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

The water quality in Moses lake has greatly improved over the past four decades in response to varying inputs of low-phosphorus Columbia River Water (CRW). Phosphorus is the key nutrient that determines the amount of algae. As total phosphorus (TP) increased, the prospect of harmful algal blooms (HABs) increases as well. That is the case for lakes and reservoirs in general and Moses Lake is no different (Welch, 2009; Welch et al., 2015).

As a result of dilution by CRW, TP in Parker Horn and south lake decreased from a spring-summer average of 152 µg/L (ppb, parts per billion) two years before to 74 µg/L during eight years after CRW input began, to 41 µg/L the following three-year period after wastewater was diverted from middle Pelican Horn (Welch et al., 1989; Welch et al., 1992). The latter reduction in TP was due largely to an increase in CRW input from an average of 105 to 178 X 10^6 m^3 (85,000 to 144,000 AF). Input of CRW continued at similar amounts through most of the 1990s, but increased further after 2000 to an average of 284 X 10^6 m^3 (230,000 AF) during 2001-2016. That increase reduced average TP in Parker Horn and south lake to 23 µg/L (Welch, 2017). More on the history of dilution and the Clean Lakes Project and sources of funds for that effort in is Welch (2017) http://mlird.org/lake/mosesreport2017-3.pdf.

Inputs of CRW were much less in 2017 and 2018 at 95 and 130 10^6 m^3 (78,000 and 106,000 AF) during spring-summer when TP in Parker Horn and south lake averaged 25 and 40 µg/L, respectively. Whole lake, volume-weighted (v-w), including Rocky Ford Arm (RFA) were nearly double those concentrations at 52 and 71 µg/L, due to the input of high-TP water from Rocky Ford Creek (RFC), most of which is in the soluble form (SRP) readily available for algal uptake. These levels were at and well above the whole-lake goal set by the WA Dept. of Ecology (WA DOE) of 50 µg/L (Carroll, 2006).

Although CRW reaches well up into RFA, it transports water from Parker Horn, which has higher TP concentrations than CRW's 7 µg/L that initially enters Parker Horn. As a result of the higher TP concentrations in 2018, HABs were a more noticeable problem and high microcystin concentrations in nearshore algal samples.

This report summarizes data from water samples collected by MLIRD at 8 sites in the lake and 4 inflows during 2017 and 2018 (Figure 1). Conditions in 2017 and 2018 are compared with previous published data (see Welch, 2017 for a more complete listing). Reasons for higher TP in 2018 than 2017 are discussed. Measures to improve water quality in Moses Lake are recommended.

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.

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 during regular monitoring events. 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. (2019).

Phosphorus Loading

External loading during May-September in kg was estimated as average surface inflow volume from respective sources multiplied by their average TP concentrations during May-September, rather than on a monthly basis, as in previous years, due to lack of monthly outflow volumes. Average flows and TP concentrations in Crab Creek (CC) and Rocky Ford Creek (RFC) in 2017 were 1.5 and 1.71 m^3/sec (53 and 60 cfs) and 42 and 149 µg/L TP, respectively. Flows and TP concentrations in CC and RFC during 2018 were, respectively, 1.9 m^3/sec (66 cfs) and 3.42 m^3/sec (120 cfs) and 43 and 141 µg/L. Total P concentrations in CC used for loading calculation were from TS3, which is downstream from RCW and represents a mixture of water from CRW, CC and RCW. Average TP in CRW was 7 µg/L. Base flow in RCW was assumed constant at 0.55 m^3/sec (19 cfs) from Carroll (2006). Flows in CC and RFC and TP in RFC were from USBR.

Internal loading was estimated from mass balance using external inflow load (I), outflow load (O), and change in average volume-weighted (v-w) lake TP_l to calculate negative (internal loading) or positive sedimentation (S):

$$S = I - O\ (\Delta\ TP_l)$$

Outflow TP mass was estimated as outflow volume X volume-weighted (v-w) lake TP. Lacking observed data, outflow volume was estimated as the average ratio (1.5) of outflow/surface inflow during 1984-1988 (Welch et al., 1989). That inflow/volume ratio varied by 21% during that 5-year period. Also, the average v-w lake TP (excluding Upper and Middle Pelican Horn) was determined from observations at 0.5 m depth, so the normally higher concentrations at depth were not included, resulting in a probable underestimate of internal loading. Thus, there is considerable uncertainty in the estimate of internal loading. Nevertheless, v-w whole-lake TP increased by 13 µg/L during May-September, 2017, and by 24.5 µg/L during that 5-month period in 2018. That difference in itself indicates internal loading in 2018 was likely at least double that in 2017.

Specific conductance (SC) was used to trace CRW in the lake and determine % lake water or % CRW according to (Welch and Patmont 1980; Cooke et al. 2005):

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

$$\%\ CRW = 100 - \%\ LW$$

Results and Discussion

Water quality during 2017 and 2018

Production of algae was much greater throughout the lake in 2018 than in 2017, despite the lake receiving more CRW in 2018; 105,000 AF versus 76,000 AF. Chlorophyll (chl), a measure of algal biomass, averaged 18 µg/L (or ppb = parts per billion) during May-September in Lower Parker Horn (TS5) and South Lake (TS6) in 2018, versus 7 µg/L in 2017 (Table 1). Average May-September chl at TS5/6 was about the same during 1986-1988 (17 µg/L). The difference between the last two years was even greater during the more productive months of July-September when average chl at the two sites was 24.6 and 6.3 µg/L in 2018 and 2017, respectively.

Measured transparency was relatively low during both 2017 and 2018 in Parker Horn and South lake (TS5/6), averaging only 1.4 m, despite much different chl concentrations (Table 1). Transparency was less than 1 m at higher chl concentrations at TS5/6 in July-August and at TS11/12 when chl was higher. There are other light attenuating substances in the lake besides chl, that cause low transparency, but most of the attenuation was usually due largely to chl concentration (Table 1).

Algal biomass as chl was greater in 2018 because total phosphorus (TP) concentration was greater, averaging 40 and 25 µg/L during May-September in 2018 and 2017, respectively (Table 1). Average TP in 1986-1988 was 41 µg/L, similar to that in 2018. As with chl, the difference in TP between 2018 and 2017 was even greater during July-September: 51 versus 30 µg/L (Table 1). Why was there more TP in 2018 than 2017? The added TP was probably due to more recycling of phosphorus from bottom sediments. Inflow TP to Parker Horn from Crab Creek (CC, TS3) was relatively constant during the summer in both years; July-September averages were 37 and 38 µg/L (Table 2). The resulting lake concentration, if the inflow from CC were the only source of phosphorus to Parker Horn, should have been actually less than the inflow concentration due to normal sedimentation loss that occurs in the lakes. That was the case in 2017; lake TP was less (- 8 µg/L) than inflow TP due to such normal loss through sedimentation (Table 2). In 2018, however, Parker Horn and South Lake TP averaged more than the inflow from CC during July-September (51 vs 37 µg/L), indicating that the net difference came from internal sources, most likely recycling from bottom sediment (Table 2). Unaccounted for sources, other than CC, such as seepage from on-site septic tank drain fields, were probably not responsible for the higher-than-inflow TP (+14 µg/L), because those sources were likely seeping at the same rate in 2018 as in 2017 when the difference between inflow and lake TP was negative (- 8 µg/L) during July-September (Table 2). There was loss of inflow TP due to sedimentation in 2018 as well, but internal loading via recycling from sediment was enough to not only equal but exceed sedimentation loss, as evidenced by lake TP exceeding inflow TP.

There was no net increase in TP in RFA (TS11/12), indicating no net recycling of phosphorus from bottom sediment (Table 2). That was also the case during 1977-1988 determined by phosphorus mass balance modeling. Internal loading, or recycling, may have occurred, but the magnitude was apparently insufficient to exceed sedimentation loss. However, the net difference between inflow and lake TP was less negative in 2018 when TP at TS11/12 was greater than in 2017, while RFC inflow TP concentration was about the same.

Algae

There were large blooms of cyanobacteria (alias blue-green algae) during summer 2018. Nearshore samples were sent to WA DOE that showed microcystin (MC) concentrations well above the state recreational guideline of 6 µg/L. The cyanobacterium *Microcystis aeruginosa* (MA) is a usual source of microcystin and it was the major blue-green bloom former in 2018 representing 60 - 98% of total blue-green biomass in Parker Horn, South lake and Rocky Ford Arm (Table 3). That species was also the dominant cyanobacteria in 2017 at 44 – 98% (Table 3).

Maximum MA concentrations during blooms averaged 169, 224 cells/ml at TS5, 6, 11 and 12 together in 2017 and 42,188 cells/ml in 2018. The higher bloom cell concentrations in 2017 suggests that MC concentrations were likely high in 2017 as well. *Microcystis* averaged 37,000 cells/ml in summer 2005, although AFA was the dominant cyanobacterium and total cyanobacteria averaged 206,000 cells/ml (Burgoon, 2006). Moderate MC concentrations > 1 µg/L were related to MA concentrations > 1,300 cells/ml in 9 western Washington lakes (Jacoby et al., 2015). The probability for moderate risk of health effects were indicated for total cyanobacteria concentrations >100,000 cells/ml (WHO, 2003).

Cyanobacteria represented most (74-94%) of the algal biovolume in Parker Horn (TS5), South Lake (TS6) and Rocky Ford Arm (RFA, TS11/12) on average during May-September, 2018, and even more so for the maximum algal biovolumes (91-98%; Table 3). Cyanobacteria averaged 57% of algal biovolume in Parker Horn and South lake during that period in 1977-1988, versus 98% during 1960-1970 prior to dilution with CRW. In 2018, cyanobacteria averaged 87% of total biovolume at those sites. They were less dominant in 2017, averaging 43% of total biovolume. Thus, 2018 was a good summer for cyanobacteria; they were even more dominant on average than during the post-dilution years (1977-1988) and in 2017, despite about the same amount of CRW.

Microcystis was by far the most dominant cyanobacterium in 2017-2018 (average maximum 78%), while *Aphanizomenon flow-aquae* (AFA), which is not a microcystin producer, and the most abundant cyanobacteria in the 1970s-1980s, was comparatively much less represented in 2017 and 2018 (Table 3). Diatoms represented more of average and maximum total biovolume in Parker Horn and South Lake in 2017, while cyanobacteria were much more dominant in RFA (Table 3). Cyanobacteria were slightly more dominant in Parker Horn and South lake than RFA in 2018.

The much higher average and maximum biovolumes of total algae and cyanobacteria observed in Parker Horn and south lake in 2017 than in 2018 may seem surprising, given the much higher average chl concentrations in 2018 than 2017, 18 vs 7 µg/L, respectively. There were only three sampling events in 2017 during July-September, compared to six in 2018 spread from May-September. That could account for some of the lower seasonal average biovolumes in 2018. However, maximums during mid to late summer were also higher in 2017. The contrast was especially evident in RFA. The higher biovolumes in 2017 than in 2018 are surprising because chl averaged higher during May-September in 2018 (18 µg/L) - more than double that in 2017 (7 µg/L). Also, the average chl:biovolume ratio in 1977-1988 was 1.9 (1-4), while the average ratio in 2018 was 4, which was within the range in 1977-1988. That ratio was ten times lower in 2017 at 0.2. Algal cell density was also 4 times greater in 2017 than in 2018. Cell biovolumes are often not closely related to chl, because cell chl can vary with light and nutrient content. In that regard, ratios of chl:TP were much greater in 2018 than in 2017; 0.45 vs 0.28. Thus, part of the reason for higher cell biovolumes in 2017 than 2018, which was the reverse for chl, may have been due to higher cell chl content in 2018. That is also indicated by the same average Secchi disc (SD) transparencies both years, but double the ratio of SD:chl in 2018, versus 2017, indicating roughly double cell chl content in 2018 (Table 1).

Effectiveness of CRW at controlling phosphorus and algae throughout the lake and year-to-year

Columbia River Water (CRW) apparently reached the upper half of Rocky Ford Arm (RFA) in sizeable quantity in 2018, as evidence by specific conductance (SC), which is useful as a conservative tracer for CRW due to its low SC. The SC in Crab Creek (CC) is 491 µS/cm and slightly lower (371 µS/cm) in Rocky Ford Creek (RFC), the two main surface inflows to RFA. Thus, calculated % CRW present in RFA at TS11 and 12 together was only slightly less (10%) using SC in CC as background lake SC rather than SC from RFC as lake background. Using SC in CC as lake background and average SC in upper RFA (TS11/12), the lake was 45% CRW in May, 2018, and 65% in July-September. More CRW was present in Parker Horn and south lake (TS5/6) in May (65%), but CRW present was the same at TS11/12 (65%) in July-September. Also, SC at TS12 was 14% more than at TS11 in May and 8% more in July-September, indicating less CRW reaches as far up RFA to TS12. However, average SC at TS11/12 (266 µS/cm) in July-September never approached the level in RFC (371 µS/cm) due to the influx of CRW, but was close in May (335 µS/cm) before CRW likely reached that far up into RFA. The low SC in RFA was due to CRW diluting the SC in inflows from RFC as well as from ground water. Ground water inputs to RFA averaged 646 µS/cm during summer 2001, according to piezometer data from DOE. More CRW reaching far up into RFA should mean greater dilution of inflow from RFC.

The amount of CRW reaching into RFA was not always proportional to total CRW inflow through CC. Average SC at TS11 was 270 µS/cm during July-September in 2001 when CRW input to CC was large at 284,000 AF. Average SC at TS11 was only slightly higher during those months in 2017 (286 µS/cm) with much less CRW input (93,000 AF), while SC at TS11 was much lower at 256 µS/cm in 2018 with similar CRW input (106,000 AF). Thus, sizable amounts of CRW have reached well up into RFA with even modest amounts of CRW input via CC.

Wind is apparently effective at transporting CRW from Parker Horn well into RFA and diluting TP, as well as SC, entering from RFC and groundwater. However, CRW has less dilution effect in upper RFA, because TP concentrations in water transported into RFA from Parker Horn were 48 µg/L TP during May-September in 2018 and 31 µg/L in 2017. Those TP levels were much higher than contained in CRW (7 µg/L) that dilutes Parker Horn and South Lake. Hence, average TP concentrations at TS11/12 were nearly double those at TS5/6 during 2018 (Table 1). The concentration of algae, indicated by chlorophyll (chl), was also proportionately higher and water transparency lower (Table 1).

Total P and chl concentrations were not always lower despite more CRW. Inputs of CRW in 2017 and 2018 were of similar magnitudes; 93,000 and 106,000 AF, respectively. Yet average May-September TPs at TS5/6 were 25 and 40 µg/L and chl 7 and 18 µg/L, respectively, during 2017 and 2018, while TPs and chl at TS11/12 were 58 and 83 µg/L and 15 and 49, during those years, respectively (Table 1). Also, relatively high TPs occurred at TS6 in 2015 and 2016 (37 and 40 µg/L) with very high CRW inflows of 350,000 and 190,000 AF, respectively. Much of the year-to-year inconsistency in diluting lake TP is due to internal loading of recycled phosphorus from bottom sediments, and those rates of internal loading are enhanced by mixing of the water column by wind, entraining high-phosphorus bottom water into the water column.

Phosphorus Loading

Internal loading of phosphorus from bottom sediments can amount to nearly 50% of the total TP input, or loading, to the lake, as observed during the 1980s, and can vary widely from year-to-year. Thus, lake TP concentrations may be considerably higher one year versus the next whether CRW inputs are sizable or not. How green the lake appears and the chance for toxin-containing cyanobacteria blooms is proportional to TP (in µg/L, or ppb) concentration, which usually increases through the summer as water temperature increases and anoxia in bottom waters develops favoring phosphorus release from bottom sediment.

External loading of TP was larger in 2018 than in 2017 (Table 4). Also, net internal loading was five times greater in 2018 than in 2017 and even greater than the average during 1984-1988 (Table 4). Internal was 47% of total loading in 2018, compared to 14% in 2017. That was partly due to the increase in v-w whole lake TP during summer in 2018 (+24.5 µg/L) being double that in 2017 (13 µg/L). That increase was due to internal recycling of phosphorus from bottom sediments into the water column, because inflow TP concentrations did not increase (Table 3).

Water temperature was not that different between the two years. Surface time-weighted average temperatures at TS5/6 and TS11/12 together were 73.1 and 74.7^0 F during July-September in 2018 and 2017, respectively. Thus, temperature was not a cause for more apparent internal loading and higher TP concentration in 2018 than in 2017.

Total P data from upper Rocky Ford Arm (TS11/12) do not indicate substantial internal loading in that area, at least to the extent that lake TP exceeded inflow TP concentration (Table 2). Lake TP concentrations were relatively similar during the summer with no apparent increase during July-September as was the case in Parker Horn and South Lake (Table 2). Total P concentrations in RFA averaged 64 µg/L during June-September in 2017 and 83 µg/L in 2018, well below the higher and consistent RFC inflow TP concentrations of about 140 μg/L (Table 3). The larger inflow from RFC in 2018, double that in 2017, likely influenced the higher TP in RFA in 2018. There would have been less time for settling inflow TP in 2018, due to less residence time in the lake. Also, three fourths of the high TP entering RFA from RFC is soluble as SRP and readily available to algae (USBR data). That fraction was the same in 2017 and 2018. The source for that high soluble fraction is apparently natural; springs above Trout Lodge hatcheries contained 97 μg/L TP of which 97% was SRP in 2017 and 2018 (USBR data).

Effects of wind and oxygen on internal phosphorus loading

Moses L, being relatively shallow, is more susceptible to internal P loading during summer than deeper lakes. That is because the water column is less resistant to wind mixing, i.e., less difference in density (temp) between surface and bottom. A useful measure of wind effect is the Osgood Index (OI): mean depth (m)/sq. root of area in square km ($OI = m/\sqrt{km^2}$). For the whole Moses Lake area, the OI is 1.06, or 1.12 excluding middle and upper Pelican Horn ($5.78\ m/\sqrt{26.59\ km^2}$). Osgood found in a study of 96 Minnesota lakes that internal loading tended to be high if that index were less than 6-7, and much less if greater than 8 (i.e., strongly thermally stratified). More strongly stratified means that wind has less chance of disturbing stratification, which tends to keep high phosphorus water contained near the bottom.

The other index that relates to internal loading, and other effects of water column stability, is relative thermal resistance to mixing (RTRM): $(D_b - D_s)/D_4 - D_5)$, where D is density determined from temperature, b is density of bottom water, s is surface water density, 4 is density of water at 4^0C and 5 is density at 5^0C.

Internal phosphorus loading was highly inversely correlated with RTRM during 1980-1988; as RTRM decreased, internal loading increased. Internal loading was especially high in 1983, 1985 and 1988 when RTRM was low. Internal loading was 56.7% of the total loading during May-September in 1985 when RTRM was 56 at TS5/6 (see pp. 43-46 in Welch et al. 1989; http://www.mlird.org/lake/MiscProjectResults.pdf).

Compared to the 1980s, the average RTRM at TS6 was 48 during June-September 2018 and 37 at TS5/6 together. That was less than half the average of 113 (range 73-172) at TS6 during that period in 1980-1988. All RTRM values in 2018 were less than the summer average at TS5/6 together during 1980-1988 (85). Temperature, and therefore density, was constant through the water column on August 13, 2018 at TS6, giving an RTRM of zero, meaning there was no resistance to wind mixing and, hence, the entrainment of phosphorus rich bottom water. Average daily wind speed was the highest during the month on August 11 (11.4 mph), just prior to the zero RTRM.

Average RTRM was 37 during 8/13-9/25 in 2018 when TP averaged 47 μg/L, as opposed to a higher RTRM (67) earlier during 6/18-8/6 when TP averaged less at 32 μg/L. Also, the highest TP observed in 2018 was 63 μg/L on 8/27 following the zero RTRM on 8/13. Thus, wind mixing that would easily destratify the water column, either totally (RTRM = zero) or partially (low RTRM), appears to have entrained high TP bottom water accounting for the high surface water TP that likely led to the high chl concentrations in 2018. Temperature profiles were not determined in

2017 when average May-September TP was much lower at 25 μg/L, but wind speeds were also lower. Daily wind speeds averaged higher during July-August in 2018 (7.5 mph) than in 2017 (6.6 mph), and with 24 days over 7 mph in 2018, versus 14 days in 2017.

The internal load is an important source of phosphorus during summer when external loads tend to be low due to low summer stream flows, although Rocky Ford Creek flow has been rather constant. Internal loading averaged 43% of total (external + internal) loading to the whole lake during May-September in 1984-1988, based on direct mass balance calculation. Internal loading to the whole lake was estimated indirectly in 2017 at 14% of total loading, and 47% in 2018 (Table 4). Internal will continue to add phosphorus to the water column even if external sources are reduced, and will probably be greater during windy summers than calm summers.

Internal loading is probably larger in lower Parker Horn and South Lake than in Rocky Ford Arm, because depth is greater allowing more area of the bottom that goes anoxic (anaerobic), which promotes high release rates of phosphorus from bottom sediments. However, other areas of the lake thermally stratify with bottom anoxic water, especially the Cascade area. Internal loading may also occur in shallower water where bottom sediments may go anoxic during periods of calm weather when surface temperature increases during the day and temporary stratification occurs. Phosphorus is also released from oxic (aerobic) sediments, but at a slower rate. Laboratory experiments with Moses Lake sediments in 1984 showed phosphorus release rates ten times higher from anoxic than oxic sediments (Okereke, 1987). The calculated internal loading from those lab release rates, and respective areas of the lake that were oxic and anoxic, was 7,013 kg during May-September. The average internal loading calculated directly from mass balance during 1984-1988 was 6,587 kg. Estimated internal loading in 2017 was 1,806 kg, and 10,836 kg in 2018 (Table 4). Internal loading was negative in 2001.

Thus, internal loading has been variable year-to-year, largely due to variation in wind speed, and the natural shallowness of Moses Lake. Shallow lakes that are deep enough to stratify, allowing bottom water to go anoxic and accumulate large concentrations of phosphorus, are susceptible to wind-caused destratification episodes during summer. Those mixing events increase TP in the water column increasing algal production. Also, results from the 1980s showed that the magnitude of TP increase in the water column due to wind was more than could be accounted for by the accumulated phosphorus in bottom anoxic water. So the effect of wind enhanced mobilization of phosphorus from shallow oxic sediments as well.

Analysis of sediment core data from Pelican Horn in 2017 and 2018

Sediment core phosphorus fractions were determined in 2 cores were taken from lower and middle Pelican Horn in 2017, before the alum treatment, and in 2 cores from the treated area of middle Pelican a in 2018, year after treatment. Concentrations are averages for 0-4 and 4-20 cm sediment depth and average values for each core are separated by a slash in Table 5.

These core results do not show an effect of the alum treatment. A properly dosed alum treatment should inactivate much or nearly all of the mobile-P, which is the fraction composed of iron-P and loosely sorbed-P that are released into the water column as internal loading.

Inactivated mobile-P would shift to Al-P. Not only did the treatment not inactivate any noticeable amount of mobile-P present in the 2017 cores, there was even more mobile-P in the 2018 cores taken at two separate sites in the center of middle Pelican – the area reported and marked as treated. The greater mobile-P in the 2018 cores, above the 2017 levels, was due to 50% of the mobile-P being loosely-sorbed-P, which was not even detected in 2017. Loosely-sorbed-P is even more easily released to overlying water as internal loading than iron-P because anoxia is not required.

Loosely-sorbed-P concentrations in the 2018 cores were exceptionally high. Why so much loosely-sorbed-P is not clear. It's probably not due to analytical error, because IEH laboratory, where both sets of cores were analyzed, also analyzed another set of cores from elsewhere at about the same time in 2018 as the Moses Lake cores and no loosely-sorbed-P was detected in those cores using the same procedure. Moreover, mobile-P fractions, along with other P fractions, added up to the Total-P (TP) in both sets of cores, as they should, and TP concentrations were about the same at 0-4 cm for both years. The large fraction of mobile-P as loosely-sorbed-P in the 2018 cores was associated with equally less calcium-P (Ca-P), which is not shown.

Also, alum treatments usually result in depleted mobile-P content in the top 1-2 cm, but there was no noticeable vertical difference in either mobile-P or Al-P concentrations in the 0-4 cm section of either core. These cores showed no signs of an alum treatment. That is surprising, because an alum floc layer was clearly evident on the sediment surface in a core taken immediately after treatment (Figure 2).

Lack of effect from the alum treatment on TP concentrations in middle Pelican Horn is not surprising given no reduction in sediment mobile-P. There was no real difference in TP concentrations between upper (45 µg/L) and middle (41 µg/L) Pelican Horn in 2017 (Welch, 2017). Average TP concentrations were even higher in upper (57 µg/L) and middle (76 µg/L) Pelican in 2018. The higher TPs were likely due to more internal loading through wind mixing in 2018 as observed in Parker Horn and south lake. Inputs of phosphorus from ground water and carp bioturbation were not likely that different between the two years.

There could be several explanations for no evidence of the treatment and the high loosely-sorbed-P concentrations.

1. The dose of about 17 mg/L Al was too low to affect the high mobile-P concentration. That dose was appropriate using mobile-P levels in the 2017 cores, taken before treatment, but sediment density in the top 10 cm in the 2018 cores, taken in the treated area, was more than double the density in the 2017 cores, causing mobile-P to be 4 times higher in the 2018 cores. Thus, alum dose should have been 75 mg/L Al or over 4 times higher to inactivate that level of mobile-P.
2. Bioturbation by the apparently large carp population may have mixed the added alum through possibly the top 20-25 cm or so, rendering the effect of the low alum dose's aluminum content to be undetectable, above the within and between core variability.
3. The high loosely-sorbed-P concentrations, responsible for the higher mobile-P fraction, may be confined to the deeper area of middle Pelican, the area that was treated, but

not cored in 2017. That is supported by the relative similarity of mobile-P (and loosely-sorbed-P) in the two 2018 cores, taken well apart in the treated area, compared with the 2017 pretreatment cores taken outside the treated area in middle and lower Pelican – north and south of I90. Those pre-treatment cores taken in 2017 had similar TP concentrations as the 2018 cores, but with much lower mobile-P. Why loosely-sorbed-P was so high in the 2018 cores may be due to the residual from decades of wastewater input.

4. Wind may have redistributed the alum floc, aided by bioturbation by carp resuspending the surficial sediment/alum floc mixture. Also, while alum floc settles rather quickly, the shallowness of Pelican Horn – 1.1 m mean depth – greatly increased the potential for wind-caused resuspention and redistribution of the floc.
5. Some combination of the above.

Summary and Conclusions

1. Lake water quality was poorer in 2018 than in 2017, because TP concentration in Parker Horn and south lake was greater. Average May-September algal biomass, as chlorophyll (chl), was 18 vs 7 µg/L and average TP was 40 vs 25 µg/L. respectively, in 2018 and 2017. The average concentrations of chl and TP in 2018 were similar to those in 1986-1988; 17 and 41 µg/L. July-September levels of chl and TP were even higher at 25 and 52 µg/L for chl and TP in 2018, but July-September levels were not higher in 2017 at 30 and 6 µg/L for TP and chl, respectively. The higher mid-to-late summer increase in TP and chl was due to greater input from internal sources of phosphorus, most likely recycling from lake-bed sediment.
2. Internal sources were probably the cause for increased mid-late-summer TP concentration, because inflow TP concentrations from Crab Creek (TS3) did not change during the summer, while TP in Parker Horn and south lake more than doubled. That indicates a net yield from internal sources, over losses from the settling of incoming TP. There was no net increase in late summer TP in 2017, nor in Rocky Ford Arm (RFA) in either 2018 or 2017. Physical-chemical processes at the sediment-water interface were the most likely source for late-summer added TP. Other possible sources, such as ground water, carp excretion/bioturbation and rooted macrophyte decay probably did not change markedly between 2017 and 2018.
3. As a result of higher mid-to-late summer TP in 2018, blooms of blue-green algae (cyanobacteria) were more noticeable than in 2017. Cyanobacteria averaged 74-98% of total algal biovolume during spring-summer in Parker Horn and south lake during 2018, even greater than in 1977-1988 (57%). The algal toxin microcystin (MC) was detected in nearshore algal mats in RFA and south lake at concentrations well above the state recreational guideline of 6 µg/L. The blue-green *Microcystis* was the most likely source of MC, since that taxon was dominant in the lake in 2018 at an average of 78% of maximum biovolume in Parker Horn, south lake and RFA. Blue-green biovolume was

actually greater in 2017 than in 2018, despite lower chl concentration in 2017. Possible reasons for that inconsistency are discussed earlier.

4. Columbia River water (CRW) that enters the lake through Rocky Coulee Wasteway (RCW) and CC reached well up into RFA during spring-summer 2018. Specific conductance (SC), which is a good indicator of low-SC CRW was less in upper RFA (TS11/12) than in RFC, 266 vs 371 µS/cm, during July-September. The fraction of CRW in that area was 65%, the same as existed in Parker Horn and south lake (TS5/6). Thus, CRW diluted TP coming from RFC and ground water, which has SC at 646 µS/cm. However, the input of CRW into RFA was not always proportional to input through RCW and CC, because transport up into RFA depends on wind. Also, while the SC tracer a relatively high fraction of CRW in upper RFA, the TP content in transported water was higher, because it came from Parker Horn with 48 µg/L in 2018 and 31 µg/L in 2017 – much higher that the 7 µg/L in CRW that dilutes Parker Horn and south lake. Thus, CRW entering from CC and RCW is much less effective at diluting RFA, with its high TP concentration, than diluting Parker Horn and south lake.
5. The higher TP concentrations in Parker Horn and south lake, as well as in RFA, in 2018 than in 2017, by 60 and 30%, respectively, were due to greater external and internal loading in 2018. While estimated external loading was 33% greater in 2018, internal was 6-fold greater and averaged 47% of total loading, similar to the average fraction during 1984-1988. Most of the increased external loading in 2018 was due to a doubling of flow in RFC; 120 cfs in 2018 vs 60 cfs in 2017. Increased internal load in 2018 may be related to more windy conditions affecting water column stability and entrainment of high TP anoxic bottom water.

   Laboratory study of sediment phosphorus (P) recycling in 1984 showed that P release from sediment overlain with anoxic (no oxygen) water was ten times greater than overlain with oxic (oxygenated) water. Calculated internal loading from respective areas in the lake in which sediment was overlain with anoxic or oxic water amounted to a total internal loading of about 7,000 kg during July-August. That was within about 500 kg of the average internal loading calculated directly from mass balance during 1984-1988.
6. Moses Lake is susceptible to wind-caused turbulence, due to its large, relatively shallow area, resulting in periodic entrainment of high-P bottom water. Also, its maximum depth of 11.5 m in the south lake area is sufficient to thermally stratify allowing anoxic, high-P water to accumulate over the bottom sediment. Strong winds can temporarily destratify the water column as was the case in August 2018 when temperature (and density) was uniform top to bottom. A useful indicator of water column stability is RTRM (relative resistance to mixing). RTRM was highly inversely correlated with internal loading during 1980-1988, with especially low RTRMs associated with high surface water TP concentrations. Also, the effect of wind on water column TP was found to be greater than simple entrainment of high-P bottom water alone. The higher internal loading in 2018 than 2017 was associated with higher daily average wind speed – 7.5 vs 6.6 mph during July-August, with 24 days over a daily average 7 mph in 2018 vs 14 days in 2017. Daily average wind speed was highest in August 2018 at 11.4 mph just prior to the destratified water column and zero RTRM and subsequent highest TP of 63 µg/L.

7. Sediment core data show that the alum treatment in May 2017 was underdosed by about four-fold. Cores taken in the central treated area of middle Pelican Horn showed no evidence of reduced mobile-P or increased aluminum concentrations in the top 0-4 cm sediment layer. That was unexpected, because alum, if properly dosed, typically depletes the mobile-P fraction, which is the source for internal loading. However, the 2018 cores revealed much higher mobile-P fraction concentrations in the treated area – about four times the mobile-P in pretreatment cores taken outside the treated area. The higher mobile-P in treated area cores was due to greater sediment density and very high loosely sorbed P that may be a legacy from decades of wastewater discharge to middle Pelican Horn. Other factors, such as sediment bioturbation and excretion by carp and ground water TP input probably contributed to the lack of alum effect. See Welch (2017) for analysis of ground water and pumping effects in Pelican Horn.

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 regularly 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 annually 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. Cyanobacteria (blue-green algae) begin to become dominant at about 30 µg/L average total phosphorus (TP) concentration during summer (Downing et al., 2001). That is also the border between eutrophy and mesotrophy. Thus, that concentration should be the goal for Moses L. The goal was 50 µg/L after the dilution project started, which was met in 1986-1988 (41 µg/L) and also DOE's tmdl goal. However, 50 µg/L is too high to avoid toxic blooms. Average summer TP in lower Parker Horn and south lake during 2017 and 2018 were 25 and 40 µg/L, respectively. Average TP concentration in upper Rocky Ford Arm was 54 and 83 µg/L during summers 2017 and 2018, respectively.
3. Propose constructing facilities to transport another 50 10 ^{6} m^3 (40,000 AF) of CRW to the main (Rocky Ford) arm from TS3 to the Connelly Park area (TS11) to dilute the high TP (140-160 µg/L) entering from Rocky Ford Creek. That should roughly reduce TP in RFA to about 30-40 µg/L from the current 54-83 µg/L observed in 2017 and 2018. The projected TP concentrations were estimated from $TP_{lake} = TP_{in}/(1 + T^{0.5})$, where TP_{lake} is the resulting lake concentration, TP_{in} is the v-w inflow concentration of RFC and CRW combined, and T is the water residence time in years (Brett and Benjamin, 2008). If delivered over 5 months, the flow would be about 3.7 m^3/sec or 130 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 design 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. An alternative for CRW to upper RFA could include a siphon from the W-20 canal about 2 miles west of the head of RFA.
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. Estimated internal loading in 2018 was over 10,000 kg and 40% of total loading. To further consider an alum treatment to this area, DO/temperature profiles should be determined at TS 5, 6, and 7 at 1 m intervals from surface to bottom during 2019, 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.

5. The area of anoxic sediment in lower Parker Horn, Cascade and south lake is roughly the area of the hypolimnion, which is estimated at about 1,260 acres or about 18% of the total lake area. Laboratory study of phosphorus release from anoxic sediment in 1984 showed that area could contribute about 3,000 kg of internal loading to the lake during the summer (Okereke, 1987). That estimated TP load was about 1/3 of the average internal load to the whole lake determined during 1984-1988, more than the internal load in 2017 and about 30% of the estimated internal load in 2018. Internal load is an important source of phosphorus causing summer blooms of cyanobacteria and was probably the cause for higher TP and chl in Parker Horn and south lake during 2018 than in 2017. A properly dosed alum treatment should inactivate about 80% of that internal load and the treatment should last on order of ten years. Alum is the proven most cost-effective way to reduce internal loading (Huser et al., 2016). However, low-phosphorus CRW also decreases the effect of internal loading because it dilutes the internal as well as the external load.

References Cited

Brett, M.T. and M. Benjamin. 2008. A reassessment of lake phosphorus retention and the nutrient loading concept in limnology. Freshwater Biol. 53:194-211.

Burgoon, P.S. 2006. 2005 summary of water sampling in Moses Lake. WQ Engineering, Wenatchee, WA.

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

Carroll, J. 2006. Moses Lake phosphorus-response model and recommendations to reduce phosphorus loading. Washington Dept. of Ecology, Olympia, WA. Pub. No. 06-03-011.

Cooke GD, Welch EB, Peterson SA, Nichols SA, 2005. Restoration of Lakes and Reservoirs. CRC Press, Boca Raton, FL, 3rd ed.

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

Eaton AD, Clesceri LS, Rice EW, Greenberg AE, Franson MAH. 2005. Standard methods for the examination of water and wastewater. 21st ed. American Public Health Association, Water Environment Federation and American Water Works Association.

Jacoby, J., M. Burghdoff, G. Williams, L. Read and J. Hardy. 2015. Dominant factors associated with microcystins in nine midlatitude, maritime lakes. Inland Waters 5:187-201.

Mathews, R.A., J. Pickens and E. Lawrence. 2019. Moses Lake algae monitoring project; 2018 final report. Inst. Watershed Studies, Huxley College of the Environment, Western Washington University.93 pp.

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

Okereke, V.O. 1987. Internal phosphorus loading and water quality projections in Moses Lake. MSE Thesis, Civil and Environmental Engineering, University of Washington, Seattle, WA.

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

Pitz, C.F. 2003. Moses Lake total maximum daily load groundwater study. WA State Dept. Ecology, Olympia, WA. Pub. No. 03-03-005.

Welch EB, CR Patmont. 1980. Lake restoration by dilution: Moses Lake, Washington. Water Res. 14:1317-1325.

Welch, E.B., C.A. Jones and R.P. Barbiero. 1989. Moses Lake quality: results of dilution, sewage diversion and BMPs – 1977 through 1988. Water Res. Ser. Tech. Rep. 118, Dept Civil & Environ. Eng., Univ. of Washington, Seattle, WA, 65 pp. http://mlird.org/cleanlakeproject.html

Welch, E.B., R.P. Barbiero, D. Bouchard and C.A. Jones. 1992. Lake trophic state change and constant algal composition following dilution and diversion. Ecol. Eng. 1:173-197.

Welch, E.B. 2009. Should nitrogen be reduced to manage eutrophication if it is growth limiting? Evidence from Moses Lake. Lake Reserv. Manage. 25:401-409.

Welch, E.B., S. Brattebo and H. Gibbons. 2015. A dramatic recovery of Lake Spokane water quality following wastewater phosphorus reduction. Lake Reserv. Manage. 31:57-165.

Welch. E.B. 2017. Moses Lake quality: Causes and Benefits of Columbia River Water. Report to the Moses Lake irrigation and Rehabilitation District. 26 pp. http://mlird.org/lake/mosesreport2017-3.pdf

Table 1. Average TP and chlorophyll in µg/L and Secchi transparency (SD) in meters in Parker Horn and South Lake (TS5/6) and upper Rocky Ford Arm (TS11/12) during 2018. Average values in 2017 are in (). Transparency due to chl calculated from equation in Carlson (1977).

	May-September		July-September	
	TS5/6	TS11/12	TS5/6	TS11/12
TP	40 (25)	83	51	91
Chl	18 (7)	49	24	54
SD	1.4 (1.4)	0.99	0.77	0.58
SD per Chl	1.07 (2.1)	0.55	0.89	0.51

Table 2. Average and maximum blue green algae (cyanobacteria) biovolumes (mm^3/L), percent () blue greens of total and average algal biovolume , and percent MA and AFA of maximum blue green biovolume during 2017 (July-September) and 2018 (May-September). MA = *Microcystis aeruginosa* and AFA = *Aphanizomenon flos-aquae*.

	2017		2018	
	TS5	TS6	TS5	TS6
Ave. blue greens (%)	16.5 (47)	10.8 (39)	5.6 (94)	1.9 (80)
Max. blue greens (%)	24.8 (61)	13.4 (22)	12.8 (94)	4.4 (91)
% MA of max. BGs	87	77	60	88
% AFA of max. BGs	0	0	7	6
	TS11	TS12	TS11	TS12
Ave. blue greens (%)	58.0 (82)	27.2 (67)	4.8 (83)	9.8 (74)
Max. blue greens (%)	144 (98)	39.2 (93)	9.8 (97)	22.6 (98)
% MA of max. BGs	48	94	71	98
% AFA of max. BGs	0.3	1.8	10	1

Table 3. Inflow and in-lake TP concentrations for Lower Parker Horn and South Lake (TS5/6) and Rocky Ford Arm (TS11/12) during 2017 and 2018. Difference between inflow and lake TP indicated in ().

							means
2018	April	May	June	July	August	September	July-September
TS/3 CC	17	37	39	39	39	34	37
TS5/6	21	15	38	53	48	53	51 (+14)
RFC hwy 17	135		166	117		145	143
TS11/12	120	60	82	106	90	78	91 (-52)
2017							
TS3		40	48	45	35	36	38
TS5/6		16	29	28	22	40	30 (-8)
RFC hwy 17	185		141	128		143	137
TS11/12		13	68	62	67	59	63 (- 74)

Table 4. 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	Net Internal	% of total
1984-1988	4,538	4,167	3,886	12,591	9,346	43
2001	2,166	2,809[1]	1,720	6,695	- 643	no net
2017	5,294	3,368[1]	2,735[2]	11,397	1,806[3]	14
2018	6,988	6,393[1]	4,225[2]	15,101	10,836[3]	47

[1]includes Trout Lodge Hatchery; [2]24 % of total TP load as in 2001 (Carroll, 2006, table 14); [3]estimated from mass balance using inflow volume X 1.5 (average surface outflow/inflow, 1984-1988) X v-w lake TP for output TP mass. CC = Crab Creek; ELC = East Low Canal; RCW = Rocky Coulee Wasteway; RFC = Rocky Ford Creek; GW = ground water.

Table 5. Phosphorus fractions in cores from Pelican Horn to determine the effect of the alum treatment in middle Pelican Horn in May, 2017. Cores were from middle and lower Pelican Horn in 2017, before treatment, and in the treated area of middle Pelican Horn in 2018 after treatment. Average concentrations of P fractions in mg/g of dry sediment are shown for each depth interval and each of the two cores separated by a /.

2017	TP mg/g	mobile P µg/g	Al-P µg/g	Al mg/g
0-4 cm	0.67/0.83	38/31	65/46	8.0/9.1
4-20 cm	0.76/0.77	37/29	51.41	8.1/7.4
2018				
0-4 cm	0.66/0.79	81/177	57/49	9.4/8.4
4-20 cm	0.72/0.67	59/88	42/32	9.4/8.6

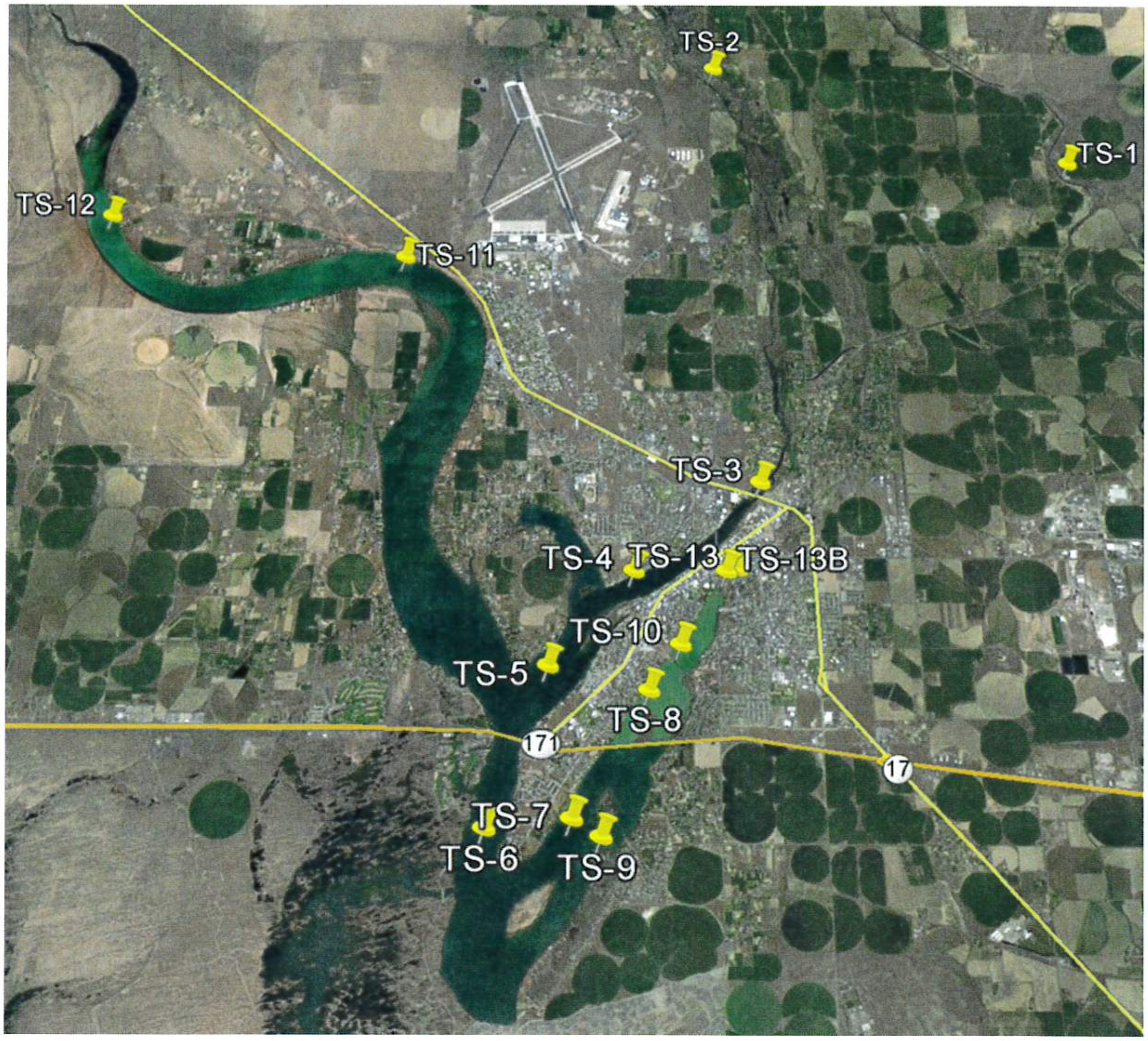

Figure 1. Sampling sites during 2017, most similar to those during 1969-1970 and 1977-1988 (Welch et al., 1989).

Figure 2. Sediment core taken immediately after the May 2017 alum treatment in middle Pelican Horn. Alum floc shown at the surface.