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Detecting phosphorus release from stormwater ponds to guide management and design

By

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Executive Summary

Stormwater ponds are widely used in urban areas to control pollution, flooding, and erosion arising from urban runoff. Despite long-standing acceptance and continued implementation of stormwater ponds to improve environmental conditions within and downstream from cities, recent research suggests that some stormwater ponds may fail to improve water quality as much as expected, exposing a lack of understanding of the processes that determine pond functioning. This two-year project investigated phosphorus dynamics in stormwater ponds in the Twin Cities Metropolitan Area to identify potential causes for elevated pond phosphorus (P) concentrations, which have the potential to be released to downstream water bodies. The goals of this study were to develop indicators for the risk of impaired phosphorus removal performance and thus gain information to better manage stormwater ponds. Research combined intensive monitoring of 17 ponds along with integrated sensor systems, sediment P release studies, hydrological monitoring, and physicochemical characterization of 64 ponds. These efforts together improved understanding of the controls on pond P concentrations and the effect of water balance on P retention by stormwater ponds.

As observed for lakes, dissolved oxygen (DO) concentrations played a key role in mediating stormwater pond P concentrations. Sediment core studies showed modest to strong sedimentary P release under anoxic conditions in overlying water, with no significant release, and in some cases P immobilization under oxic conditions. The redox-sensitive and releasable organic P forms in the sediments were found to be important in the mobilization of P under anoxic and oxic conditions. A large proportion of the ponds examined were strongly thermally and chemically stratified during summer. The same was true during winter of a small number of ponds sampled over an annual cycle. Analyses of water column thermal dynamics, wind speed, and pond characteristics revealed that many ponds were poorly mixed due to sheltering from the wind by surroundings and the presence of road salt, especially in the spring in ponds greater than 1 m in depth. Lack of wind-driven mixing led to the prevalence of low oxygen levels, a primary risk factor for internal phosphorus release. Anoxic conditions were observed in the water overlying the majority of bottom sediment area for 40% of ponds sampled at least once during the summer, and this anoxia was found to persist over most of the summer in 10 of the 17 ponds that were monitored intensively. The prevalence of low oxygen conditions was primarily related to duckweed cover and tree canopy cover. For ponds that lacked or had low duckweed cover (i.e. 0 - 10%), maximum depth played an important role, as deeper ponds characterized by anoxic bottom water layers where road salt persisted, and shallower ponds showing higher levels of DO. Ponds with duckweed showed extremely low oxygen levels during summer due to suppression of reaeration. Because these conditions

promote sedimentary phosphorus release, duckweed may create a positive feedback mechanism that promotes high phosphorus levels that further stimulate growth.

The water balance of stormwater ponds is often assumed to be neutral, where amounts of inflowing and outflowing water are approximately equal and there is little loss to evapotranspiration or groundwater. Our examination of stormwater pond water balances, however, showed substantial differences in water balances across the seven ponds we studied. We used continuous water level data as an indicator and identified large differences in the responses of pond volumes to rainfall-driven runoff or periods of drought. A small number of sites maintained steady and near-capacity volumes during dry periods, while a majority generated substantial storage volumes via non-outflow water losses (via evapotranspiration or groundwater losses) between rain events. Water level provided a good indication of the water balance of ponds during summer conditions, and because P retention is highly sensitive to water balance, water level data can also provide information to assess the likelihood of specific ponds to retain or release phosphorus.

Despite considerable variation in environmental conditions within and among stormwater ponds studied in this project, we identified effective indicators of pond water P concentration and retention. Low DO concentrations, sedimentary indicators, shallow depths, the presence of floating plants (i.e., duckweed), and heavy tree canopy cover were associated with elevated concentrations of P in surface waters. Although these indicators were all causally linked to some degree by complex mechanisms that are not yet fully understood, all are linked to underlying mechanisms that explain their association with elevated phosphorus. Water level provided a robust indicator of the tendency of ponds to retain or release inflowing water during summer. Because the water balance plays such a strong role in determining retention of P, water level measurements may provide a low-cost tool to identify ponds at risk for poor performance for P retention. Used together, indicators of surface water P concentrations and P retention could help prioritize maintenance or remediation for the many thousands of stormwater ponds in urban areas of the region.

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1. Introduction

Stormwater ponds (i.e., retention ponds, wet ponds, etc.) are one of the most common systems to address the negative impacts of urban runoff on water quality. The scientific basis for use of stormwater ponds to treat urban runoff was developed decades ago (e.g., Walker 1987, U.S. Environmental Protection Agency 1983). Such studies showed that wet pond systems (i.e., with permanent standing water) were effective at both slowing the rate of runoff and reducing downstream erosion, as well as at settling and trapping particulate pollutants and retaining them in pond sediments. Some dissolved pollutants were also seen to be effectively removed from stormwater, such as nitrate which can be removed by biotic uptake and burial or denitrification to the atmosphere. Since then, stormwater ponds have been widely added to urban landscapes, either through modification of existing water bodies or construction of new ponds.

While the water quality benefits of stormwater ponds are widely recognized, the processes that support them have not been well studied. In particular, phosphorus (P) cycling in stormwater ponds has emerged as an area of concern over the potential release of P previously trapped and stored in pond sediments. Over time, processes that lead to high concentrations of P in soils and sediments can result in unexpected losses of stored, or "legacy," P (Sharpley et al. 2013, Haygarth et al. 2014, Spears and Steinman 2020). Given the high loads and accumulation of P, stormwater ponds may be highly susceptible to remobilization of sedimentary P via internal loading from the sediments.

While relatively little information on stormwater pond P cycles exists, recent observations suggest that effects of internal loading may be widespread in stormwater ponds. For example, Taguchi et al. (2020a) observed high concentrations of P in surface waters in many ponds of the Twin Cities Metropolitan Area (TCMA) that exceeded those of urban runoff. Sønderup et al. (2016) found that retention of nutrients decreases with pond age across 66 ponds in Sweden. Research in Ontario found high bioavailability and rates of release of sediment P (Song et al. 2017, Frost et al. 2019) leading to elevated P concentrations in urban ponds. Together, these studies have exposed a lack of information about processes that retain or remobilize P in urban ponds, which have many characteristics that deviate from similar water bodies in undisturbed or agricultural settings, including co-occurrence of other pollutants, high rates of runoff inputs relative to volume and variable influences of management in urban environments. While research on stormwater ponds is expanding rapidly (see for example Chiandet and Xenopoulos 2016, Duan et al. 2016, Sharma et al. 2016) there is yet little ability to predict when and where to expect reduced P retention in ways that can be easily used by management and planning of urban water quality.

This project sought to understand P cycling in stormwater ponds in the Twin Cities region in order to provide information to improve the management of these widespread systems. The project integrated sampling to characterize conditions in a wide range of ponds, intensive monitoring using low-cost data loggers, and in-laboratory experimental studies on pond sediments. The project, active from January 2019 through December 2020, had four general goals:

(1) Identify factors that promote P trapping or release from stormwater ponds, including pond hydrodynamics (stratification, mixing, evaporation), pond and watershed attributes (volume, drainage area, dissolved oxygen concentration), and pond sediment characteristics (organic matter, redox P)
(2) Build a predictive understanding of the risk factors that lead to pond failure for P retention
(3) Develop and test inexpensive diagnostic tools to assess pond P retention to guide planning and management strategies
(4) Integrate knowledge of pond P dynamics into management guidance via a workshop, webinar, and assessment tool.

The report is organized according to six major project tasks aimed at understanding controls over pond phosphorus concentration (Tasks 1-4), pond phosphorus retention (Task 3), integration of information to identify ponds at risk of poor performance (Task 5), and dissemination of results (Task 6).

2. Methods and Materials

2.1. General Approach

To develop understanding to guide management of stormwater ponds, project data collection focused on understanding factors that regulate pond water and sediment P concentration, internal loading, and P retention. Factors that determine lentic P concentration are complex, and often involve interactions among both external watershed drivers, and internal factors such as aquatic vegetation, dissolved oxygen (DO), and mixing regimes. Our research approach sought to understand the major mechanisms regulating P cycles in stormwater ponds and to develop indicators of high P conditions to guide evaluation and management.

Rationale for data collection and analysis

Stormwater ponds are designed to store and treat runoff so that water released to downstream lakes and rivers is cleaner than inflows. For P, pond treatment is achieved via settling of particulate P, mineral sorption, and biotic uptake of dissolved PO_4, followed by settling and burial in sediments. These processes should lead to lower pond water P levels than incoming runoff on average and are the mechanisms that account for P retention in stormwater pond sediments. However, internal loading of P, or mobilization of sedimentary P back to the water column, can counteract P removals and lead to elevated P concentrations in pond surface waters and outflows. High surface water P reduces pond P retention when released downstream during storms and in baseflows.

Understanding the controls over internal loading of P in stormwater ponds, and pond surface water P, was a central focus of the project. We considered the potential role of both *watershed characteristics*, which may influence nutrient loading and accrual as well as other environmental variables within ponds, and *pond characteristics*, which affect the fate of P in ponds. The rationale for including specific variables is described briefly below, followed by detailed methods descriptions.

Watershed area and especially **connected impervious drainage** in urban watersheds (Janke et al. 2017) are strongly related to external nutrient loads. Over time, high external loads can lead to accumulation and internal loading.

Watershed land cover is well known to influence P loading to water bodies. All sites examined were located in the Twin Cities Metropolitan Area (TCMA) with land cover dominated by urban land use. Influences of specific watershed land cover characteristics were not examined in detail for several reasons. First, most sites had similar general land cover features. Second, drainage areas for each pond site could not be consistently delineated due to the lack of consistent data for pipe networks. Finally, preliminary analyses suggested that land cover was of less importance compared to other factors.

Stormwater ponds are designed to trap P. Over time, accrual of P and organic matter may lead to conditions that promote internal loading. Thus, **pond age** could be positively related to internal loading. Age was assessed relative to the most recent construction or major reconstruction relative to year 2019. Specific ages of many ponds older than 29 years since construction could not be accurately determined.

Inorganic P (i.e., phosphate; PO_4) binds with iron under oxic conditions but is released under anoxic conditions, so the characterization of the sedimentary oxygen regimes is a critical factor determining internal loading. While pond sediments are likely to be typically anoxic and difficult to characterize accurately, oxygen in the overlying water is more variable and can be used to estimate oxygen exposure of sediments. Water column **dissolved oxygen** (as well as temperature and conductivity) were determined from manual profiles at discrete depths (usually 25cm increments) from boats. Dissolved oxygen (DO) profiles and bathymetry measurements were combined to estimate the fraction of pond bottom area that was anoxic at daily, seasonal, and, for a few sites, annual scales (i.e., **anoxic factor**; Nürnberg 1995) as an integrative measure of DO regime.

Pond stratification/mixing dynamics influence oxygen and temperature regimes. Given the central role of redox conditions in determining pond P dynamics, we sought to understand the causes of pond mixing in this project. Metrics to characterize mixing and stratification regimes were derived from manual profiles of the water column (including temperature, DO, conductivity) sampled from boats at all study sites, as well as from continuous measurements via data loggers connected to sensors measuring temperature, conductivity, and wind speed at a smaller subset of sites (n = 17; Table 2.3-1). Sites were selected to include a range of pond **surface area**, **mean depth**, and **sheltering** from wind by shoreline objects (i.e., trees, banks, buildings) that influence mixing. Smaller, more sheltered, or deeper ponds were expected to resist mixing more strongly than larger, less sheltered, and shallower ponds. In addition, we used **electrical conductivity** to infer concentrations of **road salt** from winter applications. Recent work shows that high levels of salt accrue in ponds over winter and persist in bottom waters into the summer due to their higher density. This, in turn, leads to increased stratification and resistance to wind-driven mixing. Integrative metrics of mixing regimes included length of stratified period, relative thermal resistance to mixing (RTRM), comparisons of top-vs-bottom conductivity, and **mixing frequency** characterized by reductions in RTRM coincident with rainfall and observed wind.

Aquatic vegetation can strongly affect pond DO and P levels in aquatic systems. In particular, **free-floating plants** (FFP) including duckweed (*Lemna* spp.) and watermeal (*Wolffia* spp.) cover many stormwater ponds in the study region. We collected data to determine the cover and biomass in FFP and used related measurements (e.g., wind, P availability) to understand variability in distribution and abundance. Water column **chlorophyll-a** (mainly algae) was also sampled; the presence of rooted macrophytes was noted but not measured directly.

Sedimentary dissolved inorganic P (phosphate) flux is often the dominant pathway for internal loading. Both oxic and anoxic phosphate release rates were measured from seven ponds, and these data were analyzed with data from recent studies that used comparable methods.

Sediment properties such as total P and organic matter concentration may affect release rates of P and oxygen consumption. In addition, the bioavailability of sedimentary P varies with its specific mineral and organic matter association, so the relative amount of specific forms of P was also measured at a subset of sites.

2.2. Sites and types of sampling

Diverse measurements, methods, and approaches were used in the project. For clarity, we refer to the following definitions of sites and associated measurements throughout this report (Table 2.2-1):

Sediment phosphorus release sites (Dataset I) refer to study ponds where intact sediment cores were collected for lab-based incubations to measure sedimentary P flux under induced oxic and anoxic conditions. Sediment P release analyses included data from seven ponds supported directly by this project and eight sites from recent related studies. Many of these sites were also used as **intensive sampling sites** (Dataset III).

Intensive sampling sites (Dataset III) refer to study ponds with biweekly manual sampling during the warm season for water quality and water column profiles of physicochemical parameters; these sites include a subset of **monitored** ponds (Dataset II) with data loggers for continuous measurements of physical parameters to characterize mixing and stratification. Intensive monitoring was conducted during 2019, but some data from 2020, mostly supported by related projects, is included for some analyses (Table I-1).

Survey sites (Dataset IV) include a larger number of ponds sampled in 2019 to encompass more variation in pond features than present in the intensively monitored sites (Table I-1).

Extensive data sites (Dataset V) include ponds with a large number (≥15) of water quality observations on discrete dates that included total P distributed over summer or, in some cases, annually. Extensive datasets include all other data we could access from research and monitoring between 1992 to 2019. Fifty-four ponds with extensive data met these criteria, with a smaller number (8) that included annual sampling (Table I-2).

Figures and tables are referenced throughout in the report from an associated Appendix. Both are listed starting with roman numerals to distinguish them from those in the main report. In the Appendix, Table I-3 describes the sampling strategy employed for all primary study sites for this project. Table I-1 provides information for sites where existing data were used in our analyses.

Table 2.2-1. *Description and categorization of collected and acquired data*.

Dataset No.	Dataset Name	Number of Sites		Dataset Overlap
		This Project	From Past Work	
IA	Sediment Phosphorus Release	7	8	II, III
IB	Sediment Phosphorus Sampling	7 [P release] + 13 [TPI]	8 [P release]	IA, II
II	Monitoring - 2019	17	--	I, III
	Monitoring - 2020	7	--	I, III
III	Intensive Sampling	20	--	I, II
IV	Survey Sites	64	--	III
V	Extensive P Data	20	50+	I, II, III, IV

2.3. Detailed Methods

2.3.1. Study Site Categories

2.3.1.1. Survey Sites (Dataset IV)

We collected data from 64 ponds (Table I-1) to identify factors that regulate pond P concentrations. Ponds were chosen to represent a range of watershed characteristics, including size and dominant land use, and pond features, including area, volume, depth, surrounding tree cover, duckweed prevalence, and age. Survey ponds included sites studied in our previous work, where we had existing information on pond characteristics, and new sites. Most sites were in and around St. Paul and Roseville, as well as in the Riley-Purgatory Bluff Creek Watershed District (RPBCWD). Surveys were conducted in April, May, June, and August during the project's main field season in 2019, and each pond was sampled 2-4 times. Two main surveys in June and August, involved sampling of 46 and 64 ponds, respectively. The spring survey in April and May involved a smaller number (<20) and was used to study the transition from ice cover to warm summer conditions to examine effects of road salt and temperature changes on pond mixing and stratification. Water profile measurements of dissolved oxygen, temperature, and conductivity were taken concurrently with water chemistry and percent duckweed coverage for the main survey effort. Sediment organic matter and total P content were measured in 26 out of the 64 ponds.

2.3.1.2. Intensive Sampling (Dataset III)

A subset (n=20; Table 2.3-1) of the Survey Sites were sampled at a higher frequency during the field season to gain insight into seasonal dynamics and ensure a more complete characterization of ponds. Sites were sampled every 2-3 weeks from May through September or October of 2019. These sites were chosen for their range of watershed and pond characteristics and the availability of substantial data on water chemistry, phosphorus, sediment characteristics, etc. Several of the ponds had sediment phosphorus release data from recent studies (Taguchi et al. 2020a). Duckweed coverage was also measured at these sites periodically approximately biweekly from April to October.

2.3.1.3. Intensive Monitoring (Dataset II)

A subset of the Survey Ponds (Dataset IV) was selected for intensive in-situ monitoring (Table 2.3-1) in order to understand sub-daily processes expected to impact pond P concentrations and in particular the role of hydrodynamics (mixing, or the lack thereof). Data collected included time series of water level, wind speed, and a temperature profile. Some ponds included sensors with data loggers to collect continuous data on oxygen and salt concentrations. Monitoring stations were installed in a total of 17 ponds, which were chosen to represent a range of depths, sizes, and sheltering, and matched with Intensive Sampling efforts to support analyses. Of these 17 sites, two were lost or damaged beyond repair in a large storm on Aug 18, 2019, a third was located at a highly shaded site that prevented solar recharging of the battery, and a fourth experienced failure of half its thermistors, so only 13 sites could ultimately be used in detailed analyses.

2.3.1.4. Sediment Phosphorus Release and Characterization Sites (Dataset I)

A subset of the intensive sampling sites (Table 2.3-1) was investigated for sediment P release and characteristics in the laboratory to understand the potential internal P release under different environmental conditions. Seven ponds were selected for the laboratory studies such that a total of 15 ponds (Table 2.3-1) were investigated for sediment P when combined with data from previous studies (Taguchi et al. 2020a). Sediments collected from the ponds were incubated in sediment-water columns to determine the oxic and anoxic sediment P flux and then analyzed for the releasable (bioavailable) and unavailable P fractions that impact P release.

2.3.1.5. Sites with Extensive Phosphorus Data (Dataset V)

We used large numbers of observations compiled from our recent projects (926 measurements collected on 23 ponds) and seven other groups (1014 measurements collected on 31 ponds) to examine spatial and seasonal variation in stormwater pond total phosphorus (TP) concentrations in the Twin Cities Metro region. While most of the data was from recent years (2017-2019), data used includes sampling done as far back as 1992 These include our primary data collections in 2019, described in detail in this report, as well as observations of stormwater pond surface TP from 36 sites collected prior to this project (Table I-2).

Sites were selected for inclusion into this dataset if there were at least 15 or more warm-season measurements of TP in the dataset. 54 ponds with extensive data met these criteria, with a smaller number of sites (i.e., 8) that included annual sampling. Data for site characteristics (watershed area, tree canopy cover within a 50-m buffer, and pond duckweed cover), pond features (pond age, size, mean and maximum depth), and data sources were gathered (Table I-2).

Water sampling and analytical methods varied among the different groups. Descriptions of surface water sampling methods and analytical testing laboratories relevant to each group are described briefly in Table I-3. Although specific methods differed between projects and groups, the methods used to collect and analyze surface water TP were similar, allowing comparisons to be made across sites and datasets. The land use surrounding the ponds varied and ranged from low- and high-density residential, park, and school to institutional and commercial/industrial areas. Most sites in this dataset were older ponds with mean and median ages of 34 and 35 years. For about two-thirds of the ponds, whose age could not be confirmed directly, Google Earth imagery revealed that they were at least 29 years old relative to the year 2019.

Table 2.3-1*. List of In Situ-Monitored and Intensively-Sampled Pond Sites, 2019 and 2020. "Monit. Days" is the number of days with complete data collection for the Monitored Ponds (Dataset II); the number in parentheses is total days with complete ("Comp") OR partial ("Inc") data collection. Note that Sites 232, 233, and 234 were part of the intensive sampling program (Dataset III) but did not have stations installed; Site 300 was added in 2020 as an unsheltered reference site. Note also that the primary 2020 data used were water level observations for the eight 2020 sites (see Section 3).*

			Dataset II - 2019		*Dataset II - 2020*	
Name	**ID**	**Dataset(s)**	**Monit. Days, Comp (Inc)**	**Date Installed**	**Monit. Days, Comp (Inc)**	**Date Installed**
Arrow	15	II, III, IV	48 (69)	Aug 29	--	--
William Street	32	I, II, III, IV	74 (154)	Jul 17	--	--
Alameda	37	I, II, III, IV	100 (154)	Jun 26	161 (161)	May 29
Materion	40	II, III, IV	68 (68)	Aug 29	--	--
Seminary	41	II, III, IV	18 (18)	Sep 24	--	--
Cleveland/Roselawn	42	II, III, IV	77 (77)	Aug 30	93 (93)	Aug 5
RC Church	51	I, II, III, IV	83 (153)	Jun 26	--	--
Oasis	74	II, III, IV	19 (68)	Aug 29	--	--
Langton	79	II, III, IV	77 (77)	Aug 30	127 (161)	May 29
Bell	83	II, III, IV	14 (117)	Sep 5	310 (310)	Sep 5, 2019
Kasota East	116	I, II, III, IV	156 (156)	Jul 17	--	--
Swimming Pool	214	I, II, III, IV	24 (24)	Jul 25	--	--
Pond K	215	I, II, III, IV	0 (0)	Jul 25	--	--
Pond B	217	I, II, III, IV	85 (102)	Aug 23	--	--
Shoreview/Brennan	229	I, II, III, IV	132 (156)	Jul 17	161 (161)	May 29
849W	230	I, II, III, IV	83 (102)	Sep 5	149 (149)	Jun 10
BC-P4.10C	231	I, II, III, IV	99 (104)	Jul 25	100 (100)	Jul 30
Aquila	232	I, III, IV	--	--	--	--
Bren	233	I, III, IV	--	--	--	--
Shorewood 42	234	I, III, IV	--	--	--	--
35E/Larpenteur NE	300	II	--	--	108 (108)	Jul 22

2.3.2. Field Sampling and Monitoring

2.3.2.1. Field Sampling: Intensive Sampling Sites (Dataset III), Survey Sites (Dataset IV)

Field sampling methods generally followed the protocols used in a previous project by the team (Taguchi et al. 2018). Field efforts included the collection of a surface water sample (Datasets III, IV) and a hypolimnion grab sample (Dataset III) from near the pond bottom at the approximate deepest point of the pond. The hypolimnion sample was collected with a

Van Dorn sampler. The surface water samples were collected from five locations within the pond using Nalgene bottles, avoiding duckweed and algae on the surface, and combined into a single 1L composite sample of surface water. In 2020, duckweed (free-floating plants) was sampled to assess its potential influence on pond surface water TP via uptake, blocking light, and reducing oxygen diffusion. Surface-to-bottom profiles of water chemistry (temperature, DO, and specific conductivity) were measured in the pond's deepest location (if known) or approximate center using a Hach WQ40D handheld meter. Measurements were taken to the bottom of the water column, avoiding the influence of the loose sediment layer. Field assessment of the Survey Ponds included estimates of duckweed coverage (%) and other vegetation.

2.3.2.2. Field Sampling: Sediment (Dataset I)

Sediment cores were extracted from a subset of ponds (n=7; Dataset IA) among the Intensively Sampled sites (Dataset III) to be analyzed for P release in a laboratory setting (see Section 2.3.3 for methods). Five intact sediment cores, spatially distributed in the pond area, were collected from each pond by coring through holes drilled in ice (four ponds) or from a canoe (three ponds). The pond sediments with the overlying pond water were extracted into polycarbonate tubes (70 mm I.D.) using a Griffith sediment corer. After the completion of the P release study, the cores were analyzed for P fractions and organic matter content (method described under Section 2.3.3).

We also conducted a supplemental field effort to investigate a potentially simpler assessment method for pond sediments. This effort included sediment sampling to characterize organic matter and TP content. Sediment was sampled from five locations (or three or four locations in very small ponds) in a subset of ponds (n=13; Dataset IB) during Spring 2020 using a gravity sediment corer from a canoe. The top 4 cm of sediments was extruded from each sediment core and separated for analyses. The sediment samples were composited for TP analysis at the UMN Research Analytical Laboratory (RAL). Sediment samples were analyzed individually for organic matter using loss on ignition (LOI) methods.

2.3.2.3. Continuous Monitoring of Ponds with In-Situ Stations (Dataset II)

Wind speed, water level, and temperature profiles were collected continuously in a subset of ponds (n=17; Table 2.2-1) to characterize ponds' mixing and stratification behavior and to help interpret concurrent measurements of water quality. Instrumentation for the in-situ monitored ponds (Dataset II) was constructed with low-cost data loggers and sensors. At a minimum, each station included a water level sensor (ultrasonic, mounted above the water), an anemometer to measure wind speed (mounted ~3-4 feet above the water), and a set of sensors to record vertical profiles of temperature in the water, spaced 6-12 in apart. Data were logged at 10-minute intervals.

Two versions of the stations were ultimately developed. The first version suffered from damage by water leaking into instrumentation, necessitating a second, simpler, and more robust design (see Appendix II for details). These difficulties encountered in developing the stations led to delays in installations, so most data records for the sites do not begin until July 2019 (first version) or August/September 2019 (second version). A number of gaps in data collection were further caused by a power issue with the anemometers and as a result of damage from wildlife despite chicken wire covering. Despite these difficulties, we collected high-resolution data from 13 sites and were able to apply knowledge gained in 2019 to the monitoring of seven sites in 2020 for a related project and for water level analyses included in Section 4.

2.3.2.4. Bathymetry and Hypsography

Bathymetric surveys were used to determine pond depths and hypsographic curves used in data analyses. Bathymetric maps were acquired for several ponds from the cities in which they were located. For ponds lacking accurate maps, bathymetry was estimated using depth measurements collected through the ice during winter 2018-2019. Typically, several transects were made in north-south and east-west directions at roughly 5-ft intervals, with endpoints marked by GPS. Field measurements and provided maps were later digitized and analyzed (see Section 2.3.5).

2.3.2.5. Sampling for free-floating plants

To estimate the biomass and nutrient content of free-floating plants (mainly the duckweed genus *Lemna* and the watermeal genus *Wolffia*), we collected samples from eight ponds with extensive coverage in summer 2020. We refer to free-floating plants (FFP) generally throughout; most sites were dominated by duckweed species.

At each pond, samples were collected at five locations corresponding to our regular surface water sampling (see section 2.3.2.1). A sample of FFP biomass was collected at each point by inserting a circular disk 25 or 28 cm diameter with 100um nylon mesh below the duckweed layer and gently lifting it out of the water. The drained biomass sample was transferred to a plastic bag; any macroscopic tree leaves and insects were removed and discarded. Filamentous algae were sometimes present at low density and could not be easily separated from FFP biomass. Algae appeared to make a minor contribution to biomass in these samples; ponds dominated by floating filamentous algae or rooted macrophytes were not sampled for biomass. Samples were drained by inverting the bag and gently squeezing to remove excess water, or by pouring through a strainer. Samples were kept cool and transported to the laboratory where they were refrigerated until they could be dried. Some samples had to be frozen before drying for preservation, but this was generally avoided to reduce sample loss. Sample collection began in mid-July and continued until FFP cover dropped below 20% at each site.

Wet samples were dried at 60°C for up to a constant weight, weighed, and finely ground. Ground tissue samples were ashed, digested in sulfuric acid, and the extract analyzed for tissue P concentration using a colorimetric method. The absorbance of digested samples was measured on a Cary 50 Bio UV Visible spectrophotometer at 880 nm in 1 cm cells following the use of the molybdate blue/ascorbic acid reagent method for P analyses. "Apple NIST 1515" reference standards (National Institute of Standards and Technology) were used as a standard material. The amount of P in FFP biomass was calculated by scaling the measured biomass and P concentration of tissue samples collected and averaged across the five sampling locations to mass m^{-2} by adjusting according to the sampler size used. To examine the effect of FFP on water column TP, we assumed that FFP assimilated water column P from the upper 0.5 m of the water column at each site. We compared FFP vs. surface water below the plant layer by estimating the concentration of P in FFP biomass and whole water for this layer as mg/L.

2.3.3. Laboratory Sediment Phosphorus Release Study

2.3.3.1. Laboratory methods

The pond sediment cores were conducted at the St. Anthony Falls Laboratory (SAFL) for the P release study. Before starting the experiments, the water above the sediments was drained, filtered to remove particulates (1.2 um GF/F filter), and then carefully refilled into the columns. Porous air stones attached to PVC tubing were placed in the water column (~8 cm above the sediment surface) to gently bubble and mix the water column without disturbing the sediment surface.

The P release study consisted of three phases. First, the water column was kept oxic by aeration (DO > 9 mg/L). Then, air bubbling was turned off and the DO in the unmixed water column was allowed to decrease due to the sediment oxygen demand (SOD). In the third phase, the water column was mixed by bubbling ultrapure nitrogen gas to maintain anoxic conditions (DO < 1 mg/L).

Water samples for phosphate measurements were collected on a weekly basis. Samples were drawn from the approximate middle of the water overlying the sediment during mixed conditions in the oxic (first) and anoxic (third) phases. During the air off phase (second phase), water sampling was done at four points distributed across the water column depth, since a concentration gradient can develop under unmixed conditions. The four sample concentrations were used to estimate the average phosphate (soluble reactive phosphorus) concentration in the entire water column. The phosphate flux ($mg/m^2/day$) during each phase was calculated as the change in phosphate mass (where, mass = concentration × water volume) divided by the phase duration and the sediment surface area.

The DO concentrations in the unmixed water columns were measured ~8 cm above the sediment surface in each core. The DO data were fit to the Michaelis-Menten kinetic model (Michaelis and Menton 1913):

$$S = \frac{S_{max}[C_{O2}]}{K_M + [C_{O2}]}$$

where S is the substrate consumption rate, S_{max} is the maximum DO consumption rate which is fitted to match the S curve, C_{O2} is the substrate (oxygen) concentration, and K_M is the half-consumption concentration. A constant K_M of 1.4 mg/L was used for all cores based upon a regression of 60 sediment types in Walker and Snodgrass (1986). The assumption is that all DO reduction comes from the microbial oxygen demand of the sediments, so K_M represents the surface of the sediments.

2.3.3.2. Chemical Analysis of Water and Sediment Samples

Water samples collected from laboratory column studies were analyzed at the wet chemistry laboratory at SAFL. Phosphate was determined in filtered samples (0.45 um membrane filter) using the Lachat Quickchem 8000 autoanalyzer (Standard Methods 4500-PG, APHA 1995; detection limit = 5 µg/L P) and TP was determined by the ascorbic acid method (Standard Methods 4500-P, APHA 1995).

Water samples collected from field sampling were processed the same day as the collection. Samples were analyzed for TP and phosphate in both the SAFL and Finlay Laboratories, with the Finlay Lab also analyzing water samples for particulate phosphorus (PP), total dissolved phosphorus (TDP), and chlorophyll-a (chl-a). Samples for TP were collected as whole water (nonfiltered) samples from the composite sample collected in the field. For SRP and TDP, water samples were filtered through GF/F (Whatman) or membrane filters and if not analyzed within 24-48 hours, were frozen for future analysis. PP samples were filtered through GF/F filters, which were preserved by drying at elevated temperature and stored in a low-humidity chamber for later analysis. Sample handling and analytical methods were the same as in a recent project (see Taguchi et al. 2018 for details).

The pond sediments from the laboratory cores were analyzed for different types of bioavailable and unavailable P fractions contained in the total sediment P pool. The bioavailable sediment P fractions have the potential to release from the sediments under changing environmental conditions and contribute to internal loading. At least three out of the five cores from each pond were extruded into sections of 0–1, 1–2, 2–3, 3–4, 4–5, 5–7, and 7–10 cm sediment depths. Using the sequential chemical extraction procedure (method adapted from Psenner and Puckso 1988, SCWRS 2010), the amounts of different P fractions, i.e., the loosely-bound P (pore water-soluble and easily disassociated from a solid), iron-bound P (attached to an iron compound in the sediments), aluminum-bound P (attached to an aluminum compound in the sediments), mineral-bound P (attached to calcite and apatite in the sediments), labile organic P (available for microbial degradation), and residual organic P (considered recalcitrant or not available for microbial degradation), were determined. The sum of the P fractions provides the total sediment P concentrations. Water content (drying at 105 °C) and organic matter content (loss on ignition of dry sediment at 550 °C) were also determined in the sediment samples.

2.3.4. Data Analysis

2.3.4.1. Pond Attributes and Risk Indicators

Several data sources, including field and lab data collected in this project and in previous work and spatial and water quality datasets, were summarized and analyzed to assess a broad range of factors hypothesized to play a role in P retention in stormwater ponds. All factors developed and evaluated through statistical analyses (Section 2.3.6) are listed in Table 2.3-2. This section describes how these parameters were developed from the various datasets.

Table 2.3-2. *Pond attributes and potential indicators of poor phosphorus retention identified from past studies or developed in this project.*

Indicator	Potential Effect on P	Description	Ponds	Dataset(s)
Anoxic Factor	Oxygen Dynamics, Sediment Release	Fraction of sediments exposed to anoxic overlying water during monitoring season	38	I, II, III, IV
Canopy Cover	Wind Sheltering/Oxygen Dynamics, Litter Inputs	Fraction of land in buffer around pond classified as canopy and/or buildings	64	I, II, III, IV, V
Canopy Density	Wind Sheltering/Oxygen Dynamics, Litter Inputs	Density (0 - 1) of canopy in 25m buffer around pond, from LIDAR	18	I, II, III
Canopy Height	Wind Sheltering/Oxygen Dynamics, Litter Inputs	Mean height of canopy and building in 25m buffer around pond, from LIDAR	18	I, II, III
Embankment Height	Wind Sheltering/Oxygen Dynamics	Mean height of land relative to water surface in 25m buffer around pond, from LIDAR	18	I, II, III
Wind Reduction	Wind Sheltering/Oxygen Dynamics	Mean reduction in wind speed observed at pond relative to that at nearest airport	13	II
Runoff Mixing Frequency	Oxygen Dynamics, Vertical Transport	Frequency of mixing caused by runoff; specified as upper water column or top-to-bottom	13	II
Wind Mixing Frequency	Oxygen Dynamics, Vertical Transport	Frequency of mixing caused by wind; specified as upper water column or top-to-bottom	13	II
Drawdown Rate	Hydrology - Storage Recovery	Dry weather decrease in water level in the pond; specified for several periods in summer and fall	14*	II
Mean Relative Water Depth	Hydrology - Storage Capacity	Water level relative to discharge elevation; mean for all season	11	II
Outflow/Inflow Ratio	Hydrology - Storage Capacity	Rough estimate of outflow:inflow volume from changes in water level over season	11	II
Free floating plant (mainly duckweed)	Wind Sheltering/Oxygen dynamics, Organic matter inputs	Fraction of pond surface area that is covered by duckweed	64	I, III, IV, V
Sediment TP	Sediment Release	TP concentration in the upper 4 cm of sediments	25	I, III, IV, V
Sediment organic matter	Sediment Release	Organic matter concentration in the upper 4 cm of sediments	25	I, III, IV, V
Pond age	Sediment Release, Organic matter accrual, Litter inputs	Pond age since construction relative to year 2019	35	I, II, III, IV, V
Pond area	Oxygen Dynamics, Vertical Transport, Hydrology	Surface area of the pond from GIS or Google Earth	64	I, II, III, IV, V
Maximum depth	Oxygen Dynamics	Maximum depth measured from the pond surface	46	I, II, III, IV, V
Mean depth	Oxygen Dynamics	Mean of various depths measured from the pond surface	37	I, II, III, IV, V
Watershed area	External nutrient loading, Litter inputs, Sediment Release	Watershed area measured by ArcGIS	47	I, II, III, IV, V

2.3.4.2. Pond Depth and Hypsography

Bathymetric data provided direct estimates of *maximum depth* and *mean depth* in many of the ponds, as well as hypsographic curves (pond bottom area vs. depth) that were used for other analyses. From depth measurements (both field-collected and those digitized from provided maps), we generated a raster-based map of the pond depth by using a spline interpolation method in ArcGIS. A hypsographic curve was then generated from these depth maps for all ponds with bathymetric data (n = 52 ponds).

2.3.4.3. Anoxic Factor

Anoxic factor (AF) is used to describe the fraction and duration of anoxic conditions at the sediment-water interface at the scale of whole ponds or lakes. As such, AF provides a risk indicator for the release of the redox-sensitive iron-bound phosphorus fraction, which occurs as sediments become anoxic Nürnberg (1995). AF is defined by Nürnberg (1995) as "the number of days that a sediment area, equal to the whole-lake surface area, is overlain by anoxic water" and is defined as follows:

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

The right-hand side of this equation is the *anoxic fraction*, or fraction of sediment exposed to anoxic conditions. The anoxic sediment area was determined from comparing DO profiles with pond hypsographic curves determined from bathymetry and assuming lateral uniformity of DO; a DO value of 2.0 mg/L was the threshold below which water was considered anoxic. For example, in a 2m-deep pond for which the DO profile was < 2 mg/L for all locations within 0.5 m of the bottom, we assumed that all sediment in the pond located more than 1.5m from the top of the pond (2m – 0.5m) was overlain by anoxic water. The hypsographic curve (see previous section) provides the fraction of pond sediment area located at this depth or greater.

To account for different sampling frequencies and variable lengths of monitoring periods associated with our monitoring efforts, we used AF over several timescales. We calculated anoxic fraction (right-hand side of above equation) for each profile date at every site in our datasets, allowing for assessment of anoxia across sites and of changes over the season within sites. These calculations will be referred to as *anoxic fraction*, or "discrete" anoxic factor ($AF_{discrete}$). We used time-averaging of anoxic fractions over periods of interest (e.g. summer or a defined monitoring period) for individual sites. Subscripts of AF (i.e., AF_{summer}, etc.) define these periods. All anoxic fractions within the period of interest were weighted by half the length of time between the previous and next profile events, divided by the total length of the period of interest. In this way, anoxic factor could be assessed for different periods of interest and also account for different monitoring frequencies among the different sites and datasets; this was especially useful for Datasets II and III (Intensive and Monitored ponds). The anoxic fraction for interpreting oxygen conditions at sediment core locations was also done for the monitored period and summer (also described in Section 3.3).

2.3.4.4. *Mixing Events*

Water column mixing is an important process in the oxygen dynamics of lakes and was hypothesized to influence phosphorus levels and export from ponds. Mixing events were characterized in several ways for the Monitored Ponds (Dataset II), including by source of mixing (runoff vs. wind), by depth of mixing (upper pond and top-to-bottom), and by strength of mixing criteria (magnitude of reduction in stratification).

Strength of stratification was defined by relative thermal resistance to mixing (RTRM), a measure of the tendency of two “layers” of water to mix based on relative density:

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

where ρ is the density of water at layers *i* and *j* (two points in the vertical profile), relative to the density difference between water at 4˚C and at 5˚C. For RTRM > 0, the two layers are stably stratified and more stable at higher values, while at values ≤ 0 the layers are unstable and will mix. Note that no effort was made to correct for conductivity in this calculation, which would have some effect on density at high values.

RTRM was computed for two portions in the pond water column: the upper water column (top node in the profile vs. a node located 1-2 feet below that) and the whole water column (top node vs. bottom node in the profile). In this way, we characterized mixing events as involving just the upper water column or the whole water column. RTRM was computed at every time point in the temperature profile time series (10 or 15 minutes, depending on instrumentation).

Sub-hourly RTRM data were noisy, so values were screened at daily intervals for the purpose of identifying events. Mixing events were defined by a reduction in mean daily RTRM from one day to the next, with two criteria used for the strength of the mixing event based on the reduction of RTRM. Criteria 1 (“**C1**”) was a “**delta**” criteria and included days for which RTRM decreased by a value of 20 or more relative to the previous day. Criteria 2 (“**C2**”) was a “**threshold**” criteria and included days for which the previous day’s RTRM was “strong” (at a value equal to 1 standard deviation greater than the season mean RTRM) and was reduced to below this threshold.

Mixing events were further classified by source of mixing: if rainfall of > 1 cm was observed at the nearest airport to the site for the current or previous day, mixing was assumed to be induced by runoff inputs and the event was classified as a “**runoff**” mixing event. If the mean wind speed for the current or previous day was above the median wind speed for the whole monitoring period (i.e., a relatively windy day) and no rainfall was observed, it was classified as a “**wind**” mixing event.

The output of this analysis was **mixing frequency** (d^{-1}) or the total number of mixing events for each set of criteria divided by the length of the monitoring period. Frequency was used instead of absolute number of events due to the variation in length of monitoring periods among sites.

In the regression analyses, neither *runoff* nor *wind* mixing frequency were found to be great predictors of water quality variables; **total mixing frequency** (mixing frequency due to runoff and/or wind) was used instead and found to be a better predictor.

2.3.4.5. Wind Reduction

Wind speeds observed at pond monitoring stations (Dataset II) were compared to concurrently observed reference wind speeds at nearby weather stations. This **wind speed reduction** (pond station wind speed normalized by reference wind speed) is a direct measurement of approximate wind sheltering on the ponds. For sites located around Roseville, the MN State Climatology Office[1] (MNSCO) station was used as the reference wind site, while for the ponds in RPBCWD, the Flying Cloud Airport [2]was used as the reference wind site. Wind speeds at the MNSCO were corrected for height from 3m to 10m (the latter being the Flying Cloud airport wind measurement height) using Monin-Obukhov (log-law) theory, with an aerodynamic roughness of 0.10 for agricultural fields water and assuming a stable atmosphere (Stull 1988). Data were retrieved either from Iowa Environmental Mesonet (mesonet.agron.iastate.edu) or the Midwest Regional Climate Center (mrcc.illinois.edu/CLIMATE/).

2.3.4.6. Tree Canopy and Sheltering

Tree canopy was hypothesized to be an important factor for pond phosphorus dynamics through influences on wind sheltering (reduction of mixing), water level dynamics (through evapotranspiration), and direct inputs of nutrients through leaf litter. We used several spatial datasets to characterize canopy cover and sheltering around the study ponds, including 1-m resolution land cover data[3] (see also Host et al. 2016) and LIDAR data[4], both available from MN Geospatial Commons[5].

Canopy cover fraction around each pond was defined from the land cover data, which included a category for 'trees' (defined as vegetation greater than 3 m tall) as well as for 'buildings' (also with a threshold of 3 m height). Given that vegetation height and density

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[1] https://mesonet.agron.iastate.edu/sites/site.php?station=UMNM5

[2] https://mesonet.agron.iastate.edu/sites/site.php?station=FCM

[3] https://conservancy.umn.edu/handle/11299/181532

[4] http://arcgis.dnr.state.mn.us/maps/mntopo/

[5] http://gisdata.mn.gov/

were not explicitly represented in these data, we tested the use of buffers (bands around the perimeter of each pond) ranging in size from 5 m to 150 m for computing the mean canopy and canopy + building fractions within these buffers (amount of area in the given classes / total area in the buffer). We also used a buffer size that was equal to the nominal diameter of the pond (the diameter if its surface area was a circle), to account for potentially smaller areas of influence experienced by smaller ponds. Of these various buffer sizes, canopy cover in the *25 m* and *50 m* buffers were found to provide the best correlations with concentrations of most forms of P in the ponds of the Intensive and Monitoring datasets (II and III; results not shown). Results between these two buffer sizes were virtually indistinguishable. Survey ponds (Dataset IV) were assessed with the 50 m buffer, and the Intensive and Monitoring Ponds (Datasets II and III) were assessed with a 25 m buffer to be consistent with the LiDAR analysis (see next paragraph), which was an intensive effort that was not feasible to repeat for 50 m buffer in this study.

Canopy height, canopy density, and **embankment height** were parameters determined from analysis of raw LIDAR point clouds. For these analyses, we used a 25 m buffer around the ponds, selected initially by looking at tree cover in aerial photos around several ponds. Given the intensive nature of analyzing LIDAR data, we did not make an assessment of sensitivity to buffer size; further, we restricted computations to only the Intensively Sampled Ponds (Datasets II and III). *Canopy height* was calculated as the mean height of all non-ground LIDAR returns in a 25 m buffer around the pond (including both canopy and rooftops); *canopy density* was defined by the USFS as the fraction of all LIDAR returns from canopy (i.e., all non-ground returns divided by the total ground + non-ground returns; McCallum et al. 2014); and *embankment height* was defined as the mean difference in elevation between the pond water surface and the land in the buffer. Embankment height was intended to characterize the topographic sheltering around the ponds, which ranged from as little as 1 m to as much as 8 m.

2.3.4.7. Hydrologic Indicators

Water and nutrient export from stormwater ponds occurs when the water level is above the elevation of the outlet control structure. Pond water levels are regulated by water inputs relative to water losses including evaporation, seepage and outflow. When water losses are high relative to input, storage volume is generated that allows for retention of water and phosphorus from subsequent storms. Despite the importance of pond water balances in determining P retention vs release to downstream waters, little information exists to understand controls of variation in pond hydrology. However, in our previous study of pond P mass balances (Taguchi et al. 2018, Janke et al. *in prep*), we found pond hydraulic function to have a significant influence on P mass retention. In particular, antecedent storage capacity (surplus or reduced volume relative to permanent pool) was positively related to event P retention, and higher drawdown rates (observable water loss during dry periods) seemed to lead to better overall P retention at annual scale. In the current project, we did not monitor inflows and outflows of ponds, but computed relevant parameters from our continuously monitored water level in 13 ponds in 2019 (seven ponds in 2020; Dataset II; Table 2.3-1).

A continuous recording of the water level in ponds can be used to estimate the storage volume in the pond as well as the volume of water released during discharge events. This assumes that the percent increase in pond surface area with an increase in water surface elevation is small. The accuracy of this estimate can be seen in Figure 2.3-3. If the increase in pond volume with pond water surface elevation can be approximated by a linear fit (i.e., a straight line) for elevations around the outlet level, then the approximation is accurate. This assumption holds if pond shorelines are evenly sloped.

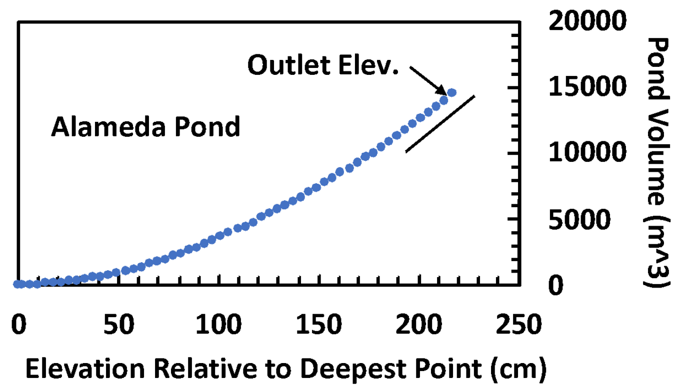

Figure 2.3-1 *Pond volume versus elevation to deepest point for Pond 37, one of seven stormwater ponds (Dataset II). The straight line is provided to assess the validity of the straight line approximation at and around the outlet elevation.*

We used the continuous recordings of pond water level relative to the outlet elevation to estimate the rate of water level decrease ("drawdown rate") at different times of the year for each pond, and to estimate the inflow versus outflow of each pond ("retention fraction"). These tasks were performed on both the 2019 and 2020 water level data. Relative water level was defined such that discharge occurred for water level >= 0. Straight lines were fit by eye to the decrease in water level that seemed to be linear, and sufficiently long to result in an accurate fit of the slope (see Results, Section 3.5.2). The relative inflow from runoff was estimated by the rapid increase in water level that would occur with a storm during the growing season in Minnesota. If the pond was below the outlet level, that runoff was used to fill up the pond.

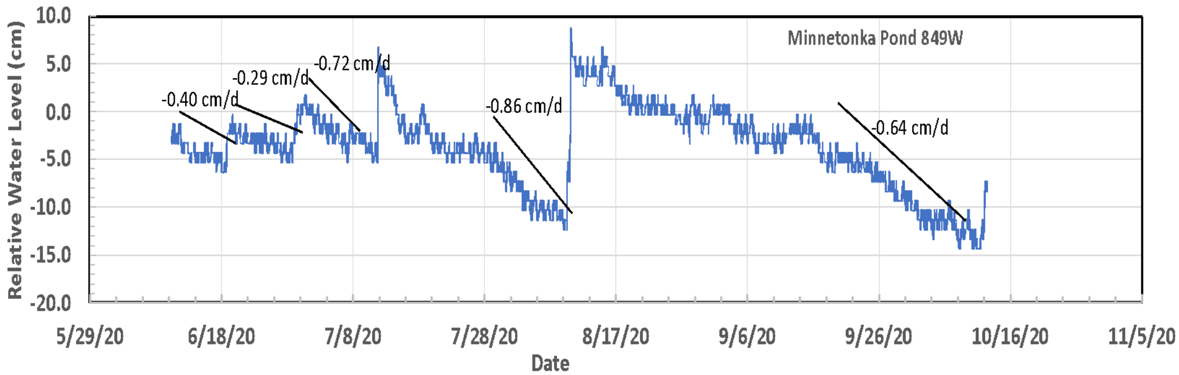

Figure 2.3-2 *Water level relative to the outlet elevation versus time for Minnetonka Pond 849W. Lines drawn next to the water level trace indicate estimated drawdown rate.*

The outflow from the pond relative to the inflow was estimated from the increase in water level above the outlet elevation. Then, the fraction of the inflow that would leave the pond is defined as:

$$Retention\ Fraction = \frac{Outflow}{Inflow} = \frac{\sum Water\ Level\ Incr\ Above\ Outlet\ Elev}{\sum All\ Water\ Level\ Increases}$$

Drawdown rates were calculated for most of the monitored ponds as a decrease in water level (cm/day) during dry periods. Several periods were used for this computation in 2019 between July 29 and September 17. The 2019 data included all 13 sites, the earlier periods included only a subset of sites due to data gaps. Values ranged from 0.50 cm/day to 4.25 cm/day. In 2020, the analysis was conducted at seven ponds between May 29 and November 5, with some variance between sites.

Retention fraction was calculated for seven ponds in the 2020 dataset, as well as for a subset of ponds monitored in 2019 for which the outlets had been located and water level elevations measured relative to the monitoring stations (n = 10 of the 13 monitored ponds in Dataset II). **Mean relative water depth** (water elevation relative to that above which discharge occurs) was also computed for the whole monitoring season at each pond.

2.3.4.8. Road *Salt (Conductivity)*

The impact on stratification in ponds from winter road salt accumulation was assessed using several methods. First, using a similar approach to McEnroe et al. (2013), the profiles of specific conductivity (SC) were characterized on each profile date by the maximum conductivity (SC_{max}), and the top-vs-bottom (btm) difference in conductivity ($SC_{dtopbtm}$), which were assumed to relate to salt accumulation and resulting stratification strength, respectively. These parameters were averaged over the season or period of interest to assess “mean” conditions, and also computed as change over the season to assess the reduction in salt levels and stratification.

A second approach to assess road salt impacts involved adjusting the stratification strength calculation relative thermal resistance to mixing, or RTRM; see above) for the effect of conductivity. Density in the RTRM formula was calculated as a function of both temperature and conductivity (“$RTRM_{tc}$”) per Novotny and Stefan (2012) by converting the conductivity to salinity and assuming that most of the ions contributing to conductivity were sodium and chloride (a valid assumption at high conductivity). Mean $RTRM_{tc}$ and season change in $RTRM_{tc}$ were computed over periods of interest (e.g., summer or monitoring period).

In the analysis of the Intensive and Monitoring Ponds Datasets (II and III), we found that period mean $\mathbf{RTRM_{tc}}$ (and to a lesser extent period mean $\mathbf{SC_{max}}$) were most strongly correlated with pond P concentrations (see Results). The other SC parameters were not used in detailed analyses. Note that $RTRM_{tc}$ was only computed from the profile data (i.e., discrete in time); we did not have sufficient SC time series to adjust the station time series RTRM values for conductivity.

2.3.4.9. Statistical Analyses

2.3.4.10. Extensive data sites (Dataset V)

We used all existing data from 54 sites to compare average seasonal surface TP concentrations. TP concentrations were first averaged for each seasonal category within individual sites before averaging across all sites, where spring ranged from April 1 to May 31, summer from June 1 to September 15, fall from September 16 to November 30, and winter from December 1 to March 31. We also examined seasonal average TP fluctuation for each site for a subset of ponds in which 8 out of 27 sites had annual representation of TP and the remaining 19 had one to two seasons (i.e., winter and/or spring) not represented. Statistical analysis for the above seasonal comparisons was conducted using one-way analysis of variance (ANOVA) on log-transformed TP which met assumptions of normality using Shapiro-wilk test and equal variance, followed by means comparison using Tukey’s HSD in JMP Pro 15. We compared average surface TP concentrations between the summer vs. cool, open water period for each site individually and for overall sites using a non-parametric Mann-Whitney U test in R. Means for summer and cool periods were calculated by site, and averages across sites were calculated from the site means. We also analyzed the potential of pond characteristics (mean and maximum pond depth, pond age, sediment organic matter, sediment TP, pond size, percent duckweed cover on the pond surface) or site features (watershed area and percent canopy cover within a 50-m buffer from the pond) to explain the difference in magnitude between summer vs. cool season TP using single linear regression (SLR) analysis. Similarly, we applied SLR analysis to examine relationships of previously described pond and site features with surface TP concentrations, which can provide information about the potential risk factors that are likely to elevate P concentrations in stormwater ponds.

2.3.4.11. Survey data sites (Dataset IV)

The variability in mean surface water TP concentrations across ponds (n = 46) was compared using categorical independent variables to develop indicators of pond TP. Combined survey dataset was utilized for this analysis. Independent variables examined were high, medium, and low levels of duckweed cover, anoxic factor, mean water column DO, maximum depth, pond age, pond area, sediment TP, and watershed canopy cover within a 50 m buffer (see Table II-2 for details on variable classification). Duckweed cover, anoxic factor and mean water column DO were averages of survey I and II. Mean TP was also derived from survey I and II average, wherein survey I, measurements were taken once over the period May 31 to early June 2019, and survey II were taken during late July or early August 2019. Means comparisons were made using one-way ANOVA once data met assumptions of normality using Shapiro-Wilk test and equal variance in JMP Pro 15, followed by Tukey HSD test.

An SLR analysis was conducted for the combined and individual survey dataset to examine the relationships between mean surface TP concentrations with various pond and watershed features as indicator variables. Pond features included pond age relative to year 2019, maximum depth, mean depth, pond area, percent duckweed cover, anoxic factor, sediment TP, sediment OM, mean water column DO, mean water column conductivity and mean water column temperature. For the combined survey dataset, average values of the individual surveys were used for surface TP, duckweed, anoxic factor, mean water column DO, conductivity and temperature. RTRM parameters were also included, i.e., mean RTRM of survey I and II and RTRM difference between survey I and II for combined surveys, and RTRM difference in top and bottom of the water column for individual surveys. Watershed features included watershed area, ratio of watershed area to pond area, and percent canopy cover within a 50m buffer. We examined relationships between these same pond and watershed variables vs. anoxic factor to better understand controls over water column DO and pond duckweed cover. In addition, we examined factors affecting variation in surface (upper 4 cm) sediment TP concentration.

Multiple linear regression (MLR) analysis was conducted to identify risk factors that lead to elevated surface TP concentrations in stormwater ponds. The package called “glmulti” was used in R software for this analysis. A full model with all the candidate variables with respect to pond and watershed characteristics to be tested was created. The “glmulti” function was used to go through every possible combination of variables in the full model to identify the best model, with sub-models ranked by the Akaike Information Criterion (AICc). For combined surveys, pond characteristics included pond age relative to year 2019, maximum depth, mean depth, pond area, mean percent duckweed cover, mean anoxic factor, sediment TP, sediment organic matter, mean RTRM and change in RTRM between survey I and II. Watershed characteristics included watershed area, ratio of watershed area to pond area and percent canopy cover within a 50m buffer. For individual surveys, all the prediction variables were the same except for the RTRM parameter, which included RTRM difference in the top and bottom profile of the water column, i.e., RTRM(Top-Btm). Paired values of anoxic factor, duckweed cover and surface TP were also used as opposed to averages. In our MLR analysis overall, it is to be noted that integrative metrics as opposed to single parameter metrics for our candidate variables were selected where relevant not only to avoid potential issues arising from collinearity, but also because they integrate more information in a single parameter and were proven to be stronger predictors of TP from the screening of SLR results (as in the case of anoxic factor vs. mean water column DO).

2.3.4.12. Sediment, Intensive Sampling and Monitoring Datasets (Datasets I, II and III)

Statistical analyses of these datasets were performed using methods similar to those used for the Extensive and Survey Datasets (see above). The first step was a correlation analysis between water quality variables (various forms of P, including both epilimnion and hypolimnion samples; anoxic factor; duckweed) and a range of potential predictors related to pond geometry (e.g. size, depth), water chemistry (dissolved oxygen, specific conductivity), stratification and mixing (RTRM, mixing frequency), and canopy cover. Results were reported in terms of Pearson r and p-value and are included in the Appendix.

SLR was used to develop models for the strongest individual predictors for each water quality variable in the correlation analysis; full models are presented in the main text for TP, AF, and duckweed (the latter two parameters chosen due to their strong correlation with TP and other forms of P), with selected models for other P forms included in the Appendix. MLR was used to construct models for TP, AF, and duckweed to develop more complete or predictive models of pond TP. Models were ranked by information criterion (AIC), and an assessment of the relative importance of various parameters was done through computation of η^2, a parameter that approximates the amount of variance explained by each parameter in a model.

A similar approach was adopted for the sediment variables (phosphate flux, various sediment phosphorus fractions, organic matter content, sediment oxygen demand) and in situ water quality variables (anoxic factor, TP and SRP concentrations). The simple linear regression analysis was done for individual cores collected from the ponds (3 to 5 cores per pond for 14 ponds, total = 51 cores) to identify the important pond sediment variables impacting internal P release in ponds. The correlations with anoxic factor were done for 10 ponds (36 cores total). The whole-pond averages (mean calculated using the data for five to six cores from a pond) were used in the regression analysis involving pond geometry (14 ponds). The results were reported in terms of Pearson r and p-value, and the scatter plots are shown in the Appendix. The SLR results were then used to develop multivariate models for predicting sediment P flux based and were assessed by the standard error (SE) and relative SE (SE divided by the mean P flux), and the model predictions were compared against actual P flux (determined in the laboratory pond cores) and models developed for lakes (Nürnberg 1988).

3. Results

3.1. Climate Conditions

Most data collection occurred in 2019 in accordance with the original 1.5-year project work plan. Record precipitation was observed statewide in 2019, and many local precipitation records were approached or exceeded in our study watersheds. Thus, we note that our 2019 results may have been influenced by larger than average amounts of runoff relative to other years. Specifically, May 1 - October 31 rainfall in 2019 was 28.5 inches at the Minneapolis-St. Paul Airport, with 24.2 inches measured over the same period in 2018 and 19.7 inches in 2020 (1981-2010 average annual rainfall is 30.6 inches, for reference; see MnSCO[6]). A project extension was granted in 2020 allowing for some additional data collection in 2020, a much drier summer. To provide some context for this work's primary study year (2019), we present in Section 3.2 below a time series of surface TP concentrations from a small set of ponds that have been sampled routinely since 2017.

3.2. Seasonality of Pond Water Quality, Duckweed, and Stratification Dynamics

Dissolved oxygen is a major driver of P dynamics in lentic ecosystems, but is influenced by interacting processes (heating, water column mixing, metabolism) that vary on daily to seasonal time scales. The dynamics of DO and factors that regulate it are presented in this section.

3.2.1. Annual TP Measurements and Longer-term Data

A small number of sites (n = 3) that were sampled by our group over annual cycles included data from multiple years (Figure 3.2-1). These measurements, which date back to June 2017, were partially supported by other projects (Janke et al. *in prep*; Taguchi et al. 2020a) or include data collected in 2018 by Finlay lab personnel. This annual sampling included data that were mostly unavailable in the extensive data analyses in Section 3.2.2.

Several features of these longer-term plots are worth noting. First, a distinct seasonality was present in the surface TP concentrations for all years of data collection across all sites, with the highest TP concentrations occurring during winter followed by a rapid drawdown or flushing in spring and with the lowest concentrations in April and May. TP generally increased over the open-water season, with peaks in late summer or fall. Secondly, while the timing of peak winter/spring concentrations were remarkably consistent across sites and years, the timing of late season peaks varied among sites. For example, Sites 32 and 51 showed late summer peaks and potentially another peak during the fall in some years, while Site 37 showed a slow decline over summer with a sharp increase in fall that would coincide with leaf drop and vegetation senescence. Some variability is present year-to-year

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[6] files.dnr.state.mn.us/natural_resources/climate/twin_cities/msp_normals_means_extremes_page3.pdf

(Site 32 in particular), which as mentioned above may be due to the high variability in precipitation (2019 was a much wetter year than 2017, 2018, or 2020; we note also that both fall 2017 and most of 2020 were drier than average.

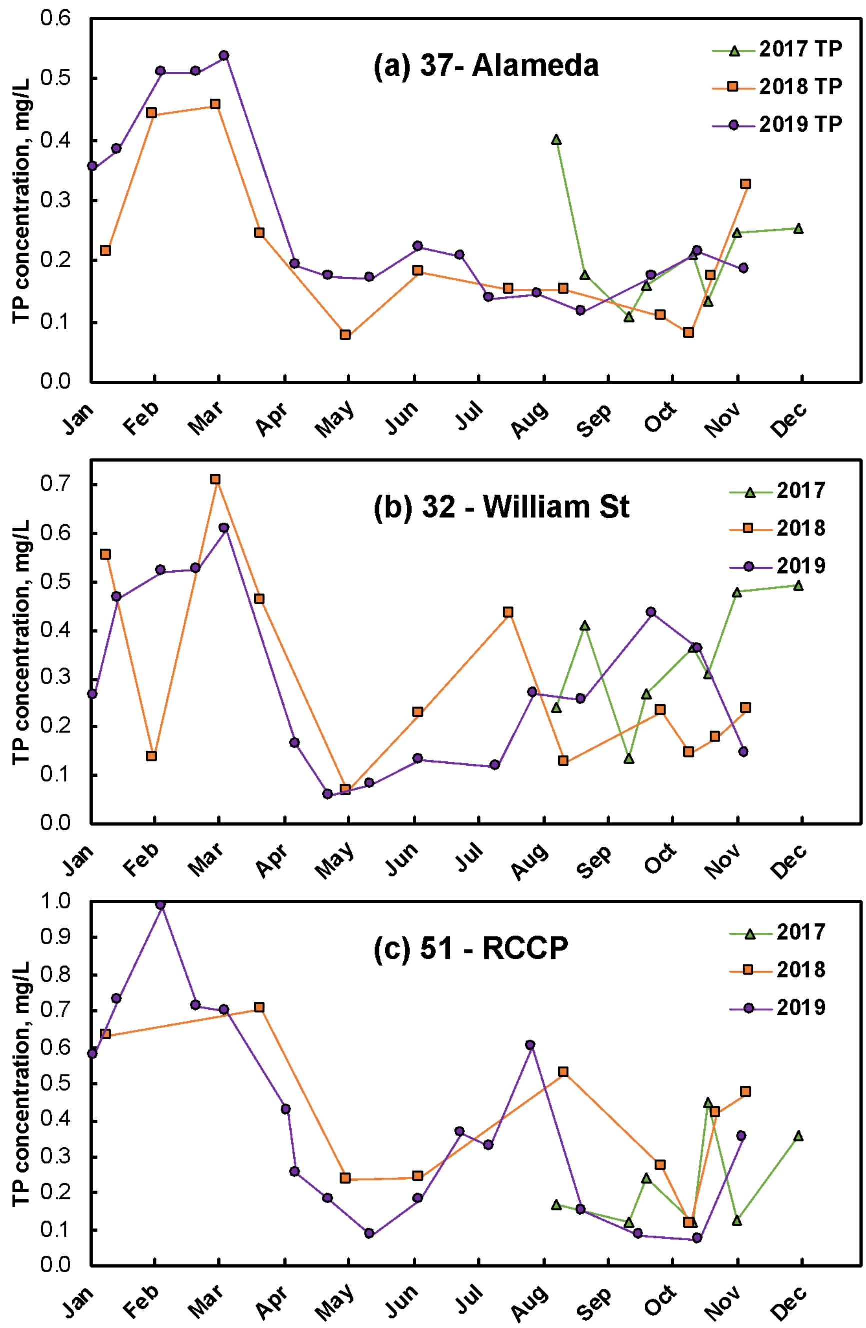
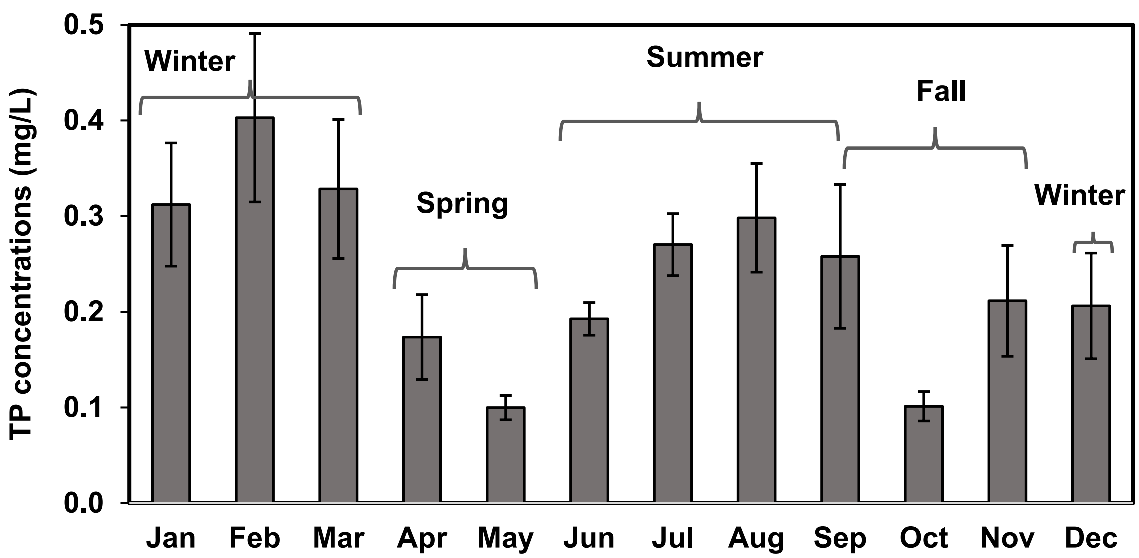


Figure 3.2-1 *Surface water TP concentrations (mg/L) measured in three ponds that have been sampled routinely since June 2017. Ponds are located in Roseville, MN and include (a) Pond 37 (Alameda), (b) Pond 32 (William Street), and (c) Pond 51 (RC Church Pond). Note difference in vertical scales.*

3.2.2. Seasonality of TP in the Extensive Pond Data (Dataset V)

Using this combined dataset, we made the following comparisons for surface water TP concentrations: 1) monthly means of all surface water TP concentration data, 2) means for winter, spring, summer, and fall, 3) means for cool open water periods (i.e., spring and fall) vs. summer months, and 4) analyses of variation in surface water TP during summer, with data that were available or could be gathered for most sites in the dataset. Datasets were not separated by year or group. We note that a few organizations have made extensive multi-year comparisons for individual ponds. Analysis of these data compared against climate data is ongoing but was beyond the scope of this project.

Pond surface water showed strong seasonal variations in TP concentration, with peak TP observed in winter and summer (Figure 3.2-2). Across sites, stormwater ponds exhibited highest TP concentrations during winter/ice-cover months (0.316 ± 0.064 mg/L) followed by summer months (0.270 ± 0.041 mg/L), with no statistical difference between the means. Fall TP (September 16 to November 30; 0.121 ± 0.014 mg/L) and spring TP (April and May; 0.117 ± 0.016 mg/L) was significantly lower compared to winter and summer TP, but did not significantly differ from each other. Additional summary statistics for this dataset are included in Table III-1.



n, sites	8	8	9	11	26	59	59	59	30	20	9	8
n, measurements	44	37	45	78	133	439	466	460	159	92	38	33

Figure 3.2-2 *Monthly mean ± SE surface TP concentrations (mg/L), associated number of sites represented in each month, and total number of measurements recorded in each month across all sites. Winter (December 1 to March 31), spring (April 1 to May 31), summer (June 1 to September 15), fall (September 16 to November 30).*

Out of the 27 ponds that contained data across two or more seasons, 16 showed evidence of consistent seasonal variation in TP in the epilimnion, although the amplitude varied strongly. These ponds with significant seasonal variation in TP showed highest peaks during winter/ice covered conditions and again in summer (Table III-2). For ponds with annual sampling (n = 8), ice covered winter dates also consistently (with exception of Pond B) exhibited the highest TP concentrations (Table III-2).

Because of the similarities in TP between spring and fall, data were combined as cool open water periods and compared to summer. Mean TP concentrations were always higher in the summer than in the cool season with the exception of three sites (Table III-3). Averaging across all sites, summer TP concentrations were 2.5 times higher than in the cool period (0.291 vs. 0.116 mg/L respectively; $p<0.01$; Figure III-1). TP concentrations in the summer season were more variable than in the cool season indicated by a higher average coefficient of variation (0.74 and 0.60 respectively; Table III-3). The magnitude of difference between the summer vs. cool season TP was weakly driven by pond duckweed cover ($R^2 = 0.16$; $p < 0.05$; $n = 26$; Figure III-2).

3.2.3. Seasonality of Water Quality in Monitoring / Intensive Sampling Ponds (Datasets II, III)

3.2.3.1. Seasonal Patterns: P, Duckweed, Stratification, and DO

Figure 3.2-3 illustrates the seasonal variation of surface total phosphorus, duckweed cover, anoxic fraction, and stratification strength (RTRM) at two ponds, Alameda (Site 37) and Arrow (Site 15), from routine sampling over the 2019 season (Jan – Nov). Alameda is a duckweed-dominated pond with heavy tree sheltering, while Arrow is a smaller and more open pond with very little duckweed cover; both have similar max depths around 2 m. These ponds show highest TP concentrations in winter and/or spring, prior to the start of the main monitoring program in late May, consistent with patterns observed in previous studies of over-winter increases in P (e.g., Taguchi et al. 2020a).

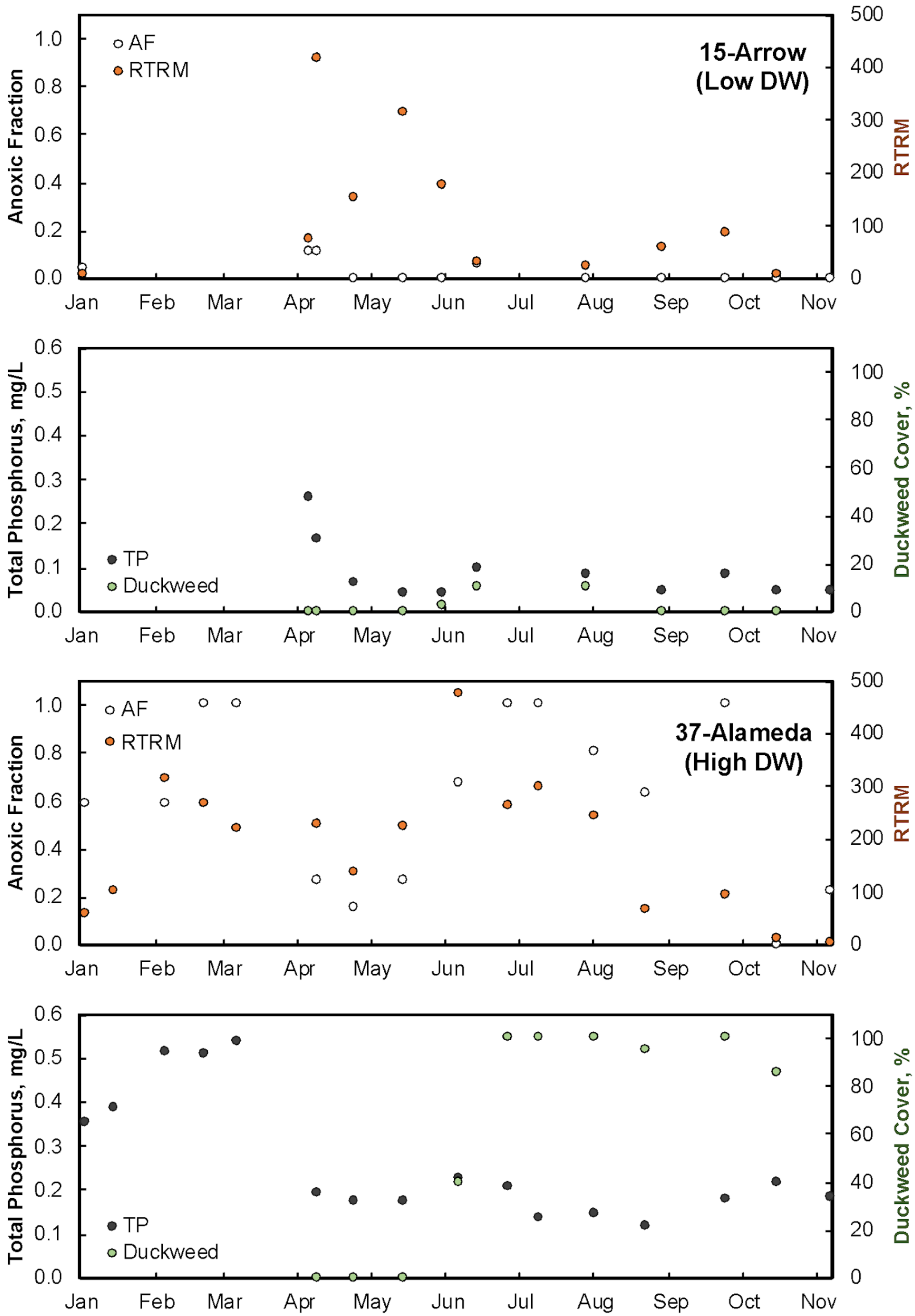


Figure 3.2-3 *2019 time series of anoxic factor (AF), duckweed cover, stratification (RTRM), and epilimnion total phosphorus at Arrow (Site 15) and Alameda (Site 37) ponds.*

DO dynamics (as assessed with anoxic fraction) were influenced by stratification strength and by duckweed cover. In particular, with the onset of heavy duckweed cover at Alameda in late June, DO decreased to anoxic levels (AF at or near 1.0) along with a drop in TP, suggesting at least some uptake by duckweed. Stratification (RTRM) also remained strong throughout the season at Alameda, enforcing anoxic conditions. Arrow, by contrast, showed low DO and low TP for much of the summer and early fall. Both ponds turned over around mid-September (note brief reduction in AF at Alameda and decline in RTRM) followed by a brief re-stratification; Alameda showed a rebound in TP coincident with a reduction in duckweed cover in October.

The contrast in stratification dynamics between the ponds are also worth noting. Both ponds accumulated road salt over the winter (mean conductivity > 1000 uS/cm through mid- to late-May at both sites), with a subsequent decrease in conductivity over the summer from flushing and diffusion. However, RTRM decreased rapidly in mid-summer at Arrow while at Alameda RTRM remained persistently high until fall turnover. This pattern at Alameda in particular suggests that early-season stratification from road salt may interact with effects of sheltering and perhaps duckweed cover to enhance stratification duration and strength, resulting in season-long suppression of DO.

Seasonal patterns in upper water temperature, duckweed cover, anoxic fraction, and stratification (RTRM) were observable across all intensively sampled ponds (Dataset III; Figure 3.2-4). As the main sampling effort did not begin until the end of May, the early season data are skewed towards a smaller set of ponds. The plot shows a gradual increase in water temperature from spring through midsummer and tracks air temperature through most of the season; a rapid increase in FFP cover also occurred through the month of June. Peak duckweed cover, anoxic fraction, and temperature occur during July and August, followed by a gradual decline in stratification, anoxic fraction, and water temperature into September and October. Most of the monitored ponds (Dataset II) turned over between Sep 9 and Sep 13, which can be seen in the low AF and RTRM values in October.

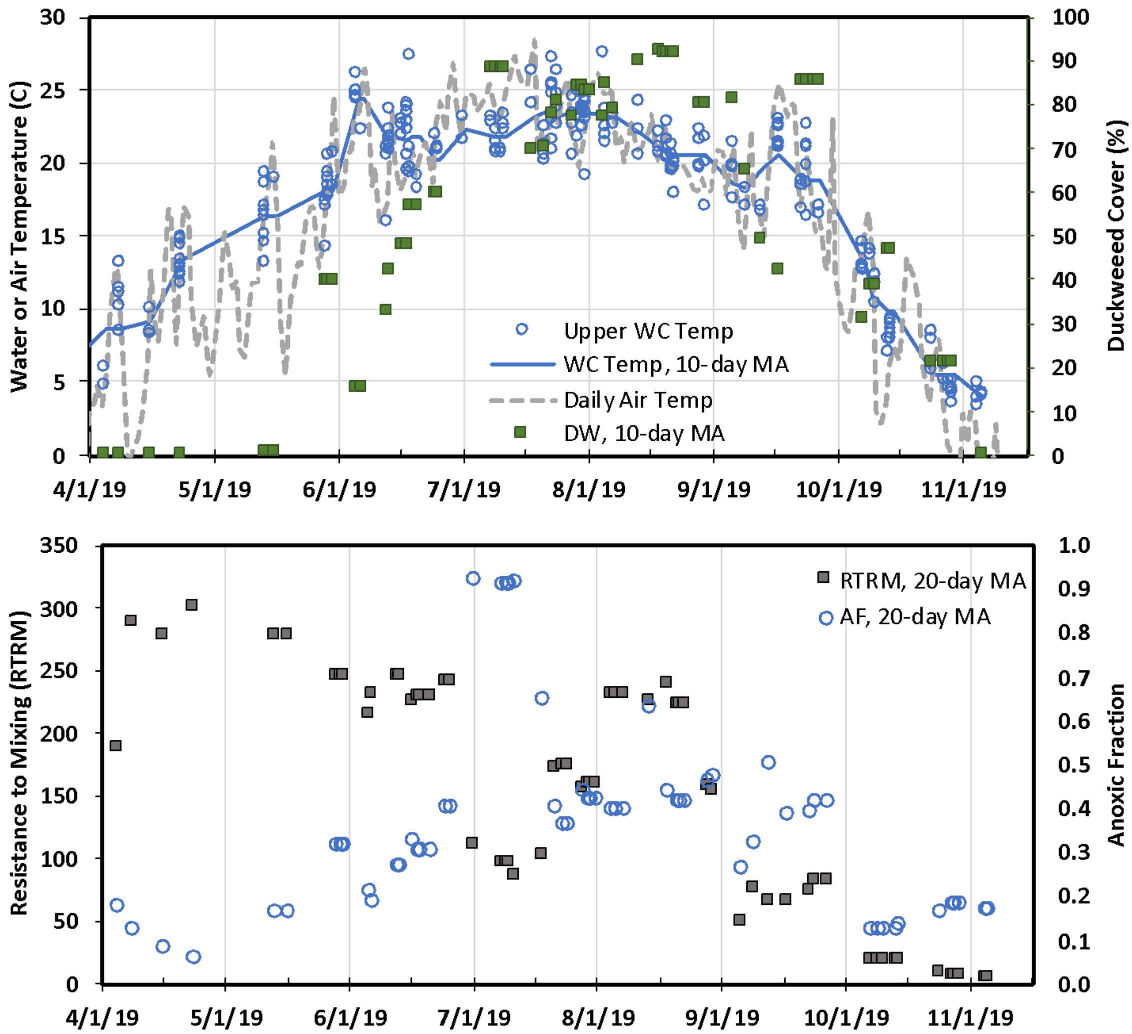


Figure 3.2-4 *Time series of water temperature, duckweed cover, resistance to mixing, and anoxic fraction across all Intensively Sampled and Monitored Ponds (Datasets II and III). Water temperature is average over the upper 50cm of water column, presented as a point for each pond and as a centered 10-day moving average. Duckweed cover (DW) is given as a 10-day moving average for ponds with median duckweed cover > 0 for the season. RTRM and Anoxic Fraction (AF) are presented as 20-day centered moving averages. Air temperature data are from Minneapolis-St. Paul International Airport.*

3.2.3.2. Patterns in Daily and Sub-Daily Observations

Thirteen of the ponds with continuous monitoring stations (Dataset II) produced usable data. Analysis of these data revealed several interesting patterns with respect to stratification and mixing at sub-seasonal (e.g., daily) time scales, though our interpretations are limited in that data from seven of the thirteen sites was collected after

Aug 27, generally within two weeks of fall turnover, due to difficulties with the initial instrumentation (See Methods).

Across these ponds, mixing frequency (as mixing events / days in monitored period, or $days^{-1}$) varied considerably depending on criteria and location (upper water column or whole water column), but ranged from 0.03 – 0.24 $days^{-1}$ (mean = 0.09 $days^{-1}$). Wind-mixing events tended to be slightly more frequent than runoff-mixing events (mean = 0.05 vs. 0.03 $days^{-1}$ across sites), with some events not falling into either classification due to simultaneously high winds and rainfall (or to thermally-induced mixing during the cool turnover period in fall). Fall turnover, defined as two consecutive days with daily mean water column (top-vs-bottom) RTRM < 10.0, was consistent across the ponds, occurring on Sep 8 or Sep 9 for 7 of the 13 monitored ponds. Four ponds turned over on Sep 13, while the most intensely-stratified (Sites 32 and 230) turned over on Sep 28 and Oct 3, respectively (Table 3.2-1). The table also shows the duration of anoxia at the sites, both as total number of days and as longest stretch with an anoxic fraction > 0.5.

Table 3.2-1 *Duration of anoxia at the Monitored and Intensive Sampling Ponds (Datasets II and III), and approximate date of turnover in 2019.*

Site	AF > 0.5 Total Days	AF > 0.5 Consecutive Days	Fall Turnover
15	0	0	9/8/19
32	137.5	105	9/29/19
37	105	105	9/13/19
40	0	0	9/9/19
42	0	0	9/8/19
51	107	107	9/9/19
74	0	0	9/13/19
79	0	0	9/8/19
83	0	0	NA
116	66.5	66.5	9/9/19
214	0	0	NA
217	15	15	9/13/19
229	88.5	61.5	9/8/19
230	117	117	10/3/19
231	115	115	9/13/19
233	125.5	125.5	NA
234	152	152	NA

An example time series of water level, rainfall, observed wind speed, and observed stratification (RTRM) are shown for two ponds, Kasota East (Site 116) and William Street (Site 32), over the period July 17 – Oct 16, 2019 in Figure 3.2-5. These ponds were both roughly the same depth (1.5 – 2 m), with the William Street pond being slightly smaller in a more sheltered setting. The plots show a slight contrast between the ponds in mixing frequency (drops in RTRM) and mixing depth (Top-vs-Bottom RTRM vs. Upper-Water-Column RTRM). In particular, the Kasota Pond turns over (Top-vs-Bottom RTRM ~ 0) several times from late August through October, including during the primary turnover (among the monitored ponds) in early September. William Street Pond, by contrast, was more strongly stratified (higher top-vs-bottom RTRM) and did not completely turn over during early September, instead remaining stratified until late September.

The plots (Figure 3.2-5) also illustrate the impact of wind and runoff on vertical mixing in the two ponds, and of calm, dry periods on the enhancement of stratification. For example, a windy series of days in late July reduced stratification strength (decreased RTRM) in both the whole water column and in the upper water column of both ponds in a "partial" mixing event, as the pond still remained stratified (RTRM >> 0). A large rainfall event on Aug 18, 2019 produced a similar drop in RTRM in both ponds, illustrating a "runoff" mixing event as incoming stormwater mixed the upper water column. By contrast, prolonged dry periods, such as early August and mid-September (evidenced by steadily decreasing water levels and low precipitation) resulted in increased stratification in the ponds (increased RTRM). The slope of the water level decrease during these dry periods is indicative of water losses in the ponds from evapotranspiration; note in particular the higher slope of water level time series during the August dry period compared to mid-September at William Street. Further assessment of this hydrologic information is found in Section 3.4.

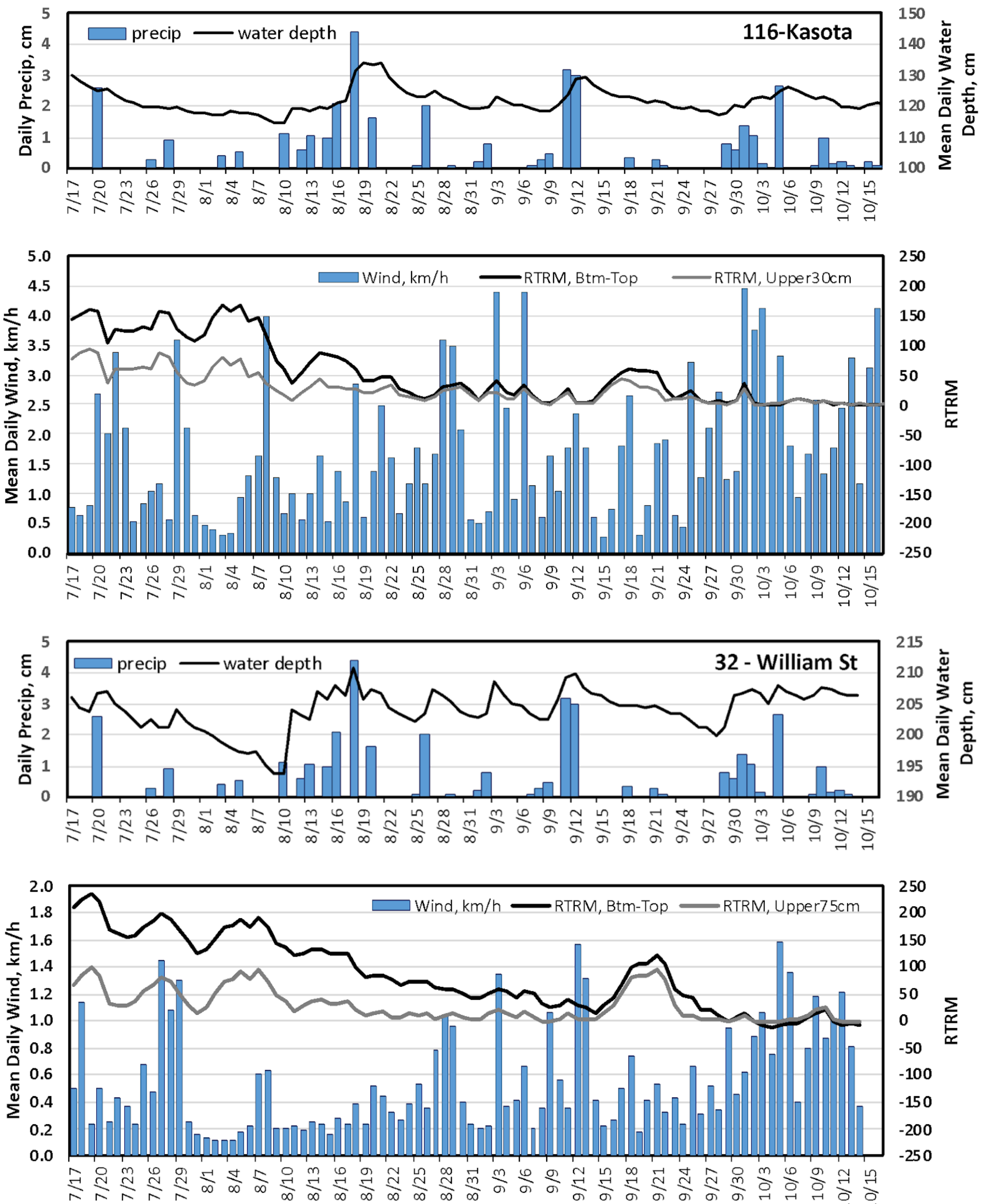


Figure 3.2-5 *Time series of daily-averaged data collected at monitoring stations (Dataset II) at Site 116 (top two plots) and Site 32 (Bottom two plots). Water depth, mean wind speed, and RTRM were measured at the sites; precipitation depth from Mpls-St. Paul Airport. RTRM shown as top-vs-bottom and for upper 30cm or 75cm of water column.*

3.3. Sediment Phosphorus Release and Characterization

3.3.1. Oxic and anoxic sediment phosphorus flux

The results of the laboratory P release study on pond sediment cores from seven ponds (part of intensively sampled and monitored ponds; Table 2.2-1) are shown in Figure 3.3-1, where the rate of change of phosphate concentration in the water column indicates phosphate flux from the sediments. Under oxic (aerated) conditions, an increase in water column phosphate concentrations was generally not observed, suggesting no phosphate release under those conditions. A small oxic phosphate flux was determined for Bren Pond sediments, possibly due to the mineralization of labile organic P in the sediments and mobilization into phosphate (Jensen and Andersen 1992).

The phosphate flux for the 7 studied ponds combined with 8 other ponds in the Twin Cities area (Dataset II and III; Table 2.2-1) indicate the wide range of anoxic sediment phosphate flux possible in ponds (Figure 3.3-2). All ponds can be expected to experience internal phosphorus release under anoxia to varying extent due to the mobilization of the redox-sensitive phosphorus in the sediment. In some ponds with a positive oxic phosphate release (Shoreview, Bren, and Point of France ponds), the sediments may release phosphorus due to the microbial degradation of organic phosphorus even under oxic conditions.

Once the air supply was switched off, the water column DO concentrations decreased due to the sediment oxygen demand in most pond sediment cores and achieved an anoxic state (DO <1 mg/L) over a period of 5 to 10 days (Figure 3.3-1). S_{max}, the maximum oxygen consumption by the biologically active sediments, ranged between 0.79 and 4.32 g/m^2/day in the sediment cores, with a relatively consistent mean S_{max} of between 1.12 and 2.79 g/m^2/day (see mean S_{max} in Figure 3.3-1). The concomitant phosphate release response to the lowered DO concentration was, however, not as consistent. Phosphate release was observed after 10 to 15 days of the air-off phase only in four ponds (Aquila Pond, BC-P4.10C, Bren Pond, and Pond 42). The sediment oxygen demand is related to aerobic microbial respiration rates. The phosphorus release from the sediments is related to anaerobic phosphorus release (Taguchi et al. 2020a); the relatively low flux from Pond K, Pond B, and Kasota East during this phase could be linked to greater oxygen availability at these sites although the mechanism driving lower flux are not clear. This may further indicate that the sediments had not previously been sufficiently exposed to a low oxygen concentration.

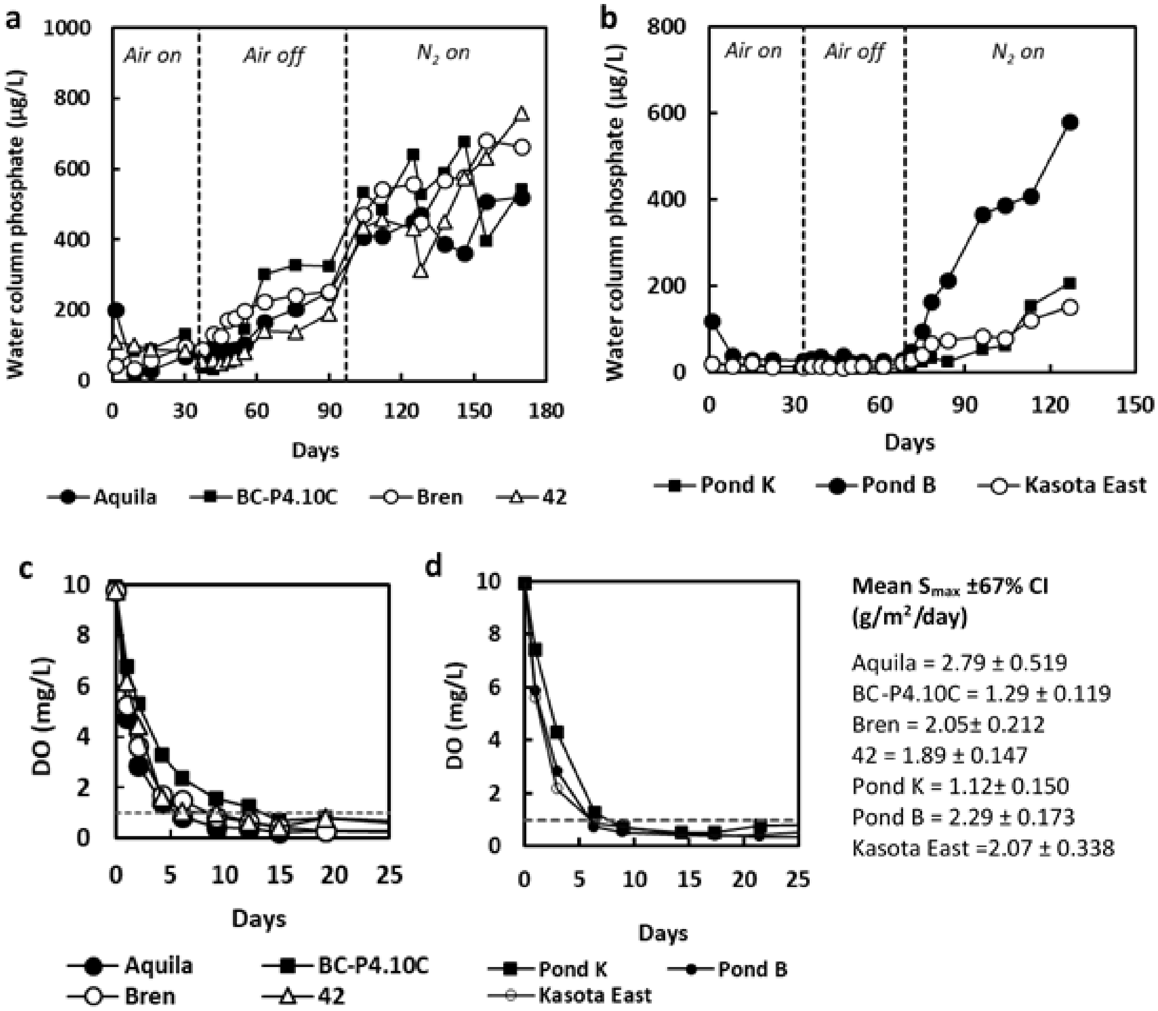


Figure 3.3-1 *Phosphate (SRP or PO_4-P) concentrations in the sediment-water columns from 7 ponds under the three phases of the laboratory phosphorus release study at 20 °C (a and b). The increase in phosphate concentration is due to sediment phosphorus release. The average concentration of five columns are plotted for each pond. The change in DO concentration 8 cm above the sediment during the air-off phase is shown in plots c and d (dashed line showing DO = 1 mg/L represents anoxic state). The sediment oxygen demands (mean S_{max} ± 67% confidence interval) determined for the pond cores are included.*

In the last phase with an anoxic water column maintained by nitrogen gas bubbling, sediment phosphate release was observed in all pond columns, including the three ponds that did not respond during the preceding air-off phase. It is possible that ~15 days of anoxic/anaerobic conditions in the water column was sufficient to develop an active anaerobic microbe population. Some variability in phosphate release rate was observed among the replicate cores collected from a pond, suggesting differences in sediment characteristics within a pond. The mean anoxic sediment phosphate flux was variable among the 7 ponds, ranging from 1.32 mg/m^2/day (Pond K) to 7.81 mg/m^2/day (Pond B).

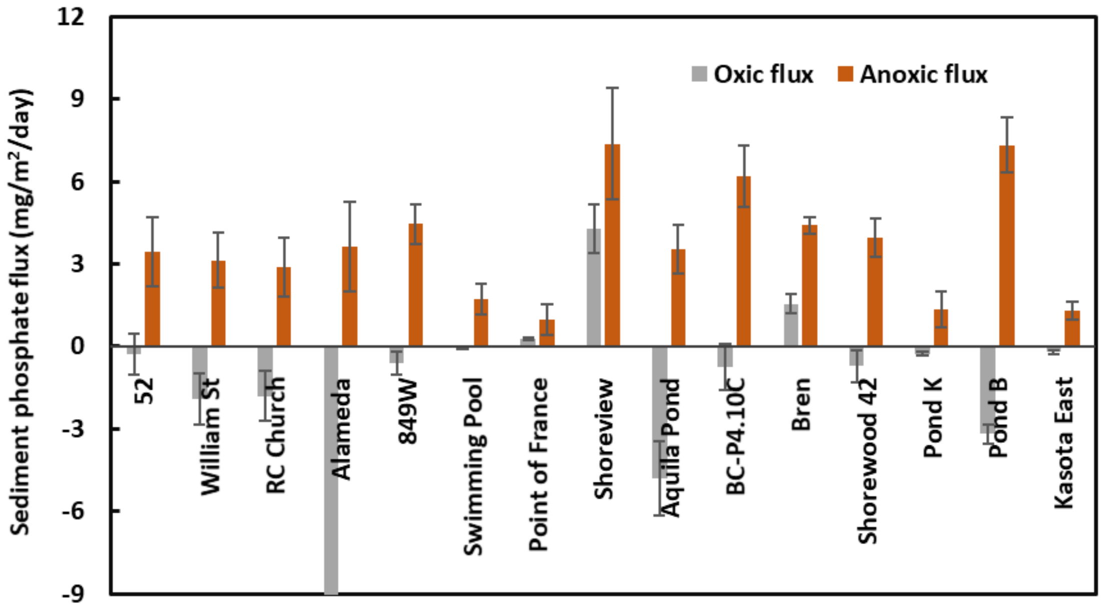


Figure 3.3-2. *Oxic and anoxic sediment phosphate flux determined for 15 ponds in the Twin Cities Metro area. Error bars are 67% confidence intervals of the mean. The oxic flux for Alameda Pond (-19.8 ± 3.37 mg/m^2/day) is out of the y-axis scale in the plot and may be subject to experimental error. Data for Ponds 52, 849W, William St, RC Church, Alameda, Swimming Pool, Point of France, and Shoreview are from previous studies.*

3.3.2. Sediment phosphorus composition

The fractionation of the total sedimentary P pool into releasable (bioavailable) and unavailable forms provided the potential for P release to occur under changing DO conditions. In the seven ponds analyzed in the current study, the redox-sensitive P forms (i.e., loosely-bound and iron-bound P) that are mostly associated with sediment phosphorus release during anoxia, ranged between 0.042 to 0.373 mg/g in the upper 4 cm of sediments (see Figure V-1 in Appendix V for concentration profiles). Labile organic P, that has the potential to become bioavailable after organic P is decomposed by bacteria, ranged between 0.051 to 0.364 mg/g. The mobile P concentration (loosely-bound P + iron-bound P + labile organic P) was the predominant P fraction (50 to 68% of total P) in two of the seven ponds.

Figure 3.3-3 illustrates the variability and the relative importance of the releasable (bioavailable) and unavailable P fractions in 15 ponds. The total P mass (sum of 6 fractions) in the pond sediments range from 0.54 to 1.7 mg/g, indicating the varying P quality of the sediments which is mostly related to the external P input to the ponds. When normalized by the TP concentration, the redox-sensitive P constituted 7.6 to 29% of the total P and the labile organic P constituted 9.3 to 43% of the total P. This means about 17% to 68% of the sediment P is potentially mobile, i.e., releasable under anoxia and by microbial decomposition, in the pond sediments. The mobile phosphorus forms thus appear to be important in several ponds and may play the largest role with respect to phosphorus dynamics in ponds. The ponds with relatively low anoxic phosphate flux were Point of France, Pond K, and Kasota East. There is no obvious correlation with the P fractions in the sediments. Pond K has the lowest and the Point of France Pond has the third lowest mobile P, but Kasota East has a higher mobile P in the sediments than many of the ponds and the Shoreview pond, with the second lowest mobile P in the sediments, has a relatively high sediment phosphorus release rate (Figure 3.3-2).

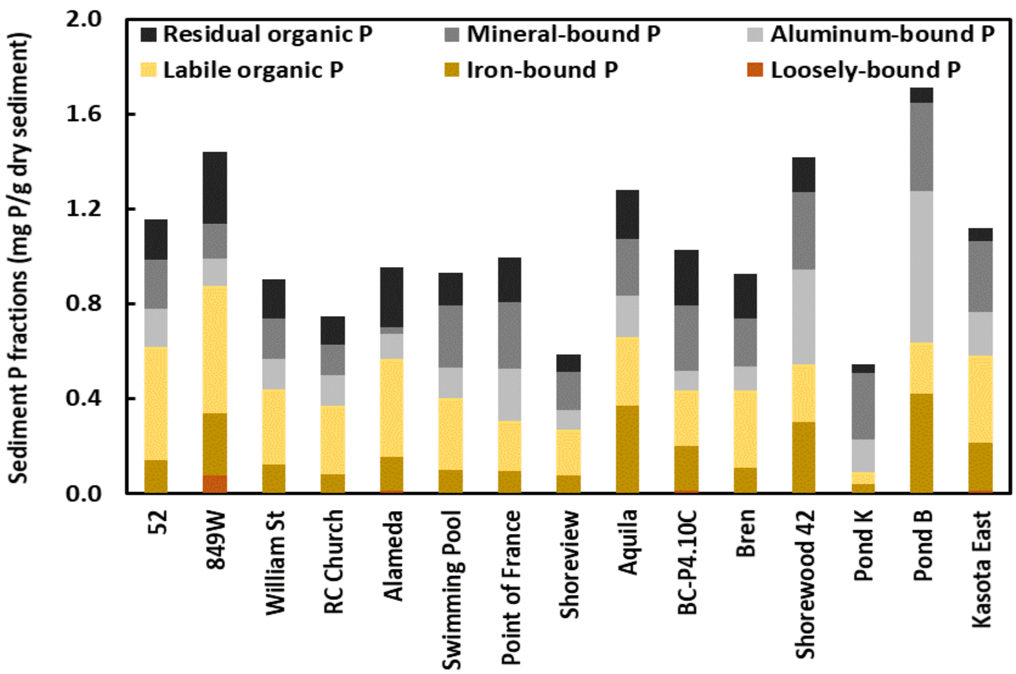


Figure 3.3-3. *Concentrations of the available and unavailable phosphorus (P) fractions in the sediments of the 15 ponds. The y-axis shows concentrations of the various P fractions on a dry-weight basis. The loosely-bound P and iron-bound P are releasable under anoxic conditions and the labile organic P can become bioavailable even under anoxic conditions, and together constitute the mobile phosphorus fraction.*

The organic matter content in the 15 ponds range from 13 to 33% (dry mass basis) in the upper 4 cm of sediments, with one exception of a very high organic content of 79% (849W). The high organic matter input to ponds, from sources such leaf litter and grass clippings, can result in the accumulation of organic P in the sediments (Janke et al. 2017, Song et al. 2017, Frost et al. 2019) that may eventually decompose and release P (Gächter et al. 1988, Golterman 2001).

3.3.3. Regression Analysis of Sediment Variables

A simple linear regression analysis was first performed to identify the important pond sediment variables impacting internal P release in ponds. For this analysis, data for 51 sediment cores from 14 ponds that have paired sediment P flux and detailed sediment characteristics were considered to investigate if significant relationships between anoxic sediment phosphate flux, sediment concentrations of bioavailable P species and total sediment P (top 4 cm of sediments), organic matter content, sediment oxygen demand (S_{max}), and anoxic factor could be established. The anoxic factor was derived for the individual sediment cores used in the laboratory study based on the DO concentration in the overlying water at the location of the cores in the respective ponds (36 cores from 10 ponds). The mean anoxic factor was calculated for the monitored period (May 29 – Nov 6) and the summer anoxic factor was calculated for the summer period (June 1 – Sep 13). Linear regression models for sediment phosphate flux and related variables are summarized in Table 3.3-1. Scatter plots showing the phosphate flux and sediment variables are included in Figure V-2 in Appendix V with correlation coefficients for the entire suite of sediment-related variables in Table V-1.

The redox-sensitive P fractions in the sediments (i.e., iron-bound P and redox-P) yielded similar correlations with the anoxic sediment phosphate flux (R^2 = 0.38 and 0.37, respectively) since the loosely-bound P concentrations were very low. The existing models for phosphorus release from lake sediments are also based on the redox-P fraction (Nurnberg 1988, Pilgrim et al. 2007, James 2011). Labile organic P alone was not related to anoxic flux (R^2 = 0.06), likely because this P form is mobilized at a slow rate only after microbial degradation occurs. The correlation between total mobile P (i.e., redox-P + labile organic P) and total P in the sediment with the anoxic flux (R^2 = 0.29 and 0.33, respectively) were similar to that of redox-P.

Table 3.3-1 *Simple linear regressions for sediment phosphate flux and sediment mobile phosphorus derived using sediment cores collected from the intensively sampled and monitored ponds (Datasets II and III). Only significant ($p < 0.05$) models are shown. AF = anoxic factor, where mean AF is for the monitoring period (May 29 – Nov 6) and summer AF is for the summer period (June 1 – Sep 13); SOD = Sediment Oxygen Demand (S_{max} determined for the laboratory sediment cores); Redox-P = Loose-P + Iron-bound P; Mobile P = Redox-P + Labile organic P. Sediment P concentrations are on a dry mass basis. N = 51 cores for most models; N = 36 for models with AF relationship.*

(1) Anoxic sediment phosphate flux (mg/m^2/day)

Predictor	Slope (SE)	Intercept (SE)	R^2	p-value
Iron-bound P (mg/g)	11.7 (2.13)	1.36 (0.448)	0.38	1.43E-06
Redox-P (mg/g)	10.8 (2.02)	1.43 (0.448)	0.37	2.44E-06
Mobile-P (mg/g)	5.93 (1.33)	0.449 (0.713)	0.29	5.03E-05
Total P (mg/g)	3.96 (0.800)	-0.788 (0.882)	0.33	9.18E-06
Mean AF	2.98 (0.762)	1.36 (0.538)	0.31	4.15E-04
Summer AF	2.68 (0.706)	1.24 (0.577)	0.30	5.74E-04

Not significant: Sediment organic content and SOD

(2) Sediment mobile P (mg/g)

Predictor	Slope (SE)	Intercept (SE)	R^2	p-value
Sediment organic matter (g/g)	0.979 (0.142)	0.212 (0.0461)	0.49	1.03E-08
SOD (g/m^2 day)	0.057 (0.0175)	0.331 (0.0561)	0.18	1.87E-03

The mean anoxic factor and the summer anoxic factor were both found to be correlated with the anoxic phosphate flux (R^2 = 0.31 and 0.30, respectively, after excluding Pond B which was deemed to be an outlier), confirming the importance of the exposure of pond sediments to oxygen (or lack thereof) in mobilizing certain sediment phosphorus species to the pond water column. It must be noted that the summer anoxic factor for the sediment coring locations was either 0 (DO > 2 mg/L; e.g., Swimming Pool and Point of France) or 1 (DO < 2 mg/L; e.g., 849W, Alameda, Bren, 42 and others; Figure V-2 in Appendix V), suggesting the persistence of low DO conditions in some ponds than others during entire summer. However, neither sediment organic matter nor sediment oxygen demand showed a significant relationship with anoxic factor for the 10 ponds analyzed, indicating the role of other variables such as duckweed cover, pond mixing, and canopy cover in determining the pond anoxic status (see Table 3.4-2). However, sediment organic matter content showed a strong relationship with labile organic P and hence mobile P (R^2 = 0.30 and 0.32, respectively; Table V-1 and Figure V-2 in Appendix V), suggesting an indirect interaction with anoxic phosphate flux. The sediment oxygen demand did not exhibit an obvious relationship with SRP flux or sediment P fractions and only a low correlation with sediment organic content (R^2 = 0.24).

The relationship between the anoxic sediment phosphate flux determined in the laboratory cores and the in situ pond phosphorus water quality was also examined. The mean anoxic phosphate flux (calculated using all 5 cores collected in a pond) alone did not show a strong relationship with the sampled concentrations of TP and SRP in the pond surface water during the summer period (R^2 = 0.045 and 0.022, respectively, not shown). The net sediment phosphate flux, determined by scaling the anoxic and oxic flux values by the summer anoxic factor (AF) for the ponds using:

$$Net\ phosphate\ flux = Anoxic\ flux\ \times AF + Oxic\ flux\ (1 - AF)$$

The calculated net flux also did not present an obvious relationship with the summertime surface TP and SRP concentrations in the ponds ($R^2 < 0.0006$, not shown).

The mean sediment phosphate flux values did not correlate strongly with pond geometry (R^2 = 0.04 for maximum pond depth and R^2 = 0.06 for DA:SA after removal of outlier Pond B with DA:SA = 698). The lack of a significant direct relationship of internal phosphate release with phosphorus concentrations and pond geometry suggests other factors such as pond hydrodynamics, duckweed presence, and watershed loading are likely playing a larger role in controlling the phosphorus concentrations in the ponds.

Based on the results of the simple linear regression analysis, the releasable sediment P fractions (redox P and mobile P), anoxic factor, sediment TP, and sediment organic matter content emerge as the important predictors of anoxic phosphate flux from the pond sediments. Table 3.3-2 summarizes multivariate models for anoxic sediment phosphate flux estimation using the identified predictors, along with plots showing the performance of the models (Figure 3.3-4). The sediment phosphorus fractions have been traditionally used in the lake literature for predicting P flux.

Table 3.3-2 *Models obtained by multiple regression analysis of anoxic sediment phosphate (SRP) flux (mg/m²/day), various sediment phosphorus fractions (mg/g dry), sediment organic content (%), and summer anoxic factor (AF) in the Intensive and Monitoring Ponds Datasets (II and III) (n = 51, except when anoxic factor is included where n = 36). Some SLR models from Table 3.3-1 are included for comparison with the multivariate models. Relative standard error is the standard error (SE) divided by the mean anoxic SRP flux.*

	Model		**Adjusted R^2**	**SE**	**Relative SE**
(a)	Anoxic SRP flux =	10.8 x Redox-P + 1.43	0.35	1.87	0.536
(b)	Anoxic SRP flux =	5.93 x Mobile-P + 0.45	0.27	1.99	0.569
(c)	Anoxic SRP flux =	-0.00103 x Organic matter + 3.98 x Sediment TP - 0.778	0.31	1.94	0.556
(d)	Anoxic SRP flux =	2.68 x Summer AF + 1.242	0.30	1.83	0.524
(e)	Anoxic SRP flux =	-0.00261 x Organic matter + 2.70 x Sediment TP + 3.15 x Summer AF - 1.96	0.41	1.73	0.494

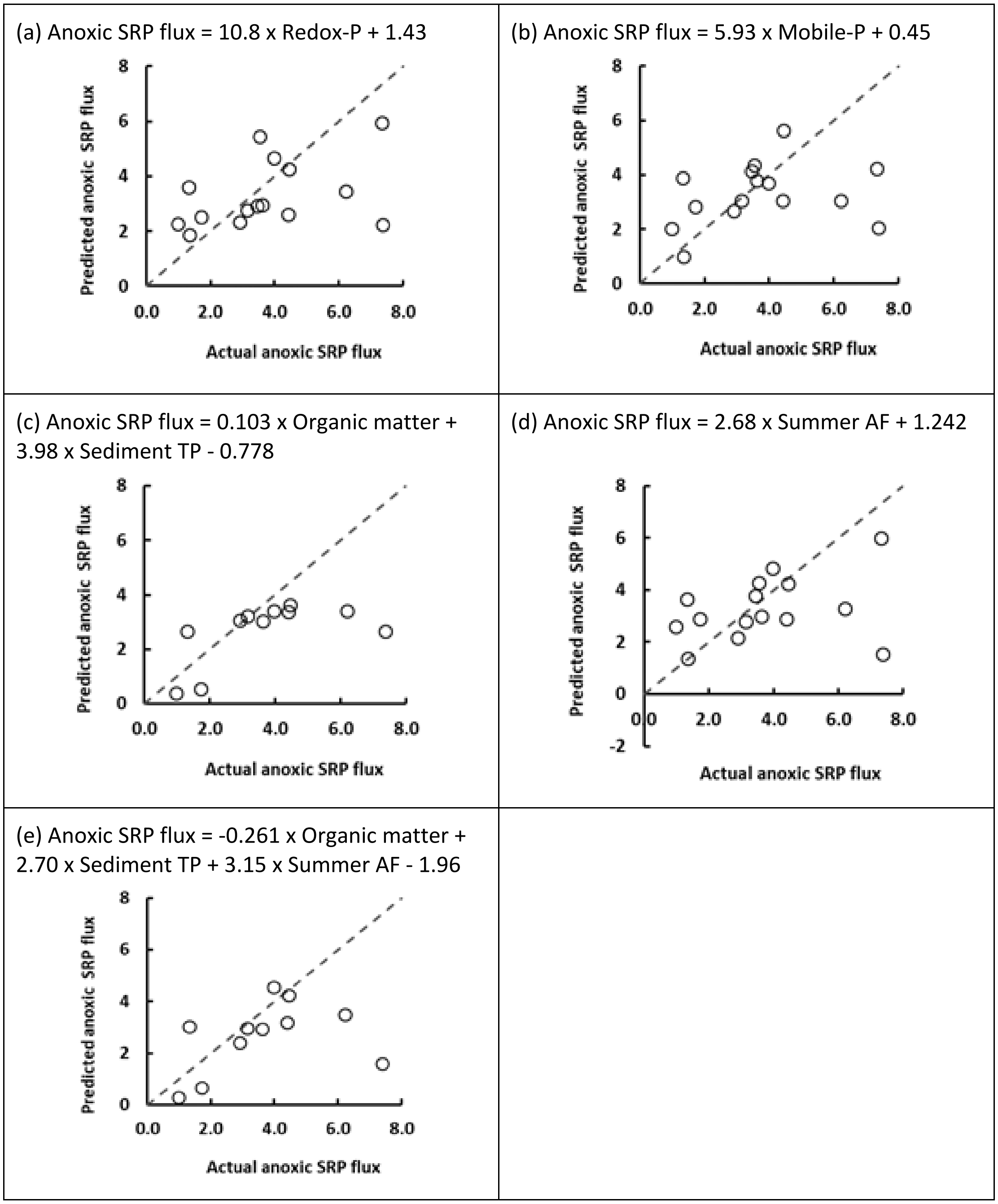


Figure 3.3-4 *Plots showing the anoxic sediment SRP flux predictions of the multivariate models in Table 3.3-2. The dotted line represents 1:1 slope to illustrate the performance of the models in predicting the sediment SRP flux when compared to the flux determined in pond sediment cores in the laboratory.*

A comparison of the regression models for ponds was made to those developed for lakes (Nürnberg 1988). The only regression model that worked well used the iron-bound P as a predictor of anoxic phosphorus flux. Our data for ponds are plotted with Nürnberg's model for lakes in Figure 3.3-5 (SE of the estimates = 2.70 for the ponds data fit to the lake model).

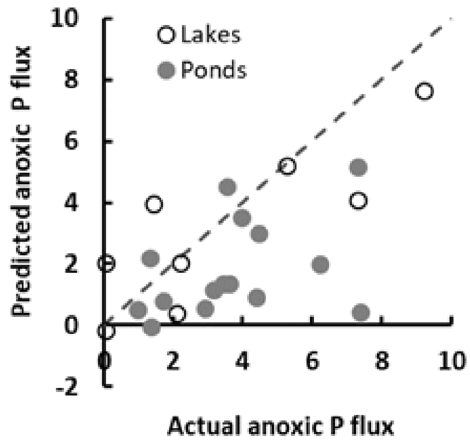


Figure 3.3-5 *Data fit of pond sediment SRP flux data to the regression model developed for lakes (TP flux = 13.72 × Iron-bound P – 0.58; Nürnberg 1988). The mean TP flux plotted for the lakes are from Nürnberg 1988 (n = 8, depicted as open circles). The 1:1 line (dashed line) is shown. The Shoreview Pond is an outlier with very low iron-bound P but high anoxic P flux determined.*

3.4. Regulators of Pond Phosphorus Water Quality

3.4.1. Controls of Pond TP in the Extensive Pond Dataset (Dataset V)

We applied simple linear regression analysis to examine the effects of pond and site features on mean surface TP concentrations in the Extensive Dataset (V). Data were not available for many parameters we examined in this project (Table I-1) across all sites. For this dataset, we were limited in examining relationships with data common to most sites including pond age and size, percent duckweed cover, watershed area and percent canopy cover within a 50 m buffer from the pond. Of these, only duckweed cover and percent canopy cover within a 50 m buffer were positively related to surface TP concentrations (Table 3.4-1). Watershed area, and pond age and size spanned wide ranges (Table I-1).

Table 3.4-1 *Simple linear regression analysis results between mean surface TP vs. percent pond duckweed and canopy cover for extensive dataset.*

Predictor	Slope (SE)	Intercept (SE)	R^2	p-value
% Duckweed cover	0.004 (0.001)	0.130 (0.048)	0.28	0.0001
% Canopy cover	0.502 (0.169)	0.031 (0.083)	0.15	0.0045

3.4.2. Controls of Pond TP in the Survey Pond Dataset (Dataset IV)

3.4.2.1. Correlation Analysis and Multiple Linear Regression (MLR)

Pearson's correlation analysis revealed highly significant relationships of surface TP concentrations with anoxic factor, canopy cover and mean water column DO across all three datasets (Table 3.4-2). Duckweed cover, maximum depth, and mean water column conductivity also exerted significant influence over TP, but not across all three datasets at the same time. Increasing duckweed cover, anoxic factor and canopy cover all led to significant increases in surface TP concentrations in all cases, except duckweed during survey I (Table 3.4-2). Survey I was conducted in the beginning of summer when conditions are not yet ideal for duckweed to be present or take off in the ponds to be able to exert a significant influence over pond TP. 25 out of 46 ponds had zero duckweed coverage, while only one pond (i.e., 849W) observed 100% duckweed coverage, and mean coverage across sites was 19% in survey I. In contrast, duckweed coverage had a strong relationship with TP in survey II, which occurred in later part of summer, where mean duckweed coverage across sites was 40% and 18 out of 63 sites had coverage of 90% or greater. Maximum depth and mean water column DO and conductivity showed negative correlation with surface TP. Predictor variables that significantly influenced surface TP concentrations and a few non-significant predictors are plotted in Figure 3.4-1. We found no clear or statistically significant relationships between surface TP concentrations and variables such as pond age, mean depth, pond area, sediment TP, sediment organic matter, mean RTRM and change in RTRM, and watershed area and watershed area: pond area.

Table 3.4-2 *Pearson correlation (Pearson r, p-value) between surface TP concentrations vs. pond and watershed features for combined survey (I and II) and individual survey I and II. AF = anoxic factor, RTRM = relative thermal resistance to mixing, DA:SA = drainage area: surface area, DW = duckweed, OM = organic matter, DO = dissolved oxygen (water column mean). N ranged from 24-63 for relationships. Significance of regressions:* * $p \leq 0.05$, ** $p \leq 0.001$ *(r-values shown in bold text in table).*

	[1]Surface TP			**[1]DW cover**			**[1]AF**		
	I & II	**I**	**II**	**I & II**	**I**	**II**	**I & II**	**I**	**II**
Age	0.26	-0.08	0.27	0.33	**0.47***	0.13	**0.54***	**0.66***	0.31
[1]AF	**0.69****	**0.72****	**0.43***	**0.85****	**0.64****	**0.82****	1.00	1.00	1.00
Canopy cover	**0.47****	**0.35***	**0.51****	**0.42***	**0.32***	**0.42****	**0.49***	**0.63***	0.23
DA:SA	0.16	0.15	0.17	-0.32	**-0.35***	-0.18	-0.11	-0.08	-0.10
Drainage area	0.02	0.0002	-0.0019	-0.27	-0.24	-0.19	0.14	0.17	-0.08
[1]DW cover	**0.53****	0.24	**0.52****	1.00	1.00	1.00	**0.85****	**0.64****	**0.82****
Max depth	**-0.44***	**-0.51****	-0.23	-0.26	-0.13	-0.18	-0.10	-0.11	-0.03
[1]Mean conductivity	**-0.38***	**-0.32***	-0.22	**-0.32***	-0.23	**-0.26***	-0.27	-0.22	-0.16
Mean depth	0.16	0.15	0.15	-0.07	-0.21	0.19	0.08	0.22	0.05
[1]Mean DO	**-0.57****	**-0.37***	**-0.36***	**-0.68****	**-0.40***	**-0.71****	**-0.81****	**-0.78****	**-0.82****
[1]Mean RTRM	-0.04	----	----	0.07	----	----	-0.15	----	----
[1]Mean temp	-0.24	-0.13	-0.14	**-0.45***	-0.14	**-0.54****	**-0.46***	-0.28	**-0.66****
Pond area	-0.05	-0.11	-0.06	0.25	0.19	0.11	0.02	0.01	-0.14
RTRM change (I-II)	0.11	----	----	-0.06	----	----	-0.09	----	----
RTRM (Top-Btm)	----	-0.25	-0.24	----	-0.24	**-0.28***	----	-0.19	-0.12
Sediment OM	0.08	-0.25	0.32	0.38	0.39	0.34	0.50	0.43	**0.46***
Sediment TP	0.20	0.13	0.21	0.16	0.19	0.14	0.10	0.19	0.04
[1]Surface TP	----	----	----	**0.53****	0.24	**0.52****	**0.68****	**0.72****	**0.43***

[1]averaged for combined surveys (I and II)

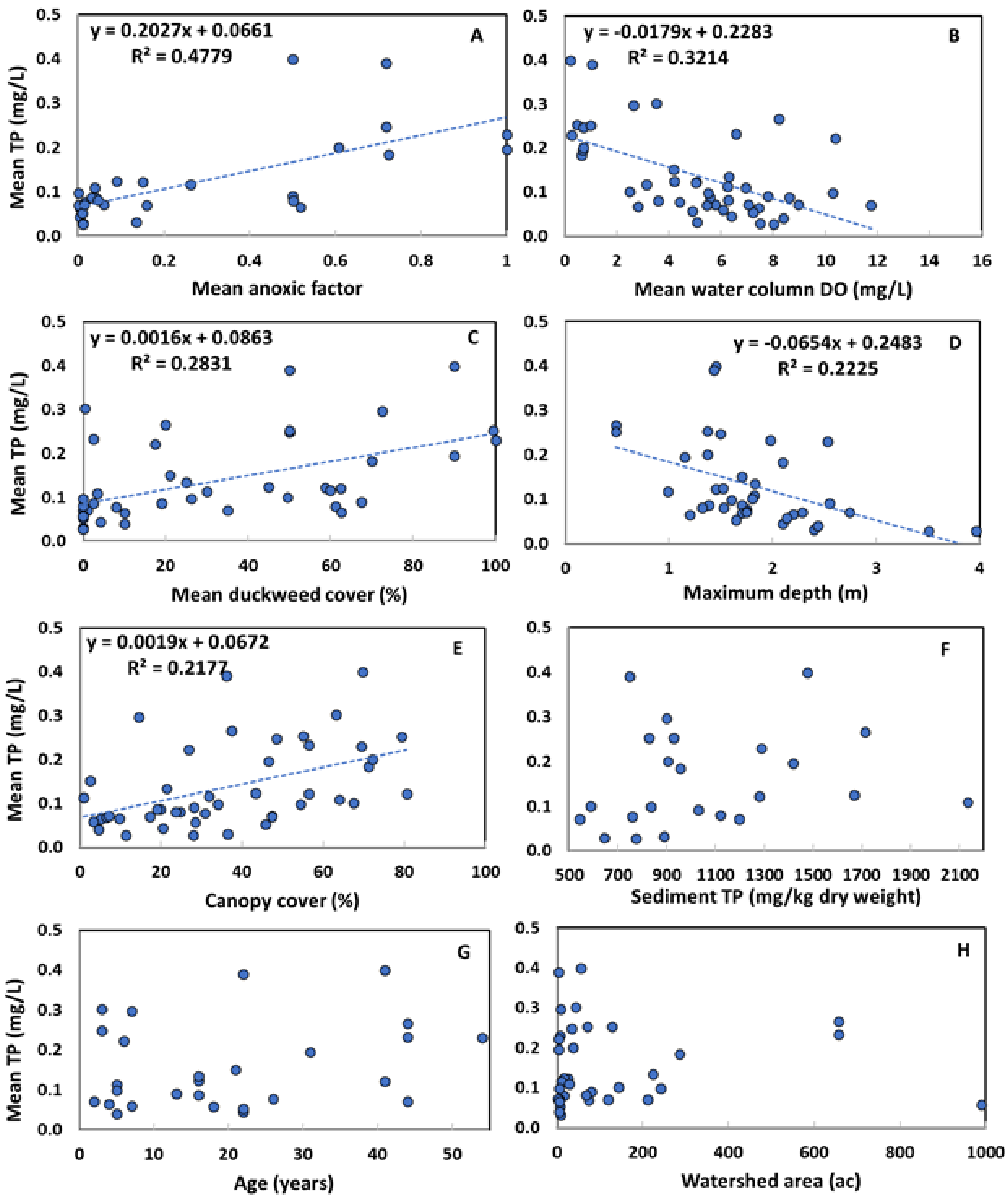


Figure 3.4-1 *Linear regression results for the combined survey dataset. Plots show relationships between mean surface TP concentrations (mg/L) with (A) mean anoxic factor, (B) mean water column DO (mg/L), (C) mean duckweed cover (%), (D) maximum depth (ft), (E) canopy cover (%), (F) sediment TP (mg/kg dry weight), (G) pond age relative to year 2019, and (H) watershed area (ac). Relationships of mean surface TP concentrations were significant with mean anoxic factor (p <0.0001), mean water column DO (p <0.0001), mean duckweed cover (p <0.001), maximum depth (p <0.01), and canopy cover (p <0.01).*

From the MLR analysis, the top 5 models ranked by AICc as the best predictors of pond surface TP concentrations for the three survey datasets are shown in Table 3.4-3. Pond area, canopy and duckweed were the variables that were identified as having the most dominant influence on TP when considered with other factors in MLR for the combined survey dataset. These three variables also appear to dominate in the best models identified for survey I and II. Pond area, however, was not a significant predictor of TP by itself in any of the survey datasets from SLR analysis (Table 3.4-2). Additional variables that were significantly related to surface TP in the individual surveys were pond maximum depth and RTRM(Top-Btm) when combined with other predictors. While pond maximum depth by itself was also significant in SLR analysis, RTRM(Top-Btm) by itself did not have a significant relationship with TP, but appears important when considered with maximum depth, canopy and duckweed for the individual surveys as shown by MLR (Tables 3.4-2, 3.4-3).

Table 3.4-3 *Multiple linear regression models for explanation of variance in surface TP concentrations across stormwater ponds as a function of pond and watershed characteristics for the three survey datasets.*

Model, Combined Surveys	**AICc**	**adj R^2**	**p-value**
size + canopy + duckweed	-97.50	0.37	<0.0001
canopy + duckweed	-95.10	0.32	<0.001
duckweed	-93.17	0.27	<0.001
size + duckweed	-92.86	0.28	<0.001
size + canopy	-92.13	0.23	<0.01
Model, Survey I	**AICc**	**adj R2**	**p-value**
max_depth + canopy	-89.30	0.43	<0.0001
size + max_depth + canopy	-88.07	0.44	<0.0001
max_depth + canopy + duckweed	-87.87	0.43	<0.0001
max_depth + canopy + RTRM(Top-Btm)	-87.31	0.43	<0.0001
size + max_depth + canopy + duckweed	-86.70	0.44	<0.001
Model, Survey II	**AICc**	**adj R2**	**p-value**
max_depth + canopy + duckweed	-74.79	0.31	<0.001
max_depth + duckweed	-73.39	0.26	<0.001
max_depth + canopy + duckweed + RTRM(Top-Btm)	-72.79	0.30	<0.001
size + max_depth + canopy + duckweed	-72.76	0.30	<0.001
size + canopy + duckweed	-72.27	0.38	<0.0001

3.4.2.2. Categorical Analysis of TP Concentrations

Given the high variability in pond TP, we used comparisons of categorical independent variables to develop indicators of pond TP. Mean surface TP concentrations were significantly higher (2 times) in ponds with high and medium levels of duckweed coverage than ponds with low duckweed coverage (Figure 3.4-2). While significantly greater TP concentrations was also observed in ponds with high levels of anoxic factor than medium and low levels (2 and 3 times higher respectively), ponds surrounded with high levels of canopy cover also exhibited significantly greater TP concentrations than medium and low levels of watershed canopy (1.5 and 2 times higher respectively). Distribution of mean TP concentrations were more variable at higher levels of duckweed and canopy coverage, and anoxic factor. Contrasting to these trends, mean TP concentrations significantly decreased at medium and high levels of water column DO and maximum depth (Figure 3.4-2). While no consistent trend in TP was observed with varying pond age and pond area, an increasing trend was observed at higher levels of sediment TP content.

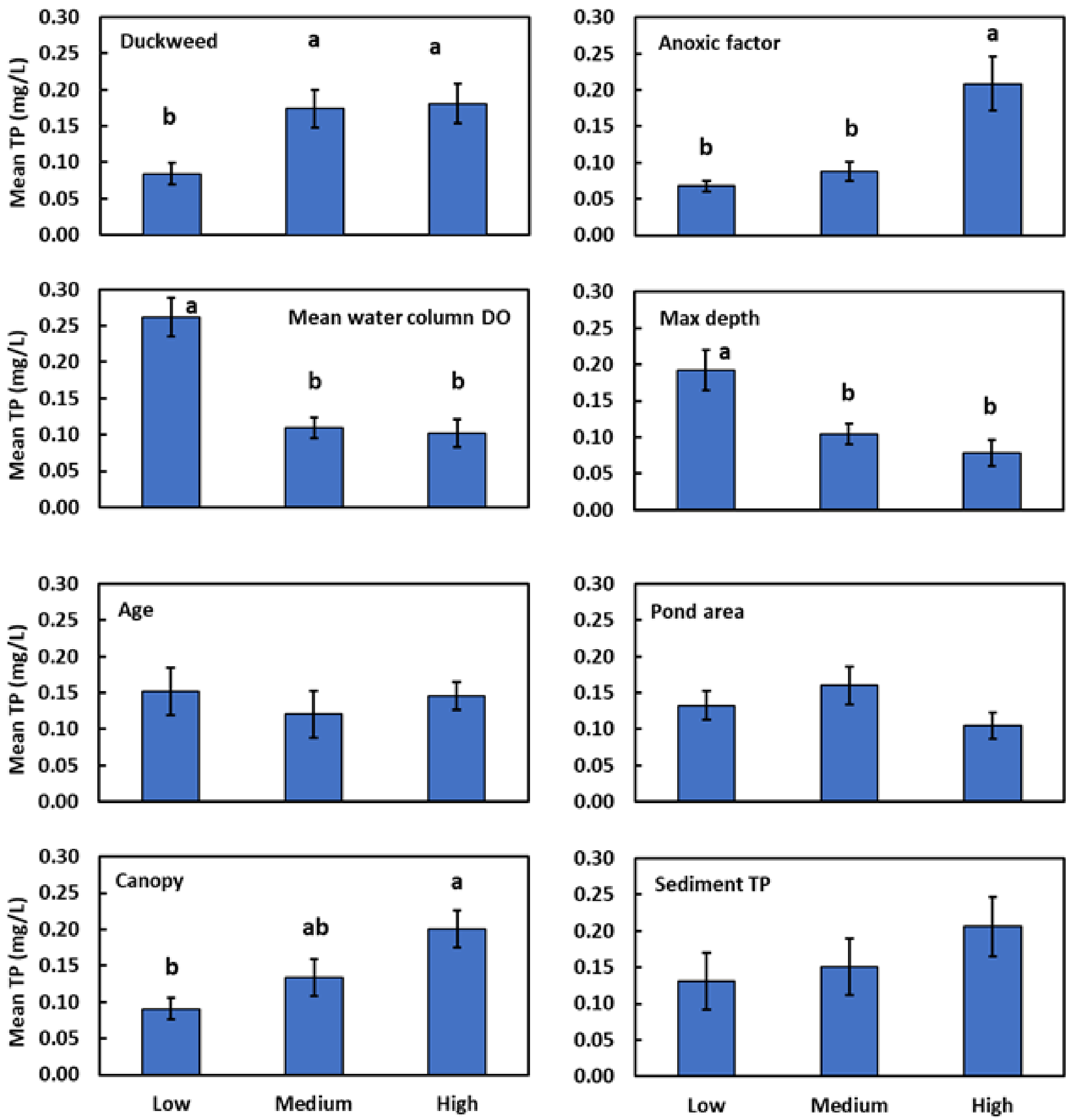


Figure 3.4-2 *Mean ± S.E. surface TP concentrations (mg/L) observed across survey ponds at varying levels of duckweed coverage, anoxic factor, mean water column DO, maximum depth, pond age, pond area, canopy cover, and sediment TP for the combined survey dataset. Lowercase letters indicate significant differences among the variable classes at p <0.05. Refer to Table II-2 for details on range, mean value and sample size of each level within each variable type.*

3.4.2.3. Controls of pond dissolved oxygen

Given the strong role of DO in the multivariate results, we conducted additional analyses to help understand controls over DO on survey ponds. We examined the relationship between paired mean anoxic factor and mean duckweed cover (%) observed for combined survey dataset (n=46). Mean values of anoxic factor and duckweed cover were derived by averaging paired data across Survey I and II observations. Mean anoxic factor and mean duckweed cover were highly significant (R^2=0.76, y = 7E-05x^2 + 0.003x + 0.0227, p <0.0001) for the combined survey dataset (Figure 3.4-3). Relationship between these two variables were also highly significant when examined for survey I (R^2 = 0.48, p = 0.0002) and survey II alone (R^2 = 0.71, p < 0.0001).

Regression results also indicated that anoxic factor was positively affected by pond canopy cover, and sometimes pond age (combined surveys and survey I) and sediment OM (survey II only) (Table 3.4-2). These results also showed duckweed cover increased significantly with increasing anoxic factor, canopy, surface TP and decreasing mean water column DO (Table 3.4-2).

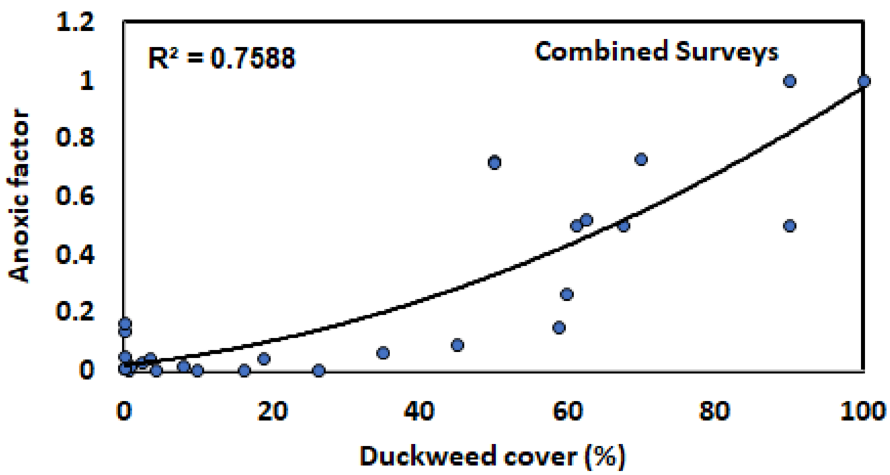


Figure 3.4-3 *Polynomial regression between paired mean anoxic factor and mean duckweed cover (%) observed for combined survey dataset (n=46).*

3.4.3. Controls of Pond P in the Monitoring / Intensive Sampling Ponds (Datasets II and III)

Results of the analysis of correlation (Pearson's r, p-value) between pond water quality variables (surface and hypolimnion TP, TDP, SRP, PP, and chl-a, as well as duckweed and anoxic factor) and factors related to oxygen, stratification, vegetation, and geometry are presented in Table IV-1 in the Appendix. Simple linear regression models for the strongest individual predictors of TP, duckweed, and anoxic factor are shown in Table 3.4-4. Multiple linear regression results are discussed at the end of this section.

3.4.3.1. Duckweed and Dissolved Oxygen

The interaction of duckweed cover, oxygen, and stratification noted in the time series data (Figure 3.2-3) was quantified statistically across the intensively sampled and monitored ponds. Duckweed cover and anoxic factor were strongly correlated when averaged over the monitoring season (May 29 – Nov 6; n = 17; r = 0.92, p < 0.001) and also when considered over the summer period (Jun 1 – Sep 13; n = 17; r = 0.92, p < 0.001) (Table 3.4-4, Table IV-1; Figure 3.4-4a). However, stratification, assessed both as change in RTRM and mean RTRM over these periods of interest, was only weakly (though positively) correlated with either duckweed cover or anoxic factor (r < 0.42), due to most ponds with lower RTRM values having high variability in oxygen regime. This may indicate that duckweed has a stronger influence on oxygen in the more weakly stratified ponds, and that other processes (sediment oxygen demand, sheltering) may be important controls on oxygen as well.

Table 3.4-4 *Single linear regression models for surface TP concentration, hypolimnion TP concentration, Anoxic Factor, and Duckweed Cover, derived from Summer 2019 (June 1 – Sep 13) data collection at the intensively sampled and monitored ponds (Datasets II and III). AF = anoxic factor, DW = duckweed cover (%), CC = canopy cover, RTRM = resistance to mixing, 'DO, mean' = water column average DO; 'Upper' refers to top ~50cm of water column, "C1" and "C2" refer to criteria used to define mixing (see Methods).*

TP Concentration, Epilimnion (mg/L)				
Predictor	**Slope (SE)**	**Int (SE)**	**R^2**	**p-value**
AF	0.109 (0.034)	0.09 (0.02)	0.41	0.0059
DO, Mean	-0.016 (0.005)	0.193 (0.019)	0.39	0.0043
DW	0.0011 (0.0004)	0.095 (0.022)	0.28	0.0167
T, Mean 50cm	-0.02 (0.01)	0.65 (0.238)	0.21	0.0486
Mixing, Top-Btm [C1]	-1.28 (0.39)	0.278 (0.045)	0.49	0.0077
TP Concentration, Hypolimnion (mg/L)				
Predictor	**Slope (SE)**	**Int (SE)**	**R^2**	**p-value**
DO, Mean	-0.038 (0.013)	0.34 (0.052)	0.34	0.0107
RTRM, Mean	0.002 (0.0005)	0.067 (0.049)	0.51	0.0009
DA:SA	0.005 (0.001)	0.089 (0.056)	0.44	0.0051
Mixing, TopBtm [C2]	-4.73 (1.33)	0.607 (0.107)	0.53	0.0046
Not significant: AF				
Anoxic Factor				
Predictor	**Slope (SE)**	**Int (SE)**	**R^2**	**p-value**
DW	0.011 (0.001)	-0.02 (0.065)	0.84	0.0000002
Mixing, Upper [C2]	-6.56 (2.84)	1.04 (0.27)	0.33	0.0412
Mixing, Top-Btm [C2]	-7.3 (3.05)	0.99 (0.24)	0.34	0.0355
CC, 25m	0.84 (0.39)	0.03 (0.21)	0.23	0.0490
Duckweed Cover				
Predictor	**Slope (SE)**	**Int (SE)**	**R^2**	**p-value**
AF	79.3 (8.8)	8.4 (5.1)	0.84	0.0000002
Mixing, Top-Btm [C2]	-649.7 (263.2)	93.6 (21)	0.36	0.0312
Canopy Height	4.9 (1.9)	22.7 (13)	0.30	0.0220
CC, 25m	74.4 (25.8)	4.4 (15.2)	0.32	0.0098

The influence of duckweed cover and oxygen (anoxic factor) on phosphorus is evident across the Intensively Sampled and Monitored ponds (n = 20). Several key relationships among summer TP, AF, and DO are shown in Figure 3.4-5, with TP showing significant relationships with AF (positive) and DO (negative). In addition to TP, surface TDP, and PP, as well as mean hypolimnion TDP and SRP were strongly related to anoxic fraction and with duckweed cover across sites (r = 0.49 – 0.68, $p < 0.05$; Table IV-1; see SLR results for TP in Table 3.4-4). These patterns held true when using parameters averaged over the monitored period (May 29 – Nov 6), with slightly weaker results during the warmer summer period (June 1 – Sep 15) and during fall (Sep 16 – Nov 6; Table IV-1).

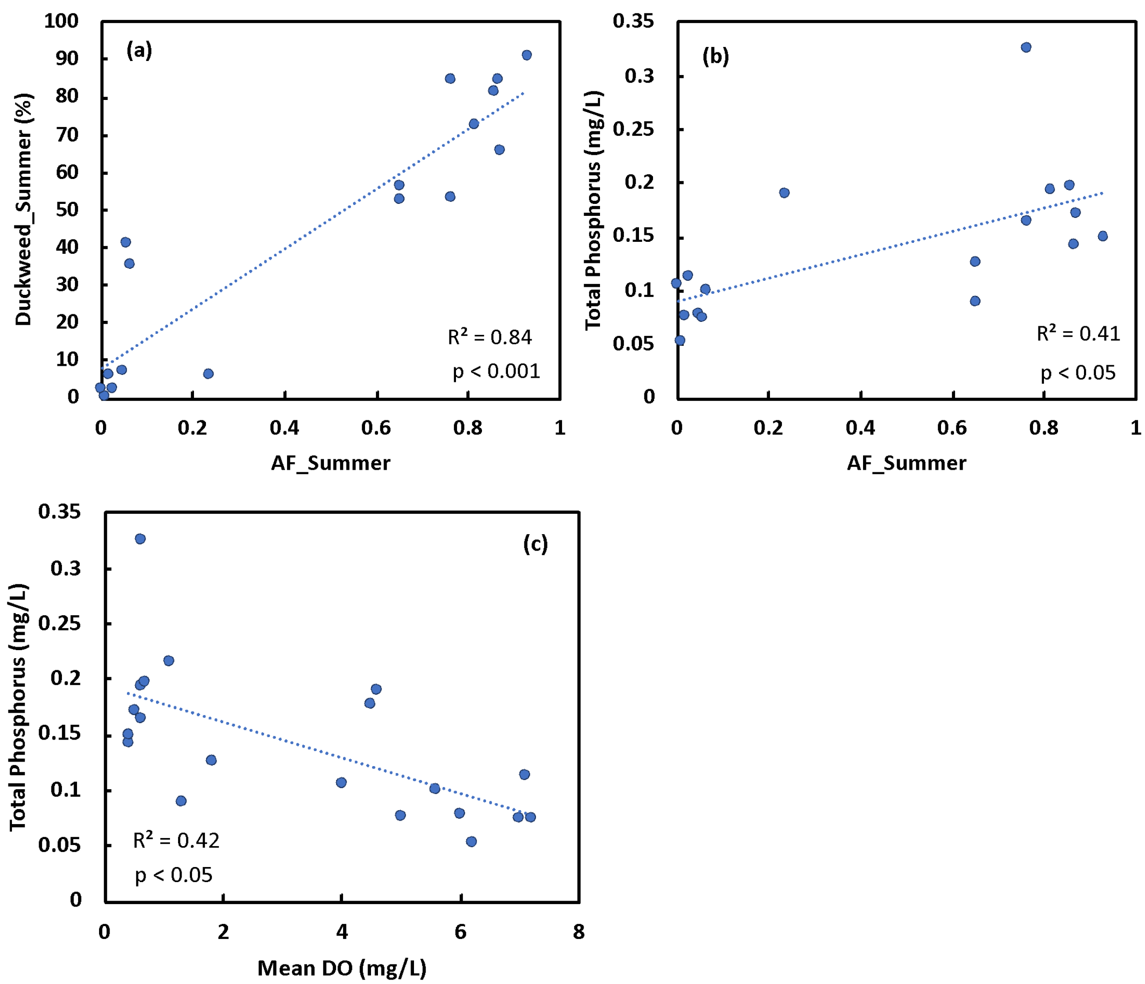


Figure 3.4-4 *Mean summer duckweed vs. anoxic factor (a), and mean summer surface TP (mg/L) vs. (b) anoxic factor and vs. (c) mean water column DO (mg/L) in the Intensive Sampling and Monitoring Sites in 2019. Each point is the summer (Jun 1 – Sep 15) mean for a site (n = 20).*

3.4.3.2. Hydrodynamics: Pond Geometry, Stratification, and Mixing

Stratification and vertical mixing were also important for explaining variation in oxygen and phosphorus across the intensively studied ponds. Higher mean RTRM, for example, was correlated with higher surface chl-a and hypolimnion TP, PP, and chl-a across the ponds over the monitored period ($n = 20$; $r = 0.52 – 0.75$, $p < 0.05$) and over summer ($n = 20$; $r = 0.58 – 0.71$, $p < 0.05$; Table IV-1). Such patterns with hypolimnetic P and chl-a could indicate settling and accumulation of phosphorus from photosynthetic origins (senesced algae or duckweed, leaf litter inputs) in a stagnant hypolimnion. By contrast, reduction in stratification (decrease in RTRM) over the monitoring period, which should be an indicator both of strength of initial stratification and of mixing effectiveness in reducing stratification strength, was positively correlated with surface and hypolimnion TP ($r > 0.64$, $p < 0.05$; result not shown). This result is perhaps contradictory with the negative relationships between mixing frequency and phosphorus concentrations (see below) but suggests that the more highly stratified ponds may be releasing more P into the water column during mixing events, or mixing events are facilitating conversion of dissolved P into particulate forms (though we note the lack of significant correlations with PP in general).

Mixing frequency, while assessed only for the ponds with continuous monitoring ($n = 13$ sites) and for a variable and limited data record across those sites (Table 2.3-1, Appendix II), showed opposite trends with respect to phosphorus. Of note, total (runoff + wind) mixing frequency, assessed using the delta criteria for top-vs-bottom (see Methods), was negatively correlated with surface TP, TDP, and SRP ($n = 13$; $r = -0.58$ to -0.76, $p < 0.05$) over the monitored and summer periods (Tables 3.4-4, IV-1). During fall (Sep 17 – Nov 6), the relationships were significant only for surface SRP and hypolimnion SRP and TDP ($r > -0.60$ to -0.73, $p < 0.05$; Table IV-1). Anoxic factor and duckweed cover were negatively correlated with top-vs-bottom and upper water column mixing frequencies ($r > -0.56$, $p < 0.05$; Table IV-1); further, pond sheltering assessed as wind reduction (ratio of airport wind speed to pond wind speed) was also positively related to TP and AF over the monitoring period ($r > 0.56$, $p < 0.05$). Together these patterns imply that more frequent vertical mixing in the ponds from both wind and runoff inputs, even if only partial with respect to the water column, led to lower dissolved P concentrations overall, higher oxygen, and lower duckweed cover. The mechanisms of P reduction could be rapid assimilation (respiration) from the presumed influx of oxygen, and potentially binding of phosphate to iron if mixing penetrates to sediment. It is also possible that more frequent mixing causes lower duckweed cover due to physical relocation into upwind areas of the pond or greater turbulence making for less ideal growth conditions. Interpreting the chain of causality in the complex interaction of wind, mixing, oxygen, and duckweed (and phosphorus) is beyond the scope of this analysis, but it is worth noting that the *wind* mixing frequencies (rather than the *total* [wind + runoff] mixing frequencies presented here) were generally poor predictors of pond water quality variables overall (results not shown).

Finally, a few relationships with pond geometry are worth mentioning. First, summer- and overall-mean surface TDP and SRP were negatively related with maximum pond depth ($r > 0.50$, $P < 0.05$; Table IV-1), which may suggest that deeper ponds were more effective at trapping P (perhaps by stronger stratification), though we note the lack of any strong correlations with any hypolimnion P forms that might suggest accumulation. Second, the ratio of drainage area to pond surface area, which is a potential indicator of pond under-sizing, was positively related to summer mean and overall mean hypolimnion TP, TDP, and SRP ($r = 0.53 – 0.76$, $p < 0.05$ after removing DA:SA outliers for ponds 41, 215, and 217; Tables 3.4-4, IV-1). This pattern may be evidence that phosphorus is accumulating in the bottom of under-sized ponds.

3.4.3.3. Canopy Effects

Tree canopy around ponds has the potential to impact phosphorus dynamics directly through leaf litter inputs and indirectly through sheltering effects that may reduce mixing by wind. Several SLR results suggest the presence of both mechanisms, the former in particular. First, greater canopy cover in a 25m buffer around the ponds (as assessed from LiDAR), was correlated with higher surface PP and duckweed cover during summer and over the monitoring period ($r > 0.48$, $p < 0.05$; Table IV-1). A similar pattern was also observed for mean canopy height around the ponds (results not shown). Higher canopy cover might facilitate more duckweed cover by reducing wind that could move duckweed into a smaller surface area of the pond, but the impact on density of duckweed cover is not known; higher PP might be caused by tree litter inputs. Canopy as assessed from the land cover data, in 10m-, 25m- and 50m- buffers, was positively related to surface TP, surface PP, duckweed cover, and hypolimnion TP over the monitoring period ($r = 0.46 – 0.60$, $p < 0.05$), with fewer and weaker relationships during summer. Instead, these relationships were strongest in fall, with most canopy metrics (regardless of buffer size) strongly correlated with surface TP, surface PP, duckweed cover, and anoxic factor ($r = 0.47 – 0.65$, $p < 0.05$). The strength of these canopy-phosphorus relationships in fall in particular suggest the trees as a source of nutrients to the ponds, rather than an impact through sheltering from wind. The general lack of relationships with wind mixing parameters from the monitoring dataset (see previous section), also suggest that wind mixing may not be as important as runoff and/or direct inputs, such as from leaf litter.

3.4.3.4. Seasonal Patterns within Sites

Most of the sites in the Monitoring and Intensive Sampling Ponds (Datasets II and III) were sampled every 2-3 weeks from late May through early November, with a few sites sampled starting in April. While the seasonality of TP and related water quality (DO, Duckweed, RTRM) were assessed qualitatively in an earlier Section (3.1), we also statistically assessed the effect of these water quality variables on TP within each site using 2019 data. While some strong patterns emerged for a few sites, in particular of TP vs. AF or duckweed vs. AF (not shown), no clear patterns emerged in the variability of the strength of these relationships across sites. While the effect of AF and duckweed on TP (and other forms of P) were quite strong when averaged over a period of interest (summer, fall, or the whole monitoring period), it is not apparent why similar effects were not present within measurements at a given site.

3.4.3.5. Developing Predictors of Pond P: Categorical Analyses

Similar to the approach for the Survey Ponds (see previous section), we summarized summer TP concentrations in the Intensive Sampling and Monitoring Ponds (Datasets II and III) by several levels (low, medium, high) of several key predictors (AF, DO, duckweed, shoreline canopy cover, pond area, pond max depth; see Appendix II). Mean and standard error of TP are shown for each category in Table 3.4-5. No assessment of significant differences was made due to low sample size, but some monotonic differences among categories are apparent, especially for AF, DO, and area.

Table 3.4-5. *Mean (standard error) of summer TP for three ranges of key pond TP predictors: duckweed, anoxic factor, canopy cover within 25m, mean water column DO, pond area, and pond max depth. See Appendix II-B for category definitions for low, medium and high for each predictor.*

	DW	AF	CC	DO	Area	Max Depth
Low	0.1 (0.013)	0.075 (0.01)	0.073 (0.016)	0.183 (0.014)	0.167 (0.018)	0.21 (0.021)
Medium	0.088 (0.012)	0.131 (0.022)	0.133 (0.017)	0.131 (0.014)	0.147 (0.016)	0.097 (0.009)
High	0.173 (0.012)	0.169 (0.014)	0.168 (0.01)	0.075 (0.009)	0.122 (0.012)	0.129 (0.014)

3.4.3.6. Determining Key Predictors of Pond P in the Intensive and Monitoring Ponds: Multiple Linear Regression (Datasets II and II)

From screening of the correlation analysis for the intensively sampled and monitored ponds (Table IV-1), a few key pond parameters have emerged for prediction of pond phosphorus (both summer concentrations and monitoring season concentrations). These parameters included anoxic factor, mean water column DO, duckweed cover, canopy+roof cover (50m buffer), canopy cover (25m buffer), mean RTRM, top-vs-bottom mixing frequency, upper mixing frequency, max depth, and ratio of drainage area to surface area. Several of these variables are likely to impact pond P through impacts to oxygen (AF, mean DO) and/or duckweed cover, both of which had the strongest relationships to pond P concentrations. These parameters were all included in the multivariate analysis along with several others identified as important to P dynamics or hydrologic retention from other analyses (see earlier in Section 3.4). Models were constructed both for P concentrations and for two strongly related variables (anoxic factor, duckweed) to develop a broader set of risk indicators for P retention.

The best multivariate models for epilimnion TP concentrations in the Intensive and Monitoring Ponds, selected by AIC, adjusted R^2, and significance of included variables, are shown in Table 3.4-6. The primary outcome of this analysis was that very few variables other than AF (or mean DO) or duckweed contribute meaningfully to explaining the variance in TP among the sites, whether considering the monitoring period or just the summer period. Some of the additional parameters that slightly improved model fitness were max depth, canopy cover (25m buffer), and ratio of drainage area:surface area. Models for anoxic factor and for duckweed were generally poor. Mixing frequency and max depth improved models for duckweed (vs. AF alone) when considering the monitoring period, while no other parameters provided improvement on the SLR models for summer AF or summer duckweed.

Table 3.4-6 *Top models by AICc, Adjusted R^2, and p-values for MLR analysis of pond TP concentrations, anoxic factor, and duckweed, in the Intensive and Monitoring Ponds Datasets (II and III). η^2 is the approximate variance in the dependent variable explained by each predictor (e.g., duckweed explains roughly 39% and depth 13% of the variability in TP for the third model below). Results are shown for (a,b,c) the monitoring period and (d) summer. Models are not shown for AF and DW for the summer period due to lack of models that explained additional variance vs. the SLR models (Table 3.4-4).*

(a) Model, Surface TP (mg/L)	AICc	ΔAICc	Adj R^2	p-value	η2
0.0012*DW + 0.083	-56.5	0.0	0.32	5.63E-03	(0.35)
0.139*CC_25m + 0.055	-55.2	1.3	0.27	1.09E-02	(0.31)
0.001*DW -0.013*Depth_max + 0.153	-53.7	2.8	0.43	4.19E-03	(0.39, 0.13)
0.12*AF + 0.081	-50.2	6.4	0.42	2.86E-03	(0.46)
0.125*AF -0.008*Depth_max + 0.122	-47.9	8.6	0.42	8.42E-03	(0.47, 0.04)
0.11*AF + 0.001*DA:SA + 0.061	-44.5	12.0	0.46	6.78E-03	(0.39, 0.07)
(b) Model, Anoxic Factor	**AICc**	**ΔAICc**	**Adj R^2**	**p-value**	**η2**
0.0097*DW + 0.00009	-12.5	0.0	0.83	2.04E-07	(0.84)
0.0079*DW + 1.09*Wind_Red -0.848	-7.3	5.1	0.84	4.15E-05	(0.73, 0.1)
(c) Model, Duckweed	**AICc**	**ΔAICc**	**Adj R^2**	**p-value**	**η2**
58.9*AF + 9.04*Depth_max - 199*Mixing_TopBtm[C1] -14.2	116.1	0.0	0.85	1.47E-04	(0.41, 0.21, 0.1)
15.6*Depth_max -582*Mixing_Upper[C2] -0.555	117.6	1.5	0.76	3.08E-04	(0.44, 0.36)
86.9*AF + 6.22	142.2	26.1	0.83	2.04E-07	(0.84)
(d) Model, Summer Surface TP (mg/L)	**AICc**	**ΔAICc**	**Adj R^2**	**p-value**	**η2**
0.0011*DW + 0.095	-52.2	0.0	0.24	1.67E-02	(0.28)
0.0011*DW -0.015*Depth_max + 0.175	-49.9	2.3	0.37	1.02E-02	(0.3, 0.16)
0.122*AF -0.015*Depth_max + 0.168	-46.3	6.0	0.46	5.61E-03	(0.45, 0.11)
0.109*AF + 0.09	-46.0	6.2	0.37	5.93E-03	(0.41)
0.107*AF + 0.001*DA:SA + 0.069	-41.7	10.6	0.43	1.03E-02	(0.41, 0.04)
0.0011*DW + 0.001*DA:SA + 0.071	-41.5	10.7	0.27	4.37E-02	(0.28, 0.04)

3.5. Water Balance in Ponds (Hydrologic Indicators)

Analyses of water level time series produced estimates of dry-weather drawdown rates, mean water level (relative to level at which discharge occurs), and a rough approximation of water balance (outflow vs. inflow). Sites for these analyses came from the Monitoring Dataset (Dataset II), 10 in 2019 and 7 in 2020. The 2019 data primarily span Late August through end of October, and are therefore likely of limited broader applicability. The 2020 data, though from fewer sites, are more complete (generally representing most of the June - October period).

3.5.1. Estimated Hydrologic Retention

Hydrologic retention was estimated from changes in water level in the ponds above and below the elevation at which discharge occurs (see Methods). We acknowledge the coarseness of this approach, which does not involve direct measurement of outflows and inflows; the approach is also not applicable to ponds with high rates of groundwater flow-through or constant water level, as the method interprets changes in water level as inflows and outflows.

The method was applied to a 2017-2018 dataset (Taguchi et al. 2020a) in which two ponds (Sites 51 and 37) with lower groundwater interaction had been monitored directly for inflow, outflow, and water level. The method was found to agree within 10% of the observed hydrologic retention over the open water period: Site 51 observed outflow/inflow ratio was 0.21 (i.e., 79% retention) compared to 0.29 by this method, and Site 37 observed outflow/inflow ratio was 0.74 (26% retention) compared to 0.70 by this method. A third pond from that 2017-2018 dataset, Site 32 (William Street Pond), had an observed outflow/inflow ratio of 0.89 (11% retention), while our method predicted an outflow ratio of 0.60. The high discrepancy is due to persistent groundwater intrusion into the drain tiles of Site 32's outlet, which would not be detected by our method.

3.5.2. 2020 Water Level Analyses

The results are given for the Shoreview Pond in Figure 3.5-1. The only sloped decrease in relative water level was 0.02 cm/d in September. This is considered to be a minor water level decrease. The rapid response of the pond outflow to high inflow (roughly one to two days for the pond level to return to the outlet elevation) indicates that the pond may not reduce peak runoff that substantially. An application of our methods indicates that roughly 98 % of the inflow contributed to the outflow during the period given in Figure 3.5-2.

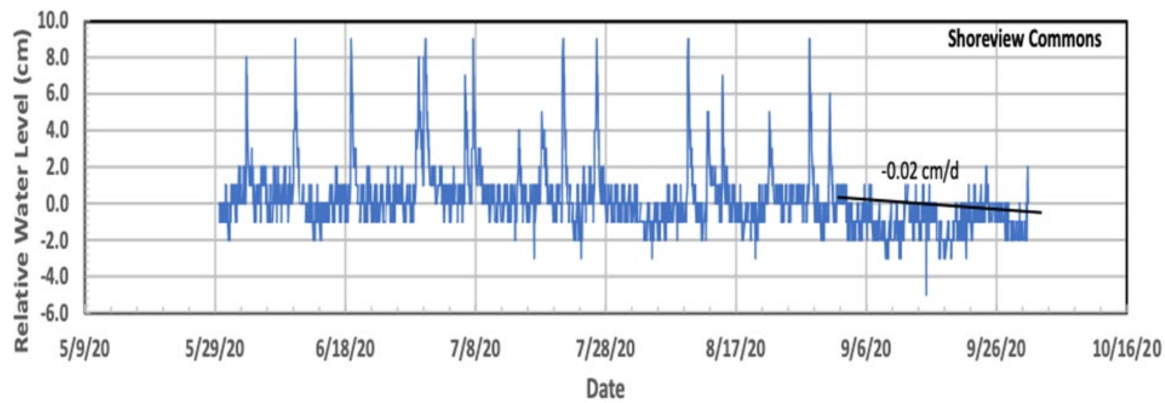


Figure 3.5-2 *Water level relative to the outlet elevation versus time for the Shoreview Pond.*

The results are given for the Minnetonka Pond 849W in Figure 3.5-3. The pond had a rate of decrease in relative water level of between 0.40 and 0.29 cm/d in June to 0.86 cm/d in August and 0.64 cm/d in September-October. The outflow response to runoff inflow required approximately 4 days for the pond level to return to the outlet elevation. Roughly 34% of the inflow that entered the pond escaped through the outflow during this period. The remainder left the pond through groundwater flow or evapotranspiration.

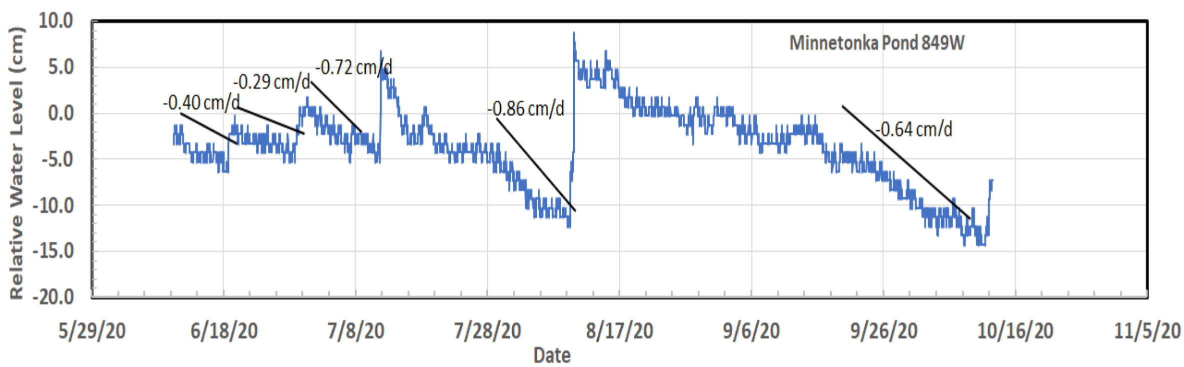


Figure 3.5-3 *Water level relative to the outlet elevation versus time for 849W.*

The results are given for the Chanhassen Pond BC-P4.10C in Figure 3.5-4. The pond has a low rate of decrease in relative water level, between 0.00 and 0.20 cm/d. The release of water from a storm required up to 4 days from its inception. Roughly 91% of the inflow exited through the outflow during this period.

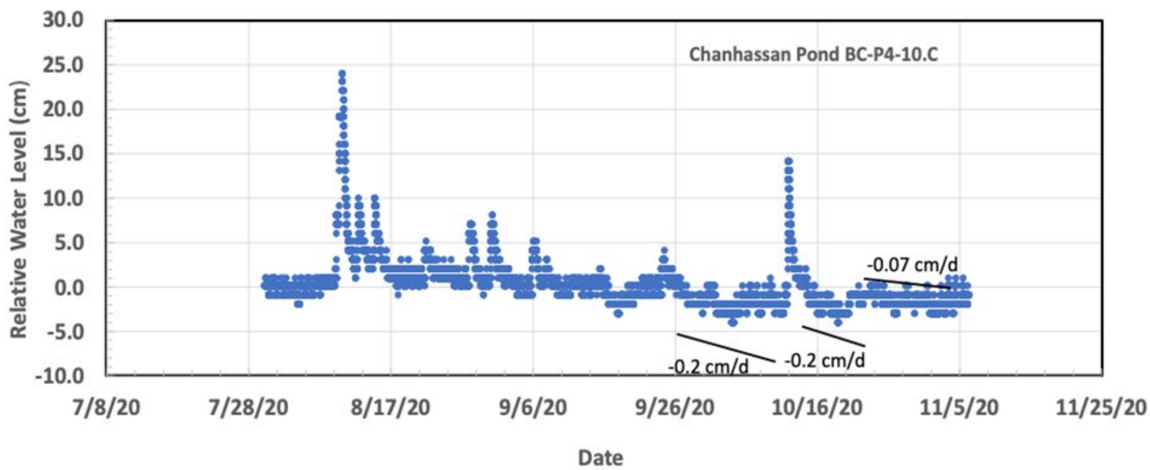


Figure 3.5-4 *Water level relative to the outlet elevation versus time for BC-P4.10C.*

As shown in Figure 3.5-5, the Pond at Cleveland and Roselawn Avenues had a peak water level decrease of 1.30 cm/d in September, a lower rate of decrease of 0.50 cm/d in September-October, and a very low rate of decrease of 0.05 cm/d in late October and early November. During the August – November period in 2020, virtually none of the runoff captured by the Cleveland-Roseville pond was released to downstream water bodies. Overall, roughly 5% of the inflow reached the outflow during this period. The remainder exited through groundwater flow or evapotranspiration.

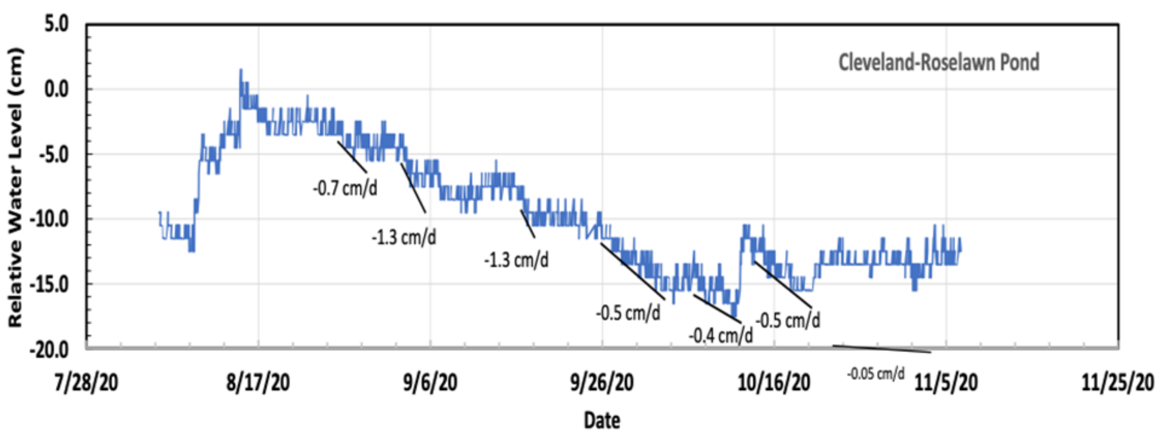


Figure 3.5-5 *Water level relative to the outlet elevation versus time for the Cleveland-Roselawn Pond.*

The Alameda Pond (Figure 3.5-6) had a fairly consistent water level decrease of between 0.70 and 1.00 cm/d, with the exception of mid to late September when it was as low as 0.30 cm/d. The quick response to rains above a relative water level of 0.20 cm indicates that this pond does not substantially reduce runoff peaks, except for times when the relative water level is below 0.20 cm water is not released. Roughly 75% of the inflow reached the outflow during the period of the measurements.

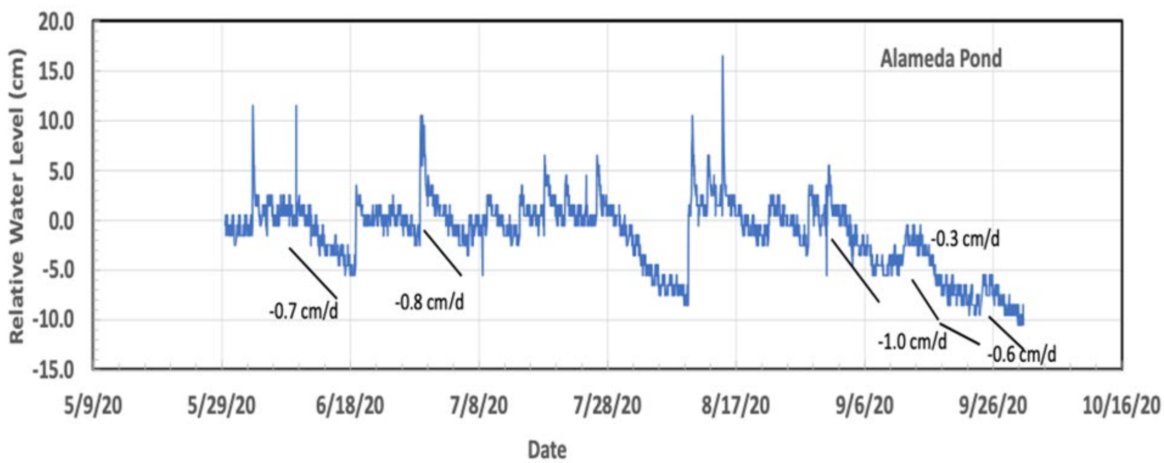


Figure 3.5-6 *Water level relative to the outlet elevation versus time for the Alameda Pond.*

The Langton Pond (Figure 3.5-7) has the largest water level drop rate of the ponds that have been surveyed. The drop was at a high of 2.50 cm/d in June-July and a minimum of 1.00 cm/d in November. There were only short periods when the relative water level was positive, indicating that there was minimal runoff release from the Langton Pond. Roughly 9% of the inflow reached the outflow during the period of the measurements.

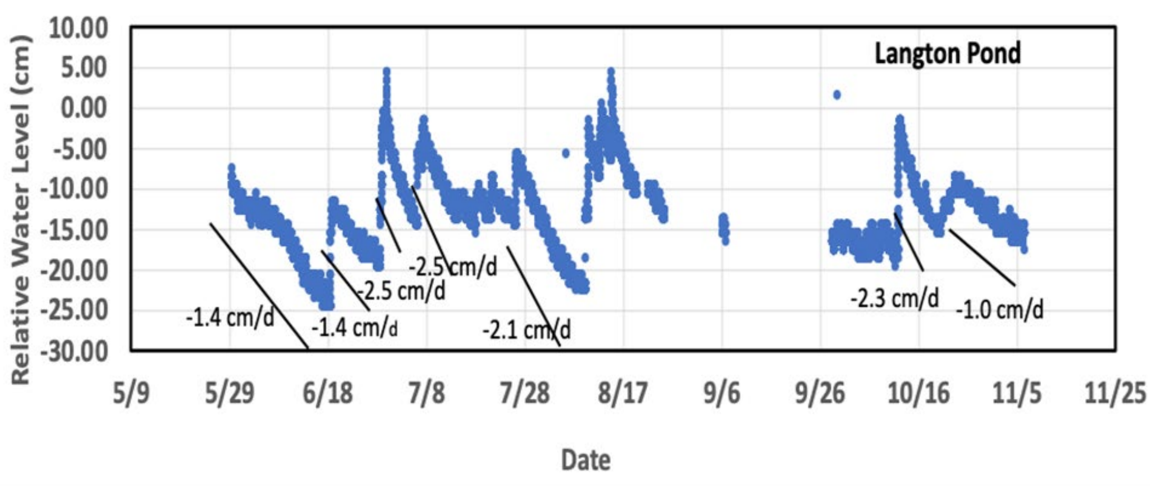


Figure 3.5-7 *Water level relative to the outlet elevation versus time for the Langton Pond.*

There was also a substantial drop in the water level of the pond at the intersection of 35E and Larpenteur (Figure 3.5-8). The peak decrease rate in mid-July was 1.60 cm/d, which decreased to 0.40 cm/d in November. The 2020 water level data indicate that water release only occurred when relative water level was 5 cm above the given outlet elevation.

Thus, roughly 9% of the inflow reached the outflow. This pond is new, has a clay liner and no emergent vegetation in 2020, so the remainder of the release would be due to evaporation. An evaporation rate of over 1 cm/d is in excess of maximum observed values in the literature, however, so the high rates of water loss at this pond remains a mystery.

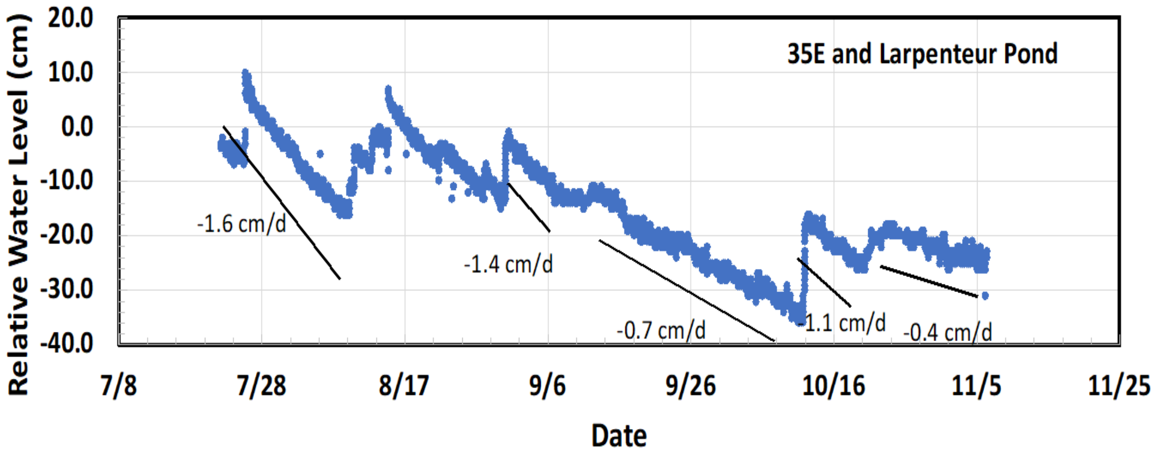


Figure 3.5-8. *Water level relative to the outlet elevation versus time for the pond northeast of Larpenteur Avenue and I-35E.*

Across sites, water level at two of the seven examined (Shoreview and BC-P4-10.C) did not show substantial water level decreases during summer. These ponds are likely to be effectively lined, minimizing exfiltration to groundwater, or have a positive groundwater inflow. The remainder of the ponds had much stronger seasonal water level decreases, with the highest decline rates observed during dry periods in July and August 2020, when evapotranspiration would be the highest. The value of the peak water level decrease rate is roughly 1.00 cm/d, with the exception of the pond at the intersection of I-35E and Larpenteur Avenue at 1.60 cm/d and the Langton pond at 2.50 cm/d. These latter two ponds are small, and most exposed to the wind of the ponds in this study, which could increase evaporation. Water loss to the groundwater table is also a possibility, even though the Langton site is clay-lined.

3.5.3 Summary of Dry-weather Drawdown Rates

Drawdown rates are summarized for the two seasons in Table 3.5-1 below. The general pattern of higher drawdown in summer and lower in fall, particularly in the 2020 data, illustrating the impact of warmer temperatures producing higher evapotranspiration rates. Some variability was present among sites and provides some clues to the ponds' water balances. In particular, Ponds 15, 79, 300, and to some extent 51, had much higher drawdown rates. These ponds were more open than the others and may have been subject to higher evaporation rates, though we note that pan evaporation rates may be on the order of 0.5 cm/day during summer, and rates at these ponds were higher than that (see MNSCO). By contrast, Ponds 37, 42, 74, and 229 had very low drawdown rates, suggesting persistent inputs of groundwater that would maintain water levels during summer. All four of these ponds are former wetlands.

***Table 3.5-1**. Dry-weather drawdown rates (cm/day) at Monitored Ponds (Dataset II) during selected periods of late 2019 and during 2020. *Note that this high drawdown rate at Pond B is likely due to direct outflow from the pond that was still occurring after a series of storms on Sep 18-19.*

		2019				2020					
Site	**Site ID**	7/29 - 8/3	9/4 - 9/7	9/14 - 9/17	9/21 - 9/28	Early June	Late June	Early July	Early Aug	Mid Sep – Mid Oct	Late Oct
Pond B	217	--	--	0.52	3.12*	--	--	--	--	--	--
Langton	79	--	1.60	0.70	1.01	1.4	1.4	2.5	2.1	--	1.0
Cleveland-Roselawn	42	--	0.31	~0	0.55	--	--	--	--	0.45	0.05
Oasis	74	--	1.30	~0	0.30	--	--	--	--	--	--
Materion	40	--	0.69	0.24	0.61	--	--	--	--	--	--
849W	230	--	--	0.25	0.35	0.40	0.29	0.72	0.86	0.64	--
Arrow	15	--	1.39	1.04	1.37	--	--	--	--	--	--
RCCP	51	0.98	--	0.24	0.99	--	--	--	--	--	--
Alameda	37	0.64	0.36	~0	0.49	0.70	--	0.80	0.80	0.63	--
William St	32	0.65	0.68	0.44	0.61	--	--	--	--	--	--
Kasota East	116	0.28	0.63	0.97	0.56	--	--	--	--	--	--
Shoreview	229	--	0.22	~0	0.28	~0	~0	~0	~0	0.02	--
BC-P4.10C	231	0.23	0.35	0.18	0.53	--	--	--	~0	0.20	0.07
35E @ Larpenteur NE	300	--	--	--	--	--	--	--	1.60	0.70	0.40

The results of this method show a wide range in outflow ratios across the monitored ponds, particularly in 2020, which was a much drier year and involved a much more complete data collection than 2019 (see next section for more details of the 2020 results). The 2019 results, while skewed towards the end of the season, which was wet and cool, show that most of the ponds were retaining very little of their inflows (generally < 20%), with the exception of two ponds (Site 51, 42). Mean relative depth was a reasonable indicator of hydrologic retention, but may be highly dependent on local watershed land and size relative to drainage area.

3.5.4 Seasonal Water Balance

Water level data were used to estimate pond water balances over the period of measurements. Results are summarized for 2019 and 2020 in Table 3.5-2 as outflow ratio (outflow/inflow) and as mean relative depth (water level relative to elevation at which discharge occurs). We found that four of the seven ponds (Minnetonka 849W, Cleveland-Roselawn, Langton and 35E-Larpenteur) retained a large proportion of inflowing water during summer. As a result, a large majority of phosphorus was retained at these sites. In contrast, two ponds (Shoreview and BC-P4.10C) had little water losses resulting in outflows that were over 90% of the inflow, leading to comparatively higher rates of phosphorus export from inflowing P and sediment release to downstream water bodies.

Our analyses show that water level data can be used to estimate the water balance of stormwater ponds during summer periods. Since the water budget of ponds plays a strong role in determining P retention or release, this simple method provides an inexpensive indicator of pond P behavior. The substantial variation in water retention across sites suggests that more research is necessary to understand controls over water losses, and indicates a possible role for pond designs to promote water retention and reduce P outflows.

Table 3.5-2. *Summary of the estimated outflow over inflow ratio and mean surface water total phosphorus concentration for the seven studied ponds during summer 2020. Sensitivity for P release was assigned according to combined influences of elevated, intermediate, and low P concentration and outflow ratios.* Outflow Ratio defined as Outflow/Inflow, estimated per method in Section 2.3.4.7.

Site	Site ID	2019		2020			Estimated Sensitivity to P Release
		Outflow Ratio	Mean Rel. Depth (cm)	Outflow Ratio*	Mean Rel. Depth (cm)	Mean Surface TP (µg/L)	
Langton	79	0.87	-1.4	0.09	-13.2	38.7	low
Cleveland-Roselawn	42	0.46	-1.38	0.05	-9.8	140.3	low
Oasis	74	0.95	0.92	--	--	--	--
Materion	40	0.93	-1.14	--	--	--	--
849W	230	0.84	0.65	0.34	-4.8	388.1	moderate
RCCP	51	0.48	-4.11	--	--	--	--
Alameda	37	0.9	-0.58	0.75	-2.8	184.5	moderate
William St	32	0.85	-0.2	--	--	--	--
Shoreview	229	0.99	1.15	0.98	-0.47	228.6	high
BC-P4.10C	231	0.79	-4.81	0.91	-0.01	152.9	high
35E @ Larpenteur NE	300	--	--	0.09	-15.2	51.8	low

3.6 Management Recommendations and Future Work

Management recommendations

While further work is necessary to develop specific recommendations, several results emerged from this project that can be used to inform management. In general, small to medium sized ponds with heavy tree canopy cover often had worse conditions with respect to water column P than larger, deeper ones. This result suggests that such ponds could be priority candidates for maintenance. However small ponds are less expensive, and also showed high rates of water retention, meaning they may maintain high P retention even when water column concentrations of P are high (Janke et al. In prep).

The table below summarizes potential concerns affecting P dynamics in urban stormwater ponds and corresponding recommendations for addressing each. Additional information to guide the implementation of these recommendations is provided by the indicated, freely-available reference material.

Table 3.6-1. *Management recommendations for ponds*

Concern	Recommendation(s)
Chloride stratification	Reduce or reroute de-icing salt [1]
Wind sheltering	Clear wind passages through large stands of tall vegetation [2]
High sediment oxygen demand	Implement street sweeping and pretreatment practices to reduce organic matter entering ponds [3] Consider sediment removal or inactivation treatments if organic matter has already accumulated [4]
Duckweed establishment	Reduce wind sheltering [2] Reduce P inputs [3]
Phosphorus export	Monitor water surface elevation and calculate water balance [5]

Sources for more information on recommendations cited above include 1] (Herb et al. 2017), [2] (Taguchi et al. 2020b), [3] (Baker et al. 2014), [4] (Erickson et al. 2018), [5] see section 3.4 *Water Balance in Ponds (Hydrologic Indicators)*; note that all topics need more work!

Recommendations for Improving Predictive Metrics

Wind Direction-weighting of Canopy Cover Around Ponds. Tree cover is often highly heterogeneous around ponds, with large gaps in tree size or spacing; further, wind does not occur from all directions equally (i.e., prevailing wind direction changes with season). We adjusted the canopy calculation to account for this heterogeneity in wind direction and canopy cover, using the 50 m buffer size (which was found to be a strong predictor of target variables among the various buffer sizes; see Results). We computed the canopy fraction for sectors in eight cardinal directions around the pond (N, NE, E, SE, etc.), and computed a "**wind-weighted" composite canopy fraction** by weighting the sector fractions by the frequency of wind observed from those directions at the MSP International Airport during the monitoring period. In this way, sheltering or gaps on the side(s) of ponds more likely to experience wind were given greater influence in the composite sheltering value. This calculation of "wind-weighted" canopy cover around ponds was done for the Monitored ponds (Datasets II) as a proof of concept, and found to provide marginally improved correlation of canopy and pond phosphorus concentrations versus the un-weighted canopy metric (see Appendix II). Future work could improve the wind-weighting to account for wind magnitude as well as different buffer sizes.

Adjustment of pond TP to account for FFP biomass. Based on 2020 FFG (i.e. duckweed) sampling, we estimate that duckweed comprises on average 50% of surface water TP in a duckweed dominated pond. This preliminary result is based on at least six measurements in five ponds (data not shown). Our sampling protocol excluded duckweed from water sample collections. **Thus our measurements of surface water TP (as well as many of those previously collected by others) underestimate the actual total TP in pond surface water, which should be defined as the sum of P in both duckweed and pond surface water.** This important point should be considered in evaluation of surface TP results presented in this report. Clearly more work is needed to understand how much P uptake by duckweed influences surface water TP particularly with respect to how much of the water column is affected by update, and thus "hidden" from only sampling the water.

Effects of aquatic macrophytes and wetlands on pond P. This project was not able to address effects of rooted aquatic vegetation in and adjacent to ponds. However, several observations were made that are relevant to indicators of pond P status that warrant future investigation. First, ponds with extensive submerged macrophytes tended to have above average water quality and lower surface water TP. Rooted macrophytes stabilize sediments and may add oxygen, potentially stabilizing sedimentary P; conversely this group is negatively impacted by excess nutrient levels, and carp. Thus the presence of rooted macrophytes may be an indicator of pond P status. Second, the presence of wetland vegetation directly adjacent to stormwater ponds may be an indicator of high pond surface levels. Wetlands are highly climate sensitive, so additional work is needed to understand our preliminary observations.

4. Dissemination and Outreach of Project Results

Project personnel have been active in outreach and dissemination with 10 local presentations, 5 national presentations, one article in Stormwater UPDATES (an online newsletter that reaches 2400 subscribers, 2/3 based in Minnesota), three refereed journal articles, and one keynote address. A list of these outreach and dissemination activities follows:

Keynote

"Ponds that Release Phosphorus." Keynote presentation by J.S. Gulliver, Pennsylvania Stormwater Symposium, Villanova, PA, October 14 – 16, 2019.

Presentations

"Maintenance of Stormwater Ponds to Increase Phosphorus Retention," Invited presentation by J.S. Gulliver, Webinar for the National Municipal Stormwater Alliance, September 15, 2020.

"Low Dissolved Oxygen in Stratified Stormwater Ponds Causes Release of Phosphorus," Presentation by V. Taguchi, 2019 World Environmental and Water Resources Congress, Pittsburgh, PA, May 19 – 23, 2019.

"Internal Loading in Stormwater Ponds as a Phosphorus Source to Downstream Waters," Invited presentation by V. Taguchi, St. Croix Watershed Research Station Research Round Table, St. Croix, MN, June 10, 2019.

"Detecting phosphorus release from stormwater ponds to guide management and design," presentation by J. Finlay, Minnesota Stormwater Research Council, July 2019.

"How to Stop Ponds from Releasing Phosphorus," Presentation by V. Taguchi, Operation and Maintenance of Stormwater Control Measures 2019, Minneapolis, MN, August 5 – 9, 2019.

"Sediment phosphorus release in stormwater ponds," Presentation by V. Taguchi, American Chemical Society Fall 2019 National Meeting & Exposition, San Diego, CA, August 25 - 29, 2019.

"How to Stop Ponds from Releasing Phosphorus," Invited presentation by V. Taguchi, Barr Engineering Surface Water Quality Practice Group Meeting, Edina, MN, September 12, 2019.

"How to Stop Ponds from Releasing Phosphorus," Invited webinar presentation by V. Taguchi, Chesapeake Stormwater Network Webcast, September 17, 2019.

“What Do We Do About Stormwater Ponds?,” Presentation by V. Taguchi, 2019 Minnesota Water Resources Conference, St. Paul, MN, October 15 – 16, 2019.

“The Challenge Of Maintaining Stormwater Treatment Practices: A Synthesis Of Recent Research And Practitioner Experience,” Presentation by A.J. Erickson, 2019 Minnesota Water Resources Conference, St. Paul, MN, October 15 – 16, 2019.

“Low Dissolved Oxygen in Stratified Stormwater Ponds Causes Release of Phosphorus,” Presentation by V. Taguchi, 2019 NALMS International Symposium, Burlington, VT, November 11–15, 2019.

"Maintaining Stormwater Ponds for Phosphorus Retention," Presentation by V. Taguchi, Minnesota Stormwater Practice and Maintenance Certification Workshop, Chaska, MN, April 30, 2020.

"The Challenge of Maintaining Stormwater Control Measures: A Synthesis of Recent Research and Practitioner Experience," Presentation by A.J. Erickson and V. Taguchi, Central States Water Environment Association's 93rd Annual Meeting in St. Paul, Minnesota, May 18-20, 2020.

"Hydrodynamics of Urban Ponds and Impacts on Phosphorus Retention," Presentation by B. Janke, Minnesota Water Resources Conference, St. Paul, MN, October 20-21, 2020.

“Understanding the Role of Dissolved Phosphorus in Managing Stormwater,” Presentation by J. Finlay, Minnesota Water Resources Conference, St. Paul, MN, October 20-21, 2020.

Publications

Erickson, A.J., V. Taguchi and J.S. Gulliver (2018). The Challenge of Maintaining Stormwater Treatment Practices, *Sustainability*, 10(10), No. 3666.

Taguchi, V.J., P.T. Weiss, J.S. Gulliver, M.R. Klein, R.M. Hozalski, L.A. Baker, J.C. Finlay, B.L. Keeler, and J.L. Nieber. (2020). It’s not easy being green, *Stormwater UPDATES*, 15 (1). http://stormwater.dl.umn.edu/updates-newsletters/updates-june-2020

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Taguchi, V.T., Olsen, T., Natarajan, P., Janke, B.D., Gulliver, J.S., Finlay, J.C. and Stefan, H.G. (2020). Internal Loading in Stormwater Ponds as a Phosphorus Source to Downstream Waters, *Limnology and Oceanography Letters*, 5(4), 322–330.

5. Summary and Conclusions

Dissolved oxygen is a well-known control over P cycling in lakes. Stormwater ponds are designed to mix often to overcome problems associated with low DO. Our studies revealed that ponds mix less frequently than expected due to effects of wind sheltering and road salt. As a consequence, a majority of ponds remain thermally and chemically stratified for much of the summer. Stratification and high sediment oxygen demand produced anoxic conditions throughout much of the water columns of many (> 40%) ponds during summer in the Twin Cities area. While the causes of unexpectedly widespread prevalence of low oxygen conditions in many ponds are not completely known, they appear to arise from combined effects of high organic matter inputs in stormwater, heavy tree cover near ponds, high aquatic primary productivity, and the presence of free floating plants.

Anoxic conditions in extensive areas of ponds promoted P release from pond sediments. Anoxic conditions were also consistently associated with elevated stormwater pond TP concentrations. As for lakes, internal loading of P occurred at high rates under anoxic conditions and was strongly related to the redox-sensitive P in the sediments. Despite high temporal and spatial variation, mean DO and anoxic fraction measurement both showed strong relationships with surface water TP. Other parameters, such as pond age, size, and depth, were weakly related to TP levels. Major findings of this project related to pond water P, emphasizing dissolved oxygen controls on TP, are integrated into a simplified depiction of stormwater pond phosphorus dynamics in Figure 5-1.

Vegetation near and in ponds played key roles in pond P (Figure 5-1). The presence of shoreline trees reduced wind-driven mixing. Trees also added organic matter to stormwater, and directly to ponds, further increasing oxygen consumption via microbial respiration. Vegetation in ponds, and especially free-floating plants (i.e., duckweed) strongly reduced oxygen availability in ponds by lowering oxygen transfer. These dynamics may create a positive feedback loop where high duckweed cover drives down DO, further increasing P availability, leading to faster duckweed growth. The role of vegetation, near and in ponds, including submerged macrophytes which were not studied in this project, clearly deserves more research attention given their central role in pond P cycles.

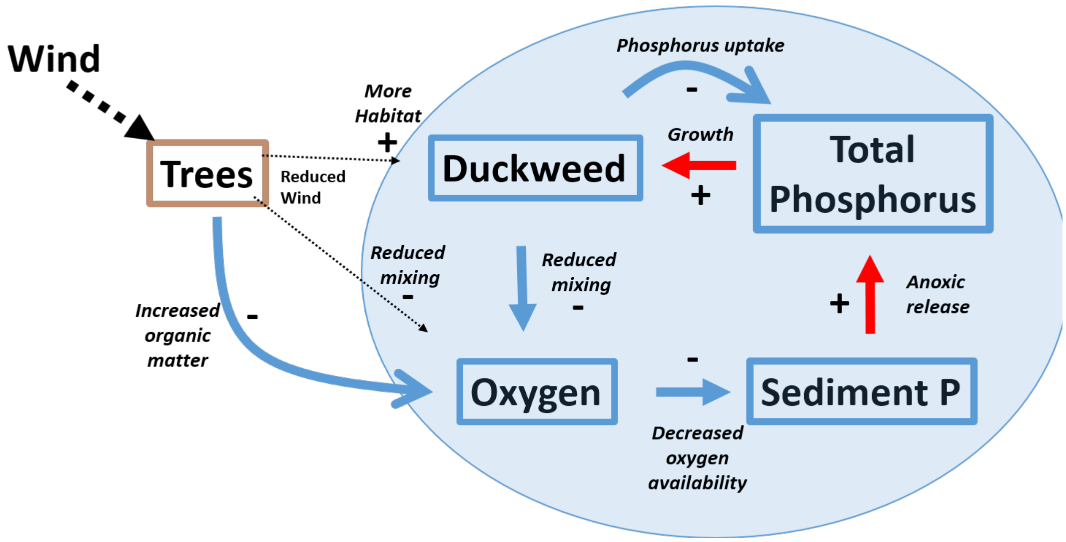

Figure 5-1 *Summary of major project findings related to dissolved oxygen controls on stormwater pond phosphorus dynamics. Negative effects (-) are indicated with* ***blue*** *lines, and positive effects (+) with* ***red*** *lines; Not all factors are represented. Dissolved oxygen concentrations are reduced by extensive duckweed coverage (i.e., FFP), high organic matter levels, and reduced mixing and aeration due to wind sheltering. Strongly anoxic conditions in sediments promotes P recycling (i.e., internal loading) from the sediments. High water column P is susceptible to export during storms and also may create a positive feedback to duckweed because growth is stimulated by high P levels. Duckweed reduces surface water P availability through uptake into biomass. Trees also played an important role through addition of organic matter to ponds and reducing wind speeds. Other factors are likely important (temperature, pond age), but these factors did not explain variation in pond water TP.*

Despite record precipitation in the year when most data were collected for the project, several variables consistently indicated the high P concentrations in stormwater ponds. Floating plant cover and shoreline tree canopy heights are both relatively easy to measure and provide strong predictions of elevated surface water P concentrations. DO concentrations, measured as water column profiles, are somewhat more labor intensive but provide the best predictors of surface water quality with respect to phosphorus. Finally, continuous measurements of water surface levels provided reliable estimates of water loss magnitudes and pathways. Pond water balances calculated by this method were highly variable and require further study to interpret fully but have potential for use as low cost indicators of stormwater pond P retention.

We conclude with a summary of our major tasks and the associated major findings from them. **Task 1** sought to *identify risk factors for elevated pond phosphorus concentrations*. We found that several parameters provided strong indicators of elevated pond water P. These indicators were all associated with factors that influence dissolved oxygen, which was mechanistically related to sedimentary P release (see Task 3, below). In **Task 2** we used *real-time data acquisition to study impacts of stratification dynamics on pond phosphorus.* These data helped understand the causes of low mixing and extensive water column stratification that contributed to low pond dissolved oxygen. Task 2 research revealed, among other results, the effect of wind sheltering on mixing and DO levels in ponds. **Task 3** *developed indicators of sedimentary P release in ponds.* Measurements of sedimentary PO_4 release rates showed high release across ponds under anoxic conditions and low (or negative) PO_4 release under oxic conditions. These results affirmed the strong role of redox in mediating internal loading in ponds. While the first three tasks were aimed at understanding internal phosphorus cycling in stormwater ponds, **Task 4** measured *variation in pond hydrology, and examined impacts of water loss on P retention*. We used continuous data for pond water level as an indicator of pond water balance. We found substantial variation in summer water balance related to climate, and large differences among sites, with some sites conservatively transferring water downstream but a majority of ponds dominated by non-outflow water losses. These high rates of water loss, while not completely understood, lead to increased P retention in stormwater ponds. In **Task 5** we used data collected in this project to *develop an approach to identify at risk ponds within watersheds to inform management activities.* High spatial and temporal variation in pond environmental conditions, complex feedback among factors affecting P cycling (including in particular the unforeseen and complex effects of duckweed and free floating plants), and threshold dynamics together prevented development of robust multivariate predictive models. However, factors affecting pond dissolved oxygen were consistently implicated in high pond water P, and our analyses identified several monitoring approaches that can be selected for use to identify ponds at risk for excess P concentration and reduced function for pond P retention. Finally *we disseminated project results using diverse outreach methods* in **Task 6.** So far the project has contributed to 16 seminar and conference presentations and three publications, with many more to come.

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