Evaluation of optimal dose and mixing regime for alum treatment of Matthiesen Creek inflow to Jameson Lake, Washington

Jillian J. Churchill and Marc W. Beutel

Washington State University, Department of Civil and Environmental Engineering, Pullman, Washington 99164

Peter S. Burgoon

Water Quality Engineering, Wenatchee, Washington 98801

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Abstract

An innovative method of reducing external phosphorus (P) loading to lakes uses engineered systems to treat lake inflows with aluminum sulfate (alum). In this study we used a series of jar tests to examine the optimal alum dose and mixing regime to remove P from Matthiesen Creek, an important external source of P to Jameson Lake. Matthiesen Creek is a good candidate for alum treatment because the creek runs year round, and the majority of P in the spring-feed creek is in the form of bioavailable dissolved P that can be efficiently captured in alum floc. The mixing regimes in this study mimicked a range of possible treatment scenarios that relied on natural turbulence in the creek or conventional mechanical mixing, and presumed the discharge of alum floc either directly to the lake or to an on-shore settling basin. Jar tests showed that an alum dose of 5 mg-Al/L was sufficient to decrease P from around 0.13 mg-P/L to below 0.02 mg-P/L for most mixing regimes. For all mixing regimes, doses of up to 20 mg-Al/L did not depress pH below the recommended minimum pH of 6. Flash mixing prior to low-intensity mixing did not enhance P removal over low-intensity mixing alone, but flash mixing alone resulted in lower levels of P removal from creek water. Jar testing with a mixture of alum-treated creek water and lake water showed that lake waters tended to inhibit P uptake by alum floc. This, combined with the fact that high pH favors the formation of the aluminate ion which could exhibit chronic toxicity to aquatic biota, suggests that discharge of alum solids directly to the lake should be avoided. We recommend an

engineered inflow treatment system on Matthiesen Creek that maintains an alum dose of 5–10 mg-Al/L under moderate mixing conditions (Gt of 1,000–3,000) with alum floc collected in an on-shore settling basin.

Key words: aluminum sulfate, alum treatment, jar test, pH, phosphorus

Alum ($Al_2(SO_4)_3 \cdot nH_2O$) is a common water quality amendment used in drinking water and wastewater treatment to enhance particulate and phosphorus (P) removal by adsorption and flocculation (Viessman and Hammer 1985). Since the early 1970s, alum has been used in lakes to prevent internal P loading by blanketing sediments with a layer of aluminum hydroxide floc that readily binds with dissolved P released from anaerobic sediments (Welch and Cooke 1999). Care must be taken regarding alum dosing, particularly in soft-water lakes, because hydrolysis of aluminum ions produces hydrogen ions, and waters with relatively low alkalinity may experience a drop in pH and resulting toxicity due to elevated concentrations of dissolved aluminum (Kennedy and Cooke 1982). Lake alum treatments are most effective in lakes in which internal nutrient loading is the dominant source that drives phytoplankton productivity (Cooke et al. 2005). Beginning in late 1980s, lake managers began to use alum as a means to decrease external P loading by treating lake inflows with alum (Harper et al. 1983, Cooke and Carlson 1986, Harper and Herr 1992). More recently, researchers have examined the efficacy of removing P from tributary inflow through the combined addition of alum and/or chemical coagulants commonly used in water treatment to enhance flocculation (Mason et al. 2005, Trejo-Gaytan et al. 2006). A recent comprehensive study of P removal by alum showed that maximum removal of soluble P is achieved at pH of 5–6, that P removal is not influenced by a wide range of salts, that temperature has little effect on alum efficiency, and that floc aging decreases the sorption capacity of alum floc (Georgantas and Grigoropoulou 2007).

In contrast to whole-lake alum treatments, treating lake inflows with alum encompasses a number of unique challenges. Whole-lake treatments are commonly based on the alkalinity of lake water and/or the content of P in lake sediments (Cooke et al. 2005, Pilgrim et al. 2007). However, there is no such clear-cut dosing guidance for lotic systems, and the success of alum treatments in such systems is sensitive to dose and mixing. For example, too low an alum dose can result in the formation of pin floc that does not settle and that contains P that is still bioavailable to phytoplankton (Pilgrim 2002). Too high an alum dose can increase operating costs and result in the over-production of solids. An appropriate level of mixing is required between the alum and the inflow to promote the formation of a floc that both retains P and settles out of the water column (Viessman and Hammer 1985). Too little mixing can poorly disperse the alum throughout the inflow resulting in poor floc formation, while too much mixing can sheer the floc apart and inhibit settling. Another important difference is that unlike a one-time whole-lake treatment, inflow treatment needs to be done on a semi-continuous and seasonal basis during times of high external P loading. Finally, when treating lake inflow with alum, solids may need to be collected in a settling basin prior to discharge of the alum-treated water to the lake. Over the long term, these solids will need to be permanently disposed of.

This study, which examined the potential to remove P from Matthiesen Creek inflow to Jameson Lake using alum, was initiated as part of a larger effort by the Foster Creek Conservation District and the Washington State Department of Ecology to examine water quality management options to control external and internal nutrient loading to Jameson Lake. This effort was spurred by a dramatic decrease in water quality in summer 2004, including a massive lake-wide bloom of the cyanobacteria *Oscillatoria rubescens*, and a subsequent fish kill due to low dissolved oxygen levels (WQE 2006). We used jar testing to: (1) determine the minimum alum dose to remove P inflow from Matthiesen Creek, (2) evaluate the effectiveness of a range of mixing regimes that mimicked the use of natural in-stream turbulence and more conventional mechanical mixing, and (3) evaluate the effect of discharging solids directly into the alkaline lake on P retention and settling of the aluminum hydroxide floc.

Methods

Study site

Jameson Lake is a eutrophic, semi-terminal alkaline lake located within an arid agricultural watershed in central Washington (Fig. 1). The lake does not discharge until its surface elevation reaches a number of outlet structures at elevations between 544.4 m and 547.4 m. In 2004 the lake had a surface elevation of 548.8 m, a maximum depth of 27.4 m, a mean depth of 10.4 m, and a volume of 2.22×10^6 m^3 (18,000 ac-ft). Jameson Lake supports a stocked trout fishery that is important to the economic development of rural Douglas County. The lake has two main inflows: Upper McCartney Creek flows into the north end of the lake from Bennett Lake, an alkaline lake upstream of Jameson Lake; Matthiesen Creek discharges into the northeast side of the lake and is fed by a natural spring roughly 1 km upstream. An ongoing water quality monitoring effort has been underway at the lake since 2002 (WQE 2006). Levels of pH in the lake range from 8 to 9, alkalinity is around 400 mg/L, Secchi disk depth ranges from 0.3 to 1.4 m, and thermal stratification and hypolimnetic hypoxia occurs from approximately June through

October. Matthiesen Creek discharge remains fairly stable throughout the year, typically ranging from 1,000 to 2,000 m^3/d (0.4–0.8 cfs), and the creek accounts for 25–50% of the annual inflow to the lake. Levels of pH in the creek are around 8.0 and alkalinity is around 170 mg/L. Total P (TP) levels in creek water are 0.10–0.3 mg-P/L, and the creek typically accounts for 50–100% of the external P load to the lake. The major source of P to the spring that feeds Matthiesen Creek is likely historical and ongoing livestock activities within its watershed. Total annual external P loading to the lake is on the order of 125 kg-P.

Internal loading likely plays a role in driving productivity in Jameson Lake. Related chamber studies performed in parallel with this alum study showed that profundal lake sediments released P at rates of 10–15 mg/m^2/d under anaerobic conditions, and that maintenance of a well-oxygenated sediment-water interface impeded sediment P release (Beutel 2007). These P release rates were typical of those in sediments from eutrophic and hyper-eutrophic lakes (Nurnberg 1988, Auer et al. 1993). Assuming a three-month anaerobic period and a profundal sediment surface area of 1 million m^2 (~250 ac), internal P loading in Jameson lake is on the order of 1,200 kg, which exceeds external loading by an order of magnitude. However, the lake has limited potential to mix internally loaded P into the photic zone during the summer and fall because of its small fetch relative to its depth. Jameson Lake has an Osgood index of mixing (OI), the ratio of mean depth (m) to the square-root of the surface area (km^2), of 22 (Osgood 1988). This value far exceeds the values of OI (<6–7) indicative of high potential for summer and fall mixing and entrainment of internally loaded nutrients into surface waters.

Mixing regimes

Alum jar tests were conducted using a Phipps and Bird PB-700™ JARTESTER under four mixing regimes that mimicked potential treatment scenarios at the site based on natural turbulence or mechanical mixing (Table 1). Mixing intensity was quantified using the parameters G (1/s), the velocity gradient, and Gt, the unitless Camp number. Both parameters are widely used in water and wastewater treatment to quantify the flocculation processes, and Gt values of 1,000–10,000 have been shown to provide optimal flocculation (Viessman and Hammer 1985). The value of G quantifies the amount of turbulent mixing and sheer stress in a system, which in turn greatly affects floc formation, and the ability of the floc to subsequently settle. The value of Gt, the product of G and the duration of mixing, quantifies the combined effects of mixing intensity and mixing duration.

The two mixing regimes of low mixing intensity mimicked direct alum injection into the creek and natural turbulence as the creek water flowed to the lake. The first mixing regime consisted of low mixing intensity of relatively short duration (Low-Short), while the second consisted of low mixing intensity of relatively long duration (Low-Long). The two low intensity mixing regimes had G values of 35/s. This in-stream G value was estimated as follows. First the headloss per unit length of creek was estimated for Matthiesen Creek based on the Manning equation for uniform flow (Lindeburg 2003, equation 19.30a):

$$\frac{H}{L} = \frac{n^2 v^2}{R^{4/3}} \qquad (1)$$

where H is head loss (m), L is creek length (m), n is the Manning roughness coefficient, v is water velocity (m/s), and R is hydraulic radius (m). Assuming a n of 0.040 (gravel bed with large boulders; Robertson et al. 1988), and using typical values of velocity (0.1 m/s) and hydraulic radius (0.04 m) for Matthiesen Creek, headloss per unit length was 0.001. The value of G was then estimated based on:

$$G = \sqrt{\frac{Q\rho gH}{\mu \forall}} = \sqrt{\frac{\rho gH}{\mu T}} = \sqrt{\frac{\rho gv}{\mu} \cdot \frac{H}{L}} \qquad (2)$$

where Q is flow rate (m^3/s), ρ is water density (1,000 kg/m^3), g is gravitational acceleration (9.8 m/s^2), and T is travel time over a given length (s). For a v of 0.1 m/s and an H/L of 0.001, G is 35/s. The Low-Short (mixing time of 30 s) and Low-Long (mixing time of 90 s) mixing regimes had Gt values of around 1,000 and 3,200 (Table 1). At a velocity of 0.1 m/s, the site of in-stream alum injection would be around 3 and 9 m, respectively, upstream of where the creek discharges to the lake for the two mixing regimes.

Based on equations (1) and (2), G in creeks (assuming uniform flow) is a complex function of Manning roughness coefficient, velocity and hydraulic radius. The value of G is sensitive to changes in v, though this sensitivity is somewhat offset by R which is expected to covary with v. In Matthiesen Creek, a quadrupling of both v and R results in a G around 100/s. In contrast, G is less sensitive to n because n typically varies from 0.025 (straight gravel bed) to 0.070 (large rocks and boulders). Increasing the assumed n for Matthiesen Creek from 0.040 to 0.070 yields a G of around 60/s. Values of G in larger creeks may be smaller because of higher R values. For example, for a creek with an R of

1.2 m, a *v* of 0.3 m/s (1 ft/s), and a *n* of 0.73 (e.g., Boundary Creek, Idaho; Robertson et al. 1988), G is 15/s. A doubling of the velocity to 0.6 m/s yields a G of around 40/s.

The two mixing regimes of high mixing intensity mimicked the use of a mechanical mixing system such as an impeller. The first mixing regime, which mimicked flash mixing of alum discharged with creek water into a settling basin or flash mixing of alum addition directly to the creek just upstream of the lake, consisted of a high mixing intensity of relatively short duration (High-Short). The second mixing regime, which mimicked flash mixing of alum added to the creek with subsequent weak mixing in the creek as it flowed to Jameson Lake, consisted of high mixing of relatively short duration followed by slow mixing of relatively long duration (Composite). This mixing regime also mimicked flash mixing of alum discharged with creek water into a settling basin with subsequent weak mixing in the basin. Flash mixing followed by weaker mixing is a typical mixing scenario used in water treatment to enhance flocculation (Viessman and Hammer 1985). The two high intensity mixing regimes were mixed at the highest mixing rate possible on the jar tester (250 rpm). This translated to a G of 350/s, a value typical of rapid mixing processes in wastewater treatment plants (Tchobanoglous and Burton 1991). The Gt values ranged from around 10,000 to 14,000 for the two high intensity mixing regimes.

Based on the Phipps and Bird rating curve of G versus mixing paddle revolutions per minute (rpm) for a standard 2-L jar test (Water and Air Research [date unknown]), G values for our 1-L jar test were scaled up based on the equation:

$$G = \sqrt{\frac{P}{\mu \forall}} \qquad (3)$$

where P is the power input (N·m/s), μ is the viscosity of the water (0.001 N·s/m^2 or kg/m·s), and $\forall$ is the volume of water (m^3). Using equation (3), the ratio of G at a volume of 1 L and 2 L can be estimated as the square root of the inverse ratio of the two volumes. Thus, for our 1-L jar tests, the manufacturer's G values were scaled up by a factor of 1.41.

Jar testing

Water samples were collected in June 2007 from Matthiesen Creek and the surface of Jameson Lake (Fig. 1). The samples were refrigerated at 11°C after collection and allowed to equilibrate to room temperature (20 °C) prior to the start of the jar testing experiments. Triplicate beakers were each filled with 1 L of creek water from a bulk storage container. The container was constantly swirled when pouring to ensure that beakers were filled with a representative sample. Beaker water was initially tested for pH using a Hach HQ40d pH meter calibrated daily with pH 7 and 10 standards. Initial water samples were collected for soluble reactive P (SRP) and TP. Beakers were loaded onto the jar tester and the target alum dose was added. We made a standard alum stock solution following the method of Cooke et al. (2005) by dissolving 1.46 g of aluminum sulfate octadecahydrate in 100 ml of water. From that stock solution, alum doses of 0, 2, 5, 10, and 20 mg-Al/L were made by diluting 1.6, 4, 8, and 16 ml, respectively, of the stock into the 1-L test jars. The jars were then mixed under the desired mixing regime.

Once mixing stopped, alum-treated creek water was allowed to settle under two settling scenarios. To mimic discharge of alum-treated creek water to the lake and subsequent settling in the lake water column, 100 ml of unsettled alum-treated creek water was gently poured into a beaker with 400 ml of lake water, and the mixture was allowed to settle for 6 hours. The dilution factor of 1:5 is typical for small creeks mixing into shallow surface waters (Fischer 1979). For the second settling scenario, 500 ml of unsettled alum-treated creek water was gently poured into an empty beaker and allowed to settle for 6 hours. This simulated settling in an on-shore settling basin. After settling, beaker water was tested for pH, and water samples were collected for SRP and TP analyses.

Standard preservation, analytical, and quality assurance procedures were used to measure TP and SRP (APHA 1998). The SRP samples were preserved by filtering through a prewashed 0.45-μm filter and freezing. The TP samples were preserved by adding sulfuric acid and storing at 11 °C. The SRP analyses were conducted on a Lachat QuikChem 8500 Flow Injector Analyzer (Method 10-115-01-1-P) based on the ascorbic acid colorimetric method, and TP was analyzed using Hach TNT 843 tubes and a Hach DR2800 spectrometer. The TP samples first underwent a permanganate digestion for one hour at 100 ^{o}C. Phosphorus levels were then estimated colorimetrically based on the ascorbic acid method. Approximate detection limits for SRP and TP, estimated as three times the standard deviation of the blanks, were 0.02 mg-P/L. Nondetect samples were reported as one-half the detection limit.

Results

Alum-treated creek water samples

The pH progressively dropped from around 8.2 to 6.5 with increasing alum dose under all mixing regimes (Fig. 2). There was little difference in pH response to alum dose in the four mixing regimes. All mixing regimes showed the same overall trend, with SRP and TP dropping as alum dose increased (Fig. 2). Initial TP levels in creek water were around 0.13 mg-P/L. Settling alone (no alum addition) resulted in a minor drop in TP to 0.8–0.11 mg-P/L, with SRP accounting for all of the remaining P. In the Low-Short, Low-Long, and Composite mixing regimes, TP and SRP were removed to below detection at alum doses >5 mg-Al/L. In contrast, in the High-Short mixing regime, TP and SRP never dropped below 0.04 mg-P/L. In all mixing regimes, at alum doses ≤2 mg-Al/L, SRP decreased substantially while TP dropped little.

Combined samples of alum-treated creek and lake water

Because of the alkaline nature of Jameson Lake (pH ~9.0, alkalinity ~400 mg/L), the initial pH of the combined samples (100 ml alum-treated creek water and 400 ml of lake water) was 8.8–9.0. The pH progressively decreased with increasing alum dose under all mixing regimes, but pH levels even at the highest alum dose of 20 mg-Al/L were all above 8.0 (Fig. 3). Initial TP levels in the combined samples were around 0.05 mg-P/L, with lake water accounting for half the P and alum-treated creek water accounting for the other half. The four mixing regimes showed somewhat different patterns in P removal. For both low mixing intensity regimes, SRP was removed to below detection above an alum dose of 10 mg-Al/L. In contrast, SRP was removed to below detection above an alum dose of 5 mg-Al/L for the High-Short mixing regime and above 2 mg-Al/L for the Composite mixing regime. The non-SRP component of TP was not substantially affected except in the Composite mixing regime, where levels dropped to around 0.01 mg-P/L at alum doses of 5 and 10 mg-Al/L.

Discussion

The jar tests with creek water showed that an alum dose of 5 mg-Al/L was sufficient to remove SRP from around 0.13 mg-P/L to below our detection limit of 0.02 mg-P/L, approximately a 90% reduction, under three of the four mixing regimes (excluding High-Short). Levels of pH at an alum dose of 5 mg-Al/L were around 7.2, well above the recommended lower pH limit of 6 for alum treatments (Pilgrim and Brezonik 2005b). Even quadrupling the dose to 20 mg-Al/L resulted in acceptable pH levels in all mixing regimes. Jar test results show that incorporating a flash-mixing phase prior to a low-intensity mixing phase resulted in no major enhancement of P removal, but that flash mixing alone was not sufficient to remove P from creek water. The poor P removal under the High-Short flash-mixing regime was likely due to a lack of a subsequent low-intensity mixing phase that would have promoted flocculation. Lower levels of P removal were also observed by Trejo-Gaytan et al. (2006) when they decreased the amount of low-intensity mixing in jar tests treating tributary inflow to Lake Tahoe with prehydrolized aluminum coagulants.

A few studies have found similar results, though few examined the range of mixing scenarios described in this study. Inflows to several small urban Florida lakes were treated with alum in an attempt to reduce external P loading from urban runoff (Harper and Herr 1992). At Lake Ella, Florida, lake managers used a novel system that injected liquid alum into stormwater pipes, with the necessary mixing occurring in the pipes and settling occurring in the lakes. At an alum dose of 5 mg-Al/L, 90% of the incoming TP and SRP were removed from lake inflow. Water quality improvements in Lake Ella included an order-of-magnitude drop in chlorophyll-a, a three-fold increase in Secchi disk depth, and improved bottom water oxygen levels. In the late 1990s, lake managers installed alum treatment facilities to treat inflow to two urban lakes in the Minneapolis and St. Paul metropolitan area (Pilgrim and Brezonik 2005a). In Fish Lake, the treatment system consisted of alum injection into a forced-main pipe with alum floc settling in a small natural pond prior to lake discharge. Jar testing performed to mimic mixing in the full-scale systems (Gt around 8,000) showed that SRP dropped to below detection at alum doses above 6 mg-Al/L. Field studies confirmed that TP removal was negligible at an alum dose of 1 mg-Al/L, but substantial at 8 mg-Al/L. Over the 5-year study period, average annual Secchi disk depth in Fish Lake increased from 1.5 m to 2.1 m. Additional jar testing by Pilgrim et al. (2007) on a range of surface water inflows to Minnesota lakes ranging in TP concentration from 0.1-0.7 mg-P/L showed >90% TP removal at alum doses of 4–8 mg-Al/L at a mixing intensity of Gt ~8,000. Mason et al. (2005) and Trejo-Gaytan et al. (2006) both saw >90% TP removal in jar tests of tributary inflows to the Salton Sea and Lake Tahoe, respectively, at an alum doses of around 4 mg-Al/L. Lake Tahoe samples underwent flash mixing followed by slow mixing (G was not reported), while the Salton Sea samples were not mixed. Ratios of Al:P at the optimal alum dose (>90% TP removal) for the studies noted above generally ranged from 20 to 60 mg-Al/mg-P. This was similar to the ratio in this study of 38.5 (5 mg-Al/L to 0.13 mg-P/L).

In our jar tests, at an alum dose of 2 mg-Al/L, we observed a decrease in SRP but not a corresponding drop in TP. Thus, SRP was being stripped out of the water but not settling out of the water column. This was likely due to the formation of a pin floc, which is small enough to remain suspended, but large enough to be trapped on a 0.45-μm filter, the working definition of nondissolved P. Pin floc can form when too little alum is added to a water sample, thereby limiting the floc building process and ultimate settling (Viessman and Hammer 1985). Pin floc formation at low alum doses has been reported in other studies examining alum treatment of lake inflows. Pilgrim and Brezonik (2005a) found that low alum doses of 1–2 mg-Al/L in jar tests with inflow to a Minnesota lake resulted in high SRP removal but little overall TP removal, and they attributed this observation to pin floc formation. In addition, they found that the P in the pin floc, while not technically soluble P, was still bioavailable to phytoplankton. They concluded that these low doses of alum would not ultimately reduce phytoplankton productivity in the lake. Kang et al. (2007) also reported the formation of pin floc and increased turbidity in jar tests with highway runoff at relatively low alum doses.

To the best of our knowledge, no study has evaluated the fate of alum floc discharged into an alkaline lake. In this study, we found that the combined sample of alum-treated creek water and lake water inhibited SRP removal for the mixing regimes with low energy. The alum dose needed to bring SRP below detection for the Low-Short and Low-Long mixing regimes increased from 5 mg-Al/L (Fig. 2) to 10 mg-Al/L (Fig. 3). The likely mechanism for this observation was that high pH at the low alum doses ($pH > 8.7$) impeded formation of SRP-adsorbing aluminum hydroxides, which are highly favored between pH 6 and 8 (Cooke et al. 2005). This was not the case for the high intensity mixing regimes, which showed SRP levels below detection at 5 mg-Al/L for the High-Short mixing regime and at 2 mg-Al/L for the Composite mixing regime. This may have been a byproduct of the higher mixing intensity initially forming a more robust floc capable of retaining sorbed P. An additional concern with discharging alum floc to alkaline waters is the formation of potentially toxic aluminate ion ($Al(OH)_4^-$) at elevated pH levels. However, little is know about the dynamics of aluminate toxicity because the bulk of toxicity studies related to alum treatment have focused on low pH conditions because alum tends to lower pH in treated water (Cooke et al. 2005).

Summary and conclusions

Alum addition to lakes is a longstanding and effective whole-lake management strategy to control internal P loading from anoxic sediments. More recently, lake managers have used alum to remove P from lake inflows as a method to lower external P loading. However, treating creek inflows with alum has a number of challenges not generally associated with lake treatment, related to dose determination, mixing requirements, the continuous and/or seasonal nature of inflow treatment, and solids management. In this study, we performed a series of jar tests to determine the optimal dose and mixing regimes needed to remove P from Matthiesen Creek inflow to Jameson Lake, and to evaluate potential discharge of alum solids directly into the lake. Based on jar test results, we recommend any engineered inflow treatment system on Matthiesen Creek should maintain an alum dose of >5 mg-Al/L under moderate mixing conditions (Gt ~ 1,000–3,000) with alum floc collected in an on-shore settling basin. The settling basin would need to be on the order of 30 m × 10 m to accommodate flows in the range of 2,000 m^3/d, with a 6-hr settling time. Additional conclusions and recommendations of the study include:

(1) An alum dose of 5 mg-Al/L was sufficient to remove the majority of SRP from creek water for all mixing regimes excluding High-Short. For all mixing regimes, doses of up to 20 mg-Al/L did not depress pH below the recommended minimum pH of 6.

(2) Incorporating flash mixing prior to low-intensity mixing did not enhance P removal, but flash mixing alone resulted in lower levels of P removal from creek water compared to low-intensity mixing.

(3) In-lake discharge of alum solids should be avoided for two reasons related to the elevated pH of surface waters in Jameson Lake. First, jar testing showed that lake waters tended to inhibit SRP uptake by alum floc, particularly at the low intensity mixing regimes. Second, high pH favors the formation of aluminate which could exhibit chronic toxicity to aquatic biota.

(4) A preliminary mass balance suggests that internal loading may be an order of magnitude greater than external loading. Treatment of lake inflow with alum is most effective for lakes where productivity is controlled by external loading. Any attempt to control external P loading from Matthiesen Creek should be coupled with efforts to better quantify the importance of internal loading and control it as necessary.

(5) If inflow alum treatment is implemented, lake managers should evaluate the potential to reuse alum solids captured in on-shore settling basins by land treating

upland areas identified as large sources of SRP to the spring that feeds Matthiesen Creek (Gallimore et al. 1999).

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Table 1.-Overview of mixing regimes.

	Possible Treatment Schemes					
Mixing Regime	Alum discharge to creek with natural turbulence	Alum discharge to settling basin with mechanical mixing	Paddle Speed (rpm)	Mixing Time (s)	G[1] (1/s)	Gt[2]
Low mixing intensity of short duration (Low-Short)	Alum discharge to creek 3 m from lake	Alum discharge with very weak mixing to settling basin	40	30	35	1,050
Low mixing intensity of long duration (Low-Long)	Alum discharge to creek 9 m from lake	Alum discharge with weak mixing to settling basin	40	90	35	3,150
High mixing intensity of short duration (High-Short)	Alum discharge to creek with flash mixing just upstream of the lake	Alum discharge to settling basin with flash mixing	250	30	350	10,500
High mixing intensity of short duration followed by slow mixing intensity of long duration (Composite)	Alum discharge to creek with flash mixing 3 m from lake	Alum discharge with flash mixing to settling basin with additional weak mixing in settling basin	250/40	30/30	350/35	13,650

[1]G, the velocity gradient, quantifies the amount of sheer stress in the system.
[2]Gt, the product of G and the duration of mixing, quantifies the combined effects of mixing intensity and mixing duration.

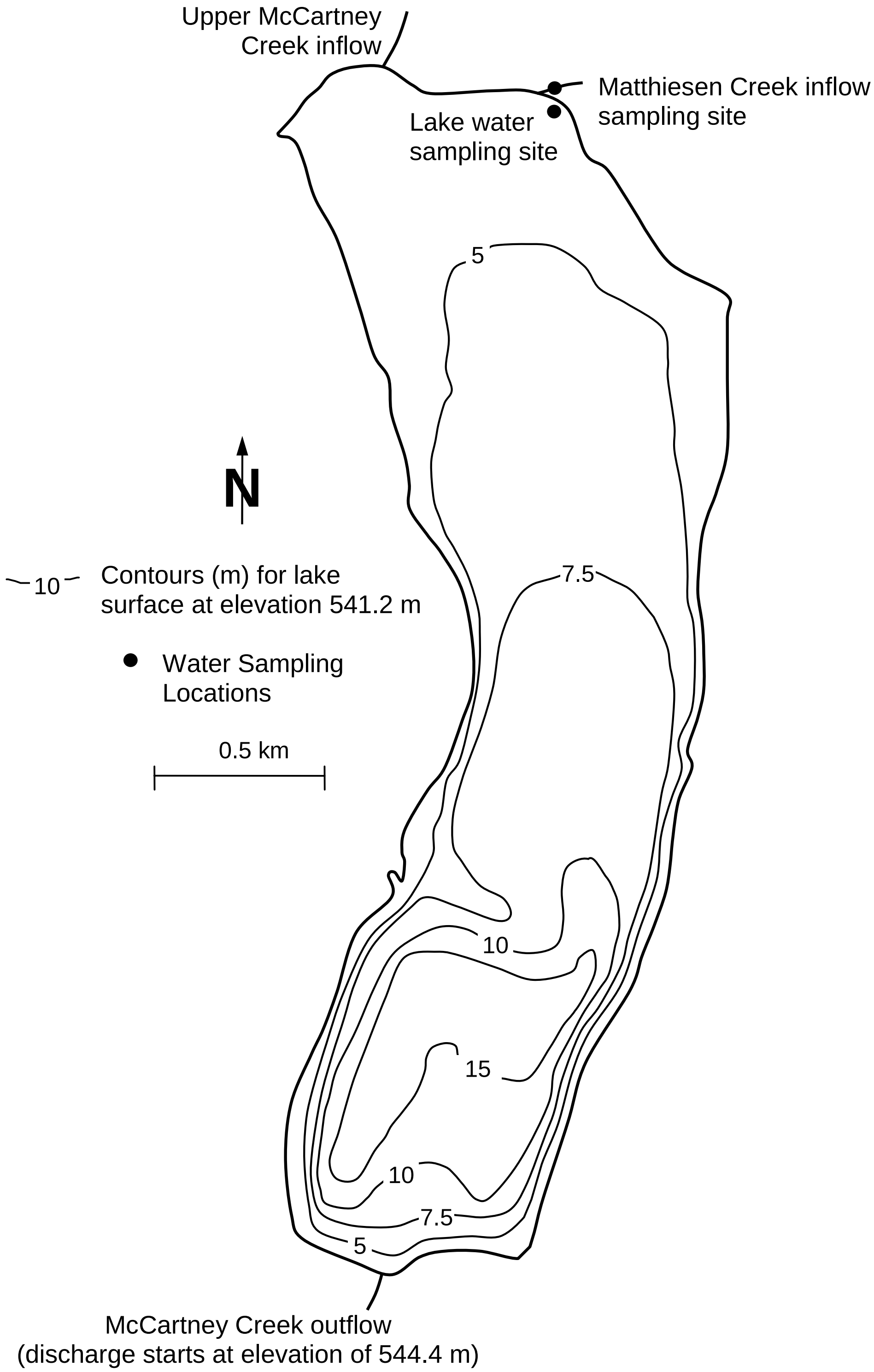

Figure 1.-Bathymetric map of Jameson Lake.

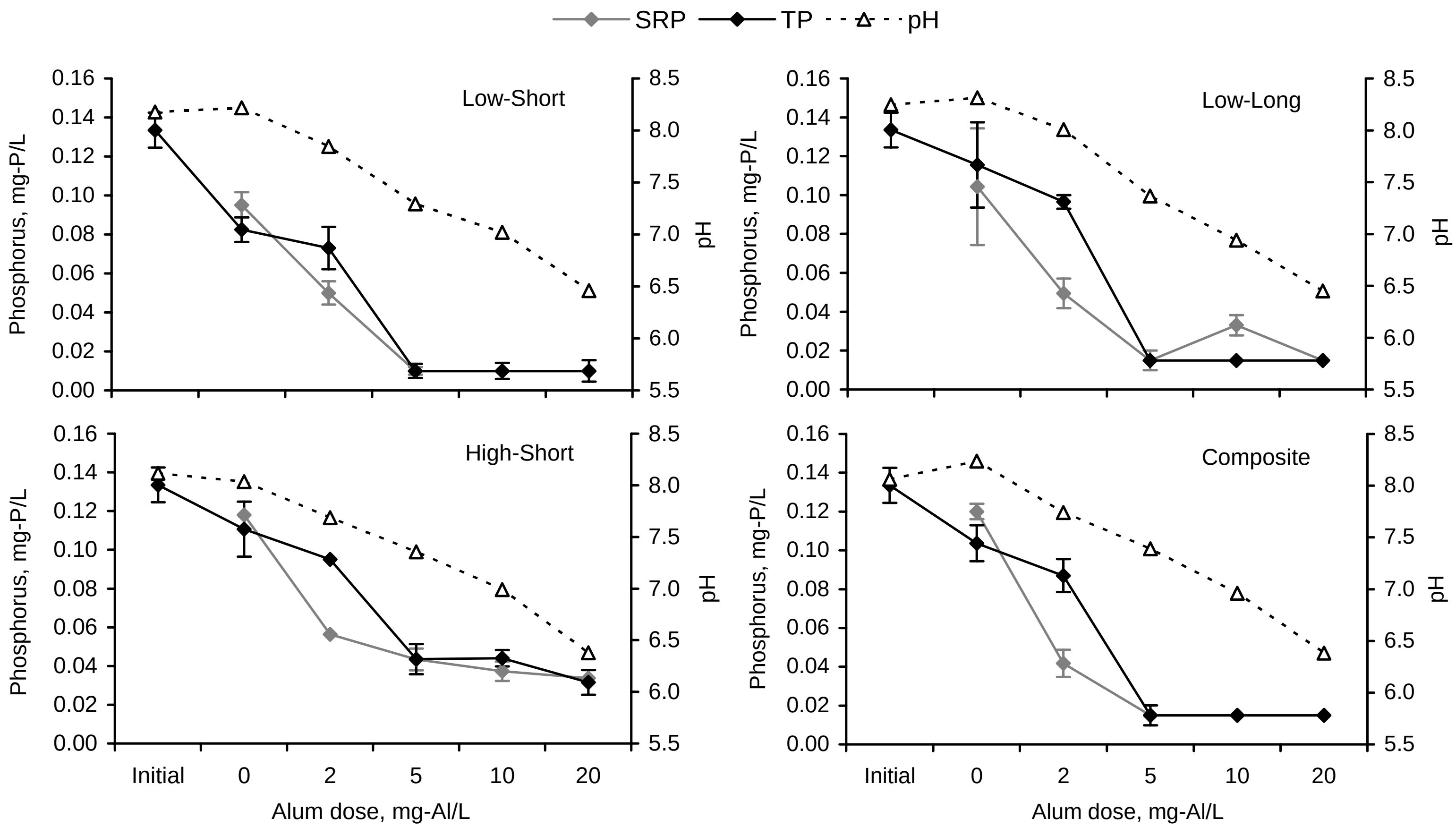

Figure 2.-Effect of alum dose on SRP, TP, and pH in jar tests with Matthiesen Creek water under four mixing regimes: (1) Low-Short, low mixing intensity of short duration; (2) Low-Long, low mixing intensity of long duration; (3) High-short, high mixing intensity of short duration; and (4) Composite, high mixing intensity of short duration followed by slow mixing of long duration. Error bars are plus/minus one standard deviation (n = 3) and are only shown where standard deviation is larger than the symbol.

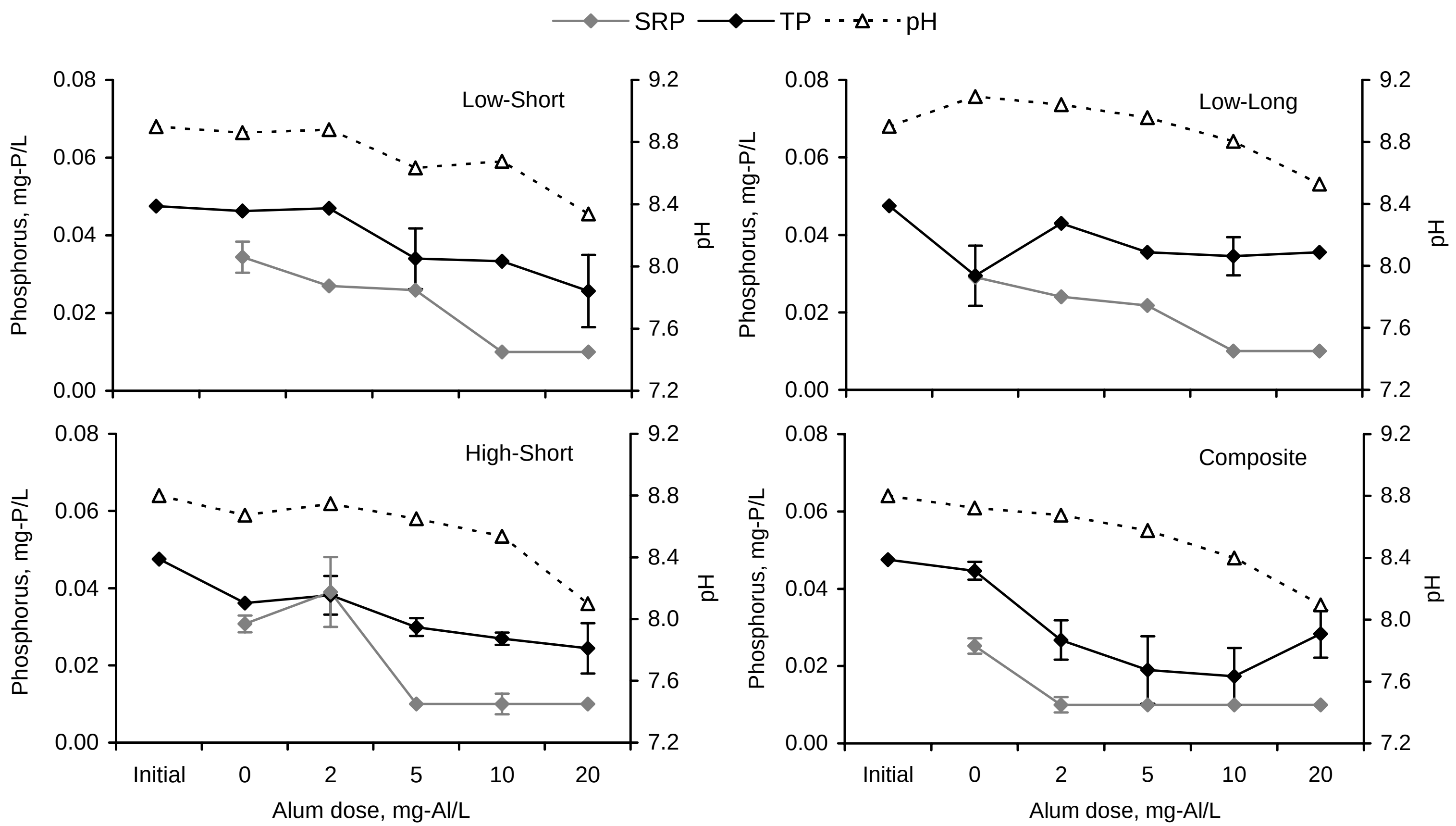

Figure 3.-Effect of alum dose on SRP, TP, and pH in jar tests with a combination of 100 ml of alum-treated Matthiesen Creek water and 400 ml of lake water. The creek water was mixed prior to addition to the lake water under three mixing regimes: (1) Low-Short, low mixing intensity of short duration; (2) Low-Long, low mixing intensity of long duration; (3) High-short, high mixing intensity of short duration; and (3) Composite, high mixing intensity of short duration followed by slow mixing of long duration. Error bars are plus/minus one standard deviation (n = 3) and are only shown where standard deviation is larger than the symbol.