Internal Phosphorus Loading, Phytoplankton Blooms, and Zooplankton Diversity In Silver Lake, NY (June-Oct 2018)

Submitted to

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Silver Lake Association Water Quality Committee
by

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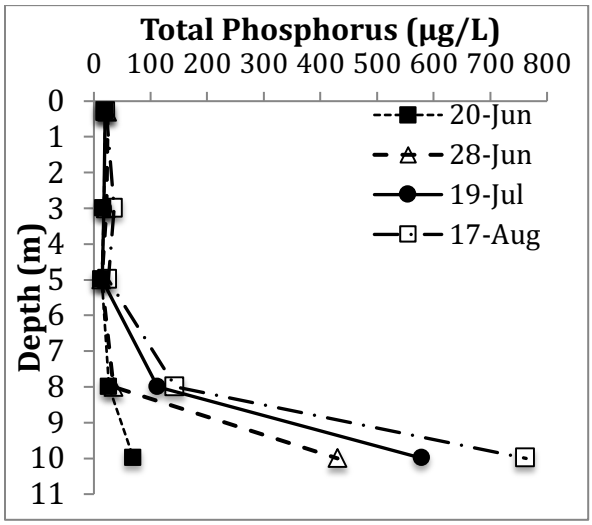
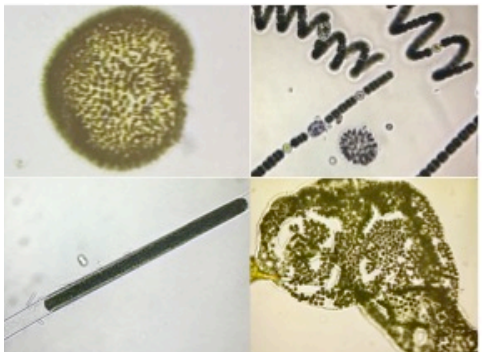
Summary

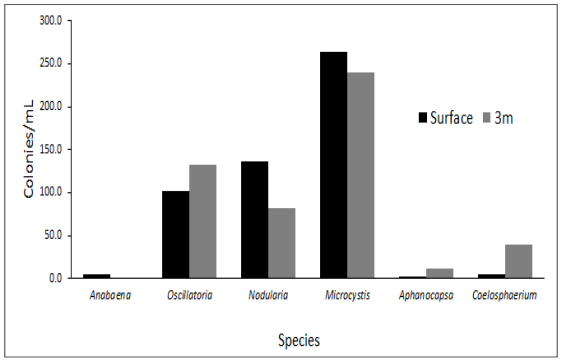
We studied the ecology of Silver Lake from June-October 2018 to document the extent of internal loading of phosphorus (P) from lake sediments, characterize the blooms of cyanobacteria that dominate the lake during the growing season, and to determine the potential role of the herbivorous zooplankton community in helping to control the magnitude of these blooms. Our study yielded three sets of discoveries that are highlighted here and described in scientific detail in subsequent sections of this report.

1. During the period of stratification there was extensive buildup of P along the lake bottom. Much of this buildup can be attributed to the release of sediment relic P from the anoxic sediments. We estimated that 3238 Kg of Total Phosphorus was released from the sediments between June and mid-August (graph below), a mass that is identical to that reported in a 2004 study of Silver Lake by SUNY-Brockport. The calculated release per square meter of lake bottom was higher than that of Conesus Lake, Sodus Bay, and Lake Erie. The proximate cause for this buildup is the development of deoxygenated conditions during the summer period of lake stratification.

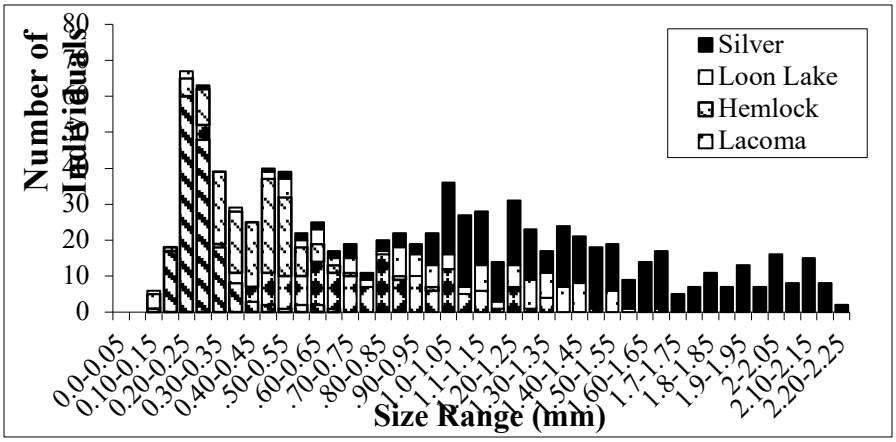


2. A significant biomass of phytoplankton persisted in the water column from June-August, but the largest blooms occurred in September and October during the fall turnover. Much of the phytoplankton biomass consisted of cyanobacteria, including potential HABs forming species such as *Waronechinia, Anabaena (*now *Dolichospermum, Nodularia, Aphanocapsa, Oscillatoria and Microcystis*, which was dominant in September and October samples (pictures and graph below).





3. The herbivorous crustacean zooplankton community of Silver Lake is diverse and abundant. Several species of the genus *Daphnia*, including *D. galeata and D. pulicaria,* reach a large size and achieve a high potential to filter water and consume phytoplankton. These *Daphnia* contribute to a large average size of 1.46 mm for the Silver Lake community compared to nearby lakes such as Conesus Lake (0.27 mm average size).



Conclusions

An excessive supply of internally loaded P is the underlying cause of the phytoplankton blooms in Silver Lake, especially in late summer and early autumn when partial and complete mixing of P from the hypolimnion has occurred. These blooms were dominated by colonies of *Microcystis* and other potentially toxic cyanobacterial species that pose a risk to Silver Lake water quality and to its status as a valuable resource for drinking water, fisheries and prime recreational use. Phosphorus management should be undertaken to improve conditions or at least prevent further loss of water quality. Use of aluminum sulfate (Alum) to deactivate the "phosphorus pump" has been recommended by other workers. New York State will likely permit use of this chemical for management of relic P within a few years and Silver Lake is an excellent candidate for treatment. Another management option for relic P is the oxygenation of deep waters by means of micro-bubble oxygen diffusers that do not disturb lake stratification. This is a more environmentally safe option but may be more difficult logistically to deploy.

Acknowledgements

We wish to express our appreciation to the many residents in the Silver Lake community and to members of the Silver Lake Association, who have been very welcoming and supportive of our studies. Special thanks go to Professor Frank Bright and Bonnie Bright for being such delightful hosts to our big groups. Thanks also to Todd Shuskey, biology teacher at Perry High School, for sharing data and for his exemplary commitment to engaging his students in meaningful scientific research.

Section I
Internal Loading of Phosphorus

Introduction

Internal loading is the process by which chemicals, including nutrients, are released from relic stores in lake sediments into the water column. Phosphorus (P) is a primary component of internally loaded chemicals. Since P is typically the most limiting nutrient in inland waters, internal P loading can have detrimental ecosystem effects by stimulating phytoplankton blooms that degrade water quality and impair ecosystem services, two of the major problems associated with eutrophication.

Internal loading of P is triggered by temperature-dependent density stratification that isolates the cold hypolimnion (deep waters) from the warm epilimnion (surface waters). Nested between the hypolimnion and the epilimnion is a region of rapidly changing temperature (and density) called the thermocline. Surface waters are rich in oxygen because of photosynthetic oxygen production and exchange with air. In a stratified water column most of the oxygen in the isolated hypolimnion is used up by microbes, which in turn respire carbon dioxide. These processes change the chemical environment of the hypolimnion, lowering the pH and creating chemically reducing conditions that cause iron and other metals in the sediments to solubilize. Much of the P adhered to the sediments is released, accumulates in pore water and then is slowly released into bottom waters, where it builds up until fall turnover. This in essence is the phenomenon of internal P loading.

Because summer water column density stratification prevents nutrient-rich bottom waters from mixing to the surface, most of the internally loaded P remains near the bottom unavailable to phytoplankton (i.e. microalgae and cyanobacteria) until the fall turnover. By that time, declining temperatures and light limitation (short days, high sun angle) rather than nutrients alone tend to limit plant growth and generally keep blooms in check. However, vertical mixing may occur during the summer season when high winds or runoff cause turbulence in the water column. Delivery of internally loaded P to surface waters combined with ample light and warm temperatures in summer may serve as a trigger for summer blooms and even harmful algal blooms (HABs) dominated by toxin-producing cyanobacteria. These summer blooms can pose more of a danger than fall blooms due to increased recreational use of lake waters.

The present study evaluates the hypothesis that summer mixing of internally loaded P from the hypolimnion may be contributing to summer cyanobacterial blooms in Silver Lake. Silver Lake is a spring fed lake; a significant amount of cold spring water enters the lake through the deep zones throughout the summer. We estimated from temperature profiles that the spring water has a temperature of about 10°C, possibly making hypolimnetic water colder and denser than it would be otherwise. Thus, an alternative to our primary hypothesis is that the stratification of the water column in Silver Lake is resistant to mixing and internally loaded P may not be an important trigger of summer cyanobacterial blooms.

To explore these hypotheses we sampled water parameters such as temperature, oxygen, oxidation-reduction potential (ORP), conductivity, pH and phosphorus in Silver Lake from June to October 2018. In addition, we set up an *in situ* temperature array in the deeper waters (~10 m depth not far from the Perry Public Beach) with sensors every 0.5-1 m depth that recorded temperature every 15 minutes. These data provided spatial and temporal resolution of temperatures in the lake and were used to detect partial water column mixing events.

Methods

Field surveys of Silver Lake were conducted on six dates throughout the 2018 summer (June 6, June 20, June 28, July 19, Sept 6, Oct 4). A Hach Hydrolab 5A Multiparameter Sonde was calibrated prior to sampling, and used to collect data on temperature, dissolved oxygen, (LDO), oxidation-reduction potential (ORP), *in vivo* chlorophyll *a*, depth, and other parameters reported in **Appendix I.** Water samples for total phosphorus (TP) and soluble reactive phosphorus (SRP) were taken using a 2.2 L Van Dorn water sampler and stored in acid washed plastic bottles. SRP samples were filtered immediately upon collection using 0.45-micron glass fiber filters. All samples were kept on ice during transportation and refrigerated upon return to SUNY-Geneseo. The first set of nutrient samples were sent to Life Sciences Laboratory Inc. for analyses. Subsequent samples were analyzed at the SUNY-Brockport Limnology Laboratory (EPA Lab Code #NY01597), which is applying for NYSDOH ELAP certification

In addition to temperature profiles collected with the Hydrolab, we deployed an *in situ* temperature sensor array over a depth of approximately 10 m near the center of Silver Lake. Starting in early June, temperature data from this array were recorded every 15 minutes and

transmitted via data cable to shore, then loaded remotely to a server at SUNY-Geneseo, where temperatures were monitored throughout the project. (iotdb.geneseo.edu/streams).

The total mass of internally loaded phosphorus during summer stratification in Silver Lake was calculated by measuring the concentration of total phosphorus (TP) at different depths in the hypolimnion at the beginning of the summer (data taken June 20 and analyzed by Life Sciences Laboratory Inc.) and the end of summer, before lake turnover begun (data taken Aug 17 by SUNY-Brockport and analyzed by Dr. Michael Chislock). The difference in hypolimnetic TP between June and August was the estimated total loading. The final measurement was taken on August 17 due to limitations in sampling opportunities and because it was important to sample before there was any large scale mixing of TP to the surface, which we believed could begin in late August as the lake approached turnover. Thus it is possible that our sampling could have underestimated the total internal loading in 2018.

The bottom surface area and volume of Silver Lake was calculated with the aid of a bathymetric map from the NYS Department of Environmental Conservation. ImageJ software (NIH) was used to measure the internal contours of the map. The area of each contour was then used to determine a volume of each stratum using the following relationship (Taube 2000):

$$V = 1/2\ H * (A_1 + A_2 + \text{sqrt}\ (A_1 x A_2))$$

V = volume
H = difference in depth between two successive depth contours
A1= area of the lake within the outer depth contour being considered
A2= area of the lake within the inner contour line under consideration

The Relative Thermal Resistance to Mixing (RTRM) was calculated as a measure of the relative stability of the water column. RTRM is a value for lake's resistance to mixing of the thermocline. This calculation uses the difference in the density of water of the upper layer and the lower layer to characterize the stability between meter depths. The following formulas were used to calculate RTRM (Kwityn 2017):

$$Density\ \left(\frac{mg}{m^3}\right) = (1000 * (1 - \left(\frac{(T + 288.9414)}{508929.2 * (T + 68.12963)}\right)) * (\frac{(T - 3.9863)^2}{1000})$$

$$RTRM = \frac{Density\ of\ upper\ layer - Density\ of\ lower\ layer}{Density\ of\ 4℃ - Density\ of\ 5℃}$$

High RTRM values indicate that a lake is not likely to mix, while low values indicate that a lake is very likely to mix. Maximum RTRM values >80 indicate strong stratification and low probability of a large mixing events (Lake Wonoscopomuc Association, 2017). Max RTRM is easily comparable between lakes since it is not dependent on depth. Temperature profile data taken from our Hydrolab and from the *in situ* temperature array were inserted into R code taken from Kwityn (2017) in order to calculate RTRM.

Results

Stratification

The water column in Silver Lake was strongly stratified throughout the summer, with the mixed layer typically reaching a depth of 5-7 m (**Figure 1**). A second set of temperature profiles from the *in situ* array shows how water column stratification begins to break down in late September. Specifically a significant wind event on the 24th of September (sustained at 10-14 mph from 10 AM to midnight, with gusts reaching 24 mph) increased the depth of the mixed layer from a depth of 8 m to 10 m. **Figure 2** shows seasonal trends in temperature recorded by the *in situ* array from June to October at various depths. The full turnover of the water column is shown to have taken place in early October. One important pattern is the instability of the temperatures at 8 m, a depth that was typically near the bottom of the thermocline. These fluctuations are an indication of mixing between deep waters and surface waters, and during those times nutrients are certainly being delivered to surface habitats occupied by cyanobacteria and other phytoplankton.

Within the mixed layer, the concentrations of oxygen were consistently near saturation capacity, between 90-130% saturation (**Figure 3**). Values above 100% represent super saturation of oxygen due to photosynthesis and surface turbulence. Below the mixed layer there was a sharp decline in oxygen concentration (corresponding to the thermocline area) so that by June 28 the hypolimnion (below depths of 7-8 meters) contained almost no oxygen, while saturation values by the end of summer were very close to zero (**Figure 4**).

The oxidation-reduction potential (ORP) is a function of the concentration of oxygen and carbon dioxide in the water. ORP values below 200 mV promote the release of phosphorus from bottom sediments. As expected, the profiles of ORP were similar to those of oxygen, with high oxidative conditions ranging from 350-450 mV prevailing in the mixed layer and the ORP

decreasing through the thermocline, finally dropping below 200 mv in the hypolimnion usually below a depth of 9 m (**Figure 5**). An important point is that these conditions were already established at depths below 9m by the time we conducted our first profile on June 20 and continued well into September when the ORP was ~100 mv at 10 m (**Figure 6**). Low ORP waters are conducive to internal loading by facilitating the release of phosphorus from the sediment into the water column, based on the principles of the Ferrous cycle (Kalf, 2002).

Water Column Phosphorus and Internal Loading

Concentrations of TP and SRP were consistently low from 20-22 µg/L in the epilimnion to approximately a depth of 7 m. As shown in **Figure 7**, the major increases and highest concentrations of TP occurred in the hypolimnion with the highest concentrations near the lake bottom at a depth of about 10 m. Phosphorus concentrations increased in the hypolimnion throughout the summer (**Table 1, Figure 7**), from 70 µg/L on June 20 to 763 µg/L at 10m on August 17. SRP values followed similar trends, remaining very low (0.0-1.5 µg/L) in surface waters and also building up in the hypolimnion where concentrations of 475 µg/L were detected by August 17 (**Table 1**).

Based on the TP measurements taken on June 20, soon after stratification, and on August 17, we estimate that internal loading of TP in Silver Lake was 3,238 kg P. The areal release rate of P per m^2 of lake bottom in the anoxic zone was calculated to be 9.4 mg /day. **Table 2** compares these results to the areal release rate of phosphorus to Conesus Lake and other regional water bodies based on a 2009 study by Makarewicz and Lewis. While Conesus Lake had a higher total amount of phosphorus added to the water column, the areal release rate for Silver Lake was higher, and given the smaller hypolimnetic volume of Silver Lake it translates into very high concentrations of phosphorous in the hypolimnion.

Water Column Stability

Silver Lake has an active underground spring source that seems to deliver cold water at a temperature of about 10°C to the hypolimnion. This was evident after the fall turnover when the water column temperature was a uniform 17.8-18.6°C while the deepest half-meter of water remained at 10°C, as it had been for most of the summer.

We hypothesized that the thermocline density boundary in Silver Lake should be relatively stable because bottom waters would be colder than typical for the region due to the supply of cold water from the spring source. This tendency is illustrated by a comparison of maximum RTRM values across spring-fed and non-spring-fed lakes in the Finger Lakes region. Maximum values >80 indicate strong stratification and low probability of a large mixing events (Lake Wonoscopomuc Association, 2017). As shown in **Table 3**, higher Max RTRM values (>94) show that spring-fed lakes are more resistant to mixing. We surmise that this higher stability is due to the addition of cold spring water at the bottom, which amplifies the density differences between surface and bottom, creating a stronger thermocline barrier. A t-test shows a significant difference between spring fed and non-spring fed Max RTRM values, with spring fed lakes having a significantly higher Max RTRM ($t=3.0253$, $df=2.0188$, $p\text{-value}=0.047$).

Silver Lake, a spring fed lake, exhibited a low resistance to mixing in late spring and early summer, with Max RTRM values of 49 on June 6 and 51 on June 28. Through the summer the Max RTRM increased and a value of 191 was recorded on July 19. As late summer cooling ensued, the Max RTRM declined as surface waters cooled, showing a value of 65 on September 6 and a value below 5 on October 4, which illustrates that the lake was very likely to mix (**Figure 8**). A slight decrease of the Max RTRM value was recorded on August 7, but overall resistance to mixing remains above 80 in mid summer, which is characteristics of a highly stratified lake.

Despite the relative stability of the water column through mid-summer, we observed episodes of mixing that clearly disrupted the thermocline density barrier. This was evident in temperature data obtained from the array of sensors deployed in the lake (**Figure 2**). For example, in mid-July there was a temperature increase of nearly 2°C at the 8 m sensor, and nearly a 1°C spike in the 10 m sensor. This indicates that there was likely an exchange of water between the epilimnion and hypolimnion, with nutrient laden waters being driven to the surface. After August 20, fluctuations at a depth of 10 m became more frequent until a major mixing event occurred on September 24. This was nearly 3 weeks before the fall turnover. These partial mid-summer mixing events typically took place in days when high winds created considerable turbulence in the water column.

Discussion

The phosphorus (P) Trophic State Index calculated for Silver Lake was 49, which classifies the lake as mesotrophic, but very close to being eutrophic (Carlson, 1977). This value is comparable to those reported for Conesus Lake, which fluctuates from high mesotrophic to mildly eutrophic. High densities and short blooms of cyanobacteria and other phytoplankton were common in Silver Lake throughout summer 2018.

Build-up of P near the bottom is almost certainly an indication of considerable internal loading from relic stores in the sediment. Total phosphorus (TP) above the thermocline stays low throughout most of the summer. Below the thermocline at 7-8 m, however, the TP and soluble reactive phosphorus (SRP) both increase steadily with depth. The calculated value for internal TP loading was 3,238 kg. This is likely an underestimate of the total seasonal loading for two reasons. First, by the time we collected the first samples in June, P build-up in the hypolimnion had already begun. Second, there is evidence that the water column mixed and disrupted the thermocline barrier in July, and those events are likely to have moved P from the hypolimnion to the surface layer. Finally, we collected our last nutrient samples on August 20 and complete water column mixing (and oxygenation) did not take place until early October. This means that internal loading likely continued for another 5-6 weeks after our last collection.

The Relative Thermal Resistance to Mixing indicates that the lake is least likely to mix in July and August, but very likely to mix at the beginning of summer, as well as the end of summer and into early autumn. The movement of the thermocline into deeper water as summer progresses into fall confirms the pattern of thermocline degradation shown in the RTRM values. However, despite the apparent stability of stratification in Silver Lake, monitoring of the temperature array showed signs of mixing from the surface to below the thermocline and near the bottom several times over the summer (**Figure 2**). When this mixing occurs, bursts of phosphorus from the hypolimnion are delivered to surface waters, where they are likely to fuel growth of cyanobacteria (e.g. *Microcystis*) that are routinely seen in summer (See report section on Cyanobacteria).

Silver Lake has been considered a high mesotrophic/eutrophic lake since CSLAP sampling began in 1986. Chl *a* levels and Secchi depths over the years indicate that the lake has been highly productive/eutrophic for all that time. Makarewicz and colleagues (2005) conducted

the first comprehensive analysis of nutrient distribution in deep water and reported that TP below 10 m reached a high of 808.1 μg/L. Maximum values of bottom TP from CSLAP studies are as high as 928 μg/L but the long-term average reported in 2017 was 196 μg/L. We believe the actual values have been much higher at least since 2004, and that the concentrations reported in the present study, near 800 μg/L, are more representative of typical maxima in recent years.

Our research confirms that internal loading is a major component of the phosphorus budget in Silver Lake. The beginning of the fall turnover clearly triggers fall HABs dominated by *Microcystis*. Moreover, we provide evidence that, even during the highly stratified midsummer period, mixing caused by winds is likely to deliver pulses of phosphorus to surface waters that are likely the source of early and mid-summer cyanobacterial growth. Cyanobacteria thrive in warmer waters, and as global temperatures increase, HABs will pose an even greater threat to water quality and lake use. The HABs problem in Silver Lake has not abated in the last 15 yr and there is no reason to believe that it will slow down in the future. Internal loading of P should be targeted as a priority for future water quality management.

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Tables and Figures

Table 1. Total Phosphorous (TP) and Soluble Reactive Phosphorus (SRP) levels in µg/L for collection dates in 2018.

Date	Nutrient Concentrations (µg/L)	0 m	3 m	5 m	8 m	10 m
June 20	**TP**	20	17	14	27	70
June 28	**TP**	24	22	14	35	430
July 19	**TP**	20	18	15	113	580
Aug 17	**TP**	22	36	25	144	763
June 28	**SRP**	1.2	1.3	0.9	14	320
July 19	**SRP**	0.6	0.5	0.7	55.4	350
Aug 17	**SRP**	1.5	1.5	0	77.8	475

Table 2. The TP areal release (per m^2) of lake bottom in Silver Lake calculated indirectly from the buildup in the hypolimnion and divided by the area of lake surface below the thermocline, TP buildup and release in Silver Lake are compared to data from other water bodies in our region taken from Makarewicz, 2009

Internal Loading	Silver Lake 2004	Silver Lake 2018	Conesus Lake 2009	Sodus Bay	Lake Erie
mg P/m^2/day	9.4	9.4	8.7	6.3	7.4
Total (kg P) to mid August	3,171	3,238	8,043		

Table 3. Maximum Thermal Resistance to Mixing for several lakes in the area including Silver Lake and two others known to have significant influx of spring water into the hypolimnion. The average Max RTRM for non-spring lakes was 62.3. The water column of spring fed lakes was much more resistant to mixing with an average max RTRM of 152 .

Lake	Classification	Max RTRM
Honeoye L.	Not Spring-Fed	59
Hemlock L.	Not Spring Fed	66
Conesus L.	Not Spring Fed	62
Loon L.	Spring-Fed	94
Silver L.	Spring-Fed	191
Lacoma L.	Spring-Fed	171

Figures

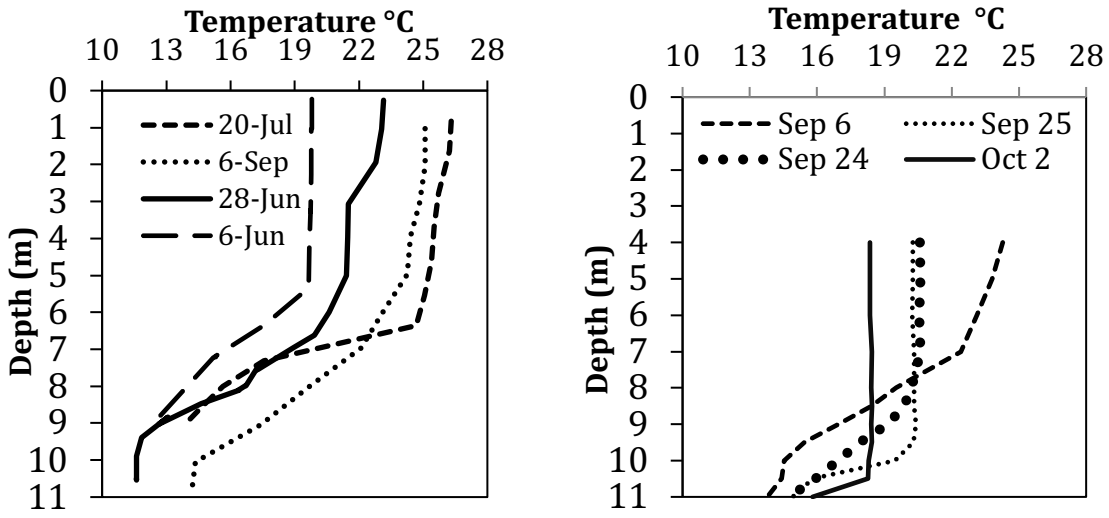

Figure 1. Silver Lake summer temperature stratification is shown here by Hydrolab temperature profiles on the left and *in situ* array sensors on the right.

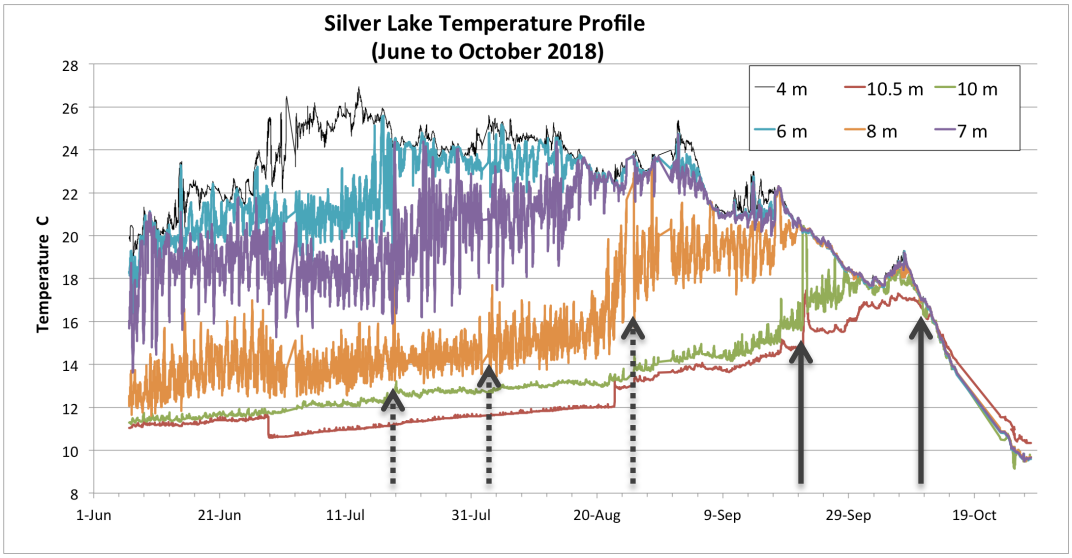

Figure 2. The *in situ* temperature array data for June-October are shown in this figure. Full turnover occurred in early October, but a significant mixing event that extended downward all the way to 10.5 m took place in late September (shown by solid arrows). Note the instability of temperatures at 8 m near the base of thermocline region (shown by dashed arrows).

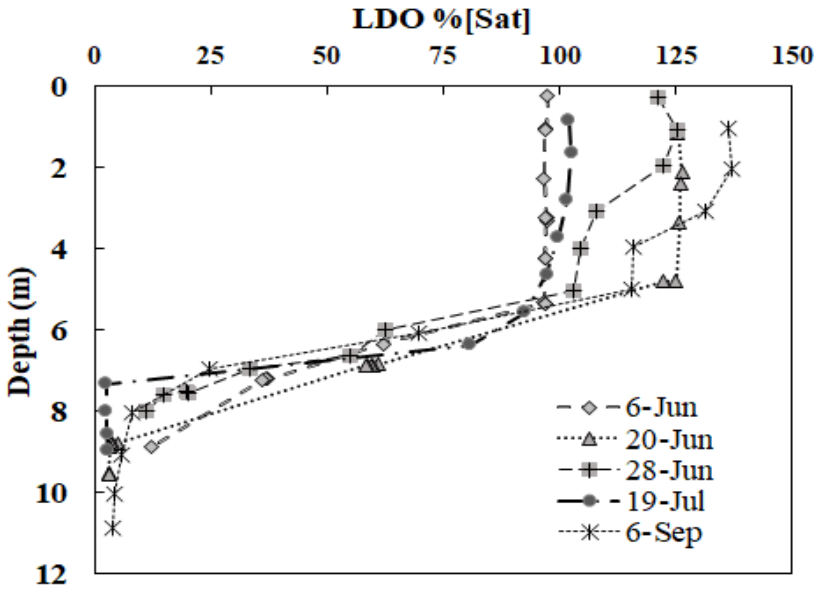

Figure 3. Oxygen saturation profiles showing the hypoxic bottom region of the lake through summer. Low oxygen conditions were already well developed when we started sampling on June 6. Hypoxic conditions near the bottom facilitate the release of phosphorous from the sediments.

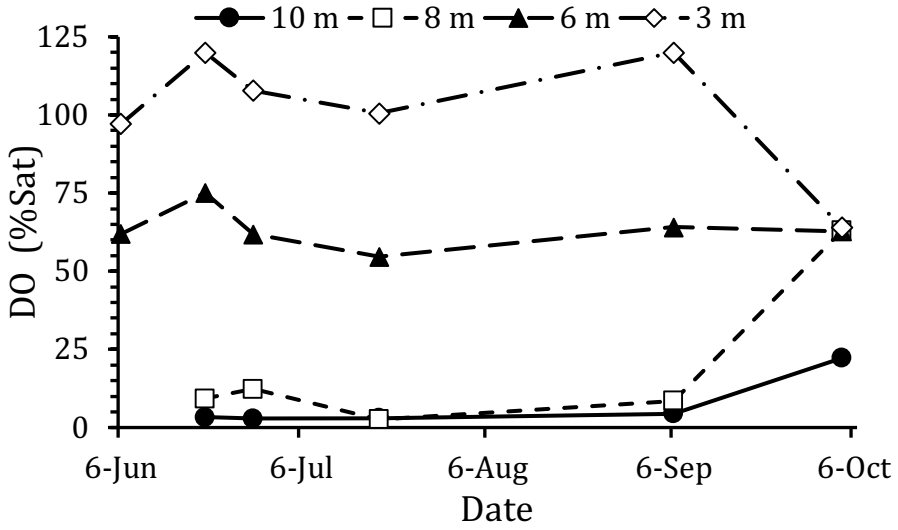

Figure 4. Oxygen saturation for selected depths, highlighting the epilimnion above the thermocline (3, 6 m) and the hypolimnion below the thermocline (8, 10 m).

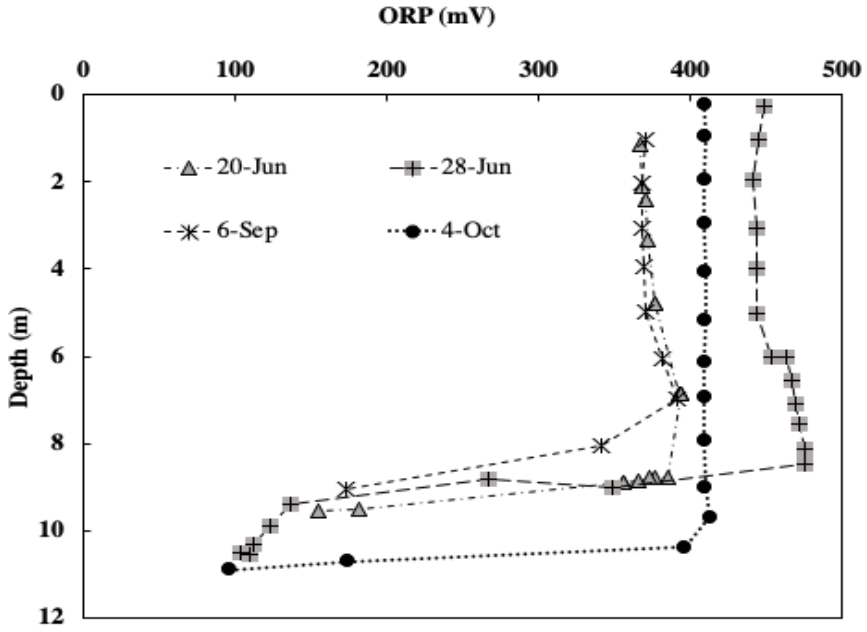

Figure 5. The oxidation-reduction potential decreases with depth below the thermocline during the summer season. Levels below 150-200 mV are indicative of a reducing environment, which facilitates P release from the sediments.

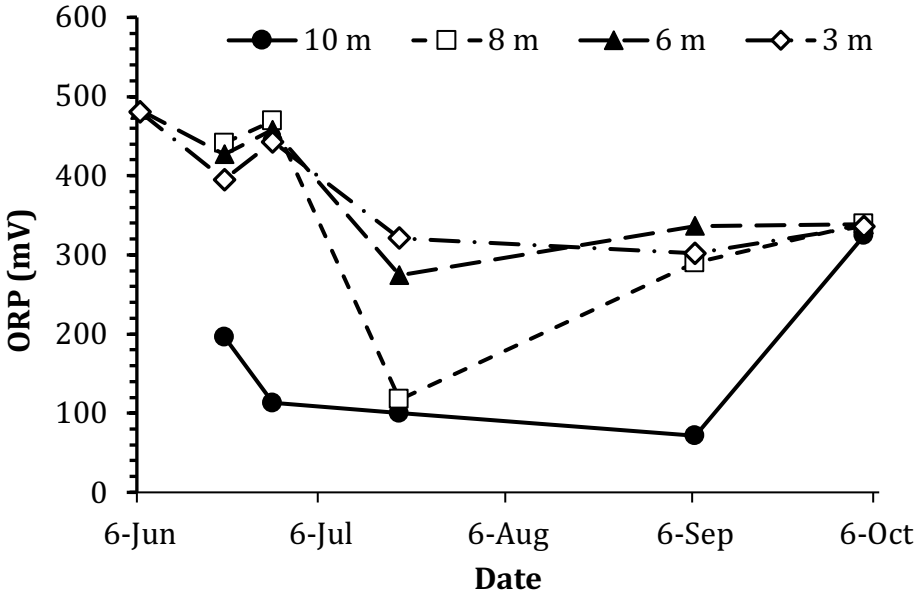

Figure 6. The Oxidation-Reduction Potential for selected depths. Reducing conditions persisted near the bottom but were higher at 8 m by 6 Sep, an indication that some mixing had occurred to those depths.

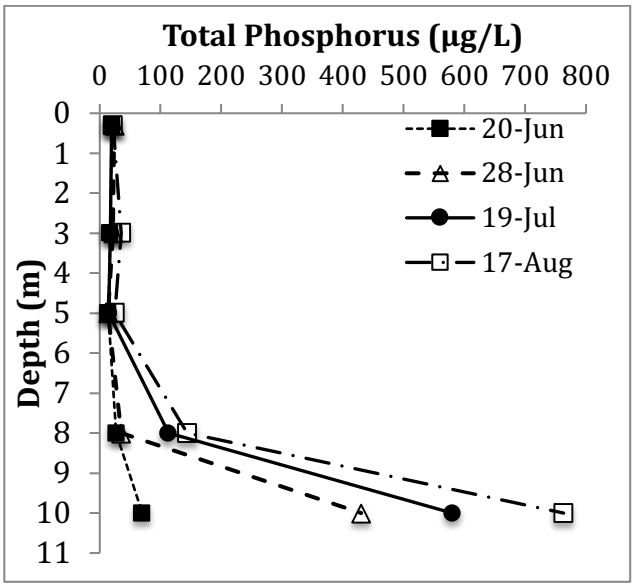

Figure 7. Profiles of total phosphorous (TP) in the water column over summer 2018. The increase in near bottom P is primarily the result of internal loading.

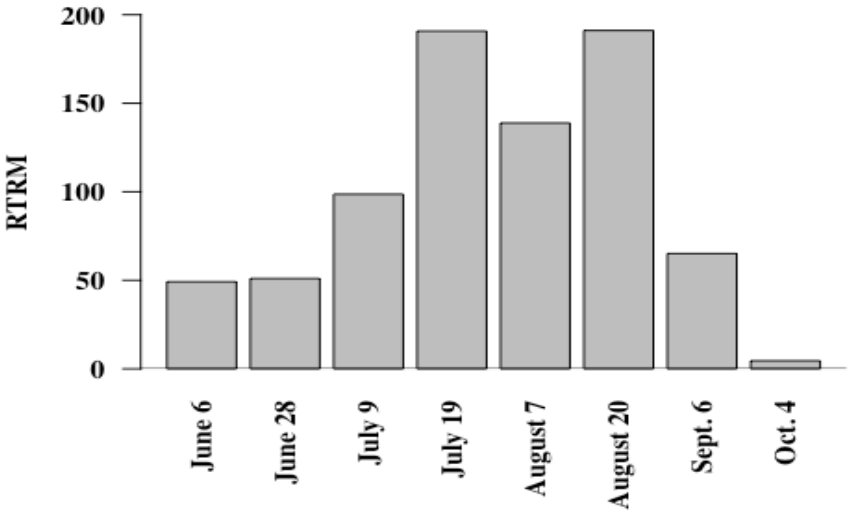

Figure 8. Maximum values of relative thermal resistance to mixing (RTRM) calculated for the period of stratification show that Silver Lake was most resistant to vertical mixing in July and August, and least resistant to vertical mixing in June, September and October.

SECTION II
SURVEY OF CYANOBACTERIA BLOOMS

Introduction

The phytoplankton community forms the base of the food web in lake ecosystems. The term phytoplankton refers to a variety of drifting single-celled and small colonial algae and photosynthetic cyanobacteria (blue-green algae). The cyanobacteria assemblage of the phytoplankton are small single cells or conspicuous colonies of tens to thousands of cells. Cyanobacterial blooms can turn into harmful algal blooms, or HABs, that occur when colonies of toxin producing cyanobacteria dominate the phytoplankton community. The toxins produced by some cyanobacteria can cause illness or death in mammals, birds, or fish that consume enough toxic cells or absorb dissolved toxins. Cases of gastroenteritis and other contact irritations in humans have been documented after individuals came into contact with contaminated recreational and municipal water supplies (Carmichael, 1981). As a group, cyanobacteria generally prefer warmer, nutrient-rich water. There is concern among scientists that blooms will increase as lakes become more eutrophic and global climate continues to warm.

Silver Lake is a meso-eutrophic lake with medium-high levels of nutrients. The Lake has been subject to cyanobacterial blooms since at least 1991, when F.X. Brown Associates (1991) reported that *Anabaena* and *Aphanizomenon* were dominant in the summer phytoplankton community. Mid-lake blooms and dense shoreline accumulations of cyanobacteria were also reported in 2005 by Makarewicz and colleagues. Blooms have continued to compromise Silver Lake water quality in recent years (e.g. CSLAP, 2017). Excess nutrient levels are a major cause of HABs. These nutrients, particularly phosphorus, could come from internal loading of legacy stores within the lake sediments, or from runoff that originates in the surrounding watershed.

The purpose of this study is to investigate the multiple facets of cyanobacterial blooms in Silver Lake. We focus on determining when blooms occurred, the species composition of the blooms, and abundance of cyanobacteria colonies.

Methods

Data collection and field surveys were conducted on six different occasions (6/4, 6/20, 6/28, 7/19, 9/6, 10/4) in Silver Lake. Water samples were collected using a 2.2 L Van Dorn sampler in the late afternoon over a depth of 10 meters. Specific depths included the surface, 3m,

6m, 9m, and 0.5-1 m above the lake bottom. The water samples were kept in plastic bottles and stored on ice for transport. Turbidity was measured using a HACH 2100Q turbidity meter in units of NTU (nephelometric turbidity units), and a 20 cm diameter Secchi-disk was used to measure Secchi depth. A Hach Hydrolab 5A Multiparameter Sonde equipped with an onboard Turner fluorometer was used to measure relative fluorescence (*in vivo* chlorophyll) at various depths. Samples for extracted chlorophyll a (chl *a*) analyses were stored in amber bottles and kept on ice during transportation. Once in the lab, these samples were filtered using a Whatman GF/F fiber filter, wrapped in aluminum foil, and stored at -20 °C. The filters were later submerged in alkaline 90% acetone, ground using a tissue grinder, and then stored in a refrigerator for an extraction time of 18 hours. After centrifugation at 3,000 rpm for 3 minutes, the clear acetone supernatant was separated and loaded into a calibrated Turner Trilogy fluorometer for determination of chl *a* concentration using the acidification method (EPA Method 445.0). A standard curve for chlorophyll in 90% alkalized acetone was created with chl *a* standard solution purchased from the Sigma Chemical Company.

Phytoplankton species were analyzed under a microscope for samples taken on September 6 and October 4 from both the surface and 3m with a VanDorn sampler, and from vertical net hauls to 7m. The different species observed were tallied to determine the relative abundance of cyanobacteria colonies (net sampler) and their concentration (VanDorn sampler). Species were identified using How To Know The Freshwater Algae, third edition, by G.W. Prescott, published by Pictured Key Nature Series.

Results

Net collections of phytoplankton from the upper 7 m or so of the water column were made during the September 6 and October 4 sampling dates. These net samples were analyzed to determine the composition of the cyanobacteria phytoplankton community. There were other types of phytoplankton present that we did not quantify. It is worth noting that the two dominant eukaryotic algal forms were dinoflagellates, almost exclusively in the genus *Ceratium*, (**Figure 1**, top left*)* and diatoms, dominated by *Fragillaria sp*. We identified cyanobacteria to the genus level and found that the dominant species were in the genera *Nodulari*a, *Microcystsis, Anabaena*, *Oscillatoria* and *Waronechinia,* shown in **Figure 1**. We did not attempt to determine cell numbers.

The seasonal pattern of phytoplankton biomass consisted of a period with minor blooms in late June, followed by a surface clear period in midsummer and then heavy blooms in late summer and fall. This trend is evident in the *in vivo* chlorophyll values, which show a minor surface spike in late June. By mid July surface waters were relatively clear as indicated by a Secchi depth of 3m and low *in vivo* chlorophyll fluorescence, but there was a deeper-water bloom just above the thermocline in July that was probably unrelated to colonial cyanobacteria, which tend to float (**Figure 2**). In contrast, the *in vivo* chlorophyll values from September 6 and October 4 were higher and indicative of a phytoplankton bloom (**Figure 2**).

Measurements of transparency (Secchi depth) and turbidity are consistent with the observations based on *in vivo* chlorophyll values. The shallowest observed Secchi depth was 1.15 m on June 28 (**Table 1**), while the deepest was 3.0 m on July 19. This July high surface Secchi disk transparency is a bit misleading because a relatively high biomass of phytoplankton had settled to about 8m, just above the thermocline (Figure 2, July 19 profile). This settlement phenomenon is not unusual during periods of calm weather. After July 19, the Secchi depth seemed to decrease and it was lowest at 1.2 m on October 4. Turbidity values at 6 m increased from June 20 to October 4, September and October having turbidities that were more than twice those observed in June and July. The extracted chlorophyll *a* concentrations also indicate the occurrence of blooms. In September, chlorophyll a concentrations were on average 20.93 µg/L at the surface, while on Oct. 4 surface chlorophyll *a* concentration was 30.56 µg/L.

We collected phytoplankton to determine the species composition in the late season blooms. Analysis showed the bloom in September was dominated by species of *Anabaena, Oscillatoria*, *Microcystis*, and *Nodularia*. Species of *Anabaena* and *Nodularia* are nitrogen-fixers, meaning they can change molecular nitrogen (N_2) into forms that are usable by other organisms. *Microcystis*, however, cannot fix its own nitrogen, so it must rely on water column supplies to meet its nitrogen requirements. It is interesting to note that the September bloom had high proportions of *Anabaena* and *Nodularia* (both nitrogen-fixers) and moderate amounts of *Microcystis*. *Microcystis* dominated the October bloom, possibly after inorganic nitrogen had been released into the water by nitrogen-fixing species.

Van Dorn water bottle samples were used to quantify the numerical abundance of colonies. On September 6, we observed a total count of 55 colonies/mL at the surface, and 99 colonies/mL at 3 m. On October 4, the total number of colonies observed at the surface was

514/mL, and 504/mL at 3m. As shown in **Figure 3,** on September 6, species of *Anabaena* were most abundant when concentrations at surface and 3 m are combined. *Microcystis* and *Nodularia* were slightly lower in total abundance. On October 4, *Microcystis* colonies were the most abundant, with *Oscillatoria* and *Nodularia* slightly less abundant while *Anabaena* concentrations were relatively low (**Figure 3**). *Microcystis* was found at extremely high concentrations on this date; at the surface there were more than 250 colonies per mL, and at 3 m, *Microcystis* was observed at about 240 colonies per mL.

Discussion

Cyanobacteria colonies were present in Silver Lake throughout the growing season. Blooms were detected in June and September with very dense concentrations present in early October. We documented the late season blooms with collections on September 6 and October 4 by showing increased levels of *in vivo* chlorophyll, and high chlorophyll *a* concentrations, shallow Secchi depths and high turbidities as shown in **Table 1**. Although the peak cyanobacterial biomass occurred in September and October, our data along with visual observations indicate that mid-late June was also a bloom period, although it was less extensive than the late summer/fall event. Between the June and Sept-Oct bloom periods we observed a midsummer clear phase in Silver Lake, with good water clarity and moderate cyanobacterial abundance.

Seasonal patterns of phytoplankton community composition frequently show that cyanobacteria are more prominent in late summer months, specifically July to October (Pickney et al., 1998). Factors such as grazing pressure, light, nutrients, and competition can influence what groups of phytoplankton dominate an aquatic ecosystem at any given time. The main component limiting cyanobacteria growth during the summer is nutrient availability, and blooms can be triggered when additional nutrients are brought into the system by runoff or internal loading. One focus of this study was to determine if the summer blooms in Silver Lake were somehow related to pulses of internally loaded P being driven to the surface by wind events. In this context it is interesting that blooms should occur early and late in the season when stratification was less stable, and that the clear phase should come in July when the thermocline was more pronounced and the water column most stable (See Section 1 on Internal Loading).

Mixing, and the delivery of internally loaded P, was more likely in June when surface waters were still warming and in late summer and fall when seasonal cooling had begun.

Colonial cyanobacteria like those in the genera *Microcystis* and *Anabaena* have been implicated as frequent causes of lake HABs worldwide. During HABs, cyanobacteria biomass is much higher than normal and the water can turn a bluish-green color due to the dominant b-g phycocyanin pigment in cyanobacteria. The aesthetic quality of lake water declines and there is increased risk of animal/human health issues as a result of contact or consumption. According to a recent study conducted at the Greater Sudbury Lakes in Canada, researchers observed significant changes in appearance and composition of phytoplankton between the months of July and September (Evans et al, 2015). The study reports that the dominant genera jncluded *Microcystis, Anabaena, Aphanizomenon, Oscillatoria, Nostoc*, *Borzia, Planktothrix, Synechocystis* and *Leptolingbia*. Of these species, *Microcystis, Anabaena*, *Aphanizomenon*, and *Oscillatoria* often form toxic blooms.

The most abundant species in Silver Lake during summer 2018 were in the genera *Microcystis*, *Oscillatoria*, *Anabaena, Coelosphaerium* and *Nodularia* for both September 6 and October 4th. *Anabaena*, *Microcystis* and *Nodularia* species dominated the September bloom. *Anabaena* and *Nodularia* are nitrogen-fixing cyanobacteria, meaning they can change nitrogen into a form that is usable to other organisms. *Microcystis*, however, cannot fix its own nitrogen, so it must rely on nitrogen-fixers or on outside sources to meet its nitrogen requirements. It is interesting to note that the September collection had high concentrations of *Anabaena* and *Nodularia* (both nitrogen-fixers) and moderate concentrations of *Microcystis* while in the dense October bloom *Microcystis* was the dominant group. By September the lake had turned over and provided ample P for phytoplankton. It is possible that the nitrogen-fixing species present in early September had fixed nitrogen into forms available to be used by *Microcystis* as it achieved dominance in October.

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Tables and Figures

Table 1. Secchi depth, turbidity, and extracted chlorophyll a measurements for Silver Lake. Measurements taken on 9/6 and 10/4 were in the midst of a bloom. Turbidity is significantly higher at all depths during the blooms. Chlorophyll a concentrations were determined during bloom periods were very high.

Date	**6/20**	**6/28**	**7/19**	**9/6**	**10/4**
Secchi Depth (m)	2.80	1.15	3	2.60	1.22
Turbidity at 0m (NTU)	3.33	6.92	--	8.12	14.4
Turbidity at 3m (NTU)	4.39	7.68	--	5.66	8.88
Turbidity at 6m (NTU)	2.49	2.28	--	6.41	5.82
Chlorophyll *a* at 0m (µg/L)	--	--	--	20.93	30.56
Chlorophyll *a* at 3m (µg/L)	--	--	--	28.43	14.92
Chlorophyll *a* at 6m (µg/L)	--	--	--	39.62	11.98

Figure 1. Abundant phytoplankton in Silver Lake. On the top left is the dinoflagellate *Ceratium*, which was common in all samples. All the other panels show photos of cyanobacteria, starting in the top row with *Waronechinia,* species of *Anabaena, Nodularia and Aphanocapsa;* second row shows *Anabaena*, *Lyngbya* and *Microcystis*; third row *Anabaena* coils and filaments and large colonies of *Microcystis*.

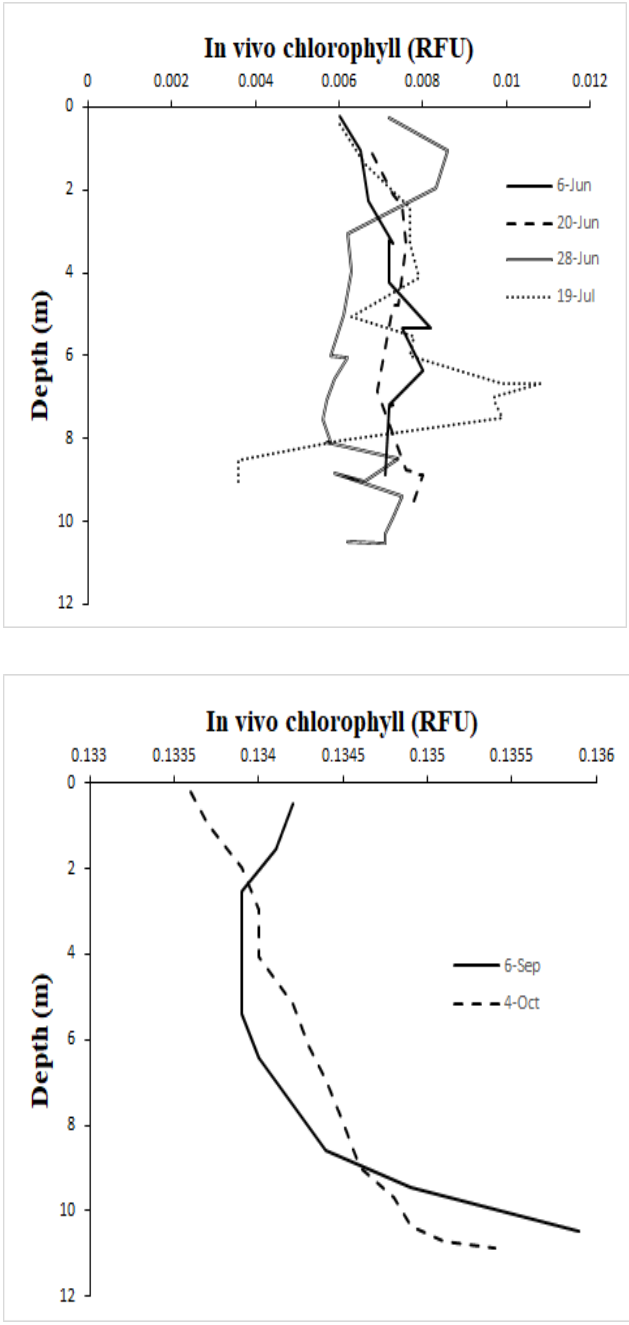
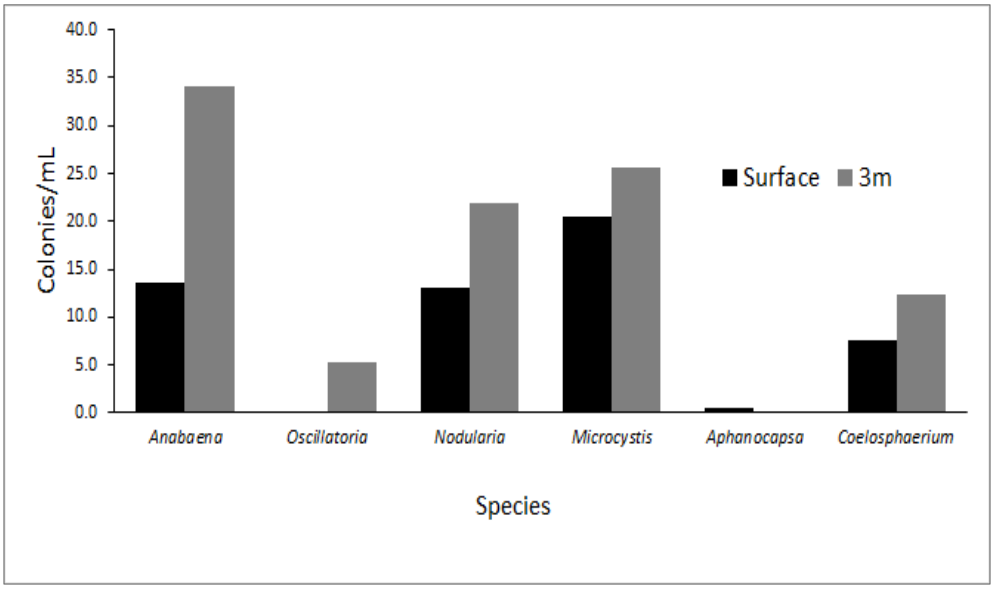


Figure 2. These are profiles of *in vivo* chlorophyll during periods of clear water (top) and during blooms (bottom). Please note that the chlorophyll (x-axis) scale for the clear water period graph is more than an order of magnitude lower than the chlorophyll scale for the bloom period graph. The increase in values below 8 m is due to temperature effects on the sensor readings.



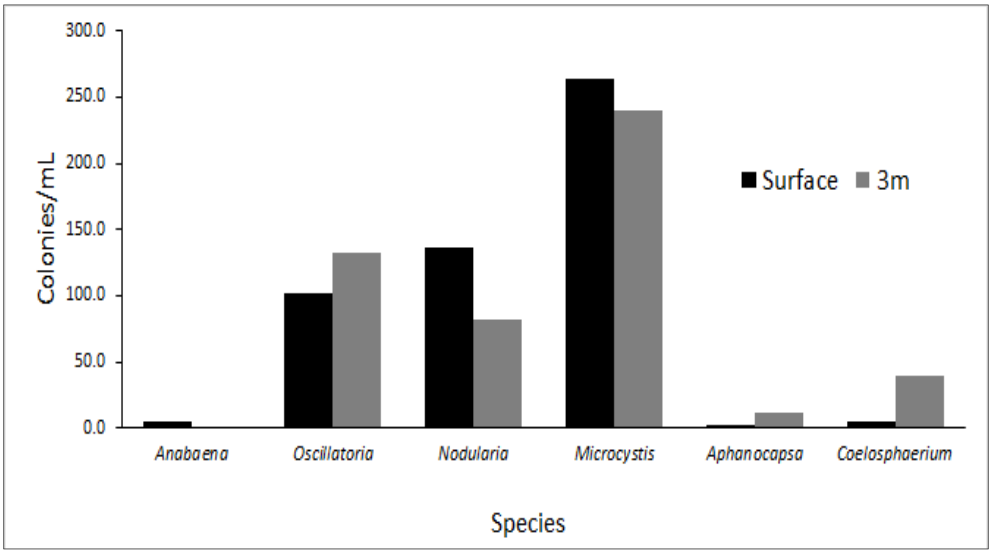


Figure 3. Bar charts showing colonies per mL for the most abundant cyanobacteria in the September collection (top) and the October collection (bottom). Colonies in the October bloom were nearly an order of magnitude denser than in September (Note the difference in the scale of the y axis between the two graphs). *Microcystis* is the dominant form in the October bloom.

Section III

Species Composition and Biomass of Crustacean Zooplankton

Introduction

Herbivorous zooplankton are the primary consumers of phytoplankton in freshwater lake communities. While rotifers dominate the zooplankton community numerically, crustaceans, by their larger size and greater filtering potential, have a more prominent role in regulating phytoplankton biomass, energy and nutrient distribution in the ecosystem (Chislock, 2012). The crustacean zooplankters, particularly larger individuals, are in turn a major food source for many smaller fish species and for the juvenile of larger species. They are thus a crucial link to the top predators in the food webs of lakes.

The dominant crustacean herbivores in temperate lakes are the water fleas (taxonomic Order Cladocera), particularly the families Daphnidae and Bosminidae. The genus *Daphnia* is comprised of the largest of the herbivorous zooplankton, ranging in size from 1mm-3mm. *Bosmina* are much smaller than *Daphnia* and are usually smaller than 0.5mm. Copepods also make up a large portion of the zooplankton community in lakes. Planktonic copepods are usually of two different types: calanoid copepods and cyclopoid copepods. They are usually between 0.5mm-1.2mm. Copepods can be omnivorous, herbivorous, and some larger species are carnivores of other zooplankton.

The water filtering and cell ingestion rates of herbivorous zooplankton have been studied extensively. The amount of water that an individual can filter in a given amount of time, known as the filtering rate (e.g. mL/individual/hour), is a function of animal size, which determines the filtration area of feeding appendages. An indirect way to estimate the filtration potential of an herbivorous crustacean is by a simple relationship where:

$$\text{Filtration rate (ml/individual/hr)} = 0.538 \times \text{length (mm)}^{1.55}$$

The food ingestion rate of a zooplankter is the filtration rate multiplied by the concentration of food cells in suspension. The largest herbivorous zooplankters such as many species of *Daphnia* are the most important filter feeders because they can process larger volumes of water, often in excess of 1.0 mL/hr. By their sheer size they can also feed on larger phytoplankton, and there is ample evidence that they are capable of ingesting colonial cyanobacteria.

There has been considerable interest in understanding the possible role of large herbivorous crustaceans as effective consumers of colonial cyanobacteria. The prevailing view among lake biologists has been that *Daphnia* and like species would not consume much cyanobacteria because of the large size of the colonies and the toxins some produce. Recent work has shown that *Daphnia* will eat cyanobacteria such as the filamentous *Anabaena* (Gilbert 1990). Feeding on the highly toxic *Microcystis* tends to slow down, but not stop filtration rates (Chislock 2014). This new perspective has sparked research into determining whether *Daphnia* and other crustacean zooplankton regulate the frequency and intensity of harmful algal blooms in lakes. It may even be possible that by eating the less toxic cyanobacteria species, zooplankton feeding may provide an advantage to toxic forms such as *Microcystis*.

Silver Lake has been subject to cyanobacterial blooms since at least 1991, when F.X. Brown Associates (1991) reported that *Anabaena* and *Aphanizomenon* were dominant in the summer phytoplankton community. Makarewicz and colleagues (2005) also reported mid lake blooms and dense shoreline accumulations of cyanobacteria. The blooms have been less severe than one might expect, given the high levels of P in the lake **(See section I, this report)**. We report here that compared to nearby lakes, Silver Lake has larger zooplankton that have a large filtration potential. It may be possible that the Silver Lake zooplankton have been feeding extensively on cyanobacteria, limiting the intensity of blooms. Consequently, the health of the zooplankton community is an important factor to consider in controlling HABs in Silver Lake.

Methods

To determine the species composition and length of zooplankton, samples were collected from the surface of Silver Lake on July 19, and September 5, 2018 by towing an 80-micron porosity plankton net vertically in the top 7 meters of the water column. The net samples were concentrated initially in 10% ethanol solution to anesthetize the animals, then gradually concentrated into 70% ethanol final solution for long term preservation. We determined the relative abundance of each species from the net samples by identifying at least 50 individuals from each collection. To determine the numerical abundance of species, samples were collected at 3 m and 6 m from Silver Lake on September 5, and October 4, 2018 L using a 2.2 L Van Dorn bottle. All Zooplankton samples were examined using an OLYMPUS CX31 microscope at 100x and 400x magnification. Specimens were identified to genus or when possible to species using

taxonomic keys in the guide *Zooplankton of the Great Lakes: A Guide to the Identification and Ecology of Common Crustaceans*, (Koda et al., 1984) as well as, *An Image Based Key to the Zooplankton of North America*, (Haney, 2013). We measured individuals to the nearest 0.01 mm using a calibrated ocular micrometer.

To analyze Van Dorn discrete volume collections (2.2 L) , fixed samples were mixed thoroughly and a 750µL subsample of the 125mL sample was pipetted onto a slide. The entire slide was scanned for organisms and a count for each species present was taken. Multiple subsamples were analyzed. These data could then be extrapolated to the whole 125mL preserved collection and then to the total volume in the Van Dorn sample from which they were concentrated. The biomass of each species was calculated using first the following relationship to calculate individual biomass: $\ln(W) = \ln(\alpha) + \beta * \ln(L)$ (W is the dry weight, L is the average length of individuals in millimeters, and α and β are species-specific coefficients as determined by Watkins et al., 2011). We also calculated filtration rates for each species by determining species -specific filtration rate using the formula: $F = 0.538 * L^{1.55}$ (L is the average length of individuals in millimeters). The potential filtration rate (mL/day) of each species was estimated by multiplying the average filtration rate per individual by the average abundance of species in the water column. Individual weight and filtration rate were multiplied times abundance to calculate a total rate for each species.

Results

We recorded nine common taxa of crustacean zooplankton in Silver Lake, most of which were identified to species. There were at least seven types of Cladocerans including larger species such as *Daphnia galeata mendotae* and *Daphnia pulicaria* **(Figure 1)**. The dominant copepods were the calanoid *Skistodiaptomus oregonensis* and the cyclopoid *Acanthocyclops vernalis* **(Figure 1).** Both dominant copepod species are reportedly omnivorous.

In the September collection the total numerical abundance of crustacean zooplankton was 70,841 ind./m^3. Among the larger species, *Skistodiaptomus oregonensis* had the highest numerical abundance with 15,646 ind./m^3 followed by *Daphnia retrocurva* (12,922 ind./m^3) and *Ceriodaphnia* (12,237 ind./m^3). Abundances of *D. g. mendotae* at 9,217 ind./m^3, *D. pulicaria* at 6,126 ind./m^3, and *Acanthocyclops vernalis* at 3,155 ind./m^3 were also high **(Table 1, Figure 2).**

These taxa were also present in the October collection, in which *Bosmina* and *Chidorus* were most abundant, reaching densities of 24,721 and 40,216 ind./m^3.

The average length of individuals in each species was used to calculate average biomass. The results for each taxon-specific biomass shown in **Table 1** are very close to biomass values reported by Dumont *et al.* (1975). These values were multiplied times the abundance of each group and summed to obtain an estimate of the total biomass of crustaceans, which was 514 mg/m^3 **(Table 1)**. At 406 mg/m^3, the Cladocera made up 78.9% of the total adult crustacean biomass. Copepod nauplii larvae were excluded from this analysis.

The average length of all mature crustaceans in Silver Lake from a sample size of 335 was 1.46 mm **(Figure 3).** Among the copepods, the calanoid species *Skistodiaptomus oregonensis* had an average length of 0.94 mm and *Acanthocyclops vernalis* had an average length of 0.89 mm **(Table 1).** The smallest Cladocera present was *Bosmina longirostris* with an average length of 0.28 mm. The Silver Lake zooplankters have the largest average length of several lakes in the region that have been studied by SUNY Geneseo **(Figure 3)**. Zooplankton in Silver Lake were roughly 1.4 times larger than zooplankton in Loon lake, which had the next largest zooplankton average size, and 5.5 times larger than the average zooplankton size in Conesus Lake. The dominant cladoceran in Conesus Lake is *B. longirostris.* The *Daphnia* (primarily *D. retrocurva*) are smaller and few in number.

The potential filtration of cladocerans in Silver Lake was calculated to be about 76% of the water column per day **(Table 2).** *Daphnia pulicaria* had the highest filtration rate with 31 mL/ind/day and given their abundance could clear 19% of the water column per day. *D. g. mendotae* had a filtration rate of 22 mL/ind./day and given their abundance would clear 20% of the water column per day.

Discussion

Silver Lake has experienced cyanobacterial blooms since at least 1991 (FX Browne Inc., 1991) and these blooms continue to be a major problem today (CSLAP Reports). Despite the preponderance of some potentially toxic phytoplankton species such as *Microcystis aeruginosa*, Silver Lake supports a thriving herbivorous zooplankton community. The large biomass and high estimated filtration rate of this community has the potential to curtail the intensity of dense phytoplankton blooms. As is evident from **(Figure 2)**, Silver Lake is dominated by much larger

zooplankton than other lakes in the surrounding area, such as Conesus and Hemlock Lakes. Looking at species compositions and sizes, it can be seen that much of the biomass of the zooplankton community was composed of *Daphnia galeata mendotae* and *D. pulicaria*, (average sizes of 1.40 mm and 1.77 mm, respectively), two large *Daphnia* species that are absent in some of the other nearby lakes, such as Conesus Lake and Hemlock Lake **(Figure 2)**.

Our results indicated that the Silver Lake zooplankton community has the potential to achieve a very high filtration rate. The Cladocerans have the capacity to clear about 75% of the water column daily **(Table 2)**. The large *Daphnia* are responsible for the high potential filtration rate. By comparison, the Cladocera in Loon Lake had a potential filtration rate of 24% (Bosch et al. 2015) and the Cladocera in Hemlock Lake had a potential filtration rate of 0.81% (Bosch et al. 2018). The fact that Silver Lake has an abundance of large zooplankton, leading to a high filtration rate, could be a major factor contributing to the control of the phytoplankton biomass, as well as the frequency and intensity of HABs in the lake.

Even with the very high filtration rate by herbivorous zooplankton, Silver Lake experiences extensive HABs (CSLAP 2017) in great part to the high release of internal (legacy) P from the sediments **(See section 1, this report)**. Therefore, it is imperative that the Silver Lake zooplankton community be safeguarded. In nearby Conesus Lake the introduction of alewife decimated populations of larger *Daphnia*, leaving behind a community with an average size of 0.47mm. Loss of the larger species lowered the potential filtration capacity of the zooplankton and increased the frequency and intensity of algal blooms (Makarewicz 2000).

The introduction of a new planktivorous fish into Silver Lake would greatly impact the populations of all three dominant *Daphnia* species, lowering the potential filtration rate by as much as 54%. The removal of such a powerful herbivorous control would likely increase the severity and frequency of the harmful algal blooms. Another measure that should be taken to ensure the prosperity of the zooplankton community is the protection of top predatory fish species that feed on zooplanktivorous “baitfish”. Therefore, it is critical that stocking and fishing regulations be established to sustain the healthy large fish populations that help maintain the health of the ecosystem.

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Tables and Figures

Table 1. The individual and total biomass of crustaceans calculated for the Sep 5 sample. Individual dry weight biomass was calculated using the formula $Ln(W) = Ln(\alpha) + \beta * Ln(L)$ (Watkins et al. 2011), where α and β are species-specific coefficients and L is the average length (mm) of the adult individuals in the population. Total biomass was calculated by multiplying the individual weight by the numerical abundance of individuals per cubic meter.

Species	Ln (α)	β	Length (mm)	Weight per Individual (µg)	Average Abundance (ind./m^3)	Total Biomass (mg/m^3)
Bosmina	3.08	3.04	0.28	0.43	3,155	1.36
Ceriodaphnia	2.562	3.34	0.87	8.10	12,237	99.09
Chydorus	4.493	3.93	0.33	1.08	381	0.41
Daphnia. g. mendotae	1.468	2.83	1.40	11.29	9,217	104.03
D. pulicaria	1.468	2.83	1.77	21.77	6,126	133.35
D. retrocurva	1.468	2.83	0.94	3.61	12,922	46.60
Diaphanosoma	1.624	3.05	0.79	2.49	8,582	21.35
Acanthocyclops	1.953	2.4	0.89	5.32	2,576	13.70
Skistodiaptomus	1.953	2.4	0.94	6.02	15,646	94.26

Table 2. Calculation of filtration rates and proportion of the water column filtered per day for Sep. 5. The filtration rate per hr was calculated using the formula $F = 0.538 * L^{1.55}$ where L is the average length (mm) of the adult individuals. The proportion of the water column cleared per day was calculated by multiplying the filtration rate (mL/ind./day) by the average abundance.

Species	Filtration Rate (mL/ind./hr)	Filtration Rate (mL/ind./day)	Average Abundance (ind./m^3)	Proportion of Water Column/ Day
Bosmina	0.07	1.68	3,155	0.006
Ceriodaphnia	0.43	10.32	12,237	0.127
Chydorus	0.09	2.16	381	0.001
Daphnia. g. mendotae	0.91	21.84	9,217	0.200
D. pulicaria	1.30	31.23	6,126	0.192
D. retrocurva	0.49	11.76	12,922	0.152
Diaphanosoma	0.37	8.88	8,582	0.077
Acanthocyclops	0.45	10.80	2,576	0.028
Skistodiaptomus	0.49	11.76	15,646	0.183

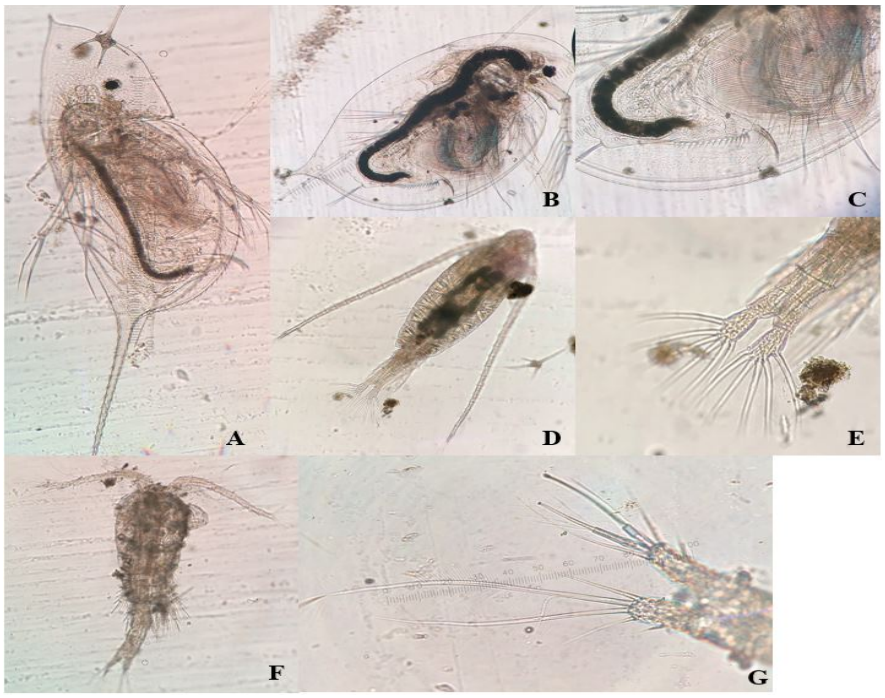

Figure 1. Abundant crustaceans in the Sep. 5 sample. Cladocerans include *Daphnia galeata mendotae* with average carapace length of 1.44 mm (A) and *D. pulicaria,* L=1.77 mm (B-C). Copepods include *Skistodiaptomus oregonensis*,(total length 0.94 mm) (D-E) and *Acanthocyclops vernialis,* length 0.89 mm (F-G).

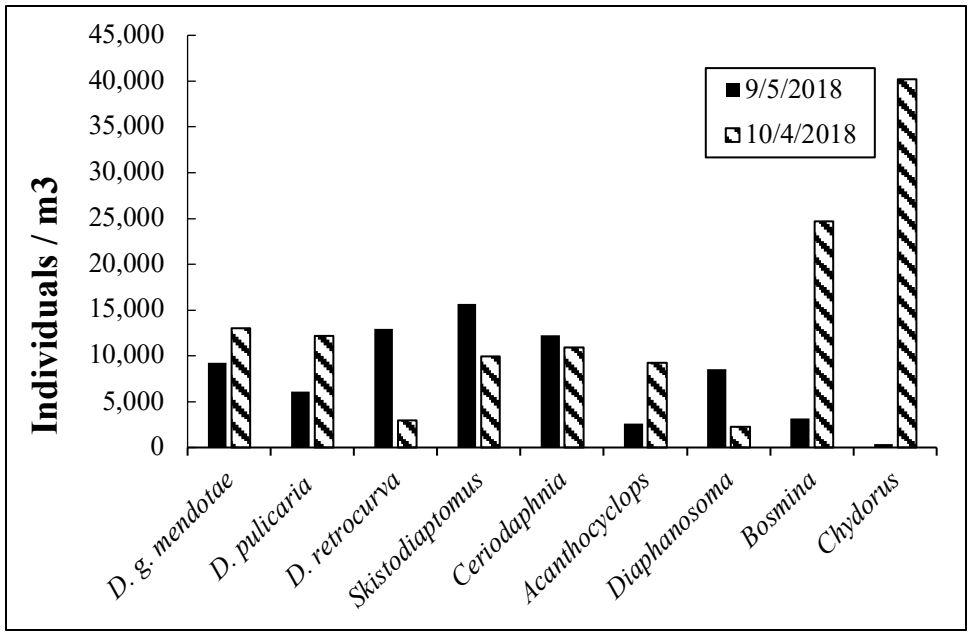

Figure 2. Comparison of the average numerical abundance of crustacean zooplankton in Silver Lake on Sep 5 and Oct. 4, based on Van Dorn bottle collections.

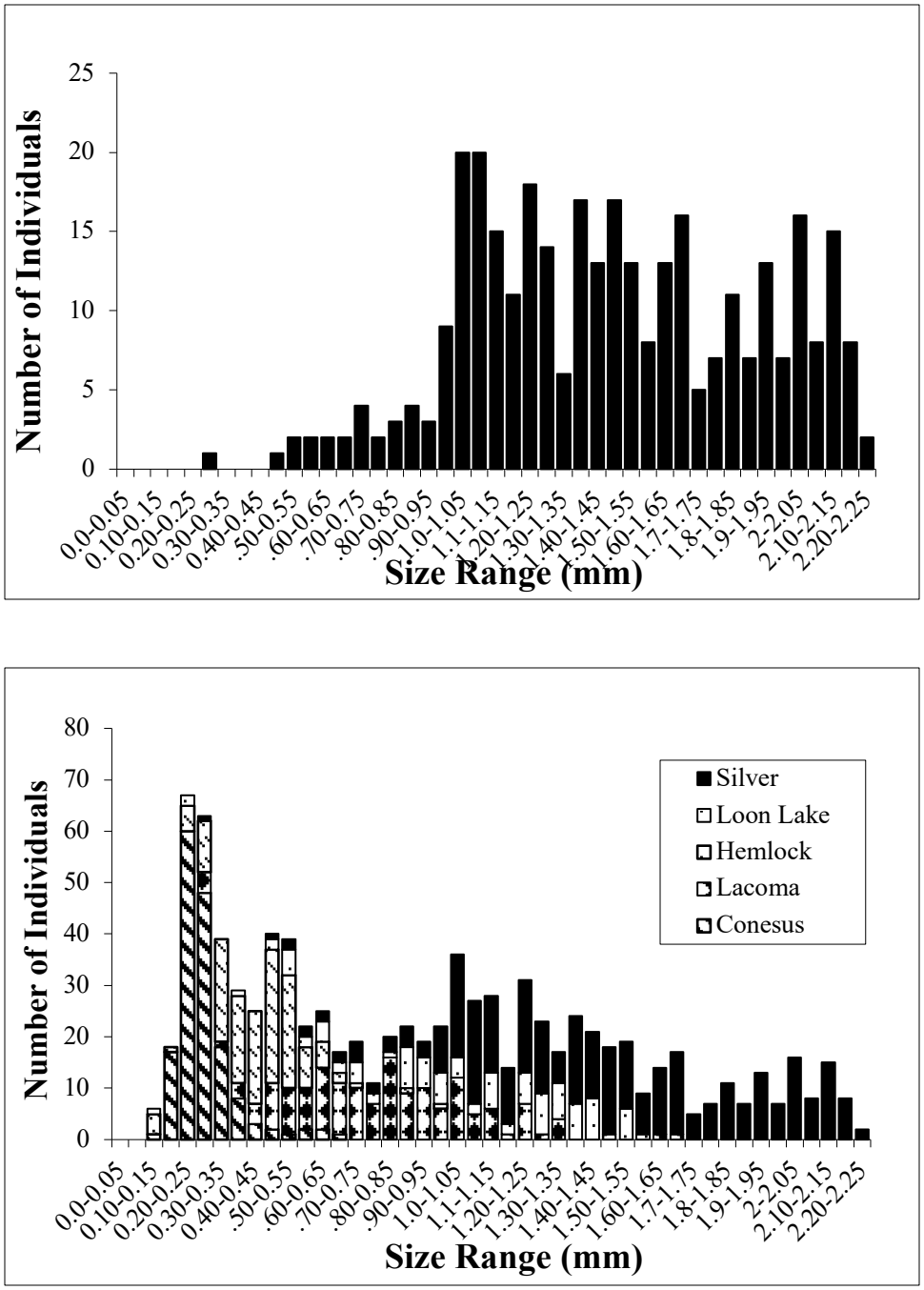

Figure 3. Length distribution of crustaceans captured with a vertical net tow. The top graph is for Silver Lake alone (Average Length: 1.46 mm). The bottom graph shows the Silver lake community in comparison to those of Loon Lake, Hemlock Lake, Lake Lacoma, and Conesus Lake. Copepod nauplius larvae were not included in this analysis.

Appendix. Hydrolab profiles taken in 2018

June 6, 3:45 PM

Depth (m)	Temp (°C)	pH (units)	ORP (mV)	SpCond (µS/cm)	PAR [µE/s/m²]	DO (% Sat)	DO (mg/l)	Chl a (V)
0.23	19.8	7.85	490	358	3528	97.5	7.44	0.006
1.03	19.8	7.93	486	358	1560	97.0	7.4	0.0065
1.05	19.8	7.95	485	358	2017	96.9	7.4	0.0065
2.27	19.8	7.97	485	358	1153	96.7	7.38	0.0067
3.3	19.8	7.99	481	358	487	97.3	7.43	0.0073
3.22	19.8	8	481	358	466	97.3	7.43	0.0072
3.22	19.7	8.01	480	358	510	97.0	7.42	0.0072
4.23	19.7	8.02	477	358	361	96.9	7.41	0.0072
4.24	19.7	8.03	476	358	310	96.9	7.41	0.0072
5.32	19.7	8.03	474	358	177	96.7	7.41	0.0082
5.34	19.7	8.05	472	358	232	96.8	7.41	0.0075
5.34	19.7	8.05	472	358	201	96.8	7.41	0.0075
6.34	17.6	7.71	483	363	133	62.0	4.96	0.008
7.19	15.3	7.6	489	366	94	37.4	3.14	0.0072
7.2	15.3	7.58	489	365	93	37.0	3.1	0.0073
7.22	15.2	7.54	490	365	81	36.1	3.04	0.0072
8.88	12.6	7.43	494	371	43	12.1	1.07	0.0071

June 20, 2:44 PM

Depth (m)	Temp (°C)	pH (units)	ORP (mV)	SpCond (µS/cm)	DO (% Sat)	DO (mg/l)	Chl a (V)
1.0	22.96	8.25	434	353	121.9	8.74	0.0069
1.0	22.91	8.3	424	353	127.2	9.13	0.0074
2.0	22.82	8.28	423	353	127.4	9.17	0.0078
3.1	22.76	8.28	418	353	126.6	9.11	0.008
4.0	22.59	8.23	417	354	120.3	8.69	0.0075
5.1	21.97	8.12	418	356	110.2	8.05	0.0065
6.0	19.66	7.83	427	361	75.1	5.75	0.0067
7.0	18.53	7.67	432	363	54.9	4.3	0.0072
8.0	14.67	7.48	441	376	9.2	0.78	0.0098
9.0	12.35	7.42	408	382	4.7	0.42	0.0083
9.9	11.46	7.37	253	387	3.7	0.34	0.0276

July 19, 3:27 PM

Depth (m)	Temp (°C)	pH (units)	ORP (mV)	SpCond (µS/cm)	PAR [µE/s/m²]	DO (% Sat)	DO (mg/l)	Chl a (V)
0.43	26.6	8.48	325	324	1990	102.2	6.85	0.0060
1.38	26.4	8.48	322	325	1118	100.7	6.78	0.0066
2.42	25.7	8.43	321	325	618	99.6	6.78	0.0077
3.3	25.5	8.4	314	325	411	99.5	6.71	0.0077
4.13	25.4	8.32	309	325	337	97.3	6.66	0.0079
5.05	25.2	8.16	295	326	188	87.4	6.01	0.0079
5.54	23.0	7.48	268	342	138	28.6	2.05	0.0078
6	22.1	7.45	245	346	71	16.9	1.23	0.0077
6.67	21.0	7.42	222	347	61	6.6	0.49	0.0099
6.69	20.4	7.42	209	349	56	6.2	0.47	0.0098
6.99	19.5	7.41	177	352	50	4.6	0.35	0.0097
7.12	19.5	7.39	174	352	49	4.6	0.35	0.0097
7.51	18.7	7.38	136	354	41	2.7	0.23	0.0099
8.04	16.2	7.38	117	362	29	2.6	0.22	0.0060
8.05	16.2	7.37	114	362	28	2.6	0.21	0.0060
8.53	14.9	7.37	100	369	23	2.5	0.21	0.0036
9.15	13.6	7.33	38	391	19	2.5	0.21	0.0036
9.17	13.6	7.32	39	393	18	2.6	0.23	0.0036

September 6, 4:18PM

Depth (m)	Temp (°C)	pH (units)	ORP (mV)	SpCond (µS/cm)	PAR [µE/s/m²]	DO (% Sat)	DO (mg/l)
0.5	19.8	8.24	371	325	260	136.4	11.24
1.54	19.8	8.26	368	325	185	136.9	11.29
2.52	19.8	8.24	368	330	64	131.2	10.87
3.43	19.8	8.14	369	337	28	115.8	9.66
4.56	19.8	8.07	370	337	12	115.4	9.65
5.42	19.8	7.95	381	349	5	69.8	5.97
6.44	19.7	7.68	391	363	3	24.9	2.17
7.51	19.7	7.38	341	406	2	8	0.73
8.57	19.7	7.19	173	456	1	5.8	0.55
9.47	19.7	7	68	495	1	4.4	0.45
10.49	19.7	6.81	48	495	0	4.1	0.42

October 4, 3:20 PM

Depth (m)	Temp (°C)	pH (units)	PAR [µE/s/m²]	DO (% Sat)	DO (mg/l)	Chl a (V)
0.2	18.6	7.85	3528	97.5	7.44	0.0060
1.0	18.5	7.93	1560	97.0	7.4	0.0065
2.0	18.4	7.95	2017	96.9	7.4	0.0065
3.0	18.3	7.97	1153	96.7	7.38	0.0067
4.1	18.3	7.99	487	97.3	7.43	0.0073
5.2	18.3	8	466	97.3	7.43	0.0072
6.1	18.2	8.01	510	97.0	7.42	0.0072
6.9	18.2	8.02	361	96.9	7.41	0.0072
7.9	18.2	8.03	310	96.9	7.41	0.0072
9.0	18.1	8.03	177	96.7	7.41	0.0082
9.7	17.9	8.05	232	96.8	7.41	0.0075
10.4	17.7	8.05	201	96.8	7.41	0.0075
10.7	17.2	7.71	133	62.0	4.96	0.008
10.9	16.8	7.6	94	37.4	3.14	0.0072