LONE LAKE ALGAE MANAGEMENT PLAN

Ecology Grant Agreement WQAL-2019-WhIsCD-00009

Prepared for
Whidbey Island Conservation District

Prepared by
Herrera Environmental Consultants, Inc

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Prepared for
Whidbey Island Conservation District
PO Box 490
Coupeville, Washington 98239

Prepared by
Herrera Environmental Consultants, Inc.
2200 Sixth Avenue, Suite 1100
Seattle, Washington 98121
Telephone: 206-441-9080

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1. INTRODUCTION AND BACKGROUND

Lone Lake a small, shallow, and polymictic lake located on the south end of Whidbey Island (Figure 1) has been experiencing a decline in water quality and increasing frequency of toxic blooms in recent years. This has stimulated interest in improving water quality and restoring native vegetation and fish habitat in the lake. As part of their commitment to improve water quality in Lone Lake, and on behalf of a Lone Lake resident, the Whidbey Island Conservation District applied for and received a Freshwater Algae Program grant from the Washington Department of Ecology (Ecology). Project goals were to monitor the lake for a 1-year period, examine hydrologic and water quality conditions and prepare water and phosphorus budgets for the lake. The objective of the sampling program was to develop a quantitative understanding of the sources of nutrients to Lone Lake, primarily phosphorus, to inform a strategy for reducing the frequency and longevity of toxic algal blooms. This project builds on past studies of the lake to develop a strategy and management plan to reduce nutrient loading and thereby reduce the frequency and duration of toxic algae blooms, restore recreational use, and maintain a high-quality fishery (WICD 2019).

Lone lake has a mean depth of 11.1 feet (3.4 meters), maximum depth of 16.7 feet (5.1 meters), and a surface area of approximately 99 acres (Table 1). The 2,430-acre watershed is mostly forested (89 percent); only 2 percent is developed (Figure 2). Forty-four shoreline residents, on septic systems, are clustered on the east side of the lake. The shoreline at the southern end of the lake is high-quality farmland in pasture and hay. The Whidbey Island Conservation District (WICD) has worked with willing landowners to around the lake between 2008 and 2011 to implement best management practices for controlling animal waste (WICD 2019).

Inflow to the lake is from several small, seasonally intermittent streams and groundwater. Lake surface elevation varies up to 2.5 feet during the year (WICD 2019). A major contributor to the high lake stage in winter is the collapse of an old, wood, outlet pipe, which limits discharge Soils are primarily loamy sand and sandy loam with some localized gravel, so they are relatively well-drained. The single outlet flows from the south end of the lake to Useless Bay, in Puget Sound (see maps in WICD 2019).

The lake is highly productive and has an excellent reputation as a trophy trout fishery (WICD 2019). Between 1996 and 2003, however, the lake was invaded by the exotic submersed plant *Egeria densa*, which reduced lake access and fish habitat quality (WICD 2019). To address the aquatic weed problem, the Lone Lake Homeowners association, in cooperation with the Island County Noxious Weed Board and Ecology, developed an aquatic vegetation management plan that included use of herbicides and grass carp to control weeds (LLHA and ICNWCB 2005).

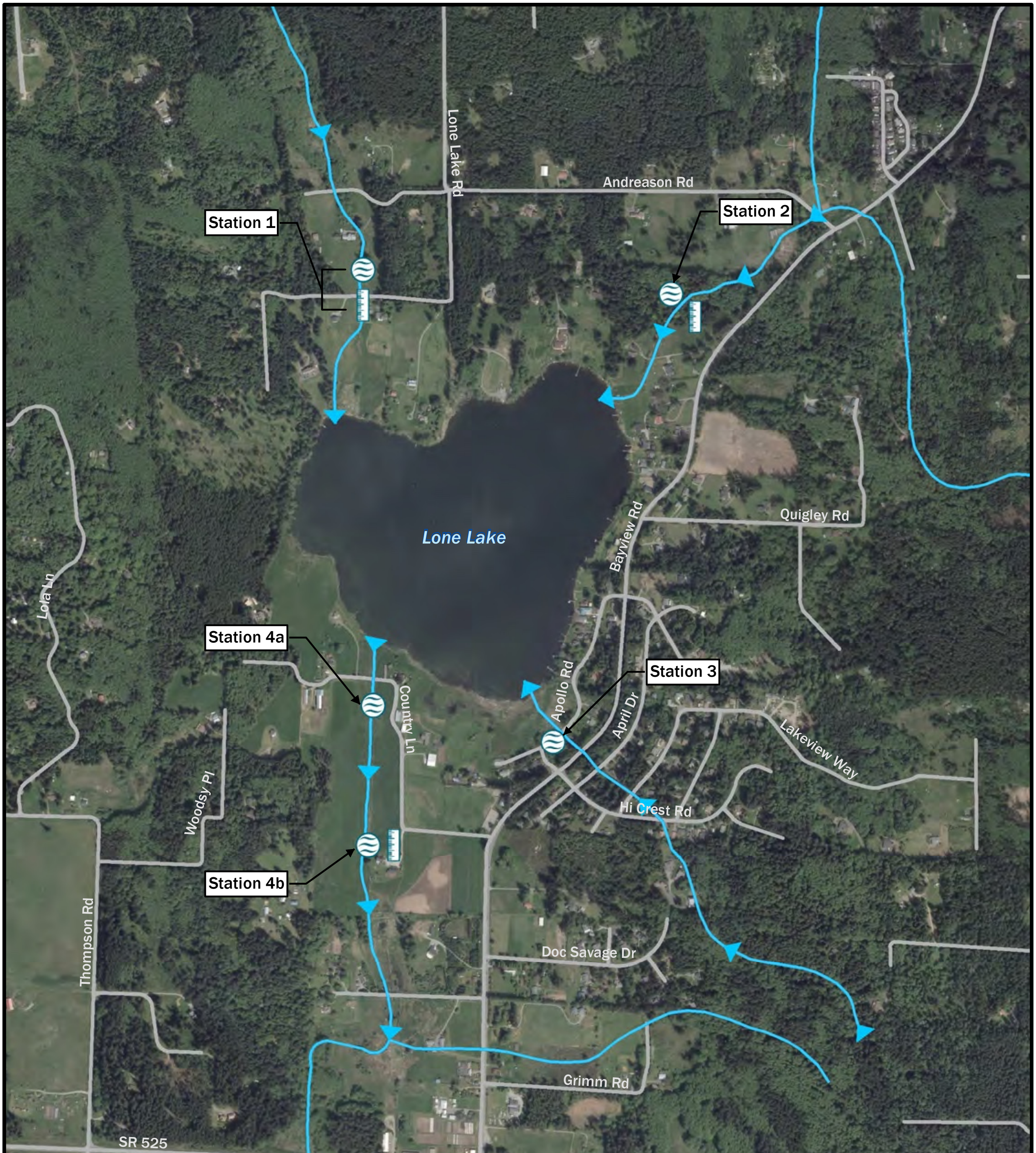

Figure 1.
Lone Lake Project Area.

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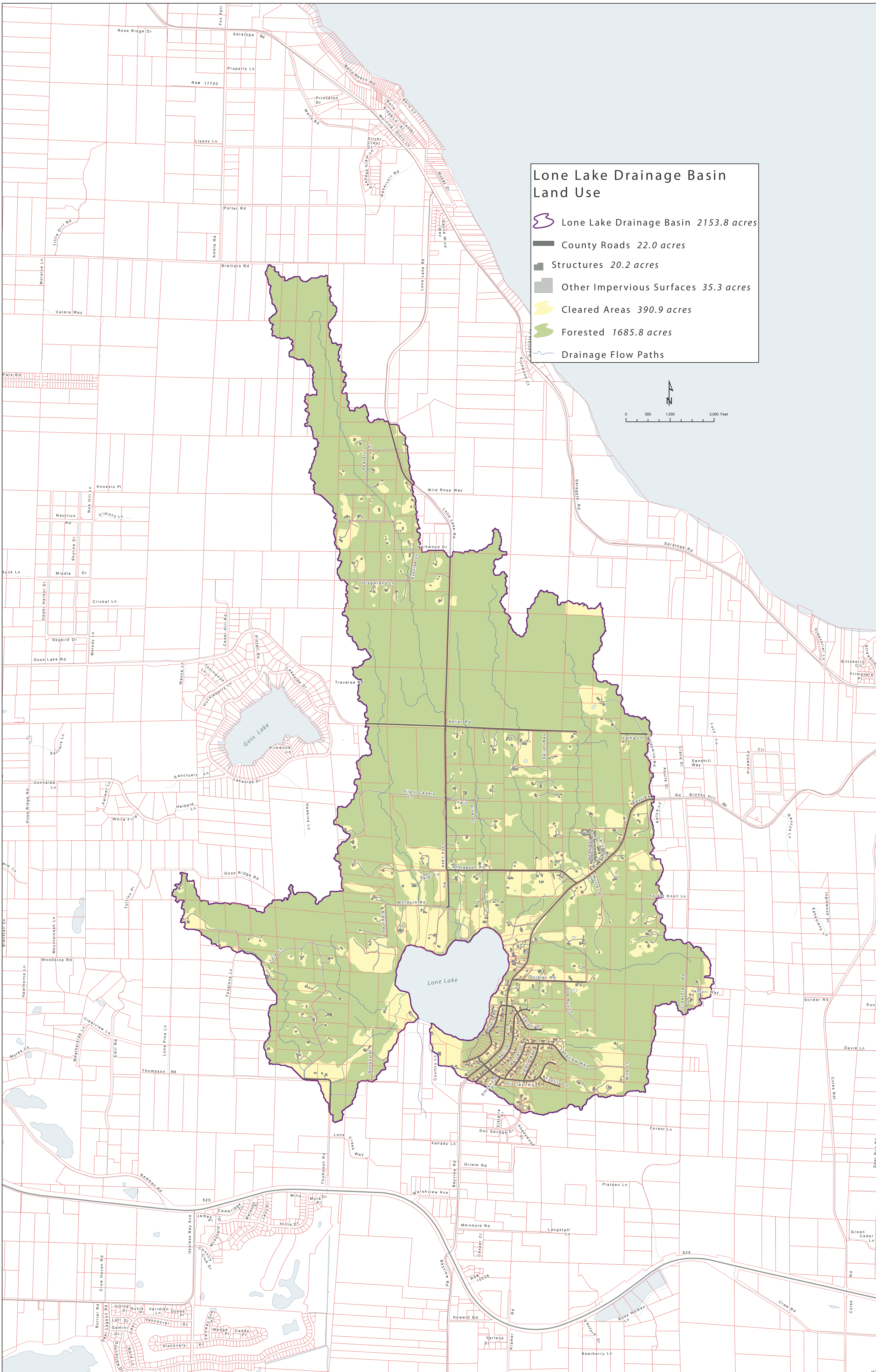


Lone Lake Drainage Basin
Land Use
Lone Lake Drainage Basin 2153.8 acres
County Roads 22.0 acres
Structures 20.2 acres
Other Impervious Surfaces 35.3 acres
Cleared Areas 390.9 acres
Forested 1685.8 acres
Drainage Flow Paths
N
0 500 1,000 2,000 Feet
Lone Lake
Goss Lake
Island County Public Works April 3, 2009

Table 1. Lone Lake Physical Data.	
Attribute	**Value**
Drainage Area	3.65 mi^2, 9.47 km^2, 2,340 acres, 947 hectares
Lake Area	99 acres, 40 hectares
Lake Volume	1,098 acre-feet, 1,354,406 m^3
Mean Depth	11.1 feet, 3.4 meters
Maximum Depth	16.7 feet, 5.1 meters
Inflow(s)	Multiple Intermittent
Outflow Channel	Single Intermittent
Trophic State Index from 2019–20	68
Trophic State Characterization	Eutrophic

Herbicide applications in 2005, 2006, and 2007 were followed by grass carp stocking in 2007. Those treatments resulted in eradication of submersed plants in the lake and a shift to a turbid, algae-dominated lake (WICD 2019). In 2016, the lake experienced a massive fish kill due to low dissolved oxygen concentrations caused by a crash of a dense algae bloom (WICD 2019). In 2017, although no fish kills were observed, dissolved oxygen concentrations of less than 2 milligrams per liter (mg/L) in surface water occurred on multiple days in mid- to late-August. The bottom-water dissolved oxygen concentration was below 2 mg/L for most of August (WICD 2019).

Toxic algae blooms have been detected in Lone Lake since algae toxin testing started in 2007 (WICD 2019). In 2017 and 2018, the lake was closed to recreational use because concentrations of two algal toxins (anatoxin-a and microcystin produced by cyanobacteria, also known as blue-green algae) exceeded state standards.

The elimination of aquatic vegetation by herbicide treatment and grass carp stocking caused substantial changes in Lone Lake chemistry. Total phosphorus (TP) in the lake averaged 0.06 mg/L in August and September from 1989 through 2007 (WICD 2019). In August and September 2008 and 2009, following the herbicide treatments and grass carp stocking, total phosphorus averaged 0.279 mg/L. Total nitrogen (TN) averaged 0.088 mg/L prior, and 2.07 mg/L after, chemical and grass carp treatments (WICD 2019). The total nitrogen to total phosphorus ratio (TN:TP) in the lake averaged 16.1 prior to treatment, and 7.3 after treatment, which suggests a shift to a nitrogen-limited state that benefits nitrogen-fixing, cyanobacteria (WICD 2019).

As a result of the degraded water quality apparently caused by the stocking of 808 grass carp in 2007, the Evergreen Fly Fishing Club and Whidbey Island Fly Fishing Club initiated extensive efforts to reduce the carp population in 2010 (J. Jacobson, Evergreen Fly Fishing Club, personal communication). The fishing clubs obtained a grass carp removal permit from the Washington Department of Fish and Wildlife and removed 41 grass carp by hook and line from 2011 through 2014, along with an additional 3 grass carp by seine netting in 2014. In 2013, they initiated a 3-year carp exclusion project showing recovery of the plant community within the exclusion pens while the rest of the lake remained predominantly barren. The fishing clubs then solicited help from bow fishers who proved to be much more effective than line or net fishing, but their efforts were limited by the poor visibility from the continued algal blooms. A total of 700 hours by volunteers removed a total of 60 carp, representing 7 percent of the total stocking amount. The grass carp population has also declined in the lake from natural death and predation by eagles and otters. Although no grass carp census or population estimate is available, native plants began to appear in the lake in the summer of 2017 (WICD 2019). In 2018, the native plant *Potamogeton praelongus* was common at depths less than 2 meters and fragments were commonly seen on the shoreline and floating at the water surface (WICD 2019). This suggests that the grass carp population has been greatly reduced because *P. praelongus* is a preferred species by grass carp.

Lone Lake is shallow and typically isothermal (same temperatures at top and bottom) and polymictic (vertically mixing). However, due to its hypereutrophic (excessive nutrient) condition it does experience periodic dissolved oxygen depletion, a concomitant decrease in pH, and an increase in specific conductance in the bottom of the water column (WICD 2019); in this case the bottom water can generally be described as the layer approximately 1 meter above the sediment. Dissolved oxygen depletion in the bottom of the water column is episodic but can occur for extended periods during the summer months, and likely results in a substantial release of phosphorus from the sediments that support summer phytoplankton production (WICD 2019).

Lake monitoring was carried out through the collaborative effort of a lake resident volunteer and the Island County Department of Natural Resources and was orchestrated by the Whidbey Island Conservation District (WICD). The lake monitoring was performed from March 2019 through February 2020 in accordance with the Quality Assurance Project Plan (WICD 2019). Herrera Environmental Consultants (Herrera) was contracted to use the monitoring results for preparation of this algae management plan.

2. LAKE WATER QUALITY

Historical data collected prior to this study were summarized in the QAPP and are included below for background information. This summary is followed by the methods and results of lake water quality monitoring conducted for the 2019–2020 study year.

2.1. Lone Lake Background Conditions

Lone Lake is shallow and typically isothermal, however, due to its hypereutrophic condition it does experience periodic dissolved oxygen depletion, a concomitant decrease in pH, and increase in specific conductance in the bottom 1 meter of the water column (Figure 3). Dissolved oxygen depletion in the bottom water is episodic but can occur for extended periods during the summer months (Figure 4), and likely results in substantial release of phosphorus from the sediments that support summer phytoplankton production. (Other mechanisms of sediment release, such as through disturbance of the sediments by bottom fish and wind-generated currents, may also occur.)

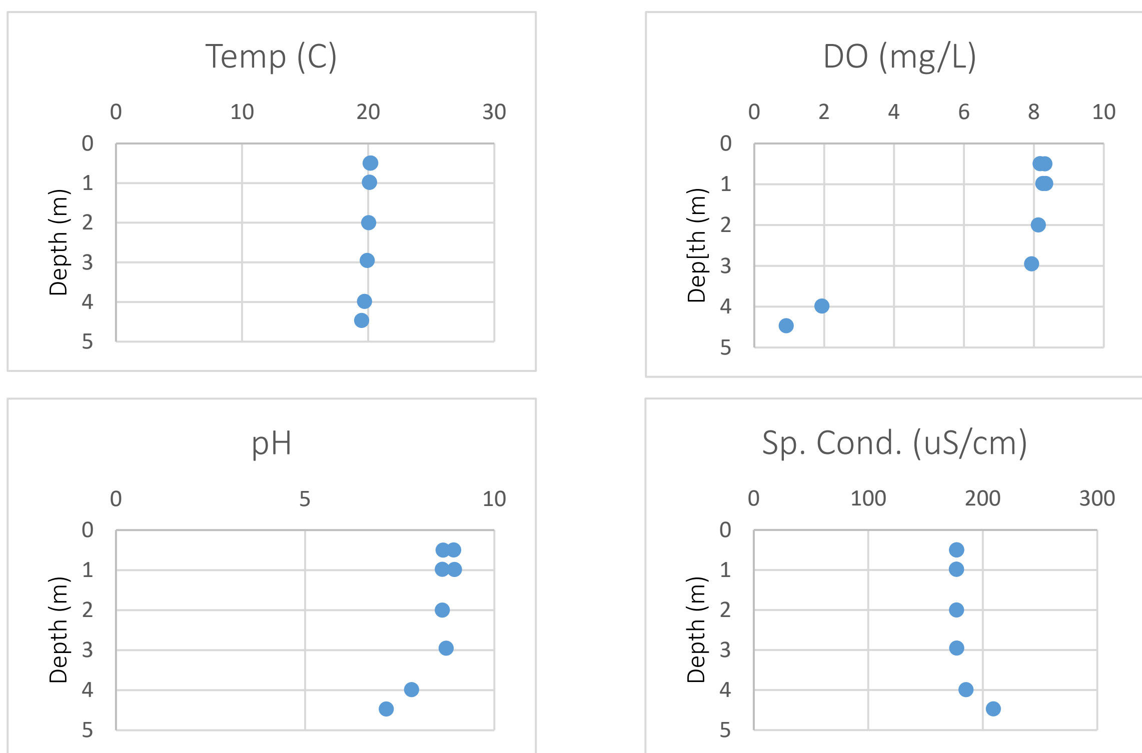


Figure 3. Profiles of Temperature, Dissolved Oxygen, pH, and Conductivity in Lone Lake on August 31, 2018.

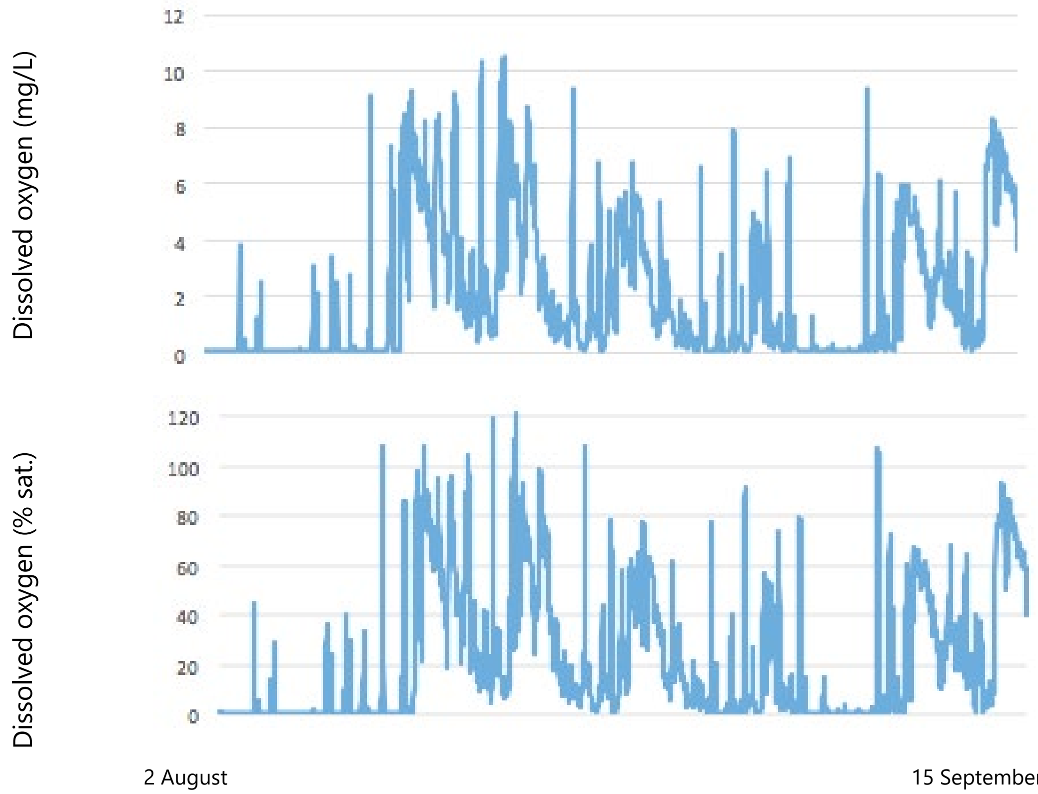


Figure 4. Dissolved Oxygen Concentration (top) and Percent Saturation (bottom) Approximately 0.5 Meter from the Bottom of Lone Lake from August 2 Through September 15, 2017.

Lake surface elevation varies up to 2.5 feet during the year (Figure 5). A major contributor to the high lake stage in winter is the collapse of an old, wood, outlet pipe, which limits discharge at the outlet. The high winter water levels may be responsible for the death of many mature trees that are noticeable on the shoreline.

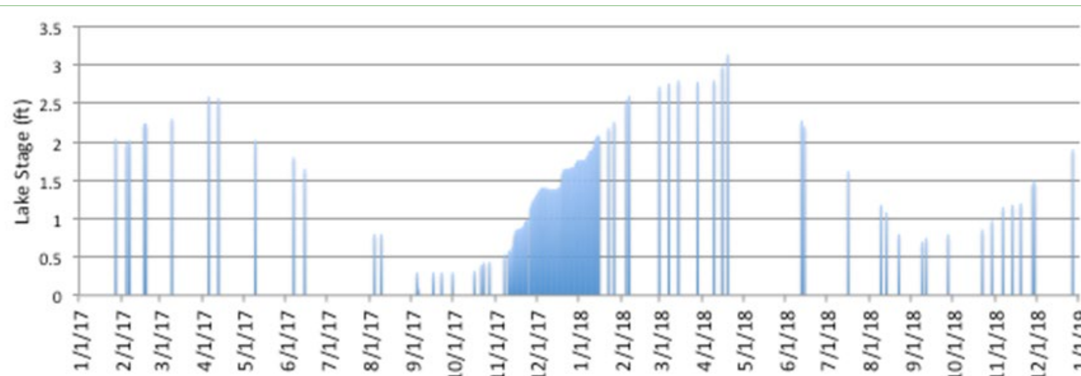


Figure 5. Lake Stage Measured at Boat Ramp (zero stage corresponds with elevation of outlet pipe).

2.2. Water Quality Study Methods

As part of this study, water quality parameters in Lone Lake were monitored during 1 calendar year from March 2019 through February 2020, to evaluate the amount of phosphorus, algae, and other important constituents in the lake in accordance with the QAPP (WICD 2019). In-lake monitoring was performed at one mid-lake station and occurred twice per month from May through October and monthly in other months. A depth-integrated sample was collected from 0.5, 2, and, 4 meter depths following procedures described in the QAPP. One grab sample was collected from the hypolimnion during the one event when dissolved oxygen was below 2 mg/L in the bottom waters. These samples were analyzed for chlorophyll a, total phosphorus, soluble reactive phosphorus, total nitrogen, ammonia nitration, and nitrate+nitrite nitrogen. Phytoplankton identification was performed on the depth-integrated sample. In-lake measurements also included collecting profile data for temperature, pH, dissolved oxygen and conductivity. Secchi depth was also measured.

All water quality samples were analyzed by Edge Analytical, which is accredited by the Washington State Department of Ecology (Ecology). A minimum of 10 percent of all field samples were collected in duplicate and a minimum of 10 percent of all laboratory analyses were measured in duplicate as part of the laboratory's quality control procedure.

The two major inflows (Stations 1 and 2; see Figure 1) to the lake were also sampled following the same schedule as the in-lake monitoring, when flow was present. Inflow measurements included discharge, total phosphorus, temperature, pH, dissolved oxygen and conductivity. Two of the sampling events were identified as storm events. Only flow measurements were made at the lake outlet as planned. The third inflow stream (Station 3) was not sampled as planned because a road culvert had been crushed and no water was observed flowing into the lake during any of the monitoring events.

2.3. Water Quality Monitoring Results

Water quality data from the 2019–2020 monitoring program are summarized below. Detailed data are included as Appendix A.

2.3.1. Temperature

Temperature profiles are presented for the one mid-lake station in Lone Lake in Figure 6. As shown, the temperature was fairly constant from top to bottom throughout the study year. By May the water was above 15 degrees Celsius (°C) and was 20 to 25 °C from June through September. To put these temperatures in context, Washington State's water quality criteria is set at 16°C for salmonid habitat (which includes trout), and 20 °C for warm-water fish (which includes fish such as bass, crappie, and perch) habitat (Washington State Surface Water Quality Standards, WAC 173-201A). The criteria are provided to put the temperature values into context with preferred fish habitat. The criteria also recognize that some water bodies may naturally exceed these conditions, which is likely the case in Lone Lake. The fact that the lake supported a trophy fishery in the past is evidence that fish can adapt and even thrive in warmer water temperatures when other environmental conditions such as clear water and a good food supply are in their favor. However, trout are not able to tolerate extremely low concentrations of dissolved oxygen as noted below.

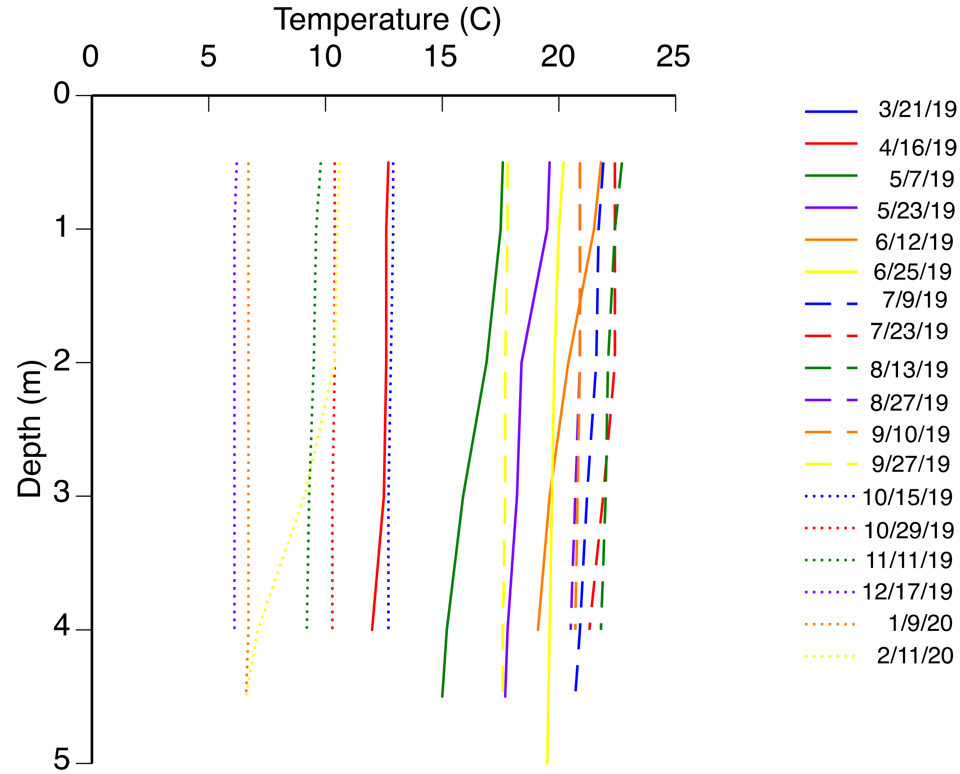


Figure 6. Temperature Profiles in Lone Lake 2019–2020.

2.3.2. Dissolved Oxygen

Dissolved oxygen is another important water quality parameter for salmonids and other aquatic organisms. Low dissolved oxygen levels can be harmful to larval life stages and respiration of juveniles and adults; therefore, it directly affects the survival of aquatic organisms. Depletion of oxygen in water bodies can also lead to a shift in the composition of the aquatic community.

Dissolved oxygen profiles are presented in Figure 7. Generally, dissolved oxygen concentrations ranged from 8 to 12 mg/L in the top 3 meters of the water column, but below this depth concentrations were below 6 mg/L for most of the summer. During the winter months of December through February, oxygen was high and constant throughout the water column. Dissolved oxygen is critical to the health of many aquatic organisms. For perspective, Washington State's water quality criteria is set at 9.5 mg/L of dissolved oxygen for salmonids (such as stocked rainbow trout) and 6.5 mg/L for warm-water fish (such as bass, crappie, and perch) (Washington State Surface Water Quality Standards, WAC 173-201A). Using these standards as a guide, the top 3 or 4 meters of water would have had adequate dissolved oxygen to support healthy fish populations.

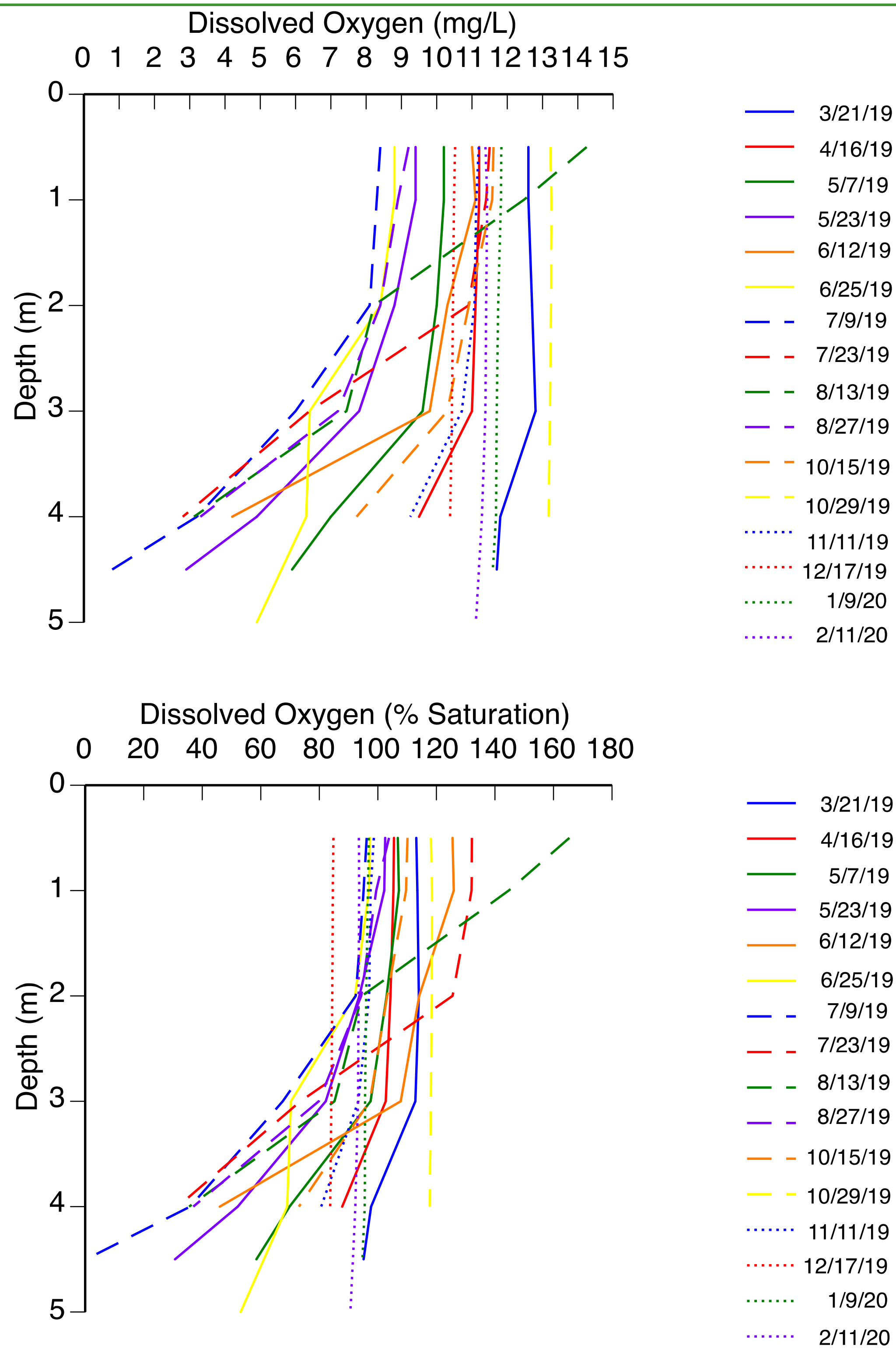
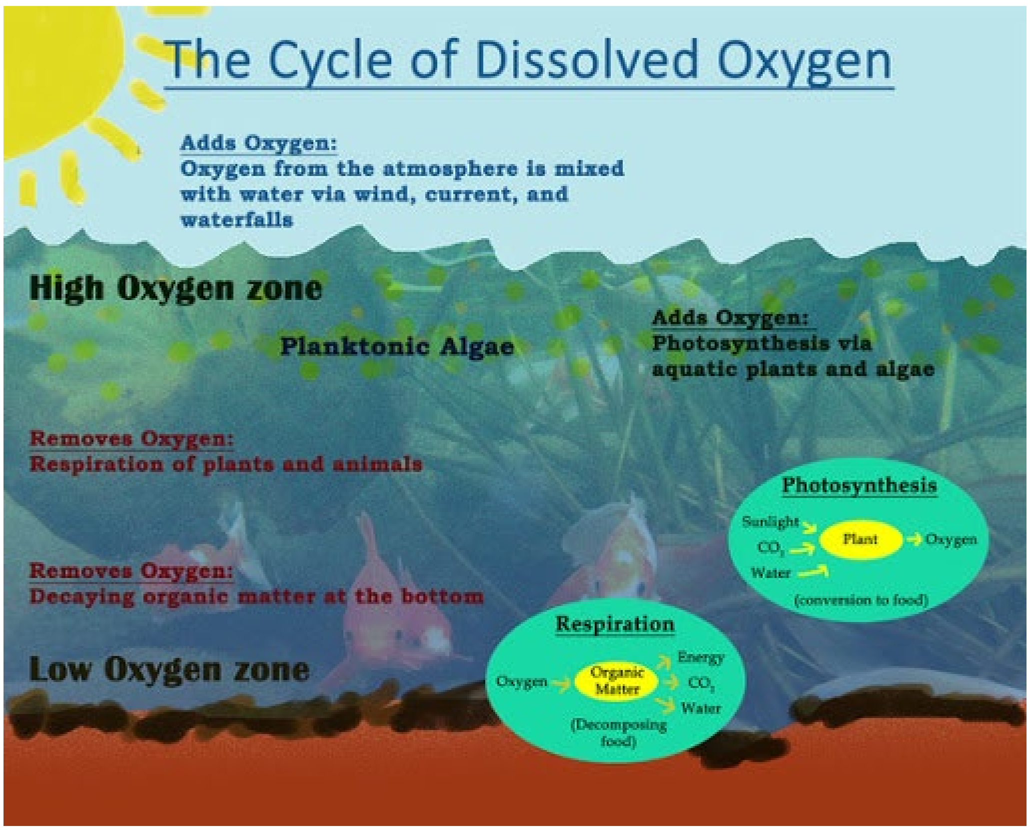


Figure 7. Dissolved Oxygen Concentration and Percent Saturation Profiles in Lone Lake 2019–2020.

The low dissolved oxygen concentrations in the bottom waters would also have contributed to release of phosphorus from the sediments. The fact that dissolved oxygen concentrations were higher in the upper portion of the water column but low nearer the sediments suggests is due to higher algae growth (an oxygen generating process) in the upper portion of the water column (i.e., in photic zone) and higher respiration and decomposition (oxygen depleting processes) near the sediments (i.e., below photic zone). Thus, although the lake is isothermal, vertical currents are not sufficient to overcome differences in photosynthesis and respiration with depth. This is exhibited in the graphic below and supported by the observed vertical differences in pH (discussed in following section).



There are frequent periods during the summer and into the fall when the lake is super-saturated (defined as levels about 110 percent saturation) with oxygen, which is harmful for aquatic life (Figure 7). During August, when dissolved oxygen concentrations were 12.5 to 14.3 mg/L in the upper 2 meters and the water was warm, the saturation level was 140 to 165 percent. These periods of super saturation would be expected to occur in conjunction with algal blooms.

2.3.3. pH and Conductivity

pH is a measure of the hydrogen ion activity in water, which can have a direct effect on aquatic organisms or an indirect effect since the toxicity of various common pollutants are markedly affected by changes in pH. Waters that have pH levels ranging from 0 to 7 are considered acidic, while waters with pH levels ranging from 7 to 14 are considered alkaline. Waters that have a pH of approximately 7 are considered neutral.

The pH profiles are presented in Figure 8. The pH generally ranged from approximately 7.5 to over 9.0 in the upper portion of the water column throughout most of the summer, generally slowly decreasing with depth in the water column. Washington State's water quality criteria sets a range of 6.5 to 8.5 for protection of aquatic life (Washington State Surface Water Quality Standards, WAC 173-201A). Therefore, these data indicate that pH was typically appropriate to support aquatic life. The exception was in mid-August when pH was above 8.5 throughout the water column because a large algae bloom and its consumption of carbon dioxide (a weak acid) was driving the pH increase.

Specific conductance or conductivity is a measure of the ability of water to conduct an electrical current, which is directly related to the content of dissolved ions in the water. Although there is no state surface water quality standard established for conductivity, this measurement is useful for identifying sources of dissolved solids and for determining the relative flow contributions attributed to groundwater, since conductivity is typically higher in groundwater than in surface water.

The conductivity of surface and bottom samples from the deep and shallow sites are also presented in Figure 8. The conductivity of Lone Lake was generally consistent from top to bottom for most of the year. During mid to late summer (July to early September) conductivity increased in bottom waters. Conductivity was also higher during summer months as dissolved solids increased from decomposition and sediment release into bottom waters. These differences in conductivity may be a result of changes in biological and chemical activities in the sediments that cause the release of charged particles or by changes in the influence of incoming water sources. For example, groundwater typically has a higher conductivity than lake water and groundwater could be more influential in the summer months due to the lack of surface water and precipitation inflows that 'dilute' its impact.

2.3.4. Secchi Depth

Secchi depth is a measure of water transparency, which is affected by color and the amount and size of algae and other particles in the water. It is one of the water quality variables used to determine the trophic state of lakes. Trophic state thresholds for Secchi depth commonly include less than 2 meters for eutrophic lakes and greater than 4 meters for oligotrophic lakes.

The Secchi depth in Lone Lake is presented in Figure 9. Secchi depth generally ranged from approximately 0.5 to over 2.5 meters for the June through September period. It was 2 meters or less on 12 of the 18 sampling dates.

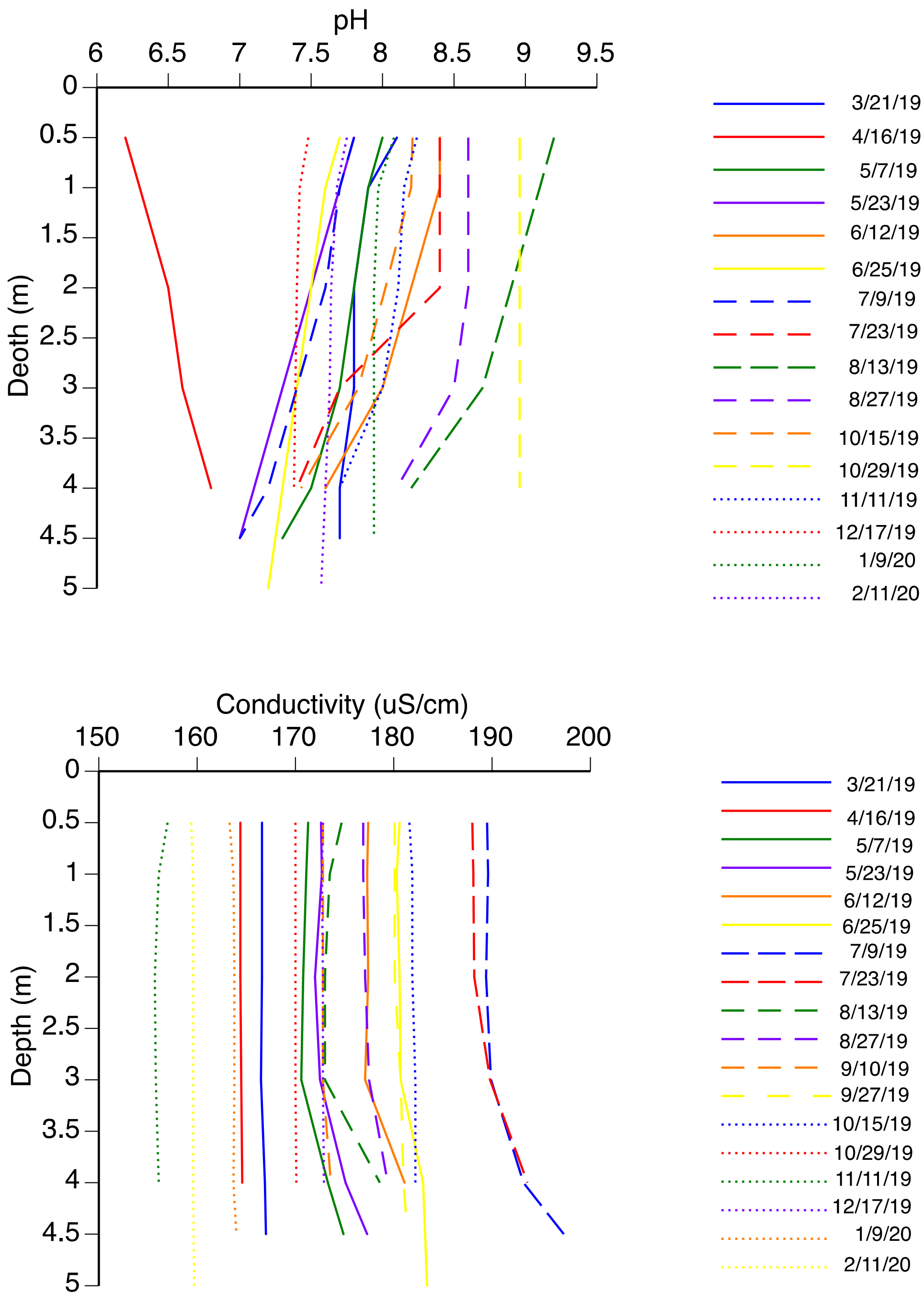


Figure 8. Conductivity and pH Profiles in Lone Lake 2019–2020.

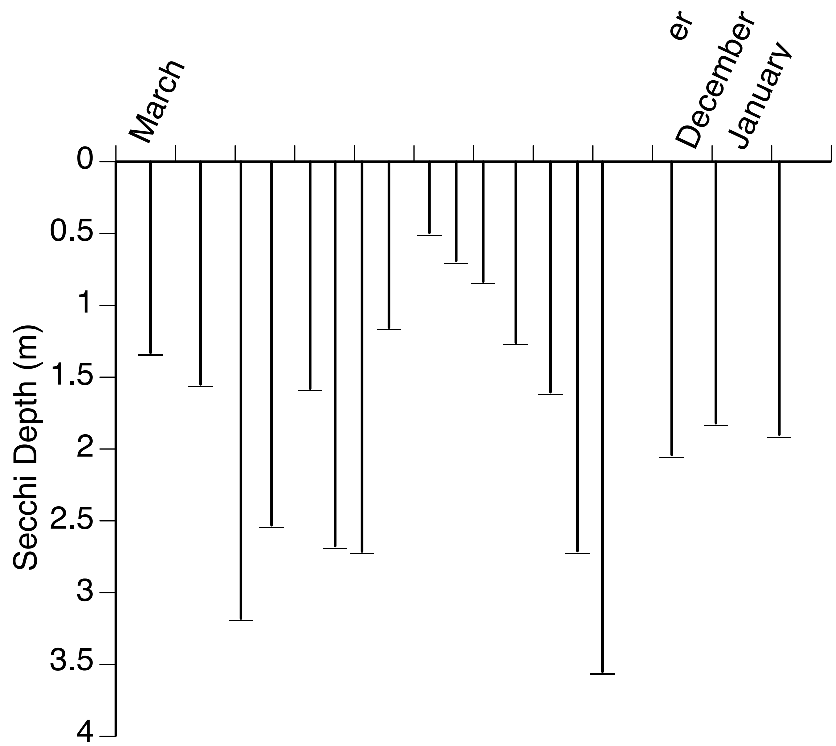


Figure 9. Secchi Depth Lone Lake 2019–2020.

2.3.5. Chlorophyll-*a*

Chlorophyll-*a* is a common measure of phytoplankton biomass. However, it is present in highly varied amounts among phytoplankton species and growth stages, and often does not relate well to other measures of phytoplankton biomass such as cell biovolume. It typically correlates well with Secchi depth transparency unless there are large amounts of suspended inorganic particles causing turbidity in a lake. The summer mean concentration of chlorophyll-*a* is one of the three variables used to determine the trophic state of lakes. Common thresholds include less than 2.6 micrograms per liter (µg/L) (or milligrams per meter cubed (mg/m^3) for oligotrophic lakes and greater than 7.2 µg/L for eutrophic lakes.

Chlorophyll-*a* concentrations are presented in Figure 10. Chlorophyll-*a* concentrations ranged from 0 to 50.5 µg/L during the summer (June through September) with an average concentration of 25.4 µg/L. The annual average concentration was 18.8 µg/L; and concentrations were well within the eutrophic range during most monitoring events. Chlorophyll-*a* concentration is an indirect measure of algal and cyanobacteria biomass since both have chlorophyll pigments. Although there were no direct measurements of algae biomass during the monitoring, it is likely that the peaks in chlorophyll concentrations shown in the figure are related to algae blooms. However, a peak in algae or an algae bloom does not necessarily correspond to a toxic algae bloom. There are many species of algae that are not capable of producing toxins that can still produce blooms, and even cyanobacteria that are capable of producing toxins do not always produce those toxins. So, the occurrence of toxic blooms cannot be predicted or inferred by chlorophyll concentrations.

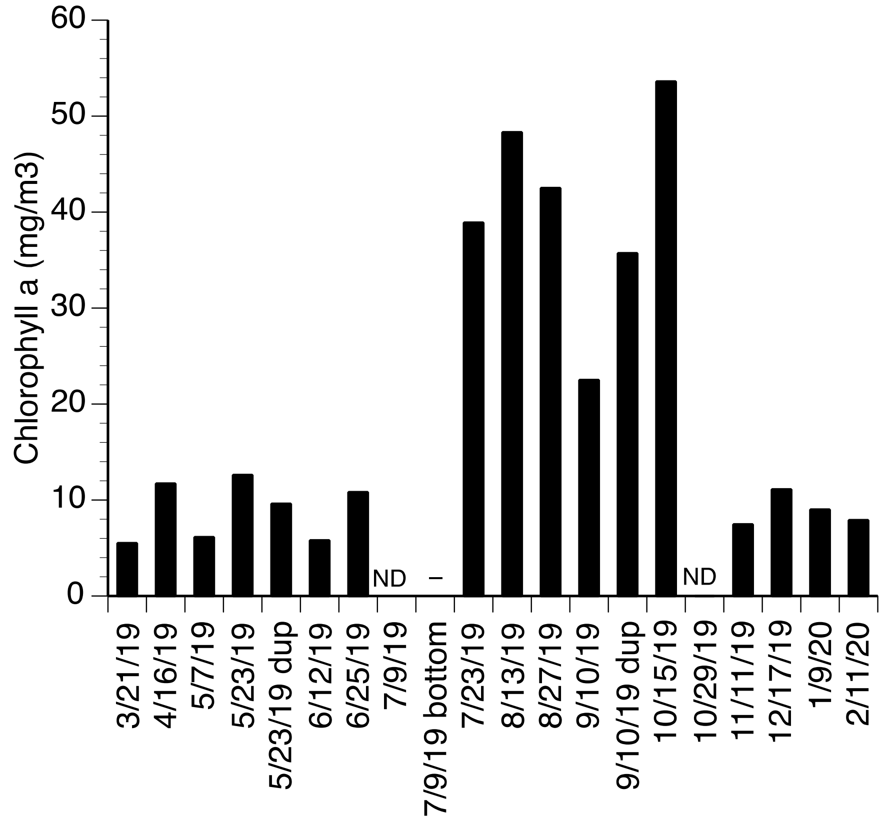


Figure 10. Surface Chlorophyll-*a* in Lone Lake 2019–2020.

2.3.6. Phosphorus

Phosphorus is a key nutrient for algae growth. Total phosphorus is a combination of inorganic and organic forms of phosphorus, which can come from natural sources or anthropogenic sources (e.g., wastewater treatment plants, septic system failures, animal manure storage, and fertilizer runoff). Phosphorus is a concern in fresh water because high levels can lead to accelerated plant growth, algal blooms, low dissolved oxygen, decreases in aquatic diversity, and eutrophication.

Soluble reactive phosphorus, also known as orthophosphate, is an inorganic fraction of phosphorus that is produced by natural processes, but also can be measured in municipal sewage. Additional sources of soluble reactive phosphorus are similar to those for total phosphorus such as septic system failure, animal waste, decaying vegetation and animals, resuspension from the bottom of a lake, and fertilizer runoff. It is a very unstable form of phosphate that is directly absorbed by aquatic vegetation and microbes such as algae.

Phosphorus is typically the most limiting nutrient for freshwater algae, meaning that control or reduction of this nutrient can be the key to reducing algae. For this reason, phosphorus is typically the focus of lake studies. Common thresholds include less than 0.012 mg/L (12 µg/L) for oligotrophic lakes and greater than 0.024 mg/L (24 µg/L) for eutrophic lakes. Washington State Surface Water Quality Standards (WAC 173-201A) established an action level of 0.020 mg/L (20 µg/L) for total phosphorus in Puget Sound lowland lakes. Summer mean concentrations greater than 0.030 mg/L (30 µg/L) generally result in undesirable algae growth that interferes with recreational uses of lakes in the Puget Sound region (Gilliom 1983).

Total phosphorus measurements include both the phosphorus that is already contained in organic matter such as algae as well as the soluble phosphorus that is still in the water and available for algae growth.

Total phosphorus (TP) concentrations are presented in Figure 11. Annual total phosphorus concentrations ranged from approximately 0.023 mg/L to 0.503 mg/L with an average concentration of 0.152 mg/L. During the June through September period that corresponds with maximum chlorophyll-*a* concentrations, the phosphorus concentrations ranged from 0.087 mg/L to 0.503 mg/L with an average of 0.272 mg/L, nearly twice the annual average concentration. These results clearly show that Lone lake concentrations are well above the State action level and the threshold for undesirable algae growth throughout the year.

The summer mean TP concentration of 0.272 mg/L is very similar to the mean TP concentration of 0.235 mg/L measured in 10 samples collected by the Western Washington University Institute of Watershed Studies from Lone Lake in 2008 through 2018 during summer months of July through September. However, the summer mean TP concentration is much higher than the mean TP concentration of 0.060 mg/L measured in nine samples collected by Ecology in 1993 through 1996 during the summer months of June through September. The four-fold increase in summer lake TP observed from 1996 to 2008 indicates that management of *Egeria densa* by herbicides and grass carp in 2005 through 2009 dramatically increased the internal loading of phosphorus in the lake and caused a shift in the alternative stable state from a clear-water lake to turbid, cyanobacteria-dominated lake.

Figure 11 also displays soluble reactive phosphorus (Soluble reactive P or SRP) concentrations over the course of the monitoring. This is the fraction of phosphorus that algae need to continue growing. The annual range in soluble reactive phosphorus was 0 to 0.290 mg/L and an average annual concentration of 0.080 mg/L. The summer period average was 0.160 mg/L. These are high concentrations relative to other lakes which is expected based on the high total phosphorus concentrations. The high summer period average indicates that there was plentiful soluble reactive phosphorus in the water column during the summer months; thus it was available to feed additional algae growth. Concentrations of soluble reactive phosphorus can sometimes be near zero during algae blooms when lakes are phosphorus limited.

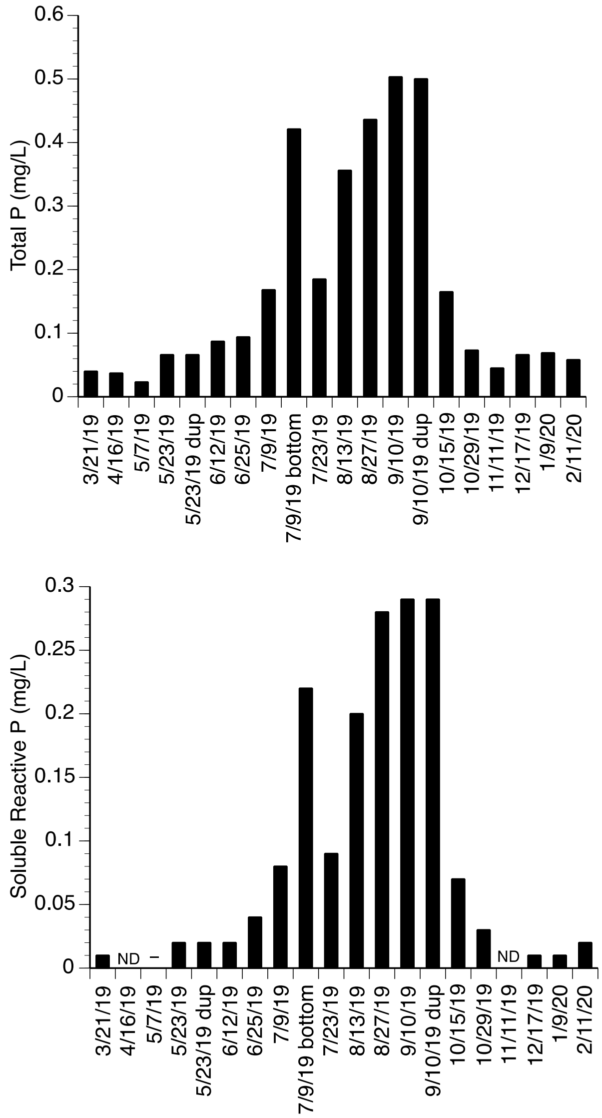


Figure 11. Total and Orthophosphate Phosphorus Concentrations in Lone Lake 2019–2020.

2.3.7. Nitrogen

Nitrogen is another important nutrient for algae growth. Total nitrogen (TN) is the sum of: organic nitrogen (which is bound to organic matter such as algae cells), and dissolved inorganic nitrogen (DIN) (which is composed of nitrate, nitrite, and ammonia nitrogen). Nitrogen is typically in plentiful supply in lakes, in part because nitrogen-fixing bacteria and many cyanobacteria species have the ability to use nitrogen gas from the atmosphere. However, nitrogen fixation is a slow and energetically-demanding process that typically occurs only when the DIN supply becomes exhausted, and several cyanobacteria genera including *Microcystis* are not able to fix nitrogen because they do not have heterocysts (Welch and Jacoby 2004).

Phytoplankton growth in eutrophic and hypereutrophic lakes is commonly limited by nitrogen because the phosphorus recycling and internal loading rate is so high due to low oxygen in bottom waters and sediments. A large data set from worldwide lakes shows that phytoplankton biomass as chlorophyll-*a* correlates with total phosphorus regardless of whether the lakes are P- or N-limited, and researchers have shown that chlorophyll-*a* is dependent on total phosphorus at concentrations less than 0.20 mg/L and that the percentage of cyanobacteria decreases with decreasing TP concentrations (Welch and Jacoby 2004). Thus, phosphorus can be regarded as the controlling nutrient well into hypereutrophy despite a tendency for nitrogen to become more likely to limit growth with increased trophic state.

Total nitrogen (Total N) concentrations, nitrate+nitrite nitrogen(NO3+NO2-N), and ammonia (NH4-N) nitrogen are presented in Figure 12. Total nitrogen concentrations ranged from 0.690 mg/L to 1.770 mg/L over the year with an average of 1.075 mg/L. Concentrations were higher in the summer months (June through September) and ranged from 1.050 to 1.770 mg/L, with an average concentration of 1,382 mg/L. The higher concentrations coincided with the period of high concentrations of phosphorus and chlorophyll-*a*, indicating that the nutrient was likely largely contained in algae cells. For perspective, trophic state thresholds suggested for summer average total nitrogen concentrations are less than 0.35 mg/L representing oligotrophic conditions, 0.35 to 0.65 mg/L for mesotrophic, 0.65 to 1.2 mg/L for eutrophic and greater than 1.2 mg/L representing hypereutrophic conditions (Welch and Jacoby 2004). Therefore, the lake would be considered hypereutrophic. in terms of total nitrogen concentrations.

The summer mean TN concentration of 1.075 mg/L is less than the mean TN concentration of 1.73 mg/L measured in 10 samples collected by the Western Washington University Institute of Watershed Studies from Lone Lake in 2008 through 2018 during summer months of July through September. However, the summer mean TN concentration is similar to the mean TN concentration of 0.97 mg/L measured in six samples collected by Ecology in 1993 through 1996 during the summer months of June through September. The small increase in summer lake TP observed from 1996 to 2008 indicates that management of *Egeria densa* by herbicides and grass carp in 2005 through 2009 did not dramatically increased the internal loading of TN like it did for TP.

Nitrate+nitrite nitrogen was below detection (less than 0.010 mg/L) or near detection limits in almost all of the samples. Since this is the form most quickly used by algae these results are not unusual, especially in a eutrophic lake.

Ammonia nitrogen concentrations are also presented in Figure 12. Ammonia nitrogen concentrations ranged from 0 mg/L to 0.150 mg/L over the year with an average of 0.046 mg/L. Concentrations were not significantly higher in the summer months (June through September) and ranged from 0.020 mg/L to 0.150 mg/L, with an average concentration of 0.060 mg/L.

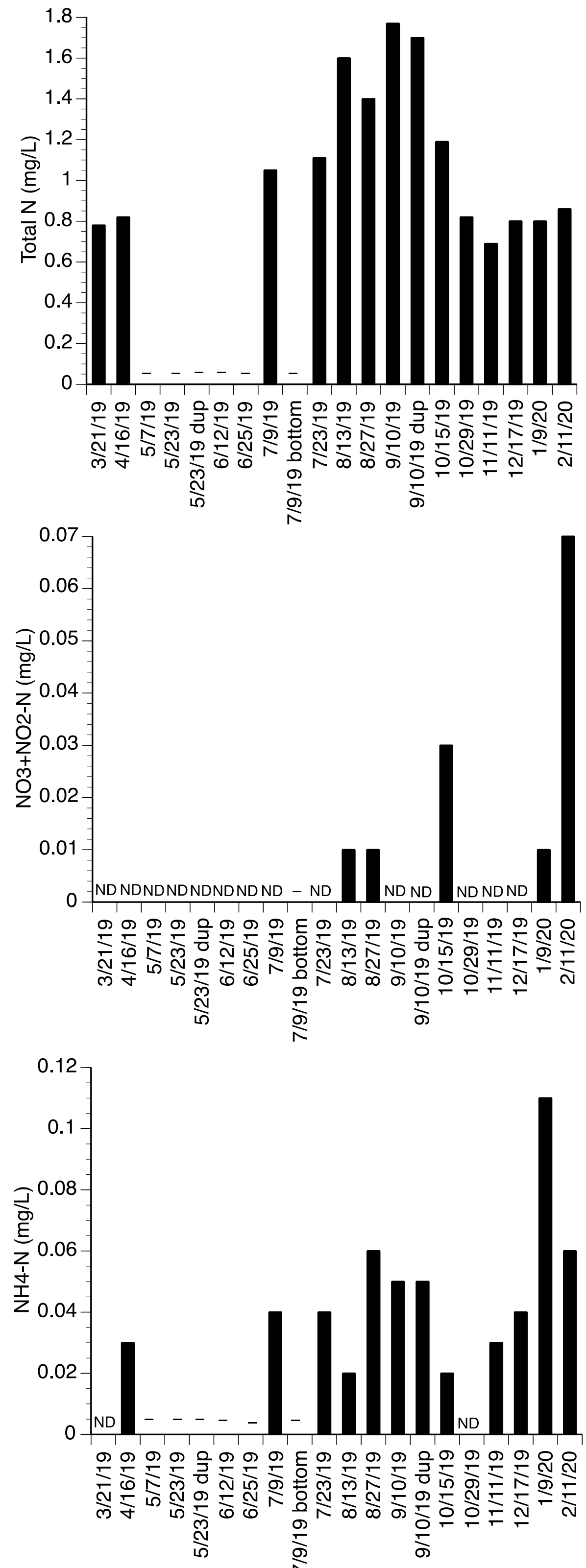


Figure 12. Total Nitrogen, Nitrate+Nitrite, and Ammonia Nitrogen Concentrations in Lone Lake 2019–2020.

Ammonia is a natural byproduct of algae decay in lakes that can accumulate at low levels before it is converted to nitrate by nitrifying bacteria. The low concentrations of ammonia nitrogen observed in Lone Lake are not indicative of septic tank inputs because ammonia nitrogen concentrations in sanitary sewage typically range from 20 to 40 mg/L, which are at least 500 times those observed in Lone Lake. Although ammonia nitrogen has not been monitored in inflow streams, relatively low total nitrogen and nitrate+nitrite nitrogen concentrations were measured in inflow stream samples collected in 2009 through 2011 that do not indicate substantial contamination by septic tank effluent. For example, a dozen samples each from inflow streams 1 and 2 exhibited mean total nitrogen concentrations of 1.2 and 2.0 mg/L, respectively, and mean nitrate+nitrite nitrogen concentrations of 0.18 and 0.36 mg/L, respectively. Thus, total nitrogen concentrations in the streams were less than or similar to those observed in Lone Lake during this study.

2.3.8. Total Nitrogen to Total Phosphorus Ratio

It is generally accepted that phosphorus is the primary limiting nutrient in lakes and nitrogen is the primary limiting nutrient in marine waters. A recent review of nutrient limitation literature concluded that while phosphorus appears to control phytoplankton growth in oligotrophic lakes over the long term (years), most lakes appear to be limited over the short term (months) by both phosphorus and nitrogen (co-limitation), and possibly by other resources such as iron (Sterner 2008). One study concluded that nutrient limitation depends on both nutrient concentrations and their ratio (Guildford and Hecky 2000). Based on nutrient relationships observed in 221 lakes, they found that phosphorus-deficient growth occurred consistently at total N:P ratios greater than 22, and nitrogen-deficient growth occurred consistently at total N:P ratios less than 9.

In Lone Lake the annual range in N:P was 3 to 22 with an average of 10, while in the critical summer period the range was only 3 to 6 with an average of 5 (Figure 13). Based on these results, Lone Lake was nitrogen limited during the critical summer period. While nitrogen limitation is an important consideration for algae management, the N:P ratio may increase as nutrient sources are reduced to where reductions in both nitrogen and phosphorus are important for reducing the occurrence of toxic algae blooms.

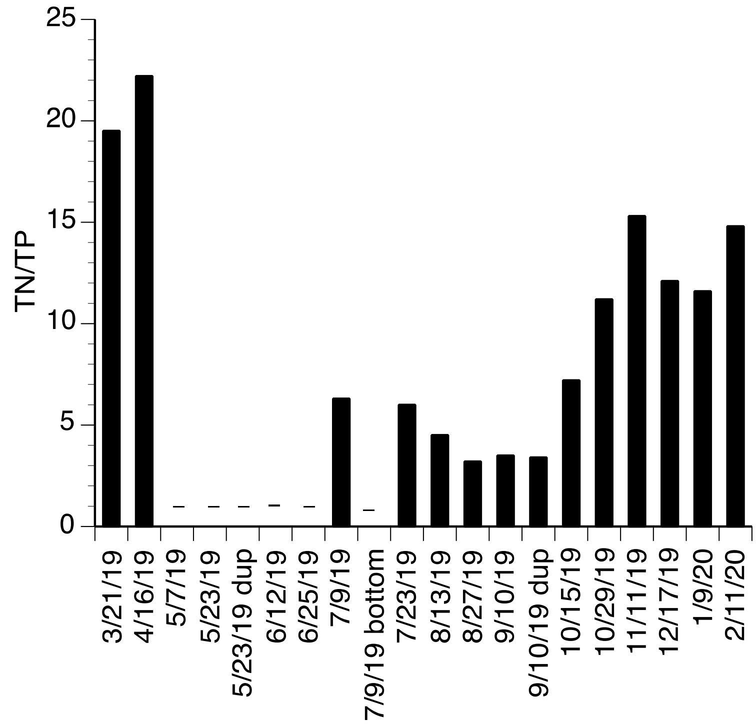


Figure 13. Ratio of Total Nitrogen to Total Phosphorus Concentrations in Lone Lake 2019–2020.

2.3.9. Trophic State Index

Lakes are classified into one of four trophic states based on increasing amounts of nutrients and algae: oligotrophic (low nutrients and productivity), mesotrophic (intermediate nutrients and productivity), eutrophic (high nutrients and productivity), and hypereutrophic (very high nutrients and productivity). Carlson's trophic state index (TSI) is commonly used to determine the trophic state based on summer (May through October) average values of Secchi depth, chlorophyll *a*, and total phosphorus in the epilimnion (surface layer) of a lake. The trophic state indices and criteria used in the evaluation are presented in Table 2.

Table 2. Carlson's Trophic State Indices and Criteria for Lakes.

Trophic Class	Trophic State Index	Secchi Depth (meters)[a]	Chlorophyll *a* (µg/L)[a]	Total Phosphorus (mg/L)[a]
Oligotrophic	<40	>4	<2.6	<0.012
Mesotrophic	40 to 50	2 to 4	2.6 to 7.2	0.012 to 0.024
Eutrophic	50 to 60	1 to 2	7.2 to 20.1	0.024 to 0.048
Hypereutrophic	>70	< 0.5	>56	>0.096

[a] Summer mean value for epilimnion.

Summer mean values and trophic state indices for Lone Lake in 2019 are presented in Table 3. Based on these data that lake is eutrophic in terms of chlorophyll-*a* (the most important of the three parameters) and Secchi depth, but nearing hypereutrophic for total phosphorus. The substantially higher TP TSI of 85 than the chlorophyll TSI of 64 is another indication of nitrogen limitation, and the lower Secchi depth TSI of 55 is indicative of low non-algal turbidity (e.g., from bacteria or clay) and particularly high algal cell concentrations of chlorophyll (e.g., from low light conditions) in the lake (Cooke et al. 2005). Based on all three parameters the overall trophic state of Lone Lake should be considered eutrophic but nearing a hypereutrophic state.

Table 3. Trophic State Indices for Lone Lake 2019.

Variable	Secchi Depth	Chlorophyll-*a*	Total Phosphorus
Summer Mean	1.45 (meters)	29.0 (µg/L)	0.272 (mg/L)
Trophic State Index	55	64	85
Trophic State Index Average	68		

2.3.10. Phytoplankton and Cyanobacteria Toxins

Phytoplankton samples were collected twice each month between June and October 2019. The algae in each sample were identified to the lowest practical taxonomic level (taxon). Each taxon was assigned a relative abundance rank, where 1 indicates presence, 2 indicates the taxon was common, and 3 indicates that it was the dominant taxon. The results are summarized in Table 4. The full report is included as Appendix B.

In June and early July diatoms and green algae were the dominant taxon, in late July through the end of September cyanobacteria dominated the algae population. This pattern is fairly typical for western Washington lakes. *Dilochospermium*, *Microcystis*, and *Woronichinia* were typically the dominant cyanobacteria and these were present throughout the monitoring period.

Washington State Department of Ecology (Ecology) has a freshwater algae monitoring program and they report results from lakes throughout Washington State on their website (<https://www.nwtoxicalgae.org/Default.aspx>). The bar charts below (Figure 14) was taken directly from that website to summarize the number of samples collected each year that exceeded state guidelines for four different types of algae derived toxins: anatoxin-*a*, cylindrospermopsin, microcystin, and saxitoxin. As shown, anatoxin-*a* concentrations have exceeded State recreation guidelines at least one time every year in the 5 most recent years of data, and in 6 of the past 11 years. Concentrations have ranged from 0.011 to 8,960 µg/L with an average of 249 µg/L during that period. This compares with the State standard of 1 µg/L.

Microcystin concentrations are also exceeded State guidelines at least one time in 4 of the last 5 years and in 7 of the 13 years it has been tested. Concentrations have ranged from below detection to 500 µg/L, and the average over the years is 30 µg/L. The maximum concentration was measured in October of 2018. This compares to the Washington State guideline of 6 µg/L. The other two toxins, cylindrospermopsin and saxitoxin, were detected in some years but never at concentrations that exceeded State standards.

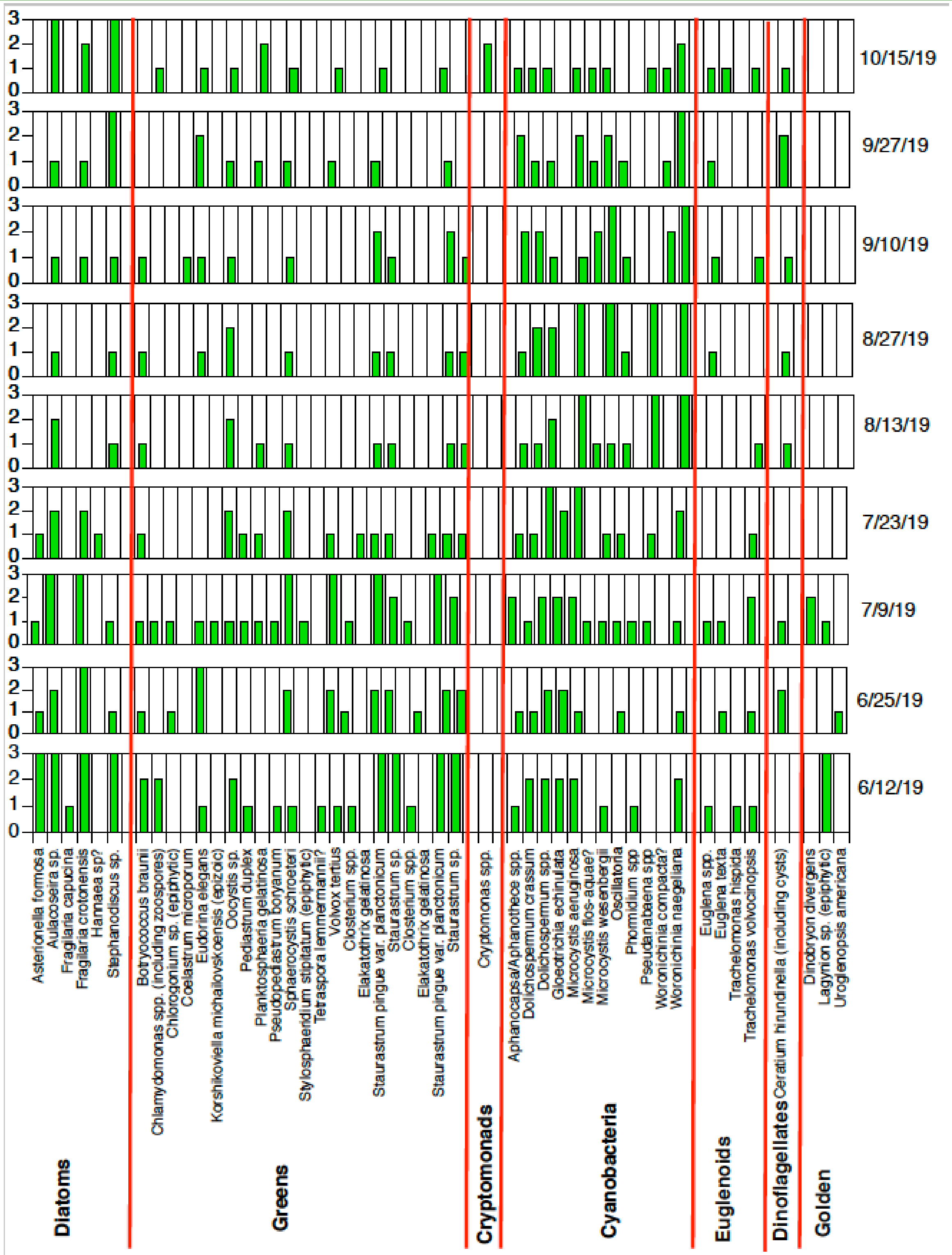


Table 4. Phytoplankton Species Presence and Dominance During June Through October 2019 Sampling Events in Lone Lake.

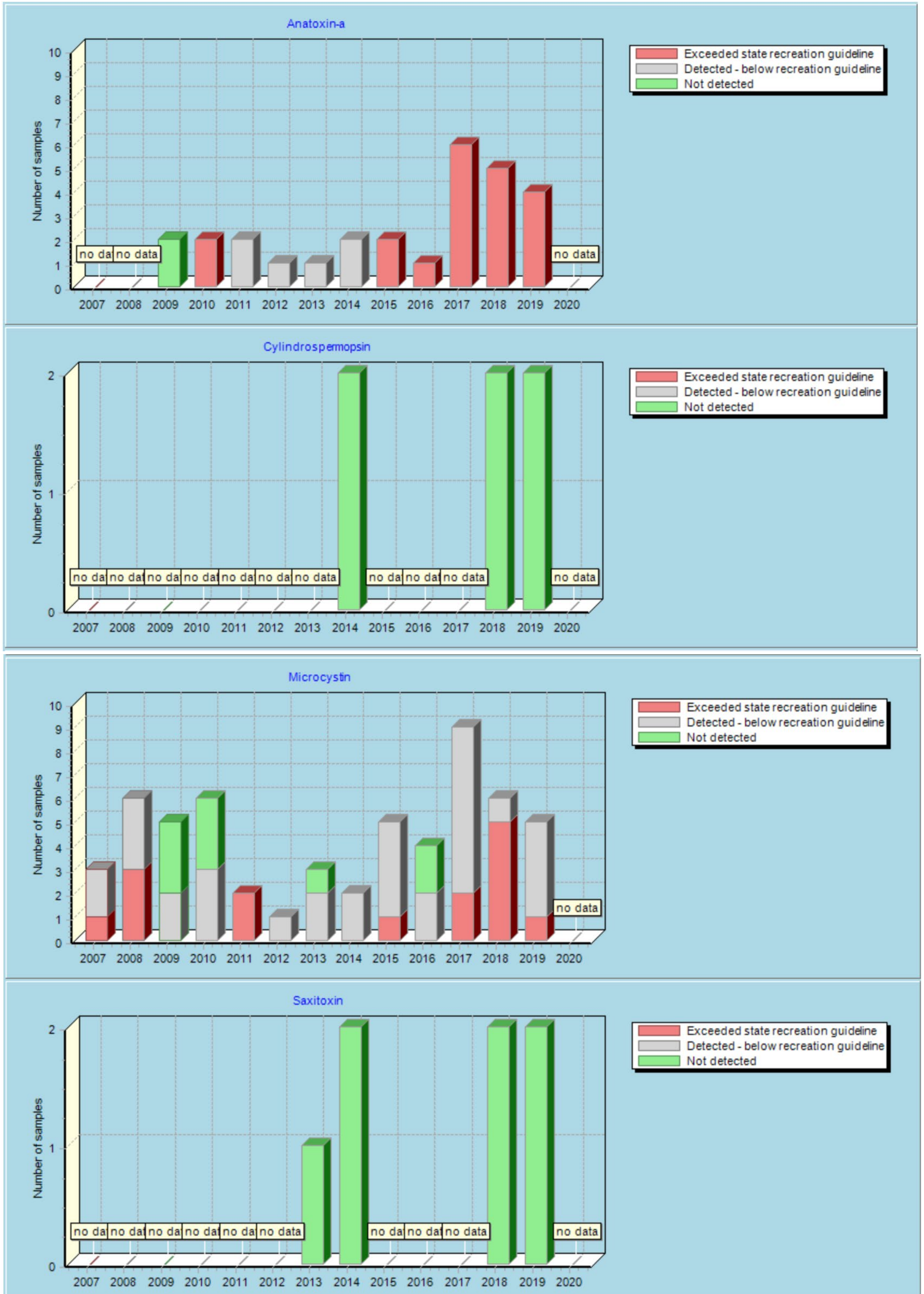


Figure 14. Sampling Results from Assessment of Algal Toxin Concentrations in Lone Lake 2007–2019.

3. SEDIMENT PHOSPHORUS

3.1. METHODS

Sediment core analysis of Lone Lake was performed to evaluate its potential for contributing to internal phosphorus loading, and to calculate the dose of alum for sediment phosphorus inactivation. Two sediment cores were collected at the central lake station using a gravity coring device. The total depth of sediment obtained was 40 centimeters (cm). The cores were sectioned into 12 depth-interval samples. The depth intervals were set at every 2 cm up to 30 cm, the final depth interval was from 38 to 40 cm. All 12 of the sediment sections from one of the cores (Core C) were analyzed for phosphorus fractions (i.e., loosely bound phosphorus, iron bound, aluminum bound, calcium bound, organic, biogenic (biologically active portion of organic phosphorus), and total phosphorus. Total iron was also measured. One section (the 10 to 12 cm depth interval) of the second core (Core B) was analyzed for all of the same analytes and used for quality assurance purposes to calculate relative percent differences (RPD) between the two cores.

Analysis of the sediment cores was performed by Aquatic Research, Inc., an Ecology-certified laboratory in Seattle, Washington. A total of 13 depth-interval samples were analyzed for solids and water, total phosphorus, iron and the following phosphorus fractions using the methods identified in the QAPP:

- Loosely bound phosphorus
- Iron bound phosphorus
- Organic phosphorus
- Aluminum bound phosphorus
- Calcium bound phosphorus
- Biogenic phosphorus

It is the mobile portion of sediment phosphorus that is of most interest because it represents that portion that may eventually be released into the lake and therefore the portion that is targeted for control through phosphorus inactivation techniques. All of the loosely bound and iron bound phosphorus as well as the biogenic portion of the organic phosphorus are the fractions that best represent the mobile portion of sediment phosphorus.

3.2. Results

The sediment core analysis results are presented in Table 5. The laboratory reports are presented in Appendix C. All requested analyses were performed, and data quality was reviewed for accuracy and precision. The detection limits were acceptable, and all analytes were detected in all samples with the exception that loosely bound phosphorus was only detected in a few the two samples from the deepest sediment sections. The recommended maximum sample holding time of 48 hours for orthophosphate analysis was exceeded by 16 days, but not the holding time of 28 days for total phosphorus. The 18-day holding time may explain the low concentrations of loosely bound phosphorus, but this labile form of phosphorus would have likely bound to iron and not affected the mobile or total phosphorus concentrations. Accuracy was acceptable based on recoveries ranging from 96 to 103 percent for quality control check samples. Precision was acceptable based on relative percent differences between laboratory duplicates of less than 15 percent for all parameters since they are within the 20 percent RPD objective defined in the QAPP. All of the sediment phosphorus fractions were lower in the QA core than in the sample core indicating that the differences may likely have been caused by actual differences in the samples. Sample results are not discarded or qualified based on field duplicate RPDs because they represent natural variability.

Lone Lake sediments have a high water content, ranging from 94 to 96 percent with no significant decrease with depth to 40 cm. They were very low in terms of percent solids (3 to 6 percent). Iron concentrations ranged from approximately 6,000 to 10,000 mg/Kg.

The iron to phosphorus (Fe:P) ratio was low (5 to 10) throughout the sediment profile. To regulate sediment phosphorus release the Fe:P ratio should exceed 10, and should exceed 15 to prevent phosphorus release from oxidized sediments (Sondergaard et al. 2003). The low Fe:P ratios within the sediment, suggests that phosphorus release should be high from sediments.

Total phosphorus concentrations decreased with depth in the soil profile, ranging from 1,552 mg/kg in the upper 2 cm to 897 mg/kg at 38 to 40 cm. Most of the sediment phosphorus was bound to organic matter at all depths. Loosely bound phosphorus was below detection in all but the two deepest samples, which is not uncommon in lake sediments because it is readily released into the water, but it also may have been due to binding of orthophosphate to iron during the 18-day holding time.

Iron bound phosphorus concentrations were high (117 to 378 mg/kg) but decreased with depth. The biogenic portion of organic phosphorus also decreased with sediment depth. The aluminum and calcium bound fractions were less variable and did not display a notable change with depth.

Concentrations of phosphorus fractions which are considered to be biologically available to microbial growth, are summarized in Table 6. Results are summarized for the top 10 cm depth intervals because this is the zone where the most biologic activity is occurring and the appropriate target for phosphorus inactivation. Active phosphorus consists of mobile phosphorus (sum of loosely- and iron-bound phosphorus) and biogenic phosphorus. Percent active phosphorus was lowest (52 percent) at the surface of the sediments and highest (70 percent) in the next 2 cm, below 4 cm there was little change. These results will be used for determining the amount of internal phosphorus loading determining an appropriate dose of alum to inactivate sediment phosphorus.

Table 5 Sediment Analysis Data for Lone Lake.

Core	CM	Solids Percent	Water Percent	Total P (mg/kg)	Loosely-Bound (mg/kg)	Iron-bound (mg/kg)	Al-Bound (mg/kg)	Biogenic P (mg/kg)	Ca-Bound (mg/kg)	Organic P (mg/kg)	Fe (mg/kg)	Fe:P
C	0–2	3.39	96.6	1552	<2.00	378	222	654	55.5	896	9843	6.3
C	2–4	3.84	96.2	1520	<2.00	308	223	686	61.3	928	9989	6.6
C	4–6	4.44	95.6	1381	<2.00	263	227	602	61.1	830	7557	5.5
C	6–8	4.59	95.4	1409	<2.00	275	213	645	61.8	859	7203	5.1
C	8–10	4.77	95.2	1482	<2.00	243	219	686	75.3	945	7370	5.0
C	10–12	4.69	95.3	1213	<2.00	222	211	567	72.8	807	7345	6.1
C	12–14	4.91	95.1	1223	<2.00	192	195	519	75.7	760	6237	5.1
C	14–16	4.92	95.1	1179	<2.00	190	207	507	80.5	703	6809	5.8
C	16–18	5.04	95	1158	<2.00	168	197	507	68.6	724	6420	5.5
C	18–20	5.29	94.7	1186	<2.00	160	207	471	73	746	6875	5.8
C	28–30	5.57	94.4	961	7.33	135	200	391	57.5	561	8367	8.7
C	38–40	5.64	94.4	897	38.6	117	211	319	62.2	468	9525	10.6
B (dup)	10–12	5.12	94.9	1002	<2.00	156	185	442	48.5	613	8009	8.0

Table 6. Average Mobile and Biogenic Sediment Phosphorus in Lone Lake.						
Core	**Depth Interval**	**Mobile Phosphorus (mg/kg)**	**Biogenic Phosphorus (mg/kg)**	**Active Phosphorus (mg/kg)**	**Total Phosphorus (mg/kg)**	**Percent Active Phosphorus**
C	0–2	156	654	810	1,552	52%
C	2–4	378	686	1064	1,520	70%
C	4–6	308	602	910	1,381	66%
C	6–8	263	645	908	1,409	64%
C	8–10	275	686	961	1,482	65%
C	Average	276	654.6	930.6	1,468.8	63%
Mobile P = loosely bound P + iron bound P; Active P = mobile P + biogenic P						

4. WATER BUDGET

4.1. METHODS

A water budget was developed in monthly time steps in order to quantify all of the inflows and outflows to/from the lake over the study year (March 2019 through February 2020). The water budget serves as a key element for developing a phosphorus model that can be used to predict the lake's response to various restoration alternatives. A water budget must balance, meaning that all of the water flowing into the lake minus all of the water flowing out of the lake will equal the change in the volume of water held in the lake over the study period, where:

$$Change\ in\ Volume\ (\Delta V) = Inflows - Outflows$$

A conceptual model of Lone Lake is presented as Figure 15.

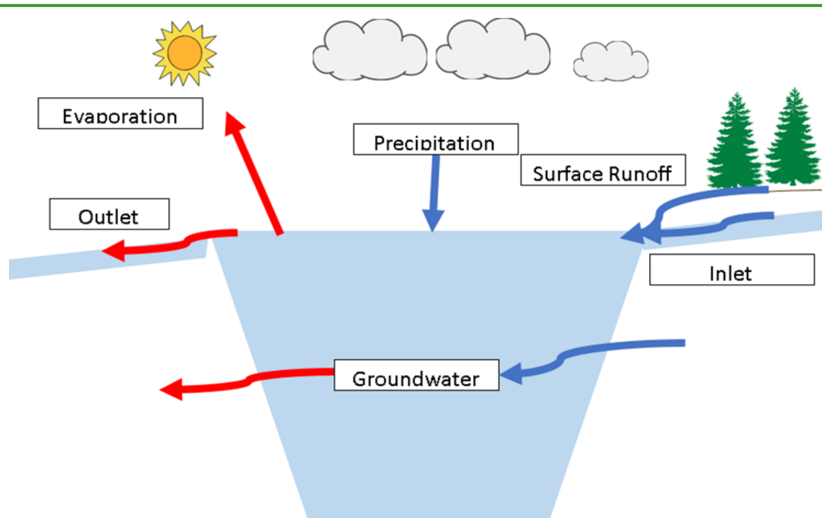


Figure 15. Conceptual Water Budget for Lone Lake.

Monthly inflow volumes into the lake included direct precipitation, base and storm flows from two inlets, and groundwater flow. Monthly outflow volumes from the lake included evaporation, lake outlet flow, and groundwater loss. The data sources and calculation methods for each inflow and outflow are described below.

4.1.1. Direct Precipitation

Direct precipitation was calculated using daily rainfall depths recorded at the Langley Weather Station (WSU 2020). The total monthly depth was multiplied by the lake surface area to get the total monthly volume.

4.1.2. Inlet Flow

Continuous depth was recorded for two inlets to the lake, and flow was measured during grab sampling events throughout the study period. The sampled flow measurements were used to develop rating curves for the inlets. Rating curves were used to calculate continuous flow. Hydrograph separation was used to calculate storm and base flow by specifying inlet-specific constants limiting the allowable daily increase in base flow. Monthly stormflow and baseflow volumes were calculated for each inlet.

Because continuous flow sampling did not begin until March 21, 2019, flow data were approximated for the beginning of March. The average flow rate for March was applied to unsampled timesteps and the two known storm events in March were approximated using storm events of similar duration and intensity in April.

No other intermittent inflows to the lake were observed during any of the monitoring events. Their potential contributions were assumed to be insignificant based on the low flows measured in the monitored streams and due to expected high infiltration of the soils in the basin.

4.1.3. Outlet Flow

Continuous depth was also recorded for the lake outlet in addition to instantaneous flow measurements. The flow measurements were used to develop a rating curve for the lake outlet to calculate continuous flow. Monthly volumes were calculated using the predicted continuous flow data.

4.1.4. Lake Evaporation

Evaporation depth was calculated using monthly average temperature and dewpoint values calculated using the daily weather data from the Langley Weather Station (WSU 2020) and the simplified Penman formula (Linacre 1977). The average monthly evaporation depth was multiplied by the surface area of the lake to get total monthly volume.

4.1.5. Lake Stage and Volume

Continuous lake stage measurements were recorded by Conservation District staff starting in 2017. The average lake stage was calculated for each month and multiplied by the lake surface area (99 acres) during the study period to determine the monthly increase or decrease in lake storage volume. Lake storage was calculated based on lake bathymetry shown in Figure 16 and the associated lake depth and volume calculations shown in Table 7.

4.1.6. Groundwater

Groundwater flows into and out of the lake were calculated by difference using the inflows and outflows described above as well as changes in lake storage volume, which were calculated using lake stage data.

4.2. Results

The monthly water budget is presented in Table 8 with inlet and outlet percentages shown in Figure 17. The two inflows account for the majority (53 percent) of the average annual inflow volume while precipitation accounted for most of the remaining inflow (43 percent). Evaporation accounts for a majority of the surface loss on a yearly basis (61 percent) and 90 percent of the surface loss during the summer months. Both surface inlets and the lake outlet dried up during summer months, indicating that precipitation, evaporation, and groundwater are driving the water budget in the summer.

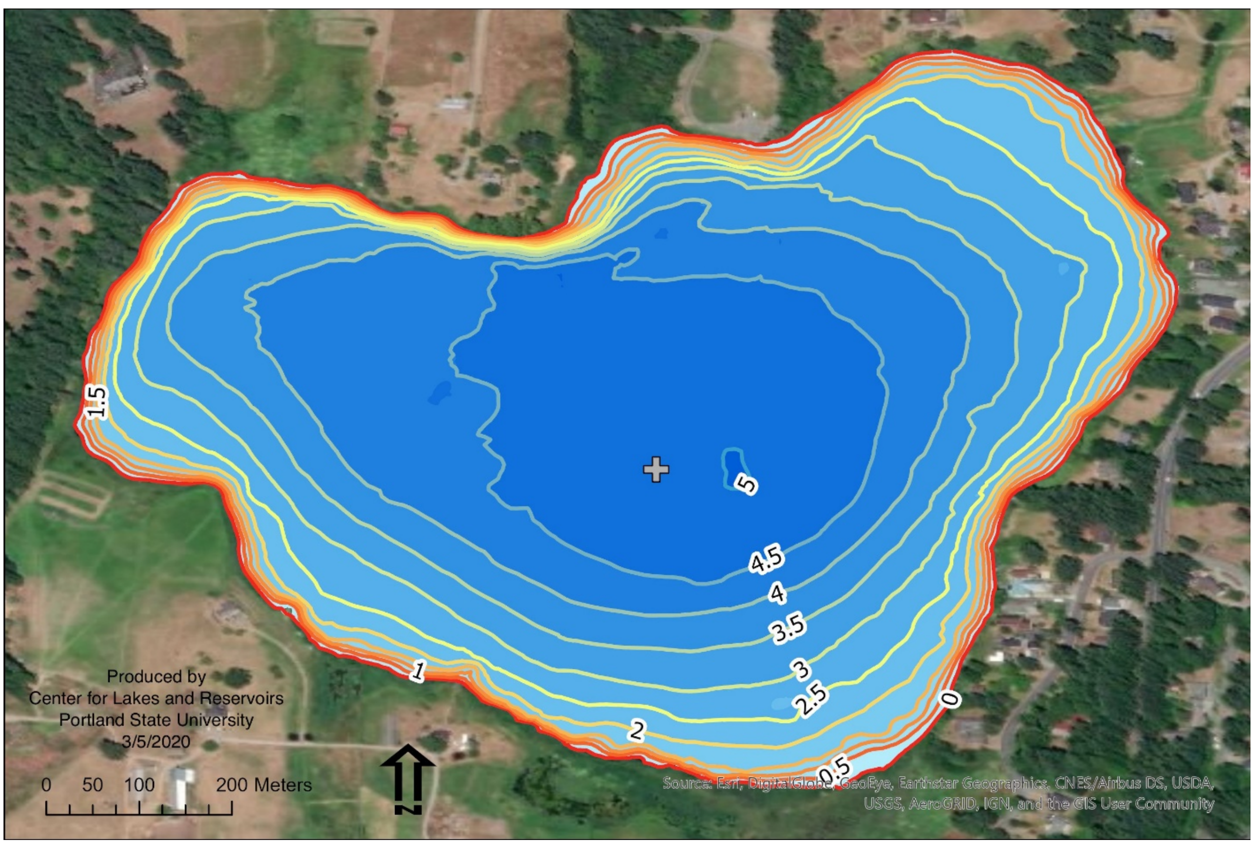
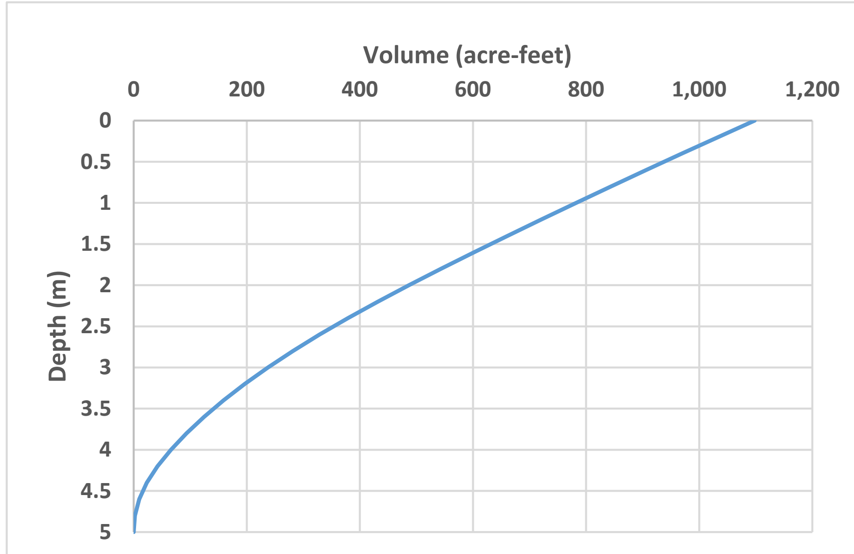


Figure 16. Lone Lake Bathymetric Map (Meters) and Sampling Location (+).

Table 7. Lone Lake Area and Volume with Depth.

Depth (meters)	Area ($meters^2$)	Volume ($meters^3$)	Area (acres)	Volume (acre-feet)
0	401,805	1,354,406	99	1,098
0.2	396,021	1,274,724	98	1,033
0.4	391,330	1,196,080	97	970
0.6	387,029	1,118,306	96	907
0.8	382,879	1,041,387	95	844
1.0	378,686	965,288	94	783
1.2	374,262	890,055	92	722
1.4	369,313	815,747	91	661
1.6	363,379	742,519	90	602
1.8	355,543	670,628	88	544
2.0	345,637	600,522	85	487
2.2	333,546	532,590	82	432
2.4	318,531	467,367	79	379
2.6	300,959	405,410	74	329
2.8	281,506	347,161	70	281
3.0	262,637	292,764	65	237
3.2	242,888	242,228	60	196
3.4	222,178	195,710	55	159
3.6	201,258	153,357	50	124
3.8	179,905	115,274	44	93
4.0	158,478	81,387	39	66
4.2	133,708	52,038	33	42
4.4	99,431	28,618	25	23
4.6	64,798	12,223	16	10
4.8	30,043	2,647	7	2
5.0	656	3	0	0



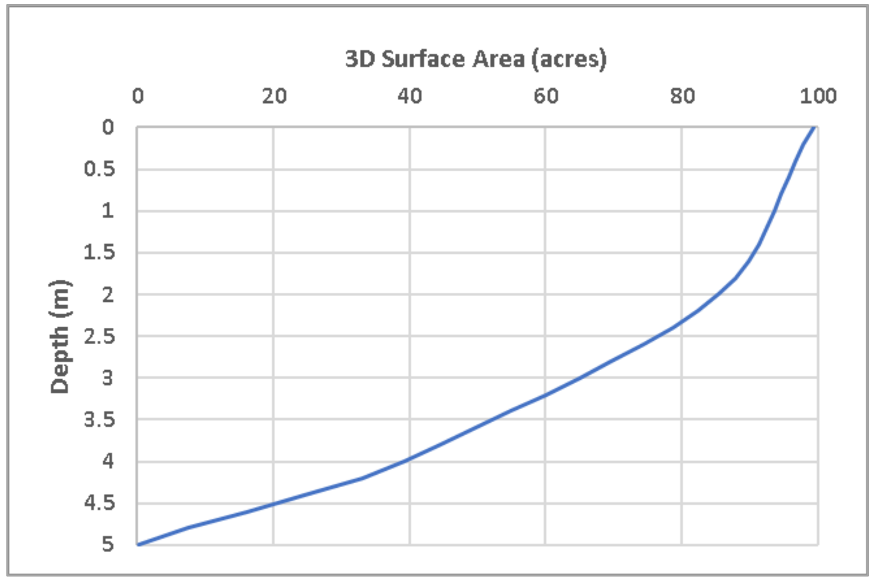
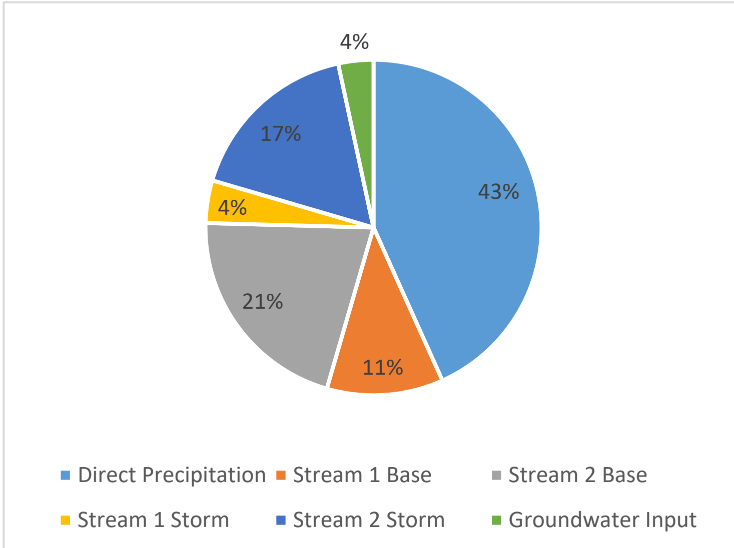


Table 8. Lone Lake Water Budget.												
	Input Volumes (m^3)						Output Volumes (m^3)			Water Balance (m^3)		
Month/ Year	Direct Precip-itation	Stream 1 Base	Stream 2 Base	Stream 1 Storm	Stream 2 Storm	Total Surface Input	Lake Evapora-tion	Outlet Stream	Total Output	Total Input Minus Total Output	Lake Storage Gain(+)/ Loss(-)	Net Ground-water Input(+)/ Output(-)
Mar/19	15,366	6,712	4,143	1,002	3,254	30,477	26,561	42,943	69,504	-39,027	12,211	-26,815
Apr/19	21,268	7,409	6,752	1,209	4,428	41,066	31,694	58,491	90,185	-49,119	-13,026	-62,144
May/19	14,043	5,393	1,183	1,112	861	22,593	45,415	15,346	60,760	-38,168	-44,368	-82,536
Jun/19	2,341	507	168	528	90	3,633	50,687	8,189	58,876	-55,243	-52,509	-107,752
Jul/19	4,681	120	1	467	12	5,281	60,603	2,147	62,751	-57,470	-43,961	-101,431
Aug/19	8,243	2	0	389	3	8,638	62,411	785	63,195	-54,557	-54,952	-109,509
Sep/19	43,351	4,591	2,155	3,240	6,303	59,640	43,659	1,619	45,278	14,362	3,663	18,025
Oct/19	22,286	6,550	4,707	1,244	4,886	39,673	25,552	2,727	28,279	11,394	14,043	25,437
Nov/19	19,844	11,529	16,432	3,271	11,602	62,678	17,001	10,368	27,369	35,308	12,822	48,130
Dec/19	39,993	11,464	32,553	5,263	43,762	133,034	15,558	22,351	37,909	95,125	39,077	134,202
Jan/20	41,621	11,564	44,631	2,716	11,92	112,459	17,744	35,552	53,296	59,163	59,836	119,000
Feb/20	47,828	6,908	23,180	6,227	23,750	107,893	13,434	60,857	74,290	33,603	133,715	167,318
Annual Totals												
Volume (m^3)	280,863	72,750	135,904	26,669	110,879	627,065	410,318	261,375	671,693	-44,628	66,552	21,924
Percent	45%	12%	22%	4%	18%	100%	61%	39%	100%	-7%	11%	3%
Summer (May–October) Totals												
Volume (m^3)	94,944	17,164	8,214	6,981	12,155	139,457	288,326	30,814	319,140	-179,682	-178,084	-357,766
Percent	68%	12%	6%	5%	9%	100%	90%	10%	100%	-56%	-56%	-112%



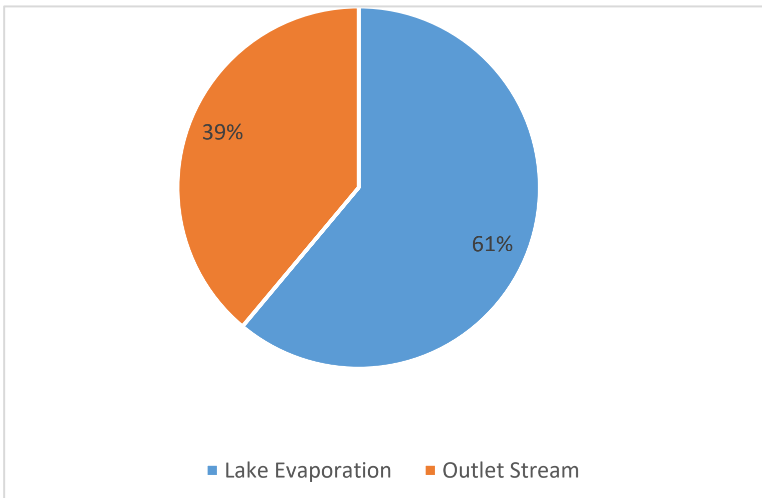


Figure 17. Lone Lake Annual Inflow (top) and Outflow (bottom), 2019–2020.

5. PHOSPHORUS BUDGET

5.1. METHODS

Using the water budget as a foundation, a phosphorus budget was created for Lone Lake that accounts for all movement of phosphorus into and out of the lake and within the lake itself. The difference between the total monthly external phosphorus inputs and outputs plus the change in phosphorus mass within the lake from the previous month equals the amount of phosphorus retained in the lake for each month, where:

$$Retention\ (\mathrm{Pret}) = Inputs\ (\mathrm{Pin}) - Outputs\ (\mathrm{Pout}) + Lake\ Storage\ Change\ (\mathrm{P\Delta L})$$

The lake phosphorus retention amount is calculated as the difference between measured total phosphorus inputs and outputs and adding the change in the amount of phosphorus stored in the lake. The lake phosphorus retention incorporates measurement errors and unmeasured sources and losses, which primarily include internal phosphorus loading and sedimentation, respectively (Steinman and Spears 2020).

5.1.1. Direct Precipitation

The total phosphorus concentration in rainfall was estimated to be 0.024 mg/L. This value is based on measured values ranging from 0.008 to 0.033 mg/L for five lakes in western Washington, and accounts for all atmospheric deposition (Ecology 2013). The total phosphorus concentration in rain was multiplied by the monthly precipitation volume to estimate phosphorus inputs from direct precipitation and other atmospheric deposition on Lone Lake.

5.1.2. Surface Inflow

Although the inflow volumes to Lone Lake were separated into storm and base flows, review of the phosphorus samples collected in Streams 1 and 2 indicated that total phosphorus concentrations did not correlate to flow. Instead, the concentrations followed seasonal patterns with higher concentrations in the spring when precipitation and base flow rates decreased and lower concentrations in the winter when precipitation and base flow rates increased. Therefore, a separate total phosphorus concentration for both base flow and storm flow was assigned to each month by averaging concentrations collected during that month or adjacent months. The average monthly total phosphorus concentration was multiplied by the respective monthly water volume separately for Steams 1 and 2 to estimate phosphorus inputs from surface water inputs to Lone Lake.

5.1.3. Outlet Flow

No lake outlet samples were collected for the study. Monthly average concentrations of total phosphorus concentrations in the lake surface samples were multiplied by the lake outlet flow volume to estimate phosphorus outputs from this source.

5.1.4. Lake Storage

The monthly amount of total phosphorus in the lake was calculated by multiplying monthly average total phosphorus concentration by monthly lake volume. Temperature profiles in the lake indicated no significant stratification of the lake, so separate values for the epilimnion and hypolimnion were not used. Monthly changes in the total amount of phosphorus in the lake were then calculated assuming there was no initial change from February to March 2019.

5.1.5. Groundwater

Groundwater phosphorus loading to the lake was calculated by multiplying the monthly volume of groundwater input (if there was a net input) by the flow-weighted average monthly concentration of total phosphorus (0.300 mg/L) for the Stream 1 and Stream 2 samples. Groundwater phosphorus export from the lake was calculated by multiplying the monthly volume of groundwater output (if there was a net output) by the monthly average concentration of total phosphorus in the lake.

5.1.6. Sediment Release

Internal phosphorus loading by sediment phosphorus release into the lake was calculated by several methods described by Nurnberg (2009) and Steinman and Spears (2020) that include the mass balance method and various sediment phosphorus release rate equations.

The mass balance method calculates the monthly increase in the amount of phosphorus that accumulates in the lake for each summer month when dissolved oxygen concentrations near the sediment surface are low and external inputs are low. The monthly sediment phosphorus release amount was calculated as the monthly increase in total phosphorus mass in the lake for June, July, and August when dissolved oxygen concentrations near the sediment surface were less than 5 mg/L and external phosphorus inputs were less than 1 kg per month. This method is similar to the hypolimnion accumulation method, which calculates the monthly increase in total phosphorus mass in the hypolimnion for those months when the dissolved oxygen concentrations are less than 2 mg/L in the hypolimnion. Accumulation in the entire lake volume and higher bottom dissolved oxygen concentrations are often used for polymictic lakes such as Lone Lake because it is recognized that sediment oxygen concentrations are much lower than those measured in the water, and sediment release also occurs from high pH conditions caused by rapid algae growth and carbon dioxide consumption.

Two sediment release equations were used based on the mobile phosphorus concentrations in the upper 10 cm of sediment in Lone Lake. The Nurnberg (1988) equation (r^2 = 0.87 for 14 lakes) is:

> Phosphorus release rate (mg/m^2/day) = 1.38 + (2.85 x Mobile Phosphorus Concentration [ug/g wet weight])

The Pilgrim et al. (2007) equation (r^2 = 0.90 for 14 lakes) is:

> Phosphorus release rate (mg/m^2/day) = 1.5 x Mobile Phosphorus Concentration (g/m^2/cm) - 0.7

The sediment release rates were multiplied by the total lake area and 90 days with unit conversion to yield mass (kg) of total phosphorus released in the study period. A release period of 90 days in June through August 2019 was selected based on dissolved oxygen concentrations and the increased amount of lake storage observed those summer months. Dissolved oxygen concentrations in bottom waters at and below 4.0 m were between 3 and 5 mg/L from May 23 through August 27, 2019, except for a low value of 0.8 mg/L at 4.5 m on July 8. As presented in the results section below, maximum increases in the amount of phosphorus stored in the lake were observed in June through August 2019. A sediment phosphorus release period of 90 days was also predicted using the Nurnberg (1996) anoxic factor equation for polymictic lakes (r^2 = 0.67 for 70 lakes) as follows:

> Anoxic factor predicted (days) = -36.2 + (50.2 x log (average summer total phosphorus concentration in lake [µg/L]) + (0.762 x mean depth [m] / square root of lake area [km^2])

5.2. RESULTS

The phosphorus budget is presented in Table 9. Internal loading of total phosphorus was estimated to total 77 kg in June through August 2019 based on the Nurnberg (1988) sediment release equation. This amount is greater than the 42 kg estimated using the Pilgrim et al. (2007) release equation and much less than the 400 kg estimated by the mass balance method. The mass balance method estimated the amount of phosphorus in the lake increased by 56 kg in June, 100 kg in July, and 244 kg in August 2019 (Table 9). Although external surface and groundwater inputs were negligible during these months, other internal inputs besides sediment release may have contributed to the observed increases in lake phosphorus (e.g., aquatic plant decay, sediment suspension, and waterfowl and fish excretion). Therefore, the 77 kg of phosphorus release was selected and proportionally distributed over the 3 months based on the observed increase the lake phosphorus storage amount.

Sediment phosphorus release represents the majority of the estimated total phosphorus input to the lake on an annual basis (41 percent) and during the summer months (75 percent). Inlets and groundwater each contributed approximately 20 percent of the phosphorus load on an annual basis, but negligible amounts during the summer algae growing season. Thus, the high total phosphorus concentrations observed in the lake inlets and their groundwater sources did not contribute a significant phosphorus load to the lake due to their relatively low input volumes.

Phosphorus inputs to lakes can be quite substantial for aquatic macrophytes and waterfowl. For example, macrophytes were estimated to contribute 40 percent of the total phosphorus input to Green Lake in Seattle when the invasive Eurasian watermilfoil covered 85 percent of the lake in 1992–1994 (Herrera 2015). Waterfowl were estimated to contribute 29 percent of the total phosphorus input, but most (26 percent) of that input was considered to be internal cycling of phosphorus in algae and plants consumed by the waterfowl. Bioturbation from European carp has been attributed to phosphorus loading in some lakes, but a carp bioturbation model for Green Lake based on its large carp population indicated that most of the disturbed sediment rapidly settles and does not contribute a substantial portion of the total phosphorus input (Herrera 2015). Green Lake is over twice the size of Lone Lake but with a similar shallow depth and a much smaller watershed (due to runoff diversion into a combined sewer system). Thus, internal inputs from macrophytes and waterfowl would be expected to be more than those to Lone Lake.

A large amount of phosphorus retention accumulated in the lake in June through September 2019. This calculated residual amount totaled 368 kg when sediment release for this same period was estimated to total 77 kg (see Table 9). Substantial aquatic macrophyte and waterfowl populations have been observed in Lone Lake and it is possible that they contributed a substantial portion of the residual phosphorus input in June through September 2019.

Phosphorus loading from waterfowl is highly variable and difficult to estimate. One study of a lake in Michigan estimated that the migratory waterfowl population of 6,500 Canada geese and 4,200 ducks (mostly mallards) contributed 88 kg of TP/year, which equates to 0.008 kg/bird/year (Manning et al. 1994). Thus, a population of 1,000 waterfowl for Lone Lake would contribute 8 kg/year or 4 percent of the estimated annual TP input. Most of the phosphorus input from waterfowl sinks to the bottom and is released from lake sediments.

The average concentration of total phosphorus measured in Stream 1 during this study was 0.131 mg/L. During monitoring performed during 2009 to 2011 the average for 12 samples was 0.060 mg/L, which is less than half of what was measured in this study. The average concentration of total phosphorus measured in Stream 2 during this study was 0.218 mg/L. During monitoring performed during 2009 to 2011 the average for 13 samples was 0.130 mg/L, which again is about half of what was measured in this study. While these findings do not impact the phosphorus budget, they do suggest that external phosphorus loadings from streams were much higher during this study year than previous study years assuming similar flow rates among years. However, the apparent increase in watershed sources of phosphorus in 2019 amounted to only 15 percent of the annual phosphorus input and 7 percent of the summer phosphorus input (see Table 9).

The high output of groundwater from the lake accounted for most (93 percent) of the phosphorus loss from the lake compared to only 7 percent loss by the outlet stream on an annual basis and a 2 percent loss by the outlet stream during the summer months (see Table 9). Because groundwater inputs and outputs were calculated as the net difference in measured inputs and outputs, the total groundwater phosphorus loadings and losses are likely higher than those reflected in the phosphorus budget.

The total summer (May through October) load of phosphorus to Lone Lake from both external and internal sources was approximately 102 kg compared to the annual load of 187 kg. Thus, 55 percent of the annual phosphorus loading occurred during the summer algae growing season.

Table 9. Lone Lake Phosphorus Budget.

Month/ Year	Surface Input Mass (kg)						Surface Output Mass (kg)	Groundwater Mass (kg)		Internal Loading (kg)	Mass Balance (kg)				
	Direct Precipi-tation	Stream 1 Base	Stream 2 Base	Stream 1 Storm	Stream 2 Storm	Total Surface Input	Outlet Stream	Ground-water Input	Ground-water Output	Sediment Release	Total Input	Total Output	Lake Storage	Change in Lake Storage	Total Retention
Mar/19	0.37	0.30	0.31	0.05	0.25	1.28	1.72	0.00	11.3	0.0	2.6	13.0	54.7	0.0	-10.5
Apr/19	0.51	0.44	0.84	0.07	0.55	2.41	2.16	0.00	26.2	0.0	4.8	28.3	50.1	-4.6	-28.1
May/19	0.34	0.85	0.37	0.17	0.27	1.99	0.68	0.00	34.7	0.0	4.0	35.4	58.3	8.2	-23.3
Jun/19	0.06	0.09	0.10	0.09	0.05	0.38	0.74	0.00	45.4	10.7	11.5	46.1	113.7	55.5	20.8
Jul/19	0.11	0.02	0.00	0.08	0.01	0.22	0.38	0.00	42.7	19.3	19.7	43.1	214.1	100.3	77.0
Aug/19	0.20	0.00	0.00	0.09	0.00	0.29	0.31	0.00	46.1	47.0	47.6	46.4	458.5	244.4	245.6
Sep/19	1.04	1.04	0.18	0.74	0.53	3.53	0.69	2.60	0.0	0.0	9.7	0.7	494.8	36.3	45.3
Oct/19	0.53	1.49	0.40	0.28	0.41	3.11	0.32	3.66	0.0	0.0	9.9	0.3	139.9	-354.9	-345.3
Nov/19	0.48	0.59	0.90	0.17	0.64	2.77	0.47	2.59	0.0	0.0	8.1	0.5	53.5	-86.4	-78.7
Dec/19	0.96	0.58	1.79	0.27	2.41	6.01	1.48	7.21	0.0	0.0	19.2	1.5	81.0	27.5	45.3
Jan/20	1.00	0.72	3.93	0.17	1.05	6.86	2.45	10.01	0.0	0.0	23.7	2.5	88.8	7.8	29.1
Feb/20	1.15	0.43	2.04	0.39	2.09	6.09	3.53	14.08	0.0	0.0	26.3	3.5	82.4	-6.4	16.3
Annual Totals															
Mass (kg)	6.7	6.6	10.9	2.6	8.3	35.0	14.9	40.2	206.4	77.0	187.1	221.3	1,889.7	27.8	-6.5
Percent	4%	4%	6%	1%	4%	19%	7%	21%	93%	41%	–	–	–	–	–
Summer (May–October) Totals															
Mass (kg)	2.3	3.5	1.0	1.5	1.3	9.5	3.1	6.3	168.9	77.0	102.3	172.0	1479.2	89.8	20.1
Percent	2%	3%	1%	1%	1%	9%	2%	6%	98%	75%	–	–	–	–	–

6. LAKE CONDITION SUMMARY

- Nutrient concentrations are very high in Lone Lake resulting in correspondingly high algal growth and chlorophyll-*a* concentrations. The lake clearly falls within the range of eutrophic conditions, verging on hypereutrophic if phosphorus alone was considered.

- Cyanobacteria dominate the algal population during the summer and they are toxin producers. Concentrations of the toxins anatoxin-*a* and microcystin have frequently exceeded state guidelines in recent years, resulting in lake closures.

- The lake appears to be nitrogen limited and there is ample soluble reactive phosphorus available in the water even during the productive summer period. This indicates that lake management activities focused on phosphorus control will need to control a large portion of the phosphorus to be effective on limiting algae growth.

- While phosphorus concentrations are very high in inflowing streams, the streams account for a very small inflow volume during summer months, and therefore the phosphorus load contributed from these streams is small during the summer months.

- Internal phosphorus loading represents the majority of the total phosphorus input to the lake on an annual basis (41 percent) and during the summer months (75 percent). And 55 percent of the total annual loading occurred during the summer months.

7. PHOSPHORUS AND ALGAE CONTROL METHODS

Given the information presented above, reasonable water quality objectives for Lone Lake would be:

- Maintaining chlorophyll-*a*, total phosphorus, and Secchi depth at or below the lowest end of the eutrophic scale in the mesotrophic range. Thus, average summertime chlorophyll-*a* concentrations would be at or below 0.072 mg/L, total phosphorus at or below 0.024 mg/L, and Secchi depth of at least 2 meters.

- Eliminating the occurrence of toxic algae blooms.

- Maintaining a healthy aquatic macrophyte community while moving the lake towards a clear state from a turbid, cyanobacteria-dominated state.

Reducing the trophic state from almost hypereutrophic to mesotrophic with a summer total phosphorus below 0.024 mg/L is predicted to reduce cyanobacteria dominance and eliminate the occurrence of toxic algae blooms because mesotrophic lakes have a diverse phytoplankton community and rarely experience toxic algae blooms. Based on the 2019 summer total phosphorus concentration of 0.272 mg/L, a 90 percent reduction of summer total phosphorus would be required to meet the water quality objective for total phosphorus at less than 0.024 mg/L. To achieve this large reduction, the entire 75 percent of the summer phosphorus input from internal phosphorus loading would need to be eliminated and additional watershed sources would need to be controlled.

A consequence of dramatically increasing water clarity is an increasing abundance of submersed aquatic macrophytes from the increased light penetration and abundant nutrient supply remaining in lake sediment. The increased abundance of submersed aquatic macrophytes will help sustain the transition to a clear water state by incorporating nutrients into rooted plant tissue rather than suspended algae biomass. The increased macrophyte abundance will improve fish habitat and perhaps return Lone Lake to its superior fishery of the past. However, it should be anticipated that native submersed plants will proliferate to a level impacting boating, swimming, and aesthetic values to some extent. recreational uses and in need of management to reduce those impacts. In addition, attached benthic algae (periphyton) could proliferate on the lake bottom from increased light penetration if the sediment nutrient supply is not sufficiently controlled.

Since the nutrient data collected in 2019 indicates that Lone Lake is nitrogen limited, it may seem counter intuitive to focus on watershed and in-lake management approaches that address control of phosphorus. As described earlier, however, large amounts of inorganic nitrogen fall into the lake directly from the atmosphere, many cyanobacteria can fix nitrogen gas from the atmosphere, and bacteria readily produce inorganic nitrogen from algae and sediment decomposition. Also, control of only nitrogen sources in a nitrogen limited lake may provide an even greater advantage to nitrogen fixing cyanobacteria when phosphorus is abundant. Furthermore, it is anticipated that Lone Lake would become phosphorus limited as nutrient inputs are controlled and its trophic state descends towards mesotrophic. Finally, controlling in-lake phosphorus inputs also reduces nitrogen inputs from less decomposition of organic matter, and controlling watershed phosphorus inputs also reduces nitrogen inputs associated with animal and human waste. Therefore, even in a nitrogen limited lake, phosphorus is still the nutrient that needs to be the focus of algae control efforts.

The following sections describe watershed management and in-lake management approaches that were considered for meeting those objectives.

7.1. Watershed Management

The Lone Lake watershed is generally largely forested (78 percent), while cleared land (typically in agriculture) comprises the other significant share (18 percent). Impervious area accounts for only 3 percent of the land cover (Figure 2). The watershed is dominated by sandy soils that provide little retention capacity. Therefore, nutrients released from activities such as onsite wastewater treatment systems, forestry, small farms, and agricultural activities can more easily be transported to the inlets and the lake. While high phosphorus concentrations were measured in the streams, the low flows in the streams were the more significant factor resulting in the overall low phosphorus loadings from the streams. In addition, shallow groundwater inputs were very low during the summer months, minimizing potential impacts from infiltration of watershed nutrients into shallow groundwaters feeding the lake. Basin delineations and more refined land use data could be used to estimate differences in potential nutrient sources between subbasins and land uses, but this effort is not be expected to significantly change the overall picture of land use and phosphorus loading without conducting a watershed monitoring program to quantify nutrient sources to streams and groundwater from specific watershed activities.

The only two known inflows to the lake (Stations 1 and 2 shown in Figure 6) were monitored as part of this study and both had very high concentrations of total phosphorus. Station 1, which appears to drain the largest portion of the watershed but is largely composed of forest, had total phosphorus concentrations ranging from 0.045 to 0.398 mg/L with an average of 0.131 mg/L. Station 2, which also drains primarily forest but appears to have more cleared land that is likely agriculture than Station 1, had even higher total phosphorus concentrations ranging from 0.055 to 1.050 mg/L with an average of 0.218 mg/L. These results indicate that agricultural activities contribute more phosphorus than forestry, as would be expected. However, it is not known if phosphorus from cleared lands primarily originate from farm animals or septic tanks, for example.

The inlet samples collected during two storm events exhibited slightly lower concentrations (0.094 mg/L) than base flow events (0.215 mg/L), based on median concentrations. The lack of significant residential development in the two monitored stream basins and the lack of increased phosphorus concentrations from runoff generated during storm events indicates that groundwater feeding these streams is high in phosphorus that is diluted by stormwater runoff rather than exacerbated by it. The high baseflow phosphorus concentrations in both streams indicates there is a relatively steady and constant source of phosphorus leaching into shallow groundwater from natural soil conditions or human activities in the watershed. Data from past monitoring on these streams indicate that phosphorus concentrations have substantially increased (doubled) in the streams in the past decade, which would have been caused by changes in human activities and not natural soil conditions.

The forest surrounding the lake does not appear to be actively managed and is not a likely source of nutrients to the streams or groundwater. As discussed earlier, the WICD has actively been working with the local agriculture community to implement BMPs to reduce animal manure and fertilizer inputs. Effective BMPs include keeping livestock out of the lake and streams, and not applying manure or other fertilizes within riparian buffers. Fencing and restoration of riparian buffers are two ways of reducing livestock inputs of nutrients from manure deposits or soil erosion. However, this work by WICD does not signify that agricultural inputs are well controlled because the work has been limited to willing landowners.

The agricultural and residential areas are served by septic systems, which could also be a phosphorus source in addition to manure and fertilizers. Incomplete adsorption of nutrients within drain fields or surface failures of septic systems, especially those located in close proximity to the lake or streams that flow into the lake, could be a source of phosphorus. Beyond the immediate perimeter of the lake, especially to the north and east where the monitored streams drain, land use is rural and therefore septic density is low. While controlling nutrient inputs from septic systems should always be an important component of lake management, the phosphorus budget results indicate that the two streams combined, represent less than 20 percent of the annual total phosphorus load and contribute no load during the critical summer period. Therefore it is unlikely that eliminating poorly functioning or failing septic systems would make a perceptible improvement to lake water quality.

Because these streams do not contribute significant phosphorus during the summer, it is unlikely that the control of these sources would be sufficient to meet the water quality objectives. However, a more thorough assessment of watershed pollutant sources should be considered. Techniques such as microbial source tracking, septic system function assessment, and other nutrient source tracing techniques should be used to better assess cost-effective source-control actions, regardless of their immediate impact to lake phosphorus loading.

7.2. In-Lake Management Techniques

The following sections summarize the most feasible lake management techniques that may be used to improve the algae community and meet the water quality objectives. All the techniques that are considered feasible have the ability to meet lake water quality objectives. There are advantages and disadvantages to each, while some are more experimental in that there are less case studies of lake applications, and there are wide differences in initial and long term costs. Table 10 provides a comparative summary of these techniques. The following section of this plan provides a brief list of lake management techniques that were not considered to be cost effective and the rationale for their elimination.

It should be understood that any lake management technique aimed at controlling algae, if successful, is likely to impact aquatic macrophyte populations. The clearer water means more sunlight for plant growth and since most plants obtain their nutrients from the sediments rather than the water, lake nutrient reduction techniques do not impact them. Lake management needs to be focused on achieving the appropriate balance between algae and plants since too much of either can be problematic. In the past, Lone Lake was a plant dominated lake but one that met beneficial uses until that plant community was invaded by an aggressive nonnative plant. Planting grass carp and use of herbicides controlled that plant problem, but caused the lake to transition from a clear water state to a turbid, cyanobacteria-dominated state. Lone Lake can be managed to achieve a better balance, but that will require also managing aquatic macrophytes.

7.2.1. Phosphorus Inactivation

7.2.1.1. Alum Treatment

Applications of aluminum sulfate (alum) applied in a sufficient dose to inactivate all mobile sediment phosphorus have been shown to be effective for at least 10 years in lakes with low watershed inputs (Cooke et al. 2005). When alum is added to water it forms a floc that grows in size and weight as it settles through the water column, sorbing inorganic phosphorus and incorporating particulate organic phosphorus through entrapment (Burrows 1977, Driscoll and Schecher 1990). The alum floc settles to the sediments where it continues to control phosphorus by sorbing additional phosphorus that is present in the sediments and thus forms a barrier to future phosphorus release from sediments into the water column. The resultant phosphorus that is bound to aluminum in the lake sediments is very stable and is thought to be permanently bound (Rydin and Welch 1998).

Table 10. Comparison of Different In-Lake Algae Management Techniques for Lone Lake.

Technique	Mode of Control	Advantages	Disadvantages	Duration	Planning Level Cost	Annual Costs
Alum	Alum removes phosphorus in the water column and blankets the sediments with a layer that blocks movement of phosphorus into the water column.	• Immediate improvement in lake algae concentrations. • Long term control of sediment derived phosphorus. • Many lake application case studies.	• Application needs to be carefully designed and controlled due potential toxic impacts.	10 years	$290,000	None.
Phoslock	Removes inorganic phosphorus from the water column, and blankets the sediments with a layer that blocks movement of phosphorus into the water column.	• Long term control of sediment derived phosphorus. • No toxicity concerns • Easy to design and permit.	• Fewer lake applications than alum, especially in the US. • Unknown duration of effectiveness. • Short term turbidity.	5–10 years	$392,000 to $451,000	$39,000 to $45,000 to cover cost for second application for a 10-year duration.
Aeration	Mixing creates conditions less advantageous to cyanobacteria growth. Whole lake or hypolimnetic systems deliver oxygen to the sediments to reduce release of sediment derived phosphorus.	• Permanent control by both mixing and oxygenation. • Depending upon design may also target sediment derived phosphorus. • Many lake applications for case studies.	• These need to be carefully designed and engineered. Poorly sized or designed applications can worsen problems. • Requires shore based electrical supply and long, air supply line. • Long term energy costs.	Permanent	$117,000 to $122,000	$26,000 Energy and maintenance costs.

Table 10 (continued). Comparison of Different In-Lake Algae Management Techniques for Lone Lake.						
Technique	**Mode of Control**	**Advantages**	**Disadvantages**	**Duration**	**Planning Level Cost**	**Annual Costs**
Nannobubble	Highly efficient oxygen transfer system that mixes and oxygenates the entire lake and water column including sediments.	● Permanent control. ● More efficient oxygen transfer. ● Transfer is effective throughout the lake from one injection point. ● Highly efficient oxygen transfer creates an oxidation effect that kills algae cells. ● Oxygen is delivered to the whole lake, including the sediments to reduce release of sediment derived phosphorus ● May increase algal mortality through oxidation as well as mixing. ● Degrades toxins. ● Oxygenates sediment.	● Requires shore- based air supply but air supply line is only needed from the equipment to the near shore area. ● Long term energy costs. ● Fewer lake case studies to confirm effectiveness.	Permanent	$204–$214K	$26,000 (Assuming same energy and maintenance cost as aeration.) There would be additional annual costs if a gas supply other than air was used.
SolarBee	Epilimnetic mixing creates conditions less advantageous to cyanobacteria growth; impacts upper layer or entire water column	● Permanent control. ● Solar-powered. Requires no shore based facility. ● No long term energy cost.	● Fewer lake case studies to confirm effectiveness. ● Less oxygen transfer than aeration or nannobubbles. ● Does not address sediment derived phosphorus.	Permanent	$165–173,000	$7,500 (Based on 5 percent of original equipment cost for maintenance.)

Table 10 (continued). Comparison of Different In-Lake Algae Management Techniques for Lone Lake.

Technique	Mode of Control	Advantages	Disadvantages	Duration	Planning Level Cost	Annual Costs
LG Sonic	Ultrasonic waves are transmitted at frequencies that provide a barrier to upward movement of algal cells into the photic zone	●Permanent control. ●Solar-powered. Requires no shore based facility. ●No long term energy cost. ●Provides real time data on lake quality.	●Few lake case studies to confirm effectiveness. ●Requires permanent contract for monitoring.	Permanent	$187 to $196K	$10,800 per year for the contract which covers maintenance.
Floating Wetlands	Biofilm growing on plant roots and within supporting matrix incorporates dissolved and particulate nutrients	●Solar powered circulation system increases removal ●Aesthetic value ●Fish and waterfowl habitat	●High cost ●Low nutrient reduction rate ●Nutrients in biofilm slough to bottom and can recycle	Permanent	$3.5 M for 2 percent lake cover	None if weeded by volunteers.
Hydrothol 191	Kills algal cells.	●Immediate, significant, predictable relief.	●Multiple applications required each summer. ●Algal cells break down and can release toxins. ●Also kills aquatic plants it comes into contact with	4 to 6 weeks	$26K to $28K (Assumes two applications per summer.)	$26K to $28K
PAK 27	Kills algal cells.	●Immediate, significant, predictable relief. ●No use restrictions or aquatic toxicity concerns. ●Does not impact aquatic plants	●Multiple applications required each summer. ●Algal cells break down and can release toxins.	4 to 6 weeks	$31K to $32K (Assumes two applications per summer.)	$31K to $32K

7.2.2. Phosphorus Inactivation

7.2.2.1. Alum Treatment

Applications of aluminum sulfate (alum) applied in a sufficient dose to inactivate all mobile sediment phosphorus have been shown to be effective for at least 10 years in lakes with low watershed inputs (Cooke et al. 2005). When alum is added to water it forms a floc that grows in size and weight as it settles through the water column, sorbing inorganic phosphorus and incorporating particulate organic phosphorus through entrapment (Burrows 1977, Driscoll and Schecher 1990). The alum floc settles to the sediments where it continues to control phosphorus by sorbing additional phosphorus that is present in the sediments and thus forms a barrier to future phosphorus release from sediments into the water column. The resultant phosphorus that is bound to aluminum in the lake sediments is very stable and is thought to be permanently bound (Rydin and Welch 1998).

Alum treatments have been used successfully in many lakes in Washington, and several strategies have been implemented in Washington and around the world to inactivate phosphorus in sediments, lakes, and from watershed inputs including the following:

- Whole lake alum dose
- Multiple small alum doses
- Microfloc alum injection
- Inflow stream alum injection

Multiple small alum doses typically cost more than a whole lake alum dose due to higher mobilization costs, and are more appropriate for lakes with high external loading that shortens the longevity of a whole lake alum dose. Multiple small alum doses are sometimes preferred over a large long-term dose for financial reasons or to reduce potential impacts of aluminum toxicity to aquatic organisms. Microfloc alum injection in a lake is more appropriate for smaller lakes with stable thermoclines, and it requires power and continued maintenance. Inflow stream alum injection (described previously) is appropriate for lakes with high external loading from one primary inflow stream.

An alum injection system would involve dosing inflowing streams with alum using a metering system calibrated with stream flow. There are few such systems in place, possibly none in Washington since they are difficult to permit. The initial planning level cost which includes system design plus the infrastructure needed to install the system (building, tanks and pipes and metering system) is estimated at $187,000, which is based on 2016 planning level cost estimates for Spanaway Lake (Brown and Caldwell 2016). There would also be annual costs for alum and energy; based on the cost estimate derived for Spanaway Lake these costs would be over

$30,000 per year. These costs are for one inflow; if both inflows were treated the costs would double.

Because internal loading is a significant source of phosphorus in Lone Lake during the summer algae bloom period, the preferred strategy would be to implement an initial whole lake alum dose to control (inactivate) phosphorus in both deep and shallow sediments, as well as phosphorus present in the water column. This aluminum dose would be applied to the entire lake area excluding shallow areas less than 5 feet deep to avoid nearshore obstructions and sediment disturbance.

Because of toxicity concerns, sodium aluminate is added along with alum to soft water lakes to prevent the pH from dropping below the lower end of the acceptable range (i.e., 6.0) and thereby killing fish from aluminum toxicity. The ratio typically used for alum and sodium aluminate is 2:1 by volume, and this ratio is assumed to be appropriate for Lone Lake. Detailed dosing and cost estimates are provided in Appendix D.

Contractor costs for materials and application (including mobilization and demobilization) have been estimated at $202,000 for a long-term (10-year) whole-lake alum treatment. In addition, there are consultant costs associated with such things as planning, design, monitoring, and reporting, which are estimated at $48,000. Allowing for a 20 percent contingency, the total estimated cost is $290,000.

7.2.2.2. Phoslock Treatment

Phoslock® is the tradename for a product that is a combination of Lanthanum, a natural but rare element in the earth, and bentonite. Because the lanthanum has a strong affinity for phosphate it is able to chemically inactive phosphate through precipitation and forms a mineral of extremely low solubility; thus permanently binding the phosphorus. Unlike alum it is not a coagulant and so it does not trap and remove particles in the water column. In fact, water can be more turbid in the days immediately following an application but decrease with time, as compared to alum which immediately clears the water. Phoslock works mainly in the sediment to bind phosphate that would normally be released to the water through decomposition or changes in sediment chemistry. It binds only to inorganic phosphate and does not address organic phosphorus. Phoslock has no known toxicity and therefore does not have the application concerns that are associated with use of alum. It is also easy to estimate dosage needed; it is based on a 100:1 ratio of Phoslock to potentially available phosphorus. While Phoslock can be applied in frequent small does to ‘strip’ the water column of inorganic phosphorus, for Lone Lake the Phoslock would be added to address sediment derived inorganic phosphorus. One of the key drawbacks to Phoslock is that there are fewer case studies of lake applications to draw from to evaluate effectiveness and duration of treatments.

Phoslock is typically applied as a slurry to the lake surface at a 100:1 ratio of Phoslock to phosphorus. Because it does not address organic phosphorus, it is best applied during winter or early spring when algae concentrations are low and phosphorus is buried in the sediments.

According to their website, Phoslock is typically applied at a rate of 55 to 100 lbs/acre-feet. A planning level cost range for a Phoslock application to Lone Lake is $125,000 to $150,000 per application. This is based on a chemical dose of approximately 54,000 pounds and includes $30,000 to $40,000 in labor for the application. (This cost range is within the range provided by a local applicator that was based on in lake phosphorus concentrations rather than the lake volume.)

Another way to calculate dose is based on the calculated internal phosphorus load of 77 kg/yr. At a ratio of 100:1, this would require 77.000 kg (over 170,000 pounds) of phoslock and cost approximately $272,000 in material plus an estimated additional $60,000 in labor for an estimated application cost of $332,000. Since the sediments are the source of phosphorus that should be targeted by the treatment and to be conservative in the cost estimating, this dose and estimation method have been selected for planning purposes.

Contractor costs associated with monitoring and reporting would be similar to alum, but the design costs would not be the same. Consultant costs for managing the contractor, monitoring effectiveness and reporting are estimated at $30,000. Therefore, an appropriate planning level cost estimate (based on sediment treatment) is $392,000 to $451,000 (allowing for a 15 percent contingency).

Re-applications would be necessary. There are fewer case studies of Phoslock on which to base long term effectiveness, but given the low external loading to Lone Lake it may be similar to what is achieved with alum.

7.2.3. Lake Aeration or Mixing

The key objective of lake aeration or mixing technologies is that the circulating or mixing motion of the water is also circulating and mixing algae cells. This provides an advantage (over natural conditions) to algae such as green algae and diatoms because under natural conditions their time in the sunlit photic zone is determined by their sinking rate, so mixing increases their time in the photic zone. Cyanobacteria have air vacuoles that provide buoyancy and allow them to remain within the photic zone for longer periods of time. Aeration or mixing reduces this advantage, although to do so requires that mixing velocities need to be high enough to overcome the buoyancy. These technologies also introduce oxygen either passively through increased mixing and turbulence of surface waters or more actively through pumping air through the water. These changes on algal community populations and oxygen levels result in other changes in the lake food web.

7.2.3.1. Traditional Aeration

There are three basic approaches to traditional lake aeration; aerating (or oxygenating) the hypolimnion, aerating the epilimnion, and aerating the entire lake (top to bottom). Hypolimnetic aeration (air injection) or oxygenation (oxygen injection) are aimed at keeping the bottom waters well oxygenated and thereby eliminating the release of phosphorus from the sediments.

This technique is considered when sediment derived internal phosphorus loading is the most significant source of phosphorus. The goal of epilimnetic aeration is to circulate only the surface waters enough to ensure that algal cells are moved below a photic zone that is shallower than the thermocline, and thereby decreasing their productivity through physical means. For both of these approaches, it is critical that the thermocline, which serves as a barrier between the epilimnion and hypolimnion, is maintained so that the phosphorus that normally accumulates in the hypolimnion is not mixed with the rest of the lake. Full lake aeration projects are focused on completely mixing the water from top to bottom. They are more appropriate for shallow lakes that are already mixed. Aerating the entire water column may also keep the surface of the sediments oxygenated and reduce release of phosphorus from the sediments which further also helps to control algae through nutrient reduction. Since Lone Lake does not have a hypolimnion, whole lake aeration is the only appropriate option.

There are a number of design considerations related to quantifying the amount of oxygen needed to meet existing and post-project oxygen demand, ensuring that the entire lake is well mixed, and avoiding turbulence at the sediment surface.

There are numerous aeration systems available that for example provide aeration along a linear path, or scattered around the lake, or as one large source near the middle of a lake. Traditional aeration systems require a power source, pump and associated plumping and have long term maintenance and energy costs. Estimated initial year costs for a system (based on inflation adjusted costs as reported in Cooke et al. 2005) are approximately $58,000. To account for engineering design costs ($50,000) and 15 to 20 percent of equipment costs to cover shoreside facilities ($8,700 to $14,000), an appropriate planning level cost range is $117,000 to $122,000. Annual operating costs are estimated at $26,000.

7.2.3.2. SolarBee

The SolarBee is a solar-energy–driven, mixing device that is used to mix either the epilimnion or the entire lake volume. Like other mixing devices it controls algae through mixing them throughout the water column. Although no air is pumped into the water, additional oxygen is added through turbulence and increased contact with air above the lake surface.

There are no significant design costs or issues associated with these; they are modular units that are easily scalable depending upon lake surface area. While SolarBees appear to primarily be used in small lakes and ponds, there have been successful applications in larger lakes, reservoirs and drinking water supplies.

Multiple SolarBee units would be required to treat the lake. Based on preliminary sizing information provided by the manufacturer, the largest unit (i.e., SolarBee SP 10000 LS) treats an area of 30 to 35 acres; therefore three of the units would be needed to treat Lone lake. The units cost $50,000 each or $150,000 for three, which includes installation. A reasonable planning level cost range is $165,000 to $173,000, based on an additional 10 to 15 percent of equipment cost. No shore-based structures would be needed and there would be no long-term energy cost.

Replacement and maintenance costs based on 5 percent of original equipment cost is estimated at $7,500 per year.

7.2.3.3. Nanobubble Aeration or Ozone

Nanobubble aeration uses compressed gas (e.g., air, ozone, carbon dioxide) to produce nanobubbles (bubbles 2,000 times smaller than a grain of salt) to circulate and aerate the water column. The key advantage of using nanobubbles versus traditional aeration technologies is that the very small bubbles move both vertically and horizontally, spreading out evenly and remaining in the water column for long periods of time (versus floating to the surface and dispersing), and therefore this technology greatly increases oxygen transfer. The entire water column is aerated including near the sediment surface, thus reducing phosphorus release from the sediments as well. The high oxygen transfer rate and resultant oxidation (through creation of ozone and other oxidative compounds) has been shown to breakdown algae cells and degrade toxins.

Similar to a Solar Bee, these are modular units that are easily scalable and there are no significant design costs associated with them. However, this is new technology; review of one manufacturer's web brochure indicated that there are approximately 80 systems installed in lakes worldwide.

Installation would require one or more shore-based generators to support multiple aeration units along the lake. (The units can be linked to avoid having multiple shore-based units). There are a number of companies that manufacture nanobubble generating devices. Based on preliminary sizing information provided by the one manufacturer (Molear), approximately two of their largest units (i.e., the Optimus SP 1000 which treats approximately 250 acre-feet) would be required at approximately $89,000 each, the equipment cost would be $178,000. There would also be one-time costs for shipping and installation. A reasonable planning level cost range (including 15 to 20 percent of equipment cost) for this particular manufacturer is estimated at $204,000 to $214,000. There would also be long term energy costs associated with this technology and costs associated with maintenance and replacement. Based on 5 percent of the original equipment costs, an annual cost of $8,900 should be assumed.

7.2.4. Ultrasound (LG Sonic)

The LG Sonic technology uses ultrasonic sound waves that create a sound barrier in the top layer of water that prevents algae from rising into the photic zone. Ultrasound does not directly kill algae cells and therefore does not contribute to increased lysing of cells and release of toxins. (There are different manufacturers of this technology, this summary was developed from materials provided by LG Sonic which appears to be the most used in lakes or reservoirs.) The LG Sonic system uses real-time monitoring to adapt the ultrasound to the optimal frequency for the algae in the lake at the time.

For Lone Lake the appropriate LG Sonic system would be an MPC Buoy system. This is a floating, solar-powered system that emits ultrasonic waves while also collecting water quality data including chlorophyll, phycocyanin (blue-green pigment), pH, turbidity, dissolved oxygen, and temperature. Based on the received data, the transmitters are activated or optimized to transmit specific ultrasonic parameters aimed at different algae types. Because the system is automated and connected to many others in the world, the algorithms it uses for algae control are continually changing and improving. The data collected and remote sensing capabilities of the units allow creation of algal distribution maps and other data summaries.

These are modular units that are easily scalable and there are no significant design costs associated with them. However, this is new technology; LG Sonics' web brochure indicates that they are being used in 20 different countries and there are numbers of case studies of successful use in reservoirs to control algae and other drinking water problems.

Each MPC-Buoy can control algae in an area of up to 1,600 feet in diameter. The manufacturer recommended a three-unit system for Lone Lake at a cost of approximately $170,000 for the system. This includes delivery, installation, buoy and anchoring system, and 1 year of monitoring. An appropriate planning level cost range is $187,000 to $196,000, which includes 10 to 15 percent of equipment cost. An annual cost of $10,800 would accrue for all following years to cover monitoring, connection services and recalibration of water quality sensors.

7.2.5. Floating Wetlands

Floating wetlands improve water quality in lakes by taking nutrients from the water that otherwise would be taken up by cyanobacteria and other phytoplankton. The principal mechanism for nutrient removal is by the biofilm growing on plant roots descending into the water from the constructed floating wetland matrix. The biofilm is composed of attached algae, bacteria, and fungi within a gelatinous matrix. In addition to dissolved nutrient uptake by the biofilm microbes, dissolved nutrients are taken up by the vascular plants themselves and the biofilm within the floating matrix, and suspended solids are adsorbed to biofilm on the plant roots. Nutrient uptake primarily occurs during the warm summer months and the biofilm ultimately sloughs off and becomes lake sediment.

The amount of nutrient removal is highly variable but generally increases directly with the wetland area, plant root surface area, water nutrient concentrations, water temperature, and dissolved oxygen concentrations (Pavlineri et al. 2017; Wang et al. 2019). A review of floating wetland function in stormwater ponds indicates that a 50 percent cover by floating wetlands reduces total phosphorus concentrations by about 50 percent and reductions decrease with increasing water depth and hydraulic loading rate (Pavlineri et al. 2017). A review of floating wetland function in eutrophic waters found an average phosphorus removal rate of 51 ±20 percent, and recommended designs covering 5 to 38 percent of the water at depths ranging from 2 to 4 feet (Wang et al. 2019).

Floating wetlands provide secondary benefits of aesthetic value and habitat for fish and wildlife. Insects graze on the biofilm; small fish feed on the insects; and the cover protects small fish from predators. Floating wetlands can be designed for waterfowl breeding habitat or can be fenced to protect new plants from waterfowl grazing.

Floating wetlands can be planted with a variety of native flowering plants, emergent plants, shrubs, and trees. Floating wetlands are easily anchored in place and should last for more than 20 years. Commercial manufacturers include Floating Islands International and Biomatrix Water, among others. Floating Islands International uses a recycled plastic matrix with polyurethane for floatation. Biomatrix Water uses a natural coir fiber matrix with recycled HDPE tubes for floatation.

Floating Islands International recommends covering at least a 2 percent cover of a lake to improve water quality. Floating wetlands cost approximately $40 per square foot (G. Fulford, Biomatrix Water, personal communication) and can be planted and installed by volunteers. The cost for floating wetlands in Lone Lake is estimated to be approximately $3.5 million based on this rate and covering 2 acres (2 percent) of Lone Lake.

7.2.6. Algaecides

Algaecides provide partial short-term algae control by killing the algae and cyanobacteria in the water column. However, all algaecides also affect other aquatic biota to varying degrees and accelerate recycling of nutrients. Algaecides are effective only while the active ingredient is in the water column and available for uptake by the algae (Cooke et al. 2005). Typically, several applications must occur within the same season to provide effective control of algae and blue-green bacteria. Algaecides do not reduce phosphorus or nitrogen concentrations and do not provide long-term control. In fact, they increase recycling of phosphorus. Currently, endothall (Hydrothol® 191) and sodium carbonate peroxyhydrate (PAK 27) are the only algaecides that can be used in the State of Washington.

Hydrothol has some use restrictions related to drinking water and toxicity to fish. The cost is approximately $250 per acre for the material and application with equates to $25,000 for Lone Lake. This plus an additional $500 is estimated for permit compliance and postings resulting in a planning level cost of $26,000 to $28,000 per application.

PAK 27 has no fishing, drinking, swimming or irrigation use restrictions. The cost is approximately $300 per acre for the material and application with equates to $30,000 for Lone Lake. This plus an additional $500 to cover permit compliance and postings results in a planning level cost of $30,500 to $32,500 per application.

If algaecides were to be used in Lone Lake, it would likely require a minimum of two treatments every summer.

7.2.7. In-lake Methods Not Addressed

There are a number of other in-lake methods for controlling algae that were not addressed because they are considered inappropriate or infeasible:

- Biomanipulation: Manipulating the food web (e.g., adding zooplankton eating fish to decrease their predation on good algae). These projects are always considered experimental because of the difficulty in predicting or controlling results. Grass carp stocking in Lone Lake is an excellent example.

- Dye: Coloring the lake with dye to decrease sunlight available for algae growth. Largely untested and likely very difficult to permit in natural lakes.

- Barley Straw: A sediment amendment thought to produce a chemical that inhibits algae growth when exposed to sunlight and in the presence of oxygen and/or favors beneficial bacteria and fungi growth over algae growth. Mechanism is poorly understood. Largely untested. Very difficult for a lakewide application.

- Dilution/Flushing: Use of a low phosphorus water supply to both dilute phosphorus and increase flushing. No nearby water supply exists.

- Hypolimnetic Oxygenation: Oxygenating the sediments to control phosphorus release from the sediments. The lack of a hypolimnion in Lone Lake makes this inappropriate.

- Hypolimnetic Withdrawal: Withdrawing water from the hypolimnion to remove phosphorus laden water. The lack of a hypolimnion in Lone Lake makes this inappropriate.

- Dredging: Removing sediment from the lake to remove the phosphorus source and increase lake depth. Difficult to permit and prohibitively expensive (multiple millions).

8. RECOMMENDED MANAGEMENT ACTIONS

Long-term reduction in the duration and frequency of toxigenic algae blooms in Lone Lake and maintenance of a healthy fishery will require a sustained, cooperative private and public effort to reduce input of legacy nutrients from the sediments in the lake and to minimize nutrient sources in the watershed. Of the feasible alternatives described above, the following in-lake management techniques are recommended for further consideration and design to meet water quality objectives (not in order of preference):

- Alum is the preferred choice for phosphorus inactivation, primarily due to the large case history of successful applications. It will result in an immediate improvement in lake quality and should be long lasting. It is also likely to provide an immediate benefit to aquatic plants by increasing light penetration. In some lakes with invasive plant species this might be a concern, but in Lone Lake it is less of a concern as long as invasive plant species are not present.

- The nanobubble and LG sonic solutions are both worth considering. While both are relatively experimental, their potential for algae control appears to be greater than traditional aeration or mechanical mixing (Solar Bee®).

In addition to in-lake actions the following are also recommended:

- A thorough assessment of watershed sources of nutrients should be undertaken. This might include microbial source tracking of farm animal and human fecal sources, septic system function assessments, and stream assessments to identify nutrient sources and illicit discharges. The significant increases in stream phosphorus concentrations measured during this study as compared to past studies is of concern and deserves careful attention.

- Aquatic macrophyte management to continue to promote the return of a diverse aquatic macrophyte community and to ensure that invasive plants are removed or controlled before they severely impact the lake ecosystem. This will require regular surveys of the aquatic macrophyte community to identify the presence of an invasive species and to track the abundance and diversity of the native aquatic macrophyte community.

- A long-term lake water quality monitoring program to track the lake’s response to management techniques and inform future decisions.

One option that should be considered is to wait to see what happens with water quality in the lake as the native aquatic plants gradually reestablish. Increased density and coverage by native submersed plants may have a significant positive effect on the lake fishery and water quality over time. This no-action alternative would allow for nature to take its course, and, given the high costs of the active management options, may be the most likely to happen.

The next step for this Plan is to gather input from the public and select a preferred alternative management scenario consisting of one or more management techniques, and identify funding sources. An engineering report or implementation plan would then be developed to refine the alternative selected based on the available funds.

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APPENDIX A

Water Quality Monitoring Data

Study_Specific_Location_ID	Field_Collection_Type	Field_Collection_Start_Date	Field_Collection_Start_Time	Sample_ID	Storm_Event_Qualifier	Result_Parameter_Name	Lab_Analysis_Date	Result_Value	Result_Value_Units
Station1	Measurement	3/21/2019				Dissolved Oxygen		11.27	mg/L
Station1	Measurement	4/30/2019				Dissolved Oxygen		10.73	mg/L
Station1	Measurement	5/30/2019				Dissolved Oxygen		10.4	mg/L
Station1	Measurement	6/13/2019				Dissolved Oxygen		9.07	mg/L
Station1	Measurement	6/27/2019				Dissolved Oxygen		9.22	mg/L
Station1	Measurement	9/27/2019				Dissolved Oxygen		11.95	mg/L
Station1	Measurement	10/17/2019				Dissolved Oxygen		11.15	mg/L
Station1	Measurement	11/21/2019				Dissolved Oxygen		12.8	mg/L
Station1	Measurement	12/17/2019				Dissolved Oxygen		12.86	mg/L
Station2	Measurement	3/21/2019				Dissolved Oxygen		11.79	mg/L
Station2	Measurement	4/30/2019				Dissolved Oxygen		11.18	mg/L
Station2	Measurement	5/30/2019				Dissolved Oxygen		10.46	mg/L
Station2	Measurement	6/27/2019				Dissolved Oxygen		11.25	mg/L
Station2	Measurement	9/27/2019				Dissolved Oxygen		11.51	mg/L
Station2	Measurement	10/17/2019				Dissolved Oxygen		10.79	mg/L
Station2	Measurement	11/21/2019				Dissolved Oxygen		12.7	mg/L
Station2	Measurement	12/17/2019				Dissolved Oxygen		12.45	mg/L
Station1	Measurement	3/21/2019				pH		8.18	pH
Station1	Measurement	4/30/2019				pH		7.99	pH
Station1	Measurement	5/30/2019				pH		8.26	pH
Station1	Measurement	6/13/2019				pH		7.92	pH
Station1	Measurement	6/27/2019				pH		6.91	pH
Station1	Measurement	9/27/2019				pH		7.63	pH
Station1	Measurement	10/17/2019				pH		7.58	pH
Station1	Measurement	11/21/2019				pH		7.69	pH
Station1	Measurement	12/17/2019				pH		7.47	pH
Station2	Measurement	3/21/2019				pH		7.88	pH
Station2	Measurement	4/30/2019				pH		7.84	pH
Station2	Measurement	5/30/2019				pH		8.9	pH
Station2	Measurement	6/27/2019				pH		7.98	pH
Station2	Measurement	9/27/2019				pH		7.73	pH
Station2	Measurement	10/17/2019				pH		7.7	pH
Station2	Measurement	11/21/2019				pH		8.01	pH
Station2	Measurement	12/17/2019				pH		7.6	pH
Station1	Measurement	3/21/2019				Salinity		0.1	ppt
Station1	Measurement	4/30/2019				Salinity		0.1	ppt
Station1	Measurement	5/30/2019				Salinity		0.1	ppt
Station1	Measurement	6/13/2019				Salinity		0.1	ppt
Station1	Measurement	6/27/2019				Salinity		0.1	ppt
Station1	Measurement	9/27/2019				Salinity		0.1	ppt
Station1	Measurement	10/17/2019				Salinity		0.17	ppt
Station1	Measurement	11/21/2019				Salinity		0.16	ppt
Station1	Measurement	12/17/2019				Salinity		0.1	ppt
Station2	Measurement	3/21/2019				Salinity		0.09	ppt
Station2	Measurement	4/30/2019				Salinity		0.1	ppt
Station2	Measurement	5/30/2019				Salinity		0.1	ppt
Station2	Measurement	6/27/2019				Salinity		0.1	ppt
Station2	Measurement	9/27/2019				Salinity		0.1	ppt

Study_Specific_Location_ID	Field_Collection_Type	Field_Collection_Start_Date	Field_Collection_Start_Time	Sample_ID	Storm_Event_Qualifier	Result_Parameter_Name	Lab_Analysis_Date	Result_Value	Result_Value_Units
Station2	Measurement	10/17/2019				Salinity		0.16	ppt
Station2	Measurement	11/21/2019				Salinity		0.16	ppt
Station2	Measurement	12/17/2019				Salinity		0.1	ppt
Station1	Measurement	3/21/2019				Specific Conductivity (at 25 deg C)		219.2	uS/cm
Station1	Measurement	4/30/2019				Specific Conductivity (at 25 deg C)		225.4	uS/cm
Station1	Measurement	5/30/2019				Specific Conductivity (at 25 deg C)		229.7	uS/cm
Station1	Measurement	6/13/2019				Specific Conductivity (at 25 deg C)		262.7	uS/cm
Station1	Measurement	6/27/2019				Specific Conductivity (at 25 deg C)		254.8	uS/cm
Station1	Measurement	9/27/2019				Specific Conductivity (at 25 deg C)		234.2	uS/cm
Station1	Measurement	10/17/2019				Specific Conductivity (at 25 deg C)		361.3	uS/cm
Station1	Measurement	11/21/2019				Specific Conductivity (at 25 deg C)		343.4	uS/cm
Station1	Measurement	12/17/2019				Specific Conductivity (at 25 deg C)		233	uS/cm
Station2	Measurement	3/21/2019				Specific Conductivity (at 25 deg C)		124.7	uS/cm
Station2	Measurement	4/30/2019				Specific Conductivity (at 25 deg C)		206.4	uS/cm
Station2	Measurement	5/30/2019				Specific Conductivity (at 25 deg C)		212.8	uS/cm
Station2	Measurement	6/27/2019				Specific Conductivity (at 25 deg C)		6062.5	uS/cm
Station2	Measurement	9/27/2019				Specific Conductivity (at 25 deg C)		217.4	uS/cm
Station2	Measurement	10/17/2019				Specific Conductivity (at 25 deg C)		341.5	uS/cm
Station2	Measurement	11/21/2019				Specific Conductivity (at 25 deg C)		329.5	uS/cm
Station2	Measurement	12/17/2019				Specific Conductivity (at 25 deg C)		227.2	uS/cm
Station1	Measurement	3/21/2019				Temperature, water		6.964	deg C
Station1	Measurement	4/30/2019				Temperature, water		9.303	deg C
Station1	Measurement	5/30/2019				Temperature, water		14.652	deg C
Station1	Measurement	6/13/2019				Temperature, water		14.821	deg C
Station1	Measurement	6/27/2019				Temperature, water		14.668	deg C
Station1	Measurement	9/27/2019				Temperature, water		13.134	deg C
Station1	Measurement	10/17/2019				Temperature, water		10.297	deg C
Station1	Measurement	11/21/2019				Temperature, water		5.175	deg C
Station1	Measurement	12/17/2019				Temperature, water		5.118	deg C
Station2	Measurement	3/21/2019				Temperature, water		6.036	deg C
Station2	Measurement	4/30/2019				Temperature, water		9.285	deg C
Station2	Measurement	5/30/2019				Temperature, water		14.662	deg C
Station2	Measurement	6/27/2019				Temperature, water		14.172	deg C
Station2	Measurement	9/27/2019				Temperature, water		12.739	deg C
Station2	Measurement	10/17/2019				Temperature, water		10.498	deg C
Station2	Measurement	11/21/2019				Temperature, water		3.658	deg C
Station2	Measurement	12/17/2019				Temperature, water		5.098	deg C
Station1	sample	3/21/2019	9:19	Station1-032119		Total Phosphorus	4/7/2019	0.045	mg/L
Station1	sample	4/30/2019	10:12	Station1-043019		Total Phosphorus	5/6/2019	0.06	mg/L
Station1	sample	5/30/2019	10:53	Station1-053019		Total Phosphorus	6/4/2019	0.157	mg/L
Station1	sample	6/13/2019	11:10	Station1-061319		Total Phosphorus	6/25/2019	0.259	mg/L
Station1	sample	6/27/2019	11:05	Station1-062719		Total Phosphorus	7/9/2019	0.11	mg/L
Station1	sample	6/27/2019	11:06	Station1-062719-d		Total Phosphorus	7/9/2019	0.075	mg/L
Station1	sample	9/27/2019	12:13	Station1-092719		Total Phosphorus	10/8/2019	0.398	mg/L
Station1	sample	10/17/2019	10:12	Station1-101719		Total Phosphorus	10/29/2019	0.056	mg/L
Station1	sample	11/21/2019		Station1-112119		Total Phosphorus			mg/L
Station1	sample	12/17/2019	10:30	Station1-121719		Total Phosphorus	12/30/2019	0.051	mg/L
Station1	sample	1/30/2020	11:09	Station1-013020		Total Phosphorus	2/5/2020	0.062	mg/L

Study_Specific_Location_ID	Field_Collection_Type	Field_Collection_Start_Date	Field_Collection_Start_Time	Sample_ID	Storm_Event_Qualifier	Result_Parameter_Name	Lab_Analysis_Date	Result_Value	Result_Value_Units
Station2	sample	3/21/2019	9:36	Station2-032119		Total Phosphorus	4/14/2019	0.076	mg/L
Station2	sample	4/30/2019	10:53	Station2-043019		Total Phosphorus	5/6/2019	0.124	mg/L
Station2	sample	5/30/2019	10:32	Station2-053019		Total Phosphorus	6/4/2019	0.309	mg/L
Station2	sample	6/13/2019	11:06	Station2-061319		Total Phosphorus	6/25/2019	1.05	mg/L
Station2	sample	6/27/2019	10:36	Station2-062719		Total Phosphorus	7/9/2019	0.09	mg/L
Station2	sample	9/27/2019	11:39	Station2-092719		Total Phosphorus	10/8/2019	0.064	mg/L
Station2	sample	10/17/2019	9:52	Station2-101719		Total Phosphorus	10/29/2019	0.105	mg/L
Station2	sample	11/21/2019		Station2-112119		Total Phosphorus			mg/L
Station2	sample	12/17/2019	10:07	Station2-121719		Total Phosphorus	12/30/2019	0.055	mg/L
Station2	sample	1/30/2020	10:30	Station2-013020		Total Phosphorus	2/5/2020	0.088	mg/L
Station1	Measurement	3/21/2019				Turbidity		1.46	NTU
Station1	Measurement	4/30/2019				Turbidity		10.87	NTU
Station1	Measurement	5/30/2019				Turbidity		1.13	NTU
Station1	Measurement	6/13/2019				Turbidity		0.32	NTU
Station1	Measurement	6/27/2019				Turbidity		0.49	NTU
Station1	Measurement	9/27/2019				Turbidity		27.61	NTU
Station1	Measurement	10/17/2019				Turbidity		0.51	NTU
Station1	Measurement	11/21/2019				Turbidity		0.2	NTU
Station1	Measurement	12/17/2019				Turbidity		0	NTU
Station2	Measurement	3/21/2019				Turbidity		8.68	NTU
Station2	Measurement	4/30/2019				Turbidity		8.18	NTU
Station2	Measurement	5/30/2019				Turbidity		7.63	NTU
Station2	Measurement	6/27/2019				Turbidity		1.74	NTU
Station2	Measurement	9/27/2019				Turbidity		14.78	NTU
Station2	Measurement	10/17/2019				Turbidity		1.68	NTU
Station2	Measurement	11/21/2019				Turbidity		3.96	NTU
Station2	Measurement	12/17/2019				Turbidity		5.08	NTU

Date	Sample	Data Type					Data Type				
	Location	Sample	Cond	Secchi	Water	Dissolved	Percent	pH	Air	Wind	Lake
	(ID Code)	Depth	uS/cm	Disk (m)	Temp oC	Oxygen	DO		Temp	Speed	Stage
		(m)				(mg/L)			(°C)	(km/h)	(m)
3/21/2019	midlake	0.5	166.6		10.6	12.6	113.1	8.1			
3/21/2019	midlake	1	166.6		10.5	12.6	113.5	7.9			
3/21/2019	midlake	2	166.6		10.4	12.7	114	7.8			
3/21/2019	midlake	3	166.5		9.1	12.8	112.8	7.8			
3/21/2019	midlake	4	166.9		7.1	11.8	97.7	7.7			
3/21/2019	midlake	4.5	167.0		6.6	11.7	95.1	7.7			
3/21/2019	midlake	2	166.5		10.3	12.7	113.3	7.7			
3/21/2019	midlake			1.33					14.4	light	
3/21/2019	midlake			1.33							
RPD			**0.08**	**0.00**	**0.97**	**0.00**	**0.62**	**1.29**			
4/16/2019	midlake	0.5	164.4		12.7	11.2	105.5	6.2			
4/16/2019	midlake	1	164.4		12.6	11.2	105.3	6.3			
4/16/2019	midlake	2	164.4		12.6	11.1	104.3	6.5			
4/16/2019	midlake	3	164.5		12.5	11	102.7	6.6			
4/16/2019	midlake	4	164.6		12	9.5	87.8	6.8			
4/16/2019	midlake	2	164.5		12.6	11	103.5	6.9			
4/16/2019	midlake			1.5					11.5	gusty	2.25
4/16/2019	midlake			1.6							
RPD			**0.08**	**6.45**	**0.00**	**0.90**	**0.77**	**5.97**			
5/7/2019	midlake	0.5	171.3		17.6	10.2	106.8	8			
5/7/2019	midlake	1	171.1		17.5	10.2	107.2	7.9			
5/7/2019	midlake	2	170.8		16.9	10	103	7.8			
5/7/2019	midlake	3	170.6		15.9	9.6	97.5	7.7			
5/7/2019	midlake	4	173.3		15.2	7	69.8	7.5			
5/7/2019	midlake	4.5	174.9		15	5.9	58.5	7.3			
5/7/2019	midlake	2	171.0		16.8	9.8	100.8	7.5			
5/7/2019	midlake			3.18					-	light	1.98
5/7/2019	midlake			3.18							
RPD			**0.08**	**0.00**	**0.59**	**2.02**	**2.16**	**3.92**			

Date	Sample	Data Type					Data Type				
	Location	Sample	Cond	Secchi	Water	Dissolved	Percent	pH	Air	Wind	Lake
	(ID Code)	Depth	uS/cm	Disk (m)	Temp oC	Oxygen	DO		Temp	Speed	Stage
		(m)				(mg/L)			(°C)	(km/h)	(m)
5/23/2019	midlake	0.5	172.6		19.6	9.4	102.5	7.8			
5/23/2019	midlake	1	172.7		19.5	9.4	102.2	7.7			
5/23/2019	midlake	2	172.0		18.4	8.8	93.7	7.5			
5/23/2019	midlake	3	172.5		18.2	7.8	82.2	7.3			
5/23/2019	midlake	4	175.1		17.8	4.9	52.1	7.1			
5/23/2019	midlake	4.5	177.3		17.7	2.9	30.7	7			
5/23/2019	midlake	2	172.2		18.4	8.6	91.2	7.1			
5/23/2019	midlake			2.51					18.4	light	-
5/23/2019	midlake			2.55							
RPD			**0.08**	**1.58**	**0.00**	**2.30**	**2.70**	**5.48**			
6/12/2019	midlake	0.5	177.4		21.8	11	125.5	8.4			
6/12/2019	midlake	1	177.3		21.5	11.1	125.9	8.4			
6/12/2019	midlake	2	177.4		20.4	10.3	114.1	8.2			
6/12/2019	midlake	3	177.1		19.6	9.8	107.8	8			
6/12/2019	midlake	4	181.1		19.1	4.2	46	7.6			
6/12/2019	midlake	2	176.9		20.5	10.3	114.6	8			
6/12/2019	midlake			1.6					20.5	light	1.56
6/12/2019	midlake			1.56							
RPD			**0.32**	**2.53**	**0.49**	**0.00**	**0.44**	**2.47**			
6/25/2019	midlake	0.5	180.6		20.2	8.8	97.3	7.7			
6/25/2019	midlake	1	180.3		20	8.8	96.9	7.6			
6/25/2019	midlake	2	180.6		19.8	8.4	92.2	7.5			
6/25/2019	midlake	3	180.7		19.7	6.4	70.3	7.4			
6/25/2019	midlake	4	183.0		19.6	6.3	69	7.3			
6/25/2019	midlake	5	183.4		19.5	4.9	53.1	7.2			
6/25/2019	midlake	2	180.4		19.8	8.3	90.8	7.3			
6/25/2019	midlake			2.65					-	light	1.32
6/25/2019	midlake			2.7							

Date	Sample	Data Type					Data Type				
	Location	Sample	Cond	Secchi	Water	Dissolved	Percent	pH	Air	Wind	Lake
	(ID Code)	Depth	uS/cm	Disk (m)	Temp oC	Oxygen	DO		Temp	Speed	Stage
		(m)				(mg/L)			(°C)	(km/h)	(m)
RPD			**0.11**	**1.87**	**0.00**	**1.20**	**1.53**	**2.70**			
7/9/2019	midlake	0.5	189.5		21.9	8.4	96.2	7.8			
7/9/2019	midlake	1	189.6		21.7	8.3	95.1	7.7			
7/9/2019	midlake	2	189.4		21.6	8.1	92.4	7.6			
7/9/2019	midlake	3	189.9		21.2	6	67.5	7.4			
7/9/2019	midlake	4	193.2		20.9	3.2	35.8	7.2			
7/9/2019	midlake	4.5	197.3		20.7	0.8	0.2	7			
7/9/2019	midlake	2	189.3		21.6	8	91.2	7.3			
7/9/2019	midlake			2.58					20.8	light	1.16
7/9/2019	midlake			2.85							
RPD			**0.05**	**9.77**	**0.00**	**1.24**	**1.31**	**4.03**			
7/23/2019	midlake	0.5	188		22.4	11.5	132.1	8.4			
7/23/2019	midlake	1	188.1		22.4	11.4	132	8.4			
7/23/2019	midlake	2	188.2		22.4	10.9	125.5	8.4			
7/23/2019	midlake	3	189.8		21.9	6.4	73.5	7.7			
7/23/2019	midlake	4	193.6		21.3	2.8	31.6	7.4			
7/23/2019	midlake	2	188.2		22.4	10.6	122.2	8.3			
7/23/2019	midlake			1.15					20.3	light	1
7/23/2019	midlake			1.16							
RPD			**0.00**	**0.87**	**0.00**	**2.79**	**2.66**	**1.20**			
8/13/2019	midlake	0.5	174.7		22.7	14.25	165.4	9.2	22.4	calm	0.68
8/13/2019	midlake	1	173.5		22.4	12.47	144.7	9.1			
8/13/2019	midlake	2	173		22.1	8.22	94.5	8.9			
8/13/2019	midlake	3	173		22	7.44	85.2	8.7			
8/13/2019	midlake	4	178.6		21.8	3.12	35.5	8.2			
8/13/2019	midlake	2	172.8		22.1	7.8	89.4	8.7			
8/13/2019	midlake			0.49							
8/13/2019	midlake			0.5							

Date	Sample	Data Type					Data Type				
	Location	Sample	Cond	Secchi	Water	Dissolved	Percent	pH	Air	Wind	Lake
	(ID Code)	Depth	uS/cm	Disk (m)	Temp oC	Oxygen	DO		Temp	Speed	Stage
		(m)				(mg/L)			(°C)	(km/h)	(m)
RPD			**0.12**	**2.02**	**0.00**	**5.24**	**5.55**	**2.27**			
8/27/2019	midlake	0.5	176.9		20.9	9.2	103.8	8.6			
8/27/2019	midlake	1	176.9		20.9	8.9	99.4	8.6			
8/27/2019	midlake	2	177.1		20.9	8.4	94.3	8.6			
8/27/2019	midlake	3	177.5		20.7	7.2	79.8	8.5			
8/27/2019	midlake	4	179.4		20.5	3.3	36.9	8.1			
8/27/2019	midlake	2	177.1		20.8	8.2	91.7	8.6			
8/27/2019	midlake			0.7					21.5	light	0.58
8/27/2019	midlake			0.68							
RPD			**0.00**	**2.90**	**0.48**	**2.41**	**2.80**	**0.00**			
9/10/2019	midlake	0.5	172.8		20.9	6.6*	73.8*	6.3*			
9/10/2019	midlake	1	172.8		20.9	6.3*	71*	6.2*			
9/10/2019	midlake	2	172.8		20.9	6.2*	69.6*	6.1*			
9/10/2019	midlake	3	172.8		20.8	6*	67.4*	6.1*			
9/10/2019	midlake	4	173.6		20.7	5.5*	61.4*	6.2*			
9/10/2019	midlake	2	172.7		20.9	6.2*	68.8*	6.1*			
9/10/2019	midlake			0.8					19.2	calm	0.62
9/10/2019	midlake			0.87							
RPD			**0.06**	**8.38**	**0.00**	**#VALUE!**	**#VALUE!**	**#VALUE!**			
9/27/2019	midlake	0.5	180.1		17.8	7.08*	74.6*	7.73*			
9/27/2019	midlake	1	180.1		17.8	7.05*	74.2*	7.72*			
9/27/2019	midlake	2	180.1		17.7	6.91*	72.7*	7.7*			
9/27/2019	midlake	3	180.7		17.7	6.29*	66.1*	7.62*			
9/27/2019	midlake	4	181		17.6	5.95*	62.5*	7.57*			
9/27/2019	midlake	4.5	181.4		17.6	5.67*	59.4*	7.49*			
9/27/2019	midlake	2	180.3		17.7	6.67*	70.1*	7.69*			
9/27/2019	midlake	0.5	180.2		17.8	6.99*	73.6*	7.74*			
9/27/2019	midlake			1.26					16	light	0.7

Date	Sample Location (ID Code)	Data Type Sample Depth (m)	Cond uS/cm	Secchi Disk (m)	Water Temp oC	Data Type Dissolved Oxygen (mg/L)	Percent DO	pH	Air Temp (°C)	Wind Speed (km/h)	Lake Stage (m)
9/27/2019	midlake			1.26							
RPD			**0.11**	**0.00**	**0.00**	**#VALUE!**	**#VALUE!**	**#VALUE!**			
10/15/2019	midiake	0.5	181.6		12.9	11.61	110.1	8.21			
10/15/2019	midiake	1	181.9		12.9	11.57	109.6	8.2			
10/15/2019	midiake	2	181.9		12.8	10.91	103.2	8.02			
10/15/2019	midiake	3	182.2		12.7	10.3	97.2	7.83			
10/15/2019	midiake	4	182.2		12.7	7.73	73	7.43			
10/15/2019	midiake	2	182.1		12.8	11.06	104.5	8.05			
10/15/2019	midiake			1.62					13.6	calm	0.7
10/15/2019	midiake			1.59							
RPD			**0.11**	**1.87**	**0.00**	**1.37**	**1.25**	**0.37**			
10/29/2019	midlake	0.5	170		10.4	13.22	118.1	8.96			
10/29/2019	midlake	1	170		10.4	13.25	118.5	8.96			
10/29/2019	midlake	2	170		10.4	13.24	118.4	8.96			
10/29/2019	midlake	3	170		10.3	13.21	118.1	8.96			
10/29/2019	midlake	4	170.1		10.3	13.17	117.7	8.96			
10/29/2019	midlake	2	170		10.4	13.22	118.1	8.97			
10/29/2019	midlake			2.7					8.9	calm	0.85
10/29/2019	midlake			2.72							
RPD			**0.00**	**0.74**	**0.00**	**0.15**	**0.25**	**0.11**			
11/11//19	midlake	0.5	157		9.8	11.19	98.6	8.24			
11/11//19	midlake	1	156.1		9.6	11.12	97.5	8.15			
11/11//19	midlake	2	155.7		9.5	11.07	97	8.11			
11/11//19	midlake	3	155.8		9.3	10.71	93.3	8			
11/11//19	midlake	4	156.1		9.2	9.25	80.6	7.7			
11/11//19	midlake	2	155.7		9.5	11.06	96.9	8.14			
11/11//19	midlake			3.58					17.2	calm	0.88
11/11//19	midlake			3.53							

Date	Sample	Data Type					Data Type				
	Location	Sample	Cond	Secchi	Water	Dissolved	Percent	pH	Air	Wind	Lake
	(ID Code)	Depth	uS/cm	Disk (m)	Temp oC	Oxygen	DO		Temp	Speed	Stage
		(m)				(mg/L)			(°C)	(km/h)	(m)
RPD			**0.00**	**1.41**	**0.00**	**0.09**	**0.10**	**0.37**			
12/17/2019	midlake										
12/17/2019	midlake										
12/17/2019	midlake	0.5	172.8		6.2	10.52	84.8	7.48			
12/17/2019	midlake	1	172.7		61	10.49	84.6	7.42			
12/17/2019	midlake	2	172.8		6.1	10.46	84.4	7.4			
12/17/2019	midlake	3	172.8		6.1	10.43	84.1	7.39			
12/17/2019	midlake	4	172.9		6.1	10.38	83.7	7.38			
12/17/2019	midlake	2	172.9		6.1	10.43	84.1	7.38			
12/17/2019	midlake			1.99					10.6	calm	1.24
12/17/2019	midlake			2.1							
RPD			**0.06**	**5.38**	**0.00**	**0.29**	**0.36**	**0.27**			
1/9/2020	midlake	0.5	163.3		6.7	11.83	96.9	8.08			
1/9/2020	midlake	1	163.7		6.7	11.82	96.6	7.97			
1/9/2020	midlake	2	163.8		6.7	11.74	96	7.94			
1/9/2020	midlake	3	163.8		6.7	11.7	95.6	7.94			
1/9/2020	midlake	4	163.7		6.7	11.68	95.5	7.94			
1/9/2020	midlake	4.5	164		6.6	11.59	94.7	7.94			
1/9/2020	midlake	2	163.7		6.7	11.72	95.8	7.9			
1/9/2020	midlake			1.82					3.8	calm	1.69
1/9/2020	midlake			1.83							
RPD			**0.06**	**0.55**	**0.00**	**0.17**	**0.21**	**0.51**			
2/11/2020	midlake	0.5	159.4		6.8	11.38	93.5	7.75			
2/11/2020	midlake	1	159.6		6.8	11.43	93.6	7.68			
2/11/2020	midlake	2	159.6		6.8	11.4	93.4	7.64			
2/11/2020	midlake	3	159.6		6.7	11.38	93.2	7.63			
2/11/2020	midlake	4	159.6		6.7	11.28	92.3	7.6			
2/11/2020	midlake	5	159.7		6.7	11.11	90.6	7.57			

Date	Sample	Data Type					Data Type				
	Location	Sample	Cond	Secchi	Water	Dissolved	Percent	pH	Air	Wind	Lake
	(ID Code)	Depth	uS/cm	Disk (m)	Temp oC	Oxygen	DO		Temp	Speed	Stage
		(m)				(mg/L)			(°C)	(km/h)	(m)
2/11/2020	midlake	2	159.6		6.7	11.37	93.1	7.61			
2/11/2020	midlake			1.9						light	2.75
2/11/2020	midlake			1.9							
RPD			**0.00**	**0.00**	**1.48**	**0.26**	**0.32**	**0.39**			

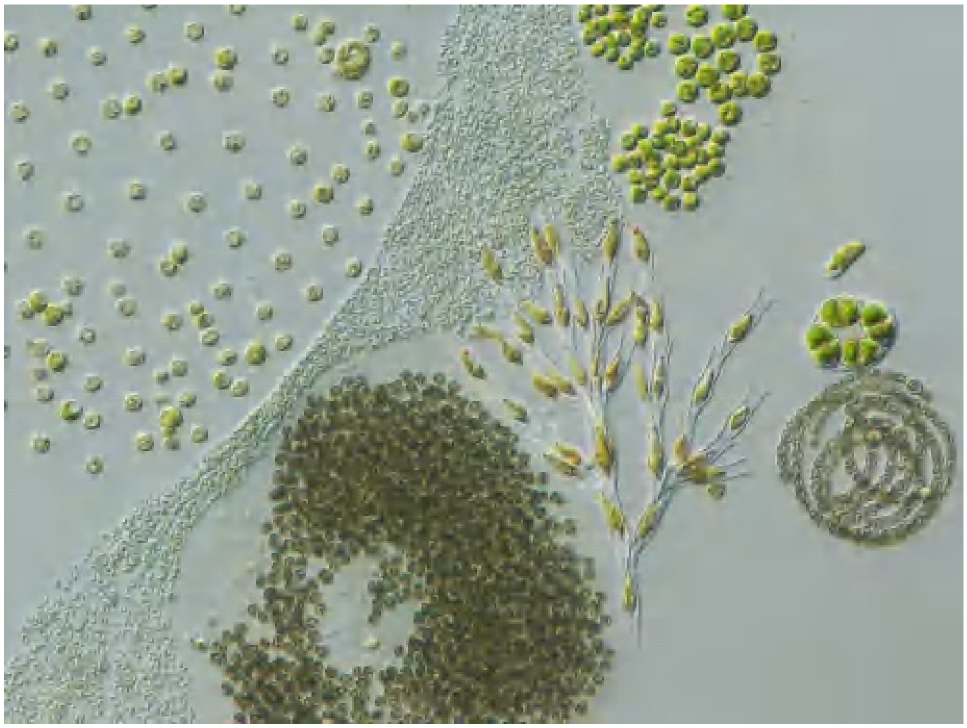
Temp decline > 1 degree/m

DO < 2 mg/L

	NO3	NH4	T NO3 NO2	SRP	TP	TKN	TN	TN/TP	chla	phaeo	Chla-phaeo (mg/m3)
3/21/2019		nd	nd	0.01	0.04	0.78	0.78	19.5	6.1	0.6	5.5
4/16/2019		0.03	nd	nd	0.037	0.82	0.82	22.2	15	3.3	11.7
5/7/2019		-	nd	-	0.023	-	-	-	6.12	nd	6.12
5/23/2019		-	nd	0.02	0.066	-	-	-	12.6	nd	12.6
5/23/19 dup		-	nd	0.02	0.066	-	-	-	12.5	2.9	9.6
6/12/2019		-	nd	0.02	0.087	-	-	-	8.18	2.4	5.78
6/25/2019		-	nd	0.04	0.094	-	-	-	10.8	nd	10.8
7/9/2019		0.04	nd	0.08	0.168	1.05	1.05	6.3	ND U	ND U	-
7/9/19 bottom		-	-	0.22	0.421	-	-	-	-	-	-
7/23/2019		0.04	nd	0.09	0.185	1.11	1.11	6.0	38.9	nd	38.9
8/13/2019		0.02	0.01	0.2	0.356	1.59	1.6	4.5	50.5	2.2	48.3
8/27/2019		0.06	0.01	0.28	0.436	1.39	1.4	3.2	47.8	5.3	42.5
9/10/2019		0.05	ND	0.29	0.503	1.77	1.77	3.5	34.3	11.8	22.5
9/10/19 dup		0.05	nd	0.29	0.5	1.7	1.7	3.4	42	6.3	35.7
9/27/2019		0.15	0.01	0.28	0.349	1.35	1.36	3.9	12.4	2.3	10.1
10/15/2019		0.02	0.03	0.07	0.165	1.16	1.19	7.2	53.6	ND	53.6
10/29/2019		ND	ND	0.03	0.073	0.82	0.82	11.2	ND	12.9	0
11/11/2019		0.03	ND	ND	0.045	0.69	0.69	15.3	7.45	ND	7.45
12/17/2019		0.04	ND	0.01	0.066	0.8	0.8	12.1	17.2	6.1	11.1
1/9/2020		0.11	0.01	0.01	0.069	0.79	0.8	11.6	13.2	4.2	9
2/11/2020		0.06	0.07	0.02	0.058	0.79	0.86	14.8	10.7	2.8	7.9

APPENDIX B

Lone Lake Phytoplankton Summary



Lone Lake Phytoplankton Summary

Dr. Robin A. Matthews

Institute for Watershed Studies
Huxley College of the Environment
Western Washington University

November 12, 2019

Funding for this project was provided by the Whidbey Island Conservation District and the Institute for Watershed Studies. We thank Matt Zupich, Whidbey Island Conservation District, and Dr. Mark Sytsma, Professor Emeritus, Portland State University, for their assistance throughout this project.

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1 Introduction

2 Methods

Sample collection: Planktonic algae samples were collected by Dr. Mark Sytsma on June 12 and 25, July 9 and 23, August 13 and 27 September 10 and 27, and October 15 and 29, 2019. The samples were collected using a 20 μm mesh phytoplankton net that was towed slightly below the lake surface for approximately 3 meters. The concentrated plankton samples were stored in a 250-mL polypropylene bottle and kept cool and away from direct sunlight until the sample could be delivered to the Institute for Watershed Studies (IWS) laboratory in Bellingham Washington. Live, unpreserved samples were brought to the laboratory on June 12, July 9, October 15, and October 29, 2019, and were examined within 48 hr. The remaining samples were preserved in 10% buffered formalin and held until they could be delivered to the IWS laboratory.

Sample processing: Live, unpreserved algae samples were opened and placed in an environmental chamber at 15° C on a 18:6 light/dark cycle until the samples could be processed. Preserved samples were kept at room temperature out of direct sunlight until they could be examined. The live and preserved algae were examined using a Nikon Eclipse 80i with phase contrast and Nomarski (DIC) objectives equipped with a Nikon DS-Fi2 digital camera.

Aliquots from the live algae collected on June 12 and July 9, 2019 were preserved in 2% glutaraldehyde for SEM analysis. The preserved samples were dewatered using sequential dilutions of 10–90% ethanol, dried using a critical point CO_2 evaporator, sputter coated, and examined using a scanning electron microscope.

Sample analysis: The algae in each sample were identified to the lowest practical taxonomic level (taxon) using standard taxonomic literature. Each taxon was assigned a relative abundance rank: rank #1 indicates taxa that were present but not abundant; rank #2 indicates taxa that were common in the sample but were not dominant; rank #3 indicates taxa that dominated the sample and were generally present in all microscopic fields of view. The ranked data from each Lone Lake samples are listed in Table 1. Additional algae data collected by IWS from 2008–2019 were used to supplement the Lone Lake algae species list (Table 2). Digital images for all algal taxa listed in Tables 1–2 are included in Figures 1–60, beginning on page 10.

Table 1: Lone Lake algal abundance in plankton samples collected 12 June - 29 October 2019.

Taxon	Sampling date				
	Jun 12	Jun 25	Jul 9	Jul 23	Aug 13
Chlorophyta (green algae)					
Botryococcus braunii	2	1	1	1	1
Chlamydomonas spp. (including zoospores)	2	0	1	0	0
Chlorococcum minutum?	0	0	0	0	0
Chlorogonium sp. (epiphytic)	0	1	1	0	0
Coelastrum microporum	0	0	0	0	0
Eudorina elegans	1	3	1	0	0
Korshikoviella michailovskoensis (epizoic)	0	0	1	0	0
Oocystis spp.	2	0	1	2	2
Pediastrum duplex	1	0	1	1	0
Planktosphaeria gelatinosa	0	0	1	1	1
Pseudopediastrum boryanum	1	0	1	0	0
Sphaerocystis schroeteri	1	2	3	2	1
Stylosphaeridium stipitatum (epiphytic)	0	0	1	0	0
Tetraspora lemmermannii?	1	0	0	0	0
Volvox tertius	1	2	3	1	0
Streptophyta (desmids and related green algae)					
Closterium acutum var. *variabile*	0	0	0	0	0
Closterium spp.	1	1	1	0	0
Elakatothrix gelatinosa	0	0	0	1	0
Staurastrum pingue var. *planctonicum*	3	2	3	1	1
Staurastrum cingulum	3	2	2	1	1
Cyanobacteria (blue-green algae)					
Aphanocapsa/Aphanothece spp.	1	1	2	1	1
Dolichospermum crassum	2	1	1	1	1
Dolichospermum spp.	2	2	2	3	2
Gloeotrichia echinulata	2	2	2	2	0
Microcystis aeruginosa	2	1	2	3	3
Microcystis flos-aquae?	0	0	1	0	1
Microcystis wesenbergii	1	0	1	1	1
Oscillatoria sp.	0	1	1	1	1
Phormidium spp.	1	0	1	0	0
Pseudanabaena spp.	0	0	1	1	3

continued on next page

Table 1: Lone Lake 2019 algal abundance, continued

Taxon	Sampling date				
	Jun 12	Jun 25	Jul 9	Jul 23	Aug 13
Cyanobacteria (blue-green algae)					
Woronichinia compacta?	0	0	0	0	0
Woronichinia naegeliana	2	1	1	2	3
Other/Bacilliariophyta (diatoms)					
Asterionella formosa	3	1	1	1	0
Aulacoseira ambigua	3	2	3	2	2
Eunotia sp.	0	0	0	1	0
Fragilaria capucina	1	0	0	0	0
Fragilaria crotonensis	3	3	3	2	0
Stephanodiscus niagarae	3	1	1	0	1
unk. filamentous diatom	0	0	0	0	0
unk. naviculoid diatom	0	0	0	0	0
Other/Cryptophyta (cryptomonads)					
Cryptomonas spp.	0	0	0	0	0
Other/Euglenophyta (euglenoids)					
Euglena spp.	1	0	1	0	0
Euglena texta	0	1	1	0	0
Trachelomonas hispida	1	0	0	0	0
Trachelomonas volvocinopsis	1	1	2	1	1
Other/Miozoa (dinoflagellates)					
Ceratium hirundinella (including cysts)	0	2	1	0	1
Other/Ochrophyta (golden algae)					
Dinobryon divergens	0	0	2	0	0
Lagynion sp. (epiphytic)	3	0	1	0	0
Mallomonas sp.	0	0	0	0	0
Uroglenopsis americana	0	1	0	0	0

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Table 1: Lone Lake 2019 algal abundance, continued

Taxon	Sampling date				
	Aug 27	Sep 10	Sep 27	Oct 15	Oct 29
Chlorophyta (green algae)					
Botryococcus braunii	1	1	0	0	0
Chlamydomonas spp. (including zoospores)	0	0	0	1	1
Chlorococcum minutum?	0	0	0	0	1
Chlorogonium sp. (epiphytic)	0	0	0	0	1
Coelastrum microporum	0	1	0	0	0
Eudorina elegans	1	1	2	1	1
Korshikoviella michailovskoensis (epizoic)	0	0	0	0	0
Oocystis spp.	2	1	1	1	1
Pediastrum duplex	0	0	0	0	1
Planktosphaeria gelatinosa	0	0	1	2	2
Pseudopediastrum boryanum	0	0	0	0	1
Sphaerocystis schroeteri	1	1	1	1	1
Stylosphaeridium stipitatum (epiphytic)	0	0	0	0	0
Tetraspora lemmermannii?	0	0	0	0	0
Volvox tertius	0	0	1	1	1
Streptophyta (desmids and related green algae)					
Closterium acutum var. variabile	0	0	0	0	1
Closterium spp.	0	0	0	0	0
Elakatothrix gelatinosa	0	0	0	0	0
Staurastrum pingue var. *planctonicum*	1	2	1	1	1
Staurastrum cingulum	1	1	0	0	1
Cyanobacteria (blue-green algae)					
Aphanocapsa/Aphanothece spp.	1	2	2	1	1
Dolichospermum crassum	2	2	1	1	1
Dolichospermum spp.	2	1	1	1	2
Gloeotrichia echinulata	0	0	0	0	0
Microcystis aeruginosa	3	1	2	1	1
Microcystis flos-aquae?	0	2	1	1	1
Microcystis wesenbergii	3	3	2	1	1
Oscillatoria sp.	1	1	1	0	0
Phormidium spp.	0	0	0	0	0
Pseudanabaena spp	3	0	0	1	0

continued on next page

Table 1: Lone Lake 2019 algal abundance, continued

Taxon	Sampling date				
	Aug 27	Sep 10	Sep 27	Oct 15	Oct 29
Cyanobacteria (blue-green algae)					
Woronichinia compacta?	0	2	1	1	1
Woronichinia naegeliana	3	3	3	2	1
Other/Bacilliariophyta (diatoms)					
Asterionella formosa	0	0	0	0	0
Aulacoseira ambigua	1	1	1	3	3
Eunotia sp.	0	0	0	0	0
Fragilaria capucina	0	0	0	0	0
Fragilaria crotonensis	0	1	1	2	2
Stephanodiscus niagarae	1	1	3	3	3
unk. filamentous diatom	0	0	0	0	1
unk. naviculoid diatom	0	0	0	0	1
Other/Cryptophyta (cryptomonads)					
Cryptomonas spp.	0	0	0	2	2
Other/Euglenophyta (euglenoids)					
Euglena spp.	1	1	1	1	2
Euglena texta	0	0	0	1	2
Trachelomonas hispida	0	0	0	0	0
Trachelomonas volvocinopsis	0	1	0	1	1
Other/Miozoa (dinoflagellates)					
Ceratium hirundinella (including cysts)	1	1	2	1	1
Other/Ochrophyta (golden algae)					
Dinobryon divergens	0	0	0	0	0
Lagynion sp. (epiphytic)	0	0	0	0	3
Mallomonas sp.	0	0	0	1	1
Uroglenopsis americana	0	0	0	0	0

Table 2: Lone Lake algae species list. Species that were collected by IWS (2008–2019) but not found in the 2019 plankton samples are indicated using an asterisk. Taxonomic authority is from Guiry and Guiry (2019).

Taxon	Taxonomic Authority
Chlorophyta (green algae)	
*Ankyra judayi**	(G.M.Smith) Fott
Botryococcus braunii	Kützing
*Characium ornithocephalum**	A.Braun
Chlamydomonas spp.	Ehrenberg
Chlorococcum minutum?	R.C.Starr
Chlorogonium sp. (epiphytic)	Ehrenberg
Coelastrum microporum	Nägeli
Eudorina elegans	Ehrenberg
Korshikoviella michailovskoensi	(Elenkin) P.C.Silva
Oedogonium sp.*	Link ex Hirn
Oocystis spp.	Nägeli ex A.Braun
Pediastrum duplex	Meyen
Planktosphaeria gelatinosa	G.M.Smith
*Pleodorina californica**	W.R.Shaw
Pseudopediastrum boryanum	(Turpin) E.Hegewald
Sphaerocystis schroeteri	Chodat
Stylosphaeridium stipitatum	(Bachmann) Geitler & Gimesi
Tetraspora lemmermannii?	Fott
*Volvox aureus**	Ehrenberg
*Volvox globator**	Linnaeus
Volvox tertius	Art.Meyer
Streptophyta (desmids and related green algae)	
Closterium acutum var. *variabile*	(Lemmermann) Willi Krieger
Closterium spp.	Nitzsch ex Ralfs
Elakatothrix gelatinosa	Wille
Staurastrum pingue var. *planctonicum*	(Teiling) Coesel & Meersters
Staurastrum cingulum	West & G.S.West G.M.Smith
Cyanobacteria (blue-green algae)	
Aphanocapsa/Aphanothece spp.	Nägeli/Nägeli
*Chroococcus*s sp.*	Nägeli
Dolichospermum crassum	(Lemmermann) P.Wacklin, L.Hoffmann & J.Komárek

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Table 2: Lone Lake algae species list, continued

Taxon	Taxonomic Authority
Cyanobacteria (blue-green algae)	
Dolichospermum spp.	(Ralfs ex Bornet & Flahault) P.Wacklin, L.Hoffmann & J.Komárek
Gloeotrichia echinulata	P.G.Richter
*Limnoraphis birgei**	(G.M.Smith) J.Komárek, E.Zapomelová, J.Smarda, J.Kopecký, E.Rejmánková, J.Woodhouse, B.A.Neilan & J.Komárková
Microcystis aeruginosa	(Kützing) Kützing
Microcystis flos-aquae?	(Wittrock) Kirchner
Microcystis wesenbergii	(Komárek) Komárek ex Komárek
Oscillatoria sp.	Vaucher ex Gomont
Phormidium spp.	Kützing ex Gomont
*Pseudanabaena mucicola**	(Naumann & Huber-Pestalozzi) Schwabe
Pseudanabaena spp.	Lauterborn
*Tolypothrix lanata**	Wartmann ex Bornet & Flahault
Woronichinia compacta?	(Lemmermann) Komárek & Hindák
Woronichinia naegeliana	(Unger) Elenkin
Other/Bacilliariophyta (diatoms)	
Asterionella formosa	Hassall
Aulacoseira ambigua	(Grunow) Simonsen
Eunotia sp.	Ehrenberg
Fragilaria capucina	Desmazières
Fragilaria crotonensis	Kitton
Stephanodiscus niagarae	Ehrenberg
Surirella sp.*	Turpin
unk. filamentous diatom	Karsten
unk. naviculoid diatom	Karsten
Other/Cryptophyta (cryptomonads)	
Cryptomonas spp.	Ehrenberg
Other/Euglenophyta (euglenoids)	
Colacium vesiculosum	Ehrenberg
Euglena spp.	Ehrenberg
Euglena texta	(Dujardin) Hübner

continued on next page

Table 2: Lone Lake algae species list, continued

Taxon	Taxonomic Authority
Other/Euglenophyta (euglenoids)	
Trachelomonas hispida	(Perty) F.Stein
Trachelomonas volvocinopsis	Svirenko
Other/Miozoa (dinoflagellates)	
Ceratium hirundinella	(O.F.Müller) Dujardin
Other/Ochrophyta (golden algae)	
Dinobryon divergens	O.E.Imhof
Lagynion sp.	Pascher
Mallomonas sp.	Perty
Uroglenopsis americana	(G.N.Calkins) Lemmermann

3 Lone Lake Phytoplankton Images

This section contains high resolution digital images of the common algae collected in plankton samples from Lone Lake during the summer of 2019. All taxonomic identifications were provided by Dr. Robin Matthews and represent my best effort to provide accurate classifications using conventional taxonomic sources and following the nomenclature in AlgaeBase (http://www.algaebase.org). All images were photographed by Dr. Matthews using a Nikon Eclipse 80i microscope with phase contrast or Nomarski (DIC) objectives or a scanning electron microscope (SEM). These images may be used for noncommercial purposes under the copyright license described at http://www.wwu.edu/iws, with appropriate credit given to the image copyright holder (Dr. Matthews) and Western Washington University. Comments, suggestions, or requests for copies of the digital images may be directed to the Institute for Watershed Studies, Western Washington University, 516 High Street, Bellingham, WA, 98225.

Unless otherwise noted, the images represent algae collected in Lone Lake as part of this project or the IWS Northwest Lakes monitoring project. If high quality images were not available for Lone Lake taxa, representative images were included from other Northwest Washington lakes. The image captions for specimens from live samples list the date when the image was created, which was usually within 48 hr of collection. The captions on SEM images list the date when the sample was processed, with the image creation date stamped at the bottom of the image.

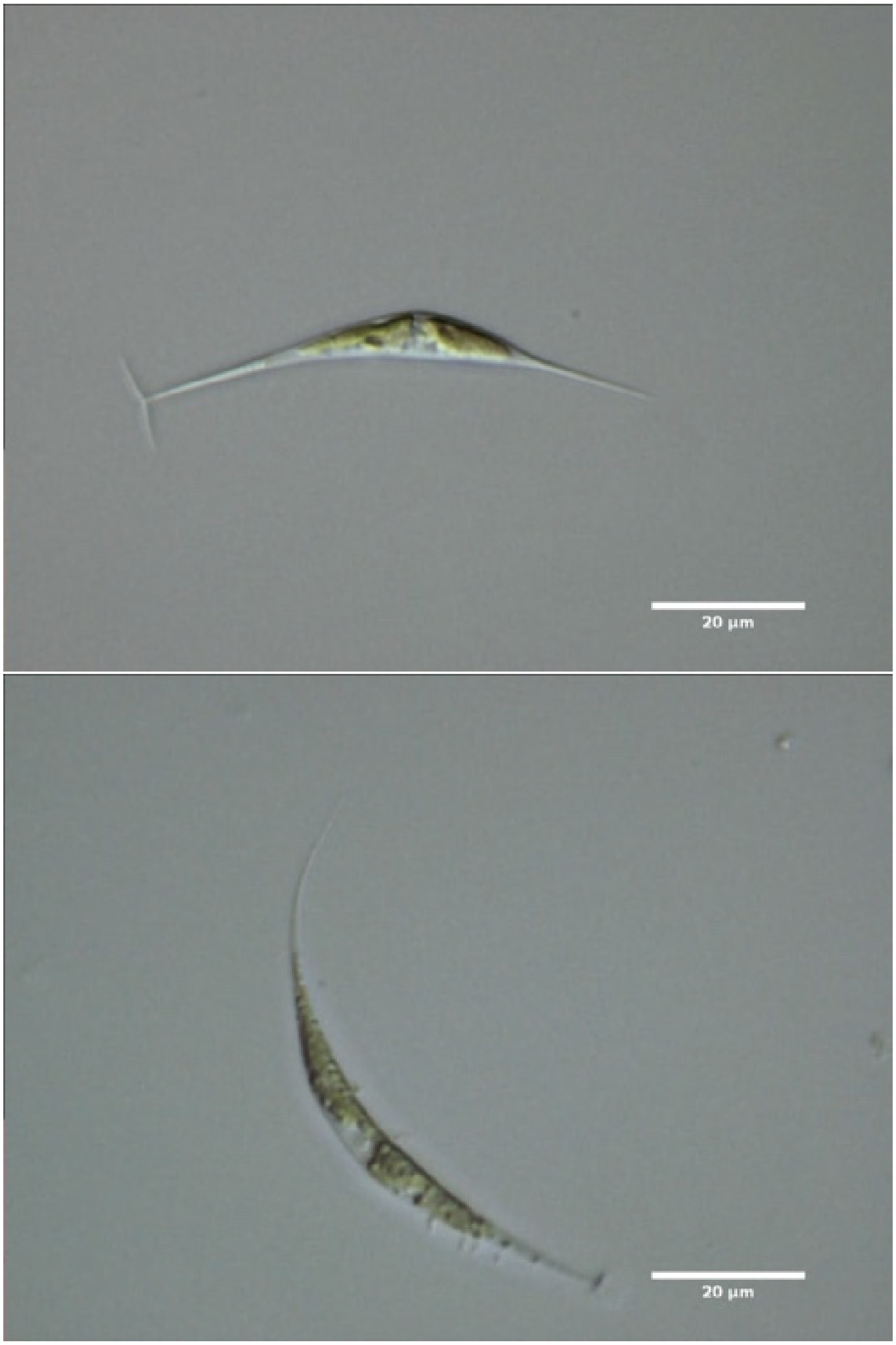


Figure 1: Chlorophyta - upper image: *Ankyra judayi* (600x DIC), September 1, 2008; lower image: *Ankyra judayi* (600x DIC), Cranberry Lake, Island County, July 7, 2010.

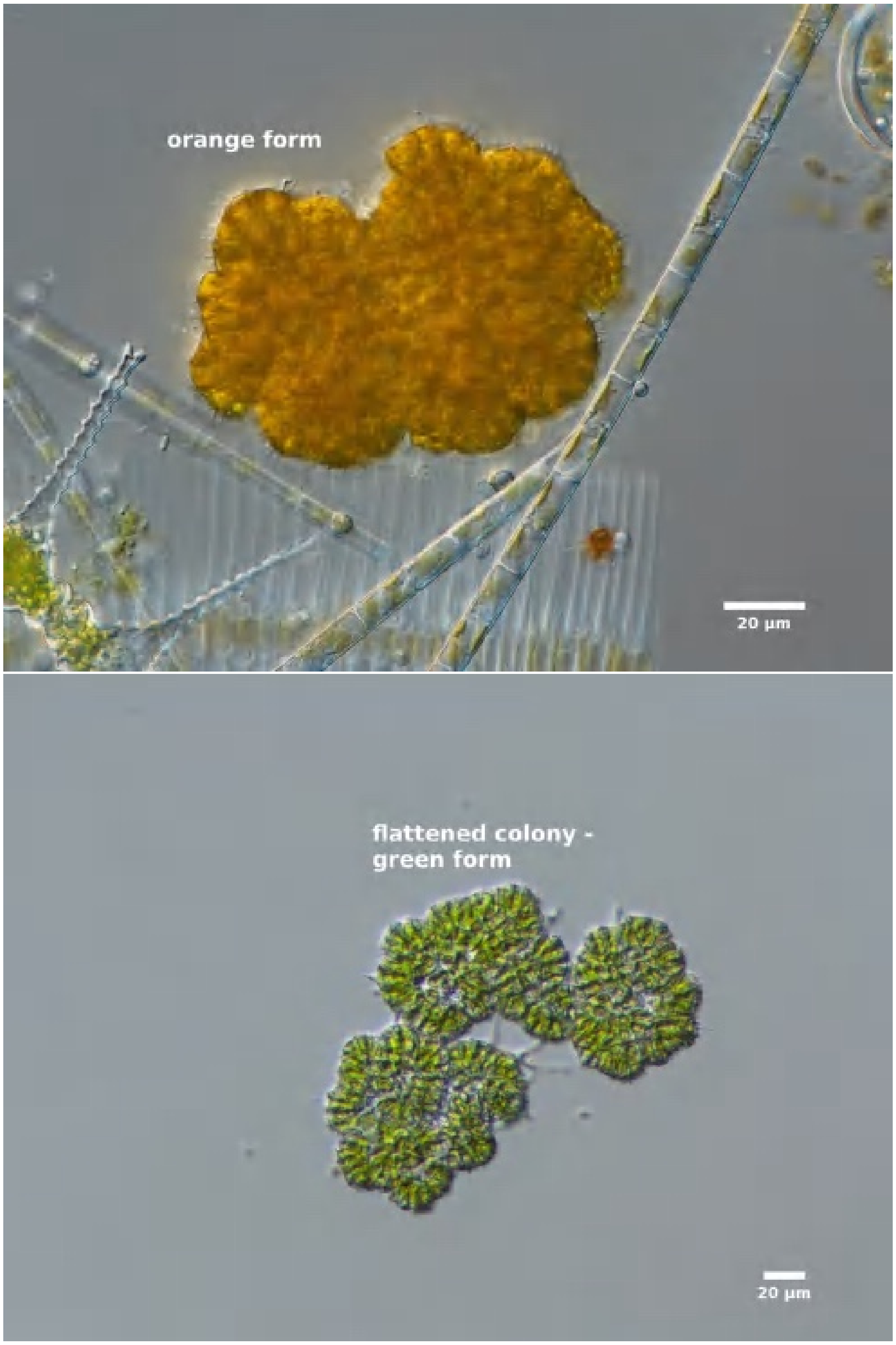


Figure 2: Chlorophyta - upper image: *Botryococcus braunii* (400x DIC), June 12, 2019; lower image: *Botryococcus braunii* (200x DIC), Heart Lake, Skagit County, July 19, 2016.

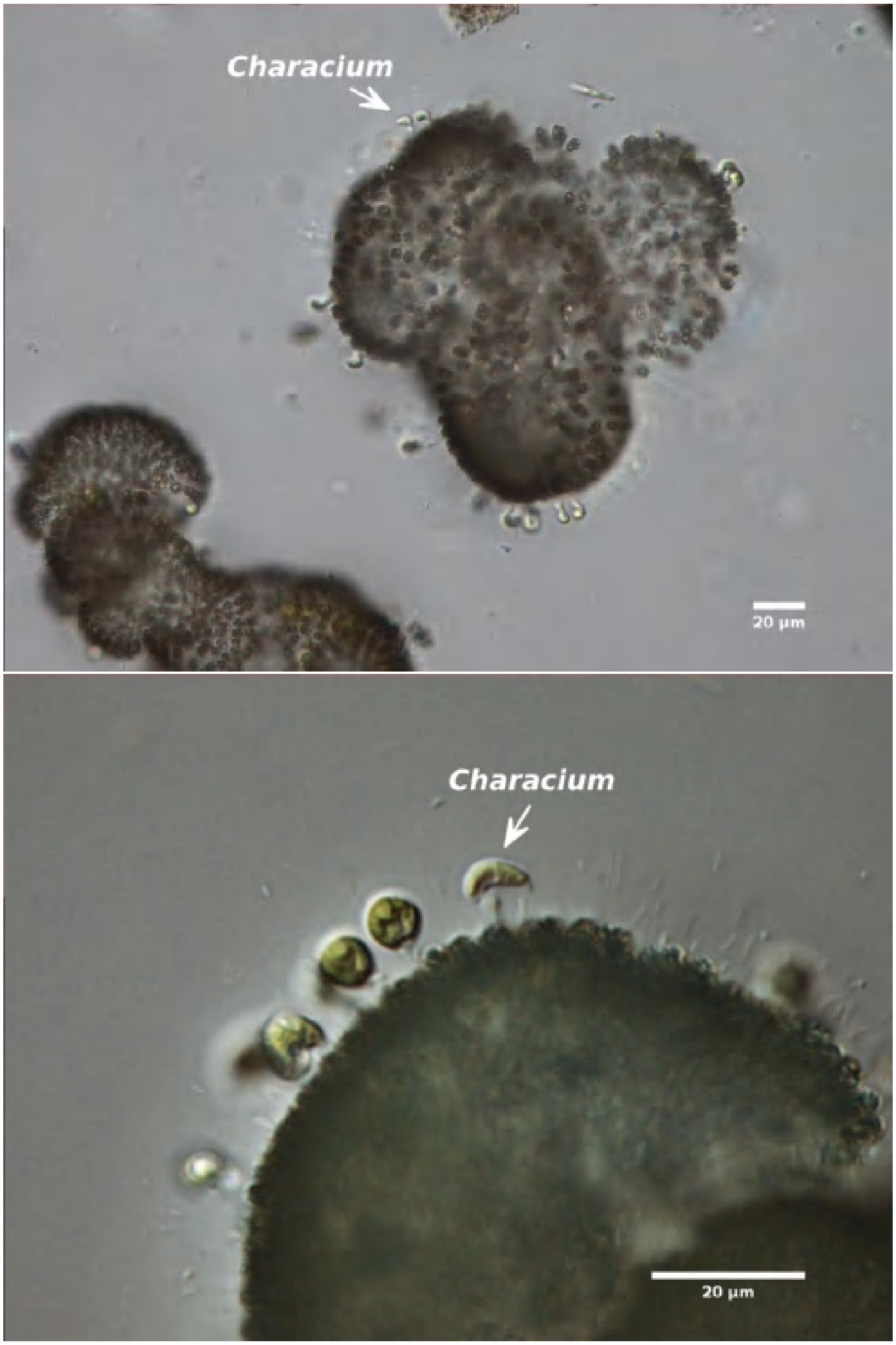


Figure 3: Chlorophyta - upper image: *Characium ornithocephalum* (200x DIC), September 30, 2009; lower image: *Characium ornithocephalum* (600x DIC), September 30, 2009.

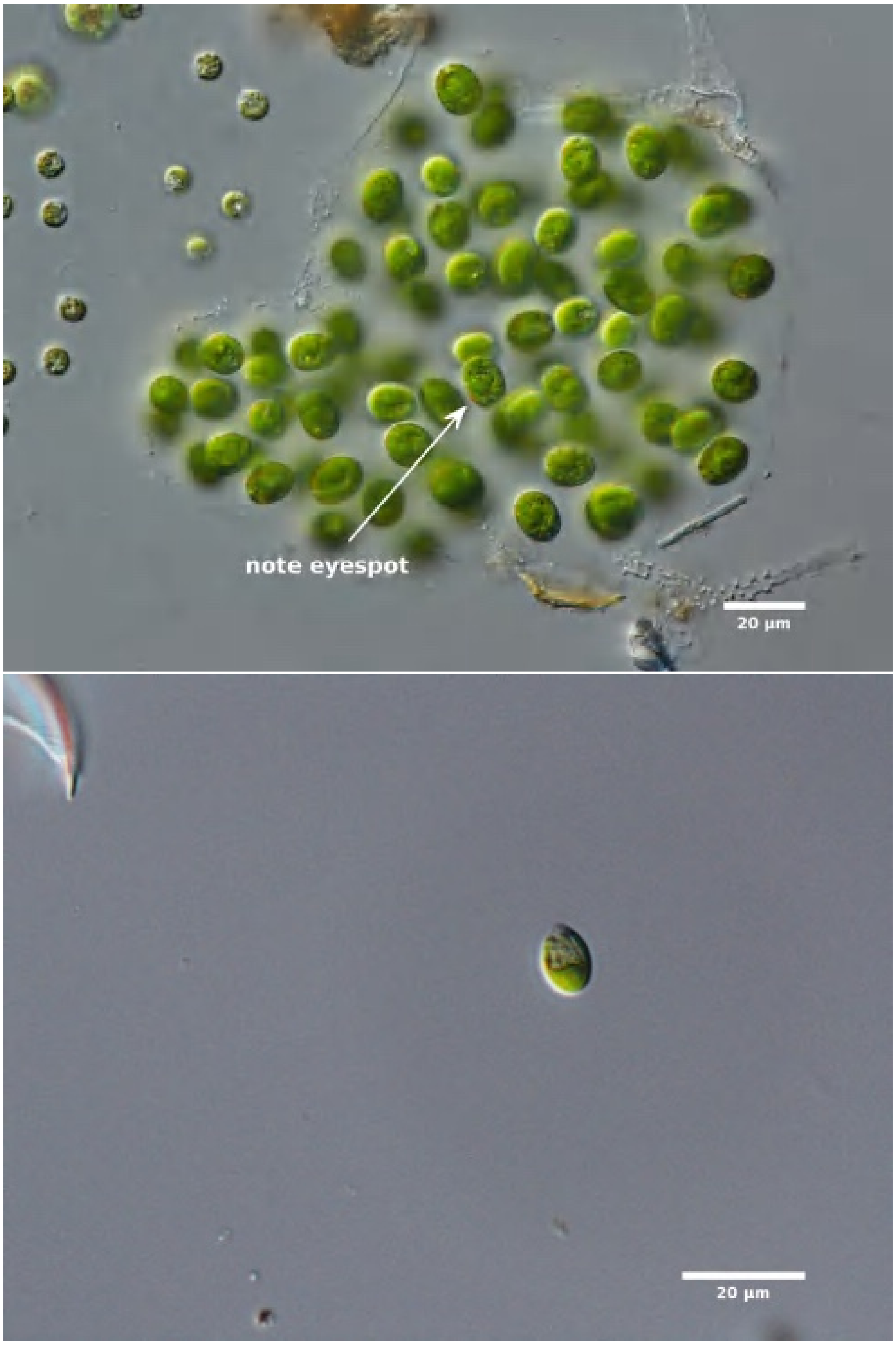


Figure 4: Chlorophyta - upper image: *Chlamydomonas* zoospores (400x DIC), July 9, 2019; lower image: *Chlamydomonas* vegetative cell (600x DIC), October 15, 2019.

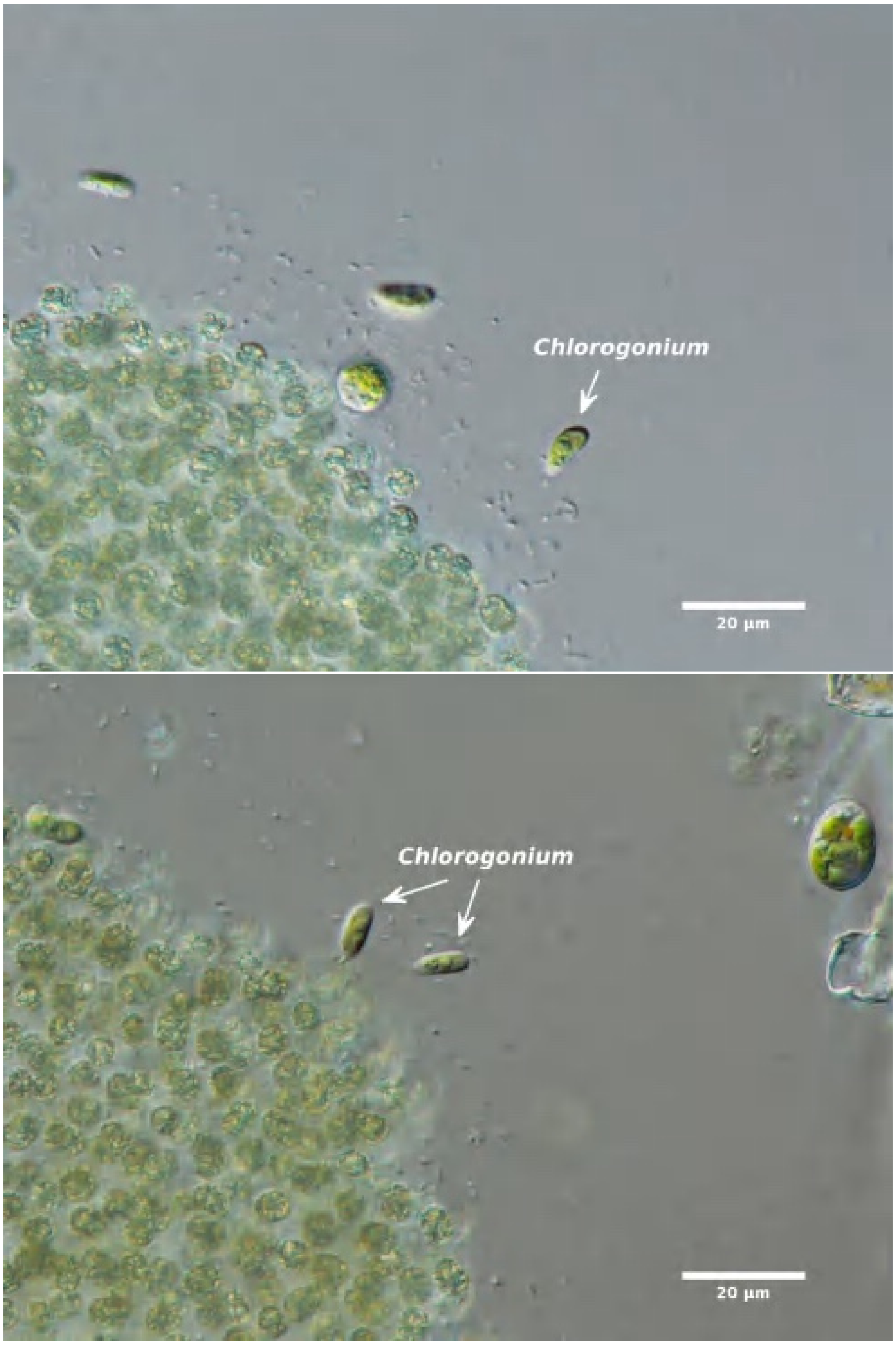


Figure 5: Chlorophyta - upper image: *Chlorogonium* (600x DIC), July 9, 2019; lower image: *Chlorogonium* (600x DIC), July 9, 2019.

Figure 6: Chlorophyta - upper/lower images: *Chlorococcum minutum*? (600x DIC), October 29, 2019.

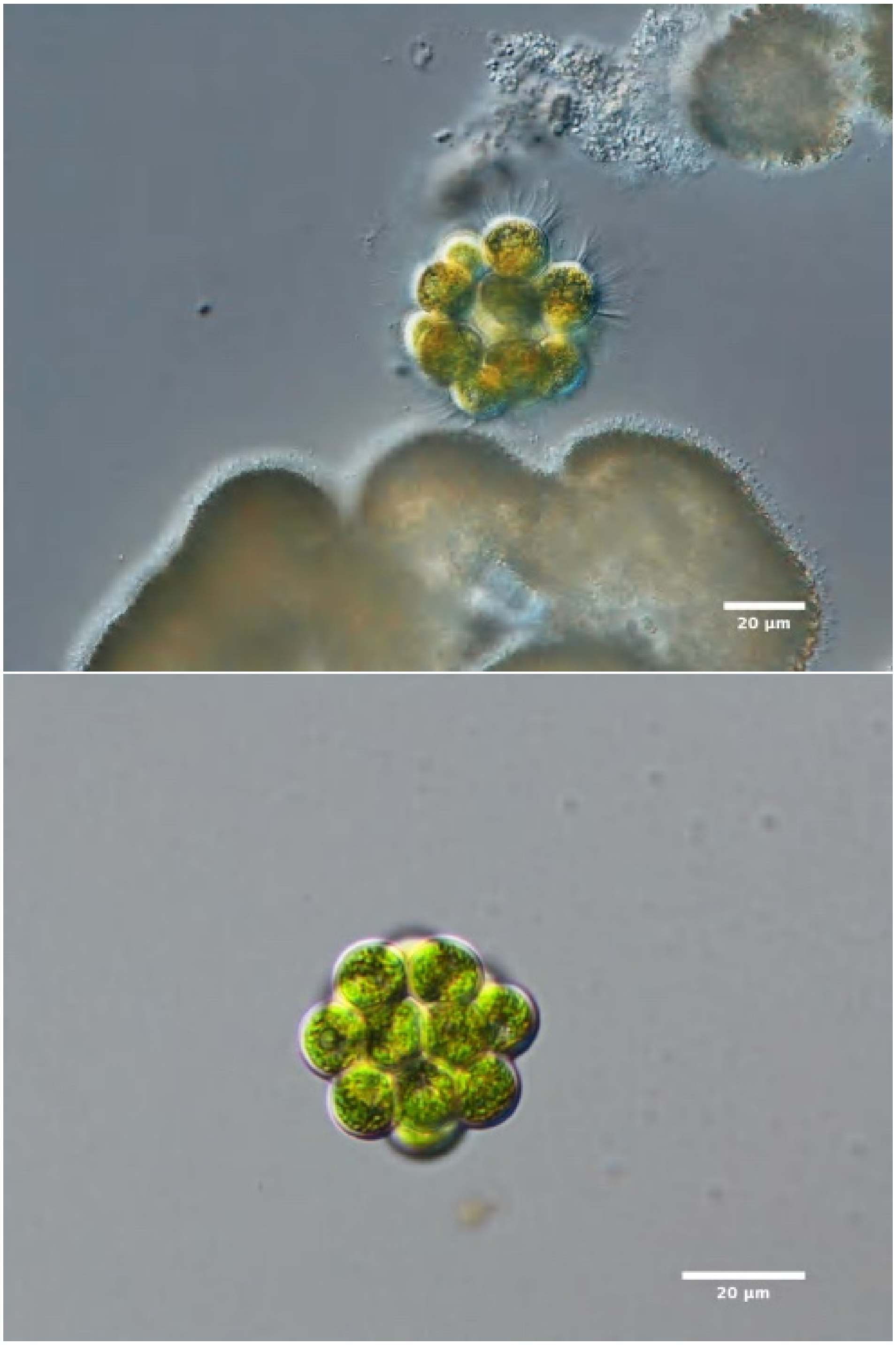


Figure 7: Chlorophyta - upper image: *Coelastrum microporum* (400x DIC), October 17, 2019; lower image: *Coelastrum microporum* (600x DIC), Heart Lake, Skagit County, June 26, 2016.

Figure 8: Chlorophyta - upper/lower images: *Eudorina elegans* (600x DIC), October 17, 2019.

Figure 9: Chlorophyta - upper image: *Korshikoviella michailovskoensis* (600x DIC), July 9, 2019; lower image: *Korshikoviella michailovskoensis* (400x DIC), Heart Lake, Skagit County, July 13, 2018.

Figure 10: Chlorophyta - upper image: *Oedogonium* (100x DIC), September 30, 2009; lower image: *Oedogonium* (200x DIC), small pond near Fairhaven Park, Whatcom County, May 20, 2015.

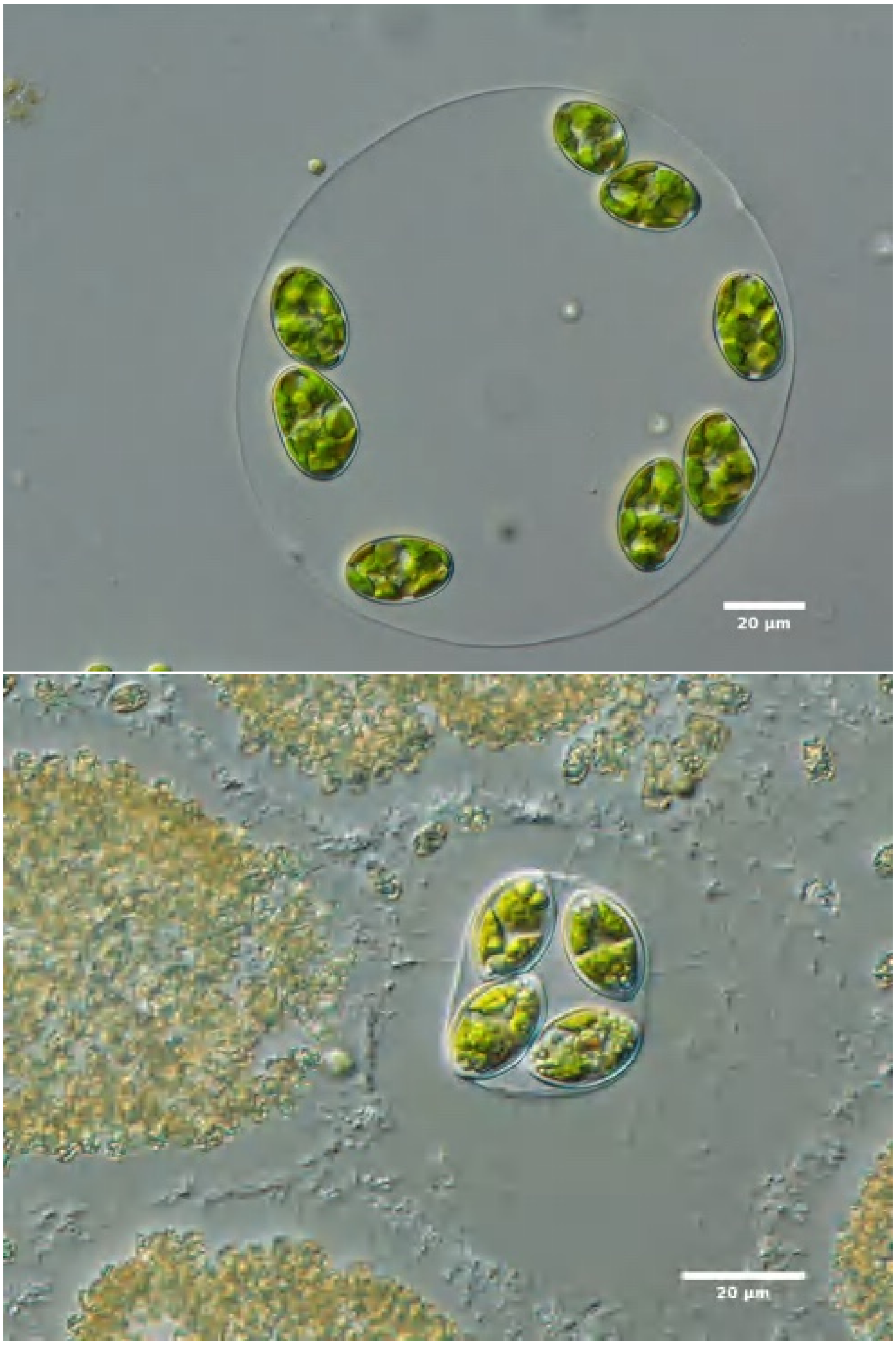


Figure 11: Chlorophyta - upper image: *Oocystis* (400x DIC), July 9, 2019; lower image: *Oocystis* (600x DIC), June 13. 2019. Note the different cell sizes that indicated there are at least two species of *Oocystis* present in Lone Lake.

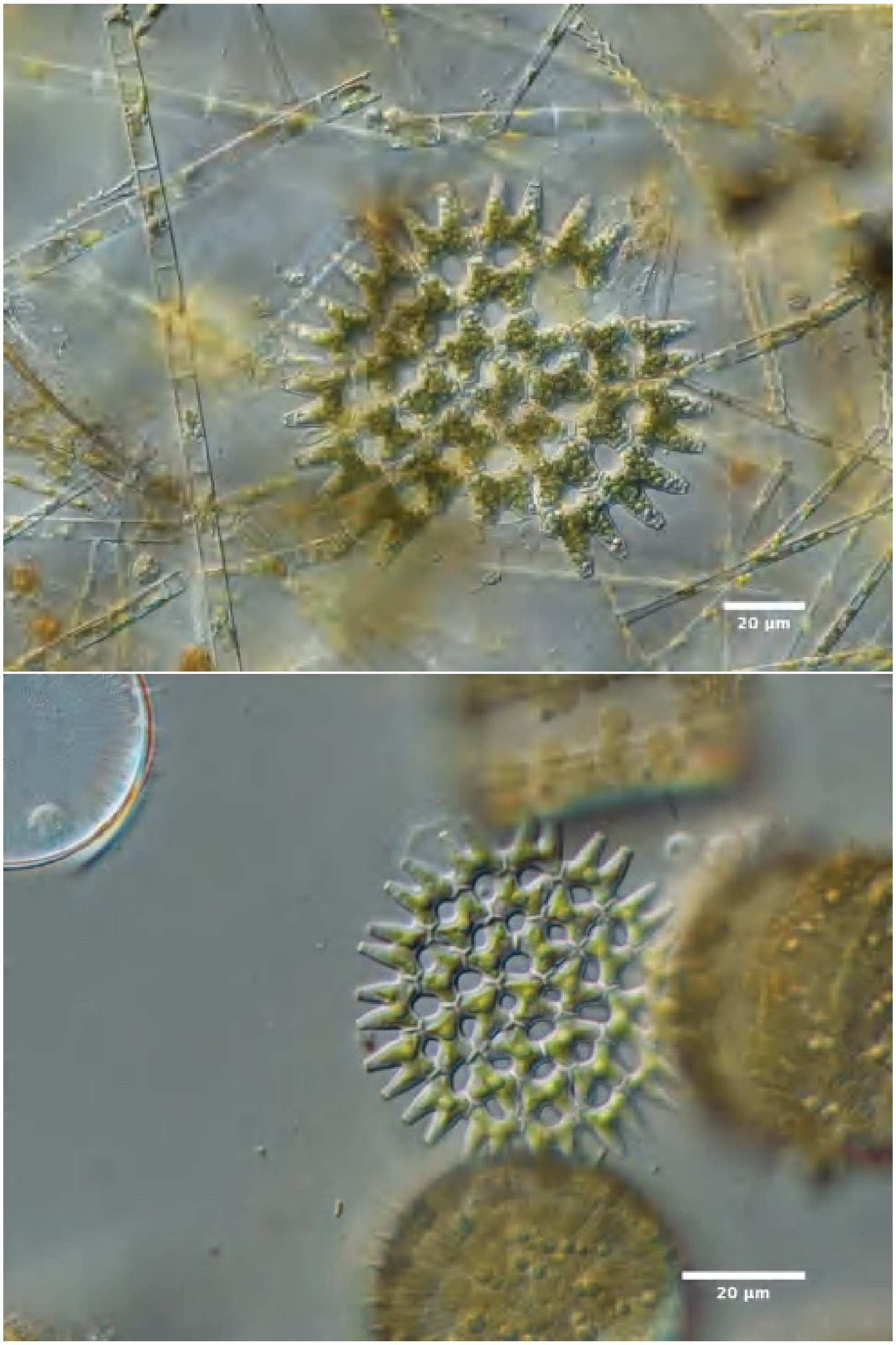


Figure 12: Chlorophyta - upper image: *Pediastrum duplex* (400x DIC), June 13, 2019; lower image: *Pediastrum duplex* (600x DIC), October 29, 2019.

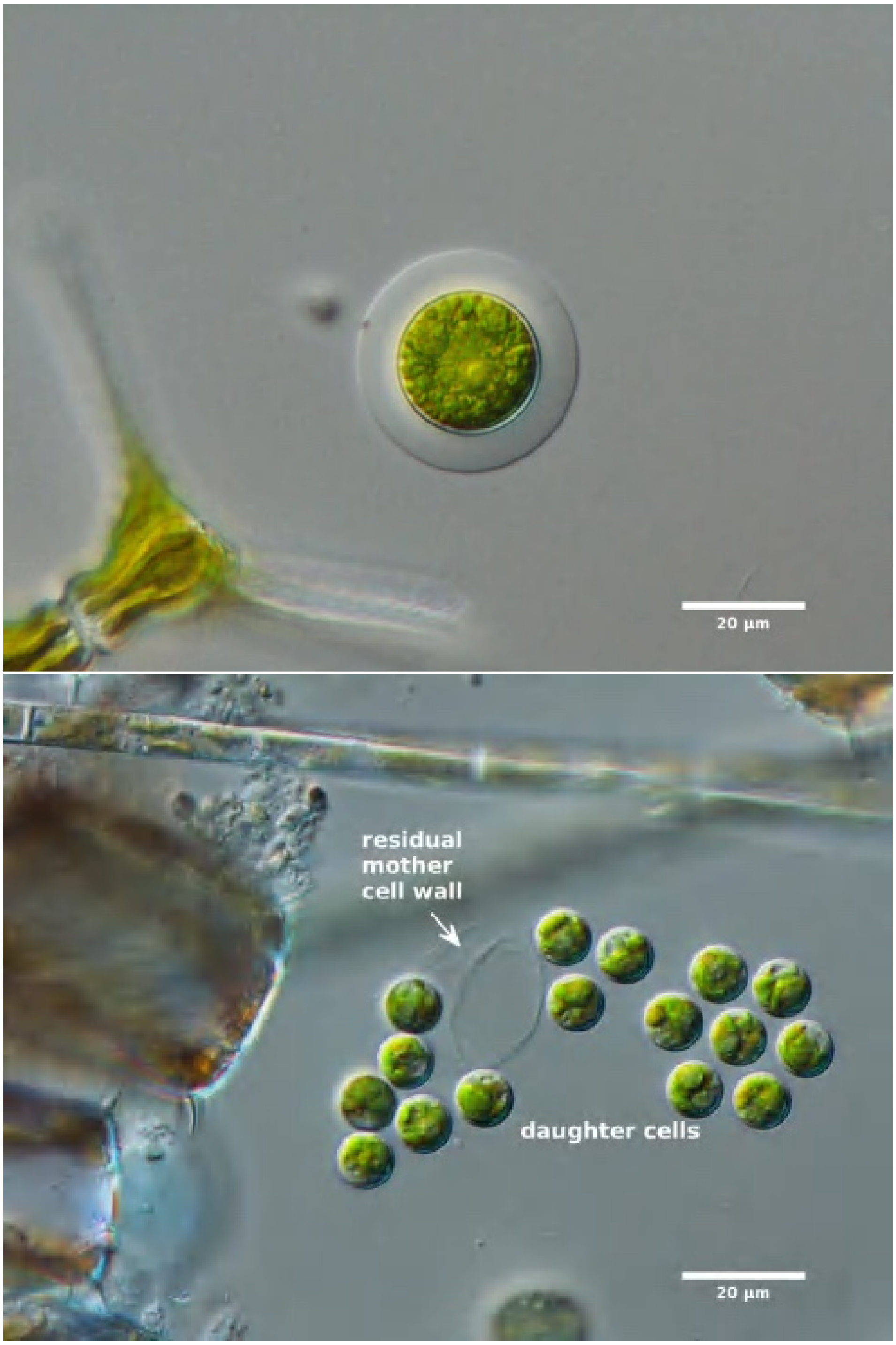


Figure 13: Chlorophyta - upper image: *Planktosphaeria gelatinosa* mother cell (600x DIC), July 9, 2019; lower image: *Planktosphaeria gelatinosa* daughter cells (600x DIC), October 15, 2019.

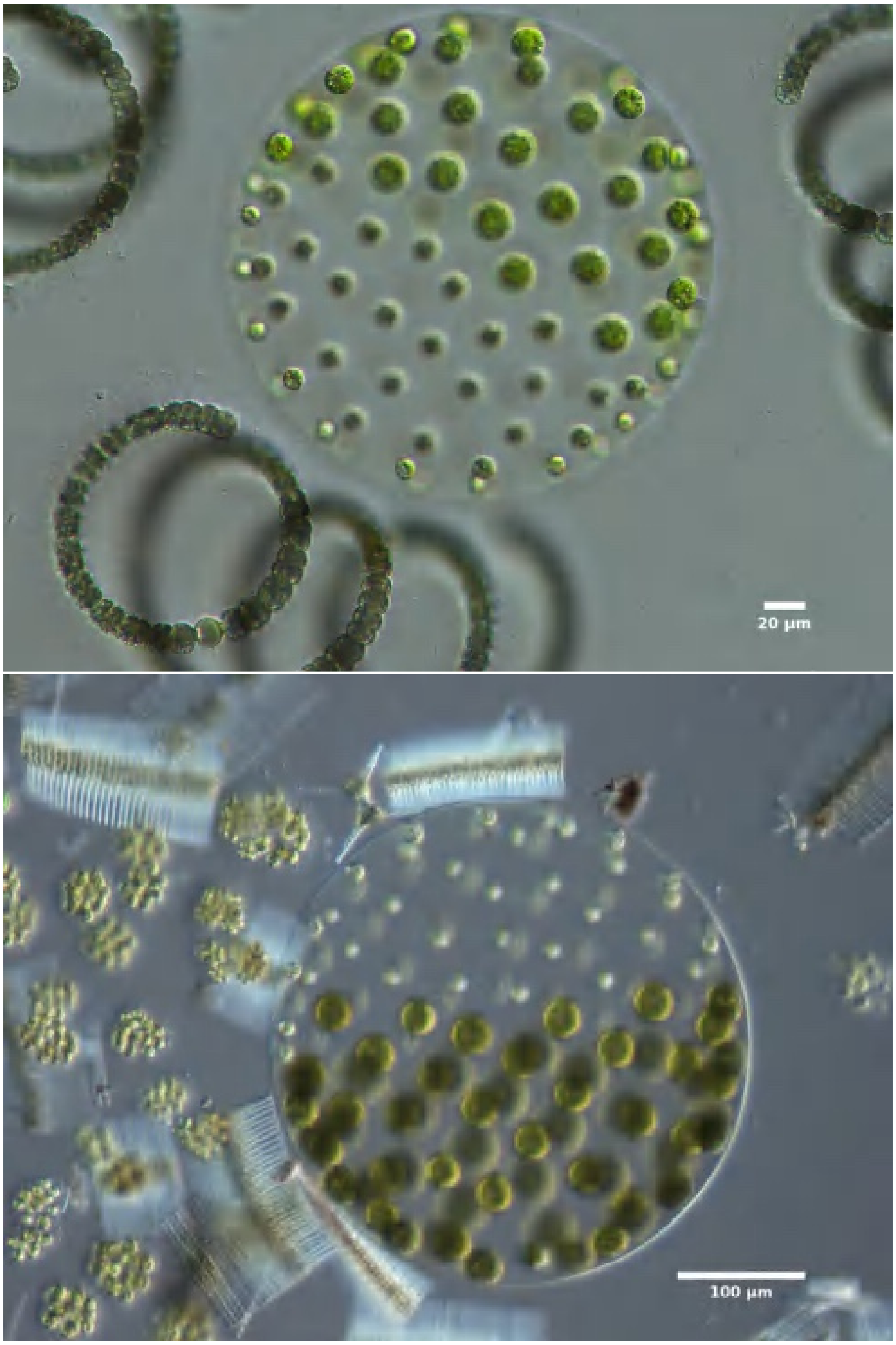


Figure 14: Chlorophyta - upper image: *Pleodorina californica* (200x DIC), August 4, 2014; lower image: *Pleodorina californica* (100x DIC), July 22, 2013.

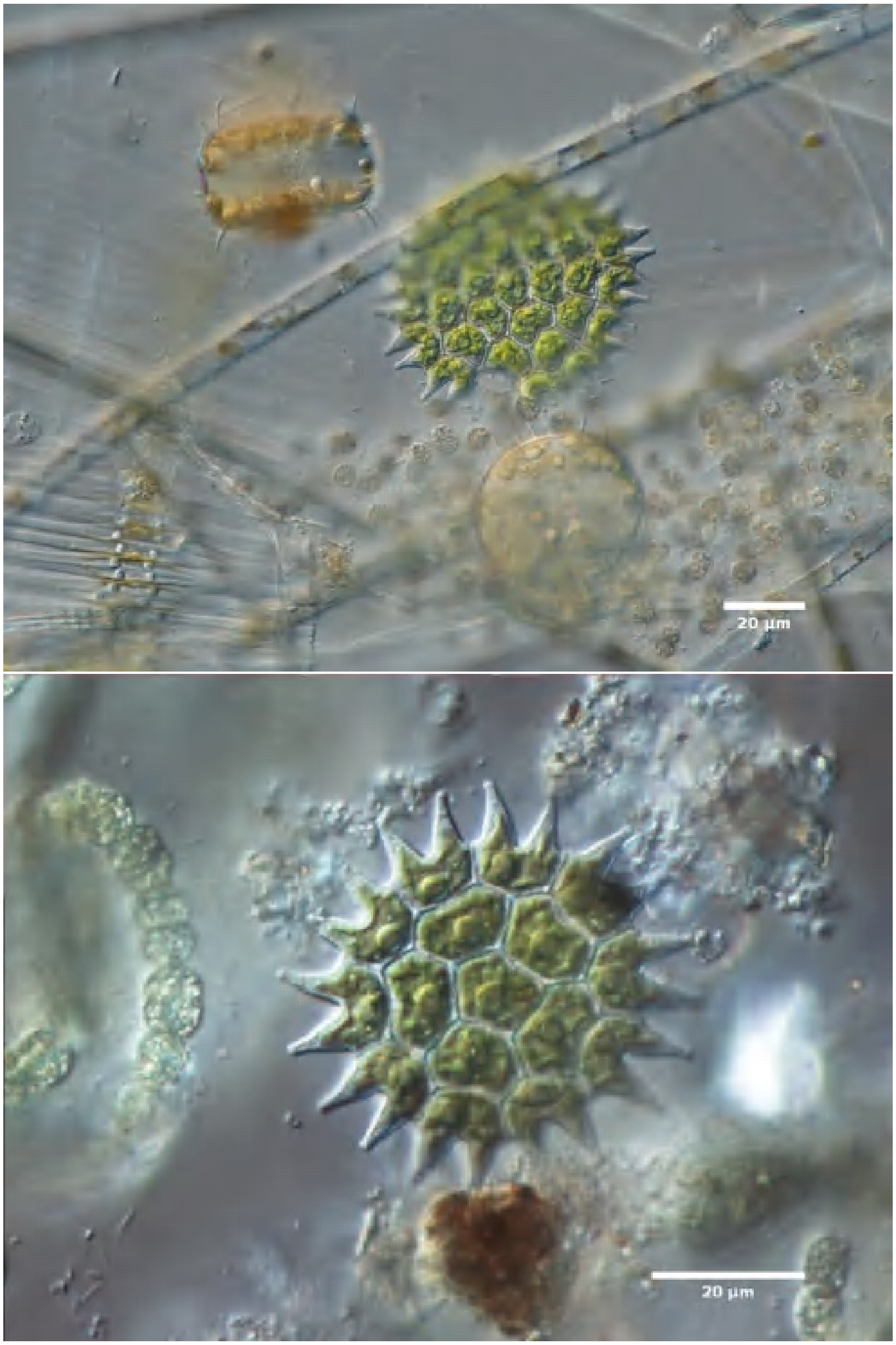


Figure 15: Chlorophyta - upper image: *Pseudopediastrum boryanum* (400x DIC), June 13, 2019; lower image: *Pseudopediastrum boryanum* (600x DIC), Wiser Lake, Whatcom County, August 19, 2009.

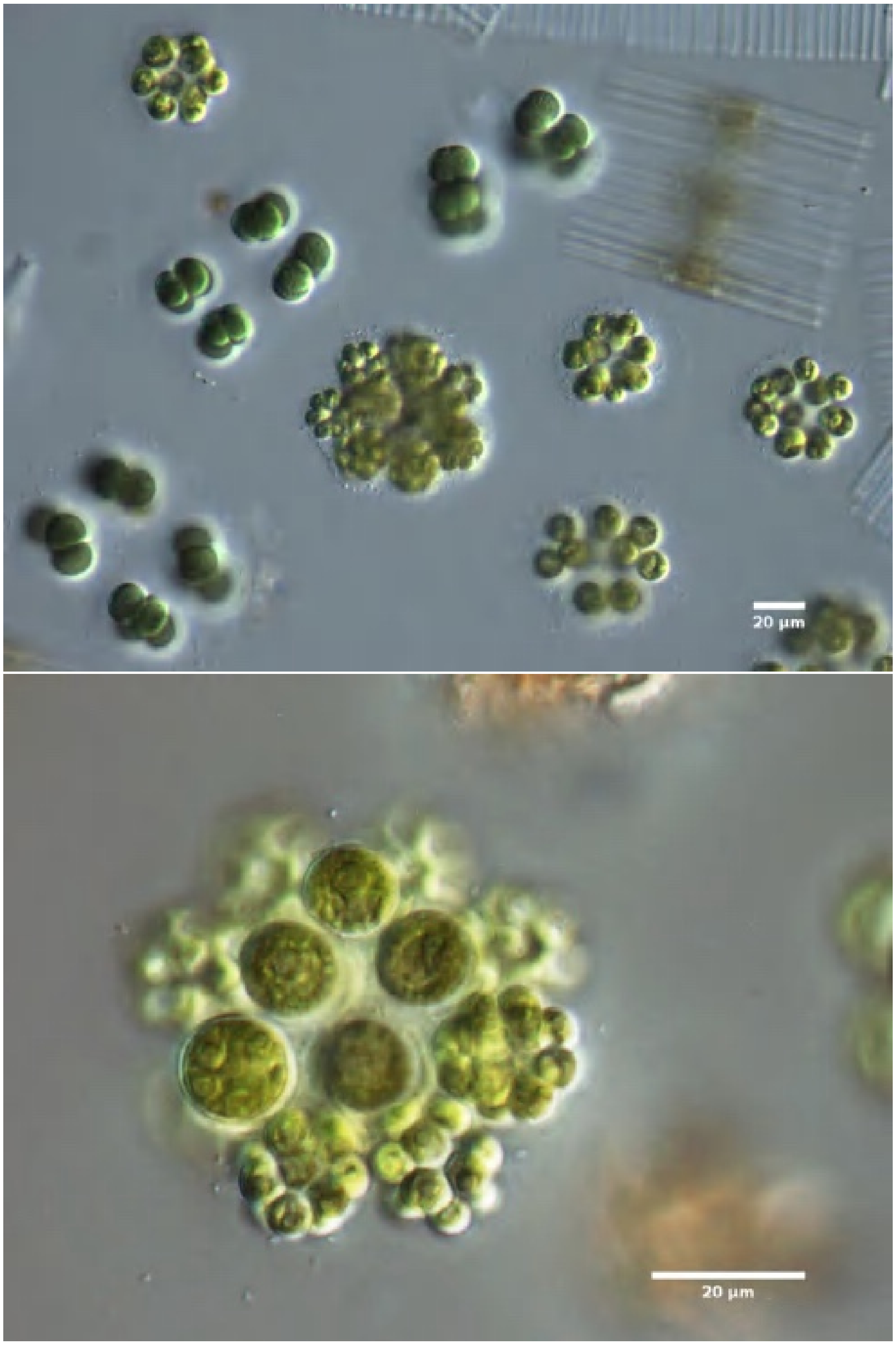


Figure 16: Chlorophyta - upper image: *Sphaerocystis schroeteri* (200x DIC), July 22, 2013; lower image: *Sphaerocystis schroeteri* (600x DIC), July 22, 2013.

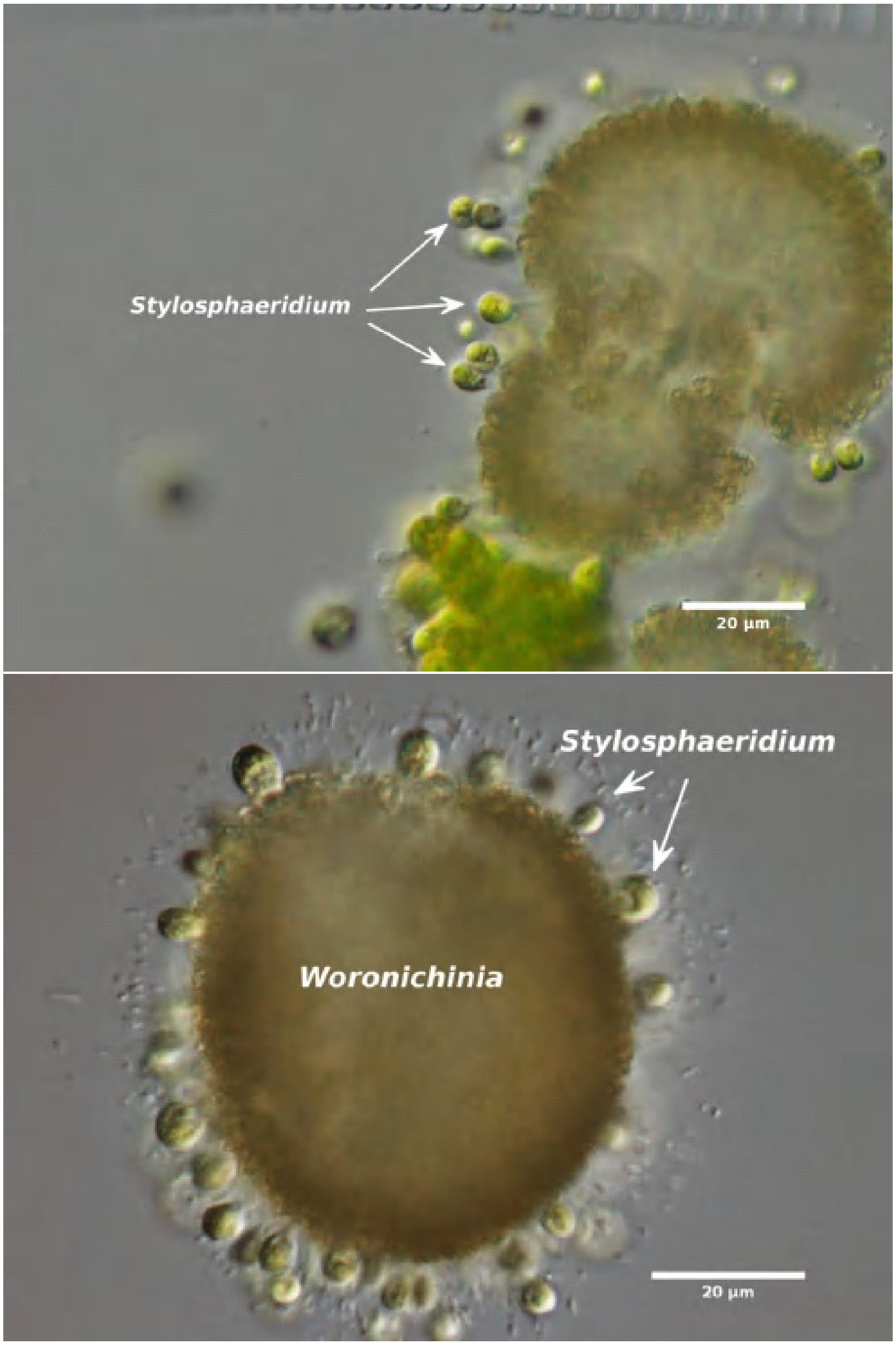


Figure 17: Chlorophyta - upper image: *Stylosphaeridium stipitatum* epiphyte on *Woronichinia* (600x DIC), July 9, 2019; lower image: *Stylosphaeridium stipitatum* epiphyte on *Woronichinia* (600x DIC), Lake Ketchum, Snohomish County, July 15, 2013.

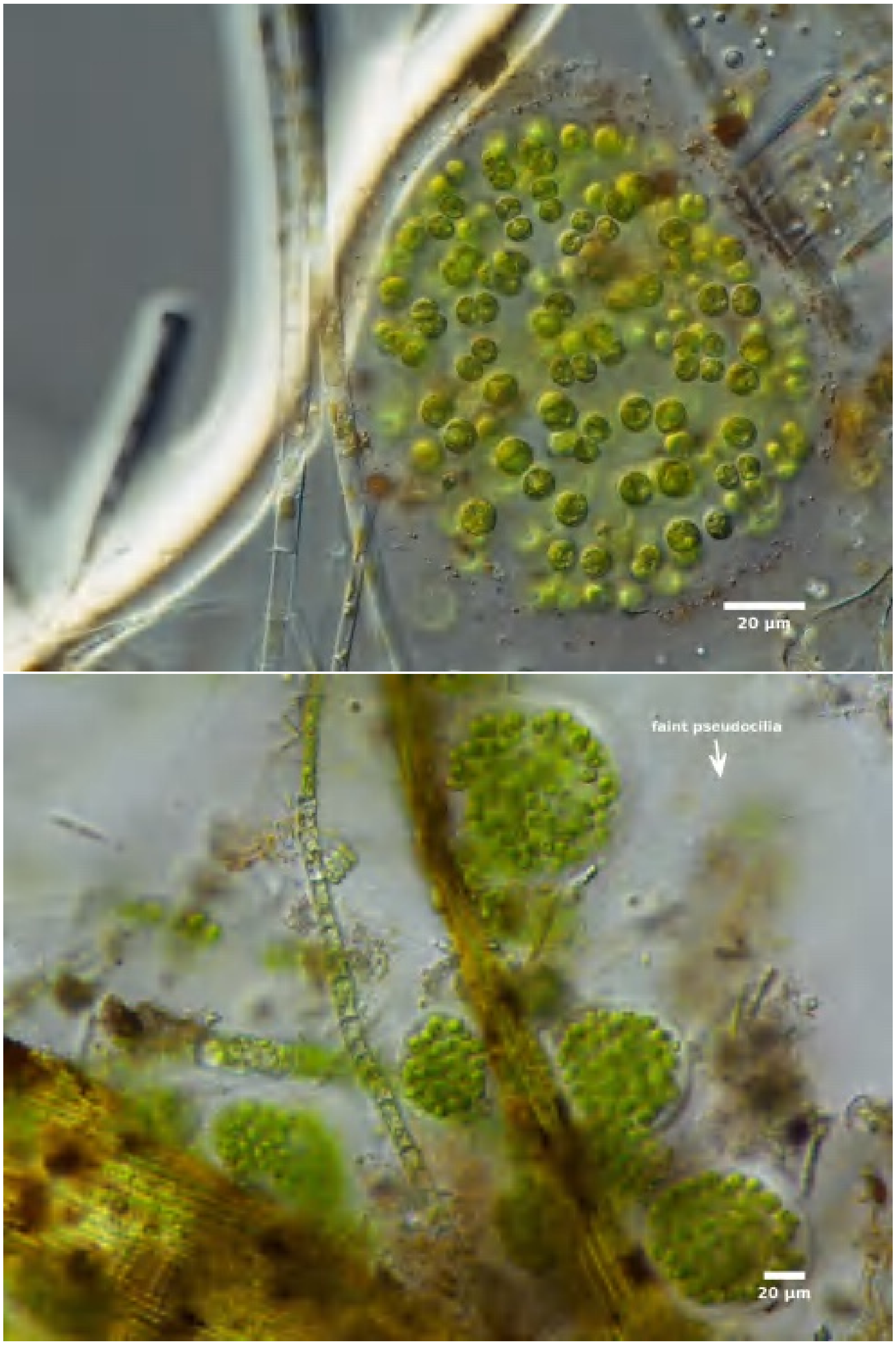


Figure 18: Chlorophyta - upper image: *Tetraspora lemmermannii*? (400x DIC), June 12, 2019; lower image: *Tetraspora lemmermannii*? (200x DIC), small pond near Fairhaven Park, Whatcom County, April 13, 2015.

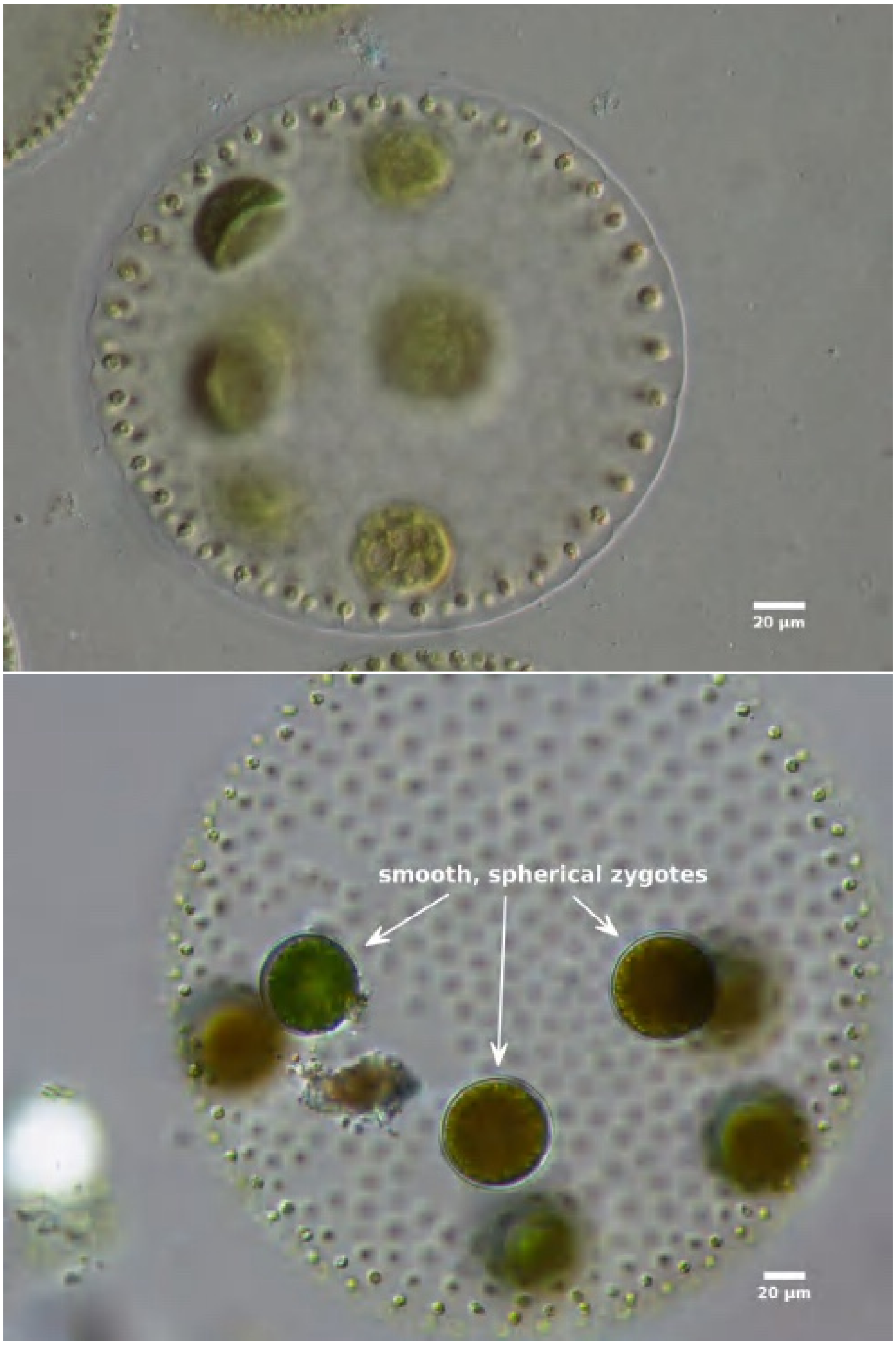


Figure 19: Chlorophyta - upper image: *Volvox aureus* (600x DIC), July 23, 2013; lower image: *Volvox aureus* zygotes (600x DIC), Myrtle Lake, Snohomish County, September 17, 2014.

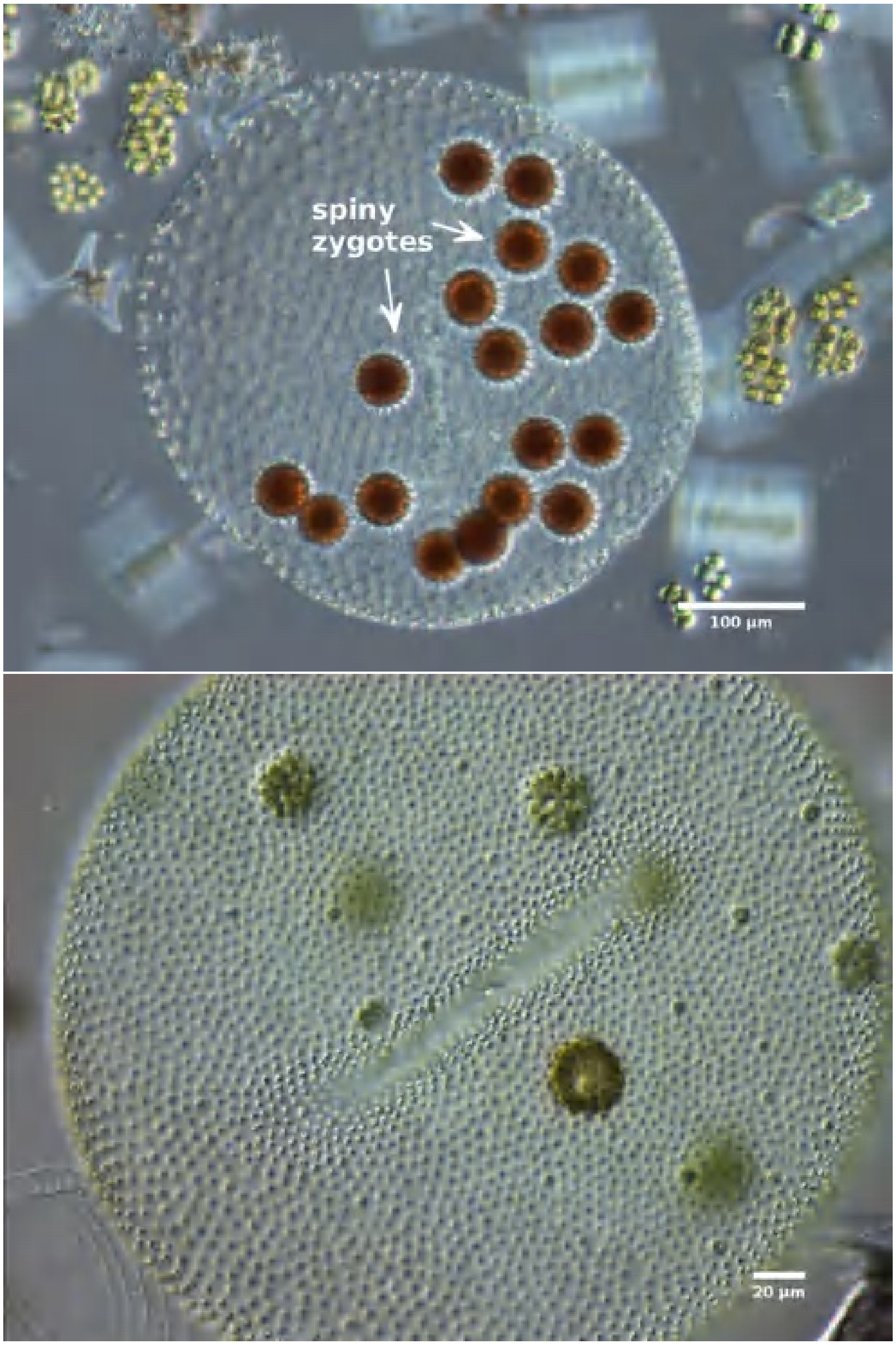


Figure 20: Chlorophyta - upper image: *Volvox globator* zygotes (100x DIC), July 22, 2013; lower image: *Volvox globator* (200x DIC), Vogler Lake, Skagit County, September 9, 2009.

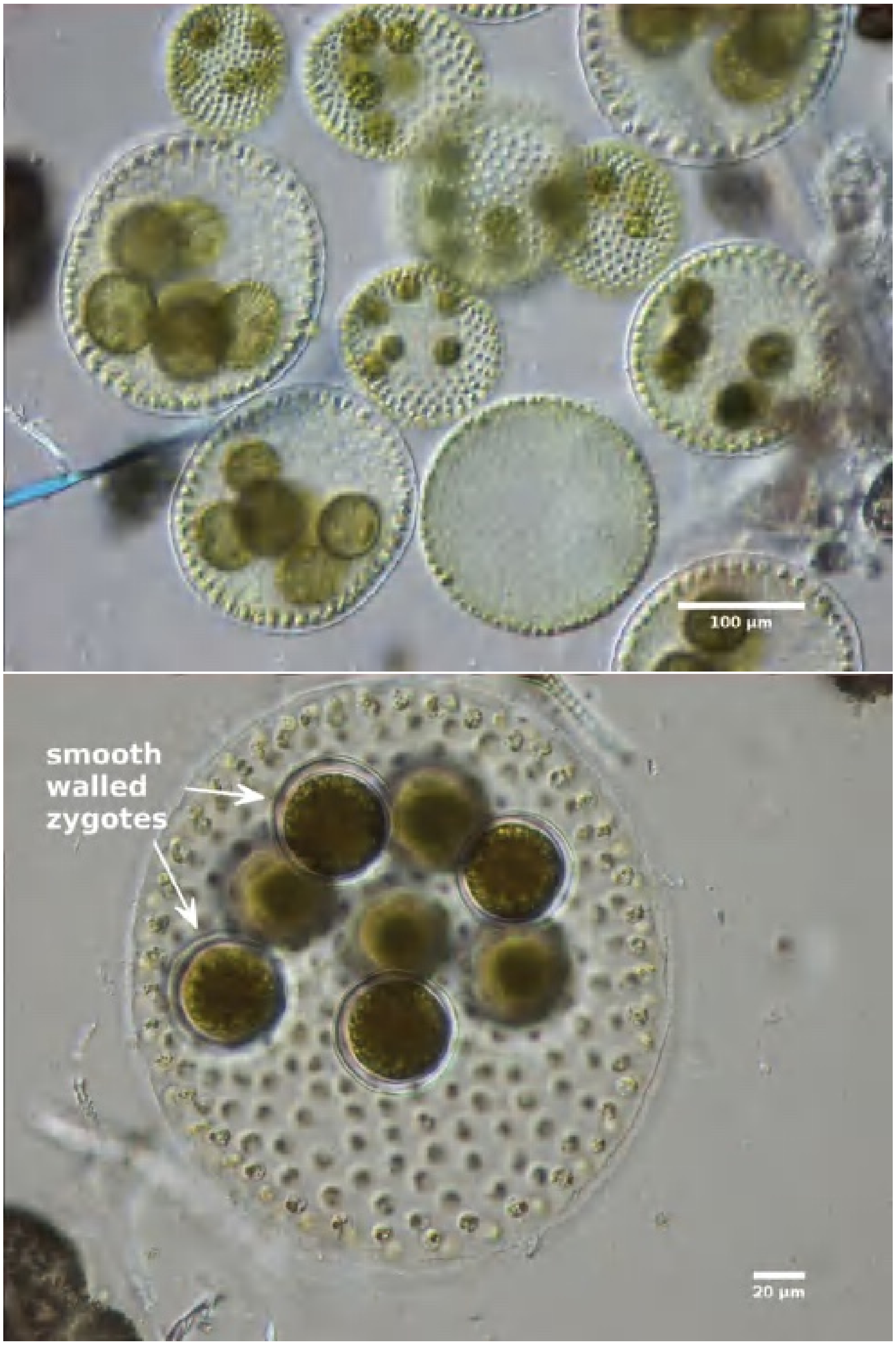


Figure 21: Chlorophyta - upper image: *Volvox tertius* bloom (100x DIC), October 1, 2009; lower image: *Volvox tertius* zygotes (200x DIC), October 1, 2009.

Figure 22: Chlorophyta/Streptophyta (desmid) - upper image: *Closterium acutum* var. *variable* (400x DIC), July 9, 2019 ; lower image: *Closterium acutum* var. *variable* (400x DIC), October 29, 2019.

Figure 23: Chlorophyta/Streptophyta (desmid) - upper image: *Closterium* (200x DIC), July 9, 2019; lower image: *Closterium* (400x DIC), June 12, 2019.

Figure 24: Chlorophyta/Streptophyta - upper image: *Elakatothrix gelatinosa* (600x DIC), October 29, 2019; lower image: *Elakatothrix gelatinosa* (400x DIC), October 2, 2019.

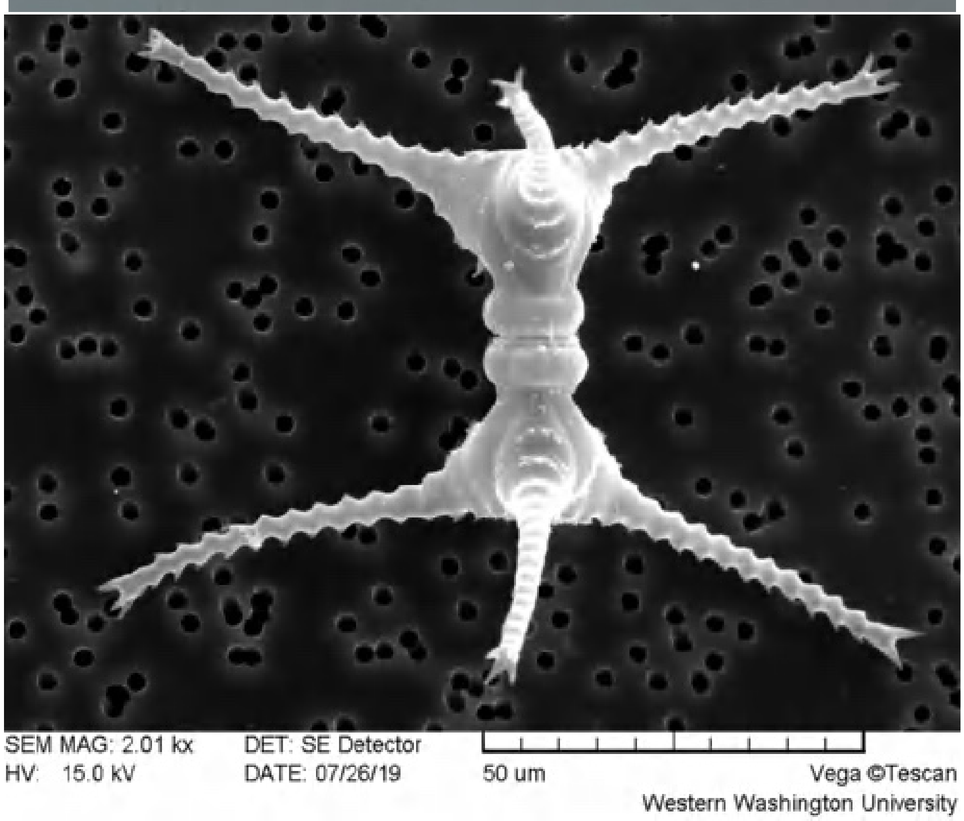


Figure 25: Chlorophyta/Streptophyta (desmid)- upper image: *Staurastrum pingue* var. *planctonicum* (400x DIC), July 9, 2019; lower image: *Staurastrum pingue* var. *planctonicum* (SEM), June 12, 2019.

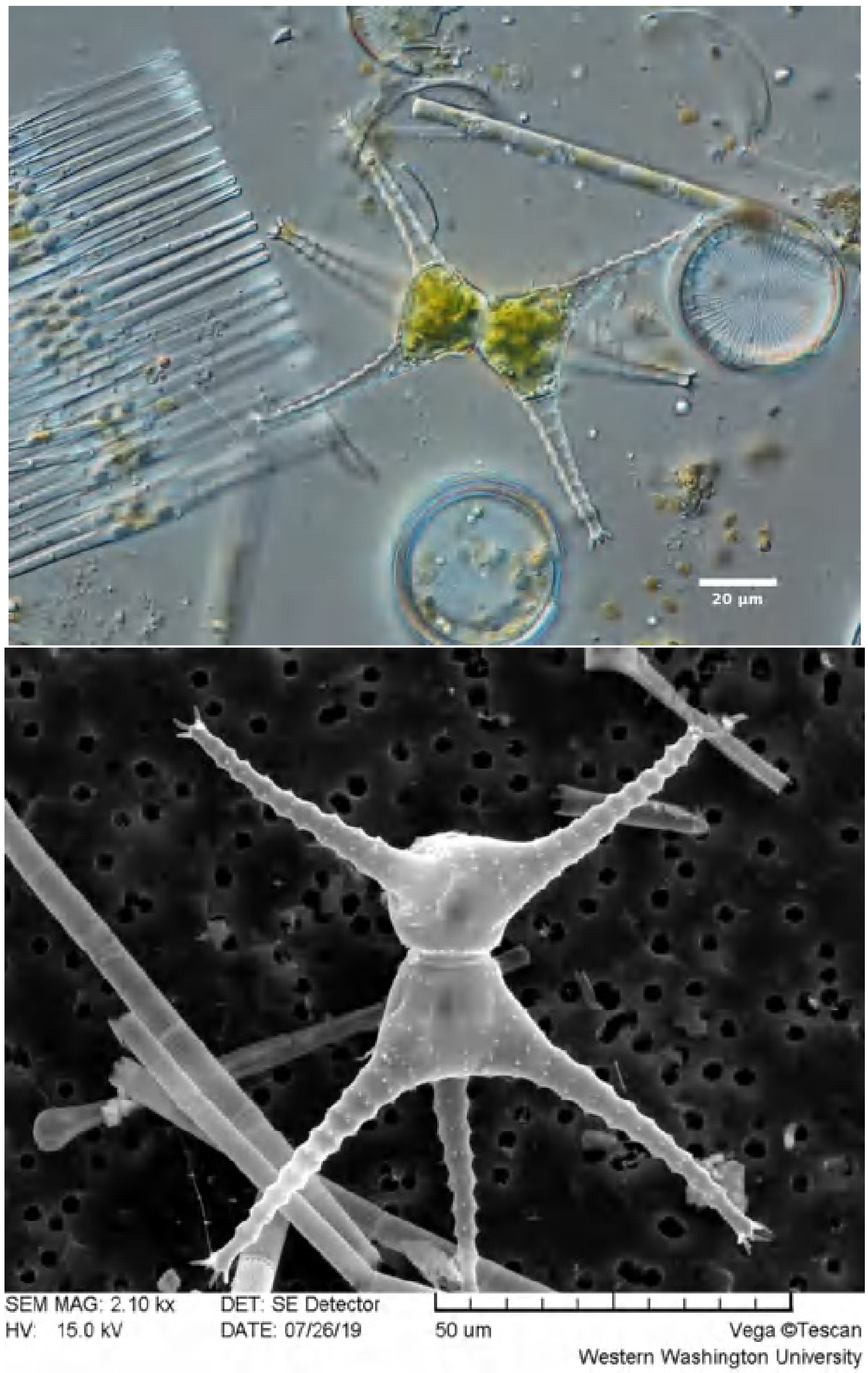


Figure 26: Chlorophyta/Streptophyta (desmid) - upper image: *Staurastrum cingulum* (400x DIC), June 13, 2019; lower image: *Staurastrum cingulum* (SEM), June 13, 2019.

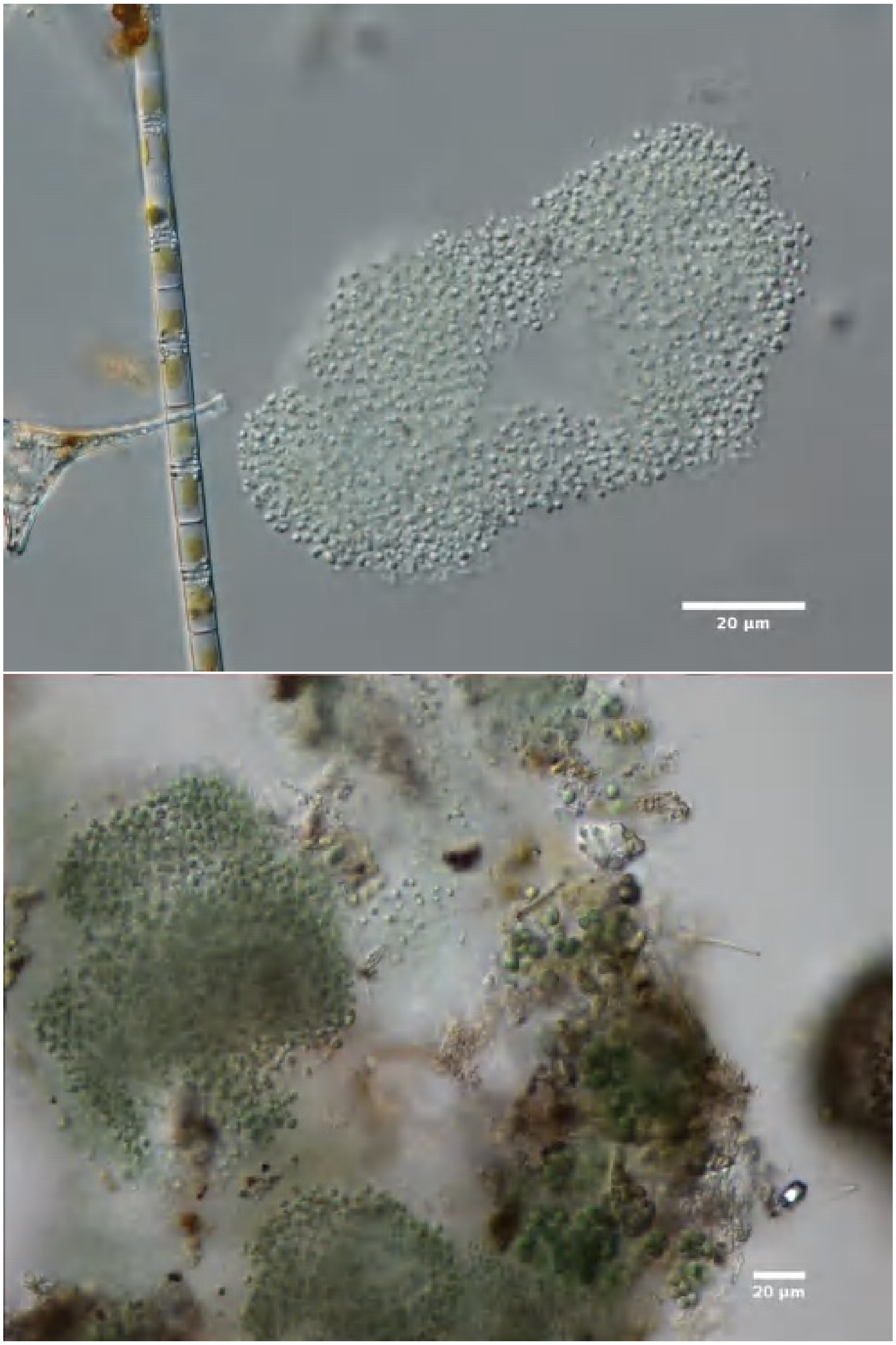


Figure 27: Cyanobacteria - upper image: *Aphanocapsa* (600x DIC), June 12, 2019; lower image: *Aphanocapsa* and *Aphanothece* (200x DIC), September 30, 2009.

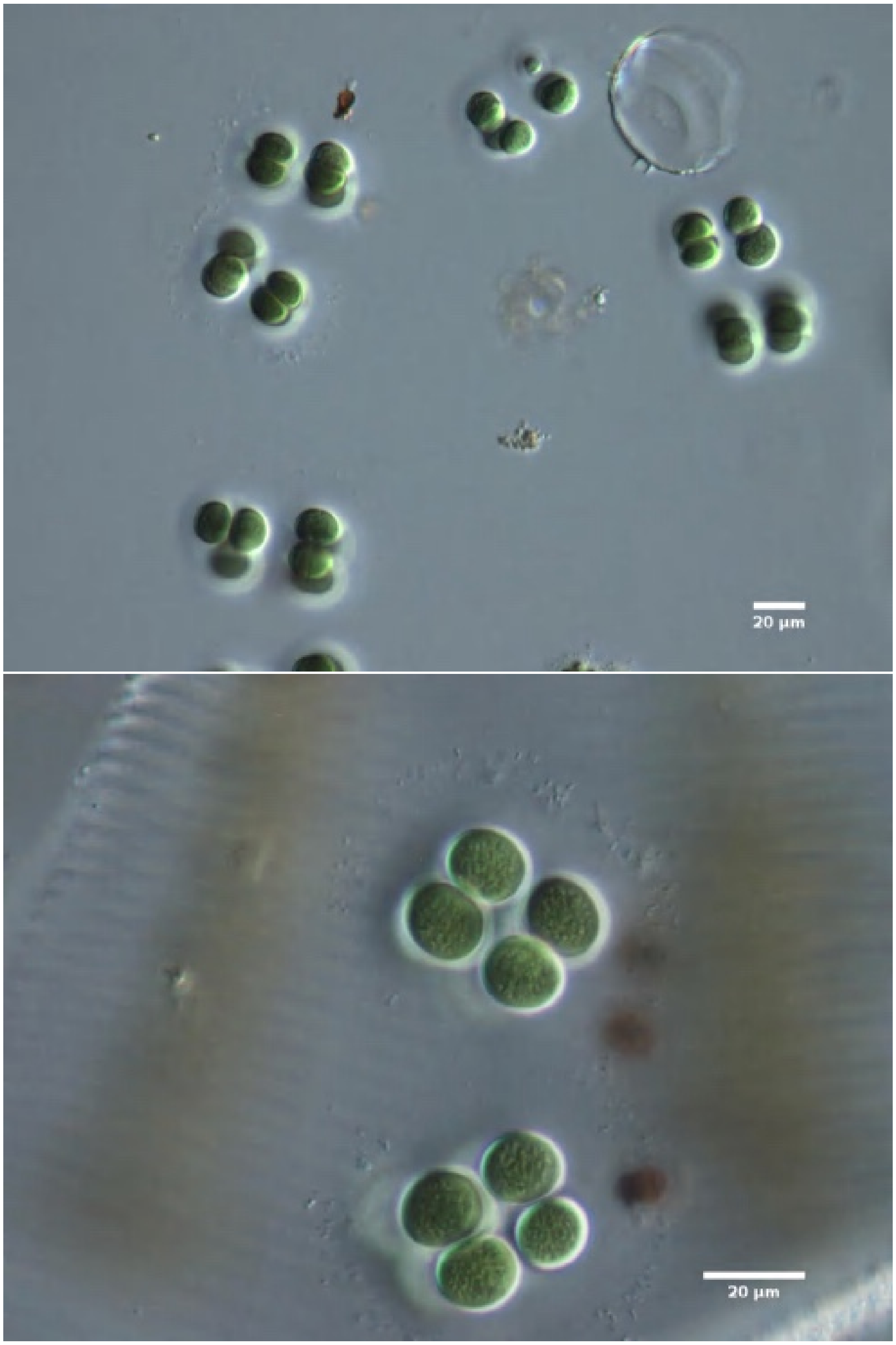


Figure 28: Cyanobacteria - upper images: *Chroococcus* (200x DIC), July 22, 2013; lower image: *Chroococcus* (400x DIC), July 22, 2013.

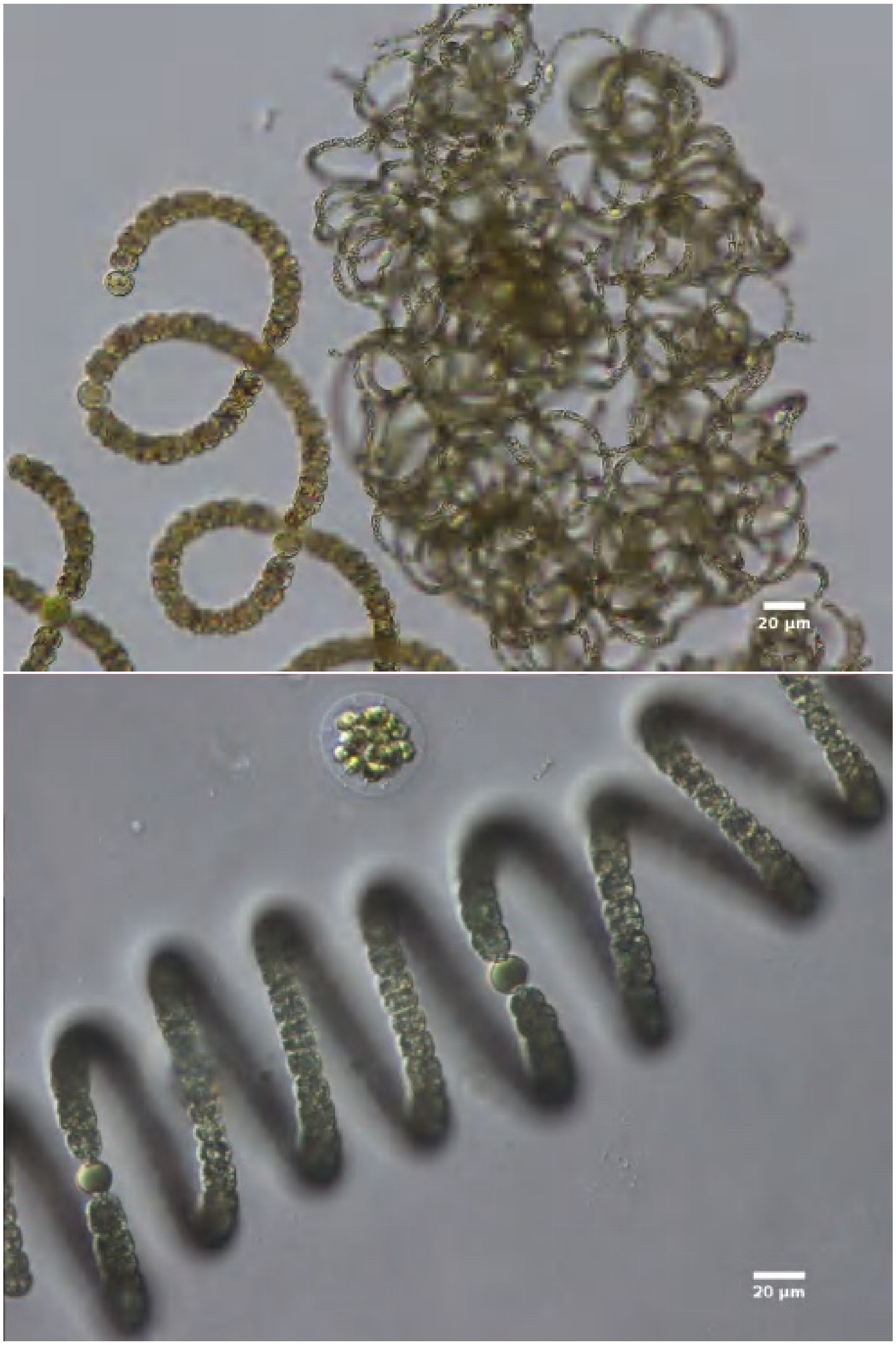


Figure 29: Cyanobacteria - upper images: Two species of *Dolichospermum* (200x DIC), August 2, 2017; lower image: *Dolichospermum crassum* (200x DIC), August 25, 2009.

Figure 30: Cyanobacteria - upper images: *Gloeotrichia echinulata* colonies (100x DIC), June 13, 2019; lower image: *Gloeotrichia echinulata* trichomes (200x DIC), June 13, 2019.

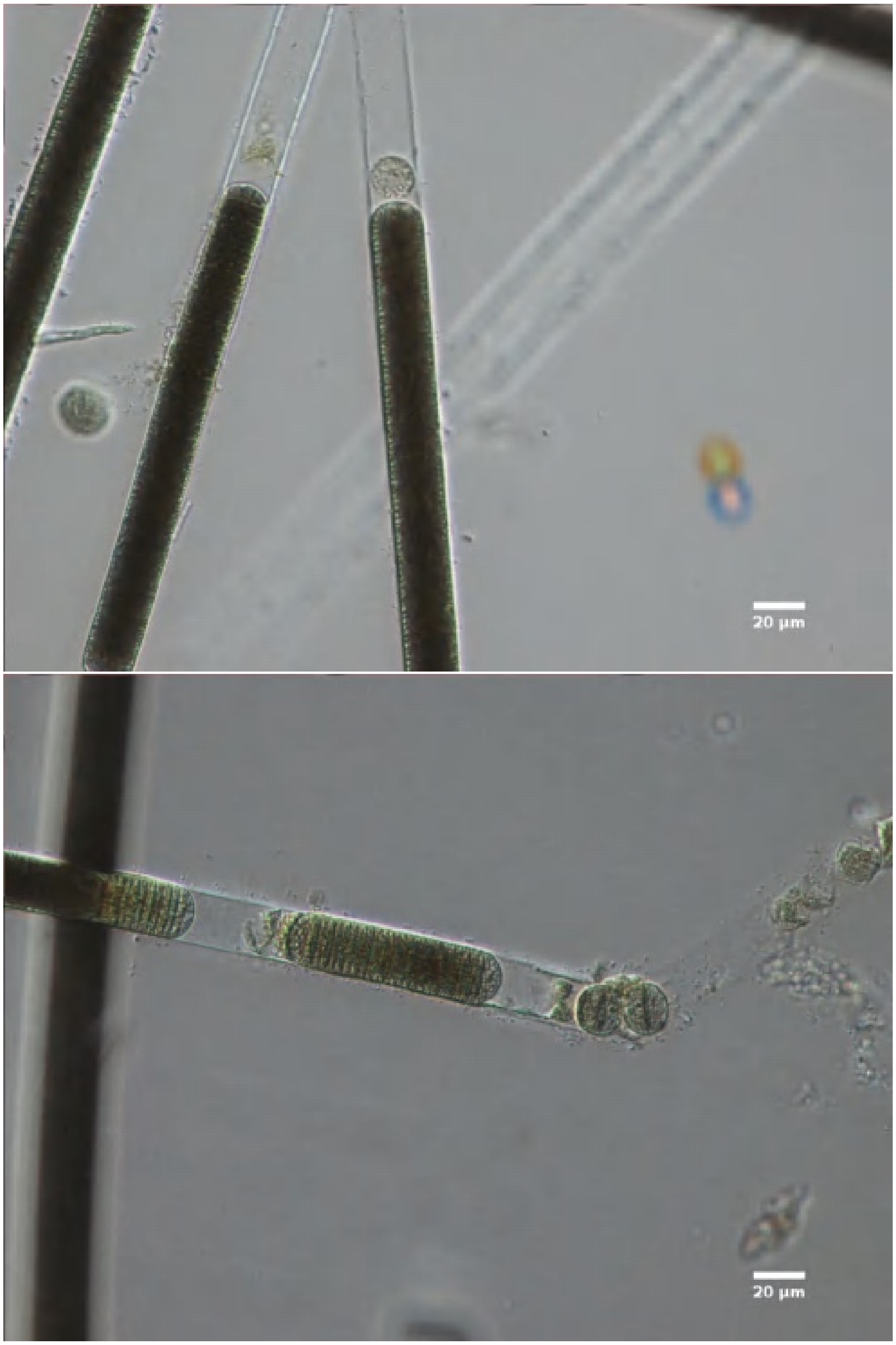


Figure 31: Cyanobacteria - upper/lower images: *Limnoraphis birgei* (200x DIC), August 28, 2008. Note presence of sheath surrounding trichome.

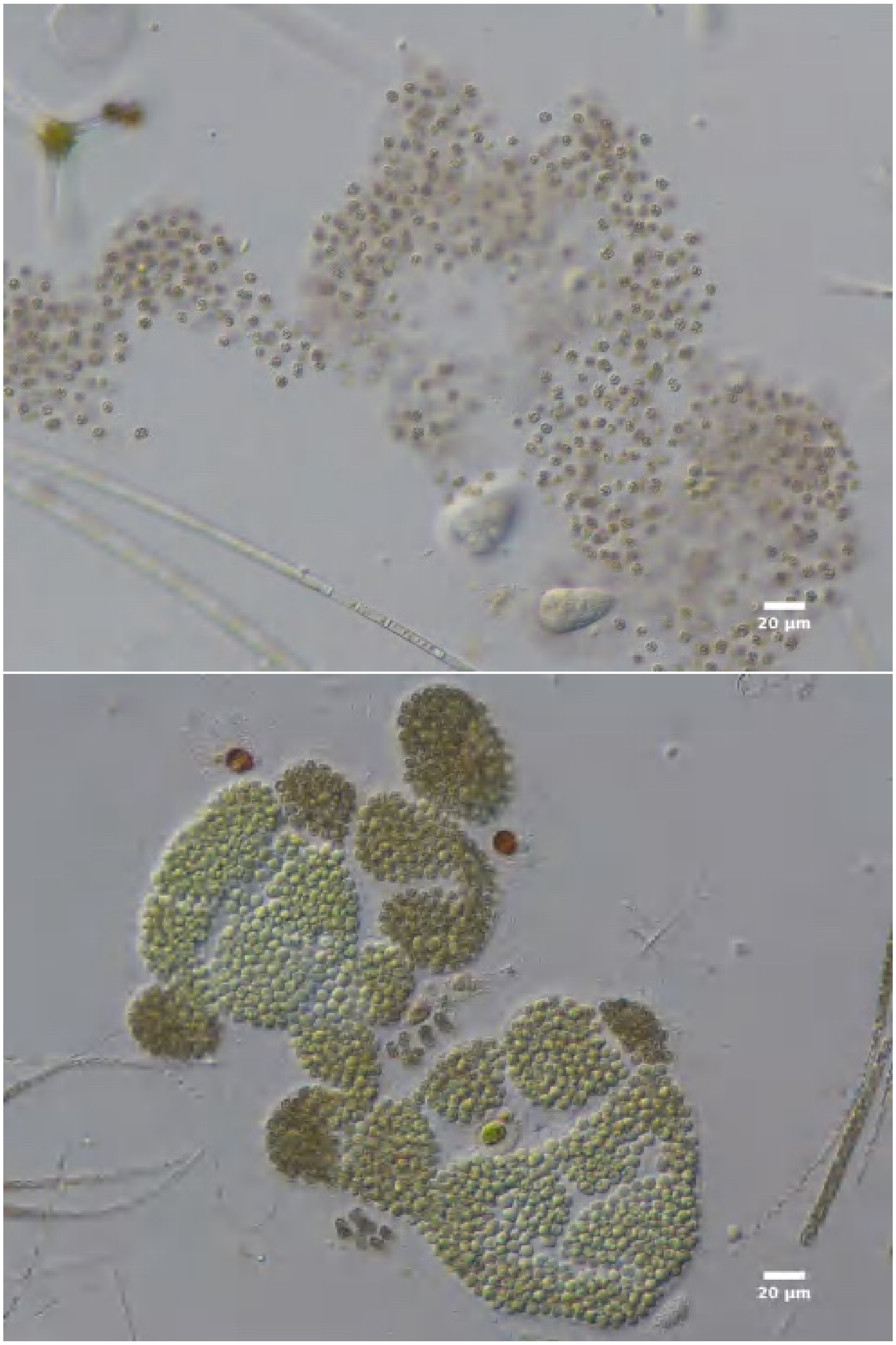


Figure 32: Cyanobacteria - upper image: *Microcystis aeruginosa* (200x DIC), June 12, 2019; lower image: Decomposing *Microcystis aeruginosa* colony (200x DIC), June 13, 2019.

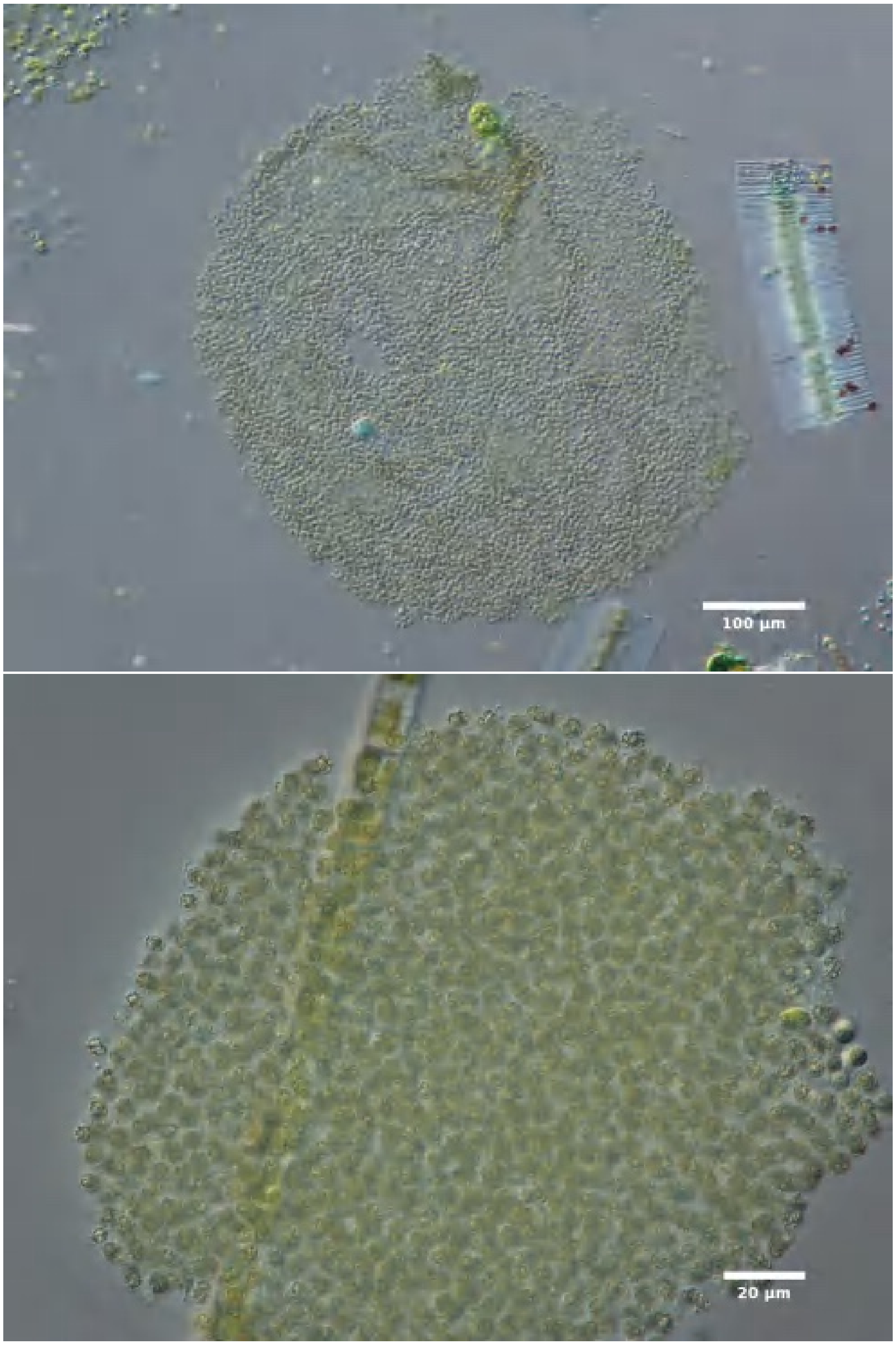


Figure 33: Cyanobacteria - upper image: *Microcystis flos-aquae*? (100x DIC), July 9, 2019; lower image: *Microcystis flos-aquae*? (400x DIC), July 9, 2019. Note tiny, densely arranged cells in spherical colony.

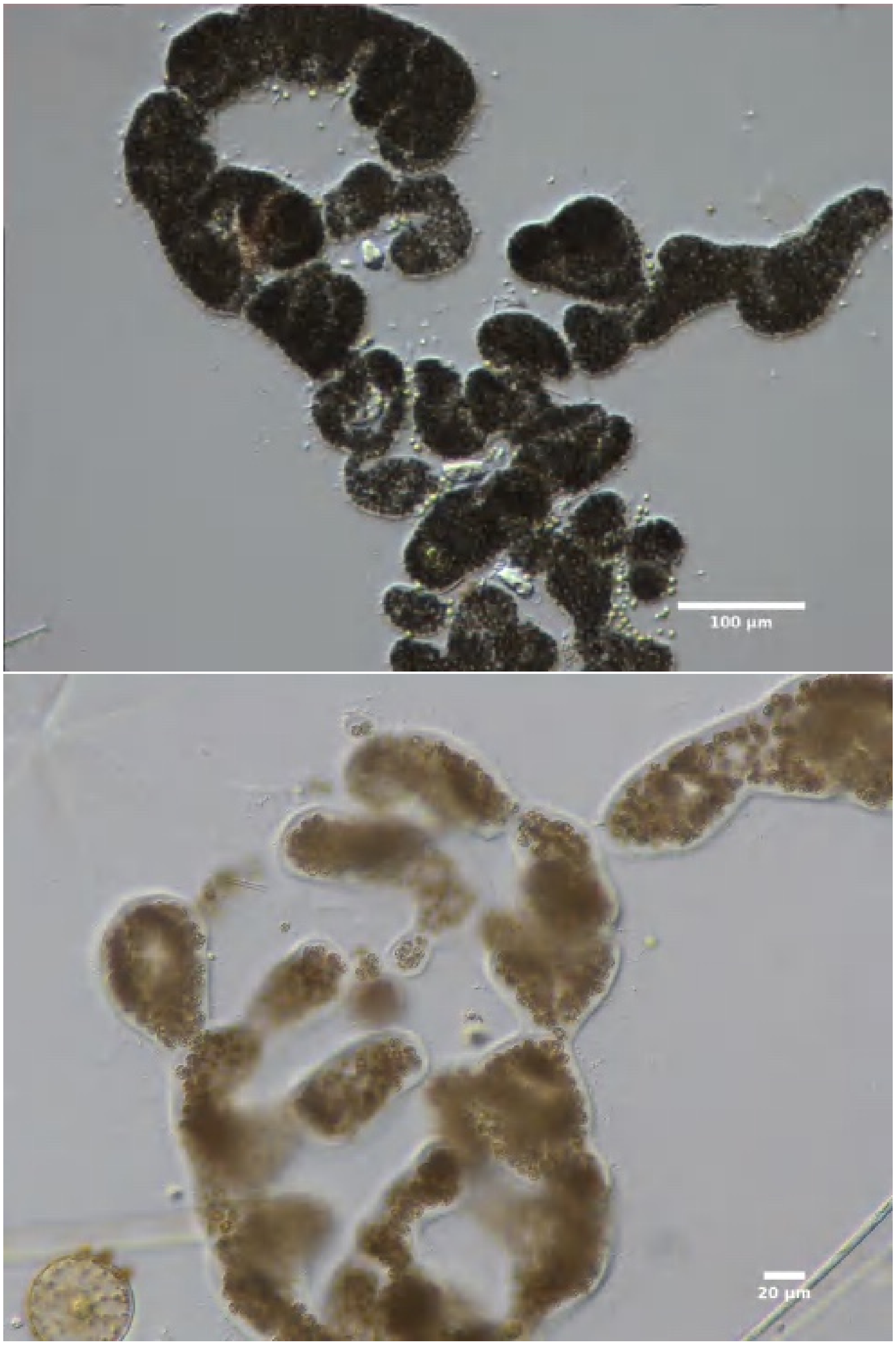


Figure 34: Cyanobacteria - upper image: *Microcystis wesenbergii* (100x DIC), July 19, 2011; lower image: *Microcystis wesenbergii* (200x DIC), June 12, 2019.

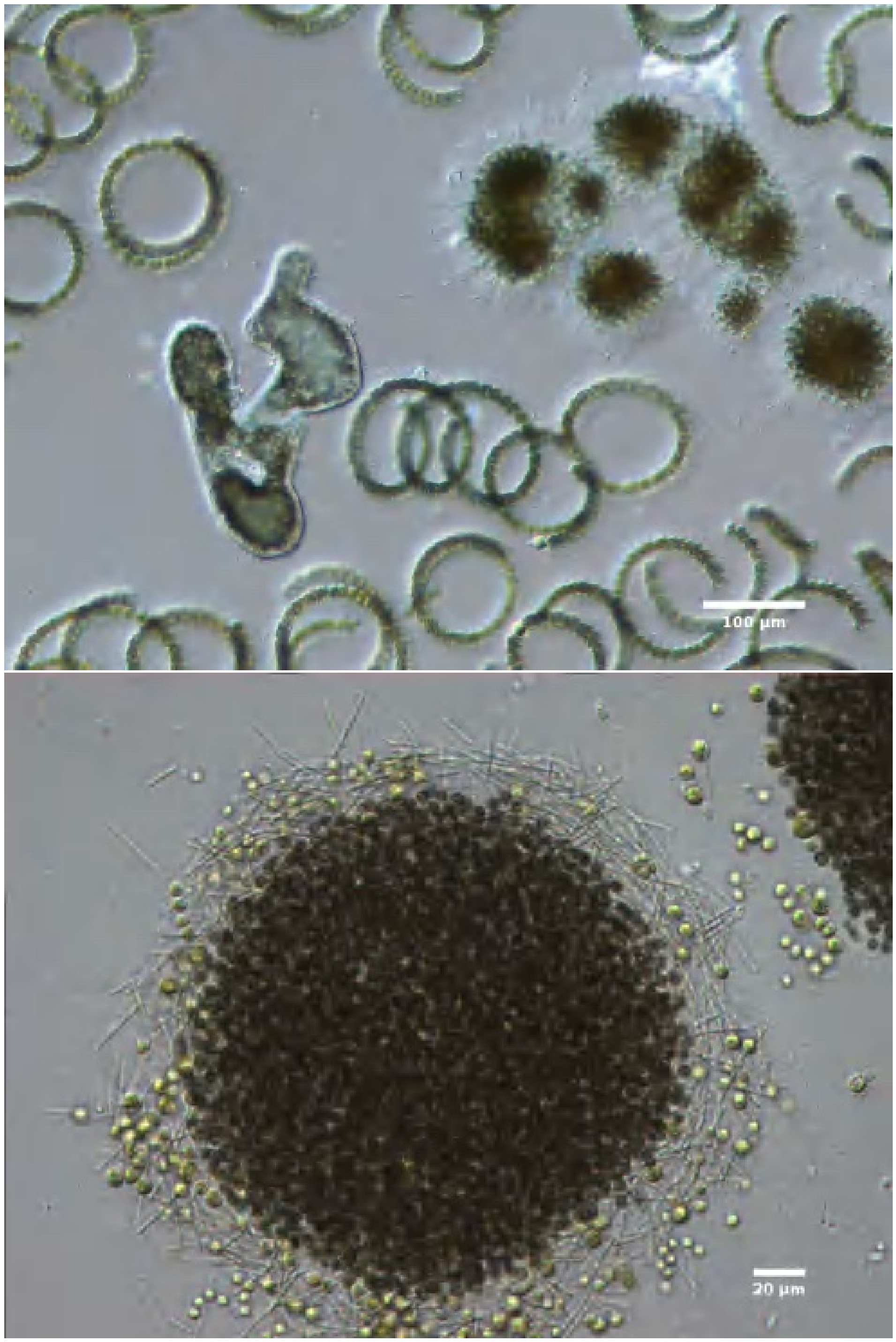


Figure 35: Cyanobacteria - upper image: *Microcystis aeruginosa* and *Microcystis wesenbergii* (100x DIC), August 4, 2014; lower image: *Microcystis* epiphytes - *Pseudanabaena* and inactive *Chlamydomonas* (200x DIC), July 19, 2011.

Figure 36: Cyanobacteria - upper image: *Oscillatoria* (400x DIC), October 2, 2019; lower image: *Oscillatoria* (200x DIC), July 9, 2019. Note absence of sheath surrounding trichome.

Figure 37: Cyanobacteria - upper image: *Phormidium* (200x DIC), September 30, 2009; lower image: *Phormidium* (200x DIC), June 12, 2019.

Figure 38: Cyanobacteria - upper image: *Pseudanabaena mucicola* surrounding *Microcystis aeruginosa* (200x DIC), August 4, 2014; lower image: *Pseudanabaena mucicola* (600x DIC), August 4, 2014.

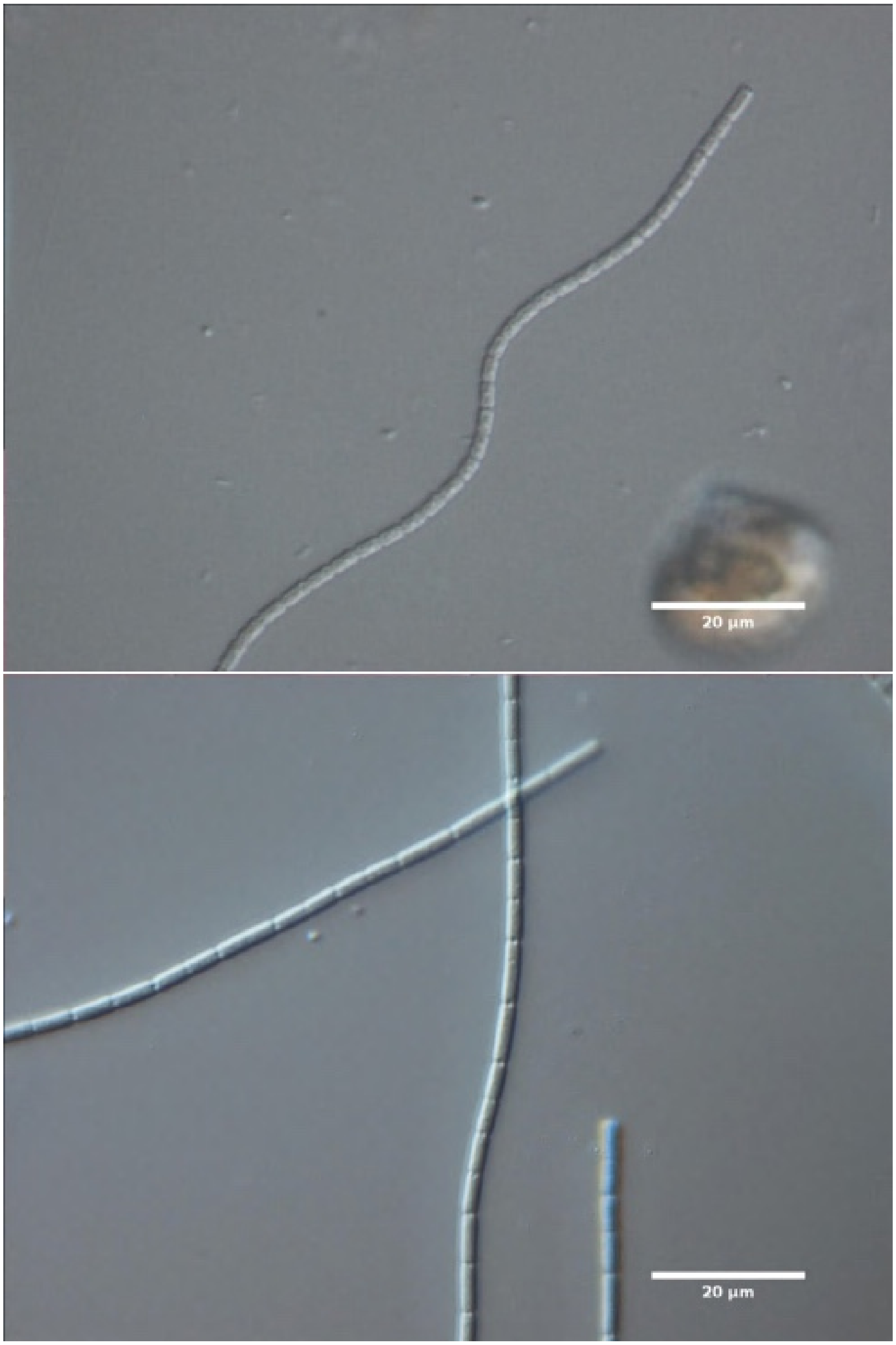


Figure 39: Cyanobacteria - upper image: *Pseudanabaena* (600x DIC), Lake Whatcom, Whatcom County, October 12, 2010; lower image: *Pseudanabaena* (600x DIC), Lake Fazon, Whatcom County, September 23, 2008.

Figure 40: Cyanobacteria - upper image: *Tolypothrix lanata* (200x DIC), September 30, 2009; lower image: *Tolypothrix lanata* (600x DIC), Tennant Lake, Whatcom County, June 22, 2017.

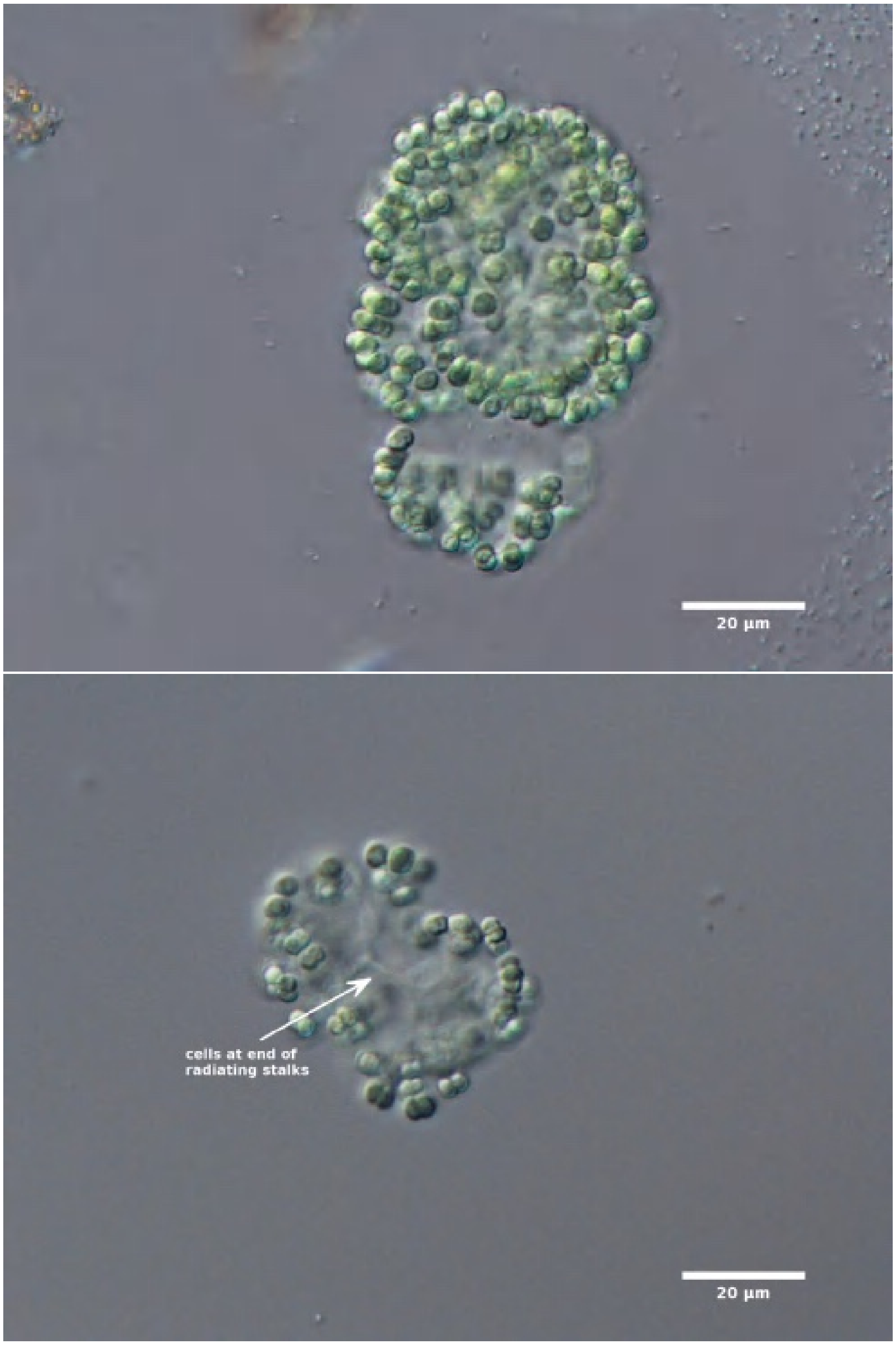


Figure 41: Cyanobacteria - upper image: *Woronichinia compacta*? (600x DIC), October 15, 2019; lower image: *Woronichinia compacta*? (600x DIC), October 29, 2019. Identification is based on presence of thick, radiating mucilage strands, these strands are absent in *Coelosphaerium* and are thinner in *Snowella*.

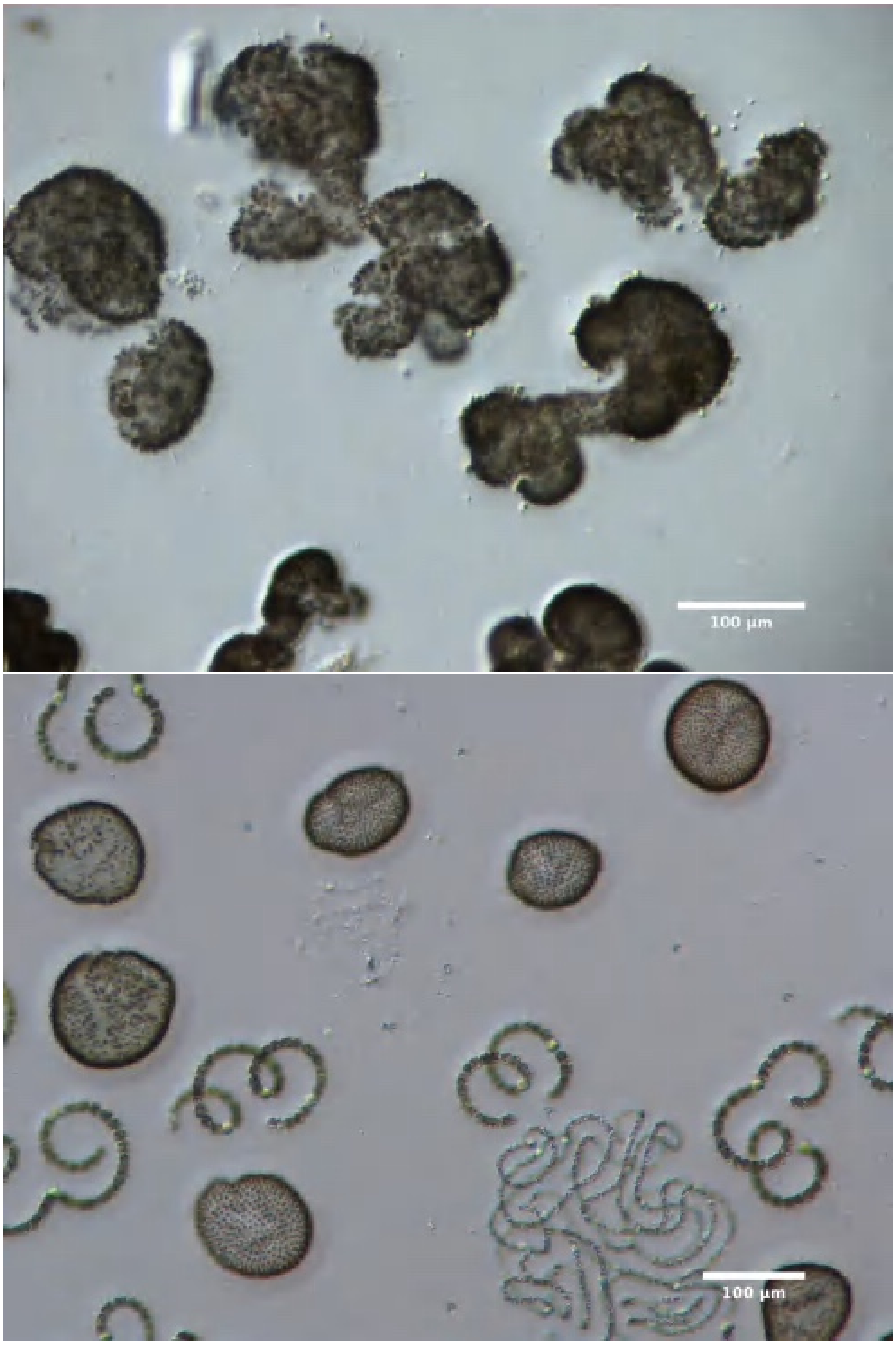


Figure 42: Cyanobacteria - upper image: *Woronichinia naegeliana* (100x DIC), September 30, 2009; lower image: *Woronichinia naegeliana* (100x DIC), Heart Lake, Skagit County, May 12, 2016.

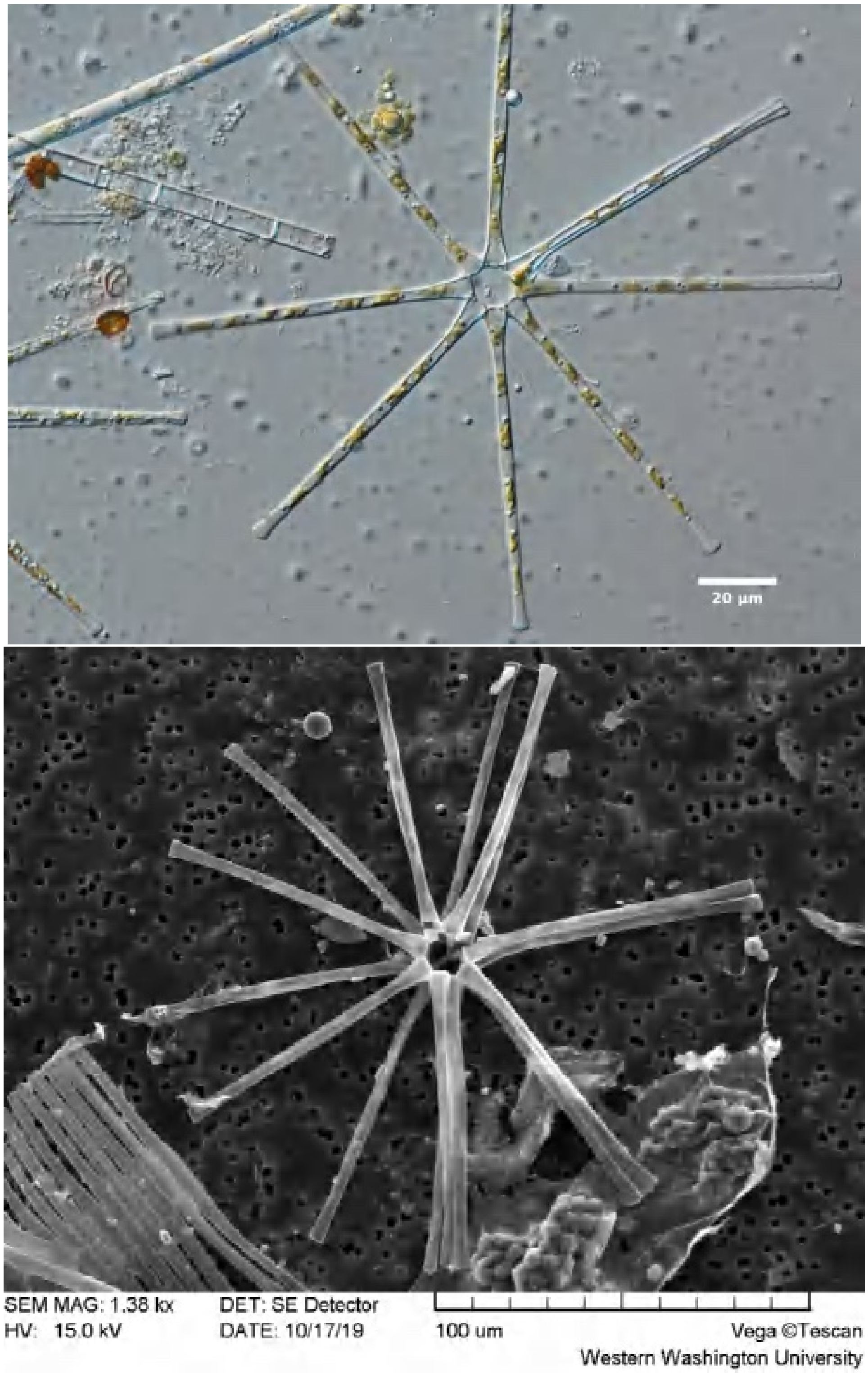
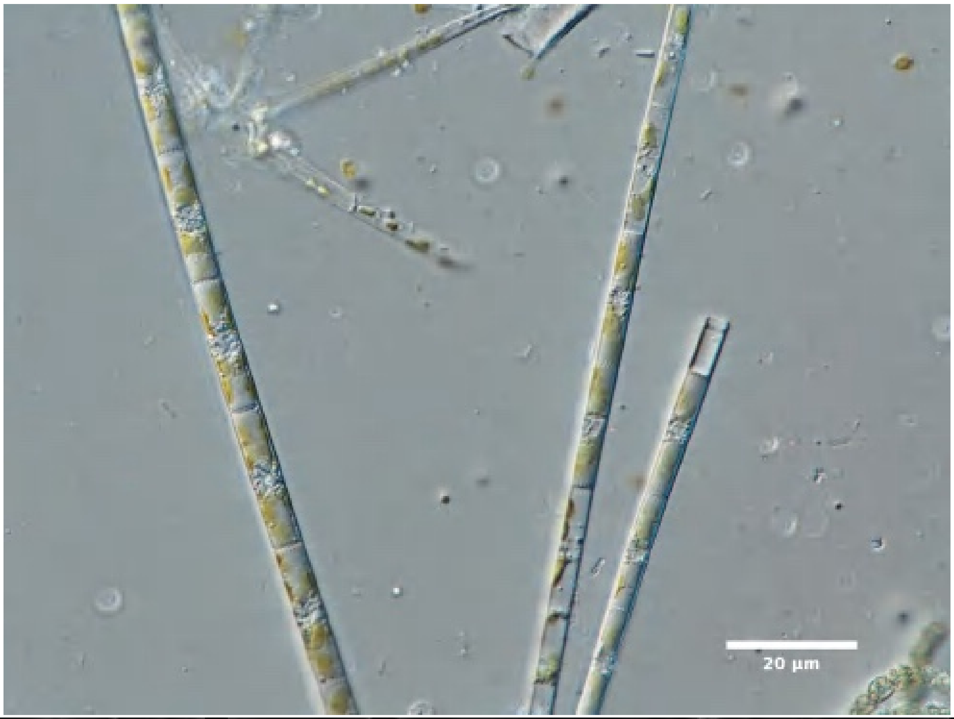


Figure 43: Other/Bacilliariophyta - upper image: *Asterionella formosa* (SEM), July 9, 2019; lower image: *Asterionella formosa* (400x DIC), June 13, 2019.



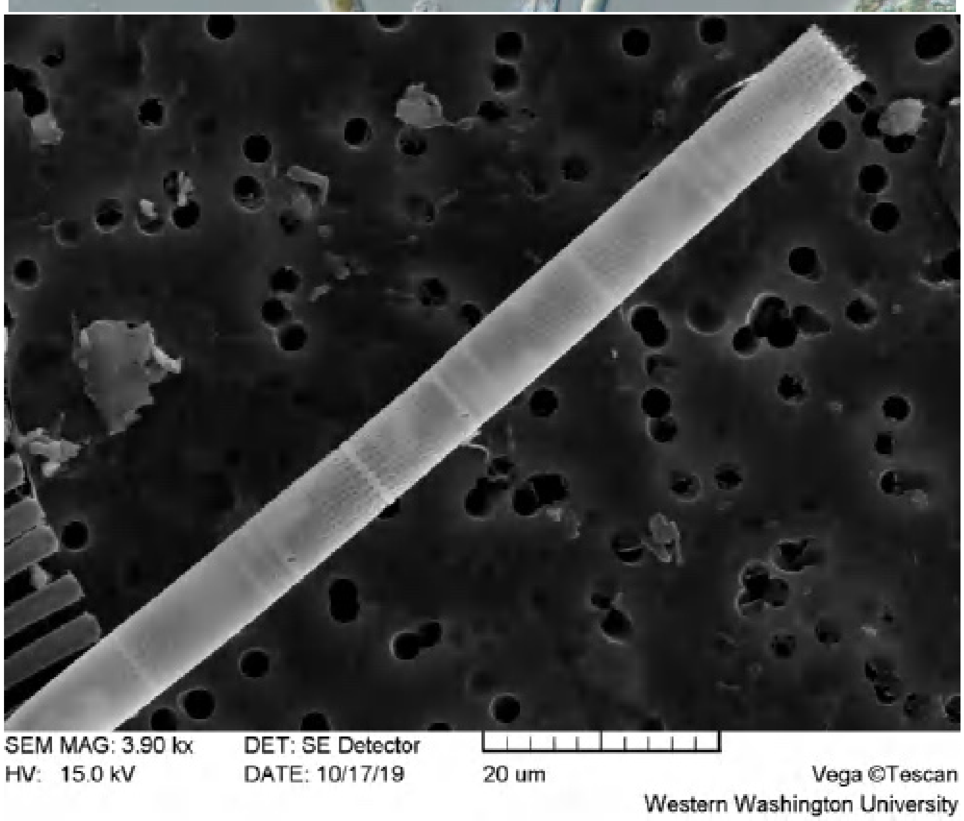


Figure 44: Other/Bacilliariophyta - upper image: *Aulacoseira ambigua* (SEM), July 9, 2019; lower image: *Aulacoseira ambigua* (600x DIC), June 13, 2019.

Figure 45: Other/Bacilliariophyta - upper/lower images: *Eunotia* (600s DIC), October 2, 2019.

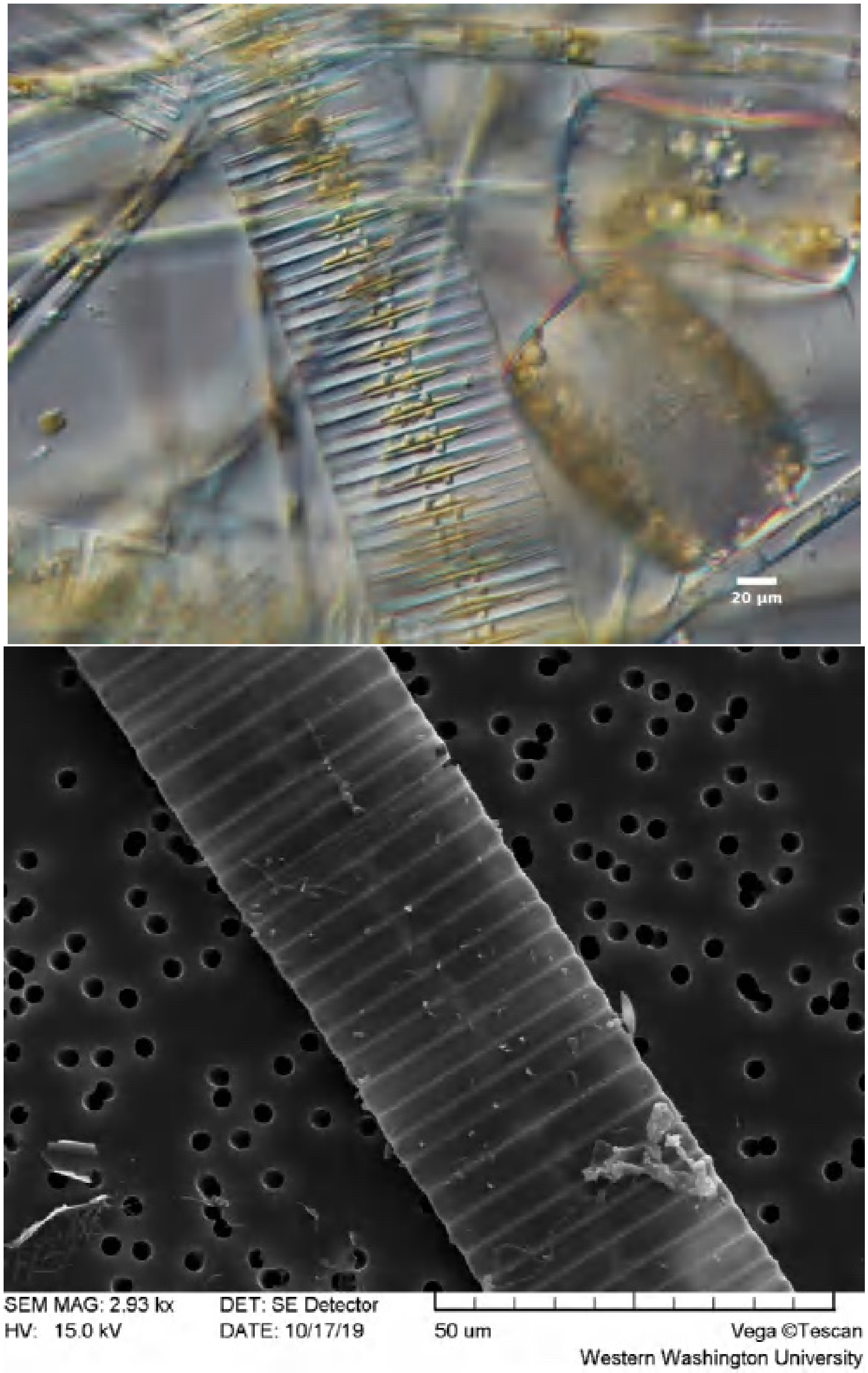


Figure 46: Other/Bacilliariophyta - upper image: *Fragilaria capucina* (SEM), July 9, 2019; lower image: *Fragilaria capucina* (200x DIC), June 13, 2019.

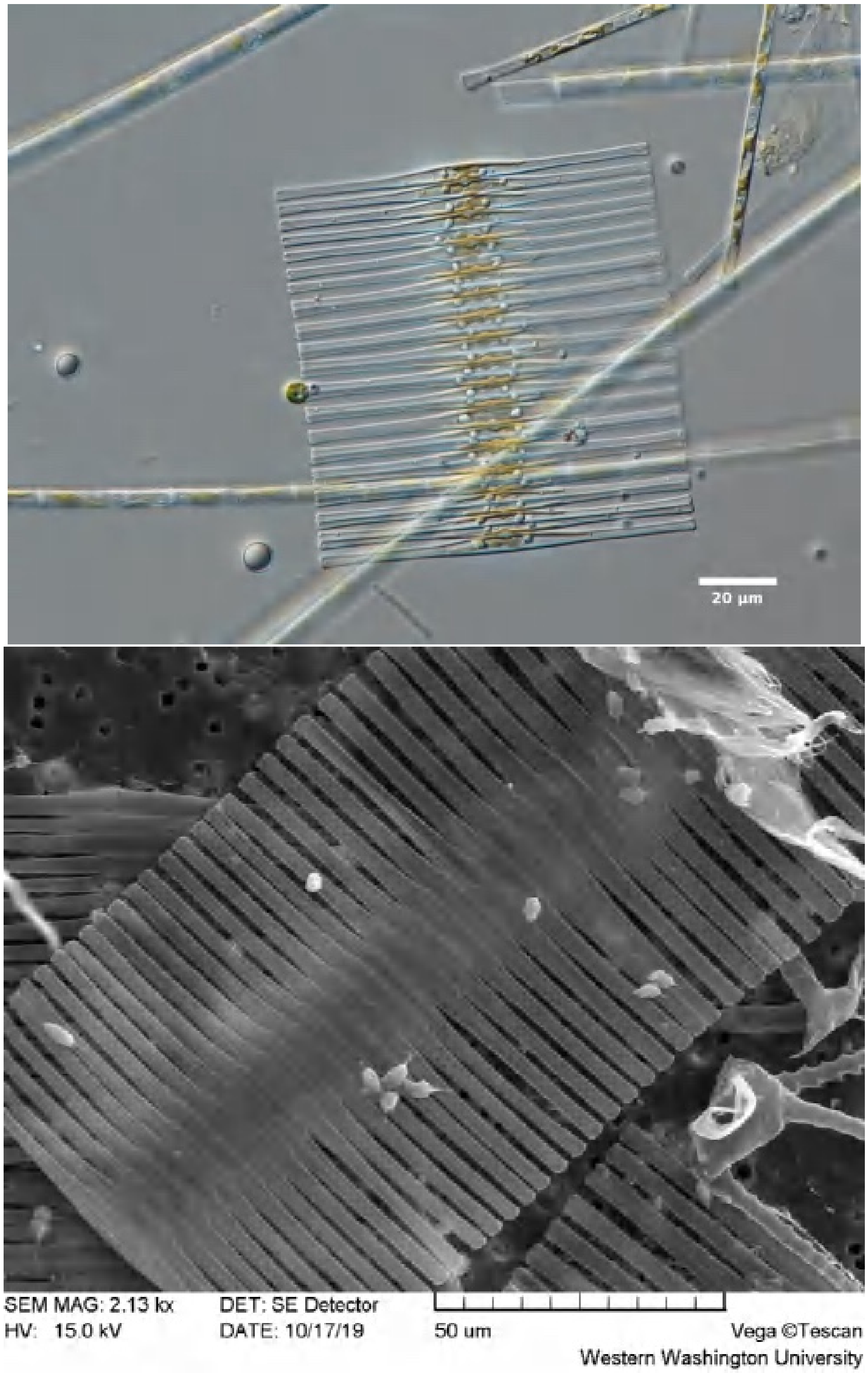


Figure 47: Other/Bacilliariophyta - upper image: *Fragilaria crotonensis* (SEM), July 9, 2019; lower image: *Fragilaria crotonensis* (200x DIC), June 12, 2019.

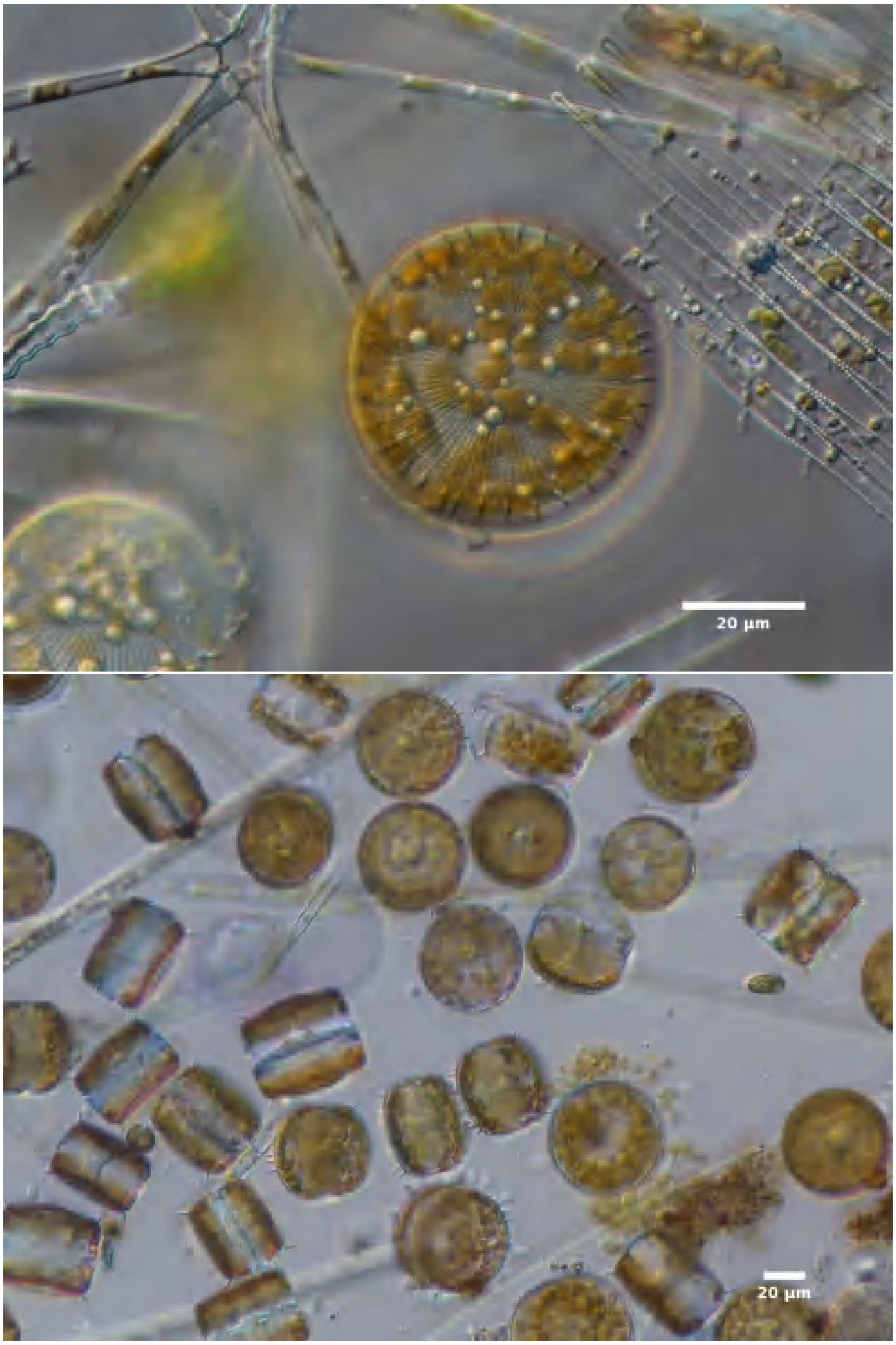


Figure 48: Other/Bacilliariophyta - upper image: *Stephanodiscus niagarae* (600x DIC), June 13, 2019; lower image: *Stephanodiscus niagarae* bloom (200x DIC), October 15, 2019.

Figure 49: Other/Bacilliariophyta - upper image: *Surirella* (400x DIC), September 30, 2009; lower image: *Surirella* (200x DIC), September 30, 2009.

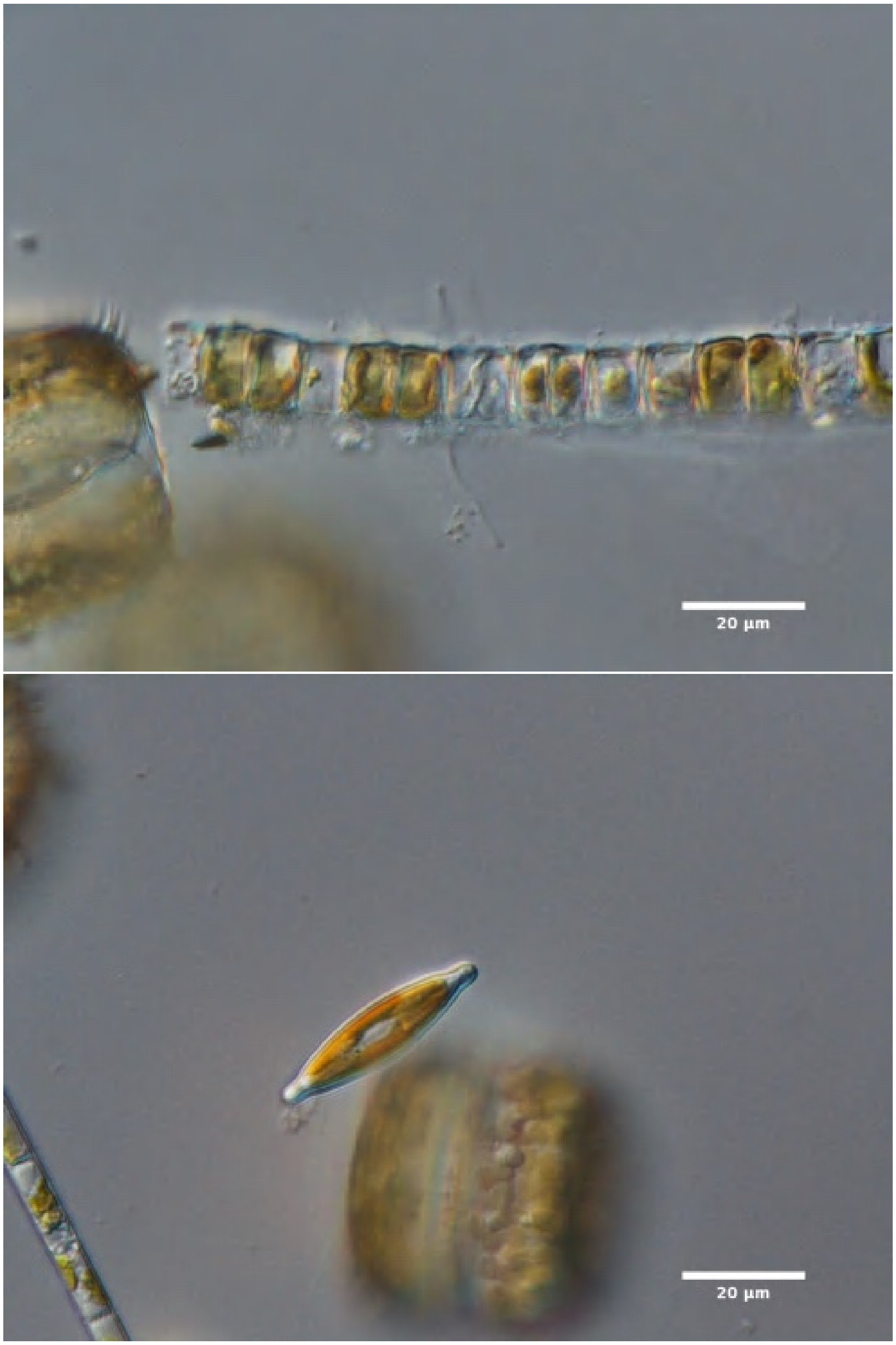


Figure 50: Other/Bacilliariophyta - upper/lower images: unknown filamentous and naviculoid diatoms (600x DIC), October 22, 2019.

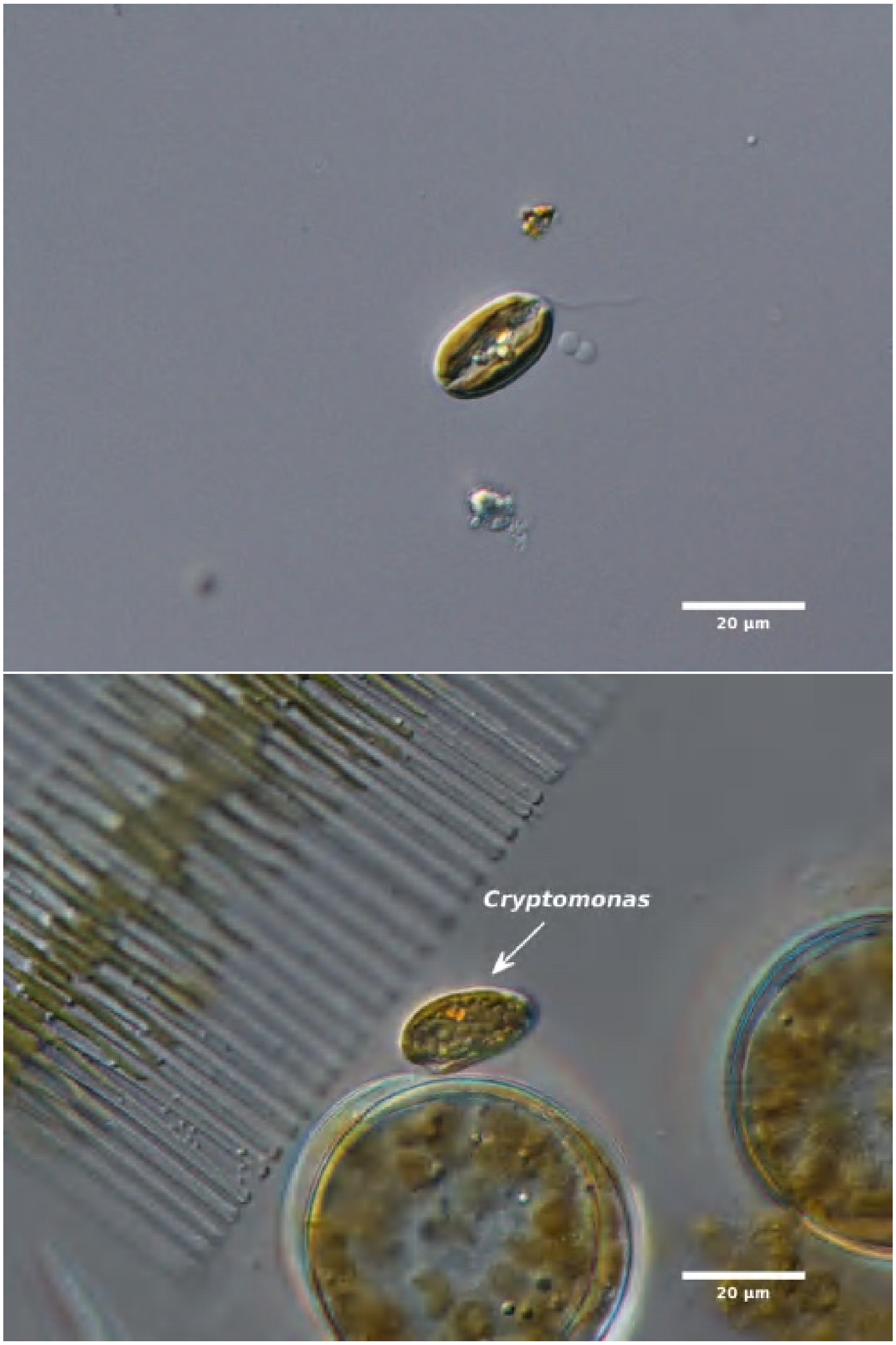


Figure 51: Other/Cryptophyta - upper/lower images: *Cryptomonas* (600x DIC), October 15, 2019.

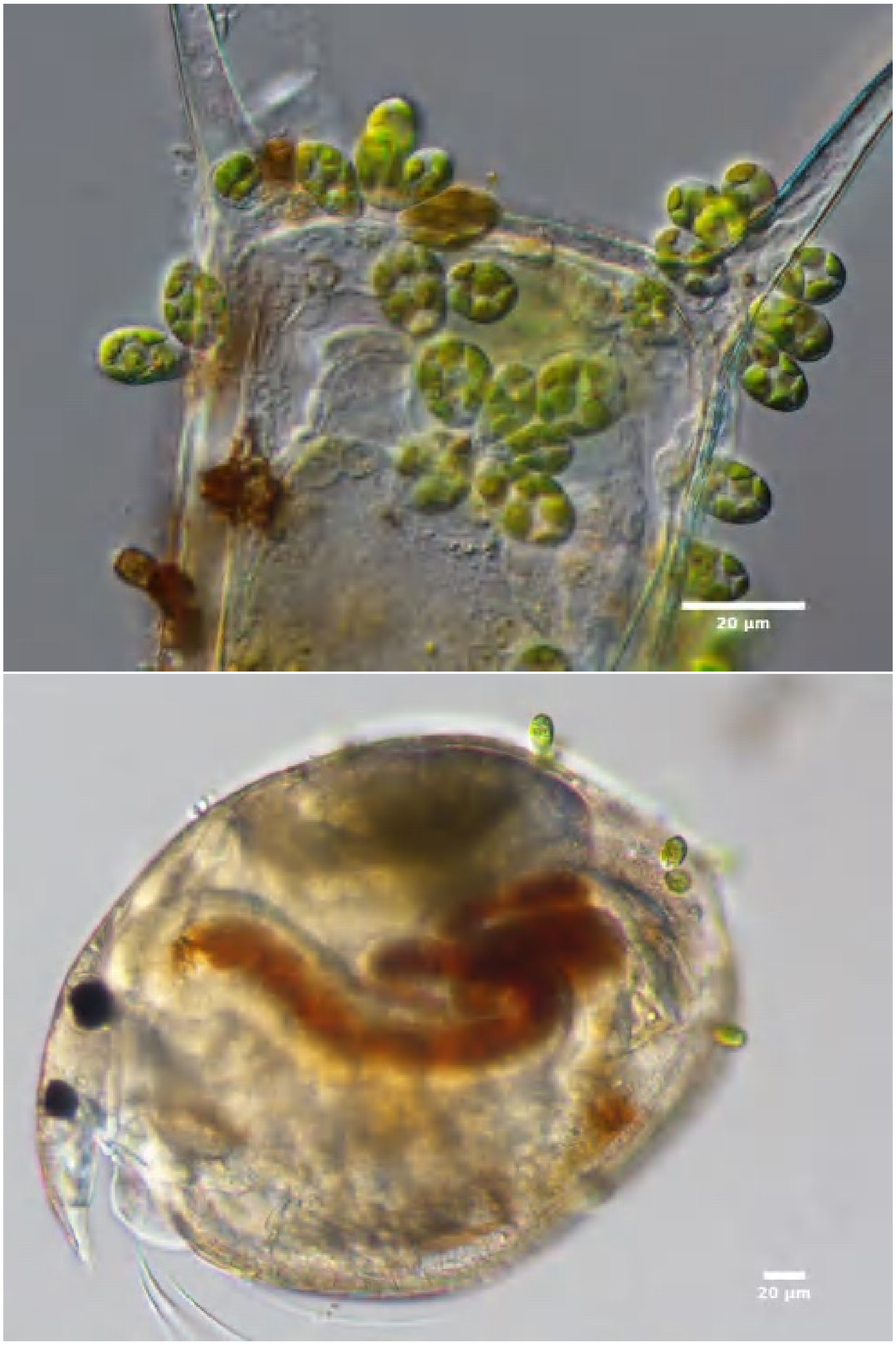


Figure 52: Other/Euglenophyta - upper image: *Colacium vesiculosum* 600x DIC, July 9, 2019; lower image: *Colacium vesiculosum* (200x DIC), Tennant Lake, Whatcom County, April 2, 2015.

Figure 53: Other/Euglenophyta - upper image: *Euglena* 400x DIC, June 13, 2019; lower image: *Euglena* (600x DIC), October 15, 2019.

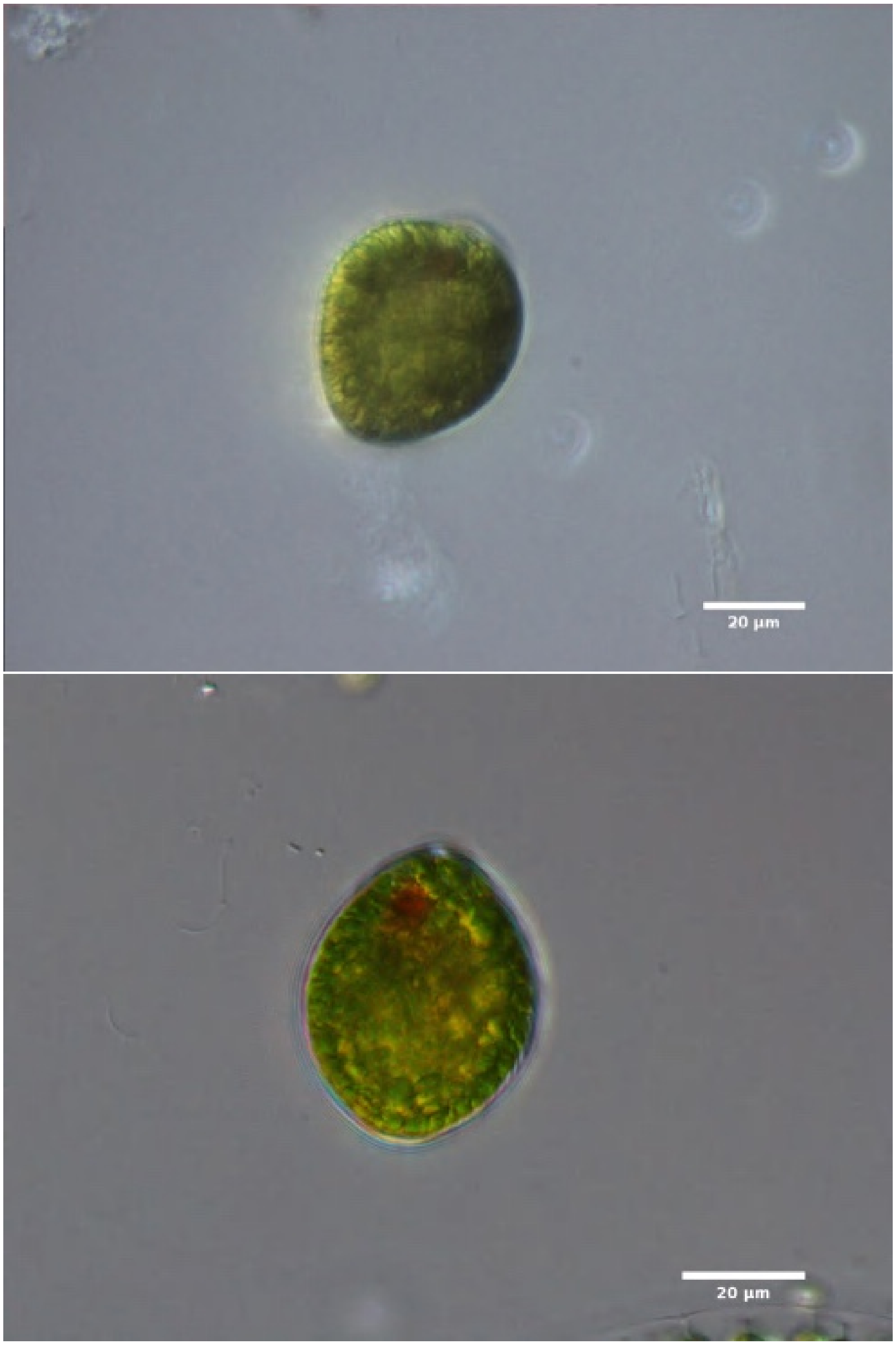


Figure 54: Other/Euglenophyta - upper image: *Euglena texta* (400x DIC), August 25, 2009; lower image: *Euglena texta* (600x DIC), July 9, 2019.

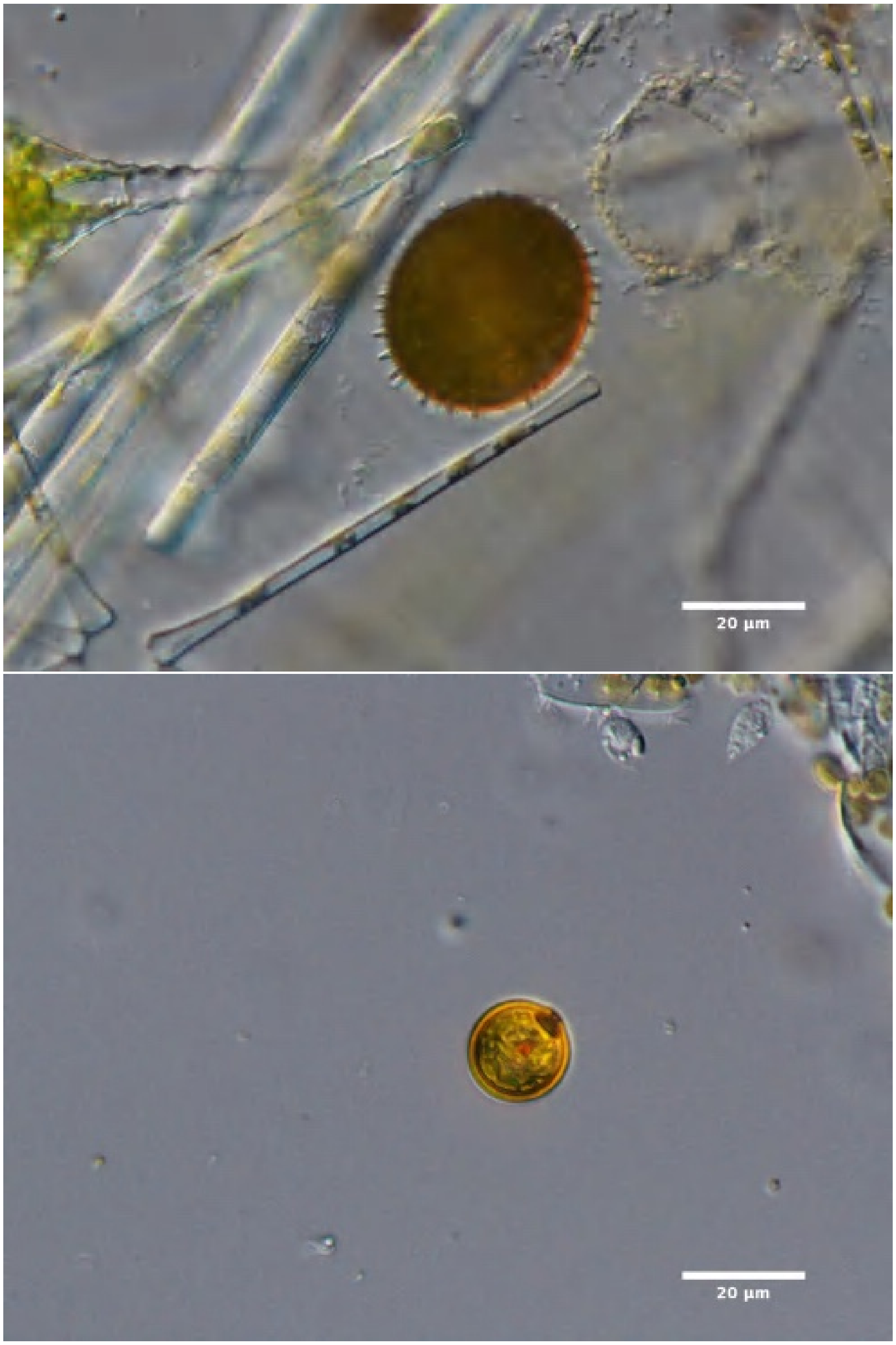


Figure 55: Other/Euglenophyta - upper image: *Trachelomonas hispida* (600x DIC), June 13, 2019; lower image: *Trachelomonas volvocinopsis* (600x DIC), October 15, 2019.

Figure 56: Other/Miozoa - upper image: *Ceratium hirundinella*¡ (200x DIC), September 30, 2009; lower image: *Ceratium hirundinella* zygote (400x DIC), July 9, 2019.

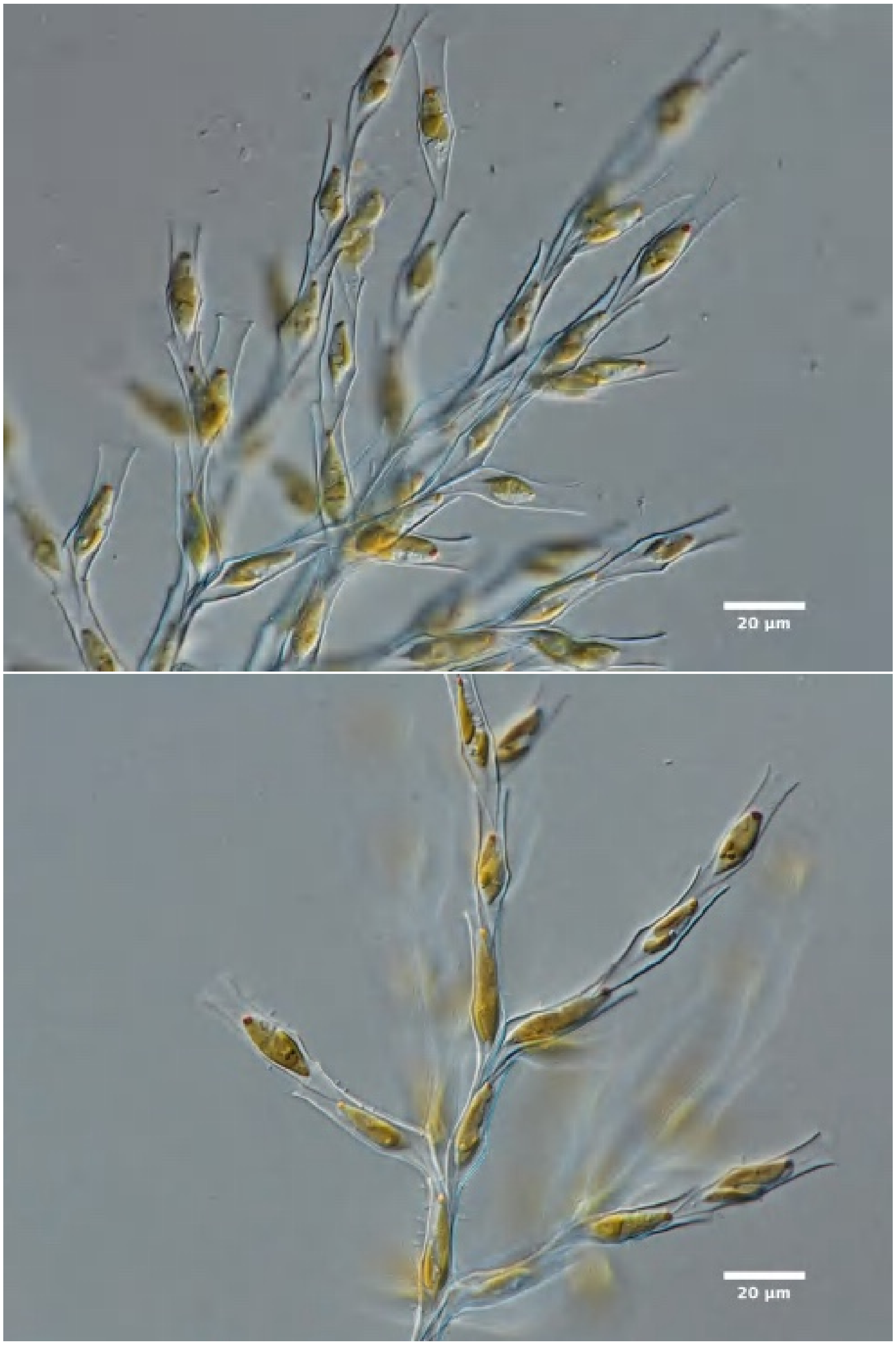


Figure 57: Other/Ochrophyta - upper image: *Dinobryon divergens* (400x DIC), July 9, 2019; lower image: *Dinobryon divergens* (400x DIC), July 9, 2019.

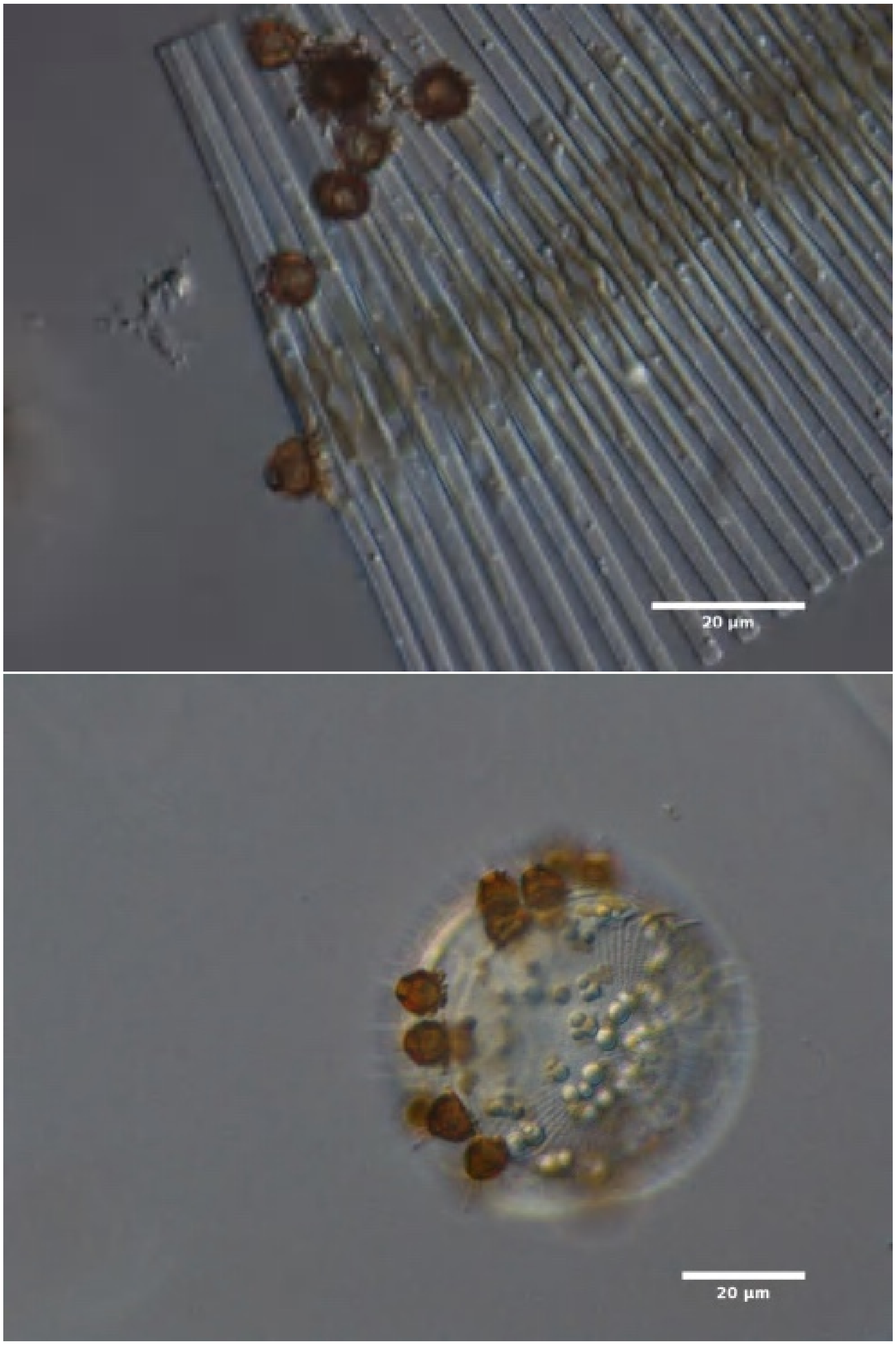


Figure 58: Other/Ochrophyta - upper image: *Lagynion* (600x DIC), July 22, 2013; lower image: *Lagynion* (600x DIC), June 12, 2019.

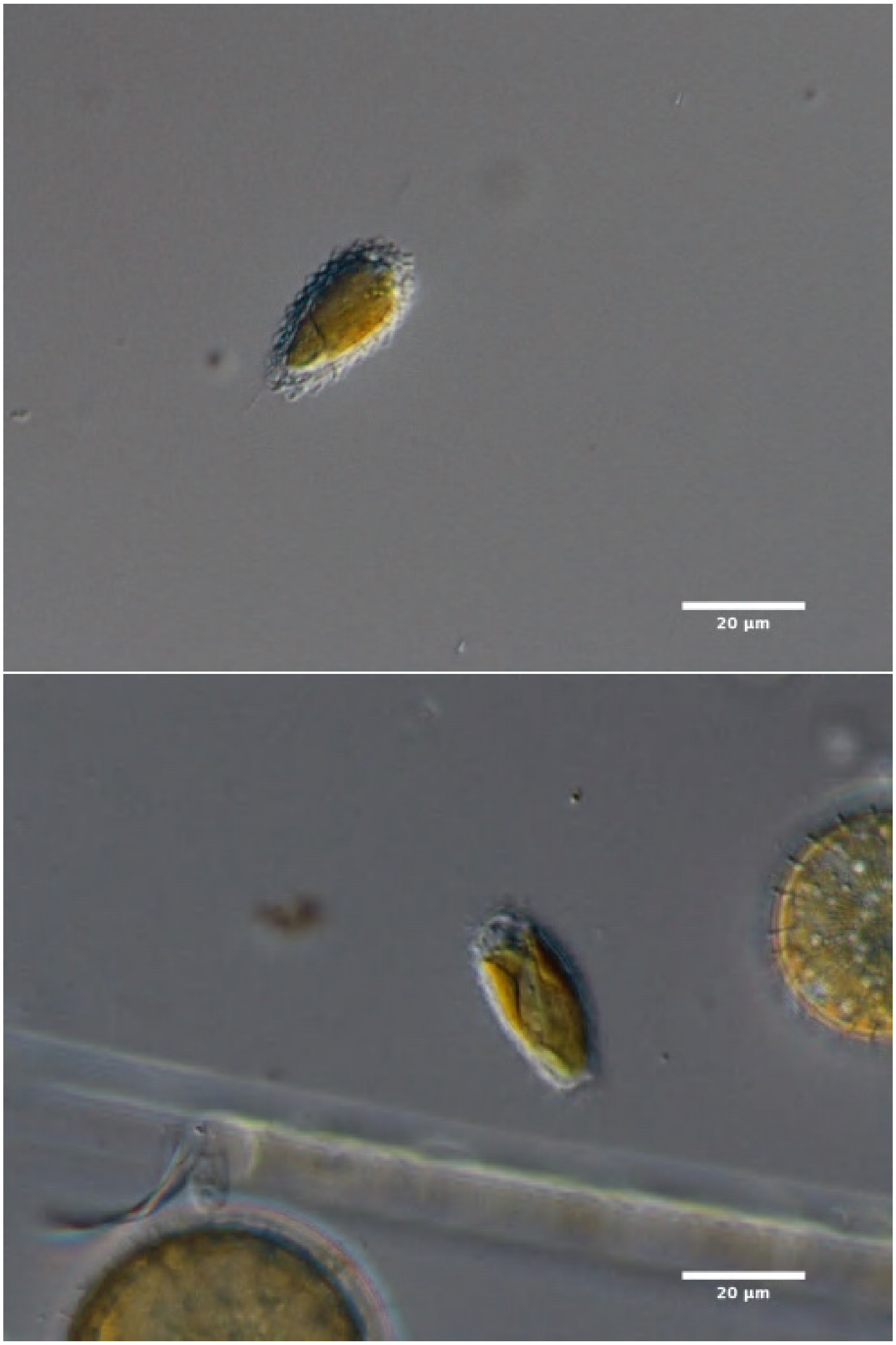


Figure 59: Other/Ochrophyta - upper image: *Mallomonas* (600x DIC), October 15, 2019; lower image: *Mallomonas* (600x DIC), October 29, 2019.

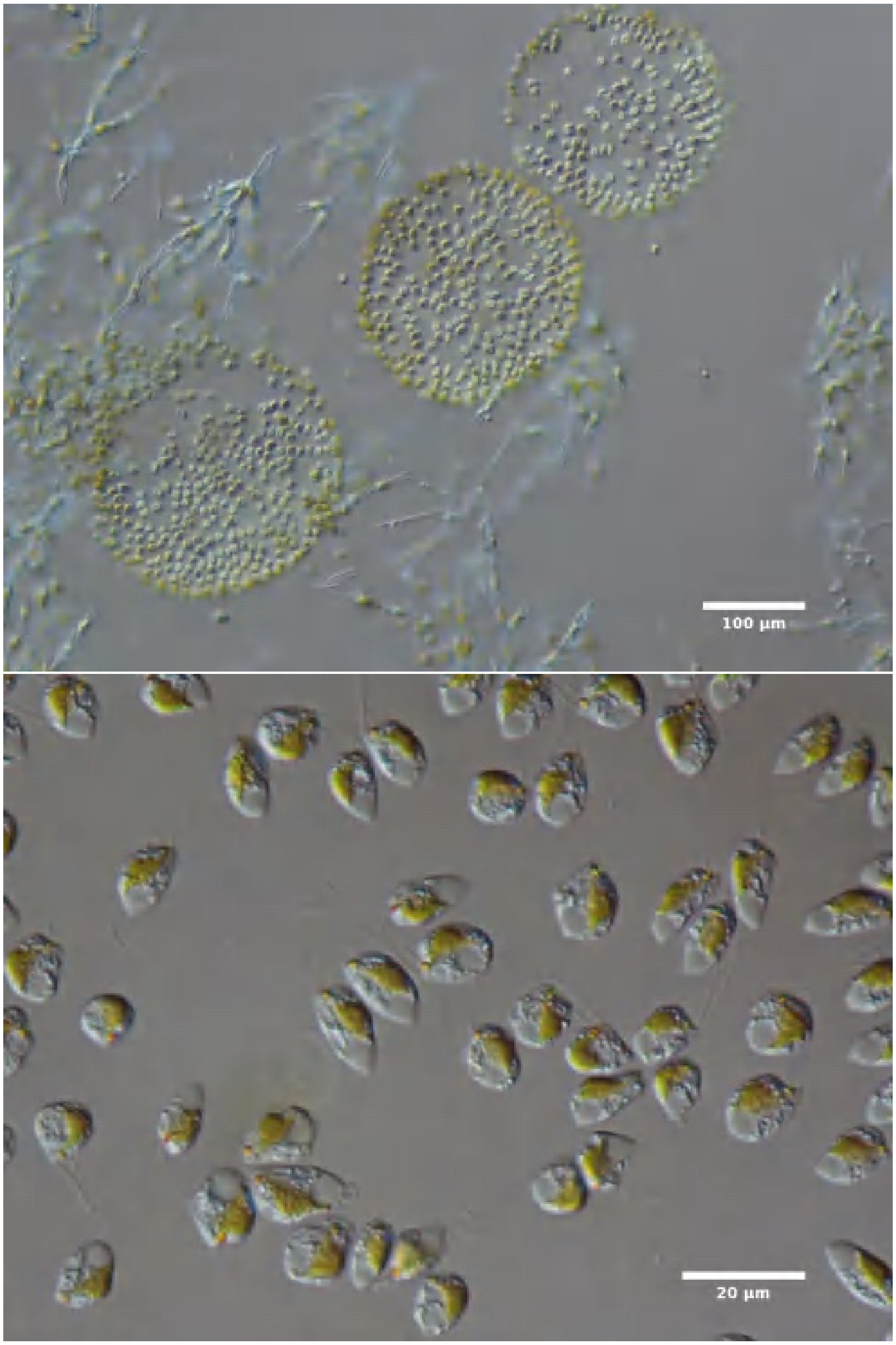


Figure 60: Other/Ochrophyta - upper image: *Uroglenopsis americana* (100x DIC), Toad Lake, Whatcom County, April 10, 2015; lower image: *Uroglenopsis americana* (600x DIC), Toad Lake, Whatcom County, April 10, 2015.

APPENDIX C

Sediment Quality Monitoring Data

IEH ANALYTICAL LABORATORIES

LABORATORY & CONSULTING SERVICES

3927 AURORA AVENUE NORTH, SEATTLE, WA 98103

PHONE: (206) 632-2715 FAX: (206) 632-2417

CASE FILE NUMBER:	**MIS056-32A**	**PAGE**	**1**
REPORT DATE:	**10/28/19**		
DATE SAMPLED:	**08/24/19**	**DATE RECEIVED:**	**08/26/19**

FINAL REPORT, LABORATORY ANALYSIS OF SELECTED PARAMETERS ON SEDIMENT SAMPLES FROM WHIDBEY ISLAND CONSERVATION DISTRICT

CASE NARRATIVE

Thirteen sediment samples were received by the laboratory in good condition and analyzed according to the chain of custody. Phosphorus fractions were determined according to the method of Rydin and Welch. Successive extractions with NH4Cl, Bicarbonate/Dithionate, NaOH, and HCL were performed and analyzed for phosphorus. One part of Organic P was determined by digesting the residue after the inorganic fractions were extracted. Organic P includes the P after the inorganic fractions plus Biogenic P. Total P is the sum of all fractions minus Biogenic P, which is part of the Organic P fraction. No difficulties were encountered in the preparation or analysis of these samples. Sample data follows, while QA/QC data is contained on subsequent pages.

SAMPLE DATA - SEDIMENTS (DRY WT. BASIS)

SAMPLE ID	% SOLIDS	% WATER	TOTAL-P (mg/kg)	LOOSELY BOUND P (NH4CL) (mg/kg)	FE BOUND P (DITHIONATE) (mg/kg)	AL BOUND P (NAOH) (mg/kg)	BIOGENIC P (mg/kg)	CA BOUND P (HCL) (mg/kg)	ORGANIC P (mg/kg)
CORE B 10-12CM	5.12%	94.9%	1002	<2.00	156	185	442	48.5	613
CORE C 0-2CM	3.39%	96.6%	1552	<2.00	378	222	654	55.5	896
CORE C 2-4CM	3.84%	96.2%	1520	<2.00	308	223	686	61.3	928
CORE C 4-6CM	4.44%	95.6%	1381	<2.00	263	227	602	61.1	830
CORE C 6-8CM	4.59%	95.4%	1409	<2.00	275	213	645	61.8	859
CORE C 8-10CM	4.77%	95.2%	1482	<2.00	243	219	686	75.3	945
CORE C 10-12CM	4.69%	95.3%	1312	<2.00	222	211	567	72.8	807
CORE C 12-14CM	4.91%	95.1%	1223	<2.00	192	195	519	75.7	760
CORE C 14-16CM	4.92%	95.1%	1179	<2.00	190	207	507	80.5	703
CORE C 16-18CM	5.04%	95.0%	1158	<2.00	168	197	507	68.6	724
CORE C 18-20CM	5.29%	94.7%	1186	<2.00	160	207	471	73.0	746
CORE C 28-30CM	5.57%	94.4%	961	7.33	135	200	391	57.5	561
CORE C 38-40CM	5.64%	94.4%	897	38.6	117	211	319	62.5	468

IEH ANALYTICAL LABORATORIES

LABORATORY & CONSULTING SERVICES
3927 AURORA AVENUE NORTH, SEATTLE, WA 98103
PHONE: (206) 632-2715 FAX: (206) 632-2417

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SEDIMENT SAMPLES FROM WHIDBEY ISLAND CONSERVATION DISTRICT			

QA/QC DATA- SEDIMENTS

QC PARAMETER	% SOLIDS	TOTAL-P	LOOSELY BOUND P (NH4CL)	FE BOUND P (DITHIONATE)	AL BOUND P (NAOH)	BIOGENIC P	CA BOUND P (HCL)	ORGANIC P
		(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)	(mg/kg)
METHOD	SM18 2540B	CALCULATED	SM18 4500PF	SM18 4500PF	SM18 4500PF	EPA 365.1	SM18 4500PF	EPA 365.1
DATE PREPARED	09/11/19	09/13/19	09/11/19	09/11/19	09/11/19	09/13/19	09/11/19	09/13/19
DATE ANALYZED	1.00%	5.00	2.00	2.00	2.00	2.00	2.00	2.00
DETECTION LIMIT								
DUPLICATE								
	CORE C 38-40CM	CORE B 10-12CM	CORE B 10-12CM	CORE B 10-12CM	CORE B 10-12CM	CORE B 10-12CM	CORE B 10-12CM	CORE B 10-12CM
SAMPLE ID	5.64%	1002	<2.00	156	185	442	48.5	613
ORIGINAL	5.78%	1018	<2.00	135	193	435	54.3	636
DUPLICATE	2.33%	1.57%	NC	14.56%	4.32%	1.76%	11.26%	3.69%
RPD								
SPIKE SAMPLE								
SAMPLE ID								
ORIGINAL								
SPIKED SAMPLE								
SPIKE ADDED	NA	NA	NA	NA	NA	NA	NA	NA
% RECOVERY								
QC CHECK								
(mg/l)								
FOUND			0.040	0.040	0.040	0.091	0.040	0.091
TRUE			0.039	0.039	0.039	0.094	0.039	0.094
% RECOVERY	NA	NA	102.56%	102.56%	102.56%	96.81%	102.56%	96.81%
BLANK	NA	NA	<2.00	<2.00	<2.00	<2.00	<2.00	<2.00

RPD = RELATIVE PERCENT DIFFERENCE.
NA = NOT APPLICABLE OR NOT AVAILABLE.
NC = NOT CALCULABLE DUE TO ONE OR MORE VALUES BEING BELOW THE DETECTION LIMIT.
OR = RECOVERY NOT CALCULABLE DUE TO SPIKE SAMPLE OUT OF RANGE OR SPIKE TO LOW RELATIVE TO SAMPLE CONCENTRATION.

SUBMITTED BY:

Damien Gadomski

Damien Gadomski
Project Manager

IEH ANALYTICAL LABORATORIES

LABORATORY & CONSULTING SERVICES

3927 AURORA AVENUE NORTH, SEATTLE, WA 98103

PHONE: (206) 632-2715 FAX: (206) 632-2417

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FINAL REPORT, LABORATORY ANALYSIS OF SELECTED PARAMETERS ON SEDIMENT SAMPLES FROM WHIDBEY ISLAND CONSERVATION DISTRICT

CASE NARRATIVE

Thirteen sediment samples were received by the laboratory and analyzed according to the chain of custody. No difficulties were encountered in the preparation or analysis of these samples. Sample data follows, while QA/QC data is contained on subsequent pages.

SAMPLE DATA - SEDIMENTS (DRY WT. BASIS)

SAMPLE ID	% SOLIDS	% WATER	IRON (mg/kg)
CORE B 10-12CM	5.12%	94.9%	8009
CORE C 0-2CM	3.39%	96.6%	9843
CORE C 2-4CM	3.84%	96.2%	9989
CORE C 4-6CM	4.44%	95.6%	7557
CORE C 6-8CM	4.59%	95.4%	7203
CORE C 8-10CM	4.77%	95.2%	7270
CORE C 10-12CM	4.69%	95.3%	7345
CORE C 12-14CM	4.91%	95.1%	6237
CORE C 14-16CM	4.92%	95.1%	6809
CORE C 16-18CM	5.04%	95.0%	6420
CORE C 18-20CM	5.29%	94.7%	6875
CORE C 28-30CM	5.57%	94.4%	8367
CORE C 38-40CM	5.64%	94.4%	9525

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FINAL REPORT, LABORATORY ANALYSIS OF SELECTED PARAMETERS ON SEDIMENT SAMPLES FROM WHIDBEY ISLAND CONSERVATION DISTRICT

QA/QC DATA- SEDIMENTS

QC PARAMETER	% SOLIDS	IRON (mg/kg)
METHOD	SM18 2540B	EPA 6010
DATE ANALYZED	09/11/19	09/05/19
DETECTION LIMIT	1.00%	2.00
DUPLICATE		
SAMPLE ID	CORE C 38-40CM	CORE C 38-40CM
ORIGINAL	5.64%	9525
DUPLICATE	5.78%	8574
RPD	2.33%	10.51%
SPIKE SAMPLE		
SAMPLE ID		
ORIGINAL		
SPIKED SAMPLE		
SPIKE ADDED		
% RECOVERY	NA	NA
QC CHECK **(mg/L)**		
FOUND		0.513
TRUE		0.500
% RECOVERY	NA	102.60%
BLANK	NA	<2.00

RPD = RELATIVE PERCENT DIFFERENCE.
NA = NOT APPLICABLE OR NOT AVAILABLE.
NC = NOT CALCULABLE DUE TO ONE OR MORE VALUES BEING BELOW THE DETECTION LIMIT.
OR = RECOVERY NOT CALCULABLE DUE TO SPIKE SAMPLE OUT OF RANGE OR SPIKE TO LOW RELATIVE TO SAMPLE CONCENTRATION.

SUBMITTED BY:

Damien Gadomski

Damien Gadomski
Project Manager

APPENDIX D

Aluminum Application and Costing Detail

ALUMINUM APPLICATION AND COSTING DETAIL

The amount (dose) of aluminum needed to inactivate sediment phosphorus is determined from the amount of mobile phosphorus in lake sediments (the source of internal loading) and the ratio of aluminum added to aluminum-bound phosphorus formed (Rydin and Welch 1998, 1999; Pilgrim et al. 2007; Huser and Pilgrim 2014). The calculated aluminum dose is then increased to account for the amount that will be used up as it moves through the water column and binds with phosphorus in the lake water.

The ratio of aluminum added to aluminum phosphorus formed has varied among lakes and researchers over time. A ratio of 20 has been successfully used in Washington lakes where the targeted amount of sediment phosphorus was based only on the mobile phosphorus concentration. A lower ratio of 8.8 parts aluminum to aluminum phosphorus formed has recently been recommended by European limnologists when active biogenic phosphorus is included in the targeted amount of sediment phosphorus to be inactivated. For Lone Lake, a slightly higher ratio of 10 parts aluminum to targeted sediment phosphorus was used to calculate the amount of aluminum added for the alum treatment and provide an additional safety factor for effectiveness longevity.

Another important consideration for calculating an alum dose is the appropriate inactivation depth. For Lone Lake, 10 cm is appropriate for long-term inactivation depth because active (mobile plus biogenic) phosphorus concentrations substantially decreased below 10 cm in the sediment core. In addition, evaluation of alum treated lakes in Washington showed that the aluminum bound phosphorus formed by the treatment was limited to the upper 10 cm of sediment in most lakes (Rydin and Welch 2000).

These recommended aluminum dose and alum treatment cost for Lone Lake is presented in Table A-1, including the assumptions discussed above. The recommended treatment scenario includes application of a full aluminum dose to inactivate sediment phosphorus and strip water column phosphorus in March or April 2021. The aluminum dose to inactivate lake sediments was calculated to be 14,607 kg Al. An additional 532 kg Al would be required to inactivate the phosphorus present in the water column. The sum of these two doses is 15,139 kg Al. Using 2 to 1 ratio for alum and sodium aluminate, a total of 35,622 gallons of alum and 17,811 gallons of sodium aluminate would be needed.

Table D-1. Lone Lake Alum Treatment Dose and Cost Estimate.		
Item	**Value**	**Basis**
Sediment Phosphorus		
Mobile P (mg/kg dw)	293	Loosly-P + Iron-P average of 0-10 cm depth in core C
Biogenic P (mg/kg dw)	655	Biogenic-P average of 0-10 cm depth in core C
Active P (mg/kg dw)	948	Sum of Mobile and Biogenic-P
Dry bulk density (g dw/cm^3)	0.0421	Average of 0-10 cm depth in cores A, B, and C
Active P (mg/cm^3)	0.040	Active P x dry bulk density x 10^{-3} kg dw/g dw
Inactivation Depth (cm)	10	Based on active P profile and observed in alum-treated WA lakes
Active P areal amount (g P/m^2)	4.0	Active P x depth x 10^{-3} g P/mg P x 10^4 cm^2/m^2
Al-P binding ratio	10	Excess Al to inactivate migrated and mineralized sediment P
Treatment area (m^2)	366,000	Lake area below 5 feet deep (91 acres)
Treatment dose (kg Al)	14,607	Active P x Al-P ratio x treatment area x 10^{-3} kg/g
Water Phosphorus		
Lake TP (ug/L = mg/m^3)	40	Epilimnion sample from March 2019
Lake Volume (m^3)	1,330,000	Lake volume excluding shallow area < 5 feet deep
Al-P binding ratio	10	Excess Al needed for TP in water
Water dose (kg Al)	532	Lake TP x Lake volume x Al-P ratio x 10^{-6} kg/mg
Aluminum Dose		
Total Al dose (kg Al)	15,139	Sum of sediment and water Al dose
Al volumetric dose (mg Al/L)	11.4	Total dose / lake volume in kg/m^3 x10^3L/m^3
Al areal dose (g Al/m^2)	41.4	Total dose / treatment area in kg/m^2 x 10^3g/kg
Material Amounts		
Al sediment+water dose (kg Al)	15,139	Sum of sediment and water Al dose
Al in Alum (kg Al)	7,837	Al dose x 0.44 kg / (0.44+0.41 kg) for 2:1 liquid ratio
Al in Sodium aluminate (kg Al)	7,303	Al dose x 0.41 kg / (0.44+0.41 kg) for 2:1 liquid ratio
Alum volume (gal)	35,622	0.22 kg Al/gal
Sodium aluminate volume (gal)	17,811	0.41 kg Al/gal (revised from 0.56 based on Heart Lk)
Cost Estimate		
Materials	$92,617	$1.10/gal alum, $3.00/gal Na aluminate (Heart 2018 +10%)
Material Application	$64,120	$1.20/gal (Heart Lake 2018 + 20%)
Submittals/Mob/Demob	$37,047	40% of materials cost
Tax	$8,336	9% of materials
Subtotal Contractor Cost	**$202,119**	Sum of materials, application, submittals/mob/demob, and tax
Consultant Planning	$18,000	Alum Treatment Plan (Herrera)
Consultant Oversight/Monitoring/Report	$30,000	4 days treatment monitoring, 1 season post-treatment monitoring, report
Contingency	$40,424	20% of contractor cost
Total Project Cost	**$290,543**	Sum of contractor, planning/design/permitting, and contingency

This aluminum dose would be applied to the entire lake area excluding shallow areas less than 5 feet deep to avoid nearshore obstructions and sediment disturbance. It is estimated to take at least two days to apply this quantity of material for a total cost of approximately $290,000. Based on 10 years of effectiveness, this cost equates to $29,000 per year to improve water quality and prevent toxic cyanobacteria blooms in Lone Lake.

Material (chemical) costs are based on $1.10/gallon for alum and $3.00/gallon for sodium aluminate. Contractor application costs are estimated at $1.20/gallon and mobilization/demobilization costs are estimated at 40 percent of the materials cost. These unit costs are 10 to 20 percent more than costs of treating Heart Lake in Skagit County in 2018 (Herrera 2019). A tax rate of 9 percent is used for materials only to derive the total contactor costs.

In addition to contractor costs, additional funds are required for a consultant to assist with planning and design, preparation of technical specifications, contractor procurement, permit application, monitoring during treatment, and reporting of the treatment observations. Consultant costs were assumed to include $18,000 for preparing an alum treatment plan and $30,000 for oversight, monitoring, and reporting of a 2-day treatment. Finally, a 20 percent contingency amount on the contractor cost is included to account for potential unexpected expenses.

The total aluminum dose is equivalent to 11.4 mg Al/L on a volumetric basis and 41.4 g Al/m2 on an aerial basis. This dose is within the range of alum treatments in Washington State (Table D-2). The volumetric dose is similar to doses for three lakes located in relatively undeveloped watersheds in nearby Skagit County (Heart, Campbell, and Erie), and lower than initial treatments of Lake Ketchum which had been enriched with livestock waste in Snohomish County.

Table D-2. Comparison of Alum Treatment Doses in Washington.

Lake (County)	Treatment Date	Volumetric Dose (mg Al/L)	Aerial Dose (g Al/m^2)	Longevity (years)[a]	Reference
Heart Lake (Skagit)	2018	10.9	26.9	unknown	Herrera 2019
Lake Campbell (Skagit)	October 1985	10.9	26	> 8	Cooke et al. 2005
Lake Erie (Skagit)	September 1985	10.9	20	> 8	Cooke et al. 2005
Black Lake (Thurston)	April 2016	1.9	13	unknown	Herrera 2017a
Lake Ketchum (Snohomish)	May 2014 March 2015	19 19	66.5 66.5	NA unknown	G. Williams (pers. comm.)
Long Lake (Thurston)	September 1983 2008 (planned)	7.7 15.2	27.7 54.9	5 unknown	Cooke et al. 2005 Tetra Tech 2006
Long Lake (Kitsap)	September 1980 September 1991 August 2006 April 2007	5.5 5.5 2.5 17.5	10.7 10.7 4.6 36.2	> 11 > 11 NA > 5	Rydin et al. 2000 Rydin et al. 2000 Tetra Tech 2010 Tetra Tech 2010

Pattison Lake (Thurston)	September 1983	7.7	30.8	7	Cooke et al. 2005
Green Lake (King)	October 1991 April 2004 April 2016	8.6 24 8.2	34 94 32	3 > 10 unknown	Herrera 2003 Herrera 2004 Herrera 2016
Phantom Lake (King)	September 1990	4.2	9.5	unknown	Rydin et al. 2000
Lake Ballinger (King)	June 1990	5.0	6.5	unknown	Rydin et al. 2000
Wapato Lake (Pierce)	July 1984 July 2008 April 2017	7.8 67.7 56.3	11.7 108 90	<1 5 unknown	Cooke et al. 2005 Herrera 2017b Herrera 2018
Medical Lake (Spokane)	Aug.-Sept. 1977	12.2	83.5	unknown	Rydin et al. 2000

[a] Cooke et al. 2005 and Herrera 2015 (for Green Lake).

mg Al/L = milligrams of aluminum per liter

g Al/m^2 = grams of aluminum per square meter

NA = not applicable