

Executive Summary of Recent Utah Lake Reports:
(Richards and Miller 2019, Williams 2019, Miller 2019 and Merritt 2019)

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Introduction

This summary is intended to provide important highlights and an integration of recent studies focused on water column loading of total and soluble P and nitrogen to the water column. Sources include P and N loading from tributaries, POTW discharges, sediment recycling and atmospheric deposition. Recent results of mass balance modelling and predictive eutrophication status based on P loading are also included. Assessing the ecological integrity of Utah Lake in relation to anthropogenic forces, invasive species, current trophic status and the potential for recovery are also discussed. Reports that are summarized include: Richards and Miller 2019, Williams 2019, Miller 2019 and Merritt 2019.

Physical setting

While one of the largest freshwater lakes in the western US (93,000 acres), Utah Lake is a tiny remnant of ancient Lake Bonneville. Since about 1850, tributary flows have been largely diverted for agricultural and municipal use, vastly reducing the flushing rate (approximately 1 volume replacement every 2 years under normal runoff) and with approximately 4 ft of evaporation each year (52% of the lake volume), the lake is continuously slightly saline. Along with overfishing of endemic fishes, introduction of invasive species, rapid urbanization of the Utah Valley, these large alterations have rapidly shifted Utah Lake to an altered stable state that includes shallow, warm water, dominance of the benthivorous carp, elevated turbidity, loss of critically important filter feeding mollusks and other invertebrates and frequent cyanobacterial blooms (cyanoHABs). Utah Lake was in a natural mesotrophic to eutrophic state as a result of being a quasi terminal lake that receives elevated P concentrations in tributaries that flow through P-rich sedimentary bedrock. Elevated P concentrations have been further exacerbated by myriad landscape disturbances associated with urbanization, such as stormwater and wastewater discharges, agricultural development and runoff, and increasing loadings of P-rich dusts from nearby west and southern desert playas from continued water diversions, a drying climate and intensive agricultural practices.

Sources of P

Utah Lake is unique in that the majority of nutrient inflows remain in the Lake; with estimates that over 90% of nutrients carried by tributary streams are captured and remain in the Lake. Total nutrient inflows to UL includes streams, wastewater treatment plants, overland flow and additional nutrient reservoirs including sediment, soils, atmospheric deposition (dust and wet deposition), and biological sources (e.g., carp).

Major tributaries drain large watershed areas that are comprised of ancient sedimentary formations (e.g. the Park City Formation), that are naturally rich in P and other limestone minerals. These P-rich tributaries predispose Utah Lake to elevated water column P concentrations and recent measurements of sediment P have documented very high total P and soluble P in the pore water. Further evidence of the transport of these P-rich sediments was documented in the delta sediments of Deer Creek Reservoir. Filled in 1940, sediments of the Provo River delta are relatively recent but have similar P concentrations as sediments in Utah Lake. Sediment P concentrations in Utah Lake ranged from 280 to 1710 mg/kg dry weight with an average of 666 mg/kg.

Phosphorus concentrations have been found to be similar between Utah Lake sediments and soils that were collected generally within a few 100 meters of the shoreline. While soil samples may not represent recent sediment deposition and concentrations, they do likely represent dust deposited from near and far field playas and disturbed agricultural lands that are reaching Utah Lake and contribute to both the nutrient and sediment load to the lake.

Two studies have evaluated the bioavailability of P in Utah lake sediments. The pore water and loosely bound (treated with NH_4Cl), represented 40 to 60% of 10 random sediment samples (Randall et al. 2019). Some samples included as much 5 mg/L SRP in the pore water. While not a part of the present study, Hogsett, (2019) measured sediment P flux in SOD chambers designed to seal to the lake bottom during incubation. They reported a lake-wide estimate of 1500 metric tons of SRP released to the water column annually. Randall, et al. 2019 noted that P concentrations in the UL water column are likely governed by sediment P sources, rather than by inflows into the lake.

Modeling the Nutrient Budget, and contribution from Wastewater Treatment Plants

Dr. LaVere Merritt, BYU Professor Emeritus, calibrated two models, the LKSIM model, which is primarily a mass balance model of water and solutes that enter the lake, and the Larsen Mersier model, which is a modification of the Vollenweider (1975) eutrophication model. Data was obtained from a comprehensive sampling program, including flow, P, N and other solute concentrations, including POTW “end of pipe” measurements collected during a joint UDWQ-Central Utah Water Conservation District sampling effort from 2009 to 2013. Additional data sets since that time include the same and some additional sites with efforts from the WFWQC and US Bureau of Reclamation (Table 1). The reduction in P load, primarily starting in 2014 is primarily due to load reductions from upgraded Timpanogos Special Service District Water Reclamation Facility and the Orem City Water Reclamation Facility to include biological nutrient removal. Both of these plants were brought online in 2013 and the POTW contribution has recently been estimated to be 120 tons/yr. Dr. Merritt’s annual loading results are listed in Table 1.

Table 1. **Utah Lake Nutrient Loadings¹ – 10 yr, 2009 – 2018 period – w/o Atmos. Dep².**

Water Yr.	Total P <u>Ton/yr</u>	Soluble Reactive P <u>Ton/yr</u>	Dis.Inorg. N <u>Ton/yr</u>
2009	264	222	2116
2010	246	210	1886
2011	309	256	2710
2012	237	202	1719
2013	240	206	1715
2014	232	198	1663
2015	227	196	1551
2016	165	132	1446
2017	199	157	1889
2018	<u>169</u>	<u>135</u>	<u>1476</u>
Avg	229	191	1815

¹With modified nutrient removal at the Timpanogos and Orem WWTPs after 2015.

Using mass balance results, the P retention coefficient of Utah Lake is about 0.9, or about 90% of all P entering the lake from tributary flow is retained. Evaluation of similar eutrophic lakes and hydrologic retention times suggests an average retention coefficient of 0.5 would be normal (Figure 1). Merritt suggests that this discrepancy supports the notion of ongoing chemical precipitation of P with various minerals, including carbonate-minerals such as apatite or redox-sensitive complexes with Fe, Al, Mn and Mg - a process that is just now beginning to receive necessary attention. For example, redox sensitive compounds are likely responsible for the elevated soluble P found in pore water samples (Randall 2019) and the net sediment efflux of P to the water column reported by Hogsett et al. (2019). This supports the conclusion of Randall (2019) where it is noted that sediment chemistry and nutrient reflux plays a major role in the nutrient budget of Utah Lake.

As mentioned above, Merritt's current use of the Larsen-Mercier model includes only tributary inflow and a P retention coefficient of only 0.5 (Figure 1). This represents a very conservative estimate of P loadings and retention. While this seems like a minor curiosity, it does not consider additional loadings from atmospheric deposition nor extremely large contribution of P to the water column from sediments. Briefly, even if 100% of P was removed from POTW discharges, plus an additional 25% removal from other tributary and nonpoint sources, the model predicts that an average of 90 ug/L would remain in tributary contributions, an amount sufficient to retain a eutrophic condition in Utah Lake.

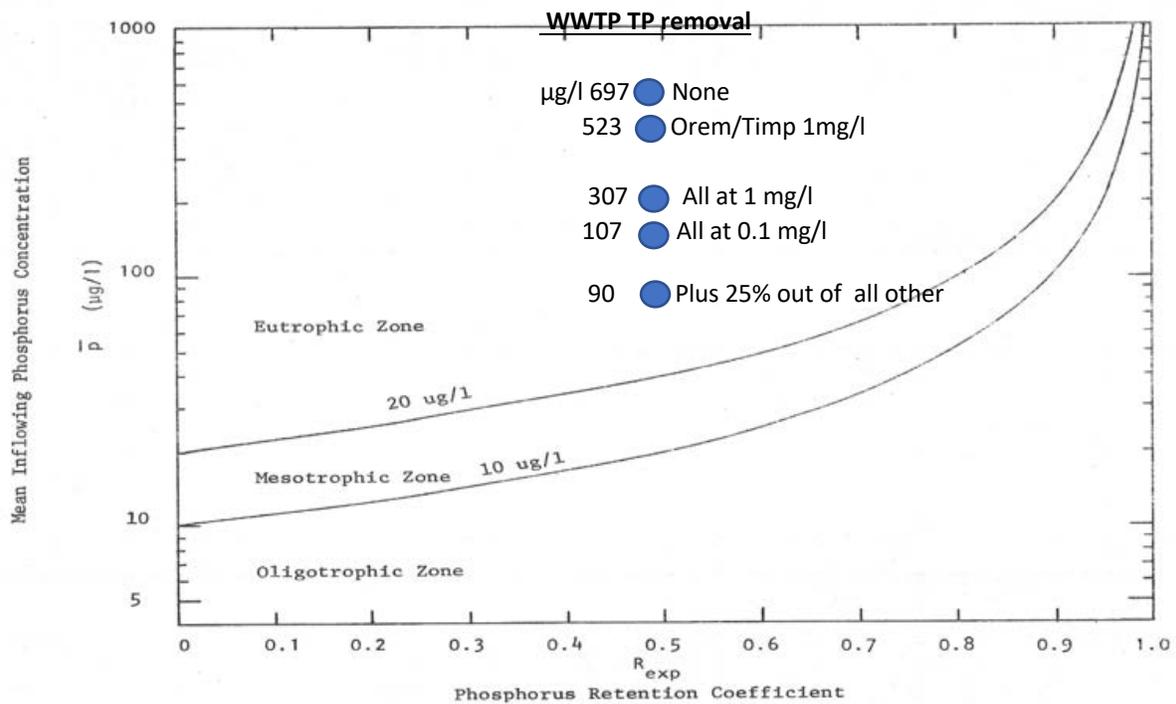


Fig. 1. Larsen-Mercier Trophic State Model for Utah Lake (w/o AD)

Dr. Merritt further noted that “in-situ concentrations of TP in the lake appear to have not changed since appreciable lake sampling began some 30 years ago, while during this period (w/o AD) the TP loadings increased perhaps 40% over that time period to 2016 and now have dropped back to about the same as 30 years ago with Phase 1 TP removal at the Timpanogos and Orem WWTPs. The long-term monthly summer averages of TP values along the middle of the main lake have remained mostly in the 50 µg/l to 70 µg/l range during this period and the outflowing Jordan River TP concentrations have been essentially constant at an avg. of about 50 µg/l. This indicates Lake TP retention (largely precipitated) in the 2018 water year was 89% (w/o AD) and 95% (w AD). Additional evidence of TP precipitation of any amount added above about 50 to 60 µg/l TP is shown in Fig. 2. This plot shows that the amount of algal growth (as indexed by Chlorophyll-a) is independent of the external TP loadings”.

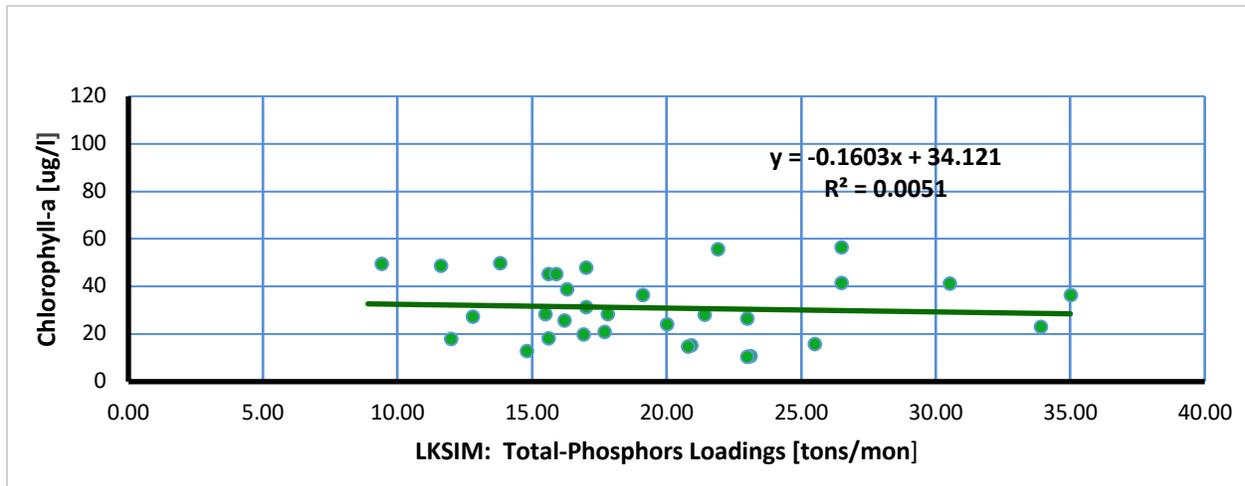


Fig. 2. Utah Lake TP Loads – Chl-a for 29 yrs (>10 & < 60) – July-Sept mon. avgs. 1990 -2018.

Atmospheric Deposition

Williams has summarized the two studies performed by graduate students of the BYU Engineering Department:

Academic studies have focused on long-range dust transport and the NADP has developed a set of guidelines to locate sampling stations at positions that are not influenced by short-range atmospheric transport. For the studies reported here, the sampling stations are located near the shore of UL to measure both long-range and short-range atmospheric transport to UL. Some of these stations do not meet all the NADP guidelines for long-range atmospheric transport studies, as short-range transport is the dominant source for UL. Olsen, et al. 1999 contains a table that documents the sample locations of both studies and whether the locations are in compliance with NADP guidelines. The majority of the guidelines are followed, though 3 of the sites have irrigated fields within 500m, one has a small gravel driveway within 25m, but it is not heavily trafficked, and one has a parking lot Figure 8 Five dust sampling locations for both [10, 12]. UTAH LAKE SOILS, SEDIMENT, AND ATMOSPHERIC DEPOSITION RESEARCH SUMMARY Page 12 60 m from the site. As noted, these do not meet all the NADP guidelines for long-range transport, but instead are sited to measure total nutrient loads to UL.

Goodman, et al. [30] published a study that evaluated dust along the Wasatch Front in about urban and the mountain snowpack. They found that urban and snow dust are similar to playa dust and that playas contribute the majority of dust to the urban Wasatch Front and mountain snowpack. They also note that urban aerosols contribute substantial amounts of anthropogenic trace elements and that these elements are soluble and readily available in the environment. The sampling locations used in this study follow NADP guidelines are designed to measure regional or global dust transport, they do not account for local transport mechanism as these do not affect regional deposition. These data are representative of regional deposition to UL and

point to playas as a major source of nutrients.” Two studies have been performed thus far. Both used samplers placed at similar locations with the second using a different way of analyzing the attenuation of dust/nutrients across the lake. As mentioned above, Strict adherence to NADP protocols was not followed. The primary reason being the NADP focuses on regional transport of airborne constituents rather than local sources. Thus, NADP samplers are placed at large distances from urban centers or other known dust sources such as lake playas or agricultural zones.

Alternatively, the goal of these two studies was to include both local and regional sources of nutrients to Utah Lake. Therefore, while sampler placement has received criticism by other NADP researchers for including this local “contamination”, estimating the contribution of this local contamination to the Utah Lake nutrient budget is a key objective of these studies with the only inherent variable left to be resolved (after important modifications of the samplers), being the appropriate attenuation equation(s) used to describe the distribution of dust/nutrients across the lake. Another large difference between NADP protocol and the sampling protocols used in the BYU studies is that NADP does not currently recognize nor provide protocols for determination of dry deposition estimates and in fact, NADP does not officially collect and report atmospheric deposition of P. Despite this controversy, this is critical in that dry deposition in arid regions has been recognized in the literature as being more significant than wet deposition-primarily from mobilized dust. For this reason, WFWQC has enlisted the services of NADP experts to assist in developing the most rigorous sample design reasonably possible to gather this necessary information.

The collection of dry deposition samples includes the complicating variable of insects, both from the emergence of aquatic insects as well as insects leaving neighboring agricultural zones such as bees and wasps. In partial resolution of this issue, Olsen et al. 2019 omitted sample data that included animal or plant debris in the collection bucket which included the omission of most of the dry deposition samples but only very few wet deposition samples. These samples represent the low estimates in the Olsen et al. data of 8 tons total P and 48 tons total N over the eight-month sampling period. Including the dry deposition samples, but removing the insects at the time of sample collection, resulted in the higher estimates of 350 tons total P and 460 tons of total N. This reveals the major reason why dry deposition measurements are so complicated.

WFWQC researchers have acknowledged the recommendation to omit insects and plant debris from the samples and sampler adjustments are now being made to install screens just above the water surface and to elevate the samplers to 2 m height in the dry deposition samplers. In addition, a sampler will be placed near the center of the lake, near Bird Island, in order to measure deposition at that location. This will provide the needed improvement in estimating the attenuation of dust/nutrients across the lake under differing atmospheric conditions such as wind direction and intensity.

Another atmospheric deposition study has been underway Dr. Wood Miller of BYU for the past three years. Measurements in this study are largely considered bulk deposition as samples are collected from rain gages within 24 hours of each rainstorm. Thus, samplers are collecting dust on the surfaces of the funnels between rain events, although it is possible that dust that falls on these surfaces may easily be remobilized and lost with wind events. Therefore, samples may be considered conservative bulk deposition estimates. The samples are initially reported in mg/L in the rainwater. Using 5 mg/L as the cutoff for outliers, average TP concentrations were 0.46 mg/l with 15 outliers. For all 370 samples, the average TP concentration was 0.77 mg/l. The highest measured TP concentrations were 1.35 and 1.16 mg/l at Genola and Lincoln Point, south of the main lake. The lowest TP concentrations in the sample set were 0.10 and 0.25 mg/l at BYU and Spanish Fork, east of the lake. The highest TP concentrations w/o > 5 mg/l outliers were 1.07 and 0.55 mg/l at Mosida and Lincoln Point, respectively, south of the main lake. The high concentrations are about 5 to 12 times higher than the low concentrations. Summer samples generally had higher concentrations than winter samples, a characteristic that is likely due to drier and more windy conditions, but also a time when insect contamination was possible. Removing the outliers resulted in an average deposition to the lake of 20 tons per year. As with the other AD program, adjustments to the samplers, including the installation of screens are being evaluated.

Factors Affecting Ecological Health and Integrity of Utah Lake

Utah Lake is the last large freshwater remnant of Lake Bonneville. It has undergone several primarily anthropogenically-induced ecosystem shifts and hysteresis over the past 150-200 years resulting in alternative states in a now analog environment.

The goal of our ecological research on Utah Lake is to increase our understanding of its ecological health and integrity, including environmental and ecological factors that can reduce its resistance and resilience and subsequently its ability to self-regulate cyanoHABs. We are doing this in a scientific holistic manner because we are well aware that focusing on only one or two stressors to the lake's ecosystem cannot possibly provide managers with enough understanding of how best to manage the lake into the future.

Such changes are primarily the result of the following stressors:

- Watershed diversions and water quality degradation
- Unnatural water level fluctuations
- Increased turbidity
- Reduction to near zero flush rates
- Changes to temperature regime
- Loss or reduction of aquatic vegetation and allelopathy
- Dominance by invasive fishes
- Watershed-wide pesticide use

- Metapopulation and metacommunity dynamics: Isolation
- Pharmaceuticals
- Urbanization, the most recent and ongoing catastrophic shift
- Loss of native zoological assemblages including mollusks
- Analog zooplankton assemblages
- Transition from native benthic ecosystem engineers to analog invasive engineers.

As a result, Utah Lake has become a poorly functioning analog of its former self. Our society has inadvertently created a much-simplified analog Utah Lake ecosystem that often favors descendants of a cyanobacterial lineage whose primordial ancestors evolved in simple, stressed systems billions of years ago, not completely unlike present Utah lake. Top-down biotic regulators, such as zooplankton, and benthic ecosystem engineers, including native mollusks and midges, have also lost most of their ability to regulate cyanobacteria blooms.

Evidence of altered States

Temperature is one of the most important environmental factors affecting cyanoHABs throughout the world and obviously in Utah Lake (Wells et al. 2015, Scheffer et al. 1997, Scheffer et al. 2001, Berger 1975, Reynolds 1988, Bissenger et al., 2008, Eppley 1972). In general, cyanoHABs increase with increased temperatures, although winter low temperatures also can influence cyanobacteria populations, including ice covered conditions. Climate change forcing will directly affect Utah Lake water temperatures and other key ecological variables and interactions that regulate cyanoHABs (Wells et al. 2015, Scheffer et al. 2001).

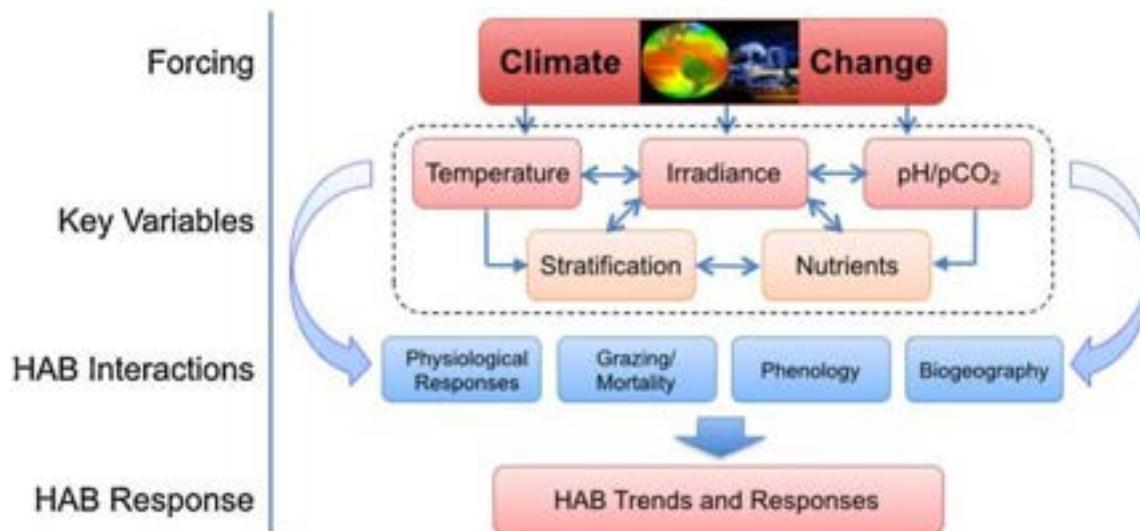


Figure3. Generalized relationships between cyanoHAB trends and responses, interactions, key variables, and climate change forcing. Taken from Wells et al. 2015.

These include cyanoHAB motility (Kamykowski and McCollum 1986), germination (Montresor and Lewis, 2006, Yamochi and Joh, 1986). Nutrient uptake, photosynthesis, and other physiological processes (Beardall and Raven 2004, Raven and Geider 1988) are all influenced by temperature (Wells et al. 2015).

Based on our research and knowledge of the Utah Lake ecosystem, we anticipate that temperature effects will cascade throughout the ecosystem as the climate changes including; affecting cyanobacteria and algal assemblages, likely increased bloom intensity and frequency, competition between phytoplankton grazer assemblages and their top down control, parasites, and planktivorous fish assemblages (Alheit et al., 2005; Edwards and Richardson, 2004; Hansen, 1991). Without a solid understanding and incorporation of the effects of changing temperature regimes, establishing nutrient criteria levels for Utah Lake will fall far short of any intended cyanoHAB management goals.

Temperature and the ability to outcompete green algae and diatoms in turbid, low light conditions, are perhaps the least considered factors controlling (favoring) the presence and dominance of cyanoHABs in Utah Lake. Yet, they are undoubtedly among the most important. Moreover, these factors are most likely exacerbated by loss of filter-feeding mollusks and other invertebrates, pesticide use, shallow depth and sediment nutrient recycling.

Our research group has collected environmental and ecological data and holistically examined many of the problems that face Utah Lake for almost a decade. We conclude that without such comprehensive and integrative knowledge of these synergistic intricacies, we could have thoughtlessly assumed that nutrient reduction alone will control future cyanobacteria blooms.

Allelopathy and CyanoHABs

An important but completely overlooked ecological factor that likely contributes to cyanoHABs in Utah Lake at least within the littoral zone, is the ecosystem - shifting reduction and loss of allelopathic chemical controls produced by submerged (SAV) and emergent aquatic vegetation (EAV; aka macrophytes) (Scheffer et al. 1993). It is well known that most species of aquatic vegetation produce allelopathic chemicals, many of which negatively affect cyanobacteria (Hilt and Gross 2008; Al-Shehri 2010; Jasser 1995; Korner and Nicklisch 2002; Santonja et al. 2018; and others). While the density and extent of submerged aquatic vegetation in Utah Lake is unknown there is substantial evidence that prior to pioneer settlement, emergent aquatic vegetation was abundant across the broad, shallow littoral zones of the lake. Allelopathic activity by this EAV likely contributed to controlling cyanoHABs pre-settlement.

Pesticide Use in the Watershed

While no studies could be found documenting pesticide concentrations in Utah Lake, the proximity of agricultural activity adjacent to several tributaries, springs and drains suggest that pesticides are reaching the lake. Particularly, the broad use of the glyphosate, Roundup®, for phragmites control could clearly be impactful on the aquatic ecosystem of Utah Lake. Hundreds of studies have shown that algae are particularly vulnerable to target and non-target herbicides. For example, several studies have shown that algae are particularly vulnerable to glyphosate-based herbicides because of their physiological and biochemical similarity with terrestrial plants (Annett, Habibi and Hontela, 2014; Tsui and Chu, 2003). Sáenz et al. (1997) showed that growth of green algae (*Scenedesmus acutus*, *S. quadricauda*) was inhibited by glyphosates and Tsui and Chu (2003) showed that the diatom species, *Skeletonema costatum* was more affected than green algae. Increased temperatures can also significantly reduce algal diversity exposed to herbicides. Obviously, benthic algae are also negatively affected by herbicides.

Cyanobacteria are much less susceptible to herbicides than are algae (diatoms and greens), which can result in their strong competitive advantage over other phytoplankton in Utah Lake. This competitive advantage combined with; 1) low light competitive advantage (see section Turbidity Favors CyanoHABs), 2) little or no lake water flushing competitive advantage (see section Flow Rates Favor CyanoHABs), 3) negative effects of pesticides on zooplankton grazers (this section), and 4) extinction prone populations and communities due to isolation. Isolation of zoobiota from potential rescue populations allows cyanobacteria to dominate the Utah Lake ecosystem. Subsequently, cyanoHABs go virtually unchecked by other remaining compromised and isolated members of the lake's aquatic community.

Development of a Multimetric Index of Biological Integrity

We have identified several dozen metrics that can be used to develop an index of biological integrity based on single or multiple stressors. The Utah Lake MIBI is composed of relatively easy to measure primary metrics specifically targeting designated beneficial uses (fisheries, shell fisheries (e.g. mollusks), birds, and the aquatic life they depend (e.g. zooplankton, benthic invertebrates) including:

1. Benthic macroinvertebrate diversity,
2. Benthic macroinvertebrate secondary production (biomass as a substitute),
3. Zooplankton diversity,
4. Zooplankton secondary production (biomass as a substitute),
5. Mollusk diversity,
6. Mollusk densities,
7. Fish condition index.

An easy-to-measure metric for recreational beneficial use (e.g. swimmable) will be to create of a DNA identification code of toxin producing cyanoHABs and develop metric baseline values. This currently remains in the development stage. Researchers are investigating which species are responsible for the various identified toxins. However, this is complicated by the increasing awareness that some strains of certain species may produce toxins while other strains do not and whether these differences are tied to specific waterbodies, water chemistry or allelopathic responses. Therefore, there is a huge need to identify the genes responsible for specific toxins and for the species and strains that inhabit Utah Lake. The goal would be to identify the potential for a bloom with the presence of the toxin-producing gene. Additional research would provide a calibration curve between gene frequency and actual toxin production in the environment. This will further support the elimination of species counts as an indicator of beneficial use support.

A brief summary of secondary metrics includes:

- Phytoplankton, zooplankton, benthic invertebrate, mollusk, and fish taxa diversity indices, e.g. evenness, effective number of taxa,
- Zooplankton family relative abundances and ratios,
- Zooplankton, benthic invertebrate, mollusk, and fish functional traits indices: Particularly for zooplankton e.g. body size; mesotrophic vs. eutrophic zooplankton taxa ratio, taxonomic group changes (Cladocera, copepods, rotifers, etc.).

The use and validity of all of the primary and secondary metrics included in the MIBI are well grounded in the ecological and bioassessment literature. WFWQC research and monitoring protocols provide the intensity and quality of data to continue developing this MIBI.

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Utah Lake Hydrology, Nutrient Loadings and Water Quality

Including results and information from the

LKSIM Model

Report to Wasatch Front Water Quality Council

By

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Dec 2019
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INTRODUCTION

LKSIM: Type of Model

The LKSIM model is conceptually a mass balance model that tracks Utah Lake's water and major dissolved ions for monthly time steps over a desired time period. The working model is the combined computer code that handles all of the data, calculations and tabulations that "runs" the conceptual model and simulates the lake's water quantities and conservative ion concentrations – water quality.

Input Data Generation

LKSIM was developed by Dr. LaVere B. Merritt many years ago to systematically handle the large amounts of data needed to simulate Utah Lake – Current simulations use 73 inflowing "tributaries," many of which are grouped inflow categories rather than specific physical surface tributaries. To use LKSIM, large quantities of meteorological, hydrological and water quality data and information must be collected, and hand correlations and calculations made to develop and prepare the computer-format data to be entered into the LKSIM program-input data files.

Conservative and Non-conservative constituents

All water is accounted for as lake inputs and outputs including surface inflows, groundwater inflows, precipitation, evaporation, and outflows. The major dissolved ions (salts) are tracked. Some are considered to be essentially "conservative" meaning that they are assumed to stay in the water as ions with no, or very little, removal. These are the cations Potassium (K⁺), Magnesium (Mg⁺⁺) and Sodium (Na⁺). Calcium (Ca⁺⁺) is the non-conservative major cation since large quantities are precipitated, to become sediment minerals, in Utah Lake. Calcium precipitation quantities are determined by reducing concentrations to observed long term monthly average values at the end of each month, the net being the precipitated amount.

The major conservative anions are Chloride (Cl⁻) and Sulfate (SO₄⁻⁻). Bicarbonate (HCO₃⁻) is the non-conservative major anion since large quantities convert to Carbonate (CO₃⁻⁻) and are precipitated. Precipitation quantities are determined as for Ca⁺⁺. Total Dissolved Solids (TDS) is the total of all of the major ions in solution. During simulation runs, TDS is corrected monthly for the precipitated Ca⁺⁺ and HCO₃⁻⁻.

Nutrient tracking

Since the LKSIM model doesn't include any reaction equations, nutrients are simply tabulated as amounts going in and out of the lake each month, with the net nutrient quantities being retained in the lake.

Years simulated

Over the years data have been collected/developed for 1930 – 2018 water years (water year is 1 Oct of prior year through 30 Sep of the water year.) LKSIM can be run for any time increment of 1 year or more. The beginning year of 1930 was a practical one since the Federal Bureau of Reclamation often used that date as a starting time for planning many area projects. Currently it takes about a man-month of time to collect a past year’s data from the various agencies and entities, and correlations and calculations done to generate the data to be entered into the computer-input files. Entry of the new prepared data and run preparation requires a few days. A LKSIM simulation run takes only a few seconds on a desktop computer.

RESULTS

Simulation for 1930-2018 (89 years)

Figure 1 is a plot of TDS and lake levels for the 89 yr (1930 – 2018) period.

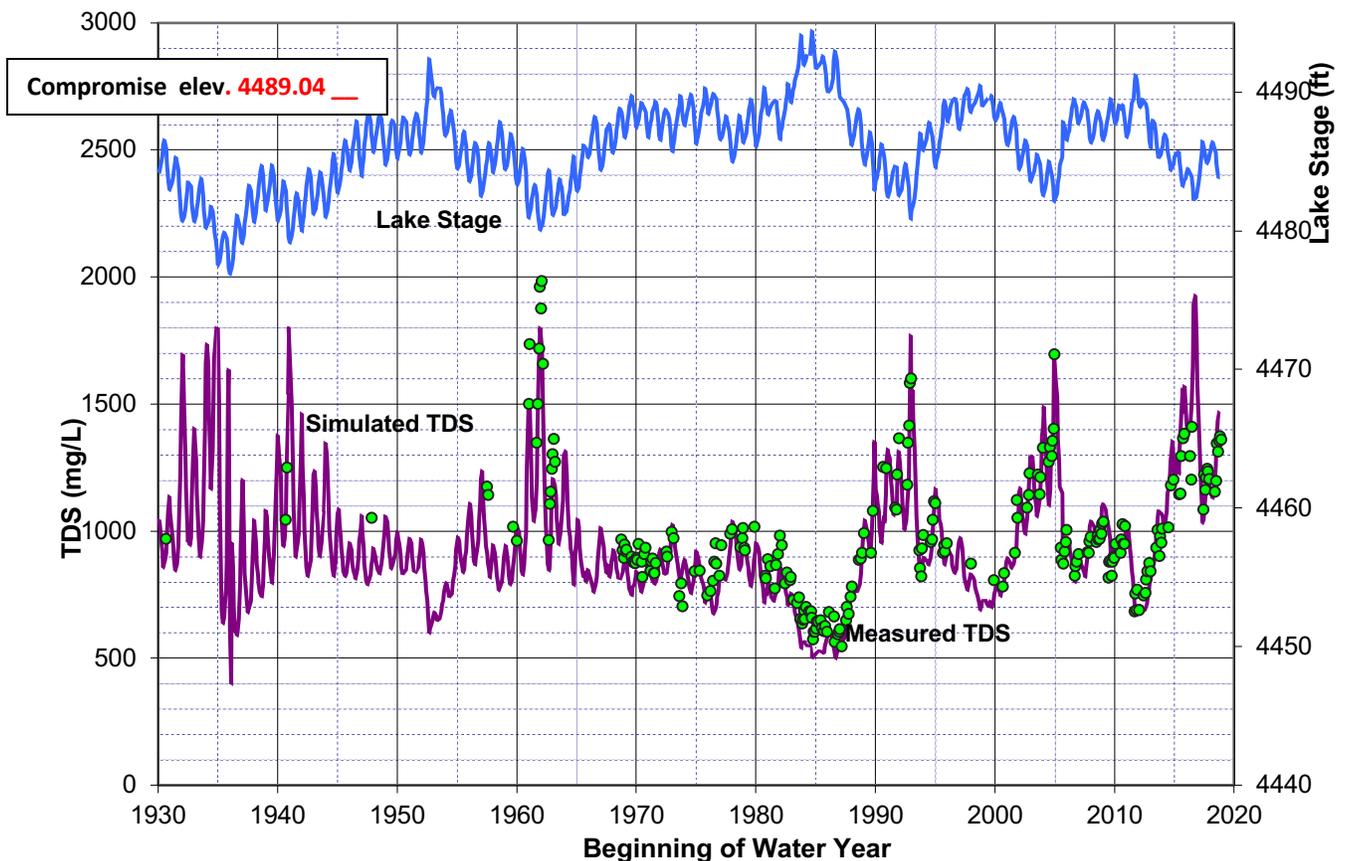


Fig. 1. Total Dissolved Solids and Water Levels in Utah Lake

In Fig 1, the upper plot line and right scale show lake levels (stage) – note that above the indicated Compromise stage at 4489.04 the lake begins to flood surrounding areas and the outlet gates must be fully open to mitigate flooding. At the lowest point in 1936 the lake was essentially dry – with a depth of only a foot or so in the shallow pond-lake in the middle of the lake bed. During more recent droughts, bottoming in 1993, 2005 and 2017, water was some 3 to 5 feet deep in the middle of the lake. The depth range is about 20 feet from the lowest level in 1936 to the highest in 1984. Mainly due to less severe droughts, lower demand to cover downstream water-rights and mitigation management by the Central Utah Water Conservancy District, both tops and bottoms are less extreme recently and the depth range is about 10 feet – at Compromise depth of 4489.04 ft the lake averages about 9 ft deep, with mid-lake depths of about 13 feet.

The continuous line in the lower plot is LKSIM simulated TDS concentration for the main lake and the small circles are average measured values in the main lake. Note that TDS varies fourfold from about 500 mg/l to 2000 mg/l over the 89 year period.

Water Balance: 2009 – 2018

Table 1 gives the LKSIM water balances for the 2009 – 2018 and 1930 – 2018 periods. About 50% of the inflowing water (Trib. + Precip.) is evaporated during the average year. During The beginning of extreme droughts considerably more water is evaporated than is coming in, and the lake shrinks towards an smaller equilibrium surface area, given its large evaporation rates.

Table 1. Utah Lake Water Balances for recent 10 years and past 89 years

<u>Component</u>	<u>Avg af/yr</u>	
	<u>2009 -2018</u>	<u>1930-2018</u>
Tributary inflows	437919.	520120 .
Precipitation	88964.	97038.
Evaporation	310175.	323943.
Tributary outflow	239663.	293622.
Beginning volume	644128. af	446658. af
Average volume	638831. af	604039. af
<u>Ending volume</u>	<u>414587. af</u>	<u>414587. af</u>

LKSIM results show that mineral precipitation averages nearly 100,000 tons/yr – Due largely to the combination of high pH, Ca++ and HCO3-. If one uses a moderately consolidated density, the precipitation is about 2 inches per 100 years over the entire lake area at compromise level; but, actually, is likely some 3 to 4 inches per 100 years in the mid part of the lake due to the deeper water column from which the precipitates are coming. If typical stream sediment-loads

are applied to the major surface tributaries, the inflowing sediment is only a few thousand tons/yr. This means the inflowing sediments are negligible as to the overall filling-in of the lake; nevertheless they are significant at the mouths of major rivers (American Fork, Provo, Hobbler Creek and Spanish Fork Rivers.)

In-Lake Phosphorus concentrations – mineral precipitation

For 2018, if annual Total Phosphorus (TP) loading is considered mixed into the total inflowing volume and then that water volume reduced some 50% by evaporation, TP concentrations in that water would be from about 470 µg/l to 950 µg/l depending on the amount of Atmospheric Deposition. (AD: 0 to 170 ton/yr for this example.) However, summer in-situ concentrations of TP in the lake appear to have not changed since appreciable lake sampling began some 30 years ago, while during this period (not including any AD loadings) the TP loadings increased perhaps 40% over that time period to 2016 and now have dropped back to about the same as 30 years ago with Phase 1 TP removal to 1 mg/l at the Timpanogos and Orem WWTPs. The long-term monthly summer averages of TP values along the middle of the main lake have remained mostly in the 50 µg/l to 70 µg/l range during this period and the outflowing Jordan River TP concentrations have been essentially constant at an avg. of about 50 µg/l. This indicates Lake TP retention (largely precipitated) in the 2018 water year was 89% (w/o AD) and 95% (w AD.)

Evidence of the lack of algal growth response to external TP loading differences is shown in Fig. 2. This plot shows that the amount of algal growth (as indexed by Chlorophyll-a) is independent of the external TP loadings. This condition is strong evidence that internal cycling is providing the phosphorus needed by algae. (Note that if all monthly average TP values were used, results are essentially the same – there are very few points above 60 µg/l, values <10 or >60 were removed since they likely represent non-presentative, low or high growth periods or samples.)

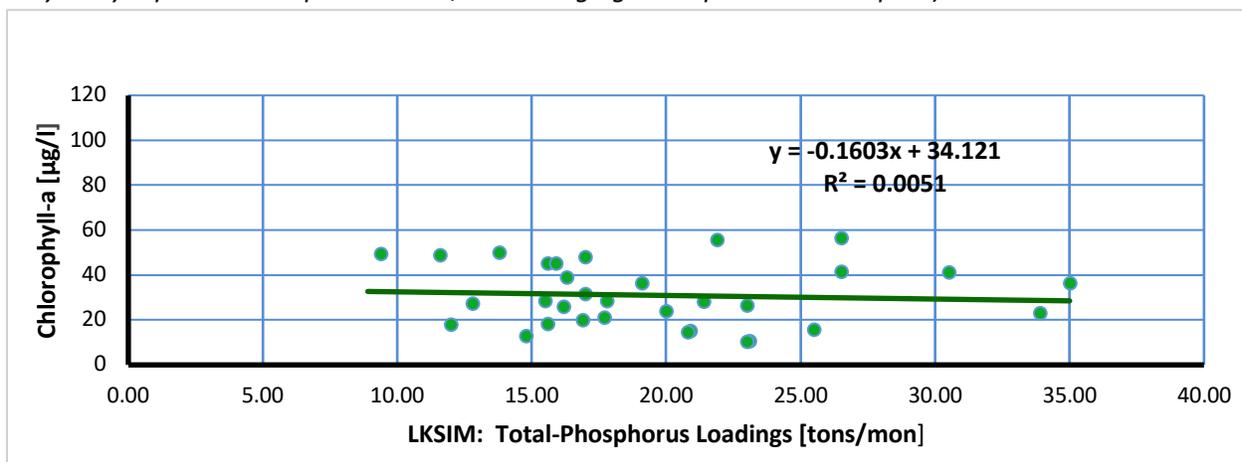


Fig. 2. Utah Lake TP Loads – Chl-a for 29 yrs (>10 & < 60) – July-Sept mon. avgs. 1990 -2018.

Conclusion: Summer TP concentrations in Utah Lake are determined by internal chemical and bio-chemical cycling and are not dependent on the amount of external loadings.

Turbidity and Lake algal growth

Although somewhat subjective at this juncture, long-term experience with Utah Lake characteristics and behavior indicates that **light limitation** due to its natural water turbidity could well be the most limiting growth factor to algae much of the time during the heaviest-growth, summer period. This is a very significant point since it brings into serious question nutrients as key growth limiting growth factors unless they can be reduced enough make the lake values more limiting than light limitation – The Larsen-Mercier trophic state model covered in a following section indicates that in-lake TP values below about 20 µg/l (20 µg/l in-situ is associated with 40 µg/l input and a retention coefficient of 0.5) would be needed to begin to reduce growth below the existing moderately eutrophic algal growth in the Lake. Obviously this TP level is well below the 50 – 70 µg/l commonly observed in the Lake during the summer, and far below the 470 – 950 µg/l in the net inflowing water in 2018. This information seems to support the light-limitation hypothesis.

Nutrient Loadings

Nutrient loadings for the last 10 years are given in Table 2. Reduced nutrient values associated with WWTP (Timpanogos and Orem) nutrient-reduction projects were assumed to be fully operational for those plants at the beginning of water year 2016 (Oct 2015).

Table 2. Utah Lake Nutrient Loadings¹ – 10 yr, 2009 – 2018 period – w/o Atmos. Dep².

Water Yr.	Total P <u>Ton/yr</u>	Soluble Reactive P <u>Ton/yr</u>	Dis.Inorg. N <u>Ton/yr</u>
2009	264	222	2116
2010	246	210	1886
2011	309	256	2710
2012	237	202	1719
2013	240	206	1715
2014	232	198	1663
2015	227	196	1551
2016	165	132	1446
2017	199	157	1889
2018	<u>169</u>	<u>135</u>	<u>1476</u>
Avg	229	191	1815

¹With modified nutrient removal at the Timpanogos and Orem WWTPs after 2015 giving ~1 mg/l TP.

²Atmos. Dep. studies are continuing — AD is very large in the Utah Lake area and is likely in the following ranges: TP: 50 to 200 ton/yr, SRP: 17 to 67 ton/yr, DIN: 150 to 600 ton/yr

General rules of thumb indicate that significant algal growth limitation occurs below TP=10 µg/l or DIN=100 µg/l. Using the 470 µg/l TP loading in 2018 (w/o AD), it is 47 times larger in Utah Lake than noted algal-growth limiting nutrient values.

The following section is included as additional evidence that phosphorus is not, and cannot be made to be, the key limiting algal-growth nutrient in Utah Lake.

Utah Lake Trophic Level – Larsen-Mercier Model

With the germination of the idea that “natural” waters needed a more prominent role in water quality management, modern scientific assessment of lake trophic levels (classification of bio-productivity) began about 70 years ago. As to lake quality, one of the most important in-lake quality factors is the amount of algae that a lake produces. For various reasons phosphorus emerged as the common nutrient often linked to possible algal-growth limitation. However, experience has repeatedly shown that phosphorus is not the actual limiting algal growth factor in many lakes.

Researchers tried various approaches in attempts to use correlation models to help understand lake trophic states and help assess possible control/change strategies. One model that evolved and has proven to be a very useful tool is the Larsen-Mercier model (1976) developed by two EPA scientists. It is particularly useful since it allows a knowledgeable user armed with perhaps only a relatively small amount of data to consider important characteristics of a lake and what might be considered to manage its trophic state via phosphorus manipulation.

The L-M model gives a fairly good projection of actual trophic state (Overall bio-productivity) if phosphorus is actually the most-limiting growth factor for algae over an annual productivity cycle. The model uses mean inflowing Total Phosphorus on the y axis and retention coefficient (R_{exp}) on the x axis. The retention coefficient (R_{exp}) is the fraction of the inflowing phosphorus load that is “lost” in the lake. Researchers also developed models for R_{exp}. Most of these R_{exp} models are functions of the water detention time and mean depth of the lake.

During development of such correlation models and in their application, it is important to realize that a lake may not be phosphorus limited and/or may have high levels of internal nutrient cycling. Nevertheless, the L-M model is one of the best “static” trophic state models to project the expected trophic status of lakes. The model is also very valuable in understanding why some lakes, such as Utah Lake, are not phosphorus limited, and whether changes might cause them to become phosphorus-limited.

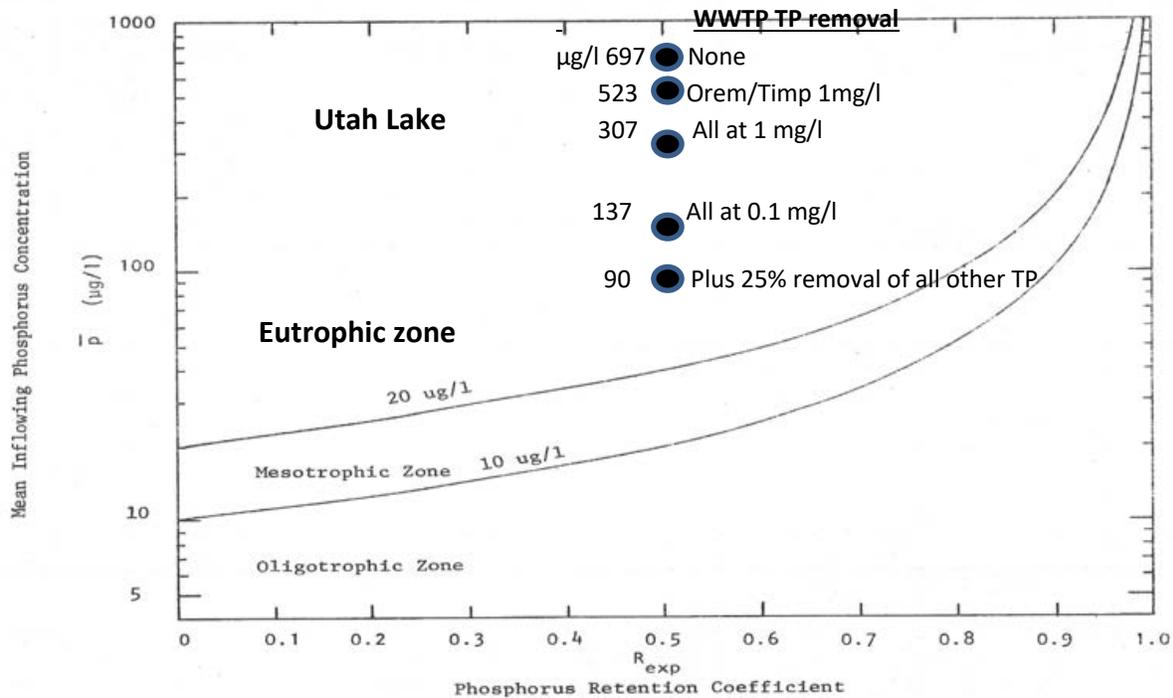


Fig. 3 Larsen-Mercier Trophic State Model showing Utah Lake TP loading for a range of conditions if the lake had a “normal” retention coeff., w/o AD – data for 2009 - 2018)

Using Utah Lake averages for the 2009 – 2018 period, the net inflowing water contained about 700 $\mu\text{g/l}$ of TP, and its R_{exp} was over 0.9 (over 90% of phosphorus retained in the lake.) **These results strongly indicate that Utah Lake is not phosphorus-limited since, if so limited, its TP average inflowing values would need to be some 40-60 $\mu\text{g/l}$ with a R_{exp} in the 0.4 – 0.6 area in order to indicate some propensity for phosphorus limitation of algal growth!**

Observations/comments:

1. Utah lake’s current actual R_{exp} is greater than 0.9. R_{exp} values for Utah Lake from correlation equations for phosphorus-limited lakes is about 0.5 (actually some 0.4 to 0.6 as its hydrology varies from year to year.) This is strong evidence that it is not an external- loading, phosphorus-limited lake since the TP retention is extremely high and far from the normal plot zone of 0.5 for an actual phosphorus-limited lake.
2. How much phosphorus would need to be removed to make the lake phosphorus-limited? Assuming limited internal cycling, to become a phosphorus-limited lake, Utah Lake would need to fit fairly well into the normal TP-limited mode, with a R_{exp} value of about 0.5. Conceptually, as the huge TP loading is reduced the R_{exp} would also decrease and as the TP-limitation zone is approached, it would approach the 0.5 R_{exp} area. This is impossible for Utah Lake as shown in Fig. 3. It also shows that even extreme TP removal from all controllable sources, likely costing \$billions, would still not

move the lake into the zone of a phosphorus-limited system; this is even more so if Atmospheric Deposition nutrients were included!

3. Utah Lake in-situ TP concentrations appear to be controlled by internal bio-chemical cycling rather than by external input loadings. Even in the absence of human activities and contributions, the natural inflowing rivers, streams and drainage waters appear to have contained some 30 to 70 $\mu\text{g/l}$ TP. When these levels are doubled in Utah Lake by evaporation of 50% of the inflowing water, external loadings would be equivalent to about a 100 $\mu\text{g/l}$ loading range. Although still doubted by some people, AD measurements around Utah Lake strongly indicate that AD nutrients are large and very likely are of a magnitude equal to all other TP sources, including the WWTPs.
4. As compared the algae needs, large phosphorus accumulations exist in the bottom sediments of Utah Lake. The lake has been accumulating it for thousands of years. Practically, that accumulation represents an essentially inexhaustible source of phosphorus to the lake. The actual phosphorus concentrations in the overlying water column depend many bio-chemical and environmental factors but these appear to be moderating phosphorus concentrations largely to a 50 to 70 $\mu\text{g/l}$ range, in-situ, during the summertime. Since actual in-situ values need to be near 10 $\mu\text{g/l}$ for significant algal-growth limitation, “Mother Nature” will not allow phosphorus to be the limiting factor to algal growth in Utah Lake!

Misc. Notes: Phosphorous values above don't include Atmospheric Deposition, values for which are still being firmed up – AD data collected over the last three years indicate that AD values are very high in the Utah Lake area, and add significantly to give much higher actual total nutrient loadings.

It is extremely important to avoid the error of assuming that all lakes are phosphorus limited – in the case of Utah Lake, erroneously assuming this might lead one to think phosphorus reductions might move down along the 0.9 Rexp line on the L-M plot. This is incorrect since Utah Lake, to be an actual phosphorus limited lake, would need to exhibit an actual Rexp in the 0.5 area as predicted by the various Rexp correlation equations. Another way to understand this point is to realize that, if one assumes the L-M model is always accurate for the actual measured Resp, Utah Lake could be made oligotrophic simply by adding more and more phosphorus until the measured Rexp value approached 1.0 and the lake is identified as oligotrophic—problem solved?

Utah Lake Research Summary:

Soils, Sediment, and Atmospheric Deposition

Developed for:
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Developed By:
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December, 2019

Executive Summary

This report reviews published literature on Utah Lake (UL) nutrient sources and loadings related to sediment, soils, and atmospheric deposition.

The major points found in these studies are:

- UL is unique in that the majority of nutrient inflows remain in the Lake
- The UL watershed contains significant phosphate deposits
- Phosphate is prevalent in lake sediments with studies documenting high levels, on the order of 1,000 ppm (or mg-P/kg dry sediment) in both Deer Creek Reservoir (upstream on the Provo River) and in UL sediments
- Up to 50% of the phosphate in UL sediments is readily available to the water column
- Measurements of near-shore soils have similar high levels of phosphorous, though less of it is readily available to the water column (10-20%)
- Measurements of phosphates in UL sediment pore water shows average values of 3.96 ppm (mg/L).
 - Pore water concentrations are an order of magnitude higher than those in the water column
 - This amount of phosphorous is very significant as a nutrient pool
- Non-anthropogenic sources may control nutrient concentrations in the water column and these concentrations may be independent of anthropogenic loadings to the lake
- Atmospheric deposition studies measured loading rates around the lake perimeter and estimated lake loads over short periods (7.5 to 8 months, depending on the study).
 - Estimates of total phosphorous and dissolved inorganic nitrogen loadings ranged from 8-350 Mg and 46-460 Mg, respectively (1 Mg = 1,000 kg) depending on assumptions
 - A second study estimated loads in the upper portion of this range
 - These loads were high for both nitrogen and phosphorous, high enough for the lake to be eutrophic on these loadings alone

UL is unique in that the majority of nutrient inflows remain in the Lake; with estimates that over 95% of nutrient inflows carried by influent streams are captured and remain in the Lake. Total nutrient inflows to UL includes influent sources such as streams and overland flow and additional nutrient reservoirs including sediment, soils, atmospheric deposition (dust and precipitation), and biological sources (i.e., carp). Biological and geochemical processes that can release or sequester nutrients from the water column govern these sources. These non-anthropogenic nutrient loadings are tens-of-times larger than those that would designate the Lake as eutrophic.

The UL watershed contains significant phosphate deposits that contribute to nutrients to the lake. Two different studies in Deer Creek Reservoir, an impoundment on the Provo River just upstream from UL and the largest tributary, found sediments with average phosphorous concentrations from of 1,107 – 2,572 mg-P/kg dry sediment with 10-20% of this phosphorous readily available to the water column [1-3]. Two different studies of UL sediments and surrounding soils found that UL sediments ranged from 280 – 1,710 mg-P/kg dry sediment with an average value of about 900 mg-P/kg dry sediment [4-6]. These studies

found that about 50% of the phosphorous in these sediments was readily available to the water column. Near-shore soils around UL were sampled and found to contain about 900 mg-P/kg dry soil, but only about 10% of this readily available to the water column.

One study measured phosphorous in the sediment pore water and the water column to start to quantify the interactions of the sediments and the lake water [7-9]. They found that the total dissolved phosphorous concentrations in the water column averaged 0.43 (\pm 0.14) mg/L and ranged from 0.03 to 1.74 mg/L. They found that pore water total dissolved phosphorous concentrations were approximately an order of magnitude higher than the water column concentrations with an average concentration of 3.96 (\pm 0.85) with a range of 0.30 to 10.8 mg/L (these averages are for samples on the east side of the lake with the range reported for the entire lake) [8, 9].

Two studies measured the atmospheric deposition rates of nitrogen and phosphorous loading to UL that includes both local (near-lake sources) and regional transport [10-12]. They found the average loading across all the sites was 8.10 mg of phosphorous $m^{-2} day^{-1}$, with the average dissolved inorganic nitrogen of 10.23 mg of nitrogen $m^{-2} day^{-1}$. The first study estimated loads values to bound the actual loads to UL over an 8 month periods and found estimates of total phosphorous and dissolved inorganic nitrogen loadings were 8-350 Mg and 46-460 Mg, respectively, depending on the assumptions used [10]. The second study used a different assumptions and measured loadings over a 7 ½ month period and estimated 58.9 tons (53.4 Mg) soluble reactive P, 162 tons (147 Mg) total phosphorous with 123.6 tons (112 Mg) nitrate-nitrogen, and 382.5 tons (347 Mg) ammonium-nitrogen, values near the higher end of the first study.

The studies reviewed in this document show that there are significant nutrient loads to UL from non-anthropogenic sources. These studies estimate that over 95% of the dissolved phosphorous load stays in the lake, contributing to the lake sediments. Sediments and atmospheric deposition represent very large nutrient pools and are readily available to the water column. Randall, et al. [8] postulates that these ***non-anthropogenic sources control nutrient concentrations in the water column and that these concentrations may be independent of dissolved anthropogenic loadings to the lake.***

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Utah Lake Research Summary:

Soils, Sediment, and Atmospheric Deposition

1. Introduction

Utah Lake (UL) is a shallow, turbid, slightly saline, eutrophic lake in a semi-arid area. It has good pollution degradation and stabilization capacity because of its shallow, well-oxygenated, high pH waters. It supports and harbors abundant wildlife as part of a productive ecosystem. The lake provides and supports a wide range of beneficial uses: ecological habitats, water storage, and recreation (e.g., boating, sailing, fishing, and hunting). Abundant wildlife and ecological richness are some of its more significant assets [13].

UL is unique in that the majority of nutrient inflows remain in the Lake; with estimates that over 95% of nutrient inflows carried by influent streams are captured and remain in the Lake. Total nutrient inflows to UL includes influent sources such as streams and overland flow and additional nutrient reservoirs including sediment, soils, atmospheric deposition (dust and precipitation), and biological sources (i.e., carp). Biological and geochemical processes that can release or sequester nutrients from the water column govern these sources. These nutrient loadings are tens of times larger than those that would designate the Lake as eutrophic.

The UL watershed contains significant phosphate deposits that contribute to nutrients to the lake. Two different studies in Deer Creek Reservoir, an impoundment on the Provo River just upstream from UL and the largest tributary, found sediments with average phosphorous concentrations from of 1,107 – 2,572 mg-P/kg dry sediment with 10-20% of this phosphorous readily available to the water column [1-3].

Two different studies of UL sediments and surrounding soils found that UL sediments ranged from 280 – 1,710 mg-P/kg dry sediment with an average value of about 900 mg-P/kg dry sediment [4-6]. These studies found that about 50% of the phosphorous in the sediments was readily available to the water column. Near-shore soils around UL were sampled and found to contain about 900 mg-P/kg dry soil, but only about 10% of this readily available to the water column. A statistical test was used and found that total phosphorous concentrations in the near-shore soils were not different than those of the sediment.

One study measured phosphorous in the pore water and the water column to start to quantify the interactions of the sediments and the lake water [7-9]. They found that the total dissolved phosphorous concentrations in the water column averaged 0.43 (\pm 0.14) mg/L and ranged from 0.03 to 1.74 mg/L. They found that pore water total dissolved phosphorous concentrations were approximately an order of magnitude higher than the water column concentrations with an average concentration of 3.96 (\pm 0.85) with a range of 0.30 to 10.8 mg/L (these averages are for samples on the east side of the lake with the range for the entire lake) [8, 9].

Two studies measured the rates of nitrogen and phosphorous loading to UL from atmospheric deposition that includes both local (near-lake sources) and regional transport

[10-12]. They found the average loading across all the sites was 8.10 mg of phosphorous m^{-2} day^{-1} , with the average dissolved inorganic nitrogen of 10.23 mg of nitrogen m^{-2} day^{-1} . They then used these rates to estimate total loads to UL.

The first study estimated high and low values to bound the actual loads to UL over an 8 month period. They estimated total phosphorous and dissolved inorganic nitrogen load ranges were 8 to 350 Mg and 46-460 Mg, respectively, depending on the assumptions used (1Mg = 1,000 kg) [10]. A second study used different assumptions and measured loadings over a 7 ½ month period and estimated 58.9 tons (53.4 Mg) soluble reactive P, 162 tons (147 Mg) total phosphorous with 123.6 tons (112 Mg) nitrate-nitrogen, and 382.5 tons (347 Mg) ammonium-nitrogen over this period, values near the higher end of the first study.

The studies reviewed in this document show that there are significant nutrient loads to UL from non-anthropogenic sources. Studies estimate that over 95% of the dissolved phosphorous load stays in the lake, contributing to the lake sediments. Sediments and atmospheric deposition represent very large nutrient pools and are readily available to the water column. Randall, et al. [8] postulates that these non-anthropogenic sources control nutrient concentrations in the water column and that these water column concentrations may be independent of dissolved anthropogenic loads from tributaries or wastewater treatment plant outflows into the lake.

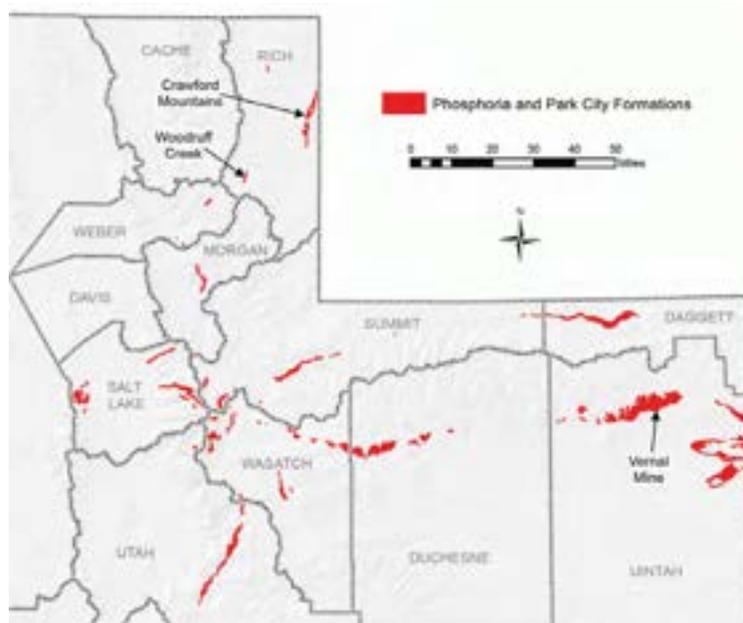


Figure 1 A map of the Phosphoria and Park City formation locations. These formations contain high concentrations of phosphorous, often in economical concentrations and are mined. The map shows current mines. The many of the deposits in Wasatch and Summit Counties are within the watersheds that contribution to UL. The figure is from [14].

This report reviews published literature on UL nutrient sources related to sediment, soils, and atmospheric deposition.

2. Phosphate

2.1 Regional Geologic Sources

According to the Utah Geological Survey [14], Utah has over 100 years of history of phosphate production, primarily for use in fertilizer. They note that potash is also produced in Utah and is another phosphorous source. They further note that while most phosphate production in the United States occurs elsewhere, Utah is a significant producer. The Utah State survey [14] states the phosphate rock quality is expressed as P_2O_5 content with the

richest phosphate rock deposits containing 30% percent P_2O_5 with Utah deposits in this range, with additional lower-grade deposits.

The phosphate resources in Utah are primarily from ocean depositions in northern Utah from the Permian (~250 million years ago) [14]. These deposits are primarily apatite, (specifically, carbonate-fluorapatite) which is the primary ore. The Meade Peak Member of the Phosphoria Formation hosts most of the economic deposits and consists of dark-colored phosphorite (a rock composed primarily of apatite), mudstone, and carbonate [14]. In the watersheds that contribute to UL, the Phosphoria Formation commonly occurs as a zone within the Permian Park City Formation [14]. Figure 1 shows that UL watersheds contain formations with high phosphorous concentrations with historic sediments in UL potentially capturing these materials and having high phosphorous levels.

In Utah, the Meade Peak Member ranges from very thin to about 225 feet thick. Rupke [14] attempts to define Utah's phosphate resources which vary widely due to differing assumptions and minimum grades, but that Utah has hundreds of millions of tons of in-place resources, with significant amounts of both economically viable, and lower grade deposits in the UL watersheds (Figure 1).

2.2 P Geochemical Fractions

In sediments and soils, P can be present in different chemical forms or pools that behave differently, these different forms are called fractions. Some fractions have P that is relatively immobile, not bioavailable, while other fractions or forms are easily mobile and bioavailable [15, 16]. For example, fractions such as apatite minerals do not influence reservoir water quality as they are not bioavailable, while other forms, such as the water-soluble or salt-extractable fractions, can easily be released into a reservoir water column [17].

Accurately measuring P concentrations in different pools is difficult, but the soil science community has developed analysis methods called fractionation, extraction, or selective dissolution methods to analyze the P concentration in different pools [17-20]. These methods use of sequence of different solvents to extract a selected P-pool from the soil or sediment and then use either centrifugation or filtration to separate the supernatant and measure the P concentration. Recent studies of UL soils, sediments, and sediment sources in the watershed quantify both total phosphorous (TP), but also determine in what forms or fractions the P is present. This provides information on how much of the TP in the soils and sediments are available to the water column in UL.

3. P in Sediments

3.1 UL Watershed P

Recent studies [4, 5, 7, 8, 10, 11, 21] have documented high phosphorous concentrations in the sediments within UL and the soils surrounding the lake. Other studies [1, 2, 6, 22, 23] have documented high phosphorous levels in the sediments of Deer Creek Reservoir (DCR), which is a reservoir upstream from UL on the Provo River. DCR is located on the Provo River, the largest inflows to UL. This section will briefly review this work.

3.1 Deer Creek Reservoir (DCR) Studies

3.1.1 Methods

DCR is located upstream from UL on the Provo River, the largest tributary to UL. The reservoir was completed in the early 1940s. Sediments that have accumulated in DCR since that time can be considered analogs of historic sediment loads from the Provo River into UL.

In 2008, Casbeer [2] took advantage of a large drawdown to support dam reconstruction that exposed nearly 2 miles of sediments in the upper reaches of the reservoir. These sediments had been accumulating since the early 1940s. This work has been reported in three publications a Master's Thesis [2], a journal article [1], and a conference paper [6]. Casbeer [2] compared his results with an earlier DCR sediment study performed by Utah State University in the mid-1980s [3].

Casbeer, et al. [1], [2] includes a detailed comparison of the results and methods of the recent and previous studies, but notes that the results were similar, though there were differences in the laboratory methods used [1, 2]. One major difference was that the earlier study included an additional fraction to determine Fe-bound or oxide-occluded P [3], in [1], P from this fraction was included in the Fr.HCl fraction and which the earlier study grouped the first three fractions in [1, 2] into one "available" fraction.

Casbeer, et al. [1] reported collecting 91 samples on a grid 100 meters (m) by 200 m with an area of approximately 750,000 m² (185 acres). These samples consisted of both surface samples and at some locations, vertical samples at different depths. They determined the amount of water soluble P for all the samples at measured P in four fractions for 19 samples. The resulting percentages were similar and Casbeer, et al. [1] states that these percentages can be used to estimate fractional P, based on TP measurements.

For the P-fractionation they followed the Moore, et al. [24] fractionation scheme which distinguishes five separate P forms. Each fraction is defined by the extracting solution used:

- Fr.W: water soluble and interstitial water P
- Fr.KCl: anion exchange sites, loosely sorbed P
- Fr.NaOH: Al- and Fe-bound or adsorbed P
- Fr.HCl: Ca-bound (apatite) P
- Fr.K₂S₂O₈: residual (mostly organic) P

Casbeer [2] notes that the final step can be performed in two ways, both requiring sediment digestion. The digestion can be performed either on the sediment residual from the fractionation sequence, or on a separate sediment aliquot. If the latter is used, the results provide the total P concentrations in the sediment and the other pools are subtracted to obtain the Fr.K₂S₂O₈ pool.

Table 1 Average water-soluble (Fr.W) P concentrations in DCR sediment with depth. Where *n* is the number of samples in each average [1, 2].

Depth	n	CP.avg (mg·kg ⁻¹)
Surface	55	5.64
6 in	15	4.43
12 in	13	4.06
2 ft	4	3.99
All	87	5.12

In general, P in the first two fractions is readily available to the water column. The 3rd fraction, Fr.NaOH, can be released from the sediments in anoxic or anaerobic iron-reducing conditions, a frequent occurrence in reservoir sediments. The other two fractions, Fr.HCl and Fr.K₂S₂O₈ are not readily available to the water column.

3.1.2 DCR Results

Of the 91 samples taken, they measured 72 samples for Fr.W, or the fraction of P that is water-soluble only and performed a complete fractionation on 19 samples.

Figure 2 and Table 1 present the results of the Fr.W (water-soluble fraction) which was determined by mixing the sediment sample with deionized water, then filtering the resulting liquid and measuring the P concentration; the sample was not shaken. The average concentration of Fr.W (water soluble P) was 5.19 mg-P/kg dry sediment in the 19 samples used for full fractionation, the average for entire 91 samples was 5.12 mg-P/kg dry sediment.

The study evaluates spatial trend by using longitudinal (along the talweg) and latitudinal (across the talweg) transects. They also performed a spatial interpolation on the results to generate a map of Fr.W P concentrations (Figure 1). This analysis showed that in general, concentrations were highest near the entrance to the reservoir and near the talweg, with concentrations dropping slightly as you moved downstream and away from the talweg (Figure 1).

The vertical distribution of the Fr.W P is shown in Table 1. This table shows that there seems to be a slight vertical trend with slightly higher concentrations near the surface (5.6 mg-P/kg dry sediment) than at a depth of 2 ft. (4.0 mg-P/kg dry sediment).

Table 2 presents the results of the fractionation analysis of 19 samples. The results show the P available to the water column is relatively high, over 180 mg-P/kg dry sediment. This is the P in the first three pools, water solution (extracted using deionized water), Fr.KCl (extracted using water with potassium chloride salt), and Fr.NaOH (which represented P that can be released under anoxic conditions). P in the first two fractions, Fr.W and Fr.KCl is readily available even if the sediments are not anoxic with concentrations of nearly 10 mg-P/kg sediment dry weight.



Figure 1 A map of the Fr.W (water soluble) P concentration in DCR, the units are in mg/g dry sediment (multiply by 10³ to obtain mg/kg). Arrows indicate the inflow talweg. The general spatial distribution follows the inflow channels (from [1]).

Table 2 Average concentration for each fraction from 19 samples in DCR exposed sediments.

Fraction	Concentration (mg/kg dry sediment)		
	Avg	Min	Max
Fr.W	4.66	2.28	9.81
Fr.KCl	4.53	2.53	11.1
Fr.NaOH	174.07	53.0	460.
Fr.HCl	926.31	128.	1,340
Fr.PFD	1460.	823.	3,230
Total (by sum)	2,572.5		

This work shows that sediments in the watershed that drains into UL have high levels of readily available P. While there was a vertical trend in the data, the trend was minor.

Casbeer [2] notes that these sample locations included sites away from the talweg where the deeper samples probably mixed with soils present before the reservoir was completed, so may not be representative of historical trends. This indicates that sediments have historically had high P levels.

3.2 UL Sediments

There have been two recent studies that quantified P concentrations in UL sediments. Both studies sampled the reservoir in 2015 and 2016. Both studies used the BYU Environmental Analytics Laboratory to measure P concentrations, with details of the methods used given in the respective publications.

The first study [4, 5] collected and analyzed 85 samples for total phosphorous (TP) by digestion. Abu-Hmeidan, et al. [4] notes that they attempted fractional analysis, but the samples were allowed to dry due to a laboratory error and the resulting data were not valid. This study collected samples near the top of the sediment layer from 0 to 4 inches in depth and noted that the top foot or so of the UL sediments are continuously re-worked, resulting in a layer of sediments that are well mixed and in good contact with the water column. They further noted that this near-surface sediment region does not exhibit significant layering. They observed little vertical variation in the first foot or so of sediment.

The second study [8, 21] took 26 co-located samples at 15 sites across UL. They measured P concentrations in three interacting lake compartments: sediment, pore water, and the water column. In this study Randall, et al. [8] measured TP by digestion and performed a fractionation analysis using a different method than the earlier DCR study [1, 2]. They also performed a mineralogy analysis and combined an x-ray diffraction (XRD) analysis that identifies mineralogy with scanning electro-

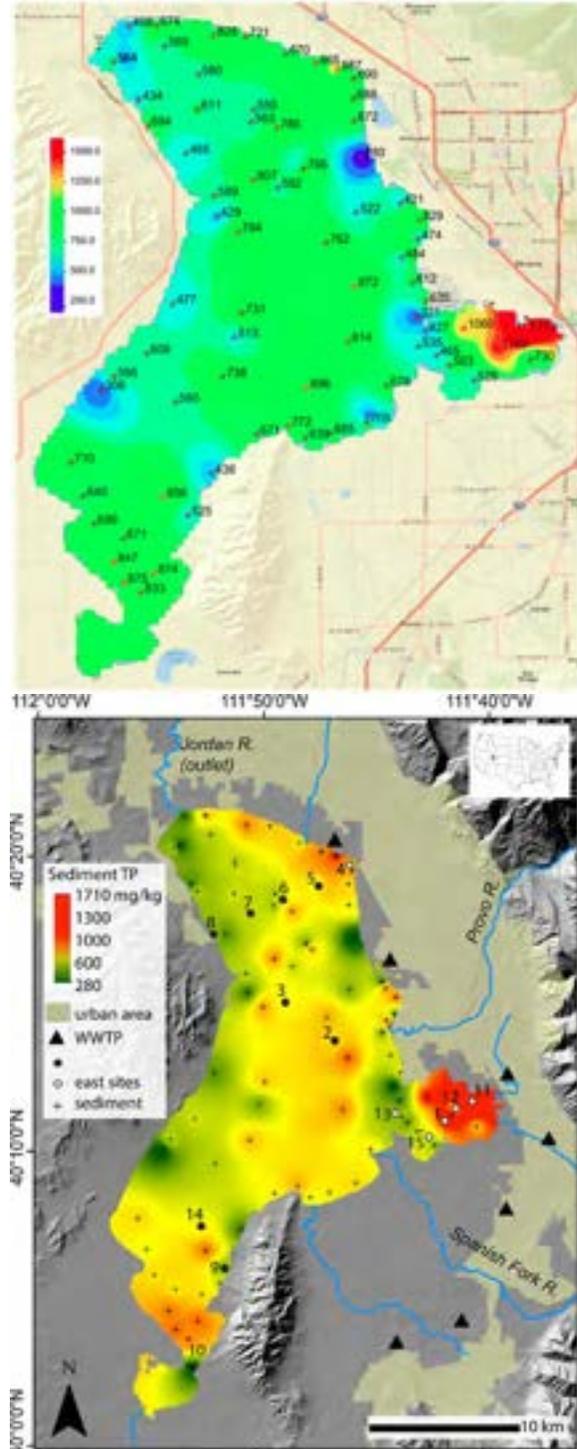


Figure 2 Two maps of TP in UL sediments, the top map is from [4] which includes 85 samples and the bottom from [8]. The second map includes samples from the first study with 26 additional samples.

showing how the P was sorbed onto the sediment. This analysis helps understand what forms the P is present.

Figure 2 shows maps of TP concentrations in the UL sediments generated by [4] (top panel) and [8] (bottom panel) who used kriging and inverse distance methods, respectively. The top panel of Figure 2 is from Abu-Hmeidan, et al. [4] and the bottom panel is from Randall, et al. [8]. The second panel, includes data from both studies [8]. There is some variation in the two maps because of slightly different data (the second panel contains mores) and the different interpolation methods, but the main difference is the color scales selected to represent the results with the second study [8] grouping most of the data into the yellow-red range.

The studies found that TP ranged from 280 – 1,710 mg-P/kg dry sediment weight. The lowest value, 280 mg-P/kg, was located on the upper east side of the reservoir and can be seen as the blue-purple “bulls-eye” in the top panel of Figure 2. This sample is located near an area of known sub-surface seeps and springs. Abu-Hmeidan, et al. [4] note that this sample was sandier with less fines than the other samples, indicating that the seeps and springs had washed the fines, and the associated P, from the sediment at that location. The location was selected because it is near a wastewater treatment plant outflow, but because of groundwater, had the lowest measured TP concentration. The samples with the highest concentrations were mostly located in shallow waters of Provo Bay. Goshen Bay, in the southern end of UL, also showed elevated concentrations, but not as high as Provo Bay.

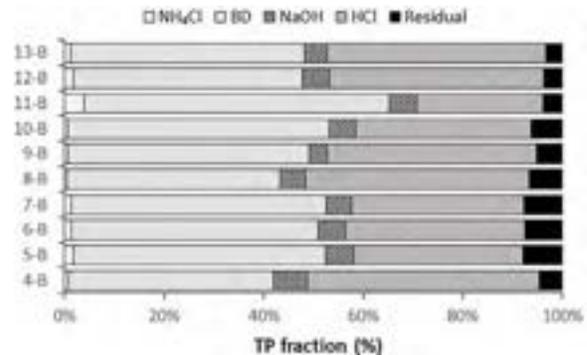


Figure 3 Fraction results for 10 sediment samples collected during May–June 2016 as % total phosphorus [8].

Randall, et al. [8] performed a fractional analysis that used different pools than those reported in the DCR study. In this UL study, the fractions measured were:

- 1) Fr.NH₄Cl (P in pore water and loosely adsorbed to surfaces);
- 2) Fr.BD (redox-sensitive P mainly bound to oxidized Fe and Mn compounds);
- 3) Fr.NaOH (P exchangeable against OH⁻ ions and P in organic matter);
- 4) Fr.HCl (P in calcium phosphate minerals and acid-soluble organic P); and
- 5) Fr.HCl.boiling (refractory organic P and non-extractable mineral P).

In this list the statement in parenthesis indicates what form of P is measured by the fraction. For this analysis, all the samples were shaken for various periods.

The first two fractions of the DCR study represent P that is included in the first fraction of this study [8]. These fractions, Fr.W and Fr.KCl (DCR) and Fr. NH₄Cl (UL) represent P readily available to the water column. The next fraction of both studies, Fr.NaOH (DCR) and Fr.BD (UL), represents P that can be released under anoxic conditions, and thus is readily released by anoxic conditions in the sediment. The Fr.NaOH fraction in the DCR analysis represents P in a different phase than the P in the Fr.NaOH fraction in the UL study because of the use of different analysis methods. The Fr.NaOH in this study is also releasable under anoxic conditions, but is more tightly bound.

Figure 3 presents the fractional analysis results from the 10 UL samples [8]. Analysis using SEM and XRF data showed that sediment P was primarily bound to oxidized Fe/Mn compounds, which was attributed to the Fr.BD fraction [8]. Randall, et al. [8] found that $49.1\% \pm 1.8\%$ of TP was associated with the Fr.BD fraction and $38.6\% \pm 2.1\%$ with the Fr.HCl fraction. As noted, the Fr.BD fraction represents P that can be released to the water column if sediments are anoxic. The study found that the percentage of P in each fraction was similar among all the samples. These percentages are very similar to the results from DCR, so we can infer that P is distributed in a similar fashion, independent of TP concentrations, or location in the watershed. This work means that if TP is measured, then the various fractional components can be estimated.

Randall, et al. [8] show that over 50% of the TP is available for release to the water column (the first two fractions in Figure 3), the majority released under anoxic conditions (Fr.BD), but up to 10% being loosely sorbed and easily released (Fr. NH_4Cl).

Randall, et al. [8] took samples of the water column and pore water co-located with the sediment samples and analyzed these water samples for TP. The water column TP concentration ranged from 0.03 to 1.74 mg/L while pore water TP concentrations were approximately an order of magnitude higher ranging from 0.30 to 10.8 mg/L [8]. Both water column and pore water concentrations were correlated with sediment samples, with high values in Provo and Goshen Bays, with lower values on the west side of UL. As the upper portion of the sediment in UL is regularly disturbed due to the shallow nature of the lake, P from the pore water is readily mixed with the water column.

Randall, et al. [8] noted *that P concentrations in the UL water column are likely governed by sediment P sources, rather than by inflows into the lake.*



Figure 4 Total phosphorus (mg-P/kg dry sediment) at each sample point. The red diamonds are on-shore soil samples. The map is not to scale [4].



Figure 5 The four sections used for analysis [4].

4. Soils

4.1 On-shore Soil Sample Results

Two studies, one primarily a sediment study [4, 5] and the other focusing on dust or particulate loads [10, 11] collected soil samples close to the shore line of UL, generally within a few 100 meters. These studies found that these near-shore soils had TP concentrations similar to those of the sediments.

Figure 4 shows the location of the soil samples, indicated by red diamonds, taken in the first study by [4] with TP concentrations in mg-P/kg dry sediment shown by the numbers next to each sample point. This maps shows that three soil samples, located near Pelican Point on the west side of the lake, have values in the highest range, from 870 – 1,114 mg-P/kg dry sediment, with the lowest sample having a concentration of 603 mg-P/kg located on the east shore.

In the second study, Reidhead [12] collected and analyzed 44 samples, from the surface to 12 inches in depth. He performed a fractional analysis on these samples in addition to measuring TP by digestion. The fractions were Fr.WS (water soluble), Fr.LB (loosely-bound), Fr.FeAl (Iron and Aluminum bound) and Fr.Ca (calcium bound). These fractions are similar to the study by Randall, et al. [8] and performed by the same laboratory.

Figure 7 shows the location of these soil samples from the second study. Reidhead [12] notes that he took samples from locations most likely to represent soils that could affect the lake. For example, samples were taken from hillsides and dunes likely to produce airborne particles). Other samples were taken in areas that did not appear to have been farmed, though some samples were taken in farmed ground as these ground represent sources close to UL. No samples were taken from corrals or pastures because of the high nutrient concentrations from animals.

The TP in these soil samples averaged 901 mg-P/kg dry soil, with a minimum concentration of 574 mg-P/kg and a maximum of 1,397 mg-P/kg. Summing the fractions, provided slightly higher values than TP measured directly, with average TP of about 1,000 mg-P/kg dry soil.

4.2 On-shore Soil Sample Analysis

To determine the difference in spatial distribution of TP in the soils and sediments, Abu-Hmeidan, et al. [4] divided the lake horizontally into four sections (Figure 5). These sections each have different dominant factors that could result in differences in phosphorous content. The lower area, Section 4



Figure 7 Location of soil samples taken by Reidhead [12].

(Figure 5) consists of the shallow southern portion of UL and does not contain any significant inflows, there are springs along the eastern shore of this section. Section 3 (Figure 5) includes Hobble Creek, a large inflow to the lake, and the outflow from the Provo wastewater treatment plant, a major P source, that both discharge into Provo Bay which is shallow and includes agriculture and feedlots on the lake shore. The eastern shore in this region, just above Provo bay, includes areas of seeps and springs and low values for TP. Section 2 (Figure 5) includes the Provo River, the main inflow to the lake, and contains the outfalls for the Orem wastewater treatment plant, a large P contributor. The eastern shore of Section 2 includes seeps and springs. Section 1 (Figure 5) contains the Jordan River outfall, the only exit point for Utah Lake, and receives the discharge from the Timpanogos Special Service district wastewater treatment plant, another large P source. Section 1 also has seeps and springs on the northwest shoreline. The on-shore, soil samples were treated as a separate section.

Table 3 and Figure 6 present the results of a statistical analysis of these samples [4]. A chi-square test was used by [4] to determine the likelihood that the data in different sections come from different populations—that is, if they are different. This analysis found are no statistically significant differences among the five groups. The chi-square analysis is a global test, and compares each section to the entire data set. Figure 6 seems to show some difference between the various sections. To explore this hypothesis, Abu-Hmeidan, et al. [4] performed a pairwise nonparametric Tukey analysis to determine if each set of pairs differed from each other. The results of this analysis show that the differences between the groups are not statistically significant. The two sections most likely to be different are the land and Group 2 samples, though the differences between these two sections, or any of the other pairs, are not statistically significant. Based on this statistical analysis Abu-Hmeidan, et al. [4] states they cannot reject the statement that the groups are the same. Or said in another way, the differences are not statistically significant. This indicates that the on-shore soil samples come from the same population distribution as the sediment samples in regards to TP.

Table 3 Statistical summary for samples from each of the four lake sections and from the on-shore soil samples [4].

Section	<i>n</i>	Avg. (ppm)	Std. Dev.
1 (Upper)	24	660	150
2 (Middle Upper)	20	631	174
3 (Middle Lower)	24	668	304
4 (Lower)	17	714	118
Land Samples	10	786	151

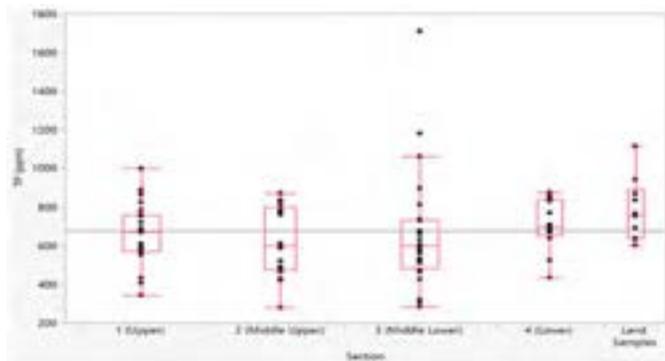


Figure 6 Box plots of total phosphorus measurements for the sediment samples in the 5 sections shown in Figure 5 and on-shore soil samples [4]. Statistically the groups are not different.

Reidhead [12] found that their soil samples have an average TP concentration of 900 mg-P/kg dry soil, with a 95% confident interval of 850 to 1,300 mg-P/kg dry soil, very similar to those from Randall, et al. [8] and Abu-Hmeidan, et al. [4].

For fractional results, Reidhead [12] found, at the 95% confident level, water soluble P is between 2.2% and 4.3%, with 0.6% to 1.1% percent loosely-bound P, and 2.5% to 4.4% aluminum and iron-bound P, with the majority of P, 90.7% to 94.2%, as calcium-bound P. These fractions are different from the sediment fractions, as the geochemistry changes when the soil dries. This analysis shows that 4.3% to 9.9% of TP in the soil is readily available to the water column.

5. Atmospheric Deposition (Dust and Rain)

5.1 Background

The finding that on-shore soil samples have similar high TP concentrations to sediment samples has implications for atmospheric deposition, including dust-borne nutrient loads UL. The soils close to UL can easily be mobilized by various processes and deposited in UL. Two recent studies have started to sample, measure, and quantify the atmospheric deposition of nutrients into UL. One study has been published as both a Master's thesis [11] and a journal article [10], the second study has only been published as a Master's thesis [12].

Atmospheric nutrient loading through wet and dry deposition is one of the least understood, pathways of nutrient transport into lakes and reservoirs, yet can be one of the most important [25-28]. UL is susceptible to atmospheric deposition due to its large surface area to volume ratio, high phosphorous levels in local soils, and proximity to Great Basin dust sources. The National Atmospheric Deposition Program (NADP), the primary atmospheric deposition monitoring program in the United States, does not measure TP in precipitation samples. As a result, few data are available to estimate atmospheric deposition of TP in Utah. However, there have been several studies performed in the Western United States with regard to nitrogen (N) deposition. In these studies, nitrate deposition in the Utah Wasatch Front produced concentrations as high as 2.0 kg/ha N [29].



Figure 8 Five dust sampling locations for both [10, 12].

Academic studies have focused on long-range dust transport and the NADP has developed a set of guidelines to locate sampling stations at positions that are not influenced by short-range atmospheric transport. For the studies reported here, the sampling stations are located near the shore of UL to measure both long-range and short-range atmospheric transport to UL. Some of these stations do not meet all the NADP guidelines for long-range atmospheric transport studies, as short-range transport is the dominant source for UL. Olsen, et al. [10] contains a table that documents the sample locations of both studies and whether the locations are in compliance with NADP guidelines. The majority of the guidelines are followed, though 3 of the sites have irrigated fields within 500m, one has a small gravel driveway within 25m, but it is not heavily trafficked, and one has a parking lot

60 m from the site. As noted, these do not meet all the NADP guidelines for long-range transport, but instead are sited to measure total nutrient loads to UL.

Goodman, et al. [30] published a study that evaluated dust along the Wasatch Front in both urban and the mountain snowpack. They found that urban and snow dust are similar to playa dust and that playas contribute the majority of dust to the urban Wasatch Front and mountain snowpack. They also note that urban aerosols contribute substantial amounts of anthropogenic trace elements and that these elements are soluble and readily available in the environment. The sampling locations used in this study follow NADP guidelines and are designed to measure regional or global dust transport, they do not account for local transport mechanism as these do not affect regional deposition. These data are representative of regional deposition to UL and point to playas as a major source of nutrients.

5.2 Atmospheric Deposition Field Methods

Olsen, et al. [10] undertook the initial study of the atmospheric deposition of P and N into UL. Reidhead [12] continued the study with some changes and improvements.

The two studies collected atmospheric deposition samples from May 2017 through November 2018 at the five locations shown in Figure 8. Olsen, et al. [10] collected and analyzed eight months of atmospheric deposition data from the five locations near Utah Lake from 5/4/2017 (May) to 12/28/2017 (December) with between 22 and 30 samples at each location. Reidhead [12] continued that study, using the same sampling locations, from 4/6/2018 (April) to 11/17/2018 (November) with 31 samples at each location. Where possible, both studies sampled weekly, though occasionally longer time periods occurred. In addition to the sampling locations around UL, Reidhead [12] collected samples at two locations on the east shore of the Great Salt Lake near Farmington Bay.



Figure 9 Example of table with wet and dry buckets [12].

They selected the sites to distribute data collection around the lake as evenly as possible to estimate atmospheric deposition. They also considered site access. They designed these samplers to operate similar to the Aerochem Metrics (ACM) bucket collectors used by the NADP.

The samplers (Figure 9) consist of two 2-gallon buckets placed on a steel table. The wet bucket is placed as is on the table to collect rain water from the storms affecting Utah Lake. The dry bucket is filled with three liters of water (four during the hottest months of the year to reduce sample loss due to evaporation) to retain soil and dust particles deposited when no precipitation is present. The table has an automated rain sensor that closes the dry bucket and opens the wet bucket when it rains. The sample buckets remain, collecting dust and soil from the atmosphere for the sampling period. For collection, a 500 mL representative sample is taken from the dry deposition bucket, and the wet sample is

combined with DI water to add up to a complete 500 mL sample and tested at the laboratory. The buckets were replaced and acid-cleaned after each sample collection.

The UL samples sites were:

- Orem Waste Water Treatment Plant (40.27595, -111.7372) 40 ft. south of a shed. There is a parking lot just north and east of the shed. There is an old, unused asphalt road 20 ft. west of the sampler, and the sampler rests on the northern end of a vacant field with little plant growth. The entire site is located within a light industrial area in Orem.
- Lakeshore/Lincoln Point (40.11291, -111.78893) is on the southern edge of a property with light farm traffic. On August 18, 2018 it was relocated to more than 100 ft. further away from the light traffic source. To the south is a large open field with light brush, with a wetland just to the east.
- Mosida (40.07712, -111.92574) is surrounded by a large, scattered brush field. In 2017 and until August 18, 2018, it was located approximately 140 ft. from the nearest road. At that time, it was moved to a more distant location some 500 ft. from that road, and over 1100 ft. from center pivot sprinkler irrigated fields.
- Saratoga Springs (40.28234, -111.8706) is located south of the city at the edge of a property (abandoned since late summer of 2018), approximately 500 ft. from the highway. A small dirt road leading to the property is rarely used. There are small structures just over 50 ft. away to the south, east and north. A large unused field borders the site on the south and west.
- Utah Lake Pump Station (40.35931, -111.8963) is perhaps the least representative site for Utah Lake for “global” AD due to the thick phragmites just under 10 yards away on two sides. Small particles from the plants occasionally enter the samples, though the plants likely mitigate construction dust from local housing developments. It is the closest site to the actual lake.

Sampling sites near Farmington Bay for comparison:

- Central Davis Water Reclamation Facility (40.9991, -111.95135) is located on a large berm of soil lakeside of the water treatment facility in Farmington. It is bordered by well-watered fields to the east and wetlands to the west. It stands next to a dirt road that occasionally receives light traffic.
- Browns Island (40.8466, -112.06777) is located east of Antelope Island on the southeast corner of the Great Salt Lake. It is on a slight berm above the wetlands just outside of the West Crystal Unit of Farmington Bay. Though the sampler at CD did not collect from May to July and the sampler at BI did not collect until September, those at Utah Lake were fairly consistent.

Though a portion of the deposition is global (from sources a great distance from the samplers themselves), the sites are expected to have different loading rates and represent local transport because of local wind patterns, climate, and nutrient sources [12].

5.3 Atmospheric Deposition Sample Results

Both studies used, the BYU Environmental Laboratory and followed procedures recommended by NAPD [31]. TP was analyzed using microwave digestion followed by inductively coupled plasma (ICP) spectroscopy. NH_4^+ was analyzed with the salicylate method and NO_3^- with the cadmium reduction method. Olsen, et al. [10] sent some duplicate samples to a separate laboratory for quality control. All analysis was done by

certified analytical laboratories following standard methods. Detailed procedures are presented in [10-12].

Table 4 Atmospheric deposition for total phosphorus (TP) results from [10]

Site	No. of Data	Mean TP Concentrations (mg/L)		Rain cm(in)/Week	Total TP Load Rate (mg m ⁻² day ⁻¹)		
		Wet	Dry		Mean	S.D.	Skew
		Lake Shore	41	0.68	0.38	0.64(0.25)	1.33
Mosida	38	0.22	1.10	0.30(0.12)	2.77	5.63	2.55
Saratoga Springs	44	0.60	5.15	0.43(0.17)	31.38	88.73	2.14
Pump Station	38	0.59	0.85	0.41(0.16)	3.78	20.14	4.68
Orem WWTP ¹	27	1.62	0.39	0.28(0.11)	1.26	2.65	3.33
Average	38	0.74	1.57	0.41(0.16)	8.10	23.82	2.70

Table 4 presents the TP results as mean TP concentrations at each sample site separated by deposition type, the weekly average precipitation, and the mean TP load rate at each sample site with standard deviation, and skewness values for the loading data. The average loading across all the sites was 8.10 mg of TP m⁻² day⁻¹, with mean values for the individual sites ranging from 1.26 to 31.38 mg of TP m⁻² day⁻¹ at Orem WWTP and Saratoga Springs, respectively. Standard deviation of the loading rates were high, larger than the mean values. The skewness values are all greater than 0, with an average value of 2.7, which indicates that the samples are skewed right. This means that the bulk of the data are in the lower range with a few larger values (i.e., long tail towards the higher values). The coefficient of variation (i.e., mean divided by standard deviation), calculated using all the data, was 0.34.

Table 5 Atmospheric deposition of dissolved inorganic nitrogen (DIN) results from [10]

Site	No. of Data	Mean DIN Concentrations (mg/L)		Rain cm(in)/Week	Total DIN Load Rate (mg m ⁻² day ⁻¹)		
		Wet	Dry		Mean	S.D.	Skew
		Lake Shore	41	4.30	1.15	0.64(0.25)	4.09
Mosida	38	2.29	1.50	0.30(0.12)	4.17	4.74	1.21
Saratoga Springs	44	4.86	6.00	0.43(0.17)	36.06	124.62	3.31
Pump Station	38	4.29	0.38	0.41(0.16)	1.59	2.33	2.31
Orem WWTP	27	7.55	1.33	0.28(0.11)	5.23	4.60	3.04
Average	38	4.66	2.07	0.41(0.16)	10.23	28.07	2.07

Table 5 presents the dissolved inorganic nitrogen (DIN) results as mean concentrations at each sample site separated by deposition type, the weekly average precipitation, and the mean DIN load rate at each sample site with standard deviation, and skewness values for the loading data. The average DIN loading of all sites is 10.23 mg of N m⁻² day⁻¹. The mean values for the individual sites range from 1.59 to 36.06 mg of N m⁻² day⁻¹ at the Pump Station and Saratoga Springs, respectively. As with the TP data, the standard deviations are high and skewness values are positive—indicating that the data have long tails to the

right (large outliers). The coefficient of variation, calculated for all the data, was 0.36, very similar to that computed for the TP data.

For the first study [10, 11], Tables 4 and 5 show that TP and DIN deposition rates at Saratoga Springs were significantly higher than at the other sites. The TP deposition rate differences between Saratoga Springs and the other sites all have p-values < 0.005 (0.5%), meaning the results are statistically different. Olsen, et al. [10] attributes this difference mainly to local nutrient sources in the area and regional weather patterns. Near the Saratoga Springs sample site there are large areas of bare earth that have high concentrations of TP. This site is also relatively close to areas with both agricultural and excavation activities at a gravel pit. In addition to these issues, storms tend to move across Utah Lake from desert areas with high dust to the northwest and west, crossing this site before reaching the lake. Other sample sites are separated from the western Utah desert areas by agricultural or urban areas that do not generate as much dust.

The second study [12], did not present mean or average results over the test period, but rather presented weekly values, in general the values are similar. These data are available in [12] along with a detailed statistical analysis of the data. The detailed data are not presented here.

5.4 Atmospheric Deposition Rates

Both studies provided order of magnitude estimates for total atmospheric deposition to UL, though they differed in the methods they used for spatial interpolation of the sample results.

Olsen, et al. [10] used kriging, with a short range of 1,000 m, to interpolate deposition rates across Utah Lake to estimate a total deposition loading over the eight month sampling period. They used the same process for both TP and DIN. Because of the large temporal variations in deposition rates, they did not estimate annual deposition, but only estimated total deposition during the sampling period using a total lake surface area of 354 km² (88,000 acres), which was the average lake area during their sampling period.

Olsen, et al. [10] assumed that deposition near the shoreline is most likely higher than deposition in the interior of the lake as near the shore deposition from local soil dust is an

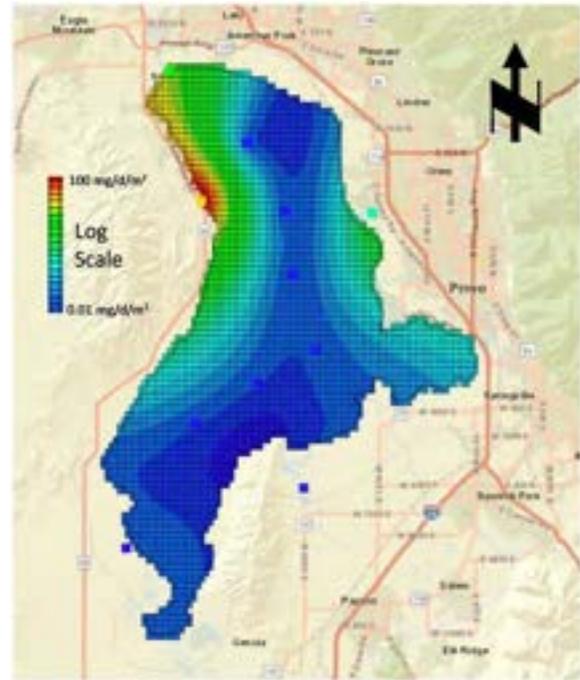


Figure 10 Example map from Olsen, et al. [10] showing the spatial deposition rates of TP for one week in May. The map shows that the pseudo sample sites in the center of the lake, which represent regional background rates, strongly influence the distribution. The colors are on a logarithmic scale with red approximately 100 mg m⁻² day⁻¹ and blue about 0.01 mg m⁻² day⁻¹ spanning four orders of magnitude. This map shows that the influence of high values at shoreline stations do not reach significantly into the lake because of the small variogram range used for kriging.

important component. In order to conservatively estimate total deposition rates, they created six “dummy” sample points along the center of the lake and assigned background deposition values for TP and DIN of $0.019 \text{ mg TP m}^{-2} \text{ week}^{-1}$ [32] and $0.112 \text{ mg DIN m}^{-2} \text{ week}^{-1}$ [31], respectively. They note that these values are at least two orders of magnitude lower than the local deposition rates measured at the lake shore. Figure 10 shows that the influence of high values at shoreline stations do not reach significantly into the lake because of the small variogram range used for kriging. The majority of the lake received loads close to background levels (*i.e.*, the blue colors in Figure 10).

Olsen, et al. [10] estimated two different total deposition values: one that included all samples collected during the study and one that excluded samples with any visible particles in the collection bucket. They assert that these two estimates provide bounding values for the actual deposition that occurred during the sampling period as the lower estimate bounds the deposition rates because it excludes all loads from dust particles, insect bodies, and other legitimate sources of nutrients and the upper estimate bounds the deposition rates because it includes some contamination sources that may be local and not significantly extend into the lake itself. Both estimates have additional conservatism because of the assumption of the short kriging range and the inclusion of multiple background (or dummy) points in the middle of the lake.

Olsen, et al. [10] estimated that the total deposition loading of TP and DIN were 350 Mg and 460 Mg, respectively, for the high estimate over the 8-month period and total deposition loading of TP and DIN were 8 Mg and 46 Mg, respectively, for the 8-month period for the low estimate.

Reidhead [12] used a different spatial interpolation method assuming a linear fall-off across the lake from all the sample sites, details are provided in [12]. For a 7.5 month period, they estimated 58.9 tons (53.4 Mg) soluble reactive P, 162 tons (147 Mg) TP, 123.6 tons (112 Mg) nitrate-nitrogen, and 382.5 tons (347 Mg) ammonium-nitrogen.

Reidhead [12] estimates are between the upper and lower bounds estimated by Olsen, et al. [10].

4. Discussion

4.1 Sediments

In their paper, Randall, et al. [8], attribute the higher sediment concentrations on the east side of UL to proximity to waste water treatment plants. This does not explain the elevated values in Goshen Bay. In the other study, Abu-Hmeidan, et al. [4] attribute the higher concentrations to shallow water bays where the sediment has high organic carbon content from decaying plant matter. They attribute the lowest values to regions of UL to areas with groundwater seeps and springs that wash the P from the sediment. The Randall *et al.* [8] paper notes that higher organic carbon content is associated with higher TP concentrations.

Figure 2 shows the results of both these studies, the figure does not support the conjecture that higher concentrations are found on the east shore, the figure clearly shows higher concentrations are associated with shallow water and swampy areas, the most prevalent are Provo and Goshen Bays. Figure 2 does show that the north-east corner has higher

values in the sediments, this area of the lake is shallow with swampy areas, and is the remnant of the delta for the drainage from American Fork Canyon and the Dry Creek area. It is also near the outfall of the Timpanogos Wastewater Treatment plant, but high values continue a significant distance south of the plant and seem to be more associated with shallow, swampy areas.

4.2 On-shore Soils

Dr. Steven Nelson of BYU expressed concerns with the paper from Abu-Hmeidan, et al. [4]. He noted that the paper had serious flaws regarding analysis of geologic historic Lake Bonneville sediments. These concerns were a result of a miss-use of terms, in that paper, the near-shore soils were referred to as “historical lake sediments” when the paper was referring to Utah Lake near-shore soils. Dr. Nelson felt that the description of these soils as historic sediments was incorrect and that a number of the conclusions in the paper were not valid as historic sediments (e.g., Lake Bonneville sediments) do not fit the descriptions. The descriptions in the paper were based on measured data from near-lake soils, the paper did not use any geologic inferences, other than to state that there are known geologic sources of phosphorous in the UL watershed.

In this document, I have been careful to call these samples “near shore soils” rather than “historic sediments”. Regardless of the designation, measurements of the soils near UL show that they have high total phosphorous levels, nearly 1,000 mg-P/kg dry soil. These data are not in dispute. As the conclusions in the papers are based on measured data, these conclusions are also valid.

4.3 Atmospheric Deposition

There have been several criticisms of the atmospheric deposition collections specifically that the collection stations do not follow the guidelines provided by the NADP. These UL sample locations were selected to quantify actual loads to the lake, both from regional and local sources. The NADP guidelines are intended to provide locations where regional or global transport can be measured and not impacted by local transport processes. In the studies reviewed here, the goal of the measurements are to quantify atmospheric deposition loadings to Utah Lake, which includes and is mostly dominated by local sources. To accurately measure the amount of nutrients deposited to the Lake, these local sources must be included in the measurements and so the sample locations cannot strictly follow NADP guidelines. The journal papers, Master’s Theses, and other reports clearly acknowledge this variance from the NADP guidelines. As noted, to help readers evaluate the locations of these measurements Olsen, et al. [10] includes a table that lists the discrepancies.

Recently, Dr. David Gay, formally associated with the NADP, reviewed the atmospheric deposition studies and also reviewed the comments made by Dr. Brahney of Utah State University. He found that Dr. Brahney’s comments were correct in the context of regional or global transport, but that he necessarily did not disagree with placement of the sampling locations for the atmospheric deposition studies I reviewed in this report.

Dr. Gay specifically commented on the placement of the sample measurements. He notes, that the sampling locations are not in compliance with NADP guidelines for regional sampling, but that in this context

“...for this particular question of flux to Utah Lake, you want to capture all of the important and significant fluxes that are contributing to deposition flux to the lake. Therefore, the contribution of local sources is important to include in the flux estimation.”

He goes on to note:

“...if all of the samplers are clearly dominated by local sources that only contribute to flux in certain parts of the lake, leaving low flux rates unaccounted for, then the total flux estimates can be overestimated..”

Dr. Gay cautions that since site selection is very influential in the estimated loads, that sites should be carefully selected to represent actual loads to the lake.

Dr. Gay provides a very detailed analysis and review of the atmospheric deposition studies that addresses many issues and provides guidance. A review of Dr. Gay's comments are not included as part of this review study.

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Appendix – Journal and Conference Papers

The appendix contains a list of the major journal papers and thesis that are the basis of this report. Where possible, a complete copy of the article is included, for some documents, such as the Theses, the entire document is not included because of length or copyright restrictions, for these documents the published abstract is included. References for all the documents are provided.

Deer Creek Studies

Included Documents

Casbeer et al. 2018

Casbeer, W.; Williams, G.; Borup, M. Phosphorus Distribution in Delta Sediments: A Unique Data Set from Deer Creek Reservoir. *Hydrology* **2018**, *5*, 58.
[Document included]

Documents Not Included

Messer et al., 1984

Messer, J.J.; Hardy, T.B.; Ihnat, J.M. *Sediment-Phosphorus Relationships in Deer Creek Reservoir*; Utah Water Research Laboratory: Utah State University, 1984. [3]

Abstract

Laboratory analyses of sediment samples recovered from Deer Creek Reservoir showed the sediments to have a moderate potential for serving as a source of phosphorus (P) for the over-lying water column under anaerobic conditions. This source could account for the occurrence or exacerbation of blue-green algal blooms in the upper end of the reservoir when the sediment surface becomes anaerobic late in the summer. It could also delay the re-sponse of the reservoir to programs that decrease external P loading, if the upper end of the reservoir still has large enough algal blooms to render the sediment-water interface anaerobic. The redox cycle of Fe is closely associated with P release in the reservoir sedi-ments, and aerobic P release rates are probably insignificant in the phosphorus budget of the reservoir. Sediment profiles indicate that the trophic state of the lake has probably not changed significantly since the reservoir was first filled. The profiles do indicate, however, that much of the P input to the lake may be in the relatively unreactive apatite-P fraction, and attempts at reducing P loading from external sources should take into account the degree of availability of the P thus removed.

Casbeer, W.C. 2009

Casbeer, W.C. Phosphorus fractionation and distribution across delta of Deer Creek Reservoir. Masters' Thesis, Brigham Young University, 2009.

Abstract

Eutrophication of reservoir systems, which causes deterioration of water quality through increased algal growth, is detrimental to our sustainable water supply and additionally impairs other beneficial reservoir uses. Limiting the amount of phosphorus (P) entering the system has been the key management tool for this problem, as P is the main limiting nutrient for plant and algal growth. These efforts have focused on controlling input of P from point sources, such as effluents from wastewater treatment plants, dairies, and industrial factories. Even in systems (such as reservoirs) with significantly reduced external P loading, however, there has been continued eutrophication and slower than

expected recovery of reservoirs in water quality restoration projects. Other nutrient sources have been studied to explain this phenomenon. The continual eutrophication has been potentially attributed to availability of nutrients from deposited sediments. This is referred to as nutrient recycling, as nutrients previously trapped within sediments may become available within the water column. Deer Creek Reservoir (DCR), a significant water supply in Utah, has had greatly improved water quality after reduction of external P loading. However, there are still large algal blooms at times as well as other water quality issues without clearly attributable causes. Part of the explanation might lie within the deposited sediments, which are present both on the sediment delta and within the reservoir. This thesis provides data that can help researchers understand what role sediment has in the continuation of water quality problems at DCR. Sediment samples were taken across the delta to define both the spatial extent and distribution of P and chemical form, or 'pool', of the P. The pools can be used to estimate the ability of the sediment-bound P to move into the water column under various conditions. Results reported here indicate that significant amounts of P are found within these sediments, though not all of it can easily become available for algal growth. We characterized P distribution by taking 91 samples on 6 transects across the exposed delta. Transects were separated by 200 m and samples were taken every 100 m along the transects. The samples were all analyzed for water soluble P content, and 19 samples were additionally characterized for KCl-, NaOH-, HCl-, and organic (by digestion) P fractions. Total P was determined for these as well by summation. The data showed that water soluble P ranged from 2.28E-03 and 9.81E-03 mg P g⁻¹ dry sediment and showed a decreasing trend along the reservoir. KCl-P ranged from 2.53E-03 and 1.10E-02, NaOH-P from 5.30E-02 to 4.60E-01, HCl-P from 1.28E-01 and 1.34E+00, and organic (residual) P from 8.23E-01 to 3.23E+00 mg g⁻¹.

Utah Lake Sediment and Soils

Included Documents

Abu-Hmeidan et al., 2018

Abu-Hmeidan, H.; Williams, G.; Miller, A. Characterizing total phosphorus in current and geologic Utah lake sediments: Implications for water quality management issues. *Hydrology* **2018**, *5*, 8.

Randall et al., 2019

Randall, M.C.; Carling, G.T.; Dastrup, D.B.; Miller, T.; Nelson, S.T.; Rey, K.A.; Hansen, N.C.; Bickmore, B.R.; Aanderud, Z.T. Sediment potentially controls in-lake phosphorus cycling and harmful cyanobacteria in shallow, eutrophic Utah Lake. *PLoS one* **2019**, *14*, e0212238.

Documents Not Included

Abu-Hmeidan, Y.H. 2017

Abu-Hmeidan, Y., Hani. Characterizing Current and Geologic Phosphorus in Utah Lake Sediment Using Field Samples, Laboratory Methods, and Statistical Analysis: Implications for Water Quality Issues. Brigham Young University, Masters Thesis, 2017.

Abstract

Phosphorus is an essential nutrient for aquatic life forms and plays a major role in the algae blooms that occur in lakes and reservoirs. It is considered a primary limiting nutrient of phytoplankton growth in streams, lakes, and reservoirs. Excess amounts of phosphorus may

cause excess growth and biomass of algae. If phosphorus is available in excess, often from sewage and industrial discharges, the high levels in a lake or reservoir can lead to eutrophication.

Utah Lake is a shallow, basin-bottom lake in a semi-arid climate with sediments that are thousands of feet thick. Starting 165 years ago, humans have been discharging wastewater into Utah Lake, which in our day has raised serious questions on how the state can mitigate the negative effects of the external nutrient loading. Even though Utah Lake receives a significant amount of anthropogenic phosphorous, there are high levels of phosphorous in geologic deposits in the area, providing a long-term natural source. This study intends to provide data on the current distribution of phosphorous in lake sediments, potential for that phosphorous to be released into the water column affecting phytoplankton growth, and how historic lake sediment phosphorous levels compare to the levels in current sediments.

Sediments play an important role in the overall metabolism of shallow lakes. They supply the water column with phosphorus and must be considered as they serve as a sink and source. More than 50 branches of surface flow discharge into Utah Lake, 15 of which are major. Based on previous data, a positive retention of phosphorus from these branches occurs in the lake, of which the sediment plays a role. Phosphorus release from sediment occurs under very complicated processes under many different conditions. Some main influential factors include the iron and calcium content, redox potential, microbial processes, turbidity, sediment resuspension, temperature, and pH.

In this study, I analyzed 85 sediment samples sampled across Utah Lake for total phosphorus. I created Geospatial maps to show the phosphorous distribution. The data showed an average phosphorus level of 666 ppm and varied in distribution throughout the lake, though the majority of the lake had levels in the 600 to 800 ppm range. There were a few samples, which had lower total phosphorus levels, in the 200 to 300 ppm range. Based on the map, I found that these lower values were in locations representing potential springs. I hypothesize that this underground water source leached some of the phosphorous from the sediments in these areas. I found that total phosphorus concentrations in current lake sediment are quite similar to phosphorus levels in historic lake sediments levels. I also performed laboratory experiments to characterize sediment-water interactions and estimate the amount of phosphorus that could be released to the water column.

Randall, M.C. 2017

Randall, M.C. Characterizing the Fate and Mobility of Phosphorus in Utah Lake Sediments. Brigham Young University, Masters Thesis, 2017.

Abstract

An increasing number of lakes worldwide are impacted by eutrophication and harmful algal blooms due to nutrient inputs. Utah Lake is a unique eutrophic freshwater lake that is naturally shallow, turbid, and alkaline with high dissolved oxygen levels. Recently, the Utah Division of Water Quality has proposed a new limitation of phosphorus (P) loading to Utah Lake from wastewater treatment plants in an effort to mitigate eutrophication. However, reducing external P loads may not lead to immediate improvements in water quality due to the legacy pool of nutrients in lake sediments. The purpose of this study was to characterize the fate and mobility of P in Utah Lake sediments to better understand P cycling in this unique system. We analyzed P speciation, mineralogy, and binding capacity in lake sediment samples collected from 15

locations across Utah Lake. P concentrations in sediment ranged from 306 to 1894 ppm, with highest concentrations in Provo Bay near the major metropolitan area. Sequential leach tests indicate that ~25-50% of P is associated with Ca ($\text{CaCO}_3/\text{Ca}_{10}(\text{PO}_4)_6(\text{OH},\text{F},\text{Cl})_2 \approx \text{P}$) and 40-60% is associated with Fe ($\text{Fe}(\text{OOH}) \approx \text{P}$). Ca-associated P was confirmed by SEM images, which showed the highest P concentrations correlating with Ca (carbonate minerals/apatite). The Ca-associated P fraction is likely immobile, but the Fe-bound P is potentially bioavailable under changing redox conditions. Batch sorption results indicate that lake sediments have a high capacity to absorb and remove P from the water column, with an average uptake of 70-96% removal over the range of 1-10 mg/L P. Mineral precipitation and sorption to bottom sediments is an efficient removal mechanism of P in Utah Lake, but a significant portion of P may be temporarily available for resuspension and cycling in surface waters. Mitigating lake eutrophication is a complex problem that goes beyond decreasing external nutrient loads to the water body and requires a better understanding in-lake P cycling.

Atmospheric Deposition

Included Documents

Olsen et al., 2018

Olsen, J.; Williams, G.; Miller, A.; Merritt, L. Measuring and calculating current atmospheric phosphorous and nitrogen loadings to Utah Lake using field samples and geostatistical analysis. *Hydrology* **2018**, *5*, 45.

Goodman, M.M. et al., 2019

Goodman, M.M. *et al.*, "Trace element chemistry of atmospheric deposition along the Wasatch Front (Utah, USA) reflects regional playa dust and local urban aerosols," *Chemical Geology*, vol. 530, p. 119317, 2019.

Documents Not Included

Olsen, J.M. 2018

Olsen, J.M. Measuring and Calculating Current Atmospheric Phosphorous and Nitrogen Loadings on Utah Lake Using Field Samples, Laboratory Methods, and Statistical Analysis: Implication for Water Quality Issues. Brigham Young University, Masters Thesis, 2018.

Abstract

Atmospheric nutrient loading and transport through precipitation and dry deposition is one of the least understood yet one of the most important pathways of nutrient transport into many lakes. These nutrients, phosphorus and nitrogen, are essential for aquatic life and often play major roles in algae blooms that occur in lakes and reservoirs. Often heavy algal growth intensifies a variety of water quality problems. Utah Lake may be even more susceptible to atmospheric deposition due to its large surface area to volume ratio and proximity to Great Basin dust sources. In this study, eight months of atmospheric deposition data were collected and analyzed from five locations near Utah Lake. Geospatial maps were created to show the temporal distribution of phosphorus and nitrogen. Evaluation of the atmospheric deposition results indicate that between 8 to 350 tons of total phosphorus and 46 to 460 tons of dissolved inorganic nitrogen were deposited onto the surface of Utah Lake over an eight-month period. Both estimates were based on assuming that the deposition decreased exponentially from the sampling station to the middle of the lake. The large difference results from using only samples with no visible particles or

insects present to give the low estimate and all samples to give the high estimate. These nutrient loading values are very significant in that it has been estimated that only about 17 tons year⁻¹ of phosphorus and about 200 tons year⁻¹ of nitrogen are needed to support a eutrophic level of algal growth in Utah Lake. Atmospheric deposition was found to be a major contributor in providing a eutrophic nutrient load to Utah Lake. Further, it is likely that the actual deposition loading is much higher than 8 tons per 8 months thus indicating that deposition alone adds a eutrophic phosphorus loading to the lake. Since conditions are similar in much of the Great Basin and other areas of Western United States, this seems to be a very significant finding relative to nutrient evaluation and feasible management scenarios. The results also indicate that one might expect to see more cyanobacteria blooms (Harmful Algal Blooms) in shallow ponds in this area if atmospheric deposition is the main source of nutrients, since N to P ratios are low and thus more situations arise where a shortage of ionic nitrogen favors these nitrogen-fixing cyanobacteria.

Reidhead, J.G. 2019

Reidhead, J.G. Significance of the Rates of Atmospheric Deposition around Utah Lake and Phosphorus-Fractionation of Local Soils. Brigham Young University, Masters Thesis, 2019.

Abstract

Eutrophic Utah Lake receives a large nutrient load from a variety of sources, including treated wastewater discharges, runoff and tributaries, recycling from bottom sediments and Atmospheric Deposition (AD). AD was the focus of this study and was comprised of two complementary parts. First was a study of nitrogen and phosphorus depositions from the atmosphere, and second was a study of phosphorus as contained in soils near Utah Lake via fractionation methods. The soil samples were found to contain approximately 1,000 mg-P/kg soil for total phosphorus (TP). A separate phosphorus (P) fractionation gave slightly higher values, excluding the residual P, we are 95% confident that one gram of sample soil contains between 2.2 and 4.3percent water soluble P, 0.6 to 1.1 percent loosely-bound P, 2.5 to 4.4 percent aluminum andiron-bound P, and 90.7 to 94.2 percent calcium-bound P.AD results indicate that during the period from April 1 to Nov 17, 2018, Utah Lake received approximately 58 tons of soluble reactive P, 153 tons of TP, 118 tons of nitrogen (N)from nitrate, and 387 tons of N from ammonium via AD. Nutrient quantities from AD are very large compared to the 17 ton/yr of P needed for a eutrophic loading to the lake. Because of the very large overall nutrient loading to Utah Lake, it is likely that some other limiting growth factors are controlling algal growth.

Article

Phosphorus Distribution in Delta Sediments: A Unique Data Set from Deer Creek Reservoir

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Abstract: Recently, Deer Creek Reservoir (DCR) underwent a large drawdown to support dam reconstruction. This event exposed sediments inundated by the reservoir, since dam completion in the early 1940s. This event allowed us to take sediment data samples and evaluate them for phosphorous (P) content. It is difficult for normal reservoir sediment studies to have sediment samples at high spatial resolution because of access. During the drawdown, we collected 91 samples on a grid 100 m in one direction and 200 m in the other. This grid defined an area of approximately 750,000 m² (185 acre). We took both surface samples, and at some sites, vertical samples. We determined water soluble P for all the samples, and P in four other reservoirs or fractions for 19 samples. Results showed water soluble P in the range of 2.28×10^{-3} to 9.81×10^{-3} , KCl-P from 2.53×10^{-3} to 1.10×10^{-2} , NaOH-P from 5.30×10^{-2} to 4.60×10^{-1} , HCl-P from 1.28×10^{-1} to 1.34, and residual (mostly organic) P from 8.23×10^{-1} to 3.23 mg/g. We provide this data set to the community to support and encourage research in this area. We hope this data set will be used and analyzed to support other research efforts.

Keywords: sediments; phosphorous; available nutrients; water quality; reservoir management

1. Introduction

Detrimental reservoir eutrophication often caused by nutrients affects water use [1,2]. Reservoir water quality managers often limit nutrients, such as phosphorus (P), to the reservoir. This is difficult, as P comes from many sources, including dust particulate matter [1,3–8]. Often, these approaches have minimal impact to water quality [9]. One potential reason is sediments that can release P to the water column [9–14] reducing the effectiveness of other measures [15]. Reservoirs are susceptible to nutrient recycling because of sediment delta re-cutting and residence times [16].

Most reservoirs do not have any, or at best, have limited data on nutrients in the sediments [17]. Even though P is a major driver of eutrophication, the literature has minimal information on sediment P distributions [18], likely because of the difficulty in collecting these data [1,19]. For most reservoirs, the sediment delta is of higher concern because of the annual drawdown and sediment re-cutting cycle [20,21]. Fabre [22] showed that sediments in the reservoir delta can be resuspended by adding nutrients to the water column. Most reported sediment P studies use a few core samples or laboratory experiments [12,23–26], and cannot evaluate spatial distribution patterns or impacts.

Deer Creek Reservoir (DCR) is an important water supply reservoir in northern Utah. The reservoir was completed in the early 1940s [27]. Owing to DCR's importance as a water supply reservoir, researchers and government agencies have performed several water quality studies [28–32].

The DCR drawdown for dam reconstruction allowed access to approximately 5 km of the sediment delta that had previously been inundated. This exposed area is shown in Figure 1, and it allowed access to most of the sediment delta. The drawdown, shown in Figure 1, exposed over 5 km of sediments.

allowed access to most of the sediment delta. The drawdown, shown in Figure 1, exposed over 5 km of sediments. This allowed for easier sample collection on a high-resolution grid, over a large portion of the sediment delta. There has been another study that evaluated the P distribution in DCR sediments, but it only used 4 locations [33,34]. The previous study did not evaluate spatial distributions, even though the data indicated that there were variations [34]. A historical DCR drawdown in 2001, recut the sediment delta and caused taste and odor problems because of the resulting geosmin release [32]. This indicated that delta re-cutting impacts DCR water quality.

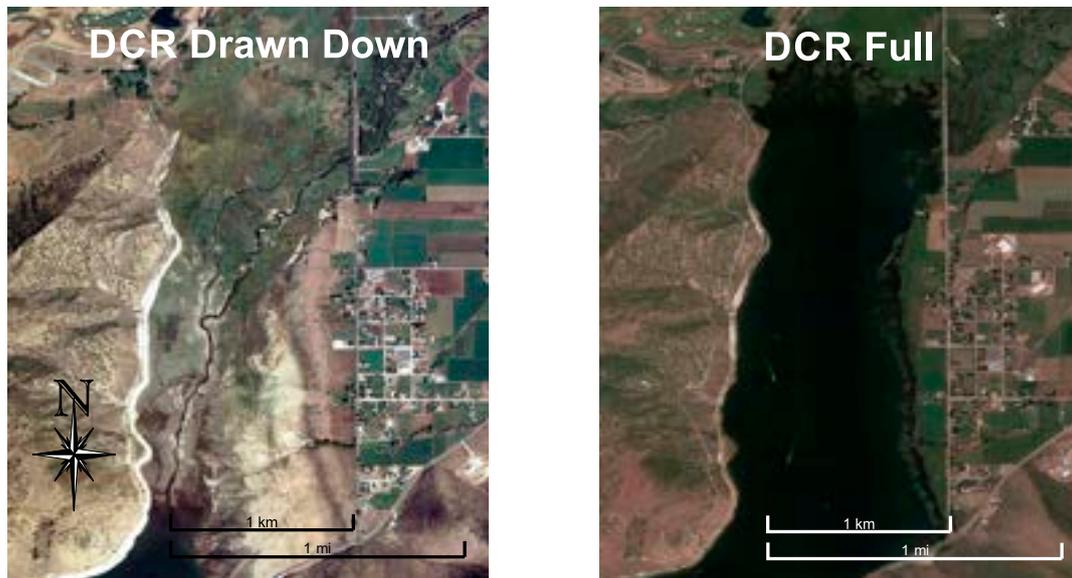


Figure 1. Deer Creek reservoir during the construction drawdown (left panel) and full (right panel). The draw down was extensive, exposing over a 3 km of sediments.

Our work provides field data that is useful for understanding how sediments affect nutrient dynamics within the DCR. The DCR is a large reservoir with a large sediment delta. The data along with their spatial distribution we expect this dataset will help researchers generalize geosmin release to other reservoirs with similar problems, because this data set provides detailed characterization of the distribution of a highly spatially distributed nutrient. This data set also helps address an important area of concern in reservoirs.

We provided this data set, with detailed method descriptions, to the community as a resource to support research into this important topic. Better understanding of the impacts of internal P cycling, and the role played by sediment resuspension and recycling can help reservoir and water quality managers to better address nutrient issues. This work addresses resuspension by providing a high-resolution data set, which includes fractionation results. Understanding P fractionation is important, as much of the P in sediments is not available to the water column for resuspension. We compare results from samples collected on exposed sediment delta (delta exposed for about a year) to results from a 25-year-old, long-term, submerged sediment delta (delta submerged for 25 years) to provide very basic insight into potential differences. Both sets of samples were collected in the region of the reservoir.

2. Collection and Analysis

2.1. Sample Collection

We collected sediment samples at the locations shown in Figure 2. Figure 1 shows the delta during the drawdown; Figure 2 shows the sample locations with the reservoir full. The samples were taken at 100 and 200 m spacing for distances along and between the transects, respectively. We took two different types of sample, i.e., 57 surface (S) samples (white squares in Figure 1) and 15 samples that included both surface and vertical samples (black crosses in white squares in Figure 1). We did

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 included both surface and vertical samples (black crosses in white squares in Figure 1). We did not sample locations 1 through 6 or location 60 because they were beyond the delta limits. For 11 of the 15 vertical samples, we sampled at the surface, at 6 inches and at 1 foot. At four of the 15 vertical locations (points 12, 20, 30, and 42), we took an additional sample at 2 feet.

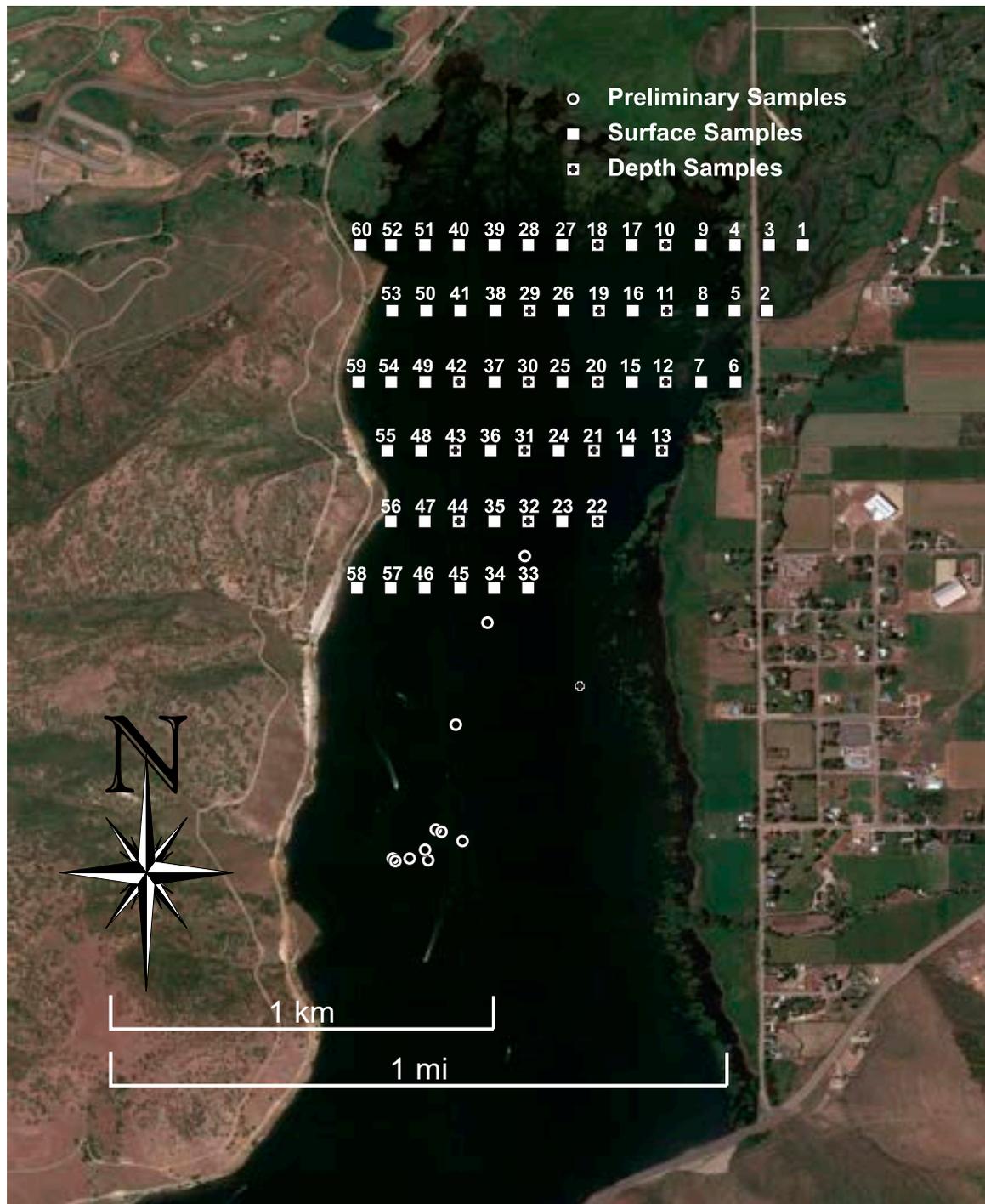


Figure 2. Sediment sample locations showing preliminary samples (circles), surface samples (squares), and samples at multiple depths (squares with cross).

We made field trips over a 4 week summer period in 2008. We used hand augers to obtain the samples. We kept the samples refrigerated soon after collection until analysis. We performed the analysis within one week of collection. For the majority ($n = 91$) of the samples, we only analyzed the concentration of P in the water-soluble pool or fraction. In 19 of the samples, we analyzed P in four additional pools or fractions. The full results are available in the electronic supplement to this article.

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2.2. Geochemical Fractionation

In sediments, P can be present in different chemical forms or pools. Different pools behave differently in the environment, with some relatively immobile and not bioavailable, and other forms easily mobile and bioavailable [35,36]. Forms such as apatite minerals do not influence reservoir water quality as they are not bioavailable, whilst other forms, such as the water-soluble or salt-extractable pools, can easily be released into the reservoir water column [37].

It is difficult to measure accurately P content in different pools, but in the soil science community, they have developed analysis methods called fractionation, extraction, or selective dissolution methods to analyze the P concentration in different pools [37–41]. These methods mix different solvents with the sediment in sequence to extract a selected P-pool, and then they use either centrifugation or filtration to separate the supernatant. P concentration is measured from this liquid.

We followed the Moore and Coale [42] fractionation scheme. This scheme distinguishes five separate P forms, defined by the extracting solution used:

1. Fr.W: water soluble and interstitial water P
2. Fr.KCl: anion exchange sites, loosely sorbed P
3. Fr.NaOH: Al- and Fe-bound or adsorbed P
4. Fr.HCl: Ca-bound (apatite) P
5. Fr.K₂S₂O₈: residual (mostly organic) P

The final step can be performed in two ways, where either case requires sediment digestion. This digestion can either be performed on the sediment residual, or a separate sediment aliquot. If the latter is used, the results provide the total P concentrations in the sediment. Regarding sediment digestion, we followed the Standard Methods persulfate digestion [43]. In general, P in the first three fractions is readily available to the water column. The 3rd fraction, Fr.NaOH, can be released from the sediments in anoxic or anaerobic iron-reducing conditions, a frequent occurrence in reservoir sediments.

The methods we used are shown in Table 1. Table 1 also presents the methods used in the earlier DCR study [34]. There is one major difference, the earlier study included an additional fraction to determine Fe-bound or oxide-occluded P [34]. In our results, P from this fraction was included in the Fr.HCl fraction. This is somewhat conservative, and we assumed that the Fr.HCl is generally not bioavailable, whilst the Fe-bound P could potentially be released under reducing conditions.

Table 1. Comparison of fractionation schemes used in this and the previous Deer Creek study. The table provides the extracting solution and shaking time for both studies. The first and last P fractions (Fr.W and Fr.K₂S₂O₈) steps did not include shaking. In the table, “current” refers to this study, “previous” refers to the study by Messer, Hardy and Ihnat [34].

P Fraction	Current Solvent	t_{shake} (h)	
		Current/Previous	Previous Solvent
Fr.W	DI Water	-	-
Fr.KCl	1.0 M KCl	2/2	1.0 N NH ₄ Cl
Fr.NaOH	0.1 M NaOH	17/18	0.1 N NaOH
Fr.CBD	-	-	CBD *
Fr.HCl	0.5 M HCl	24/18	0.5 M HCl
Fr.K ₂ S ₂ O ₈	Persulfate	-	Persulfate

* CBD is a Na-citrate, Na-bicarbonate, and Na-dithionite solvent.

We used 250 mL Nalgene polypropylene tubes in the fractionation process, with ~10.0 g sediment samples. We shook the samples using a Cole Parmer 51,704 series shaker and centrifuged them with a Sorvall Superspeed RC2-B ultracentrifuge. We filtered suspended solids from supernatants by vacuum filtration with 0.45 µm Geotech geofilters, using a Nalgene reusable filter holder. This size traditionally defines the boundary between dissolved and suspended solids. We prepared 161 samples (including duplicates) for measurement.

2.3. Measurement

We used a standard colorimetric analysis to measure P in the form of orthophosphates in water and wastewater [43]. Specifically, we used the Hach amino acid method (Hach method 8178) with a Hach DR5000 spectrophotometer, for measurement of P in solution after each fractionation step [44]. Hach method 8178 is a colorimetric analysis following Standard Methods [43], which measures dissolved orthophosphates in a range from 0.23 to 30.00 mg·L⁻¹. The Hach method uses factory-supplied calibration curves to convert intensity measurements to concentrations.

2.4. Sediment Concentration Conversion

To determine P content of the sediment, we converted the measured P solution concentrations to sediment concentrations with the following equation:

$$C_{P.in.sed} = \frac{C_{P.in.wat} \frac{1}{1000} V_{liq} D}{m_{dry.sed}} \quad (1)$$

where: $C_{P.in.wat}$ is the P concentration in solution (mg·L⁻¹); V_{liq} is the extractant volume (L); D is the dilution factor; $m_{dry.sed}$ is the dry sediment mass (g); and $C_{P.in.sed}$ is the P concentration in dry sediment (mg·g⁻¹). We adjusted for entrained solution and lost sediment, to calculate the values used in this equation. We determined sediment water content and used that data to calculate the dry mass of sediment ($m_{dry.sed}$) used in the fractionation.

3. Results

We analyzed 161 solutions for P concentrations. Of the 91 samples, we analyzed 72 for water-soluble P and Fr.W only. We completed a full fractionation for the remaining 19 samples. Complete fractionation determines P content in all five pools. We provide the complete results and coordinates for all sample locations in the electronic Supplemental File associated with this manuscript.

3.1. Fr.W (Water Soluble Fraction)

In the 72 samples analyzed only for Fr.W, the average P concentration was 5.19×10^{-3} and 5.12×10^{-3} mg/g dry sediment, for the 72 samples only analyzed for Fr.W and for all the samples, respectively. The standard deviation, σ , of the Fr.W only samples was 3.95×10^{-3} , which was relatively high ~76% of average. This high standard deviation indicated variation in P-distribution.

Tables 2–4 present P with variations by depth, longitudinally, and laterally, respectively. Table 2 presents average Fr.W concentration data by depth (6 inches, 1 foot, and 2 feet) and it indicates a decreasing trend with depth.

In Table 3, the sampling area is presented as transects (as shown in Figure 1), with the first being the most upstream, or nearest in the reservoir inflow. By observation, the data showed a general trend of decreasing concentrations in the downstream direction, though transect 2 is a notable exception. Table 4 presents the data spatially organized by columns (as shown in Figure 1), with column 1 on the right when facing upstream. In general, these data showed higher concentrations on the right side and lower concentrations on the left side across the width of the delta.

Table 2. The average water-soluble (Fr.W) P concentrations in the sediment. The results are presented in the order of sample depth.

Depth	<i>n</i>	$C_{P,avg}$ (mg·g ⁻¹)
Surface	55	5.64×10^{-3}
6 in	15	4.43×10^{-3}
12 in	13	4.06×10^{-3}
2 ft	4	3.99×10^{-3}
All	87	5.12×10^{-3}

Table 3. Longitudinal variation in average surface Fr.W P concentrations, with the samples presented as transects with transect 1 being the most upstream.

Transect	<i>n</i>	$C_{P,avg}$ (mg·g ⁻¹)
1	10	7.29×10^{-3}
2	10	4.81×10^{-3}
3	11	6.31×10^{-3}
4	8	5.59×10^{-3}
5	5	3.82×10^{-3}
6	7	4.03×10^{-3}

Table 4. Lateral variation in average surface Fr.W P concentrations. Data are presented starting from the right side of the delta (looking upstream). Column 1 contains points 7–9 seen in Figure 1.

Column	<i>n</i>	$C_{P,avg}$ (mg·g ⁻¹)	Group
1	3	1.53×10^{-2}	High
2	4	5.88×10^{-3}	
3	4	9.68×10^{-3}	
4	5	7.62×10^{-3}	
5	5	4.10×10^{-3}	Low
6	5	3.47×10^{-3}	
7	6	4.54×10^{-3}	
8	6	4.73×10^{-3}	
9	6	4.04×10^{-3}	
10	6	3.10×10^{-3}	
11	2	3.23×10^{-3}	

Tables 2–4 indicate directional trends. Whilst there are various potential processes that could cause these trends, we did not collect data to explore this question.

We estimated a continuous P-distribution map to visualize P concentration across the delta, we used kriging implemented in the Groundwater Modeling System (GMS) [45]. Using Kriging, we interpolated an estimated Fr.W P distribution map using the Fr.W P concentrations ($n = 55$). Figure 3 shows this distribution as contour data.

The direction trends implied by Tables 3 and 4, are more clearly shown in Figure 3. P-concentration profiles seemed to follow the existing talweg. Maps of other P-pools were similar. We attribute the difference to the smaller number of samples available to guide the estimation. We do not present these maps, as they are based on a relatively few number of samples. However, we provide the data for other researchers to explore these issues.

This work is preliminary in nature, and further geostatistical analysis should confirm and expand on some of these initial findings regarding anisotropy of P content across the delta, and the potential causes of such variation. Additional study could use co-kriging or other methods to better estimate other P-fraction concentrations at sample locations, where we only measured the Fr.W fraction.

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Figure 3. Contours of the Fr.W (water-soluble) fraction from the surface samples. Arrows indicate the inflow talweg, which is better seen in the left panel of Figure 1. The general spatial distribution follows the inflow channels.

3.2. Complete Fractionation

Table 5 presents the average P concentrations in each of the five fractions. We performed complete fractionation on 19 samples. We provide the complete data set in the electronic supplement.

Table 5. Average sediment P concentrations for various fractions. The first two pools tend to be found in much lower concentrations than the other pools, with the organic P (from Fr.K₂S₂O₈) found in greatest concentrations.

Pool	n	$C_{P,avg} \text{ (mg}\cdot\text{g}^{-1}\text{)}$
Fr.W	19	4.70×10^{-3}
Fr.KCl	18	4.93×10^{-3}
Fr.NaOH	19	1.82×10^{-1}
Fr.HCl	18	9.39×10^{-1}
Fr.K ₂ S ₂ O ₈	19	1.46

The sediments, on average, contained much less P in the water soluble (Fr.W) and KCl-extractable pools than in the other pools, whilst there was significantly more (over two orders of magnitude) in the NaOH-extractable, the HCl-extractable, and the residual P pools. Apatite P, extracted with HCl (Fr.HCl), was present in significant amounts, as expected from results of previous work [33,34], and because these sediments were largely calcareous [32]. The residual P (organic, plus P not extracted in previous steps) was the largest fraction present.

3.3. Comparison with Previous Study

Table 6 compares average sediment P concentrations for similar fractions of the current and the previous Messer DCR study [34]. Concentrations reported from Messer et al. [34] (their Table 2) were from their upper sampling location, which is likely within our study area based on their map. The fractionation schemes were slightly different as shown in Table 6, not only in extractants used, but also in other aspects, such as shaking time and quantity of sediment extracted. Additionally, sediments collected in the previous study [34] were taken from the bottom of the reservoir, whilst those

used in this study were taken from the exposed delta. Obviously, there is a substantial time difference (~25 years) between the two studies, and several changes have occurred (e.g., completion of the Jordanelle Reservoir upstream, and limitations on influent P based on management reforms), which could have affected P content of sediments.

Table 6. Fractionation results from current and previous studies by Messer, Hardy and Ihnat [34]. The previous study reports the sum of Fr.NaOH and Fr.CBD as non-apatite inorganic-P (NAIP).

$C_{P,avg}$ (mg·kg ⁻¹)			
This Study	This Study	Messer *	Messer
Fr.W	4.66	-	-
Fr.KCl	4.53	-	-
Fr.W+Fr.KCl	9.19	54.3	Fr.NH ₄ Cl
Fr.NaOH	174.07	155.2	Fr.NaOH
-	-	355.8	Fr.NaOH + Fr.CBD
Fr.HCl	926.31	542.0	Fr.HCl
Fr.PFD	1460	-	Total (NAIP + Fr.HCl)
Total (by sum)	2572.5	1107.3 *	Total (by sum)
-	-	825.6	Total (by digestion)

* We calculated total P by sum, for the previous work; though this did not include organic P. Note that the current total-P was higher than total-P by digestion as completed by Reference [34].

As shown in Table 6, Messer et al., [34] performed one step (Fr.NH₄Cl), whilst we performed two steps (Fr.W and Fr.KCl) to start the fractionation. The sediments contained much less P from these pools in the current study, though this might be reasonable when considering that our sample sediments came from the exposed delta, whilst theirs were from the hypolimnion of the reservoir (exposed v. in solution). Levels of Fr.NaOH P were very comparable, with slightly higher amounts in the current work. This is most likely due to the distinct environments, where P may resorb to or co-precipitate with Fe minerals upon drying of sediments. Apatite P content was much higher in the sediments from the current work. This might be due to pools, perhaps from the CBD step that we did not perform, dissolving into solution with the strong acid. Authigenically formed apatite might be part of the explanation as well; these Ca-P complexes are more likely to dissolve in solution than detrital forms of apatite. Alternatively, the greater amounts of apatite could provide evidence that limitations on external P loading have worked, since this P pool was not regulated as other pools were [34].

There were significant amounts of residual P present in both studies, and this is another point of departure between the two studies. In the current work, we calculated total P by the summation of concentrations of individual pools, whilst in the previous work total P was determined by digestion of a separate aliquot of sediment. To compare, we could calculate residual P for the previous study by subtracting inorganic P from the total P they obtained. However, they extracted more inorganic P (897.8 mg·kg⁻¹ by summation of Fr.NaOH, Fr.CBD, and Fr.HCl) than they obtained by total digestion (825.8 mg·kg⁻¹) (see Table 6 for summary of results), so we do not place any confidence in this comparison. The anomaly in their data might be due to resorption of P in the fractionation process or because of P entrained in solutions.

Our study and the previous study used different fractionation schemes. Initially it seems that we should be able to compare P concentrations by cumulative sums, but Wang, et al. [46] show that the fractions extracted by various schemes are not well defined. They note that each fractionation scheme yielded operationally defined pools, that cannot be used to identify specific P compounds. This means that quantitative comparisons between different schemes is difficult, and comparisons are limited to qualitative discussions. One specific issue that affects different schemes is reabsorption to CaCO₃ [46]. As DCR sediments are calcareous, this could significantly affect the behavior of different fractionation schemes.

4. Conclusions

This work makes available a unique data set for researchers. It is part of a larger project attempting to understand, characterize, and model water quality issues at DCR. We gathered these data to characterize how deposited delta sediments might provide a recycled nutrient, specifically P source to the water column, in the case of reservoir flooding (refilling after withdrawal) over the exposed delta. This study provided useful data and insights for further work in this area.

Fr.W P concentrations have spatial trends both horizontally and vertically. Based on our limited data, the other fractions of P behaved similarly [47], which was supported by P-distribution maps. By observation, it appeared that the general ratios between the P-fractions was relatively constant, allowing co-kriging or other methods to be used to evaluate spatial distributions in more detail. As noted, we used kriging to characterize these distributions that follow the general Fr.W patterns.

We hope that researchers will use this unique data set to better understand processes in reservoir sediment deltas, including re-cutting and resuspension and the resulting spatial trends. This can inform reservoir managers about the impact that reservoir drawdowns could have, as the drawdowns affect nutrient concentrations due to re-cutting and resuspension of delta sediments.

Our initial results suggest that there is more apatite-P in the exposed delta sediments than in submerged sediments sampled previously, based on comparison of our results with those of Messer et al. [34]. This could have repercussions on refilling reservoirs, as some of the apatite-P found in delta sediments might be of authigenic (precipitated in area previously covered by water), instead of detrital origin. This type of process might have implications in the Western United States, where the extended droughts over recent years have significantly lowered many reservoirs exposing delta sediments. This modified P might more easily become available as the reservoir is refilled, releasing additional nutrients to the water column.

This increased apatite fraction may also indicate that the external P loading reductions started in 1981, as described by Reference [32], have been successful. Apatite-P is the least bioavailable pool due to high insolubility, so Messer et al. [34] recommended that this pool not be targeted for reduction. Our data showed an increased percentage for this pool and it may be evidence of reduction in the other P pools that were targeted for reduction. We expect to explore these issues further, and hope that by providing these field data to the research community, we can spur additional insight.

Supplementary Materials: The following are available online at <http://www.mdpi.com/2306-5338/5/4/58/s1>. Table S1: Coordinates for sample locations; Table S2: Summary of sediment P concentrations ($\text{mg}\cdot\text{g}^{-1}$) for Sample Locations 7–29; and Table S3: Summary of sediment P concentrations ($\text{mg}\cdot\text{g}^{-1}$) for sample locations 30–59.

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Article

Characterizing Total Phosphorus in Current and Geologic Utah Lake Sediments: Implications for Water Quality Management Issues

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Abstract: Utah Lake is highly eutrophic with large phosphorous inflows and a large internal phosphorous reservoir in the sediment. There are debates over whether this phosphorous is from geologic or more recent anthropogenic sources. This study characterizes total phosphorous in geologic and current lake sediments to attempt to address that question. The average total phosphorous concentrations in the lake sediment were 666 ppm, with most samples in the 600 to 800 ppm range with a few larger values. Concentrations in historic geologic sediments were not statistically different from lake sediments. A spatial analysis showed that phosphorous distributions appeared continuous from the lake to the shore and that high and low values could be attributed to areas of seeps and springs (low) or feed lots and waste water discharge (high). These results indicate that geologic sediments without anthropogenic impacts are not statistically different than current lake sediments. The high values indicate that internal natural phosphorous loadings could be significant and the impaired state may be relatively insensitive to external anthropogenic loadings. If this is the case, then mitigation efforts to address anthropogenic sources may have minimal impacts. This case study presents an impaired water body where non-anthropogenic nutrient sources are significant and shows that reservoir management decisions should consider these non-anthropogenic phosphorous sources relative to anthropogenic sources. This study can serve as a template for evaluating the importance of geologic phosphorous sources for management decisions.

Keywords: reservoir management; sediment phosphorous; non-anthropogenic phosphorous; sediment-water interaction

1. Introduction

1.1. Study Goals and Importance

Utah Lake has extensive algal blooms, is highly productive, highly eutrophic, and nutrient rich [1,2]. The Utah Division of Water Quality has determined that phosphorous inflows, in the form of phosphate, significantly contributes to the impairment of Utah Lake [1,2] and are considering a number of mitigation efforts, all of which carry significant costs in both monetary and resource terms. These mitigation efforts assume that external phosphorus loading, mainly from anthropogenic sources, significantly contribute to Utah Lake's impaired state. We present evidence that internal natural phosphorous loadings could be a significant factor, and that the impaired state of the lake may be relatively insensitive to external anthropogenic loadings.

Utah Lake has high phosphorous inflow loadings, on the order of 300 tons/year, with an estimated 80% of this total, 188.8 tons/year, from wastewater treatment plants (WWTP). Two WWTPs, Provo City and Orem City, are estimated to contribute 71.6 and 79.3 tons/year, respectively, or approximately 50%

of the total annual phosphorous load. The estimated Utah Lake outfalls of phosphorous are on the order of only 85 tons/year [1,2] indicating the lake acts as a phosphorous sink. Our research shows that total phosphorous in the lake sediment is very high, but most importantly, phosphorous concentrations in geologic lake sediments, deposited prior to human impacts, are not statistically different than current lake sediments; they have similar levels. This indicates that the anthropogenic loading is not significantly affecting phosphorous levels in lake sediments. It is likely that these high sediment phosphorous levels directly contribute to phosphorous levels in the water column. We infer from these data, and some simple lab experiments, that water column phosphorous levels are likely moderated by phosphorous in the sediment, not by inflows. If this is the case, then costly mitigation efforts at wastewater treatment plants will have a minimal impact on Utah Lake water quality. Our results indicate that phosphorous levels in the sediment are from geologic origins. The Utah Lake watersheds include the Park City formation, which is an upper Carboniferous formation of phosphate-bearing shales about 100 feet thick (30 m) [3]. It is rich in phosphates and includes deposits of economic phosphorite ores mined in the Utah Lake watershed [4]. The Park City formation is part of a region called the western phosphate field that includes the Phosphoria, Park City, and Embar formations [5–7].

This study is important to reservoir and lake management as it provides a case study of an impaired water body where non-anthropogenic nutrient sources are significant and should be considered when evaluating mitigation efforts. There are other studies in this area such as [8], which evaluates non-anthropogenic and anthropogenic nitrogen sources for water quality impacts. Reservoir management decisions to address water quality issues made without considering these non-anthropogenic phosphorous sources may not be optimal and may be ineffective. This study does not attempt to estimate phosphorous loading to Utah Lake from the sediments, as that would require significant field and laboratory work to characterize phosphorous release and transport processes within the sediments and from the sediment to the water column. It presents total phosphorous amounts in the lake and geologic lake sediments, discusses the observed spatial distributions, and presents results from some preliminary laboratory studies that indicate the relative magnitude of sediments as phosphorous sources.

1.2. Phosphorous

Phosphorous is a primary limiting nutrient of phytoplankton growth in many streams, lakes, and reservoirs including Utah Lake [9]. Phosphorous can come from both anthropogenic sources such as sewage, WWTP discharge, industrial discharges, and fertilizers and natural sources such as precipitation or geologic materials [9].

This study presents the results of a field sampling campaign to characterize the amount of phosphorous present in both current and geologic Utah Lake sediments. Samples were taken from current lake sediments and from geologic (older than 10,000 years) sediments on the lake shore and analyzed for total phosphorous content. The results were evaluated to determine if the differences among the different measurements were statistically significant. These results can then be used to infer if the high phosphorous levels in the lake sediments are from anthropogenic activities or geologic in origin. While this study is unique to Utah Lake, many other impaired lakes and reservoirs exist in regions that have high geologic phosphorous content. This study can serve as a template for evaluating the importance of geologic sources for management decisions.

The Vollenweider model [10] is widely used to evaluate reservoir health. It estimates reservoir phosphorous concentrations using a mass balance approach that includes phosphorous inflow relating in-lake phosphorus (P_{lake}) inlet concentrations (P_{in}) and hydraulic residence time (tw):

$$P_{lake} = \frac{P_{in}}{1 + \sqrt{tw}}. \quad (1)$$

This model assumes reservoir phosphorous levels are only due to phosphorous inflows and the amount of time required to flush the impoundment. However, this model is not applicable for water

bodies that have significant internal phosphorous sources such as Utah Lake [11]. In these reservoirs, in-lake water column phosphorous levels may be independent of inflow concentrations and moderated by interactions with the sediments. Søndergaard et al. [12] describe a mass balance on phosphorous in the sediment and state that when phosphorous flux from the sediment to the lake exceeds phosphorous inflows, and then reduction in lake inflows may have little to no effect on water quality conditions [12]. That is, phosphorous levels in the water column may be controlled by sediment interactions rather than external sources. Lake sediments can act as either a source or a sink for phosphorous in the water column depending on the amount of dissolved oxygen, water column phosphorous concentrations, and sediment phosphorous concentrations [13].

1.3. Phosphorus Release from Sediments

Phosphorous sediment-water interaction is a complicated and governed by different processes depending on lake and chemical compositions [12]. Parameters that affect phosphorous partitioning between the water column and the sediment include sediment dry weight, organic content, and iron (Fe) content along with the concentration of other elements with the capacity to bind and release phosphorus. Water parameters include pH, reduction potential (pE), total dissolved solids, and others. Release mechanisms are governed by parameters such as: Fe:P ratio, reduction–oxidization (Redox) conditions, resuspension, turbidity, biological activity, microbial processes, temperature, and pH [12].

Phosphorus in sediments can be in several fractions including water-soluble, salt-extractable (sorbed), iron-bound, organic, and pH-sensitive (generally inorganic or mineral) [14–16]. Phosphorous in the water-soluble fraction is generally in equilibrium with sediment pore water and diffuses easily into the water column. Salt-extractable phosphorous is typically sorbed onto the sediment and can be released by ion exchange mechanisms (salt). Iron-bound phosphorous is bound to iron minerals in the sediments, typically iron-hydroxides which are significantly more soluble under reducing conditions. Organic phosphorous is typically found in detritus from living organism, while the mineral fraction is typically mineral phosphorous that is not bio-available or easily dissolved. The first two fractions, water-soluble and salt-extractable tend towards equilibrium in a shallow, well mixed lake with sediment resuspension, like Utah Lake [14,16].

Iron reduction and associated phosphorous release are governed by aeration levels, with anaerobic or anoxic conditions creating conditions that contribute to the release of phosphorous [17]. This is often catalyzed by anaerobic microbial processes. Redox conditions regulate phosphorus release into pore water for the fraction of phosphorous bond to the iron minerals. When Fe(III) is reduced into Fe(II), the iron mineral solubility is significantly higher, up to two orders of magnitude, resulting in iron dissolution. As the iron dissolves, phosphorus bound to the iron minerals is released into water [18,19]. In 1936 and 1947, Einsele [20] and Mortimer [19], respectively, showed that under oxygenated conditions sediments retain phosphorus by fixation to Fe(III), whereas, under anaerobic or reduced conditions, phosphorus is released when the iron is reduced to a more soluble form, Fe(II). Ruban and Demare [21] concluded that phosphorus release was avoided as long as dissolved oxygen concentration stayed above 0.5 mg/L. Recently, Petticrew and Arocena [22] showed that phosphorus release rates under reducing conditions are strongly correlated to iron bound phosphorus [12], and Jensen et al. [23] concluded that retention capacity is proportional to the Fe:P ratio if the ratio exceeds 15 by weight while Caraco et al. [24] stated that the ratio can be less than 10. Jensen, Kristensen, Jeppesen and Skytthe [23] analyzed sediment from 12 different lakes and found that for aerobic sediments the phosphate concentration is related to the Fe:P ratio and total iron controls the maximum adsorption capacity [25]. Phosphorous releases are also affected by calcium concentrations with release rates more affected by pH than redox conditions [26].

1.4. Biological Activity

Many of the chemical processes that affect phosphorous equilibrium and release rates from sediments are relatively slow, but can be catalyzed by biological activity, which significantly increases

the rate at which these processes occur. For example, iron reduction or oxidation can be relatively slow, on the order of months to years or longer; biological processes can significantly increase this rate to time scales on the order of hours to days [27]. Biological activity also can create more reducing conditions through decomposition of organic matter using existing oxygen.

Biological processes can make phosphorous from the organic fraction available to the water column as microbial processes consume oxygen and release phosphorous from the decomposition of organic materials [17]. This means that biological processes can release phosphorous under both oxidizing (organic fraction) and reducing conditions (iron-bound fraction) depending on conditions. If sufficient material (food) is available, aerobic processes continue until oxygen and nitrate are exhausted as electron acceptors, then iron or sulfates are used, resulting in reducing conditions, causing iron dissolution and release of iron-bound phosphorous as discussed. During calm, warm summer periods, especially when large amounts of biomass are decomposing, biological activity is increased, releasing phosphorous from organic materials and potentially causing anoxic events to occur releasing iron-bound phosphorous in the sediments [12,17]. Temperature also affects phosphorus release from sediments, as higher temperatures can increase microbial activity creating anoxic or anaerobic conditions and catalyzing mineral dissolution [19,28–30].

1.5. Resuspension

In shallow lakes, such as Utah Lake, physical processes, such as boating, wave action, and carp feeding, regularly disturb the sediment, mixing phosphorous-rich sediments with the water column. Resuspension is a mechanical process that permits settled sediments to be redistributed in the water column and enhances phosphorus exchange between the suspended material and the water column [31]. A number of factors cause resuspension including waves [32], wind [33,34], ice cover, turbidity [35], and climate change [36]. Carp and other benthivorous fish have a large impact on sediment resuspension and nutrient concentrations in the water column. Their rooting activities cause sediment resuspension as they search for food and digestive activities release phosphorus into the water. Utah Lake has a significant carp population that are bottom feeders and create significant sediment disturbance, which can lead to higher phosphorous content in the water column and algal growth in lakes [2].

Experiments have shown that internal reservoir phosphorus loading encouraged by resuspension to be 8 to 10 times greater [37] and 20 to 30 times greater [34] than that from undistributed sediments. During resuspension, the majority of the released phosphorous is not from the iron-bound fraction because reservoir water is aerobic, but more often from the water-soluble and salt-extractable fractions and is not dependent on the aerobic state of the water. If reservoir water is anaerobic or anoxic, then iron-bound phosphorous can also be released to the water column.

2. Utah Lake

2.1. Background Information

Utah Lake (Figures 1 and 2) is considered the third largest freshwater lake, by the surface area, west of the Mississippi with a surface area of about 150 square miles (384.5 km²). Volume is about 870,000 acre-feet (1100 million cubic meters). It is a shallow lake with an average depth of 9 ft. (2.74 m) and a maximum depth of about 12 feet (3.6 m) in non-drought years. During drought years, the average depth is shallower. The lake lies in Utah valley surrounded by the major cities of Provo, Orem, and Saratoga Springs.

Utah Lake is a popular recreation area because of its proximity to the metropolitan areas of Provo and Orem. Approximately 150,000–200,000 people visit Utah Lake each year for its shoreline recreation, fishing, and water activities. It also supports agriculture uses as it is used to irrigate about 50,000 acres (202.3 km²) yearly.

Utah Lake has fifteen key tributaries with the majority of inflows from the Provo and Spanish Fork Rivers, which together account for approximately 60% of the inflow. Other inflows are from springs, groundwater, overland flow, and precipitation. The Jordan River is the only surface outlet and is located on the north end of the Lake [38]. Utah Lake is a semi-terminal lake with the outfall only about half the inflow, even though there is a significant groundwater contribution to the lake. Evaporation plays a large role in the hydrology of Utah Lake, as almost 41% of its water is lost due to evaporation each year. The lake has high sedimentation rates, with an estimated 60% to 80% of the sediment being calcite mainly coming from mineral precipitation from the water column [39,40]. This high evaporation rate coupled with the very shallow nature of Utah Lake causes the high dissolved solids in the lake water, essentially maintaining dissolved solids at the saturation level for calcite. Average annual precipitation in the area is about 19.8 in. (50.3 cm). Table 1 provides an overview of the hydrology of Utah Lake.



Figure 1. Utah Lake located in northern Utah and is a large, freshwater lake with a surface area of about 150 square miles.



Figure 2. The Jordan River basin that contains both the inflows and outflows for Utah Lake (figure created and data from [41]).

Table 1. Hydrologic summary of Utah Lake.

Tributary	Avg. Flow (Acre-Feet/Year)	Outflow	Avg. Flow (Acre-Feet/Year)
Provo River	+151,000	Evaporation	-349,800
Spanish Fork River	+99,700	Jordan River	-428,200
Benjamin Slough	+36,700		
Provo Other Inflows	+134,200	Evaporation	-349,800
Spanish Fork River	+99,700	Jordan River	-428,200
Benjamin Slough	+36,700	Total Outflow	-778,000
Other Inflows	+134,200		
Total Inflow	+829,300	Total Outflow	-778,000

Utah Valley is a sediment-filled basin that contained the historic Lake Bonneville, which occupied the basin between about 23,000 and 10,000 years ago. This historic lake filled Utah valley with lake sediments that created a nearly flat lake bottom and relatively flat shoreline [38]. The greatest lake level elevation of 5090 feet (1550 m) was achieved about 16,000 years ago (the Bonneville level). Then, about 13,000 to 14,000 years ago, the level dropped to about 4740 feet (1440 m) (the Provo level). After the ice age, lake levels declined to about 4250 feet (1295 m) (the Gilbert level) about 10,000 years ago [38]. Sediments from the Gilbert level are the geologic sediments sampled in this study. The current lake level is about 4490 feet (1370 m) above sea level, higher than the Gilbert level because of recent sedimentation.

2.2. Sediment

Utah Lake is shallow with an average depth of 9.2 feet (3 m). In drought years, this can be significantly less. In Utah Lake wind action, recreation, and carp resuspend sediments in the water column, potentially releasing phosphorous [42]. These processes resuspend the sediments to create a lake bottom composed of soft ooze rather than more compacted sediments [40]. According to Horns [38], there generally is no distinct boundary between water and sediment at the bottom of Utah Lake with wave action mixing sediment up to a foot deep (0.3 m). This creates conditions where phosphorous stored in the sediment significantly contributes to phosphorous in the water column. Utah Lake sediment is high in organic content and is generally anaerobic.

phosphorous stored in the sediment significantly contributes to phosphorous in the water column. Utah Lake sediment is high in organic content and is generally anaerobic.

Based on seismic profiling, the average sedimentation rate over the last 10,000 years was between 0.8 mm/year and 1.5 mm/year [38,40]. Studies involving dandelion pollen, a species introduced in 1849, estimate sedimentation rates of about 1.38 cm per year over the last 150 years [37]. Recent studies estimate the long-term sedimentation rate to be about 1 mm/year, with a possible increase over the past 150 years to about 2 mm/year [37,39]. The sediments below Utah Lake extend to a depth as great as 10,000 feet (3000 m) [38] with deep sediments and shore sediments over 100 feet (30 m) above current lake levels from historic Lake Bonneville. These sediments present a large potential reservoir of geologic phosphorous.

2.3. Utah Lake External Phosphorous Loading

Different studies have estimated external phosphorous loading into Utah Lake and have identified loadings provided from streams, springs, drains, wastewater plants, and other sources. The State of Utah Department of Environmental Quality (DEQ) Division of Water Quality conducted a study to calculate total phosphorous loads for each inflow to the lake [1]. Evaporation and precipitation were ignored in these calculations assuming that they have negligible phosphorous amounts.

Figure 3 presents a summary the DEQ study estimating the total phosphorous loading for Utah Lake [1]. The total phosphorous inflow DEQ estimated and anthropogenic sources for Utah estimates that anthropogenic sources, mostly WWTPs provide about 77% of the total phosphorous intake. They estimate the total phosphorous intake is approximately 297.6 tons/year while the only phosphorous outflow is the Jordan River with a total phosphorous outflow of approximately 83.5 tons/year. These estimates show that Utah Lake is a phosphorous sink. Even with Utah Lake acting as a large phosphorous sink, we later show that total phosphorous levels in current lake sediments are not statistically different than levels of geologic lake sediments.

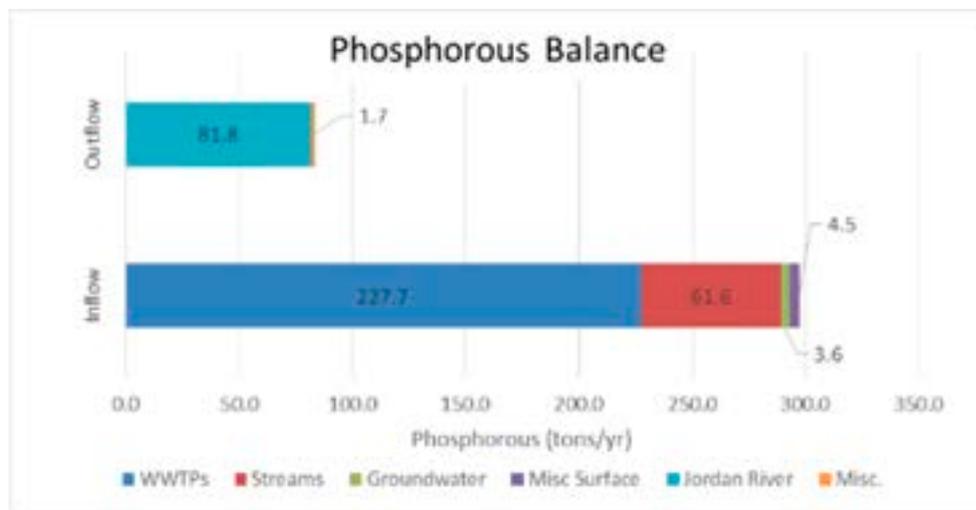


Figure 3. Summary of the total phosphorous load and exports from Utah Lake [1].

3. Methods

3.1. Sampling Methods and Locations

We considered many factors during the design and implementation of the sampling process. We wanted to representatively sample the lake with expected data to have higher phosphorous concentrations and patterns, and also characterize regions of the lake we expected to have high phosphorous amounts because of inflows. Because of the size of Utah Lake, access was also a consideration. We took 36 samples in 2015 and 49 samples in 2016. Samples from 2015 were reported in [43].

Sampling occurred during two separate summer field campaigns, June through August, in 2015 and 2016. We took 36 samples in 2015 and 49 samples in 2016. Samples from 2015 were reported in [43].

We sampled semi-random locations in the body of the lake, and more or less equally spaced locations around the lake perimeter (Figure 4). We sampled with higher density near inlets or points where overland flow was likely. This included additional points to represent the locations of the inflows from WWTP locations and the outlet. We took ten samples of geologic sediments. Five of these were on the northwest side of the lake in areas that exhibited very little anthropogenic impact—that is, there did not appear to be any indication of agriculture use, construction or animal containment (e.g., corrals for fenced areas). The two samples on the west side of the lake were in areas which had no visible indications of recent impacts, but it was more difficult to determine because of land cover. These geologic sediment samples are from lake deposits about 10,000 years ago.

Figure 4 shows the distribution and location of these sample points. The figure indicates the lake sediment points sampled in 2015 as blue squares, the lake sediment points sampled in 2016 as red circles, and land samples as black triangles that represent geologic lake sediments.

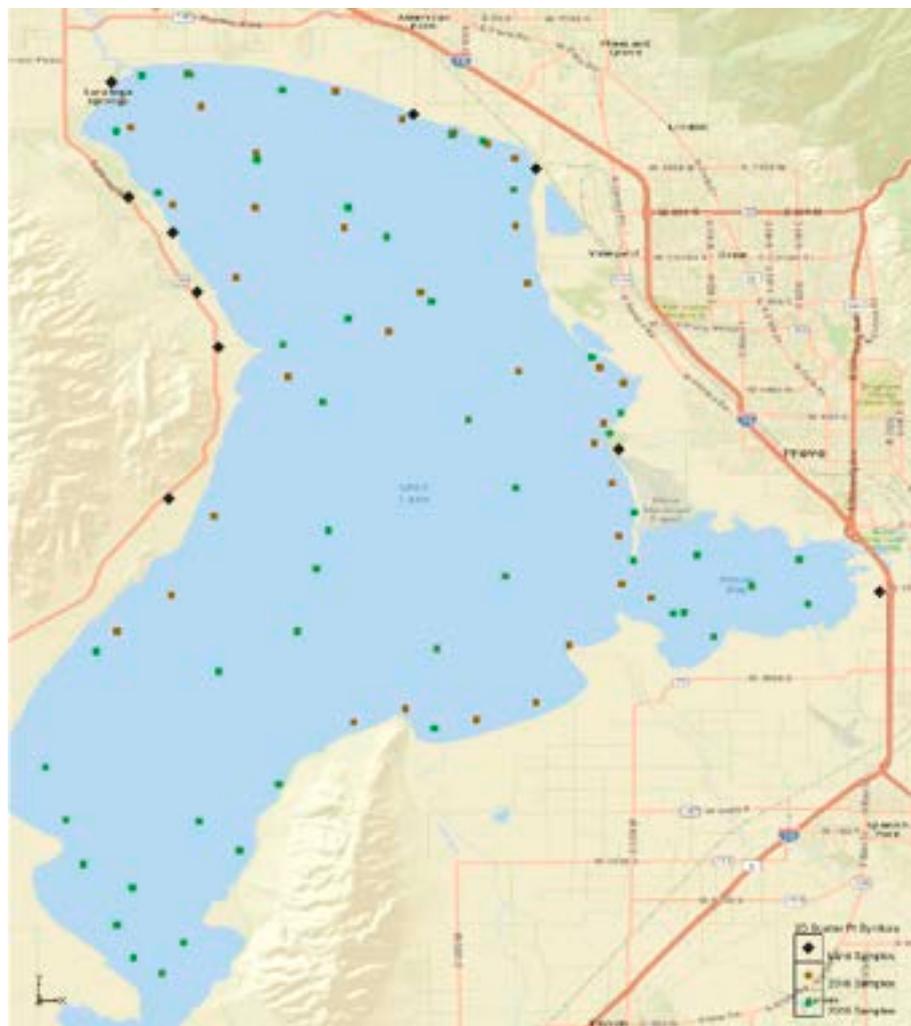


Figure 4. A map showing the location of the sediment and soil sample points, the figure indicates sediment points sampled in 2015 as blue squares, sediment points sampled in 2016 as red circles, and samples taken on the shore as black triangles, which represent geologic lake sediments.

Samples were collected near the top of the sediment layer from 0 to 4 inches in depth. As noted, the top foot or so of the Utah lake sediments are continuously re-worked, resulting in a layer of

sediments that are well mixed and in good contact with the water column. This near-surface sediment region does not exhibit significant layering. Based on field observations, we saw little vertical variation in the first foot or so of the sediment.

Once collected, we stored samples in Ziploc bags, labeled with number, location, and date, and kept in a cooler. We transported samples to the Brigham Young University (BYU) Civil Engineering Environmental Laboratory and stored the samples in a refrigerated room until analysis.

3.2. Chemical Analysis Methods

The analysis for total phosphorous was done by the BYU Environmental Analytics Laboratory (Provo, UT, USA). Samples were digested with an Ethos OZ Microwave digestion system (Milestone, Fatebenefratelli 1/5, 24010, Sorisole (BG), Italy) to dissolve the sediment and release phosphorous. The results were filtered using a 0.45 μm filter and analyzed using a Thermo Scientific iCAP™ 7400 ICP-OES Analyzer (ThermoFisher, Boston, MA, USA) with a phosphorus detection limit of 1.55 $\mu\text{g/L}$. The dry-weight total phosphorous was then calculated and reported as mg/kg (mg of phosphorous per kg of sediment), which is parts-per-million in mass units (ppm).

The Environmental Analytics Laboratory also analyzed the 36 samples taken in 2016 to determine how the phosphorous was distributed in five different pools or fractions. These fractions were water soluble, salt extractable, iron-bound, mineral, and organic. The first three pools represent phosphorous that can be released by sediment resuspension and anaerobic conditions. Unfortunately, these samples were allowed to dry and an incorrect laboratory method was used so we could not use the results, and they are not presented.

3.3. Statistical Analysis Methods

We divided the lake into regions, based on spatial areas (described below) and used two different statistical tests to determine if the samples in these regions came from a single population or if there were differences in the populations.

We used common statistical and spatial analysis tests [44–49]. We performed a chi-square test to determine the likelihood that data in the different lake sections came from different populations. Specifically, we tested the null hypothesis that all sections have the same average level of phosphorus [43]. We performed a stochastic pairwise nonparametric Tukey analysis [50,51], which compared pairs of samples from all the sets to determine if they differed from each other. Sample pairs were randomly picked. To develop spatial maps of phosphorous extent, we used the Shepard's interpolation method implemented in the Groundwater Modeling System (GMS) version 10.2 software package (Aquaveo, Provo, UT, USA) [52]. Shepard's method is an inverse distance weighting approach that includes permeable and absolute barriers [53].

4. Results

4.1. Sampling Results

Total phosphorus concentration varied throughout the lake, ranging from 280 to 1710 ppm, with an average value of 666 ppm. Figure 5 shows the results and sampling points. The average total phosphorous was 719 ppm, 604 ppm and 786 ppm for sediments sampled in 2015, 2016, and land samples, respectively. Statistical analysis (below) showed that these values are not significantly different. In addition to population statistics (discussed below), there are some differences that can be explained by physical processes. The samples with significantly low values were all taken in areas of groundwater or spring discharge to Utah Lake. We attribute these significantly lower phosphorous levels to leaching or flushing of phosphorous from the sediment by the groundwater. Groundwater is anoxic and can remove phosphorous in the first three fractions, water-soluble, salt-extractable, and iron-sorbed. The three highest samples were taken from sediments near a corral/feedlot complex with significant animal waste.



Figure 5. Total phosphorus in ppm at each sample point. Some points between 2015 and 2016 are close and labels overlap. The map is not to scale.

4.2. Data Analysis

For analysis purposes, we divided the lake horizontally into four sections as shown in Figure 6.

These sections each have different dominant factors that could result in differences in phosphorous content. For analysis purposes, we divided the lake horizontally into four sections as shown in Figure 6. These sections each have different dominant factors that could result in differences in phosphorous content. The lower area, labeled 4 in Figure 6, consists of the shallow southern portion and does not contain any significant inflows. There are springs along the eastern shore of this section. Section 3 (Figure 6) includes Hobbie Creek large inflow to the lake. The eastern shore, just above the bay, includes areas of seeps and springs. The large bay on the east side of this section is shallow and includes agriculture and feedlots on the lake shore. Section 2 (Figure 6) includes the Provo River, the main inflow to the lake, and contains the outfalls for the Provo and Orem wastewater treatment plants, the largest phosphorus contributors to the reservoir. The eastern shore of this area includes areas of seeps and springs. The large bay on the east side of this area is shallow and includes agriculture and feedlots on the lake shore. Section 1 (Figure 6) contains the Jordan River outfall, the only exit point for Utah Lake. This section also has seeps and springs on the northwest shoreline.

Table 2 presents the descriptive statistics calculated for the samples taken in each section, while Figure 7 shows the data as well grouped, with each section having one to three low values except for the land samples, which do not have low values. With the exception of Section 3, all the

data fall within 1.5 IQR of the lower or upper quartiles, indicating low dispersion in the data. Section 3 has high outliers, which we attribute to feedlot runoff (discussed below). We attribute the low values in each section to samples taken in areas of seeps and springs (discussed below). The land section does not contain any outliers. Figure 7 shows the data with 1.5 IQR from lower and upper center quartile for each section. Samples were taken in seeps or springs or high values as on samples taken near feedlots for the land samples. Section 4 has the highest average phosphorous concentration along with the highest values for the 1st and 3rd quartiles, indicating low dispersion in the data. Section 3 has high outliers, which we attribute to feedlot runoff (discussed below). We attribute the low values in each section to samples taken in areas of seeps and springs (discussed below). The land section does not contain low values as no samples were taken in seeps or springs or high values as no samples were taken near feedlots or animal yards. Section 4 has the highest average phosphorous

We used a chi-square test to determine the likelihood that the data in different sections come from different populations—that is, if they are different. We constructed the different lake sections and land section, based on some general differences in physical properties; in these different physical properties affect the total phosphorous levels. Then, the chi-squared test should indicate the values

We tested the null hypothesis that all sections have the same average level of total phosphorus. We obtained a *p*-value of 0.1210, and this is well above the standard confidence level of 0.05 (95%). With a *p*-value this large, we fail to reject the null hypothesis and cannot state that there are any differences between the different sections. There are no statistically significant differences among the five groups.

The chi-square analysis is a global test, looking at a single section compared to the entire data set. Figure 7 seems to show some difference between the various sections. To explore this hypothesis, we performed a pairwise nonparametric Tukey analysis to compare each set of pairs to determine if they differed from each other.



Figure 6. The four sections used for analysis of the Utah Lake sediment data. Figure 6. The four sections used for analysis of the Utah Lake sediment data.

Table 2. Statistical summary for samples from each of the four lake sections and from the land area. Table 2. Statistical summary for samples from each of the four lake sections and from the land area.

Section	# of Samples	Avg. (ppm)	Standard Deviation (Std)
1 (Upper)	24	660	150
2 (Middle Upper)	20	631	174
3 (Middle Lower)	24	668	304
4 (Lower)	17	714	118
Land Samples	10	786	151

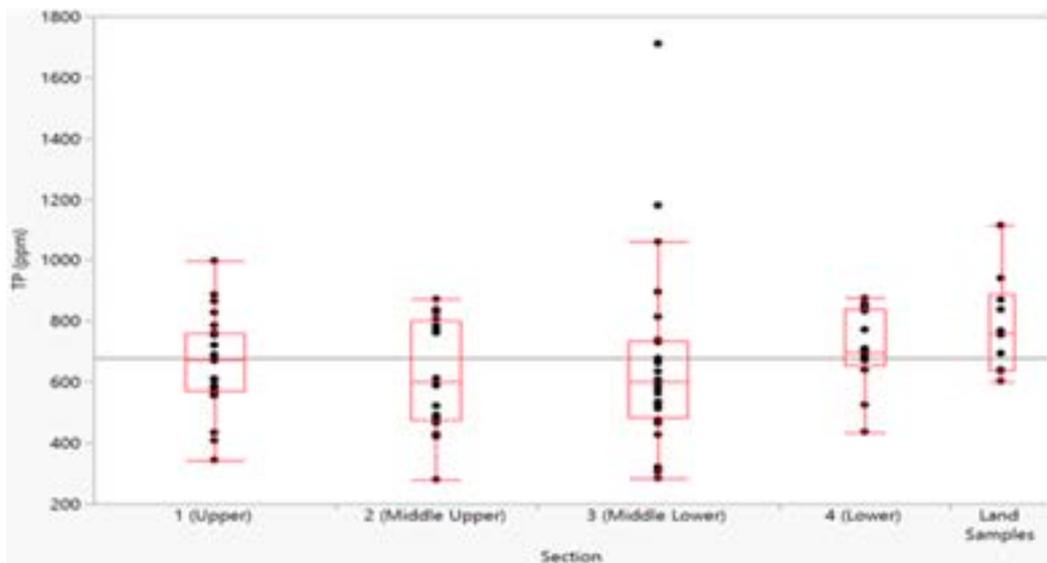


Figure 7. Box plots of total phosphorus measurements for each of the sections shown in Figure 6 and land samples.

Table 3 presents the Tukey results ordered from least likely to reject the null hypothesis to most likely to reject the null hypothesis—in other words, from least likely to be the same to most likely to be the same. The results show that the differences between the groups are not statistically significant. The sections most likely to be different are the land and Group 2 samples. The *p*-value for a comparison of these two sections is 0.29, significantly larger than the standard confidence level of 95% (*p*-value of 0.05). With a *p*-value of 0.29, we can state that the differences between these two sections, and any of the other pairs, are not statistically significant. We do note that there were only 10 land samples. This small number of samples is most likely the cause of the smaller *p*-values for the differences of the land samples with the other groups. However, even these smaller values are significantly larger than the *p*-value for a standard confidence level of 95%. We cannot reject the statement that the groups are the same, and the differences are not statistically different. The small number of samples does increase the uncertainty some. While it would have been beneficial to have more land samples, the focus of the sampling program was on lake sediments. Additional land samples could be addressed in the future but is not as important as evaluating sediment–water flux rates and phosphorous loads from sediment–water interactions, erosion from surface runoff, and short-range dust transport.

Table 3. Statistical summary of difference tests among the areas.

Area	Area	Difference	Std Error of Diff.	<i>p</i> -Value
Land	2 (Mid Upper)	154.6	78.7	0.29
Land	1 (Upper)	126.0	76.5	0.47
Land	3 (Mid Lower)	101.0	77.0	0.68
4 (Lower)	2 (Mid Upper)	82.3	67.0	0.33
Land	4 (Lower)	72.3	81.0	0.90
4 (Lower)	1 (Upper)	73.8	64.4	0.92
3 (Mid Lower)	2 (Mid Upper)	53.6	62.1	0.91
4 (Lower)	3 (Mid Lower)	28.7	65.0	0.99
1 (Upper)	2 (Mid Upper)	28.6	61.5	0.99
3 (Mid Lower)	1 (Upper)	25.0	59.3	0.99
3 (Mid Lower)	1 (Upper)	25.0	59.3	0.99

4.3. Geostatistical Analysis

We noted spatial patterns in the data. While statistical analysis showed no difference among the four lake sections and the land, there are areas of the lake that present different spatial distributions. For example, all the high values in Section 3 are spatially correlated and located in Provo Bay. To

4.3. Geostatistical Analysis

We noted spatial patterns in the data. While statistical analysis showed no difference among the four lake sections and the land, there are areas of the lake that present different spatial distributions. For example, all the high values in Section 3 are spatially correlated and located in Provo Bay. To analyze the spatial distribution patterns and attempt to assign these differences to various physical processes, we used geostatistical models to create phosphorous distribution maps.

We created these maps using all the total phosphorous data using the Shepard's interpolation method implemented in the GMS version 10.2 software package [52]. These spatial maps show phosphorus distribution patterns in and around the lake. We created two maps:

- A map using only points inside the lake—only lake sediment samples (Figure 8),
- A map using points inside and outside the lake—both lake sediment and shore sediment samples (Figure 9)

These maps show the distribution of phosphorous in Utah Lake sediments and how that distribution relates to distributions in the shore sediments.

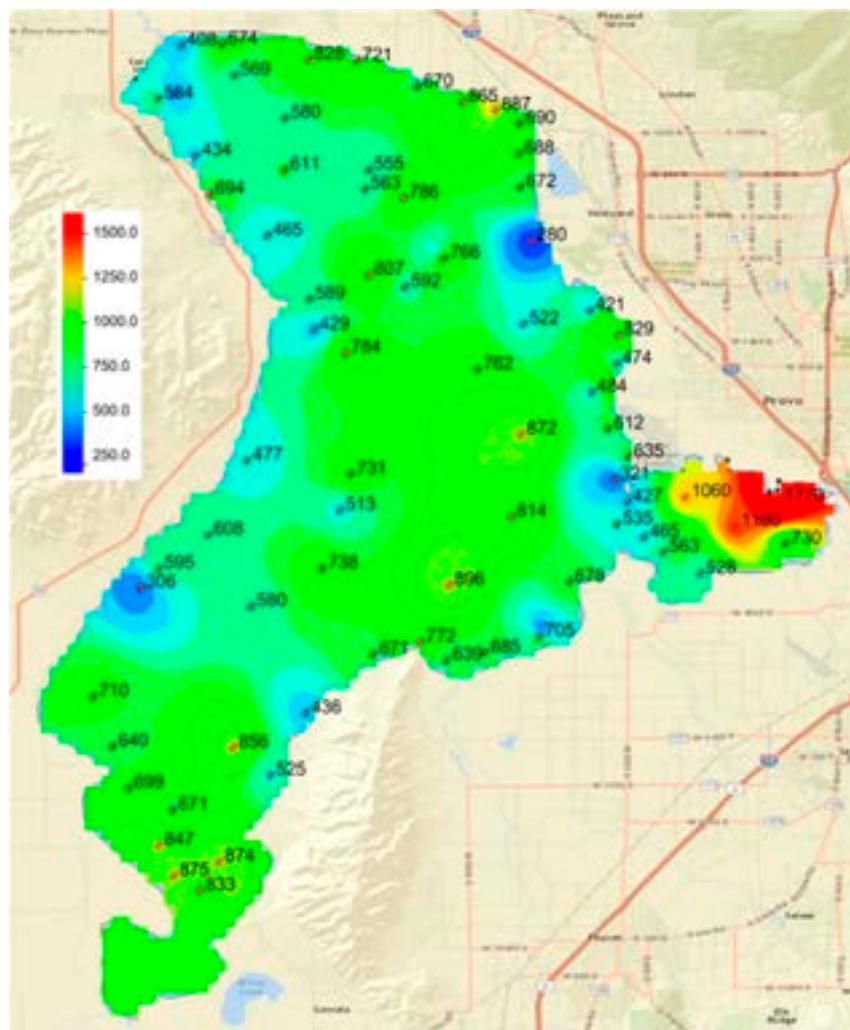


Figure 8. A geostatistical Model showing total phosphorous distribution in Utah Lake developed by interpolating along the sediment sample points inside the lake. The colors represent the interpolated values, the numbers indicate total phosphorous concentrations at the sample points. Both the legend and the numerical values represent total phosphorous amounts in ppm, and the map is not to scale.

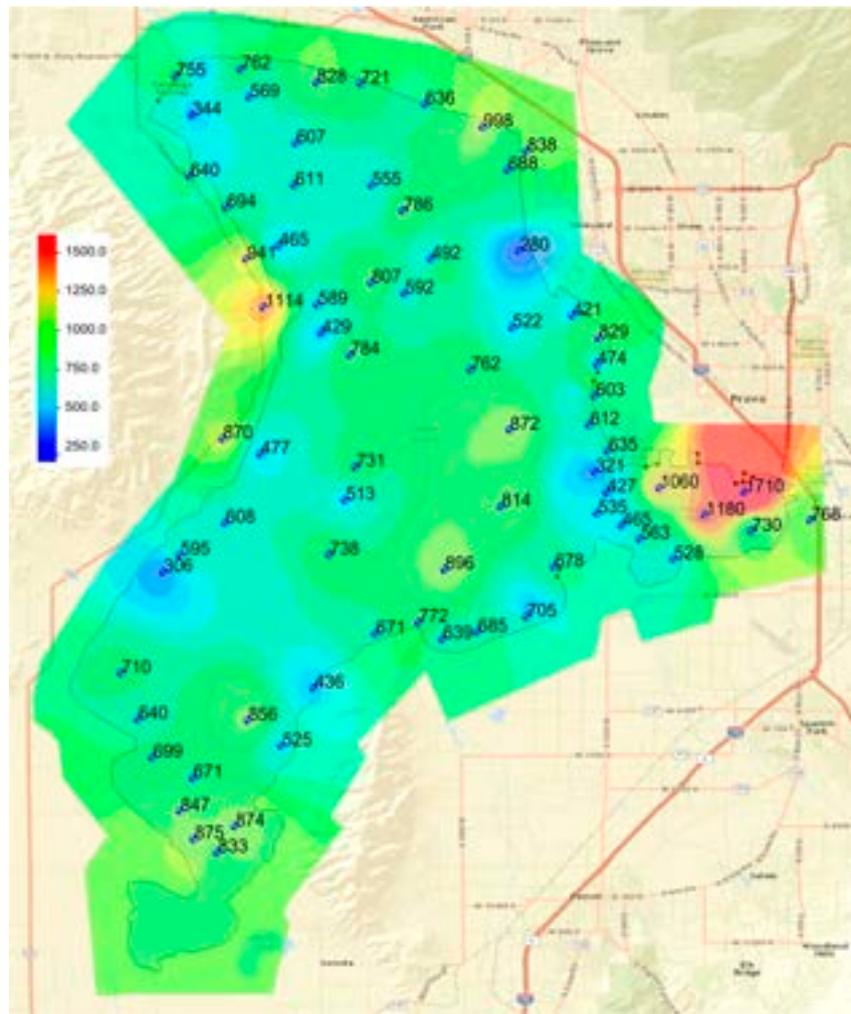


Figure 9. A geostatistical Model showing total phosphorous distribution in Utah Lake, developed by interpolating among the sediment sample points inside the lake as well as samples taken on the shore. The numbers indicate total phosphorous concentrations at the sample points. Both the colors in the legend and the numerical values represent total phosphorous amounts in ppm, and the map is not to scale. This map shows that the total phosphorous distribution appears to be continuous from the current lake sediments to the historic sediments. This indicates that phosphorous levels historically are similar to current values.

We visually evaluated the two maps to determine if it appeared that the spatial phosphorous distribution inside the lake was visually different from the distribution on the shore—that is, did the distribution appear to be continuous or did there appear to be some form of discontinuity between the lake distribution and the shore samples? Figure 9, which is based on both in-lake and on-shore samples, shows that there are no discontinuities at the shore line. In fact, the phosphorous distribution seems to naturally extend from the lake to the geologic sediments on-shore with no visible anomalies. This provides additional evidence to support the hypothesis that recent processes have had a minimal impact on the spatial distribution of phosphorous in current or geologic lake sediment phosphorous levels.

Figures 8 and 9 do present some interesting spatial patterns. First, note the location of the samples representing the low values in the lake. One sample with a value of 280 ppm, located on the eastern shore in Section 2 (middle north), was taken because this is the approximate location of the inflow from

the Orem Wastewater treatment plant, and we expected this area to have elevated phosphorous levels. As noted above, the locations of all the low-level phosphorous samples occur in areas of Utah Lake known to have seeps and springs. By observation, we found the physical properties of these samples were also different. Most lake sediment samples felt like clay and were smooth and sticky. The samples with low phosphorous levels were more granular and had a sandy feel. During sampling in 2015, these sample locations were in Utah Lake; however, in 2016, Utah Lake was at historic low levels and some of these sample locations were exposed. We visited the sample location near the outfall of the Orem Treatment plant, and found that seeps and springs were extensive in this area. We attribute low phosphorous levels in these samples to ground water removing the water-soluble, salt-extractable, and iron-sorbed phosphorous fractions.

Figures 7 and 8 also show areas with higher than normal phosphorous concentrations, the main one being Provo Bay on the southeastern portion of the lake. Both maps show sediments with very high phosphorous concentrations on the north shore of Provo Bay. This bay is a shallow wetland and contains a feedlot on the east side of the bay on the banks of Hobble Creek and receives the influent from the Provo City WWTP in the northeast corner. These locations are very near where the highest samples are located. This feedlot or corral has existed for at least 40 years and probably dates to early settlement in the area, which is the late 1800s. The State of Utah calculated phosphorous loads from the Provo and Orem city WWTPs as 71.6 and 79.3 tons/year, respectively. These are the two largest phosphorous sources for the lake.

We attribute the high phosphorous levels in this area to runoff from this cattle operation. Input from the Provo City WWTP most likely also impacts these sediment levels, but we do not see similar high values near the discharge of the Orem City WWTP, which releases a similar phosphorous load. There is one land sample that has a high phosphorous concentration located on the western shore of the lake. It is likely that this sample was also impacted by livestock operations, though we do not have evidence to that effect. While we have no direct evidence, there are some faint indications of historic corrals and fences in the area.

The spatial distribution of phosphorous concentrations indicates that patterns extend from the current in-lake sediments to the historic lakebed geologic sediments. This implies that the current lake processes are having a minimal impact on phosphorous contents. The distributions indicate spatial correlation among the samples with high phosphorous content. We attribute these high values to livestock operations. This is based on significant evidence in Provo Bay and general inference for the sample on the eastern shore.

5. Phosphorous Release Experiment

5.1. Lab Experiments of Phosphorus Release

To examine the potential for sediment phosphorous contributions to the water column, we conducted two lab experiments. We designed these experiments to provide indications of potential phosphorus contributions and do not present these experiments as good models for actual lake processes. However, they do indicate that lake sediments could be a significant source of phosphorous to the water column.

For these experiments, we used a shore sediment sample and a lake sediment sample. Both samples were stored in Ziploc® bags in a refrigerator for about a week prior to the experiment. The samples were divided into four lake samples and four shore samples, and placed into eight beakers. We covered the samples with two inches of water, four samples with distilled water containing salt (two sediment and two shore samples) and four samples with Provo River water (two sediment and two shore samples). At this point, there were four bottles for each sediment type, two with distilled-salt water and two with Provo River water. After adding the water, we manually mixed one of each combination once every three days. The distilled salt water experiment continued for a period of one month, while the Provo River water experiment extended for two weeks. The other beakers were not

disturbed during these periods. At the end of the experiments, we took water samples, filtered them through a 0.45 micron filter, and measured phosphorous levels using an ICP (Thermo Scientific iCAP™ 7400 ICP-OES Analyzer) at the BYU Environmental Lab. The lake water was meant to be an analog for the water column in Utah Lake. The salt water was to determine if salt-extractable phosphorous was available. We did not try to replicate reducing conditions. Shaking the bottles was an analog for sediment resuspension. Again, we were not trying to model actual lake conditions, only determine if these release mechanisms were feasible and determine approximate magnitudes.

5.2. Lab Experiments Results

Table 4 shows the results of the experiments. All results are in mg/L. Water samples were taken after the suspended sediment was allowed to settle for one to two days, then filtered on a 0.45 µm filter. This was done so we only measured the dissolved phosphorous content, not phosphorous sorbed on suspended solids. The initial concentration of phosphorus in Provo River water was 0.05 mg/L before being added to the beakers.

Table 4. Laboratory experimental results.

Solution Type	Total Phosphorous (mg/L)			
	Land Soil Not Shaken	Land Soil Shaken	Lake Sediment Not Shaken	Lake Sediment Shaken
Provo River Water	2.60	5.10	0.24	0.24
Distilled Water and Salt	7.70	14.0	3.00	19.0

Several trends are apparent from these results, more phosphorous is released with salt water than with Provo River water, and more phosphorous is released from the land samples than from the lake sediment samples. In addition, as expected, resuspended (shaken) sediments released more phosphorous. The salt water releases significantly more phosphorous from the sediment than the Provo river water. For the on-shore sediments, the salt water extracts about three times as much phosphorous as the Provo River water. For the lake sediments, this ranges from 10 times for the non-disturbed samples to 100 times for the shaken samples.

Significantly more research is required to determine the actual mechanisms responsible for these differences; however, we can hypothesize as to the processes that could cause these results. Salt water could extract both the phosphorous in the interstitial water and the ion-sorbed phosphorous, while the Provo River water probably does not extract much of the ion-sorbed material. Shaking the beakers simulates mechanical resuspension of the sediments. Without resuspension, even if phosphorus is released into the pore water of the sample, it must diffuse through the sediment to the sediment surface in the beaker to be released. Shaking eliminates the need for the diffusion.

For the lake sediments with Provo River water, there was essentially no difference between the shaken and non-disturbed samples. While the water concentration is lower, it is still about five times that of the river water alone. We hypothesize that the interstitial water in the lake sediments quickly comes to equilibrium with the Provo River water. This could indicate that the water-soluble fraction is smaller in the current lake sediments. Even though one beaker was not shaken, the sediments were mixed with the water when the beaker was initially filled. The beakers were not air tight, and there was significant head space in the beakers. We do not think the water was ever anaerobic, meaning that even if the sediments became anaerobic, the phosphorous would have precipitated with the iron when it came into contact with the oxidizing conditions in the water above the sediments.

The land samples with Provo River water had significantly higher phosphorous concentrations than the lake sediments. We attribute this to significantly higher concentrations of water-dissolvable phosphorous in this soil. The agitated land sample had about twice the concentration of the non-shaken sample. We think this may be due to diffusion in the non-shaken sample.

The salt water samples showed significantly higher phosphorous levels. The values for the shaken land and sediment samples were similar, 14 and 19 ppm, respectively. The non-shaken land and sediment samples were 7.7 and 3.0 ppm, respectively. This implies that there is a significant ion-bound (or sorbed) fraction of phosphorous available in both the land and lake samples and that the amounts are similar. Again, we attribute the smaller concentrations in the non-shaken samples to diffusion, which limits the release.

6. Discussion

The sampling results show that Utah Lake sediments have large concentrations of total phosphorus with an average total phosphorous concentration of 666 ppm. The concentrations in historic geologic sediments are similar, but higher, with an average total phosphorous concentration of 786 ppm. Simple laboratory experiments implied significant amounts of this phosphorous exists in water soluble or salt-extractable (sorbed) fractions that are easily released to the water column.

Phosphorous balance results show that Utah Lake acts as a phosphorous sink, and significantly more phosphorous enters the lake than leaves. However, statistical analysis shows that phosphorous content in different areas of the lake are not significantly different from each other. More importantly, phosphorous levels in current lake sediments are not statistically different than total phosphorous levels for geologic sediments (land samples). This implies that phosphorous inflows to Utah Lake, though large, may not be significant compared to existing phosphorous levels. Additional studies would be required to determine how inflows affect water column phosphorous levels and fluxes from sediments. The northeast portion of Provo Bay is the location of the three highest samples. They have total phosphorous content of 1060, 1180, and 1710 ppm, with the highest value near the inlets of Mill Race Creek, which receives water from the Provo WWTP and Hobble Creek, which has a small feedlot near the shore of the bay. Samples near the outfall of the Orem City WWTP did not have similar high values. The sample nearest the Orem City plant outfall (shown on upper east shore of Utah Lake in Figures 8 and 9) had one of the lower values of total phosphorous at 280 ppm. However, we attribute this low value to an area of groundwater seeps and springs into the lake.

We showed, using statistical analysis and spatial maps, that geologic sediments and current sediments have total phosphorus levels that are not statistically different from each other. Thus, the current phosphorous load coming into the lake does not seem to significantly impact sediment phosphorus levels and may not have large impacts on phosphorous concentrations in the water column. As noted, this may not be the case in the northeast corner of Provo Bay where it appears that anthropogenic sources have had impacts on sediment phosphorous concentrations.

The water bottle experiments were not quantitative in regards to how reservoir and land sediments affect water column phosphorous levels, but they do show that lake sediments are potential phosphorous reservoirs for the water column. These experiments imply that the concentration of phosphorous in Utah Lake waters could potentially be partially governed by water–sediment interactions, rather than tributary inflows. The bottle tests of the historic sediments indicate that overland flow and the associated erosion and short-distance dust transport could be a significant contributor to phosphorous inflows. Interestingly, the amounts of phosphorous in the sediments from erosion are similar to existing lake sediments but seem to have more phosphorous in the fractions that are easier water extractable. We did not run an anaerobic experiment to evaluate the iron-sorbed fraction, but we expect another large reservoir of phosphorous in this fraction.

This research documents the large phosphorous reservoirs in both current and historic lake sediments and highlights the need to determine actual phosphorous loadings to Utah Lake from these sources. This work demonstrates the need for additional research to understand and quantify the processes and rates of phosphorous flows into and out of the sediments, and quantify loads from surface runoff and associated erosion and atmospheric loadings.

7. Conclusions

This report characterizes phosphorous levels in current and geologic Utah Lake sediments. Statistically, the geologic lake shore sediments are not different from current lake sediments. This implies that, while Utah Lake accepts a significant phosphorous inflow and acts as a phosphorous sink, this probably does not significantly affect sediment phosphorous levels and that these sediments could have significant impact on water column phosphorous concentrations.

Simple bottle tests showed that the phosphorous in the sediments could easily be released to the water column. These facts, combined with Utah Lake physical properties (shallow, carp content, thick sediments, and wind) that cause significant sediment resuspension, increase the probability that the phosphorus in the sediment significantly impacts lake water quality.

Phosphorus release from sediment occurs under very complicated processes under many different conditions. Thus, more knowledge is needed to develop a conclusion for the phosphorus release mechanism in Utah Lake and the relative impacts of historic high phosphorus levels in Lake Sediments versus the current high inflow loadings.

Future work should better quantify sediment release rates, provide a better understanding of how the phosphorous is stored in the sediments, and improve geochemical models for how phosphorous moves between sediments and the water column. It should evaluate loads from overland flow and associated erosion and from short-distance dust transport. This work is needed, as historic lake sediments on the shoreline have high phosphorous concentrations, which are based on our simple bottle experiments, may be easily transferred to the water column.

The main management issue to determine how these large phosphorus concentrations in lake and historic sediments affect phosphorous concentrations in the water column and the relationship with external loads. Research needs to evaluate if this a buffered system, in regards to phosphorous, where changing external loads have little impact on the lake phosphorous levels. Research needs to quantify the potential and actual total phosphorous flux from the sediments to the water column and what factors influence those flux rates. Example research topics include quantifying fluxes from sediment resuspension due to wind, boat, and carp actions along with quantifying potential releases from anaerobic conditions. These fluxes need to be estimated and compared to known loadings for the lake. Current estimates show that Utah Lake is a sink for phosphorous, with over 60% of the estimated load remaining in the lake. This implies that phosphorous geochemistry of the sediment–water system needs to be better understood. Research questions such as “are the sediments and water column in near equilibrium conditions” need to be answered. This is important because if this system is in a near equilibrium state, the phosphorous reservoir in the sediment is very large and external loadings may have little impact on water phosphorous concentrations in the lake. Research is required to determine if phosphorous concentrations in the water column are relatively stable and buffered by the sediment phosphorous reservoir or closely tied to phosphorous inflows.

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Conflicts of Interest: The authors declare no conflict of interest.

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RESEARCH ARTICLE

Sediment potentially controls in-lake phosphorus cycling and harmful cyanobacteria in shallow, eutrophic Utah Lake

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Abstract

Lakes worldwide are impacted by eutrophication and harmful algal or cyanobacteria blooms (HABs) due to excessive nutrients, including legacy P released from sediments in shallow lakes. Utah Lake (northern Utah, USA) is a shallow lake with urban development primarily on the east side of the watershed, providing an opportunity to evaluate HABs in relation to a gradient of legacy sediment P. In this study, we investigated sediment composition and P concentrations in sediment, pore water, and the water column in relation to blooms of harmful cyanobacteria species. Sediments on the east side of the lake had P concentrations up to 1710 mg/kg, corresponding to elevated P concentrations in pore water (up to 10.8 mg/L) and overlying water column (up to 1.7 mg/L). Sediment P concentrations were positively correlated with Fe₂O₃, CaO, and organic matter abundance, and inversely correlated with SiO₂, demonstrating the importance of sediment composition for P sorption and mineral precipitation. Although the sediment contained <3% Fe₂O₃ by weight, approximately half of the sediment P was associated with redox-sensitive Fe oxide/hydroxide minerals that could be released to the water column under reducing conditions. Cyanobacteria cell counts indicate that blooms of *Aphanizomenon flos-aquae* and *Dolichospermum flosaquae* species tend to occur on the east side of Utah Lake, corresponding to areas with elevated P concentrations in the sediment, pore water, and water column. Our findings suggest that shallow lake eutrophication may be a function of P in legacy sediments that contribute to observed HABs in specific locations of shallow lakes.

Introduction

Harmful algal or cyanobacteria blooms (HABs) remain a substantial water quality concern as reoccurring blooms impact human health, cause a decline in lake recreation value, and create ecological problems, especially in shallow lakes that are prone to water level fluctuations and eutrophication. HABs are triggered by increased levels of total lake phosphorus (P) and

nitrogen (N) from anthropogenic nutrient pollution [1]. HABs are especially problematic in unstratified shallow lakes because of strong interactions between the water and land, atmosphere, and sediment [2–4]. However, many HAB dynamics remain elusive, such as the amount of total lake P necessary to produce blooms in response to external P loading or internal P release from legacy sediments [5]. Internal P fluxes from sediments to the overlying water column often result in time lags for restoration of shallow lakes after reduction in external nutrient loads [6–10].

Shallow lakes are prone to P release given high surface area to volume ratio, making sediment-water interactions a key process in dissolved P exchange [4, 8]. Sediments in shallow lakes act as a net P sink [11] but may serve as a temporary P source depending on the physicochemical properties of the sediments and overlying water [12]. P release is regulated by the interactions among dissolved oxygen, pH, temperature, microbial activity, and pore water P concentrations [11, 13]. Reducing conditions in sediment drive reductive dissolution of Fe (oxy)hydroxide minerals, releasing Fe oxide-bound P from the sediment [14]. Loosely bound P in sediments or dissolved P in pore water become available after resuspension by carp bioturbation or wind [15, 16]. These chemical and physical processes underscore the need to understand P speciation in sediments and interactions with water chemistry.

The interconnectedness of P in sediments and the water column may influence HABs. The microbial species responsible for the deleterious effects of HABs are due to cyanobacteria since cyanobacteria alone produce cyanotoxins, which are the primary water quality concern. To bloom, certain cyanobacterial species may exploit the relatively cooler temperatures in spring and/or shady versus sunny conditions [17]; others may generate their own nitrogen as N-fixers and outcompete species relying on inorganic N [18]; and still others may rely on different forms of P (i.e., soluble reactive P, organic P, mineral occluded P) [19]. Often it is a combination of conditions that cause certain cyanobacteria to bloom, but these conditions are consistently linked to some form of P. For example, modeling of distribution of six cyanobacterial species in five shallow eutrophic lakes over two years, the abundance of five cyanobacteria (two *Planktothrix*, two *Aphanizomenon*, and an *Anabaena* species) were linked to total P, and two species (*Aphanizomenon* and *Anabaena* species) to soluble reactive P in the water column [17]. The exact amount of total lake P to elicit blooms varies substantially due to the internal storage and cycling of P in sediments. Thus, with the potential for shallow lakes to internally cycle P and the sensitivity of cyanobacteria to different forms of P, the interconnectedness of P in sediments and water column may dictate blooms.

In this study, we investigated P concentrations and speciation with potential links to HAB production in Utah Lake, a shallow, eutrophic, freshwater lake experiencing lake closures in recent years due to HABs. Utah Lake is the third largest freshwater lake in the western U.S. with an area of 375 km² but an average depth of only 3 m (maximum depth of 6 m) under normal lake water levels. Nearly all the urban development is located on the east side of the watershed, providing an opportunity to evaluate HABs in relation to a gradient of P concentrations in the water column and legacy sediments between the east and west sides of the lake. To understand the linkages between P and HABs, our study was undertaken to: (1) evaluate P concentrations in sediment, pore water, and the water column; (2) characterize sediment composition, P speciation, and potential for P release; and (3) compare P spatial variability with counts of three common cyanobacterial species creating HABs in Utah Lake. Further, to investigate the impacts of urban nutrient inputs, special attention was given to sediments near the wastewater treatment plant (WWTP) outfalls occurring on the east side of Utah Lake.

Materials and methods

Study area

Utah Lake (Fig 1), a remnant of Pleistocene Lake Bonneville, is located in rapidly urbanizing Utah Valley, with a population >500,000 on the east side of the lake that is expected to double by 2050. The lake is popular for recreation but was closed during portions of 2016 and 2017 due to the presence of HABs. As a basin bottom lake in a rapidly urbanizing area, the lake receives nutrients from agricultural runoff, wastewater effluent, natural P in the local geology, and atmospheric deposition [20–22]. Seven WWTPs discharge into the east side of Utah Lake and waste streams are only expected to grow with increasing population. In particular, Provo Bay (Fig 1) receives effluent from three WWTPs. The majority of surface water inputs occur on the east side of the lake, including the Provo and Spanish Fork Rivers (Fig 1). Sedimentation rates average about 1.4 mm/yr—primarily from precipitation of carbonate minerals from the alkaline lake water [23]—potentially providing an important control on P concentrations in the water column by co-precipitation.

To combat eutrophication and HABs, the Utah Division of Water Quality has proposed a new rule to limit P in WWTP effluent to 1 mg/L by 2020 [24] and ultimately to 0.1 mg/L, at a capital cost ranging from \$24 million to over \$1 billion statewide depending on the level of treatment [25]. Given the legacy loading of P to Utah Lake, it is unclear whether this increased treatment would lead to immediate or long-term improvements in water quality.

Water and sediment sampling

To evaluate the potential for P cycling in Utah Lake, we measured P concentrations in three interacting lake compartments: sediment, pore water, and the water column. A total of 26 co-located samples of all three compartments were collected from 15 sites across Utah Lake during October 2015 (“A”), May/June 2016 (“B”), August 2016 (“C”), and November 2016 (“D”), with an emphasis on sediment near the effluent of WWTPs on the east side of the lake (Fig 1). Sample sites 1, 4, 11–13, and 15 (“east”) were located near WWTP inputs and sample sites 2–3, 5–10, and 14 (“west”) were located in the main body of the lake. Not all sites were visited during each sampling event, resulting in thirteen samples from both the east and west side of the lake. While the east versus west groupings are somewhat arbitrary, we found that it was the most useful way to separate the sites based on proximity to urban nutrient sources on the east side of Utah Lake. Lake sediments were collected using a stainless-steel Ekman dredge to isolate the top 5 cm of sediment, and pore water was extracted by centrifugation. Water column samples were taken from the middle of the water column using a peristaltic pump with FEP tubing. No specific permission was required for the field work because Utah Lake is located on public state land and the sampling did not include endangered or protected species.

Sediment mineralogy and chemistry

To characterize sediment composition, we measured TP, organic matter, carbonate abundance, mineralogy and trace/major element chemistry in all samples ($n = 26$). TP was analyzed by total digestion using USEPA Method 3052. Briefly, 8 mL concentrated HNO_3 and 2 mL concentrated H_2O_2 were added to 0.1 g of sediment and microwave digested at 180°C for 15 min for TP analysis by Thermo Scientific iCAP 7400 ICP-OES. Organic matter and carbonate abundance were determined by mass loss on ignition (LOI) after combustion in a muffle furnace at 550°C and 1000°C, respectively, for 4h [26]. Mineralogy was determined using a Rigaku MiniFlex 600 x-ray diffraction (XRD) instrument. XRD patterns were evaluated using Rietveld methods in the Rigaku PDXL2 software using crystallographic information files

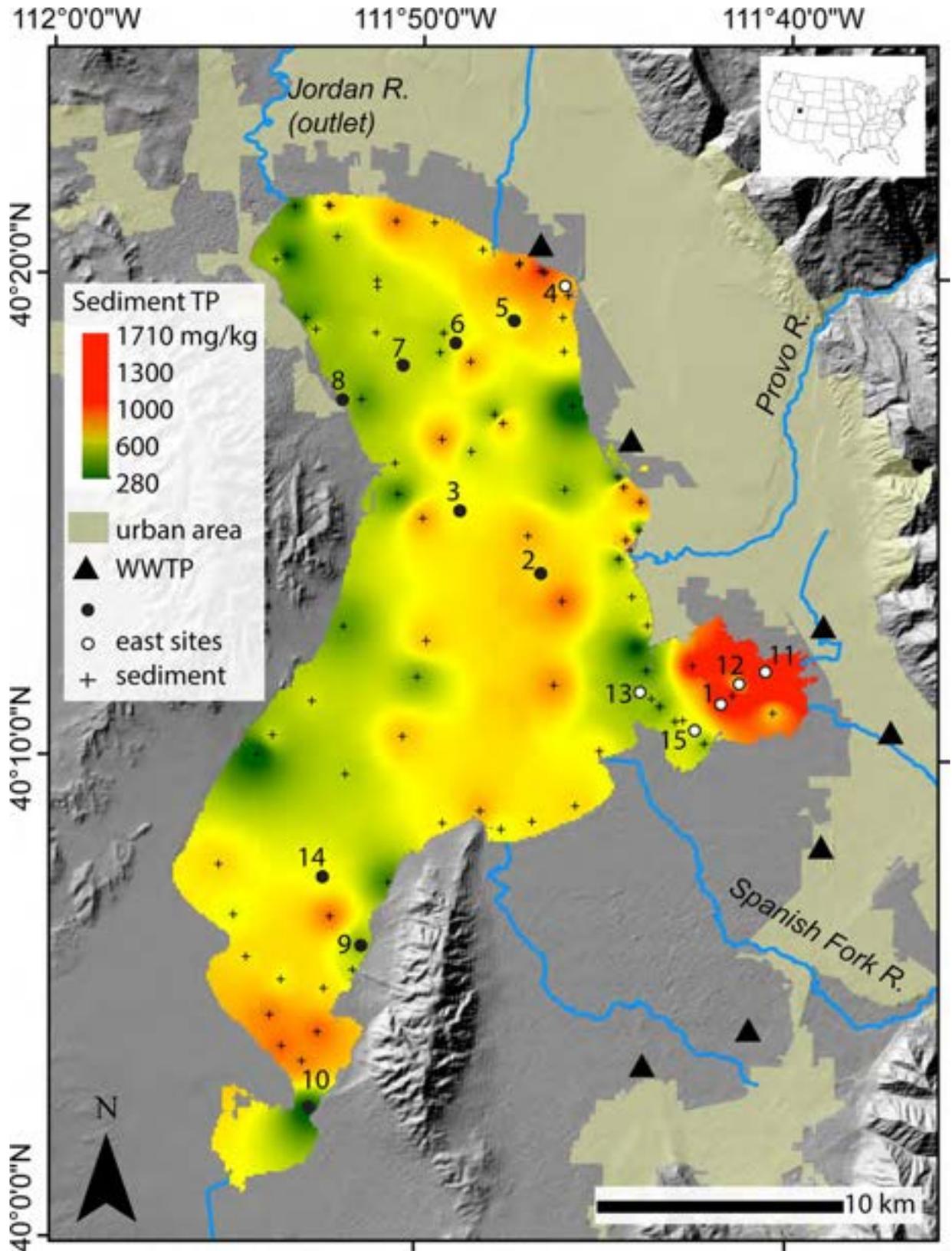


Fig 1. Location map of Utah Lake. Map shows interpolated 2015–2016 total phosphorus (TP) concentrations in Utah Lake sediments, the Wasatch Front urban area, wastewater treatment plants (WWTPs), tributaries, and the Jordan River outlet. Sample locations identified with circles and number label were sampled for surface water, pore water, and sediment (black circle = west and white circle = east) while sample locations identified with a cross were sampled for sediment only by Abu-Hmeidan et al. [21]. The urban area is defined according to the 2010 US census.

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obtained from the American Mineralogist Crystal Structure Database. Minerals with <3% abundance were excluded, and mineral abundances were normalized to 100% after adding organic matter from LOI 550°C measurements. An example XRD pattern is provided in the Supporting Information (S1 Fig). Major and trace element concentrations in sediment were analyzed by x-ray fluorescence (XRF) with a Rigaku Primus wavelength dispersive instrument on samples that were powdered, compacted into pressed pellets, and melted into glass disks. The XRF provided major element oxide weight percentages (SiO₂, TiO₂, Al₂O₃, Fe₂O₃, MnO, MgO, CaO, Na₂O, K₂O, and P₂O₅) and trace elements concentrations (Ba, Ce, Cl, Cr, Cu, F, Ga, La, Nb, Nd, Ni, Pb, Rb, S, Sc, Sm, Sr, Th, U, V, Y, Zn, and Zr). Oxide weight percentages were recalculated after adding carbonate abundance from LOI 1000°C measurements. Scanning electron microscope (SEM) images were taken for sample 1-A (collected from the middle of Provo Bay) to provide an in-situ and non-invasive view of elemental composition at the sediment grain surfaces. Scanning images and dot maps of Ca, Fe, and P were created using energy dispersive spectroscopy capabilities of the XL30 FEI environmental SEM. Raw TP concentrations and XRF/XRD data are provided in the Supporting Information (S1 and S2 Tables, respectively).

Sequential extraction of sediment samples

Sediment samples from May–June 2016 ($n = 10$) were subjected to a sequential extraction procedure to investigate P speciation in lake sediments. Sediment P species were determined using a sequential extraction scheme for calcite rich lakes from Hupfer et al. [27], modified after Psenner et al. [28], on 2.5 g of wet sediment. Extraction steps included: 1) 1 M NH₄Cl (deoxygenated–N₂ purged) shaken for 0.5h to extract P in pore water and loosely adsorbed to surfaces; 2) 0.11 M BD (bicarbonate/dithionite-buffered to a pH 7 using NaHCO₃) shaken for 1 h to remove redox-sensitive P mainly bound to oxidized Fe and Mn compounds; 3) 1M NaOH shaken for 16 h to remove P exchangeable against OH⁻ ions and P in organic matter; 4) 0.5 HCl shaken for 16 h to remove P in calcium phosphate minerals and acid-soluble organic P; and 5) 1 M boiling HCl for 0.25 h after a 550°C ignition for 2 h to evaluate the refractory organic P and nonextractable mineral P. The sediment was rinsed with 1 M NH₄Cl between each extraction steps. After each extraction step, the supernatant (including the 1 M NH₄Cl rinse) was filtered through a 0.45-μm nylon filter and analyzed for TP by ICP-OES. Raw data from the sequential extraction experiments are provided in the Supporting Information (S3 Table).

Pore water and water column chemistry

To evaluate the relatively mobile form of P in Utah Lake, we measured total dissolved P (TDP; defined as the P fraction passing through a 0.45 μm filter) and trace/major element chemistry in pore water and water column samples. Filtered water samples were analyzed for TDP and other trace/major elements (As, Ca, Ba, Fe, K, Mg, Mn, Mo, Na, Si, Sr, and V) by ICP-OES. Major anions (F⁻, Cl⁻, NO₃⁻, HPO₄²⁻, SO₄²⁻) were analyzed using a Dionex ICS-90 ion chromatograph. Alkalinity (assumed to be HCO₃⁻) was measured on water column samples by titration. Major ion charge balances were within an acceptable charge balance error of ±5% for

all water column samples (S4 Table). Raw water column and pore water chemistry data, including major ion and trace element concentrations, are provided in the Supporting Information (S4 and S5 Tables, respectively).

HAB species abundance

To identify the potential influence of P on HABs, we evaluated the abundance of total cyanobacteria and three dominant cyanobacteria species in relation to P concentrations across Utah Lake. Total cyanobacteria and specific species were counted via semi-automated imaging flow cytometry (Phycotech Inc.). All samples were collected during the summers of 2016 and 2017 from the upper 0.5 m of the water column and preserved with Lugols solution prior to shipment for counting. Cell counts were collected as part of the HAB monitoring project carried out by the Utah Division of Water Quality and values are publicly available (<https://deq.utah.gov/Divisions/dwq/health-advisory/harmful-algal-blooms/bloom-events/index.htm>). Some of the highest cell counts were measured at the marinas (Lindon, Provo, and Lincoln Beach; S2 Fig) but they were excluded from our analyses because we lacked P data at these sites. The marinas are small, semi-closed systems that may not represent processes occurring in the main body of the lake or Provo Bay.

Statistics

To investigate relationships between TP concentrations and sediment composition (i.e., organic matter, CaO, SiO₂, Fe₂O₃), we conducted linear regression analyses using the *lm* function in R (<http://www.R-project.org>). Further, we tested for differences in TDP concentrations in the water column and pore water between the west and east side of Utah Lake with t-tests in R.

Results

Sediment total phosphorus highest on east side of Utah Lake

Sediment TP concentrations in Utah Lake ranged from 280 to 1710 mg/kg, with higher concentrations on the east side of the lake relative to the west side (Fig 1). To create the map shown in Fig 1, we combined our results with those of Abu-Hmeidan et al. [21], who collected 84 sediment samples over the same time period and analyzed TP in the same laboratory as our samples. Elevated TP concentrations (>900 mg/kg) were found in sediment near WWTP effluent, particularly in Provo Bay. Total P concentrations in sediment ranged from 300–900 mg/kg throughout much of the lake, even far from WWTP inputs, representing a moderate to high background in the lake sediments. Notably, parts of the east shore near the inlet of Provo River and Spanish Fork River had low sediment TP concentrations.

Relationship between sediment composition and TP concentrations

In addition to sample location, TP concentrations were partially controlled by sediment composition. Specifically, P₂O₅ concentrations were positively correlated with organic matter (measured as LOI 550°C) and CaO abundance, and inversely correlated with SiO₂ (Fig 2). The relationships were especially strong in the east side samples, with R² values ≥0.95 (F_{1,11} > 200, p < 0.001) for P₂O₅ versus LOI 550°C, CaO, and SiO₂. In contrast, the relationships between P₂O₅ and organic matter, CaO, and SiO₂ were significant but weaker for the west side samples (R² values ranging from 0.54 to 0.84, F_{1,11} < 60, p < 0.005). P₂O₅ was also positively correlated with Fe₂O₃ abundance, although this relationship was not significant for the east side samples.

The XRD analysis shows that sediments are primarily composed of calcite, dolomite, and quartz, with small amounts of organic matter (S2 Table). Calcite mass ranged from 12.7–

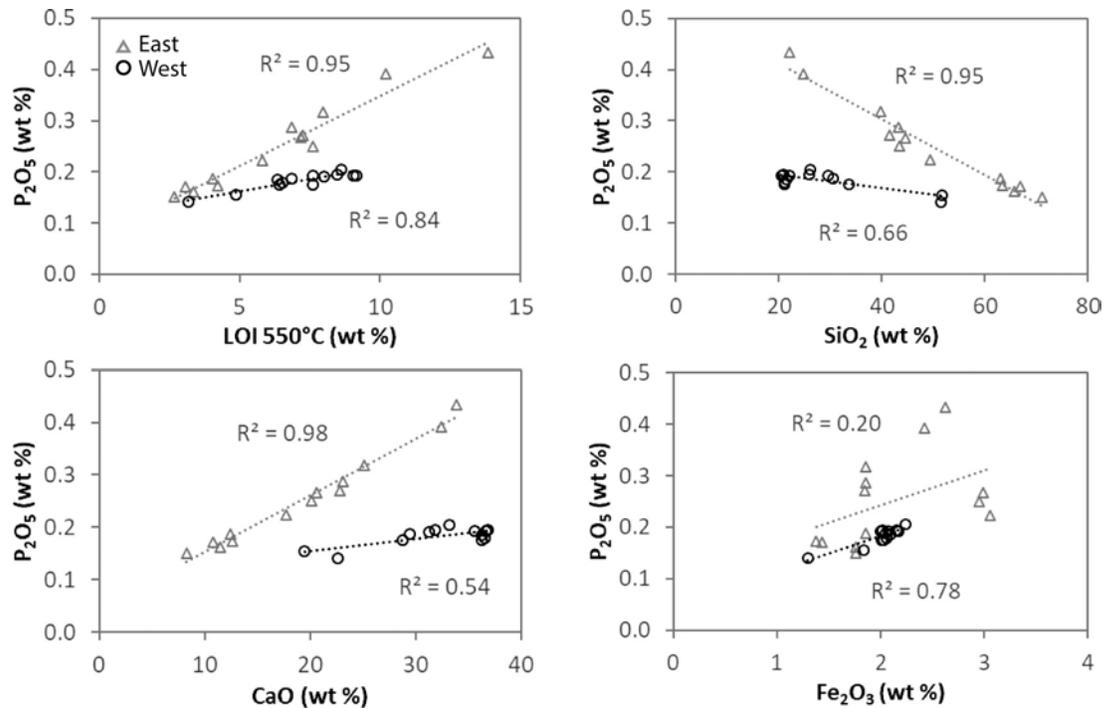


Fig 2. P_2O_5 abundance versus LOI 550°C, CaO, and SiO_2 . LOI 550°C is a measure of organic matter concentration. West samples ($n = 13$) were collected from sites 2–3, 5–10, and 14 and east samples ($n = 13$) were collected from sites 1, 4, 11–13, and 15.

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75.1% ($45.9\% \pm 4.1\%$; mean \pm standard error), dolomite from 2.0–29.8% ($7.8\% \pm 1.3\%$), and quartz from 10.3–77.0% ($39.5\% \pm 4.9\%$). Organic matter (from LOI 550°C measurements) ranged from 2.7–13.8% ($6.7\% \pm 0.6\%$). Peaks for clay minerals (illite, smectite, and kaolinite) and feldspars were small to absent, indicating low abundance, so they were neglected from the analysis. With the exception of clays and feldspars, our sediment mineralogy results are similar to values reported by Hogsett [29]. Notably, there was no clear evidence for aragonite in any of the samples. Likewise, Fe oxy/hydroxide minerals were in low enough abundance that they were not observed by XRD, which is not unexpected given that Fe_2O_3 comprised only ~1–3% of sediment mass based on the XRF data. Although Fe concentrations were low, the presence of Fe was confirmed by the SEM images, which showed the distribution of elemental Ca, Fe, and P in lake sediment (Fig 3).

Phosphorus speciation in lake sediments

Sediment P was primarily bound to oxidized Fe/Mn compounds (BD fraction) and Ca phosphate minerals or acid-soluble organic P (HCl fraction; Fig 4). On average, $49.1\% \pm 1.8\%$ of TP was associated with the BD fraction (range: 41–61%; $n = 10$) and $38.6\% \pm 2.1\%$ with the HCl fraction (range: 25–47%). Given low Mn concentrations in Utah Lake sediment (≤ 0.06 wt. %; S2 Table), the BD fraction likely represents P associated with Fe rather than Mn. Likewise, given that the sediments are calcite-rich, the HCl fraction is likely dominated by P from calcium phosphate minerals rather than organic P. The other three fractions, including NH_4Cl (loosely bound), NaOH (exchangeable P and P in organic matter), and the residual leach step (refractory organic P), accounted for an average of only 12.4% of TP in the sediment samples. Although concentrations were higher in the east samples, the percentage of P in each fraction was similar between the east and west samples.

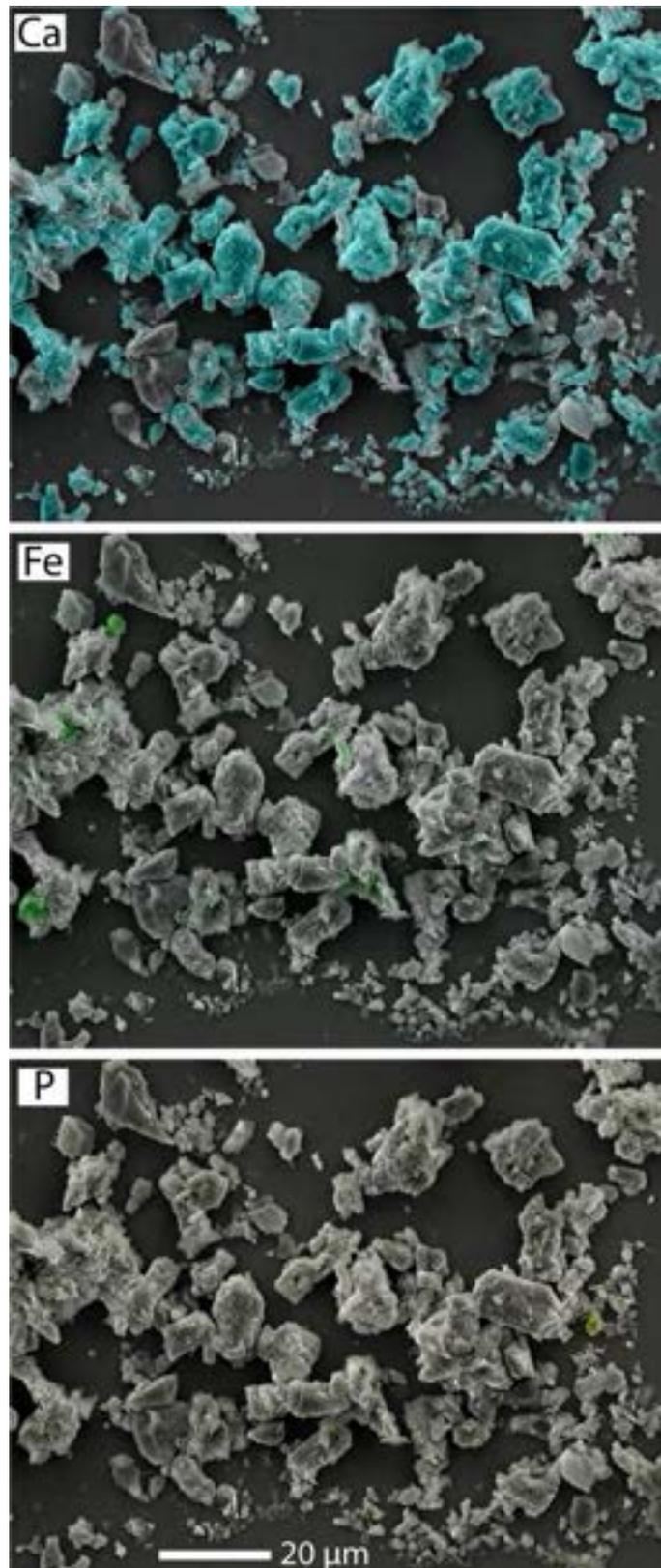


Fig 3. Backscattered SEM image of Utah Lake sediment. The sample is overlain with energy dispersive spectroscopy dot maps of Ca (blue), Fe (green), and P (yellow). Image was taken for sediment sample 1-A collected from the middle of Provo Bay in October 2015.

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Water column and pore water phosphorus concentrations

TDP concentrations in the water column and pore water were higher on the east side of Utah Lake near WWTP inputs relative to the west side (Fig 5). Concentrations in the east and west samples were significantly different for water column (t-test, $t = 2.66$, $p = 0.021$, $df = 12.0$) and pore water samples (t-test, $t = 3.65$, $p = 0.003$, $df = 12.4$). In the water column, TDP concentrations ranged from 0.04 to 1.74 mg/L in the east samples (0.43 ± 0.14 mg/L) compared with 0.03 to 0.09 mg/L in the west (0.05 ± 0.01 mg/L). Pore water TDP concentrations were approximately an order of magnitude higher than the water column concentrations and were higher on the east side relative to the west, ranging from 0.30 to 10.8 mg/L in the east samples (3.96 ± 0.85 mg/L) compared with 0.40 to 1.61 mg/L in the west (0.83 ± 0.11 mg/L).

HABs prominent on east side of the lake

HABs were prominent on the east side of Utah Lake (Fig 6) where P concentrations were elevated in the sediment, pore water, and water column. A map showing sample site information is provided in the Supporting Information (S2 Fig). In the summer months of 2017, total cyanobacteria cell counts were at least 2.4-times higher in the east ($143,244 \pm 27,128$) than west side ($60,638 \pm 18,155$) of the lake (Fig 6E). Two cyanobacteria species, *Aphanizomenom flos-aquae* and *Dolichospermum flosaquae* dominated the total cell counts, with additional minor contributions of *Microcystis aeruginosa*. *Aphanizomenom flos-aquae* cell counts were consistently 3.2-times higher in the east ($112,826 \pm 25,270$) relative to the west side ($57,703 \pm 17,117$;

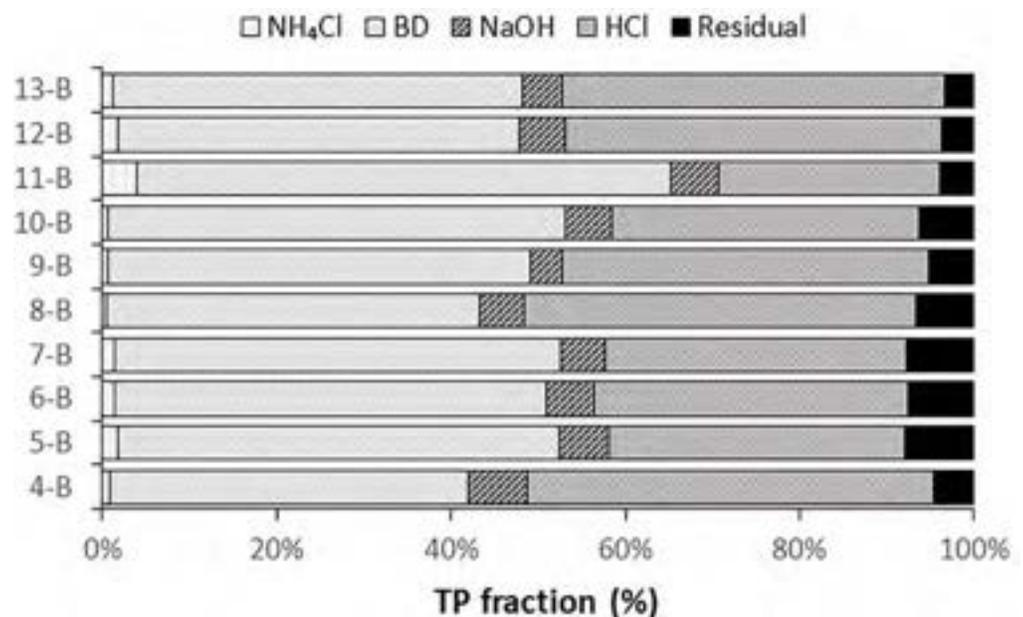


Fig 4. Sediment sequential extraction results. Extraction results for 10 sediment samples collected during May–June 2016. Results are expressed as % total phosphorus NH₄Cl = P in pore water and adsorbed loosely to surfaces, BD = redox-sensitive P mainly bound to oxidized Fe and Mn compounds, NaOH = P exchangeable against OH⁻ ions or bound in organic matter, HCl = calcium phosphate minerals, and the residual is refractory organic P and nonextractable mineral P.

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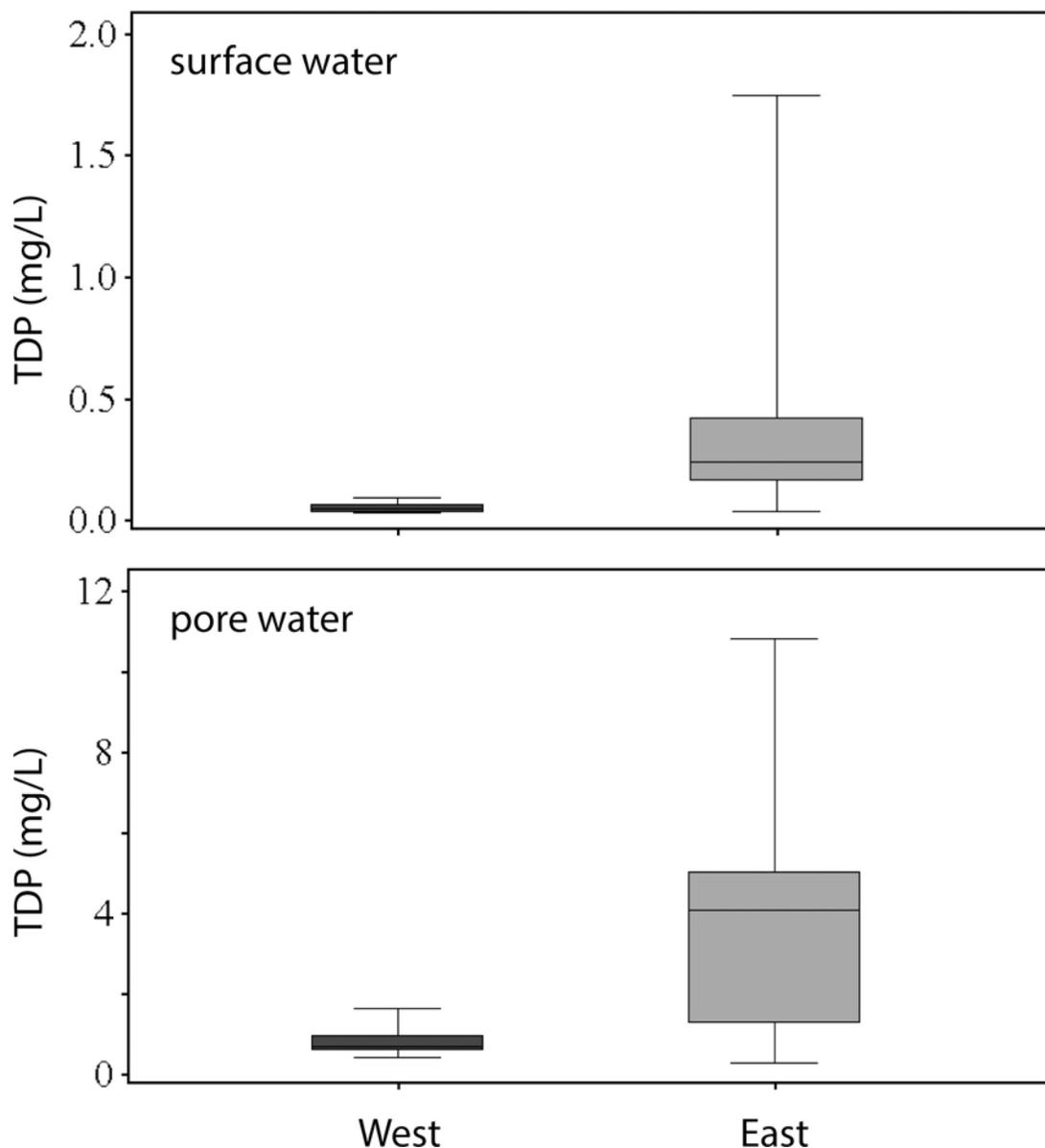


Fig 5. Total dissolved phosphorus (TDP) concentrations in surface water (top) and pore water (bottom). West samples ($n = 13$) were collected from sites 2–3, 5–10, and 14. East samples ($n = 13$) were collected from sites 1, 4, 11–13, and 15. For each sample type, TDP concentrations were averaged from the four sampling periods.

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Fig 6F), and along with *Dolichospermum flosaquae* contributed to blooms in July and August (Fig 6G). Alternatively, in early July of 2016, one immense bloom, predominantly of *Aphanizomenon flos-aquae*, occurred in three of the six east side locations where the cell counts were in the millions (34,193,162; 1,892,112; 1,418,070; Fig 6B). A bloom of *Microcystis aeruginosa* occurred along the east side of the lake later in the summer of 2017 in August (cell counts east = $7,211 \pm 2,590$; cell counts west = 910 ± 273). Blooms occurred infrequently in 2017 in west locations, but in 2016 one west side location experienced a minor bloom of *Microcystis aeruginosa* (5,631 cells) in late summer (Fig 6D).

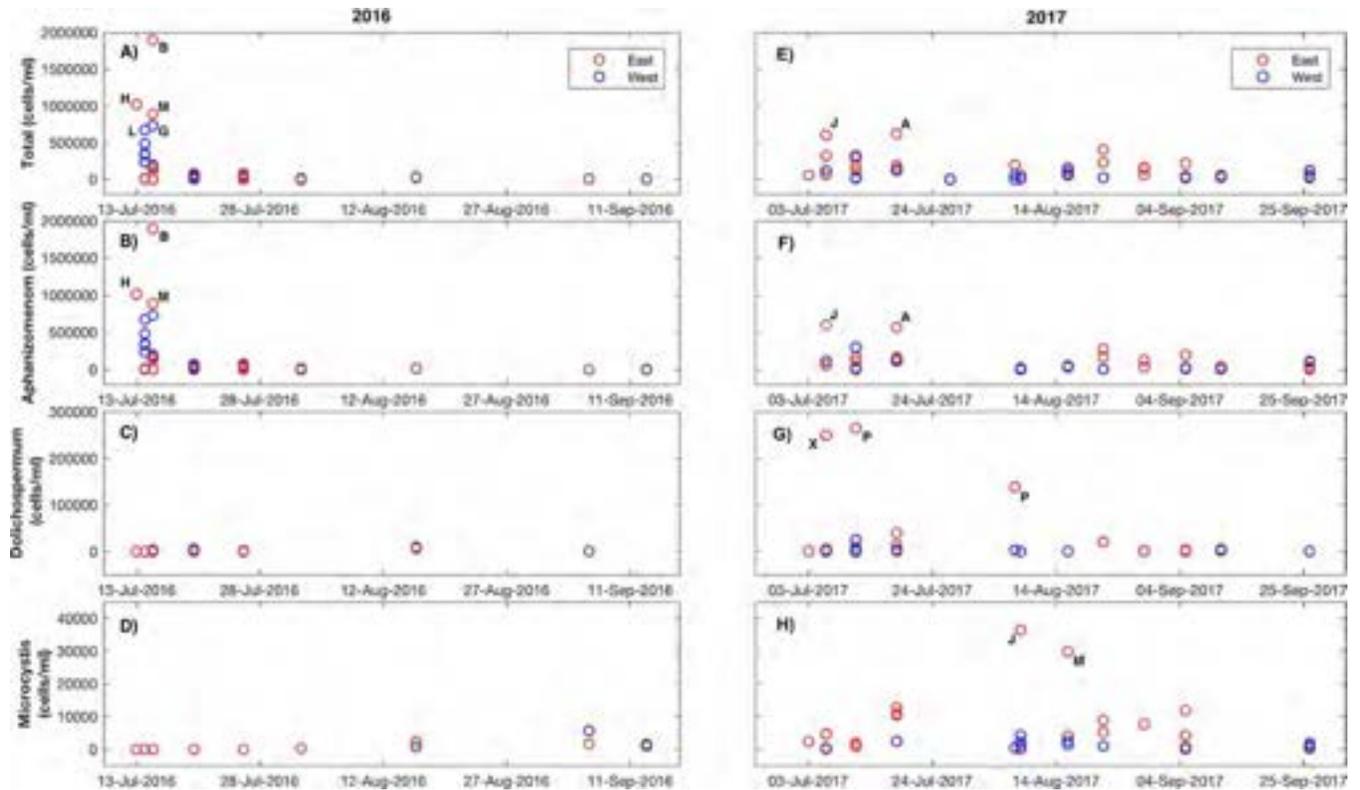


Fig 6. The abundance of total cyanobacteria in Utah Lake during summer 2016 and 2017. (A, E), *Aphanizomenon flos-aquae* (B, F), *Dolichospermum flosaquae* species (C, G), and *Microcystis aeruginosa* (D, H) cell counts for the summer of 2016 ($n = 46$) and 2017 ($n = 52$) across Utah Lake. Red circles and blue circles represent cell counts via semi-automated imaging flow cytometry from individual samples from the east and west side of the lake. Site labels correspond to locations shown in S2 Fig.

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Discussion

Spatial variability of phosphorus in Utah Lake

Phosphorus concentrations in pore water, water column, and sediment were generally highest on the east side of Utah Lake, reflecting legacy and current nutrient inputs from the urban area and WWTPs (Figs 1 and 5). Lake sediments contain legacy P from decades of nutrient inputs from the urban area, including WWTP effluent on the east side of Utah Lake and Provo Bay. Sediment TP concentrations in Utah Lake are similar to values reported for shallow, hypereutrophic Lake Taihu in China (range: 304–1553 mg/kg, $n = 30$) [30]. Elevated TP concentrations in pore water, which reflect elevated sediment TP concentrations, may contribute to P flux to the water column through diffusion or sediment resuspension. Notably, not all sediment on the east side of the lake contained elevated P concentrations. Sediment near the mouths of the Provo and Spanish Fork Rivers, which also coincide with WWTP outfalls (Fig 1), contained relatively low TP concentrations. The rivers likely contribute abundant silicate minerals, resulting in sediment with low capacity for P sorption (as indicated by the negative relationship between P and SiO_2 ; Fig 2), or the rivers have low P concentrations that dilute the P budget. These are also areas where groundwater with low P concentrations discharges to the lakebed [21].

Total P concentrations in sediments were a function of proximity to WWTP outfalls and the mineral content or sorption capacity of the sediments. Sediment mineralogy was dominated by quartz, dolomite, and calcite (S2 Table), varying in abundance based on the relative

inputs of detrital quartz and dolomite from tributaries and endogenic calcite that precipitates from the calcite-saturated water column. Organic matter, Ca, and Fe content control the amount of P retained in sediments through sorption or mineral precipitation (Fig 2). The role of organic matter in complexation of P is not clear since sequential leaching results showed a minimal amount of P in the organic matter (1M NaOH) fraction. The correlation may reflect co-located organic matter- and Ca- or Fe-rich sediments rather than interactions with organic matter. It is also possible that a fraction of the organic matter was coprecipitated with Fe(III) oxides and oxyhydroxides such that P in organic matter was released with the BD fraction rather than the 1 M NaOH fraction [31]. Notably, although Fe concentrations represented only a small fraction of total sediment composition (with Fe_2O_3 typically < 3% by weight; Fig 2), the sequential extraction results indicate that Fe retains approximately half of the sediment P by sorption to Fe (oxy)-hydroxides ($\text{Fe}(\text{OOH}) \approx \text{P}$) (Fig 4). Sorption of P by Fe-oxide minerals was further demonstrated by an SEM image of sediment from Provo Bay, which shows the association of P with Fe-rich sediment grains (Fig 3). In Fig 3, the colored dot maps indicate the presence of specific elements (blue for Ca, green for Fe, and yellow for P) where P is dispersed across Ca- and Fe-rich grains.

The association of P with different minerals affects the subsequent mobility in sediments and potential flux to the water column. Phosphorus was primarily associated with Fe ($\text{Fe}(\text{OOH})$) and Ca (CaCO_3 or $\text{Ca}_{10}(\text{PO}_4)_6(\text{OH},\text{F},\text{Cl})_2 \approx \text{P}$) from the BD and HCl fractions (Fig 4). These minerals act as sinks to sequester P from water column to the sediment [32, 33]. Coprecipitation of P with calcite and apatite minerals is strong at neutral to alkaline pH [34]. Utah Lake is an alkaline lake with pH values typically over 8 and is buffered by high bicarbonate concentrations. The Ca-associated P is likely precipitated with calcite ($\text{CaCO}_3 \approx \text{P}$) or apatite ($\text{Ca}_{10}(\text{PO}_4)_6(\text{OH},\text{F},\text{Cl})_2$) minerals. Although sediment chemistry is not kinetically conducive to precipitation of apatite, authigenic apatite mineral precipitation may occur as diatoms store polyphosphates inside their cells and form Ca phosphate minerals [35]. The majority of Ca-associated P may act as a permanent sink for P in Utah Lake with alkaline pH values. In contrast, the Fe-bound P pool is mobile as P may be released from sediments under anoxic conditions with the reductive dissolution of Fe-oxide minerals [36]. Anoxic conditions may develop locally near the sediment-water interface, particularly during summer when microbial activity is high. Although we did not evaluate seasonal differences in P fractionation in this study, it is possible that the Fe-bound fraction may decrease as P is released under summertime anoxic conditions. While flux rates from the sediment to overlying water column are unknown, the sediment-water interface is potentially a major controlling factor of P cycling in Utah Lake. Thus, quantifying P fluxes from sediment to overlying water is an important area for future study.

Elevated TDP concentrations in the water column and pore water were found on the east side of the lake, which also contained the highest TP concentrations in sediment. The relationship suggests that P flux from legacy sediments controls P concentrations in the pore water and water column or it reflects P movement from water to sediment. The sediments may act as a net sink or source depending on equilibrium P concentrations and changing geochemical conditions at the sediment-water interface. Notably, while sediment and water P concentrations were spatially related, there was less variability in sediment compared with water. Sediment concentrations varied six-fold across the lake (280–1710 mg/kg) whereas water column and pore water concentrations varied by nearly two orders of magnitude (0.07–1.9 and 0.3–10.8 mg/L, respectively). The greater variability in water concentrations may reflect biogeochemical processing in the water column and pore water. TDP concentrations were typically 2- to 4-times higher in pore water relative to the water column, likely because of reducing conditions in the sediment as P is released with reduced Fe [37].

Phosphorus may lay a foundation for HABs

The elevated levels of P in the east side of Utah Lake appear to lay a foundation primarily for two species, *Aphanizomenon flos-aquae* and *Dolichosperma flosaquae*, to contribute to HABs. *Aphanizomenon* and *Dolichosperma* species are often triggered by TDP in the water column of otherwise P-limited systems [38, 39]. Both species may utilize high levels of P to generate biomass and satisfy their stoichiometric requirements for growth by fixing atmospheric N. *Aphanizomenon flos-aquae*, dominate HABs when nitrate:soluble reactive P ratios are relatively low [19, 40]. Little is known regarding the conditions that promote *Dolichospermum flosaquae* growth. *Dolichospermum* species may co-dominate blooms with other *Aphanizomenon* species like *Aphanizomenon gracile* and may be triggered by similar lake chemical conditions (Wood et al. 2017). TDP and TP are not the only factors leading to blooms as seen in the difference in bloom characteristics from 2016 to 2017. Multiple factors are necessary for blooms to occur, including N concentrations and meteorological conditions, but it seems that legacy P generated from multiple WWTPs over decades plays a role. Future research needs to identify the meteorological conditions and chemical thresholds, including those associated with multiple forms of P, allowing cyanobacteria species like *Aphanizomenon flos-aquae* to bloom in Utah Lake.

Implications for lake restoration

Improvement in Utah Lake water quality would likely be delayed after a reduction in external P loads from WWTPs due to P release from lake sediments. Internal P cycling is commonly responsible for delayed response to decreasing external nutrient loads, with lakes typically reaching a new equilibrium within 10–15 years after nutrient reductions [6]. However, with high alkalinity, high pH, oxygenated and calcite-saturated waters, Utah Lake may represent the best-case scenario for a self-cleaning system to remove P from the water column by mineral precipitation in calcite. Ultimately the P-rich calcite may convert to thermodynamically stable apatite minerals and be further removed from the P cycle.

In addition to limiting external nutrient inputs, geo-engineering techniques could be considered for selected parts of Utah Lake to limit internal P cycling and HABs [41]. For example, a number of studies have shown field-scale success using lanthanum modified bentonite to limit P release from sediments [42]. Aluminum salts are another option for treating sediments but would likely not be effective in Utah Lake given the high pH and susceptibility of sediment resuspension [43, 44]. After carefully considering these and other factors, portions of the lake with elevated P concentrations in sediment, such as Provo Bay, could be targeted for P treatment. The combination of stepwise nutrient reductions and geo-engineering methods may be implemented as part of a cost-effective lake restoration program.

Conclusion

Understanding the fate and mobility of phosphorus (P) in shallow lake sediments is essential for evaluating P cycling and potential impacts on water quality. In this study, we analyzed the water column, pore water, and sediment in Utah Lake to determine primary P sinks and the potential for internal P cycling. Our results suggest that majority of P in sediments is associated with calcite or apatite and Fe oxy/hydroxide minerals. Whereas the calcite- and apatite-bound P fraction is likely immobile in the sediments, the Fe-associated fraction is potentially released from sediments under reducing conditions. Sediment, pore water, and water column had the highest P concentrations on the east side of the lake near urban inputs. Pore water concentrations were an order of magnitude higher than the water column, demonstrating the potential for high flux rates between the bottom sediments and overlying water. Blooms of harmful

cyanobacteria are also prevalent in areas of elevated P concentrations, indicating a link between legacy P pools and HABs. Our study has implications for water quality restoration plans of an increasing number of eutrophic lakes near developing cities, where nutrient pollution is critical.

Supporting information

S1 Fig. Representative XRD pattern for Utah Lake sediment showing prominent quartz and calcite peaks. This pattern is for sample 3-D collected in November 2016. The height of the mineral peaks represents the abundance of minerals present in the sample.
(TIF)

S2 Fig. Sampling sites for cyanobacteria cell count data shown in Fig 6.
(TIF)

S1 Table. Sediment total P concentrations measured after sample digestion.
(XLSX)

S2 Table. Sediment chemistry and mineralogy.
(XLSX)

S3 Table. Sediment sequential extraction data.
(XLSX)

S4 Table. Water column chemistry.
(XLSX)

S5 Table. Pore water chemistry.
(XLSX)

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Article

Measuring and Calculating Current Atmospheric Phosphorous and Nitrogen Loadings to Utah Lake Using Field Samples and Geostatistical Analysis

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Abstract: Atmospheric nutrient loading through wet and dry deposition is one of the least understood, yet can be one of the most important, pathways of nutrient transport into lakes and reservoirs. Nutrients, specifically phosphorus and nitrogen, are essential for aquatic life but in excess can cause accelerated algae growth and eutrophication and can be a major factor that causes harmful algal blooms (HABs) that occur in lakes and reservoirs. Utah Lake is subject to eutrophication and HABs. It is susceptible to atmospheric deposition due to its large surface area to volume ratio, high phosphorous levels in local soils, and proximity to Great Basin dust sources. In this study we collected and analyzed eight months of atmospheric deposition data from five locations near Utah Lake. Our data showed that atmospheric deposition to Utah Lake over the 8-month period was between 8 to 350 Mg (metric tonne) of total phosphorus and 46 to 460 Mg of dissolved inorganic nitrogen. This large range is based on which samples were used in the estimate with the larger numbers including results from “contaminated samples”. These nutrient loading values are significant for Utah Lake in that it has been estimated that only about 17 Mg year⁻¹ of phosphorus and about 200 Mg year⁻¹ of nitrogen are needed to support a eutrophic level of algal growth. We found that atmospheric deposition is a major contributor to the eutrophic nutrient load of Utah Lake.

Keywords: Utah Lake; phosphorus; nitrogen; atmospheric deposition; eutrophication; harmful algal blooms

1. Introduction

1.1. Study Goals and Importance

Nutrients have been the focus of considerable scientific research in relation to eutrophication and also the growth of harmful algal blooms (HABs) [1,2]. Nutrients can enter an aquatic ecosystem in a variety of ways: surface tributaries, storm and wastewater discharges, groundwater, and from the atmosphere. Of these different avenues, atmospheric deposition is commonly neglected during the formation of a nutrient budget as in most areas it is not considered a significant source. Recently, with the increased focus on eutrophication of water bodies by nitrogen (N) and phosphorus (P), studies have found that atmospheric deposition is an important part of the total nutrient budget for many aquatic ecosystems [3–6].

Shallow lakes such as Utah Lake might be even more susceptible to atmospheric deposition because of their large surface area to volume ratios. Despite these characteristics, atmospheric deposition is commonly neglected with respect to the nutrient budgets of Utah Lake [7]. The objective of this work is to: (1) quantify the atmospheric loading of total phosphorous (TP) and dissolved

inorganic nitrogen (DIN) to Utah Lake, (2) evaluate the spatial variability of TP and DIN loads over the surface of Utah Lake, and (3) characterize the temporal variability of TP and DIN loads among seasons.

This case study is important as it clearly demonstrates, with field-sample data, the importance and magnitude of atmospheric nutrient loadings. For Utah Lake, and we suspect for many other lakes and reservoirs, reducing nutrient loads from other sources may be costly and because atmospheric loads were neglected, may not produce the expected results. In the case of Utah Lake, we show that atmospheric loads can support a eutrophic level of algae growth.

1.2. Utah Lake Water Quality Background

There is increased interest recently in nutrient loadings to Utah Lake [8]. There is a controversy about the cost and effectiveness of reducing the large nutrient loadings from waste water treatment plants (WWTPs). There is agreement that the WWTPs contribute significant loads to Utah Lake, but there is disagreement on whether reducing or even eliminating these loads would significantly affect algal growth in Utah Lake. Currently phosphorous is not a limiting nutrient in Utah Lake and is present in excess. The debate is whether reducing WWTP TP loadings would be sufficient to reduce phosphorus loads such that they could be limiting to overall algal growth, as well as to limit HABs [7–9]. Many well-understood factors can be limiting for excessive algal growth and the occurrence of HABs. These include light levels, water clarity, water temperatures, slow-moving or stagnant waters, and high nutrient levels [10–12]. Of these factors, only nutrients can usually be controlled. In Utah Lake many of these factors affect algal growth and depending on climatology and seasonal effects can be limiting [13,14]. For example, at certain times of the year, light penetration (Secchi depth measurements of only a few centimeters) can be so low as to be a limiting factor in algal growth [7,8,15–17].

In 2004, Utah Lake was listed on Utah's §303(d) list for exceedance of state criteria for total phosphorus (TP) and total dissolved solids (TDS) concentrations. The State of Utah's Division of Water Quality conducted a study to calculate total phosphorus loadings to Utah Lake. The sources that were included in this study were tributaries (including WWTPs), groundwater, springs, and miscellaneous surface flows. Of the total phosphorus loading, the tributaries were found to contribute 97.2% [7]. Atmospheric deposition was not considered, mainly because little to no information was available. Since that time, Abu-Hmeidan, Williams and Miller [15] found that historic sediments in the lake have high phosphorous concentrations that affect water quality. Laboratory tests showed that Utah Lake sediments can contribute anywhere between 0.24 to 19 mg/L of TP to water in laboratory columns. This study also showed that soils on the Utah Lake shore are historic sediments that have high phosphorous values, up to 1000 mg/kg with average values over 700 mg/kg [15].

Merritt and Miller [9] completed an extensive eutrophic and nutrient loading evaluation for Utah Lake. They confirmed TP loadings from the earlier analysis by the Utah Department of Water Quality (DWQ) and studies by the engineering firm Psomas based on water inflows. They evaluated the impact of these loadings using the Larsen and Mercier Trophic State Model and the Carlson Trophic State Index Model [18,19]. They concluded that the actual lake state is currently moderately eutrophic, but P and N loadings estimated from water inflows to the lake are some 15 to 20 times larger than needed to support a eutrophic level of algal growth. They also found that for these estimated loadings, the N/P ratio was approximately 8, which indicates that nitrogen would be more limiting than P if nutrients are the limiting factor. They noted that this may have some impact, though tenuous, on the presence of HABs in the lake at times. Overall, Merritt and Miller [9] concluded that phosphorus and nitrogen loadings from water inflows could not be reduced sufficiently to make nutrients the limiting factors to algal growth. They reported that light limitation, via the high natural turbidity of the lake, was likely the overall limiting factor. They recommended quantification of atmospheric nutrient deposition to better understand the feasibility of nutrient control studies and make these studies more accurate. These recommendations were submitted before recent research that indicates that lake sediments may also be a large nutrient source to Utah Lake [15]. Based on the Larsen–Mercier Model, they concluded that only 17 Mg (tons) year⁻¹ of phosphorus was needed to support the current moderately eutrophic

state of the lake (Note on units, English tons from previous studies have been directly converted to Metric tonnes (Mg) which is 1000 kg. This was done to maintain significant figures as the nutrient estimates have significant uncertainty and a more precise conversion would have resulting in numbers with higher precision than is warranted by the data). Using a N/P ratio of 12 for the algae, this means about 200 Mg (tons) year⁻¹ of nitrogen would be needed by Utah Lake algae. Current estimated nutrient loadings are much higher than this, even without considering atmospheric deposition or sediment sources, and Merritt and Miller [9] noted it would not be feasible to reduce nutrient loadings to this level by improving WWTP effluent concentrations.

1.3. Deposition Classification and Collection

The National Atmospheric Deposition Program (NADP), the primary atmospheric deposition monitoring program in the United States, does not measure TP in precipitation samples. As a result, little data are available for atmospheric deposition of TP in Utah. However, there have been several studies performed in the Western United States with regard to nitrogen deposition. In these studies, nitrate deposition in the Utah Wasatch Front produced concentrations as high as 2.0 kg/ha N [20].

There are three main classifications of atmospheric deposition: bulk deposition, wet deposition, and dry deposition. Bulk deposition has been the most common form of atmospheric collection for decades [21,22]. Recently, it has become increasingly common to collect both wet and dry deposition. Wet deposition is defined as the process by which atmospheric pollutants become associated with cloud or precipitation droplets (or particles) and subsequently delivered to the earth's surface during precipitation. Dry deposition is the process whereby gases and aerosols are deposited directly from the atmosphere to surfaces. Dry deposition sources principally include particulates from the burning of fossil fuels, forest fires, wind erosion of soil, aerosols from agricultural practices, and plant pollen [3,4,23].

Sampling devices used in the collection of atmospheric deposition vary, depending on the type of deposition to be measured. It is common for researchers to collect samples using wet-dry samplers. In these samplers, both wet and dry deposition samples are collected using two buckets and a moisture sensor. The bucket corresponding to either wet or dry deposition is left open to the atmosphere depending on whether wet deposition or dry deposition is being collected. The moisture sensor is used to close one bucket and open the other when conditions change [24].

1.4. Utah Lake Description

Utah Lake (see Figures 1 and 2) is a shallow, eutrophic, basin-bottom lake in a semi-arid region. It is the third largest freshwater lake west of the Mississippi River in surface area and has the largest surface area of any freshwater lake in Utah. Utah Lake rests in Utah Valley and is generally bounded by several municipal areas on the south, east, and north, and a sparsely populated area to the west. It has an average depth of 2.7 m (9 ft.), a max depth of 4.3 m (14 ft.), and a surface area of approximately 364 km² (90,000 acres).

Inflow to Utah Lake is approximately 800,000,000 m³ per year (725,000 acre-feet) with significant variation [7]. The primary sources of water to the lake are snowpack runoff from the Wasatch and Uinta Mountains and small streams and creeks. In addition to precipitation and groundwater, the lake has at least fifteen key tributaries. The major tributaries include the American Fork River, Benjamin Slough, Hobble Creek, Mill Race Creek, Provo River, and the Spanish Fork River. The Provo River and the Spanish Fork River account for approximately 60% of the inflow into Utah Lake [7]. The Jordan River is the only surface outlet and discharges northward toward the Great Salt Lake.

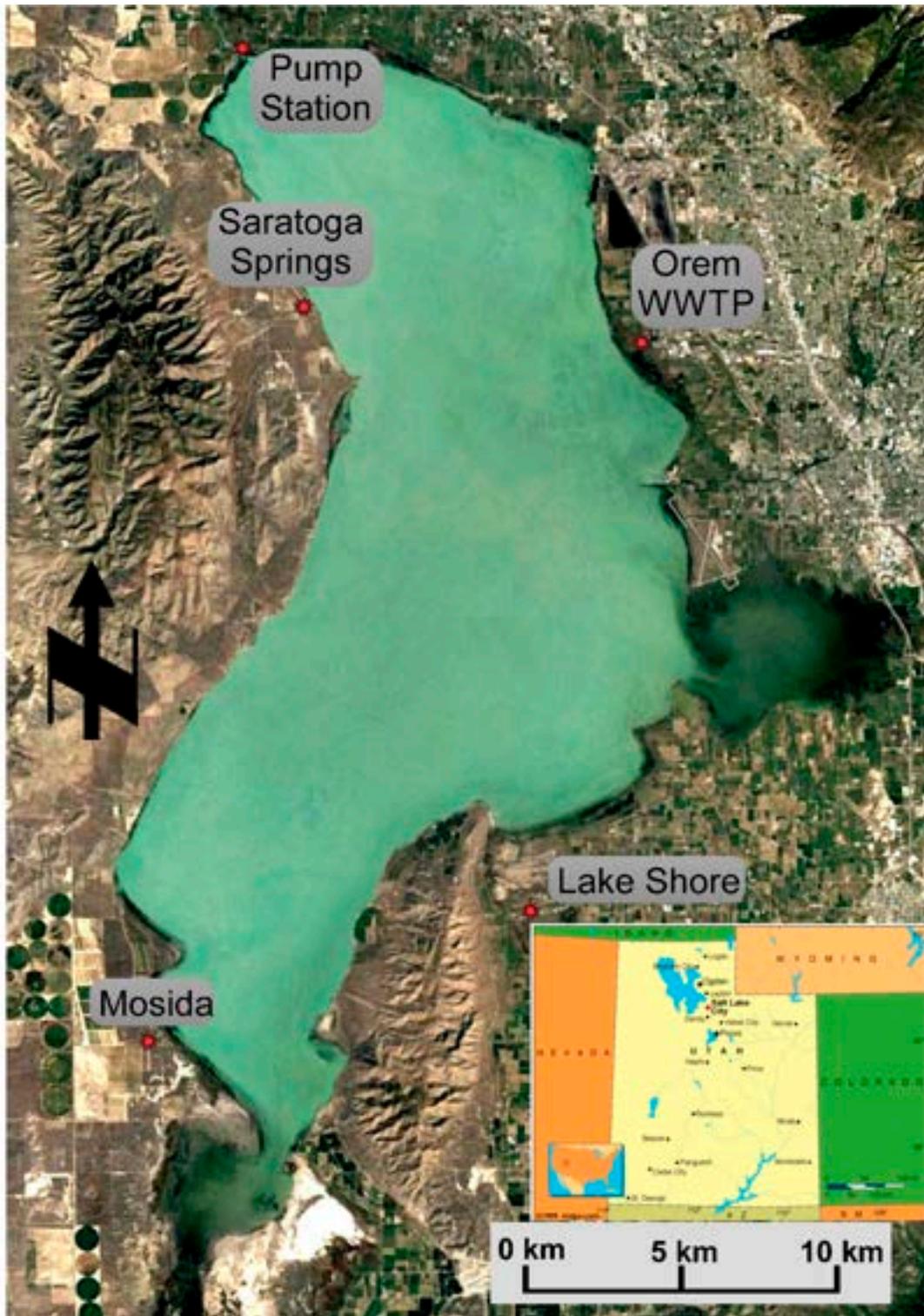


Figure 1. Utah Lake is a large freshwater lake in northern Utah which has a surface area of about 360 square km (140 square miles) with an average depth of 2.7 m (9 feet) and a maximum depth of only 4.3 m (14 feet). Also shown on the map are the five atmospheric deposition sampling sites used in this study.



Figure 2. The Jordan River basin that contains both the inflows and outflow for Utah Lake (figure created from [25]).

The area surrounding Utah Lake has an average annual precipitation of approximately 33 cm (13 inches) [26]. Approximately 50% of the water flowing into Utah Lake is lost to evaporation each year. This high evaporation rate of the water flowing into Utah Lake is lost to evaporation each year. This high evaporation rate of dissolved solids (TDS) and calcium carbonate (CaCO₃) in the water results in a high concentration of dissolved solids (TDS) and calcium carbonate (CaCO₃) in the water. The high concentration of dissolved solids (TDS) and calcium carbonate (CaCO₃) in the water results in a high concentration of suspended sediments in the water. The high concentration of suspended sediments in the water results in a high concentration of suspended sediments in the water. The high concentration of suspended sediments in the water results in a high concentration of suspended sediments in the water.

2. Materials and Methods

2. Materials and Methods

2.1. Sampling Methods and Locations

2.1. Sampling Methods and Locations

To characterize atmospheric depositional patterns on Utah Lake, we established a network of five sampling stations (see Figure 1) where we deployed Utah Lake automated samplers. All of these five locations are situated near the Utah Lake shore line. We distributed the locations around the

were situated near the Utah Lake shore line. We distributed the locations around the parameter of Utah Lake to obtain a representative measure of the total atmospheric deposition across the lake surface. We selected the sites to distribute around the lake as evenly as possible and based on site access. We designed these samplers to operate similar to the Aerochem Metrics (ACM) bucket collectors used by the NADP. Placement of each sampler followed protocols similar to those established by the NADP [24]. Table 1 provides a site information for each sampling site and indicates if the site is in compliance with NADP, or if it is not, why that site is not compliant with the NADP protocol.

Table 1. Site information and compliance of each site with NADP site selection protocols. Not all protocols were followed as we are interested in local transport in addition to long-range transport.

Issue	Lake Shore	Mosida	Saratoga Springs	Pump Station	Orem WWTP
Latitude	−111.787781	−111.927626	−111.868827	−111.895347	−111.735528
Longitude	40.11229	40.076452	40.283815	40.359414	40.276158
Irrigation sources	Compliant	Central Pivot irrigation 380 m from collector	Compliant	Central Pivot irrigation 500 m from collector	Wheel line irrigation 500 m from site
≥5 m from Equipment	Solar Panel	Solar Panel	Solar Panel	Solar Panel	Solar Panel
≥5 m from Collector	3 m from collector	Compliant	Compliant	Compliant	Compliant
≥10 m from Collector	Access road is 7 m from collector	Compliant	Compliant	Compliant	Compliant
≥20 m from Collector	Horse corral 10 m from collector	Compliant	Compliant	Compliant	Compliant
≥30 m from Collector	Farm Shed 15 m from collector	Compliant	Small gravel driveway 25 m from collector	Compliant	Compliant
≥100 m from Collector	Compliant	Compliant	Compliant	Compliant	Parking lot 60 m from collector
≥500 m from Collector	Compliant	Compliant	Compliant	Compliant	Compliant
≥1 km from Collector	Compliant	Compliant	Compliant	Compliant	Compliant
NADP Site Classification	R	I	S	S	U

Each automated sampler consisted of two polyethylene buckets on an elevated table with a moveable cover. These buckets measured 0.0615 m² at the bucket opening and were 35 cm high. Movement of the cover was initiated by a moisture sensor. This moisture sensor was calibrated to ensure that dew would not trigger the movement of the bucket cover. During dry periods the cover shielded the wet-deposition bucket and when activated by rain or snow, the cover shielded the dry deposition bucket and exposed the wet deposition bucket.

2.2. Sample Collection and Chemical Analysis Methods

We collected samples at the five sites around Utah Lake from 4 May 2017 to 28 December 2017. We collected samples weekly from each location in accordance with NADP protocol, and during each

collection we replaced the buckets with clean buckets. In order to simulate the collecting properties of a wet lake surface, we pre-loaded the bottom of the dry-deposition bucket with 3 L of deionized water [4,27]. We choose to use 3 L as a compromise between simulating water surface in the context of this standard sampling device while offering low evaporative loss with a sample volume small enough to yield detectable concentrations and deposition rates.

We cleaned the graduated cylinders, funnels, bottles, buckets, and all other equipment in contact with the samples with phosphorus-free detergent and we acid-washed them with 10% HCl solution. We stored equipment in plastic bags until used for sample collection. We analyzed each batch of deionized water that was added to samplers and any detectable nutrient concentrations were subtracted from deposition estimates.

To extract samples from the buckets in the laboratory we used small brushes to scrub the sides of the deposition buckets and swirled the water to ensure that the contents and particles were removed. If needed, we added deionized water to bring the wet sample volume to 500 mL to yield sufficient analytical volume. We stored samples at 4 °C until analyzed.

Due to the fact that NO_2^- concentrations are commonly small relative to NO_3^- , we will refer to the sum of these two values as NO_3^- . Similarly, we will reference $\text{NH}_3 +$ and NH_4^+ amounts as simply NH_4^+ . We define dissolved inorganic nitrogen (DIN) as $\text{NO}_3^- + \text{NH}_4^+$. We express all measurements as a mass per square meter per time on the basis of collector surface area.

We labeled a sample as contaminated (i.e., “contaminated sample”) if any of the following conditions occurred: overlapped samples (dry buckets that collected some rainwater), bulk deposition samples (a combined wet and dry deposition sample, due to sampler malfunction), samples with visible contamination (bird droppings, insects, obvious algal growth), and samples that were collected longer than a week. Both contaminated and uncontaminated samples were used in analysis to describe a possible range of atmospheric loading to Utah Lake and we report both results. For contaminated samples, any visible debris, such as vegetative matter or bug bodies, were removed from the sample before analysis. We debated whether to leave the debris in the samples because they do represent a nutrient source for the lake, but we decided to err on the side of caution and removed the debris. Even though the debris were removed, these samples were likely impacted because nutrients from the debris could dissolve in the water in the sample bucket.

2.3. Sample Analysis

Field samples were analyzed by the BYU Environmental Analytical Laboratory with some TP duplicates sent to ChemTech-Ford Laboratory, a commercial laboratory, for quality checks. We were not able to send duplicate DIN samples to ChemTech-Ford, as the field samples were not acidified and too much time elapsed between sampling and sending samples for analysis by the commercial laboratory. All duplicate TP samples were within expected ranges. Both laboratories followed procedures recommended by NAPD [24]. TP was analyzed using microwave digestion followed by inductively coupled plasma (ICP) spectroscopy. NH_4^+ was analyzed with the salicylate method and NO_3^- with the cadmium reduction method. All analysis was done by certified analytical laboratories following standard methods.

2.4. Atmospheric Load Calculations and Geostatistical Methods

To calculate atmospheric nutrient spatial loading rates at each sample point we multiplied the nutrient concentration of each sample (mg/L) by the volume of the sample (L) to give milligrams of nutrients deposited in the sample bucket. If distilled, deionized water was added to bring samples up to analytical volume, we subtracted any measurable nutrient concentration in the distilled, deionized water from the sample concentration. We then calculated unit area deposition rates dividing the total deposition mass (mg) by the surface area of the sampler (0.0615 m^2) and the time represented by the sample (for example 1 week).

We used the measured TP and DIN unit deposition rates to create spatial models of nutrient distribution patterns on the lake. To calculate total deposition, we used a total lake surface area of 354 km² (88,000 acres), which was the average lake area during our sampling period.

We believe that deposition near the shoreline is most likely higher than deposition in the interior of the lake as near the shore deposition from local soil dust is an important component. Most atmospheric deposition programs are not interested this type of local transport and view it as contamination. Most atmospheric deposition studies generally try to only measure deposition rates from long-range nutrient transport [24,28]. We expect that local transport does affect deposition near the center portions of the lake but that these rates are probably lower than those near shore. In order to conservatively estimate total deposition rates, we created six “dummy” sample points along the center of the lake (blue squares in Figure 3) and assigned background deposition values for TP and DIN of 0.019 mg TP m⁻² week⁻¹ [28] and 0.112 mg DIN m⁻² week⁻¹ [24], respectively. These values are at least two orders of magnitude lower than the local deposition rates measured at the lake shore.

Figure 3 shows the data locations used to create the spatial model. The green squares along the shore are local sample locations: Lake Shore, Mosida, Saratoga Springs, Pump Station, and Orem WWTP. We assigned background deposition rates described above to the six blue squares in the center of the lake (see Figure 3).

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Figure 3. Utah Lake coverage showing the eleven points used in interpolation.

3. Results

To interpolate between the sample points we used simple kriging with an exponential variogram that had a range of 1000 m to represent the decrease in the deposition rate as distance increased onto the lake from the shoreline [3,29]. We interpolated data from the sample points, both measures and background onto a 2D grid with 3398 cells which represents the lake surface. Each of the cells Table 2 shows the number of samples that were collected from each site.

Table 2. Number of dry and wet samples collected at each location.

Site	Number of Samples		
	Dry	Wet	Total

had an area of 101,722 m² with dimensions approximately 381 m by 267 m (1250 feet by 875 feet). This approach means that the estimated deposition rates tend toward the average deposition rate for cells away from the sample points. In this case the average rate would be the average of the six pseudo points (long-range background) and the five sample points, this would result in data closer to the long-range background deposition rates, significantly lower than any rates measures at the shoreline stations. We used the Groundwater Modeling System (GMS) geostatistical software developed by Aquaveo for computation and to create the spatial distribution maps [30].

For each week, we interpolated deposition loadings onto the 2D grid to estimate deposition for that week. For sites that had missing values for any week, we used the average of the remaining sites for that week. To compute the total weekly load, we multiplied the deposition rate for each cell by the cell area to give a nutrient loading in each cell. We then summed the atmospheric nutrient loadings for each cell to compute the total nutrient loading for Utah Lake (mg week⁻¹) for that week. We converted the loading units from mg/week to Mg/week for convenience. We repeated this process for every week in which we collected data over the 31-week study period. We summed the nutrient loading for each week (31 weeks) to yield a total atmospheric nutrient loading for Utah Lake for the eight months. We used the same process for both TP and DIN.

3. Results

3.1. Sampling Results

From May to December, we collected 162 dry deposition samples and 59 wet deposition samples. Table 2 shows the number of samples that were collected from each site.

Table 2. Number of dry and wet samples collected at each location.

Site	Number of Samples		
	Dry	Wet	Total
Lake Shore	29	12	41
Mosida	28	10	38
Saratoga Springs	30	14	44
Pump Station	28	10	38
Orem WWTP	22	5	27

3.2. Phosphorous Load Calculation

Table 3 presents the TP results as mean TP concentrations at each sample site separated by deposition type, the weekly average precipitation, and the mean TP load rate at each sample site with standard deviation, and skewness values for the loading data. The average loading across all the sites was 8.10 mg of TP m⁻² day⁻¹, with mean values for the individual sites ranging from 1.26 to 31.38 mg of TP m⁻² day⁻¹ at Orem WWTP and Saratoga Springs, respectively. Standard deviation of the loading rates were high, larger than the mean values. The skewness values are all greater than 0, with an average value of 2.7, which indicates that the samples are skewed right. This means that the bulk of the data are in the lower range with a few larger values (i.e., long tail towards the higher values). The coefficient of variation (i.e., mean divided by standard deviation), calculated using all the data, was 0.34.

Table 3. Summary for total phosphorous (TP) concentration and load data from May 2017 through December 2017.

Site	No. of Data	Mean TP Concentrations (mg/L)		Rain cm(in)/Week	Total TP Load Rate (mg m ⁻² day ⁻¹)		
		Wet	Dry		Mean	S.D.	Skewness
Lake Shore	41	0.68	0.38	0.64(0.25)	1.33	1.95	0.82
Mosida	38	0.22	0.85	0.30(0.12)	2.77	5.63	2.55
Saratoga Springs	44	0.60	5.15	0.43(0.17)	31.38	88.73	2.14
Pump Station	38	0.59	0.85	0.41(0.16)	3.78	20.14	4.68
Orem WWTP ¹	27	1.62	0.39	0.28(0.11)	1.26	2.65	3.33
Average	38	0.74	1.57	0.41(0.16)	8.10	23.82	2.70

3.3. Nitrogen Load Calculation

Table 4 presents the DIN results to mean DIN concentrations at each sample site separated by deposition type, the weekly average precipitation, and the mean DIN load rate at each sample site with standard deviation, and skewness values for the loading data. The average DIN loading of all sites is 10.23 mg of N m⁻² day⁻¹. The mean values for the individual sites range from 1.59 to 36.06 mg of N m⁻² day⁻¹ at the Pump Station and Saratoga Springs, respectively. As with the TP data, the standard deviations are high and skewness values are positive—indicating that the data have long tails to the right (large outliers). The mean values for the individual sites range from 1.59 to 36.06 mg of N m⁻² day⁻¹ at the Pump Station and Saratoga Springs, respectively. As with the TP data, the standard deviations are high and skewness values are positive—indicating that the data have long tails to the right (large outliers). The coefficient of variation, calculated for all the data, was 0.36, very similar to that computed for the TP data.

Table 4. Summary for dissolved inorganic nitrogen (DIN) deposition data from May 2017 through December 2017.

Site	No. of Data	Mean DIN Concentrations (mg/L)		Rain cm(in)/Week	Total DIN Load Rate (mg m ⁻² day ⁻¹)		
		Wet	Dry		Mean	S.D.	Skewness
Lake Shore	41	4.30	2.29	1.50(0.64)	4.06	4.07	1.21
Mosida	38	2.29	4.86	0.30(0.12)	4.36	124.22	3.31
Saratoga Springs	44	4.86	4.29	0.43(0.17)	124.63	233.31	2.31
Pump Station	38	4.29	7.55	0.41(0.16)	2.33	4.60	3.04
Orem WWTP ¹	27	7.55	4.66	0.28(0.11)	4.10	28.07	2.07
Average	38	4.66	2.07	0.41(0.16)	28.07	2.07	

3.4. Site Comparison

Deposition rates varied both spatially and temporally among sample locations. Figures 4 and 5 shows the temporal variation for TP and DIN, respectively for each sample site. Precipitation data showed no significant differences among the sites. Dry deposition rates showed few site-to-site differences aside from the Saratoga Springs site.

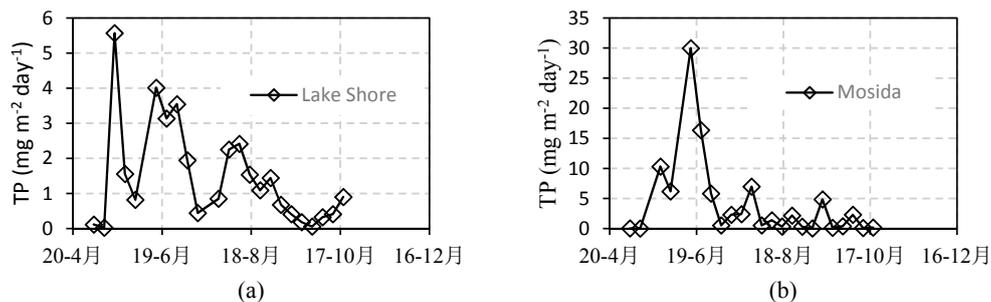


Figure 4. Cont.

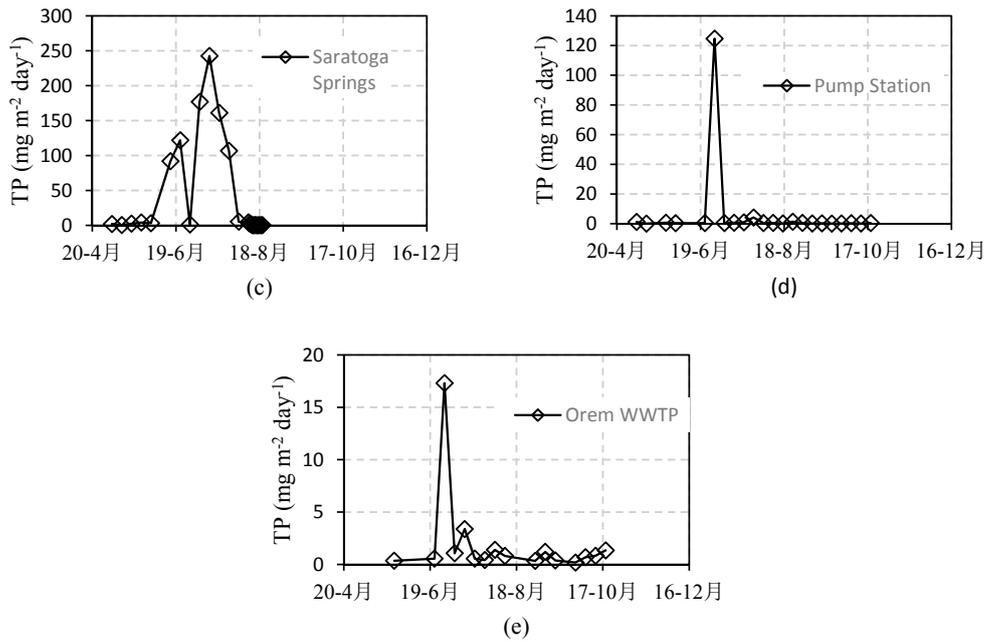


Figure 4. Temporal variations in TP by sample locations: (a) Lake Shore; (b) Mosida; (c) Saratoga Springs; (d) Pump Station; (e) Orem WWTP.

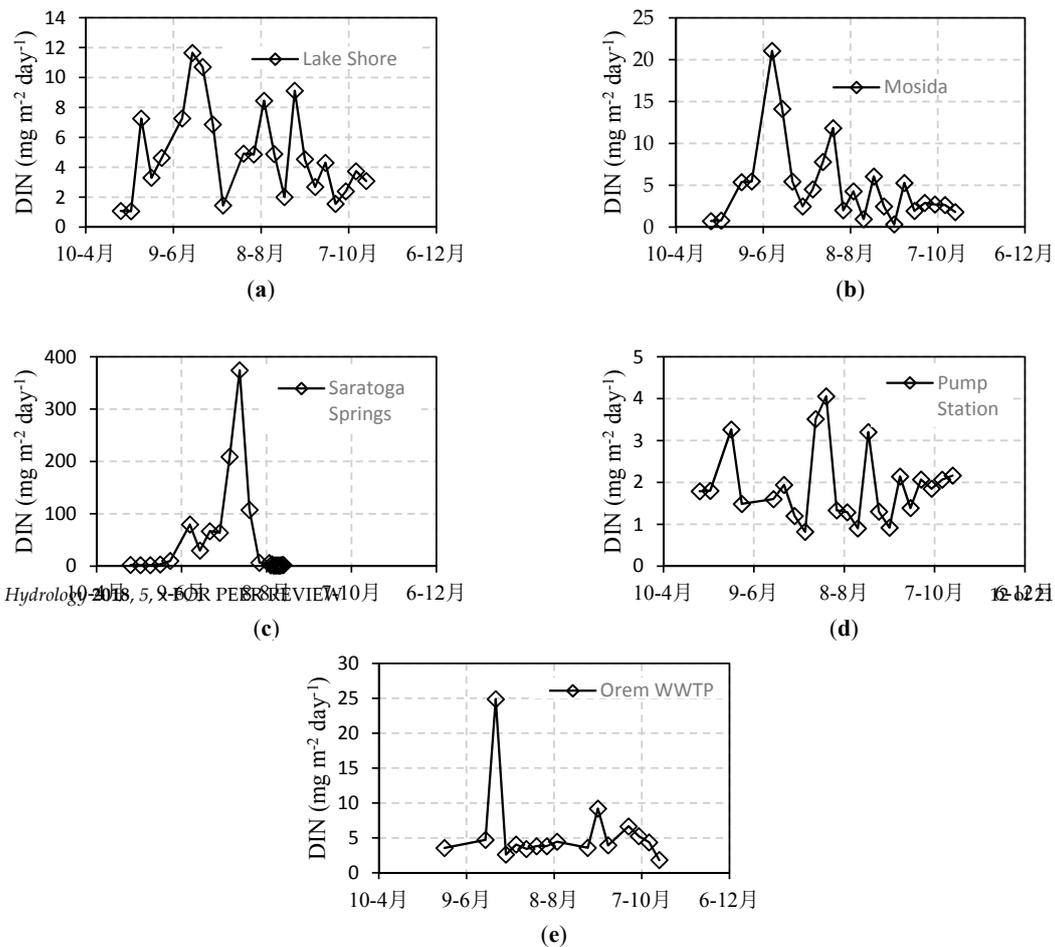


Figure 5. Temporal variations in DIN by sample locations: (a) Lake Shore; (b) Mosida; (c) Saratoga Springs; (d) Pump Station; (e) Orem WWTP.

Tables 3 and 4 show that TP and DIN deposition rates at Saratoga Springs were significantly higher than at the other sites. The TP deposition rate differences between Saratoga Springs and the other sites all have p -values < 0.005 (0.5%). Traditionally, p -values < 0.05 (5%) are used to determine if the difference is statistically significant, our p -values are an order of magnitude lower, indicating that it is unlikely that the difference is due to chance. We attribute this difference mainly to local

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3.5. Total Deposition

As described in the geostatistical methods section, we used kriging to interpolate deposition rates across Utah Lake to estimate a total deposition loading over the eight month sampling period. We used the same process for both TP and DIN. Because of the large temporal variations in deposition rates, we did not estimate annual deposition, but only estimated total deposition during the sampling period.

We estimated two different total deposition values. For the higher estimate, we included all samples collected during the study. For the lower estimate, we excluded all the samples we deemed contaminated from the analysis. Contamination was defined as the presence of any visible particles in the collection bucket. These two estimates provide bounding values for the actual deposition that occurred during the sampling period. We believe the lower estimate bounds the deposition rates because it excludes all loads from dust particles, insect bodies, and other legitimate sources of nutrients. We believe the upper estimate bounds the deposition rates because it includes some contamination sources that may be local and not significantly extend into the lake itself.

For the higher value (computed using all the samples), we estimated that the total deposition loading of TP and DIN were 350 Mg (tons) and 460 Mg (tons), respectively, over the 8-month period. For the lower value (contaminated samples excluded), the total deposition loading of TP and DIN were 8 Mg (tons) and 46 Mg (tons), respectively, for the 8-month period.

Figures 6–9 show the weekly atmospheric deposition rates for TP and DIN over the study period. Figures 6 and 7 present the lower and upper estimate values for TP and DIN, respectively. Because of the scale, it is difficult to see the variation in the lower estimate values. Figures 8 and 9 present the weekly lower estimates for TP and DIN, respectively. These figures show considerable temporal variation in both the high and low estimates, the upper estimate has the highest rates in the June–August period while the lower estimate has higher rates in the November–December period. In the latter period, November–December, the differences between the upper and lower bounds are small as contaminated samples are rare during this time.

Figure 10 shows an example distribution of TP loading for the week ending 11 May 2017. Figure 6 shows that for this week, the difference between the maximum and minimum TP loading estimates was relatively small. In Figure 10, deposition rates range from approximately $100 \text{ mg m}^{-2} \text{ day}^{-1}$ at the Saratoga Springs site (northwest shore) to approximately $0.01 \text{ mg m}^{-2} \text{ day}^{-1}$ near the center and southeast portion of the lake—approximately four orders of magnitude lower. This map shows that our choice of variogram parameters, specifically the smaller range value, used with the kriging algorithm limits the spatial influence of the high samples, with the majority of the lake tending toward background levels (blue in the figure). The green color represents values about two orders of magnitude below the red values.

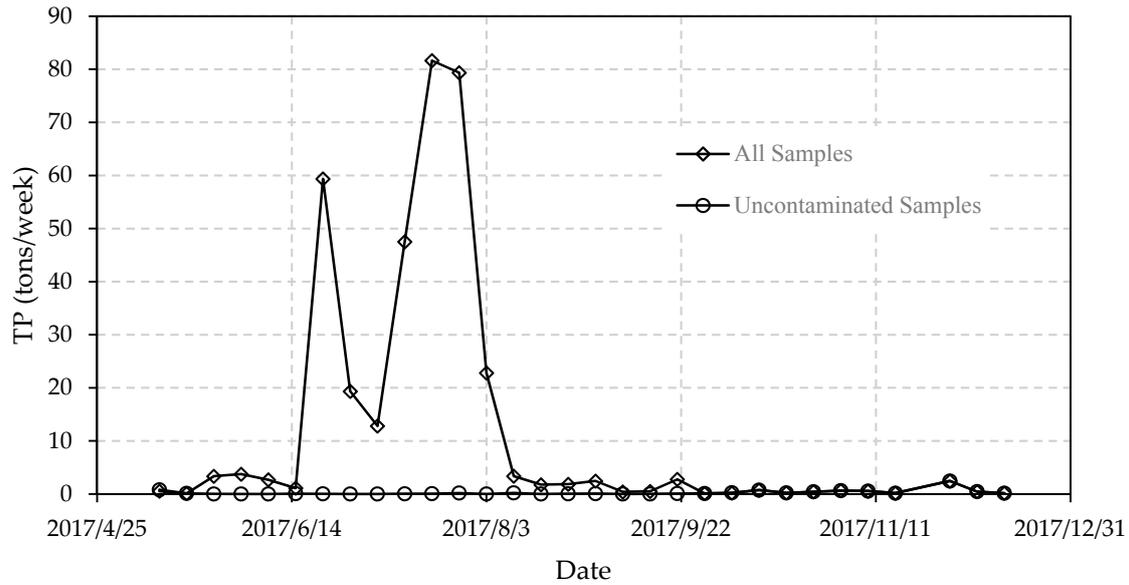


Figure 6. Utah Lake TP seasonal loading weekly estimates, both high (all samples) and low (uncontaminated samples) estimates.

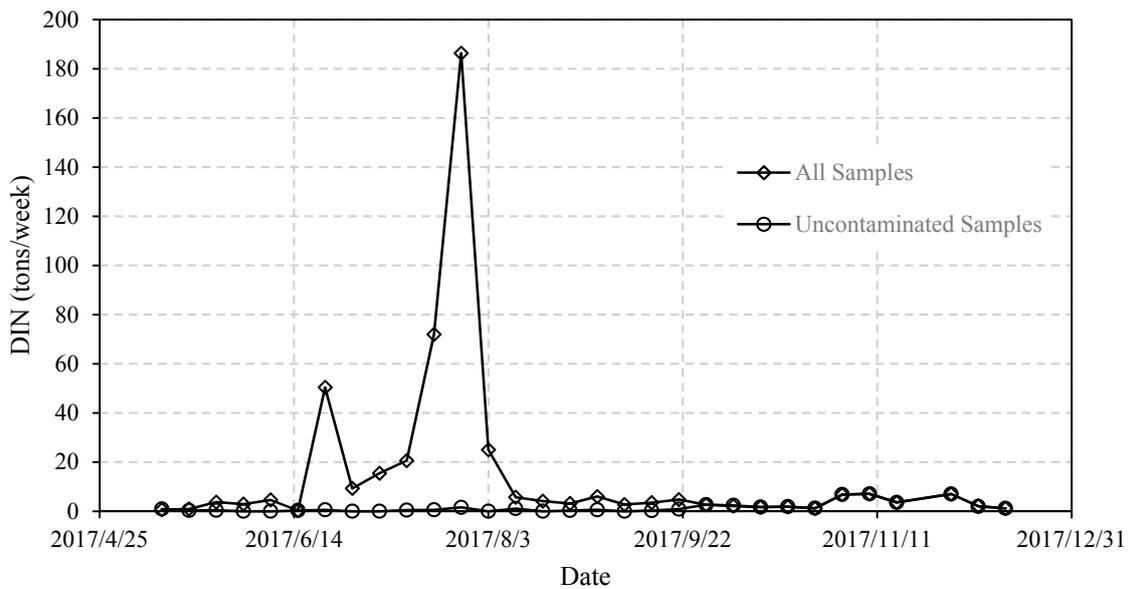


Figure 7. Utah Lake DIN seasonal loading weekly estimates, both high (all samples) and low (uncontaminated samples) estimates.

4. Discussion

4.1. Wet and Dry Deposition Rates

Wet deposition rates are more spatially correlated while dry deposition rates show more local variation. Wet deposition is the result of nutrients dissolved in precipitation; while there is some local influence on this process, most nutrients are dissolved into the rain higher in the atmosphere, away from most local variation. This explains why these data are more similar. Dry deposition rates are heavily influenced by local processes and phenomena, such as dust clouds or other near-ground sources such as insects. In addition, the dry deposition sample buckets had water in the bucket to trap the dry material. Some of the material trapped was larger in size, such as wind-borne vegetative matter

or insects. The largest dry deposition rates occurred at Saratoga Springs during the summer months with rates significantly higher than any of the other sites (see Figure 4). We attribute some of these high values to a terrestrial bee, *Halictidae Lasioglossum*. During the summer period, sample buckets had numerous bee bodies in the water. As noted above, these bodies were removed before sample analysis, but having been present in the water during the week, they significant raise the amount of nutrients in the samples. Also, as noted, the Saratoga Springs site had more local contribution from dust—which is also higher in the dry summer months. This, and other site differences, appeared to be mainly based on site characteristics; high TP deposition rates were found near agricultural and excavation areas. The west side of Utah Lake is impacted by a large gravel pit and agricultural practices, and these dust sources combined with high phosphorous levels in the soil are likely contributing to higher levels of TP.

Other studies have reached similar conclusions that local sources can contribute significant nutrient

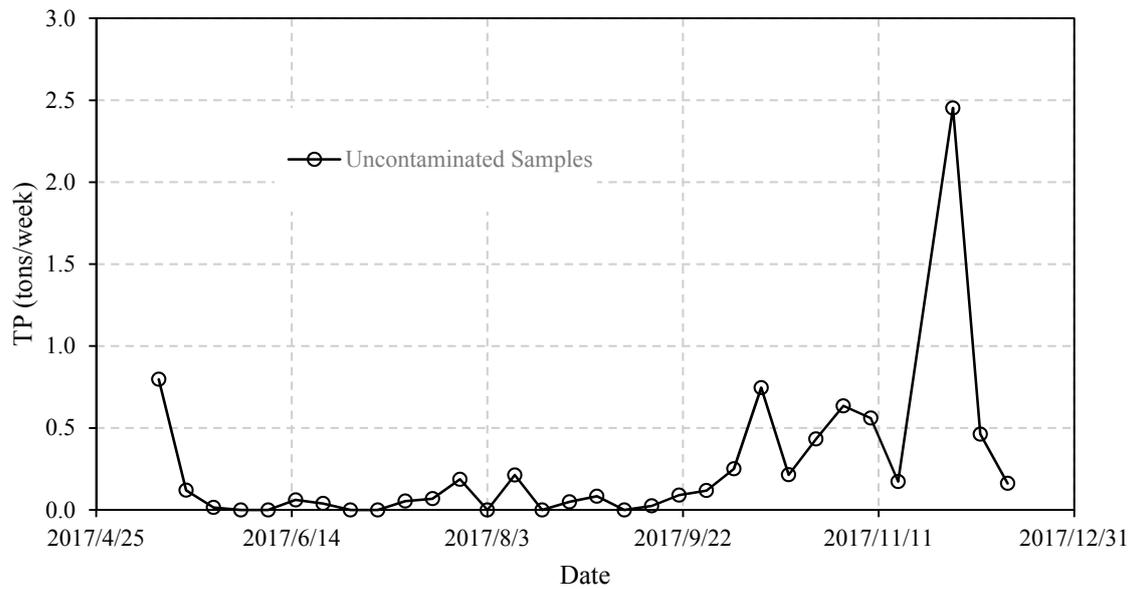


Figure 8. TP lower estimates of weekly loadings (uncontaminated samples).
 Figure 8. TP lower estimates of weekly loadings (uncontaminated samples).

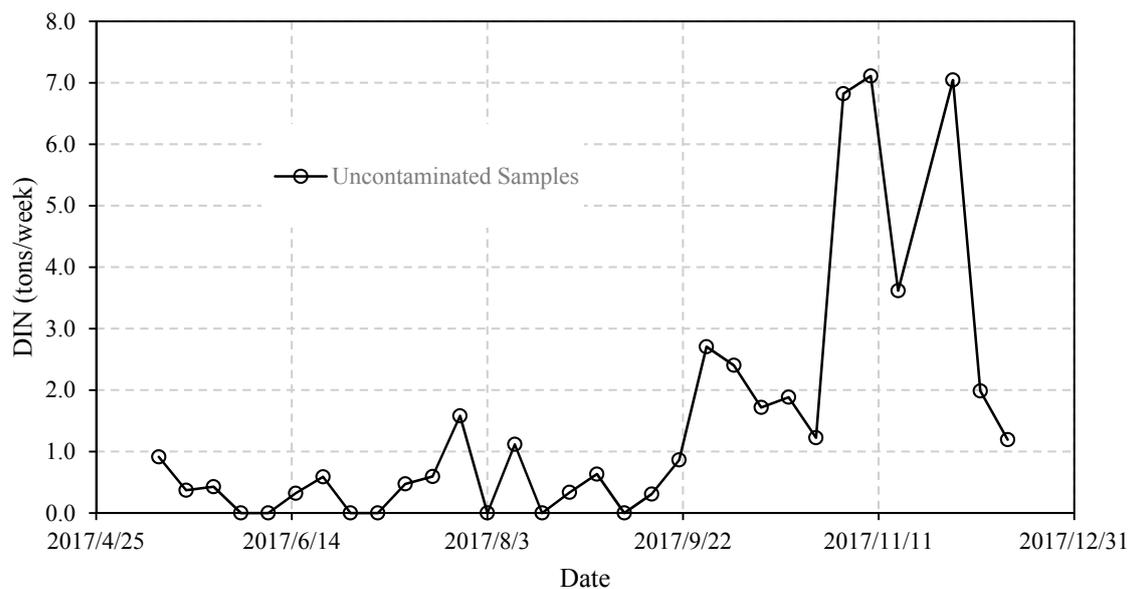


Figure 9. DIN lower estimates of weekly loadings (uncontaminated samples).
 Figure 9. DIN lower estimates of weekly loadings (uncontaminated samples).
 Figure 9. DIN lower estimates of weekly loadings (uncontaminated samples).

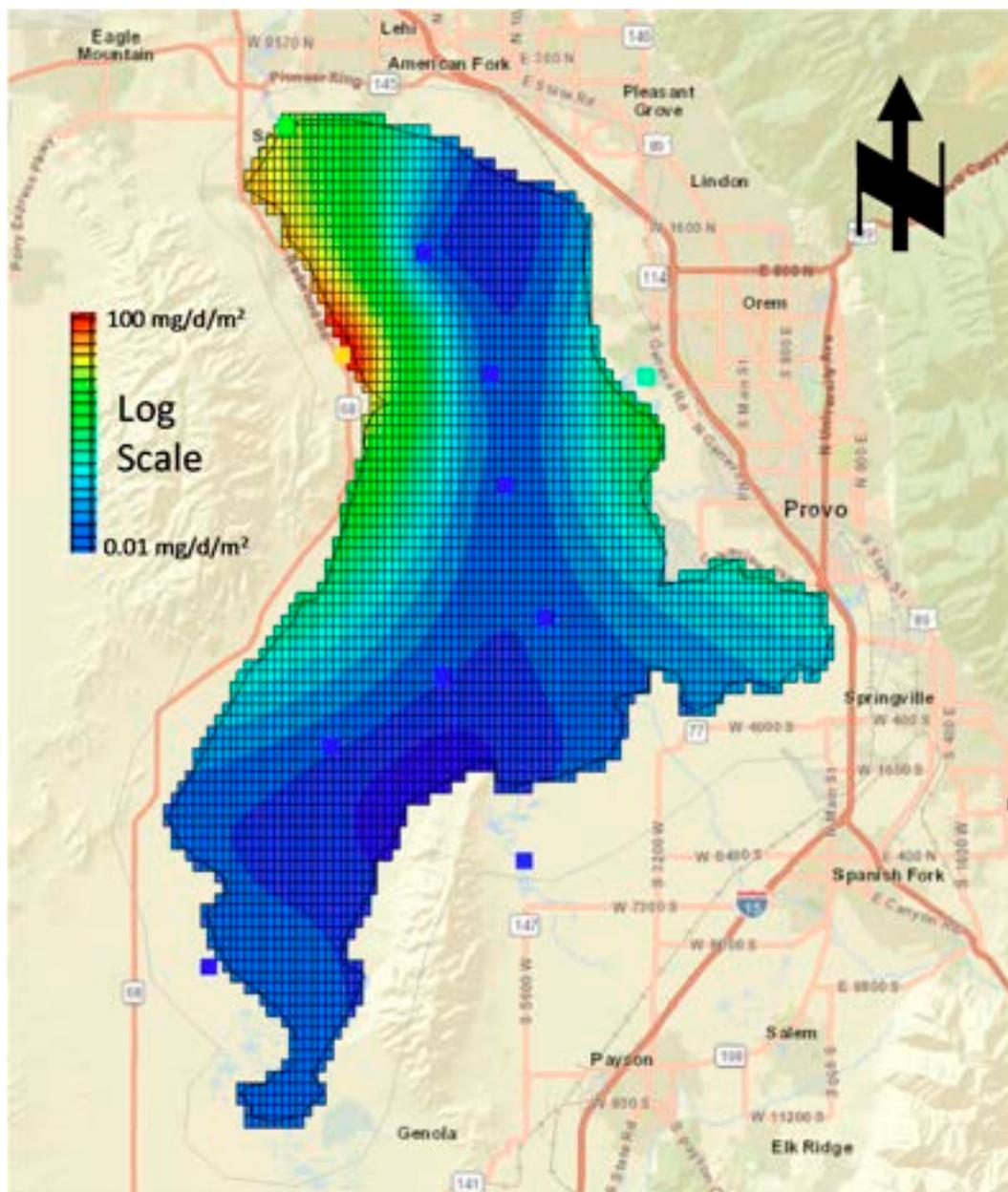


Figure 10. Example map showing the spatial deposition rates for TP for the period of 4 May 2017 through 11 May 2017. The map shows that the pseudo sample sites in the center of the lake, which represent regional background rates, strongly influence the distribution. The colors are on a logarithmic scale with red approximately $100 \text{ mg m}^{-2} \text{ day}^{-1}$ and blue about $0.01 \text{ mg m}^{-2} \text{ day}^{-1}$, spanning four orders of magnitude. This map shows that the influence of high values at shoreline stations do not reach significantly into the lake because of the small variogram range used for kriging.

4. Discussion

4.2. Local Sources versus Global Sources

4.1. Wet and Dry Deposition Rates

As discussed, most atmospheric transport methods are designed to minimize contributions from local sources and are intended to characterize long-range transport [24]. An important issue in this study is how far onto the lake the influence from local nutrient sources extends and at which point nutrients associated with long-range transport (i.e., regional/global) become the major source. Wet deposition is the result of nutrients dissolved in precipitation; while there is some local influence on this process, most nutrients are dissolved into the rain higher in the atmosphere, away from most local variation. This explains why these data are more similar. Dry deposition rates are from most local variation. This explains why these data are more similar. Dry deposition rates are We designed this study to characterize local source contributions because Utah Lake is surrounded by

urban, industrial, and agricultural land uses, and soils in the area have high nutrient content. Standard methods developed to study long-range nutrient transport would view these sample sites as being compromised because they are close to dust sources, agricultural regions, and other activities that can cause nutrients to become airborne [24]. However, since we are developing a nutrient loading based on atmospheric deposition, these local sources are important and represent actual contributions to nutrient loads on the lake; local sources are not viewed as contaminated samples. Our sampling sites were not selected to be free of local sources but rather to be representative of atmospheric deposition on the lake surface.

The high spatial variability in TP loads (see Table 3) suggests that significant portions of the TP deposition load is from local sources and affected by local conditions at the sampling sites. For example, the maximum average TP load was observed at the Saratoga Springs site ($31.38 \text{ mg of TP m}^{-2} \text{ day}^{-1}$). We attribute these higher nutrient loadings to the proximity to local dust sources, industrial and agricultural practices, and local soils.

Local sources are highest when dust is suspended during windy storms. In this region, most storms approach from a westerly direction, first arriving on the western shore (represented by the Saratoga Springs site). The Orem WWTP site is on the eastern shore and local dust clouds generated during these storms need to cross the approximately 10-km (6-mile) wide lake before reaching this sample point. The Orem WWTP site has the minimum average TP loading measured during this study of $1.26 \text{ mg m}^{-2} \text{ day}^{-1}$. It is likely that TP deposition rates decrease across the lake in some fashion from the 31.38 to $1.26 \text{ mg m}^{-2} \text{ day}^{-1}$ measured at Saratoga Springs and the Orem WWTP, respectively.

To bound our estimates, as noted above, we assumed that TP deposition rates near the center of the lake were only $0.019 \text{ mg m}^{-2} \text{ day}^{-1}$, which is the value reported in the literature for regional long-range transport [28]. This regional value is two orders of magnitude below the smallest rate we measured of $1.59 \text{ mg m}^{-2} \text{ day}^{-1}$, averaged over the study period, at the Pump Station site. This indicates that our low estimate may be significantly below actual deposition rates. Sampling sites near the lake center will be required to resolve this question.

4.3. Total Deposition

TP and DIN seasonal deposition rates are similar with higher deposition during the late spring and summer months and lower rates during the winter months. Both TP and DIN deposition rates were dominated by dry deposition (i.e., particulate matter rather than dissolved). Maximum deposition occurred in July and early August. This temporal pattern of TP and DIN deposition is shown in Figures 6 and 7, respectively.

These seasonal trends agree with many other studies which found that deposition increased in early summer and decreased over the remainder of the summer [32–34]. In a review of studies on TP deposition from long-range transport, deposition rates showed a wide range, with estimated values between 0.007 to $1.7 \text{ mg m}^{-2} \text{ day}^{-1}$ [35]. Our uncontaminated TP samples used for the lower bound estimate, fall within this range of values. The contaminated TP samples used in the upper bound estimate exceeded this range for 7 weeks during the study period, from 22 June 2017 to 8 August 2017. The period with these higher values is shown on Figure 6, but with units of total deposition ($\text{Mg (tons) week}^{-1}$) as the unit rate was converted to lake deposition rates.

Literature-reported nitrogen deposition ranges were smaller than those of TP deposition. We compared our DIN data to the NADP wet deposition data for 2014 in Utah. NADP-reported rates ranged from 2.14 to $8.42 \text{ mg m}^{-2} \text{ year}^{-1}$ [24] while our measured rates ranged from 0.04 to $24.82 \text{ mg m}^{-2} \text{ year}^{-1}$. Our measured values appear to be reasonable when compared to other studies [31,33], again with the higher values used for the upper bound being attributed to local sources.

We did not attempt to determine what percentage of these loads were bio-available. We expect that the majority of the DIN is in bioavailable forms as is the wet TP loadings as the majority of these samples only contained dissolved phosphate. The dry TP loadings contained significant local soils and dust with a portion of the phosphorous in a mineral form. Local sediment and soil samples

have phosphorous concentrations in the 600–1900 mg/kg range with 40–60% of this phosphorous in the water soluble, salt extractable, or iron-bound phases, all of which can easily release bioavailable phosphorous to the water column [15,36,37].

4.4. Contamination

Contamination is a problem in deposition studies, mainly due to a lack of agreement as to how to delineate the boundaries for the area being studied as this defines what is regarded as contamination. For example, long-range transport studies view local sources as contamination that do not contribute to additional loading because the nutrients are from the same areas—that is, no new nutrients are introduced or transported into the region. For these studies, local sources are contamination because they represent internal recycling within the local system, not new nutrients [23]. For long-range transport studies, these local contamination sources include excavation and agriculture and also includes nutrients from birds and insects.

For this study, we are attempting to bound nutrient loads to the water column in a lake or reservoir. For this purpose, the study should include all of the sources that provide nutrients to the lake, especially during the main algal-growth season. This definition includes shoreline or other local sources as nutrients from these sources provide additional loads to the lake. These differences in the goals and scope of the studies make the selection of sample sites different. For long-range studies, sites should be isolated, away from local dust sources or activities that could contribute local nutrients to the sample bucket as outlines by the NADP [24]. For studies trying to characterize nutrient loads to lakes or reservoirs, local sources, such as dust from shoreline soils and activities such as agriculture or excavations, can contribute significant loads to the water column. For this study, we viewed all these local sources as legitimate nutrient sources which increase the total load to the water column.

In this study, 43% of the collected samples were contaminated to some degree. Insects were the main source of contamination, with the terrestrial bee *Halictidae Lasioglossum* as the most common insect.

4.5. Upper and Lower Bound Estimates

Our upper estimate was computing using both contaminated and uncontaminated samples. Our lower estimate did not use contaminated samples. The contaminated samples mostly contained contributions from birds and insects with little plant materials. We manually removed all large particulates (such as insect bodies) from the samples before analysis.

Our lower estimate was computed using only the uncontaminated samples. These samples still represent both long-range and short-range transport because of the location of the sampling sites near the lake shore. That is, we expect them to be influenced by local dust sources (e.g., high nutrient soils), agricultural or other anthropogenic sources. We do not expect this loading to only be deposited near the shore, but to be carried significant distances into the lake. To provide a conservative lower estimate, we assumed that deposition rates approached long-range deposition values at the middle of the lake. We feel this is a conservative lower bound because wind transport is mainly west-to-east, samples on the eastern shore had deposition rates more than two orders of magnitude higher than the background, long-range deposition values we used as characteristic for the center of the lake. Actually, deposition values are probably closer to those measured on the eastern shore, though we have no data to support this assertion.

4.6. Future Work

More work is required to characterize deposition rates within Utah Lake. We expect them to be higher than our lower bound estimate as we believe that local sources influence deposition across the entire lake, not just the near shore area. We also expect deposition rates to vary, not only within a year, but over longer periods. For example, dry hot years would likely have higher dust source contributions than wet years, though the wind associated with storms during wet years may also cause

higher rates. A long-term study is required to better understand actual deposition. We do think that our upper and lower estimates with their associated large range, effectively bound the actual rates.

5. Conclusions

Many studies have indicated that atmospheric deposition can be an important component of the nutrient budget of a lake. We collected nutrient data weekly at five locations, situated around Utah Lake. The sample locations were selected to characterize atmospheric deposition. The sites are approximately equally spaced around the lake with locations chosen to match the main local land use. We collected both wet and dry nutrient deposition using methods and samplers following recommendations from the National Atmospheric Deposition Program (NADP) [24]. Almost half of our samples, 43%, showed contamination based on NADP guidelines. However, for this study, we viewed this contamination as legitimate nutrient sources to the water column—not as local nutrient recycling as does the NADP. Rather than trying to provide a best estimate for nutrient deposition to Utah Lake, we estimated a lower and upper bound. The lower bound was estimated using only uncontaminated samples and the upper bound was estimated using all the samples. To address depletion as local sources moved across the lake, we assumed that the center of the lake only experienced deposition from the long-range background—a rate more than two orders of magnitude below our lowest rates measured on the eastern shore and nearly four orders of magnitude from the highest rates measured on the western shore. We did not estimate annual rates, as we only had 8 months of data and the data showed significant temporal variation.

Our estimates of the 8-month nutrient load indicate that the atmosphere contributes between 8 to 350 Mg (tons) of TP and 46 to 460 Mg (tons) of DIN during this period. Dry TP deposition represents the bulk of seasonal deposition and the most important fraction of total deposition. The largest atmospheric nutrient deposition loading occurs during the summer months which is the most critical time for phytoplankton growth. Since the summer is when the algal growth is the greatest, it is important to note that the impact of atmospheric deposition to Utah Lake is even greater than if it were spread evenly during the year.

Studies of Utah Lake have concluded that about 17 Mg (tons) of phosphorus per year and 200 Mg (tons) of nitrogen per year would be sufficient to support the algal growth currently occurring in the lake, which is deemed to be negatively impacted from high nutrient loads [9]. Even our low estimates of 8 and 46 Mg (tons) for only 8 months are approximately $\frac{1}{2}$ and $\frac{1}{4}$ of the TP and DIN loads, respectively estimated to support eutrophic conditions. We assert that these lower bound estimates are significantly below the actual values because of our conservative use of the long-range transport values for the center of the lake. Our high bound estimates of 350 and 460 Mg (tons) of TP and DIN, respectively, are well above the loads estimated to support eutrophic conditions. We feel that this higher estimate is an upper bound, but that these values are not unrealistic.

This study shows that atmospheric nutrient deposition contributes sufficient loadings to support eutrophic lake conditions. While this study was specific to Utah Lake, these results should guide water managers at other locations. Most lake total maximum daily load (TMDL) studies do not include atmospheric deposition. This could be a major oversight and may indicate why some water bodies do not respond as well as expected to nutrient reductions in inflows though we have no data on other locations.

Author Contributions: J.M.O.: Designed the sample equipment, performed the field work, and did most of the data analysis, one of two main authors. G.P.W.: Assisted with sample analysis, designed the data analytical methods, and helped with study design, one of two main authors. A.W.M.: Performed the study design, provided expertise on lake and reservoir processes, and provided data and information on Utah Lake hydrology. L.M.: Provided insight and expertise on Utah Lake and historical issues associated with nutrients and algal blooms.

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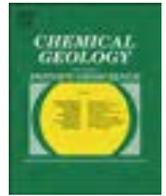
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Trace element chemistry of atmospheric deposition along the Wasatch Front (Utah, USA) reflects regional playa dust and local urban aerosols

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ABSTRACT

Atmospheric deposition to urban areas and mountain snowpack often contains a mixture of playa dust and anthropogenic aerosols, yet the relative contribution of trace elements from each source is not well understood. To evaluate dust contributions from regional playas to an urban area, we sampled playa dust sources, urban dust deposition, and snow dust deposition across the Wasatch Front in northern Utah, USA. Dust samples were analyzed for trace and major element chemistry, grain size distribution, and mineralogy. Playa, urban, and snow dust samples contained similar mineralogy, dominated by silicate, carbonate, and evaporite minerals. Grain size distribution between playa, urban, and snow dust samples was also similar, suggesting that playas are a primary dust source for the region. Principal component analysis and enrichment diagrams revealed similar chemistry of playa, urban, and snow dust with some exceptions. Compared with playa dust, urban and snow dust had lower concentrations of Li, Na, Sr, U, Mg, and Ca and higher concentrations of Fe, Al, Be, Sb, Se, Mo, Cr, La, and Cu. The first set of elements are found in evaporite- and carbonate-minerals while the second set of elements are sourced from anthropogenic activities. Mass balance calculations suggest that 90% of the dust mass deposited to the Wasatch Front is from playas, with small additions from local sources that alter the dust chemistry. Sequential leaching of dust samples showed that some elements were readily leached with weak acids, including B, K, Na, Sr, Ca, U, Mo, Cd, and Se, suggesting that they are environmentally available and may serve as important macronutrients or contaminants to ecosystems. This is the first study to directly compare the geochemistry of playa dust sources with dust deposition to urban areas and mountain snowpack in the western US, with implications for understanding how water diversions, land use changes, and population growth may affect the regional dust cycle in the future.

1. Introduction

Saline lakes around the world are drying because of water diversions and climate change, exposing areas of dry lakebed and playas that are major sources of mineral dust (Goudie, 2009; Prospero et al., 2002; Reynolds et al., 2007; Skiles et al., 2018; Steenburgh et al., 2012; Wurtsbaugh et al., 2017). For example, increased dust emissions at Owens (dry) Lake in California have resulted in PM₁₀ (particulate matter < 10 μm) exceedances and deposition of harmful metals in downwind communities (Cahill et al., 1996; Gill, 1996; Reheis, 1997; Reheis et al., 2009, 2002). Persistent drought in Africa in the 1960s dried up part of Lake Chad, forming the world's largest dust source in the Bodélé Depression (Washington et al., 2003). The hypersaline Great

Salt Lake in northern Utah is currently at historic low levels and continues to recede because of drought and river diversions, increasing the lakebed surface area available for dust mobilization (Wurtsbaugh et al., 2017). Mineral dust emitted from playas has a variety of potentially harmful effects to the environment and human health. For example, dust carries an array of organisms and metals that may affect air quality and water resources (Carling et al., 2012; Dastrup et al., 2018; Kellogg and Griffin, 2006; McTainsh and Strong, 2007). Dust causes diseases such as asthma, pneumonia, and valley fever in humans (Derbyshire, 2007; Goudie, 2014; Pope et al., 1991), increases the frequency and intensity of harmful algal blooms in lakes (Brahney et al., 2015; Zhang, 1994), and causes earlier snowmelt and decreased runoff from mountain snowpack (Painter et al., 2010).

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Dust is typically a mixture of atmospheric particles from multiple sources (Lawrence and Neff, 2009). The bulk composition of mineral dust includes silicate, carbonate, and evaporite minerals with similar concentrations of crustal elements (Al, Ca, Fe, K, Mg, Na, and Si) across samples (Abed et al., 2009; Kubilay and Saydam, 1995; Reheis et al., 2009). Variation in dust chemistry is largely found in trace element concentrations (Ben-Israel et al., 2015; Zhao et al., 2015), such as the enrichment of Se in dust from the Salton Sea lakebed in California (Frie et al., 2017; Williams, 1935). The PM_{2.5} (particulate matter < 2.5 μm) fraction of dust is typically enriched in trace elements relative to larger size fractions (Aarons et al., 2017; Das et al., 2015). In urban settings, anthropogenic sources of particulate matter include petroleum refining, combustion, vehicles, construction, gravel pits, and mining (Ajmone-Marsan and Biasioli, 2010; Gunawardana et al., 2012). Urban aerosols combine with mineral dust to create a mixture that is enriched in trace elements (Das et al., 2015; Divrikli et al., 2003; Lee et al., 1972; Li et al., 2001; Samara and Voutsas, 2005). Differences in the chemistry of dust samples are evaluated by principal component analysis (PCA), where principal components (PCs) are used to characterize specific urban aerosol sources (Gunawardana et al., 2012; Meza-Figueroa et al., 2007; Tokaloğlu and Kartal, 2006). Sequential leaching of dust samples is used to evaluate element availability or mobility in the environment, particularly for potentially harmful anthropogenic-enriched elements (Carling et al., 2012; Dastrup et al., 2018; Lawrence et al., 2010).

The purpose of this study is to determine the relative contributions of playa dust and urban aerosols to atmospheric deposition along the Wasatch Front in northern Utah, USA (Fig. 1). Specific objectives are to: 1) compare the geochemistry and mineralogy of regional playa dust sources to urban and snow dust deposition; 2) quantify trace and major element enrichment in urban and snow dust deposition relative to playa dust; 3) evaluate seasonal variability in urban dust deposition fluxes and chemistry; and 4) characterize the environmental availability of trace and major elements in urban and snow dust deposition. In northern Utah, more than two million people live along the north-south trending Wasatch Front at the eastern edge of the arid Basin and Range province (Fig. 1). The Wasatch Front experiences regular dust storms due to proximity to regional dust sources (Hahnenberger and Nicoll, 2012; Jewell and Nicoll, 2011; Mallia et al., 2017; Reynolds et al., 2014; Steenburgh et al., 2012). Backward-trajectory modeling, satellite images, and atmospheric dust transport models show that western Utah playas, including the increasingly exposed lakebed of the Great Salt Lake, the Sevier Desert, and Great Salt Lake Desert, are the main dust sources to the Wasatch Front (Dastrup et al., 2018; Hahnenberger and Nicoll, 2014; Skiles et al., 2018) (Fig. 1). Playa dust and urban aerosols are deposited along the Wasatch Front and to Wasatch and Uinta Mountain snowpack, which is the primary water source for the region (Fig. 1).

2. Materials and methods

2.1. Study area description

Playas and lakes of the Sevier Desert and Great Salt Lake Desert are remnants of Pleistocene Lake Bonneville (Oviatt, 2015) (Fig. 1). Great Salt Lake reached its lowest level in recorded history in November 2016, with ~50% of the lakebed exposed relative to 1847 (Wurtsbaugh et al., 2017). Sevier Lake contained water until the mid-1800s when it was desiccated because of irrigation diversions and has remained mostly dry until present, with the exception of a high snowmelt runoff period during 1984 and 1985 (Oviatt, 1988).

Dust events in northern Utah typically occur during the spring and fall seasons, with an average of 4.3–4.7 events per year (Hahnenberger and Nicoll, 2012; Steenburgh et al., 2012). Most dust events are associated with cold fronts or a baroclinic troughs that produce southwesterly winds above threshold friction velocities to entrain sediments (Hahnenberger and Nicoll, 2012; Jewell and Nicoll, 2011; Steenburgh

et al., 2012). Southwesterly winds and dust are funneled by the Basin and Range topography toward the urban Wasatch Front (Hahnenberger and Nicoll, 2012).

2.2. Playa dust source sampling

To characterize the composition of playa dust, we sampled 15 representative playa dust source locations during 2016–2017, including Sevier Dry Lake (SDL1, SDL2, and SDL3), Tule Valley (TV1 and TV2), Wah Wah Valley (WW), Sunstone Knoll (SK), Fumarole Butte (FB), Pismire Wash (PW), Fish Springs (FS), Dugway Proving Grounds (DPG), and the dry lakebed of Great Salt Lake (GSL1, GSL2, GSL3, and GSL4) (Fig. 1). At each location we deployed BSNE (Big Springs Number Eight) samplers, which are traps installed at a fixed height above the ground and directed into the wind by a fin (Fryrear, 1986). The selection of sites was intended to provide spatial coverage across western Utah. Within each playa, we placed the BSNE samplers in locations > 1 km from roads and > 1 km from the edge of the playa in an area that had at least several kilometers of fetch to the south to maximize the amount of dust that would be collected during strong southerly winds. We used two different BSNE setups, including a single-port sampler with a trap at a height of 50 cm above the playa surface and a four-port sampler with trap heights of 10 cm, 15 cm, 20 cm and 50 cm. Photographs of the BSNE samplers are provided in the Supplementary material (Fig. S1). Dust samples were collected from each site after several months of deployment to obtain sufficient material for geochemical analysis, with one to three total deployments per site (Supplementary material, Table S1). Dust from each port was collected in a separate Ziploc bag by flushing material from the sampler with ultra-pure water.

2.3. Urban dust deposition sampling

To characterize dust deposition along the Wasatch Front, we collected samples at Provo, Salt Lake City (SLC), Ogden, and Logan (Fig. 1). At each location, we deployed a passive dust collector that was constructed from a large plastic tote lined with a plastic bag and covered with an acid-washed plastic screen and marbles to provide a surface for dust deposition (Reheis and Kihl, 1995). The collectors were placed on the rooftops of four-story university buildings where they were less likely to be impacted by local dust or disturbed by people relative to locations near ground level. Photographs of the passive collectors are provided in the Supplementary material (Fig. S2). The collectors were deployed for two-month periods during fall 2015, spring 2016, and from June 2017 through September 2018, for a total of nine dust deposition sampling seasons (Supplementary material, Table S2). An additional collector was placed at the Provo location for eight days in fall 2015 and 18 days in spring 2016 to capture specific dust events that were predicted based on the forecasted strong southerly winds. In all, we collected 35 urban dust samples, including ten from Provo, eight from SLC, eight from Ogden, and nine from Logan. One sample each from Provo, SLC, and Ogden was lost during field work or sample preparation. Dust deposition flux rates were calculated by dividing the dust mass (g) by the sample collection area (0.2624 m²) and deployment time (two months), expressed as g/m²/month.

2.4. Snow dust deposition sampling

To compare dust chemistry in the urban area with dust in adjacent mountains, we sampled dust deposition from snowpack in the Wasatch and Uinta Mountains during spring 2016, 2017, and 2018 (Fig. 1). To collect dust, we excavated a snow pit from the snow surface to the ground and identified dust layers in the snowpack. A photograph of a typical snow pit and dust layer is provided in the Supplementary material (Fig. S3). Each dust layer was sampled by packing 2 L FLPE bottles with dusty snow. Some locations produced more than one sample per season due to multiple dust layers within the snowpack,

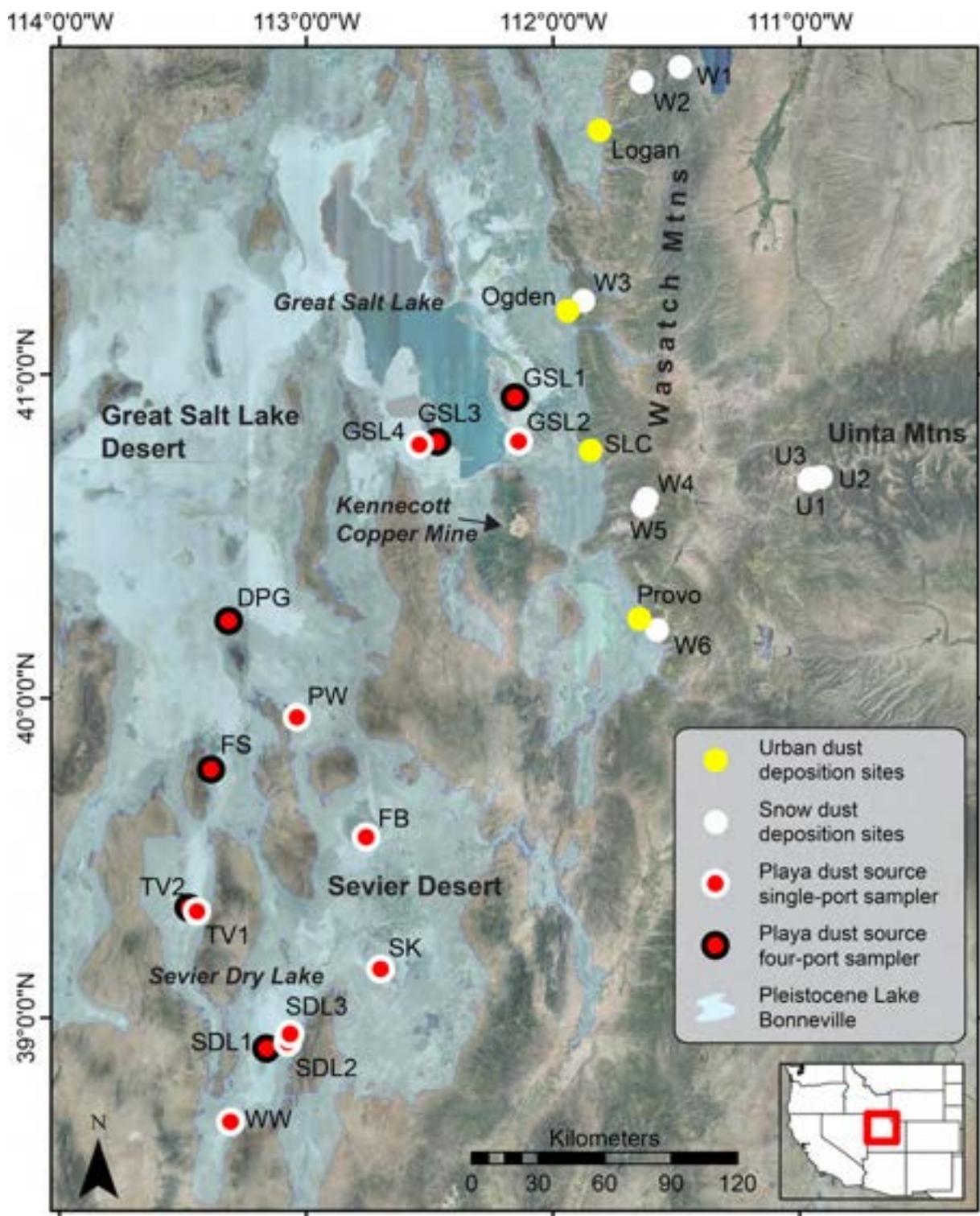


Fig. 1. Playa, urban, and snow dust sampling locations in Utah, USA. Playa dust samplers were placed in dry lakebeds that are the remnant of Lake Bonneville, including the Great Salt Lake and Sevier Dry Lake. Urban dust deposition collectors were placed on rooftops of buildings, and snow samples were collected from mountain snowpack. Abbreviations: GSL = Great Salt Lake, DPG = Dugway Proving Ground, PW = Pismire Wash, FS = Fish Springs, TV = Tule Valley, SDL = Sevier Dry Lake, WW = Wah Wah, SK = Sunstone Knoll, FB = Fumarole Butte, W = Wasatch, U = Uintas, SLC = Salt Lake City.

each of which likely represents a single dust event or storm. A total of 22 snow dust samples were collected, including three in 2016, six in 2017, and 13 in 2018 (Supplementary material, Table S3).

2.5. Dust sample analyses

Dust samples were analyzed for trace and major element concentrations, mineral abundance, and grain size distribution. Prior to analyses, the dust and water slurries from playa, urban, and snow dust samples were transferred to acid-washed 2 L FLPE bottles. Samples

were dried by evaporating at 50 °C in a laminar flow hood and treated with 30% hydrogen peroxide to remove organic matter. Mineral aggregates in the dust samples were gently broken apart using a clean disposable plastic spatula. Because playa dust source samples contained visibly large grains that were not representative of potentially far-travelled dust, a subset of bulk dust source samples was wet-sieved through a 52- μm nylon mesh filter screen before analysis (hereafter called “fine playa” samples). In all, 69 unique dust source samples were analyzed for trace and major element chemistry, including 28 bulk samples, 32 fine samples, and 9 samples for both bulk and fine fractions.

Dust samples were analyzed for trace and major element concentrations following a four-step sequential leaching procedure. To each ~200 mg sample aliquot, we added 4 mL of 1 M ammonium acetate buffered to pH = 7 and stirred vigorously. After reaction for ~20 h, the samples were centrifuged at 5000 rpm for 3–5 minutes and the supernatant was decanted to a new tube for analysis. The dust sample was then rinsed with an additional 1 mL of 1 M ammonium acetate, stirred, centrifuged, and decanted to the analysis tube. These procedures were repeated on each sample using 1 M acetic acid and 1 M nitric acid. Then, 1 mL of aqua regia (1:3 ratio of concentrated nitric acid to hydrochloric acid) was added to the remaining dust sample and reacted for three days. The purpose of the sequential leaching procedure was to target the exchangeable and water-soluble fractions with ammonium acetate, the carbonate mineral fraction with acetic acid, the feldspar and clay mineral fraction with nitric acid, and the residual minerals with aqua regia, although not all silicate and refractory minerals were completely dissolved (Carling et al., 2012; Dastrup et al., 2018; Lawrence and Neff, 2009). The total concentration of each element was calculated as the sum of the four leaching steps.

Each of the four leachates was analyzed for trace and major element concentrations using an Agilent 7500ce quadrupole inductively coupled plasma mass spectrometer (ICP-MS) with a collision cell, a double-pass spray chamber with perfluoroalkoxy (PFA) nebulizer (0.1 mL/min), a quartz torch, and platinum cones. Concentrations were measured for the following 46 elements: Ag, Al, As, B, Ba, Be, Ca, Cd, Ce, Co, Cr, Cs, Cu, Dy, Er, Eu, Fe, Gd, Ho, K, La, Li, Lu, Mg, Mn, Mo, Na, Nd, Ni, Pb, Pr, Rb, Sb, Sc, Se, Sm, Sr, Tb, Th, Ti, Tl, U, V, Y, Yb, and Zn. The detection limit (DL) was determined as three times the standard deviation of all blanks analyzed throughout each run. A US Geological Survey (USGS) standard reference sample (T-205) and National Institute of Standards and Technology (NIST) standard reference material (SRM 1643e) were analyzed multiple times in each run together with the samples as a continuing calibration verification. The long-term reproducibility for T-205 and SRM 1643e shows that our results are accurate within 10% for most elements. Raw trace and major element data are provided in the Supplementary material (Table S4).

Dust mineralogy was evaluated on a subset of five fine playa and five dust deposition samples by x-ray diffraction (XRD). The ten samples were selected to provide a representative suite of mineralogy across the sample set while considering only samples that had sufficient mass for the XRD analysis. Samples were analyzed on zero background holders with a Rigaku MiniFlex 600 XRD. Resulting patterns were quantitatively interpreted using the Reference Intensity Ratio (RIR) method (Hubbard et al., 1976) with Rigaku PDXL2 software. Raw mineralogy data are provided in the Supplementary material (Table S5).

A subset of ten bulk playa, fine playa, urban, and snow samples were analyzed for grain size distribution. The 40 samples were selected to provide a representative suite of grain size distributions across the sample set while considering only samples that had sufficient mass for the analysis. Each sample was dispersed using sodium hexametaphosphate and sonified to effectively separate aggregates. Grain size distribution was measured by laser scattering with a Horiba LA-950, which has an effective range of 50 nm to 3 mm, using the methodology of Munroe et al. (2015). Raw particle size data are provided in the Supplementary material (Table S6).

2.6. Data quality control

To prepare the dataset for statistical analyses, we removed specific elements and adjusted concentrations for elements with detection limit issues. For instance, Sc was removed because it was below DL in nearly all samples in all leaching steps. For other elements, specific values < DL were set as $\frac{1}{2}$ the DL. Additionally, Pb and Zn were not considered when evaluating playa chemistry because of anomalously high concentrations in some samples that were probably a result of contamination from the metal and/or paint on the BSNE samplers (Reynolds et al., 2014).

2.7. Data analysis

Differences in chemistry between dust types were evaluated using the PCA function in Matlab. Elements were assigned a score for each PC, where elements with higher scores contribute more significantly to a given PC. Two PCA ordinations were run, one comparing the chemistry of fine playa dust (n = 41) with bulk playa dust (n = 37) and another comparing the chemistry of fine playa dust (n = 41) with urban dust (n = 35) and snow dust (n = 22). Each run used total element concentrations from the four sequential leaching steps.

Urban and snow dust chemistry data were compared to regional playa dust using enrichment diagrams and plots of sequential leaching fractions. We compared to regional playa chemistry rather than crustal values to provide a more robust indication of anthropogenic enrichment (Reimann and de Caritat, 2005). To calculate relative enrichment or depletion for each element, median total element concentrations in snow (n = 22), Provo (n = 10), SLC (n = 8), Ogden (n = 8), and Logan (n = 9) dust samples were divided by the median total element concentrations in fine playa dust samples (n = 41). Sequential leaching fractions were calculated using the median element concentration from each leaching step for fine playa (n = 41), urban (n = 22), and snow (n = 35) dust samples.

3. Results

3.1. Similar mineralogy and grain size distributions between playa, urban, and snow dust samples

Grain size distributions were similar between fine playa, snow, and urban dust samples (Fig. 2). All samples were primarily composed of particles < 200 μm in diameter, with peaks at 10 μm for playa dust and ~20 μm for urban and snow dust. Urban and snow dust had a larger average grain size relative to playa dust, but the differences were within the error bars. A comparison between bulk playa and fine playa samples (Supplementary material, Fig. S4) shows that the wet sieving protocol successfully removed large particles (> 200 μm) that are not representative of aeolian dust.

The fine playa, urban, and snow dust samples contained a similar suite of minerals including quartz, halite, gypsum, calcite, aragonite, dolomite, and feldspar (Supplementary material, Fig. S5). Within this suite of minerals, the samples were dominated by different minerals. For example, fine playa samples from TV and FB contained > 49% calcite, SDL contained 71% halite, GSL contained 45% aragonite, and DPG contained 51% quartz. Only a small amount of gypsum (< 5%) was observed in the fine playa samples. Urban dust samples contained 20–65% quartz with a mix of halite, gypsum, calcite, aragonite, dolomite, and feldspar. The only urban sample with significant (> 1%) amounts of aragonite was Logan with 34%. The snow dust sample from the Uintas (U2) contained a smaller fraction of calcite, halite, and gypsum relative to the urban samples.

3.2. Similar trace element chemistry among playa dust sources

The PCA results showed that both bulk and fine playa dust samples

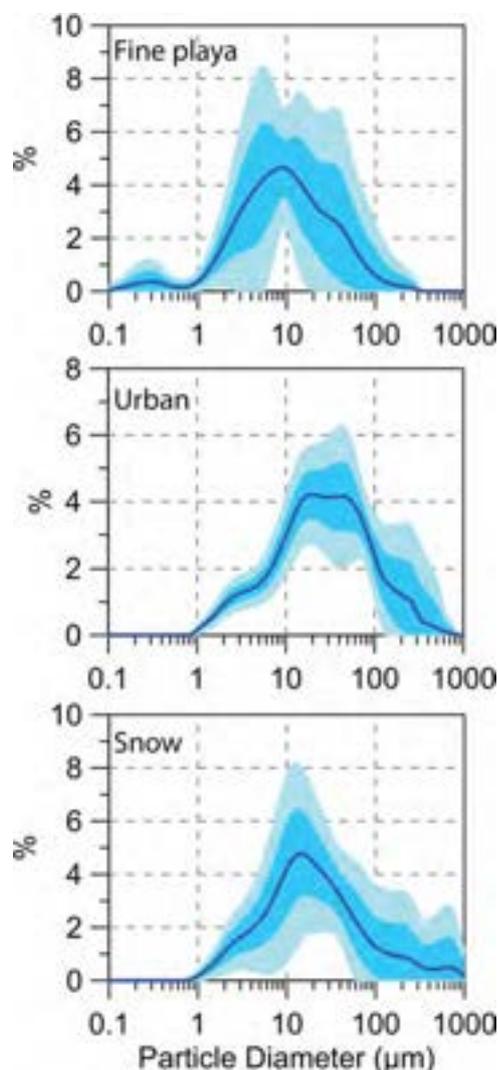


Fig. 2. Average (dark blue line) grain size distribution of fine playa ($n = 10$), urban ($n = 10$), and snow dust ($n = 10$) samples expressed as percentages in each size class. Lighter shades of blue represent 1- σ and 2- σ ranges (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article).

had similar trace and major element composition (Supplementary material, Fig. S6). A two-axis PCA explained 63.8% of the variance in the dataset (49.9% in PC1 and 13.9% in PC2). PC1 is primarily explained by Be, Al, Mn, Fe, Co, Y, rare earth elements (REEs), and Th, while PC2 is explained by Li, As, Se, Rb, Sr, Mo, Cd, Cs, and U. None of the playas had unique overall chemistry as samples from the different playas overlapped in PCA space. Nine samples were analyzed for both fine and bulk fraction (connected by tie lines in the figure) with similar PC1 and PC2 scores, revealing similar chemistry between the size fractions. An exception was one of the GSL sample pairs, connected by a dashed line, which had a different chemistry between the bulk and fine samples. Given the overall similar chemistry of the bulk and fine playa samples, only the fine playa samples are considered in the remainder of this report to simplify the comparisons between playa sources and fine-grained urban and snow dust deposition.

3.3. Urban and snow dust deposition enriched in specific trace elements relative to playa dust

Fine playa dust was geochemically similar to snow and urban dust based on the PCA results (Fig. 3). In the PCA comparing fine playa,

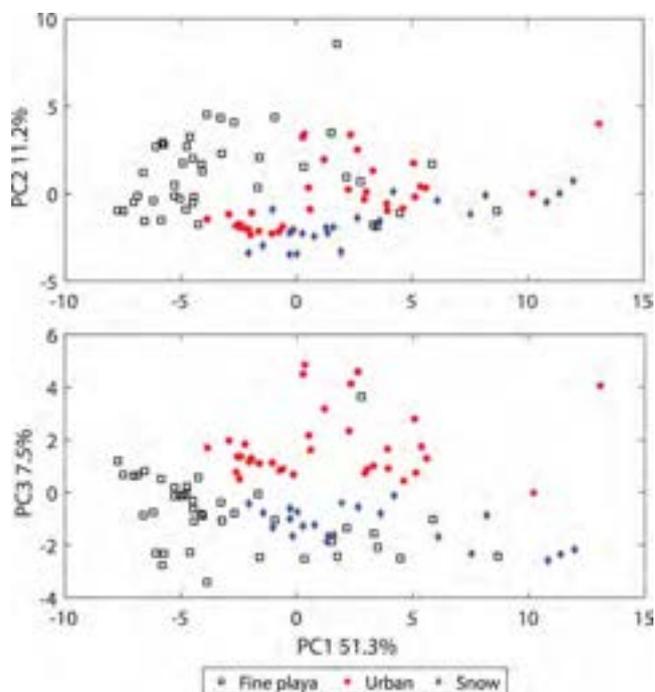


Fig. 3. PCA for fine playa ($n = 41$), urban ($n = 35$), and snow ($n = 22$) dust samples based on total element concentrations. The top panel shows PC1 versus PC2 and the bottom panel shows PC1 versus PC3. PC1 accounts for 51.3% of the total variance and is primarily explained by Be, Al, V, Mn, Fe, Co, Y, and REEs. PC2 accounts for 11.2% of the total variance and is primarily explained by Li, B, Mg, K, Ca, Rb, Sr, and Cs. PC3 accounts for 7.5% of the variance and is explained by Se, Cd, and Sb.

urban, and snow dust, three PCs explain 70% of the variability in the dataset (51.3% in PC1, 11.2% in PC2, and 7.5% in PC3). PC1 is primarily explained by Be, Al, V, Mn, Fe, Co, Y, and REEs, PC2 is explained by Li, B, Mg, K, Ca, Rb, Sr, and Cs, and PC3 is explained by Se, Cd, and Sb. Fine playa dust was generally defined by lower PC1 scores (hence lower concentrations of PC1 elements) and higher PC2 scores (higher concentrations of PC2 elements) relative to urban and snow dust. The urban samples generally had higher PC3 scores relative to fine playa and snow dust samples.

Specific elements were either depleted or enriched in urban and snow dust relative to playa dust (Fig. 4). Given the inherent variability in the dataset comparing dust deposition chemistry relative to multiple playa dust sources, we only consider elements with enrichment factors < 0.5 or > 2 times that of fine playa samples as relatively depleted or enriched. Depleted elements (enrichment factor < 0.5) include Li, Na, Sr, U, and Mg, and enriched elements (enrichment factors > 2) include Fe, Mn, Co, and Cu. Other elements were depleted or enriched in either snow dust or urban dust but not both sample types. For example, Ca and B were depleted in snow dust and Al, Be, Th, and Ti were enriched in snow dust but not in urban dust. In contrast, Sb, Se, Mo, Cd, and Cr were enriched in urban dust but not in snow dust. Notably, La was enriched in the urban SLC samples but not in the other urban or snow samples. Box plots with concentrations of specific “depleted” or “enriched” samples are shown in Figs. 5 and 6 to show the variability within each sample type.

3.4. Seasonal variability in dust flux rates and trace element chemistry

Urban dust deposition flux rates varied by season and location. Monthly dust fluxes at individual dust deposition sites varied from 0.5 to 3.8 g/m²/month (Supplementary material, Table S7). Seasonally, the highest flux rates were observed in March–May 2018 with an average of 2.9 g/m²/month at the four locations and the lowest were observed

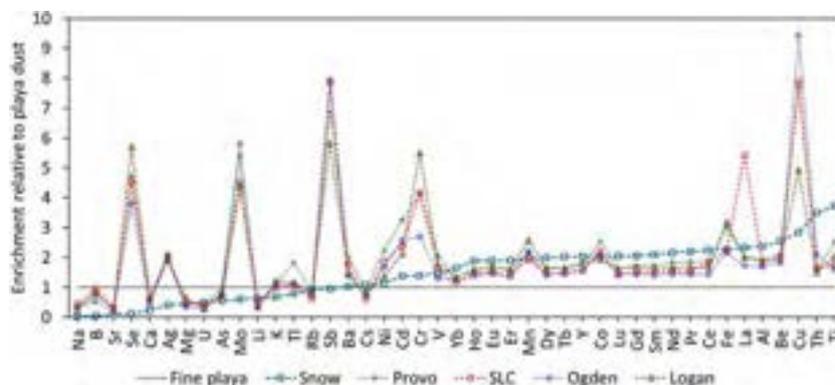


Fig. 4. Enrichment diagram comparing median total element concentrations in snow (n = 22), Provo (n = 10), SLC (n = 8), Ogden (n = 8), and Logan (n = 9) dust samples relative to median concentrations in fine playa dust samples (n = 41). Elements are sorted by element enrichment factors for snow dust.

in September–November 2015 samples with an average of 0.7 g/m²/month. Average flux rates for all sampling periods were 2.0, 2.6, 2.4, and 1.8 g/m²/month at Provo, SLC, Ogden, and Logan, respectively. The annual flux rates calculated for the period of August 2017 through September 2018 (normalized to 365 days) were 28.7, 34.9, 33.8, and 24.7 g/m²/yr for Provo, SLC, Ogden, and Logan, respectively.

Urban dust deposition samples had similar chemistry across seasons at all locations, with some elements showing seasonal patterns at all sites (Supplementary material, Fig. S7). Notably, there was a peak in Sr and Na concentrations and a dip in Co, Se, and REE concentrations at all four urban locations during the January–March 2018 sample period. Element concentrations were similar across the four urban sites during each season, particularly for REE and other trace element concentrations in the January–March 2018 samples. In contrast, Sr concentrations were variable across sites, with consistently lowest concentrations at Logan and variable concentrations at Provo.

3.5. Sequential leaching fractions indicate environmentally available trace and major elements

To evaluate relative environmental availability of specific elements, element fractions were compared between leaching steps (Fig. 7). The fine playa samples had the highest fraction of most elements in the ammonium acetate and acetic acid fractions (i.e., the playa samples were the most soluble), followed by urban and then snow dust samples. Nearly 100% of Na and over 80% of Sr and Se was extracted in the ammonium acetate and acetic acid steps in all sample types. Other readily leached elements include Ca, U, Cd, Mo, and B, with > 50% dissolved in the first two leaching steps. Insoluble elements that were primarily extracted with aqua regia include Cr, Cs, Al, Fe, and Ti.

4. Discussion

4.1. Wasatch Front dust deposition is a mixture of playa dust and urban aerosols

Similarities in grain size distribution, mineralogy, and chemistry

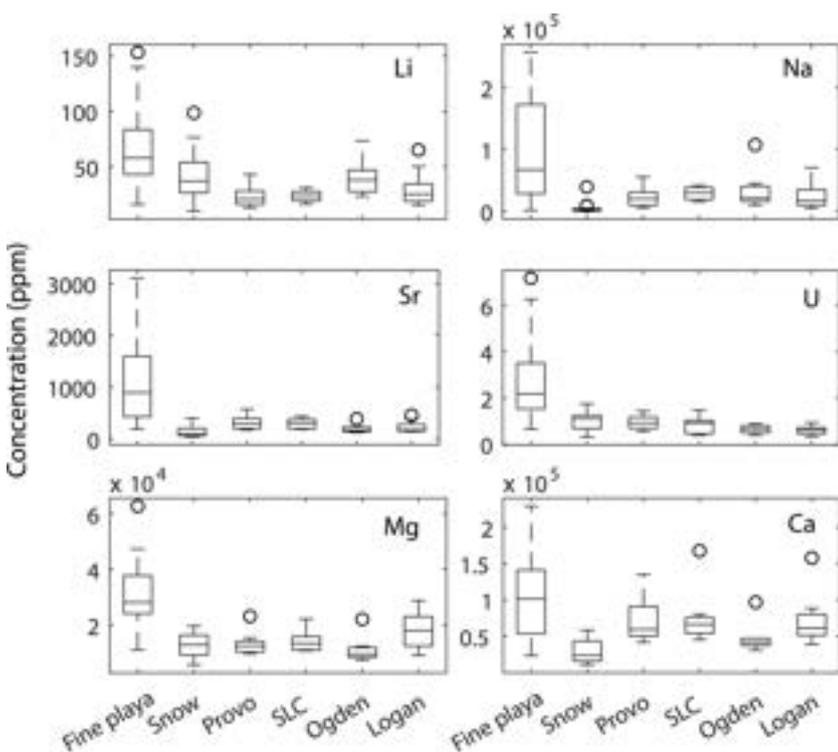


Fig. 5. Box plots showing total concentrations for elements that are depleted in urban and snow samples relative to fine playa. Data from fine playa (n = 41), snow (n = 22), Provo (n = 10), SLC (n = 8), Ogden (n = 8), and Logan (n = 9) samples. The bottom and top edges of each box indicate the 25th and 75th percentiles, respectively, and the central line is the median. The whiskers extend to the extreme points that are not considered outliers, and the outliers are plotted as individual circles.

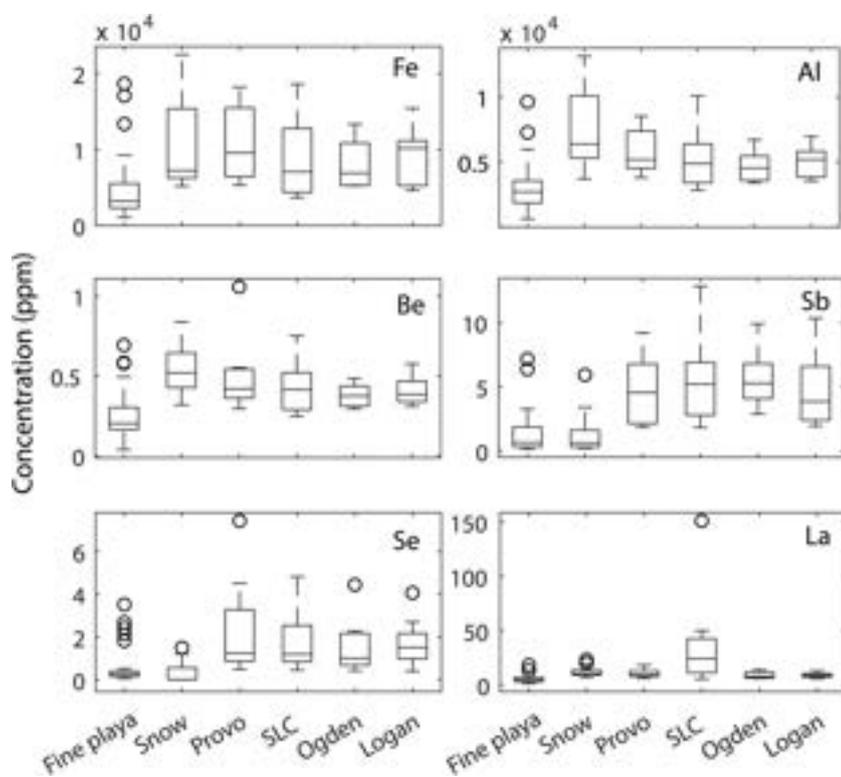


Fig. 6. Box plots showing total concentrations for elements that are enriched in urban and snow dust samples relative to fine playa dust samples. Data from fine playa ($n = 41$), snow ($n = 22$), Provo ($n = 10$), SLC ($n = 8$), Ogden ($n = 8$), and Logan ($n = 9$) samples. The bottom and top edges of each box indicate the 25th and 75th percentiles, respectively, and the central line is the median. The whiskers extend to the extreme points that are not considered outliers, and the outliers are plotted as individual circles.

between playa dust sources and urban and snow dust deposition suggest that regional playas are a major dust source to the Wasatch Front. Grain size distributions in urban and snow dust samples were similar to fine playa samples, ranging from 2 μm to 125 μm with an average size of 10–20 μm (Fig. 2). Playa dust emissions likely contain additional material < 10 μm but these particles are not efficiently trapped by the BSNE samplers (Mendez et al., 2016). The urban and snow dust samples were also composed of a similar suite of minerals as playa dust sources, including quartz, halite, gypsum, calcite, aragonite, dolomite, and feldspar (Supplementary material, Fig. S5). This suite of minerals is similar to that described in other studies of dust in northern Utah (Dastrup et al., 2018; Munroe et al., 2015). The relative abundance of different minerals in dust sources and dust deposition is not expected to be identical because soluble evaporite minerals such as halite and gypsum are not effectively transported regional distances. Carbonate minerals (calcite and aragonite) could also partially dissolve in low pH water during transport from playas to the Wasatch Front. On the other hand, the relatively insoluble silicate minerals are more effectively transported regional distances and are expected to be found at higher percentages in dust deposition. In support of this idea, the PCA shows that urban and snow samples contain relatively higher concentrations of elements associated with silicate minerals in PC1 (Be, Al, V, Mn, Fe, Co, Y, and REEs) and lower concentrations of elements associated with evaporite and carbonate minerals in PC2 (Li, B, Mg, K, Ca, Rb, Sr, and Cs) (Fig. 3). Higher concentrations of Li, Na, Mg, Ca, Sr, and U in playa samples are also shown in the enrichment diagram (Fig. 4) and box plots (Fig. 5). Further, nearly all elements were more soluble (i.e., were leached with ammonium acetate and acetic acid) in the playa samples relative to the urban and snow dust samples (Fig. 7).

Global dust is likely not a substantial component of Wasatch Front dust deposition. Modern global dust sources such as north Africa and Asia are the primary mineral and nutrient dust sources to many parts of the world (Bartholet, 2012; Takemura et al., 2002). Asian aerosols arriving in North America are generally within the < 2.5 μm size range (Fairlie et al., 2007; VanCuren and Cahill, 2002) and African dust in western North America generally consists of < 1 μm particles at low concentrations (Perry et al., 1997). In our samples, the < 2 μm fraction

makes up < 4% of the urban and snow dust particles (Fig. 2), indicating that global dust sources do not contribute a substantial fraction of the total dust deposition to the Wasatch Front.

Dust chemistry indicates that dust deposition to the Wasatch Front is a mixture of playa dust and urban anthropogenic aerosols. Most elements were found at similar concentrations (within a factor of two) in the playa and urban/snow dust samples (Fig. 4). However, several anthropogenic trace elements were enriched in snow and urban dust relative to playa dust. For example, Cu concentrations were enriched in snow and urban dust samples, suggesting that a local anthropogenic Cu source adds to the “background” playa dust. Urban areas generate Cu from manufacturing, production of electronic equipment, and waste (Ajmone-Marsan and Biasoli, 2010; Lincoln et al., 2007; Wong et al., 2007), as well as from gasoline and oil lubricants (Li et al., 2001). Another source of anthropogenic Cu contamination in Utah may be from local mining industries. The Kennecott Copper Mine, one of the largest open-pit copper mines in the world, is located adjacent to the Wasatch Front (Fig. 1). Tailings piles and mining operations proximal to Provo, SLC, and Ogden may contribute to the 40–50% higher Cu concentrations relative to Logan (Fig. 4). Other mines also exist throughout western Utah that may contribute to the observed Cu enrichment (Reynolds et al., 2014).

A subset of anthropogenic trace elements, including Se, Ag, Sb, and Cd were enriched in urban dust samples but not in snow dust samples (Figs. 4 and 6). Further, the urban samples were enriched in PC3 on the PCA diagram due to elevated concentrations of Se, Sb, and Cd (Fig. 3). The snow samples were collected from sites that are distal from the urban area and at a higher elevation, suggesting that the snow dust is less impacted by local anthropogenic contamination compared with the urban samples. Further, the urban samples represent wet and dry deposition over a two-month period whereas snow dust samples mainly represent dry deposition from a single event, giving the urban samples a greater opportunity for picking up elements from contaminant sources. Anthropogenic sources of Se such as industrial processes and fossil fuel combustion account for > 65% of total Se emissions (Wen and Carignan, 2007). The majority of anthropogenic Se is sourced from coal combustion and copper refining and smelting (Mosher and Duce, 1987).

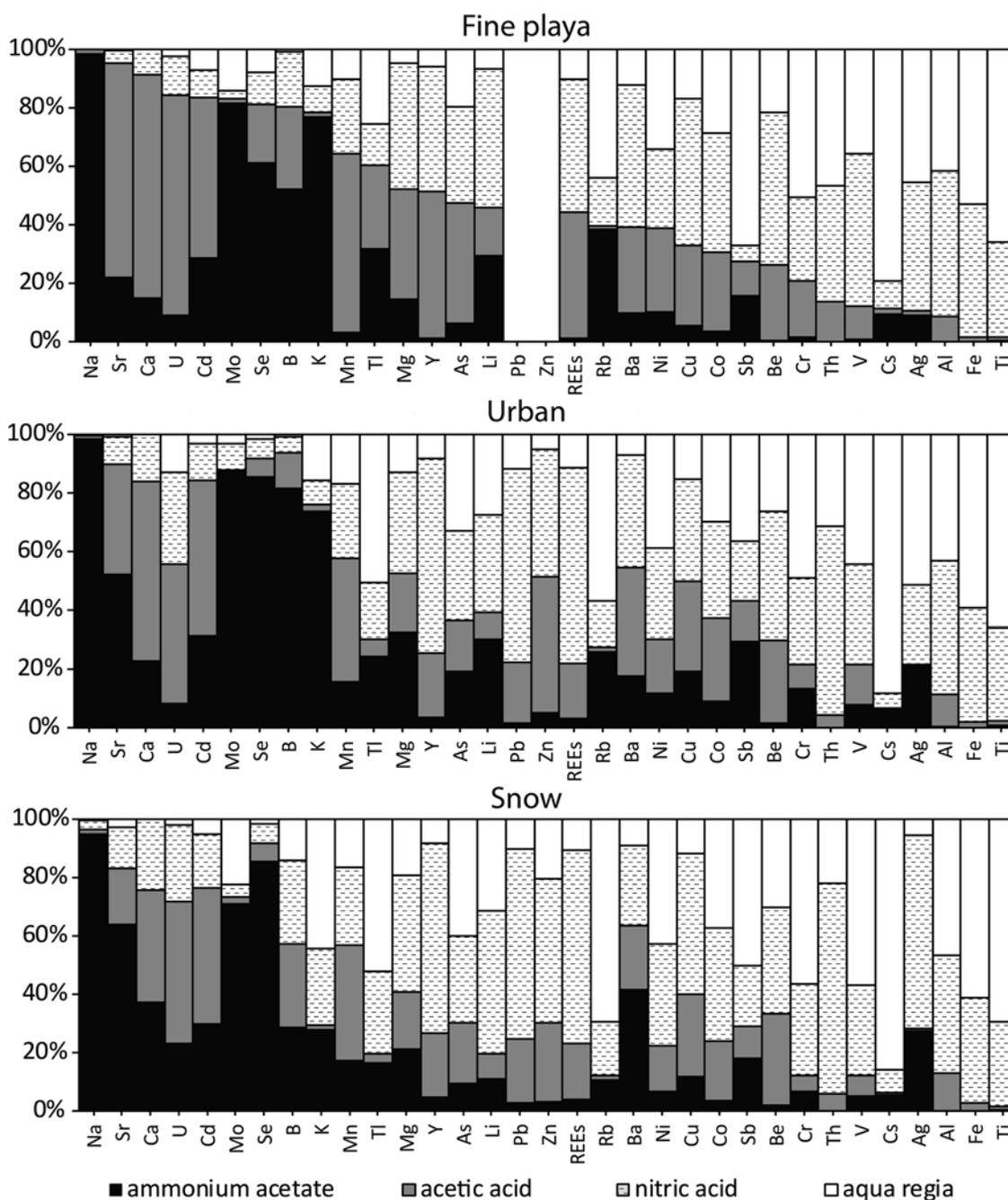


Fig. 7. Sequential leaching element fractions (expressed as a percentage of the total concentration) for fine playa ($n = 41$), urban ($n = 35$), and snow ($n = 22$) dust samples. The elements are sorted based on the combined fraction of ammonium acetate and acetic acid in the fine playa samples. Percent contributions from ammonium acetate and acetic acid leaching steps represent the environmentally available fraction. To simplify the plot, REEs (Yb, Lu, Er, Ho, Dy, Tb, Eu, Ce, Gd, La, Sm, Nd, Pr) were averaged because they had similar fractions in each leaching step in the fine playa, urban, and snow dust samples.

Silver is often correlated with Se because it is also derived from coal combustion (Lee et al., 1994; Salmon et al., 1978). Automotive brake abrasion dust and ash from waste incineration contribute Sb (Dietl et al., 1997; Iijima et al., 2009; van Velzen et al., 1998v; von Uexküll et al., 2005v). The majority of Cd in air emissions from urban areas is sourced from smelting/metallurgical processing and waste incineration (Fishbein, 1981). Other Cd sources include coating and plating in plastics, production of automobile radiators, and wear and emissions from tires, fuel, and oils (Ajmone-Marsan and Biasioli, 2010; Fishbein, 1981).

Lanthanum showed a unique enrichment pattern with elevated concentrations in SLC relative to other urban areas (Figs. 4 and 6). Oil refineries and petrochemical emissions are the primary La source in

urban areas (Kulkarni et al., 2006; Moreno et al., 2010, 2008; Olmez and Gordon, 1985). Typical La/Ce ratios in natural crustal rock ranges from 0.4 to 0.6 (Moreno et al., 2010; Rudnick and Gao, 2003). La/Ce ratios for urban dust samples were near crustal values in Provo, Ogden, and Logan (0.52 – 0.58) yet over 2.1 in SLC. The four-fold enrichment in the La/Ce ratio at SLC is likely due to oil refineries that are located < 10 km to the west of the SLC sampling site (Fig. 1).

To estimate dust contributions from playa sources relative to urban aerosol sources, we created a simple mass mixing model. Assuming the total dust mass deposited along the Wasatch Front is the sum of playa dust mass and urban aerosol mass, then for a given element X,

$$\begin{aligned} & \text{Conc}_{X_Playa} * \text{Mass}_{Playa} + \text{Conc}_{X_UrbanAerosol} * \text{Mass}_{UrbanAerosol} \\ & = \text{Conc}_{X_Total} * \text{Mass}_{Total} \end{aligned}$$

where Conc_{X_Total} is the concentration of element X in a dust deposition sample and Mass_{Total} is the mass of a dust deposition sample. For urban aerosol concentrations, we used PM_{10} data collected from high-volume air samplers in central Barcelona, Spain (Moreno et al., 2006). While Barcelona is not a perfect comparison with the Wasatch Front, it is the best available trace element dataset from an urban area with a large population. Assuming similar urban aerosol element concentrations between the Wasatch Front and Barcelona, the relative mass contribution from playas was estimated based on median playa and urban dust concentrations of Ti, V, Cr, Mn, Ni, Cu, Se, Mo, Cd, Sb, and Ba. Using these elements, the fraction of dust from playa sources ranged from 47 to 98%, with a mean of 91%. These calculations suggest that a large majority of the dust mass deposited along the Wasatch Front is sourced from regional playas. Only a small fraction of anthropogenic aerosols is required to generate the observed differences in urban dust relative to playa “background” dust. These results confirm the findings of other studies that show the importance of playa dust to the Wasatch Front based on backward-trajectory modeling (Mallia et al., 2017; Skiles et al., 2018) and satellite imagery (Hahnenberger and Nicoll, 2012; Steenburgh et al., 2012).

4.2. Dust deposition flux rates across the Wasatch Front are relatively higher than other US locations

Dust fluxes and element concentrations varied seasonally and spatially throughout the study. Our measured dust flux rates of 24–35 g/m²/yr in urban samples are higher than in other parts of the US but lower than major global dust areas. Dust fluxes averaged 3.5 g/m²/yr during 2011–2013 in the Uinta Mountains (Munroe et al., 2015), 2–20 g/m²/yr from 1984 to 1999 in the southwestern US (Reheis, 2006), and 5–10 g/m²/yr in the San Juan Mountains in Colorado (Lawrence et al., 2010). In the Negev Desert, Israel, dust accumulation rates ranged from 120 to 300 g/m²/yr (Offer and Goossens, 2001). In our samples, flux rates were typically higher in spring months, corresponding to the season with higher frequency and/or intensity of dust storms (Hahnenberger and Nicoll, 2012; Jewell and Nicoll, 2011). Flux rates were higher at SLC and Ogden relative to Provo and Logan, suggesting a greater influence of playa and/or urban aerosol inputs to those locations. Ogden and SLC are proximal to Great Salt Lake and likely receive more dust from nearby playa sediments. SLC also likely receives the most local anthropogenic dust because it has the highest population density in Utah, the most industry, and is closest to the Kennecott Copper Mine.

Seasonal differences in urban dust chemistry may indicate relative contributions from playa versus local anthropogenic dust inputs. The peak in Sr and Na concentrations and dip in Co, Se, and REE concentrations during the January–March sampling period suggests a larger input of playa dust and a dilution of urban aerosols (Supplementary material, Fig. S7). Specifically, we observed a large dust event on 18 February 2018 with significantly decreased visibility across northern Utah. The event darkened mountain snowpack and the local avalanche center reported multiple observations of the dusty snow layer. This dust event, among others during January through March 2018, likely contributed to increase in Sr and Na concentrations and decrease in urban-sourced elements like Co and Se.

4.3. Implications of dust deposition for human and ecosystem health

The sequential leaching results have implications for understanding the environmental availability of specific elements (Fig. 7). Elements that are leached by ammonium acetate and acetic acid are likely readily mobilized into soils or water bodies. These “available” elements are also more likely to impact human health as they are inhaled and dissolved

into the bloodstream. Readily reacted minerals include evaporite minerals such as halite and gypsum in the ammonium acetate leaching step and carbonate minerals such as calcite, aragonite, and dolomite in the acetic acid leaching step (Dastrup et al., 2018). Because playas have higher concentrations of soluble minerals compared with urban and snow dust samples, elements commonly found in those minerals such as Na, B, Ca, Sr, and U have high (> 50%) fractions in ammonium acetate and acetic acid leaching steps (Fig. 7). Additionally, Se and Cd have high environmental availability (> 60% leached in the ammonium acetate and acetic acid steps) in playa, urban, and snow dust samples (Fig. 7), with potentially negative consequences for ecosystems.

Immobile elements that are primarily leached by aqua regia also play an important role in the ecosystem because they aid in soil formation (Dastrup et al., 2018; Lawrence et al., 2013). The immobile elements, including Cr, Cs, Fe, and Ti, likely accumulate in soils rather than being dissolved in aquatic systems. On geologic time scales, soils with anomalously high concentrations of metals such as Cr and Ti are attributed to dust deposition rather than local bedrock weathering (Lawrence et al., 2013; Munroe, 2014).

5. Conclusions

Dust deposition is often a mixture of mineral dust and anthropogenic aerosols. In our study on northern Utah dust, the suite of silicate, carbonate, and evaporite minerals and grain size distributions in playa dust sources were similar to urban and snow dust deposition, suggesting that playas are the dominant source of dust to the Wasatch Front. Playas are enriched in Ca, Li, Mg, Na, Sr, and U from evaporite and carbonate minerals. Dust deposition is enriched in several trace elements, including Se, Ag, Cd, Sb, and La, which are sourced from industrial activity, mining, and vehicles. Based on mass balance calculations using element concentrations, playas contribute ~90% of urban dust with only minor contributions from urban aerosols. Dust deposition fluxes were greatest in the spring, while concentrations were similar year-round for most elements. Many elements were readily leached with weak acids, including potentially harmful elements such as Cd and Se, suggesting that dust may play an important role in trace element cycling in terrestrial and aquatic ecosystems. By comparing dust sources with dust deposition in a major urban area, our study has implications for understanding dust transport mechanisms from source to sink and understanding the major components of urban atmospheric deposition. A deeper understanding of dust sources is useful for understanding how water diversions, land use changes, and population growth may affect the regional dust cycle in the future.

Data availability

All raw data associated with this study are contained in the supplementary tables.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:<https://doi.org/10.1016/j.chemgeo.2019>.

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**Updated Interim Report on Nutrients in
Precipitation on Utah Lake
-Bulk Deposition-**

Report to Wasatch Front Water Quality Council

By
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Dec 2019
Provo, Utah

Updated Interim Report on Nutrients in Precipitation on Utah Lake

Wood Miller – 19 December 2019

The purpose of this study is to evaluate the impact of atmospheric deposition of nutrients onto the surface of Utah Lake. It appears that no nutrient budget for Utah Lake in the past has accounted for the phosphorus and nitrogen from the precipitation on the lake. This contribution of TP and TN to the nutrient budget is not insignificant. We have found that the phosphorus in the rain on the lake alone is likely enough phosphorus to keep the lake eutrophic and produce algae blooms.

I have included the most relevant tables, figures, and maps into the body of this report. All other tables, figures, and maps referred to in the report are included in the Appendix.

We have collected precipitation samples from the rain and snow storms over the past 3 years at 9 locations around Utah Lake. Over 380 samples have been obtained and all of them have been analyzed for Total Phosphorus TP (and Ortho-Phosphorus during 2019) and Total Nitrogen TN concentrations at Chemtech-Ford Laboratories. We are very appreciative to Paul Ellingson and the personnel at Chemtech-Ford for having all the analysis done at no charge.

Figure A shows our 9 precipitation sampling sites around Utah Lake. Locations #1 BYU, #7 Utah Lake Lehi, and #9 Spanish Fork are official NOAA NWS weather stations. They save the water collected in their 3 ft high gages for me each day it rains and I obtain the water from them within 1 or 2 days. They keep the precipitation sample cool and sealed, but they probably don't clean out their collector tube very well between daily measurements. At these stations, the funnel is used in the summer, but not in the winter. We think the evaporation from the collector is minimal because the precipitation is retrieved within hours of the rain event. Any TP falling into the collector between storms would have presumably also fallen on the lake.

The other 6 sampling sites shown on Figure A have standard precipitation gages which we set up for this bulk atmospheric deposition research project. The rain collectors consist of ~ 5 in. diameter plastic tubes along with ~ 1.5 ft. diameter plastic funnels above the tubes. The funnels provide for more water collection from each storm. These collectors are all in open spaces and are all over 6 ft. above the ground.

The tubes and funnels are washed out each time we take a sample. We remove all bugs and debris from the sample at the time of sampling. We do not filter the samples, but we take off the floating material and do not pour out the settled material. We put the cleaned water in acid-fixed sample bottles, keep them cool, and take them to Chemtech-Ford Lab within a few days. Precipitation samples are collected in a timely manner; on the same day or within 1 or 2 days after a significant and measureable storm.

We understand that our sampling is a “first order approximation of bulk deposition” and does not follow the NADP protocol. Our objective since we started sampling 3 years ago has been to collect precipitation samples around the lake and estimate the bulk atmospheric nutrient deposition. Possibly our method and collectors would be considered an “NADP Provisional Method.” Or as Dr. David Gay put it, “the study does show a low-cost method for collecting basic deposition information for the area.”

Chemtech-Ford Laboratories is certified by the State of Utah department of health environmental laboratory certification program. Figure B is the Lab Certification Certificate. For our bulk atmospheric deposition project, the Lab runs the following methods: Ortho-Phosphorus by Standard Method SM 4500 P-E, Total Phosphorus by Standard Method SM 4500 P-E/F, Nitrate+Nitrite by Standard Method SM 4500 NO₃-F, Ammonia by Standard Method SM 4500 NH₃-H, and Total Inorganic Nitrogen is calculated by adding the Nitrate+Nitrite and the Ammonia.

As shown on Figure A, site #1 BYU is located in a campus urban area. Site # 2 Lincoln Pt is located on the Utah Lake shoreline in an orchard area. Site #3 Pelican Pt is also on the lake shoreline in an open grassland area. Site #4 Genola is in an orchard area near the far south end of the lake. Site #5 Elberta is farther south of the lake in farm and open grassland area. Site #6 Mosida is very near the lake in open grassland area. Site #7 Utah Lake Lehi is in a farm area on the north lakeshore near the pump station and the Jordan River outlet. Site #8 Orem is in the WWTP area and within open park areas. Site #9 Spanish Fork is in the foothills area also within open park areas.

Sites #2 to #4 and #6 to #8 are all very close to the lake. Sites 3, 7 & 8 are on the north shore while Sites 2, 4 & 6 are on the south shore. Site #5 collects the precipitation from the southwest which is the direction of the prevailing pre-storm winds in Utah Valley. Figures C and D are 2 typical wind roses for the Provo Airport. These southwest winds potentially carry dust from the Utah west desert and dry lake beds. The only sampling stations which are in urban areas are #1 BYU and #8 Orem. We suggest that the water collected at all these locations represents the quantity and quality of the precipitation across and around the lake very well.

We are continuing to collect data and draw conclusions about this bulk deposition. We realize there are questions about just how representative of actual surrounding areas the samples are. How much dry-deposition nutrients might be washed down into the actual gage by the rain/snow as compared to before that precipitation enters the gage? And what conditions or features might make a station more or less susceptible to accumulation of dry deposition being included in the sample?

We recognize that it is almost impossible to get just a separate, accurate wet component without using an automated device that opens only during actual precipitation and closes when none. Therefore, our bulk deposition results do include some dry deposition but are largely for the wet deposition. These results may be somewhat different than the total wet and dry components that graduate students Jacob Olsen, Josh Reidhead, and Seth Barrus have been measuring.

We have developed a large and valuable spreadsheet with all the Utah Lake precipitation phosphorus and nitrogen data for the study of bulk atmospheric nutrient deposition on Utah Lake. We have summarized much of the data in the spreadsheet into Tables 1a and 1b which give TP, Ortho-P and TN averages at all 9 locations and the overall summary averages.

Figure 1a and Map 1a show the TP averages from Table 1a and Figure 1e and Map 1e show the TN averages from Table 1b. As the figures and maps show, the average TP and TN concentrations are quite different at the different locations.

Table 1a and 1b and the figures also show the concentrations without outliers. For TP values, outliers are > 1 and > 5 mg/l, and for TN values, outliers are > 10 mg/l. The results for average concentrations without outliers for TP are given in Figure 1a and Maps 1aa and 1ab, and for TN in Figure 1e and Map 1ea. What we are calling outliers are values which seem to be unreasonably high. We don't exactly know why they are so high. There are no field records which might indicate excessive contamination or mishandling or other reasons. In some, even many cases, they may be actual measured high concentrations.

Tables 2 – 10 (not given in this report) are separate tables with all the sample data and averages for each of the 9 locations. Each table is respectively followed by 5 graphs, 45 graphs in total, (2a – 10e) (not given here) which show the averages in the tables. Only the table and 5 graphs for Lincoln Pt., Table 3 and Figures 3a – 3e, are included with this report as examples.

There is also the need to observe and compare the available weather data associated with these precipitation samples. We have added to the spreadsheet the actual precipitation amounts at the BYU and Spanish Fork weather stations on the day of and day before our measurements. And we included the wind directions at the Salt Lake Airport, the nearest wind data station, on the day of and day before our measurements.

As shown on Table 1a, the highest TP concentrations w/o > 1 mg/l outliers are 0.33 and 0.31 mg/l at Elberta and Mosida, south and west of the lake. Most rain storms, along with dust storms, come from the southwest. The lowest TP concentrations w/o > 1 mg/l outliers are 0.10 and 0.14 mg/l at BYU and Spanish Fork, east and away from the lake. The high concentrations are about 3 times higher than the low concentrations. There are no TP or TN outliers at the BYU location.

As shown on Table 1b, the highest TN concentration w/o > 10 mg/l outliers, is 2.64 mg/l at Mosida. The lowest TN concentrations w/o > 10 mg/l outliers is 1.51 mg/l at Spanish Fork, away from the lake. BYU in its urban setting is not low in this case. The high concentration is only about 2 times higher than the low concentration.

Initially we wanted the averages of only the samples less than 1 mg/l for TP, which is 0.22 mg/l, and less than 10 mg/l for TN, which is 2.15 mg/l. All the higher concentration samples were considered to be outliers. For example, there are 61 TP samples greater than 1 mg/l, or 61 outliers, or 16 % of all 370 samples, and 15 TN samples greater than 10 mg/l, or 15 outliers, or 4% of all 354 samples.

However, we determined that our initial TP threshold for so-called outliers may be too low; that there is the distinct possibility of having reasonable TP concentrations higher than 1 mg/l. Insects and other large or small debris were completely removed from all the samples, without filtering. But still there are high concentration sample results. Possibly parts of insects and other debris are dissolved in the sample, but often the rain collected is just plain dirty, particularly rain collected during and after our very common dusty southwest windstorms.

Therefore, Table 1a also shows that we have determined the TP averages for the samples less than 5 mg/l to be 0.46 mg/l with 15 outliers, and for all the 370 samples to be 0.77 mg/l, with no outliers. Notice that the average TN concentration w/o 15 outliers (TN > 10 mg/l) is 2.15 mg/l in Table 1b, and the average TN concentration for all data, no outliers, is 3.08 mg/l.

As shown on Table 1a, the highest TP concentrations w/o > 5 mg/l outliers are 1.07 and 0.55 mg/l at Mosida and Lincoln Pt, south of the main lake. Many storms come from the south. The lowest TP concentrations w/o > 5 mg/l outliers are 0.10 and 0.25 mg/l again at BYU and Spanish Fork, east and away from the lake. The high concentrations are about 2 to 10 times higher than the low concentrations.

As also shown on Table 1a, the highest TP concentrations w/ no outliers are 1.35 and 1.16 mg/l at Genola and Lincoln Pt, south of the main lake. The lowest TP concentrations w/ no outliers are 0.10 and 0.25 mg/l at BYU and Spanish Fork, away from the lake. The high concentrations are about 5 to 12 times higher than the low concentrations.

As shown on Table 1b, the highest TN concentrations w/ no outliers are 5.43 and 5.15 mg/l at Mosida and Lincoln Pt. The lowest TN concentrations w/ no outliers are 1.51 and 1.98 mg/l at Spanish Fork and Elberta. The high concentrations are about 4 times higher than the low concentrations.

We also determined the average concentrations at each of the 9 locations for all the summer (Apr-Sep) samples and for all the winter (Oct-Mar) samples. These TP results are shown in Table 1a, on Figure 1b and Maps 1ba, 1bb & 1bc for summer, and Figure 1c and Maps 1ca, 1cb & 1cc for winter. The TN results are also in Table 1b, on Figure 1f and Maps 1fa & 1fb for summer, and Figure 1g and Maps 1ga & 1gb for winter.

The tables and figures also show the overall averages of all the summer and winter data at all the stations. Most of the summer and winter concentrations are quite different, with the summer values generally being higher, sometimes very much higher. The averages for all the data are of course in between the summer and winter averages.

For the TP, the summer overall averages are about 2 or 3 times higher than the winter overall averages. The individual locations' TP summer averages are as much as 4 or 5 times higher than winter averages. For the TN, the summer averages are generally only slightly higher than the winter averages.

Figure 1i shows the Ortho-P along with the TP < 5 mg/l averages from Table 1a for comparison. Map 1i shows the contours of Ortho-P averages on the lake based on the data from the 8 sites (BYU has no O-P data). As the figure and map show, the average Ortho-P concentrations are fairly similar at the different locations, except for Mosida. No outliers were considered for Ortho-P samples.

We also determined the average Ortho-P concentrations at each of the 8 locations for all the summer (Apr-Sep) samples and for all the winter (Oct-Mar) samples. These O-P results are again given in Table 1a, on Figure 1j and Map 1j for summer, and Figure 1k and Map1k for winter. Figure 1l is for the loadings (tons/year) of O-P.

The tables and figures also show the overall averages of all the summer and winter data. Most of the summer and winter concentrations are quite different among the locations. Table 1a shows that at 5 sites the summer values are higher than winter, at 1 site the winter value is higher, and at 2 sites they are about the same. Therefore, the Maps 1j & 1k look quite different. The summer overall average is about 4 times higher than the winter overall average. The averages for the all data are of course in between the summer and winter averages.

The results show that the average TP concentration of 0.77 mg/l with all samples is almost 4 times larger than the average TP concentration of 0.22 mg/l without the 61 outliers of TP > 1 mg/l. We have calculated that a concentration of about 0.2 mg/l of TP in the approximate 12 inches/year of precipitation on an approximate 84,000 ac lake results in a loading of more than 20 tons of TP/year on the lake. At a concentration of ~ 4 times (~ 0.8 mg/l), the loading on the lake would be more than 80 tons of TP/year.

These loadings are also shown in Tables 1a and 1b and on Figures 1d, 1l and 1h for TP, Ortho-P and TN respectively. Of course, the loading values are dependent and vary with lake surface area. The area varied by about +/- 6 % each year during this study period, or between ~ 79,000 and 89,000 acres. Therefore, the 0.2 mg/l TP loadings would vary by about the same range, +/- 6 %, or ~ 19 and 21 tons of TP/year. This variance is within the range of the approximations for the calculations.

In conclusion, I have compared my results from 3 years of collecting and analyzing bulk TP data with the approximate but representative results of our graduate students who have for the most part followed the NAPD rules and protocol in their TP sampling and analyzing. Jacob Olsen conducted his research during 2017 and Josh Reidhead during 2018. Figure 11a is the map of the sampling sites around Utah Lake with the overall average bulk TP concentrations from each of the 3 studies listed below the site names. Data are given at my 9 sites and at the students' 5 sites which are at approximately the same locations as 5 of my 9.

Figure 11b is a bar graph which shows the overall average TP concentrations at all the locations from the 3 studies. The Reidhead (2017) avg concentrations are generally the highest, with Olsen (2018) next, and my (Miller, 2017 – 2019) concentrations the lowest. TP concentration values at the 5 common locations are all within less than 1/3 of an order of magnitude of each other; a range of 1 to 3 mg/l. It's reassuring that all these TP values are quite similar.

Other observations and evaluations on our study of the bulk atmospheric nutrient depositions are given below:

Dr. LaVere Merritt wrote, "Perspective on atmospheric nutrients in Utah Valley:

Concentrations of nutrients needed to become firmly limiting to algae growth: TP = 0.01 - 0.02 mg/l and DN = 0.10 - 0.15 mg/l.

... to support eutrophic algae growth: TP = 0.03 - 0.04 mg/l & DN = 0.30 - 0.35 mg/l

... in Utah valley precipitation Dec 2016-Apr 2017: TP = 0.12 mg/l & DN = 2.2 mg/l"

Now: Dec 2016 - Mar 2019: TP ~ 0.22 mg/l w/o TP > 1 outliers, and ~ 0.8 w/ all data.

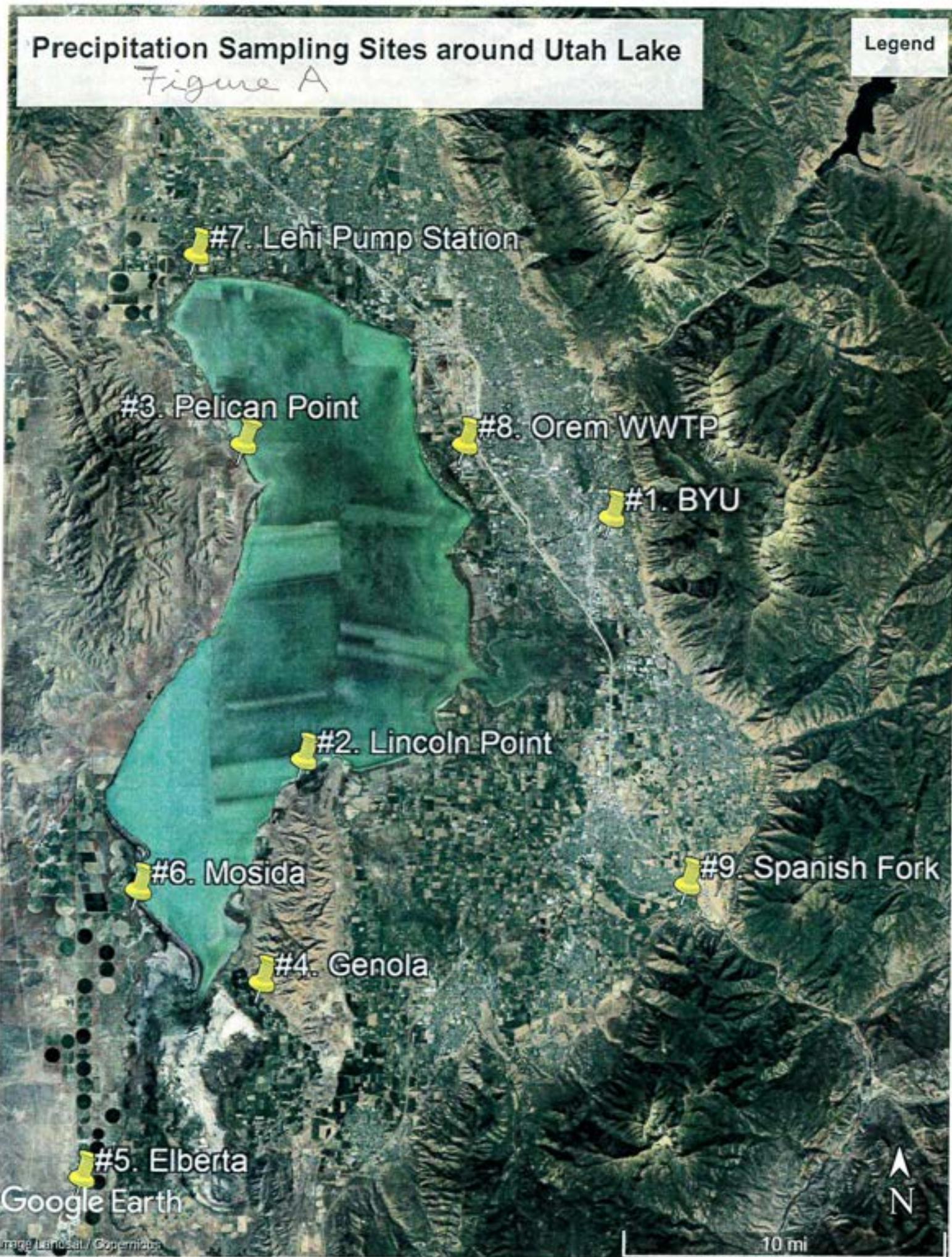
TN ~ 2.2 mg/l w/o TN > 10 outliers, and ~ 3.3 w/ all data.

Dr. LaVere Merritt also wrote, "The precipitation nutrient results tell us that there is about 3 to 4 times more phosphorus and nitrogen in the wet precipitation than needed to support eutrophic algal growth levels. When we add the dust deposition then this increases to many times more. This means that essentially all of the surface ponds and lakes in our lowland areas are receiving far more nutrients than the algae can use--so nutrients are not the issue and essentially nothing can be done to change the amount of algae growing in these waters-chemical treatment of the larger waters is just too expensive or difficult also".

Precipitation Sampling Sites around Utah Lake

Figure A

Legend





[U24] DELTA
Windrose Plot [Time Domain: Apr,]
Period of Record: 01 Apr 1973 - 30 Apr 2012

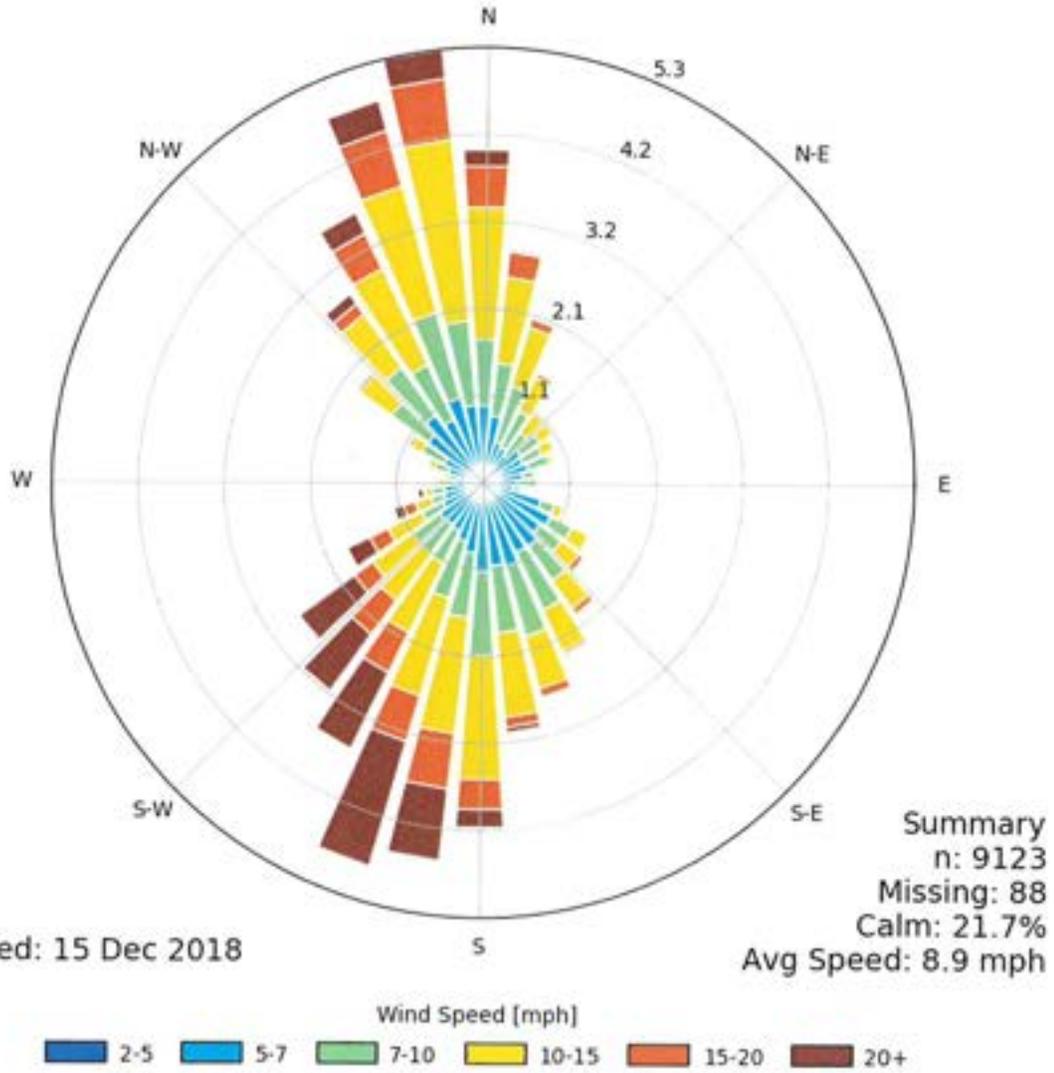


Figure D

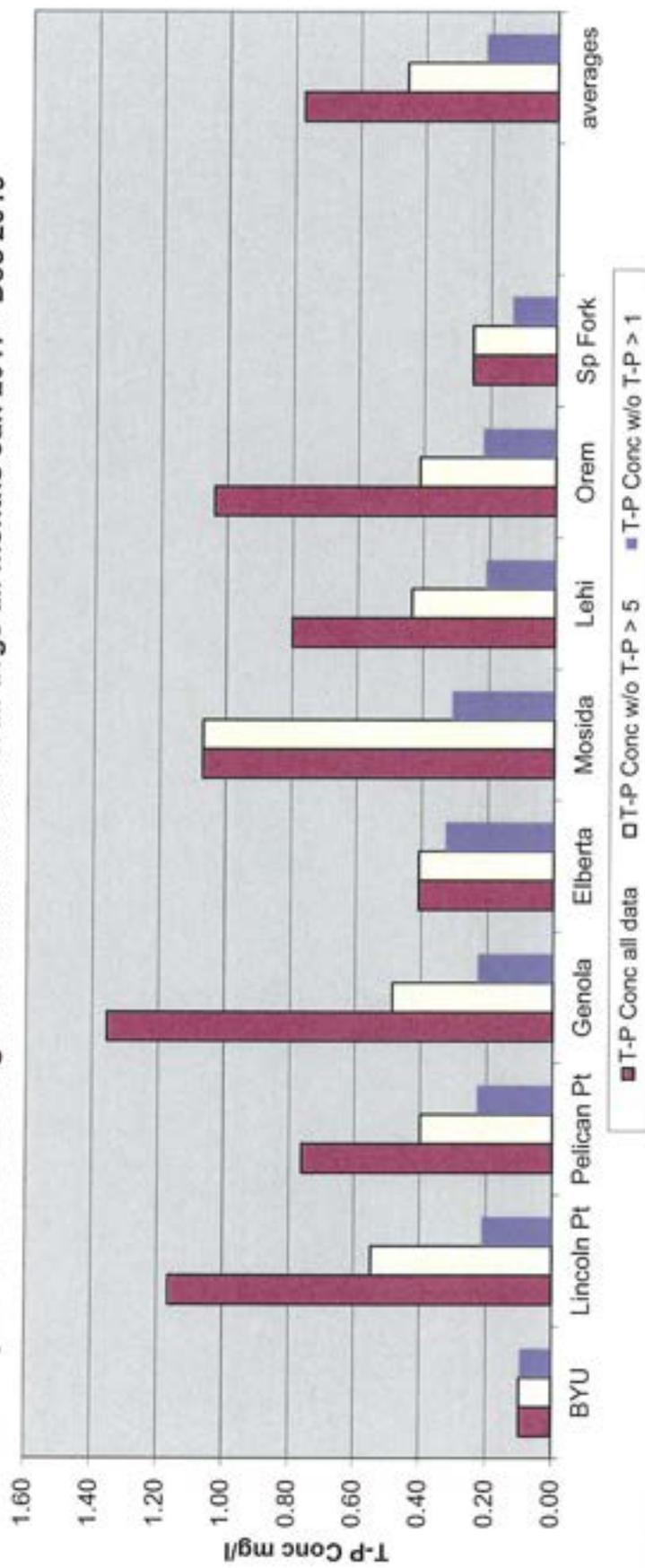
Table 1a. Averages of all 9 locations from all nutrient samples for the whole year and for summer (Apr-Sept) and winter (Oct-Mar).

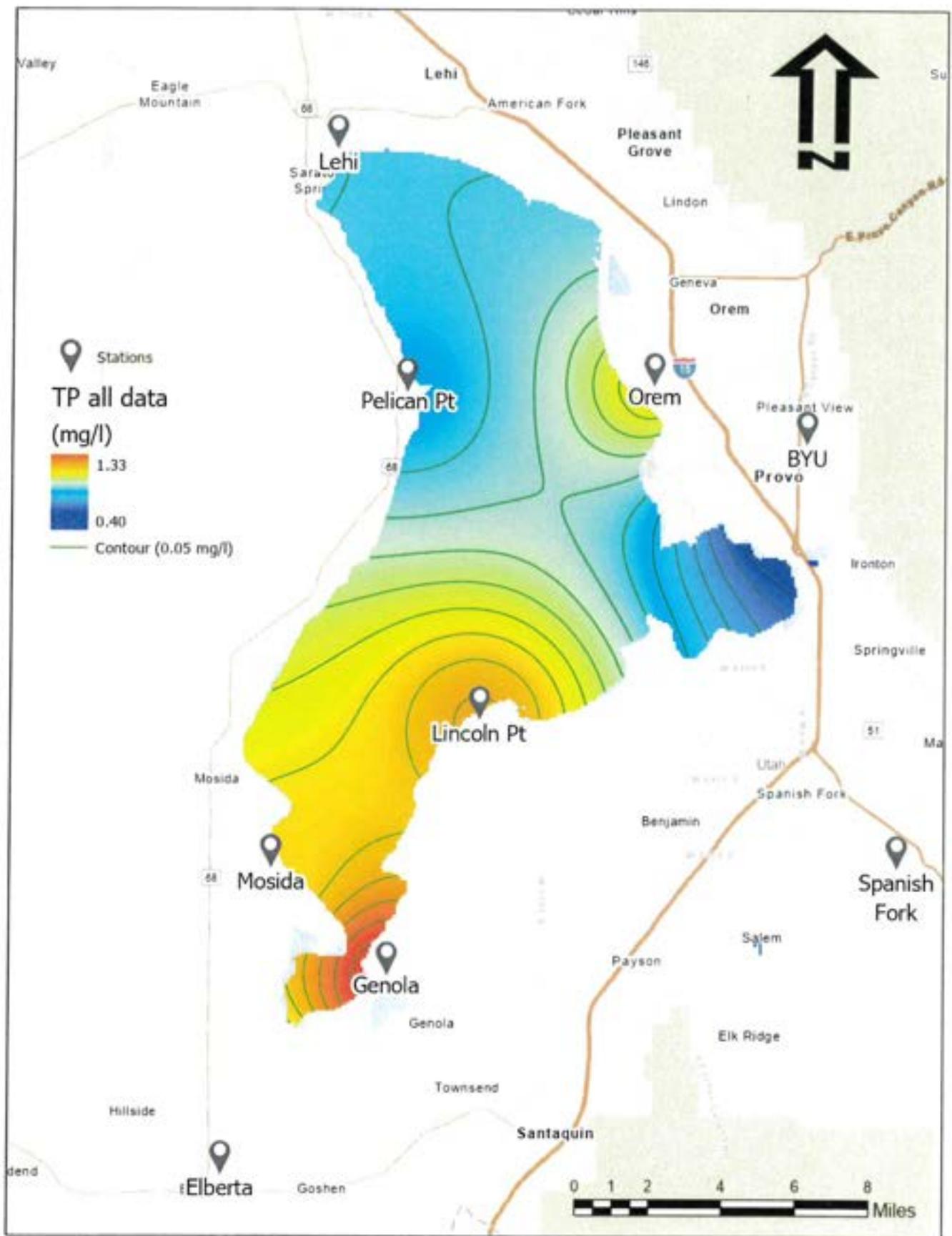
Location	Total Phos (mg/l)		all TP >1 mg/l >5 mg/l	Ortho-P (mg/l)		all Ortho-P samples								
	all data	summer	winter	TP < 1	summer	winter	TP < 5	summer	winter	TP		summer	winter	
BYU	0.10	0.12	0.09	0.10	0.12	0.09	0.10	0.12	0.09	43	0	0.00	0.00	0
Lincoln Pt	1.16	1.70	0.52	0.21	0.32	0.12	0.55	0.80	0.27	44	12	0.54	0.73	4
Pelican Pt	0.76	0.80	0.72	0.23	0.23	0.24	0.40	0.43	0.37	38	6	0.11	0.11	2
Genola	1.35	2.06	0.21	0.23	0.25	0.21	0.49	0.70	0.21	42	10	0.14	0.20	5
Elberta	0.41	0.39	0.43	0.33	0.35	0.30	0.41	0.39	0.43	42	3	0.12	0.12	0
Mosida	1.07	1.50	0.34	0.31	0.38	0.24	1.07	1.50	0.34	35	11	0.95	1.21	0
Lehi	0.80	1.12	0.38	0.21	0.27	0.15	0.43	0.48	0.38	46	9	0.11	0.11	2
Orem	1.04	1.42	0.33	0.22	0.24	0.20	0.41	0.46	0.33	40	8	0.20	0.24	2
Sp Fork	0.25	0.40	0.09	0.14	0.18	0.09	0.25	0.40	0.09	40	2	0.09	0.12	0
averages above avg no.samples	0.77	1.06	0.35	0.22	0.26	0.18	0.46	0.59	0.28	370	61	0.25	0.31	15
tonsTP/yr at avg area 83,800 ac & 12"/yr rain or 6"/half yr at given avg TP conc.	370	202	168	309	154	155	355	189	166	383 including BDL as of Dec 16	13	70	46	24
	87.9	60.1	19.7	25.1	14.7	10.3	51.9	33.3	15.8	28.4	17.9	4.6		

Table 1b. Averages for Nitrogen

Location	Total Nitro (mg/l)		Total Nitro (mg/l)		Total Nitro (mg/l)		all TN samples	TN outliers >10 mg/l
	all data	summer	winter	TN <10	summer	winter		
BYU	2.24	2.20	2.26	2.24	2.20	2.26	39	0
Lincoln Pt	5.15	5.75	4.12	2.32	2.56	1.92	41	6
Pelican Pt	4.56	6.21	2.74	2.33	2.37	2.30	38	2
Genola	2.08	2.63	1.23	1.82	2.21	1.23	38	1
Elberta	1.98	1.65	2.32	1.98	1.65	2.32	36	0
Mosida	5.43	5.57	5.21	2.64	2.62	2.67	34	4
Lehi	2.45	3.10	1.73	2.45	3.10	1.73	44	0
Orem	2.33	2.60	1.87	2.08	2.21	1.87	38	2
Sp Fork	1.51	1.87	1.14	1.51	1.87	1.14	46	0
averages	3.08	3.51	2.51	2.15	2.31	1.94	354	15
no.samples	354	193	161	339	182	157	382 including 28 BDL as of Dec 16	
tonsTN/yr at avg area 83,800 ac & 12"/yr rain or 6"/half yr at given avg TN conc.	350.9	199.7	143.2	245.2	131.5	110.4		

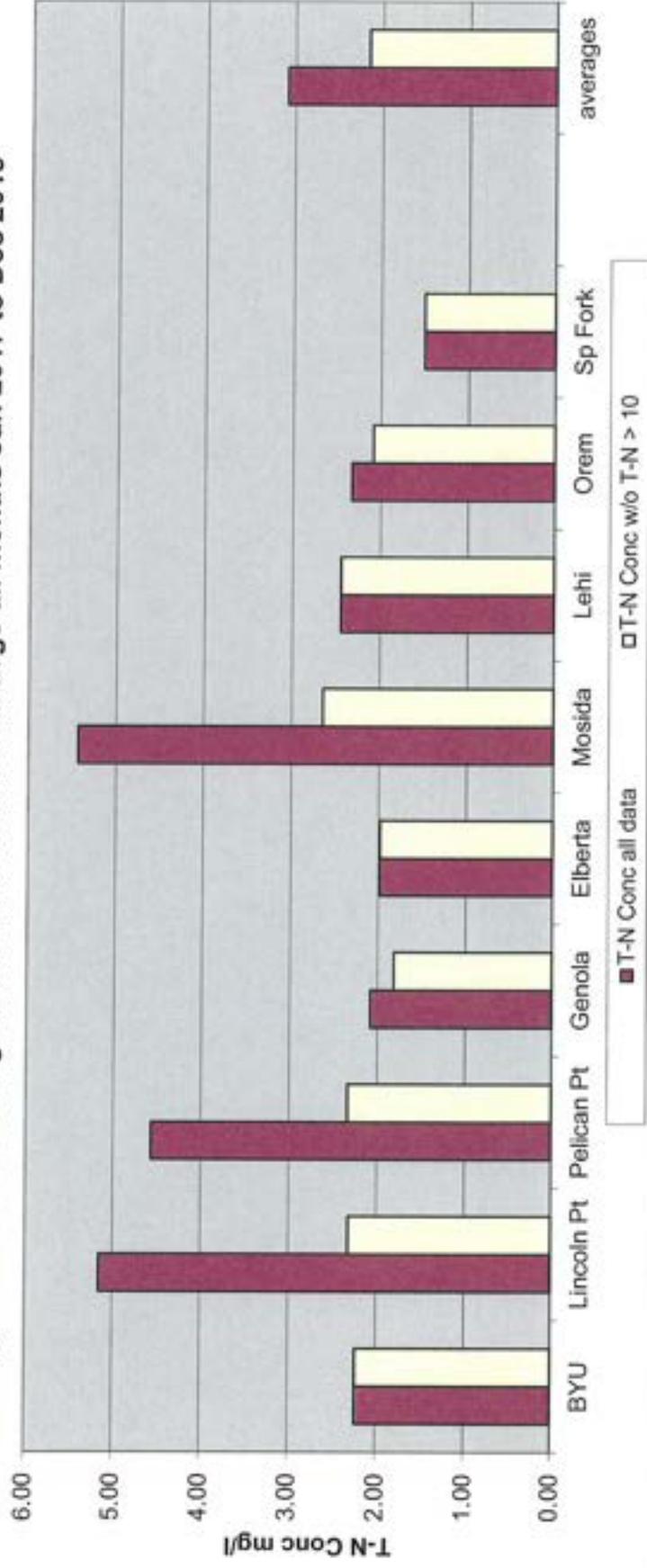
Figure 1a. T-P Conc avgs at 9 locations & overall avgs all months Jan 2017 - Dec 2019

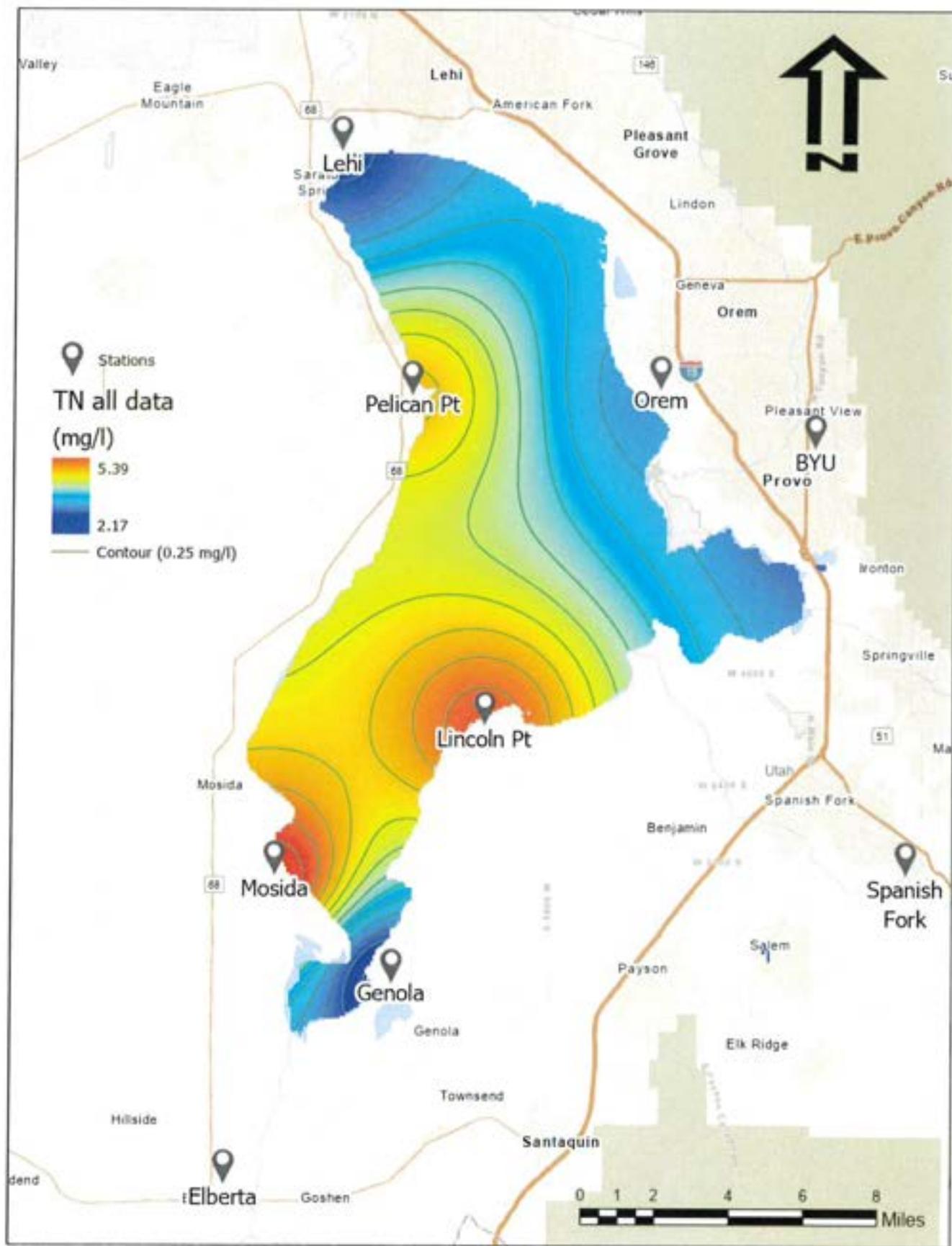




Map 1a

Figure 1e. T-N Conc avgs at 9 locations & overall avgs all months Jan 2017 to Dec 2019





Map 1e

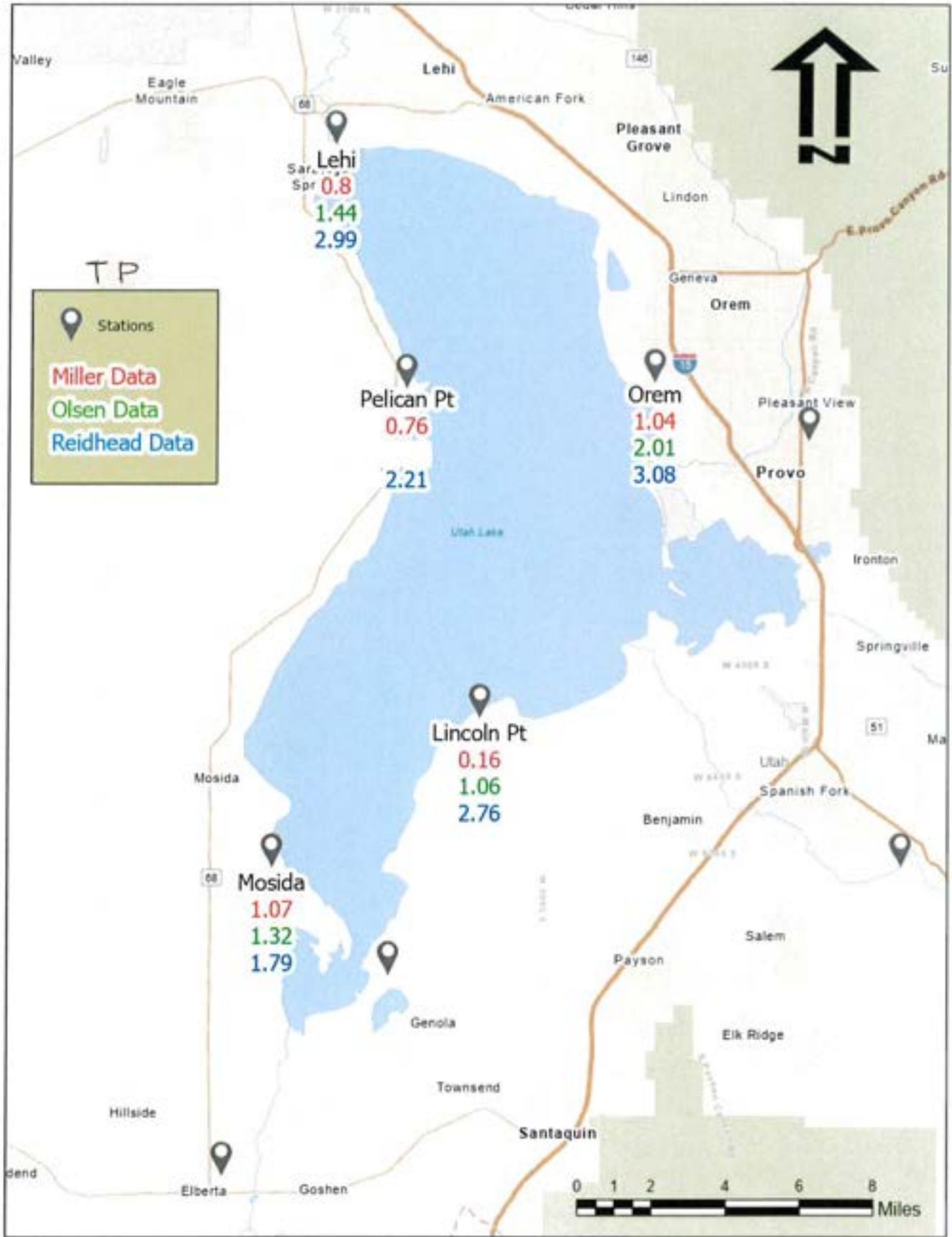
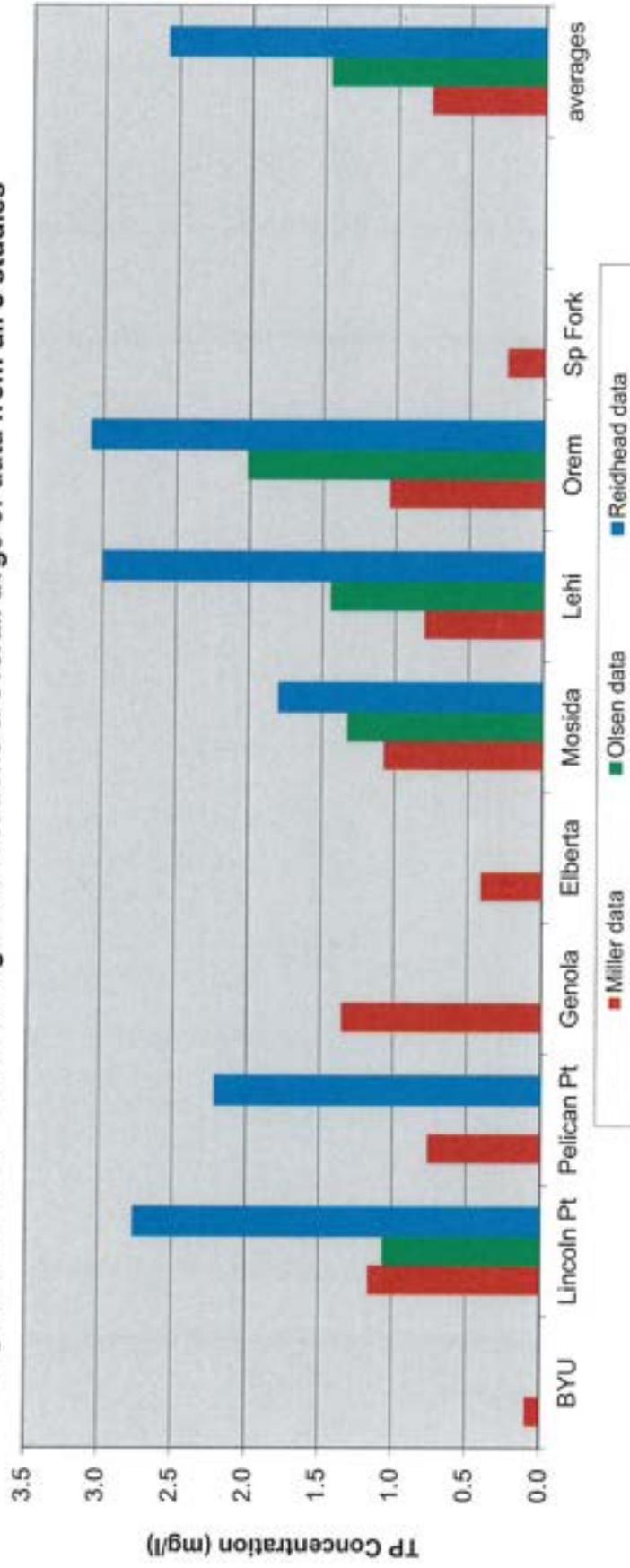


Figure 11a

Figure 11b. "Bulk" TP Conc avgs at 9 locations & overall avgs of data from all 3 studies



APPENDIX

State of Utah

Department of Health

Environmental Laboratory Certification Program

Certification is hereby granted to

Chemtech Ford Laboratory

9632 South 500 West
Sandy, UT 84070

*Has conformed with the
2009 TNI Standard*

*Scope of accreditation is limited to the
State of Utah accredited fields that accompany
this Certificate*

EPA Number: UT00027

Expiration Date: 6/30/2020

Certificate Number: UT000272019-24



Robyn M. Atkinson, Ph.D, HCLD
Director, Utah Public Health Laboratory



Continued accredited status depends on successful ongoing participation in the program.

Figure B



[U24] DELTA
Windrose Plot [Time Domain: Aug,]
Period of Record: 01 Aug 1973 - 01 Sep 2012

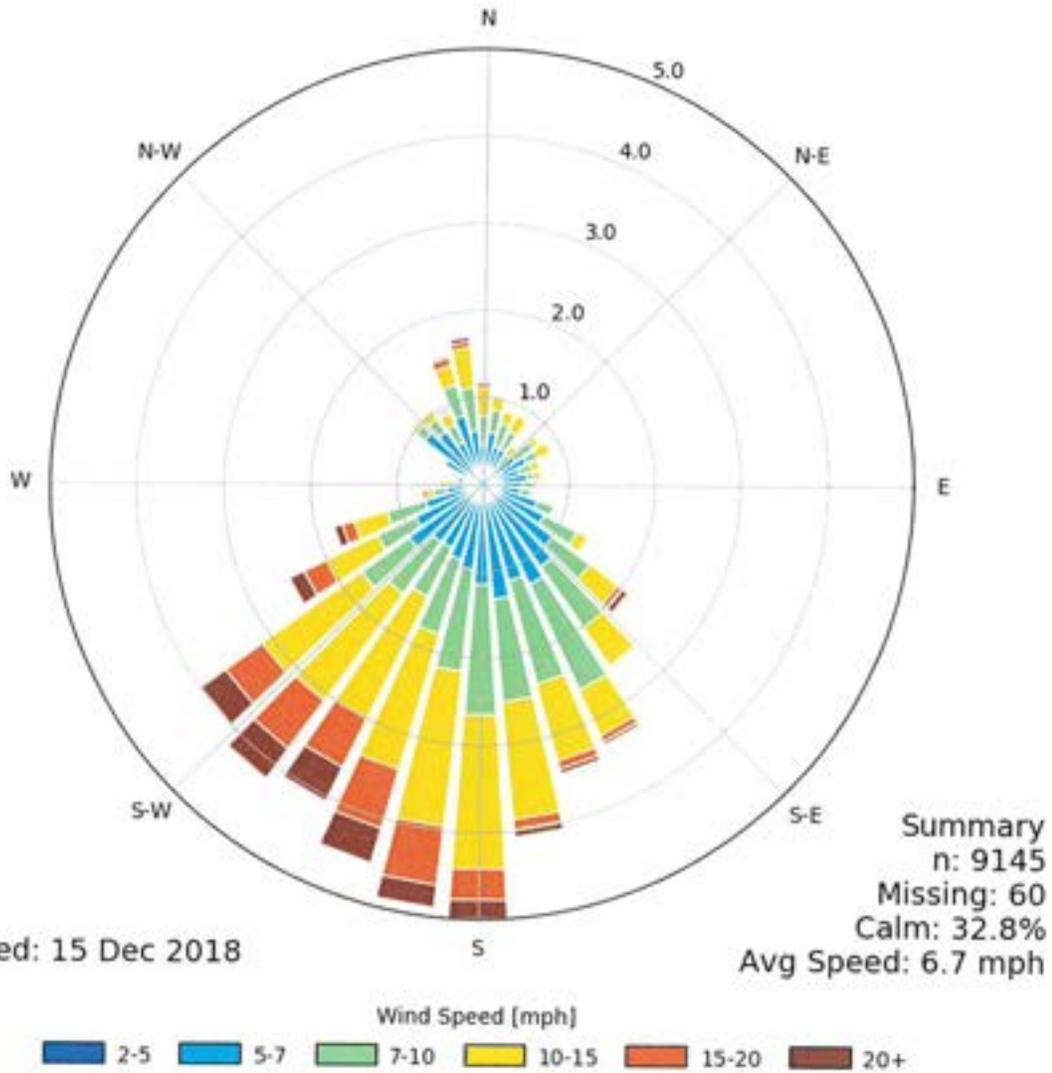
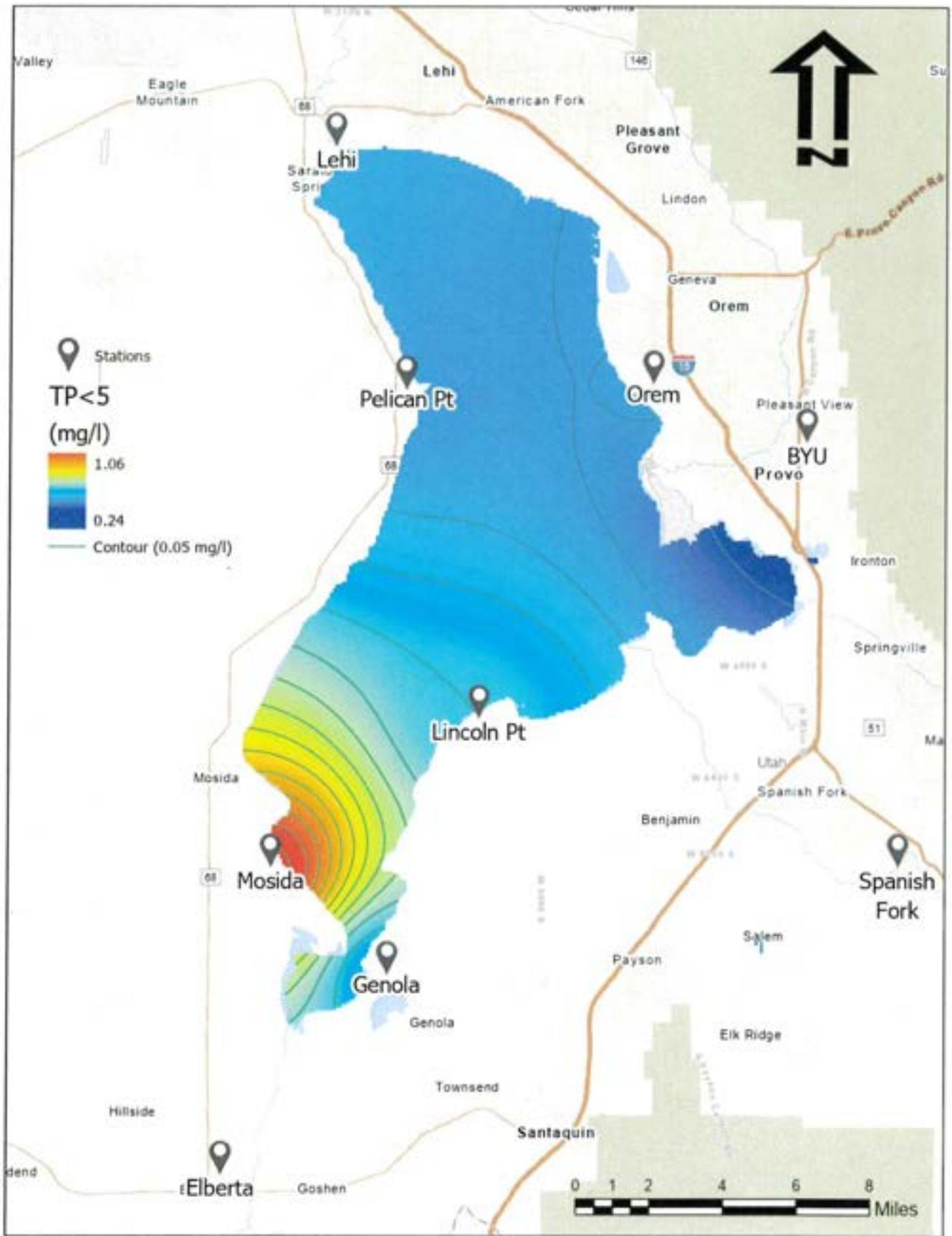
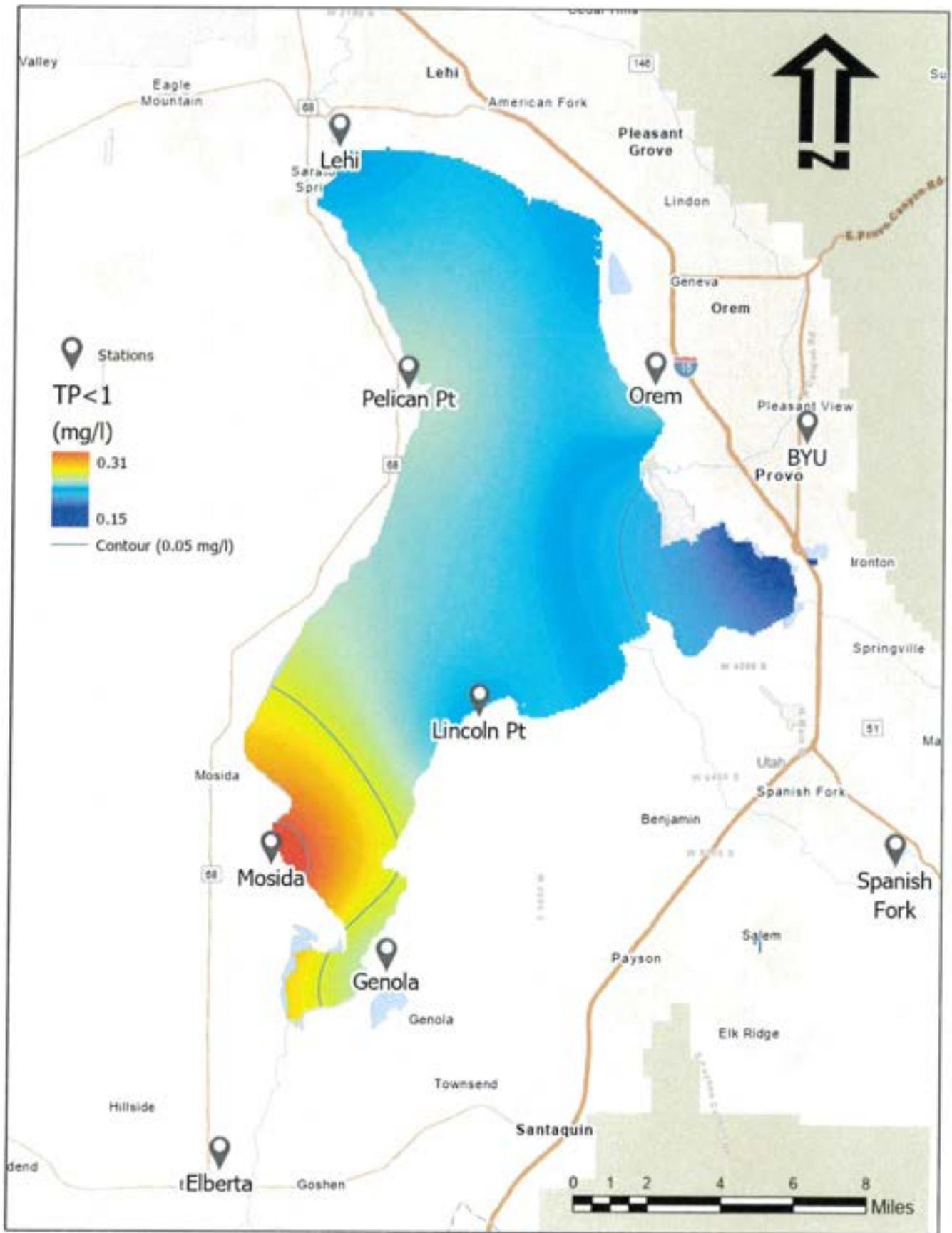


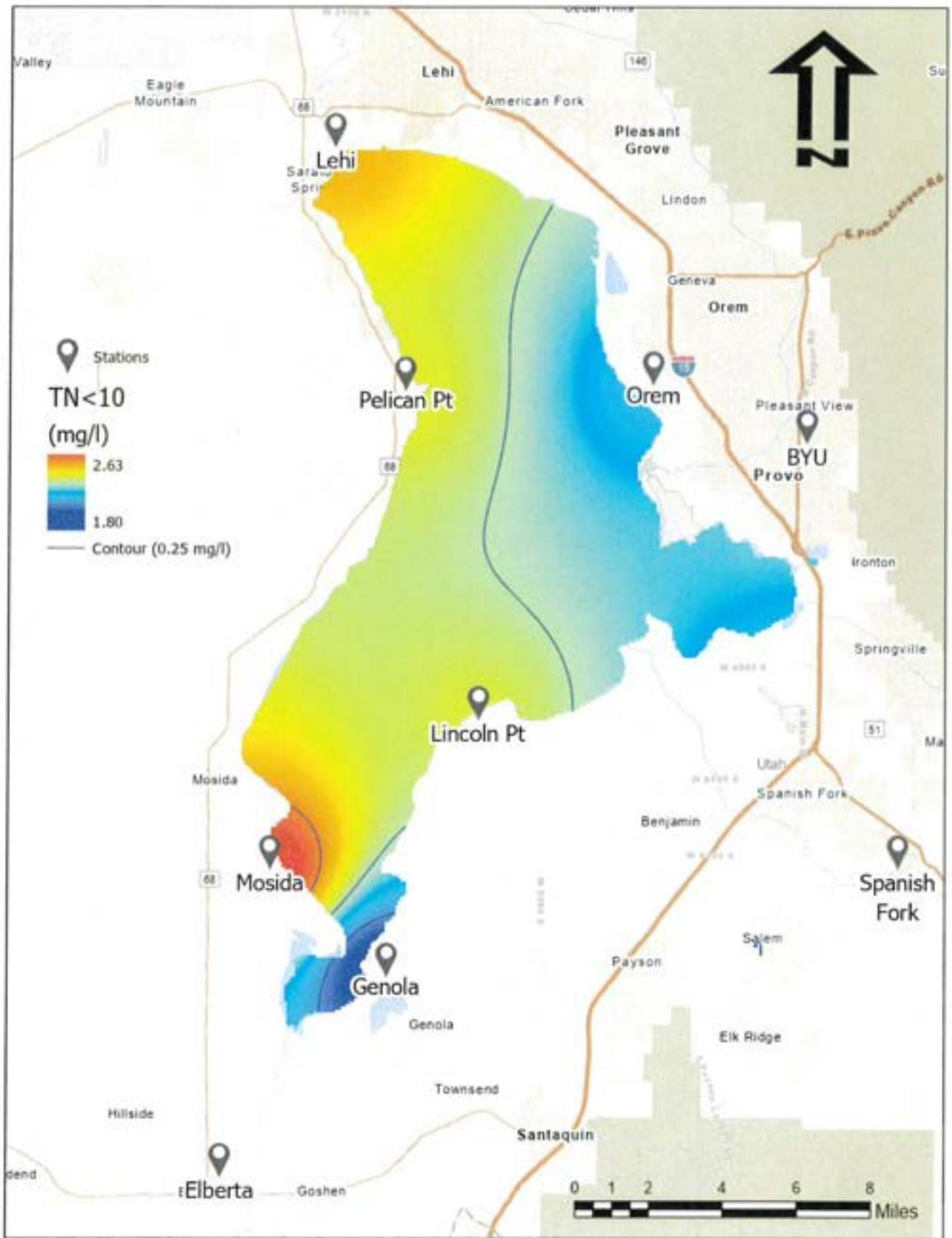
Figure C



Map 1aa



Map 1 a b



Map 1ea

Table 3. Lincoln Point

too big	Sampling Date	TP>1 outliers	Total Phos (mg/l) w/o outlrs	TP>5 outliers	Total Phos (mg/l) w/o outlrs	Total Phos (mg/l) all data	OrthoPhos (mg/l) all data	TN>10 outliers	Total Nitro (mg/l) w/o outlrs	Total Nitro (mg/l) all data
	10-Feb-17		0.08		0.08	0.08			1.42	1.42
	22-Feb-17	1.96			1.96	1.96		24.40		24.40
	27-Feb-17		0.17		0.17	0.17			5.31	5.31
	5-Mar-17		0.20		0.20	0.20			4.83	4.83
	23-Mar-17		0.37		0.37	0.37			3.06	3.06
	27-Mar-17		0.08		0.08	0.08			1.35	1.35
	30-Mar-17		0.06		0.06	0.06			2.46	2.46
	8-Apr-17		0.07		0.07	0.07			2.11	2.11
	19-Apr-17		0.07		0.07	0.07			0.95	0.95
	21-Apr-17		0.06		0.06	0.06			5.03	5.03
	25-Apr-17		0.06		0.06	0.06			1.00	1.00
	6-May-17		0.37		0.37	0.37			1.50	1.50
	17-May-17	8.90		8.90		8.90			6.90	6.90
	21-May-17	1.40			1.40	1.40			1.70	1.70
25	13-Jun-17							14.00		14.00
	25-Jul-17	8.80		8.80		8.80		23.60		23.60
21	10-Aug-17							21.40		21.40
	15-Sep-17		0.69		0.69	0.69			1.00	1.00
	24-Sep-17		0.18		0.18	0.18			1.05	1.05
	5-Nov-17	1.10			1.10	1.10		BDL	BDL	
	17-Nov-17		0.18		0.18	0.18			0.90	0.90
	9-Jan-18		0.10		0.10	0.10			1.20	1.20
	15-Feb-18		0.03		0.03	0.03			2.40	2.40
	16-Mar-18		0.01		0.01	0.01			0.50	0.50
	23-Mar-18		0.03		0.03	0.03			0.50	0.50
	7-Apr-18	1.60			1.60	1.60			1.20	1.20
	20-Apr-18		0.55		0.55	0.55			0.40	0.40
	30-Apr-18		0.49		0.49	0.49			1.30	1.30
	3-May-18		0.23		0.23	0.23			1.70	1.70
	11-May-18		0.18		0.18	0.18			2.70	2.70
	22-Aug-18	6.30		6.30		6.30		34.20		34.20
	3-Oct-18	5.30		5.30		5.30		12.40		12.40
	10-Oct-18		0.09		0.09	0.09			0.70	0.70
	30-Nov-18		0.40		0.40	0.40			0.30	0.30
	18-Jan-19		0.04		0.04	0.04	0.01	BDL	BDL	
	6-Mar-19		0.11		0.11	0.11	0.02	BDL	BDL	
	29-Mar-19		0.04		0.04	0.04	0.04	BDL	BDL	
	10-Apr-19		0.57		0.57	0.57	0.29		2.60	2.60
	21-Apr-19		0.60		0.60	0.60	0.10		1.20	1.20
	7-May-19		0.23		0.23	0.23	0.08		1.30	1.30
	21-May-19		0.39		0.39	0.39	0.27		1.00	1.00
	21-Jun-19	2.20			2.20	2.20	0.51		3.60	3.60
	1-Aug-19	3.70			3.70	3.70	2.20		9.60	9.60
	9-Aug-19	1.40			1.40	1.40	1.00		4.80	4.80
	11-Sep-19	1.70			1.70	1.70	1.40		3.70	3.70
	20-Nov-19		0.12		0.12	0.12	BDL	BDL	BDL	
	count	12.00	32.00	4.00	40.00	44.00	11.00	6.00		41.00
	averages		0.214		0.548	1.164	0.538		2.322	5.153
	summer (Apr-Sep) avgs		0.316		0.797	1.698	0.731		2.561	5.752
	winter (Oct-Mar) avgs		0.124		0.272	0.524	0.023		1.918	4.115
	summer count					24.00			22.00	26.00
	winter count					20.00			13.00	15.00

Figure 3a. Lincoln Pt. T-P Conc all data

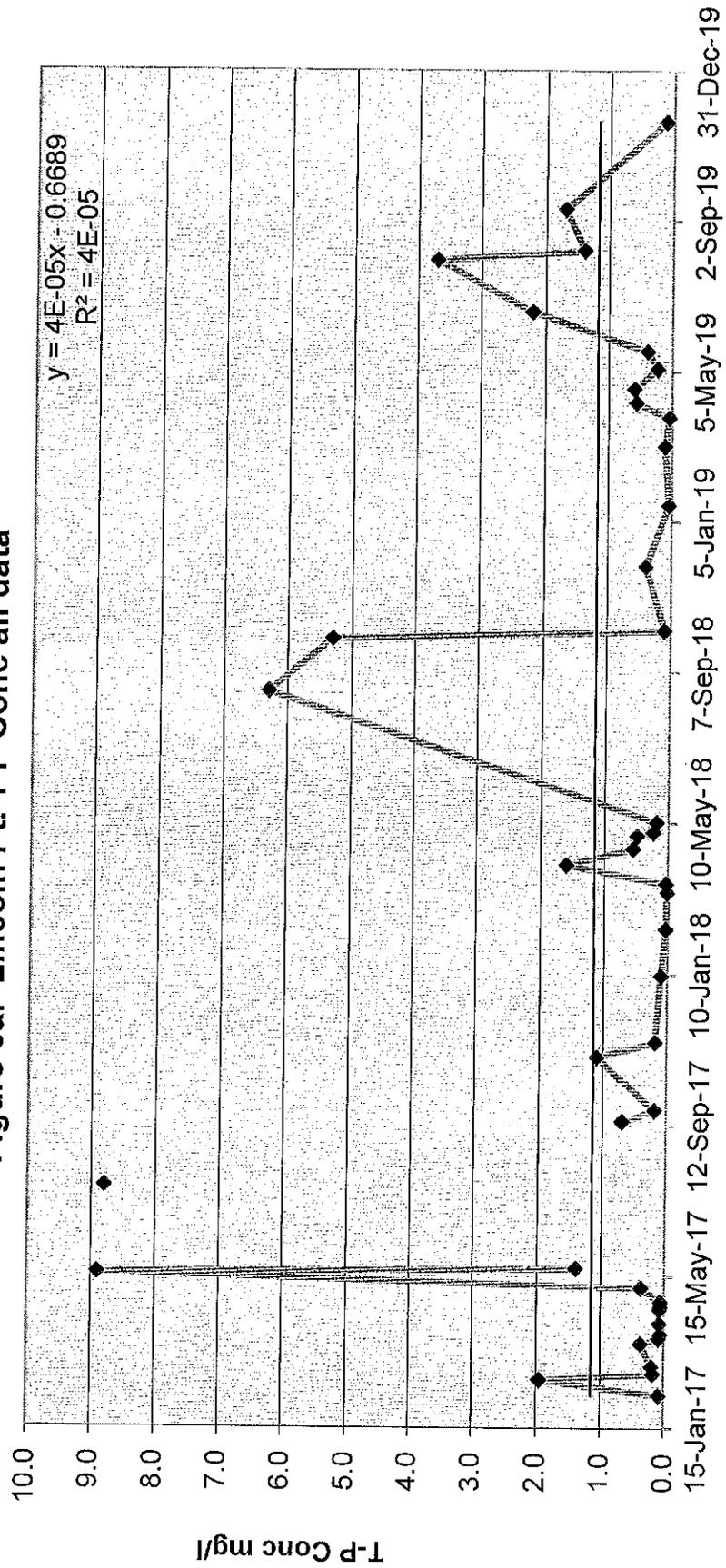


Figure 3b. Lincoln Pt. T-P Conc w/o T-P > 5

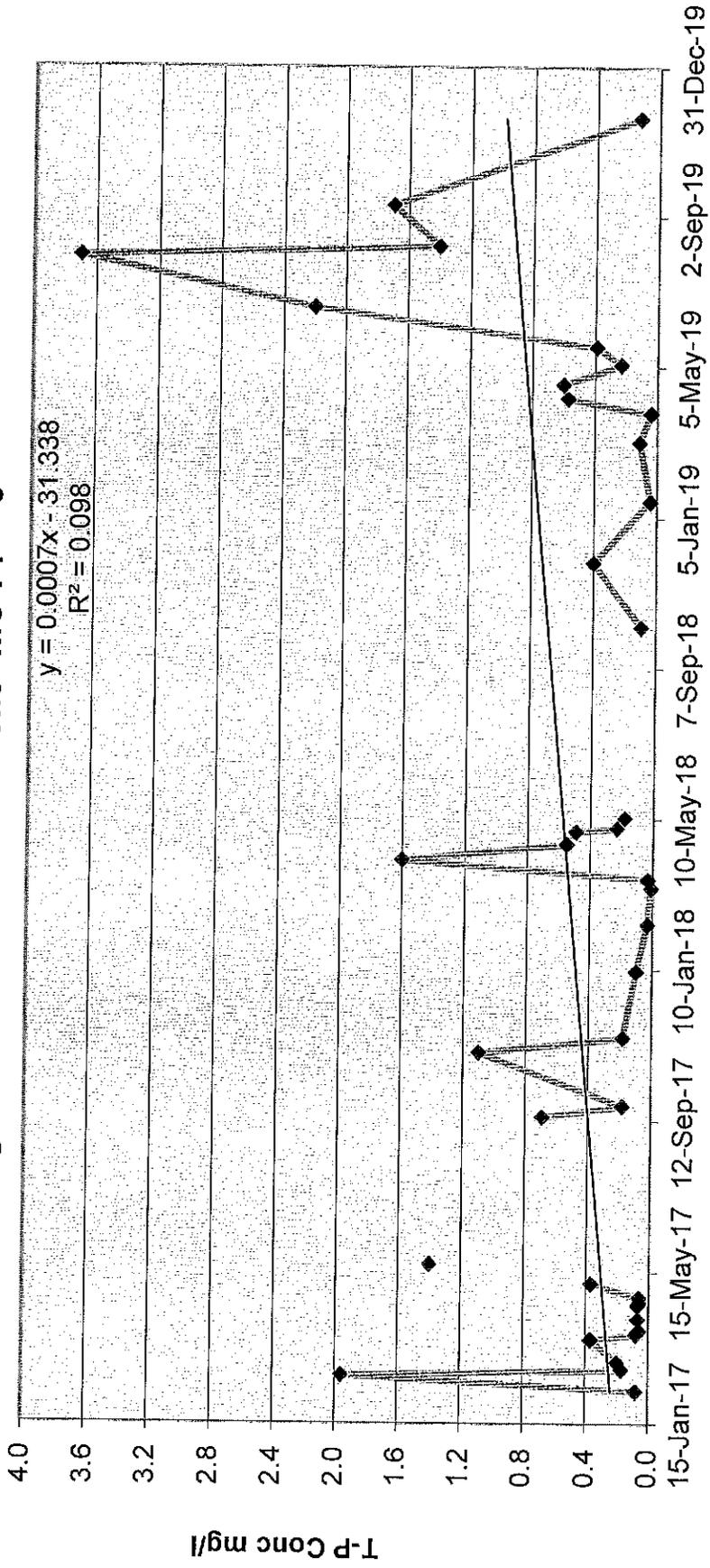


Figure 3c. Lincoln Pt. T-P Conc w/o T-P > 1

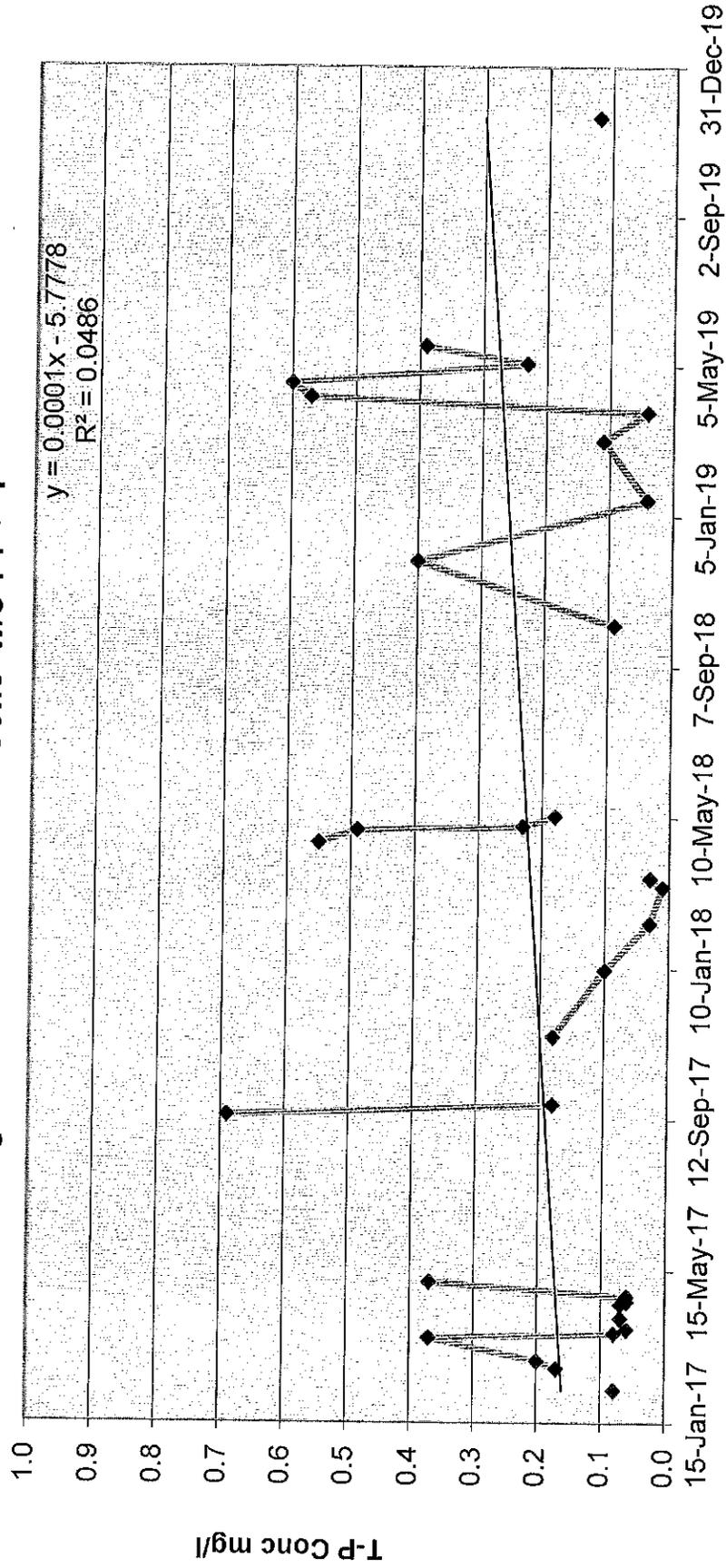


Figure 3d. Lincoln Pt. T-N Conc all data

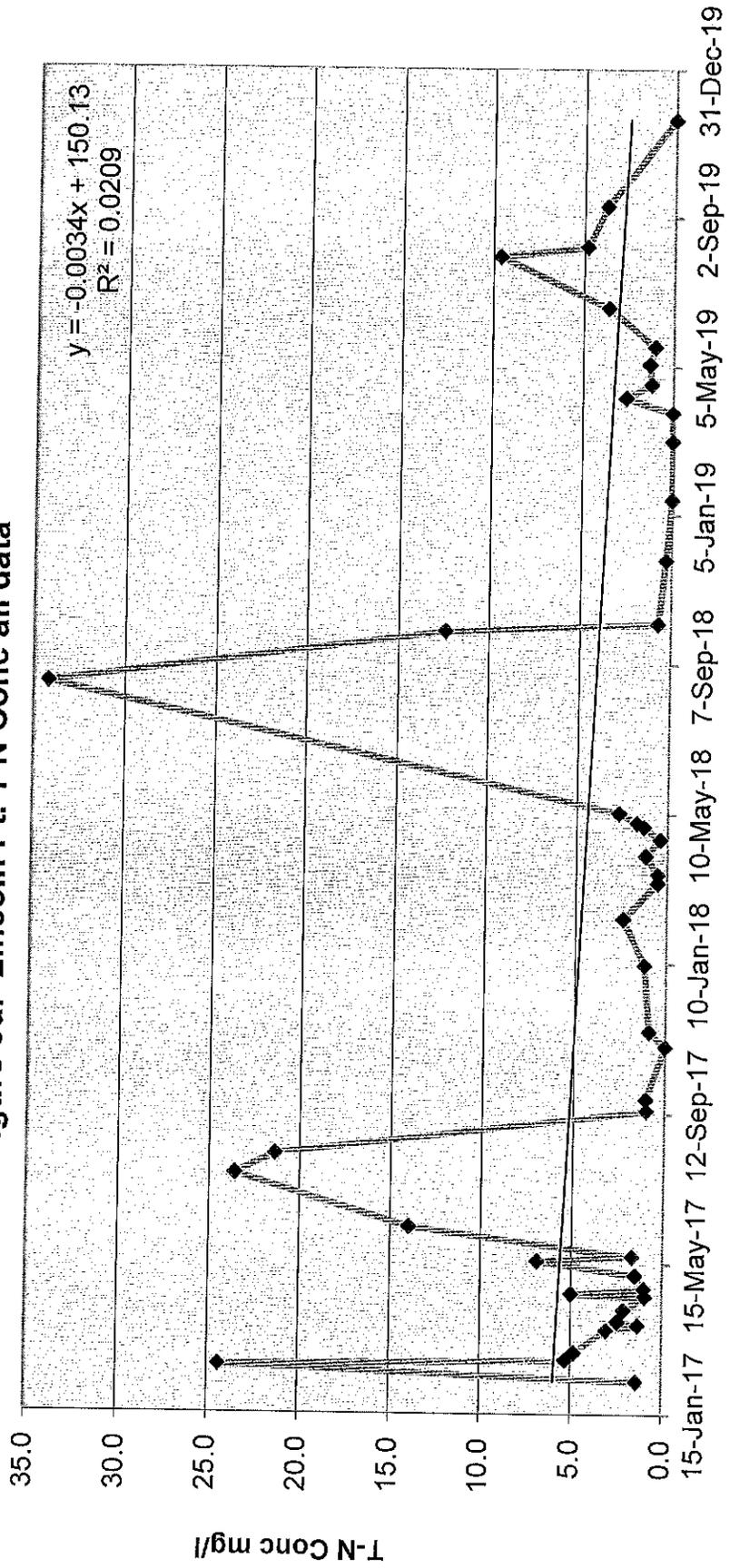


Figure 3e. Lincoln Pt. T-N Conc w/o T-N > 10

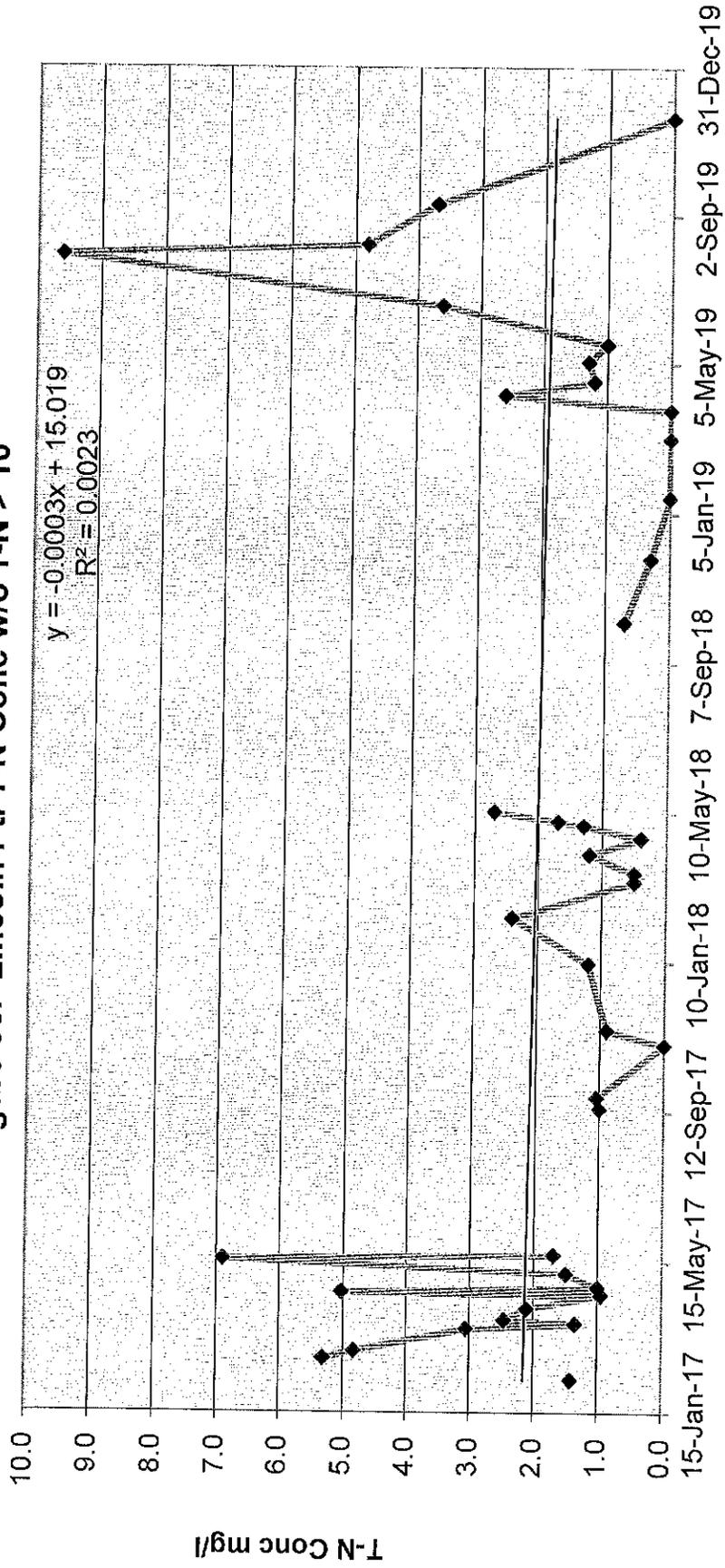
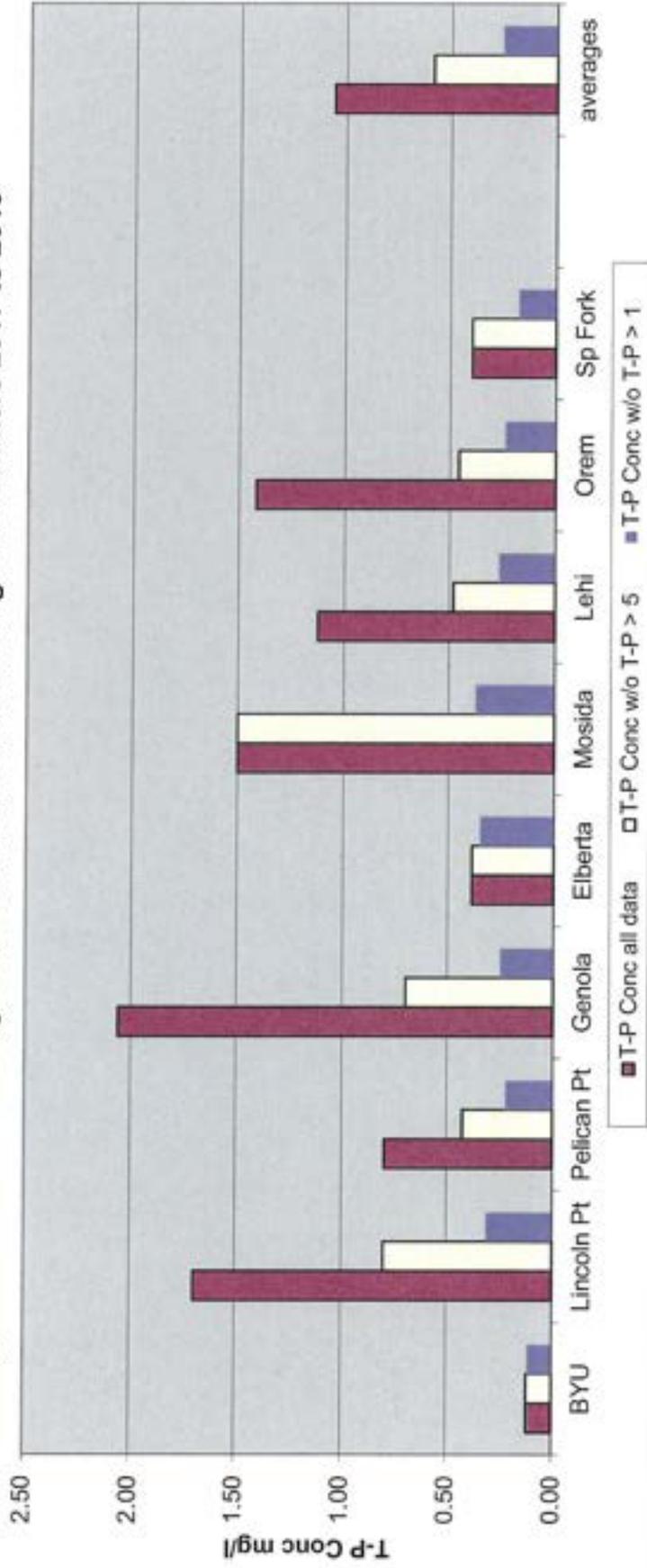
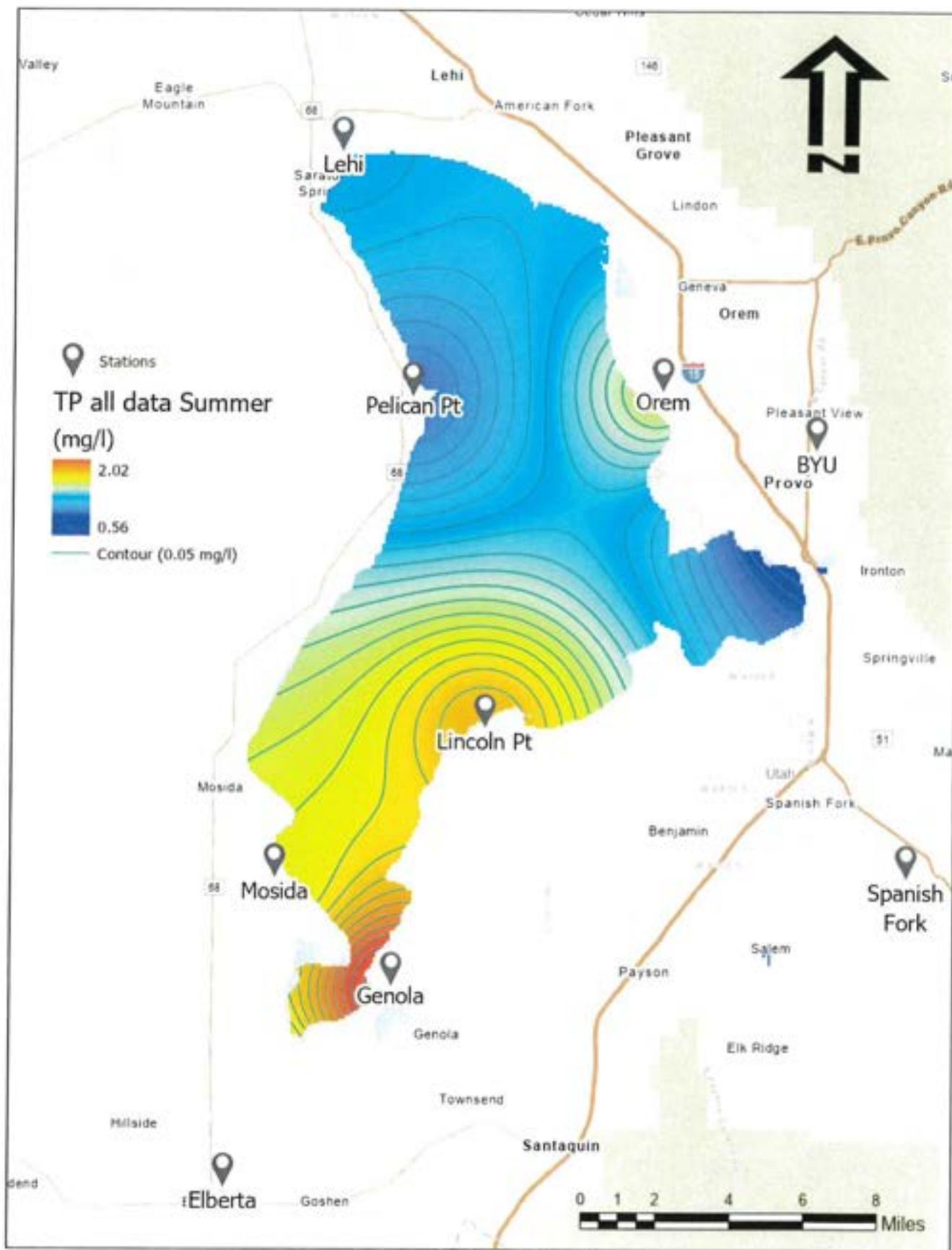
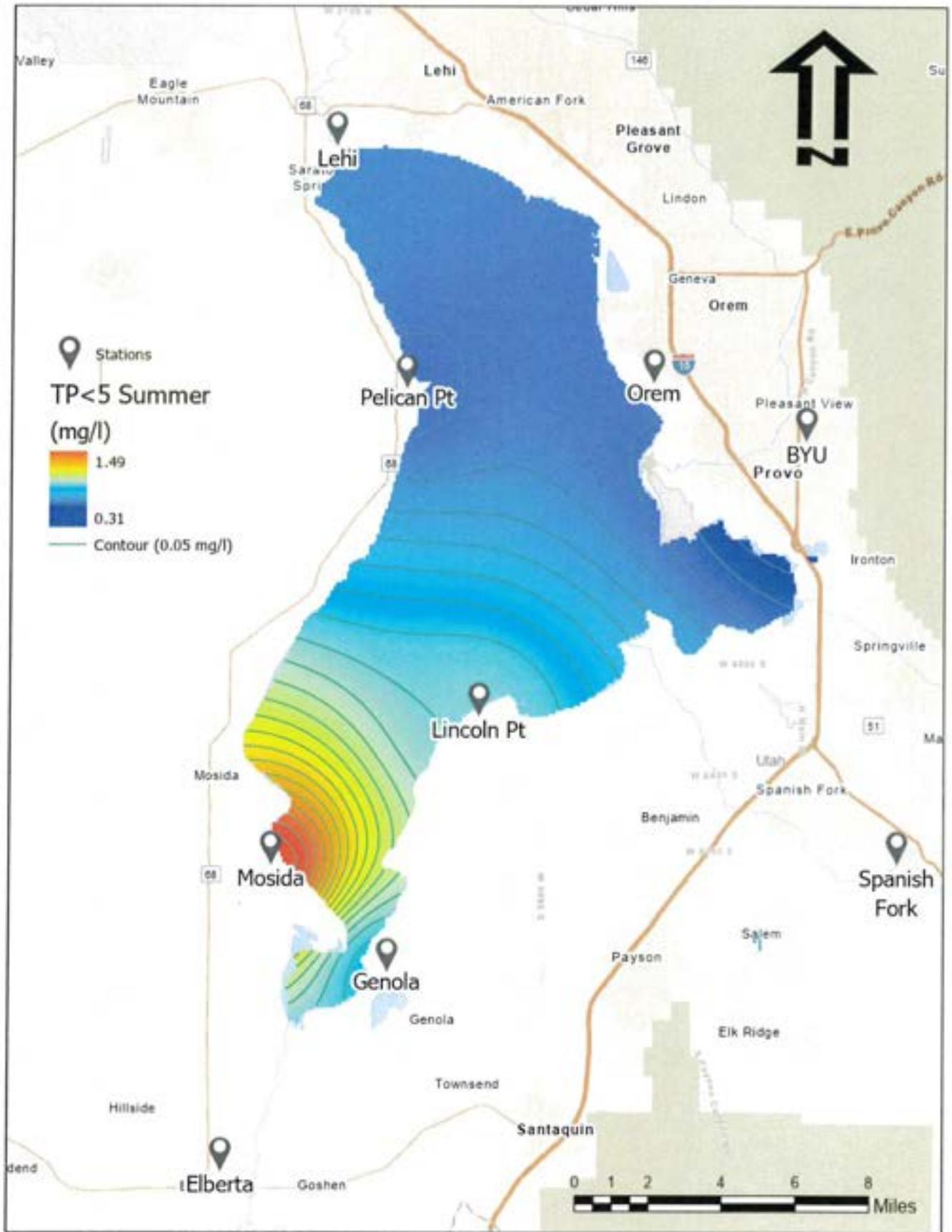


Figure 1b. T-P Conc averages at 9 locations & overall avgs for summers 2017 to 2019

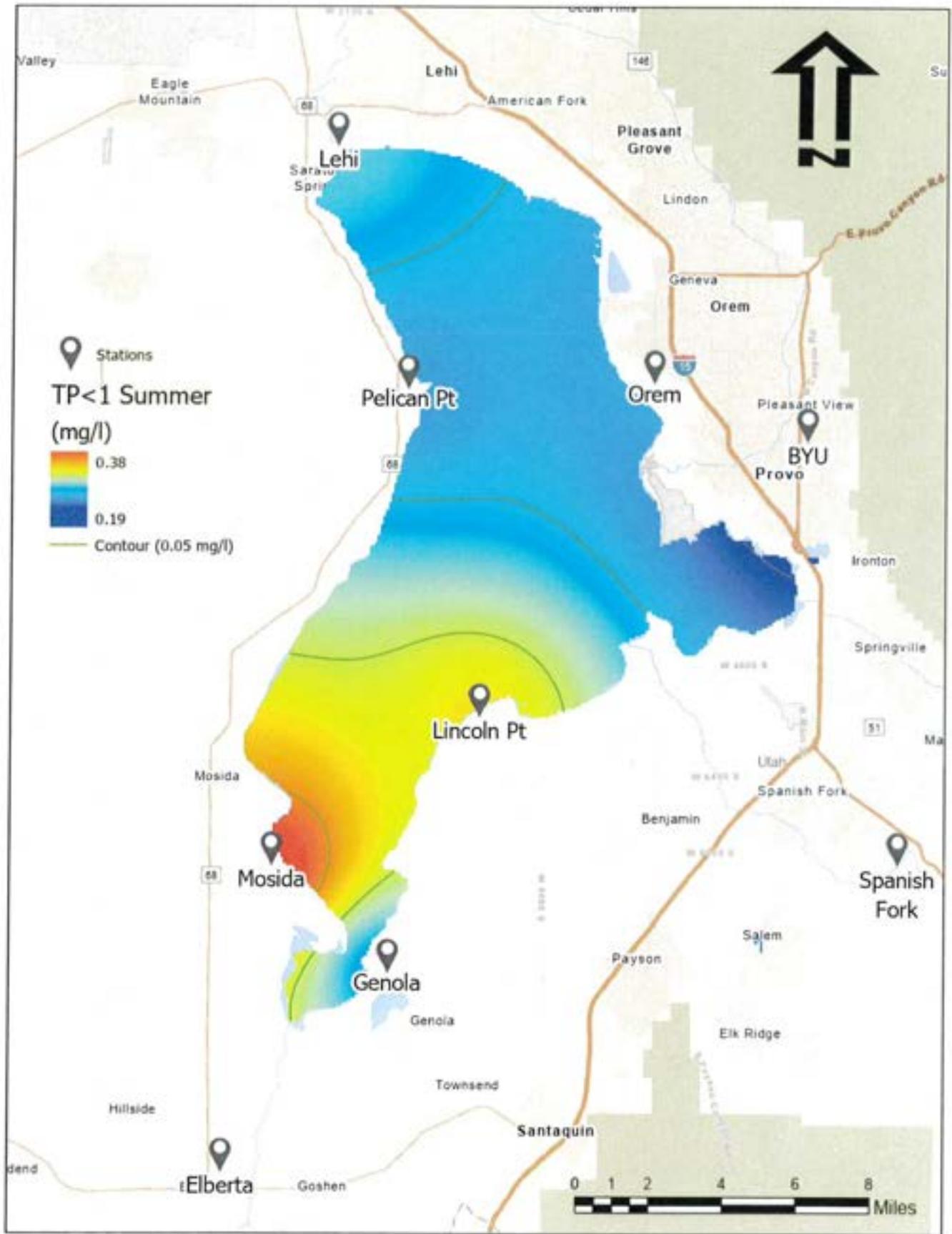




Map 1ba

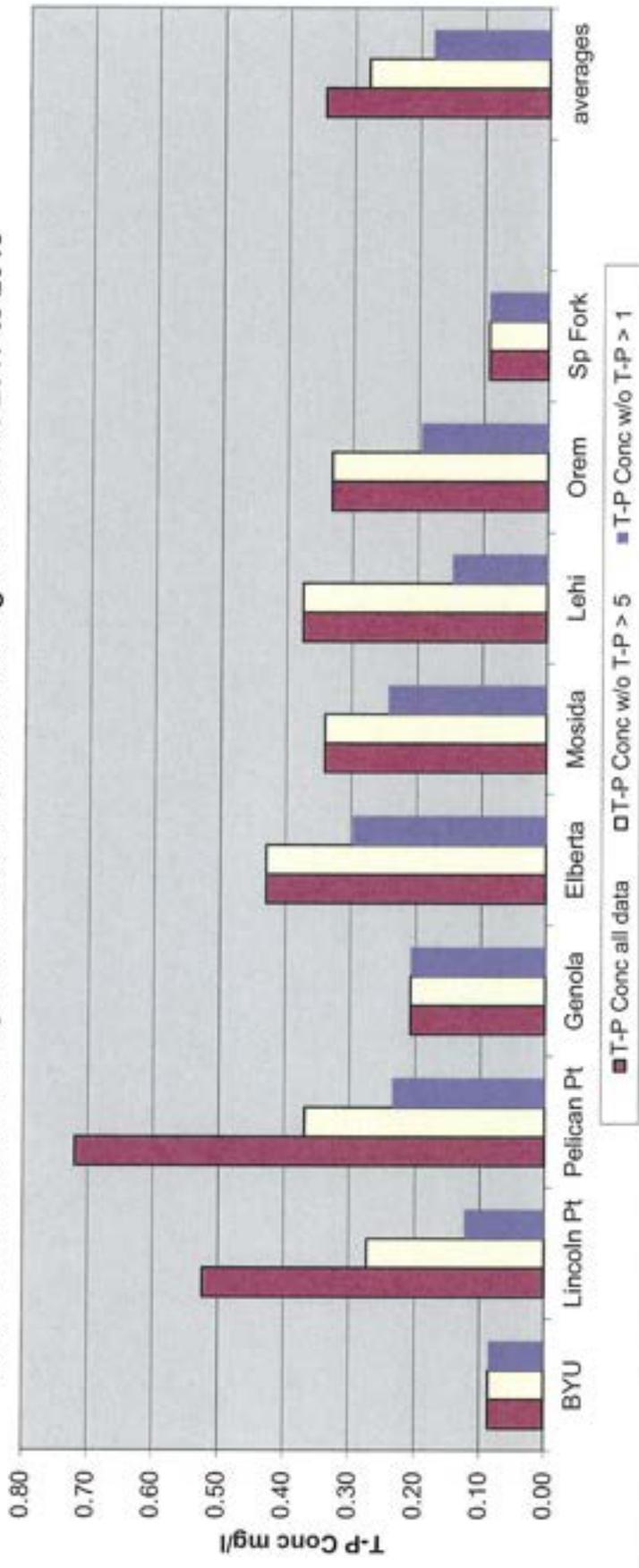


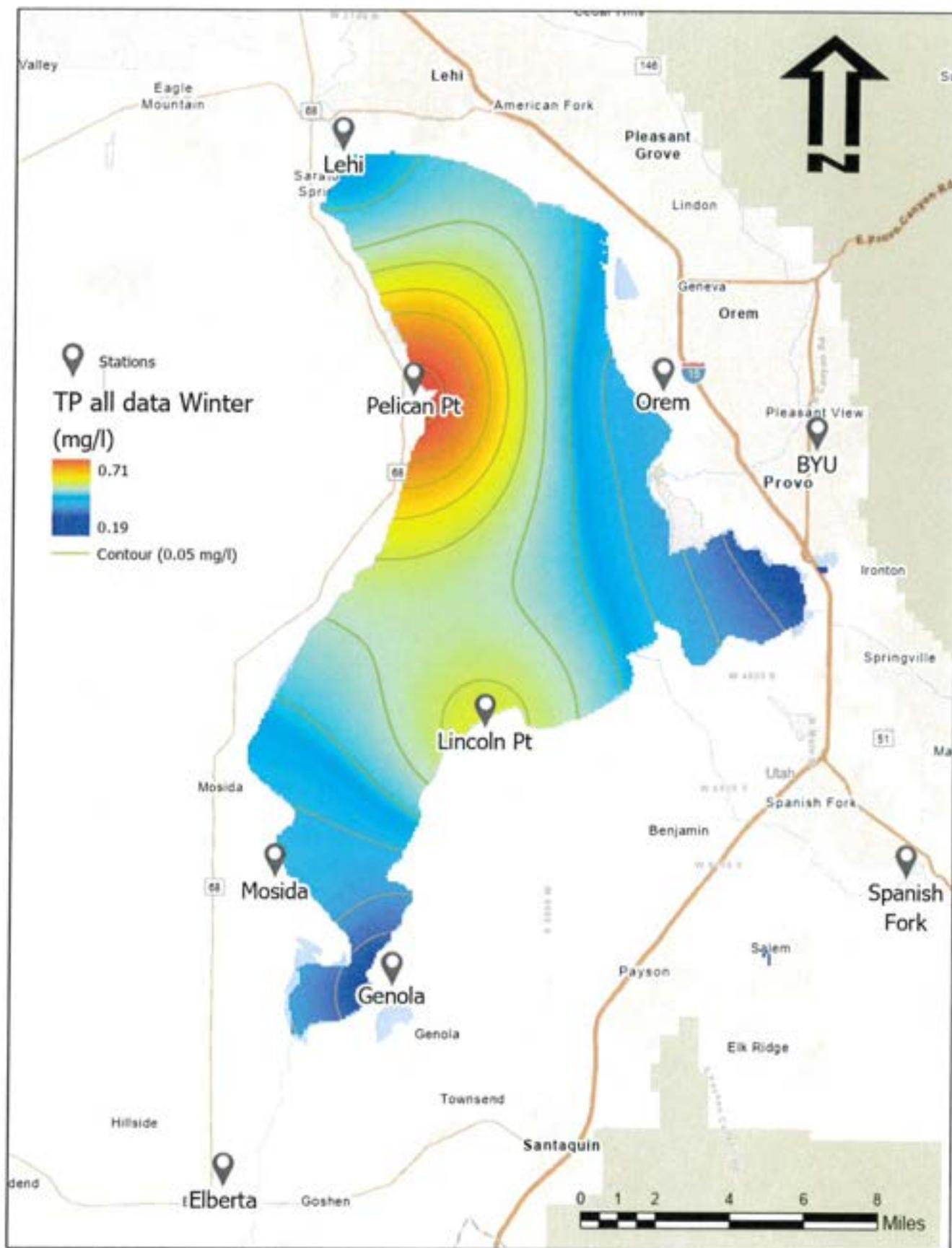
Map 1bb



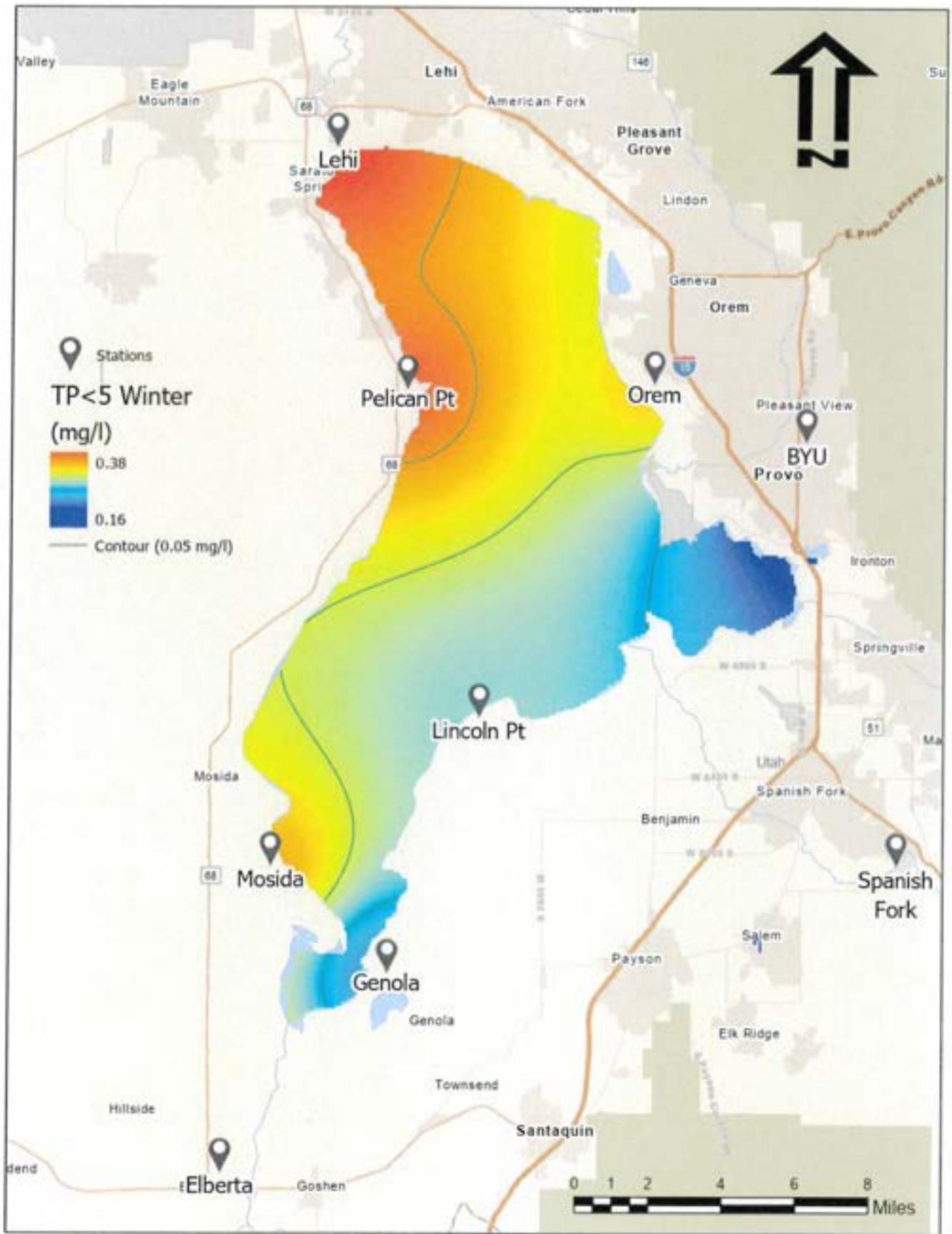
Map 15c

Figure 1c. T-P Conc averages at 9 locations & overall avgs for winters 2017 to 2019

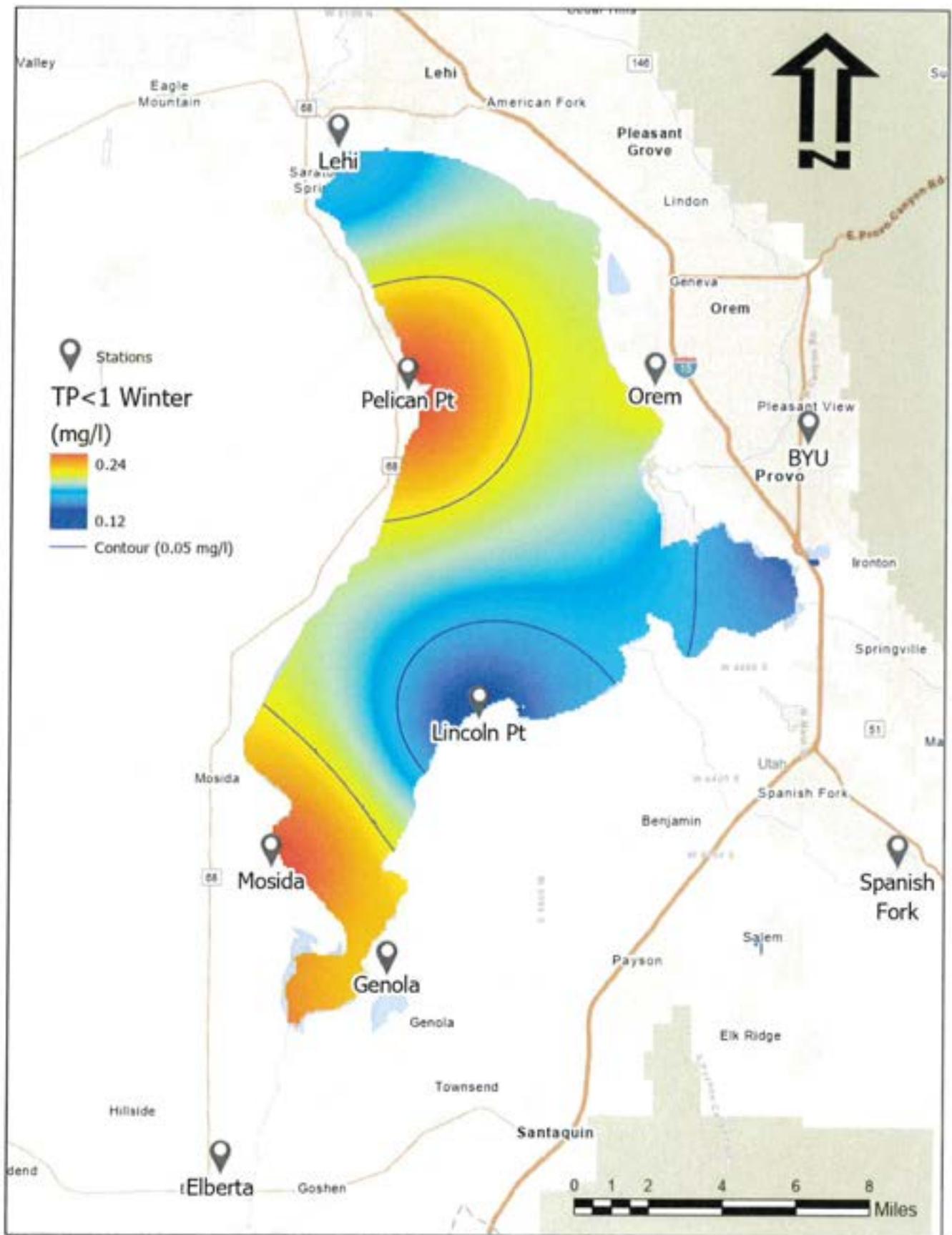




Map 1ca

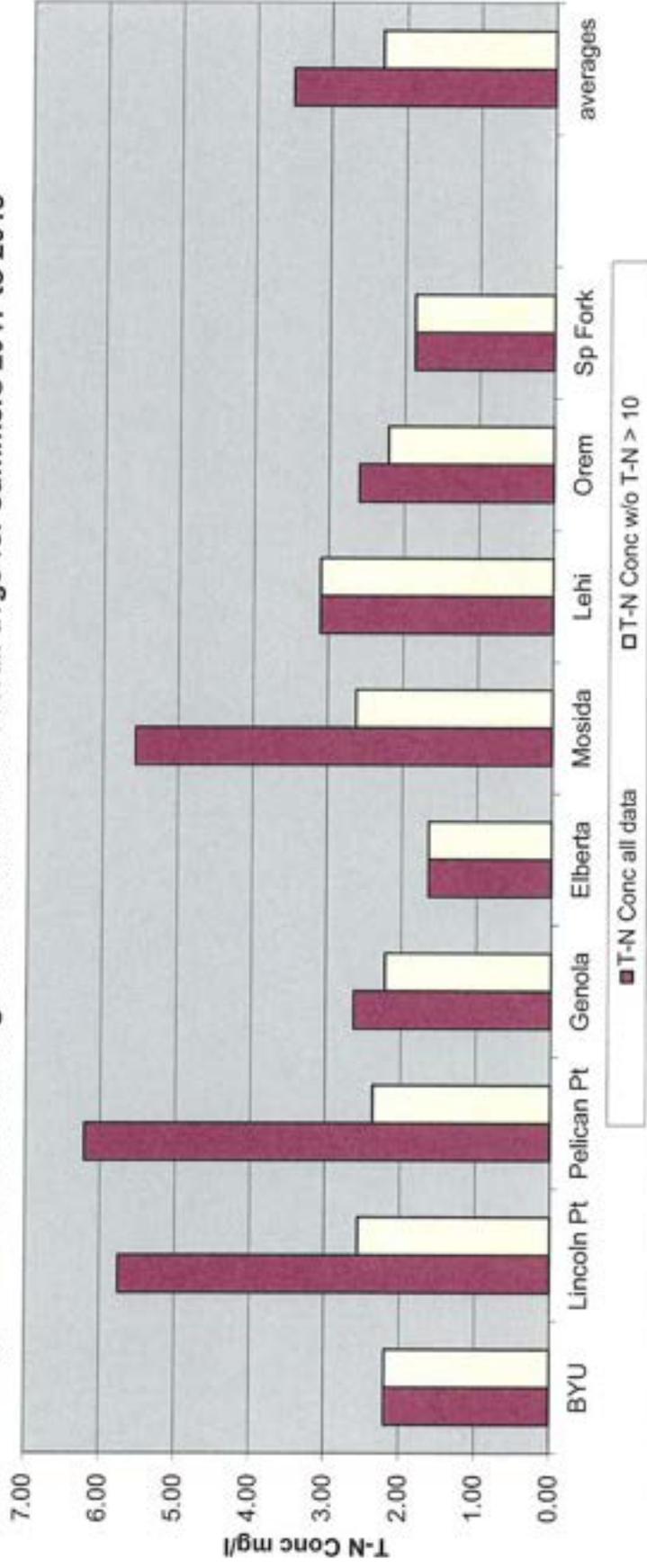


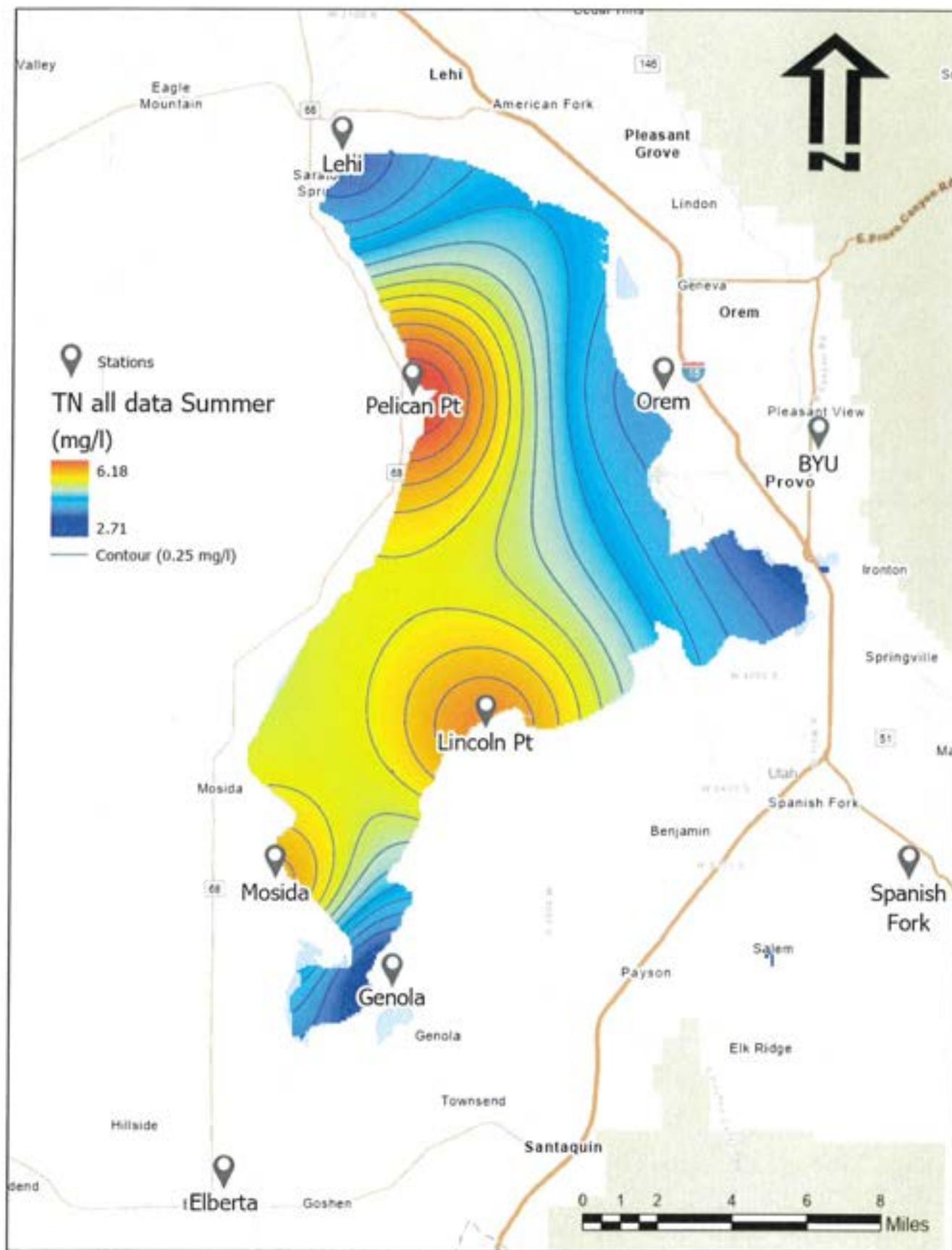
Map 1cb



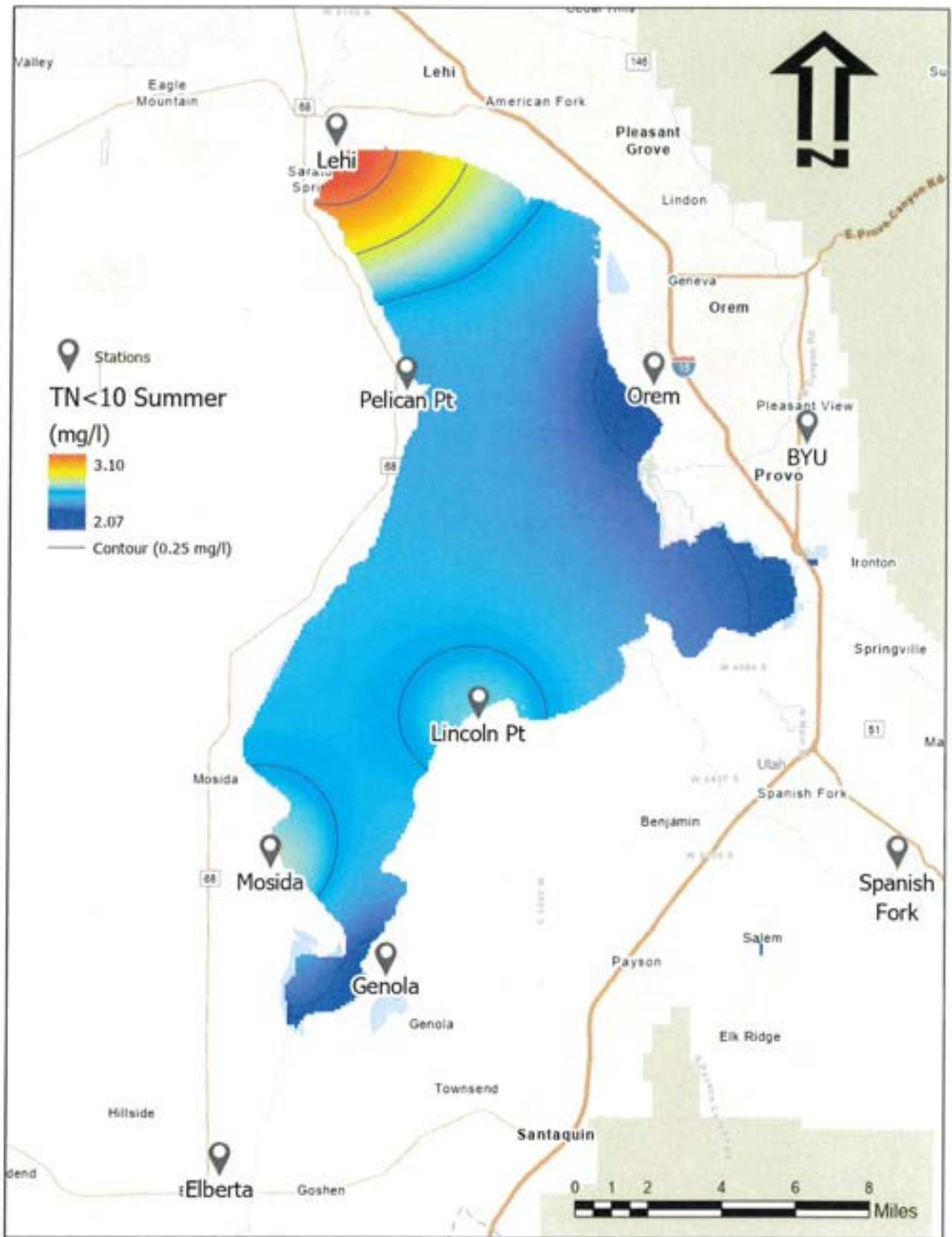
Map 1cc

Figure 1f. T-N Conc averages at 9 locations & overall avgs for summers 2017 to 2019



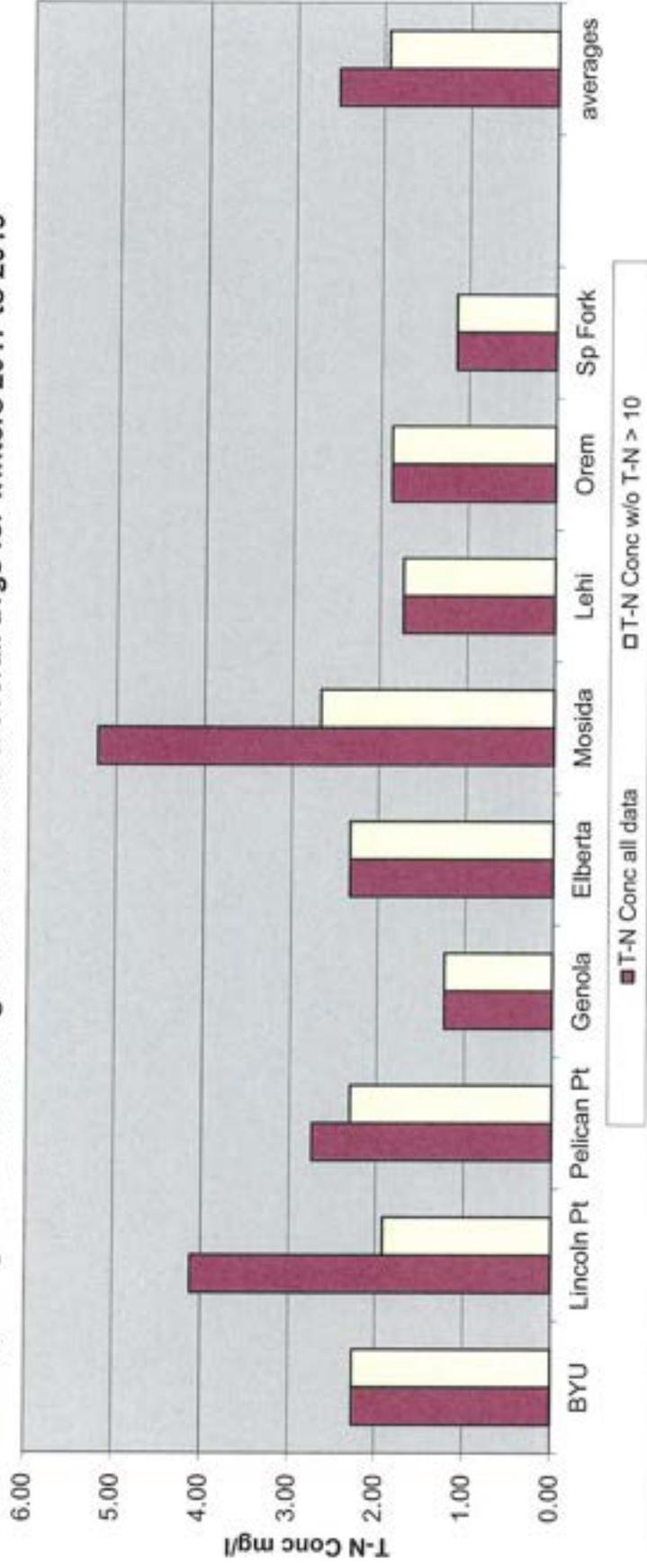


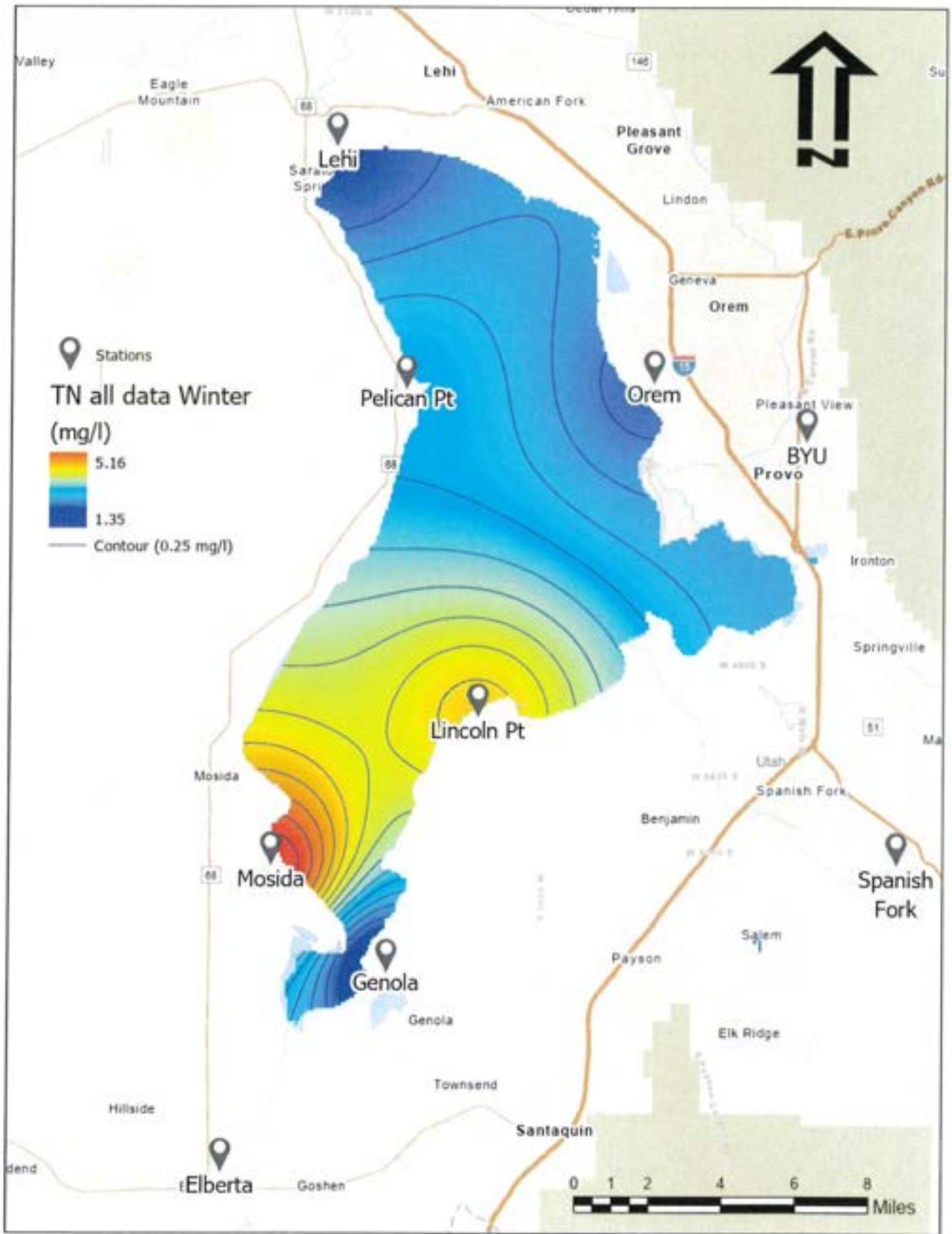
Map 1 fa



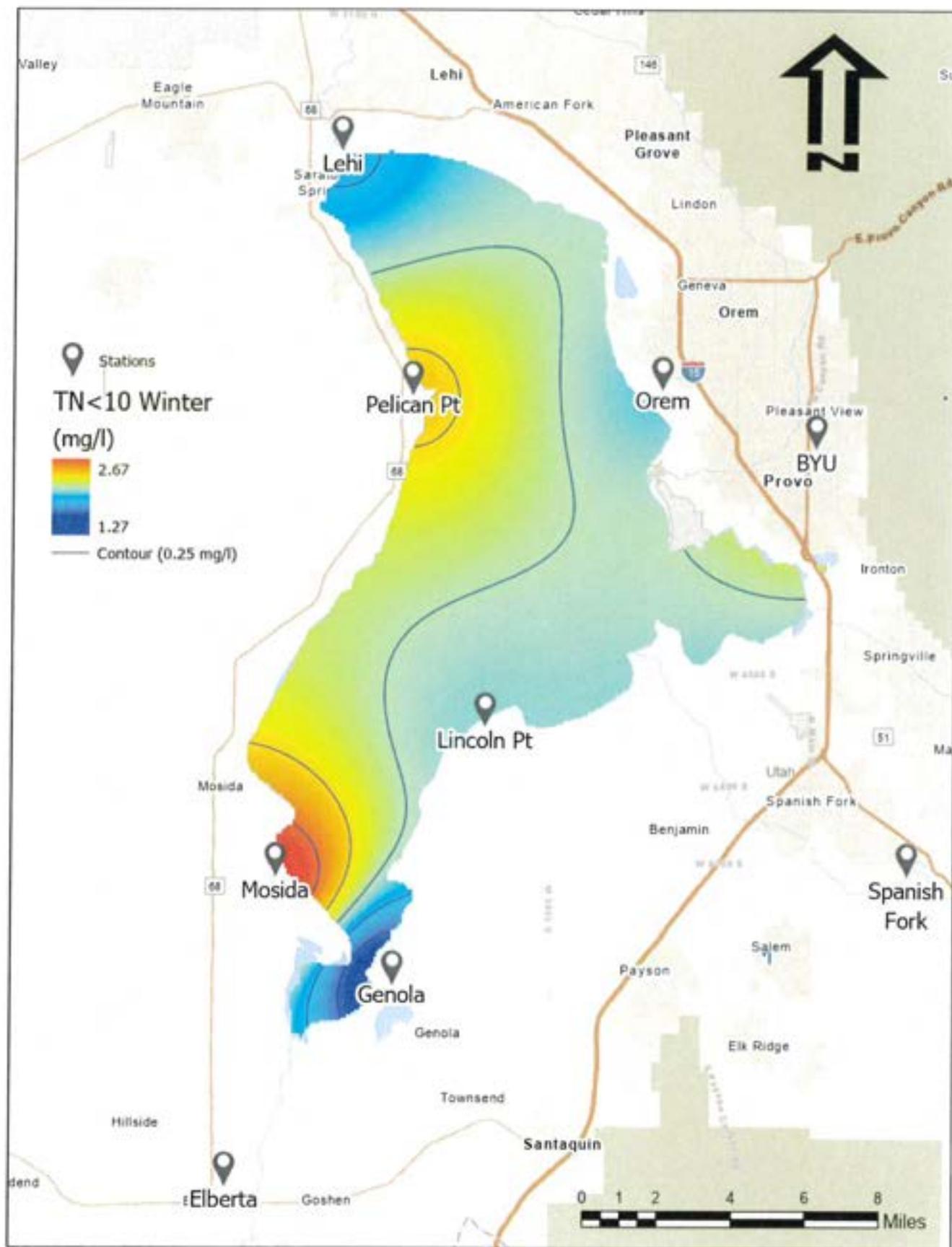
Map 1fb

Figure 1g. T-N Conc averages at 9 locations & overall avgs for winters 2017 to 2019



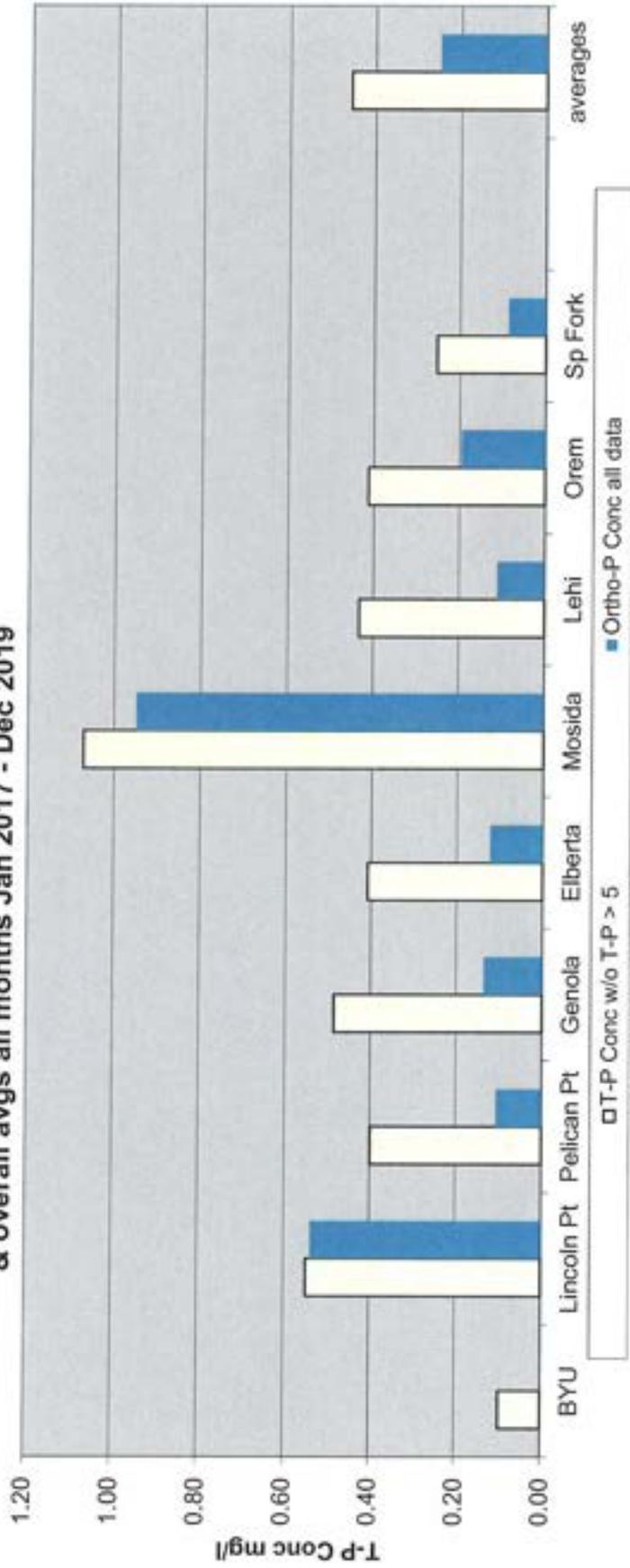


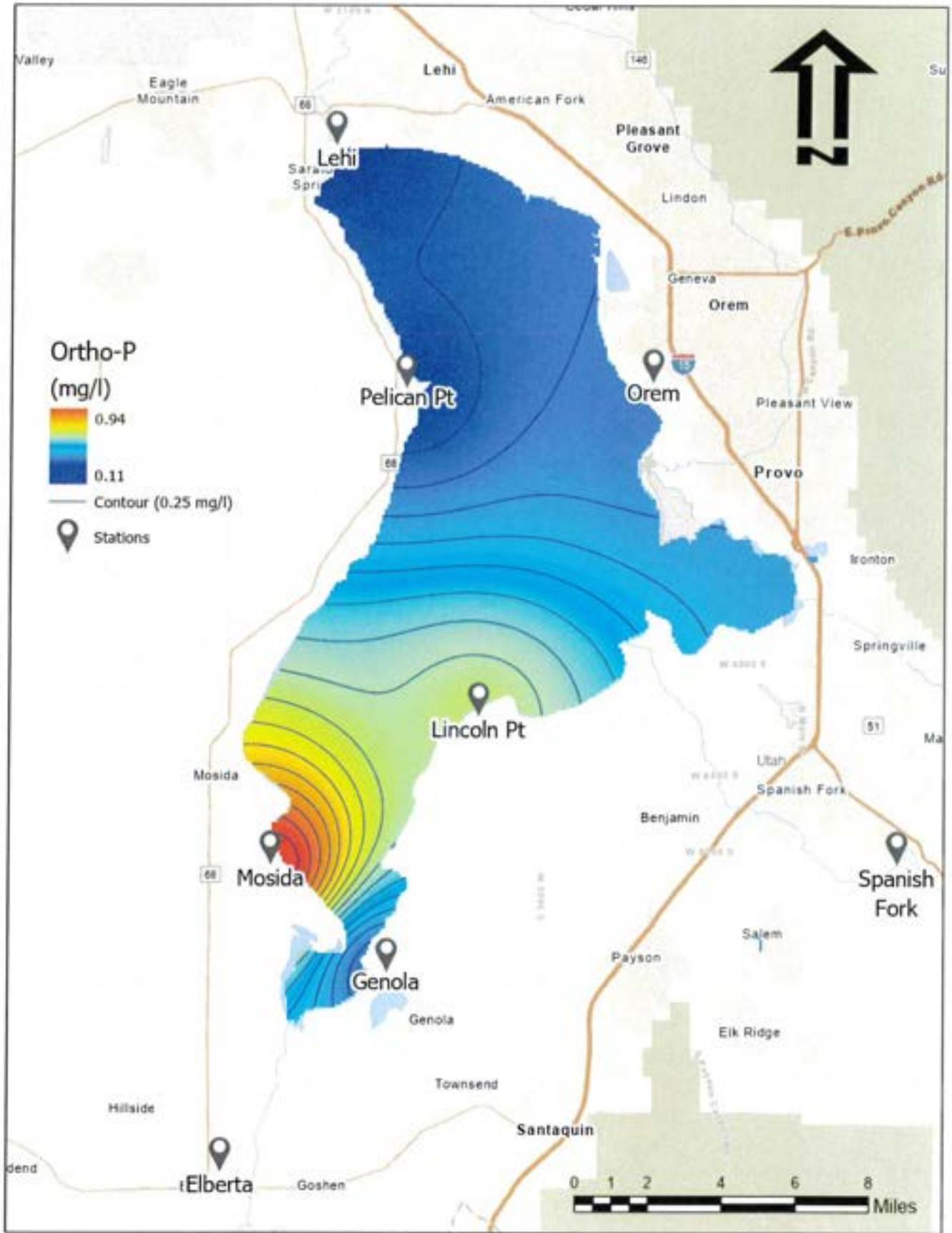
Map 1ga



Map 1gb

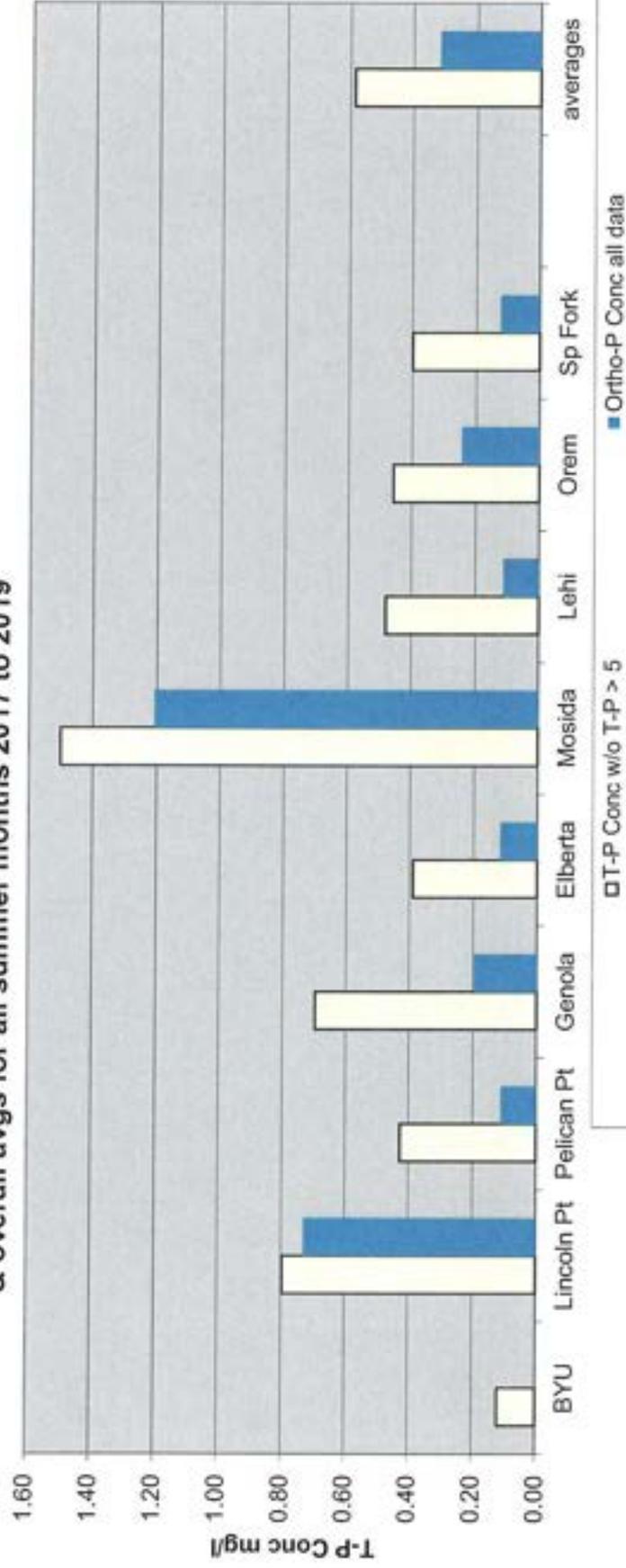
Figure 1i. T-P < 5 mg/l & Ortho-P Conc avgs at 8 locations
& overall avgs all months Jan 2017 - Dec 2019

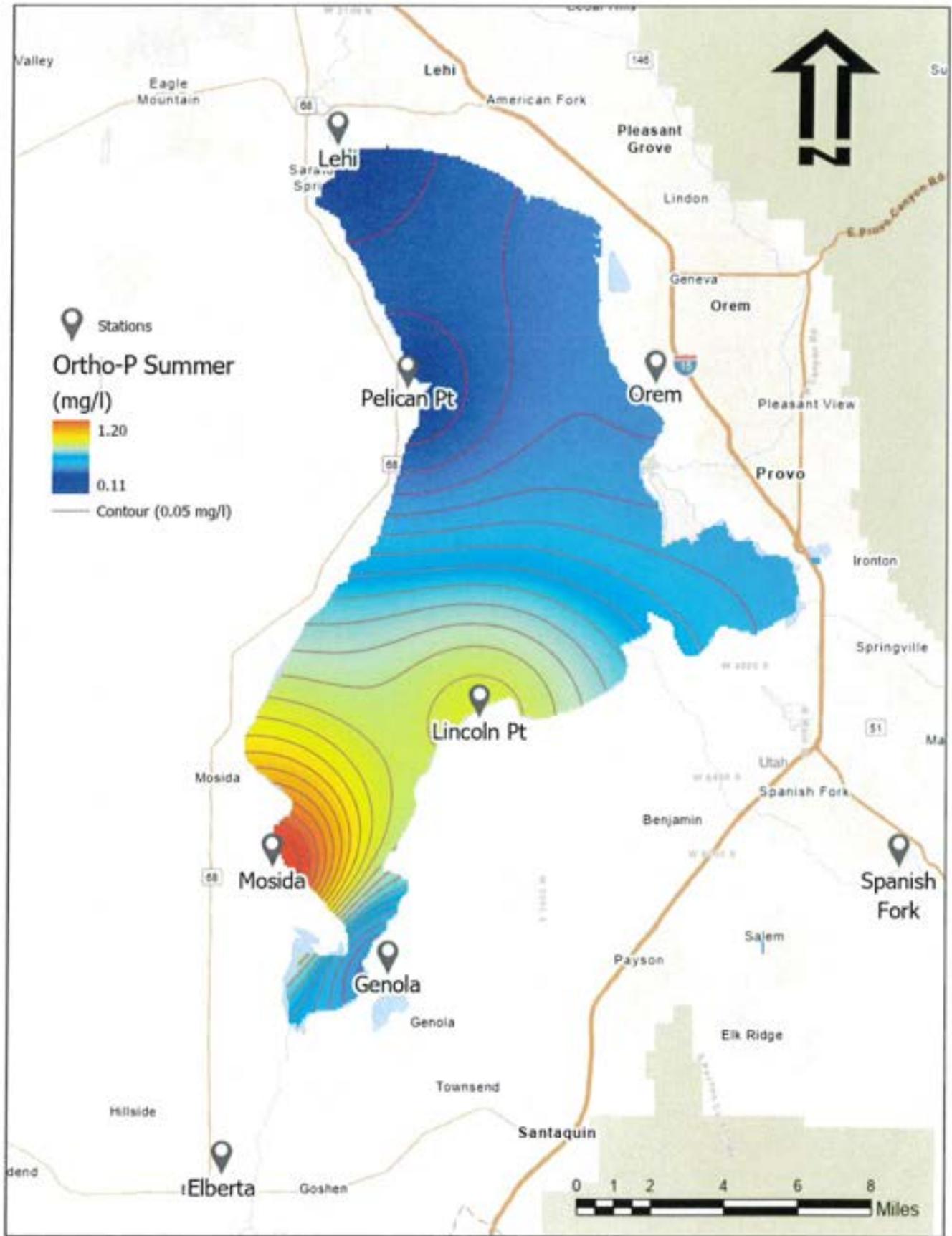




Map 1i

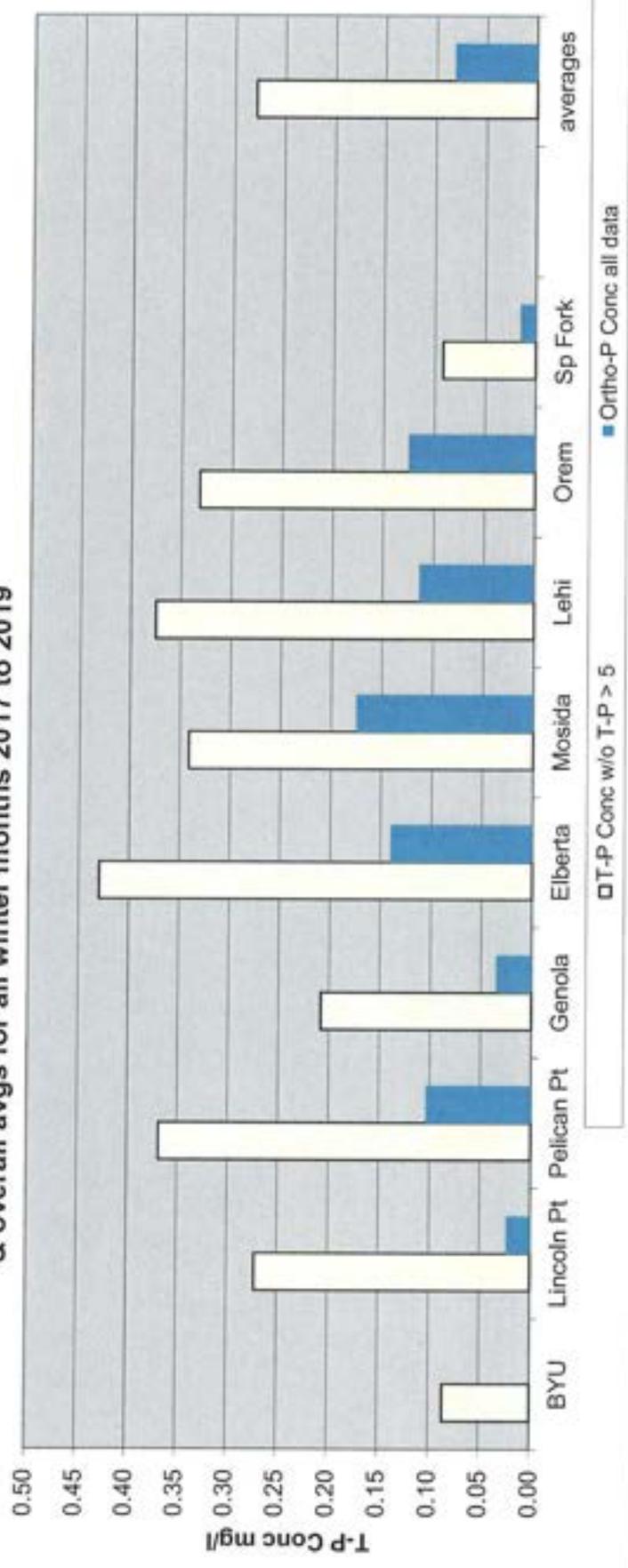
Figure 1j. T-P < 5 mg/l & Ortho-P Conc averages at 8 locations & overall avgs for all summer months 2017 to 2019

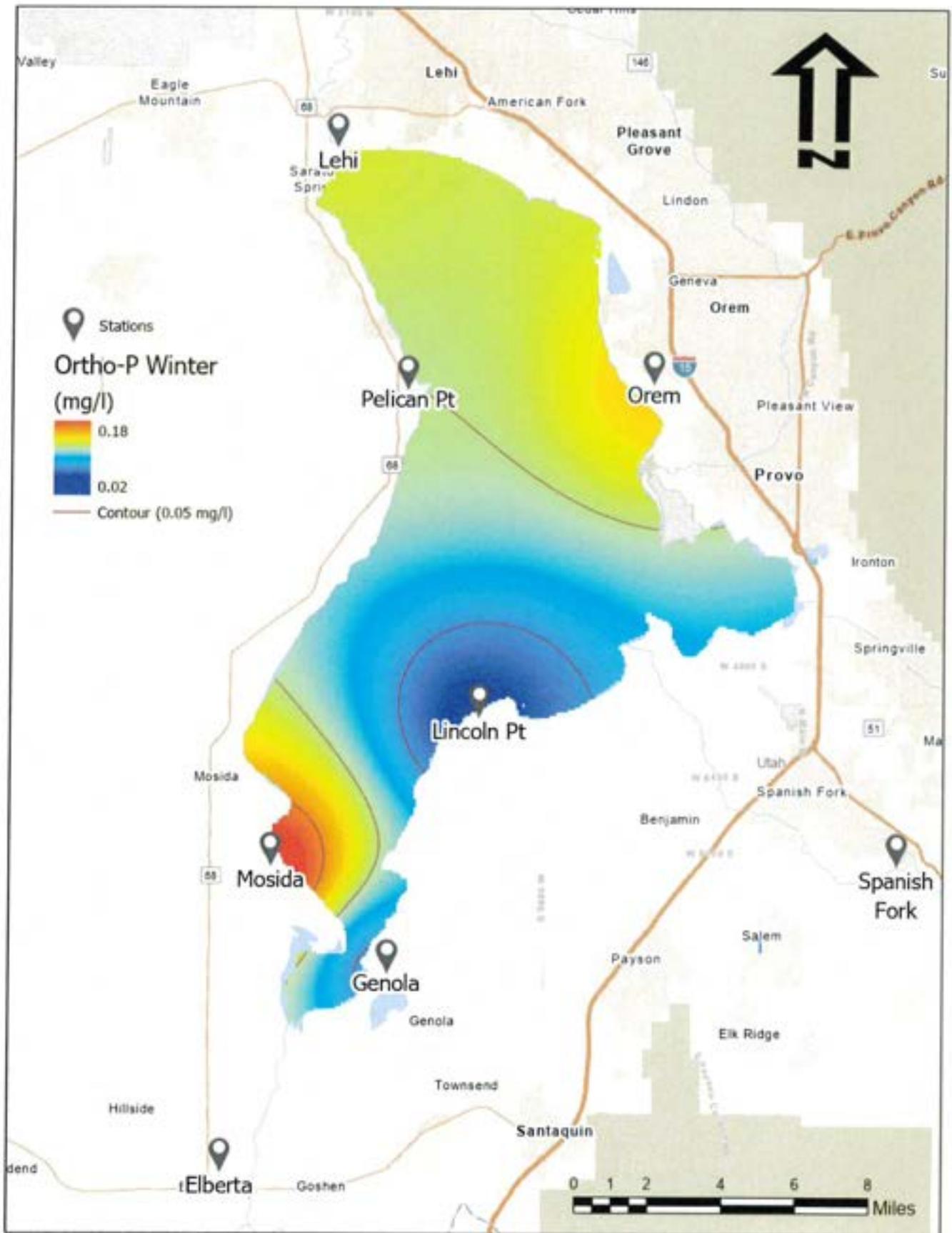




Map 1j

Figure 1k. T-P < 5 mg/l & Ortho-P Conc averages at 8 locations & overall avgs for all winter months 2017 to 2019





Map 1k

Figure 1d. T-P Loads (tons/yr) at 2-yr avg lake area 83,800 ac & 12"/yr or 6"/half yr precip & all data, T-P < 1, T-P < 5, summer & winter overall avg T-P conc 2017 to 2019

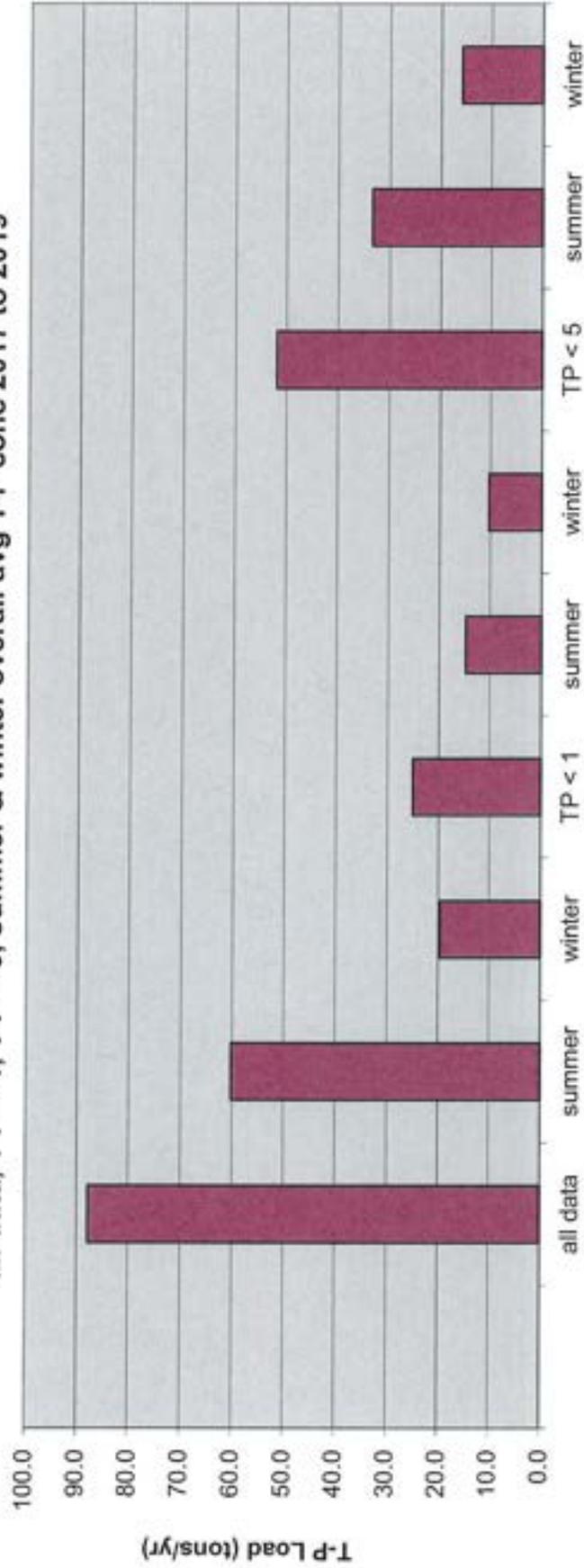


Figure 11. T-P & Ortho-P Loads (tons/yr) at 2-yr avg lake area 83,800 ac & 12"/yr or 6"/half yr precip & all data, T-P <1, T-P <5, sum & win overall avg T-P conc 2017 to 2019

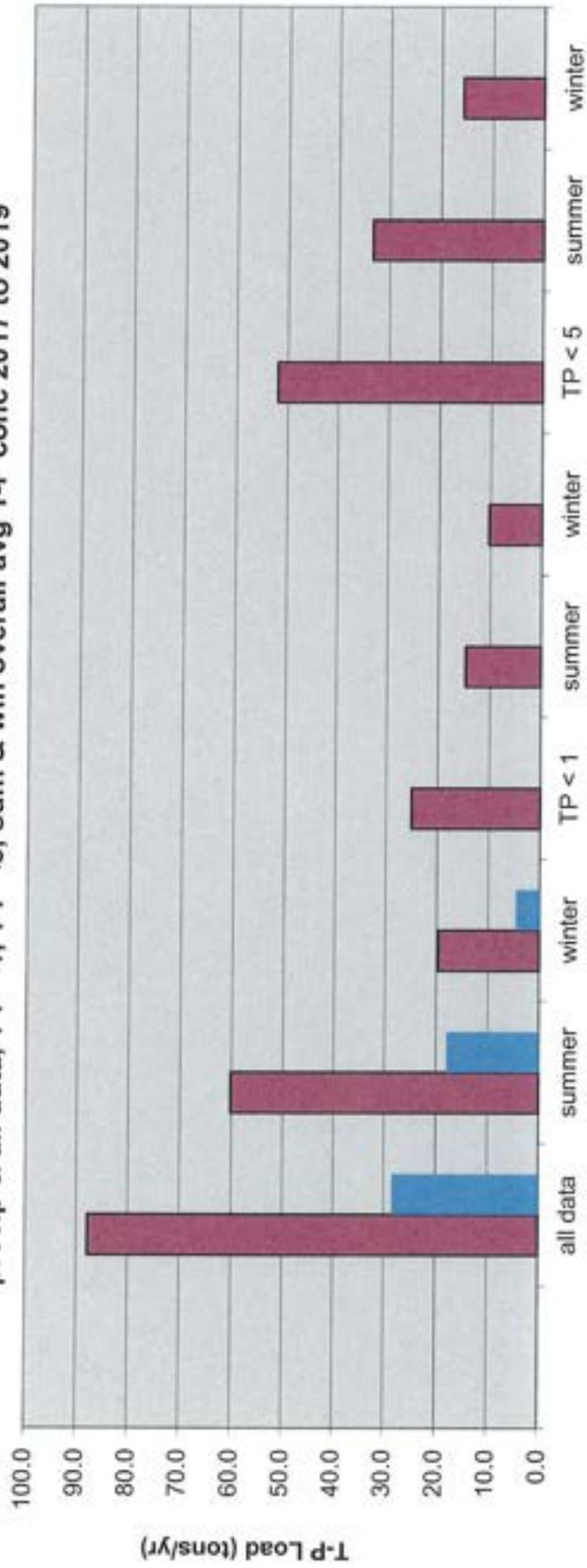
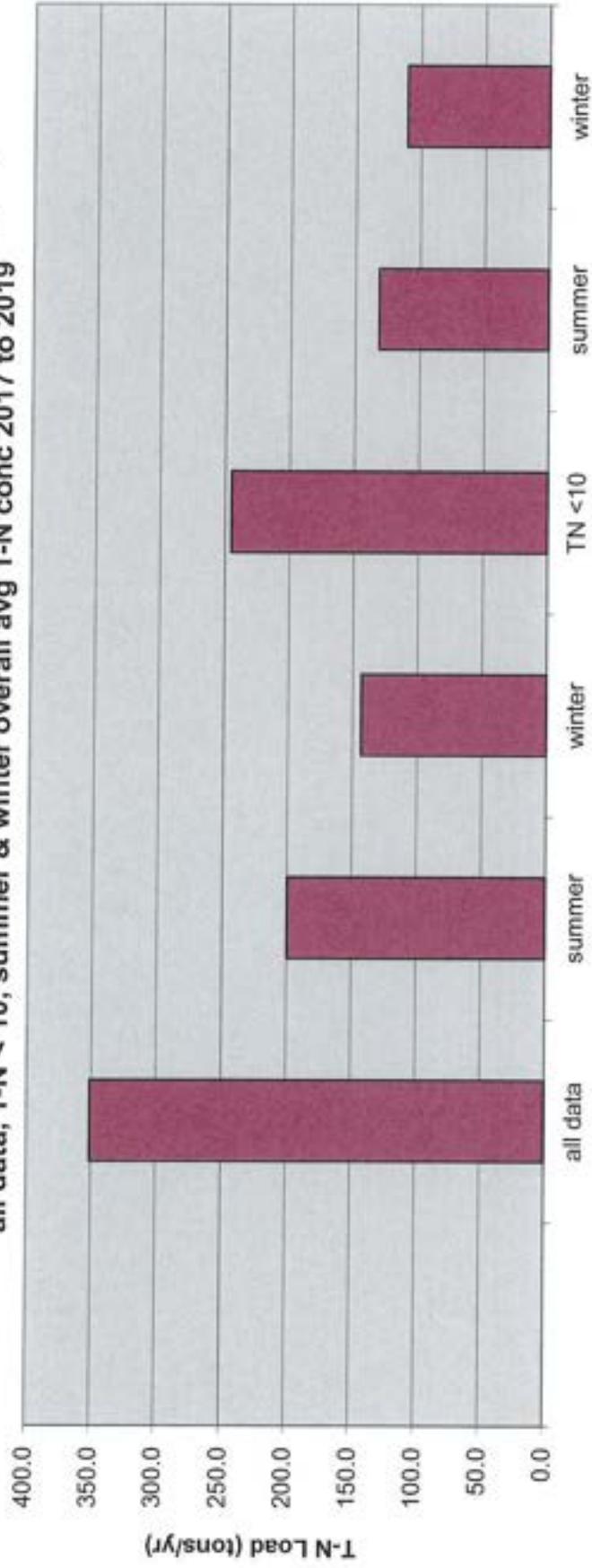


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*Factors Effecting the Ecological Health and Integrity of
Utah Lake with a Focus on the Relationships between
Water Column Regulators, Benthic Ecosystem Engineers,
and CyanoHABs*



File name: Ecological health and integrity of Utah Lake Progress Report 2019 Version 2.3

Progress Report 2019

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Cover photos:

Top photo: Utah Lake at Vineyard during low water in 2016. Clear water in foreground and up until pelicans visible in the distance is from abundant freshwater springs flowing off of the Wasatch Range that for the most part no longer flow directly into the lake but have lost their integrity due to unprecedented urban development.

Bottom photo: Utah Lake at Sandy Beach looking SW into Goshen Bay during low water in 2016. This was a relatively low water year and the mollusk hunter pictured would have been knee to waist deep in water at the same time period in 2019.

Summary

Utah Lake is the last large freshwater remnant of Lake Bonneville. It has undergone several primarily anthropogenically induced ecosystem shifts and hysteresis over the past 150-200 years resulting in alternative states in a now analog environment. Subsequently, the lake has lost most of its ecological integrity. Utah Lake's ecological health has also been severely compromised with a concomitant reduction in its resistance and resilience to ongoing and future perturbations. A multitude of stressors have prompted this loss, precluding its ability to self-regulate. These multiple stressors have liberated complex ecosystem constraints that until recently provided guards against the competitive advantages to cyanobacteria, allowing for frequent blooms. Stressors that are discussed in this progress report include:

- Watershed diversions and water quality degradation
- Water level fluctuations
- Turbidity
- Near zero flush rates
- Changes to temperature regime
- Loss or reduction of aquatic vegetation and allelopathy
- Dominance by invasive fishes
- Watershed-wide pesticide use
- Metapopulation and metacommunity dynamics: Isolation
- Pharmaceuticals
- Urbanization, the most recent and ongoing catastrophic shift
- Loss of native zoological assemblages including mollusks
- Analog zooplankton assemblages
- Transition from native benthic ecosystem engineers to analog invasive engineers.

As a result, Utah Lake has become a poorly functioning analog of its former self. Our society has inadvertently created a much-simplified analog Utah Lake ecosystem that often favors descendants of a cyanobacterial lineage whose primordial ancestors evolved in simple, stressed systems billions of years ago, not completely unlike present Utah lake.

Top-down biotic regulators, such as zooplankton, and benthic ecosystem engineers, including native mollusks and midges, have also lost most of their ability to regulate cyanobacteria blooms. Top-down regulators and benthic ecosystem engineers are the primary focus of this progress report and our research.

Almost no research has been conducted on zooplankton assemblages in Utah Lake. Their life histories, ecology, importance to the functioning of the lake, and even their taxonomies are poorly documented. A much needed and invaluable taxonomic update is included in the appendices. In response to the lack of ecological information, we are compiling and analyzing much needed research on zooplankton in the lake. Included in this progress report is our taxonomic evaluation and revision of zooplankton in the lake. Continued data collection will allow us to develop a definitive taxonomic key to be used

by other researchers. We are also the only researchers that we are aware of, that conduct research on the importance of mollusks and midges to the functioning of Utah Lake, including their role in regulating cyanoHABs.

Our research group has collected environmental and ecological data and holistically examined many of the problems that face Utah Lake for almost a decade. We conclude that without such comprehensive and integrative knowledge of these synergistic intricacies, we could have thoughtlessly assumed that nutrient reduction alone will control future cyanobacteria blooms.

We also have determined that the best way to understand and monitor the biological health of Utah Lake is to develop a Multimetric Index of Biological Integrity (MIBI). This index contains dozens of important metrics (measures) and is an ongoing part of our Utah Lake research. The preliminary MIBI is included as an appendix in this project report.

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Goals

The goal of our ecological research on Utah Lake is to increase our understanding of its ecological health and integrity, including environmental and ecological factors that can reduce its resistance and resilience and subsequently its ability to self-regulate cyanoHABs. We are doing this in a scientific holistic manner because we are well aware that focusing on only one or two stressors to the lake's ecosystem cannot possibly provide managers with enough understanding of how best to manage the lake into the future. As one major end point, we can use this knowledge to develop a suite of relevant metrics that can be used to monitor changes in Utah Lake's ecological health and address the causes.

Introduction and Background

Lake Bonneville: A Lake No More?

Utah Lake is one of the last remnant puddles of an ancient lake that until recently, in geologic time, spanned over 83,000 km² and at times was over 300 m deep; Lake Bonneville. Lake Bonneville was created when volcanic eruptions in southeast Idaho blocked an adolescent version of the Bear River and diverted its flow into central Utah, during the Pleistocene Epoch, between 130,000 and 30,000 years ago (Chronic 1990, Stokes 1986). Deep grabens¹ filled with diverted Bear River water and formed Lake Bonneville (Chronic 1990, Stokes 1986). At times, Lake Bonneville was as large as one of the largest of the Great Lakes, Lake Michigan and often was even deeper. Lake Bonneville and its remains sit on a limestone basin that in some locations is five miles thick. This limestone was deposited from highly productive, nutrient rich, shallow seas that ebbed and flowed in and out of existence starting in the Cambrian Period and lasted throughout the Pennsylvanian Period in the Paleozoic Era, some 540 to 250 million years ago (Chronic, 1990, Stokes 1986). Dozens of these ancient nutrient laden limestone layers deposited by marine life over the eons are clearly visible along the uplifted Wasatch Range (Figure 1).

Lake Bonneville water levels raised and lowered for thousands of years depending on climatic and geologic conditions, reaching its maximum height about 16,000 years ago. Then around 14,500 years ago, Lake Bonneville breached near Red Rock Pass, Idaho creating a cataclysmic flood into the Snake River, known as the Great Bonneville

¹ Grabens are depressed blocks of the Earth's crust bordered by parallel faults formed by range-faulting.

Flood. The level of Lake Bonneville water then remained below its outlet due to a drying climate starting about 10,000 years ago, eventually resulting in remnant puddles; Great Salt Lake, Utah Lake, Sevier Lake, Rush Lake, and Parowan Lake (Chronic, 1990, Stokes 1986). Terminal Great Salt Lake became salty, Utah Lake remains fresh- slightly brackish because of its limited outflow into the Jordan River, and Sevier Lake is now a dry lakebed. Utah Lake and Great Salt Lake are the only large waterbodies that remain of Lake Bonneville, and Utah Lake is the only large bodied freshwater relict remaining.



Figure 1. Mt Timpanogos in the Wasatch Range near Utah Lake showing the many stratifications of nutrient rich, ancient shallow seas that formed in the Paleozoic Era before being uplifted.

Utah Lake

“A lake that endured for millennia has been ecologically devastated within 150 years.”
(Janetski 1990)

The ecology of Utah Lake is tied to its past and the ecology of Lake Bonneville. Utah Lake is a slightly saline- eutrophic to hypereutrophic- alkaline-turbid- shallow-temperate lake with an average depth of about 1.5 to 2.8 m. It is about 40 km long by 21 km wide, with a surface area of about 384 km². Thirteen fish were native to Utah Lake, including the top predator, Bonneville cutthroat trout. Only two native fish species remain, the Utah sucker (*Catostomus ardens*) and the critically endangered June sucker

(*Chasmistes liorus*) (Carter 2005, Heckmann et al. 1981). Because it is shallow and has a large surface area, evaporation plays a key ecological role. Even though evaporation accounts for > 42% of its outflow; large amounts of evaporative losses are typical of shallow lakes residing within semi-arid to arid climates.

Although Utah Lake historically functioned as a natural shallow lake ecosystem, it has undergone what are known as ecological hysteresis² (Nikanorov and Sukhorukov 2008, Beisner et al. 2003) and catastrophic ecosystem shifts³ (Scheffer et al. 2001, Beisner et al. 2003). This has resulted in dramatic transitions known as ‘alternative stable states’⁴, primarily driven by human economic activity. The most important factors that contributed to these shifts include; sedimentation, loss of submerged aquatic vegetation, abrupt transition from a native fishery to non-native fishery most notable the introduction of benthic feeding carp, loss of native fish and invertebrates including losses of critically important filter feeding bivalves and algal grazer snails, water diversions, increased nutrients and toxic metals, man-made water- level fluctuations, and multiple other types of pollution and pollutants⁵ (Deseret News 1967, Carter 2005, Janetski 1990). Utah Lake is no longer a natural lake but is managed as an *operational water supply reservoir*

² Hysteresis “implies that communities and ecosystems might be easily pushed into some configurations from which it may prove much more difficult for them to recover” (Beisner et al. 2003). Hysteresis is “where the observed equilibrium of a system cannot be predicted solely based on environmental variables, but also requires knowledge of the system's past history”. (Wikipedia accessed November 24, 2019).

³ All ecosystems are exposed to gradual changes in climate, nutrient loading, habitat fragmentation or biotic exploitation. Nature is usually assumed to respond to gradual change in a smooth way. However, studies have shown that smooth change can be interrupted by sudden drastic switches to a contrasting state (see Footnote 4). Although diverse events can trigger such shifts, studies show that a loss of resilience usually paves the way for a switch to an alternative state. This suggests that strategies for sustainable management of such ecosystems should focus on maintaining resilience (Scheffer et al. 2001).

⁴ Marten Scheffer (2009) provided a simple explanation for alternative stable states, “Suppose that you are in a canoe and gradually lean farther and farther over to one side to look at something interesting underwater. Leaning over too far may cause you to capsize and end up in an alternative stable state upside down. Although the details of the theory of alternative stable states may appear tricky, several key properties can be seen in this simple example. For instance, returning from the capsized state requires more than just leaning a bit less to the side. It is difficult to see the tipping point coming, as the position of the boat may change relatively little up until the critical point. Also, close to the tipping point resilience of the upright position is small, and minor disturbances such as a small wave can tip the balance.

⁵ EPA’s general definition of pollution is the man-made or man-induced alteration of the chemical, physical, biological, and radiological integrity of water. Whereas, pollutant is dredged spoil, solid waste, incinerator residue, sewage, garbage, sewage sludge, munitions, chemical wastes, biological materials, radioactive materials, heat, wrecked or discarded equipment, rock, sand, cellar dirt and industrial, municipal, and agricultural waste discharged into water (<https://www.epa.gov/cwa-404/clean-water-act-section-502-general-definitions>).

because of a man-made dam at its outlet into the Jordan River built in 1872 and created to serve large water demands.

The Utah Lake ecosystem prior to Mormon settlement in the mid-1800s certainly was not at all what it is today. Utah Lake was arguably the most scenic and productive cold-water fishery in the western USA prior to Mormon settlement (Janetski 1990, Carter 2005, de Escalante 1776) and was a unique and ecologically important part of Utah's (and the nation's) natural heritage.

Bonneville cutthroat trout and twelve other fish species thrived by the millions. More species of freshwater mollusks called Utah lake home than anywhere in western North America and based on our understanding of freshwater mollusk ecology, they likely dictated most of its ecological functioning. Birds, wildlife, and Native Americans thrived in its environs. In fact, historical records show that the first Mormon settlers likely would not have survived their first few winters in Utah if not for the tremendous native fishery in Utah Lake, particularly the Bonneville cutthroat trout fishery, and the generosity of Native Americans residing along its shores (Carter 2005, Heckmann et al. 1981). The following are a few excerpts from Janetski (1990) and Carter (2005) regarding Utah Lake prior to settlement:

“... the valley and the borders of the lake of the Timpanogos (Utah Lake) ... is the most pleasant, beautiful and fertile in all of New Spain The lake and the rivers which empty into the lake abound in many kinds of choice fish; there are to be seen there very large white geese, many varieties of duck, and other kinds of beautiful birds never seen elsewhere; beavers, otters, seals, and other animals which seem to be ermines by the softness and the whiteness of their fur.” (Escalante 1776).

“I was at Utah Lake last week and of all the fisheries I ever saw, that exceeds all. I saw thousands caught by hand, both by Indians and whites. I could buy a hundred, which each weigh a pound, for a piece of tobacco as large as my finger. They simply put their hand into the stream and throw them out as fast as they can pick them up Five thousand barrels of fish might be secured there annually”(Pratt 1849 in Janetski 1990).

“Indeed, so great was the number of suckers and mullets passing continuously upstream that often the river would be full from bank to bank as thick as they could swim for hours and sometimes days together.”— George Washington Bean, 1854.

Although Utah Lake continues to be highly productive and is sanctuary for thousands of migratory birds and the last remnant wild population of June Sucker; its native fauna has all but disappeared and its waters are now primarily dominated by taxa such as cyanobacteria, algae, worms (oligochaetes), midges (chironomids), and introduced fishes. Utah Lake is now a highly regulated and abused reservoir ecosystem that has undergone human induced ecological hysteresis and catastrophic shifts and no

longer resembles its natural self, pre-settlement. Consequently, Utah Lake is biologically impoverished and polluted. Understandably, Utah Lake is now severely underappreciated by naïve citizens and managers unfamiliar with its exceptional past.

In this progress report, we discuss historical and ongoing factors that cause(d) ecosystem shifts that degrade(d) and reduce(d) its resistance and resilience⁶ to perturbation, its ecological health and integrity, and its inability to self-regulate cyanoHABs. We then discuss the relationships between water column regulators, benthic ecosystem engineers, and cyanoHABs⁷ based on our ongoing research of Utah Lake's ecology. We submit that "absence of fundamental knowledge of the mechanisms driving harmful algal blooms frustrates most hope of forecasting their future prevalence" (Wells et al. 2015) or for developing relevant nutrient criteria. This progress report is an important addition to this limited knowledge.

Ecological Health and Integrity

The biological⁸ and ecological integrity⁹, of the majority of large temperature lakes throughout the world have been lost or severely compromised and the status of their ecological health is questionable. Although ecological integrity and ecological health are often used interchangeably and their definitions simplified to meet management goals, they are not the same (Karr 1993, 1996). According to Karr (1996):

"Integrity implies an unimpaired condition or the quality or state of being complete or undivided; it implies correspondence with some original condition. Health, on the other hand, implies a flourishing condition, well-being, vitality, or prosperity". "An ecosystem is healthy when it performs all its vital functions normally and properly; a healthy ecosystem is resilient, able to recover from many stresses; a healthy ecosystem requires minimal outside care" (Karr 1996).

Regrettably, and despite its ancient lineage, Utah Lake is now a highly regulated and abused reservoir ecosystem and is consequently biologically impoverished. According to Karr (1996), "if biotic impoverishment is the problem, then protecting the integrity of" Utah Lake's "biological system(s) must be the goal". However, it is difficult to find and

⁶ Resistance is the property of communities or populations to remain "essentially unchanged" when subject to disturbance. Resilience is the capacity of a population/community of organisms or an ecosystem to respond to a perturbation or disturbance by resisting damage and recovering quickly.

⁷ There is a misunderstanding that harmful algal blooms in Utah Lake are caused by algae. In fact, blooms in the lake are caused by cyanobacteria, not algae. We, therefore, refer to harmful algal blooms as cyanoHABs throughout this report.

⁸ Biological integrity refers to the capacity to support and maintain a balanced, integrated, adaptive biological system having the full range of elements (genes, species, assemblages) and processes (mutation, demography, biotic interactions, nutrient and energy dynamics, and metapopulation processes) expected in the natural habitat ... (Angermeier and Karr 1994, Karr and Dudley 1981, Karr et al. 1986).

⁹ Ecological integrity is the sum of physical, chemical, and biological integrity (Karr 1993, 1996).

quantify any of Utah Lake's remaining ecological integrity, although as long as it is alive it must have some hidden shred of integrity.

The ecological health of Utah Lake on the other hand, given its age, genealogy, and historical and current abuse, appears to be quantifiable and salvageable. All lake ecosystems have limited and somewhat predictable lifespans (Wetzel 2001, Cole and Weihe 2016). All lakes evolve, grow old, and die. Because Utah Lake evolved with nutrient rich geological parent materials and is now for the most part a terminal reservoir with high surface area to depth ratio, severely limited flushing, in an arid environment with large evaporation rates; nutrient and chemical concentrations continue to increase as it journeys towards the end of its life. Increasing primary productivity, often eutrophic in spring and autumn and hyper eutrophic in summer is the inevitable 'stable state' for a lake with ecological characteristics and history, such as Utah Lake.

Utah Lake is not managed for its health or integrity but is principally managed for water use and consumption ever since it was first settled by non-native Americans. Regulations have been established for Utah Lake to protect water quality 'designated beneficial uses' including 'warm water fisheries' and recreational use, however there is a large difference in how the lake is managed and how regulations are applied.

Dozens of measures (metrics) of Utah Lake's current ecological health derived from other lakes throughout the world are available from the literature and from ours and other's past and present research on the lake. Given our limited knowledge of Utah Lake's ecosystem function, these metrics can be accurately but imprecisely quantified and allow us to monitor changes to Utah Lake's ecological health.

In this progress report we review environmental factors that contribute(d) to ecosystem shifts in Utah Lake and that affect its health and integrity including self-regulation and resistance and resilience to cyanohABs. We further evaluate important interactions between phytoplankton, zooplankton, and invasive fishes, and the importance of benthic ecosystem engineers for maintaining the lakes health. These results will allow us to greatly improve and refine scientifically valid metrics and indices urgently needed to monitor Utah Lake's ecological health.

Factors Affecting Utah Lake's Ecosystem Shifts, Integrity, Health, and CyanohABs

Watershed Diversions and Degradation Favors CyanohABs

One of the leading causes of Utah Lake's loss of ecological integrity, declining health, and vulnerability to perturbation has been and continues to be water diversions throughout the watershed, starting in its headwaters. Utah Lake resides in an arid to semi-arid environment, has a very large surface area to volume ratio, and loses a sizable portion of its water to evaporation. Utah Lake's ecosystem depends on the surrounding watershed for most of its water, predominantly from the Wasatch Range of the Middle

Rocky Mountains in the form of surface water and groundwater. However, much of this water is diverted before it reaches the lake and/or is polluted before entering the lake. Loss of a natural water regime has negative ecological effects that reverberate throughout Utah Lake's ecosystem, including its ability to self-regulate cyanoHABs. Again, Utah Lake is not a naturally functioning lake but a water supply reservoir. There is a large difference between how natural lake ecological function and reservoir ecological function that has been documented by countless research publications, white papers, and management agencies. Entire journals are devoted to this subject including those that caution applying regulatory assessments based on natural lake ecosystems.

Rivers

The major tributaries to Utah Lake, including the Provo River, Spanish Fork, Beer Creek, Carrant Creek, and American Fork River are all highly regulated. When their flows into Utah Lake are not completely curtailed and disconnected, they are severely reduced and do not follow a natural flow regime that a normally functioning lake would recognize. As with lakes, it is well documented that flow regulation degrades a river's integrity and threatens its health. In addition to flow regulations and unnatural flow regimes, all of these tributaries are also polluted by other types of pollutants and have had their ecologic integrity and health further conceded.



Figure 2. Timpanogos Creek a headwater tributary to the Provo River, the highly regulated main tributary to Utah Lake. This tributary originates on Mt Timpanogos (Figure 1) and flows through mostly limestone bedrock.

Spring Tributaries

The condition and health of spring tributaries and ground water inputs to Utah Lake is poorly understood. We are compiling information on these critically important water sources and have observed that most spring tributaries are in poor health. We have collected ecological data on one abused spring tributary near Vineyard and are currently analyzing this data and will be able to monitor any ecological changes in the future.

Fluctuating Water Levels Favor CyanoHABs

It is well known, that rapidly fluctuating lake (reservoir) levels, particularly unpredictable fluctuations, are detrimental to biota and ecological health, including resistance and resiliency (Thornton et al. 1990, Cowx 2008). Utah Lake levels fluctuate substantially both intra- and inter-annually, naturally and anthropogenically. The lake is nourished mostly by snow fed rivers and streams, springs, and ground water. Snowpack levels and rain events vary between years and effect Utah Lake levels. Utah Lake is a highly regulated reservoir and is principally managed for downstream agriculture use. Consequently, regulators have designated a lake compromise level¹⁰ of 1368.4 meters. Utah Lake's tributaries are also managed for agriculture use and domestic use; much of the tributary water is diverted before it reaches the lake. Weather and climate related lake levels are exacerbated by ecologically unpredictable human demands.



Figure 3. Utah Lake water elevations from 1884 until 2016. Note extreme low levels in 1935, which had catastrophic effects on the lake's biota. Red line is 'compromise' level = 1368 meters.

¹⁰ As a result of the 1983-1984 flooding, a lawsuit was filed for compensation due to flooding based upon breach of contract of the previous compromise level. In 1985, a new compromise level was reached which governed the maximum level of the lake. The new level was chosen to be 4,489 feet (1,368 m) above sea level. When the water level in Utah Lake exceeds this level, the Jordan River pumps and gates are left open. The new compromise level also means that the lake's elevation was below Jordan River's stream bed (https://en.wikipedia.org/wiki/Utah_Lake).

Low Lake Levels

In 2016, snowpack levels were low and water demand was high. This caused lake levels to drop significantly. Utah Lake levels fell to about 2.13 ft below compromise level, the lowest it has been in over 23 years. The southern portion of Goshen Bay went completely dry in late summer for several months causing the clay and mud substrate to bake kiln-dry hard (Figure 4, Figure 5). Obviously, very few if any sessile, aquatic obligate organisms survived (e.g. midge larvae, mollusks, oligochaetes, etc.) (Figure 6). Those that were able, moved to wetted habitats. It also takes many years for benthic macroinvertebrate assemblages to fully recolonize these large areas once water returns, which translates into a substantial loss of benthic secondary production and ecosystem function that resonates throughout the food web.



Figure 4. Exposed sun baked clay and mud substrate of southern Goshen Bay, 2016. Several square miles were affected. Obviously, very few if any benthic invertebrates survived, and complete recolonization will take several years.



Figure 5. Exposed sun baked clay and mud substrate of southern Goshen Bay, 2016. Several square miles were affected. A few early successional ‘weeds’ attempted to colonize these areas.



Figure 6. Invasive Asian Clam (*Corbicula* sp.) stranded during low water levels along the shoreline of Utah Lake near Vineyard, UT, summer 2016. Hundreds of these clams were observed stranded and eventually either perished due to desiccation or were preyed upon.

Low lake levels in 2016 revealed the existence and importance of large quantities of groundwater spring flows, particularly along the lake’s eastern shores (Figure 7, and cover images).



Figure 8. Exposed springs flowing into Utah Lake, near Vineyard, UT 2016. The entire open areas behind the front row of cottonwood trees in foreground including those trees behind the front row have been lost to housing subdivisions. No open areas remain. The spring water sources are now mostly paved over and filled and any remaining ecological integrity destroyed.

Low lake levels also expose fine shoreline sediments that are highly erosional and only colonized by early succession plants (cover image 1), including invasive phragmites and tamarisk. Low water years can expose large amounts of habitat and large quantities of benthic macroinvertebrates (e.g. midge larvae, corixids, etc.) beneficial to wading shorebirds (Figure 9). Low water years also restrict adult fishes from competing with wading shorebirds for macroinvertebrates in shallow habitat a net benefit to wading birds and more importantly allowing macroinvertebrate populations to increase as a result of reduced fish predation. However, the clay content of the exposed habitat in some sections of the lake has been known to trap shorebirds and waterfowl, much to their demise (Richards and Miller, personal observations). Low water levels in Utah Lake can exacerbate cyanoHABs by increasing nutrient concentrations, increasing temperatures, and altering the food web, etc. or reduce cyanoHABs by allowing benthic invertebrate induced sediment aeration to continue or increase.



Figure 9. White faced ibis gorging itself on adult midges and ephydrid flies during the low water year, 2016. Low lake levels allow wading birds access to shallow sections of Utah Lake that otherwise would be inundated during high lake levels.

High Lake Levels

During high water years, near shore areas of Utah Lake are inundated and the water is too deep for wading shorebirds, which have to seek food resources and wading habitat elsewhere. 2019 was a relatively wet year and lake levels remained high and near compromise. Lake elevation at the time of this report was 1367.7 meter or about 30 cm below compromise. 2019 high waters inundated shoreline areas dominated by invasive phragmites and tamarisk and created wetlands that were hospitable to many organisms including zooplankton, macroinvertebrates, and frogs (primarily native Boreal Chorus Frogs and invasive American Bullfrogs). Inundated aquatic vegetation also provided fish spawning habitat and nurseries for native and invasive larval and juvenile fishes. Zooplankton were able to take advantage of inundated phragmites in 2019. The following link to a YouTube video shows how abundant zooplankton can be within inundated phragmites stands:

Zooplankton in inundated phragmites video

https://youtu.be/orJ2_ILDb7I.

Flooded phragmites in 2019 also provided exceptional spawning habitat for invasive carp. Personal observations revealed in autumn 2019 to be a banner year for juvenile fish recruitment in inundated vegetation and we predict it to be a very strong year class for several species and will have food web effects for several years.

The following two links to YouTube videos provides an understanding of how important high lake levels are to sustaining large populations of this nuisance species:

Carp spawn video 1

<https://youtu.be/kefBvIv5Fhw>

Carp spawn video 2

<https://www.youtube.com/watch?v=-HnwlZ1otH0>

Any assessment of Utah Lake's ecological health needs to consider lake level fluctuations and their ecological impacts, for better or worse.

Note: The following sections on additional environmental conditions that favor cyanoHABs and in particular the following three sections; [Turbidity Favors CyanoHABs](#), [High Levels of Solar Radiation and Bouancy Favors CyanoHABs](#), and [Limited Flush Rates Favor CyanoHABs](#) are brief literature review synopses. There is a large and increasing amount of research being conducted on cyanobacteria ecology and the environmental conditions responsible for blooms in a response to ever increasing cyanoHABs, worldwide. Consequently, some of what we report in the following sections may become outdated or elaborated upon as our knowledge rapidly increases.

[Turbidity Favors CyanoHABs](#)

Sediments in Utah Lake are easily resuspended in the water column by even the slightest wind and wave action (DWQ Utah Lake Science Panel 2019, Randall et al. 2017). These suspended solids (total suspended solids = TSS = turbidity) can remain in the water column for several days or longer after strong wind events (Richards and Miller, personal observations) and dissolved solids constantly maintain turbidity in this eutrophic lake (see video below).

<https://youtu.be/z07JTJfin7c>

Turbidity significantly affects biological activity and productivity and directly influences the occurrence and abundance of cyanobacteria because they are considered superior competitors to green algae under low light conditions (i.e. high turbidity) (Smith 1986, Jensen et al. 1994, Scheffer et al. 1997). In addition to being superior light competitors, cyanobacteria also promote low light conditions by having a higher turbidity per unit of phosphorus than other phytoplankton, thus they increase shading starting at the onset of blooms (Scheffer et al. 1997). It is important to understand that this phenomenon occurs in eutrophic lakes regardless of water column nutrient levels (Scheffer et al. 1997)(Figure 10). Given Utah Lake's almost constant high turbidity, it is one of the more important mechanisms determining cyanoHABs in the lake.

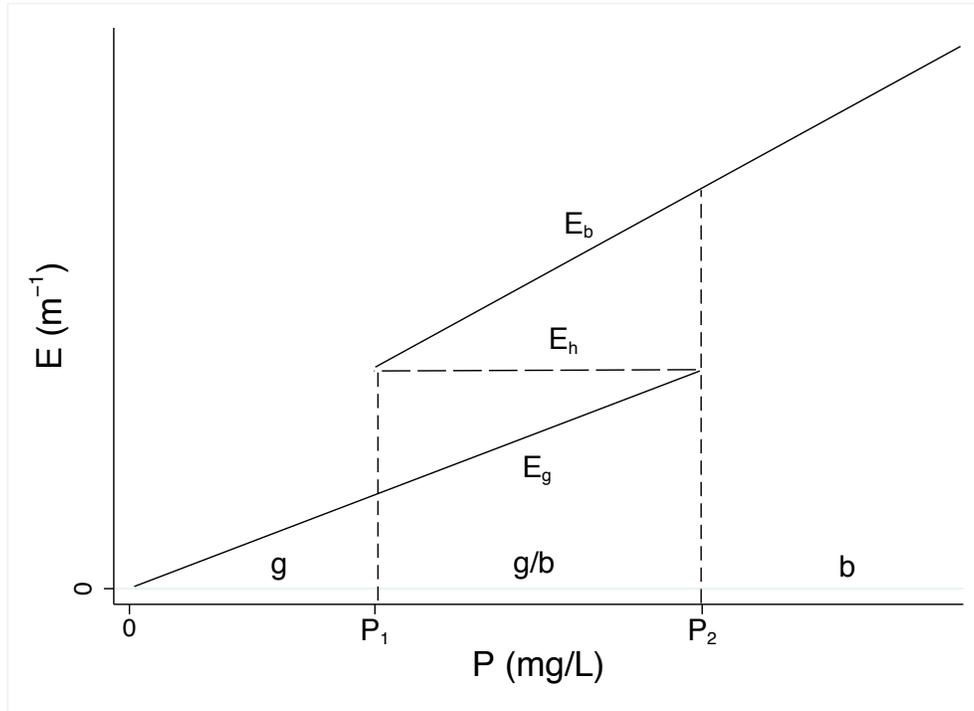


Figure 10. Green algae/cyanoHAB hysteresis as the response of turbidity, E (m^{-1}) with respect to phosphorus, P (mg/L) from Scheffer et al. 1997. “Starting in the region [b] where a cyanobacterial monoculture is the only stable state, a shift to the alternative state will only occur after the lower bifurcation line is passed. On the other hand, starting from the region [g] where the cyanobacterial monoculture is unstable, a switch to cyanobacteria will only occur after the upper bifurcation line is crossed. In the region [g/b] (P_1 to P_2) where two alternative equilibrium states are stable, the system will stay in its current equilibrium state, unless perturbations bring it within the basin of attraction of the alternative state.” It can be seen from this diagram that a transition from blue- green to green dominance is always associated with a conspicuous drop in turbidity, and that in the vicinity of the breakpoints a small variation of a control parameter can be sufficient to induce the transition.

Very preliminary data based on extremely limited samples sizes support the literature that phytoplankton blooms increase turbidity in Utah Lake (Figure 11, Figure 12). We are in the process of compiling and analyzing additional data from WFWQC and DWQ.

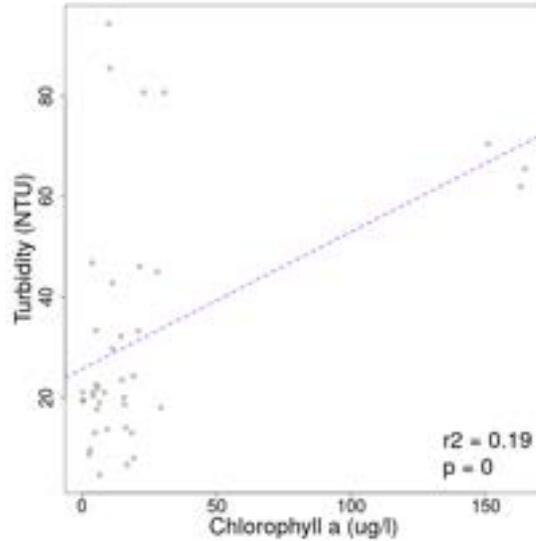


Figure 11. Very preliminary analysis of relation between Chlorophyll a and turbidity in Utah Lake. From UDWQ Shiny app (<https://udwq.shinyapps.io/UtahLakeDataExplorer/>). High turbidity at low *Chl- a* levels show sediment-based turbidity, whereas the three high turbidities at high *Chl- a* levels may show effects of phytoplankton bloom on turbidity. More analyses are required due to very low sample size, particularly in the Chlorophyll a range between 40 to 125 ug/l provided by UDWQ consequently the regression line and results should be considered irrelevant.

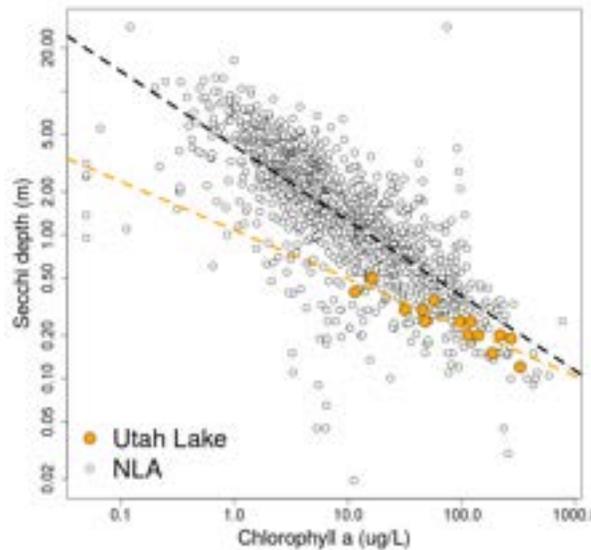


Figure 12. Very preliminary analysis of relation between Chlorophyll a and turbidity in Provo Bay. From UDWQ Shiny app (<https://udwq.shinyapps.io/UtahLakeDataExplorer/>). More analyses are required due to very small sample size. Yellow = data from Provo Bay; black open circles and line = data from National Lake Assessment Program. Provo Bay regression line (yellow dashed line) should be ignored and more relevant statistical analyses performed on this and future data.

The following figure, Figure 13, represents phosphorus levels (TP)(mg/L) in the water column of Utah Lake across all locations from 1989 to 2018 compiled by UDWQ and graphed by month. The red reference lines are 0.08 and 0.17 mg/L TP that represent hysteresis cutoffs of green algae/cyanobacteria relationships between turbidity and TP at

a flush rate of 0.1/day (retention time of 10 days) based on Scheffer et al. 1997 (see Figure 10 above and Figure 9A in Scheffer et al. 1997). At $TP \cong 0.17$ mg/L, cyanobacteria will dominate the lake even if turbidity is zero at a relatively high flush rate of 0.1/day. At lower flush rates, cyanobacteria will dominate even at lower TP levels. At $TP \cong 0.08$ mg/L or higher, cyanobacteria will tend to dominate when turbidity is at moderate levels reported by Scheffer et al. (1997) (i.e. $E(m^{-1}) \cong 0.9$) at a flush rate of 0.1/day.

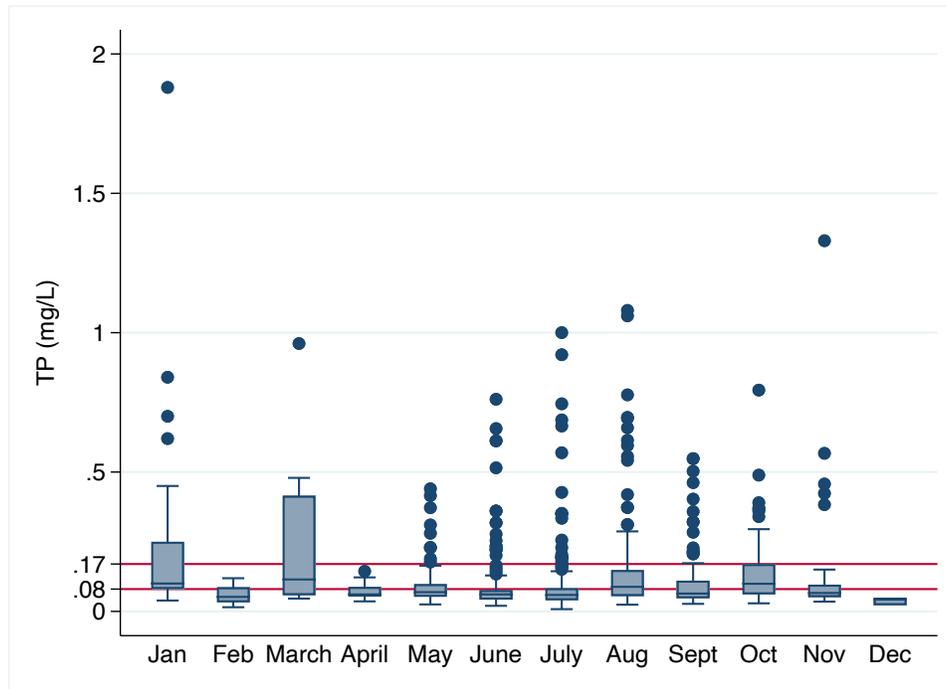


Figure 13. Monthly TP (mg/L) levels in Utah Lake based on UDWQ data from 1989 to 2018. Median, 25th, 75th, range, and outliers. Red reference lines are Scheffer et al. (1997) green/cyanobacteria hysteresis cutoffs at flush rate = 0.1/day (retention time = 10 days). N = 1112 TP recordings.

From Figure 13, we can see that much of the time, TP levels in Utah Lake favor cyanoHABs given its high turbidity and low flush rate (long retention time) and we can infer that lake wide TP reduction would have to be substantial to reduce cyanoHABs. However, other factors can influence cyanoHABs (see all sections of this report). At this time, we have not calculated retention times for Utah Lake, although preliminary calculations show the retention time to be between 1 and 2 years. Clearly, a retention time of one year, predisposes Utah lake to cyanoHABs as nutrients will continually accumulate in the water column and sediments and the shallowness provides easy access of recycled nutrients to reach the photic zone. We also do not know turbidity and light extinction coefficient parameters for any of the green algae or cyanobacteria that cause blooms in Utah Lake but assume those values will be similar to those used by Scheffer et al. (1997). We plan to employ a marine photometer in our regular surveys starting in 2020.

Solar Radiation and Buoyancy Favor CyanoHABs

Not only are many cyanobacteria species superior competitors for low light attenuation, they also are able to withstand high solar radiation and can regulate their buoyancy to maximize irradiance levels (Berg and Sutula 2015). After all, cyanobacteria have been present and evolving in aquatic ecosystems since the beginning of life on Earth and have easily withstood several mass extinctions and the test of time.

Cyanobacteria genera that occur in Utah Lake including *Microcystis*, *Planktothrix* and *Aphanizomenon* have the ability to regulate their buoyancy by a combination of gas vesicles and carbohydrate storage products (Berg and Sutula 2015). Gas vesicles allows positive buoyancy whereas carbohydrate storage products have the opposite affect (Walsby 1994, 2005) and the type and amount of carbohydrate storage products produced varies by species and irradiance level (Berg and Sutula 2015, Visser et al. 1997, Wallace and Hamilton 1999). Cyanobacteria can become neutrally buoyant and adjust upward lift by balancing gas vacuoles and carbohydrate storage products (Walsby et al. 2004), thus they can control their vertical position in the water column by regulating the amount of carbohydrate storage products consumed (Berg and Sutula 2015, Konopka et al. 1987, Wallace and Hamilton 1999). In order to maximize their growth potential, filamentous cyanobacteria can sink or float at speeds up to 0.3 m per day in order to position themselves at just the right depth (Walsby 2005). Obviously, nutrient availability affects carbohydrate production and buoyancy regulation; nitrogen starved cells have excess carbohydrate stores and tend to lose buoyancy more easily than nutrient sufficient cells (Brookes and Ganf 2001).

High irradiance levels are in part why cyanoHABs occur more frequently in summer. Adjusting buoyancy also allows many species of cyanobacteria to settle down on the benthic substrate at night and actually acquire sediment-derived nutrients. Cyanobacteria can acquire unoxidized iron and free phosphorus when the surface layer of sediments becomes hypoxic or anoxic. This transition between sediment anoxia and oxic conditions plays a very important role in whether green algae or cyanobacteria dominate in Utah Lake. We reported this important phenomenon in a section of our previous progress report (Richards and Miller 2019), which we have attached in **Appendix 1**, including how the presence or absence of chironomid larvae can influence these interactions.

Limited Flush Rates Favor CyanoHABs

High flush rates also reduce the probability of cyanoHABs because of their relatively slow growth rates (Scheffer et al. 1997, Figure 14). Alternatively, loss or reduction of flushing increases cyanoHABs (Jónasson and Adalsteinsson 1979, Einarsson et al. 2004). Unfortunately, Utah Lake rarely meets compromise level (see Figure 3) and unless lake

water is actually physically pumped out of the lake, it most often functions as a terminal lake with near zero flush rate for a good portion of the year.

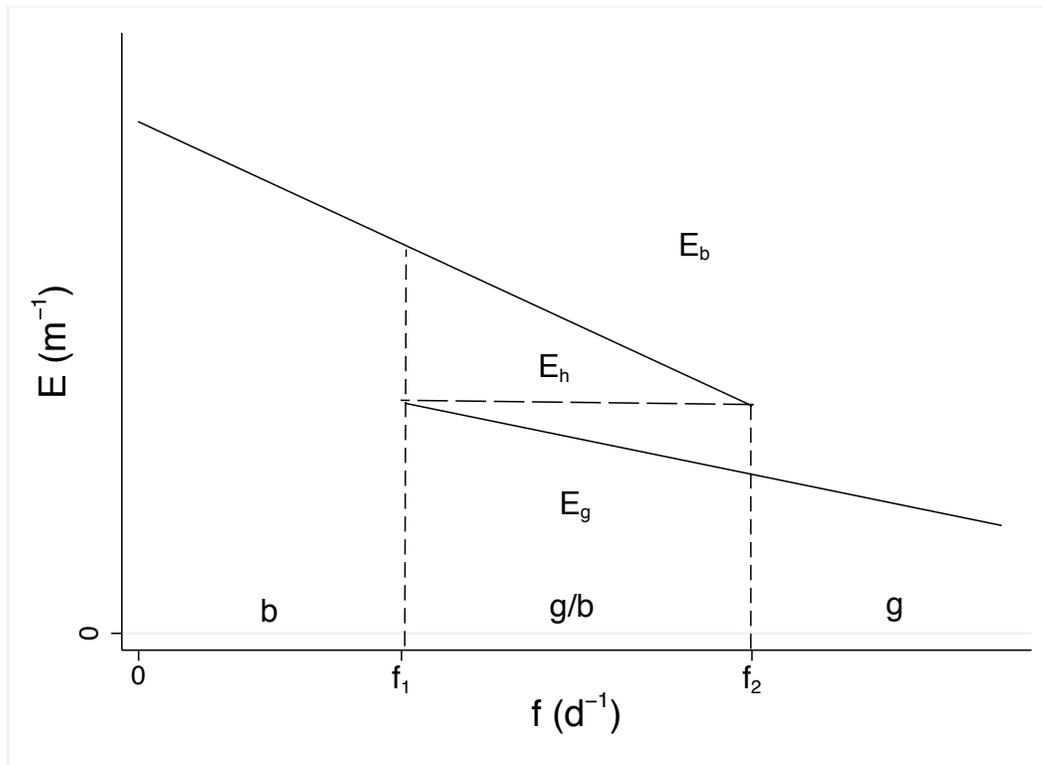


Figure 14. Green algae/cyanoHAB hysteresis as the response of turbidity, E (m^{-1}) with respect to phosphorus, flush rate, f (d^{-1}) from Scheffer et al. 1997. “Starting in the region [b] where a cyanobacterial monoculture is the only stable state, a shift to the alternative state will only occur after the lower flush rate bifurcation line is passed. On the other hand, starting from the region [g] where the cyanobacterial monoculture is unstable, a switch to cyanobacteria will only occur after the upper flush rate bifurcation line is crossed. In the region [g/b] (f_1 to f_2) where two alternative equilibrium states are stable, the system will stay in its current equilibrium state, unless perturbations bring it within the basin of attraction of the alternative state.” It can be seen from this diagram that a transition from blue- green to green dominance is always associated with a conspicuous drop in turbidity, and that in the vicinity of the breakpoints a small variation of a control parameter can be sufficient to induce the transition.

Scheffer et al. (1997) noted in their study that beyond a flush rate of $\approx 18\%$ lake volume per day, cyanobacteria were predicted to be absent irrespective of the nutrient level. Again, Utah Lake has flush rates much lower than 18% per day with rates that are often close to 0% per day.

The importance of flush rates controlling cyanoHABs in Utah Lake can be clearly shown by comparing environmental conditions between Utah Lake and downstream impounded wetland ponds in Great Salt Lake (GSL) within the same drainage (Table 1).

Table 1. Comparison of several environmental factors between Utah Lake and downstream Great Salt Lake (GSL) impounded wetland ponds that may be responsible for cyanoHABs in Utah Lake but not the ponds. All values are best estimates at this time and will be updated pending further analyses.

	Utah Lake	GSL Impounded Wetland Ponds
Mean depth (m)	≈ 2	≈ 2
Water column TP¹ (mg/l) (mean and 95%CI)	0.11 (0.10; 0.12)	1-3
Estimated volume flush rates² (% day⁻¹)	0 to << 18	≥ 18
Fish production/biomass	Very high	Very high
Aquatic vegetation cover	Sparse	Abundant
Water source	Utah Lake	Mostly Utah Lake during summer, POTWs, tributaries
CyanoHABs	Frequent in summer	Absent

¹Utah Lake water column TP estimates from UDWQ online database. N = 1112 recordings. GSL impounded wetlands ponds TP estimates from sources including the Jordan River and State Canal at closest locations directly upstream of ponds.

²Lake volume flush rates estimated from Miller and Richards personal observations and needs quantification. Flush rate = 0.18 day⁻¹ is from Scheffer et al. (1997) when cyanobacteria were predicted to be absent irrespective of the nutrient level.

GSL wetland ponds receive much of their water from POTW discharge and Utah Lake via the Jordan River, especially in summer that transports large amounts of phytoplankton originating from Utah Lake, including cyanoHABs (Richards 2019, Miller 2013). These ponds receive over an order of magnitude as much TP than what is in the water column in Utah Lake (Table 1). Interestingly, cyanoHABs have not been reported in GSL wetland ponds, whereas Utah Lake has frequent blooms in late summer. Flush rates are the only the major environmental difference between these water bodies which provides strong evidence that low to nonexistent flushing of Utah Lake directly contributes to cyanoHABs within the lake. We have initiated a study comparing these two water bodies focusing on causal reasons for absence of cyanoHABs in wetland ponds and abundance of blooms in Utah Lake.

It appears that nutrient reduction alone cannot decrease cyanobacteria dominance in Utah Lake or unless strong management actions are implemented. For example, unrealistic efforts to reduce wind and wave derived ultra-fine sediment suspension turbidity via massive wind breaks and/or intensive pumping and flushing of water out of the system: Both inefficient and unconventional strategies. Alternatively, turbidity caused by invasive species derived benthic bioturbation can be mediated with careful planning, as can implementation of other biomanipulation methods.

Even though turbidity, solar radiation, and flush rates in shallow lakes such as Utah Lake often supersede nutrients in regulating cyanoHABs; recently extirpated top down (higher trophic level) regulators, such as native bivalves and zooplankton assemblages,

reduced cyanoHAB frequency and intensity in the lake. However, Utah Lake is an out-of-balance analog with weakened resistance and resilience. Subsequently, cyanoHABs will continue to occur or may increase in frequency and intensity. It therefore becomes imperative to understand the current health of Utah Lake and in particular, the ecology and interactions between top down controllers of cyanoHABs in Utah Lake including, water column regulators and benthic ecosystem engineers. This understanding may then provide managers with more efficient alternative methods for reducing cyanoHAB frequency and intensity in the lake.

Temperature and CyanoHABs

Temperature is also an important environmental factor affecting cyanoHABs throughout the world and obviously in Utah Lake (Wells et al. 2015, Scheffer et al. 1997, Scheffer et al. 2001, Berger 1975, Reynolds 1988, Bissenger et al., 2008, Eppley 1972). In general, cyanoHABs increase with increased temperatures, although winter low temperatures also can influence cyanobacteria populations, including ice covered conditions. Climate change forcing will directly affect Utah Lake water temperatures and other key ecological variables and interactions that regulate cyanoHABs (Wells et al. 2015, Scheffer et al. 2001) (Figure 15).

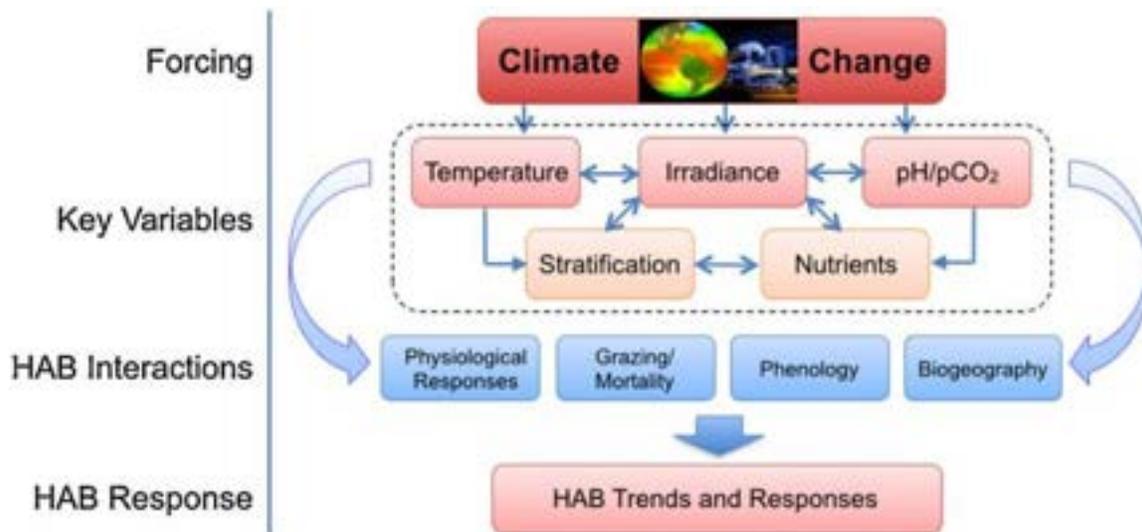


Figure 15. Generalized relationships between cyanoHAB trends and responses, interactions, key variables, and climate change forcing. Taken from Wells et al. 2015.

It is expected that warming water temperatures will increase potential growth rates of cyanoHABs (Wells et al. 2015). CyanoHAB motility (Kamykowski and McCollum 1986), germination (Montesor and Lewis, 2006, Yamochi and Joh, 1986), nutrient uptake, photosynthesis, and other physiological processes (Beardall and Raven 2004, Raven and Geider 1988) are all influenced by temperature (Wells et al. 2015). For example, Li (1980) showed that the optimal temperature for phytoplankton

photosynthesis is generally greater than the optimum for growth. The biogeographical breadth of phytoplankton species niches, including cyanoHAB species, is largely determined by temperature regimes (Longhurst 1998, Okolodkov 1999, 2005, Wells et al. 2015), including often underappreciated regulation by winter low temperatures (Scheffer et al. 1997, Berger 1975, Reynolds 1988).

The effects of winter low temperatures and duration of ice cover on cyanoHAB and other phytoplankton succession on the community level food web in Utah Lake has been grossly understudied. However, Richards and Miller (2019c) and Richards and Miller (2017) reported seasonal succession of phytoplankton in the lake in the recent past. The Wasatch Front Water Quality Council is increasing its winter ecological sampling efforts to better understand this phenomenon.

Obviously, cyanoHAB species, can show strong intraspecific differences in temperature tolerance and responses (de Boer, 2005, Wells et al. 2015). Ecological evolution prescribes that all phytoplankton species throughout the world, including Utah Lake, have “a temperature niche described by optimal, lethal and tolerable water temperatures at which cells do not grow well, but can survive” (de Boer, 2005, Fehling et al. 2004, Magaña and Villareal 2006, Rhodes and O’Kelly 1994, Wells et al. 2015).

Based on our research and knowledge of the Utah Lake ecosystem, we anticipate that temperature effects will cascade throughout the ecosystem as the climate changes including; affecting cyanobacteria and algal assemblages, likely increased bloom intensity and frequency, competition between phytoplankton grazer assemblages and their top down control, parasites, and planktivorous fish assemblages (Alheit et al., 2005; Edwards and Richardson, 2004; Hansen, 1991). Without a solid understanding and incorporation of the effects of changing temperature regimes, establishing nutrient criteria levels for Utah Lake will fall far short of any intended cyanoHAB management goals.



Figure 16. Unseasonably cold temperatures in October 2019 resulted in a rapid freeze of shorelines of Provo Bay, Utah Lake. Two days prior to taking this photo, air temperatures were 65° F and zoo biota, particularly juvenile fishes were abundant (https://youtu.be/5qYfa4_a_lk). Remaining emergent aquatic vegetation stands, including phragmites stands after aggressive control and removal projects provided some refugia for biota from the freeze. Note open water towards the center of the bay. One week later the ice was gone.

Allelopathy and CyanoHABs

An important but completely overlooked ecological factor that likely contributes to cyanoHABs in Utah Lake at least within the littoral zone, is the ecosystem- shifting reduction and loss of allelopathic chemical controls produced by submerged (SAV) and emergent (EAV) aquatic vegetation (aka macrophytes) (Scheffer et al. 1993). It is well known that most species of aquatic vegetation produce allelopathic chemicals, many of which negatively affect cyanobacteria (Hilt and Gross 2008; Al-Shehri 2010; Jasser 1995; Korner and Nicklisch 2002; Santonja et al. 2018; and others). According to Gross et al. (2007): “We now have ample evidence that low phytoplankton and epiphyte densities in shallow eutrophic lakes might be the result of complex abiotic and biotic feedback mechanisms, and allelopathy is considered as one alternative mechanism (Scheffer et al. 1993).”

It has been well documented that phytoplankton (including cyanobacteria) exhibit different, species specific and strain specific sensitivities to macrophyte allelochemicals (Gross 2003; Hilt and Gross 2008; Korner and Nicklisch 2002; Mulderij et al. 2005; van Donk and van de Bund 2002; Al-Shehri 2010; Jasser 1995; Santonja et al. 2018). For

example, chlorophytes (green algae) appear to be less sensitive, whereas diatoms and cyanobacteria are often significantly inhibited by macrophyte allelochemicals (Hilt and Gross 2008; Al-Shehri 2010; Jasser 1995; Korner and Nicklisch 2002). Mulderij et al. (2005) reported a higher sensitivity of toxic than non-toxic *Microcystis aeruginosa* strains among cyanobacteria to aquatic vegetation allelochemicals. There is a general consensus among chemical ecologists that macrophyte derived allelochemicals can be strong drivers of phytoplankton communities (Santonja et al. 2018) and that invasive macrophytes can have differing effects on cyanobacteria than native species.

We have found no literature that evaluated allelochemicals produced by invasive *Phragmites* sp. on cyanobacteria, the dominant EAV along the shores of Utah Lake, although our review was only cursory. However, we have observed (and have limited unpublished data) that water clarity and zooplankton abundances are substantially greater in inundated *Phragmites* sp. stands in Provo Bay and other areas of Utah Lake compared with areas directly adjacent to the stands devoid of macrophytes and that wind disturbance does not appear to fully explain these differences. See the following YouTube video to see how water clarity is improved in *Phragmites* stands which allows zooplankton to prosper:

https://www.youtube.com/watch?v=orJ2_ILDb7I

The following video shows a green algal bloom on Provo Bay in June 2017 just outside of phragmites and cattail stands.

<https://www.youtube.com/watch?v=Jqc44-EvzRA>

Several more specific examples of allelochemicals effects on algae and cyanobacteria follow. *Myriophyllum* sp. (water milfoil) are known to produce polyphenols that play a major role in suppressing phytoplankton growth (Gross 2003b; Hilt 2006; Hilt et al. 2006; Bauer et al. 2009). *Myriophyllum spicatum*, a species of milfoil that grows in Utah, contains up to 30% polyphenols based on dry weight in apical meristems and exhibits a strong inhibitory action against various cyanobacteria and algae (Gross et al., 1996; Korner and Nicklisch, 2002; Bauer et al. 2009). Algicidal phenolic compounds produced by *Myriophyllum* sp. defensive use against phytoplankton has been shown in aquatic habitats and in situ experiments and may provide an advantage for submerged plants in the competition with phytoplankton for light and likely contribute to the stabilization of clear water states in macrophyte dominated shallow eutrophic lakes (Hilt and Gross, 2008; Bauer et al. 2009). Bauer et al. (2009) showed that crude extracts of apical meristems of *Myriophyllum* sp. always inhibited the growth of *Anabaena variabilis*, although the active chemical compounds were seasonally variable.

Santonja et al. (2018) found that allelopathic effects of aqueous leaf extracts of *Ludwigia hexapetala* (a South American species of water primrose) had strong effects on

the photosynthetic activity of three target phytoplankton strains of cyanobacteria; *Scenedesmus communis*, a toxic *Microcystis aeruginosa* strain and a non-toxic *Microcystis aeruginosa* strain. They identified three flavonoid glycosides myricitrin, prunin and quercitrin as the main secondary compounds and suggested that these chemicals could favor the photosynthetic activity of toxic cyanobacteria in spring and reduce their photosynthetic activity in summer, potentially leading to drastic changes in the phytoplankton communities.

SAV and EAV macrophyte dominated conditions in shallow eutrophic Utah Lake may have been the norm, pre-settlement mid-1800's. Based on the literature and our limited personal observations, we posit that macrophyte allelochemical defenses helped control cyanoHABs in the lake in the past but at present their role has diminished. We also submit that observations of early explorers that documented Utah Lake as having clear water were based on shoreline observations that likely were influenced by SAV and EAV effects on water turbidity, including allelopathy.

Any holistic scientifically based management actions taken to reduce cyanoHABs in the lake must address SAV and EAV allelopathy and incorporate this knowledge. For example, there is an ongoing phragmites removal program along the shores of Utah Lake via several intrusive control methods, even though to our knowledge no rigorous BACI based monitoring program has or is being conducted that measures ecosystem effects of phragmites removal, including allelopathic control of cyanoHABs.

Watershed-wide Pesticide Use Favors CyanoHABs

The effects of pesticides on the environment have been well documented since Rachel Carson's 1962 book *Silent Spring*, the environmental text that "changed the world" (Carson 2002). Since 1962, we have come to understand that targeted and non-targeted application of pesticides within a watershed can have severe negative direct and indirect effects, and short-term pulse or chronic low-level effects on aquatic ecosystems (Van Dijk et al. 2013, Carson et al. 1962, countless others). The effects of pesticides can range from effects on individuals to populations to communities to entire ecosystems and their food webs, including ecosystem functions (Van Dijk et al. 2013, Englert et al. 2012). Community-level effects can occur even at low levels of toxicants (Liess and Beketov 2011).

Pesticides can act individually or synergistically and there are often large phylogenetic differences in toxicity. For example, freshwater ostracods had 48-h LC₅₀s for a pesticide that were 50–120 times lower than that of *Daphnia magna* (Sánchez-Bayo and Goka, 2006b) and the midge, *Chironomus tentans* LC₅₀s were four orders of magnitude lower than those for *D. magna*.

In addition, the inert components used in a pesticide formulation can also be toxic to aquatic organisms (Dunkel and Richards 1998). Aquatic organisms are constantly being exposed to residues of chemicals present in water, from which they cannot escape. The

following figure (Figure 17) illustrates of modes of entry of pesticides into aquatic ecosystems.

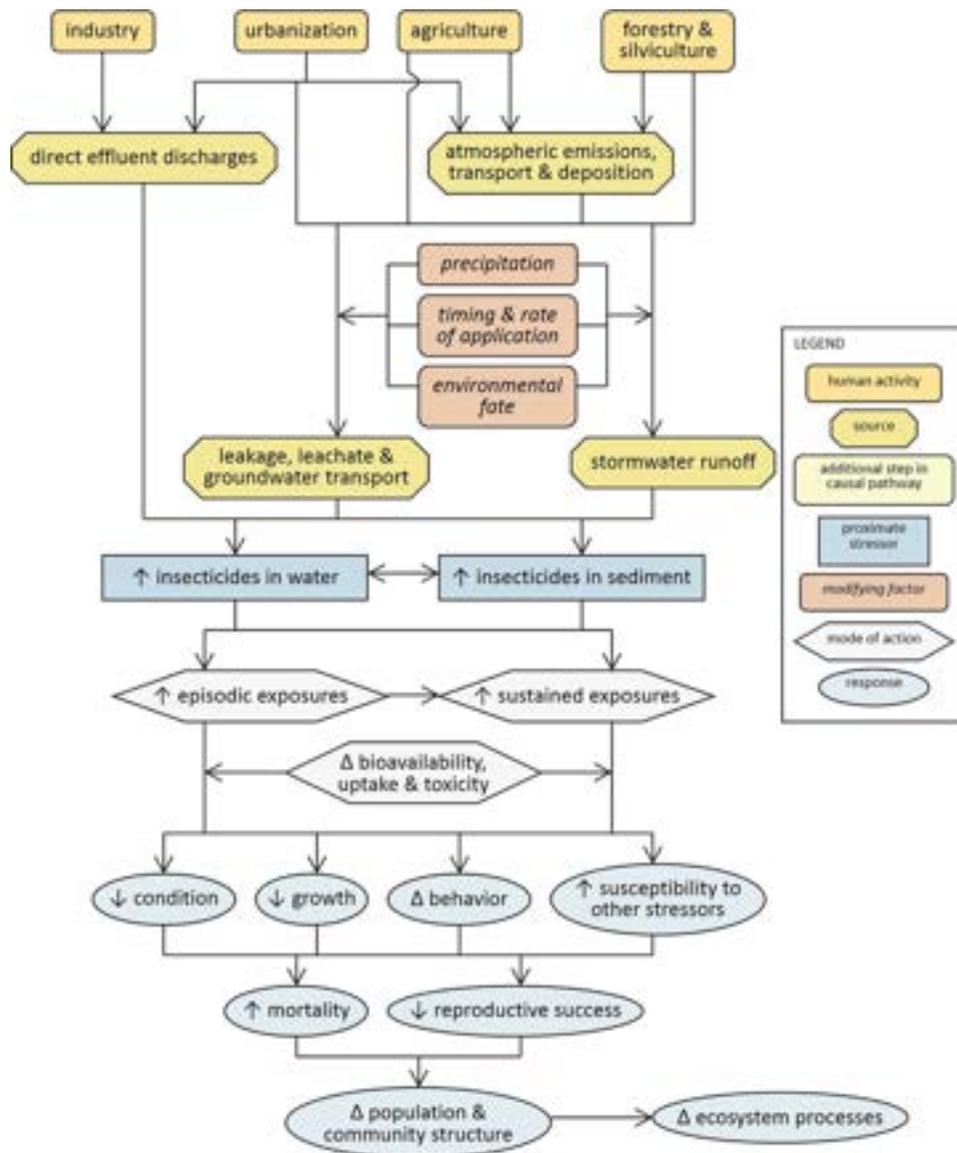


Figure 17. Conceptual model of mode of entry of pesticides into aquatic ecosystems from USEPA 2015.

Examples of the effects of pesticide use on aquatic organisms are voluminous. For example, long-term alterations in aquatic invertebrate community structure can occur after only a single pulse contamination of an aquatic ecosystem (Beketov et al. 2008, Van Dijk et al. 2013). Mohr et al. (2012) reported that negative effects on caddisflies occurred after a single pulse of the pesticide imidacloprid, and Diptera and Ephemeroptera larvae were affected after repeated pulses (Mohr et al. 2012, Van Dijk et al. 2013). This same pesticide caused zooplankton, benthic nekton and entire neuston communities to become

significantly less abundant than non-treated controls (Hayasaka et al. 2011, Van Dijk et al. 2013).

Cumulative effects can act synergistically with other chemicals. For instance, Chen et al (2009) reported that eight days' exposure to a mixture of the nonylphenol polyethoxylate, R-11 and imidacloprid resulted in the crustacean *Ceriodaphnia dubia* population size that was three times smaller than with R-11 alone, and 13 times smaller than with imidacloprid alone (Van Dijk et al. 2013). Cumulative effects with exposure time become more relevant when organisms are constantly exposed to low levels of many contaminants (Tennekes and Sanchez-Bayo 2011). Even the lowest single pesticide concentrations can have toxic effects if sustained over a long period. For example, the difference in LC₅₀ between only 2- and 5-day exposures was three orders of magnitude for freshwater ostracod *Cypridopsis vidua*, (Sánchez- Bayo, 2009). This trend toward lower LC₅₀s with increasing exposure time also occurs with other species including *Daphnia magna*, *Gammarus* amphipods, black fly larvae, alderflies, mayfly and dragonfly nymphs (Beketov and Liess, 2008a; Roessink et al., 2013, Van Dijk et al. 2013). Kuncce et al. (2015) showed that survival of midge larvae (*Chironomus riparius*) was reduced in waters contaminated with mixtures of neonicotinoids (imidacloprid, thiacloprid) and pyrethroid insecticides (deltamethrin and esfenvalerate). Cumulative and synergistic effects do not always occur rapidly but may result in what is known as “delayed mortality” (Beketov and Liess, 2008a), which can occur many days, weeks, or even months after exposure.

Sublethal effects of pesticides also occurs and can reverberate throughout the food web (Desneux et al 2007, Van Dijk et al. 2013). Sublethal effects can affect neurophysiology, larval development, molting, adult longevity, immunology, fecundity, sex ratio, mobility, orientation, feeding behavior, and oviposition behavior (Desneux et al. 2007, Van Dijk et al. 2013). All of these sub-lethal effects have the potential to produce population level and community level impacts on ecosystems (Van Dijk et al. 2013).

Direct and indirect negative effects on individual species affects communities, food webs, and ecosystem function (Van Dijk et al. 2013). For example, direct negative effects on zooplankton species within the community can have indirect effects on growth suppression in fish feeding on the zooplankton species. Colombo et al. (2013) showed that the disappearance of chironomid larvae from pesticide use brought about increases in *Radix* sp. snails. Pestana et al. (2009) documented that the overall biodiversity of aquatic communities is negatively affected by pesticide use.

Many aquatic populations can be decimated by pesticides and recovery rates can be either slow or, if there is competition with other species, does not take place at all (Liess et al. 2013, Van Dijk et al. 2013). This is particularly true for univoltine or semivoltine species (Liess et al. 2013, Van Dijk et al. 2013). In addition, the structure of aquatic communities does not always revert to the original condition (i.e. hysteresis, resilience)

because some species disappear while others increase in numbers (Beketov et al., 2008; Hayasaka et al., 2012a).

Hundreds of studies have shown that algae are particularly vulnerable to target and non-target herbicides. For example, several studies have shown that algae are particularly vulnerable to glyphosate-based herbicides because of their physiological and biochemical similarity with terrestrial plants (Annett, Habibi and Hontela, 2014; Tsui and Chu, 2003). Sáenz et al. (1997) showed that growth of green algae (*Scenedesmus acutus*, *S. quadricauda*) was inhibited by glyphosates and Tsui and Chu (2003) showed that the diatom species, *Skeletonema costatum* was more affected than green algae. Increased temperatures can also significantly reduce algal diversity exposed to herbicides. Obviously, benthic algae are also negatively affected by herbicides.

Cyanobacteria are much less susceptible to herbicides than are algae (diatoms and greens), which can result in their strong competitive advantage over other phytoplankton in Utah Lake. This competitive advantage combined with; 1) low light competitive advantage (see section Turbidity Favors CyanoHABs), 2) little or no lake water flushing competitive advantage (see section Flow Rates Favor CyanoHABs), 3) negative effects of pesticides on zooplankton grazers (this section), and 4) extinction prone populations and communities due to isolation (see following section Metapopulation and Metacommunity Dynamics: Isolation Favors CyanoHABs) allows cyanobacteria to dominate the Utah Lake ecosystem. Subsequently, cyanoHABs go virtually unchecked by other remaining compromised members of the lake’s aquatic community.

Utah County has a long history of pesticide use and is one of the leading pesticide applicators in Utah with an extensive list of pesticides used (Eisele et al. 1989). Pesticides have been heavily applied near Utah Lake for decades (including DDT), particularly on its southern end with intensive agricultural land use. Tributaries that transport pesticides to Utah Lake from agricultural southern lands include Beer Creek, Spanish Fork, Currant Creek, etc. Forested lands and silvicultural lands within the watershed are also sources of pesticides to the lake, as well as unmonitored urban industrial and household sources and atmospheric drift. A very short list compiled by the author in a 1998 publication of several pesticides that have been used in agricultural and forested land and their effects on macroinvertebrates is in

Table 2.

Table 2. Effects of several pesticides on aquatic macroinvertebrates frequently applied to agricultural and forested areas. From Dunkel and Richards 1998.

Pesticide Name	Type	Aquatic invertebrate tested	Lethal Concentration	Duration	Citation
2,4-D	Systemic Pyridine herbicide	Amphipods	LC ₅₀ 2.1-6.8 ppm	24h	1
Picloram	Systemic Pyridine herbicide	<i>Gammarus lacustris</i> (Amphipod)	LC ₅₀ 2.1-6.8 ppm	24h	2

Fenvalerate	Synthetic pyrethroid	<i>Gammarus lacustris</i> (Amphipod)	LC ₅₀ 0.00003 ppm to 0.000093 ppm	96h and 24h	3
Diflubenzuron	Growth regulator	<i>Skwala</i> sp. (Plecoptera)	LC ₅₀ 57.5 ppm	96h	4
Malathion	Organophosphate	Various macroinvertebrates	LC ₅₀ 0.000069 to 3.0 ppm	96h	5
Carbaryl	Carbamate	Various macroinvertebrates	LC ₅₀ 0.0048 to 0.28 ppm	96h	6
Azinphosmethyl	Organophosphate	Various macroinvertebrates	LC ₅₀ 0.00015 to 0.021 ppm	96h	7
Glyphosate R		amphipod	LC ₅₀ 43 ppm	96h	8
Flucythrinate	Synthetic pyrethroid	amphipod	LC ₅₀ 0.00022 ppm	96h	9
Flucythrinate	Synthetic pyrethroid	<i>Brachycentrus americanus</i> (Trichoptera)	40 to 80% immobilization	Almost immediately	10
Flucythrinate	Synthetic pyrethroid	<i>Pteronarys dorsata</i> (Plecoptera)	LC ₅₀ > 0.0000007 mg/liter	12d	11

^{1,4,5,6,7,8}Meehan 1991; ^{2,3} Anderson 1982, Smith and Stratton 1986; ^{9,10, 11} Anderson and Shubat 1984

There is ongoing use and misuse of pesticides to control mosquitos and phragmites directly adjacent to the shores of Utah Lake. Several dozen pest control businesses advertise aquatic insect control using pesticides, including at least one company that promotes controlling midges and mayflies (D.C. Richards, personal observation, see <https://www.orkin.com/other/mayflies> for a leading pest control business view on mayflies as pests). These applications certainly have a negative indirect on non-target organisms including midges and green algae, two important regulators of cyanoHABs in Utah Lake. As documented in this report and other reports (Richards and Miller 2019b, 2019c, 2017, Richards 2019a, 2019b, 2018, 2018b, and 2016), zooplankton are the top down regulators of phytoplankton and midge larvae are the dominant benthic ecosystem engineers in Utah Lake. Both these groups, as well as algae (excluding cyanobacteria) are highly susceptible to pesticides entering the lake and any negative effects restrict their important contributions to Utah Lake’s food web, maintaining its ecosystem function, resistance and resilience, and as natural controls of cyanoHABs.

Metapopulation and Metacommunity Dynamics: Isolation Favors CyanoHABs

Ecological integrity as noted earlier, is the sum of physical, chemical, and biological integrity (Karr 1993, 1996). Biological integrity includes the full range of elements that specifically include metapopulation and metacommunity processes (Angermeier and Karr, 1994; Frey, 1975; Karr and Dudley, 1981; Karr et al., 1986).

Metapopulations and metacommunities are not completely isolated nor completely connected with other populations or communities but are those that ‘blink in and out of extinction’ due to limited dispersal and connectivity. They rely on rescue from nearby populations and communities for their survival. Therefore, limited amounts of dispersal and connectivity decrease the extinction risk of isolation. In addition, if environmental

stochasticity is not completely correlated between metapopulations and metacommunities extinction risk is reduced.

Dispersal, connectivity, and the level of demographic and environmental stochasticity correlation determines metapopulation and metacommunity viability (Altermatt 2011, Oliver et al. 2013). Isolated populations and communities on the other hand, are those that have no dispersal and connectivity to other populations or communities resulting in much higher extinction probabilities either through demographic or environmental stochasticity (Hanski 1999, MacArthur and Wilson 1967, Fagan et al. 2002, Strayer 2008).

Metapopulation viability is also determined by the relationship between suitable and unsuitable habitat. Suitable habitat can be occupied or unoccupied by populations and communities, likewise unsuitable habitat may be occupied or unoccupied. The proportion of suitable habitat that is occupied is a major driver in population and community viability. Suitable habitat may be unoccupied solely due to lack of dispersal and connectivity from other populations or communities (Strayer 2008). For example, water quality conditions that became unsuitable in the recent past from human activities may now be or may become suitable in the future but if there no dispersal and connectivity between populations and communities these suitable habitats will remained unoccupied. Habitat connectivity is key to ecosystem resistance and long-term resilience, as is biodiversity (Oliver et al. 2015).

It is likely that many extant populations once occupied most of the suitable habitats in Utah Lake prior to settlement but now have near- zero occupancy rates due to loss of connectivity and dispersal from other systems. Loss of habitat also increases dispersal distances between populations, which causes a loss in the proportion of unoccupied suitable habitat and therefore increases the extinction debt (Strayer 2008). Loss of dispersal ability and loss of suitable habitat are not additive but multiplicative (Strayer 2008). For example, if only 30% of suitable habitat remains and only 60% remains occupied due to reduced migration for example, then only 18% of the previously suitable habitats that were once occupied are now unoccupied.

Some of the native specie's populations in Utah Lake may be at such critically low densities that they may also have entered what is known as the 'extinction vortex' (Gilpen and Soule 1986), where in addition to the factors just described; genetic factors such as inbreeding depression, genetic drift, and 'mutational meltdown' (Lynch and Burger 1993) and demographic and environmental stochasticity combine in positive feedback loops that accelerate their extinction probabilities (Lynch et al. 1993, and Lynch and Gabriel 1990, Mock et al. 2004, Fagen and Holmes 2006).

Utah Lake for the most part can be considered to be isolated from other suitable habitats; thus, its populations and communities are isolated. As a remnant of Lake Bonneville, Utah Lake's biological community evolved in relative isolation from other lakes its size and has gradually adapted and evolved during the environmental and

ecological transition from Lake Bonneville to Utah Lake. The only slightly similar large water body in proximity and connectivity to Utah Lake is downstream Farmington Bay of Great Salt Lake, a recently transitioned saltwater water body to fresh/brackish water body. Any aquatic dispersal and rescue of populations and communities in Utah Lake from Farmington Bay would have to be upstream through the Jordan River, which is often not physically connected to Utah Lake due to a dam and low lake water levels or via aerial dispersal of adult insects, although winds are typically from the south in summer when aerial insect adults are present. Because Utah Lake is now a lentic warm water system during summer, very few upstream connections exist because most upstream communities are lotic, cold water adapted. Consequently, population and community extinction risks are highly elevated in the lake.

As one example, Utah Lake's fish assemblage is now dominated by invasive species. Its benthic invertebrates and zooplankton have evolved with the community level effects of these invasive species without the support of rescue populations. Given all of the other stresses imposed on the lake, Utah Lake is now a poor analog of its former self. Utah Lake's simplified, depauperate, isolated, analog biological community cannot function properly enough to self-control cyanoHABs and biological rescue efforts do not appear to be forthcoming.

Urbanization: Most Recent and Ongoing Catastrophic Shift Favors CyanoHABs

Utah is experiencing a burgeoning exponential human population growth; Utah County is ground zero (Coon 2017). Utah's population was recorded at 11,380 in 1850, ten years later it was 40,273, a 250% increase and by 1900 it had grown to 276,749. Utah's population is roughly 3.22 million as of 2019 and by 2060 it is projected to have 6.84 million people (World Population Review 2019a). Utah County's human population was 2,206 in 1850. Today it is 606,503, a 27,500% increase and rapidly growing (Figure 18) (World Population Review 2019b).

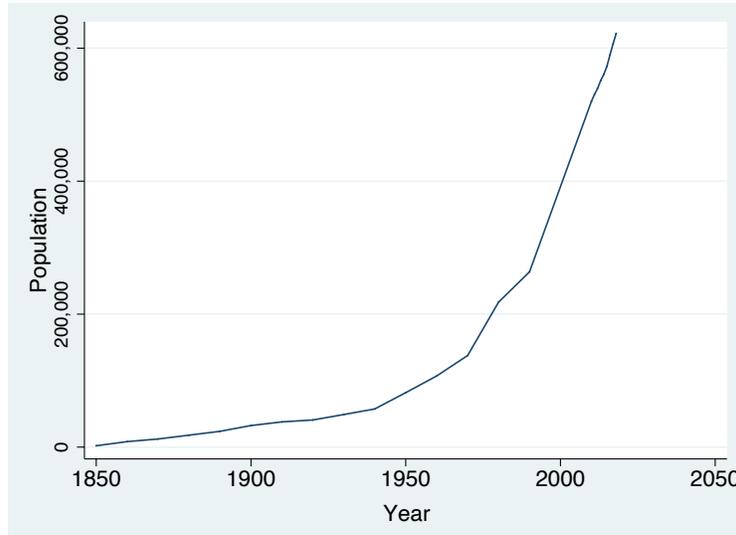


Figure 18. Exponential human population growth since Mormon settlement in 1850s in Utah County, UT (data extracted from World Population Review 2019b).

The following is a YouTube link to a video of the unprecedented urban development occurring along the shores of Utah Lake at Vineyard, UT. This area had dozens of spring feed groundwater tributaries feeding the lake whose fate is now unknown:

<https://youtu.be/88CWhlhBc-8>

The last remaining old growth cottonwood ‘parks’ are also being cleared to make room for subdivisions (Figure 19).

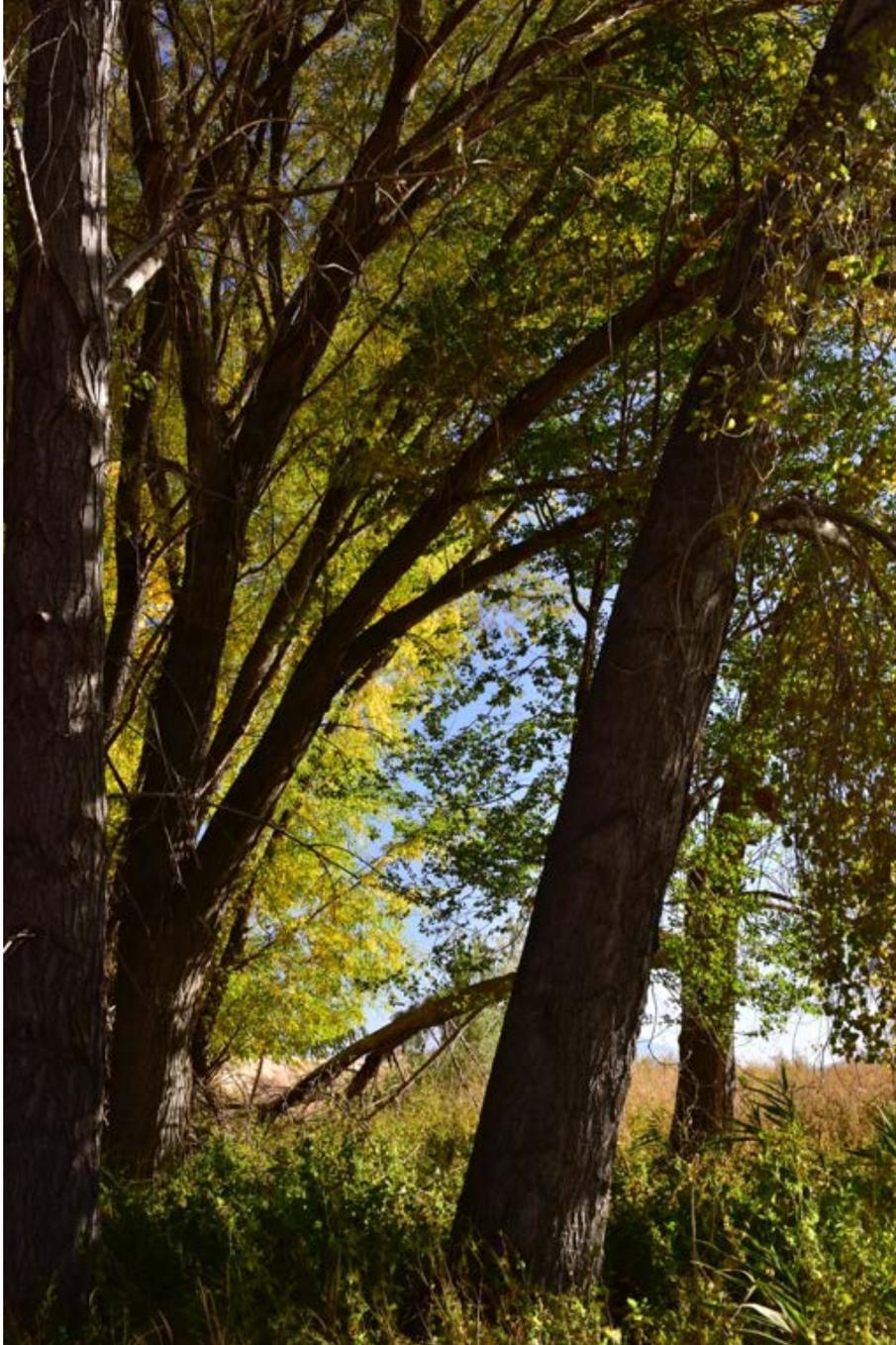


Figure 19. One of last remaining native Fremont cottonwood groves along the shores of Utah Lake, near Vineyard, UT. Acres of these groves have been recently removed to make room for housing and industrial development to support Utah County's burgeoning human population. These riparian forests were instrumental in maintaining Utah Lake's health and integrity and their role is now likely insignificant and they can be considered relict or ghost species and ecosystems due to compounding extinction debt.

Freshwater springs along the shores of Utah Lake and their wetlands are now being lost to urbanization. The following YouTube video documents one of the few remaining

springs that has been polluted by urbanization but may still have some remaining health, resistance, and resilience.

<https://youtu.be/7PZe-HQFzml>

This spring has been further polluted after increased housing development since the video was recorded.

Utah is the second driest state in the U.S., consequently water demands are high, particularly in the metropolitan areas surrounding Salt Lake City and Utah County. With projected population growth estimated to be in the millions in the next few decades; water demands are only expected to get higher and perhaps unsustainable. Urbanization, industrialization, housing development, and agriculture are all putting unprecedented pressure on water resources on Utah Lake's watershed. In addition, unregulated and unenforced pollutants and pollution are entering its waters via urban runoff, agriculture, household, and industrial wastes.

Unmeasured amounts of oil based industrial chemicals enter the lake via non-point urban surface and groundwater runoff during and after housing, business, and industrial construction. Pesticides enter via non-point agricultural runoff from increased agricultural production (see Pesticide section). Pharmaceutical chemicals mostly enter the lake via POTW effluent.

Pharmaceuticals: An underestimated disruptor of Utah Lake's ecosystem.

It is estimated that aquatic macroinvertebrates can have at least sixty-one kinds of multi-class pharmaceuticals in their tissues (Richmond 2018, Rodríguez-Mozaz 2015). *The bugs are on drugs, and so are the fish.*

According to Richmond et al. (2015):

“Although we do not yet understand the direct and indirect effects of these compounds, either singly or in complex mixtures, on fish and wildlife, a growing body of research demonstrates that many pharmaceuticals disrupt ecological interactions, functions and communities (Richmond et al. 2017). For example, amphetamines and antidepressants in stream water can disrupt the timing of emergence of aquatic insects (Lee et al. 2016, Richmond et al. 2016); psycholeptics such as Valium™ and the illicit drugs amphetamine and LSD can compromise the web-building ability of spiders (Witt 1971); and fish behaviour is altered when consuming prey contaminated with an antidepressant (Brodin et al. 2014). While pharmaceuticals have been detected in trace amounts (part per trillion concentrations) in surface waters around the world for over two decades, it was widely assumed that these concentrations pose little risk to the aquatic biota because environmental concentrations are usually well below lethal concentrations, and many of these compounds degrade rapidly in the environment, suggesting low risk of persistence and biomagnification potential (reviewed by Richmond et al. 2017). In light of these assumptions, it was surprising that we detected such a diverse suite of pharmaceuticals in aquatic invertebrates and at such

high concentrations (part per million concentrations of total pharmaceuticals in invertebrates at the most contaminated sites).”

It’s fairly common knowledge that Utah is one of the top states among prescription drug use and abuse in the U.S (<https://turningpointcenters.com/2012/10/29/prevalence-of-prescription-drug-abuse-in-utah/>, <https://www.recoveryconnection.com/substance-abuse-statistics-by-state/utah/>). Although unmeasured, it should come as no surprise that many of these chemicals are entering Utah Lake’s food web and like other waterbodies throughout the world are having a negative effect on ecosystems. It is apparent that Utah Lake’s ecosystem in its present state cannot hope to cope with urbanization and all of the other stressors discussed throughout this introduction and that most likely, Utah Lake has lost its ability to self-regulate cyanoHABs, and its resilience to future perturbations weakened.

Wetlands and Phragmites

Utah Lake wetlands are an integral part of the healthy functioning of the lake. They provide invaluable ecosystem services including improving water quality, however most of these wetlands have been lost to development and are continuing to be lost. As is with the rest of Utah Lake’s ecosystem, wetlands health has been compromised. We have not quantitatively evaluated any wetland data for inclusion in this progress report but provide a brief anecdotal discussion on our observations focusing on the invasive strain of the common reed, *Phragmites australis*.

The invasive subspecies (or strain depending on which authority cited) of the common reed, *Phragmites australis australis* plays a critical role in Utah Lake’s wetland ecology. Although, it is thought by many to be a devastating ecological threat, as it is now the dominant analog wetland emergent plant species; Phragmites provides important ecosystem services that native species are currently challenged to do including; shoreline protection from wind and waves that are all too common on Utah Lake, shelter and habitat for zooplankton and juvenile fishes, reduced turbidity, and switching algal assemblages from water column phytoplankton to sediment surface periphyton, etc. Utah Lake historically supported vast stands of native Phragmites, and every effort should be made to reintroduce and promote their recovery.

Native phragmites, bulrush, cattails (*Typha* sp.), and other species gave way to invasive phragmites and Salt Cedar (*Tamarix* sp.), etc. in Utah Lake’s wetlands over the last 150 years or so. Recently, there is an intensive phragmites management control/removal effort ongoing along the shores of Utah Lake. However, as far as we know, no formal BACI type quantitative monitoring or evaluation of the effects of phragmites removal on the wetland’s ecological functioning, food web or role in regulating cyanoHABs has occurred or is planned. Without such monitoring there is no scientific possibility of evaluating ecological benefits or impacts of phragmites removal.

We have observed what we conclude are important ecological benefits of phragmites stands in Utah Lake some of which were briefly discussed in this report. The following two videos demonstrate how phragmites can protect shorelines from wave action on the lake:

<https://youtu.be/gUnh2QnMOSo>

<https://youtu.be/ihI0gj1A8s8>

Appendix 5 is a nine-page opinion report that Dr. Richards was asked to provide to members of the USDA, APHIS Technical Advisory Group for Field Release for Biological Control Agents of Phragmites, early 2019. Dr. Richards has past scientific experience developing and using biocontrols on weeds and field experience observing phragmites ecology and habitat on Utah Lake. The opinion report is titled, “A Case for Adaptive Holistic Management of Phragmites in the Jordan River Drainage and Southern Utah”.

Obviously, much more research is needed to evaluate the importance of remaining wetlands and their plant communities to the healthy functioning of Utah Lake ecosystem, including services provided by phragmites.

Water Column Regulators and the Ecosystem Engineers

Water column and benthic sediments in shallow lakes, unlike deep lakes, are in close proximity, intimately linked, and do not function in isolation from each other (Scheffer 1998). However, water column and benthic environs operate under different sets of ecological conditions and guidelines. For example, water column nutrient cycling interacts directly with atmospheric deposition, whereas nutrient cycling within the benthos is dependent on nutrient obtained from the sediments and nutrients ‘falling’ out of the water column. Benthos are also more likely to experience anoxic conditions more frequently than the water column. Biotic assemblages also differ between these habitats and conditions.

Zooplankton and pelagic fishes are the most important top down regulators of nutrients and phytoplankton (including cyanoHABs) within the water column. Grazing by zooplankton on phytoplankton and predation of fishes on zooplankton recycle nutrients within the water column, often within a matter of minutes (Sailey et al. 2015). Zooplankton assemblages in Utah Lake are spatially and temporally heterogenous and exhibit a somewhat predictive seasonal succession that is directly and indirectly related to phytoplankton dynamics (see all Richards reports in Literature Cited section, this report, and Sailey et al. 2015). Zooplankton assemblages in the lake are also apparently strongly influenced by planktivorous fishes, predominately invasive species such as carp and the

juvenile stages of most other species. Predation on zooplankton likely has reduced their ability to regulate phytoplankton assemblages in Utah Lake (Richards and Miller 2019b, Richards 2019b, Sondergaard et al. 2008; Wetzel 2001; Cole and Weihe 2016; Havens et al. 2015a, 2015b; Gophen 1990; Cooke et al. 2016).

Ecosystem engineers are species that have disproportionate influences on their ecosystems (Paine 1966). A single species of ecosystem engineer can alter the physical environment, which can have cascading consequences for the whole system (Jones et al. 1994, Bertness and Leonard 1997, Wright and Jones 2006). They also have other direct and indirect effects within a community through competition, herbivory, predation, etc. (Jones et al. 1997, Bertness et al. 1999, Hastings et al. 2007, Largaespada et al. 2012, Phillips et al. 2019). Three benthic ecosystem engineers have had a disproportionate influence on the ecology of Utah Lake and have transitioned in importance over recent times due to human induced impacts: mollusks- to chironomids (midges)- to invasive fishes, particularly carp and catfish (see Richards reports in Literature Cited section, and this report). More details of ecosystem effects of these engineers are presented in following sections.

Water column regulators and nutrient cycling

Nutrient cycling and recycling occur rapidly in the water column portion of the food web consisting of phytoplankton, zooplankton, and fishes (Sailley et al. 2015). Nutrients also enter the water column via atmospheric deposition, wastewater treatment facilities, surface water, groundwater, and sediments. Because Utah Lake has a large surface area, is shallow, has frequent local inversions (Figure 20), and strong wind storms that deliver nutrients to the lake from surrounding nutrient rich soils; the amounts and effects of atmospheric nutrient deposition can be substantial and are presently being quantified by researchers at Wasatch Front Water Quality Council. The shallow nature and high amounts of background nutrients in the lakebed results in a substantial and important influx of nutrients from benthic sediments into the water column (Scheffer 1998, Hogsett and Goel 2013, Hogsett et al. 2019). In fact, Hogsett et al. (2019) reported that sediment phosphorous loading in Utah Lake was 5 time higher than external loading and was estimated at 1500 tons year⁻¹. For the most part, nutrient limitation does not seem to be a problem in Utah Lake due to continuous release from sediments, except perhaps during summer cyanoHABs (Hogsett et al. 2019).



Figure 20. Common inversion layer over Provo Bay and southern Utah Lake. High concentrations of nutrient and other pollutants can enter the surface water of the lake during these times. Atmospheric deposition of nutrients into Utah Lake are presently being quantified by WFWQC scientists.

Water Column Regulators: Zooplankton and Invasive Fishes

Zooplankton

Zooplankton grazers are the number one water column regulator of phytoplankton, including cyanoHABs (Iglesias et al. 2007, Scheffer 1998). Zooplankton frequently move between habitats including daily horizontal migration; subsequently zooplankton are a vital linkage between the pelagic, benthic and littoral zones (Vander Zanden and Vadeboncoeur 2002, Jones and Waldron, 2003). However, very little is known about the spatial and temporal patterns of zooplankton in Utah Lake despite their critically important relationships with: nutrient cycling, phytoplankton (including cyanoHABs) ecology and population dynamics, and other food web components in the lake (Richards and Miller 2017, Richards 2018).

Zooplankton obviously have top down grazing effects on phytoplankton and cyanobacteria and in turn are affected by these (bottom up effects) (Iglesias et al. 2007). Zooplankton also have different modes of feeding including grazing and predation, some of which prey upon other zooplankton. Most zooplankton are selective feeders. All of these complex interactions directly and indirectly influence nutrient cycling in the water

column. Zooplankton excretion and respiration of nitrogen, phosphorus, and ammonia is immediately available and consumed by phytoplankton, often within minutes. This phytoplankton-zooplankton component of water column nutrient cycling has been well documented and known by limnologists and ecologists for several decades and is likely an important driver of cyanobacteria blooms in Utah Lake (Iglesias et al. 2007, Scheffer 1998).

Medium- and large-sized cladocerans, typically *Daphnia* spp. can markedly reduce phytoplankton biomass (Jeppesen et al. 1990, Scheffer 1998), even in communities dominated by cyanobacteria (Jeppesen et al. 2003, Lampert et al. 1986, Brooks and Dodson 1965, Gorokhova and Engstrom-Ost 2009, and Hogfors et al. 2014). *Daphnia* spp. have the ability to feed on bacteria, protozoa, phytoplankton and even some small zooplankton, highlighting their important role in freshwater food webs (Yin et al. 2010). It has been demonstrated that intensive zooplankton grazing can promote a clear-water state (Scheffer 1998). For example, grazing by *Daphnia* sp. has been reported to be responsible for spring clearing in temperate lakes (Meijer et al. 1999).

Phytoplankton assemblages can have a bottom up control on zooplankton assemblages via several mechanisms, including relative abundance, digestibility, nutrient content, etc. Conversely, zooplankton assemblages can have a top down control on phytoplankton assemblages via selective and non-selective grazing and contrary to past assumptions, it has become apparent that zooplankton routinely and selectively rely on cyanoHABs in their diets. Consequently, zooplankton assemblages can shift phytoplankton assemblages toward better adapted cyanobacteria consumer species (Motwani et al. 2017, Woodland et al. 2013, Koski et al. 2002, Vehmaa et al. 2013, Gorokhova and Engstrom-Ost 2009, Hogfors et al. 2014, Ger et al. 2016).

Utah Lake appears to support a diverse zooplankton assemblage that varies spatially and temporally (Richards and Miller 2017). There are >> 20 zooplankton taxa occurring in Utah Lake including; cladocera, copepod, and rotifer taxa from several functional groups, each with different life history and feeding strategies (Richards and Miller 2017, Richards 2019, Marshall 2019, and unpublished data). The taxonomy of Utah Lake's zooplankton has never been fully documented and verified. Because of this gap, zooplankton taxonomy is under revision by OreoHelix Ecological and River Continuum Concepts, Manhattan, MT (see Appendix 4). It is of utmost importance to correctly identify zooplankton taxa in the lake.

Unfortunately, zooplankton assemblages in Utah Lake have also undergone bottlenecks and assemblage shifts, including those stressors discussed in the previous sections that have resulted in Utah Lake's zooplankton assemblages becoming analogs of past natural assemblages and that may no longer be able to regulate cyanoHABs. One of the most important factors not discussed so far has been and continues to be predation on zooplankton by planktivorous invasive fish and how this affects cyanoHABs.

Invasive Fish Predation on Zooplankton Favors CyanoHABs

Planktivorous fish predation has strong deleterious effects on zooplankton prey. Planktivory also negatively affects entire zooplankton assemblages and often initiates trophic cascades throughout the food web (Carpenter and Kitchell, 1996, Scheffer and Jeppesen, 1998; Jeppesen et al., 1998; Moss et al., 1998, Iglesias et al. 2007). This can be especially catastrophic if planktivorous fish are invasive and the native zooplankton assemblages haven't evolved with invaders.

Planktivory is thought to be the main factor controlling the spatial distribution, abundance and body size of zooplankton in shallow lakes (e.g. Scheffer, 1998; Burks et al., 2002, Iglesias et al. 2007) and often induces major shifts in the size distribution of zooplankton (Hrbáček et al., 1961; Brooks and Dodson, 1965) or behavioural shifts (Timms and Moss, 1984; Schriver et al., 1995; Lauridsen and Lodge, 1996; Burks et al., 2002; Romare and Hansson, 2003). For example, in Lake Blanca, Uruguay, the small size of the dominant cladocerans and the dominance by copepods and rotifers likely reflect the extremely high abundance of planktivorous fish predators (Iglesias et al. 2007). The effect of planktivory on decreased zooplankton size can increase the likelihood of cyanoHABs. This is because larger sized zooplankton are often better at feeding on larger strands of algal particularly cyanobacteria (Carpenter and Kitchell 1988, Caroni 2010, Jeppesen et al. 2011, Attayde and Bozelli 1998, Carpenter et al. 1985, Jeppesen et al 2000, Jeppesen et al 2003, Lamper et al 1986, Gannon and Stemberger 1978, others). Sarnelle (2007) also reported that high abundances of generalist grazers (i.e., *Daphnia*) may control blooms when released from planktivorous fish predation (Ger et al. 2016).

We have conducted preliminary analysis of zooplankton body sizes in Utah Lake and found that they are substantially smaller than expected, indicating that invasive planktivores have altered the zooplankton assemblages in the lake, which could be contributing to cyanoHABs. See Appendix 3: *Spatial and Temporal Variability of Zooplankton Body Lengths in Utah Lake, Technical Memo* for results of this analysis. We are also incorporating zooplankton body size metrics into our Multimeric Index of Biological Integrity for Utah Lake (Appendix 2) because of its importance to the lake's ecological health.

Zooplankton survival often depends on heterogenous habitat to avoid planktivores. Consequently, zooplankton assemblages and abundances often differ between littoral submerged and emergent aquatic vegetated habitat and open water habitat; planktivory will have different effects depending on type of habitat. Almost all small juvenile fish in Utah Lake are planktivores and tend to seek refuge in aquatic vegetation from larger piscivorous fish, subsequently increasing planktivory. However, aquatic vegetation is also a refuge for zooplankton. Aquatic vegetation also decreases turbidity and improves clarity and visibility, either through decreased sediment turbulence or phytoplankton allelopathy or both. Clearer water in vegetated habitat also has less phytoplankton abundance food resources for zooplankton, either through increased grazing by

zooplankton or allelopathy or both. In Utah Lake, open water habitat is defined by turbidity and can provide zooplankton cover from visual planktivores and an abundance of phytoplankton food resource. Tradeoffs are inevitable. However, given the very high seasonal abundance of juvenile planktivorous fishes, no habitat may provide significant refuge for zooplankton in the lake at those times (Iglesias et al. 2007). It has been our observation that the most devastating impact of non-native juvenile fish planktivores occurs during clear water conditions in autumn in shallow habitats where phragmites and other aquatic vegetations have been physically removed and juvenile fishes are schooling. See video:

https://youtu.be/5qYfa4_a_lk

Fish planktivory on zooplankton obviously occurred with native fishes (i.e. June Sucker) in Utah Lake in the past. However, in the past, mussels and clams were likely another dominant predator (via filtration) of phytoplankton, particularly during times when zooplankton abundances were reduced by fish planktivory and these bivalves likely helped control cyanoHABs (see following section). Bivalves also eat smaller sized zooplankton which results in an average larger size zooplankton assemblage, which in turn eat more phytoplankton especially the larger size phytoplankton (e.g. cyanoHABs). (Marroni et al. 2016, Caraco et al. 1997, Prins and Escaravage 2005, Newell et al. 2007).

Invasive common carp (*Cyprinus carpio*) are a major disruptor of Utah Lake's ecosystem including their impacts on zooplankton. Meijer et al. (1990), Khan (2003), and Britton et al. (2007) reported that up to 25% of the biomass ingested by carp can consist of zooplankton in other lakes. Carp can therefore affect the zooplankton assemblages in many ways, by direct predation (Miller and Crowl, 2006), by consuming macroinvertebrates that themselves are zooplankton predators (Khan, 2003), through loss of macrophytes that provide shelter, and by increasing phytoplankton biomass and promoting cyanobacterial blooms (Parkos et al., 2003) (Raposeiro et al. 2017). Furthermore, resuspension of sediment particles can interfere with the filtering apparatus of cladocerans (Kirk and Gilbert, 1990), and bioturbation may also affect zooplankton dormant stages in sediments, negatively effecting emergence patterns (Angeler et al., 2002) (Florian et al. 2016).

As we have reported in Richards and Miller (2019b) (see Appendix 2), many studies have shown that removal or reduction of planktivorous fish populations could be used to enhance zooplankton grazing on algae, including reduction of cyanoHABs and thereby creating a clearwater state (Gulati, 1990; Hansson et al., 1998a; Søndergaard et al., 2007, 2008). This has led many researchers and managers to recommend biomanipulation as a relatively inexpensive remedy for controlling algal blooms compared to attempts at whole drainage nutrient control (Riedel-Lehrke 1997; Cooke 1986; Jeppesen et al. 2007; Richards 2019a).

Transition of Benthic Ecosystem Engineers: Bivalves to Midges to Invasive Fishes

Each benthic taxon in Utah Lake uses different sediment engineering methods according to their evolutionary resume. There is no redundancy, and each is critically unique. However, invasive fish benthic engineers are not subtle executing their methods and do more harm than good.

Mollusks

Freshwater mollusks include bivalves (mussels and clams) and gastropods (snails). Their diversity and abundances in the depauperate western USA peak in the Utah Lake-Jordan River drainage and the surrounding area (Richards 2017, Richards 2014). Utah Lake's mollusk diversity and abundances are due to a Lake Bonneville heritage, abundant nutrients, high pH and high CaCO₃ levels originating from the thick limestone base rock in the watershed which they require (see Introduction and Background: *Lake Bonneville: A Lake No More?*).

Native mollusks were the dominant benthic ecosystem engineers in Utah Lake when early explorers and Mormon settlers first arrived in the 1800s. Native mollusks were also responsible for much of the water column functioning (Richards 2014, 2017, 2018b, and 2019a). Unfortunately, their role as keystone species and ecosystem engineers has been eliminated.

Historically, there were two species of mussels native to Utah Lake, the now 'critically imperiled' *Margaritifera falcata*, common name Western Pearlshell (Figure 21) and the 'imperiled' *Anodonta californiensis/nutalliana*, common name Floater (Figure 22). Both species are extinct in the lake, however small, isolated, remnant, imperiled populations of *A. californiensis/nutalliana* still occur in upstream creeks (Richards 2017). Highly weathered shells of *A. californiensis/nutalliana* can still be found along Utah Lake shores (Figure 25) but no shells of *M. falcata* have been found in more than one hundred years. Both species were abundant and large enough sized to be important food items to Native Americans and early settlers. Native mussels were an important part of Utah Lake's fisheries, although *M. falcata* was usually not eaten (Chamberlain and Jones 1929, Janetski 1990). Native mussel and other bivalve and snail shells from the region were highly prized and traded throughout the west. Thus, Utah Lake's native mollusk fisheries¹¹ were a critical component of Native American economies and an important part of the early settler economy. Utah Department of

¹¹ In fisheries – the term fish is used as a collective term, and includes **mollusks**, crustaceans and any aquatic animal which is harvested. Fishery is defined as: The industry or occupation devoted to the catching, processing, or selling of fish, **shellfish**, or other aquatic animals. Shellfish include mussels, clams, and snails.

Natural Resources (2007) reported that Utah Lake likely had the largest population of *A. californiensis/nutalliana* in the entire state but have now been extirpated from the lake. One likely reason for their disappearance from Utah Lake was predation by the apex predator, the introduced European carp (*Cyprinus carpio*), which still occurs in the lake by the tens of millions and will likely not be completely eliminated into the foreseeable future.

Native clams (Family Sphaeriidae) population data is completely lacking from Utah Lake. However, we have found hundreds of weathered native clam shells throughout the lake and tributary sediments (Figure 24). Regrettably, we have not found any live native clams. Their abundances in Utah Lake were likely as great or greater than native mussels because unlike native mussels, native clams do not require secondary fish hosts to complete their life cycle. We suspect that similarly to native mussels, Utah Lake's native clam populations are either extinct or very close to extinction.

Utah Lake's native snail populations likely consisted of more than a dozen species, all of which had a different ecological niche and provided vital ecosystem functions in the lake (Richards 2014, 2017, 2018b, and 2019a). Almost all of these taxa are extinct in Utah Lake including; several species of springsnails in the genus, *Pyrgulopsis*, the pebble snail, *Fluminicola coloradoensis* (Figure 28), two valvata species, *Valvata humeralis*, the glossy valvata, and *Valvata utahensis*, the Utah round mouth snail or desert snail, *Planorbella binney*, the Coarse Rams-Horn (take and add photo), and the iconic *Helisoma newberryi newberry*, the Great Basin Ramshorn (Figure 26, Figure 27). Utah Lake probably supported the largest population of *Planorbella binney* in Utah (Oliver and Bosworth 1999). Several other tolerant snail species still exist in the lake, including those in the families Lymnaeidae and Physidae. We continue to find thousands of the now extinct snail shells along Utah Lake's shorelines and in benthic sediments during routine benthic sampling.

We have documented that native mollusk shells form a layer about 1 to 3 cm thick within the benthic sediments just under an organic layer, especially in Provo Bay. We are currently estimating relative abundances of these extant and extinct species from several locations in the lake in an effort to understand recent past environmental spatial patterns in Utah Lake. We are also collaborating with several researchers examining sediment core samples from the lake to estimate when most of these mollusk species went extinct.



Figure 21. *Margaritifera falcata* from Big Cottonwood Creek, Salt Lake County, UT, circa 1880. These specimens are housed in the Natural History Museum of Utah, Salt Lake City, collected by Dr. Orson Howard, Professor Biology at University of Utah, in the late 1880's and were apparently fairly common (Richards personal examination of museum specimens) however, the exact location of collection in Big Cottonwood Creek was not documented. These historical specimens were identified by Dr. Howard as *Margaritana margaritifera*, which was later revised to *Margaritifera falcata*.



Figure 22. One of the few remaining live adult *Anodonta* found lying on the surface of what was mostly comprised of thousands of invasive Asian clams, *Corbicula*, in Currant Creek, a former tributary to Utah Lake, August 2016.



Figure 23. Weathered *Anodonta* shell exposed during low water levels in Utah Lake near SW corner of Provo Airport, July 29, 2016.



Figure 24. Remnants of the diverse, abundant, and unique bivalve assemblage of Utah Lake exposed in the substrate of a spring creek entering the lake near Vineyard, UT in 2016. At least three species of bivalves are visible, two species of fingernail clams and the mussel *Anodonta* sp., which is considered extinct in Utah Lake. Native fingernail clams may also now be extinct in the lake and its tributaries.

These biofilterers are presumed to have occurred in such high abundance that they regularly filtered the entire water column of the lake in a few days and thus were instrumental in controlling cyanoHABs. This spring creek has now been polluted by urban development and these shell fragments are now covered in thick sediment.



Figure 25. Mollusk shell remnants (two white bands) piled along the eastern shoreline of Goshen Bay, Utah Lake. Shells are mostly native snails, but native clams and mussels occur as well. These remnants show that mollusks in Utah Lake were much more diverse, abundant, and unique than at present and were the dominant benthic ecosystem engineers of Utah Lake until the recent past.

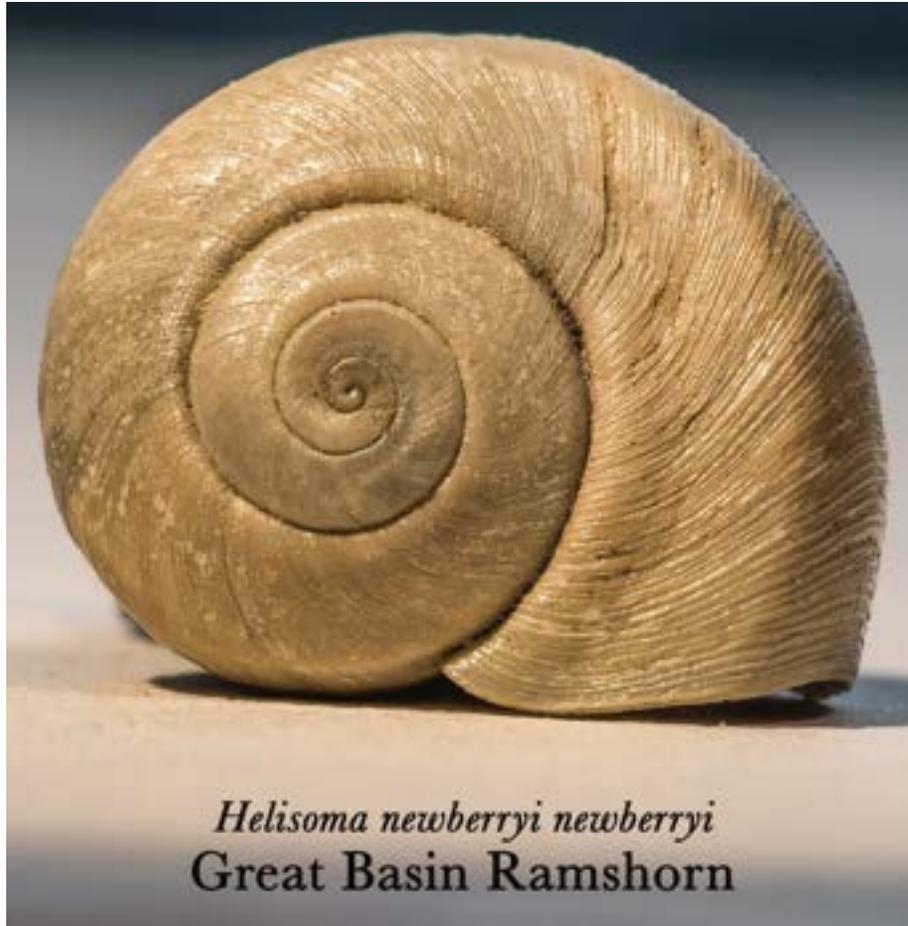


Figure 26. Shell of *Helisoma newberryi newberryi*, the Great Basin Ramshorn, a former abundant resident of Utah Lake. No live specimens of this subspecies have been found in the lake for many years and its viability in Utah is questionable but presumably it is extinct. Snail assemblages in Utah Lake were instrumental in regulating benthic algal assemblages and nutrient transfer into the food web and their past role in the ecosystem is now vastly diminished. Presently, only two or three pollution tolerant species survive.



Figure 27. Remnant shells of *Helisoma newberryi newberryi*, The Great Basin Ramshorn and *Valvata utahensis*, the Utah Round Mouth Snail or Desert Snail collected from Utah Lake in 2019.



Figure 28. Empty shells of the prosobranch snail, *Fluminicola coloradoensis*, Green River Pebblesnail found in several locations in Utah Lake during our mollusk surveys. Scale lines are 1 mm.

Native mollusks were the consummate ecosystem engineers in Utah Lake and until recently (200 to 150 years ago) governed almost all of its ecosystem functions. Due to human economic activities, however, most native mollusks in Utah Lake have been extirpated and no longer provide those critical ecosystem services, including the regulation of cyanoHABs.

Utah Lake's two native mussel species and several clam species likely dominated the benthic invertebrate community responsible for water column nutrient cycling in Utah Lake both numerically and in terms of biomass ≥ 150 years ago. They performed both the function of particle removers from the water column and regulated other biota involved in water purification, including algae, bacteria, and fungi in the sediments (Newell 1988; Newell and Ott 2013). They also controlled the key process of oxidation of organic matter particularly the major oxidizer, bacteria (Wetzel 2001; Sorokin et al. 1999; Ostroumov 2005). Native mussels likely directly reduced the amount of particulate organic matter (POM) available to be remineralized by pelagic consumers and bacterioplankton in the lake (Newell et al. 2005). Bivalves are world renowned for the ability to filter large volumes of water and Utah Lake's two native mussels and several species of clams were likely able to filter the entire lake's water column in just a few days (Richards 2014, 2017, 2018b, and 2019a).

Native freshwater mollusks likely constituted the largest portion of benthic standing crop biomass in Utah Lake. Consequently, mollusks were the primary contributors to Ca and CO₃ cycling in the lake. They were also major regulators of other nutrients, including phosphorus, and trace metals (Malathi and Thippeswamy 2013; Mann, 1964; Negus, 1966; Cameron et al. 1979; Liu et al. 2010).

As most of us know, mollusk shells are mostly composed of CaCO₃ and it is reported that shell composition in freshwater mollusks is typically $> 0.95 \text{ g CaCO}_3 \text{ g}^{-1} \text{ shell}$ (Richards 2018; Ademolu et al. 2016; Malathi and Thippeswamy 2013; Mann, 1964; Negus, 1966; Cameron et al. 1979; Liu et al. 2010; Cameron et al. 1979). Mollusks also incorporate minerals into their body tissue. For example, Malathi and Thippeswamy (2013) reported mineral contents of body tissue in the freshwater mussel *Parreysia*

corrugata and showed that mineral contents were substantial and varied seasonally. Liu et al. (2010) found that soft tissues of *Anodonta* sp. contained 19 g Mn kg⁻¹ tissue dry weight and ≤ 19 g Fe kg⁻¹ tissue dry weight.

We have documented the substantial ecosystem effects that invasive *Corbicula* and *Potamopyrgus* can have in the Jordan River, including their influence on Ca, CO₃, nutrients, and mineral cycling and infer that native mollusks had similar effects in Utah Lake in the past (Richards 2018). Mollusks in Utah Lake could have also had substantial effects on phosphorus cycling. For example, it is becoming more common for water treatment facilities to use powdered mollusk shells to remove aqueous P from wastewater (Abeynaiike 2011; Xiong et al. 2015). Abeynaiike (2011) found that with excess of partially calcined pyrolysed mussel shells, at a concentration of 5 g L⁻¹, more than 95% phosphate removal can be achieved.

Our basic understanding of mollusk chemical interactions with their environment suggests that in the past, native mollusks in Utah Lake strongly influenced sediment chemistry in the lake and that sediment, pore-water, and water column chemistry now observed in the lake is a direct result of the loss of native mollusks (Richards et al. 2019). This is particularly important for understanding phosphorus chemistry levels and interactions in the lake as they may relate to cyanoHABs. We cite Randall et al. (2019) for a detailed description of current sediment P in Utah Lake to help illustrate our concern that in the past, native mollusks regulated P and other chemical conditions including pH, CaCO₃, and indirectly regulated cyanoHABs but due to extirpation, are no longer able to do so.

Randall et al. (2019) reported that sediment P in Utah Lake:

“... was primarily bound to oxidized Fe/Mn compounds (BD fraction) and Ca phosphate minerals or acid-soluble organic P (HCl fraction). On average, 49.1% \pm 1.8% of TN was associated with the BD fraction (range: 41–61%) and 38.6% \pm 2.1% with the HCl fraction (range: 25–47%). Given low Mn concentrations in Utah Lake sediment (.06 wt. %), the BD fraction likely represents P associated with Fe rather than Mn. Likewise, given that the sediments are calcite-rich, the HCl fraction is likely dominated by P from calcium phosphate minerals rather than organic P.”

And that,

“The association of P with different minerals affects the subsequent mobility in sediments and potential flux to the water column. Phosphorus was primarily associated with Fe (Fe (OOH)) and Ca (CaCO₃ or Ca₁₀(PO₄)₆(OH,F,Cl)₂ \approx P) from the BD and HCl fractions. These minerals act as sinks to sequester P from water column to the sediment. Coprecipitation of P with calcite and apatite minerals is strong at neutral to alkaline pH. Utah Lake is an alkaline lake with pH values typically over 8 and is buffered by high bicarbonate concentrations. The Ca-associated P is likely precipitated with calcite (CaCO₃ \approx P) or apatite (Ca₁₀(PO₄)₆(OH,F,Cl)₂) minerals. Although sediment chemistry is not kinetically conducive to precipitation of apatite, authigenic apatite mineral precipitation may occur as diatoms store polyphosphates inside their cells and form Ca phosphate minerals [35]. The majority of Ca-associated P may act as a permanent sink for P in Utah Lake with alkaline pH values. In

contrast, the Fe-bound P pool is mobile as P may be released from sediments under anoxic conditions with the reductive dissolution of Fe-oxide minerals. Anoxic conditions may develop locally near the sediment-water interface, particularly during summer when microbial activity is high. Although we did not evaluate seasonal differences in P fractionation in this study, it is possible that the Fe-bound fraction may decrease as P is released under summertime anoxic conditions. While flux rates from the sediment to overlying water column are unknown, the sediment-water interface is potentially a major controlling factor of P cycling in Utah Lake.”

This remarkable similarity between sediment chemistry in Utah Lake reported by Randall et al. (2019) and mollusk chemical interactions with their environment strongly suggests that native mollusks were instrumental in regulating nutrient cycling and sediment chemistry in Utah Lake. We suggest that mollusks also regulated alkalinity in the lake via their sequestration of CaCO₃. These strong and synergistic associations between sediment chemistry and mollusks in Utah Lake requires further analyses perhaps starting with conceptual models and is especially important for understanding the regulation of cyanoHABs in the lake.

Midges

After the inevitable decline and demise of Utah Lake’s native molluscan fishery over the past 100 years or so, remaining non-molluscan benthic invertebrates became the dominant native benthic ecosystem engineers, especially chironomids (midges) and oligochaetes (aquatic worms). Non-molluscan benthic macroinvertebrates are now one of the most critical components of the ecology and ecosystem functioning of Utah Lake. They are a major link between sediment chemistry, water column chemistry, nutrient cycling, benthic algae, phytoplankton, and Utah Lake’s food web including non-game and game fish, June suckers, waterfowl, shorebirds, etc. Benthic invertebrates provide underappreciated but invaluable ecosystem services and are keystone taxa instrumental for the functioning of Utah Lake’ ecosystem.

Midge larvae (Family Chironomidae; Class Insecta) dominate the benthic ecosystem in Utah Lake and can often comprise 80-90% of the benthic invertebrate biomass with a standing crop wet weight biomass of 700 to 3000 U.S. short tons, which can be produced in one summer month (Richards and Miller 2019c). By their sheer volume, biomass, secondary production, and ecology; midge larvae are the benthic/sediment ecosystem engineers responsible for much of the lake’s benthic/sediment function and interaction with the water column (Richards and Miller 2019c, Holker et al. 2015); and has been reported by Randal et al. (2017) and Hogsett et al. (2019), the sediment water interface appears to be a major controlling factor of P recycling.

Adult midges also transfer energy and nutrients out of Utah Lake into surrounding wetlands after larval pupation and adults become airborne. Midge swarms along the shoreline of Utah Lake are often intense with tens of thousands of adults participating in their mating rituals. The following two videos show such swarms along the lake’s shores:

<https://www.youtube.com/watch?v=vVSgmNQS9YI>

and

<https://youtu.be/aE4nThbiY6s>

Adult midges also rest in shoreline vegetation between mating (Figure 29) and before females release eggs back into the lake.



Figure 29. Adult male midge (Chironomidae) resting on a wild iris in wetlands along the eastern shore of Utah Lake, July 2019. This stand of wild irises was destroyed in 2019 during the phragmites removal program.

Although midge densities are extremely high in Utah Lake, they are often much higher in Farmington Bay (Figure 30) and are nothing compared to densities and swarms that can occur in Lake Myvatn, Iceland¹².

¹² Lake Myvatn literally translates to Midge Lake.



Figure 30. Adult midge swarm at Farmington Bay, Great Salt Lake wetland ponds. Swarms appear to be dark funnel clouds along the wetland horizon and are not controlled burning.

The following video shows a typical midge swarm in Lake Myvatn:

<https://www.youtube.com/watch?v=E0BhQm27RA4>.

It has become clear that several dominant benthic taxa, primarily chironomids (midges), can alter benthic ecosystem function and play a key role in the timing and intensity of cyanoHABs in lake ecosystems. However, this relationship has received very little attention, particularly in Utah Lake. For refresher, the section on the relation between midge larvae and cyanoHABs that we reported in our 2016 Progress Report has been added in Appendix 1. In the following section, we discuss our latest literature findings on just how important midge larvae can be to benthic ecosystem functions, including cyanoHABs in Utah Lake.

Substrate Stabilization and Structure, Net Ecosystem Production, and cyanoHABs

Larval midge tubes are constructed from silk similar to the kind of silk produced by spiders, which has very strong tensile strength and ductility. Midge larvae also produce connecting networks of silk that stabilizes the substrate and provides three-dimensional structure to the sediment (Olafsson and Paterson 2004, H€olker et al. 2015). Midge larvae can reach very high densities in Utah Lake, which certainly helps stabilize the substrate and increases structure (Figure 31).



Figure 31. Thousands of different midge larval instar tubes in Provo Bay, Utah Lake. These larvae help stabilize the easily disturbed substrate, provide three-dimensional structure, and the larvae actively oxygenate the sediments including Fe near the sediment water boundary layer. Tubes are likely either *Chironomus* sp. or *Tanytus* sp. or both. This photo was taken during a low water year when water levels were shallow enough that large insectivorous fish were excluded, and predation was reduced allowing midge populations to maintain high densities and to continue to provide valuable ecosystem services other than just as fish food.

Midge larval tubes increase sediment shear strength subsequently reducing resuspension and turbidity. Ólafsson and Paterson (2004) documented that *Tanytarsus gracilentus* (midge) larvae in Lake Myvatn, Iceland modified the surface sediment by tube building and showed that shear strength of the sediment surface, and hence resistance to erosion, increased significantly with increased densities of *T. gracilentus* larvae (Phillips et al. 2019).

Sediment stabilization is critical for Utah Lake because among other things, sediments and nutrients are easily suspended and affect turbidity and nutrient availability, which often favors cyanoHABs (see the previous section on Turbidity). Midge larval tubes provide three-dimensional structure that also increases habitat for small microorganisms and algae. By providing stable substrate for algae, larval midge tubes indirectly increase gross primary production (GPP) in the sediment, although by

consuming algae, midges may inhibit GPP. Midge larvae can also stimulate microbial respiration (RESP) by oxygenating the sediment. (Phillips et al. 2019, Holker et al. 2015). Therefore, the overall effect of midge larvae on net ecosystem production (NEP) depends on the balance between their effects on GPP and RESP, which is also affected by light conditions (Phillips et al. 2019) (Figure 32).

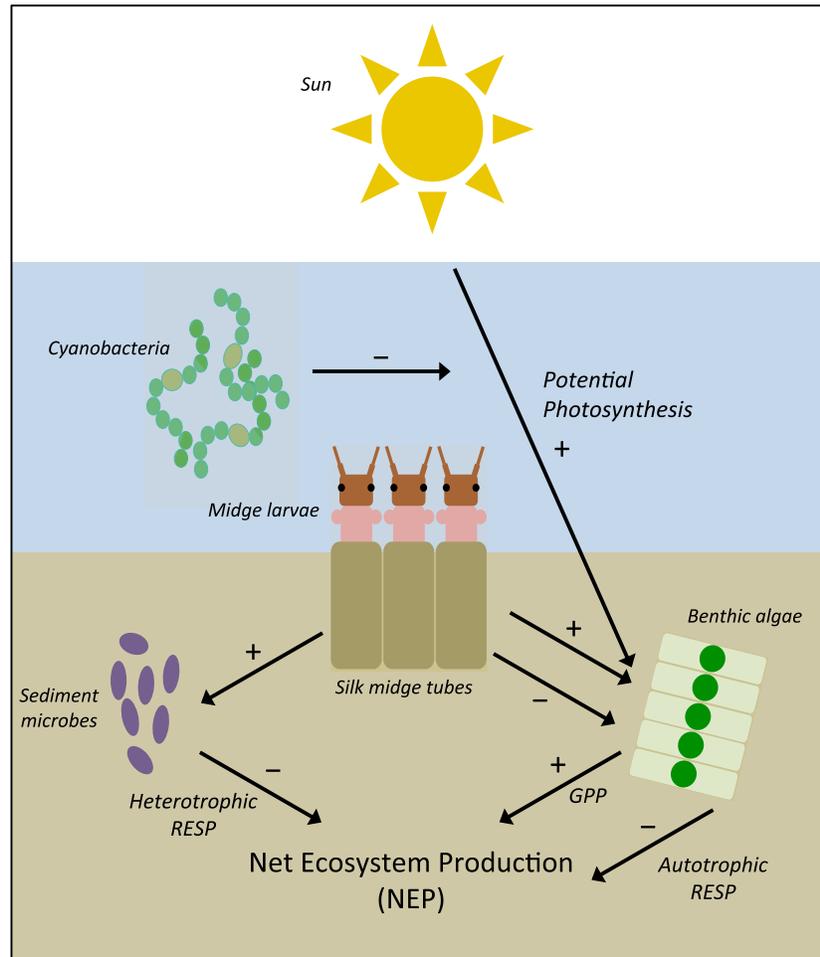


Figure 32, Midge larvae alter benthic ecosystem function. This figure and caption were taken from Philipps et al. 2019. "Midges can alter benthic ecosystem function. Larval midges build silk tubes that provide a substrate for algal growth and increase gross primary production (GPP) in the sediment. However, midges may inhibit GPP through consumption of algae. Furthermore, midges can stimulate microbial respiration (RESP) by oxygenating the sediment. Gross primary production and RESP have opposite effects on net ecosystem production (NEP), so the effect of midges on NEP depends on the balance between their effects on GPP and RESP. We hypothesized that light mediates this balance, because the positive effects of midges on GPP would decline as photosynthesis became more limited by light. Episodic cyanobacterial blooms have a negative effect on benthic light levels, which could result in spatiotemporal variation in the net effects of midges on benthic production."

Midge larvae and cyanoHABs

We briefly reported how midge larvae could affect cyanoHABs in Utah Lake in our Utah Lake 2016 Progress Report, including how circumstantially, cyanoHABs appeared to cycle out of sync with larval abundance (Richards and Miller 2019c, and Appendix 1 of this report). Philips et al. (2019) further elaborated on these effects more recently (Figure 32). Einarsson and Örnólfsson (2004) also reported that intense cyanoHABs blooms (*Aphanizomenon flos-aquae*) always occurred in years of low chironomid populations but sometimes developed in other years in Lake Myvatn and similar to our current findings; cyclic patterns of dominant midges varied several orders of magnitude. Einarsson and Örnólfsson (2004) also suggested that cyclic patterns of midges were not likely due to climate. *Tanytarsus gracilentus* in Lake Myvatn showed cyclic population fluctuation with three peaks occurring during a 20-year period. Body size of *T. gracilentus* fluctuated with population size but in an opposite fashion and with a time lag in Lake Myvatn again, similar to our findings in Utah Lake (Richards and Miller 2019). *T. gracilentus* body size and abundance and predator abundance in Lake Myvatn suggested that the population fluctuations were driven by interaction with resources and not by predator-prey interactions (Einarsson et al. 2002). However, there are only two major predacious fish in Lake Myvatn, three-spined stickleback (*Gasterosteus aculeatus*) and Arctic charr (*Salvelinus alpinus*) (Einarsson et al. 2004), whereas there are several invasive fish species in Utah Lake that are avid midge larvae hunters; common carp (*Cyprinus carpio*), white bass (*Morone chrysops*), channel catfish (*Ictalurus punctatus*), and black bullhead (*Ameiurus melas*), to name a few. Common carp biomass is measured in the tens of tons in Utah Lake and this species alone can regulate or decimate midge populations within the lake.

We agree with the midge researchers on Lake Myvatn that the underlying mechanisms for midge cycles are not fully understood and that further investigation is required and that by sheer abundance, midges may be one of the major regulating factors in the long-term dynamics of Lake Myvatn and Utah Lake ecosystems. We also agree with our Icelandic colleagues that “for effective conservation, the only sound strategy seems to be to avoid interfering with the basic components of the ecosystem” (Phillips et al. 2019).

The effects of midge larval on ecosystem respiration (RESP) and GPP vary seasonally with greater effects in summer during increased temperatures. Baranov et al. (2016) showed that RESP in sediments with and without chironomids did not differ at 5⁰ C, but at 30⁰ C sediment respiration in microcosms with 2000 chironomid larvae per m² was 4.9 times higher than in uninhabited sediments. This is a somewhat lower density of larvae than what we typically find in Utah Lake and compared to their results suggest that midge larval effects on RESP may be higher in Utah Lake.

Warm summer water temperatures result in faster midge larval development, shorter life cycles, additional generations per year and higher reproduction rates—all resulting in higher larval densities and intensified ecosystem effects (Hamburger et al. 1995;

Eggermont and Heiri 2012). With large densities, especially in eutrophic water bodies with warm water, midge larvae burrowing, and ventilation activities can dramatically impact freshwater biogeochemistry (Morad et al. 2010). For example, in shallow Lake Muggelse in Germany (mean depth 5 m, relatively similar to Utah Lake mean depth) a volume equivalent to the total water column of the lake is pumped by chironomids through their burrows, once a week (Morad et al. 2010). This rate is likely similar to Utah Lake. That is, during certain times of year when midge larvae are at relatively high densities and are active, they can pump the entire water column of Utah Lake through the sediments, perhaps weekly or less. Baranov et al. (2016) concluded that high densities of chironomids in shallow lakes can significantly intensify sediment respiration, especially in warm and well-oxygenated systems. This effect is most pronounced in shallow, non-stratified lakes such as Utah Lake and is consistent with sediment chemistry findings by Hogsett et al. (2019).

Very few studies have been conducted on the benthic invertebrate assemblages in Utah Lake (Barnes and Toole 1981, Spencer and Denton 2003, Shiozawa and Barnes 1977) and none were conducted at the level and intensity that is presently being accomplished by this group. No study has ever examined the role of benthic invertebrates on HABs in Utah Lake, this is the first. Our research is also an important element of sediment chemistry, nutrients, and food web models that are presently being conducted by us and others on Utah Lake. Preliminary results of our research are leading to valuable insights on the role of benthic macroinvertebrates in the ecology and ecosystem functioning.

Invasive Fishes: Benthic Ecosystem Disruptors

Invasive benthivorous fish can alter the benthic sediment ecosystem more drastically than did mollusks and midges in Utah Lake. It is well known that common carp (*Cyprinus carpio*) are one of the most important pelagic and benthic ecosystem disruptors in Utah Lake. Carp are omnivorous and feed largely on macrophytes and invertebrates. In Utah Lake, feeding activity by adult carp and catfish involves rooting in the sediment and disrupting and destabilizing natural processes, causing large amounts of bioturbation. Native mollusks and midge larvae on the other hand, stabilized sediments and reduced turbidity. Benthic feeding invasive fishes also destabilize Utah Lake sediments in other ways that have not been fully documented or quantified. However, others researchers have reported that in general, the net impact of ecosystem engineers depends on the balance between diverse and potentially conflicting effects, which itself can depend on the environmental context and the species involved (Bertness et al. 1999, Norkko et al. 2006, Daleo and Iribarne 2009, Brown and Lawson 2010, Lathlean and McQuaid 2017). The context dependence of ecosystem engineering means that environmental variation can produce large differences in engineer effects through space and time (Wright et al. 2006, Hastings et al. 2007). Ecosystem responses of engineering to environmental

conditions are likely nonlinear (as is true for many ecological processes) and the shape of this nonlinearity could have large consequences for the overall effects of analog engineering in variable environments (Ruel et al. 1999), such as Utah Lake. We strongly suggest that additional research is needed comparing the now analog benthic ecosystem engineer invasive fishes with past molluskan engineers and remaining midge engineers in Utah Lake.

Invasive Dojo: Game Changer

The occurrence of Oriental Weatherfish, *Misgurnus anguillicaudatus* (a.k.a. Dojo) in the Utah Lake drainage is also very concerning. We recently documented this invasive species in the Jordan River just downstream of Utah Lake (Richards 2019). Dojo were common throughout the upper Jordan River in our 2019 electrofishing survey and occurred in mostly in silty, heavily vegetated habitat in slower sections of the river.



Figure 33. Image of Invasive Dojo (*Misgurnus anguillicaudatus*) from nas.er.usgs.gov. Zachary Randall, Florida Museum (UF 236274) ©.

Dojo life history and ecology are poorly documented in North America, including ELS. However, multiple authors have found that this species has caused reductions in macroinvertebrate populations, altered aquatic habitats, and are vectors for certain fish parasites (<https://www.fws.gov/Fisheries/ANS/erss/highrisk/Misgurnus-anguillicaudatus-WEB-8-21-12.pdf>). Dojo are considered r-selected organisms that mature quickly, reproduce multiple times in their lifetimes and throughout the year and produce a great number of offspring. In their native range, these fish spawn multiple times per year during a spawning season that lasts from mid-April until mid-October and appear to be altering their breeding season based on different environmental conditions in their non-native ranges (Frable 2008). Dojo tolerate and thrive in a very wide range of temperature and oxygen conditions and their thermal tolerances range from 3.7 °C to 19.3 °C (Logan et al. 1996). They are able to survive in low oxygen and hypoxic conditions and are known hosts of non-native digenean trematode flukes and viral pathogens (Lintermans et al. 1990; Frable 2008). Keller and Lake (2007) demonstrated in the laboratory that oriental weatherfish caused significant levels of increased turbidity and water column nitrogen. Dojo are likely altering the Jordan River ecosystem and will continue to do so

as their population increases. These ecological and life history factors could contribute heavily in the facilitation of harmful algal blooms and eutrophication of many water bodies. Dojo could become a severe problem in Utah Lake, if and when they become established; if they haven't done so already. It is our opinion that Dojo will inevitably become established and will thrive in Utah Lake and will cause major ecosystem shifts that could exacerbate cyanoHABs.

Recommendations

It has become obvious that continued research and gathering of basic ecological data on Utah Lake is essential based on our cumulative years of ecological research and ongoing data synthesis. Subsequently we recommend the following:

1. Continued ecological data collection focusing on gaps in our knowledge, some of which have been discussed in this progress report.
2. Continued refinement of Utah Lake zooplankton taxonomy, without which, valid analyses are impossible.
3. Continued development of our Utah Lake Multimetric Index of Biological Integrity and population of metrics within the index. This will be an invaluable tool for managers.

Acknowledgements

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Appendices

Appendix 1. Midge larvae and cyanoHABs section from Richards and Miller 2019.

Benthic Macroinvertebrates and HABs

The relationship between benthic macroinvertebrates, particularly worms and midges, and harmful algal blooms has received very little attention. In this section, we discuss the latest science on just how important these interactions are to Utah Lake HABs.

For several decades it has been recognized that anoxia is a pre-condition for cyanobacteria blooms in eutrophic waters (Trimbee and Prepas 1988) and that warm temperatures and stable water columns promote anoxia (Paerl, 1988; Zhang and Prepas, 1996). However, the role of Fe in cyanobacteria blooms has been severely underappreciated and the role of midge (Chironomidae) larvae in regulating Fe availability has been even less so.

Anoxia and Fe

Molot et al. (2014) proposed that the role of anoxia and ferrous iron was critical for

cyanobacteria bloom formation. Their model can be summarized as follows:

“The model has several critical concepts: (i) P regulates biomass and productivity in fresh waters until excessive loading renders a system N-limited or light-limited, but it is the availability of ferrous ions (Fe^{2+}) that regulates the ability of cyanobacteria to compete with its eukaryotic competitors; (ii) Fe^{2+} diffusing from anoxic sediments is a major Fe source for cyanobacteria, which acquire it by migrating downwards into Fe^{2+} -rich anoxic waters from oxygenated waters; and (iii) subsequent cyanobacterial siderophore production provides a supply of Fe^{3+} for reduction at cyanobacteria cell membranes that leads to very low Fe^{3+} concentrations in the mixing zone.

When light and temperature are physiologically suitable for cyanobacteria growth, bloom onset is regulated by the onset of internal Fe^{2+} loading which in turn is controlled by anoxia, reducible Fe content of surface sediments and sulphate reduction rate.”

Figure 34 (taken from Molot et al. 2014) illustrates this concept.

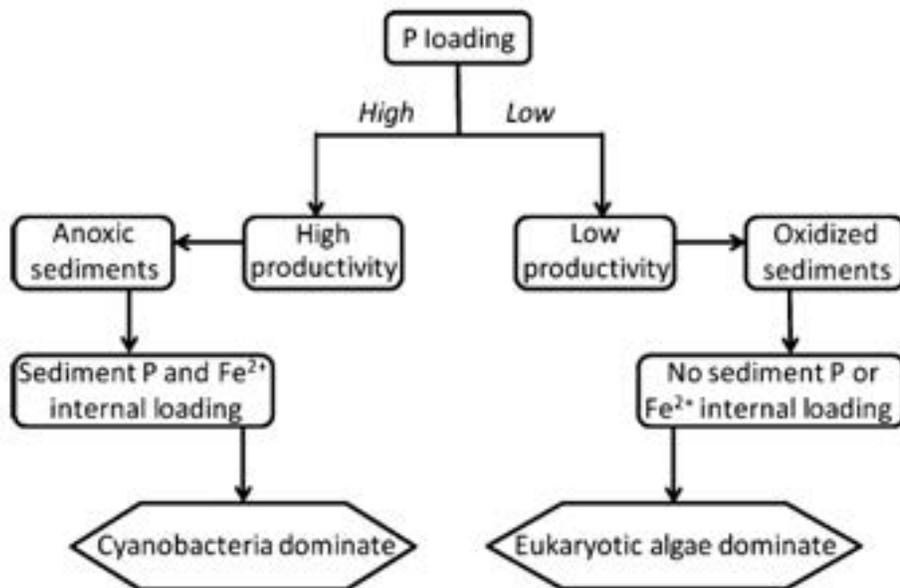


Figure 34. Simplified conceptual diagram of the modified phosphorus eutrophication model of cyanobacteria bloom formation for systems lacking naturally anoxic surficial sediments. The only factor controlling Fe^{2+} production shown here is anoxia at the sediment water interface.

Figure 35 (taken directly from Molot et al. 2014) illustrates the processes that promote Fe delivery to cyanobacteria and thereby promote cyanobacteria dominance in lakes.

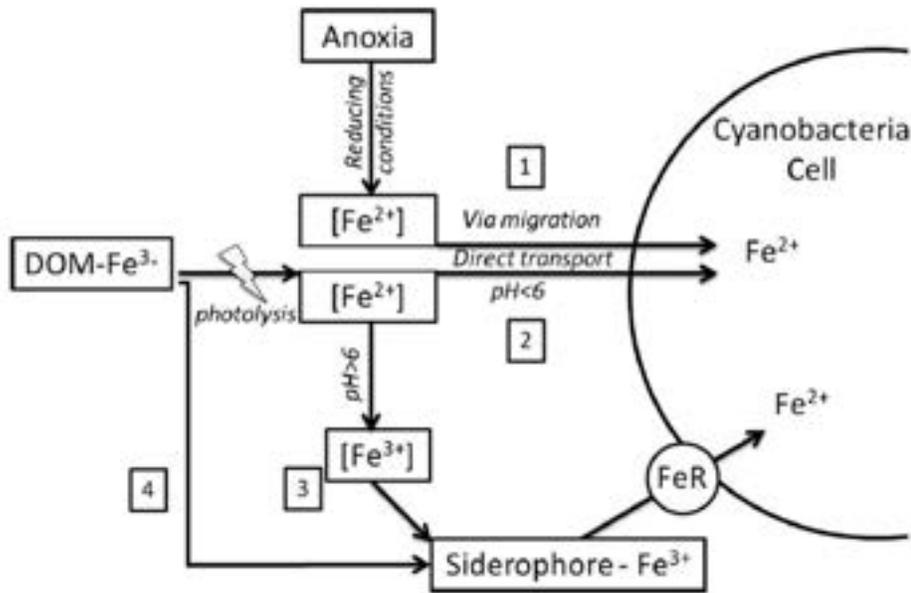


Figure 35. Anoxia: systems with anoxic sediments will experience Fe^{2+} flux into anoxic waters. Migrating cyanobacteria can acquire Fe^{2+} for direct transport into cells. (2) Photoreduction: DOM- Fe^{3+} can be photo-reduced, giving rise to Fe^{2+} that is available for direct Fe^{2+} transport into phytoplankton cells, but the transport rate is pH dependent. Acidity affects rates of abiotic oxidation by dissolved O_2 and at pH < 6 Fe re-oxidation may be low enough to give rise to a pool of transportable Fe^{2+} . At higher pH, much of it is probably re-oxidised before transport. (3) and (4) Fe-scavenging (or acquisition) system: siderophores are produced by cyanobacteria that can (3) bind free soluble Fe^{3+} and (4) cleave Fe^{3+} from DOM complexes. Scavenged Fe^{3+} is then delivered to the cell membrane, creating a pool of Fe only accessible by cyanobacteria. Fe^{3+} is reduced by the Fe reducing system (FeR) before transport across the inner membrane. The Fe^{2+} pool is shown as two separate pools - in anoxic waters (internal loading) and in the mixing layer (photo-reduction).

Midge larvae

Midge larvae can actively oxygenate the sediments near the sediment/water boundary, including converting Fe^{2+} to Fe^{3+} (Holker et al. 2015) (Figure 36). Midge larvae are extremely abundant in Utah Lake sediments and can thus have a tremendous effect on Fe conversion. However, midge larvae pupate and then leave the lake as adults and are therefore not always present actively aerating the sediment. Subsequently, when larvae pupate and leave sediments as adults, larval tubes collapse and Fe^{3+} reduces to Fe^{2+} (Figure 36).

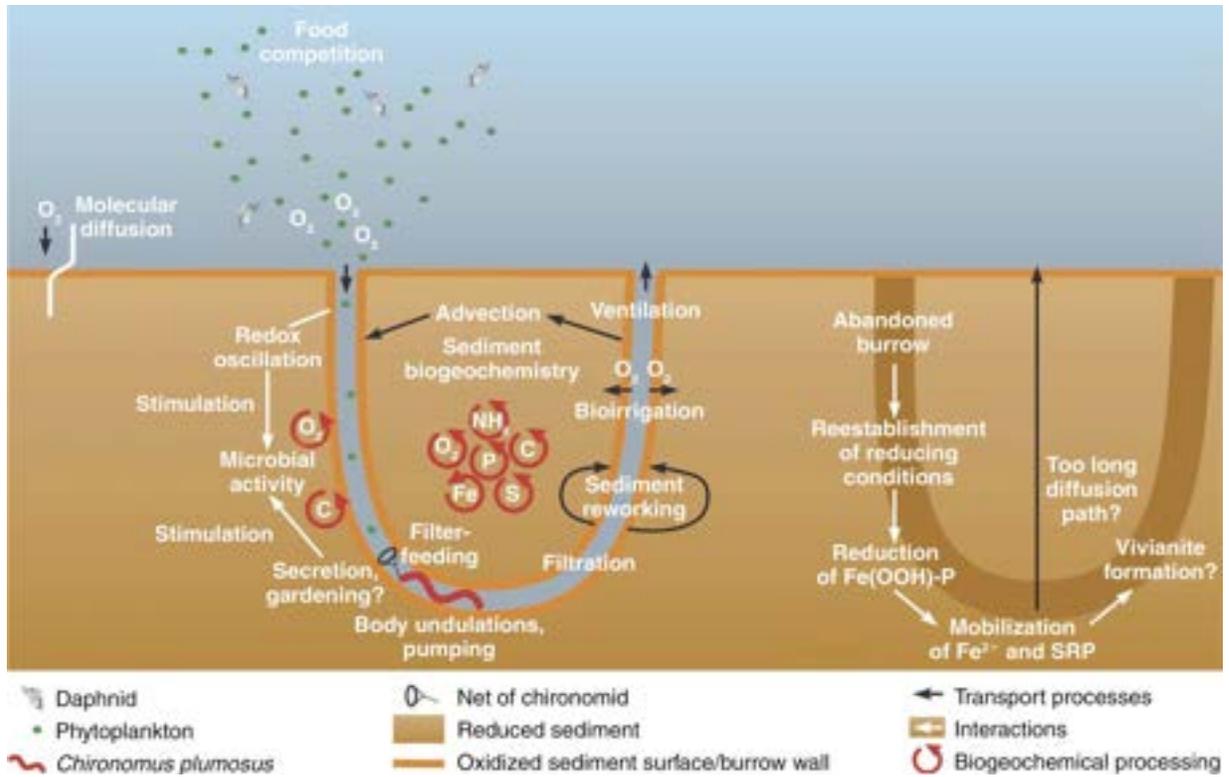


Figure 36. From: Holker et al. 2015. Tube-dwelling invertebrates: tiny ecosystem engineers have large effects in lake ecosystems

A Provisional Multi-Metric Index of Biological Integrity (MIBI) to Assess Water Quality in Utah Lake centered on Regulatory Directives

Draft Technical Report



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Regarding society’s poor track record of environmental protection:

“The complex reasons for failure center on the hubris of a society that behaves as if it could repeal the laws of nature. Plans generated by economists, technologists, engineers, and ecologists have too often assumed that lost or damaged components of ecological systems are unimportant or can be repaired or replaced.” J. R. Karr, 1996

Cover image: Female cyclopoid copepod.

http://www.ulrichhopp.de/bilder/kleinkrebse/Kleink_03_Mesocyclops_leuckarti_003.jpg

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Introduction

Multimetric indices of biological integrity (MIBIs) are a type of bioassessment that rely on empirical knowledge of how a wide range of biological attributes respond to varying degrees of human influence (Karr 1993, Karr and Chu 1997). The most useful MIBIs explicitly embrace several attributes of the biotic assemblages, including; taxa richness (diversity) and composition, indicator taxa (e.g., tolerant and intolerant groups), population dynamics, production, and an assessment of processes that include trophic structure, feeding strategies and other functional traits (Allen and Polimene 2011; Calow 1987; Cao and Hawkins 2019). The goal of a MIBI is to measure and evaluate the consequences of human actions on biological systems (Karr 1993, Karr and Chu 1997) however, it should be emphasized that bioassessments, including MIBIs, are not science but are the link between scientists and managers, and thus some level of subjectivity (e.g. professional judgment and management objectives) is inherent and cannot be completely avoided. MIBIs are evaluative precursors to more intensive, stressor specific, monitoring programs. They are assessment tools not monitoring tools and should not be used as such, although more comprehensive MIBIs such as the one presented in this report can help guide managers as to the types and causes of impairment.

Utah Lake

Utah Lake is an underappreciated, unique, and ecologically important part of Utah's (and the nation's) natural heritage. It is one of the few freshwater remnants of pluvial Lake Bonneville, that likely outsized Lake Michigan in size and volume. Utah Lake has until recently supported one of the most diverse and productive molluscan faunas in the western USA with perhaps twenty snail, clam, and mussel taxa. These mollusks likely dictated much of Utah Lake's ecosystem function (Richards and Miller 2017; Richards and Miller 2019; Richards 2016, 2018, 2019). Unfortunately, the majority of these molluscan taxa have been extirpated from the lake and their populations have been drastically reduced throughout most of its drainage (Richards and Miller 2019; Richards and Miller 2017; Richards 2016, 2017, 2018, 2019a). Utah Lake was also once home to at least a dozen native fishes, including the Bonneville Cutthroat Trout (extirpated), Utah Lake Sculpin (extinct), and June Sucker (endangered) due to in part its ancient lineage and isolation from other large bodied freshwater lakes. Most native fishes have been extirpated from Utah Lake.

Regrettably, Utah Lake is now a highly regulated and abused reservoir ecosystem that has undergone human induced ecological hysteresis and catastrophic shifts and no longer resembles its natural self, pre-Mormon settlement. Consequently, Utah Lake is biologically impoverished. According to Karr (1996), "if biotic impoverishment is the problem, then protecting the integrity of" Utah Lake's "biological system(s) must be the goal".

Sections 101(a) of the Clean Water Act (1987) legally mandates USEPA to protect the physical chemical, and **biological integrity**^{13, 14} of our nation's waters. In addition, the Clean Water Act necessitates protection and enhancement of shellfisheries¹⁵, which many managers fail to realize, includes freshwater mollusks (mussels, snails, clams).

Under the provision of the Clean Water Act, the Utah Division of Water Quality (UDWQ) is mandated to protect Utah Lake's three designated biological beneficial uses:

1. Warm-water fisheries,
2. other aquatic life (e.g. bird populations), and the
3. aquatic life they depend on (UDWQ 2019).

UDWQ is also required to protect for recreational beneficial use of Utah Lake; the main impairment is considered toxin-producing cyanoHABs, and for agricultural uses¹⁶.

Because Utah Lake: 1) is the last freshwater remnant of pluvial Lake Bonneville, 2) its large size (surface area \approx 100,000 acres) in an semi-arid climate, 3) its unique molluscan and fish diversity heritage, and 4) no other 'reference' water bodies with which to compare; the index of metrics (MIBI) and baseline values presented in this report are site specific for Utah Lake. The index can, however, be modified for other lentic waters and will be for Farmington Bay of Great Salt Lake.

Metrics

Primary Metrics

The Utah Lake MIBI is composed of relatively easy to measure primary metrics specifically targeting designated beneficial uses (fisheries, shell fisheries (e.g. mollusks), birds, and the aquatic life they depend (e.g. zooplankton, benthic invertebrates) including:

1. Benthic macroinvertebrate diversity,

¹³ We adhere to the following definition of biological integrity throughout this document and during all of our research endeavors: **Biological integrity** refers to the capacity to support and maintain a balanced, integrated, adaptive biological system having the full range of elements (genes, species, assemblages) and processes (mutation, demography, biotic interactions, nutrient and energy dynamics, and metapopulation processes) expected in the natural habitat ... (Angermeier and Karr, 1994; Frey, 1975; Karr and Dudley, 1981; Karr et al., 1986).

¹⁴ The combination of physical, chemical, and biological integrity = ecological integrity (Karr 1996).

¹⁵ The Clean Water Act (1987) states that: "It is the national goal that wherever attainable, an interim goal of water quality which provides for the protection and propagation of fish, **shellfish**, and wildlife and provides for recreation in and on the water be achieved."

¹⁶ UDWQ (2019) documents that, "The state classifies waters based on their uses and develops water quality standards to protect those uses. Utah's designated uses include drinking water, recreation, aquatic wildlife, and agriculture. Utah Lake is protected for the following designated uses:

- 2B: Infrequent primary contact recreation such as boating, wading, or similar uses
- 3B: Warm-water species of game fish, including the necessary aquatic organisms in their food chain
- 3D: Other aquatic wildlife.
- 4: Agricultural uses including irrigation of crops and stock watering" (UDWQ 2019).

2. Benthic macroinvertebrate secondary production (biomass as a substitute),
3. Zooplankton diversity,
4. Zooplankton secondary production (biomass as a substitute),
5. Mollusk diversity,
6. Mollusk densities,
7. Fish condition index.

An easy to measure metric for recreational beneficial use (e.g. swimmable) will be:

1. Creation of a DNA identification code of toxin producing cyanoHABs and develop metric baseline values.

The MIBI emphasizes the importance of Utah Lake's unique molluscan fauna, the importance of benthic invertebrates, particularly chironomids to its fisheries and birds, and the importance of zooplankton to its fisheries. Present values of each metric reported by Utah Lake researchers (e.g. Richards and Miller 2017; Richards 2016, 2018, 2019, UDWQ, and others) and/or those reported in the literature will be used as baseline values in which to compare future changes.

Secondary Metrics

There are also several dozen secondary metrics, including functional trait based metrics that are increasingly recognized as equally important or superior to taxa based metrics (Allen and Polimene 2011; Calow 1987; Dehling and Stouffer 2018; Monteiro and Faria 2018; Hayden et al. 2019), that will help fine tune and support the primary metrics and allow managers to better understand the levels and types of impairments affecting the lake.

A brief summary of secondary metrics includes:

- Phytoplankton, zooplankton, benthic invertebrate, mollusk, and fish taxa diversity indices, e.g. evenness, effective number of taxa,
- Zooplankton family relative abundances and ratios,
- Zooplankton, benthic invertebrate, mollusk, and fish functional traits indices: Particularly for zooplankton e.g. body size; mesotrophic vs. eutrophic zooplankton taxa ratio, taxonomic group changes (Cladocera, copepods, rotifers, etc.).

The use and validity of all of the primary and secondary metrics included in the MIBI (Table 1) are well grounded in the ecological and bioassessment literature (see Unabridged Literature Cited and References section).

Baseline Values

All of the metrics listed in Table 1 (Provo Bay specific as an example) will be populated from values based on recent and present conditions. These will be considered baseline scores to evaluate changes. Some metrics will increase or decrease depending on changes in water quality. No overall score(s) will be derived as is frequently done in other MIBIs. We contend that there is no statistical or ecological rationale for weighting each metric and then subjectively combining them into a final score, therefore, we consider each metric as stand-alone. Each metric will either respond separately to different types and levels of

impairment or compliment or add support to other metrics. Avoiding an overall score will allow researchers and managers the ability to observe more subtle changes in conditions and act accordingly.

Less Eutrophic Utah Lake Goal

Many of the metrics values will directly or indirectly change if and when Utah Lake moves along the primary production gradient from the current highly productive 'hyper eutrophic' condition to a lesser productive hyper eutrophic to eutrophic condition, as is the management goal of several agencies, including USEPA and UDWQ. Some metrics may have already exceeded a productivity threshold. For example, Utah Lake benthic invertebrate secondary production may or may not have exceeded a threshold value due to hyper-eutrophic conditions and could increase when primary production (e.g. eutrophication) is lowered. The appropriate value for benthic invertebrate secondary production would therefore be its maximum obtainable to protect for the designated beneficial uses of warm-water fisheries and bird populations.

Spatially and Temporally Derived MIBIs

We have confirmed that biological components including phytoplankton, zooplankton, benthic invertebrates, and fisheries, etc. vary both spatially and temporally in the lake (Richards and Miller 2017; Richards 2016, 2018, 2019, unpublished data and observations). MIBI baseline scores presented in the Table 1 example will therefore be provided for three locations that are mostly ecologically distinct based on others and our research:

- 1) Provo Bay,
- 2) Goshen Bay, and
- 3) Utah Lake proper.

That is, separate MIBIs will be required for each of the three sections of the lake.

Although marinas including Lindon Marina, Utah Lake State Park Marina, and Lincoln Marina experience quite different ecologies and baseline metric values compared to the other two locations, they will not have separate MIBIs. It is apparent that marinas function as a type of pollutant and should be treated as such.

The example MIBI presented in Table 1 includes metrics that reflect the temporal component of Provo Bay's ecology. Recommended times of year such as annual, seasonal, or monthly metric measurement are included in the MIBI.

Focus on Zooplankton Metrics

Zooplankton are a main focus of this MIBI. Zooplankton are in the pivotal position of transferring nutrients throughout aquatic food webs (bottom-up, top-down, trophic cascades) (Caroni and Irvine 2010; García-Chicote et al. 2018) and thus play an essential ecological role within Utah Lake. Zooplankton have a proportionally high indicator value that cannot be encompassed by phytoplankton or fish metrics (Carpenter et al. 1985;

Jeppesen et al. 2011; García-Chicote et al. 2018; Naselli-Flores and Rossetti, 2010; Barnett and Beisner 2007). In addition, the response of zooplankton assemblage structure metrics can be both to specific disturbances and chronic changes ((Attayde and Bozelli, 1998; Cairns et al., 1993; García-Chicote et al. 2018). Subsequently, these ecological roles of zooplankton in Utah Lake are explicitly and implicitly captured in the MIBI.

Zooplankton Taxonomy

There are only about twenty or so zooplankton taxa in Utah Lake (Richards 2019; Marshall 2019), which makes species level identification relatively easy for trained taxonomists or geneticist using DNA barcoding. We consider the Marshall (2019) report to be the definitive taxonomic reference for Utah Lake zooplankton until further modified. This reference was used to develop baseline zooplankton metric scores presented in the MIBI (Table 1 example for Provo Bay).

Fish Assemblage Imbalance

Utah Lake's native fish assemblage no longer exists. Thirteen native species occurred in the lake upon arrival of Mormon settlers in the mid 1800s. The Bonneville Cutthroat Trout, Bonneville Redside Shiner, Mottled Sculpin, Utah Lake Sculpin, Leatherside Chub, Utah Chub, Speckled Dace, Longnose Dace, Mountain Whitefish, and Mountain Sucker no longer exist in Utah Lake. The analog Utah Lake fish assemblage is now dominated by introduced species including Carp, Largemouth Bass, White Bass, Black Bullhead, Channel Catfish, Walleye, Goldfish, Yellow Perch, Blue Gill, and Black Crappie. The fish assemblage in the lake is most certainly out of balance.

We have incorporated several metrics that reflect this imbalance directly and several indirectly. It is well known that planktivorous fish can alter entire lake food webs primarily by preferentially consuming larger bodied zooplankton which in turn preferentially prefer feeding on larger phytoplankton including cyanophytes (Sondergaard et al. 2008; Wetzel 2001; Cole and Weihe 2016; Havens et al. 2015a, 2015b; Gophen 1990; Cooke et al. 2016). All of the fish species currently residing in Utah Lake are planktivorous at least during their juvenile stages. Many studies have shown that removal or reduction of planktivorous fish populations improves water quality including reduction of cyanoHABs leading many researchers and managers to recommend biomanipulation as a relatively inexpensive remedy for controlling algal blooms compared to attempts at whole drainage nutrient control (Riedel-Lehrke 1997; Cooke 1986; Jeppesen et al. 2007; Richards 2019a). We have shown that at least one metric, *zooplankton body length* is reduced in Utah Lake compared to other temperate lakes and that body lengths of zooplankton in the lake vary temporally and spatially in a pattern consistent with planktivore feeding (Appendix 1). Several of the zooplankton metrics in the example MIBI (Table 1) will respond to changes in fish assemblage composition especially if a fisheries biomanipulation program is initiated in Utah Lake.

Species Variability as a Function of Ecosystem Stability

Individual plankton species abundances and assemblage composition variability increases disproportionately to other commonly measured environmental variables as ecosystems become more and more out-of-balance and unstable (e.g. loss of diversity; increased nutrients; other pollution and pollutants; trophic cascades; altered food webs; etc.) (Cottingham et al. 2000; Ptacnik et al. 2008; Zohary 2004; Thomas et al. 2018). The well-established population dynamics literature shows that widely fluctuating populations are a good indicator of disturbance and that at low population levels, extinction risk increases with increased variability (e.g. demographic stochasticity, environmental stochasticity) (Melbourne and Hastings 2008; Vucetich et al. 2000; Pimm et al. 1988). Many phyto- and zooplankton taxa in Utah Lake occur at low abundances that are highly variable (see relevant Richards citations). These taxa are more susceptible to extinction and are inherently useful indicators of impaired conditions. Several metrics in Table 1 reflect low taxa abundance and variability (e.g. CV metrics reflect the well-known theoretical predictions that extinction risk increases with an increase in temporal coefficient of variation in population size (CV) (Pimm et al. 1998)).

The development and refinement of this MIBI is designed to be a collaborative effort between agencies including UDWR fisheries program, UDWQ, WFWQC, and others.

The following table, Tables 1 is our proposed MIBI for Provo Bay and functions as a working guideline for Wasatch Front Water Quality Council researchers and their contractors who are collecting data on Utah Lake.

Table 3. Example of proposed multimetric index of biological integrity (MIBI) template for monitoring Utah Lake. Provo Bay MIBI. Justification of metrics used in this MIBI can be found in Unabridged Literature Cited and Selected Reference Section. Metric values are in the process of being populated in this MIBI. TBD = To Be Determined.

Provo Bay	Metric	Baseline Value	Improvement Change
Phytoplankton	All Divisions		
	<i>Chl A</i> ⁹ (monthly mean and 90% CI)	Jan: Feb: March: April: May: June: July: Aug: Sept: Oct Nov: Dec:	Decrease
	<i>Total biovolume</i> (cells L ⁻¹) (monthly mean and 90% CI) ¹¹	Jan: 326 (91; 561) Feb: 2945 (1,989; 3,900) March: 7,333 (4,239; 10,427) April: 10,988 (6,024; 15,952) May: 75,806 (unk.; 179,259) June: 93,746 (unk.; 190,318) July: 2,289,270 (597,856; 3,980,684) Aug: 606,535 (397,855; 815,215)	Decrease

		Sept: 668,899 (407,730; 930,068) Oct/Nov: 423,521 (290,408; 556,634) Dec: unknown	
	<i>Total biovolume CV</i>	Jan: Feb: March: April: May: June: July: Aug: Sept: Oct Nov: Dec:	Decrease
	<i>Toxin level ($\mu\text{g L}^{-1}$)¹² (monthly mean and 90% CI)</i>	Jan: Feb: March: April: May: June: July: Aug: Sept: Oct Nov: Dec:	Decrease
	<i>Mean cell size (V) ($\mu\text{m}^3 \text{ cell}^{-1}$) (monthly mean and 90% CI)</i>	Jan: Feb: March: April: May: June: July:	Decrease

		Aug: Sept: Oct Nov: Dec:	
	<i>Mean cellular C content (pg C cell⁻¹) (monthly mean and 90% CI)</i>	Jan: Feb: March: April: May: June: July: Aug: Sept: Oct Nov: Dec:	Increase
	<i>Mean C content/mean cell volume (C/V) (pg C μm⁻³) (monthly mean and 90% CI)</i>	Jan: Feb: March: April: May: June: July: Aug: Sept: Oct Nov: Dec:	Increase
	Taxa Based Diversity ¹⁵		
	<i>Richness (seasonal mean and 90% CI)</i>	Winter Spring: Summer: Autumn:	Increase
	<i>Evenness</i>	Winter	Increase/Decrease ¹⁷

	(seasonal mean and 90% CI)	Spring: Summer: Autumn:	
	<i>ENT</i> ¹³ (seasonal mean and 90% CI)	Winter: Spring: Summer: Autumn:	Increase/Decrease ¹⁷
	Division Based Biovolume ¹¹		
	<i>Proportion biovolume Cyanophytes</i> (cells L ⁻¹) (monthly mean and 90% CI) ¹¹	Jan: 0.00 Feb: 0.08 (0.00, 0.16) March: 0.09 (0.00, 0.18) April: 0.09 (0.04, 0.14) May: 0.21 (0.00, 0.44) June: 0.69 (0.59, 0.79) July: 0.87 (0.80, 0.95) Aug: 0.54 (0.46, 0.62) Sept: 0.68 (0.53, 0.83) Oct/Nov: 0.14 (0.05, 0.24) Dec:	Decrease
	<i>Proportion biovolume Chlorophytes</i> (cells L ⁻¹) (monthly mean and 90% CI) ¹¹	Jan: 0.08 (0.00, 0.017) Feb: 0.10 (0.00, 0.24) March: 0.11 (0.05, 0.18) April: 0.44 (0.33, 0.54) May: 0.34 (0.10, 0.58) June: 0.22 (0.15, 0.29) July: 0.11 (0.04, 0.19) Aug: 0.40 (0.32, 0.49) Sept: 0.24 (0.13, 0.36) Oct/Nov: 0.83 (0.73, 0.94) Dec: unknown	Increase in summer months
	<i>Proportion biovolume Bacillariophytes</i> (cells L ⁻¹) (monthly mean and 90% CI) ¹¹	Jan: 0.90 (0.82, 0.99) Feb: 0.82 (0.66, 0.98) March: 0.79 (0.69, 0.89) April: 0.45 (0.34, 0.55) May: 0.44 (0.18, 0.71)	Increase

		June: 0.08 (0.03, 0.13) July: 0.01 (0.00, 0.02) Aug: 0.05 (0.03, 0.06) Sept: 0.08 (0.02, 0.14) Oct/Nov: 0.02 (0.01, 0.03) Dec: unknown	
	<i>Proportion biovolume other Divisions</i> (cells L ⁻¹) (monthly mean and 90% CI) ¹¹	Jan: 0.01 (0.00, 0.04) Feb: 0.00 March: 0.01 (0.00, 0.01) April: 0.03 (0.01, 0.05) May: 0.01 (0.01, 0.02) June: 0.01 (0.00, 0.01) July: 0.01 (0.00, 0.01) Aug: 0.01 (0.00, 0.02) Sept: 0.00 Oct/Nov: 0.00 Dec: unknown	Increase
	Division Based Diversity		
	<i>Proportion Cyanophyte Taxa</i> (seasonal mean and 90% CI)	Winter Spring: Summer: Autumn:	Decrease
	<i>Proportion Chlorophyte Taxa</i> (seasonal monthly and 90% CI)	Winter Spring: Summer: Autumn:	Increase
	<i>Proportion Bacillariophyte Taxa</i> (seasonal mean and 90% CI)	Winter Spring: Summer: Autumn:	Increase
	<i>Proportion other Divisions Taxa</i> (seasonal mean and 90% CI)	Winter Spring: Summer: Autumn:	Increase

Zooplankton	Assemblage Level Body Size		
	<i>Length</i> (mm) (seasonal mean and 90% CI) ⁵	Winter: 0.66 (0.38, 0.95) ¹ Spring: 0.68 (0.60, 0.75) Summer: 0.77 (0.72, 0.82) Autumn: 0.89 (0.72, 1.06)	Increase
	<i>CV length</i> ⁵ (seasonal)	Winter: 0.55 Spring: 0.30 Summer: 0.15 Autumn: 0.18	Decrease
	<i>Body mass</i> (mg) (seasonal mean and 90% CI)	TBD ¹⁰	Increase
	<i>Biovolume</i> (mm ³) (seasonal mean and 90% CI)	TBD ¹⁰	Increase
	Assemblage Level Production		
	<i>Biomass</i> (mg L ⁻¹) (seasonal mean and 90% CI)	Winter: Spring: Summer: Autumn:	Increase
	<i>Biomass</i> (mg L ⁻¹) <i>CV</i> (seasonal)	Winter: Spring: Summer: Autumn:	Decrease
	Assemblage Level Growth/Reproduction		
	<i>Potential Growth Rate</i> (seasonal mean and 90% CI)	TBD from Literature ⁴	
	<i>Reproduction Type/Frequency</i> (seasonal mean and 90% CI)	TBD from Literature ⁴	
	<i>Offspring Size/Number</i> (seasonal mean and 90% CI)	TBD from Literature ⁴	
	Assemblage Level Consumption		
	<i>Clearance Rate</i> (seasonal mean and 90% CI)	TBD from Literature ⁴	
	<i>Food Size Range</i> (seasonal mean and 90% CI)	TBD from Literature ⁴	

	<i>Sloppy Feeding</i> (seasonal mean and 90% CI)	TBD from Literature ⁴	
	Assemblage Level Predator Avoidance		
	<i>Vertical Migration</i> (seasonal mean and 90% CI)	TBD from Literature ⁴	Decrease
	<i>Escape Response</i> (seasonal mean and 90% CI)	TBD from Literature ⁴	
	<i>Transparency</i> (seasonal mean and 90% CI)	TBD from Literature ⁴	Decrease
	<i>Cyclomorphosis/Defense</i> (seasonal mean and 90% CI)	TBD from Literature ⁴	Decrease
	Assemblage Level Waste/Loss		
	<i>Egestion Rate (C, N, P)</i> (seasonal mean and 90% CI)	TBD from Literature ⁴	
	<i>Fecal Pellet Sedimentation Rate (C, N, P)</i> (seasonal mean and 90% CI)	TBD from Literature ⁴	
	<i>Excretion Rate (NH₄, PO₄)</i> (seasonal mean and 90% CI)	TBD from Literature ⁴	
	Assemblage Level Metabolism		
	<i>Respiration Rate</i> (seasonal mean and 90% CI)	TBD from Literature ⁴	
	<i>Digestion</i> (seasonal mean and 90% CI)	TBD from Literature ⁴	
	<i>Assimilation</i> (seasonal mean and 90% CI)	TBD from Literature ⁴	
	Diversity ²		
	<i>Taxa Richness</i> (annual)	6.79 (6.30, 7.29)	Increase
	<i>Taxa Evenness</i> (annual)	0.59 (0.55, 0.64)	Increase/Decrease ¹⁷
	<i>ENT</i> (annual)	3.18 (2.95, 3.40)	Increase/Decrease ¹⁷
	Group Relative Abundance		

	<i>Proportion Rotifera</i> (seasonal)	Winter: 0.14 (0.05; 0.22) Spring: 0.13 (0.09; 0.17) Summer: 0.10 (.06; 0.14) Autumn: 0.03 (0.00; 0.06)	
	<i>Proportion Rotifera CV</i> (seasonal)	Winter: 0.99 Spring: 1.02 Summer: 1.00 Autumn: 1.62	Decrease
	<i>Proportion Cladocera</i> (seasonal)	Winter: 0.27 (0.24; 0.30) Spring: 0.33 (0.30; 0.36) Summer: 0.35 (0.31; 0.39) Autumn: 0.34 (0.29; 0.38)	
	<i>Proportion Cladocera CV</i> (seasonal)	Winter: 0.20 Spring: 0.33 Summer: 0.30 Autumn: 0.30	Decrease
	<i>Proportion Calanoida</i> (seasonal)	Winter: 0.15 (0.08; 0.22) Spring: 0.10 (0.08; 0.13) Summer: 0.05 (0.02; 0.07) Autumn: 0.12 (0.04; 0.21)	
	<i>Proportion Calanoida CV</i> (seasonal)	Winter: 0.73 Spring: 0.91 Summer: 1.47 Autumn: 0.84	Decrease
	<i>Proportion Cyclopoida</i> (seasonal)	Winter: 0.22 (0.16; 0.28) Spring: 0.28 (0.23; 0.30) Summer: 0.32 (0.25; 0.39) Autumn: 0.29 (0.12; 0.45)	
	<i>Proportion Cyclopoida CV</i> (seasonal)	Winter: 0.42 Spring: 0.29 Summer: 0.58 Autumn: 0.70	Decrease
	<i>Proportion Harpacticoida</i> (seasonal)	Winter: 0.03 (0.00; 0.06) Spring: 0.01 (0.00; 0.01) Summer: 0.02 (0.00; 0.03)	

		Autumn: 0.05 (0.00; 0.10)	
	<i>Proportion Harpacticoida CV</i> (seasonal)	Winter: 1.61 Spring: 4.1 Summer: 2.15 Autumn: 1.16	Decrease
	<i>Proportion Daphnia sp.</i> (seasonal)	Winter: 0.19 (0.13; 0.24) Spring: 0.16 (0.14; 0.18) Summer: 0.17 (0.14; 0.19) Autumn: 0.18 (0.10; 0.26)	Increase
	<i>Proportion Daphnia sp. CV</i> (seasonal)	Winter: 0.49 Spring: 0.49 Summer: 0.43 Autumn: 0.55	Decrease
	Zooplankton-phytoplankton relationships		
	<i>Z:P ratio</i> (zooplankton biomass to phytoplankton biomass) (seasonal mean and 90% CI)	Winter: Spring: Summer: Autumn:	Increase
Non-Molluscan Benthic Invertebrates	Diversity		
	<i>Taxa Richness</i> (seasonal)	Winter: Spring: Summer: Autumn:	Increase
	<i>Taxa Evenness</i> (seasonal)	Winter: Spring: Summer: Autumn:	Increase
	<i>ENT</i> (seasonal)	Winter: Spring: Summer: Autumn:	Increase
	Production		
	<i>Total biomass</i> (mg dry weight m ⁻²)	Winter: na	Increase

	(seasonal)	Spring: na Summer: 10,546 () Autumn: 10,961 ()	
	<i>Total biomass CV</i> (seasonal)	Winter: na Spring: na Summer: 0.84 Autumn: 0.89	Decrease
	<i>Chironominae biomass</i> (mg dry weight m ⁻²) (seasonal)	Winter: na Spring: na Summer: 3,304 () Autumn: 8,827 ()	Increase
	<i>Chironominae biomass CV</i> (seasonal)	Winter: na Spring: na Summer: 1.23 Autumn: 1.01	Decrease
	<i>Tanypodinae biomass</i> (mg dry weight m ⁻²) (seasonal)	Winter: na Spring: na Summer: 6975 () Autumn: 1372 ()	Increase
	<i>Tanypodinae biomass CV</i> (seasonal)	Winter: na Spring: na Summer: 1.23 Autumn: 1.03	Decrease
	<i>Oligochaete biomass</i> (mg dry weight m ⁻²) (seasonal)	Winter: na Spring: na Summer: 267 () Autumn: 761 ()	Increase
	<i>Oligochaete biomass CV</i>	Winter: na Spring: na Summer: 1.28 Autumn: 0.66	Decrease
	<i>Corixid biomass</i> (mg dry weight m ⁻²) (seasonal)	Winter: na Spring: na Summer: na Autumn: na	Increase

	<i>Corixid biomass CV</i> (seasonal)	Winter: na Spring: na Summer: na Autumn: na	Decrease
Mollusks	Diversity ¹⁶		
	<i>Native gastropod richness</i>	Autumn: 2	Increase
	<i>Invasive gastropod richness</i>	Autumn: 0	Maintain
	<i>Pulmonate richness</i>	Autumn: 2	Increase
	<i>Non-pulmonate richness</i>	Autumn: 0	Increase
	<i>Native bivalve richness</i>	Autumn: 0	Increase
	<i>Invasive bivalve richness</i>	Autumn: 1	Decrease
Fishes	Condition		
	<i>Biological Condition Index³</i> (seasonal)	TBD	Increase
	Diversity		
	<i>Proportion planktivore taxa</i> (yearly)	TBD ³	Decrease
	<i>Proportion piscivore taxa</i> (yearly)	TBD ³	Increase
	<i>Proportion benthic taxa</i> (yearly)	TBD ³	Decrease
	<i>Proportion invasive taxa</i> (yearly)	TBD ³	Decrease
Submerged Aquatic Vegetation	Abundance ⁷		
	<i>Proportion substrate cover</i> (yearly)	TBD	Increase
	Diversity ⁷		
	<i>Taxa Richness</i> (yearly)	TBD	Increase

¹ More data needed to reduce variability estimates

² Taxa diversity metrics S, E, and ENT use annual value because there were no significant differences between seasons using bootstrapped (N = 500) mean and 90% CIs.

- ³ Consultation with UDWR fisheries biologists needed for metric values
- ⁴ Metric values need to be determined from literature and then confirmed with Utah Lake empirical values
- ⁵ Zooplankton body lengths and CV metrics derived from Richards 2019 literature review and need to be confirmed with empirical data from future samples.
- ⁶ Further refinements and justification of seasonal body length sub- metrics are in Appendix 1.
- ⁷ Collaborative research needs to be initiated asap to estimate SAV metrics at all three locations
- ⁸ Temperature data to be acquired from UDWQ Utah Lake database
- ⁹ Data to be compiled from WFWQC and UDWQ Utah Lake database
- ¹⁰ To be determined empirically
- ¹¹ Monthly means and 90% CIs based on lake wide values for 2017 only. Need to compile data from WFWQC and UDWQ and re analyzed
- ¹² A DNA based measure of toxin level detection is suggested
- ¹³ ENT = effective number of taxa = exponentiated Shannon Diversity Index (H) (Jost 2006; Chao et al. 2010)
- ¹⁵ Phytoplankton taxa diversity metric means and 90% CI values will be derived from UDWQ database from Rushforth Phycology and after taxonomic status and synonymies are accounted and adjusted for (see Richards 2018b for taxonomic updates).
- ¹⁶ Utah Lake mollusk diversity metric values derived from Richards 2017 and unpublished data
- ¹⁷ Evenness and ENT may either increase or decrease with changes in conditions and need to be evaluated based on baseline values (Cao and Hawkins 2019)

Discussion

This is a provisional MIBI illustrating metrics specific to Provo Bay but will include Goshen Bay and Utah Lake proper metric values (presently being populated). More literature review, data compilation, and consultation with fisheries biologist and other Utah Lake researchers will be essential to modify, evaluate, and complete it. Once metric values are populated for each location, researchers and managers will be able to confidently evaluate changes to the biological and ecological condition of Utah Lake as opposed to depending on professional judgment or highly simplified indices comprised of only a few easy to measure generalized metrics. Utah Lake is a unique body of water in the western USA with a remnant unique native biota that deserves our best efforts to assess and then monitor its present state. It is our responsibility to maintain and improve Utah Lake's condition and protect its biological and ecological integrity, including its beneficial uses for this and future generations.

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DRAFT

Spatial and Temporal Variability of Zooplankton Body Lengths in Utah Lake

Technical Memo

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To:
Wasatch Front Water Quality, Council, Salt Lake City, UT

April 16, 2019

Introduction

Body lengths of zooplankton are a widely used metric for evaluating conditions in lakes. Zooplankton body lengths typically decrease with increased; temperature, eutrophication, DIN:DP ratio, pollutants, fish predation, and interactions between these factors (Havens and Hanazato 1993; Havens et al. 2015; Havens and Beaver 2011; Trommer and Stibor 2019; Barnett and Beisner 2007; Gliwicz and Lampert 1990; Richman and Dodson 1983; Gillooly and Stanley 2000; others). Body size is extremely important in algal bloom dynamics because larger sized zooplankton are often better at feeding on larger strands of

algal particularly cyanobacteria, therefore the loss of larger sized zooplankton may result in cyanoHABs (Carpenter and Kitchell 1988; Caroni 2010; Jeppesen et al. 2011; Attayde and Bozelli 1998 ; Carpenter et al. 1985; Jeppeson et al 2000; Jeppesen et al 2003; Lamper et al 1986; Gannon and Stemberger 1978; others). Richards (2019a) is developing a multi-metric index of biological integrity to monitor water quality in Utah Lake and Farmington Bay using zooplankton body length as an important metric.

Even though water quality managers are very concerned about cyanoHABs in Utah Lake, there have been no estimates of zooplankton body length spatial and temporal patterns in in Utah Lake, despite their well-known importance as a metric for monitoring water quality. We have remedied this situation by conducting statistical analyses on spatial and temporal patterns of zooplankton body lengths in Utah Lake with results presented in this memo and Richards (2019a in progress).

Methods

We used zooplankton data collected from Wasatch Front Water Quality Council and OreoHelix Consulting over the last several years as was presented in Richards 2019b. We then determined sample weighted zooplankton body lengths based on abundance data and lengths reported in Richards (2109b). We then conducted best-fit regression analyses, marginal analyses, and predicted mean and 95% CI body lengths for each location and month in Utah Lake.

Results

Zooplankton body lengths significantly varied spatially and temporally in Utah Lake with a relatively small to medium- small mean length = 0.85 mm (std. dev. = 0.19). Overall, body lengths were smallest from March through August and mostly significantly smaller than the mean (Figure 37). Body lengths were also significantly smaller than average in Provo Bay, Lindon Marina, and Utah Lake State Park Marina and significantly larger than average in the mid sections of the lake (labeled LP) (Figure 38).

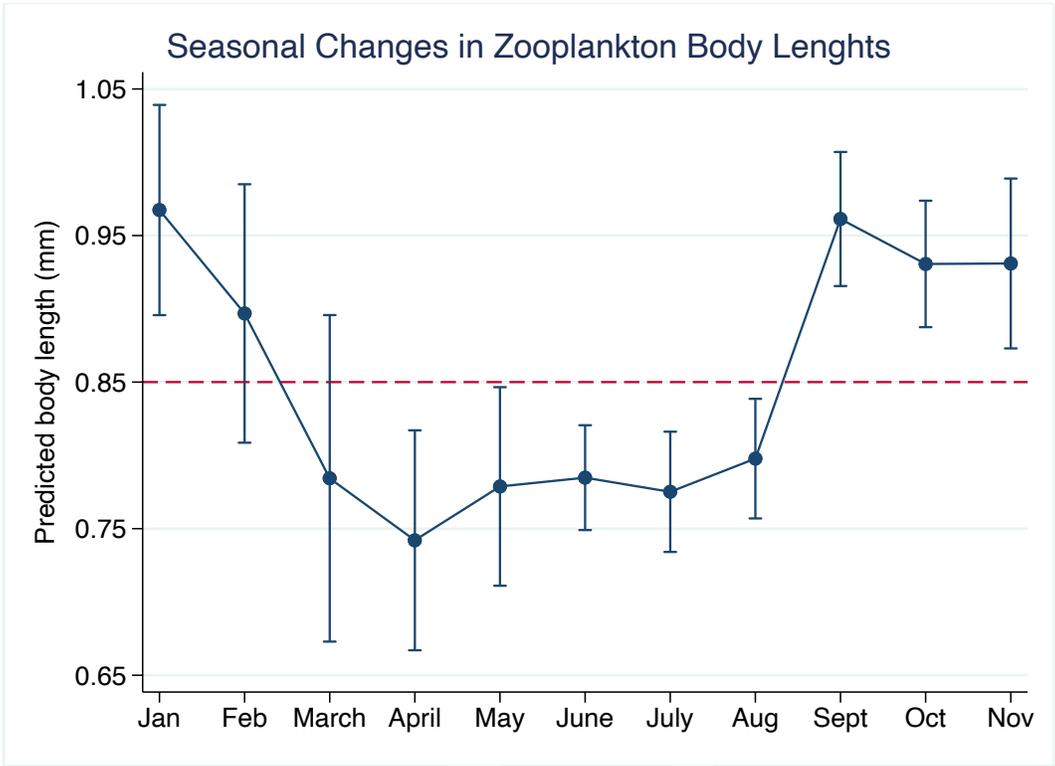


Figure 37. Predicted mean and 95% CIs for zooplankton body lengths from best fit regression models in Utah Lake seasonally.

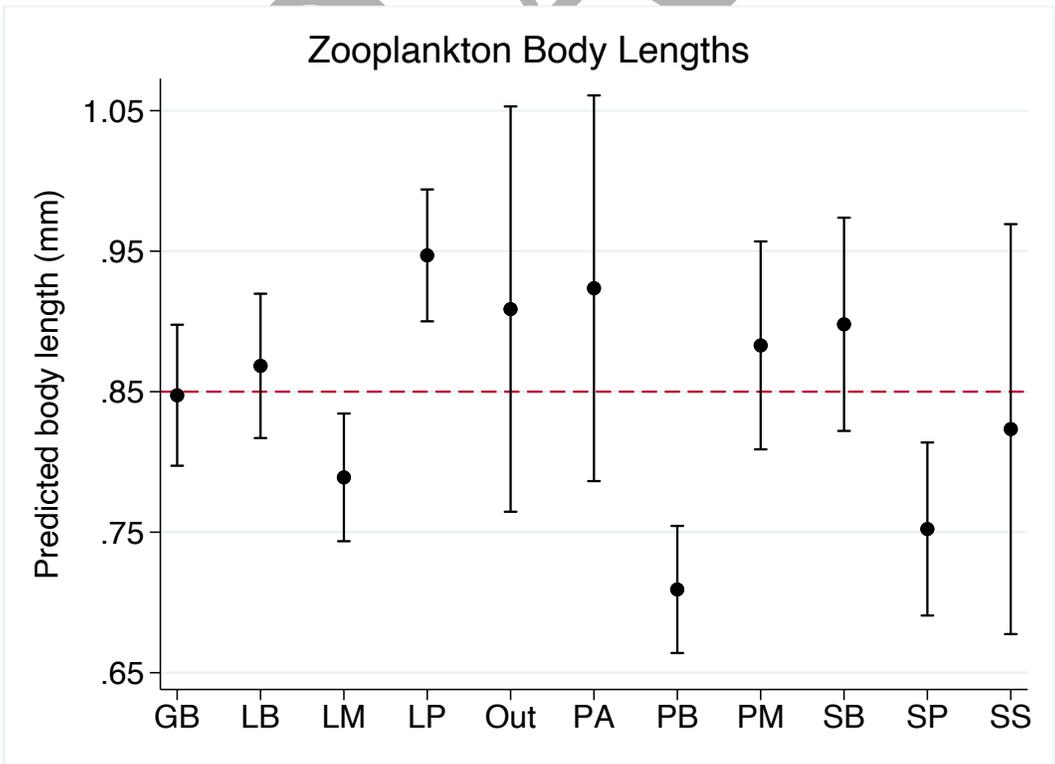


Figure 38. Predicted mean and 95% CIs for zooplankton body lengths from best fit regression models in Utah Lake by location.

Zooplankton body lengths were relatively uniformly small in Lindon Marina throughout the seasons (Figure 39) but significantly smaller than average in Provo Bay in March and April (Figure 40) and smaller than average from May to August at Utah Lake State Park Marina (Figure 41).

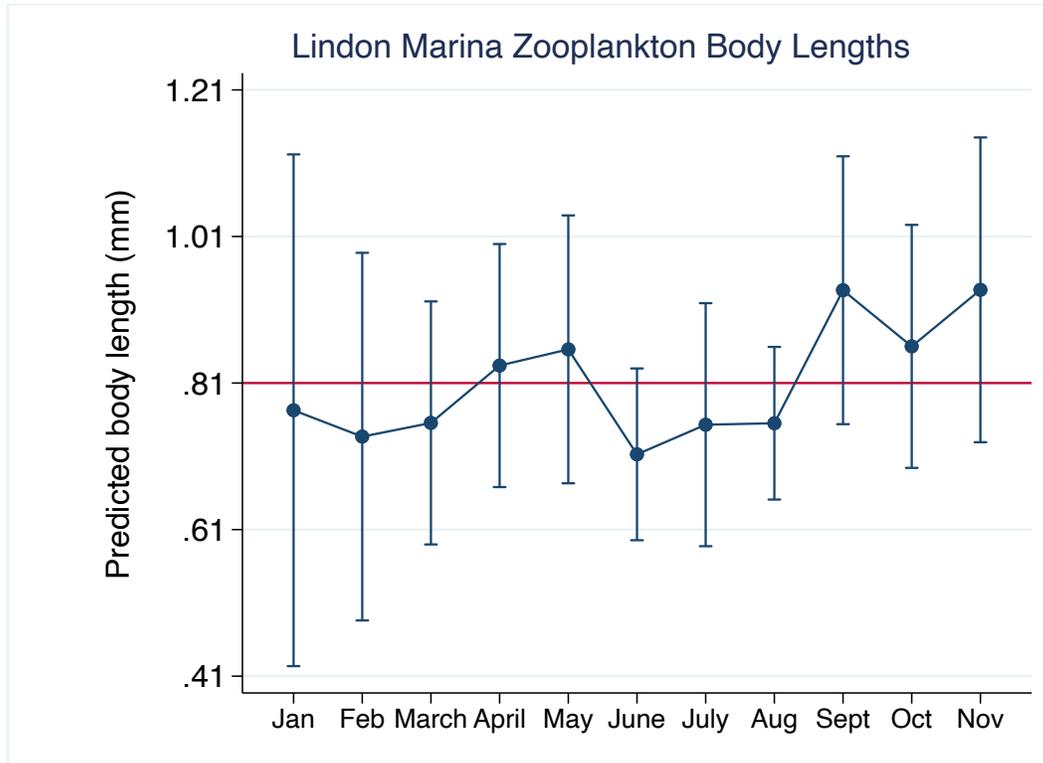


Figure 39. Predicted mean and 95% CIs for zooplankton body lengths from best fit regression models in Lindon Marina.

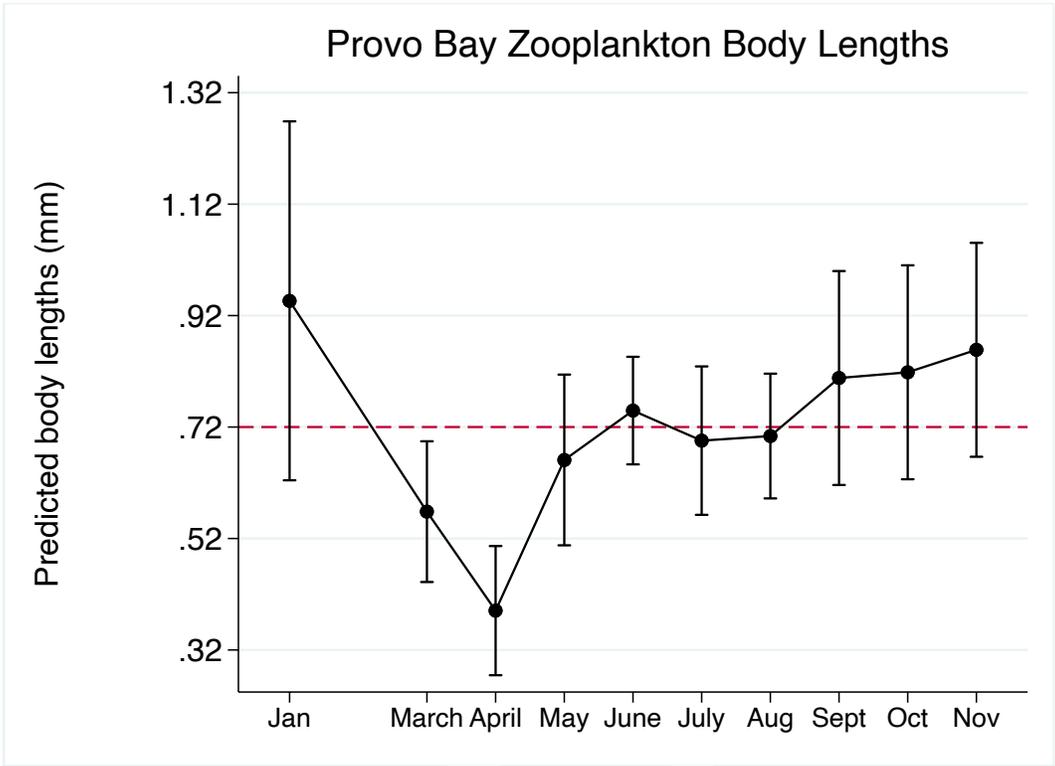


Figure 40. Predicted mean and 95% CIs for zooplankton body lengths from best fit regression models in Provo Bay.

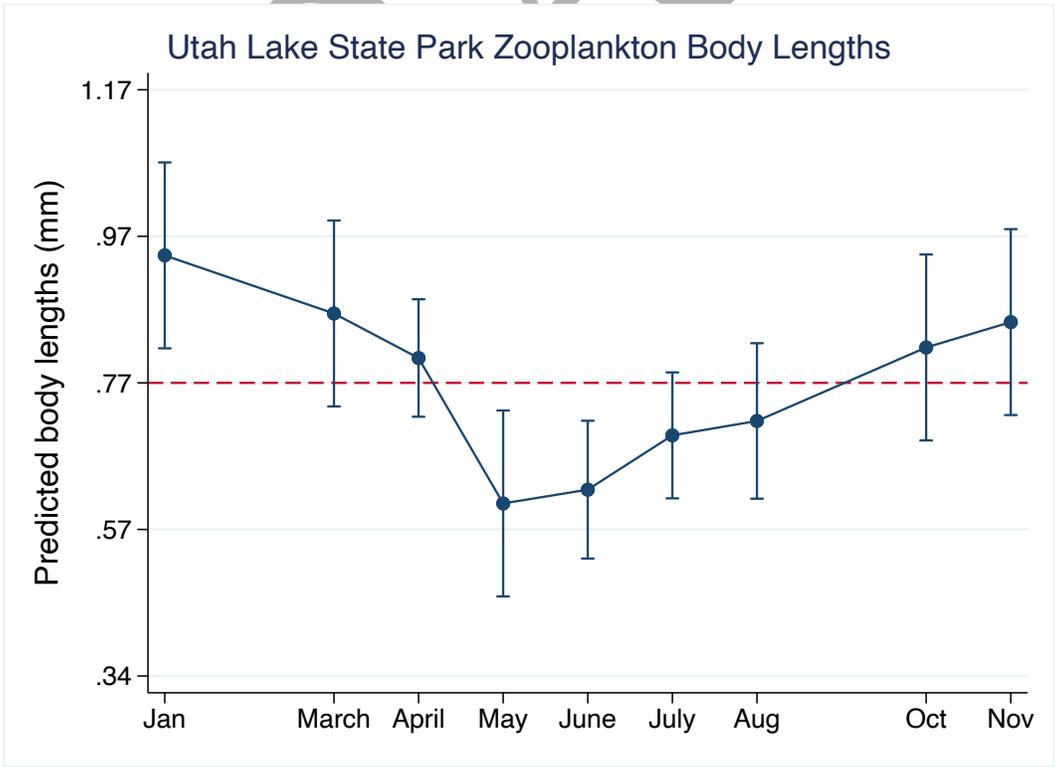


Figure 41. Predicted mean and 95% CIs for zooplankton body lengths from best fit regression models in Utah Lake State Park marina.

Discussion

Results presented in this memo show that zooplankton body lengths were relatively small compared to other temperate lakes (see References) and are a highly useful metric for monitoring water quality in Utah Lake. Body lengths significantly varied spatially and temporally. We suggest that other than the typical seasonal progression of zooplankton assemblages, was also due to the effects of planktivorous fish predation. Body lengths were typically smallest from spring through summer when fish are most actively feeding and when water clarity was often the best for visual planktivore feeding. The reason zooplankton body size was smallest in Provo Bay compared to most other sites was likely because this bay has the greatest planktivorous fish densities in the entire lake. Planktivorous fish prefer larger sized zooplankton.

2019 was a relatively high-water year in Utah Lake that resulted in a highly successful carp spawn and subsequently a boom in YOY juvenile carp production (Richards personal observation). Planktivorous juvenile carp require substantially more energy/individual body mass during growth than do larger adult carp maintaining body mass, thus zooplankton consumption rates should be higher in 2019 and subsequent years until the 2019 carp age class reaches adulthood than in previous less successful spawn years. Other planktivorous fish in the lake may have also produced more YOY than previous years. This phenomenon may alter zooplankton biomass and assemblage structure and requires careful monitoring.

Increased body sizes in late summer reported here were possibly due to increased algal bloom induced turbidity in mid-summer, which reduced visual ability of planktivores to find larger zooplankton, and subsequently allowed larger zooplankton such as *Daphnia* sp. to be able to consume larger sized phytoplankton, which then was partially responsible for decreased algal blooms in late summer/early autumn (Carpenter 1988; Chislock et al. 2019; Richards 2019b). Other causal factors for reduced zooplankton body size in Utah Lake are under investigation, including relationships between zooplankton body size and phytoplankton traits, pollution effects, and temperature effects. Results of these analyses will be directly applicable to the MIBI that is being produced by Richards (2019a).

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DRAFT

Appendix 4. Laboratory Observations Regarding Identifications and likely Synonymies among Zooplankton from Utah Lake (2017-2018)



Laboratory Observations Regarding Identifications and likely Synonymies among Zooplankton from Utah Lake (2017-2018)

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Prepared for:
Oreohelix Consulting AND Wasatch Front Water Quality Council

March 29, 2019



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Acknowledgements:

Special thanks to Theron Miller and David Richards for realizing there was a problem in taxonomy and providing us the opportunity to address it.

Cover Photo:

Daphnia galeata mendotae from Utah Lake. Although the specimen has a pointy, extended, and partially re-curved head, the clearly visible ocellus, positioned midway down the head, clearly separates this (and several other) species from *Daphnia retrocurva*—the only previously reported *Daphnia* from this “group” (see report) in Utah Lake.

“The beginning of wisdom is to call things by their proper name.”

— Confucius

Laboratory Observations Regarding Identifications and likely Synonymies among Zooplankton from Utah Lake

March 29, 2019

Brett D. Marshall, MSc.

River Continuum Concepts, Inc.

Abstract

We identified 106 zooplankton samples collected from Utah Lake, UT by David Richards (OreoHelix Consulting). There were some conflicts among the historic taxa lists provided by two morphological taxonomists and genetic taxonomy. Here we detail our notes to provide insights on how to best optimize the operational taxonomic units for working with data describing zooplankton assemblages of Utah Lake, including the most defensible method combining of data from multiple sources.

Suggested citation:

Marshall, B. D. 2019. Laboratory Observations Regarding Identifications and likely Synonymies among Zooplankton from Utah Lake. Prepared for OreoHelix Consulting, and Wasatch Front Water Quality Council, Salt Lake City, UT 84114.

Introduction

One of the great challenges of ecological science is minimizing unwanted sources of uncertainty so that our studies can be used to elucidate the causes of variation and covariation within and among biological populations interacting with their abiotic environment. The uncertainty of observations is exacerbated when two (or more) different taxonomists yield different species lists; did the species composition change between sampling events or are the differences simply artefacts of different methods in sample processing?

In this report, we provide information critical to the effective and defensible analysis of zooplankton data from Utah Lake. The species lists from four data-sources are in conflict and this report recommends which taxa should be synonymized and which should be retained as operational taxonomic units based on practical applications of microscopy, laboratory processing, and practical taxonomic effort. The results balance the current taxonomic literature vs. practical aspects of taxonomy and ecology to ensure the same taxonomic standards can be effectively used for processing samples in future years.

Methods

Data sources were from Richards and Miller (2017), Richards (2017a), and Richards (2017b). Richards 2016 used data provided by Larry Gray (the Gray data hereafter) and were sometimes in conflict with the taxa list provided by another taxonomist from LimnoPro (Richards 2017a (the LimnoPro dataset hereafter)). Both of these sources were sometimes in conflict with the taxa list comprised by the analysis of DNA Barcodes (Richards 2017b). Our focus was to ensure our taxonomic laboratory effort reconciled these sources and to understand why the differences occurred so that synonymies among the datasets are credibly and responsibly addressed. Thus, the taxonomy involved multiple keys and sometimes primary literature on taxonomic revisions and species description—much more effort than a routine bench taxonomy project. The Results Section of this report details some likely sources of variation in taxonomy referring to data from Gray, LP (LimnoPro), DNA, and RCC (our current accompanying dataset). The first data source, compiled by Gray, consisted of 4 copepod taxa, 10 cladoceran taxa, and 3 rotifer taxa (Table 1). Gray’s dataset was the basis for the LimnoPro dataset (Table 2), which consisted of 4 copepod taxa, 11 cladoceran taxa, and 3 rotifer taxa. The taxa lists were similar to each other but may not have been completely independent. Both datasets deviated considerably from the DNA dataset (Table 3) which consisted of 6 copepod taxa, 23 cladoceran taxa, 3 rotifer taxa. However, it should be noted that only a sub-sample of samples sent to LimnoPro were included in the taxa list.

Table 1. Data from the Gray dataset (Richards 2016) included the following taxa.

Order	Family	Gray Taxa
Cyclopoida	Cyclopidae	<i>Eucyclops agilis</i>
Cyclopoida	Cyclopidae	<i>Acanthocyclops robustus</i>
Calanoida	Diaptomidae	<i>Leptodiaptomus sicilis</i>
Harpacticoida	Canthocamptidae	<i>Attheyella</i>
Cladocera	Daphniidae	<i>Ceriodaphnia quadrangula</i>
Cladocera	Daphniidae	<i>Daphnia pulex</i>
Cladocera	Daphniidae	<i>Daphnia retrocurva</i>
Cladocera	Chydoridae	<i>Pleuroxus striatus</i>
Cladocera	Chydoridae	<i>Chydorus sphaericus</i>
Cladocera	Chydoridae	<i>Leydigia leydigi</i>
Cladocera	Sididae	<i>Diaphanosoma</i> sp.
Cladocera	Bosminiidae	<i>Bosmina longirostris</i>
Cladocera	Ilyocryptidae	<i>Ilyocryptus</i> sp.
Cladocera	Leptodoridae	<i>Leptodora kindti</i>

Plioma	Brachionidae	<i>Brachionus</i>
Plioma	Brachionidae	<i>Keratella</i>
Plioma	Asplanchnidae	<i>Asplanchna</i>

Table 2. Data from the LimnoPro dataset (Richards 2017a) included the following taxa.

Order	Family	LimnoPro Taxa
Cyclopoida	Cyclopidae	<i>Microcyclops rubellus</i>
Cyclopoida	Cyclopidae	<i>Acanthocyclops robustus</i>
Calanoida	Diaptomidae	<i>Leptodiaptomus sicilis</i>
<u>Harpacticoida</u>	<u>Canthocamptidae</u>	<u><i>Cletocamptus</i></u> sp.
Cladocera	Daphniidae	<i>Ceriodaphnia</i> <i>quadrangula</i>
Cladocera	Daphniidae	<i>Daphnia pulex</i>
Cladocera	Daphniidae	<i>Daphnia retrocurva</i>
Cladocera	Daphniidae	<i>Simocephalus vetulus</i>
Cladocera	Chydoridae	<i>Pleuroxus aduncus</i>
Cladocera	Chydoridae	<i>Chydorus sphaericus</i>
Cladocera	Chydoridae	<i>Leydigia leydigi</i>
Cladocera	Moinidae	<i>Moina macrocarpa</i>
Cladocera	Sididae	<i>Diaphanosoma</i> <i>brachyurum</i>
Cladocera	Bosminiidae	<i>Bosmina longirostris</i>
Cladocera	Leptodoridae	<i>Leptodora kindti</i>
Plioma	Brachionidae	<i>Brachionus plicatilis</i>
Plioma	Brachionidae	<i>Brachionus calyciflorus</i>
Plioma	Asplanchnidae	<i>Asplanchna</i> sp.

Table 3. Data from the DNA dataset (Richards 2017b) included the following taxa. The list was modified to only include zooplankton.

CLASS	Family	Species
c__Branchiopoda	f__Artemiidae	s__Artemia franciscana
c__Branchiopoda	f__Bosminidae	s__Bosmina sp. BOLD:AAV0686
c__Branchiopoda	f__Bosminidae	s__Bosmina lieperi
c__Branchiopoda	f__Chydoridae	s__Chydorus brevilabris
c__Branchiopoda	f__Chydoridae	s__Leydigia lousi
c__Branchiopoda	f__Daphniidae	s__Daphnia magna
c__Branchiopoda	f__Daphniidae	s__Ceriodaphnia dubia
c__Branchiopoda	f__Daphniidae	s__Simocephalus sp. BOLD:AAB1538
c__Branchiopoda	f__Daphniidae	s__Simocephalus cf. punctatus 3 NA
c__Branchiopoda	f__Daphniidae	s__Daphnia galeata
c__Branchiopoda	f__Daphniidae	s__Ceriodaphnia cf. acanthina AS29b3
c__Branchiopoda	f__Daphniidae	s__Ceriodaphnia cf. laticaudata ZMXII-611
c__Branchiopoda	f__Daphniidae	s__Ceriodaphnia sp. BOLD:AAB6934
c__Branchiopoda	f__Daphniidae	s__Simocephalus cf. punctatus 1 NA
c__Branchiopoda	f__Daphniidae	s__Ceriodaphnia cf. laticaudata BOLD:AAB5055
c__Branchiopoda	f__Daphniidae	s__Ceriodaphnia sp. BOLD:AAB5055
c__Branchiopoda	f__Daphniidae	s__Ceriodaphnia sp. HE-627
c__Branchiopoda	f__Daphniidae	s__Ceriodaphnia cf. laticaudata AS43e4
c__Branchiopoda	f__Daphniidae	s__Ceriodaphnia cf. laticaudata HE-615.1
c__Branchiopoda	f__Daphniidae	s__Daphnia ambigua
c__Branchiopoda	f__Daphniidae	s__Simocephalus punctatus
c__Branchiopoda	f__Moinidae	s__Moina mongolica
c__Branchiopoda	f__Sididae	s__Diaphanosoma cf. heberti BOLD:AAB9853
c__Branchiopoda	f__Sididae	s__Latonopsis cf. australis AS40c3
c__Maxillopoda	f__Cyclopidae	s__Acanthocyclops americanus
c__Maxillopoda	f__Cyclopidae	s__Cyclopidae sp. BOLD:AAG9780
c__Maxillopoda	f__Diaptomidae	s__Leptodiaptomus cf. sicilis JAAR-2015
c__Maxillopoda	f__None	s__Calanoida sp. HE-071
c__Maxillopoda	f__None	s__Cyclopoida sp. HE-241

c__Maxillopoda	f__None	s__Calanoida sp. HE-073
c__Monogononta	f__Brachionidae	s__Brachionus sp. Almenara
c__Monogononta	f__Brachionidae	s__Brachionus plicatilis group sp. MEG-2012
c__Monogononta	f__Brachionidae	s__Brachionus urceolaris

Results

General

Our analysis of 106 zooplankton samples from Utah Lake included 7 copepod taxa, 16 cladoceran taxa, and 4 rotifer taxa—in addition to some aberrant species such as insects (Table 4). This is the RCC dataset. Our dataset found more taxa than either of the other datasets because, we were driven to understand and reconcile the differences among the other sites; we made many slides, used multiple keys, and where keys failed went to primary taxonomic literature, including taxonomic revisions and species descriptions (Table 4). Thus, the effort was above and beyond routine standard bench taxonomy. However, the effort will allow for better¹⁷ routine standard bench taxonomy in the future. Table 4. Taxa list from the analysis by the River Continuum Concepts (RCC dataset hereafter) from samples collected in 2017-2018, and processed in late 2018.

Order	Family	RCC Taxa
Cyclopoida	Cyclopidae	<i>Microcyclops rubellus</i>
Cyclopoida	Cyclopidae	<i>Acanthocyclops americanus</i>
Calanoida	Diaptomidae	<i>Leptodiaptomus sicilis</i> female
Calanoida	Diaptomidae	<i>Leptodiaptomus sicilis</i> male
Harpacticoida	Harpacticoida	Harpacticoida
Harpacticoida	Canthocamptidae	Canthocamptidae
Harpacticoida	Canthocamptidae	<i>Attheyella</i> sp.
Harpacticoida	Laophontidae	<i>Onychocamptus mohammed</i>
Cladocera	Daphniidae	<i>Ceriodaphnia</i> sp.
Cladocera	Daphniidae	<i>Ceriodaphnia dubia</i>
Cladocera	Daphniidae	<i>Daphnia</i> sp.
Cladocera	Daphniidae	<i>Daphnia mendotae</i>
Cladocera	Daphniidae	<i>Daphnia retrocurva</i>

¹⁷ Better: more resolute, more consistent, more reproducible, and more efficient.

Cladocera	Daphniidae	<i>Daphnia ambigua</i>
Cladocera	Daphniidae	<i>Daphnia magna</i>
Cladocera	Daphniidae	<i>Simocephalus</i> sp.
Cladocera	Chydoridae	<i>Leydigia louisi</i>
Cladocera	Chydoridae	<i>Alona setulosa</i>
Cladocera	Moinidae	<i>Moina micrura</i>
Cladocera	Sididae	<i>Diaphanosoma</i> cf. <i>heberti</i>
Cladocera	Bosminiidae	<i>Bosmina longirostris</i> complex
Cladocera	Macrothricidae	<i>Macrothrix</i> sp.
Cladocera	Ilyocryptidae	<i>Ilyocryptus</i> sp.
Cladocera	Leptodoridae	<i>Leptodora kindti</i>
Plioma	Brachionidae	<i>Brachionus calyciflorus</i>
Plioma	Brachionidae	<i>Brachionus variabilis</i>
Plioma	Brachionidae	<i>Brachionus</i> sp. Almenara
Plioma	Asplanchnidae	<i>Asplanchna</i> sp.

Copepoda

Cyclopoda: *Acanthocyclops*

The case of *Acanthocyclops* presents a significant conundrum in taxonomic synonymies for the Utah Lake zooplankton dataset. The Gray and LimnoPro datasets reported the presence of *A. robustus*, whereas the DNA assessment of samples from Utah Lake reported only *A. americanus*.

In our initial identifications of *Acanthocyclops* using keys of Thorpe and Rogers (2016), we agreed with datasets provided by Gray and by LimnoPro that the *Acanthocyclops* in Utah Lake appeared to be *A. robustus*. However, Thorpe and Rogers (2016) and several other sources (e.g., Haney et al. 2013) do not list *A. americanus* among the fauna of North America even though the worldwide consensus is that the species occurs here. The story of this species is an interesting mix of history and biology. In the late 1800s, the “*Acanthocyclops robustus* group” was comprised of three highly varied (among species) and variable (within species) species: *A. vernalis*, *A. robustus*, and *A. americanus*. *A. americanus* was newest of the three species and the only species originally described in the USA, whereas the other two species were described from Europe. In the 1920s, *A. americanus* was found in Europe and the whole idea of geographic isolation as the basis of taxonomy came into question. Over the next half century, systematists worked with an extensive network of subspecies designations of the *A. robustus* group, that ultimately culminated with Kiefer (1976) combining *A. americanus* and *A. robustus* into a single species— with *A. robustus* having seniority by a few months. Thus, all *A. americanus* became *A. robustus* and all records of *A. americanus* gradually disappeared. In the 1990s, a resurgence of ‘dividing’ taxonomy began as European biologists began

separating the species of the group again. Over time, the ecological differences among the species-complex had been noticed by biologists. This eventually led to Miracle et al. (2013) publishing their molecular-genetic assessment of the *Acanthocyclops robustus* group, along with the definitive physical characteristics to separate the species. This allowed major ecological differences among the species of the *A. robustus* group to be further elucidated. Miracle et al. (2013) report that these three species (*A. vernalis*, *A. robustus*, *A. americanus*) remain valid, while they synonymized several newer European species back into *A. americanus*.

For the study of zooplankton in Utah Lake, the differentiation of *A. robustus* from *A. americanus* is especially important because they usually occupy different ecological niches, with the former preferring a more littoral-zone habitat, and the later preferring pelagic/limnetic habitat (e.g., Miracle et al.2013). Thus, changes in the distribution, abundance, and production of these species would probably be important in describing the changing ecology of Utah Lake. Although the datasets of Gray and LimnoPro were not incorrect in their determination of *A. robustus* (because there are no published keys of North American fauna that include *A. americanus*), using the morphological characters defined by Miracle et al.

(2013) we were clearly able to associate these specimens to the currently recognized species of *A. americanus* and bring these dataset into the modern standards of taxonomy.

We believe the two names *A. americanus* and *A. robustus* should be synonyms in the current datasets from Utah Lake because (1) the DNA taxa-list was generated using subsamples of the samples processed by LimnoPro, but did not find any trace of *A. robustus*, and (2) because the limitations in the keys to North American fauna prevented LimnoPro and Gray from arriving at the determination of *A. americanus* (Table 5). I usually do not promote following nouveau taxonomy because changing names for the sake of changing names is detrimental to the long-term integrity of datasets. However, the taxonomy of this group has followed a long process over the last century and it also corresponds to differences in ecological niche that might be important for understanding the changing ecology of Utah Lake. Regardless of the name used, I believe that both names (*A. americanus* and *A. Robustus*) describe the same taxon—at least usually.

Table 5. Synonymies in the genus *Acanthocyclops*. ‘X’ marks determinations in each of the datasets considered. The final recommended taxon (O.T.U.) is in the bottom row highlighted.

Species	Gray	LimnoPro	DNA	RCC
<i>A. vernalis</i>				
<i>A. robustus</i>	X	X		
<i>A. americanus</i>			X	X

Recommend: <i>A. americanus</i>				
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Cyclopidae: *Eucyclops agilis*

Gray reported the species *Eucyclops agilis* from Utah Lake. This was not found in any other dataset.

However, the taxon is superficially similar to *Microcyclops rubellus* that was found by LimnoPro, RCC and DNA. Moreover, not only is the size and shape similar, the course antennal morphology (that would have been used under a dissecting scope to ‘morpho-type’ *Microcyclops rubellus* incorrectly as *Eucyclops agilis*) is similar for both species (Table 6).

Table 6. Synonymies in the genera *Eucyclops* and *Microcyclops*. ‘X’ marks determinations in each of the datasets considered. The final recommended taxon (O.T.U.) is in the bottom row highlighted

Species	Gray	LimnoPro	DNA	RCC
<i>Microcyclops rubellus</i>		X	X	X
<i>Eucyclops agilis</i>	X			
<i>Microcyclops rubellus</i>				

Calanoida: *Leptodiaptomus sicilis*

The only Calanoida among the Gray and LimnoPro datasets was the species *Leptodiaptomus sicilis*. The DNA dataset found *Leptodiaptomus sicilis* but also several hits for a vague order-level taxon “Calanoida sp.” The only Calanoida species we found was *Leptodiaptomus sicilis* (Table 6). We differentiated the sex of the species because different keys were used. This may ultimately be useful information.

Table 6. Synonymies in the genera *Eucyclops* and *Microcyclops*. ‘X’ marks determinations in each of the datasets considered. The final recommended taxon (O.T.U.) is in the bottom row highlighted

Species	Gray	LimnoPro	DNA	RCC

<i>Leptodiaptomus sicilis</i>	X	X	X	X
“Calanoida sp.”			X	
<i>Leptodiaptomus sicilis</i>				

Harpacticoid copepods

The harpacticoids were represented by only one taxon in the Gray dataset (*Attheyella* sp.), and only one taxon (and only a few individuals (~5/10,000)) in the LimnoPro dataset (*Cletocamptus* sp.). Although both datasets reported a different taxon, it would be presumptuous to assume that both taxonomists were looking at the same species and arrived at different names. These are uncommon constituents of plankton samples and the DNA dataset reported zero harpacticoids. In truth, the specimens were under represented to provide meaningful analysis because they are usually benthic-zone inhabitants. There are too few of them for conventional statistics, and the underlying assumption of multivariate statistics is that samples not containing a taxon are from habitats not suitable for that taxon; their occurrence is aberrant in plankton samples. Subsampling may further omit harpacticoid specimens from the taxa list. The most efficient way to deal with these aberrations is to either lump them up into a more inclusive taxon (e.g., Harpacticoida) or to exclude them from analysis all together. We favor inclusion in the dataset because their occurrence in plankton could indicate a benthic disturbance or ecological event. However, their use as such an indicator is probably best at the level of Harpacticoida. Incidentally, they were only identified as harpacticoida sp. in the DNA dataset also.

Cladocerans

Daphnia

The *Daphnia* were reported as from two groups of morphologically similar taxa. The groups were arbitrarily assigned as Group-1 and Group-2 to facilitate discussion of likely synonymies, which was important because we found some serious problems among the identities assigned to *Daphnia*—these problems are best addressed by dealing with different groups separately. Group-1 was comprised of *Daphnia* specimens with different degrees of expanded head capsule (i.e., ‘helmet’) and included *D. retrocurva*, *D. galeata*, and *D. mendotae*. Group-2 included taxa without as expanded of a helmet though occasionally a point of variable size occurred off the head. The Group-2

taxa were *D. pulex*, *D. magna*, and *D. ambigua*. Both groups had conflicts among the morphology datasets and the DNA dataset.

Daphnia: Group-1

Both the Gray and the LimnoPro datasets reported *Daphnia retrocurva* as the sole member of this group, whereas the DNA dataset reported *D. galeata* as the only member of this group. Most of the specimens processed in the RCC dataset were identified as *Daphnia mendotae*. *D. mendotae* is a synonym for *Daphnia galeata mendotae*. The North American subspecies *D. g. mendotae* is the sibling to the European *D. galeata galeata* (historically *D. galeata*). Thus, the specimens identified as *D. mendotae* in the RCC dataset is congruent with specimens called *D. galeata* in the DNA dataset, which recognized the eventual synonymy of *D. mendotae* and *D. galeata*, but not the subspecies differentiation of continental populations.

D. g. mendotae is a highly variable species, and there is some curvature in the head that superficially resembles that of underdeveloped *D. retrocurva*. However, finer examination of the head curvature and the presence of an ocellus that is clearly visible under higher magnification and on slide-mounted specimens assigns these specimens clearly to *D. g. mendotae*--not *D. retrocurva*. The *D. g. mendotae* are clearly separable from *D. retrocurva* even when the head is curved (or the *D. retrocurva* is under developed) by the presence of an ocellus below the compound eye, which is lacking in *D. retrocurva*. Unfortunately, the key of Thorpe and Rogers (2016) incorrectly suggest that *D. retrocurva* has an obvious ocellus, further confounding taxonomists using this key.

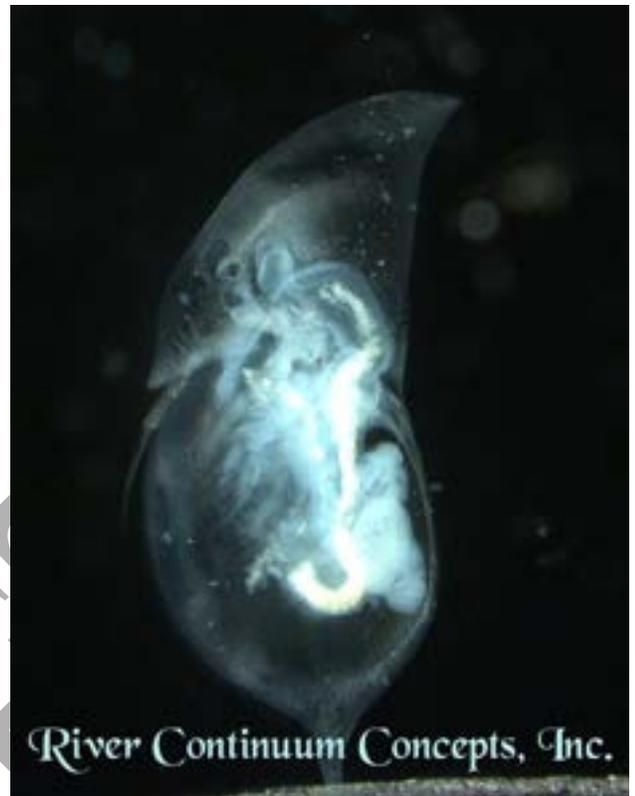


Figure 1. *Daphnia retrocurva* from Utah Lake.

Our impression is that most of the specimens identified by LimnoPro and Gray as *D. retrocurva* were probably mostly misidentified specimens of *D. g. mendotae*—the dominant *Daphnia* in our samples. However, we eventually found several true specimens *D. retrocurva* (Fig. 1) among the samples processed at RCC. Since these were less abundant in many samples that we processed, the species was probably omitted from the DNA dataset simply because of subsampling. Datasets from Gray and LimnoPro probably overestimated the abundance of *D. retrocurva* while omitting the occurrence of *D. g. mendotae* (Table 7).

For combining datasets, consider calling them *Daphnia* Group-1. Alternatively, accept the fact that *D. retrocurva* was uncommon, and call them all *D. galeata mendotae*—accepting the fact that an unknown (and variable) error increases the abundance of the taxon historically from (0-50%). To be clear, we would continue to differentiate *D. retrocurva* from *D. galeata mendotae* in future datasets, but to compare with historical datasets, investigators will need to group them.

Table 7. Synonymies in the genus *Daphnia* (Group-1). ‘X’ marks determinations in each of the datasets considered. The final recommended taxon (O.T.U.) is in the bottom row highlighted.

Species	Gray	LimnoPro	DNA	RCC
<i>D. retrocurva</i>	X	X		X

<i>D. galeata</i>			X	
<i>D. mendotae</i>				X
Recommend: <i>D. galeata mendotae</i> (<i>D. retrocurva</i> separate)				

Daphnia: Group-2

The datasets from Gray and LimnoPro both identified the Group-2 *Daphnia* species as *Daphnia pulex* whereas the DNA dataset found no *D. pulex*, but rather *D. magna* and *D. ambigua* (Table 8). Although the separation of these species should be routine, the keys of Balcer et al. (1984) do not include *D. magna*, which prefers more eutrophic waters than the Great Lakes of the northcentral USA and Canada. Therefore, Balcer et al.'s (1984) key would incorrectly identify any *D. magna* specimens to the species of *D. pulex* because of the spatial intent of the key. Microscopic differentiation of *D. magna* from *D. pulex* requires slide mounting for smaller specimens, but larger *D. magna* specimens jump out from other species, especially in slide-mounted specimens (but also under ~80-100x on dissecting scopes), because of the sigmoid structure of the postabdomen. The direction and angle of the carapace-head union separates *D. magna* from *D. ambigua*. Had the investigators (Gray, LimnoPro) used slide-mounted specimens and used a broad taxonomic key, this determination should have been straight forward.

For combining datasets, consider *D. magna*, *D. ambigua*, and *D. pulex* synonymous with *Daphnia* Group-2 or 'D. magna / D. ambigua.'

Table 8. Synonymies in the genus *Daphnia* (Group-2). 'X' marks determinations in each of the datasets considered. The final recommended taxon (O.T.U.) is in the bottom row highlighted.

Species	Gray	LimnoPro	DNA	RCC
<i>D. pulex</i>	X	X		
<i>D. magna</i>			X	X
<i>D. ambigua</i>			X	X
Recommend: <i>D. magna / D. ambigua</i>				

Ceriodaphnia

The *Ceriodaphnia* of Utah Lake were identified by both the Gray and LimnoPro datasets as only *C. quadrangula*. However, this taxon was not validated by DNA evidence, and although we reviewed many *Ceriodaphnia*, we were unable to find any *C. quadrangula* in the samples. This includes not only thousands of *Ceriodaphnia* from Utah Lake, but also thousands from Farmington Bay of Great Salt Lake—no *C. quadrangula*. We believe this maybe from sloppy taxonomic error because you cannot possibly determine a specimen to be *C. quadrangula* if it has a fenestra. All the specimens we observed had a fenestra, and therefore could not be keyed to *C. quadrangula* by anyone carefully examining the specimens—though this usually requires slide mounting.

The DNA dataset, included *C. dubia*, *C. acanthina*, *C. laticaudata*, and *Ceriodaphnia* sp.. These species can only be identified by microscopic analysis at high magnification (i.e., slide mounting). The specimens we identified in the RCC dataset included *C. dubia* and when occasionally specimens that could not be identified because of their maturity or condition (*Ceriodaphnia* sp.). Upon cursory examination, the shape of these *Ceriodaphnia* gestalts similar to *C. quadrangula*, and the body appears to lack reticulations (as per *C. quadrangula*). However, microscopic examination of finer features indicates that (in addition to fenestra mentioned above) they have a light and variable reticulation of the carapace that also matches *C. dubia*. It is possible that a few other species were present and that we found them as *Ceriodaphnia* sp.. One answer to this problem is simply to call all *Ceriodaphnia* the single genus-level taxon, *Ceriodaphnia* sp., as recommended in the table below (Table 9).

Table 9. Synonymies in the genus *Ceriodaphnia*. ‘X’ marks determinations in each of the datasets considered. The final recommended taxon (O.T.U.) is in the bottom row highlighted.

Species	Gray	LimnoPro	DNA	RCC
<i>C. quadrangula</i>	X	X		
<i>C. dubia</i>			X	X
<i>C. cf. acanthina</i>			X	
<i>C. cf. laticaudata</i> (x3)			X	
<i>Ceriodaphnia</i> sp. (x3)			X	X
Recommend: <i>Ceriodaphnia</i> sp.				

The dataset from Gray used the ambiguous *Diaphanosoma* whereas, LimnoPro identified all *Diaphanosoma* as *D. brachyurum*. However, the DNA dataset found zero *D. brachyurum*, rather relating the DNA sequences to a different species, *D. cf. heberti*; this is a species not represented in any North American key (e.g., Thorpe and Rogers 2016, Haney et al. 2013, Pennak 1978, etc.). Thus, it was impossible for taxonomists in North America to identify this species using standard bench references. Additionally, the species *D. heberti* was not listed as a taxon on ITIS, or WORMS (lists of current taxa). When systematic study determines that an ITIS-listed species is determined to be invalid, or a synonym for another species, they remain listed but are flagged as invalid taxa. Thus, when we find a taxon completely omitted from these species lists, it usually means they were missed simply as an oversight. We have reached out to several authors on the topic including the original author that described *D. heberti* (Korovchinsky 2002). When we found these specimens, we understood why they called them *D. brachyurum*; but we also disagreed with their final determination because the fit just wasn't right. First, the most-current key to North American *Diaphanosoma* (Thorpe and Rogers 2016) is broken. Page 458, couplet 5, “posterioventral margin of valves with denticles,” vs. couplet 5', “posterioventral margin of valves with denticles...” One of the coupling choices is a typographical error. Additionally, the prior coupling (4) states that the dorsal antennal ramus [has] 7 or 8 setae—so the extension of coupling 5' continuing, “...distal antennomere [=ramus] with 7 or 8 setae,” is redundant, and self-validating. Every species with 7 or 8 setae on dorsal antennal ramus will key (ambiguously) to this one species.

In Pennak (1978), these specimens are clearly identified as the now defunct species, *D. leuchtenbergianum*, which was synonymized (and then split into several other species) with *D. bergei* (Košíněk 1981). But working through the Pennak (1978) key may have revealed why the Gray and Limnopro datasets designated Utah Lake *Diaphanosoma* to the species *D. brachyurum*. Pennak (1978) differentiates *D. brachyurum* from *D. leuchtenbergianum* (= *D. bergei* et al.) by the eye being near the anterior margin of the head, whereas the key by Haney et al. (2013) specifies only that they eye is near the margin of the head. Thus, the key of Haney et al. (2013) would (incorrectly) take specimens from Utah Lake, with eyes near the ventral margin of the head to species *D. brachyurum*. The key by Balcer et al. (1984) did not provide species level keys, deferring to Košínek (1981).

The most recent functional key (given that the key of Thorpe and Rogers (2016) is broken) to United States *Diphanosoma* is by Haney et al. (2013) who separated the two similar species based on the “proximity of the eye to the margin of the head” for *D. brachyurum*. This was presented and interpreted in one important coupling to mean near any margin of the head, but in species descriptions at the end of the key, *D. brachyurum* should have a very large eye, near the anterior margin of the head, vs. *D. bergei* which has a smaller eye (Fig. 2), located more centrally or ventrally-marginally (Košíněk 1981, Haney et al. 2013).

If not for DNA evidence for another species, we would have left these taxa and the *D. brachyurum* / *D. bergei* complex in the RCC dataset also. But to understand the validity

of this observation, we needed to know the rationale for the difference in the DNA dataset, which found *D. heberti* but zero *D. Brachyurum*. The original species description of *D. heberti* (Korovchinsky 2002) considers the morphology antennal spines and they are quite different between species. Specifically, *D. heberti* has a single short and stout spine where all others have a long skinny one. Barcoding for this species has been done in Mexico and Canada, but since North American keys excluded the species, no American biologists report it; the distribution is an arc ending at the US border. Korovchinsky (pers comm) verified that this is still a valid taxon (Table 10).

Table 10. Synonymies in the genus *Diaphanosoma*. ‘X’ marks determinations in each of the datasets considered. The final recommended taxon (O.T.U.) is in the bottom row highlighted. Asterisks indicate that using different references produced different results. Thorpe and Rogers (2016) placed these specimens in the *D. bergei* complex. Whereas Pennak (1978) placed them in the defunct species *D. leuchtenbergianum*.

<i>Diaphanosoma</i> sp.	X			
<i>Diaphanosoma brachyurum</i>		X		
<i>Diaphanosoma heberti</i>			X	X
<i>Diaphanosoma leuchtenbergianum</i>				*
<i>Diaphanosoma bergei</i>				*
Recommend: <i>D. cf. heberti</i>				



Figure 2. *Diaphanosoma* cf. *heberti* from Utah Lake. Note that truncated head is not typical—specimen was damaged by handling.

Moina

The dataset of Gray reported no *Moina* species at all. Given the abundance of this species, they were probably grouped with another taxon through error in the Gray dataset. *Moina* and *Ceriodaphnia* both have a notch behind the head that could cause some confusion. However, the gestalt body form of *Moina* is more similar to *Diaphanosoma* in that the shape of the carapace and head is comparable. Additionally, the carapace is often parted wide at the ventral side, reminiscent of the wing-like appearance of the *Diaphanosoma* carapace. Moreover, since the ultimate determination of *Moina* was *M. micrura*, which is very small, specimens may have been thought to be immature *Diaphanosoma*—explaining why the taxon ‘*Diaphanosoma* sp.’ was used by Gray. Given the high densities and importance of the species, this taxonomic error should be addressed.

The dataset from LimnoPro reported 380 *Moina macrocarpa*. *Moina macrocarpa* is not a valid taxon. It is however, a very popular misspelling, so popular that the misspelling has made it into journal articles, magazine articles and web pages. The valid species is *M. macrocopa* (q.v., Goulden 1968, Haney 2013, Thorpe and Rogers (2016), ITIS (2018), and WORMS (2018)). Since it is cultured for fish culture, popular magazines help propagate the misspelling. However popular, *M. macrocopa* is a relatively large and hairy beast of a *Moina*. Those observed in Utah Lake are much smaller and delicate.

The RCC dataset reports these as *M. micrura*. The specimens of Utah Lake key to this taxon by multiple keys, including: Thorpe and Rogers 2016, Haney 2013, and the original systematic revision of worldwide *Moina* taxonomy by Clyde Goulden (1968) at the Academy of Natural Sciences. It is necessary to view slide-mounted specimens at high magnification to differentiate these specific *Moina* species; we do not believe that Gray or LimnoPro prepared slides for most specimens, but rather rolled the dice using the pop culture supposition that all *Moina* are *M. macrocarpa* [sic]. In addition to being larger and hairier, *M. macrocopa* has large spines on the first leg. Whereas, *M. micrura* have plumose setules on the last and penultimate articles of the first leg. This is relatively easy to distinguish *IF* slides are prepared (Table 11).

The case of Utah Lake *Moina* is more interesting in that it highlights one of the pitfalls of DNA taxonomy. Specifically, the DNA dataset found no *M. micrura* but did find an occurrence of *Moina mongolica*, which is strictly an Old-World halophile (saltwater-dweller). In the systematic revision of Moinidae (Goulden 1968) the history of taxonomy was considered briefly. Goulden (1968) described how several other OldWorld species were synonymized into *Moina micrura*, those of the USA are typically the *M. m. typica* subspecies. However, many of the records of *Moina mongolica* from fresher waters in central Russia (late 1800s-early1900s) were very likely misidentified *M. micrura* (Goulden 1968). Thus, there are two possible paths whereby the DNA database would identify *M. mongolica*. First and most likely, *M. micrura* specimens from old museum collections (improperly identified as per Goulden 1968) were incorrectly barcoded as *M. mongolica*, and this code persisted in the DNA database based on erroneous taxonomy. Another (less-likely) scenario could occur whereby *M. mongolica* had come as an invader to Great Salt Lake and spread to other regional aquatic systems. In our Review of Farmington Bay zooplankton, we found that *Moina micrura* was the only Moinidae species present (so far). If halophile *M. mongolica* were to invade Utah’s waters, Great Salt Lake would be the most likely epicenter of the invasion—and our analysis of zooplankton there also only found *M. micrura*. Therefore, we believe the correct identity of *Moina* in Utah Lake to be *M. micrura* despite DNA evidence. This identity is confirmed up by every taxonomic key we examined (e.g., Goulden 1968, Pennak 1978, Haney 2013, Thorpe and Rogers 2016). Regardless of the nature of the error, the moinid specimens we have observed differ from *M. macrocopa* (or *M. marcocarpa* [sic]) in many ways. The over use of historic local species-lists developed without mounted specimens, and cavalier taxonomy without slide validation caused a taxonomic (and indeed typographic) error to have persisted among zooplankton datasets of the region for many years.

Table 11. Synonymies in the genus *Moina*. ‘X’ marks determinations in each of the datasets considered. The final recommended taxon (O.T.U.) is in the bottom row highlighted. *Gray probably added these to *Diaphanosoma* sp., which is (likely) why he used the ambiguous genus identity.

Moina Taxonomic Synonymies				
Species	Gray*	LimnoPro	DNA	RCC

<i>Moina macrocarpa</i> [sic]		X		
<i>Moina mongolica</i>			X	
<i>Moina micrura</i>				X
Recommend: <i>Moina</i> cf. <i>micrura</i>				

Bosminidae: Bosmina longirostris.

All the morphological taxonomic determinations of Bosminidae resulted in the same conclusion: the ubiquitous *Bosmina longirostris*. However, the DNA dataset called all *Bosmina* sp. the single species *Bosmina leideri*. Thorpe and Rogers (2016) note that the determination of *B. longirostris* is actually a complex of cryptic species *B. longirostris*, *B. freyi*, and *B. leideri*—species that cannot currently be morphologically separated. We do not believe these taxonomic units to be in any conflict. The ambiguity of this species complex should be acknowledged by including the ‘complex’ after the species name. However, since the species apparently can not be separated, one wonders how a DNA database record for *B. leideri* came into existence. We prefer the more ambiguous “*Bosmina longirostris* complex” because it accommodates the future resolution of these species, while being independently replicable by other investigators (Table 12).

Table 12. Synonymies in the genus *Bosmina*. ‘X’ marks determinations in each of the datasets considered. The final recommended taxon (O.T.U.) is in the bottom row highlighted.

Species	Gray	LimnoPro	DNA	RCC
<i>Bosmina longirostris</i>	X	X		X
<i>Bosmina leideri</i>			X	
Recommend: <i>Bosmina longirostris</i> complex				

Chydoridae: Leydigia sp.

Gray and LimnoPro both identified *Leydigia* to the species *Leydigia leydigi*. However, this species was not identified by DNA, which instead found *L. lousi* [sic.], which was misspelled in the database (*L. louisii*). The most current key (Thorpe and Rogers 2016)

supposedly separates *L. leydigi* from *L. lousi mexicana*. However, the key uses only the characters that Kotov (2003) used to separate *L. leydigi* from *L. lousi* in general (including all subspecies), completely ignoring the characters of Kotov et al. (2003) that separate *L. lousi lousi* from *L. lousi mexicana*. Thus, the key of Thorpe and Rogers (2016) is incomplete and erroneous. We recommend that the best taxon for this group is to ignore the sub-species designation in the key of Thorpe and Rogers (2016) because this will allow independent validation / confirmation / replication of taxonomic effort without referring investigators to the ever-changing sea of primary systematic literature (Table 13).

The actual current separation of *L. l. mexicana* from *L. l. lousi* involves the location of chitinous insertions within the swimming setae (far from bases in *L. l. mexicana*, near bases in *L. l. lousi*). Additionally, the shape of ocelli differs markedly between the two different sub species. Thus, it is possible to separate these subspecies relatively reliably, under high magnification. We will continue to report the subspecies, but for combining datasets, investigators should assume that historical *L. leydigi* are probably *L. lousi*—and further they are probably *L. lousi lousi*.

As to the nature of the errors from *Leydigia leydigi*, we believe that taxonomists (Gray, LimnoPro) saw the large, swooping circular postabdomen under dissecting scope, and assumed that the lateral spines (which they could not see) were small (*Leydigia leydigi*) when in fact they were, proportionally large (*Leydigia lousi*).

Table 13. Synonymies in the genus *Leydigia*. ‘X’ marks determinations in each of the datasets considered. The final recommended taxon (O.T.U.) is in the bottom row highlighted.

Species	Gray	LimnoPro	DNA	RCC
<i>Leydigia leydigi</i>	X	X		
<i>Leydigia lousi</i> [sic.]			X	
<i>Leydigia lousi mexicana</i>				
<i>Leydigia lousi lousi</i>				X
Recommend: <i>Leydigia lousi lousi</i>				

Chydoridae: Pleuroxus

We did not find any *Pleuroxus* from Utah Lake in the 2017-2018 samples we identified, but both Gray and LimnoPro did find this genus—though they disagreed on the species determination. Since we did not see their specimens, thoughts presented here are more conjecture than in the other taxonomic groups where we were pretty sure that we were talking about the same specimens. *Alona* (RCC dataset) could have been included here in Gray and LimnoPro.

Chydoridae: *Chydorus spaericus*

Both Gray and LimnoPro identified *Chydorus spaericus* from Utah Lake. We did not find any *Chydorus* in the RCC 2017-2018 data, but the DNA dataset found *Chydorus brevilabris*. Both are small and roundish. Belyaeva and Taylor (2008) reported many cryptic species, recognized by molecular markers and ontology, occur in the ‘*Chydorus spaericus* complex.’

Chydoridae: *Alona setulosa*

We found 20 specimens of *Alona setulosa* (out of more than 23,000 identified specimens we examined for the Utah Lake dataset). It makes sense that this species was omitted from DNA analysis due to subsampling and splitting. If the other datasets included *A. setulosa*, it would be difficult to imagine how they might have misclassified it. However, it has a superficial appearance to other Chydoridae. It is especially similar to *Leydigia* sp.—both *Alona* sp. and *Leydigia* sp. are in the subfamily Aloninae—if a few specimens were in the older datasets, they were probably added to the more common *Leydigia*. The effect of this error is probably small, since among our samples, *Alona* comprised only about 0.08% of the annual community of zooplankton. Or *Alona* (some or all) may have been included with *Pleuroxus* sp..

Rotifera

Rotifers were generally under sampled or at least under represented. It might be best to lump them into a very wide taxon, perhaps Rotifera. Regardless of how they are dealt with, we did not have enough specimens to make recommendations on the taxonomy of the group.

Results: Recommended Tabulated Synonymies by Dataset

We summarized the synonymies from the previous section, to recommend the best practical grouping for each dataset. Taxa shaded in orange require taxonomic reclassification with other datasets.

1. Gray Data required the most changes (below).

Family	Gray Taxa	New Recommended O.T.U.
Cyclopidae	<i>Eucyclops agilis</i>	<i>Microcyclops rubellus</i>
Cyclopidae	<i>Acanthocyclops robustus</i>	<i>Acanthocyclops americanus</i>
Diaptomidae	<i>Leptodiaptomus sicilis</i>	<i>Leptodiaptomus sicilis</i>
Canthocamptidae	<i>Attheyella</i>	Harpacticoida

Daphniidae	<i>Ceriodaphnia quadrangula</i>	<i>Ceriodaphnia</i> sp. / <i>Ceriodaphnia dubia</i>
Daphniidae	<i>Daphnia pulex</i>	<i>Daphnia magna</i> / <i>Daphnia ambigua</i>
Daphniidae	<i>Daphnia retrocurva</i>	<i>Daphnia galeata mendotae</i> / <i>D. retrocurva</i>
Chydoridae	<i>Pleuroxus striatus</i>	<i>Pleuroxus</i> sp.
Chydoridae	<i>Chydorus sphaericus</i>	<i>Chydorus brevilabris</i> (per DNA only)
Chydoridae	<i>Leydigia leydigi</i>	<i>Leydigia louisii</i> / (also <i>Alona</i> sp).
Sididae	<i>Diaphanosoma</i> sp.	<i>Diaphanosoma</i> cf. <i>heberti</i> / Also: <i>Moina micrura</i>
Bosminiidae	<i>Bosmina longirostris</i>	<i>Bosmina longirostris</i> complex
Ilyocryptidae	<i>Ilyocryptus</i> sp.	<i>Ilyocryptus</i> sp.
Leptodoridae	<i>Leptodora kindti</i>	<i>Leptodora kindti</i>

Brachionidae *Brachionus* Rotifera? Brachionidae
Keratella Rotifera? Asplanchnidae *Asplanchna*
 Rotifera?

2. Data From LimnoPro (below)

Order	Family	LimnoPro Taxa	New Recommended O.T.U.
Cyclopoida	Cyclopidae	<i>Microcyclops rubellus</i>	<i>Microcyclops rubellus</i>
Cyclopoida	Cyclopidae	<i>Acanthocyclops robustus</i>	<i>Acanthocyclops americanus</i>
Calanoida	Diaptomidae	<i>Leptodiaptomus sicilis</i>	<i>Leptodiaptomus sicilis</i>
Harpacticoida	Canthocamptidae	<i>Cletocamptus</i> sp.	Harpacticoida
Cladocera	Daphniidae	<i>Ceriodaphnia quadrangula</i>	<i>Ceriodaphnia</i> sp. / <i>Ceriodaphnia dubia</i>
Cladocera	Daphniidae	<i>Daphnia pulex</i>	<i>Daphnia magna</i> / <i>Daphnia ambigua</i>
Cladocera	Daphniidae	<i>Daphnia retrocurva</i>	<i>Daphnia galeata mendotae</i> / <i>D. retrocurva</i>
Cladocera	Daphniidae	<i>Simocephalus vetulus</i>	<i>Simocephalus</i> sp.
Cladocera	Chydoridae	<i>Pleuroxus aduncus</i>	<i>Pleuroxus</i> sp.
Cladocera	Chydoridae	<i>Chydorus sphaericus</i>	<i>Chydorus brevilabris</i> (per DNA only)

Cladocera	Chydoridae	<i>Leydigia leydigi</i>	<i>Leydigia lousisi</i>
Cladocera	Moinidae	<i>Moina macrocarpa</i>	<i>Moina micrura</i>
Cladocera	Sididae	<i>Diaphanosoma brachyurum</i>	<i>Diaphanosoma cf. heberti</i>
Cladocera	Bosminiidae	<i>Bosmina longirostris</i>	<i>Bosmina longirostris</i> complex
Cladocera	Leptodoridae	<i>Leptodora kindti</i>	<i>Leptodora kindti</i>
Plioma	Brachionidae	<i>Brachionus plicatilis</i>	Rotifera?
Plioma	Brachionidae	<i>Brachionus calyciflorus</i>	Rotifera?
Plioma	Asplanchnidae	<i>Asplanchna sp.</i>	Rotifera?

Conclusions

We documented a rationale for the systematic and defensible integration of data from different sources for Utah Lake and beyond. Without this information, the underlying assumptions of grouping taxa might appear arbitrary. This document provides a framework for discussing potential taxonomic synonyms and Operational Taxonomic Units (O.T.U.s) without each and every Utah Lake study requiring a comprehensive literature review to support laboratory operations; investigators can simply cite this document and explain why they agree or disagree with our recommended taxonomic units. Similarly, changes and deviations from procedure can refer to this document for context.

We also found numerous apparent taxonomic errors. Some were due to observer error, and some might be due to using inappropriate regional keys, but many were due to the quality of taxonomic keys not keeping pace with the changing state of the science. Therefore, in addition to providing a framework for systematic grouping of animals, we also have worked out the current ideal taxonomic effort for Utah Lake zooplankton studies. This will streamline our laboratory work in the future. Furthermore, we have found that many of the same issues may apply to other regional waterbodies, such as Farmington Bay. This allows for broader application of our findings.

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DRAFT

Appendix 5. A Case for Adaptive Holistic Management of Phragmites in the Jordan River Drainage and Southern Utah

A Case for Adaptive Holistic Management of Phragmites in the Jordan River Drainage and Southern Utah

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Figure 42. Phragmites along the shore of Utah Lake, UT photo courtesy of the Utah Lake Commission.

Background

Wetlands in the Jordan River drainage, Utah include those surrounding Utah Lake, the fourth largest freshwater lake in the western U.S. and those surrounding the southern portion of Great Salt Lake, primarily Farmington Bay. Phragmites has invaded all of wetlands in the drainage and its impacts are substantial. I have been conducting ecological research on Jordan River drainage Utah wetlands for about 9 years, including phragmites research. My expertise includes ecological studies on; phytoplankton, zooplankton,

aquatic macroinvertebrate assemblages, fisheries, nutrients, and invasive species, as well as serving as a consultant for management of these critical wetlands. Invasive species research that I have conducted includes impacts on native aquatic threatened and endangered species and ecosystems (see Richards, Richards et al., and Hoven and Richards in Literature Cited) and developing biocontrol agents including chrysomelid beetles, the weevil (*Rhinocyllus conicus*) and hawk moths (Family Sphingidae) for controlling St John's wort (*Hypericum perforatum*), leafy spurge (*Euphorbia esula*), Spotted Knapweed (*Centaurea maculosa*), and Canadian musk thistle (*Carduus nutans*).

Wetlands in the Jordan River drainage are some of the most important and last remaining wetlands in the Central Flyway and are designated World Natural Heritage sites, primarily because of the importance to shorebirds and waterfowl. Not surprisingly, the vast majority of managers in the area consider invasive phragmites to be a major problem. One of these negative effects that I am analyzing with several other researchers is the loss of shorebird nesting habitat due to expanding phragmites stands.

All managers of these wetlands also agree however, that the number one problem facing these wetlands is chronic lack of water. Most suggest that the second most important problem is excessive nutrients.

Having had the privilege of spending much time researching these wetlands, I have a much different opinion on phragmites than most and will discuss some of its vastly underappreciated ecological benefits. I will also discuss some of the best ways to manage this invasive grass focusing on Provo Bay, Utah Lake and Farmington Bay, Great Salt Lake.

Ecological benefits of invasive Phragmites

Both Provo Bay and Farmington Bay are paltry analogs of a once incomparable and incredibly diverse Jordan River drainage ecosystem, pre-Mormon settlement, mid-1800s (Richards 2018a). At present, during years with normal precipitation, both bays have enough water to flood invasive phragmites stands up to about one meter in depth that lasts throughout most of the year. However, in summer both bays typically dry and phragmites stands are elevated on dry ground several inches above water level. Cattails (*Typhus* sp.) and submerged aquatic vegetation (e.g. *Stuckenia* sp., *Ruppia* sp., etc.) cannot survive in these drying conditions. Therefore, many natives, including cattails, are not losing habitat as much to phragmites invasion but to increasingly drying conditions from chronic dewatering and now, climate change.

Phragmites and soil conditions

Stands of this invasive grass provide shading that cools and helps maintain damp soils and cools surface water during summer when daytime air temperatures regularly exceed 100° F. Surface ground temperatures reach or exceed air temperatures, soils quickly dry, and top soil layers are lost to wind action in locations where phragmites stands have been

physically removed by managers. In stands of phragmites, soil temperatures remain cooler and moister (Richards unpublished data). Of real concern to citizens in the Greater Salt Lake City Metropolitan Area is degradation of air quality due to wind born particles from exposed Great Salt Lake sediments, most of which contain toxic metals including zinc, cadmium, lead, and mercury that can easily become windborne. Although not quantified, phragmites stands likely help prevent some of this loss and help protect air quality. More research is needed to evaluate transpiration loss from phragmites vs. evaporative loss without such stands including soil loss and changes in air quality.

Phragmites and Aquatic Faunal Assemblages

Provo Bay and Farmington Bay wetlands provide crucial habitat for aquatic species, typically starting from shortly after ice out (Provo Bay) and continuing throughout spring into early-mid summer when they are inundated. Utah Lake and Farmington Bay waters are eutrophic to hypereutrophic. Utah Lake is one of the most productive warm water fisheries in the western USA and is the most productive lake in Utah. Provo Bay has the highest primary and secondary production in Utah Lake. Hundreds of acres of phragmites in Provo Bay contribute to this productivity. When inundated, phragmites stands provide premier habitat for; zooplankton, free swimming and benthic macroinvertebrates (e.g. midges, snails, beetles, bugs, etc.), and act as nurseries for small fishes including larvae, juveniles, and minnows. Large predatory fish cannot penetrate the stands, although several broadcast spawners (i.e. White Bass, Carp, Channel Catfish etc.) utilize phragmites for egg laying. We have collected dozens of biotic samples within these stands and directly outside of the stands and estimate that invertebrate and small fish densities and biomass are at least an order of magnitude greater in phragmites stands than outside. Loss of these stands would dramatically reduce fisheries productivity in Provo Bay and Utah Lake. Because Farmington Bay is more saline than Provo Bay it does not sustain a warm water fishery, other than invasive carp, but does support one of the greatest concentrations of shorebirds, wading birds, and waterfowl in the western USA, all of which depend on its aquatic invertebrate food base.

I have estimated that Utah Lake produces 1000 to 6000 tons of chironomids (midge) biomass seasonally and that Provo Bay often produces 20,000 midge larvae/m² of lake bottom (Richards 2017). Likewise, Farmington Bay. Midges are the dominant food source for higher trophic levels including aquatic and terrestrial. Migratory cliff swallows and barn swallows almost exclusively feed on adult midges during late summer/early autumn. Dozens of other bird species also depend on this harvest. Adult midge swarms in Provo Bay and Farmington Bay are spectacular. These mating swarms are often misidentified as smoke from fires and can contain tens of millions of male and female midges per swarm with dozens of swarms occurring at once, each rising from the shoreline up to perhaps 100 to 200 feet in altitude (D.C. Richards personal observations). Adult midges rely on the thick stands of phragmites as shelter after emerging from the bays and as resting substrate between mating. I have begun measuring and comparing midge densities in phragmites and other plants including cattails, bulrush, and invasive salt cedar. Midge densities are greatest per surface area on salt cedar due to its many branches and surface area but phragmites stands support the greatest midge density per

unit land area due to its height and density (unpublished data). I postulate that midge densities would be much lower in Provo Bay and Farmington Bay without stands of phragmites to provide shelter to adults. This would have severe negative consequences on fish and bird populations dependent on these aquatic insects. Other invertebrate species in Provo Bay and Farmington Bay also depend on phragmites, including several Odonata taxa (dragonflies and damselflies), aphids, spiders, beetles and bugs, to name a few (personal observations).

Phragmites allelopathic chemicals can reduce algal blooms

Phragmites is known to produce allelopathic chemicals that reduce interspecific competition from other plants, including algae. Provo Bay and Farmington Bay have massive algal blooms during summer, including potentially harmful cyanobacteria blooms. Algal blooms are often non-existent within phragmites stands except during the most severe blooms. Water clarity is much better within stands than out, which I attribute more to allelopathic chemicals than to reduced wind induced sediment disturbance and have much lower levels of algae within (unpublished data).

Phragmites and Other Wildlife

The thick stands of phragmites in Provo Bay and Farmington Bay cover hundreds of acres and provide shelter and cover for many mammals, small and large. Native mule deer populations in Utah and throughout the west are in decline and what was once prime mule deer habitat in these highly urbanized areas now supports relatively few and small populations. However, mule deer survive in Provo Bay and Farmington Bay wetlands in part due to the security provided by dense stands of phragmites. Remaining agriculture lands near the bays provide food resources for mule deer, while phragmites provides visual shelter from hunters. Utah is a recreational hunting culture. These small deer populations would be eradicated without this shelter. Several agencies are involved in restoration projects along the Jordan River that joins Utah Lake and Great Salt Lake and one of their goals is to provide habitat for mule deer. I have observed dozens of deer using phragmites on Provo Bay and Farmington Bay to safely move between other habitats. There are deer trails through all but the thickest stands. I have personally observed other mammals utilizing phragmites include; beaver, muskrat, skunk, coyote, mink, voles, and other rodents. Amphibians also thrive in phragmites stands in the bays. Boreal Chorus Frogs occur by the millions and use the stands for mating and egg laying; their croaking during mating gets so loud at times that it can drown out the sounds of low flying commercial jets at Salt Lake City airport. Threatened Columbia Spotted Frog and the invasive American Bullfrog also thrive in phragmites during high water. Hundreds of flocking birds such as grackles and starlings utilize the thick phragmites stands to feed and avoid harriers, falcons, and other raptors, even during the coldest winter days. Several species of waterfowl rely on security and food resources provided by phragmites stands to nest and raise young including; American coots, Canada geese, and mallards. Great Horned owls and other owl species patrol phragmites at night for small rodents in Provo Bay and Farmington Bay. Even though it is virtually a monoculture within phragmites stands; the ecosystem is alive with life.

Phragmites and Nutrient Removal

The second biggest threat to the Provo Bay and Farmington Bay ecosystems, real or perceived, is nutrients. Water quality agencies including Utah Division of Water Quality and USEPA have been focusing their attention on reducing nutrient inputs into these bays. However, phragmites is the number one reducer of nutrients into the bays. We have documented at least a tenfold reduction of phosphorus in water treatment effluent as it passes through phragmites wetlands and enters into Provo and Farmington Bays. Phragmites also uptakes excessive nutrients from the soil and stores them as plant material. Point and non-point sources of nutrients may be contributing to algal blooms in the bays and phragmites is by far the most cost-effective method of reducing these sources of nutrients.

Adaptive Phragmites Management: A Holistic Approach

Poorly planned, poorly enacted, poorly coordinated, and myopic phragmites management in the Jordan River drainage is a problem. Multiple government agencies from city to county to state to federal and numerous ‘commissions’ have differing agendas when it comes to managing phragmites. The most common methods used in the drainage are mowing, crushing, spraying with herbicides, and grazing by cattle with very little coordination between agencies and with no formalized plan as to how best revegetate with native species. The mantra by government agencies in Utah is; ‘phragmites is bad and it must be eliminated’. Most citizens that I discuss phragmites with tend to think phragmites is not a problem and actually is esthetically pleasing, unless of course they are allergic to it. Obviously, the truth is somewhere in between.

Several promising approaches to managing phragmites include but are not limited to: grazing by cattle, harvesting as food resource for livestock, harvesting for cultural use by Native Americans, and boutique paper. Managers at the Farmington Bay Waterfowl Management Area near Salt Lake City are using cattle grazing to remove phragmites to improve shorebird and waterfowl habitat. Results are promising and it appears that once cattle have been introduced to Phragmites, they prefer it to other forage. It has been estimated that phragmites contains 18% protein which is higher than the most widely used cattle forage crop, alfalfa. Phragmites however is harder to digest than alfalfa and a simple remedy would be to harvest and produce easier to digest pellets. A business opportunity waiting to be implemented.

Continued harvesting of phragmites either by direct grazing or as a sustainable crop would remove nutrients tied up in the plants and thus remove nutrients from the soil and eventually reduce the impacts of nutrients on water quality and possibly reducing the intensity of harmful algal blooms. An estimated 1500 tons of ortho phosphates are released into Utah Lake annually from lakebed sediments alone (Hogsett et. al. 2019). Unpublished data from the Wasatch Front Water Quality Council has estimated that perhaps up to 500 tons or more are deposited annually on the lake from the atmosphere. There is also a legacy of natural and anthropomorphic nutrient overload in these

wetlands. Sustainable harvest of phragmites is the most prudent management option for reducing nutrients in the wetlands and the waters of Utah Lake and Great Salt Lake.

Southern Utah continues to experience drought conditions, with the exception of 2019. Global climate change models predict continued and intensifying drought into the foreseeable future in southern Utah and the southwest. Native Americans, including Navajo, Ute Mountain, and Hopi Tribes are particularly hard hit by these droughts and often do not have enough water to grow forage for their livestock. Harvesting phragmites from Utah Lake and Great Salt Lake wetlands could easily meet Native American livestock needs. I have been in conversations with members of this tribe and they support this idea. In addition, Native Americans have used and continue to use native phragmites and other grasses for many items including baskets, housing material, food, medicine, and other uses. Adobe building material is composed of grass and mud and phragmites is an excellent adobe building material, including traditional Navajo hogans. Indeed, several Native Americans regularly make the trip from southern Utah to Provo Bay to harvest pickup truck loads of phragmites for multiple uses. Every effort should be made to increase these uses to benefit tribes and reduce impacts of invasive phragmites in wetlands.

Of course, phragmites has been used for centuries throughout the world to make paper. Boutique paper made from unique plant materials is a thriving business and phragmites harvested from Utah Lake and Great Salt Lake wetlands would be ideal start up to one or more entrepreneurial business minded citizens.

It is time for us to stop considering invasive phragmites as a nuisance and begin to utilize it as a crop to help meet the needs of our now universally altered analog ecosystems and promote sustainable economies where it has become established. Adaptive and holistic management of phragmites will require a landscape ecology perspective (e.g. patch dynamics vs. mono single species stands) and an unbiased understanding of its negative impacts and more importantly its beneficial ecosystem services that it now provides (Kiviat 2013).

Remaining Native Phragmites Stands and Unnecessary Introduction of Biocontrol Agents

There are very few remaining stands of native phragmites in the Jordan River drainage and southern Utah. They are ecologically, culturally, and esthetically important. These remaining stands need protection including that from invasive phragmites. However, introducing biocontrol agents to control invasive phragmites is not a rational idea given the management options that I have listed above and more importantly because: The literature is ripe with instances where introduced biocontrol agents either failed to control targeted invasives or had severe direct or Indirect negative effects on ecosystems (Simberloff and Stiling 1996a; 1996b; Strong and Pemberton 2000; Louda et al 2003; Louda 1997; Boettner 2000; Henneman and Memmott 2001; Louda and O'Brian 2002; Pearson and Callaway 2003). Once established it is near impossible to eliminate invasive species, including those used for biocontrol.

Managers consistently consider biocontrols as alternatives to controlling invasive species, including insect predators on wetland invasive plants. Introduced tamarisk beetles have had limited impact on invasive salt cedar in Utah, primarily due to misapplication and a poor understanding of the biocontrols ecology (Dr. Tim Graham, Moab, UT). In the case of releasing phragmites biocontrols, what is going to keep the biocontrols from spreading to native phragmites? (nothing). The Jordan River drainage and other parts of Utah where both native and invasive phragmites occur often get extremely windy and also have long periods of time with little or no winds. Given enough time, it is inevitable that insect biocontrols will be transported to native stands and their populations will become established in native phragmites stands, if they are released. Functional extinction of native phragmites would be a tragic loss to Utah's native ecological and cultural heritage. Much better options are available.

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