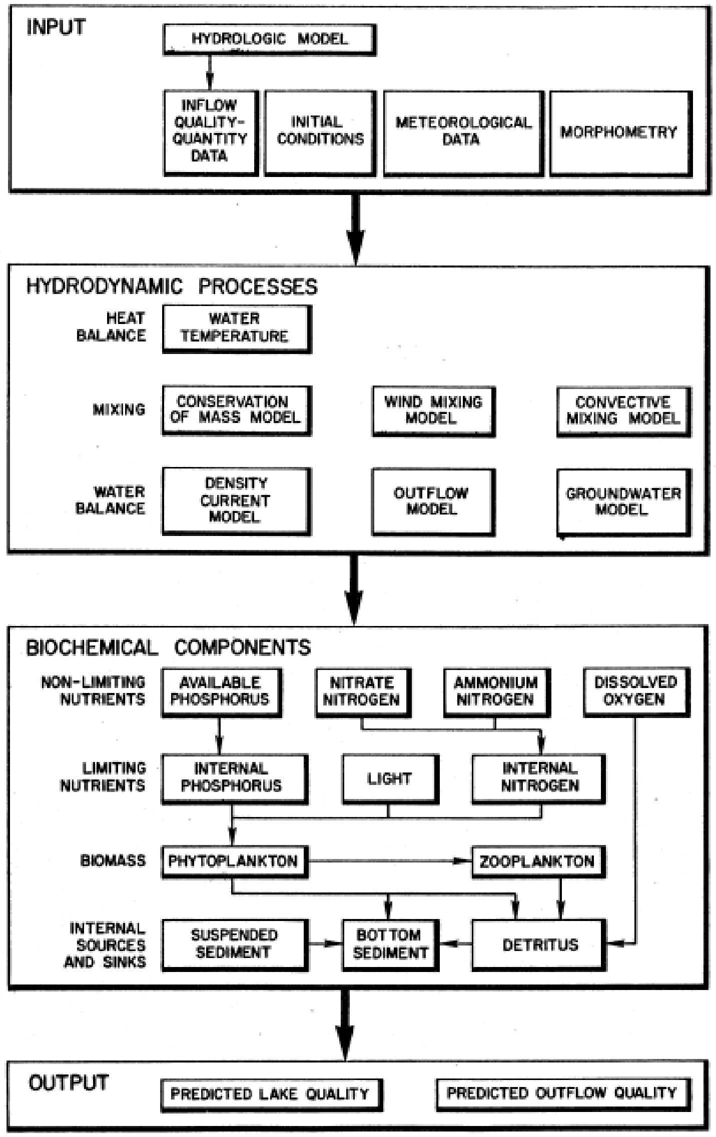


Figure 3-1. Physical, chemical, and biological processes included in the MinLake model (Riley 1988).

3.2 Calibration of 1-D Pond Model

3.2.1 Study Site

The initial MinPond calibration was done for the 2017 monitoring year in Alameda. Alameda is a natural wetland that was altered to allow for permanent stormwater storage. This 1.05-ha (2.55 acres) historic wetland was selected as the initial site because of available data from prior studies (Oberts 1998; Herb et al. 2017; Taguchi et al. 2018, 2020). Alameda receives drainage from 115 ha (288 ac) of mostly residential land, with some commercial/institutional developments, conveyed through a 1-m diameter inlet pipe and a solid rectangular weir outlet structure (Figure 3-2; Herb et al. 2017).

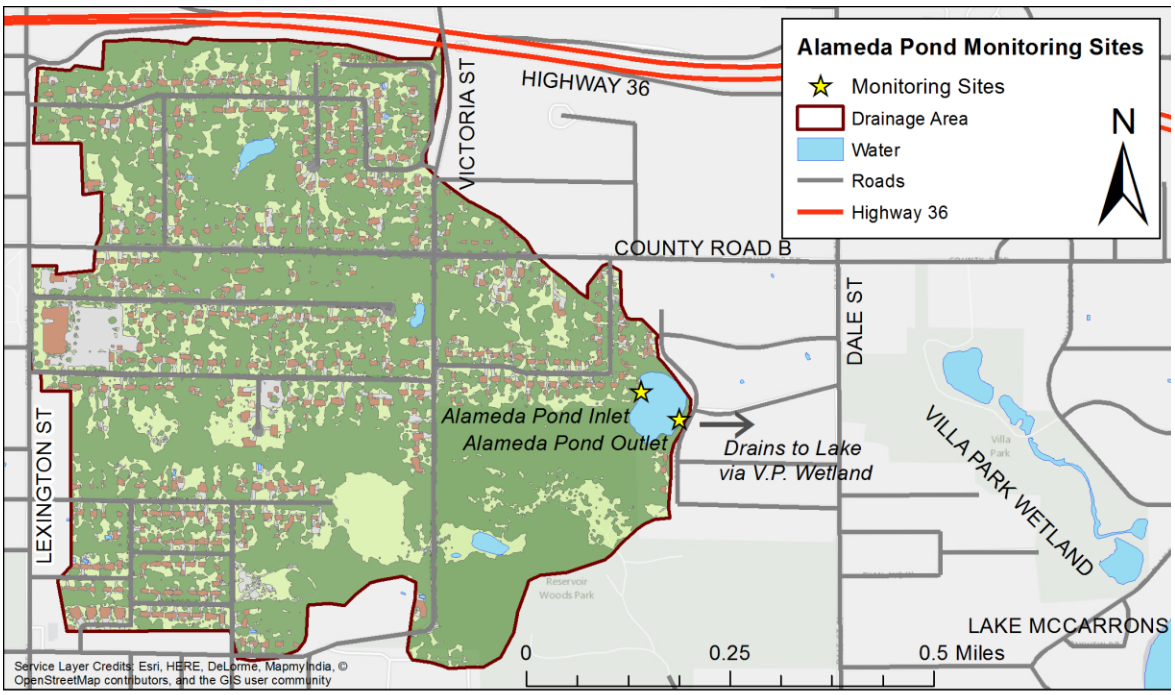


Figure 3-2. Watershed of the Alameda historic wetland site and locations of inlet and outlet monitoring sites at the site (figure from Herb et al. 2017).

3.2.2 Available Pond Field Data

Pond inflow (rate, temperature, conductivity) at Alameda was monitored in 2017 (Herb et al. 2017), bi-weekly profiles of temperature, dissolved oxygen, and conductivity were taken, as well and phosphorus and chlorophyll measurements from the pond water column and grab samples of the inflow (Taguchi et al. 2018). Water surface elevation was also available (Taguchi et al. 2018; Capitol Region Watershed District n.d.). Previously developed pond bathymetry data (Taguchi et al. 2022) were used to develop the depth-surface area relationship used by MinLake to determine the area and volume of each water layer in the simulation. Sediment oxygen demand was based on data from laboratory sediment core incubations (Taguchi et al. 2020). Hourly weather data, including solar radiation, is available from the University of Minnesota Baker Observatory (unpublished).

3.2.2.1 Specified Inflows

Specified inflow to the MinLake model includes hourly flow rate, temperature, chloride concentration, dissolved phosphorus concentration, and detritus concentration (as BOD). Flow rate, temperature, and conductivity were monitored in 2017 at Alameda. Conductivity measurements were converted to chloride concentrations using relationships developed in Herb et al. (2017). Phosphorus and BOD inflow concentrations were estimated as monthly averages from available grab samples (Taguchi et al. 2018).

For final calibration, the 1-D model for Alameda was run for the period 4/15/2017 to 10/31/2017 using nominally 0.1 m thick water layers. The number and thickness of the layers is automatically adjusted in MinLake as water level changes to maintain a layer thickness close to 0.1 m.

3.2.2.2 Water Level /Water Balance Simulation

The MinLake model simulated the dynamics of water level during inflow/outflow events and the slow decrease in water level due to evaporation (Figure 3-3). There is indication that, particularly later in the year, there is an additional slow outflow from the pond, likely groundwater outflow. This additional outflow is unlikely to affect the results obtained by the model. A specified groundwater outflow could be added to model in the future.

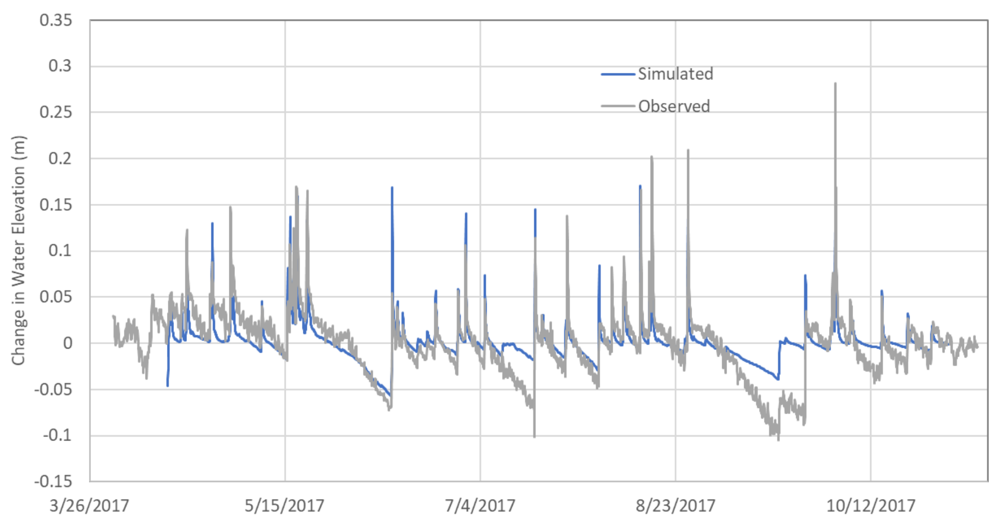


Figure 3-3. Simulated and observed water level in Alameda, 2017.

3.2.3 Water Temperature Simulations

Preliminary simulations of Alameda had significant errors in water temperature compared to observed temperature profiles. These errors were determined to be, in part, because the MinLake model does not consider natural convection processes in the surface heat transfer formation. In cases where the water temperature exceeds air temperature, buoyancy-driven air currents above the pond can increase surface heat transfer above values predicted by forced convection (wind) alone (Ryan et al. 1974). Natural convection becomes an important process for wind-sheltered water bodies with reduced forced convection. The surface heat transfer formation in MinPond was modified to add a term for natural convection, as follows:

$$H_c = C_0(N_{cn} + F_{cn})(T_s - T_a) \quad (3.1)$$

$$F_{cn} = c_1 U \quad (3.2)$$

$$N_{cn} = c_2(T_s - T_a)^{0.33} \; for \; T_s > T_a \quad (3.3)$$

$$N_{cn} = 0 \; for \; T_s \leq T_a \quad (3.4)$$

Where H_c is convective heat transfer, T_s and T_a are the water surface temperature and air temperature, respectively, F_{cn} and N_{cn} are the forced convection and natural convection coefficients, respectively, and c_0, c_1, and c_2 are constants taken from Ryan et al. (1974).

With the modified heat transfer formulation, MinPond was calibrated to the observed water temperature profiles. Table 3-1 summarizes the parameters that were adjusted to give the lowest overall root-mean-square error of 1.5 °C. The water light attenuation coefficient value given (2.5 m^{-1}) is for the case with no algae in the simulation. With the algae model turned on, the water light attenuation coefficient was reduced to 1.5 m^{-1}, since algae provide additional light attenuation. The wind speed multiplier, which accounts for wind sheltering around the pond, gave the best RMSE at a value of 0.5. In the previous study (Taguchi et al. 2022), analysis of wind data taken at a monitoring station in Alameda suggested a wind multiplier of 0.25. Using a wind multiplier of 0.25 in MinPond increased the water temperature RMSE slightly, from 1.5 °C to 1.6 °C. Figure 3-4 gives examples of the simulated and observed temperature profiles in Alameda in 2017.

Table 3-1. Model parameters adjusted for the water temperature calibration.

Parameter	Calibrated Value
Water light attenuation coefficient	2.5 m^{-1}
Hypolimnetic diffusion coefficient for heat	0.026 m^2/day
Wind speed multiplier	0.25-0.5

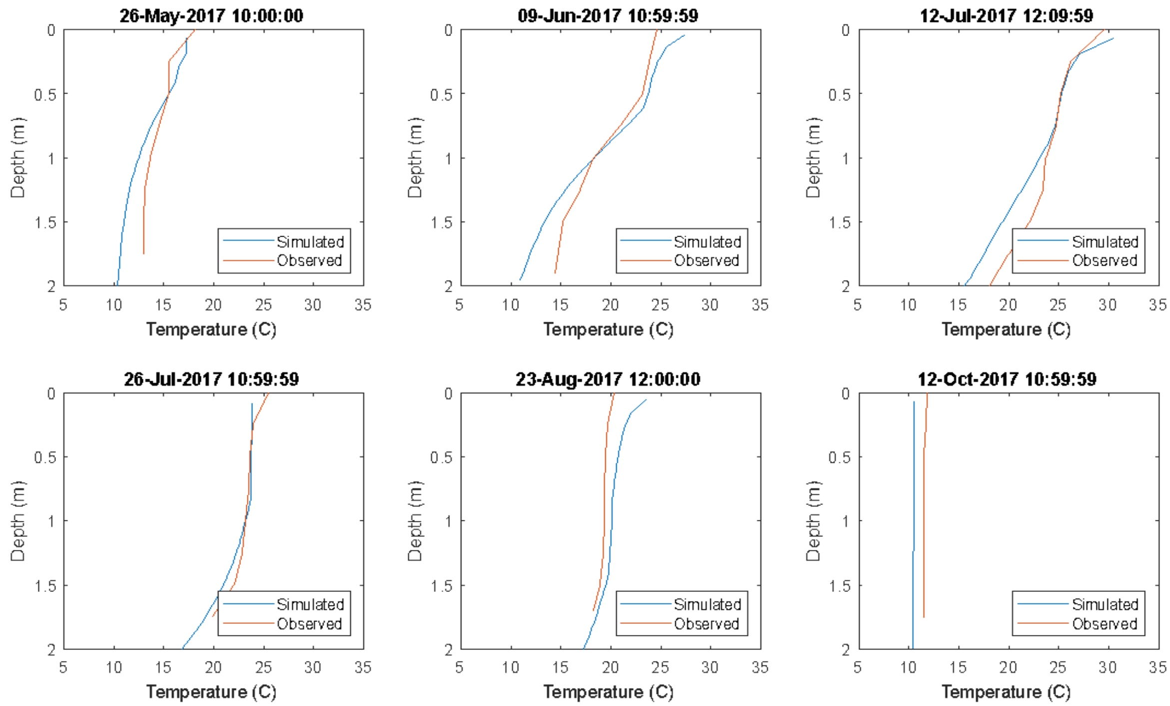


Figure 3-4. Examples of simulated and observed temperature profiles in Alameda in 2017.

3.2.4 Dissolved Oxygen Simulations

Preliminary pond simulations using a sediment oxygen demand coefficient of 5 $g/m^2/day$ (Table 3-2; Taguchi et al. 2018) produced an anoxic water column for much of the year, not matching observations. Similar to surface heat transfer, it was hypothesized that natural convection processes are important for oxygen transfer with the air (Holgerson et al. 2016), and a natural convection term was added to the oxygen transfer formation in MinPond, using a formulation very similar to Equations 1-3. This brought the simulated oxygen balance into better agreement with observations (Figure 3-5). Examples of simulated and observed dissolved oxygen profiles are given in Figure 3-6.

Table 3-2. Model parameters adjusted for the dissolved oxygen calibration.

Parameter	Calibrated Value
Sediment oxygen demand	5 $g/m^2/day$
Hypolimnetic diffusion coefficient for dissolved substances	0.026 m^2/day
Oxygen re-aeration coefficient	0.25-0.5

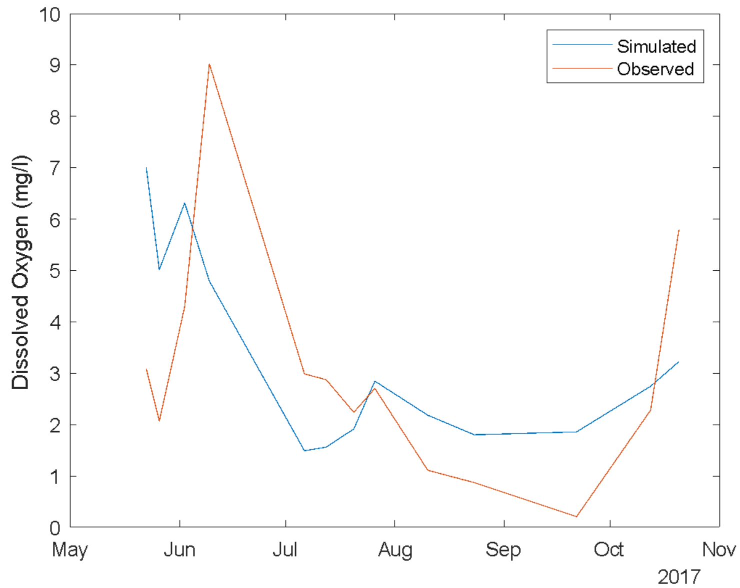


Figure 3-5. Simulated and observed volume-averaged dissolved oxygen concentration in Alameda, 2017. Volume-averaging is averaging over depth weighted by the pond area at each depth.

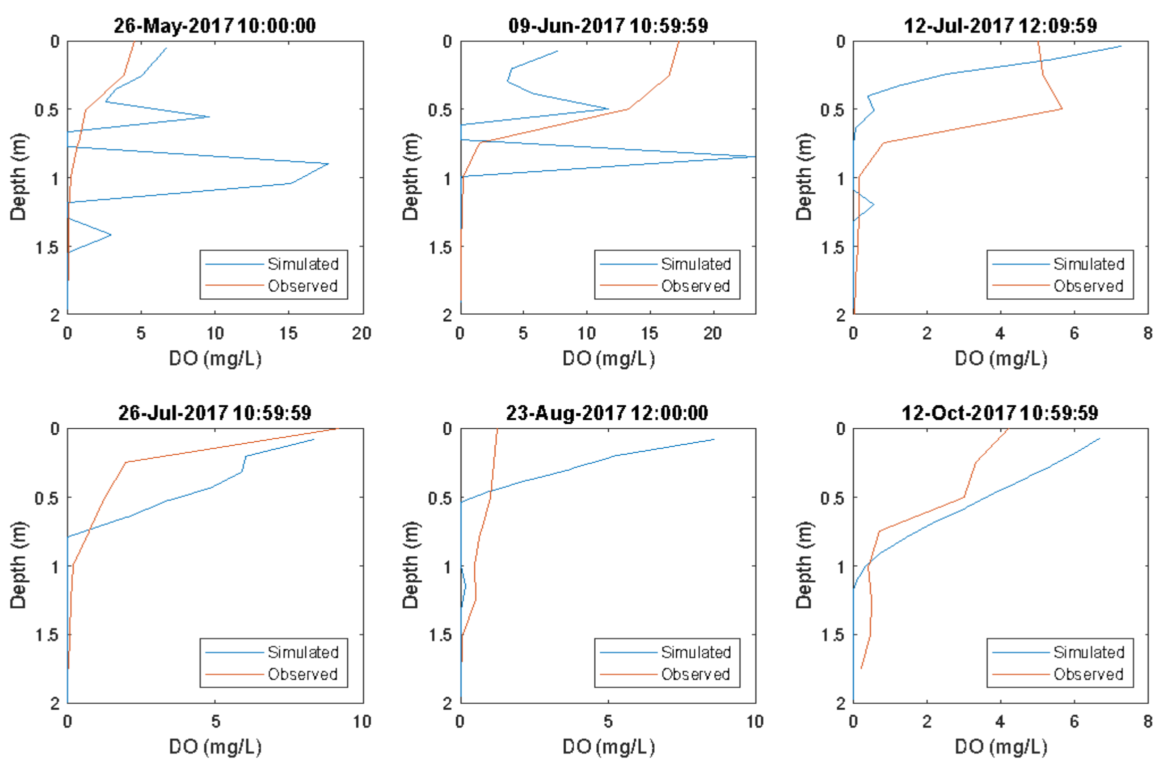


Figure 3-6. Examples of simulated and observed dissolved oxygen profiles in Alameda in 2017.

3.2.5 Chloride Simulations

The 1-D MinPond effectively simulated the persistence of the bottom chloride layer over the year, and flushing of chloride in October. The main parameter controlling chloride vertical diffusion in the water column is the diffusion coefficient for dissolved substances, set to 0.026 m^2/day (Table 3-2). Example simulated and observed chloride profiles are given in Figure 3-7.

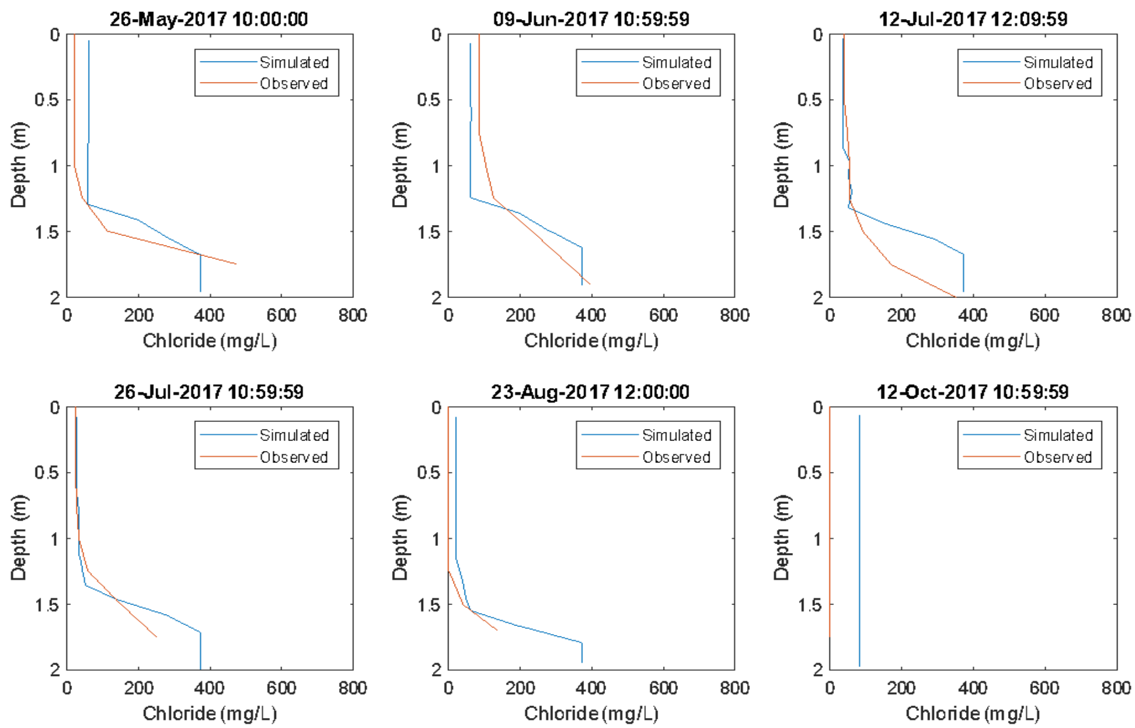


Figure 3-7. Examples of simulated and observed chloride profiles in Alameda in 2017.

3.2.6 Phosphorus Simulations

The main model parameter to be set for simulations of phosphorus is the anoxic sediment release rate. Based on previous lab data (Taguchi et al. 2018), this was set to 5 mg/m^2/day for Alameda. In-pond total phosphorus (TP) measurements are compared to simulated TP profiles for 2017 in Figure 3-8. The estimated phosphorus inflows and the sediment release rates give about the right amount of phosphorus in the water column, but the simulated profiles are difficult to verify with the existing field observations.

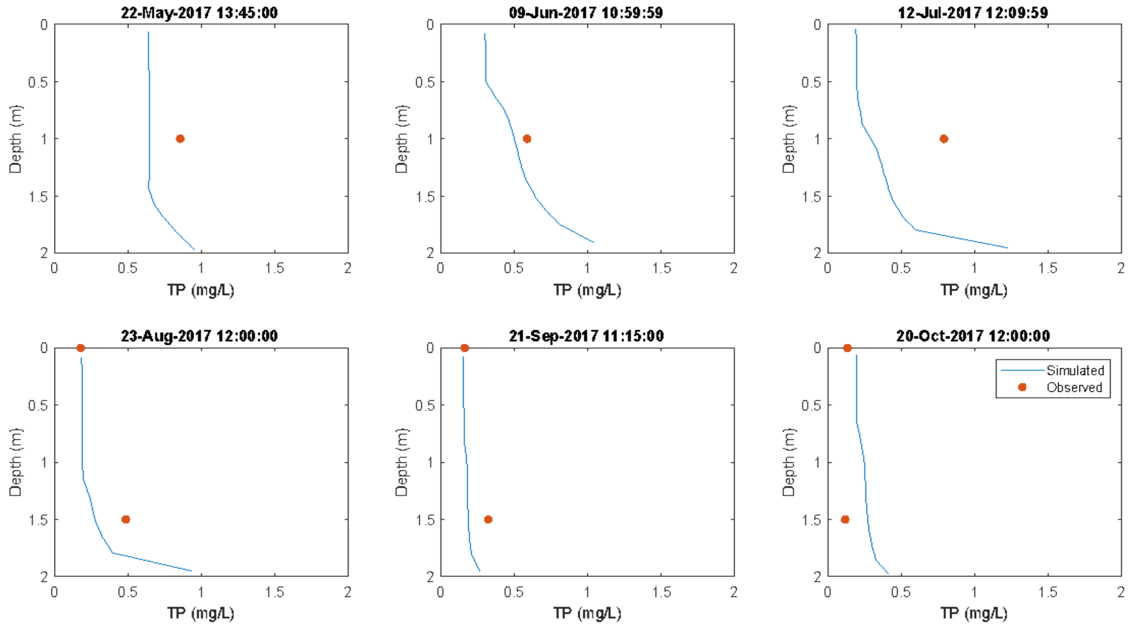


Figure 3-8. Examples of simulated and observed total phosphorus (TP) profiles in Alameda in 2017. The observed TP data for May, June, and July are depth-averaged, while the August, September, and October TP data were discrete epilimnion and hypolimnetic samples.

3.3 MinPond Refinement and Verification

MinPond was refined and verified by:

- simulating hydrology and water quality in Alameda for additional monitoring years, including preliminary refinements to the model to include the effects of duckweed on surface heat transfer
- simulating the water budget and water quality in Camden Central, utilizing monitoring data taken under this project along with additional data supplied by the Minneapolis Park Board

3.3.1 Refinements to MinPond

MinPond was used to simulate water budget and water quality in Alameda for 2021 and 2022. The monitoring data available for these years includes continuous in-pond monitoring of the vertical temperature profile, water level, and wind speed, and conductivity at one depth. Additional monitoring data included bi-weekly profiles of temperature and conductivity, water chemistry measurements (TP, SRP), and measurements of floating plant cover and density. The floating plant measurements at Alameda and other Twin Cities metro ponds (data collected in this project and Janke et al. 2023) were used to refine MinPond surface heat transfer component, as follows:

- The surface albedo (the fraction of solar radiation reflected by the water surface) was adjusted over the range from 0.07 (no floating plant cover) to 0.25 (full floating plant cover; van Marrewijk 2017) for the fraction of the pond covered with floating plants.

- Based on light attenuation measurements taken at Twin Cities metro ponds, portions of a pond covered by floating plants were assumed to attenuate 70% of non-reflected solar radiation in the upper few cm of the water column.

For MinPond, the effects of floating plants were included as an area-weighted average, e.g. for 50% duckweed coverage, the surface albedo is taken as the average of 0.07 and 0.25 = 0.16. Other effects of duckweed on pond heat transfer, oxygen transfer, and mixing probably exist, however, insufficient data is currently available to estimate these effects. Including the effect of duckweed on surface albedo and light attenuation resulted in measurable improvements in the temperature simulations for Alameda (Figure 3-9). For example, for the 2020 simulations, the overall root-mean-square error of the temperature simulations improved from 3.9 °C to 1.9 °C.

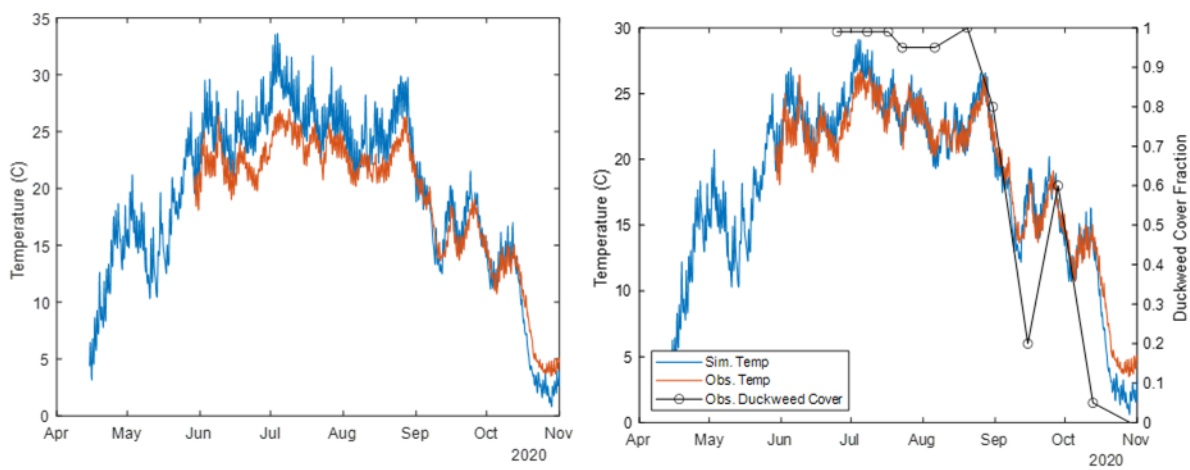


Figure 3-9. Simulated and observed near-surface water temperatures (0.25 m depth) at Alameda for models omitting (left) and including (right) the effects of duckweed cover on surface albedo and light attenuation.

3.3.2 Modeling of Camden Central

MinPond was used to simulate the water budget and water quality in Camden Central for the 2023 monitoring year, using monitoring data collected in this project and by the Minneapolis Park & Recreation Board (MPRB). Camden Central is newer (construction year of 2007) than Alameda, has lower wind sheltering than Alameda, is treated with a herbicide during each season, and consequently does not have significant floating plant cover. Camden Central also has a Hennepin County meteorological station within 50 m (shown in aerial photo of Camden in Figures 2-1).

MinPond for Camden Central was assembled using bathymetry data supplied by Minneapolis Park and Recreation Board (MPRB), and monitoring data taken by the MPRB was used to develop inflow phosphorus and chloride concentrations. Inflow rates were estimated based on the pond level data taken at the pond monitoring station and the known pond bathymetry.

MinPond was run for the 2023 season at Camden Central using the assembled inflow time series and 2023 climate data from the Hennepin County station. The model was run using the same coefficients as the Alameda pond model, except that 1) the floating plant cover was set to zero for Camden Central, and 2) the wind sheltering multiplier was increased from 0.25 to 0.55 (less wind sheltering). The Camden Central data set had the unique feature of having both on-pond wind data from the pond station and an immediately adjacent meteorological tower – this gave a rather direct measure of wind sheltering. Figure 3-10 plots the ratio of the on-pond wind speed to the met tower wind speed in 2023 – the ratio varies by direction, with higher ratios (less wind sheltering) along the longer east-west direction of the pond. The weighted average (by number of observations) ratio is 0.60, close to the calibrated wind sheltering coefficient (0.55) used in Camden Central model.

Overall, the simulation results for Camden (Figure 3-11and Figure 3-12) were better compared to the Alameda pond simulations, likely because of the lack of duckweed cover at Camden. The root-mean-square error of the temperature simulations was 1.0 ^{o}C for Camden, compared to 1.9 ^{o}C for Alameda.

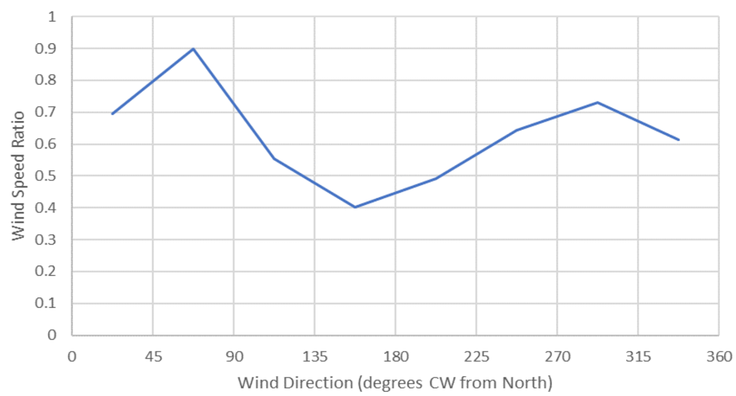


Figure 3-10. Ratio of the on-pond wind speed to the adjacent met tower wind speed, for eight directional bins.

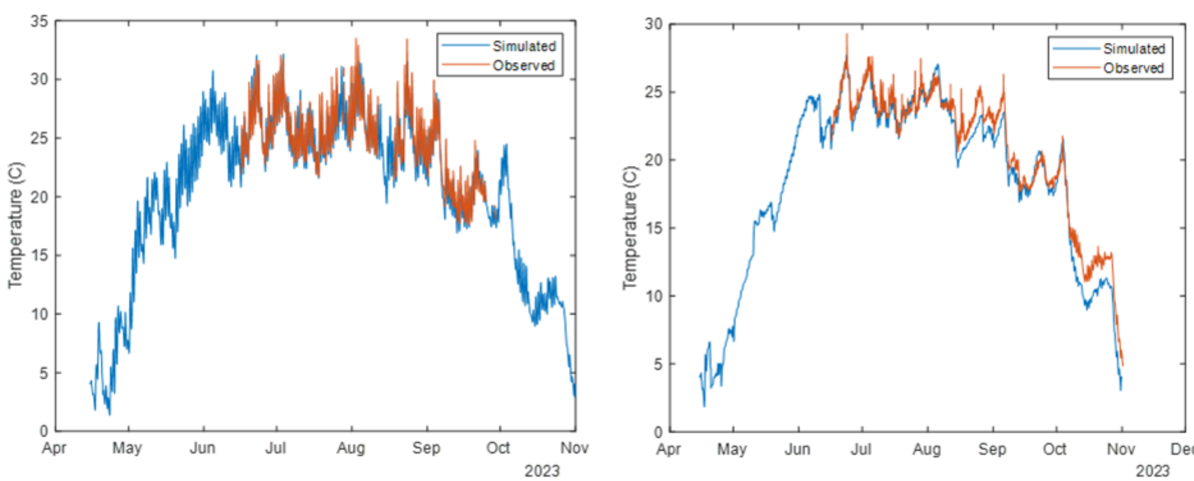


Figure 3-11. Temperature simulation results in comparison to the temperature monitoring data taken at the Camden pond station in 2023, for 0.25 m depth (left) and 1.0 m depth (right). The overall root-mean-square error of the temperature simulation was 1.0 ^{o}C.

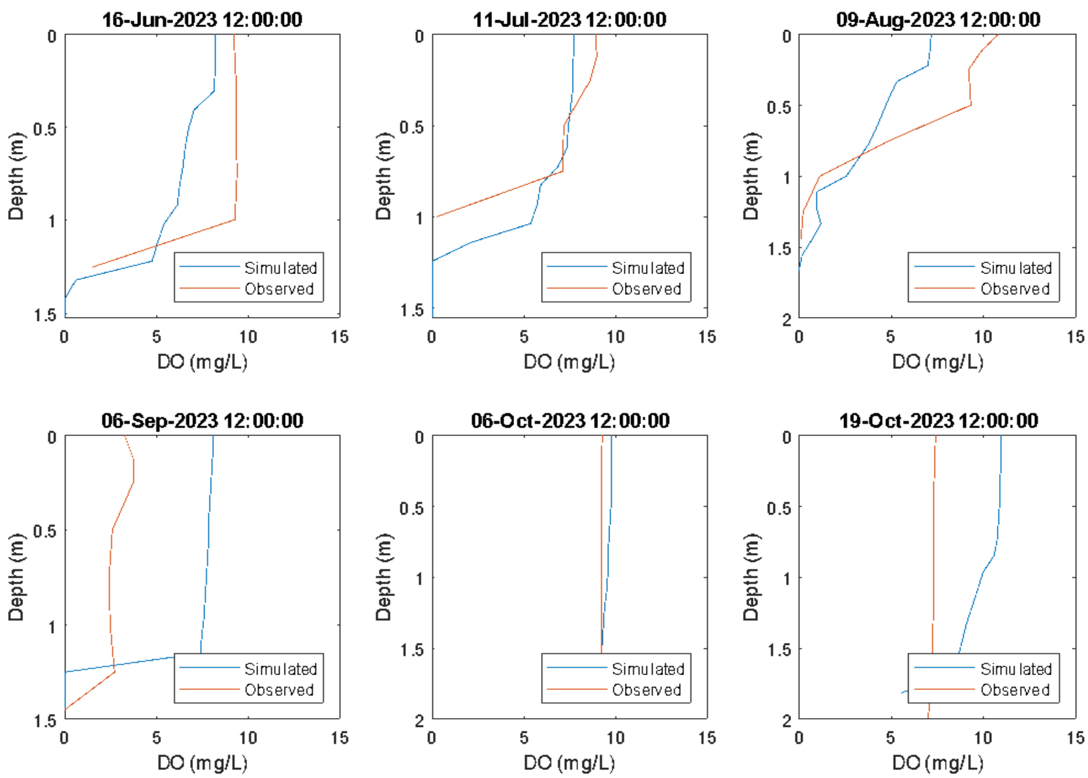


Figure 3-12. Simulated dissolved oxygen (DO) profiles in comparison to the observed profiles taken at the monitoring station at Camden Central in 2023.

Chapter 4: Determining Effectiveness of Remediation Techniques using the 1-D Retention Pond Phosphorus Model

In this chapter, we present computer simulation results using the previously developed MinPond (Chapter 3) for six remediation techniques applied to stormwater ponds with and without duckweed coverage. Finally, the pond simulations are summarized and compiled with costing information to evaluate the cost effectiveness of different pond remediation techniques in reducing phosphorus export from ponds.

4.1 Updates to MinPond

4.1.1 Floating Plants

It has become clear to us that floating-leaf macrophytes such as duckweed (*Lemna*) and water meal (*Wolffia*) play an important role in the phosphorus dynamics of stormwater ponds and historic wetlands (Figure 4-1; Janke et al. 2023). To address this, we added a module to MinPond that simulates the growth and senescence of generic floating-leaf macrophytes, herein called duckweed. The purposes of adding the duckweed growth model were to 1) make the simulations of mitigation strategies more realistic and 2) increase our understanding of how duckweed affects the phosphorus budget in a pond or historic wetland.

The duckweed growth model is similar to the algal growth model that was already included in MinPond, and includes growth as a function of light availability, temperature, and phosphorus availability. The main difference between the duckweed growth model and the algal growth model is that the duckweed model simulates a single layer of plants at the water surface, while the algal growth model simulates a concentration of algae over the entire water column. The duckweed biomass is assumed to cover the entire pond or historic wetland, and is represented in MinPond as a mass per unit area, where the mass is specified as grams dry weight to correspond to typical measurement analysis. The various parameters used in the duckweed growth model were either obtained from literature, obtained from data taken in this and partner University of Minnesota pond projects, or determined via calibration of the model outputs to observed duckweed biomass. An initial duckweed biomass is specified at the start of each model simulation year and allowed to grow along with (and in competition with) algae. Further details on the equations and parameters used in the duckweed model are given in Appendix C.

The effects of the duckweed biomass on the pond or historic wetland include the following:

- Additional light is attenuated at the surface (see Appendix C)
- Surface albedo (light reflection) is increased to 0.25 (van Marrewijk 2017)
- Surface oxygen transfer is reduced (Morris and Barker 1977). Oxygen produced by duckweed photosynthesis is assumed lost to the atmosphere (Pokorný and Rejmánková 1983).

- Duckweed growth takes up dissolved phosphorus from the water column (see Appendix C) and duckweed senescence adds phosphorus to the water column.
- Duckweed loss through the pond or historic wetland outlet is included in the phosphorus export calculations

Figure 4-1. Photo of Shoreview Commons with free-floating macrophyte (duckweed) cover. Photo credit: Emma Squires (2025).

4.1.2 Submersed, Rooted Macrophytes

A number of stormwater ponds and historic wetlands in the Twin Cities region contain a significant number of rooted, submerged macrophytes such as pondweed and coontail (Figure 4-2). These ponds have been the subject of study in Finlay et al. (2026). In the present project, we added a rooted macrophyte growth model to the model to study how the presence of these plants may affect dissolved oxygen and nutrient retention. A rooted macrophyte growth model was adapted from a similar lake model recently developed in the USGS Aquatic Invasive Species project[5], "Managing water quality and invasive macrophytes to promote healthy native aquatic plant communities", PIs Ray Newman and Jeff Peterson.

The rooted macrophyte growth model predicts the growth of plants at different depths in a pond or historic wetland as a function of available light and temperature, as detailed in Appendix C. The modeled effects of plants on pond or historic wetland water include:

- Plant growth produces dissolved oxygen in the water column
- Plant respiration at night consumes oxygen in the water column
- Plant senescence and decay produce detritus, which consumes oxygen
- Plant biomass attenuates light (in competition with algae) and dissipates wind mixing energy (Herb and Stefan 2005) – both these processes tend to increase thermal stratification

[5] https://wrc.umn.edu/projects/managing-invasive-macrophytes

Figure 4-2. Photo of Briarcroft pond with rooted macrophytes. Photo credit: Andrew Ratz (2024).

4.2 MinPond Simulations to Evaluate Mitigation Strategies

We used MinPond to simulate the effect of several mitigation strategies on in-pond TP concentrations and phosphorus export to downstream water bodies. The following six mitigation scenarios were considered:

1. Chemical sediment treatment
2. Mechanical aeration
3. Wind sheltering reduction
4. Watershed load reduction
5. Smart water level control
6. Clay lining

For each scenario, MinPond was run for 10 years of simulations (2014-2023) for two or more variations of the treatment practice. For each variation of a treatment practice, 10-year means and standard deviations of in-pond phosphorus concentration, phosphorus export, anoxic sediment area fraction, and mixed layer depth were recorded to evaluate the effect of each treatment practice on pond or historic wetland performance.

The hourly inflow rates for the ten years of simulations were developed based on inflow data at the Alameda historic wetland. Inflow rates were either directly observed at the inlet or back-calculated from observed water levels. A regression relationship between hourly precipitation and hourly inflow rates was then developed (Equation 4.1, Figure 4-3), to enable estimation of inflow rates to the historic wetland for any year with local precipitation data.

$$Q_{in} = c_0 \cdot (P_0 + c_1 \cdot P_{-1} + c_2 \cdot P_{-2}) \qquad (4.1)$$

Where Q_{in} is the inflow rate, P_o is the precipitation depth at the time of the inflow measurement, P_{-1} and P_{-2} are the precipitation depths at 1 and 2 hours prior to the inflow measurement, respectively, and c_0,

c_1, c_2 are constants. For the historic wetland Alameda, c_0=0.057, c_1=0.88, and c_2=0.39, with Q_{in} in m^3/s and P in cm. For the other ponds and historic wetlands, these flow rates were linearly scaled based on the impervious area ratio to Alameda.

The phosphorus concentrations of the inflows were based on data from the metro-area stormwater database (Finlay et al. 2024). Data for TP and SRP concentration were extracted for watersheds in the range of 50 to 500 acres, and relationships were developed between runoff concentration and the 14-day total precipitation prior to the sampling event. The phosphorus concentration data were log transformed prior to fitting. The exponential fits for TP and SRP are listed and illustrated in Figure 4-4. Lower precipitation in the two weeks prior to sampling leads to higher TP and SRP concentrations. The combination of the fits for inflow rate and concentration were used to generate inflow time series to MinPond with representative variability based on the observed precipitation times series for each year of simulation.

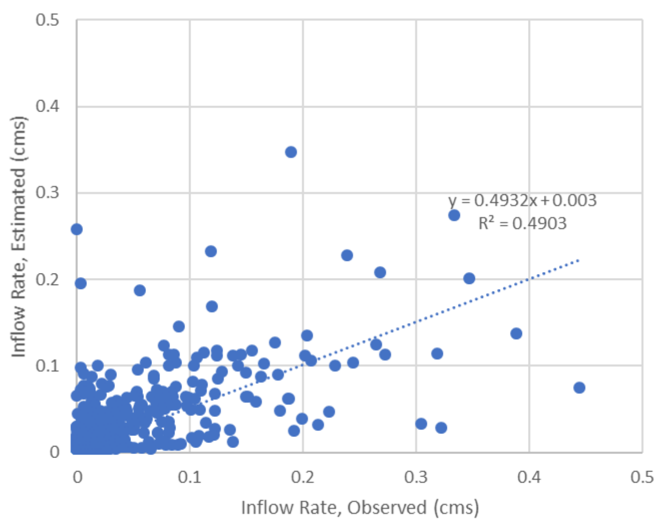


Figure 4-3. Predicted vs. observed hourly inflow rates to Alameda for 2020 and 2021 data. The predicted flow is based on observed precipitation (Equation 4.1). The units of inflow are cubic meters per second (cms).

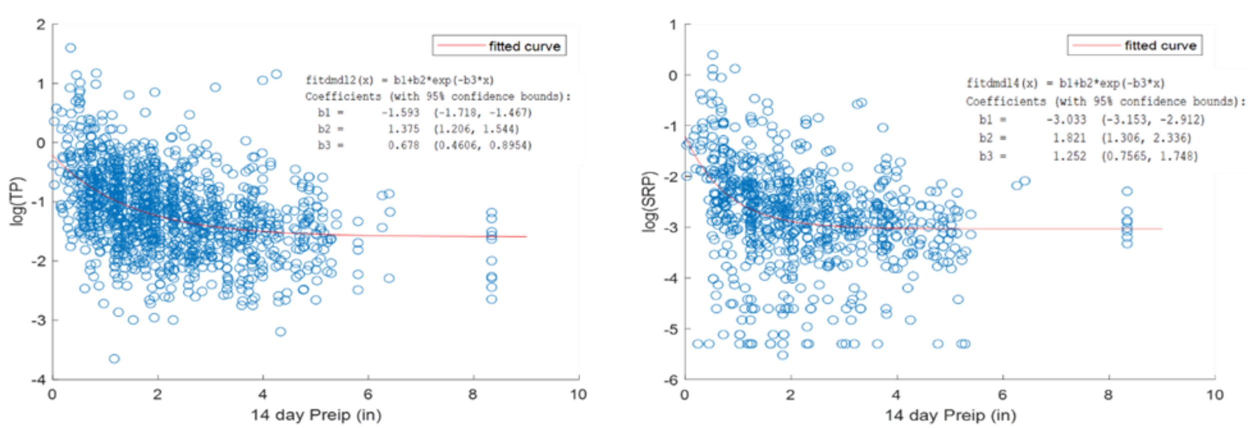


Figure 4-4. Fits of observed log-transformed TP and SRP to 14-day total precipitation.

The implementation of each treatment practice in MinPond was as follows:

4.2.1 Chemical sediment treatment

We considered chemical treatments of pond and historic wetland sediments that bind phosphorus in the sediments and thus reduce the sediment soluble reactive phosphorus (SRP) release rates. The two assessed treatment types included alum and iron filings, and for both treatment types, we estimated both effectiveness (dosage) and cost of applications. The effect of chemical treatments on pond and historic wetland performance was simulated by reducing the specified anoxic release rate, as measured in previous University of Minnesota pond and historic wetland projects (Taguchi et al. 2022).

4.2.2 Mechanical Aeration

The effect of mechanical aeration (air bubble plumes) on pond and historic wetland mixing, dissolved oxygen, and the phosphorus budget was added to MinPond using information on lake aerator design from Schladow (1993). Schladow gives design curves that relate the efficiency (η) of an aerator to water depth, thermal stratification, and the aerator flow rate, where the efficiency is defined as:

$$\eta = \frac{\Delta PE}{W_{iso}} \qquad (4.2)$$

where ΔPE is the change in potential energy of stratification in the lake and W_{iso} is the isothermal work created by the aerator pump. The isothermal work was estimated by Schladow as:

$$W_{iso} = 2.203\, p_a\, Q_T\, \Delta t \log\left(\frac{p_h}{p_a}\right) \qquad (4.3)$$

where p_a and p_h are the surface pressure and pressure at the depth of the diffuser outlet, Q_T is the flow rate of the diffuser pump, and Δt is the time step. For the relatively low water depths of typical stormwater ponds and historic wetlands, the aerator efficiencies are predicted to be quite low, on the order of 1% or lower – we assumed a 1% efficiency, meaning that only 1% of the aerator work leads to destratification of the water body. Since these aerator design curves were developed for deeper lakes and reservoirs, the predicted efficiency for a pond and historic wetland is relatively uncertain.

Using Equations 4.2 and 4.3, the energy available to destratify a pond and historic wetland can be estimated as a function of the pond and historic wetland depth and the aerator flow rate. The aeration energy was then directly added to the wind mixing energy in MinPond, to calculate the increase in the mixed layer depth associated with the aerator. This assumed that vertical mixing due to aeration processes is similar to wind mixing.

A second, more complex model for mixing in lakes using aerators was also tested for a subset of two ponds (Alameda and Camden Central). This aerator model was developed for MINLAKE and tested for several lakes (Zic and Stefan 1989). Briefly, the bubble aeration model considers a vertical bubble plume from a diffuser, and calculates the flume diameter and velocities, the entrainment of water as the plume rises, the radial jet that is formed near the water surface, and the plunging of the plume to an intermediate depth. The Zic/Stefan bubble plume model was adapted for use with MinPond. For Alameda, a single diffuser with 5 CFM (2.4 l/s) flow rate was modeled, while the Camden Central model (a larger pond) was run using 2 diffusers with 5 CFM per diffuser.

4.2.3 Wind Sheltering Reduction

The effect of changes to wind sheltering on the vertical mixing and phosphorus dynamics in stormwater ponds or historic wetlands was examined with MinPond by by changing the wind sheltering coefficients, linearly related to the reduction in wind velocity, for each pond over a range of values (0.4 to 0.8) that correspond to the range of values observed in field measurements. All ponds and historic wetlands analyzed were subjected to the same range of wind sheltering, although small ponds and historic wetlands will tend to be more sheltered than larger ponds and historic wetlands. For ponds and historic wetlands with duckweed, we did not include a function to reduce or eliminate the duckweed coverage for low wind sheltering cases, or include the dynamics of the short-term response of duckweed to particularly windy conditions, due to a lack of quantitative data.

4.2.4 Watershed Load Reduction

Two types of watershed load reduction were evaluated for their ability to reduce phosphorus export from stormwater ponds and historic wetlands:

1) Nutrient load reduction, corresponding to, for example, street sweeping practices. The inflow nutrient concentrations were scaled over a range of possible reductions of up to 80%.
2) Runoff volume load reduction, corresponding to, for example, the addition of infiltration practices. Runoff nutrient concentrations were kept unchanged from the baseline condition.

For each case, the nutrient or volume scaling factor was fixed over the ten years of simulation.

4.2.5 Smart Water Level Control

Smart water level control in stormwater ponds and historic wetlands is typically used to partially drain ponds and historic wetlands prior to storms, to increase storage capacity. We examined the effect of such a strategy on the phosphorus export of a pond or historic wetland, assuming the pond or historic wetland was drained to half of the control depth for storm events of 1.5 inches or greater.

4.2.6 Clay Lining

Clay liners in stormwater ponds are typically used to prevent excessive infiltration of water to shallow groundwater, to prevent groundwater contamination or to prevent contaminated groundwater from entering the pond. To examine how including a clay liner in a pond construction design or retrofit could affect phosphorus export, we eliminated the specified groundwater outflow rate for Alameda (the calibrated Alameda MinPond has a net outflow to groundwater). To generalize this case a bit more, we also considered what happens to the phosphorus budget of a pond or historic wetland if there is a net positive groundwater input, producing a net baseflow out of the pond or historic wetland. We assumed an input of water only, with no nutrient concentration. This case could also give an indication of the response of a pond or historic wetland to a steady pumped inflow.

4.3 Pond Selection for Mitigation Analysis

The six ponds and historic wetlands selected for mitigation analysis are given in Table 2-1 and Figure 2-1, including two ponds that have high wind sheltering and high duckweed coverage (Alameda, Shoreview Commons, Canterbury Oaks), one pond with low wind sheltering and little or no duckweed (Camden Central), and a pond and a historic wetland with significant rooted macrophytes (LP-53, Briarcroft). Four of the ponds are historic wetlands that have been converted to capture stormwater runoff. We included two historic wetlands (Alameda, Shoreview Commons) that were modeled in the previous project (Taguchi et al. 2022) to examine the effect of the duckweed simulations included in the new pond model, MinPond, used in this study. Models for each of the six ponds and historic wetlands were assembled using available pond/historic wetland bathymetry and watershed information (Table 2-1). Some outlet structure information was available for Alameda, Camden Central, and Shoreview Commons that was used in the respective models, with all being surface outlets. For the remaining ponds and historic wetlands, a surface outlet weir was assumed and sized to give approximately 24-hour drawdown.

4.4 Results of Pond Mitigation Analysis

4.4.1 Duckweed and Rooted Plant Simulations

Figure 4-5 gives an example simulation of duckweed and algal biomass in Alameda over a ten-year period. The maximum duckweed biomass varies from year to year, based on varying climate conditions (mainly air temperature and precipitation). Higher duckweed biomass tends to suppress algal biomass via competition for light and for phosphorus. To compare the pond and historic wetland phosphorus dynamics with and without duckweed, the Alameda pond/historic wetland model was run both with the duckweed growth module on (as in Figure 4-4) and with duckweed growth off. The effect of the presence of duckweed growth on simulated phosphorus (TP) export varied from year to year (Figure 4-6), but the 10-year average TP export was 9.2 kg/year with duckweed and 7.9 kg/year without duckweed (16% higher).

An example of rooted plant biomass simulations is given in Figure 4-7, for the LP-53 historic wetland. The LP-53 model was run with and without plants, to help quantify the effect of the rooted plants on phosphorus dynamics. Figure 4-8 shows that the rooted plants slightly increased TP export – this is likely due to the plants taking up P from the sediment during growth and then releasing P into the water column during senescence and decay.

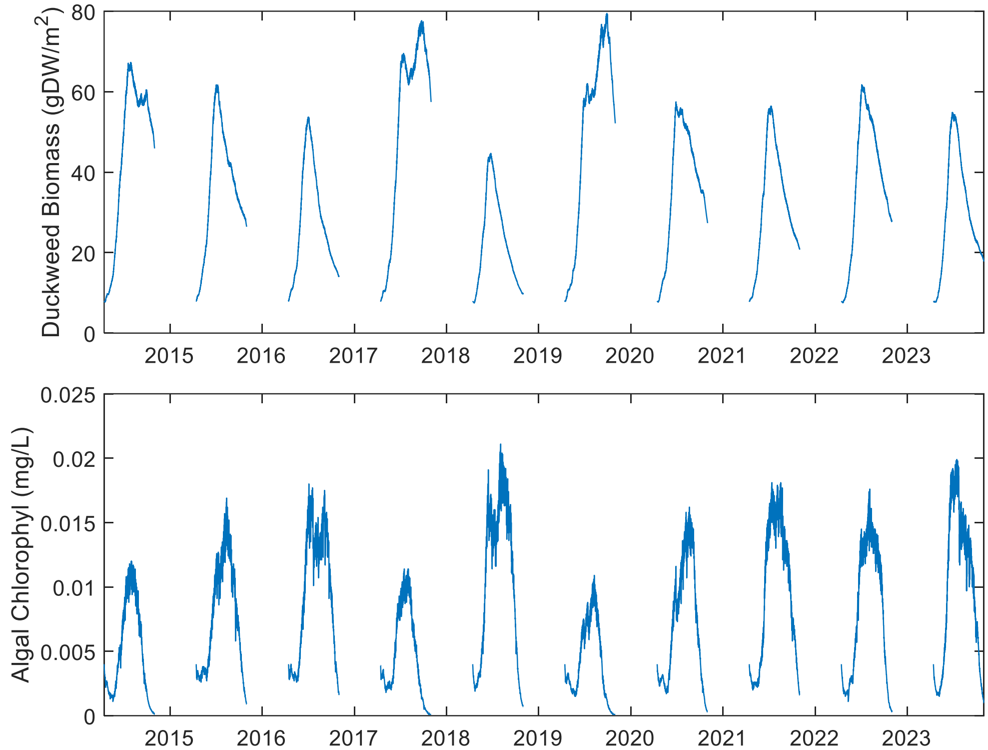


Figure 4-5. Simulated duckweed biomass and algal concentration over 10 years in Alameda historic wetland.

Figure 4-6. Simulated annual total phosphorus export from the Alameda historic wetland with and without duckweed growth.

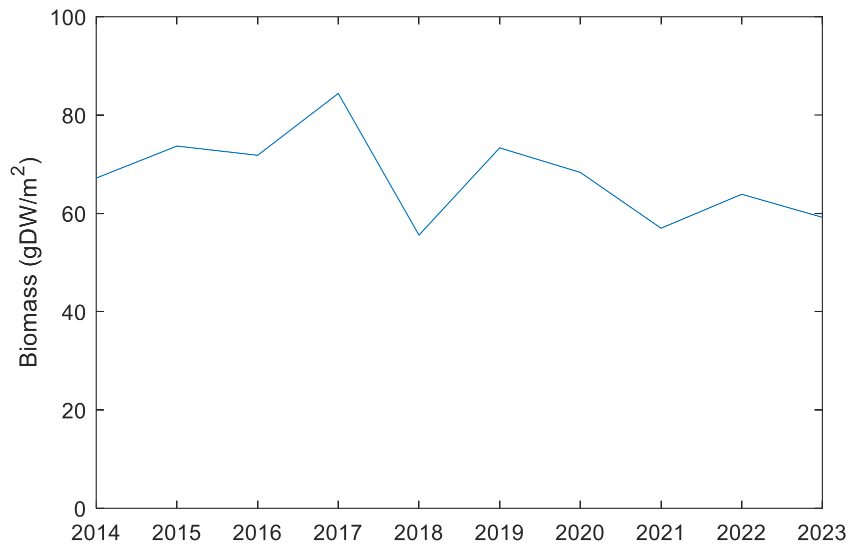


Figure 4-7. Simulated mean annual rooted plant biomass (in grams dry weight per square meter) for LP-53, for 2014-2023.

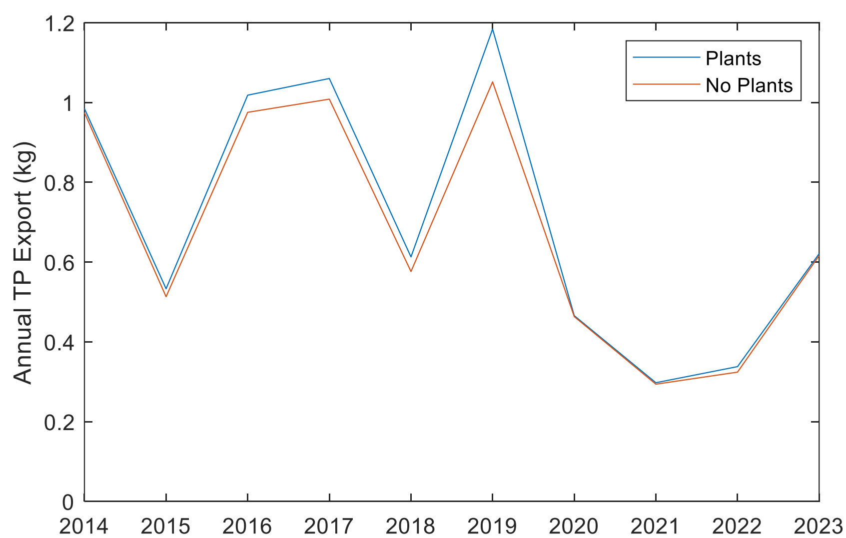


Figure 4-8. Simulated annual total phosphorus export from LP-53 with and without rooted plant growth.

4.4.2 Chemical Sediment Treatment

Pond and historic wetland simulations were run for a range of anoxic sediment release rates, using a scaling factor to reduce the nominal release rates assumed for each pond and historic wetland (Table 4-2). If the dissolved oxygen at the sediment-water interface was not anoxic (DO >1 mg/L), it was assumed that there was no release of TP from sediments, which explains the low TP export at Camden Central and Briarcroft. The simulated variation of TP export and in-pond phosphorus concentration with sediment release rate is shown in Figure 4-9 for each of the four ponds and historic wetlands. These results were then interpolated to give expected changes in phosphorus export for 1) iron filings treatment and 2) alum treatments, as summarized in Table 4-1 and Table 4-2. Iron filings treatments reduced phosphorus export by 8.3 to 18.0%, while alum treatments reduced phosphorus export by 11.3 to 25.9%. The biggest reductions in TP export were for Alameda and LP-53, which had the highest nominal release rates (Table 4-1).

Table 4-1. Assumed anoxic sediment release rates for the untreated pond/historic wetland and for iron filings and alum treatments (mg/m²/day).

	Alameda	Camden Central	Canterbury Oaks	Shoreview Commons	LP-53	Briarcroft
Original	7.52	3.97	3.97	3.18	7.10	5.0*
Iron	3.38	1.75	1.75	1.43	3.10	2.2
Alum	1.73	0.87	0.87	0.73	0.01	1.1

*No lab data available, estimated

Table 4-2. Simulated total phosphorus export for each sediment treatment and the %change in export from the original case (i.e., before treatment).

Case	Parameter	Alameda	Camden Central	Canterbury Oaks	Shoreview Commons	LP-53	Briarcroft
Original	TP out (kg/year)	9.4	16.4	1.4	10.5	0.55	9.2
Iron	TP out (kg/year)	7.7	14.7	1.2	9.3	0.47	8.4
	% Change	-18.0	-9.8	-16.1	-11.6	-13.7	-8.3
Alum	TP out (kg/year)	7.2	14.1	1.1	8.8	0.4	8.2
	% Change	-23.5	-13.9	-21.9	-16.2	-25.9	-11.3

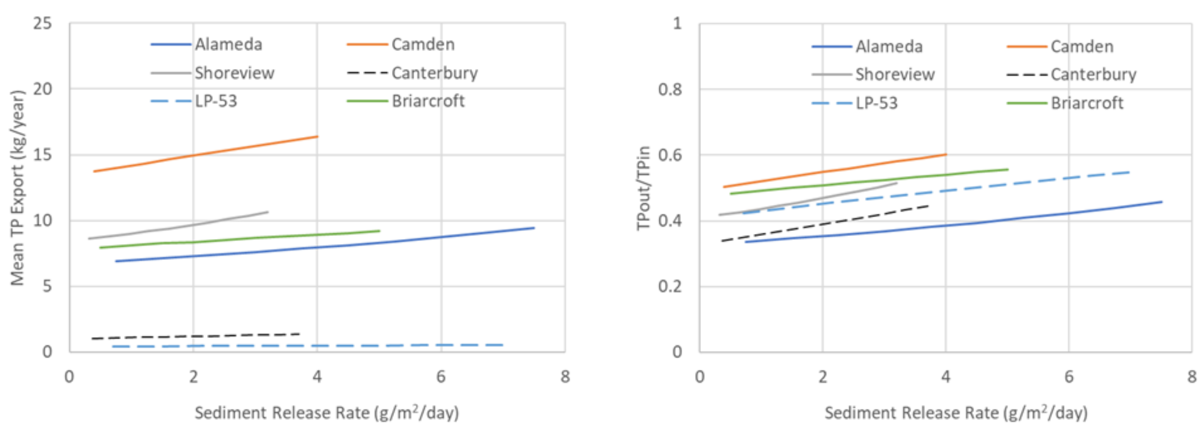


Figure 4-9. Mean annual total phosphorus export and ratio of export to import for varying anoxic sediment release rates. The highest plotted release rate value for each pond/historic wetland is the assumed nominal (original) value for that pond/historic wetland, as given in Table 2-1.

4.4.3 Mechanical Aeration

The pond/historic wetland simulations did show that the addition of bubble-plume type aeration increase mean mixed layer depth, as expected (Figure 4-10), with greater increases in the smaller water bodies (Canterbury Oaks, Briarcroft) for a given aerator flow rate. However, the aerator was not found to be effective in reducing phosphorus export for LP-53 and Briarcroft, while aeration did reduce TP export in the other four ponds and historic wetlands, by 7.1% (Shoreview Commons) to 27% (Camden Central), as summarized in Table 4-3. Figures 4-10 and 4-11 show data for aeration flow rates up to 40 l/s – higher flow rates were simulated to induce complete mixing in ponds and historic wetlands, but the trends in TP export did not change substantially.

The mean August vertical profiles of DO and TP, provided in Figure 4-12 for LP-53 and Camden Central, indicate the impact of mixing the water column. Dissolved oxygen concentration was impacted in both LP-53 and Camden Central, but TP was substantially impacted in Camden Central. Aeration had little effect on TP in LP-53 because there was a low anoxic factor and little release of phosphorus from the sediments.

For the ponds and historic wetlands that did not respond as well to aeration, this is likely because while the aerator increases mixed layer depth, it also transports high TP water up from the bottom, where it is more likely to be exported through the outlet. LP-53 and Briarcroft, with an increase TP export with aeration, both had macrophytes and they were also the deepest water bodies simulated (Table 2-1).

At the assumed 1% mixing efficiency, only two of the ponds and historic wetlands became well mixed for aerator air flow rate of up to 10 L/s at standard pressure (1 atmosphere) and temperature (25 °C). However, the results given in Table 4-3 for the Zic/Stefan aerator model tell a somewhat different story. The 13.5% reduction in TP export in Alameda was achieved using a single diffuser with 5 CFM (2.4 l/s) flow rate, about 5X less air flow than the flow rate required for the same TP reduction using the simple aerator model (25 l/s). Similarly, the 8.8% reduction in TP export at Camden Central was achieved using 2 diffusers at 5 CFM each (4.8 l/s total), compared to 21 l/s required (4.4X higher) to obtain an 8.8% reduction using the simple aerator model. This suggests that a mixing efficiency closer to 4 to 5% may be appropriate in the simple aerator model. The efficiency of aerator mixing for shallow water bodies needs to be further investigated, as well as the interactions between aerators, duckweed, and macrophytes.

Table 4-3. Summary of % change in pond/historic wetland TP export for aeration compared to no aeration. Results for the simple aerator model are given for an air flow rate sufficient for complete mixing, while the results for the Zic/Stefan model are for select flow rates that achieved partial mixing of the ponds/wetlands.

Pond/Historic Wetland	Simple Aerator Model		Zic/Stefan Aerator Model	
	Change in TP Export (%)	Air Flow Rate (l/s)	Change in TP Export (%)	Air Flow Rate (l/s)
Alameda	-16.7	80	-13.5	2.4
Briarcroft	+2.1	25		
Camden Central	-27.0	50	-8.8	4.8
Canterbury Oaks	-10.8	35		
LP-53	+8.4	120		
Shoreview Commons	-7.1	80		

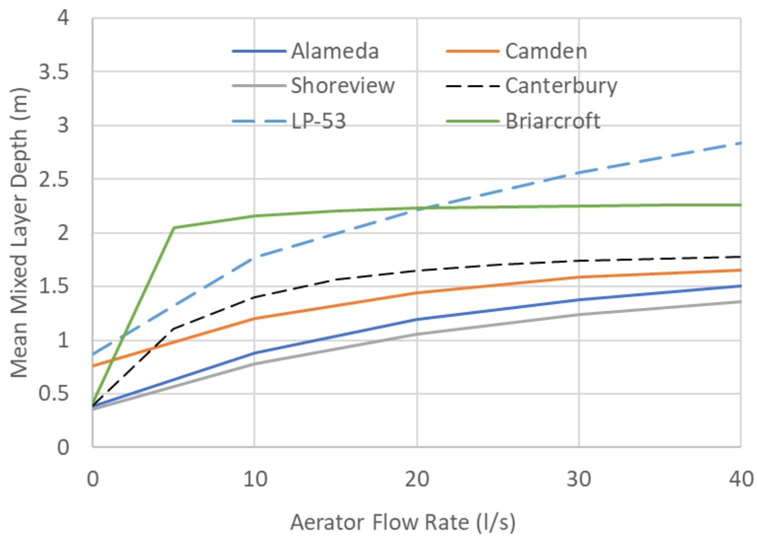


Figure 4-10. Ten-year average mixed layer depth versus aerator flow rate for the four ponds and historic wetlands simulated. For reference, a commercial aerator was specified for Alameda with a total flow rate of 4.8 CFM = 2.2 l/s.

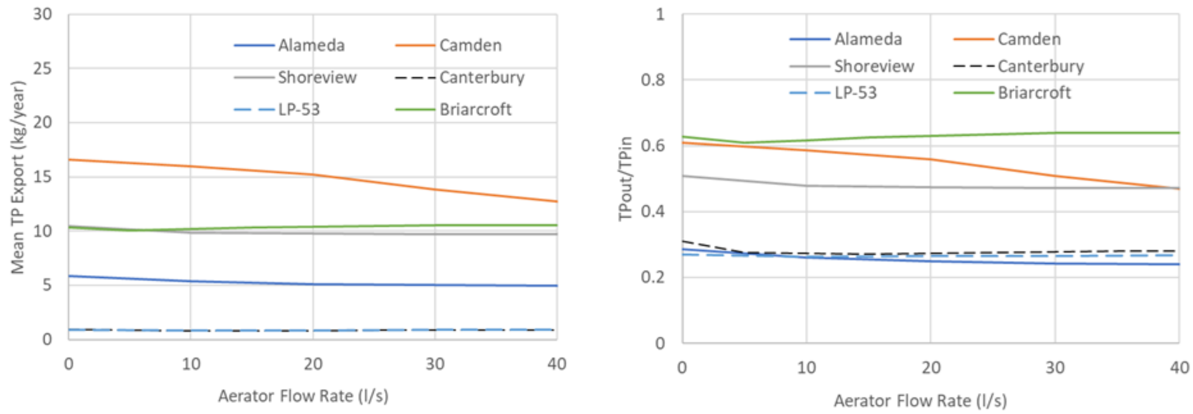


Figure 4-11. Ten-year average TP export versus aerator flow rate and TP input/TP export ratio versus aerator flow rate for the four ponds and historic wetlands simulated.

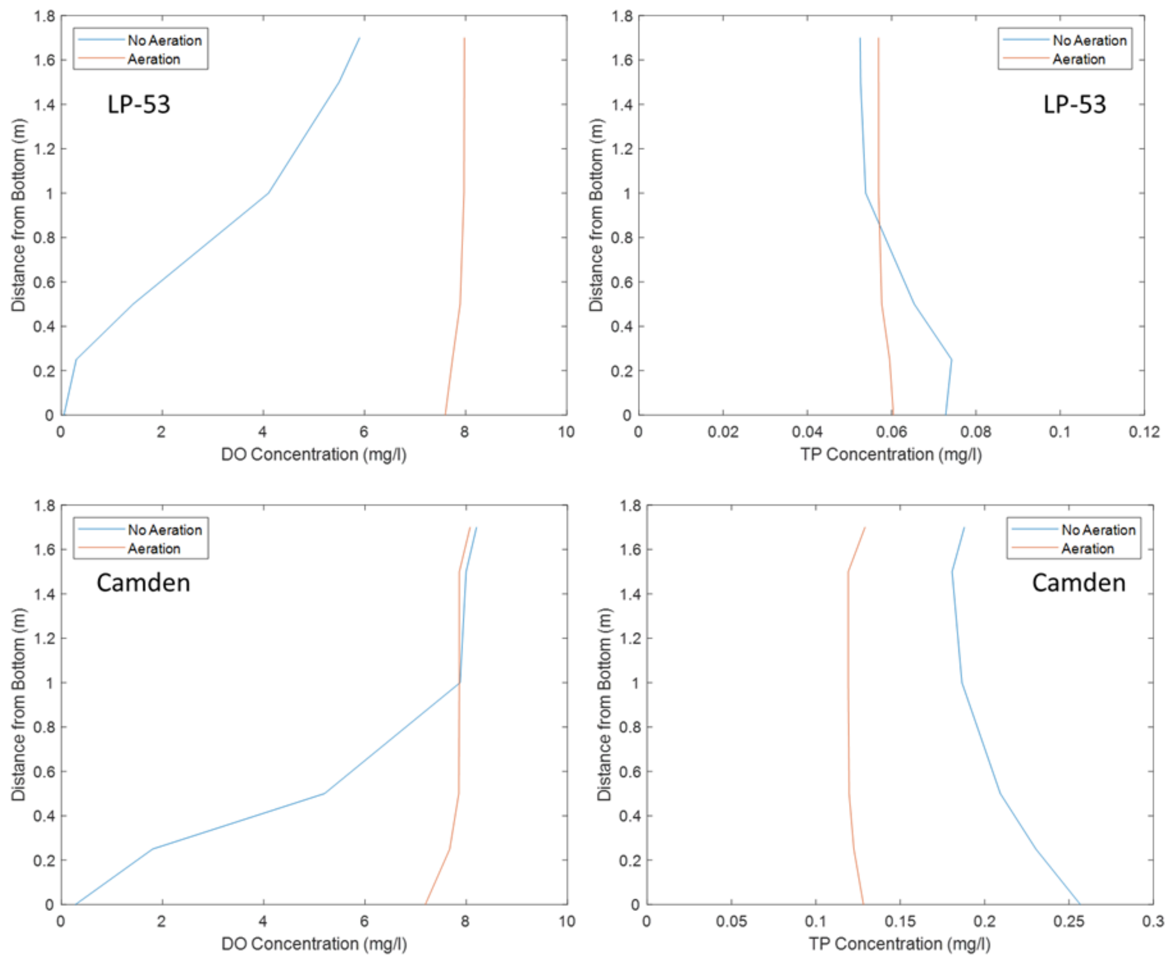


Figure 4-12. Mean August dissolved oxygen and TP profiles for aerated and un-aerated conditions in LP-53 and Camden Central.

4.4.4 Wind Sheltering Reduction

The simulated response of the four ponds and historic wetlands to changes in wind sheltering are shown in Figure 4-13 and Figure 4-14. The response of mixed-layer depth and plant biomass are given in Figure 4-13. For the algae-dominated pond (Camden Central), the mixed layer depth reduces with increasing wind sheltering, an expected result. The unexpected, slight positive slope of mixed layer depth with wind sheltering on the duckweed covered historic wetlands (Alameda, Canterbury Oaks, Shoreview Commons) and the ponds and historic wetlands with rooted macrophytes (LP-53, Briarcroft) can be explained as follows:

- As sheltering increases, the surface water temperature increases due to less convective cooling and evaporation.
- Higher surface temperature reduces duckweed and rooted plant biomass (Figure 4-13), as more days are above the optimum growth temperature.
- The reduced duckweed and rooted plant biomass promote greater wind mixing, count-acting the decrease in wind mixing energy with increased sheltering.

It should be noted that the response of duckweed-covered ponds and historic wetlands to changes in wind sheltering assumes that the duckweed remains as wind sheltering is lowered. For larger ponds and historic wetlands, in particular, reduced wind sheltering may reduce duckweed coverage or eliminate it completely, but an example of this has not been observed in the field to quantify the wind conditions required to suppress duckweed.

Figure 4-14 gives results for the change in the pond/historic wetland 10-year mean TP export over a range of wind sheltering factors, along with the TP export normalized to the TP input to each pond/historic wetland. All ponds and historic wetlands show slight decreases in TP export as wind sheltering is decreased (and wind mixing increases).

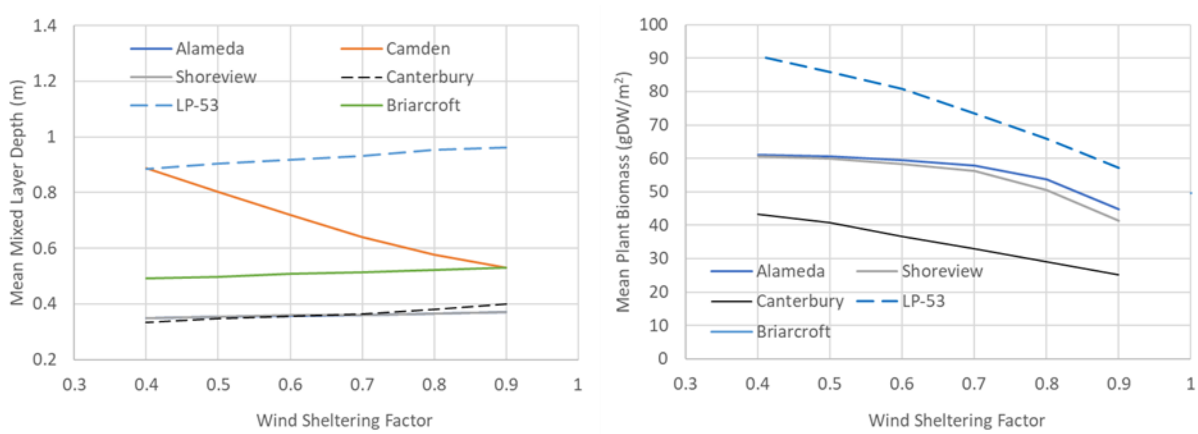


Figure 4-13. Ten-year average mean mixed layer depth versus wind sheltering factor and mean plant biomass versus wind sheltering factor for the six ponds and historic wetlands simulated. Higher wind sheltering factor implies lower wind speeds over the pond/historic wetland. Plant biomass is duckweed for Alameda, Shoreview Commons, and Canterbury, and is rooted plants for LP-53 and Briarcroft.

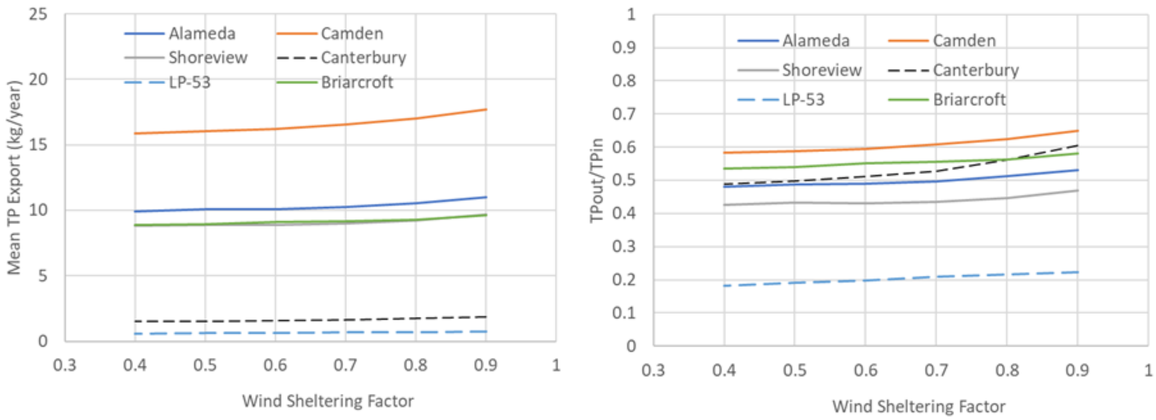


Figure 4-14. Ten-year average TP export versus wind sheltering factor (left panel) and TP input/TP export ratio versus wind sheltering factor (right panel) for the six ponds and historic wetlands simulated. Higher wind sheltering factor implies lower wind speeds over the pond/historic wetland.

4.4.5 Watershed Load Reduction

The response of pond and historic wetland phosphorus export to reductions in watershed loading were simulated for both volume reduction (e.g. adding infiltration practices) and nutrient concentration reduction (e.g. street sweeping). Both volume reduction and nutrient concentration reduction were effective at reducing TP export (Figure 4-15) in all ponds and historic wetlands. A 25% reduction in inflow volume (corresponding to a volume reduction factor of 0.75) produced reductions in TP export of 21 to 33%, while a 25% reduction in nutrient concentrations (corresponding to a concentration reduction factor of 0.75) produced reductions in TP export of 2.3 to 21.9% (Table 4-4). The response of the ratio of outflow TP to inflow TP was more complex (Figure 4-16). As external loading is reduced, internal loading becomes a larger fraction of the TP export, tending to increase the TP output/input ratio.

Table 4-4. Summary of TP export reductions for reductions in watershed volume and nutrient loads. Negative values of ΔTP represent a reduction in TP export from a pond/historic wetland.

	Alameda	Camden Central	Canterbury Oaks	Shoreview Commons	LP-53	Briarcroft
Nominal TP Export (kg/yr)	9.4	16.4	10.6	1.4	0.55	9.8
TP Export, 25% Volume Reduction (kg/yr)	6.4	11.0	7.7	1.0	0.44	7.2
ΔTP, 25% Volume Reduction (kg/yr)	-3.0	-5.4	-3.0	-0.35	-0.11	-2.5
% TP Reduction	-32.4	-32.9	-27.8	-25.8	-20.9	-26.0
TP Export, 25% Nutrient Reduction (kg/yr)	8.0	13.9	8.7	1.2	0.54	7.6
ΔTP, 25% Nutrient Reduction (kg/yr)	-1.4	-2.5	-1.9	-0.2	-0.01	-2.1
% TP Reduction	-15.4	-15.0	-18.1	-11.0	-2.3	-21.9

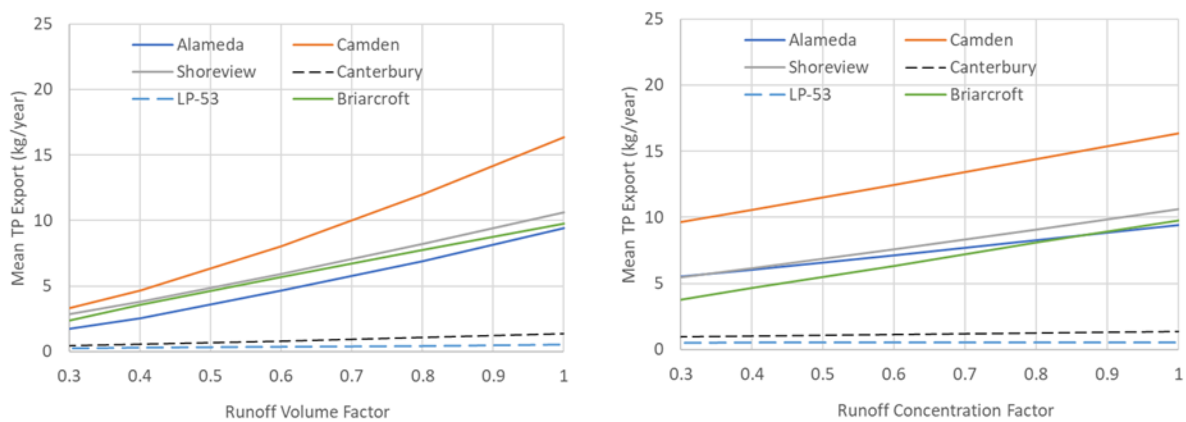


Figure 4-15. Ten-year average TP export versus runoff volume factor (left panel) and versus nutrient concentration factor (right panel) for the six ponds and historic wetlands simulated. The runoff volume factor was used to uniformly scale the input flow rates (leaving inflow concentrations unchanged), while the nutrient concentration factor was used to uniformly scale the inflow nutrient concentrations (leaving inflow volumes unchanged).

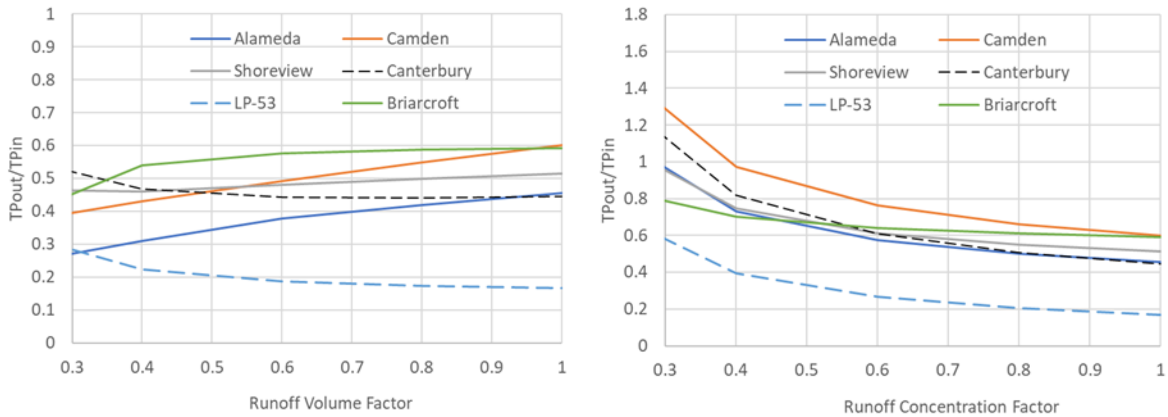


Figure 4-16. Ten-year average TP output/input ratio versus runoff volume factor (left panel) and versus nutrient concentration factor (right panel) for the six ponds and historic wetlands simulated.

4.4.6 Smart Water Level Control

The smart water level controls were simulated for four ponds and historic wetlands to mimic strategies for reducing flooding from large storms, by increasing pond/historic wetland storage prior to storm events of 1.5 inches or greater. This was accomplished by temporarily lowering the elevation of the outlet structure 24 hours prior to each storm of 1.5 inches depth or greater, and then resetting the outlet height just prior to the storm onset. These water level control strategies were found to increase phosphorus export by 6.8, 8.9, 2.7, and 8.0% in Alameda, Camden Central, Shoreview Commons, and Canterbury Oaks, respectively. Examples of the simulated year-year variability in TP export with and without the smart controls are given in Figure 4-17 for Alameda and Camden Central. It is likely that the smart control increases phosphorus export by withdrawing high TP water from deep in the pond and historic wetland during the pre-storm draw down, when the outlet elevation is set to half-depth.

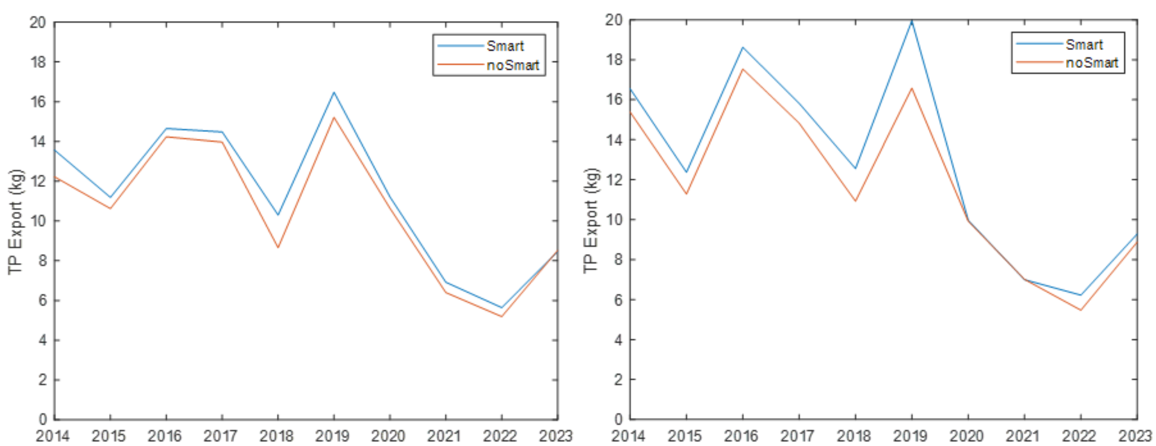


Figure 4-17. Annual total TP export for 10 years of simulations, with and without the smart level control strategy, for Alameda (left panel) and Camden Central (right panel).

4.4.7 Clay Liners, Groundwater, and Baseflow

We examined the effect of a clay liner on Alameda, as Alameda has a net 0.02 cfs (0.6 L.s) outflow of groundwater as part of MinPond water budget calibration. Alameda also has a baseflow at the inlet on the order of 0.1 cfs (2.8 L/s). To generalize the results, we simulated a range of positive and negative groundwater inputs to Alameda and Camden Central, and found these groundwater flows affected phosphorus export rather strongly, as shown in Figure 4-18. If a clay liner were added to Alameda, TP exports are predicted to increase about 18%. Positive values of groundwater input in Figure 4-18 can be interpreted as a pond or historic wetland with baseflow at the inlet, pump inputs (e.g. Camden Central), or ponds and historic wetlands with net groundwater inflow – these cases are predicted to further increase TP export. The results for the algae-dominated Camden Central pond were very similar to the duckweed dominated Alameda historic wetland. Note that this analysis does not take into account potential differences in the sediment nutrient release rates of lined vs. unlined ponds. We did not consider phosphorus entering or exiting the pond via the groundwater flows.

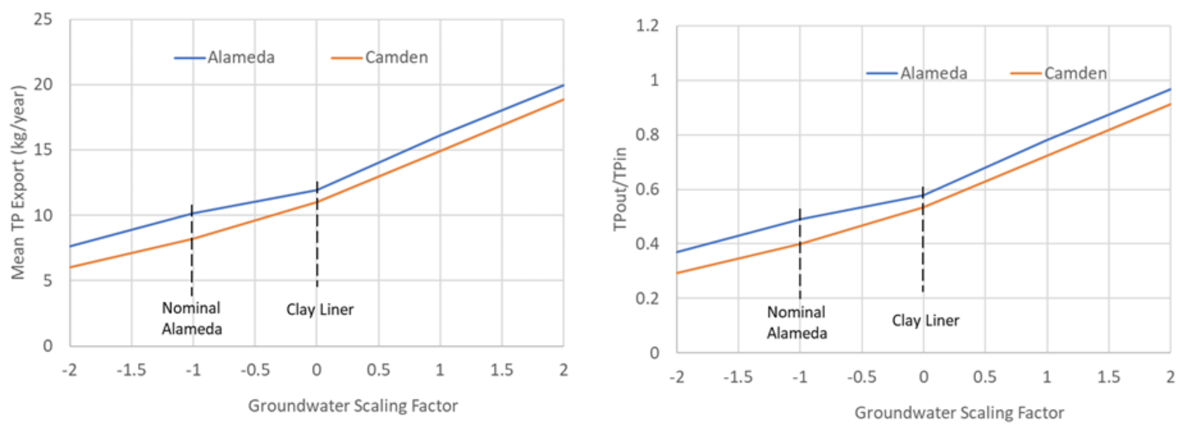


Figure 4-18. Ten-year average TP export versus groundwater input scaling factor (left panel) and TP input/TP export ratio versus groundwater input scaling factor (right panel) for Alameda and Camden Central. The nominal (calibrated) groundwater outflow rate in Alameda is approximately 0.02 cfs (0.6 L/s), equivalent to 0.17 in (4.4 mm) of evaporation per day.

4.5 Cost and performance comparisons of the pond mitigation methods

Costing was not performed for mitigation practices found to be ineffective in reducing pond TP export (wind sheltering reduction, mechanical aeration, smart level control).

This section compares the cost and performance of the pond and historic wetland mitigation strategies that were effective in reducing phosphorus export: chemical sediment treatments and water load reduction. Costing information for chemical sediment treatment and watershed load reduction were adapted from Taguchi et al. 2022, as detailed in Appendix D. Remediating groundwater inputs and baseflow inputs also have the potential to reduce pond and historic wetland TP export, but are too site-specific to cost. Overall, the watershed volume reduction mitigation strategy through installation of infiltration facilities was the most expensive, approaching $30,000/kg P for the larger ponds and historic wetlands and even higher for the small wetland (Canterbury Oaks). The costs associated with LP-53 can be considered an outlier, because the nutrient export and changes in nutrient export for the various treatment practices were so small. The 25% nutrient reduction strategy was the least expensive, ranging from $1,800 to $3,800/kg P. The chemical mitigation strategies have intermediate costs ranging from $3,700 to $8,300/kg P, with Canterbury Oaks somewhat higher. Cost numbers for Canterbury Oaks are presented in Table 4-5, but are likely more uncertain for the relatively small water body and watershed.

Table 4-5. Summary of pond and historic wetland TP export reductions and the associated annual costs over 10 years of simulated phosphorus export.

Case	Parameter	Alameda	Camden Central	Canterbury Oaks	Shoreview Commons	LP-53	Briarcroft
Original Base Case	TP export (kg/year)	9.4	16.4	1.4	10.5	0.55	9.2
Iron Filings Treatment	Cost ($/kg P)	$5,800	$7,800	$27,500	$8,300	$166,300	$4,300
Alum Treatment	Cost ($/kg P)	$7,300	$5,200	$7,300	$5,900	$136,000	$3,700
25% Volume Reduction	Cost ($/kg P)	$13,200	$8,900	$34,000	$13,200	$112,700	$13,700
Intense Street Sweeping	Cost ($/kg P)	$3,600	$2,600	$3,800	$2,600	$76,000	$1,800

4.6 Discussion

MinPond results suggest that reducing wind sheltering and mixing with bubble generators may not lead to reductions in TP export for many ponds and historic wetlands. Although increased mixing energy transports oxygen towards the bottom, and should reduce sediment anoxia and P release, it also transports high P concentration water and BOD upwards in the water column, which can lead to more transport of P through a surface outlet and reduce oxygen levels in the mixed layer. The Camden Central pond, which is phytoplankton dominated, and the Briarcroft pond, which has moderate macrophyte coverage, did show moderate reductions in TP export for both reduced wind sheltering and aeration cases. The 1% mixing efficiency for ponds and historic wetlands calculated from the equations provided by Schladow (1993) for deeper lakes needs further research. The interactions of aerators with duckweed and rooted macrophytes also needs further research – the results of this study indicate that duckweed suppresses wind-driven vertical mixing, but the effect of these plants on aeration is not clear.

A smart control can increase phosphorus export by withdrawing high TP water from deep in the pond or historic wetland during the pre-storm draw down, when the outlet elevation is set to half-depth.

Chemical sediment treatments were found to be effective for reducing TP export from ponds and historic wetlands, by reducing the internal loading of P from sediments. Although the costs of alum and iron filing treatments were estimated to be fairly comparable (on the order of $5000/kg P), more data are needed on the longevity of these treatments to better assess long-term costs.

Both volume reduction (adding infiltration practices) and nutrient reduction through increased street sweeping were found to be effective in reducing TP export from stormwater ponds and historic wetlands, however, street sweeping was found to be more cost effective (~$5000/kg P) than volume reduction (~$13,000/kg P). However, volume reduction strategies are likely to have other benefits not included in this study, such as reduced downstream infrastructure costs and reduced flooding.

Analysis of stormwater ponds and historic wetlands with and without duckweed suggested that the presence of duckweed may moderately increase TP export from a pond or historic wetland (e.g. Figure 4-7). Although the pond or historic wetland includes duckweed growth, senescence, and decay of duckweed over the season, there is very little information available on what happens to duckweed biomass in late fall and over-winter, and how that affects the nutrient budget of a pond or historic wetland. The presence of rooted plants in a pond or historic wetland was also predicted to increase TP export from a pond or historic wetland (Figure 4-9) by drawing nutrients from the sediment during growth and then releasing them in the water column during senescence and decay. However, some species of rooted macrophytes, such as coontail, are known to obtain nutrients from the water column, which may have a quite different effect on the phosphorus budget of a pond or historic wetland.

Chapter 5: Conclusions and Recommendations

The major findings and potential use of the model developed in this project are summarized below:

1) The field data and laboratory column studies (Chapter 2) completed as a part of this project filled gaps in the understanding of sediment phosphorus release and high phosphorus concentration risk factors in ponds. Phosphate (SRP) is the primary limiting nutrient in stormwater ponds and historic wetlands that receive stormwater runoff. Total phosphorus is important because 1) it will be degraded into phosphate by microbes in the water and sediment and 2) it is indicative of phosphate that has been taken up by plant matter in the pond or historic wetland.
2) Floating plant cover (duckweed and water meal) appears to enhance stratification in ponds. This is supported by comparing Camden Central (without floating plants) to Alameda and Canterbury Oaks (Table 2-2). The surface layer in Camden Central was generally larger than in Alameda and Canterbury Oaks, but this did not show up in the anoxic factor (fraction of a period in which DO was less than 1 mg/L in the bottom waters), where Camden Central had a similar anoxic factor to Shoreview Commons and Canterbury Oaks. Duckweed cover may influence DO dynamics in ponds/historic wetlands, but not necessarily across the entire water column. This effect was exacerbated in the 2023 season, when there were less frequent stormwater inputs to provide direct mixing, and the high chloride concentration in bottom waters persisted throughout the summer, which exasperated stratification. In both the 2023 and 2024 field seasons, higher duckweed cover in Alameda and Canterbury indicated anoxia, while duckweed-free Camden Central and LP-53 were more well mixed and oxic.
3) Partial mixing events appear to be relatively frequent in Camden Central and LP-53 and less frequent in Alameda, indicating the importance of sheltering, size of the pond and atmospheric heat flux in causing partial mixing of the ponds (solar radiation heating the pond during the day, back-radiation cooling the pond surface at night, producing convective mixing). In a dry year such as 2023, the role of infrequent stormwater runoff events for water column mixing and P dynamics in the ponds was made more apparent. During wet periods, at least partial mixing was observed in all the ponds. Wind mixing events appeared to be less frequent in the stratified and sheltered ponds, with intermittent mixing (from wind or heat flux) observed in the open ponds (Camden Central). Partial mixing, however, did not substantially reduce the anoxic factor in the Camden Central pond, which means that it did not substantially reduce phosphorus flux from the sediments.
4) The modeling effort undertaken in this study re-emphasized the **cost-effectiveness of street sweeping** as a means to reduce TP export from stormwater ponds and historic wetlands to receiving water bodies (Table 4-5 and Section 4.4.5). The first 25% reduction in TP input to the pond was simulated, a fairly low value for intensive street sweeping efforts. The cost/kg of TP reduction in pond export was as low as $1800 (for Briarcroft) and always below the other options.
5) The cost effectiveness of **chemical sediment treatment** (Table 4-5 and Section 4.4.2) was found to be the next best option. Alum treatment was slightly more cost-effective than iron filings in five out of six cases, and preliminary observations indicated that alum treatment may have a longer effective life, although possibly by only a few years.

6) Reducing wind sheltering and mixing with bubble generators may not lead to reductions in TP export for many ponds and historic wetlands (Section 4.4.3 and Section 4.4.4). Although increased mixing energy transports oxygen toward the bottom, and should reduce sediment anoxia and P release, it also transports high P concentration water and BOD upward in the water column, which can lead to more transport of P through a surface outlet and reduce oxygen levels in the mixed layer. Two of six ponds/historic wetlands did show moderate reductions in TP export for both reduced aeration cases.
7) Microbubble/nanobubble oxygen diffusers that act as an oxygen source near the sediment but do not mix the pond or historic wetland may be more effective than standard bubble diffusers in reducing phosphorus export from stormwater ponds and historic wetlands – the effect would be expected to be similar to chemical sediment treatments (Section 4.4.2).
8) The volume-reduction (adding infiltration practices) strategy was found to be effective in reducing TP export from stormwater ponds and historic wetlands but was found to be the least cost-effective of the four strategies (~$13,000/kg P). However, volume reduction strategies are likely to have other benefits not included in this study, such as reduced downstream infrastructure costs and reduced flooding.
9) Smart water level control strategies designed for flood mitigation were found to be ineffective for reducing phosphorus export from ponds and historic wetlands and instead may increase phosphorus export. It is likely that the smart control increases phosphorus export by withdrawing high TP water from deep in the pond or historic wetland during the pre-storm draw down, when the outlet elevation is set to half-depth.

5.1 Recommendations

Based on the findings summarized above and documented in this report, as well as the knowledge gained in this project, we make the following recommendations in decreasing order of cost-effectiveness:

7) A major water-quality goal of stormwater ponds and historic wetlands that receive stormwater runoff is to reduce phosphorus exported to receiving water bodies. To achieve this water-quality goal, the manager needs to reduce inputs of phosphorus to the pond/historic wetland, which comes from watershed runoff and can also come from internal loading from the sediments.
8) An intense street sweeping regime is the most cost-effective means of reducing phosphorus export from a stormwater pond or historic wetland that serves as a stormwater practice by decreasing the load coming from the watershed. If a community does not have an intense street sweeping program and wants to remediate high phosphorus concentrations discharging into receiving waters, this is a cost-effective means of doing so.
9) If the stormwater pond or historic wetland is stratified, with high concentrations of phosphorus below the surface layer, an outlet designed to avoid transporting the high phosphorus concentrations to the receiving water body is important. An outlet should be designed to avoid the release of both floating plants and bottom waters.

10) Chemical treatment of the sediments is another effective means of reducing phosphorus export only when the phosphorus release from the sediments is substantial. A sediment release study will inform on this topic. The sediments will need to be retreated every five to ten years, due to the high organic load that these ponds and historic wetlands experience.
11) Oxygen addition to the pond or historic wetland via microbubbles will reduce phosphorus release from the sediments if the oxygen addition is sufficient.
12) Bubble aeration that is successful in destratifying the water column can bring oxygen down to the sediments and reduce a high phosphorus release from the sediments. However, if the aerator does not fully destratify the water column, the result could be a higher phosphorus export, as the vertical transport of phosphorus will be enhanced by the aerators.

References

APHA. (1995). *Standard methods for the examination of water and wastewater*, 19th Ed. American Public Health Association (APHA), the American Water Works Association (AWWA), and the Water Environment Federation (WEF, former Water Pollution Control Federation or WPCF), Washington, DC.

Baker, L., Kalinowsky, P., Hobbie, S., Bintner, R., & Buyarski, C. (2014, March/April). Quantifying nutrient removal by enhanced street sweeping. *Stormwater Solutions*. Retrieved from StormH20.com

Capitol Region Watershed District. (n.d.). Water data reporting tool. CRWD. http://waterdata.capitolregionwd.org/applications/login.html?publicuser=Guest#waterdata/stationoverview

Chambers, P. A., Prepas, E. E., Bothwell, M. L., & Hamilton, H. R. (1989). Roots versus shoots in nutrient uptake by aquatic macrophytes in flowing waters*. Canadian Journal of Fisheries and Aquatic Sciences, 46*(3), 435-439.

Erickson, A. J., Herb, W. R., Gallagher, N. D., Weiss, P. T., Wilson, B. N., & Gulliver, J. S. (2023). *Climate change adaptation of urban stormwater infrastructure* (Final report No. 2023-21). Minnesota Department of Transportation, St. Paul, MN.

Fang, X., & Stefan, H. G. (2009). Simulations of climate effects on water temperature, dissolved oxygen, and ice and snow covers in lakes of the contiguous US under past and future climate scenarios. *Limnology and Oceanography, 54*, 2359-2370.

Finlay, J. C., Janke, B. D., Trojan, M., Wilson, B., & Marek-Spartz, M. (2024). *Leveraging Minnesota's stormwater data for improved modeling and management of water quality in cities (Final report)*. Minnesota Stormwater Research Council, St. Paul, MN. https://hdl.handle.net/11299/259975

Finlay, J. C., Janke, B. D., Natarajan, P., Larkin, D. J., & Distel, J. (2026). *Managing urban pond vegetation to enhance water quality benefits* (Final report). Minnesota Stormwater Research Council, St. Paul, MN. https://wrc.umn.edu/projects/managing-veg

Ford, D. E., & Stefan, H. G. (1980). Thermal predictions using an integral energy model. *Journal of the Hydraulics Division, 106*, 39-55.

Gulliver, J. S. & Stefan, H. G. (1982). Lake phytoplankton model with destratification. *Journal of Environmental Engineering Division, 108*(5), 864-882.

Herb, W. R., & Stefan, H. G. (2003). Integral growth of submersed macrophytes in varying light regimes. *Ecological Modelling, 168*(1-2), 77-100.

Herb, W. R., & Stefan, H. G. (2005). Model for wind-driven vertical mixing in a shallow lake with submersed macrophytes. *Journal of Hydraulic Engineering*, *131*(6), 488-496.

Herb, W.R., B. Janke, O. Mohseni, & Stefan, H.G. (2009). Simulation of temperature mitigation by a stormwater detention pond. *Journal of the American Water Resources Association, 45*(5), 1164-1178.

Herb, W. R., Janke, B. D., & Stefan, H. G. (2017). *Study of de-icing salt accumulation and transport through a watershed.* Minnesota Department of Transportation, Saint Paul, MN.

Holgerson, M. A., Zappa, C. J., & Raymond, P. A. (2016). Substantial overnight reaeration by convective cooling discovered in pond ecosystems. *Geophysical Research Letters, 43*(15), 8044-8051.

Hootsmans, M. J. M. (1994). A growth model for *Potamogeton pectinatus* L. In W. van Vierssen, M. J. M. Hootsmans, & J. Vermaat.(Eds.), *Lake Veluwe, a macrophyte dominated system under eutrophication stress* (pp. 250–286). Kluwer Academic Publishers, Netherlands.

Huser, B. J., Egemose, S., Harper, H., Hupfer, M., Jensen, H., Pilgrim, K. M., Reitzel, K., Rydin, E., & Futter, M. (2016). Longevity and effectiveness of aluminum addition to reduce sediment phosphorus release and restore lake water quality. *Water Research, 97*, 122–132. https://doi.org./10.1016/j.watres.2015.06.051

Janke, B. D., Natarajan, P., Shrestha, P., Taguchi, V. T., Finlay, J. C., & Gulliver, J. S. (2021). *Detecting phosphorus release from stormwater ponds to guide management and design* (Project report No. 597). MN Stormwater Research Council, St. Paul, MN. conservancy.umn.edu/handle/11299/218852pdf

Janke, B. D., Finlay, J. C., Taguchi, V. J., & Gulliver, J. S. (2022). Hydrologic processes regulate nutrient retention in stormwater detention ponds. *Science of the Total Environment, 823,* 153722. https://doi.org/10.1016/j.scitotenv.2022.153722

Janke, B., Natarajan, P., Gulliver, J. S. & Finlay, J. C., (2023). *Stormwater pond maintenance, and wetland management for phosphorus retention* (MnDOT Report No. 2023-25). Minnesota Department of Transportation, St. Paul, MN. https://hdl.handle.net/11299/256430

Jensen, H. S., & Andersen, F. O. (1992). Importance of temperature, nitrate, and pH for phosphate release from aerobic sediments of four shallow eutrophic lakes. *Limnology and Oceanography, 37*(3), 577-589.

Kalinosky, P., Baker, L. A., & Hobbie, S. E. (2014). *User support manual: Estimating nutrient removal by enhanced street sweeping. Prepared for Minnesota Pollution Control Agency and U.S. Environmental Protection Agency.* https://wrl.mnpals.net/islandora/object/WRLrepository%3A2446

Lasfar, S., Monette, F., Millette, L., & Azzouz, A. (2007). Intrinsic growth rate: A new approach to evaluate the effects of temperature, photoperiod and phosphorus–nitrogen concentrations on duckweed growth under controlled eutrophication. *Water Research, 41*(11), 2333-2340.

Moore T. L., Gulliver, J. S., Stack, L., & Simpson, M. H. (2016). Stormwater management and climate change: Vulnerability and capacity for adaptation in urban and suburban contexts. *Climatic Change, 138(*3), 491-504.

Morris, P. F., & Barker, W. G. (1977). Oxygen transport rates through mats of Lemna minor and Wolffia sp. and oxygen tension within and below the mat. *Canadian Journal of Botany, 55*(14), 1926-1932.

Minnesota Pollution Control Agency (MPCA). (2021). Pond inventory for stormwater manual. MPCA. https://stormwater.pca.state.mn.us/index.php?title=File:Pond_Inventory_for_Stormwater_Manual.xlsx

Natarajan, P., Gulliver, J. S., & Arnold, W. A. (2017). *Internal phosphorus load reduction with iron filings*. (SAFL project report No. 582). University of Minnesota, Minneapolis, MN. http://hdl.handle.net/11299/195677

Natarajan, P., & Gulliver, J. S. (2022). *Assessment of internal phosphorus release and treatment with iron filings in five RPBCWD ponds* (SAFL project report No. 601). University of Minnesota, Minneapolis, MN. https://conservancy.umn.edu/handle/11299/228409

Natarajan, P. N., Janke, B. D., & Finlay, J. C. (2025). *Enhancement and validation of a stormwater pond assessment tool* (Final report). Minnesota Stormwater Research Council, St. Paul, MN. https://hdl.handle.net/11299/273518

Oberts, G. L. (1998). Long-term reductions in removal effectiveness: Lake McCarrons wetland treatment system. E*ngineering Approaches to Ecosystem Restoration Conference Proceedings: 1998 Wetlands Engineering & River Restoration Conference, Denver, CO, 628-633.*

Psenner R., & Puckso R. (1988). Phosphorus fractionation: Advantages and limits of the method for the study of sediment P origins and interactions. *Archiv für Hydrobiologie. Beiheft Ergebnisse der Limnologie, 30,* 43–59.

Pokorný, J., & Rejmánková, E. (1983). Oxygen regime in a fishpond with duckweeds (Lemnaceae) and Ceratophyllum. *Aquatic Botany, 17*(2), 125-137.

Richardson, D., Holgerson, M., Bortolotti, L., Bodmer, P., Daly, J., Edwards, D., Gallagher, K., Gannon, K., Gavin, A., Hovel, R., Finlay, J., Finlay, K., Hornbach, D., Janke, B., Leone, A., & Mesman, J. (2026). Ponds are neither dimictic nor polymictic: A variable transition season between stratified summers and mixed autumns. *Limnology and Oceanography Letters*. (In Review).

Riley, M. J., & Stefan, H. G. (1987). *Dynamic lake water quality simulation model "MINLAKE"* (SAFL Project Report No. 263). University of Minnesota, Minneapolis, MN.

Riley, M., & Stefan, H. G. (1987). MINLAKE: A dynamic lake water quality simulation model. *Ecological Modeling, 43,* 155-182.

Ronkainen, F. (2021). Investigation effects of Lemna removal on oxygen and phosphorus recovery in urban ponds. *Minnesota Undergraduate Research & Academic Journal, 4*(5). https://pubs.lib.umn.edu/index.php/muraj/article/view/3665/2780

Ryan, P. J., Harleman, D. R., & Stolzenbach, K. D. (1974). Surface heat loss from cooling ponds. *Water Resources Research, 10*(5), 930-938.

Schladow, S. G. (1993). Lake destratification by bubble-plume systems: Design methodology. *Journal of Hydraulic Engineering, 119*(3), 350-368.

St. Croix Watershed Research Station (SCWRS). (2010). *Sediment phosphorus extraction procedure high sample throughput*. D.R. Engstrom, modified by Robert Dietz and Michelle Natarajan (2015). St. Croix Watershed Research Station, Marine on St. Croix, MN.

Stantec. (2022). *Pond survey, maintenance assessment and internal loading analysis.* Stantec Consulting Services, Inc., Golden Valley, Minnesota.

Stefan, H., & Ford, D. E. (1975). Temperature dynamics in dimictic lakes. *Journal of the Hydraulics Division*, *101*(HYl), 97-114.

Steinman, A. D., & Spears, B. M. (2019). *Internal phosphorus loading in lakes: Causes, case studies, and management.* J. Ross Publishing.

Taguchi, V., Olsen, T., Janke, B., Stefan, H. G., & Finlay, J. (2018). *Stormwater research priorities and pond maintenance, Objective 3: Phosphorus release from stormwater ponds*. Minnesota Pollution Control Agency, St. Paul, MN.

Taguchi, V. J., Olsen, T. A., Natarajan, P., Janke, B. D., Gulliver, J. S., Finlay, J. C., & Stefan, H. G. (2020). Internal loading in stormwater ponds as a phosphorus source to downstream waters. *Limnology and Oceanography Letters, 5*, 322-330.

Taguchi, V. J., Janke, B. D., Herb, W. R., Gulliver, J. S., Finlay, J. C., & Natarajan, P. (2022). *Wet pond maintenance for phosphorus retention* (Final report No. 2022-20). Minnesota Department of Transportation, St. Paul, MN. https://hdl.handle.net/11299/227893

U.S. Bureau of Labor Statistics. (2021). CPI inflation calculator. US BLS. https://www.bls.gov/data/inflation_calculator.htm

Van Gerven, L. P., de Klein, J. J., Gerla, D. J., Kooi, B. W., Kuiper, J. J., & Mooij, W. M. (2015). Competition for light and nutrients in layered communities of aquatic plants. *The American Naturalist, 186*(1), 72-83.

van Marrewijk, B. (2017). Modelling the production of duckweed in combination with economic possibilities (Doctoral dissertation, BSc thesis), Wageningen UR.

Wells, S. A. (Ed.). (2019). *CE-QUAL-W2: A two-dimensional, laterally averaged, hydrodynamic and water quality model, version 4.2, user manual*. Department of Civil and Environmental Engineering, Portland State University.

Weiss P. T., Gulliver J. S., & Erickson A. J. (2007). Cost and pollutant removal of storm-water treatment practices. *Journal of Water Resources Planning and Management, 133*(3), 218-229.

West, D., & Stefan, H. G. (1998). *Simulation of lake water quality using a one-dimensional model with watershed input. Model description and application to Lake Riley and Lake Elmo* (Project report No. 430). University of Minnesota, Minneapolis, MN.

Zic, K., & Stefan, H. G. (1989). *Lake destratification by air bubble columns* (Project report No. 265). University of Minnesota, Minneapolis, MN.

Appendix A

Pond Classification System

1 Introduction

The purpose of this appendix is to describe a classification system for phosphorus retention in ponds receiving stormwater runoff. To do this, we analyzed the urban pond data from the current and past projects to identify controls and indicators of pond function, and used this information to develop a stormwater pond classification system. **This classification focused on identification of pond types with similar total phosphorus (TP) concentration (an indicator of P retention) and on pond maintenance approaches as a resource for pond management and decision making.** The focus and potential application of the pond classification system is therefore a companion to the already-developed Pond Assessment Tool (Janke et al. 2023, Natarajan et al. 2025), providing a method for a more comprehensive perspective on pond phosphorus. The primary benefit of the classification system would be to facilitate assessment of a large number of pond sites. Ponds could then be scored and ranked using the Pond Assessment Tool to prioritize further monitoring or maintenance. The Pond Assessment Tool and the proposed Pond Classification System were developed for use by practitioners and stormwater managers. Both tools provide a rapid assessment method to identify and screen at-risk ponds and further prioritize those ponds for remediation or other management measures to improve their water quality functionality.

Given the complexity of small ponds, our analysis for Task 5 considered a subset of potential factors related to pond water quality functionality (described in Table A-1) that were originally considered in the Pond Assessment Tool, and the analysis does not address more complex factors related to aquatic vegetation type (e.g., submerged vs. emergent plants) or processes such as pond water column mixing that rely on continuous data collection or monitoring. This larger set of potential factors and analytical approaches are listed in Table A-1 to help guide further work in this area. The impact of various vegetation types and implications for pond classification and management will be addressed by another project (Finlay et al. 2026[6]), with results potentially integrated into the classification system and the Pond Assessment Tool later.

[6] https://wrc.umn.edu/projects/managing-veg

Table A-1. Potential predictors of pond surface water TP concentration, based on results of previous studies. Parameters highlighted in blue were used in the CART analysis; *retained by CART as important for the classification.

Parameter	Description	Method or Data Source	Included in CART
DW	Duckweed cover as fraction of surface area	Field observation	Y*
Area_ac	Pond surface area in acres	Aerial imagery or drawings	Y*
Origin	Constructed Pond or Historic Wetland	From pond owner or from historic imagery	Y*
Age	Year that pond was built, or if a natural waterbody , when it was connected by storm sewers	From pond owner or from historic imagery	Y
Max_d	Max depth in pond in feet	Bathymetry, profile, or from pond owner	Y
Mean_d	Mean depth in pond in feet	Bathymetry, profile, or from pond owner [incomplete]	Y*
Rel_d	Relative depth, another measure of water column stability	Max depth divided by nominal diameter of pond	N
Osgood	Osgood index, measure of water column stability	Mean depth in m divided by square root of surface area in sq. km	N
Topo_cpy_ht	Mean height in m of canopy, buildings, and ground surface relative to pond water surface within 50m buffer	Classified LIDAR point cloud data (2022)	N
Fetch_h	Fetch relative to tree height	Pond nominal diameter divided by Topo_cpy_ht	Y
Cpy_dens	Mean canopy density in a 50m buffer around pond	Canopy Returns / Total Returns in a grid cell (LIDAR point cloud data)	Y
Imp_500	Total impervious percentage in 500m buffer of pond	NLCD 2023 Impervious layer	Y
Cpy_500	Fraction of 500m buffer classified as Canopy	TCMA 1-m Land Cover (UMN 2015)	N
Res_500	Fraction of 500m buffer classified as a Residential land use	2020 Metropolitan Council Generalized Land Use	Y
DW	Duckweed cover as fraction of surface area	Field observation	Y*
Emerg	Emergent vegetation (e.g., cattails) as fraction of pond surface area	Aerial imagery or MNDNR Wetland Inventory [incomplete]	N
SAV	Submerged aquatic vegetation cover as fraction of surface area	Field observation [incomplete]	N

2 Methods

The pond classification system was based on insights from past projects (Taguchi et al. 2018; Janke et al. 2021; Taguchi et al. 2022; Janke et al. 2023), in which we grouped ponds based on characteristics that were found to be important to total phosphorus (TP) concentrations observed in surface waters of the ponds in the dataset. The Pond Assessment Tool (Janke et al. 2023; Natarajan et al. 2025) that resulted from that work included a high-level screening and scoring component that was based on several pond characteristics that could largely be assessed from pond and watershed information, namely duckweed cover, emergent vegetation cover, pond age, watershed land use, shoreline canopy cover, and presence of hydric soils.

For the pond classification system, we refined the previous approach in two ways:

(1) using a recursive *Classification and Regression Tree* (CART) analysis to guide the pond classification, which is a more robust approach compared to the manual characterization used in the original work, and an approach suitable for the complex, non-linear relationships between pond TP and pond or watershed characteristics; and

(2) using a subset of pond data that were primarily collected in our past projects, along with robust collections by the city of Bloomington, which provided more similarity of data collection frequency and methods across sites (n = 66 sites total). This approach contrasts with the larger, more sparsely-collected dataset used in the original Pond Assessment Tool (n ~ 200 sites).

The CART technique attempts to split a dataset into groupings where constituents are more similar based on values of the predictor variables, minimizing the mean squared error between response (surface TP) and predictors (site characteristics) within the two groups at a split. Splits start with the predictor variable of highest correlation with the response and continue to occur on sub-groups in a branching fashion as long as the overall model error decreases. The final tree is typically over-fit, and some approaches recommend selecting a 'pruned' tree (i.e., a version of the tree with fewer splits) that has the lowest cross-validation error as determined by a non-exhaustive, leave-one-out approach. For larger datasets, CART would include splitting a dataset into a model training (80%) and a testing (20%) portion. For further details, the University of Minnesota-Morris includes a high-level overview with a pair of examples[7].

Our relatively small dataset (n = 65 sites) was used solely for model training, as we were most interested in producing a basis for a pond classification, and less interested in predictive capacity. Further, given the sensitivity of CART to small changes in inputs, we used the CART results as a guide rather than as a prescription for classification. Thus, we selected a tree that was not based on minimizing cross-validation error and instead on producing relatively distinct groups based on pond surface TP concentration, which aligned with insights from past results and used a subset of our highest quality input data (both surface TP concentrations and pond parameters).

[7] https://mnstats.morris.umn.edu/multivariatestatistics/cart.html

The CART analysis was carried out using the 'rcart' package in R (Therneau et al. 2025[8]), following guidance from Guild (2025[9]). We used surface total phosphorus (TP) concentration as the response variable, and selected a subset of pond sites and watershed characteristics that were known from previous work to be important predictors of pond surface TP (Table 1). In particular, we expected factors such as duckweed cover, pond depth, pond surface area, pond age, and wind sheltering to be important to pond TP water quality.

Once the ponds had been classified into groups, important sediment and water quality variables with relevance to pond management, such as water column dissolved oxygen (DO), anoxic sediment P flux, sediment organic matter content, sediment TP, sediment bioavailable P, and sediment oxygen demand, were summarized for each split in the classification tree to provide additional context for the splits. These variables were also summarized within each group to provide support for the potential management actions that were recommended to address surface water TP in the pond groups that had moderate or high levels of TP (generally > 0.1 mg/L).

2.1 Datasets

Pond water quality data were assembled from the monitoring efforts of the current project, as well as several past projects by the authors (Taguchi et al. 2018; Janke et al. 2021; Taguchi et al. 2022; Janke et al. 2023). These core data include 42 sites that have been studied for at least one full season (May – October) or have at least five total TP samples between 2016 and 2025. Monitoring typically comprised of bi-weekly sampling of the pond for surface water TP concentration and profiling (temperature, conductivity, and dissolved oxygen), and sediment analysis (lab-determined anoxic sediment phosphate release rate and sediment oxygen demand (SOD), and sediment characteristics like organic matter content, sediment TP, and bioavailable P in sediment at 30 ponds). From 2019 onwards, many of these sites also had field observations of duckweed coverage on the pond surface.

To this core dataset we also added monitoring data collected by several cities with intensive pond monitoring programs that included, at the very least, surface water TP concentration and basic pond information (depth, surface area, age), and potentially some sediment characteristics (organic matter content, anoxic sediment phosphate release rate, bioavailable P). These cities included Bloomington (J. Distel, pers. comm.), Richfield (Mattias Oddsson, pers. comm.), and Minneapolis (Stantec 2022), with 47 sites total among the three cities, bringing the total potential site count to 89.

Pond and watershed characteristics important to TP concentration (also listed in in Table 1):

- pond origin (constructed pond or historic wetland)
- pond age
- pond surface area
- pond mean and/or maximum depth
- topographic and tree canopy sheltering

[8] https://cran.r-project.org/web/packages/rpart/index.html
[9] https://rpubs.com/camguild/803096

- watershed land use and land cover fractions
- duckweed cover fraction.

Pond origin and age were obtained from pond owners and from viewing historical aerial imagery either in Google Earth or from several sources compiled by MN Geospatial Information Services[10]. Pond surface area, if not provided by the pond owners, was determined either from aerial imagery in Google Earth or using the MN DNR's updated wetland inventory map[11]. Pond depth was provided by pond owners, or in some cases, estimated from maximum profile depths during site visits. Watershed land use and land cover fractions were determined in GIS using 500 m buffers around the ponds, per methods in Natarajan et al. (2025). Duckweed cover was taken from direct field observations where possible, which included both our core sites as well as many of the Bloomington ponds, as the city included this in site visit protocols from 2021 onward. Duckweed cover was also estimated from aerial and satellite imagery using Google Earth and Planet Labs, respectively (Eva Forsline and Julia Grabow, unpublished data; Finlay et al. 2026). These cover estimates were used primarily to help fill out duckweed cover estimates for sites with low numbers of field observations.

Sheltering: One new parameter added to this analysis was the estimate of sheltering from topography and tree canopy, which was computed using the latest LIDAR data collected in the state of Minnesota (during 2022 for the Twin Cities metro area[12]). This massive dataset is maintained by the USGS 3DEP program and hosted on Amazon Web Services[13]. Using a Python script adapted from opentopography.org[14], pre-classified LIDAR point cloud data were downloaded for a 50 m buffer around each pond and analyzed to determine the mean height of the land surface (topography), tree canopy, and buildings in the buffer around the pond, relative to the pond water surface elevation, in meters (Figure 1). This metric provided a measure of the pond's entrenchment in the landscape, with relevance to wind sheltering, which has been found to impact oxygen and mixing in the ponds as well as duckweed distribution (Finlay et al.2026). Wind "fetch" was estimated for each pond by normalizing the pond's nominal diameter by this mean topography-plus-canopy height, i.e., expressing the pond's diameter in number of tree heights.

Other parameters with relevance to water quality or water column stability, such as Osgood index (defined as mean depth divided by square root of surface area) and relative depth (defined as max depth divided by nominal diameter of pond), were also included in the analysis but ultimately not proven useful for classification in this dataset. Parameters related to other forms of pond vegetation, such as emergent vegetation cover, lily pads, and submerged aquatic vegetation, while important to pond oxygen and water quality, were not included here as these parameters are being refined and analyzed in another project (Finlay et al. 2026).

---

[10] https://www.mngeo.state.mn.us/chouse/airphoto/compare_characteristics.html
[11] https://gisdata.mn.gov/dataset/water-nat-wetlands-inv-2009-2014
[12] https://www.mngeo.state.mn.us/chouse/elevation/lidar_gen2.html
[13] https://registry.opendata.aws/usgs-lidar/
[14] https://github.com/OpenTopography/OT_3DEP_Workflows

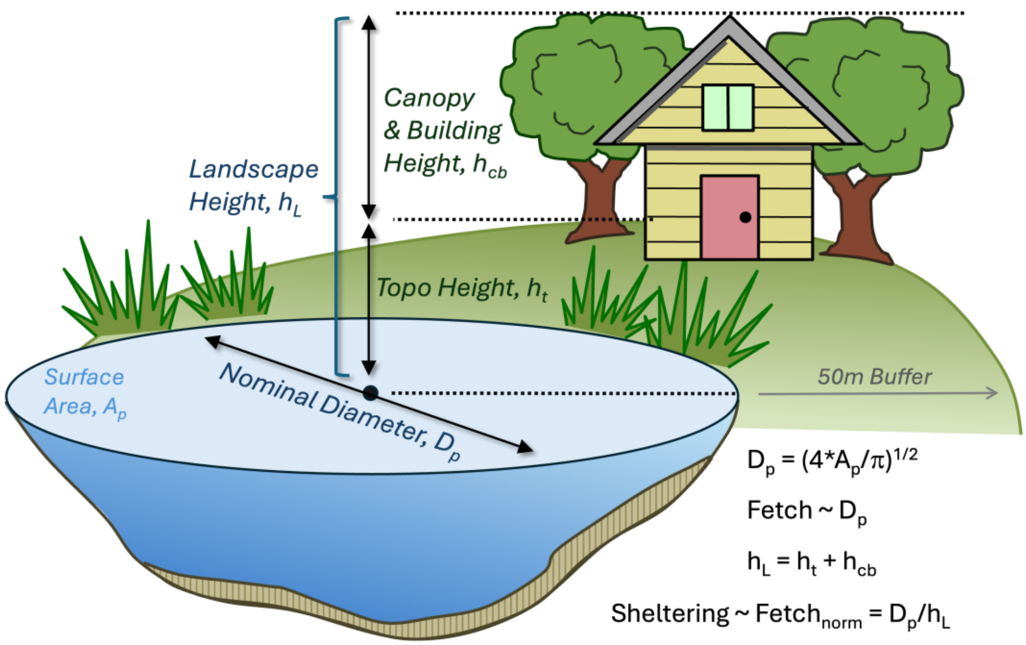


Figure A-1. Schematic illustrating wind sheltering ('normalized fetch') calculation on ponds using LIDAR data to compute height of the land surface, buildings, and trees above the pond's surface. Figure from Finlay et al. (2026).

2.2 Data Processing and Model Setup

Total phosphorus concentrations (TP, mg/L) for each site were first averaged into mean summer (defined here as June – September) and mean season (May – October inclusive) values for each year of data collection. Years without at least 5 samples were then removed; this criteria would exclude all sites without roughly one sample per month to minimize the introduction of bias from a short record. Then all years were averaged together to produce a mean annual summer and mean annual season TP concentration for each site, with each year of data collection assigned equal weights. Duckweed cover (as fraction of pond surface area) and dissolved oxygen concentration (as water column mean concentration in mg/L) were averaged using the same process.

Given the importance of duckweed cover as a predictor or classifying variable (Janke et al. 2022; Natarajan et al. 2025), we excluded sites that did not have duckweed cover observations from either field observations or a similar number of aerial/satellite photos (roughly 5 per season). This approach resulted in the elimination of some Bloomington sites; we further excluded the Minneapolis and Richfield sites due to lack of information on management practices (herbicide in particular), which would have a strong impact on duckweed cover. In total, these restrictions reduced the number of sites included in the analysis from 89 to 68 within the season data set and to 66 within the summer dataset, and resulted in the CART analysis using duckweed cover as the first splitting variable. Other variables included as potential predictors in the CART analysis included pond origin, pond age, normalized fetch, pond surface area, pond mean depth, pond max depth, canopy density, watershed impervious cover (500 m buffer), and residential land use (500 m buffer).

We developed trees using both summer and season mean values. Considering the importance of summer period for pond phosphorus water quality, especially the likelihood of anoxic sediment phosphorus release, we selected a pond classification tree based on summer mean value. With the first split based on a threshold of duckweed cover (duckweed cover fraction < 0.19), later splits were based on pond origin and surface area within the "lower" duckweed cover sites, and based on mean depth within the "higher" duckweed cover sites. This grouping helped address a potential limitation of the early version of the Pond Assessment Tool, which included models primarily based on duckweed cover. The other variables tested in the model were not important to classification in this dataset.

3 Pond Classification System

The regression tree resulting from the CART analysis on the subset of 66 sites with summer means of TP is shown in Figure 2 and forms the basis for a pond classification system that results in **five pond categories** based primarily on duckweed cover, with pond origin and surface area used to provide some differentiation among ponds with low duckweed and with mean depth to divide among ponds with high duckweed. These five pond categories include, from lowest mean TP to highest mean TP:

Group 1: Low duckweed cover, constructed ponds (n=11)

Group 2: Low duckweed cover, historic wetlands, large surface area (n=19)

Group 3: Low duckweed cover, historic wetlands, small surface area (n=7)

Group 4: High duckweed cover, deeper depth (n=11)

Group 5: High duckweed cover, shallow depth (n=18)

The differences in water quality (duckweed cover, water column TP, water column DO) and sediment variables (anoxic sediment P flux, SOD, sediment organic matter content, sediment bioavailable P) among the five pond groups are illustrated in Figures 2 to 5.

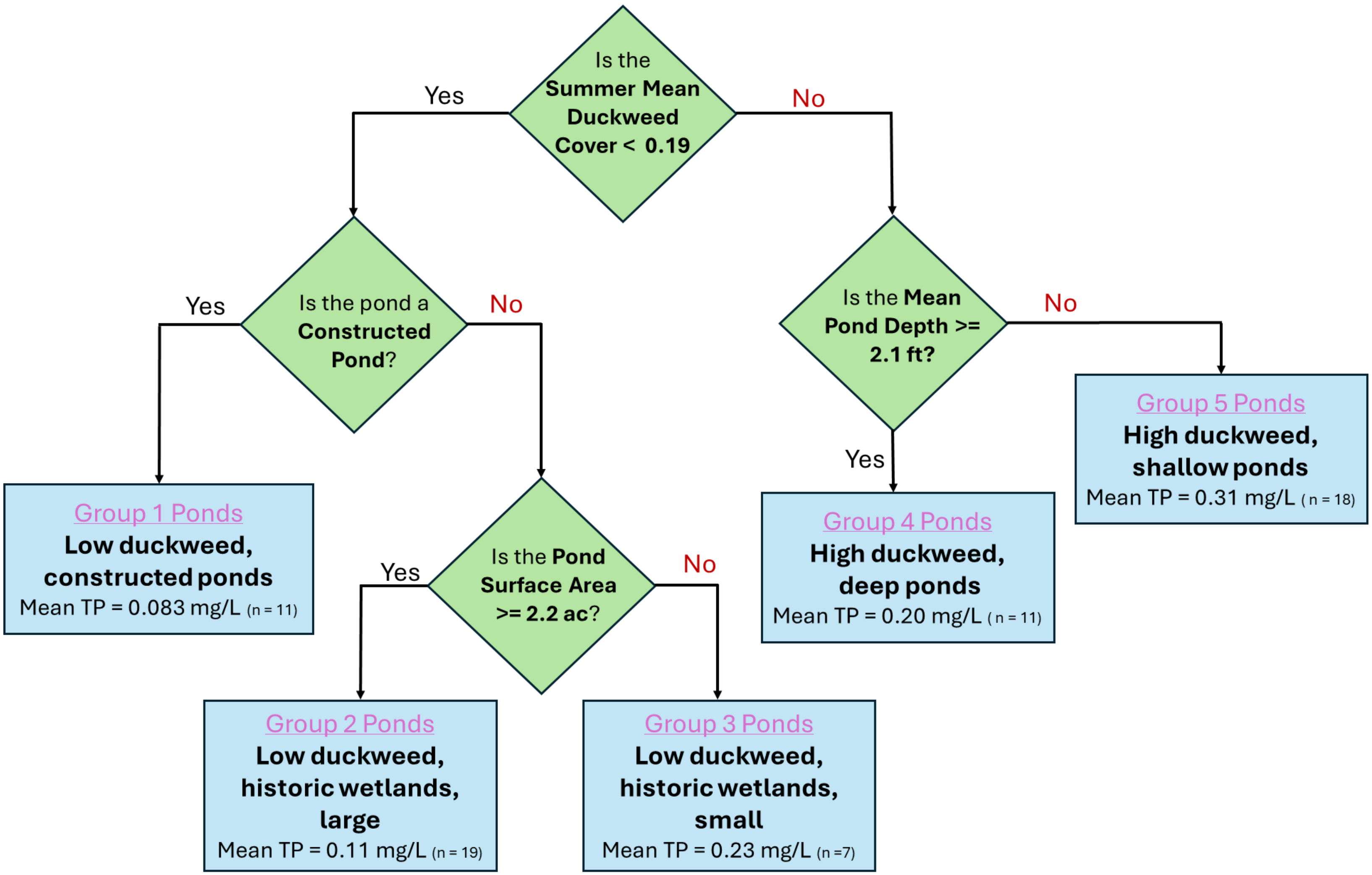


Figure A-2. Regression tree (adapted from CART analysis) showing classification of pond summer (June – September) mean surface total phosphorus concentration (TP, mg/L) based on season mean duckweed cover (fraction of pond area), pond origin (constructed pond vs. historic wetland), mean depth, and pond surface area. The five groups of ponds display the number of ponds (n) within the group and the mean TP concentration of the group.

3.1 Low duckweed cover ponds (Groups 1, 2, 3) vs. High duckweed cover ponds (Groups 4, 5)

The first division in the classification tree is based on the mean duckweed cover (DW) during the summer period (June – September), dividing the ponds into “low” and “high” duckweed cover sites. While the low duckweed cover ponds (mean DW = 0.027; Groups 1, 2 ,3 combined) contained mean summer surface TP (Mean TP) of 0.12 mg/L, the high duckweed cover ponds (mean = 0.74; Group 4 and 5 combined) contained 0.27 mg/L mean TP (Figure A-3). When compared to the low duckweed sites, the high duckweed ponds tended to have a much lower average dissolved oxygen (Mean DO) concentration in the water column and the sediments showed a higher potential for P release under anoxic conditions (Sediment P Flux), with relatively higher bioavailable (mobile) P mass (Sediment Mobile P). Sediment oxygen demand (SOD) and organic matter content (Sediment OM) were also relatively higher in the high duckweed ponds. Sediment TP and sediment (labile) organic P mass were, however, not much different.

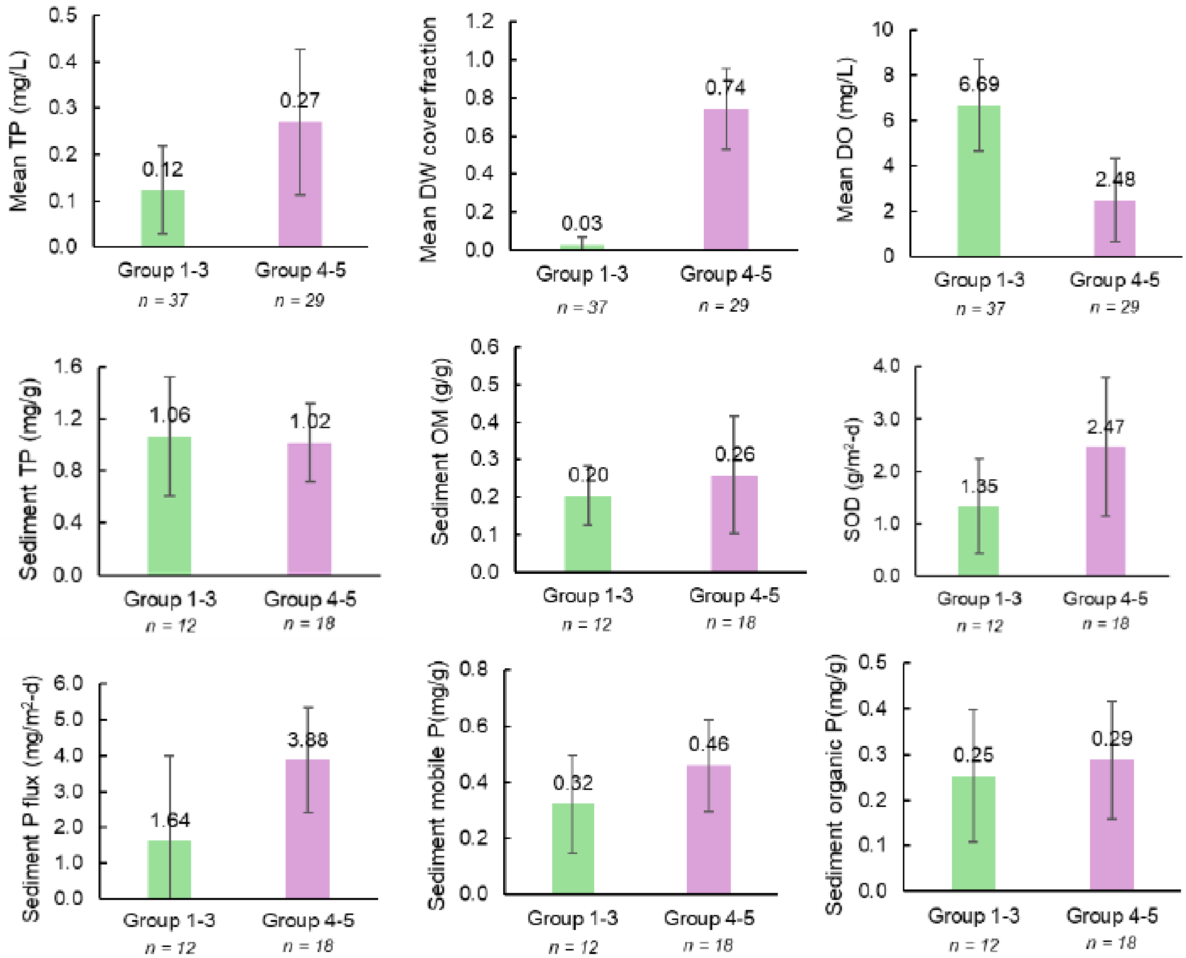


Figure A-3. Plots showing the season mean for TP, duckweed cover and water column DO, and sediment variables for pond groups with low duckweed (Group 1-3) and high duckweed (Group 4-5) as shown in the regression tree (Figure 1). Error bars represent the standard deviation of the mean. The number of ponds (n) within each sub-group is included.

3.2 Low duckweed cover ponds, that are constructed ponds (Group 1) vs. historic wetlands (Groups 2, 3)

Within the low duckweed cover ponds, the classification tree divides ponds based on their origin, i.e., whether they are constructed ponds or historic wetlands. It must be noted that the CART analysis combined sites with unknown origin (because determinations could not be done by satellite imagery also) with the historic wetlands; the pond age of all unknown origin sites were <1991. Compared to historic wetlands, constructed ponds tended to have lower water column TP and their sediments contained less TP, organic matter, SOD and substantially less labile organic P mass (Figure A-4). Anoxic sediment P release and bioavailable P mass in sediment did not show a difference based on the pond origin, which was an unexpected result. The mean water column DO concentration was similar and well above the anoxic threshold in the pond sub-groups analyzed, which can be largely attributed to the very low duckweed cover in all ponds (mean DW < 0.04; plot not shown).

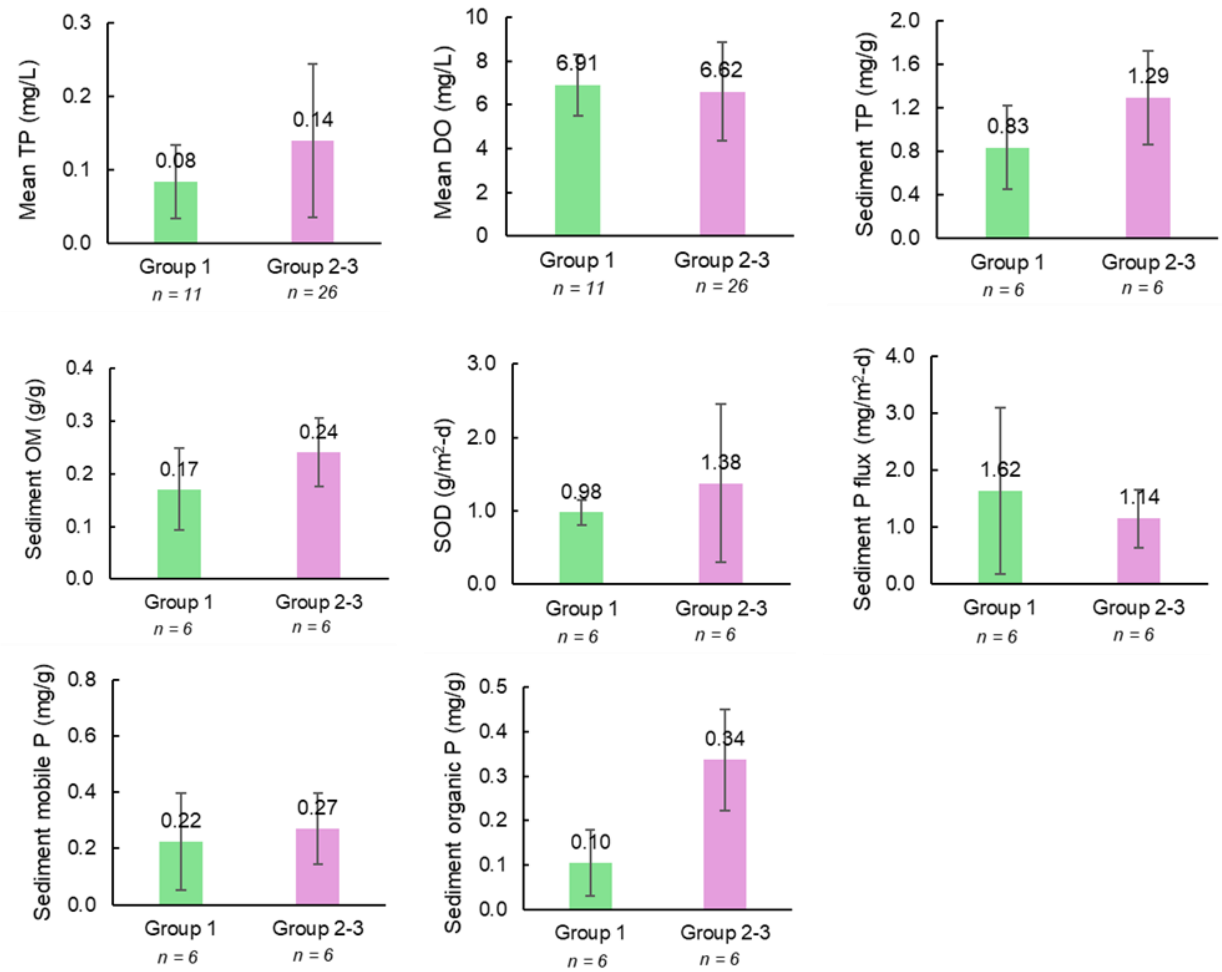


Figure A-4. Plots showing the summer mean for TP, water column DO, and sediment variables for pond groups characterized as constructed ponds (Group 1) and historic wetlands (Group 2-3) as shown in the regression tree (Figure 1). Error bars represent the standard deviation of the mean. The number of ponds (n) within each sub-group is included.

3.3 Low duckweed cover, historic wetlands/unknown origin ponds, that have large surface area (Group 2) vs. small surface area (Group 3)

Within the low duckweed cover ponds that are historic wetlands, CART further divides the ponds based on their surface area, with a cut-off value of 2.2 ac. The smaller ponds (≤ 2.2 ac) tended to have a higher mean summer TP than larger ponds, but water column DO concentrations were not much different, owing to the low duckweed cover in summer (Figure A-5). Comparison of sediment characteristics could not be done for the two groups as data from only one pond was available for large ponds (Group 3).

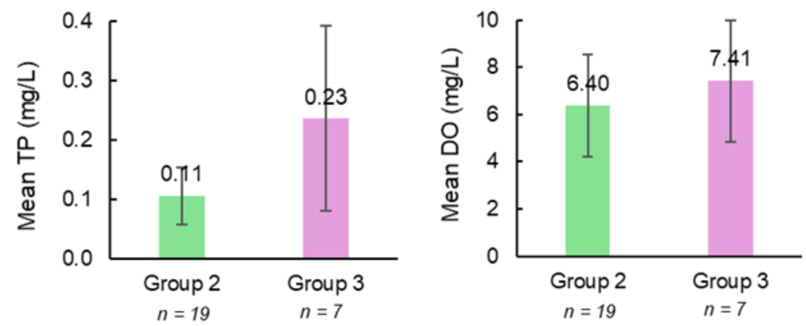


Figure A-5. Plots showing the summer mean for TP and water column DO for pond groups characterized as historic wetlands with small (Group 2) and large surface area (Group 2) as shown in the regression tree (Figure 1). Error bars represent the standard deviation of the mean. The number of ponds (n) within each sub-group is included. Sediment variables were not plotted due to the low site count

3.4 High duckweed cover ponds that are deep ponds (Group 4) vs. shallow ponds (Group 5)

Within ponds with a high summertime duckweed cover, the classification tree further divides the ponds based on their mean depth into deep (≥ 2.1 ft; Group 4) and shallow ponds (Group 5), where a higher mean summer TP is observed in shallow ponds (Figure A-6). Under high duckweed coverage, the deep ponds tend to have a slightly lower water column DO and their sediments may have a slightly higher potential to release P under anoxia. Other sediment variables did not appear to show differences based on mean pond depth.

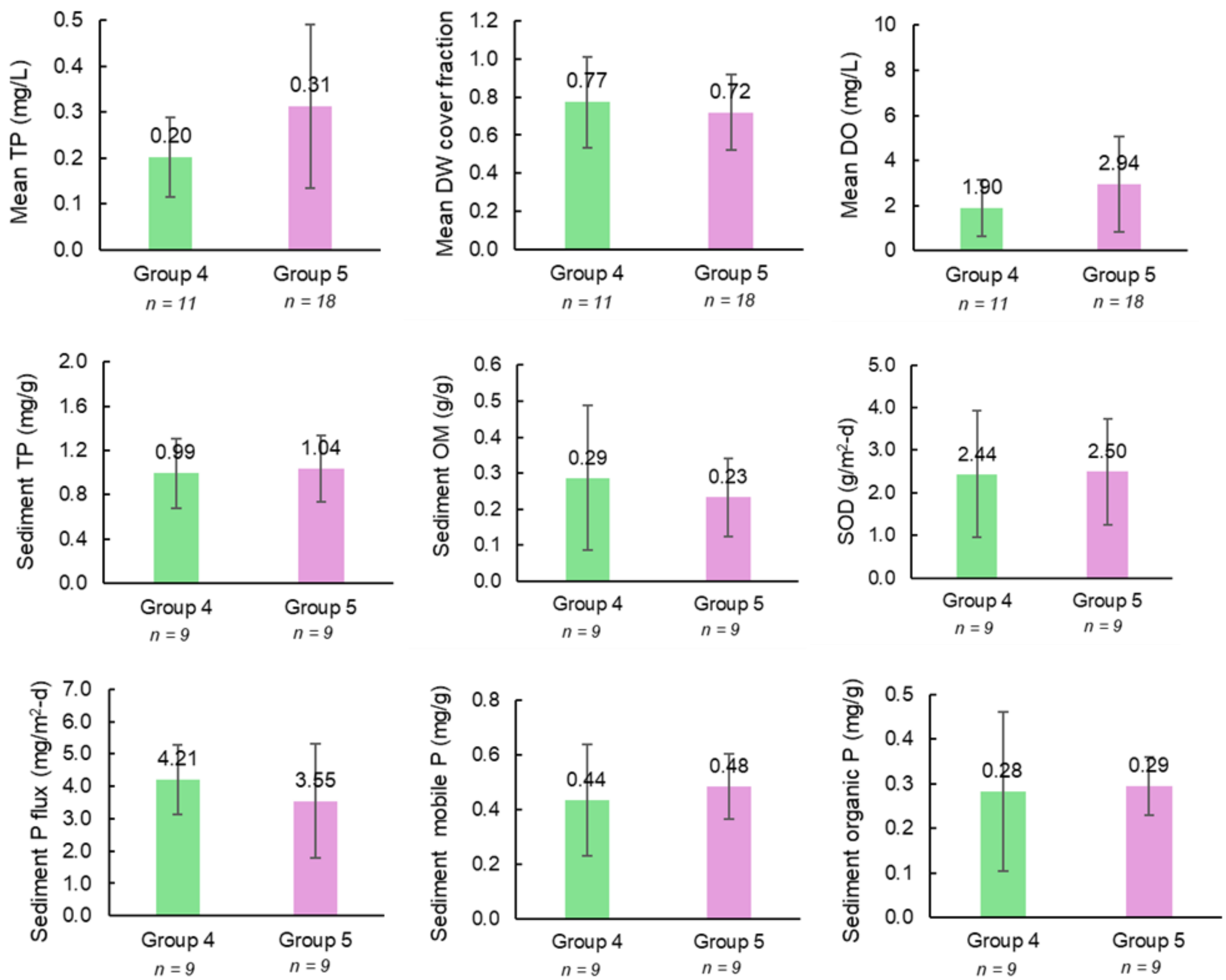


Figure A-6. Plots showing the summer mean for TP, duckweed cover, water column DO, and sediment variables for pond groups with high duckweed cover that are deep (Group 4) and shallow (Group 5) as shown in the regression tree (Figure 2). Error bars represent the standard deviation of the mean. The number of ponds (n) within each sub-group is included.

To summarize, the regression tree aligns with our broader expectations of these stormwater ponds, in that **ponds with high duckweed cover tended to have the highest TP due to the strong feedback of dense duckweed cover on anoxia (DO < 2 mg/L) and likely resulting in internal P loading from sediments**. In a field experiment involving manual removal of duckweed from a small pond (area = 0.1 ac), the pond responded rapidly to removal of duckweed, with rising oxygen levels and decreasing surface water TP (Ronkainen 2021); this suggests that a heavy duckweed cover is an indicator of anoxic conditions in the pond. Within the large group of ponds having some or no duckweed cover, constructed ponds tended to have lower water column TP than historic wetlands. Within those historic wetlands with low duckweed cover, the smaller ponds tended to have higher concentrations of TP than the larger ponds. Under high duckweed cover, shallow ponds tended to have higher concentrations of TP than deeper ponds. Since pond water column DO is strongly controlled by susceptibility to mixing or aeration by wind (i.e., larger or more open sites) and also by the presence or absence of rooted macrophytes and algae, we acknowledge that there are likely some other factors that should be considered in future classifications.

4 Management Recommendations

In this section we provide a list of potential maintenance options for ponds that specifically address the pond parameters in the pond classification system (i.e., duckweed, pond surface area, pond depth), using information largely adapted from Janke et al. (2023). We emphasize that the purpose of this section is to provide potential maintenance or treatment options vis-à-vis definitive maintenance actions to take. Feasibility or cost-effectiveness of pond remediation or management practices are discussed in greater detail in this project's main report.

Removal of Free-Floating Plants: Removal of nutrient-rich duckweed biomass is expected to enable nutrient removal and increased water column dissolved oxygen, especially given the strong negative association of duckweed versus oxygen and phosphorus levels of ponds. Treatment options to remove duckweed could therefore help reduce phosphorus export risk of ponds. Alum treatment to reduce duckweed via reduction in phosphorus in an urban pond has been done successfully (Osgood 2012). While there is some evidence of higher oxygen and lower TP following duckweed removal in a small urban pond (Ronkainen 2021), impacts of mechanical removal are being investigated in a current project by Finlay et al. (2026)[15].

Dredging: Dredging is a routine maintenance practice to mechanically remove accumulated sediments to restore pond capacity. Change in pond bathymetry or geometry (e.g., volume, depth, surface area) due to dredging can impact the hydrodynamics of ponds through effects on mixing, stratification, and hydrologic retention, with resulting impacts to oxygen and phosphorus dynamics. Potential impacts of dredging on sediment phosphorus release through removal of sediment P and organic matter from the ponds are being investigated in a current project by Gulliver et al. (2024-2027)[16].

Chemical Sediment Treatment: Treatment of sediments with chemicals or other amendments (such as alum, iron, spent lime) that can bind phosphorus is a method to reduce internal P release and water column P concentration and associated algal and floating plant growth. Although effectiveness of alum and iron treatment of lake sediments has been well-established, potential impacts and longevity of these chemical treatments in stormwater ponds are being investigated in a current project by Gulliver et al. (2023-2026)[17].

Aeration: Aeration of a pond is expected to de-stratify the pond water column and enhance oxygen delivery to the pond bottom, which can potentially minimize anoxic sediment P release. While aeration was found to be a cost-effective measure for improving pond phosphorus retention in a modeling

[15] https://wrc.umn.edu/projects/managing-veg
[16] https://researchprojects.dot.state.mn.us/projectpages/pages/lrrbProjectDetails.jsf?id=29689&type=CONTRACT
[17] https://wrc.umn.edu/projects/remediation-effectiveness

exercise (Taguchi et al. 2022), field performance and cost-effectiveness of various aerator/mixing systems will be investigated in a demonstration project by Gulliver et al. (2025-2028)[18].

Watershed Management: Watershed management includes methods to reduce inflow concentrations and volumes, typically by installation of structural stormwater control measures (SCMs) and implementation of non-structural practices like street sweeping.

5 Integration of the Pond Classification System into the Pond Assessment Tool

We propose adding the pond classification system to the Pond Assessment Tool as either an alternative to the Tool's current Screening tool (Tool 1-A), or as a reconnaissance or pre-screening tool that would be used to identify and group ponds with potential for poor P retention. A workflow might look like that shown in Figure A-7. The classification system could be used first to quickly assess a large number of sites and group them based on typical (average) surface TP concentrations associated with the groups. Ponds could then be scored and ranked using the Screening Tool (1-A) to prioritize further monitoring or maintenance. Either Tool could be used alone for a small number of sites, but the primary benefit of the classification system would be to facilitate assessment of a large number of pond sites.

The classification system has not been integrated with the Pond Assessment Tool as of the writing of this report, but pending feedback from the Advisory Panel, we will include some version of it in a future release of the Tool.

[18] https://wrc.umn.edu/projects/aeration-remediation

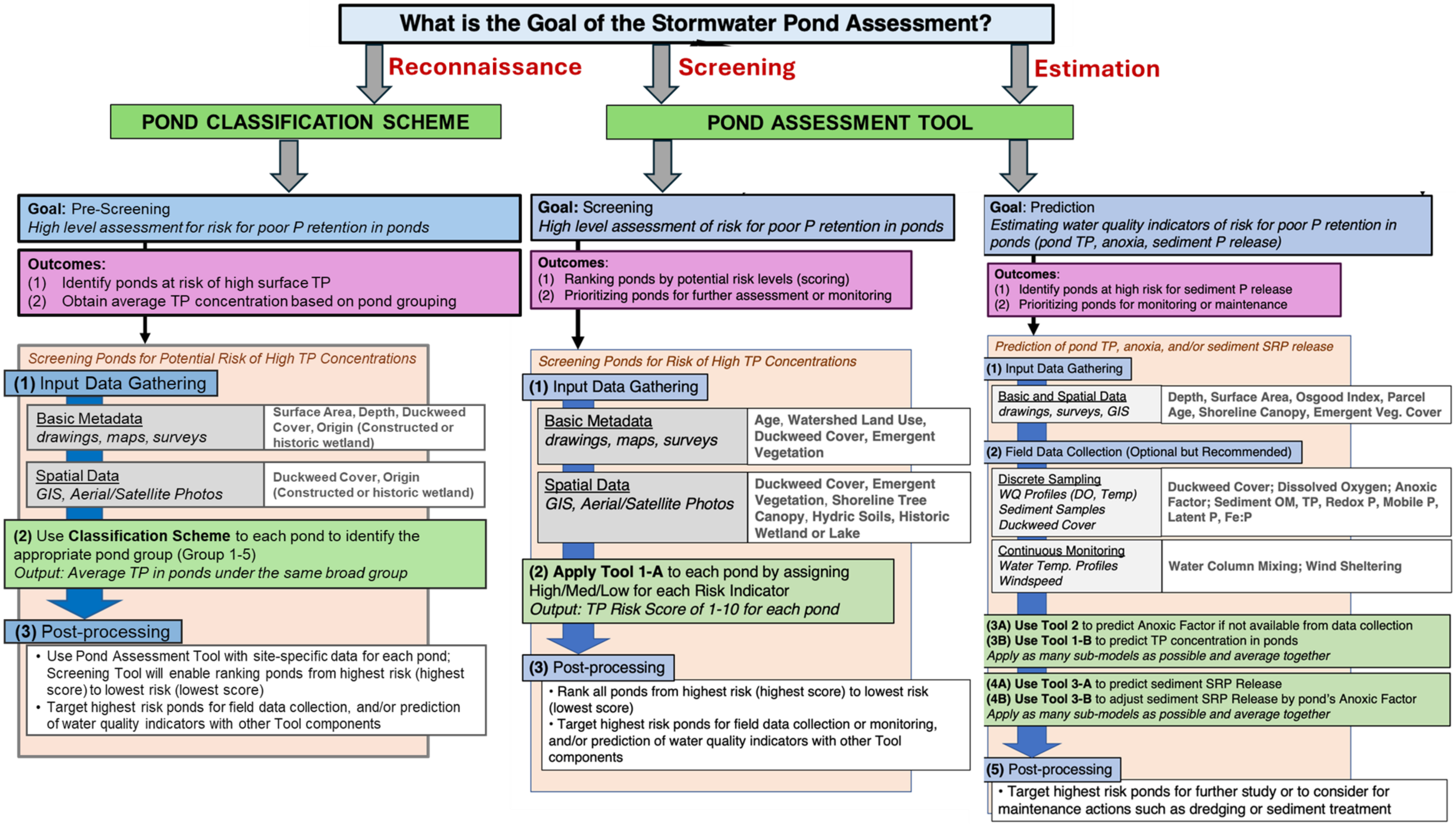


Figure A-7. Proposed integration of the pond classification system with the Pond Assessment Tool (workflow adapted from Natarajan et al. 2025).

Appendix B
Field Sampling Photos and Data from 2023 and 2024, and Laboratory Phosphorus Release Study Data

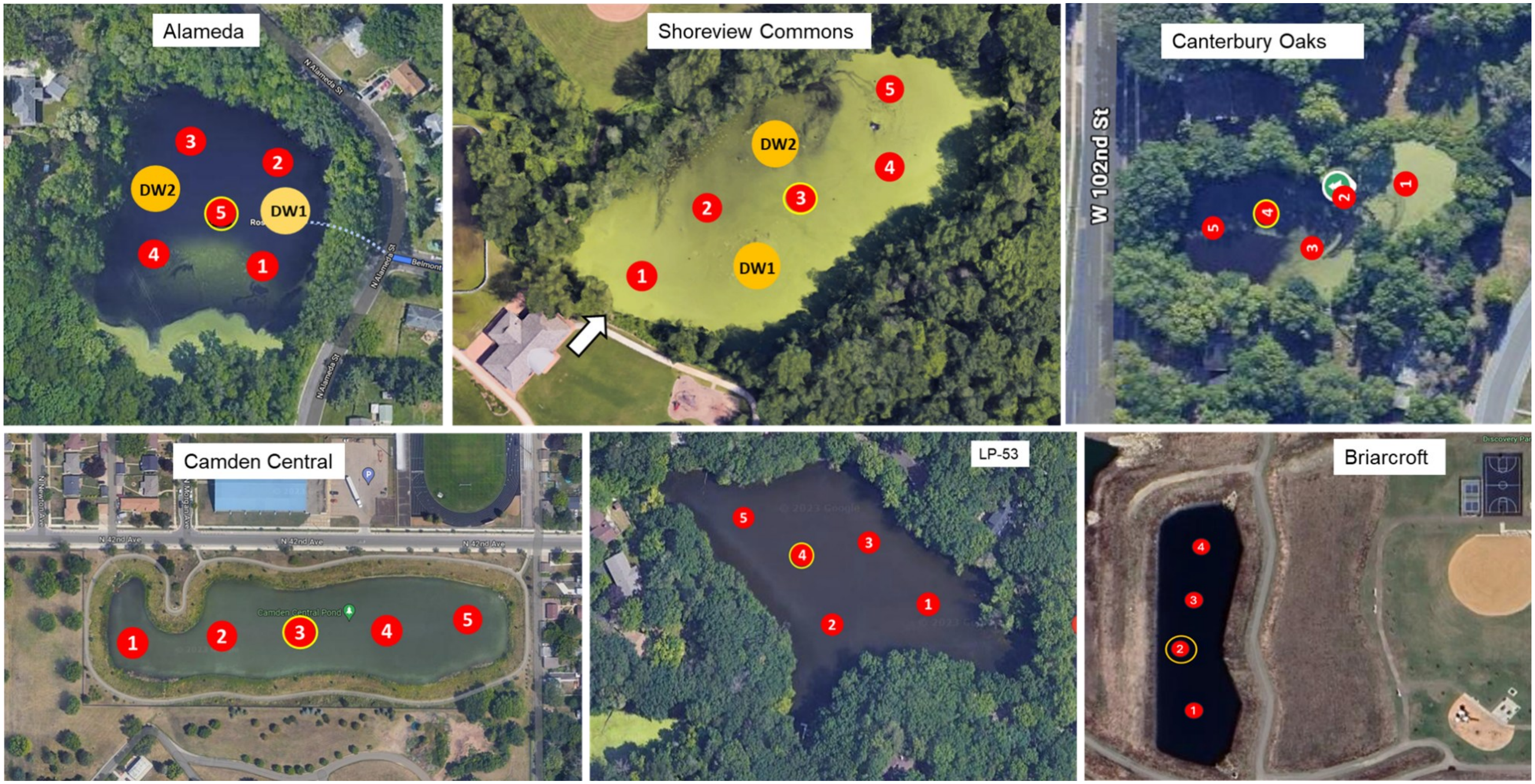


Figure B-1 Maps showing the water sampling locations in Alameda, Shoreview Commons, Camden Central, Canterbury Oaks, LP-53, and Briarcroft. Locations 1 to 5 are in the open water, where the deepest location (highlighted with yellow circle) was used for measuring the vertical profiles of DO, temperature and specific conductivity and for the hypolimnion water sample collection. Surface water samples from 1 to 5 were composited into one sample in the field. In Alameda and Shoreview Commons, Locations DW1 and DW2 were for duckweed sampling in addition to Locations 1-5.

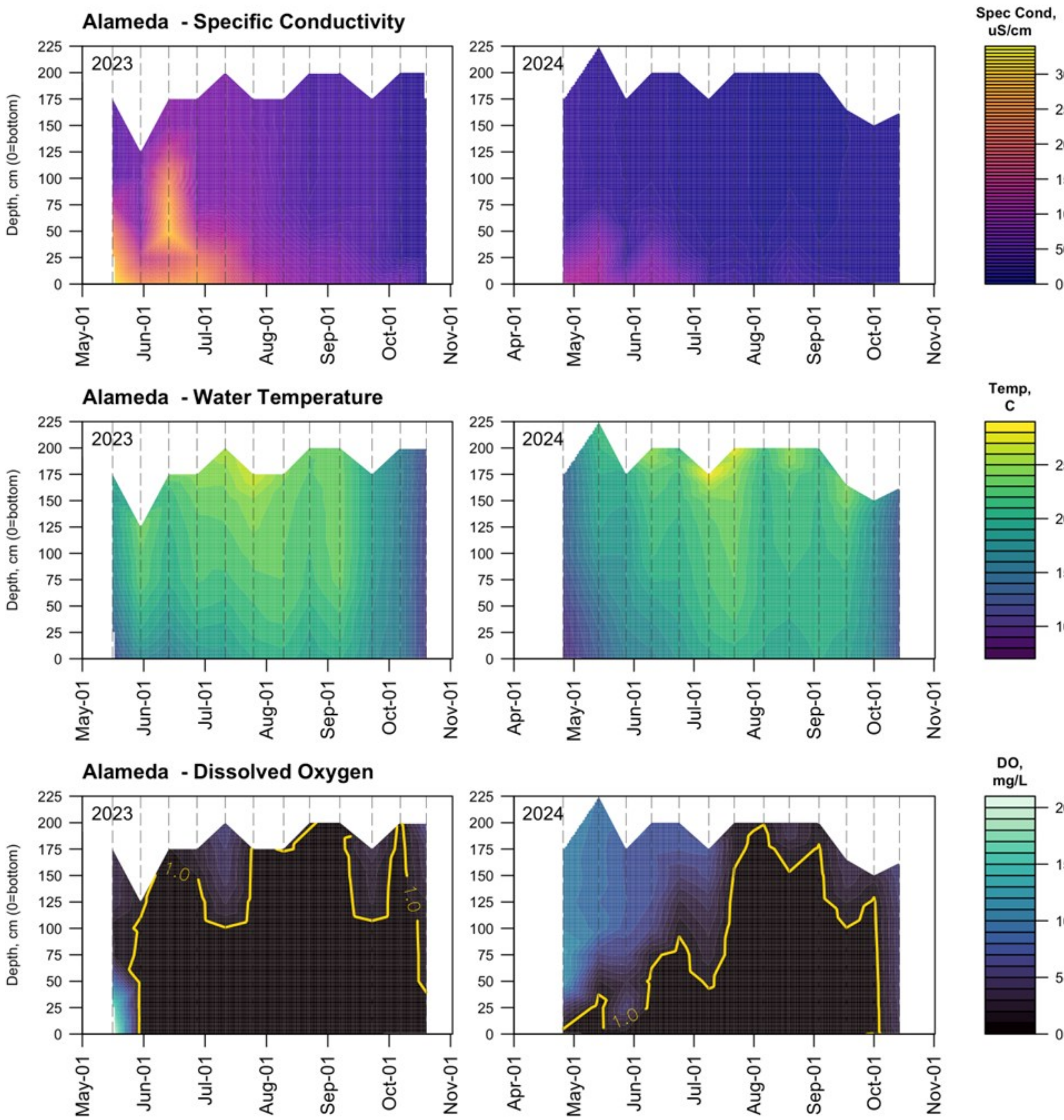


Figure B-2. Specific conductivity (top row), temperature (middle row), and dissolved oxygen (bottom row) contour plots for Alameda: Color indicates values per the scale at right, with water depth relative to the pond bottom on the y-axis and time along the x-axis (left column is 2023 and right column is 2024). Vertical dashed lines are dates of site visits when profiles were collected; linear interpolation used to fill in the gaps between profile dates. On the DO plot, a contour for 1.0 mg/L indicates levels below which the pond/historic wetland is anoxic.

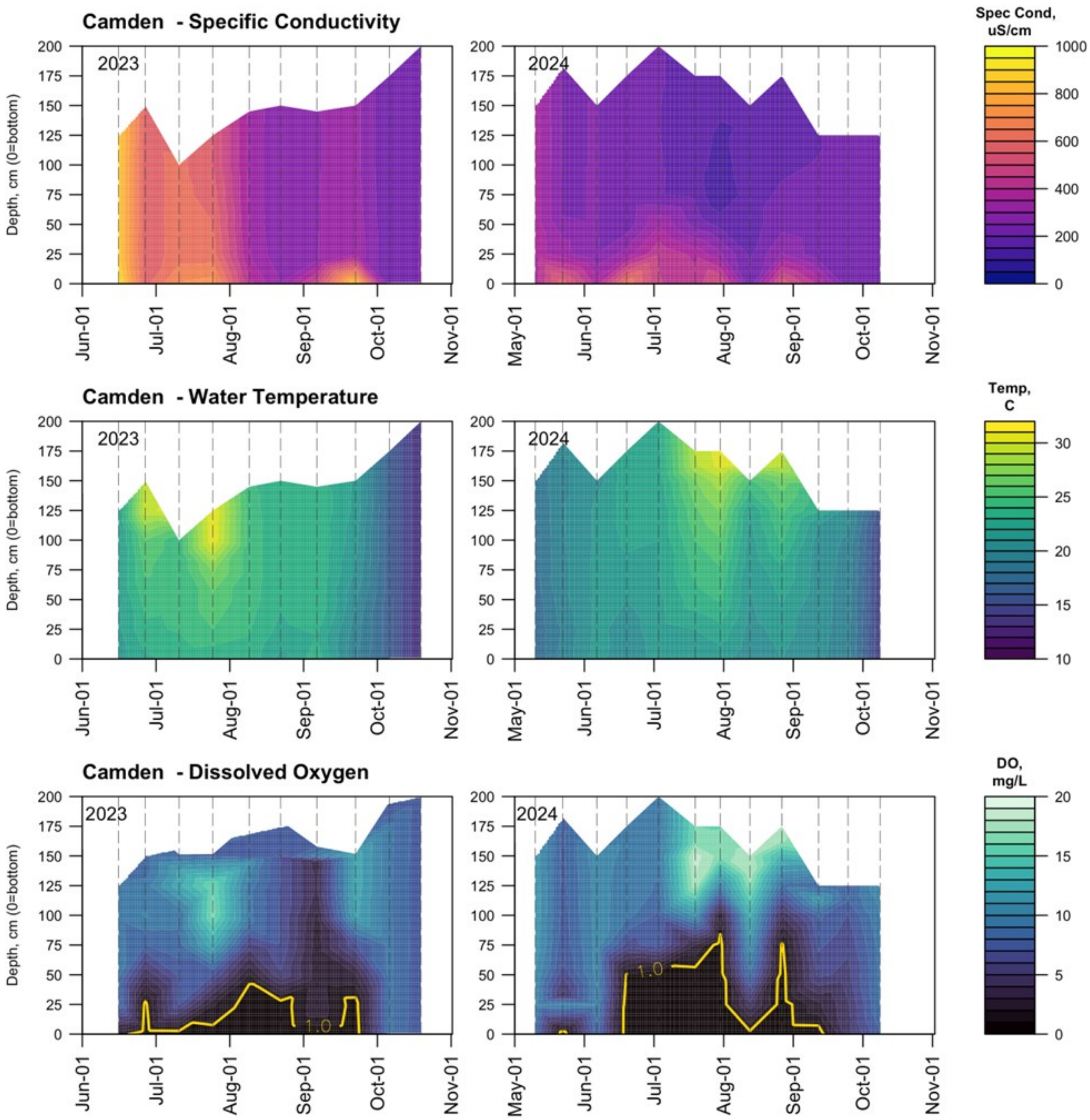


Figure B-3. Specific conductivity (top row), temperature (middle row), and dissolved oxygen (bottom row) contour plots for Camden Central: Color indicates values per the scale at right, with water depth relative to the pond bottom on the y-axis and time along the x-axis (left column is 2023 and right column is 2024). Vertical dashed lines are dates of site visits when profiles were collected; linear interpolation used to fill in the gaps between profile dates. On the dissolved oxygen plot, a contour for 1.0 mg/L indicates levels below which the pond is considered anoxic.

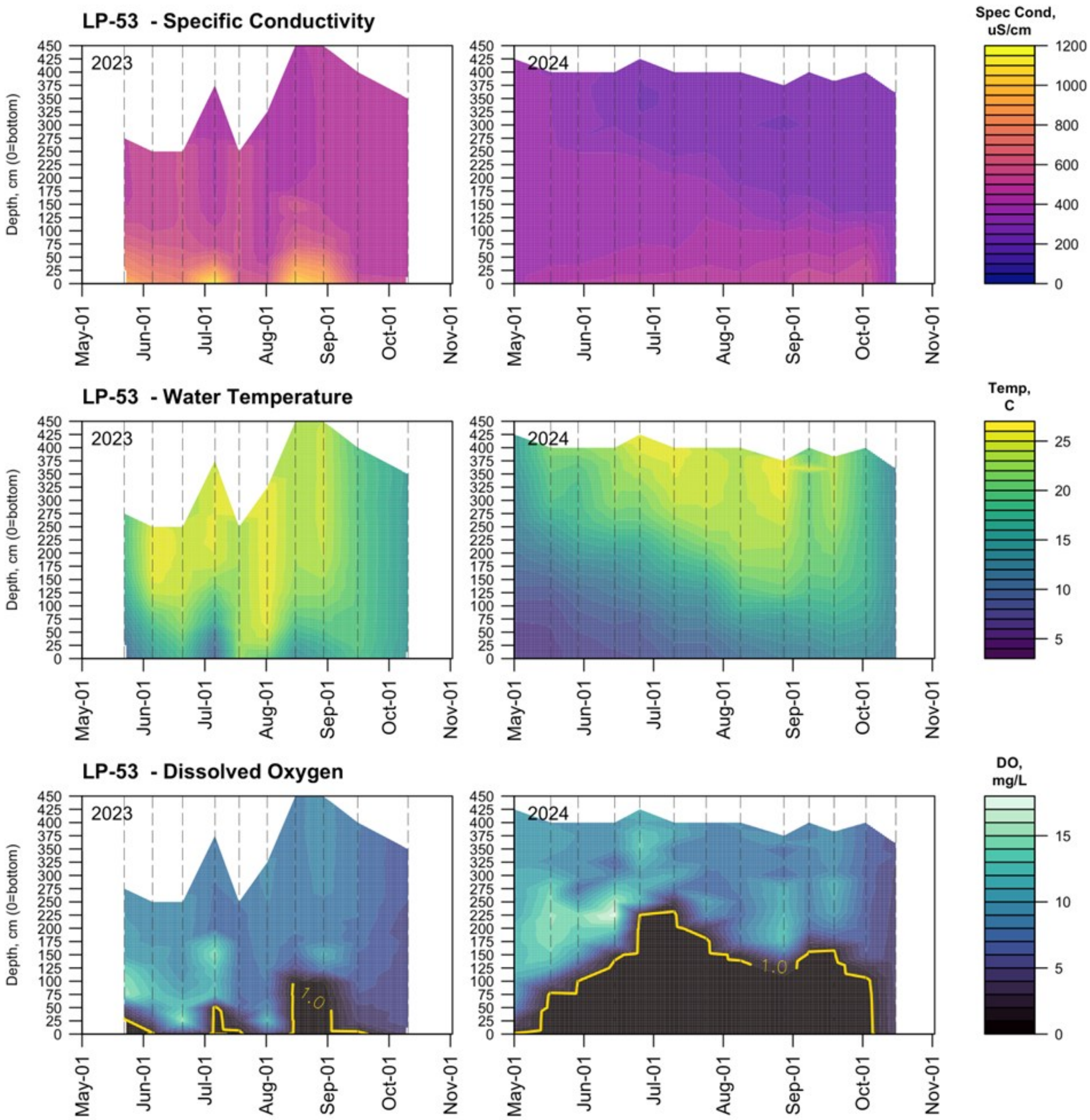


Figure B-4. Specific conductivity (top row), temperature (middle row), and dissolved oxygen (bottom row) contour plots for LP-53: Color indicates values per the scale at right, with water depth relative to the pond bottom on the y-axis and time along the x-axis (left column is 2023 and right column is 2024). Vertical dashed lines are dates of site visits when profiles were collected; linear interpolation used to fill in the gaps between profile dates. On the dissolved oxygen plot, a contour for 1.0 mg/L indicates levels below which the pond/historic wetland is considered anoxic. Note that the water level changes in 2023 are likely due to inconsistent collection of profiles with respect to location in the pond, reflecting different depths in profile location rather than large fluctuations in water level over time.

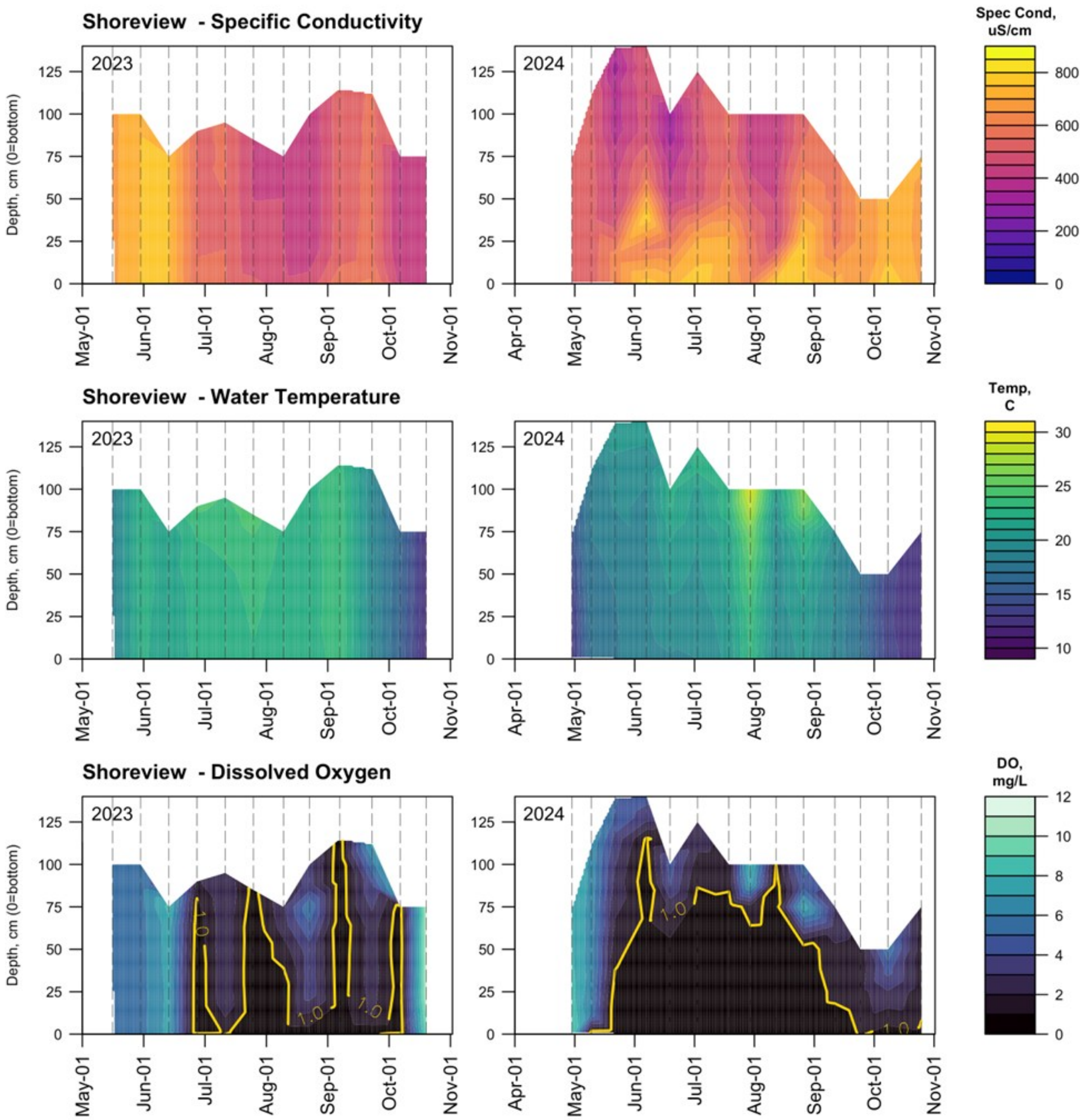


Figure B-5. Specific conductivity (top row), temperature (middle row), and dissolved oxygen (bottom row) contour plots for Shoreview Commons: Color indicates values per the scale at right, with water depth relative to the pond bottom on the y-axis and time along the x-axis (left column is 2023 and right column is 2024). Vertical dashed lines are dates of site visits when profiles were collected; linear interpolation used to fill in the gaps between profile dates. On the dissolved oxygen plot, a contour for 1.0 mg/L indicates levels below which the pond/historic wetland is considered anoxic.

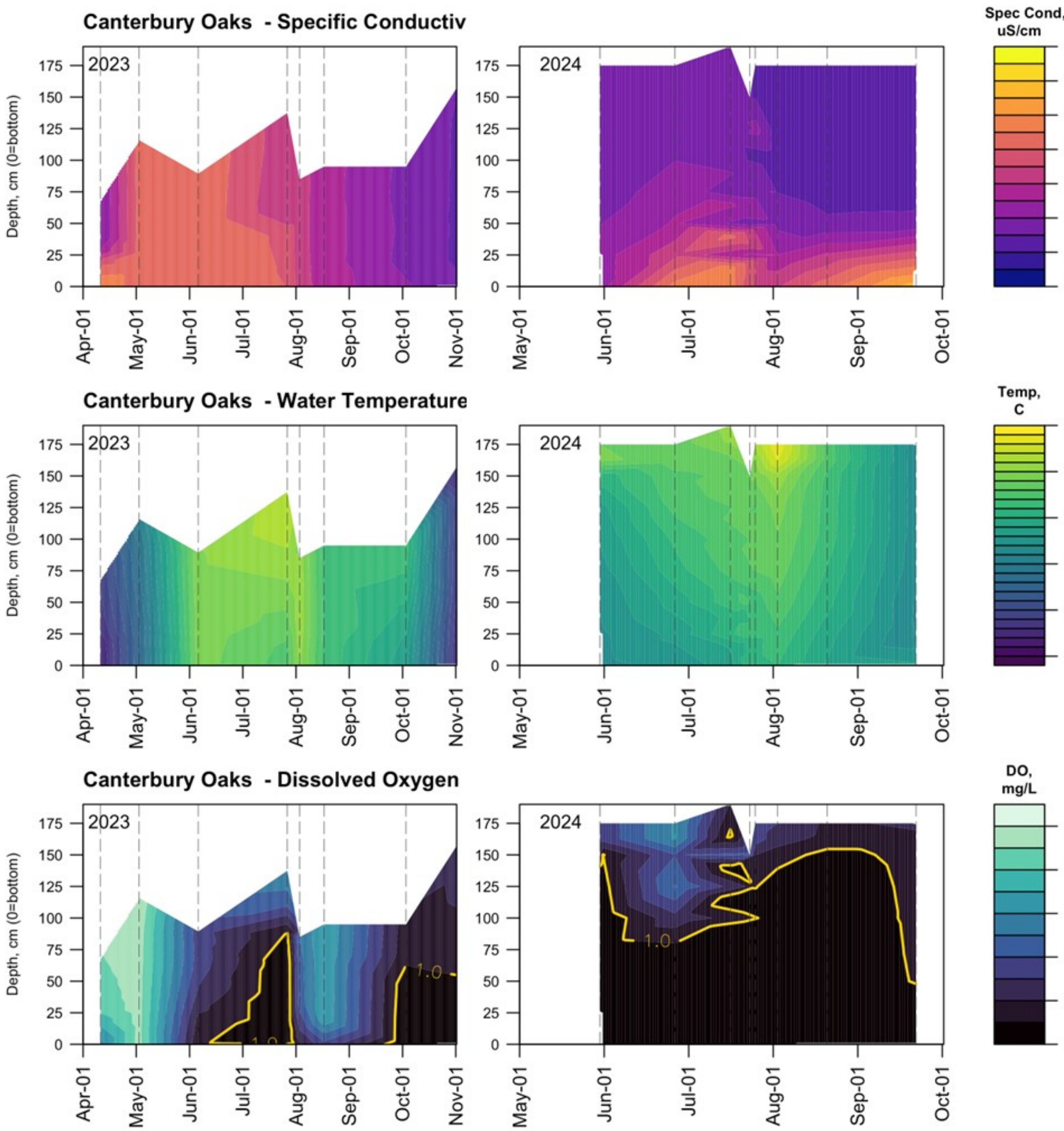


Figure B-6. Specific conductivity (top row), temperature (middle row), and dissolved oxygen (bottom row) contour plots for Canterbury Oaks pond: Color indicates values per the scale at right, with water depth relative to the pond bottom on the y-axis and time along the x-axis (left column is 2023 and right column is 2024). Vertical dashed lines are dates of site visits when profiles were collected; linear interpolation used to fill in the gaps between profile dates. On the dissolved oxygen plot, a contour for 1.0 mg/L indicates levels below which the pond/historic wetland is considered anoxic.

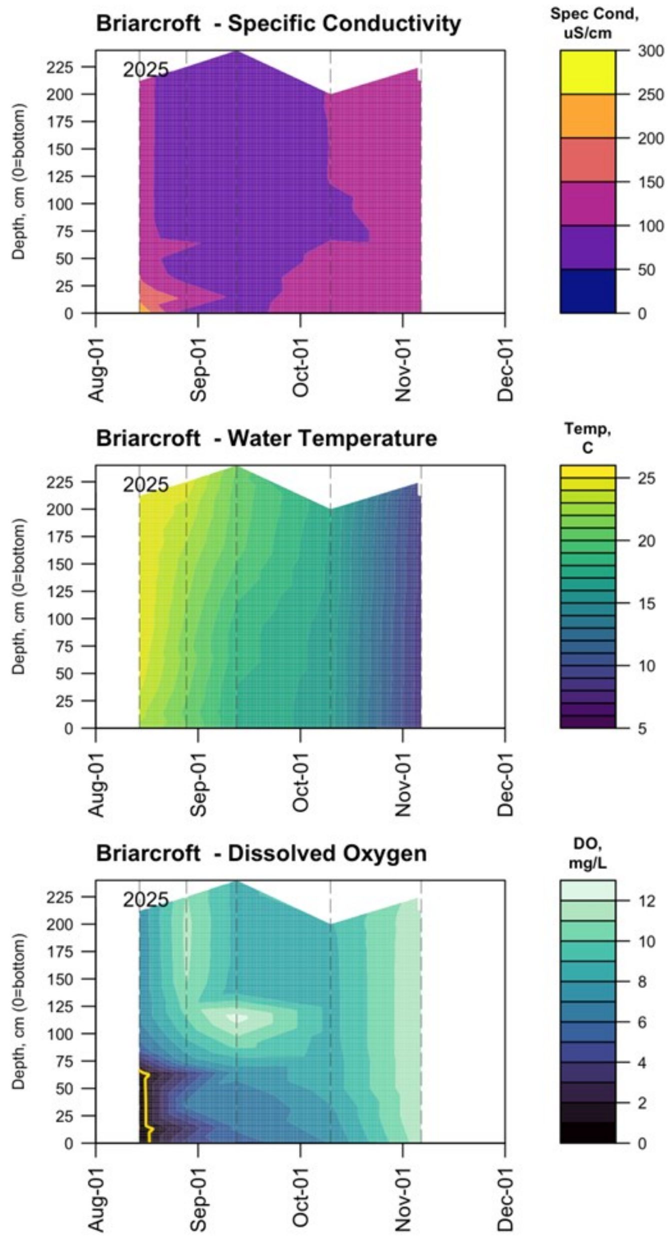


Figure B-7. Specific conductivity (top row), temperature (middle row), and dissolved oxygen (bottom row) contour plots for Briarcroft: Color indicates values per the scale at right, with water depth relative to the pond bottom on the y-axis and time along the x-axis (note the year is 2025). Vertical dashed lines are dates of site visits when profiles were collected; linear interpolation used to fill in the gaps between profile dates. On the dissolved oxygen plot, a contour for 1.0 mg/L indicates levels below which the pond is considered anoxic.

Continuous Monitoring at Pond Sites

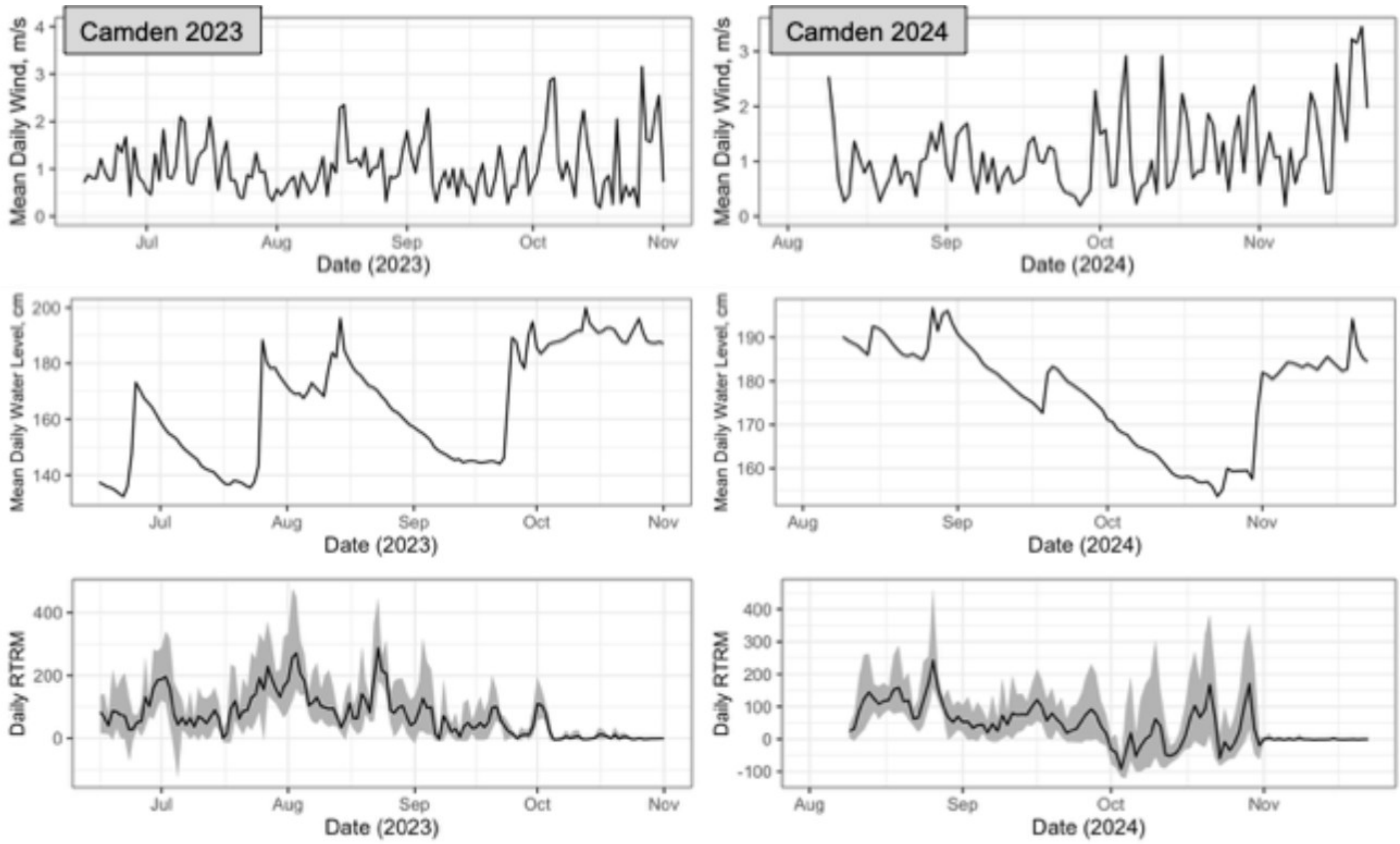


Figure B-8. Mean daily wind speed (m/s) (top row), mean daily water level (cm) (middle row), and daily relative thermal resistance to mixing (RTRM) as mean (black line) minimum and maximum (dark gray envelope), observed at the Camden Central pond in 2023 (left column) and 2024 (right column).

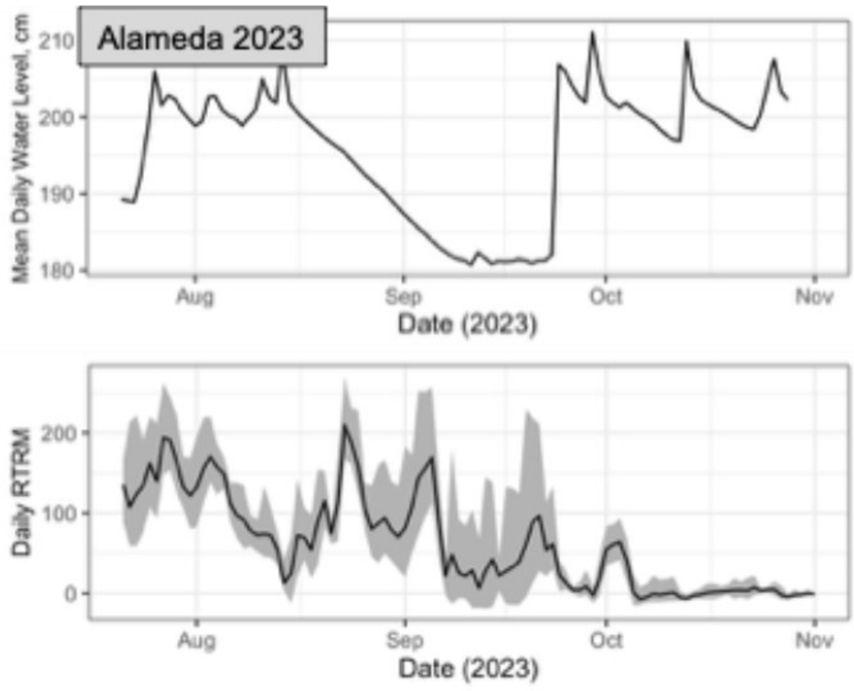


Figure B-9. Mean daily water level (cm) (top), and daily relative thermal resistance to mixing (RTRM) as mean (black line) and minimum and maximum (dark gray envelope), observed at Alameda in 2023. No anemometer was installed at the site in either 2023 or 2024. Failure of several temperature sensors at the site in 2024 resulted in limited applicability for the water level data collected in 2024, which is not shown.

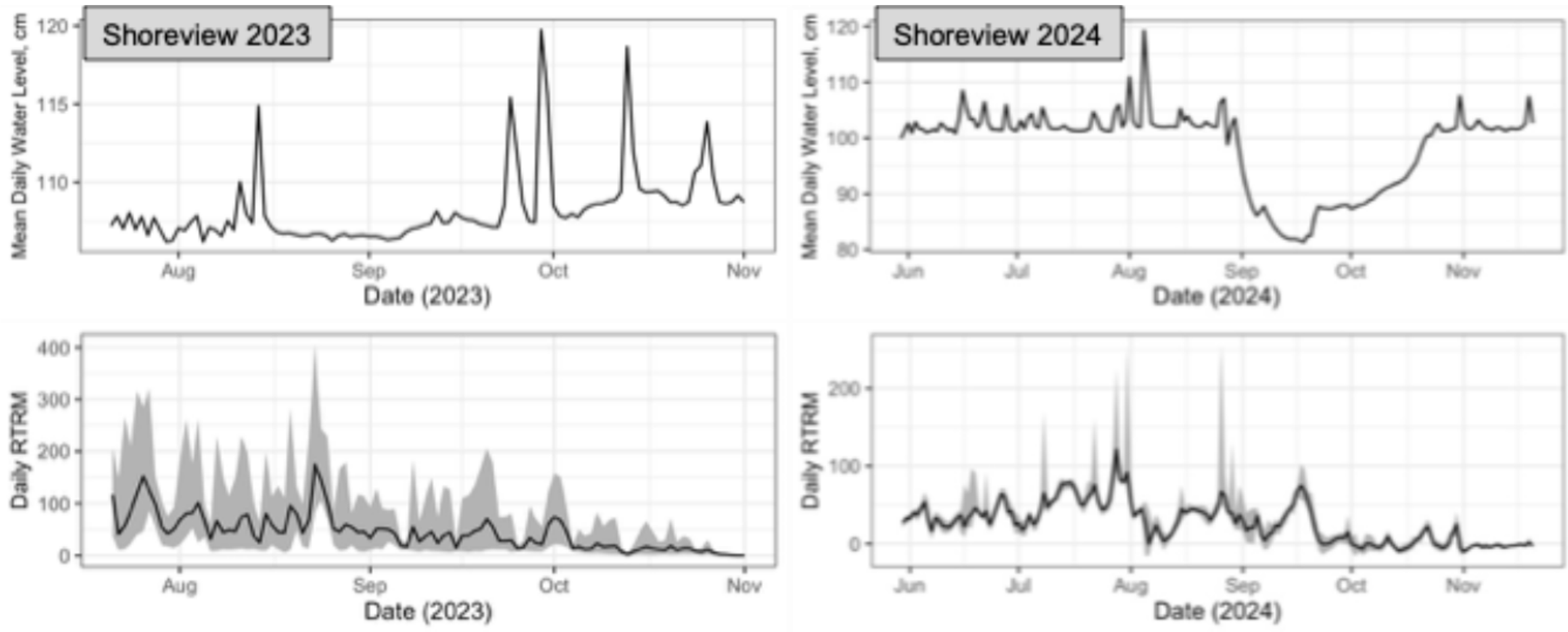


Figure B-10. Mean daily water level (cm) (top row), and daily relative thermal resistance to mixing (RTRM) as mean (black line) minimum and maximum (dark gray envelope) (bottom row), observed at Shoreview Commons in 2023 (left column) and 2024 (right column). Note that no anemometer was installed at the site in either year.

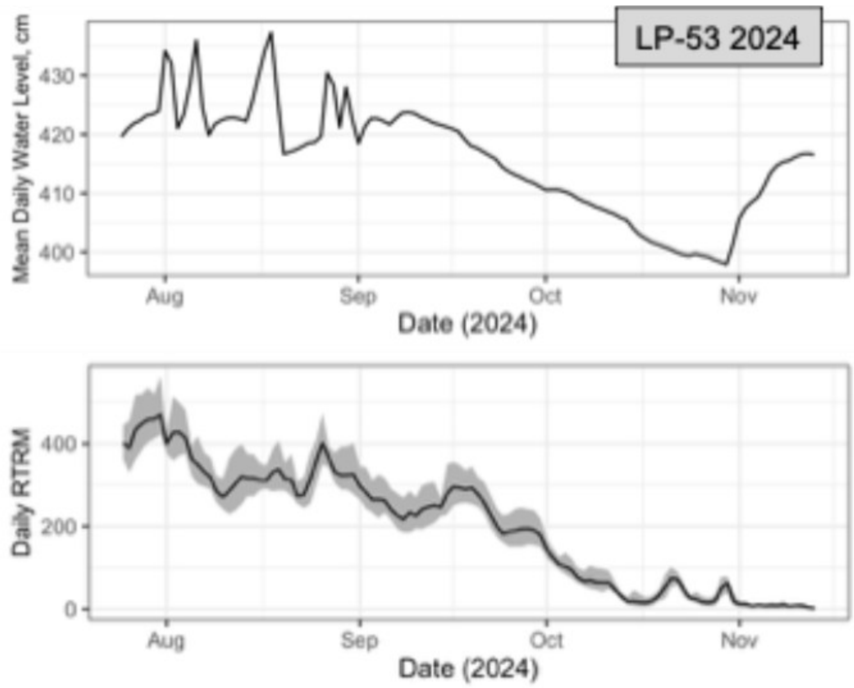


Figure B-11. Mean daily water level (cm) (top row), and daily relative thermal resistance to mixing (RTRM) as mean (black line) minimum and maximum (dark gray envelope) (bottom row), observed at LP-53 in 2024. Note that no data were collected at the site in 2023. Damage to the initial instrumentation in 2024 that included the anemometer resulted in no wind data being collected at the site in 2024.

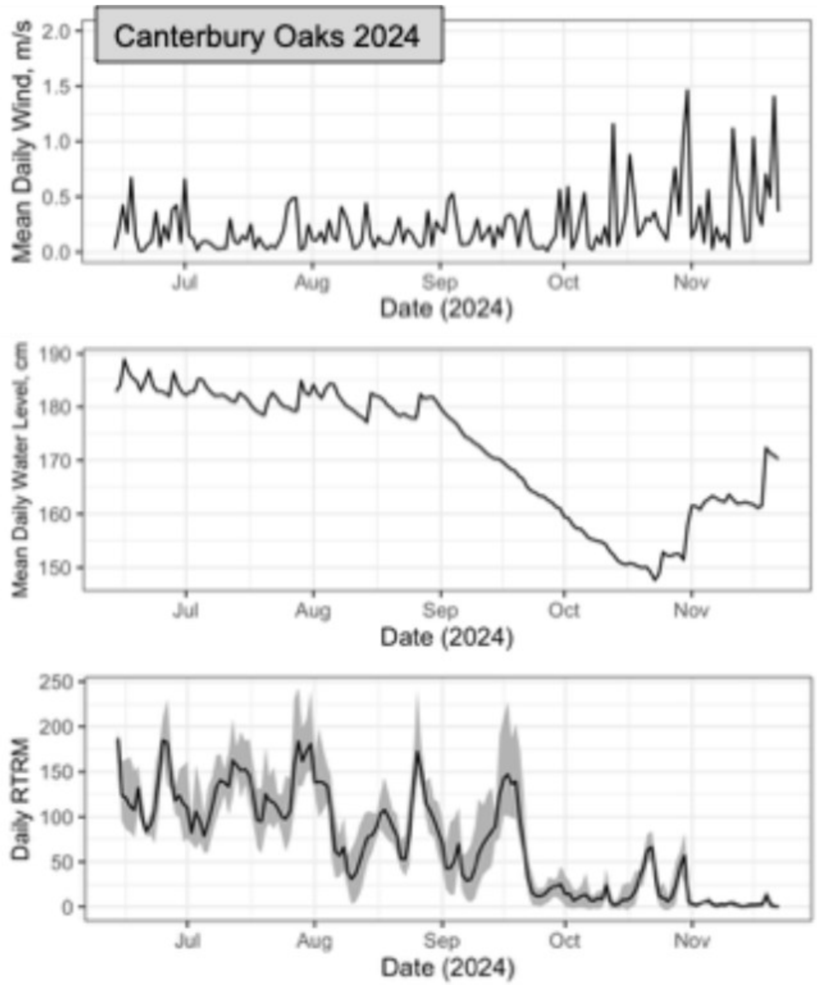


Figure B-12. Mean daily wind speed (m/s) (top row), mean daily water level (cm) (middle row), and daily relative thermal resistance to mixing (RTRM) as mean (black line) and minimum and maximum (dark gray envelope) (bottom row), observed at Canterbury Oaks during the 2024 field season. Note that a full station was not installed at the site in 2023, so those data are not shown here. The site was instrumented and monitored in 2025 (not shown) as part of another project.

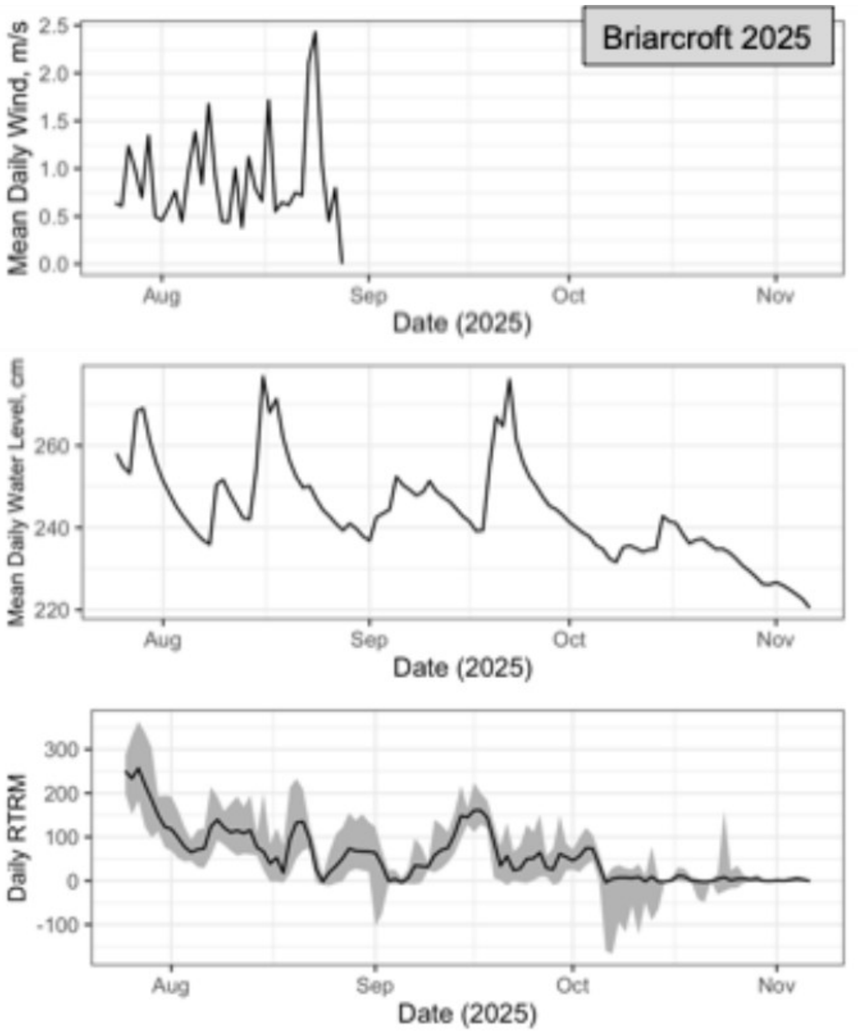
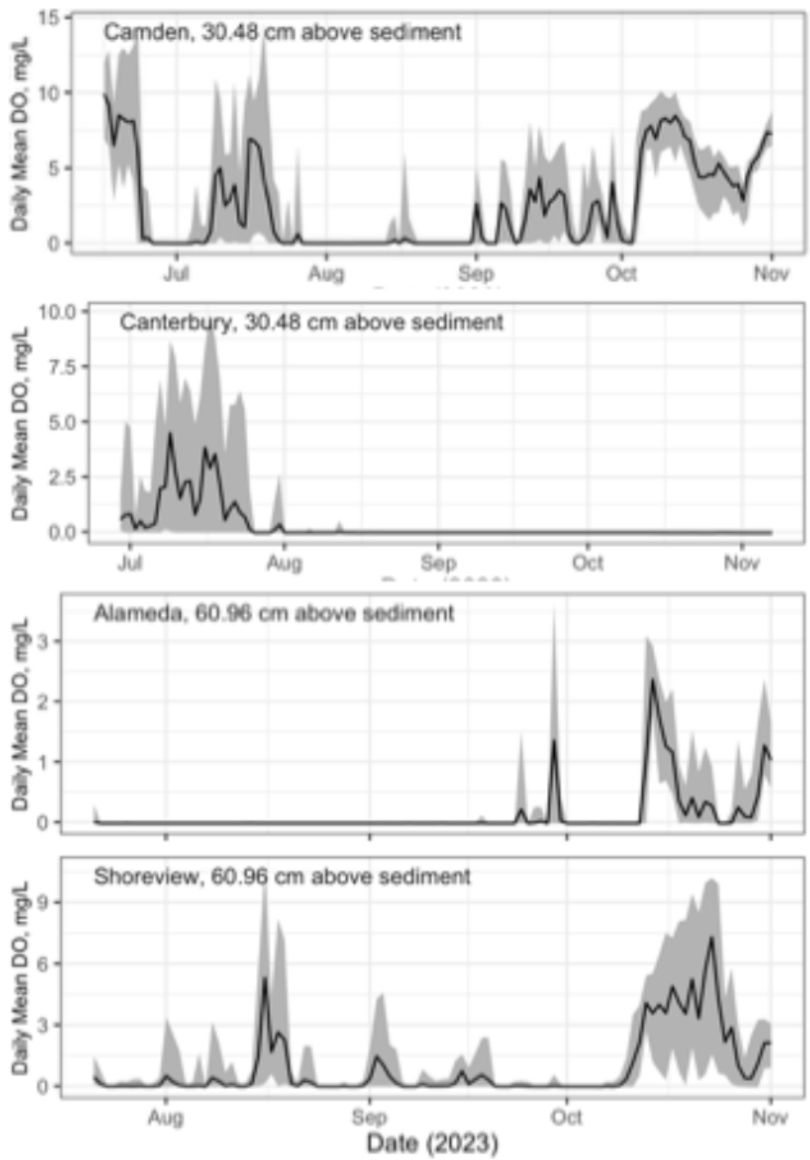


Figure B-13. Mean daily wind speed (m/s) (top row), mean daily water level (cm) (middle row), and daily relative thermal resistance to mixing (RTRM) as mean (black line) and minimum and maximum (dark gray envelope) (bottom row), observed at Briarcroft during the 2025 field season. The site was instrumented and monitored as part of another project. Note that the anemometer failed in August and a replacement was not available.



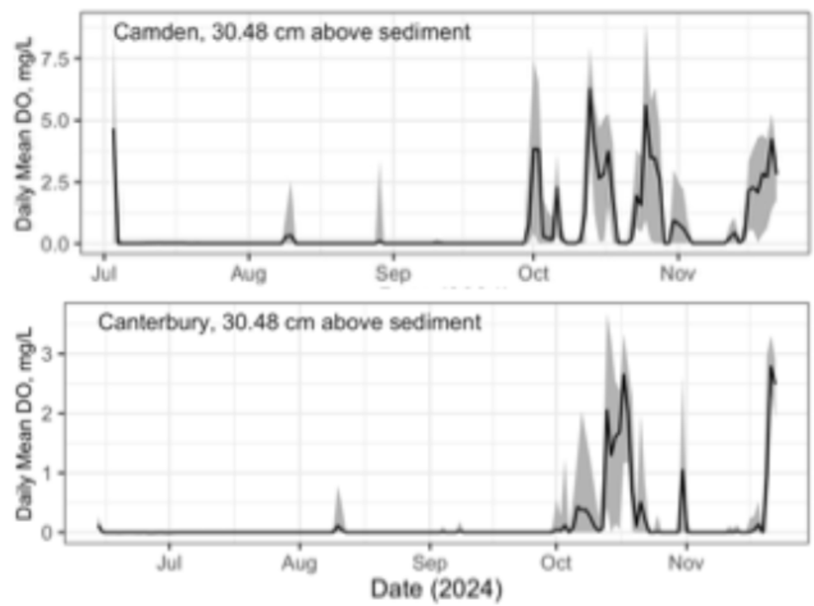


Figure B-14. Daily mean, minimum, and maximum dissolved oxygen concentration (mg/L) observed near the bottom of several of the monitored pond and historic wetland sites. Daily minimum and maximum illustrated by gray envelope around black line (mean DO). Left column shows DO observed during 2023 monitoring season, while the right column (two sites) shows DO observed during 2024 monitoring season. Note that no DO values above zero were observed at the LP-53 site (depth ~ 100 cm above sediment) during the entire season (July – November 2024).

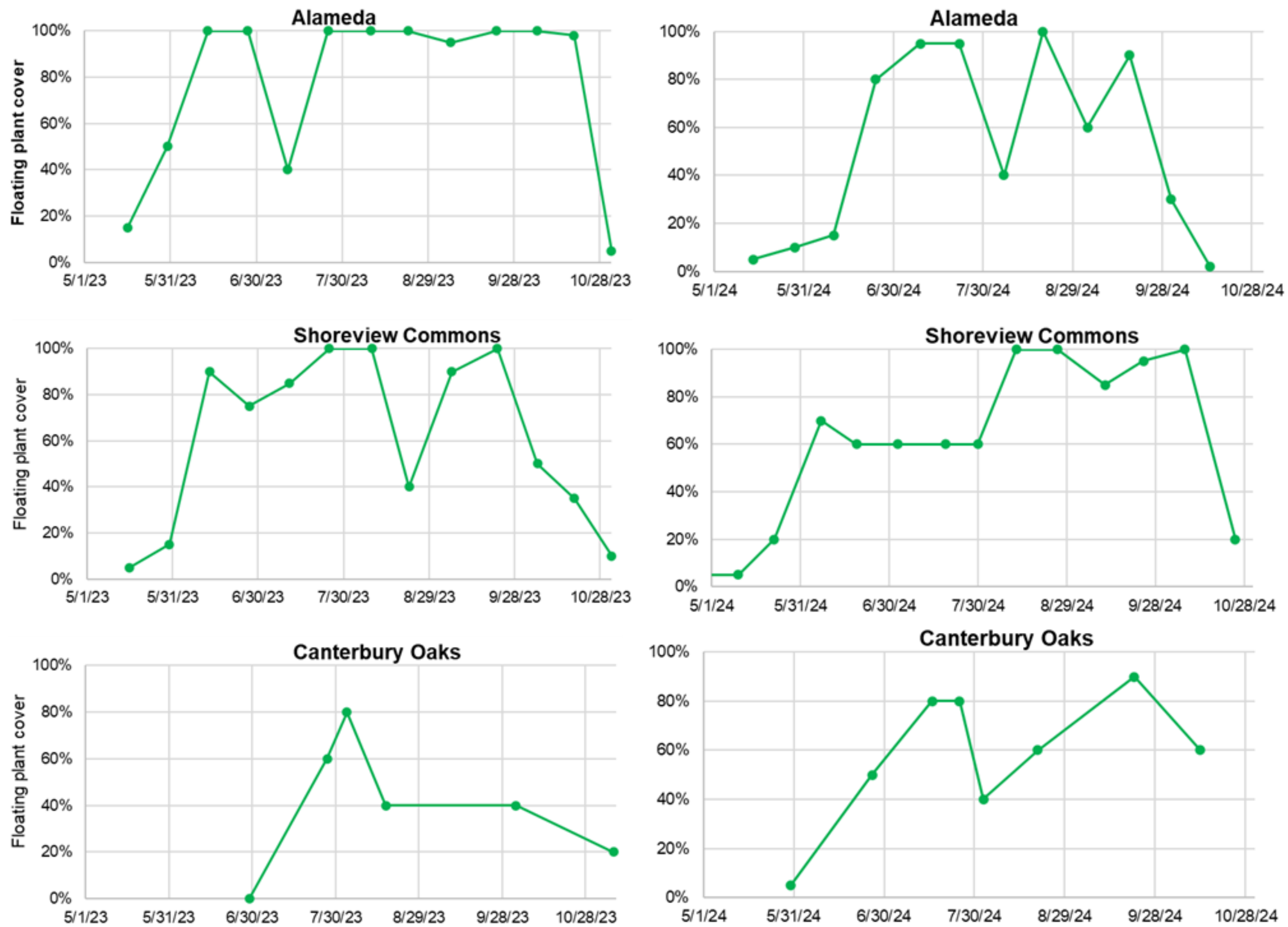
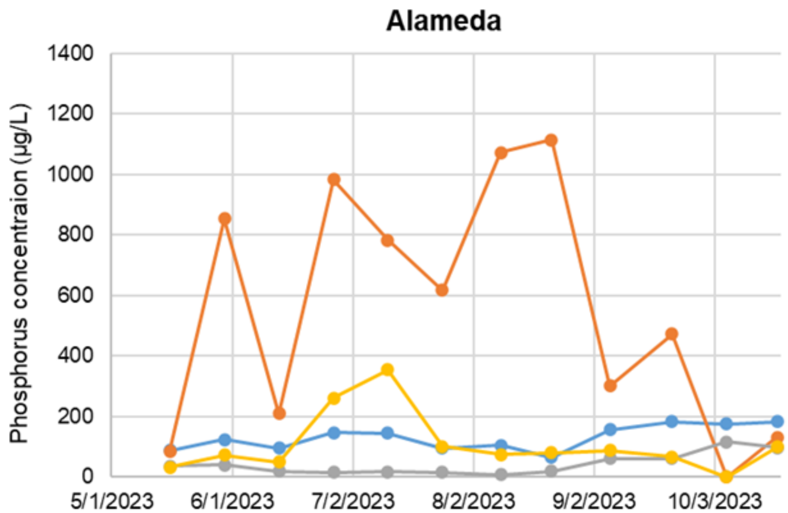
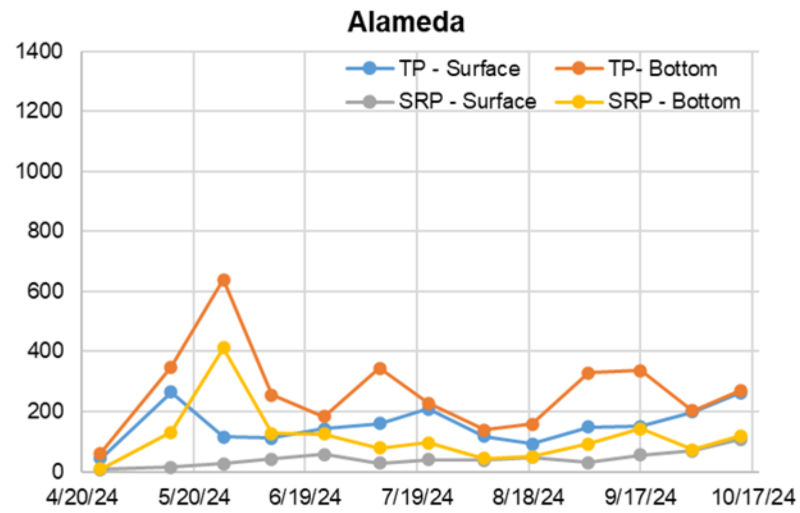
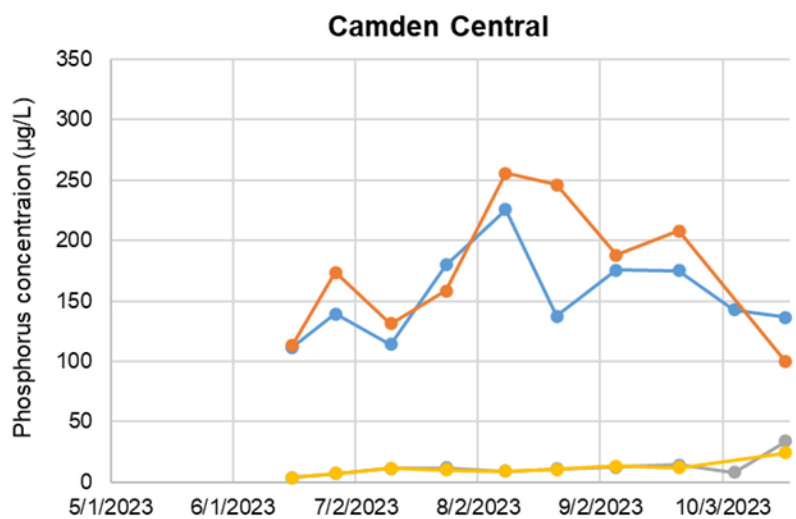
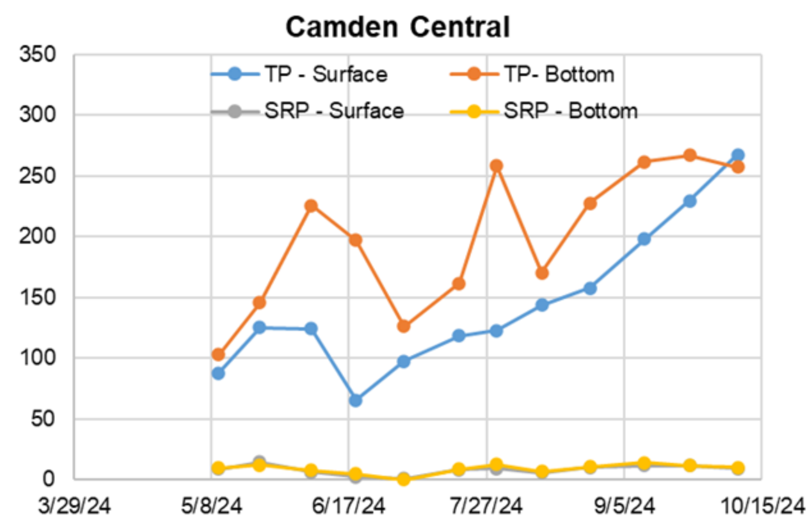


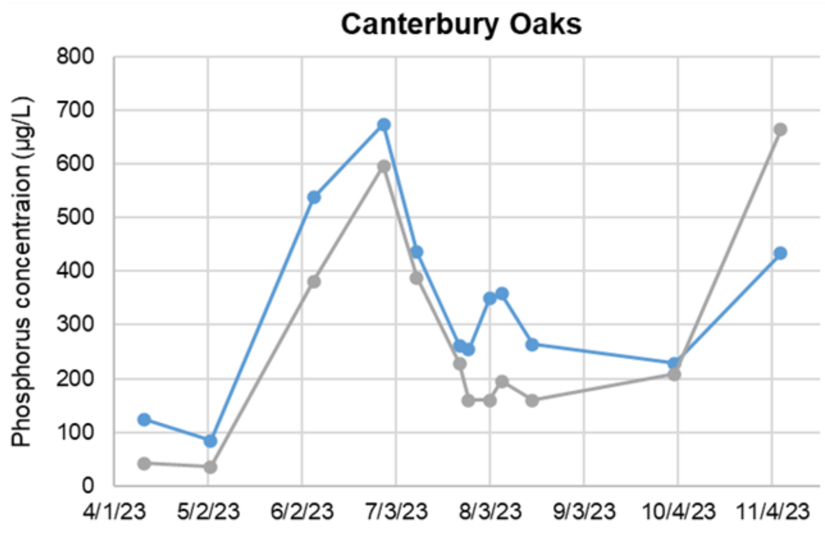
Figure B-15. Floating plant cover on the surface of Alameda, Shoreview Commons, and Canterbury Oaks during the 2023 and 2024 field seasons. Floating plants include duckweed and watermeal. Values shown are visual estimates by the field crew at the time of site visits. There were little to no floating plants on Camden and LP-53.

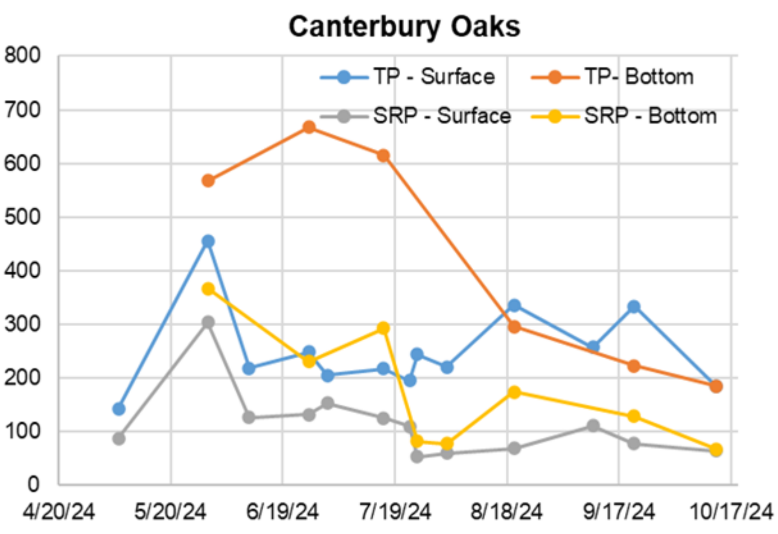

Alameda
Phosphorus concentraion (µg/L)
1400
1200
1000
800
600
400
200
0
5/1/2023
6/1/2023
7/2/2023
8/2/2023
9/2/2023
10/3/2023

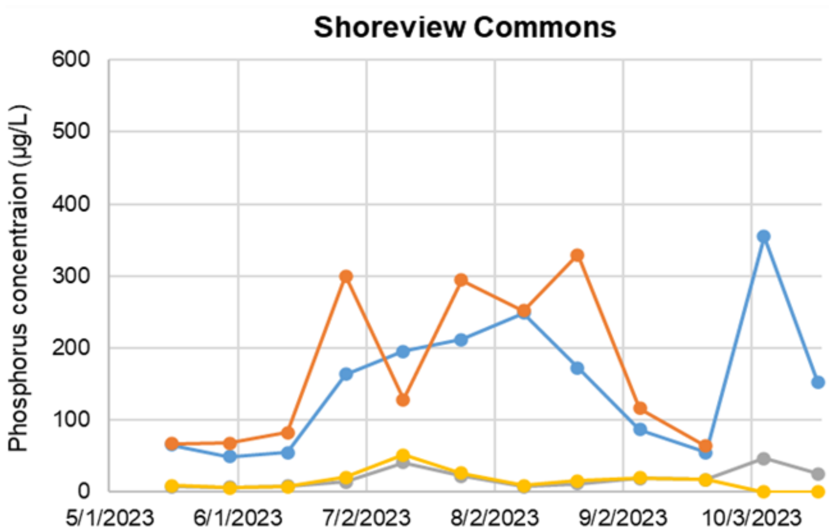

Alameda
TP - Surface
TP- Bottom
SRP - Surface
SRP - Bottom
1400
1200
1000
800
600
400
200
0
4/20/24
5/20/24
6/19/24
7/19/24
8/18/24
9/17/24
10/17/24

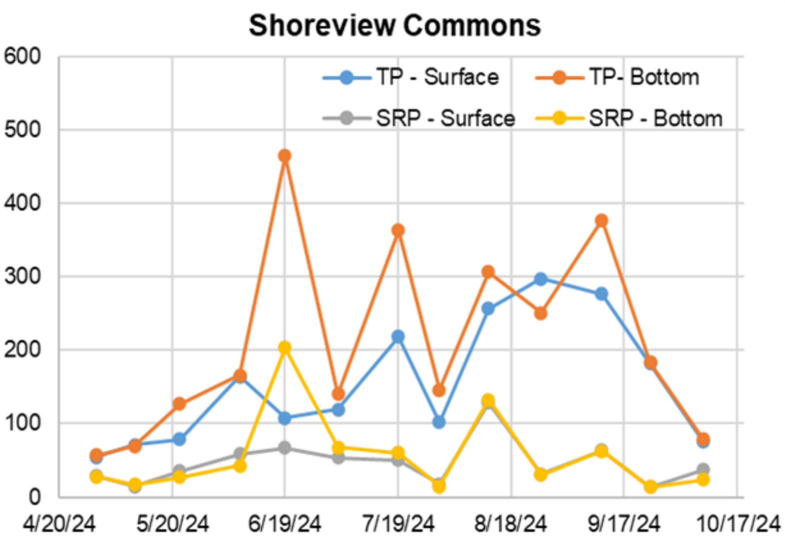

Camden Central
Phosphorus concentraion (µg/L)
350
300
250
200
150
100
50
0
5/1/2023
6/1/2023
7/2/2023
8/2/2023
9/2/2023
10/3/2023


Camden Central
TP - Surface
TP- Bottom
SRP - Surface
SRP - Bottom
350
300
250
200
150
100
50
0
3/29/24
5/8/24
6/17/24
7/27/24
9/5/24
10/15/24


Canterbury Oaks
Phosphorus concentraion (µg/L)
0
100
200
300
400
500
600
700
800
4/1/23
5/2/23
6/2/23
7/3/23
8/3/23
9/3/23
10/4/23
11/4/23


Canterbury Oaks
TP - Surface
TP- Bottom
SRP - Surface
SRP - Bottom
0
100
200
300
400
500
600
700
800
4/20/24
5/20/24
6/19/24
7/19/24
8/18/24
9/17/24
10/17/24


Shoreview Commons
Phosphorus concentraion (µg/L)
0
100
200
300
400
500
600
5/1/2023
6/1/2023
7/2/2023
8/2/2023
9/2/2023
10/3/2023


Shoreview Commons
TP - Surface
TP- Bottom
SRP - Surface
SRP - Bottom
0
100
200
300
400
500
600
4/20/24
5/20/24
6/19/24
7/19/24
8/18/24
9/17/24
10/17/24

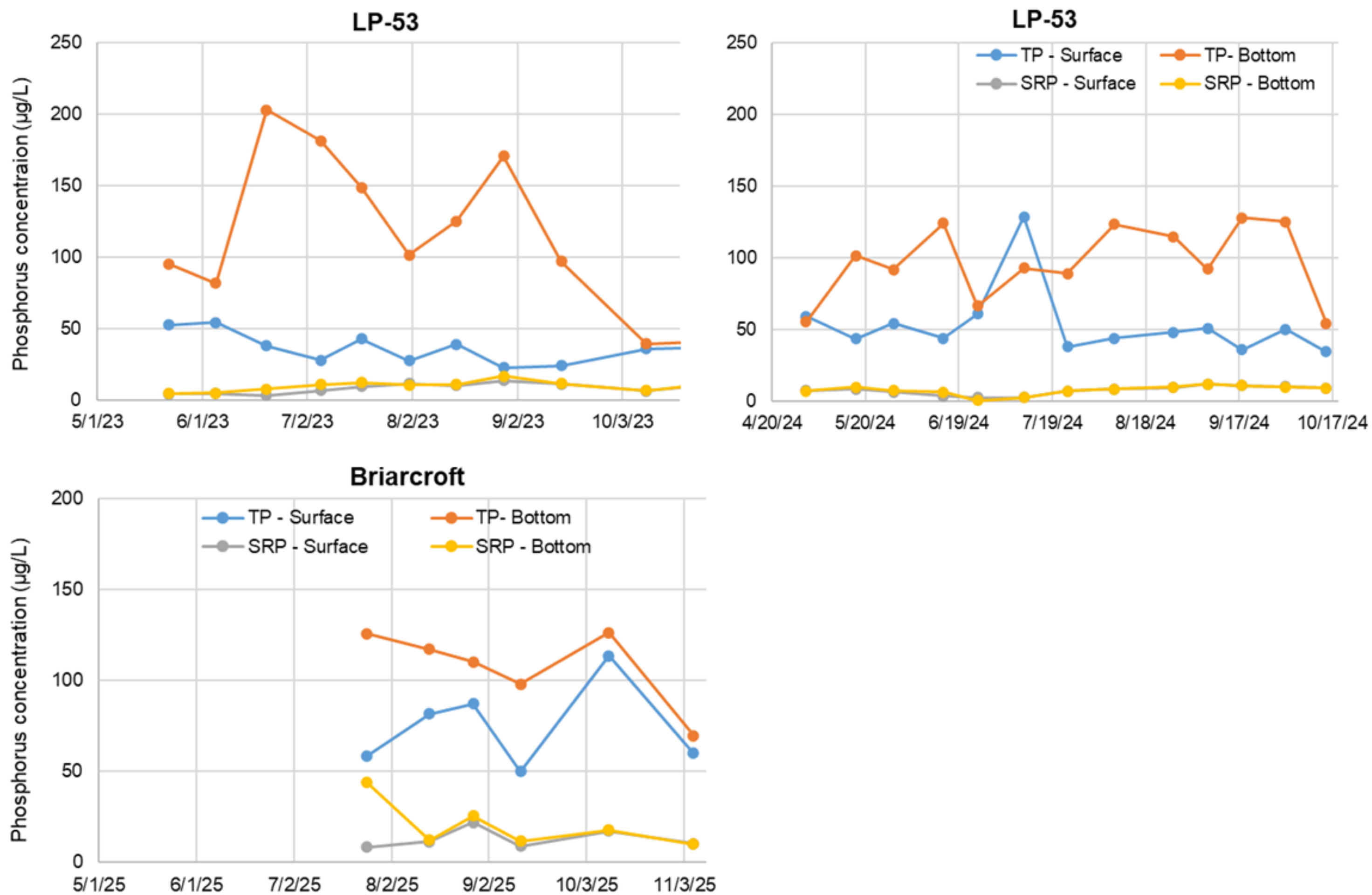


Figure B-16. Phosphorus water quality at Alameda, Camden Central, Canterbury Oaks, Shoreview Commons, and LP-53 during the 2023 and 2024 field seasons. Concentrations of total phosphorus (TP) and soluble reactive phosphorus (SRP) in the surface and bottom waters are plotted. Note difference in Y-axis scale in the plots. Concentrations are post-treatment for Shoreview Commons (treated with iron filings in 2021) and LP-53 (treated with alum in 2019). The 2023 data for Canterbury Oaks and 2025 data for Briarcroft were collected as part of the Finlay et al. (2026) study.

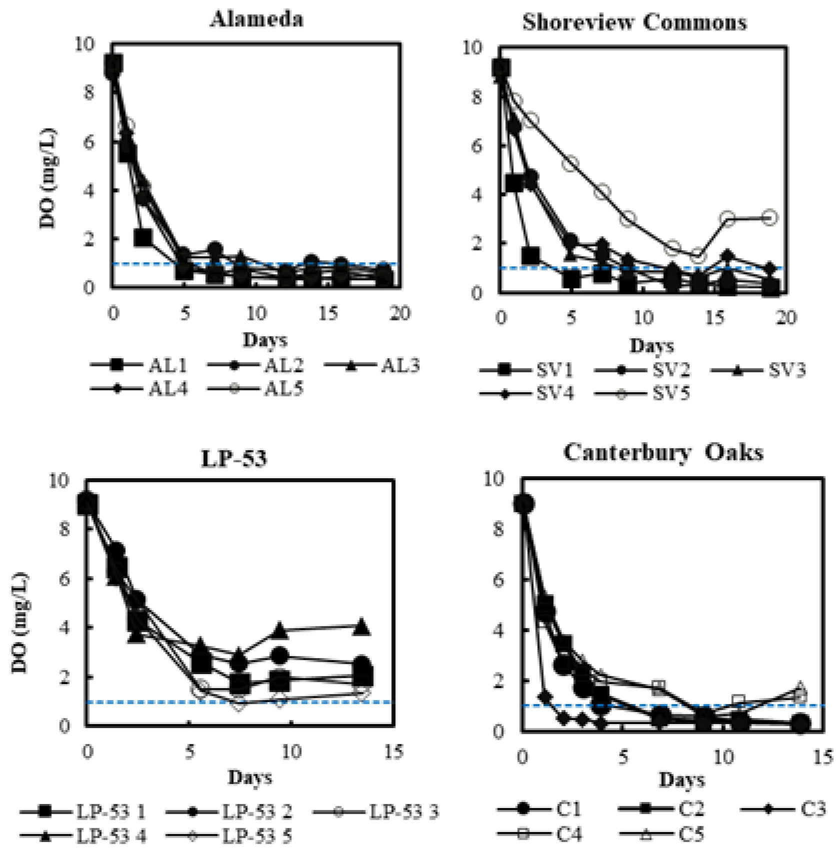


Figure B-17. Water column dissolved oxygen (DO) concentrations after air supply was switched off in the sediment cores from the five ponds and historic wetlands. The DO measurements were taken ~8 cm above the sediment surface. The 1 mg/L DO (dashed line) represents anoxic state.

Appendix C

Duckweed and Submersed Macrophyte Growth Model

Duckweed

The duckweed growth rate (μ) is formulated in terms of a maximum growth rate (μ_{max}), the surface water temperature (T), the light level (I), and the internal phosphorus concentration in the duckweed (N).

$$\mu = \mu_{max} \cdot f1(T) \cdot f2(I) \cdot f3(N) \tag{C-1}$$

Temperature Function

The temperature function for duckweed growth is formulated with two parameters, T_{opt} and T_{max}, in a similar manner to the algae growth temperature function (Riley and Stefan 1987):

$$f1(T) = exp\left(-2.3 \cdot \left(\frac{T - T_{opt}}{T_{opt}}\right)^2\right), T < T_{opt} \tag{C-2}$$

$$f1(T) = exp\left(-2.3 \cdot \left(\frac{T - T_{opt}}{T_{max} - T_{opt}}\right)^2\right), T > T_{opt} \tag{C-3}$$

Where T_{opt} is the optimum growth temperature and T_{max} is the maximum temperature for growth.

Light response

The light response of duckweed growth is calculated as an integral over the thickness of the duckweed, to take into account self-shading, where biomass at the bottom of the layer receives less light than the top of the layer – see Figure B.1.

$$\mu(I) = \frac{I}{I + \lambda} \tag{C-4}$$

$$I(z) = I_o exp(-c_2\, m(z)) \tag{C-5}$$

$$f2(I) = \frac{1}{m_1} \int_0^{m_1} \frac{I_o\, exp(-c_2\, m(z))}{I_o\, exp(-c_2\, m(z)) + \lambda} dm = \frac{1}{m_1 c_2}\, ln\left(\frac{\lambda / I_o + 1}{\lambda / I_o + exp(-c_2\, m_1)}\right) \tag{C-6}$$

where m is the duckweed biomass density (g/m^3), m1 is the total duckweed biomass per unit area (g/m^2), λ is the light half-saturation constant, and c_2 is the light attenuation constant for duckweed.

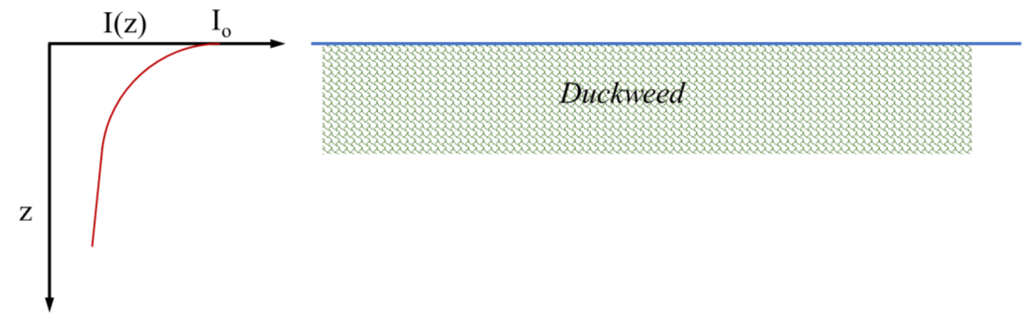


Figure C-1. Attenuation of light (I) over the thickness of a duckweed layer.

<u>Phosphorus function</u>

The response of growth rate to phosphorus availability includes the concept of luxury uptake, meaning the duckweed can uptake more phosphorus than is required for growth and store it. The growth model uses an equation for the uptake of phosphorus and a second equation for the effect of the internal phosphorus concentration on growth rate. These equations are very similar to the equations used for algal growth rate in MinPond (Riley and Stefan 1987).

$$f3(N) = \frac{N - m1 \cdot R_{min}}{N} \qquad \text{(C-7)}$$

$$u = \frac{\partial N}{\partial t} = u_{max} \cdot \frac{m1 \cdot R_{max} - N}{m1 \cdot (R_{max} - R_{min})} \cdot \frac{S}{\lambda_s + S} \qquad \text{(C-8)}$$

where N is the internal phosphorus concentration, t is time, m1 is the duckweed biomass, R is the duckweed biomass phosphorus ratio, bounded by R_{min} and R_{max}, u is the phosphorus uptake rate, u_{max} is the maximum phosphorus uptake rate, S is the available phosphorus concentration in the surface water layer, λ_s is the half-saturation constant for phosphorus uptake

Table C-1. Summary of duckweed model coefficients.

Parameter	Description	Value	Source
μ_{max} (day^{-1})	Maximum growth rate	0.35	Lasfar et al. (2007)
T_{opt} (°C)	Optimum growth temperature	22	Lasfar et al. (2007)
T_{max} (°C)	Maximum growth temperature	32	Lasfar et al. (2007)
λ (µE/m^2/day)	Half-saturation constant for light	350	Van Gerven et al. (2015)
c_2 (m^2 gDW^{-1})	Light attenuation coefficient	0.02	U of M data
R_{max} (gP/gDW)	Maximum phosphorus ratio	0.001	U of M data
R_{min} (gP/gDW)	Minimum phosphorus ratio	0.005	U of M data
λ_s (mg/l)	Half-saturation constant for phosphorus uptake	0.03	Calibration
u_{max} (gP/gDW/day)	Maximum phosphorus uptake rate	0.00017	Calibration

Submersed Macrophytes

The submersed macrophyte growth model used in MinPond was adapted from a model developed for a concurrent USGS Aquatic Invasive Species project ("Managing water quality and invasive macrophytes to promote healthy native aquatic plant communities", PIs Ray Newman and Jeff Peterson). MinPond has a series of sediment layers defined, independent of the water layers (Figure B.2). The model allows a group (cohort) of plants to grow, rooted in each of the sediment layers. An initial biomass is specified for each plant cohort, and the cohort grows over the season as a function of available light, temperature, and nutrients.

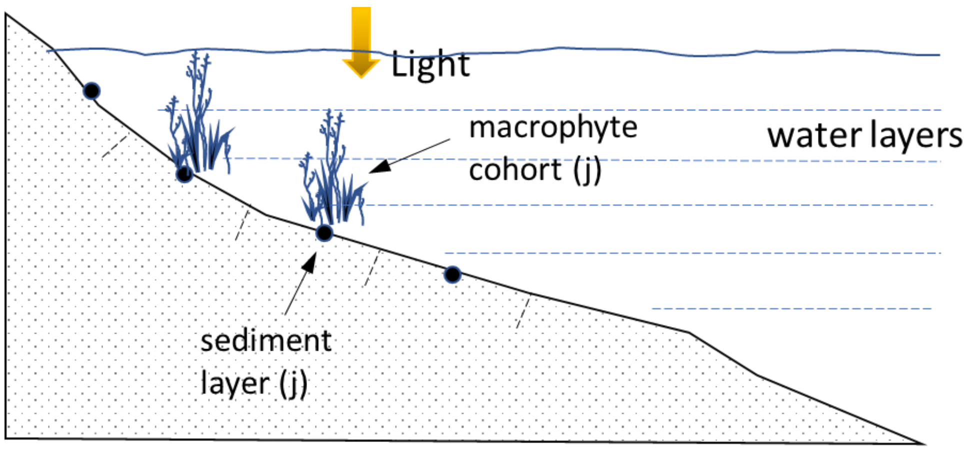


Figure C-2. Diagram of water layers, sediment layers, and submersed macrophyte cohorts.

As the macrophyte cohort grows upward in the water column, the plant mass is divided into a series of vertical layers. The rate of growth in each layer is calculated based on the local water temperature, light availability, and nutrient availability. The net growth rate includes additional terms for temperature-dependent plant respiration and a fixed mortality rate, to account for plant senescence and grazing (details not shown). The overall net growth of the cohort is then used to increase the plant height towards the water surface.

Temperature Function

The temperature function for macrophyte growth is identical to the MINLAKE algae growth model, with two parameters, T_{opt} and T_{max}, (Riley and Stefan 1987):

$$f1(T) = exp\left(-2.3 \cdot \left(\frac{T-T_{opt}}{T_{opt}}\right)^2\right), T < T_{opt} \qquad \text{(C-9)}$$

$$f1(T) = exp\left(-2.3 \cdot \left(\frac{T-T_{opt}}{T_{max}-T_{opt}}\right)^2\right), T > T_{opt} \qquad \text{(C-10)}$$

Where T_{opt} is the optimum growth temperature and T_{max} is the maximum temperature for growth.

Light response

The light response of macrophyte growth is calculated for each layer in each plant cohort. The local light conditions at each depth take into account background attenuation by the water, attenuation by algae, and attenuation by the plant itself (self-shading).

$$\mu_m(I) = \frac{I}{I+\lambda_m} \quad \text{(C-11)}$$

$$I(z) = I_o exp(-K_T \cdot z) \quad \text{(C-12)}$$

$$K_T = K_W + k_a \cdot Chl + k_m \cdot m \quad \text{(C-13)}$$

where μ_m is the plant growth rate, I is the local light intensity, Io is the surface light intensity, λ_m is the light half-saturation coefficient for the plants, K_T is the total light attenuation coefficient, K_W is the background attenuation of the water, k_a is the specific attenuation rate of algae, Chl is the algae concentration, k_m is the specific attenuation rate of the plant biomass, and m is the plant density (g/m^3).

Phosphorus function

For this analysis, the rooted plants are assumed to obtain their nutrients from the sediment, and were therefore not nutrient limited (Chambers et al. 1989).

Table C-2. Summary of rooted plant model coefficients

Parameter	Description	Value	Source
μ_m (day^{-1})	Maximum growth rate	0.35	Calibration
T_{opt} (°C)	Optimum growth temperature	20	Hootsmans 1994
T_{max} (°C)	Maximum growth temperature	30	Hootsmans 1994
λ_m (µE/m^2/day)	Half-saturation constant for light	200	Herb & Stefan, 2003
k_m (m^2 gDW^{-1})	Light attenuation coefficient	0.024	Herb & Stefan, 2003

Appendix D

Information on the Cost of Remediation Treatments

Costing information for sediment chemical treatments (alum, iron filings) and watershed load reductions were taken and adapted from Taguchi et al. (2022), as follows.

Alum Treatment

As described in Taguchi et al. (2022), we used water quality data from four alum-treated lakes in the Riley Purgatory Bluff Creek Watershed District (RPBCWD), MN to estimate an average change in hypolimnetic total phosphorus (TP) concentrations in response to alum. The average reduction in TP concentration (available data before treatment vs. available data after treatment) was 77% (71% minimum; 86% maximum). For the purposes of modeling the chemical treatment of sediments scenario, the measured sediment phosphorus release rate coefficient was reduced by 77% for the medium application scenario.

Alum dosages for Alameda and Shoreview Commons wetlands were previous estimated in Taguchi et al. (2022) based on sediment data for each pond or historic wetland, targeting a 20:1 Al:P ratio. These values were then applied only to the portion of the sediments likely to experience anoxic conditions (the deeper portions) rather than the entire sediment surface area. Alum dosages for Camden Central and Canterbury Oaks were estimated from the Alameda/Shoreview Commons numbers assuming the application rates are proportional to pond or historic wetland area and anoxic sediment release rate, as summarized in Table D-1.

Cost estimates were based on values from alum applications in lakes in the City of Eagan, MN (data provided by the City of Eagan). Material cost estimates were \$0.55/liter (\$2.10/gal) for alum and \$1.48/liter (\$5.60/gal) for sodium aluminate buffer solution (one part buffer for every two parts alum) when aluminum toxicity due to pH change is of concern. For the ponds and historic wetlands in this study, no buffer was included in the cost estimates, but a 10% contingency was factored in. A major variable in chemical treatments for ponds and historic wetlands are mobilization costs. Based on pond or historic wetland and lake treatments in Eagan, the mean multiplier for material costs to arrive at the total costs was 2.6 (1.4 minimum; 4.6 maximum). For the purpose of the cost estimates in this study, we used a multiplier value of 3. Estimating the frequency of reapplication is difficult because of the lack of long-term data on stormwater pond or historic wetland applications as well as the high site-to-site variability. Based on extensive data from lakes (Huser et al. 2016), we assumed a 10-year lifespan.

Iron Filings Treatment

We considered iron filings as an alternative to alum for chemical treatment of pond or historic wetland sediments. The concept is to overload the sediments with elemental iron so that sufficient iron of various charges is present to restrict release even under low DO conditions (Natarajan et al. 2021). For this study, we assumed that the results from a recent iron filings application in the Shoreview Commons historic wetland are most applicable, with a 55% reduction in SRP release rates. Differences in longevity between alum and iron filings applications are likely but as of yet unknown. For the purposes of cost estimates in this study, we are assuming that either treatment would last for 10 years before reapplication became necessary.

The Shoreview Commons historic wetland application of iron filings was dosed at a rate of 0.58 kg/m^2 (0.12 lb/ft^2), which is somewhat greater than the dosage of 0.50 kg/m^2 (0.10 lb/ft^2) derived from laboratory study results (Natarajan et al. 2017). The cost estimates assume a material cost of $0.93/kg ($0.42/lb) iron and an approximate shipping cost of $558 from the vendor to Minneapolis (Section 2). Unlike with alum, no specialized crew is required to mobilize to the historic wetland site and deploy specialized equipment. We estimated the cost of municipal staff time and the use of the necessary equipment at $3,000, although the true amount could be greater if additional or more specialized staff or equipment are found to be necessary.

Table D-1. Pond or historic wetland characteristics used to estimate alum and iron filings dosages and the associated costs.

	Alameda	Camden Central	Canterbury Oaks	Shoreview Commons	LP-53	Briarcroft
Pond/Historic Wetland Area (ac)	2.9	4.1	0.84	2.9	3.9	1.0
Anoxic Sediment Release Rate ($mg/m^2/day$)	7.5	4.0	3.7	3.2	7.1	5.0**
Alum Dosage (liters)	8,680	6,560	1,240	5,580	11,080	2000
Alum Treatment Total Cost	$16,000	$12,000	$2,200*	$10,000	$20,400	$3,680*
Iron Filings Dosage (kg)	6,800	9,620	1,970	6,800	9,140	2,340
Iron Filing Total Cost	$9,900	$13,200	$5,500	$9,900	$13,300	$3,400

*This cost was extrapolated from larger ponds and historic wetlands, and is likely too low.
**Estimated

Watershed-based methods

<u>Volume Reduction</u>

Phosphorus load reduction from street sweeping and associated costs were estimated based on a municipal street sweeping study by Kalinosky et al. (2014) in Prior Lake, Minnesota. They studied street sweeping along routes with low-, medium-, and high- tree canopy densities at frequencies of one, two, and four times a month over two years and tabulated costs including labor, fuel, and operation and maintenance of the street sweeper vehicles; the cost of purchasing each street sweeper vehicle was not included. The overall average was found to be $707/kg ($321/lb) of TP mass removed (costs adjusted for inflation from 2014 USD to 2021 USD values (U.S. Bureau of Labor Statistics 2021).

The cost of volume reduction (here modeled as the 25% Volume reduction scenario) used data from Weiss et al. (2007), based on the construction costs of infiltration basin practices and 20 years of operation and maintenance costs, with prices calculated according to the water quality volume (WQV), in units of cubic meters, treated by the practice:

$Infiltration\ Basin\ Cost\ (2005\ USD) = 1281 * \ WQV^{0.634}$ (Weiss et al. 2007) (D-1)

The required water quality volume needed to capture 25% of the total runoff input to each pond or historic wetland or historic wetland was calculated as follows:

1. An Excel spreadsheet was set up with the 10-year flow input time series (hourly time step)
2. The flow was routed through a storage volume, representing a single biofiltration unit upstream of the pond/historic wetland
3. A simple water budget model was used to calculate the storage volume and outflow (overflow), assuming the infiltration rate was sufficient to drain the water quality volume in 24 hours
4. The storage volume was adjusted to achieve a 25% reduction in volume reaching the pond
5. Equation C.1 was then used to estimate a cost, assuming either a single BMP with the entire required storage volume or 10 BMPs spread over the watershed.

We adjusted tabulated cost values by a factor of 1.46 for inflation from 2005 USD to 2021 USD values (U.S. Bureau of Labor Statistics 2021). Additional formula coefficients were provided by Weiss et al. (2007) for lower and upper 67% confidence interval values in order to capture the wide variability in cost estimates for watershed-based method – the cost confidence intervals were approximately +100%, -35% of the nominal costs given in Table D-2. We also multiplied the costs calculated using the Weiss et al. (2007) by a factor of 0.85 to yield 10-year costs rather than 20-year costs, to compare them against the other remediation strategies evaluated in this study.

Nutrient Reduction

An estimate was made for the cost of TP mass reduction using municipal street sweeping, as studied by Kalinosky et al. (2014) in Prior Lake, Minnesota. They studied street sweeping along routes with low-, medium-, and high- tree canopy densities at frequencies of one, two, and four times a month over two years and tabulated costs including labor, fuel, and operation and maintenance of the street sweeper vehicles; the cost of purchasing each street sweeper vehicle was not included. The overall average was found to be $707/kg ($321/lb) of TP mass removed (costs adjusted for inflation from 2014 USD to 2021 USD values (U.S. Bureau of Labor Statistics 2021). In specific months, cost values ranged from an average of $107/kg ($49/lb) TP for an area with high tree canopy density being swept twice a month in October to an average of approximately $1,675/kg ($760/lb) for an area with low tree canopy density being swept four times a month in July. The cost numbers of Kalinosky et al. (2014) are somewhat lower than those given by Baker et al. (2014), which estimated costs of street sweeping between $200/kg P ($100/lb P) and $1320/kg P ($600/lb P) removed – this study was also based on data from Prior Lake, MN, but also includes estimated costs for the sweepers.

We used an averaged cost from the Baker et al. (2014) study ($1100/kg) to estimate the cost to reduced TP loading to each pond/historic wetland by 25% (Table D-2).

Table D-2. Summary of information used to estimate costs of watershed-based flow reduction and nutrient reduction mitigation.

	Alameda	Shoreview Commons	Camden Central	Canterbury Oaks	LP-53	Briarcroft
Mean Annual Inflow (m^3/year)	58,400	58,400	77,100	8,760	4,440	8,330
Mean Inflow TP Concentration (mg/l)	0.31	0.31	0.31	0.31	0.31	0.31
Mean TP Mass Inflow (kg/year)	18.1	18.1	23.9	2.72	2.76	13.77
25% Volume Reduction (m^3/year)	14,600	14,600	19,300	2,190	2,220	11,100
Required Water Quality Volume (m^3)	160	160	215	24	25.4	127
Estimate Cost, 1 BMP	$39,600	$39,600	$47,800	$11,900	$12,400	$34,300
Estimate Cost, 10 BMPs	$91,900	$91,900	$111,100	$27,700	$28,700	$79,600
25% Nutrient Reduction (kg P/year)	4.5	4.5	6.0	0.68	0.69	3.44
25% Nutrient Reduction Cost	$4,980	$4,980	$6,580	$750	$760	$3790