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Phosphorus Sorption Experiments: Impacts on
Water Column Phosphate Concentrations
in Utah Lake

Rebecca Layne Butters

A thesis submitted to the faculty of
Brigham Young University
in partial fulfillment of the requirements for the degree of
Master of Science

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PHOSPHORUS SORPTION EXPERIMENTS: IMPACTS ON WATER COLUMN PHOSPHATE CONCENTRATIONS IN UTAH LAKE

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Department of Civil and Construction Engineering
Master of Science

BYU ENGINEERING



Abstract

Utah Lake, located in north-central Utah, USA, is a freshwater lake with a large surface area, shallow depth, and phosphorus-rich sediment with significant water-sediment interaction. These factors cause unique phosphorus behavior relative to other lake ecosystems, governed by sediment-water interactions. We hypothesize that benthic sediments and suspended solids act, by sorption processes, to maintain water column P concentrations that are largely insensitive to P inflows. We conducted a Phosphorus Addition Experiment to characterize sorbed P fluxes in the lake. Our experimental procedure included bench-scale laboratory experiments testing Utah Lake water and sediment with dilute and augmented P levels, as well as in situ field experiments in large mesocosms with augmented P. We analyzed field data collected when spring runoff diluted in-lake P concentrations to observe dilution response in situ. We assumed that P present in a filtered water sample consisted of dissolved P, and that the P present in an unfiltered sample included sorbed, colloidal, biomass-associated, and particulate P as well as dissolved P. We found that filtered water concentrations returned to a common dissolved water column P concentration of approximately 0.01-0.03 mg/L after augmentation or depletion of initial P concentrations. Our experiments demonstrated that water column P concentrations in Utah Lake are relatively insensitive to external P loadings. Rather, strong sorption and precipitation processes maintain relatively constant dissolved water column P concentrations and compensate for changes in nutrient loads and lake volume. Our conclusions inform discourse regarding implementing phosphorus-input-limiting strategies in the restoration of the Utah Lake ecosystem.

Keywords: Utah Lake, phosphorus exchange, eutrophication, sorption, harmful algal blooms (HABs)

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1 Introduction

1.1 *Study Motivation*

In freshwater lakes, nutrient concentrations and cycling, especially those of phosphorus (P), are governed by complicated processes. Generally, these processes can be described using a mass balance model. However, Utah Lake water geochemistry, including the role of benthic sediments and suspended solids as a sorption system, can affect nutrient concentrations in the water column, which in turn affects the activity of aquatic residents, including algae, fish, and other freshwater life [1]. Utah Lake has complicated geochemistry; the shallow lake has high suspended and dissolved solids. The water is saturated with respect to calcite, with high concentrations of several nutrients, including P, that are usually limiting nutrients in other lake ecosystems [2, 3]. In Utah Lake, it is estimated that as much as 95% of P nutrient loadings do not leave the lake [4, 5]. Lakebed sediments have high P concentrations, mostly from geologic sources [6], which strongly influences water column concentrations [7, 8]. This makes Utah Lake a unique and understudied case, as shallow, sorption-dominated systems with nutrient-rich sediment are rare and not as heavily studied as more typical lake conditions.

As part of ongoing efforts to restore and preserve the health of the Utah Lake ecosystem, the focus of most discourse has been on reducing future nutrient loads to make the lake less eutrophic. There is considerable reason to have confidence in this approach, as it has been tested extensively and found to be successful in a variety of ecosystems across the world [9]. Reduction of P loads is a widely accepted strategy for the prevention and remediation of lake eutrophication [9]. P has also been established as a primary cause of Harmful Algal Blooms (HABs) in ecosystems worldwide, including Utah Lake, generating a well-founded concern for the impact of high P levels on lake health [7, 10]. Current research on Utah Lake indicates that nutrient loadings to the lake are above limitation levels, with concentrations tens of times larger than the concentration standard

for trophic state index (TSI) classification for eutrophic conditions [6]. Utah Lake P concentrations are high enough that decreasing the amount of P in the water column would likely have little impact on the current ecosystem, as there is sufficient P stored in lakebed sediments to sustain algal growth in a sorption-dominated system [4, 5, 7].

Utah Lake is unique when considering the behavior of P in the water column because the lake sediments contain P concentrations ranging from between 600 and 1,700 mg/kg, with maximum values on the order of 1,710 mg/kg [7, 8]. Wave action and bioturbation in the shallow lake result in extremely high turbidity levels, which allows the water column to regularly interact and equilibrate with sorbed P in the suspended sediments.

P strongly sorbs to solids when the dissolved P in the water and sorbed P on the solids act in equilibrium. This means that P sorbed onto sediments can be transferred between the sorbed and liquid phases in response to changing water column conditions [11]. This equilibrium process behaves as a “buffer” that minimizes changes in water column concentration related to external loads. The water column concentration instead is governed by the total amount of P in both the sediments and water column. This equilibrium can change in response to a variety of influences, including changes in temperature, the concentration of dissolved oxygen, pH, and the relative concentration of P in the sorbed and liquid phases [12]. Sorption has been observed in Utah Lake by researchers examining trends in the appearance of HABs [7, 13].

Given the presence of this buffered system in Utah Lake, it follows that any reduction in P input by point sources would likely be compensated for by the release of P reserves from the lakebed sediment into the water column, depending on the strength and details of this sorption system. To better understand sorption processes that occur between the water column and sediments in Utah Lake, we designed the Utah Lake Nutrient Cycling Studies Phosphorus Addition Experiment to characterize P fluxes between the lakebed sediment and the water column in Utah Lake. We also evaluated the change in water column concentrations in response to externally imposed changes in lake volume and P loading.

1.2 Historical P Interactions

1.2.1 Anthropogenic Contributions

There are seven wastewater treatment plants that discharge their effluent to Utah Lake, either directly or into one of its tributaries [7]. Raw sewage was discharged directly to Utah Lake as late as 1967, with discharges since then being subsequently regulated by the National Pollution Discharge Permits (NPDES) [5].

Beginning in the mid-1960's, eutrophication became a clear concern for both the government and the general population in the USA. After P was identified as a primary nutrient of concern for eutrophication and as other water quality concerns mounted in the country, the government responded with legislation including the Clean Water Act of 1972 [14]. To enforce this new legislation, the US EPA began to enact specific limits on phosphate content in lakes, streams, and other water bodies. Since then, rules have been established across the country to limit P contribution to waterways from anthropogenic sources, including phosphate fertilizers, detergents, and wastewater treatment plant (WWTP) outfalls [14]. The development, evaluation, and adjustment of these P rules has been of principal importance for WWTP design and maintenance in the decades since eutrophication became a widespread concern.

In 2016, the Utah Department of Water Quality proposed a new set of requirements, known as Rule 317-1-3, for waste discharges effective as of January 1, 2020, which limits the concentration of P in non-lagoon WWTP effluents to a maximum of 1 mg/L. The P concentration in this rule refers to total P (TP), which refers to water column P that is dissolved, sorbed to particulate matter, or included in biomass suspended in the water column. Dissolved P (DP) refers to only that portion of P present in the water column that remains after filtration on a membrane filter of 0.45 μm porosity [15]. Previously, there had been no universally applied limits on phosphate inputs by sewage treatment plants on Utah Lake, though historical average phosphate concentrations in WWTP influents were between 5-7 mg/L, depending on the treatment strategies used by each plant [16]. Estimated effluent concentrations for WWTPs similar to Timpanogos Special Service District (TSSD), a WWTP located on the northeastern end of the lake, were around 2.5-3 mg/L before Total Maximum Daily Loads (TMDLs) were established for Utah Lake [16]. TMDLs are set for water bodies based on the ability of the receiving water body to perform its designated beneficial functions [16]. In past years, specific limits had only been placed on the phosphate content for WWTP effluents discharged directly into a receiving water body for which TMDLs had been established, a group in which Utah Lake has not always been included. The new 2020 regulation marked a state-wide generalization of P limits that had not been the norm prior. These regulations included Utah Lake and set limits on all its sources of wastewater effluent for the first time. Our study began after this change, in 2021, with P dosing experiments beginning the following year. We want to note that most regulations are written for phosphate (PO_4), which is a bioavailable ion. In this study, we used an analytical technique that measured elemental P. Most DP is present as phosphate, but P sorbed onto solids or as part of

phosphate-containing complexes or minerals, which is measured as part of TP, may not be present as phosphate.

1.2.2 Environmental Contributions

Although steps have been taken to reduce anthropogenic P loads, Utah Lake continues to receive large contributions of P from the local environment. Soils surrounding the lake have high P concentrations, similar to those in the lakebed sediments, with some measured levels as high as 1710 mg/kg [7]. The Park City Formation, a geological unit composed of P-bearing shales, is found present throughout the Utah Lake watershed and serves as an abundant source of detrital phosphate to surface waters, which continually transport those phosphates to Utah Lake [8]. Utah Lake's large surface area and shallow depth also make it extremely susceptible to influence by atmospheric deposition (AD). AD of P-bearing particulates is estimated to contribute between 15 and 300 tons of P to the lake annually [17-19].

P loads from geologic sources and AD are difficult to characterize and control. However, reservoir sediments upstream from Utah Lake and the surrounding lacustrine soils whose P makeup was established before anthropogenic sources of P became significant in Utah County contain P levels that are not statistically different from P levels in current lakebed sediments, which indicates that high-P levels in the sediment are geologic in origin as opposed to being predominantly impacted by anthropogenic P inputs to the lake since European settlement [8]. This indicates that anthropogenic sources of P, while large, may not be the major influence on Utah Lake water column P concentrations. For this reason, it is important to consider environmental sources of P when studying Utah Lake and to conduct research which characterizes P behavior in the lake to inform remediation strategies, as methods that only rely on limiting external P inputs from point sources may have minimal impact.

1.2.3 Historic P Trends

We examined historic water quality data archived in the State of Utah Ambient Water Quality Monitoring System (AWQMS) to evaluate trends in lake-wide PO_4 concentration since 1990 [20]. The AWQMS data include both dissolved PO_4 (DPO_4) and total PO_4 (TPO_4) concentration records, which designations are identified as Dissolved and Total fractions in Figure 1-1, which shows the mean daily concentration of PO_4 in each fraction as measured at multiple sample locations across Utah Lake from 1990 to 2022.

It is clear from Figure 1-1 that while DPO_4 concentrations have remained relatively stable in the last three decades, there appears to be a rise in TPO_4 concentration in recent years. This rise is not entirely

unexpected, as the population surrounding Utah Lake has experienced an estimated 1.3 - 2% annual growth rate since 2010 [21]. This can increase both anthropogenic and geologic contributions of P to Utah Lake. We would expect a corresponding increase in DPO_4 with these increased loads, however, we do not see this increase in Figure 1-1. Another potential explanation is that with recent growing concern around tracking instances of HABs on Utah Lake, it is possible that sampling events in recent years have more frequently coincided with algae bloom tracking activities, which are typically associated with high P concentrations. It is expected that the TPO_4 concentration fluctuates more than the DPO_4 concentration in the water column because the TPO_4 concentration reflects P that is sorbed onto suspended sediments in the water column, for which P has been shown to have a greater affinity [13].

General trends indicate that in the last thirty years, DPO_4 levels have remained relatively low and consistent, close to an average value of 0.034 mg/L, with a standard deviation of 0.035 mg/L. By contrast, TPO_4 had an average value of 0.092 mg/L and a larger standard deviation of .194 mg/L, indicating more variability in this fraction. Figure 1-2 is a distribution plot demonstrating this variation, with some extreme values excluded from view, but not from consideration, to enable close examination of the data. The behavior of P in both the dissolved and total fractions is important to consider when applying the results of our work to remediating P concentrations in the lake.

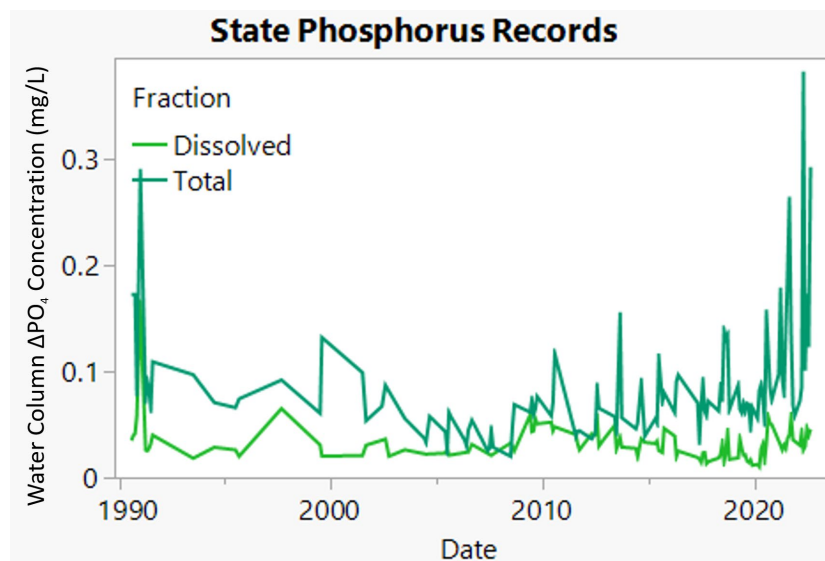


Figure 1-1 AQWMS PO_4 Measurements (1990-2022)

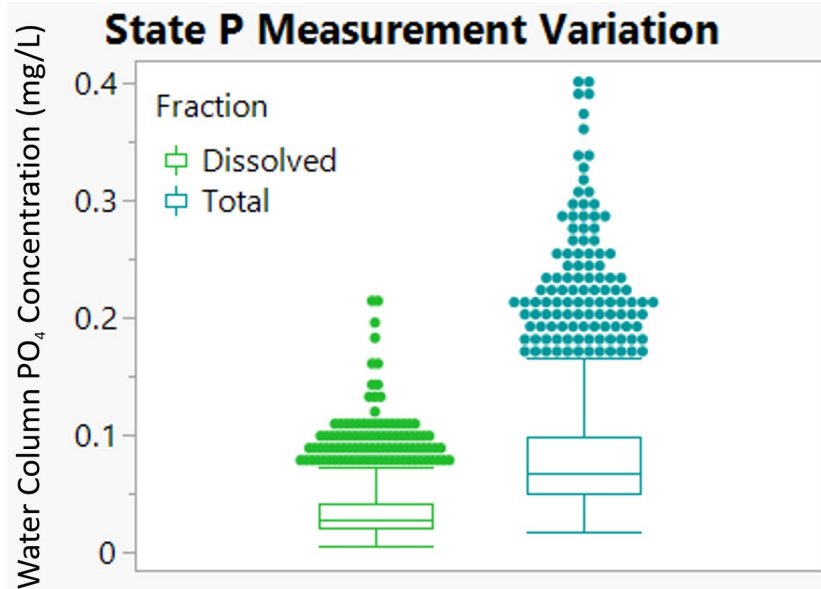


Figure 1-2 Variation in AQWMS P Measurements

1.3 Research Objectives

Our study objectives are to:

1. Quantify typical P partitioning between dissolved (water column) and sorbed (lakebed sediment) phases,
2. Investigate the rate at which P partitioning between the dissolved and sorbed phases occurs in response to changes in the environment, and
3. Improve our understanding of P loadings in Utah Lake.

2 Methods

We examined P sorption processes between lakebed sediments and the lake water column by three methods:

1. Bench-Scale experiments (BSIs), which informed method development and permitted detailed investigation of P response to water column P augmentation and dilution under laboratory conditions,
2. In Situ experiments (ISEs), which permitted direct testing of our hypothesis in Utah Lake by tracking P response to augmented concentrations in lake water contained in a large, semi-isolated mesocosm under natural conditions, and
3. Lake Dilution Tracking experiments (LDTs), which permitted indirect testing of our hypothesis on Utah Lake by tracking how lake P concentrations acted in response to dilution of the lake by larger than normal amounts of spring runoff during a record water-year.

2.1 *Definition of P Fractions*

We examined the behavior of P in terms of concentration in multiple P storage fractions in the Utah Lake system. How P is sorbed or mineralized in the solid fractions affects its interaction with water column P concentrations. The solid fraction in which P is stored affects its bioavailability, which in turn has impacts on the overall health of the lake. Examination of P behavior in lake systems by characterizing P fractions is a well-established procedure in scientific studies, but which fractions are examined are at the discretion of the researchers [22, 23]. All of our P measurements will be reported in terms of mg/L for liquid P concentrations or in terms of mg/kg for solid P concentrations for ease of comparison.

2.1.1 Water Column P Forms

We will discuss P in the water column in terms of five different forms, including Dissolved P (DP) and Total P (TP) measured as the concentration of elemental P, and Dissolved Reactive P (DPO_4), Total Reactive P (TPO_4), and Total P (TPO) measured as the concentration of phosphate. The terms phosphorus (P) and phosphate (PO_4) are often used interchangeably, because most naturally-occurring dissolved P exists as orthophosphate (PO_4) [24]. PO_4 is the nutrient form of P which is taken up by algae and bacteria, and its availability can impact the growth of these microorganisms in aquatic systems [25-27]. It is important to note that the measurements of the concentrations of P and PO_4 discussed in this paper are different from each other when expressed as mg/L because the molecular mass of PO_4 is roughly three times larger than the elemental mass of P per mole. Because of this difference and because some of the P measurement techniques we used measured elemental P, while others measured PO_4 , we acknowledge the relevant distinctions between P and PO_4 for the purposes of this study. However, we are not concerned by our use of both measurements, as the distinction is not important when evaluating sorption. The distinction between P and PO_4 measurements is clearly noted in all relevant figures and tables for this report.

DP represents the P that remains in the water column after filtration with a filter of 0.45 μm porosity. While the majority of the P in this form is likely PO_4 , we measured and will discuss elemental P concentrations. TP represents any P in the water column, including DP and any P sorbed to solids, present in minerals or organisms, or associated with other particulates suspended in the water column.

DPO_4 represents the PO_4 that remains in the water column after filtration with a filter of 0.45 μm porosity [28]. This measurement differs from DP in that it measures the phosphate ion, PO_4 , directly, rather than measuring elemental P. In a similar manner, TPO_4 represents any PO_4 in the water column, including DPO_4 and any PO_4 that is sorbed onto suspended particulate matter or otherwise associated with minerals, organisms, or any other particulates in the water column [28]. Finally, TPO refers to a measure of the total amount of phosphates present in any form in a water sample, including as part of molecules like adenosine triphosphate or metaphosphate [28].

2.1.2 Sediment P Fractions

We will discuss P in the lakebed sediment in terms of six total fractions, including Water-Soluble P (WS P), Loosely-Sorbed P (LS P), Iron- and Aluminum-Bound P (FAS P), Calcium-Bound P (CS P), Residual P (RS P), and Total P (TS P).

WS P represents the fraction of P present in lakebed sediments that is dissolved in sediment porewater. LS P represents the fraction of P that is sorbed to lakebed sediments by simple ion interactions, which can be undone by anion exchange. FAS P represents the fraction of P bound to aluminum (Al) or iron (Fe) in the lakebed sediments, which can be released under anoxic conditions. CS P represents the fraction of P that has precipitated in association with calcium (Ca), and mostly acts as a P sink rather than source. RS P represents the fraction of P which consists of mostly organic P, which is the P that remains in the sediment after fractionation steps have been performed to remove P associated with other sediment fractions. Finally, TS P represents all P that is sorbed or precipitated in the sediment across all fractions.

These fractions represent different forms of P sorbed to or part of the solid phase. Some of these fractions can move between the solid and liquid phases by sorption processes, by precipitation and dissolution, or by bio-uptake. For example, the CS P fraction typically acts as a near-permanent sink for P in Utah Lake, as P in this fraction is co-precipitated with calcite and is only released if the calcite is dissolved under low pH conditions. The P associated with Fe and Al in the lakebed sediments is also co-precipitated, but is more likely to be available for eventual release to the water column as these Fe and Al minerals are soluble under anoxic conditions [7]. The P in the WS and LS fractions are governed by equilibrium sorption processes between the solid and liquid phases, making these P fractions very mobile.

The six fractions we selected to examine were originally identified by Moore et al., and later used by Casbeer et al. on Deer Creek Lake sediments, just upstream from Utah Lake [22, 23]. We followed the methods outlined by Moore et al., with some modifications for practical application to our experimental apparatuses [29].

2.2 *Sample Analysis Methods*

2.2.1 *Elemental P Analysis*

We measured the element content of each of our samples by Inductively Coupled Plasma – Optical Emission Spectrometry (ICP-OES), in the BYU Environmental Analytical Laboratory (EAL). Students who operated the ICP-OES equipment, a Thermo Scientific™ 7400 ICP-OES Radial Analyzer, were trained by EAL personnel and developed standardized methods to ensure the preservation of sample integrity during the analysis process. ICP-OES measured the concentration of elemental P in the sample. The minimal detectable quantity of P is 5.66 µg/L or 5.66 parts-per-billion. Typical P concentrations were greater than .01 mg/L (10 µg/L), which is above the detection limit for our analysis equipment.

We performed our measurements for DP on filtered samples obtained using vacuum filtration on filters with a pore size of .45 μm before ICP analysis. For TP measurements, we digested each sample using EPA Method 3015a, which is a concentrated nitric acid digestion under elevated temperatures and pressures. After digestion and a small dilution to ensure complete sample transfer from the digestion apparatus to the analysis apparatus, these samples were run on the ICP-OES equipment. Both the filtered and digested samples were analyzed by an analysis method which measured the concentration of up to 25 elements, including P.

Because our TP measurements were conducted on diluted samples, we corrected our measured concentrations to accurately represent TP concentrations before dilution.

2.2.2 PO_4 Analysis

Our analysis procedure included measuring several water quality metrics, including PO_4 concentration, using Hach spectrometry. This analysis was conducted at the TSSD laboratory using the vendor-supplied (Hach) TNTplus methods associated with each analyte. A DR 3900 Hach Spectrometer was used to examine the concentration of PO_4 in each sample. This analysis was conducted solely on our ISE samples, and due to errors in sample handling during the 2023 sampling season, we could not consider these samples in the examination of our 2023 results. For this reason, PO_4 samples will only be considered during the discussion of our 2022 ISE.

2.3 *Bench-Scale Phosphorous Experiments (BSIs)*

2.3.1 Introduction

We designed our bench-scale experiments to mimic lake conditions as closely as possible. We conducted experiments with P concentrations deliberately higher and lower than Utah Lake conditions while limiting and monitoring the influence of potential confounding variables, including pH and temperature. We performed three iterations of our experiment, and gradually changed our experimental design as we learned more about the behavior of P in the bench-scale environment.

2.3.2 Bench-Scale Iteration One (BSI-I)

We conducted the Bench-Scale Iteration One (BSI-I) experiment in August of 2022 which served principally to evaluate the adequacy of our experiment design and to identify any needed changes for future experiments. In BSI-I, we examined lake P behavior under two water column conditions: an excess of P, and a deficit of P, relative to typical P concentrations we measured in one of our in situ mesocosms (referred to

in this paper as C07) which, during the summer of 2022, was found to be about 0.098 mg/L. We conducted this test on a short timescale of 24 hours.

To simulate both excess and deficient P loads we placed a 50 g layer of lakebed sediment in six 250 mL Erlenmeyer flasks and then filled each flask with 200 mL of lake water taken from C07. We obtained the lake water for our experiments following standard field sampling methods from the Utah Lake Nutrient Cycling Studies project. We obtained our sediment by diving to collect a sample of the top layer of undisturbed lakebed sediments near our in situ study area. Both water and sediment samples were promptly stored in cool, dark conditions until use, which was less than 24 hours later.

To simulate excess (higher-than-usual) P loads, we added varying levels of P to the lake water (Table 2-1). We used Easy Peasy Triple Super Phosphate Fertilizer, which has a concentration of about 46% Calcium Phosphate ($\text{Ca}_3(\text{PO}_4)_2$), as a P source after crushing and mixing controlled amounts with DI water to create dosage solutions of known concentration. We chose this fertilizer because it contains P in combination with Ca, which we did not expect would interact significantly with other constituents of the water column or sediments because Utah Lake water is already saturated with Ca [7, 13]. Although $\text{Ca}_3(\text{PO}_4)_2$ is generally insoluble in water, we were able to create dosage solutions with elevated P by our mixing methods. We found the resulting solutions had elevated levels of TP proportionate to the amount of fertilizer we had added. Our dosing estimations were not perfect predictors of eventual P concentration in our experimental solutions due to the variability of P concentration in lake water, but our strategy worked well enough to create effective dosage solutions that enabled us to create experimental solutions of varying P concentration. We measured actual P concentrations in our dosage solutions and experimental solutions using ICP-OES.

Table 2-1 BSI-I excess P condition sample design

Flask #	Soil Added (g)	Fertilizer Added (g)	$\text{Ca}_3(\text{PO}_4)_2$ Added (g)	Water Volume (ml)	Starting Tp Conc. (mg/L)	Time Shaken (hr)
1A	50	None	None	200	7.145	1
2A	50	0.005	0.002	200	51.88	1
3A	50	0.053	0.024	200	239.1	1
4A	50	None	None	200	7.535	24
5A	50	0.005	0.002	200	68.66	24
6A	50	0.053	0.024	200	283.0	24

To roughly simulate P-deficient conditions, we placed 50 g of lakebed sediment in six experimental flasks, then filled each flask with 200 mL of

DI water. We added varying concentrations of P to the DI water to simulate different diluted P levels (Table 2-2). To mimic mixing conditions in Utah Lake, we attached our flasks to a shaker table which ran at a moderate speed of 100-120 rpm for either 1 hour or 24 hours. For both the excess and deficient P conditions, we duplicated each treatment, including our controls. One duplicate was agitated for 1 hour, while the other duplicate was agitated for 24 hours.

Table 2-2 BSI-I deficient P condition sample design

Flask #	Soil Added (g)	Fertilizer Added (g)	Ca ₃ (PO ₄) ₂ Added (g)	Water Volume (ml)	Starting Tp Conc. (mg/L)	Time Shaken (hr)
1B	50	None	None	200	0.484	1
2B	50	0.005	0.002	200	17.06	1
3B	50	0.053	0.024	200	170.6	1
4B	50	None	None	200	0.147	24
5B	50	0.005	0.002	200	10.93	24
6B	50	0.053	0.024	200	134.8	24

For laboratory blank control samples, we filled two flasks with lake water with no lakebed sediment or added P (Table 2-3). Again, one duplicate was assigned to the 24-hour group, and the other was assigned to the 1-hour group.

Table 2-3 BSI-I control sample design

Flask #	Soil Added (g)	Fertilizer Added (g)	CA ₃ (PO ₄) ₂ Added (g)	Water Volume (ml)	Starting Tp Conc. (mg/L)	Time Shaken (hr)
7	0	0	0	200	7.646	1
8	0	0	0	200	6.824	24

To measure initial baseline concentrations, we extracted 40 mL of supernatant sample from each flask before agitation began at the start of the experiment for ICP-OES examination.

We attached aquatic tape thermometers to our flasks to monitor temperature. After agitation for either 1 or 24 hours, we removed our flasks from the shaker table and extracted samples for ICP-OES analysis. Table 2-1, Table 2-2, and Table 2-3 list the agitation time for each sample in the Time Shaken column. We selected these extraction periods to provide insight into the kinetics, or rate, at which P exchange occurred between lakebed sediments and the water column.

2.3.3 Bench-Scale Iteration Two (BSI-II)

In BSI-II, conducted in February of 2023, we used P concentrations that more realistically mimicked potential water-column P levels in Utah Lake. These concentrations better characterized the interactions that could occur in the lake. We altered our methods to better mimic lake conditions by using lake water for every sample, including samples that would represent diluted conditions, rather than using DI water.

Because it was winter, we obtained the lake water and sediment samples by walking about 100 yards out onto the ice as close to our sampling location as we could get while maintaining reasonably safe sampling conditions. We chipped through the ice to access the underlying water and sediment by using a dip-sampler and a small shovel, respectively (Figure 2-1). We quickly stored our samples in cool, dark conditions until use or froze the samples within 24 hours of sampling.



Figure 2-1 Winter sample collection from the frozen surface of Utah Lake

For BSI-II, we used four different P concentrations, two with deficient P and two with excess P, each of which were duplicated to make two groups of eight test flasks. The concentrations we used in BSI-II are listed in Table 2-4 and Table 2-5. These tables include a column that details the percent of the background concentration represented by each sample. The background TP concentration was about 0.182 mg/L in the water used for this experiment. To establish baseline concentrations, before mixing our lake water with sediment, we extracted 20 mL of supernatant sample from each flask for ICP analysis.



Figure 2-2 Experimental flasks on a shaker table

To simulate excess P-loads, we placed a 50 g layer of lakebed sediment in eight 250 mL Erlenmeyer flasks and filled the flasks with 200 mL of lake water. We added varying levels of our dosage solution to raise the P concentration in each sample (Table 2-4).

Table 2-4 BSI-II excess P condition design

Flask #	Soil Added (g)	Water Volume (ml)	Starting Tp Conc. (% Of Background)	Starting Tp Conc. (mg/L)	Time Shaken (Hr)
1A	50	200	103%	0.235	1
2A	50	200	98%	0.224	1
3A	50	200	127%	0.290	1
4A	50	200	140%	0.282	1
5A	50	200	102%	0.231	24
6A	50	200	112%	0.254	24
7A	50	200	126%	0.287	24
8A	50	200	133%	0.303	24

To simulate P deficit conditions in the water column, we placed 50 g of lakebed sediment on the bottom of eight 250 mL Erlenmeyer flasks and filled the flasks with 80 mL of lake diluted lake water. We diluted the water using a two-step process: 1) we added calcite (CaCO_3) to a large quantity of lake water to sorb and precipitate water column PO_4 ; 2) we created bench-scale samples by adding sufficient dosage solution to create experimental solutions of our desired P-deficient concentrations for the experiments. These concentrations were all below the average lake concentration. Due to lack of CaCO_3 , we reduced the amount of water in

these flasks to 80 mL rather than the larger 200 mL used in BSI-I. We measured the resulting P concentration before the experiments began. This process was not exact, but we were able to obtain several samples of varying concentrations of relatively low P for use in this experiment (Table 2-5).

Table 2-5 BSI-II deficient P condition design

Flask #	Sediment Added (g)	Water Volume (ml)	Starting Tp Conc. (% Of Background)	Actual Starting Tp Conc. (mg/L)	Time Shaken (hr)
1B	50	80	16%	0.036	1
2B	50	80	9%	0.020	1
3B	50	80	32%	0.074	1
4B	50	80	39%	0.089	1
5B	50	80	21%	0.049	24
6B	50	80	13%	0.031	24
7B	50	80	19%	0.044	24
8B	50	80	31%	0.071	24

As controls, we included two flasks filled with lake water and no lakebed sediment or added P (Table 2-6).

Table 2-6 BSI-II control condition design

Flask #	Sediment Added (g)	Water Volume (ml)	Starting Tp Conc. (mg/L)	Time Shaken (hr)
1C	0	200	0.203	1
2C	0	200	0.241	24

We placed our samples on a shaker table at 100-120 rpm and measured half of our samples after one hour of agitation. We measured the remaining samples after 24 hours of agitation. For the duration of mixing, we monitored temperature and pH using digital probes at regular three-hour intervals.

In addition to examining water column TP concentrations, we examined the change in P concentration in each of our experimental lakebed sediments for BSI-II.

We conducted ICP-OES analysis of P levels in water and sediments using EPA Methods 3015a and 3051 for aqueous and solid samples, respectively. We prepared our sediment samples for analysis by first isolating multiple fractions of sediment-bound P according to our microfractionation method developed and refined for use on Utah Lake sediments by Moore, Casbeer, and Taggart [13, 22, 29]. Based on this

microfractionation method, we measured the concentrations of P in the WS, LS, FAS, CS, RS, and TS P fractions for each sediment sample.

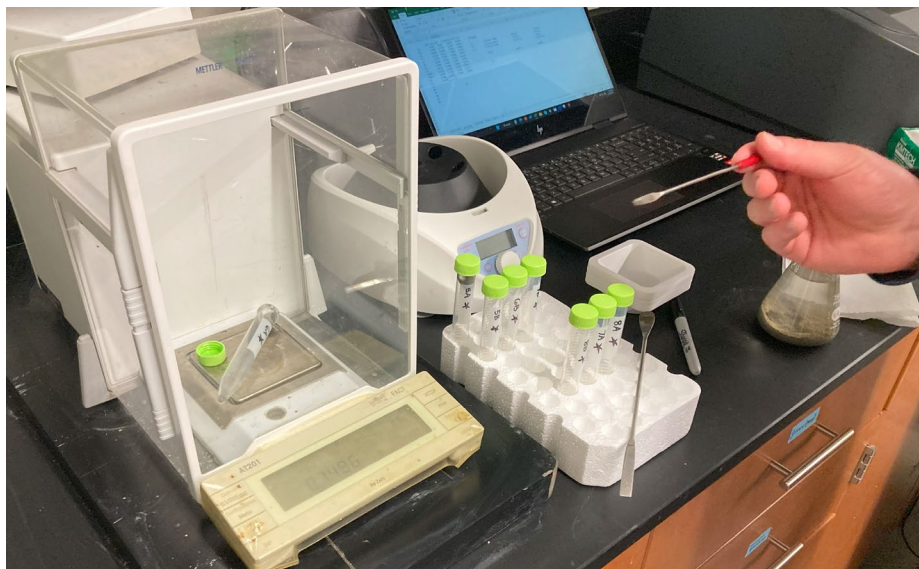


Figure 2-3 Sediment samples are weighed and prepared for fractionation

2.3.4 Bench-Scale Iteration Three (BSI-III)

In Bench-Scale Iteration Three (BSI-III), we increased the length of time we agitated the flasks to examine P concentrations over long periods of time under conditions which permitted our samples to arrive nearer to equilibrium. We expected that these conditions would be more similar to the time frames we observed in Utah Lake. We agitated lake water, without the addition of P, with lakebed sediment for one week prior to dosing to ensure that the water and sediment were in equilibrium before starting the experiments. After one week, we measured the concentration of P in the water column before beginning the addition or removal of P from the lake water. After adding or removing P from the water in the flasks, we agitated the flasks for another week at speeds of 100-120 rpm while monitoring pH and temperature at regular three-day intervals.

To simulate excess and deficient P concentrations, we placed 75 g of lakebed sediment in eight 250 mL Erlenmeyer flasks and filled the flasks with 200 mL of lake water. After the week-long acclimation period, we dosed our lake water with varying levels of P to artificially raise or lower the phosphorus concentration in each sample (Table 2-7). Due to taking some samples associated with the acclimation period, the volume of water in our experimental flasks during the second week was 145 mL. This volume is recorded in Table 2-7.

Table 2-7 BSI-III excess and deficient P condition designs

Flask #	Soil Added (g)	Water Volume (ml)	Starting Tp Conc. (% Of Background)	Starting Tp Conc. (mg/L)	Time Shaken (hr)
1A	75	145	32%	0.047	7
2A	75	145	191%	0.165	7
3A	75	145	114%	0.276	7
4A	75	145	256%	0.369	7
1B	75	145	111%	0.160	7
2B	75	145	130%	0.187	7
3B	75	145	217%	0.313	7
4B	75	145	238%	0.344	7

As controls, we included two flasks filled with lake water containing no sediment or added P, samples 1C and 2C. We also examined the change in P concentration in the different lakebed sediment P storage fractions, similar to our procedure in BSI-II.

2.4 *In Situ Experiments*

2.4.1 Experimental Apparatus

Our in situ experiments relied on specialized mesocosms called limnocorrals (Figure 2-4), also referred to as corrals, that we have built and installed annually on the northern shore of Utah Lake during summer and fall months since 2021 as part of the Utah Lake Nutrient Cycling Studies Experiments.

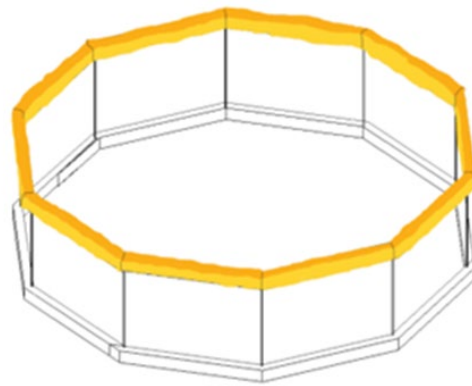


Figure 2-4 Simplified rendering of a limnocorral with yellow floats

Limnocorrals create a semi-isolated water column in the lake in which experiments can be conducted while limiting the influence of potential

confounding variables like fish activity, dilution by mixing with the whole lake, and wave action. Our limnocorrals are 10 m (33 feet) wide decagons, with ten sides, each about 3 m (10 ft) long and about 10 cm (8 inches) wide and tall. The floats along each side of our limnocorrals are connected by a decagonal frame made up of hollow metal poles. The decagon floats on the lake surface, forming a roughly circular shape. Attached to the metal frame of the limnocorrals are 3-m (10-ft) semipermeable skirts made of thick, tightly woven plastic mesh which extend from the frame down to the lakebed as a means of isolating a cylindrical volume of lake water, which varied depending on the location of the limnocorral and the overall lake volume. A flexible rubber tube lines the bottom of the skirts to prevent the walls of the limnocorrals from billowing out of shape in response to wave action, and a heavy chain is attached to the rubber tube to help seal the skirts to the lakebed. Sandbags placed on flaps on the bottom of the skirts also help seal them to the lakebed and prevent lake wildlife, like carp, from burrowing into the study area. The frames of our limnocorrals are anchored to the lakebed by multiple sets of ropes and anchors attached to the frame. In 2023, we placed hollow metal poles into the lakebed sediment at the inside of every other corner along the frame as an additional measure to help prevent the skirts from billowing. We conducted in situ experiments inside of these limnocorrals in 2022 and 2023, with the corral arrangements for these seasons shown in Figure 2-5.

In our experiment, we dosed the limnocorrals with excess P, and monitored P concentrations over time to evaluate whether sorption processes had acted to mitigate the increase in P concentration. An essential factor to analyze our data was quantifying the flushing rate of the corrals, or the rate at which water turnover occurred in the semi-isolated water column. We wanted to distinguish the decrease in P concentration due to sorption from the decrease in P concentration due to dilution or mixing with the rest of the lake. To quantify lake mixing or residence time, we used rhodamine dye, which we can measure down to the parts-per-trillion range. Rhodamine is an inert tracing dye, the dilution of which can be used to track mixing in water bodies. Because rhodamine does break down in sunlight, we ran several experiments and determined that the degradation that occurred due to sunlight was not significant over the time scales of our experiment given the limited light penetration beyond the surface waters of the lake. To validate our experimental design and methods, we dosed our experimental limnocorrals with rhodamine early in the 2022 season, before we had applied any experimental treatments to the corrals. We tracked the concentration of rhodamine in the corral over time and determined the average rate of decline in rhodamine within the corral in $\mu\text{g/L-day}$. Later, during experiments with P additions, we compared the rate of P concentration decline in our experimental

limnocorral in mg/L-day to the rate of rhodamine decline we observed in that corral. We assumed that P decline includes both dilution and sorption while rhodamine decline includes only dilution. This method allowed us to quantify the decline in P and separate it from the decline due to normal corral flushing, or sorption.

In 2023, our corrals were frequently damaged due to inclement weather and extreme water surface elevation changes. This damage allowed lake water to mix more easily with the isolated water volume. To quantify this action, we conducted our rhodamine dilution measurements coincident with our P dose experiments in order to allow us to characterize the efficacy and replicability of our experiments.



Figure 2-5 River and corral sampling sites and corral layouts in 2022 and 2023; rhodamine can be seen in the 2023 insert as pink water in the corrals

2.4.2 In Situ Experiment One (ISE-I)

For In Situ Experiment One (ISE-I), which began on August 3, 2022, we added sufficient PO_4 to our experimental corral to increase the background concentration by two orders of magnitude over lake background, which we measured at about 0.0974 mg/L. We accomplished this by adding about 1.5 kg of the same fertilizer we used in our BSIs, powdered and mixed with 4 L of DI water and 10 mL of rhodamine. We added this mixture to our experimental limnocorral, designated as C07. We simultaneously monitored P concentration in a control limnocorral designated as C10, which received no treatment (Figure 2-6).



Figure 2-6 ISE-I experimental and control limnocorrals

To add the mixture evenly to the corral, we used five-gallon garden sprayers filled with four gallons of the concentrated P solution combined with 10 ml of rhodamine dye to quantify lake mixing and dilution (Figure 2-7). After the application, we hand-mixed the corrals gently using boat oars.



Figure 2-7 Researchers dose an experimental limnocorral with P and rhodamine

After the P application, we sampled C07 and C10 simultaneously for the next three days and continued to observe corral conditions for the following two weeks. Based on our bench-scale tests, we knew that conditions would change more rapidly in the first few hours after dosing, so we conducted more frequent sampling early in the study period and gradually tapered off. Our sampling schedule is outlined in Figure 2-8.

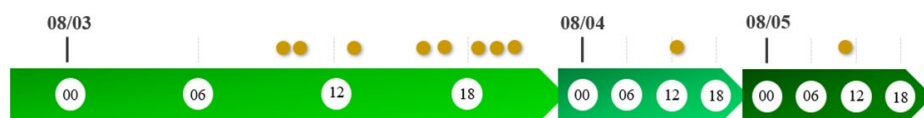


Figure 2-8 ISE-I sampling schedule in which sampling events are denoted by yellow dots, each arrow represents a day, and the 24-hour time is marked in white circles to split each day into six-hour segments

Our field sampling procedure included a collection step and a treatment step. We sampled C07 and C10 on an approximate hourly basis for the remainder of the dosage day. We then took one sample per day for the two days after dosage application. We used a dip sampler to reach into the middle of the corral for collecting water samples. We triple-rinsed the dip sampler in limnocorral lake water, then collected sample water that we stored in 0.5 L sample bottles, which were also triple-rinsed with limnocorral lake water before final sample collection. In the treatment step, which was unique to ISE-I and ISE-II, we immediately dosed the samples with 2 mL of Lugol's Iodine Solution to prevent continued growth of microorganisms that could alter P concentrations.

After sample collection, we delivered the samples to the TSSD lab to be tested for TPO_4 , DPO_4 , and TPO. We also ran filtered and unfiltered samples through the ICP-OES machine at the BYU Environmental Analytics Lab to measure elemental DP and TP concentrations over time.

2.4.3 In Situ Experiment Two (ISE-II)

In Situ Experiment Two (ISE-II), which began on August 17, 2022, was a near exact replicate of ISE-I in terms of dosage and sample collection. Our sampling schedule, outlined in Figure 2-9, was slightly different due to variances in lake conditions that permitted more frequent sampling than in ISE-I.

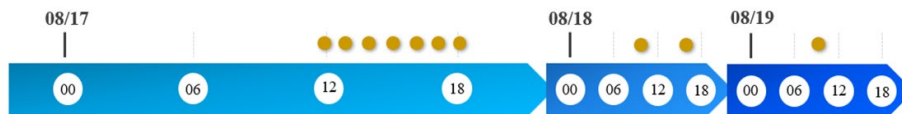


Figure 2-9 ISE-II sampling schedule

2.4.4 In Situ Experiment Three (ISE-III) and In Situ Experiment Four (ISE-IV)

We attempted to conduct two more limnecorral experiments in 2023, with minor adjustments that included only dosing the experimental corral to double the background P concentration and taking soil samples in addition to water samples during the experimental period. Unfortunately, due to severe equipment impairment from inclement weather, our results are not reliable and are not considered in the synthesis of our conclusions. We created a brief examination of our experimental results, which is available to interested parties, but is not included in this report.

In these experiments, we only doubled the background concentration of P in the experimental limnecorrals. Based on ISE-I and ISE-II, we found this concentration to be a more reasonable dose because it was within the normal upper limits of typical spikes in P concentration due to natural processes within the lake. We used the same fertilizer as our source of PO_4 and applied it to our corrals by the same methods as in previous iterations, which included an application and mixing phase. Our sample collection procedure differed slightly in that we chose to forgo the use of Lugol's Iodine Solution to stop algal growth after sample collection in favor of only placing the freshly collected samples in a cool, dark cooler before transferring them to a dark freezer within 24 hours. We chose to make this simplification to our sample collection methods to allow for more reliable examination of rhodamine concentration in the corrals over time as we found that Lugol's Iodine Solution interfered with the rhodamine measurement. It also may have had a minor interference with our elemental concentration measurements due to the iodine interacting with and creating complexes with dissolved elemental constituents. This would not affect ICP-OES measurements, but it could impact PO_4 measurements at the TSSD laboratory. We were able to reliably collect and analyze samples for rhodamine analysis using separate bottles. We transferred these samples directly to a cool, dark storage place before direct analysis on a fluorometer.

2.5 Lake Dilution Tracking (LDT)

In 2023, we performed the Lake Dilution Tracking (LDT) experiment. This water year had record precipitation in the Utah Lake watershed and resulted in high inflows to the lake [30]. Flows in the spring of 2023 rapidly filled the lake, raising the lake level to its maximum allowable elevation by late June, starting from a low elevation in May. During this time period, we monitored TP and DP concentrations in Utah Lake and the TP and DP concentrations of three of Utah Lake's major tributaries (Provo River, Spanish Fork River, and Hobbie Creek) nearly weekly, and the concentration of TP and DP in inflowing streams routinely measured lower than concentrations in the lake. This rapid filling of the lake should have diluted the P concentrations in the water column. We conducted this monitoring experiment for six months from April 2023 to October 2023 to observe how the lake's P concentrations would respond to being diluted by these large inflows.

We followed the same sampling procedures used at our in situ corrals to collect our water samples. We used 0.5 L sterile plastic bottles to collect both lake and river water samples. We used a dip sampler and waders to enable sample collection towards the center of the rivers in order to minimize sample contamination by researchers. We collected river samples from the same sampling area in each river using a dip sampler on a weekly or biweekly basis in portions of the rivers where sediment was not disturbed by the sampler and had no visible impact from plant or animal influences (Figure 2-10). At each sampling site, we collected bottle blank samples to account for any influence that air quality near the sampling site may have had on the water sample. Bottle blanks were obtained by pouring DI water back and forth between two bottles while allowing maximum air exposure in order to isolate the concentrations of elements in the air that may have affected the concentration we observed in our surface water samples. These blanks were monitored for unusually high P values that might have affected the reliability of our water samples.

All our sampling sites were as close as we could reasonably get to the mouth of the river so that we could create the most accurate record possible of the rivers' P concentrations upon entering the lake. Our sampling locations are marked on a map in Figure 2-5.

After sample collection, we took the samples to the TSSD laboratory to be sorted into filtered and unfiltered samples. We analyzed the filtered and unfiltered samples using ICP-OES to track DP and TP concentrations over the experimental period. When samples were not filtered or analyzed immediately, they were stored in a dark freezer until thawing for analysis.



Figure 2-10 A researcher collects river samples using a dip sampler

3 Results

3.1 *Bench-Scale Experiments*

3.1.1 Water Column Data

3.1.1.1 *BSI-I*

Typical water column TP concentrations in Utah Lake are on the order of 0.1 mg/L or less. BSI-I used concentrations in the 100s of mg/L range, 1,000 times larger than lake concentrations. In hindsight, these concentrations are not useful for characterizing Utah Lake behavior. However, BSI-I yielded background information and data, which provided insight into what areas we should investigate for subsequent experimental iterations. In general, the P concentrations we used in BSI-I were too high and not representative of Utah Lake.

Figure 3-1 summarizes the change in TP concentration for BSI-I. These data are difficult to analyze since some expected trends occur, but there are also some unexpected results.

Two interesting processes are apparent in the BSI-I data: First, the total amount of P in the water column did not remain constant over time, even in experimental flasks, which are a closed system. This indicated that an exchange occurred between the water column and lakebed sediments in our samples that resulted in P being stored in or released from the sediment, which was the only other sink or source of P in the system. These data are from ICP-OES measurements, which measure elemental P. The samples increased or decreased in water column TP concentrations, though not always in the expected directions. Most sample pairs exhibited changes in the same direction, indicating a potential relationship between the concentration of water column P at the start of our agitation period and whether that concentration would be associated with an increase or decrease in water column TP concentration over time. Second, the rate of change in TP concentration over one hour was much higher than the rate of change in TP concentration that occurred in the 1-hour samples' 24-hour

counterparts. This difference indicated that alteration of TP concentration in the water column in response to sudden input changes likely occurs quickly at first, before gradually slowing, which is a common phenomenon for chemical reactions. From this data, it would appear that the rate of P exchange that occurs between sediments and the water column over time may be described by 1st order kinetics, rather than 0th order or constant kinetics, which yield a flat line.

We believe that the large positive change exhibited by the highest concentration pairs occurred because undissolved fertilizer in the water increased the P concentration. This change could also have been caused by complex formations that extracted P from fractions usually not involved in sorption. We do not have any data for the attribution of this P-source, though undissolved fertilizer is the most likely way to explain these results. We assume this was an experimental error associated with our unrealistic dosage levels.

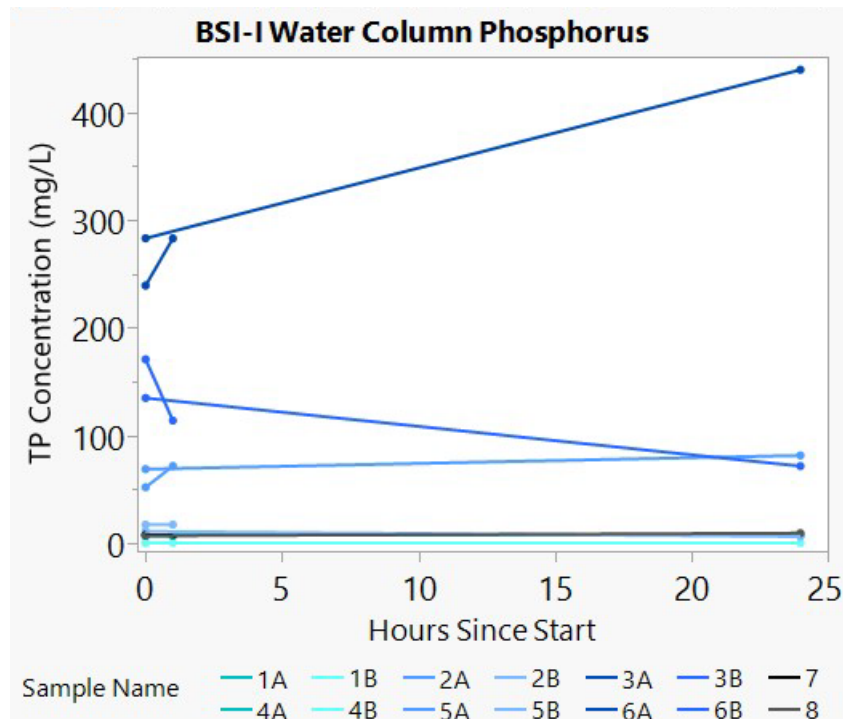


Figure 3-1 BSI-I results indicated a variable rate of P exchange

3.1.1.2 BSI-II

BSI-II investigated P exchange processes in response to more realistic alterations of background TP concentration, from 0.02 mg/L to 0.30 mg/L, or about three times below to three times above the typical Utah Lake concentration of 0.1 mg/L. BSI-II demonstrated a clear tendency for the water column TP concentration to trend towards a common equilibrium TP concentration following the addition of excess or deficient amounts of

P, simulating a dosing or dilution event (Figure 3-2). This process supports our assumption that a buffered sorption system exists in Utah Lake which maintains background P levels in the water column at a relatively stable equilibrium. This is accomplished by sediments releasing stored P into the water column or taking up excess DP in response to excess or deficient water column P concentrations compared to typical conditions.

As in BSI-I, the rate of change in our 1-hour samples was much higher than the rate of change in our 24-hour samples, indicating that the kinetics of the process follow a power function, likely 1st or 2nd order. To better characterize the kinetics, we averaged the final TP concentration values of our 1-hour samples with the initial TP concentrations of their 24-hour counterparts so that we could plot 1-hour results with 24-hour results together, creating an approximate visual representation of the kinetic rate (Figure 3-3).

Figure 3-2 and Figure 3-3 show that all our 24-hour samples behaved as expected, but some of our 1-hour samples behaved contrary to our expectations. Three of our P-deficient, or diluted, 1-hour samples decreased in P concentration during the first hour instead of increasing, as we expected. However, similar starting concentrations sampled after 24-hours behaved as expected by trending toward the typical background concentration. The unexpected behavior we saw in our 1-hour samples was likely due to some initial interactions between the freshly combined water and sediment samples, for example, sorption with the WS fraction, before the remaining sediment P processes were able to dominate. These initial reactions may have been quick, with the more dominant processes taking more time. All but one of our 24-hour samples appeared to be approaching a common point that matched typical background lake concentrations. The one exception, sample 7B, showed a significant and disproportionate increase in TP concentration relative to the behavior of the other samples. This may have been because of undissolved fertilizer, but we have no data to verify this assumption.

Based on our results in BSI-II, we determined that our samples should be permitted to interact for a longer period of time before sampling for analysis, as our 24-hour samples all behaved as expected, while the behavior of their 1-hour counterparts occasionally contradicted our expectations. We used this information to design BSI-III and the in-lake ISEs.

Figure 3-4 shows the changes in sediment P concentrations for BSI-II. These changes will be discussed in more detail in Section 4.1.2, along with a discussion of the changes in sediment P concentrations observed during BSI-III.

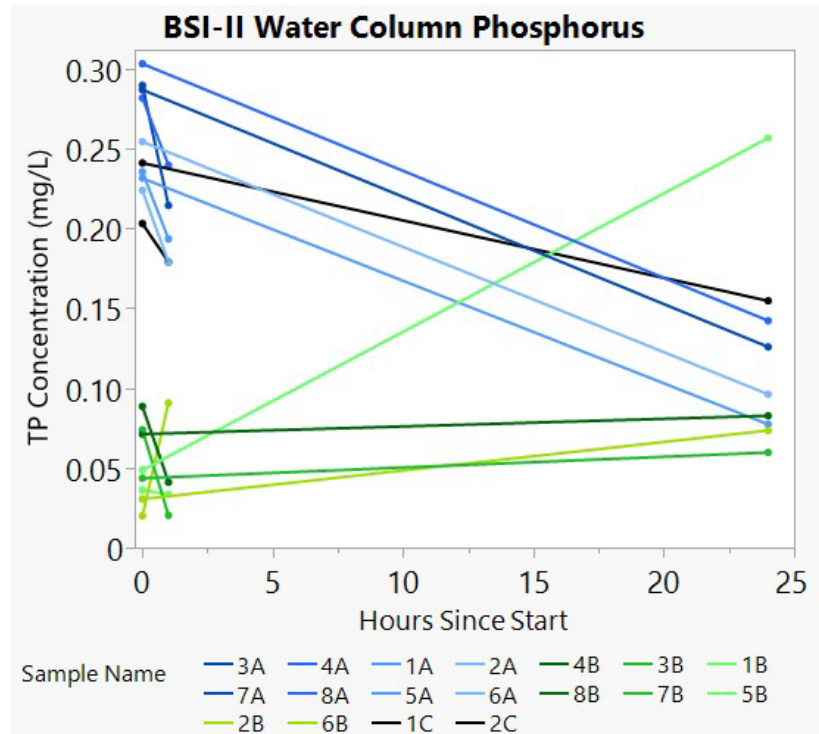


Figure 3-2 BSI-II water column TP trended towards a common concentration close to 0.1 mg/L

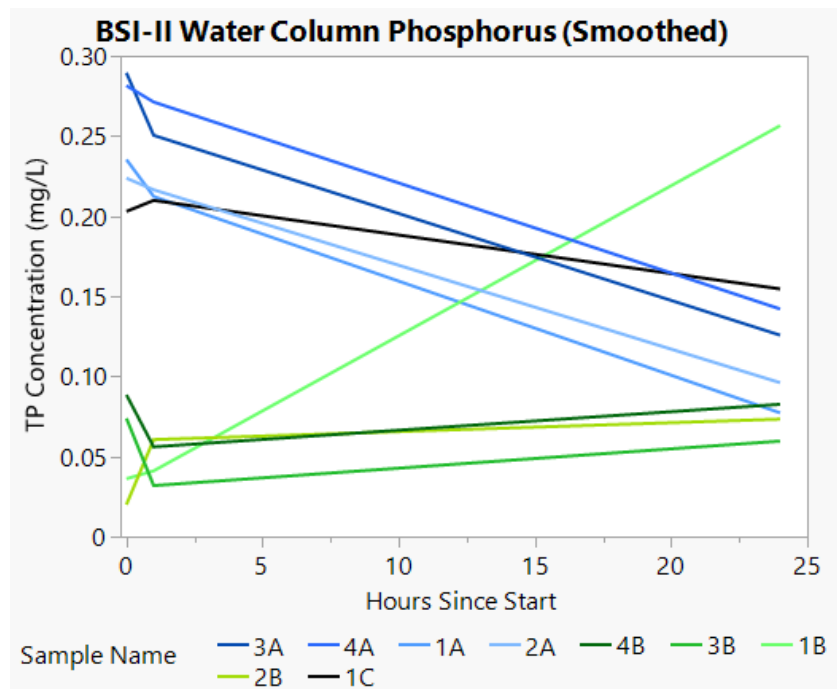


Figure 3-3 BSI-II water column TP with combined counterpart curves

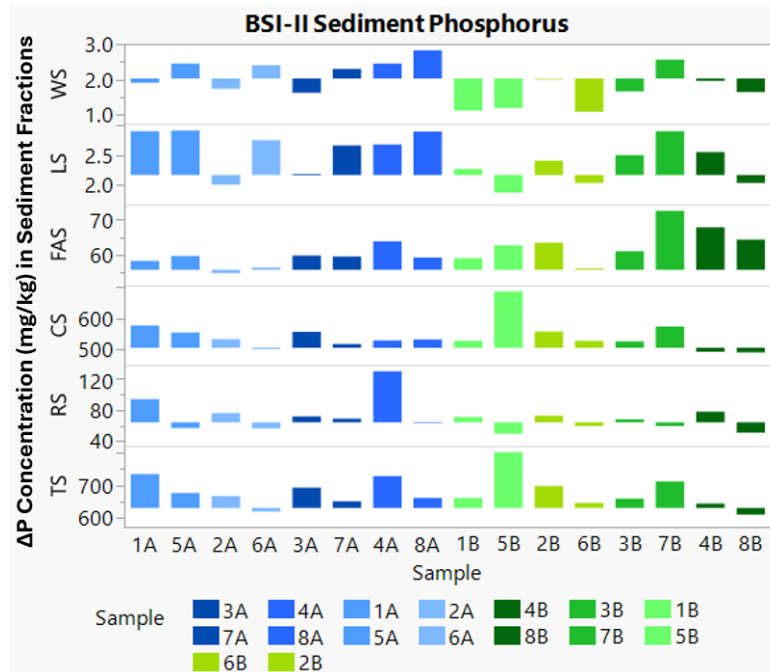


Figure 3-4 BSI-II sediment results indicate an inverse relationship between sediment and water column P change in the LS and WS fractions

3.1.1.3 BSI-III

BSI-III demonstrated results similar to those in BSI-II. In this iteration, we allowed all our samples to run for seven days in order to permit them to reach an equilibrium concentration. As shown in Figure 3-5, all of our samples trended toward a common final TP concentration in the water column close to 0.1 mg/L. Figure 3-6 displays the observed change in P in each of our measured P storage fractions in the sediment.

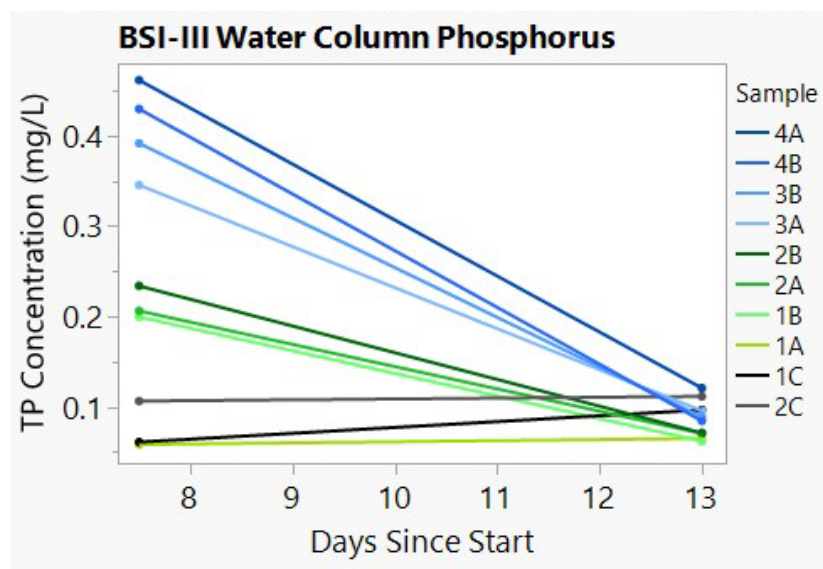


Figure 3-5 BSI-III water column TP trended towards a common concentration

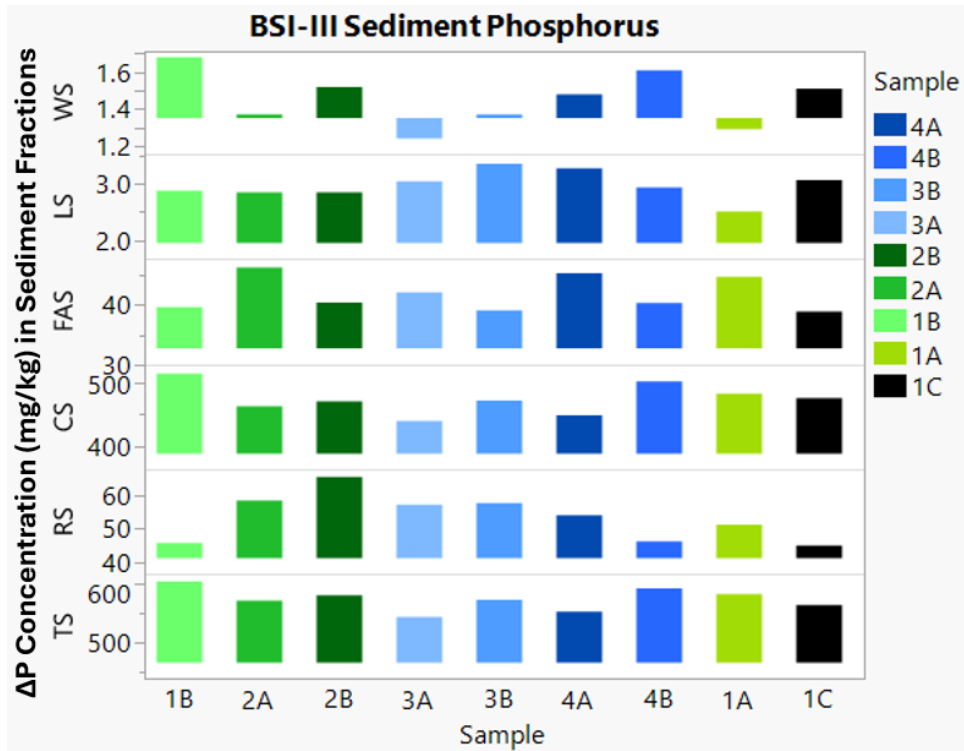


Figure 3-6 BSI_III sediment results indicate an inverse relationship between sediment and water column P change in the WS fraction

3.1.2 Sediment Concentration Changes

Figure 3-4 and Figure 3-6 show the changes in P concentration (mg/kg) in the various sediment fractions for BSI-II and BSI-III, respectively. These data provide interesting insight into the implications of our water column data, because in many cases, the changes that we observed in P storage fractions of interest occurred opposite to the change in P concentration that we observed in the water column in each experimental flask. We expected to see the largest concentration changes associated with P sorption occur in the WS, LS, and FAS fractions of P in the lakebed sediments, as these fractions are most susceptible to sorption processes. In order to undergo P exchange with the water column, the WS and LS fractions only need contact with lake water under typical lake conditions, which can occur during sediment resuspension or just water column-sediment contact. The FAS fraction can release or bind P under anoxic or aerobic conditions, respectively. Most sediment in Utah Lake is anoxic, while the water column is aerobic [8].

Both Figure 3-4 and Figure 3-6 demonstrate that the change in P concentration in the WS fraction was nearly always in the inverse direction relative to the change in TP concentration in the water column. In BSI-II, nearly 69% of all samples demonstrated an inverse direction of change in P concentration between the water column and WS P. In addition, nearly

88% of our 24-hour samples demonstrated inverse direction of change between the water column and WS P, which is to be expected as our 24-hour samples were permitted more time to reach a TP concentration close to or matching equilibrium. The change in P concentration in the LS P fraction also behaved as expected (56% of all samples and 88% of long-term samples behaved in accordance with our assumptions.). In BSI-III, 78% of all sediment samples exhibited inverse changes between water column and sediment P concentration in all fractions, with WS P having slight differences from the rest of the fractions regarding which samples behaved contrary to our expectations. These results align with our expectations that the lakebed sediment will act as a source or sink of P by increasing or decreasing water column TP following dosage or dilution events, respectively, especially due to P exchange between the water column and the WS, LS, and FAS P storage fractions.

3.2 *In Situ Experiments*

3.2.1 In Situ Experiment 1 (ISE-I)

Our results from ISE-I demonstrated a quick decline in P concentration after adding excess P, as shown by both ICP-OES data (elemental P) and results from the TSSD laboratory measuring PO_4 . For our ISE-I analysis, we only present data from our PO_4 measurements.

Although our original dosing activities caused an expected spike in water column concentration on August 3rd, 2022, Figure 3-7 shows that both the DPO_4 and TPO_4 fractions in our experimental corral (C07) experienced a secondary rise or spike in P on August 5th. However, this spike occurred simultaneously in our control corral (C10), which received no added P. Based on the control, we assume that this spike was due to external conditions not related to isolated processes in the corral. We did not see this spike in TPO in either corral C07 or C10.

The rate of change in P concentration in our experimental corral, not considering the secondary spike in P, was -2.68 mg/L-day, -4.29 mg/L-day, and -2.54 mg/L-day for TPO, DPO_4 , and TPO_4 , respectively. The rate of change in rhodamine concentrations measured in our experimental corral was routinely between -0.53 and -0.79 $\mu\text{g/L-day}$ when measured earlier in the summer. The rate of change in P concentrations is significantly higher than that of rhodamine, which measured the rate of dilution in the corrals. Because the corral flushing rate was estimated to be much slower than the rate of P dilution in our experimental corral, we determined that the decline in P that we observed in our corral was due to interactions within the corral, rather than normal corral dilution.

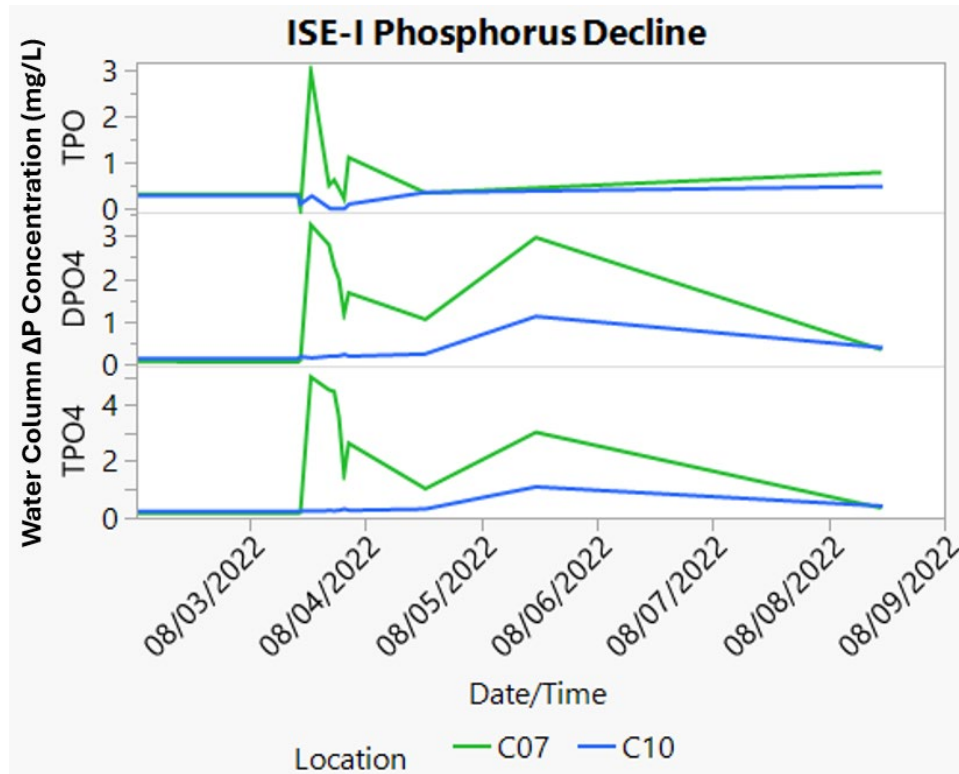


Figure 3-7 ISE-I results demonstrated evidence that P levels quickly return to normal after dosing events

3.3 LDT

For the duration of our observation period, lake levels climbed dramatically due to the unusual amount of runoff entering the lake (Figure 3-8). In late April, at the beginning of our observation period, lake volume was approximately $7.27 \times 10^8 \text{ m}^3$ (589,000 ac-ft) (approximately 68% of the legal maximum level). By July, lake volume had peaked at approximately $1.05 \times 10^9 \text{ m}^3$ (848,000 ac-ft) (~97% of maximum level). This change represented a 44% increase in volume in just three months at the beginning of our experiment. Lake levels remained close to the maximum allowable level for the rest of our data collection period, culminating in an early October volume of approximately $9.33 \times 10^8 \text{ m}^3$ (756,000 ac-ft) (~87% of the maximum). During the same period of time, inflowing rivers contained TP and DP concentrations that were low compared to contemporary TP and DP concentrations in the lake. Based on this rapid filling with relatively dilute influent water, we would expect Utah Lake P concentrations to become diluted over the course of our experiment despite continued P loadings from WWTPs and AD.

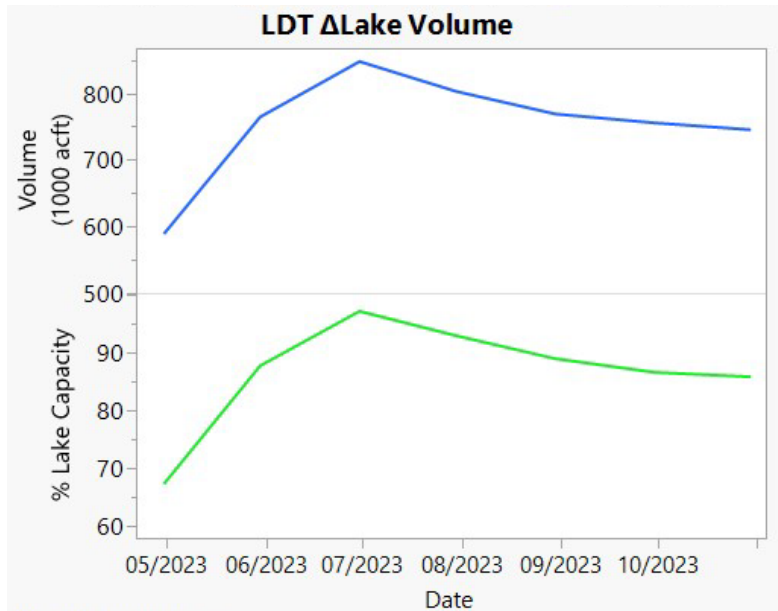


Figure 3-8 Lake volume increased greatly during the LDT period

From April to October of 2023, we monitored three inflowing tributaries (Provo River, Hobbie Creek, and Spanish Fork River), which provided the majority of inflows to the lake during our sampling period. These flows contributed large amounts of water, while the P concentrations in the inflows were significantly lower than background P concentrations in Utah Lake. Hobbie Creek flowed at an average rate of 3.60 cms (127 cfs) during our experimental period, with a maximum flow of 21.5 cms (761 cfs) in mid-May, contributing a total of more than 58.6M m³ (47,500 ac-ft) of water to Utah Lake from late April to October (~9% of total inflow to the lake from the same period). Provo River contributed a total of 273M m³ (221,000 ac-ft) of water (~40% of total inflow) at an average rate of 13.0 cms (458 cfs) with a maximum rate of 47.9 cms (1,690 cfs), which the river achieved in late June, right before the lake reached its peak volume in July. Similarly, Spanish Fork River contributed an estimated 236M m³ (191,000 ac-ft) of water (~34% of total inflow) at an average rate of 14.4 cms (508 cfs), reaching a maximum flow rate of 53.5 cms (1,890 cfs) in mid-May. The river flow data presented in this section were obtained from the CUAHSI HydroClient service [31].

In 2023, from April 25th through October 31st, Utah Lake received an estimated 684M cms (554000 ac-ft) of inflows [32]. Our monitored rivers, taken together, contributed around 83% of the volume of those inflows during our collection period. This proportion is in accordance with previous estimates that the Provo and Spanish Fork Rivers alone typically account for approximately 60% of annual Utah Lake inflows [8]. At the same time, inflow volumes dwarfed outflow volumes, which measured at

just 129M m³ (105000 ac-ft) during the same period, being smaller than the total inflows by 500%, which should have significantly diluted P concentrations in Utah Lake [32].

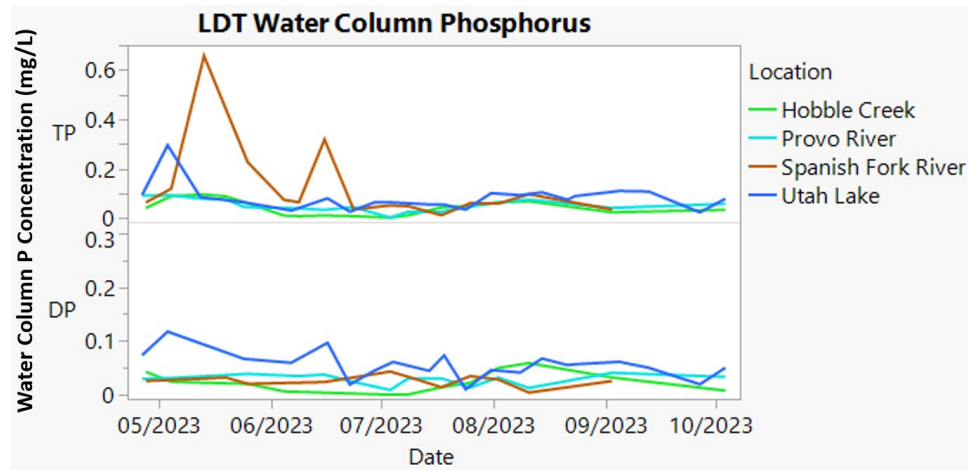


Figure 3-9 LDT results track TP and DP concentration in Utah Lake and three major inflowing tributaries

Table 3-1 Median DP and TP concentrations in Utah Lake and three major tributaries during the LDT period

Water Body	Median DP Conc. (mg/L)	Median TP Conc. (mg/L)
Utah Lake	0.0511	0.0928
Provo River	0.0304	0.0528
Spanish Fork River	0.0274	0.0636
Hobble Creek	0.0213	0.0435

Figure 3-9 and Table 3-1 demonstrate that the TP and DP concentrations measured in Provo River, Hobble Creek, and Spanish Fork River were consistently less than TP and DP concentrations in the lake for the duration of our five-month sampling period. Additionally, DP concentrations in the lake water never fell to and remained at lower concentrations than in any inflowing tributary for longer than one weekly sample at a time. While in-lake P concentrations showed a decline from May through July (during the peak lake volume change), they were relatively constant from about mid-July through the end of our sampling period. After July, lake volumes decreased slightly, but lake P concentrations remained essentially constant. Based on traditional mass-balance approaches to predicting P behavior, we would expect lake P concentrations to be reduced based on dilution. While they did decrease for a short period, after July, they rebounded to previous background

concentrations by the end of our observation period. We attribute this lag to kinetics, with sorption processes taking place over time.

3.4 *Isotherm (K_d) Fitting*

3.4.1 BSI-II Isotherm

Based on our results from BSI-II, we created several K_d curves, or isotherms, which would describe the binding affinity of P to various P storage fractions in the lakebed sediments in Utah Lake. For this dataset, we noticed a clear sorption trend when we compared the total P concentration in the three most active fractions of sediment, WS, LS, and FAS P, to the final TP concentration in the water column. We created four K_d curves based on these data. To do this, we created a dataset which attributed the observed change in water column P amount in mg to each of our fractions of interest. Three of the four fraction groups we chose to investigate in this way were: 1) WS P, 2) WS and LS P together, and 3) WS, LS, and FAS P together. We selected these fractions because they are the P fractions in lakebed sediments that can be reasonably expected to interact with the water column. The fourth fraction we investigated was TS P, which helped us observe the overall affinity of P for lakebed sediments demonstrated in our experiment. We performed our analysis by calculating the change in water column TP during our experimental period in mg, then applying that change in the inverse direction to the measured initial P amount in mg in each of our sediment fraction groups. This helped us create four datasets of theoretical final P concentrations (mg/kg) in each fraction, as if all the P change that occurred in the water column was the result of interactions with only that fraction or fraction group. We plotted our theoretical final sediment P concentrations against our observed final water column TP concentrations and identified the power equation which created the line of best fit for our dataset. This equation represents the equation of the K_d curve, with the slope term indicating the proportion of P that was observed to be associated with the sediment fractions of interest in relation to the amount of P which was observed to be associated with the water column, indicating an affinity of P for the lakebed sediments over the water column in the buffered system.

In our BSI-II investigation, we noticed an outlying value associated with one of our samples (5B), so we removed this value before creating our K_d curves. Our K_d curves based on BSI-II data represent only datapoints from our 24-hour sample set, as these samples were agitated for a long enough time to approach equilibrium.

BSI-II Isotherms

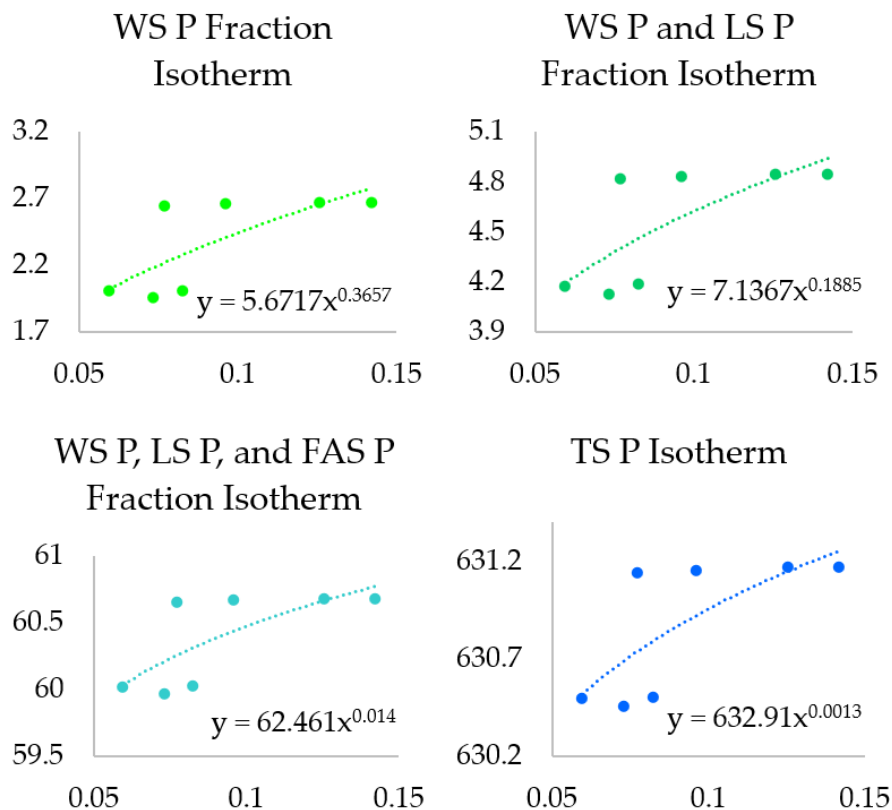


Figure 3-10 BSI-II Isotherms based on the WS, LS, FAS, and TS P fractions

From our analyses, we identified K_d values, or distribution coefficients, of 5.7, 7.1, 62.5, and 632.9 for the WS P, WS P and LS P together, WS P, LS P, and FAS P together, and TS P fraction groups, respectively (Figure 3-10).

3.4.2 BSI-III Isotherm

Using the same method as in our BSI-II investigation, we created several K_d curves based on the WS, LS, FAS, and TS P fractions in our sediment data and their relationship to the final TP concentration in the water column. Our analysis returned K_d curves that clearly identified which sediment P fractions interacted the most consistently with water column P. In this case, we identified K_d values, or distribution coefficients, of 6.1, 6.5, 38.72, and 467.8 for the WS P, WS P and LS P together, WS P, LS P, and FAS P together, and TS P fraction groups, respectively. The data presented in these K_d curves represent data from all of our BSI-III samples, which were agitated for seven days, long enough to reasonably reach equilibrium in our experimental apparatuses (Figure 3-11).

These distribution coefficients demonstrate a higher P affinity for all P storage fractions in the sediment than for the water column, but this affinity is variable depending on the storage fraction investigated and aligns with our expectations. The fraction groups which we expected to interact most with the water column (WS P, and WS and LS P together) routinely returned the lowest distribution coefficient of our groups, indicating that though P has a higher affinity for these sediment fractions, the relative affinity that P exhibits for these fractions is small compared to the affinity indicated by the high distribution coefficients returned by our WS, LS, and FAS P together and TS P group analyses.

BSI-III Isotherms

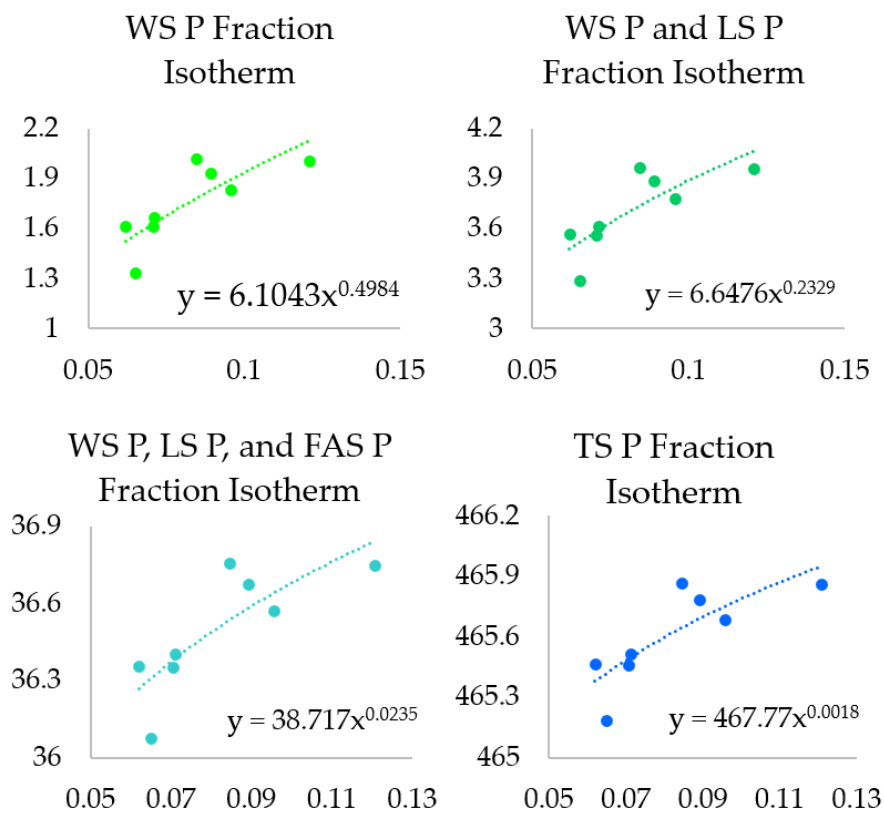


Figure 3-11 BSI_III Isotherms based on the WS, LS, FAS, and TS P fractions

4 Discussion

4.1 *Bench-Scale Experiments*

4.1.1 BSI-I

We used unrealistic P concentrations for BSI-I, but the experiment provided useful insight into what trends we should investigate and helped us design our successive experiments. BSI-I data (Figure 3-1) show how P concentrations changed in the water column over time.

Two interesting trends were visible from BSI-I data: first, it became obvious that the concentration of P in the water column after a dosing or dilution event does not remain constant over time when permitted to mix with sediment, even in a closed system like our flasks, which had P in significant excess of any levels that could occur in Utah Lake. Interestingly, some samples significantly increased water column TP concentration, an outcome which we did not expect. We attribute that change to experimental error, most likely because of undissolved fertilizer in the dosage solution. However, most samples exhibited changes, both in the short term (1-hour) and long-term (24-hour) measurements, and similar TP concentrations exhibited changes in the same direction. This indicated a relationship between the TP concentration originally in the water column and whether that concentration is associated with an increase or decrease in P as the lake water moves back towards an equilibrium concentration. Second, the slope of the change in TP concentration observed in our 1-hour samples was much steeper than the slope of the change in TP concentration that occurred in their 24-hour counterparts. This difference indicated that the alteration of water column P concentration in response to a sudden change likely occurs quickly at first, before gradually slowing, indicating that the kinetics of this interaction probably follow a 1st order process.

4.1.2 BSI-II

We used more realistic TP concentrations for BSI-II. The results support our hypothesis of a buffered system existing and dominating water column P concentrations in the lake. Samples that were dosed to higher-than-background P concentrations, denoted in Figure 3-2 in shades of blue, all demonstrated a decline in water column P concentrations. For the most part, our diluted samples behaved in the opposite manner, as demonstrated by the rising slopes of all of our long-term green sample lines and half of our short-term green sample lines. Additionally, our samples once again indicated that the rate of change in P concentration in the samples is much higher in the first hour than in the overall process.

In addition, our isotherms based on all but one outlier 24-hour sample demonstrated reliable, realistic distribution coefficients which describe P affinity in the water column and in the sediments. The fractions of lowest P affinity for sediments (WS P and LS P) both had a distribution coefficient near six, which aligns with expectations that P stored in these fractions is more accessible to the water column than P stored in other fractions by more complex processes, such as P in the FAS P fraction, which had a much higher distribution coefficient (Figure 3-10).

Though these results are encouraging, we suspected that the less-than-ideal alignment of our results with a typical isotherm curve was indicative of a failure in our samples to reach equilibrium during our experiment time frame. For this reason, we chose to continue our K_d curve investigation using our BSI-III dataset, which was conducted over seven days rather than only 24 hours.

4.1.3 BSI-III

The comparison in Figure 3-11 clearly demonstrates the usefulness of isotherms in determining the affinity of P for sediments to inform conclusions about P flux in Utah Lake. These curves align well with our results from BSI-II, but also fall more in line with the shape of a typical isotherm, which lends more confidence to the possibility of their use as a descriptor of P sorption in Utah Lake and supports our assumption that our samples were able to achieve a state closer to equilibrium in BSI-III. Additionally, these samples aligned with our ideal K_d curves without a need to exclude any outlier values, indicating that all samples were able to achieve an equilibrium or near-equilibrium state, as opposed to the results from BSI-II. Based on these data, we can conclude that P in Utah Lake has an affinity for the LS and WS fractions of lakebed sediment which is around six times greater than P's affinity for the water column. The fact that P has a higher affinity for these fractions supports what we already know about the nature of Utah Lake as a buffered system. Additionally, our

distribution coefficients for the groups including the FAS and TS P fractions demonstrate that our assumptions about the relatively high availability of P to the water column from specifically the WS and LS fractions are justified. These distribution coefficients also offer an explanation for the observed resiliency of historic P concentrations, especially DP concentrations, in the Utah Lake water column, as these strong sediment-P interactions cause large changes in lake P content to affect sediment concentrations more significantly than water column concentrations.

4.2 ISE-I

ISE-I successfully indicated that the water-column concentration of TP and DP will decrease quickly in response to substantially increased PO₄ concentrations in the water column. The change we observed in our in situ mesocosm occurred more quickly than the rate of water flow through the mesocosm itself, indicating that mixing with the lower lake water P concentrations outside of the mesocosm was not the only cause of this decline (Table 4-1).

Table 4-1 Rate of change in PO₄ and rhodamine in C07

Δ Rhodamine Concentration (μg/L/day)	Δ TPO Concentration (mg/L/day)	Δ DPO₄ Concentration (mg/L/day)	Δ TPO₄ Concentration (mg/L/day)
-0.527	-2.68	-4.29	-2.54

Due to inclement weather which led to severe equipment failure, we were unable to reliably replicate our ISE experiments. However, our results in ISE-I align with previous conclusions about P concentration in Utah Lake. We also witnessed a much quicker change in DPO₄ than in TPO₄, which indicates that lake PO₄ concentration will more quickly adjust in the dissolved fraction than as total PO₄. This supports our hypothesis that lake P concentrations are maintained by sediment-water interactions that will manifest in relatively stable DP concentrations and aligns well with historical Utah Lake P concentration data, as described in Section 1.2.3.

4.3 LDT Experiment

Our LDT Experiment demonstrated that water column P concentrations in the lake are maintained in response to dilution in much of the same way that they are maintained in response to dosing. Despite consistent inputs of dilute water to Utah Lake by the inflows we observed, Utah Lake P, especially DP, concentrations remained relatively stable throughout the summer. As supported by our examination of AQWMS data, TP

measurements during our LDT experiment exhibited more variation than DP measurements, but both remained relatively consistent throughout the summer. TP measurements in the lake had an average of .09 mg/L with a standard deviation of .066 mg/L, while DP measurements averaged .059 mg/L with a lower standard deviation of .028 mg/L (Figure 4-1). Additionally, initial and final TP and DP measurements in the lake were very similar to each other, demonstrating the stability of water column P measurements over long time spans in response to whole-lake dilution, as shown in Figure 3-9 and Table 4-2.

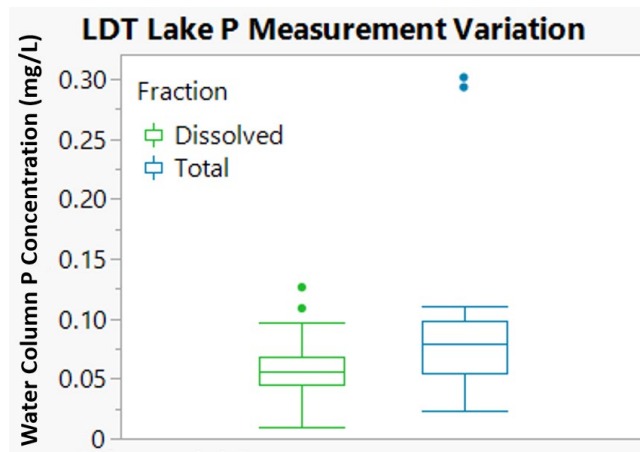


Figure 4-1 Variation in LDT lake P measurements

Table 4-2 LDT P concentration statistics

Parameter	DP Concentration (mg/L)	TP Concentration (mg/L)
Minimum	.01	.023
Maximum	.126	.298
Initial	.052	.091
Final	.05	.078

5 Conclusion

Our research provides considerable evidence that P concentrations in the Utah Lake water column, especially in the dissolved fraction, are maintained by interactions between the water column and lakebed sediments. P concentration in the water column is therefore relatively insensitive to external P inputs. We demonstrated through both bench-scale and in situ testing that this relationship is at play whether water column P concentrations have been elevated or reduced.

Our study has laid considerable groundwork for future studies in characterizing the rate of P exchange between sediments and the water column in Utah Lake. We demonstrated in our BSI and ISE experiments that P decline will occur more quickly right after a dosing or dilution event before the rate of change slowly tapers off as the P concentration in the water column approaches typical background levels. We also established work characterizing the affinity of P for lakebed sediments and demonstrated that P will interact with some sediment P fractions more than others in response to a dosage or dilution event in the water column. Based on our results from BSI-II and BSI-III, P has a higher affinity for the lakebed sediments than for the water column. P will especially interact with the WS, LS, and FAS P fractions to reach an equilibrium P concentration in the water column in response to a dosage or dilution event.

Our work informs current investigations into Utah Lake remediation efforts in response to the lake's eutrophic state. The cycling behavior of P in the lake system is an essential factor for consideration when making attempts to reduce water column P because the tendency of P to escape from lakebed sediment to replenish P in the water column following a dilution event will counter attempts to limit or reverse lake eutrophication by reducing P inputs. By investigating P cycling trends, our research enables concerned parties to approach attempts to resolve lake eutrophication in a strategic and accurately foresighted way.

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