Project Report #602

Wet Pond Maintenance for Phosphorus Retention: LRRB 2019 KB 03 MnDOT Agreement No. 1034035

Final Report

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Executive Summary

This report 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-2E, 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, verified on data from the Shoreview Commons pond and applied to maintenance and remediation strategies for the Alameda, Shoreview Commons, Langton, and Minnetonka 849W ponds. Costs of maintenance or remediation strategies were estimated, and the cost per reduction in total phosphorus release was calculated.

We make the following observations from these pond simulations (Table S-1) and the subsequent calculations:

1. Under anoxic conditions (low dissolved oxygen concentration) with relatively high sediment phosphate release rates, sediment treatment with alum or iron filings to reduce internal phosphorus loading is a good option to reduce water column total phosphorus concentration and associated algal and floating plant growth. The export of phosphorus to receiving water bodies, however, should also consider the magnitude of outflow. Minnetonka 849, for example, has relatively low outflow from the pond and thus low phosphorus export (Figure S-1a).
2. Sediment treatment to reduce phosphate release is generally not the most effective way to treat newer ponds, such as the Langton pond, simply because their sediment phosphate release rate is generally low (Figures S-1a and S-1b).

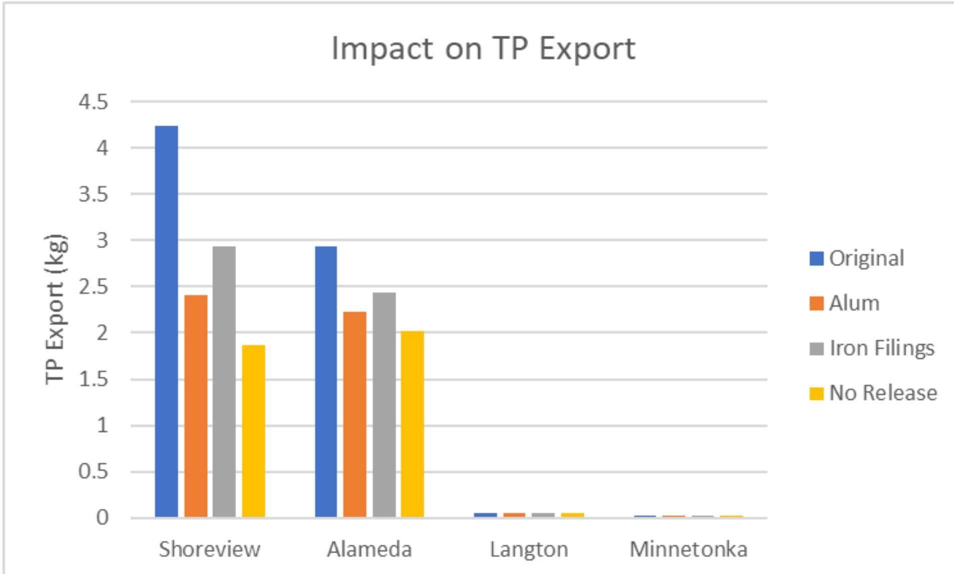

Figure S-1a. *Bar plot of simulation results of cumulative total phosphorus (TP) export mass for each pond model under various sediment chemical treatment scenarios. Further details are available in Section 5.3.*

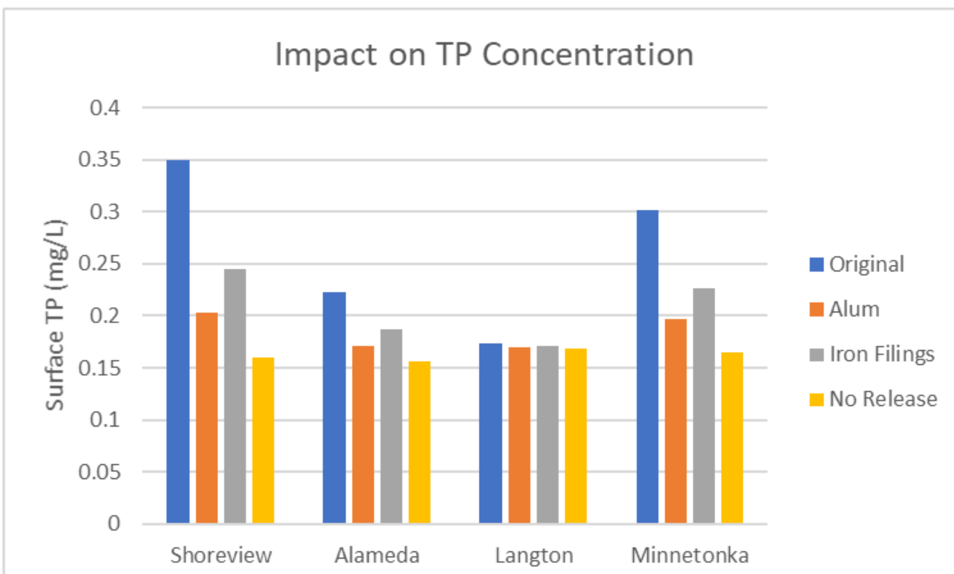

Figure S-1b. *Bar plot of simulation results of mean surface total phosphorus (TP) concentrations for each pond model under various sediment chemical treatment scenarios. Further details are available in Section 5.3.*

3. Mechanical aeration designed to destratify the pond can be an effective treatment for water column concentration of total phosphorus (TP) at stormwater retention ponds. Again, outflow must also be considered to estimate a reduction in the export of phosphorus.
4. The Shoreview Commons pond had sediment releases of phosphate under oxic conditions. In this case, 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.

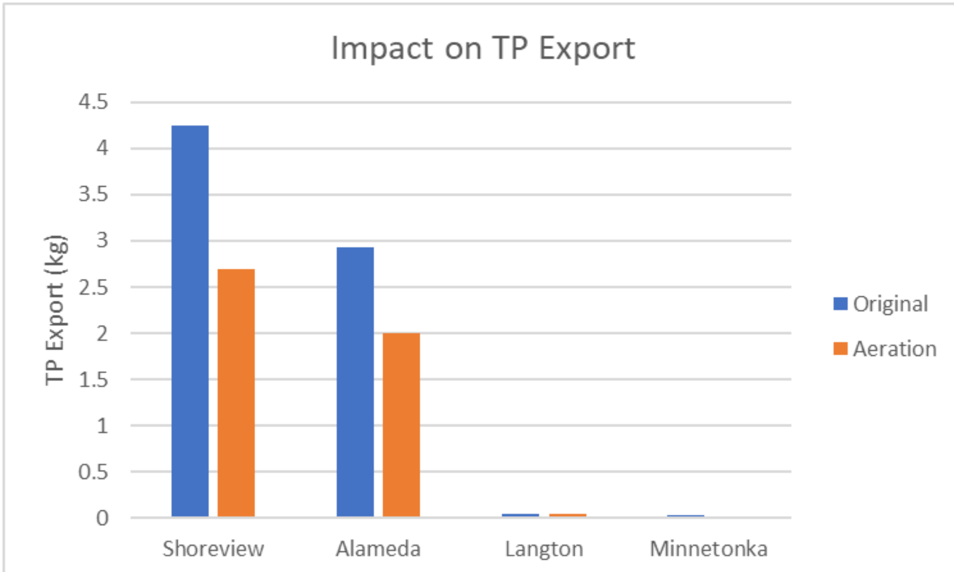

Figure S-2a. *Bar plot of simulation results of cumulative total phosphorus (TP) export mass for each pond model under mechanical aeration scenarios. Further details are available in Section 5.5.*

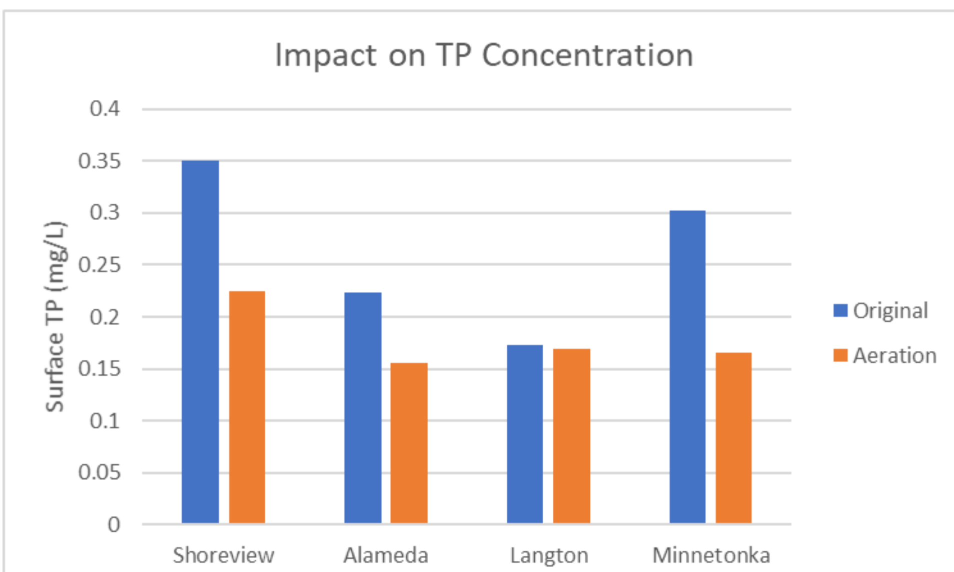

Figure S-2b. *Bar plot of simulation results of mean surface total phosphorus (TP) concentrations for each pond model under mechanical aeration scenarios. Further details are available in Section 5.5.*

5. Watershed-based methods (reducing inflow concentrations and volumes) were effective for all pond exports (Figure S-3a), which was expected since the stormwater TP inflows were a major component of the overall TP mass balance in each pond. Reducing inflow volumes

(through the installation of infiltration practices) led to increased TP concentrations in the model's ponds since constituents in the pond water were not as diluted by inflows, which had lower TP concentration relative to pond water (Figure S-3b). 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 reduced both in-pond concentrations and overall pond TP export in a more predictable way.

6. Volume inflow reduction and phosphorus concentration reduction, however, are generally less cost-effective at reducing total phosphorus concentration than sediment treatment and aeration when sediment phosphate release rate is high. They are more effective on newer ponds with a low phosphate release rate (See Section 5.7).

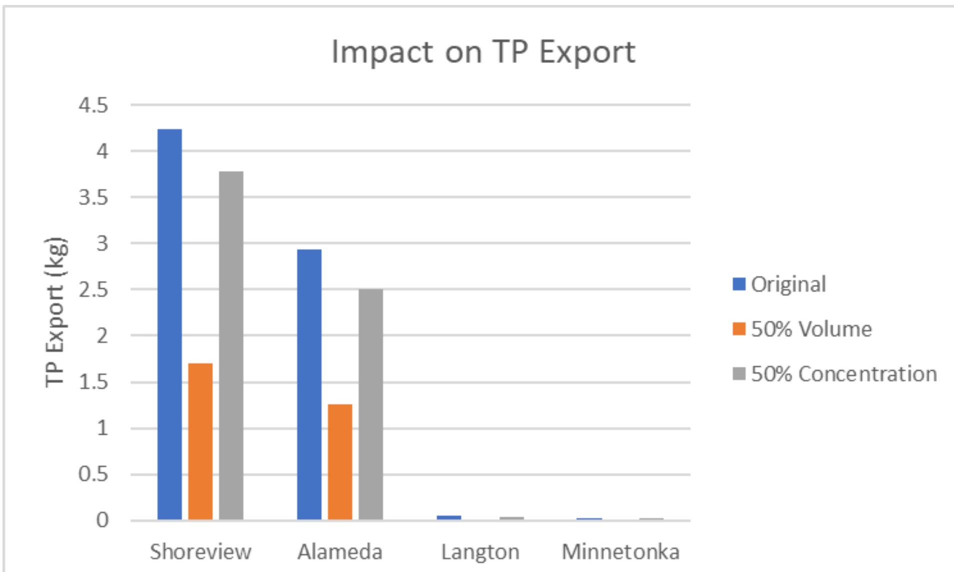

Figure S-3a. *Bar plot of simulation results of cumulative total phosphorus (TP) export mass for each pond model under various watershed-based treatment scenarios. Further details are available in Section 5.7.*

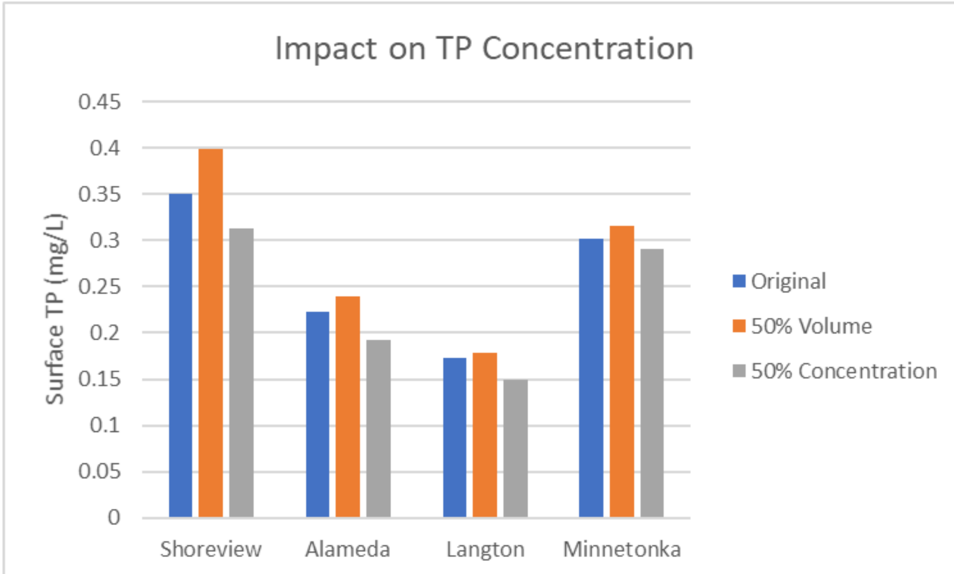

Figure S-3b. *Bar plot of simulation results of mean surface total phosphorus (TP) concentrations for each pond model under various watershed-based treatment scenarios. Further details are available in Section 5.7.*

7. We 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 (Figures S-4a and S-4b). This could result from one or all of three observations: ponds have a shot wind fetch, relative to lakes; the banks on the pond cause sufficient separation of the wind and sheltering of the ponds to reduce the potential wind shear; and the upwind roughness, such as houses, trees around the houses and 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.

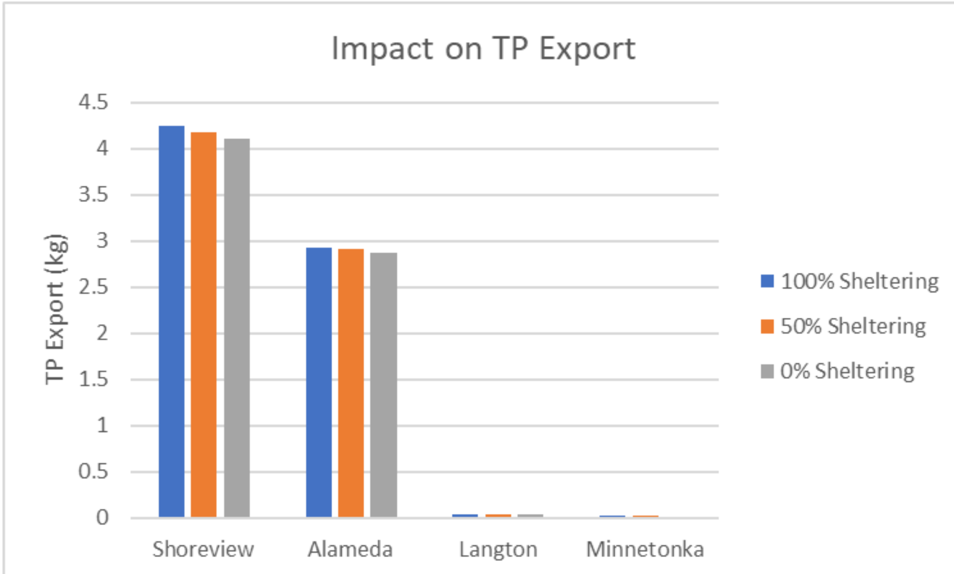

Figure S-4a. *Bar plot of simulation results of cumulative total phosphorus (TP) export mass for each pond model under various wind sheltering reduction scenarios. Further details are available in Section 5.6.*

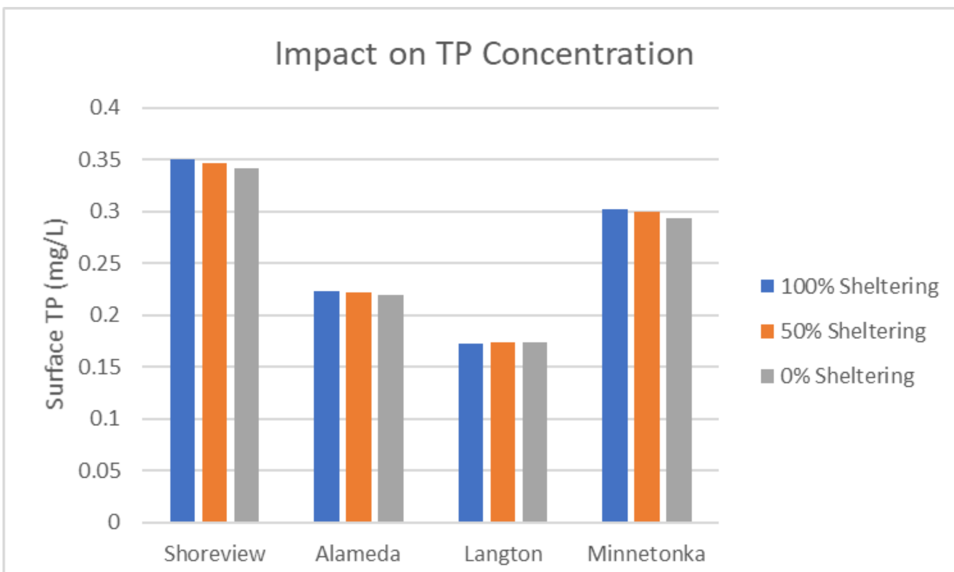

Figure S-4b. *Bar plot of simulation results of mean surface total phosphorus (TP) concentrations for each pond model under various wind sheltering reduction scenarios. Further details are available in Section 5.6.*

8. Pond depth and morphometry was identified as a key control over P concentration. TP export increased as ponds aged and filled in and decreased after dredging (Figure S-5a). The impact on TP concentration in the pond was more variable (Figure S-5b). Dredging was required to maintain pond depth, but we found that dredging was not a cost-effective treatment for phosphorus reduction in the water column or the reduction of phosphorus

export from ponds (See Section 5.6). Building a shallower pond with greater surface area (equal volume) had a similar TP export to the original filled pond with higher TP concentrations in the pond. A deep pond with less surface area had lower TP export and concentrations than the original (actual) ponds.

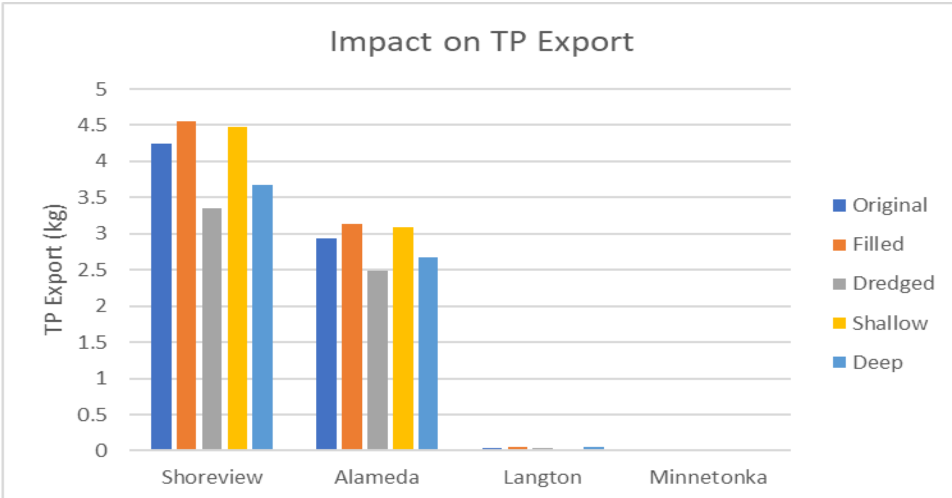

Figure S-5a. *Bar plot of simulation results of cumulative total phosphorus (TP) export mass for each pond model under various bathymetry modification scenarios. Further details are available in Section 5.6. The Minnetonka pond model becomes unstable under the bathymetry modification scenarios, and thus no scenario results are shown on the bar plot.*

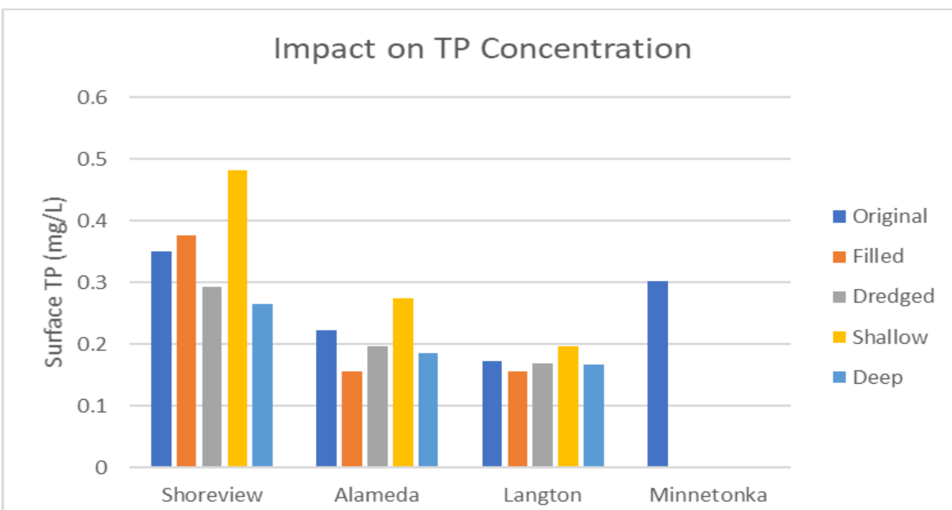

Figure S-5b. *Bar plot of simulation results of mean surface total phosphorus (TP) concentrations for each pond model under various bathymetry modification scenarios. Further details are available in Section 5.6. The Minnetonka pond model becomes unstable under the bathymetry modification scenarios, and thus no scenario results are shown on the bar plot.*

9. An iron-enhanced sand filter pond-perimeter trench is effective at reducing TP export (Figure S-6) but is not a cost-effective treatment to reduce export of phosphorus for the four ponds in our study. 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.

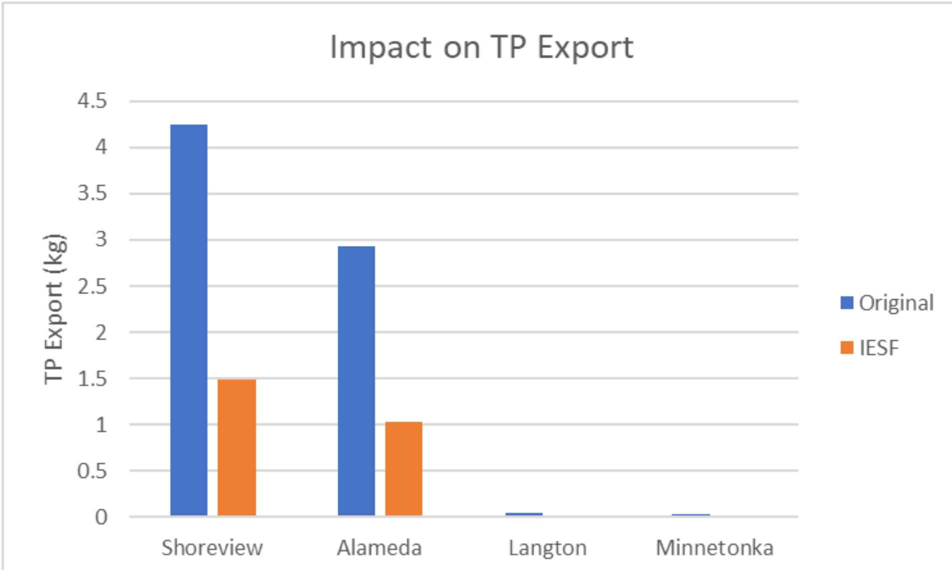

Figure S-6a. *Bar plot of simulation results of cumulative total phosphorus (TP) export mass for each pond model under iron-enhanced sand filter (IESF) bench implementation scenarios. Further details are available in Section 5.9.*

A summary or the effectiveness of various remediation strategies is provided in Table S-1. Through these maintenance activities, we believe it is possible to return stormwater ponds to their original water-quality performance. For best results, we recommend pairing watershed-based methods with the in-pond 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 proper application of all these practices requires the oxic and anoxic sediment phosphate release rates, without which the modeling in this study would be overly hypothetical.

Table S-1. Summary table of simulated remediation strategies for modeled ponds. Effectiveness is evaluated against a threshold of 10% improvement from the base simulation scenario for each pond. Cost-effectiveness is evaluated against a threshold of 10x the overall most cost-effective value (Shoreview alum application at $230 per kg TP export; Minnetonka alum application at $150 per % TP conc.). 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 the Conclusions and Recommendations section.

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

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 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. Ponds store runoff and settle solids along with associated pollutants to the bottom of the pond. However, there is increasing evidence that many ponds may no longer be providing the water-quality benefits of the original design (Taguchi et al. 2018a, 2018b, 2020a, 2020b). Some ponds can re-release phosphorus trapped in the bottom sediments back into the water column (i.e., internal phosphorus loading), primarily under low dissolved oxygen conditions. Phosphorus export from stormwater ponds may be affected by these internal processes related to oxygen and mixing dynamics, as well as sediment chemistry and hydrology. Some of the controls of internal loading can be addressed by maintenance practices, such as adding aerators or alum treatments. Since ponds are part of the watershed network that delivers runoff containing phosphorus into lakes and streams, it is critical to develop effective approaches to maintain them, especially for older ponds, and develop methods to improve their functionality.

This project investigates maintenance and re-design measures required to eliminate phosphorus pollution from stormwater ponds through modeling and data analysis. Results are used to develop guidelines and recommendations to enable stormwater practitioners to cost-effectively manage phosphorus loading and discharge from ponds. We investigated two types of stormwater ponds: upland stormwater ponds and stormwater wetlands (Gulliver et al. 2021). Upland (or constructed) stormwater ponds are considered 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. Maintenance activities in these ponds are not subject to state regulatory programs unless the pond is a designated public water. Stormwater wetlands (or natural wetlands deepened to allow for stormwater ponding) are a historical wetland area that has been modified or is managed to produce a clean water service for a downstream water body that still meets the regulatory definition of a wetland. Typical maintenance activities in stormwater wetlands such as sediment removal, culvert repairs, etc. are generally allowed without the need for a permit from the regulatory programs provided there is no filling, drainage, or excavation aside from sediment removal. In this report, the term stormwater ponds or retention ponds will refer to both types of water bodies.

In this report, we first review the literature to identify the potential cost and effectiveness of various remediation scenarios. We also contact maintenance personnel and lake treatment companies to guide our examination of the literature. We then develop relationships between pond performance and common design and situation characteristics based on existing research, i.e., determining which remediation strategy is most cost-effective.

Both ponds and wetlands receiving stormwater provide water-quality improvements by settling and retaining pollutants associated with suspended solids, where phosphorus is often the primary pollutant target. We demonstrate in our recent research the potential for phosphorus remobilization from pond/wetland sediments under conditions of high sediment phosphorus and organic matter combined with periods of low dissolved oxygen (Taguchi et al. 2018a). Pond oxygen dynamics, which are affected by complex interactions of stratification, water column mixing, and ecosystem dynamics in ponds, have emerged as a key factor in understanding pond phosphorus cycling. To better understand the complexity of processes affecting dissolved oxygen, and their impact on phosphorus management strategies, we have applied and analyzed a detailed water-quality model (CE-QUAL-W2) in conjunction with supplementary wind sheltering model (ANSYS-Fluent) and information to estimate the cost and effectiveness of various remediation strategies in each of four ponds: Alameda pond in Roseville, Shoreview Commons pond in Shoreview, Pond 849A in Minnetonka, and the Langton pond (upstream) in Roseville. The Langton pond is an upstream constructed stormwater pond and the other three are stormwater wetlands.

We used field measurements to verify the performance of the application of CE-QUAL-W2 to retention ponds. Watershed characteristics, pond area, land cover tree canopy, etc. are taken from GIS data that have been collected. We made the following measurements to verify given aspects of the application that vary within ponds, such as periodic stratification and unusually high wind sheltering relative to lakes:

(1) Water-quality measurements, such as total phosphorus and dissolved oxygen concentrations

(2) Stratification measurements, such as temperature, specific conductivity, and wind velocity

(3) Spatial characteristics, such as tree canopy density and watershed parameters

We then used the computer model CE-QUAL-W2 to simulate the phosphorus concentration and stratification in a retention pond. The model incorporated information on transport of phosphorus (P), chloride (from road salt), heat and dissolved oxygen (DO) with the runoff into the retention pond, wind sheltering and stratification of the retention pond, release of P from the sediments and discharge of P downstream into receiving water bodies. To assess these characteristics in a systematic way, we developed standard pond configurations and modeled the following seven remediation techniques in CE-QUAL-W2 (Figure 1):

(2) **Chemical treatment of sediments** with substances such as alum or iron to keep phosphorus in bottom sediments

(3) **Reorientation of outlet works** to draw water from lower in the water column to target low dissolved oxygen concentrations and reduce pond anoxia as well as targeting high chloride concentration and reduce pond stratification

(4) **Mechanical aeration** sufficient to mix the pond/wetland and reduce or eliminate stratification

(5) **Wind sheltering reduction** around ponds/wetlands to promote wind mixing and reduce or eliminate stratification

(6) **Watershed-based methods** to reduce the quantity of solids inflow and phosphorus loading

(7) **Bathymetry modification** of pond depths and surface areas to promote wind mixing and reduce pond phosphorus export

(8) **Iron-enhanced sand filter bench implementation** to treat pond effluent for phosphorus

Results from this study can be used to maintain and apply design retrofits to existing and new ponds to improve pond performance and benefits for use along roadways throughout Minnesota and the United States.

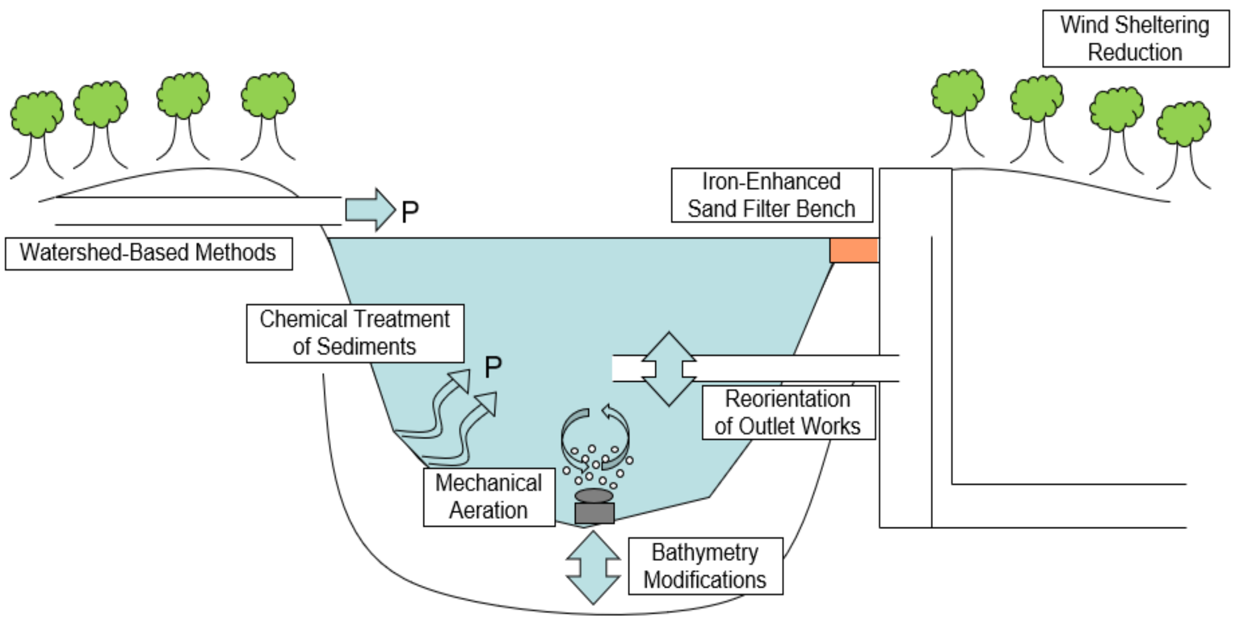

Figure 1. *Conceptual diagram of the seven remediation strategies.*

1.1 Benefits

Stormwater ponds are widely implemented stormwater control measures for runoff quantity and quality control in urban areas and are used to remove pollutants such as solids, nutrients, metals and hydrocarbons from runoff through the settling of particles. Arguably, the most important of these pollutants is phosphorus because it is the limiting nutrient restraining algal blooms in freshwater, including the growth of toxic cyanobacteria. Pond sediments typically act as sinks for phosphorus; however, low DO concentration above the sediments can trigger the release of previously buried phosphorus. The lack of DO in the pond may be due to thermal stratification, chemical stratification due to road salt inputs, poor mixing resulting from sheltering from canopy around the pond, or a combination of all three. Low DO conditions are widely known to induce sediment phosphorus release. However, the presence of anoxic conditions in shallow ponds (Taguchi et al, 2018a, 2018b) is surprising and poorly understood since ponds are assumed to frequently mix. The result is increased phosphorus concentrations in the pond outflows that diminish or even negate their intended function. Given the large number and popularity of ponds implemented for stormwater treatment in Minnesota, pond re-design and maintenance measures that can improve the phosphorus retention in ponds and limit the impairment of receiving surface water bodies is necessary. The proposed research will employ modeling studies to investigate the benefits of different types of pond maintenance actions and re-design so that phosphorus pollution from ponds to receiving water bodies will be eliminated, thereby allowing environmental benefits of ponds to be fully realized. The maintenance and re-design recommendations will also aid cost-effective operation of ponds.

2 Review of Previous Research on the Seven Remediation Scenarios

We reviewed the existing literature to identify the potential cost and effectiveness of the seven remediation scenarios. Additionally, we contacted maintenance personnel and lake treatment companies to guide our examination of the literature for preliminary cost and effectiveness estimates of the seven remediation scenarios. The available data encountered by these literature review efforts are somewhat limited and discussed below, separated by remediation scenario. For comparison, the costs and effectiveness of sediment removal via dredging or excavation are also briefly discussed.

2.1 Chemical treatment of sediments

Various chemical treatments are applied to lakes and smaller ponds/wetlands for phosphorus control. These typically include some form of aluminum, calcium, iron, lanthanum, or other metals. Different chemicals will be more or less effective under different conditions and likewise remain effective for longer or shorter lengths of time. As with any treatment method, managing external inputs of phosphorus is also essential to long-term success (Steinman and Spears, 2019).

Chemical treatments can either inactivate phosphorus directly or by oxidizing the upper sediment layer. Oxidizing chemicals provide alternative electron acceptors in the absence of oxygen to prevent iron reduction and the release of iron-bound phosphorus. Various applications of calcium, nitrate, iron, chloride and lime have been able to reduce sediment oxygen demand and phosphorus release (Steinman and Spears, 2019).
Phosphorus inactivation treatments are more common and have primarily involved aluminum (typically as alum) applications. Recent applications of iron filings have also occurred.

2.1.1 Alum applications

Alum treatments have been conducted in waterbodies for internal phosphorus load reduction and water quality improvement. It is assumed that the applied cost (excluding mobilization) is $1.80 to $2.00 per gallon for alum, and $5.60 per gallon for sodium aluminate (data based on lakes treated in Eagan and Eden Prairie). In Eagan, one upland stormwater pond and four natural wetlands deepened to allow for stormwater ponding were treated with alum in summer 2019, as part of capital improvement projects aiming to treat internal loading within the waterbodies themselves and thereby manage phosphorus load input to downstream lakes. The alum treatment was done by HAB Aquatic Solutions (Lincoln, NE), who used a smaller barge specially built for applying alum in smaller waterbodies. The waterbodies received different doses of alum that was applied over treatment areas ranging between 1.1 and 7.1 ac. The total lump sum cost for the contractor to conduct a buffered alum treatment in the five ponds/wetlands was $173,199, which included costs for alum and sodium aluminate buffer, mobilization, all equipment, material, work and labor and applicable taxes required to complete the application. Normalized by the total amount of buffered alum (alum + sodium aluminate), the cost of application including mobilization is $14/gal for the waterbodies. The cost of application including mobilization was lower at $4.41/gal in lakes, likely because of the difference in scale of application and other site-specific factors such as entry into multiple ponds.

Pre- and post-treatment assessment of water quality has been done in some lakes, but water quality changes in ponds/wetlands treating stormwater are yet to be monitored. Five lakes in the RPBCWD were treated with alum between 2012 and 2019, and the alum treatment reduced the average TP concentrations in the lake as given in Table 1:

Table 1. Pre- and post-treatment concentrations of total phosphorus for lakes in the Riley Purgatory Bluff Creek Watershed District.

Lake	Treatment Year	Avg Epilimnion TP (mg/L)			Avg Hypolimnion TP (mg/L)		
		Pre-alum	Post-alum	Change	Pre-alum	Post-alum	Change
Riley	2016	0.056	0.026	53%	0.502	0.146	71%
Lotus	2018	0.056	0.040	29%	0.429	0.059	86%
Rice Marsh	2018	0.081	0.029	65%	0.107	0.033	70%
Round	2012, 2018	0.040	0.037	7.5%	0.916	0.160	83%
Hyland	2019	0.073	0.030	59%	n/a	n/a	

2.1.2 On-going iron filings applications

Trial applications of iron filings as a phosphorus control measure have been completed at four locations during the winter of 2019-20 and 2020-21 within the Twin Cities Metro area of Minnesota. The cities of Minnetonka, Shoreview, and Eden Prairie (natural wetland deepened to allow for stormwater ponding) and Chanhassen (constructed stormwater pond) have conducted applications to date. The recommended dosing is roughly 4460 lbs iron per acre of pond/wetland surface area. The three cities used a 2- to 3-person crew and a tractor with a mounted fertilizer spreader to apply the recommended dose of iron filings across the surface of a frozen pond/wetland in 2-4 hours. Material costs for these applications are summarized in Table 2. The effectiveness of each treatment is therefore as yet unknown.

The material cost of iron filings varies annually but is approximately $0.42/lb and with shipping approximates $2000/acre of pond/wetland surface area treated. Additional costs relating to logistics labor for application, etc. have not been identified.

Table 2. Material costs for iron filings treatment of four ponds in the Minneapolis-St. Paul Metropolitan Area.

City	Pond/wetland Area (acre)	Dosing (lb/acre)	Unit Cost (/lb)	Material Mass (lb)	Material Cost	Shipping Cost	Total Cost
Chanhassen	1.07	4461	$0.42	4773	$2005	$558	$2563
Eden Prairie	2.17	4461	$0.42	9680	$4066	$558	$4624
Minnetonka	1.63	4653	$0.42	7584	$3185	$558	$3743
Shoreview	2.90	5172	$0.42	15000	$6300	$1240	$7540

2.2 Reorientation of outlet works

This management strategy broadly involves the design of pond/wetland outlet structures to manipulate hydrologic conditions within and downstream of the waterbody, generally focused on increasing storage volumes and detention times and modulating outflows to mimic natural hydrology, prevent flooding, or adapt to prolonged dry periods (Quigley and Lefkowitz, 2015). Maintenance options have traditionally included the installation of multiple outlet structures, which can increase the hydraulic residence time by allowing the temporary storage volume of stormwater to drain slowly through an orifice or perforated pipe outlet (Watt et al., 2004; Schwartz et al. 2017). In lake management, selective withdrawal from the hypolimnion of stratified lakes can be used to remove phosphorus-rich, anoxic water (Nürnberg, 1987). To prevent negative impacts from phosphorus export, however, such a strategy would need to be coupled with some type of effluent treatment to remove phosphorus (Nürnberg, 2007), although this may not be cost-effective in periodically mixed systems because TP would not accumulate in the hypolimnion (Nürnberg, 2019). It may also be possible to selectively withdraw chloride-rich water as a destratification measure in cold-climates with heavy winter road salt applications that accumulate in waterbodies treating stormwater (McEnroe et al., 2013). However, Chiandet and Xenopoulos (2016) examined 50 waterbodies treating stormwater and found that 80% of them were stratified despite the majority of them having outlet structures that drained from the hypolimnion.

Recently, there has been a particular focus on automated controls (valves) for increasing residence time (e.g., via closure of lower-elevation outlets) or for increasing storage capacity by drawing-down the water level prior to a forecasted storm (Kerkez et al. 2016). With active controls, the “permanent” pool in a pond/wetland can be maintained at a higher level for the majority of the time and lowered only when necessary. This may prove beneficial for phosphorus capture because Shamsudin et al. (2017) found that waterbodies treating stormwater with passive outlet controls had pollutant removal rates correlating with permanent pool volumes rather than temporary storage volumes. Such continuous monitoring and adaptive controls (CMAC) in particular have received much recent attention due to the increased availability and robustness of low-cost data loggers, water quality sensors and cellular-connected controllers. These systems provide the capability to operate, program, and continuously monitor waterbodies remotely via cellular data connections, and automatically employ management strategies adapted to current waterbody conditions and immediate future weather forecasts.

A major advantage of such systems is the ability to increase effective storage in an existing pond/wetland (through strategic drawdowns, assuming downstream storage is not an issue) without having to excavate the pond/wetland to increase physical storage capacity. The other advantages of such systems have been evaluated in several pilot projects (WERF 2014). Quigley and Lefkowitz (2015) write: “Stormwater BMPs with forecast-based adaptive control achieve better pollutant removal and runoff reduction outcomes because, among other benefits, they can increase the amount of time that stormwater remains in the treatment facility without compromising capture rate while also reducing the frequency of erosive flows. Further, the technology used to deploy the CMAC also collects performance continuously, allowing for accurate and precise quantification of a BMP’s actual (not theoretical) performance.”
The results of some pilot studies, modeling assessments, and one local project are described briefly below. However, while these results generally seem to show the promise of adaptive control practices for improving effectiveness of stormwater management, the technology is still relatively early in its development and adoption. Thus, very little field data are currently available on the cost-effectiveness of adaptive controls in waterbodies treating stormwater, especially with respect to phosphorus removal.

Water Research Foundation CMAC Pilot Projects (Quigley et al. 2014)

The final report was inaccessible as of this review, but the executive summary (https://www.waterrf.org/system/files/resource/2019-08/INFR1R11ES.pdf) states that the primary focus was on rainwater harvesting and small-scale infiltration practices across several case studies. The summary implies that the results show promise for broader cost-effective stormwater management. There was no mention of ponds and wetlands in the executive summary.

Quigley and Lefkowitz (2015)

In this document, the authors lay out a rationale for incorporating CMAC systems into stormwater management in the Chesapeake Bay watershed. The authors also point out that current crediting systems are based on the design treatment volume of a waterbody treating stormwater; CMAC systems increase treatable volume and therefore should be credited appropriately (based on observations from the systems). The results of several modeling and pilot field studies follow:

1. Simulation of pond/wetland performance with a modification to the outlet structure in which a computer-controlled valve (OptiRTC-type system; see below) was used to drain water from the pond/wetland prior to storm events and/or during dry periods; pond/wetland residence time was increased from 12 to 270 hours, and discharge volumes were reduced by 74% (Opti 2015).
2. Several studies (Gaborit et al. 2013; Carpenter et al. 2014; Klenzendorf et al. 2015) investigating the use of CMAC systems on existing infiltration basins, as well as dry

pond-to-wet pond conversions, were shown to provide benefits in terms of retention of TSS (39% to 90% reduction), nitrate (73%), and ammonium (10% to 84%).

3. Another study found that using a CMAC system to increase residence time of water in a pond/wetland through timed releases under a range of simulated scenarios, without adding any additional storage, achieved a 48% - 68% increase in particulate removal and a 50% reduction in peak flows (Muchalla et al. 2014).

Opti RTC

Opti (optirtc.com) is a company that provides CMAC systems across the U.S. One example of a local Twin Cities installation is the Curtiss Pond project in Falcon Heights (Capitol Region Watershed District), installed in 2015. The system was designed to alleviate flooding in a local park, which was serviced by an undersized stormwater-treating pond/wetland. The pond/wetland was outfitted with an OptiRTC computer controller that operates an outflow valve. The controller has an internet connection, and when a National Weather Service forecast calls for 12.5 mm (0.5-inch) or greater rainfall with > 70% probability, the computer opens the valve to draw down water level by up to 2 feet (Fossum, 2019). The outlet is connected to an underground infiltration gallery, which provides additional storage and infiltration capacity without reducing the park's useable area. The total project cost was $559,000, $70,000 of which was for the OptiRTC system (14%). This adaptive design provided an estimated 58% increase in volume storage (503 cu-m or 17,772 cu ft) vs. a system with only the new infiltration gallery, for a 14% increase in cost (Fossum 2019).

A proposed OptiRTC system in Brevard County, Florida (BCNRM 2017), which was to be added as a retrofit to an existing infiltration basin as a pilot project, was expected to achieve an improvement of 42% total nitrogen (TN) removal, or an additional 21.8 kg (48 lbs) per year of TN removal at a cost of $181,000 (~$57 per kg TN per year over 30 years in 2017 dollars). This removal estimate was based primarily on an increase in residence time.

Colby Lake Watershed

A stormwater retrofit assessment study was done for the Colby Lake watershed in Woodbury, MN. The study modeled expected costs and TP removal from modifying pond/wetland outlet structures to increase storage volume and hydraulic residence time. The first pond/wetland, CL1N3_1, was expected to capture an additional 1.6 kg TP per year for a design and installation cost of $183,000, or $115/kg. The second, CL1E6_2, was expected to capture an additional 0.4 kg TP per year for a cost of $131,500, or $329,000/kg. These costs are based on an assumption of $3.00/ft^2 of waterbody surface area ($52,900/hectare or $130,680/ac) (Washington Conservation District, 2016).

2.3 Mechanical aeration

Aeration and oxygenation are two distinct strategies for increasing the dissolved oxygen content in the hypolimnion of waterbodies with or without necessarily destratifying the water column (Steinman and Spears, 2019). In lake management, it may be advantageous to protect lower water temperatures or to prevent higher TP concentrations in the hypolimnion from mixing with the epilimnion. Aeration involves exposing hypolimnetic water to air, while oxygenation involves injecting oxygen. Both are widespread lake management techniques, although with mixed success. The theory of oxic phosphorus immobilization in sediments requires sufficient iron to be present in sediments and sufficient oxygen to be made available. Site-specific characteristics, undersized aeration/oxygenation systems, or excessive external loading of TP could therefore make this remediation strategy ineffective (Steinman and Spears, 2019).

In addition to researching the available literature, we created a standard set of questions regarding experience with mechanical aeration and attempted to interview, by phone and email, as many entities responsible for managing stormwater ponds in Minnesota as we could reach. In the end, we contacted 18 entities and received responses from 13 of them. Municipal respondents included the cities of Bloomington, Chanhassen, Chaska, Eagan, Eden Prairie, Edina, Minnetonka, Mound, Ramsey, Shoreview, Shorewood, and St. Cloud. Regulatory respondents included Dakota County, Hennepin County, Minnesota Department of Transportation (MnDOT), Riley Purgatory Bluff Creek Watershed District (RPBCWD), Ramsey-Washington Metro Watershed District (RWMWD), and Shakopee Mdewakanton Sioux Community. Consultant respondents included Jacobs and HTPO Consulting.

Our surveys revealed that approximately half of our respondents had no experience with mechanical aeration. The Cities of Eagan, Edina, and Mound only had aeration experience with respect to lake applications. The City of Chanhassen reported that many ornamental fountains have been installed on small private waterbodies by homeowners associations, but these did not seem to improve water quality. According to Watt et al. (2004), ornamental fountains are not effective aerators and can result in sediment resuspension. Of those entities with mechanical aeration experience, only Jacobs, Shoreview, and Bloomington had specific data to share with us regarding design, costs, and performance of mechanical aeration systems. These are described below.

Jacobs

David Austin, PE, of Jacobs recommend sintered rubber hose (soaker hose) diffusers because they are robust and easily refurbished every few years. Currently, Jacobs is using a battery-free solar-powered aerator system available on Amazon.com for $4,399 (https://tinyurl.com/u4wk8np) and able to treat up to 2 acres of pond/wetland surface area. Steve McComas of Blue Water Science recommended a similarly sized aeration pump from Pentair available for $3,151.71, including six diffusers (https://tinyurl.com/v6c8u34).

City of Shoreview, MN

An ornamental fountain (2-hp or 1.5 kW) and two aeration units (3/4-hp and 1-hp or 0.56-kW and 0.75 kW) were installed in a 1.2 hectare (2.9-acre) wetland (natural wetland deepened to allow for stormwater ponding) for aesthetic and water quality improvement purposes at a material cost of $5000-$6000 and a total project cost of $10,000. Primary motivators for the implementations were excessive free-floating macrophyte (FFM) cover and foul smells. These aeration units appear to function on the basis of water circulation rather than bubble diffusion. The pond was not monitored prior to or following the implementation of these units, but no apparent improvements in water quality were noted. It appears that the wetland is too shallow for these systems to be effective, requiring further application of aeration units.

City of Bloomington, MN

An aeration system was installed on a 13 hectare (32-acre) lake in the 1970s as part of a groundwater source project to maintain the lake water level at a cost of $40,000 including the drilling of a 41 m (135-ft) well. In 2009, the system was reconfigured to recirculate water from the lake through a 1,140 Lpm (300 gpm) pump which releases it over a stair-step aeration structure. The primary motivation for this change was stormwater management. The reconfiguration did require a $30,000 dredging project to remove sediment from the pump intake area. Since then, winter fish kills appear to have been reduced.

2.4 Wind sheltering reduction

Chiandet and Xenopoulos (2016) studied 50 constructed stormwater ponds and 5 natural ponds and found that ponds with more sinuous shorelines (lower surface area: perimeter ratio) had better water quality, which they attributed to greater macrophyte populations along the shoreline. They also found that length and length:width ratio were important, but they were unsure of specific recommendations to make. In terms of wind sheltering, they found that larger fetch lengths in the direction of prevailing winds could enhance mixing and decrease stratification. They acknowledged the apparent contradiction between wanting increasing mixing to elevate DO concentration while wanting to enhance settling and reduce resuspension. Their analysis suggested that an optimal pond depth would be 0.8-1.2 m.

Bentzen et al. (2009) studied a stormwater-treating waterbody with high wind exposure and recommended that vegetation be implemented to enhance wind sheltering or that the pond/wetland be deepened in order to prevent sediments from being exposed to flow velocities where resuspension became possible. They cited a critical wind value of 8 m/s for waterbodies shallower than 1 m.

One potential, although untested, method to reduce stratification and improve hypolimnetic dissolved oxygen concentrations is to reduce wind sheltering. When constructed stormwater ponds are first built, they are typically exposed to wind mixing as a result of land clearing, and the long-standing assumption has been that shallow ponds are fully mixed and oxic (McEnroe et al., 2003; Chen et al., 2019). But as time passes, trees and tall buildings may emerge and shelter constructed stormwater ponds from wind mixing (Erickson et al., 2018; McEnroe et al., 2013; Herb et al. 2006). Trees are able to shelter the water surface for a distance extending 35-100 times the tree height (Markfort et al., 2014). In small waterbodies treating stormwater, it is not uncommon for the height of surrounding trees to be on the same order as the fetch length of the waterbodies. Xenopoulos and Schindler (2001) have also observed that annual wind velocities over lakes increased following forest fires and clear-cutting operations that effectively reduced sheltering from winds coming from specific directions. Furthermore, they found that water mixing only increased in smaller lakes because larger lakes had sufficiently large fetches to provide sufficient wind exposure. Even with enhanced wind exposure, however, it is still possible that only high wind speeds will cause sufficient mixing to oxygenate the hypolimnion of shallow stormwater-treating waterbodies (Chen et al., 2019).

Despite potential benefits to phosphorus retention, remediation efforts on stormwater-treating waterbodies have often focused on reducing wind exposure due to concerns relating to fine sediment resuspension (Andradottir, 2017; Bentzen et al. 2009; Watt et al., 2004). Wind mixing can also generate currents that enhance the movement of water between the inlet and outlet of a pond/wetland, thereby decreasing the residence time (Andradottir, 2017).

Tree removal efforts may also face potential stakeholder resistance. Although many stormwater-treating waterbodies are artificial, constructed ponds or else artificially-modified natural wetlands, many stormwater-treating waterbodies have the appearance of being “natural” due to the presence of wildlife and mature vegetation. As such, tree-removal operations may face opposition from nearby residents. One approach to address concerns is to replace tree habitat with diverse, native habitat or low-lying pollinator species that can allow wind to pass through corridors distributed around the waterbody (Taguchi et al., 2020a). If effective, this remediation strategy could be one of the most cost-effective.

2.5 Watershed-based methods

The traditional form of pretreatment in most stormwater-treating waterbodies is a sediment forebay near the inlet structure that can remove larger sediment particles. This allows for more localized sedimentation that can simplify regular dredging to preserve water storage volumes (Marsalek et al., 2008). Still, many older constructed stormwater ponds were built without forebays (Anderson et al. 2002).The coarse sediments captured by forebays generally lower toxic metal concentrations and nutrient concentration (Blecken et al., 2017; McNett and Hunt, 2011). However, Schifman et al. (2018) found evidence to the contrary.

Various other pretreatment devices and watershed management strategies have varied pollutant removal abilities and costs for implementation and maintenance. One common strategy is the implementation of various green infrastructure throughout a watershed to reduce water volumes, peak flows, and pollutant loads that reach sensitive waterbodies further downstream (Taguchi et al. 2020a). Applying the same principles in the drainage areas of stormwater ponds would increase their ability to effectively manage stormwater by reducing the burden of pollutant accumulation over time (Erickson et al. 2018a). Such watershed management approaches also increase resilience to changes in drainage area land use or precipitation patterns (Moore et al. 2016). An increasingly popular approach to cost-effectively managing watershed nutrient loads is targeted street-sweeping for gross-solids (e.g., leaves and grass clippings) (Kalinosky et al. 2014).

2.6 Iron-enhanced sand filter bench implementation

Sand filters are one of the oldest water-treatment techniques and were formally developed as a stormwater management in the 1980s (Taguchi et al. 2020a). They function by physically trapping particulate pollutants and adsorbing a limited amount of dissolved pollutants. Recently, various enhancements have been added to sand filter media in order to increase the adsorption capacity of dissolved pollutants including phosphorus (Erickson et al. 2018a). In particular, iron-filings have become a common filtration media enhancement in Minnesota for the treatment of dissolved phosphorus (Erickson et al. 2012). One popular application of iron-enhanced sand filters (IESFs) has been in trenches along the perimeter bench of stormwater ponds (Belden and Fossum 2018).

2.7 Bathymetry Modification

Stormwater ponds perform many functions, so it is difficult for their forms to be optimized for all of the desired functions at once (Erickson et al. 2018a). In the case of phosphorus management, pond volumes, depths, surface areas, and flow path lengths are just a few of the characteristics that drive important physical, chemical, and ecological mechanisms controlling phosphorus retention.

Brink and Karnish (2018) analyzed stormwater pond (both constructed stormwater ponds and natural wetlands deepened to allow for stormwater ponding) data from International Stormwater BMP Database (http://bmpdatabase.org/) and found that all metal removal efficiencies correlated with TSS removal, reinforcing the importance of particulate removal. They also found that different conclusions could be drawn depending on whether performance was evaluated on the basis of concentration reduction (making ponds/wetlands receiving large influent concentrations appear to be performing better than ponds/wetlands receiving low influent concentrations, even if both ponds/wetlands have the same effluent concentrations) or pollutant mass removal. In terms of pond/wetland morphology, Brink and Karnish found that larger permanent pool volumes correlated with greater pollutant mass removal, regardless of the temporary storage volumes. They also found that larger pond/wetland surface areas relative to watershed drainage areas also resulted in improved performance.

Ferrara et al. (2018) conducted a modeling analysis of a hypothetical stormwater-treating waterbody, focusing on inlet and outlet locations (on opposite side of a rectangular basin and in various configurations relative to the centerline). They found that hydraulic residence time and sedimentation potential were greatest when the flow from the inlet is able to reattach to the basin sidewall before reaching the outlet, and that this may be more likely to occur if the inlet and outlet are placed on the same side of the basin centerline.

Watt et al. (2004) stated that in-line stormwater-treating waterbodies (those with continuous inflows outside of rainfall events) will have a reduced ability for particulate sedimentation because of the reduced hydraulic residence time associated with having a continuous discharge or the permanent pool.

Jansons and Law (2007) found that teardrop- and kidney- shaped stormwater-treating waterbodies had longer hydraulic residence times than rectangular basins of the same surface area. Glenn and Bartell (2010) defined a dimensionless quantity, the Short-Circuiting Index (SCI), based on pond/wetland detention time, geometry, and inflow that can be used to design stormwater-treating waterbodies with minimal potential for hydraulic short-circuiting.

The presence of bottom-feeding fish (often carp but also bullheads, minnows, and other species) have also been attributed to water turbidity and is believed to increase sediment phosphorus release (Webster et al. 2001; Parkos et al. 2003). How much phosphorus is contributed by fish depends on the feeding behavior of the fish (Andersson et al. 1988), the density of fish living in a pond/wetland (Roberts et al. 1995), and the size of the fish (Driver et al. 2005). Depending on the phosphorus loading a pond/wetland receives, fish activity can make up a significant portion of the total load. However, waste excretion generally comprises a greater proportion of mobilized phosphorus than any sediment suspension (Hart and Harding 2015). Still, it remains unclear whether fish removal is an effective or efficient management strategy to improve water quality. Fish removal can be difficult, depending on the size and physical characteristics of a pond/wetland, and may become a regular operation rather than a one-time treatment. In one study, Bajer and Sorensen (2015) noted that although turbidity was greatly reduced following carp removal, total phosphorus concentrations did not appear to be affected. This may be because fish primarily translocate phosphorus from sediments or vegetation into the water but do not actually add new phosphorus from any outside source (Andersson et al. 1988). Still, regular removal of fish would permanently remove any phosphorus that had been assimilated into the fish biomass, although this may not be cost-effective compared to other management strategies.

Ideal bathymetry configurations are still not well understood, and beneficial principles are not always incorporated into stormwater pond designs. One common retrofit to improve some characteristics (e.g., increase the flow path length between the inlet and outlet structures to allow more treatment time) is the implementation of flow-lengthening baffles (Erickson et al. 2018a). To retroactively address more foundational components of a pond’s bathymetry, it may be necessary to apply geotechnical modifications, potentially in the form of dredging.

Dredging in stormwater-treating waterbodies has been performed mainly for maintenance and restoring storage capacity by removing sediment deltas formed at the inlets. This removal is recommended to occur once every ten years (Erickson et al., 2013), although this is seldom done due to financial and logistical limitations (Erickson et al., 2018a). This type of large-scale maintenance is estimated to cost 80% of the original pond/wetland construction cost (Clary and Piza, 2017). Much of the costs are determined by sediment disposal costs and site restoration, although steps can be taken to minimize both (Erickson et al., 2018a). Occasionally, ponds/wetlands are dewatered to allow excavation as opposed to dredging. Each method has its own advantages and disadvantages, and either can result in substantial environmental impacts in the absence of proper planning (Steinman and Spears, 2019). However, Hosomi and Sudo (1992), in a modeling effort, calculated that phosphorus external loading reduction, artificial aeration, and phosphorus inactivation may be more effective at prevent phosphorus internal loading than sediment dredging. Whether this holds true likely varies due to site-specific factors.

Molenwiel Pond in The Netherlands

A stormwater-treating waterbody (size not specified) was dredged to improve water quality and eliminate cyanobacterial blooms. Two separate dredging operations (one month apart) were required to successfully remove all sediment. Intact sediment core incubations were able to confirm that sediment TP release was greatly reduced (from approximately 14 mg/m^2/day to approximately 5 mg/m^2/day) following the removal of 850 m^3 (660 cu-yd) of sediment. However, the cost of dredging was $139,000 US and the cost of restoration following dredging was $241,000. Furthermore, the water quality is expected to gradually deteriorate until it is once again experiencing cyanobacterial blooms because external sources of phosphorus have not been controlled (Steinman and Spears, 2019).

City of Eagan, MN

Over the past decade, 30-35 stormwater-treating waterbodies have been dredged in the City of Eagan. Dredging cost approximately $43,000 per waterbody, which includes costs for special handling and disposal of contaminated sediments. The costs are high because nearly 50% of the basins have PAH-contaminated soils in the city.

Coon Creek Watershed District / Anoka Conservation District

Various assessment methods applied in the Pleasure Creek watershed in Anoka, MN, including GIS, site visits, WinSLAMM modeling, and construction and O&M cost estimations were able to provide sediment volume-specific estimates of dredging costs depending on contamination levels: $15.4/cu-m($20/cu-yd), $27/cu-m ($35/cu-yd), and $38.6/cu-m ($50/cu-yd) for Level 1, Level 2, and Level 3 type excavated soils (Level 1: suitable for local re-use; Level 2: Suitable for industrial use; Level 3: contaminated waste disposal required). A proposed dredging at Pond 304 was estimated to cost $2070 (Level 1), $3160 (Level 2), or $4,250 (Level 3) per kg TP per year over 30 years (Anoka Conservation District, 2016).

Colby Lake Watershed / Washington Conservation District

Various assessment methods applied in Colby Lake watershed in Woodbury, MN, including GIS, site visits, P8 modeling, and construction and O&M cost estimations were able to provide TP removal cost efficiency estimates relating to sediment dredging ranging from $4,100 - $11,900 per kg TP per year over 30 years (Washington Conservation District, 2016).

St. Cloud Pond 52A

The City of St. Cloud dredged Pond 52A in 2018 with 992 metric tonnes (902 U.S. tons) of sediment removed over 85-90% of the pond. The pond was built in 1998, with a surface area of 0.21 hectares (0.52 acres). The total dredging was 0.4 m over the pond, and it was found that 4 of 5 subsequent sediment cores still had black sediment with high organic content at 20 cm depth. Thus, the high organic sediment was over 60 cm, resulting in a sediment deposition of over 3 cm per year.

3 Development of a Retention Pond Phosphorus Model

A hydrodynamic and water quality model is necessary to replicate pond/wetland conditions observed in the field and predict the impact that each of the seven remediation scenarios could have on these conditions. Of particular interest are conditions conducive to sediment phosphorus release. Sediment phosphorus release can be affected by a variety of factors, but it is largely understood to depend heavily on the concentration of dissolved oxygen in benthic water adjacent to the sediment surface (Steinman and Spears 2019).

Dissolved oxygen can be affected by stormwater inflows, sediment oxygen demand, biological oxygen demand, physical mixing, chemical diffusion, and open water surface area. These processes are sensitive to water temperatures, atmospheric conditions, stratification, and water surface exposure to atmospheric exchange, which could be reduced in the presence of ice cover or thick floating vegetation. Stratification, meanwhile, is a product of differences in water density relating to temperature and concentrations of dissolved solids including winter applied road salt.

3.1 Numerical Model Development

3.1.1 Hydrodynamic and Water Quality Model

CE-QUAL-W2 is a two-dimensional, laterally-averaged, hydrodynamic and water quality model developed by the US Army Corps of Engineers in the 1970s and has continued to evolve since then. It has been applied extensively to rivers, lakes, reservoirs, estuaries, and other water bodies or combinations of water bodies. Detailed information on CE-QUAL-W2 can be found in the User Manual (Wells 2019).

A CE-QUAL-W2 model is comprised of cross-sectional “segments” divided into depth “layers.” The bathymetry is, therefore, defined by specifying the width of each layer present in each segment (Figure 2). The model calculates the variation of flow rates, temperature, and other quantities over depth and the length direction, assuming the water body is well mixed in the width direction. For application to stormwater ponds, the length direction is defined as the direction from the pond inlet to the outlet. The model is unsteady, calculating the time variation of the physical and chemical parameters over time in response to varying flow inputs and weather conditions.

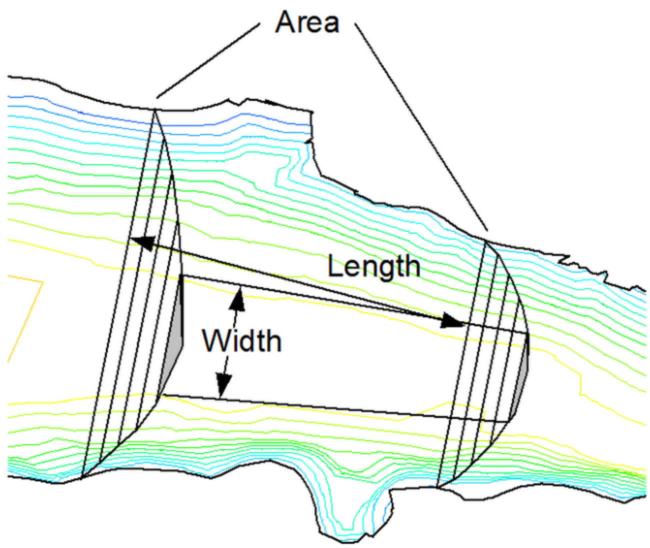

Figure 2. *A visual depiction of segment layers in CE-QUAL-W2 compared to bathymetric contour data. From "WMS:CE-QUAL-W2 Bathymetry" (2018).*

The hydrodynamic model uses Leonard's ULTIMATE numerical transport scheme algorithm and is capable of predicting surface water elevations, water velocities, and temperatures, which affect water density. For this reason, salinity and dissolved solids are also included. Additional water quality computations occur after the hydrodynamic computations and can include a large variety of water quality parameters relating to chemistry and biology (Wells 2019).

Water temperature, including stratification, is affected by temperature, dissolved solids, salinity, wind velocity, solar radiation, and other factors. CE-QUAL-W2 includes an atmospheric heat exchange model that takes into account different forms of solar radiation and heat exchange from the waterbody (Wells 2019). The model uses weather data at, for example, hourly time steps, to calculate the time variation of surface heat transfer between the water and the atmosphere.

Dissolved oxygen in the water column is modeled from the balance of sources (surface aeration, water inflows, photosynthetic production) with sinks (biological oxygen demand, sediment oxygen demand, water outflows). The distribution of oxygen within the water column is also subject to wind mixing and temperature stratification. Sediment oxygen demand can be defined as a zero-order reaction that varies only with temperature or as a first-order reaction that considers the settling of organic matter. Depending on the availability of additional sediment data, sediment diagenesis kinetics between the sediment and overlying water can also be calculated (Wells 2019).

Phosphorus is considered in various forms both in sediments and dissolved in water. The model also considers physical, chemical, and biological drivers of fluxes between different phosphorus forms (Figure 3).

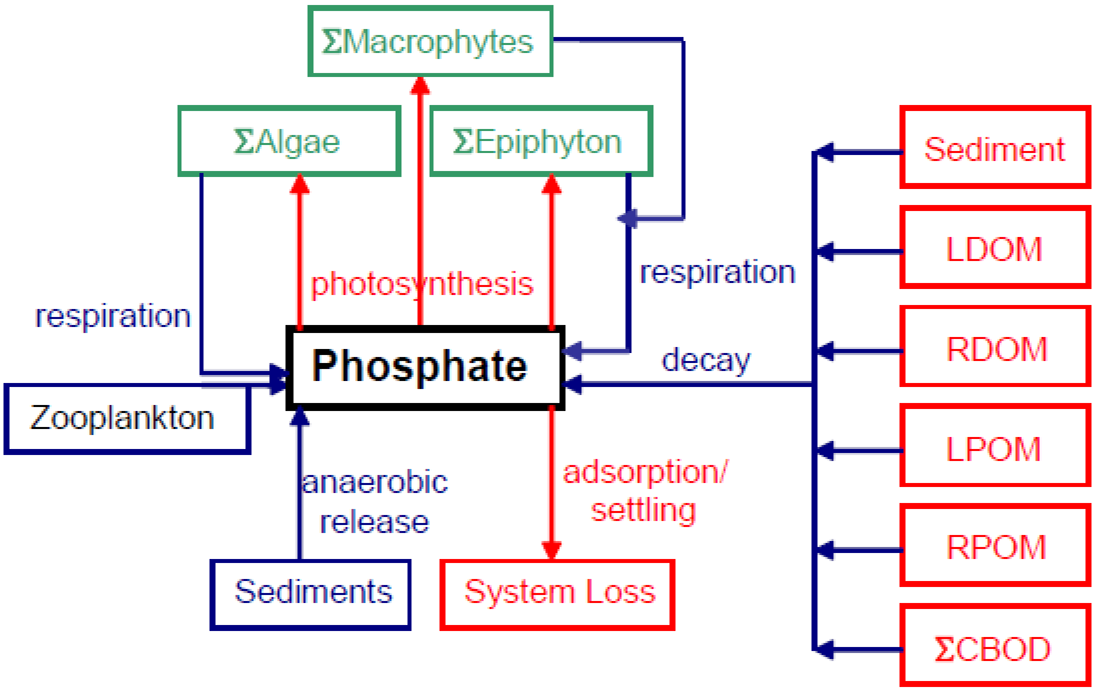

Figure 3. *A schematic of phosphorus fluxes between various model compartments. From Wells (2019). LDOM is labile dissolved organic matter; RDOM is refractory dissolved organic matter; LPOM is labile particulate organic matter; RPOM is refractory particulate organic matter; and CBOD is carbonaceous biochemical oxygen demand.*

3.1.2 Calibration Study Site

Alameda pond is a natural wetland that was altered to allow for permanent stormwater storage. This 1.05-ha pond/wetland was selected as the initial site because of available data from prior studies (Herb et al. 2017; Janke et al. 2022; Oberts 1998; Taguchi et al. 2018a, 2020b). Herb et al. describe the pond/wetland as draining 115 ha of mostly residential land with some commercial/institutional developments conveyed through a 40-inch diameter inlet pipe and a solid rectangular weir outlet structure.

3.1.3 Data Collection

Building a CE-QUAL-W2 model requires data to inform the bathymetric geometry, initial conditions, boundary conditions, hydraulic parameters, kinetic parameters, and model calibration. Inflow and outflow measurements were available for the Alameda pond from 7/1/2015 to 6/30/2016 along with corresponding chloride concentration measurements (Herb et al. 2017). Additionally, inflow and outflow measurements are available from 7/21/2017 to 5/29/2018 with corresponding nutrient concentration measurements (Taguchi et al. 2018a). Water surface elevation is available from 8/30/2016 to 10/28/2019 (Taguchi et al. 2018a) and from 7/19/2017 to 6/8/2020 (Capitol Region Watershed District n.d.). And for calibration purposes, intermittent water quality profile measurements of temperature, electrical conductivity, and dissolved oxygen are available from 12/12/2016 to 10/15/2019 (Taguchi et al. 2018a).

Geo-referenced depth measurements were provided by WSB Engineering (not published). This dataset consisted of GPS coordinates with associated depth measurements. CE-QUAL-W2 does not include a native process for reformatting bathymetric data into the required format of segments and layers. To facilitate this process, we developed a MATLAB (*MATLAB* 2019) script that allows the user to specify the pond/wetland inlet and outlet locations, the number of cross-sectional segments to be extracted along the inlet-outlet transect (Figure 4), and the number of depth layers for which widths are to be calculated against the bathymetric data (Figure 2). This automated process greatly simplifies the process of modifying the bathymetric geometry or the model computational grid resolution. The script also allows the user to specify, if necessary, the approximate boundary geometry (shores and banks) that are typically excluded from bathymetric surveys but are necessary for flood storage when the hydrodynamic model is run (Figure 5). The output of the MATLAB script is a finalized bathymetry input file that can be interpreted by CE-QUAL-W2 and verified for accuracy in the native pre-processor.

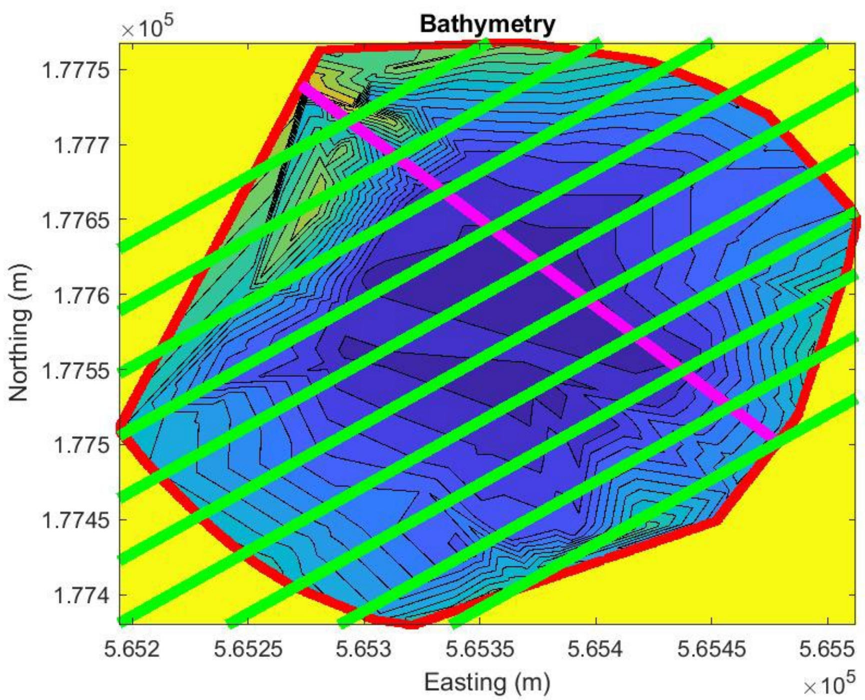

Figure 4. *A visual representation of the inlet-outlet transect (magenta) and cross-sections (green) along which bathymetric data is to be extracted from the elevation contours (blue-to-yellow gradient bounded by red).*

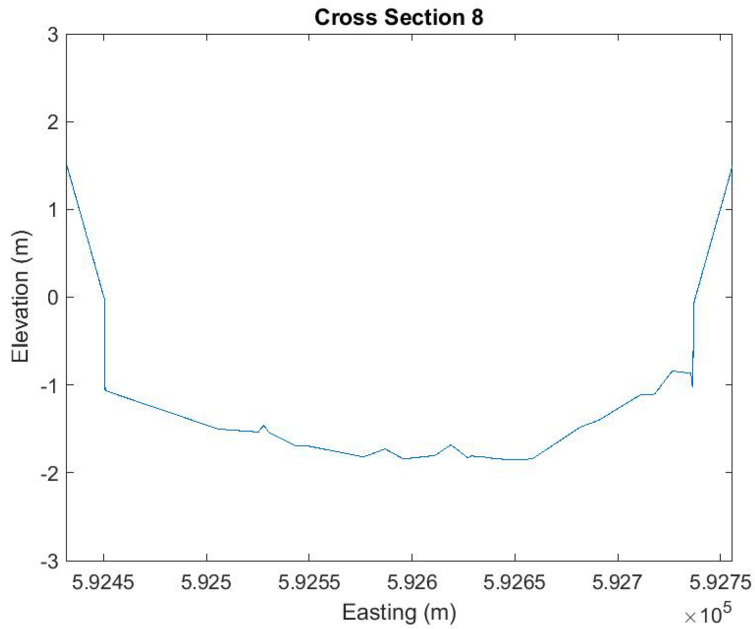

Figure 5. *An elevation profile along one cross-section combining values interpolated from bathymetric data and user-defined bank geometry. Elevation 0 is the normal water surface level.*

Inflow time series data were available from a previous study (Herb et al. 2017). Therefore, accurate inflow values could be specified for specific times during the model run. In the absence of these data, inflow hydrographs could be generated for typical conditions using a watershed modeling program such as EPA SWMM (Rossman 2015) or P8 (Walker and Walker 2017). Sediment oxygen demand is based on data from laboratory sediment core incubations (Taguchi et al. 2020b). And weather data, including solar radiation, is available from the University of Minnesota Baker Observatory (unpublished).

3.2 Model Configuration and Implementation

For this study, the hydrodynamic and water quality parameters of interest are dissolved oxygen, phosphorus, and temperature. Phosphorus dynamics depend heavily on dissolved oxygen concentrations, and dissolved oxygen depends heavily on stratification dynamics. Therefore, the first priority has been to improve the accuracy of stratification and mixing dynamics.

One approach has been to refine the computational grid (vertical element thickness and number of cross-sectional elements) to reduce the effect of numerical diffusion and other unexpected behaviors. Refining the vertical grid from 0.175-m thick elements (orange) to 0.145-m thick elements (red) strongly affected the model behavior, whereas additional refinement to 0.100-m thick elements (green) did not greatly impact the model behavior (Figure 6). A finer grid resolution greatly increases the computational run time, so further optimization of vertical and horizontal grid resolution will be undertaken to maximize model accuracy without unnecessarily increasing the model complexity.

Figure 6. *Comparison of electrical conductivity measurements at various depths (surface, middle, and bottom of the water column) and influence of modifying vertical grid resolution.*

We were able to improve agreement between field observations of thermal stratification and simulated values from the model by increasing the value of the Solar Radiation Absorbed at Surface (BETA) parameter (Figure 7). We believe that more realistic representations of light absorption will allow the model to more accurately represent the impact of FFM presence on light attenuation, heat inputs, thermal stratification, and dissolved oxygen in the water column. Varying BETA values over the course of the year (as FFM cover expands and contracts) should allow for a finer calibration of the impact of FFM on model dynamics.

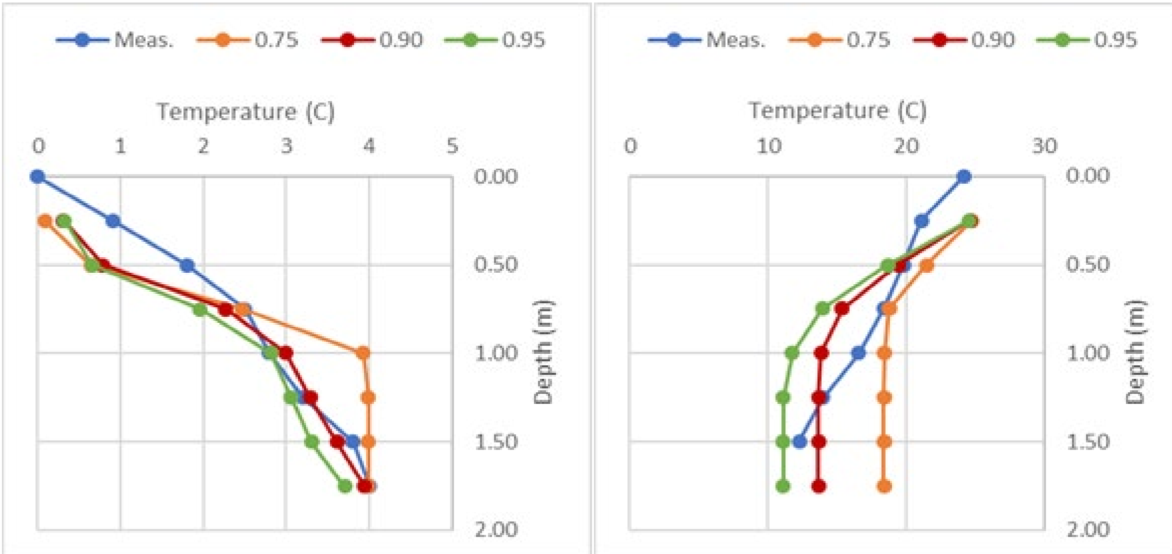

Figure 7. *Temperature profile measurements for Alameda pond on 1/31/18 (left) and 5/27/18 (right) comparing measured values (blue) to simulated values (orange, red, and green) with different Solar Radiation Absorbed at Surface (BETA) values. BETA=0.9 gives the best prediction on 1/13/18. Further refinement on chloride concentrations will improve 5/27/18 predictions.*

Field observations of pond water surface elevation showed steady and gradual drop over periods with little or no precipitation. These water level decreases are in excess of expected evaporation rates, which can likely be attributed to exfiltration to groundwater. Experimentation with defining lateral withdrawal flows (QWD) across all segments has led to better agreement between the observed and simulated water surface elevations over these periods (Figure 8).

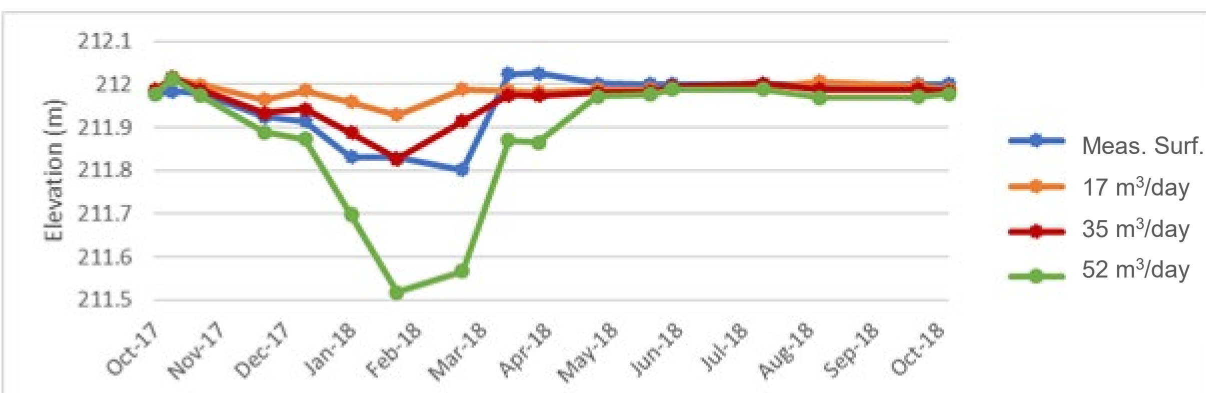

Figure 8. *Water surface elevation measurements for Alameda pond from 10/12/17 to 10/11/18 comparing measured values (blue) to simulated values (orange, red, and green) with different selective withdrawal flow values, showing that, in this case, groundwater flows of 0.0004 m^3/s will adequately simulate groundwater withdrawals. Flow values are distributed evenly across the cross-sectional pond segments.*

4 Field Calibration and Verification of the Retention Pond Model

4.1 Field Data Acquisition

Hydrologic and water quality data, collected by the authors in previous projects, were used to parameterize and calibrate the CE-QUAL-W2 model for the Alameda pond in Roseville, MN. These data collection efforts are described briefly here; further details can be found in reports describing the previous projects (Herb et al. 2017, Taguchi et al. 2018a).

4.2 Site Description

Alameda pond (Figures 9 and 10) is a 11700-m^2 (2.9-ac) pond located upstream of the Villa Park Wetland (Lake McCarrons watershed) in Roseville, MN. The pond's drainage area is roughly 1,153,000 m^2 (285 ac) of mostly residential land use, with a few ponds and wetlands located within (and connected to) the drainage network. It borders a park area to the southwest and is completely enclosed by mature tree canopy. It was historically a wetland that was connected to the area's storm drain network, likely in the 1960's. It has never received any dredging maintenance. The pond's maximum depth is roughly 2.1 m (7 ft), with a mean depth of 1.4 m (4.5 ft). It has a single inlet (1-m (40-in) diameter concrete pipe) and the outlet is a rectangular weir roughly 0.9 m (3 ft) wide.

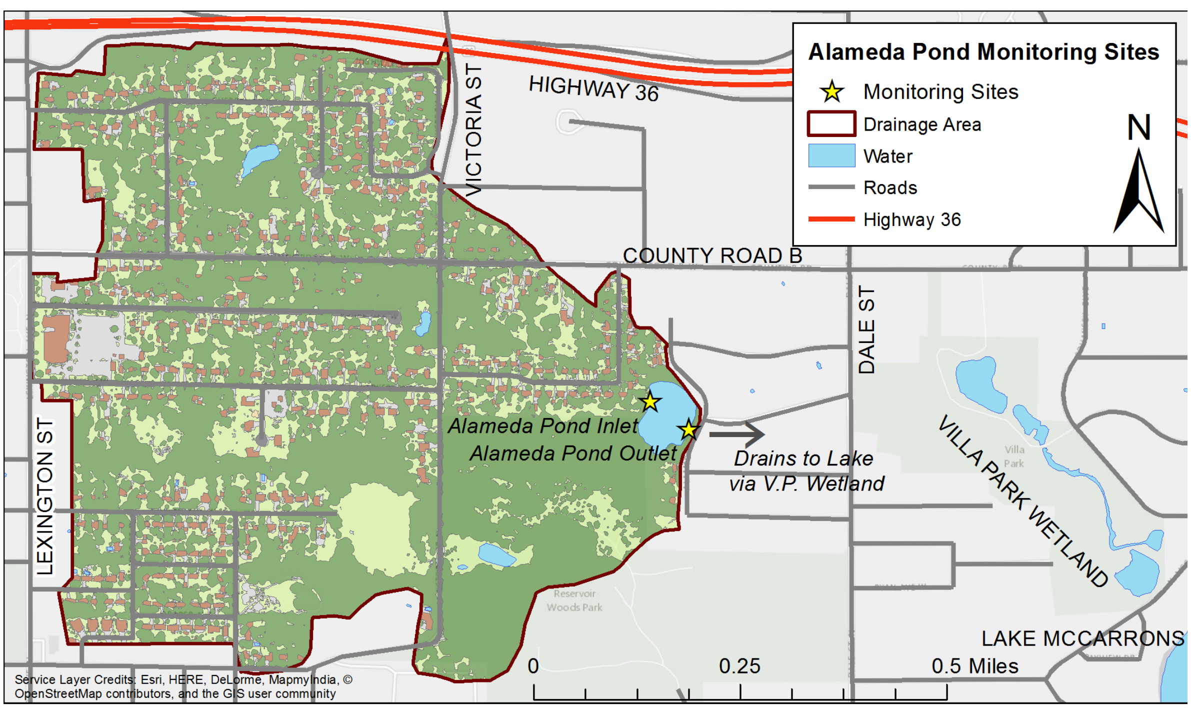

Figure 9. *Watershed of the Alameda pond site and location of inlet and outlet monitoring sites at the pond. Figure from Herb et al. (2017).*

Figure 10. *Photo of Alameda pond with partial free-floating macrophyte cover in May 2018. Photo credit: Vinicius Taguchi (2018).*

4.3 Data Collection

The Alameda pond has been the subject of several overlapping studies since 2015. The earliest (Herb et al. 2017) from 2015 – 2017 included monitoring for basic hydrology and electrical conductivity to study road salt accumulation in and release from the pond. Water level measurements were recorded in the pond (Solinst Levelogger), with electrical conductivity (Hobo Onset Conductivity probe) monitored in the outlet pipe; water level was converted to discharge based on a weir equation applied at the outlet. Inlet monitoring of discharge and conductivity were carried out with an ISCO 4150 flow logger and Hobo Onset Conductivity probe. Profiles of temperature, conductivity, and dissolved oxygen measurements were taken periodically in the pond center, with samples collected from the pond water as well as from outflows and inflows. These samples were measured for chloride concentration.

A second study in the pond, conducted from January 2017 – June 2018, focused on intensive monitoring of inflows and outflows to better understand the fluxes of phosphorus within and from the pond (Taguchi et al. 2018a). ISCO autosamplers were installed at the inlet (July 2017) and at the outlet (January 2017) to record flows year-round and collect runoff and snowmelt samples, which were analyzed for various forms of phosphorus. Conductivity loggers (Onset Hobo) were installed at the pond's inlet and outlet, and water level was logged using a Solinst Levelogger. The Capitol Region Watershed District also monitored water levels at the site beginning in 2018. Profiles of temperature, conductivity, and dissolved oxygen measurements were recorded every few weeks from the center of the pond year-round over the study duration, and epilimnion and hypolimnion water samples were analyzed for various forms of phosphorus (see Taguchi et al., 2018a, for details of field and analytical methods). Inflow and outflow monitoring ended in July 2018 at the conclusion of the project, though some sparse data collection (profiles and water column sampling) was carried out until the next project began in 2019.

Data from the two aforementioned projects, Herb et al. (2017) and Taguchi et al. (2018a), were used in model development. Data types and collection periods are shown in Table 3. The most relevant and detailed data were collected during 2017, the period used to develop, parameterize, and calibrate the CE-QUAL-W2 model for the pond. The 2016 period included hydrology, temperature, and conductivity data, but only a single profile of TP concentration measurements on October 13th; all of these data were used for verification of the model. The 2018 period included P data but inflow measurements ended mid-summer, and thus it was not deemed sufficient for model verification.

Table 3. *Type and timeline (by year and quarter) of monitoring data collected in two previous projects that were used in the development of the CE-QUAL-W2 model for the Alameda pond. 'x' indicates relevant data were collected for all or most of that year's quarter. '(x)' indicates that there are gaps in the data or that some of the data are of poor quality due to monitoring constraints. Profile data were generally periodic (every 2-6 weeks, depending on time of year, and included temperature, specific conductivity, and temperature). The first study (Herb et al. 2017) was conducted from 2015-2017, overlapping during the summer field season of 2017 with the second study (Taguchi et al. 2018a). Note that outlet discharge was estimated from pond water level and a weir equation applied to the outlet for 2016, and sparse grab sampling was used to measure P at the Inlet site in early 2017.*

		2016				2017				2018	
Location	**Measurement**	**Q1**	**Q2**	**Q3**	**Q4**	**Q1**	**Q2**	**Q3**	**Q4**	**Q1**	**Q2**
Inlet	Discharge	x	x	x	x	x	x	x	x	x	x
	Conductivity	x	x	x	x	x	x	x	x	x	x
	Temperature	x	x	x	x	x	x	x	x	x	x
	Phosphorus					(x)	(x)	x	x	x	x
Outlet	Discharge	(x)	(x)	(x)	(x)	x	x	x	x	x	x
	Conductivity	x	x	x	x	x	x	x	x	x	x
	Temperature	x	x	x	x	x	x	x	x	x	x
	Phosphorus					x	x	x	x	x	x
Pond Center	Water Level	x	x	x	x	x	x	x	x	x	x
	Profiles	x	x	x	x	x	x	x	x	x	x
	Phosphorus				(x)	x	x	x	x	x	x

4.4 Description of Model Calibration

In order to verify the model, it was necessary to first calibrate it by adjusting various unmeasured parameters to replicate observed conditions. Once the model output matched observed measurements, a different set of input conditions for a distinct monitoring period was run through the model without additional calibration. This second set of model outputs was again compared against observed measurements as a verification step.

The model calibration process was three-fold. First, known environmental condition values were entered into the model. These included pond bathymetry, incoming flow rates, incoming concentrations of water quality constituents (e.g., chloride and phosphorus), meteorological data, sediment phosphorus release rates, and initial conditions based on the earliest available field observations for the calibration monitoring period. Second, a coarse adjustment of model parameters was undertaken by making qualitative comparisons of model output to field observation measurements. And third, finer adjustments were made to ambiguous model parameters by conducting a sensitivity analysis across a reasonable range of values for each parameter. In each iteration, the model output was quantitatively compared to the field observation measurements on the basis of a root mean square error (RMSE) calculation (Equation 1) for each measurement profile (Figure 11). Simulation parameters were then adjusted to minimize the RMSE disagreement between measured and predicted values.

$$RMSE = \sqrt{\frac{\sum(prediction - measurement)^2}{number\ of\ paired\ predictions\ and\ measurements}} \quad \textbf{(Eq. 1)}$$

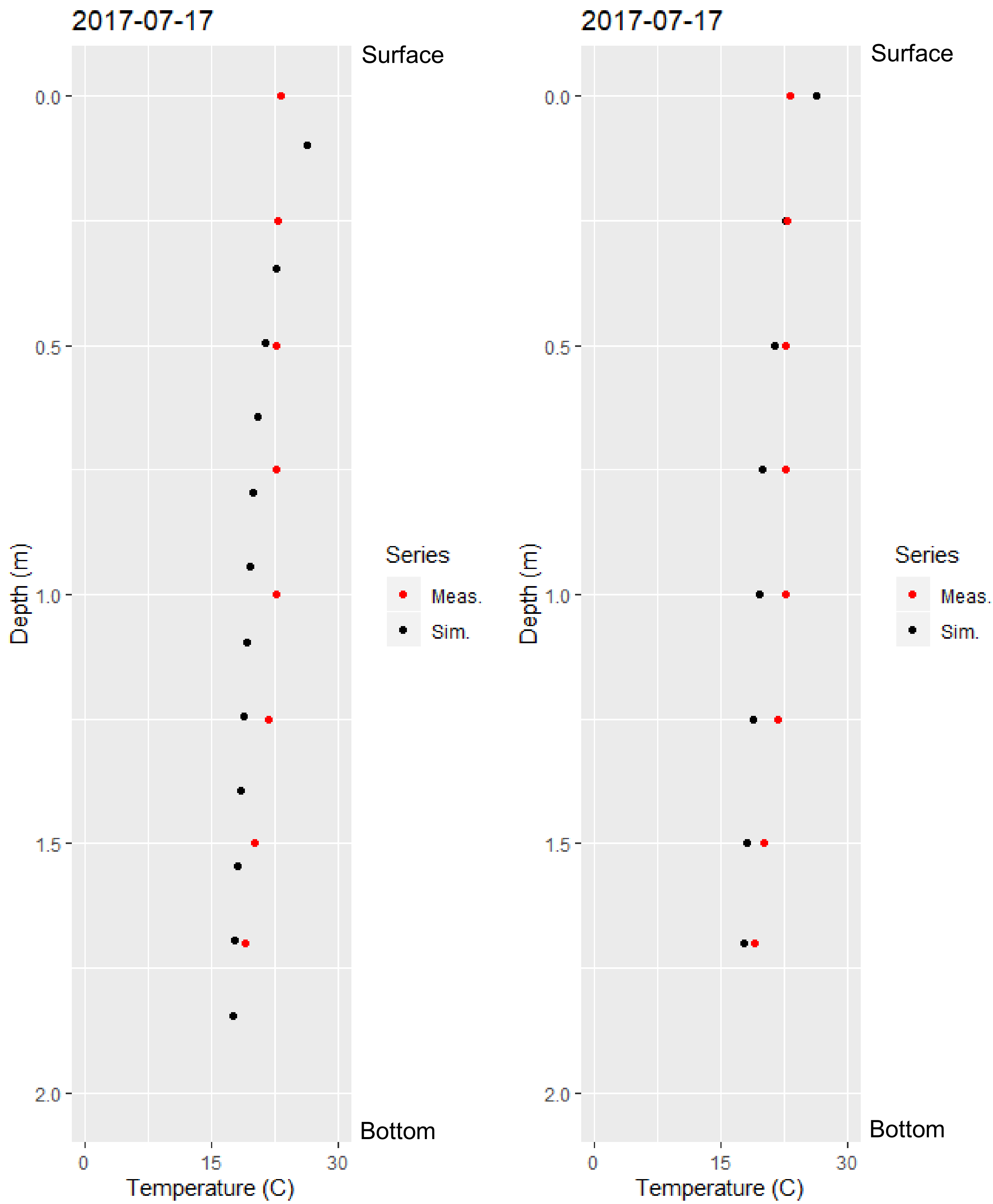

Figure 11. *Illustration of process for taking simulation outputs (left) and interpolating values for specific elevations corresponding to measured data points (right) to derive paired values for RMSE calculations.*

4.5 Model Calibration

This section describes changes to fundamental model parameters used to achieve a good fit of the field observations required to accurately model a small, shallow stormwater pond using CE-QUAL-W2.

4.5.1 Vertical Grid Resolution

A sensitivity analysis undertaken during the model creation phase (described in Section 3) determined that a very fine vertical grid resolution was required to accurately capture the strength of the water column stratification due to the differences in density across the observed strong gradients of chloride concentrations. It was found that a vertical grid spacing of 15 centimeters was the optimal size for the greatest improvements to the model performance without requiring so much computation time as to make the model impractical for iterative analyses. The vertical grid spacing was set at the smallest value that would result in super computer run periods of less than one day.

4.5.2 Wind Sheltering

Previous work (Herb et al. 2017) found that a drastic reduction in wind mixing was required for a CE-QUAL-W2 pond model to replicate the strong and lasting water column stratification that has been observed in the target pond and various others (Taguchi et al. 2018a). This model had used the same static wind sheltering coefficients (WSC) employed by Herb et al. of 0.01 (1% of wind energy reaching the water surface) for each model cross section. Figure 12 provides an example of these calibrations to chloride concentration.

In parallel with the work on the CE-QUAL pond model, we also undertook research on modeling wind sheltering of ponds. The goal of this research was to enable estimation of wind sheltering of ponds and lakes without field monitoring data. A commercial computational fluid dynamics (CFD) package (ANSYS-Fluent) was used to simulate wind flow from land surfaces (with vegetation and topography) on to ponds. The main outputs of the CFD models were the wind velocity distribution over the pond and the shear stress distribution on the pond surface, which is a surrogate for wind mixing energy. For example, a decrease of shear stress on a lake surface indicates more wind sheltering, less wind mixing, and more stratification. A number of simulations were run using two- and three-dimensional ANSYS-Fluent models to examine the effects of tree height and density and the pond bank height on wind sheltering. An example two-dimensional simulation output is given in Figure 13, showing the wind velocity and shear stress distributions as the wind transitions from a land surface to a water surface, with a line of trees and an embankment at the shoreline. A series of model runs simulated a range of tree and embankment heights, tree densities, single tree lines versus continuous forests, and the distance of the trees from the water edge. The results are compiled into design curves that show the importance of the different variables on the average shear stress on the water surface.

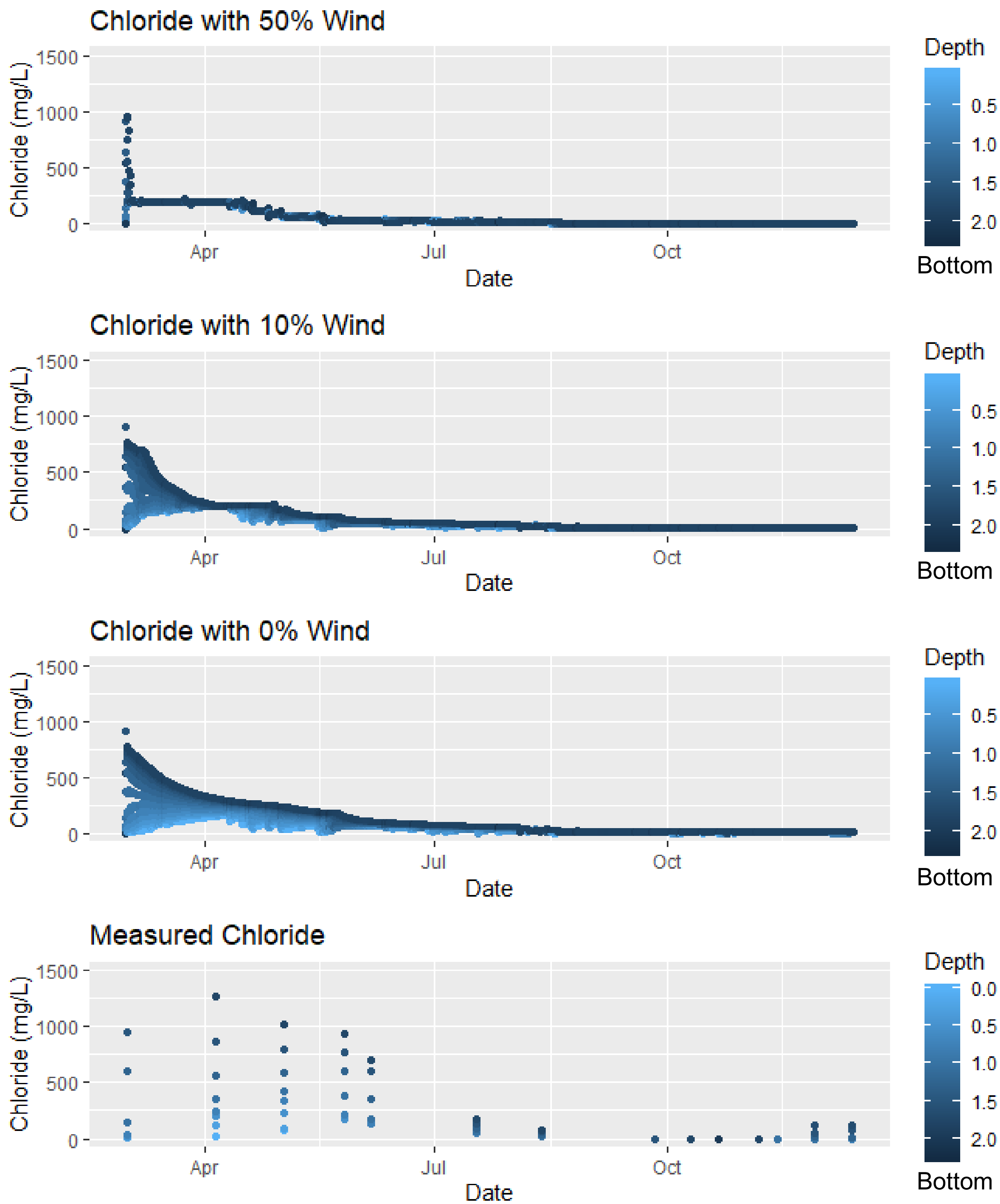

Figure 12. *Comparison of model chloride results with partial wind sheltering coefficient (WSC = 0.50) near-total wind sheltering (WSC = 0.10) and total wind sheltering (WSC = 0.00).*

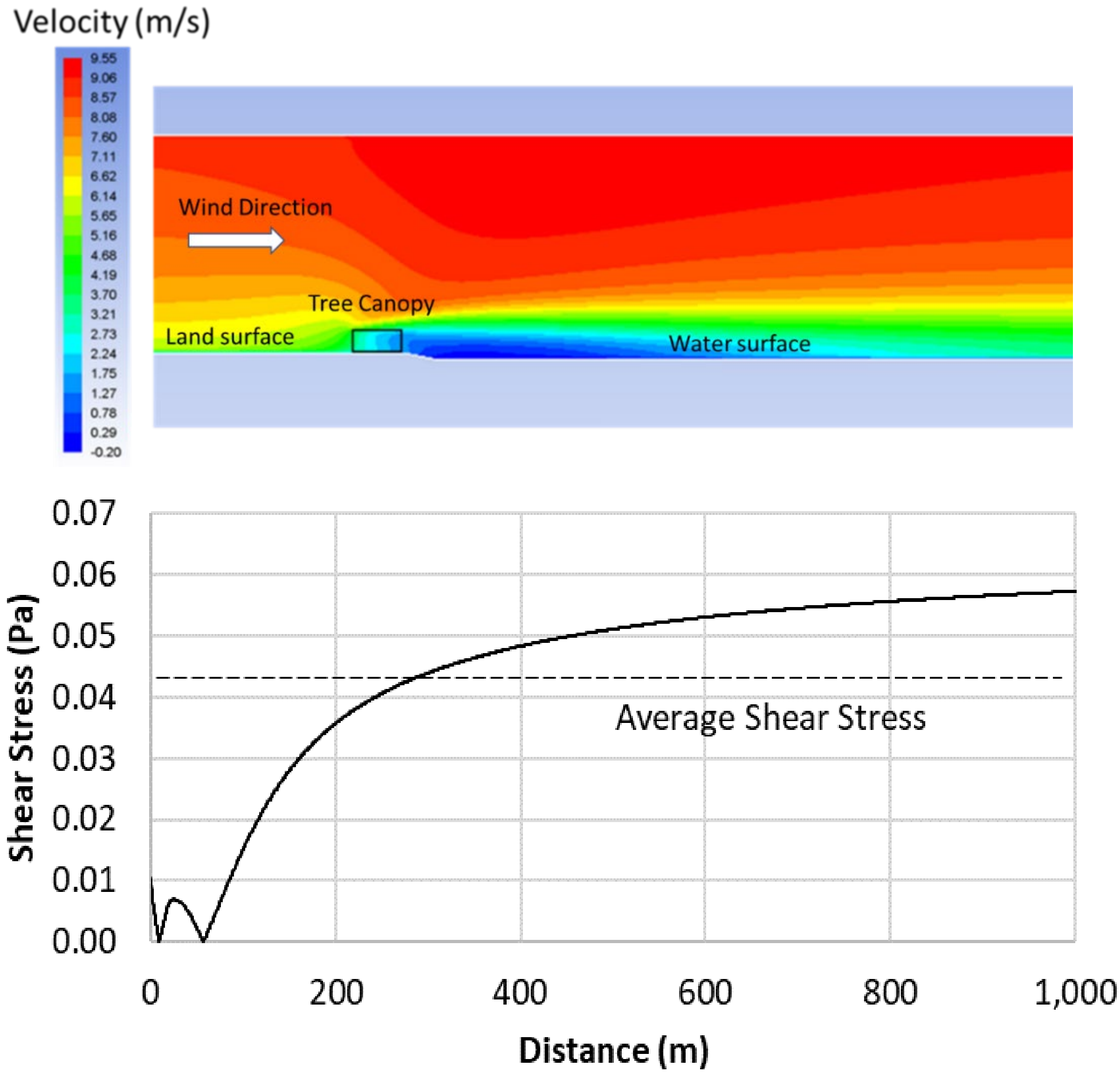

Figure 13. *Sample output from a Fluent model, showing the wind velocity distribution (upper panel) going from upwind (left), over a 20-m (66-ft) wide line of trees, and onto a pond or lake surface, and the corresponding shear stress distribution on the water surface (lower panel).*

4.5.3 Diffusion and Dispersion

In addition to a refined vertical grid resolution, it was found that the modeled chloride concentration gradients were rapidly becoming homogenized (i.e., weakened stratification with more similar chloride concentrations across the water column) in a way that did not resemble the field observation measurements (Figure 14). It was found that the model behavior, compared to measurements, could be improved by removing all diffusion and dispersion within the model. More specifically, the coefficients for longitudinal eddy viscosity (AX), longitudinal eddy diffusivity (DX), and maximum value for vertical eddy viscosity (AZMAX) were all set to zero. A few tangentially related coefficients were also set to zero such as the coefficient of bottom heat exchange (CBHE), interfacial friction factor (FI), and heat lost to sediments that is added back to water column (TSEDF). Numerical diffusion, caused by the discretization of the model, remains in the model, which represented diffusion and dispersion in the field applications.

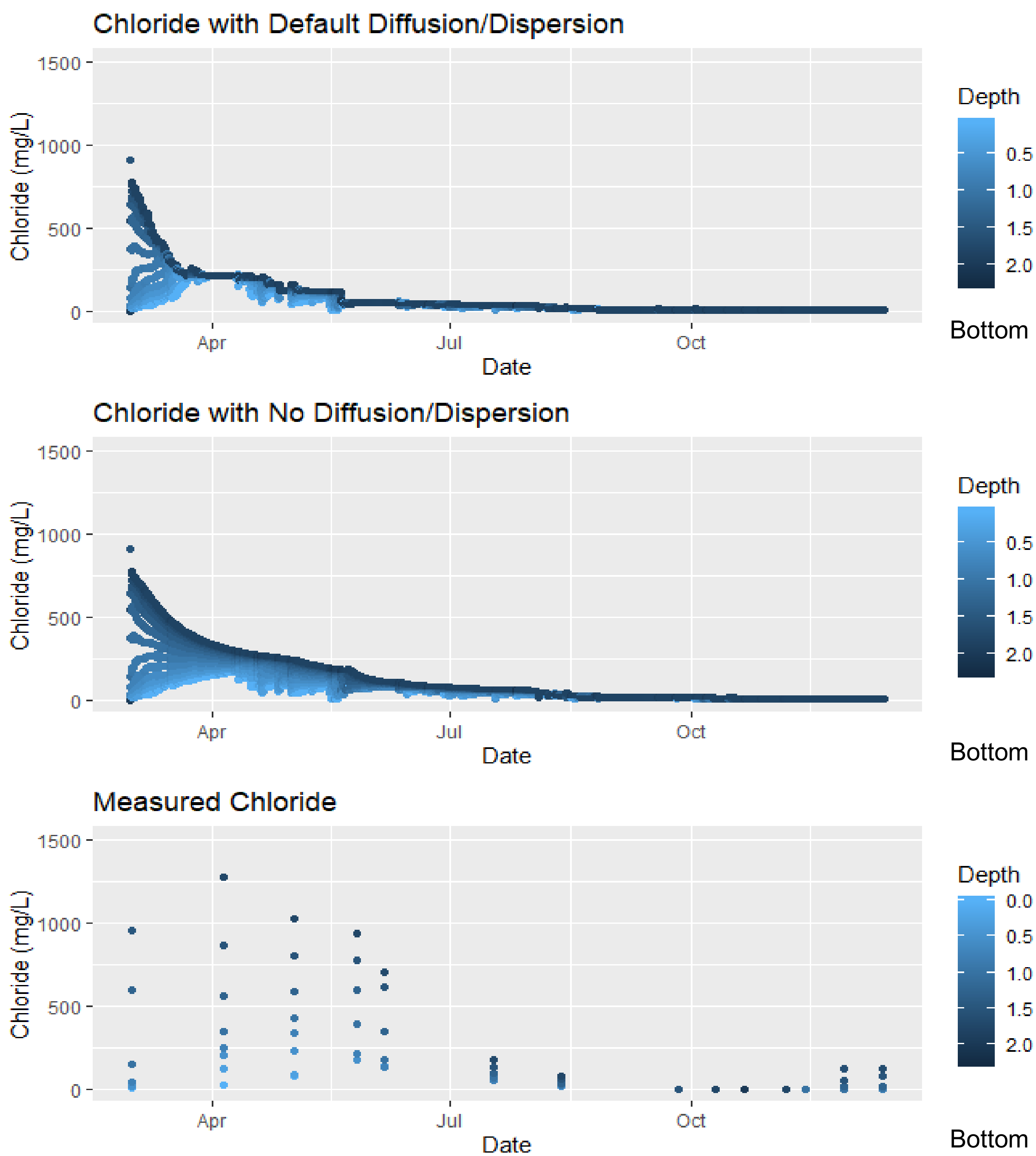

Figure 14. *Comparison of model chloride results with default diffusion/dispersion parameters and with no diffusion/dispersion.*

4.5.4 Particulate Settling

Despite the substantial reductions to vertical diffusion, the model output showed that phosphorus concentrations in the water column were decreasing more rapidly than what was observed in the field observations (Figure 15). It was determined that, since a large portion of the phosphorus entering stormwater ponds is in a particulate form, it was possible that the particles may have been settling too quickly using the default model parameters. A sensitivity analysis revealed that changing the particulate organic matter settling coefficient (POMS) from the default rate of 0.1 m/day (0.3 ft/day) to 0.0 m/day (0.0 ft/day) yielded improved results. We believe that this because particulate TP loading to stormwater ponds is largely organic (Taguchi et al. 2018b), and organic particulate matter settles much more slowly than inorganic particulate matter. Phytoplankton and floating plants, in fact, can have a negative settling velocity.

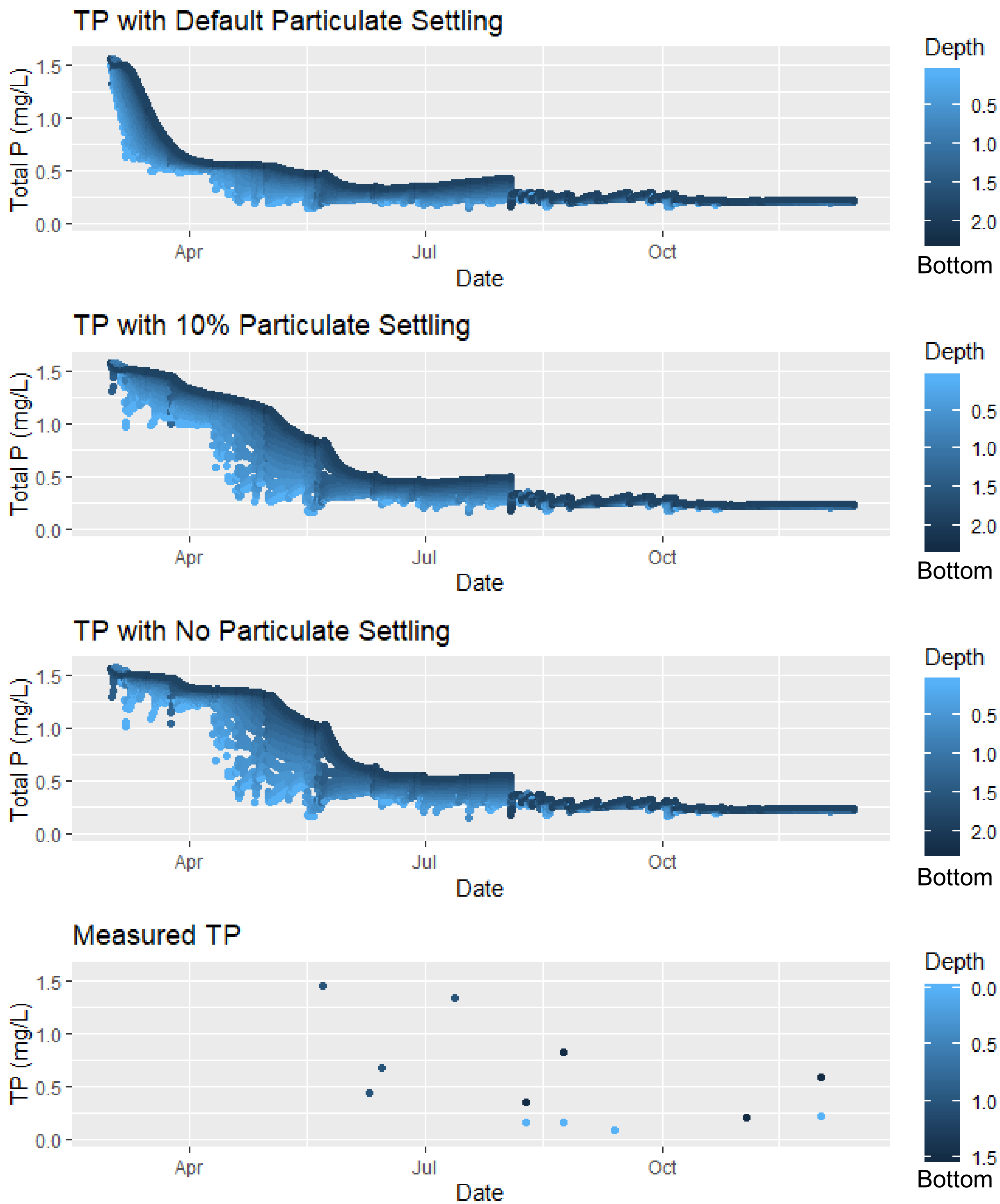

Figure 15. *Comparison of model total phosphorus (TP) results with default particulate settling (POMS = 0.1 m/day), 10% particulate setting (POMS = 0.01 m/day) and no particulate settling (POMS = 0.0 m/day).*

4.5.5 Light Extinction

It was known from prior field monitoring efforts (Figure 16) that many urban stormwater ponds in the study area are covered by free-floating macrophytes (FFM) including duckweed (Lemna spp.) and watermeal (Wolffia spp.). The effects of this seasonal vegetative cover on stormwater ponds are not well understood, although recent research efforts (Janke et al. 2022) are finding that the presence of FFM in ponds was found to coincide with low dissolved oxygen in the water column, heavy tree canopy cover, and elevated levels of dissolved phosphorus.

Figure 16. *Water quality sampling of a Minnesota stormwater pond with free-floating macrophyte cover. Photo credit: Tasha Spencer (2020).*

An attempt to emulate light limitation due to FFM cover, which proved successful, used a dynamic light extinction coefficient (EXH2O), which did not require any source code modification. EXH2O describes the fraction of light that is lost per meter of depth, meaning it allows light energy dissipation to be specified higher or lower in the water column according to FFM presence. The specific values used (1.45 m^{-1} (0.442 ft^{-1}) during FFM presence and 0.85 m^{-1} (0.259 ft^{-1}) during FFM absence) were determined through a sensitivity analysis. Using this strategy, stronger or weaker EXH2O values were defined at specific dates corresponding to the presence or absence of FFM cover (Figure 17). Using a static EXH2O equal to the average of the two values (1.15 m^{-1}; 0.350 ft^{-1}) was a reasonable approximation for most of the simulated monitoring season, but the dynamic EXH2O values (0.85 m^{-1} (0.259 ft^{-1}) and 1.45 m^{-1} (0.442 ft^{-1})) were still a slightly better for most comparison dates (based on RMSE).

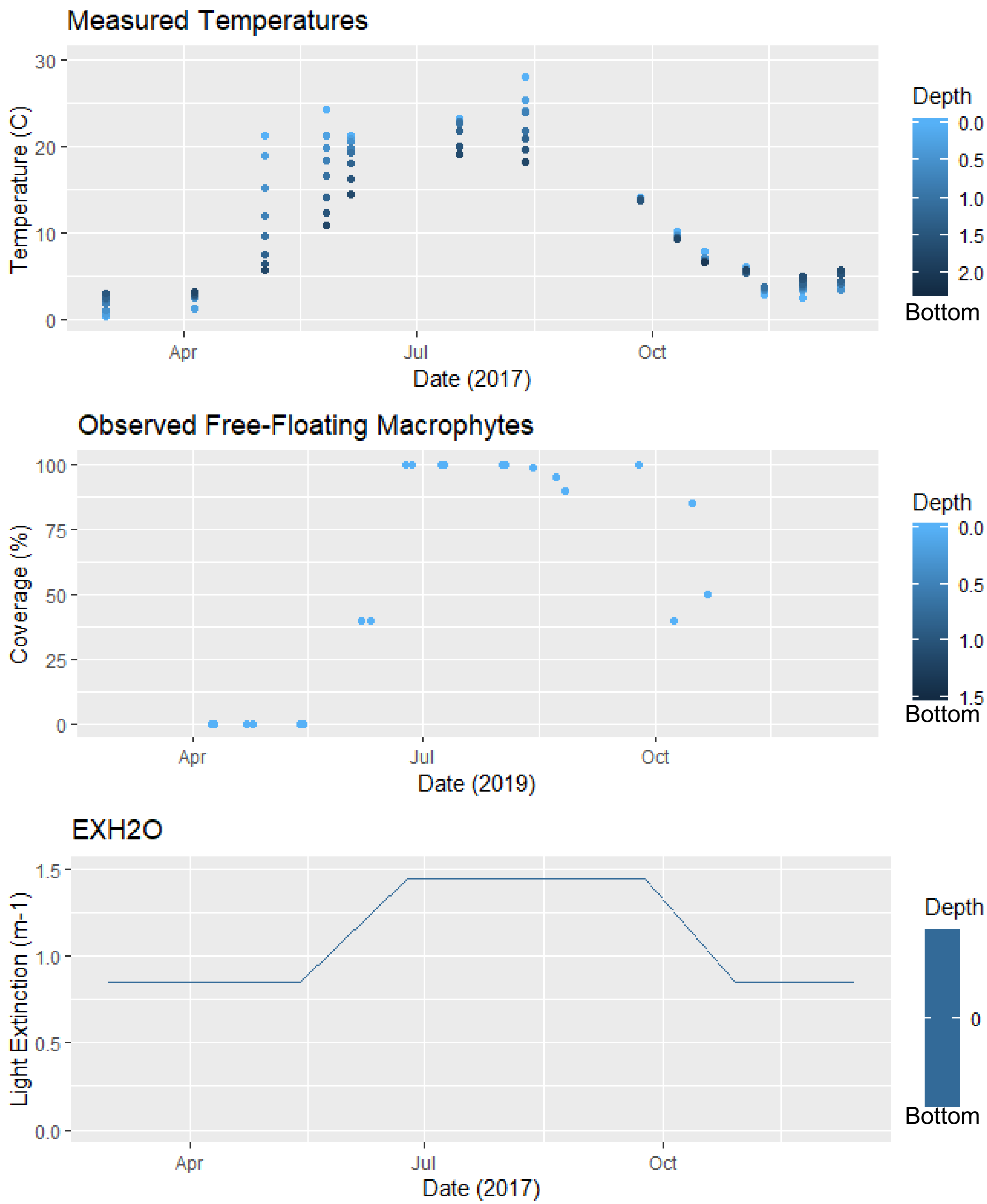

Figure 17. *A comparison of measured temperatures, observed free-floating macrophyte cover, and assigned EXH2O values for simulation.*

With these adjustments, simulated temperature profiles approximate both the magnitude of measured temperatures and the presence or absence of thermal stratification (Figures 18a and 18b). Exact values differ somewhat, but this was expected because temperatures vary throughout the day and measured profiles may not be perfectly temporally matched with simulated profiles. The model calibration for the remaining parameters is discussed in Appendix A, and the verification of the model performance is discussed in Appendix B.

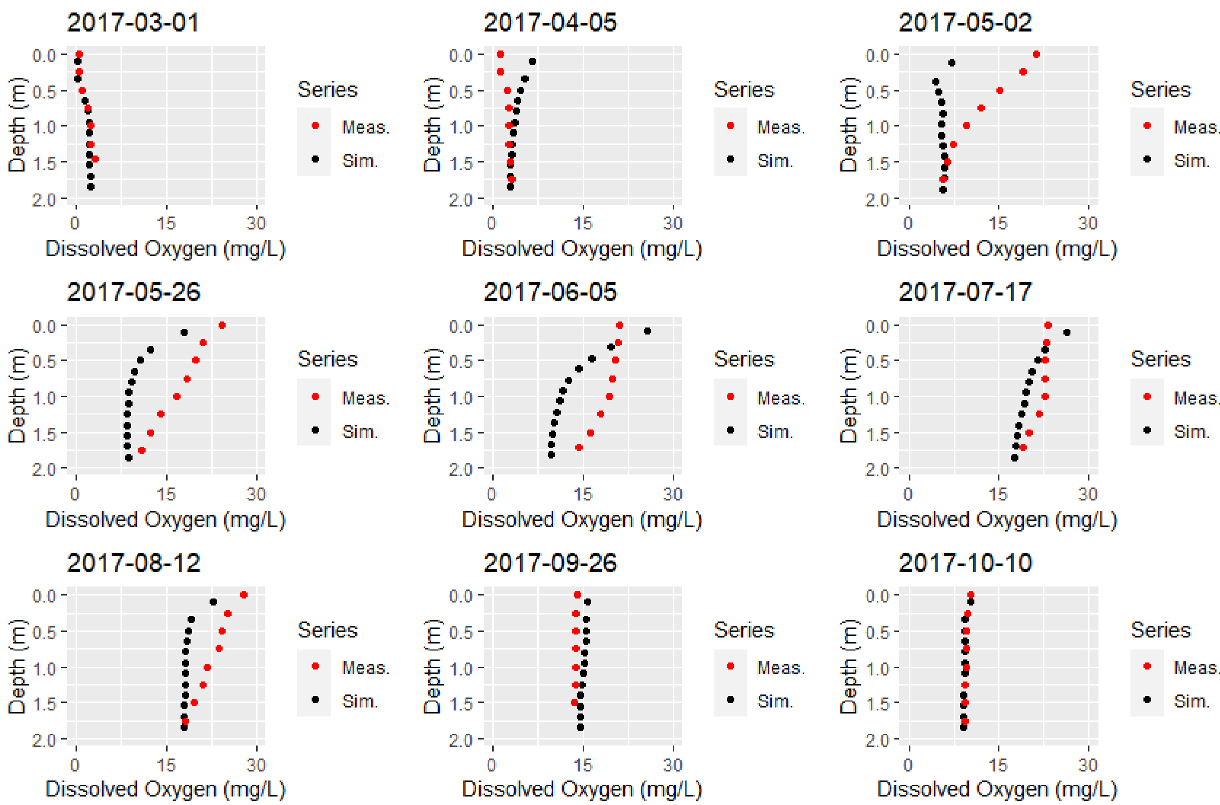

Figure 18a. *Comparison of measured temperature profiles to corresponding temperature profiles from model output. Part 1 of 2.*

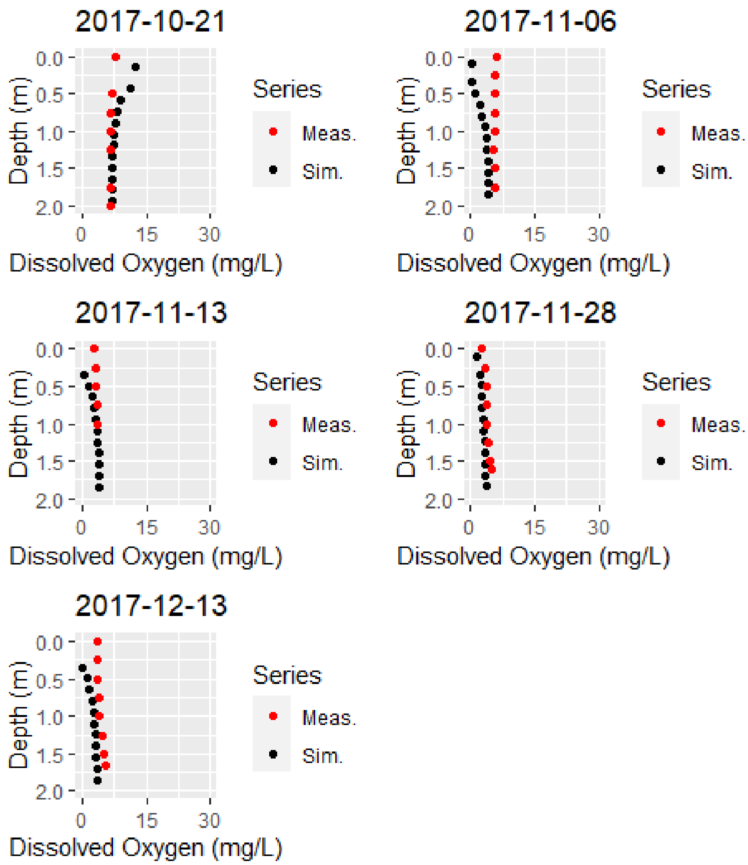

Figure 18b. *Comparison of measured temperature profiles to corresponding temperature profiles from model output. Part 2 of 2.*

5 Pond Phosphorus Remediation Results

A pond-by-pond modeling effort would be highly intensive with respect to data inputs and model creation labor and would be unlikely to be undertaken by pond management practitioners outside of the academic research setting. We, therefore, sought a more generalizable approach that would result in observations that could be more easily applied with data that local management groups would be likely to have. Often, pond management practitioners do not have access to all of the required data for all of the ponds they wish to model (e.g., robust and continuous inflow measurements including volumes and water quality constituent concentrations, frequent measurements of in-pond water quality constituent concentrations, continuous measurements of wind velocities over the pond surface, etc.). It is common engineering practice, however, to apply existing models to new scenarios by making reasonable engineering assumptions.

In order to generate the most useful recommendations for urban stormwater pond design, maintenance, and remediation through this modeling effort, we generalized the pond model and simplified the permutations thereof for each pond and remediation strategy. This generalization involved modeling each pond over the same warm-season timespan with identical initial conditions, meteorological inputs, and inflow concentrations. By controlling for as many variables as possible between simulations, we are able to compare and contrast each pond to each remediation scenario as well as comparing the different remediation strategies more broadly in our discussion of general recommendations.

In this Section, we set up each of the four modeled ponds based off of a verified model of the Shoreview pond because this pond exhibits features common in many constructed stormwater ponds (e.g., inlet and outlet located on opposite ends). Furthermore, the weir-controlled riser structure at the outlet is connected to the main pool of the pond by a submerged outlet pipe. This design allows the outlet pipe elevation to be modified as a remediation strategy in our modeling efforts.

The Shoreview pond model simulation occurs over the timespan from June 6, 2019 to September 12, 2019 for which we have warm-season water quality monitoring data gathered as part of this study as well as inflow and outflow data provided by the Ramsey-Washington Metro Watershed District. We used the same warm-season, data-rich period as the simulation period for all of our pond models. We kept many of the fundamental parameters involved in creating a stormwater pond model in CE-QUAL-W2 (e.g., vertical grid resolution) consistent across ponds as well as parameters that would have required significant effort to calibrate (e.g., light extinction coefficients and accommodations for free-floating macrophytes (FFM)).

The following additional model parameters were made consistent across ponds:

- Inlet structure elevations relative to normal water surface level
- Outlet structure elevations (except when modified as a treatment scenario) relative to normal water surface level
- Inlet and outlet structure dimensions
- Inflow concentrations (except when modified as a treatment scenario)

- Meteorological inputs
- Initial in-pond constituent concentrations

The following model parameters were unique to each pond:

- Inflow volumes (scaled by watershed impervious area)
- Bathymetry
- Pond orientation relative to north
- Sediment oxygen demand
- Sediment phosphorus release

5.1 Scenario Evaluation Summary

Each scenario was defined according to the parameter changes we developed for each model, based on realistic expectations and observed environmental variability. We defined these changes in a consistent manner so that each pond may be directly compared. We then ran the model with each set of parameters. When necessary, we made minor modifications to parameters that define computation speed, such as the maximum timestep, to preserve model stability under new conditions. We post-processed the model output to calculate performance metrics for each remediation strategy indicative of phosphorus, dissolved oxygen, and stratification conditions. These metrics are summarized for each test scenario and compared against the baseline condition of each pond model on the basis of percent improvement and estimated cost. We examined eight performance metrics:

- **SRP Release:** Model-estimated cumulative sediment release of Soluble Reactive Phosphorus mass over the simulation period (kg). *Why Important:* Management goal. Supplies phosphate to algae, and impacts water column TP concentration, which is the sum of particulate and dissolved phosphorus.
- **DO Consumption:** Model-estimated cumulative sediment consumption of dissolved oxygen mass over the simulation period (kg). *Why Important:* Impacts SRP release from sediment due to greater release under reduced oxygen conditions.
- **Mean and Median TP:** Mean and median surface layer and water column-averaged TP concentration over the simulation period (mg/L). *Why Important*: Management goal.
- **Mean and Median RTRM:** Normalized mean and median difference in density between the top and bottom of the water column expressed as Relative Thermal Resistance to Mixing to communicate strength of stratification over the simulation period (unitless). *Why Important:* Impacts oxygen dynamics by isolating the benthic layer from the oxygen in the atmosphere.
- **Mean and Median DO:** Mean and median benthic and water column-averaged Dissolved Oxygen concentration over the simulation period (mg/L). *Why Important:* Low DO can lead to SRP release from sediments.
- **Days No Oxia:** Number of days without any discrete benthic DO concentrations above 2 mg/L (anoxia threshold) over the simulation period (days). *Why Important:* Higher DO can minimize SRP release from sediments.

- **Days Any Anoxia:** Number of days with any discrete benthic DO concentrations below 2 mg/L (anoxia threshold) over the simulation period (days). *Why Important:* Low DO can lead to SRP release from sediments.
- **TP Export:** Model-estimated cumulative export of Total Phosphorus mass over the simulation period (kg). *Why Important:* Management goal, determines phosphorus retention functionality of pond, impacts downstream water bodies.

Results based on some of these metrics (Mean and Median TP, Days No Oxia, Days Any Anoxia, TP Export) are tabulated and discussed within the body of the report. Others (SRP Release, DO Consumption, Mean and Median TP, Mean and Median RTRM, Mean and Median DO, Days No Oxia, Days Any Anoxia) are listed in Appendices C through G.

5.2 Modeled Ponds

We selected four ponds, monitored as part of this project and for surface water quality and sediment phosphorus release in previous work (Janke et al. 2022; Taguchi et al. 2020b), for the model application task. These ponds included: (1) Alameda pond and (2) Langton Lake Upstream Pond, both located in Roseville, (3) Shoreview Commons Pond, located in Shoreview, and (4) 849w Pond, located in Minnetonka and hereafter referred to as the "Minnetonka pond." The four ponds encompassed a range in surface area (~650 – 12,000 m^2 or ~0.16 – 2.97 ac) and volume (~460 – 14,500 m^3; or ~121,500 – 3,830,500 gal), as well as age (the Langton pond was constructed in 2017 while the Alameda, Shoreview, and Minnetonka ponds are former wetlands converted to stormwater ponds 32 – 60+ years ago). The ponds also varied slightly in land use context, with Alameda and Minnetonka in single-family residential neighborhoods, Shoreview in a residential area with an adjacent open (turf) park and a recreation center parking lot, and Langton in a more commercial / industrial land use near Langton Lake (see aerial photos in Figure 19). Alameda, Shoreview, and Minnetonka ponds were all heavily sheltered by mature tree cover on shore and in the vicinity (8.4 – 12.9 m average tree height within a 50-m buffer or 28 – 42 ft height with 160-ft buffer); Alameda also has substantial topographic sheltering due to being located in a depression (mean embankment height = 3.9 m or 12.8 ft). Langton, being a new pond, has no on-shore tree cover but is located near mature (but sparse) tree cover and warehouse-type buildings. Pond characteristics relevant to the modeling study are summarized in Table 4.

Setup of the CE-QUAL-W2 model for the ponds also required bathymetric information in order to define the 2-D computational grid for each pond (or, for some analyses, the dimensions to which to scale the existing model). For the Shoreview, Langton, and Minnetonka ponds, bathymetric maps were developed from through-the-ice depth surveys in late winter of 2018, while a map from a contracted survey was provided for the Alameda pond by the city of Roseville. Pond bathymetry for the four ponds is shown in Figure 20.

We collected supporting data at each site, including continuous measurements of wind speed, water level, and water column temperature profiles at roughly 25-cm (9.8-in) intervals using mid-pond instrumentation stations. The 2020 supporting data were described in the Section 4, where the data were used to help set up and calibrate the CE-QUAL-W2 model for the Alameda pond.

The same monitoring was carried out on the Alameda and Shoreview ponds in 2021 as part of a related project; wind speed data from both 2020 and 2021 were utilized in the wind sheltering reduction scenario analysis in this project. To use these wind data, wind speeds collected on the ponds using LaCrosse TX-23U anemometers were calibrated to more robust instrumentation (RM Young anemometer) at the St. Anthony Falls Laboratory using a side-by-side comparison.

Table 4. *Physical characteristics of the three ponds included in the modeling task. Mean shore canopy height and mean embankment height are estimated within a 50-m buffer of each pond using LIDAR data (https://www.dnr.state.mn.us/maps/mntopo/index.html); embankment height is the ground elevation height above mean water level. For the Langton pond, LIDAR data are older than the pond so canopy estimate could not be competed, with embankment estimate from pond drawings.*

	Shoreview Pond	**Alameda Pond**	**Langton Pond**	**Minnetonka Pond**
Surface Area, m^2 (ac)	11,700 (2.9)	11,700 (2.9)	650 (0.2)	6,900 (1.7)
Mean Depth, m (ft)	0.61 (2.0)	1.39 (4.6)	0.75 (2.5)	0.74 (2.4)
Max Depth, m (ft)	1.80 (5.9)	2.10 (6.9)	1.75 (5.7)	2.55 (8.3)
Volume, m^3 (ac-ft)	7,100 (5.7)	14,500 (11.8)	460 (0.4)	4,700 (3.8)
Mean Shore Canopy Height, m (ft)	12.9 (42.3)	8.4 (27.5)	NA*	11.7 (38.4)
Mean Embankment Height, m (ft)	1.0 (3.3)	3.9 (12.8)	~2.0* (~6.6)	2.1 (6.9)
Drainage Area, m^2 (ac)	583,000 (144)	1,153,000 (285)	8,300 (2.1)	27,900 (6.9)
Watershed Loading Ratio**	50:1	98:1	11:1	4:1
Impervious Drainage Area m^2 (ac)	233,100 (58)	230,700 (57)	7,700 (1.9)	27,900 (6.9)***
Impervious Watershed Loading Ratio**	20:1	20:1	10:1	4:1
Drainage Area Land Use	Parking Lot, Residential, Wooded, Park	Residential, Wooded	Industrial, Street	Residential, Wooded
Age, years	32	~60	4	61

*The Langton pond was constructed too recently to appear on LiDAR data.
**Ratio of drainage area to pond surface area
***The actual estimate is 5,300 m^2 (1.3 ac), but this value is thought to be incorrect based on model performance. Model results were more reasonable when the impervious drainage area was estimated as the entire 27,900- m^2 (6.9-ac) drainage area, which could mean the true drainage area is underestimated or that there are unknown groundwater contributions.

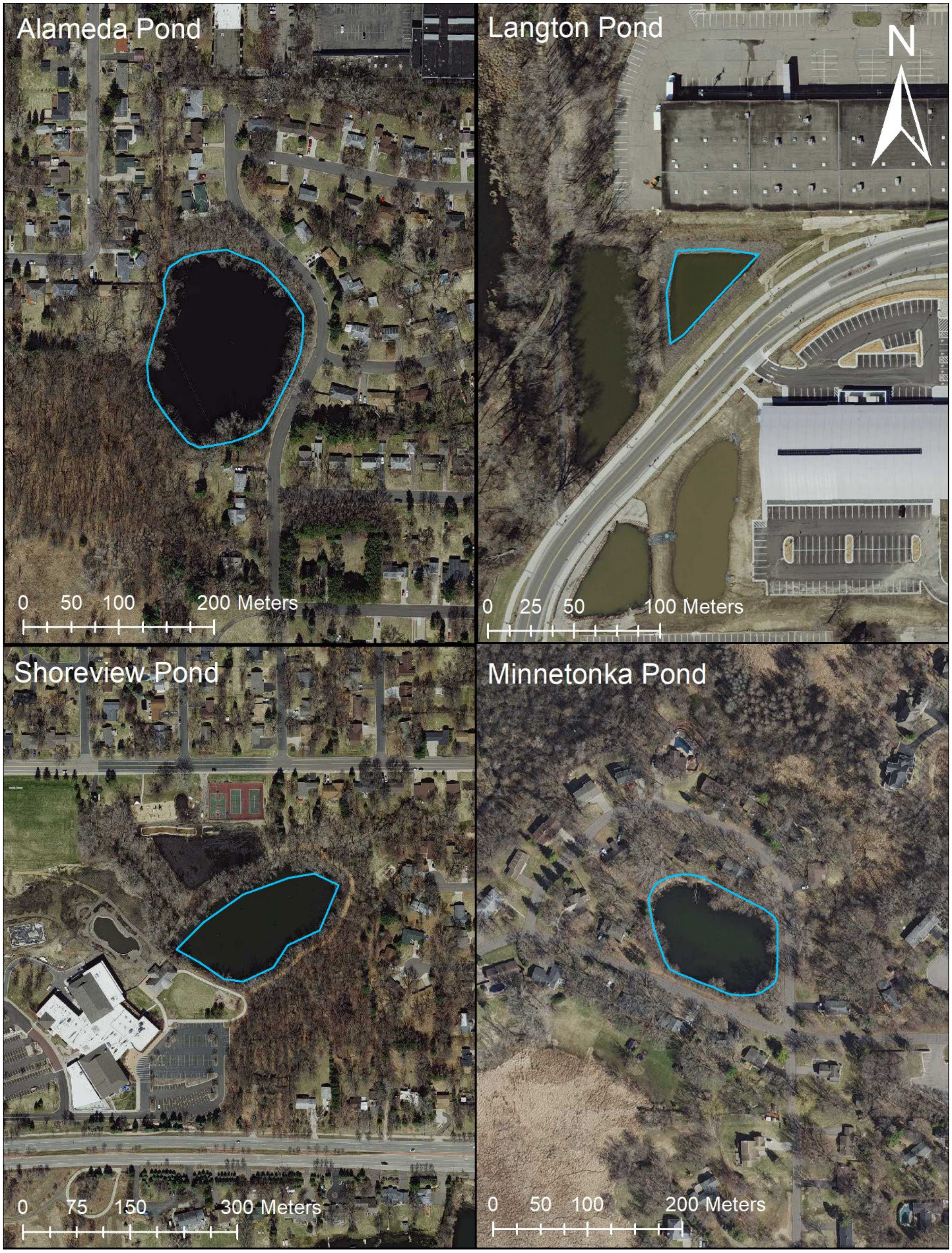

Figure 19. *Aerial views of the four monitored ponds. Imagery from 2020, downloaded from MnGeo (https://www.mngeo.state.mn.us/chouse/wms/geo_image_server.html).*

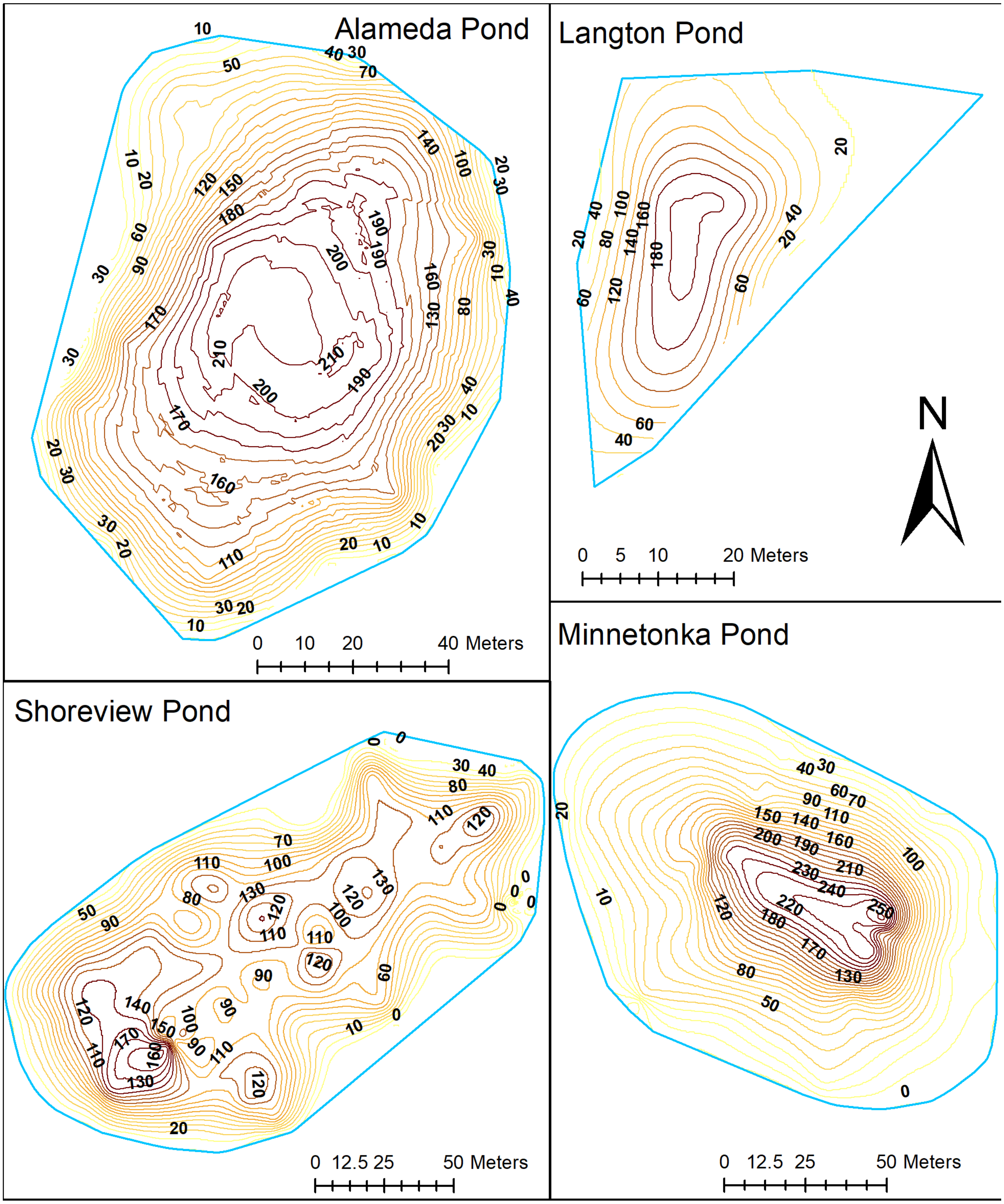

Figure 20. *Bathymetric maps of the four ponds, showing depth contours in centimeters.*

5.3 Chemical Treatment of Sediments

We considered chemical treatments of pond sediments because they will bind phosphorus in the sediments and thus reduce the sediment soluble reactive phosphorus (SRP) release rates. The two assessed treatment types included alum and iron filings, and for both treatment types, we estimated both effectiveness (dosage) and cost of applications.

5.3.1 Alum Treatment

As provided in Section II, we used water quality data from four alum-treated lakes in the Riley Purgatory Bluff Creek Watershed District (RPBCWD), MN to estimate an average change in hypolimnetic total phosphorus (TP) concentrations in response to alum. The influence of alum was calculated by comparing the pre- and post- treatment concentrations (Data provided by RPBCWD; data shown in Tables H-1). The average reduction in TP concentration (available data before treatment vs. available data after treatment) was 77% (71% minimum; 86% maximum). For the purposes of modeling the chemical treatment of sediments scenario, the measured sediment phosphorus release rate coefficient was reduced by 77% for the medium application scenario. We also considered an additional No Sediment Release scenario to provide the theoretical limits of treatment effectiveness (no cost estimate is given for this scenario). Sediment phosphorus release rates were measured using the method of Taguchi et al. (2018a).

We calculated alum dosages to capture and immobilize both the redox-sensitive phosphorus (redox-P) and labile organic phosphorus (labile-P), together referred to as mobile phosphorus (mobile-P), in the sediments (Reitzel et al. 2005) based on sediment data for each pond. Each alum dose was considered based on the mobile-P concentration and the total bulk density in the top 4 cm (1.6 in) of sediment for each pond (Rydin and Welch 1998; sediment data for the modeled ponds are provided in Table H-2). Some of the ponds have dense sediments (from 12% up to 52% solids content), which means that a high alum dose is required for the high area-based mobile-P content and that there is a possibility of alum floc not mixing into the sediments (James 2011). There is also evidence indicating that the alum floc loses its phosphorus-binding capacity as it ages, meaning that alum applications will only be effective for a limited period of time. Many practitioners, therefore, recommend using a lower alum dose (i.e., a dose based on a low ratio of alum (Al) to sediment P; de Vicente et al. 2008) that can be reapplied more frequently. We are operating on the recommendation to apply a lower alum dose of roughly 20:1 Al:P and repeat the dosing as necessary. These values were then applied only to the portion of the pond sediments likely to experience anoxic conditions (the deeper portions or the pond) rather than the entire sediment surface area.

Cost estimates are based on values from alum applications in lakes in the City of Eagan, MN (data provided by the City of Eagan). Material cost estimates are \$0.55/liter (\$2.10/gal) for alum and \$1.48/liter (\$5.60/gal) for sodium aluminate buffer solution (one part buffer for every two parts alum) when aluminum toxicity due to pH change is of concern. For the ponds in this study, no buffer is included in the cost estimates, but a 10% contingency has been factored in. A major variable in chemical treatments for ponds are mobilization costs. Based on pond and lake treatments in Eagan, the mean multiplier for material costs to arrive at the total costs was 2.6 (1.4 minimum; 4.6 maximum). For the purpose of the cost estimates in this study, we used a multiplier value of 3. Estimating the frequency of reapplication is difficult because of the lack of long-term data on stormwater pond applications as well as the high site-to-site variability. Based on extensive data from lakes (Huser et al. 2016), we are assuming a 10-year lifespan. The scenarios, adjusted SRP flux values, and corresponding alum dose amounts and cost estimates are described in Table 5.

Table 5. *Alum doses calculated for each remediation scenario. Total Costs include all anticipated costs over a 10-year period.*

Shoreview Pond				
Scenario	**SRP Flux** mg/m^2/day	**Alum Dose** liters (gal)	**Material Cost***	**Total Cost***
Original	3.18	-	-	-
Alum Application	0.73	5,580 (1,474)	$3,400	$10,000
No Sediment Release	0.00	-	-	-
Alameda Pond				
Scenario	**SRP Flux** mg/m^2/day	**Alum Dose** liters (gal)	**Material Cost***	**Total Cost***
Original	7.52	-	-	-
Alum Application	1.73	8,680 (2,293)	$5,300	$16,000
No Sediment Release	0.00	-	-	-
Langton Pond				
Scenario	**SRP Flux** mg/m^2/day	**Alum Dose** liters (gal)	**Material Cost***	**Total Cost***
Original	0.63	-	-	-
Alum	0.14	538 (142)	$330	**
No Sediment Release	0.00	-	-	-
Minnetonka Pond				
Scenario	**SRP Flux** mg/m^2/day	**Alum Dose** liters (gal)	**Material Cost***	**Total Cost***
Original	5.62	-	-	-
Alum Application	1.29	2,800 (751)	$1700	$5,200
No Sediment Release	0.00	-	-	-

*Costs rounded to two significant figures.

**No cost data available to accurately estimate total costs for small ponds.

5.3.2 Iron Filings Treatment

We considered iron filings as an alternative to alum for chemical treatment of pond sediments. The concept is to overload the sediments with elemental iron so that sufficient iron of various charges is present to restrict release even under low DO conditions (Natarajan, et al. 2021). The iron filings applications that have been implemented so far (Section 2) allow us to consider one to three years of pre-treatment data and one year of post-treatment data. This is a short period to calculate average performance rates. For this study, we assume that the results from a recent iron filings application in the Shoreview pond are most applicable, with a 55% reduction in SRP release rates. The theoretical No Sediment Release scenario with no sediment phosphorus release is the same as in the case of alum. Differences in longevity between alum and iron filings applications are likely but as of yet unknown. For the purposes of cost estimates in this study, we are assuming that either treatment would last for 10 years before reapplication became necessary based on extensive data from alum treatments in lakes (Huser et al. 2016).

The Shoreview pond application of iron filings was dosed at a rate of 0.58 kg/m^2 (0.12 lb/ft^2), which is somewhat greater than the dosage of 0.50 kg/m^2 (0.10 lb/ft^2) derived from laboratory study results (Natarajan et al. 2017). The cost estimates assume a material cost of $0.93/kg ($0.42/lb) iron and an approximate shipping cost of $558 from the vendor to Minneapolis (Section 2). Unlike with alum, no specialized crew is required to mobilize to the pond site and deploy specialized equipment. Municipal staff and lawn-care equipment are sufficient to evenly disperse iron filings across a frozen pond surface, or municipal staff and a small watercraft in the case of a warm-weather application. We estimated the cost of municipal staff time and the use of the necessary equipment at $3,000, although the true amount could be greater if additional or more specialized staff or equipment are found to be necessary. If the application were contracted to an outside company, the total cost multiplier would likely be on the order of 3, similar to that of the alum material costs (Table 6).

Table 6. *Iron filings doses calculated for each remediation scenario. Total Costs include all anticipated costs over a 10-year period.*

Shoreview Pond				
Scenario	**SRP Flux** mg/m^2/day	**Iron Filings Dose** kg (lb)	**Material Cost***	**Total Cost***
Original	3.18	-	-	-
Iron Filings Application	1.43	6,800 (15,000)	$6,300	$9,900
No Sediment Release	0.00	-	-	-
Alameda Pond				
Scenario	**SRP Flux** mg/m^2/day	**Iron Filings Dose** kg (lb)	**Material Cost***	**Total Cost***
Original	7.52	-	-	-
Iron Filings Application	3.38	6,800 (15,000)	$6,300	$9,900
No Sediment Release	0.00	-	-	-
Langton Pond				
Scenario	**SRP Flux** mg/m^2/day	**Iron Filings Dose** kg (lb)	**Material Cost***	**Total Cost***
Original	0.63	-	-	-
Iron Filings Application	0.28	380 (830)	$350	$3,900
No Sediment Release	0.00	-	-	-
Minnetonka Pond				
Scenario	**SRP Flux** mg/m^2/day	**Iron Filings Dose** kg (lb)	**Material Cost***	**Total Cost***
Original	5.62	-	-	-
Iron Filings Application	2.53	4,000 (8,800)	$3,700	$7,300
No Sediment Release	0.00	-	-	-

*Costs rounded to two significant figures.

5.3.3 Results of Chemical Treatment of Sediments

Surface TP concentrations in the Shoreview, Alameda, and Minnetonka ponds responded as expected to reduced sediment SRP release rates from alum and iron filing treatments, with greater surface TP reductions resulting from higher doses. The surface TP concentrations in the Langton pond, however, were not greatly affected by chemical treatment (Table 7) because this is a younger pond with little accumulated sediments and a very low sediment phosphorus release rate (Table 2).

Table 7. *Surface total phosphorus (TP) concentrations for chemical treatment of sediments.*

Shoreview Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.350 ± 0.010	-	0.360	-
Alum Application	0.203 ± 0.004	-42%	0.207	-43%
Iron Filings Application	0.245 ± 0.005	-30%	0.251	-30%
No Sediment Release	0.160 ± 0.002	-54%	0.162	-55%
Alameda Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.223 ± 0.004	-	0.228	-
Alum Application	0.171 ± 0.002	-23%	0.174	-24%
Iron Filings Application	0.187 ± 0.003	-16%	0.189	-17%
No Sediment Release	0.156 ± 0.001	-30%	0.158	-31%
Langton Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.173 ± 0.002	-	0.177	-
Alum Application	0.170 ± 0.002	-2%	0.174	-2%
Iron Filings Application	0.171 ± 0.002	-1%	0.175	-1%
No Sediment Release	0.169 ± 0.002	-2%	0.172	-3%
Minnetonka Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.302 ± 0.009	-	0.311	-
Alum Application	0.197 ± 0.004	-35%	0.202	-35%
Iron Filings Application	0.227 ± 0.005	-25%	0.233	-25%
No Sediment Release	0.165 ± 0.002	-45%	0.168	-46%

Overall, reducing sediment anoxic phosphorus fluxes through chemical treatment of sediments appears to be an effective strategy to reduce phosphorus export, the primary goal of most pond management actions. TP is still exported under No Sediment Release scenarios due to the continued import of TP from watershed loading. Cost-effectiveness (both as cost per kg (lb) removed and per percent reduction in TP export) also varied across the ponds (Table 8) due to already low TP export values (Minnetonka and Langton ponds) being difficult to improve, with Shoreview being the most cost-effective for chemical treatment. Cost per % ∆ TP is a calculation of the total scenario cost divided by the percent improvement in TP export mass relative to the original condition; this is a representative metric of overall improvement independent or original values. Cost per kg (lb) TP is a calculation of the total scenario cost divided by the TP export mass reduction in number of kilograms (pounds); this is a common metric for water quality decision-making, but values are inflated by the low original TP export values. In the Langton pond, there is minimal benefit to treating the sediments because the phosphorus release rate is already so low compared to the other ponds.

Table 8. *Results for total phosphorus (TP) export of chemical treatment of sediments. TP Exports are calculated only for the approximately 3-month simulation period, while Total Costs include all anticipated costs for each remediation strategy over a 10-year period. Cost per % ∆ TP is a calculation of the total scenario cost divided by the percent improvement in TP export mass relative to the original condition. Cost per kg (lb) TP is a calculation of the total scenario cost divided by the TP export mass reduction in number of kg (lb).*

Shoreview Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	4.2 (9.4)	-	-	-
Alum Application	2.4 (5.3)	-43%	$240	$5,600 ($2,500)
Iron Filings Application	2.9 (6.5)	-31%	$320	$7,600 ($3,400)
No Sediment Release	1.9(4.1)	-56%	-	-
Alameda Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	2.9 (6.5)	-	-	-
Alum Application	2.2 (4.9)	-24%	$660	$23,000 ($10,000)
Iron Filings Application	2.4 (5.4)	-17%	$580	$20,000 ($9,000)
No Sediment Release	2.0 (4.4)	-31%	-	-
Langton Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	0.045 (0.099)	-	-	-
Alum Application	0.044 (0.097)	-2%	$530	$1,200,000 ($540,000)
Iron Filings Application	0.044 (0.098)	-1%	$3,000	$6,700,000 ($3,000,000)
No Sediment Release	0.044 (0.098)	-2%	-	-
Minnetonka Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	0.026 (0.058)	-	-	-
Alum Application	0.021 (0.047)	-19%	$270	$1,000,000 ($460,000)
Iron Filings Application	0.023 (0.050)	-14%	$530	$2,000,000 ($900,000)
No Sediment Release	0.020 (0.043)	-25%	-	-

*Costs rounded to two significant figures.

To place the phosphorus export of the remediation strategy on the same time frame as the costs, we observe that approximately 41% of the precipitation that fell between 1991 and 2020 occurred in the months June, July, and August according to data available from the US National Weather Service (https://www.weather.gov/wrh/climate?wfo=mpx). Roughly equating precipitation to export of phosphorus, then the approximately 3-month simulation TP export values represent 41% of the annual pond TP export. Table 9 provides 1-Year and 10-Year TP export values as rough approximations based on the assumption that cost per kg (lb) TP is a calculation of the 10-year scenario cost divided by the approximated 10-Year TP export mass reduction.

Table 9. *Evaluation of cost-effectiveness over a 10-year span for chemical treatment of sediments. 1-Year and 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 kilograms (pounds); this is a common metric for water quality decision-making, but values are inflated by the low original TP export values. All values are rounded to two significant figures.*

Shoreview Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	4.2 (9.4)	10 (23)	100 (230)	-
Alum Application	2.4 (5.3)	5.9 (13)	81 (130)	$230 ($100)
Iron Filings Application	2.9 (6.5)	7.2 (16)	59 (160)	$310 ($140)
No Sediment Release	1.9 (4.1)	4.6 (10)	88 (100)	-
Alameda Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	2.9 (6.5)	7.1 (16)	71 (160)	-
Alum Application	2.2 (4.9)	5.4 (12)	54 (120)	$920 ($420)
Iron Filings Application	2.4 (5.4)	5.9 (13)	59 (130)	$820 ($370)
No Sediment Release	2.0 (4.4)	4.9 (11)	49 (110)	-
Langton Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	0.045 (0.099)	0.11 (0.24)	1.1 (2.4)	-
Alum Application	0.044 (0.097)	0.11 (0.24)	1.1 (2.4)	$49,000 ($22,000)
Iron Filings Application	0.044 (0.098)	0.11 (0.24)	1.1 (2.4)	$280,000 ($130,000)
No Sediment Release	0.044 (0.098)	0.11 (0.24)	1.1 (2.4)	-
Minnetonka Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	0.026 (0.058)	0.064 (0.14)	0.64 (1.4)	-
Alum Application	0.021 (0.047)	0.052 (0.11)	0.52 (1.1)	$42,000 ($19,000)
Iron Filings Application	0.023 (0.050)	0.055 (0.12)	0.55 (1.2)	$82,000 ($37,000)
No Sediment Release	0.020 (0.043)	0.048 (0.11)	0.48 (1.1)	-

5.4 Reorientation of Outlet Works

This section considers the modification of pond outlet structures such that water is withdrawn from the pond at different elevations in the water column. The goal of such an 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.

5.4.1 Modeling Approach

For the sake of consistency, we configured all outlet structures as a single outlet pipe leading to a riser structure with a weir controlling the permanent pool elevation at 272 m (892.4 ft) (Figure 21). The base scenario had the invert of the outlet pipe at 0.1-m (0.33-ft) below the permanent pool elevation (for model stability); 271.9 m (892.1 ft). The center and bottom outlet scenarios had the outlet pipe invert elevations halfway through and at the bottom of each water column, respectively, relative to the bottom of the main body of the pond (avoiding smaller localized depressions or holes, even if deeper). For ponds where the last pond segment was not sufficiently deep to accommodate the deepest outlet pipe elevation, the outlet pipe was routed to the nearest pond segment (ponds span segments 5-15, upstream to downstream) that was at the desired depth (Table 10).

Outlet pipes received water from this same segment in each scenario so that we could evaluate the impact of changing the elevation of the pond outlet structure in isolation. Cells downstream of the rerouted pipe inlet cell were required by CE-QUAL-W2 to be active (part of the waterbody) and were, therefore, defined with a width of 0.01 m (0.033 ft) (to minimize their impact since they do not actually exist). This would simulate a scenario where a flexible pipe is extended from the outlet structure to a deeper part of the pond, a relatively simple and inexpensive remediation. In order to avoid conflating variables, the defined outlet pipe length, diameter, and other characteristics were kept consistent across ponds. We estimated total retrofit costs at \$65.88/m^2 (\$6.12/ft^2) of pond surface area based on information from the Washington Conservation District (2016), in which increasing size and complexity of outlet structures are factored into estimates. These values include median installation costs of \$37.29/m^2 (\$3.46/ft^2), design costs of 40% above construction, installation oversight costs of \$243 over three one-hour visits, ten years of marginal contracted annual maintenance costs based on a 30-year average rate of \$0.14/m^2 (\$0.01/ft^2) per year; costs adjusted for inflation from 2016 USD to 2021 USD values (U.S. Bureau of Labor Statistics 2021).

Table 10. *Outlet works reorientation costs for each pond. Total Costs include all anticipated costs over a 10-year period. All costs were adjusted by the CPI to 2021.*

Shoreview Pond			
Scenario	**Outlet Invert Elevation** m (ft)	**Surface Area** m^2 (ft^2)	**Total Cost***
Original	271.9 (892.1)	11,700 (126,300)	-
Center	271.5 (890.7)	11,700 (126,300)	$770,000
Bottom	270.9 (888.8)	11,700 (126,300)	$770,000
Alameda Pond			
Scenario	**Outlet Invert Elevation** m (ft)	**Surface Area** m^2 (ft^2)	**Total Cost***
Original	271.9 (892.1)	11,700 (125,900)	-
Center	271.1 (889.4)	11,700 (125,900)	$770,000
Bottom	270.2 (886.5)	11,700 (125,900)	$770,000
Langton Pond			
Scenario	**Outlet Invert Elevation** m (ft)	**Surface Area** m^2 (ft^2)	**Total Cost***
Original	271.9 (892.1)	650 (7,000)	-
Center	271.3 (890.1)	650 (7,000)	$43,000
Bottom	270.5 (887.5)	650 (7,00)	$43,000
Minnetonka Pond			
Scenario	**Outlet Invert Elevation** m (ft)	**Surface Area** m^2 (ft^2)	**Total Cost***
Original	271.9 (892.1)	6,900 (74,100)	-
Center	270.7 (888.3)	8,900 (74,100)	$450,000
Bottom	269.7 (884.8)	8,900 (74,100)	$450,000

*Costs rounded to two significant figures.

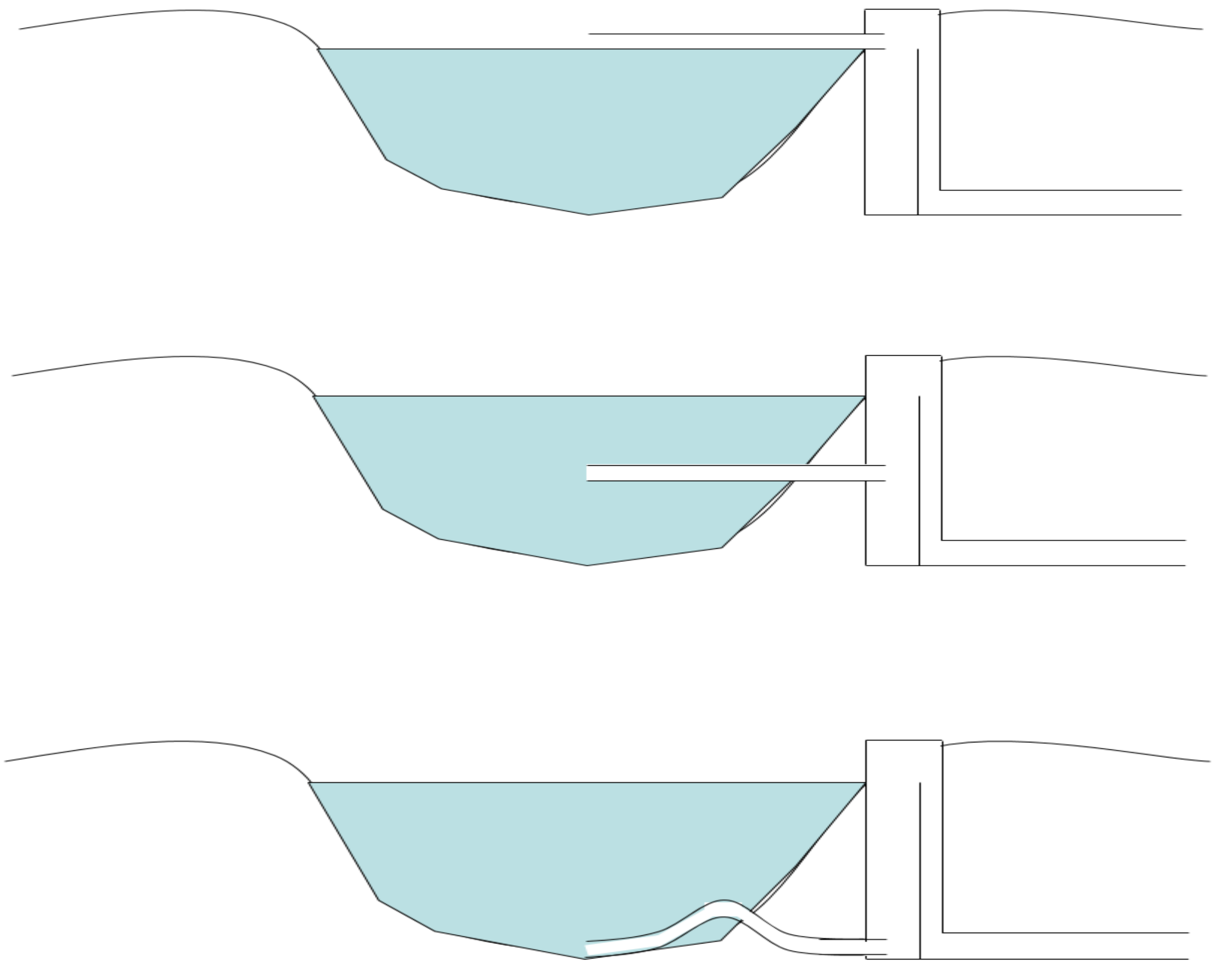

Figure 21. *Conceptual diagram of outlet works reorientation in Top, Center, and Bottom outlet scenarios.*

5.4.2 Results of Outlet Reorientation

Changing the depth of the withdrawal produced a slight increase in the TP export from the ponds and was, therefore, an ineffective remediation for these three scenarios (Table 11). The inflow was not sufficiently large and frequent to allow this technique to evacuate the water at the bottom of the pond before the relatively high sediment oxygen demand reduced the DO concentration towards zero mg/L. The likely mechanism of increased P export was the transport of P released from the sediment (or settled from inflows), which suggests that this management technique might be more effective if combined with a chemical treatment that prevents sediment release of P or is applied to a pond with a lower residence time (volume/inflow rate). This remediation technique will not be explored further in this report.

Table 11. *Results for total phosphorus (TP) export for outlet reorientation. TP Exports are calculated only for the approximately 3-month simulation period, while Total Costs include all anticipated costs for each remediation strategy over a 10-year period. Cost per % ∆ TP is a calculation of the total scenario cost divided by the percent improvement in TP export mass relative to the original condition. Cost per kg (lb) TP is a calculation of the total scenario cost divided by the TP export mass reduction in number of kg (lb). All costs were adjusted by the CPI to 2021.*

Shoreview Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP**	**Cost per kg (lb) TP**
Original	4.2 (9.4)	-	-	-
Center	4.3 (9.5)	1%	*	*
Bottom	4.3 (9.5)	1%	*	*
Alameda Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP**	**Cost per kg (lb) TP**
Original	2.9 (6.5)	-	-	-
Center	3.1 (6.9)	7%	*	*
Bottom	3.0 (6.5)	1%	*	*
Langton Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP**	**Cost per kg (lb) TP**
Original	0.045 (0.10)	-	-	-
Center	0.045 (0.10)	0%	*	*
Bottom	0.045 (0.10)	0%	*	*
Minnetonka Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP**	**Cost per kg (lb) TP**
Original	0.026 (0.06)	-	-	-
Center	0.029 (0.06)	9%	*	*
Bottom	0.028 (0.06)	6%	*	*

*No measured improvements in TP export.

5.5 Mechanical Aeration

This remediation technique aims to weaken or remove water column stratification and potentially expose sediments to more oxygenated water from the pond surface to promote phosphorus burial, or at least prevent anoxic phosphorus release. Various strategies exist for mechanically reducing stratification and aerating a water body using bubble plumes from air diffusers. In cases where complete destratification is not feasible or not desired, it is possible to mechanically oxygenate a water body while preserving stratification by injecting liquid oxygen directly. It must be noted that oxygenation is cost-prohibitive for most stormwater pond applications, particularly when pond managers are seeking solutions for many ponds under their care. It must also be noted that most surface fountains are primarily ornamental and do not provide as much aeration or destratification to the majority of the pond bottom as bubble plumes.

5.5.1 Modeling Approach

The version of CE-QUAL-W2 we are using in this modeling study (version 4.2) includes capabilities to model oxygenation but not aeration. The bubble plume dynamics necessary to model mechanical aeration are planned for a future version of CE-QUAL-W2 that has not yet been released at the time of writing (Wells 2019). We, therefore, sought to calculate aeration requirements for our study ponds external to the CE-QUAL-W2 model. We used the method of Lorenzen and Fast (1977), which allowed us to calculate the necessary air flow to entrain sufficient water and fully displace the volume of the water body within a reasonable amount of time. This method is often simplified to a rule-of-thumb airflow requirement of 9.2 m^3/min/km^2 (6,300 gal/min/mi^2) of waterbody surface area.

In deep lakes and reservoirs, the number of aeration diffusers is less important than the air flow rate because bubble plumes are able to efficiently disperse and interact with the majority of the water volume as the bubbles rise. Most stormwater ponds, however, are too shallow for bubble plumes to affect a large radius. As a starting point for our design recommendations, we used depth-sensitive aeration diffuser density recommendations provided by pond aeration specialists at Algae Control Canada (www.algaecontrol.ca) based on their experience (Table 12).

Table 12. *Aeration diffuser density recommendations from Algae Control Canada.*

Water Depth m (ft)	**Water Body Area Per Diffuser** m^2 (ac)
2 – 3 (6.6 – 9.8)	1,300 – 2,000 (0.32 – 0.49)
1 – 2 (3.3 – 6.6)	1,000 – 1,300 (0.25 – 0.32)
1 (3.3) or less	1,000 (0.25) or less

5.5.2 Cost Estimates

For the purpose of estimating costs to aerate each pond with the prerequisite number of diffusers, we requested itemized costs from Vertex Aquatic Solutions (www.vertexaquaticsolutions.com) (Table 13).

Table 13. *Aeration compressor pricing based on Vertex Aquatic Solutions 2021 Retail Pricing Catalog.*

Compressor Size hp	**Airflow Capacity** m^3/min (cfm)	**Compressor Cost***	**Cabinet Size**	**Cabinet Cost***
⅓	0.071 (2.5)	$1,800	Small	$390
½	0.122 (4.3)	$2,600	Medium	$480
¾	0.158 (5.6)	$2,700	Medium	$480
1	0.244 (8.6)	$4,300	Large	$580
1-½	0.317 (11.2)	$4,600	Large	$580

*Costs rounded to two significant figures.

We approximated total costs from quote and pricing documents provided by Vertex Aquatic Solutions (Table 14). Aeration system designs were based on airflow requirements, and cost estimates include the approximate electricity consumption over a 10-year lifespan assuming continuous operation for six (warm weather) months per year with an average cost of $0.07/kWH and an additional monthly summer surcharge of $14.79 per kW. As a way to estimate the impact that successfully aerating each pond would have on phosphorus dynamics, we ran each pond model with modified sediment phosphorus flux values measured for each of the four ponds in individual laboratory incubations using the method of Taguchi et al. (2018a). These oxic flux values differ from the more traditional anoxic flux values (Table 14) in that they were all greatly reduced from their anoxic counterparts. Some, notably the Alameda pond, had strongly negative oxic flux values, meaning that the pond sediments would act as a sink of phosphorus; we have approximated all negative fluxes as 0 for the sake of a conservative modeling approach since it is unknown how long negative fluxes would be maintained in the field.

Table 14. *Aeration system cost estimates. Total Costs include all anticipated costs over a 10-year period.*

Pond	**Oxic SRP Flux** mg/m^2/day	**Required Diffusers**	**Airflow per Diffuser at Depth** m^3/min (cfm)	**Required Compressor(s)** hp	**Compressor(s) Cost***	**Total Cost***
Shoreview	2.65	9	0.037 (1.3)	1-½ + ½	$18,000	$26,000
Alameda	0.00**	8	0.037 (1.3)	1-½ + ⅓	$14,000	$21,000
Langton	0.00**	1	0.059 (2.1)	⅓	$2,800	$4,100
Minnetonka	0.00**	5	0.059 (2.1)	1-½ + ½	$13,000	$21,000

*Costs rounded to two significant figures.

**No oxic SRP release detected.

5.5.3 Results of Mechanical Aeration for Remediation

By modeling the effect of mechanical aeration on sediment SRP release rates, we are able to quantify the potential impact of this remediation strategy on the net TP export for each pond (Table 15). This remediation strategy appears to be effective for ponds where large anoxic SRP flux values (Table 5) can be replaced with smaller oxic SRP values (Table 14). It is difficult to evaluate its true cost-effectiveness without knowing the ultimate operation and maintenance costs that will be incurred, but a rough assessment has been made based on implementation costs alone (Table 15).

To place the phosphorus export of the remediation strategy on the same time frame as the costs, we observe that approximately 41% of the precipitation that fell between 1991 and 2020 occurred in the months June, July, and August according to data available from the US National Weather Service (https://www.weather.gov/wrh/climate?wfo=mpx). Roughly equating precipitation to export of phosphorus, then the approximately 3-month simulation TP export values represent 41% of the annual pond TP export. Table 16 provides 1-Year and 10-Year TP export values as rough approximations based on the assumption that cost per kg (lb) TP is a calculation of the 10-year scenario cost divided by the approximated 10-Year TP export mass reduction.

Table 15. *Results for total phosphorus (TP) export for aeration. TP Exports are calculated only for the approximately 3-month simulation period, while Total Costs include all anticipated costs for each remediation strategy over a 10-year period. Cost per % ∆ TP is a calculation of the total scenario cost divided by the percent improvement in TP export mass relative to the original condition. Cost per kg (lb) TP is a calculation of the total scenario cost divided by the TP export mass reduction in number of kg (lb).*

Shoreview Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	4.2 (9.4)	-	-	-
Mechanical Aeration	2.7 (5.9)	-37%	$700	$16,000 ($7,400)
Alameda Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	2.9 (6.5)	-	-	-
Mechanical Aeration	2.0 (4.4)	-56%	$670	$23,000 ($10,000)
Langton Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	0.045 (0.099)	-	-	-
Mechanical Aeration	0.044 (0.097)	-2%	$2,200	$4,900,000 ($2,200,000)
Minnetonka Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	0.026 (0.058)	-	-	-
Mechanical Aeration	0.020 (0.043)	-25%	$830	$3,100,000 ($1,400,000)

*Costs rounded to two significant figures.

Table 16. *Evaluation of cost-effectiveness over a 10-year span for mechanical aeration. 1-Year and 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).* All values are rounded to two significant figures.

Shoreview Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	4.2 (9.4)	10 (23)	100 (230)	-
Mechanical Aeration	2.7 (5.9)	6.6 (14)	66 (140)	$670 ($300)
Alameda Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	2.9 (6.5)	7.1 (16)	71 (160)	-
Mechanical Aeration	2.0 (4.4)	4.9 (11)	49 (110)	$940 ($430)
Langton Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	0.045 (0.099)	0.11 (0.24)	1.1 (2.4)	-
Mechanical Aeration	0.044 (0.097)	0.11 (0.24)	1.1 (2.4)	$200,000 ($92,000)
Minnetonka Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	0.026 (0.058)	0.064 (0.14)	0.64 (1.4)	-
Mechanical Aeration	0.020 (0.043)	0.048 (0.11)	0.48 (1.1)	$130,000 ($59,000)

5.6 Wind Sheltering Reduction

Wind plays an important role in the mechanical mixing of surface waters, and in shallow water bodies may provide mixing of the water column. The benefit of mixing to stormwater ponds, as with other remediation techniques (e.g., reorientation of outlet works, mechanical aeration), is to reduce stratification strength and expose the entire pond to oxygen in the atmosphere in order to prevent anoxic sediment release of phosphorus.

5.6.1 Modeling Approach

We modeled wind exposure for each pond using the commercial computational fluid dynamics (CFD) package ANSYS-Fluent. The CFD models were used to determine wind sheltering coefficients for each pond, taking into account both fetch (open water distance) and wind sheltering from surrounding trees and terrain. We placed one anemometer on each pond at a roughly 3-foot height above the water to measure wind velocities over the pond. We then calibrated the CFD models for each pond by adjusting the tree density (porosity) around the pond and by adjusting the incoming wind speed profile, based on the roughness characteristics of the surrounding land (forest, mixed development, etc.). An example simulation is shown in Figures 22, 23a, and 23b. The ratio of the wind speed measured on the pond to the airport wind speed was used as the calibration parameter, for wind data binned into eight 45° directional bins. Based on wind calibration data taken at the St. Anthony Falls Laboratory (SAFL) with multiple anemometers, the pond anemometer wind speeds ($U_{measured}$) were adjusted as:

$$U_{corrected} = U_{measured} + 0.469 \cdot (1 - \exp(-16.34 * U_{measured})) \quad \textbf{(Eq. 2)}$$

We then calculated a wind sheltering coefficient for each directional bin:

$$C_{sheltering} = \frac{\sqrt[2]{\tau/\tau_1}}{f(fetch)} \quad \textbf{(Eq. 3)}$$

where τ is the modeled average shear stress on the pond, τ_1 is the equilibrium shear stress far downwind , and $f(fetch)$ is the fetch correction used internally in CE-QUAL-W2 (Wells 2019). The $C_{sheltering}$ coefficient is used as a multiplicative correction to the airport wind speeds used as inputs to the CE-QUAL models. Shear stress is used as the basis for correcting wind speeds, because wind shear stress drives the vertical mixing processes in ponds. The square root takes into account that shear stress varies as wind speed squared. Since the CFD model includes the effect of fetch on shear stress, and CE-QUAL also does a fetch correction internally (Wells 2019), the $f(fetch)$ function was added so that the wind speed is not corrected for sheltering twice. In CE-QUAL, the default fetch function is:

$$f(fetch) = \left(\frac{5ZB+4.6052}{3ZB+9.2103}\right) \quad \textbf{(Eq. 4)}$$

where $ZB = 0.8\, ln(fetch/2) - 1.0718$.

Figure 22. *Conceptual diagram of wind sheltering grid in ANSYS-Fluent. The highlighted numbers (0.3, 0.5, 0.6) represent the relative tree densities input to the Fluent model.*

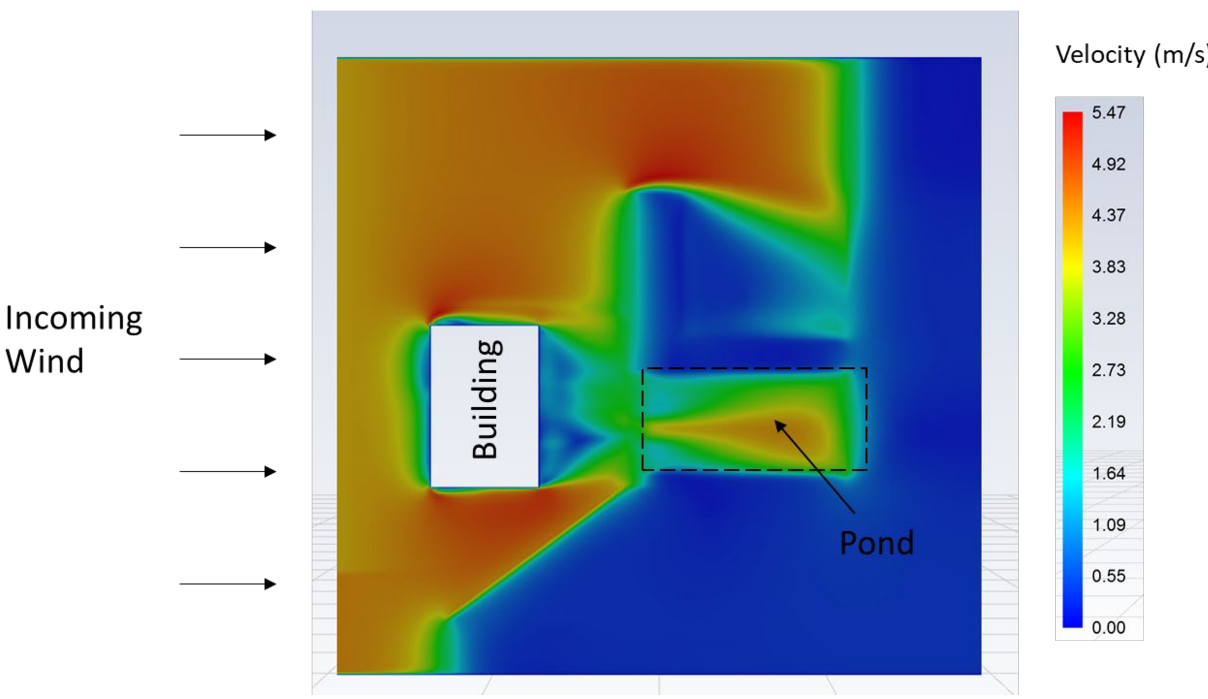

Figure 23a. *Simulated wind velocity distribution in and around the Shoreview pond (map view). The blue areas are forested and sheltered areas with low wind velocity. The simulation domain is approximately 300 m long x 300 m wide x 100 m tall (980 ft x 980 ft x 320 ft).*

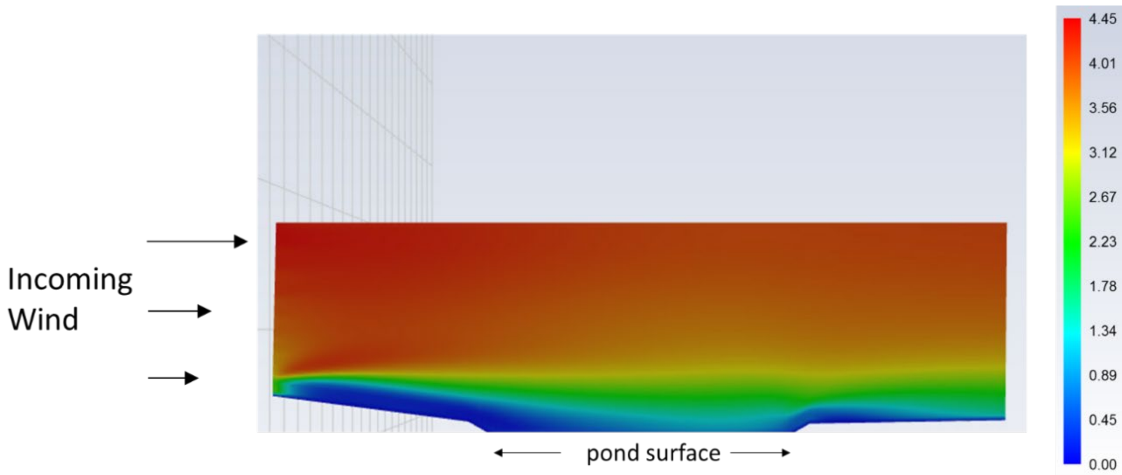

Figure 23b. *Simulated wind velocity distribution in and around the Alameda pond (cross-section view along pond centerline), showing the approximate pond embankment and the surrounding topography. The blue areas are forested and sheltered areas with low wind velocity.*

The ANSYS-FLUENT modeling effort has simulated the baseline and scenario wind sheltering coefficients for the Shoreview pond and our current modeling effort applies the same wind sheltering conditions to each pond. Based upon the results and the expense of operating a three-dimensional fluid mechanics simulation, we based all wind sheltering coefficients on the Shoreview pond. Bulk treatments were evaluated by calculating wind sheltering coefficients under a 100% tree removal scenario (sheltering from only topography and non-tree obstructions) and calculating an intermediate wind sheltering coefficient to represent an intermediate tree removal level (Table 17). We applied these same treatments to the Alameda and Langton ponds in order to gauge their sensitivity to wind sheltering.

For the purpose of estimating costs, we used information provided by the City of Roseville, MN (Ryan Johnson, personal communication) and decided on an approximate rate of \$4.32 per m^2 (\$0.40 per ft^2) of tree removal area. We assume that this treatment would have to be repeated approximately every 10 years.

Table 17. *Wind sheltering coefficients (WSCs) for bulk reductions. Total Costs include all anticipated costs over a 10-year period.*

Shoreview Pond					
Wind Direction	**Original**	**50% Reduction**	**∆ WSC**	**100% Reduction**	**∆ WSC**
North	0.236	0.226	-4%	0.216	-8%
Northeast	0.269	0.304	13%	0.338	26%
East	0.266	0.304	14%	0.341	28%
Southeast	0.361	0.625	73%	0.889	146%
South	0.412	0.510	24%	0.608	48%
Southwest	0.332	0.321	-3%	0.310	-7%
West	0.256	0.282	10%	0.307	20%
Northwest	0.268	0.370	38%	0.472	76%
Approx. Tree Removal – m^2 (ft^2)	-	33,500 (360,900)	-	67,100 (721,800)	-
Cost	-	$140,000	-	$290,000	-

*Costs rounded to two significant figures.

In addition to the bulk wind sheltering scenarios examined for each pond, we explored more opportunistic targeted tree removal scenarios for the Shoreview pond. We modified the 3-D wind sheltering model in ANSYS-Fluent to reflect each scenario and generated new wind sheltering coefficients (Table 18) that we subsequently applied to CE-QUAL-W2 for comparison. Visual representations of Cases A, B, and C for the Shoreview pond are depicted in Figures 24 - 26.

Table 18. *Wind sheltering coefficients (WSCs) for specific scenarios. Total Costs include all anticipated costs over a 10-year period.*

Shoreview Pond						
Wind Direction	**Case A**	**∆ WSC**	**Case B**	**∆ WSC**	**Case C**	**∆ WSC**
North	0.226	-4%	0.227	-4%	0.221	-6%
Northeast	0.264	-2%	0.264	-2%	0.363	35%
East	0.264	1%	0.269	-1%	0.276	4%
Southeast	0.379	7%	0.388	5%	0.410	14%
South	0.393	-1%	0.408	-5%	0.396	-4%
Southwest	0.375	10%	0.365	13%	0.406	22%
West	0.276	20%	0.307	8%	0.309	21%
Northwest	0.426	63%	0.438	59%	0.425	59%
Approx. Tree Removal m^2 (ft^2)	2,200 (23,700)	-	9,900 (106,400)	-	15,700 (169,300)	-
Cost	$9,500	-	$43,000	-	$68,000	-

*Costs rounded to two significant figures.

Figure 24. *Conceptual diagram of the Case A potential tree removal scenario for the Shoreview pond. This scenario targets a thin line of trees shielding the pond from the direction of prevailing wind (northwest).*

Figure 25. *Conceptual diagram of the Case B potential tree removal scenario for the Shoreview pond. This scenario more broadly exposes the pond to winds from the northern and western directions by removing a large swath of trees that are not adjacent to residential properties.*

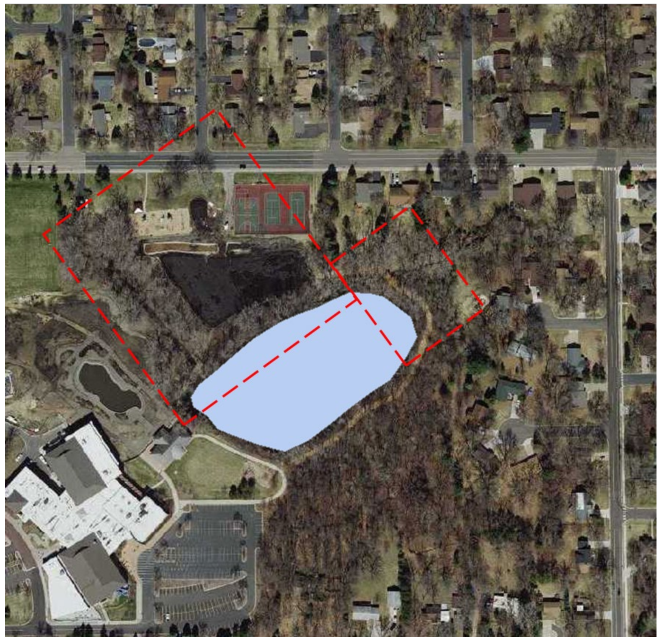

Figure 26. *Conceptual diagram of the Case C potential tree removal scenario for the Shoreview pond. This scenario expands the tree removal area to encompass the northern half of the pond with limited tree removal adjacent to residential properties.*

5.6.2 Results of Reduced Sheltering

The wind sheltering reduction scenarios we evaluated did not greatly impact anoxia in the ponds, even at 100% reduction of sheltering from trees (Table 19). Similarly, while the tree removal scenarios we evaluated for each pond improved mixing, they were insufficient to fully destratify the ponds. As a result, TP export saw minimal improvements, and the unit cost of those improvements was quite high (Table 20). Wind sheltering reduction as a remediation strategy may be more effective when used in tandem with other strategies. Another approach would be to maintain ponds in an unsheltered state from their initial construction, and avoid placing them in locations with high banks, as this would be a more cost-effective way to derive benefits from increased wind exposure. We did not attempt to compare the results on a 10-year phosphorus export reduction basis, as the results of this remediation were not cost-effective.

Table 19. *Anoxic days resulting from a reduction of wind sheltering.*

Shoreview Pond				
Scenario	**Days No Oxia**	**∆ Days No Oxia**	**Days Any Anoxia**	**∆ Days Any Anoxia**
Original	92	-	99	-
50% Reduction	92	0%	99	0%
100% Reduction	90	-2%	99	0%
Case A	91	-1%	99	0%
Case B	92	0%	99	0%
Case C	92	0%	99	0%
Alameda Pond				
Scenario	**Days No Oxia**	**∆ Days No Oxia**	**Days Any Anoxia**	**∆ Days Any Anoxia**
Original	88	-	99	-
50% Reduction	88	0%	99	0%
100% Reduction	88	0%	99	0%
Langton Pond				
Scenario	**Days No Oxia**	**∆ Days No Oxia**	**Days Any Anoxia**	**∆ Days Any Anoxia**
Original	33	-	84	-
50% Reduction	29	-12%	83	-1%
100% Reduction	28	-15%	79	-6%
Minnetonka Pond				
Scenario	**Days No Oxia**	**∆ Days No Oxia**	**Days Any Anoxia**	**∆ Days Any Anoxia**
Original	99	-	99	-
50% Reduction	99	0%	99	0%
100% Reduction	99	0%	99	0%

Table 20. *Results for total phosphorus (TP) export from wind sheltering reduction. TP Exports are calculated only for the approximately 3-month simulation period, while Total Costs include all anticipated costs for each remediation strategy over a 10-year period. Cost per % ∆ TP is a calculation of the total scenario cost divided by the percent improvement in TP export mass relative to the original condition. Cost per kg (lb) TP is a calculation of the total scenario cost divided by the TP export mass reduction in number of kg (lb).*

Shoreview Pond				
Scenario	**TP Export kg (lb)**	**∆**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	4.2 (9.4)	-	-	-
50% Reduction	4.2 (9.2)	-2%	$95,000	$2,200,000 ($1,000,000)
100% Reduction	4.1 (9.1)	-3%	$88,000	$2,100,000 ($940,000)
Case A	4.2 (9.3)	-1%	$16,000	$380,000 ($170,000)
Case B	4.2 (9.3)	0%	$100,000	$2,400,000 ($1,100,000)
Case C	4.2 (9.3)	-1%	$76,000	$1,800,000 ($810,000)
Alameda Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	2.9 (6.5)	-	-	-
50% Reduction	2.9 (6.4)	-1%	**	**
100% Reduction	2.9 (6.3)	-2%	**	**
Langton Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	0.045 (0.099)	-	-	-
50% Reduction	0.045 (0.099)	0%	**	**
100% Reduction	0.044 (0.098)	-1%	**	**
Minnetonka Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	0.026 (0.058)	-	-	-
50% Reduction	0.023 (0.051)	-12%	**	**
100% Reduction	0.019 (0.043)	-26%	**	**

*Costs rounded to two significant figures.

**Cost data not available.

5.7 Watershed-Based Methods

In addition to pond maintenance and remediation, stormwater management also typically includes efforts to reduce stormwater export through watershed improvements. Therefore, it is of interest to evaluate how each pond would respond to watershed improvements that target water volume (e.g., infiltration stormwater practices) or that target pollutant concentrations and mass loads (e.g., filtration stormwater practices, enhanced street sweeping).

5.7.1 Modeling Approach

Changes to watershed loading were evaluated on the basis of percent reductions to volumetric and concentration loading (Table 21). Costs for volume and concentration reduction strategies are extremely variable depending on watershed constraints, local regulations, and preferred strategies or practice types.

One estimate that could be made on the cost of TP mass reduction (here modeled as the 50% Concentration scenario) was on the basis of increased municipal street sweeping as studied by Kalinosky et al. (2014) in Prior Lake, MN. They studied street sweeping along routes with low-, medium-, and high- tree canopy densities at frequencies of one, two, and four times a month over two years and tabulated costs including labor, fuel, and operation and maintenance of the street sweeper vehicles; the cost of purchasing each street sweeper vehicle was not included. The overall average was found to be $707/kg ($321/lb) of TP mass removed (costs adjusted for inflation from 2014 USD to 2021 USD values (U.S. Bureau of Labor Statistics 2021)). By month, cost values ranged from an average of $107/kg ($49/lb) TP for an area with high tree canopy density being swept twice a month in October to an average of approximately $1,675/kg ($760/lb) for low tree canopy density being swept four times a month in July. According to data available from the US NWS (https://www.weather.gov/wrh/climate?wfo=mpx), approximately 41% of the precipitation that fell between 1991 and 2020 occurred in the months June, July, and August. We used this factor to scale the watershed TP reductions during our simulation period to annual values for the purpose of estimating 10-year costs to be compared with the other remediation strategies in this study (Table 21).

The cost of volume reduction (modeled as the 50% Volume scenario) was similar to that used by Moore et al. (2016) and attributed to Weiss et al. (2007) based on the construction costs of bioinfiltration practices and 20 years of operation and maintenance costs, with prices calculated according to the water quality volume (WQV), in units of cubic meters, treated by the practice:

$$Biofiltration\ Practice\ Cost\ (2005\ USD) = 1542 * WQV^{0.776}$$ (Weiss et al. 2007) **(Eq. 5)**

We adjusted tabulated cost values in Table 21 for inflation from 2005 USD to 2021 USD values (U.S. Bureau of Labor Statistics 2021). Additional formula coefficients were provided by Weiss et al. (2007) for lower and upper 67% confidence interval values in order to capture the wide variability in cost estimates for watershed-based methods (Table 21). The divided the costs calculated using the Weiss et al. (2007) by a factor of 2 to yield 10-year costs rather than 20-year costs so we could compare them against the other remediation strategies being evaluated in this study.

Table 21. *Scenarios for watershed-based methods. Inflow mass estimated over 10 years. 50% Volume Reduction scenarios refer to the implementation of infiltration SCMs and 50% Concentration Reduction scenarios refer to the implementation of street-sweeping. Total Costs include all anticipated over a 10-year period. All costs were adjusted by the CPI to 2021.*

Shoreview Pond						
Scenario	**Simulated Inflow Vol.** m^3 (gal)	**Mean Inflow TP Conc.** mg/L	**Inflow TP Mass** kg (lb)	**Average Cost Estimate***	**Low Cost Estimate***	**High Cost Estimate***
Original	104 (27,400)	0.378	270 (590)	-	-	-
50% Volume	52 (13,700)	0.378	130 (300)	$23,000	$15,000	$47,000
50% Concentration	104 (27,400)	0.189	130 (300)	$95,000	$14,000	$220,000
Alameda Pond						
Scenario	**Simulated Inflow Vol.** m^3 (gal)	**Mean Inflow TP Conc.** mg/L	**Inflow TP Mass** kg (lb)	**Average Cost Estimate***	**Low Cost Estimate***	**High Cost Estimate***
Original	104 (27,400)	0.378	270 (590)	-	-	-
50% Volume	52 (13,700)	0.378	130 (300)	$23,000	$15,000	$47,000
50% Concentration	104 (27,400)	0.189	130 (300)	$95,000	$14,000	$220,000
Langton Pond						
Scenario	**Simulated Inflow Vol.** m^3 (gal)	**Mean Inflow TP Conc.** mg/L	**Inflow TP Mass** kg (lb)	**Average Cost Estimate***	**Low Cost Estimate***	**High Cost Estimate***
Original	3.1 (820)	0.378	16 (36)	-	-	-
50% Volume	1.6 (410)	0.378	8.0 (18)	$1,500	$900	$3,700
50% Concentration	3.1 (820)	0.189	8.0 (18)	**	**	**
Minnetonka Pond						
Scenario	**Simulated Inflow Vol.** m^3 (gal)	**Mean Inflow TP Conc.** mg/L	**Inflow TP Mass** kg (lb)	**Average Cost Estimate***	**Low Cost Estimate***	**High Cost Estimate***
Original	13.5 (3,600)	0.378	65 (140)	-	-	-
50% Volume	6.8 (1,800)	0.378	32 (72)	$4,800	$2,900	$10,800
50% Concentration	13.5 (3,600)	0.189	32 (72)	**	**	**

*Costs rounded to two significant figures.
**Data not available for small watersheds.

5.7.2 Results of Watershed-Based Methods for Remediation

All three modeled ponds had a reduction in mean surface TP concentration (Table 22) – and mean water column TP concentration (Table 23) -- for a reduction of incoming TP concentration, highlighting the importance of watershed P source reduction strategies. Conversely, the three ponds all showed an increase in surface/water column TP (very slight for Langton and Alameda ponds) for the scenario of a reduction of volume of inflows to the ponds, which would be a potential outcome of widespread implementation of infiltration-based stormwater management practices. This result is likely the result of a lack of dilution and flushing of in-pond TP by inflows.

Table 22. *Results for surface total phosphorus (TP) concentrations. 50% Volume Reduction scenarios refer to the implementation of infiltration SCMs and 50% Concentration Reduction scenarios refer to the implementation of street-sweeping.*

Shoreview Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.350 ± 0.010	-	0.360	-
50% Volume	0.399 ± 0.014	14%	0.409	14%
50% Concentration	0.313 ± 0.008	-11%	0.316	-12%
Alameda Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.223 ± 0.004	-	0.228	-
50% Volume	0.239 ± 0.005	7%	0.245	7%
50% Concentration	0.192 ± 0.002	-14%	0.194	-15%
Langton Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.173 ± 0.002	-	0.177	-
50% Volume	0.178 ± 0.002	3%	0.181	2%
50% Concentration	0.150 ± 0.001	-14%	0.150	-15%
Minnetonka Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.302 ± 0.009	-	0.311	-
50% Volume	0.316 ± 0.011	5%	0.323	4%
50% Concentration	0.291 ± 0.009	-4%	0.299	-4%

Table 23. *Results for water column-averaged total phosphorus (TP) concentrations. 50% Volume Reduction scenarios refer to the implementation of infiltration SCMs and 50% Concentration Reduction scenarios refer to the implementation of street-sweeping.*

Shoreview Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.345 ± 0.010	-	0.358	-
50% Volume	0.399 ± 0.013	16%	0.412	15%
50% Concentration	0.307 ± 0.007	-11%	0.312	-13%
Alameda Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.211 ± 0.004	-	0.210	-
50% Volume	0.235 ± 0.005	11%	0.242	15%
50% Concentration	0.178 ± 0.002	-15%	0.179	-15%
Langton Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.172 ± 0.002	-	0.175	-
50% Volume	0.177 ± 0.002	3%	0.180	3%
50% Concentration	0.148 ± 0.001	-14%	0.149	-15%
Minnetonka Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.313 ± 0.009	-	0.323	-
50% Volume	0.326 ± 0.010	4%	0.329	2%
50% Concentration	0.302 ± 0.008	-4%	0.311	-4%

Despite modeling this remediation strategy at a 50% reduction to external P loading and no change to internal P loading, the cumulative impact on TP export appears to be greater than 50% reduction in the case of the volume reduction scenarios (Table 24). This is because ponds receiving less inflow generally experience less outflow and, therefore, retain a higher fraction of the P they receive even if water column TP concentrations increase (Janke et al. 2022).

Improvements to TP export were more moderate in the concentration reduction scenarios because internal P loading would still be exported from the pond at the same rate since outflow volumes were not reduced. Still, cost estimates suggest that concentration reduction strategies may be slightly more cost-effective than volume-reduction strategies because of the differences in implementation costs (Street-Sweeping versus building infiltration facilities). Actual decision-making between the two approaches will need to carefully consider the implementation costs in specific watersheds.

Table 24. *Results for total phosphorus (TP) export of watershed treatment. TP Exports are calculated only for the approximately 3-month simulation period, while Total Costs include all anticipated costs for each remediation strategy over a 10-year period. Cost per % ∆ TP is a calculation of the total scenario cost divided by the percent improvement in TP export mass relative to the original condition. Cost per kg (lb) TP is a calculation of the total scenario cost divided by the TP export mass reduction in number of kg (lb). All costs were adjusted by the CPI to 2021.*

Shoreview Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % Improvement***	**Cost per kg (lb) TP***
Original	4.2 (9.4)	-	-	-
50% Volume	1.7 (3.8)	-60%	$390	$9,200 ($4,200)
50% Concentration	3.8 (8.3)	-11%	$600	$200,000 ($92,000)
Alameda Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	2.9 (6.5)	-	-	-
50% Volume	1.3 (2.8)	-57%	$410	$14,000 ($6,400)
50% Concentration	2.5 (5.5)	-15%	$450	$220,000 ($100,000)
Langton Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	0.045 (0.099)	-	-	-
50% Volume	0.004 (0.009)	-91%	$17	$38,000 ($17,000)
50% Concentration	0.039 (0.085)	-14%	**	**
Minnetonka Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	0.026 (0.058)	-	-	-
50% Volume	0.000 (0.000)	-100%	$48	$180,000 ($83,000)
50% Concentration	0.026 (0.057)	-2%	**	**

*Costs rounded to two significant figures.

**Data not available for small watersheds.

To place the phosphorus export of the remediation strategy on the same time frame as the costs, we observe that approximately 41% of the precipitation that fell between 1991 and 2020 occurred in the months June, July, and August according to data available from the US National Weather Service (https://www.weather.gov/wrh/climate?wfo=mpx). Roughly equating precipitation to export of phosphorus, then the approximately 3-month simulation TP export values represent 41% of the annual pond TP export. Table 25 provides 1-Year and 10-Year TP export values as rough approximations based on the assumption that cost per kg (lb) TP is a calculation of the 10-year scenario cost divided by the approximated 10-Year TP export mass reduction.

Table 25. *Evaluation of cost-effectiveness over a 10-year span of watershed reduction scenarios. 1-Year and 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 kilograms (pounds); this is a common metric for water quality decision-making, but values are inflated by the low original TP export values. All values are rounded to two significant figures.*

Shoreview Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	4.2 (9.4)	10 (23)	100 (230)	-
50% Volume	1.7 (3.8)	4.2 (9.2)	42 (92)	$380 ($170)
50% Concentration	3.8 (8.3)	9.2 (20)	92 (200)	$8,300 ($3,800)
Alameda Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	2.9 (6.5)	7.1 (16)	71 (160)	-
50% Volume	1.3 (2.8)	3.1 (6.8)	31 (68)	$580 ($260)
50% Concentration	2.5 (5.5)	6.1 (13)	61 (130)	$9,100 ($4,100)
Langton Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	0.045 (0.099)	0.11 (0.24)	1.1 (2.4)	-
50% Volume	0.004 (0.009)	0.010 (0.022)	0.10 (0.22)	$1,500 ($700)
50% Concentration	0.039 (0.085)	0.094 (0.21)	0.94 (2.1)	*
Minnetonka Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	0.026 (0.058)	0.064 (0.14)	0.64 (1.4)	-
50% Volume	0.000 (0.000)	0.000 (0.00)	0.00 (0.0)	$7,500 ($3,400)
50% Concentration	0.026 (0.057)	0.063 (0.14)	0.63 (1.4)	*

*Data not available for small watersheds.

5.8 Bathymetry Modification

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. Maintenance practices that effect pond bathymetry include dredging (sediment removal) and retrofitting (enlarging or reducing surface area); ponds also tend to fill in with sediment over time (sediment accumulation). Here, we evaluated the effects of these of pond bathymetry modifications on phosphorus dynamics.

5.8.1 Modeling Approach

We considered possible modifications to pond bathymetries within two distinct contexts: (1) changes to an existing pond and (2) redesigns for new ponds (Table 26). In the case of changes to existing ponds, pond depths (maximum depths ranging 1.65-1.96 m or 5.41-6.43 ft) were modified without any adjustments being made to the pond surface areas to mimic “dredging” and “filling” scenarios: ponds are deepened to twice their original depths (pond volumes doubled; maximum depths ranging 3.30-3.92 m or 10.83-12.86 ft) in one scenario and made shallower to be half their original depths (pond volumes halved; maximum depths ranging 0.83-0.98 m or 2.72-3.22 ft) in another scenario (Figure 27). In the case of redesigns for new ponds, the depths of each pond were modified with adjustments made to pond surface areas in order to preserve pond volumes. These are considered “redesign” scenarios in which similarly sized ponds could be constructed with different dimensions: in one scenario the ponds are deepened to twice their original depths, and in another scenario the ponds are made shallower to be half their original depths (Figure 28). Most of the evaluated modifications would occur naturally, in the case of sedimentation, or at the time of new construction, in the case of redesigns, and therefore do not have any costs that we can estimate.

There are data available to estimate the cost of pond dredging, and multiple cities were canvassed to develop a representative estimated dredging cost. However, potential contaminants in pond sediments that are hazardous to human health, when present in concentrations exceeding the MPCA prescribed limit, could increase the cost of sediment disposal by two to eight times (Kyser 2018) as sediment associated with dredging would be classified as hazardous waste and would need to be disposed of accordingly. In addition, dewatering and site preparation can increase costs per cubic yard of material, depending upon the size of the pond. Our estimate of pond sedimentation rates from one typical pond (St. Cloud Pond 52) is 3 cm/year (1.2 in/yr). We are basing all cost estimates on a 10-year life, so the 10-year sedimentation would be 30 cm (1 ft). Dredging cost invoices are estimated as a cost per ton of dredged material. Pond sediment typically has a specific gravity of 1.2 (weight of sediment over weight of water). Thus, the cubic yards of sediment collected after 10 years would be:

$$\frac{1\ ft\ sed\left(43{,}560\frac{ft^2}{acre}\right)62.4\,\frac{lbs}{ft^3}}{1.2\frac{sed}{water}\,(2200\,\frac{lbs}{ton})} = 1030\ \frac{tons}{acre} \qquad \textbf{(Eq. 6)}$$

The cost of dredging and disposing of the material typically varies from \$32 to \$41 per ton, although costs have been \$130/ton for smaller ponds in a difficult area (RWMWD, 2022; St. Cloud, 2022; Stantec, 2022). At \$36 per ton, the cost of dredging a typical pond of 1 or more acres surface area would be \$37,000/acre (\$9/m^2) after 10 years of sedimentation and \$134,000/acre (\$33/m^2) for ponds smaller than 1 acre. The calculated costs are shown in Table 27.

Table 26. *Descriptions of bathymetry modification scenarios. Total Costs include all anticipated costs over a 10-year period.*

Typical Dredging (Every 10 Years)				
Pond	**Surface Area** m^2 (ac)	**Dredging Mass** tons	**Cost per Ton**	**Total Cost***
Shoreview	11,700 (2.9)	3,000	\$36	\$110,000
Alameda	11,700 (2.9)	3,000	\$36	\$110,000
Langton	650 (0.16)	160	\$130	\$21,000
Minnetonka	6,900 (1.7)	1,800	\$36	\$63,000
Dredged Remediation Scenario (Depth Doubled)				
Pond	**Change in Depth** m (ft)	**Change in Volume** m^3 (gal)	**Dredging Mass** tons	**Total Cost***
Shoreview	1.8 (6.0)	10,000 (2,700,000)	13,000	\$480,000
Alameda	2.0 (6.4)	11,000 (3,000,000)	15,000	\$540,000
Langton	1.7 (5.4)	630 (170,000)	830	\$110,000
Minnetonka	2.6 (8.6)	1,000 (270,000)	1,400	\$49,000

*Costs rounded to two significant figures.

Table 27. *Descriptions of bathymetry modification scenarios. Total Costs include all anticipated costs over a 10-year period.*

Shoreview Pond				
Scenario	**Model Depth** m (ft)	**Model Volume** m^3 (gal)	**Procedure**	**Total Cost***
Original	1.82 (5.97)	10,027 (2,648,900)	-	-
Filled	0.91 (2.99)	5,091 (1,344,900)	Sedimentation	$0
Dredged	3.64 (11.94)	20,133 (5,318,600)	Dredging	$480,000
Redesign Shallow	0.91 (2.99)	10,151 (2,681,600)	New Construction	**
Redesign Deep	3.64 (11.94)	10,111 (2,671,000)	New Construction	**
Alameda Pond				
Scenario	**Model Depth** m (ft)	**Model Volume** m^3 (gal)	**Procedure**	**Total Cost***
Original	1.96 (6.43)	11,114 (2,936,000)	-	-
Filled	0.98 (3.22)	5,508 (1,455,100)	Sedimentation	$0
Dredged	3.92 (12.86)	22,411 (5,920,400)	Dredging	$540,000
Redesign Shallow	0.98 (3.22)	10,992 (2,903,800)	New Construction	**
Redesign Deep	3.92 (12.86)	11,237 (2,968,500)	New Construction	**
Langton Pond				
Scenario	**Model Depth** m (ft)	**Model Volume** m^3 (gal)	**Procedure**	**Total Cost***
Original	1.65 (5.41)	542 (143,200)	-	-
Filled	0.83 (2.71)	254 (67,100)	Sedimentation	$0
Dredged	3.30 (10.83)	1,169 (308,800)	Dredging	$110,000
Redesign Shallow	0.83 (2.71)	501 (312,400)	New Construction	**
Redesign Deep	3.30 (10.83)	591 (156,100)	New Construction	**
Minnetonka Pond				
Scenario	**Model Depth** m (ft)	**Model Volume** m^3 (gal)	**Procedure**	**Total Cost***
Original	2.62 (8.60)	955 (252,300)	-	-
Filled	1.31 (4.30)	468 (123,700)	Sedimentation	$0
Dredged	5.24 (17.19)	1,992 (526,200)	Dredging	$49,000
Redesign Shallow	1.31 (4.30)	928 (245,200)	New Construction	**
Redesign Deep	5.24 (17.19)	1004 (265,000)	New Construction	**

*Costs rounded to two significant figures.

**Data not available.

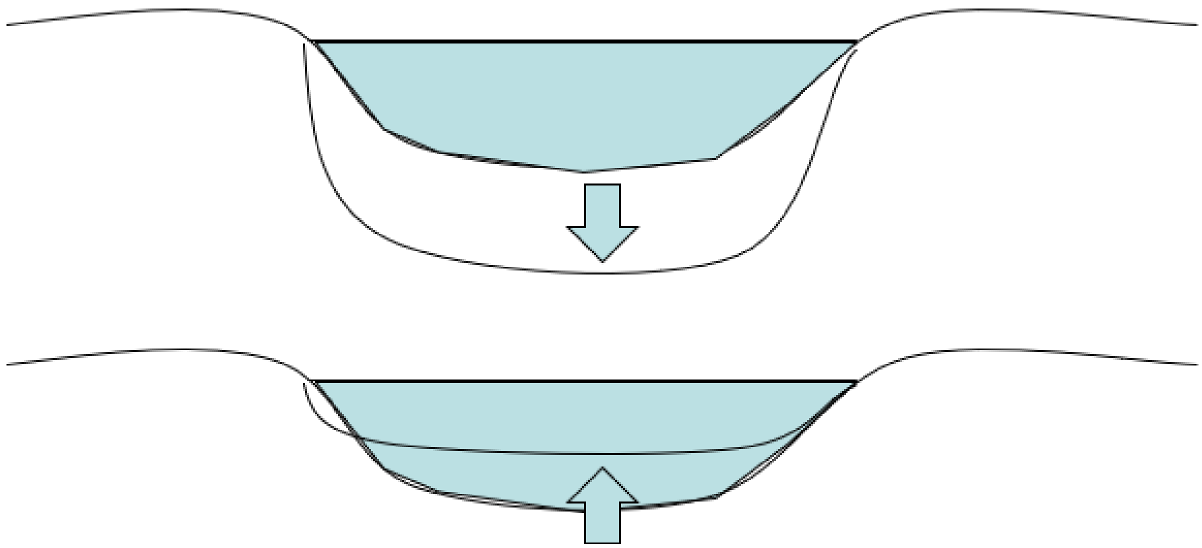

Figure 27. *Conceptual diagram of bathymetry modification in dredging/filling scenarios: dredging (top) and filling (bottom).*

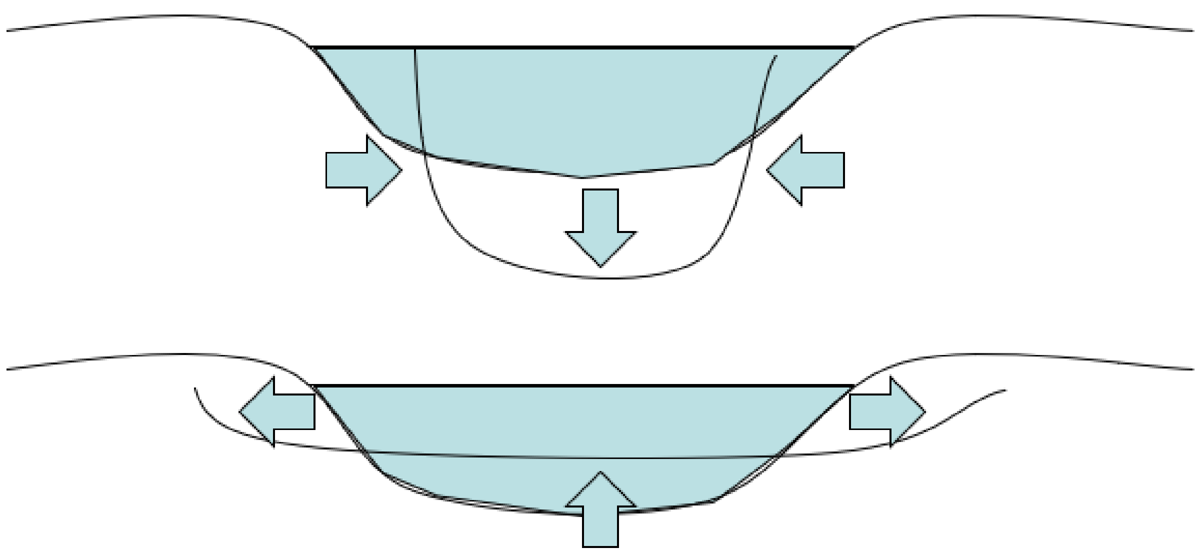

Figure 28. *Conceptual diagram of bathymetry modification in redesign scenarios: re-design "deep" (top) and re-design "shallow" (bottom).*

5.8.2 Results of Bathymetry Modification Scenarios

Sediment flux results of bathymetry modification are variable (Table 28). For the Langton pond, SRP release was decreased in both the Filled and Redesign Shallow scenarios presumably because the shallower water column exposed benthic sediments to higher oxygen. In the Shoreview and Alameda ponds, however, we saw the magnitude (mass) of SRP release decrease in the Redesign Deep scenarios and increase in the Redesign Shallow scenarios. These results suggest that the benefits realized in having the Shoreview and Alameda benthic sediments closer to the surface were overwhelmed by the additional SRP release that occurred in the Redesign Shallow scenarios, which greatly increased the pond surface areas and therefore the benthic sediment surface areas that releases SRP. The shallower scenarios would intuitively have promoted wind mixing and the impact thereof was seen in the stratification strength of the ponds being reduced in these scenarios (Table G-1). The impact to benthic DO concentrations, however, did not reflect this reduction (Table G-2) because any additional DO was consumed by the high sediment oxygen demand (Table 28). The Redesign Deep scenarios, which increased water column depth and greatly reduced the benthic sediment surface areas in these ponds, produced conditions that would be expected to result in higher per-area SRP release, i.e., increased stratification strength (RTRM) and a corresponding decrease in benthic DO (see Appendix G). Yet, the effect of reduced benthic surface area had a stronger impact and led to lower overall SRP release. The discrepancy between the behaviors of the Langton pond and Shoreview and Alameda ponds is likely greatly influenced by the fact that the Shoreview and Alameda ponds are strongly anoxic whereas the Langton pond is frequently oxic. The Minnetonka pond model became unstable when evaluating these remediation strategies, so no output values were able to be determined.

Table 28. *Sediment fluxes with changes in depth and pond redesign for bathometry modification scenarios.*

Shoreview Pond				
Scenario	**SRP Release** kg (lb)	**Δ SRP Release**	**DO Consumption** kg (lb)	**Δ DO Consumption**
Original	5.1 (11.3)	-	3600 (7900)	-
Filled	3.8 (8.5)	-25%	4200 (9200)	17%
Dredged	6.2 (13.6)	21%	3100 (6900)	-12%
Redesign Shallow	7.5 (16.5)	46%	8300 (18000)	132%
Redesign Deep	3.2 (7.1)	-37%	1400 (3200)	-59%
Alameda Pond				
Scenario	**SRP Release** kg (lb)	**Δ SRP Release**	**DO Consumption** kg (lb)	**Δ DO Consumption**
Original	2.0 (4.4)	-	1600 (3600)	-
Filled	1.5 (3.3)	-24%	2300 (5100)	41%
Dredged	2.4 (5.2)	19%	1100 (2400)	-32%
Redesign Shallow	2.9 (6.3)	45%	4900 (11000)	202%
Redesign Deep	1.2 (2.6)	-41%	460 (1000)	-72%
Langton Pond				
Scenario	**SRP Release** kg (lb)	**Δ SRP Release**	**DO Consumption** kg (lb)	**Δ DO Consumption**
Original	0.0035 (0.0078)	-	80 (180)	-
Filled	0.0018 (0.0040)	-49%	82 (180)	3%
Dredged	0.0095 (0.0209)	169%	91 (200)	14%
Redesign Shallow	0.0027 (0.0060)	-23%	170 (370)	109%
Redesign Deep	0.0058 (0.0127)	64%	44 (96)	-45%
Minnetonka Pond				
Scenario	**SRP Release** kg (lb)	**Δ SRP Release**	**DO Consumption** kg (lb)	**Δ DO Consumption**
Original	1.5 (3.3)	-	2100 (4700)	-
Filled	*	*	*	*
Dredged	*	*	*	*
Redesign Shallow	*	*	*	*
Redesign Deep	*	*	*	*

*Data unavailable due to model instability.

The degree to which surface and water column-averaged TP concentrations were affected by changes in bathymetry were variable (Tables 29 and 30). The scenarios that consistently produced increased TP concentrations were the Redesign Shallow scenarios, presumably because of the increased benthic sediment surface area from which SRP flux can occur combined with a shallower water column in which the released P is distributed. Conversely, the Dredging and Deep Redesign scenarios tended to produce lower surface and water column TP concentrations, likely due to a higher ratio of pond volume to sediment-released P compared to the baseline scenario (i.e., dilution). Interestingly, only in the Shoreview pond did a scenario of filling in (reduced volume + shallower depth), an assessment of the impacts of pond aging, produce higher in-pond TP concentrations. Effects of filling in on DO and RTRM were variable across the ponds (see Appendix G). For the Shoreview pond, DO increased and RTRM decreased in this scenario, which would be expected to reduce water column TP, rather than increase, as observed in the simulations. This suggests that the effects of filling in of older stormwater ponds may have wide-ranging effects.

Table 29. *Surface total phosphorus (TP) concentrations with changes in depth and pond redesign.*

Shoreview Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.350 ± 0.010	-	0.360	-
Filled	0.376 ± 0.009	7%	0.384	7%
Dredged	0.293 ± 0.008	-16%	0.297	-18%
Redesign Shallow	0.482 ± 0.016	38%	0.495	38%
Redesign Deep	0.266 ± 0.006	-24%	0.270	-25%
Alameda Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.223 ± 0.004	-	0.228	-
Filled	0.156 ± 0.001	-30%	0.158	-31%
Dredged	0.196 ± 0.003	-12%	0.198	-13%
Redesign Shallow	0.275 ± 0.007	24%	0.280	23%
Redesign Deep	0.186 ± 0.003	-16%	0.188	-18%
Langton Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.173 ± 0.002	-	0.177	-
Filled	0.156 ± 0.001	-10%	0.158	-11%
Dredged	0.170 ± 0.002	-2%	0.173	-2%
Redesign Shallow	0.196 ± 0.004	13%	0.198	12%
Redesign Deep	0.168 ± 0.002	-3%	0.171	-3%
Minnetonka Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.302 ± 0.009	-	0.311	-
Filled	*	*	*	*
Dredged	*	*	*	*
Redesign Shallow	*	*	*	*
Redesign Deep	*	*	*	*

*Data unavailable due to model instability.

Table 30. *Water column-averaged total phosphorus (TP) concentrations with changes in depth and pond redesign.*

Shoreview Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.345 ± 0.010	-	0.358	-
Filled	0.368 ± 0.009	7%	0.372	4%
Dredged	0.293 ± 0.008	-15%	0.302	-16%
Redesign Shallow	0.481 ± 0.016	39%	0.490	37%
Redesign Deep	0.251 ± 0.006	-27%	0.251	-30%
Alameda Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.211 ± 0.006	-	0.210	-
Filled	0.145 ± 0.008	-31%	0.145	-31%
Dredged	0.187 ± 0.003	-11%	0.186	-11%
Redesign Shallow	0.268 ± 0.006	27%	0.270	29%
Redesign Deep	0.164 ± 0.005	-22%	0.163	-22%
Langton Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.172 ± 0.006	-	0.175	-
Filled	0.180 ± 0.008	5%	0.184	5%
Dredged	0.169 ± 0.003	-2%	0.174	-1%
Redesign Shallow	0.197 ± 0.006	15%	0.201	15%
Redesign Deep	0.163 ± 0.005	-5%	0.162	-8%
Minnetonka Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.313 ± 0.009	-	0.323	-
Filled	*	*	*	*
Dredged	*	*	*	*
Redesign Shallow	*	*	*	*
Redesign Deep	*	*	*	*

*Data unavailable due to model instability.

Despite variable effects of bathymetry modifications on in-pond TP concentrations, TP export mass was reduced consistently under Dredged scenarios for all ponds (Table 31). We attribute this to the increased water volumes and subsequently increased water residence times that promote net phosphorus retention. As discussed earlier, the Redesign Deep scenarios improved TP retention for the Shoreview and Alameda ponds by reducing the benthic sediment surface areas that consistently release SRP under the frequently anoxic conditions. The

Langton pond, on the other hand, had improved TP retention under the Redesign Shallow scenario because of frequent oxic conditions promoted by the shallower pond. It is important to note, however, that our model did not account for sediment resuspension, emergent macrophyte growth in shallow water, or other factors that could have resulted in greater TP export under shallow pond conditions that what our model results suggest.

With respect to the Filled scenario, we observed a slight increase in exported TP in all three ponds (7% - 9%), despite variable and somewhat unintuitive impacts to pond TP concentrations, benthic DO, and stratification strength (RTRM). This result suggests that aging ponds, i.e., those accumulating sediment without a corresponding expansion in surface area, may pose a slight risk of reduced TP retention performance.

To place the phosphorus export of the remediation strategy on the same time frame as the costs, we observe that approximately 41% of the precipitation that fell between 1991 and 2020 occurred in the months June, July, and August according to data available from the US National Weather Service (https://www.weather.gov/wrh/climate?wfo=mpx). Roughly equating precipitation to export of phosphorus, then the approximately 3-month simulation TP export values represent 41% of the annual pond TP export. Table 32 provides 1-Year and 10-Year TP export values as rough approximations based on the assumption that cost per kg (lb) TP is a calculation of the 10-year scenario cost divided by the approximated 10-Year TP export mass reduction.

Table 31. *Total phosphorus (TP) export for bathymetry modification scenarios. TP Exports are calculated only for the approximately 3-month simulation period, while Total Costs include all anticipated costs for each remediation strategy over a 10-year period. Cost per % ∆ TP is a calculation of the total scenario cost divided by the percent improvement in TP export mass relative to the original condition. Cost per kg (lb) TP is a calculation of the total scenario cost divided by the TP export mass reduction in number of kg (lb).*

Shoreview Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	4.2 (9.4)	-	-	-
Filled	4.6 (10.0)	7%	$0	$0 ($0)
Dredged	3.3 (7.4)	-21%	$4,000	$93,000 ($42,000)
Redesign Shallow	4.5 (9.9)	5%	**	**
Redesign Deep	3.7 (8.1)	-13%	**	**
Alameda Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	2.9 (6.5)	-	-	-
Filled	3.1 (6.9)	7%	$0	$0 ($0)
Dredged	2.5 (5.5)	-15%	$5,600	$190,000 ($86,000)
Redesign Shallow	3.1 (6.8)	5%	**	**
Redesign Deep	2.7 (5.9)	-9%	**	**
Langton Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	0.045 (0.099)	-	-	-
Filled	0.049 (0.108)	9%	$0	$0 ($0)
Dredged	0.031 (0.068)	-31%	$1,400	$3,100,000 ($1,400,000)
Redesign Shallow	0.015 (0.033)	-67%	**	**
Redesign Deep	0.054 (0.119)	20%	**	**
Minnetonka Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	0.026 (0.058)	-	-	-
Filled	***	***	***	***
Dredged	***	***	***	***
Redesign Shallow	***	***	***	***
Redesign Deep	***	***	***	***

*Costs rounded to two significant figures.

**No data available.

***Data unavailable due to model instability.

Table 32. *Cost-effectiveness over a 10-year span for bathymetry modification scenarios*. All values are rounded to two significant figures.

Shoreview Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	4.2 (9.4)	10 (23)	100 (230)	-
Filled	4.6 (10.0)	11 (25)	110 (250)	$0 ($0)
Dredged	3.3 (7.4)	8.2 (18)	82 (180)	$3,800 ($1,700)
Redesign Shallow	4.5 (9.9)	11 (24)	110 (240)	*
Redesign Deep	3.7 (8.1)	9.0 (20)	90 (200)	*
Alameda Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	2.9 (6.5)	7.1 (16)	71 (160)	-
Filled	3.1 (6.9)	7.6 (17)	76 (170)	$0 ($0)
Dredged	2.5 (5.5)	6.1 (13)	61 (130)	$7,800 ($3,500)
Redesign Shallow	3.1 (6.8)	75 (17)	75 (170)	*
Redesign Deep	2.7 (5.9)	65 (14)	65 (140)	*
Langton Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	0.045 (0.099)	0.11 (0.24)	1.1 (2.4)	-
Filled	0.049 (0.108)	0.12 (0.26)	1.2 (2.6)	$0 ($0)
Dredged	0.031 (0.068)	0.076 (0.17)	0.76 (1.7)	$130,000 ($58,000)
Redesign Shallow	0.015 (0.033)	0.037 (0.081)	0.37 (0.81)	*
Redesign Deep	0.054 (0.119)	0.13 (0.29)	1.3 (2.9)	*
Minnetonka Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	0.026 (0.058)	0.064 (0.14)	0.64 (1.4)	-
Filled	**	**	**	**
Dredged	**	**	**	**
Redesign Shallow	**	**	**	**
Redesign Deep	**	**	**	**

*No data available.

**Data unavailable due to model instability.

5.9 Iron-Enhanced Sand Filter Bench Implementation

One means of reducing TP release from a pond is to install a pond-perimeter iron-enhanced sand filter (IESF) in the shallow-depth bench that often encircles stormwater ponds (Erickson et al. 2018). An IESF bench is designed to approximately 0.3 – 0.6 m (1 - 2 ft) below the original permanent pool elevation of a pond, allowing it to dry between storm events and filter water from the pond temporary storage volume between the original and new permanent pool elevations during storm events. The pond overflow structure will then be 0.3 – 0.6 m (1 – 2 ft) above the IESF pond-perimeter bench and will allow excess stormwater inflows to be diverted during larger storms. The Minnesota Stormwater Manual (MPCA 2022) specifies a 65% TP retention for these pond-perimeter benches (which are typically in the “tier 1” category).

5.9.1 Modeling Approach

Depending on the configuration of the outlet structure on a stormwater pond, the addition of a filter bench could impact the hydraulic behavior of a pond akin to the addition of an orifice outlet beneath the primary overflow weir. We used the Minnesota Stormwater Manual’s performance value of 65% TP removal, which accounts for both hydraulic and chemical components of an IESF bench. We applied this treatment rate to the outflow TP concentrations determined by our models for each pond. Of course, relying on this performance value still requires that the IESF must be properly designed and constructed and cannot be beyond the useful life of the media, corresponding to approximately 200 m (656 ft) of water having passed through the filter (Erickson et al. 2018). We approximated installation costs as $415.45/m^2$ ($38.60/ft^2$) based on data from previous projects of the Capitol Region Watershed District (Personal Communication with Bob Fossum, 2022). On top of this, we added a 15% cost for design and engineering and assumed that operation and maintenance would equate 50% of the construction cost over a 10-year lifespan. We sized each IESF bench to treat 1 ft (0.3 m) of ponding depth from the pond assuming the IESF could process a rate of 12-ft (3.7 m) of treatment depth over 48 hours (Erickson et al. 2007). The calculated values are shown in Table 33.

Table 33. *Iron-Enhanced Sand Filter Bench cost estimates. Total Costs include all anticipated costs over a 10-year period.*

Pond	Pond Surface Area m^2 (ac)	Treatment Volume m^3 (gal)	IESF Area m^2 (ft^2)	Total Cost*
Shoreview	11,700 (2.90)	3,600 (945,000)	980 (10,500)	$670,000
Alameda	11,700 (2.89)	3,600 (942,000)	970 (10,500)	$670,000
Langton	650 (0.16)	200 (52,000)	50 (580)	$37,000
Minnetonka	6,900 (1.7)	2,100 (554,000)	570 (6,200)	$393,000

*Costs rounded to two significant figures.

5.9.2 Results of Iron-Enhanced Sand Filter Bench Implementation

Filtration practices are greatly limited by the water volume and pollutant load they are intended to treat. In the case of pond-perimeter filtration benches, the treatment volume consists of the entire pond surface area multiplied by the ponding depth above the surface of the filtration bench. We sized an IESF bench to be able to treat 0.3 m (1 ft) of ponding depth in the study ponds so as to be able to provide approximately 65% of TP outflow reduction. The costs (Table 33) to treat this entire volume were extremely large, especially when considered with the anticipated reductions in TP export (Table 34). IESFs would be most cost-effective in ponds that were exporting large amounts of phosphorus or as a pretreatment practice used to reduce the pollutant load entering the ponds (see Watershed-Based Methods). We recognize that we were not able to optimize the size of the pond-perimeter trenches and that such an optimization may result in improved cost-effectiveness. For that reason we did not attempt to compare the results on a 10-year phosphorus export reduction basis, as the results of this remediation were not cost-effective.

Table 34. *Total phosphorus (TP) export for iron-enhanced sand filter bench implementation. TP Exports are calculated only for the approximately 3-month simulation period, while Total Costs include all anticipated costs for each remediation strategy over a 10-year period. Cost per % ∆ TP is a calculation of the total scenario cost divided by the percent improvement in TP export mass relative to the original condition. Cost per kg (lb) TP is a calculation of the total scenario cost divided by the TP export mass reduction in number of kg (lb).*

Shoreview Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	4.2 (9.4)	-	-	-
IESF Bench	1.5 (3.3)	-65%	$10,000	$240,000 ($110,000)
Alameda Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	2.9 (6.5)	-	-	-
IESF Bench	1.0 (2.3)	-65%	$10,000	$350,000 ($160,000)
Langton Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	0.045 (0.099)	-	-	-
IESF Bench	0.016 (0.035)	-65%	$570	$1,300,000 ($580,000)
Minnetonka Pond				
Scenario	**TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**Cost per kg (lb) TP***
Original	0.026 (0.058)	-	-	-
IESF Bench	0.009 (0.020)	-65%	$6,000	$23,000,000 ($10,000,000)

*Costs rounded to two significant figures.

Table 35. *Evaluation of cost-effectiveness over a 10-year span. 1-Year and 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 kilograms (pounds).* All values are rounded to two significant figures.

Shoreview Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	4.2 (9.4)	10 (23)	100 (230)	-
IESF Bench	1.5 (3.3)	3.6 (8.0)	36 (80)	$10,000 ($4,500)
Alameda Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	2.9 (6.5)	7.1 (16)	71 (160)	-
IESF Bench	1.0 (2.3)	2.5 (5.5)	25 (55)	$14,000 ($6,500)
Langton Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	0.045 (0.099)	0.11 (0.24)	1.1 (2.4)	-
IESF Bench	0.016 (0.035)	0.038 (0.084)	0.38 (0.84)	$52,000 ($24,000)
Minnetonka Pond				
Scenario	**Simulation TP Export** kg (lb)	**1-Year TP Export** kg (lb)	**10-Year TP Export** kg (lb)	**10-Year Cost per kg (lb) TP**
Original	0.026 (0.058)	0.064 (0.14)	0.64 (1.4)	-
IESF Bench	0.009 (0.020)	0.022 (0.050)	0.22 (0.50)	$950,000 ($430,000)

6 Conclusions and Recommendations

This modeling exercise has many limitations; as British statistician George E. P. Box (1987) explained, "*Remember that all models are wrong; the practical question is how wrong do they have to be to not be useful.*"

This report details the four modeled ponds: Alameda, Shoreview Commons, Langton Lake Upstream, and Minnetonka 849w. Although there are things that can be learned by this exercise, it would be helpful to similarly model additional ponds to help generalize results to other ponds with different characteristics and watershed contexts. Each pond will have a unique response to environmental stimuli. To reduce the individuality of the response to the important factors of this report, many of the fundamental parameters involved in creating a stormwater pond model were kept consistent across ponds as were parameters that would have required significant effort to calibrate. These were inlet and outlet structure elevations relative to normal water surface level, inlet and outlet structure dimensions, inflow concentrations (except when modified as a treatment), meteorological inputs, initial in-pond constituent concentrations, and wind sheltering coefficients (except when modified as a treatment). The parameters that were unique to each pond were inflow volumes (scaled by watershed impervious area), bathymetry, pond orientation, thickness of cross-sections, sediment oxygen demand, and sediment phosphorus release.

We simulated seven remediation scenarios for the four ponds: chemical treatment of sediments with either alum or elemental iron filings, outlet reorientation, mechanical mixing, reduced wind sheltering, watershed modifications to reduce volume and reduce concentration, bathymetry modifications, and iron-enhanced sand filter bench implementation. The impacts of the most cost-effective and realistic remediation scenarios were summarized in Table 36 for comparison, with costs approximated as the expected 10-year cost of each remediation strategy.

Table 36. *Summary of cost-effectiveness of the most successful and realistic scenarios of each remediation strategy for each modeled pond over a 10-year span. 1-Year and 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). All values are rounded to two significant figures.*

Alum Application				
Pond	**10-Year TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**10-Year Cost per kg (lb) TP***
Shoreview	81 (130)	-43%	$240	$230 ($100)
Alameda	54 (120)	-24%	$660	$920 ($420)
Langton	1.1 (2.4)	-2%	$530	$49,000 ($22,000)
Minnetonka	0.52 (1.1)	-19%	$270	$42,000 ($19,000)
Iron Filings Application				
Pond	**10-Year TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**10-Year Cost per kg (lb) TP***
Shoreview	59 (160)	-31%	$320	$310 ($140)
Alameda	59 (130)	-17%	$580	$820 ($370)
Langton	1.1 (2.4)	-1%	$3,000	$280,000 ($130,000)
Minnetonka	0.55 (1.2)	-14%	$530	$82,000 ($37,000)
Mechanical Aeration				
Pond	**10-Year TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**10-Year Cost per kg (lb) TP***
Shoreview	66 (140)	-37%	$700	$670 ($300)
Alameda	49 (110)	-56%	$670	$940 ($430)
Langton	1.1 (2.4)	-2%	$2,200	$200,000 ($92,000)
Minnetonka	0.48 (1.1)	-25%	$830	$130,000 ($59,000)
50% Watershed Volume Reduction				
Pond	**10-Year TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**10-Year Cost per kg (lb) TP***
Shoreview	42 (92)	-60%	$390	$380 ($170)
Alameda	31 (68)	-57%	$410	$580 ($260)
Langton	0.10 (0.22)	-91%	$17	$1,500 ($700)
Minnetonka	0.00 (0.00)	-100%	$48	$7,500 ($3,400)
50% Watershed Concentration Reduction				
Pond	**10-Year TP Export** kg (lb)	**∆ TP Export**	**Cost per % ∆ TP***	**10-Year Cost per kg (lb) TP***
Shoreview	92 (200)	-11%	$600	$8,300 ($3,800)
Alameda	61 (130)	-15%	$450	$9,100 ($4,100)
Langton	0.94 (2.1)	-14%	**	**
Minnetonka	0.63 (1.4)	-2%	**	**

*Costs rounded to two significant figures.

**Data not available for small watersheds.

The responses of each pond to the modeled remediation strategies were varied. In particular, we found that the Langton pond, which is relatively young and has accumulated little phosphorus-rich sediment so far, exports little TP and did not substantially improve under most remediation strategies. The Minnetonka pond also had very low TP export although it is older than the Langton pond; this is attributed to its low drainage area to pond area loading ratio resulting in very low outflow volume. Under the modeled wind sheltering reduction scenarios, the Langton pond saw a decrease in anoxic days owing to increased wind mixing. By comparison, the Alameda and Shoreview ponds were not greatly improved under the modeled wind sheltering reduction scenarios owing to substantial sediment oxygen demand and the topographic sheltering supplied by the pond banks. The Minnetonka pond, however, had the greatest reduction in export TP resulting from wind sheltering removal, potentially due to increased wind-facilitated evaporation. Modification of the pond outlet to pull water off the bottom was not successful because of insufficient flow-through, where the pond re-established stratification before pond withdrawal could have much of an effect.

The Shoreview and Alameda ponds behaved similarly and responded well to remediation strategies that targeted anoxic sediment release since both ponds had high anoxic sediment SRP flux values owing to their age and accumulation of phosphorus-rich sediments. For this reason, both ponds also responded well to the Dredging and Redesign Deep scenarios (Tables 29 through 32) of bathymetry modifications where the water column depth was increased to ~3-4 m (~10-13 ft) from typical pond depths of ~1.5-2 m (~5-6.5 ft); the result of this added depth was that the sediment TP release was less likely to reach the pond surface. The cost per mass released from the pond, however, was higher for dredging than many other alternatives.

Watershed-based methods (reducing inflow concentrations and volumes) were effective for all ponds, which was expected since the stormwater TP inflows were a major component of the overall TP mass balance in each pond. The Minnetonka pond had its TP export reduced to 0 kg (0 lb) because it did not experience any outflow under the 50% inflow scenario. Reducing inflow volumes led to increased TP concentrations in the ponds since constituents in the pond water were not as diluted by inflow volumes. 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. A modeling study by Small et al. (2019) demonstrated a similar impact to lake TP in nearby Lake Como (St. Paul, MN) in response to reduced inputs of runoff (volume) and P loading from the watershed. Reducing inflow TP concentrations without modifying inflow volumes reduced both in-pond concentrations and overall pond TP export in a more predictable way, although it may be less effective in ponds with smaller drainage area to surface area loading ratios such as the Minnetonka pond.

Cost per kg (lb) TP exported is a common metric for evaluating remediation strategies. Note that when the original TP mass values were low, the cost per kg (lb) TP export values were typically high. In such cases, evaluating the cost per % improvement in TP mass offered a normalized evaluation of the different remediation strategies. Looking at Table 36, the most cost-effective remediation strategies for the Shoreview pond appeared to be treating the sediments to reduce phosphorus release through either an iron filings application or an alum application and implementing watershed-based strategies that reduce pond inflow volumes. An application of mechanical aeration at the Shoreview pond could be performed in conjunction with sediment treatment, but would not be as effective without sediment treatment because the sediments were found to release phosphorus under both anaerobic and aerobic conditions. Treating sediments for the Alameda pond had less of a cost advantage over mechanical aeration, as the cost per kg of phosphorus removed from export was about equal for alum addition and mechanical aeration, with iron filings addition being somewhat cheaper; watershed-based volume reduction was again a viable option. As for the Langton pond, its minimal sediment phosphorus release and very small watershed made it an interesting case to compare against the other ponds. According to cost effectiveness, the preferred methods for the Langton pond were watershed improvements, although cost estimates were difficult for such a small watershed, whereas in-pond treatments were not expected to be cost-effective since the Langton pond did not experience very much sediment phosphorus release. Cost-effectiveness was similarly difficult to evaluate for the Minnetonka pond because of its low TP inflows and limited outflow volume and TP export; however, the pond did have substantial sediment phosphorus release. The most cost-effective strategy for the Minnetonka pond was watershed-based volume reduction, although it is important to note that under the modeled scenario, the Minnetonka pond did not experience any outflow at all during the simulation period; this terminal pond scenario could adversely impact the pond water quality and aquatic ecosystem.

The discussion in this report is based on stormwater ponds seen as runoff treatment technology, which performance is quantified on the basis of reduction in outflow phosphorus compared to inflow phosphorus. If a stormwater pond is also seen as an aesthetic amenity, however, then reduction in phosphorus concentration within the ponds themselves becomes an important factor, as estimated by the difference in mean TP concentration in the ponds. While Table 36 describes the cost-effectiveness of each remediation scenario relative to outflow TP reduction, Table 37 describes the same data but reports the change in TP concentration within the ponds and the cost-effectiveness of each remediation scenario in effecting these changes. This distinction between the two tables is highlighted by the example of the Minnetonka, where Table 37 shows that it can be cost-effectively be remediated as a neighborhood amenity (targeting in-pond TP concentration) as compared to its relative costliness when remediated as a stormwater treatment practice (marginal improvements to already low TP export being prohibitively expensive). Remediation recommendations for each stormwater pond will vary depending on its unique stresses, context, and performance goals, so it is helpful to examine options from multiple perspectives.

Table 37. *Summary of estimated cost-effectiveness in reducing surface TP concentration of the most successful and realistic scenarios of each remediation strategy for each modeled pond. The mean % differences in surface TP are used in this table.* All values are rounded to two significant figures.

Alum Application				
Pond	**TP (mg/L)**	**TP Conc. Reduction (%)**	**10-Year Cost**	**10-Year Cost per % TP Reduction ($/%)**
Shoreview	0.20	42%	$10,000	$290
Alameda	0.17	23%	$16,000	$700
Langton	0.17	2%	*	*
Minnetonka	0.20	35%	$5,200	$150
Iron Filings Application				
Pond	**TP (mg/L)**	**TP Conc. Reduction (%)**	**10-Year Cost**	**10-Year Cost per % TP Reduction ($/%)**
Shoreview	0.25	30%	$9,900	$330
Alameda	0.19	16%	$9,900	$620
Langton	0.17	1%	$3,900	$3,900
Minnetonka	0.23	25%	$7,300	$292
Mechanical Aeration				
Pond	**TP (mg/L)**	**TP Conc. Reduction (%)**	**10-Year Cost**	**10-Year Cost per % TP Reduction ($/%)**
Shoreview	0.16	54%	$26,000	$480
Alameda	0.16	30%	$21,000	$700
Langton	0.17	2%	$4,100	$2,100
Minnetonka	0.17	45%	$21,000	$470
50% Watershed Volume Reduction				
Pond	**TP (mg/L)**	**TP Conc. Reduction (%)**	**10-Year Cost**	**10-Year Cost per % TP Reduction ($/%)**
Shoreview	0.40	-14%	$23,000	-$1,700
Alameda	0.24	-7%	$23,000	-$3,200
Langton	0.18	-3%	$1,500	-$600
Minnetonka	0.32	-5%	$4,800	-$1,100
50% Watershed Concentration Reduction				
Pond	**TP (mg/L)**	**TP Conc. Reduction (%)**	**10-Year Cost**	**10-Year Cost per % TP Reduction ($/%)**
Shoreview	0.31	11%	$95,000	$8,900
Alameda	0.19	14%	$95,000	$6,800
Langton	0.15	14%	*	*
Minnetonka	0.29	4%	*	*

*Data not available for small watersheds.

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Appendix A

Pond Model Calibration

Appendix A. Pond Model Calibration

Chloride Concentration

Despite the extensive efforts made to accurately represent observed chloride concentrations in the field, the capabilities of the model seem somewhat limited (Figures A-1a and A-1b). The current model accurately depicts the presence or absence of chloride stratification through much of the year. However, the specific chloride concentrations and magnitude of chloride stratification are not accurate in the early summer months. This could be caused by the vertical numerical diffusion in the model, which is a function of the vertical grid spacing. Still, this is useful information for predicting anoxic conditions at the sediment-water interface that could cause sediment phosphorus release.

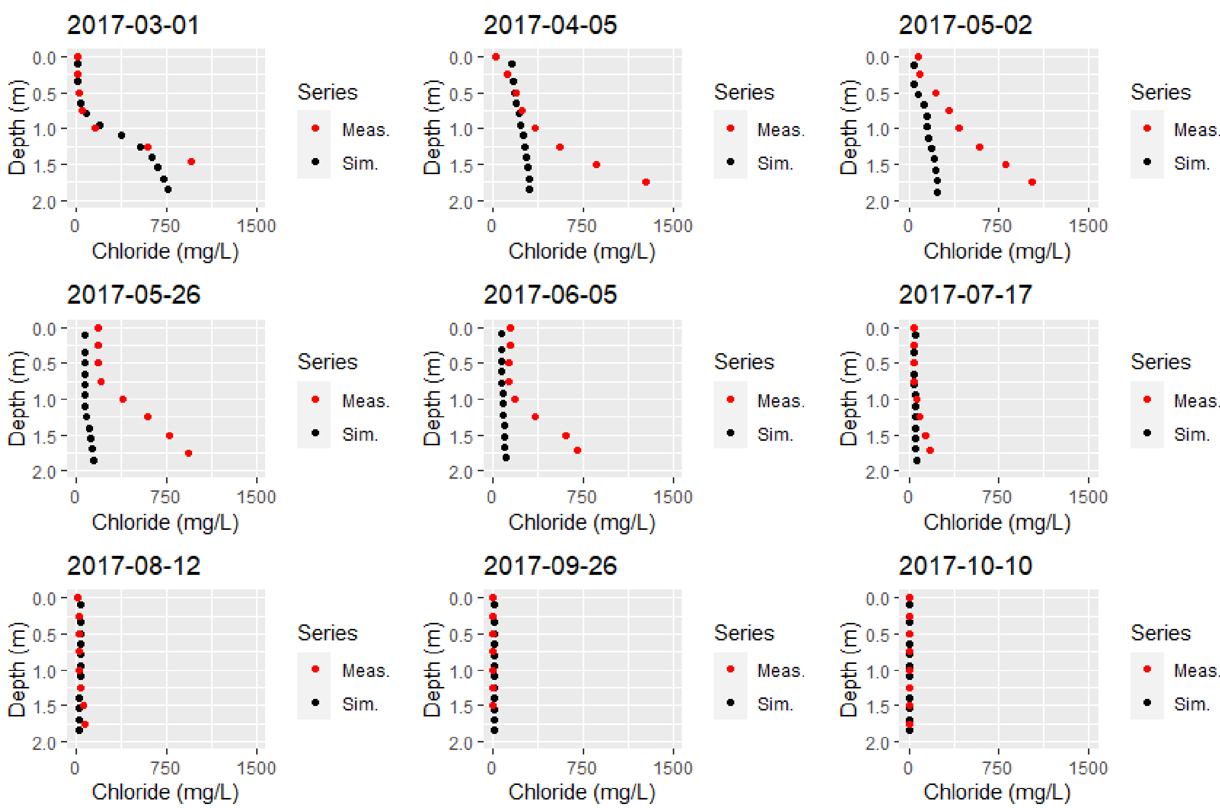

Figure A-1a. *Comparison of measured chloride concentration profiles to corresponding chloride concentration profiles from model output. Part 1 of 2.*

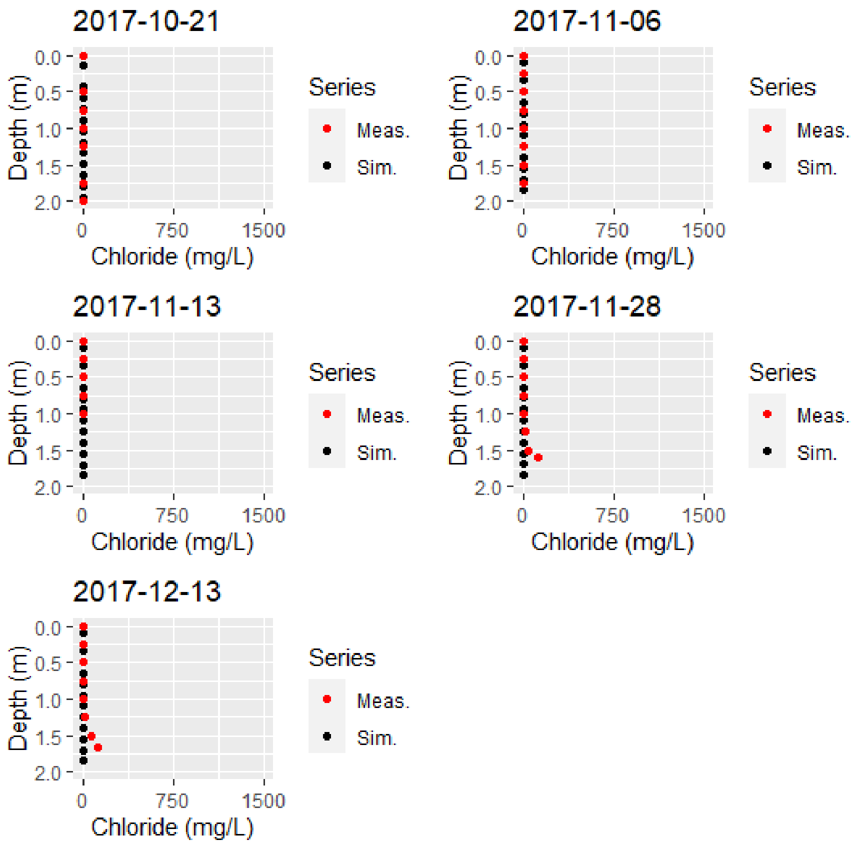

Figure A-1b. *Comparison of measured chloride concentration profiles to corresponding chloride concentration profiles from model output. Part 2 of 2.*

Dissolved Oxygen

The primary modeling goal with respect to dissolved oxygen is to identify the presence or absence of anoxic conditions at the sediment-water interface, as this strongly influences phosphorus dynamics within freshwater bodies. The current model does appear to accurately capture the lasting anoxic conditions along the sediment-water interface of the target pond (Figures A-2a and A-2b). One limitation that was encountered was that the model currently predicts substantial reaeration of the upper water column due to exposure to atmospheric oxygen. This does not match our field measurements, which often showed anoxic conditions throughout the water column. One potential remedy would be to increase the biological oxygen demand in the model pond water so as to negate any reaeration, but much of the resistance to reaeration in the field is likely due to the substantial free-floating macrophyte (FFM) cover observed on the Alameda pond. During the warm-season months, the FFM forms a thick mat and appears to act as a membrane that limits gas exchange between the water surface and the atmosphere.

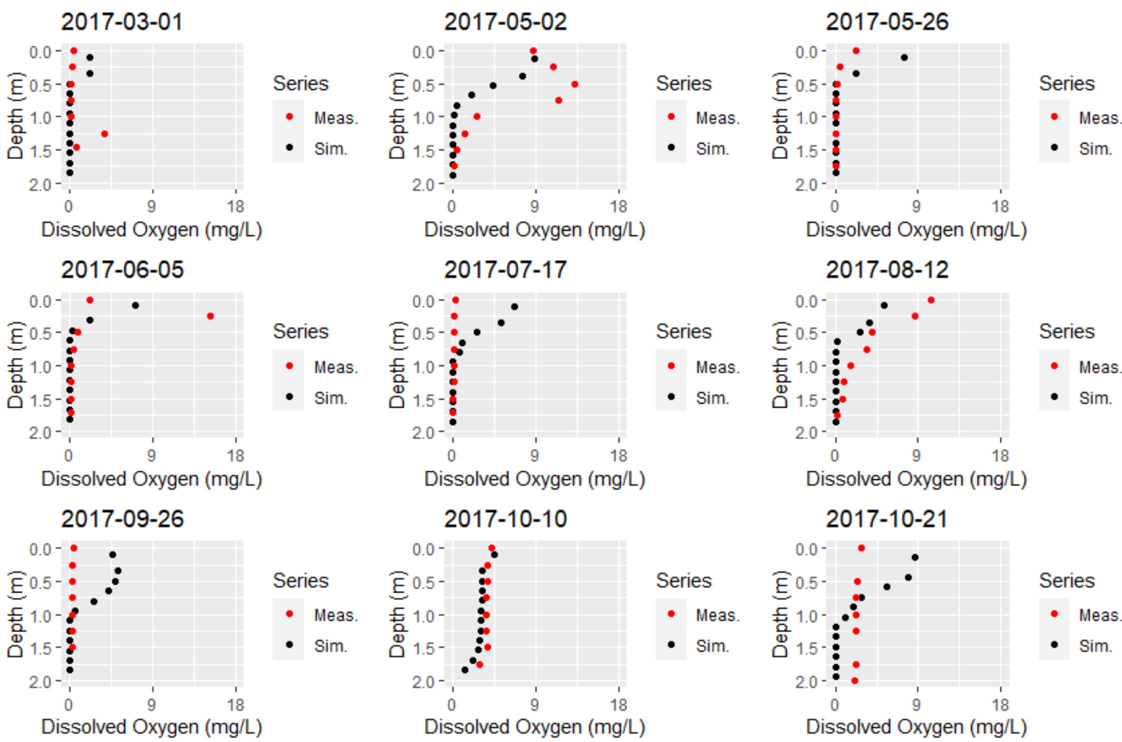

Figure A-2a. *Comparison of measured dissolved oxygen concentration profiles to corresponding dissolved oxygen concentration profiles from model output. Part 1 of 2.*

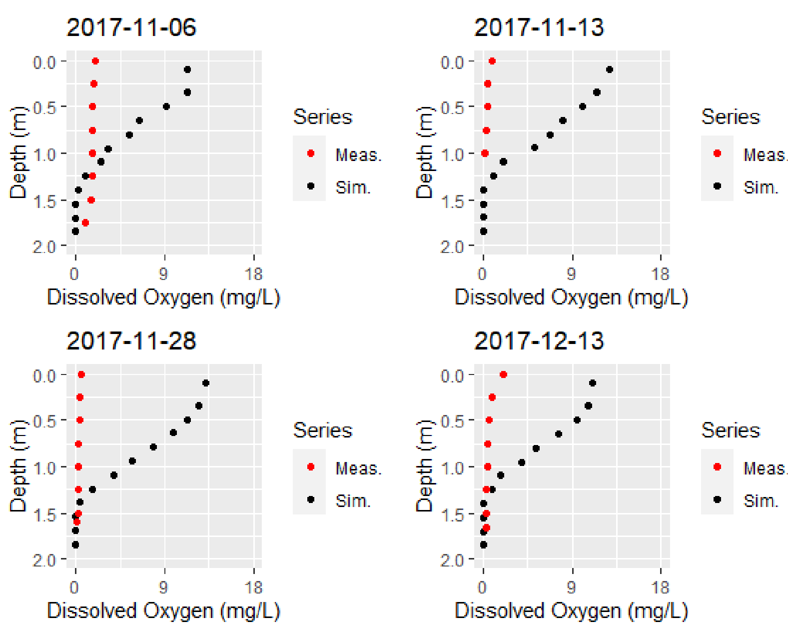

Figure A-2b. *Comparison of measured dissolved oxygen concentration profiles to corresponding dissolved oxygen concentration profiles from model output. Part 2 of 2.*

Soluble Reactive Phosphorus

Soluble reactive phosphorus (SRP) measurements in the field are extremely sensitive to in situ conditions, sampling methods, and analytical procedures. As such, individual SRP measurements are not as informative as long term patterns of observations. The current model appears to predict SRP concentrations reasonably well (Figures A-3a and A-3b), and occasionally both overpredicts and underpredicts specific values.

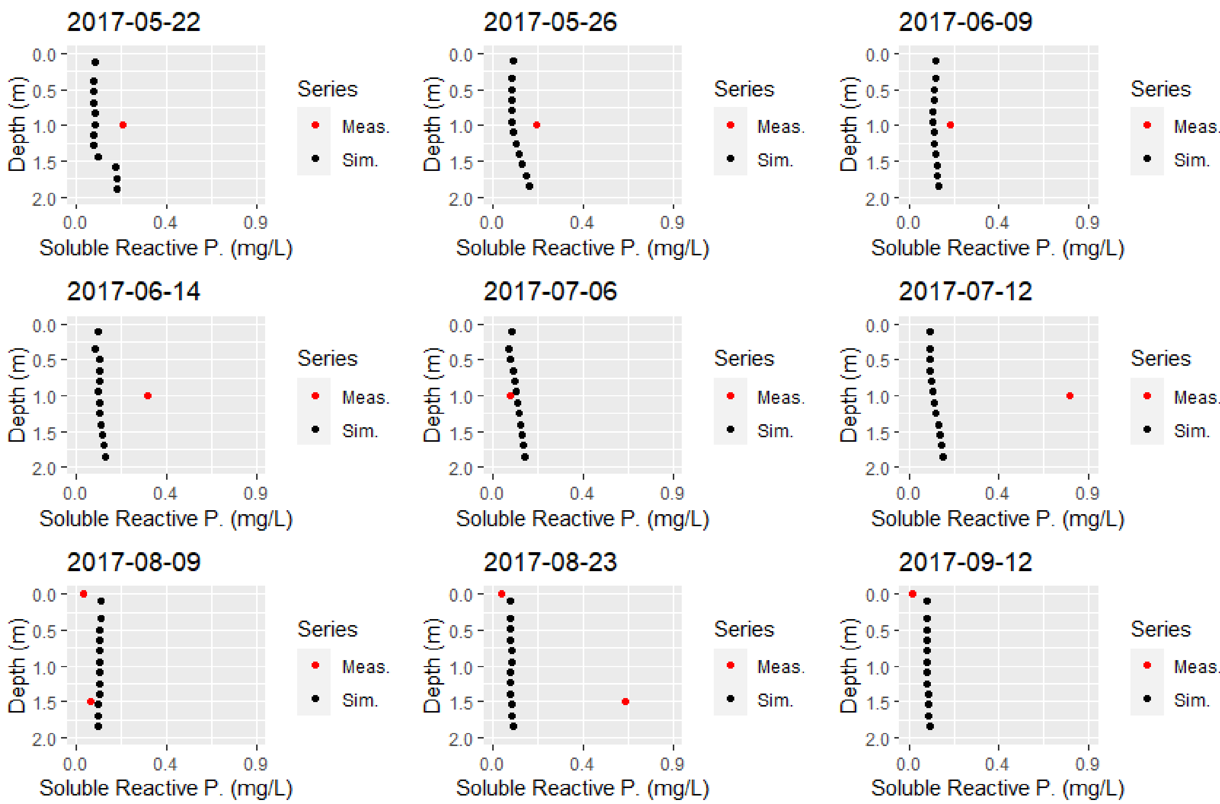

Figure A-3a. *Comparison of measured soluble reactive phosphorus concentration profiles to corresponding soluble reactive phosphorus concentration profiles from model output. Part 1 of 2.*

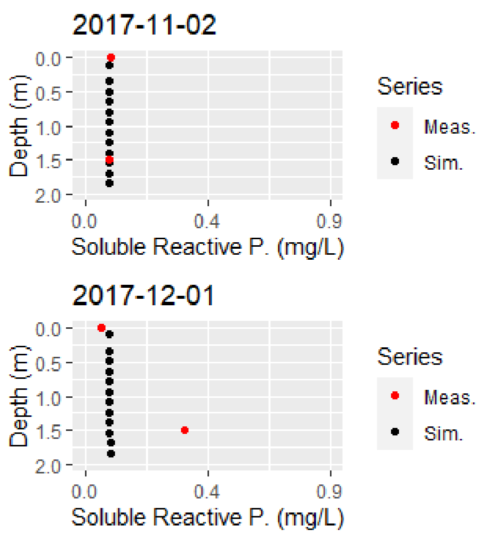

Figure A-3b. *Comparison of measured soluble reactive phosphorus concentration profiles to corresponding soluble reactive phosphorus concentration profiles from model output. Part 2 of 2.*

Total Phosphorus

Total phosphorus (TP) field measurements are not quite as sensitive as SRP measurements because TP includes SRP and various other forms of phosphorus, and there is no sink of TP. Soluble reactive phosphorus, on the other hand, can change form for various chemical and biological reasons that may increase or decrease measured concentrations while TP concentrations remain unchanged. Still, TP concentrations in a pond remain variable over time and should primarily be used to evaluate long term patterns of observations. The model appears to predict TP concentrations well (Figure A-4), but it appears to occasionally underpredict measured TP concentrations substantially (see May 22nd and July 12th in Figure A-4). This could also be a sampling problem, as much of the total phosphorus in mid-summer is contained in the floating vegetation of Alameda Pond, and is challenging to sample accurately.

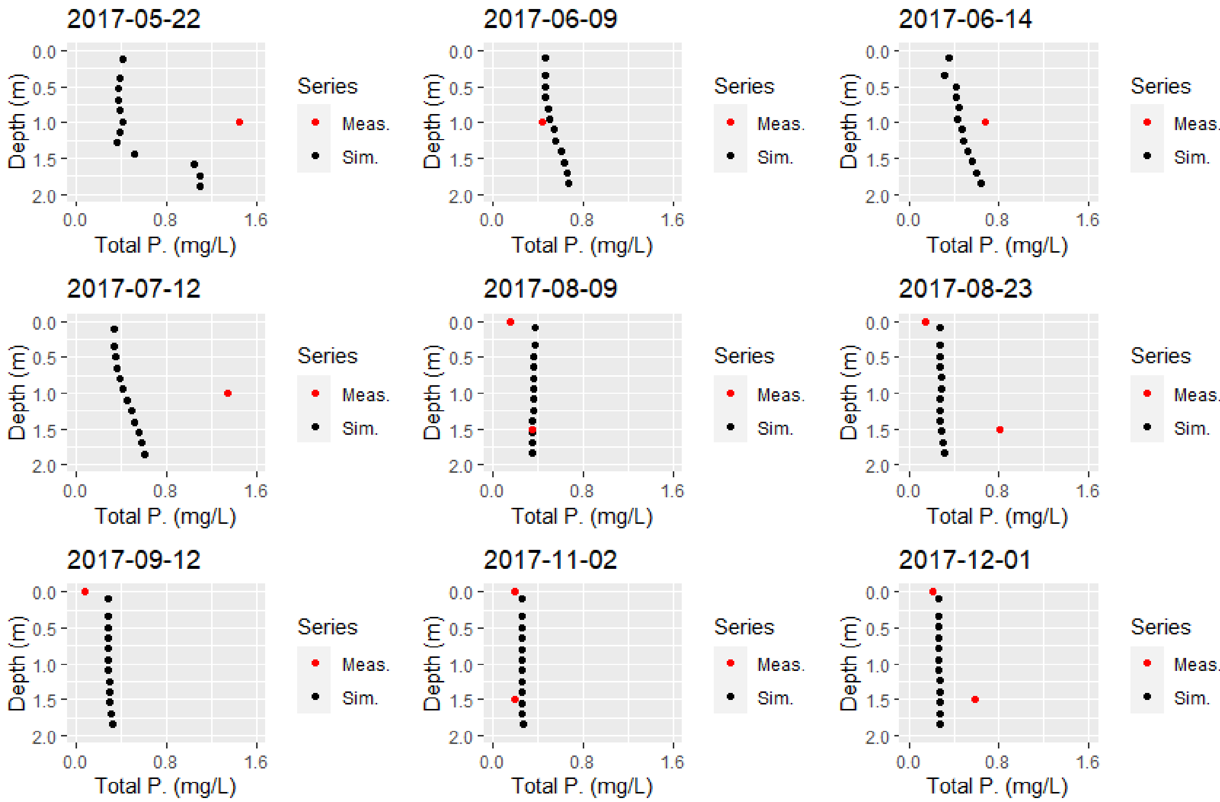

Figure A-4. *Comparison of measured total phosphorus concentration profiles to corresponding total phosphorus concentration profiles from model output.*

Summary of Calibration Results

The calibration results are summarized in Table A-1. Temperature, dissolved oxygen and TP were represented well, with two dates. Soluble reactive phosphorus is an intermediate compound that is used in plant growth, so it quickly enters the tissue of suspended and floating plants. It, therefore, is variable and notoriusly difficult to predict. Chloride was also not predicted throughout the season well, but is a conservative substance and not subject to similar variability. We found that the problem with chloride concentrations is that there was more vertical numerical diffusion in the pond model than the diffusion that physically occurs in the pond. However, the stratification dynamics were well represented even with this numerical diffusion, so we did not pursue more accurate simulation of chloride concentrations further.

Table A-1. *Summary of model calibration.*

Constituent	Performance	Notes
Temperature	Good approximation of temperatures. Good representation of presence/absence of stratification.	
Chloride	Poor approximation of concentrations. Reasonable approximation of presence/absence of stratification.	Exceptional chloride gradients in stormwater ponds challenge the capabilities of CE-QUAL-W2.
Dissolved Oxygen	Excellent approximation of benthic concentrations. Poor approximation of surface concentrations.	Surface concentrations are less important for this modeling effort but may be improvable.
Soluble Reactive Phosphorus	Reasonable approximation of concentrations.	Field measurements are highly sensitive and variable.
Total Phosphorus	Reasonable approximation of concentrations.	Greatly underpredicted on May 22nd and July 12th.

Appendix B

Pond Model Verification

Appendix B. Pond Model Verification

Once the model had been calibrated against the most complete set of field observation measurements in the Alameda pond, the model was configured for the Shoreview pond and verified against the next-most complete set of field data. The purpose of this verification step was to assess whether the pond model was accurately predicting pond conditions based on the input data it was given. There is always the possibility of "over-fitting" the model in the calibration step such that the model is only accurate under a narrow range of conditions and is not useful under more general conditions; over-fitting the model should be avoided for the model to be useful in meeting the project goals. In this case, the flexibility of the model was evaluated by applying it to a different pond.

Temperature and Thermostratification

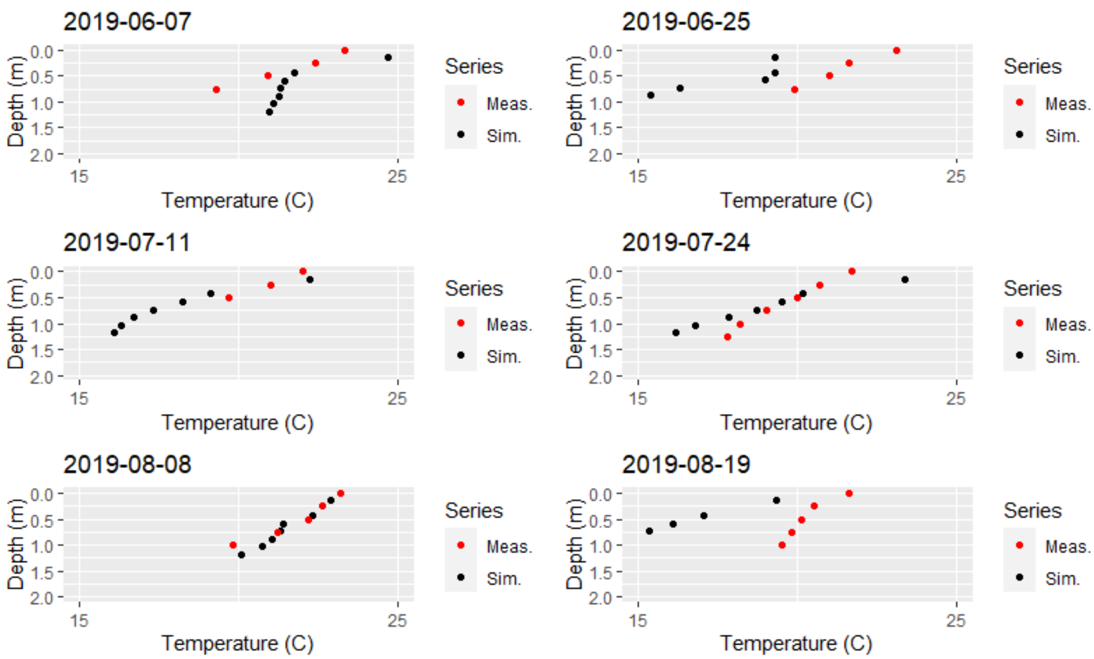

Figure B-1. *Comparison of measured temperature profiles to corresponding temperature profiles from model output.*

Chloride and Chemostratification

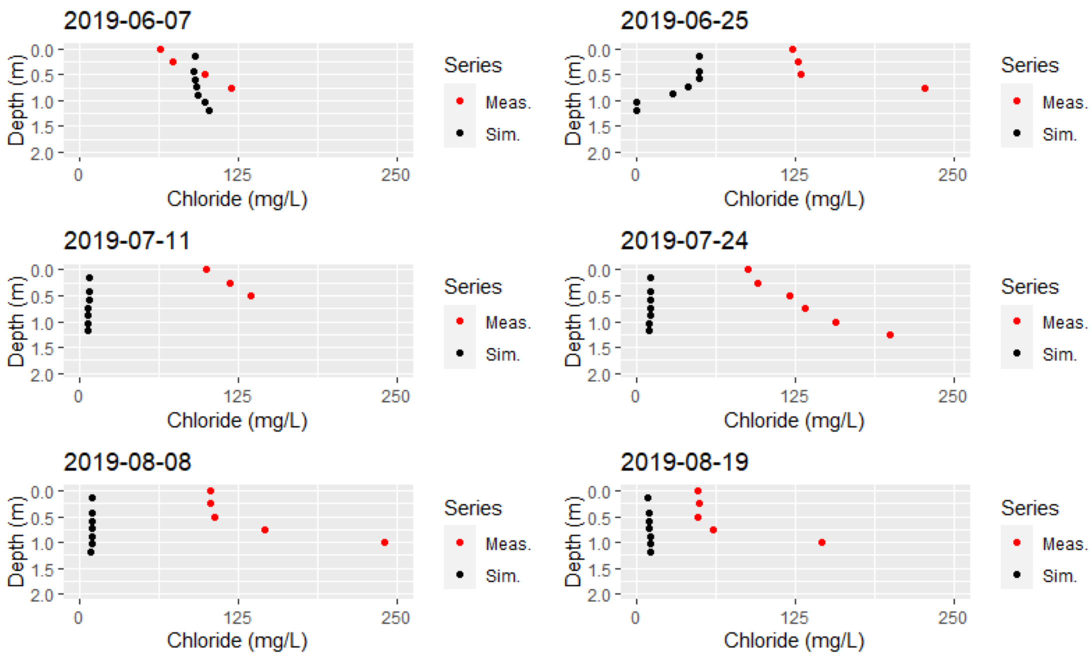

Figure B-2. *Comparison of chloride concentration profiles derived from electrical conductance measurements to corresponding chloride concentration profiles from model output.*

Dissolved Oxygen and Anoxia

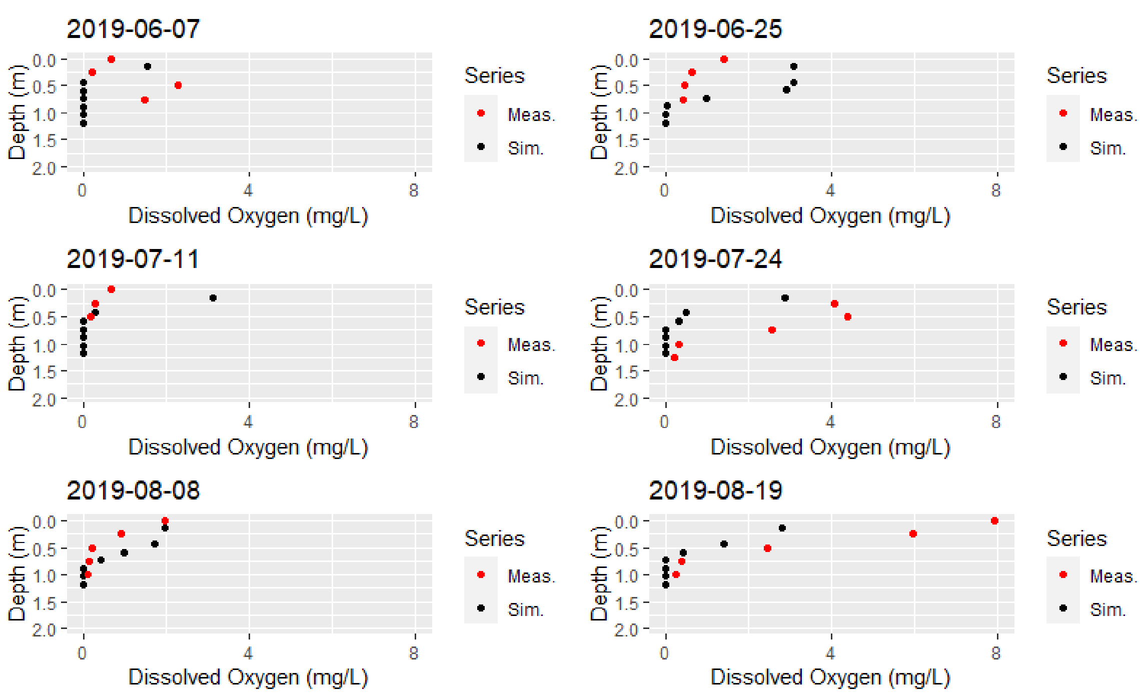

Figure B-3. *Comparison of measured dissolved oxygen concentration profiles to corresponding dissolved oxygen concentration profiles from model output.*

Soluble Reactive Phosphorus

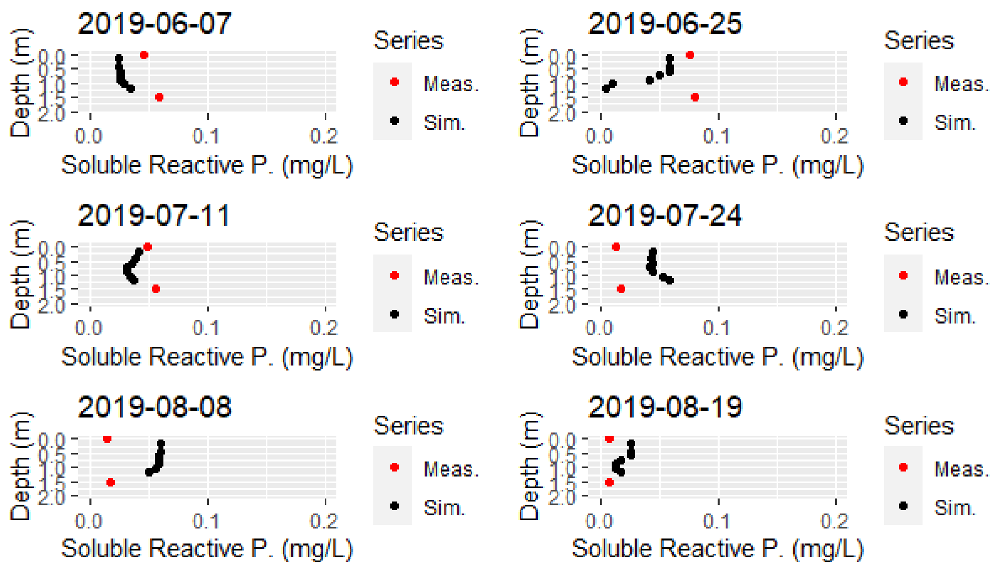

Figure B-4. *Comparison of measured soluble reactive phosphorus concentration profiles to corresponding soluble reactive phosphorus concentration profiles from model output.*

Total Phosphorus

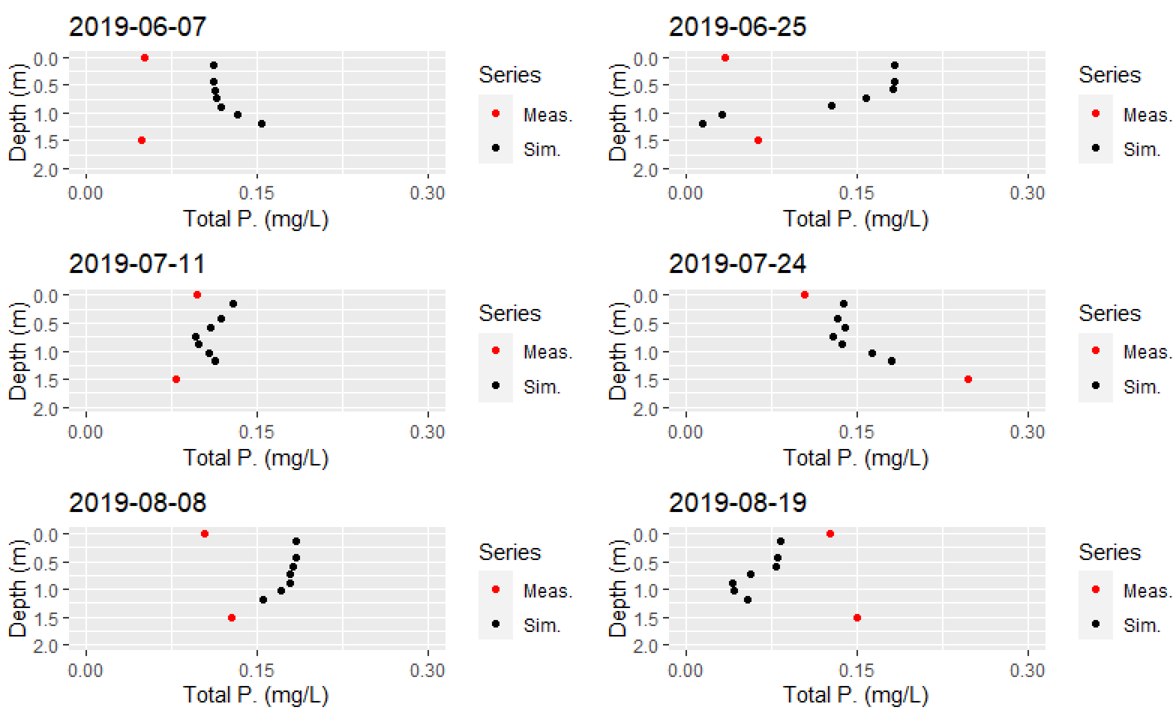

Figure B-5. *Comparison of measured total phosphorus concentration profiles to corresponding total phosphorus concentration profiles from model output.*

Table B-1. *Summary of model verification.*

Constituent	Performance
Temperature	Reasonable approximation of temperatures and stratification strength (slope).
Chloride	Accuracy quickly depreciated. Chloride did not remain in the pond system as long as in field observations.
Dissolved Oxygen	Good approximation of dissolved oxygen concentrations and anoxia presence.
Soluble Reactive Phosphorus	Good but variable accuracy in approximating soluble reactive phosphorus concentrations.
Total Phosphorus	Good but variable accuracy in approximating total phosphorus concentrations.

Appendix C

Chemical Treatment of Sediments

Appendix C. Chemical Treatment of Sediments

Table C-1. *Results for sediment fluxes.*

Shoreview Pond				
Scenario	**SRP Release** kg	**∆ SRP Release**	**DO Consumption** kg	**∆ DO Consumption**
Original	5.1	-	3600	-
Alum Application	1.2	-77%	3600	0%
Iron Filings Application	2.3	-55%	3600	0%
No Sediment Release	0.0	-100%	3600	0%
Alameda Pond				
Scenario	**SRP Release** kg	**∆ SRP Release**	**DO Consumption** kg	**∆ DO Consumption**
Original	2.0	-	1600	-
Alum Application	0.45	-77%	1600	0%
Iron Filings Application	0.91	-54%	1600	0%
No Sediment Release	0.0	-100%	1600	0%
Langton Pond				
Scenario	**SRP Release** kg	**∆ SRP Release**	**DO Consumption** kg	**∆ DO Consumption**
Original	0.0035	-	80	-
Alum Application	0.00080	-77%	80	0%
Iron Filings Application	0.0016	-55%	80	0%
No Sediment Release	0.0000	-100%	80	0%
Minnetonka Pond				
Scenario	**SRP Release** kg	**∆ SRP Release**	**DO Consumption** kg	**∆ DO Consumption**
Original	1.5	-	2100	-
Alum Application	0.35	-77%	2100	0%
Iron Filings Application	0.67	-55%	2100	0%
No Sediment Release	0.00	-100%	2100	0%

Table C-2. *Results for water column-averaged total phosphorus (TP) concentrations.*

Shoreview Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.345 ± 0.010	-	0.358	-
Alum Application	0.198 ± 0.003	-43%	0.202	-43%
Iron Filings Application	0.241 ± 0.005	-30%	0.247	-31%
No Sediment Release	0.155 ± 0.005	-55%	0.155	-57%
Alameda Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.211 ± 0.004	-	0.210	-
Alum Application	0.160 ± 0.002	-24%	0.158	-25%
Iron Filings Application	0.175 ± 0.003	-17%	0.171	-18%
No Sediment Release	0.145 ± 0.002	-31%	0.145	-31%
Langton Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.172 ± 0.002	-	0.175	-
Alum Application	0.169 ± 0.002	-2%	0.171	-2%
Iron Filings Application	0.170 ± 0.002	-1%	0.172	-2%
No Sediment Release	0.168 ± 0.002	-3%	0.170	-3%
Minnetonka Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.302 ± 0.009	-	0.311	-
Alum Application	0.197 ± 0.004	-35%	0.202	-35%
Iron Filings Application	0.227 ± 0.005	-25%	0.233	-25%
No Sediment Release	0.165 ± 0.002	-45%	0.168	-46%

Appendix D

Reorientation of Outlet Works

Appendix D. Reorientation of Outlet Works

Table D-1. *Reorientation of Outlet Works: Results for sediment fluxes.*

Shoreview Pond				
Scenario	**SRP Release** kg	**Δ SRP Release**	**DO Consumption** kg	**Δ DO Consumption**
Original	5.1	-	3600	-
Center	5.1	0%	3600	0%
Bottom	5.1	0%	3600	0%
Alameda Pond				
Scenario	**SRP Release** kg	**Δ SRP Release**	**DO Consumption** kg	**Δ DO Consumption**
Original	2.0	-	1600	-
Center	2.0	0%	1600	0%
Bottom	2.0	0%	1600	0%
Langton Pond				
Scenario	**SRP Release** kg	**Δ SRP Release**	**DO Consumption** kg	**Δ DO Consumption**
Original	0.0035	-	80	-
Center	0.0035	0%	80	0%
Bottom	0.0035	0%	80	0%
Minnetonka Pond				
Scenario	**SRP Release** kg	**Δ SRP Release**	**DO Consumption** kg	**Δ DO Consumption**
Original	1.5	-	2100	-
Center	1.5	0%	2100	0%
Bottom	1.5	0%	2100	0%

Table D-2. *Reorientation of Outlet Works: Results for surface total phosphorus (TP) concentrations.*

Shoreview Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.350 ± 0.010	-	0.360	-
Center	0.350 ± 0.010	0%	0.360	0%
Bottom	0.350 ± 0.010	0%	0.360	0%
Alameda Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.223 ± 0.004	-	0.228	-
Center	0.223 ± 0.004	0%	0.228	0%
Bottom	0.223 ± 0.004	0%	0.228	0%
Langton Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.173 ± 0.002	-	0.177	-
Center	0.173 ± 0.002	0%	0.177	0%
Bottom	0.173 ± 0.002	0%	0.177	0%
Minnetonka Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.302 ± 0.009	-	0.311	-
Center	0.302 ± 0.009	0%	0.311	0%
Bottom	0.302 ± 0.009	0%	0.311	0%

Table D-3. *Reorientation of Outlet Works: Results for water column-averaged total phosphorus (TP) concentrations.*

Shoreview Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.345 ± 0.010	-	0.358	-
Center	0.345 ± 0.010	0%	0.358	0%
Bottom	0.345 ± 0.010	0%	0.358	0%
Alameda Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.211 ± 0.004	-	0.210	-
Center	0.211 ± 0.004	0%	0.210	0%
Bottom	0.211 ± 0.004	0%	0.210	0%
Langton Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.172 ± 0.002	-	0.175	-
Center	0.172 ± 0.002	0%	0.175	0%
Bottom	0.172 ± 0.002	0%	0.175	0%
Minnetonka Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.313 ± 0.009	-	0.323	-
Center	0.313 ± 0.009	0%	0.323	0%
Bottom	0.313 ± 0.009	0%	0.323	0%

Table D-4. *Reorientation of Outlet Works: Results for relative thermal resistance to mixing (RTRM).*

Shoreview Pond				
Scenario	**Mean RTRM**	**∆ Mean RTRM**	**Median RTRM**	**∆ Median RTRM**
Original	0.080 ± 0.006	-	0.067	-
Center	0.080 ± 0.006	0%	0.067	0%
Bottom	0.080 ± 0.006	0%	0.067	0%
Alameda Pond				
Scenario	**Mean RTRM**	**∆ Mean RTRM**	**Median RTRM**	**∆ Median RTRM**
Original	0.108 ± 0.007	-	0.094	-
Center	0.108 ± 0.007	0%	0.094	0%
Bottom	0.108 ± 0.007	0%	0.094	0%
Langton Pond				
Scenario	**Mean RTRM**	**∆ Mean RTRM**	**Median RTRM**	**∆ Median RTRM**
Original	0.071 ± 0.006	-	0.057	-
Center	0.071 ± 0.006	0%	0.056	0%
Bottom	0.071 ± 0.006	0%	0.057	0%
Minnetonka Pond				
Scenario	**Mean RTRM**	**∆ Mean RTRM**	**Median RTRM**	**∆ Median RTRM**
Original	0.105 ± 0.008	-	0.092	-
Center	0.106 ± 0.008	0%	0.092	0%
Bottom	0.106 ± 0.008	0%	0.092	0%

Table D-5. *Reorientation of Outlet Works: Results for benthic dissolved oxygen (DO) concentrations.*

Shoreview Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	0.164 ± 0.061	-	0.000	-
Center	0.165 ± 0.061	0%	0.000	0%
Bottom	0.165 ± 0.061	0%	0.000	0%
Alameda Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	0.177 ± 0.076	-	0.000	-
Center	0.177 ± 0.076	0%	0.000	0%
Bottom	0.177 ± 0.076	0%	0.000	0%
Langton Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	2.205 ± 0.224	-	1.796	-
Center	2.205 ± 0.224	0%	1.795	0%
Bottom	2.205 ± 0.224	0%	1.795	0%
Minnetonka Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	0.055 ± 0.225	-	0.000	-
Center	0.055 ± 0.224	0%	0.000	0%
Bottom	0.055 ± 0.225	0%	0.000	0%

Table D-6. *Reorientation of Outlet Works: Results for water column-averaged dissolved oxygen (DO) concentrations.*

Shoreview Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	1.445 ± 0.044	-	1.497	-
Center	1.446 ± 0.044	0%	1.496	0%
Bottom	1.446 ± 0.044	0%	1.496	0%
Alameda Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	1.606 ± 0.057	-	1.725	-
Center	1.608 ± 0.057	0%	1.730	0%
Bottom	1.607 ± 0.057	0%	1.730	0%
Langton Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	4.248 ± 0.135	-	4.408	-
Center	4.248 ± 0.135	0%	4.408	0%
Bottom	4.248 ± 0.135	0%	4.408	0%
Minnetonka Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	0.620 ± 0.031	-	0.580	-
Center	0.609 ± 0.031	-2%	0.562	-3%
Bottom	0.620 ± 0.031	0%	0.580	0%

Table D-7. *Reorientation of Outlet Works: Results for anoxic days.*

Scenario	Days No Oxia	∆ Days No Oxia	Days Any Anoxia	∆ Days Any Anoxia
Shoreview Pond				
Original	92	-	99	-
Center	92	0%	99	0%
Bottom	92	0%	99	0%
Alameda Pond				
Scenario	**Days No Oxia**	**∆ Days No Oxia**	**Days Any Anoxia**	**∆ Days Any Anoxia**
Original	88	-	99	-
Center	88	0%	99	0%
Bottom	88	0%	99	0%
Langton Pond				
Scenario	**Days No Oxia**	**∆ Days No Oxia**	**Days Any Anoxia**	**∆ Days Any Anoxia**
Original	33	-	84	-
Center	33	0%	84	0%
Bottom	33	0%	84	0%
Minnetonka Pond				
Scenario	**Days No Oxia**	**∆ Days No Oxia**	**Days Any Anoxia**	**∆ Days Any Anoxia**
Original	99	-	99	-
Center	99	0%	99	0%
Bottom	99	0%	99	0%

Appendix E

Wind Sheltering Reduction

Appendix E. Wind Sheltering Reduction

Table E-1. *Results for sediment fluxes.*

Shoreview Pond				
Scenario	**SRP Release** kg	**∆ SRP Release**	**DO Consumption** kg	**∆ DO Consumption**
Original	5.1	-	3600	-
50% Reduction	5.0	-2%	3600	2%
100% Reduction	4.8	-5%	3700	5%
Case A	5.1	-1%	3600	1%
Case B	5.1	-1%	3600	1%
Case C	5.0	-2%	3600	2%
Alameda Pond				
Scenario	**SRP Release kg**	**∆ SRP Release**	**DO Consumption kg**	**∆ DO Consumption**
Original	2.0	-	1600	-
50% Reduction	2.0	-2%	1700	5%
100% Reduction	1.9	-6%	1800	12%
Langton Pond				
Scenario	**SRP Release kg**	**∆ SRP Release**	**DO Consumption kg**	**∆ DO Consumption**
Original	0.0035	-	80	-
50% Reduction	0.0033	-7%	80	0%
100% Reduction	0.0030	-16%	80	1%
Minnetonka Pond				
Scenario	**SRP Release kg**	**∆ SRP Release**	**DO Consumption kg**	**∆ DO Consumption**
Original	1.5	-	2100	-
50% Reduction	1.4	-4%	2100	2%
100% Reduction	1.4	-9%	2200	4%

Table E-2. *Wind sheltering reduction: Results for surface total phosphorus (TP) concentrations.*

Shoreview Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.350 ± 0.010	-	0.360	-
50% Reduction	0.346 ± 0.010	-1%	0.356	-1%
100% Reduction	0.341 ± 0.010	-3%	0.351	-3%
Case A	0.349 ± 0.010	0%	0.359	0%
Case B	0.349 ± 0.010	0%	0.359	0%
Case C	0.348 ± 0.010	-1%	0.358	-1%
Alameda Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.223 ± 0.004	-	0.228	-
50% Reduction	0.222 ± 0.004	0%	0.226	-1%
100% Reduction	0.220 ± 0.004	-1%	0.224	-2%
Langton Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.173 ± 0.002	-	0.177	-
50% Reduction	0.174 ± 0.002	0%	0.177	0%
100% Reduction	0.174 ± 0.002	0%	0.177	0%
Minnetonka Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.302 ± 0.009	-	0.311	-
50% Reduction	0.299 ± 0.009	-1%	0.307	-1%
100% Reduction	0.293 ± 0.009	-3%	0.300	-4%

Table E-3. *Wind sheltering reduction: Results for water column-averaged total phosphorus (TP) concentrations.*

Shoreview Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.345 ± 0.010	-	0.358	-
50% Reduction	0.341 ± 0.010	-1%	0.355	-1%
100% Reduction	0.337 ± 0.009	-2%	0.349	-3%
Case A	0.344 ± 0.010	0%	0.358	0%
Case B	0.344 ± 0.010	0%	0.357	0%
Case C	0.343 ± 0.010	-1%	0.357	0%
Alameda Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.211 ± 0.004	-	0.210	-
50% Reduction	0.211 ± 0.004	0%	0.209	0%
100% Reduction	0.210 ± 0.004	-1%	0.209	0%
Langton Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.172 ± 0.002	-	0.175	-
50% Reduction	0.172 ± 0.002	0%	0.175	0%
100% Reduction	0.172 ± 0.002	0%	0.175	0%
Minnetonka Pond				
Scenario	**Mean TP** mg/L	**∆ Mean TP**	**Median TP** mg/L	**∆ Median TP**
Original	0.313 ± 0.009	-	0.323	-
50% Reduction	0.309 ± 0.009	-1%	0.318	-1%
100% Reduction	0.302 ± 0.009	-4%	0.309	-4%

Table E-4. *Wind sheltering reduction: Results for relative thermal resistance to mixing (RTRM).*

Scenario	Mean RTRM	∆ Mean RTRM	Median RTRM	∆ Median RTRM
Shoreview Pond				
Scenario	**Mean RTRM**	**∆ Mean RTRM**	**Median RTRM**	**∆ Median RTRM**
Original	0.080 ± 0.006	-	0.0665	-
50% Reduction	0.077 ± 0.006	-3%	0.066	-2%
100% Reduction	0.075 ± 0.006	-6%	0.0634	-5%
Case A	0.078 ± 0.006	-2%	0.0651	-2%
Case B	0.078 ± 0.006	-2%	0.0646	-3%
Case C	0.078 ± 0.006	-2%	0.064	-4%
Alameda Pond				
Scenario	**Mean RTRM**	**∆ Mean RTRM**	**Median RTRM**	**∆ Median RTRM**
Original	0.108 ± 0.007	-	0.094	-
50% Reduction	0.106 ± 0.007	-1%	0.093	-2%
100% Reduction	0.103 ± 0.007	-4%	0.091	-4%
Langton Pond				
Scenario	**Mean RTRM**	**∆ Mean RTRM**	**Median RTRM**	**∆ Median RTRM**
Original	0.071 ± 0.006	-	0.057	-
50% Reduction	0.069 ± 0.006	-2%	0.055	-3%
100% Reduction	0.067 ± 0.006	-5%	0.053	-7%
Minnetonka Pond				
Scenario	**Mean RTRM**	**∆ Mean RTRM**	**Median RTRM**	**∆ Median RTRM**
Original	0.105 ± 0.008	-	0.092	-
50% Reduction	0.103 ± 0.008	-2%	0.088	-4%
100% Reduction	0.102 ± 0.007	-3%	0.087	-6%

Table E-5. *Wind sheltering reduction: Results for benthic dissolved oxygen (DO) concentrations.*

Shoreview Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	0.164 ± 0.061	-	0.000	-
50% Reduction	0.161 ± 0.060	-2%	0.000	0%
100% Reduction	0.175 ± 0.062	7%	0.000	0%
Case A	0.166 ± 0.062	1%	0.000	0%
Case B	0.156 ± 0.060	-5%	0.000	0%
Case C	0.166 ± 0.061	1%	0.000	0%
Alameda Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	0.177 ± 0.076	-	0.000	-
50% Reduction	0.182 ± 0.077	3%	0.000	0%
100% Reduction	0.192 ± 0.080	8%	0.000	0%
Langton Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	2.205 ± 0.224	-	1.796	-
50% Reduction	2.272 ± 0.224	3%	1.985	11%
100% Reduction	2.422 ± 0.228	10%	2.140	19%
Minnetonka Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	0.055 ± 0.025	-	0.000	-
50% Reduction	0.053 ± 0.024	-4%	0.000	0%
100% Reduction	0.059 ± 0.025	8%	0.000	0%

Table E-6. *Wind sheltering reduction: Results for water column-averaged dissolved oxygen (DO) concentrations.*

Shoreview Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	1.445 ± 0.044	-	1.497	-
50% Reduction	1.447 ± 0.044	0%	1.498	0%
100% Reduction	1.474 ± 0.045	2%	1.533	2%
Case A	1.442 ± 0.045	0%	1.493	0%
Case B	1.443 ± 0.044	0%	1.494	0%
Case C	1.444 ± 0.044	0%	1.496	0%
Alameda Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	1.606 ± 0.057	-	1.725	-
50% Reduction	1.570 ± 0.059	-2%	1.710	-1%
100% Reduction	1.561 ± 0.060	-3%	1.695	-2%
Langton Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	4.248± 0.135	-	4.408	-
50% Reduction	4.331 ± 0.135	2%	4.486	2%
100% Reduction	4.445 ± 0.136	5%	4.540	3%
Minnetonka Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	0.620 ± 0.031	-	0.580	-
50% Reduction	0.627 ± 0.030	1%	0.597	3%
100% Reduction	0.656 ± 0.032	6%	0.619	7%

Appendix F

Watershed-Based Methods

Appendix F. Watershed-Based Methods

Table F-1. *Results for sediment fluxes.*

Shoreview Pond				
Scenario	**SRP Release** kg	**Δ SRP Release**	**DO Consumption** kg	**Δ DO Consumption**
Original	5.1	-	3600	-
50% Volume	4.9	-3%	3600	0%
50% Concentration	5.0	-2%	3600	2%
Alameda Pond				
Scenario	**SRP Release** kg	**Δ SRP Release**	**DO Consumption** kg	**Δ DO Consumption**
Original	2.0	-	1600	-
50% Volume	1.9	-5%	1800	12%
50% Concentration	1.9	-2%	1700	5%
Langton Pond				
Scenario	**SRP Release** kg	**Δ SRP Release**	**DO Consumption** kg	**Δ DO Consumption**
Original	0.0035	-	80	-
50% Volume	0.0031	-12 %	80	1%
50% Concentration	0.0033	-6%	80	0%
Minnetonka Pond				
Scenario	**SRP Release** kg	**Δ SRP Release**	**DO Consumption** kg	**Δ DO Consumption**
Original	1.5	-	2100	-
50% Volume	1.4	-4%	2100	-4%
50% Concentration	1.5	-1%	2100	1%

Table F-2. *Results for relative thermal resistance to mixing (RTRM).*

Shoreview Pond				
Scenario	**Mean RTRM**	**∆ Mean RTRM**	**Median RTRM**	**∆ Median RTRM**
Original	0.080 ± 0.006	-	0.067	-
50% Volume	0.064 ± 0.005	-20%	0.053	-20%
50% Concentration	0.080 ± 0.006	0%	0.067	0%
Alameda Pond				
Scenario	**Mean RTRM**	**∆ Mean RTRM**	**Median RTRM**	**∆ Median RTRM**
Original	0.108 ± 0.007	-	0.094	-
50% Volume	0.081 ± 0.006	-25%	0.068	-28%
50% Concentration	0.108 ± 0.007	0%	0.094	0%
Langton Pond				
Scenario	**Mean RTRM**	**∆ Mean RTRM**	**Median RTRM**	**∆ Median RTRM**
Original	0.071 ± 0.006	-	0.057	-
50% Volume	0.070 ± 0.006	-1%	0.056	-1%
50% Concentration	0.071 ± 0.006	0%	0.057	0%
Minnetonka Pond				
Scenario	**Mean RTRM**	**∆ Mean RTRM**	**Median RTRM**	**∆ Median RTRM**
Original	0.105 ± 0.008	-	0.092	-
50% Volume	0.107 ± 0.008	2%	0.094	3%
50% Concentration	0.105 ± 0.008	0%	0.092	0%

Table F-3. *Results for benthic dissolved oxygen (DO) concentrations.*

Shoreview Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	0.164 ± 0.061	-	0.000	-
50% Volume	0.157 ± 0.051	-5%	0.000	0%
50% Concentration	0.180 ± 0.065	10%	0.000	0%
Alameda Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	0.177 ± 0.076	-	0.000	-
50% Volume	0.158 ± 0.058	-11%	0.000	0%
50% Concentration	0.194 ± 0.080	10%	0.000	0%
Langton Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	2.205 ± 0.224	-	1.796	-
50% Volume	2.354 ± 0.231	7%	1.842	3%
50% Concentration	2.322 ± 0.229	5%	1.866	4%
Minnetonka Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	0.055 ± 0.025	-	0.000	-
50% Volume	0.056 ± 0.024	2%	0.000	0%
50% Concentration	0.057 ± 0.025	3%	0.000	0%

Table F-4. *Results for water column-averaged dissolved oxygen (DO) concentrations.*

Shoreview Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	1.445 ± 0.044	-	1.497	-
50% Volume	1.505 ± 0.046	4%	1.572	5%
50% Concentration	1.551 ± 0.047	7%	1.605	7%
Alameda Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	1.606 ± 0.057	-	1.725	-
50% Volume	1.630 ± 0.059	2%	1.761	2%
50% Concentration	1.759 ± 0.062	10%	1.938	12%
Langton Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	4.248 ± 0.135	-	4.408	-
50% Volume	4.462 ± 0.135	5%	4.627	5%
50% Concentration	4.401 ± 0.137	4%	4.571	4%
Minnetonka Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	0.620 ± 0.031	-	0.580	-
50% Volume	0.647 ± 0.030	4%	0.618	6%
50% Concentration	0.633 ± 0.031	2%	0.598	3%

Table F-5. *Results for anoxic days.*

Shoreview Pond				
Scenario	**Days No Oxia**	**∆ Days No Oxia**	**Days Any Anoxia**	**∆ Days Any Anoxia**
Original	92	-	99	-
50% Volume	91	-1%	99	0%
50% Concentration	86	-7%	99	0%
Alameda Pond				
Scenario	**Days No Oxia**	**∆ Days No Oxia**	**Days Any Anoxia**	**∆ Days Any Anoxia**
Original	88	-	99	-
50% Volume	91	3%	99	0%
50% Concentration	88	0%	99	0%
Langton Pond				
Scenario	**Days No Oxia**	**∆ Days No Oxia**	**Days Any Anoxia**	**∆ Days Any Anoxia**
Original	33	-	84	-
50% Volume	33	0%	84	0%
50% Concentration	33	0%	82	-2%
Minnetonka Pond				
Scenario	**Days No Oxia**	**∆ Days No Oxia**	**Days Any Anoxia**	**∆ Days Any Anoxia**
Original	99	-	99	-
50% Volume	99	0%	99	0%
50% Concentration	99	0%	99	0%

*Data not available due to model instability.

Appendix G

Bathymetry Modification

Appendix G. Bathymetry Modification

Table G-1. *Results for relative thermal resistance to mixing (RTRM).*

Shoreview Pond				
Scenario	**Mean RTRM**	**∆ Mean RTRM**	**Median RTRM**	**∆ Median RTRM**
Original	0.080 ± 0.006	-	0.067	-
Filled	0.052 ± 0.005	-35%	0.043	-36%
Dredged	0.127 ± 0.008	60%	0.118	78%
Redesign Shallow	0.047 ± 0.004	-40%	0.041	-38%
Redesign Deep	0.144 ± 0.009	81%	0.139	108%
Alameda Pond				
Scenario	**Mean RTRM**	**∆ Mean RTRM**	**Median RTRM**	**∆ Median RTRM**
Original	0.108 ± 0.007	-	0.094	-
Filled	0.108 ± 0.007	0%	0.094	0%
Dredged	0.158 ± 0.009	47%	0.159	69%
Redesign Shallow	0.059 ± 0.005	-45%	0.052	-45%
Redesign Deep	0.176 ± 0.009	63%	0.182	93%
Langton Pond				
Scenario	**Mean RTRM**	**∆ Mean RTRM**	**Median RTRM**	**∆ Median RTRM**
Original	0.071 ± 0.006	-	0.057	-
Filled	0.080 ± 0.006	14%	0.073	28%
Dredged	0.089 ± 0.007	26%	0.070	24%
Redesign Shallow	0.075 ± 0.006	7%	0.065	14%
Redesign Deep	0.093 ± 0.007	32%	0.078	38%
Minnetonka Pond				
Scenario	**Mean RTRM**	**∆ Mean RTRM**	**Median RTRM**	**∆ Median RTRM**
Original	0.105 ± 0.008	-	0.092	-
Filled	*	*	*	*
Dredged	*	*	*	*
Redesign Shallow	*	*	*	*
Redesign Deep	*	*	*	*

*Data not available due to model instability.

Table G-2. *Results for benthic dissolved oxygen (DO) concentrations.*

Shoreview Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	0.164 ± 0.061	-	0.000	-
Filled	0.553 ± 0.098	236%	0.000	0%
Dredged	0.065 ± 0.046	-61%	0.000	0%
Redesign Shallow	0.611 ± 0.096	272%	0.000	0%
Redesign Deep	0.104 ± 0.059	-37%	0.000	0%
Alameda Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	0.177 ± 0.076	-	0.000	-
Filled	0.177 ± 0.076	0%	0.000	0%
Dredged	0.058 ± 0.037	-67%	0.000	0%
Redesign Shallow	0.207 ± 0.068	17%	0.000	0%
Redesign Deep	0.119 ± 0.070	-33%	0.000	0%
Langton Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	2.205 ± 0.224	-	1.796	-
Filled	3.228 ± 0.207	46%	3.288	83%
Dredged	1.079 ± 0.191	-51%	0.000	-100%
Redesign Shallow	3.632 ± 0.209	65%	3.853	115%
Redesign Deep	0.863 ± 0.174	-61%	0.000	-100%
Minnetonka Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	0.055 ± 0.025	-	0.000	-
Filled	*	*	*	*
Dredged	*	*	*	*
Redesign Shallow	*	*	*	*
Redesign Deep	*	*	*	*

*Data not available due to model instability.

Table G-3. *Results for water column-averaged dissolved oxygen (DO) concentrations.*

Scenario	Mean DO mg/L	∆ Mean DO	Median DO mg/L	∆ Median DO
Shoreview Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	1.445 ± 0.044	-	1.497	-
Filled	1.425 ± 0.067	-1%	1.435	-4%
Dredged	1.240 ± 0.046	-14%	1.353	-10%
Redesign Shallow	1.521 ± 0.065	5%	1.547	3%
Redesign Deep	1.090 ± 0.041	-25%	1.163	-22%
Alameda Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	1.606 ± 0.057	-	1.725	-
Filled	1.606 ± 1.542	0%	1.725	0%
Dredged	1.622 ± 0.075	1%	1.950	13%
Redesign Shallow	1.129 ± 0.055	-30%	1.083	-37%
Redesign Deep	1.326 ± 0.061	-17%	1.454	-16%
Langton Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	4.248 ± 0.135	-	4.408	-
Filled	4.790 ± 0.122	13%	4.917	12%
Dredged	3.421 ± 0.167	-19%	3.393	-23%
Redesign Shallow	5.127 ± 0.117	21%	5.229	19%
Redesign Deep	3.030 ± 0.159	-29%	2.862	-35%
Minnetonka Pond				
Scenario	**Mean DO** mg/L	**∆ Mean DO**	**Median DO** mg/L	**∆ Median DO**
Original	0.620 ± 0.031	-	0.580	-
Filled	*	*	*	*
Dredged	*	*	*	*
Redesign Shallow	*	*	*	*
Redesign Deep	*	*	*	*

*Data not available due to model instability.

Table G-4. *Results for anoxic days.*

Shoreview Pond				
Scenario	**Days No Oxia**	**∆ Days No Oxia**	**Days Any Anoxia**	**∆ Days Any Anoxia**
Original	92	-	99	-
Filled	57	-38%	98	-1%
Dredged	95	3%	99	0%
Redesign Shallow	49	-47%	99	0%
Redesign Deep	91	-1%	99	0%
Alameda Pond				
Scenario	**Days No Oxia**	**∆ Days No Oxia**	**Days Any Anoxia**	**∆ Days Any Anoxia**
Original	88	-	99	-
Filled	88	0%	99	0%
Dredged	95	8%	99	0%
Redesign Shallow	86	-2%	99	0%
Redesign Deep	90	2%	99	0%
Langton Pond				
Scenario	**Days No Oxia**	**∆ Days No Oxia**	**Days Any Anoxia**	**∆ Days Any Anoxia**
Original	33	-	84	-
Filled	10	-70%	71	-15%
Dredged	64	94%	90	7%
Redesign Shallow	6	-82%	57	-32%
Redesign Deep	70	112%	92	10%
Minnetonka Pond				
Scenario	**Days No Oxia**	**∆ Days No Oxia**	**Days Any Anoxia**	**∆ Days Any Anoxia**
Original	99	-	99	-
Filled	*	*	*	*
Dredged	*	*	*	*
Redesign Shallow	*	*	*	*
Redesign Deep	*	*	*	*

*Data not available due to model instability.

Appendix H

Supplementary Data

Appendix H. Supplementary Data

Table H-1. *Phosphorus water quality data for four RPBCWD lakes treated with alum. The average concentrations were calculated for "n" years before and after the date of alum application and the n value is provided in parenthesis for each lake. The biweekly/monthly P data for the lakes are available through the MPCA EQuIS database <https://webapp.pca.state.mn.us/surface-water/search>.*

Lake	**Sample Location**	**Pre-Treatment TP** (mg/L) [n]	**Post-Treatment TP** (mg/L) [n]	**Reduction in TP**
Riley	Epilimnion	0.056 [2.3]	0.026 [3.3]	53.1%
	Hypolimnion	0.502	0.146	70.9%
Lotus	Epilimnion	0.056 [4.3]	0.040 [1.0]	29.3%
	Hypolimnion	0.429	0.059	86.3%
Rice Marsh	Epilimnion	0.081 [4.6]	0.029 [1.0]	64.6%
	Hypolimnion	0.107	0.033	69.5%
Round	Epilimnion	0.040 [2.7]	0.037 [5.8]	7.5%
	Hypolimnion	0.916	0.160	82.6%
Hyland	Epilimnion	0.073 [5.0]	0.030 [0.3]	59.1%

Table H-2. *Sediment phosphorus (P), moisture content, and bulk density data for the modeled ponds. Concentrations provided are average over the upper 4 cm depth of sediments. Mobile P is the sum of redox-P and labile organic-P mass (dry weight basis) in the sediments.*

Pond	**Mobile P** (mg/g)	**Moisture Content** (%)	**Bulk Density** (g/cm^3)
Shoreview	0.27	85	1.12
Alameda	0.57	88	1.09
Langton	0.10	48	1.60
Minnetonka	0.97	96	1.01