Lake Stevens Alum Treatment Effectiveness Study

November 2022

Prepared for:
The City of Lake Stevens

Prepared by:
Tetra Tech Inc.
with review by Snohomish County Surface Water Management

TABLE OF CONTENTS

1. INTRODUCTION AND BACKGROUND

1.1. MANAGEMENT HISTORY

The City of Lake Stevens (City) and Snohomish County (County) have been working to reduce the effects of excessive phosphorus loading to Lake Stevens for several decades. A water quality analysis was completed in 2012 that examined long-term options for managing water quality of the lake (Snohomish County and Tetra Tech, 2012). Specifically, the study compared the effectiveness of continued operation of a hypolimnetic aeration system and the implementation of annual aluminum sulfate (alum) treatments. In the 2012 study, the nutrient dynamics of the lake were analyzed and the relative contributions of external (watershed) and internal (sediments) loading to the lake were identified at present and extrapolated to the future. The 2012 study determined approximately 45% of the annual phosphorus load was from the lake sediments (i.e., internal loading) and 40% was from ongoing inputs from watershed runoff (i.e., external loading) with the remainder coming from groundwater and precipitation (Snohomish County and Tetra Tech, 2012). The results of the 2012 study recommended that the hypolimnetic aerator be decommissioned and annual low-dose alum treatments be implemented to address excess phosphorus in the lake from both internal and external loading. The annual alum treatments would remove a portion of the external phosphorus load that entered the lake each year by stripping phosphorus out of the water column. Over time, the alum would also gradually inactivate sediment phosphorus to reduce internal loading. Based on the amount of phosphorus in the lake sediments, it was anticipated to take nine years of small alum treatments to inactivate the sediment phosphorus in the top 4 cm and reduce internal phosphorus loading. Controlling both the internal and external phosphorus load each year would result in a decrease in algae, particularly, cyanobacteria blooms that occurred in the spring and late summer/fall (Snohomish County and Tetra Tech, 2012).

The City's 2013 Phosphorus Management Plan (City of Lake Stevens, 2013) outlined a series of actions to reduce both internal and external phosphorus loading to the lake. Small annual alum treatments were proposed over the next 9 - 12 years to inactivate sediment phosphorus and reduce phosphorus in the water column from external loading sources. The first annual alum treatment was completed in June of 2013. Since then, treatments have been conducted annually through 2020. In 2016, the annual treatments were shifted from spring to fall. Overall, 430,000 gallons of alum have been applied to Lake Stevens from 2013 - 2020. The aluminum dose for each treatment was 0.15 mg Al/L except for 2020 when the dose was lowered to 0.12 mg Al/L for budgetary reasons. In total, 1.17 mg Al/L was applied to the lake over an eight-year time period.

2. WATER QUALITY ANALYSIS

2.1. PURPOSE

The original goals of lake restoration at Lake Stevens were to reduce the overall amount of phosphorus in the lake to subsequently reduce algal blooms, especially toxic blooms that pose a risk to lake users. While the aerator showed some initial effectiveness in reducing the total phosphorus (TP) and associated blooms, it's effectiveness decreased through the years leading to the decision to replace it with annual alum treatments. The main purpose of this water quality analysis is to assess the effectiveness of alum compared to aeration in meeting the overall project goals. More specifically, this section evaluates the effectiveness of alum treatment to successfully inactivate phosphorus from the following sources:

1) external phosphorus loading that enters the lake each year;
2) phosphorus left over from internal loading from the previous year; and
3) phosphorus in the sediments (cumulatively over time).

In addition, this analysis looks at all historic water quality data to determine if the lake is meeting the original restoration goals of reduced TP and reduced occurrences of algae and potentially toxic blooms compared to the time before any restoration efforts, including alum and aeration.

2.2. METHODS

The primary quality parameters analyzed were water clarity, TP, soluble reactive phosphorus (SRP), chlorophyll *a* (chl) and algal toxins. Monthly depth profiles of temperature and dissolved oxygen were also used to determine lake stratification. Lake Stevens raw data can be viewed and downloaded from the County's online water quality database at www.lakes.surfacewater.info. Algal toxin data for Lake Stevens can be found online at www.nwtoxicalgae.org.

Water quality sampling frequencies and depths have varied over the years (). The variation is a result of changes in sampling entities starting with consultants for specific studies and volunteer community efforts (1986 - 1996), the former Drainage District 8 (1997 - 2006) and Snohomish County/City of Lake Stevens (2007 - present) (

Table 1). Given the high variability in data collection, two different time periods of data were analyzed to meet the study objectives.

2007 - 2021: Aeration to Alum Change

Snohomish County and the City of Lake Stevens began partnering to collect water quality data in 2007 and continue ongoing collection. Data collected during this timeframe was the most consistently collected and followed a quality assurance plan compared to the data collected prior to 2007. Data collected during this timeframe was used to understand the water quality changes resulting from the shift from aeration to alum treatments. This period spans the last five years that the lake aerator was in operation (2007 - 2012) and the first nine years of annual alum treatments (2013 - 2021). The data set includes samples at several lake depths which allows for volume-weighting of the phosphorus data. Volume-weighting provides a better whole-lake understanding of the changes in phosphorus concentrations.

From 2007 - 2021, volume-weighted TP concentrations were calculated for the epilimnion from 1, 5 and 10 meters (m). TP concentrations in the hypolimnion were calculated as a volume-weighted average from 20, 30 and 40 m. Additionally, volume-weighted average epilimnetic TP and hypolimnetic averages were determined for the summer period of May - October to reflect stratified water quality conditions during the recreational season (**Table 2**). Chl concentrations were calculated as an average of 1, 5 and 10 m samples and were not volume-weighted. Finally, algal blooms were tested when reported by County staff and submitted to the State Department of Ecology's free testing program. Samples were analyzed for the liver toxin, microcystin, and the neurotoxin, anatoxin-a.

1986 - 2021: Pre-aeration to Present

While the depths of sample collection varied greatly, most years included phosphorus samples from 1 and 40 m and chl samples from 1 m. These data were analyzed to look at the long-term changes of all restoration efforts including pre-aeration, aeration, and alum treatments. Composite samples taken in 1989 and 1990 - 1992 were not directly comparable and were excluded from the dataset.

Table 1. History of water quality sampling for TP and chl, 1986 - 2021.

Time Period	Total Phosphorus							Chlorophyll *a*			
	1m	5m	10m	0-10m	20m	30m	40m	1m	5m	10m	0-10m
1986 - 1987 (Ja, F, Ma)	X	X	X				X	X	X		
1989 (D)											X
1990 (F, Ap, Jun, Au, O, D)				X			X				X
1991 (F, Ap, Jul, Aug, D)				X			X				X
1992 (F, Ap, Au, O)				X			X				X
1993 (May, Jun, Jul, Au, S, O)	X	X	X								X
1994 - 1996	X	X	X								X
1997 - 2003	X	X	X		X	X	X	X	X	X	
2004 - 2006	X						X	X			
2007 - 2021	X	X	X		X	X	X	X	X	X	

[1] Parameters were collected on a monthly basis unless otherwise denoted

[2] Samples were collected as a composite sample from multiple depths

[3] Samples were collected at 1, 5, 10, 20, 30 and 40 m from Apr - Nov and at 1, 10, 20 and 40 m from Dec - Mar; TP and chl data from 2018 were rejected due to quality assurance concerns.

2.3. RESULTS AND DISCUSSION

2.3.1. Phosphorus

Phosphorus is the main nutrient of concern in lakes as it is directly related to the growth of algae, including potentially toxic cyanobacteria. In most Washington lakes, phosphorus is the “limiting nutrient,” meaning that it controls the primary productivity in the lake, especially in highly productive lakes (Welch and Jacoby, 2001). Phosphorus was primarily measured as TP. SRP was also taken from the 40 m sample location. SRP is the dissolved inorganic phosphorus component that is found as phosphate in the water. SRP is bioavailable for plant and algae growth. TP includes SRP as well as particulate and other phosphorus forms. Particulate and other phosphorus forms (i.e., organic) are not immediately available for algal growth but over time can be released into the lake in more readily available forms.

Epilimnion (Upper Waters)

TP concentrations in the epilimnion were calculated as a volume-weighted average from 1, 5 and 10 m. During the last six years of hypolimnetic aeration (2007 - 2012), TP concentrations in the epilimnion were highly variable throughout the year ranging from 7.5 to 36 µg/L (Figure 1). The concentrations typically increased each winter following lake turnover. They remained high through March or April after which they decreased through the summer. After the implementation of annual alum treatments (2013 - 2021), TP was lower and less variable, ranging from 3 to 17 µg/L (Table 2**;** Figure 1). The large phosphorus peaks following lake turnover were not seen after 2013. Instead, peaks appeared to be later in spring, likely corresponding with external phosphorus loading from seasonal rainfall. Looking at just the summer volume-weighted epilimnetic TP, concentrations were reduced from an average of 12.8 ± 2.2 µg/L during aeration to 7.0 ± 1.2 µg/L with the annual alum treatments (Table 2).

Year-round phosphorus changes in the lake were also examined by looking at the 1 m changes for the entire 1989 - 2021 dataset. The data are shown in a box and whisker plot with time periods broken up between pre-alum, early aeration, late aeration, and alum (). TP values did not change significantly from pre-aeration through the aeration period, though the distributions are skewed slightly higher during aeration. With the onset of alum treatments, the 1 m TP significantly decreases compared to all other time periods. With alum treatments, the annual variation and the seasonal peaks are lower which likely is a result of reduced seasonal pulses following lake turnover (). It should be noted that the pre-aeration values are very limited as the data from 1990 - 1993 were not comparable and were excluded. For this reason, the 40 m dataset, which includes all years, provides a better measure of the aerator effectiveness.

Looking at just the summer recreational season, there is not a clear trend before and after aeration. However, annual averages are consistently lower after alum treatments (

Figure 3). Before alum, TP fluctuated above and below 12 µg/L and averaged around 12 µg/L. From 2013 - 2021, TP averages in the top 1 m have been consistently at or below 10 µg/L averaging around 7.0 µg/L (

Figure 3).

Lakes can also be classified by their trophic state based on mean summer epilimnetic TP values. Based on ranges from Nurberg (1996), lakes cal be classified as:

- oligotrophic – less than 10 µg/L or less
- mesotrphic - 10 to 30 µg/L
- eutrophic – 30 to 100 µg/L
- hypereutrophic – greater than 100 µg/L

Lake Stevens shifted from mesotrophic conditions prior to alum treatments to an oligotrophic state after the start of alum treatments. Given its size and morphology, Lake Stevens was likely an oligotrophic lake historically. The lake also shifted post-alum from Good to Excellent conditions as defined by the County in comparison to other area lakes (Snohomish County, 2021).

Table 2. Lake Stevens mean summer volume-weighted TP, chl, and Secchi disk 2007 - 2021.

Year	Mean Summer (May - Oct)				
	Whole-Lake Volume-Weighted TP (ug/L)	Epilimnion (0 - 10m) Volume-Weighted TP Concentration (ug/L)	Hypolimnion (20m - 47m) Volume-Weighted TP Concentration (ug/L)	Epilimnion (0 - 10m) Chl (ug/L)	Secchi Disk Depth (m)
2007	16.6	10.9	30.6	3.2	6.4
2008	21.2	15.1	37.3	3.5	6.1
2009	12.0	10.1	16.5	3.5	6.3
2010	22.6	14.9	39.0	5.8	5.5
2011	18.7	14.0	31.3	4.1	5.5
2012	16.4	11.5	27.5	4.1	6.5
2013 (Alum Started)	13.0	8.4	23.5	3.1	6.6
2014	9.3	7.8	11.9	2.6	7.7
2015	7.1	6.9	7.5	2.5	6.9
2016	8.9	8.4	9.7	3.7	6.3

Year	Mean Summer (May - Oct)				
	Whole-Lake Volume-Weighted TP (ug/L)	**Epilimnion (0 - 10m) Volume-Weighted TP Concentration (ug/L)**	**Hypolimnion (20m - 47m) Volume-Weighted TP Concentration (ug/L)**	**Epilimnion (0 - 10m) Chl (ug/L)**	**Secchi Disk Depth (m)**
2017 (May - July)	7.2	6.6	5.6	2.5	8
2018	no data	no data	no data	no data	7.2
2019	5.9	5.8	5.6	1.4	9.9
2020	7.7	7.2	7.7	2.5	8.3
2021	5.9	5.3	5.6	2.3	9.2
Pre-alum (2007 - 2012)	17.9 ± 3.8	12.8 ± 2.2	30.4 ± 8.0	4.1 ± 0.9	6.1 ± 0.4
Post-alum (2013 - 2021)	8.1 ± 2.3	7.0 ± 1.2	9.6 ± 6.0	2.6 ± 0.7	7.8 ± 1.3

Figure 1. Volume-weighted epilimnetic TP concentrations in Lake Stevens, 2007 - 2021.

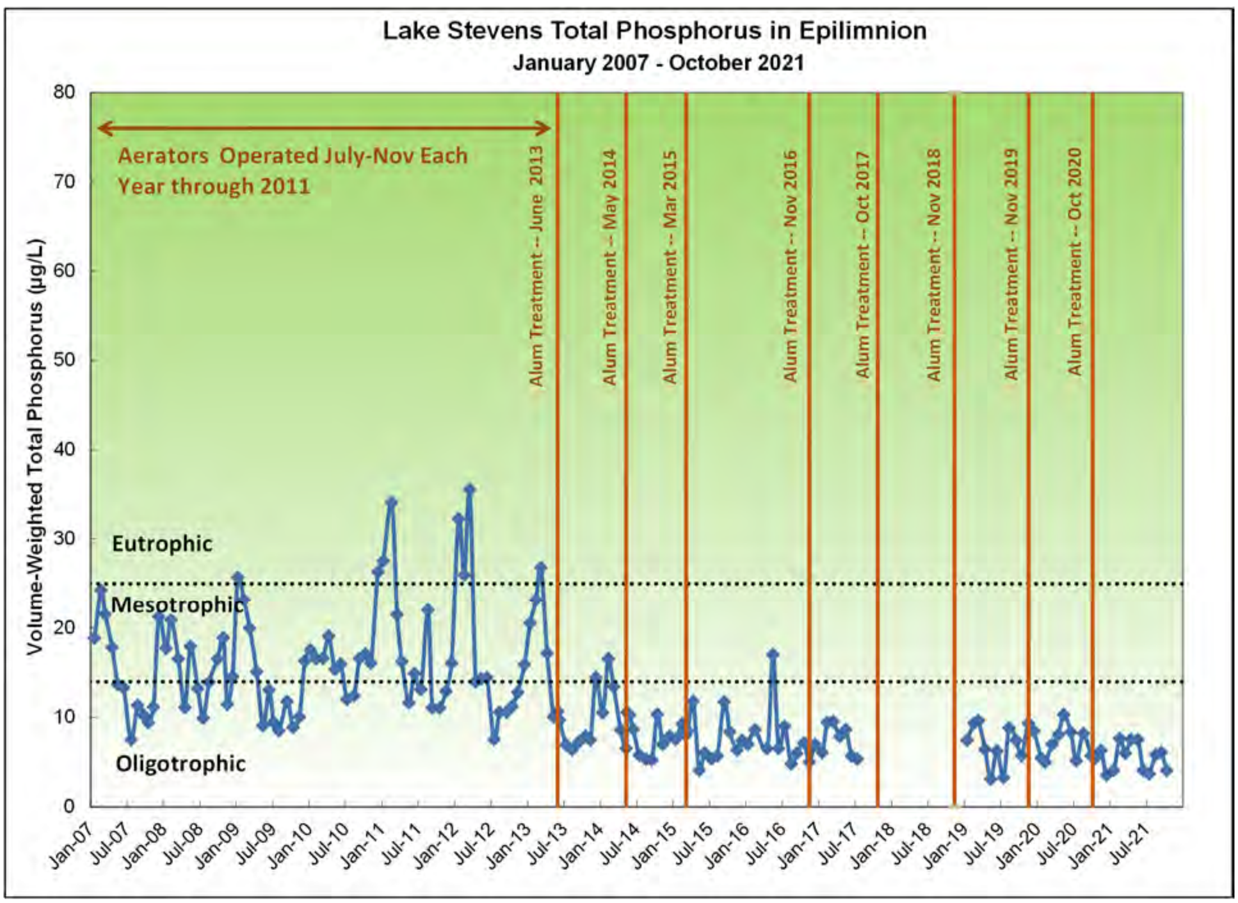


Figure 2: TP Concentrations at 1m In Lake Stevens, 1986 - 2021

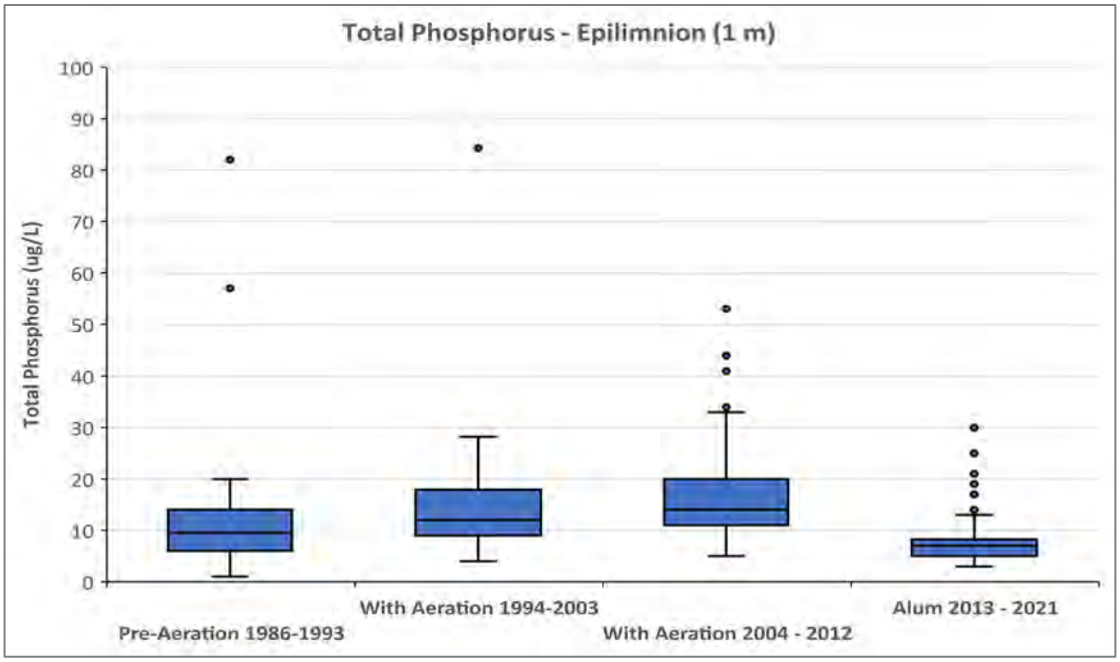
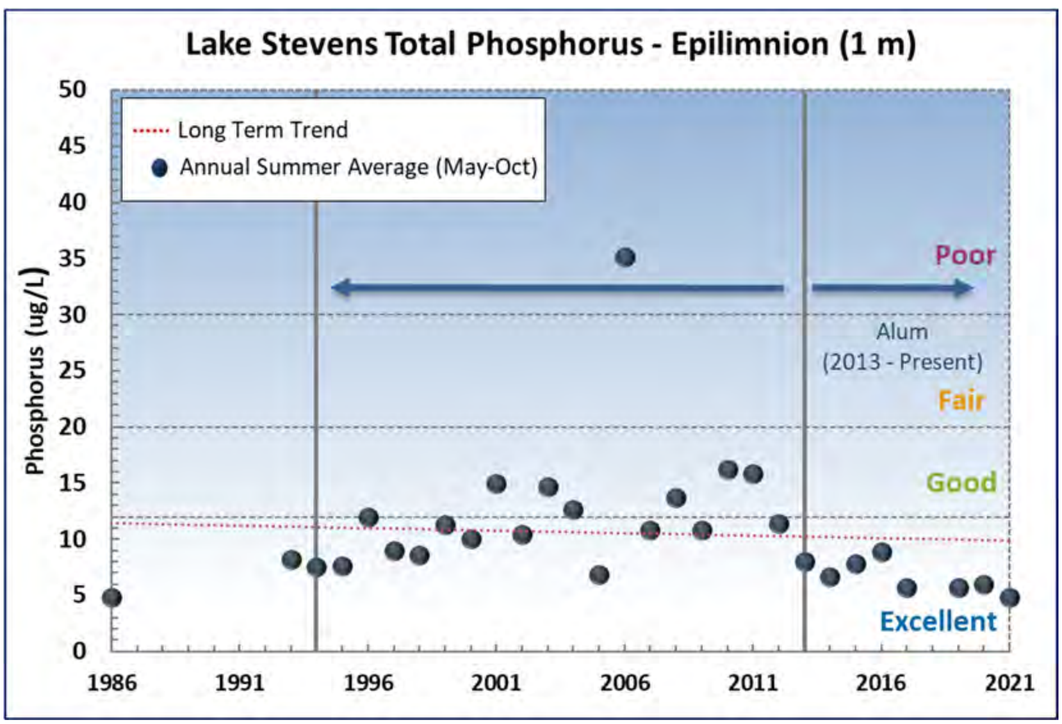


Figure 3. Summer (May - Oct) average TP concentrations at 1 m in Lake Stevens, 1986 - 2021.



Total Phosphorus - Hypolimnion

TP concentrations in the hypolimnion were calculated as a volume-weighted average from 20, 30 and 40 m for 2007 - 2021 (Table 2; Figure 4). Similar to the epilimnion, there was a decrease in both the average and range of hypolimnetic TP following the implementation of alum treatments. Pre-alum volume-weighted hypolimnetic TP ranged from 12.6 to 52 µg/L compared to 3.5 to 27 µg/L after the start of alum treatments (Figure 4). The summer average volume-weighted hypolimnetic TP decreased 68%, from 30.4 ± 8.0 µg/L before alum treatments to 9.6 ± 6.0 µg/L after alum treatments (Table 2).

Not only was there an overall decrease in hypolimnetic TP, but there was also a reduction in the seasonal peaks of phosphorus. Prior to alum when the lake aerator was still operating, there were consistently late summer or early fall TP peaks (Figure 4). These peaks indicated that phosphorus was still being released from the sediments despite aeration. Since alum treatments began, there are only occasional small TP increases in the late summer.

From 1989 - 2021 there has been a decrease in year-round TP at 40 m (Figure 5). Maximum TP at 40 m in Lake Stevens prior to aeration was usually greater than 100 µg/L and in 1991 reached 189 µg/L. With the start of aeration, TP at 40 m was greatly reduced although maximums were still occasionally high, and concentrations were more variable compared to after the start of alum treatments. The outlier TP concentrations greater than 40 µg/L post-alum treatment, shown in Figure 5, were all from 2013, which was the first year of alum treatment.

The long-term TP summer averages at 40 m have also decreased at Lake Stevens (Figure 5). There was an initial drop in TP following the first year of aeration, but TP began to rise in the later aerator years. The decrease in effectiveness was previously identified in the 2012 study due to a lack of iron (Fe) to bind the phosphorus. This conclusion led to the transition from aeration to alum treatment (Snohomish County and Tetra Tech, 2012). Except for the first year of alum treatments (2013), post-alum 40 m TP has been below 25 µg/L averaging around 18 µg/L. The decrease also represents a shift from Good to Excellent conditions as defined by the County in comparison to other lakes (Snohomish County, 2021). There are no trophic classifications established for hypolimnetic TP values.

Figure 4. Volume-weighted hypolimnetic TP concentrations in Lake Stevens, 2007 - 2021.

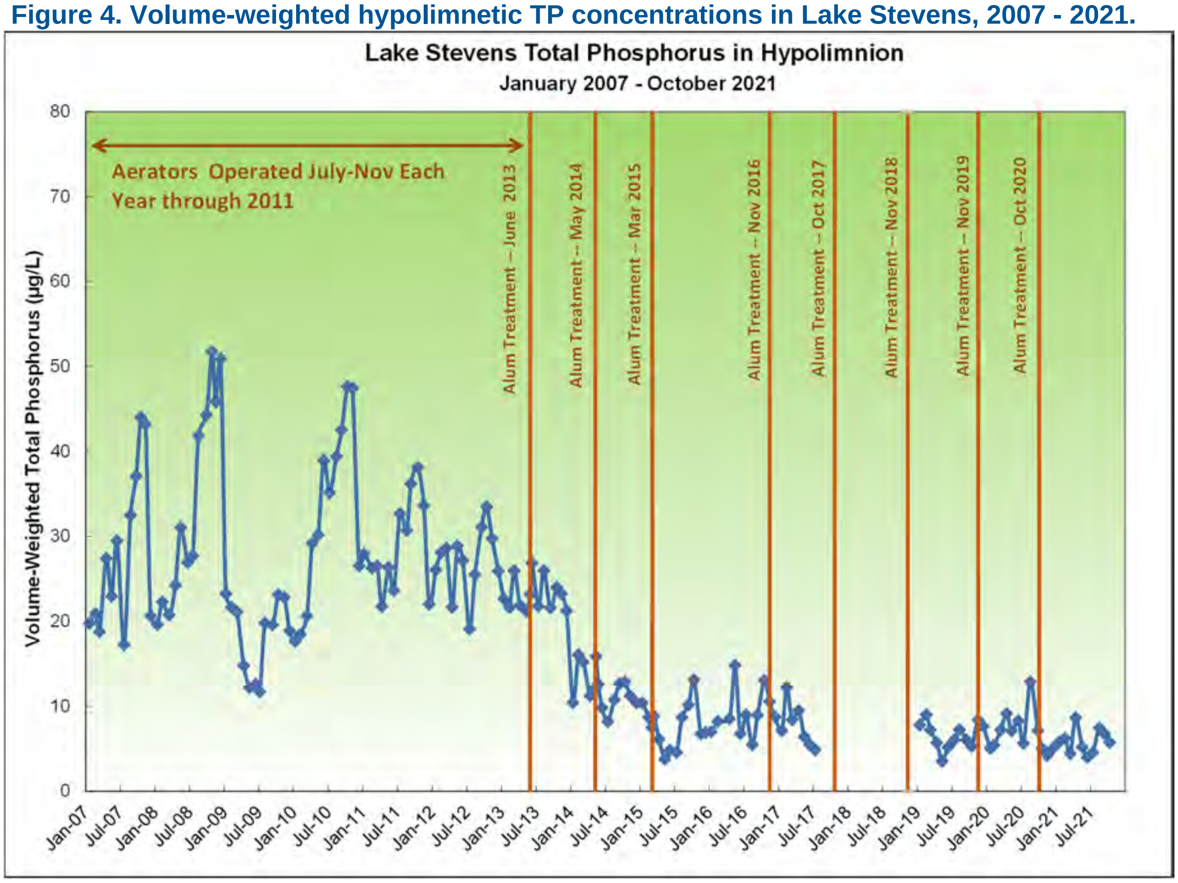
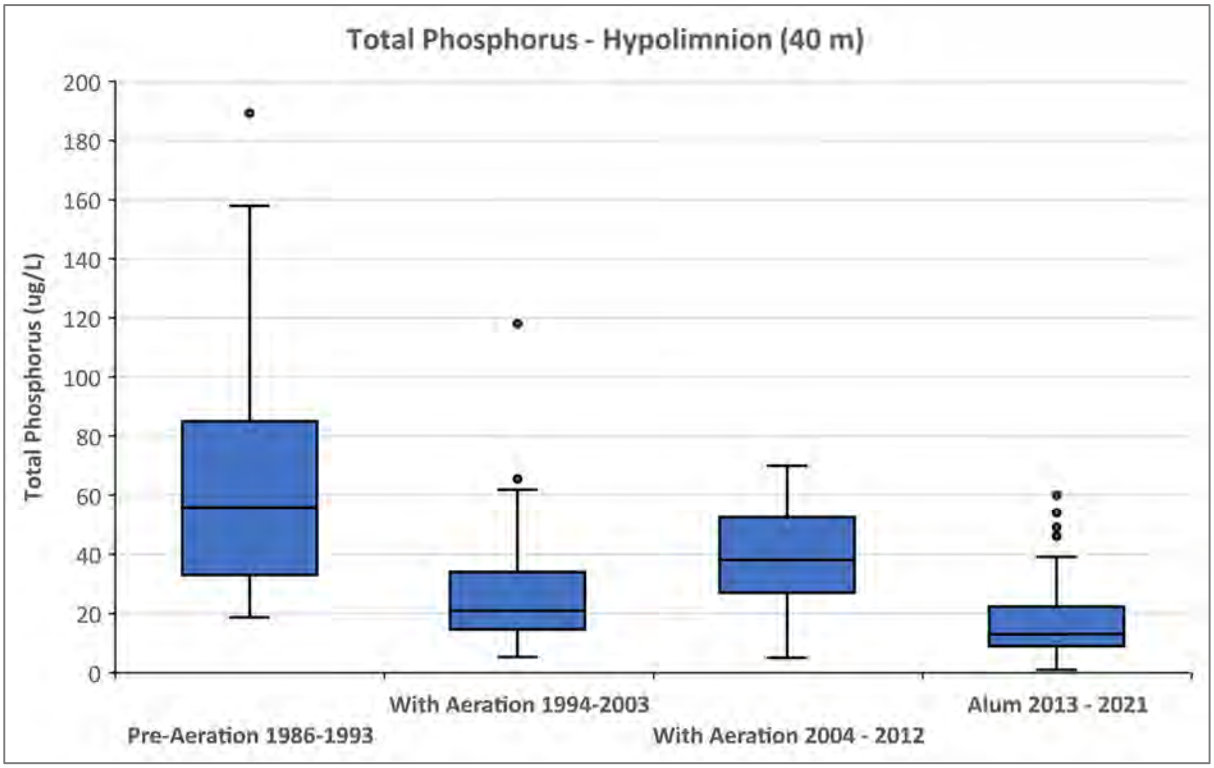


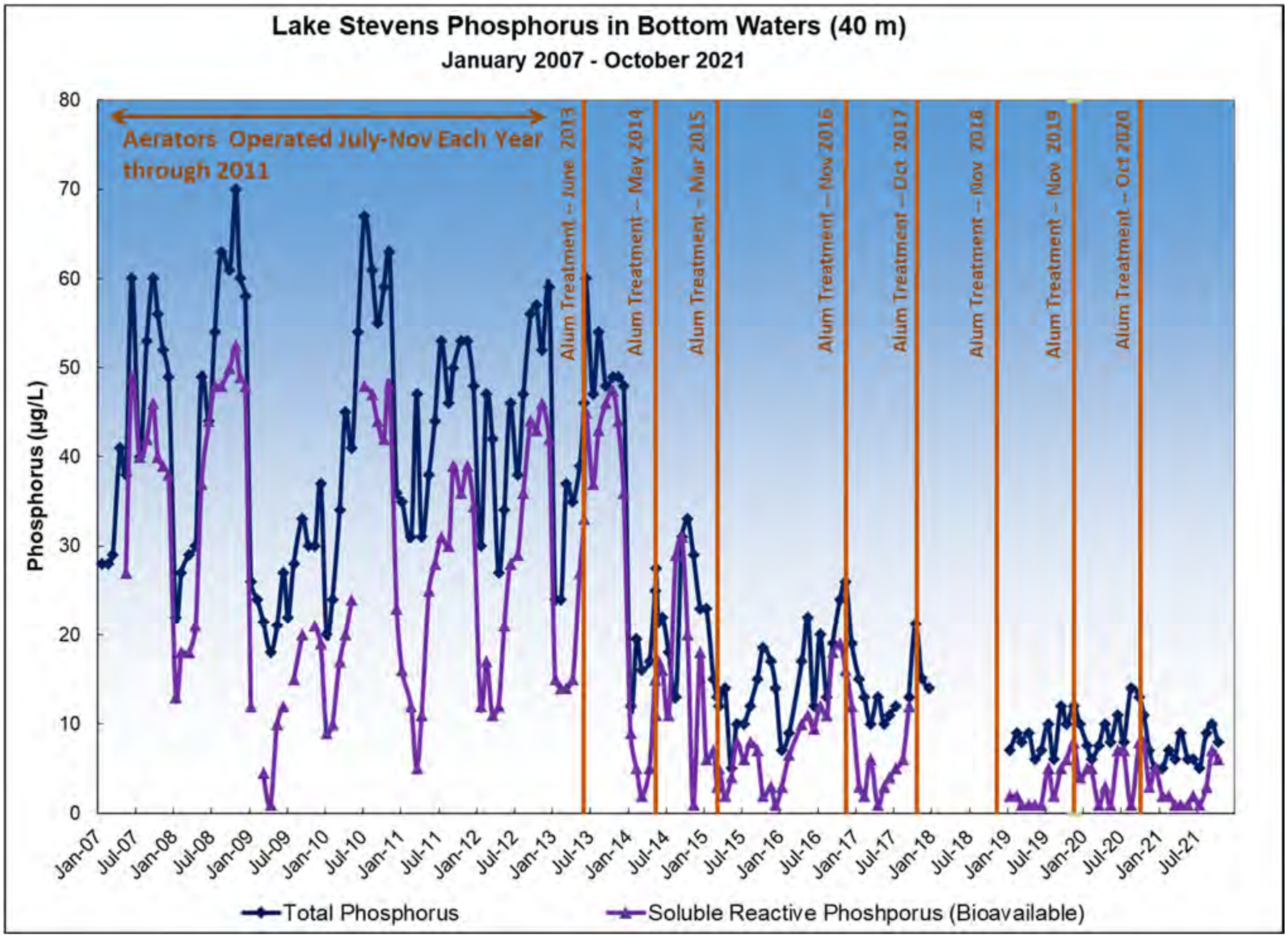
Figure 5. TP concentrations at 40 m in Lake Stevens, 1986 - 2021.



Soluble Reactive Phosphorus - Hypolimnion

Phosphorus released from the sediments is typically in the form of SRP. Therefore, the amount of phosphorus that builds up in the hypolimnion, or the amount of internal loading, can also be seen by looking at the maximum TP and SRP concentrations at 40 m near the lake bottom. The TP maximum at 40 m declined from around 60 to 70 µg/L before alum treatments to 10 to 20 µg/L after alum treatments (Figure 6). For SRP, the 40 m maximum before alum treatments ranged from 21 to 53 µg/L. The 21 µg/L value was from 2009 and was much lower than all other years which typically had SRP peaks around 40 to 50 µg/L. In 2013 and 2014, right after the start of alum treatments, SRP maximums were higher than other post-alum years at 47 and 31 µg/L, respectively. In subsequent years, SRP maximums at 40 m were lower, ranging from 6 to 19 µg/L (Figure 6), despite late summer and fall anoxia in the hypolimnion, which normally would have resulted in higher SRP concentrations due to phosphorus release from the sediments. The decrease in both maximum TP and SRP concentrations at 40 m suggests a decrease in phosphorus release from the sediments. Sediment release rates (SRR) are further discussed in Section 3.5.

Figure 6. TP and SRP concentrations at 40 m in Lake Stevens, 2007 - 2021.



2.3.2. Chlorophyll *a* (chl)

Chlorophyll a (chl) is a measure of algal biomass in the water column. Epilimnetic chl concentrations in Lake Stevens were calculated as an average of the 1, 5 and 10 m samples for 2007 - 2021 (

Table 1 and Table 2; Figure 7). Chl concentrations were low, consistently below 6.0 µg/L, even before the start of alum treatments (Figure 7). Looking at just summer chl concentrations, mean May - October chl averaged 4.1 ± 0.9 µg/L from 2007 - 2012 before alum compared to 2.6 ± 0.7 µg/L after alum (Table 2), thus, confirming algal biomass in Lake Stevens was low before alum treatments, and the start of alum treatments decreased the algal biomass even further.

While chl values were generally low throughout the year, there are typically spring peaks in chl corresponding to the higher levels of epilimnetic phosphorus from both external loading and turnover from the hypolimnion. Prior to alum treatments, the maximum average chl was 9 µg/L or greater in most years with a high of 27 µg/L in March 2011. In 2014 and 2015, the first two springs following alum treatment, chl concentrations were similar to those before alum with larger peaks (Figure 7). From 2016 - 2021, however, chl concentrations were much lower, peaking at around 3 to 6 µg/L in the spring. The timing of annual alum treatments, earlier in the year in 2013 - 2015 versus in late fall in 2016 - 2020, may account for the higher post-alum chl peaks in 2014 - 2015 because the fall treatments appear to have been more effective at reducing the hypolimnetic phosphorus that would have been entrained in the epilimnion during fall turnover.

Historically, average chl in the epilimnion of Lake Stevens has been consistently at or below 9 µg/L (Figure 8). However, peak chl values were much higher prior to aeration in 1986 - 1993, as well as towards the end of the aeration period in 2004 - 2012. Peak chl post-alum treatments has at times been greater than 9 µg/L, but those were spring peaks during the first few years of alum as discussed previously. Chl concentrations post-alum treatments have mostly fallen below the oligotrophic boundary of 3.5 µg/L with an annual median of 2.5 µg/L and an annual average of 3.3 µg/L (Figure 8).

Lakes can also be classified by trophic state using chl. Oligotrophic waterbodies have mean summer chl concentrations of 3.5 µg/L or less, while eutrophic, or highly productive, waterbodies have mean summer chl concentrations of 9.0 µg/L or more (Nurnberg, 1996). Based on these chl ratings, Lake Stevens shifted from a mesotrophic system pre-alum to an oligotrophic system after treatments began (Figure 9). This trend parallels the shift seen in trophic classification based on TP concentrations.

Figure 7. Monthly chl concentrations in the epilimnion of Lake Stevens, 1986 - 2021.

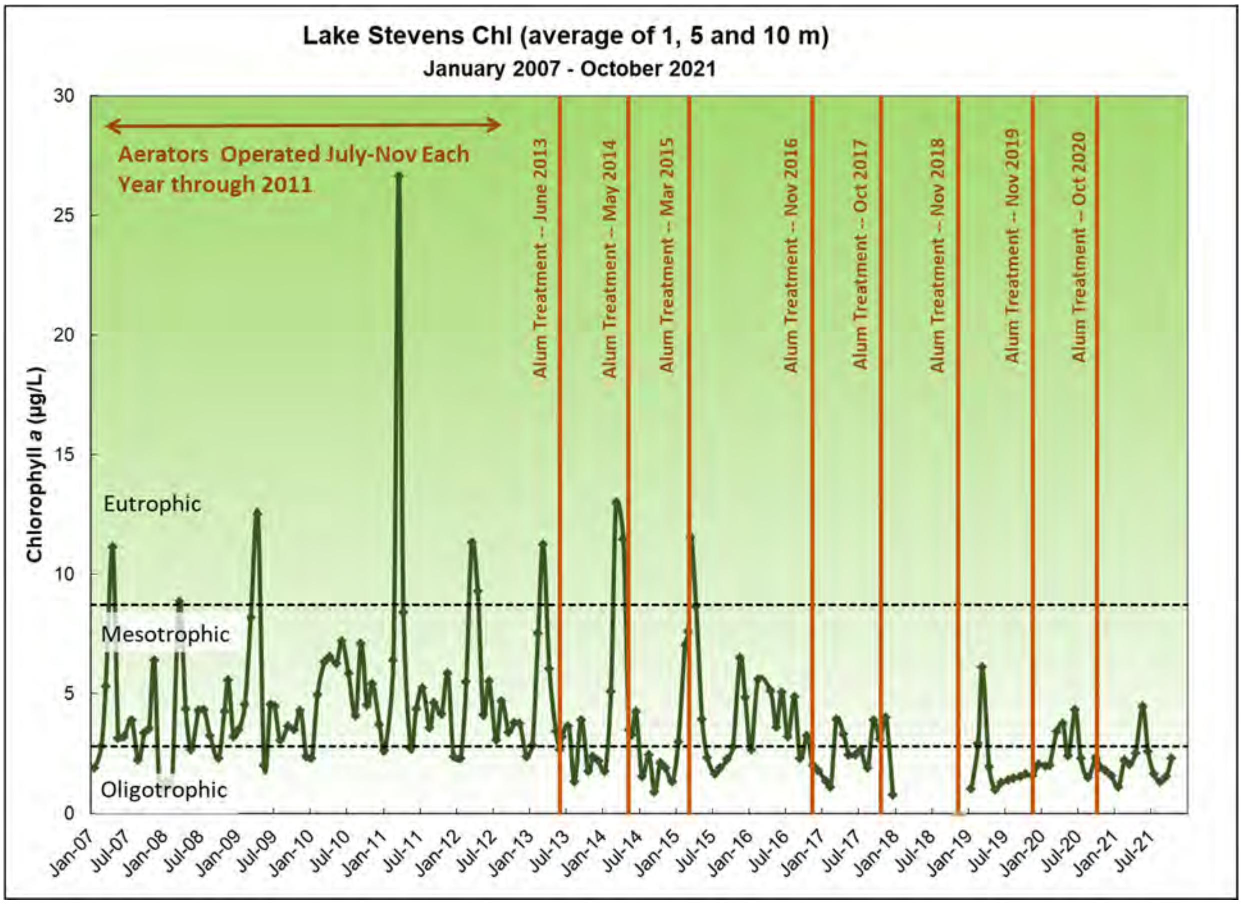


Figure 8. Chl concentrations in the epilimnion of Lake Stevens, 1986 - 2021.

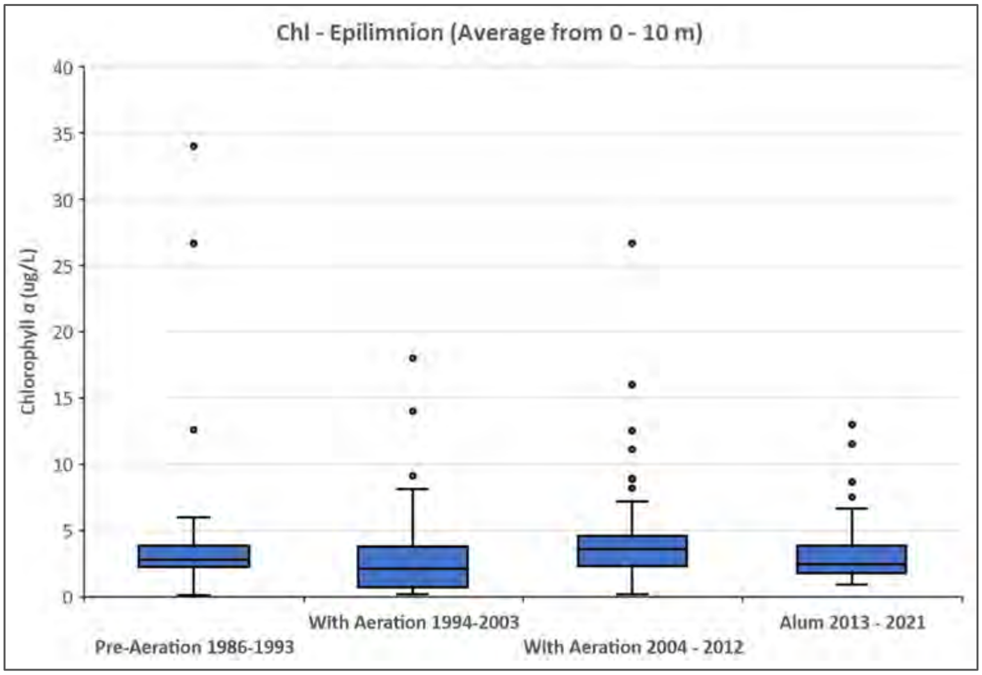
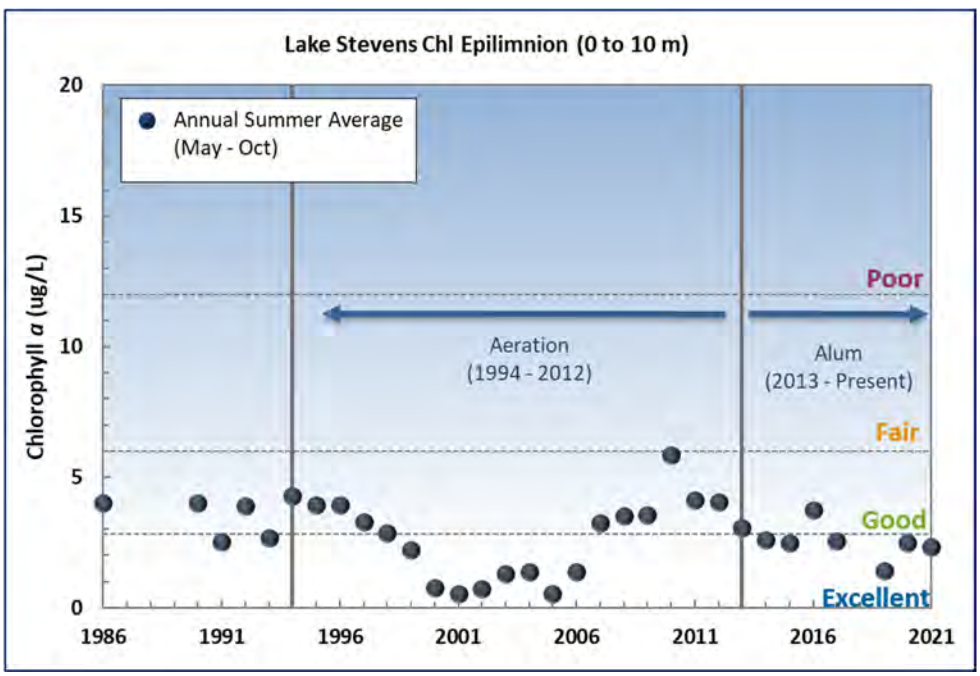


Figure 9. Summer average chl concentrations in the epilimnion of Lake Stevens, 1986 - 2021.



2.3.3. Toxic Algae (Cyanobacteria)

The term “toxic algae” refers to a particular type of algae found in lakes and streams called blue-green algae or cyanobacteria. This type of algae is a natural part of lakes, but sometimes it can grow rapidly or bloom often as a result of high phosphorus levels in the lake. The algae will begin to accumulate as a surface scum which often resembles blue or green paint. Some species of cyanobacteria can produce toxins that can harm people or pets swimming in or drinking the water.

Since the 1980’s, there have been anecdotal reports of Lake Stevens having a history of extensive algal blooms. This was one of the main factors leading to lake restoration work. For example, the 1983 Phase I Restoration Study indicated, “Dense persistent algal blooms were noted throughout the lake…” (Reid, Middleton & Associates). However, there was not consistent monitoring of the lake for algal blooms until 2007.

Since 2007, the lake has experienced occasional nuisance blue-green algae blooms, especially in the late winter and early spring. During each of the the winters of 2011, 2013 and 2015, the algal blooms have been toxic on at least one occasion. Each time, the toxin was microcystin, a liver toxin. In each of these years, toxin values exceeded the WA state recreational standard of 8 µg/L with annual maximums of 28.6 and 23.8 µg/L, respectively. These algae blooms occurred during the off-season for swimming in the lake, but toxins can still affect pets that drink lake water during harmful winter blooms. No blooms have been reported since May of 2016. The lack of significant blooms corresponds to the decrease in TP concentrations following the alum treatments.

2.3.4. Water Clarity (Secchi Disk)

Water clarity is measured with a black and white Secchi disk that is lowered into the water until the distinction in colors is no longer visible. Water clarity varies with several factors including lake depth and size, water color and suspended particles in the water including sediment and algae.

Lake Stevens has consistently had high water clarity since aeration began in 1994. Summer average clarity usually exceeds 6 m (Figure 11). There are occasional low clarity measurements that likely correlate with algal blooms and usually occur in the spring (Figure 10).

Water clarity has continued to increase following alum treatments with mean summer (May - October) clarity increasing from an average of 6.1 m to 7.8 m from pre-alum treatment levels to post-alum treatment, respectively (Table 2). In the past three years, average summer water clarity has been the deepest measured, ranging from 8.3 to 9.9 m (Figure 10). During that period only one value was less than 6 m recorded in May 2021 (Figure 10).

The observed high-water clarity is expected for lakes with similar morphometry to Lake Stevens as it is a deep lake with a large surface area. Lake Stevens also has very little natural color which leads to higher water clarity. During 2010 - 2011, the water color of the lake averaged 6 pcu (platinum-cobalt color units), the second lowest reading in the County. This indicates that water color does not play an important role in the water clarity at Lake Stevens. Water clarity at Lake Stevens.

The last factor that impacts water clarity at Lake Stevens is particles suspended in the water such as algae and sediment. Prior to lake restoration activities, the lake suffered from intermittent algae blooms and overall higher levels of algae as indicated by the chl which typically casue lower water transparency. The lake restoration activities starting with aeration and then alum is expected to improve water clarity as they are designed to reduce phosphorus and thereby reduce algal growth. Alum can also lead to short-term improvements in water clarity as alum binds with particulate and suspended matter from the water column to form a flocculant (floc) which removes the particles and settles to the bottom. As expected, aeration led to a small improvement in water clarity and alum led to further improvements.

Similar improvements in water clarity have been observed in regional lakes following restoration activities. Lake Washington underwent significant changes after wastewater was diverted from the lake. Higher water clarity occurred several years after the wastewater was diverted. The change corresponded to a shift in the dominant zooplankton species to *Daphnia*, which increased the grazing rate of algae. A decrease in cyanobacteria (*Oscillatoria*) was also observed. The Stevens clarity changes may be related to the increases of biomass of large zooplankton grazers described in Section 4.3.2.

While there were shifts in the water clarity following lake restoration treatments, Lake Stevens had even higher than expected water clarity prior to lake restoration activities. Based on the empirical relationships between the Secchi and chl measurements taken prior to the start of alum treatments, the expected clarity of Lake Stevens would be closer to 3 or 4 m (Carlson, 1977). There are similar examples of regional near-oligotrophic lakes with higher than expected water clarity relative to chl. For instance, water clarity was greater than predicted for both Lake Washington (6.9 m, 3 µg/L chl) and Lake Sammamish (5.2 m, 3.6 µg/L chl) (Edmonson and Lehman, 1981; Welch et al., 2019). Higher than expected clarity in Lake Stevens may also be partly related to its long water retention time (5.6 years). For example, clarity increased in Lake Sammamish partly due to a 23% increase in water residence (Welch et al., 2019).

Figure 10. Secchi disk depths measured in Lake Stevens, 2007 - 2021.

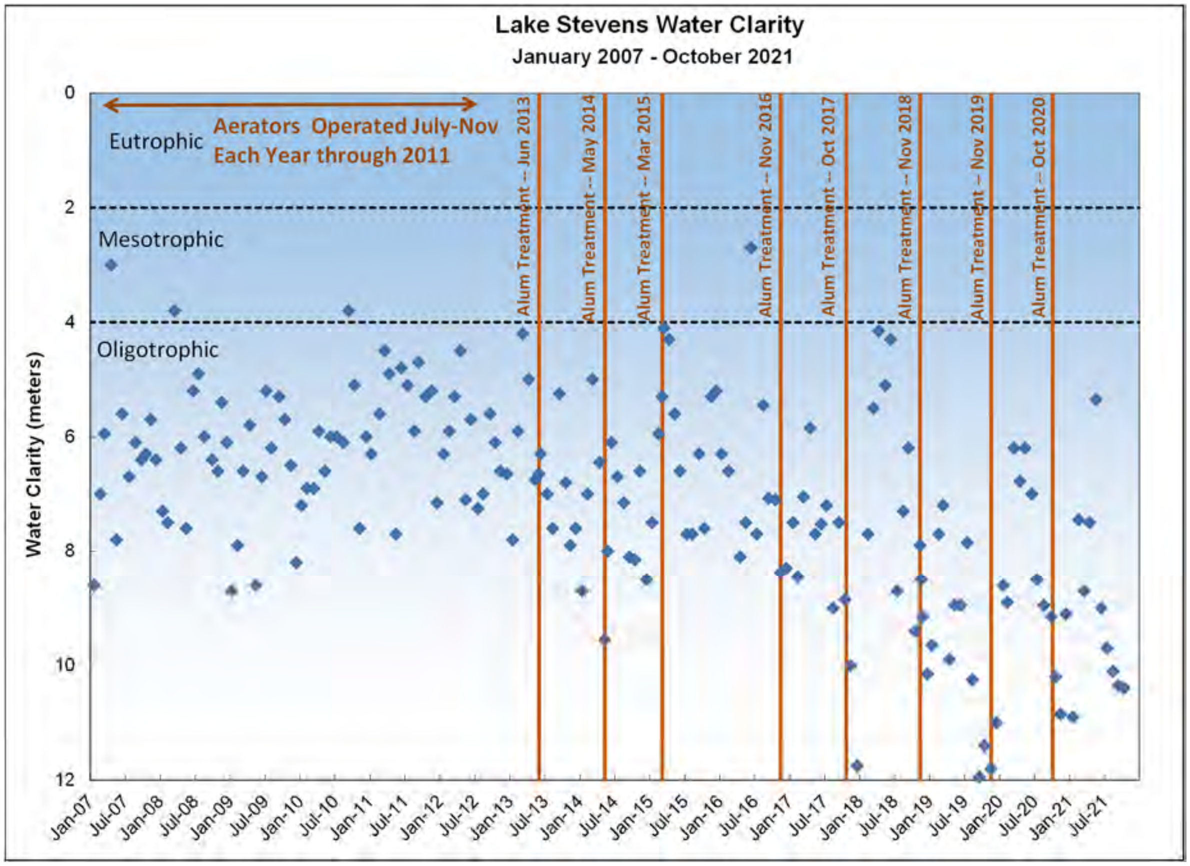
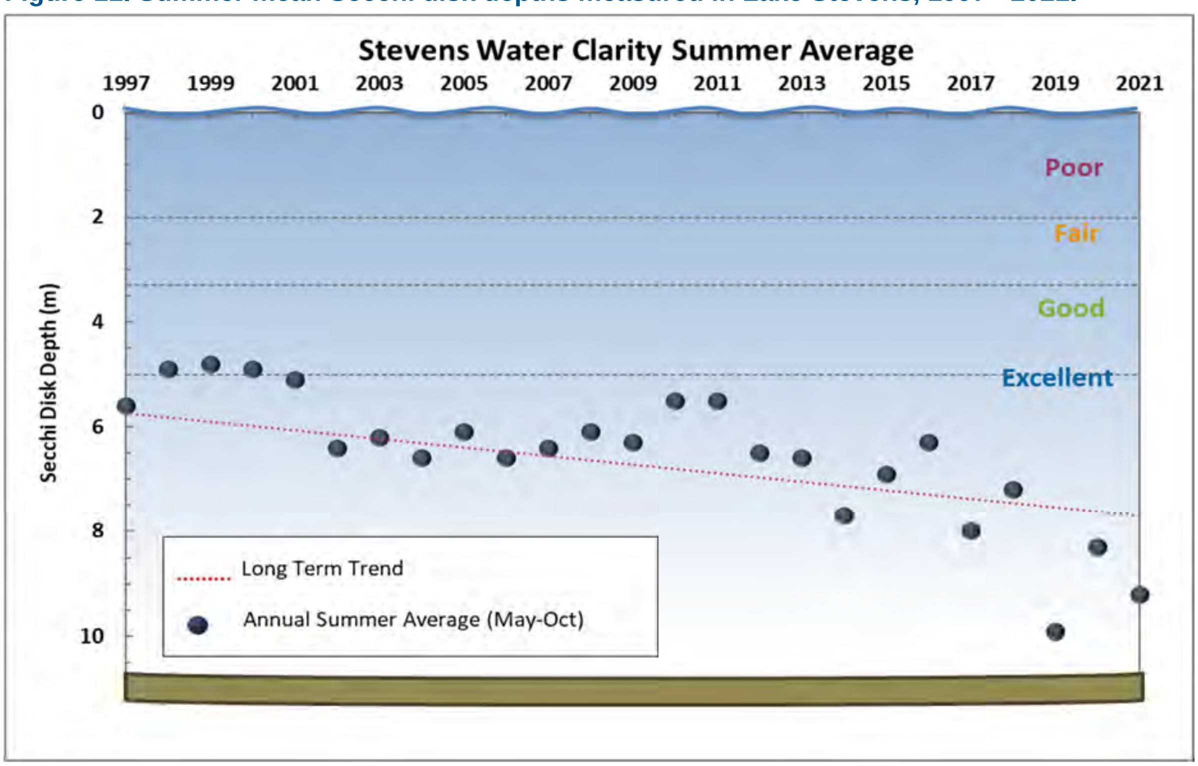


Figure 11. Summer mean Secchi disk depths measured in Lake Stevens, 1997 - 2021.



2.4. WATER QUALITY CONCLUSIONS

The alum treatments are meeting the original lake management goals of reducing the overall amount of phosphorus in the lake, which has reduced algal blooms and especially toxic blooms that pose a risk to lake users. In comparison to the late years of operating the aerator in the lake, hypolimnetic and epilimnetic phosphorus concentrations have been reduced by one-half to two-thirds. Chl concentrations in the lake have also decreased, and there has been no incidence of potentially toxic cyanobacteria blooms since 2016. SRP in the hypolimnion also decreased, indicating reduced internal loading of phosphorus.

The alum treatments have reduced the effect of external phosphorus loading by binding with phosphorus and removing it out of the water column and immobilizing that phosphorus when it settles out in the sediment. Without the annual alum treatments, any phosphorus that enters the lake from external sources may increase the sediment phosphorus concentration over time. In 2012, it was determined that 40% of the annual phosphorus load to the lake was from external loading. The City has been actively working to reduce external phosphorus loads through best management practices and source control programs. There is no quantitative data for the amount of phosphorus that enters the lake, therefore, the magnitude and timing of the external loading is unknown. Future water quality monitoring for TP from the lake inlets is recommended to determine the magnitude and timing of external phosphorus loading.

The annual alum treatments were initially designed for spring application to remove most of the hypolimnetic phosphorus entrained into the epilimnion during fall turnover. The timing for alum treatments moved from a spring application to a fall application in 2016. The change in alum treatment timing has proven more effective at reducing the hypolimnetic phosphorus that would be entrained into the epilimnion during fall turnover and subsequently has reduced the magnitude of the spring algae bloom as well as the summer algal biomass. It is recommended that alum treatments continue to be implemented in the fall.

3. SEDIMENT CORE ANALYSIS

3.1. SEDIMENT ANALYSIS OBJECTIVES

The City manages phosphorus in Lake Stevens according to the 2013 Phosphorus Management Plan (City of Lake Stevens, 2013). The 2013 plan outlines the use of annual alum treatments to immobilize the phosphorus in the lake sediments as well as remove phosphorus from the water column. The purpose of this sediment study is to assess the effect of annual alum treatments on reducing the potential for phosphorus release from the lake sediments and make recommendations for future phosphorus management techniques that will be used to update the 2013 Phosphorus Management Plan for Lake Stevens.

Two different strategies were used to assess the impact of the alum treatments. First, the SRR was calculated for each year to identify changes in the SRR before treating the lake with alum and after the start of alum treatments. Next, sediment cores were collected in 2021 and analyzed to determine 1) the overall phosphorus concentration in the sediment, 2) the different forms of phosphorus, and 3) the concentrations of analytes to which phosphorus can bind (e.g. iron, aluminum). These results were then compared to previous sediment studies to understand how much of the legacy phosphorus had been neutralized and how much is still available for potential release from lake sediments. Collectively, the analysis will help to determine the future potential for phosphorus sediment release and guide future phosphorus management activities.

3.2. SEDIMENT CORE METHODS

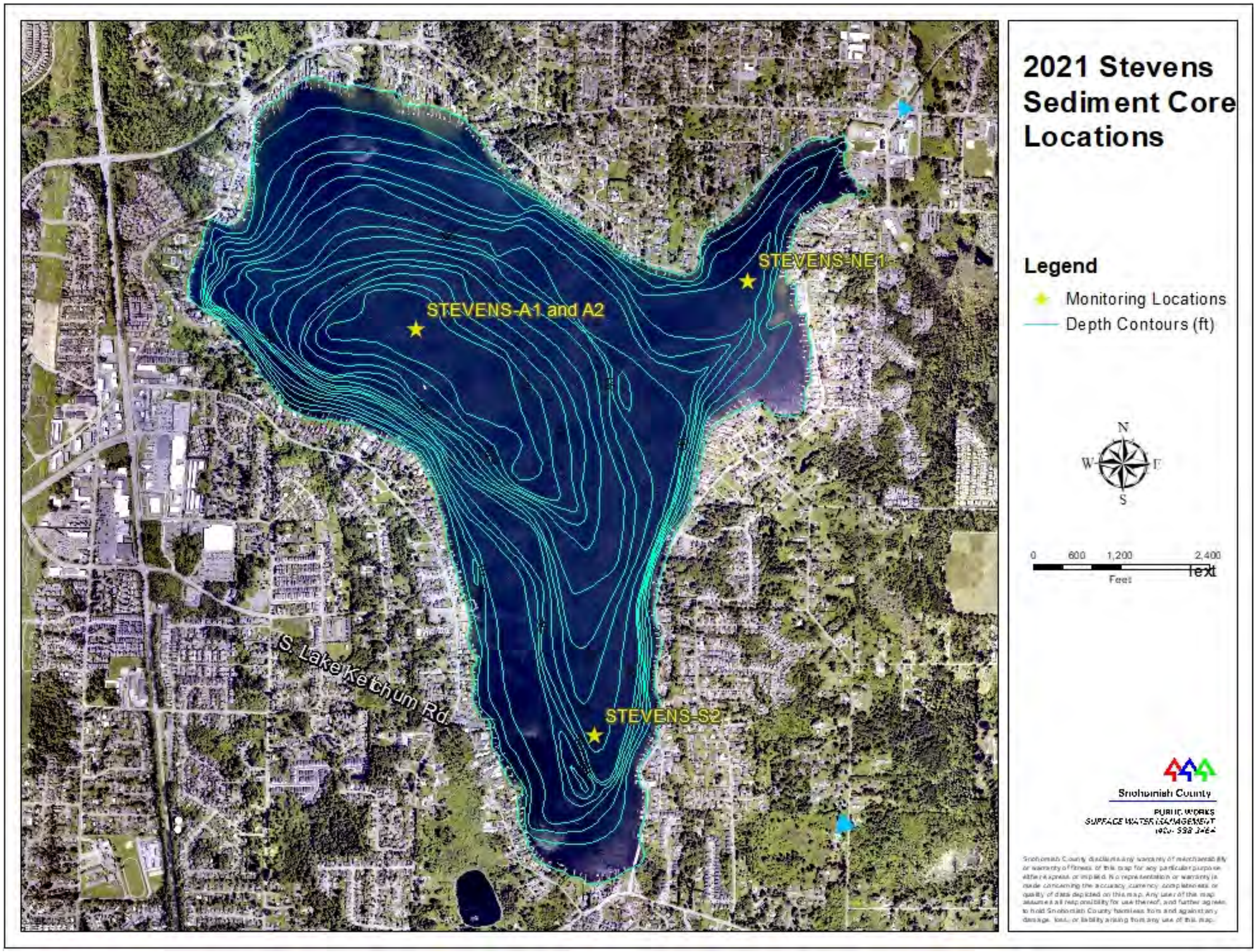
The County collected three sediment cores from three regions in Lake Stevens on July 15, 2021 using a piston corer. The sediment cores were collected at the same approximate locations as the 2009 sediment study (Tetra Tech, 2009) – one core from the deepest hypolimnetic sediment near the historical aerator at approximately 160 ft (A1), one core from the epilimnion zone near the outlet bay at 40 feet (NE1), and one core from the south end of the lake at 80 ft (S2) (Figure 12). The sediment cores were delivered to IEH Analytical Laboratories (formerly Aquatic Research) and analyzed for the following parameters at depths of 0-2, 2-4, 4-6, 6-10, 10-15, 15-20, 20-25, 25-30 and 30-35 centimeters (cm):

- Total phosphorus (TP)
- Biogenic phosphorus
- Total organic phosphorus
- Labile (loosely-bound) phosphorus
- Iron-bound phosphorus (Fe-P)
- Aluminum-bound phosphorus (Al-P)
- Calcium-bound phosphorus (Ca-P)
- Total Aluminum
- Total Iron
- Total Calcium
- % solids
- % water
- % volatile solids

To determine the different phosphorus sediment fractions, a series of sediment digestions were used as outlined in Rydin and Welch (1998). Additional details on the sediment collection procedures and laboratory analysis can be found in the Quality Assurance Project Plan for Lake Stevens Sediment Study (Snohomish County, 2021).

The rate of increase in hypolimnetic phosphorus, or the net SRR, was calculated for each summer during 2007 - 2021. The volume-weighted hypolimnetic TP concentration, determined using TP from 20 m to 40 m, was converted to hypolimnetic mass using the estimated volume of the hypolimnion from 20 m and deeper. The SRR was calculated as the rate of change in hypolimnetic TP mass over a specified period during stratification, usually April/May to October/November. This is divided by the hypolimnetic area contributing to the release of phosphorus. For comparison, SRRs were calculated using two different hypolimnetic areas; the entire hypolimnetic area from 20 m and deeper, and the deepest area only below 40 m. SRR are expressed in mg/m^2-day.

Figure 12. Lake Stevens sediment core locations, 2021.



3.3. PHOSPHORUS SEDIMENT FRACTION BACKGROUND

In addition to analyzing the TP in lake sediments, it is also important to identify the different fractions of phosphorus in the lake sediments. As shown in Figure 13, TP is comprised of several phosphorus fractions which can be divided into two main categories, available phosphorus, and stable phosphorus, as follows:

Available Phosphorus

Available phosphorus is the portion of the sediment phosphorus that is available to be recycled back into the lake as internal loading. It is comprised of mobile phosphorus (mobile-P) and biogenic phosphorus (biogenic-P).

- **Mobile-P** is made up of loosely-sorbed and iron-bound phosphorus.
    - **Loosely-sorbed phosphorus (Loosely-sorbed P)** is phosphorus that is not chemically bound but physically attracted to other particles like organics and clay. It can be released when sediment is re-suspended and is not dependent upon oxygen conditions. It is typically only a small portion of the mobile phosphorus.
    - **Iron-bound phosphorus (Fe-P)** – The other portion of mobile phosphorus is iron-bound phosphorus. It can be released or mobilized during anoxic (i.e., lack of dissolved oxygen [DO]) conditions. Under oxic conditions (i.e., presence of DO), iron exists in the ferric form to which phosphorus chemically binds. In anoxic conditions, iron is reduced, and phosphorus is released. This results in an upward diffusion of phosphorus through the sediments to the overlying water, contributing to internal loading.

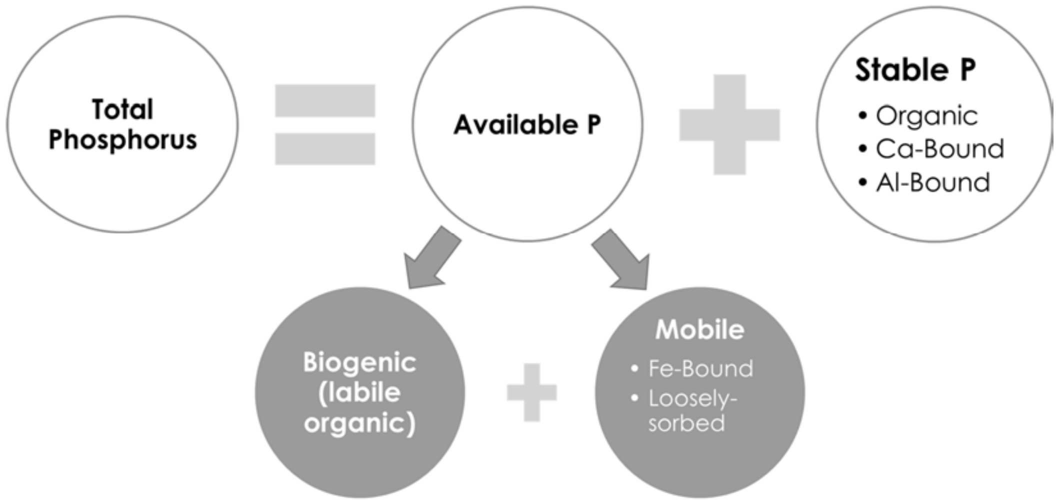
- **Biogenic-P** is largely composed of organic material which has not completely decayed to refractory (stable) organic material. This material may also include some bacteria, fungus, and algal cells that have recently died or are still viable. It also consists of some labile-phosphorus which is already soluble. Unlike mobile-P, biogenic-P can become available in both anoxic and oxic conditions and is released at a higher rate in oxic conditions. This is because aerobic metabolism is generally faster than anaerobic metabolism. A portion of biogenic-P can become available in any given year and contribute to internal loading. However, the specific mechanisms that drive biogenic-P release are not fully understood and are still being explored. Recent studies have found that 8 to 30% of total biogenic-P can mobilize and become soluble (NALMS, 2019). Note that the 2009 sediment study did not include biogenic-P as it was conducted prior to the inclusion of biogenic-P in routine sediment core analysis.

Stable Phosphorus

Stable phosphorus is the portion of the sediment phosphorus that is unavailable for recycling back into the water column as internal loading under normal conditions within sediments typical of Western Washington lakes. It is comprised of the following:

- **Aluminum-bound phosphorus (Al-P)** is stable under typical ranges of pH (6.0 to 7.8 in the sediments). This renders phosphorus inactive and unavailable to overlying water, regardless of DO at the sediment-water interface and in the overlying water (Cooke et al., 2005). The goal of alum treatments is to convert mobile phosphorus into aluminum-bound phosphorus instead of iron-bound or mobile forms such as loosely-bound and biogenic-p.
- **Calcium-bound phosphorus (Ca-P)** is also a stable bond not affected by redox conditions and remains in the sediment under typical lake conditions.
- **Organic phosphorus (organic-P)** is composed of fully decayed or refractory organic matter that is no longer readily released from the sediment under pH ranges typically found in the lake bottom. However, a portion of the organic-P, measured as biogenic-P, may be available for release into the water column.

Figure 13. Phosphorus fractions analyzed in sediment cores.



3.4. SEDIMENT ANALYSIS RESULTS AND DISCUSSION

Sediment core analytical results were completed by IEH Analytical laboratories on October 8, 2021 and are shown in Table 3 and Table 4. The phosphorus fractions, aluminum, and iron were compared to sediment data collected in the 2009 sediment study (Tetra Tech) as well as in 1981 (Reid, Middleton & Assoc.,1983). The 1981 profile is assumed to begin at the 15 cm depth of the 2021 and 2009 cores based on the estimated sedimentation rate that would result in the burial of the older sediments.

3.4.1. Sediment Core Extraction Limitations

The sediment phosphorus fractionation methodology used to analyze the sediment cores was developed by Rydin and Welch (1998) which is the standard method for determining sediment phosphorus fractions. However, based on the results from Lake Stevens and other similar lakes, it has recently been identified that the less stable forms of aluminum minerals that are present after repeated alum treatments (amorphous aluminum hydroxide) may release phosphorus during the extraction process. Specifically, it was discovered that the Fe-P extraction process may release some of the phosphorus from the freshly formed aluminum mineral to the extraction solution (Huser, 2021; personal communication). This could lead to an overestimation of Fe-P in the sediments, especially near the surface with freshly formed Al-P. It may also lead to an underestimation of Al-P, although that has not been confirmed.

The issue with the sediment analysis has only recently emerged as typical past analysis has been performed on lakes without alum treatments or those where core samples are taken several years after an alum treatment where the aluminum in these sediments is typically more stable. Lake Stevens is one of a handful of lakes in Western Washington that have been treated with a low annual dose of alum, and the same analysis issues have been observed in other sediment cores from similar lakes. For example, the atypical results from Lake Stevens have been observed in a similar sediment study conducted at Lake Ketchum, Washington. Lake Ketchum was also treated with small annual alum doses and has also observed a reduction in SRR. Experiments are currently being designed to study the potential issues in the fractionation method and to determine the appropriate method revisions to correct the issues (Huser, 2021; personal communication).

Given the newly identified extraction issue, the results for Fe-P in the Lake Stevens cores presented should be considered biased high. Therefore, the Fe-P results should not be used to determine the remaining amount of mobile phosphorus in the Lake Stevens sediments, nor should they be used to assess the effectiveness of the treatments in converting Fe-P to Al-P. In addition, results for Al-P may also be affected by this potential extraction issue and should be considered estimates. All other sediment results presented should be unaffected.

3.4.2. Total Phosphorus (TP)

TP was determined as the sum of the various phosphorus fractions for each sediment interval. TP concentrations in the surface sediments were higher in 2021 compared to 2009 at both the deep (A1) and south lake (S1) stations (sample locations, Figure 12). At depths of 6 cm and greater, TP was consistently lower than at the surface and was similar to the concentrations measured in 2009 (Figure 14).

Higher concentrations of TP in 2021, compared to 2009, may be a result of continued external phosphorus loading from the lake watershed. However, given the large difference in cores, it is more likely that the high TP concentrations in 2021 may have resulted from upward diffusion of SRP from the lower depths of lake sediments. Upward diffusion occurs through the sediment porewater and is known to occur following alum treatments. Mobile-p was found to diffuse from deeper sediment layers to bind with the added aluminum floc in both Lake Süsser See, Germany (Lewandowski et al., 2003) and in Green Lake, Seattle, Washington (Welch et al., 2017). In Lake Stevens, the upward diffusion is likely because lower sediments had higher concentrations of TP based on the 1981 cores. TP concentrations in 1981 ranged from approximately 10,000 to 15,000 mg/kg in the top 4 cm of the core (as shown at 15 cm depth in Figure 14). The historic core TP levels are much higher than most comparable lakes. For example, TP in the top 5 cm of Lake Washington had historic sediment concentrations of 1,000 to 2,000 mg/kg and only increased to 5,500 mg/kg, with direct discharges into the lake of 11 wastewater treatment plants (Shapiro et al., 1971).

3.4.3. Stable Phosphorus

As discussed previously, the three main stable forms of phosphorus in lake sediments are Al-P, Ca-P and stable organic phosphorus. Table 3 shows the full sediment profile of each of these sediment fractions. The focus of this analysis will be on Al-P, as the goal of the alum treatments is to convert phosphorus that is in available forms to stable Al-P.

Aluminum and Aluminum-Bound Phosphorus

To understand Al-P, it is helpful to first look at the aluminum concentrations in the sediments. Aluminum concentrations would be expected to increase given the annual addition of aluminum through the alum treatments. The annual alum treatments from 2013 - 2019 added a total of 21 g/m2 of aluminum to the lake. Figure 15 shows aluminum sediment profiles in 2021 compared to 2009 and 1981. Aluminum in the top 6 cm of the deep core (A1) nearly doubled in 2021 compared to 2009 (average 15,900 mg/kg versus 9,400 mg/kg). However, aluminum concentrations in the other two 2021 cores decreased slightly or were similar to those measured in 2009.

Results were similar for Al-P (Figure 16). In the top 6 cm at the deep station (A1), Al-P increased from an average of 600 mg/kg in 2009 to 4,000 mg/kg in 2021 (Figure 16). There was also an increase at the shallow NE1 station from 243 mg/kg to 388 mg/kg in 2021. Conversely, at the south station (S1), Al-P in the top 6 cm was less in 2021, 351 mg/kg, compared with a 2009 average of 465 mg/kg. At 10 cm and below, Al-P was similar to levels in 2009 at all stations.

The larger increase in total aluminum and Al-P observed in the deepest portion of the lake compared to shallower areas is likely due to low-density particles, including alum floc, migrating toward deeper areas. This focusing of low-density particles is a typical phenomenon observed in lakes.

Aluminum to Phosphorus ratio (Al:P)

The ratio of Al:P indicates how much sediment phosphorus is bound with aluminum and is unavailable to be released back into the water column. The ratios of Al to P ranged from 1.7 to 7.1 in the deep core (A1), 5.8 to 10.9 in the shallow core (NE1), and 5.8 to 11.4 in the south core (S2) (Table 4). These ratios are similar to those in

2009 which had ratios of 5:1 to 10:1, and higher than those near the surface in 1981, which had an average 3.2:1 in the top 8 cm (Tetra Tech, 2009). The increase in Al:P means that more of the sediment phosphorus is bound to aluminum than in 1981 and is unavailable for recycling back into the water column. Also, the sediment with high phosphorus, relative to aluminum, is now buried by more recent sediment. However, the low Al:P ratio at depth may mean some of the deeper high phosphorus could diffuse upward to the surface sediments as it is not bound with aluminum.

Another indicator of alum treatment effectiveness is the amount of aluminum added relative to the amount of Al-P formed. The annual alum treatments from 2013 - 2019 added a total of 21 g/m^2 of aluminum to the lake. The amount of Al-P formed from the annual alum treatments, based on the increase in Al-P in the top 6 cm at the deep station (A1), was 16 g/m^2, which was the most notable increase. Therefore, the ratio of Al added to Al-P formed is 1.3:1. This low ratio indicates that the alum was very efficient at binding sediment phosphorus. The ratio observed in Lake Stevens is lower than usually observed in other lakes following sediment inactivation treatments. The average Al added to Al-P formed ratio observed in eight Washington lakes at 7 to 21 years after alum treatments was 10.7 ± 0.7 (Rydin et al., 2000). For example, the Al added to Al-P formed ratio in Green Lake, Seattle, Washington, went from 15:1 to 3.7:1 over 11 years after the start of alum treatments in the top 25 cm. The decrease in Al added to Al-P formed ratio in Green Lake indicates that the aluminum added continued to bind phosphorus diffused from deeper sediments and from external input. Lowered Al added to Al-P formed ratios mean greater continued efficiency of added alum to bind phosphorus over time. However, it should be noted that the amount of Al added to Green Lake was significantly higher than that added to Lake Stevens.

3.4.4. Available Phosphorus

The available phosphorus in lake sediments is the amount of phosphorus that could be potentially released into the water column. It is a key factor for understanding the potential of phosphorus release from lake sediments and is used to plan for and determine dosing for alum treatments.

Available phosphorus is the combination of mobile-p and a portion of the biogenic-p. Mobile-p is comprised of loosely-sorbed and Fe-P. Concentrations of loosely-sorbed P were mostly negligible in both 2009 and 2021 with levels being just above or below the detection limit of 2 mg/kg (Table 3). Therefore, this section will primarily focus on Fe-P and biogenic-P.

Iron-Bound Phosphorus

Iron-bound phosphorus is the portion of the available fraction that is usually the major mobile source of internal loading in lakes. Accordingly, the conversion of Fe-P to stable Al-P is the primary objective of alum treatments (Tetra Tech, 2009; Tetra Tech, 2012). A decrease in Fe-P corresponding to the observed increase in Al-P has been documented in sediment core profiles at several alum-treated lakes. A laboratory-analyzed sediment core from Lake Delavan, Wisconsin, showed that Fe-P decreased from 400 mg/kg to zero with the addition of 100 mg/g of Al (Cooke et al., 2005). Following two alum treatments at Green Lake, Washington (one in 1998 and one in 2004), the Al has continued to bind diffused phosphorus from depth and external inputs over time. At the index station in Green Lake, mobile phosphorus went from 450 to 650 mg/kg in 1998 to around 100 mg/kg after the 2004 treatment to near zero in 2015, 11 years post-treatment (Rydin and Welch, 1999; Welch et al., 2017).

The sediment core results from Lake Stevens are contrary to expectations where, unlike iron concentrations, Fe-P concentrations increased in the top 6 cm at all three sediment core stations (Figure 17). Concentrations increased ten-fold at the deep station (A1) from an average of 350 mg/kg in 2009 to 4,000 mg/kg in 2021. Concentrations at the shallow station (NE1) and shallow station (S2) nearly doubled from 2009 with averages increasing from 95 to 170 mg/kg and 334 to 616 mg/kg, respectively. The concentrations at depths below 6 cm were mostly higher in 2021 at the A1 and NE1 stations, but the magnitude of the difference was much smaller than in the surface sediments. The shallow S2 station had similar results in 2009 and 2021 at the lower depths.

Not only are the Fe-P results contrary to expectations, but they also do not align with other water quality metrics. A high and increasing amount of sediment Fe-P would indicate there is a growing amount of available phosphorus that would lead to higher SRR. As discussed previously, and in the companion water quality memo, both hypolimnetic TP and SRP concentrations and SRRs are much lower now following the eight years of alum treatments.

As indicated previously, the higher-than-expected Fe-P concentrations in the Lake Stevens sediment cores are likely overestimated based on the sediment extraction issue previously discussed. Given the potential issue in the methodology, the Fe-P results should not be used. Since the true amount of available phosphorus remaining in the sediments could not be determined, the observed changes in hypolimnetic TP and SRP and the measured SRRs should be used to assess the impact of the annual alum treatments.

Iron to Phosphorus ratios

The ratio of total Fe to TP determines the ability of a lake to successfully bind external phosphorus in the sediments. The amount of total Fe determined in the sediment cores is not impacted by the potential extraction methodology issue that resulted in higher-than-expected Fe-P concentrations. The average Fe:P ratios in 2021 ranged from 6.2 to 8.3, with the lowest ratio in the deep core, A1, and the highest ratio in the core near the south end of the lake (S1) (**Table 4**). The average Fe:P ratio in the 2009 cores was slightly higher at 11:1 (Tetra Tech, 2009). Ratios in both years are below the 15:1 ratio needed to bind phosphorus. The 15:1 ratio is well established from research in lakes around the world (Cooke et al., 2005). Below this Fe:P ratio, iron is not the primary control of phosphorus internal loading. The 1981 sediment analysis found that the Fe:P ratios in surface sediments (0 to 4 cm) averaged only 4.6, less than the ratios in either 2009 or 2021. Sediment profiles of iron are shown in Figure 18.

Biogenic phosphorus

Biogenic-P is considered part of the mobile phosphorus pool and a portion of it, usually about a third, is used to determine the aluminum dose for a sediment inactivation alum treatment. However, it has not been shown to change following alum treatment in other lakes. Biogenic-P in the top 20 cm ranged from 300 to 450 mg/kg in Green Lake at the deep index station and did not change following treatment (Welch et al., 2017). Biogenic-P in Lake Stevens sediment, top 25 cm, was similar to Green Lake, ranging from 127 to 1000 mg/kg, with an average of 370 mg/kg. Biogenic-P measured in Lake Ketchum sediments in 2009 and 2021 averaged 400 and 450 mg/kg, respectively, just slightly higher than in Lake Stevens. There is no biogenic-P data from Lake Stevens cores collected in 2009; however, organic-P was determined. On average biogenic-P in Lake Stevens made up 66% of the total sediment organic-P in 2021. Based on this percentage, biogenic-P in the 2009 cores can be estimated from organic-P concentrations. Thus, the average biogenic-P in the top 25 cm in 2009 was estimated to be 400 mg/kg, just slightly higher than the average from 2021. This indicates that there most likely has not been a depletion of biogenic-P following alum treatment. Furthermore, the biogenic-P in the sediment does not appear to have a large contribution to internal loading or buildup of phosphorus in the hypolimnion.

Figure 14. Sediment TP content in the top 35 cm before and after hypolimnetic aeration ceased in 2012 and after annual alum treatments started in 2013. Site locations are A1 (deep - 160 ft), S2 (south end - 80 ft), and NE1 (epilimnion zone near outlet bay - 40 ft).

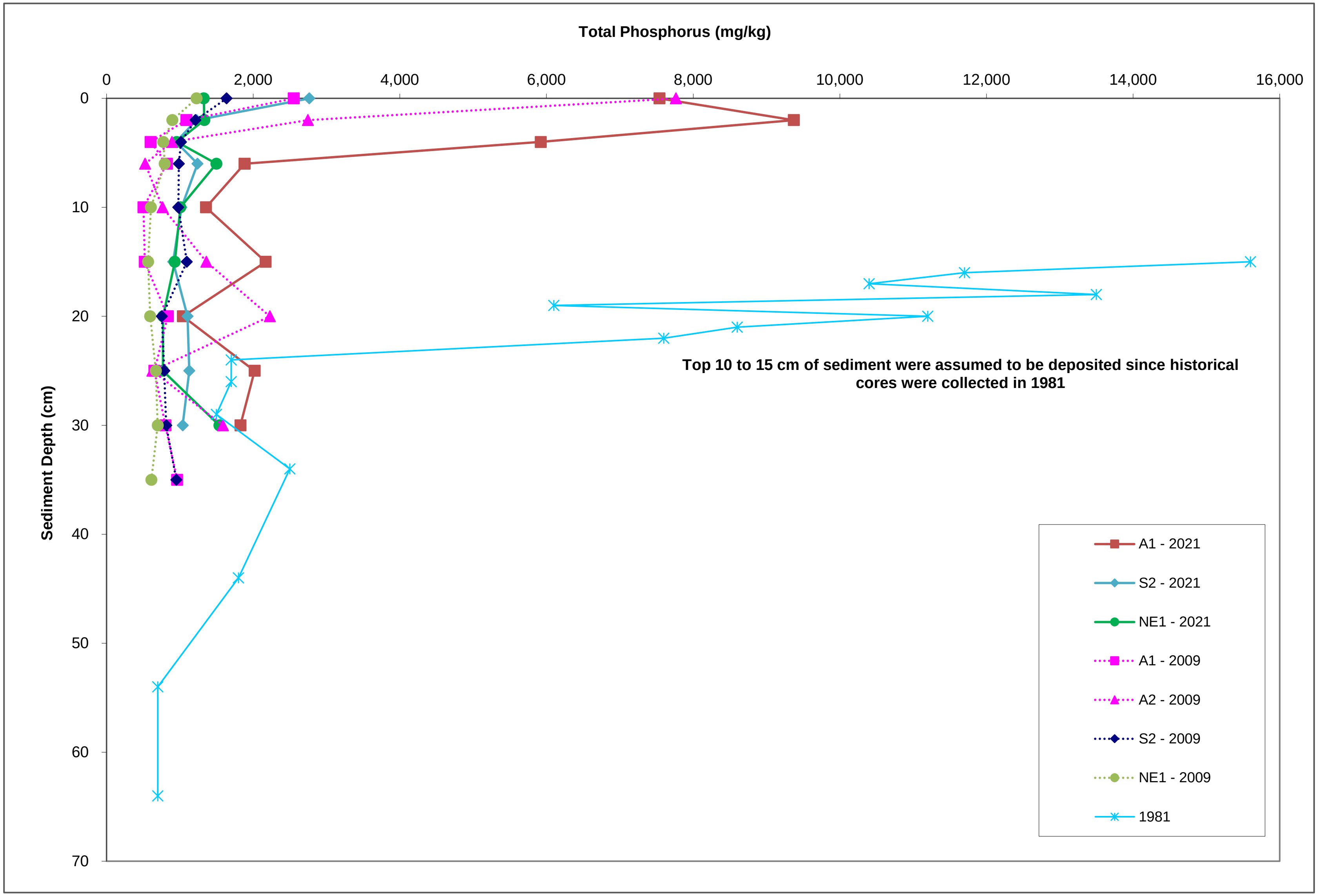


Figure 15. Sediment Al content in the top 35 cm before and after hypolimnetic aeration ceased in 2012 and after annual alum treatments started in 2013. Site locations are A1 (deep - 160 ft), S2 (south end - 80 ft), and NE1 (epilimnion zone near outlet bay - 40 ft).

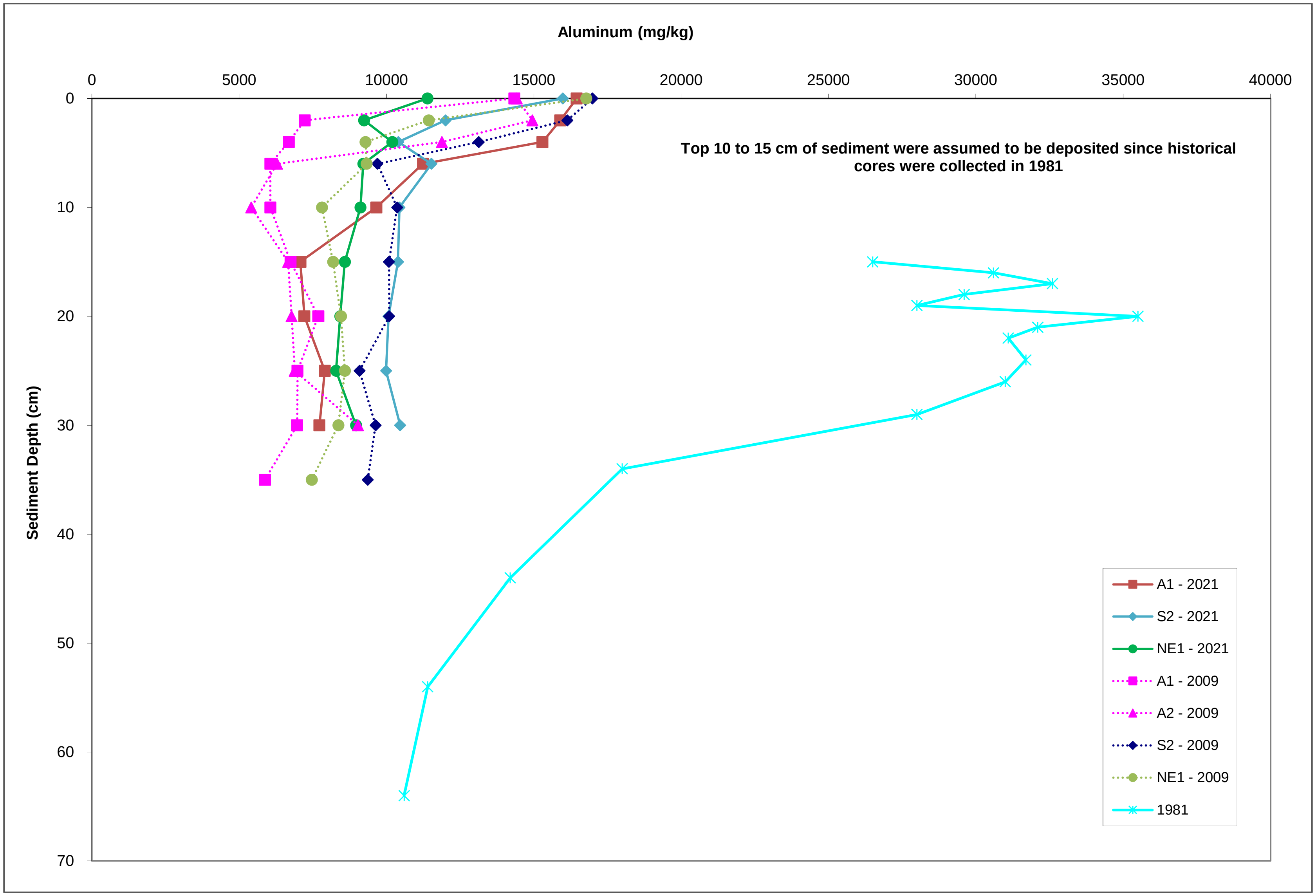


Figure 16: Sediment Al-P content in the top 35 cm before and after hypolimnetic aeration ceased in 2012 and after annual alum treatments started in 2013. Site locations are A1 (deep - 160 ft), S2 (south end - 80 ft), and NE1 (epilimnion zone near outlet bay - 40 ft).

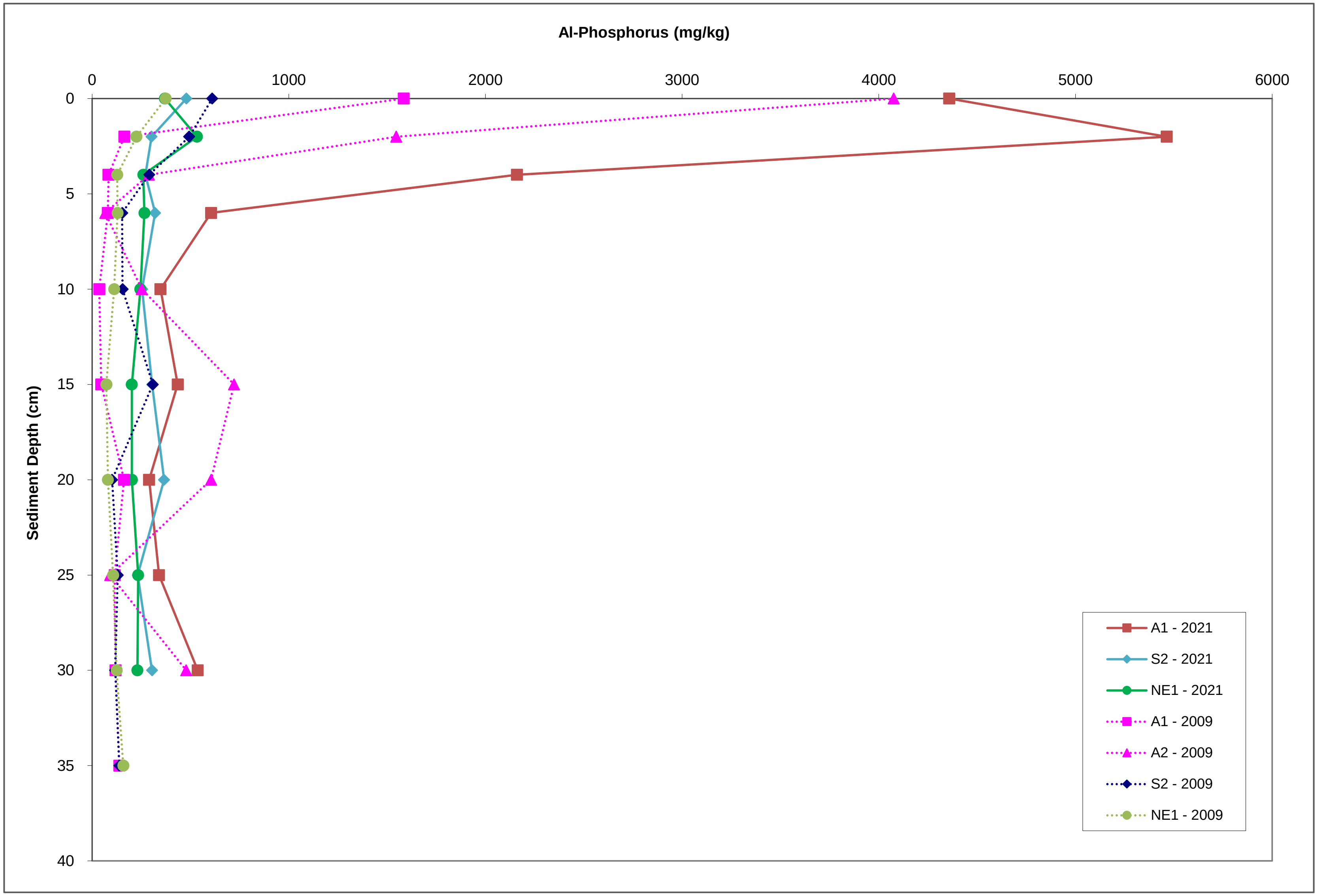


Figure 17. Sediment Fe-P content in the top 35 cm before and after hypolimnetic aeration ceased in 2012 and after annual alum treatments started in 2013. Site locations are A1 (deep - 160 ft), S2 (south end - 80 ft), and NE1 (epilimnion zone near outlet).

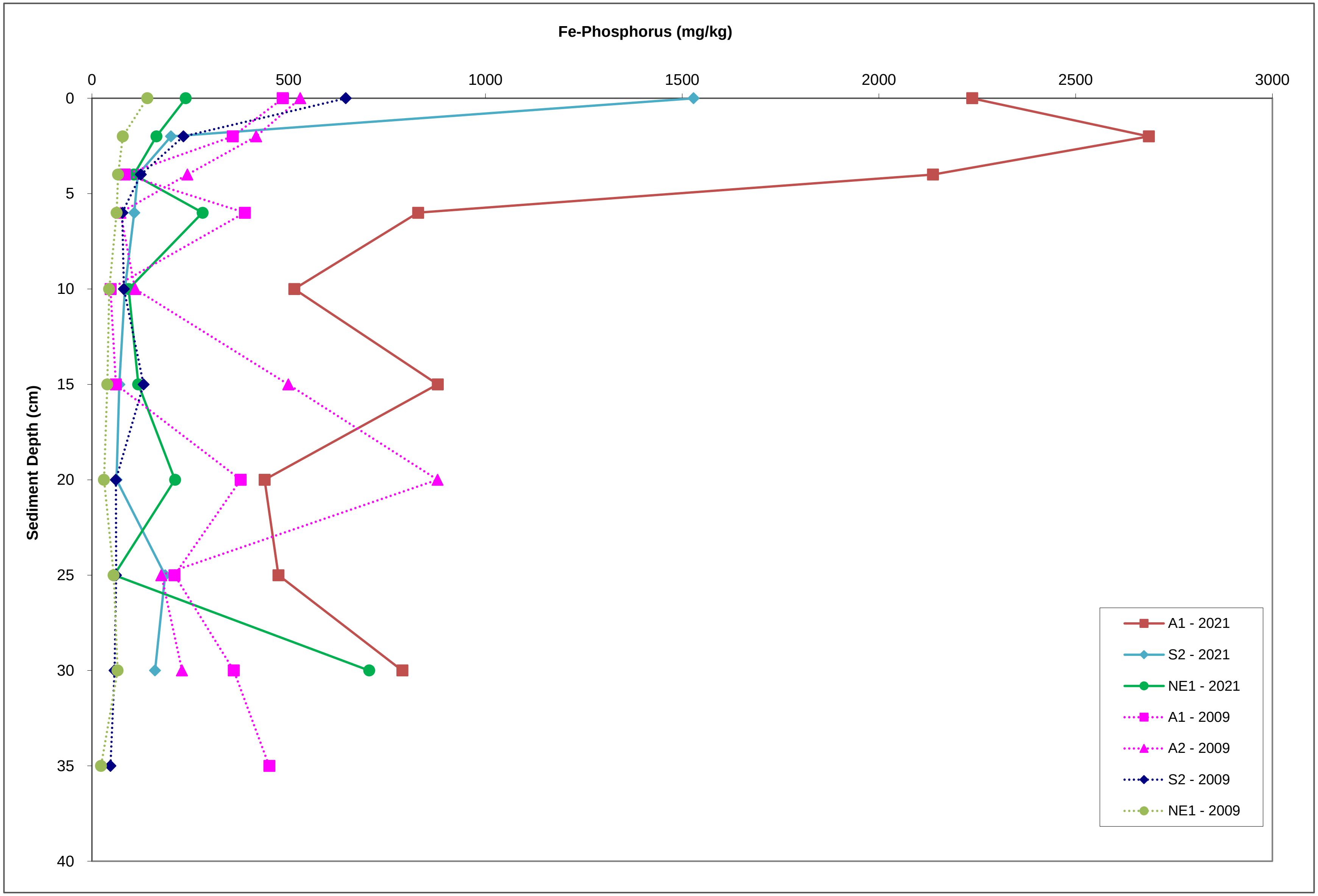
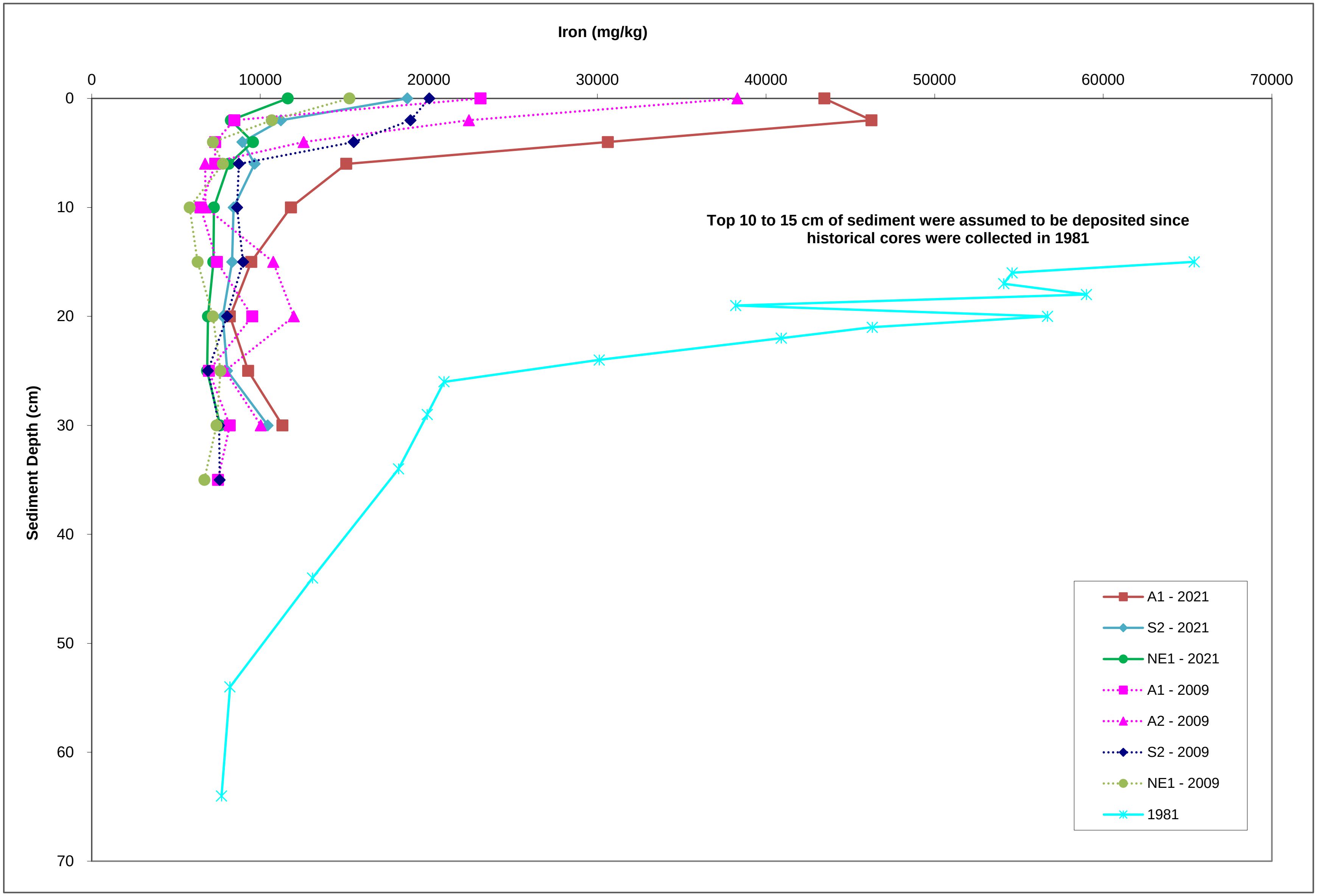


Figure 18. Sediment Fe content in the top 35 cm before and after hypolimnetic aeration ceased in 2012 and after annual alum treatments started in 2013. Site locations are A1 (deep - 160 ft), S2 (south end - 80 ft), and NE1 (epilimnion zone near outlet bay).



3.5. SEDIMENT RELEASE RATE (SRR) RESULTS AND DISCUSSION

The SRR over the summer declined in 2013 with the start of annual alum applications and has remained low to near zero (Table 5; Figure 19). The post-alum treatments SRR (average 0.19 mg/m2-day) declined by more than three-fourths, although year-to-year variability was ± 80%. The SRRs in 2020 and 2021 were very low, zero and 0.05 mg/m2-day, respectively (

Table 1). The large variability in post-alum treatment SRRs could be due to difficulty in detecting the low post-alum treatment SRRs. Prior to the start of alum treatments, SRR (average 1.08 mg/m2-day) varied less (± 25%) but was much higher than post-alum. The SRR decreased from before the start of alum treatments and after the start of alum treatments by the same proportion at 40 m (4.4 to 0.78 mg/m2-day) as it did for the whole-hypolimnetic area (Table 5). The decline in SRR resulted in a subsequent decline in mean summer (May - October) hypolimnetic TP by two-thirds from 30 to 9.6 µg/L. Late summer maximums at 40 m (near the bottom) also declined, from 60 - 70 µg/L to 10 - 20 µg/L (Figure 6). The annual alum treatments have resulted in the consistent reduction in SRR and decline in hypolimnetic TP.

The year-to-year variations in SRR may also have been due to the length and timing of thermal stratification and bottom DO. Profiles of DO were variable year-to-year. Bottom water was less anoxic (< 2 mg/L DO) in 2009 - 2011 (1.43 mg/L to 3.1 mg/L) with aeration but was consistently near zero in 2012. Anoxia occurred for a longer period in 2013 than in subsequent years but SRR was low (0.22 mg/m2-day). The addition of alum starting in 2013 was likely the reason for low SRRs even with a longer period of anoxia. Minimum DOs in 2013 - 2021 were usually near zero however SRRs were very low, indicating minimal to zero release of phosphorus from the sediments. The low SRRs observed since alum treatments started in 2013 are due to the binding of sediment phosphorus with aluminum, which remains bound even under anoxic conditions.

The hypolimnetic TP concentrations and SRRs are comparable to those in Lake Sammamish, a mesotrophic lake in Western Washington; average whole-hypolimnetic TP of 19 µg/L and SRR of 2.4 mg/m2-day (Welch et al., 2019). Prior to wastewater diversion, the whole-hypolimnetic TP and SRR in Lake Sammamish were higher at 58 µg/L and 10 mg/m2-day. Lake Sammamish still has greater hypolimnetic DO depletion, TP, and SRR than Lake Stevens, partly because Lake Sammamish is shallower (31 m versus 47 m maximum depth) with a greater anoxic volume; the bottom 10 m goes anoxic by mid-summer (Figure 4). It has been over 40 years following wastewater diversion and any further decline in Lake Sammamish's SRR is unlikely. A similar residual SRR would probably be the case for Lake Stevens without the annual alum treatments.

Another comparison relates to availability of hypolimnetic phosphorus to the epilimnion through diffusion and entrainment. The Osgood index (mean depth [m]/√surface area [km2]) for Lake Stevens is double that for Lake Sammamish (9.4 versus 3.9). Entrainment of hypolimnetic phosphorus into the epilimnion was observed to increase at indices less than 6 to 7 (Osgood, 1988). Thus, high hypolimnetic phosphorus is not as apt to reach the epilimnion and be available for algal uptake during the summer in Lake Stevens. Nevertheless, epilimnion TP also declined after the start of annual alum treatments; from a May - October average of 11.3 to 7.3 µg/L. This decrease is partly due to binding of phosphorus from the epilimnion following spring alum treatments.

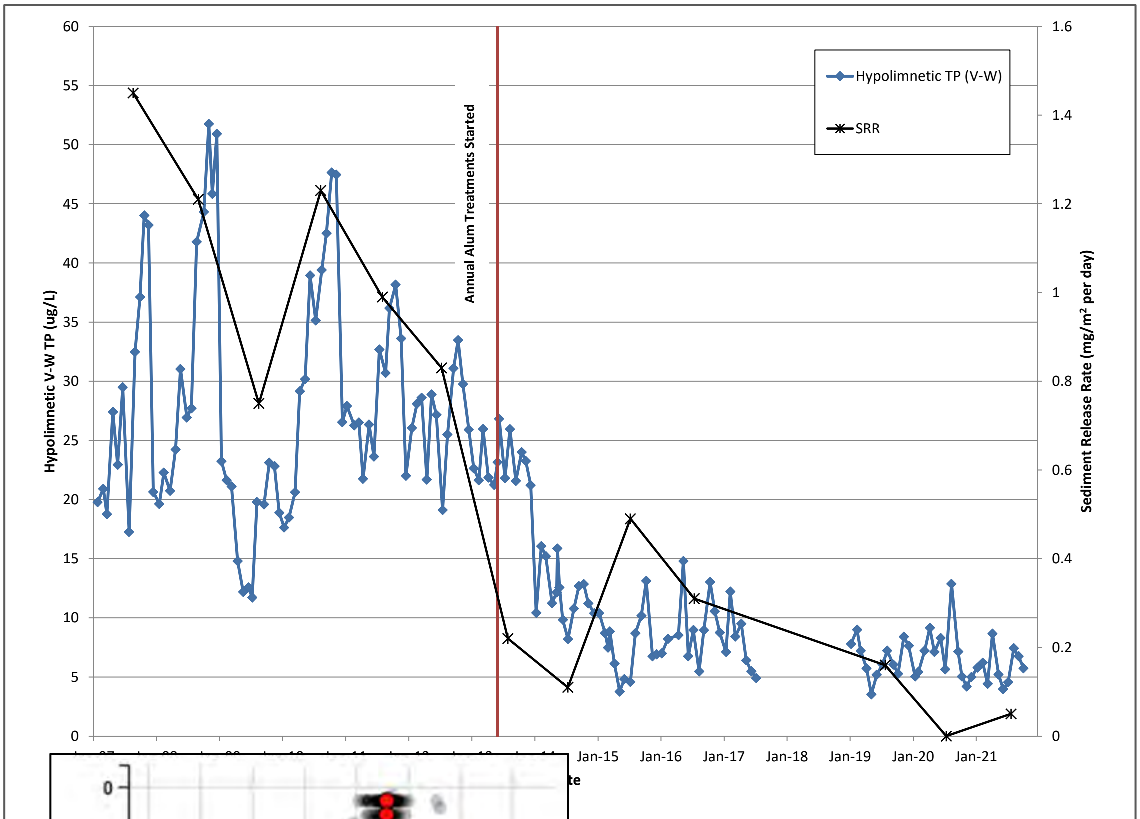
The annual alum dose starting in 2013 has been 0.15 mg Al/L each year (0.12 mg Al/L in 2020). The dose needed to inactivate mobile sediment phosphorus in the top 4 cm was calculated at 1.6 mg Al/L (Tetra Tech, 2009; Snohomish County and Tetra Tech, 2012, page 26). Eight years of annual water column stripping amounts to 1.17 mg Al/L – about three-fourths of the initial dose for sediment phosphorus inactivation in the top 4 cm of sediment. At a dose of 0.15 mg Al/L, it would take another 3 annual treatments to reach the full sediment inactivation dose calculated for the top 4 cm.

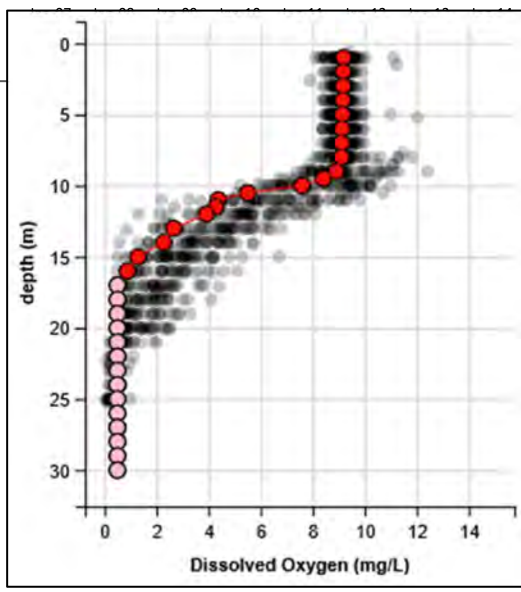
To inactivate mobile-P in the top 36 cm (which contains high concentrations of legacy phosphorus that could diffuse upwards over time) and strip the water column of TP, the dose was calculated at 10.4 mg Al/L (Tetra Tech, 2009, page 17). The data on hypolimnetic phosphorus decrease (maximums of 60 - 70 µg/L to 10 - 20 µg/L) and SRR decrease by over three-fourths indicate that sediment phosphorus inactivation has partially occurred in Lake Stevens. Also, the SRR has remained consistently low (0.19 mg/m^2-day) for the past nine years with some year-to-year variation. Given the low release rate – only 14% of the Lake Sammamish rate – further reduction in SRR in Lake Stevens seems unlikely, even if alum treatment continues. However, if alum applications are discontinued, there is a potential for SRR to increase if mobile phosphorus in deeper sediments migrates upward, or as future phosphorus inputs build up in the lake sediments.

Figure 19. Volume-weighted hypolimnetic SRR in Lake Stevens, 2007 - 2021.

Figure 20. DO concentrations in Lake Sammamish (DeGasperi, personal communication, King County 2021).

3.6. SEDIMENT ANALYSIS CONCLUSIONS





With respect to lake sediments, the purpose of the annual alum treatments was to neutralize the available phosphorus in the lake sediments to prevent internal loading as well as bind ongoing phosphorus inputs from the lake watershed to prevent future build-up of available phosphorus in the sediment. The low SRR and hypolimnetic TP are evidence that the alum treatments have effectively bound sediment phosphorus making it unavailable to diffuse into the overlying water. The reduction in internal loading is shown by the decrease in hypolimnetic TP to around 10 µg/L post-alum. This is very low level despite anoxic conditions that should produce maximum release of phosphorus from the sediments. Additionally, the SRR decreased to a low, apparent equilibrium level post-alum.

Unfortunately, the sediment core analysis found that mobile phosphorus available for release as internal loading increased rather than decreased as expected. As discussed previously, these results are not considered reliable and are likely the results of a lab analysis issue which has been recently detected at other similar lakes with ongoing alum treatments. Despite these results, the SRR is a reliable observed measurement of internal loading since it integrates the effects of anoxia, diffusion through sediment porewater and the movement of overlying water and circulation across the sediment-water interface on phosphorus transport from the sediments. Thus, SRR should be used to evaluate the effectiveness of alum at inactivating sediment phosphorus and reducing internal loading. If the majority of sediment phosphorus in the sediments has been neutralized, the SRR can reasonably be expected to remain at a low level without further treatment, even with continued anoxia, as was observed in Lake Sammamish. However, the alum floc layer will probably settle through the sediment and newly added sediment could increase SRR in the future as external phosphorus loading continues.

The most uncertainty from the sediment core results is the effectiveness of the alum treatments to neutralize phosphorus in the deeper sediments. Based on the 2009 sediment cores, there is a potentially large source of phosphorus in deeper sediments. Over time, the phosphorus could migrate or diffuse upwards to surface sediments. Mobile phosphorus was found to diffuse from deeper sediment layers to bind with the added aluminum floc in both Lake Süsser See, Germany (Lewandowski et al., 2003) and in Green Lake, Seattle, Washington (Welch et al., 2017). Reassessment of the phosphorus concentrations in deeper cores of Lake Stevens is recommended once the lab analysis issues have been resolved. This would show if the high phosphorus at depth has depleted over time or whether it is locked in place and not migrating.

Table 3. Lake Stevens sediment phosphorus data, 2021.

SAMPLE ID		% SOLIDS	% WATER	VS (%)	TOTAL-P (mg/kg)	LOOSELY BOUND P (mg/kg)	FE BOUND P (mg/kg)	AL BOUND P (mg/kg)	BIOGENIC P (mg/kg)	CA BOUND P (mg/kg)	ORGANIC P (mg/kg)	Mobile-P (mg/kg)
STEVENS-A1 0-2 CM		7.31%	92.7%	28.3%	7541	2.0	2237	4358	155	619	327	2238.9
STEVENS-A1 2-4 CM		8.30%	91.7%	28.1%	9373	2.0	2686	5463	257	757	467	2687.7
STEVENS-A1 4-6 CM		9.97%	90.0%	27.5%	5923	2.0	2137	2160	1031	425	1201	2139.2
STEVENS-A1 6-10 CM		10.5%	89.5%	36.0%	1885	2.0	829	605	195	131	319	830.9
STEVENS-A1 10-15 CM		9.85%	90.1%	30.3%	1357	2.0	514	348	225	149	346	516.3
STEVENS-A1 15-20 CM		10.4%	89.6%	28.6%	2171	2.0	879	436	676	59.2	798	880.7
STEVENS-A1 20-25 CM		10.7%	89.3%	27.5%	1043	2.0	439	290	153	77.8	236	441.0
STEVENS-A1 25-30 CM		11.8%	88.2%	27.0%	2020	2.0	474	340	713	367	839	476.0
STEVENS-A1 30-35 CM		12.4%	87.6%	28.1%	1827	2.0	789	537	243	148	353	790.8
STEVENS-NE1 0-2 CM		8.00%	92.0%	37.7%	1328	2.0	238	370	408	136	583	240.4
STEVENS-NE1 2-4 CM		7.07%	92.9%	39.5%	1334	2.0	164	534	341	96.5	540	165.8
STEVENS-NE1 4-6 CM		7.88%	92.1%	38.5%	960	2.0	107	261	318	113	480	108.5
STEVENS-NE1 6-10 CM		8.25%	91.8%	41.1%	1499	2.0	281	266	634	128	824	283.2
STEVENS-NE1 10-15 CM		8.91%	91.1%	41.3%	1008	2.0	93.3	246	355	97.9	571	95.3
STEVENS-NE1 15-20 CM		9.72%	90.3%	42.2%	932	2.0	118	202	316	94.1	518	119.6

SAMPLE ID		% SOLIDS	% WATER	VS (%)	TOTAL-P (mg/kg)	LOOSELY BOUND P (mg/kg)	FE BOUND P (mg/kg)	AL BOUND P (mg/kg)	BIOGENIC P (mg/kg)	CA BOUND P (mg/kg)	ORGANIC P (mg/kg)	Mobile-P (mg/kg)
STEVENS-NE1 20-25 CM		10.0%	90.0%	40.7%	772	2.0	211	203	127	86.4	272	213.1
STEVENS-NE1 25-30 CM		11.0%	89.0%	37.0%	771	2.0	56.2	234	200	122	358	58.2
STEVENS-NE1 30-35 CM		10.5%	89.5%	39.1%	1540	2.0	705	231	337	119	485	706.8
STEVENS-S2 0-2 CM		7.82%	92.2%	32.5%	2766	2.0	1529	480	389	141	616	1531.0
STEVENS-S2 2-4 CM		6.91%	93.1%	37.4%	1227	2.0	201	302	405	102	622	202.9
STEVENS-S2 4-6 CM		8.08%	91.9%	36.1%	947	2.0	117	272	269	101	457	118.8
STEVENS-S2 6-10 CM		9.06%	90.9%	38.8%	1241	2.0	108	321	517	132	681	109.6
STEVENS-S2 10-15 CM		9.72%	90.3%	38.2%	1022	2.0	83.7	255	403	105	579	85.7
STEVENS-S2 15-20 CM		10.7%	89.3%	35.6%	909	2.0	69.7	306	265	107	427	71.7
STEVENS-S2 20-25 CM		11.6%	88.4%	33.9%	1105	2.0	62.6	366	384	128	548	64.6
STEVENS-S2 25-30 CM		11.9%	88.1%	30.0%	1130	2.0	186	232	401	153	559	188.0
STEVENS-S2 30-35 CM		11.6%	88.4%	35.0%	1040	2.0	160	305	280	134	441	162.3

Table 4. Lake Stevens sediment metals data, 2021.

SAMPLE ID	% SOLIDS	% WATER	VS (%)	ALUMINUM (mg/kg)	IRON (mg/kg)	CALCIUM (mg/kg)	Fe:P RATIO	Al:P RATIO
STEVENS-A1 0-2 CM	7.31%	92.7%	28.3%	16454	43453	4564	5.8	2.2
STEVENS-A1 2-4 CM	8.30%	91.7%	28.1%	15897	46251	4171	4.9	1.7
STEVENS-A1 4-6 CM	9.97%	90.0%	27.5%	15292	30613	3620	5.2	2.6
STEVENS-A1 6-10 CM	10.5%	89.5%	36.0%	11239	15100	2957	8.0	6.0
STEVENS-A1 10-15 CM	9.85%	90.1%	30.3%	9659	11817	3223	8.7	7.1
STEVENS-A1 15-20 CM	10.4%	89.6%	28.6%	7078	9463	2241	4.4	3.3
STEVENS-A1 20-25 CM	10.7%	89.3%	27.5%	7217	8201	2285	7.9	6.9
STEVENS-A1 25-30 CM	11.8%	88.2%	27.0%	7907	9280	2691	4.6	3.9
STEVENS-A1 30-35 CM	12.4%	87.6%	28.1%	7726	11314	2630	6.2	4.2
STEVENS-NE1 0-2 CM	8.00%	92.0%	37.7%	11391	11628	4676	8.8	8.6
STEVENS-NE1 2-4 CM	7.07%	92.9%	39.5%	9242	8248	4628	6.2	6.9
STEVENS-NE1 4-6 CM	7.88%	92.1%	38.5%	10196	9560	5133	10.0	10.6
STEVENS-NE1 6-10 CM	8.25%	91.8%	41.1%	9211	8134	4963	5.4	6.1
STEVENS-NE1 10-15 CM	8.91%	91.1%	41.3%	9119	7251	4950	7.2	9.0
STEVENS-NE1 15-20 CM	9.72%	90.3%	42.2%	8589	7218	6007	7.7	9.2
STEVENS-NE1 20-25 CM	10.0%	90.0%	40.7%	8429	6898	5845	8.9	10.9
STEVENS-NE1 25-30 CM	11.0%	89.0%	37.0%	8293	6840	5263	8.9	10.8
STEVENS-NE1 30-35 CM	10.5%	89.5%	39.1%	8967	7613	6213	4.9	5.8
STEVENS-S2 0-2 CM	7.82%	92.2%	32.5%	15983	18717	5295	6.8	5.8
STEVENS-S2 2-4 CM	6.91%	93.1%	37.4%	12003	11223	4730	9.1	9.8
STEVENS-S2 4-6 CM	8.08%	91.9%	36.1%	10402	8941	4384	9.4	11.0
STEVENS-S2 6-10 CM	9.06%	90.9%	38.8%	11525	9661	4743	7.8	9.3
STEVENS-S2 10-15 CM	9.72%	90.3%	38.2%	10440	8413	4359	8.2	10.2
STEVENS-S2 15-20 CM	10.7%	89.3%	35.6%	10391	8326	4232	9.2	11.4
STEVENS-S2 20-25 CM	11.6%	88.4%	33.9%	10067	7773	4131	7.0	9.1
STEVENS-S2 25-30 CM	11.9%	88.1%	30.0%	9989	8035	4674	7.1	8.8
STEVENS-S2 30-35 CM	11.6%	88.4%	35.0%	10465	10440	4988	10.0	10.1

Table 5. Lake Stevens SRR, 2007 - 2021.

Date/Year	SRR @ whole-hypo area (mg/m^2-day)	SRR @ 40 m area (mg/m^2-day)
5/21/2007 - 11/16/2007	1.45	5.9
5/19/2008 - 12/16/2008	1.21	4.91
5/18/2009 - 11/16/2009	0.75	3.05
5/13/2010 - 11/10/2010	1.23	4.98
5/19/2011 - 10/20/2011	0.99	4
4/8/2012 - 10/17/2012	0.83	3.38
5/14/2013 - 10/21/2013	0.22	0.9
4/15/2014 - 10/16/2014	0.11	0.46
4/13/2015 - 10/14/2015	0.49	1.99
4/18/2016 - 10/19/2016	0.31	1.27
4/15/2019 - 11/18/2019	0.16	0.64
5/13/2020 - 9/28/2020	0.00	0.01
5/20/2021 - 10/12/2021	0.05	0.19
AVERAGE PRE-ALUM (2007 - 2012)	**1.08**	**4.37**
AVERAGE POST-ALUM (2013 - 2021)	**0.19**	**0.78**

4. BIOLOGICAL ANALYSIS

4.1. BIOLOGICAL ANALYSIS OBJECTIVES

Historically, Lake Stevens suffered from blooms of algae that created unsightly scums and harmed lake recreation. Blooms of cyanobacteria (also known as blue-green algae) were of particular concern as they have the potential to produce toxins that can affect the health of people and pets recreating in the lake. Cyanobacteria blooms also impact water quality by reducing clarity and decreasing DO which can affect fish and other aquatic life. Cyanobacteria can affect a lake's food web as they are largely inedible compared to other more desirable algal species. The goal of ongoing restoration efforts is to reduce phosphorus in the lake which, in turn, reduces algae and harmful algal blooms and improves overall lake health.

The purpose of the biological analysis is to determine if the restoration goals have been met by examining the changes in the phytoplankton and zooplankton communities. Phytoplankton are the microscopic photosynthetic organisms that form the base of the food chain and include cyanobacteria and algae. Zooplankton are the tiny animals that feed on phytoplankton and are consumed by fish and other aquatic organisms.

The main objectives of the analysis were to:

- assess the effectiveness of the alum treatments in reducing the population of potentially toxic cyanobacteria
- evaluate the short and long-term impacts of the alum treatments on the zooplankton community including how it relates to the kokanee fishery

The results of this analysis will help the City evaluate the alum treatment's success and continue adaptive management of the lake into the future. It will also provide the basis to ensure timing of future alum treatments has minimal impact to the kokanee fishery which has been a concern of recreational anglers.

4.2. BIOLOGICAL ANALYSIS METHODS

4.2.1. Data Sources

There are limited biological data collected at Lake Stevens. Routine collection of biological data has not occurred due to budget constraints. However, biological data has been collected during intermittent studies conducted in the last three decades. Biological data from these studies represent three different periods in the lake restoration history as follows:

- **Pre-Restoration Period (1986 - 1987):** Phytoplankton and zooplankton were collected monthly from February 1986 - March 1987 as part of a Phase IIA Restoration analysis (KCM, 1987). Phytoplankton samples were collected at an open water, deep location as a composite sample of 0 to 10 meters and were analyzed for identification, abundance and biovolume. Zooplankton samples were collected from the same sample location using a vertical 40 m tow. Samples were analyzed for identification, abundance, and biomass.

- **Late Aeration/Pre-alum Period (2009 - 2011):** As part of the Regional Examination of Harmful Algal Blooms (REHAB) study, weekly phytoplankton samples were collected bimonthly from May - October each year during 2009 - 2011. The samples were collected from the shoreline area of Lake Stevens Davies Beach (formerly Wyatt Park) as the REHAB study purpose was to identify the presence of, and health risk, from potentially toxic cyanobacteria species at regional lakes. Water Environmental Services conducted the analysis including identification and abundance. The biovolumes for the dataset were not directly measured. Instead, values were estimated based on a twenty-year historical phytoplankton dataset from the lake monitoring program in King County, Washington. This period represents the lake conditions after 15 years of aerator operation, which began in 1994. It corresponds to the 2009 study which concluded the aerator was no longer effectively controlling internal phosphorus loading from the lake sediments (Tetra Tech, 2009).

Post-alum (2016 - 2018): From 2016 - 2018, the County collected monthly phytoplankton samples from the same shoreline location as the REHAB study. Samples were preserved with an iodine Lugol's solution. Zooplankton samples were collected from the same deep-water station as the monthly water quality sampling. A vertical tow sample was taken from a depth of 35 m using a 74 µm mesh zooplankton net with a 30 cm diameter hoop. Samples were preserved with a glycerin alcohol solution. Both phytoplankton and zooplankton samples were analyzed by Advanced Eco-Solutions for identification, abundance, biovolume (phytoplankton), and biomass (zooplankton). This data is from the period three years after alum began in 2013 and includes two before-and-after alum treatment comparisons.

Fish stocking data of kokanee and rainbow trout from 2010 - 2021 was also obtained from the Washington Department of Fish and Wildlife. Unfortunately, stocking data prior to 2010 could not be located for this study.

4.2.2. Data Limitations

The lack of long-term consistent biological data will limit the scope of conclusions that are drawn from this analysis. Biological parameters can be highly variable seasonally and between years because of differing climate conditions as well as other factors such as fish stocking. Additionally, it may take multiple seasons for the lake biological community to stabilize following changes in lake conditions that may occur from lake restoration activities.

Another limitation of the analysis stems from different phytoplankton sample collection methods used at different points in time. The samples collected for phytoplankton analysis in 1986 - 1987 were collected from the open water as a composite of the top 10 m of the water column, while later samples were collected at the shoreline. Cyanobacteria have pseudo-vacuoles (gas-vacuoles) that expand/contract allowing cells to be buoyant to maximize photosynthetic production and thus can collect on the water surface and be blown onshore by wind. This can cause differences in cell abundance and biovolumes due to wind speed, direction, solar intensity and variability. Cyanobacteria can also recruit phosphorus from bottoms sediment and carry it with them. Recruitment of cyanobacteria cells from bottom sediments was determined in Lake Sammamish following a toxic bloom that had microcystin levels higher in shoreline samples than in the open water (Kenworthy, 2000; Johnston and Jacoby, 2003). Therefore, shoreline samples may be disproportionately higher in cyanobacteria as compared to other algae such as diatoms (bacillariophytes) or green algae (chlorophytes). Overall, these factors can cause the shoreline samples to be especially unreliable in comparison with the earlier 10 m composite, and potentially even unreliable for year-to-year comparison.

The historic and recent phytoplankton data can be compared, however, by looking at phyla dominance, both by abundance and biovolume. Cyanobacteria cells are usually much smaller than most diatoms and green algae, so biovolume is best to use for determining dominance relative to biomass. However, when cyanobacteria are a significant percentage of the total cell density, there is a water quality and food-chain stress that leads to accelerated eutrophication of a waterbody and the community ecosystem falls out of balance. Thus, evaluating cyanobacteria abundance can also provide valuable information when comparing datasets.

4.3. BIOLOGICAL ANALYSIS RESULTS AND DISCUSSION

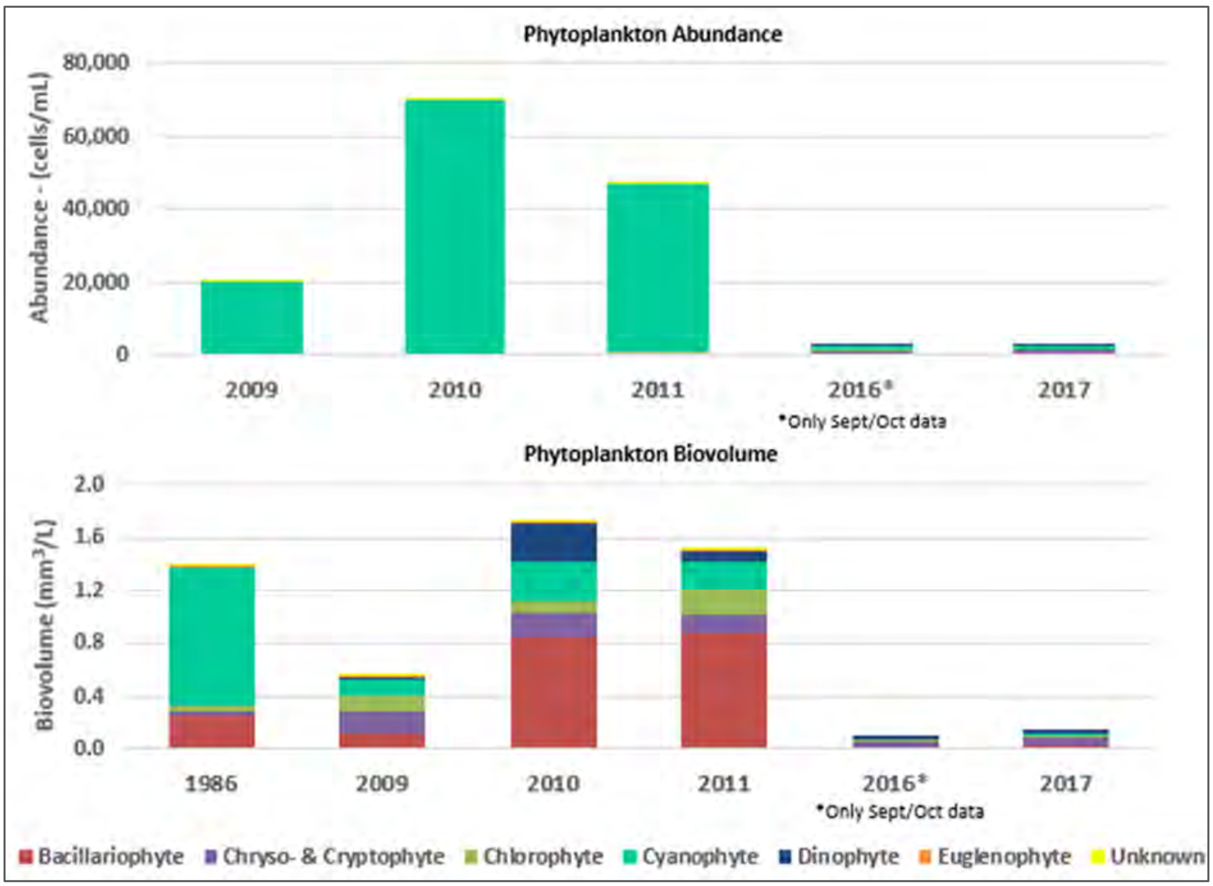
4.3.1. Phytoplankton Community

To see the differences in the phytoplankton community, individual taxa were grouped to the taxonomic level of phylum. The main phyla found in Lake Stevens are as follows:

- Bacillariophyta – diatoms
- Chryso & Cryptophyta – golden algae (no longer includes diatoms)
- Chlorophyta – green algae
- Cyanophyta – cyanobacteria
- Dinophyta – dinoflagellates
- Euglenophyta – euglenas

Despite the potential limitations in comparing shoreline samples between years, the overall abundance of phytoplankton is significantly lower when comparing summer samples from the pre-alum period (2009 - 2011) and the post-alum period (2016 - 2017) (Table 9). Pre-alum samples had summer average abundances between 20,000 and 70,000 cells/mL in 2009 - 2012 (Figure 21; Table 9). The 2017 post-alum summer average was 2,764 cells/mL, an order of magnitude lower than pre-alum. The 2016 average was also low at 2,549 cells/mL, but only includes samples from September and October. A similar decrease was observed in total phytoplankton biovolumes, a surrogate for the biomass of algae. Total summer biovolumes were 0.57, 1.74 and 1.52 mm3/L in 2009 - 2011 compared to 0.15 mm3/L in 2017 (Figure 21;Table 13)

Figure 21: Changes in Lake Stevens phytoplankton phyla pre-alum (2009 - 2012) and post-alum (2016 - 2017) by abundance (top graph) and biovolume (bottom graph).

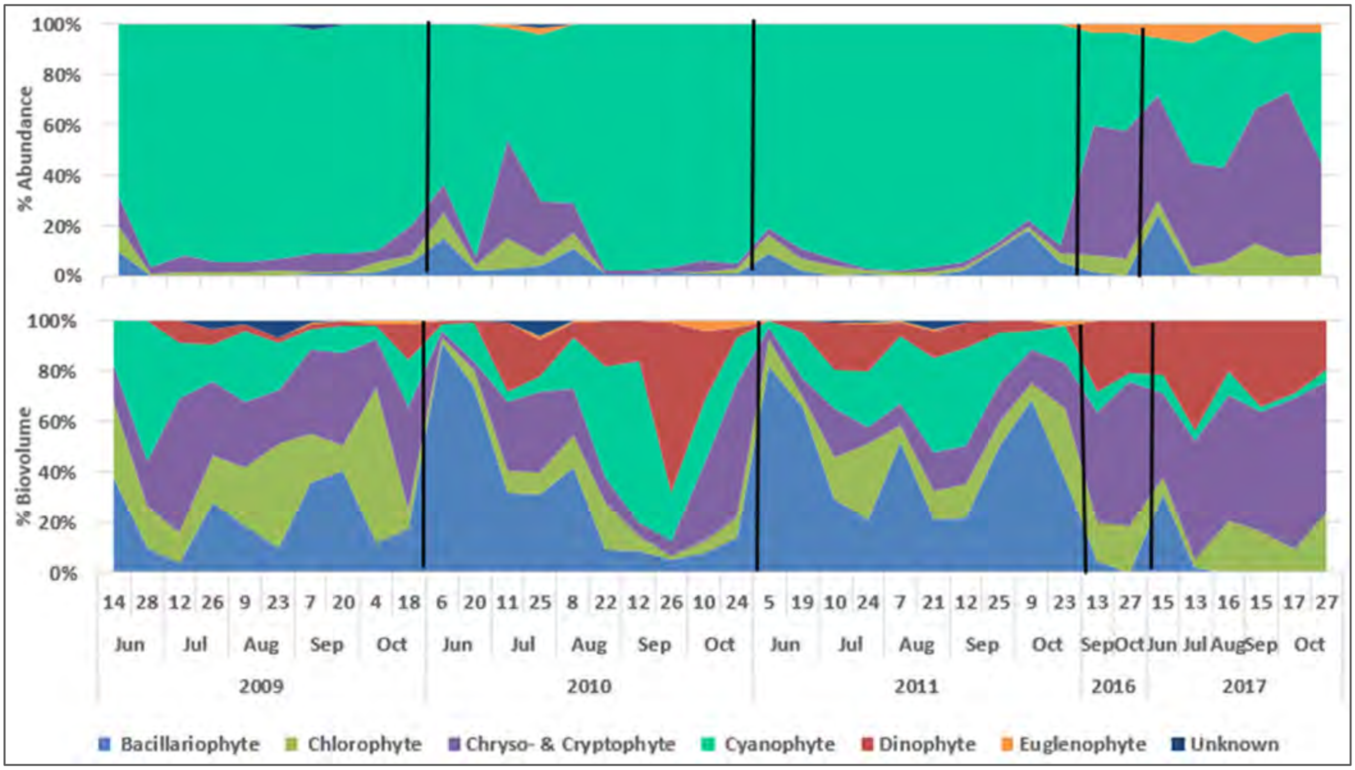


Not only has total phytoplankton decreased but cyanobacteria also comprised a lower proportion of the community both in abundance and biovolume post-alum (Figure 22; Table 7 and

Table 10). In the summers of 2010 - 2012, cyanobacteria cells accounted for 91 to 92% of the total summer average compared to 37% in 2017. There were corresponding increases in 2017 of chrysophytes that accounted for 41% of the population, followed by bacillariophytes (11%) and chlorophytes (6%) (Figure 22; Table 7). When looking at biovolumes, cyanobacteria was not as dominant in the lake community pre-alum as it has a much smaller cell size compared to taxa in other phyla. However, the summer cyanobacteria biovolumes still decreased from 14 - 19% of the community to 4% in 2017 (Figure 22;

Table 10).

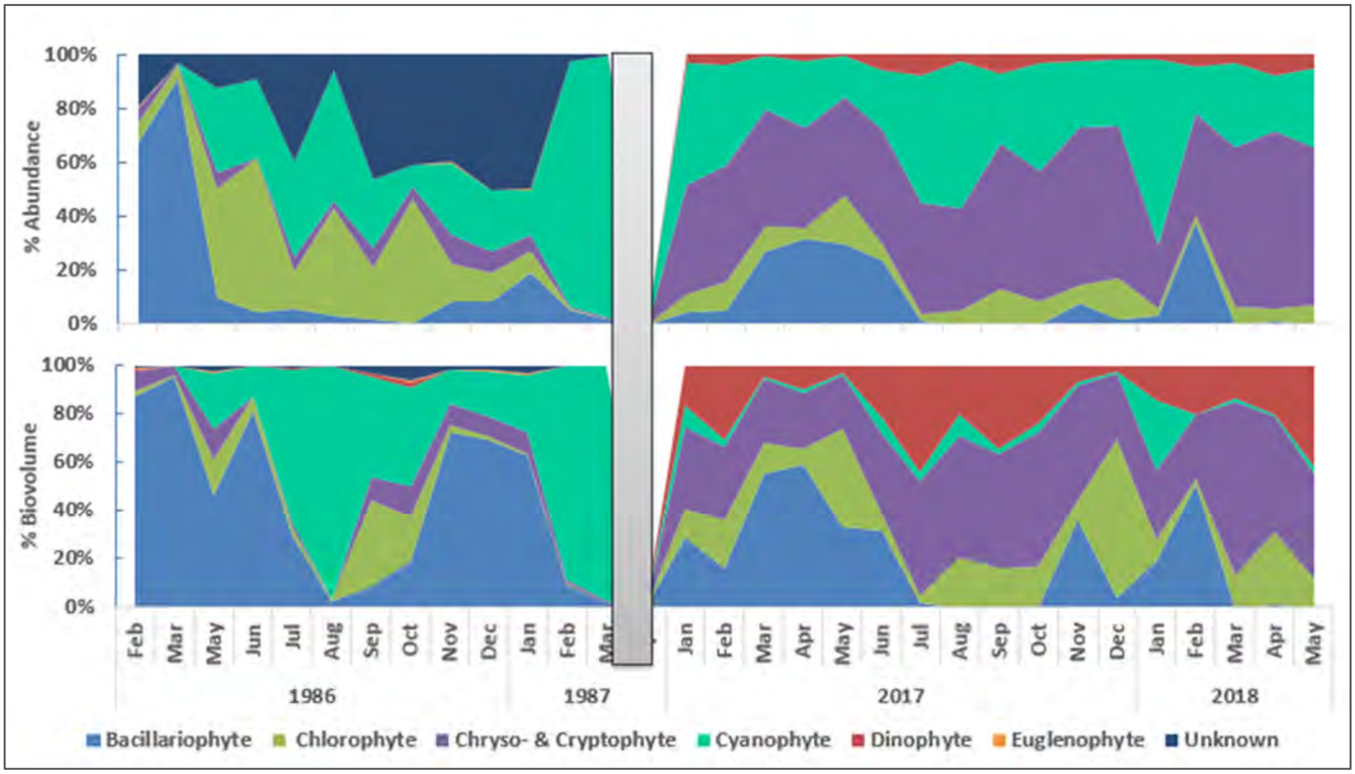
Figure 22: Summer progression of Lake Stevens phytoplankton communities by phyla pre-alum (2009 - 2011) and post-alum (2016 - 2017). Top graph is % abundance; bottom graph is % biovolume.



Changes in the cyanobacteria population across the entire year can be evaluated by comparing the 1986 - 1987 data to the 2017 - 2018 data. While overall phytoplankton abundance was lower in the historical data (Figure 23), cyanobacteria cells were present through much of the year and dominated the phytoplankton community in February and March in 1987 (Figure 23; Table 8). When looking at cyanobacteria cell abundance, the proportion of cyanobacteria does not appear to change significantly from the 1980's to the present except when comparing the 1987 winter data. The 1986 abundance data are especially difficult to interpret as one algal species could not be identified yet comprises a large portion of the algal population (Figure 23).

When compared to the 2016 - 2018 data, there is consistently lower abundance of cyanobacteria cells through much of the year. Despite the lack of a clear difference in the proportion of algae, the net decrease in overall phytoplankton does indicate that there should be fewer cyanobacteria cells currently as compared to the 1980's.

Figure 23: Annual progression of Lake Stevens phytoplankton communities by phyla pre-restoration (1986 - 1987) and post-alum (2016 - 2017). Top graph is % abundance; bottom graph is % biovolume.

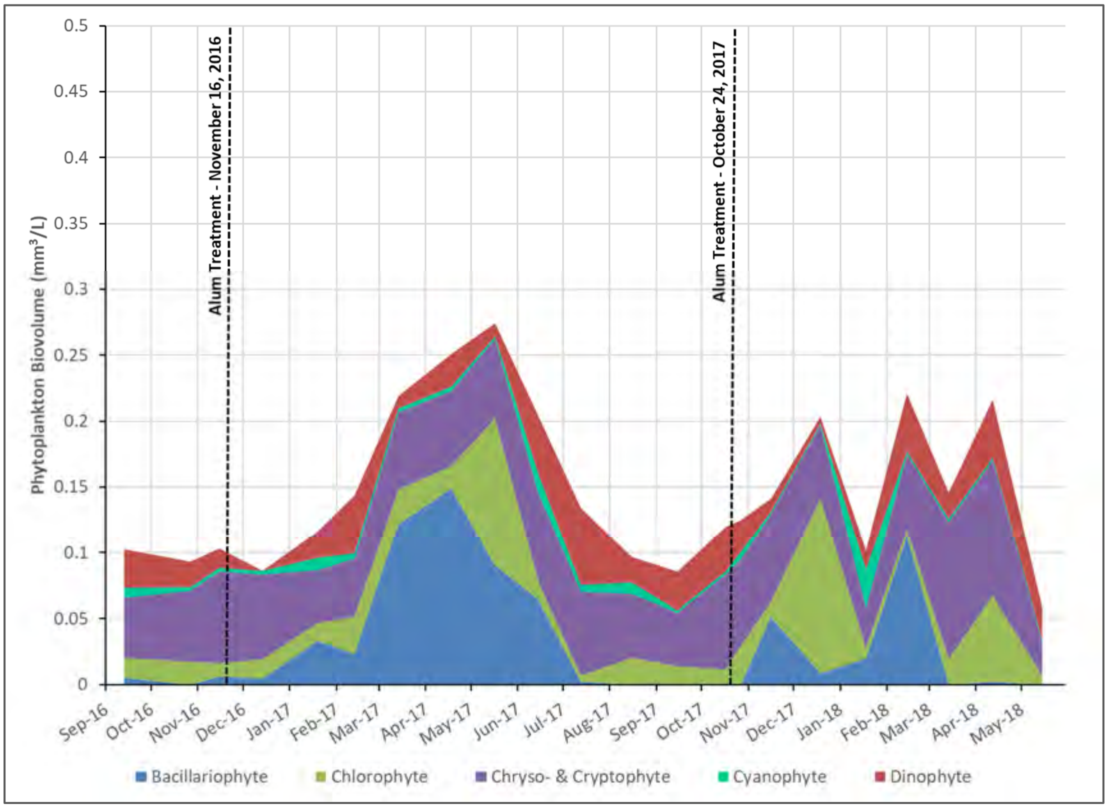


Finally, while the goal of alum treatments is to reduce phosphorus and decrease the amount cyanobacteria in the lake, there is concern that too much phytoplankton loss could disrupt the food base of the lake. Phytoplankton can be bound up in the alum flocculant during treatment and settle to the bottom causing short-term declines in phytoplankton. Based on the 2016 - 2018 data, it does not appear that the off-target impacts of alum on phytoplankton are significant. In 2016, there was a very small decrease in biovolumes one-month post-treatment and in 2017 there was an increase (Figure 24). Cyanobacteria were also less abundant in spring 2017 after the 2016 fall alum treatment and roughly equal to diatoms (bacillariophyte) (Figure 24).

The reduction in phytoplankton abundance and biovolume is consistent with the observed decreases in epilimnetic TP and chl observed following the alum treatments. TP decreased from an average of 12.8 to 7 µg/L and chl from 4 to 2.6 µg/L (Table 2). In addition, the reduction of cyanobacteria dominance is also likely a result of the lower TP values as the Lake Stevens finding are consistent with a large dataset relationship between TP and cyanobacteria percent composition (Downing et al., 2001). Their data showed that the risk for 50% cyanobacteria was very low (0 to 10%) at TP less than 10 µg/L. Low TP does not preclude cyanobacteria, but their presence is minimized with low TP concentrations.

Overall, the alum treatments do appear to be meeting the goal of reducing the abundance of cyanobacteria and potentially toxic algae blooms. Furthermore, cyanobacteria are a lower proportion of the algal community and have been replaced by more desirable species which provide a superior food source for zooplankton and fish.

Figure 24: 2016 - 2017 Lake Stevens monthly phytoplankton biovolumes by phylum shown with annual alum treatment dates.



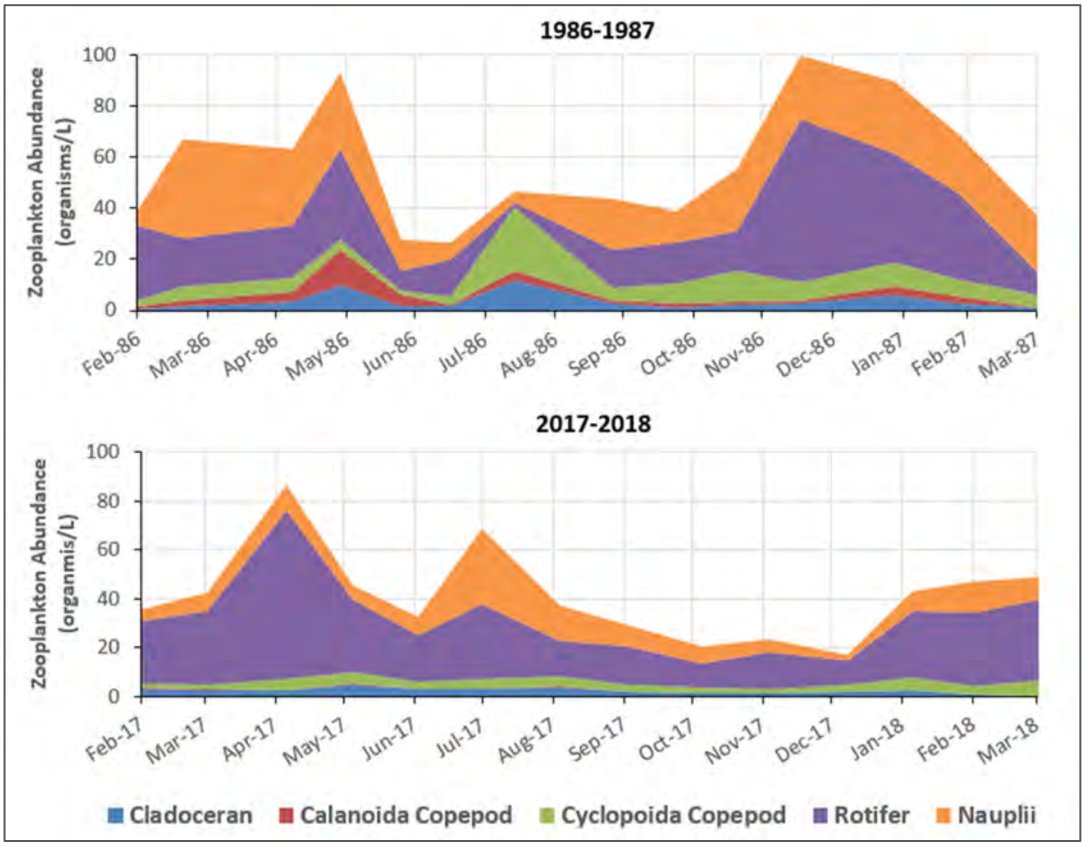
4.3.2. Zooplankton Community

There is limited data available to assess the effect of alum on the zooplankton community in Lake Stevens. There is one historic dataset before alum, 1986 - 1987 and one dataset post-alum, 2016 - 2018. Fortunately, the zooplankton samples collected in 1986 - 1987 and in 2016 - 2018 were collected using similar methods and are therefore directly comparable. Annual averages of both density and biomass were calculated for February 1986 - March 1987 and February 2017 - March 2018 to ensure comparison of equal time periods.

To see the differences in the zooplankton community, individual identified taxa were categorized into five main groups: rotifers, cyclopoid copepods, calanoid copepods, nauplii and cladocerans. Rotifers are typically small and primarily consume organic particles and bacteria. Copepods are larger in size than rotifers and are divided into cyclopoid copepods and calanoid copepods. Nauplii are copepod juveniles that lack characteristics to appropriately be classified. Finally, cladocerans are larger and include taxa such as *Daphnia*. Cladocerans are the most desirable food source for many planktivorous fish followed by.

The average and cyclopoid copepods decreased from 10.0 organisms/L to 3.6 organisms/L after alum. On the other hand, rotifers were similarly abundant before and after treatments, with a slight increase in average densities from 20.2 organisms/L in 1986 - 1987 to 25.8 organisms/L post-alum.

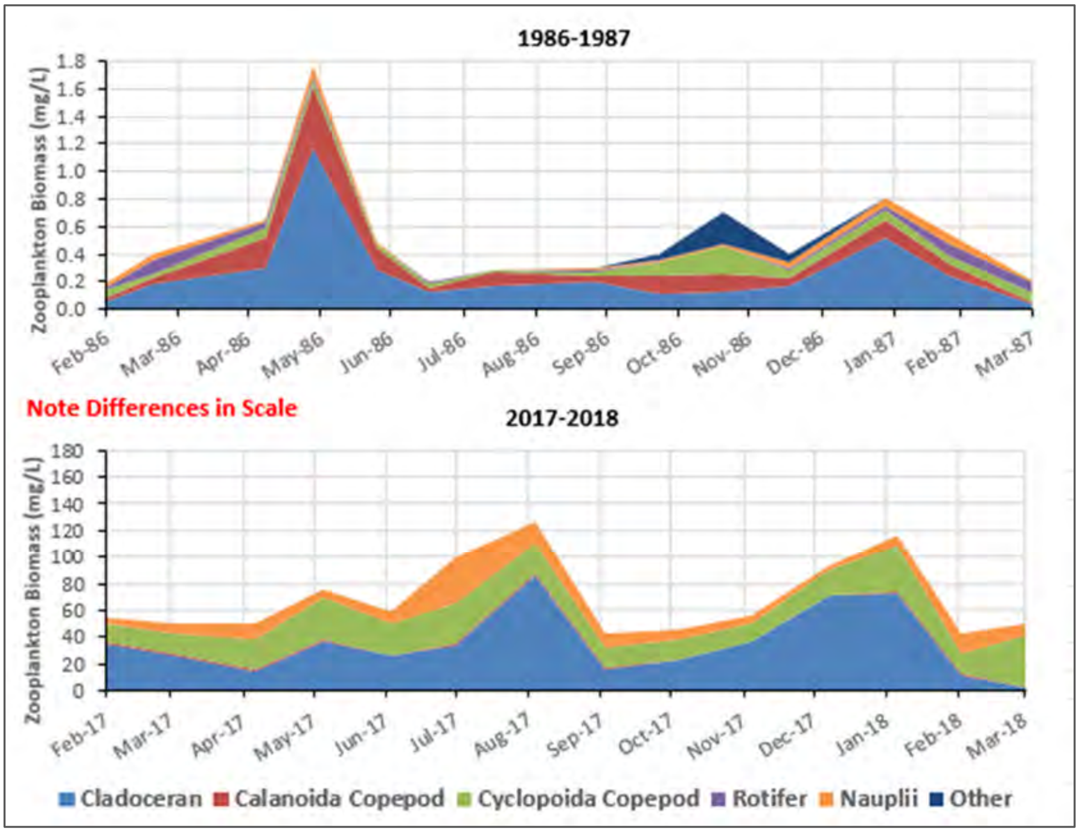
Figure 25: Lake Stevens zooplankton abundance by group pre-alum in 1986 - 1987 (Top figure) and post-alum in 2017 - 2018 (bottom figure).



While the overall density of zooplankton may have decreased, the biomass significantly increased following alum treatments with an annual average of all zooplankton increasing from 0.05 mg/L to were less abundant but larger in average size after alum. Rotifers, on the other hand, saw a slight decrease in biomass per organism from 0.014 to 0.0002 mg/L. However, the rotifers are so small in comparison that they contribute less than 1% of the total biomass.

While the total biomass changes were large, smaller shifts were observed in the community composition (Figure 26, Table 16, Table 17). Cladoceran before and after alum treatment comprised just under 50% of the biomass. The community of copepods did shift with calanoid copepods decreasing from 21% to 2% of the community while cyclopoid copepods increased from 15% to 34%). Rotifers decreased from 7% of the biomass to 0.01%.

Figure 26: Lake Stevens zooplankton biomass by group pre-alum in 1986 - 1987 (top figure) and post-alum in 2017 - 2018 (bottom figure). Note the 100X difference in scale between time periods.



While restoration efforts may be one influence on zooplankton abundance and structure, another potential factor is the relative size and number of planktivorous fish planted in Lake Stevens. Unfortunately, fish stocking dates, numbers and weights are only available from Washington Department of Fish and Wildlife from 2010 - 2021, so their impact cannot be analyzed over the entire dataset.

Within the past ten years, there were steady stocking rates for kokanee with annual estimates of around 150,000 fish, or about 280/ha (note the lake surface area is 530 ha). Small rainbow trout (~0.04 lbs) were stocked eight times over the past ten years with the last planting in March 2015. Five large rainbow trout stockings between 2011 and 2015 added about 1,700 to 2,500 pounds of fish each time at about 115 fish/ha. In biomass, fish stockings since 2010 have averaged about 0.8 lbs/ha (0.4 kg/ha) and 2.6 lbs/ha (1.2 kg/ha) of kokanee and rainbow, respectively.

The stocking rates observed when the recent zooplankton data were collected are very low. The rate is likely low because Lake Stevens is an oligotrophic lake meaning it is naturally less productive and would not have sufficient phytoplankton to sustain larger populations. Kokanee and small rainbow trout select for large zooplankton, like *Daphnia* in the cladoceran group, but at these low stocking rates there may not have had a large effect on zooplankton abundance. Because there is no zooplankton data available for the time when both kokanee and rainbow trout were planted, it is impossible to determine if the stocking of both these fish species together may have previously had an impact on the zooplankton abundance and community.

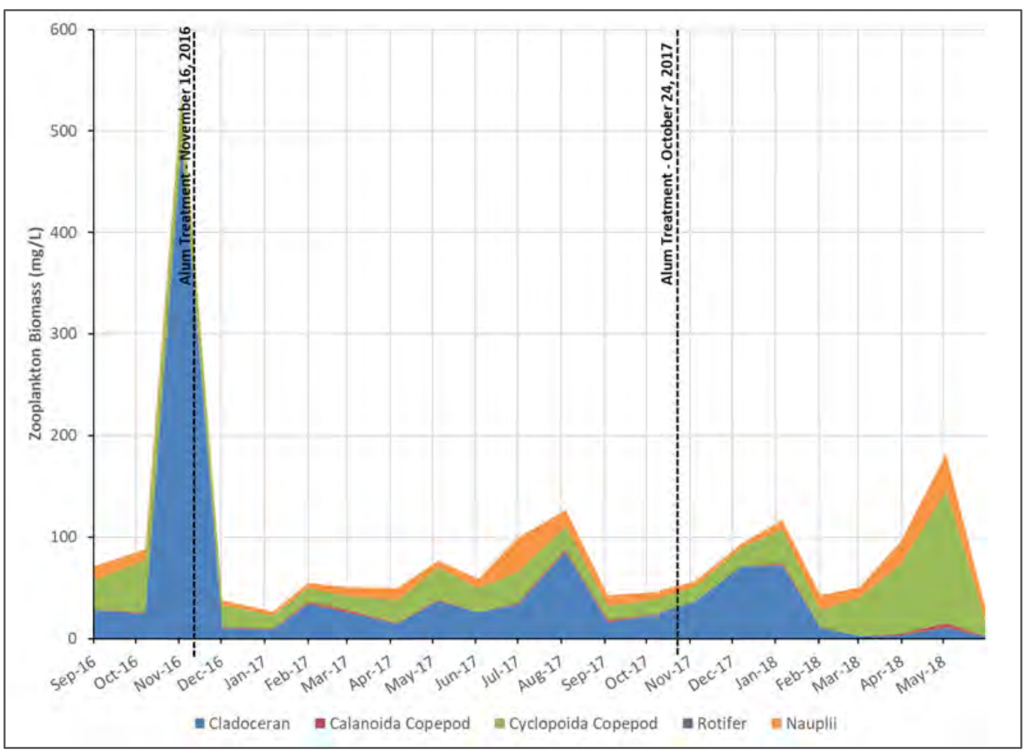
Overall, the restoration efforts do appear to have caused long-term changes in the zooplankton community by slightly decreasing overall density of zooplankton but dramatically increasing the biomass. The changes appear to be due to increases in the average size of individual zooplankton. Additionally, rotifer biomass has decreased and been replaced by larger copepod populations.

The second goal of zooplankton analysis was to evaluate potential impact of alum treatments on the zooplankton community as it relates to supporting the kokanee fishery. Like phytoplankton, zooplankton can be caught in the alum flocculent during treatment and settle out to the bottom causing short-term impacts. Long-term, alum treatments can indirectly influence zooplankton by decreasing phosphorus and, in turn, phytoplankton which are an important food source. However, alum could also affect the phytoplankton community by decreasing cyanobacteria, increasing the abundance of more desirable algal species, and improving zooplankton grazing opportunities.

Samples taken before and after the 2016 and 2017 alum treatments can be used to evaluate the short-term impacts (Figure 27). In 2016, there was a peak in cladoceran density and biomass that was occurring before the fall alum treatment that declined sharply following the treatment In contrast to the 2016 treatment, cladoceran density and biomass increased following the fall 2017 treatment. In both years, treatment corresponds to a period of destratification for the lake, making it difficult to assess which factor was influencing the zooplankton population.

With only two years of zooplankton data collected right before and right after an alum treatment, it is difficult to determine whether the alum treatments have short-term impacts to zooplankton abundance. However, the annual alum treatments do not appear to directly inhibit nor enhance the cladoceran population dynamics; rather, it appears that water column mixing dynamics and potentially fish stocking may have more of a short-term impact on the zooplankton community than the alum treatments.

Figure 27: Lake Stevens monthly zooplankton biomass with annual alum treatment dates.



4.4. BIOLOGICAL ANALYSIS CONCLUSIONS

The intended goal of lake restoration, including the alum treatments, was to reduce the abundance of potentially toxic cyanobacteria and associated algal blooms. Since alum treatments began, there has been a significant reduction in the overall abundance and biomass of phytoplankton. The lower level of algae in the water is supported by the lower chl values. Most importantly, there has also been a shift in the community assemblage from a cyanobacteria-dominated system to a more diverse system that includes a higher proportion of chryospyhtes/cryptophytes and dinophytes. The shift in the dominance of cyanobacteria abundance can be seen as compared to before restoration and compared to the late years of aeration. The shift away from cyanobacteria dominated system has also resulted in the absence of toxic algal bloom reports starting in 2016 - the present.

An additional goal was to determine if the alum treatments are causing negative impacts to the zooplankton community as it relates to the kokanee fishery. The limited data available show that the abundance of desirable zooplankton (cladocerans and copepods) decreased while the biomass increased meaning the zooplankton are less numerous but larger following treatments. In addition, total biomass of zooplankton was also much higher compared to pre-restoration values. The data were not sufficient to understand any potential short-term impacts as the two years of data had conflicting results with one year showing lower zooplankton immediately following treatments and another year showing increases. The difference between years may also be a difference in seasonal factors that also affect zooplankton such as timing of seasonal mixing and fish stocking.

The biological analysis is limited because of the lack of consistent monitoring data. To better evaluate long-term changes, it is recommended that the City begin regular collection of biological data to inform the impact of management activities and environmental conditions on the overall lake health.

Table 6. Lake Stevens phytoplankton abundance, 2009 - 2011. Collected as part of REHAB study from shoreline.

Date		Density (#/mL)								% Cyanophyte
		Bacillariophyte	Chlorophyte	Chryso- & Cryptophyte	Cyanophyte	Dinophyte	Euglenophyte	Unknown	Total	
6/14/2009		113	180	137	9726	1			10,157	96%
6/28/2009		7	151	242	35390				35,790	99%
7/12/2009		3	291	616	42042	6			42,958	98%
7/26/2009		17	238	209	17543	9		8	18,024	97%
8/9/2009		20	225	181	19187	4		3	19,620	98%
8/23/2009		8	475	173	27497	3		8	28,164	98%
9/7/2009		11	121	150	13544	2	1	40	13,869	98%
9/20/2009		37	74	324	8387	5			8,827	95%
10/4/2009		42	183	143	7865	1	4		8,238	95%
10/18/2009		201	120	434	16998	3	4		17,760	96%
6/6/2010		448	583	355	29852	4			31,242	96%
6/20/2010		137	206	189	64302	2			64,836	99%
7/11/2010		25	95	310	4950	12		2	5,394	92%
7/25/2010		52	142	284	9760	29	2	20	10,289	95%
8/8/2010		362	227	401	37324	2	2	1	38,319	97%
8/22/2010		93	419	156	134195	5	2		134,870	99%
9/12/2010		105	169	157	228171	8	1		228,611	100%
9/26/2010		163	84	192	145815	31	6	1	146,292	100%
10/10/2010		60	98	321	28024	6	9		28,518	98%
10/24/2010		102	218	228	13717	1	5		14,271	96%
6/5/2011		710	1456	197	23290		3		25,656	91%
6/19/2011		148	1061	287	46270	3	4		47,773	97%
7/10/2011		24	260	139	12714	2		1	13,140	97%
7/24/2011		69	308	99	63672	5	3	3	64,159	99%
8/7/2011		35	216	70	74150	2	1	1	74,475	100%
8/21/2011		10	74	122	66316	5	1	5	66,533	100%
9/12/2011		116	139	126	53070	2		1	53,454	99%
9/25/2011		1297	138	301	79955	7	2		81,700	98%
10/9/2011		1317	69	203	16574	3			18,166	91%
10/23/2011		292	240	173	23520	1	6		24,232	97%

Table 7. Lake Stevens phytoplankton abundance, 2016 - 2018. Samples collected at the shoreline.

Date	Density (#/mL)						% Cyanophytes
	Bacillariophyte	Chlorophyte	Chryso- & Cryptophyte	Cyanophyte	Dinophyte	Total	
9/13/2016	48.8	195.1	1536.7	1097.6	97.6	2976	37%
10/27/2016		146.3	1048.8	804.9	73.2	2073	39%
11/15/2016	48.8	146.3	1170.8	780.5	48.8	2195	36%
12/14/2016	48.8	170.7	1268.3	756.1		2244	34%
1/19/2017	97.6	146.3	902.5	1000.0	73.2	2220	45%
2/13/2017	146.3	292.7	1195.2	1048.8	97.6	2781	38%
3/14/2017	1000.0	341.5	1585.4	731.7	24.4	3683	20%
4/18/2017	1170.8	146.3	1341.5	902.5	97.6	3659	25%
5/17/2017	902.5	536.6	1097.6	439.0	24.4	3000	15%
6/15/2017	609.8	146.3	1073.2	561.0	146.3	2537	22%
7/13/2017	24.4	73.2	1048.8	1219.6	195.1	2561	48%
8/16/2017		170.7	1219.6	1756.2	73.2	3220	55%
9/15/2017		219.5	902.5	439.0	122.0	1683	26%
10/17/2017		170.7	1463.5	536.6	73.2	2244	24%
10/27/2017		268.3	1122.0	1585.4	97.6	3073	52%
11/16/2017	146.3	122.0	1097.6	463.4	48.8	1878	25%
12/19/2017	24.4	195.1	731.7	317.1	24.4	1293	25%
1/18/2018	73.2	73.2	585.4	1683.0	48.8	2464	68%
2/14/2018	853.7	48.8	853.7	390.3	97.6	2244	17%
3/14/2018		146.3	1268.3	658.6	73.2	2146	31%
4/12/2018	24.4	97.6	1439.1	439.0	170.7	2171	20%
5/15/2018		97.6	780.5	390.3	73.2	1342	29%

Table 8. Lake Stevens phytoplankton abundance, 1986 - 1987. Collected from deep, open water station and composited over top 10 m.

Date	Density (#/mL)								% Cyanophyte
	Bacillariophyte	Chlorophyte	Chryso- & Cryptophyte	Cyanophyte	Dinophyte	Euglenophyte	Unknown	Total	
2/28/1986	374.6	40.2	30.1		0.6	2.6	107.6	556	0%
3/20/1986	3265.7	180.3	34.9			1.7	105.1	3,588	0%
5/7/1986	169.1	411.9	51.7	120.3		1	160.9	915	13%
5/28/1986	20.0	346.2	50.7	461.7			75.5	954	48%
6/24/1986	38.3	451.1	0.8	229.1			74.7	794	29%
7/16/1986	20.3	46.1	18.3	115.7	0.2		133.3	334	35%
8/13/1986	19.2	214.9	17.5	265.5	0.3		30	547	49%
9/25/1986	5.8	67.8	25.4	87.9	0.1		159.3	346	25%
10/23/1986	7.1	497.0	45.0	88.7	0.2	1.9	443.3	1,083	8%
11/19/1986	54.8	92.3	70.6	178.4	0.1	1.6	264.4	662	27%
12/17/1986	44.5	50.4	42.6	111.4		1.9	251.9	503	22%
1/27/1987	57.5	25.6	17.9	50.3		1.1	150.5	303	17%
2/24/1987	226.5	35.6	17.8	3843.8			107.1	4,231	91%
3/31/1987	419.6		76.2	20898.4			96.2	21,490	97%

Table 9. Lake Stevens summer (June - October) phytoplankton abundance, 1986, 2009 - 2011, and 2017. Samples collected in 1987 were from deep, open water station and composited over top 10 m. Samples collected in 2009 - 2011 and 2017 were from the shoreline.

Date	Density (#/mL)							
	Bacillariophyte	Chlorophyte	Chryso- & Cryptophyte	Cyanophyte	Dinophyte	Euglenophyte	Unknown	Total
Jun - Oct 1986	18	255	21	157	0	2	168	623
Jun - Oct 2009	46	206	261	19,818	4	3	15	20,352
Jun - Oct 2010	155	224	259	69,611	10	4	6	70,269
Jun - Oct 2011	402	396	172	45,953	3	3	2	46,931
Jun - Oct 2017	317	175	1,138	1,016	118			2,764

Table 10. Lake Stevens phytoplankton biovolume, 2016 - 2018. Samples collected at the shoreline.

Date	Biovolume (mm^3/L)						% Cyanophytes
	Bacillariophyte	Chlorophyte	Chryso- & Cryptophyte	Cyanophyte	Dinophyte	Total	
9/13/2016	0.005	0.015	0.045	0.008	0.029	0.10	8%
10/27/2016		0.018	0.053	0.003	0.020	0.09	3%
11/15/2016	0.006	0.010	0.069	0.003	0.015	0.10	3%
12/14/2016	0.004	0.015	0.064	0.003		0.09	3%
1/19/2017	0.033	0.013	0.040	0.010	0.020	0.12	8%
2/13/2017	0.023	0.028	0.044	0.004	0.044	0.14	3%
3/14/2017	0.122	0.026	0.058	0.003	0.010	0.22	1%
4/18/2017	0.149	0.017	0.057	0.004	0.024	0.25	1%
5/17/2017	0.090	0.112	0.061	0.002	0.010	0.27	1%
6/15/2017	0.064	0.013	0.068	0.015	0.044	0.20	7%
7/13/2017	0.002	0.004	0.063	0.005	0.059	0.13	4%
8/16/2017		0.020	0.048	0.009	0.020	0.10	9%
9/15/2017		0.014	0.040	0.002	0.029	0.09	2%
10/17/2017		0.011	0.071	0.002	0.034	0.12	2%
10/27/2017		0.030	0.065	0.006	0.024	0.12	5%
11/16/2017	0.051	0.011	0.067	0.002	0.010	0.14	1%
12/19/2017	0.009	0.132	0.056	0.001	0.005	0.20	1%
1/18/2018	0.020	0.009	0.030	0.029	0.015	0.10	29%
2/14/2018	0.113	0.005	0.057	0.002	0.044	0.22	1%
3/14/2018		0.020	0.104	0.002	0.020	0.15	2%
4/12/2018	0.002	0.065	0.104	0.002	0.044	0.22	1%
5/15/2018		0.007	0.025	0.002	0.024	0.06	3%

Table 11. Lake Stevens phytoplankton biovolume, 2009 - 2011. Collected as part of REHAB study from shoreline.

Date	Biovolume (mm^3/L)								% Cyanophyte
	Bacillariophyte	Chlorophyte	Chryso- & Cryptophyte	Cyanophyte	Dinophyte	Euglenophyte	Unknown	Total	
6/14/2009	0.226	0.179	0.086	0.104	0.002			0.6	17%
6/28/2009	0.054	0.088	0.096	0.298				0.5	56%
7/12/2009	0.027	0.077	0.349	0.144	0.060			0.7	22%
7/26/2009	0.128	0.085	0.135	0.068	0.027		0.016	0.5	15%
8/9/2009	0.084	0.104	0.119	0.127	0.012		0.006	0.5	28%
8/23/2009	0.047	0.199	0.104	0.089	0.011		0.032	0.5	19%
9/7/2009	0.116	0.060	0.107	0.026	0.006	0.003	0.003	0.3	8%
9/20/2009	0.237	0.058	0.210	0.063	0.014			0.6	11%
10/4/2009	0.080	0.408	0.127	0.037	0.002	0.012		0.7	6%
10/18/2009	0.136	0.067	0.320	0.154	0.110	0.012		0.8	19%
6/6/2010	5.336	0.122	0.142	0.151	0.100			5.9	3%
6/20/2010	1.730	0.152	0.088	0.345	0.028			2.3	15%
7/11/2010	0.273	0.074	0.235	0.030	0.236		0.008	0.9	4%
7/25/2010	0.238	0.069	0.244	0.047	0.113	0.006	0.050	0.8	6%
8/8/2010	0.433	0.129	0.191	0.205	0.065	0.006	0.004	1.0	20%
8/22/2010	0.096	0.195	0.106	0.461	0.186	0.006		1.1	44%
9/12/2010	0.133	0.085	0.083	0.977	0.244	0.003		1.5	64%
9/26/2010	0.130	0.033	0.157	0.492	1.695	0.023	0.004	2.5	19%
10/10/2010	0.059	0.037	0.230	0.187	0.200	0.034		0.7	25%
10/24/2010	0.078	0.044	0.301	0.099	0.025	0.015		0.6	18%
6/5/2011	4.157	0.572	0.231	0.133		0.009		5.1	3%
6/19/2011	1.525	0.098	0.169	0.401	0.110	0.012		2.3	17%
7/10/2011	0.188	0.109	0.122	0.101	0.120		0.004	0.6	16%
7/24/2011	0.280	0.387	0.083	0.291	0.242	0.009	0.012	1.3	22%
8/7/2011	0.455	0.060	0.077	0.231	0.050	0.003	0.004	0.9	26%
8/21/2011	0.121	0.065	0.091	0.213	0.064	0.003	0.020	0.6	37%
9/12/2011	0.132	0.082	0.093	0.233	0.065		0.004	0.6	38%
9/25/2011	0.753	0.139	0.237	0.298	0.065	0.006		1.5	20%
10/9/2011	0.838	0.086	0.168	0.086	0.053			1.2	7%
10/23/2011	0.322	0.249	0.154	0.126	0.005	0.018		0.9	14%

Table 12. Lake Stevens phytoplankton biovolume, 1986 - 1987. Collected from deep station and composited over top 10 m.

Date	Biovolume (mm^3/L)								% Cyanophyte
	Bacillariophyte	Chlorophyte	Chryso- & Cryptophyte	Cyanophyte	Dinophyte	Euglenophyte	Unknown	Total	
2/28/1986	0.5204	0.0119	0.0489		0.0013	0.0053	0.0068	0.6	0%
3/20/1986	2.1298	0.0134	0.0761			0.0034	0.0073	2.2	0%
5/7/1986	0.4749	0.1026	0.1064	0.0570		0.0020	0.0123	0.8	8%
5/28/1986	0.1463	0.0966	0.0549	0.2516			0.0063	0.6	45%
6/24/1986	0.7706	0.0611	0.0014	0.1147			0.0046	1.0	12%
7/16/1986	0.2580	0.0269	0.0196	0.5481	0.0082		0.0083	0.9	63%
8/13/1986	0.1296	0.0184	0.0266	4.2139	0.0163		0.0019	4.4	96%
9/25/1986	0.0250	0.1048	0.0271	0.1213	0.0041		0.0099	0.3	42%
10/23/1986	0.0822	0.0812	0.0556	0.1767	0.0094	0.0038	0.0276	0.4	40%
11/19/1986	0.7959	0.0294	0.0964	0.1524	0.0038	0.0031	0.0164	1.1	14%
12/17/1986	0.5421	0.0134	0.0648	0.1421		0.0037	0.0166	0.8	18%
1/27/1987	0.1967	0.0034	0.0272	0.0748		0.0022	0.0096	0.3	24%
2/24/1987	0.2716	0.0056	0.0427	2.4271			0.0079	2.8	88%
3/31/1987	0.5861		0.1901	24.7988			0.0060	25.6	97%

Table 13. Lake Stevens summer (June - October) phytoplankton abundance, 1986, 2009 - 2011, and 2016-2017. Samples collected in 1987 were from deep, open water station and composited over top 10 m. Samples collected in 2009 - 2011 and 2017 were from shoreline.

Date	Biovolume (mm3/L)							
	Bacillariophyte	Chlorophyte	Chryso- & Cryptophyte	Cyanophyte	Dinophyte	Euglenophyte	Unknown	Total
Jun - Oct 1986	0.253	0.058	0.026	1.035	0.010	0.004	0.010	1.40
Jun - Oct 2009	0.113	0.133	0.166	0.111	0.027	0.009	0.014	0.57
Jun - Oct 2010	0.850	0.094	0.178	0.299	0.289	0.013	0.017	1.74
Jun - Oct 2011	0.939	0.178	0.141	0.221	0.096	0.009	0.009	1.59
Sep-Oct 2016	0.005	0.017	0.049	0.005	0.024			0.10
Jun - Oct 2017	0.033	0.016	0.059	0.006	0.035			0.15

Table 14. Lake Stevens zooplankton density (# organisms/L), 1986 - 1987.

Date	Density (#/L)						
	Cladoceran	Calanoida Copepod	Cyclopoida Copepod	Nauplii	Rotifer	Other	Total
2/28/1986	0.7	1.0	2.5	29.0	6.0		39.1
3/20/1986	1.9	2.2	5.6	18.7	38.6		67.0
5/7/1986	3.3	3.8	6.0	20.1	30.2		63.3
5/28/1986	10.1	13.7	4.2	35.4	29.7		93.0
6/24/1986	1.9	4.1	1.8	7.7	12.3		27.7
7/16/1986	1.9	0.6	2.9	14.6	6.7		26.6
8/13/1986	11.9	3.7	24.7	1.9	4.7		46.8
9/25/1986	2.9	1.1	4.9	14.9	20.0		43.7
10/23/1986	1.2	1.6	7.7	16.0	12.5	0.0	39.0
11/19/1986	2.0	1.5	12.2	15.3	24.6	0.03	55.7
12/17/1986	3.0	0.8	7.2	63.7	24.9	0.01	99.5
1/27/1987	6.2	3.3	9.6	42.8	27.8		89.6
2/24/1987	2.8	3.0	6.2	34.1	23.0		69.1
3/31/1987	0.9	0.4	4.8	8.9	21.9		36.9
AVERAGE	3.6	2.9	7.2	23.1	20.2	0.0	56.9

Table 15. Lake Stevens zooplankton density (# organisms/L), 2016 - 2018.

Date	Density (#/L)					
	Cladoceran	Calanoida Copepod	Cyclopoida Copepod	Nauplii	Rotifer	Total
9/13/2016	1.5		4.0	11.7	19.1	36.4
10/20/2016	4.4	0.3	8.0	9.2	5.2	27.0
11/15/2016	18.1	0.8	5.9	6.7	4.0	35.6
Alum Treatment - 11/16/2016						
12/14/2016	1.3	0.3	2.9	4.7	14.0	23.2
1/19/2017	1.1	0.1	2.0	3.2	12.6	19.0
2/13/2017	3.0	0.3	2.2	4.6	25.1	35.3
3/14/2017	2.4	0.3	1.8	7.6	30.6	42.7
4/18/2017	2.4	0.2	4.4	10.5	69.0	86.5
5/17/2017	4.7	0.4	4.9	5.4	29.7	45.1
6/15/2017	3.0	0.1	3.0	7.9	18.8	32.8
7/13/2017	2.9	0.3	4.0	30.6	31.0	68.7
8/16/2017	3.8	0.3	3.9	14.6	14.8	37.4
9/15/2017	1.7	0.4	2.4	9.2	15.7	29.4
10/17/2017	1.1	0.1	2.3	6.7	10.2	20.3
Alum Treatment - 10/24/2017						
11/16/2017	1.2	0.0	2.1	5.6	14.7	23.5
12/20/2017	1.9	0.1	2.5	2.8	9.9	17.2
1/18/2018	2.3	0.3	4.9	7.7	27.7	42.9
2/14/2018	0.6	0.2	3.1	12.9	30.6	47.4
3/14/2018	0.3	0.1	6.1	9.0	33.2	48.6
4/12/2018	1.3	0.2	8.5	19.6	55.4	85.1
5/15/2018	4.5	0.5	20.7	32.1	303.2	361.1
6/12/2018	0.4	0.3	2.0	10.7	17.0	30.4
Average (Feb. 2017 - Mar. 2018)	**2.2**	**0.2**	**3.4**	**9.6**	**25.8**	**41.3**

Table 16. Lake Stevens zooplankton biomass (mg/L), 1986 - 1987.

Date	Biomass (mg/L)						
	Cladoceran	Calanoida Copepod	Cyclopoida Copepod	Nauplii	Rotifer	Other	Total
2/28/1986	0.05	0.04	0.06	0.03	0.01		0.2
3/20/1986	0.19	0.04	0.03	0.04	0.10		0.4
5/7/1986	0.30	0.21	0.08	0.02	0.04		0.6
5/28/1986	1.17	0.45	0.04	0.08	0.02		1.8
6/24/1986	0.29	0.16	0.02	0.01	0.01		0.5
7/16/1986	0.12	0.03	0.04	0.01	0.01		0.2
8/13/1986	0.17	0.10	0.01	0.00	0.00		0.3
9/25/1986	0.20	0.04	0.04	0.01	0.01		0.3
10/23/1986	0.11	0.13	0.10	0.01	0.00	0.04	0.4
11/19/1986	0.12	0.14	0.20	0.01	0.01	0.23	0.7
12/17/1986	0.17	0.05	0.07	0.05	0.01	0.06	0.4
1/27/1987	0.52	0.13	0.06	0.06	0.02		0.8
2/24/1987	0.24	0.09	0.05	0.08	0.09		0.5
3/31/1987	0.04	0.02	0.07	0.01	0.07		0.2
AVERAGE	**0.3**	**0.1**	**0.1**	**0.03**	**0.03**	**0.1**	**0.5**

Table 17. Lake Stevens zooplankton biomass (mg/L), 2016 - 2018.

Date	Biomass (mg/L)					
	Cladoceran	Calanoida Copepod	Cyclopoida Copepod	Nauplii	Rotifer	Total
9/13/2016	28.3		29.5	13.0	0.003	70.8
10/20/2016	24.0	1.7	51.8	10.2	0.001	87.6
11/15/2016	488.7	3.6	37.1	7.4	0.001	536.8
12/14/2016	10.1	1.3	21.3	5.2	0.001	37.9
1/19/2017	9.0	0.8	13.4	3.5	0.006	26.7
2/13/2017	34.5	1.9	13.3	5.2	0.003	54.9
3/14/2017	25.7	1.9	14.3	8.4	0.004	50.3
4/18/2017	14.7	0.8	22.3	11.6	0.010	49.4
5/17/2017	37.2	1.6	31.5	6.0	0.003	76.3
6/15/2017	25.5	0.4	24.0	8.8	0.004	58.6
7/13/2017	33.6	1.1	30.9	33.9	0.007	99.4
8/16/2017	85.9	2.0	22.8	16.2	0.002	126.8
9/15/2017	16.2	1.6	14.6	10.2	0.005	42.6
10/17/2017	23.1	0.3	14.6	7.4	0.001	45.4
11/16/2017	36.5	0.1	13.4	6.2	0.004	56.3
12/20/2017	70.5	0.5	18.9	3.1	0.003	93.1
1/18/2018	72.6	1.5	33.7	8.6	0.006	116.4
2/14/2018	10.9	1.0	16.1	14.2	0.005	42.3
3/14/2018	2.4	0.2	37.7	9.9	0.010	50.3
4/12/2018	3.0	1.4	68.6	21.7	0.015	94.6
5/15/2018	11.0	4.3	130.3	35.6	0.019	181.3
6/12/2018	1.4	1.3	15.4	11.8	0.003	30.0
Average (Feb. 2017 - Mar. 2018)	**34.9**	**11.1**	**22.0**	**10.7**	**0.005**	**68.7**

5. LAKE MANAGEMENT RECOMMENDATIONS

5.1. IN-LAKE MANAGEMENT RECOMMENDATIONS

The annual alum treatments, which started in 2013, have been highly successful in improving the health of Lake Stevens and reducing the frequency of potentially toxic algal blooms. The alum has nearly eliminated internal loading by inactivating phosphorus in the lake sediments. The treatments have also reduced the effect of ongoing external phosphorus loading to the lake by stripping phosphorus out of the water column. When phosphorus is removed from the water column with alum, it settles to the lake bottom and is permanently bound to aluminum and not available to be recycled back into the lake as internal loading.

Given the success of the alum treatments to date, one management option considered was to stop all future alum treatments. However, this option is not recommended as there is still a high degree of uncertainty of the potential lake response to ongoing external phosphorus loading and potential future sediment release. Without the annual treatments, any phosphorus that enters the lake from external phosphorus inputs will continue, and sources will recharge the sediment phosphorus reserves over time. The response at Lake Stevens is also magnified because of the lake's naturally low level of summer flushing. Since there is no current information on the amount of external pollution to the lake, it cannot be estimated how long it would be until alum treatments would again be needed to maintain the improved water quality conditions.

Therefore, the recommended management approach is for the City to continue the small alum treatments, but at a reduced frequency of every 2 to 4 years. The approach will provide cost savings for the City while still ensuring that lake water quality is maintained. Spreading out the treatments will also allow the City to identify the lake's response in the years without alum treatments and adapt treatment frequency.

The new, lower-frequency treatments should be applied at least until the full sediment inactivation dose originally recommended in the 2012 study has been applied to the lake. The original dose of 1.17 mg Al/L was based on inactivating all mobile phosphorus in the top 4 cm of the lake. As of 2020, 1.17 mg Al/L or about three-fourths of the treatment has been applied. At the current rate of 0.15 mg Al/L, it would take approximately three more treatments to achieve the full dose. When the full dose is applied, the need for future alum treatments should be reviewed and further informed by additional monitoring data collected during that period.

5.1.1. Future Treatment Frequency

It is recommended that the next alum treatment occur in the fall of 2022 since there was no alum application in 2021. Treatments should then continue every 2 to 4 years. To determine the need for a treatment, the City should examine the lake monitoring data each year to determine if a treatment is needed. While professional judgement should be used when interpreting data to make management decisions, two recommended metrics that to conduct more frequent treatments are as follows:

- **Spring phosphorus concentrations** – Elevated spring concentrations can be a good indicator for reduced effectiveness of alum treatments. They can reflect 1) a surge in nutrients from the lake watershed or 2) a return of "holdover phosphorus" or phosphorus released from the lake sediment the previous year. March TP concentrations that exceed 20 ug/L would be approaching pre-alum treatment spring levels and indicate treatment is needed.

- **High annual precipitation** – Years with higher precipitation are going to increase the external load of pollution washing into the lake. The maintenance treatments are recommended, in part, to neutralize ongoing phosphorus pollution into the lake. Therefore, if there is higher than average rainfall, a maintenance treatment should be considered.

If neither of the above triggers are met and data do not otherwise indicate phosphorus increases, maintenance alum treatments should be conducted at least every fourth year to ensure that the external phosphorus load is neutralized, and the lake conditions remain stable and healthy.

5.1.2. Future Treatment Dosing

The dose of each treatment should continue at a rate at or close to 0.15 mg Al/L. This level of dosing has proven to be effective at curtailing internal loading and reducing water column phosphorus concentrations. However, the dose could be slightly adjusted up or down based on observed water column TP concentrations, annual variability in precipitation and budget limitations.

5.1.3. Future Treatment Timing

Future treatments should be conducted in the fall when phosphorus concentrations in the hypolimnion are the highest. A fall treatment will continue to be the most effective at reducing hypolimnetic TP which would normally be released during the fall and winter and fuel spring algae blooms.

5.2. WATERSHED AND EXTERNAL LOADING MANAGEMENT

The external loading of phosphorus to Lake Stevens will likely not decrease in the future, especially considering the effects of climate change and the atmospheric increases, unless there are significant efforts to implement behavioral changes within the lake community. Public outreach and education of the lake community should focus on the importance of reducing nutrient loading to the lake from landscape practices, land-uses, and building maintenance and include best management practices community members can implement. The City should continue their efforts to reduce phosphorus pollution within the watershed and external loading to the lake.

5.3. MONITORING

Lake Stevens has a robust long-term water quality and lake level dataset that has been instrumental in guiding lake management decisions. It is strongly recommended that the City continue the routine monitoring program that began in 2007 as a partnership between the City and the County. Continued monitoring will provide the necessary data to track conditions in the lake. It is also recommended that routine biological sampling, as well as inlet sampling be added to the monitoring program to address data gaps identified during data review. Finally, it is recommended that the sediment core study be repeated once the analysis issue has been resolved. Overall, an ongoing monitoring program will help the City and community evaluate the effectiveness of management activities and ensure future sound management decisions.

5.3.1. Lake Water Quality and Lake Levels

At a minimum, it is recommended that the City should continue to follow the water quality monitoring program implemented by the County which includes monthly monitoring as follows, and as detailed in the Quality Assurance Management Plan (QAMP):

- Lake profiles of DO, temperature, conductivity and pH
- Water samples
- TP at 1, 5, 10, 20, 30 and 40 m
- SRP at 40 m
- Total nitrogen at 1 m
- Chl at 1, 5, and 10 mm
- Water clarity, measured with a Secchi disk
- True water color (taken for two years every decade)

In addition, the City should also continue to collect continuous lake level measurements which was started under the City/County partnership. This data will ensure the City is able to implement its lake level management plan.

It should be noted that starting in 2021, there are new water quality monitoring requirements to satisfy the City's Washington State Department of Ecology Aquatic Plant and Algae Management permit. The monitoring required for any whole or partial lake alum treatment includes pH measurements prior to any alum addition, throughout the duration of treatment, and for 24 hours following treatment completion. The City must also monitor for total and dissolved aluminum, dissolved organic carbon (DOC), and total hardness when conducting an alum treatment. Specific details of the timing and number of samples for this additional monitoring are included in the permit.

5.3.2. Biological

It is recommended that the City begin routine monitoring of the biological community in the lake. Historic data is sparse and has not been consistently collected making it difficult to effectively assess the efficacy of past management efforts on the lake ecology. Not only will these data help the City assess the biological diversity and health of the lake, but they will also improve the ability of the City to adaptively manage the lake. Specifically, phytoplankton monitoring will ensure the management efforts are effectively reducing and preventing harmful toxic algae blooms. Zooplankton analysis will provide valuable information on the direct effects of alum treatments on the zooplankton community and how it relates to the kokanee stocking and fishery productivity. It will also provide information on the indirect alum effects such as improvements to the phytoplankton community structure which enable larger zooplankton production desirable for fisheries. Detecting changes in the biological community will be especially important with the changes in climate that our region will likely experience in the coming decades.

Biological monitoring should include collecting samples for phytoplankton and zooplankton identification and enumeration. Phytoplankton samples should be collected monthly from the upper waters of the established lake monitoring station. It is recommended that equal amounts of water should be collected from 1, 5 and 10 m and composited into one sample for phytoplankton analysis. Zooplankton samples should also be collected from the established lake monitoring station with a vertical tow of a 74 µm net from 35 m to the water surface.

5.3.3. Lake Sediments

It is highly recommended that the sediment of Lake Stevens be reassessed in 5 years to evaluate the adaptive management program to ensure the program is the most cost effective in meeting the management goals of preserving and enhancing the beneficial uses of Lake Stevens. The main reason is to determine if there is a remaining pool of high phosphorus in deeper sediments. If so, phosphorus could migrate or diffuse upwards to surface sediments and be released should alum treatments cease. Reassessment of the phosphorus concentrations in deeper cores of Lake Stevens would show if the high phosphorus at depth has depleted over time or whether it is locked in place and not migrating. It will also confirm if all of the phosphorus from the sediments is effectively neutralized with the alum treatments at the time of the new sediment study.

Sediment cores should be collected in the same three locations as in 2009 and 2021 and analyzed for the same phosphorus fractions and metals. An external study is currently being designed to evaluate the current phosphorus fraction methodology and how it performs on newly treated sediments. Revisions to the phosphorus fraction methodology are expected within the next year. Additionally, separate analysis of TP for each sediment increment is recommended using the EPA 6010 method. Typically, TP is calculated as the sum of the phosphorus fractions. It is recommended that for each sediment increment, TP be determined using both methods. To determine whether the high pool of phosphorus in the deeper sediments is migrating upwards, the core collected at the deep station should be at least 50 cm in length.

6. REFERENCES

Carlson, R.E. 1977. A trophic state index for lakes. Limnol. Oceanogr. 22:361-368.

City of Lake Stevens, 2013. Phosphorus Management. Revised February 6, 2013.

Cooke, G.D., E.B. Welch, S.A. Peterson, and S.A. Nichols. 2005. Restoration and Management of Lakes and Reservoirs. Taylor & Francis 3rd Edition.

DeGasperi, C. 2021. Personal communication. King County.

Downing, J.A., S.B. Watson, and E. McCauley. 2001. Predicting cyanobacteria dominance in lakes. Can. J. Fish Aquat. Sci. 58:1905-1908.

Edmondson, W.T. and J.R. Lehman. 1981. The effect of changes in the nutrient income on the condition of Lake Washington. Limnol. Oceanogr. 26:1-28.

Huser, B. 2021. Personal communication. Swedish University of Agricultural Sciences.

Johnston, B.R. and J.M. Jacoby. 2003. Cyanobacterial toxicity and migration in a mesotrophic lake in western Washington, USA. Hydrobiologia 495: 79-91.

KCM. 1987. Lake Stevens Restoration Phase IIA. Prepared for City of Lake Stevens. December 1987. 154pp.

Kenworthy, B.R. 2000. The factors associated with cyanobacterial toxin production in Lake Sammamish during the summers of 1997 and 1999. MS Thesis, University of Washington, Department of Civil and Environmental Engineering, Seattle, Washington.

Lewandowski, J., I. Schauser, and M. Hupfer. 2003. Long term effects of phosphorus precipitations with alum in hypereutrophic Lake Susser See (Germany). Water Res. 37:3194-3204.

North American Lake Management Society (NALMS). 2019. Presentation and discussion at Lake and Pond Phosphorus Inactivation and Interception Workshop. Dick Osgood, Harry Gibbons, John Holtz, and Joe Bischoff. NALMS annual conference. Burlington, VT. Nov 15, 2019.

Nurnberg, G.K. 1996. Trophic state of clear and colored, soft-and hardwater lakes with special consideration of nutrients, anoxia, phytoplankton and fish. Lake Reserv. Manage. 12:432-447.

Osgood, R.A. 1988. Lake mixes and internal phosphorus dynamics. Arch. Hydrobiol. 113:629-638.

Perakis, S.S., E.B. Welch and J.M. Jacoby. 1996. Sediment-to-water blue-green algal recruitment in response to alum and environmental factors. Hydrobiologia 318:165-177.

Reid, Middleton & Assoc. 1983. Phase 1 Lake Stevens Restoration. City of Lake Stevens.

Rydin, E. and E.B. Welch. 1998. Aluminum dose required to inactivate phosphate in lake sediments. Water Res. 32:3969-2976.

Rydin, E. and E.B. Welch. 1999. Dosing alum to Wisconsin lake sediments based on possible in vivo formation of aluminum bound phosphorus. Lake and Reserv. Manage. 15:324-331.

Rydin, E., B. Huser, and E.B. Welch. 2000. Amount of phosphorus inactivated by alum in Washington Lakes. Limnol. Oceanogr. 45:226-230.

Shapiro, J. W.T. Edmondson, and D.E. Allison. 1971. Changes in the chemical composition of sediments of Lake Washington, 1958-1970. Limnol. Oceanogr. 16(2):437-452.

Snohomish County and Tetra Tech, Inc. 2012. Lake Stevens Hypolimnetic Aeration and Alum Treatments Analysis: Technical Memorandum. Prepared for City of Lake Stevens.

Snohomish County. 2021. Lake Stevens 2021 Health Report. https://snohomishcountywa.gov/5393

Tetra Tech, Inc. 2009. Lake Stevens Sediment Quality Investigation Technical Memorandum. Prepared for Snohomish County Public Works Surface Water Management.

Welch, E.B., H.L. Gibbons, S.K. Brattebo, and H.A. Corson-Rikert. 2017. Progressive conversion of sediment mobile phosphorus to aluminum phosphorus. Lake and Reserv. Manage. 33:2, 205-210.

Welch, E.B., D. Bouchard., J. Frodge, and J. M. Jacoby. 2019. Stability of Lake Sammamish phosphorus despite land use changes. Lake and Reserv. Manage. 35(2):167-180.

Welch, E.B. and J.M. Jacoby. 2001. On determining the principal source of phosphorus causing summer algal blooms in western Washington lakes. Lake and Reservoir Management 17:55

.