Moses Lake Water Quality:
Causes and Benefits of Columbia River Water

Prepared for Moses Lake Irrigation and Rehabilitation District

By

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Introduction

Lake Quality Improvement

Moses Lake was hypereutrophic during the 1960s to the mid-1970s. Total phosphorus (TP) concentration averaged 152 µg/L and chlorophyll (chl) 58 µg/L in Lower Parker Horn and the lower lake, between I-90 and the outlet, during spring-summer 1969-1970 (Welch et al., 1992). Those concentrations were well above the eutrophic-hypereutrophic boundaries for TP and chl of 100 and 30 µg/L, respectively, so lake water quality was highly degraded Nürnberg, 1996). Harmful algal blooms (HABs) of blue green algae, *Aphanizomenon* and *Microcystis*, occurred throughout the summer, forming unsightly surface scums and were probably toxic. Although growth of algae requires nitrogen in larger quantities than phosphorus, the latter is the key nutrient that drives eutrophication of lakes world wide and Moses Lake is not an exception (Welch, 2009).

The potential benefit of adding Columbia River water (CRW) to the lake was recognized in the early 1960s, because CRW was low in phosphorus and nitrogen and the infrastructure was in place to route CRW from the East Low Canal through Rocky Coulee Wasteway (RCW) and Crab Creek (CC) into Parker Horn. Transporting large quantities of CRW with a low TP concentration of 20 µg/L would dilute the high-TP lake water and proportionately reduce the concentration of algae (chl). There was precedent for using RCW as a feed route for irrigation water through the lake, but the quantities had been at various low rates since 1956, averaging only 5 X 10^6 m^3 (4,050 AF) in 1969-1970 when the lake was hypereutrophic. The amount of water needed to improve lake water quality was substantial; in-lake experiments showed that algal biomass could be reduced in proportion to CRW added (Welch et al., 1972). So, to reduce lake TP to the mesotrophic boundary of 30 µg/L from around 150 µg/L would require replacing most of the lake water with CRW, because phosphorus would still be entering the lake from ground water and recycled from bottom sediments.

Funds became available as a Clean Lakes Project from the US Environmental Protection Agency (USEPA), through the Washington Dept. of Ecology (WADOE), in cooperation with the US Bureau of Reclamation (USBR), to increase the transport of CRW through the lake with the goal of reducing spring-summer average TP to 50 µg/L that would produce a transparency of 1.5 meters. As a result, TP in Lower Parker Horn and the lower lake was reduced by one half during 1977-1986 when CRW input averaged 20-fold over the pre-project rate, and below 50 µg/L during 1986-1988 when CRW was over 30-fold above the pre-project rate (Welch et al., 1989; Welch et al., 1992). Total P has declined further since 2000 as CRW input increased 60-fold on average over the pre-project rate.

As part of the Clean Lakes Project, a pump was installed in Upper Parker Horn in 1982 that transported CRW into Pelican Horn. Also, domestic wastewater was diverted from Middle Pelican Horn in 1984, which initially reduced TP from 600 to 80 µg/L (Welch et al., 1989).

There was also a shift in the lake's fish populations associated with improvement in water quality from 1977-2001. The catch fraction of crappie and bluegills decreased markedly, while largemouth bass decreased some, while smallmouth bass, walleye and brown bullhead increased several-fold (Welch, 2009).

While increasing CRW input successfully diluted lake TP and greatly improved water quality in the 1970s and 1980s, further improvements were considered prudent, so a TMDL (total maximum daily load) study was conducted by WADOE in 2001 (Carroll, 2006). The proposed TMDL goal was a lake TP of 35 µg/L to be achieved by reductions in external sources. Average lake TP during 1986-1988 had approached that goal at 41 µg/L in Lower Parker Horn and south lake (Welch et al., 1989).

This report updates the water quality and trophic status of Moses Lake with data from WADOE in 2001, USBR from the 1990s through 2017 and MLIRD in 2017. USBR data are from the lake's inlets and outlet, while WADOE and MLIRD data are from multiple sites in the lake, corresponding to those during 1969-1988, as well as inlets, sampled by the University of Washington (Welch et al., 1989).

Methods

Water Sample Collection and Analysis

Water samples were collected in 2017 with a Van Dorn bottle by MLIRD personnel at a depth of 0.5 m at nine lake sites during May-September (Figure 1). Sampling at TS12 was not begun until July. Inflows at two sites on CC (TS2, TS3), east low canal (TS1), seepage to upper Pelican Horn (TS13) and a ground water spring (TS9) were grabbed-sampled. Upper Pelican Horn was sampled near its outlet to middle Pelican Horn. Sample frequency was twice in May, June and September and once in July, August and the last event was October 2. Collections were in clean, pre-acid rinsed bottles.

Samples were shipped on ice to IEH Analytical Laboratories, Seattle, WA, for analysis of total phosphorus (TP), nitrate-nitrogen (NO_3-N) and sodium (Na). Chlorophyll was determined in the same lake samples on residue following filtration in the laboratory. Analytical procedures were according to standard methods (Eaton et al., 2005). Specific conductance (SC) was determined *in situ* at all sites except TS9 coincident with water sampling.

Water samples for algae identification and enumeration were also collected from the Van Dorn bottle water, coincident with the sample for other constituents. Samples for algae were collected in July, August and October 2 only. Algal abundance was determined as cells/ml and also expressed as biovolume in mm^3/L based on measured cell volumes of individual species observed. Methods are described in detail by Matthews et al. (2018).

Phosphorus Loading

External loading during May-September, 2017, was estimated as average surface inflow volume from respective sources multiplied by their average TP concentrations during the five months, rather than on a monthly basis, as in previous years, due to lack of monthly TP data on inflow streams. Average flow and TP concentrations in CC and Rocky Ford Creek (RFC) were 1.5 and 1.71 m^3/sec (53 and 60 cfs) and 42 and 149 µg/L TP, respectively. Total P concentrations in CC were from the five observations by USBR, as well as by MLIRD. Average TP in CRW was 7 µg/L, observed by MLIRD, and 87 µg/L in RCW base flow from Carroll (2006).

Internal loading for 2017 was estimated as the difference between external load and outflow load, which was estimated as outflow volume X volume-weighted (v-w) lake TP (50.8 µg/L). Lacking observed data, outflow volume was estimated as the average ratio (1.5) of outflow/surface inflow during1984-1988. That inflow/volume ratio varied by 21% during 1984-1988. Also, the average v-w lake TP (excluding Upper and Middle Pelican Horn) was determined from observations at 0.5 m, so the normally higher concentrations at depth were not included, resulting in an underestimate of internal loading. Thus, there is considerable uncertainty in the estimate of internal loading.

Results and Discussion

Forty years of dilution

The nutrient characteristics of the two major surface inflows to the lake, Crab Creek and Rocky Ford Creek, are shown in Table 1. Both TP and SRP concentrations at the Highway 17 gauging site on Crab Creek have averaged much higher over the past 14 years than in the 1980s. Nitrate was also reportedly higher. The nearly six-fold increase in SRP is important because that is the form of P readily available for algae. Nevertheless, that effect was minimized when large volumes of CRW, with a SRP of only 4 µg/L (Table 1), were transported through Parker Horn during the spring-summer periods.

The average inflow rate of CRW was 325 x 10^6 m^3 (263 x 10^3 AF) over an average of 208 days during 2002-2016 (18 m^3/sec, 633 cfs)). That flow would replace half the lake's volume contained in the lake sections, Cascade, Parker Horn (TS4/5) and south lake (TS6/7), at about 2.0 %/day. Normal CC and RFC inflow at 3.2 m^3/sec (112 cfs) would replace that half-lake volume at less than two tenths the rate; 0.36 %/day. Thus, once CRW were added during the spring and early summer, the resulting mixture of CRW and lake water was replaced very slowly given the relatively slow inflow rate of the two creeks – on the order of 600 days to replace 90% of the half-lake volume. Of course, ground water with relatively high SRP also enters the lake, although that flow is less than the surface inflows.

The RFC inflow to the upper main arm (TS12) contained very high TP concentrations with a high fraction that was available SRP (78%). While recent P concentrations were similar to those in the 1980s, nitrate has doubled (Table 1). The reason for higher nitrate is unclear, but possible due to higher rates of fertilizer application to farm land. However, the higher nitrate-N concentrations have probably not caused a greater algal biomass, because the ratio of NO_3-N:SRP was well above the Redfield ratio of 7:1, which defines the usual demand by algae for those two macro nutrients. Thus, P was still the key nutrient in shortest supply, which determines algal growth and biomass formation. To illustrate the importance of phosphorus, wastewater phosphorus was reduced in Spokane wastewater to the Spokane River and Lake Spokane in 1977 and lake quality quickly improved over a 7-year period from hypereutrophic before P reduction to borderline meso-eutrophic and eventually to borderline oligo-mesotrophic as inflow TP continued to decline, while inflow nitrogen increased (Welch et al., 2015). Thus, increased nitrogen had no effect on lake algae and water quality because the lowered TP was the nutrient exerting control on algal production and lake quality.

Specific conductance (SC) is a good tracer of CRW, which is low in SC in contrast to CC (Table 2). Moses Lake is a hard-water lake, meaning its alkalinity and hardness are relatively high, and consequently its dissolved ion content that is measured by specific conductance (SC) is respectively high. CRW can be traced in the lake because it has a SC around one-fourth the levels in CC, RFC and the lake (Table 2). Thus, lake SC was highly inversely correlated with CRW input over the past 40 years, because the high rate of CRW inflow replaced lake water (Figure 2). Percent lake water (or CRW) was calculated by (Cooke et al., 2005):

$$100\,[(LW - ELCW) / (CCW - ELCW)] = \%\ LW$$

In 2017, SC at TS5 and TS6 averaged 288 µS/cm during May-September. That represents about 60% CRW and 40% original lake water, and largely accounts for the low average TP concentration of 25 µg/L observed in that area of the lake. Hence, TP was also inversely related to CRW inflow over the past 40 years (Figure 3).

Despite the relatively low input of CRW (93 X 10^6 m^3) in 2017, SC in lower Parker Horn (TS5) and south lake (TS6) increased from 267 to 299 µS/cm, or only 64% during spring-summer, ending up at 55% CRW in that lake area. In contrast, SC was lower at 254 µS/cm or 68% CRW through August in 2016, which had a two and half-fold greater input of CRW (234 X 10^6 m^3).

Trophic state indicators in 2017 vs 1970s-1980s

Total P markedly decreased in Parker Horn and south lake during the 1970s and 1980s due to the input of low-P CRW and decreased even further as CRW inputs more than doubled (~ 100 to 250 x 10^6 m^3) after 2000 (Table 3; Fig. 4). The average of 20 µg/L TP during 2002-2016 is based on four samples collected by USBR from March through September. Previous average TPs included both lower Parker Horn (TS5) and south lake (TS6), which together represents half the lake volume that was affected most by CRW. Average spring-summer (mid-April to mid-October) TPs from USBR data (sample number = 4) during 2002-2017 and the MLIRD (May-September, n=9) in 2017 were both 25 µg/L, which tended to validate using the 15 years of USBR data as representative of lower Parker Horn and south lake water quality.

Algal biomass, indicated by chlorophyll (chl) content, averaged 7 µg/L at TS5/6 during spring summer 2017. At the ratio of chl:TP (0.35) during 2017, average chl during 2002-2016 likely averaged about 6 µg/L (Table 3). These levels are much lower than in the 1970s-1980s, and correspond to lower TP. The ratio of chl:TP of 0.35 agrees with the average for world lakes, indicating that chl is dependent on TP in Moses Lake.

Transparency is usually dependent on the concentration of algae, because that is the major source of particulate matter in water that absorbs and scatters light, which determines the depth of visibility measured by a Secchi disc (SD). Average spring-summer transparency in lower Parker Horn and south lake in the past ranged from 0.7 m before to around 1.5 m after dilution by CRW (Table 3). Observed transparency was consistently 40% greater than predicted from a relationship between chl and transparency: $SD = 7.7/chl^{0.68}$ (Carlson, 1977).

Average observed transparency at TS5/6 in 2017 was only 1.4 m, which was 2/3 of that predicted from the low chl of 7 µg/L using the above equation (Table 3). The lower than expected transparency may indicate the presence of more non-algal particulate matter than in the past.

The trophic state in the half volume of Moses Lake most affected by CRW is now mid to lower mesotrophic, as indicated by TP and chl, compared to hypereutrophic before dilution and eutrophic after the start of dilution in the 1970s and 1980s (Table 3). However, transparency, at less than 2 m, still indicates eutrophy (Nürnberg, 1996).

Total P and chl were much higher in other areas in the lake in 2017, indicating a hypereutrophic state. The upper main arm (TS12) had average TP and chl concentrations of 86 and 26 µg/L, respectively, which were slightly higher than average levels at TS11 in the 1970s-1980s of 70 and 22 µg/L, respectively. There are no prior data from TS12. Also, average TP and chl in middle Pelican Horn (TS8) were quite high in 2017 at 42 and 13 µg/L, respectively, although lower than in 1985-1988 (after sewage diversion) with average TP and chl at 81 and 14 ug/L, respectively. The chl:TP ratios in the upper main arm and middle Pelican Horn currently were around 0.30, while that ratio in middle Pelican Horn in the 1980s was only 0.17, about half that expected. That low ratio in middle Pelican indicated severe nitrogen limitation, which is currently the case as indicated by nitrate concentrations that remained at the detection limit during summer 2017. Depletion of available nitrogen occurred because the most abundant algal species were not nitrogen fixers and the phosphorus concentration was more than adequate.

Algae

Cyanobacteria were composed primarily by *Microcystis aeriganosa* (MA) in lower Parker Horn (TS5) and south lake (TS6) during 2017, averaging 51% of total plankton algal biovolume in 5 of the 6 samples collected during mid-July through September. Percent cyanobacteria was only 5% in the sixth sample due to the large biovolume of a diatom. That average summer percentage may have been lower had the sites been sampled from May-September, because cyanobacteria have usually reached maximums in August and September as water temperature and water column stability increase. *Aphanizomenon flos aquae* usually preceded and was more abundant than *Microcystis* in the past and likely was more abundant in the spring and early summer that was not sampled in 2017. These are the usual two important nuisance species, scum forming species that form harmful algal blooms (HABs) in eutrophic lakes (Fig. 5).

Total cyanobacteria biovolume ranged from about 11 to 25 mm^3/L at TS5/6 and averaged 14 mm^3/L during mid-July through September. Total biovolume of all algae averaged 67 mm^3/L, which was due mostly to a large biomass of a diatom (*Suriella*) at TS5 in July that minimized the percent cyanobacteria to 5%. Biovolume of cyanobacteria was lower at those sites (TS5/6) during 1986-1988, averaging about 7 mm^3/L, although their % of total was higher averaging 79% during the whole summer, May-September (Welch et al., 1992). While biovolume of cyanobacteria at TS5/6 averaged lower in 1986-1988 than in 2017 (7 vs 14 mm^3/L), the reverse occurred with average chl, which was much higher in 1986-1988 than 2017 – 17 vs 7 µg/L (Table 3). Again, that may be due to the shorter sampling period for algae in 2017. Cyanobacteria were more abundant in the main arm in 2017 (TS11/12) than at other sites at biovolumes ranging from 14-144 mm^3/L and averaged 74% of the total at those two sites (Table 4).

Cyanobacteria have never been well represented in Pelican Horn (TS8) where they averaged only about 6 mm^3/L and 8% of total biovolume in 2017, and about 1 mm^3/L and 9% of the total during May-September in 1986-1988. Green algae, such as *Oosystis* and *Scenedesmus*, have usually dominated the plankton algae in Pelican Horn, both before and after the diversion of wastewater in 1984 (Figure 6). *Oosystis* was represented in 2017, but most of the biovolume in Pelican Horn was composed of diatoms, such as *Aulacoseira*. These small greens absorb and scatter more light through the water column than do cyanobacteria, which tend to clump and accumulate in patches on the surface, allowing more transmittance of light. Those small green algae result in the more apparent turbid conditions in Pelican Horn than in the rest of the lake, evident in Figure 1, although the green shade throughout the lake is unnatural. The main portion of Parker Horn and south lake (TS5, 6 and 7) actually appears bluer to the eye. Lakes having a green shade are usually glacially fed.

Microcystis aeriganosa produces microcystin, a liver toxin that has poisoned livestock, pets and is a threat to the health of users of lakes for recreation and drinking water world-wide (WHO, 2003). Microcystin (MC) was determined in 9 western Washington lakes with moderate to high (>6 µg/L) MC concentrations that were related to *Microcystis aeriganosa* concentrations greater than 1,300 cells/ml and TN:TP ratios $\leq$ 25.7:1 (Jacoby et al., 2015). The average abundance of *Microcystis aeriganosa* at T5/6 during 2017 was 60,320 cells/ml and double that at TS11/12 (120,738 cells/ml). MC was not determined in 2017, but the high cell concentrations of *Microcystis aeriganosa* indicate that MC was probably present in substantial amounts at these sites in 2017. Ratios of TN:TP averaged 15 during 1995-2016 and 21 in 2017 at TS5/6.

Effect of Pumping CRW to Pelican Horn

Pumping of Parker Horn water to Pelican Horn in the 1980s, when CRW was present in upper Parker Horn (TS4), had little effect on water quality. There was essentially no difference in average TP in either upper Pelican (TS10) or middle Pelican (TS8) when CRW was or was not present in upper Parker and the pump ran continuously during spring and summer (Table 4). That is, exchanging middle Pelican water at 11%/day by pumping water with 29 µg/L TP from upper Parker did not lower TP in either section of Pelican Horn. However, TP in middle Pelican was even much lower than in upper Pelican, through which the pumped water flowed first. Pumping low-TP water should have decreased TP in both upper and middle Pelican, but TP was not significantly different with CRW present or absent. Even more surprising was the much higher TP in upper Pelican, where low-TP water entered, than in middle Pelican (Table 5). With pumping on for an average of 182 days, and an exchange of 11%/day, each summer during 1986-1988 there was ample time to replace Pelican Horn water many times. Therefore, either the input of TP from ground water or internally from bottom sediment, with its legacy from wastewater input, must have countered the effect of pumping low-TP water into both upper and middle Pelican Horn.

Specific conductance (SC) shows a similar pattern, although pumping low-SC CRW had a slightly greater effect at lowering SC in both lake sections, especially in middle Pelican in 1986-1988 (Table 5). Given that SC is conservative and therefore a good tracer of CRW, and not supplied internally, ground water inflow likely had a much greater effect maintaining SC levels than pumping low-SC CRW had on lowering SC, given that SC in upper Pelican was only slightly lower with CRW in upper Parker being present than absent. While pumping low-SC CRW had a slight effect on SC in upper Pelican, there was no effect on TP, indicating that ground water and internal loading both probably controlled TP in the 1980s, despite pumped inflow from upper Parker with one-fifth the TP concentration that existed in upper Pelican.

SC was only slightly lower in middle Pelican (TS8) in 2017 with the pump on (72 days) than off, while SC was lower with pumping on in 1986-1988, probably because the pump ran continuously in the 1980s (Table 6). Nevertheless, by lowering middle Pelican SC only 40% of the difference in SC, with and without low-SC CRW input (344-191 vs 344-284 µS/cm), indicates that ground water contributed substantially to SC in the lake in the 1980s.

TP was much lower in both upper and middle Pelican in 2017 than in the 1980s with the pump either off or on. That may mean recovery (i.e., reduction in TP) in these two lake sections has occurred in the intervening 30+ years since wastewater was diverted from middle Pelican in 1984. That is, internal loading of P from bottom sediment has apparently decreased, which is the usual pattern of long-term response to a large reduction in wastewater P input; middle Pelican contained nearly 1,000 µg/L of TP prior to diversion. That amounts to a 96% decrease in TP, which is about what high-level, advanced wastewater treatment would have achieved (removal to 50 µg/L TP in the effluent). However, Pelican Horn still appears rather green, because the biomass of algae has not declined with the decline of TP. Biomass of algae, as indicated by chl, is still about the same as in the 1980s; 13 µg/L average in 2017 vs 14 µg/L during 1985-1988. Chl averaged slightly less at TS6/7 in 2017 at 7 µg/L with much lower TP, despite 2.5 times less CRW than in 1985-1988. Chl was 50 µg/L in middle Pelican before wastewater diversion.

Ground Water

Ground water sampling with lake-bed piezometers in 2001 showed an average SC of 860 µS/cm at two sites on the east side and one on the west of Pelican Horn and an average SRP concentration of 200 µg/L at two of those sites, one site on the east and one on the west (Pitz, 2003). Samples from another site on the west side of middle Pelican in vicinity of the outfall, from which wastewater was diverted, had an average SC of 1,400 µS/cm and SRP of 1,200 µg/L. Thus, there were ground water sources along both sides of Pelican Horn in 2001 that contained SC at 2.5-4 and SRP at 4-25 times the current lake concentrations and, depending on ground water flow, may have countered the effect of pumping to dilute P in that lake area.

Ground water flow into upper and middle Pelican Horn was determined by sodium mass balance in 1982 at 0.26 m^3/sec (9.1 cfs, Carlson, 1983). At that flow, ground water inflow would have replaced Pelican Horn water 1.9 times during the summer, while continuous pumping from upper Parker at 1.4 m^3/sec (50 cfs) would replace Pelican Horn water 10.3 times during summer. The combined pumping and ground water flows, with their respective SC levels, should have produced a SC in the lake of around 346 µS/cm, assuming an average of 995 µS/cm from the four ground water sites in 2001 and 228 µS/cm at TS4 during the 72 days of pumping in 2017. That estimated SC is close to the time-weighted average of 352 µS/cm observed in middle Pelican Horn in 2017 (Table 6).

Ground water flow rate apparently has changed little since 1982. A sodium (Na) mass balance with 2017 data indicates a ground water flow rate of 0.21 m^3/sec, similar to the calculated rate of 0.26 m^3/sec in 1982 (Carlson, 1983). Therefore, this assessment indicates that ground water is probably a substantial source of P to Pelican Horn. The four ground water sampling sites in 2001 had high SRP concentrations – average 200 µg/L (Pitz, 2003). The ground water sampling site used in 1982 and 2017 (TS9) had less TP than the sites had SRP in 2001 (76 in 2017 and 48 µg/L TP in 1982). Total P is the sum of particulate P and soluble reactive P (SRP). The high SRPs at the piezometer sites in 2001 were associated with low dissolved oxygen (DO) and anoxic conditions at two of the four sites, which would allow high SRP to persist (Pitz, 2003). Such high concentrations under anoxic conditions would not persist once the ground water entered the oxygenated lake. One of the four ground water sites in 2001, which was proximal to that in 1982 and 2017 (TS9), had an SRP of 44 µg/L. That was more than double the level in 1982 (19 µg/L). Thus, ground water P concentrations determined at TS9 have increased since 1982; average TP was 76 µg/L in 2017 vs 48 µg/L in 1982.

Assuming an equilibrium existed between inflow and lake TP in middle Pelican Horn in 2017, even though the pump ran only half the time (72 days), with 76 µg/L entering in ground water (TS9) and 26 µg/L at TS4 when the pump was on, the calculated mixed TP (or inflow concentration) would have been 38 µg/L, which is close to the time-and area-weighted average observed of 43 µg/L during summer (Table 5). Thus, ground water was an important source of P to Pelican Horn in 2017. Internal loading from sediment was probably important also because some of the 38 µg/L inflow concentration would have been lost to sedimentation, resulting in a lake concentration around 27 µg/L ($TP_{lake} = TP_{inflow}/1 + 0.15^{0.5}$), where 0.15 is the residence time fraction for 150 days per year; Brett and Benjamin, 2008). The difference between 38 and 27 is 11 µg/L, which would be likely due to recycling from bottom sediment, or internal loading. A ground water TP at TS9 in 2017 may not have been entirely representative for Pelican Horn, but probably more so than 200 µg/L SRP observed on average in 2001. Such a high ground water inflow concentration would have produced a mixed inflow of 60 µg/L and 43 µg/L in the lake, equaling the observed lake concentration, which is unlikely, because much of the ground water sampled in 2001 was anoxic so those high SRPs would not have persisted in the lake, and internal loading would have provided some input.

Sodium and nitrate-N were much higher in ground water at TS9 – averaging 65.5 and 14 mg/L, respectively – than in 1986-1988 (2.6 and 6 mg/L). Sodium now is near the level in sewage effluent prior to diversion (82 mg/L) and nitrate-N is much greater than usually occurs in treated sewage effluent (7 mg/L). Both constituents are soluble and readily pass through soil. The high and increased levels suggest that unsewered development on the east side of Pelican Horn may be providing a source through leaching from septic tank drain fields.

Alum Treatment of Middle Pelican Horn

The central half of middle Pelican Horn was treated with alum (aluminum sulfate) by Aquatic Solutions on May 30, 2017. Alum has been used in hundreds of lakes, mostly in the US, but also in Europe, over the past 50 years. The purpose is to inactivate phosphorus (P) in bottom sediments to reduce recycling or internal loading. Alum treatments have been widely successful throughout the world (Cooke et al., 2005; Huser et al., 2015). The dose added to the middle half (100 acres) of middle Pelican was about 17 mg/L of Al.

Sediment cores were collected outside the treated area in middle Pelican, north of I90, and another in lower Pelican south of I90 on the day of treatment and analyzed for P fractions by IEH Analytical Laboratories in Seattle. The P fraction most responsible for internal loading is complexed with iron (Fe-P) and is considered mobile. That fraction averaged 0.038 and 0.029 mg/g dry weight in the top 10 cm of the cores in middle and lower Pelican Horn, respectively. Corrected for density, concentrations were 17.7 and 21.8 $\mu g/cm^3$. To inactivate that mobile-P fraction would require alum at about 15 mg/L as Al. The treatment added 17 mg/L, so most of the mobile-P in half of middle Pelican should have been inactivated. Actual effectiveness of the treatment on TP in that lake section is unclear because TP data are unavailable from middle Pelican during previous recent years for comparison. However, there was little difference in average TP concentrations between untreated upper and treated middle Pelican Horn (45 vs 41 µg/L, time-weighted), respectively, whether pumping was on or off (Table 5). Treatment effectiveness on lake TP was minimized given that only half the area was treated and wind would have mixed water from the untreated area with water in the treated area. Also, TP would have been added from ground water and sediment bioturbation by carp may have minimized alum's effect. A future sediment core from the treated area should show the fraction of mobile-P that was inactivated with the treatment.

Sediment was analyzed in Pelican Horn in 1983-1985 (Okereke, 1987). Total P averaged 0.98 mg/g in the top 10 cm then and 0.76 mg/g dry-weight in the two cores from 2017. Thus, sediment TP has apparently decreased over the 33 years since diversion of wastewater.

Phosphorus Loading

External TP loading in 2001 was about half the average during 1984-1988, which followed diversion of wastewater (Table 7). Loading in 2001 was well below the range during 1984-1988

(9,964-15,170 kg; Welch et al., 1989; Jones and Welch, 1990). Also, estimates of loading from ground water in 1984-1988 were double that in 2001, as was loading from CC and CRW (Carroll, 2006). Less external loading in 2001 was not due to dilution of lake water from 70% more CRW. Added CRW increases load (concentration x flow), but with much lower inflow TP concentration (20 μg/L) than the inflow streams (63 μg/L). Exceptionally low inflows probably explain much of the lower total loading in 2001; CC and RFC flows were low (20^{th} percentile) during 2001 (Carroll, 2006). External loading in 2017 was similar to that in 2001, despite less input from CC+ELC+RCW due to less CRW (Table 7).

Internal loading varied considerably, year to year during 1984-1988, from 2,720 to 19, 845 kg and there was no internal loading in 1978 and 1980, with the latter due to the Mt. St. Helens ash layer on the sediment that blocked P release. However, no internal loading in 2001 (i.e., negative outflow-inflow load), may be due to the lack of lake data in September when TP tends to increase, as anoxia develops at depth and P release from bottom sediment increases. For example, TP at TS5/6 increased to an average of 40 μg/L in September 2017 from 23 μg/L during May-August, and that was at 5 m, which did not include the probable increased concentrations at depth. Much of that late summer increase likely came from sediment P release or internal loading.

Although the estimate of internal loading in 2017 is uncertain, due to unknow outflow volume, it was nonetheless probably positive, especially given that no TP data below 0.5 m were available to determine average, v-w lake TP (Table 7). Another approach to estimate the effect of internal loading on lake TP, indicates that about 60% of observed whole lake TP may have come internally from bottom sediment (Table 8). Predicted lake TP from average inflow concentration (41 μg/L) and lake water residence time (0.92 yr) gives 20 μg/L, while observed TP was 51 μg/L in 2017, leaving a balance of 31 μg/L that likely came from internal loading (61%). The fraction from internal loading in 1984-1988 was 45% (28 μg/L) using the same model to estimate lake TP from inflow concentration (Brett and Benjamin, 2008). The fraction of total loading that was internal in 1984-1988, determined directly from mass balance, was 43%, which is similar to that estimated using the model (45%, Table 7). Thus, there was probably substantial internal loading in 2017.

Positive internal loading, even after over 30 years of CRW input and after wastewater diversion, represents a continual threat to lake water quality, particularly as the summer progresses. Continual CRW input annually is essential to dilute internal as well as external loading to maintain acceptable lake water quality for recreation and water use and minimize the frequency and abundance of HABs. Without CRW in 2017, average v-w whole lake TP would have been on the order of 67 μg/L (using the model), instead of 51 μg/L as observed (Table 8).

Average TP at TS5/6, which was used as the indicator of the effect of CRW in the 1970s and 1980s, would likely have been on the order of 90 μg/L, around four times the 25 μg/L average in 2017 and the 20 μg/L average during 2002-2016 (Figure 3). The average lake TP concentrations at TS9 in 1996-1998 was 93 μg/L when CRW inputs were relatively low averaging 77 10^6 m^3, one fourth the 2002-2016 average (Figure 4; Table 3). However, TP was surprising low in 2017, given the relatively low CRW input of 93 x 10^6 m^3 (Table 3). There is considerable variability in lake TP for any given CRW inflow, although consistently low for inputs above 250 x 10^6 m^3, as indicated in Figure 3.

Summary

1. Phosphorus concentrations in lower CC (Hwy 17) have increased substantially during the past 14 years compared to the 1980s, while TP in RFC has not changed. However, nitrate increased in both CC and RFC, although that should not have caused increased algae, which is related more to TP concentration. The increased nitrate may be due to increased use of fertilizer.
2. Inflow of CRW averaged much higher during 2002-2016 than in the 1980s. The higher average inflow of CRW of 325 x 10^6 m^3 (263 x 10^3 AF) over an average of 208 days during 2002-2016, was 50-100% more than in the 1970s and 1980s. That higher recent input to half the lake volume, which was affected most directly by CRW input, would have been replaced at about 2%/day. Most of that low-TP water would have remained in the lake all summer, being replaced slowly by CC and RFC normal inflow, even if CRW input were largely terminated after early June, which was the case in 2017, when total CRW inflow was only 93 x 10^6 m^3 (76 x 10^3 AF).
3. Persistence of CRW in 2017 was demonstrated by the low average SC at TS6 of 288 µS/cm that represents 60% low-TP CRW and 40% original lake water with high SC and TP. With 2.5-fold more inflow of CRW in 2016 (234 x 10^6 m^3; 189 x 10^3 AF) than in 2017, average SC was lower at 254 µS/cm representing 68% CRW.
4. Total P concentration in lower Parker Horn (TS5) and south lake (TS6) has decreased markedly over the past 40 years since the Clean Lakes Project began; from six-fold less TP since before the project (1969-1970), to three-fold less TP since the initial post-project period (before wastewater diversion) during 1977-1984, and two-fold less TP since 1986-1988. The whole lake was hypereutrophic (TP > 100 µg/L) before the project began, eutrophic (> 30 µg/L) after, and mesotrophic (< 30, > 10 µg/L) since the 1980s.
5. The concentration of algae (and bloom magnitude) has also declined in proportion to decreased TP since the Clean Lakes Program began, with chl, the indicator of algal biomass, decreasing two and a half, three and eight-fold during the respective periods noted in 4 above. Transparency, or visibility through the water determined by lowering a black and white Secchi disc, did not improve in 2017 compared to the post project period. Transparency is usually closely dependent on chl concentration and should have been on the order of 2 m, consistent with the low chl concentration of 7 µg/L, which indicates mesotrophy (>1 <2 m). Cause for the discrepancy is uncertain, but not due to measurement error. The Secchi measurement is a prime indicator of lake quality and usually very reliable.
6. Other areas of the lake, less affected by low-TP CRW, like the upper main arm, remained hypereutrophic with much larger average TP and chl concentrations than in those areas more affected by CRW (TS5/6/7). Average TP and chl were ever higher at TS11 (Connelly Park) than in 1986-1988, and CRW reaches into that area. Although Pelican Horn water quality (TP, chl) had greatly improved since wastewater was diverted in 1984, it was still

eutrophic with much higher chl and lower transparency than in Parker Horn and south lake.

7. Cyanobacteria were abundant during mid-July-September at all sites in the lake except Pelican Horn, where green algae and diatoms dominated. *Microcystis aeriganosa* (MA) was the most cyanobacteria. It produces microcystin (MC), a liver toxin, and is a major constituent of harmful algal blooms (HABs). Cell concentrations of MA were well above those producing moderate to high MC concentrations in 9 western Washington lakes. Cyanobacteria biomass was much higher in the main arm (TS11/12) than at other sites in 2017, consistent with higher TP and chl. Biomass of cyanobacteria averaged higher at TS5/6 in 2017 than in 1986-1988, despite lower TP and chl in 2017, probably because algal composition was determined only from mid-July through September in 2017, when cyanobacteria were usually more abundant than in spring-early summer. Nevertheless, there are still HABs in the lake, some of which may be toxic, despite the marked improvement in water quality resulting from increased CRW inputs and these blooms often produced floating scums nearshore, as was the case at Connelly Park (Figure 7).
8. Pumping low-TP CRW from upper Parker Horn (TS4) into Pelican Horn had no effect on TP concentrations in 1986-1988. TP concentrations were not different with the pump on or off and not lower in upper Pelican, where CRW inflow enters, than in middle Pelican. However, SC was less with the pump on than off, although higher in upper than in middle Pelican. Apparently, ground water input with high SC (860 µS/cm in 2001) must have partially offset the effect of pumping low-SC CRW. Even less effect from pumping low-TP CRW means that inputs to Pelican from both ground water and internal loading from bottom sediment completely offset the effect of pumping. A similar pattern occurred in 2017, although TP was much lower in both upper and middle Pelican, probably due to much reduced internal loading from bottom sediments 33 years after wastewater diversion from middle Pelican. That is supported by lower sediment-P concentrations from cores taken in the 1980s and in 2017.
9. Ground water supplied much of the TP to Pelican Horn in 1982 and 2017 as evidenced by sodium balances, with the difference between observed lake TP and that expected from ground water having been due to internal loading from sediment. Sodium mass balance indicated similar (within 20%) ground water flows to Pelican Horn in 1982 and 2017. The flow rates are also reasonable from a comparison of ground water **SC** in 2001 and lake SC in 2017.
10. The alum treatment to half of middle Pelican Horn in May, 2017, may have inactivated most of the mobile-P, which is the source for internal loading, as evidenced from sediment cores collected from middle untreated part and lower Pelican Horn in May compared with the dose actually added. However, the treatment's effect was not apparent, probably due to wind mixing with the untreated area and contribution of TP from ground water and sediment bioturbation by carp, although data from previous recent years were unavailable for comparison. Future cores should show the fraction of mobile-P that was inactivated.
11. Phosphorus loading in 2017 from external sources was similar to that in 2001, but about half the average during 1984-1988, after wastewater was diverted. Loading in 2001 was low due to lower than normal flows in CC and RFC and low in 2017 due to

much less CRW. Internal loading due to P release from bottom sediment was likely as high in 2017 as in 1984-1988. Lack of internal loading reported for 2001 was probably due to minimal lake TP from late summer when internal loading is greatest due to developing anoxic conditions overlying the bottom sediment. Increased water column TP in late summer reflects internal loading in a mass balance analysis. While the actual amount of internal loading in 2017 is uncertain, due to lack of observed outflow volume necessary for a mass balance, that source was nevertheless substantial.

That a substantial internal loading occurred in 2017 was substantiated by the difference between whole lake v-w TP concentration observed and that predicted by a simple mass balance model using observed inflow TP concentration and water residence time. That approach resulted in 45% of TP loading coming from internal sources in 1984-1988 compared with 43% calculated directly from mass balance. Percent internal loading was 61 in 2017 using the same model approach and 46 from mass balance with estimated outflow volume.

The apparent persistence of internal P loading after 30 years of dilution with CRW input is important to summer lake water quality. Large inputs of CRW dilute the internal as well as external sources of TP. Without CRW in 2017, whole lake v-w TP concentration would have been an estimated 67 µg/L instead of the observed 51 µg/L, and probably much higher at TS5/6 than the observed 25 µg/L, and likely much larger algal blooms than observed.

Recommendations

1. Determine the toxicity (microcystin) level of cyanobacteria blooms (HABs) in Parker Horn and south lake (TS5/6) and main arm (TS11/12) by collecting near shore samples of scums and sending samples to Department of Ecology for analysis.
2. A minimum of 100 x 10^6 m^3 (81 x 10^3 AF) of CRW is needed to dilute external and internal TP loading to Parker Horn and south lake (TS5/6) and provide acceptable water quality (TP = 30 µg/L) and minimize HABs.
3. Propose constructing facilities to transport another 40 10^6 m^3 of CRW to the main (Rocky Ford) arm from TS3 to the Connelly Park area (TS11) to dilute the high TP (180 µg/L) entering from Rocky Ford Creek. That should roughly reduce lake TP to about half the current 80 µg/L. If delivered over 6 months, the flow would be about 2.5 m^3/sec or 90 cfs over a distance of about 4 miles. That quantity may not be possible given constraints on CRW availability and construction costs. More in-depth analysis is necessary to further consider such a project, considering ground water sources of P and lake response. Delivering CRW to upper main arm was originally in the Clean Lakes Project.
4. Treat the anoxic area in Parker Horn and/or south lake (TS5/6) with alum to inactivate sediment mobile P and reduce internal P loading that contributes to poor water quality in late summer as DO gradually depletes over the deepest area promoting the release of mobile P. This source can be substantial, especially with increased wind mixing that entrains bottom P-rich water into the lighted zone. Such instability produced the highest

internal loading in 1985 that occurred during the 1970s-1980s; 19, 845 kg, which was 3 times the 11-year average. To further consider an alum treatment to this area, DO/temperature profiles should be determined at TS5,6, and 7 at 1 m intervals from surface to bottom during 2018, in order to determine the anoxic area. A bathometric map of lower Parker Horn and south lake is also needed to accurately determine the area for treatment.

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Table 1. Average nutrient concentrations in inflows to Moses Lake during 1986-1988 (Welch et al., 1988) compared with 2003-2017 in ().

	TP	SRP	NO_3-N
Crab Creek	47 (75)	7 (40)	685 (950)
Rocky Ford Creek	164 (165)	107 (131)	669 (1405)
East Low Canal (CRW)	19 (7[1])	4 (NA[1])	30 (10[1*])

*detection limit; [1]2017 only

Table 2. Average specific conductance in µS/cm used as tracer of CRW (Columbia River water) throughout Moses Lake.

CRW	142	
Moses L natural	445	(1969-1970)
Crab Creek	491	
Rocky Ford Cr	371	

Table 3. Annual Columbia River water (CRW) inflows, half-lake volumes (77.8 X 10^6 m^3) replaced and average spring-summer total phosphorus (TP) and chlorophyll (chl) concentrations and transparency (SD) in the south lake (TS 6) and Lower Parker Horn (TS5). USBR data (n=4) from TS6 only. *Spring-summer average TP from the north outlet (USBR) equaled TP in lower Parker Horn (TS5) and south lake (TS6) together (MLIRD) in 2017.

		10^6 m^3	AFX10^3	0.5 Lake Volumes	µg/L		m
	Years	CRW	CRW	0.5	TP	CHL	SD
UW	69-70	5	4	0.37	152	58	0.7
UW	77-84	105	85	1.36	74	21	1.4
UW	86-88	178	144	2.3	41	17	1.6
DOE	01	284	230	3.7	17	11	NA
USBR	02-16	325	263	4.8	23	NA	NA
MLIRD	2017	93	76	1.2	25*	7	1.4

Table 4. Biovolume in mm^3/L of plankton algae and cyanobacteria at six sites in mid to late summer, 2017.

	TS5	TS6	TS8	TS10	TS11	TS12
			July 17			
Total	243	19.3	61.8	ns	147	59.4
cyanobacteria	13.0	10.7	5.4	ns	144	28.5
% cyanos	5	56	9		98	48
			September 5			
Total	15.2	61.9	81.0	61.5	26.3	42.3
cyanobacteria	11.6	13.4	5.9	2.4	14.1	39.2
% cyanos	76	22	7	4	53	93
			October 2			
Total	40.9	21.2	90.6	31.6	16.5	23.5
Cyanobacteria	24.8	8.2	6.5	4.2	15.5	13.9
% cyanos	61	39	7	13	94	59
			Seasonal average			
Total	99.8	34.1	79.5	46.6	62.4	41.7
Cyanobacteria	16.5	10.8	5.9	3.3	58.0	27.2
% cyanos	47	39	7.7	3.7	81.7	66.7

Table 5. Average TP (µg/L) in Upper (TS10) and Middle (TS8) Pelican Horns during spring-summer 1986-1988 with continuous pumping of CRW with 29 µg/L TP from Upper Parker to Upper and Middle Pelican at 1.42 m^3/sec (50 cfs) for an average of 182 days/year, and with/without pumping in 2017 when pump ran for 72 days. Number of samples = n).

	Upper Pelican	Middle Pelican
CRW present 1986-1988 (n=16)	153	70
CRW absent 1986-1988 (n=22)	145	67
Pump on 2017 (n=3)	56	37
Pump off 2017 (n=6)	39	43

Table 6. Average SC (µS/cm) in Upper (TS10) and Middle (TS8) Pelican Horns during spring-summer 1987-1988 with continuous pumping of CRW with 191 µS/cm from Upper Parker to Upper Pelican at average of 182 days/year, and with/without pumping in 2017 when pump ran for 72 days. Number of samples = n.

	Upper Pelican	Middle Pelican
CRW present 1986-1988 (n=9)	339	284
CRW absent 1986-1988 (n=14)	362	344
Pump on 2017 (n=3)	NA	337
Pump off 2017 (n=5)	NA	361

Table 7. Average TP loading in kg after wastewater diversion in spring-summer (May-September). Loading from 1984-1988 from Welch et al. (1989); 2001 from Carroll (2006); 2017 using TP data collected by MLIRD and recorded flow.

	CC+ELC+RCW	RFC	GW	Total External	Internal
1984-1988	4,538	4,167	3,886	12,591	9,346
2001	2,166	2,809[1]	1,720	6,695	- 643
2017	1,484	3,799[1]	1,688[2]	6,951	5,851[3]

[1]includes Trough Lodge Hatchery (from Carroll, 2006); [2]25% of total TP load as in 2001 (Carroll, 2006, table 14); [3]estimated from 2017 inflow volume X 1.5 (average outflow/inflow, 1984-1988) X 50.8 mg/m^3 v-w lake TP = 12,802 kg outflow; 12,802 – 6,951 = 5,851. CC = Crab Creek; ELC = East Low Canal; RCW = Rocky Coulee Wasteway; RFC = Rocky Ford Creek; GW = ground water.

Table 8. Average TP load and volume inflow (May-September), v-w inflow TP, water residence time, observed v-w whole-lake TP, predicted v-w whole-lake TP, and lake TP due to internal loading as difference between predicted and observed lake TP. Inflow includes ground water at 18% of total as in 2001 (Carroll, 2006). Lake volume excluding Upper and Lower Pelican Horn = 153.9 m^3; 1,234 m^3 = 1 AF.

	TP load	inflow	TP_{in}	T	TP_{lake}	$TP_{lakepred}$	$TP_{internal}$	% internal
	kg	10^6 m^3	µg/L	yr	µg/L	µg/L	µg/L	
1984-1988	12,591	199	63	0.77	62	34	28	45
2001	6,695	340	20	0.45	20	12	8	40
2017	6,951	168	41	0.92	51	20	31	61
2017 wo/CRW	6,300	74.7	84	2.1	67	34	33	49

TP_{lake} volume – weighted; $TP_{lakepred}$ predicted by $TP_{lake} = TP_{in}/(1 + T^{0.5})$; $TP_{internal} = TP_{lake} - TP_{lakepred}$; TP_{in} = TP load/inflow volume; T= water residence time (lake volume/inflow rate) for 5/12 year.

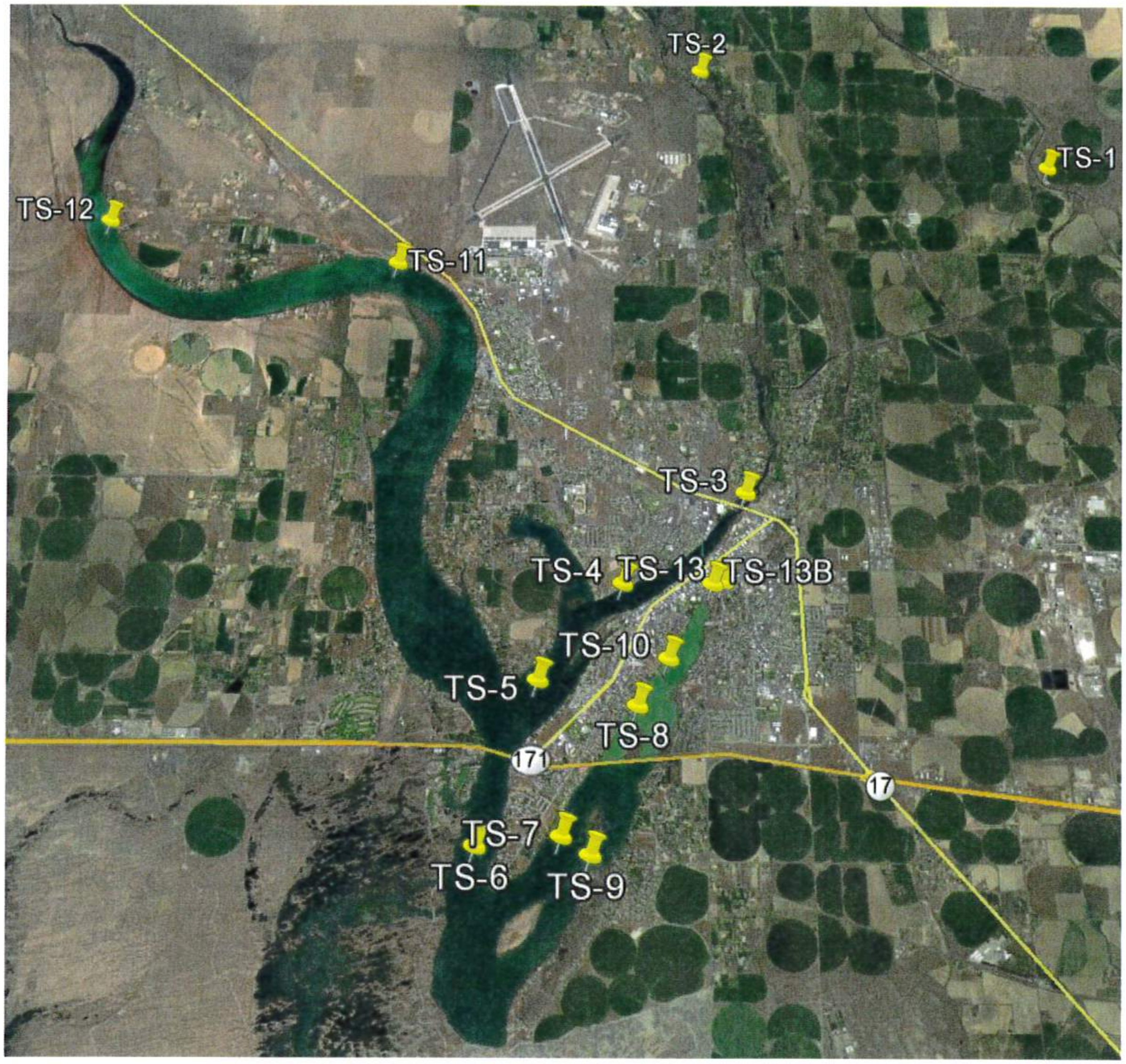

Figure 1. Sampling sites during 2017, most similar to those during 1969-1970 and 1977-1988 (Welch et al., 1989).

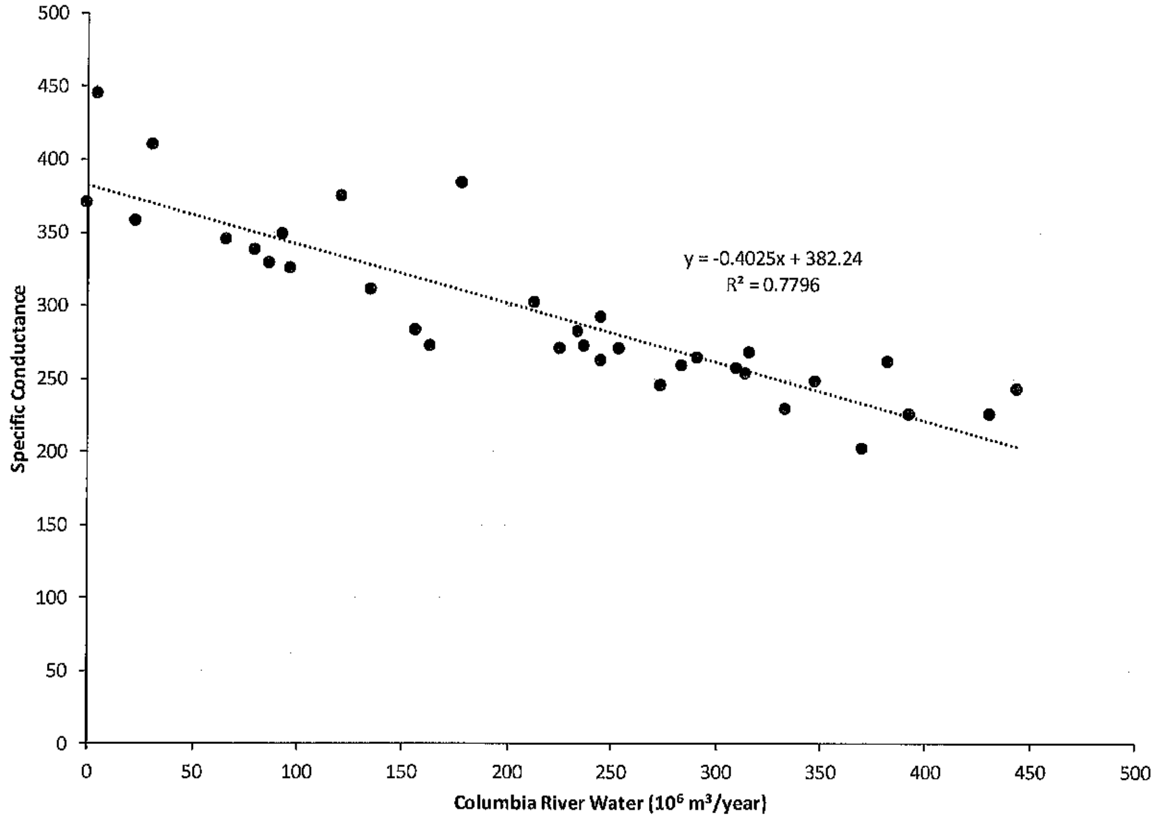

Figure 2. Relation between CRW input and spring-summer average specific conductance in μS/cm in lower Parker Horn (TS5) and the south lake (TS6) during 1969-1988 (May-September) from Welch et al. (1992) and south lake near north outlet during 1995-2017 from USBR (mid-April to mid-October).

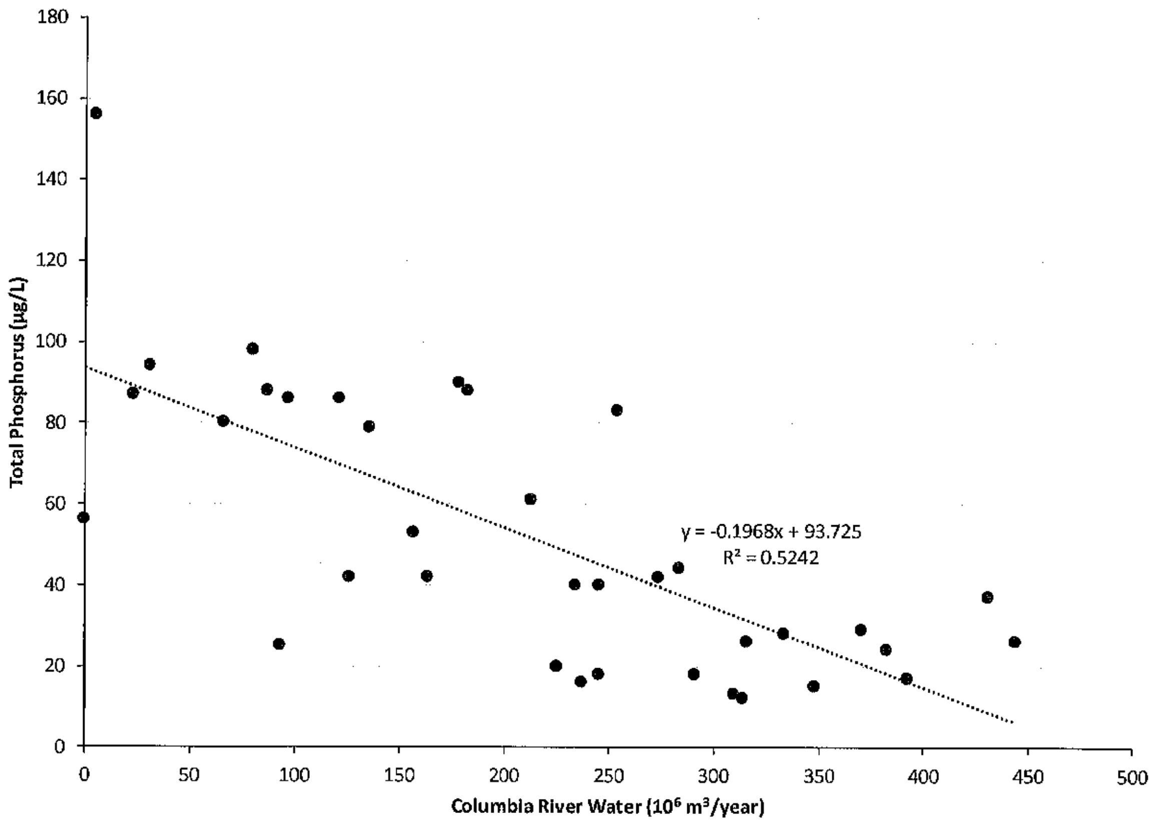

Table 3. Relation between CRW input and spring-summer average TP concentration in lower Parker Horn (TS5) and the lower lake (TS6) during 1969-1988 (May-September) from Welch et al. (1992) and south lake near north outlet during 1995-2017 from USBR (mid-April to mid-October).

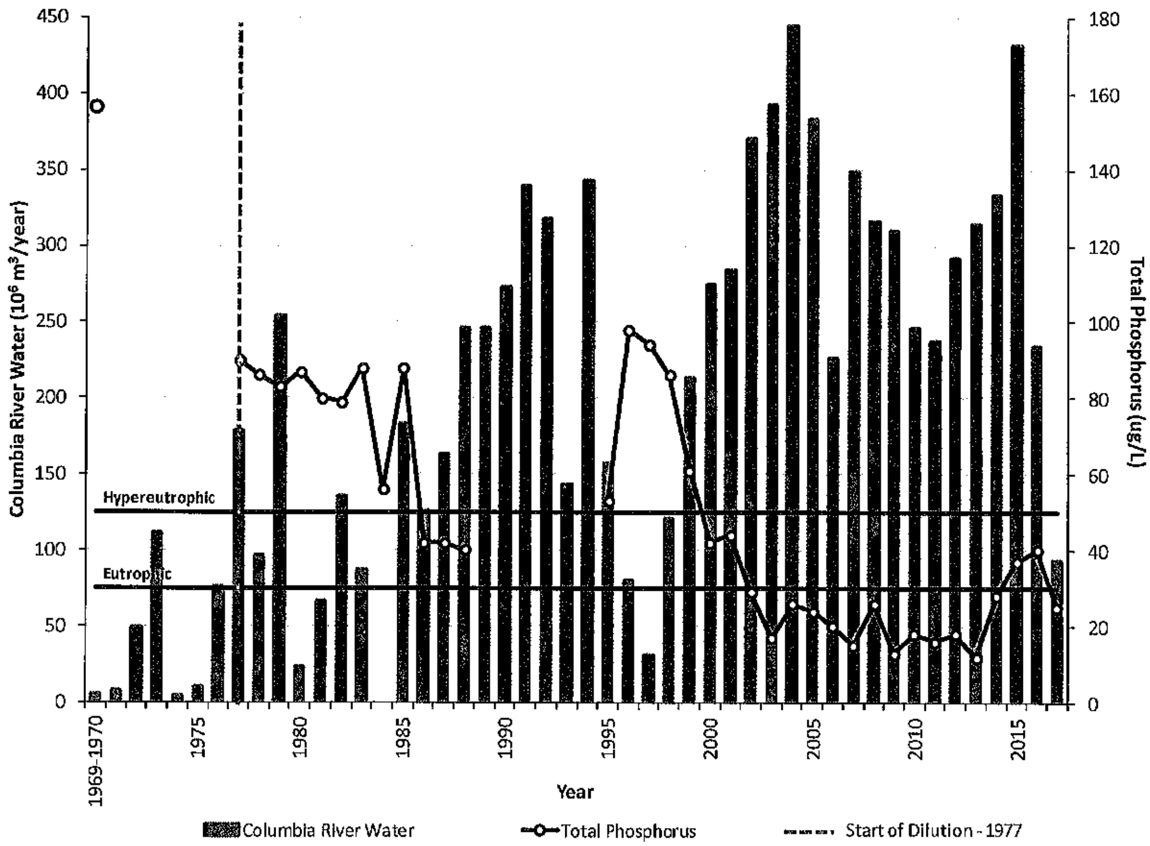

Figure 4. Inflow of Columbia River water into Moses Lake over nearly 50 years and spring-summer average spring-summer TP concentration in lower Parker Horn (TS5) and south lake (TS6) during 1969-1988 (May-September) from Welch et al. (1992) and south lake near north outlet during 1995-2017 from USBR (mid-April to mid-October).

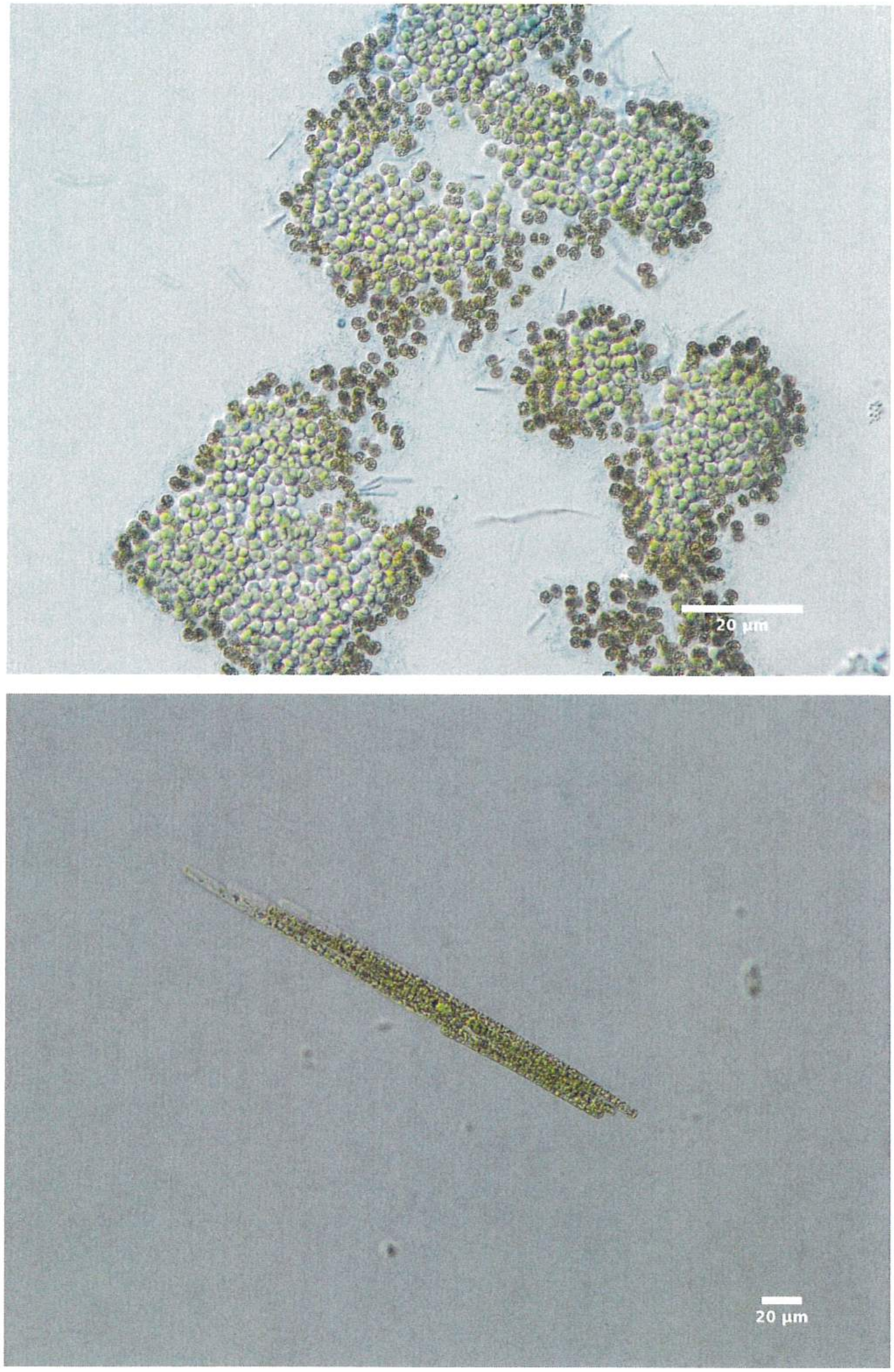

Figure 5. The cyanobacteria *Microcystis aeruginosa* top and *Aphanizomenon flos aquae* bottom, respectively 600x and 200x magnification. These are bloom forming HABs that often form surface scums.

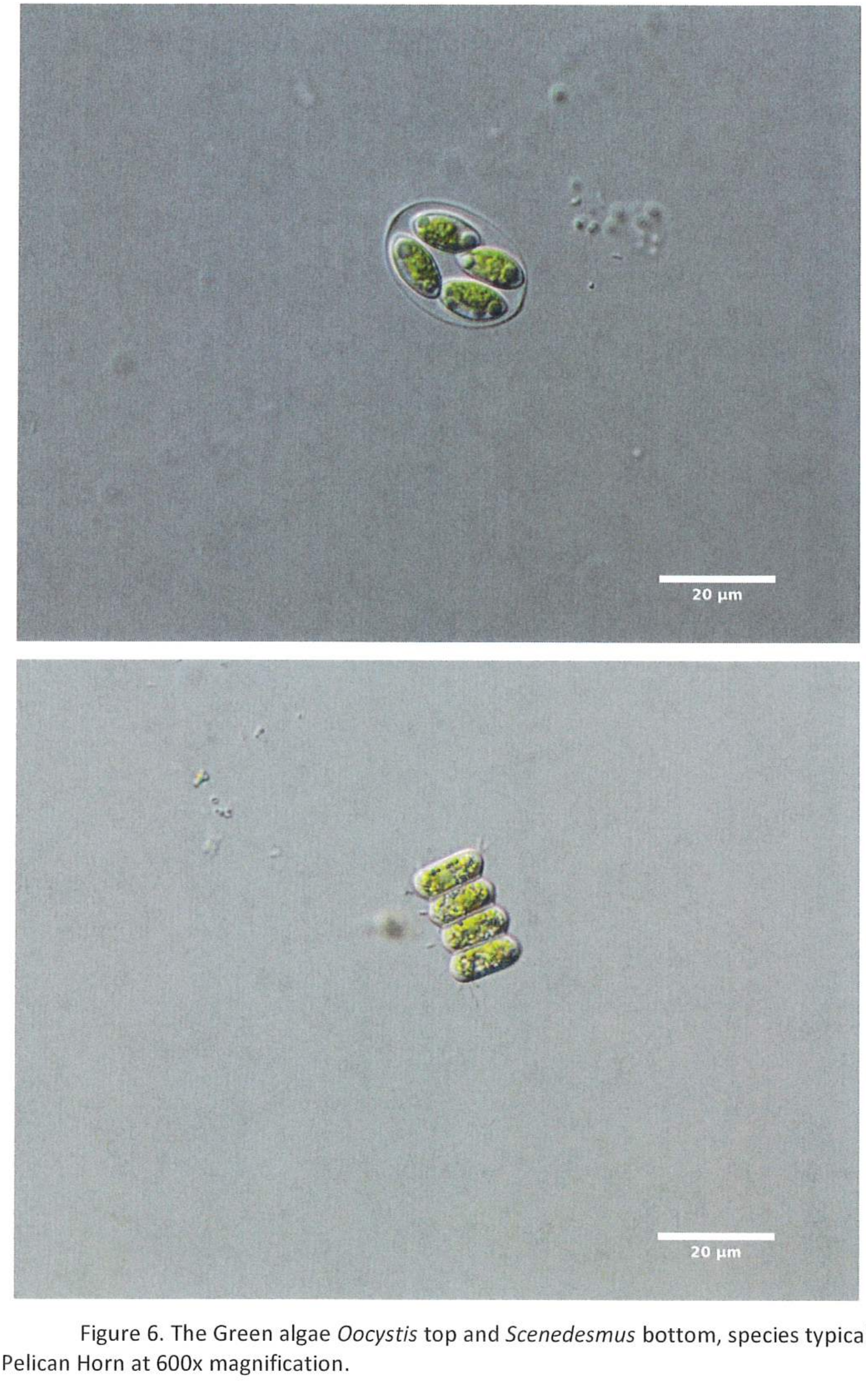

Figure 6. The Green algae *Oocystis* top and *Scenedesmus* bottom, species typical in Pelican Horn at 600x magnification.

Figure 7. Cyanobacteria surface scum, formed in open water and blown near shore by wind at the boat launch Connelly Park near TS11. The scum was probably composed of organisms shown in Figure 5.