OPTIMAL MIXING REGIMES FOR ALUM TREATMENT OF CREEK INFLOW TO

JAMESON LAKE, WASHINGTON

By

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Abstract

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Hydrated aluminum sulfate (alum) has been used since the early 1970s for treating water

column and sediment phosphorus (P) in lakes. A new and innovative method of

combating eutrophication in lakes uses engineered treatment systems to inject alum into

stream inflow to reduce external P loading. 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 entering Jameson Lake, WA. Alum doses of 0, 2, 5, 10 and 20 mg-

Al/L were applied under four mixing regimes: (1) slow mixing of short duration, (2) slow

mixing of long duration, (3) high mixing of short duration, and (4) high mixing of short

duration followed by slow mixing of long duration, and were tested to determine

maximum P removal efficiency. Tests were also completed to determine the need for a

settling basin at the creek outlet, compared to the treated water directly entering the lake.

Overall removal efficiency was high in all mixing regimes, with 45-98% soluble reactive

P and 15-89% total P removed from the creek water, and a decrease in overall lake total P

(23-79%) after addition of treated creek water. The mixing regimes with a slow stirring period had the greater efficiencies after settling without lake water, but high mixing was more efficient when the treated creek water settled in lake water.

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Dedication

This thesis is dedicated to my mother and father

who provided both emotional and financial support over the years.

INTRODUCTION

Alum ($Al_2(SO_4)_3*18H_2O$), a common water quality amendment in drinking water and wastewater facilities, enhances particulate and phosphorus (P) removal by chemical flocculation. With mixing in water, the aluminum salt dissociates and the aluminum forms different aluminum hydroxides. The pH changes associated with this reaction are due to the loss of hydroxyl ions from solution from the formation of the hydroxides. The hydroxides that form below a pH of 6 are positively charged, the main hydroxide that forms between pH of 6 and 8 is $Al(OH)_3(s)$, and hydroxides formed above a pH of 8 are negatively charged. The charged ions are soluble in water and can be toxic in aquatic biota. The formation of solid aluminum hydroxide particles is used in water and wastewater treatment because it both adsorbs soluble reactive P (SRP) and enhances settling of particulate P (Droste 1997). Since the early 1970s, alum has been used in lakes to remove water column P and prevent P release from lake sediments by forming a blanket layer on the sediment surface which can adsorb P without releasing it under anoxic conditions (Cooke et al. 2005).

In a review of traditional whole-lake treatments, Welch and Jacoby (1999) noted that alum treatments can be very effective, lasting up to 15 years, but in some cases the treatments were completely ineffective showing no improvements one year after addition. A controlling parameter identified within that study of the effectiveness of alum treatments was the source of P. Internal cycling of P from lake sediments can be effectively treated by whole-lake alum application, while external sources of P to lakes, such as urban and agricultural runoff, require some other treatment option and/or implementation of watershed management strategies (Welch and Jacoby 2001).

In late the 1980s, lake managers began to use alum as a means to remove P from lake inflows, thereby decreasing external P loading and enhancing water quality. 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). Due to the lack of available land to build conventional stormwater treatment management systems, lake managers used a novel system which injected liquid alum into the stormwater pipes, with the necessary mixing occurring in the pipes and settling occurring in the lakes. This process decreased orthophosphate, known analytically as SRP, by at least 90 percent and total phosphorus (TP) in the lake by 89 percent. No negative consequences related to the in-lake disposal of the resulting aluminum hydroxide floc was reported.

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 Tanners Lake, alum floc was deposited in a settling pond prior to the lake, and in Fish Lake, alum floc was released into a small natural pond, where it reportedly had the added benefit of inhibiting sediment P release under anoxic conditions. TP removal in the inflows ranged from 60-80%, and TP in the lakes decreased by at least 50% over a 5-year period. In Fish Lake, average annual Secchi disk depth increased from 1.5 m to 2.1 m.

Some recent studies have examined the efficacy of removing P from tributary inflow through a combined addition of alum and different forms of polyacrylamide, synthetic polymers commonly used in water treatment to enhance flocculation (Mason et al. 1998, Kang et al. 2007). These researchers showed that polyacrylamide can increase

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hardy floc formation in more rapid mixing environments, and enhance floc deposition when no low mixing or stagnant period is provided.

In whole-lake treatments, alum is added to the water column allowed to settle to the bottom of the lake where if covers lake sediments and forms a sorption barrier to soluble P effluxing from anoxic sediments. In contrast to whole-lake alum treatments, treating natural stream and creek inflows with alum encompasses a number of unique challenges. First, determining the alum dose for inflow treatment is not as straight forward as for whole-lake treatment (Pilgrim et al. 2007), which is commonly based on the alkalinity of lake water and/or the content of P in lake sediments (Cooke et al. 2005). Second, mixing is required between the added alum and the inflow to promote the formation of a floc that both retains P and settles out of the water column (Droste 1997).

Another contrast to whole-lake treatments is that inflow treatment is not a one time event, but needs to be applied when external loading is high, commonly during peak flows in the spring. A final challenge is the determination of where the treated water should be allowed to settle, in the lake where the alum floc could potentially have unanticipated consequences (e.g., filling in shallow areas with resulting macrophyte growth), or in a settling basin with related management efforts and expense related to final disposal of solids. In essence, alum treatment of lake inflow is akin to alum treatment of water and wastewater, where a continuous input of a target concentration followed by a particular mixing regime is optimized to promote the removal of particulates and associated P, as well as dissolved P, via flocculation and settling (Hammer and Hammer 1996).

The objectives of this study, which examined the potential to remove P from Matthiesen Creek inflow to Jameson Lake, were three-fold: (1) to determine the alum dose to substantially reduce P in inflow from creek P, (2) to find the most effective mixing regime to enhance P removal, and (3) to evaluate simulated settling scenarios for the treated water.

MATERIALS AND METHODS

Study Site

Jameson Lake is a eutrophic alkaline lake located within an arid, agricultural watershed in central Washington (Fig. 1). The lake has a maximum depth of 30 m and a surface area 2.14 km^2. A change in agricultural practices in the watershed had lead to an increase in the lake's volume over the past 50 years. The rising water has lead to an increase in surface area by 60%, flooding old farms and houses. Another aspect of the lake hydrology is an average annual change in surface water elevation of 0.45 m which reflects the area's wetter winter and spring months and drier summer months (Anchor Environmental 2006). Another factor in the increasing lake volume is the high elevation of the outlet. The lake water needs to pass through an old irrigation ditch at the south end and the lake can only drain when the water level is greater than the ditch. Occasional flooding in the watershed has caused an increase in external P loading which is likely related to the increase in P incorporated into the excess water as it travels through the agricultural watershed. In 2005, this process led to hyper-eutrophic conditions in the lake (Fig. 2), and has led to extreme summer algal blooms, low dissolved oxygen concentrations, and fish-kills (WQE 2006).

There are two main inflows into Jameson Lake, McCartney Creek and Matthiesen Creek. Upper McCartney Creek flow into the north end of the lake from Bennett Lake, an alkaline lake to the north. Matthiesen Creek, which is feed by a natural spring roughly 1 km upstream, flows into the northeast side of the lake. Matthiesen Creek discharge

remains fairly stable year round and delivers 25-50% of the total inflow to the lake. The creek inflow accounts for 50-99% of the external P load to the lake, with the larger loading occurring during the spring months (WQE 2006). Livestock within the creek's watershed is presumed to be the major P source in Matthiesen Creek (PGG 2004). Rural Douglas County has embarked on an extensive effort to restore its natural resources and promote environmental stewardship (FCCD 2008), including restoration efforts towards Jameson Lake. Due to the growing interest in the water quality of the region, and the recent severe degradation in water quality in Jameson Lake, this pilot study was initiated to determine an appropriate treatment scheme for the Matthiesen Creek, with potential mixing requirements and solids disposal taken into consideration.

Mixing Regimes

Jar tests were conducted under a number of regimes which mimicked potential treatment scenarios at the site. Mixing regimes were developed to model the impacts of mechanical mixing devices installed at the site as well as simpler methods using the natural turbulence of the stream for floc formation. To quantify the mixing, the velocity gradient, G (1/s), and the unitless parameter, Gt, were used as design parameters to match existing technologies. Both parameters are widely used in wastewater treatment flocculation processes as measurable quantities which are related to the total number of collisions between floc particles, causing aggregation into larger particles that settle out of solution (Reynolds and Richards 1996). The higher the amount of collisions, the more likely a heartier floc will form, although there is an upper limit in which too great a magnitude will result in appreciable shear stress which can tear the floc apart. The velocity gradient can be related to power associated by the equation:

$$G = \sqrt{\frac{P}{\mu V}} \qquad\qquad (1)$$

where G is the velocity gradient, P is the power input (Nm/s), μ is the viscosity of the

water (Ns/m^2), and V is the volume of water (m^3). G values in the experiments were

extrapolated from manufacturer provided information. G values in streams are dependent

on channel depth and width, flow rate, and head loss (Mason et al. 2005). Velocity

gradients occur in both vertical and transverse directions, further complicating mixing

quantification (Belatos 1979). Natural stream flow is normally fully turbulent, with very

turbulent zones behind obstructions (boulders) creating wake zones of intense mixing

(Tritico and Hotchkiss 2005).

Laboratory Simulation of Mixing Regimes

The study included an evaluation of four mixing regimes (Table 1). A regime of

slow mixing of short duration (SS) was designed to replicate a liquid alum side-stream

entering the creek with mixing from the natural turbulence of the creek water, or with a

simple mixing device (ie. a weir). The slow mixing had a typical G value from

wastewater flocculation processes (35 1/s) (Tchobanoglous and Burton 1991).

The second regime was slow mixing of long duration (SL) and replicated the side-

stream injection at a point further upstream of the lake, thus allowing more mixing to

occur. Based on an average creek velocity of 0.165 m/s (WQE unpublished data), the

injections for SS and SL would be 5.8 and 15 m upstream of the discharge point,

respectively. The advantage to the natural mixing is that much less electricity would be

required to deliver the alum and remove P, making the process more sustainable.

The impact of adding a mixing impeller was investigated in the final two mixing

regimes. The high mixing of short duration (HS) regime represented a flash mixing

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process just prior to the outlet of the creek. The HS regime had a G value typical of rapid

mixing processes in wastewater treatment plants (350 1/s), and was tested for a higher

contact time period which elevated the Gt value (10,500) (Tchobanoglous and Burton

1991). The impeller option was further evaluated by following the rapid mixing process

by a slower mixing (HSSL). The real-world application of this scenario would be an

impeller placed a distance upstream of the lake inflow, with the treated water cascading

down the stream channel. This is a common water treatment procedure for flocculation

(Reynolds and Richards 1996), and avoids the potential for poor floc formation

sometimes observed when flash mixing is not followed by slow mixing (Trejo-Graytan et

al. 2006).

Water Quality Monitoring

A preliminary round of water samples were collected from Matthiesen Creek in

May of 2007. A second set of water samples were collected for more extensive testing in

June of 2007 from both Matthiesen Creek and Jameson Lake (Fig. 3). 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. June jar tests were conducted as

follows. Triplicate beakers were each filled with 1 liter 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 and was calibrated daily with pH 3, 7, and 10 standards. Water

samples were collected for pretreatment SRP and TP analyses. Beakers were loaded onto

a Phipps and Bird jar tester and the target alum dose was added (Fig. 4). The alum used in

the experiments was in solution form as described in Cooke et al. (2005). It was made by

dissolving 1.46 g of aluminum sulfate octadecahydrate in 100 mL of water. From that

stock solution, alum doses were 0, 2, 5, 10, and 20 mg-Al/L were made by diluting 1.6 mL, 4 mL, 8 mL and 16 mL, respectively, of the stock into the 1 L test jars. This range is typical of those used in lake and creek alum treatment studies (DeGasperi et al. 1993, Pilgrim and Brezonik 2005b, Cooke et al. 2005).

The jars were then mixed at the desired target mixing regime. Once the mixing stopped, alum-treated creek water was allowed to settle under two settling scenarios. To mimic discharging the alum-treated creek water into the lake, 100 mL of the water was gently poured into a previously measured 400 mL of lake water to simulate allowing the treated water to directly enter the lake. Another 500 mL of the mixed creek water with alum was gently poured into a separate empty beaker. This simulated the settling of alum-treated creek water in an on-shore settling basin, which would allow for a smaller impact on the lake.

After settling for 6 hours, a duration that typically allow for nearly complete floc settling (Pilgrim and Brezonik 2005a), the beaker water was tested for pH, and water samples were collected for post-treatment SRP and TP analyses. SRP samples were preserved by filtering through a 0.45µm filter and freezing; TP samples were preserved by adding sulfuric acid and storing at 11 °C (APHA 1998). SRP analyses were conducted on a Lachat QuikChem 8500 Flow Injector Analyzer (Method 10-115-01-1-P). TP was measured using Hach TNT 843 tubes and measured on Hach DR2800 spectrometer. Quality assurance procedures entailed comparisons to external standards (SRP only, less than 8% difference), on-going precision recovery samples (SRP only, less than 3% difference), and duplicates (TP only, less than 10% difference). Approximate detection

limits (three times the standard deviation of the blanks) for SRP and TP were below 0.02 mg-P/L. Non-detect samples were reported as one-half the detection limit.

The preliminary jar tests conducted in May were done in a similar fashion with the following four differences: (1) tests were done without replication, (2) tests included only two mixing regimes of 50 rpm for 60 s (Gt of 2,600) and 250 rpm for 60 s (Gt of 20,400), (3) only TP was evaluated, and (4) no combined creek water and lake water jar tests were conducted.

RESULTS

Preliminary May Jar Tests

Preliminary jar tests in May showed that alum addition had a profound effect on pH and TP levels in creek samples. The pH dropped from 8.3 to 7.0 under low mixing and to 6.0 under high mixing (Fig. 5). TP in creek water was around 0.2 mg-P/L. Just allowing the samples to settle resulted in a drop of TP to 0.15 mg-P/L (Fig. 6). At low alum doses (2 and 5 mg Al/L), TP removal increased with increasing alum dose and enhanced mixing. At higher doses (10 and 20 mg Al/L) TP removal decreased with increasing alum dose and TP removal was fairly similar between mixing regimes. Optimal TP removal from ~0.2 to ~0.05 mg/L, a 75% decrease, occurred at a dose of 5 mg Al/L with enhanced mixing and resulted in an acceptable pH drop. A similar level of removal was attained with an alum dose of 10 mg Al/L without enhanced mixing.

pH in June Jar Tests

In the June jar tests, pH in creek water progressively dropped from around 8.2 to 6.5 with increasing alum dose under all mixing regimes (Fig. 7). There was very little change in pH response to the four mixing regimes, with almost identical pH decreases at each alum dose. All mixing regimes had a pH of around 7 at an alum dose of 10 mg-Al/L. All mixing regimes for the creek water were maintained within the non-toxic range of pH 6-8. The pH in the combined lake water and treated creek water showed a slight decrease with increasing alum dose, but stayed above 8 at all doses and all mixing regimes. The high alkalinity lake water (pH ~9) provided a large buffering capacity and hence the

smallest changes in pH were observed especially in the low alum doses. Note the very small error bars in Fig. 7, indicating that the triplicate treatments all behaved very similarly.

TP and SRP in June Jar Tests

All of the mixing regimes showed the same overall trend for SRP and TP for the samples representing a settling basin (Fig. 8). TP levels in the untreated jar samples were around 0.1 mg-P/L, which was lower than the 0.15 mg-P/L levels observed in May. TP appeared to consist entirely of dissolved P since SRP equaled TP. In the SS, SL and HSSL mixing regimes (Fig. 7 A, B and D, respectively) P was completely removed at an alum dose of 5 mg-AL/L. TP never got below around 0.04 mg-P/L in the HS jars. An alum dose of 2 mg-Al/L had an interesting trend in all four of the mixing regimes. While almost all of the TP in the lake was available as SRP initially, every mixing regime had more TP than SRP at the 2 mg-Al/L dose.

The settling scenario representing the solids directly entering the lake provided a much different TP removal profile (Fig. 9). The combined creek and lake water had an initial TP concentration of around 0.04 mg-P/L. This TP content is a combination of TP in the lake water and the addition of a 1:5 dilution of creek water which contained around 0.1 mg-P/L. So around half the TP was from the creek water (one fifth of 0.1 mg-P/L is 0.02) with the other half pre-existing in the lake water. In the SS regime (Fig. 9A) only the highest dose (20 mg-Al/L) showed substantial TP removal. The SL regime (Fig. 9B) did not produce much decrease in TP at any Al dose. The HS regime (Fig. 9C) was the only regime to drop the TP concentration to below the detection limit at all doses above 5 mg-Al/L. In the HSSL regime (Fig. 9D) TP decreased to around 0.02 mg-P/L at alum doses of 5 mg-Al/L and higher.

<center>**DISCUSSION**</center>

Changes in pH due to Alum Dose

In our study, we monitored the pH change to determine if the natural buffering capacity of the lake and creek water could support alum addition without the use of chemical buffers. Outside this range, aluminum hydroxide ions are formed which are water soluble and potentially toxic to aquatic biota (Cooke et al. 2005, Pilgrim and Brezonik 2005b). Other lake inflow treatment studies (Harper and Herr 1992, Haggard et al. 2004) have used chemical buffers to avoid any toxicity effects of the aluminum ions. Kang et al. (2007) limited their studies to alum doses that would not result in acidified water. Both Mason et al. (2005) and Pilgrim et al. (2005a) did not report pH values associated with their jar tests. In a follow-up study, Pilgrim and Brezonik (2005b) modeled the expected pH changes occurring in stream water with respect to alkalinity and natural organic matter in the water. The study clearly shows the increase in aqueous Al below pH 5.5 and above pH 8. This study raises concern for the discharge of alum floc to directly to Jameson Lake, which has a pH around 9.

Optimal Alum Dose and Mixing Regime

While 2 mg-Al/L decreased SRP by more than 50% in the June creek water test jars, it proved to be ineffective at decreasing TP in all mixing regimes. This was most likely due to the formation of a pin-floc, which is not large enough floc to settle out of solution yet too large to pass through the 0.45μm filter used for SRP analysis. Therefore, P within this pin-floc would be detected in TP measurements. This same phenomenon seemed to occur in three out of four of the lake and creek water mixture mixing regimes,

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namely the slow mixing of short duration, the slow mixing of long duration and the high mixing of short duration. A greater Gt (potential for collisions) may increase the floc formation, which would provide for more TP removal. In an inflow treatment to an urban lake in Minnesota, Pilgrim and Brezonik (2005a) also noted that a low alum dose, 1 mg-Al/L, provided sufficient SRP removal from their jar tests, but did not remove the TP. In the field, the researchers found that a 1 mg-Al/L dose was not able to remove sufficient P, and a higher dose of 8 mg-Al/L was implemented. This dose ultimately led to a drop in TP from 0.047 to 0.026 mg-P/L in lake inflow, thereby provided better P removal. Pin-floc formation can occur at higher alum doses as well. Kang et al. (2007) used a very small velocity gradient (9.3 1/s) to mix alum into first flush storm-water, and noticed pin floc around 5 mg-Al/L.

The jar tests showed that an alum dose of 5 mg-Al/L was as effective at removing SRP and TP as the 20 mg-Al/L dose, with at least 75% removal efficiency. A number of related studies have found similar results, though few examined the range of mixing scenarios as we did in this study. Mason et al. (2005) performed jar tests on river water flowing into the Salton Sea and found that 4 mg-Al/L was able to reduce the SRP by 80% and removed almost all of the TP. The SRP was enhanced even more when alum was applied with different polyacrylamide polymers to create a stronger floc. When looking at the turbidity of storm-water runoff, Kang et al. (2007) also had higher removal efficiencies with the alum salts and polymers. Harper and Herr (1992) used a 5.3 mg-Al/L dose to eliminate greater than 85% of the incoming SRP and TP in urban storm-water ponds in Florida.

Our results are very similar to those of Pilgrim and Brezonik (2005a) which reported similar initial TP values. An alum dose of 6 mg-Al/L dropped the TP concentration to below 0.02 mg-P/L. The authors used a 103 rpm mixing speed for 30 s in their jar tests to analyze for alum efficiency which would have a G value that falls between the fast and slow mixing in our jar tests. The mixing regime used in that study had G values from the full scale systems in between our minimum and maximum values (184, 203, and 144 1/s). These results lead us to believe that our results will scale-up well to a full sized treatment facility.

This study incorporated mixing effects on P removal. The only mixing regime that did not reduce SRP and TP in the creek water to below detection limits was HS. However, the HS regime had the greatest TP removal in the creek:lake water mixture (Fig. 9). This increase in removal is most likely due floc formation being facilitated by the inevitable mixing that occurred when the two water samples were combined; without this extra mixing floc development would have been hindered, as in the creek water settling (Fig. 8). TP in the other mixing regimes does not present the same characteristic because floc development was complete or nearly complete before the treated water entered the lake water. In fact, this additional mixing may have caused the HSSL mixing regime to tear apart due to high shear stress on the large floc particles. There is no evidence within the creek water mixing that this occurred, and therefore it can only be explained by the added mixing involved in the attempt to combine the lake and creek water. Our results agree with those of Trejo-Gaytan et al. (2006) who argued that a period of slow mixing is important in order to obtain maximum P removal. When the treated water settles in the original containers, the mixing regimes having a slow mixing period

had the highest removal efficiencies. If the treated water directly enters the lake, then the slow mixing period will prevent the alum floc from adsorbing P from the lake water column.

Effects of Settling Scenario on P removal

Each settling scenario has its advantages. The alternative of sending the treated water into the lake has two significant advantages. First, the stream P concentrations are too low to saturate the alum's removal efficiency, therefore allowing lake P to be bound in the floc and removed from the water column. The second significant advantage is that if the alum floc is allowed to settle onto the lake sediments, it will prevent the release of lake sediment P into the water under anoxic conditions (Haggard et al. 2004, Pilgrim and Brezonik 2005a).

The settling basin scenario also has significant advantages. The pH of the creek stayed within the 6-8 range that has been noted as a safe range for biota due to the presence of soluble Al species (Hayden and Rubin 1974, Barbiero et al. 1988, Cooke et al. 2005, Pilgrim and Brezonik 2005b). This is not true for the lake, where its high pH and high alkalinity buffer the lake water and any released alum may form toxic aluminum anions. Creation of a settling basin would also allow for storage of the floc, which can cause anoxic conditions to prevail, and kill off benthic invertebrates within the basin, but protect the lake's biota (Barbiero et al. 1988, Pilgrim and Brezonik 2005a, Pilgrim and Brezonik 2005b).

Other alternatives to alum are ferric chloride and synthetic polymers. Ferric chloride has been shown to resist releasing bound P in wetland mesocosms, although problems may arise in a shallow settling basin (Sherwood and Qualls 2001). The use of

synthetic polymers has proved to be more efficient than alum at removing TP with

shorter settling times (5 minutes) although this is a more expensive method (Mason et al.

2005).

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, including: (1) dose determination, (2) mixing requirements, (3) the continuous and/or seasonal nature of inflow treatment, and (4) 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 settling of alum floc in the lake.

From these tests, several conclusions can be drawn. For the creek water jars, which represent the construction of a settling basin at the site, an alum dose of 5 mg-Al/L lowered the P concentrations in the creek water to non-detect in all but the HS mixing regime. The 5 mg-Al/L dose is also recommended based on the formation of the pin-floc at 2 mg-Al/L. The alum dose at this site can be increased to 20 mg-Al/L without resulting in a pH low enough to form positively charged toxic aluminum species (< 6).

The lake: creek mixture, which represents allowing the treated water to directly enter the lake, has some important differences. The pH of the system never dropped below 8, which means there is the potential for toxic aluminum ions. The only mixing regime to effectively remove P was HS, and to a lesser extent HSSL.

Before implementation of this method at the site, a number of recommendations should be taken into consideration. A P mass balance within the lake needs be conducted. If internal cycling is an important P source, then installing a lake inflow treatment system

will not effectively eliminate eutrophication problems. For Matthiesen Creek water, alum doses of 5-20 mg-Al/L will be efficient at removing P without dropping the pH below the toxic threshold of 6. Treatment of Matthiesen Creek could use three of the four mixing regimes, SS, SL, or HSSL. The SS regime is recommended due to the minimal mechanical mixing involved.

If direct discharge into the lake is desired, then the HS mixing regime should be used. In our jar tests, it was the only regime to remove P to below the detection limits. Before installation, an evaluation of the toxicity of negatively charged aluminum ions should be conducted.

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TABLES

Table 1. Overview of Mixing Regimes

Mixing Regime	Application	Paddle Speed (rpm)	Mixing Time (s)	G (1/s)	Gt
Slow Mixing of Short Duration (SS)	Discharge to creek near lake	40	30	35	1050
Slow Mixing of Long Duration (SL)	Discharge upstream in creek	40	90	35	3150
High Mixing of Short Duration (HS)	Discharge to creek with flash mixing	250	30	350	10500
High Mixing of Short Duration and Slow Mixing of Long Duration (HSSL)	Discharge upstream in creek with flash mixing	250	30	350	13650
		40	90	35	

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Figure 1. Photo of Jamison Lake looking in a northeastern direction towards the Matthiesen Creek outlet.

Figure 2. Algal bloom caused by excessive P loading in Jameson Lake, WA. View facing the lake's southern outlet.
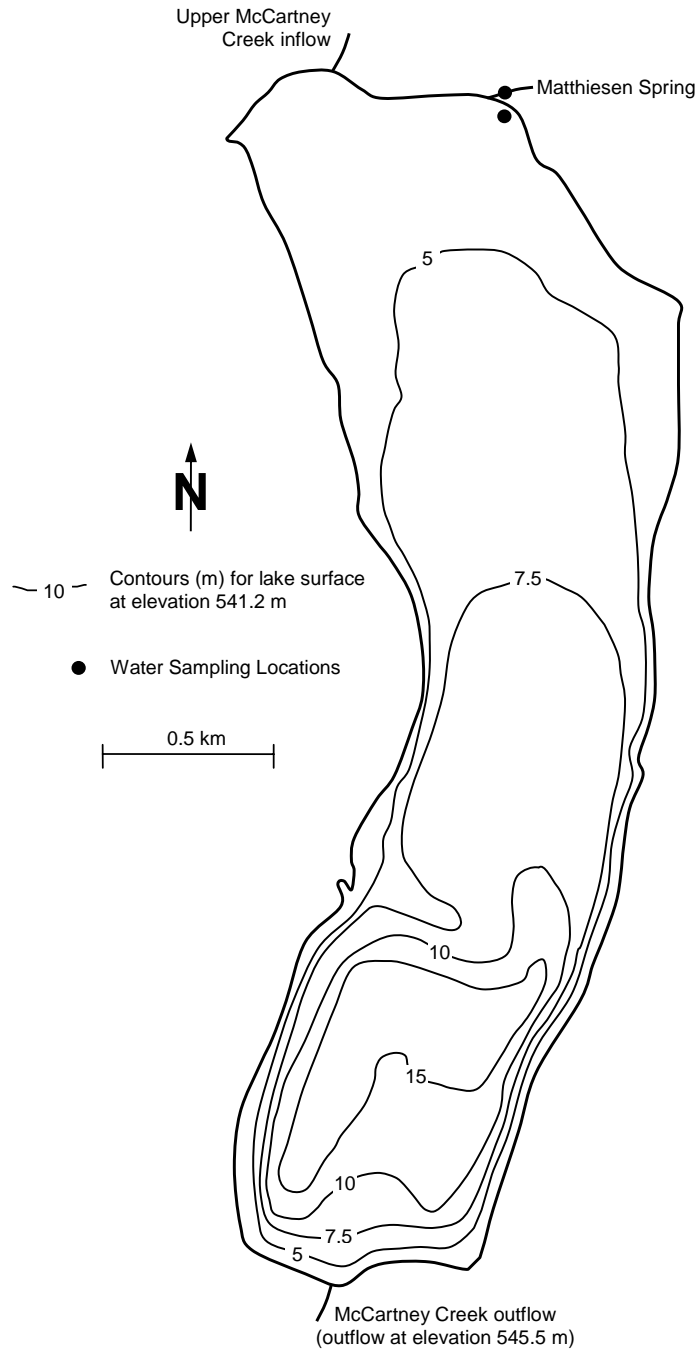
Upper McCartney
Creek inflow

Matthiesen Spring

5

N

— 10 — Contours (m) for lake surface
at elevation 541.2 m

● Water Sampling Locations

0.5 km

7.5

10

15

10

7.5

5

McCartney Creek outflow
(outflow at elevation 545.5 m)

Figure 3. Bathymetry map of Jameson Lake showing sampling locations.

Figure 4. Photo of mixing apparatus and jars of creek water being tested.
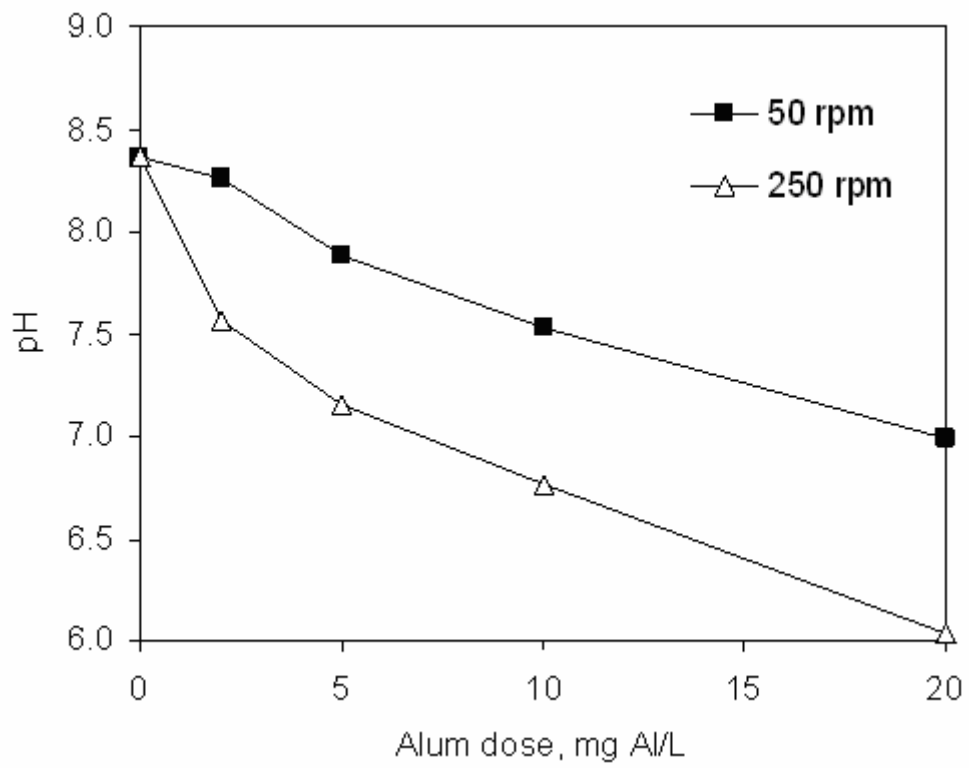
Figure 5. Effect of alum dose and mixing regime on pH in preliminary May jar tests.
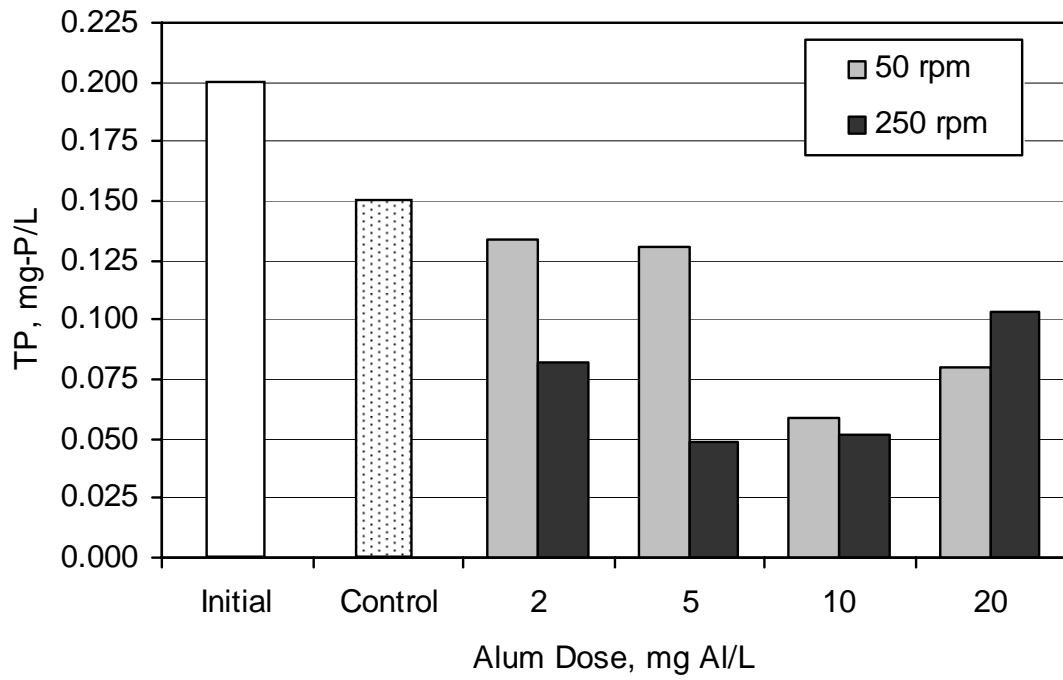
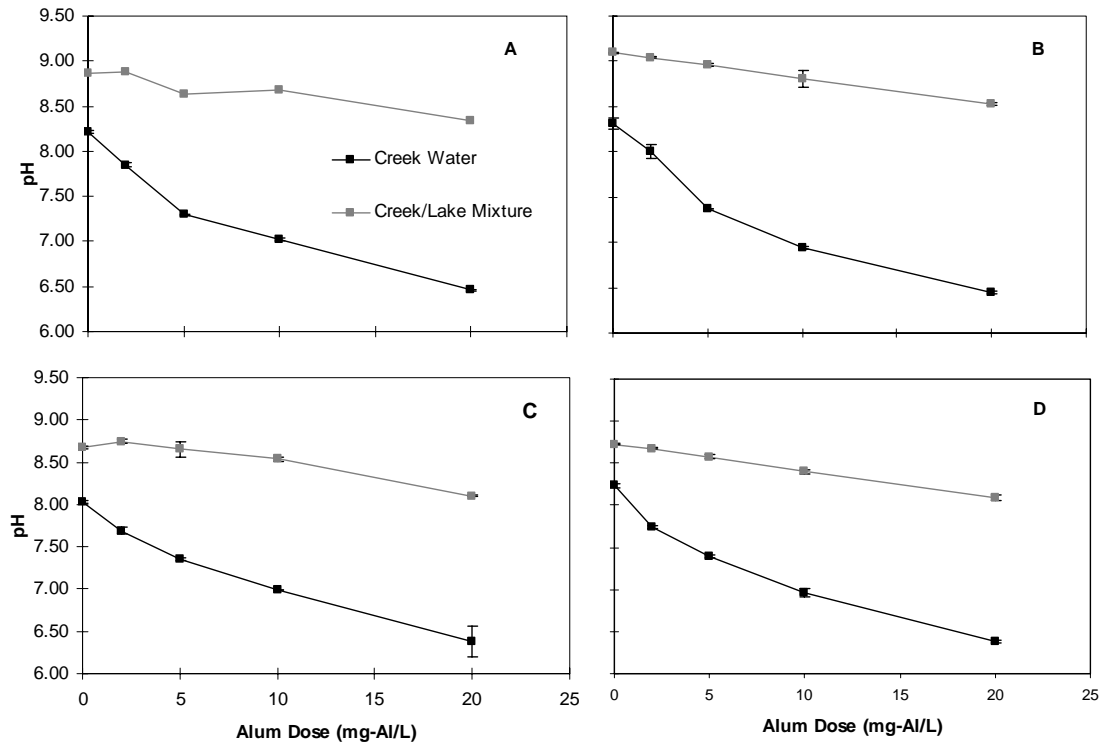Figure 6. Effect of alum dose and mixing regime on TP in preliminary May jar tests.

Figure 7. Effect of alum dose and mixing regime and settling scenario on pH. Initial pH, and the final pH for the two settling configurations (discharge into the lake and discharge into a settling basin) shown. (A) Slow mixing of short duration (SS). (B) Slow mixing of long duration (SL). (C) High mixing of short duration (HS). (D) High mixing of short duration followed by slow mixing of long duration (HSSL). Plus and minus one standard deviation shown, where deviation is larger than the symbol.
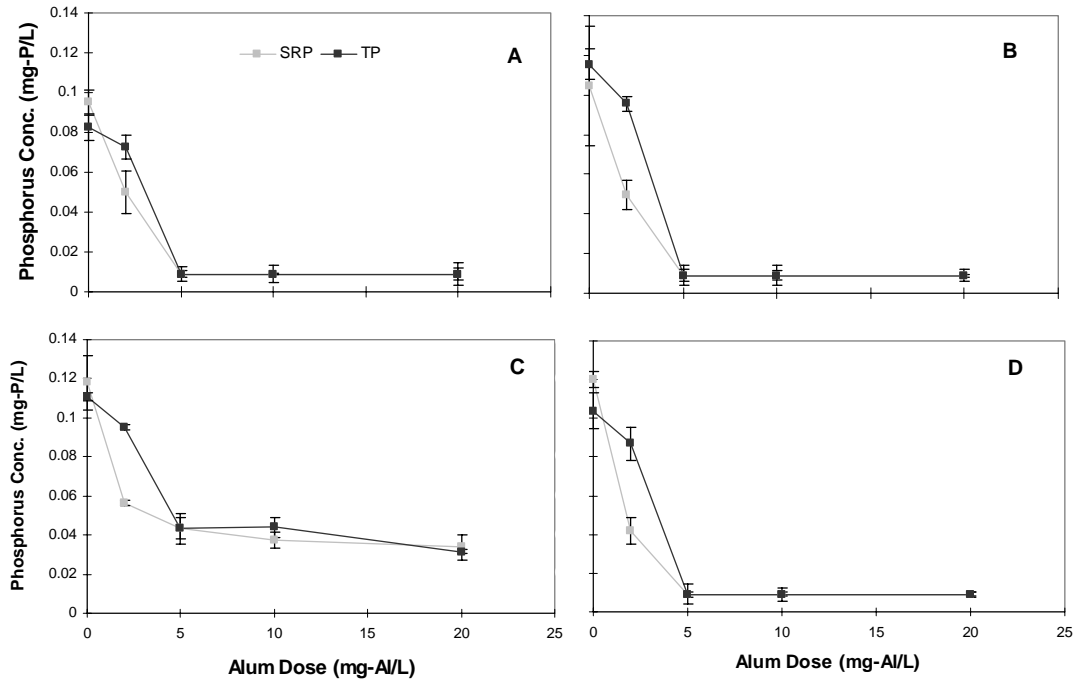
Figure 8. Effect of alum dose, and mixing regime on SRP and TP in Jameson Lake jar test. (A) Slow mixing of short duration (SS). (B) Slow mixing of long duration (SL). (C) High mixing of short duration (HS). (D) High mixing of short duration followed by slow mixing of long duration (HSSL). Plus and minus one standard deviation shown, where deviation is larger than the symbol. Non-detects are reported as one half the detection limit.
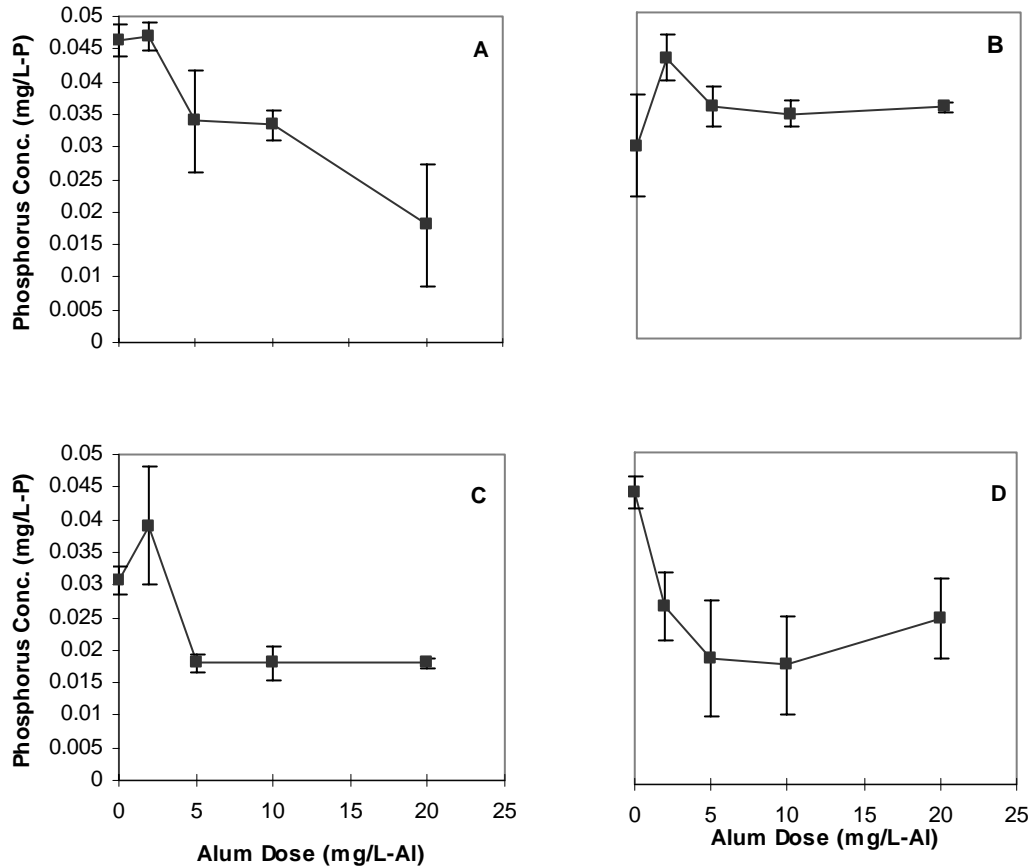
Figure 9. Effect of Jameson Lake water on TP from alum-treated Matthiesen Creek water. Jar tests were designed to replicate addition of treated creek water directly into the lake. (A) Slow mixing of short duration (SS). (B) Slow mixing of long duration (SL). (C) High mixing of short duration (HS). (D) High mixing of short duration followed by slow mixing of long duration (HSSL). Plus and minus one standard deviation shown, where deviation is larger than the symbol. Non-detects are reported as one half the detection limit.