DEPARTMENT OF TRANSPORTATION

Stormwater Pond Maintenance, and Wetland Management for Phosphorus Retention

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Research Project
Final Report 2023-25

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Reduction in phosphorus is critical because phosphate, a dissolved form of phosphorus, sustains algal and cyanobacteria growth and causes a wide range of water-quality impairments in the ponds and downstream waters including algal blooms, excess floating plants, taste, and odor problems. Many stormwater ponds and wetlands that treat stormwater appear to be less effective than expected or originally intended in phosphorus retention, a key function of these ponds in urban environments. There is evidence that many old ponds are releasing phosphorus from bottom sediments at high rates and likely exporting phosphorus to downstream surface water bodies. A major outcome of this project is a pond Assessment Tool to assess the risk of high phosphorus concentrations in ponds and sediment release of phosphorus. The tool is based on 20 ponds with detailed water quality and phosphorus release measurements and a meta-analysis of 230 ponds in the Twin Cities metro area. Other outcomes included a working definition of a constructed stormwater pond and a wetland treating stormwater in the framework of water-body regulations, the development of recommendations for stormwater pond maintenance and wetland management, and an update to the sections on the constructed stormwater ponds section of the *2009 Stormwater Maintenance BMP Guide*.

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Stormwater Pond Maintenance, and Wetland Management for Phosphorus Retention

Final Report

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Definitions

In this report, we address the water quality in stormwater ponds, which is an indicator of pond performance for water quality treatment and closely related to the water quality released from these stormwater ponds to downstream waters. The following are definitions of terms that will be used throughout this document:

Constructed stormwater pond (stormwater pond)**:** An upland constructed water body of less than 10 acres in surface area which has infrastructure to deliver and release stormwater runoff. Such ponds typically contain standing water for a majority of the time. These ponds are not subject to Wetland Conservation Act (WCA) or Clean Water Act (CWA) 404 regulations; however, Public Waters Work Permit Program (PWWPP) regulations apply to any of these that are identified as public waters on the DNR's Public Waters Inventory. Such sites are considered to be a treatment device and must follow design requirements in MPCA's general construction stormwater permit when projects create one or more acres of new impervious surface. These constructed stormwater ponds and other treatment practices are also regulated under the MPCA Municipal Separate Storm Sewer System (MS4) program where applicable.

Wetland treating stormwater: a historical wetland area that has been modified or is managed to produce a clean water service for a downstream water body, and still meets the regulatory definition of a wetland (typically a Type 5 wetland). These wetlands are subject to WCA regulations and potentially CWA 404 and 401 requirements if they are Waters of the U.S. They are subject to PWWPP regulations if they are or are within designated public waters. Please see Chapter 2 for a more complete description.

Ponds: For the purpose of this report, both constructed stormwater ponds and wetlands that treat stormwater will be referred to as ponds. The differentiation between the two types of water bodies is primarily regulatory, and does not affect the purpose of this document.

Pond 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 0.91 m (3 ft) deep.

Open water pond: A pond as above, with a majority of open water (without emergent vegetation). These are typically between 0.61 and 3.1 m (2 and 10 feet) in depth.

Executive Summary

Background and Objectives

Stormwater ponds and wetlands across a wide variety of characteristics (e.g., watershed history, age, size, depth, aquatic vegetation) are used to treat stormwater to remove a significant portion of sediment and associated pollutants in runoff (e.g., phosphorus, nitrogen, hydrocarbons, and metals) prior to the runoff being released downstream. An important contemporary management issue is that some stormwater ponds appear to be less effective than expected or originally intended in phosphorus (P) retention, a key function of these stormwater control measures (SCMs) in urban environments. There is evidence that many old constructed stormwater ponds and wetlands treating stormwater are releasing phosphorus from bottom sediments at high rates and likely exporting phosphorus to downstream surface water bodies (Taguchi et al. 2018, 2020). The potential for phosphorus re-release from accumulated sediments in ponds and factors related to climate, watershed, pond characteristics that regulate phosphorus release or burial are important in controlling phosphorus dynamics in ponds (Janke et al. 2021). Reduction in phosphorus is critical because phosphate, a dissolved form of phosphorus, sustains algal and cyanobacteria growth and causes a wide range of water quality impairments in the ponds and downstream waters including algal blooms, excess floating plants, taste, and odor problems.

The overall objective of this project is to define, identify, and evaluate risk factors to water quality performance, primarily phosphorus retention, of stormwater ponds and wetlands treating stormwater. The main project objectives and tasks are to:

1) Define existing stormwater ponds and wetlands treating stormwater, develop field studies to fill knowledge gaps, and synthesis existing data to support project deliverables;

2) Establish robust relationships between pond/wetland features and functionality through broad field assessment, directed laboratory studies, and data synthesis;

3) Develop a spreadsheet tool to evaluate site-specific indicators of functionality (or failure) for performance for phosphorus treatment; and

4) Recommend maintenance or management protocols for stormwater ponds and wetlands that treat stormwater; this will include an update to the 2009 LRRB Research Implementation Committee (RIC) report “Stormwater Maintenance Best Management Practices [BMP] Resource Guide” for stormwater ponds and wetlands that treat stormwater with regard to phosphorus release from sediments, as well as managing polycyclic aromatic hydrocarbon (PAH) concerns in pond and wetland dredging.

By understanding the effectiveness and risk factors that determine effectiveness of constructed stormwater ponds and wetlands that treat stormwater for phosphorus removal, the project will help develop and prioritize maintenance and management activities to provide optimal performance for phosphorus retention in these stormwater bodies. The information will help stormwater managers who are responsible for evaluating and maintaining stormwater ponds and wetlands that treat stormwater. Ultimately, better management will guide allocation and more efficient use of limited resources to address environmental impacts of stormwater.

Overview of Methods and Activities

Major project methods and activities include the following:

1) With the aid of the Technical Advisory Panel, a working definition of constructed stormwater pond and wetland treating stormwater in the framework of water body regulations for purposes of study in this project, the development of outcomes and to assist in stormwater practitioners in piloting a path through the multitude of regulations. Results are presented in Chapter 2 of this report.

2) A field survey was conducted over one field season to monitor phosphorus, water quality and in situ conditions (dissolved oxygen, temperature, specific conductivity) in 6 ponds chosen to represent pond features that were under-represented in our past studies, including the presence of emergent vegetation (20 to 86%) and free-floating vegetation (20 to 86%). The data collected revealed the complex nature of ponds with emergent vegetation, as the surface TP concentrations were more variable and higher in the vegetated portions when compared to the open water portion of the ponds. These results are presented in Chapter 3 of this report, with documentation in Appendix A.

3) Intensive monitoring of 5 ponds (selected from the survey task 3) was performed during the second field season; monitoring included bi-weekly water sampling and measurement of in-situ conditions, as well as continuous monitoring of temperature profiles, conductivity, wind speed and wind direction over the pond to assess stratification and water mixing. The correlation between free-floating vegetation, oxygen levels (as anoxic factor) and TP became more apparent with the new data. These results are presented in Chapter 3 of this report, with documentation in Appendix A.

4) Sediment cores extracted from the 5 ponds identified in task (4) were incubated in the laboratory to determine the oxic and anoxic sediment phosphorus release rates, and the mass of various sediment phosphorus forms (i.e., bioavailable vs. unavailable P). We were able to evaluate the prevalence of phosphate release from the sediments and establish relationships between sediment chemistry and risk factors for phosphate release. These results are presented in Chapter 4 of this report, with documentation in Appendix B.

5) Existing pond data were aggregated, reviewed, and synthesized to inform project objectives; these data included water quality (phosphorus, dissolved oxygen, temperature, conductivity) and pond and/or watershed characteristics (metadata) in about 230 water bodies in the Twin Cities metropolitan area over approximately the past 15 years. We were able to establish broad patterns related to categories of indicators important to total phosphorus (TP) concentration in ponds. These categories were related to potential watershed inputs (surrounding land use and land cover, including tree canopy), hydrogeologic setting (proximity to hydric soils or historic water bodies), water body characteristics (depth, vegetation cover), and classification (MnDNR/National Wetland

Inventory and Metro Mosquito Control District Circular 39). These results are presented in Chapter 5 of this report.

6) Using the new field data and data for 15 intensively-monitored ponds collected during previous projects (Janke et al. 2020; Taguchi et al. 2018, 2020), we further strengthened the existing relationships between phosphorus conditions and important risk factors (pond age, pond depth, tree cover, vegetation cover, watershed/land use type, sediment characteristics, environmental and climatic conditions). These results are presented in Chapter 5 of this report.

7) The Pond Assessment Tool was developed to identify and evaluate indicators of functionality (or failure), primarily focused on phosphorus, i.e., higher phosphorus concentrations, lower dissolved oxygen concentrations, and higher sediment phosphorus release in ponds. While the broad patterns established from data analysis of 230 water bodies formed the basis of the screening tool, statistical analysis of field data provided regression models for predicting pond TP, pond anoxia, and sediment release. These results are presented in Chapter 6 of this report, with a tutorial in Appendix C.

8) Recommendations for stormwater pond maintenance and wetland management were proposed, based on project results, data synthesis, and a literature review of PAHs in pond sediments, construction and maintenance costs, and runoff storage/settling alternatives. We complemented and supported this activity with information from a recent Local Road Research Board (LRRB) funded project "Wet Pond Maintenance for Phosphorus Retention" (Taguchi et al. 2022) in which maintenance measures to limit phosphorus release in ponds were investigated. These results are presented in Appendix D of this report.

9) Using project results, a review and update to the sections on constructed stormwater ponds incorporating maintenance activities to reduce sediment release of phosphorus were provided for two existing documents: 2009 Stormwater Maintenance BMP Guide, and 2011 Decision Tree for Stormwater BMPs. These results are presented in Appendix E of this report.

Outcomes and Benefits

The Pond Assessment Tool (Chapter 6 and Appendix C) is the major outcome of this project. The research conducted in this project combined with an existing pond dataset based upon previous research by the authors enabled the development of the multi-level pond assessment toolbox for evaluating ponds at risk for poor phosphorus water quality (see Figure ES-1 for tool framework). The tool uses factors or risk indicators related to a number of processes important to phosphorus dynamics in ponds, including phosphorus or sediment inputs to ponds (related to watershed), oxygen dynamics and vertical transport within ponds (through mixing or stratification processes), vegetation, sediment characteristics, legacies of sediment, organic matter, and phosphorus inputs.

The tool includes several assessment options, using input data that are either basic (drawings, plans, aerial photographs, site visits), sampling and monitoring data (periodic grab sampling/profiling of pond water) and sediment sampling. The first assessment (Tool 1-A) is a method of screening ponds for levels of indicators related to risk of high pond water TP using basic site data. The second assessment (Tool 1-

B) provides predictions (numerical estimates) of pond phosphorus concentrations using input of more detailed sampling and/or monitoring data. The assessment of pond oxygen status (Tool 2) predicts pond anoxia using anoxic factor or dissolved oxygen concentrations, which can indicate risk of potential sediment phosphorus release. The third assessment (Tool 3) provides models to predict anoxic sediment phosphate release using a combination of basic data and detailed sampling data.

We combined the Pond Assessment Tool with recommendations on pond maintenance methods to address the risk indicators identified by the tool and guide efforts to improve phosphorus removal in existing ponds (Appendix D). Together, the assessment tool and the maintenance framework will help cities, watershed managers, and state agencies to prioritize, allocate, and more efficiently use limited resources for environmental management. Thus, the research has resulted in three primary benefits to the state of Minnesota:

1) Environmental Aspects: Cost-effective management of constructed stormwater ponds and wetlands treating stormwater for optimal treatment performance will improve runoff quality and allow the allocation of resources for other environmental projects.

2) Operations and Maintenance Savings: Understanding factors affecting performance, and developing tools to evaluate proper functionality and guide maintenance will help prioritize constructed stormwater ponds and wetlands treating stormwater that yield the most benefit to water quality.

3) Reduce Road User Cost: Measures to minimize pollutant load from stormwater ponds and wetlands treating stormwater will reduce the cost for treating downstream lakes that receive polluted runoff. This will allow municipalities to reduce taxes to road users.

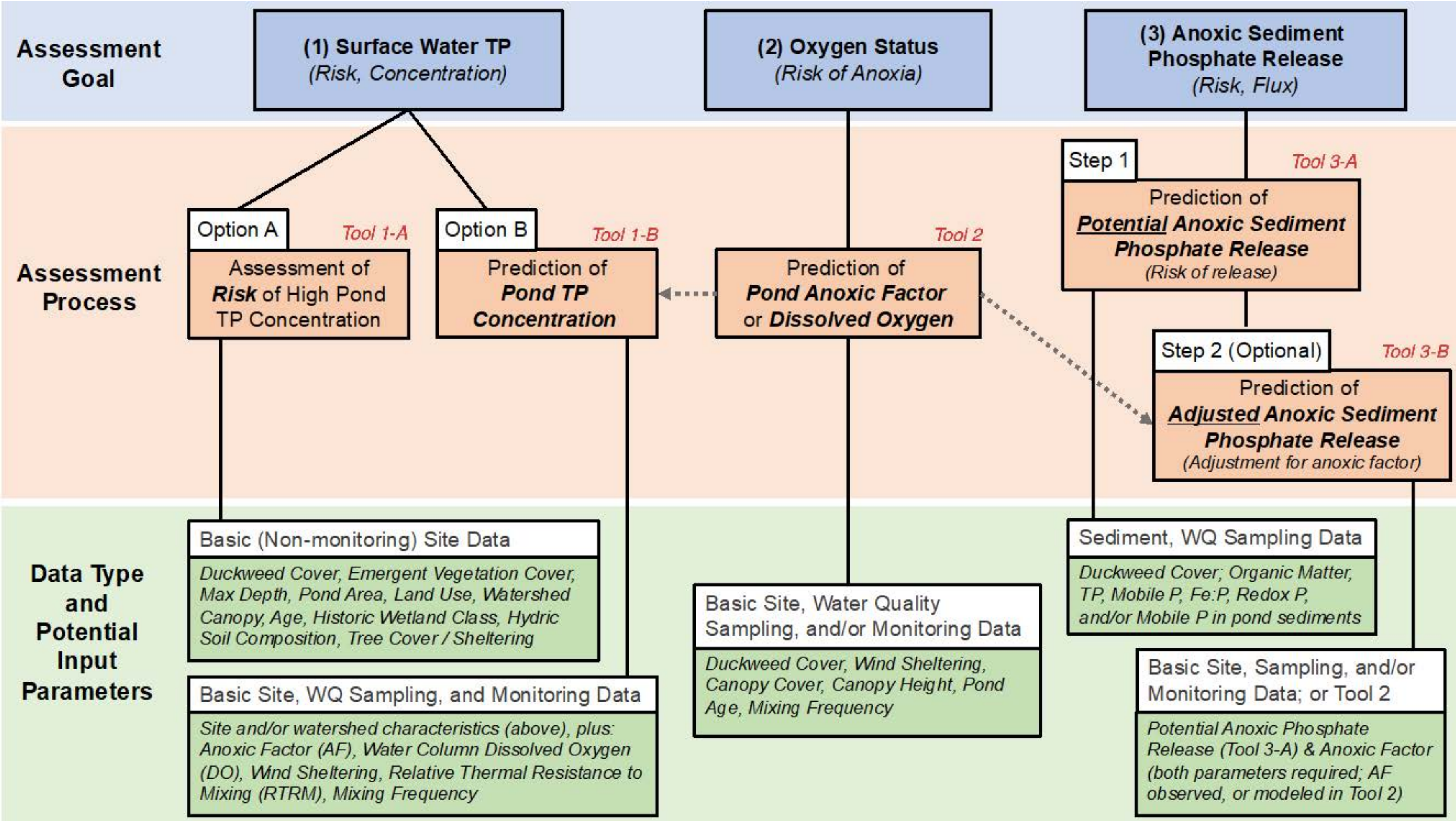

Figure ES.1 Flowchart of the Pond Assessment Tool showing the assessment goals, assessment process for each goal, and input data parameters for the assessment process.

Chapter 1: Introduction

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. Stormwater ponds and wetlands treating stormwater store runoff and settle solids along with associated pollutants to the bottom of the pond. However, there is increasing evidence that many of these ponds may no longer be providing the water-quality benefits of the original design (Taguchi et al. 2018, 2020). Phosphorus export from stormwater ponds and wetlands treating stormwater is affected by internal processes related to oxygen and mixing dynamics, as well as sediment chemistry and hydrology. Some ponds can release phosphorus trapped in the bottom sediments back into the water column, primarily under low dissolved oxygen conditions, contributing to internal load of phosphorus. The process of internal loading can be addressed by remediation techniques, such as adding aerators or alum treatments. Yet, little is known about the potential for phosphorus re-release from the wide range of pond practices being used for stormwater management, including both open water ponds and ponds with emergent vegetation, nor about the potential factors (related to e.g., climate, watershed, pond characteristics) that regulate phosphorus release or burial (Pant 2020).

Reduction in phosphorus is critical because phosphate, a dissolved form of phosphorus, sustains algal and cyanobacteria growth and causes a wide range of water quality impairments in stormwater ponds and downstream receiving waters, including algal blooms, excessive free-floating plants, taste, and odor problems. Therefore, it is critical to develop effective approaches to both assess and maintain ponds, especially for older ponds, and develop methods to improve their functionality.

1.2 Objectives

The primary objective of this project was to define, identify, and evaluate risk factors to water quality performance (primarily phosphorus retention) of the diverse stormwater ponds and wetlands treating stormwater. The goal of developing this understanding is to help develop and prioritize maintenance and management activities to provide optimal performance for phosphorus retention in stormwater ponds. Better management will guide allocation and more efficient use of limited resources to address environmental impacts of stormwater. Project sub-objectives are to:

(i) Establish robust relationships between pond features and functionality through broad field assessment, directed laboratory studies and data synthesis;

(ii) Develop a spreadsheet tool to evaluate site-specific indicators of functionality (or failure) for performance for phosphorus treatment; and

(iii) Recommend maintenance protocols for stormwater ponds and suggest management options; this includes an update to the 2009 LRRB Research Implementation Committee (RIC) report “Stormwater Maintenance Best Management Practices [BMP] Resource Guide” for stormwater ponds with regard to phosphorus release from sediments and managing Polycyclic Aromatic Hydrocarbons (PAH) concerns in pond dredging.

This project will expand our limited understanding of effectiveness of stormwater ponds and wetlands treating stormwater for phosphorus removal, identify factors that determine effectiveness, and inform stormwater managers who need to evaluate and maintain these pond practices.

1.3 Prior Research on Phosphorus in Stormwater Ponds and Wetlands Treating Stormwater

Stormwater ponds and wetlands treating stormwater are primarily designed to detain and treat urban stormwater runoff. Phosphorus is one of the critical pollutants in runoff because phosphorus is the limiting nutrient for primary production in temperate freshwaters (Schindler 1977). These ponds typically act as sinks for phosphorus washed off from fertilizers, degrading organic matter, animal wastes, and other non-point sources in the surrounding watershed area.

Our research team was made aware of unusually high phosphorus concentration in some ponds in 2014, when representatives with the Riley Purgatory Bluff Creek Watershed District (RPBCWD) came to the University of Minnesota to ask for our perspective on the phosphorus concentrations that they had measured in 98 stormwater ponds and wetlands treating stormwater, sampled in five cities between 2010 and 2013. The results showed total phosphorus (TP) concentrations ranging between <0.010 mg/L to 8.1 mg/L during summer months (mean = 0.41 mg/L; median = 0.19 mg/L, 95% confidence interval = 0.47 mg/L; n = 748 water samples; Forster et al. 2012; RPBCWD 2014). On average, the TP levels were higher than levels expected in pond outflows and were higher than typical inflowing concentrations (Taguchi et al. 2018, 2020). The MPCA's TP water quality standard for lakes in the North Central Hardwood Forest ecoregion is set at 0.040 to 0.060 mg/L to prevent eutrophic conditions.

An opportunity to investigate the high phosphorus concentrations was presented in 2016, in the form of funding from the Clean Water Fund through the Minnesota Pollution Control Agency (MPCA). We discovered that many ponds were stratified at 0.31 m (1 ft) of depth or less, and were not mixing down to the sediments during much of the spring and summer period (Olsen 2017; Taguchi et al. 2018), which then had little exposure to the oxygen in the air. Stormwater ponds are designed to mix, so the lack of mixing even following rain events was surprising, and is likely a major factor in the observed low DO conditions with phosphorus release in urban ponds. The organic loading into the ponds is also relatively high, with leaves, grass clippings, and other debris coming with the stormwater and settling into the sediments at the bottom of the pond, resulting in an unusually high sediment oxygen demand as microbes in the sediments broke down the organic material. The combination of no mixing and high sediment oxygen demand resulted in an anaerobic environment (dissolved oxygen < 1 mg/L) where phosphorus is released from the sediments as iron (Fe) shifts from the ferric (Fe^{+3}) to ferrous (Fe^{+2}) form. This was confirmed by the laboratory pond sediment cores from nearly 15 ponds in the Twin Cities area that demonstrated phosphorus release from the anoxic sediments at rates that can substantially increase pond phosphorus concentrations (Janke et al. 2021). With this sediment release of phosphorus, the ponds will no longer be providing the water quality benefits of the original design (Taguchi et al. 2020).

While data for this report is mostly derived from small water bodies, we note that larger, flow-through water bodies (Simplified Hydrogeomorphic Classification; Brinson 1993; Kloiber et al. 2019) appear to be problematic for lake water quality through release of dissolved phosphorus. Watersheds with open channel flow through areas of extensive urban versions of these water bodies appear to combine conditions that can mobilize phosphorus with high potential for phosphorus transport during summertime conditions. Three noteworthy case studies from the Twin Cities metro area, briefly described below, illustrate the potential for low phosphorus retention in flow-through shallow ponds with emergent vegetation.

In the first study, Janke and Finlay (2015) assessed phosphorus balances at the Villa Park Wetland (Roseville, MN) using monitoring data from 2006 to 2012. Their analyses showed very low phosphorus retention (near zero) due to release of dissolved phosphorus, especially during late summer. Although the data were from April to November, it is likely that annual results were similar since winter conditions are often anoxic in organic rich sediments in the region. Dredging to remove phosphorus rich sediment and historical drainage from one part of the water body was implemented to reduce phosphorus release. However, comparison of data before (2006-2012) and after (2016-2019) dredging showed only small improvements in phosphorus retention by the water body (CRWD 2019, 2020).

In a second study of shallow, flow-through ponds with emergent vegetation, research on the Twin Lakes - Shingle Creek watershed[1] shows that large loads to Twin Lake derived from shallow ponds (e.g. see this article[2] in a local newspaper). Efforts are ongoing to develop controls of phosphorus release via shallow pond treatment and outlet modification (Wenck 2014). A third study of Six Mile Creek watershed (Six Mile Creek Diagnostic Study; Wenck 2013) similarly showed that large shallow ponds with emergent vegetation along the creek were a major source of phosphorus to some lakes along the Creek.

While flow-through shallow ponds appear to have low phosphorus retention and may even be net exporters of phosphorus, we focus the current analysis on more hydrologically isolated ponds due to their prevalence across the state and the Twin Cities metro in particular, and ubiquitous use in stormwater management. We note also that the use of the term “isolated” here does not refer to closed basins or preclude connection to the stormwater drainage system; the term refers to the lack of direct (in-line) connection to a stream or other open channel. In reality, many of these ponds have inlets and outlets that are connected to storm drains.

[1] http://www.shinglecreek.org/tmdls.html
[2] https://www.hometownsource.com/sun_post/community/brooklyncenter/for-twin-lake-going-with-the-flow-spells-trouble/article_c7e071a6-d936-11e7-b858-d704f3a138b4.html

1.4 Outline of Project Approach

This project expanded the previous investigation to include additional stormwater ponds and wetlands treating stormwater, with a range of attributes identified (in part) from the previous project that investigated 15 ponds (Janke et al. 2021) as risk factors for phosphorus release (for example, pond age, design, tree cover, and watershed factors). In particular, new sites included wetlands that were historically altered for stormwater management (i.e., wetlands treating stormwater). We complemented this project with information from the Local Road Research Board (LRRB)-funded project “Wet Pond Maintenance for Phosphorus Retention” in which we investigated how stormwater ponds and wetlands treating stormwater should be maintained to limit phosphorus release to receiving water bodies (Taguchi et al. 2022).

Major project activities included:

1) With the aid of the Technical Advisory Panel, a working definition of the various wet stormwater management practices (specifically, constructed vs. natural, wetland vs. pond) was developed for purposes of study in this project and for the development of outcomes (Chapter 2);

2) Field data collection in ponds during two field seasons (2020 and 2021): efforts included approximately bi-weekly surveys to monitor phosphorus water quality and in situ conditions (dissolved oxygen, temperature, specific conductivity) in six ponds chosen to represent a range of pond characteristics (2020), as well as intensive monitoring of a subset of five ponds during the second field season (2021); intensive monitoring included continuous monitoring of temperature profiles, water level, wind speed/direction over the pond to assess stratification and water mixing (Chapter 3, Appendix A);

3) Laboratory quantification of phosphorus release from pond sediments and identification of sediment phosphorus forms using sediment cores extracted from the five intensively-studied ponds (Chapter 4, Appendix B);

4) Review and synthesis of the existing pond data set to support analysis objectives; these data included water quality (phosphorus, dissolved oxygen, conductivity) and water body and/or watershed characteristics (metadata) (Chapter 5);

5) Statistical analysis: development of robust relationships between phosphorus release potential and important risk factors (pond age, pond depth, outlet structure, tree cover, watershed type, sediment characteristics, environmental and climatic conditions) to predict pond functionality (Chapter 5);

6) Pond Assessment Tool development, from the field data collection and statistical analysis, to assess these risk factors for constructed stormwater ponds and wetlands treating stormwater and to identify indicators of functionality (or failure) (Chapter 6, Appendix C);

7) Development of recommendations for stormwater pond maintenance and wetland management, based on project results, data synthesis, and a literature review of PAHs in pond sediments, construction and maintenance costs, and runoff storage/settling alternatives (Appendix D).

8) A review and update to the sections on stormwater ponds incorporating maintenance activities to reduce sediment release of phosphorus were provided for two existing documents: 2009 Stormwater Maintenance BMP Guide, and 2011 Decision Tree for Stormwater BMPs (Appendix E).

1.5 Benefits

The research conducted in this project included data collection to evaluate the prevalence of the release of phosphate from the sediments of constructed stormwater ponds and wetlands treating stormwater, and a tool to evaluate their phosphorus treatment benefits (or detriments). The project will guide future efforts to develop methods to manage or maintain ponds to provide optimal performance for phosphorus removal, helping cities, watershed managers, and state agencies to prioritize, allocate, and more efficiently use limited resources for environmental management.

Primary benefits to the state of Minnesota include:

Environmental: Cost-effective management of constructed stormwater ponds and wetlands treating stormwater for optimal treatment performance will improve runoff quality and allow the allocation of resources for other environmental projects.

Operations and Maintenance Savings: Understanding factors affecting performance, and developing tools to evaluate proper functionality and guide maintenance will help prioritize constructed stormwater ponds and wetlands treating stormwater that yield the most benefit to water quality.

Reduce Road User Cost: Measures to minimize pollutant load from constructed stormwater ponds and wetlands treating stormwater will reduce the cost for treating downstream lakes that receive polluted runoff. This will allow municipalities to reduce taxes to road users.

Chapter 2: Wetland Regulations: When and How They are Applied to Ponds

A diversity of small water bodies are used in wet stormwater management practices to treat runoff, ranging from constructed stormwater ponds to wetlands. There is considerable variability among organizations and watershed managers in terms of classification and functional definitions of these water bodies. The definition of constructed stormwater ponds versus wetlands is an important issue because of state and federal wetland regulations that could influence stormwater maintenance practices. These definitions and associated regulatory implications need to be clarified for the project report. The objective is to define the existing wet stormwater management practices, within the context of landscape features and existing judgment, to guide the field study in this project, synthesis of existing data, and development of key deliverables. For this purpose, the Technical Advisory Panel (TAP) held a meeting with representatives of the Board of Soil and Water Resources (BWSR), Minnesota Pollution Control Agency (MPCA), and Department of Natural Resources (DNR) to advise on definitions and related regulations for constructed stormwater ponds versus wetlands. We summarize the result of this meeting in section, including the development of a definition framework for this project.

2.1 State of Minnesota Statutes on Wetlands and Public Waters

The State of Minnesota has given BWSR jurisdiction over wetlands and the MDNR jurisdiction over public waters. In addition, the MPCA regulates pollutant discharges to “waters of the state” as defined in MN Statue 115. The following is taken from the 2020 publication of Minnesota Statutes 103G.005:

“Wetland type" means a wetland type classified according to Wetlands of the United States, United States Fish and Wildlife Service Circular 39 (1971 edition), as summarized in this subdivision.

> *(1) "Type 1 wetlands" are seasonally flooded basins or flats in which soil is covered with water or is waterlogged during variable seasonal periods but usually is well-drained during much of the growing season. Type 1 wetlands are located in depressions and in overflow bottomlands along watercourses, and in which vegetation varies greatly according to season and duration of flooding and includes bottomland hardwoods as well as herbaceous growths.*
>
> *(2) "Type 2 wetlands" are inland fresh meadows in which soil is usually without standing water during most of the growing season but is waterlogged within at least a few inches of surface. Vegetation includes grasses, sedges, rushes, and various broad-leafed plants. Meadows may fill shallow basins, sloughs, or farmland sags, or these meadows may border shallow marshes on the landward side.*
>
> *(3) "Type 3 wetlands" are inland shallow fresh marshes in which soil is usually waterlogged early during a growing season and often covered with as much as six inches or more of water. Vegetation includes grasses, bulrushes, spikerushes, and various other marsh plants such as cattails, arrowheads, pickerelweed, and smartweeds. These marshes may nearly fill shallow lake basins or sloughs, or may border deep marshes on the landward side and are also common as seep areas on irrigated lands.*

(4) "Type 4 wetlands" are inland deep fresh marshes in which soil is usually covered with six inches to three feet or more of water during the growing season. Vegetation includes cattails, reeds, bulrushes, spikerushes, and wild rice. In open areas, pondweeds, naiads, coontail, water milfoils, waterweeds, duckweeds, waterlilies, or spatterdocks may occur. These deep marshes may completely fill shallow lake basins, potholes, limestone sinks, and sloughs, or they may border open water in such depressions.

(5) "Type 5 wetlands" are inland open fresh water, shallow ponds, and reservoirs in which water is usually less than ten feet deep and is fringed by a border of emergent vegetation similar to open areas of type 4 wetland.

(6) "Type 6 wetlands" are shrub swamps in which soil is usually waterlogged during growing season and is often covered with as much as six inches of water. Vegetation includes alders, willows, buttonbush, dogwoods, and swamp-privet. This type occurs mostly along sluggish streams and occasionally on floodplains.

(7) "Type 7 wetlands" are wooded swamps in which soil is waterlogged at least to within a few inches of the surface during growing season and is often covered with as much as one foot of water. This type occurs mostly along sluggish streams, on floodplains, on flat uplands, and in shallow basins. Trees include tamarack, arborvitae, black spruce, balsam, red maple, and black ash. Northern evergreen swamps usually have a thick ground cover of mosses. Deciduous swamps frequently support beds of duckweeds and smartweeds.

(8) "Type 8 wetlands" are bogs in which soil is usually waterlogged and supports a spongy covering of mosses. This type occurs mostly in shallow basins, on flat uplands, and along sluggish streams. Vegetation is woody or herbaceous or both. Typical plants are heath shrubs, sphagnum moss, and sedges. In the north, leatherleaf, Labrador-tea, cranberries, Carex, and cottongrass are often present. Scattered, often stunted, black spruce and tamarack may occur.

From Minnesota Statute Chapter 115.01, Subd. 22 *"Waters of the state" means all streams, lakes, ponds, marshes, watercourses, waterways, wells, springs, reservoirs, aquifers, irrigation systems, drainage systems and all other bodies or accumulations of water, surface or underground, natural or artificial, public or private, which are contained within, flow through, or border upon the state or any portion thereof.*

The definition of public waters is provided in Minnesota Statutes Section 103G.005 Subd. 15:

"(1) water basins assigned a shoreland management classification by the commissioner under sections 103F.201 to103F.221;

(2) waters of the state that have been finally determined to be public waters or navigable waters by a court of competent jurisdiction;

(3) meandered lakes, excluding lakes that have been legally drained;

(4) water basins previously designated by the commissioner for management for a specific purpose such as trout lakes and game lakes pursuant to applicable laws;

(5) water basins designated as scientific and natural areas under section 84.033;

(6) water basins located within and totally surrounded by publicly owned lands;

(7) water basins where the state of Minnesota or the federal government holds title to any of the beds or shores, unless the owner declares that the water is not necessary for the purposes of the public ownership;

(8) water basins where there is a publicly owned and controlled access that is intended to provide for public access to the water basin;

(9) natural and altered watercourses with a total drainage area greater than two square miles;

(10) natural and altered watercourses designated by the commissioner as trout streams; and

(11) public waters wetlands, unless the statute expressly states otherwise.

(12) Public waters are not determined exclusively by the proprietorship of the underlying, overlying, or surrounding land or by whether it is a body or stream of water that was navigable in fact or susceptible of being used as a highway for commerce at the time this state was admitted to the union."

Subd. 15a. Public waters wetlands.

"Public waters wetlands" means all types 3, 4, and 5 wetlands, as defined in United States Fish and Wildlife Service Circular No. 39 (1971 edition), not included within the definition of public waters, that are ten or more acres in size in unincorporated areas or 2-1/2 or more acres in incorporated areas (Circular 39 is referenced as Shaw and Fredine 1971).

2.2 Implementation of Wetlands Regulations

Wetlands are lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water. By definition, wetlands must:

1. have a predominance of hydric soils;
2. be inundated or saturated by surface water or groundwater at a frequency and duration sufficient to support a prevalence of hydrophytic vegetation typically adapted for life in saturated soil conditions; and
3. under normal circumstances, support a prevalence of hydrophytic vegetation.

Basins that do not support emergent vegetation with maximum water depths exceeding 8.2 feet (2.5 m) under normal conditions are considered deep water habitats and not wetlands. Wet detention basins used for stormwater treatment are either wetlands, deep water habitats, or sites that combine both. Table 2.1 summarizes regulatory aspects of wetlands and deep-water habitats in Minnesota.

Table 2.1 Summary of regulatory aspects of wetlands and deep-water habitats in Minnesota.

Aquatic Resource	Regulatory Program	Regulatory Triggers	Rule/Statutory Basis
Wetlands (except those in Public Waters)	Wetland Conservation Act	Fill, drainage, excavation	MN Rules 8420
Public Waters (includes specific wetlands, deep water habitats, ephemeral streams, and open channels)	Public Waters Work Permit Program	Flow and morphology change in course, current, or cross section	MN Rules 6115
Waters of the U.S. (includes all aquatic resources)	CWA Section 404	Discharge of dredged or fill material	Section 404 Federal Clean Water Act
Waters of the State (includes all aquatic resources)	CWA Section 401 and State Water Quality Standards	CWA 404 Permit	Section 401 Federal Clean Water Act and MN Statute 115, Rules 7050 & 7052

The primary wetland regulatory programs in Minnesota are the DNR's Public Waters Work Permit Program, the Wetland Conservation Act (administered by BWSR), Section 404 of the Clean Water Act (administered by the U.S. Army Corps of Engineers), and Section 401 of the Clean Water Act (administered by MPCA). Note that this list does not include MS4, Minnesota General Stormwater Permit for construction activity (CSW permit) and industrial stormwater permits or other potential NPDES/SDS permits, all administered by the MPCA. The following is a summary of each program as it relates to stormwater management ponds:

1. Public Waters Work Permit Program (PWWPP) – A permitting program that regulates a defined set of waters and wetlands (includes wetlands, streams, deep water habitats, lakes, and ditches) referred to collectively as "Public Waters". Public waters are all water basins and watercourses that meet the criteria set forth in Minnesota Statutes, Section 103G.005, subd. 15 that are identified on Public Waters Inventory (PWI) maps and lists. PWI maps and PWI lists are available by county on the DNR's website. Any stormwater management pond that is part of a designated public water is subject to PWWPP regulatory requirements.

2. Wetland Conservation Act (WCA) – A statewide regulatory program for all wetlands that are not public waters per above. Most wet detention ponds meet the definition of a wetland and are subject to WCA regulations. However, WCA specifically excludes actions in stormwater wetlands from wetland regulatory provisions if the stormwater wetland was created in an upland location.

Uplands are defined as areas with non-hydric soils. WCA includes specific provisions related to maintenance of wetlands used for stormwater management.

3. Clean Waters Act (CWA) 404 – A federal program administered by the U.S. Army Corps of Engineers (Corps) that regulates discharges into aquatic resources that meet the definition of Waters of the U.S. Stormwater ponds that were created in uplands or otherwise are not Waters of the U.S. are not subject to program regulations.

4. CWA 401 – A federal regulatory provision administered by the MPCA via state rules for water quality standards. Projects triggering the need for a CWA 404 permit must comply with state water quality standards as determined by the MPCA via the application of Minnesota Rules 7050 and 7052.

5. Minnesota Rules 7050.0186 – Wetland Standards and Mitigation is regulated by the MPCA through the 401 program mentioned above and by the CSW Permit. The CSW permit accepts determinations made by other jurisdictional authorities (U.S. Army Corps, DNR and WCA) that show the wetland mitigation sequence has been followed. Where there are impacts from a construction activity not addressed in one of the permits or other determinations (e.g. permanent inundation, significant degradation of water quality, excavation, filling, draining) CSW permittees must go through a mitigation sequence. Common examples include excavation in a type 1,2,6,7, and 8 wetland.

2.3 Summary of Technical Advisory Panel Meeting

The following guidelines emerged from the 9 Feb, 2021, meeting of the Technical Advisory Panel (TAP):

2.1.1 Definitions

Constructed Stormwater Pond – a pond constructed in upland to treat stormwater, which must be documented as built in upland. Municipalities should be including these upland ponds in their stormwater utility documentation during the design process. These ponds are not subject to WCA or CWA 404 regulations. PWWPP regulations apply to any of these that are identified as public waters on the DNR's Public Waters Inventory. Such sites are considered to be a treatment device and must follow design requirements in MPCA's general construction stormwater permit when projects create one or more acres of new impervious surface. These constructed stormwater ponds and other treatment practices are also regulated under the MPCA Municipal Separate Storm Sewer System (MS4) program where applicable. Characteristics of the underlying mapped soils may indicate if the pond was constructed in an upland location. Otherwise, landscape position, historic aerial photography, construction plans, and other sources of information must be used by the appropriate individuals to make this determination.

Upland Constructed Wetland – a water body constructed upland with predominantly wetland-like characteristics (e.g. promotion of macrophyte growth; greater residence time, flow-path length, and/or higher surface area relative to depth than ponds) for the purpose of treating stormwater inflows and/or wastewater. This definition is included here because a constructed wetland is considered to be distinct from a stormwater pond in the treatment field. The two are similar in terms of regulations in

Minnesota. An upland constructed wetland must follow design requirements in MPCA's general construction stormwater permit when projects create one or more acres of new impervious surface. These constructed stormwater wetlands and other treatment practices are also regulated under the MPCA MS4 program where applicable (i.e. MS4 tier 1 cities). The soils may be hydric, but the original soils were not hydric. These sites are maintained and operated as a stormwater treatment facility.

Wetlands Treating Stormwater – a historical wetland area that has been modified or is managed to produce a clean water service for a downstream water body, and still meets the regulatory definition of a wetland. These wetlands are subject to WCA regulations and potentially CWA 404 and 401 requirements if they are Waters of the U.S. They are subject to PWWPP regulations if they are or are within designated public waters. Any pond classified as a wetland (types 1 through 8) where construction or maintenance activities disturb one or more acres of land need to obtain the MPCA general stormwater permit for construction activity and may need to follow the mitigative sequence found in Minnesota Rule 7050.0186.

Deepwater Habitats used for Stormwater Treatment – historically inundated areas (wetlands, lakes, ponds) that have been modified or is managed to produce a clean water service for a downstream water body, but are too deep to be considered wetlands. These areas are not subject to WCA regulations, but are potentially subject to CWA 404 and 401 regulations if they are Waters of the U.S. They are subject to PWWPP regulations if they are or are within designated public waters.

2.1.2 Maintenance of Stormwater Ponds and Management of Wetlands Treating Stormwater

Constructed Stormwater Ponds – Maintenance activities in these ponds are not subject to the aforementioned regulatory programs unless the pond is a designated public water.

Upland Constructed Wetland – Maintenance activities in constructed wetlands are not subject to the aforementioned regulatory programs unless the constructed wetland is a designated public water.

Wetlands Treating Stormwater – Typical management activities in these wetlands such as sediment removal and culvert repairs are generally allowed without the need for a permit from the regulatory programs provided that there is no filling, drainage, or excavation aside from sediment removal. Management projects on stormwater wetlands that are public waters should coordinate with the DNR prior to the beginning of the project, to confirm they do not need a permit. Maintenance projects on stormwater wetlands that are not public waters may proceed without WCA approval; however, it is advisable that municipalities coordinate with the Local Government Unit (LGU) responsible for implementing the WCA to confirm compliance. Such coordination can happen on a project by project basis or at an annual meeting to discuss the year's maintenance projects.

Deepwater Habitats used for Stormwater Treatment – Typical maintenance activities in these areas do not need a permit from the regulatory programs provided the area is not a public water and there is no filling proposed. Maintenance projects on public waters should coordinate with the DNR prior to maintenance activities to confirm they do not need a permit.

2.1.3 Construction/Alteration Activities on Stormwater Ponds and Wetlands Treating Stormwater

Constructed Stormwater Ponds – Construction/alteration activities involving filling, draining, excavation, or other modification in these ponds is not subject to the aforementioned regulatory programs unless the pond is a designated public water. Construction/alteration activities in stormwater ponds in upland areas is not subject to wetland or public water regulations unless it indirectly drains another wetland or public water.

Upland Constructed Wetland – Construction/alteration activities such as filling, draining, and excavation, in constructed wetlands is not subject to the aforementioned regulatory programs unless the wetland is a designated public water. Construction/alteration activities in stormwater wetlands in upland areas is not subject to wetland/water regulations unless it indirectly drains another wetland or public water.

Wetlands Treating Stormwater – Construction activities involving filling, draining, and/or excavation (aside from sediment removal) in these wetlands is subject to WCA regulations. A pre-application meeting with the local WCA Technical Evaluation Panel (TEP) charged with advising the WCA local government unit is recommended to determine the type of WCA approval needed for the project. Filling within these wetlands may trigger CWA 404 and 401 jurisdiction and the need for permits from the Corps and certification from MPCA. If the CSW permit is required for a project, excavation in wetland types 1,2,6,7,8 (WCA determinations would be accepted on type 3,4 and 5) are required to go through the mitigation sequence in Minnesota Rule 7050.0186 and will be required to mitigate for lost wetland type. For example, excavating in a type one wetland to create a pond will still result in an area that retains wetland characteristics, but the specific designated use of a type one wetland is lost (plant species, habitat) and a treatment practice has been created. Such projects proposed on stormwater wetlands that are public waters should coordinate with the DNR to determine the need for a PWWPP permit. These projects likely would also require permitting under MPCA stormwater program as an MS4.

Deepwater Habitats used for Stormwater Treatment – Construction activities involving filling in these areas may trigger CWA 404 and 401 jurisdiction and the need for permits from the Corps and certification from MPCA. Such projects proposed in public waters should coordinate with the DNR to determine the need for a PWWPP permit.

(Circular 39 is referenced as Shaw and Fredine, 1971).

2.4 Issues with the Wetlands Definition

There are still some unresolved issues with regard to the definitions and regulations of wetlands. First, there is no quantitative definition of a “predominance of hydric soils.” Second, there is no statement about when the soil data is taken. Presumably, the data would be taken before construction commences, but many soils have hydric and non-hydric periods. Third, the GIS reference layers that exist in the NCRS SSURGO database do not necessarily agree with our current knowledge of soil history.

Fourth, it is stated above that wetlands must have a prevalence of hydrophytic vegetation, but prevalence is not defined quantitatively, such as a percentage cover. Finally, the “appropriate individuals” in the definition of upland stormwater ponds needs to be more accurately described.

Our ongoing work in this and other related projects suggests some challenges that may present difficulties in determining where and how wetland regulations apply to “stormwater ponds” in Minnesota. We wish to note two potentially important issues with wetland and stormwater pond classification. First, soil data in the most widely used database (USDS-NRCS) are often missing in older urban areas, or sometimes inaccurate in urbanized landscapes. For such areas, historic maps may suffice but are dependent on availability and data quality. Second, many urban stormwater sites have both deep water habitat that lacks extensive vegetation cover and heavily vegetated areas, making clear distinction between the two difficult. Inconsistencies among depth thresholds separating deep water habitat from wetlands between agencies further complicate the subject. Circular 39 (Shaw and Fredine 1971) defines maximum depth as “usually less than 10 feet deep” yet other sources have maximum depths of 8.2 feet (2.5 m). Further, determining depth of some ponds may be difficult due to the presence of deep unconsolidated sediments.

2.5 Conclusions on Applications of Wetland Definitions

The application of wetland and aquatic resource regulations to basins used for stormwater management is complex. The type of basin (wetland vs deep water habitat), origin of the basin (historically wet area versus constructed in an upland location), regulatory designation (public water vs non-public water), whether the wetland was mitigated for when it was turned into a wetland treating stormwater and activity type proposed (maintenance vs construction) determine the scope and type of regulations that apply. Municipalities should know where all of the public waters are located in their jurisdiction, and document these waters as part of their stormwater management system. Additional information can be found BWSR (2021). The local TEP is a good resource for determining which regulations apply (if any) to a given pond or wetland and how they may apply.

Chapter 3: Field Data Collection in Study Ponds

This chapter summarizes the data collected from field monitoring over two field seasons (2020 and 2021). Two approaches were employed in the data collection: first, a survey of six stormwater ponds and wetlands treating stormwater was carried out during the first field season (July – November, 2020), with results used to identify a smaller set of sites (5 ponds) used in a second, more intensive monitoring approach during the next field season (May – November, 2021). With substantial monitoring data available from previous projects, a primary motivation of the field data collection in this project was to focus on ponds that were not well-represented in past data collection. This included ponds with open water and very little on-shore tree cover, as well as ponds with emergent vegetation (i.e., with wetland-like features). The second field season also presented an opportunity to assess the impact to phosphorus of an iron filings treatment applied to the sediments of the Shoreview Commons Pond during winter 2020-21.

Data collection efforts supported two primary objectives:

1. Investigate differences in water quality and risk indicators between ponds with open water and those with emergent vegetation (that are also generally shallower); and

2. Contribute to development of a set of indicators for ponds related to risk of high phosphorus levels and potential for phosphorus release (developed in Chapter 5)

3.1 Site Selection and Description

Pond sites were selected to represent a range of characteristics affecting phosphorus retention, as identified in previous work (Janke et al. 2021): tree cover/wind sheltering, duckweed, emergent vegetation, water depth, and hydrogeologic setting. Of these, emergent vegetation was a particular emphasis for this project, in addition to other criteria including (1) availability of existing water quality data and information (depth, bathymetry, water samples), and (2) access. Selected stormwater ponds/wetlands treating stormwater and relevant characteristics are given in Table 3.1; with detailed maps of the ponds shown in Figure 3.1 and Appendix A.

Six pond sites were selected for field data collection in Field Season 1 (which began on July 1, 2020). Five pond sites sampled in Field Season 1 were selected for monitoring in Field Season 2 (May - November, 2021), and were instrumented for continuous monitoring (see Methods Section 3.2). Two additional ponds (Langton and 35E/Larpenteur) that had very little shoreline vegetation cover and no duckweed were added in Field Season 2, as most pond sites in our previous data collections were heavily tree-sheltered with frequently dense duckweed cover. These two open water ponds were monitored using the approach employed in Field Season 1 (sampling and vertical profiling only; see methods below).

Three of the ponds monitored in Field Season 2 (Aquila, Duck, and Wetland-1) represent more wetland-like sites, as all three had substantial emergent vegetation and duckweed, and were located in areas with hydric soils. The other two ponds, Alameda and Shoreview Commons, were studied intensively in previous work (Taguchi et al. 2020; Janke et al. 2021); both were highly tree-sheltered ponds and tended to have high (but variable) duckweed cover in most years. Alameda also served as a control site for climate variability, as this pond had been monitored intensively since 2017. Land use setting for these five pond sites was primarily residential, with Langton and 35E/Larpenteur dominated by commercial land use and roads.

Table 3.1 Characteristics of stormwater ponds and wetlands treating stormwater studied as part Field Monitoring and Laboratory Sediment P Release activities in this project. For field monitoring tasks, 'sampling only' 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 sites were cored as part of previous work (indicated by years prior to 2021). Sites cored in 2021 were included in this study. *Ages of pond sites are the year of construction or year connected to the stormwater network, relative to 2019.

Pond	Field Season 1 (2020)	Field Season 2 (2021)	Sediment Core Study	Area, ac	Max Depth, m	Drainage Area, ac	Age*, yr	Emergent Veg %	Hydric Soils %	Canopy % (50 m Buffer)	Historic Wetland	Circular 39 Class	Dominant Land Use
Cavell Pond	Sampling Only	NA	NA	5.22	0.31	14.4	> 29	55	100	48	Yes	4	Residential
LU-P2.4B	Sampling Only	NA	NA	2.78	0.76	20.6	NA	20	100	63	Yes	4	Residential
Corpus Christi Pond	Sampling Only	NA	NA	0.12	0.31	NA	NA	NA	0	NA	No	NA	Institutional
Aquila	Sampling Only	Monitoring	2019	2.85	0.49	13.9	41	50	3	81	Yes	4	Residential
Duck Pond S	Sampling Only	Monitoring	2021	9.43	0.67	29.1	28	86	80	40	Yes	4	Residential
Wetland-1	Sampling Only	Monitoring	2021	3.05	1.0	11.8	32	73	100	68	Yes	4	Residential
Shoreview Commons	NA	Monitoring	2018	2.9	1.2	144	30	0	95	68	Yes	4	Residential
Alameda	NA	Monitoring	2017	2.89	2.1	285	70	0	0	71	Yes	4	Residential

Pond	Field Season 1 (2020)	Field Season 2 (2021)	Sediment Core Study	Area, ac	Max Depth, m	Drainage Area, ac	Age*, yr	Emergent Veg %	Hydric Soils %	Canopy % (50 m Buffer)	Historic Wetland	Circular 39 Class	Dominant Land Use
Langton	NA	Sampling Only	2020	0.16	1.7	2.06	2	0	0	< 25	No	NA	Residential / Commercial
35E / Larpenteur	NA	Sampling Only	2020	0.8	1.8	NA	4	0	0	< 10	No	NA	Residential / Highway

Figure 3.1 Locations of pond monitoring sites in Field Season 2 and aerial photographs (MN Geospatial Commons). See Appendix A for detailed sampling maps.

3.2 Field Sampling and Monitoring Methods

The purpose of field sampling and monitoring efforts in the pond 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. 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 (can impact oxygen dynamics and phosphorus levels).

3.1.1 Sampling Protocols

For the six ponds with emergent vegetation (Cavell, LU-P2.4B, Corpus Christi, Aquila, Duck, and Wetland-1; Table 3.1), field sampling methods were developed to capture the expected high spatial variability in DO and phosphorus concentrations between open water column and emergent vegetation locations. Sampling was performed in 4-7 locations in the open water area (depending on the pond size) and along 4-6 lateral transects (generally three points along a transect) in the emergent vegetation zones. Sampling locations and transects for the three ponds with emergent vegetation are shown in Appendix A. Given the variable water depth at a site and among the sampled sites, the water temperature, dissolved oxygen (DO), and specific conductivity measurements were taken at the surface (~0 cm) and approximately 12.5 cm below the surface at multiple locations in the open water areas. Only the surface measurements were taken in the emergent vegetation areas. Vertical profiles of temperature, dissolved oxygen, and conductivity were collected in the deepest location in the open water area. Surface (epilimnion) grab water samples were collected at all five locations sampled in the open water and composited in the field. Water samples were also collected at each of the three points along a transect in the emergent vegetation areas using a pole sampler with Nalgene bottle. Water was generally very shallow in the emergent vegetation portions of the pond and at times was impossible to sample without entraining high amounts of muck and debris. A hypolimnion water sample was collected at the deepest location where the vertical profile was taken.

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).

Field crews also made notes about the extent (fraction of open water) and density of duckweed (*Lemna*) and watermeal (*Wolffia*) cover during each site visit at all ponds.

3.1.2 Pond Monitoring Methods

Five ponds (Alameda, Shoreview Commons, Aquila, Duck, and Wetland-1; Table 3.1) were monitored continuously in Field Season 2 for water level, wind speed, and vertical temperature profile using the same instrumentation and approach as in the first field season. Monitoring stations were installed near the pond's deepest point and consisted of an anemometer (LaCrosse TX-23U) mounted roughly 0.91 m above the water surface, ultrasonic distance gauge for water level measurement, and a thermistor chain of 3 to 6 nodes spaced roughly 15 to 46 cm apart in the vertical direction (spacing determined by water depth). Data were logged at 10- or 15-minute intervals with an Arduino-type data logger connected to a solar panel. For the two deeper ponds (Alameda and Shoreview Commons), discrete Onset Hobo data loggers were installed for conductivity at a depth of roughly 15 cm above the bottom to measure flushing of road salt, and dissolved oxygen (DO) was logged at a depth of roughly 30 cm below the surface at the Shoreview Commons pond site. The two supplemental ponds, Langton and 35E/Larpenteur, did not include any loggers for continuous monitoring.

All seven ponds 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. For the four ponds with primarily open water (Alameda, Shoreview Commons, Langton, and 35E/Larpenteur), a single water quality profile was collected near the pond's deepest point, along with one water sample collected from near the pond bottom (hypolimnion) using a Kemmerer type sampler[3]. Surface (epilimnion) water samples were collected just below the water surface, avoiding as much duckweed as possible, from five locations and composited into a single sample in the field. Sampling locations are shown in Appendix A.

3.3 Data Collection Summary

In Field Season 1, the six pond sites were visited 1-2 times each during the period August 4 to November 4, 2020. While ten sites were called for by the proposal in this initial survey, complications and delays due to Covid-19 reduced this initial data collection to six ponds, and work began later than intended. In Field Season 2 (2021), the seven selected pond sites were routinely sampled from May 20 – Nov 15, 2021, with a total of 9 to 13 site visits per pond (Table 3.2). Two of the shallow ponds with emergent vegetation (Aquila and Duck) were too difficult to access and sample accurately during the driest parts of the summer, and thus had fewer total visits. Monitoring equipment was installed in Alameda and Shoreview Commons ponds on June 10, with later installations at the other three primary sites (Duck and Wetland-1 in early July, Aquila in mid-July). Equipment was removed from all sites between Nov 10 and Nov 16, 2021. At Duck, the monitoring station failed completely, and therefore no monitoring data (wind speed, water level, temperature profile) are available from this site. Monitoring at the other stations was complete through the duration of the field season, with the exception of the Wetland-1,

[3] https://www.wildco.com/what-are-kemmerer-bottles/

where the bottom three temperature nodes were permanently damaged by wildlife (likely muskrats) in mid-August.

Table 3.2 Summary of field data collection at the seven pond sites in Field Season 2 (May – November 2021). *For Turnover, general pattern is given (intermittent or frequent), but if the pond was stratified through most of season then date of fall turnover is given. **Temperature data at Wetland-1 incomplete after Aug 16 due to damage by wildlife.

Pond	Site Visits	Site Visits		Monitoring Station		Turnover*	Duckweed Cover
		First	Last	Start	End		
Aquila	11	5/26/21	11/10/21	7/16/21	11/10/21	intermittent	High
Duck Pond S	9	5/26/21	11/10/21	NA	NA	intermittent	High
Wetland-1	12	6/1/21	11/15/21	7/2/21	11/12/21*	11/1/21	High
Shoreview	12	5/24/21	11/5/21	6/10/21	11/16/21	intermittent	Low
Alameda	12	5/24/21	11/8/21	6/10/21	11/16/21	10/25/21	High
Langton	10	5/24/21	11/5/21	NA	NA	intermittent	None
35E/Larpenteur	11	6/1/21	11/8/21	NA	NA	frequent	None

3.4 Data Quality and Data Processing

Climate: Weather was among the hottest and driest on record for the Twin Cities[4] in Field Season 2 (2021), with June – October mean air temperature roughly 2.8 °C above average (21 °C observed vs. 18.2 °C, the 1981-2010 normal at Minneapolis-St. Paul International Airport); only 33.5 cm of rainfall was observed over the period (vs. the 1981-2010 normal of 46 cm), of which 13.2 cm fell during an 11-day stretch of August-September.[5] The substantial drought caused extremely low water levels at some of the sites (Aquila and Duck) that made access difficult, and also caused some concern for generalization of monitoring results.

Due to low water levels at the ponds with emergent vegetation, it was often difficult to collect water samples without entraining muck or vegetation at the transect locations, and some samples with extremely high TP concentrations likely reflect entrainment of debris in the samples. The water samples that were found to contain large sediment and/or duckweed mass are not included in summary values.

[4] https://www.mprnews.org/story/2021/09/22/happy-autumn-summer-was-warmest-on-record-for-twin-cities
[5] https://www.dnr.state.mn.us/climate/twin_cities/listings.html

For open water, the surface measurements (DO, temperature, conductivity) taken at the 4-7 locations were averaged to obtain the mean for that sampling event. The average for the emergent vegetation zone was calculated using data at all transects (i.e., 3 points per transect x 4 transects = 12 locations total).

Dissolved Oxygen: At the Shoreview Commons pond site, substantial bio-fouling of the dissolved oxygen logger made these data unreliable and are not presented in the report. The temperature data recorded by the logger were usable and were combined with the profile station data for that site.

Wind: With respect to wind data, we note that the anemometer used by the pond monitoring stations (LaCrosse TX-23U) was found in a concurrent project to under-observe actual wind conditions when compared side-by-side to a more robust RM Young brand anemometer. The analysis was carried out at the St. Anthony Falls Laboratory in fall of 2021, and a correction equation for the wind speed was developed (Taguchi et al. 2022):

$$U\_corr = U\_raw + 0.469*(1 - e^{-16.34*U\_raw})$$

where *U_corr* is the corrected wind speed (m/s), and *U_raw* is the raw wind speed measured by the LaCrosse anemometer (m/s). Wind speed data presented in this report have been corrected with this equation.

3.5 Field Season 1: Summary of Results

The mean surface concentrations of DO, temperature, specific conductivity, and phosphorus species in the open water and vegetated locations for each of the six sites are shown in Appendix A. The mean surface TP concentration in the open water portion was typically much lower than in the water sampled within emergent vegetation, and this observation was consistent across all six sites (Figure 3.2; note that summary and analysis excluded water samples with entrained sediment, which was more common among the samples from emergent vegetation). Sediments in the vegetated area were expected to be more anoxic because of low mixing due to wind sheltering. Duckweed and watermeal were observed at all sites during the sampling period and covered a large area of the open water (50 to 90%) at most of the sites, with their areal extent and density variable within a given site and toward the end of the season. This may have produced low estimates of phosphorus concentrations, as P in duckweed was not included in those calculations. In contrast to TP, mean TDP and SRP concentrations were similar for the open water and vegetated portions at most of the sites, and were usually much lower overall compared to TP (Appendix A).

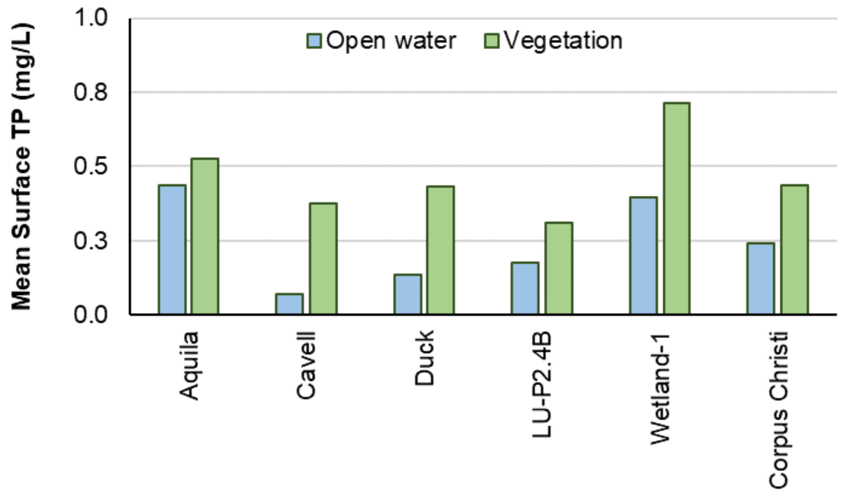

Figure 3.2 Mean surface TP concentrations in the open water locations and vegetated locations in the ponds sampled during Field Season 1 (August to November 2020).

The mean surface concentrations of DO, temperature, specific conductivity, and phosphorus species in the open water and vegetated locations for each of the six sites are plotted in Appendix A. The in situ DO conditions were spatially variable within the sites, with conditions in the open water area exhibiting less variability than the vegetated area with primarily emergent plants. The presence of free-floating plants (duckweed and watermeal) could have contributed to this variability. Specific conductivity levels were higher at two sites (~800 µS/cm; Aquila and Cavell ponds) than the other sites, suggesting differences in watershed chloride input and/or export from the ponds. The surface water temperature and specific conductivity levels were less spatially variable within the open water areas.

3.6 Field Season 2: Stratification and Dissolved Oxygen Dynamics

To illustrate contrasting stratification dynamics in the ponds, consider the temperature contours for Alameda, 35E/Larpenteur, and Aquila (Figure 3.3). These ponds varied in terms of setting and characteristics: Alameda is a relatively deep, open water pond that is heavily sheltered by trees and covered by duckweed; 35E/Larpenteur is a newer pond of similar depth with no tree sheltering, emergent vegetation, or duckweed cover; and Aquila is shallow and tree sheltered, roughly half covered by emergent vegetation, with persistent duckweed. Alameda maintained persistent temperature stratification throughout the season until fall, when it turned over in early September, while 35E/Larpenteur remained completely mixed throughout the season, likely due to the influence of wind mixing (as precipitation events were infrequent). Aquila, though much shallower, did exhibit some stratification in mid-summer (when duckweed cover was highest), but was mostly well-mixed throughout the season.

The effect of these stratification and mixing dynamics on oxygen levels in the ponds also varied across these three sites. Figure 3.4 shows extreme levels of anoxia (< 1 mg/L) in Alameda throughout the water column, persistent through the entire summer. The slight recovery of DO near the surface in August coincided with a very brief wet period that caused mixing and entrainment of oxygen by inflows; otherwise the pond was especially stagnant during the rest of the season. The 35E/Larpenteur pond, by contrast, remained highly oxygenated throughout the monitoring period, as would be expected from the well-mixed water column. The DO dynamics in Aquila are more interesting, as the period of observed temperature stratification (late June through early August) also coincided with anoxia in the pond; as water level increased with the August storms, the water column quickly reoxygenated and remained that way for most of the fall.

These patterns illustrate the impact on pond oxygen dynamics of several interacting factors, including wind access (affected by tree sheltering and perhaps duckweed), direct mixing by stormwater 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.

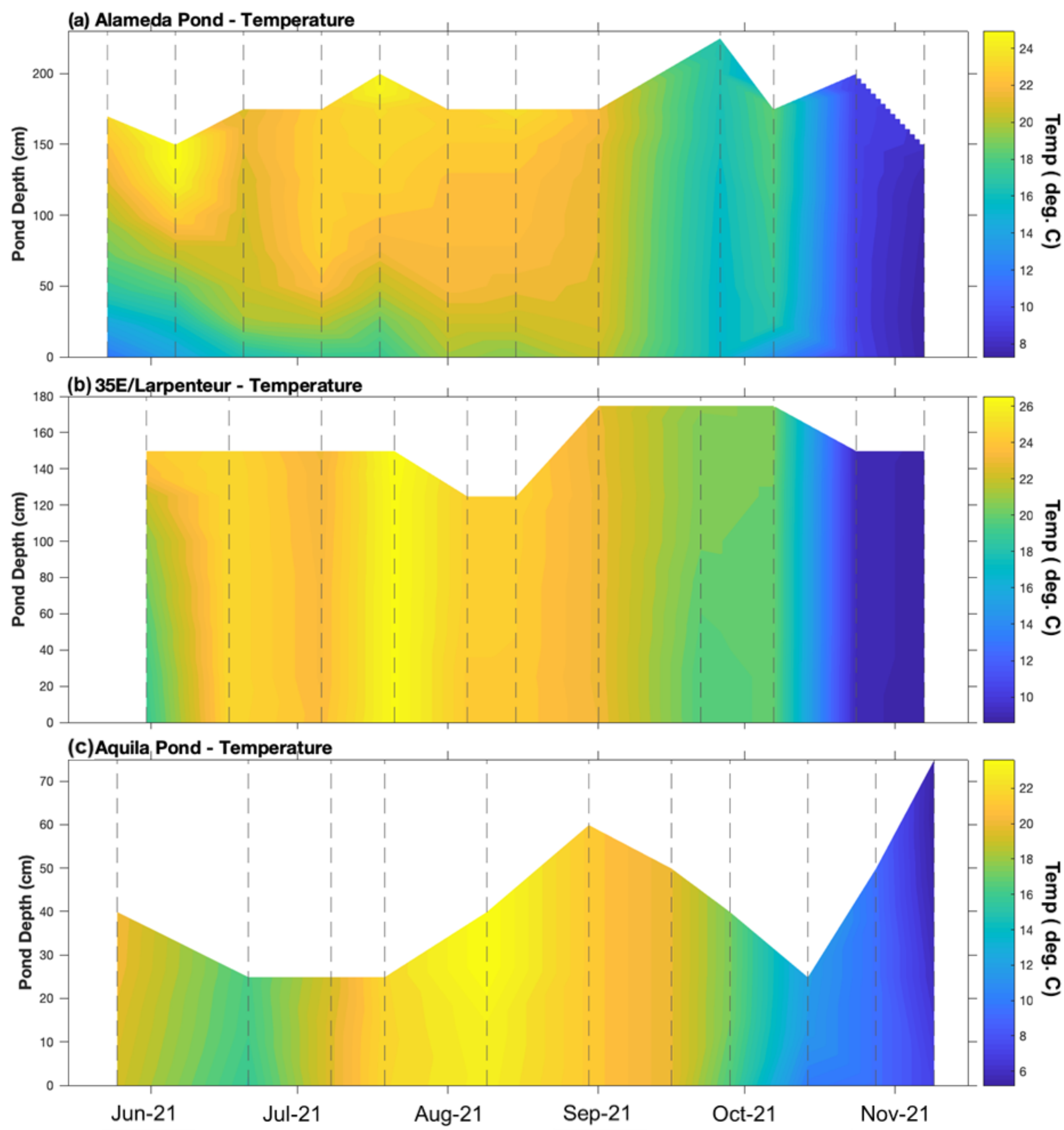

Figure 3.3 Temperature contour plots for three ponds: (a) Alameda, (b) 35E/Larpenteur, and (c) Aquila. Color indicates temperature per the scale at right (note difference in scales among plots), with water depth relative to the pond bottom on the y-axis, and time along the x-axis. Vertical dashed lines are dates of site visits when profiles were collected; linear interpolation used to fill in the gaps between profile dates.

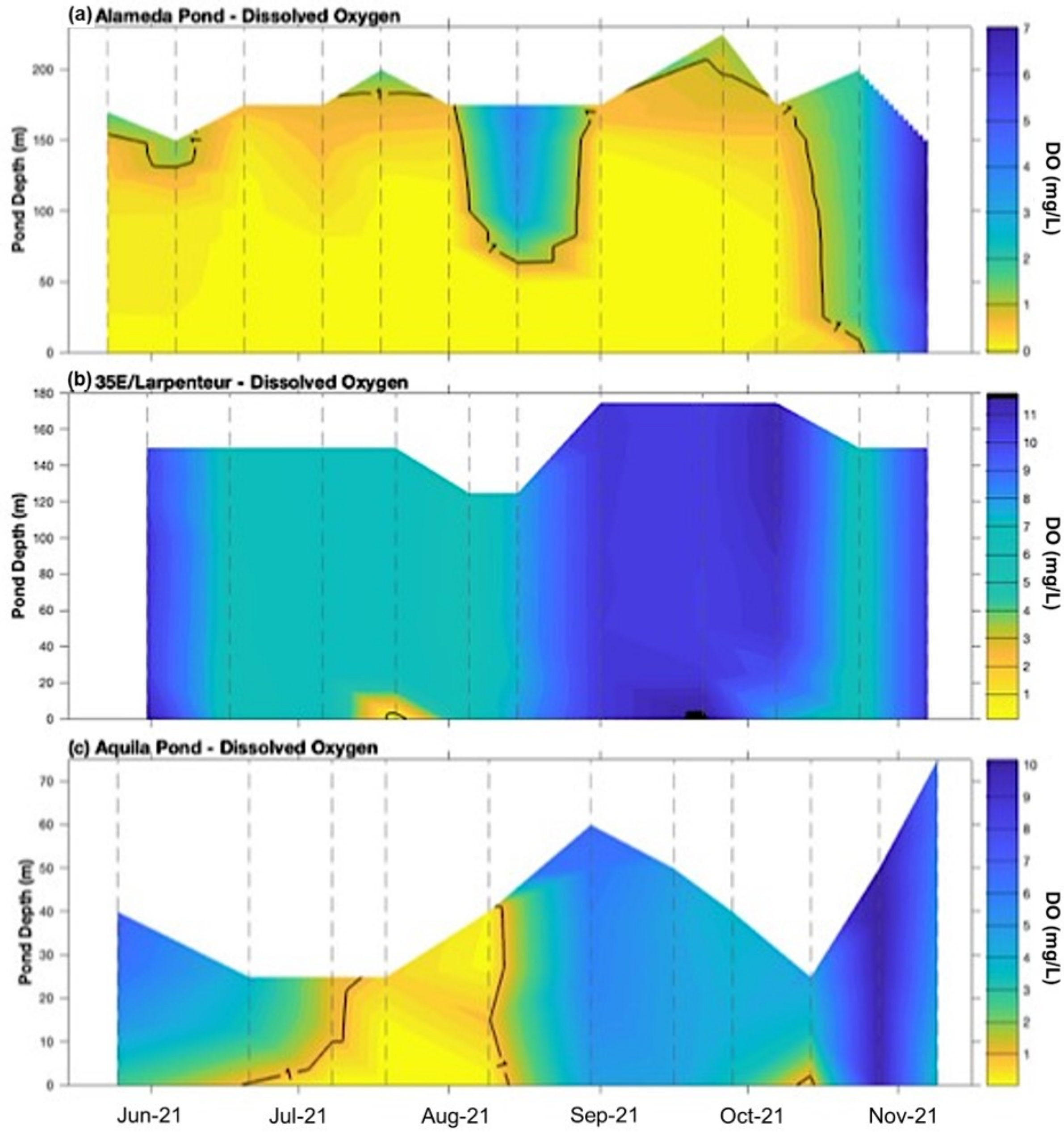

Figure 3.4 Dissolved oxygen (DO) contour plots for three ponds: (a) Alameda, (b) 35E/Larpenteur, and (c) Aquila. Color indicates DO per the scale at right (note difference in scales among plots), with water depth relative to the pond bottom on the y-axis, and time along the x-axis. Vertical dashed lines are dates of site visits when profiles were collected; linear interpolation used to fill in the gaps between profile dates. A contour for 1.0 mg/L indicates levels below which the pond is considered anoxic.

3.7 Field Season 2: Time Series of Phosphorus and Water Quality

In this section, the time series data for the ponds, including (as available) rainfall (cm), pond water depth (cm), relative thermal resistance to mixing (RTRM) for the upper water column and for the whole water column ('top-btm', where "btm" indicates pond bottom), wind speed (km/h), surface TP and SRP (mg/L), hypolimnion TP and SRP (mg/L), surface DO (mg/L), and duckweed cover (%) are presented. Note that water level, RTRM, and wind speed were collected by the pond monitoring stations, which have a shorter record than the data collected on sampling trips to the ponds (e.g., phosphorus and duckweed cover; see Methods in Section 3.2).

We present the time series for selected ponds to illustrate the effects on DO and phosphorus dynamics of pond setting (duckweed, emergent vegetation) and of the special case of non-routine pond maintenance (iron filings treatment of sediments at Shoreview Commons Pond). Data for Wetland-1, 35E/Larpenteur, Langton, and Duck are shown in Appendix A.

Time series data for Alameda (Figure 3.5) show in particular the importance of dense duckweed cover on water column oxygen dynamics, especially in such a dry year as 2021. Duckweed cover ramped up quickly in early summer and was persistent and dense until mid to late October. Water column DO was extremely low during this period, producing top-to-bottom anoxic conditions for much of the summer, with the exception of late August, a wet period when runoff inputs appear to have oxygenated the upper water column and eroded stratification strength (indicated by decreases in water column RTRM during the same period). Water column stratification was otherwise relatively strong throughout the season, with additional mixing indicated by substantial decreases in RTRM for windy periods in early- and mid-July. Surface TP dynamics show a slight response to these mixing events or mid-summer stagnation; late spring and early fall concentrations were generally higher.

In contrast to Alameda, with persistent duckweed cover and anoxia, water quality dynamics at Aquila (Figure 3.6) illustrate the impacts of variable duckweed cover on oxygen. This pond experienced dense duckweed cover in mid-summer (July and August), and during this period, this pond's entire water column was anoxic, and stratified (with relatively high RTRM) despite being only 40 – 50 cm deep at this time. Inputs of stormwater during August, shown by large increases in pond water level, also appeared to oxygenate the water column all the way to the sediments, and duckweed cover generally decreased from this point onward. Stratification for the rest of the season was weak (indicated by RTRM fluctuating around a value of 0). Surface and hypolimnion TP (and to a lesser extent SRP) also increased during the onset of this wetter period, likely reflecting watershed inputs and mixing of the hypolimnion. Phosphorus concentrations decreased over the remainder of the season, which was extremely dry. Similar to Alameda, some evidence of wind mixing is indicated by decreases in RTRM during a windy period in late July.

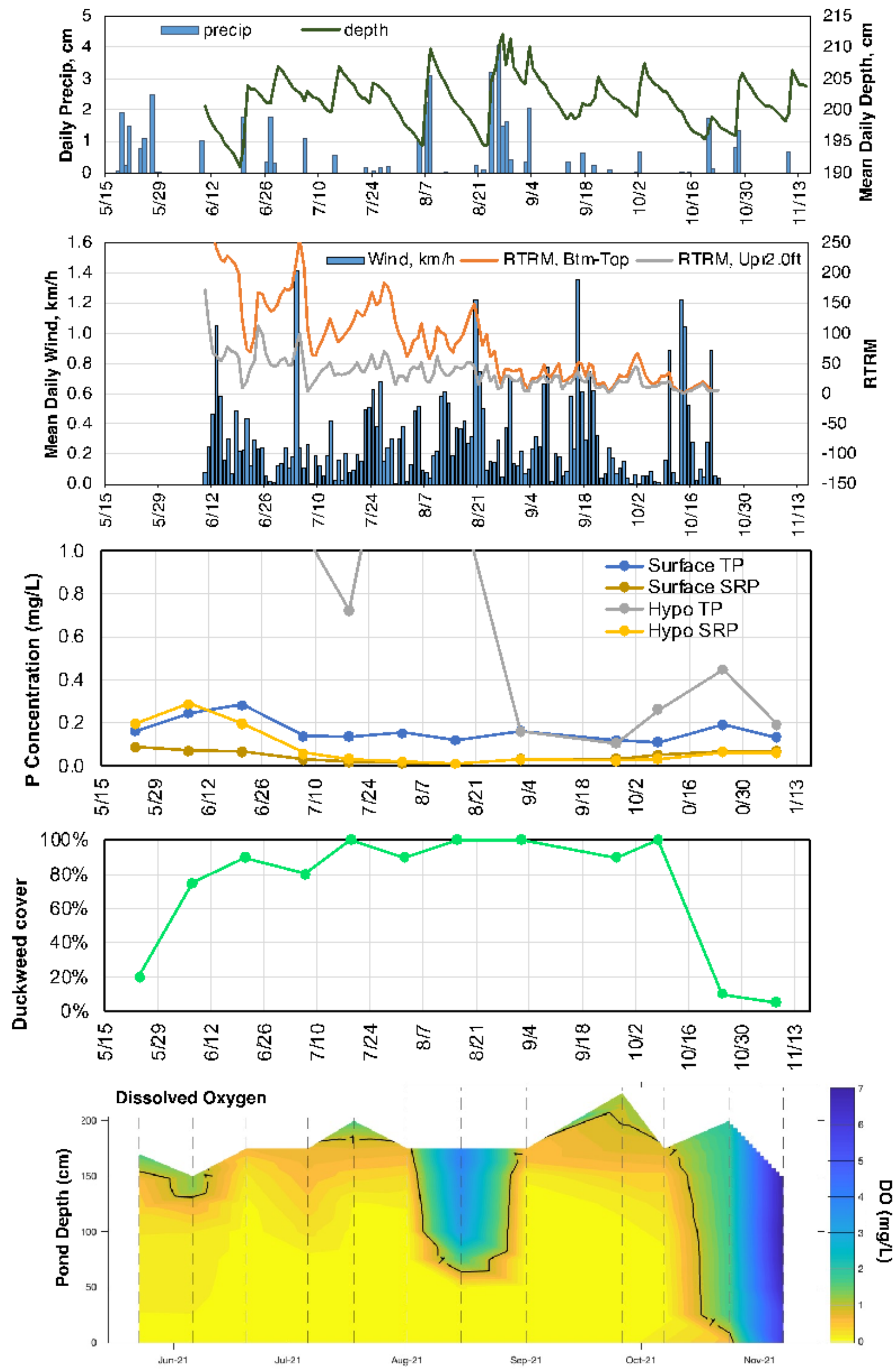

Figure 3.5 Alameda Pond time series of rain (cm; from MSP Airport), water depth (cm), wind speed (km/h), RTRM (for upper water column and for whole water column 'top-btm'), surface and hypolimnion TP and SRP (mg/L), duckweed cover (%), DO profiles (mg/L; as contours. RTRM, wind, and water level measured by monitoring stations and averaged into daily values). Other data from site visits. Hypolimnion TP > 1 mg/L are not shown so that surface TP dynamics can be seen (TP concentrations were 2.5 mg/L on 5/24, 1.84 mg/L on 6/7, 1.59 mg/L on 6/21, 1.98 mg/L on 8/2, and 1.31 mg/L on 8/16).

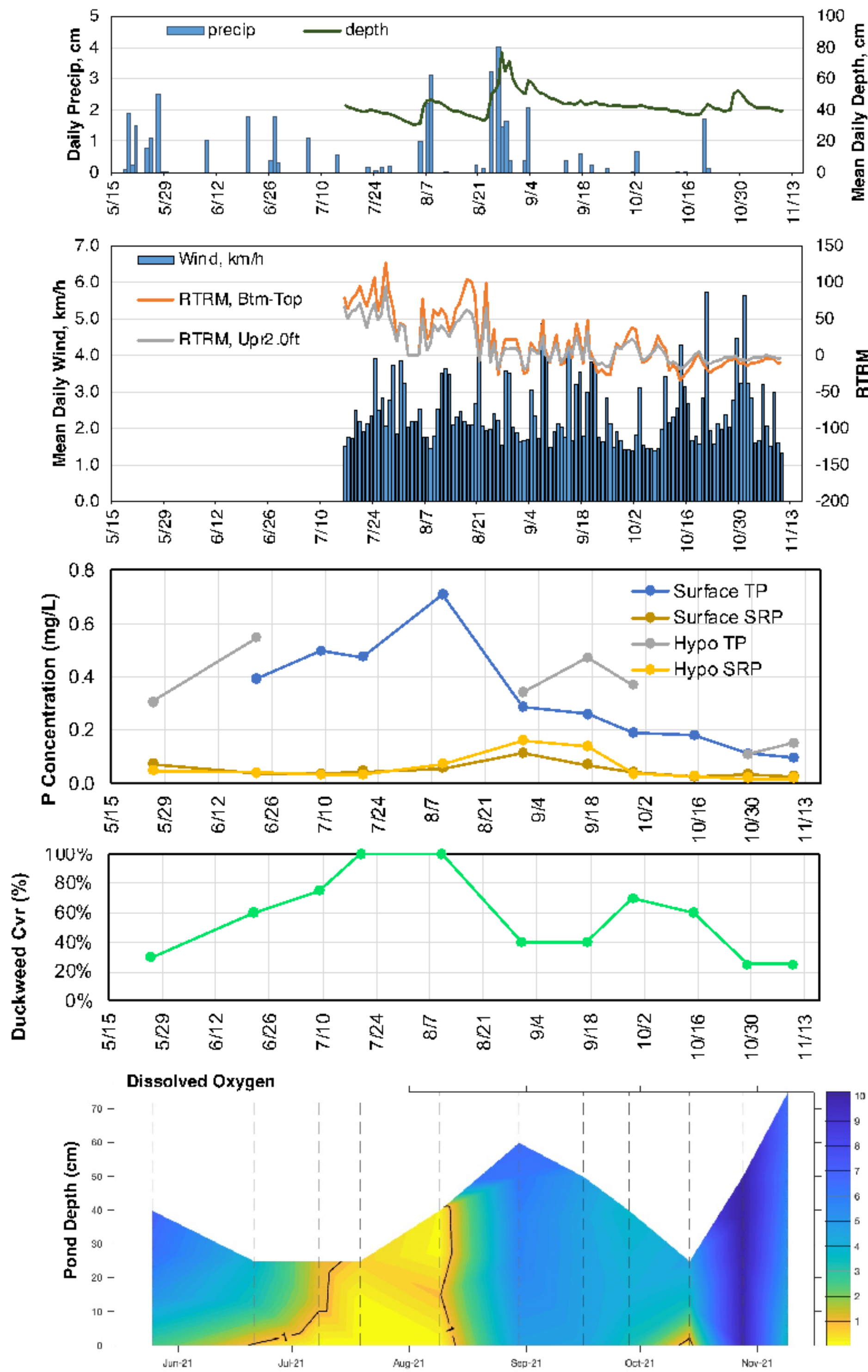

Figure 3.6 Aquila Pond time series of rain (cm; from MSP Airport), water depth (cm), wind speed (km/h), RTRM (for upper water column and for whole water column 'top-btm'), surface and hypolimnion TP and SRP (mg/L), duckweed cover (%), DO profiles (mg/L; as contours). RTRM, wind, and water level measured by monitoring stations and averaged into daily values. Other data collected on site visits.

Results for the other ponds with emergent vegetation (Wetland-1 and Duck) as well as the newer and unsheltered ponds (35E/Larpenteur and Langton) are shown in Appendix A. Wetland-1 showed patterns similar to Aquila, but with more persistent duckweed cover and resulting anoxia over the season, with a brief period of oxygenation associated with temporary lower duckweed cover in August. Duck, much like Wetland-1, was dominated by duckweed cover and water column anoxia, with slightly higher summer and early fall P concentrations relative to late spring and late fall, and a slight (but short-lived) increase in SRP during the wet period from late August to early September, illustrating the rapid uptake of bioavailable P in a productive system. The two unsheltered ponds (35E/Larpenteur and Langton) had very low P concentrations overall, which were potentially driven by a lack of duckweed cover, well-mixed water columns (likely from wind and convective mixing), and low watershed inputs in a dry year.

Lastly, the time series for Shoreview Commons Figure 3.7) is presented as a special case of a post-treatment pond, as iron filings were applied to the pond in winter 2020-2021 to bind dissolved P in sediments. In contrast to previous years (Appendix A), this pond did not maintain temperature stratification, as the fluctuating RTRM values indicate. RTRM fluctuations did not appear to respond to changes in duckweed cover, rainfall inputs, or wind events, and thus the pond's temperature dynamics were likely dominated by diurnal heat fluxes (solar radiation during the day, convective mixing at night). Lower density and less persistent duckweed cover would have provided less shading of the water column. Oxygen and duckweed showed roughly opposite trends, with low DO at peak duckweed cover in July, followed by increasing DO as duckweed senesced through fall. A DO peak observed in late August coincided with a period of high runoff inputs and a deeper water column, along with higher TP concentrations in both the hypolimnion and epilimnion; this pattern likely reflected an oxygenated water column combined with new watershed inputs of P.

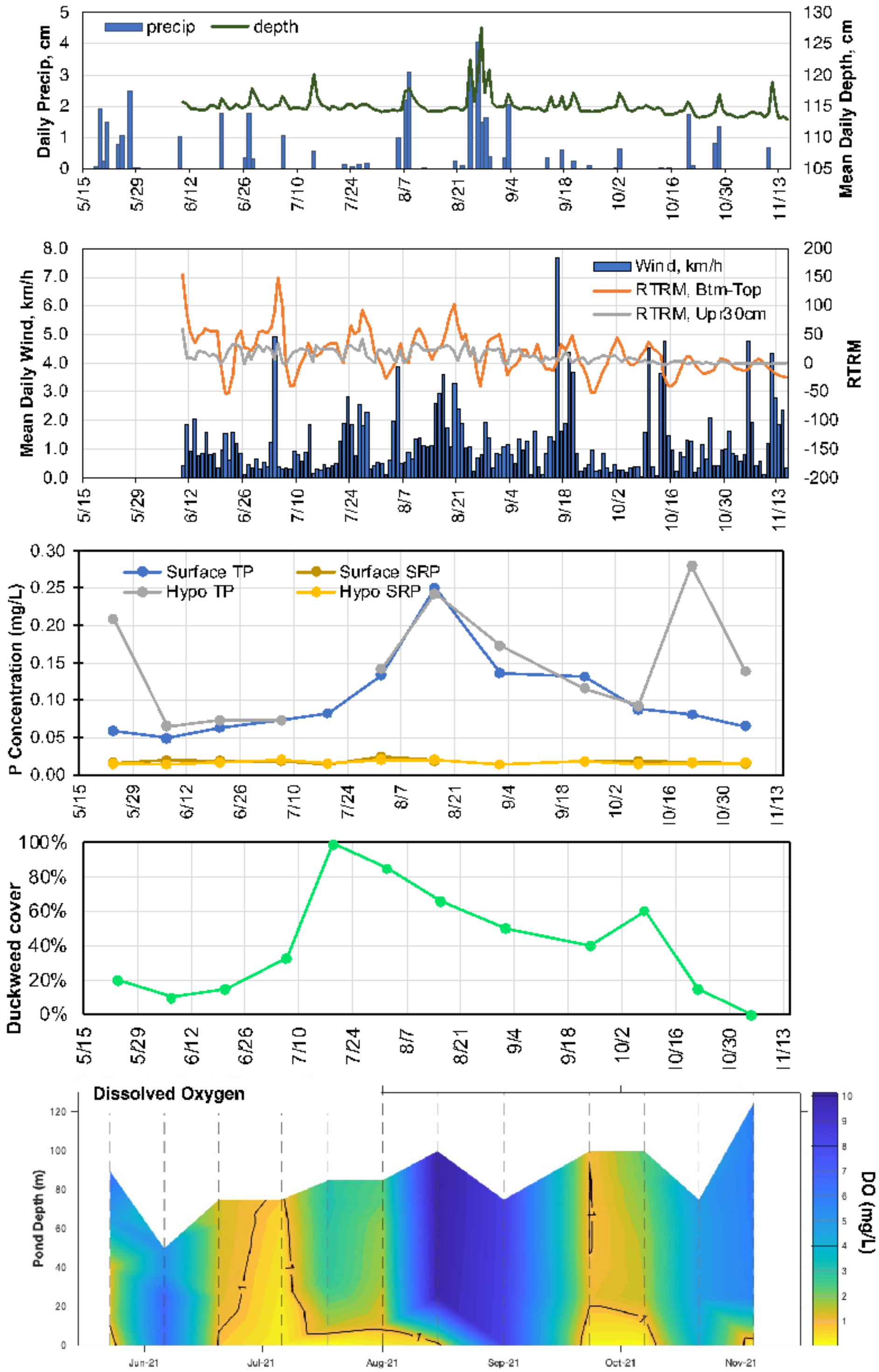

Figure 3.7 Shoreview Commons Pond time series of rain (cm; from MSP Airport), water depth (cm), wind speed (km/h), RTRM (for upper water column and for whole water column 'top-btm'), surface and hypolimnion TP and SRP (mg/L), duckweed cover (%), DO profiles (mg/L; as contours. RTRM, wind, and water level measured by the monitoring stations and averaged into daily values). Other data collected on site visits.

3.8 Field Season 2: Water Quality Patterns Across Pond Sites and Influence of Emergent Vegetation

One of the objectives of selecting sites with a range of emergent vegetation (0 to 86%) was to examine differences in water quality parameters between vegetation-dominated and open ponds, and between the open water and emergent vegetation portions of the ponds with vegetation. In Figure 3.8, we show patterns across ponds in season-mean surface DO, temperature, conductivity, and TP concentrations in the open water and in the emergent vegetation portion (3 ponds) for Field Season 2 (2021).

Dissolved oxygen (DO) was affected by duckweed cover and near-shore sheltering, as near-surface DO was much higher in the open, duckweed-free ponds (35E/Larpenteur and Langton) than in the other sites, especially those with extensive duckweed cover (Alameda, Wetland-1). These ponds dominated by persistent whole-water column anoxia (Alameda, Wetland-1) have a lower season-averaged surface DO than the ponds that were generally anoxic but mixed frequently (Aquila, Duck, Shoreview Commons). The largest variation in surface DO (error bars in DO plot; Figure 3.8) were at the shallow sites (Aquila, Duck, Wetland-1), where sediment oxygen demand might have more readily impacted surface DO.

Temperature was also affected by duckweed and sheltering; the 35E/Larpenteur and Langton ponds, which have no tree sheltering, emergent vegetation, or duckweed cover, were warmer at the surface than the other ponds (Figure 3.8). Although shallower in depth, the open water areas in Aquila and Duck ponds had the lowest surface water temperature, likely due to the combined effects of thick duckweed cover and tree sheltering. The variation in surface temperature is approximately the same at all the sites.

Surface TP concentrations in the ponds appeared to be affected by both emergent vegetation and duckweed. In the three ponds with emergent vegetation (Aquila, Duck, Wetland-1; 50-86% emergent vegetation cover), mean surface TP was about two times the mean TP in Alameda and Shoreview Commons (no emergent but dense duckweed) and six times the mean TP in 35E/Larpenteur and Langton (no emergent vegetation or duckweed; Figure 3.8). The total dissolved P (TDP) and soluble reactive P (SRP) concentrations were higher in the Aquila and Duck ponds than the other sites, however TDP and SRP observations are complicated by their being utilized in vegetation growth. Note also that the low mean phosphorus concentrations in Shoreview Commons is post-treatment. High hypolimnetic TP was observed in Alameda and Duck ponds that were persistently anoxic in summer 2021 (see Appendix A), highlighting the possibility of sediment P release due to low DO conditions.

Differences in water quality were also observed between the open water and vegetated portions of the three ponds with emergent vegetation studied in Field Season 2 (Figure 3.8). Importantly, mean TP was higher and more variable in the vegetated portions of the ponds, similar to observations in the surveys of Field Season 1 (see Appendix A). Further, surface TP in the emergent vegetation areas was consistently higher than the surface TP in the open water for every sampling event at each of the three sites (data not shown). For TDP and SRP, concentrations were once again higher in the vegetation than at the open water, although the differences were not as large as for TP, especially at Duck when compared to Aquila and Wetland-1.

An explanation for the higher TP in the vegetated areas is not obvious from the other water quality data (DO and temperature). Surface DO and temperature were actually lower in the open water than in the vegetation zone at two of the three ponds, with higher variation of DO in the open water pool (likely due to changes in duckweed density over the season). Differences in surface temperature between the two zones was prominent only at Aquila, where the emergent vegetation area was warmer by 1.6 °C than the open water pool. While broad conclusions cannot be drawn from these results, the patterns strongly suggest that emergent vegetation may play an important role in phosphorus mobilization in ponds.

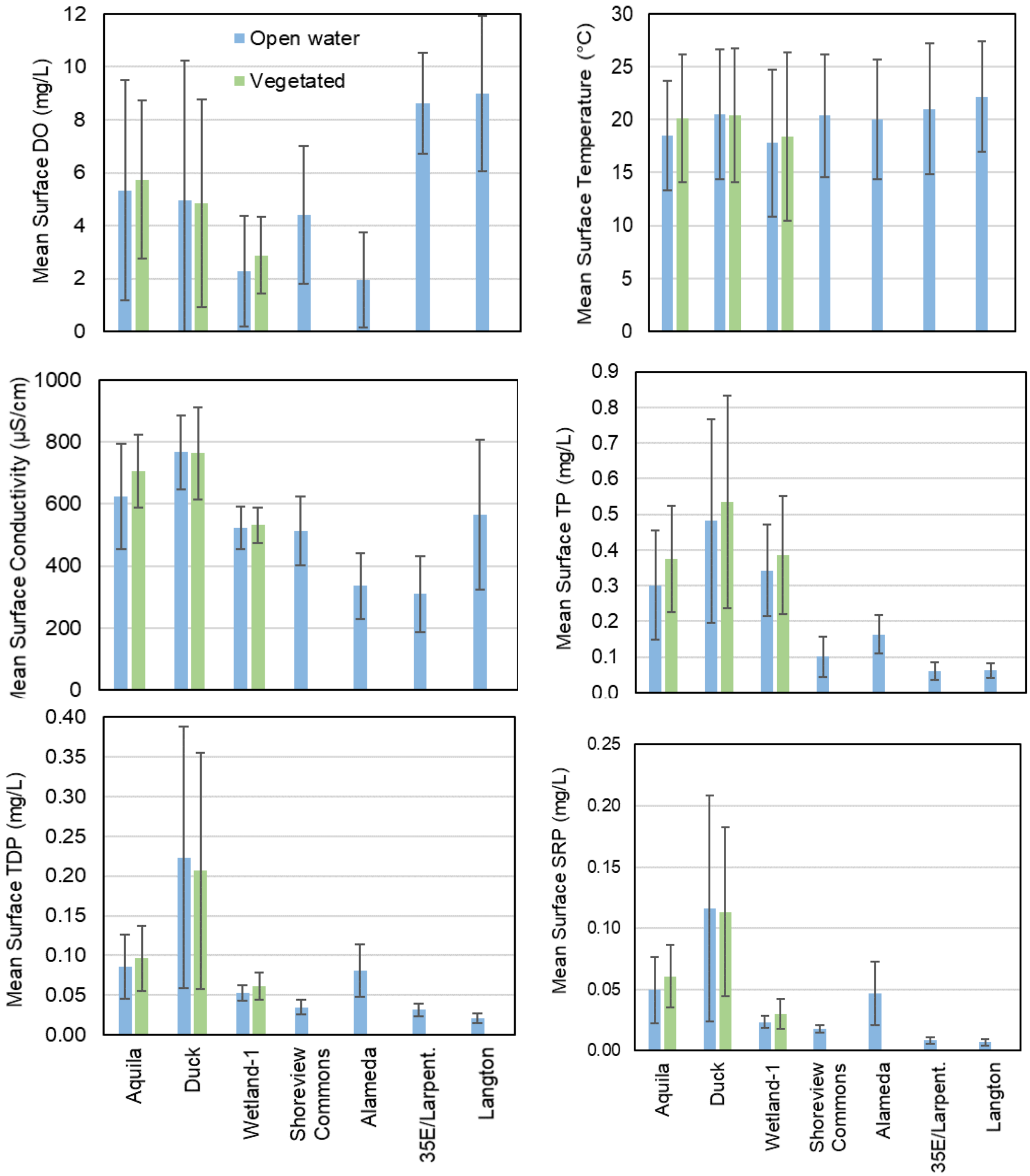

Figure 3.8 Mean and standard deviation of surface concentrations of DO, temperature, specific conductivity, and phosphorus species in the open water ponds (Shoreview Commons, Alameda, 35E/Larpenteur, Langton) and ponds with emergent vegetation (Aquila, Duck, Wetland-1) during Field Season 2 (May – Nov, 2021). At the three ponds with emergent vegetation, data for paired sampling in the open water and emergent vegetation zones are plotted.

3.9 Year-to-Year Comparison (Field Season 1 vs. Field Season 2)

Data collected in Field Season 2 (2021) and Field Season 1 (2020) were compared to illustrate any differences across the monitored sites due to climate, recognizing that both 2020 and 2021 were dry years. Water quality in the open water ponds was compared to the water quality in the open water portion of the ponds with emergent vegetation during the two seasons (Figure 3.9). The 2020 data were excluded for Duck and Wetland-1 because measurements were available only for one or two events in late fall of 2020.

The mean surface DO concentrations were lower in most ponds in 2021 than 2020; this was especially obvious at Aquila which was predominantly anoxic and had thick duckweed cover during the summer period. Alameda had low surface DO in both years and this pond has been found to be strongly-stratified, persistently anoxic top-to-bottom, and fully-covered by thick duckweed for much of the summer period every year. At 35E/Larpenteur and Langton, although the 2021 surface DO was slightly lower than 2020, water columns tended to be well-oxygenated and well-mixed.

Surface water temperatures in 2021 were higher than in 2020, likely because of record warm air temperatures in 2021; Aquila was an exception to this observation. Since Aquila had both lower surface DO and surface temperature, the effects of other parameters (duckweed, weather/wind) need to be considered. Conductivity levels in 2021 were higher in Aquila, Alameda, 35E/Larpenteur, and Langton; this is likely related to the watershed chloride input to the ponds. Except in Alameda and 35E/Larpenteur, surface conductivity was high (> 500 µS/cm) at all sites during both years (see conductivity contour plots in Appendix A).

Surface TP concentrations at the ponds were, however, largely unaffected by differences in climate between the 2020 and 2021 seasons. TDP and SRP trends were similar (data not shown). 2021 was characterized by a predominantly dry period from June to July and a wet period from late August to early September. As shown earlier in the time series plots, surface TP increased during this wetter period, likely reflecting watershed inputs.

3.9.1 Potential Effects of Iron Filings Addition in the Shoreview Commons Pond

The Shoreview Commons Pond was an exception to the seasonal trends largely observed at other ponds, specifically for surface DO and TP concentrations, when comparing 2020 and 2021 data. We suspect that the observations made in 2021 are a direct response to the iron filings treatment applied at the Shoreview Commons Pond in February 2021 rather than factors that can be attributed to climatic variation. In 2020 (see Appendix A), the pond had a primarily anoxic water column from June to October, with periodic but only a brief increase in surface DO (summer anoxic factor of 0.56; Appendix A). Duckweed was dense and at 100% cover in June that decreased to 50% by September. In 2021 (Figure 3.7), the pond did not maintain temperature stratification, had lower density and less persistent duckweed cover, and low DO only at peak duckweed cover in July 2021 (summer anoxic factor of 0.32; Appendix A). When compared to the 2020 surface TP (and TDP and SRP) concentrations, the pond water contained lower phosphorus concentrations throughout the 2021 season; while the mean surface TP in 2021 (0.10 mg/L) was nearly half the 2020 concentration (0.24 mg/L), the 2021 TDP and SRP were about 60% of 2021 levels (hypolimnion concentrations were also low in 2021 vs. 2020). The most likely explanation is that the pond contained low phosphorus due to lowered internal P release (effect of iron filings addition) and thus a low overall availability of P for uptake by duckweed in 2021. The lower duckweed cover likely manifested an increased surface DO in 2021 vs. 2020.

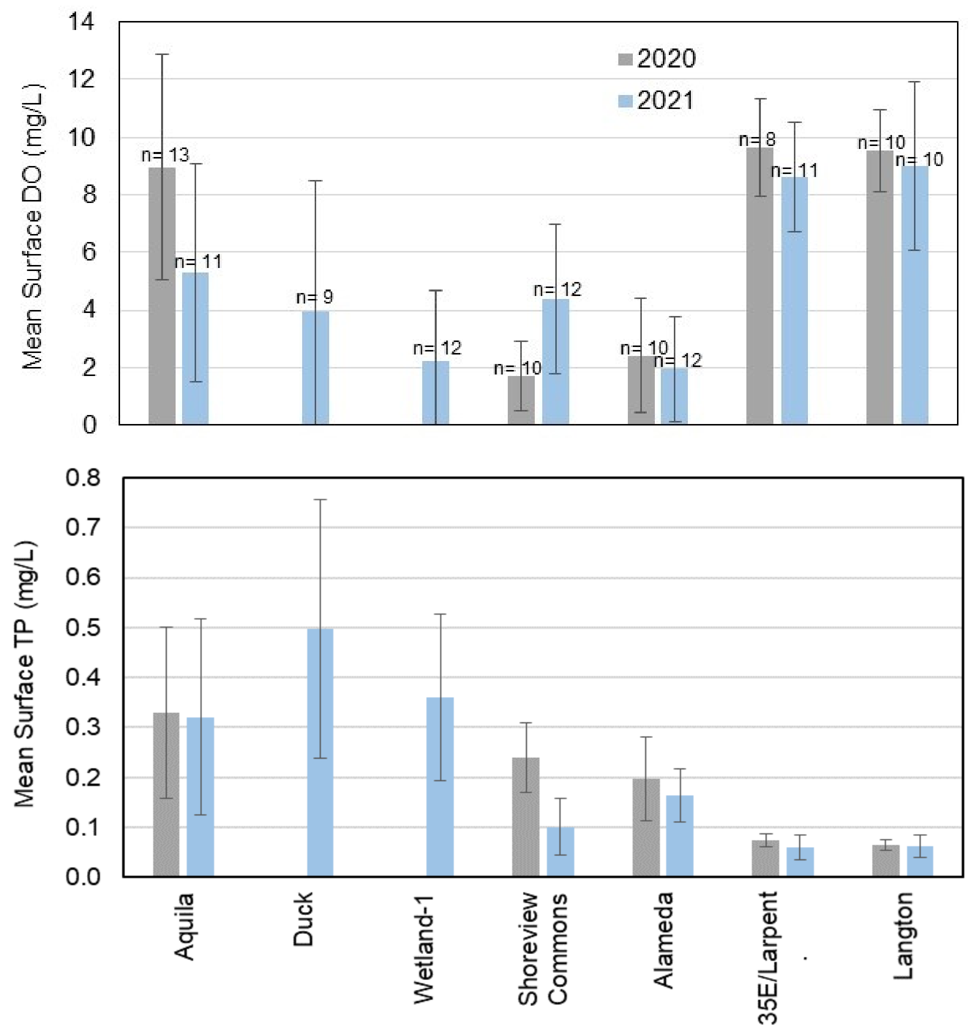

Figure 3.9 Comparison of mean surface concentrations of DO (top) and total phosphorus (TP) (bottom) at the monitored ponds during the 2020 and 2021 field seasons. Error bars represent standard deviation of the mean. 'n' is the number of sampled events. 2020 seasonal mean not available for Duck and Wetland-1.

Chapter 4: Laboratory Phosphorus Release Study Results

The purpose of the laboratory experiments was to determine the phosphorus flux from the pond sediments under oxic and anoxic conditions and to analyze the sediment phosphorus chemistry. For sediment phosphorus release, a subset of five ponds from the seven monitored ponds were chosen (Table 3.1): Aquila, Duck, Wetland-1, Shoreview Commons, and Alameda. We have included data on Langton and 35E/Larpenteur pond sediments (unpublished data from our previous work) to support the discussion of the results for the five ponds.

4.1 Methods

In each pond, intact sediment cores with overlying pond water were collected from five locations distributed in the open water portion of the pond. While Duck and Wetland-1 were cored in July 2021, the other ponds (Aquila, Shoreview Commons, and Alameda) were cored as part of previous studies, as indicated in Table 3.1.

The phosphorus (P) release experiment 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). The calculated flux is for the lab incubation temperature of ~20 °C.

After the completion of the P release experiment, the upper 10 cm depth of the sediment cores were sectioned for chemical analysis. Using the sequential phosphorus extraction procedure (method adapted from Psenner and Puckso 1988), the total sedimentary phosphorus pool was fractionated into the bioavailable and unavailable phosphorus forms. The bioavailable phosphorus species are releasable under low DO conditions and consist of the loosely-bound P (porewater-soluble and easily disassociated from a solid), iron-bound P (P attached to iron minerals), and labile organic P (organic P available for microbial degradation). The aluminum-bound P (P attached to aluminum minerals), mineral-bound P (P attached to primarily calcite and apatite), and residual organic P (P considered recalcitrant or not available for microbial degradation) are the relatively unavailable since they are insensitive to changes in DO conditions. The sediment moisture content (drying at 105 °C) and sediment organic content (loss on ignition at 550 °C) were also determined.

4.2 Results

4.1.1 Oxic and Anoxic Phosphorus Release

The phosphorus release study results for the five ponds (Aquila, Duck, Wetland-1, Shoreview Commons, and Alameda) are shown in Figure 4.1, where the rate of change of phosphate concentration in the water column indicates phosphate release rate. Under oxic (aerated) conditions, the initial water column phosphate concentrations decreased in the columns from the Alameda, Aquila, Duck, Wetland-1, which means sediment phosphate release does not occur under oxic conditions in these pond cores. Only the columns from Shoreview Commons (taken before iron filings treatment) showed oxic phosphate release from the sediments, indicating that there is a high mineralization of labile organic P in these sediments (Jensen and Andersen 1992).

Once the air supply was switched off, the water column DO concentrations started decreasing due to the sediment oxygen demand (Appendix A). The rate of water column DO consumption was rapid in the cores from Alameda, with DO concentration reaching < 0.6 mg/L within 2 days in all cores. In the other pond cores, the water columns approached the anoxic state (DO <1 mg/L) over a period of 2 to 5 days. 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. 2018). This is supported by the observed concomitant increase in water column phosphate due to sediment phosphate release under the reduced DO conditions. The phosphorus release began after 4 days in some pond cores (Shoreview Commons, Wetland-1, some cores from Duck) but took up to 15 days in others (Aquila and some cores from Alameda) during the air off phase (Figure 4.1).

In the last phase with an anoxic water column, phosphate release continued to occur in all sediment cores from Aquila, Shoreview Commons, and Wetland-1. Sediments from Alameda exhibited more variability in phosphate release during the last phase, as some cores did not exhibit an increase in water column phosphorus. Also, it appeared that most of the releasable phosphate in the sediments had already been released under the previous air off phase in two of the cores from Alameda. Two columns from Duck (DP3 and DP4) that already contained a high water column phosphate exhibited minimal accumulation of phosphate from the sediments under anoxic conditions; this could be due to equilibrium conditions between water column and sediment porewater but it was not verified by measurements.

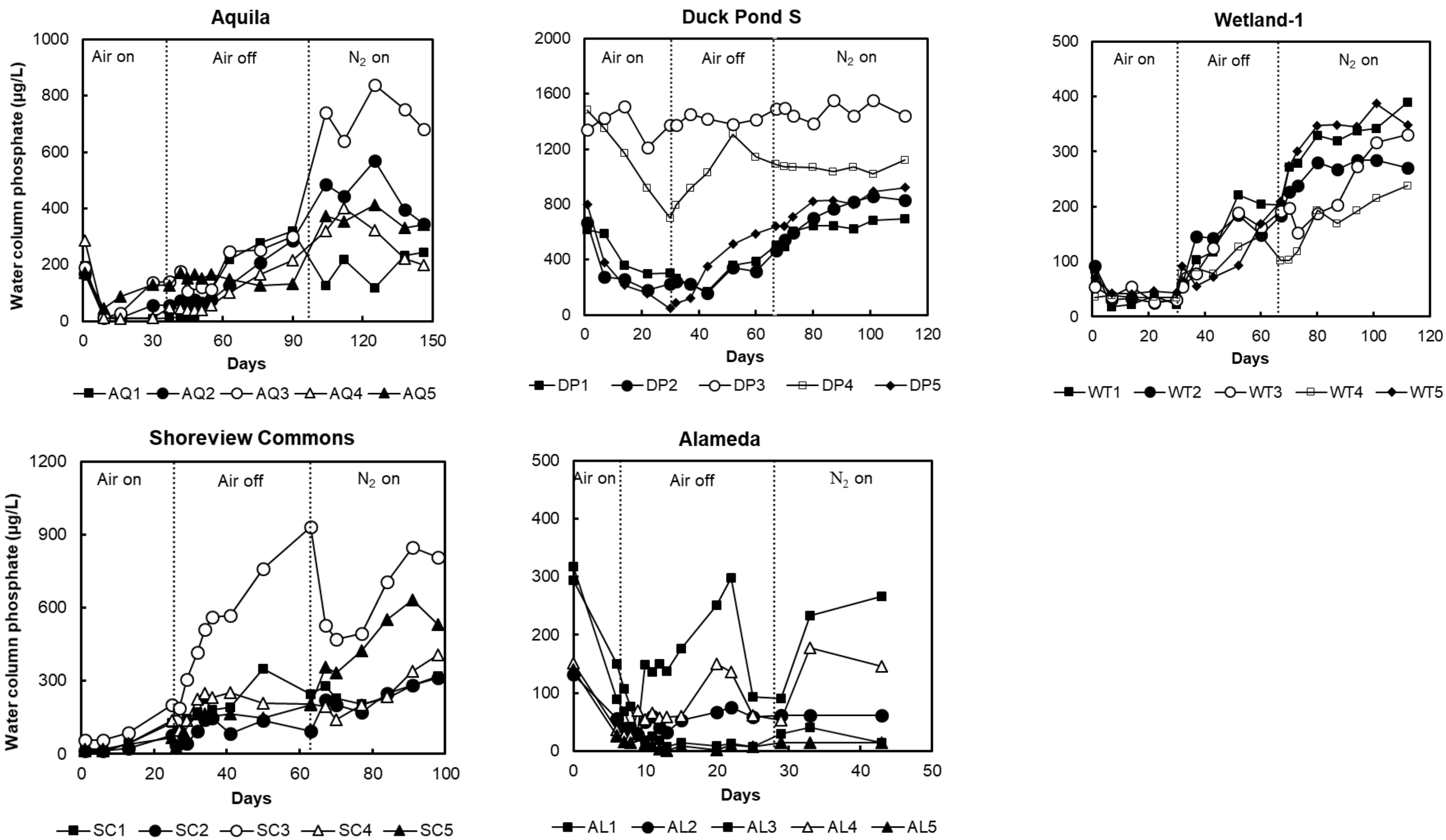

Figure 4.1 Laboratory phosphorus release study results showing the ortho-phosphate concentrations in the water columns of the five sediment cores from each pond under the air bubbling (oxic), air off, and N_2 bubbling (anoxic) phases at 20°C. Notice the difference in y-axis scale in the plots. Data for Aquila, Alameda, and Shoreview Commons are from previous studies (data reported in Taguchi et al. 2020; Janke et al. 2021).

The mean oxic and anoxic sediment phosphate release rates for the five ponds are summarized in Table 4.1. Under oxic conditions, the phosphate flux was negative for Alameda, Aquila, Duck, and Wetland-1. The high oxic phosphate flux of 4.28 mg/m^2/day for Shoreview Commons before iron filings treatment suggests a high mineralization of labile organic P in these sediments (Jensen and Andersen 1992). The anoxic phosphate flux values for the five ponds are moderately high and indicate the risk for internal phosphorus loading in these ponds under summertime anoxia. Previous work (unpublished data) on sediments from the relatively new ponds, Langton and 35E/Larpenteur, determined that these sediments contain low organic matter and have a low potential for anoxic sediment phosphorus release (Table 4.1). The range of observed P flux in the seven ponds highlight the differences in sediment quality among other risk factors.

Table 4.1 Sediment phosphate flux, sediment oxygen demand (S_{max}), and sediment organic content in the pond sediments. Values listed are mean ± 67% confidence interval for five sediment cores from each pond. The oxic flux for Alameda may be subject to experimental error. Data for Langton and 35E/Larpenteur ponds are from previous work (unpublished). 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)	S_{max} (g/m^2/day)	Sediment organic
Alameda	-19.8 ± 2.98	3.64 ± 1.63	5.19 ± 0.591	28%
Aquila	-4.09 ± 1.57	3.54 ± 0.888	2.79 ± 0.519	28%
Shoreview Commons	4.28 ± 0.869	5.93 ± 1.03	4.64 ± 0.687	34%
Duck Pond S	-14.1 ± 4.43	5.52 ± 0.68	3.15 ± 0.254	34%
Wetland-1	-1.10 ± 0.477	3.52 ± 0.546	3.42 ± 0.369	35%
Langton	-0.047 ± 0.011	0.628 ± 0.383	1.02 ± 0.119	4.8%
35E/Larpenteur	-0.165 ± 0.132	1.22 ± 1.02	0.798 ± 0.180	12%

4.1.2 Sediment Phosphorus Fractions

The sediment phosphorus chemistry was analyzed to determine the amounts of biologically-available sediment P species that release under low DO conditions (loosely-bound and iron-bound P) or by bacterial mineralization into phosphate (labile organic P), and the phosphorus species that are generally not available for sediment P release (aluminum-bound, mineral-bound and residual organic P). The vertical profiles of the various phosphorus fractions in the sediments of the five ponds and the sediment physical characteristics are provided in Appendix B.

The mean concentrations of the phosphorus fractions in the top 4 cm depth of the pond sediments are shown in Figure 4.2. Loosely-bound P was measured in low concentrations in all five ponds. The iron-bound P was relatively low (< 0.15 mg P/g) in the upper sediments of Alameda and Shoreview Commons. A much higher concentration of iron-bound P was found in Wetland-1 sediments (0.30 mg P/g in the 0-2 cm depth) and in Aquila sediments (0.53 mg P/g in the 3-7 cm depth). Labile organic P was found to be high in the sediments of Alameda, Duck, and Wetland-1, which is in agreement with the high organic content of these sediments (organic matter content = 28% in Alameda, 34% in Duck and 35% in Wetland-1). In fact, labile organic P was the major mobile P fraction in these pond sediments. In Aquila and Shoreview Commons, labile P was comparatively lower in concentration, although sediments contained high organic content (organic matter content = 28% in Aquila, 34% in Shoreview Commons). The unavailable forms of P were found to be significant in some pond sediments; for example, mineral-bound P in Aquila and Shoreview Commons, aluminum-bound P in Wetland-1, residual organic P in Alameda and Duck were the P fractions present at significant concentrations.

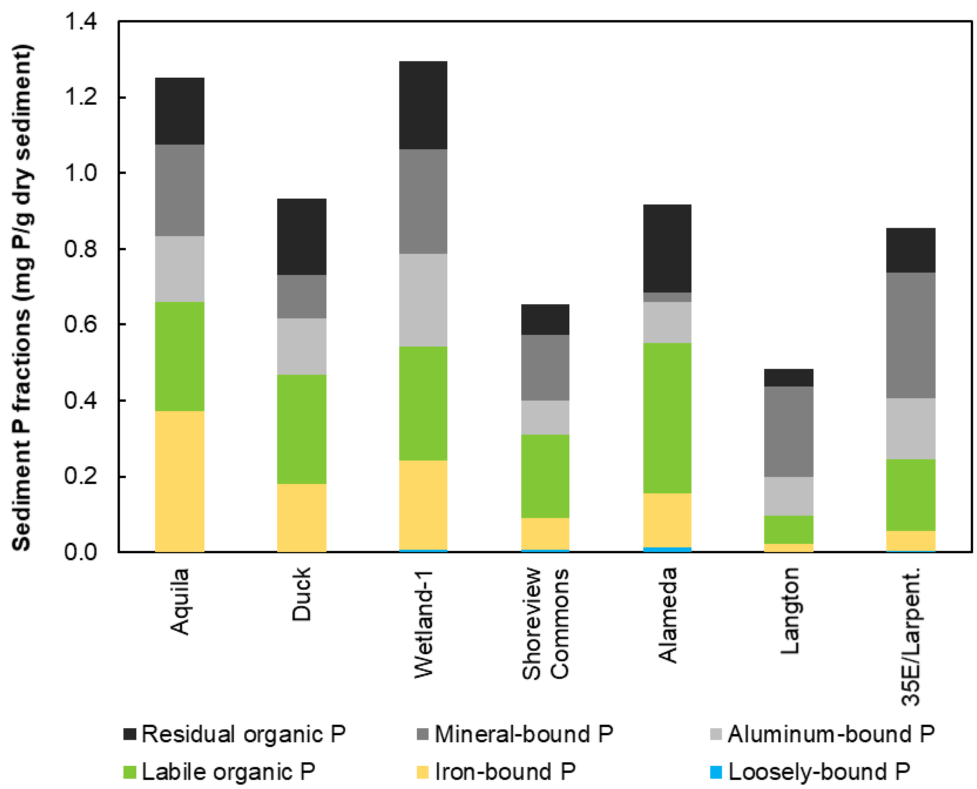

Figure 4.2 Concentrations of the sediment phosphorus fractions in the five ponds selected for laboratory study. Data for Langton and 35E/Larpenteur ponds are unpublished data from previous work. The average composition in the upper 4 cm depth of sediments in 3-5 cores from each pond is shown. Sum of the six fractions provides total P in the sediments. The loosely-bound P, iron-bound P and labile organic P are the mobile P in the sediments and other forms are relatively unavailable under anoxic conditions.

Figure 4.3 illustrates the relative mass of the various P forms in the sediments and that the ponds contain different amounts of bioavailable P that have the potential to release under low DO conditions (redox-sensitive P) and after organic phosphorus is broken down by microbes. The mean mobile P concentration (loosely-bound + iron-bound + labile organic-P) in the top 4 cm depth was determined to be 0.550 mg/g in Alameda, 0.660 mg/g in Aquila, 0.271 mg/g in Shoreview Commons, 0.518 mg/g in Duck, and 0.579 mg/g in Wetland-1. The mobile P thus constituted 61% of total P in Alameda, 50% of total P in Aquila, 46% of total P in Shoreview Commons, 60% of total P in Duck, and 51% of total P in Wetland-1 sediments. The mobile phosphorus forms thus appear to be important and may play a large role with respect to phosphorus dynamics in these ponds under anoxic water conditions. The contrast in sediment phosphorus composition in the newer ponds is evident by the low mobile P mass of 20% in Langton and 29% in 35E/Larpenteur.

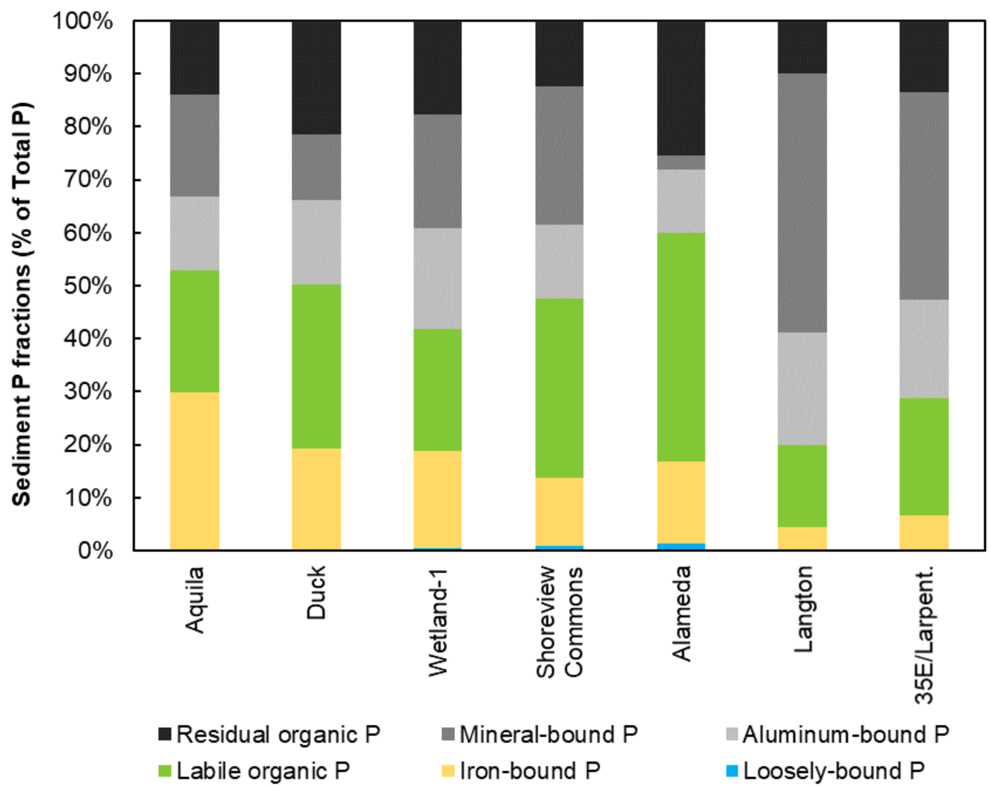

Figure 4.3 Composition of the sediment total phosphorus (P) in the five ponds selected for laboratory study. Data for Langton and 35E/Larpenteur ponds are unpublished data from previous work. The average composition in the upper 4 cm depth of sediments in 3-5 cores from each pond is shown. The sum of loosely-bound P, iron-bound P and labile organic P represents the mobile P in the sediments, where this P is released by redox changes and bacterial mineralization. The other P forms (aluminum-bound, mineral-bound and residual organic P) are generally not available for sediment P release.

Chapter 5: Data Synthesis and Analysis of Phosphorus in Ponds in Support of the Pond Assessment Tool

5.1 Developing Indicators of Risk of High Phosphorus in Stormwater Ponds and Wetlands Treating Stormwater

5.1.1 Introduction

In a past project funded by the Minnesota Stormwater Research Council (Janke et al. 2021), we investigated potential factors to explain high phosphorus (P) concentrations and internal P loading observed in the surface water of a limited number of ponds. These factors or “risk indicators” included pond and watershed characteristics developed from spatial data (e.g., land use, tree cover) and pond data (e.g., depth, size, volume, duckweed cover, dissolved oxygen) that would indicate more complex factors like old age, accumulation of organic matter and sediment, watershed disturbance, or sheltering from tree canopy, which based on previous work (e.g., Taguchi, et al. 2018, 2020) would be expected to cause high P levels, in particular from internal loading. In the smaller data set of that project, some patterns were found between pond total phosphorus (TP), internal loading and indicators related to dissolved oxygen, duckweed cover, and near-shore tree canopy cover. We found, however, that the dataset was too limited to result in strong correlations, because of the complexity and interdependency of the indicators.

A major goal of the current project was to expand and refine those initial analyses by continuing monitoring for two additional years in some ponds, including two years of monitoring in vegetated shallow waters (see Chapter **Error! Reference source not found.**), performing sediment P release studies in both sets of ponds (see Chapter 4) and aggregating additional pond TP data acquired from watershed managers (n > 200 ponds; see Section 0). Our focus was on understanding the extent to which characteristics of the watershed (land cover, land use, hydric soils) and waterbody (free-floating and emergent vegetation, age, size, depth) were useful predictors of observed TP concentrations and sediment P release in shallow water bodies, as well as the extent to which categories could be developed from these indicators for classifying relatively high or low levels of pond TP and sediment P release. Given the complex nature of pond hydrology and nutrient cycling, and the variability observed in the TP data from the previous project, we expected a categorical analysis to be more useful for characterizing ponds than a detailed statistical (regression) analysis.

In this section we describe the factors considered important to pond P dynamics, along with likely mechanism(s) of influence and a brief description of how we defined or assessed the indicators (Table 5.1). A broad range of processes are described by these indicators, including P or sediment inputs to ponds, oxygen dynamics and vertical transport within ponds (through mixing or stratification processes), sediment characteristics, and legacies of sediment, organic matter, and P inputs. The analyses investigating the influence of these indicators are provided in previous reports (Taguchi et al. 2018; Janke et al. 2021; Natarajan and Gulliver 2022), with a summary in Section 5.3.

Table 5.1 Potential indicators of risk for poor performance for retention of phosphorus by ponds, considered by current project. Mechanisms of effect and method of calculation are also described.

Indicator	Potential Effect on P	Description
Anoxic Factor	Oxygen Dynamics, Sediment Release	Fraction of sediments exposed to anoxic overlying water during monitoring season
Shoreline Canopy Cover	Wind Sheltering/Oxygen Dynamics, Litter Inputs	Fraction of land in 25 m buffer around pond classified as canopy and/or buildings
Shoreline Canopy Height	Wind Sheltering/Oxygen Dynamics, Litter Inputs	Mean height of canopy and building in 25 m buffer around pond, from LIDAR
Embankment Height	Wind Sheltering/Oxygen Dynamics	Mean height of land relative to water surface in 25 m buffer around pond, from LIDAR
Wind Reduction	Wind Sheltering/Oxygen Dynamics	Mean reduction in wind speed observed at pond relative to that at nearest airport
Mixing Frequency	Oxygen Dynamics, Vertical Transport	Fraction of days that pond was mixed, based on a minimum RTRM threshold
Free-floating plant (mainly duckweed)	Wind sheltering, Oxygen dynamics, Organic matter inputs	Fraction of pond surface area that is covered by duckweed
Emergent Vegetation Cover	Oxygen dynamics, organic matter, mixing	Fraction of pond surface area covered by rooted, emergent macrophytes
Sediment TP	Sediment P Release	TP concentration in the upper 4 cm depth of sediments
Sediment organic matter	Sediment P Release	Organic matter concentration in the upper 4 cm depth of sediments

Indicator	Potential Effect on P	Description
Sediment Fe:P	Sediment P Release	Total Iron to Total Phosphorus mass ratio in the upper 4 cm depth of sediments
Sediment Redox-P & Labile Organic P	Sediment P Release	Redox-P or Labile Organic P in the upper 4 cm depth of sediments
Pond age	Sediment Release, Organic matter accrual, Litter inputs	Pond age since construction or connection to storm drains relative to year 2021
Pond area	Oxygen Dynamics, Vertical Transport, Hydrology	Surface area of the pond
Maximum depth	Oxygen Dynamics	Maximum depth measured from the pond surface
Mean depth	Oxygen Dynamics	Mean of various depths measured from the pond surface
Land Cover	Phosphorus and sediment inputs to pond	Land cover (pavement, grass, canopy) in a 500 m vicinity of pond
Land Use	Phosphorus and sediment inputs to pond	Land use (residential, commercial, other) in 500 m vicinity of pond
Watershed area	External nutrient inputs, Litter inputs	Drainage area contributing to pond

5.1.2 Risk Indicators: Description, Purpose, and Calculation Methods

Risk indicators related to watershed factors, hydrogeologic classification, and pond characteristics were derived from analysis of several spatial and aerial imagery datasets, most of which were available from the Minnesota Geospatial Commons[6] or Google Earth, with other datasets on water quality or pond morphology and construction age being provided by cities and agencies. Many of these analyses were carried out by manually inspecting relevant imagery and spatial data for each site, though as possible, watershed characteristics were determined through an automatic spatial analysis (buffering) technique (see example in Figure 5.2). Risk indicators are described below along with motivation for their inclusion and a brief description of their method of calculation. Datasets (I, II, III) pertaining to the analyses are described in the next section (Section 0).

5.1.2.1 Construction or Connection Age

Newly constructed ponds may start under conditions of low sediment P and high binding capacity, and accumulate P and organic matter over time. Natural ponds or wetlands modified to receive stormwater runoff may also undergo changes, including P accumulation, that could result in high rates of internal P loading. In our dataset, construction dates were often unavailable for ponds older than about 1990, though among these older ponds some dates were provided either for construction (for new ponds) or for when existing ponds were connected to storm drains. Ponds of undetermined age were investigated using historical aerial imagery in Google Earth, with pond age approximated using the year when a pond first showed up in the aerial imagery. The earliest imagery was 1991, and many ponds were visible in this year. With such uncertainty around pond age, we used a categorical approach, with ponds binned by date of construction or connection to storm drains: "1990 or Before" (pond age ≥ 30 years relative to 2021, including ponds visible in 1991 aerial imagery), "1991-2010" (pond age = 10 to 30 years, relative to 2021), and "After 2010" (pond age ≤ 10 years, relative to 2021). In this analysis, we made no distinction between ponds constructed new vs. existing ponds connected to drains, nor did we adjust age for any major pond maintenance or retrofits (e.g., dredging).

5.1.2.2 Land Use

Land use is an important indicator for watershed development history and for magnitude of phosphorus and sediment in runoff inputs to ponds. Land use was determined for a 500 m buffer around the ponds (Dataset II) using the Metropolitan Council's 2020 land use map for the TCMA[7]. A 500 m buffer was used as an approximation of pond watershed area, given that very few ponds in our larger dataset (Dataset II; n = 230 ponds) had information on watershed delineations.

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[6] https://gisdata.mn.gov/

[7] https://gisdata.mn.gov/dataset/us-mn-state-metc-plan-generl-lnduse2020

5.1.2.3 Hydric Soils and Historic Water Body Classification

Urban ponds set in hydric soils or in former wetlands or lakes may have higher risk for elevated phosphorus. In particular, hydric soils or soils in former wetlands and historic lakes (extant or drained) may be associated with higher levels of organic matter and sediment TP from legacy inputs, which would potentially contribute to loading to ponds in addition to current watershed inputs. For this analysis, we used a 100 m buffer around the ponds to describe the vicinity, using SSURGO[8] to look for presence of hydric soils near the ponds (typically described at levels of 0%, 5%, 45%, 95%, or 100%), and a historic waterbody layer[9] to look for historic wetland classification. We acknowledge that urban soils data are relatively incomplete, and thus this analysis was considered approximate.

5.1.2.4 Water Body Classification

Several classification systems exist to describe the open water, vegetation, and sediment characteristics of shallow water bodies, including the Cowardin, Circular 39, and Specialized Plant Community Classification systems, all of which have been applied to shallow water bodies in the state of Minnesota by the DNR (Kloiber et al. 2019)[10]. We also used the classification system of the Metro Mosquito Control District (MMCD)[11], who maps smaller wet areas in the TCMA and ground truths the locations of many of these sites. MMCD classifies wet areas based on Circular 39 descriptions of water permanence and applies observations of shore slope, substrate, and vegetation type and extent, providing additional detail not found in the DNR classifications.

5.1.2.5 Emergent Vegetation Cover

Emergent vegetation impedes mixing and promotes anoxia (e.g., Winikoff and Finlay 2023), and therefore could have a positive effect on pond surface water TP. Ponds dominated by emergent vegetation also tend to be shallower, warmer, and rich in organic matter, all factors associated with internal P loading. We use the term “emergent vegetation” to denote emergent rooted vegetation (such as cattail, reed) in ponds. This category does not include floating plants (such as duckweed). Emergent vegetation cover was estimated as the proportion of permanently inundated areas dominated by emergent plants, using one of two methods. The primary method used the MN DNR Wetland Inventory[12], which includes identification of open water (Cowardin ‘UB’ or ‘AB’ classification) and emergent vegetation (Cowardin ‘EM1’ classification) areas of water bodies. For ponds that did not show up in this map, recent aerial imagery (Google Earth) was used to approximate emergent cover fractions.

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[8] https://catalog.data.gov/dataset/gridded-soil-survey-geographic-database-gssurgo-minnesota
[9] https://gisdata.mn.gov/dataset/water-hist-hydrography
[10] https://gisdata.mn.gov/dataset/water-nat-wetlands-inv-2009-2014
[11] https://gisdata.mn.gov/dataset/org-mmcd-env-wetland-mosquito-wet-areas
[12] https://gisdata.mn.gov/dataset/water-nat-wetlands-inv-2009-2014

5.1.2.6 Duckweed and Duckweed Cover

We have found in our recent research that duckweed and other free-floating plants dramatically reduce oxygen levels in ponds, potentially leading to internal P loading. We use the term *duckweed* to describe free-floating aquatic vegetation in the ponds (regardless of pond type) that includes both duckweed and water meal. “Duckweed coverage” is a visual observation of the extent of duckweed cover on the surface of the pond and is expressed in terms of percent pond area. We made multiple site visits (typically 8-12) throughout the open-water season (May to October) to note the duckweed cover on the pond surface and obtained a time-averaged cover fraction (Dataset I). An alternate method to estimate duckweed cover is via aerial imagery such as Google Earth, using several images collected over a season or season (preferred method), or at least from a single point in time (e.g., midsummer; June to August) when duckweed cover would be prevalent. For the larger pond dataset (Dataset II) that lacked direct observations, we based duckweed cover on a single date in August 2015 using Google Earth.

5.1.2.7 Land Cover: Tree Canopy

Land cover (such as grass, trees, and pavement) in a pond’s watershed may be important indicators of development history and of runoff inputs of phosphorus and sediment to ponds. **Tree canopy** in particular may have a range of direct and indirect effects on urban pond phosphorus levels. Extensive tree cover may contribute to higher levels of organic matter and TP inputs to ponds through leaf litter inputs to stormwater runoff. Tree cover may also be highly associated with residential land use, which may cause both higher pond TP and runoff TP concentrations due to intensity of human activities, such as fertilization, pet waste, and yard waste. We used a 1-m resolution land cover layer for the TCMA[13] to determine land cover (e.g., tree cover, grass, pavement) within 500 m of the water bodies (n = 230; Dataset II). A 500 m buffer was used as an approximation of pond watershed area, given that few ponds in our dataset (Dataset II; n = 230 ponds) had information on watershed delineations.

5.1.2.8 Shoreline Tree Canopy

Shoreline tree canopy may have a range of direct and indirect effects on pond phosphorus levels. Greater tree cover in the vicinity of ponds serves to reduce wind speed and lower mixing or oxygenation rates, which may contribute to internal loading through suppression of DO levels in ponds. The extent of tree canopy cover and height of tree canopy on pond shorelines was defined using raw LIDAR data (from MnTOPO[14]) extracted from a 25 m buffer around ponds. Using LASTOOLS[15] (which requires a license for large datasets), we determined **shoreline canopy height**, defined as the arithmetic average height (relative to the ground DEM) of all LIDAR returns classified as “canopy” within the 25 m buffer. **Shoreline canopy cover** was defined as the percentage of all LIDAR returns in the buffer that were reflected back from canopy. To quantify the impact of adjacent topographic sheltering, we also computed

[13] https://gisdata.mn.gov/dataset/base-landcover-twincities
[14] https://www.dnr.state.mn.us/maps/mntopo/index.html
[15] https://rapidlasso.com/lastools/

embankment height as the difference in elevation between the pond water surface and the mean ground elevation in the 25 m buffer, using the digital elevation model provided with the LIDAR data. Note that we found canopy height and canopy cover to be highly correlated in our dataset (r = 0.74; n ~ 20 ponds, Dataset I); we used shoreline canopy cover in our regression analysis (Section 5.3) but we recommend using whichever parameter is easier to calculate.

5.1.2.9 Wind Sheltering

A potentially important mechanism for mixing (and reaeration) of pond water columns is through direct mechanical mixing by wind (with runoff inputs and heat exchange the other primary mixing processes). Wind sheltering (wind reduction), which may be more substantial in smaller ponds surrounded by mature trees, may promote anoxia through reduced mixing and/or promotion of duckweed growth, contributing to anoxic sediment release of phosphate within the pond. Tree cover, buildings, and topography in the vicinity and on the shore of ponds can contribute to wind sheltering. Currently, we do not have a cost-effective method to model or estimate this wind sheltering effect for multiple ponds (see Taguchi et al. 2022 for a description of the technique to model wind sheltering and impacts to water column mixing in ponds). However, we measured wind directly using monitoring stations located on the ponds that were equipped with anemometers. We defined **wind sheltering** as the reduction of wind speed observed at the pond relative to concurrent observations at the Minneapolis-St. Paul Airport, averaged over either the summer period (June – Aug) or the entire monitoring period (May – Oct). This observational approach was used in a previous project and is described in more detail in Janke et al. (2021).

5.1.2.10 Relative Thermal Resistance to Mixing (RTRM), Mixing Frequency

Dissolved oxygen in ponds, especially near the sediments, has a strong influence on pond TP through its effect on sediment P release. Pond water column mixing is crucial for introducing oxygen to the lower water column, but highly stratified ponds, which are common in our study area (Janke et al. 2021a) are more resistant to mixing and may at greater risk for high TP. Mixing dynamics, including stratification strength, in ponds can be characterized using changes in density gradients, calculated from the pond temperature time series data. The temperature data are used to calculate the Relative Thermal Resistance to Mixing (RTRM), which is defined as:

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

where ρ is the density of water at layers *i* and *j* (two points in the vertical profile), relative to the density difference between water at 4˚C and at 5˚C. We calculated RTRM for the upper 30-50 cm of the water column and for the entire water column (top vs. bottom of the profile), averaged over the summer data record (June – Aug).

The more positive the RTRM, the more stable the water column, and RTRM near zero indicates an unstable water column that will mix. Holgerson et al. (2022) determined a density gradient value of 0.287 $kg/m^3/m$ to define a threshold for a pond to be considered mixed. This threshold corresponded to an RTRM value of roughly 9 – 65 (dependent on pond depth), and was used to determine the **mixing frequency** of the pond as fraction of days during the monitoring period that a pond mixed (i.e., pond's RTRM went below this threshold; see Figure 5.1).

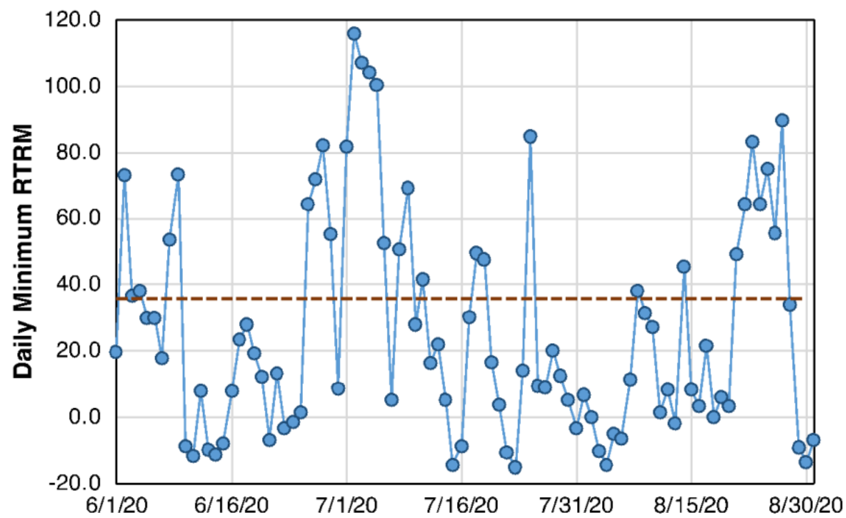


Figure 5.1 Daily Minimum Stratification Strength, as Relative Thermal Resistance to Mixing (RTRM) between the top and bottom of the pond water column at the Shoreview Commons Pond (see Janke et al. 2021a). Dashed line represents the RTRM threshold below which a pond is considered "mixed" per Holgerson et al. 2022; for this summer season (June 1 – Aug 30), the pond was "mixed" on 66% of all days in the record (mixing frequency = 0.66).

5.1.2.11 Anoxic Factor and Dissolved Oxygen

Anoxic factor (AF) is a metric that can be used to characterize the oxygen status of lakes and ponds, to help identify potential for internal P loading. AF is defined by Nurnberg (1995) as "the number of days that a sediment area, equal to the whole-lake surface area, is overlain by anoxic water", is a measure of the persistence of anoxia and exposure of sediment to anoxic conditions. The equation for AF is:

$$AF = \left(\frac{duration\ of\ anoxia}{duration\ of\ monitoring\ period}\right) \times \left(\frac{anoxic\ sediment\ area}{pond\ surface\ area}\right)$$

where we used 2.0 mg/L as the threshold for anoxia. The anoxic sediment area is determined from hypsographic curves derived from bathymetry for each of the ponds (see calculation example in tutorial provided with the report). We calculated AF for the summer period (June 1 – Aug 31) or the entire field season (May 1 to Nov 1) using the DO profiles collected in the ponds. Anoxic Factor (AF) of 1.0 means the entire pond bottom is exposed to anoxia over the entire duration, while AF of zero means the pond water column is completely oxygenated over the entire duration.

5.1.2.12 Sediment Phosphorus Release

Pond sediments can release phosphorus (P) under anoxic (or low dissolved oxygen) conditions. Phosphorus release under oxic conditions is typically negligible, but can occur due to the microbial degradation of organic phosphorus in the sediments and mobilization into phosphate (Jensen and Andersen 1992). We determined the sediment phosphorus release rates (in mg P/m^2 sediment area/day) in the laboratory by performing column studies using intact sediment cores with overlying water columns that were kept under controlled oxic and anoxic conditions (see experiment description in Janke et al. 2021; Results from this study in Chapter 4).

5.1.2.13 Sediment Phosphorus Fractions

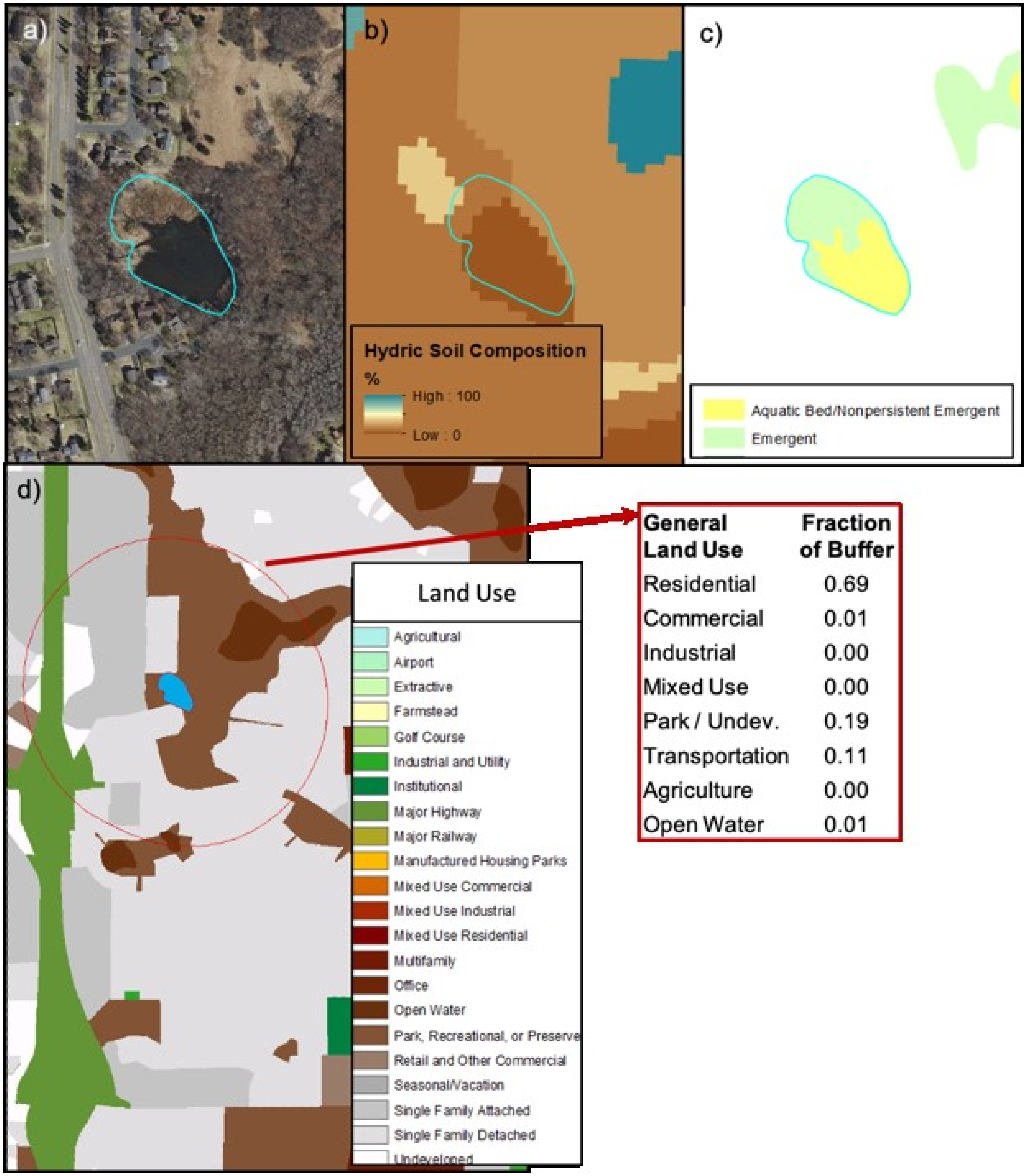
The total sedimentary phosphorus (P) pool consists of biologically-available P species that release under low DO conditions (loosely-bound and iron-bound P) or by bacterial mineralization into phosphate (labile organic P), and the phosphorus species that are generally not available for sediment P release (aluminum-bound, mineral-bound and residual organic P). The sediment phosphorus fractionation is determined by the sequential phosphorus extraction method (Psenner and Puckso 1988). In the present analysis, we used the mean P mass (mg P/g dry sediment) in the top 4 cm depth of the pond sediment cores.

5.1.2.14 Sediment Organic Matter Content

Sediment organic matter (expressed as % dry sediment mass) influences oxygen demand and may indicate organic P pools in aquatic sediment. The high organic matter input to ponds, from sources such leaf litter and grass clippings, can result in the accumulation of organic P in the sediments (Janke et al. 2017; Song et al. 2017; Frost et al. 2019) that may eventually decompose and release phosphorus (Gächter et al. 1988; Golterman 2001). In simple terms, “mucky” sediments are rich in organic matter and are thus more likely to contain high phosphorus content. In the present analysis, we used the mean percent organic matter content in the top 4 cm depth of the pond sediment cores.

5.1.2.15 Sediment Metal Concentrations

The metals (iron, aluminum, calcium) play a role in binding phosphorus in the sediments. Phosphorus bound to iron is considered redox-sensitive (i.e., sensitive to DO fluctuations), while aluminum- and calcium-bound P are not affected by changes in oxygen levels in the pond. We used the concentrations of total iron, aluminum, calcium, and phosphorus (determined by ICP analysis) in the top 4 cm depth of sediment cores for our data analysis.



General Land Use	Fraction of Buffer
Residential	0.69
Commercial	0.01
Industrial	0.00
Mixed Use	0.00
Park / Undev.	0.19
Transportation	0.11
Agriculture	0.00
Open Water	0.01

Figure 5.2 Example of spatial data sets and land use - buffer analysis for Aquila Pond in Bloomington (Table 3.1): (a) aerial imagery (2020; www.mngeo.state.mn.us), (b) hydric soil composition (gSSURGO), (c) Cowardin wetland classification (showing ~50% open water/aquatic bed and ~50% emergent vegetation), and (d) 2016 land use in a 500m buffer around the water body, with summary of land use fraction in table at right.

5.2 Pond Water Quality Datasets

Several urban pond sediment and water quality datasets were collected and assembled by the authors in this and previous projects to support analysis of risk indicators for poor phosphorus retention in ponds treating stormwater. These datasets included intensive monitoring and sampling efforts, widespread sampling surveys of ponds, as well as sediment sampling and coring for laboratory incubation studies of phosphorus release. The three primary datasets are described below:

Dataset I (Intensive Pond Monitoring): Twenty-five stormwater ponds and wetlands treating stormwater comprise the most intensive dataset. These ponds are located throughout the Twin Cities Metro Area (TCMA) and have been studied since 2016, with most sites having one to three years of data collection. Research included routine (roughly bi-weekly) warm-season water sampling and profiling for temperature, dissolved oxygen, and conductivity; continuous monitoring of temperature, water level, and wind speed above the pond; routine assessments of duckweed cover; and collection of relevant site data (bathymetry, construction age). Sediment cores and sediment samples were taken from most of these sites (see Dataset III). A goal of these studies was to investigate factors related to pond characteristics (e.g., land use, age, tree sheltering) and water quality dynamics (e.g., anoxia, stratification, duckweed cover) that influence pond surface water TP concentrations, and guide development of regression models for TP prediction. Results of this previous research are described in papers and reports (Taguchi et al. 2018, 2020; Janke et al. 2021, 2022).

Dataset II (Aggregated Pond TP): This dataset, acquired from our past projects and from several cities, agencies, and practitioners across the TCMA (Table 5.2), includes surface water total phosphorus (TP) observed in 230 ponds across the Twin Cities metropolitan area over the past 15 years. This dataset was used to examine patterns in surface water P related to land cover, land use, hydrogeologic setting, and pond characteristics like age, depth, and vegetation cover. Results of this work are the primary input for the categorical assessment for surface water TP used by the Pond Assessment Tool (described in Chapter Chapter 6:).

Dataset III (Pond Sediment): The third primary dataset consists of sediment-related data collected by the authors. Over several previous projects dating back to 2016, sediment cores have now been collected from over 20 ponds (all of which are included among the Dataset I ponds) and incubated in laboratory settings under oxic and anoxic conditions to quantify sediment phosphorus release. Sediment samples were also collected from these ponds, in addition to a few others, to help relate phosphorus release to standard sediment characteristics such as organic matter and total phosphorus concentration metals content. Detailed fractionation analysis to determine sediment phosphorus forms was made on these sediments. Results of these efforts are described in literature from several previous projects (Taguchi et al. 2018, 2020; Janke et al. 2021; Natarajan and Gulliver 2022).

Table 5.2 Two primary shallow waterbody datasets analyzed for TP concentrations in this project as part of 'Dataset II', with constituent subsets based on data source, number of sites, and years represented. *Note that in the assembled dataset, some sites were represented in more than one subset, e.g., most of the Bloomington sites were also part of the RPBCWD dataset (so the sum of site numbers in the subsets will exceed the total number of sites). No site was included more than once in analyses.

Dataset	*Subset*	**Number of Sites**	**Years Represented**	**Source**
II-A. Task 3 Dataset		**233***	**1992 - 2020**	
Riley-Purgatory Bluff Creek Watershed District		*100*	*2011 - 2015*	RPBCWD Staff
Capitol Region Watershed District		*20*	*2007 - 2020*	CRWD Staff, Online [1]
Mississippi Watershed Management Organization		*3*	*2008 - 2017*	MPCA Surface Water Data Viewer [2]
City of Bloomington		*43*	*2009 - 2020*	Jack Distel, Steve Gurney
City of Eagan		*4*	*1992 - 2008*	Eric Macbeth
Previous Work by Authors (TCMA Urban Core)		*89*	*2011 - 2020*	Taguchi et al. 2020 [3], Janke et al. 2021 [4], unpublished
II-B. MPCA Database		**222**	**1995 - 2017**	MPCA (Mark Gernes)
Twin Cities Metro		*46*	*1995 - 2017*	
Non-Metro		*176*	*1995 - 2017*	

[1] http://waterdata.capitolregionwd.org/applications/login.html?publicuser=Guest#waterdata/stationoverview
[2] https://webapp.pca.state.mn.us/wqd/surface-water
[3] https://doi.org/10.13020/p338-vx49
[4] https://hdl.handle.net/11299/218751

5.3 Data Analysis: Drivers of Pond Water Quality

5.3.1 Drivers of Pond Phosphorus, Anoxia, and Sediment Release from Analysis of Aggregated Datasets

We aggregated data from this project and our previous projects (Taguchi et al. 2020; Janke et al. 2021) and analyzed effects of various parameters on pond phosphorus. Below we describe a subset of parameters from the broader Risk Indicator List (Table 5.1) that represent the most useful and strong predictors of pond surface water TP, anoxia, or sediment phosphate release, as determined by our analyses. While most of the original list of indicators (Table 5.1) had some predictive power for pond TP or anoxia, several were excluded from the tool due to relative data scarcity (e.g., watershed area, mean depth), high covariance (e.g., canopy height and canopy cover), or weaker predictive power (e.g., embankment height, land cover such as grass or pavement). The strongest parameters are used in the development of the Pond Assessment Tool (Chapter Chapter 6:).

Here we describe the strongest risk indicators and their implication or impacts to pond surface water TP concentrations, oxygen levels, and sediment phosphate release, most of which are included in the Pond Assessment Tool. A few indicators, including wetland classification and geography (TCMA vs. greater Minnesota), were weaker indicators of pond TP, and are included for reference at the end of this section.

5.3.1.1 Construction / Connection Age

We found **pond age** to be an indicator of elevated TP, though the effect was highly variable (Dataset II, n = 230 sites; Figure 5.3). Urban ponds constructed prior to 1991 or of undetermined age (pond age ≥ 30 years, relative to 2021) had slightly higher TP compared to the newest ponds constructed since 2010. Constructed ponds built between 1990 and 2010 tended to have the highest and most variable TP concentrations. In the intensively monitored ponds (Dataset III, n = 22), the older (> 30 years) pond sediments contained higher levels of total phosphorus and organic matter content, which indirectly influence the phosphorus release from anoxic sediments (bottom panel of Figure 5.3). The trend is an increase in sediment TP and organic matter with pond age. Therefore, age appears to have effects on surface water TP through the effects of higher sediment TP and organic matter in the pond sediments. Further, the results suggest that there is a risk for high surface TP in ponds constructed in the early 1990's, when development and pond construction was extensive.

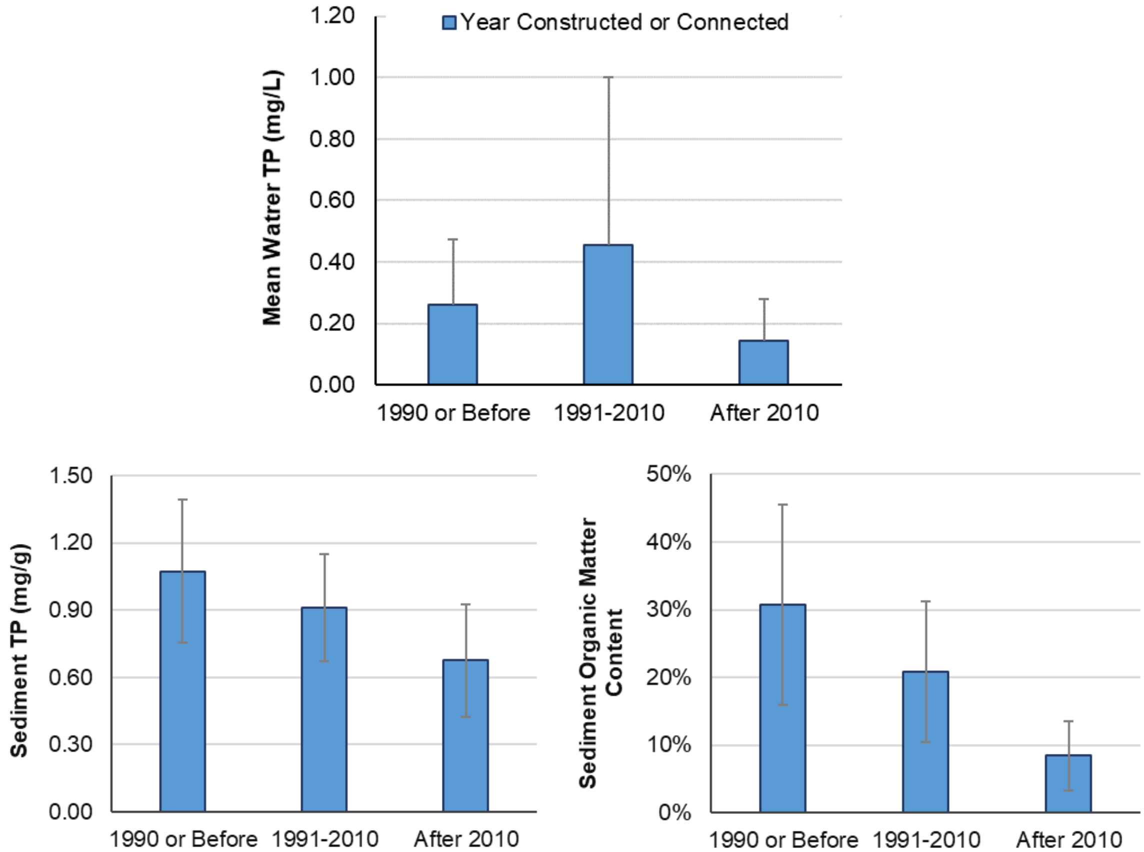


Figure 5.3 Mean and standard deviation of surface water TP concentration (n = 230; top panel) and sediment TP and organic matter content (n = 22; bottom panels) in urban ponds, based on the year of construction or connection to storm drains. Note that in the top panel, the "1990 or Before" group includes ponds with undetermined age (i.e., construction data unavailable) that were visible in 1991 Google Earth Imagery. Construction year can be converted to age based on year 2021 (for example, a pond built after 2010 is ≤ 10 years old).

5.3.1.2 Land Use

Residential land use was by far the most common land use in the vicinity of the urban ponds, and was also associated with the highest and most variable surface water TP in water bodies (Figure 5.4). Residential land use is associated with higher runoff TP concentrations due to intensity of human activities and land disturbance, such as fertilization, pet waste, and yard waste. While the sample size is small, the high TP concentrations associated with "open water" (i.e., surrounded by lake or wetland) and "park/undeveloped" land use likely indicate ponds located in areas with extensive wetland complexes, hydric soils, and historic lakes or wetlands -- factors which were also found to indicate higher levels of pond TP (see section below). The land use classes of commercial, industrial and transportation (airport, railroad, and highway) were associated with lower TP concentrations in the ponds, likely due to lower organic matter and P concentrations in runoff from these land uses.

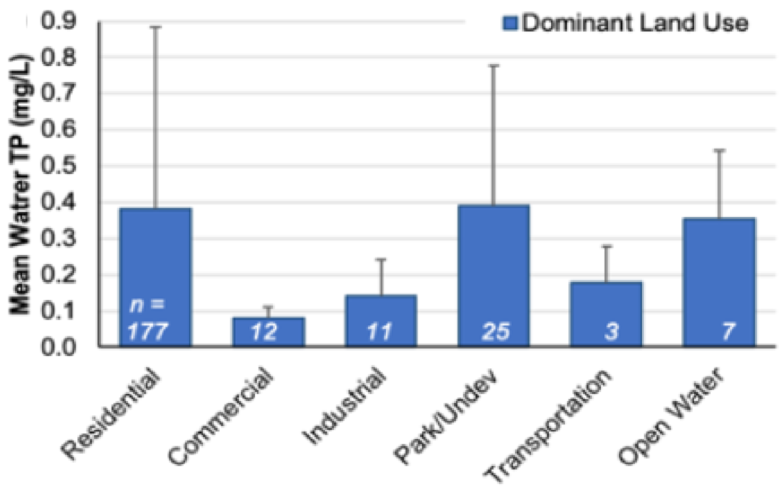


Figure 5.4 Mean and standard deviation of surface water TP concentration (mg/L) based on the dominant land use within a 500 m buffer of the ponds (i.e., land use with the greatest area fraction in the buffer; n = 235 ponds, Dataset II).

5.3.1.3 Hydric Soils and Historic Water Body Classification

Waterbodies that were located in areas of hydric soils or near historic water bodies tended to have much higher TP than those in well drained soils (Figure 5.5). For example, water bodies with greater than 40% hydric soils had more than twice the TP concentration of those in non-hydric soils. Many ponds were not constructed, but were historically part of wetland or lake complexes, and had storm drains connected to them as the surrounding area was developed. Such ponds tended to have slightly higher TP concentrations than ponds located or constructed in areas not historically considered parts of lakes or wetlands.

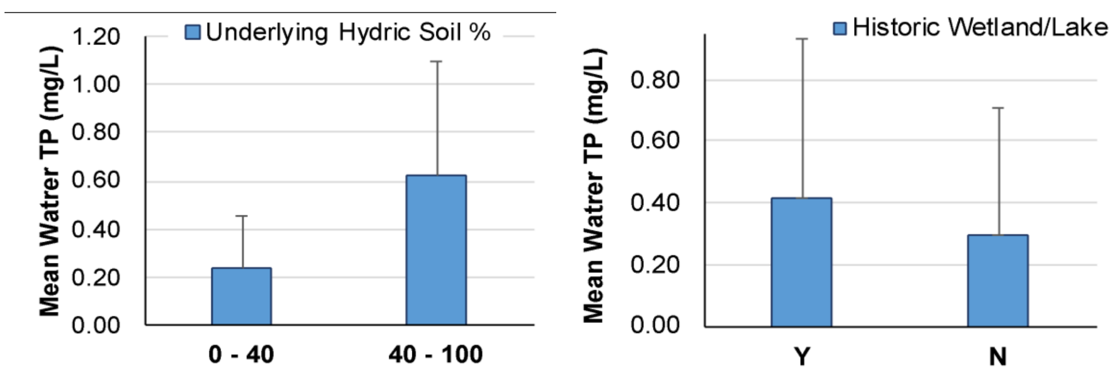


Figure 5.5 Mean and standard deviation of surface water TP (mg/L) across pond sites (n = 230; Dataset II), for two categorizations based on hydrogeologic setting: (a) highest hydric soil composition in and around the pond (as %), and (b) proximity to historic lakes or water bodies (as Yes or No).

5.3.1.4 Emergent Vegetation Cover

Emergent vegetation cover was associated with elevated TP in ponds, as ponds with more than 50% cover in emergent had roughly 2.5 times the mean TP of ponds with less than 50% emergent cover (n = 225, Dataset II; Figure 5.6). Emergent vegetation promotes anoxia (Winikoff and Finlay 2023), and higher TP in emergent-dominated ponds may indicate greater anoxic sediment phosphorus release. Other factors potentially associated with emergent-dominated ponds that may influence sediment phosphorus release, such as shallow depth or higher sediment organic matter, have not been well quantified.

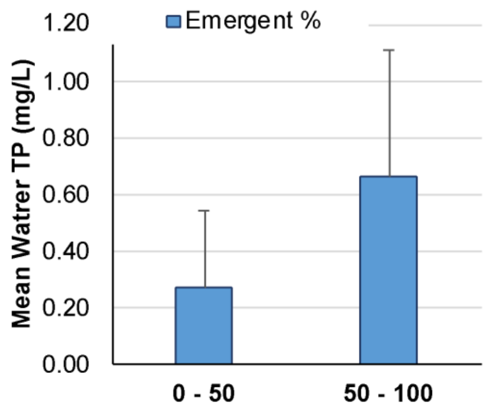


Figure 5.6 Mean and standard deviation of surface water TP concentration (mg/L) in pond sites with low (<50%) and high (>50%) emergent cover (n =225; Dataset II). Emergent vegetation are typically rooted wetland plants and exclude free-floating plants like duckweed.

5.3.1.5 Duckweed and Duckweed Cover

In our analyses, we found strong positive relationships between duckweed cover and pond TP. In the larger dataset (Dataset II), ponds with greater than 50% duckweed cover had roughly twice the mean TP concentration of ponds with lower or no duckweed present (Figure), while in our monitored ponds dataset (Dataset I), pond TP showed significant relationships with anoxic factor (AF) (positive) and DO (negative; Figure 5.7). While anoxic factor has a more direct, mechanistic link to pond TP and sediment release than duckweed (whose effects are likely to be indirect as mediated through strong impacts to oxygen), the generally strong relationships with duckweed (see bottom panel of Figure) present an opportunity for low cost monitoring of relevant pond conditions in the TCMA. Duckweed cover is far easier to assess over a season than dissolved oxygen profiles. Additionally, duckweed is expected to impact pond phosphorus concentrations due to direct uptake of P into plant tissue during growth periods and source during senescence, effects of which have not been quantified in our analysis. However, we note that some larger ponds may also exist in turbid phytoplankton dominated conditions (e.g., Vitense et al. 2019). Thus, while duckweed indicates elevated P and low oxygen, its absence cannot be used exclusively to eliminate the potential for elevated P and low oxygen conditions.

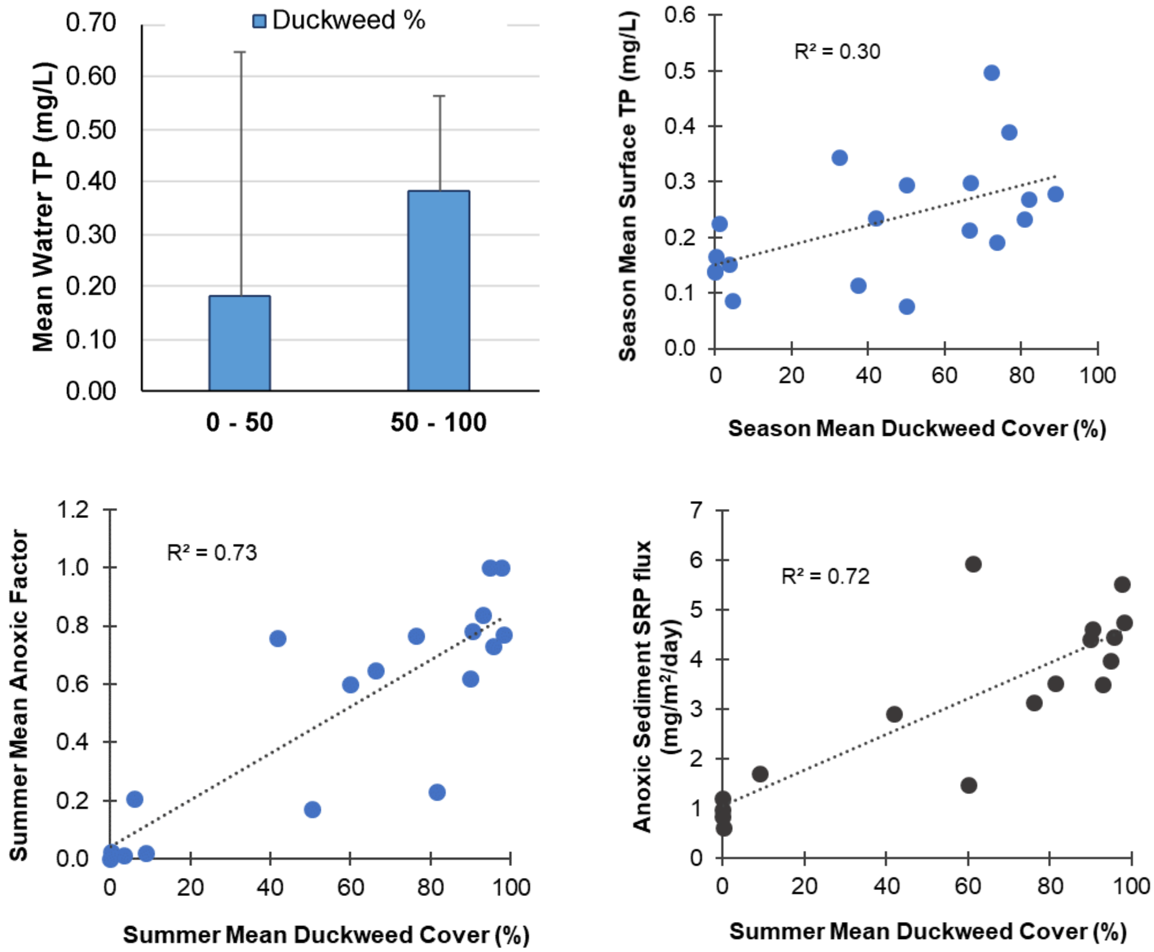


Figure 5.7 Top panel: Mean surface water TP concentration (mg/L) in [left] pond sites with low (< 50%) and high (>50%) duckweed cover, as percentage of total open water area (n =139; Dataset II), and [right] as a function of season (May to October) mean duckweed cover (n = 19; Dataset I). Bottom panel: Summer (June to Aug) mean anoxic factor is strongly influenced by summer mean duckweed cover (%) [n = 20; Dataset III; left]. The correlation between summer duckweed cover and anoxic P flux in laboratory sediments [right] can be explained by their inter-relationship with anoxic factor (n = 19; Dataset III).

5.3.1.6 Land Cover: Tree Canopy

While land cover composition was variable among sites, most cover types (including impervious area and grass cover) were weak predictors of pond TP concentration. However, high tree cover (> 60%) was associated with elevated mean TP relative to water bodies with lower amounts of tree cover, while ponds with lower tree cover (< 20%) had the lowest and least variable TP concentrations (Figure 5.8).

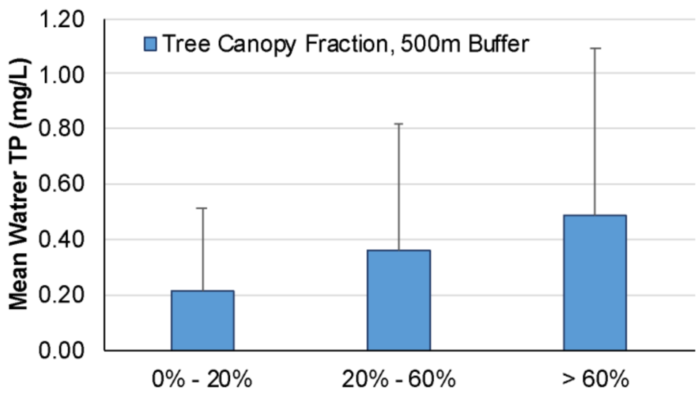


Figure 5.8 Mean surface water TP (mg/L) observed in ponds (n = 230; Dataset II), grouped by percentage canopy cover in a 500 m buffer.

5.3.1.7 Shoreline Tree Canopy

We found that shoreline tree canopy may contribute to internal loading through suppression of DO levels in ponds, as seen by an increase in season (May – Oct) mean anoxic factor with greater tree height around the pond (Figure 5.9, left). We found a similarly positive (though weaker) relationship between shoreline tree cover fraction and season (May – Oct) mean surface water TP (Figure 5.9, right), further suggesting a role of tree canopy in suppressing DO and causing internal release of P. Tree cover near the shore may also contribute phosphorus and organic matter directly to the ponds through leaf litter deposition, and taller trees may contribute more than shorter trees, but it is not clear how the magnitude of leaf litter inputs compares to runoff inputs or internal loading. Embankment height was not a strong predictor of pond TP.

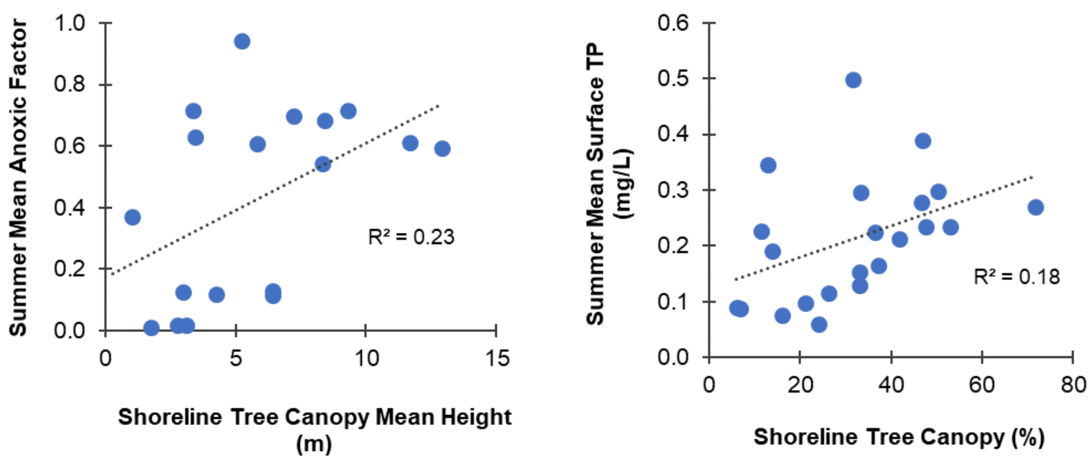


Figure 5.9 Season (May to Oct) mean anoxic factor versus shoreline tree canopy mean height, determined as the average height of all canopy LIDAR returns in a 25 m buffer around the pond (n = 19; Dataset I). Right: Mean season (May to Oct) TP concentration in ponds (n = 22; Dataset I) as a function of shoreline tree canopy (percent canopy in a 25 m buffer, determined from LIDAR).

5.3.1.8 Wind Sheltering

While a small sample size (n = 12, Dataset I), we found that pond water column dissolved oxygen (mg/L) was negatively affected by observed wind sheltering (Figure 5.10, left), suggesting that a lack of wind mixing could promote anoxia in ponds. We also found a positive relationship between wind sheltering and duckweed cover (Figure 5.10, right), indicating that more sheltered ponds may enable high duckweed growth, which in turn strongly promotes anoxia and associated high TP. We do not include this latter relationship in the models but have provided it here to illustrate the complexity of sheltering and oxygen dynamics in ponds.

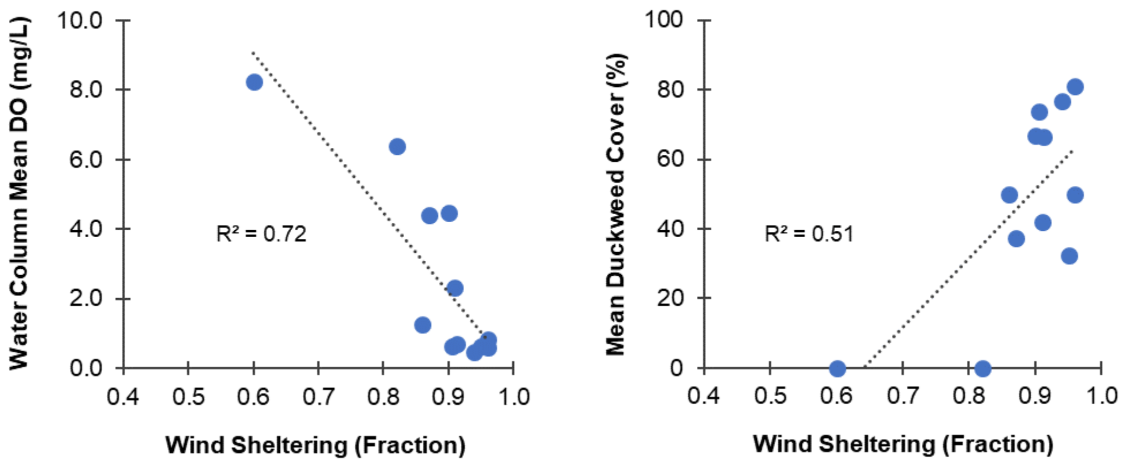


Figure 5.10 Left: Water column season (May – Oct) mean dissolved oxygen concentration (DO, mg/L) in ponds (n = 12; Dataset I) as a function of wind sheltering (reduction of wind observed at airport vs. observed at pond monitoring station over the May – Oct season). Right: season (May – Oct) mean duckweed cover vs. wind sheltering (n = 12; Dataset I).

5.3.1.9 Relative Thermal Resistance to Mixing (RTRM), Mixing Frequency

Mixing frequency, as a fraction of summer days that a pond "mixed" (daily minimum top vs. bottom RTRM < mixing threshold), had a strong negative affect on anoxic factor (Figure 5.11). This indicates that pond water column mixing, regardless of cause (e.g., wind, runoff, or heat exchange), is effectively reducing stratification strength and anoxia in ponds. We note that the season- and summer-mean RTRM values, which indicate stratification strength, also showed strong relationships with anoxic factor and water column dissolved oxygen, but these analyses are on-going and not included in the tool at this time.

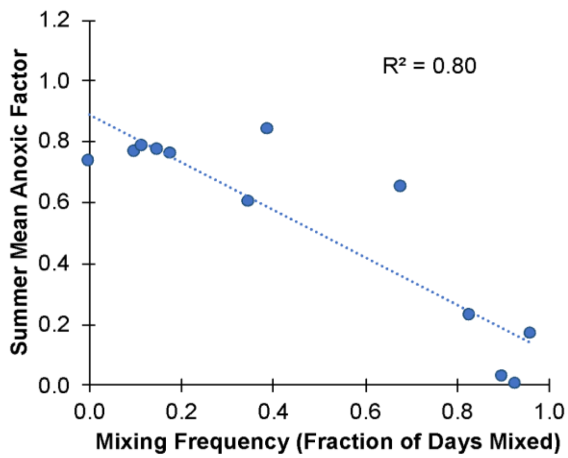
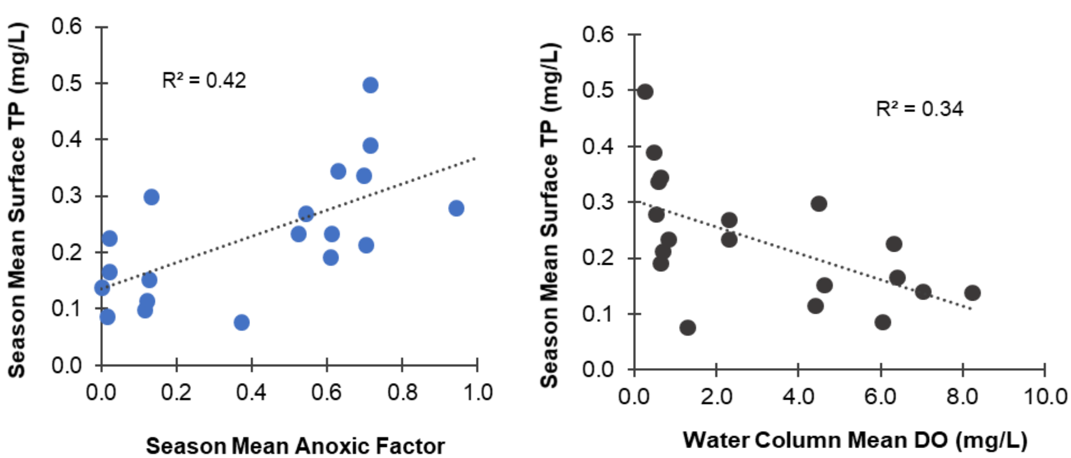


Figure 5.11 Mixing frequency, as a fraction of summer (June – Aug) days that a pond "mixed" (where mixing means the daily minimum top vs. bottom RTRM was less than the mixing threshold value for the pond), versus summer mean anoxic factor (n = 12; Dataset I).

5.3.1.10 Anoxic Factor and Dissolved Oxygen

Anoxic Factor (AF) emerged as one of the strongest indicators of pond surface water TP during the open-water season (May to Nov) (Figure 5.12 top panel). AF was also directly relevant to the magnitude and risk of sediment phosphate release due to the exposure of pond sediments to oxygen (or lack thereof) in mobilizing certain sediment phosphorus species to the pond water column (Figure 5.12 bottom panel). Similarly, water column mean dissolved oxygen concentration (mg/L) was negatively related to surface water TP (Figure top panel). This parameter is easier to calculate than AF but is potentially less directly related to the risk of sediment phosphorus release since it does not weight the profile by the exposure to pond sediment, as does AF.



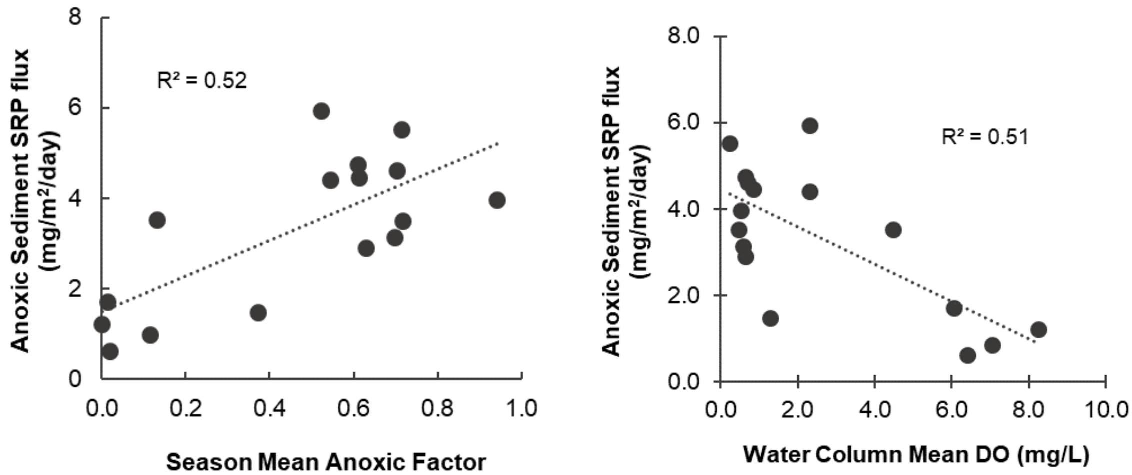


Figure 5.12 Top panel: Mean season (May-Oct) surface water TP (mg/L) vs. mean season anoxic factor and mean water column DO observed in the set of intensively monitored ponds (n=19; Dataset I). Each point represents one site, with data averaged over all seasons of data collection (typically one to five years during 2017-2021). Bottom panel: Anoxic phosphate release rate from sediment cores vs. season anoxic factor and mean water column DO observed in the monitored ponds (n = 19; Dataset III).

5.3.1.11 Sediment Phosphorus Release

A high rate of phosphorus release under anoxic conditions indicates that there is a risk for substantial internal phosphorus loading, especially in ponds that experience pervasive and persistent summertime anoxia. This is because the depletion in DO concentration and the resulting increase in sediment P release can result in an increase in pond phosphorus concentrations in the surface and hypolimnion. Ponds that remain highly oxygenated throughout the growing season and consist of organic-poor sediments are at much lower risk for internal loading. We observed a range of anoxic sediment phosphate release in the laboratory sediment cores of ponds in the TCMA (Figure 5.13), indicating the effects of different sediment characteristics on the pond sediment phosphorus release. The response to anoxia is related to the mobility of phosphorus in the sediments (i.e., concentrations of redox-sensitive P and labile organic P), which is mostly related to the external phosphorus input to the ponds over a number of years.

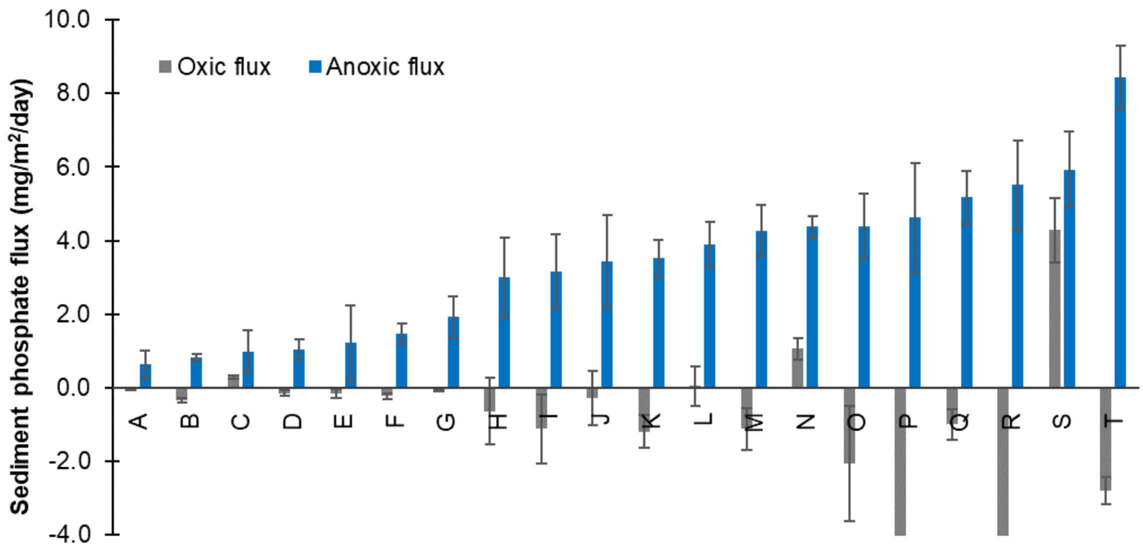


Figure 5.13 Mean anoxic and oxic sediment phosphate release rates measured in the laboratory sediment cores from a set of intensively monitored ponds (n =20; Dataset III). Error bars represent 67% confidence interval of the mean (3-5 cores per pond). Oxic phosphate flux for ponds P (-19.8) and R (-14.1) are beyond the Y-axis scale.

5.3.1.12 Sediment Phosphorus Fractions

The anoxic phosphorus release is well correlated with the biologically-available or releasable P fractions (i.e., loosely-bound P, iron-bound P, labile organic P) in the sediments (Figure 5.14), and thus appear to be important in playing the largest role with respect to phosphorus dynamics in urban ponds. The existing models for phosphorus release from lake sediments are based on the redox-P fraction (Nurnberg 1988, James 2011). The labile organic P, although mobilized at a slow rate only after microbial degradation occurs, was found at relatively high concentrations in some ponds (Janke et al. 2021). The risk of anoxic P release is greater in ponds that experience pervasive and persistent summertime anoxia.

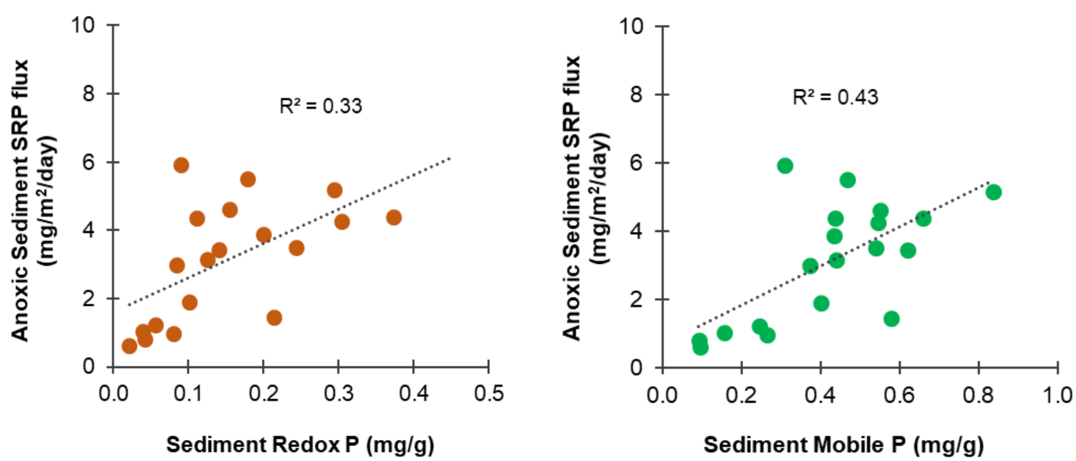


Figure 5.14 Relationship of sediment phosphorus composition of redox-P (loosely-bound P + iron-bound P) and mobile P (redox-P + labile organic P) with anoxic sediment phosphate release rates (SRP flux) observed in the monitored ponds (n =20; Dataset III).

5.3.1.13 Sediment Organic Matter Content

We found stronger relationship between pond sediment organic content and sediment labile organic P mass (that releases after bacterial degradation) than with anoxic P flux (Figure 5.15), suggesting a more indirect but important control over the release of phosphorus from the sediments under low oxygen conditions. Older ponds (> 31 years) are expected to have a higher accumulation of organic-rich sediments when compared to newer ponds (< 10 years).

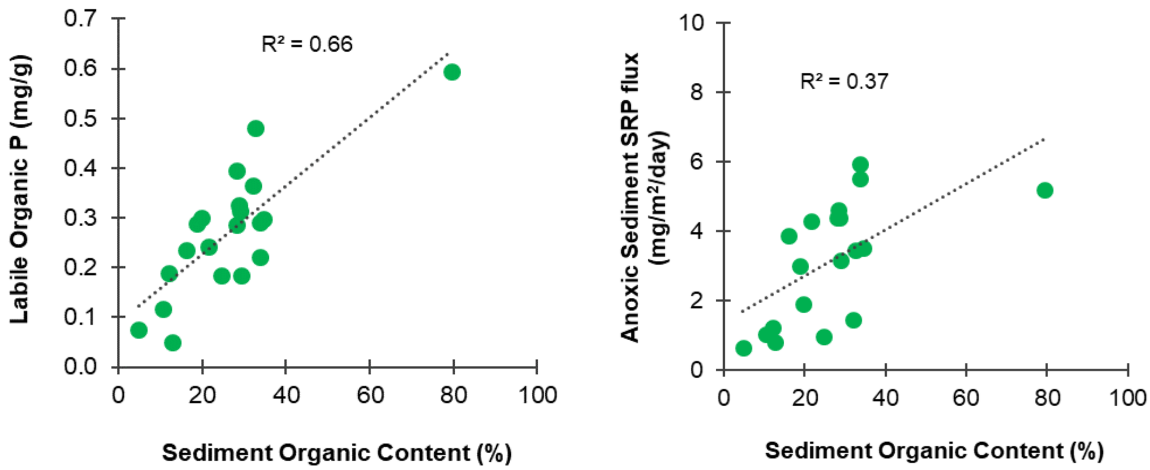


Figure 5.15 Relationship of sediment organic matter content with labile organic P mass in sediments and with anoxic sediment phosphate release rates (SRP flux) observed in the monitored ponds (n =19; Dataset III).

5.3.1.14 Sediment Metal Concentrations

We examined the effects of sediment iron concentration (in terms of total Fe: total P mass ratio) on the phosphorus release from pond sediments and found that a higher Fe: total P mass ratio was associated with lower P release and mobile P mass (redox-P + labile organic P) in the pond sediments (Figure 5.16).

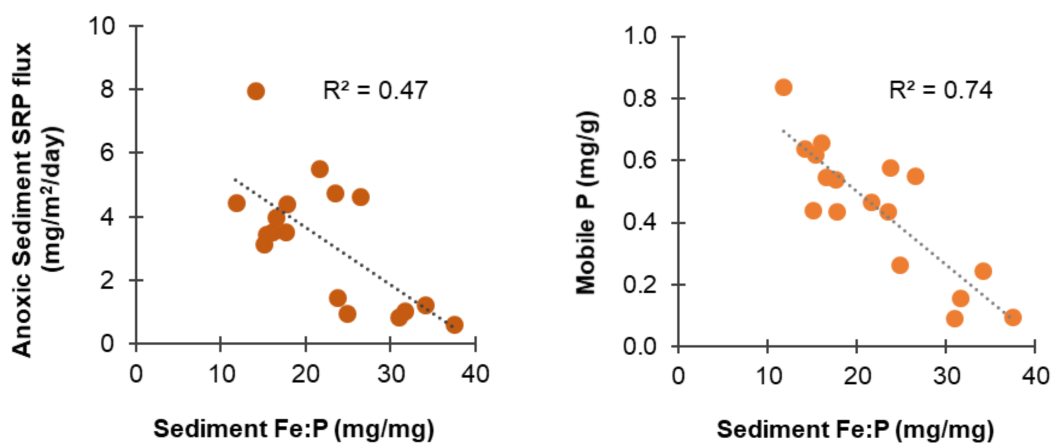


Figure 5.16 Relationship of sediment total iron (Fe):total phosphorus (P) mass ratio with anoxic sediment phosphate release rates (SRP flux) and mobile P mass in the sediments of monitored ponds (n =18; Dataset III).

5.3.2 Indicators not included in the Pond Assessment Tool (Chapter 6)

5.3.2.1 Shallow Water Body Classification

Surface water TP in the TCMA ponds (Dataset II-A) was also grouped by wetland classification in the Simplified Plant Community Classification (SPCC) system, which for the ponds in this data set, had nearly complete overlap with analogous categories in the Cowardin and Circular 39 systems (see Kloiber et al. 2019). The lowest mean TP was observed for the deeper and more open water category ("shallow open water community" in SPCC, corresponding roughly to Class 5 in Circular 39; n =133), at roughly 56% of mean TP of the "deep marsh" category in SPCC (corresponding roughly to Class 4 in Circular 39; n = 63) (Figure 5.17). The former class may be indicative of deeper, more pond-like systems with greater stratification and burial potential, with the latter class more likely to be vegetated and shallow. Note that the "uncategorized" water bodies (n = 37) likely correspond primarily to sites in the "seasonally flooded" and "shallow marsh" categories in the SPCC system but were excluded in this analysis due to the initial focus on water bodies with open water categories or that were permanently inundated. That this category has the highest mean TP may be due to the shallow, anoxic, and/or productive conditions likely present in these systems.

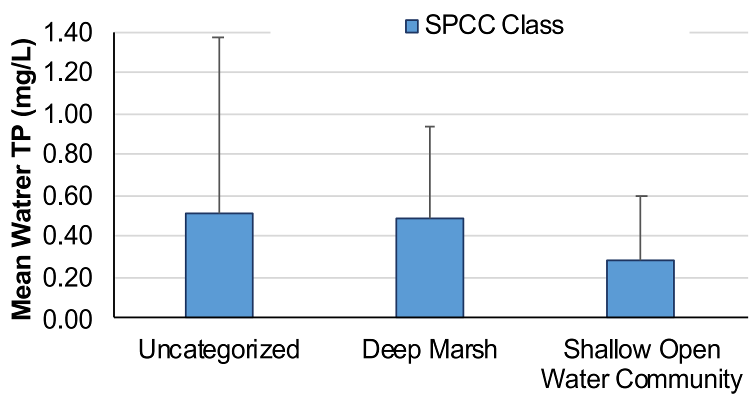
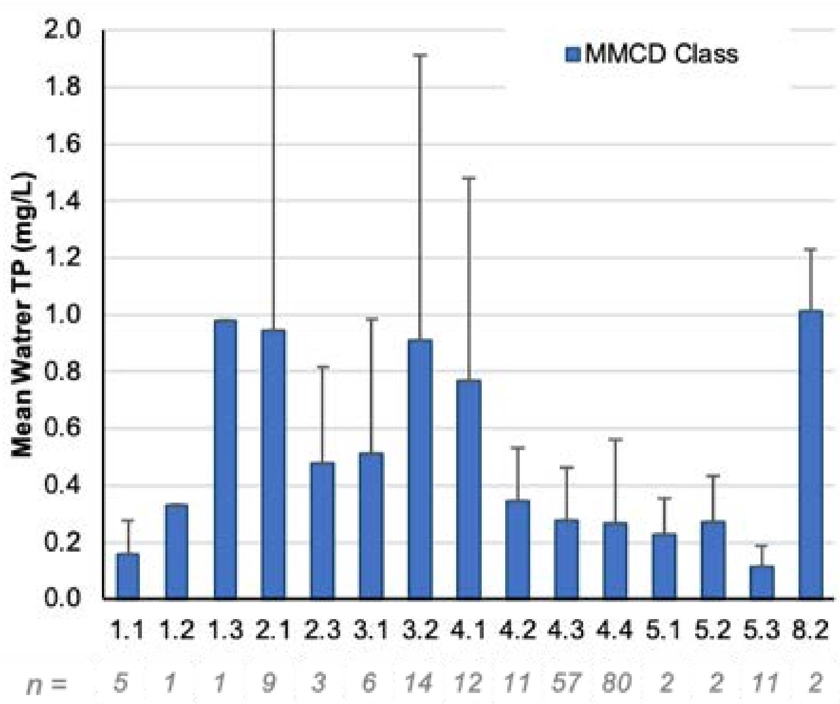


Figure 5.17 Surface water TP concentration (mg/L) as mean + SD, categorized by Simplified Plant Community Classification (n = 199; Dataset II-A). Note that most water bodies in the "uncategorized" class likely belonged to the "seasonally flooded" or "shallow marsh" classifications.

We also examined TP in the ponds as categorized by the system used by the Metro Mosquito Control District (MMCD). This system classifies water bodies and wet areas based on Circular 39 classification of water permanence, with further classification based on type of vegetation present in the ponds (see Section 0). Mean and standard deviation of TP are shown for these MMCD classifications in Figure 5.18. The highest TP concentrations were generally observed for Class 3 ponds (temporary water but rarely dry) in the Circular 39 classification, though this class had far fewer ponds than the Class 4 ponds (permanent water, shallow), which were most numerous. Within classes 3, 4, and 5, the more vegetated ponds (classes 3.1, 3.2, 4.1, 4.2, 5.1, and 5.2) tended to have higher concentrations than the open water categories (4.3, 4.4, 5.3), suggesting as in the previous analyses (Figure 5.6) that more vegetated ponds may produce conditions of high TP. Seasonal wetlands, or those categorized as having non-permanent water, also appeared to have higher concentrations of TP.



MMCD "X.Y" Class Codes

X	Circular 39 Classification
1	Temp Water, Usually Dry
2	Temp Water, Usually Waterlogged
3	Temp Water, Rarely Dries Out
4	Permanent Water, Shallow
5	Permanent Water, Deep
8	Bog
Y	**MMCD Vegetation Class***
1	Grass and sedge, open water < 50%
2	Mostly cattails, open water < 50%
3	Open water > 50%, vegetated fringe
4	Mostly open water, does not support fish

Figure 5.18 Surface water TP concentration (mg/L) as mean + SD, categorized by Metro Mosquito Control District. Classification shown in table on right side of plot; gray numbers at bottom are the number of sites that fell into each category. Note the standard deviation of Class 2.1 is 1.22 mg/L. *Vegetation classification descriptions are shown for the water body classes with permanent water (Classes 3 and 4, the most common in our dataset); different descriptions exist for the seasonal and bog classes.

5.3.2.2 Water Body Classification: Metro Area vs. Greater Minnesota

Using data obtained from the MPCA's statewide database of sampled water bodies (Dataset II-B; Table 5.2), we repeated the classification of water bodies and TP concentrations to provide statewide context for our metro dataset (Dataset II-A). Note that of the 222 water bodies in the MPCA database, 46 were located within the metro area; of these, there may be some overlap with our Metro dataset but this overlap should not be enough to influence the analysis.

Within the MPCA dataset, there appeared to be no appreciable difference among the metro-area and non-metro water bodies (Figure 5.19). Furthermore, very little difference was present in mean TP in the statewide (MPCA) water bodies among three categories in the Cowardin classification system, though the shallower water bodies (Class 4) and those with less permanent water (Class 3) tended to have slightly higher variability in TP than the deeper, open water bodies (Figure 5.19). These results contrast with our TCMA dataset (Dataset II-A), in which we observed higher and more variable TP in the more vegetated and shallower water body classes (Figure 5.6, Figure 5.18); also, mean TP was roughly 80% higher in our metro ponds than in the MPCA's metro ponds. Together these results suggest that the more urban and more shallow the water body, the higher the water column TP, though we also acknowledge the limits of this sort of broad categorization without more intensive knowledge of the water body sites.

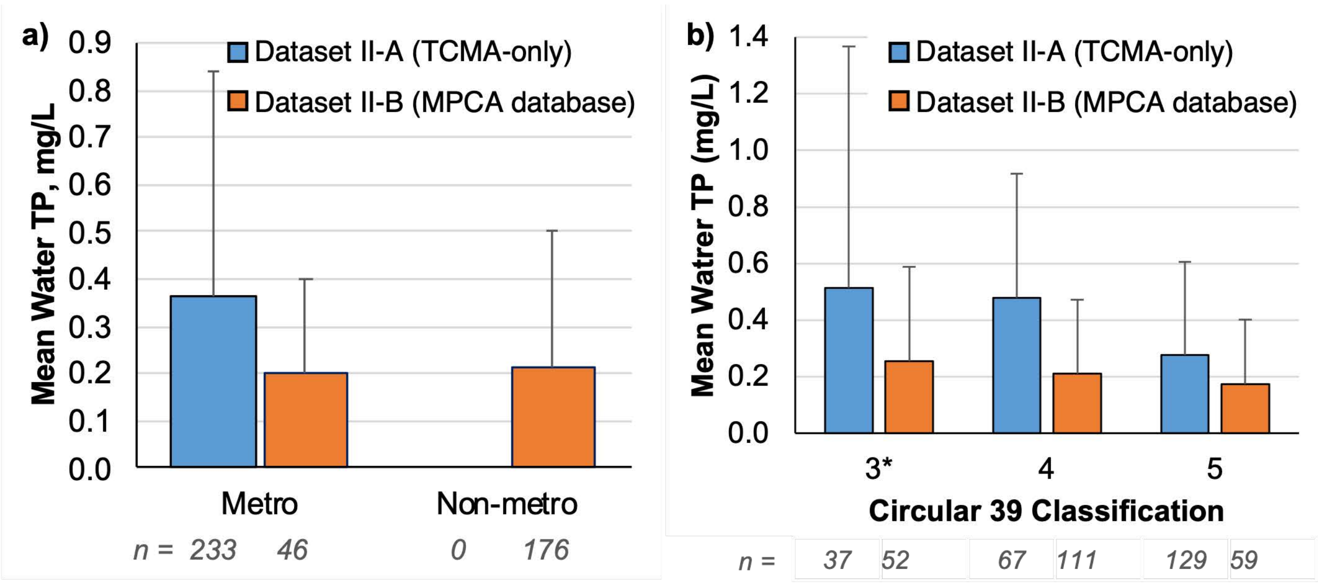


Figure 5.19 Comparison of surface water TP concentration (as mean + SD in mg/L) between the two water body datasets: the TCMA Task 3 Dataset and the MPCA's statewide water body database (see Error! Reference source not found.**). (a) Metro vs. non-metro water bodies in the two datasets, and (b) comparison of the two datasets based on Circular 39 classification. * For the TCMA dataset, we did not classify water bodies if they were not Class 4 or Class 5, but most of these uncategorized water bodies were likely in Class 3 (temporary water but rarely dry).**

The effect of ecoregion on mean TP in the metro vs. non-metro water bodies in the MPCA database (Dataset II-B) appears to be potentially important, as illustrated by a plot of mean TP based on EPA Type 4 Ecoregion (EPA, 2013[16]; Figure 5.20). There is variability among ecoregions, with lower TP generally observed for the less-organic moraine and outwash regions. However, while the metro water bodies tended to mostly fall into these moderate TP categories (which might suggest that ecoregion / hydrogeology does not explain the difference in our metro dataset vs. the MPCA dataset), 13 of the MPCA's metro ponds fell into the ecoregion (51i: Big Woods) with the second-highest mean TP of all ecoregions. We did not assess our TCMA dataset (Dataset II-A) for ecoregion, but a large number of our sites may be within this ecoregion (which includes parts of Hennepin and Scott counties), which may partly explain the higher TP observed in our TCMA Dataset vs. the MPCA database.

[16] https://www.epa.gov/eco-research/level-iii-and-iv-ecoregions-continental-united-states

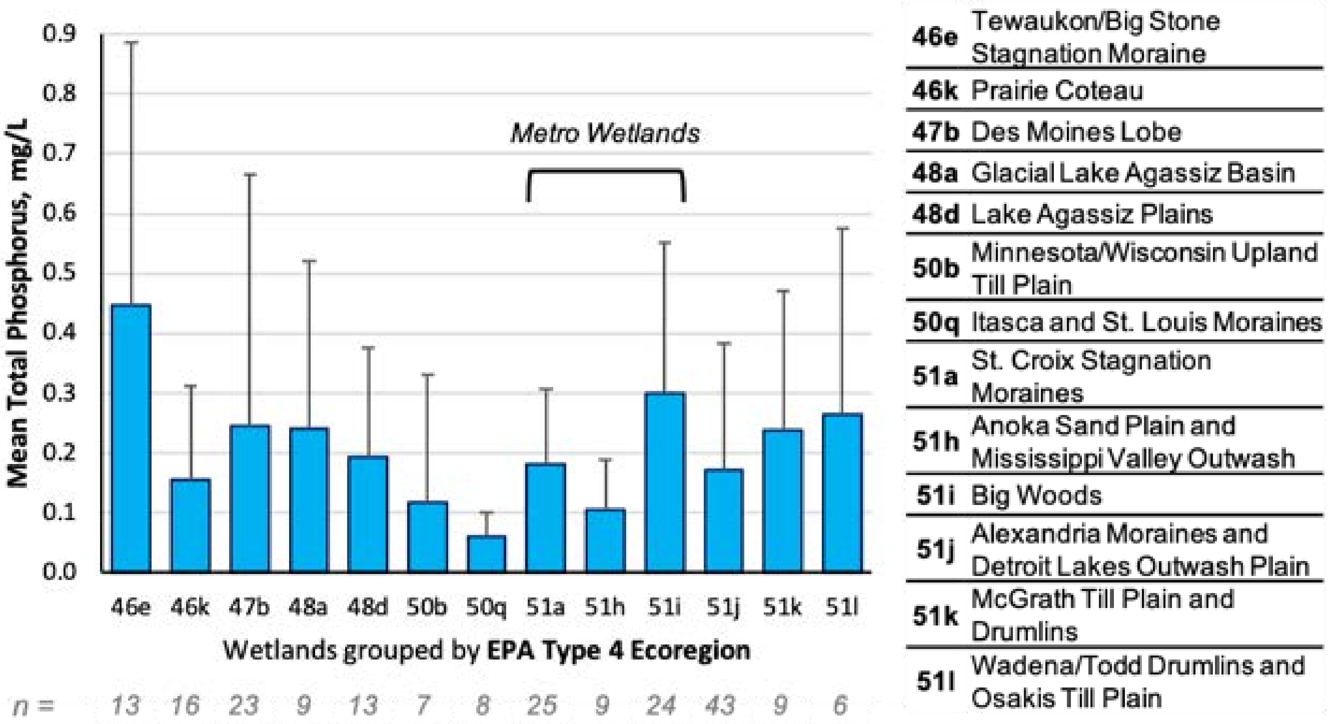


Figure 5.20 Surface water TP concentration (mg/L) as mean + SD by EPA Type 4 Ecoregion, in Dataset II-B (database of MPCA monitoring across the state). Ecoregion definitions provided at right. Metro-area water bodies fell primarily into the three ecoregions indicated (51a, 51h, and 51i), though these ecoregions extend beyond the metro and included some greater MN ponds as well.

5.3.3 Development of Regression Models to Predict Surface Water TP, Anoxia, and Sediment SRP Release in Ponds from Risk Indicators

In our previous work (Janke et al. 2021), we analyzed and reported the impacts of pond anoxic factor, duckweed cover, and sediment characteristics on the pond TP concentrations and sediment SRP flux in 14 intensively sampled and monitored ponds in Twin Cities. In that analysis, we applied single and multiple linear regression analysis to investigate sets of predictors for pond TP and sediment SRP flux. In this project, we have updated these regressions with data collected in 2020 and 2021 (for a total of up to 25 intensively studied pond sites, depending on analysis) to provide the models for a Pond Assessment Tool developed as a primary deliverable of this project (see Chapter Chapter 6:).

The approach to developing regression models is described in detail in Janke et al. (2021), but briefly, we began with a large set of physical, watershed, water quality, and sediment parameters that were expected to have some influence on pond water phosphorus concentrations and sediment phosphate release. These parameters are listed in Table 5.1. In Section 5.3.1, we also detailed the parameters that emerged as the most useful predictors of pond TP and sediment P release in simple linear regression or even just as categorical variables (like dominant land use). For the development of pond TP and pond oxygen models, the strong predictors identified were included in an exhaustive multivariate analysis using the 'glmulti' package in R, which performs ANOVA on all combinations of input variables and ranks models based on corrected Akaike Information Criteria (AICc). Models with inflated variance (variance inflation factor > 5) were thrown out, as were those in which any model coefficients were not significant at $p < 0.1$ or in which any included parameters were co-variate. For example, given high correlation between duckweed cover and anoxic factor, no models included both parameters. Given a relatively small sample size (n = 20-25), we also eliminated any models with more than 3 input variables due to over-constraint. We repeated this modeling exercise for summertime (June – Aug) TP, season (May – Oct) TP, summertime Anoxic Factor, season Anoxic Factor, and summertime dissolved oxygen. For pond sediment phosphate release model development, we adopted a similar approach and performed the multivariate analysis using the ‘regression’ package in MS Excel.

Ultimately, selection of models for inclusion in the Pond Assessment Tool was based on consideration of AICc and adjusted R^2 (in R) or adjusted R^2 (in Excel) versus similar models, with the goal of including models with input data types that could be acquired with a range of resource levels by local watershed management organizations. For the Pond TP and Pond Oxygen models, three “classes” of models are included: simple linear regression models (those with a single input parameter), multivariate models using site and watershed characterization parameters (generally higher uncertainty but simpler inputs), and multivariate models using more complex inputs (typically monitoring data) to provide the lowest levels of uncertainty. For Sediment Phosphorus Release, proposed models include simple linear regression models (observation data or sediment data input) and multivariate models that require either simple inputs (simple sediment characteristics) or complex inputs (detailed sediment analytical data). We also include models to estimate the complex inputs using simple sediment characteristics. The chosen models are shown in Table 5.3 below along with relevant model descriptions of goodness of fit.

We expected sediment variables (organic matter, sediment TP, P fractions) to impact pond P concentrations due to their relationship with sediment P release; however, the dataset analyzed (n = 20 ponds) showed weak correlations between observed surface TP and sediment variables considered (maximum linear $R^2 < 0.17$), and thus did not yield a strong model for estimation of pond surface TP using sediment parameters. It is likely that other parameters such as anoxic factor, pond depth, and duckweed cover have a much stronger and direct impact on pond P concentrations and pond oxygen status than the sediment parameters.

Table 5.3 Simple and Multiple Linear Regression Models for Surface Water TP (mg/L), Anoxic Factor, Water Column Dissolved Oxygen (mg/L),and Sediment Phosphate Release (mg/m^2/day) included in the Assessment Tool. Parameter definitions are included.

	Std Error	Adj. R^2	p-value
Summer (June - Aug) Mean Surface TP (mg/L)			
0.237*(AF) + 0.146	0.11	0.37	0.0021
0.253*(AF) + 0.068*(DMEAN/SAREA^0.5) + 0.290	0.11	0.41	0.0094
Season (May - Oct) Mean Surface TP (mg/L)			
0.0011*(DW) + 0.0020*(EMERG) + 0.154	0.081	0.44	0.0038
0.0015*(DW) - 0.077*(DMAX) + 0.290	0.082	0.42	0.0049
-0.0265*(SAREA) + 0.00259*(CPYCVR) + 0.00214*(EMERG) + 0.152	0.079	0.50	0.0018
0.202*(AF) - 0.0846*(DMAX) + 0.290	0.072	0.56	0.0080
0.150*(AF) - 0.0249*(SAREA) + 0.00219*(EMERG) + 0.183	0.062	0.67	0.0003
-0.0208*(DO_WC) + 0.00150*(EMERG) + 0.260	0.081	0.36	0.00095
0.049*(PFLUX_ADJ) + 0.13	0.084	0.47	0.0014
Summer (June - Aug) Anoxic Factor			
0.00802*(DW) + 0.0408	0.19	0.72	<0.0001
0.00824*(DW) - 0.058*(SAREA) + 0.129	0.18	0.63	<0.0001
Season (May - Oct) Anoxic Factor			
0.00789*(DW) + 0.055	0.18	0.76	<0.0001
-0.112*(DO_WC) + 0.735	0.12	0.87	<0.0001
-0.120*(DO_WC) + 0.00271*(CPYCVR) + 0.651	0.10	0.90	<0.0001
Season (May - Oct) Water Column Dissolved Oxygen (mg/L)			
-13.0*(WINDSH) + 4.12*(MIXED) + 12.1	0.86	0.89	<0.0001
1.51*(DMAX) + 7.57*(MIXED) - 3.67	1.11	0.82	0.0002
Anoxic Sediment Phosphate Release (mg/m^2/day; lab conditions)			
0.0359*(DW) + 1.06	0.94	0.72	1.81E-05
3.36*(AF) + 1.40	1.17	0.54	0.0012
-0.429*(DO_WC) + 4.4	1.22	0.51	0.0020
0.014*(OM) + 4.59*(SED TP) - 1.65	1.36	0.55	0.0007
12.5*(REDOXP) + 1.32	1.48	0.51	0.0004
6.85*(MOBP) + 0.413	1.60	0.40	0.0016
11.7*(REDOXP) + 2.14*(LABORGP) + 0.876	1.50	0.47	0.0017
REDOXP = -0.012*(FE:P) + 0.442	0.086	0.55	0.0007
MOBP = -0.024*(FE:P) + 0.973	0.11	0.74	9.73E-06
MOBP = 0.0058*(OM) + 0.393*(SEDTP) - 0.101	0.073	0.86	1.63E-08
LABORGP = 0.0068*(OM) + 0.0899	0.078	0.66	1.44E-05

Parameter Definitions	
AF	Anoxic Factor
DMEAN	Mean Depth, m
DMAX	Pond Max Depth, m
SAREA	Pond Surface Area, m^2
EMERG	Percent Pond Covered by Emergent Veg.
DW	Percent Pond Area Covered by Duckweed
CPYCVR	Percent of Pond Shore Covered by Canopy (LIDAR)
DO_WC	Water Column DO, Mean over Season (May - Oct), mg/L
PFLUX_ADJ	Adjusted Phosphate Flux in mg/m^2/day (Tool 3-B)
WINDSH	Observed Wind Reduction, pond vs. airport
MIXED	Fraction of Days Pond Mixed (based on RTRM)
OM	Sediment Organic Matter, %
SEDTP	Sediment TP, mg/g
REDOXP	Sediment Redox P, mg/g
MOBP	Sediment Mobile P, mg/g
LABORGP	Sediment Labile Organic P, mg/g
FE:P	Sediment Fe:P (mass ratio)

Chapter 6: Pond and Wetland Performance Evaluation Tool

In this chapter, we present the Pond Assessment Tool, a spreadsheet tool for assessing urban ponds for phosphorus water quality that is based upon the results of the Data Synthesis and Analysis of Phosphorus in Ponds task (Chapter 5). ***The overall goal of the Tool is to evaluate and apply specific risk indicators of high surface water total phosphorus (TP), anoxia, and internal phosphorus loading due to sediment phosphorus release.*** The tool included a set of models to predict the phosphorus risk indicators using available data and understanding of controlling processes. The tool is based on data collected as part of our previous work (Taguchi et al. 2018, 2020; Janke et al. 2021, Natarajan and Gulliver 2022), the current project, and related literature. The Tool was developed with flexibility so that it can be refined and enhanced as future information is available.

6.1 Pond Assessment Tool Description

The primary goals of the Pond Assessment Tool are:

1. To estimate pond surface water total phosphorus (TP) concentration and risk
2. To determine pond oxygen status (risk of anoxia)
3. To estimate pond sediment phosphate (P) release

The assessment process steps for the identified goals are illustrated in Figure 6.1. The tool includes several models to choose from based on input data availability. The inp*ut data types* are:

(i) *basic site data* on the pond and watershed that are typically readily available or accessible to the pond owner (drawings, plans, aerial photographs, site visits) or require a more intensive data collection method (GIS/Spatial data analysis);

(ii) *sampling data* that involve periodic grab sampling/profiling of pond water and sediment sampling; and

(iii) *monitoring data* that involve data collection using continuous monitoring stations or loggers installed in the pond.

Goal 1: Surface Water TP

The tool employs two levels of assessment for this goal, based on the type of input data. The first assessment (Tool 1-A) is a method of **screening** ponds for levels of indicators related to **risk** of high pond water TP. This assessment utilizes basic site data (such as duckweed cover or presence of hydric soils). The second assessment (Tool 1-B) provides **predictions** (numerical estimates) of pond phosphorus concentrations, using input of more detailed sampling and/or monitoring data that require periodic data collection at the pond sites along with basic site data.

Goal 2: Oxygen Status

The assessment of pond oxygen status (Tool 2) **predicts pond anoxia** in terms of anoxic factor, a measure of the extent and duration of exposure of pond sediments to low oxygen conditions, which can indicate risk of potential sediment phosphorus release. Anoxic factor can be estimated from sampling and monitoring data (dissolved oxygen profiles) and basic site data (pond bathymetry), or estimated from regression models incorporated into Tool 2. Tool 2 also provides estimates of water column dissolved oxygen from site or watershed characteristics. Tool 2 serves as an input to predictions of Goal 1 (Pond TP concentration) and Goal 3 (Sediment phosphate release).

Goal 3: Anoxic Sediment Phosphate Release

We propose two steps for the **prediction of phosphorus release** from pond sediments that contribute to internal phosphorus loading in ponds. The first step (Tool 3-A) provides models to predict potential anoxic sediment phosphate release as observed with laboratory columns of pond sediment and water, using sediment and water quality sampling data and basic site data (duckweed cover). Tool 3-A includes models that require input sediment parameters that are relatively simple to analyze in an analytical services laboratory (sediment TP, organic content, metal concentrations), as well as models that require input parameters from rigorous sediment analysis (redox-P, mobile-P). If a rigorous sediment analysis is not performed, models are provided to estimate the rigorous sediment parameters using simple sediment parameters as inputs. Given the variable oxygen status across ponds due to a multitude of factors, we recognize the importance of pond oxygen status on the in situ sediment phosphorus release rate and hence the internal load generated within the pond. The second step (Tool 3-B) adjusts the potential anoxic phosphate release (output from Tool 3-A) using pond anoxic factor (observed in field, or modeled in Tool 2) to predict the anoxic sediment phosphate release that might be observed under field conditions.

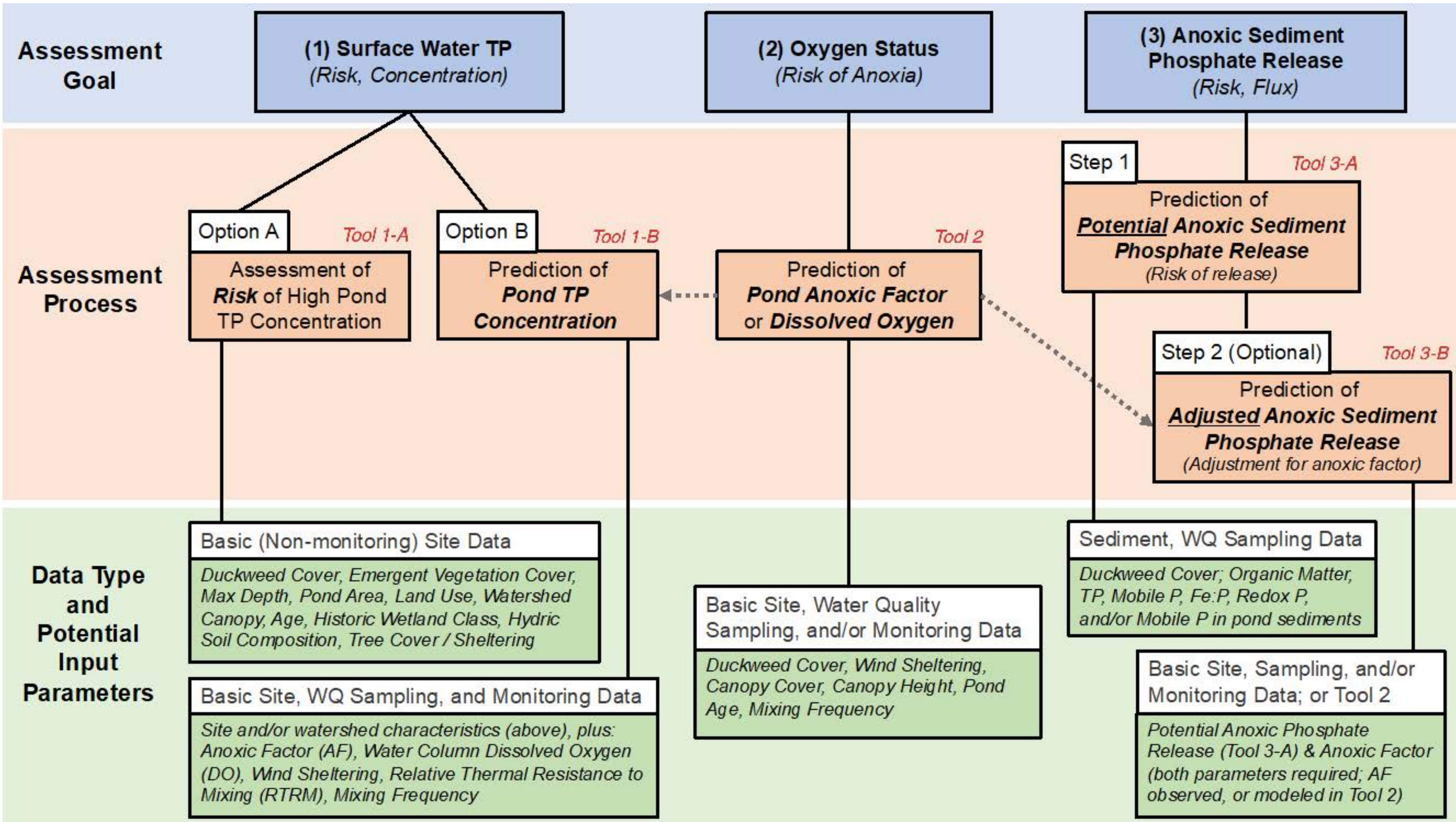


Figure 6.1 Flowchart of the Pond Assessment Tool showing the assessment goals, assessment process for each goal, and the potential input data parameters for the assessment process.

6.2 Reference Datasets

The assessment tool was developed using several urban pond sediment and water quality datasets, which have been collected and assembled by the authors in this and previous projects. These datasets included intensive monitoring and sampling efforts, widespread sampling surveys of ponds, as well as sediment sampling and coring for laboratory incubation studies of phosphorus release. The three primary datasets are described in the tool spreadsheet and in Section 0.

6.3 Limitations of the Current Version of the Pond Assessment Tool

The assessment tool represents a synthesis of the work completed over the last several years to better understand the drivers of phosphorus dynamics in ponds. An important outcome of that research has been the realization that ponds are incredibly complex systems, and while certain factors (such as duckweed cover, extent of anoxia, and sediment conditions) appear to have substantial influence on pond phosphorus cycling, we acknowledge that there are other drivers of which we have less understanding. Research on phosphorus retention in ponds is on-going, and future versions of the model can be informed by input from new studies.

We highlight several areas where our analyses may be limited. First, we encountered a sparsity of some types of data including recent bathymetry (needed for anoxic factor and mean depth calculations) and pond construction age. These were identified as important metrics for assessing pond condition but were not widely available in Dataset II. In addition, some of the variables not assessed in our work include submerged aquatic vegetation as well as overall impacts of vegetation on phosphorus cycling (assimilation and re-release after senescence) within the pond system. Submerged aquatic vegetation in particular is poorly documented in ponds, and is not easily assessed from aerial imagery or wetland classification systems as are free-floating vegetation (e.g., duckweed and watermeal) and emergent vegetation (rooted macrophytes such as cattails). Furthermore, while likely less common in small urban ponds than in shallow lakes ponds may also exist in a turbid phytoplankton-dominated state (Vitense et al. 2019), which would lead to high P even if certain risk indicators (anoxia, hydric soils, or shoreline canopy) were low.

Lastly, we point out that some collected data have not been analyzed in detail, and additional risk indicators may emerge from on-going data analyses and model development. In particular, the continuous monitoring datasets are large, and we've used these data to develop general descriptors of pond stratification and mixing dynamics, typically averaged over a season or summer. Work remains to investigate these data for evidence of shorter time scales of mixing and causes (e.g., runoff inputs, wind, or surface heat exchange). In addition, duckweed and watermeal have been sampled in the field and analyzed for TP concentrations, but these data have yet to be included in an estimate of "effective" surface water TP (i.e., a value that includes both water concentration and phosphorus bound up in duckweed).Thus, we emphasize that while the assessment tool does include some simple and multivariate models for prediction of pond phosphorus concentration and sediment phosphate release rates, these models should be considered approximate; most of the models still have considerable unexplained variance. To provide some context for results, we include goodness of fit and standard error for the models, as well as the ranges of values for the input parameters among the ponds used to develop the models. These latter values are intended to bracket the likely range of input values for which the models could be assumed "valid" without introducing an unknown and potentially large uncertainty. Lastly, while ponds with higher TP concentrations and higher rate of internal P release typically indicate that ponds are likely to be at risk of poor performance for P removal, a hydrologic analysis would be required to accurately assess a pond's actual P export to receiving water bodies (Janke et al. 2022). Such an assessment was beyond the scope of this tool, but a future version could include the use of water level data to help approximate a pond's hydrologic (water) balance and assess the risk of P export.

6.4 Description of Model Input Parameters

Table 6.1 describes parameters representing the most useful and strong predictors of pond surface water TP, anoxia, or sediment phosphate release, as determined by our analyses. The original list of potential indicators is presented in Table 5.1 (Chapter Chapter 5:), along with information on the purpose of these parameters, how they were determined for our analyses and their reference dataset(s), and the implication or impacts to pond surface water TP concentrations, oxygen levels, and sediment phosphate release. The strongest parameters from those analyses are used by the Assessment Tool, and include parameters used in both the "simple" (categorical) assessment of pond surface water TP (Tool 1-A) and in the predictive (regression) models of the other tool components for estimating surface water TP concentration (Tool 1-B), pond anoxic factor (Tool 2), and sediment phosphate release (Tools 3-A, 3-B). The models in the toolbox are shown in Table 5.3 along with relevant model descriptions of goodness of fit.

Note that several parameters (e.g., duckweed) are used in both the simple tool and the predictive tools. Note also that *not* all parameters are needed to use the tool; certain parameters (such as duckweed cover, pond age, and land use) may be adequate to provide some information and understanding of risks for high TP, anoxia, or internal loading in ponds.

Table 6.1 Potential input parameters to the Pond Assessment Tool, representing the strongest indicators of poor phosphorus retention in stormwater ponds and wetlands treating stormwater (high surface water TP, low dissolved oxygen, and high anoxic sediment SRP release).

Indicator	Potential Effect on P	Description
Anoxic Factor	Oxygen Dynamics, Sediment Release	Fraction of sediments exposed to anoxic overlying water during monitoring season
Shoreline Canopy Cover	Wind Sheltering/Oxygen Dynamics, Litter Inputs	Fraction of land in 25 m buffer around pond classified as canopy and/or buildings
Wind Reduction	Wind Sheltering/Oxygen Dynamics	Mean reduction in wind speed observed at pond relative to that at nearest airport
Mixing Frequency	Oxygen Dynamics, Vertical Transport	Fraction of days that pond was mixed, based on a minimum RTRM threshold
Free-floating plant (mainly duckweed)	Wind Sheltering/Oxygen dynamics, Organic matter inputs	Fraction of pond surface area that is covered by duckweed
Emergent Vegetation Cover	Oxygen dynamics, organic matter, mixing	Fraction of pond surface area covered by rooted, emergent macrophytes
Sediment TP	Sediment P Release	TP concentration in the upper 4 cm depth of sediments
Sediment organic matter	Sediment P Release	Organic matter concentration in the upper 4 cm depth of sediments
Sediment Fe:P	Sediment P Release	Total Iron to Total Phosphorus mass ratio in the upper 4 cm depth of sediments
Sediment Redox-P & Labile Organic P	Sediment P Release	Redox-P or Labile Organic P in the upper 4 cm depth of sediments
Pond age	Sediment Release, Organic matter accrual, Litter inputs	Pond age since construction or connection to storm drains relative to year 2021
Pond area	Oxygen Dynamics, Vertical Transport, Hydrology	Surface area of the pond
Maximum depth	Oxygen Dynamics	Maximum depth measured from the pond surface
Mean depth	Oxygen Dynamics	Mean of various depths measured from the pond surface
Land Cover	Phosphorus and sediment inputs to pond	Land cover (pavement, grass, canopy) in a 500 m vicinity of pond
Land Use	Phosphorus and sediment inputs to pond	Land use (residential, commercial, other) in 500 m vicinity of pond

6.5 Assessment of Benefits and Costs Associated with Pond Monitoring

The Pond Assessment Tool has the ability to provide a low-cost method to assess risk (and provide estimates) of high water column phosphorus, sediment phosphate release, and anoxia in ponds treating stormwater using readily-available spatial, water quality, and pond data. We will compare the cost savings arising from utilizing the Pond Assessment Tool components for evaluating pond conditions vis-à-vis full-scale field monitoring or grab sampling to assess pollutant treatment performance. Three methods of evaluating pond phosphorus water quality have been considered and they vary in the required effort and scale of data collection, analysis, and information obtained.

Method 1: Inflow and outflow monitoring of ponds to estimate phosphorus mass retention (i.e., input – output) in the pond. This method is the most intensive monitoring method and yields pollutant concentrations and mass loads for estimating metrics such as pollutant removal efficiency, and hydrologic loading (and retention) of the pond. Supplementing the event monitoring with loggers installed within the pond will yield information on the pond conditions (stratification and water level) between rainfall events.

Assumptions: 15 rainfall events are sampled each year to collect complete composite water samples at the inlet and outlet for each event, and the samples are chemically analyzed for pollutant concentrations (TSS, TP, DP, DN, chloride); monitoring ideally is for three years to account for seasonal- and climate-driven variations in rainfall and phosphorus loading at the site. A continuous monitoring station, to record water level, temperature, and dissolved oxygen concentration, is also installed in the pond.

Estimated cost: For a pond site that has 1 inlet and 1 outlet, the cost of monitoring inflows and outflows in a pond is estimated to be $95,500 in the first year, with increased resources needed when more inlets or outlets are present. The cost is for purchase of automated flow measurement equipment and continuous monitoring station loggers, labor for installation and maintenance of equipment, collection of water samples during flow periods, and chemical analysis of the samples collected. In the second and third year, expenses will primarily be for water sample collection and chemical analysis and estimated to be $43,500 per year (Costs estimated by William Selbig, USGS Upper Midwest Water Science Center, Madison, WI). The total cost for three-year of monitoring a hydrologically simple pond is $182,500.

Limitations: The P mass retention (or release) and between-event pond DO levels can indicate the potential internal phosphorus loading in the pond, and the current functioning of the pond for pollutant removal. To confirm and quantify the internal P load, however, either lab studies on sediment phosphorus release rate, or additional analysis of the sediment quality (organic content, total phosphorus and metal levels) need to be performed. The sediment work (cost provided under Method 2) will yield independent estimates of internal loading in the pond.

Method 2: Grab sampling to monitor the in situ water quality conditions including sediment sampling to estimate internal phosphorus loading. This method will yield data on pond phosphorus concentrations, pond oxygen and mixing status, and pond sediment phosphorus characteristics, which are helpful for understanding the risk for poor phosphorus water quality conditions.

Assumptions: Regular visits to the pond site during the growing season (2 site visits per month from April to October, 14 visits total) to collect surface and bottom water quality data (dissolved oxygen, temperature, conductivity profiles, and concentrations of phosphorus species of total P, total dissolved P, soluble reactive P) and chloride; pond sampling will be done for three field seasons to account for seasonal variation in rainfall and phosphorus loading at the pond. A continuous monitoring station to record water level, temperature, and dissolved oxygen level, is also installed in the pond. Sediment sampling is done one time to collect five sediment cores for laboratory phosphorus release study and chemical analysis of sediments (upper 10 cm depth) to characterize sediment phosphorus fractions, organic matter content, and metal concentrations.

Estimated cost: The cost of regular grab sampling at a pond will be about $17,000 in the first field season; the cost includes field probes, continuous monitoring station loggers, labor for installation and maintenance of equipment, bi-weekly data collection, and chemical analysis of the samples collected. In the second and third year, expenses will primarily be for water sample collection and chemical analysis, estimated to be $13,000 per field season. For sediment analysis work, the $8,700 expense encompasses the sediment phosphorus release rate study, sediment P chemistry analysis, and metal analysis. The total cost is for three years is estimated to be $51,700.

Limitations: The treatment performance of the pond (the export of phosphorus) during flow periods is not known without inflow-outflow monitoring. Additional analysis of the water balance or at a minimum, water level data, are needed to estimate hydraulic retention. Laboratory sediment P release data are needed with the pond's DO regime to determine if the pond has a net retention (or release) of phosphorus (Janke et al. 2021, 2022).

Method 3: Low-cost monitoring informed by the Pond Assessment Tool. This method involves two steps: first, application of the Tool to screen and prioritize pond sites to monitor for water quality and other parameters (using Tool 1-A), and second, collection of field data required as inputs to apply models in the tool (Tools 1-B, 2, 3). The data input to the models are parameters that can be collected with relatively less effort, such as water column DO concentrations, and simpler to measure by any analytical services laboratory (sediment TP, organic content, metal concentrations). Note that tool components for pond TP, oxygen status, and sediment P release (Tools 1-B, 2, 3) have models that use basic site data as inputs (e.g., pond depth, pond area, duckweed cover) which can be derived from pond drawings and aerial imagery without the need for field data collection. Still, field measurements (pond DO, sediment) enable the use of models with lower uncertainty and are recommended when parameters are available or easy to gather.

Assumptions: It is assumed that the user has access to the basic site data (pond drawings, plans, aerial photographs) required to run the Screening tool (Tool 1-A). First, the user will screen multiple pond sites for the identified risk factors and use the results to create a priority list that targets ponds at the highest risk for poor phosphorus water quality. Then, field sampling at the selected pond sites is carried out to collect the limited data necessary to run the models. To use Tool 1-A and Tool 2, at least 5 site visits are required during the growing season to collect DO and temperature profiles and note floating plant coverage. We also include installation of a continuous monitoring station to record water level, temperature, and dissolved oxygen level, in the pond. The field monitoring provides data necessary to calculate anoxic factor (a measure of sediment exposure to oxygen, which is highly relevant to sediment P release). To use Tool 3-A, sediment sampling is done once; bulk sediment (say, samples from five locations within the pond composited into a single sample) will be analyzed for sediment TP, organic matter content, and metal concentrations at a certified laboratory.

Estimated Cost: There is no cost for collecting basic site data, as they are generally readily available to the user. The cost for field data collection is estimated to be $6,400 for one field season, which is for field probes, continuous monitoring station loggers, and labor for data collection. If water samples are also collected during site visits, chemical analysis of the samples will cost an additional $3,100 (which is not included in the $6400 estimate earlier). Sediment collection and analysis cost is estimated to be $600. Data collection from one field season can provide the inputs needed for the Pond Assessment Tool. Future monitoring will cost $1,650 per season (excluding water sample analysis). These costs would only be required if the pond is deemed to be at risk for high phosphorus concentrations of sediment phosphorus release by Tool 1-A. Roughly 50 to 70% of the ponds will likely be eliminated by being classified as low risk for internal loading of phosphorus and high phosphorus concentrations. The total cost is estimated to be $13,400 per pond fully assessed.

Limitations: The treatment performance of the pond (the export of phosphorus) during flow periods is not known without inflow-outflow monitoring. Results and interpretations are dependent on the available data and data inputs, and the uncertainty associated with the models in the Pond Assessment Tool. The net retention (or release) of P mass is still dependent on the hydraulic retention in the pond, the determination of which requires additional analysis of the pond water level data (e.g., Janke et al. 2021, 2022).

6.6 Tutorial for the Pond Assessment Tool

We have provided a tutorial showing the application of the Pond Assessment Tool for the Alameda Pond, located in Roseville, MN, as an example. The tutorial, provided in Appendix C, illustrates the evaluation of Alameda Pond for high phosphorus risks depending on data availability for Tools 1, 2, and 3. The tutorial includes guidelines on the methods of data collection and/or extraction for model input.

Chapter 7: Conclusions and Recommendations

Stormwater ponds and wetlands that treat stormwater are not always operating as designed or as expected with regard to phosphorus retention. This is because many ponds are stratified at 0.5 m (1.6 ft) or less, creating conditions for low dissolved oxygen concentration over the sediments and sediment phosphorus release. This report enables the assessment of risk factors with regard to sediment phosphorus release and resulting high phosphorus concentration in these ponds. The major findings and potential use of the products developed in this project are summarized below:

1) A working definition of the various wet stormwater management practices, specifically, constructed stormwater ponds and wetlands treating stormwater is contained in Chapter 2 of this report. This a stand-alone document that describes the wetland definitions and the regulatory processes around stormwater ponds and wetlands treating stormwater, providing further clarification of important distinctions that affect the management of small water bodies that treat stormwater.
2) The field data and laboratory column studies (Chapters 3 and 4) completed as a part of this project filled gaps in the understanding of sediment phosphorus release and high phosphorus concentration risk factors in ponds, and were also important in informing and developing the Pond Assessment Tool.
3) The meta-analysis of existing pond data (Chapter 5.2) as it related to phosphorus concentrations of approximately 230 stormwater ponds and wetlands that treat stormwater in the Twin Cities metropolitan area over approximately the past 15 years were aggregated, reviewed, and synthesized into a form helpful for project objectives and were important in informing and developing the Pond Assessment Tool (Chapter 6).
4) Using the new field data, laboratory column studies, meta-analysis of existing pond data, and data for 15 intensively-monitored ponds collected during previous projects (Taguchi et al. 2018, 2020; Janke et al. 2021; Natarajan and Gulliver 2022), we further strengthened the existing relationships between phosphorus conditions and important risk factors (pond age, pond depth, tree cover, vegetation cover, watershed/land use type, sediment characteristics, environmental and climatic conditions), and enabled the development of the Pond Assessment Tool.
5) The Pond Assessment Tool, developed to identify and evaluate indicators of functionality (or failure) with regard to phosphorus retention, incorporates the broad patterns established from data analysis of 230 water bodies to form the basis of the screening tool as well as a statistical analysis of field data that provided regression models for predicting pond TP concentration, pond anoxia, and sediment P release. The Pond Assessment Tool (Chapter 6) is designed to be a stand-alone spreadsheet tool, complete with a user's guide.
6) The Pond Assessment Tool can help organize the assessment of ponds for risk of sediment phosphorus release and high phosphorus concentration. Its use can enable cost-effective management of constructed stormwater ponds and wetlands treating stormwater for optimal treatment performance, will improve runoff quality and allow the allocation of resources for other environmental projects (Chapter 6).

7) The project was complemented and supported with information from a recent Local Road Research Board (LRRB) funded project “Wet Pond Maintenance for Phosphorus Retention” (Taguchi et al. 2022) in which maintenance measures to limit phosphorus release in ponds were investigated. This enabled a review and update to the sections on constructed stormwater ponds (Appendix D) incorporating maintenance activities to reduce sediment release of phosphorus for two existing LRRB documents: Stormwater Maintenance Best Management Practices Resource Guide (Marti et al. 2009), and Decision Tree for Stormwater BMPs (Marti et al. 2011).
8) Updates to the report “Stormwater Maintenance Best Management Practices Resource Guide” were developed for only the sections that are related to stormwater ponds and wetlands treating stormwater (Appendix E). A separate LRRB project, “Updating the 2009 Stormwater BMP Maintenance Resource Guide,” funded from July 1, 2022 to June 30, 2024, is designed to update the entire guide to incorporate over a decade’s worth of knowledge gained, new stormwater BMPs and management techniques that have been developed and refined, and to address revisions to regulations that govern stormwater runoff in the State of Minnesota. The sections on stormwater ponds and wetlands treating stormwater will be included in this update.

7.1 Recommendations

Based upon the finding and products summarized above and documented in this report, and the knowledge gained in this project, we make the following recommendations:

1) There is some concern about the regulatory perspective on stormwater ponds versus wetlands that treat stormwater. **We recommend that a pamphlet that documents these definitions and regulatory perspectives be developed from Chapter 2 of this document.**
2) The Pond Assessment Tool is designed to be helpful for managers and other professionals who are responsible for maintaining ponds. It will help them improve phosphorus retention in ponds. **We recommend that a separate document be developed from the Pond Assessment Tool and the User’s Guide to the Pond Assessment Tool for managers to use in their goal of improving phosphorus management.**
3) The stormwater pond update will also be incorporated into a report for a project, Updating the 2009 Stormwater BMP Maintenance Resource Guide, for the LRRB. **We recommend that major dissemination of the sections updating guidance on stormwater ponds and wetlands treating stormwater be deferred until the update of the 2009 document is complete, June 30, 2024.** The portion of Appendix E that incorporates the Update to the Stormwater Pond 2009 Stormwater Maintenance BMP Guide can be made available to practitioners who request it, or can be downloaded from this report.
4) **Based on the project findings, we recommend few substantial changes to the 2011 LRRB report “Decision Tree for Stormwater BMPs,”** because the document is related to BMP selection in new installations, rather than maintenance of existing installations. If the LRRB would like to update this document, a separate project is recommended.

References

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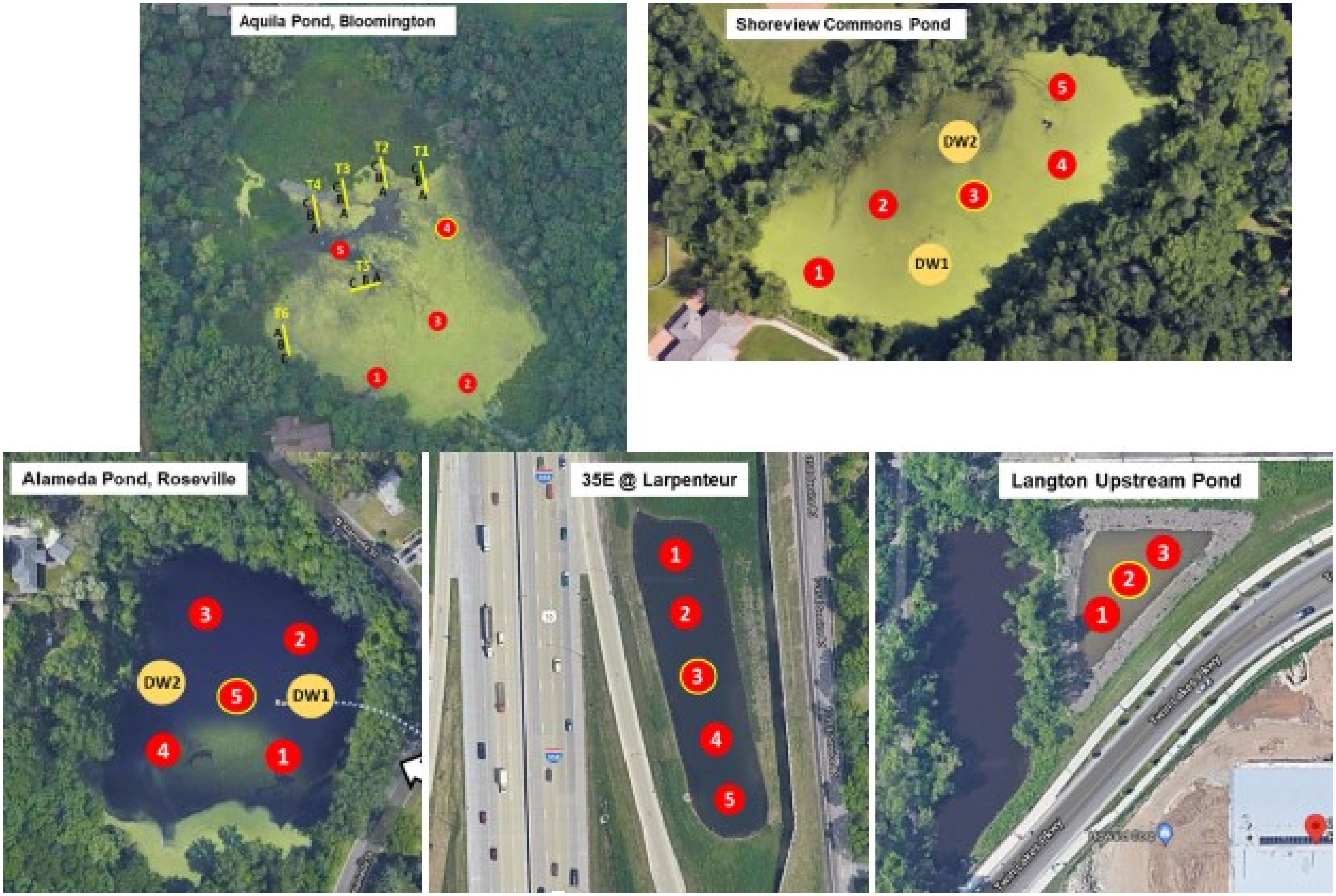
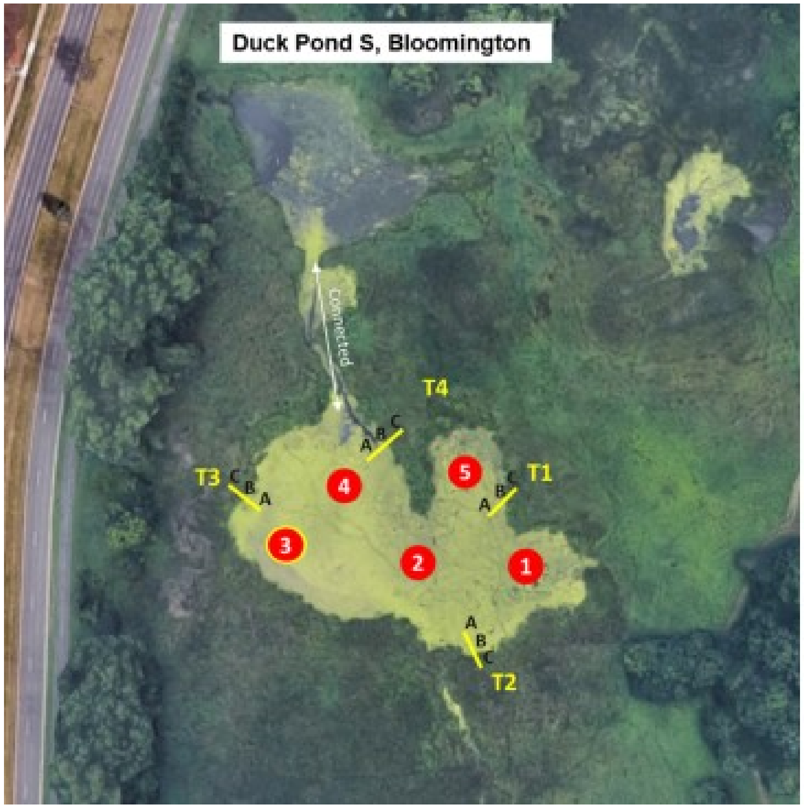
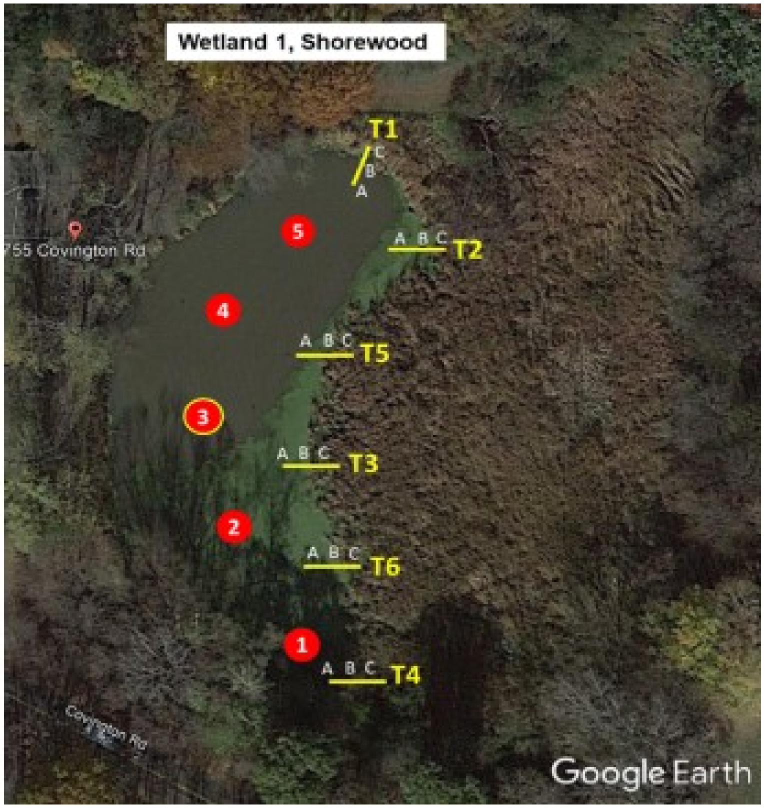
Appendix A
Field Sampling Photos and Data from 2020 and 2021



Aquila Pond, Bloomington
T1
T2
T3
T4
T5
T6
A
B
C
1
2
3
4
5
Shoreview Commons Pond
DW1
DW2
1
2
3
4
5
Alameda Pond, Roseville
DW1
DW2
1
2
3
4
5
35E @ Larpenteur
1
2
3
4
5
Langton Upstream Pond
1
2
3


Duck Pond S, Bloomington
Connected
T4
A B C
T3 C B A
4
5
A B C T1
3
2
1
A B C
T2


Wetland 1, Shorewood
T1
C B A
5
A B C T2
755 Covington Rd
4
A B C T5
3
A B C T3
2
A B C T6
1
A B C T4
Covington Rd
Google Earth

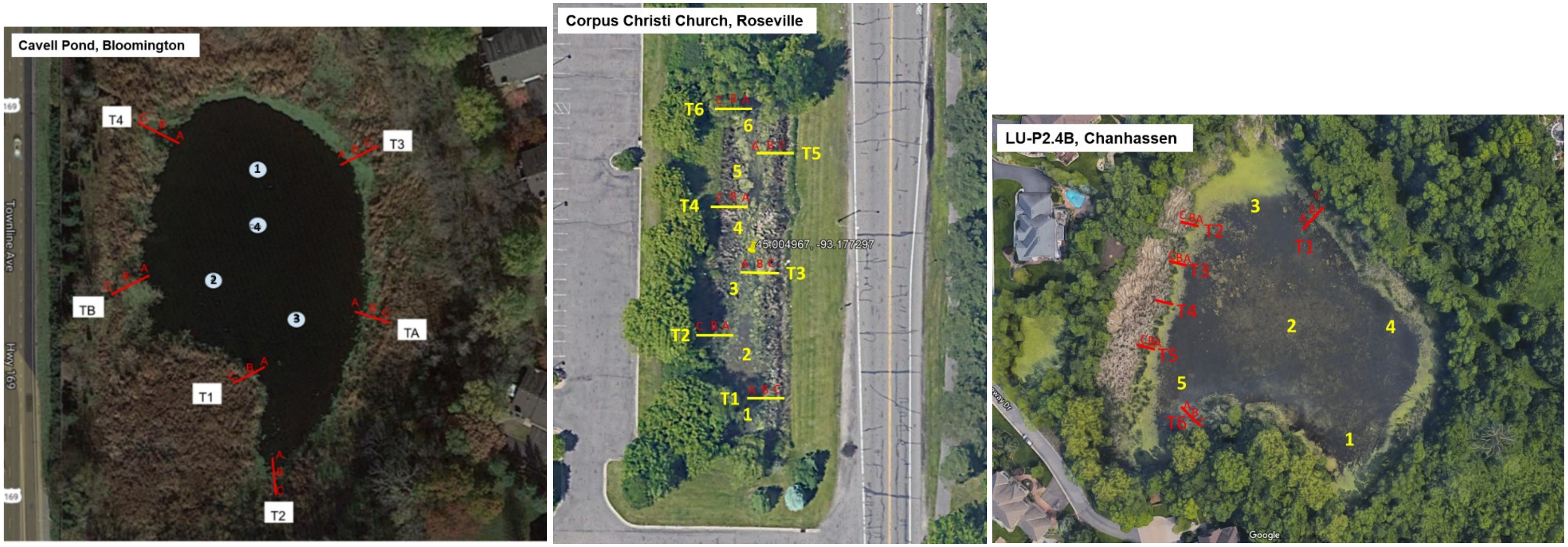
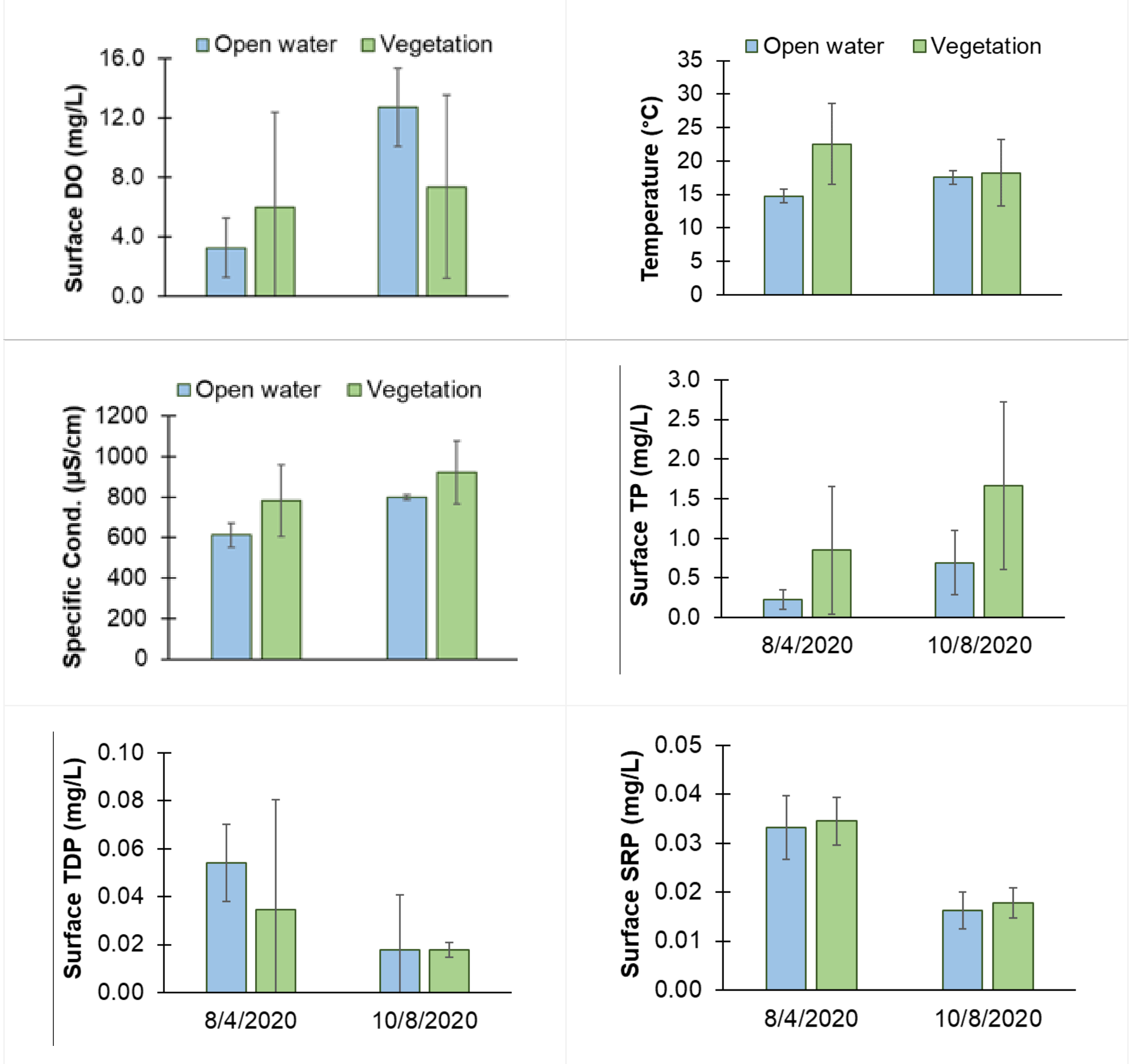


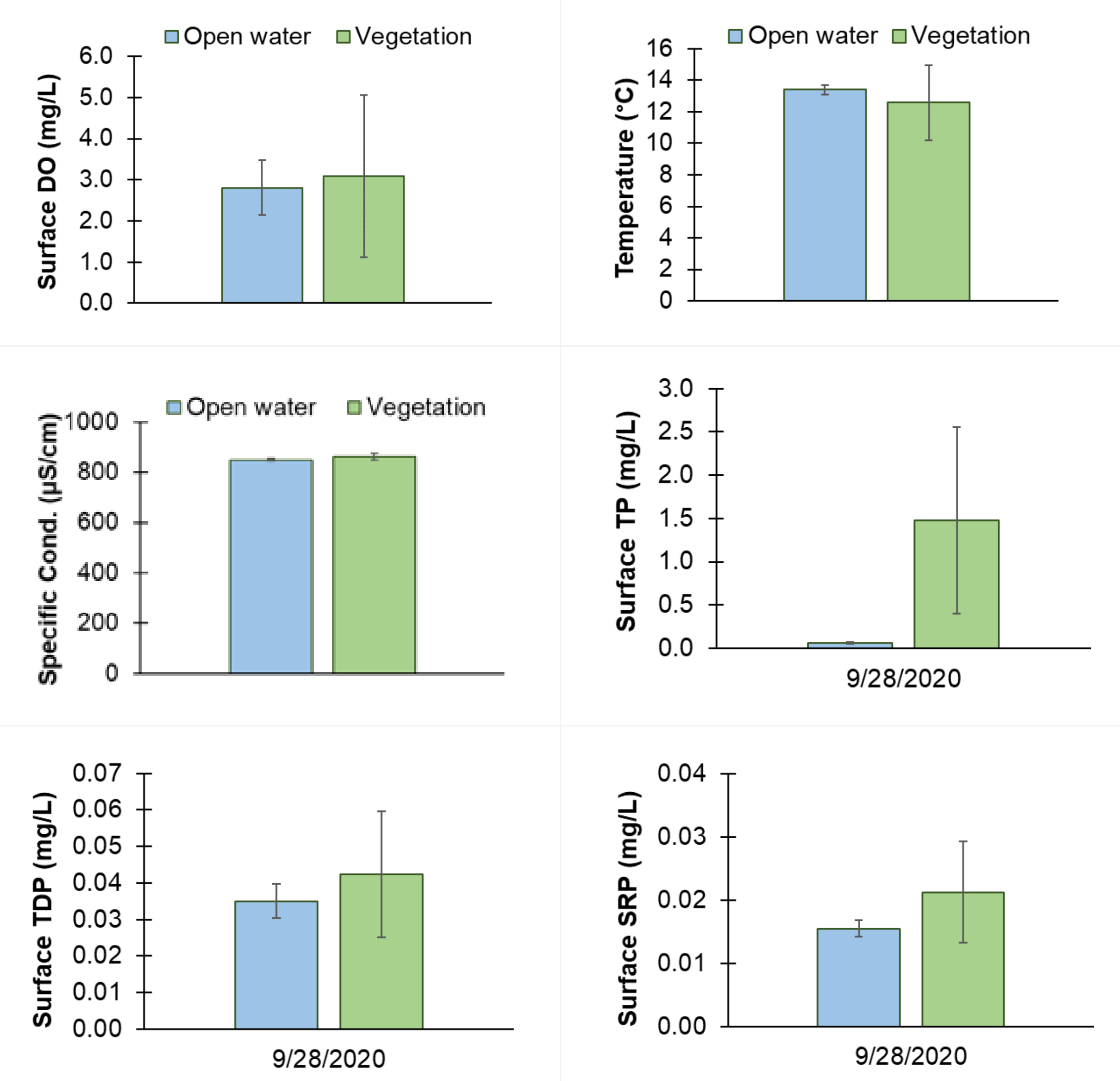
Figure A.1 Maps showing the water sampling locations in Aquila Pond, Shoreview Commons Ponds, Alameda Pond, 35E/Larpenteur, Langton Pond, Duck Pond S, Wetland-1, Cavell Pond, Corpus Christi Church, and LU-P2.4B. Locations 1 to 5 are in the open water, where the deepest location (highlighted with yellow 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 the ponds with emergent vegetation (Duck Pond S, Wetland_1, Aquila, Cavell Pond, Corpus Christi Church, and LU-P2.4B), additional sampling was done in the vegetated area along 4-6 transects (T1 to T6) with three locations (A, B, C) sampled per transect. In Shoreview and Alameda ponds, Locations DW1 and DW2 were for duckweed sampling in addition to Locations 1-5.

Aquila Pond

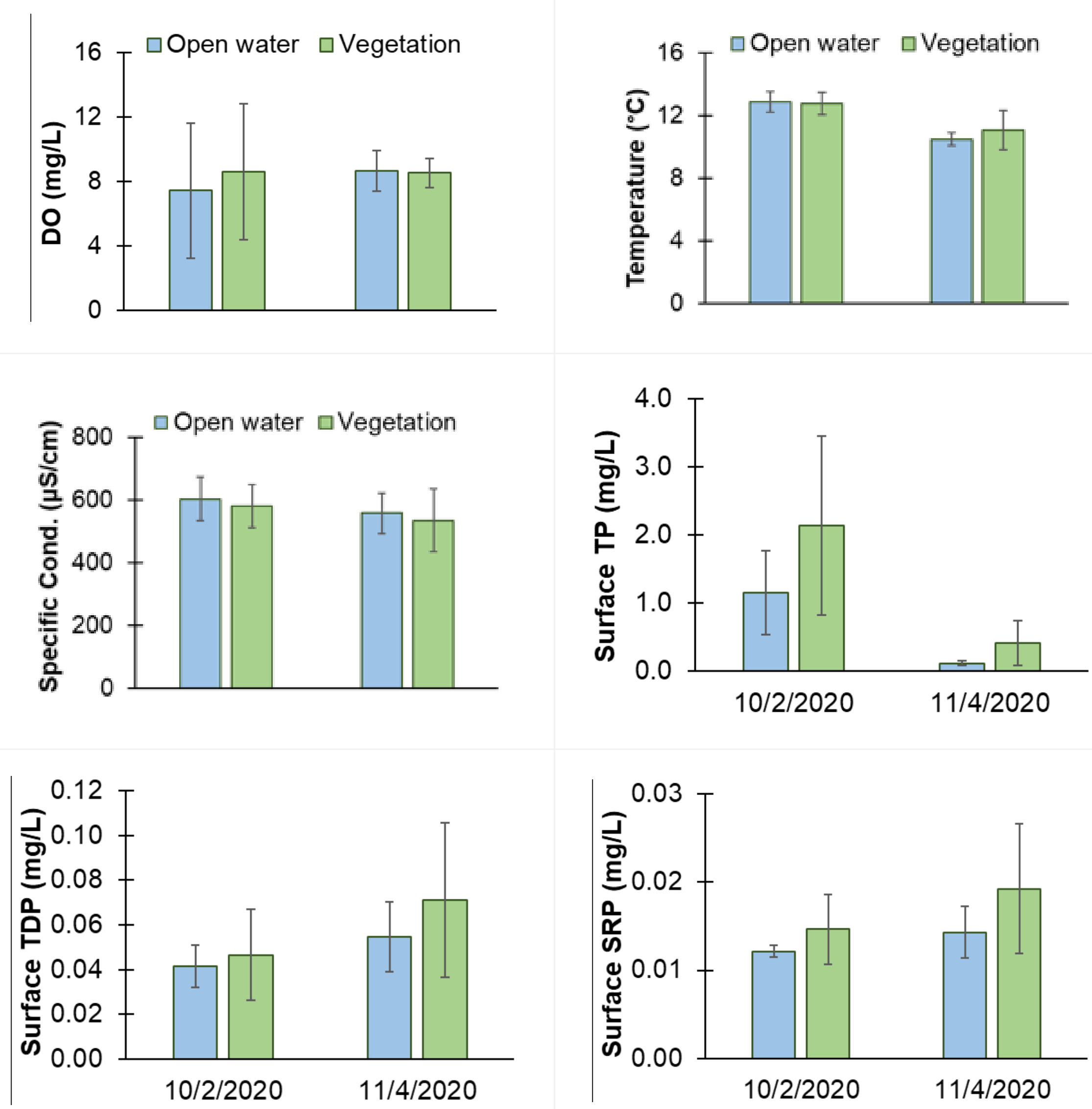


Open water
Vegetation
Surface DO (mg/L)
16.0
12.0
8.0
4.0
0.0
Temperature (°C)
35
30
25
20
15
10
5
0
Specific Cond. (µS/cm)
1200
1000
800
600
400
200
0
Surface TP (mg/L)
3.0
2.5
2.0
1.5
1.0
0.5
0.0
8/4/2020
10/8/2020
Surface TDP (mg/L)
0.10
0.08
0.06
0.04
0.02
0.00
Surface SRP (mg/L)
0.05
0.04
0.03
0.02
0.01
0.00

Cavell Pond

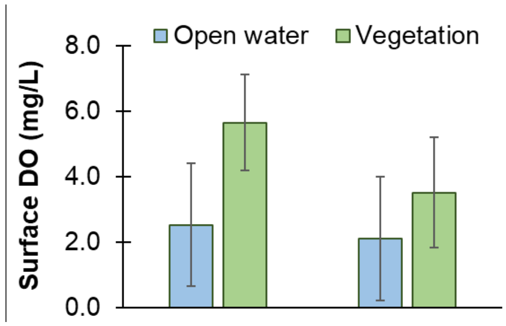

Open water
Vegetation
Surface DO (mg/L)
6.0
5.0
4.0
3.0
2.0
1.0
0.0
Open water
Vegetation
Temperature (°C)
16
14
12
10
8
6
4
2
0
Open water
Vegetation
Specific Cond. (µS/cm)
1000
800
600
400
200
0
Surface TP (mg/L)
3.0
2.5
2.0
1.5
1.0
0.5
0.0
9/28/2020
Surface TDP (mg/L)
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0.00
9/28/2020
Surface SRP (mg/L)
0.04
0.03
0.02
0.01
0.00
9/28/2020

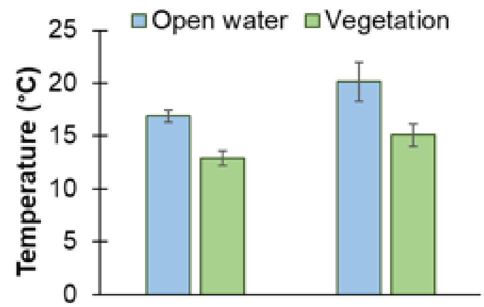
Duck Pond S

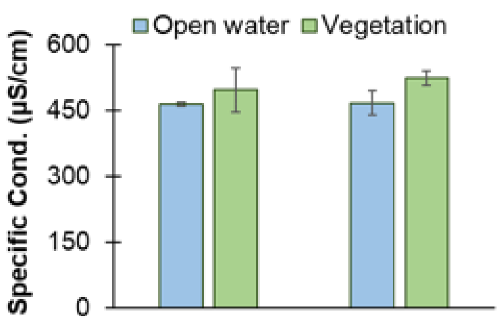


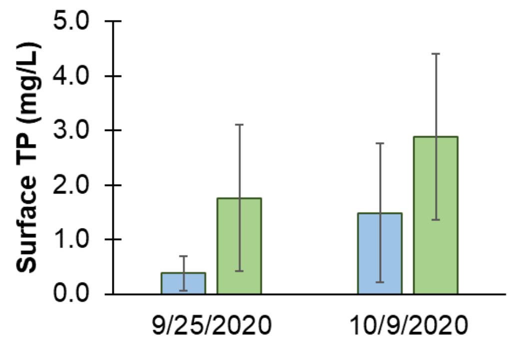
Open water
Vegetation
DO (mg/L)
16
12
8
4
0
Temperature (°C)
16
12
8
4
0
Specific Cond. (µS/cm)
800
600
400
200
0
Surface TP (mg/L)
4.0
3.0
2.0
1.0
0.0
10/2/2020
11/4/2020
Surface TDP (mg/L)
0.12
0.10
0.08
0.06
0.04
0.02
0.00
Surface SRP (mg/L)
0.03
0.02
0.01
0.00

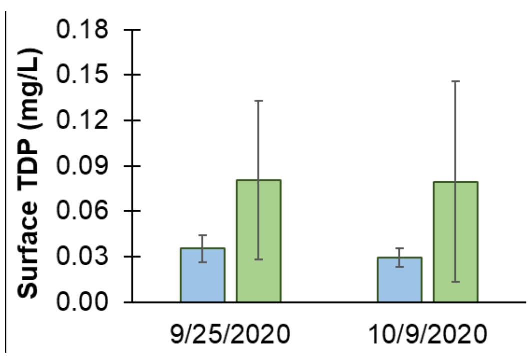
Wetland-1

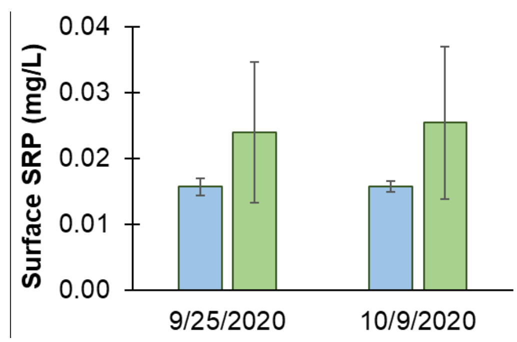

Open water
Vegetation
Surface DO (mg/L)
8.0
6.0
4.0
2.0
0.0


Open water
Vegetation
Temperature (°C)
25
20
15
10
5
0


Open water
Vegetation
Specific Cond. (µS/cm)
600
450
300
150
0


Surface TP (mg/L)
5.0
4.0
3.0
2.0
1.0
0.0
9/25/2020
10/9/2020


Surface TDP (mg/L)
0.18
0.15
0.12
0.09
0.06
0.03
0.00
9/25/2020
10/9/2020


Surface SRP (mg/L)
0.04
0.03
0.02
0.01
0.00
9/25/2020
10/9/2020

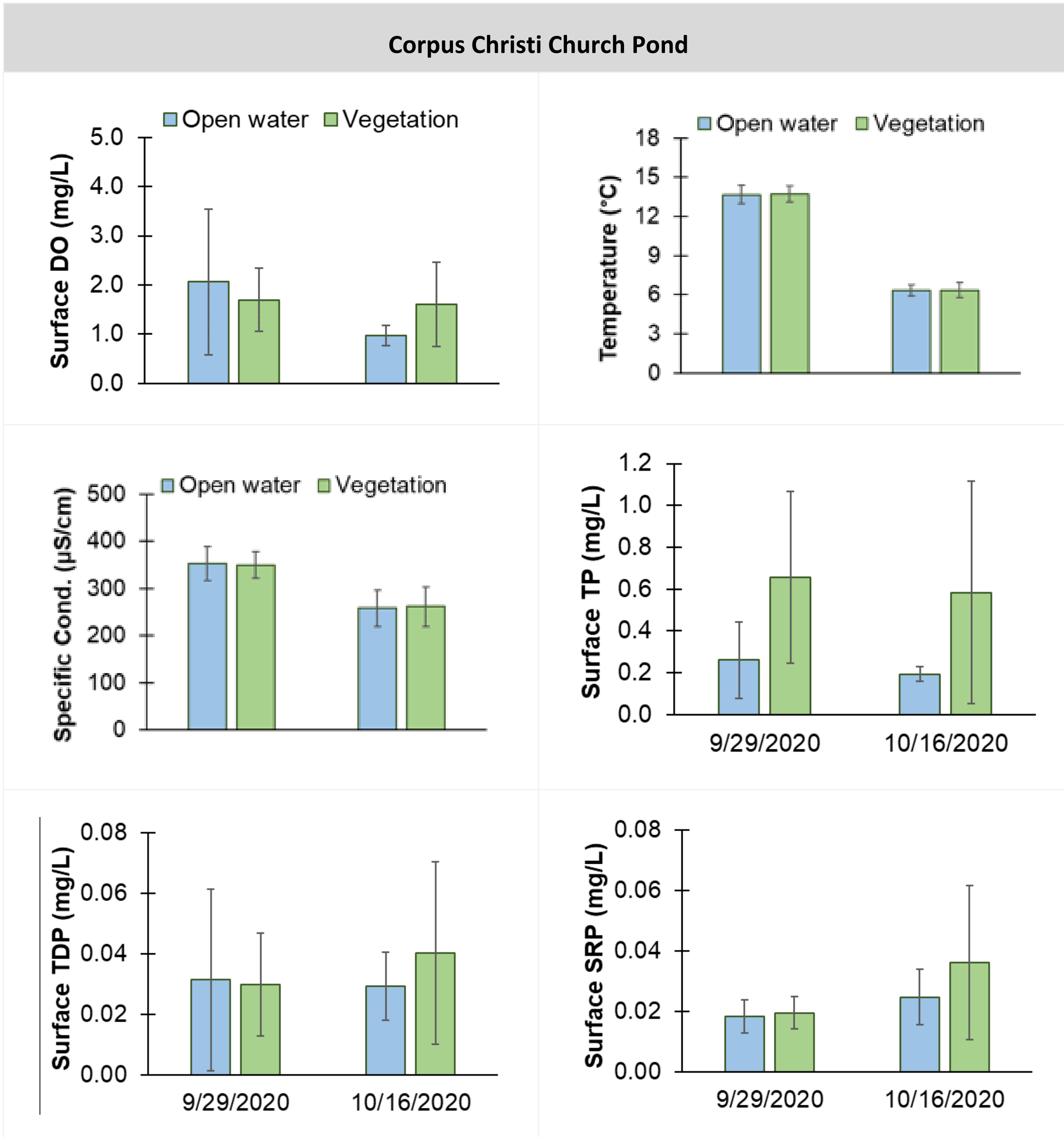


Figure A.2 Mean surface water DO, temperature, specific conductivity, total phosphorus (TP), total dissolved P (TDP), and soluble reactive P (SRP) concentrations in the open water and vegetated areas in the six ponds sampled during Field Season 1 (2020). The mean is based on 4-7 open water locations and 4-6 lateral transects along emergent vegetation at each site. Error bars represent standard deviation of the mean value.

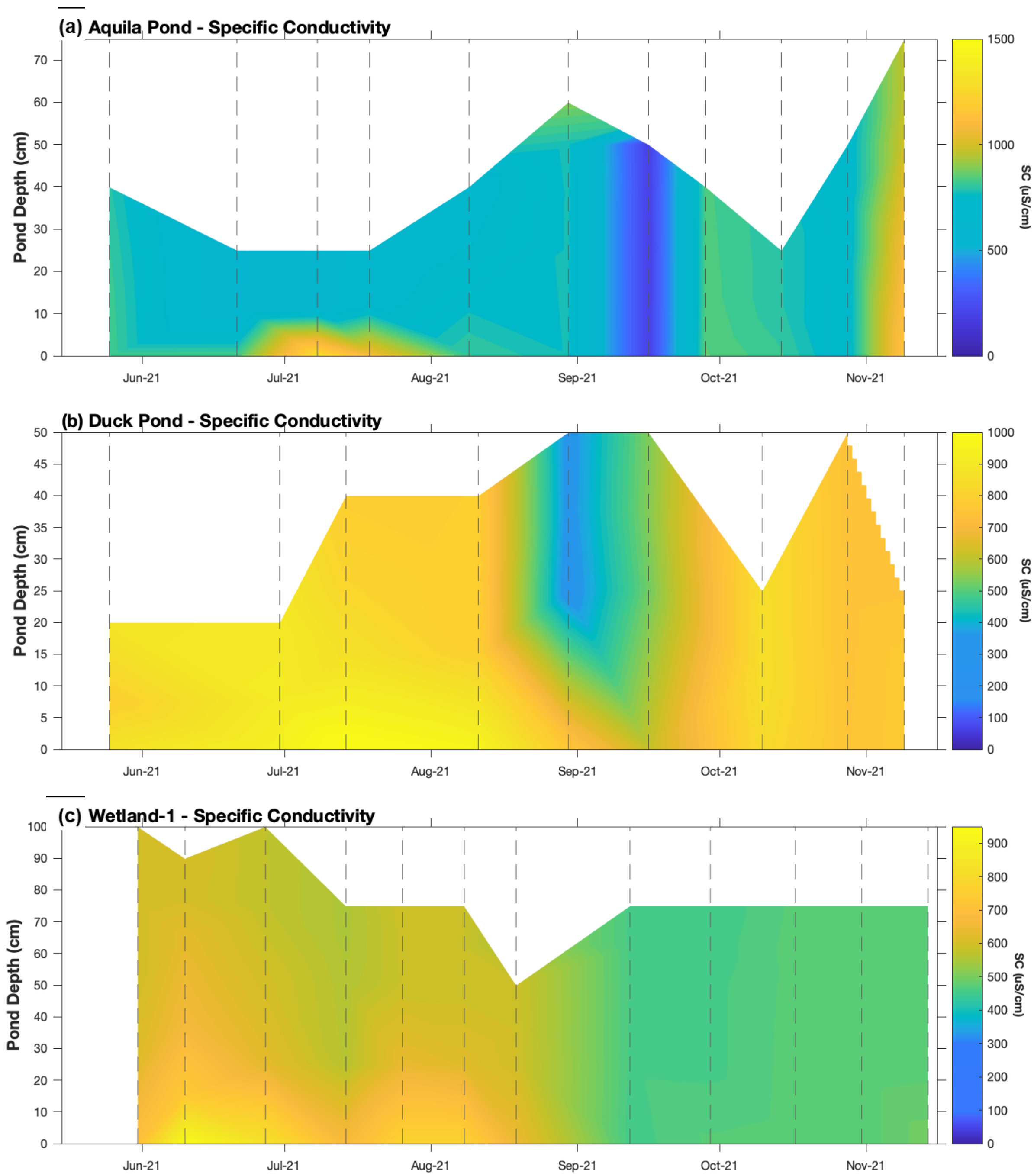


Figure A.3 Time series of specific conductivity (uS/cm) in the vegetated study ponds (Aquila, Duck Pond S, and Wetland-1) as a function of depth, plotted as color (note difference in scales across sites). Linear interpolation used to fill gaps in space (vertical axis) and time (horizontal axis). Profile dates indicated by vertical dashed lines.

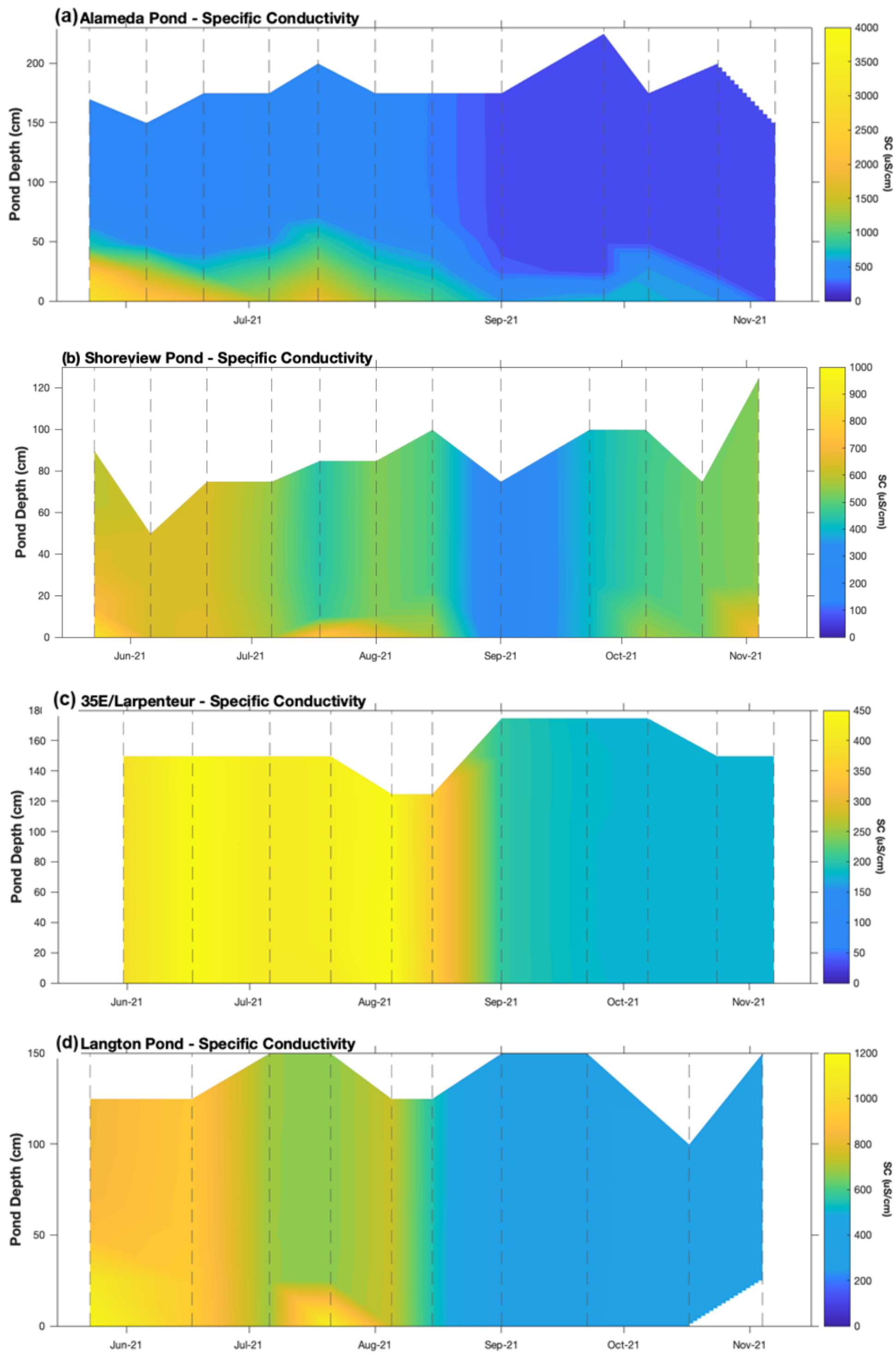


Figure A.4 Time series of specific conductivity (uS/cm) in the un-vegetated study ponds (Alameda, Shoreview Commons, 35E/Larpenteur, and Langton) as a function of depth, plotted as color (note difference in scales across sites). Linear interpolation used to fill gaps in space (vertical axis) and time (horizontal axis). Profile dates indicated by vertical dashed lines.

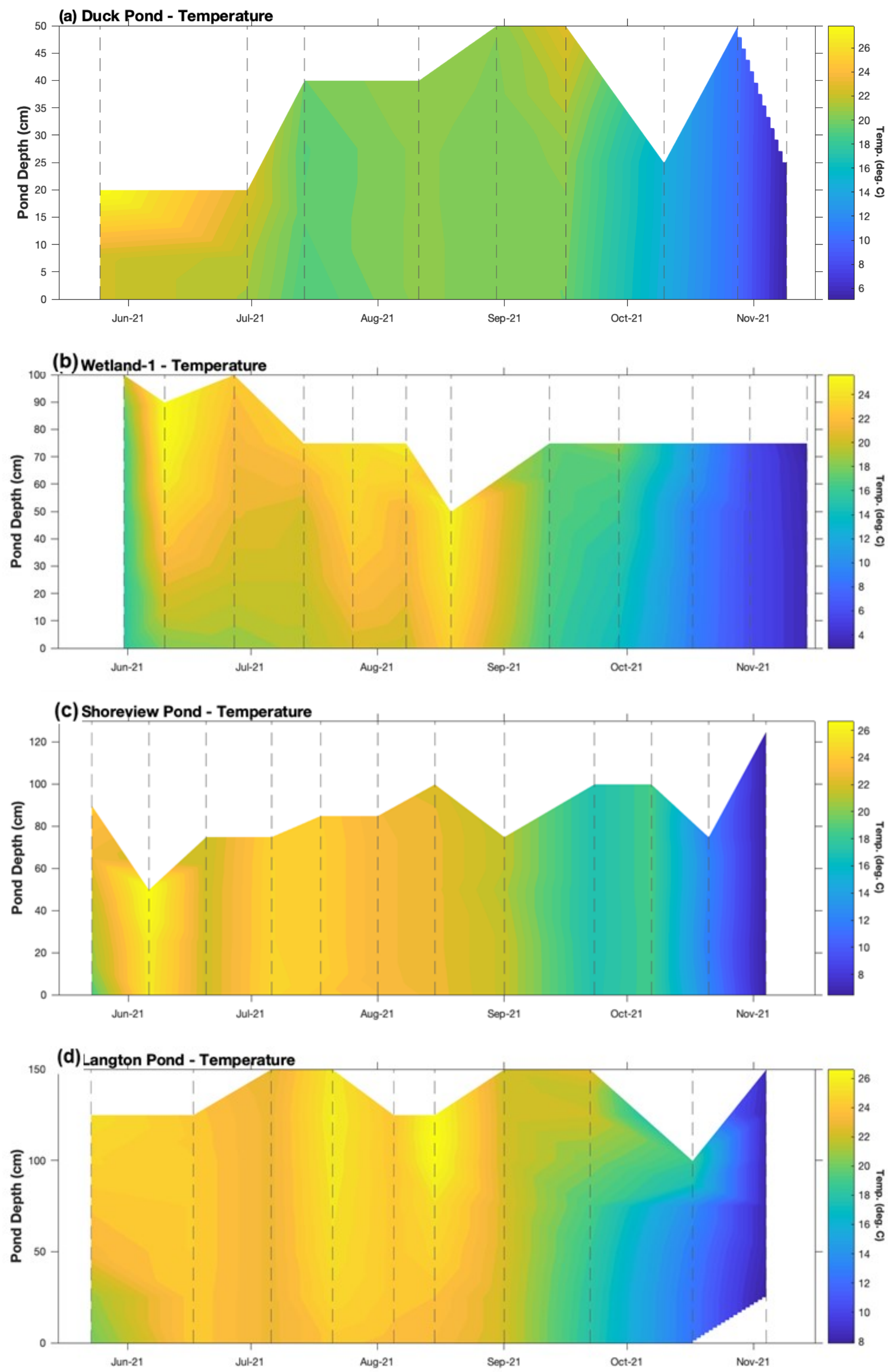


Figure A.5 Time series of temperature (˚C) in four study ponds (Duck Pond S, Wetland-1, Shoreview Commons, Langton) as a function of depth, plotted as color (note difference in scales across sites). Linear interpolation used to fill gaps in space (vertical axis) and time (horizontal axis). Profile dates indicated by vertical dashed lines.

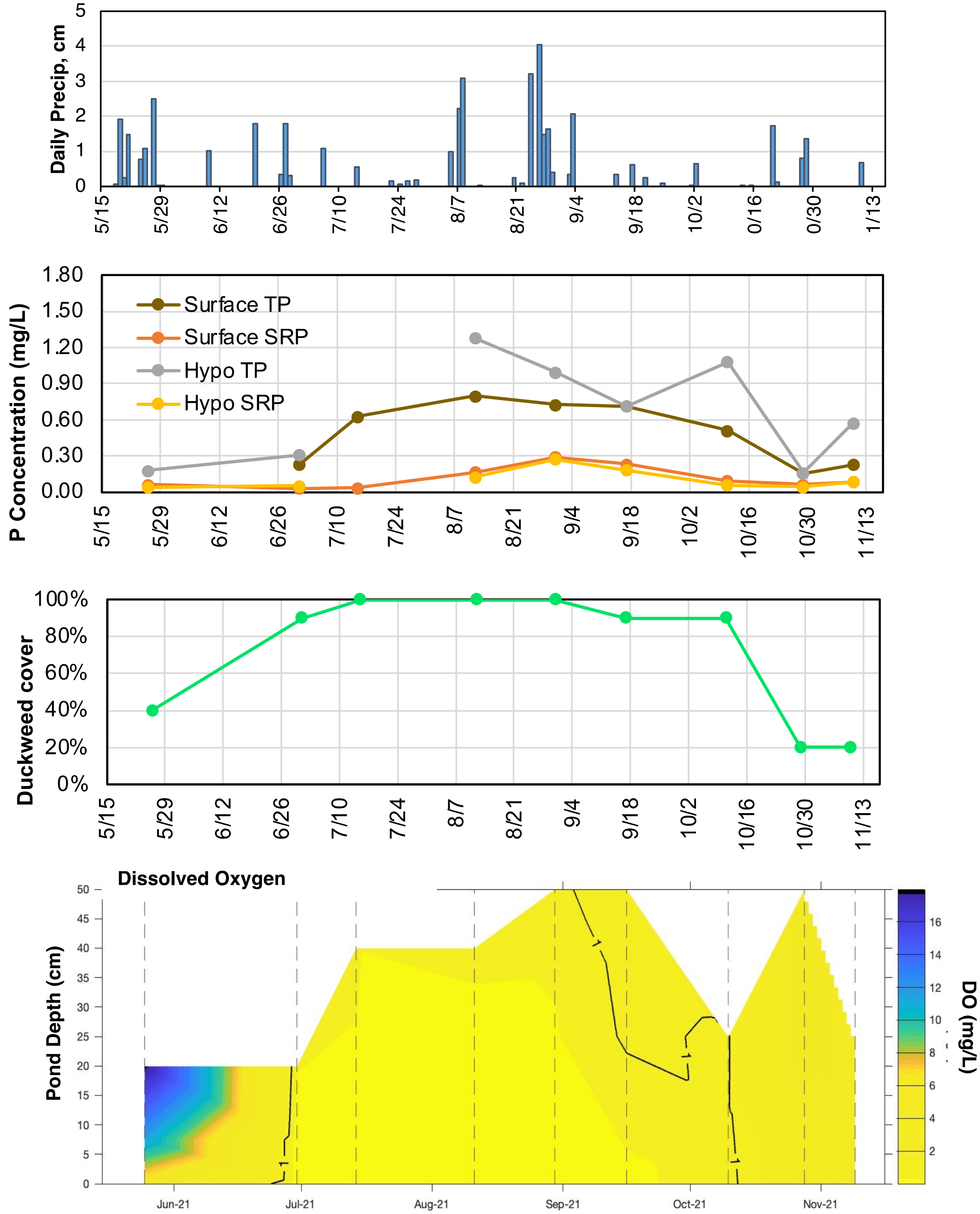


Figure A.6 Time series of rain (cm; from MSP Airport), surface and hypolimnion TP and SRP (mg/L), duckweed cover (%), DO profiles (mg/L; plotted as contours), at Duck Pond S. Data collected on site visits (shown by points in TP and Duckweed plots, as vertical dashed lines in DO contour plot) in 2021.

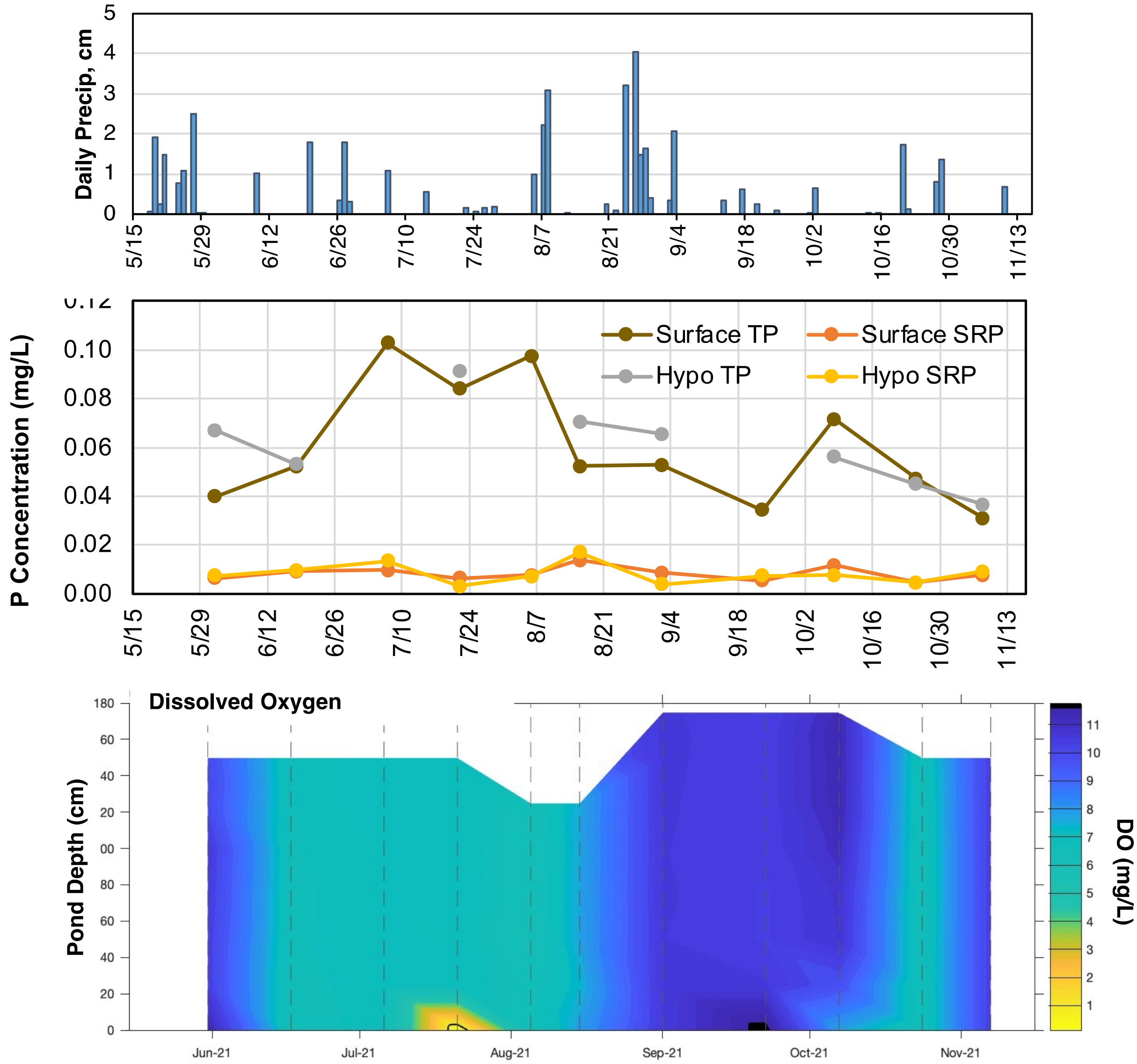


Figure A.7 Time series of rain (cm; from MSP Airport), surface and hypolimnion TP and SRP (mg/L), and DO profiles (mg/L; plotted as contours) at 35E/Larpenteur Pond. Data collected on site visits (shown by points in TP plot, as vertical dashed lines in DO contour plot) in 2021. No duckweed cover was observed on this pond.

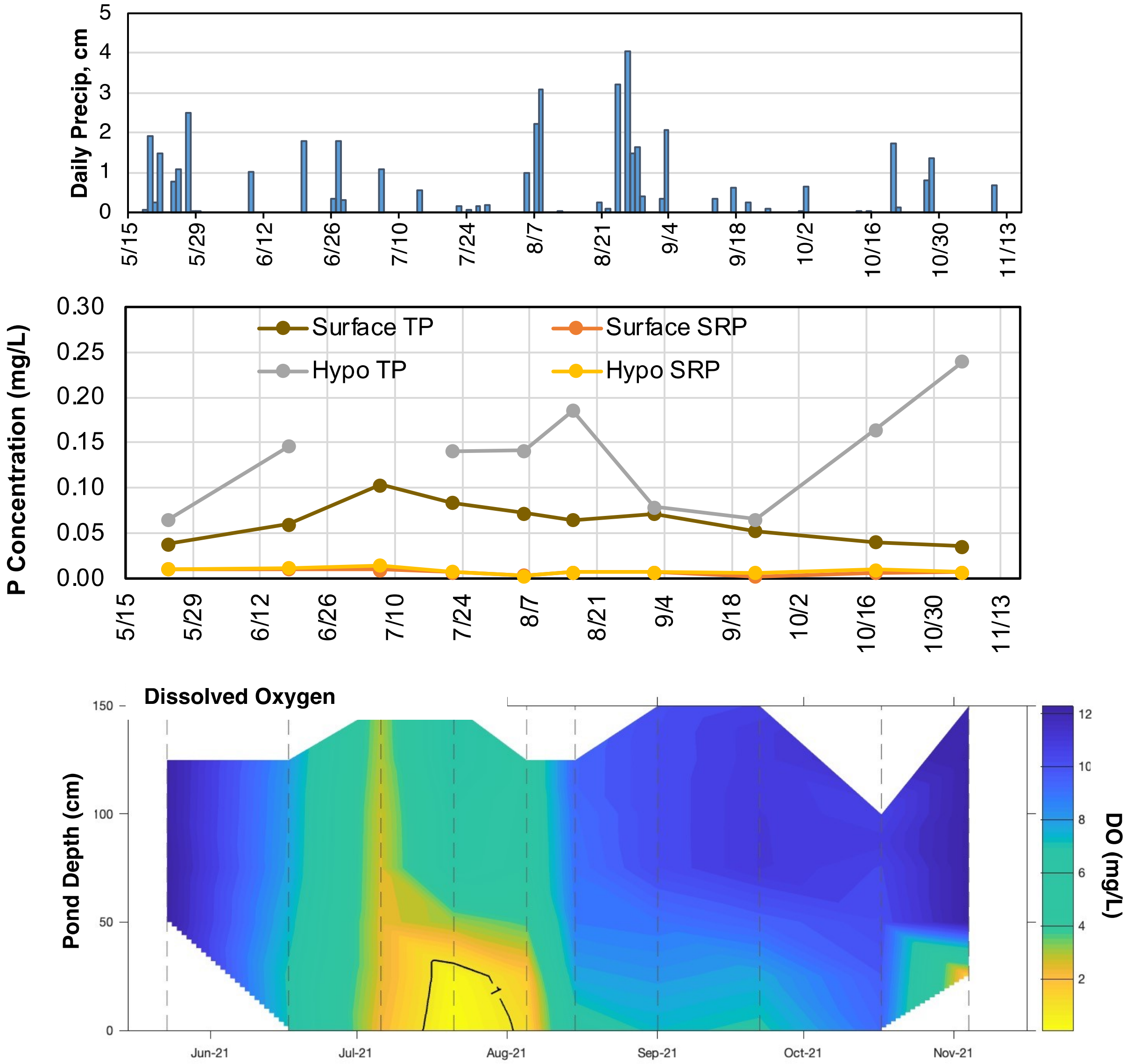


Figure A.8 Time series of rain (cm; from MSP Airport), surface and hypolimnion TP and SRP (mg/L), and DO profiles (mg/L; plotted as contours) at Langton Pond. Data collected on site visits (shown by points in TP plot, as vertical dashed lines in DO contour plot) in 2021. No duckweed cover was observed on this pond.

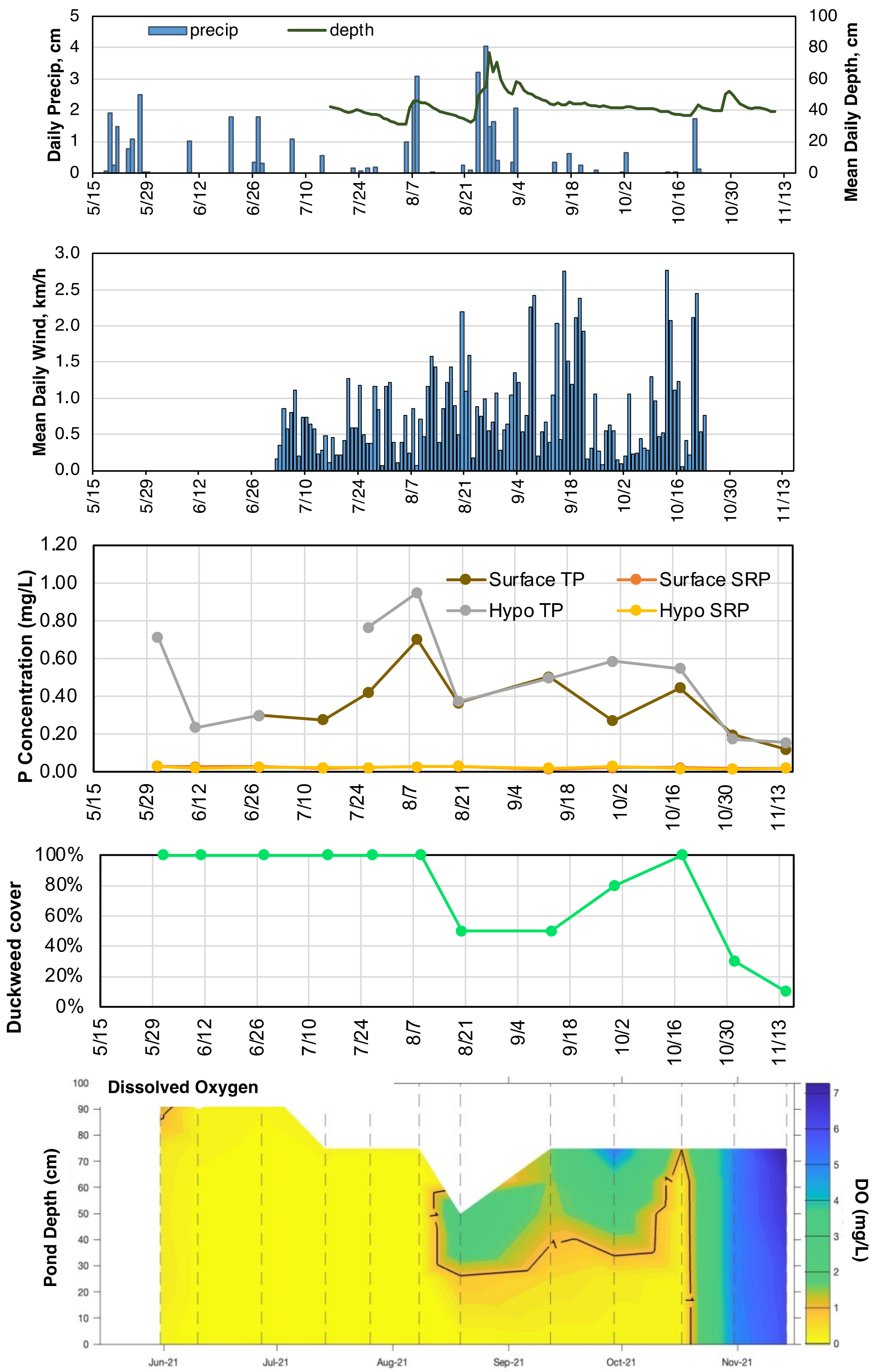


Figure A.9 Wetland-1 time series of rain (cm; from MSP Airport), water depth (cm), wind speed (km/h), surface and hypolimnion TP and SRP (mg/L), duckweed cover (%), DO profiles (mg/L; as contours),. Wind and water level measured by the monitoring stations and averaged into daily values for plotting. Other data collected on site visits. No RTRM due to loss of data (damage to temperature sensors).

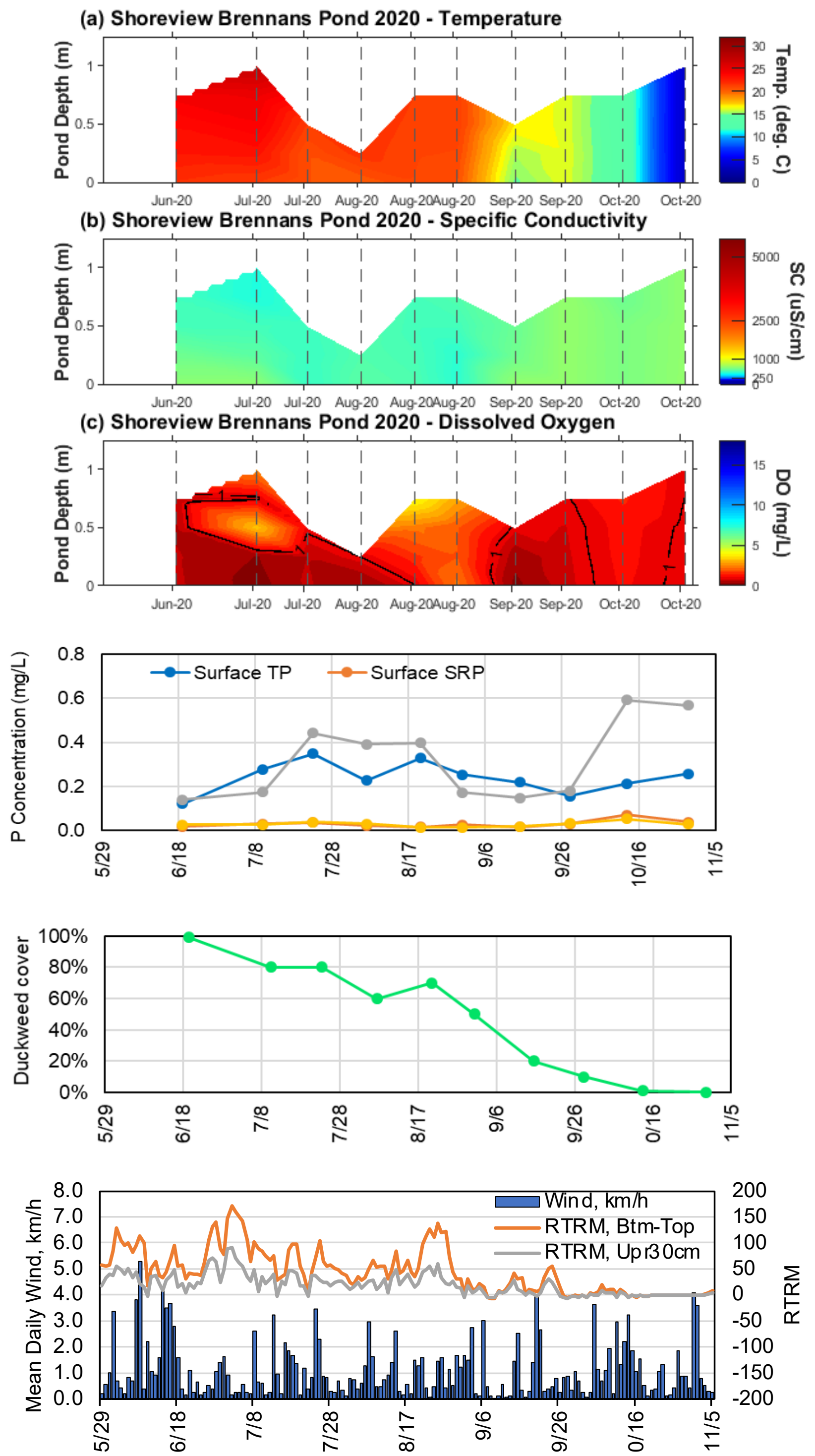


Figure A.10 Time series of temperature, conductivity, DO profiles (plotted as contours), surface and hypolimnion TP and SRP (mg/L), duckweed cover (%), RTRM, and wind at the Shoreview Commons Pond in 2020. Data collected on site visits (shown by points in TP and Duckweed plots, vertical dashed lines in contours) in 2020; wind and RTRM from monitoring station.

Table A.1 Summary of mean phosphorus concentrations in the surface water and hypolimnion of the ponds during the May to November period in 2020 and 2021.In the ponds with emergent vegetation (Aquila, Duck, Wetland_1), concentrations provided are for the open water area. Summer anoxic factor (AF; see definition in the main report) was calculated for the June to August period.

Pond	Field Season	Summer AF	Duckweed cover (% area)	Mean surface TP (mg/L)	Mean surface TDP (mg/L)	Mean surface SRP (mg/L)	Mean surface TDP:TP	Mean surface SRP:TDP	Mean hypo TP (mg/L)	Mean hypo TDP (mg/L)	Mean hypo SRP (mg/L)	Mean hypo TDP:TP	Mean hypo SRP:TDP
Aquila	2020	NA	77%	0.330	0.059	0.032	0.228	0.433	0.306	0.051	0.032	0.217	0.530
	2021	0.66	57%	0.321	0.096	0.049	0.342	0.553	0.329	0.113	0.058	0.380	0.549
Duck Pond S	2020	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	2021	1.0	72%	0.498	0.196	0.116	0.425	0.573	0.660	0.183	0.107	0.320	0.632
Wetland-1	2020	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	2021	0.84	77%	0.360	0.070	0.023	0.207	0.362	0.481	0.072	0.023	0.158	0.488
Shoreview Commons	2020	0.56	47%	0.240	0.056	0.031	0.259	0.592	0.321	0.060	0.029	0.232	0.511
	2021	0.32	45%	0.101	0.035	0.018	0.400	0.539	0.146	0.038	0.017	0.325	0.469
Alameda	2020	NA	68%	0.196	0.060	0.031	0.346	0.475	0.719	0.115	0.029	0.209	0.413
	2021	0.89	72%	0.164	0.081	0.047	0.496	0.536	1.011	0.322	0.085	0.326	0.396
35E/ Larpenteur	2020	NA	0%	0.074	0.025	0.008	0.326	0.508	0.099	0.031	0.008	0.328	0.428
	2021	0.0	0%	0.061	0.032	0.008	0.586	0.291	0.061	0.034	0.008	0.561	0.271
Langton	2020	NA	0%	0.065	0.024	0.008	0.371	0.480	0.110	0.032	0.009	0.300	0.654
	2021	0.04	0%	0.062	0.021	0.007	0.365	0.328	0.136	0.023	0.008	0.202	0.394

Appendix B
Laboratory Phosphorus Release Study Data and Pond Sediment Analysis Data

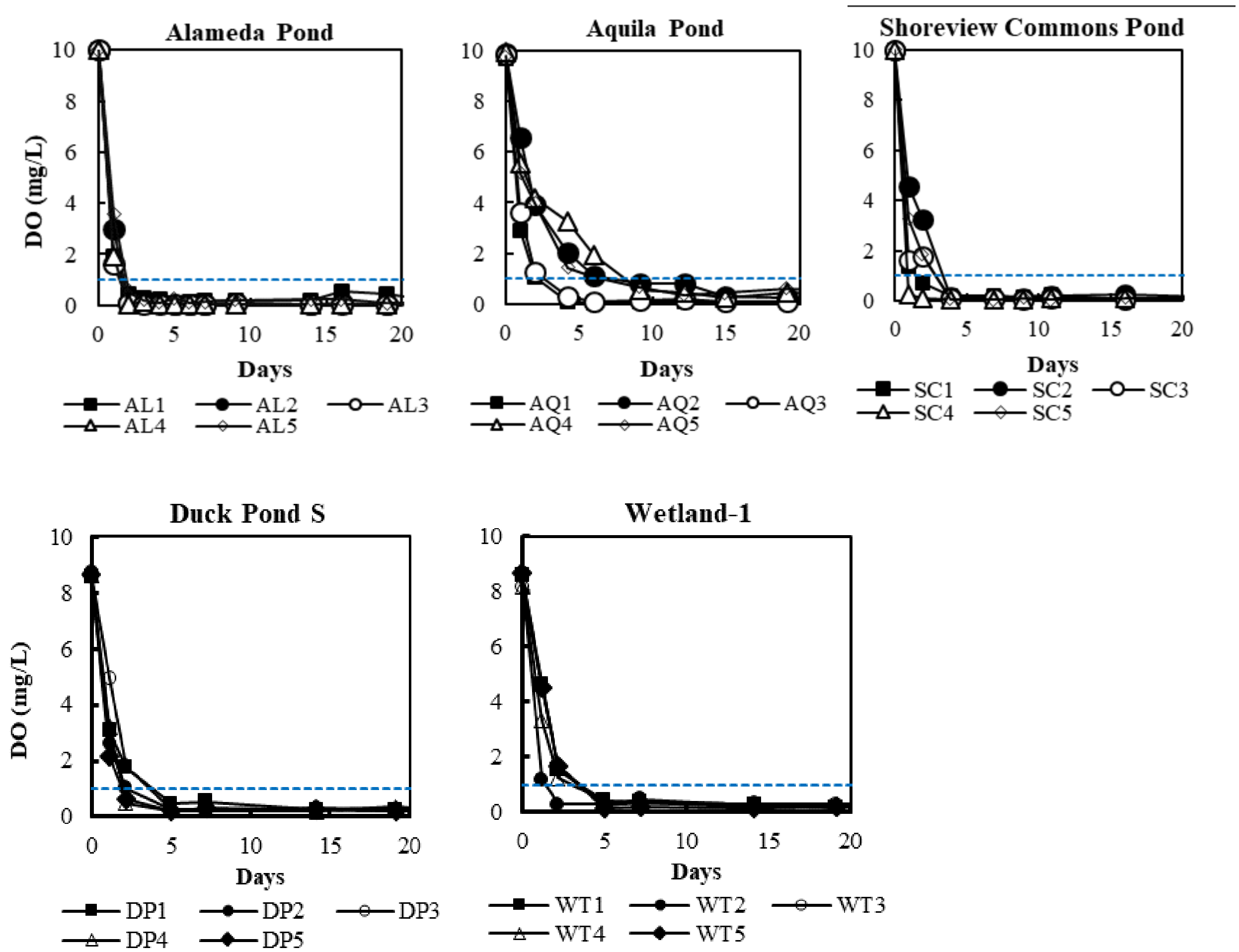


Figure B.1 Water column dissolved oxygen (DO) concentrations after air supply was switched off in the sediment cores from the five ponds. The DO measurements were taken ~8 cm above the sediment surface. The 1 mg/L DO (dashed line) represents anoxic state. Data for Alameda, Aquila and Shoreview Commons ponds are reported in Janke et al. (2021).

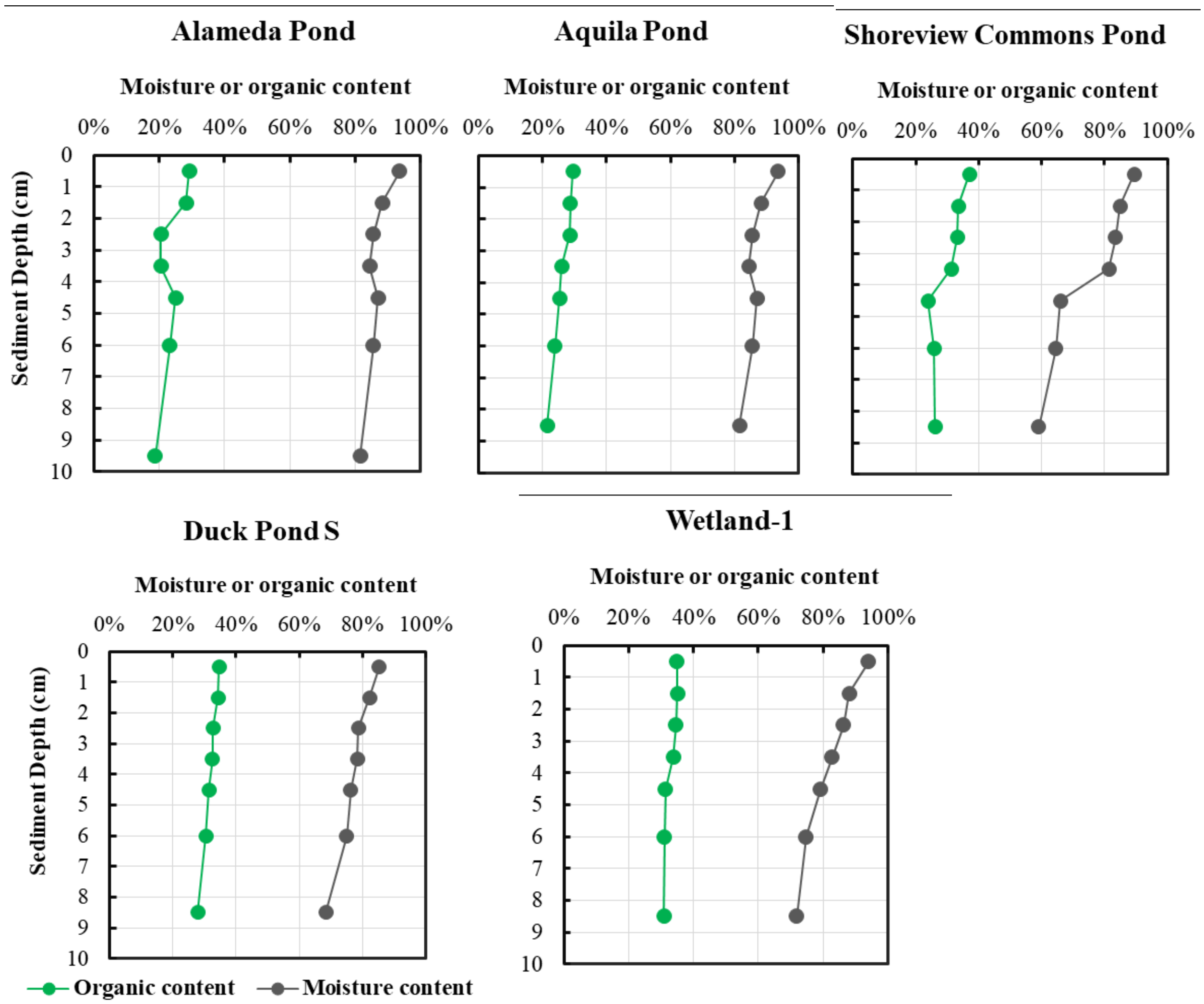


Figure B.2 Vertical variations in the sediment moisture content and organic matter content in the pond sediments. The 0 cm sediment depth represents the sediment-water interface. Values are plotted in the mid-point of a given sediment depth interval (for example, concentration for 0-2 cm depth is plotted at 1 cm). Duck Pond S and Wetland-1 were part of the current study. Data for other ponds are from previous studies (Janke et al. 2021).

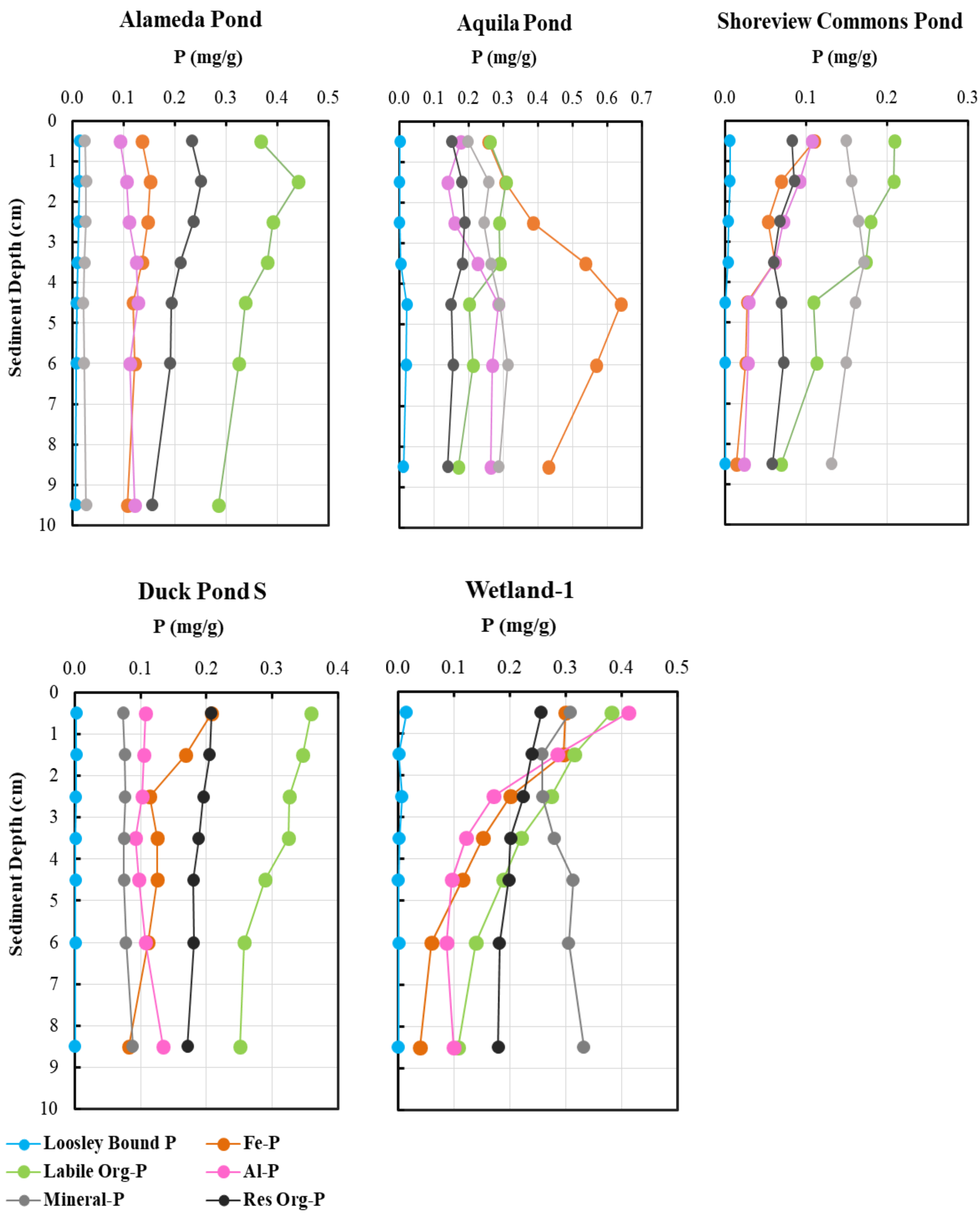


Figure B.3 Concentrations of the phosphorus (P) fractions in the upper 10 cm depth of sediments in five ponds. The average concentration (dry-weight basis) in 3-5 sediment cores from each pond are plotted. The loosely-bound P, iron-bound P (Fe-P) and labile organic-P are the mobile P forms. The sediment depth 0 cm represents the sediment-water interface. Values are plotted in the mid-point of a given sediment depth interval (for example, concentration for 0-2 cm depth is plotted at 1 cm). Notice difference in X-axis scale in the plots.

Appendix C
User Guide for Pond Assessment Tool v1.0

Illustration for Alameda Pond, Roseville, MN

This user guide uses the Alameda pond site as an example to illustrate:

1. Sample data collection and data extraction methods
2. Application of the available tools, including multi-level assessment depending on data availability

Case 1. Simple Assessment of Surface TP (Tool 1-A)

Basic Information Provided by City:

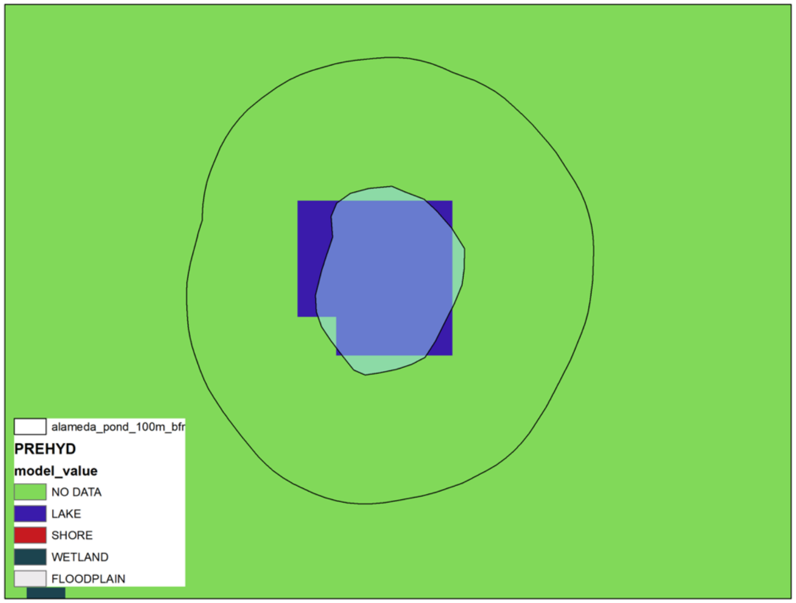
- Storm drains connected to the pond in the 1960's when surrounding area was developed ("old" pond, < 1990)
- Pond characteristics known:
  surface area = 2.89 ac
  drainage area = 285 ac
  max depth = 2.25 m
  mean depth = 1.23 m

How to evaluate hydrogeologic setting: *proximity to historic water bodies* and *hydric soils*

Basic GIS Buffer Analysis (Using 100m Buffer)

MnDOT Historical Waterbody Layer

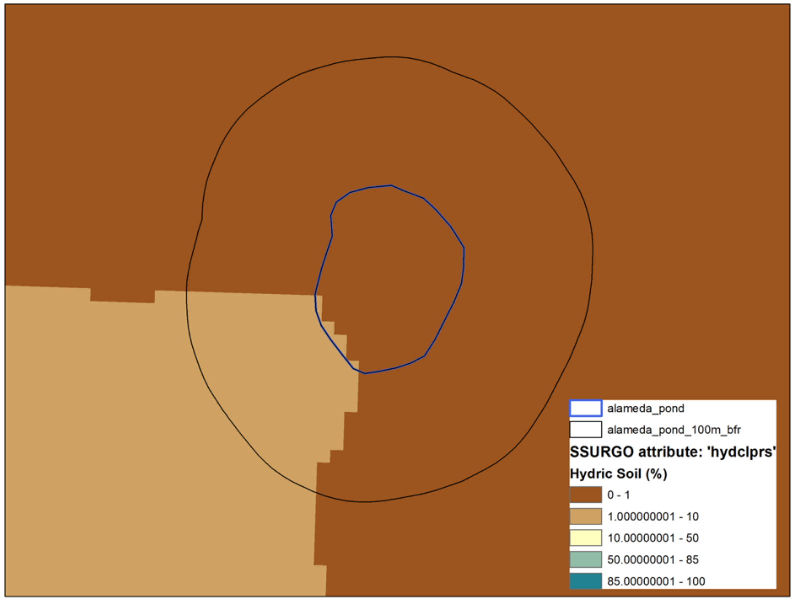
https://gisdata.mn.gov/dataset/water-hist-hydrography



→ Pond is a historic water body (Lake)

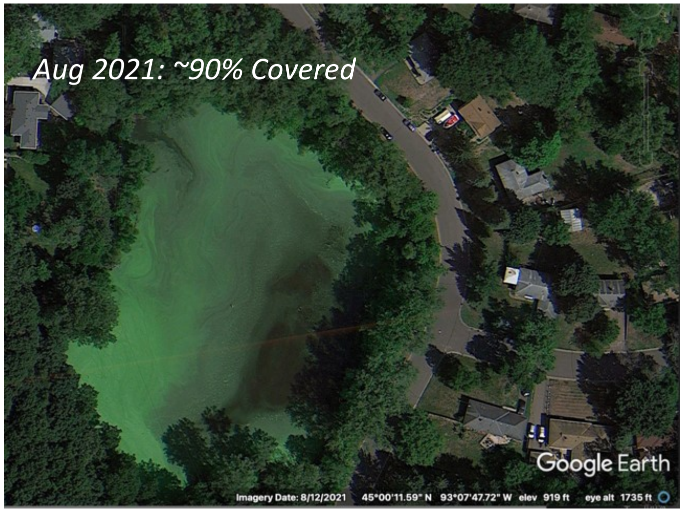
gridded SSURGO using 'hydclsprs' attribute (percentage of soil map unit that is classified as 'hydric')

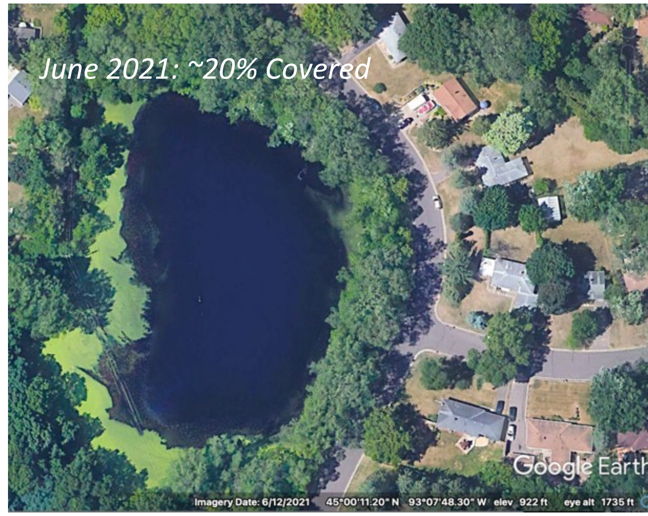
https://catalog.data.gov/dataset/gridded-soil-survey-geographic-database-gssurgo-minnesota

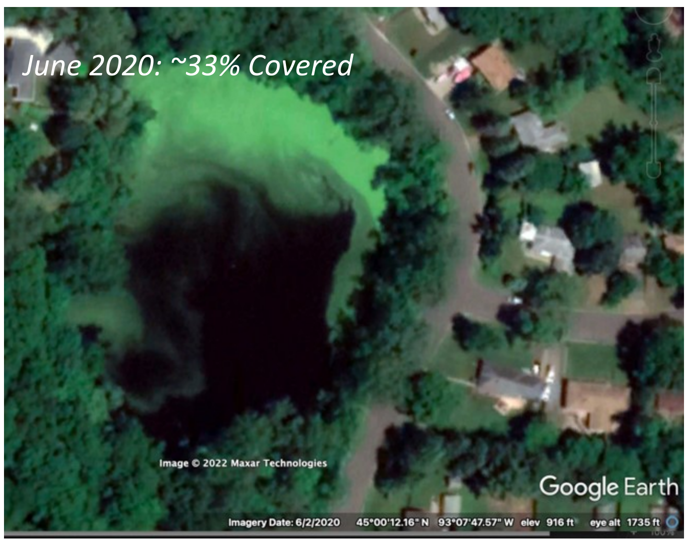


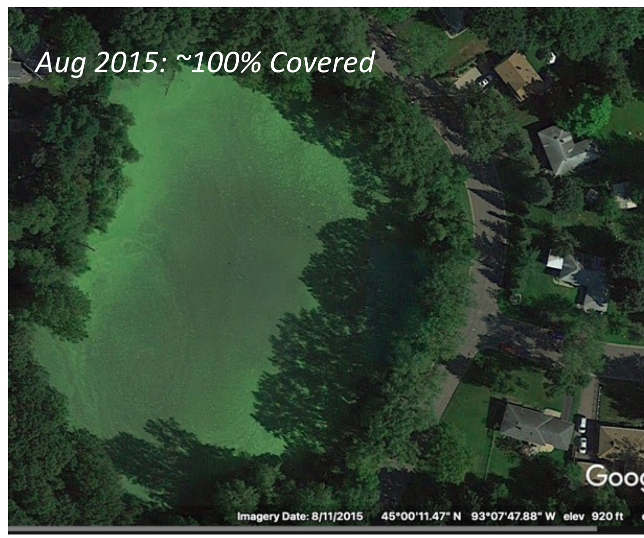
→ Low presence of hydric soils under / near the pond (<10%)

How to evaluate emergent vegetation cover and duckweed cover in the pond

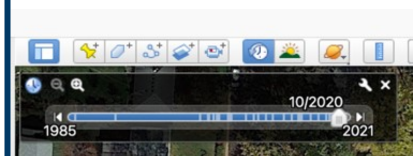








Using Selected Summer Aerial Images in Google Earth (Imagery Date Slider)

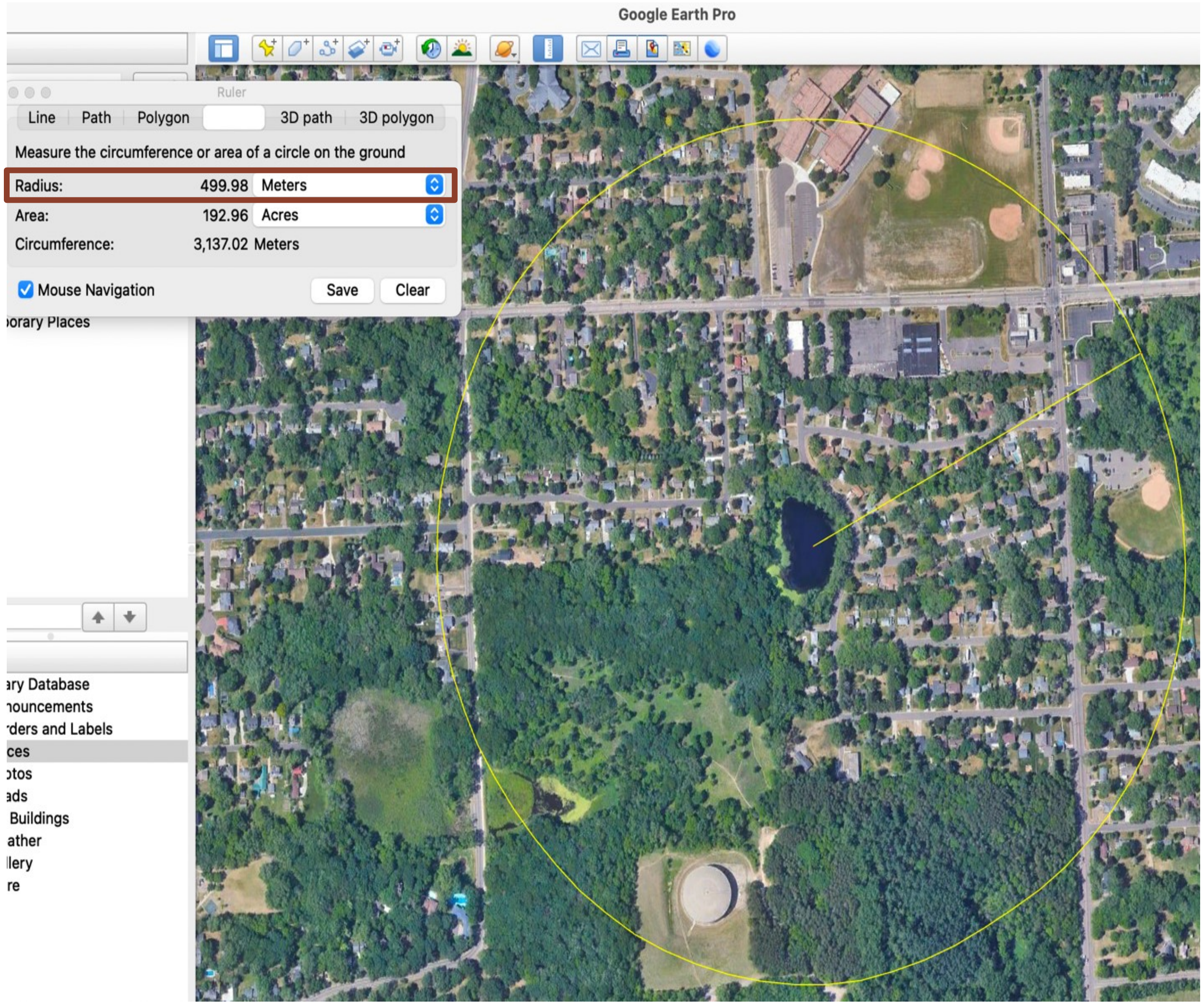


Average Duckweed Cover: 60%+

No Emergent Vegetation

***Note**: During three seasons of periodic observations at this pond, mean summer (June-Aug) cover was 90%, and mean season (May-Nov) cover was 67%. Thus, direct observation is preferred to aerial imagery.*

How to evaluate canopy cover and major land use (Option 1)



Using the Circle

Tool in Google Earth

with Radius ~500m

Canopy Cover

Likely >60%

Majority Land Use

is Residential / Park

How to evaluate surrounding land use using GIS Buffer Analysis (Option 2)

Optional: Land Use Basic GIS Buffer Analysis (Using 500m Buffer)

<u>Met Council General Land Use Layer</u>

https://gisdata.mn.gov/dataset/us-mn-state-metc-plan-generl-lnduse2020

→ *Confirms original estimate from Google Earth that dominant land uses are residential + park*

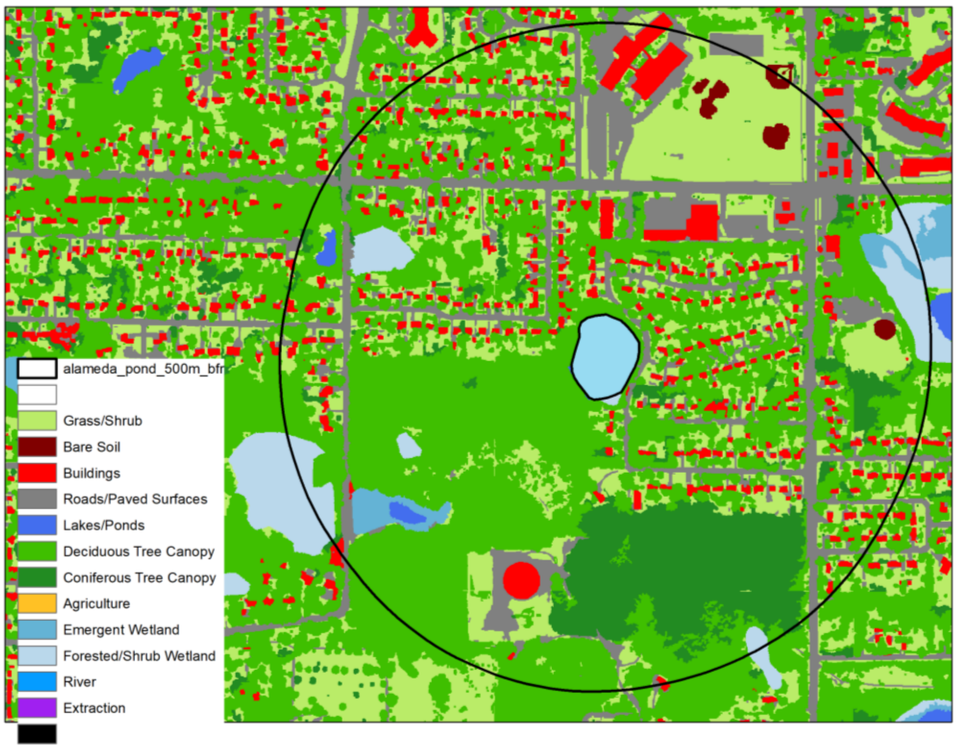
→ *If not obvious from visual inspection, can use a Zonal Histogram command to output pixel counts by land use within the pond's buffer ('zone')*

How to evaluate ca`nopy cover using GIS Buffer Analysis (Option 2)

Optional: Canopy Cover from Basic GIS Buffer Analysis (Using 500m Buffer)

TCMA Land Cover (1 m Resolution)

https://gisdata.mn.gov/dataset/base-landcover-twincities



Zonal Histogram command (ArcGIS)

- Zone defined by 500m Buffer

- Input Raster is Land Cover layer

Category	Pixel Count	Percent of Total
Grass/Shrub	**93679**	**16.9%**
Bare Soil	**1629**	**0.3%**
Buildings	**33271**	**6.0%**
Roads/Paved Surfaces	**69699**	**12.5%**
Lakes/Ponds	**1510**	**0.3%**
Deciduous Tree Canopy	294849	53.1%
Coniferous Tree Canopy	42477	7.6%
Agriculture	**0**	**0.0%**
Emergent Wetland	**6969**	**1.3%**
Forested/Shrub Wetland	**11535**	**2.1%**

Total Canopy Cover ~61%

→ Confirms original estimate from Google Earth that canopy cover is >60% in the pond's vicinity

Application of Tool 1A-TP

Assessment Level: 1

Goal: Risk of High Total Phosphorus Concentration in Pond Surface Water

Factor	Factor Level (Risk of High TP) Low	Medium	High	Unknown	Data Source
Construction Year		x			City drawings show sewers connected to pond in 1960's; historical aerial photos in Google Earth show pond present in ~1994
	After 2010	*Before 1990*	*1990-2010*		
Underlying or Nearby Hydric Soil	x				Little or no hydric soils in vicinity of pond according to GIS overlay analysis
	0-40%	*NA*	*40-100%*		
Historic Wetland			x		historic lake according to MN Historical Hydrography Layer
	No	*NA*	*Yes*		
Emergent Vegetation Cover	x				Viewed aerial images (in Google Earth) over past five years, and pond has no cattails or other emergent veg
	0-50%	*NA*	*>50%*		
Duckweed Cover			x		Viewed aerial images (in Google Earth) for May to October period of five seasonal years, estimated the average duckweed coverage
	0-50%	*NA*	*>50%*		
Watershed Tree Canopy			x		GIS overlay analysis showed > 60% tree cover
	< 20%	*20% - 60%*	*> 60%*		
Dominant Watershed Land Use			x		Based on aerial imagery (in Google Maps)
	Comm	*Ind/Transp*	*Residential*		

Interpretation: Alameda Pond is at medium to high risk for high surface water TP

Case 2. More Advanced Input Data Preparation for Prediction of Pond TP (Tool 1-B) and Sediment Phosphate Release (Tool 3)

Basic Information Provided by city:

- Bathymetry survey data (depth at various locations in pond) – needed for anoxic factor calculation

Supporting Steps Illustrated in this Section:

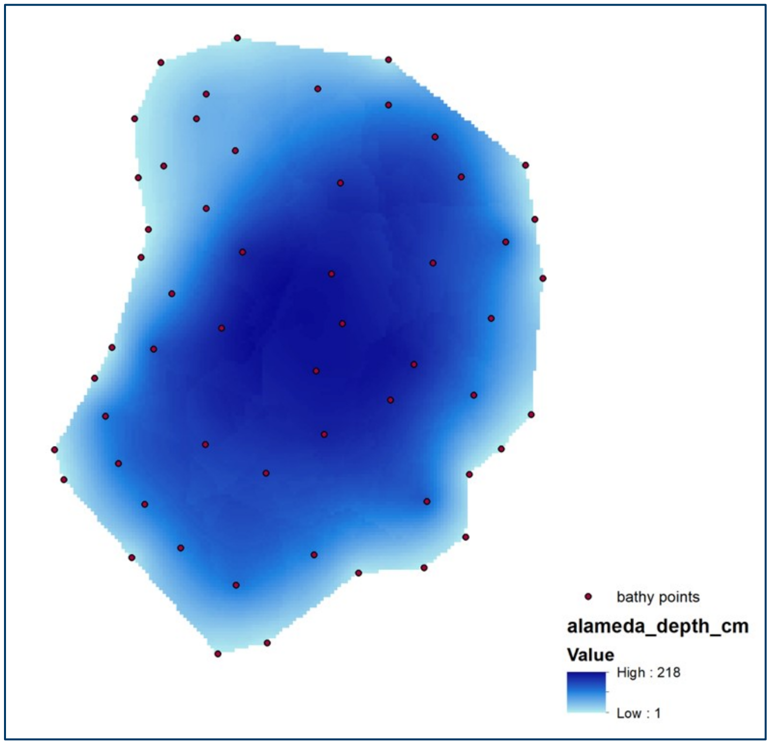
- Anoxic Factor Calculation
- Option 1 (Preferred Method): Direct Calculation from Observational Data
- Option 2: Using Tool 2 to Model AF from Duckweed Cover
- Anoxic Sediment Phosphate Release Rate (Tools 3A and 3B)

How to calculate the pond's Anoxic Factor

Anoxic Factor Calculation Option 1: From Bathymetry and Dissolved Oxygen profiles

Step 1: Convert Pond Bathymetry into Hypsography (Area vs. Depth)

(A) Use kriging (GIS) to convert depth measurements at individual points into a continuous map of depth



(B) Use Zonal Histogram to output pixel count for each depth in the pond *[sum of pixels] * [pixel area] at a given depth is approximately the area of the pond bottom located at that depth*

(C) Aggregate depths into bins as needed to create an area vs. depth curve *- note that depth is relative to top of pond in the example below, such that depth of 0 cm corresponds to pond's surface area when pond is at max depth*

Depth, cm	Cumulative Area, m^2
0	4454.8
4	4441.8
8	4406.5
12	4359.0
16	4300.3
20	4244.0
…	…
180	1037.8
184	931.0
188	812.0
192	697.5
196	591.3
200	478.5
204	366.8
208	238.3
212	69.8
216	3.0

Anoxic Factor Calculation (Option 1: From Bathymetry and Dissolved Oxygen profiles)

Step 2: For each profile date, determine area and area fraction of pond bottom corresponding to each depth in the profile

(A)

Date	Depth, cm	DO, mg/L
6/21/21	0	0.7
6/21/21	12.5	0.38
6/21/21	25	0.22
6/21/21	50	0.06
6/21/21	75	0.03
6/21/21	100	0.03
6/21/21	125	0.02
6/21/21	150	0
6/21/21	175	0
7/19/21	0	1.73
7/19/21	12.5	1.12
7/19/21	25	0.81
7/19/21	50	0.44
7/19/21	75	0.09
7/19/21	100	0.06
7/19/21	125	0.01
7/19/21	150	0
7/19/21	175	0
7/19/21	200	0
8/16/21	0	4.02
8/16/21	12.5	4.08
8/16/21	25	3.75
8/16/21	50	3.62
8/16/21	75	3.36
8/16/21	100	1.75
8/16/21	125	0.11
8/16/21	150	0
8/16/21	175	0

(B)

Area @ Depth (m2)	Area Fraction
3893.8	0.190
3662.3	0.178
3349.0	0.163
2959.0	0.144
2561.3	0.125
2079.3	0.101
1324.0	0.065
697.5	0.034
0.0	0.000
4300.3	0.170
4118.0	0.163
3893.8	0.154
3349.0	0.132
2959.0	0.117
2561.3	0.101
2079.3	0.082
1324.0	0.052
697.5	0.028
0.0	0.000
3893.8	0.190
3662.3	0.178
3349.0	0.163
2959.0	0.144
2561.3	0.125
2079.3	0.101
1324.0	0.065
697.5	0.034
0.0	0.000

Depth, cm	Cumulative Area, m^2
0 - 3	4454.8
4 - 7	4441.8
8 - 11	4406.5
12 - 15	4359.0
16 - 19	4300.3
20 - 23	4244.0
…	…
180 - 183	1037.8
184 - 187	931.0
188 - 191	812.0
192 - 195	697.5
196 - 199	591.3
200 - 203	478.5
204 - 207	366.8
208 - 211	238.3
212 - 215	69.8
216 - 218	3.0

(A) **Raw Data**: Collect several profiles of Dissolved Oxygen over the course of the open water season, ideally near the pond's deepest point (truncated example shown here with 3x points)

(B) **Assign Sediment Area and Area Fraction to Each Depth in the Profile:** Using the depth-area curve from Step 1, look up the area fraction that corresponds to the profile depth. Optional: account for difference between hypsographic max depth and pond depth on the profile date, by adding that difference to the lookup value. (Example: on 7/19, max profile depth is 200 cm, hypsography max depth is 218 cm; difference is 18cm, so add that to the lookup value when referencing the depth-area curve. A value of 0 cm in the profile gets assigned an area, 4300m^2, *corresponding to a depth of 18cm in the depth-area curve)*

Anoxic Factor Calculation (Option 1: From Bathymetry and Dissolved Oxygen profiles)

Step 3: Calculate fraction of pond bottom exposed to anoxic conditions (anoxic fraction) for each profile date, and aggregate multiple values into Anoxic Factor

Date	Depth, cm	DO, mg/L	Area @ Depth (m2)	Area Fraction
6/21/21	0	0.7	3893.8	0.190
6/21/21	12.5	0.38	3662.3	0.178
6/21/21	25	0.22	3349.0	0.163
6/21/21	50	0.06	2959.0	0.144
6/21/21	75	0.03	2561.3	0.125
6/21/21	100	0.03	2079.3	0.101
6/21/21	125	0.02	1324.0	0.065
6/21/21	150	0	697.5	0.034
6/21/21	175	0	0.0	0.000
7/19/21	**0**	**1.73**	**4300.3**	**0.170**
7/19/21	**12.5**	**1.12**	**4118.0**	**0.163**
7/19/21	**25**	**0.81**	**3893.8**	**0.154**
7/19/21	**50**	**0.44**	**3349.0**	**0.132**
7/19/21	**75**	**0.09**	**2959.0**	**0.117**
7/19/21	**100**	**0.06**	**2561.3**	**0.101**
7/19/21	**125**	**0.01**	**2079.3**	**0.082**
7/19/21	**150**	**0**	**1324.0**	**0.052**
7/19/21	**175**	**0**	**697.5**	**0.028**
7/19/21	**200**	**0**	**0.0**	**0.000**
8/16/21	0	4.02	3893.8	0.190
8/16/21	12.5	4.08	3662.3	0.178
8/16/21	25	3.75	3349.0	0.163
8/16/21	50	3.62	2959.0	0.144
8/16/21	75	3.36	2561.3	0.125
8/16/21	100	1.75	2079.3	0.101
8/16/21	125	0.11	1324.0	0.065
8/16/21	150	0	697.5	0.034
8/16/21	175	0	0.0	0.000

(A)

Anoxic Fraction
1.0
1.0
0.11

(A) **Anoxic Fraction**: For each profile date, determine the fraction of pond sediment area for which DO < 2 mg/L *(e.g., sum the 'Area Fraction' column within each profile date for rows with DO < 2.0)*

(B)

Anoxic Factor ~ 0.73

Interpretation: From June 1 to Aug 31, roughly 73% of the pond sediments were exposed to anoxic conditions (DO < 2 mg/L)

(B) **Average the Anoxic Fractions together over the entire period of data collection.** If profiles were relatively evenly spaced out over the monitoring season, they can be averaged arithmetically. If spacing was more irregular, each value should be weighted by roughly the number of days represented by the measurement (assume the value sits in the middle of the time period between the previous and next profile date).

In our example, the values were collected mid-month in June, July, and August, so they can be averaged together without weighting and assuming they are representative for the June 1 – Aug 31 period.

Ideally, more frequent profiles would be measured. Three might be considered the bare minimum.

Application of Tool 2 - Oxygen

Anoxic Factor Calculation Option 2: Using Tool 2

(A) Determine duckweed cover on pond. See earlier section on using Google Earth imagery; season average from multiple site visits is the preferred method. *Note that model is primarily valid on ponds with a range of duckweed cover; duckweed-free ponds will have lower but more variable anoxic factors*.

(B) Enter 60 into Model 1 of the Oxygen Tool (Tool 2). The model estimates an anoxic factor (AF) of 0.58; considerably less than the value determined from data collection (0.73).

<u>**Tool 2-Oxygen**</u>

Assessment Level: **1**

Goal: **Pond Anoxic Factor, or Water Column Dissolved Oxygen**

Anoxic factor is a time-integrated measure of how much of the pond sediment is exposed to overlying anoxic water (0.0 - 1.0); the more frequently or extensive the anoxia, the greater risk for sediment release of P. It is also a strong predictor of pond water TP.

Water column-averaged dissolved oxygen is a slightly weaker but similarly useful predictor of pond TP and of sediment release.

		Model Parameter(s)				Model Output			
	Model	Value	Name	Definition	Notes or Data Source*	Dependent Variable	Units	Description	Modeled Value
1	[AF] = A*[DW] + B					**AF: Anoxic Factor**	--	Season Mean Anoxic Factor	**0.58**
	A = 0.0097	**60**	**DW: Duckweed Cover**	Percent of pond area covered by duckweed in midsummer, or averaged over season	Aerial photos, site visit(s)			*Season (~May-Nov) mean*	
	B = 0.00009								

Input (Duckweed Cover, %)

Output (AF)

Application of Tool 1B - TP

Assessment Level: 2 – Goal: Total Phosphorus Concentration in Pond Surface Water

Prediction of Pond Surface Water TP

Using available data (max depth, anoxic factor, duckweed cover), two options for models are considered here.

Case A: 'Limited' data case. Assuming that only duckweed cover could be observed, Model 1 of Tool 1B could be used to predict Pond TP for Alameda Pond. We use the value of duckweed cover (60%) estimated previously from Google Earth.
This model produces an estimated value of **0.16 mg/L for TP.**

Tool 1B-TP

Assessment Level: 2
Goal: Total Phosphorus Concentration in Surface Water
This calculator provides several models for estimating mean and likely range of pond surface water TP. Higher TP values indicate ponds may be at risk of poor performance for P removal, but a hydrologic analysis would be required to accurately assess a pond's P mass budget.

	Model	Model Parameter(s)				Model Output			
	Model	Value	Name	Definition	Notes or Data Source*	Dependent Variable	Units	Description	Modeled Value
1	[TP] = A*[DW] + B					**TP: Total Phosphorus**	mg/L	Season Mean Surface TP	0.16
	A = 0.0012	**60**	**DW: Duckweed Cover**	Percent of pond area covered by duckweed in midsummer, or averaged over season	Aerial photos, site visit(s)			*Season: ~May - Nov*	
	B = 0.083								
2	[TP] = A*[DW] + B*[DMAX] + C					**TP: Total Phosphorus**	mg/L	Summer Mean Surface TP	0.28

Case B: Observational data available; using anoxic factor, survey data (surface area and emergent vegetation cover).
For this example, we used Model 7 of Tool 1B, which uses anoxic factor, pond surface area, and emergent vegetation cover as inputs.
This model produces an estimated value of **0.22 mg/L for TP** (and has a smaller standard error)**.**

7	[TP] = A*[AF] + B*[SAREA] + C*[EMERG] + D					**TP: Total Phosphorus**	mg/L	Season Mean Surface TP	**0.22**
	A = 0.150	**0.73**	**AF: Anoxic Factor**	Anoxic Factor over warm season [0.0 - 1.0]	Pond plans or drawings, records, site visit(s), or Tool 2-Oxygen			*Season: ~May - Nov*	
	B = -0.0249	**2.89**	**SAREA: Pond Surface Area**	Pond surface area, in acres [0.10 - 6.91 ac]	Pond plans or drawings, records				
	C = 0.00219	**0**	**EMERG: Emergent Veg Cover**	Percent of pond area covered by emergent vegetation [0 - 86%]	Aerial photos				
	D = 0.183								

The actual season (May – Nov) mean TP value at Alameda Pond over the duration of our studies was 0.21 mg/L

Application of Tool 3A - SedRel

Calculating Potential Anoxic Sediment Phosphate Release

Examples are shown for using available data (anoxic factor, duckweed cover) and sediment sampling data (organic matter, TP in the sediments).

Case A: 'Limited' data case. Assuming that only duckweed cover could be observed, Model 1 of Tool 3A could be used to predict potential anoxic sediment phosphate release for Alameda Pond. We use the value of duckweed cover (60%) estimated previously from Google Earth. This model produces an estimated value of **3.22 mg/m²/day for potential anoxic sediment phosphate release**.

Tool 3A-SedRel

Assessment Level: 1

Goal: Anoxic Sediment Release of Phosphorus (Potentail value under controlled lab setting)

Indicator of risk for release of sediment-bound P under anoxic conditions. Models on this sheet are for prediction of anoxic sediment release observed in 20 ponds (nearly 100 sediment cores) in laboratory column experiments, and indicate risk of internal P release in ponds but are not a complete estimate of anoxic P release that might be observed in the field.

The P release rates predicted here should be adjusted by observed or modeled pond anoxic factor (AF) to determine an effective anoxic P release rate (Tool 3B-SedRel).

		Model Parameter(s)				Model Output			
	Model	Value	Name	Definition [Range of Parameter in Reference Ponds]	Notes or Data Source*	Dependent Variable	Units	Description	Modeled Value
1	[PFLUX] = A*[DW] + B A = 0.036 B = 1.06	60	**DW: Duckweed Cover**	Percent of pond area covered by duckweed in midsummer, or averaged over season [0% - 98%]	Aerial photos, site visit(s)	**PFLUX: Phosphate flux**	mg/m²/day	Anoxic sediment P release rate, mg/m²/day	**3.22**

Case B: Sediment sampling data available; bulk sediment (~top 5 cm depth) analyzed for organic matter content (loss-on-ignition) and total phosphorus (TP, by ICP analysis). For this example, Model 4 of Tool 3B can be used, which uses sediment organic matter content and sediment TP as inputs. This model produces a **potential anoxic sediment phosphate release of 4.67 mg/m²/day.**

	Model	Value	Name	Definition	Notes or Data Source	Dependent Variable	Units	Description	Modeled Value
4	[PFLUX] = A*[OM] + B*[SEDTP] + C A = 0.014	28	**OM: Organic matter content**	Percent sediment organic matter content (determined by LOI), averaged over 0-4 cm sediment depth [4.9 - 79%]	Bulk or core sediment sampling	**PFLUX: Phosphate flux**	mg/m²/day	Anoxic sediment P release rate, mg/m²/day	**4.67**
	B = 4.59 C = -1.65	1.29	**SEDTP: Sediment TP**	Sediment total phosphorus (TP) mass, averaged over 0-4 cm sediment depth; mg P/g sediment (dry weight) [0.499 - 1.71 mg/g]	Bulk or core sediment sampling				

The actual (lab-measured) value of anoxic sediment phosphate release for Alameda Pond was 4.63 mg/m²/day

Application of Tool 3B - SedRel

Calculating Adjusted Anoxic Sediment Phosphate Release

Estimate Adjusted Anoxic Sediment Release of Phosphorus: Optional step, if pond anoxic factor is known (observational data or predicted by Tool 2).

Case A. 'Limited' data case. Assuming that only duckweed cover could be observed, Tool 2 was used to predict anoxic factor (AF) of 0.52 for Alameda Pond. The anoxic sediment phosphate release of 3.22 mg/m^2/day predicted by Model 1 of Tool 3A (using duckweed cover as input) adjusted by the AF yielded an anoxic sediment phosphate release of **1.67 mg/m^2/day under field conditions** in the Alameda Pond**.**

Tool 3B-SedRel

Assessment Level: 2

Goal: Adjusted Anoxic Sediment Release of Phosphorus

The sediment phosphorus release rate predicted by Tool 3A-SedRel can be adjusted here by field-observed anoxic factor or by modeled pond anoxic factor (Tool 2-Oxygen) to determine an effective anoxic P release rate. A pond with sediment characteristics prone to sediment P release (high P release rate) may function acceptably for TP removal provided it has low anoxia (low AF).
This tool can be used to determine the impact of combined risk factors of (1) sediment characteristics prone to sediment release, and (2) ponds prone to frequent or prolonged anoxia.

		Model Parameter(s)				Model Output			
	Model	Value	Name	Definition	Notes or Data Source*	Dependent Variable	Units	Description	Modeled Value
1	[PFLUX_ADJ] = [AF]*[PFLUX]	0.52	AF: Anoxic Factor	Time- and space-integrated estimate of sediment exposure to low oxygen conditions (< 2.0 mg/L); dimensionless	Pond plans or drawings, records, site visit(s), or using Tool 2-Oxygen	PFLUX_ADJ: Phosphate flux	mg/m^2/day	Anoxic sediment P release rate, mg/m^2/day *Adjusted for pond anoxic factor*	1.674
		3.22	PFLUX: Phosphate flux	Anoxic sediment P release rate, mg/m^2/day	Predicted anoxic sediment P release rate (Tool 3A-SedRel), or measured in laboratory				

Case B: Sediment Sampling data and Observational data available; using sediment characteristics and anoxic factor.
For this example, we calculated pond anoxic factor (AF) of 0.73 from the pond bathymetry and in situ dissolved oxygen (DO) profiles.
The anoxic sediment phosphate release predicted from Tool 3A (4.67 mg/m^2/day) was adjusted by the AF (0.73) to yield an anoxic sediment phosphate release of **3.41 mg/m^2/day under field conditions in the Alameda Pond.**

		Model Parameter(s)				Model Output			
	Model	Value	Name	Definition	Notes or Data Source*	Dependent Variable	Units	Description	Modeled Value
1	[PFLUX_ADJ] = [AF]*[PFLUX]	0.73	AF: Anoxic Factor	Time- and space-integrated estimate of sediment exposure to low oxygen conditions (< 2.0 mg/L); dimensionless	Pond plans or drawings, records, site visit(s), or using Tool 2-Oxygen	PFLUX_ADJ: Phosphate flux	mg/m^2/day	Anoxic sediment P release rate, mg/m^2/day *Adjusted for pond anoxic factor*	3.409
		4.67	PFLUX: Phosphate flux	Anoxic sediment P release rate, mg/m^2/day	Predicted anoxic sediment P release rate (Tool 3A-SedRel), or measured in laboratory				

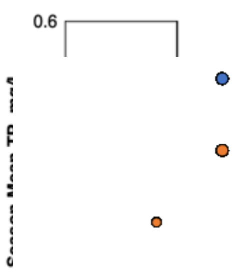
The anoxic sediment phosphate release under field conditions will provide an improved estimate of internal phosphorus loading in the pond.

Interpreting Results from All Tools

Tool 1-A: Medium/High Risk of High TP

Factor	Low	Medium	High	Unknown
	Factor Level (Risk of High TP)			
Construction Year		x		
	After 2010	*Before 1990*	*1990-2010*	
Underlying or Nearby Hydric Soils	x			
	0-40%	*NA*	*40-100%*	
Historic Wetland or Lake			x	
	No	*NA*	*Yes*	
Emergent Vegetation Cover	x			
	0-50%	*NA*	*>50%*	
Duckweed Cover			x	
	0-50%	*NA*	*>50%*	
Watershed Tree Canopy			x	
	< 20%	*20% - 60%*	*> 60%*	
Dominant Watershed Land Use			x	
	Comm	*Ind/Transp*	*Res*	

Tool 1-B: Pond TP Concentration (mg/L)



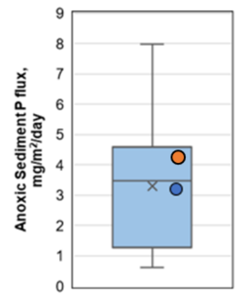
Box plot: Observed Mean TP across 25 study ponds (Dataset I; see Report)

- Estimate of Mean TP using modeled Anoxic Factor (Tool 2): **0.22 mg/L**
- Estimate of Mean TP using observed Anoxic Factor: **0.22 mg/L**

Interpretation: Predicted pond TP is close to average TP concentration in this dataset, so risk of high TP is likely moderate

(Actual Observed TP: 0.21 mg/L)

Tool 3-A: Sediment Phosphate Release Rate (Potential Value)



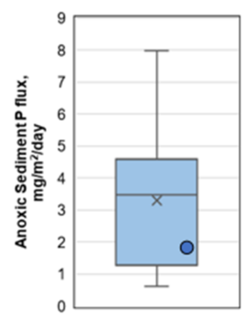
Box plot: Lab-observed release rates in 20 study ponds (Dataset III; see Report)

- Estimate of Anoxic P Release using Model 1: **3.22 mg/m²/d**
- Estimate of Anoxic P Release using Model 4: **4.67 mg/m²/d**

Interpretation: Potential sediment phosphate release rate is relatively high for this pond

(Actual observed rate in lab: 4.63 mg/m²/d)

Tool 3-A: Sediment Phosphate Release Rate (Adjusted for Field Conditions)



- Estimate of Anoxic P Release using Modeled AF: **1.67 mg/m²/d**
- Estimate of Anoxic P Release using Observed AF: **3.41 mg/m²/d**

Interpretation: A wide range of possible anoxic release rates are possible at this site, but a combination of persistent anoxia (AF ~ 0.73 from observations) and sediment prone to higher phosphate release indicate a pond with high risk of internal loading, even with a moderate surface water TP concentration.

Together, these factors suggest pond water quality will likely get worse with time due to internal loading.

Appendix D
Stormwater Pond Maintenance and Wetland Management Guide

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1 Introduction

This document presents a guide on maintenance strategies for primarily managing the phosphorus water quality and the sediment concentration of polycyclic aromatic hydrocarbons (PAHs) in urban stormwater ponds and wetlands. It has been found that many ponds that treat stormwater have not been reducing phosphorus concentration at expected levels (Taguchi et al. 2020), and that PAHs in the sediment of roughly one third of the ponds that treat stormwater require placement in a confined disposal facility after dredging activities (Huang et al. 2019). Since phosphorus retention in ponds is a major water quality control for Minnesota's streams and lakes, and all ponds eventually need to be dredged, these are two important issues for the management of ponds that treat stormwater.

2 Phosphorus Retention by Ponds that Treat Stormwater

This section of the guide is intended to be used in conjunction with the Pond Assessment Tool that was developed for the evaluation of ponds for specific risks of high surface water total phosphorus (TP), anoxia, and internal phosphorus loading (Janke et al. 2023). We use results of the modeling exercise on various pond maintenance actions (Taguchi et al. 2022) to recommend specific strategies for addressing risk indicators that directly or indirectly impact the phosphorus water quality including the oxygen status in ponds.

2.1 Description of Risk Indicators

Table D.1 lists all potential factors considered as important to pond phosphorus dynamics, along with likely mechanism(s) of influence and a brief description of how we defined or assessed the indicators. A broad range of processes are described by these indicators, including phosphorus or sediment inputs to ponds, oxygen dynamics and vertical transport within ponds or wetlands (through mixing or stratification processes), sediment characteristics, and legacies of sediment, organic matter, and phosphorus inputs. The definitions and analyses investigating the influence of these indicators are provided in Janke et al. (2023) and in previous research projects (Taguchi et al. 2018; Janke et al. 2021).

Table D.1 Potential indicators of risk for poor performance for retention of phosphorus by ponds considered by current analysis. Indicators that emerged as important, or significant in regression analyses are shown in bold text.

Indicator	Potential Effect on P	Description
Anoxic Factor	Oxygen Dynamics, Sediment Release	Fraction of sediments exposed to anoxic overlying water during monitoring season
Shoreline Canopy Cover	Wind Sheltering/Oxygen Dynamics, Litter Inputs	Fraction of land in 25m buffer around pond classified as canopy and/or buildings
Shoreline Canopy Height	Wind Sheltering/Oxygen Dynamics, Litter Inputs	Mean height of canopy and building in 25 m buffer around pond, from LIDAR
Embankment Height	Wind Sheltering/Oxygen Dynamics	Mean height of land relative to water surface in 25 m buffer around pond, from LIDAR
Wind Reduction	Wind Sheltering/Oxygen Dynamics	Mean reduction in wind speed observed at pond relative to that at nearest airport
Mixing Frequency	Oxygen Dynamics, Vertical Transport	Fraction of days that pond was mixed, based on a minimum RTRM threshold
Free floating plant (mainly duckweed)	Wind Sheltering/Oxygen dynamics, Organic matter inputs	Fraction of pond surface area that is covered by duckweed
Emergent Vegetation Cover	Oxygen dynamics, organic matter, mixing	Fraction of pond surface area covered by rooted, emergent macrophytes
Sediment TP	Sediment P Release	TP concentration in the upper 4 cm of sediments
Sediment organic matter	Sediment P Release	Organic matter concentration in the upper 4 cm of sediments
Sediment Fe:P	Sediment P Release	Total Iron to Total Phosphorus mass ratio in the upper 4 cm of sediments
Sediment Redox-P & Labile Organic P	Sediment P Release	Redox-P or Labile Organic P in the upper 4 cm of sediments
Pond age	Sediment Release, Organic matter accrual, Litter inputs	Pond age since construction or connection to storm drains relative to year 2021
Pond area	Oxygen Dynamics, Vertical Transport, Hydrology	Surface area of the pond
Maximum depth	Oxygen Dynamics	Maximum depth measured from the pond surface
Mean depth	Oxygen Dynamics	Mean of various depths measured from the pond surface
Land Cover	Phosphorus and sediment inputs to pond	Land cover (pavement, grass, canopy) in a 500m vicinity of pond

Indicator	Potential Effect on P	Description
Land Use	Phosphorus and sediment inputs to pond	Land use (residential, commercial, etc.) in 500m vicinity of pond
Watershed area	External nutrient inputs, Litter inputs, Sediment Release	Drainage area contributing to pond

Only a subset of parameters from Table 1 represented the most useful and strong predictors of pond surface water TP, anoxia, or sediment phosphate release and were used in the Pond Assessment Tool developed in Janke et al. (2023). The reasons for exclusion of some parameters include relative data scarcity (e.g., watershed area, mean depth), high covariance (e.g., canopy height and canopy cover), or weaker predictive power (e.g., embankment height, land cover such as grass or pavement).

2.2 Pond Assessment Tool Summary

The Pond Assessment Tool was developed to assess three main goals:

1. To estimate pond surface water total phosphorus (TP) concentration and risk
2. To determine pond oxygen status (risk of anoxia)
3. To estimate pond sediment phosphate (P) release

Figure D.1 illustrates the assessment process steps for the identified goals. The tool includes several model options, using input data that are either basic (drawings, plans, aerial photographs, site visits) or sampling and monitoring data (periodic grab sampling/profiling of pond water and sediment sampling, or using continuous monitoring stations or loggers installed in the pond). The assessment tool was developed using several urban pond or wetland sediment and water quality datasets, which have been collected and assembled by the authors in this and previous projects. These datasets included intensive monitoring and sampling efforts, widespread sampling surveys of ponds, as well as sediment sampling and coring for laboratory incubation studies of phosphorus release (Taguchi et al. 2018, 2020; Janke et al. 2021, 2022a, 2023).

Goal 1: Surface Water TP

The tool employs two levels of assessment for this goal, based on the type of input data. The first assessment (Tool 1-A) is a method of screening ponds for levels of indicators related to risk of high pond water TP. This assessment utilizes basic site data (such as duckweed cover or presence of hydric soils). The second assessment (Tool 1-B) provides predictions (numerical estimates) of pond phosphorus concentrations, using input of more detailed sampling and/or monitoring data that require periodic data collection at the pond sites along with basic site data (Table 2).

Goal 2: Oxygen Status

The assessment of pond oxygen status (Tool 2) predicts pond anoxia in terms of anoxic factor, a measure of the extent and duration of exposure of pond sediments to low oxygen conditions, which can indicate risk of potential sediment phosphorus release. Anoxic factor can be estimated from sampling and monitoring data (dissolved oxygen profiles) and basic site data (pond bathymetry), or estimated from regression models incorporated into Tool 2. Tool 2 also provides estimates of water column dissolved oxygen from site or watershed characteristics. Tool 2 serves as an input to predictions of Goal 1 (Pond TP concentration) and Goal 3 (Sediment phosphate release).

Goal 3: Anoxic Sediment Phosphate Release

The tool has two steps for the prediction of phosphorus release from pond sediments that contribute to internal phosphorus loading in ponds. The first step (Tool 3-A) provides models to predict potential anoxic sediment phosphate release as observed with laboratory columns of pond sediment and water, using sediment and water quality sampling data and basic site data (e.g., duckweed cover, anoxic factor; Table 2). Tool 3-A includes models that require input sediment parameters that are relatively simple to analyze in an analytical services laboratory (sediment TP, organic content, metal concentrations), as well as models that require input parameters from rigorous sediment analysis (redox-P, mobile-P). If a rigorous sediment analysis is not performed, models are provided to estimate the rigorous sediment parameters using simple sediment parameters as inputs. Given the variable oxygen status across ponds due to a multitude of factors, we recognize the importance of pond oxygen status on the in situ sediment phosphorus release rate and hence the internal load generated within the pond. The second step (Tool 3-B) adjusts the potential anoxic phosphate release (output from Tool 3-A) using pond anoxic factor (observed in field, or modeled in Tool 2) to predict the anoxic sediment phosphate release that might be observed under field conditions.

The models in the assessment toolbox are shown in Table D.2 along with relevant model descriptions of goodness of fit. Note that several parameters (e.g., duckweed) are used in both the simple tool and the predictive tools. Note also that not all parameters are needed to use the tool; certain parameters (such as duckweed cover, pond age, and land use) may be adequate to provide some information and understanding of risks for high TP, anoxia, or internal loading in ponds.

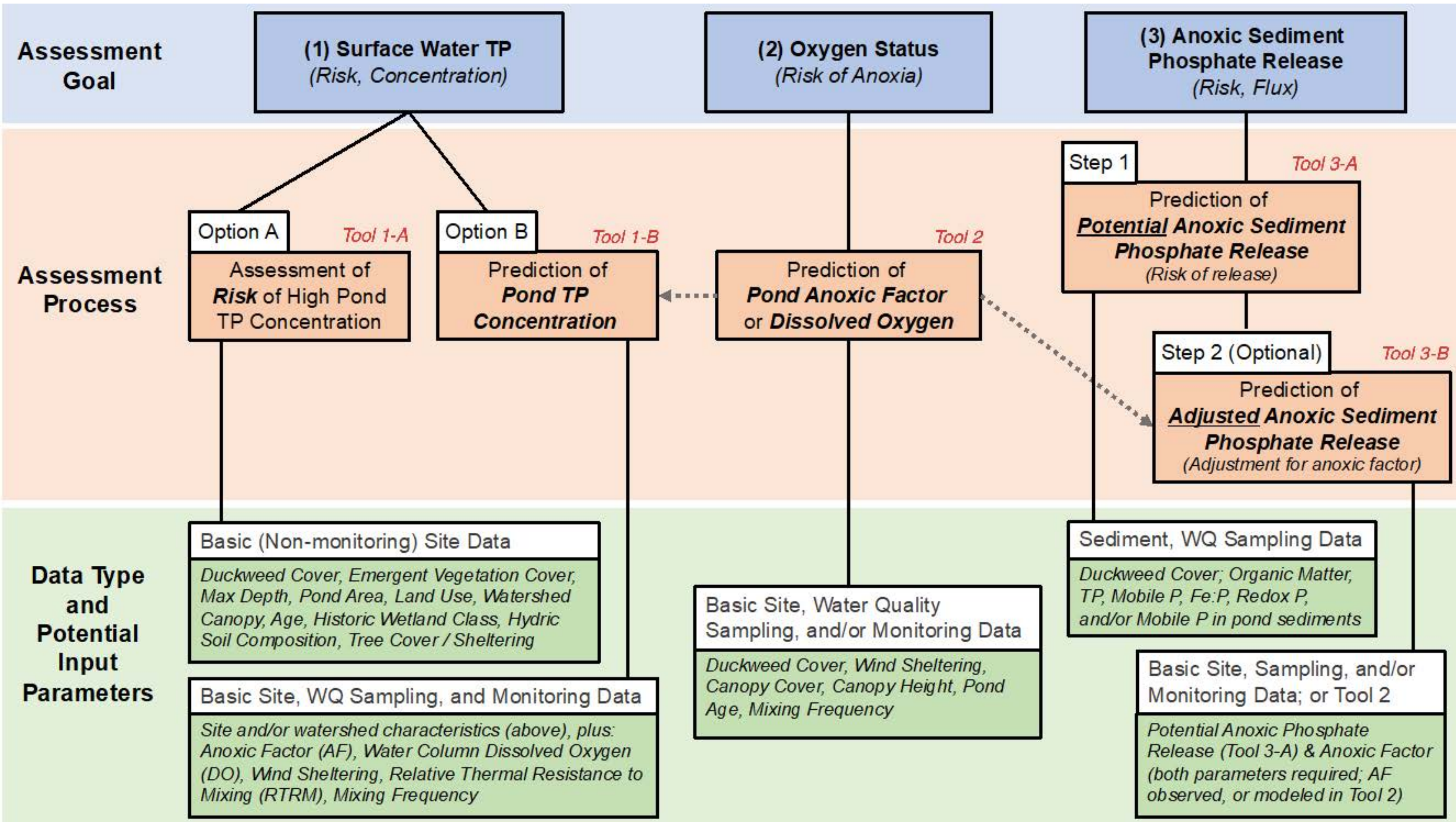

Figure D.1 Flowchart of the Pond Assessment Tool showing the assessment goals, assessment process for each goal, and the potential input data parameters for the assessment process.

Table D.2 Simple and Multiple Linear Regression Models for Surface Water TP (mg/L), Anoxic Factor, Water Column Dissolved Oxygen (mg/L), and Sediment Phosphate Release (mg/m^2/day) included in the Assessment Tool. Parameter definitions included in the table below.

	Std Error	Adj. R^2	p-value
Model, Summer (June - Aug) Mean Surface TP (mg/L)			
0.237*(AF) + 0.146	0.11	0.37	0.0021
0.253*(AF) + 0.068*(DMEAN/SAREA^0.5) + 0.290	0.11	0.41	0.0094
Model, Season (June - Aug) Mean Surface TP (mg/L)			
0.0011*(DW) + 0.0020*(EMERG) + 0.154	0.081	0.44	0.0038
0.0015*(DW) - 0.077*(DMAX) + 0.290	0.082	0.42	0.0049
-0.0265*(SAREA) + 0.00259*(CPYCVR) + 0.00214*(EMERG) + 0.152	0.079	0.50	0.0018
0.202*(AF) - 0.0846*(DMAX) + 0.290	0.072	0.56	0.008
0.150*(AF) - 0.0249*(SAREA) + 0.00219*(EMERG) + 0.183	0.062	0.67	0.0003
-0.0208*(DO_WC) + 0.00150*(EMERG) + 0.260	0.081	0.36	0.00095
0.049*(PFLUX_ADJ) + 0.13	0.084	0.47	0.0014
Model, Summer (June - Aug) Anoxic Factor			
0.00802*(DW) + 0.0408	0.19	0.72	<0.0001
0.00824*(DW) - 0.058*(SAREA) + 0.129	0.18	0.63	<0.0001
Model, Season (May - Oct) Anoxic Factor			
0.00789*(DW) + 0.055	0.18	0.76	<0.0001
-0.112*(DO_WC) + 0.735	0.12	0.87	<0.0001
-0.120*(DO_WC) + 0.00271*(CPYCVR) + 0.651	0.10	0.90	<0.0001
Model, Season (May - Oct) Water Column Dissolved Oxygen (mg/L)			
-13.0*(WINDSH) + 4.12*(MIXED) + 12.1	0.86	0.89	<0.0001
1.51*(DMAX) + 7.57*(MIXED) - 3.67	1.11	0.82	0.0002
Model, Anoxic Sediment Phosphate Release (mg/m^2/day; lab conditions)			
0.0359*(DW) + 1.06	0.94	0.72	1.8E-05
3.36*(AF) + 1.40	1.17	0.54	0.0012
-0.429*(DO_WC) + 4.4	1.22	0.51	0.0020
0.014*(OM) + 4.59*(SED TP) - 1.65	1.36	0.55	0.0007
12.5*(REDOXP) + 1.32	1.48	0.51	0.0004
6.85*(MOBP) + 0.413	1.60	0.40	0.0016
11.7*(REDOXP) + 2.14*(LABORGP) + 0.876	1.50	0.47	0.0017
REDOXP =-0.012*(FE:P) + 0.442	0.086	0.55	0.0007
MOBP = -0.024*(FE:P) + 0.973	0.11	0.74	9.7E-06
MOBP = 0.0058*(OM) + 0.393*(SEDTP) - 0.101	0.073	0.86	1.6E-08
LABORGP = 0.0068*(OM) + 0.0899	0.078	0.66	1.4E-05

Parameter Definitions	
AF	Anoxic Factor
DMEAN	Mean Depth, m
DMAX	Pond Max Depth, m
SAREA	Pond Surface Area, m^2
EMERG	Percent Pond Covered by Emergent Veg.
DW	Percent Pond Area Covered by Duckweed
CPYCVR	Percent of Pond Shore Covered by Canopy (LIDAR)
DO_WC	Water Column DO, Mean over Season (May - Oct), mg/L
PFLUX_ADJ	Adjusted Phosphate Flux in $mg/m^2/day$ (Tool 3-B)
WINDSH	Observed Wind Reduction, pond vs. airport
MIXED	Fraction of Days Pond Mixed (based on RTRM)
OM	Sediment Organic Matter, %
SEDTP	Sediment TP, mg/g
REDOXP	Sediment Redox P, mg/g
MOBP	Sediment Mobile P, mg/g
LABORGP	Sediment Labile Organic P, mg/g
FE:P	Sediment Fe:P (mass ratio)

2.3 Pond Maintenance Options to Address Indicators for Risk of Poor Phosphorus Retention

This section provides a description of potential maintenance options for ponds with high levels of risk indicators for poor phosphorus removal performance (see Table D.1 above). In Table D.3 below, these potential maintenance options are cross-referenced to the specific risk indicators and related assessment goals of the Pond Assessment Tool (Figure D.1). The pond maintenance options considered for improving conditions for phosphorus retention include the following:

Conventional / Tested Maintenance Options:

1. **Aeration:** typically consist of mechanical practices (e.g., aerators) aimed at de-stratifying the pond water column and exposing sediments to oxygen.
2. **Dredging:** removal of accumulated sediment and associated organic matter and phosphorus, to improve pond storage and reduce risk of internal loading.
3. **Chemical Treatment:** application of substances such as alum or iron filings to trap phosphorus in pond sediments and prevent internal loading.

4. **Pond Expansion:** increasing pond surface area to promote wind-driven mixing and increase hydraulic residence time.
5. **Outlet Modification:** practices designed to draw water from lower in the water column to remove low-oxygen water and promote de-stratification; also includes practices such as screens on weirs to prevent export of floating vegetation, and filtration practices such iron-enhanced filter benches to polish dissolved phosphorus from outflows.

Experimental Maintenance Options (Require Further Research or Testing):

6. **Canopy Thinning:** removal or thinning of shoreline tree canopy to promote wind-driven mixing.
7. **Removal of Free-Floating Plants:** mechanical removal or chemical treatment of floating vegetation (such as duckweed) to remove phosphorus and organic matter associated with biomass, and to promote higher dissolved oxygen levels.
8. **Vegetation Harvesting**: harvesting of emergent vegetation such as cattails to remove phosphorus and promote mixing.
9. **Adaptive Control / Real Time Control:** remotely operated valves on outlets that can draw down water levels prior to incoming storms to improve storage and retention.

Details of each maintenance option are provided in this section, with several of the more established options (#1 through #5 in the list above) relying on the work of a recent pond modeling study (Taguchi et al. 2022). The final four treatment and maintenance options in the list (#6 through #9) are considered experimental or are still in the early stages of testing and implementation. A few of these options were addressed in part by the earlier modeling study, with further information coming from a broad stormwater pond literature review (Janke et al. 2022b) and from additional literature reviewed for this document. A summary of the earlier modeling study (Taguchi et al. 2022) is provided in the next section (Section 2.4) for reference, along with an analysis of cost-effectiveness for several of the maintenance options.

Table D.3 List of known maintenance options to address pond risk indicators for poor phosphorus retention, specific to the goals of the Pond Assessment Tool: (1) high water column TP ("water TP"), (2) low dissolved oxygen ("Oxygen"), and (3) sediment SRP release ("Sed P Release"). The last three indicators in the list are related to water retention (Janke et al. 2022a) and not currently included in the Assessment Tool. Risk indicators (Column 1) include those used in the Pond Assessment Tool (see Table 2 for descriptions), with relevant assessment tool goals given in Column 2 (see also Figure 1). * indicates maintenance options that are largely untested or experimental, and require further research before the practice could be definitively recommended (see Section 2.5 for details).

Risk Indicator [see Table 1]	Risk Category (Assessment Tool Goal) [see Figure 1]	Maintenance Options
Anoxic Factor	(1) Water TP, (2) Oxygen, (3) Sed P Release	Aeration; Dredging; Watershed Management
Water Column DO	(1) Water TP, (2) Oxygen, (3) Sed P Release	Aeration; Dredging; Watershed Management
Free floating plants (FFP) e.g., duckweed	(1) Water TP, (2) Oxygen, (3) Sed P Release	*FFP Removal*;* Aeration; Outlet Filtration
Emergent Vegetation Cover	(1) Water TP	Vegetation Removal*
Pond age	(1) Water TP	Dredging; Chemical Sediment Treatment
Land Cover	(1) Water TP	Watershed Management
Land Use	(1) Water TP	Watershed Management
Hydric Soils	(1) Water TP	NA
Historic Wetland / Waterbody	(1) Water TP	NA
Shoreline Canopy Cover	(1) Water TP, (2) Oxygen	*Canopy Thinning**
Pond area	(1) Water TP, (2) Oxygen	Pond Expansion
Maximum depth	(1) Water TP, (2) Oxygen	Dredging
Wind Reduction	(2) Oxygen	*Canopy Thinning**
Mixing Frequency	(2) Oxygen	Aeration; *Canopy Thinning*; FFP Removal**

Risk Indicator [see Table 1]	Risk Category (Assessment Tool Goal) [see Figure 1]	Maintenance Options
Sediment TP	(3) Sed P Release	Chemical Sediment Treatment; Dredging
Sediment organic matter	(3) Sed P Release	Chemical Sediment Treatment; Dredging
Sediment Fe:P	(3) Sed P Release	Chemical Sediment Treatment; Dredging
Sediment Redox-P & Labile Organic P	(3) Sed P Release	Chemical Sediment Treatment; Dredging
Drawdown Rate	*Retention*	Dredging; Outlet Modification; *Adaptive Control**
Mean Relative Water Depth	*Retention*	Dredging; Outlet Modification; *Adaptive Control*;* Watershed Management
Outflow/Inflow Ratio	*Retention*	Dredging; Outlet Modification; *Adaptive Control*;* Watershed Management

2.3.1 Aeration

Aeration of a pond involves de-stratifying the pond water column to enhance oxygen delivery to the pond bottom, which can promote redox- and biologically- driven binding of phosphorus to sediments (in particular, binding of soluble reactive phosphorus to iron oxides). Several options may be considered, but generally involve the use of pumps and diffusers to bubble atmospheric air into the water column from the pond bottom, introducing oxygen and promoting mixing from the pond surface. Note that fountains are not considered aerators, as they are largely ornamental and do not provide much mixing except at the very pond surface. Other methods that could provide aeration benefits include thinning of the tree canopy and removal of free floating plants (duckweed), if present, which would potentially enhance wind-driven diffusion. These options are considered in more detail separately.

Performance and cost-effectiveness of a mechanical aeration approach were considered by Taguchi et al. (2022) and presented in Figure D.2 below. In the four modeled ponds of that study, mechanical aeration designed to destratify the pond was an effective treatment for water column concentration of total phosphorus (TP), with modeled reductions of 2% to 54% (Figure D.2a), which translated to roughly 2% to 56% reductions in hydrologically exported mass of TP (Figure D.2b). Absolute value of mass reduction at the Langton and Minnetonka ponds were low, due to rarity of export from these two ponds. We note also that the Shoreview Commons pond had sediment releases of phosphate under oxic conditions, so aeration alone was less effective because oxygenation of the water above the sediments was not enough to eliminate sediment phosphate release. Aeration still resulted in a substantial reduction in TP export and TP concentration in the Shoreview Commons pond, however.

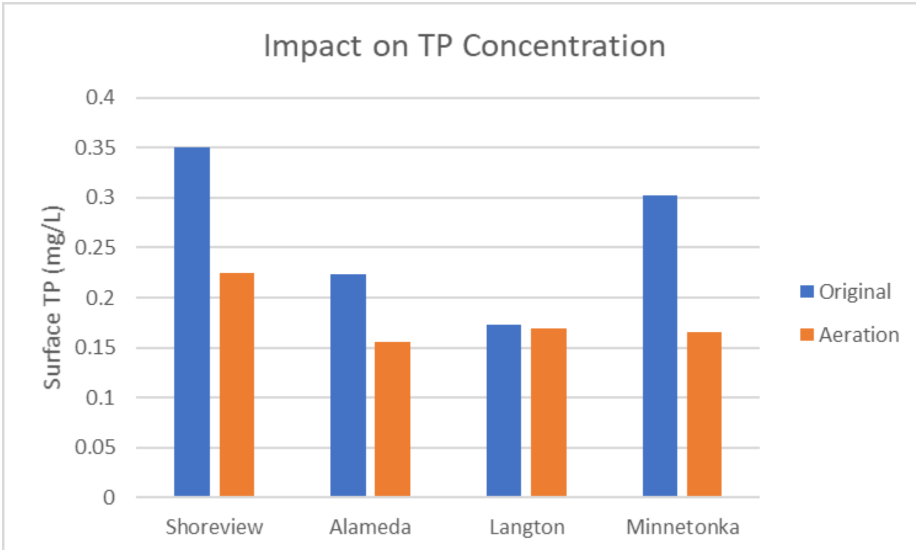

Figure D.2a Bar plot of simulation results of mean surface total phosphorus (TP) concentrations for each pond model under mechanical aeration scenarios (Taguchi et al. 2022).

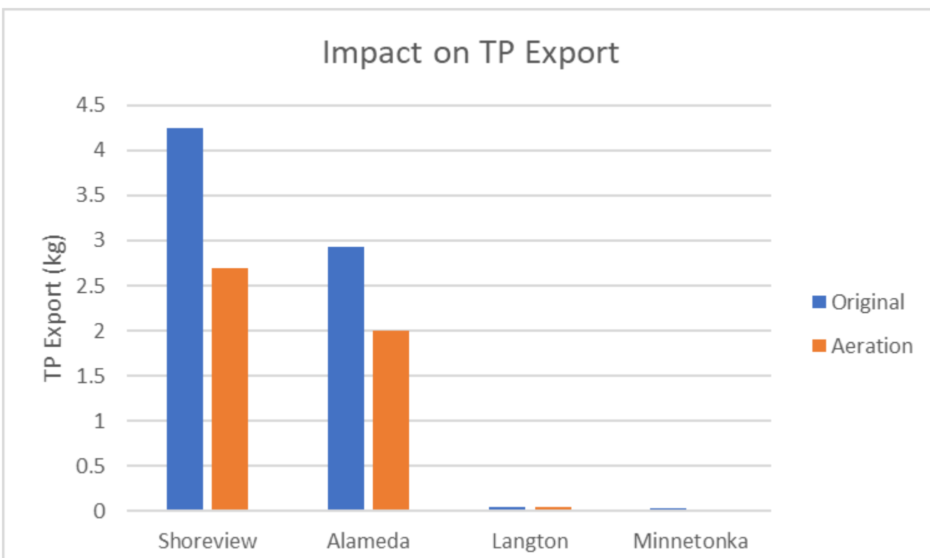

Figure D.2b Bar plot of simulation results of cumulative total phosphorus (TP) export mass for each pond model under mechanical aeration scenarios (Taguchi et al. 2022).

2.3.2 Dredging

Dredging is a maintenance practice that affects pond bathymetry through mechanical removal of accumulated sediments, as well as providing a permanent removal of phosphorus and organic matter from the ponds. Pond bathymetry or geometry (e.g., volume, depth, surface area) can impact the hydrodynamics of ponds through effects on mixing, stratification, and hydrologic retention, with resulting impacts to oxygen and phosphorus dynamics. A key issue and research need for dredging is the disposal of dredged sediment, which is often contaminated with heavy metals and PAHs. This contamination issue is addressed in a short literature review in Section 3.

In the modeling study (Taguchi et al. 2022), the impacts of dredging were assessed through changes to pond geometry; an aging pond was simulated through "filling" of sediment to decrease depth and volume, while the "dredging" scenario was simulated by an increase in depth and volume. TP export increased as ponds aged and filled in (original" vs. "filled"; Figure D.3a) and decreased after dredging ("original" vs. "dredged"; Figure D.3a), with observed reductions of 15% to 31% TP export across three of the ponds. The impact on TP concentration in the pond was more variable (Figure D.3b), but were generally lower in the dredged scenario vs. current conditions. Two other scenarios of pond bathymetry modification were simulated (expansion of surface area ("shallow") and reduction of surface area ("deep"); these are considered in more detail in Section 2.3.5.

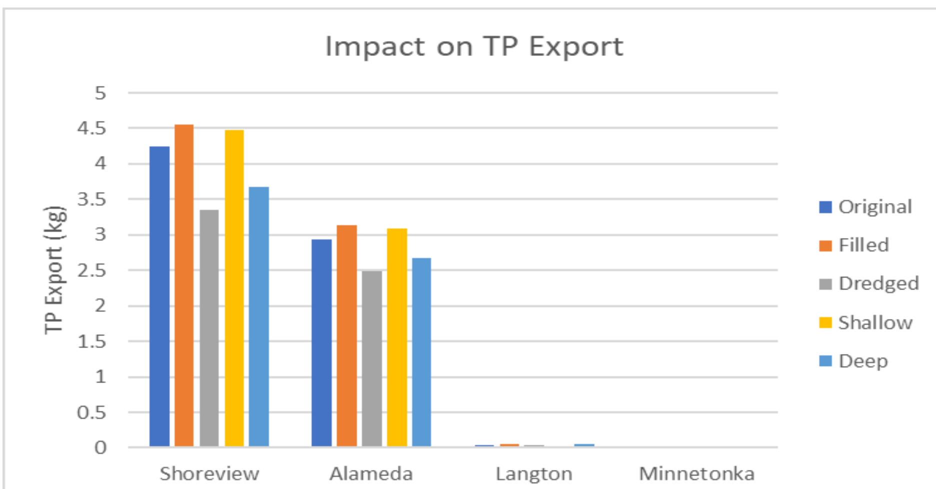

Figure D.3a Bar plot of simulation results of cumulative total phosphorus (TP) export mass for each pond model under various bathymetry modification scenarios. The Minnetonka pond model becomes unstable under the bathymetry modification scenarios, and thus no scenario results are shown on the bar plot (Taguchi et al. 2022).

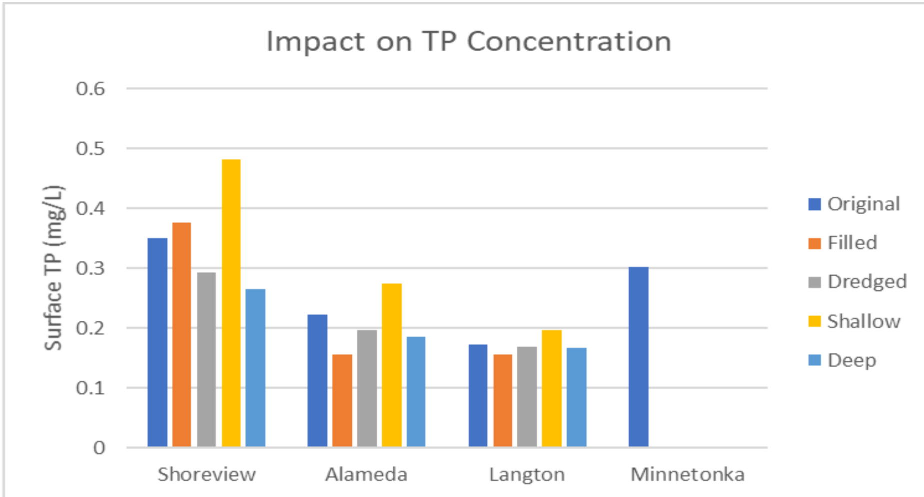

Figure D.3b Bar plot of simulation results of mean surface total phosphorus (TP) concentrations for each pond model under various bathymetry modification scenarios. The Minnetonka pond model becomes unstable under the bathymetry modification scenarios, and thus no scenario results are shown on the bar plot (Taguchi et al. 2022).

2.3.3 Watershed Management

Watershed management includes methods that reduce inflow concentrations and volumes (typically structural BMP practices). In the Taguchi et al. (2022) modeling study, these methods were effective for reducing pond TP export in the four study ponds (Figure D.4a), which was expected since the stormwater TP inflows were a major component of the overall TP mass balance in each pond. Reducing inflow volumes (e.g., through the installation of infiltration practices) led to the greatest benefit for reduction of TP export, with simulated reductions of 50% to 100%. However, TP concentrations in the ponds increased slightly with reduced volume of inflows since constituents in the pond water were not as diluted by inflows, which had lower TP concentration relative to pond water (Figure D.4b). This approach resulted in very low TP export but would be problematic for ponds treated as amenities where pond water quality is also a priority. Reducing inflow TP concentrations without modifying inflow volumes, which might result from source-reduction practices such as street-sweeping or phosphorus fertilizer bans, produced smaller but more predictable benefits to reduction of in-pond concentrations and pond TP export (roughly 2% to 15% reductions across ponds).

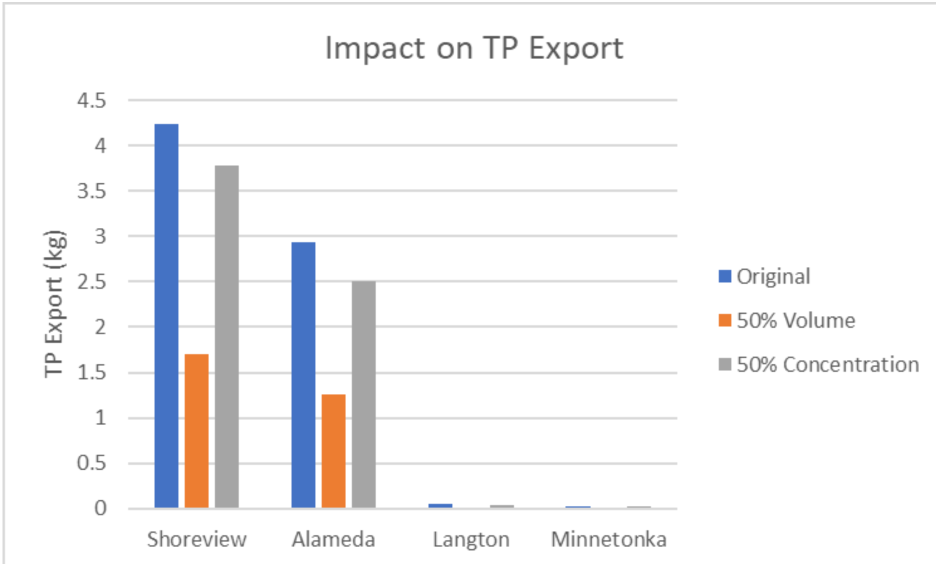

Figure D.4a Bar plot of simulation results of cumulative total phosphorus (TP) export mass for each pond model under various watershed-based treatment scenarios (Taguchi et al. 2022).

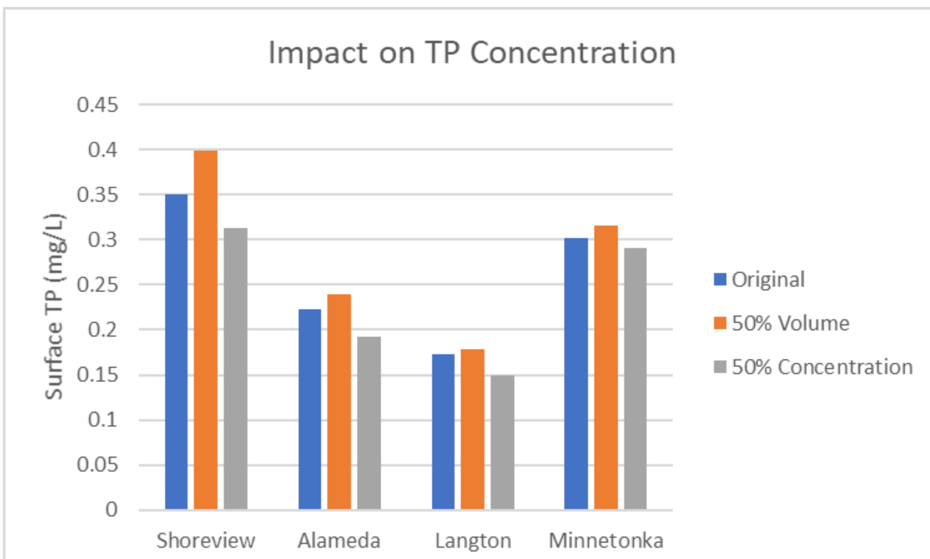

Figure D.4b Bar plot of simulation results of mean surface total phosphorus (TP) concentrations for each pond model under various watershed-based treatment scenarios (Taguchi et al. 2022).

2.3.4 Chemical Sediment Treatment

Under anoxic conditions (low dissolved oxygen concentration) with relatively high sediment phosphate release rates, sediment treatment with chemicals or other amendments to reduce internal phosphorus loading would be a good option to reduce water column total phosphorus concentration and associated algal and floating plant growth. Several chemical treatment options have been tested on ponds to date, including iron filings (Natarajan and Gulliver 2022), alum (Osgood 2012), and spent lime (Kuster et al. 2022; Wilson 2022). In the modeling study (Taguchi et al. 2022), treatment with alum and iron filings were both considered and found to be effective for reducing TP export from the ponds (2% to 43% reduction; Figure 5a) and for reducing in-pond TP concentrations (1% to 42% reduction; Figure 5b). In the Minnetonka and Langton ponds, absolute impacts to TP export were low due to relatively low outflow rates from the ponds. Further, sediment treatment to reduce phosphate release is generally not an effective way to treat newer ponds, such as the Langton pond, simply because their sediment phosphate release rate is generally low (Figure D.5a and 5b).

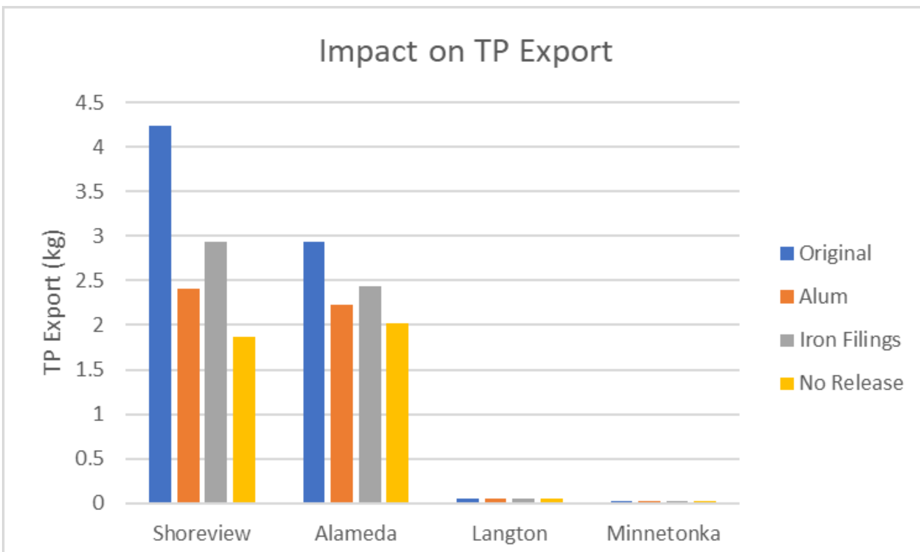

Figure D.5a Bar plot of simulation results of cumulative total phosphorus (TP) export mass for each pond model under various sediment chemical treatment scenarios (Taguchi et al. 2022).

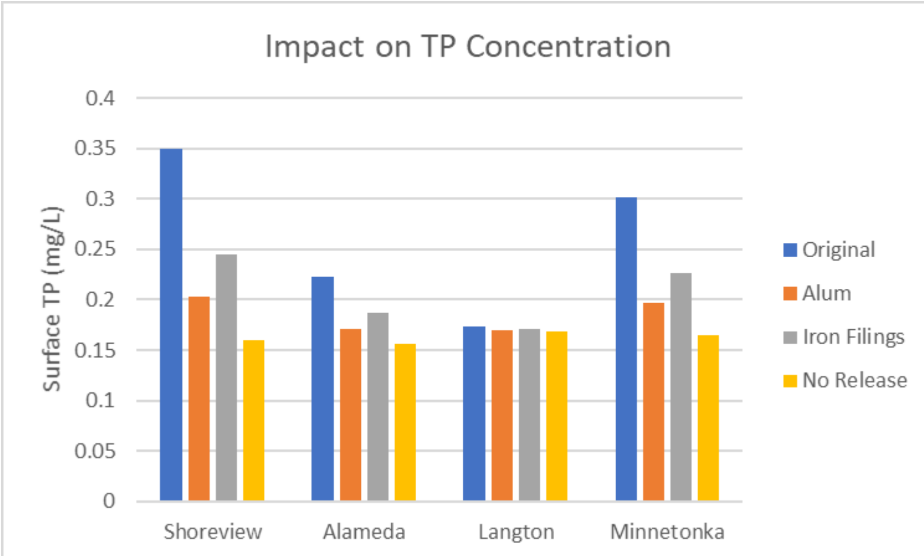

Figure D.5b Bar plot of simulation results of mean surface total phosphorus (TP) concentrations for each pond model under various sediment chemical treatment scenarios (Taguchi et al. 2022).

2.3.5 Pond Expansion

Pond expansion is a management option to increase pond surface area, which would promote greater exposure to wind and provide benefits of greater oxygenation and perhaps suppression of free floating plants. Benthic area would also be increased in the expansion scenario. We note that this is likely a less feasible maintenance operation in most cases due to the additional land requirements needed to expand a pond, and might be better considered as a "re-design" of ponds.

In the modeling study (Taguchi et al. 2022), two pond bathymetry modifications were considered: an expansion in which the pond volume was unchanged but the pond was made shallower ("shallow" in Figure D.4) as well as a contraction in which pond volume was similarly unchanged but made deeper ("Deep" in Figure D.4). Building a shallower pond with greater surface area ("shallow" scenario) produced a similar but slightly lower TP export relative to the "filled" pond, but resulted in higher in-pond TP concentrations due to greater benthic area over which to contribute sediment SRP release. In the contraction scenario (a deeper pond with less surface area; "deep" in Figure D.4) lower TP export and concentrations were simulated than in the original ponds, likely due to higher stratification strength and less water column mixing.

2.3.6 Outlet Modification

We considered two scenarios of outlet modification: (1) an iron-enhanced sand filter bench, used to remove soluble phosphorus as water leaves the pond, and (2) the modification of pond outlet structures such that water is withdrawn from the pond at different elevations in the water column. The goal of the latter approach is to remove low-oxygen water from the bottom of the pond to weaken stratification and draw more oxygenated water to the sediments, which is expected to provide a reduction in sediment-released P.

Iron-enhanced sand filters (IESF) are increasingly being used as an outflow treatment or polishing practice in stormwater ponds due to their ability to remove dissolved P (Erickson et al. 2018), which could help alleviate the effects of internal P release. Taguchi et al. (2022) simulated an IESF bench sized to treat 0.3 m (1 ft) of ponding depth in the study ponds, which provided a TP reduction of approximately 65% in the ponds (Figure D.6). Filter size was not varied in the simulation, and construction of a filter as used in the model would be potentially costly. Cost-effectiveness of all management options is addressed in the next section.

For the second modification (withdrawal depth), three elevations within the water column were considered by the modeling study of Taguchi et al. (2022): surface, middle, and bottom of the water column. Top (surface) was considered the baseline scenario. Results of the model study suggested very little positive impact from a bottom- or middle-withdrawal strategy, with TP export actually increasing by 1% to 9% relative to baseline in these scenarios across the modeled ponds. The likely mechanism of the increased export was withdrawal and export of sediment-released P; therefore, **the withdrawal-depth modification is not likely an effective strategy in ponds with high anoxic factors or suspected of having high rates of sediment P release.**

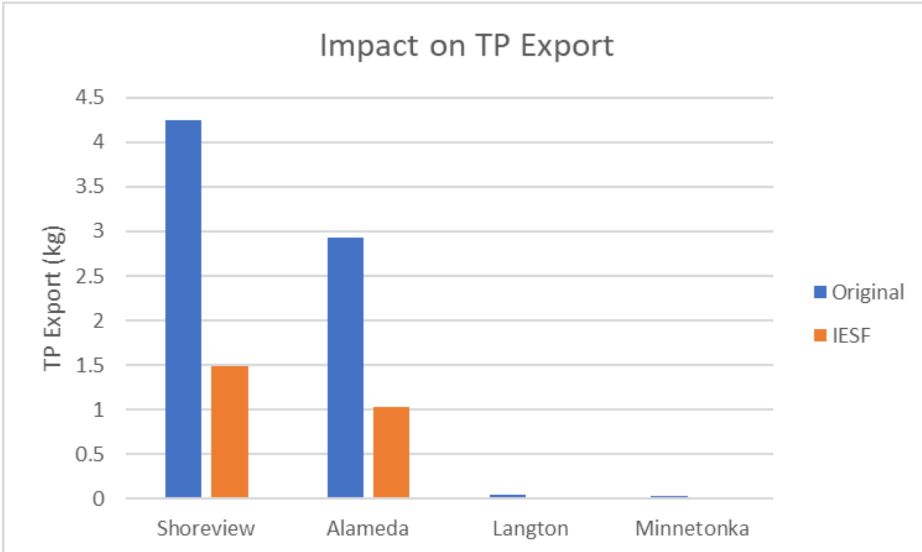

Figure D.6 Bar plot of simulation results of cumulative total phosphorus (TP) export mass for each pond model under iron-enhanced sand filter (IESF) bench implementation scenarios. Taguchi et al. (2022).

2.3.7 Canopy Thinning – Experimental

Wind plays an important role in energy transfer that results in the mechanical mixing of surface waters. The benefit of mixing to stormwater ponds is reduced stratification strength and exposure of more of the pond to oxygen, which should prevent anoxic sediment release of phosphorus. Taguchi et al. (2022) simulated several scenarios of wind sheltering reduction through removal of tree canopy around ponds, using a combination of the pond water quality model and a three-dimensional fluid dynamic model to simulate wind speed distribution above the pond. Scenarios considered included a 50% reduction and 100% reduction of tree sheltering, as well as selective removal of trees (along one or two shorelines oriented with the prevailing wind direction) on the Shoreview pond.

It was found that reduction of wind sheltering did not provide much additional mixing of the ponds, and therefore did not substantially reduce total water column phosphorus concentration (Figure D.7). This could result from one or all of three observations: 1) ponds have a short wind fetch, relative to lakes; 2) the banks on the pond cause sufficient separation of the wind and sheltering of the ponds to reduce the potential wind shear; and 3) the upwind roughness, such as houses, trees around the houses and other buildings, have a fairly large effect on the wind's ability to generate shear stress on the ponds. Wind has a greater effect on ponds more exposed to wind, such as the Langton pond, where wind sheltering reduction scenarios resulted in the Langton pond having a decrease in anoxic days and an increase in oxic days owing to increased wind mixing. **The influence of sheltering on mixing and DO requires further investigation.**

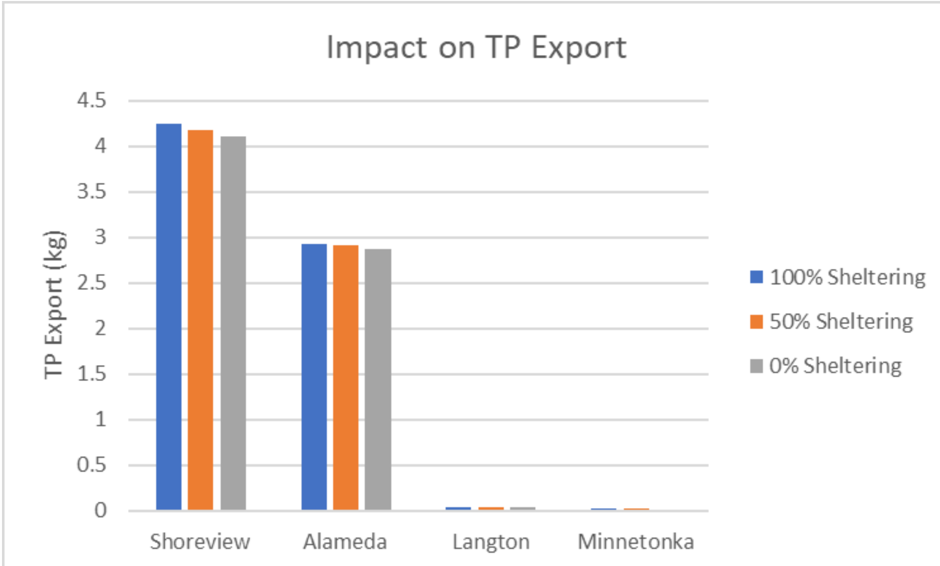

Figure D.7a Bar plot of simulation results of cumulative total phosphorus (TP) export mass for each pond model under various wind sheltering reduction scenarios (Taguchi et al. 2022).

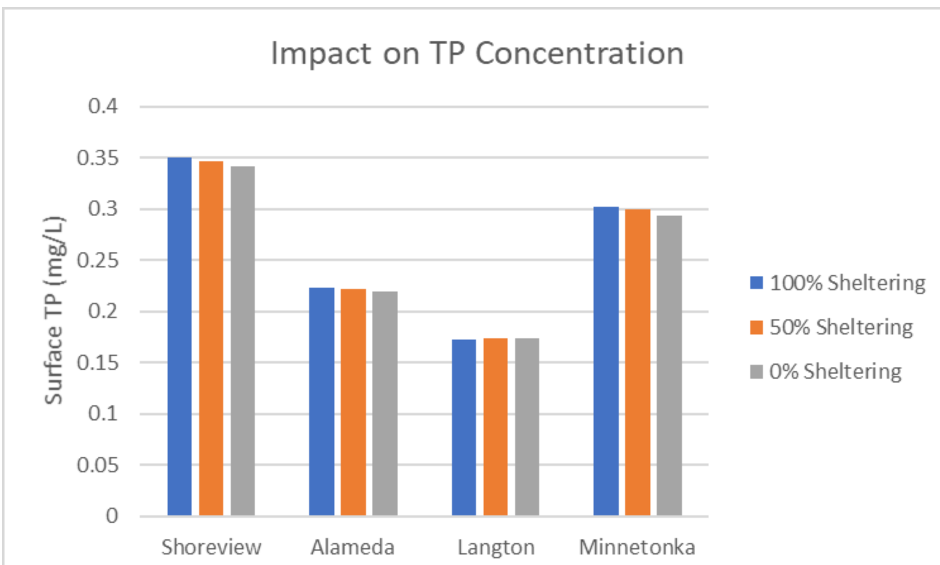

Figure D.7b Bar plot of simulation results of mean surface total phosphorus (TP) concentrations for each pond model under various wind sheltering reduction scenarios (Taguchi et al. 2022).

2.3.8 Removal of Free-Floating Plants (e.g., duckweed) – Experimental

Given the apparent strong negative association of duckweed versus oxygen and phosphorus levels of ponds in our study, treatment options to remove duckweed could prove effective at reducing phosphorus export risk of ponds. Duckweed tissue is nutrient rich (Bonomo et al. 1997; Perniel et al. 1998; Liu et al. 2017), so removal of biomass should have effects through both nutrient removal and increased oxygen, and burial of P in ponds. We are aware of no published examples of mechanical removal for water quality improvements, although a recent UMN undergraduate project found higher oxygen and lower total phosphorus following duckweed removal in a small urban pond (Ronkainen 2021). Research on nutrient removal by Lemnaceae, in other fields (biofuels, remediation, animal nutrition) may be helpful in informing such an approach. Duckweed thus represents an opportunity for management of nutrients in stormwater ponds, though options to remove duckweed or to utilize it for nutrient removal are admittedly difficult (Perniel et al. 1998; Alahmady et al. 2013).

Alum treatments have been successfully applied to an urban pond to reduce duckweed via reduction in P (Osgood 2012). Because duckweed has a high requirement for P, strategies to reduce P below the threshold for rapid growth (Giblin et al. 2014), such as through aeration or chemical treatment of sediments, may be effective for also reducing duckweed density, with presumably positive resulting effects on dissolved oxygen and P dynamics in ponds. **However, removal or management of free-floating plants for improved phosphorus removal in ponds remains a research need.**

2.3.9 Vegetation Harvesting – Experimental

Harvesting of macrophytes from ponds has been considered for permanent removal of phosphorus, as well as other pollutants such as chloride and heavy metals. This option might be especially attractive in ponds with invasive species such as hybrid cattails, whose removal might improve other ecosystem services of ponds.

Alsadi (2019) demonstrated significant removal of phosphorus through annual harvesting of macrophytes from an agricultural treatment wetland system in Minnesota, but it is unclear if this approach would work in an urban stormwater setting. Floating treatment wetlands, commonly used in wastewater treatment and being adapted to stormwater management, also have potential to provide permanent phosphorus removal through regular harvesting (White, 2021; Ge et al. 2016; Wang et al. 2014). Potential nutrient removal is commonly attributed to nutrient accumulation in plant tissues and has rarely been assessed directly, however, and the suitability of floating treatment wetlands in northern climates (where ponds freeze) has been largely untested (but see Tharp et al. 2016). Further, the manual labor required for vegetation removal may make it prohibitive to employ at large scale. **However, the high potential for phosphorus removal and for improved pond conditions through vegetation harvesting warrants further study.**

2.3.10 Adaptive Control / Real Time Control – Experimental

This management option involves modifying the outlet structure of a pond in a way that water level can be drawn down through a remotely operated valve in response to forecast storms, which provides additional water storage (and thus, retention capacity and residence time for removal of phosphorus, sediment, and other pollutants). This emerging technology is still in its infancy, but has shown potential in demonstration applications in St. Paul, MN[17], Michigan (Kerkez et al. 2016) and elsewhere (Schmitt et al. 2020). **The high potential for use in retrofits and improvement to retention suggest that these technologies would greatly benefit from further research into overcoming the hurdles in implementation and development**, which include costs of construction and data transmission, design of mathematical models or rules to govern adaptive systems, and integration with forecasting systems (Brasil et al. 2021).

2.4 Summary of Pond Maintenance Modeling Study and Assessment of Cost-Effectiveness *(Revised summary from Taguchi et al. 2022)*

Modeling Study Summary: A recent report by Taguchi, et al. (2022) from a preceding project considers the outcomes of the pond maintenance strategies of sediment treatment to reduce internal loading of phosphorus, mechanical aeration, alteration of pond outlet to pull water off the bottom, reduction of wind sheltering, dredging, outlet treatment by iron enhanced sand filtration, and reduction of phosphorus loading from the watershed. The strategies were analyzed with the model CE-QUAL-W2, where inputs to the model were initial conditions, morphology, inflow rate and total phosphorus and soluble reactive phosphorus concentrations, sediment oxygen demand, sediment release of phosphate, and meteorological conditions. The model as applied in this research simulated stratification, wind mixing, outflow and vertical profiles of temperature, dissolved oxygen, chloride, soluble reactive phosphorus, and total phosphorus. The model was calibrated on data from Alameda pond in Roseville, verified on data from the Shoreview Commons pond and applied to maintenance and remediation strategies for the Alameda, Shoreview Commons, Langton (Roseville), and Minnetonka 849W ponds. Costs of maintenance or remediation strategies were estimated, and the cost per reduction in total phosphorus release was calculated.

Cost-Effectiveness: A summary or the effectiveness of various remediation strategies for four ponds is provided in Table D.4 with associated costs provided in Table D.5. Through these maintenance activities, it is believed to be possible to return stormwater ponds to their original water-quality performance. For best results, Taguchi et al. (2022) recommend pairing watershed-based methods with the in-pond

[17] https://www.capitolregionwd.org/projects/curtiss-pond-improvement-project/

methods found to be effective (chemical treatments and mechanical aeration). Understanding how the different components of overall pond phosphorus dynamics interact is key to reducing TP export over the lifespan of ponds. Inexpensive routine maintenance practices, like street sweeping and preventing tall vegetation such as trees from establishing, could result in substantial cost savings by preventing the need for more expensive acute remediation strategies.

A few notes are included below on the cost-effectiveness and long-term implications of several treatments. For a more detailed discussion of the effectiveness of various treatments, see Taguchi et al. 2022.

- **Chemical treatment** (e.g., alum, iron filings) may be effective, but longevity in ponds is unknown. More work is needed to better understand chemical types, dosing, application strategies, and long-term performance of such treatments in stormwater ponds.

- **Dredging** was required to maintain pond depth, but dredging was not a cost-effective treatment for phosphorus reduction in the water column or the reduction of phosphorus export from ponds.

- **Watershed Management:** Volume inflow reduction and phosphorus concentration reduction are well known strategies for reducing nutrient pollution to lakes. Such actions are clearly helpful to improving pond retention but may not improve surface TP and were generally less cost-effective at reducing total phosphorus concentration than sediment treatment and aeration when sediment phosphate release rate was high. Further, implementation may be difficult to achieve, given limited space for new stormwater infrastructure in older urban areas.

- **Iron-Enhanced Sand Filter Bench:** The bench was found to be effective at reducing TP export (Figure D.7) but was not a cost-effective treatment to reduce export of phosphorus for the four ponds in the modeling study (Table D.3). A pumping system, designed to use the pond-perimeter trench more frequently, may provide an improved benefit/cost ratio. An iron enhanced sand filter is best used as a treatment train polishing step to remove phosphate before entering the receiving water body, but only if phosphate concentrations are high.

Table D.4 Summary table of simulated remediation strategies for modeled ponds. Effectiveness was evaluated against a threshold of 10% improvement from the base simulation scenario for each pond. Cost-effectiveness was evaluated against a threshold of 10x the overall most cost-effective value. One order of magnitude is the basis for all thresholds to be conservative given the variability and uncertainty in cost estimates. Further details are available in Taguchi et al. (2022).

Remediation Strategy	Addressing TP Export				Addressing Surface TP Conc.			
Pond:	Sh.	Al.	La.	Mi.	Sh.	Al.	La.	Mi.
Sediment Treatment – Alum	✓	✓	×	(✓)	✓	✓	×	✓
Sediment Treatment – Iron	✓	✓	×	(✓)	✓	✓	×	✓
Outlet Reorientation – Center	×	×	×	×	×	×	×	×
Outlet Reorientation – Bottom	×	×	×	×	×	×	×	×
Mechanical Aeration	✓	✓	×	(✓)	✓	✓	×	✓
Wind Sheltering Reduction – 50%	×	×	×	(✓)	×	×	×	×
Wind Sheltering Reduction – 100%	×	×	×	(✓)	×	×	×	×
Watershed Vol. Reduction	✓	✓	✓	(✓)	×	(✓)	×	×
Watershed Conc. Reduction	✓	✓	(✓)	×	(✓)	(✓)	(✓)	×
Bathymetry – Filling	×	×	×	□	×	(✓)	(✓)	□
Bathymetry – Dredging	(✓)	(✓)	(✓)	□	(✓)	(✓)	×	□
Bathymetry – Shallow Red.	×	×	(✓)	□	×	×	×	□
Bathymetry – Deep Red.	(✓)	×	×	□	(✓)	(✓)	×	□
IESF Bench	(✓)	(✓)	(✓)	(✓)	×	×	×	×

✓	(✓)	×
Effective Cost-Effective	Effective **NOT** Cost-Effective	**NOT** Effective

Table D.5 Summary table of cost estimates of the most successful and realistic simulated remediation strategies for modeled ponds over a 10-year span. 10-Year TP export values are rough approximations based on the assumption that the approximately 3- month simulation TP export values represent 41% of the annual pond TP export. Cost per kg (lb) TP is a calculation of the 10-year scenario cost divided by the approximated 10-Year TP export mass reduction in number of kg (lb). Cost per % TP conc. Is a calculation of the 10-year scenario cost divided by the reduction in mean surface TP concentrations. Shoreview alum application at $230 per kg TP export; Minnetonka alum application at $150 per % TP conc. All dollar values are rounded to two significant figures and presented as 2021 USD. All mass and percent values are rounded to one decimal place. N/A values indicate where cost estimations were not possible or when no improvement occurred. Further details are available in Taguchi et al. (2022).

Alum Application					
Pond	**10-Year Cost**	**10-Year kg (lb) TP Export Reduction**	**Cost per 10-Year kg (lb) TP Export Reduction**	**% TP Conc. Reduction**	**Cost per % TP Conc. ($/%) Reduction**
Shoreview	$10,000	44.8 (98.7)	$230 ($100)	42%	$290
Alameda	$16,000	17.2 (37.9)	$920 ($420)	23%	$700
Langton	N/A	0.0 (0.0)	N/A	2%	N/A
Minnetonka	$5,200	0.1 (0.3)	> $10,000	35%	$150
Iron Filings Application					
Pond	**10-Year Cost**	**10-Year kg (lb) TP Export Reduction**	**Cost per 10-Year kg (lb) TP Export Reduction**	**% TP Conc. Reduction**	**Cost per % TP Conc. ($/%) Reduction**
Shoreview	$9,900	31.9 (70.3)	$310 ($140)	30%	$330
Alameda	$9,900	12.1 (26.6)	$820 ($370)	16%	$620
Langton	$3,900	0.0 (0.0)	> $10,000	1%	$3,900
Minnetonka	$7,300	0.1 (0.2)	> $10,000	25%	$290
Mechanical Aeration					
Pond	**10-Year Cost**	**10-Year kg (lb) TP Export Reduction**	**Cost per 10-Year kg (lb) TP Export Reduction**	**% TP Conc. Reduction**	**Cost per % TP Conc. ($/%) Reduction**
Shoreview	$26,000	38.0 (83.7)	$670 ($300)	54%	$480
Alameda	$21,000	22.3 (49.2)	$940 ($430)	30%	$700
Langton	$4,100	0.0 (0.0)	> $10,000	2%	$2,100
Minnetonka	$21,000	0.2 (0.4)	> $10,000	45%	$470
50% Watershed Volume Reduction					
Pond	**10-Year Cost**	**10-Year kg (lb) TP Export Reduction**	**Cost per 10-Year kg (lb) TP Export Reduction**	**% TP Conc. Reduction**	**Cost per % TP Conc. ($/%) Reduction**
Shoreview	$23,000	61.9 (136.5)	$380 ($170)	-14%	N/A
Alameda	$23,000	40.8 (89.9)	$580 ($260)	-7%	N/A
Langton	$1,500	1.0 (2.2)	$1,500 ($700)	-3%	N/A
Minnetonka	$4,800	0.6 (1.4)	$7,500 ($3,400)	-5%	N/A
50% Watershed Construction Reduction					
Pond	**10-Year Cost**	**10-Year kg (lb) TP Export Reduction**	**Cost per 10-Year kg (lb) TP Export Reduction**	**% TP Conc. Reduction**	**Cost per % TP Conc. ($/%) Reduction**
Shoreview	$95,000	11.4 (25.1)	$8,300 ($3,800)	11%	$8,900
Alameda	$95,000	10.4 (23.0)	$9,100 ($4,100)	14%	$6,800
Langton	N/A	0.2 (0.3)	N/A	14%	N/A
Minnetonka	N/A	0.0 (0.0)	N/A	4%	N/A

For comparison, we have provided construction and maintenance estimation curves (excluding land costs) for stormwater ponds and stormwater wetlands in Figure 8.

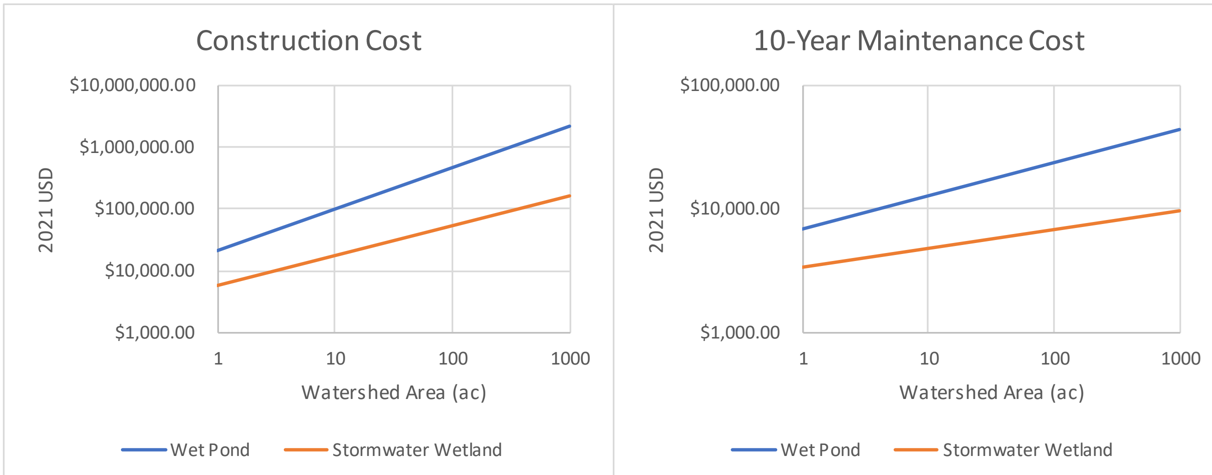

Figure D.8 Construction and maintenance cost curves excluding land costs for stormwater ponds and wetlands developed by Wossink and Hunt (2003) using data from North Carolina, as presented in Clary and Piza (2017). 10-year maintenance estimates were calculated by dividing the provided 20-year maintenance cost curves by a factor of 2. The original cost curves were presented in 2003 USD values which were converted to 2021 USD using the U.S. Bureau of Labor Statistics (2022) Consumer Price Index Inflation Calculator.

Additional estimates of TP removal costs by alternative sedimentation practices are provided by Kertesz et al. (2013) based on a computational fluid dynamics modeling effort. After converting 2013 USD values to 2021 USD values (for consistency with the above costs) using the U.S. Bureau of Labor Statistics (2022) Consumer Price Index Inflation Calculator, the results were as follows: $10/kg-TP ($5/lb-TP) using a screened hydrodynamic separator, $26/kg-TP ($12/lb-TP) using a sump, and $39/kg-TP ($17/lb-TP) using a radial cartridge filter. These devices are generally regarded as pre-treatment because they remove larger particulates (sumps and separators) or require a large footprint for the discharge treated (filter systems), but do present an effective technology to have in a treatment train.

3 Polycyclic Aromatic Hydrocarbons in the Sediments of Ponds that Treat Stormwater (Revised from Janke et al. 2022b)

3.1 Background and Performance Trends

Polycyclic aromatic hydrocarbons (PAHs) originate from natural and anthropogenic sources in the environment. In urban settings, the primary cause of PAH pollution is often due to anthropogenic sources including combustion products (incomplete combustion of petroleum, oil, coal and wood), motor vehicles (leaking motor oil, gasoline, tire and brake wear), road debris, road surface abrasion, seal coat materials for asphalt paved parking lots, and industrial processes (Polta et al. 2006). An increased degradation of sediment quality due to PAH contamination has been observed in urban watersheds due to increasing urban sprawl over the past two decades (Van Metre et al. 2000). PAHs are pollutants of great concern due to their toxic and carcinogenic effects, persistent nature and ability to bioaccumulate.

In stormwater ponds, PAHs will mostly settle out to the bottom sediments. Due to their extremely low solubility and hydrophobic nature, most PAHs, especially the heavy-weight PAHs, are associated primarily with particulate matter and are thus more predominant in urban stormwater runoff than their dissolved forms (Bathi et al. 1999, Hwang and Foster 2006). In the review of literature, several studies have documented the presence of PAHs in stormwater pond sediments in Minnesota (Polta et al. 2006, Crane 2014, Huang et al. 2019) and beyond (Kamalakkan et al. 2005, Flanagan et al. 2021), indicating ponds act as sinks for PAHs. The studies have largely focused on evaluating the sediments for PAH contamination and thus reported the sediment concentrations of PAHs, including the type of PAHs, the spatial and vertical distribution in the pond and nature of particle associations; however, a quantitative measure of performance (i.e., percent reduction in runoff) has not been reported.

Overall, the studies report an extreme variability in PAH concentrations in pond sediments. Typically, higher concentrations of PAHs are measured in finer size particles than the coarser fraction since PAHs are mostly associated with organic aggregates rather than mineral forms (El-Mufleh et al. 2013). The differences in particle associations not only results in spatial differences in the amount and type of PAHs from inlet to the outlet of the pond, but also increases the possibility of dispersion and accumulation of PAHs beyond the pond (Kamalakkan et al. 2005). While some pond sediments are dominated by heavy PAHs (4-6 molecular rings), other sediments contain light-weight PAHs (2-3 rings), suggesting the presence of different sources of PAHs to the ponds (i.e., combustion processes vs. fossil fuel leaks vs. coal tar sealcoat). The influence of major land use type on the PAH source apportionment and contamination levels are possible, i.e., lower PAHs in ponds in residential catchments than in industrial/commercial and road catchments (Kamalakkan et al. 2005, Benardin-Souibgui et al. 2018), although significant correlations with land use and pond characteristic were not always found (Crane 2014, Huang et al. 2017, Flanagan 2021). The deposition of PAHs relative to heavy metals (Cr, Mo, Mn, Co, Zn, Fe, etc.) was also not conclusive in one study (Kamalakkan et al. 2005) likely due to different physical and chemical characteristics and transport processes. The variabilities and lack of trends could be due to the complex chemical structure of PAHs that dictates their behavior, distribution, and fate in the environment.

3.2 Mechanistic Drivers

The persistence of PAHs in the environment is dependent on a variety of factors, such as the chemical structure, concentration, dispersion, and bioavailability of the PAH. Additionally, environmental factors such as pH, temperature, soil type and structure, and presence of adequate levels of oxygen, nutrients and water for the activity of the PAH-degrading microbial community (bacteria and fungi) control the fate and persistence of PAHs (Bamforth and Singleton 2005). PAHs are readily degraded by photooxidation and reaction with atmospheric oxidants. However, PAHs can be associated with other pollutants such as hydrocarbons and heavy metals (Kamalakkan et al. 2005) that are more readily degraded aerobically than PAHs, which increases the overall residence time of PAH in the environment (Bamforth and Singleton 2005). Under anaerobic conditions, PAH degradation is severely limited which means, PAHs will be very stable and have prolonged existence in an anoxic environment such as pond sediments.

3.3 Assessment Methods

The most common method for PAH assessment is by sediment sampling, as PAHs are mostly bound to sediments. Sediments collected by dredging or coring are subject to analysis for a suite of PAHs and evaluated in terms of benzo[a]pyrene equivalent concentration (which is determined using the relative potency factors provided by the MPCA; Polta et al. 2006) that is compared to industrial soil reference value (SRV) to evaluate the extent of contamination. The dissolution (bioavailability) of the PAHs are typically assessed separately via bench-scale or laboratory studies.

3.4 Maintenance Recommendations and Treatments

Accumulation of PAHs in pond sediment has important implications for sediment management (i.e., removal and disposal). Design measures, such as sediment forebays, to reduce sediment load input to ponds could limit the PAH accumulation in the pond (Schifman et al. 2018, Flanagan et al. 2021). One way to improve sediment removal rates in ponds, as well as pond hydrologic functioning, is to intercept sediments before they enter ponds using pretreatment devices. A variety of pond pretreatment devices exist, including vegetative filter strips, hydrodynamic separators, and sediment forebays (Marsalek et al. 1992, Taguchi et al. 2020b). These devices can be relatively inexpensive to install and, if maintained, can significantly delay the need for costly pond dredging. Regular sediment removal from pretreatment devices also represents a substantial pollutant removal mechanism that can reduce contaminant accumulation rates and internal loading. However, such devices require routine maintenance, and there may be costs associated with sediment disposal, particularly if deemed hazardous waste due to the presence of toxic metals and PAHs.

Bioremediation of PAH-contaminated sediments by indigenous microbial population can be achieved by adjusting the water, air, and nutrient supply (biostimulation) and by inoculating with microorganisms with known pollutant-transforming abilities (bioaugmentation). Examples of these processes are land-farming and biopiling (several studies cited in Bamforth and Singleton 2005). Kyser et al. (2010) conducted bench-scale experiments using compost to biodegrade PAHs in pond sediments but the results were less successful. Huang et al. (2019) conducted a preliminary treatability study in the laboratory that showed reduction in PAH bioavailability by the addition of powdered activated carbon to sediments. There are no known regulations, however, that consider bioavailability in the determination of toxic concentrations. Studies outside the pond literature have studied the ability of a consortium of bacteria and fungi for the bioremediation of PAHs (Bamforth and Singleton 2005, Sheng et al. 2009). While aerobic degradation of PAHs is well documented, there seems to be minimal possibility of biodegradation under denitrifying, sulfate-reducing and methanogenic conditions that are often found in stormwater ponds. One potential disadvantage of in situ anaerobic bioremediation of PAH, however, would be the concomitant release of redox-sensitive nutrient species such as phosphate from the sediments. Phytoremediation may be difficult and inadequate due to sensitivity of plant species to PAHs (Díaz-Ramirez et al. 2003), thus combining the use of plants and plant growth promoting bacteria for clean-up of PAH-contaminated soil has been suggested (Glick 2003, Sheng et al. 2009).

3.5 Knowledge Gaps and Research Needs

One of the consistent findings in the literature is the effect of material composition (organic vs. inorganic) of runoff particulates and sediments on the PAH associations. While some studies have focused on the nature of PAHs in different fractions of the sediment, only a few studies have investigated the nature of the particles themselves (El-Mufleh et al. 2013). Methods to identify the specific signatures of the sources of anthropogenic particles in the pond sediments could be used as a surrogate for chemical analysis for PAH (Flanagan et al. 2021).

Remediation measures for PAHs in stormwater ponds was not found in the current literature search, and only selected research reports (Kyser et al. 2010, Huang et al. 2019) were included in this review. There is a substantial research need for PAH remediation measures for pond sediments. Currently, the main means of disposing of dredged sediments contaminated with PAHs is disposal in a confined disposal facility. This increases the overall cost of dredging by roughly a factor of three.

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Appendix E
Review and Update of Stormwater Ponds Sections in 2009 Stormwater Maintenance BMP Guide and 2011 Decision Tree for Stormwater BMPs

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1 Introduction

This document presents suggested updates to two previous documents that outline guidance on stormwater BMP selection (Marti et al. 2011) and on maintenance and inspection procedures for stormwater BMPs (Marti et al. 2009). These recommendations concern only the sections of those two documents that are related to stormwater ponds, and are based on the pond maintenance guide and the Pond Assessment Tool described in Janke et al. (2023). Datasets and analyses supporting development of the tool and the maintenance recommendations are also published in Janke et al. (2023).

2 Suggested Updates to the 2009 LRRB Report Stormwater Maintenance BMP Resource Guide

2.1 Definition

A diversity of small water bodies are used in wet stormwater management practices to treat runoff, ranging from constructed stormwater ponds to natural wetlands (MPCA 2022). The term “Constructed Stormwater pond” typically refers to any upland (without hydric soils) constructed basin that is built for the purpose of capturing and storing stormwater runoff, either temporarily or for an extended period of time, in order to prevent or mitigate downstream water quantity or quality impacts (Janke et al. 2023). Prior to the Wetland Conservation Act of 1991, wetlands were often plumbed to be utilized as stormwater treatment BMPs. Existing wetlands are no longer used as new treatment BMPs except as a final device in a stormwater treatment train. Because wetlands are regulated waterbodies, maintenance activities typically require permits and removal of sediment can only be done to a specific elevation (see Chapter 2 in Janke et al. 2023).

2.2 Description

Constructed stormwater ponds and wetlands that treat stormwater (stormwater ponds and wetlands) are typically installed as an end-of-pipe BMP at the downstream end of a subwatershed. The primary function of a stormwater pond and wetland is to remove a significant portion of sediment and associated pollutants from stormwater runoff prior to it being released downstream. They also can be used to attenuate peak discharges from a site, and in some cases, reduce the flows to predevelopment rates.

Virtually all stormwater ponds and wetlands have an outlet control structure. These can take various forms and can be constructed of a variety of materials depending on the specific requirements for the pond/wetland and depending on the surrounding topography. A typical outlet control structure will provide skimming so that floatables are retained in the pond. It will typically also control the rate of discharge from the pond and provide an emergency overflow for large storm events.

There are several distinct sub-categories of stormwater ponds and wetlands that are discussed in more detail below.

2.2.1 Sub-categories of Stormwater Ponds

National Urban Runoff Program (NURP) or wet extended detention pond:

A combination of permanent pool storage and extended detention storage above the permanent pool to provide additional water quality and rate control (MPCA 2022).

Multi-cell pond with subsets (Micro-pool extended detention pond, Wetland/Pond combo):

Some stormwater ponds are designed with a micro-pool where the water first enters the pond. The micro-pool prevents resuspension of previously-settled sediments and clogging. Stormwater ponds can also be designed as multiple pond systems that create longer pollutant removal pathways (MPCA 2022).

Lined detention ponds (MPCA 2022):

Lined ponds are generally used in circumstances where a permanent pool is needed but difficult to maintain due to site conditions, or where seepage from the pond into the groundwater would otherwise occur but must be avoided. This includes:

- Areas with Hydrologic Group A soils, gravel, or fractured bedrock
- Potential Stormwater Hotspots (PSHs)
- Karst terrain
- Wellhead Protection Areas or other sensitive groundwater recharge areas

Lined ponds also may be used when an open water pond aesthetic is desired.

Liners can be constructed of a layer of compacted clay or of a variety of proprietary synthetic materials. In most cases, it is desirable to protect the liner from drying out or injury from sharp objects. Sometimes a covering material, typically sand, is placed over the liner.

Wet Ponds:

The above ponds are generally grouped together as “Wet ponds,” which will be the categorization followed herein.

Dry detention pond:

A dry detention pond has no permanent pool; it is designed to temporarily detain stormwater runoff and allow large sediment particles and associated pollutants to settle out. Water is gradually released through an outlet into the storm drain system. Dry ponds are highly susceptible to sediment resuspension and generally are designed for rate control (MPCA 2022).

2.3 Benefits/Limitations of Stormwater Ponds

2.3.1 Benefits (MPCA 2022)

- Able to effectively reduce many pollutant loads and control runoff flow rates
- Well developed design procedure
- Potential wildlife habitat and aesthetic enhancement
- May be used as temporary sedimentation basin during construction

2.3.2 Limitations (MPCA 2022)

- Relatively large space requirement
- Tends to increase water temperature and may cause downstream thermal impact
- Potential for odor
- Problematic for areas of low relief, high water table, near-surface bedrock, wellhead protection areas or source water protection areas without a liner
- Sediments can release phosphate (internal loading) which will increase algal and floating plant growth and may increase the export of phosphorus to receiving water bodies (Taguchi et al. 2020).

2.4 Assessment Activities – Stormwater Ponds

Whenever work is performed on a stormwater pond, it is essential for maintenance crews to know:

- If a permit from another regulatory agency is required to perform work on the pond.
- The extent of excavation that is allowed (by city, county or other regulatory agencies).
- If the Minnesota Pollution Control Agency will require a dredged materials permit or notification.
- If dredged, materials may require testing to determine acceptable disposal methods. Metals and polycyclic aromatic hydrocarbons (PAHs) tend to collect in the sediments.

Ponds are designed to retain solids by settling, but other pollutants (e.g., phosphorus, hydrocarbons, and metals) attached to retained solids (by adsorption, etc.) are also retained by sedimentation practices. Therefore, a topic of concern for sedimentation practices is the analysis of pollutants that may be sorbed to (and subsequently desorbed from) retained solids. Measuring and comparing the solid-bound pollutant concentrations at the influent and in the retained solids, or the dissolved pollutant concentrations at the influent and effluent, may give insight into processes occurring within the device, such as sorption or desorption of pollutants. There are three types of assessment (Erickson et al. 2013) that are relevant to ponds, depending upon the goal of the assessment: Visual inspection, capacity testing and monitoring, as described below.

2.4.1 Visual inspection

Visual inspection of sedimentation practices should include inspection and documentation of the amount and distribution of retained solids. For example, a large deposit of solids at the inflow location of a dry pond may alter the inflow conditions or increase re-suspension of solids. See the standard procedures at the end of this chapter for detailed instructions about visual inspection of sedimentation practices. In addition, inspect all inlet and outlet structures for clogging and/or structural damage. Schedule removal of debris and to repair/replace outlet structure, if necessary. The outflow location(s) should also be inspected to make sure a tailwater condition is not impeding discharge from the pond.

Visual inspection can also be used to assess the presence of high phosphorus concentrations in wet ponds. Janke et al. (2023) found that a good indicator of high phosphorus concentration in Minnesota is floating plant cover with plants such as duckweed (lemna) and watermeal (wolffia). The relationship found for wet ponds is given in Figure E.1.

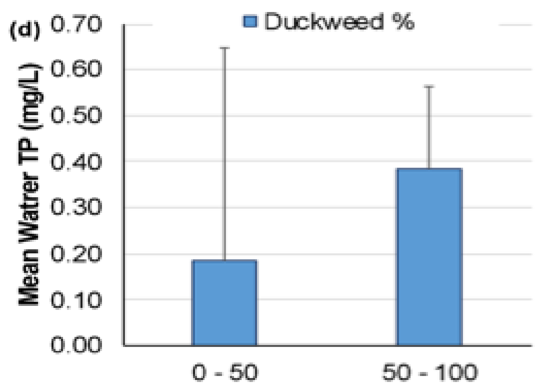

Figure E.1 Floating plant cover in mid-summer, defined as the fraction of pond surface area covered by small, free-floating plants (Lemna and Wolffia), tended to be strongly associated with higher TP concentrations in ponds (Janke et al. 2023).

2.4.2 Capacity testing

Capacity testing can be applied to sedimentation practices to estimate sediment storage capacity. All sedimentation practices can be assessed with sediment retention tests if adequate access is available. The infiltration capacity of dry ponds can also be assessed with hydraulic conductivity measurements.

Dry ponds

Hydraulic conductivity measurements of dry ponds is used to estimate the rate at which stored water infiltrates into the soil, which can be used to estimate the runoff volume reduction by infiltration (Erickson et al. 2013) A single point measurement with a falling head infiltrometer (Tecca et al. 2022) can take between 30 seconds and several hours, depending on the soil characteristics of the dry pond. Hydraulic conductivity measurements for a single dry pond can require one day to one week to complete. These hydraulic conductivity tests should be performed shortly after construction to establish a baseline for future tests and to investigate or identify construction impacts on infiltration capacity.

Sediment retention tests are used to estimate the depth and, subsequently, volume of sediment retained in a dry pond. Surface elevations are measured either with a level and level rod or a total station (i.e., surveying equipment), and the corresponding longitude and latitude are recorded either with GPS or with a total station. Using the basin topography and the original topography (from as-built plans or design drawings), the amount of sediment retained in the dry pond can be estimated. The amount of retained sediment can be compared to the design capacity to determine the available sediment retention capacity and to estimate when the pond will require maintenance (i.e., sediment cleanout). One to three days are typically required for each dry pond to perform sediment retention assessment.

Wet ponds

Sediment retention tests can be performed on a wet pond to estimate the depth and subsequently volume of sediment retained. Bottom elevations in a wet pond are measured with a sonar depth measurement device. The water surface can be used as a local elevation standard if a staff gauge has been installed in the pond to measure water surface elevation. Sonar depth measurements can be made in the winter when the wet pond is covered with sufficient ice to traverse or in the summer from a boat or while using waders. Corresponding longitude and latitude are recorded either with GPS or with a total station. Using the basin topography and the original topography (from as-built plans or design drawings), the amount of sediment retained in the pond can be estimated. The volume of retained sediment can be compared to the design capacity to determine the available sediment retention capacity and to estimate when the pond will require maintenance (i.e., sediment cleanout). As with dry ponds, it is recommended that these tests be performed soon after construction is complete to develop as-built plans as a benchmark for future assessment. If repeat measurements are made over time, the sediment accumulation rate can be estimated.

Level 2 assessment (sediment retention testing) of wet ponds is used to determine the amount of solids captured by sedimentation. If repeat measurements are made over time, the sediment accumulation rate can be estimated. Level 3 assessment (synthetic runoff tests), however, can estimate hydraulic behavior (via conservative tracer), the pollutant removal efficiency, or both.

2.4.3 Monitoring

Guidelines for monitoring can be found in Erickson et al. (2013), chapters 4, 5, and 6.

Dry ponds

By monitoring dry ponds, one can assess the peak flow reduction and pollutant removal efficiency. Measuring and comparing inflow and outflow hydrographs for a dry pond can give an estimate of the reduction in peak flow for a given storm event and, therefore, an estimate of the hydraulic effectiveness of the stormwater BMP. Results from sampling and analyzing stormwater samples from the inflow and outflow can be used to estimate the pollutant removal effectiveness.

Wet ponds

Monitoring of wet ponds is well documented (Wu et al. 1996, Comings et al. 2000, Koob 2002, Mallin et al. 2002). Short-circuiting within a wet pond can be estimated by monitoring the movement of a naturally occurring conservative tracer, such as chloride, as it moves through a wet pond if a sufficient pulse in concentration has occurred at the inlet. Comparing the inflow and outflow tracer concentration versus time curves can determine if, and to what extent, short-circuiting may be occurring.

Another form of monitoring is in-pond monitoring for water quality. The water column can be sampled periodically for nutrient analysis (phosphorus, and nitrogen species), temperature, dissolved oxygen concentration, and conductivity, as an indicator of chloride concentration. Chlorophyl a and floating plant samples can also be taken. Many ponds are stratified at as little as 1 ft depth, so taking measurements and samples near the top and bottom of the water column is prudent.

2.5 Maintenance Activities

Maintenance of ponds often involves sediment and trash removal, fixing clogged pipes, or addressing invasive vegetation. The performance of a pond is often dependent upon the continuous maintenance activities undertaken. The pond will eventually lose the performance that it had as a new pond unless it is maintained on a regular basis.

2.5.1 Actions

Dry Ponds

Dry ponds can be effective at retaining suspended solids and pollutants that typically adsorb to solids. After continued operation, retained solids will eventually need to be removed. The dry pond must be regularly inspected to determine its condition. The required frequency of inspection and maintenance is dependent on the watershed land use (e.g. urban, rural, farm, etc.), construction activities present and rainfall amounts and intensity. However, it is recommended that visual inspection and any associated maintenance be performed at least once per year.

If any level of assessment reveals that a dry pond is not draining a runoff event less than or equal to the design storm volume within the specified design time, the following measures should be taken.

1. Inspect all outlet structures for clogging and/or structural damage. Remove debris and repair/replace outlet structure, if necessary.
2. Inspect the outflow location(s) to make sure a tailwater condition is not impeding discharge from the pond. If this is the case, the tailwater should be eliminated or the outlet modified in such a way that drainage occurs within the desired time.
3. If the pond still does not drain within the specified design time, the hydraulics of the pond should be reevaluated and the geometry and outlet structure redesigned.

Hunt and Lord (2006) discuss the maintenance requirements of wetlands and wet ponds. Even though Hunt and Lord do not specifically discuss dry ponds, their recommendations that do apply to both wet ponds and dry ponds are reproduced in Table E.1.

Table E.1 Maintenance Requirements and Frequencies for Dry Ponds (selected from Hunt and Lord 2006).

Task	Frequency	Notes
Remove retained sediment	Variable (Once every 5 to 10 years is typical in stable watersheds)	In unstable watersheds (i.e. those with active construction), the frequency can be once per year
Monitor sediment depth	Once per year	Can be performed with capacity testing
Maintain outlet structures	Once per month and after every storm over 2 inches	Follow visual inspection guidelines
Remove floating trash and debris	Once per month	Increase frequency, if needed
Remove vegetation from dam top and faces, if applicable	Once per year	Increase frequency, if needed

Dry ponds are most effective in retaining suspended solids and pollutants that tend to adsorb to solids and are usually not implemented to reduce temperature impacts or achieve retention of dissolved pollutants. Thus, the following discussion only considers solids and pollutants which typically sorb to solids. If a dry pond is not retaining suspended solids (or other sorbed pollutants) at expected levels, the following steps should be taken.

1. Check to make sure that the desired levels of pollutant retention are realistic. For example, if the target pollutant is total phosphorous and the runoff entering the pond contains a large fraction of dissolved phosphorous, large retention rates of total phosphorous may not be possible for a device that retains pollutant mostly by sedimentation. Or, if the sediment size distribution contains an uncharacteristically large fraction of fines, the hydraulic retention time may not be adequate to achieve the desired retention rate. If retention of the desired pollutant is not realistic, consider implementing another stormwater BMP to achieve desired results.

2. Perform a sediment capacity test to determine the remaining sediment storage capacity of the pond. If there is no remaining capacity or if the capacity is nearly exhausted, the deposited sediment should be removed to allow for additional storage.

3. If there is adequate storage capacity remaining in the pond and pollutant removal is still below expected values, a tracer study should be performed to determine if short-circuiting is occurring. If short-circuiting is occurring, consider adding one or more baffles or retrofitting the pond to redirect the flow of runoff in a way that eliminates or minimizes short-circuiting.

Wet Ponds

Wet ponds can be effective at retaining suspended solids and pollutants that typically adsorb to solids but are not as effective at retaining dissolved pollutants. Because after continued operation retained solids will need to be removed, the wet pond must be inspected regularly to determine its condition. The required frequency of inspection and maintenance is dependent on the watershed land use (e.g. urban, rural, farm, etc.), construction activities in the watershed and rainfall amounts and intensity. However, it is recommended that visual inspection and any associated maintenance be performed at least once per year.

Hunt and Lord (2006) discuss the maintenance requirements of wetlands and wet ponds. The recommendations that apply to wet ponds are reproduced in Table E.2.

Table E.2 Maintenance Requirements and Frequencies for Wet Ponds *(revised from Hunt and Lord 2006)*.

Task	Frequency	Notes
Remove all sediment from forebay and deep pool (dredging)	Variable (Once every 5 to 10 years is typical in stable watersheds)	In unstable watersheds (i.e. those with active construction), the frequency is typically once a year.
Monitor sediment depth in forebay and deep pools	Once per year	Can be performed with capacity testing
Maintain outlet structures, if required	Once per month and after every storm over 2 inches	Follow visual inspection guidelines
Remove floating trash and debris	Once per month	Increase frequency, if needed
Remove vegetation from dam top and faces, if applicable	Once per year	Increase frequency, if needed
Mow wet pond perimeter	From every week to once per year	--
Remove muskrats and beavers, if present	Inspect at least monthly	Destroy muskrat holes whenever present. Contact a professional trapper to remove beavers

Capacity testing or monitoring are required to determine if a wet pond is retaining pollutants as expected. Wet ponds are most effective in retaining suspended solids and pollutants that tend to adsorb to solids and are usually not implemented to reduce temperature impacts or achieve retention of dissolved pollutants. One exception is where algae or floating plants take up dissolved nutrients and turn them into particulate nutrients, resulting in the dissolved nutrients in a wet pond being lower than in the inflow. If a wet pond is not retaining suspended solids or other sorbed pollutants at expected levels, the following steps should be taken.

1. Check to make sure that the desired levels of pollutant retention are realistic. For example, if the target pollutant is total phosphorous and the runoff entering the pond contains a large fraction of dissolved phosphorous, large retention rates of total phosphorous may not be possible for a device that retains pollutant mostly by sedimentation. Or, if the sediment size distribution contains an uncharacteristically large fraction of fines, the hydraulic retention

time may not be adequate to achieve the desired retention rate. If retention of the desired pollutant is not realistic, consider implementing another stormwater BMP to achieve desired results.

2. Perform a sediment capacity test to determine the remaining sediment storage capacity of the pond. If there is no remaining capacity or if the capacity is nearly exhausted, the entrained sediment must be removed to allow for additional storage.

3. If there is adequate storage capacity remaining in the pond and pollutant removal is still below expected values, a tracer study should be performed to determine if short-circuiting is occurring. If short-circuiting is occurring, consider adding one or more baffles or retrofitting the pond to redirect the flow of water in a way that eliminates or minimizes short-circuiting.

One aspect of many wet ponds that have been noticed is that they can stratify from chlorides and temperature at 1 ft depth or less (Taguchi et al. 2020). This stratification cuts off the bottom layers of the pond from oxygen in the atmosphere, and the high organic content of the sediments causes the dissolved oxygen concentration to approach zero. These low oxygen concentrations create anoxic conditions, causing the release of phosphate from the sediments (internal loading of phosphorus) which will reduce the phosphorus retention of wet ponds. Taguchi et al. (2022) have studied maintenance techniques to reduce internal phosphorus loading, and found that sealing the sediments with alum or iron filings and mixing through aeration to be the most cost-effective techniques.

2.5.2 Factors Affecting Pond Performance

Maintenance efforts for sedimentation practices are typically focused on sediment and trash removal, fixing clogged pipes, and addressing invasive vegetation. Table E.3 lists the percent of responding municipalities that indicated the listed factors most frequently caused deterioration of BMP performance. The most frequent factors causing deterioration of performance were sediment buildup, litter and debris, pipe clogging and invasive vegetation. Bank erosion, groundwater level and structural problems, however, can cause serious and rapid deterioration of performance when present.

Table E.3 Percent of Respondents who indicated the listed factor frequently caused deterioration of Sedimentation Practice performance. Taken from Kang et al. (2008).

Factor	Dry Ponds	Wet Ponds
Sediment Buildup	24%	26%
Litter/Debris	31%	19%
Pipe Clogging	18%	21%
Invasive Vegetation	16%	10%
Bank Erosion	8%	11%
Groundwater Level	2%	7%
Structural Problems	0	7%

2.6 Recommendations

2.6.1 Assessment

Visual inspection is recommended for assessment of all ponds at regular intervals, at least once per year. If there is floating plant coverage, consideration should be given to reducing internal phosphorus loading. Capacity testing is recommended for assessment of infiltration in dry ponds or for the assessment of sediment accumulation in wet ponds. Monitoring is recommended when capacity testing do not meet the goals of the assessment program.

2.6.2 Maintenance

Wet and dry ponds are designed to be effective at retaining suspended solids and pollutants that typically adsorb to solids. After continued operation, the retained solids need to be removed. The ponds must therefore be regularly inspected to determine its condition. The required frequency of inspection and maintenance is dependent on the watershed land use (e.g. urban, rural, farm, etc.), construction activities in the watershed and rainfall amounts and intensity. However, it is recommended that visual inspection and any associated maintenance be performed at least once per year.

3 Suggested Updates to the 2011 LRRB Report Decision Tree for Stormwater BMPs

The 2011 LRRB report (Marti et al. 2011) outlines a decision tree to be used for BMP selection in new installations, as a function of space constraints, permit or regulatory requirements for total suspended solids (TSS) and total phosphorus (TP) removal, land use context, and special site considerations such as contaminated soils, wellhead protection areas, depth to groundwater, etc. Given that the Pond Assessment Tool and the resulting pond maintenance guidance document developed in Janke et al. (2023) are intended for existing ponds rather than new ponds, there are few substantial changes that we could recommend to the decision tree. Furthermore, we note that new or young ponds were in the lowest risk category for high surface water TP concentrations in our data set (see Figure E.2 below). Therefore, we do not have any changes or updates to recommend for the Decision Tree document. However, we include a few considerations below with respect to life expectancy and site suitability for pond BMPs, based on the categorical assessment of ponds included in the Assessment Tool. These considerations, which primarily have to do with BMP installations at sites in areas of residential land use or with heavy tree cover, may warrant further investigation in future studies.

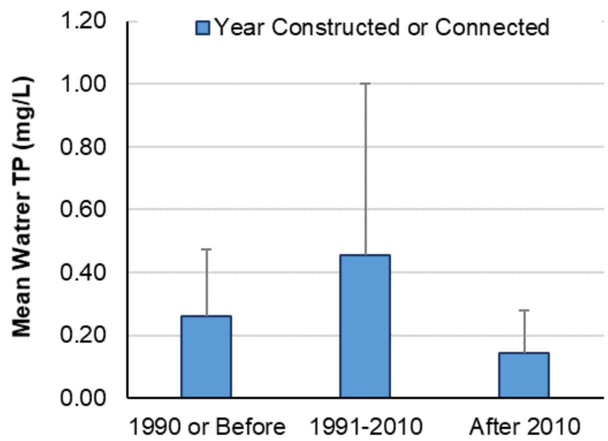

Figure E.2 Mean and standard deviation of surface water TP concentration (n = 230) in TCMA urban ponds, based on the year of construction or connection to storm drains. "1990 or Before" group includes ponds with undetermined age (i.e., construction data unavailable) that were visible in 1991 Google Earth imagery. Construction year can be converted to age based on year 2021 (for example, a pond built after 2010 is ≤ 10 years old). From Janke et al. (2023).

3.1 Project Setting (Question 2.4 in Marti et al. 2011)

The project setting category in the decision tree allows for consideration of the surrounding land use in siting of a BMP, and includes three categories: (1) central business district, (2) residential/ suburban/ campus/ commercial, and (3) rural/undeveloped. Based on our dataset presented in previous task reports (and in Figure E.3 below), we note that ponds in residential areas tended to have the highest surface water TP concentrations. The mechanisms underlying the pattern are not completely understood (and may be due in part to correlation with high tree cover), and we note also that high TP concentrations in ponds do not necessarily indicate poor TP mass retention (Janke et al. 2022). Yet, this pattern suggests that alternative BMPs with high TP retention might be considered in residential areas (such as multiple infiltration- or bioretention-type BMPs) despite potentially higher maintenance needs due to potential clogging from organic inputs. If rate control is a primary concern stormwater ponds would still be recommended.

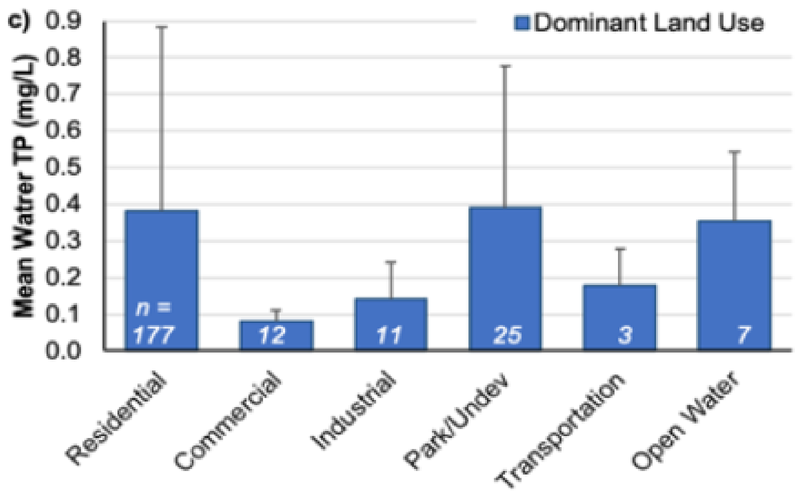

Figure E.3 Mean and standard deviation of surface water TP concentration (mg/L) based on the dominant land use within a 500 m buffer of the ponds (i.e., land use with the greatest area fraction in the buffer; n = 235 ponds; Janke et al. 2023).

3.2 Special Considerations (Question 2.5 in Marti et al. 2011)

The special considerations category in the decision tree factors in site characteristics that might influence suitability of different BMP types, such as shallow depth to groundwater, contaminated soils, locations in wellhead protection areas or karst geology. Based on our analysis for the Pond Assessment Tool, special consideration might be added for areas with high tree canopy, as we observed substantially higher TP in ponds with high tree cover (> 60%) within a 500-m proximity of the ponds (Figure E.4). In case of high tree canopy cover near the site, alternative BMPs such as infiltration or bioretention might be recommended over ponds, unless rate control is a necessity. As above, we note that infiltration and filtration BMPs might be susceptible to clogging from leaf litter and organic inputs and potentially require more maintenance; further, high TP concentrations may not necessarily indicate poor TP retention. These topics require further study before recommendations could be made with certainty.

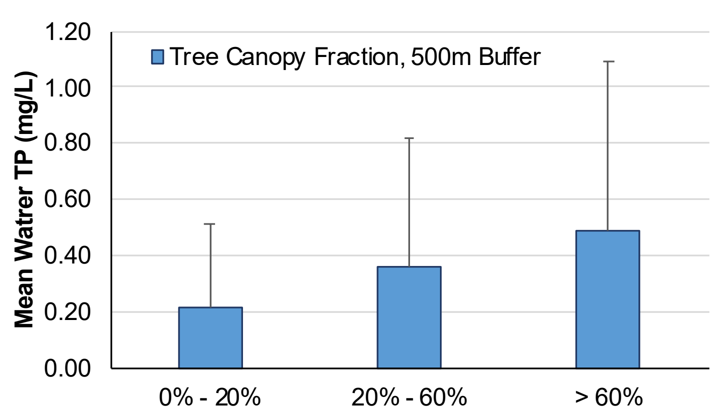

Figure E.4 Mean surface water TP (mg/L) observed in ponds (n = 230; Janke et al. 2023), grouped by percentage canopy cover in a 500 m buffer.

3.3 Cost, Life Cycle, and Maintenance Burden (Table 5 in Marti et al. 2011)

Table 5 in Marti et al. (2011) outlined costs, cost-effectiveness, life expectancy, and maintenance requirements for the various BMP types included in the document, without consideration of land use or land cover factors as was done in the decision tree. One comment we make here is that in the case of residential and/or high tree canopy watersheds, the life expectancy of the phosphorus control function of stormwater pond BMPs may be reduced relative to the other land use categories (highly developed and rural), as well as in the case of high tree cover (which is often associated with residential land use). The life cycle costs associated with a pond in residential and/or high tree cover contexts would also more likely be higher, with more frequent and/or costly maintenance required.

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