OPTIMIZATION OF RESIDUALS MANAGEMENT AT A DRINKING WATER TREATMENT FACILITY

by

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ABSTRACT

Drinking water treatment processes that use aluminum sulfate as a coagulant produce residuals streams with high levels of aluminum which must be treated before being discharged to the environment, and the treatment process produces a solids residuals stream that must subsequently be managed and disposed of. The goal of this thesis is to examine the effectiveness of a residuals treatment process currently in use at a WTP and explore options for optimizing treatment. Another objective was to characterize the solids stream residuals in order to identify potential options for use or disposal with consideration given to amendments that would improve any characteristics that may limit options for use or disposal and to identify the source or sources of elevated chromium in the solids residuals.

A tracer study was conducted to evaluate the hydraulic retention time in the treatment lagoons and determined that short circuiting was occurring. A sampling program was conducted to determine the effectiveness of existing treatment and found that the aluminum in the effluent from the treatment lagoons was primarily in particulate form and the concentration often exceeded 100 µg/L. Bench-scale settling tests were conducted with polymer addition and cation ratio adjustment to determine if treatment could be optimized. The addition of cationic, anionic and non-ionic polymers were all found to increase aluminum removal from CFBW. Cation ratio adjustment was not found to increase aluminum removal from the CFBW. The solids stream residuals were characterized to assist with evaluating potential use or disposal options, and metals concentrations in the solids residuals exceeded CCME commercial and industrial soil quality guidelines for arsenic, chromium, and selenium. The solids stream residuals were blended with blast furnace slag to determine whether amending the residuals with slag could reduce the leachability of metals. It was found to reduce the level of manganese that was leached from the soil. A sampling program was conducted to identify the source or sources of chromium in the solids stream residuals. The aluminum sulfate was calculated to be an insignificant contributor to the chromium load in the sludge. As a significant volume of raw water is processed through the plant, an annual average chromium concentration as low as 0.86 µg/L could account for the remainder of the chromium load in the sludge.

LIST OF ABBREVIATIONS AND SYMBOLS USED

µg	microgram
°C	degrees Celsius
%	percent
Al	aluminum
$Al(OH)_3$	aluminum hydroxide
Alum	aluminum sulfate
BCMOE	British Columbia Ministry of Environment
Ca	calcium
$CaCO_3$	calcium carbonate
$CaCl_2$	calcium chloride
CCME	Canadian Council of Ministers of the Environment
CO_2	carbon dioxide
Cr	chromium
CSTR	continuously stirred tank reactor
CSQG	Canadian soil quality guidelines
DAF	dissolved air flotation
FBW	filter backwash water
FWAL	freshwater aquatic life
hr	hour
ICP-MS	inductively coupled plasma mass spectrometer
IGPM	imperial gallons per minute
K	potassium
$KMnO_4$	potassium permanganate
kg	kilogram
L	litres
m	metre
m^3	cubic metres
meq	milliequivalent
Mg	magnesium

$MgCO_3$	magnesium carbonate
mg	milligrams
mL	millilitre
Na	sodium
NOM	natural organic matter
NSE	Nova Scotia Environment
NTU	nephelometric turbidity unit
PAC	polyaluminum chloride
RPM	revolutions per minute
RTD	residence time distribution
RWT	rhodamine wt
s	second
SPLP	synthetic precipitation leaching procedure
TSS	total suspended solids
USGPM	united states gallons per minute

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CHAPTER 1 INTRODUCTION

Drinking water treatment residuals are waste streams that are produced during the drinking water treatment process. They are comprised of any impurities that are removed during the water treatment process, chemicals that are added during treatment, and water. Drinking water treatment residuals can include organic and inorganic solids, algae, bacteria, viruses, colloids, dissolved salts, and water treatment chemicals (Crittenden et al., 2012). Improvements in technology can result in more matter being removed from the raw water during the treatment process and an increase in chemicals used during the treatment process which result in an increase in residuals that must be managed. In conventional treatment processes, drinking water treatment residuals include filter backwash water (FBW), filter-to-waste water, and sludge from clarifiers. These residuals streams can amount to 3 to 5 % of the total volume of raw water treated by the plant (Crittenden et al., 2012). FBW generally comprises the largest component of the residuals stream (2 to 5 % of plant flow) but contains less than 10% of the removed solids; clarifier sludge generally comprises only 0.1 to 0.3 % of plant flow but contains the majority of the removed solids (Crittenden et al., 2012). FBW is generally characterized as having a total suspended solids (TSS) concentration ranging from 100 to 1000 mg/L, an initial settling velocity of between 0.06 and 0.15 m/hr and a pH between 7.2 – 7.8 (Crittenden et al., 2012).

Historically, water treatment plant residuals were discharged to surface water without treatment (Crittenden et al., 2012). Options for management of water treatment plant residuals include direct discharge to sanitary sewer or a treatment process that involves solid liquid separation (Crittenden et al., 2012). After treatment, the clarified liquid stream can be discharged to surface water or a sanitary sewer, and the solids stream can be monofilled, landfilled, or land applied. Water treatment plant residuals have also been recycled to the source water or the head of the plant; however, there are concerns related to the concentrated impurities contained in the residuals.

One common issue with residuals management at conventional drinking water treatment plants that use an aluminum based coagulant include the adequate removal of aluminum prior to discharge of the liquid stream to surface water. The aluminum originates from the addition of a coagulant to the process (aluminum sulfate or polyaluminum chloride). Another common issue relates to disposal options for the sludge that is produced. Water treatment plants produce a large quantity of sludge that must be disposed of. Landfill disposal uses up volume in the landfill and many communities are trying to decrease the amount of material that is disposed of in landfills. There may be environmental concerns associated with land application of water treatment plant sludge due to the composition of the sludge and its leaching properties. The use of sludge as daily cover in a landfill may represent a beneficial reuse option that reduces the need to use clean soil as daily cover while recycling a material that would otherwise be a waste product.

1.1 RESEARCH RATIONALE

A water treatment plant (WTP) in Nova Scotia produces residuals that must be treated, including FBW and clarifier sludge. Combined, these flows are referred to as combined filter backwash water (CFBW) and amount to more than 10% of the total water treated at the plant. Residuals treatment at the plant includes sedimentation in two parallel treatment lagoons after which the clarified liquid stream is discharged to a brook, downstream of the raw water intake. Samples collected by plant operations staff in 2014 indicated that the aluminum concentration in the discharge from the lagoons ranged from 13.05 μg/L to 354 μg/L with an average of 120 μg/L. The Canadian Council of Ministers of the Environment (CCME) fresh water aquatic life (FWAL) criterion for aluminum is 100 μg/L if the pH is greater than or equal to 6.5 (CCME, 1999b). There is no criterion for marine waters. The downstream portion of the brook where the treatment lagoons discharge has tidal influence from the Atlantic Ocean and may be considered a brackish environment.

The plant has been limited in options for disposal of the sludge due to the presence of metals in exceedance of the CCME soil quality guidelines, in particular for chromium which has a criterion of 64 mg/kg for residential properties and 87 mg/kg for commercial or industrial properties (CCME, 1999c).

1.2 RESEARCH OBJECTIVES

The principle objective of this research is to study the management of residuals at a drinking water treatment plant in Atlantic Canada in order to identify areas where there is opportunity for improvement, and evaluate potential methods for improvement. This research has been broken down into 4 sub-objectives:

1. **Evaluate existing liquid stream residuals treatment**.
   This task included a study of the hydraulic retention time in the lagoons. A sampling program was also conducted to evaluate aluminum removal in the lagoons including whether the aluminum concentration in the discharge is in dissolved or particulate form and whether it varies with time as a result of the backwash cycle.
2. **Optimization of liquid stream residuals treatment.**
   This task comprised a series of bench-scale settling tests in order to determine whether the CFBW could be amended by chemical addition in order to improve aluminum removal in the existing lagoons.
3. **Characterize solids residual (sludge) in order to evaluate potential uses or disposal options.**
   The solids residual was characterized, including solids content, metals content and leaching characteristics in order to assist with identifying potential options for use or disposal. In addition, amendments were considered in order to improve characteristics of the solids residual that could limit disposal or use options such as leachability.
4. **Conduct investigation to determine source of elevated chromium in the solids residual.**
   An elevated level of chromium that exceeds CCME soil quality guidelines has previously been identified in the solids residual by plant operations staff and has limited reuse and disposal options. A sampling program was conducted in order to identify the source or sources of chromium.

CHAPTER 2 LITERATURE REVIEW

A literature review was performed of the characteristics of different residuals streams, treatment of drinking water residuals, solids residual disposal, and drinking water residuals regulations in Nova Scotia.

2.1 DRINKING WATER RESIDUALS

Drinking water treatment residuals are waste products that are produced during the drinking water treatment process. They are comprised of matter that is removed during the water treatment process and the water that transports it and include impurities that are found in the raw water such as organic and inorganic solids, algae, bacteria, viruses, colloids, dissolved salts as well as chemicals that are added as part of the treatment process (Crittenden et al., 2012). In a conventional treatment plant, residuals include filter backwash water (FBW), filter-to-waste water, and sludge from the clarifiers; the volume of residuals that are produced can be as high as 3 to 5 percent of the raw water volume (Crittenden et al., 2012). Filter backwash water makes up the largest portion of the residuals (Crittenden et al., 2012).

2.1.1 Filter Backwash Water

Filter backwash water (FBW) is a waste stream that is produced when filters are backwashed. It is comprised of water and any material that is removed by the filter and is typically high in suspended solids. The concentration of suspended solids in the FBW is dependent on the efficiency of treatment processes upstream of the filters such as coagulation/flocculation, clarification, the efficiency of the filtration process, and the volume of water used to backwash the filters (CBCL, 2004). The time between filter backwashes generally ranges from 24 to 72 hours (Crittenden et al., 2012). The backwash cycle of filters results in high volume intermittent flows of FBW. FBW generally comprises the largest component of the residuals stream (2 to 5 % of plant flow) but contains less than 10% of the removed solids (Crittenden et al., 2012). FBW is generally characterized as having a total suspended solids (TSS) concentration ranging from 100 to 1000 mg/L, an initial settling velocity of between 0.06 and 0.15 m/h and a pH between 7.2 – 7.8 (Crittenden et al., 2012). In Atlantic Canada treatment plants, the suspended solids in the FBW can be characterized as light and slow to settle due to a high organic content in the metal hydroxide flocs (CBCL, 2004).

2.1.2 Filter-to-waste Water

After the filters have been backwashed, the flow reverses to produce treated water. The water that is initially produced by a filter after it has been backwashed can be high in total suspended solids (TSS). For this reason, the water that is initially produced after backwash is usually treated as a residual or waste product and disposed of.

2.1.3 Clarifier Sludge

In conventional treatment plants, flocs generated during coagulation/flocculation are removed from the water in clarifiers by the process of sedimentation. Flocs settle to the bottom of the clarifier where they are continuously removed. The sludge that is produced is made up of settled floc containing concentrated impurities from the raw water as well as chemicals that have been added as coagulants and flocculant aids. Clarifier sludge generally comprises only 0.1 to 0.3 % of plant flow but contains the majority of the removed solids (Crittenden et al., 2012).

2.2 DRINKING WATER RESIDUALS TREATMENT

Options for management of drinking water treatment residuals include direct discharge to a sanitary sewer or a treatment process that involves separating the residuals into a solid and liquid stream prior to discharge; the liquid stream is then either discharged to a sanitary sewer or surface water body, and the solids stream is either landfilled, monofilled, or land applied.

2.2.1 Treatment Options

Some common unit operations that are used in the treatment of residuals (Crittenden et al., 2012) are:

- Flow equalization to mitigate the impact of high volume intermittent FBW flows to surface water or downstream unit operations;
- Chemical addition (typically polymer) to promote the removal of particulate and/or dewatering of solids;
- Settling ponds/lagoons, sedimentation, dissolved air flotation (DAF), thickeners, membrane filtration, granular filtration, can be used for solid liquid separation;
- Centrifuges, vacuum filters, filter press, drying beds can be used for dewatering solids residual;

- The liquid stream can be recycled to the source water or the head of the plant, or discharged to surface water or sanitary sewer for disposal;
- The solids stream can be monofilled, landfilled or land applied for disposal.

2.2.2 Treatment in Nova Scotia

A summary of the residuals treatment processes at selected conventional, DAF, and direct filtration plants in Nova Scotia is provided below:

- Glace Bay Water Treatment Plant - conventional treatment
  FBW, filter-to-waste water and clarifier sludge are directed to two parallel treatment lagoons where sedimentation occurs. The clarified liquid stream is discharged to McAskill's Brook downstream of the source water reservoir. Once yearly, the solids residual is pumped from the lagoons into a drying bed. In the past, solids residual from the drying bed has been blended with soil and disposed of in the municipal landfill.
- New Waterford Water Treatment Plant - dissolved air flotation (DAF) treatment
  All residuals produced at the plant which include filter backwash water and sludge from the DAF tanks, are discharged directly to the sanitary sewer.
- J. Douglas Kline Water Treatment Plant - direct filtration treatment
  FBW is directed to two parallel engineered treatment lagoons where sedimentation occurs followed by a natural wetland. The clarified liquid stream is discharged to Little Pockwock Lake. The solids residual from the lagoons is pumped to drying beds.
- Lake Major Water Treatment Plant - conventional treatment
  Clarifier sludge and FBW flows to a mixed equalization tank followed by thickeners where cationic polymer is added. Sludge from the thickeners is pumped to a centrifuge for dewatering and additional cationic polymer is added. The liquid stream from the centrifuge is recycled back to the equalization tank. The solids residual from the centrifuge is transported by truck to a drying bed. Supernatant from the thickener and effluent from the drying bed is discharged back to Lake Major after the chlorine has been neutralized with Vita-d-chlor.
- Windsor Water Treatment Plant - dissolved air flotation (DAF) treatment

Cationic polymer is added to the FBW in the FBW holding tank at a concentration of approximately 6.25 µL of polymer per litre of FBW. FBW is then pumped from the holding tank to two treatment lagoons in series. The clarified liquid stream from the lagoons is discharged to Falls Brook. The solids residual from the lagoons is dewatered in geotubes and stored onsite.

2.3 RESIDUALS TREATMENT CONSIDERATIONS

A common issue with residuals management involves ensuring adequate treatment of the liquid stream prior to discharge. Adequate treatment of the liquid stream can be affected by the water chemistry as pHs that are much higher or lower than the minimum solubility point of aluminum will cause the aluminum to be in dissolved form. It can also be affected by the nature of the solids in the filter backwash water. Small or light particles can be slow to settle or inefficient to remove through sedimentation. In addition, less than optimal hydraulics in settling ponds/ lagoons can lead to short circuiting which reduces the hydraulic retention time of water in the ponds. During the design phase for treatment lagoons, an assumption is made regarding the solids content of the solids residual. If the lagoons are producing solids residual with a lower solids content than assumed, this can cause the lagoons to fill up with solids residual much faster than intended. For instance, if a design assumption was made that the solids residual would have a 3% solids content and the lagoons are producing solids residual with 1% solids content, the lagoons will fill up with solids residual 3 times faster than intended by the designer.

2.3.1 Settling Pond Hydraulics

In practice, settling ponds or lagoons do not operate at their theoretical hydraulic retention time. The hydraulics of settling ponds/ lagoons are affected by short circuiting due to dead zones, solids residual accumulation, unsteady inflow, and wind. The most common cause of poor performance in lagoons is short circuiting (NRC, 2004). Some methods of improving the flow distribution in lagoons include installing baffles and relocating or reconfiguring the inlet and outlet. A tracer test can be used to experimentally determine the amount of short circuiting occurring in a lagoon or settling pond.

The ideal settling pond hydraulics would follow a plug flow residence pattern (Thackson et al., 1987). However, settling pond hydraulics do not follow the ideal conditions of either a plug flow reactor or continuously stirred tank reactor (CSTR) but rather fall somewhere in between. Ideal CSTR flow involves the flow being thoroughly mixed upon entering the lagoon and the concentration at the outlet being equal to the concentration at any point in the lagoon. Ideal plug flow involves zero mixing in the longitudinal direction of flow; all atoms exiting the system have been inside the system for an identical length of time (Fogler, 2006). Figure 1 provides the E(Θ) curve for CSTRs in series (for 2, 4, 10 and ∞ CSTRs). As the number of CSTRs in series increases, the curve approaches that of an ideal plug flow reactor with the curve for ∞ CSTRs in series being equivalent to an ideal plug flow reactor. The E(Θ) curve is the residence time distribution (RTD) function, with Θ being equal to time divided by the theoretical retention time of the system. The area under the curve between any two points represents the fraction of matter leaving the system that has resided in the system for an amount of time between those two points (Fogler, 2006).

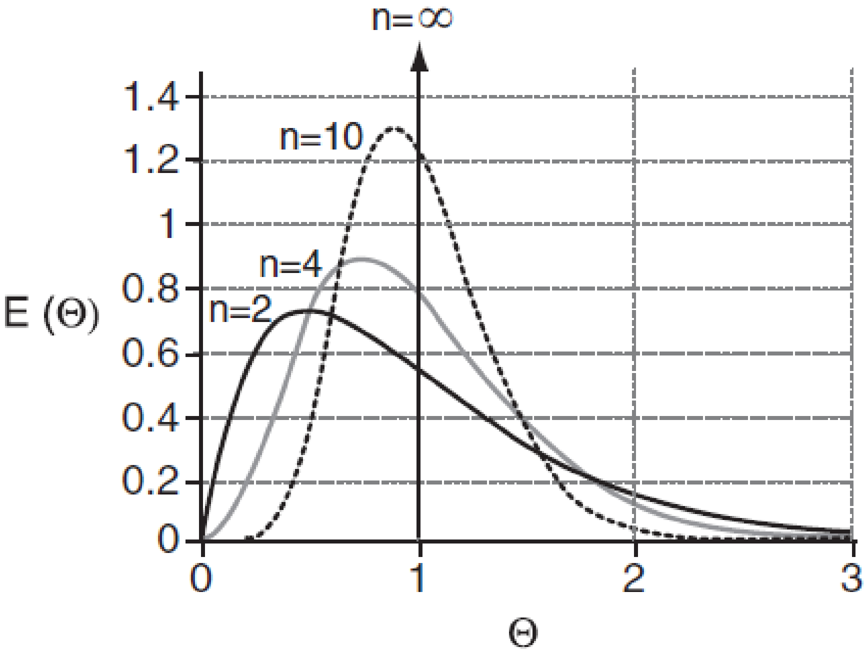

Figure 1 Theoretical Residence Time Distribution of CSTRs in Series (from Fogler, 2006)

The hydraulic efficiency of a settling pond is expressed as the mean residence time divided by the theoretical residence time, and a hydraulic efficiency of less than 0.3 or 0.4 represents short circuiting (Thackson et al., 1987).

2.3.2 Aluminum Chemistry

Aluminum is added to the treatment process as aluminum sulfate (alum) to act as a coagulant. The most common form of alum that is used as a coagulant is $Al_2(SO_4)_3 \cdot 14H_2O$ (Crittenden et al., 2012). Alum is a metal salt and its purpose is to destabilize particles in the water. Aluminum has a unique solubility curve in that it becomes increasing soluble at both high and low pHs. Gensemer and Playle (1999) state that aluminum is relatively insoluble between pH 6.0 and 8.0; their finding was based on Figure 2, which was reproduced from Schecher and McAvory (1992) who conducted a simulation of aluminum speciation at 15°C. Driscoll and Letterman (1995) reference a pH of 6.5 as the point of minimum solubility of aluminum. The solubility of aluminum increases with decreasing temperature (Driscoll and Postek, 1996). Aluminum hydroxide solid $Al(OH)_3$ is the predominant species of aluminum in the pH range of 6 to 8 (Gensemer and Playle, 1999).

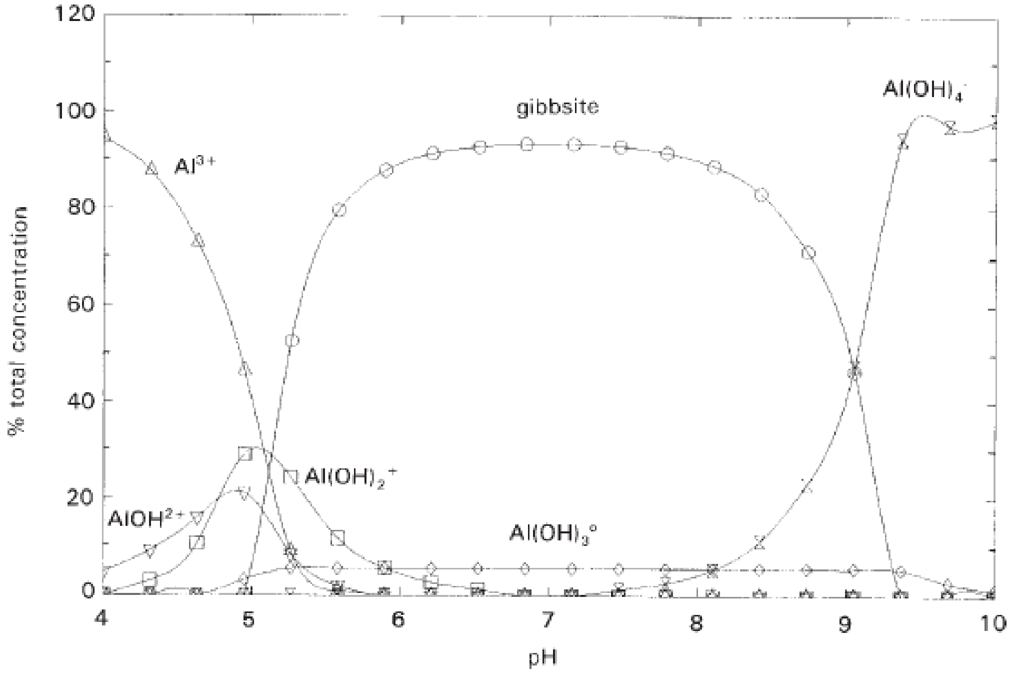

Figure 2 Solubility Diagram for Aluminum (Source: Schecher and McAvory, 1992)

The solids residual produced at drinking water treatment plants that employ alum or poly aluminum chloride (PACl) as a coagulant are generally very high in aluminum. An analysis of alum water treatment plant solids residual ranging from fresh to two years old by Agyin-Birikorang and O'Connor (2009) found the aluminum concentration of fresh and two year old solids residual to be 152+/-20.1 g/kg and 137+/- 1.59 g/kg, respectively. The pH of fresh solids residual was 5.12±0.22 while the pH of two year old solids residual was 5.94±0.45. There has been significant research regarding the use of aluminum to amend soils including the use of water treatment residuals containing aluminum (Udeigwe et al., 2011). Aluminum has been shown to reduce the mobilization of phosphorous, zinc, copper, lead, arsenic, DOC and pathogens (Udeigwe et al., 2011).

2.3.2.1 Published Aluminum Water Quality Criteria

The Canadian Council of Ministers of the Environment (CCME) have established the Canadian Water Quality Guidelines for the Protection of Aquatic Life which includes an aluminum guideline for freshwater of 5 μg/L if the pH is less than 6.5 and 100 μg/L if the pH is greater than or equal to 6.5 (CCME, 1999b). The guideline is based on the aluminum guideline originally published in the Canadian Water Quality Guidelines (CCREM 1987). There is no aluminum guideline provided for marine waters. The Health Canada guidelines for Canadian Drinking Water Quality include an operational guidance value of 100 μg/L for conventional treatment and 200 μg/L for other treatment types (Health Canada, 2014). The United States (US) Environmental Protection Agency (EPA) National Recommended Water Quality Aquatic Life Criteria for aluminum in freshwater with a pH of between 6.5 and 9.0 is 750 μg/L as an acute guideline (1 hour average) and 87 μg/L as a chronic guideline (4 day average). There is no saltwater guideline provided.

2.3.3 Settling Efficiency

Settling and sedimentation are interchangeable terms that refer to a physical process where suspended particles that are heavier than water are removed from solution by gravitational settling (Metcalf and Eddy, 2014).The aim of sedimentation is to produce a sufficiently clarified liquid stream and a solids residual with a high solids content (Boon and Dolan, 1998). Factors that affect the efficiency of particle removal by settling include particle size, particle sphericity, particle density, and viscosity of the fluid. Two important factors that determine the efficiency of settling are the rate of sedimentation and the solids concentration of the sediment (Rhodes, 2008). Repulsive particles take longer to settle but produce a sediment that contains a lower water content, and attractive particles settle more quickly but produce a sediment that contains a higher water content, as shown in Figure 3.

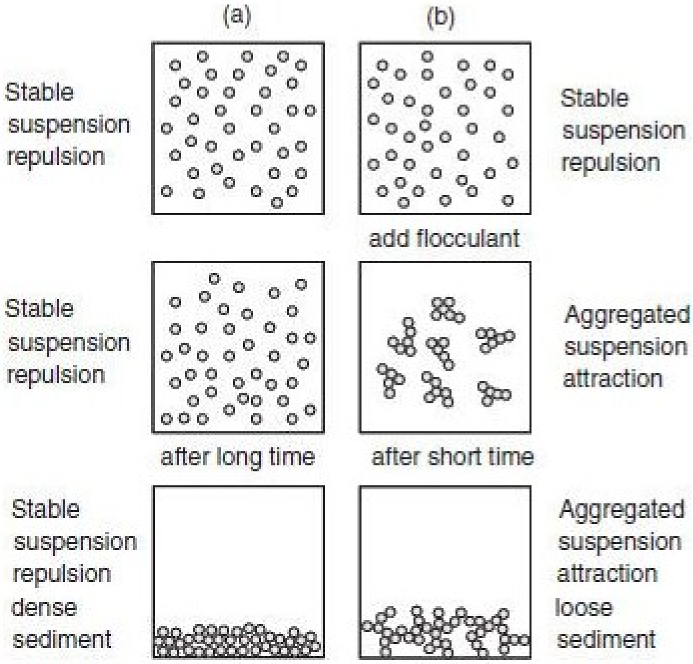

Figure 3 Settling of Repulsive and Attractive Particles (Source: Rhodes, 2008)

A British Columbia Ministry of Environment publication (BCMOE, 2001) states that the smallest particle that can be removed by plain sedimentation has a particle diameter of 0.005 to 0.01 mm. Smaller particles can be removed through sedimentation with the aid of coagulation and flocculation. Coagulation is a process by which particles are destabilized by eliminating the surface charge on a particle through addition of a chemical. The duration of the coagulation process is typically less than 10 seconds. Polymers can also be added as a coagulant aid to improve the coagulation process. Flocculation is a process by which particles that have been destabilized by coagulation are aggregated into larger particles known as floc for increased settling efficiency. In addition, these larger particles fall on smaller particles during settling and effectively sweep them from solution which results in the removal of small or light particles that would otherwise settle very slowly. The duration of the flocculation process ranges from 20 to 45 minutes. (Crittenden et al., 2012).

Crittenden et al. (2012) provides a range of settling velocities for different sizes of aluminum floc in water at 15 °C which has been reproduced in Table 1.

Table 1 Settling Velocity of Aluminum Floc in Water

Floc Type	Settling Velocity at 15 °C (m/h)
Small fragile alum floc	2 – 4
Medium-sized alum floc	3 – 5
Large alum floc	4 – 5.5

Issues that affect sedimentation efficiency include short circuiting, temperature gradients, wind effects, inlet energy dissipation and outlet currents (Crittenden et al., 2012). Two principal factors in the effectiveness of settling ponds/ lagoons are the avoidance of short circuiting, and the avoidance of disturbing previously settled solids (Boon and Dolan, 1995). Bourgeois et al. (2004b) found that the optimum retention time for thickening of water treatment plant residuals was between 0.8 and 1 day. Beyond 1 day, the metal concentrations increased due to resolubilization of metals. However, it is unclear if this applies to aluminum or only to iron and manganese as reducing conditions could cause the resolubilization of iron and manganese, but aluminum has only one valence state. CBCL (2004) indicates that lagoons that are used for treatment of water treatment plant residuals should provide a retention time of between 15 and 30 days.

2.3.4 Polymer Addition to Improve Settling

Polymers are “long-chain molecules consisting of repeating chemical units with a structure designed to provide distinctive physicochemical properties” (Crittenden et al., 2012). They are also sometimes referred to as polyelectrolytes. Synthetic organic polymers come in anionic, cationic or non-ionic types and are also often classified by molecular weight.

Polymers can be used to improve settling of particles. This is done by two mechanisms: they can act as a coagulant and destabilize particles, and they can promote the formation of larger flocs that are more resistant to shear by forming a bridge between particles (Crittenden et al., 2012). Polymers can also be used to improve dewatering of solids residual. However, they also add mass to the solids residual.

Polymer selection, dose, and mixing conditions are typically based on empirical testing. For solids residual conditioning, the polymer selection is dependent on the properties of the solids residual, the properties of the polymer, and the mixing environment (Crittenden et al., 2012). Polymer bridging is the primary mechanism involved in solids residual conditioning. Higher molecular weight polymers are generally more effective for solids residual conditioning (Crittenden et al., 2012). Crittenden et al. (2012) provides the following notes on the effect of different dosing conditions for polymers (Table 2) and the effect of different mixing conditions for polymers (Table 3):

Table 2 Effect of Polymer Dose Conditions (from Crittenden et al., 2012)

Polymer Dose	**Effect**
Too low	Nonadsorbed polymer ends will adsorb on destabilized particle, causing it to restabilize
Optimum	Particles are destabilized and can flocculate
Too high	All adsorption sites will be used and particle will not flocculate

Table 3 Effect of Polymer Mixing Conditions (from Crittenden et al., 2012)

Mixing Conditions	**Effect**
Inadequate	Nonadsorbed polymer ends will adsorb on destabilized particle, causing it to restabilize
Too long / intense	Floc will break up

Results of a number of studies that looked at polymer conditioning of water treatment plant residuals are summarized in Table 4 and described further below.

Table 4 Summary of Polymer Conditioning Studies

Study	Finding
Cornwall and Lee (1994)	Non-ionic polymer increased removal of giardia sized particles from FBW using sedimentation.
Arora et al. (2001)	0.5 mg/L dose of both cationic and anionic polymers improved particle and turbidity removal during settling.
Wood (2014)	Optimum aluminum removal from FBW via settling was determined to be achieved with a 5 mg/L dose of medium molecular weight polymer and 1.5 hours of settling time.
Follett (2012)	The addition of cationic polymer upstream of filters reduced ripening time and maximum turbidity during ripening.
Zhao (2003)	Anionic polymer Magnafloc LT25 substantially increased floc size in alum sludge but floc size eventually plateaued with increasing dose.
Zhao (2004a)	Increasing dose of anionic polymer Magnafloc LT25 increased rate of settling until optimum dose was reached. The optimum polymer dose was associated with the minimum supernatant viscosity.
Ma et al. (2007)	There was no considerable improvement in sludge dewaterability using both anionic and cationic polymer compared to using only one type.

Cornwall and Lee (1994) found that the addition of non-ionic polymer increased removal of giardia sized particles from FBW using sedimentation. Arora et al. (2001) found that both cationic and anionic polymers at a dose of 0.5 mg/L were effective at improving turbidity removal and particle removal during settling. Jar tests conducted by Wood (2014) determined optimal aluminum removal from FBW via settling with a 5 mg/L dose of cationic medium molecular weight polymer and a settling time of 1.5 hours. Under these conditions, the aluminum in the FBW was reduced to 101 µg/L.

Follett (2012) conducted bench-scale tests to determine the impact of cationic polymer addition to FBW upstream of the filters at a plant where filter-to-waste infrastructure was not available and found that it reduced ripening time and the maximum turbidity during ripening.

Zhao (2003) found that a substantial increase in floc size resulted from the addition of the anionic polymer Magnafloc LT25 to alum solids residual; however, the floc size plateaued at a certain polymer dose. Polymer doses in the study ranged from 0 to 30 mg/L and experimental conditions involved a 30s rapid mix period followed by a 1 minute slow mix period. 10 mg/L was found to be the optimum dose for obtaining the lowest specific resistance to filtration. However, dewatering tests using an air pressure plate apparatus showed no significant change in dewatering extent with polymer dose. The study also found that the degree of compactness of the floc was insensitive to the polymer dose.

Zhao (2004a) used the minimum interfacial height of sediment after 30 minutes of settling in 100 mL measuring cylinders to determine an optimum polymer dose. Anionic polymer Magnafloc LT25 was the polymer used in the study. Polymer doses ranging from 2 to 30 mg/L were used and experimental conditions involved a 30s rapid mix period followed by a 1 minute slow mix period. Polymer doses of greater than 15 mg/L were found to result in rapid settling of floc and a coarsely structured matrix of solids residual. The height of the solids residual layer decreased with time due to compression. For polymer doses under the optimum dose of 10 mg/L, increasing the polymer dose was shown to increase the rate of settling. The optimum polymer dose was also associated with a minimum viscosity of the supernatant. Supernatant viscosity decreased with increasing polymer dose until the optimum polymer dose which was attributed to a reduction in fine particles in the supernatant. Supernatant viscosity increased with increasing polymer dose above the optimum polymer dose which was attributed to excess polymer in the water. The amount of residual polymer in the supernatant remained constant until the optimum polymer dose, after which it increased rapidly with increasing polymer dose.

Ma et al. (2007) explored the conditioning of alum solids residual using a cationic polymer followed by an anionic polymer and also evaluated each polymer alone. The study found that there was no considerable improvement in solids residual dewaterability compared to using one type of polymer. This was attributed to the fact that for alum solids residual, interparticle bridging is thought to be the main factor in polymer conditioning rather than charge neutralization. The optimum dose of the cationic polymer was found to be 50 mg/L and the optimum dose of the anionic polymer was found to be 30 mg/L. The TSS of the solids residual in the study was 2985 mg/L.

2.3.5 Cation Ratio Adjustment to Improve Settling

Bourgeois et al. (2004a) conducted a study on the performance of sedimentation of drinking water residuals with optimal cation ratios. The study indicated that the quality of the settled water could be improved through the addition of divalent cations. The optimum ratio of monovalent to divalent cations for sedimentation was identified as being 0.33:1. Charge-neutralization precipitation was proposed as the mechanism by which settling was improved; the divalent cations were thought to destabilize the negatively charged color molecules within the FBW.

Trias et al. (2004) identified the optimization of cation ratios as a factor that impacts the settling behavior and dewaterability of solids residual. This study was based on the concept of a bridging action between particles due to divalent cations such as Ca^{2+} and Mg^{2+}. This study involved laboratory experiments to determine the impact that Ca^{2+} and Mg^{2+} ions have on settling and dewaterability of liquid hog manure. The results indicated that although settling and dewatering improved, the improvements were inconsistent and not statistically significant.

Higgins and Novak (1997) demonstrated that the activated sludge dewatering properties were correlated with the ratio of monovalent to divalent cations and calcium to magnesium. This study found that a ratio of monovalent to divalent cations greater than 2:1 resulted in poor dewatering. A ratio of 1:1 for monovalent to divalent cations and 1:1 for calcium to magnesium was suggested for optimum settling and dewatering. The study also concluded that cation addition can reduce the polymer demand by 30 – 75%.

Novak et al. (1998) studied an activated sludge treatment plant with regard to the impact that the cation content of the wastewater had on settling and dewatering. It was found that a ratio of monovalent to divalent cations of greater than 2:1 on a milliequivalent basis resulted in poor dewatering properties and increased polymer requirements. The study also showed that when the concentration of sodium and potassium were less than 10 and 0.1 meq/L, respectively, there was a substantial improvement in settling and dewatering; however, under these conditions the addition of magnesium did not further improve settling or dewatering.

Hundt and O'Melia (1988) found that the addition of calcium as $CaCl_2$ improved the removal of fulvic acid. Two possible explanations proposed by the authors included calcium reducing the charge on the fulvic acid making the molecule more hydrophobic, or a reduction in electrostatic repulsion between molecules at the oxide surface.

2.3.6 Impact of Mixing on Settling

The efficiency of particle removal by settling can be impacted by turbulence. Settling can be inhibited by turbulence due to the re-entrainment of particles that have already settled and by keeping particles suspended (Janssen, 2008). However, settling can also be improved by turbulence due to the increased collision between particles aiding flocculation.

As described in Section 2.3.4, inadequate mixing of polymer in the water will result in the nonadsorbed polymer ends adsorbing onto destabilized particles causing them to restabilize, and mixing that is too intense or for too long a duration will cause floc to break up (Crittenden et al., 2012). In addition, mixing can impact floc formation as mixing is one mechanism that causes contact between particles for them to flocculate (Crittenden et al., 2012).

2.4 SOLIDS RESIDUAL MANAGEMENT AND DISPOSAL CONSIDERATIONS

Other common issues with residuals management include adequate dewatering of the solids residual, and finding an appropriate reuse or disposal option for the solids residual. Use or disposal options of the solids residual can be limited by the characteristics of the solids residual including the presence of bacteria and high metals concentrations as well as the leaching characteristics of the solids residual. Disposal of water treatment plant solids residual in a landfill uses up volume in the landfill and many communities are trying to decrease the amount of material that is disposed of in landfills. The use of solids residual as daily cover in a landfill may represent a beneficial reuse option that reduces the need to use clean soil as daily cover while recycling a material that would otherwise be a waste product.

2.4.1 Solids Residual Chemistry

Drinking water treatment using aluminum sulfate (alum) or poly aluminum chloride (PAC) produces a solids residual with a high concentration of aluminum. The resulting aluminum concentration in the solids residual is dependent on the coagulant dose and the raw water characteristics. In addition to aluminum, the solids residual contains impurities that are removed during treatment such as organic matter, iron, manganese and bacteria. The solids content of the solids residual depends on the method that has been used for dewatering.

2.4.1.1 Published Soil Quality Criteria

The Canadian Council of Ministers of the Environment (CCME) publishes Canadian Soil Quality Guidelines (CSQG) for parkland, residential, commercial, and industrial sites for the protection of environmental and human health. The CSQG for metals are summarized in Table 5 (CCME, 1999c).

Table 5 CCME Soil Quality Guidelines

Parameter (Acid Extractable)	CCME Soil Quality Guidelines (mg/kg)		
	Residential/ parkland	Commercial	Industrial
Aluminum	-	-	-
Antimony	20	40	40
Arsenic	12	12	12
Barium	500	2000	2000
Beryllium	4	8	8
Bismuth	-	-	-
Boron	-	-	-
Cadmium	10	22	22
Calcium	-	-	-
Chromium	64	87	87
Cobalt	50	300	300
Copper	63	91	91
Iron	-	-	-
Lead	140	260	600
Lithium	-	-	-
Magnesium	-	-	-
Manganese	-	-	-
Mercury	6.6	24	50
Molybdenum	10	40	40
Nickel	50	50	50
Phosphorus	-	-	-
Potassium	-	-	-
Rubidium	-	-	-
Selenium	1	2.9	2.9
Silver	20	40	40
Sodium	-	-	-
Strontium	-	-	-
Thallium	1	1	1
Tin	50	300	300
Titanium	-	-	-
Uranium	23	33	300
Vanadium	130	130	130
Zinc	200	360	360

2.4.2 Chromium Chemistry

According to CCME (1999a) the chromium content in soil ranges from 10 to 100 mg/kg. Chromium can exist in 9 different oxidation states; however, trivalent and hexavalent chromium are the most common (CCME, 1999a). Trivalent chromium occurs naturally, but the primary source of hexavalent chromium is pollution. Sources of chromium include metal finishing, corrosion control, leather tanning, wood treatment, the production of pigments, the production of stainless steel, the manufacture of bricks, fossil fuel combustion, raw sewage, urban runoff, and pulp and paper production (CCME, 1999a; Pawlisz, 1997). Trivalent chromium is strongly adsorbed by clay particles, organic matter in soil, and by metal oxyhydroxides (CCME, 1999a). Conventional drinking water treatment removes trivalent chromium from water but does not remove hexavalent chromium (Pawlisz, 1997). Hexavalent chromium is highly soluble and does not readily adsorb onto soil particles (Pawlisz, 1997). Freshwater bodies in Canada generally contain less than 5 µg/L of chromium and Canadian soils generally contain between 10 and 100 mg/kg (Pawlisz, 1997). Hexavalent chromium is expected to be the dominant species in drinking water where oxidants such as chlorine or chloramine are present (McNeill et al., 2012). Trivalent chromium has a low solubility between pH 7 and 10 and the minimum solubility of trivalent chromium of approximately 1 µg/L occurs at pH 8 (Rai et al., 1987; McNeill et al., 2012). McNeill et al. (2012) identified five forms in which trivalent chromium can be present in surface water:

- as soluble trivalent chromium;
- as $Cr(OH)_3$ solid;
- sorbed to the surface of $Fe(OH)_3$ and other oxides;
- fixed inside oxides; and
- complexed with NOM.

Hexavalent chromium, on the other hand, is most likely to be present in a soluble form (McNeill et al., 2012).

A study of the 100 largest drinking water sources in the US by the US Geological Survey in 1962 found that the total chromium concentration in the source water ranged from less than the detection limit to 35 µg/L with a median concentration of 0.43 µg/L (McNeill et al., 2012).

McNeill et al. (2012) suggested that chromium may be a trace contaminant in chemicals used in water treatment. A study by Eyring et al. (2002) noted that alum used at a water treatment plant in Durham, N.C., contained 9.5 mg/L of chromium (Eyring et al., 2002; Bishop et al., 1987). Eyring et al. (2002) also analyzed 9 samples of alum from 3 different suppliers for metal impurities; all 9 samples had chromium results less than 4 mg/L with 8 of the results less than 2 mg/L. There was variability in the concentration of impurities noted between supplies and between batches from the same supplier. The Committee on Water Treatment Chemicals (1982) recommended a maximum impurity content for chromium in aluminum sulfate as 30 mg/kg. A news release by the Missouri Department of Natural Resources in 2010 noted that a sample of treated water contained 0.6 µg/L of hexavalent chromium while the raw water contained 0.1 µg/L (MDNR, 2010). According to McNeill et al. (2012) the suspected source of the increase in hexavalent chromium was the lime or alum that was added during the treatment process. McNeill et al. (2012) also noted that other sources of chromium in drinking water could be the result of leaching from or reaction with distribution system materials such as cast iron, cement, or stainless steel.

Coagulation and filtration, ion exchange, adsorption and membrane filtration are treatment options that can be used to remove chromium during drinking water treatment; however, coagulation and filtration is the most common and effective method of chromium removal (Sharma et al., 2008).

2.4.2.1 Published Chromium Water Quality Criteria

The Canadian Council of Ministers of the Environment (CCME) Canadian Water Quality Guidelines for the Protection of Aquatic Life include a chromium guideline for freshwater of 1 µg/L for hexavalent chromium and 8.9 µg/L for trivalent chromium and a chromium guideline for marine waters of 1.5 µg/L for hexavalent chromium and 56 µg/L for trivalent chromium (CCME, 1999b). The Health Canada guidelines for Canadian Drinking Water Quality include a maximum acceptable concentration of 50 µg/L (Health Canada, 2014). The US EPA National Recommended Water Quality Aquatic Life Criteria for freshwater for chromium is 11 µg/L for hexavalent chromium and 74 µg/L for trivalent chromium as a chronic guideline and 16 µg/L for hexavalent chromium and 570 µg/L for trivalent chromium as an acute guideline. There is no saltwater guideline provided for trivalent chromium. The saltwater guideline for hexavalent chromium is 1100 µg/L as an acute guideline and 50 µg/L as a chronic guideline.

2.4.3 Slag as a Source of Alkalinity

Blast furnace slag is a by-product of the steel industry and is in abundant supply in Sydney, NS. CBCL completed a study in 2005 of potential slag applications (CBCL, 2005). This study described blast-furnace slag as being mildly alkaline with a pH in solution of between 8 and 10. Slag was identified as an effective alkalinity source to treat acid mine water as it yields several hundred times more alkalinity than limestone. The study referenced a comparison by Conestoga Rovers & Associates (2001) of the metals content of the slag to CCME guidelines which identified exceedances for selenium and beryllium but concluded that the use of slag would not pose an unacceptable health risk to the public in residential settings. The study also included the results of leachate testing of the slag that was performed by Jacques Whitford in 2005.

According to the National Slag Association (2003), the composition of slag consists of oxides of calcium, iron, silicon, aluminum, magnesium and manganese in complexes of calcium silicates, aluminosilicates and aluminoferrite with calcium oxide and silicon dioxide making up the largest portion. Uses of slag include concrete, parking lots, walkways, driveways, production of Portland cement, soil remineralization, pH supplement or liming agent, and treating acidic runoff (National Slag Association, 2003).

2.4.4 Soil Amendments to Improve Leaching Characteristics

Soil pH is one factor that can control the mobilization of contaminants (Udeigwe et al., 2011). The solubility of many metals is dependent on the pH of the solution. The synthetic precipitation leaching procedure (SPLP) test approximates leaching that would occur as a result of slightly acidic rainwater. The addition of alkalinity to the solids residual could neutralize acidity in the rainwater and reduce the leaching of metals from the solids residual. An experiment where lime was added to soils showed that there was a relationship between exchangeable aluminum and the soil pH; a higher pH was associated with lower exchangeable aluminum (Belkacem and Nys, 1997). A study by Strand et al. (2002) that compared leaching results from soils exposed to simulated “normal rain” at a pH of 5.3 to those exposed to simulated “acid rain” at a pH of 4.3 found that more aluminum was leached from soils receiving the simulated “acid rain”.

The application of crushed limestone is a common method of reducing the acidity of soil, and the application of gypsum and dolomite can also be used (Belkacem and Nys, 1997). Slag is a waste product that is produced during the production of steel and is in abundant supply in Cape Breton. Slag can be used as a source of alkalinity for soil or water (CBCL, 2005; National Slag Association, 2003).

2.5 DRINKING WATER RESIDUALS REGULATION IN NOVA SCOTIA

The Nova Scotia Environment (NSE) document “Nova Scotia Treatment Standards for Municipal Drinking Water Systems” was approved on March 12, 2012. The document states that its purpose is to “set out the minimum requirements that apply to municipal drinking water systems in Nova Scotia” (NSE, 2012). There is a section in this report on the management of waste systems that discuss requirements for the management of filter backwash water and filter backwash and clarifier solids. The document requires that filter backwash water be discharged to an approved location and cannot be discharged to the raw water intake pumps or inlet structures or upstream of the raw water intake (NSE, 2012).

For discharge to a freshwater watercourse, the Treatment Standards require:

- TSS not to exceed 5 mg/L over naturally occurring clear flow background watercourse concentration which is to be calculated as the 90th percentile value from a minimum of 12 monthly clear flow samples;
- Chlorine residual shall not exceed 0.02 mg/L;
- pH to be between 6.5 and 9.0; and
- Discharge to be non-acutely lethal as per “Reference Method for Determining Acute Lethality to Rainbow Trout” (NSE, 2012).

In addition, the Treatment Standards (NSE, 2012) state that the following options may be considered for setting discharge criteria limits for metals:

- Meet the values set by Canadian Council of Ministers of the Environment (CCME) Canadian Water Quality Guidelines for the Protection of Aquatic Life;
- If naturally occurring background concentrations are higher than the values set by the above guideline, NSE may allow discharge criteria limits to be set at the 90th percentile of the watercourse's background concentrations. This approach requires a minimum of 12 monthly samples from the watercourse to establish background concentrations;
- If the above requirement is not attainable, NSE may allow a 10 percent increase above the 90th percentile;
- If the above requirement is not attainable, a study shall be completed to recommend "end of pipe" discharge criteria limits.

The discharge criteria limits for metals must be met at the end of the pipe in 95% of samples, and sampling frequency shall be at least monthly or as required by NSE (NSE, 2012).

For discharge to a municipal wastewater system, the Treatment Standards require:

- The utility must ensure that there is capacity in the wastewater system; and
- The utility must contact NSE to see what other requirements must be met (NSE, 2012).

For discharge to a marine or brackish environment, the Treatment Standards require:

- The utility must contact NSE to see what other requirements must be met (NSE, 2012).

Treatment Standards also require that filter backwash and clarifier solids be disposed in accordance with a solids disposal plan that has been prepared by the municipal water utility and is acceptable to NSE (NSE, 2012).

CHAPTER 3 BACKGROUND

3.1 SITE OVERVIEW

The WTP has been in operation since 2003 and produces 9.4 million litres of water per day in order to serve a population of 23,760 (CBRMa). The design treatment capacity is 17.7 million litres per day gross capacity and 15.9 million litres per day net capacity (Dillon, 2003). Treatment at the plant includes filtration and clarification for organics, turbidity, manganese, and iron.

3.1.1 Source Water

Source water for the Treatment Plant is supplied from a man-made reservoir that was formed by a dam on a brook. It was previously established as a water supply for an industrial operation. The reservoir has been characterized as having similar water quality to a former source water supply that was used by the community but with higher concentrations of color, turbidity, iron, manganese and organic carbon; however, the reservoir has a safe yield of 20.5 million gallons per day compared to the 4.0 million gallon per day yield from the former source water supply (Dillon, 2003). The reservoir watershed covers an area of 3700 hectares (CBRMb). The intake was constructed in 1972/1973 and is comprised of a concrete intake with a 30” wood stave intake line (Dillon, 2003). An orthophotograph of the Reservoir is shown in Figure 4.

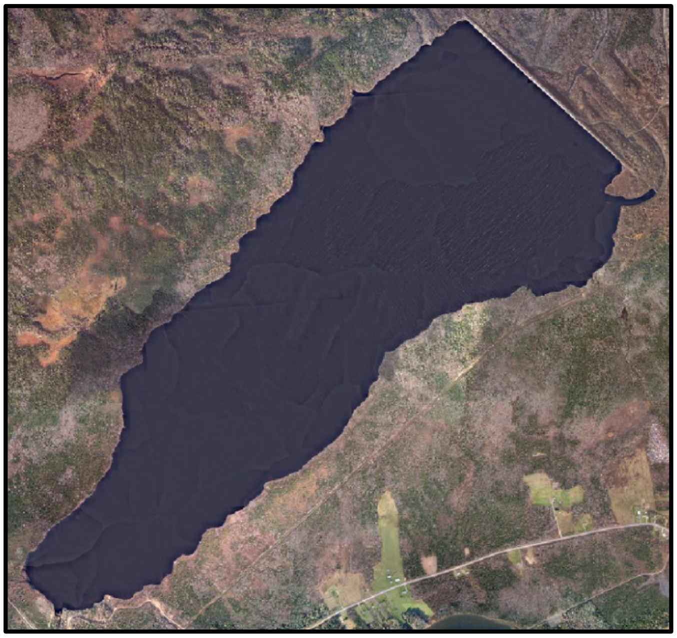

Figure 4 Orthophotograph of Reservoir

3.1.2 Treatment Process

Water from the reservoir is treated for organics, turbidity, manganese, iron, bacteria, viruses and protozoa. The treatment process includes a strainer, coagulation, flocculation, filtration and chlorination (Figure 5). Raw water first flows through a duplex strainer, followed by two pre-treatment tanks in series. Lime ($CaCO_3$) is added to the first pre-treatment tank to adjust the pH to approximately 11 for oxidation. Potassium permanganate ($KMnO_4$) is added to the first pre-treatment tank to oxidize iron and manganese so that they precipitate from solution. Carbon dioxide (CO_2) is added to the second pre-treatment tank to adjust the pH to approximately 6.9 for coagulation. Aluminum sulfate is added to the second pre-treatment tank as a coagulant and polymer is added as a coagulant aid. After leaving the second pre-treatment tank, water flows in parallel through two flocculation trains. Each flocculation train is comprised of three flocculation tanks in series. After flocculation, water flows through two sedimentation tanks in parallel with water from one flocculation train flowing through one sedimentation tank and water from the other flocculation train flowing through another sedimentation tank. After the water leaves the sedimentation tanks it flows into a distribution channel where it is distributed to three dual media filter tanks in parallel. Each filter tank contains anthracite, sand, support gravel, and filter block. After filtration, water flows through a chlorine contact chamber where chlorine is added for disinfection. The water then flows into two clear wells.

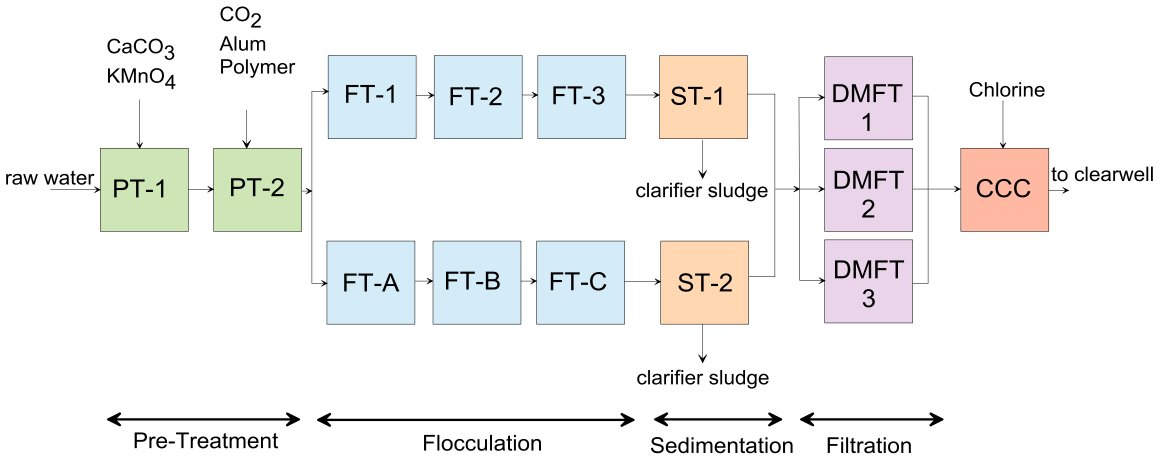

Figure 5 Treatment Process Block Flow Diagram

PT = Pre-treatment Tank

FT = Flocculation Tank

ST = Sedimentation Tank

DMFT = Dual Media Filter Tank

CCC = Chlorine Contact Chamber

3.1.3 Filter Backwash Process

Each of the three filters is backwashed every 72 hours resulting in one filter being backwashed per day. The other two factors that trigger a filter backwash are head loss and filter effluent turbidity. A filter would be backwashed if the turbidity was greater than 0.2 NTU or the head loss was greater than 8.5 feet; however, in practice this is a rare occurrence. The filter being backwashed is first drained of water and then an air scour is conducted. The filter is then backwashed under a low flow condition at a rate of 2100 IGPM for two minutes while the air scour continues. The air scour is stopped and the filter is backwashed under a high flow condition at a rate of 4800 IGPM for 10 minutes. It is backwashed for an additional five minutes at a rate of 1200 IGPM. The flow then reverses at a rate of 300 IGPM for two minutes. This is known as the rinse or filter-to-waste stage. After two minutes, provided the turbidity is less than 0.2 NTU, the waste valve closes and water flows from the filter to the clear well. If the turbidity is not less than 0.2 NTU, the filter to waste process continues until it is. During the backwash process the filter bed is fluidized. The intention is for the low flow backwash to just fluidize the bed, and for the high flow backwash to cause the bed to expand by 25 to 30% (Dillon, 2003). The total volume of water used per backwash is 58,800 imperial gallons (267,210L) over a duration of 19 minutes. The Operations and Maintenance Manual suggested a total backwash volume of 22,000 to 33,000 imperial gallons (Dillon, 2003). This means that there is a higher volume of liquid residuals being produced per backwash than was anticipated by the designer.

Filter backwash water (FBW) and filter to waste water leaves the filters and flows from the backwash channel through a 24” diameter polyvinyl chloride (pvc) pipe to manhole 5. Clarifier sludge is pumped continuously from the clarifiers at a rate of 50 USGPM and flows through a 2” diameter pvc pipe to manhole 5. Combined filter backwash water

(CFBW) is a term used to describe the liquid residuals stream that contains both filter backwash water and clarifier sludge. Sodium bisulfite is added to the CFBW for dechlorination via a diffuser located in manhole 5. Water leaves manhole 5 and flows via a 24” pvc pipe to manhole 6 where the flow is split and flows via two 24” pipes to the lagoons. The inlets to the lagoons are located at the upstream outer corner of either lagoon and are situated on the bottom of the lagoons. The outlets from the lagoons are located at the downstream inner corner of each lagoon and consist of an adjustable weir. Water flows from the outlet of each lagoon to manhole 7. It leaves manhole 7 and flows to the discharge at the brook via a 24” pipe. The discharge is situated downstream of the reservoir that is used as the source water supply for the plant. Each lagoon is 132 feet wide and 282 feet long. The depth from the design water level to the base of the lagoon is 6.25 feet. Although the flow to the lagoons from filter backwash operations is intermittent, the clarifier sludge is pumped continuously to the lagoons and the lagoons are discharging continuously. Plant operations staff collect a sample from the discharge to the brook weekly. A diagram of the filter backwash process is given in Figure 6.

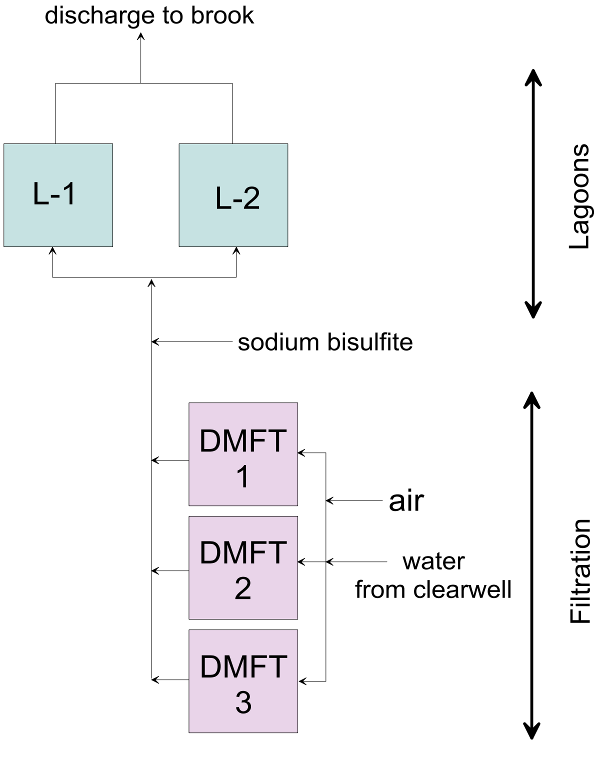

Figure 6 Backwash Process Block Flow Diagram

L = Lagoon
DMFT = Dual Media Filter Tank

3.1.4 Residuals Management

A clarified liquid stream is discharged from the treatment lagoons to the brook, downstream of the reservoir. Solids residual is removed from the treatment lagoons annually. In order to dewater the lagoons, the valve on the inlet of one lagoon is closed and the pond is dewatered by pumping the water and solids residual from the lagoon to the drying bed. The process is then repeated with the other lagoon. The solids residual naturally undergoes a seasonal freeze and thaw process on the drying bed in order to increase the solids content of the solids residual. An orthophotograph showing the WTP, the two parallel treatment lagoons, and the drying bed is shown in Figure 7.

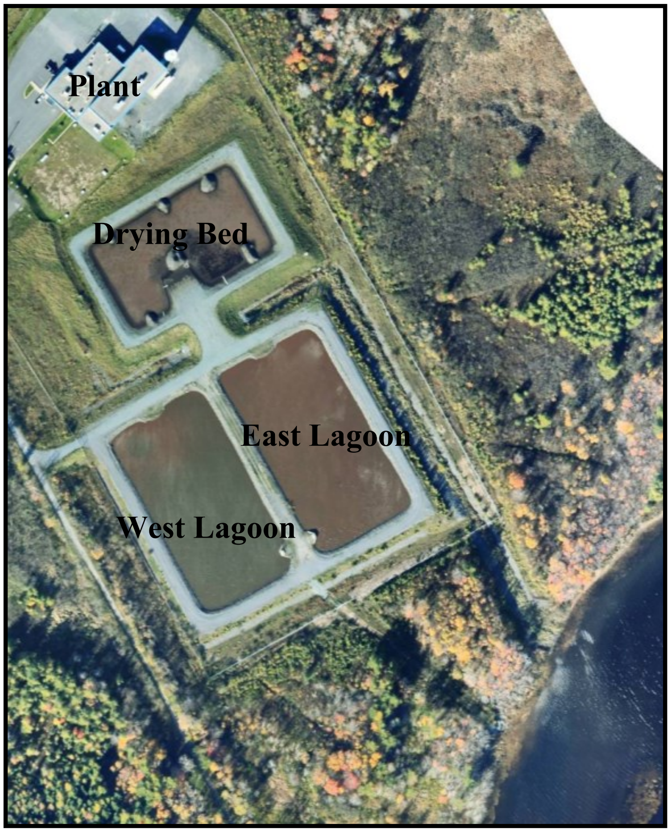

Figure 7 Orthophotograph of the WTP, Lagoons and Drying Bed

3.2 ALUMINUM DISCHARGE TARGET IN LIQUID STREAM

The Part V Approval to Operate does not contain a numerical discharge target for aluminum in the lagoon effluent. The CCME FWAL guideline for aluminum is 100 µg/L for water with a pH of greater than 6.5 and 5 µg/L if the pH is less than 6.5. There is no CCME water quality guideline for marine waters. The section of the brook where the lagoon effluent is discharged has tidal influence from the Atlantic Ocean. The Nova Scotia Environment (NSE) document "Nova Scotia Treatment Standards for Municipal Drinking Water Systems", approved on March 12, 2012, requires that for discharge to freshwater, the TSS must not exceed 5 mg/L over the naturally occurring clear flow background concentration, chlorine must not exceed 0.02 mg/L, pH must be between 6.5 and 9.0, and the discharge must be non-acutely lethal (NSE, 2012). Discharge to freshwater must also meet CCME FWAL criteria for metals, unless the naturally occurring background concentration is higher, then the discharge criteria may be the 90th percentile of the background concentration. If that criteria cannot be reached, a 10% increase above the 90th percentile may be allowed. If that criteria cannot be reached, a study is required to recommend discharge criteria limits. The Treatment Standards require the metals discharge criteria to be met at the end of pipe in 95% of samples. Samples must be collected at least monthly. For discharge to a municipal wastewater system or brackish environment, NSE must be contacted to determine the discharge requirements.

3.3 ISSUES IDENTIFIED WITH RESIDUALS MANAGEMENT

One of the issues that have been identified at the site related to residuals management involves finding a solution for disposal or reuse of the solids residual from the drying bed. Exceedances of CCME soil quality guidelines, particularly for chromium, have limited disposal options for the solids residual in the drying bed. The chromium concentration in the solids residual has ranged from 69 to 130 mg/kg. The CCME soil quality guideline for chromium for commercial/industrial sites is 87 mg/kg. Another issue relates to the aluminum levels in the discharge from the lagoons. Sampling conducted by plant operations staff has indicated that the aluminum levels in the discharge in 2014 ranged from 13.05 µg/L to 354 µg/L with an average of 120 µg/L. The CCME Freshwater Aquatic Life Guideline for aluminum in receiving waters with a pH of greater than or equal to 6.5 is 100 µg/L. In addition, it has been identified that during periods of increased turbulence in the lagoons there is less efficient settling or re-entrainment of floc resulting in floc discharging from the lagoons.

3.4 PREVIOUS WORK RELATED TO RESIDUALS MANAGEMENT AT THE SITE

In 2008, CBCL Ltd completed a study to evaluate the aluminum and TSS levels in the lagoon discharge with the purpose of informing the development of site-specific discharge limits. This report indicated that the aluminum concentration in the brook upstream of the lagoon discharge ranged from 120 µg/L to 270 µg/L with an average of 164 µg/L (CBCL, 2008).

CHAPTER 4 MATERIALS AND METHODS

4.1 EVALUATE LIQUID STREAM RESIDUALS TREATMENT

4.1.1 Characterization of Filter Backwash Water

Samples of combined filter backwash water (CFBW), which is comprised of clarifier sludge and filter backwash water, were collected on 10 occasions (July 17; September 9, 11; October 15, 16; November 4, 12, 18; December 8, 2014; and May 2, 2015). Samples were collected from Manhole 6, situated immediately upstream of the treatment lagoons; samples were collected after the backwash water was observed to be flowing through the manhole which occurs approximately 45 minutes after the backwash is initiated. It is important to note that as the CFBW was obtained as a grab sample, and the FBW cycle has a duration of 19 minutes, the sample represents a "snapshot" of the CFBW chemistry. A flow proportional composite sample would be more representative of the CFBW chemistry. Clarifier sludge is discharging continuously while FBW discharges over a period of 19 minutes once daily.

Samples were stored at 4 °C prior to analysis. Samples to be analyzed for total suspended solids (TSS) were not preserved. Samples to be analyzed for metals were preserved by lowering the pH to less than 2 with trace metal grade nitric acid (A509-P212, Fisher Scientific, Ottawa, Ontario). Samples were analyzed for TSS using method 2420 APHA. This involved filtering a volume of sample through a glass fibre filter that had previously been weighed. The filter with accumulated solids was then oven dried at 104 °C in an oven for 1 hour and weighed. The drying and weighing procedure was repeated until consecutive measurements were within the lesser of 0.5 mg or 4% of the previous measurement. The difference in mass between the dried filter and the original mass of the filter is equal to the weight of solids accumulated on the filter. This mass is divided by the volume of sample that was filtered through the filter to determine the TSS. The FBW samples were also digested with trace metal grade nitric acid (A509-P212, Fisher Scientific, Ottawa, Ontario) prior to being analyzed for chromium and aluminum using an ICP-MS. Standard Method 3030E was followed for nitric acid digestion. The reportable detection limits of the ICP-MS are provided in Table 6.

Table 6 Analytical Detection Limit of ICP-MS for Aluminum and Chromium

Parameter	Detection Limit (µg/L)
Aluminum (Al)	4
Calcium (Ca)	10
Chromium (Cr)	0.4
Iron (Fe)	7
Magnesium (Mg)	10
Manganese (Mn)	0.8
Potassium (K)	10
Sodium (Na)	10

4.1.2 Tracer Study

A tracer was used to study the flow of water through the lagoons and to determine the residence time distribution (RTD) of the lagoons. Rhodamine WT (RWT) was used as a tracer. The methodology was based on a pulse input experiment as described by Folger (2006) in which an amount of tracer is injected into the inflow in as short a time as possible. A YSI 6130 RWT sensor was used with a YSI 6920 multi-parameter sonde to measure the concentration of RWT in the water. The tracer was injected into manhole 6 which is situated upstream of the treatment lagoons. From the manhole, the flow splits into two streams which flow into the lagoons at the outer corner of each lagoon. The inflows are located on the bottom of the lagoons. Water flows through the lagoons before discharging into an outlet structure at the inner downstream corner of each lagoon. Each stream then flows to manhole 7. From manhole 7, the water flows through a 24” diameter pipe before discharging to the brook. The sonde could not be deployed at the discharge to the brook for security reasons as it is located outside the fenced in property boundary of the plant. The sonde was deployed at the outlet structure of one of the lagoons to determine the RTD of the lagoon. As the tracer was injected in manhole 6, the tracer was split between each of the lagoons but was only monitored at the outlet of one. The amount of tracer that was injected was determined based on the estimated volume of water in the lagoons and the detection limit of the instrument. The YSI 6130 RWT sensor has a range of 0 – 200 μg/L with a resolution of 0.1 μg/L and an accuracy of the greater of 5% or 1 μg/L.

The first tracer study was performed on May 23, 2014 when the lagoons had approximately 9 months of solids residual accumulation. During the tracer study, 2L of RWT was diluted into water from the lagoon prior to being injected into manhole 6. This was done in order to adjust the density to be closer to that of the water in the lagoons. The tracer concentration was monitored at the outlet of the west lagoon at a rate of one reading every minute for approximately 28 hours.

The second tracer study was performed on December 11, 2014 when the lagoons had approximately 3 months of solids residual accumulation. During the tracer study, 2L of RWT was diluted into the water in manhole 6 just prior to the backwash flow reaching the manhole in order to adjust the density to be closer to that of the water in the lagoons. The tracer concentration was monitored at the outlet of the east lagoon at a rate of one reading every 5 minutes for approximately 100 hours. The tracer was monitored at the outlet of the east lagoon during the second tracer study because the west lagoon was covered with a layer of ice on the day the test was started.

4.1.3 Lagoon Effluent Sampling

In order to determine the variation in aluminum concentration with time in the effluent from the treatment lagoons, samples were collected periodically during the day from the area in front of the discharge structure of one of the treatment lagoons. This exercise was first conducted on July 11, 2014 when the lagoons had approximately 11 months accumulation of solids residual and was repeated on November 10, 2014 after the solids residual had been pumped from the lagoons and there was had approximately 2 months accumulation of solids residual. Samples were dispensed into 10 mL tubes. Aliquots of selected samples were filtered with 0.45 micron filters. The filtered sample results represent the dissolved portion of aluminum while the unfiltered sample results represent total aluminum. All samples were preserved with trace metal grade nitric acid (A509-P212, Fisher Scientific, Ottawa Ontario) by adjusting the pH to less than 2. Samples were stored at 4 °C prior to analysis. All samples were analyzed for aluminum using an inductively coupled plasma mass spectrometer (ICP-MS). The detection limit of the instrument for aluminum is 4 μg/L.

The aluminum results from the lagoon effluent sampling conducted by the plant operations staff at the discharge from the lagoons to the brook during 2014 were also obtained. The method used by plant staff to analyze aluminum uses a Hach DR 2800 spectrophotometer and Method 8326 – Eriochrome Cyanine R Method.

4.2 OPTIMIZATION OF LIQUID STREAM RESIDUALS TREATMENT

A series of settling tests were conducted in order to evaluate whether settling of particulate aluminum in CFBW could be improved through amendment of the CFBW with chemical addition. On the day of each test, a 10L container was filled with a sample of combined filter backwash water that was collected from manhole 6 which is situated immediately upstream of the lagoons. The sample was collected after flow was observed in the manhole which occurs approximately 45 minutes after the backwash is initiated. It is important to note that as the CFBW was obtained as a grab sample, and the FBW cycle has a duration of 19 minutes, the sample represents a “snapshot” of the CFBW chemistry. A flow proportional composite sample would be more representative of the CFBW chemistry.

4.2.1 Polymer Settling Tests

A jar test apparatus (7790-901, Phipps & Bird, Richmond Virginia) was used for all settling tests. Five of the jars were filled with 750 mL of combined filter backwash water. Four polymers were used in the settling tests as listed in Table 7. Polymers were selected to cover the categories of anionic, non-ionic and cationic polymers in order to determine if one type exhibited superior results. The four polymers selected are currently used at water treatment plants in Nova Scotia, either in the drinking water treatment process or the residuals treatment process.

Table 7 Polymers Used in Settling Tests

Polymer	Charge	Molecular Weight	Source (use)
Polyfloc AP1103	anionic	high MW	Lake Major (water treatment)
Magnafloc LT20	non-ionic	high MW	Glace Bay (water treatment)
Novus 2667	cationic	high MW	Lake Major (residuals treatment)
Superfloc C-1592RS	cationic	high MW	Windsor (residuals treatment)

After the specified dose of polymer was added to each jar, the jars were mixed at 300 RPM (revolutions per minute) for a duration of one minute. The doses and sample intervals used in each of the polymer settling tests are summarized in the following Table 8. The doses were chosen based on the range of polymer doses tested by Wood (2014).

Table 8 Polymer Doses and Sample Intervals Used in Settling Tests

<table>
<tr><th rowspan="2">Doses</th><th colspan="5">Polymer (mg/L)</th></tr>
<tr><td>0</td><td>3</td><td>5</td><td>12</td><td>25</td></tr>
<tr><th rowspan="2">Samples</th><th colspan="5">Settling Time (hours)</th></tr>
<tr><td>0</td><td>1</td><td>2</td><td>3</td></tr>
</table>

In order to determine whether the mixing that occurs in the lagoon might have an impact, either positive or negative, on the efficiency of particle removal via settling, one set of tests was conducted with the jars allowed to settle with no mixing after the one minute mix period, and another set of tests was conducted with the jars allowed to settle with mixing at 5 revolutions per minute (RPM). A quantitative estimate of the level of mixing that occurs naturally in the lagoons was not available so 5 RPM was selected as it was the lowest mixing speed on the jar test apparatus. Samples were collected from the jars after the specified settling time. For the settling tests with cationic polymer Superfloc C-1592RS, an aliquot of each sample collected after three hours of settling was filtered through a 0.45 µm filter while the rest of the sample remained unfiltered. The filtered sample results represent the dissolved portion of aluminum while the unfiltered sample results represent total aluminum. All samples were preserved by adjusting the pH to less than 2 with trace metal grade nitric acid (A509-P212, Fisher Scientific, Ottawa Ontario). Samples were stored at 4 °C prior to analysis. All samples were analyzed for aluminum using an inductively coupled plasma mass spectrometer (ICP-MS). The analytical detection limit for aluminum with the ICP-MS is 4 µg/L. CFBW samples, representing 0 hours of settling time, were digested with trace metal grade nitric acid (A509-P212, Fisher Scientific, Ottawa Ontario) prior to analysis due to the presence of floc. Standard Method 3030E was followed for nitric acid digestion. In order to determine whether settling efficiency with settling at 0 RPM and 5 RPM were statistically different, an analysis of variance was conducted using Microsoft Excel 2013.

4.2.2 Cation Ratio Optimization Settling Test

A jar test apparatus (7790-901, Phipps & Bird, Richmond Virginia) was used for the settling test. Each jar was filled with 750 mL of combined filter backwash water. The concentrations of monovalent (sodium and potassium) and divalent (calcium and magnesium) cations in the CFBW were estimated based on previous samples. Based on the estimated concentrations, doses of calcium carbonate and sodium chloride were chosen to provide a range of M:D ratios in the jar tests in meq/L. After the specified doses of calcium carbonate ($CaCO_3$) and/or sodium chloride (NaCl) were added to each jar, the jars were mixed at 300 RPM (revolutions per minute) for a duration of one minute followed by two stages of flocculation: 10 minutes at 40 RPM and 10 minutes at 20 RPM. The jars were then allowed to settle with no mixing. This procedure is consistent with the procedure followed in Bourgeois et al. (2004a). The doses used in the settling tests are summarized in Tables 9 and 10. The resulting M:D ratios are also provided; these values were based on the actual measured concentrations of monovalent and divalent cations in the CFBW plus the monovalent and divalent cations added as calcium carbonate and sodium chloride. M:D ratios tested ranged from approximately 0.1 to 3.

Table 9 Doses of Calcium Carbonate and Sodium Chloride Used in Settling Test 1

Jar	$CaCO_3$ (mg/L)	NaCl (mg/L)	M:D Ratio
1	0	0	0.22
2	130	0	0.09
3	0	30	0.48
4	0	90	1.01
5	0	330	3.11

Table 10 Doses of Calcium Carbonate and Sodium Chloride Used in Settling Test 2

Jar	$CaCO_3$ (mg/L)	NaCl (mg/L)	M:D Ratio
1	0	0	0.22
2	170	0	0.08
3	0	30	0.48
4	0	100	1.10
5	0	310	2.94

Samples were collected from the jars after the specified settling time (0, 0.5, 1, 1.5, 2, 3 and 24 hours). An aliquot of each sample collected after 3 hours and 24 hours of settling time was filtered through a 0.45 micron filter while the rest of the sample remained unfiltered. The filtered sample results represent the dissolved portion of aluminum while the unfiltered sample results represent total aluminum. All samples were preserved by adjusting the pH to less than 2 with trace metal grade nitric acid (A509-P212, Fisher Scientific, Ottawa Ontario). Samples were stored at 4 °C prior to analysis. All samples were analyzed for aluminum using an inductively coupled plasma mass spectrometer (ICP-MS). In addition, the CFBW samples were analyzed for magnesium, calcium, potassium and sodium using an ICP-MS. CFBW samples, representing 0 hours of settling time, were digested with trace metal grade nitric acid (A509-P212, Fisher Scientific, Ottawa Ontario) prior to analysis due to the presence of floc. Standard Method 3030E was followed for nitric acid digestion. The analytical detection limits for the ICP-MS for the parameters analyzed are listed in Table 11.

Table 11 ICP-MS Analytical Detection Limits

Parameter	Detection Limit (µg/L)
Aluminum (Al)	4
Calcium (Ca)	10
Magnesium (Mg)	10
Potassium (K)	10
Sodium (Na)	10

4.3 CHARACTERIZATION OF SOLIDS RESIDUAL (SLUDGE)

4.3.1 Solids Content of Solids residual

In order to determine the solids content of the solids residual that is produced in the treatment lagoons, in-situ solids residual samples were collected from each treatment lagoon using a Sludge Judge sampler from Cole Parmer. Samples were collected adjacent to the outlet structures as this was the only location where a core sample could be obtained without disturbing the solids residual layer. Three core samples were collected from each treatment lagoon on June 11, 2014 when there was approximately 10 months of solids residual accumulation in the lagoons. The samples consisted of two visually distinct solids residual layers that appeared to differ in solids content, and a layer of water at the top. The majority of the solids residual in the core sample appeared to have a very low solids content; however, there was a thin layer at the bottom of the sampler that appeared to have a much higher solids content. The water layer was decanted from the sample and each of the two layers of solids residual was analyzed separately. In-situ solids residual samples were also collected as grab samples from the drying bed. There were also two visually distinct layers of solids residual in the drying bed that appeared to differ in solids content and each layer was analyzed separately. The lower layer of solids residual in the drying bed appeared moist and malleable, and the upper layer of solids residual in the drying bed appeared dry and crumbly. To determine the solids content of the samples, an aliquot of each sample was weighed in a metal weighing dish. The samples were then oven dried at 104 °C for 24 hours. The dried solids were weighed on a scale. The solids content was calculated as the mass of dried solids divided by the initial mass of the sample.

4.3.2 Metals Content of Solids Residual

Samples were collected on July 30, 2014 and November 20, 2014 from two different batches of solids residual in the drying bed. The sample collected on July 30, 2014 was of solids residual that had accumulated in the lagoons from May 2012 to August 2013 and had been drying in the drying bed since August 2013. The sample collected on November 20, 2014 was of solids residual that had accumulated in the lagoons from August 2013 to September 2014 and had been drying in the drying bed since September 2014. The solids residual samples were collected as a grab sample from the drying bed and submitted to Maxxam Analytics in Sydney, Nova Scotia to be analyzed for metals. The method used by Maxxam Analytics for metals analysis was EPA 6020A. Sample results were also obtained from water treatment plant staff for a sample that had been collected from the drying bed on May 30, 2013. This sample was of solids residual that had accumulated in the lagoons from September 2011 and had been drying in the drying bed since May 2012. The results of the metals analysis are reported in units of mg/kg on a dry weight basis. The solids residual samples collected from the drying bed are summarized in Table 12.

Table 12 Drying Bed Solids Residual Samples

Date Collected	Start of Accumulation in Lagoon	Date pumped to Drying Bed
May 30, 2013	September 2011	May 2012
July 30, 2014	May 2012	August 2013
Nov 20, 2014	August 2013	September 2014

4.3.3 Leachability Testing

Samples were collected of solids residual in the drying bed on November 20, 2014. The samples were of solids residual that had accumulated in the lagoons from August 2013 to September 2014 and had been drying in the drying bed since September 2014.

The solids residual samples were collected as a grab sample from the drying bed and submitted to Maxxam Analytics in Sydney, Nova Scotia to be analyzed. The samples were analyzed for synthetic precipitate leaching procedure (SPLP) and metals analysis. The analytical procedure conducted by the laboratory involves analyzing a mass of the sample on an as-received basis and the results are not determined on a dry-weight basis. Therefore, the samples were dried in an oven at 104 °C for 48 hours prior to being submitted to the laboratory in order to determine the SPLP result on a dry weight basis so that the result was not dependent on the water content of the solids residual. The methods used by Maxxam Analytics were US EPA 1312 m for SPLP extraction and US EPA 6020A R1 m for metals analysis.

An additional sample from the solids residual collected on November 20, 2014 was submitted to the laboratory for SPLP analysis on an as-collected basis in order to determine the impact the drying procedure had on the results of the SPLP analysis.

4.3.4 Solids Residual Amendment

Solids residual samples collected on November 20, 2014 were blended with air cooled blast furnace slag in order to determine the impact that amending the solids residual with slag would have on the leaching characteristics of the solids residual. Synthetic precipitation leaching procedure (SPLP) is a laboratory test that approximates the leaching that would occur due to slightly acidic precipitation. This test is performed on the material as it is received, so a sample of solids residual from the drying bed was first dried in an oven at 104 °C for 48 hours before blending with slag in order to obtain a test result that is not impacted by the solids content of the solids residual. Four samples of approximately 250 mL were prepared from the dried solids residual. Air-cooled blast furnace slag was added to the samples in the following amounts on a mass percent basis: 0%, 2%, 5% and 10%. The samples were then submitted to Maxxam Analytics in Sydney, NS in order to be analyzed for SPLP with metals analysis. The methods used by Maxxam Analytics were US EPA 1312 m for SPLP extraction, and US EPA 6020A R1 m for metals analysis.

4.4 INVESTIGATION TO DETERMINE SOURCE OF CHROMIUM IN SOLIDS RESIDUAL

4.4.1 Sampling Program

In order to determine the source of the chromium in the solids residual, samples were collected from various locations along the treatment train including raw water, pre-treatment tank 1, pre-treatment tank 2, flocculation tank, clarifier, post-filtered water, clearwell, and lagoon discharge to the Brook. Samples were collected on May 12, May 14 and May 21, 2014. The discharge to the brook was not sampled on May 14, 2014 due to access issues. Samples were also collected of the polymer and alum that are added to the process in pre-treatment tank 2. On December 31, 2014 samples were collected from 7 streams that flow into the reservoir just upstream of the reservoir. The location of samples is shown on Figure A1 in Appendix A.

Samples were collected in 10 mL tubes and each sample was preserved by adjusting the pH to less than 2 with trace metal grade nitric acid (A509-P212, Fisher Scientific, Ottawa Ontario). Selected samples were digested with trace metal grade nitric acid prior to analysis. Standard Method 3030E was followed for nitric acid digestion. Samples were stored at 4 °C prior to being analyzed. Water samples were analyzed for chromium using an inductively coupled plasma mass spectrometer (ICP-MS) instrument. The method detection limit for chromium is 0.4 µg/L.

4.4.2 Mass Balance

A mass balance of the water treatment process (Figure 8) was conducted in order to:

1. Provide an estimate of the solids residual production rate; and

2. Determine whether the chromium concentration in the solids residual could be explained by low levels of chromium in the raw water.

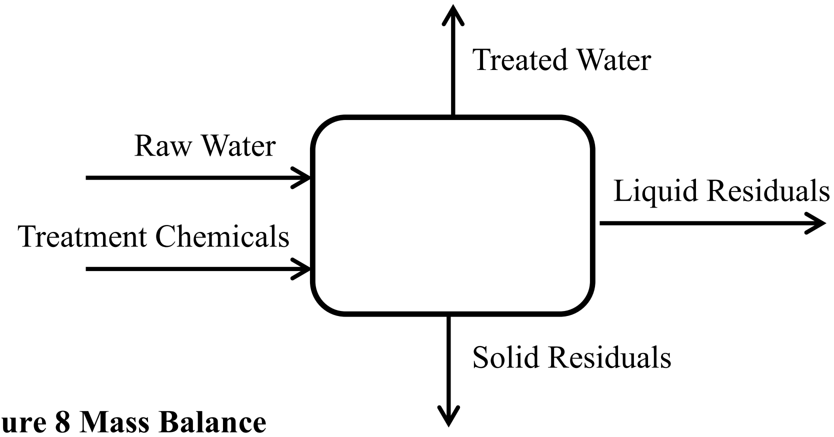

Figure 8 Mass Balance

Aluminum enters the system in the raw water and in the aluminum sulfate which is added as a coagulant. It leaves the system in the lagoon effluent, in the treated water, and in the solids residual. A mass balance calculation of the aluminum load in the solids residual and the aluminum concentration from a metals analysis of the solids residual were used to calculate an estimated solids residual production rate. A “high” and “low” solids residual production rate was calculated using the lowest and highest measured aluminum concentration in the drying bed solids residual, and low and high estimates of raw and treated water flowrates.

The calculated solids residual production rate, the chromium concentration from a metals analysis of the solids residual, the chromium load in the aluminum sulfate coagulant, and the chromium load in the raw water were used in a mass balance calculation of chromium to determine the daily chromium load to the system that is unexplained by the chromium load in the alum and raw water.

Next, the calculated solids residual production rate, the chromium concentration from a metals analysis of the solids residual and the chromium load in the aluminum sulfate coagulant were used in a mass balance calculation of chromium in order to determine what the chromium load in the raw water would have to be in order to account for the elevated chromium level in the solids residual that is not accounted for by the aluminum sulfate. This analysis was conducted as all digested chromium samples of raw water were collected in November and December and there may be seasonal variation.

CHAPTER 5 RESULTS

5.1 EVALUATE LIQUID STREAM RESIDUALS TREATMENT

5.1.1 Characterization of Filter Backwash Water

Twelve combined filter backwash samples were collected on 10 different occasions between July 17, 2014 and May 2, 2015. The samples were analyzed for TSS, chromium, aluminum, potassium, sodium, calcium, magnesium, iron, manganese and pH although not all samples were analyzed for every parameter. The results are presented in Table 13. With the exception of the December 8, 2014 samples, samples were collected from Manhole 6 after flow was observed in the manhole which occurs approximately 45 minutes after the backwash is initiated. On December 8, 2014, three consecutive samples were collected during approximately the first 5 minutes after flow was observed in the manhole. It is important to note that as the CFBW was obtained as a grab sample, and the FBW cycle has a duration of 19 minutes, the CFBW samples collected represent a "snapshot" of the CFBW chemistry. A flow proportional composite sample would be more representative of the CFBW chemistry.

Table 13 Analytical Results of CFBW

Date	TSS	Al	Cr	K	Na	Ca	Mg	Fe	Mn	pH
	mg/L	mg/L	µg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	units
17-Jul-14	-	70.3	-	0.51	10.71	24.13	0.68	-	-	-
9-Sep-14	-	191	13.19	-	-	-	-	-	-	-
11-Sep-14	1367	330	15.65	-	-	-	-	-	-	-
15-Oct-14	1400	399	16.75	-	-	-	-	-	-	-
16-Oct-14	2333	798	31.29	-	-	-	-	-	-	-
4-Nov-14	3000	525	34.79	2.14	10.16	31.62	5.84	-	-	-
12-Nov-14	2333	528	27.54	-	-	-	-	-	-	-
18-Nov-14	2333	377	19.39	-	-	-	-	-	-	-
8-Dec-14	1321	255	19.03	-	-	-	-	-	-	-
8-Dec-14	400	94	5.117	-	-	-	-	-	-	-
8-Dec-14	300	109	8.835	-	-	-	-	-	-	-
2-May-15	1667	376	8.254	2.20	8.50	32.00	4.30	75.9	37.4	6.25
min	300	70	5.117	0.51	8.50	24.13	0.68	75.9	37.4	6.25
max	3000	798	34.79	2.20	10.71	32.00	5.84	75.9	37.4	6.25
average	1645	338	18.17	1.62	9.79	29.25	3.61	75.9	37.4	6.25

TSS measurements ranged from 300 to 3000 mg/L with an average of 1643 mg/L. Aluminum concentrations in the CFBW ranged from 70 to 798 mg/L with an average of 331 mg/L. Chromium concentrations in the CFBW ranged from 5.117 µg/L to 34.79 µg/L with an average of 18.17 µg/L. The CFBW also contains iron and manganese. A sample collected on May 2, 2015 had an iron concentration of 75.9 mg/L and a manganese concentration of 37.4 mg/L. The concentrations of TSS, aluminum and chromium in the second and third samples that were taken during the FBW cycle on December 8, 2014 are lower than all other measurements. To highlight the variation in water quality of CFBW over the duration of the FBW cycle, the TSS and aluminum concentrations in the three consecutive CFBW samples collected on December 8, 2014 are plotted on Figure 9.

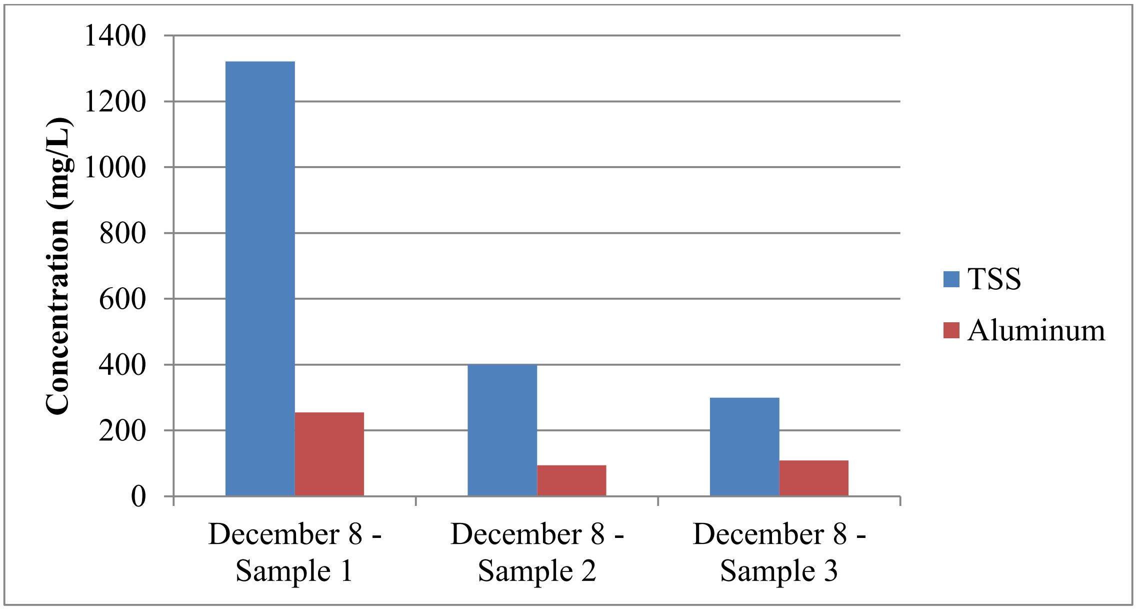

Figure 9 Variations in FBW Chemistry during Backwash Cycle

The CFBW samples collected from manhole 6 contain both FBW and clarifier sludge. Clarifier sludge is continuously discharged to the lagoons while FBW is discharged intermittently. A sample was also taken of the sludge at the bottom of the clarifiers and analyzed for TSS, aluminum and chromium. The results are displayed in Table 14.

Table 14 Analytical Results of Clarifier Sludge

Date	TSS	Al	Cr
	mg/L	mg/L	µg/L
18-Nov-14	1667	269	11.7

The TSS, aluminum and chromium results from the clarifier sludge sample are all within the ranges of the CFBW sample results.

5.1.2 Tracer Study

5.1.2.1 Initial Tracer Test to Establish Protocols

The theoretical residence time of the treatment lagoons is 24.4 days with no solids residual accumulated in the lagoons, and 11.6 days with 1 m of solids residual accumulated in the lagoons. The first tracer study was conducted on May 23, 2014. At the time of this test there was approximately 9 months of solids residual accumulated in the lagoons. It is estimated that 9 months accumulation of solids residual would result in between 0.5 and 0.7m of solids residual in the lagoons based on the discussion presented later in Section 6.3.1. During this study, Rhodamine-WT (RWT) was monitored at the outlet of the western lagoon at a rate of one measurement per minute for approximately 28 hours. A normalized plot of the tracer concentration measured at the outlet of the western lagoon with time is provided in Figures 10 and 11.

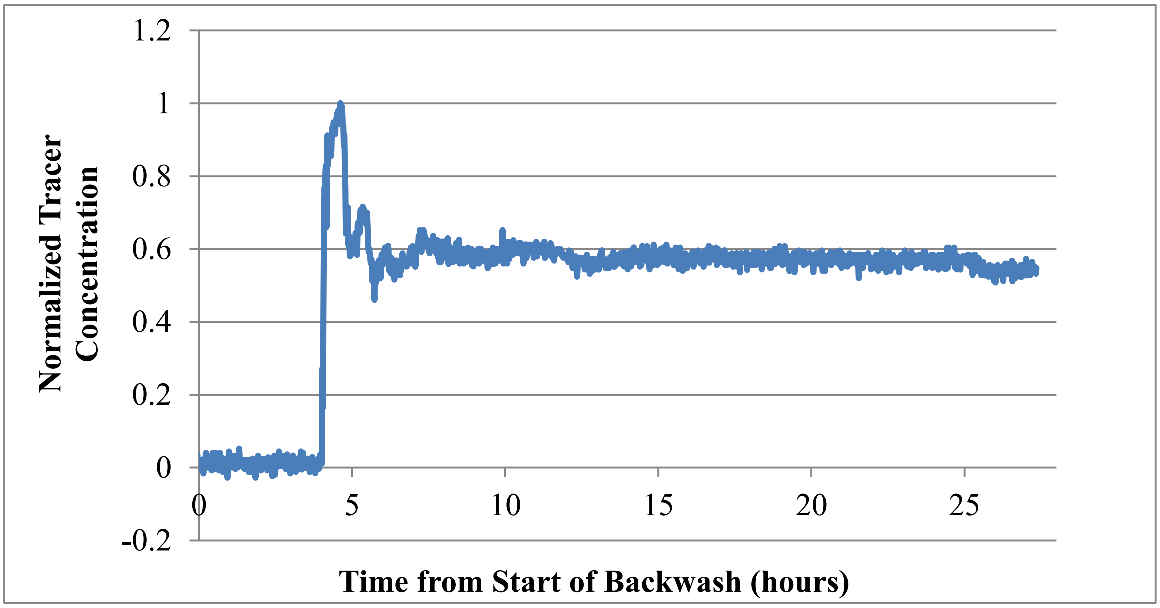

Figure 10 May 23, 2014 Tracer Test

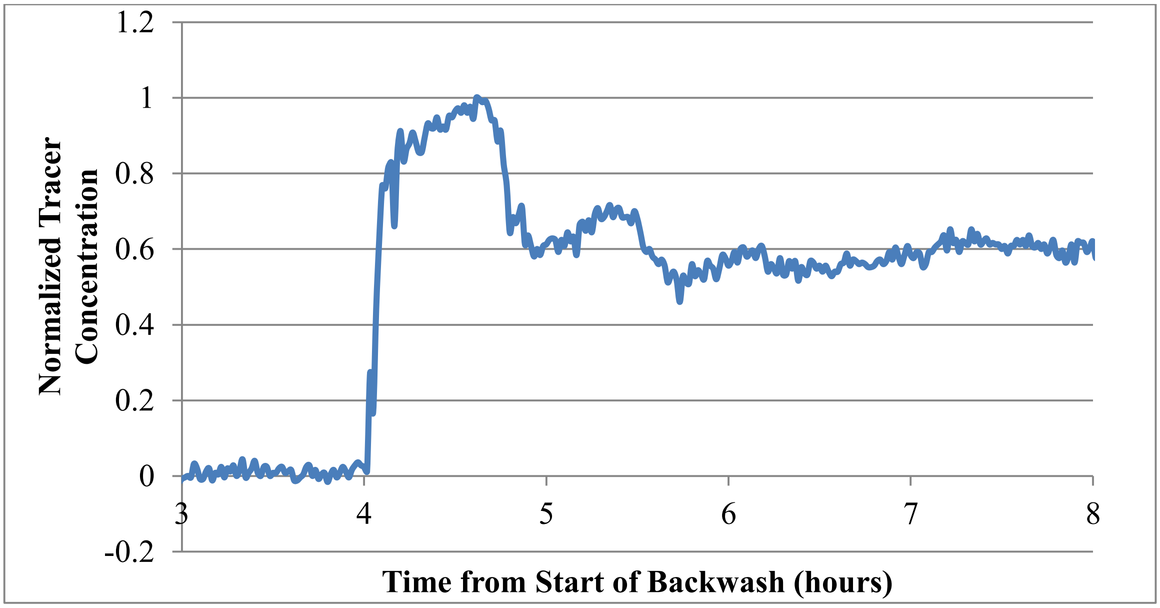

Figure 11 May 23, 2014 Tracer Test

The RWT was detected at the outlet of the lagoon approximately 240 minutes after the backwash was initiated and approximately 200 minutes after flow was observed in manhole 6. The minimum residence time is slightly less than 200 minutes as the flow observed in the manhole is thought to be due to the high flowrate backwash which occurs after the low flow backwash period. The concentration of RWT in the outlet reached a maximum 277 minutes after backwash was initiated and 234 minutes after flow was observed in manhole 6. There was a second peak in RWT concentration at the outlet 323 minutes after backwash was initiated and 280 minutes after flow was observed in manhole 6. After the second peak, the RWT concentration remained fairly steady for the remainder of the test which is associated with a fully mixed condition. The mean residence time could not be calculated for this tracer study as the duration of the test was not long enough.

5.1.2.2 Tracer Test to Evaluate Lagoon Hydraulics

A second tracer study was conducted on December 11, 2014 with the objective of monitoring the concentration of RWT at the discharge structure for a longer period of time as well as after the solids residual had been pumped from the lagoons. At the time of this test there was approximately 3 months of solids residual accumulated in the lagoons. It is estimated that 3 months accumulation of solids residual would result in approximately 0.2m of solids residual in the lagoons based on the discussion presented later in Section 6.3.1. During this study, RWT was monitored at the outlet of the eastern lagoon as the western lagoon was covered with a layer of ice on the day the test was started. RWT was monitored at the lagoon outlet at a rate of one measurement every 5 minutes for approximately 100 hours. A normalized plot of the tracer concentration measured at the outlet of the eastern lagoon with time is provided in Figure 12.

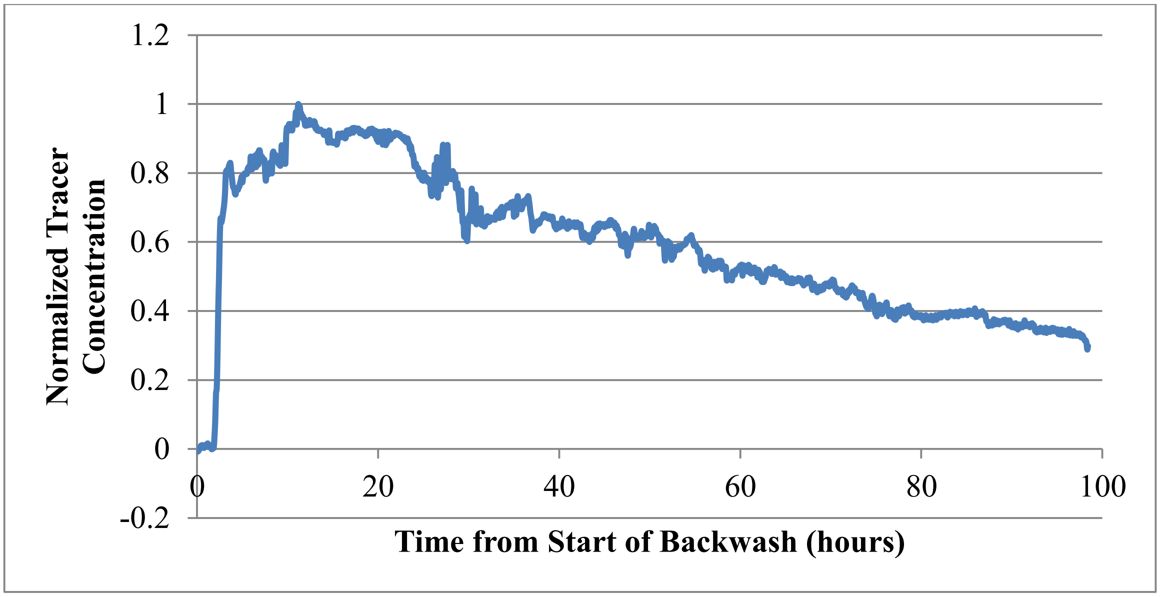

Figure 12 December 11, 2014 Tracer Test

The tracer concentration during the second tracer study reached a maximum and then decreased approximately linearly afterward. The RWT was detected at the outlet of the lagoon approximately 125 minutes after the backwash was initiated and approximately 85 minutes after flow was observed in manhole 6. The minimum residence time is slightly less than 85 minutes as the flow observed in the manhole is thought to be due to the high flowrate backwash which occurs after the low flow backwash period. The concentration of RWT at the outlet reached a maximum at 670 minutes after backwash was initiated and 630 minutes after flow was observed in manhole 6. After the peak was reached, the tracer concentration decreased at a rate of approximately 0.72% per hour (Figure 13).

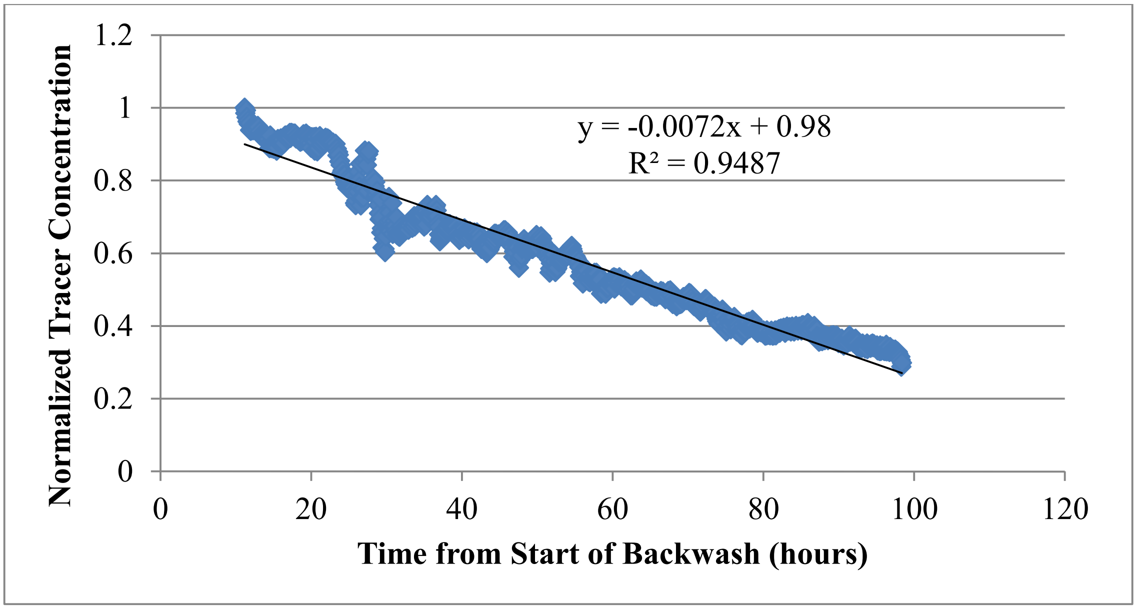

Figure 13 Rate of Decrease in Tracer Concentration December 11, 2014

The C(t) curve is a plot of the measured tracer concentration at the outlet of the lagoon with time (Figure 14). The blue portion of the curve represents the measured concentration of tracer at the outlet of the lagoon; the red portion of the curve represents the projected concentration of tracer at the outlet of the lagoon after monitoring was ceased. This projection was determined based on a line of best fit through the concentration data from the time of the maximum tracer concentration until the time monitoring was ceased.

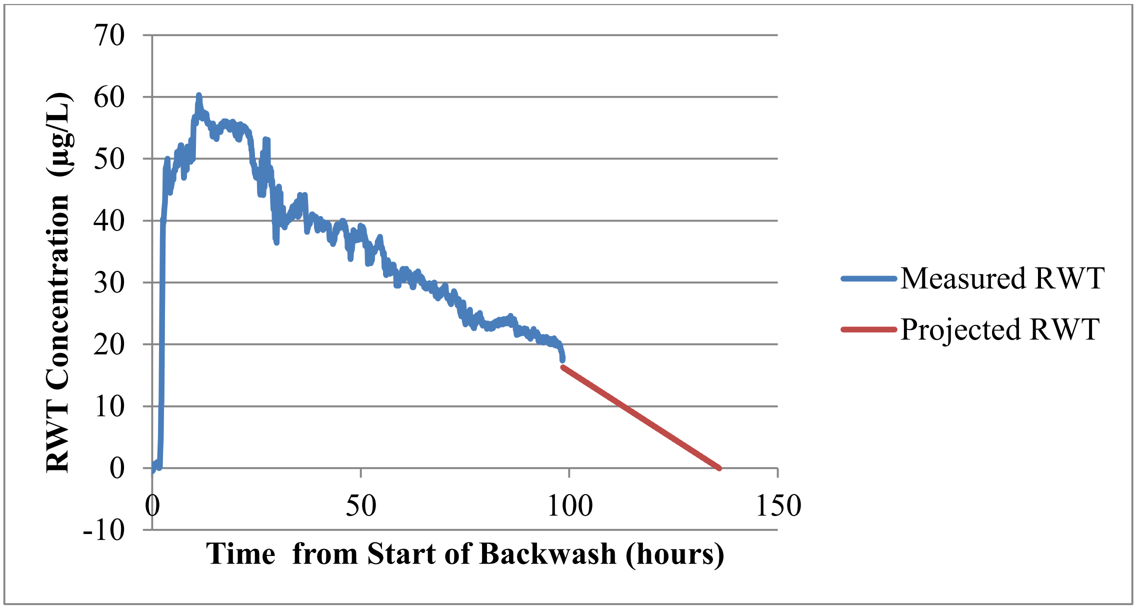

Figure 14 C(t) Curve for December 11, 2014 Tracer Test

The E(t) curve (Figure 15) represents the residence time distribution function and is obtained by the following equation (Fogler, 2006):

$$E(t) = \frac{C(t)}{\int_0^\infty C(t)dt}$$

Using the Trapezoidal Rule for numerical integration of the C(t) data with a 5 minute time step, the area under the C(t) curve was calculated to be 3824.8 µg·hr/L. This value was used to construct the E(t) curve (Figure 15). The area under the E(t) curve between any two points represents the fraction of matter leaving the system that has resided in the system for an amount of time between those two points; for instance the area under the E(t) curve between 0 and time t represents the fraction of matter exiting the system that has resided in the system for less than time t (Fogler, 2006). The below E(t) curve (Figure 15) approximates CSTRs in series; the theoretical normalized E(t) curve for CSTRs in series was provided in Figure 1.

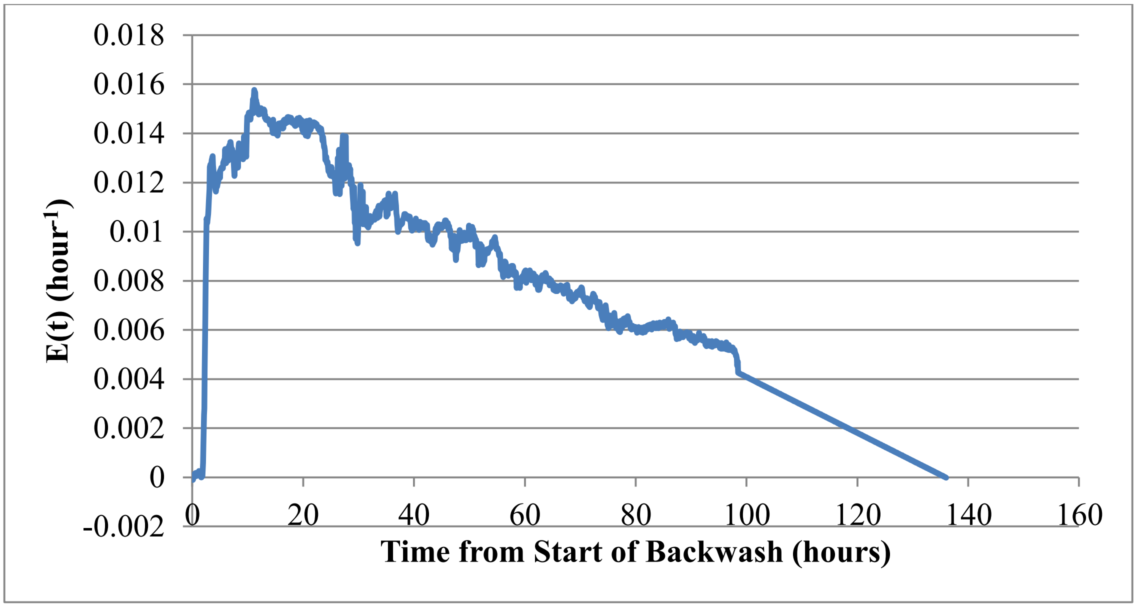

Figure 15 E(t) Curve for December 11, 2014 Tracer Test

The F(t) curve (Figure 16) represents the cumulative distribution function and is obtained by the following equation (Fogler, 2006):

$$F(t) = \int_0^t E(t)dt$$

Using the Trapezoidal Rule for numerical integration of the E(t) data with a 5 minute time step, the area under the E(t) curve from 0 to time t was calculated at each 5 minute increment to construct the F(t) curve (Figure 16). The F(t) curve is useful for quickly determining the length of time it takes a given fraction of matter to exit the system.

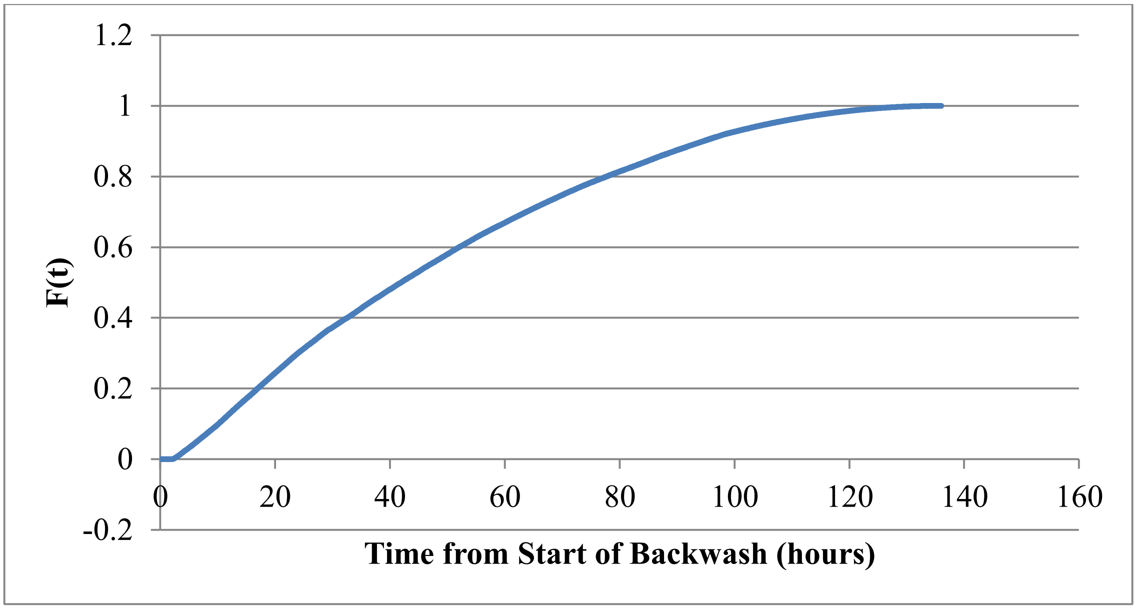

Figure 16 F(t) Curve for December 11, 2014 Tracer Test

From the F(t) curve it was determined that, 20%, 50% and 80% of the water spent less than 17, 42 and 77.75 hours in the lagoon, respectively.

The mean residence time in the lagoon was calculated to be 47.5 hours based on the following formula using the trapezoidal rule for numerical integration with a 5 minute time step.

$$t_m = \int_0^\infty tE(t)dt$$

5.1.3 Lagoon Effluent Sampling

The results of the effluent discharge samples collected by plant operations staff during 2014 were obtained. Figure 17 shows the aluminum concentrations in the effluent discharge samples collected during 2014. The aluminum concentrations ranged from 13.05 to 354 µg/L with an average of 120 µg/L. There did not appear to be a seasonal trend in the concentrations.

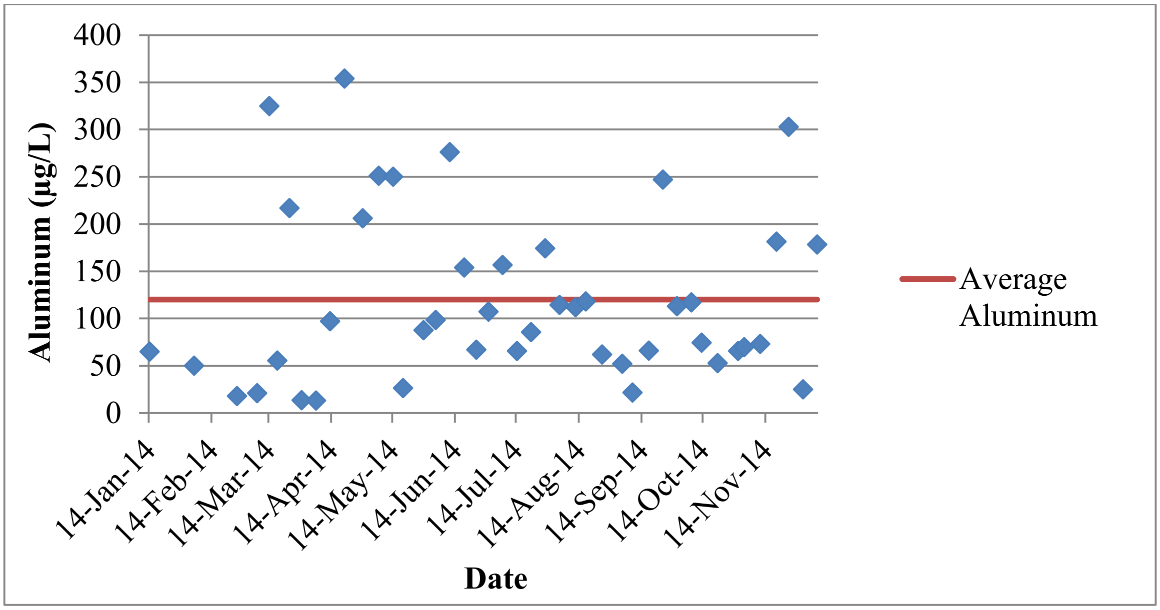

Figure 17 Aluminum in Lagoon Effluent from 2014 Sampling

As the above samples were collected at a frequency of one per week, the data could not show whether there was a variation in aluminum concentration as a result of the backwash cycle. In order to determine if the aluminum concentration in the effluent varied with time in relation to the backwash cycle, samples were collected every half hour for 5.5 hours on July 11, 2014 (Figure 18). At this time there was approximately 11 months accumulation of solids residual in the lagoons. Selected samples were filtered through a 0.45 micron filter in order to determine the concentration of aluminum in the dissolved phase. The total aluminum concentration ranged from 168 µg/L to 298 µg/L with an average of 195 µg/L. The dissolved aluminum concentration ranged from 24.47 µg/L to 67.64 µg/L with an average of 37.12 µg/L. A plot of the normalized tracer concentration during the May 23, 2014 tracer test was also included in Figure 18. The point where tracer is first detected at the outlet is the minimum residence time of water in the lagoon. It was expected that elevated aluminum concentrations in the effluent would be associated with the minimum residence time of water in the lagoon as the aluminum would have had less time to settle. The tracer was first detected at the outlet of the lagoon after approximately 4 hours from the start of backwash, and the maximum aluminum concentration was detected after 5.75 hours. The tracer test and the aluminum samples were not conducted on the same day so some variation was expected.

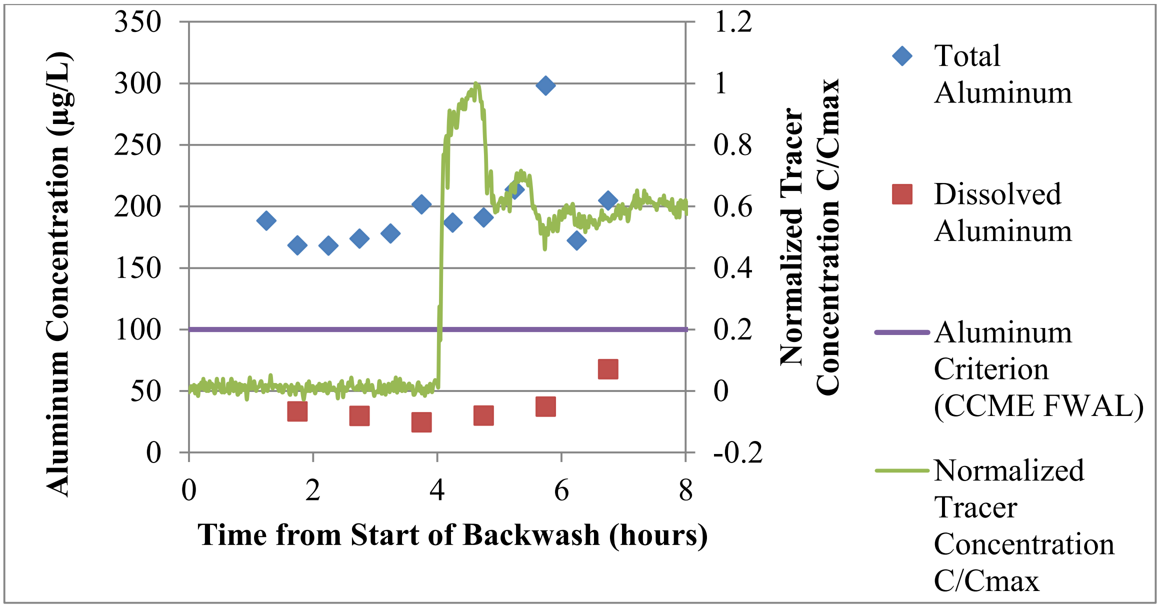

Figure 18 Aluminum Concentrations in Lagoon Discharge on July 11, 2014

For samples where both a total and dissolved aluminum analysis was performed, the dissolved portion of the aluminum concentration ranged from 12% to 33% of the total aluminum concentration with an average of 18%.

Samples were also collected every half hour for 4 hours on November 10, 2014. At this time there was approximately 2 months accumulation of solids residual in the lagoons. The same procedure was followed. The total aluminum concentration ranged from 98.01 µg/L to 128.9 µg/L with an average of 113.5 µg/L. The dissolved aluminum concentration ranged from 16.24 µg/L to 17.74 µg/L with an average of 16.87 µg/L. The results are shown on Figure 19.

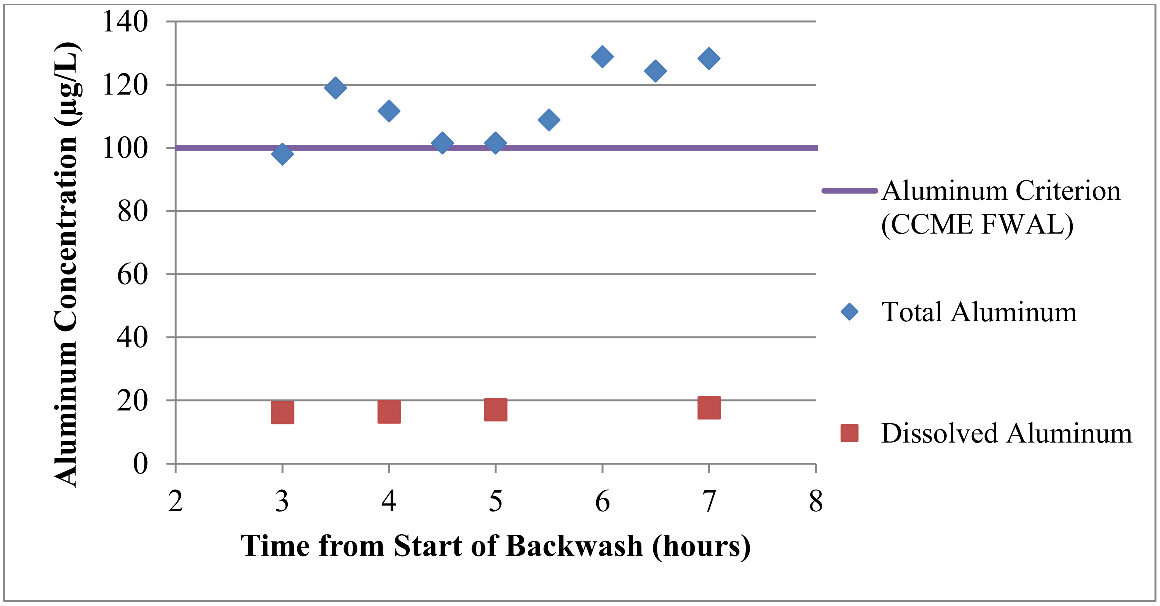

Figure 19 Aluminum Concentrations in Lagoon Discharge on November 10, 2014

For samples where both a total and dissolved aluminum analysis was performed, the dissolved portion of the aluminum concentration ranged from 14% to 17% of the total aluminum concentration with an average of 15%.

5.2 OPTIMIZATION OF LIQUID STREAM RESIDUALS TREATMENT

5.2.1 Polymer Settling Test

Figure 20 shows the aluminum concentration after 1 and 3 hours of settling at 0 RPM with doses of the cationic polymer Superfloc C-1592RS ranging from 0 to 25 mg/L. The concentration of Al in the CFBW used for this test was 377,000 µg/L.

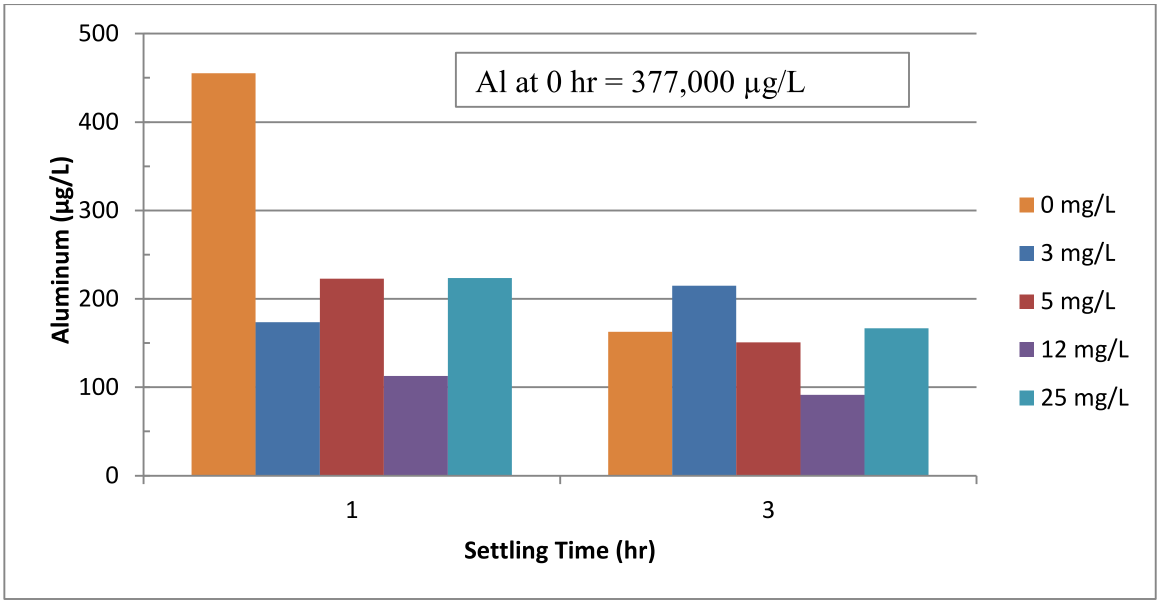

Figure 20 Settling Test with Superfloc C-1592RS at 0 RPM

There does not appear to be a trend of increased removal of aluminum with increased polymer dose. However, after three hours of sampling, the 5, 10 and 12 mg/L doses had aluminum concentrations of 150.9, 83.79 and 91.25 µg/L, respectively compared to an aluminum concentration of 162.7 µg/L with no polymer.

Figure 21 shows the aluminum concentration after 1, 3, and 24 hours of settling at 0 RPM with doses of the cationic polymer Novus 2667 ranging from 0 to 25 mg/L. The concentration of Al in the CFBW used for this test was 109,400 µg/L.

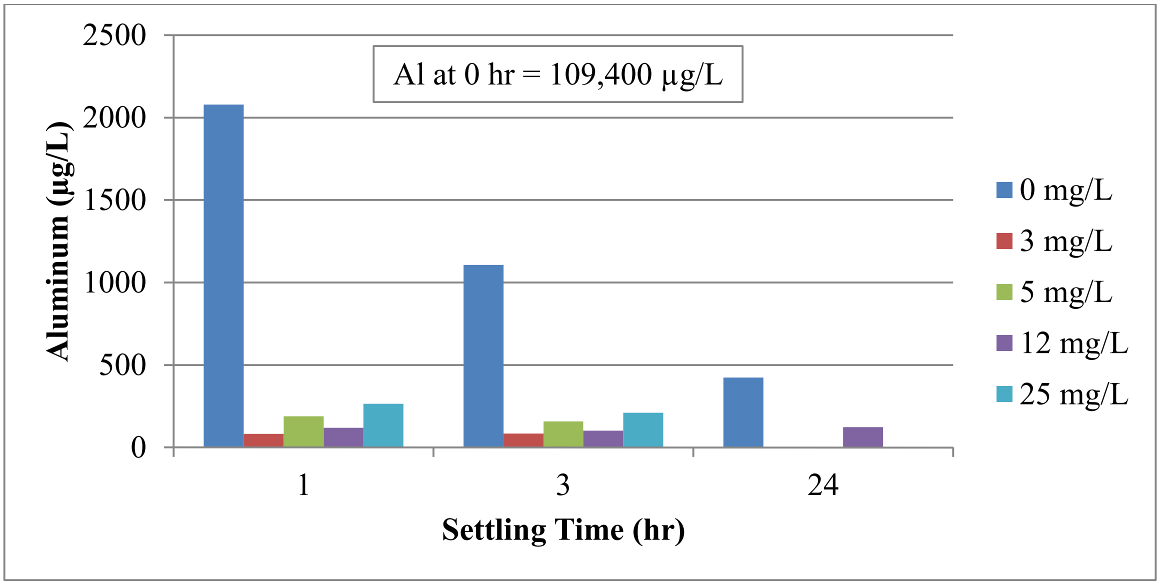

Figure 21 Settling Test with Novus 2667 at 0 RPM

All doses of polymer Novus 2667 showed significant removal after 1 and 3 hours of settling compared to no polymer. After 1 hour, aluminum concentrations with varying polymer doses ranged from 82.13 to 263.9 µg/L compared to a concentration of 2079 µg/L with no polymer. After 3 hours, aluminum concentrations with varying polymer doses ranged from 83.78 to 187.8 µg/L compared to a concentration of 1107 µg/L with no polymer. After 24 hours, the aluminum concentration with a polymer dose of 12 mg/L was 123.3 µg/L compared to a concentration of 423.7 µg/L with no polymer. Samples could not be taken after 24 hours from the jars with the other polymer doses because the floc was floating on the surface of the water. The 3 mg/L dose had the best aluminum removal at both the 1 and 3 hour sample intervals with concentrations of 82.13 and 83.78 µg/L.

Figure 22 shows the aluminum concentration after 1 and 3 hours of settling at 0 RPM with doses of the anionic polymer Polyfloc AP1103 ranging from 0 to 25 mg/L. The concentration of Al in the CFBW used for this test was 254,700 µg/L.

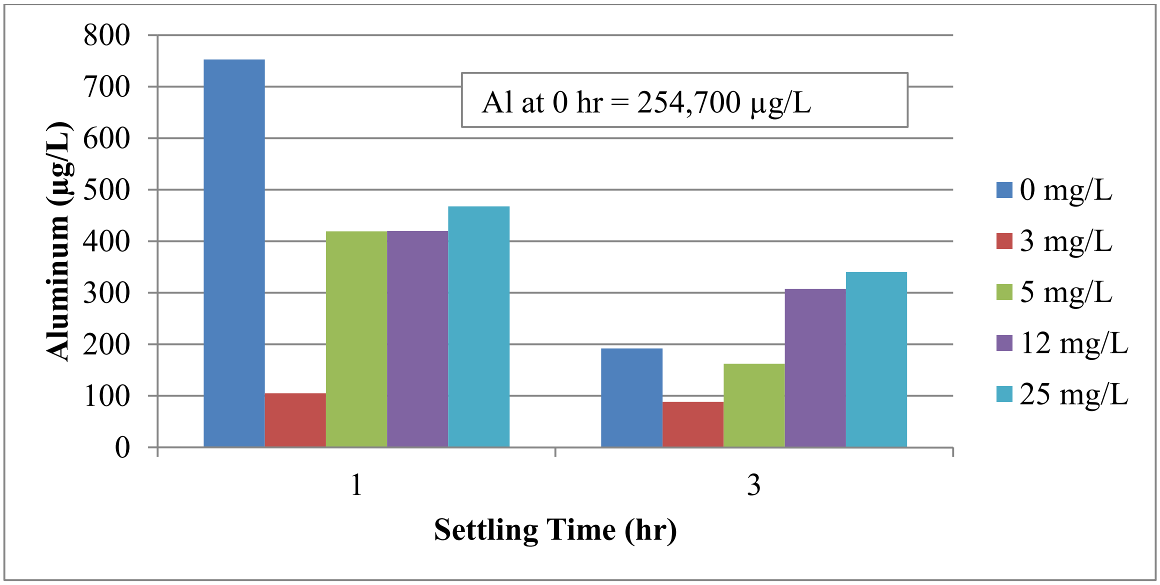

Figure 22 Settling Test with Polyfloc AP1103 at 0 RPM

The 3 mg/L dose of polymer had the best removal of aluminum both the 1 and 3 hour sample interval with aluminum concentrations of 105.5 and 88.28 µg/L compared to aluminum concentrations of 752.7 and 192.3 µg/L with no polymer.

Figure 23 shows the aluminum concentration after 1 and 3 hours of settling at 0 RPM with doses of the non-ionic polymer Magnafloc LT20 ranging from 0 to 25 mg/L. The concentration of Al in the CFBW used for this test was 94,110 µg/L.

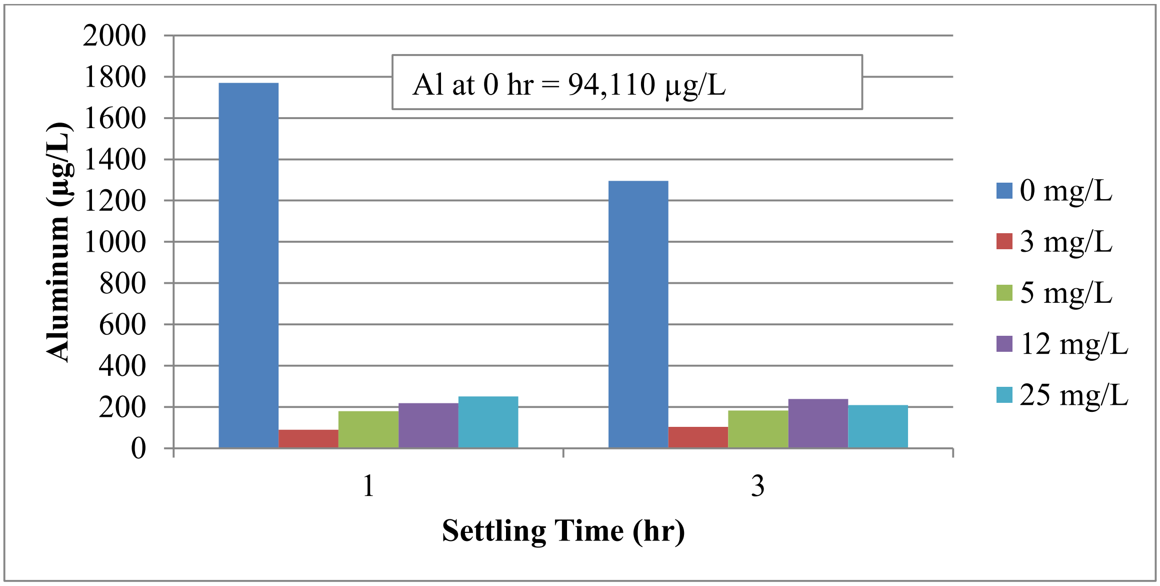

Figure 23 Settling Test with Magnafloc LT20 at 0 RPM

All doses of polymer Magnafloc LT20 showed significant removal after 1 and 3 hours of settling compared to no polymer. After 1 hour, aluminum concentrations with varying polymer doses ranged from 89.59 to 250.2 µg/L compared to a concentration of 1771 µg/L with no polymer. After 3 hours, aluminum concentrations with varying polymer doses ranged from 103.8 to 238 µg/L compared to a concentration of 1295 µg/L with no polymer. The 3 mg/L dose had the best aluminum removal at both the 1 and 3 hour sample interval with concentrations of 89.59 and 103.8 µg/L.

Figure 24 shows the aluminum concentration after 3 hours of settling at 0 RPM with doses of all polymers tested ranging from 0 to 25 mg/L.

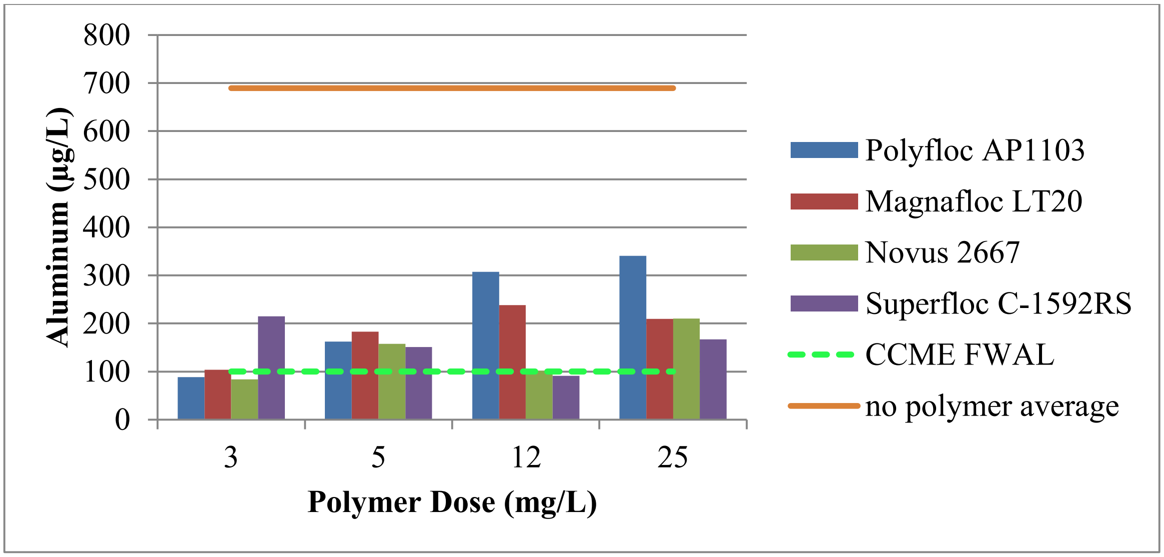

Figure 24 Aluminum Removal after 3 Hours of Settling at 0 RPM

There were 3 tests that reached an aluminum level of less than the CCME FWAL guideline of 100 µg/L. The 3 mg/L dose of cationic polymer Novus 2667 resulted in an aluminum concentration after 3 hours of 83.78 µg/L. The 12 mg/L dose of cationic polymer Superfloc C-1592RS resulted in an aluminum concentration of 91.25 µg/L. The 3 mg/L dose of anionic polymer Polyfloc AP1103 resulted in an aluminum concentration of 88.28 µg/L. None of the settling tests conducted without polymer added reached an aluminum level of less than 100 µg/L; the average aluminum concentration was 689 µg/L.

It is difficult to make a direct comparison between different polymers as the starting concentration of aluminum in the FBW was different for each test.

The results of settling tests conducted with the polymers above with settling at 5RPM are presented as Figures A2 through A6 in Appendix A.

5.2.1.1 Comparison of Polymer Settling With and Without Mixing

Table 15 compares the aluminum concentration at 3 hours with all doses tested for each polymer with 0 RPM and 5 RPM mixing during settling.

Table 15 Aluminum Concentration at 3 hours with 0 RPM and 5 RPM Mixing

Dose (mg/L)	Aluminum Concentration at 3 hr (µg/L)							
	Polyfloc AP1103		Magnafloc LT20		Novus 2667		Superfloc C-1592RS	
	0 RPM	5 RPM	0 RPM	5 RPM	0 RPM	5 RPM	0 RPM	5 RPM
0	192.3	2110	1295	446.5	1107	184	162.7	1250
3	88.28	142.7	103.8	214.1	83.78	148.3	214.7	1409
5	162.6	131.3	182.6	114.1	157.3	121.5	150.9	976.5
12	307.4	225	238	172.2	101.5	97.29	91.25	1796
25	340.5	843.7	209.3	188.5	209.9	144	166.8	131.7

Figures 25 through 28 show the aluminum concentration after 3 hours of settling at 0 RPM and 5 RPM with doses of each polymer tested ranging from 0 to 25 mg/L. A one-way ANOVA found that the settling efficiency between settling at 0 RPM and 5 RPM with and without polymer added was not statistically significant ($p = 0.07$ and $p = 0.58$, respectively).

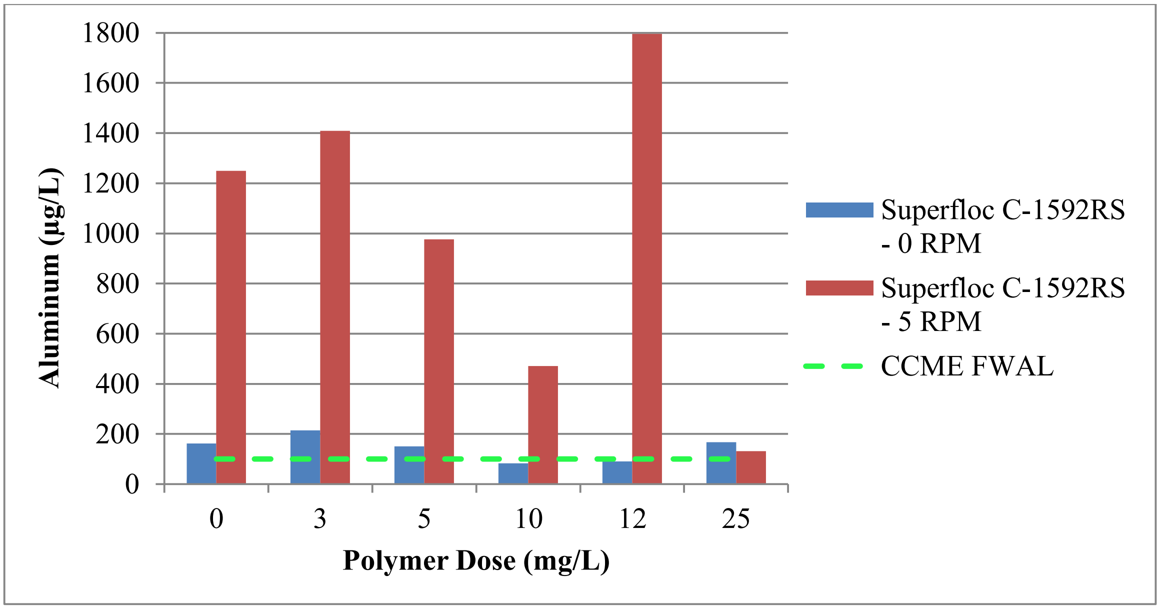

Figure 25 Aluminum Concentration with 0 RPM and 5 RPM Mixing with Superfloc C-1592RS

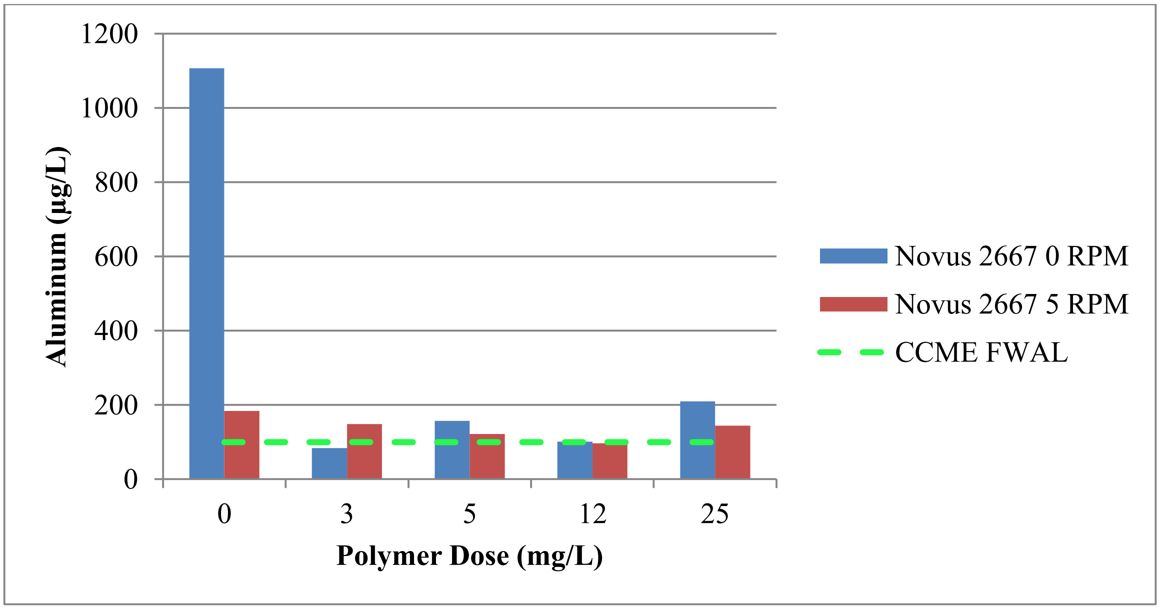

Figure 26 Aluminum Concentration with 0 RPM and 5 RPM Mixing with Novus 2667

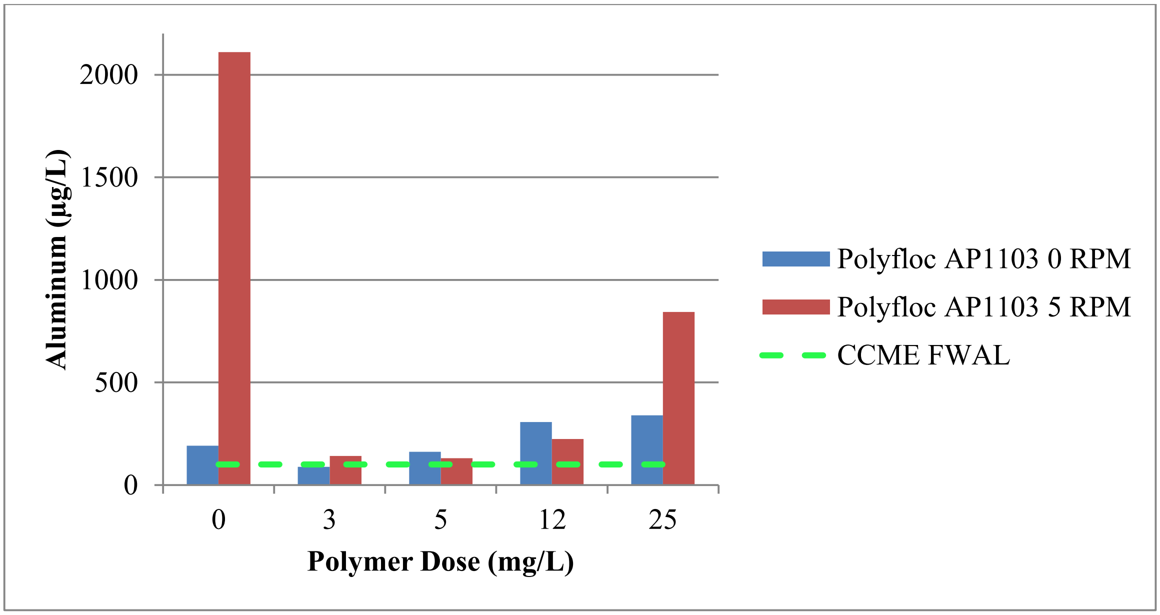

Figure 27 Aluminum Concentration with 0 RPM and 5 RPM Mixing with Polyfloc AP1103

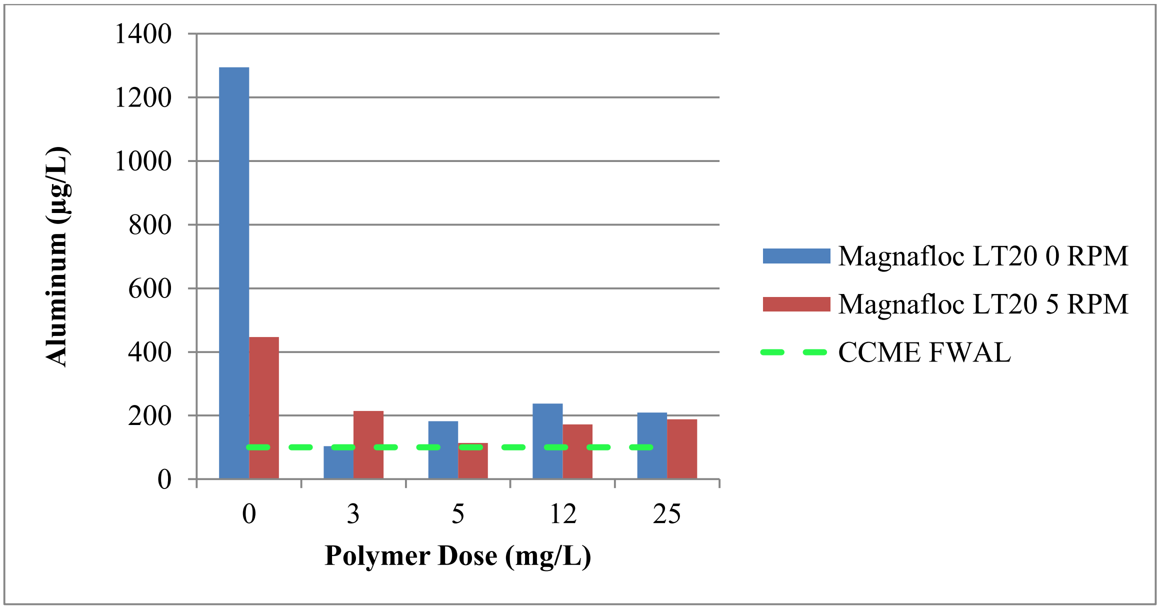

Figure 28 Aluminum Concentration with 0 RPM and 5 RPM Mixing with Magnafloc LT20

5.2.2 Cation Ratio Optimization Settling Test Results

The total concentrations of potassium, sodium, calcium and magnesium in each of the jars during the jar test are provided below in Table 16 along with the calculated M:D ratio and the aluminum concentration after 3 and 24 hours of settling. The calculated M:D ratio is the ratio of the sum of the concentrations of sodium and potassium in meq/L to the sum of the concentrations of calcium and magnesium in meq/L. The concentrations include the measured concentrations in the unamended CFBW and the chemical doses that were added during the test.

Table 16 Cation Ratio Results for Settling Test

Date	K	Na	Ca	Mg	M:D Ratio	Al (3 hr)	Al (24 hr)
	mg/L	mg/L	mg/L	mg/L	unitless	µg/L	µg/L
02-May-15	2.2	8.5	32.0	4.3	0.22	4020	1053
02-May-15	2.2	8.5	84.1	4.3	0.09	3379	1246
02-May-15	2.2	20.3	32.0	4.3	0.48	2782	1012
02-May-15	2.2	43.9	32.0	4.3	1.01	2623	891.2
02-May-15	2.2	138.3	32.0	4.3	3.11	1842	756
02-May-15	2.2	8.5	32.0	4.3	0.22	2725	1192
02-May-15	2.2	8.5	100.1	4.3	0.08	3294	1107
02-May-15	2.2	20.3	32.0	4.3	0.48	2201	868.6
02-May-15	2.2	47.8	32.0	4.3	1.10	2372	763.5
02-May-15	2.2	130.5	32.0	4.3	2.94	1393	705

The aluminum concentrations in each jar after each settling interval are listed in Table B1 in Appendix B. The results are for total aluminum unless otherwise specified. Filtered samples were collected and analyzed after three hours and 24 hours of settling.

The aluminum concentrations after 3 and 24 hours of settling are plotted against the ratio of monovalent to divalent ions in meq/L in Figure 29. There was no trend observed that indicated that aluminum removal improved with a decreasing M:D ratio. Rather, aluminum removal appeared to increase with increasing M:D ratio.

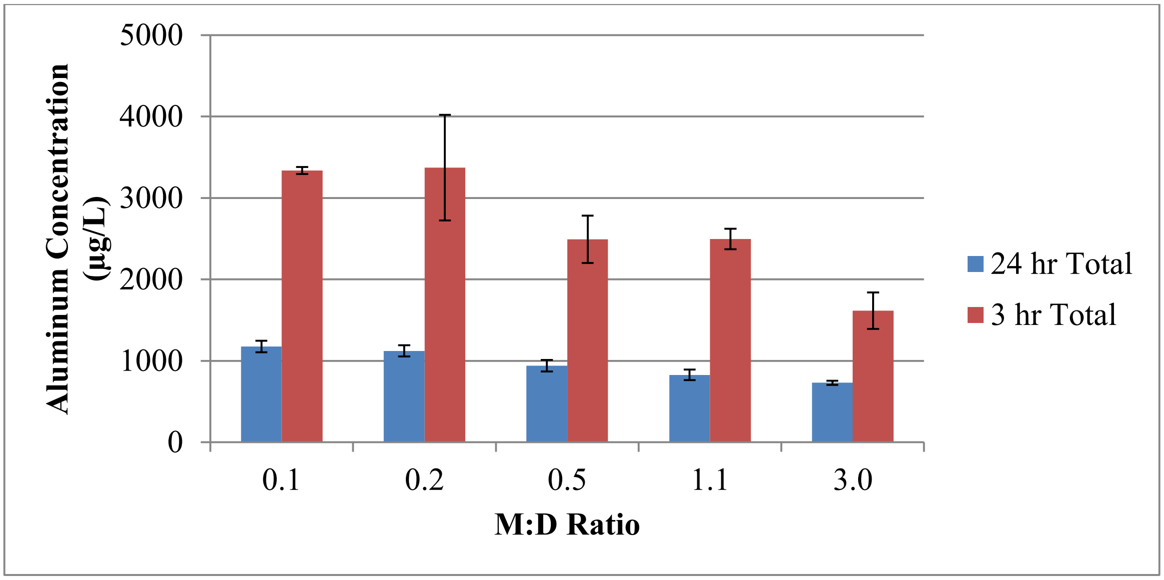

Figure 29 Aluminum Concentrations with Varying M:D Ratios

In order to determine whether the variation in aluminum removal was due to an impact to the settling efficiency or an impact to the aluminum solubility, a plot of the total aluminum concentration and dissolved aluminum concentration after 24 hours of settling is shown below in Figure 30. Although there is variation in dissolved aluminum concentration, the decrease in total aluminum concentration that was noted with increasing M:D ratios was not explained by a corresponding decrease in aluminum solubility.

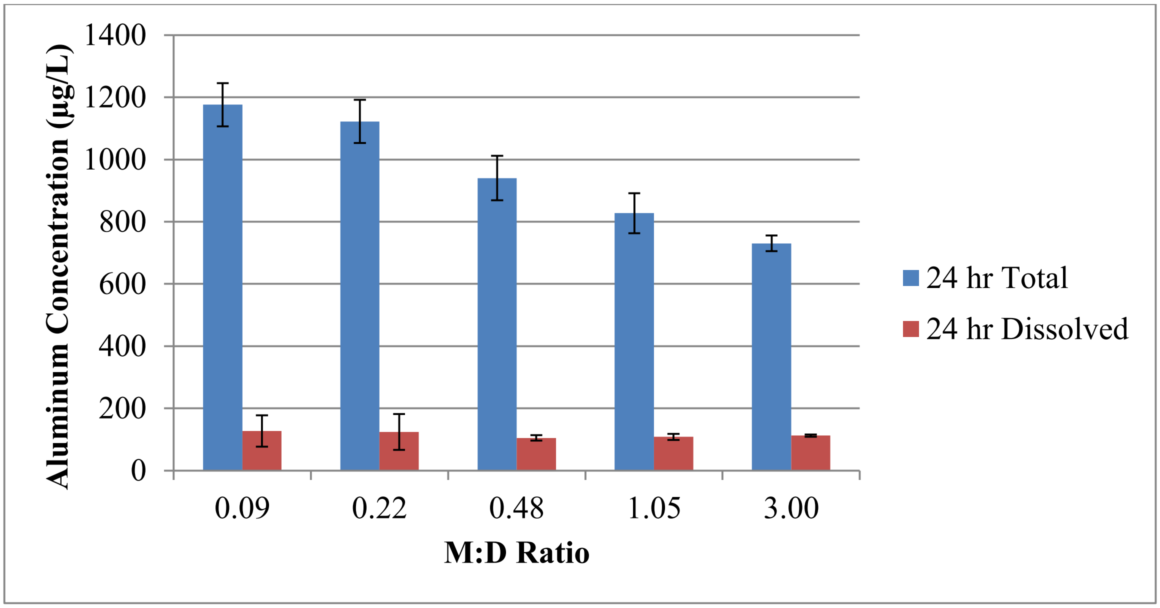

Figure 30 Total and Dissolved Aluminum after 24 hours of Settling

In order to determine whether there was an increase in dissolved aluminum during the test, a plot of the dissolved aluminum concentration after 3 hours of settling and the dissolved aluminum concentration after 24 hours of settling is shown below in Figure 31. The average aluminum concentration at each M:D ratio increased from 3 hours to 24 hours of settling.

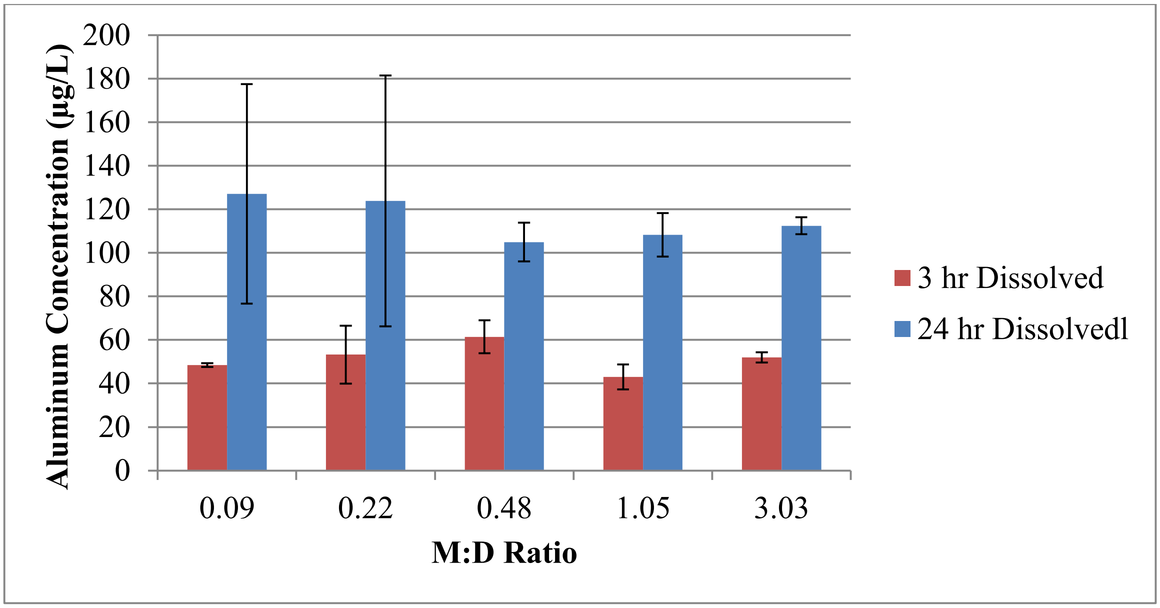

Figure 31 Dissolved Aluminum after 3 and 24 hours of Settling

5.3 CHARACTERIZATION OF SOLIDS RESIDUAL (SLUDGE)

5.3.1 Solids Content of Solids Residual

The results of the solids content analysis of the solids residual in the lagoons and drying bed is summarized in Table 17 and Figure 32.

Table 17 Solids Content of Solids Residual

Source	Layer	Solids Content (wt%)	Average (wt%)
Lagoon	Top	1.15	1.0
		1.01	
		0.96	
		0.95	
	Bottom	2.81	2.7
		3.00	
		2.54	
		2.47	
Drying Bed	Top	35.92	36.9
		35.55	
		39.36	
	Bottom	15.93	24.2
		25.10	
		31.65	

The solids residual in the lagoons was found to have an average solids content of 1% by mass with the exception of a thin layer at the bottom which was found to have an average solids content of 2.7% by mass. The solids residual in the drying bed was observed to have two distinct layers: the top layer had an average solid content of 36.9% by mass and the bottom layer had an average solids content of 24.2% by mass.

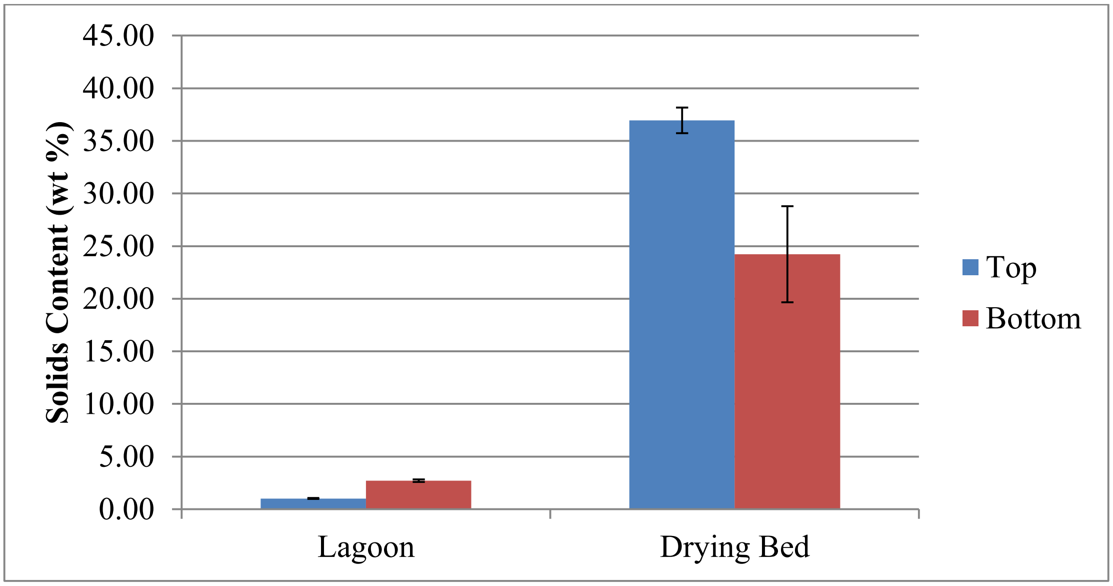

Figure 32 Solids Content of Solids Residual

5.3.2 Metals Content of Solids Residual

The analytical results of the metals concentrations in the solids residual from the drying bed from May 30, 2013, July 30, 2014 and November 20, 2014 are provided in Table 18 for the metals that were detected. The full table of results is included as Table B2 in Appendix B. Results are compared to CCME Soil Quality Guidelines for Residential/Parkland, Commercial and Industrial Sites as a reference (CCME, 1999c).

Table 18 Metals Content of Solids Residual – Detected Metals

Parameter	CCME Soil Quality Guidelines		Sample Results		
	Residential / parkland	Commercial	30-May-13	30-Jul-14	20-Nov-14
Aluminum (Al)	NG	NG	160000	180000	190000
Arsenic (As)	12	12	7.4	**15**	**15**
Barium (Ba)	500	2000	74	160	40
Beryllium (Be)	4	8	1	<2.0	<2.0
Cadmium (Cd)	10	22	0.46	<0.30	<0.30
Calcium (Ca)	NG	NG	4100	4700	2800
Chromium (Cr)	64	87	**110**	**130**	**69**
Cobalt (Co)	50	300	5.6	5.0	4.1
Copper (Cu)	63	91	<10	8.4	7.9
Iron (Fe)	NG	NG	19000	33000	30000
Lead (Pb)	140	260	12	12	11
Magnesium (Mg)	NG	NG	380	1000	850
Manganese (Mn)	NG	NG	7300	9400	5300
Mercury (Hg)	6.6	24	<0.1	<0.10	0.12
Nickel (Ni)	50	50	9.7	9.2	9.1
Phosphorus (P)	NG	NG	700	NM	910
Potassium (K)	NG	NG	<350	100	<100
Selenium (Se)	1	2.9	**3.2**	**1.7**	**1.7**
Strontium (Sr)	NG	NG	17	23	13
Titanium (Ti)	NG	NG	110	NM	NM
Uranium (U)	23	33	2.7	2.2	1.9
Vanadium (V)	130	130	56	85	59
Zinc (Zn)	200	360	180	170	140

BOLD	Exceedance of the CCME Commercial Soil Quality Guidelines.
BOLD	Exceedance of the CCME Residential Soil Quality Guidelines.

NG: no applicable guideline; NM: parameter was not measured.

Selenium exceeded the CCME soil quality guideline of 1 mg/kg for residential sites in all three samples and exceeded the CCME soil quality guideline of 2.9 mg/kg for commercial and industrial sites in one sample. The exceedances of the residential soil quality guideline ranged from 2.9 to 3.6 times the guideline and the exceedance of the commercial and industrial soil quality guideline was 1.2 times the guideline.

Arsenic exceeded the CCME soil quality guideline of 12 mg/kg for residential, commercial and industrial sites in two of the three samples. The exceedances of the residential, commercial and industrial soil quality guideline were 1.3 times the guideline. However, there have been reports that the background soil concentration in Cape Breton for arsenic is above CCME soil quality guidelines. Background sampling conducted by JDAC as part of the Human Health Risk Assessment for the Area North of Coke Ovens found that the arithmetic mean of the urban reference area surface soil concentration of arsenic was 28 mg/kg (JDAC, 2001).

Chromium exceeded the CCME soil quality guideline of 64 mg/kg for residential sites in all three samples and exceeded the CCME soil quality guideline of 87 mg/kg for commercial and industrial sites in two of the three samples. The exceedances of the residential soil quality guideline ranged from 1.1 to 2.0 times the guideline. The exceedances of the commercial and industrial soil quality guideline ranged from 1.3 to 1.5 times the guideline. The source of the elevated chromium levels was studied as part of this thesis.

5.3.3 Leachability Testing

Solids residual samples were collected in November 2014 from a batch of solids residual in the drying bed that had accumulated in the lagoons from September 2013 to September 2014 and had been drying in the drying bed since September 2014. As leachate analysis is performed on the samples on an as-received basis, two samples were dried in an oven at 104 °C for 48 hours prior to analysis and one sample was analyzed as collected. The results of the detected parameters in the SPLP analysis plus chromium are presented in Table 19. The full analytical results are presented as Table B3 in Appendix B. Analytical results were compared to CCME FWAL and Marine water guidelines for reference.

Table 19 SPLP Analysis of Solids Residual

Metals (Leachable)	CCME Marine (µg/L)	CCME FWAL (µg/L)	SPLP (µg/L)		
			20-Nov-14	20-Nov-14(3)	20-Nov-14(3)
Aluminum	NG	100	69	**210**	**210**
Arsenic	12.5	5	<2.0	**7.5**	**12**
Barium	NG	NG	24	130	140
Calcium	NG	NG	6000	17000	25000
Chromium	56(1)	8.9(1)	<2.0	<2.0	<2.0
Cobalt	NG	NG	<1.0	<1.0	1.4
Magnesium	NG	NG	300	1400	9200
Manganese	NG	NG	9700	22000	51000
Potassium	NG	NG	140	960	2000
Sodium	NG	NG	1700	2600	4100
Strontium	NG	NG	5.1	41	77
Zinc	NG	30	<5.0	<20	21

Notes

BOLD Indicates an exceedance of the CCME FWAL criteria.

(1) Criteria is for total chromium, not hexavalent chromium.

(2) Criteria is dependent on water hardness. Listed criteria is for unknown water hardness. (3) Sample was oven dried at 104 °C prior to being analyzed.

The dried November 20, 2014 samples exceeded CCME FWAL guidelines for aluminum and arsenic; there were no exceedances of CCME marine guidelines. The November 20, 2014 SPLP sample that was not dried prior to analysis did not exceed CCME FWAL or marine guidelines. The chromium concentration was less than the detection limit for each of the samples.

5.3.4 Solids Residual Amendments

Solids residual from the drying bed was collected on November 20, 2014 and dried in an oven at 104 °C for 48 hours. Samples of the oven dried solids residual were blended with 2%, 5% and 10% slag obtained from Portside Aggregates in Sydney, Nova Scotia and submitted to Maxxam Analytics to be analyzed for SPLP with metals analysis. A sample with no added slag was also submitted for the same analysis. The analytical results for all metals that were detected in the leachate plus chromium are provided in Table 20.

Table 20 Solids Residual Amendment SPLP Analytical Results

Metals (Leachable)	Units	Slag Composition (wt%)			
	wt%	**0**	**2**	**5**	**10**
Aluminum (Al)	µg/L	210	230	150	180
Arsenic (As)	µg/L	7.5	7.6	7.2	6.5
Barium (Ba)	µg/L	130	110	120	88
Calcium (Ca)	µg/L	17000	34000	61000	74000
Chromium	µg/L	<2.0	<2.0	<2.0	<2.0
Magnesium (Mg)	µg/L	1400	1500	1500	1400
Manganese (Mn)	µg/L	22000	17000	12000	8600
Potassium (K)	µg/L	960	950	780	830
Sodium (Na)	µg/L	2600	3700	4400	4100
Strontium (Sr)	µg/L	41	68	120	140

Plots of the metals concentration in the leachate at each slag concentration are provided in Figures 33, 34 and 35. A plot of the pH of the leachate at each slag concentration is provided in Figure 36. Manganese was the only parameter that showed a consistent trend of decreasing leachability with increasing slag concentration. The manganese concentration varied from 22000 µg/L with 0% slag to 8600 µg/L with 10% slag, a decrease of 61%. A trend of increasing leachability with increasing slag concentration was noted for calcium and strontium. The calcium concentration ranged from 17000 µg/L with 0% slag to 74000 µg/L with 10% slag, an increase of 335%. The strontium concentration ranged from 41 µg/L with 0% slag to 140 µg/L with 10% slag, an increase of 241%. This could be due to the presence of calcium and strontium in the slag. The pH ranged from 6.22 with 0% slag to 7.45 with 10% slag.

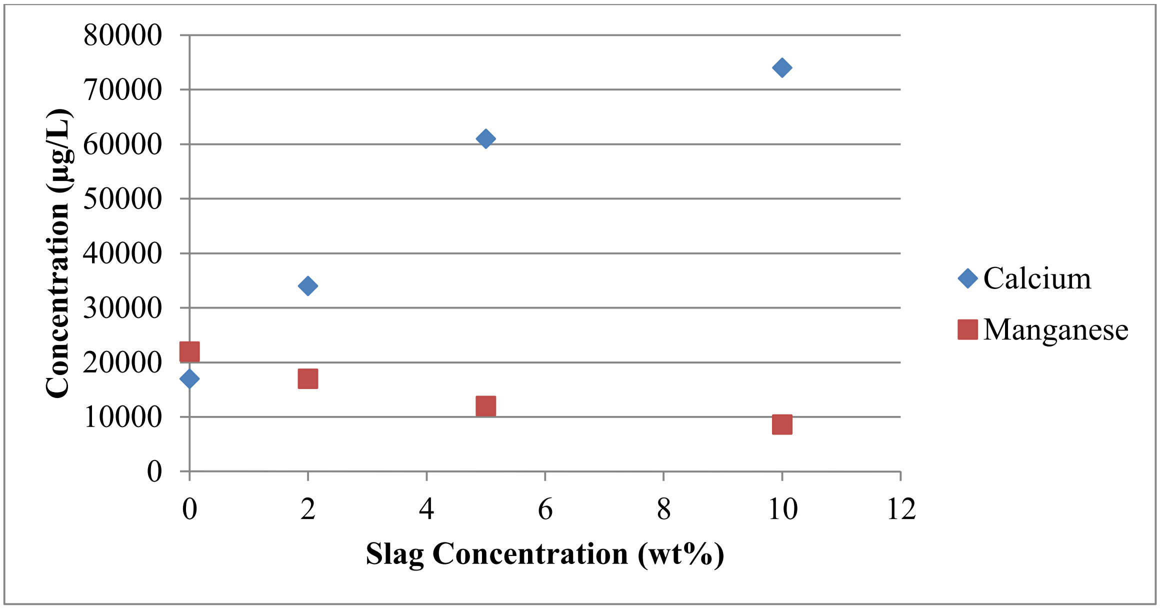

Figure 33 Calcium and Manganese Concentrations in Leachate

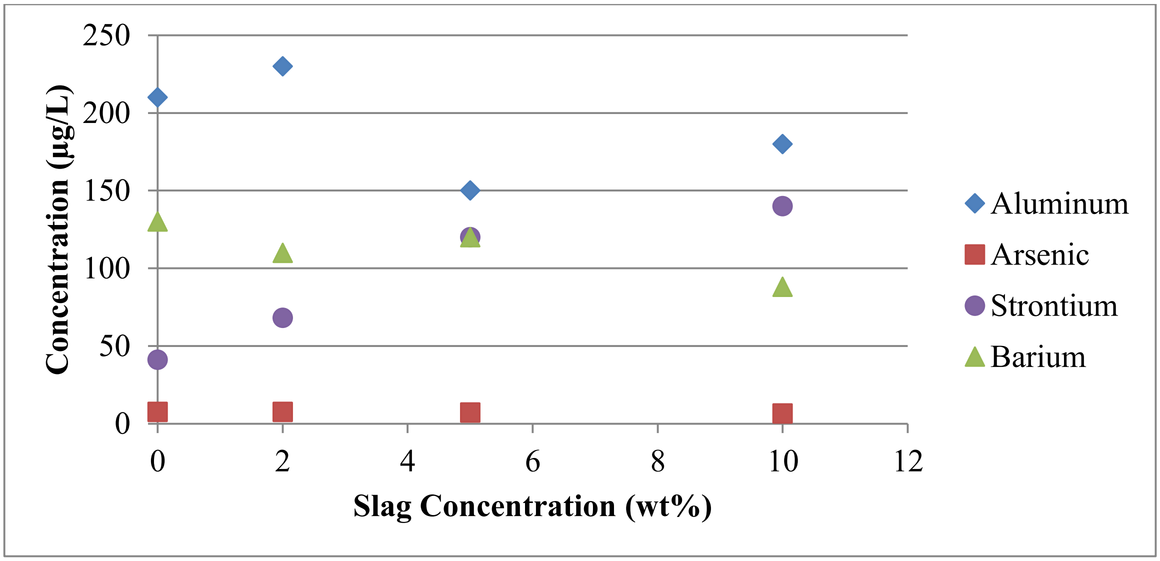

Figure 34 Aluminum, Arsenic, Strontium and Barium Concentrations in Leachate

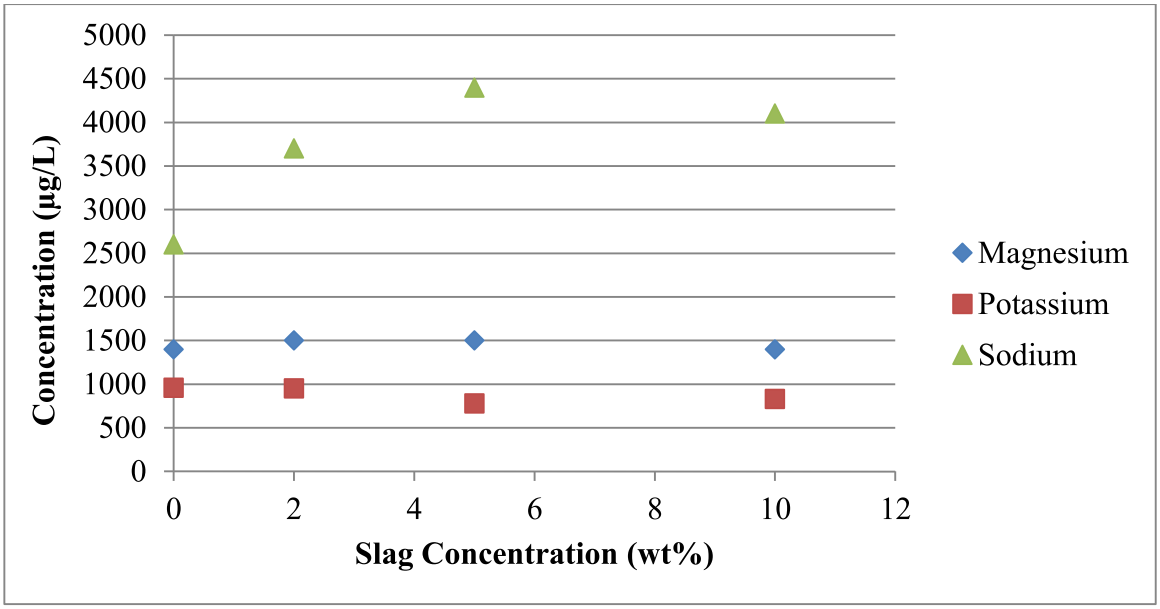

Figure 35 Magnesium, Potassium and Sodium Concentrations in Leachate

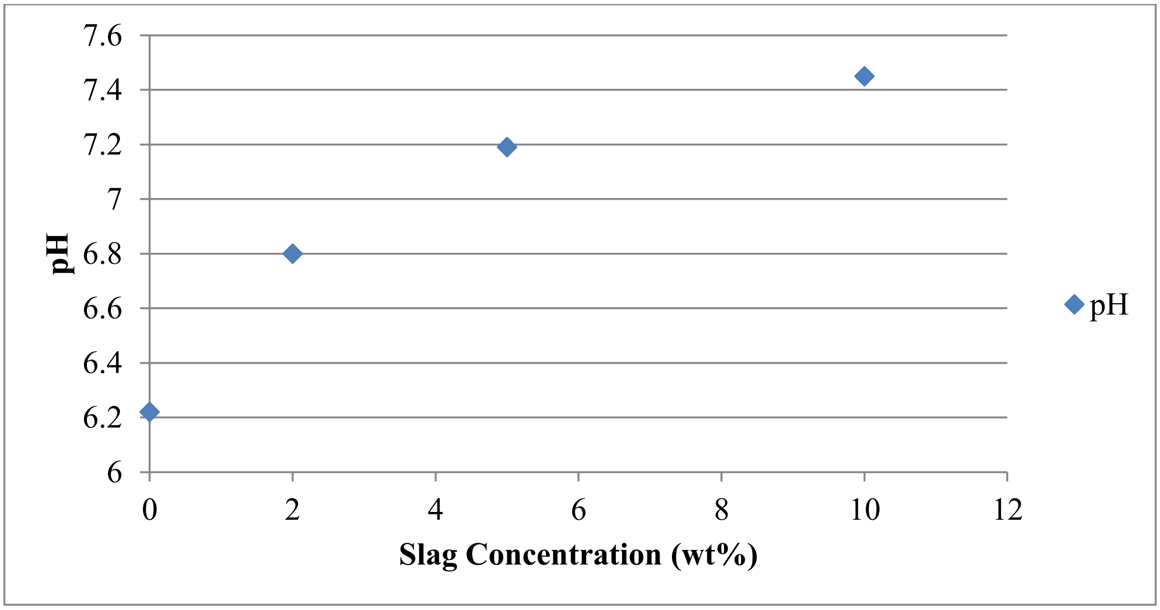

Figure 36 pH of Leachate with Slag Amendment

5.4 INVESTIGATION TO DETERMINE SOURCE OF CHROMIUM IN SOLIDS RESIDUAL

A sampling program was conducted to identify potential sources of the elevated levels of chromium in the solids residual. A mass balance was used to determine whether chromium levels detected in plant flows could account for the elevated chromium levels in the solids residual. As part of the mass balance, an estimate of the solids residual production rate was calculated.

5.4.1 Sampling Program

5.4.1.1 Plant Flow Sample Results

The chromium results of all water samples collected throughout the treatment process are provided in Table 21. Additional raw water samples were collected in November and December 2014 and digested with nitric acid prior to analysis in order to determine whether there was chromium present above the detection that was associated with colloids and was not being measured by the ICP-MS.

Table 21 Plant Flow Analytical Results for Chromium

Location	Units	May 12 2014	May 14 2014	May 21 2014	Nov 18 2014	Dec 9 2014	Dec 12 2014
Raw	µg/L	<0.4	<0.4	<0.4	<0.4	<0.4	<0.4
	µg/L	<0.4	<0.4	<0.4	0.636(1)	0.509(1)	<0.4(1)
Pre-treatment Tank 1	µg/L	<0.4	<0.4	<0.4	-	-	-
Pre-treatment Tank 2	µg/L	<0.4	<0.4	<0.4	-	-	-
Flocculant Tank	µg/L	<0.4	<0.4	<0.4	-	-	-
Clarifier	µg/L	<0.4	<0.4	<0.4	-	-	-
Post-filter	µg/L	<0.4	1.913	<0.4	-	-	-
Clearwell	µg/L	<0.4	<0.4	<0.4	-	-	-
Lagoon Effluent	µg/L	<0.4	-	<0.4	-	-	-

Notes:
(1) Sample was digested prior to analysis.

Most sample results were below the detection limit of the instrument for chromium with the exception of a post-filtered sample on May 14, 2014 and the raw water samples collected on November 18 and December 9 that had been digested with nitric acid. The ICP-MS results showed that although they were below the detection limit of the instrument, the chromium results for pre-treatment tank 2 appeared elevated above the other results. As polymer and aluminum sulfate (alum) are added to the process in pre-treatment tank 2, samples of these chemicals were collected to analyze for chromium. The sample results are provided in Tables 22 and 23.

Table 22 Alum Analytical Results for Chromium

Sample	Chromium µg/L
alum 1	866.3
alum 2	768.6
alum 3	802.2
alum 4	761.8
alum 5	757.9
Average:	**791.4**

Table 23 Polymer Analytical Results for Chromium

Sample	Chromium µg/L
polymer 1	0.691
polymer 2	0.788
Average:	**0.740**

The average chromium concentration for the Alum samples was 791.4 µg/L. The average chromium concentration for the polymer samples was 0.740 µg/L. Samples of CFBW and clarifier sludge were analyzed in order to determine the level of chromium in the residuals streams and confirm that the chromium in the solids residual originated from the residuals streams and not from a source exterior to the treatment process. The analytical results are presented in Table 24. All residuals samples were digested with nitric acid prior to analysis.

Table 24 Residuals Streams Analytical Results for Chromium

Sample	Clarifier Sludge	FBW
	µg/L	µg/L
09-Sep-14		13.19(1)
11-Sep-14		15.65(1)
15-Oct-14		16.75(1)
16-Oct-14		31.29(1)
04-Nov-14		34.79(1)
12-Nov-14		27.54(1)
18-Nov-14	11.7(1)	19.39(1)
08-Dec-14		19.03(1)
08-Dec-14		5.117(1)
08-Dec-14		8.835(1)

Notes:
(1) Sample was digested prior to analysis.

5.4.1.2 Source Water Sample Results

A sample was collected from each of 7 streams upstream of the reservoir to determine if chromium was present in an amount higher than the detection limit in any of the streams. The locations of the samples are shown on the attached Figure A1 in Appendix A. Chromium was detected in the stream labelled SS-1 that flows into the reservoir from the north east and the stream labelled SS-6 that flows into the reservoir from the south west in levels above the detection limit of 0.4 µg/L as shown in Table 25. The results of the samples from the rest of the streams were less than the reportable detection limit. The stream samples were also analyzed for aluminum, manganese and iron in order to identify any potential sources of impacted water flowing into the reservoir and the results are presented in Figure 37.

Table 25 Analytical Results for Chromium in Source Water Streams

Sample Location	Chromium (µg/L)
Stream Sample 1 (SS1)	0.515
Stream Sample 2 (SS2)	<0.4
Stream Sample 3 (SS3)	<0.4
Stream Sample 4 (SS4)	<0.4
Stream Sample 5 (SS5)	<0.4
Stream Sample 6 (SS6)	0.448
Stream Sample 7 (SS7)	<0.4

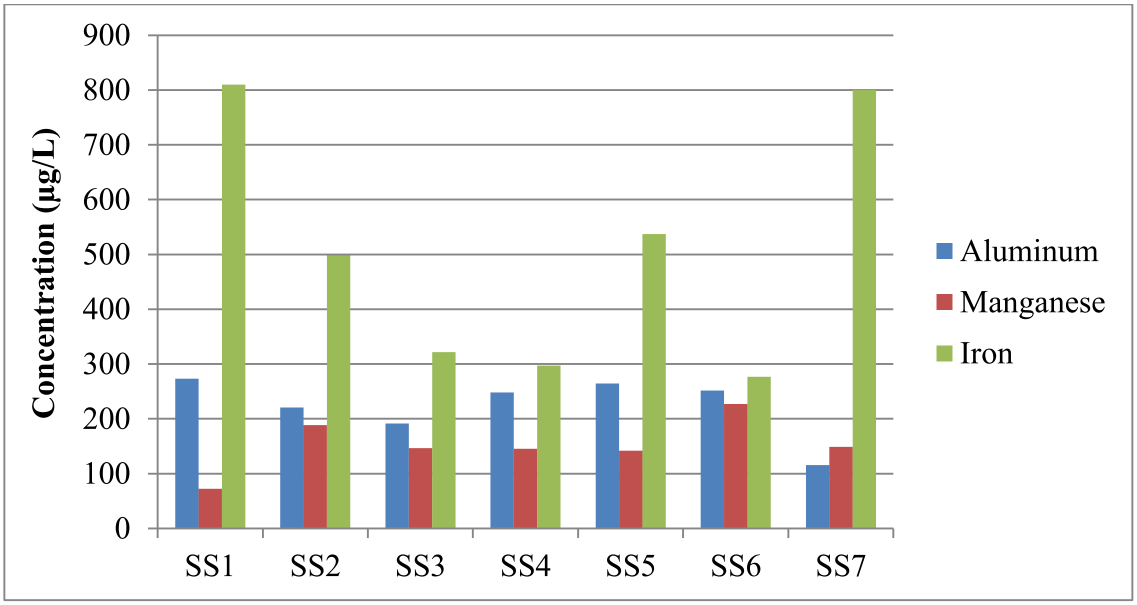

Figure 37 Analytical Results for Metals in Source Water Streams

Aluminum was detected in all streams in levels ranging from 115.5 to 273.1 µg/L. All sample results for aluminum were above the CCME FWAL guideline of 100 µg/L for waters with a pH of greater than 6.5. Manganese was detected in all streams in levels ranging from 72.27 to 227 µg/L. There is no CCME FWAL guideline for manganese. Iron was detected in all streams in levels ranging from 276.6 to 810.1 µg/L. All sample results for iron except sample SS6 were above the CCME FWAL guideline of 300 µg/L.

5.4.2 Mass Balance

A mass balance was performed in order to determine if the level of chromium in the solids residual could be explained by the chromium concentration in the aluminum sulfate as well as to determine what the chromium concentration would have to be in the raw water to explain the level of chromium in the solids residual. The raw water and treated water flow meters at the plant reported annual daily average flowrates of 9230 m^3/day and 9418 m^3/day, respectively. The treated flow rate should be lower than the raw water flowrate due to water being removed from the process as residuals which have a total daily flowrate of 540 m^3/day. The discrepancy could be due to the inaccuracy of the flow meters. Therefore, mass balance calculations were done using low and high estimates of the plant flowrates as summarized in Table 26.

Table 26 Summary of Plant Flowrates for Mass Balance

Flow	Units	Low	High
Raw Water	m^3/day	9230(1)	9958(2)
Treated Water	m^3/day	8690(3)	9418(4)
Residuals Flow	m^3/day	540(5)	

Notes:
(1) From raw water flow meter.
(2) Sum of flow from treated water flow meter and residuals flow.
(3) Difference in flow between raw water flow meter and residuals flow.
(4) From treated water flow meter.
(5) Sum of clarifier sludge flow, filter backwash flow, and filter to waste.

The chromium load in the alum can be determined by multiplying the alum dose by the chromium concentration in the alum by the raw water flowrate. The estimated chromium load from the aluminum sulfate is between 0.31 and 0.34 g/day as shown in Table 27. See Table B4 in Appendix B for values used in the calculation.

Table 27 Chromium Load from Aluminum Sulfate

Parameter	Units	Result
Chromium Load from Aluminum Sulfate (low)	**g/day**	**0.31**
Chromium Load from Aluminum Sulfate (high)	**g/day**	**0.34**

The aluminum load in the alum can be determined by multiplying the alum dose by the amount of aluminum in the alum by the raw water flowrate. The estimated aluminum load from the alum was determined to be between 23.5 and 25.4 kg/day as shown in Table 28. See Table B5 in Appendix B for values used in the calculation.

Table 28 Aluminum Load from Aluminum Sulfate

Parameter	Units	Result
Aluminum Load from Aluminum Sulfate (low)	**kg/day**	**23.5**
Aluminum Load from Aluminum Sulfate (high)	**kg/day**	**25.4**

The estimated aluminum load in the lagoon effluent of 0.06 kg/day was determined by multiplying the annual average aluminum concentration in the lagoon effluent by the flowrate of the lagoon effluent (Table 29). The estimated aluminum load in the raw water of 1.4 to 1.5 kg/day was determined by multiplying the aluminum concentration in the raw water by the flowrate of the raw water (Table 30). The estimated aluminum load in the treated water of between 0.24 to 0.26 kg/day was determined by multiplying the aluminum concentration in the treated water by the flowrate of the treated water (Table 31).

Table 29 Aluminum Load in Lagoon Effluent

Parameter	Units	Result
Aluminum Concentration in Lagoon Effluent (yearly average)	µg/L	120
Lagoon Effluent Flow	LPD	540000
Aluminum Load in Lagoon Effluent	**kg/day**	**0.06**

Table 30 Aluminum Load from Raw Water

Parameter	Units	Result
Raw Flowrate (low)	LPD	9230000
Raw Flowrate (high)	LPD	9958000
Aluminum Concentration in Raw	µg/L	149
Aluminum Load from Raw (low)	**kg/day**	**1.4**
Aluminum Load from Raw (high)	**kg/day**	**1.5**

Table 31 Aluminum Load in Treated Water

Parameter	**Units**	**Result**
Aluminum Concentration in Treated Water	µg/L	28
Treated Water Flow (low)	LPD	8690000
Treated Water Flow (high)	LPD	9418000
Aluminum Load in Treated Water (low)	**kg/day**	**0.24**
Aluminum Load in Treated Water (high)	**kg/day**	**0.26**

Solids Residual Production Rate (Dry Basis)

It is difficult to determine the solids residual production rate as the solids residual is comprised of settled solids from the filter backwash water and clarifier sludge. The flowrate and TSS of the filter backwash water varies during the backwash cycle and it is difficult to get a flow proportionate composite sample for TSS analysis. Therefore, an alternative approach was taken to estimate the amount of solids residual produced. As the aluminum concentration primarily comes from the aluminum sulfate dose, and there are also measurements available for the aluminum concentration in the raw water, treated water, lagoon effluent, and drying bed solids residual (on a dry basis), an aluminum balance was performed in order to estimate the production rate of solids residual on a dry basis. Figure 38 provides a visual representation of the mass balance that was performed. The blue labels represent known parameters and the red label represents the unknown parameter.

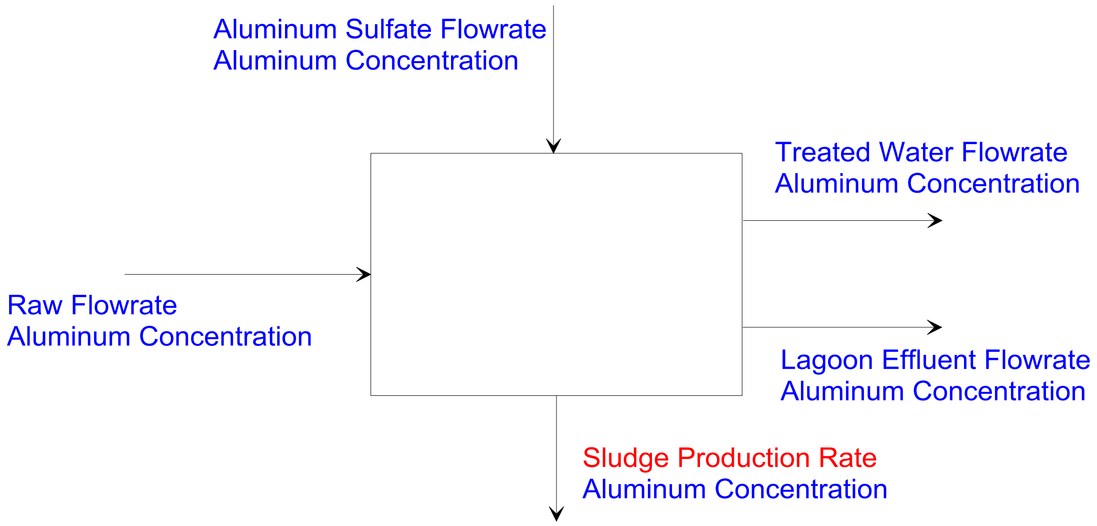

Figure 38 Mass Balance to Determine Solids Residual Production Rate

A “low” estimate of the calculated aluminum load in the solids residual was determined by using “low” values for inputs from the system (aluminum sulfate, raw water) and “high” values for outputs to the system (lagoon effluent, treated water). In contrast, a “high” estimate of the calculated aluminum load in the solids residual was determined by using “high” values for inputs to the system and “low” values for outputs from the system. The aluminum load in the solids residual was calculated to be between 24.6 and 26.6 kg/day (Table 32). The solids residual production rate was determined by dividing the aluminum load in the solids residual by the aluminum concentration in the solids residual. A “low” estimate of the solids residual production rate was obtained using the low aluminum load and the high aluminum concentration, and a “high” estimate was obtained using the high aluminum load and the low aluminum concentration. The solids residual production rate was calculated to be between 130 and 166 kg/day on a dry basis, which corresponds to 432 to 554 kg/day with a solids content of 30% by mass (Table 33).

Table 32 Mass Balance to Determine Aluminum Load in Solids Residual

Parameter	Units	Result	
		Low	High
Aluminum Load from Aluminum Sulfate	kg/day	23.56	25.42
Aluminum Load from Raw Water	kg/day	1.38	1.48
Aluminum Load in Treated Water	kg/day	0.24	0.26
Aluminum Load in Lagoon Effluent	kg/day	0.06	
Calculated Aluminum Load in Solids Residual	**kg/day**	**24.6**	**26.6**

Table 33 Mass Balance to Determine Solids Residual Production Rate

Parameter	Units	Result	
		Low	High
Aluminum Concentration in Solids Residual	g Al/kg Solids Residual	160	190
Aluminum Load In Solids Residual	kg/day	24.61	26.60
Solids Residual Production Rate dry basis	kg/day	130	166
Solids Residual Production Estimate at 30% Solids	**kg/day**	**432**	**554**

The calculated solids residual production rate, the chromium concentration from a metals analysis of the solids residual, the chromium load in the aluminum sulfate coagulant, and the chromium load in the raw water were used in a mass balance calculation of chromium to determine the daily chromium load to the system that is unexplained by the chromium load in the alum and raw water. A range of values was calculated for the unexplained chromium load by using "high" values for the inputs and "low" values for the outputs to determine the low end of the range and "low values" for inputs and "high" values for outputs to determine the high end of the range. The highest chromium concentration measured in the raw water of 0.636 µg/L and a chromium concentration of zero were used in calculating the high and low chromium load in the raw water, respectively. The chromium load in the solids residual that is unexplained by the chromium concentration in the raw water and the alum is within the range of 2.3 and 21.3 g/day (Table 34).

Table 34 Unexplained Chromium Load

Parameter	**Units**	**Result**	
		Low	**High**
Solids Residual Production Rate (dry basis)	kg/day	130	166
Chromium Concentration in Solids Residual	mg/kg	69	130
Chromium Load in Solids Residual	g/day	8.94	21.61
Chromium Load in Lagoon Effluent	g/day	0	
Chromium Load in Treated	g/day	0	
Chromium Load from Aluminum Sulfate	g/day	0.31	0.34
Chromium Load from Raw	g/day	0	6.33
Unexplained Chromium Load	**g/day**	**2.3**	**21.3**

As all digested chromium samples of raw water were collected in November and December, a second mass balance was performed in order to determine what the average annual chromium concentration in the raw water would have to be to account for the portion of chromium in the solids residual that is not explained by the chromium load from the aluminum sulfate. Figure 39 provides a visual representation of the mass balance that was performed. The blue labels represent known parameters and the red label represents the unknown parameter. The chromium load in the lagoon effluent and the treated water were assumed to be zero.

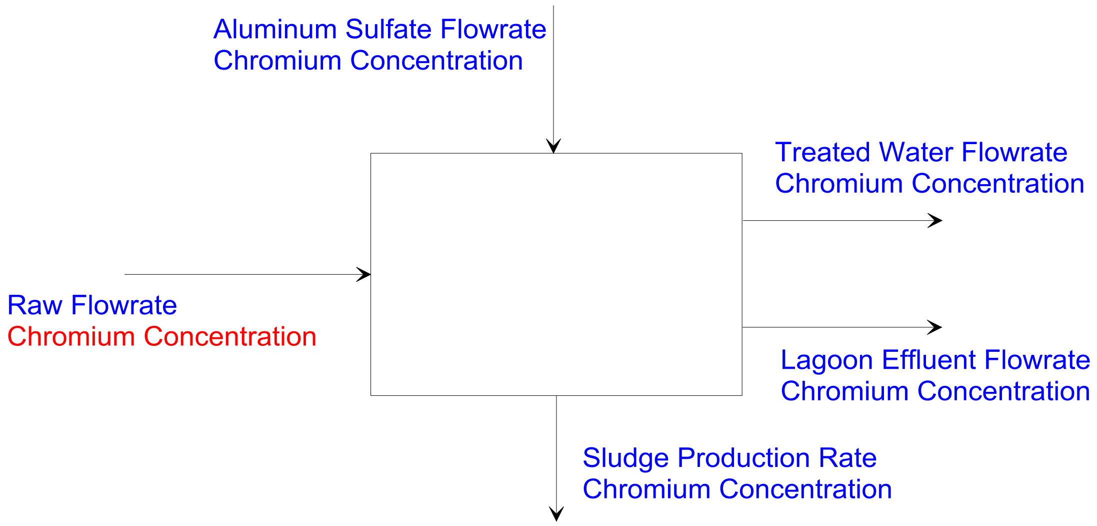

Figure 39 Mass Balance to Determine Required Chromium Concentration in Raw Water

A “low” estimate of the chromium load in the solids residual was determined by multiplying the low estimate of the solids residual production rate by the lowest chromium concentration result for the solids residual. A “high” estimate of the chromium load in the solids residual was determined by multiplying the high estimate of the solids residual production rate by the highest chromium concentration result for the solids residual.

A “low” estimate of what the chromium concentration would have to be in the raw water to account for the unexplained portion of the chromium in the solids residual was determined by using the low estimate of the chromium load in the solids residual and the high estimate of chromium load from the aluminum sulfate. A “high” estimate was obtained using the high estimate of the chromium load in the solids residual and the low estimate of the chromium load from the aluminum sulfate. The chromium load in the raw water would have to be between 0.86 and 2.3 µg/L to account for the chromium load in the solids residual that is not explained by the chromium load from the aluminum sulfate (Table 35).

Table 35 Mass Balance to Determine Required Chromium Concentration in Raw

Parameter	Units	Result	
		Low	High
Solids Residual Production Rate (dry basis)	kg/day	130	166
Chromium Concentration in Solids Residual	mg/kg	69	130
Chromium Load in Solids Residual	g/day	8.94	21.61
Chromium Load in Lagoon Effluent	g/day	0	
Chromium Load in Treated	g/day	0	
Chromium Load from Aluminum Sulfate	g/day	0.31	0.34
Required Chromium Load from Raw	**g/day**	**8.60**	**21.30**
Raw Flowrate	LPD	9230000	9958000
Required Chromium Concentration in Raw	**µg/L**	**0.86**	**2.3**

CHAPTER 6 DISCUSSION

6.1 EVALUATE LIQUID STREAM RESIDUALS TREATMENT

6.1.1 Characterization of Filter Backwash Water

At the WTP, the FBW flow is 267 m^3/day and the clarifier sludge flow is 273 m^3/day, each representing between 2.7% and 2.9% of the total volume of water processed at the plant for a total of between 5.4% and 5.8%. A range is provided because there are two estimates of the volume of water processed at the plant, which is discussed in Section 5.4.2. ASCE and AWWA (1996) estimates that filter backwash water comprises 2 to 5 % of the total water treated at the plant. Crittenden et al. (2012) estimates that residuals comprise 3 to 5% of the total water treated at the plant with filter backwash water ranging from 2 to 5% and clarifier sludge ranging from 0.08 to 0.3%. According to data provided by Bourgeois et al. (2004a), the Lake Major Water Supply Plant has a residuals stream that comprises 1.42% of the water treated at the plant including 0.74% for clarifier sludge and 0.68% for filter backwash water. The Victoria Park Water Treatment Plant in Truro, NS has a residuals stream that comprises 5.4% of the water treated at the plant including 2.7% for clarifier sludge and 2.7% for filter backwash water (Bourgeois et al., 2004a). Arora et al. (2001) found that in a survey of water treatment plants in the US, FBW ranged from 2 to 10% of plant flow with an average of 2.5%.

TSS measurements of filter backwash water ranged from 300 to 3000 mg/L with an average of 1643 mg/L. Three consecutive CFBW samples collected during a single backwash cycle on December 8, 2014 had TSS concentrations of 1321, 400, and 300 mg/L, indicating that there is a large variation in CFBW water quality over the duration of the backwash cycle. The TSS of clarifier sludge was measured as 1667 mg/L. ASCE and AWWA (1996) estimate that FBW generally contains between 50 and 400 mg/L of TSS. Crittenden et al. (2012) estimates that FBW contains on average between 100 and 1000 mg/L of TSS. Bourgeois et al. (2004a) found the TSS of CFBW water from Lake Major Water Supply Plant and Victoria Park to be 120 and 490 mg/L, respectively.

The aluminum concentration in the CFBW at the WTP ranged from 94 to 798 mg/L with an average of 361 mg/L. The aluminum concentration in a sample of clarifier sludge at the WTP was 269 mg/L. Bourgeois et al. (2004a) found that the composite residuals stream at the Lake Major Water Supply Plant contained 17 mg/L of aluminum and the composite residuals stream at Victoria Park contained 61 mg/L of aluminum. Arora et al. (2001) found that aluminum in FBW ranged from non-detect to 145.8 mg/L with an average of 14.8 mg/L.

The chromium concentration in the CFBW at the WTP ranged from 5.117 to 34.79 µg/L with an average of 19.16 µg/L. The chromium concentration in a sample of clarifier sludge at the WTP was 11.7 µg/L. Iron and manganese concentrations of 75.9 mg/L and 37.4 mg/L, respectively, were measured in a May 2, 2015 sample of CFBW. This is expected to be due to their presence in the raw water, which is removed during the treatment process.

6.1.2 Tracer Study

Assuming the residuals flows are evenly distributed between the two lagoons, the theoretical residence time of the treatment lagoons is 24.4 days with no solids residual accumulated in the lagoons, and 11.6 days with 1m of solids residual accumulated in the lagoons. The first tracer study, conducted with approximately 9 months accumulation of solids residual in the lagoons, had a minimum residence time of 200 minutes and the maximum tracer concentration was reached at a residence time of 234 minutes. The second tracer study, conducted with approximately 3 months of solids residual accumulated in the lagoons, had a minimum residence time of 85 minutes. The maximum tracer concentration, however, was not reached until a residence time of 630 minutes. Based on an extrapolation of the tracer curve, it is projected that after 5.7 days, all tracer would have exited the lagoon.

It is evident from both tracer tests that there is significant short circuiting occurring in the lagoons. The maximum tracer concentration being reached at an increased retention time during the second test is likely a result of the lower level of solids residual accumulation in the lagoon during this test.

The hydraulic efficiency of a settling pond is expressed as the mean residence time divided by the theoretical residence time and a hydraulic efficiency of less than 0.3 or 0.4 represents short circuiting (Thackson et al., 1987). During the second tracer test, the mean residence time was 47.5 hours and the theoretical retention time based on an estimated 0.175 to 0.225 m of solids residual accumulation in the lagoons in 3 months (see Section 6.3.1 for discussion on rate of solids residual accumulation) was calculated to be between 516 and 532.8 hours. This results in a hydraulic efficiency of 0.09 and indicates that short circuiting is occurring in the lagoons.

6.1.3 Lagoon Effluent Sampling

The aluminum concentration in lagoon effluent samples conducted by plant operations staff in 2014 ranged from 13.05 to 354 µg/L with an average of 120 µg/L. There did not appear to be a seasonal trend in the concentrations. The average of 120 µg/L is above the CCME FWAL criteria of 100 µg/L for water with a pH of greater than or equal to 6.5; however, a study conducted by CBCL (2008), found that the background aluminum concentration in the brook upstream of the lagoon discharge ranged from 120 µg/L to 270 µg/L with an average of 164 µg/L. Of the 44 lagoon effluent samples collected by plant operations staff in 2014, there were 20 samples that exceeded the CCME FWAL criteria of 100 µg/L, 14 samples that exceeded the minimum background concentration measured by CBCL Ltd. of 120 µg/L, and 12 samples that exceeded the average background concentration measured by CBCL Ltd. of 164 µg/L.

Samples collected on July 11 when there was approximately 11 months accumulation of solids residual had a total aluminum concentration that ranged from 168.2 to 298 µg/L. Samples collected on November 10, 2014 when there was approximately 2 months accumulation of solids residual had a total aluminum concentration that ranged from 98.01 to 128.3 µg/L. During both sample events, the aluminum was primarily in particulate form. This information provided direction for additional experiments in that bench scale experiments were focused on improving settling of particulate aluminum rather than adjusting the pH of the water to reduce the solubility of aluminum.

6.2 OPTIMIZATION OF LIQUID STREAM RESIDUALS TREATMENT

6.2.1 Polymer Settling Test

ASCE and AWWA (1996) state that anionic high molecular weight polymers are most successful for use in thickening and dewatering of water treatment residuals. According to Crittenden et al. (2012) most types are polymers are successful in conditioning solids residuals; however, higher molecular weight polymers are generally more effective. Research has found that non-ionic polymer (Cornwall and Lee, 1994), cationic polymer (Arora et al., 2001; Wood, 2014) and anionic polymers (Arora et al., 2001) were successful in improving particle removal via settling. Settling tests were conducted with anionic, non-ionic and cationic high molecular weight polymers in doses ranging from 3 mg/L to 25 mg/L. Optimum polymer doses found in research varies considerably, from 0.5 mg/L (Arora et al., 2001) to 50 mg/L (Ma et al., 2007) and is dependent on both the properties of the solids residual and the properties of the polymer. Therefore, polymer selection is usually conducted using empirical testing.

After 3 hours of settling time with no mixing during settling, a 12 mg/L dose of cationic polymer Superfloc C-1592RS, a 3 mg/L dose of cationic polymer Novus 2667, and a 3 mg/L dose of anionic polymer Polyfloc AP1103 were all successful in reaching an aluminum concentration of less than the CCME FWAL guideline of 100 µg/L with aluminum concentrations of 91.25, 83.78, and 88.28 µg/L, respectively. None of the jars without polymer added were successful in reducing the aluminum concentration to less than 100 µg/L. After 3 hours of settling time with 5RPM mixing during settling, a 12 mg/L dose of cationic polymer Novus 2667 was successful in reaching an aluminum concentration of less than the CCME FWAL guideline of 100 µg/L with an aluminum concentration of 97.29 µg/L. None of the jars without polymer were successful in reducing the aluminum concentration below 100 µg/L.

The addition of polymer to CFBW at the WTP could be used to improve aluminum removal in the lagoons. All polymers tested were successful in reducing the aluminum concentration compared to the jars with no polymer; however, only the cationic and anionic polymers tested were successful in reducing the aluminum concentration below 100 µg/L. Cationic polymer Novus 2667 performed the best during both the 0 RPM settling test and the 5 RPM settling test. The optimal dose should be refined through settling tests with a flow proportional composite sample as the optimal dose is likely overestimated due to the grab samples coinciding with the high flow backwash period resulting in a high level of aluminum and TSS in the CFBW.

One important consideration is that the addition of polymer would increase the mass of solids residual to be managed. A 3 mg/L dose of polymer could increase the mass of solids residual by 592 kg per year on a dry basis. This would represent an increase in solids residual production of between 0.98 and 1.25%.

6.2.1.1 Comparison of Polymer Settling With and Without Mixing

Mixing can provide either negative or positive impacts to particle removal during settling. Turbulence caused by mixing can impede settling by causing particles that have already settled to become re-entrained and by preventing particles from settling (Janssen, 2008). However, improved flocculation can result from the increase in particle collisions caused by turbulence.

Settling tests were conducted with no polymer and doses of high molecular weight cationic, anionic and non-ionic polymer in doses ranging from 3 to 25 mg/L with settling at 0 RPM. The settling tests were repeated with settling at 5 RPM. For the jars without polymer added, after 1 hour of settling, 2 of the 4 tests had lower aluminum concentrations with 0 RPM settling and 2 had lower aluminum concentrations with 5 RPM settling. After 2 hours of settling, 1 of the 4 tests had a lower aluminum concentration with 0 RPM settling had 3 had lower aluminum concentrations with 5 RPM settling. After 3 hours of settling, 2 of the 4 tests had lower aluminum concentrations with 0 RPM settling and 2 of the tests had lower aluminum concentrations with 5 RPM settling. Similarly, at varying doses of cationic, anionic and non-ionic high molecular weight polymer, there does not seem to be a consistent trend in improved settling at either 0 or 5 RPM settling. A one-way ANOVA found that the settling efficiency between settling at 0 RPM and 5 RPM with and without polymer added was not statistically significant ($p = 0.07$ and $p = 0.58$, respectively).

6.2.2 Cation Ratio Optimization Settling Test

Settling tests were conducted with M:D ratios ranging from 0.1 to 3. The unamended CFBW had an M:D ratio of 0.24, 0.22 and 0.22 in the three samples analyzed for monovalent and divalent cations. None of the jar tests conducted with cation ratio adjustment were successful in reaching an aluminum concentration of less than the CCME FWAL guideline of 100 µg/L after 3 or 24 hours of settling.

After 2, 3, and 24 hours of settling, there was a general trend of increased aluminum removal with increasing M:D ratio. Trias et al. (2004) found that improvements in settleability of liquid hog manure through cation ratio adjustment were inconsistent and not statistically significant. Higgins and Novak (1997) found the optimum ratio for M:D and Ca:Mg to be 1.0. Bourgeois et al. (2004a) determined that lower M:D ratios increased aluminum removal through settling and found the optimum ratio for M:D to be 0.33.

Novak et al. (1998) studied an activated sludge treatment plant with regard to the impact that the cation content of the wastewater had on settling and dewatering. It was found that a ratio of monovalent to divalent cations of greater than 2:1 on a milliequivalent basis resulted in poor dewatering properties and increased polymer requirements. The study also showed that when the concentration of sodium and potassium were less than 10 and 0.1 meq/L, respectively, there was a substantial improvement in settling and dewatering; however, under these conditions the addition of magnesium did not further improve settling or dewatering. The average concentration of sodium and potassium in the CFBW at the WTP were measured to be 0.39 and 0.06 meq/L, respectively. The concentration of sodium and potassium for all jar tests were less than 10 and 1 meq/L, respectively.

Cation ratio adjustment was not found to be a feasible solution for improving aluminum removal from the CFBW at the WTP as cation ratio adjustment did not result in aluminum concentrations below the CCME FWAL guideline of 100 µg/L, and the concentration of sodium and potassium in the CFBW is below the level under which Novak et al. (1998) identified there was no settling improvement with an increase in divalent cations.

6.3 CHARACTERIZATION OF SOLIDS RESIDUAL

6.3.1 Solids Content of Solids Residual

The lagoons generally produce a solids residual with a solids content of 1% on a mass basis. There was a thin layer of solids residual with a solids content of 2.7% on a mass basis at the bottom of the lagoons. The increased solids content of the bottom layer could be due to compression, or solids residual that had been present in the lagoon prior to the previous winter with freezing causing the increase in solids content.

The solids content of the solids residual that is produced depends on the turbidity of the raw water, the coagulant dose and the mechanism of coagulation (ASCE and AWWA, 1996). Higher levels of suspended solids in the raw water produce solids residual with a higher solids content. When the coagulation mechanism is adsorption-charge neutralization, the process uses a lower alum dose and results in a higher solids content solids residual due to a higher percentage of suspended solids and a lower percentage of aluminum hydroxide (ASCE and AWWA, 1996). When the coagulation mechanism is sweep coagulation, the process uses a higher alum dose and results in a solids residual that is more difficult to thicken and dewater due to a higher percentage of aluminum hydroxide in the solids residual (ASCE and AWWA, 1996). Cornwall et al. (1987) state that gravity thickening of coagulant solids residual results in a solids content between 3 and 4%. According to ASCE and AWWA (1996) lagoons typically produce a solids residual with a solids content of 6 to 10% for metal hydroxide solids. ASCE and AWWA (1996) provides the settled solids content of alum solids residual from four treatment plants in Missouri which are summarized in Table 36.

Table 36 Solids content of settled alum solids residual (ASCE and AWWA, 1996; Calkins and Novak, 1973)

Location	Settled Solids (%)
Higginsville	3.1
Macon	3.4
Moberly	6.3
Kirksville	7.8

Knocke et al. (1987) found that a reduction in coagulation pH or coagulant dose or an increase in raw water turbidity increased the dewaterability of the solids residual. The solids content of the solids residual at various experimental conditions for coagulant dose, pH and coagulation mechanism are provided in Table 37.

Table 37 Coagulation Mechanism and Solids Content of Solids Residual (ASCE and AWWA, 1996; Knocke et al., 1987)

Raw Turbidity	Coagulant Dose	Coagulation pH	Mechanism	Thickened Solids Content	Dewatered Solids Content
NTU	mg/L		-	%	%
40	10	6.2	Adsorption-charge neutralization	6.0	42
40	15	6.3	Adsorption-charge neutralization	5.5	22
7	40	6.5	Mixed	1.0	15
7	75	7.1	Enmeshment	1.0	11
7	75	8.1	Enmeshment	0.5	9

According to the following Figure 40 from Knocke et al. (1987) the coagulation dose of 57 mg/L and coagulation pH of 6.9 used at the WTP is associated with sweep or enmeshment coagulation.

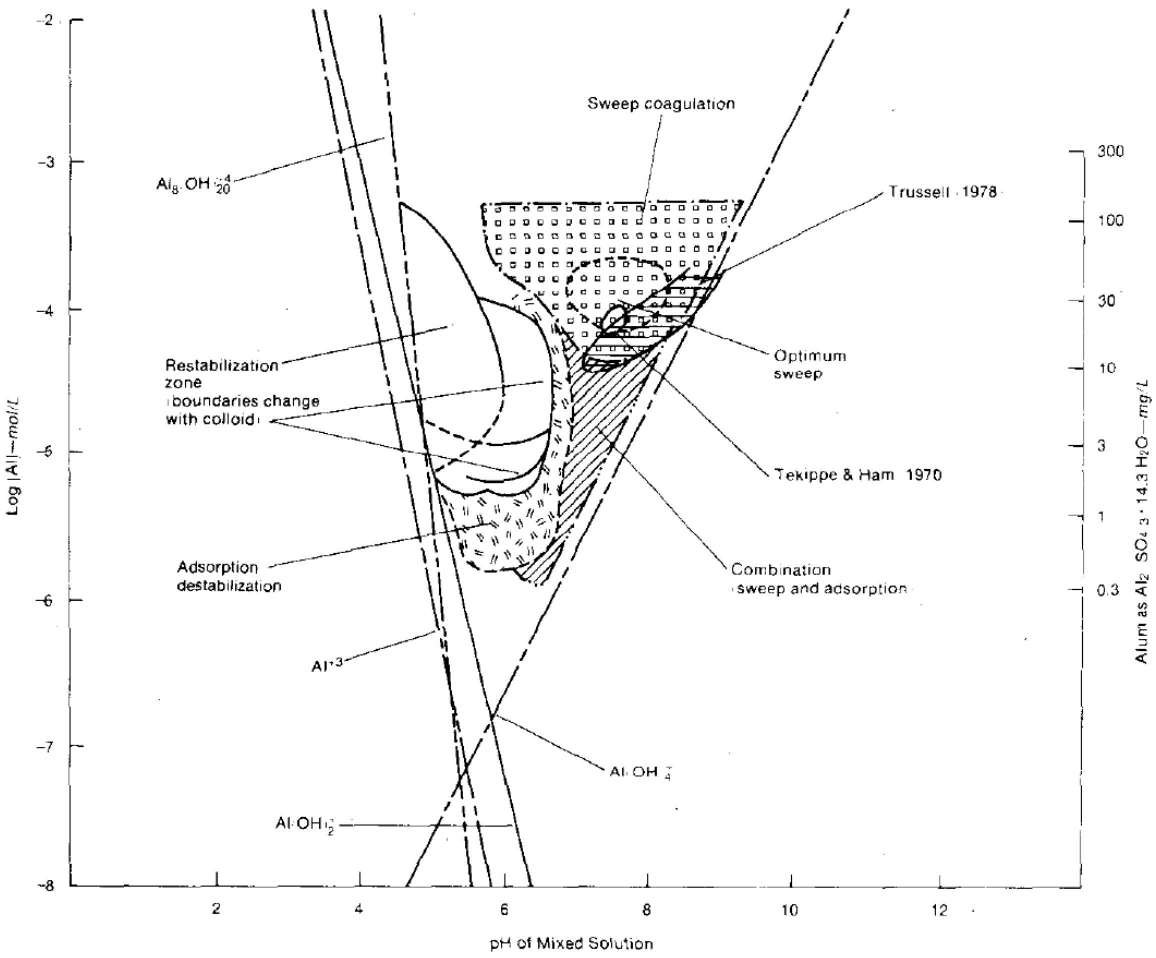

Figure 40 Coagulation Diagram for Alum Coagulation (Source: Knocke et al., 1987)

The solids residual in the drying bed was observed to have two distinct layers: the top layer had an average solid content of 36.9% by mass and the bottom layer had an average solids content of 24.2% by mass. ASCE and AWWA (1996) provide the following solids content requirements for various disposal options of water treatment plant residuals, summarized in Table 38.

Table 38 Required Solids Residual Solids Content for Various Disposal Options

Disposal Option	Solids Content Range (%)
Land Application	<1 – 15
Landfill	<15 – 25
Monofill	>25
Discharge to Sewer	<1 – 8
Direct Stream Discharge	<1 – 8
Residual Reuse	<1 – >25

The drying bed removes enough water to produces a solids residual with a high enough solids content for disposal. However, the low solids content of the solids residual produced by the lagoons results in them filling up with solids residual twice as fast as would occur with a solids content of 2% as assumed during the design phase. In addition, any water content in the solids residual leaving the drying bed represents additional mass that must be managed and disposed of.

The O&M Manual for the WTP (Dillon, 2003) recommended that solids residual be removed from the lagoons when it reached a thickness of 2 feet. Based on a solids content of 1%, an area of 3458 m^2 per lagoon, a solids residual production rate of 130 and 166 kg/day on a dry basis, and an assumed specific gravity of solids of 2.2, the rate of solids residual accumulation in the lagoons is estimated to be between 0.7 and 0.9 m/year. This means it would take between 36 and 47 weeks for 2 feet of solids residual to accumulate in the lagoons. The current practice is to remove solids residual from the lagoons annually.

6.3.2 Metals Content of Solids Residual

The metals content of three different “batches” of solids residual from the drying bed was analyzed. The highest metals concentrations in the solids residual are aluminum, iron, and manganese. These metals comprise an average of 17.7, 2.7, and 0.7% of the solids residual, respectively, by weight on a dry basis. The high aluminum concentration is primarily a result of the aluminum sulfate that is used as a coagulant in the water treatment process and to a much lesser extent due to aluminum in the source water. An analysis by Agyin-Birikorang and O’Connor (2009) of alum water treatment plant solids residual found fresh solids residual to contain 15.2% +/- 2.01 aluminum. The average aluminum concentration of 17.7% in the WTP solids residual is just above their upper limit of 17.21%. A study by Wang et al. (2014) found that solids residual from the Beijing City No. 9 Waterworks which used Al and Fe salts as coagulants contained 10.7, 11.0, and 0.862% aluminum, iron and manganese, respectively, by weight on a dry basis. The manganese level in the solids residual is due to the presence of manganese in the raw water and its removal by either the clarifiers or filters during the treatment process. The remainder of the solids residual is likely composed of clay, silt, algae, organic matter, and trace metals.

When comparing the metals content of the solids residual to CCME soil quality guidelines (CCME, 1999c), it exceeds residential, commercial, and industrial guidelines for arsenic in 2 of 3 samples; it exceeds residential soil quality guidelines for chromium in 3 of 3 samples and commercial and industrial soil quality guidelines for chromium in 2 of 3 samples; and it exceeds residential soil quality guidelines for selenium in 3 of 3 samples and commercial and industrial soil quality guidelines for selenium in 1 of 3 samples. The exceedances are all close to the guidelines with the highest concentration of arsenic, chromium, and selenium being 25%, 49% and 10%, respectively, above the commercial/industrial soil quality guideline. The average concentrations in the WTP solids residual for arsenic, chromium and selenium were 12.5 mg/kg, 103 mg/kg and 2.2 mg/kg, respectively. A study by Wang et al. (2014) found that solids residual from the Beijing City No. 9 Waterworks contained 106 mg/kg of arsenic and 726 mg/kg of chromium; selenium was not detected.

6.3.3 Leachability Testing of Solids Residual

Whether the water treatment plant solids residual is going to be monofilled, landfilled, or land applied, the leachability of the solids residual is one factor that helps to demonstrate how it will impact the environment it is placed in. The synthetic precipitation leaching procedure (SPLP) test is the most relevant leachate test to estimate the leaching behavior of the solids residual in a monofill or land application scenario. This test approximates leaching due to slightly acidic precipitation.

There are no regulatory criteria for SPLP results in Nova Scotia or from CCME. However, the results have been compared to CCME FWAL and CCME marine water criteria for reference.

Three samples of solids residual from the drying bed collected on November 20, 2014 were analyzed for SPLP. As leachate analysis is performed on the samples on an as-received basis, two of the samples were dried in an oven at 104°C for 48 hours prior to analysis and one sample was analyzed as collected. It was expected that the samples that were dried prior to analysis would have higher concentrations of metals in the leachate due to a higher mass of solids in the sample. The SPLP results of the November 20, 2014 samples that were dried prior to analysis exceeded CCME FWAL criteria for aluminum and arsenic only. There were no exceedances for the sample that was analyzed without being dried.

In addition to the leaching characteristics, the amount of contaminants that will leach from residuals also depends on the hydraulic conductivity of the residuals. The hydraulic conductivity of alum solids residual is very low with values ranging from 10^{-4} to 10^{-6} m/day with the effective stress increasing from 2 to 540 kPA (O'Kelly, 2010).

Based on the results of the leaching analysis, it is suggested that SPLP analysis should be performed on each batch of solids residual prior to land application due to the potential leaching of aluminum, and arsenic and the potential for variability in leaching characteristics between different batches of solids residual. The test should be conducted on the solids residual as collected, at the solids content which it will be applied as the SPLP results are dependent on the solids content of the solids residual.

6.3.4 Solids Residual Amendment

When solids residual from the drying bed was amended with 0, 2, 5 and 10% blast furnace slag by weight, the only metal that exhibited a trend of a decrease in concentration in the leachate with increasing slag content was manganese. The manganese concentration in the leachate decreased from 22,000 µg/L with 0% slag to 8,600 µg/L with 10% slag. This represents a reduction in manganese concentration of 60%. The pH concentration of the sample increased from 6.22 with 0% slag to 7.45 with

10% slag. The decrease in manganese concentration in the leachate is likely due to a decrease in solubility of manganese with an increase in pH. Manganese is highly soluble in acidic and neutral waters (Silva et al., 2012; Bamforth et al., 2006) and the solubility decreases with increasing pH. The solubility of manganese cannot be explained by pH alone, however. Manganese solubility and the precipitates that are formed are also dependent on whether it is in an oxidizing or reducing environment (Bamforth et al., 2006). Acidic and reducing conditions can reduce insoluble Mn^{4+} oxides to soluble Mn^{2+} (Heal, 2001). These results are consistent with Cetin and Aydilek (2013) which found that the blending of soil with fly ash resulted in an increase in pH which increased the concentration of boron, molybdenum and selenium and decreased the concentration of manganese in the leachate.

Calcium and strontium exhibited a trend of an increase in concentration in the leachate with increasing slag concentration. There was an increase in concentration in the leachate with 10% slag of 335% and 241% for calcium and strontium, respectively, compared to 0% slag. This may be due to the calcium and strontium leaching from the slag itself rather than the slag causing an increase in leaching of calcium and strontium from the solids residual. Calcium oxide makes up the largest portion of the composition of slag (National Slag Association, 2003). Sample results contained in a report by CBCL (2005), identified the presence of strontium in SYSCO slag at a concentration of 421 mg/kg and in leachate produced from the slag at a concentration of 1950 µg/L.

There was no apparent trend in aluminum concentration in the leachate with increasing slag composition in the solids residual. This is likely due to the fact that across the pH range of solids residual in the test, pH of 6.22 with 0% slag to pH of 7.45 with 10% slag, aluminum is relatively insoluble and the solubility remains fairly constant. Chromium and selenium were not detected in any of the leachate samples during this experiment.

A study by Wang et al. (2014) found that pH is an important factor in the release of metals from water treatment residuals and most metals were more easily released under low pH conditions. It was found that release of the following metals was increased under low pH conditions: Ba, Be, Ca, Cd, Co, Cr, Fe, Mg, Mn, Pb, Sr and Zn. The release of the following metals was increased under high pH conditions: As, Mo and V. The release of the following metals was increased under both high and low pH conditions: Al, Cu and Ni. The study recommended a pH range between 6 and 9 for low release of metals.

6.4 Investigation to Determine Source of Chromium in Solids Residual

Chromium has been found in the drying bed solids residual in concentrations ranging from 69 to 130 mg/kg. Chromium was found in the alum in concentrations ranging from 757.9 to 866.3 µg/L with an average of 791.4 µg/L. Chromium was detected in the raw water in concentrations ranging from <0.4 µg/L to 0.636 µg/L. A mass balance was conducted in order to determine whether these chromium levels in the alum and raw water could account for the chromium concentration in the solids residual. Of the estimated 8.9 to 21.6 g/day load of chromium in the solids residual, the aluminum sulfate provides between 0.31 and 0.34 g/day of chromium, and the highest measured concentration of chromium in the raw water would provide between 5.9 and 6.3 g/day of chromium. This calculation leaves between 2.3 and 15.4 g/day of chromium unaccounted for. The deficit could be due to higher chromium concentrations in previous batches of alum, although this is unlikely as with the small flowrate of alum per day, the concentration would have to be almost an order of magnitude higher than measured. However, alum used at a water treatment plant in Durham, N.C. contained 9.5 mg/L of chromium (Eyring et al., 2002) which is 12 times the average concentration in the alum used at the WTP. Another possible explanation could be seasonal variation in chromium concentration in the raw water. The mass balance determined that the average annual chromium concentration in the raw water would have to be between 0.86 and 2.3 µg/L to account for the chromium in the solids residual not contributed by the alum. This level is 1.4 to 3.6 times higher than the highest chromium concentration measured in the raw water; however, there were only 3 raw water samples analyzed for chromium after digestion with nitric acid and they were all collected in November and December 2014. Another explanation could be an alternative source of chromium that has not been identified. The chromium concentration in any treatment chemicals would have to be substantially higher than the concentration in the raw water would have to be due to the low flow rates of treatment chemicals compared to the raw water flow. The chromium deficit in the calculation could also be due to inaccuracy in inputs and assumptions used in the mass balance calculation. McNeill et al. (2012) noted that other sources of chromium in drinking water could be the result of leaching from or reaction with distribution system materials such as cast iron, cement, or stainless steel. Table 39 lists the plant materials upstream of the lagoons. Concrete, ductile iron, and stainless steel can be sources of chromium.

Table 39 Plant Materials Upstream of Lagoons

Component	**Material**
Raw water intake	Concrete
Raw water intake line	Wood stave
Intake Screen	Aluminum
Process Piping	Primarily ductile iron and PVC with some stainless
Pre-treatment tanks	Concrete
Floc tank	Concrete
Clarifiers	Concrete
Plate settlers	Fibre-reinforced plastic
Filter tanks	Concrete
Filter media	anthracite, sand, support gravel, filter block
Chlorine contact chamber	Reinforced concrete
Clearwell	Reinforced concrete
Backwash troughs	PVC
Piping to Lagoons	PVC
Lagoon inlet piping	Ductile iron

CHAPTER 7 CONCLUSIONS AND RECOMMENDATIONS

7.1 EVALUATE LIQUID STREAM RESIDUALS TREATMENT

The volume of combined filter backwash water (CFBW) produced, which comprises between 5.4 and 5.8% of the water treated at the plant, makes up a slightly larger proportion of treated water than values found in literature. There is a large variation in TSS and aluminum concentration in the CFBW over the duration of the filter backwash cycle. This is likely due to the majority of particles being removed from the filter near the beginning of the filter backwash cycle. The clarifier sludge chemistry is similar to the samples taken at the beginning of the filter backwash cycle. Although the clarifier sludge is continuously discharging, samples collected later during the filter backwash cycle would be largely comprised of the cleaner FBW that would be expected toward the end of the filter backwash cycle. The TSS and aluminum concentrations measured in the CFBW are higher than the values found in the literature. This could be due to the timing of the grab samples, as flow proportional composite samples were not collected, or could be due to the high dose of alum that is associated with sweep coagulation.

The results of the tracer study indicated that the mean hydraulic retention time with 3 months of solids residual accumulation is 47.5 hours which means that short circuiting is occurring in the lagoons. The minimum retention time in the lagoons for the tracer test with 3 months of sludge accumulation was 85 minutes.

The aluminum concentrations in the lagoon effluent as measured by plant operations staff during 2014 had an average of 120 µg/L. This is above the CCME FWAL guideline of 100 µg/L for waters with a pH of greater than or equal to 6.5; however, the location of the lagoon discharge is within a section of the brook that has tidal influence from the Atlantic Ocean and may be considered a brackish environment. In addition, a 2008 study conducted by CBCL Ltd. found that the background concentration of aluminum measured in the brook upstream of the lagoon effluent discharge ranged from 120 to 270 µg/L with an average of 164 µg/L. The aluminum concentration in samples collected on December 31, 2014 from 8 streams that flow into the reservoir, which is the source water for the plant and is situated upstream of the lagoon effluent discharge, ranged from 115.5 to 273.1 µg/L with an average of 223.5 µg/L. The range of aluminum concentrations in the discharge on a day when there was 2 months of solids residual accumulation in the lagoon (98.01 to 128.3 μg/L) was found to be less than the range of aluminum concentrations when there was 11 months of solids residual accumulation (168.2 to 298 μg/L). This could be due to either the reduced hydraulic retention time associated with greater solids residual accumulation, re-entrainment of settled particles due to the higher solids residual-water interface, or a combination of both. However, there was no seasonal trend in aluminum concentrations noted in the 2014 lagoon effluent samples collected by plant operations staff. The aluminum in the lagoon effluent was found to be primarily in particulate form which means the aluminum levels are due to particles not being effectively removed via settling rather than as a result of increased solubility due to the pH of the water. Although the concentrations of aluminum in the lagoon effluent are generally greater than the CCME FWAL criterion, this criterion may not be applicable, and they are generally within the range of background concentrations of aluminum in the brook.

7.2 OPTIMIZATION OF LIQUID STREAM RESIDUALS TREATMENT

It was found that high molecular weight polymer could be used to improve aluminum removal from CFBW via settling to levels below the CCME FWAL guideline which is 100 μg/L for waters with a pH of 6.5 or greater. None of the jar tests conducted without polymer were successful in reducing the aluminum concentration below this level. High molecular weight cationic polymer NOVUS 2667 exhibited the best results in the experiments conducted. The optimum dose should be confirmed through settling tests with a flow-proportional composite CFBW sample as the TSS and aluminum concentration was shown to decrease as the FBW cycle progressed. Additional considerations with the use of polymer for CFBW treatment include operational and capital costs, increased solids residual production, mixing requirements, and residual polymer in the water that is discharged to the environment. Results of a series of duplicate settling tests conducted with settling at 0 RPM compared to settling at 5 RPM indicated that there was no consistent trend in improved settling with or without mixing during settling. The difference in settling efficiency between settling at 0RPM and 5RPM, with and without the addition of polymer, was not found to be statistically significant ($p=0.07$ and $p = 0.58$, respectively).

Cation ratio adjustment was determined to not be an appropriate method to improve aluminum removal from CFBW. The concentration of aluminum was not reduced below 100 µg /L for any of the cation ratio adjustments tested, and the concentrations of sodium and potassium in the CFBW are below the levels that Novak et al. (1998) identified as being associated with no improvement in settling and dewatering with the addition of the divalent cation magnesium. However, it is important to note that the study by Novak et al. (1998) involved activated sludge and not alum residuals.

7.3 CHARACTERIZATION OF SOLIDS RESIDUAL (SLUDGE)

The solids residual production rate was calculated to be between 130 and 166 kg/day on a dry basis. The lagoons produce solids residual with a solids content of 1%. The low solids content is likely due to the source water characteristics as well as the mechanism of sweep coagulation. The accumulation rate of solids residual in the lagoons was estimated to be between 1.4 and 1.7 m/year. The drying beds were effective at increasing the solids content of the solids residual to between 24% (lower layer) and 37% (upper layer). With an average solids content of 30%, this results in a solids residual production rate ranging from 158 to 202 tonnes per year.

Metals concentrations in the solids residual exceeded CCME soil quality guidelines for commercial and industrial land for arsenic, chromium, and selenium; however, none of the parameters exceeded CSQGs in all three samples. In addition, the arsenic concentrations were all less than the natural background concentration for arsenic in soil determined by JDAC (2001). The exceedances ranged from 10 to 50% above the CSQG and blending the solids residual with soil would likely result in the metals concentrations being below the CSQGs.

SPLP testing was conducted on the solids residual and a metals analysis was performed to determine the potential for metals to leach from the solids residual if it were land applied. The results were compared to CCME FWAL criteria as a point of reference and exceeded criteria for aluminum and arsenic. However, it is important to note that the SPLP testing was conducted on oven dried solids residuals, and solids residuals submitted at the solids content at which they are produced by the drying bed would result in lower concentrations for the test result. In addition, the quantity of metals that would leach from the solids residual if it were land applied would also depend on the amount of water that passes through the solids residual which is dependent on precipitation, topography, evapotranspiration, and the permeability of the solids residual.

The addition of air-cooled blast furnace slag was found to reduce the level of manganese that was leached from the residuals but did not have an effect on the aluminum level. There was an increase in calcium and strontium in the leachate with increasing slag concentration, likely due to the components leaching from the slag itself.

7.4 INVESTIGATION TO DETERMINE SOURCE OF CHROMIUM IN SOLIDS RESIDUAL

The chromium load in the solids residual was not explained by the chromium load in the alum or the raw water. The chromium load in the alum was found to only account for between 1 and 4% of the chromium load in the solids residual. The average annual chromium concentration in the raw water would have to be between 0.86 and 2.31 µg/L to explain the remainder of the chromium load in the solids residual. The highest measured chromium concentration in the raw water was 0.636 µg/L; however, there may be seasonal variation. To put the chromium concentrations in the raw water in context, The Health Canada guidelines for Canadian Drinking Water Quality for chromium in treated water is 50 µg/L (Health Canada, 2014) which is 79 times higher than the maximum measured chromium concentration in the raw water. Although the raw water does not have to be treated for chromium to meet the guideline, it is being removed in the treatment process and accumulating in the solids residual resulting in chromium concentrations in the solids residual being above CCME soil quality guidelines.

7.5 RECOMMENDATIONS

Future work at the WTP could involve a risk assessment of the discharge to the brook. This could include preparation of a dilution model and determination of an appropriate mixing zone in the receiving water and would also consider the background concentration of aluminum in the receiving water. It could also include determining whether the section of the brook where the discharge is located is considered a freshwater, brackish, or marine environment.

Between 16 and 19% of the solids residual produced by the plant on a mass basis (dry) is elemental aluminum. Aluminum is likely present in the residuals in a form with a higher molar mass than elemental aluminum which therefore would account for a larger percentage of the total residuals. For instance, if it was in the form $Al(OH)_{3(s)}$, it would account for between 46 and 55% of the solids residual produced by the plant on a mass basis (dry). As the majority of the aluminum load comes from the aluminum sulfate, there may be an opportunity to decrease the production rate of residuals by reducing the coagulant dose. Jar test experiments could be used to determine whether the coagulant dose could be reduced without impacting treated water quality. Reducing coagulant dose would also reduce the plant's chemical costs.

Additional characterization could be conducted of the source water reservoir to determine the relative contributions of the streams that flow into the reservoir and whether the aluminum, iron, manganese and chromium concentrations in the streams vary seasonally.

If polymer dosing was going to be implemented, additional jar tests should be conducted with a flow proportional composite combined filter backwash water sample in order to determine the optimum dose. In addition, the level of residual polymer in the discharge and whether it is deleterious to the receiving water should be considered.

Additional research could include determining whether polymer dosing impacts the leaching characteristics of the solids residual. It would also be beneficial to determine a way to increase the solids content of the solids residual that is produced in the lagoons.

An experiment could be conducted to determine whether stainless steel, ductile iron, and concrete materials in the plant are contributing to the chromium levels detected in the solids residual.

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APPENDIX A ADDITIONAL FIGURES

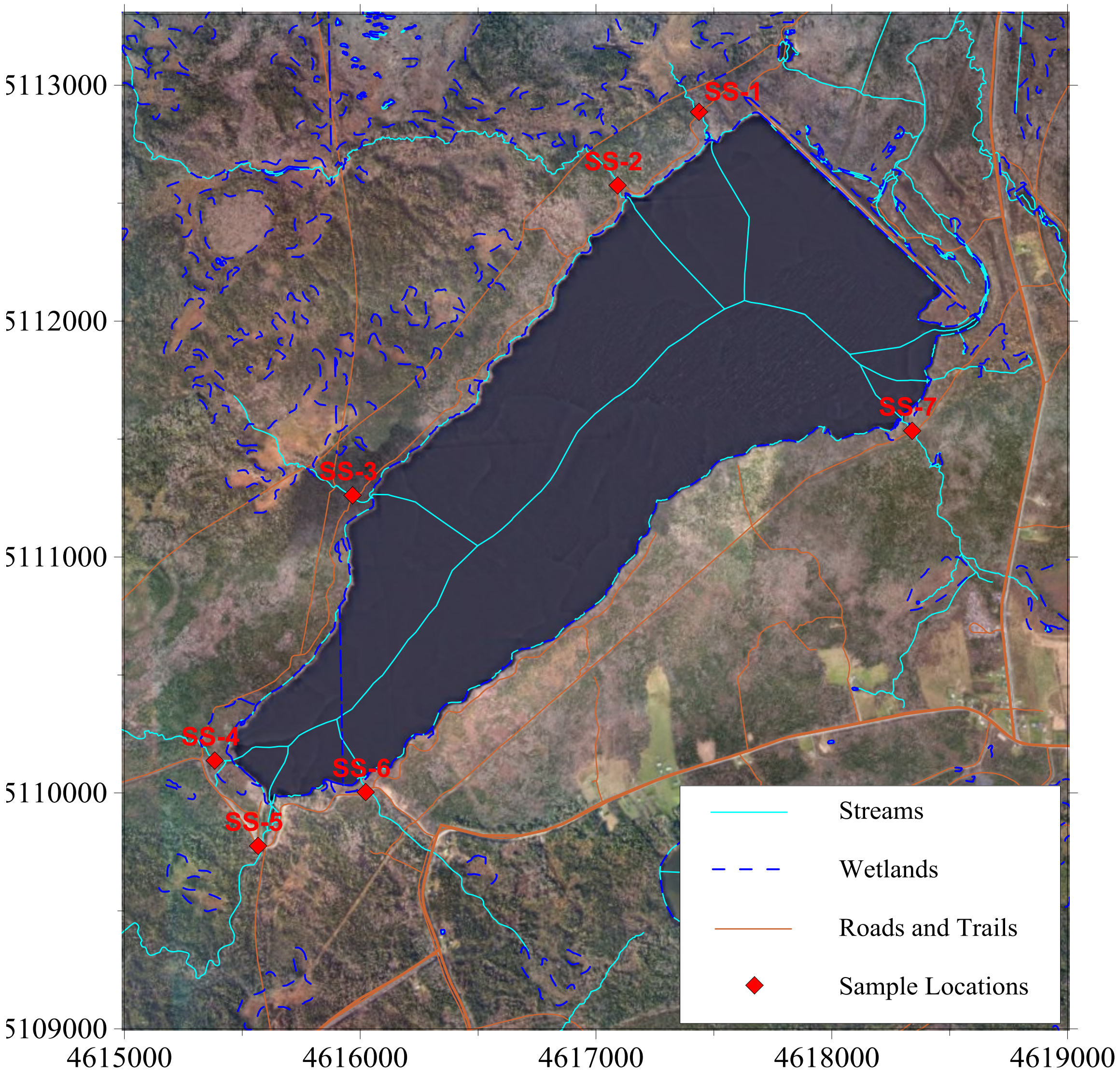

Figure A1 Surface Water Sample Locations

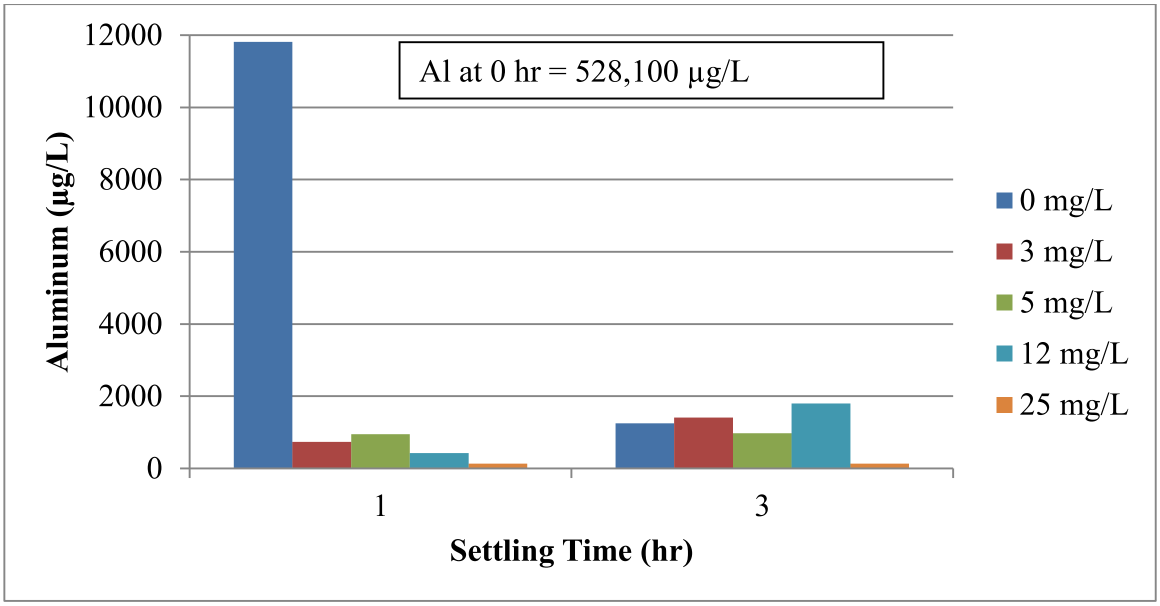

Figure A2 Settling Test with Superfloc C-1592RS at 5 RPM

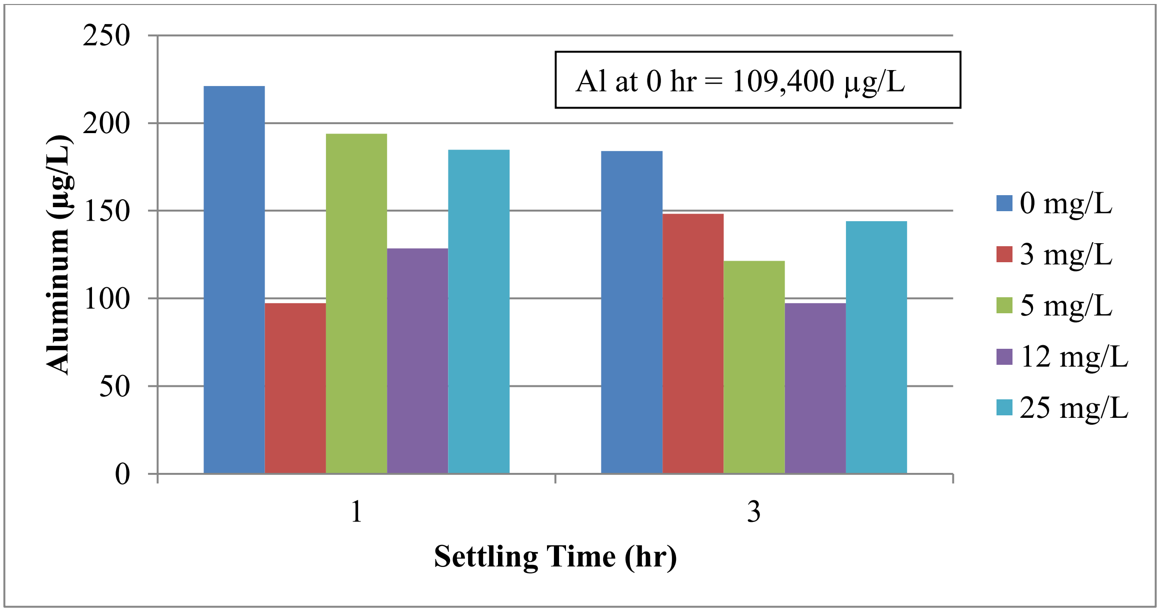

Figure A3 Settling Test with Novus 2667 at 5 RPM

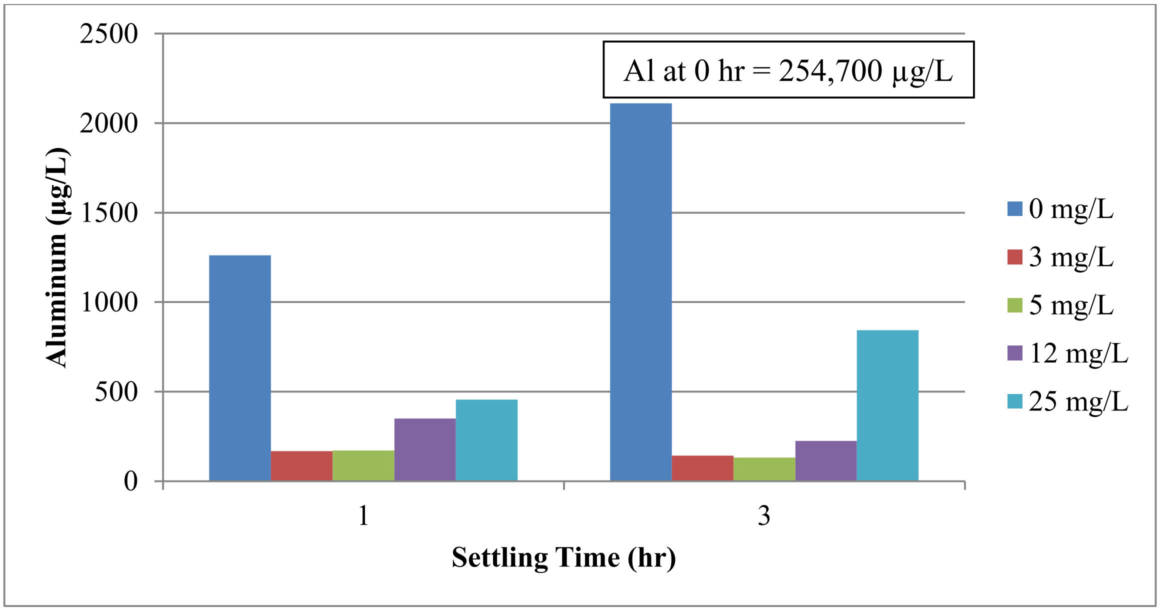

Figure A4 Settling Test with Polyfloc AP1103 at 5 RPM

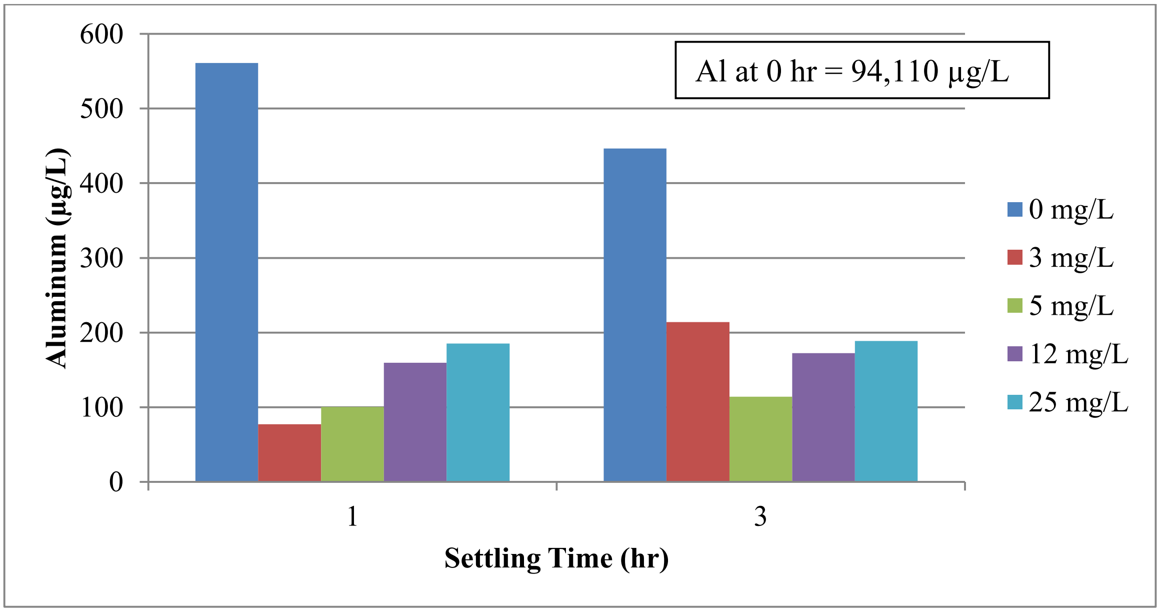

Figure A5 Settling Test with Magnafloc LT20 at 5 RPM

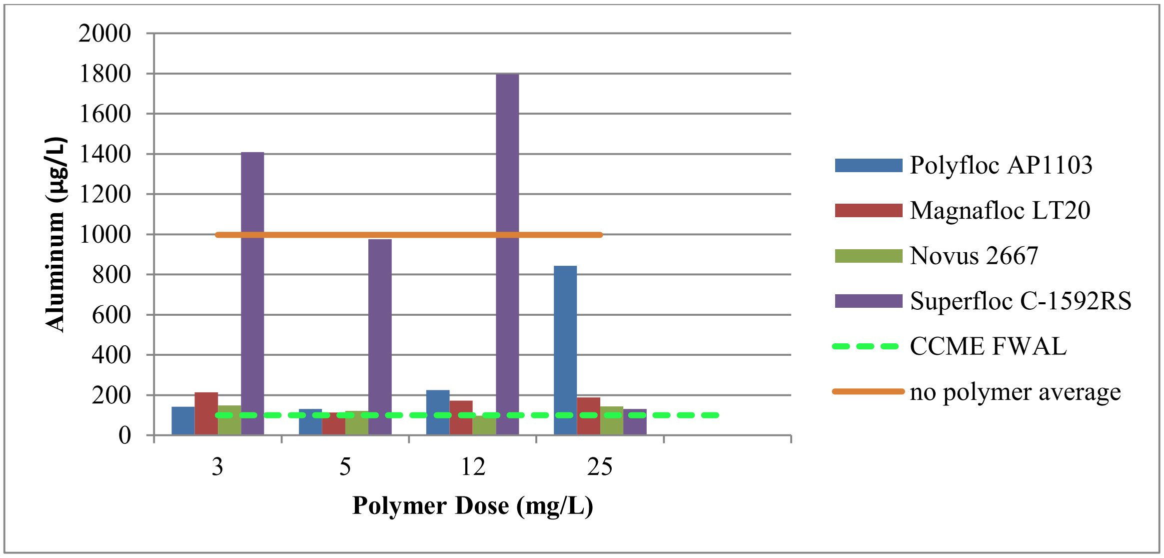

Figure A6 Aluminum Removal after 3 Hours of Settling at 5 RPM

APPENDIX B ADDITIONAL DATA

Table B1 Cation Ratio Adjustment Data

Date	Aluminum Concentration (µg/L)								M:D
Time (hr):	0	0.5	1	2	3	24	3f	24f	
02-May-15	376150	5693	3693	5363	3379	1246	47.56	76.62	0.09
02-May-15	376150	5516	6045	5111	4020	1053	66.49	66.24	0.22
02-May-15	376150	4941	2718	5370	2782	1012	53.9	96.09	0.48
02-May-15	376150	4621	2429	5362	2623	891.2	37.28	98.26	1.01
02-May-15	376150	2560	2513	4355	1842	756	54.27	108.5	3.11
02-May-15	376150	6902	7680	6735	3294	1107	49.34	177.5	0.08
02-May-15	376150	6774	4307	3511	2725	1192	39.97	181.4	0.22
02-May-15	376150	6084	6003	3965	2201	868.6	68.97	113.8	0.48
02-May-15	376150	5352	3627	3630	2372	763.5	48.68	118.2	1.10
02-May-15	376150	4813	2993	2187	1393	705	49.56	116.3	2.94

Table B2 Metals Content of Solids Residual

Parameter	CCME Soil Quality Guidelines		Sample Results		
	Residential/ parkland	Commercial	30-May-13	30-Jul-14	20-Nov-14
Aluminum (Al)	NG	NG	160000	180000	190000
Antimony (Sb)	20	40	<1	<2.0	<2.0
Arsenic (As)	12	12	7.4	**15**	**15**
Barium (Ba)	500	2000	74	160	40
Beryllium (Be)	4	8	1	<2.0	<2.0
Bismuth (Bi)	NG	NG	NM	<2.0	<2.0
Boron (B)	NG	NG	<7	<50	<50
Cadmium (Cd)	10	22	0.46	<0.30	<0.30
Calcium (Ca)	NG	NG	4100	4700	2800
Chromium (Cr)	64	87	**110**	**130**	**69**
Cobalt (Co)	50	300	5.6	5.0	4.1
Copper (Cu)	63	91	<10	8.4	7.9
Iron (Fe)	NG	NG	19000	33000	30000
Lead (Pb)	140	260	12	12	11
Lithium (Li)	NG	NG	<1	<2.0	<2.0
Magnesium (Mg)	NG	NG	380	1000	850
Manganese (Mn)	NG	NG	7300	9400	5300
Mercury (Hg)	6.6	24	<0.1	<0.10	0.12
Molybdenum (Mo)	10	40	<1	<2.0	<2.0
Nickel (Ni)	50	50	9.7	9.2	9.1
Phosphorus (P)	NG	NG	700	NM	910
Potassium (K)	NG	NG	<350	100	<100
Rubidium (Rb)	NG	NG	NM	<2.0	<2.0
Selenium (Se)	1	2.9	**3.2**	**1.7**	**1.7**
Silver (Ag)	20	40	<1	<0.50	<0.50
Sodium (Na)	NG	NG	<350	<100	<100
Strontium (Sr)	NG	NG	17	23	13
Thallium (Tl)	1	1	<0.7	<0.10	<0.10
Tin (Sn)	50	300	<10	<2.0	<2.0
Titanium (Ti)	NG	NG	110	NM	NM
Uranium (U)	23	33	2.7	2.2	1.9
Vanadium (V)	130	130	56	85	59
Zinc (Zn)	200	360	180	170	140

BOLD	Exceedance of the CCME Commercial Soil Quality Guidelines.
BOLD	Exceedance of the CCME Residential Soil Quality Guidelines.

NG: no applicable guideline; NM: parameter was not measured.

Table B3 SPLP Analysis of Solids Residual

Metals (Leachable)	CCME Marine (µg/L)	CCME FWAL (µg/L)	SPLP (µg/L)		
			20-Nov-14	20-Nov-14(3)	20-Nov-14(3)
pH			6.54	6.22	5.80
Aluminum	NG	100	69	**210**	**210**
Antimony	NG	NG	<2.0	<2.0	<2.0
Arsenic	12.5	5	<2.0	**7.5**	**12**
Barium	NG	NG	24	130	140
Beryllium	NG	NG	<2.0	<2	<2.0
Boron	NG	1500	-	-	-
Cadmium	0.12	0.09	**<0.30**	**<0.30**	**<0.30**
Calcium	NG	NG	6000	17000	25000
Chromium	56(1)	8.9(1)	<2.0	<2.0	<2.0
Cobalt	NG	NG	<1.0	<1.0	1.4
Copper	NG	2(2)	<2.0	<2.0	<2.0
Iron	NG	300	<50	<50	<50
Lead	NG	1(2)	<0.50	<0.50	<0.50
Lithium	NG	NG	<2.0	<2.0	<2.0
Magnesium	NG	NG	300	1400	9200
Manganese	NG	NG	9700	22000	51000
Molybdenum	NG	73	<2.0	<2.0	<2.0
Nickel	NG	25(2)	<2.0	<2.0	<2.0
Phosphorus	NG	NG	<100	<100	<100
Potassium	NG	NG	140	960	2000
Selenium	NG	1	<1.0	<1.0	<1.0
Silver	NG	0.1	**<0.50**	**<0.50**	**<0.50**
Sodium	NG	NG	1700	2600	4100
Strontium	NG	NG	5.1	41	77
Thallium	NG	0.8	<0.10	<0.10	<0.10
Tin	NG	NG	<2.0	<2.0	<2.0
Uranium	NG	15	<0.10	<0.10	<0.10
Vanadium	NG	NG	<2.0	<2.0	<2.0
Zinc	NG	30	<5.0	<20	21

BOLD Indicates an exceedance of the CCME FWAL criteria.

(1) Criteria is for total chromium, not hexavalent chromium.

(2) Criteria is dependent on water hardness. Listed criteria is for unknown water hardness. (3) Sample was oven dried at 104 °C prior to being analyzed.

Table B4 Chromium Load from Aluminum Sulfate

Parameter	Units	Result
Raw Flowrate (low)	LPD	9230000
Raw Flowrate (high)	LPD	9958000
Chromium in Aluminum Sulfate	mg/L	0.7914
Aluminum Sulfate Dose	mg/L Raw	57
Aluminum Sulfate SG	unitless	1.335
Aluminum Sulfate Load (low)	LPD	394.09
Aluminum Sulfate Load (high)	LPD	425.17
Chromium Load from Aluminum Sulfate (low)	**g/day**	**0.31**
Chromium Load from Aluminum Sulfate (high)	**g/day**	**0.34**

Table B5 Aluminum Load from Aluminum Sulfate

Parameter	Units	Result
Raw Flowrate (low)	LPD	9230000
Raw Flowrate (high)	LPD	9958000
Aluminum Sulfate Dose	mg/L Raw	57
Aluminum Sulfate Molar Mass	g/mol	584.31
Aluminum Molar Mass	g/mol	26.98
Aluminum Sulfate Composition	% $Al_2(SO_4)_3 \cdot 14H_20$	48.50
Aluminum Sulfate Load (low)	kg/day $Al_2(SO_4)_3 \cdot 14H_20$	255.16
Aluminum Sulfate Load (low)	mol/day $Al_2(SO_4)_3 \cdot 14H_20$	436.69
Aluminum Sulfate Load (high)	kg/day $Al_2(SO_4)_3 \cdot 14H_20$	275.29
Aluminum Sulfate Load (high)	mol/day $Al_2(SO_4)_3 \cdot 14H_20$	471.13
Aluminum Load from Aluminum Sulfate (low)	**kg/day**	**23.56**
Aluminum Load from Aluminum Sulfate (high)	**kg/day**	**25.42**