Removal of Aluminium in Filter Backwash Water: A Treatment Optimization Case Study

by

Megan Elizabeth Wood

Submitted in partial fulfilment of the requirements
for the degree of Master of Applied Science

at

Dalhousie University
Halifax, Nova Scotia
April 2014

DEDICATION

I dedicate this thesis to my mother, Susan Wood, for her support, love and motivation throughout my academic career.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABSTRACT

This is a study of filter backwash water handling methods and the development of an optimization strategy for the current treatment process at a water treatment plant in Nova Scotia. The aluminum concentration at the point of regulation was found to always exceed the guideline of 184 μg/L with an average discharge of 669±471 μg/L. Tracer studies showed significant short-circuiting with a minimum retention time of five hours and forty-nine minutes for the entire treatment system. The treatment lagoons were modeled using fluid dynamics and different baffle placements were compared; two evenly spaced longitudinal baffles displayed the biggest improvement in retention time. Bench-scale settling tests determined an optimal polymer dose of 5 mg/L using a cationic medium molecular weight polymer with a settling time of 1.5 hours, which is representative of the initial discharge of the two baffle CFD model. The aluminum concentration with this combination of improvements was reduced to 101 μg/L.

List of Abbreviations and Symbols Used

2D Two-dimensional

3D Three-dimensional

Al Aluminum

Alum Aluminum Sulphate

BCs Boundary Conditions

c Concentration

CCME Canadian Council of Ministers of Environment

CFD Computational Fluid Dynamics

CV Control volume

CWQG Canadian Water Quality Guidelines

d_{H_2O} Depth of the water

DOC Dissolved Organic Carbon

dt size of the time step

FBW Filter Backwash Water

FIDAP Fluid Dynamics Analysis Package

HRT Hydraulic Retention Time

JDKWTP J. Douglas Water Treatment Plant

kg Kilograms

g grams

MDL Minimum detection limit

mg miligrams

mL mililitres

Mn manganse

N number of repeats

NA not available

ND non-detect

NTU Nephelmetric Turbidity Units

P phosphorus

P_{H_2O} water pressure

P_{air} air pressure

PWG Potable Water Grade

QUICK Quadratic Upstream Interpolation for Convective Kinetics

RANS Reynold’s Averaged Navier Stokes Equations

Re Reynold’s number

RSM Reynold’s stress equation model

RTD Retention time distribution

SD Standard deviation

SIMPLE Semi-Implicit Method for Pressure-Linked Equations

T temperature

t Time

t_{10} the time it takes 10% of the mass of the tracer has passed through the outlet

t_{90} the time it takes 90% of the mass of the tracer has passed through the outlet

TOC Total organic carbon

TSS Total Suspended Solids

UDF User defined function

USEPA United States Environmental Protection Agency

VOF Volume of fluid

WSP Waste stabilization pond

WTP Water treatment plant

WWTP Wastewater treatment plant

$\bar{\varepsilon}$ Average energy dissipated per unit of mass

ε Local turbulent kinetic energy dissipation

ρ Density

μ Dynamic viscosity

ν Kinematic viscosity

ACKNOWLEDGEMENTS

First and foremost I would like to thank my supervisor Dr. Graham Gagnon for giving me this opportunity and for his guidance throughout this project. I am incredibly grateful and honored to have been part of your research group.

My thesis project was made possible by funding from the NSERC/Halifax Water Industrial Research Chair Program, which includes the following industrial partners: Halifax Water, LuminUltra Technologies Ltd., Cape Breton Regional Municipality and CBCL Ltd.

I would also like to thank Dr. Graham Gagnon and Amina Stoddart for giving me the opportunity to run the JD Kline pilot plant, especially to Amina for her trust and patience while I ran the pilot. It was a fun and invaluable learning experience.

I would like to thank the JD Kline WTP staff especially Peter Flinn for making this project happen. Another thanks for making me feel welcomed and all of your help while I ran the pilot. I know that everything you have taught me about running a WTP will be incredibly useful in my professional development in the future.

Thank you to my committee Dr. Rob Jamieson and Dr. Craig Lake for reviewing my thesis. In addition, your extra help and technical guidance along the way was very much appreciated.

I send many thanks to the research team including Jessica Campbell, Heather Daurie, and Elliot Wright for your help in the lab and running my samples. I would like to thank Wendy Krkosek for her efforts with the STEWARD program. I have learned a lot from the seminars; thanks to you for organizing quality speakers. The symposiums were always a lot of fun and full of learning experiences. Special thanks to Tarra Chartrand for all of your help keeping things organized in the group. To everyone else in the research group thank you for your friendship over these past two years and making this an enjoyable process.

Yamuna Srinivasan Vadasarukkai thank you for your friendship, mentorship, and teaching me the ways of CFD. I could not have done this without you.

Finally, I would like to thank my friends and family afar. All the love sent by any form possible, snail-mail, e-mail, texts, long phone calls and visits were always much appreciated. Ending every conversation with, when are you coming ‘home’, forced me to stay focused and for that I am grateful.

Chapter 1: INTRODUCTION

1.1 BACKGROUND

Wastes produced at drinking water treatment plants (WTP) include liquid, solid and gaseous materials removed during the water treatment (MWH 2005). Filter backwash water (FBW) is the wastewater produced as a result of the filter cleaning (backwashing) process at a drinking WTP and it usually accounts for 2-5% of WTPs production (MWH 2005). FBW is highly concentrated in drinking water treatment chemicals and materials from the source water removed during filtration. It usually contains approximately 50 to 1,000 mg/L of solids (Droste 1997). The constituents of concern in FBW include pathogenic microorganisms, turbidity particles, total organic carbon, and heavy metals (MWH 2005). Therefore, FBW must be treated before being discharged into receiving waters to prevent pollution and detrimental effects to the aquatic species and environment.

Aluminum sulphate is a common coagulation chemical in the water treatment industry. As a result, the FBW from WTPs that use this chemical is highly concentrated in aluminum. Aluminum is the third most abundant metal on earth, making up 8% of the earth's crust (Sposito 1996). When aluminum becomes mobilized, it can have toxic effects on the aquatic species living in surface waters. For example, acid rain can increase aluminum in the waters by releasing aluminum normally bound in soils and rocks. Due to the recent acidification of waters systems many studies have investigated the toxicity of aluminum on aquatic species (Dennis and Clair 2012; Gundersen *et al.,* 1994; Booth *et al.,* 1988; Wood *et al.,* 1990; Driscoll 1985). The increase in aluminum in the receiving waters and the toxic effects it can have on aquatic species has lead the Canadian Council of Ministers of the Environment (CCME) to impose aluminum discharge guidelines. The Canadian Water Quality Guidelines (CWQG) for the Protection of Aquatic Life for aluminum are as follows (CCME 1999):

5 μg/L at pH<6.5, $[Ca^{2+}]$< 4 mg/L and DOC< 2 mg/L

100 μg/L at pH ≥6.5, $[Ca^{2+}]$ > 4 mg/L and DOC ≥ 2 mg/L

The aluminum concentrations in lakes in Nova Scotia are typically higher than the CWQG this is possibly due to the acidification of the waters (Dennis and Clair 2012). In the Halifax region, the aluminum concentrations are in the range of 131 to 219 µg/L (Halifax Water 2013; CCME 2005). The guidelines do not stipulate which water quality parameter (i.e. pH, Ca^{+} concentration or DOC) is the overriding factor or whether aluminum is in total or dissolved forms. The guideline does suggest that aluminum toxicity is largely dependent on other water quality parameters as aluminum interactions are quite complex. In contrast, the USEPA regulates the dissolved aluminum without regard for other water quality parameters. The regulation is set at 87 µg/L on a four-day average and 750 µg/L on a one-hour average of acid soluble aluminum (USEPA 2005).

The Nova Scotia Treatment Standards for Municipal Drinking Water Systems requires treatment of FBW prior to discharge into receiving waters to prevent pollution of the receiving waters (Nova Scotia Environment 2012). Lagoons paired with a wetland to receive lagoon effluent can be a suitable FBW treatment method, as they have low capital cost (Shilton 2005). Metal removal mechanisms in lagoons include sedimentation, adsorption, biological uptake and precipitation. Lagoons can also provide evident treatment for FBW. However, typical lagoons studies monitor performance in terms of biological oxygen demand BOD removal and little work has been done to quantify such systems in terms of removal of other pollutants (Thirumurthi 1974; Shilton 2005). Inadequately designed lagoons can result in the discharge of water with remaining pollutants. There are few studies that have looked at the use of lagoons for removal of heavy metals and even fewer studies that have looked at wetlands for the treatment of water high in aluminum from sources other than acid mine drainage (Kadlec and Wallace 2009; Shilton 2005).

There are many reasons for FBW to go inadequately treated. The properties and volume of FBW produced can only be determined once the WTP starts producing water; therefore, the system can often be under designed. Another reason for the under designed FBW treatment systems is that they are often put into place without considerable deliberation, as the treatment of FBW is not the main purpose of a plant. Finally, many WTPs were designed and commissioned before the implementation of discharge guidelines and the discharge guidelines have become more stringent over the years. For these reasons, retrofits and upgrades to the FBW handling systems often are required.

Improvements to the hydraulic performance of lagoons are often sought when attempting to improve treatment efficiency because contaminate concentration will decrease with increased retention time (Shilton 2005). Due to the simplicity of lagoon design, the hydraulic performance can be easily improved. Hydraulics are improved by creating conditions that lend themselves to a more plug flow nature, including multiple ponds in series, increasing the length to width ratio, adding baffles and dissipating inlet flow rate (Shilton 2005).

Redesign without knowing the extent of the benefits can be expensive and possibly futile. Therefore, modeling retrofit scenarios to determine the effects on the hydraulics of the lagoons can be efficient in both time and money. Computational fluid dynamics (CFD) can produce models of comparable accuracy to physical scale models (Muisa *et al.,* 2011; Baawain, *et al.,* 2006). However, CFD provides much greater detail in the flow patterns, including eddies, short circuiting, dead zones, and velocity quantification at each point within the domain than a physical model (Hamzah *et al.,* 1997; Camnasio *et al.,* 2011). Since velocity governs transport, sedimentation and re-suspension of particulates, CFD could be instrumental in the design of tanks used for sedimentation including lagoons used for the treatment of FBW (Abdulla *et al.,* 1995; Camnasio *et al.,* 2011).

1.2 Research Rationale

This thesis focuses on the treatment of the filter backwash wastewater (FBW) at the J. Douglas Kline Water Treatment Plant (JDKWTP) in Halifax, Nova Scotia. This water treatment plant (WTP) produces approximately 2,000 m^3 of high aluminum FBW every day. To treat the FBW, the plant takes a passive approach and utilizes the vast quantity of land surrounding the WTP owned by the utility. A two-step process for treatment of the FBW is made up of engineered lagoons followed by a natural wetland.

Currently the FBW treatment at the JDKWTP monitoring is limited to a weekly water quality sample. Sampling conducted by a third party suggests that the aluminum regulation is regularly exceeded. For these reasons, it is necessary to monitor the current treatment process to identify deficiencies and to use this data to develop a solution for elevated aluminum levels.

1.3 Research Objectives

The main objectives of this work were as follows:

1. Hydraulically characterize the filter backwash water treatment site at the JDKWTP;
2. Assess current water quality moving through the system;
3. Create a fluid dynamics model of the engineered lagoons to determine the hydraulic characteristic within the lagoon;
4. Optimize fluid mechanics in the lagoon by adding flow restriction (e.g., baffles) to maximize treatment potential; and
5. Experimentally determine if treatment chemicals (e.g., polymer type; polymer dosage) can assist in treatment performance.

1.4 THESIS STRUCTURE

This thesis is structured into the following five chapters:

Chapter 2 is a review of relevant literature of topics covered in this thesis. It discusses other FBW handling methods at various WTPs to put the JDKWTP methods into context. The different types of hydraulic models are compared to show the pros and cons of each. The evolution of CFD use in the water treatment industry is discussed to demonstrate where the field currently is. Finally, studies comparing various water matrices to determine aluminum toxicity and cases of wetlands employed to treat waters high in aluminum are discussed.

Chapter 3 provides an overview of the JDKWTP process, the filter backwashing process and the FBW treatment site. This chapter also describes the analytical methods used for the water samples, and the discharge criteria for the FBW at the JDKWTP.

Chapter 4 discusses the results from the hydraulic characterization of the JDKWTP FBW lagoons including tracer studies of the lagoons and wetland. This chapter also discusses and compares water quality throughout the wetland CWQG and compliance guidelines.

Chapter 5 analyzes the results from the CFD model of the engineered lagoons. Various baffle placements within the lagoons were simulated to determine the effect they would have on the flow patterns and retention times within the lagoons. Using the retention times computed from the CFD results bench-scale settling tests were used to determine the impact of the retention time changes on water quality. Additionally, the results from the bench-scale polymer experiment are presented.

Chapter 6 concludes the research and summarizes the findings. Possible next steps and suggestions for future research are also discussed in this chapter.

Chapter 2: LITERATURE REVIEW

2.1 ALUMINUM CHEMISTRY

Aluminum in water can come from many different sources; it can occur naturally in surface water, a small portion is in hydrated ionic form but most of it is complexed with silicates (Kadlec and Wallace 2009). It is also released from the soil and bedrock due to acidification of waters. Aluminum is added in the water treatment process in the form of aluminum sulphate or alum ($Al_2(SO_4)_3 14H_2O$) as a coagulant to destabilize colloidal particles (Droste 1997). During coagulation, reaction [2.1] takes place, the alum reacts with the calcium hydroxide or lime and aluminum hydroxide is produced and consequently filtered out ending up in the FBW (Droste 1997).

$$Al_2(SO_4)_3 14H_2O + 3Ca(OH)_2 \rightarrow 2Al(OH)_3 + CaSO_4 + 14H_2O \quad [2.1]$$

If the FBW is not properly treated, the aluminum that remains in the water can have toxic effects on the aquatic species in the receiving waters. Aluminum toxicity in water to biota is determined by the chemical makeup of the water. Solubility of aluminum is dependent on the pH of the water (Figure 2.1). Aluminum is most soluble at a pH greater than 9 and at a pH of 4 to 5. Above a pH of five solubility decreases to a peak in insolubility at a pH of 7 and then solubility increases (Kaldec and Wallace 2005). Aluminum precipitates out of solution as amorphous $Al(OH)_{3(s)}$ (or as its mineral name, gibbsite).

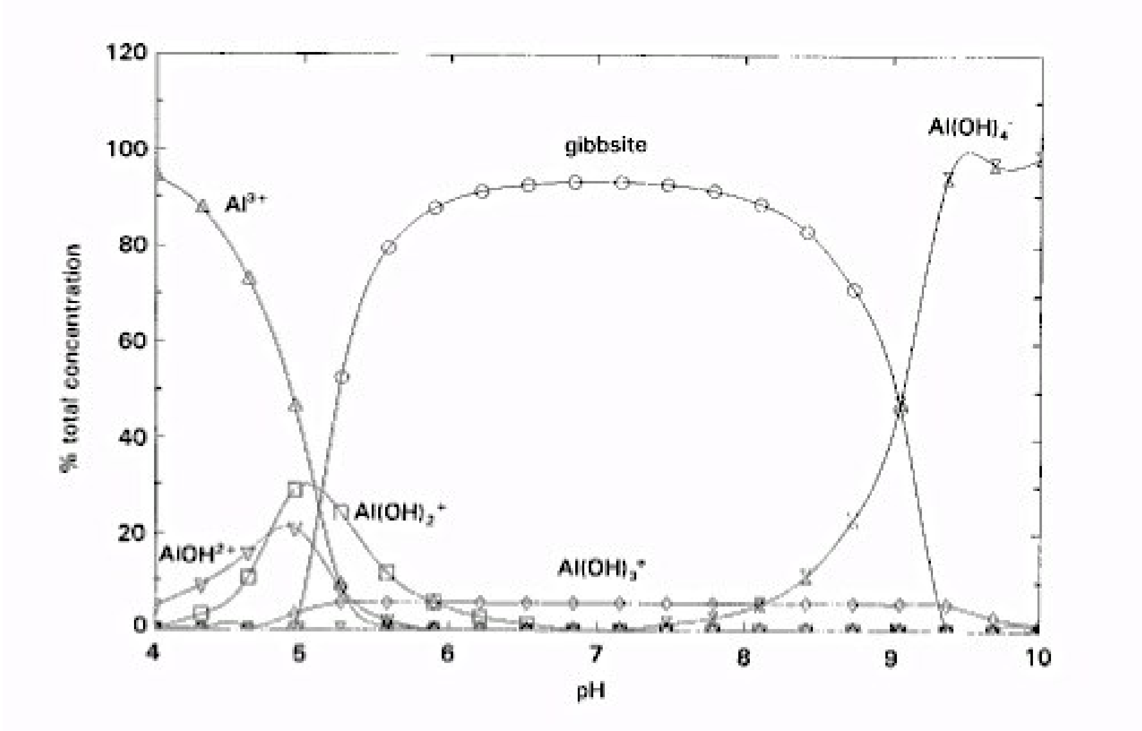

FIGURE 2.1 SOLUBILITY OF ALUMINUM AT 15°C BASED ON EQUILIBRIUM CONSTANTS AFTER ONE HOUR OF REACTION TIME (GENSEMER, 2000).

2.2 Aluminum Toxicity

The effect of aluminum in water has been comprehensively studied in North Eastern United States of America, Scandinavia, and Eastern Canada due to acidification of water bodies releasing aluminum from the bedrock (Baldigo *et al.,* 2007; Dennis and Clair 2012; Driscoll *et al.,* 1980; Driscoll 1985). Aluminum can have toxic effects across all trophic levels.

2.2.1 Effects of Aluminum in the Water on Planktonic Species

Aluminum has been shown to affect planktonic species at concentrations far lower than more upper trophic level aquatic life; this is because aluminum does not biomagnify (Kadlec and Wallace 2009). Many researchers have looked at the effect of decreasing pH and increasing aluminum concentration on phytoplankton's phosphate uptake. They found that phosphate uptake was more inhibited at pH ~6 with elevated aluminum (250 μg/L to 400 μg/L) than at a more acidic pH (~4.5) with elevated aluminum (Nalekwajko and Paul 1985; Parent and Campbell 1994; Gensemer 2000; Pettersson *et al.,* 1988).

2.2.2 Effects of Aluminum in the Water on Fish

The toxicity of aluminum to aquatic species is highly dependent on the water quality. The toxicity of aluminum is dependent on the organics or inorganics ligands it is bound with, the concentration of aluminum, and the temperature and the pH of the water (Wauer *et al.,* 2004; Ingersoll *et al.,* 1990).

Muisa *et al.,* (2011) performed a field study of fish living in waters with elevated aluminum (31.18 ± 21.5 mg/L) caused by WTP FBW discharge and found that the gills and muscles had relatively low aluminum concentrations, whereas the highest amount of aluminum was found in the liver and kidneys.

The mechanism that causes aluminum toxicity is under debate due to the fact that it is hard to differentiate between the symptoms that are primarily due to the elevated aluminum concentrations and those that are secondary (Poleo 1995). However, the proposed mechanism is due to the microenvironment created in the waters nearest to the fish gills caused by gas exchange across the surface of the gill. The pH of the water nearest the gills is slightly more basic with the release of NH_4 causing the aluminum to precipitate and bind with the gills. The precipitated aluminum on the gills causes suffocation of the fish (Walt 2000; Gensemer and Playle 1999). Calcium acts to buffer the effects of the aluminum by competing with the aluminum for the binding sites on the gills (Wood *et al.,* 1998; Gensemer and Playle 1999). Dissolved organic carbon (DOC) can also reduce toxicity by complexing with the aluminum reducing the amount of aluminum in the solution.

2.2.3 Effects of Aluminum in the Water on Humans

Aluminum has been loosely correlated with Alzheimer’s and other neurodegenerative diseases due to increased aluminum concentrations found in the brains of sufferers. However, few conclusive studies have been performed due to the host of environmental factors that are linked with these diseases (Droste 1997; Kozłowski *et al.,* 2006).

McLachlan *et al.,* (1996) analyzed brains of people who had died with neurodegenerative diseases and the amount of aluminum in the drinking water of their ten previous residences. It was found that based on 1991 Canadian population statistics approximately 15,000 to 26,000 Alzheimer's cases could have been prevented if aluminum concentration in tap water was consistently below 0.1 mg/L. This is lower than the aluminum guideline for a direct filtration plant, which is set at 0.2 mg/L (Health Canada 2010).

Gauthier *et al.,* (2000) looked at the concentration of various aluminum species in the drinking water supply of 68 Alzheimer's patients of the Saguenay-Lac-Saint-Jean region in Quebec and found no significant correlation between aluminum and Alzheimer's.

Frecker (1991) studied aluminum in the drinking water and looked at causes of death in communities in Newfoundland. No communities exceeded the drinking water quality guideline of 0.2 mg/L (Health Canada 2010). However, the community of Newtown with the highest aluminum concentration of 0.165 mg/L and most acidic water (pH of 5.2) had the highest dementia death rates (Frecker 1991).

2.3 WETLAND

The Nova Scotia Environment Act (1994a) defines a wetland as land commonly referred to as a marsh, swamp, fen or bog. These either periodically or permanently have a water table at, near or above the land's surface or that is saturated with water, and sustains aquatic processes as indicated by the presence of poorly drained soils, hydrophytic vegetation and biological activities adapted to wet conditions. Nova Scotia currently has approximately 360,000-ha of freshwater wetlands (Nova Scotia Environment 2011). Wetlands provide a wide range of benefits to the environment including mitigating storm flows, and storing and removing high concentrations of pollutants (Nova Scotia Environment 2011).

2.4 WETLANDS FOR ALUMINUM REMOVAL

Metals are removed through various mechanisms in wetlands including sedimentation, adsorption to biomass, bioaccumulation into the biomass, precipitation and chelation

(Shilton 2005). The most common removal method is through the metals adsorbing to solids and settling (Shilton 2005).

Goulet *et al.,* (2005) used mesocosms to determine the bioaccumulation of Al in different macrophytes with Al smelter wastewater. A 54-66% decrease in total aluminum from 1.05-1.66 mg/L influent to 0.48-0.55 mg/L effluent was observed. The main cause in decrease in aluminum concentration was attributed to deposition of particulate and there was not a significant decrease in dissolved aluminum. The aluminum uptake by the macrophytes was not found to be detrimental (Goulet *et al.,* 2005).

Wieder *et al.,* (1990) also looked at wetlands ability to treat acid coalmine drainage, which contained 10 mg/L of aluminum, with pilot scale wetlands. The first two weeks that the wetlands were run there was an accumulation of aluminum. For the following 15 weeks of the study the wetlands became a source of aluminum (Wieder *et al.,* 1990).

Wieder (1989) did a survey of constructed wetlands used to treat coalmine drainage in the Eastern United States. Twenty sites that were reporting influent and effluent aluminum concentration were analyzed. It was found that the median aluminum removal was 47.7% with an average influent of 21.2 ± 5.5 mg/L of total aluminum and an average effluent of 10.8±3.0 mg/L (Wieder 1989).

Kaggwa *et al.,* (2001) studied a wetland used as the receiving waters for alum sludge from the Gaba II WTP in Uganda. The average aluminum concentration in the water directly downstream of the discharge was 1.54 mg/L. The study showed root abnormalities in the plants and phosphorus deficiencies in the vegetation (Kaggwa et al., 2001).

A model by Flanagan *et al.,* (1994) was used to predict the water quality results (specifically aluminum and iron concentrations) for a constructed nine-cell treatment wetland in Ohio for acid mine drainage based on two years of preconstruction data. The model was zero dimensional accounting for diffusion, stoichiometric relationships, precipitation, role of vegetation, and pore water. Two scenarios were simulated for aluminum retention; circumneutral pH substrate with aluminum retention 63% increasing to 93% with increasing age of the and low pH substrate with aluminum retention ranging from 0 to 60% depending on the season.

Once the wetland was constructed, Mitsch and Wise (1998) analyzed its' performance and compared the results to preconstruction and modeled water quality. When comparing aluminum concentrations before and after construction at the outflow there was a 55% decrease (from 62±14 mg/L to 28±6 mg/L of total aluminum) (Mitsch and Wise 1998). The constructed wetland provided an average aluminum removal of 39.1%, which does fall in the generous range predicted by Flanagan *et al.,* (1994).

2.5 RESIDUALS

Residuals are the wastes produced at all WTPs. They are made up of organic, inorganic, algae, bacteria, viruses, colloids, and precipitated chemicals from the raw water and from the treatment process (Montgomery 1985). Treatment of residuals is required prior to discharge into the environment because they can be toxic to the aquatic species, raise pH of the waters, and increase turbidity (MWH 2005). The method through which the residuals are produced determines their properties and classification. Sedimentation processes, and filter and screen scrapings produce a more solid waste or sludge whereas filter backwashing, membrane filter waste and ion exchange brines are typically a liquid waste (MWH 2005). For brevity sake, this thesis will focus on the liquid waste produced from filter backwashing or filter backwash water (FBW).

2.6 FILTER BACKWASHING

Filter backwashing is the process of removing particles from the filter pores by pumping filtered water backwards through the filter at a rate high enough to fluidize the filter media and remove the particles in the pores. Filters are typically backwashed based on one of three events; either filter effluent greater than 0.2 NTU, a head loss of 1.8m to 3m or a set run time (Crittenden and Montgomery 2005). The water produced during the backwash (filter backwash water) can represent up to three percent of a water treatment plant's water production (Cornwell and Lee 1994). FBW quality is highly dependent on the source water, amount of water used in the backwash, the treatment train and its' efficiency. Regardless, it is typically high in organics, coagulant chemicals and suspended solids (CBCL 2004; Ecuru *et al.,* 2011).

2.7 FBW HANDLING METHODS

FBW handling refers to the method of dewatering and disposal of the FBW produced at the WTP (MWH 2005). The ultimate goal of FBW handling is to dewater the solids as much as possible prior to disposal to save on the cost of transporting the waste to the utility (MWH 2005). Ultimately waste is usually disposed of at a landfill and transporting the solids is often the most costly step of the FBW handling. Various methods are used to dewater FBW to reduce the liquid content to a solids concentration of 35% before being landfilled (Montgomery 1985).

Flow equalization tanks are often used as the first step in the treatment process of FBW to mitigate the effects of extreme differences between the high and low flow periods and some settling. Coagulants can be added such as polymers to promote settling (MWH 2005). Other thickening methods include dissolved air flotation, centrifugation, and gravity belt (Droste 1997). High rate sedimentation is frequently used to dewater FBW, as it is a more cost-effective method than mechanical dewatering. The FBW is first flocculated with polymer or a coagulant and allowed to settle in a clarifier with lamella plate settlers (MWH 2005). The thickened solids can be dewatered with lagoons, drying beds, or vacuum filters. Dissolved air flotation (DAF) has been shown to effectively treat FBW especially those characterized by light organic material and high in metal hydroxides (Cornwell 2001). Bench-scale experiments in Bourgeois *et al.* in 2004 demonstrated that FBW could be treated to produce water of equal or better quality through sedimentation, DAF, gravity thickening or ultrafiltration. For WTPs located with proximity to a wastewater treatment plant (WWTP), the FBW can be fed via sewer to the WWTP. The addition of FBW to the WWTP process is cost-effective for the WTP and it enhances the solids separation at the WWTP (Droste 1997). Due to the fact that all FBW has unique properties, the optimal treatment method will also be unique (Kaggwa *et al.,* 2001; Novak and Langford 1977).

2.8 Relevant Cases of FBW Handling at Utilities

The following is a summary of methods of FBW treatment employed at various WTPs around the world:

At the Kingston WTP (Kingston, Ontario), FBW and sludge from the settling tanks are directed to equalization tanks where Magnafloc 120L is added to promote settling. The water then flows through plate settlers and is dechlorinated with 30% bi-sulphite solution. The supernatant is discharged into Lake Ontario (Utilities Kingston 2012).

At the Harara WTP in Zimbabwe, 110,000 m^3 of waste made up of FBW and sludge from settling tanks is produced daily. The alum based waste is directly discharged into the Manyame River (Muisa, *et al.,* 2011). Aluminum levels in the water (38-69 mg/L), sediments (100-120 mg/g) and the aluminum in the fish directly after the discharge point was significantly elevated but returned to ‘normal’ levels (water 2 mg/L, and sediments 8 mg/g) 8 km downstream (Muisa, *et al.,* 2011).

The Negeri Sembilan WTP in Malaysia is a conventional plant that uses alum as a coagulant. The waste stream is directly discharged the into the Linggi River, which is heavily polluted due to urbanization surrounding it (Hamzah *et al.,* 1997). Aluminum levels in water and sediments were monitored upstream and downstream of waste discharge point (Abdulla *et al.,* 1995). It was found that 2 km downstream of the waste discharge the aluminum concentrations were elevated (average 33 mg/kg in the sediment and 0.84 mg/L in the water). Aluminum concentration returned to very close to upstream levels 15 km downstream (average 24 mg/kg in the sediment and 0.69 mg/L in the water) (Abdulla *et al.,* 1995).

At the Kampala WTP in Uganda, 14,000 m^3 of FBW is produced by the conventional plant and directly discharged into the Gaba swamp, which is used by the community for fishing and agriculture (Ecuru *et al.,* 2011). The aluminum levels in the wetlands decreased from 1.54 mg/L to 0.17 mg/L within 250 m of the discharge point (Kaggwa *et al.,* 2001).

2.9 FBW HANDLING IN ATLANTIC CANADA

In Atlantic Canada, FBW must be discharged downstream of the raw water intake and cannot be recycled to the head of the plant (Nova Scotia Environment 2012). In comparison, the USEPA allows recycling as long as the FBW is returned to the head of the plant so that it passes through all of the plant processes (USEPA 2001) as per the Filter Backwash Rule.

The water produced from dewatering the FBW can be discharged to receiving waters or wastewater treatment plants. If the water is being discharged into the environment, the water must be sufficiently treated to meet increasingly stringent discharge guidelines set in Canada by the Canadian Council of Ministers of the Environment (CCME). Typically, in Atlantic Canada, FBW is dewatered passively, to reduce the cost of hauling to a landfill. A common passive treatment method includes engineered lagoons used for settling. The Atlantic Canada design guidelines for lagoons used for treatment of WTP wastes are as follows (CBCL 2004):

- HRT of between 15 and 30 days
- Two years of sludge storage
- Minimum of two lagoons in parallel
- Located in an area free of flooding
- Minimum depth of 1.5m
- Minimum freeboard of 0.9m
- Adjustable decanting device
- Low permeability liner
- Effluent sampling location
- Outlets and inlets located to minimize short circuiting
- Volume of ten times the volume of water discharged during a twenty-four hour period
- Minimum length to width ratio of 4:1
- Width to depth ratio of 3:1
- Velocity to be dissipated at the inlet

Other methods may be required in addition to lagoons for treatment of FBW high in aluminum as it can be difficult to dewater due to the lightweight nature of the floc particles (CBCL 2004).

2.10 FBW CONDITIONING

Conditioning FBW has been shown to increase settling rates and improve supernatant water quality (William *et al.,* 2007; NRC 2004). Synthetic polymers are commonly used for flocculating a wide variety of wastewaters particles and in the coagulation or flocculation steps of drinking water treatment (Böhm and Kulicke 1996; Bolto and Gregory 2007). They have been found to promote sedimentation by increasing floc particle density and thus reduce suspended solids in the supernatant (NRC 2004). There are two steps to the general mechanisms at work, first a reduction in charge repulsion occurs and then the aggregation of the particles (Böhm and Kulicke 1996).

Polymers are defined as substances made up of molecules of a high molecular weight of a long chain of repeating units. They are generally classified by ionic charge, (cationic, anionic, and non-ionic) and by molecular weight (MW), either high $>10^5$, medium 10^5-10^6, and low $<10^4$ g/mol (Bolto and Gregory 2007). The properties of the polymer dictate the driving coagulation mechanism for floc formation. When the polymer has the opposite charge from the surface charge on the particles, the polymer is absorbed on to the particle via electrostatic attraction (Moody 2012). In contrast, when the polymer and particles have the same charge but the particle has metal ion available, a salt linkage occurs (Moody 2012). Nonionic polymers typically employ hydrogen bonds with particles in the water (Moody 2012). Typically, high charge density and low to moderate molecular weight polymers exhibit charge neutralization and adsorption. In this case, when too much polymer is added, particles will gain a positive charge and re-stabilize (Droste 1997).

Many studies have been conducted to determine possible benefits of using a polymer to condition FBW and optimize dose and type of polymer. Novak *et al.,* (1977) found that anionic higher molecular weight polymers were best suited to conditioning WTP sludge. The polymer was used to promote drying in sand drying beds and it was found that the polymer conditioning improved sludge dewatering (Novak and Langford 1977).

Bache and Zhao (2001) attempted to optimize alum sludge settling using Magnfloc LT25 an anionic polymer with a new settling test easily performed by WTP personnel. The sludge was conditioned with the polymer, allowed to settle for thirty minutes in 100 mL graduated cylinder and performance was measured in terms of height of water-sludge interface. It was found that 10 mg/L provided the greatest amount of settling (Kadlec 1994; Bache and Zhao 2001). Zhao (2004) looked at polymer to enhance settling of alum sludge from a clarifier. A dose of 10 mg/L was again found to be optimal measured in terms of supernatant turbidity. A settling test was performed with varying container sizes of 100 mL, 500 mL and 1 L cylinders and at low doses the settling was found to be independent of container size (AWWA 1996; Zhao 2004).

Adding polymer to the FBW before backwashing the filters has been found to improve filter performance by reducing filter ripening time, the maximum turbidity during ripening and wasted water during ripening (Yapijakis 1982; Logsdon *et al.,* 2002). The polymer worked by increasing adhesion of particles to the filter media (Logsdon *et al.,* 2002). This addition of polymer then went on to improve settling in the FBW sedimentation units (Yapijakis 1982).

2.11 HYDRAULIC MODELING

Hydraulic modeling is employed for engineering purposes often as a preconstruction method to determine how a hydraulic structure will function under various flow conditions. There are three general methods of modeling hydraulic systems; empirically, physically, or hydro-dynamically. Empirical models use a black box approach with statistical or empirical equations to predict the effluent water quality. Physical models employ similitude to scale the hydraulic system to a size appropriate for the laboratory setting. These models work well for retrofitting. Finally, hydrodynamic models or more specifically CFD use Navier-Stokes equations to represent the patterns of flow within the system (Grayman *et al.,* 1996). The type of model used is dependent upon the information the modeler is looking to gain.

2.12 CFD MODELING OVERVIEW

Computational fluid dynamics is the analysis of flow usually of fluid or heat with the within a system (Versteeg and Malalasekera 2007). It combines the study of fluid mechanics, numerical methods and computer science. CFD can be applied to engineered tanks to determine flow patterns within them based on user defined boundary conditions. Flow within tanks of the water treatment industry has traditionally been idealized thus leading to poorly designed tanks. With the use of CFD, the flow can be quantified and improvements to tank design can be made (Walt *et al.,* 2000) identified eight benefits of designing tanks based on CFD results rather than empirical or physical models. These are summarized in Table 2.1 below.

TABLE 2.1 BENEFITS OF DESIGNING BASED ON CFD RESULTS (WALT *ET AL.,* 2000)

Traditional Design Approach	CFD Design Approach
The internal hydraulics was assumed to be plug flow	It is not necessary to assume a flow type as the internal hydraulics can be simulated
Only inlet and outlet conditions are considered as little information is available on the internal hydraulics	Inlet conditions can be used to determine the outlet condition a priori by modeling the internal hydraulics
Designs are based on process specific empirical data	CFD is based on generalized fundamental physical relations
Safety factors are used to overcome the discrepancy between the theory and reality. Process tanks can therefore either over-sized or under-sized.	Safety factors can be limited as there is less uncertainty in terms of the tank hydraulics.
The success of a design can only be determined a posterior	The success of design can be determined a priori
The effect of geometry and internal obstructions on hydraulics can only be determined experimentally and a posteriori on a full scale tank	The effect of geometry and internal obstructions on hydraulics can be simulated. The effect is therefore known a priori
Scale problems plague pilot scale tests	CFD techniques enables the designer to simulate tank performance with actual tank dimensions
Performance indicators are used as a design input that cannot guide design improvements	Performance indicators can be calculated based on CFD results

CFD is able to model flow within user-defined space in either two or three dimensions (2D or 3D, respectively). The flow is represented as changes in velocity and pressure based on the boundary conditions. The changes in velocity and pressure are calculated for each control-volume (CV) of the volume using a Eularian approach. The Eularian approach involves tracking the changes in velocity and pressure in a CV as opposed to the Lagrangian approach, which tracks the motion of a particle of fluid (Versteeg and Malalasekera 2007).

Navier-stokes equations, along with turbulence, species transport, and or multiphase equations are solved through a series of chosen discretization schemes for every CV (Versteeg and Malalasekera 2007). The equations are solved using the finite volume method where the cell values are calculated based on the values of the surrounding CV. Through a series of simplifications and assumptions of the properties of the fluid within the defined volume, the flow parameters can be computed more easily. Commonly, fluids in the water treatment industry are assumed to be incompressible and Newtonian in nature (Wood *et al.,* 1998). When stress and strain are linearly related by viscosity, the fluid is considered Newtonian in nature (Anderson *et al.,* 2009).

2.13 An Evolution of CFD Models of Water Treatment Tanks

CFD has only recently been applied to the WTP industry as it has been made more practical with the advent of commercially available codes. In 1995 one of the first CFD papers looking at pond hydraulics by Wood *et al., 1995* gave a glimpse into the possibilities of this technology. The software at this time was new and the hardware was relatively limited with a mere 1.2 GB hard drive; this resulted in an overly simplified 2D model of laminar flow.

Wood *et al.* (1995) used a finite element approach with the application of FIDAP 7.0 software to model three retrofit scenarios in 2D; two different baffle placements and an aerator were compared against the current rectangular pond geometry with a length to width ratio of 2:1. It was found that a baffle placed close to the inlet increased the stagnation zone; a baffle placed closer to the outlet at two-thirds the width of the pond was found to prevent the short-circuiting that had appeared in the unbaffled pond. In the case of the modeled aerator, a large circular flow around the aerator pattern was created and large dead zones around the perimeter of the pond. No validations for these models were performed.

Wood *et al.,* in 1998 again used FIDAP to model a 2D waste stabilization pond (WSP) except this time they were able to model the turbulent flow regime with a *k*-ϵ turbulence model. They attempted to validate the CFD with the tracer study performed by Mangleson *et al.,* (1972) by using the same geometry and boundary conditions. A simulated tracer study was performed using a transient simulation and a slug injection of a fluid with identical properties to water to represent the tracer. Retention time distribution (RTD) curves were not comparable in most cases and this was attributed to the inability to model the inlet in 3D.

Baawain *et al.,* (2006) created a CFD model of a physical model setup by Hurtig (2003) of E.L. Smith WTP chlorine contact tank. The CFD model was created using the CFX^{TM} software package with a k-ω turbulence model in 3D. Baawain *et al.* compared the CFD tracer studies to the rhodamine tracer studies performing by Hurting (2003) for two different baffle arrangements. Baawain was able to achieve excellent matching RTD and demonstrated that CFD is a cost effective method of modeling retrofits to WSPs. With the use of the CFD model a 73% increase in t_{10} was achieved by changing the inlet design from one with a diameter of 44 mm to seven with diameters of 6 mm.

Similarly He *et al.,* 2004 compared a CFD model to a physical model. Ansys Fluent® was used to model a combined sewer overflow (CSO) in Northern Toronto and compared it to a physical model of the CSO. It was found that the CFD and physical model compared well.

2.14 Hydraulic Flow Patterns in Engineered Reactors

Hydraulics of the wetland is a large determinate in the system's ability to provide treatment to the water (Kadlec and Wallace 1996; Shilton 2005). To apply empirical equations to the wide range of situations, wetlands are simplified and represented as reactors. Engineered reactors are designed based on idealized performance equations. The two most common idealizations are plug flow and continuously stirred (AWWA 1996). Plug flow reactors represent an ideal reactor in which no mixing occurs along the flow path and thus all the fluid elements leave the reactor at the theoretical hydraulic retention time (HRT) (Thirumurthi 1969; Levenspiel 1999). The theoretical hydraulic retention time is defined by the ratio of the pond volume to the inflow rate (Equation 2.2) (Shilton 2005).

$$\theta_{theoretical} = \frac{V}{Q_{in}} \qquad [2.2]$$

Where V is the volume of pond;

and Q_{in} is the flow rate into the pond.

In contrast, continuously stirred tank reactors represent an ideal reactor where the concentration is uniform throughout the entire reactor (Levenspiel 1999). In reality reactors generally behave somewhere between the two ideals due to unsteady flow rates, wind and shear stress at the side and bottoms of the reactor (Thackston *et al.,* 1987).

To determine where on the reactor continuum a reactor falls on the retention time, distribution (RTD) curves are commonly used (Holland *et al.,* 2004). RTD can also be used to compare, quantify and determine the efficiency of a reactor or wetland. Figure 2.5 displays idealized plug flow and CSTR RTD curve.

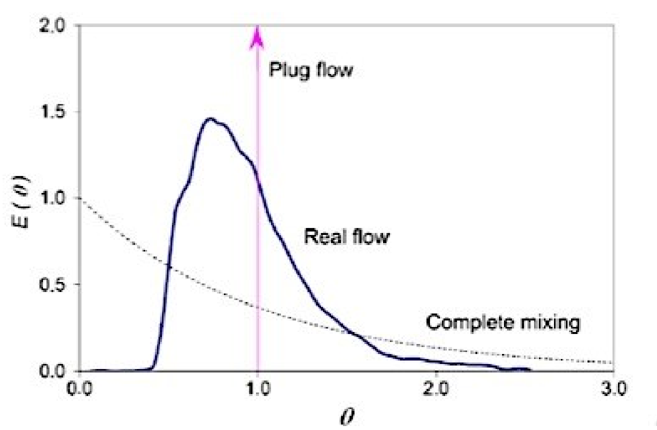

FIGURE 2.2 IDEALIZED RTD CURVES FOR PLUG FLOW AND CSTR TANKS (ADAPTED FROM TEIXEIRA *ET AL.,* 2008).

RTD curves are created through pulse input hydraulic tracer studies (Shilton 2005). Tracer studies are a simple method of developing a more complete understanding of the flow patterns occurring in the reactor. They are conducted with a concentrated pulse of a inert conservative chemical injected at the inlet and the concentration is measured at the outlet (AWWA 1996). The plot of the concentration of the tracer concentration at the outlet over time is the RTD curve (Shilton 2005). Using these curves various hydraulic indices can be derived including:

- Mean retention time
- Dispersion number
- Short circuiting factor
- Theoretical volume
- Time of 10% discharge and 90% discharge
- Dispersion index

Deviations from the ideal plug flow are represented in the reactor as areas of short-circuiting and dead zones. Dead zones are defined as areas within the reactor where the velocity is significantly less than the average and significant recirculation occurs (Thackston *et al.,* 1987). Although the water in these areas receive a high degree of settling time, these areas are not utilized by the bulk of the water and thus reduce the effective volume of the reactor (Thackston *et al.,* 1987). In contrast, short-circuiting is defined as the portion of flow that leaves the reactor in a time of 30 to 40% of the theoretical HRT (Thackston *et al.,* 1987).

Many different retrofit solutions have been developed and modeled to improve the hydraulics of reactors including baffles, changing the inlet outlet placement, berms and aerators (Shitlon 2005).

2.15 APPLICATION OF BAFFLES

Baffles are a simple retrofit option for reducing flow rate and improve the performance of a kind of waste stabilization pond through hydraulics. Baffles act to breakup the flow with the pond to prevent short-circuiting (Shilton 2005). Baffles can be placed in an infinite number of arrangements and many studies have been conducted to optimize baffle placement.

Mangleson *et al.,* (1972) performed the most extensive WSP layout analysis. Nine different baffle arrangements were looked at, as well as nine different inlet and outlet placements. It was found that increasing the length to width ratio by adding baffles provided the greatest increase in hydraulic performance with six longitudinal baffles providing a 30% increase in hydraulic efficiency (Mangleson *et al.,* 1972).

Sah (2011) created a 3D model Deltft3D to compare the effects of a baffled and unbaffled pond from a water quality perspective. It was found that there was no improvement in the water quality with the addition of baffles.

Transverse baffles, which extend a certain width across the pond, are the most commonly used baffle placement. Watters *et al.,* (1973) used a physical scale model to test inlet scenarios and transverse baffle placements in terms of pond hydraulics. Horizontal baffles across the length of the pond were tested; three baffle widths were tested- 50, 70 and 90% width of the pond. It was found that 50% caused increased in the short-circuiting with the baffles not being long enough to force the flow into a snaking pattern. In contrast, the 90% length baffles increased the dead-space in the pond. Thus, 70% width baffles were determined to be the optimal length of the three tested.

Shilton and Harrison (2003) compared the performance varying the number of baffles in a physical scale modeled WSP. No baffles, one, two, four, six and eight baffles at 70% width were compared in terms of reduction of coliform. It was found that the coliform reduction increased with increasing number of baffles with the eight baffles pond having the highest removal. The improvement between two and six baffles was not significant and would not warrant the extra cost. The effect of unevenly spaced baffles was also modeled and it was found that the evenly spaced baffles provided a superior removal efficiency (Shilton and Harrison 2003).

Chapter 3: METHODOLOGY

3.1 STUDY SITE OVERVIEW

3.1.1 J. DOUGLAS KLINE WTP OVERVIEW

The J. Douglas Kline WTP (JDKWTP) in Upper Hammonds Plains Nova Scotia, Canada provides drinking water to Halifax, Bedford, Waverly, Timberlea, Fall River and Sackville. Figure 3.1 describes the water treatment process occurring at the JDKWTP. The water is pumped in from Pockwock Lake, located in a protected watershed. Pockwock Lake water is characterized by low pH, low alkalinity, and low turbidity. The water is treated via screening, coagulation, flocculation, dual media filtration and chlorination. There are three premix tanks where lime ($CaCO_3$) is added in the first tank for pH adjustment up to approximately 10; potassium permanganate is also added ($KMnO_4$) to oxidize iron and manganese. The second mix tank is used for contact time and mixing for oxidization. In the third tank, carbon dioxide is added to bring the pH back down to about 5.5 then 8 mg/L of aluminum sulphate (alum $Al_2(SO_4)_3 14H_2O$) is added as the coagulant and in the winter months a polymer (Magnafloc LT20) is also added to enhance floc strength. The water is then fed into four identical flocculation trains each consisting of two parallel three stage tapered hydraulic flocculation processes. The flocs are then filtered out with the eight dual media filters that consist of 0.6 m of sand and 0.3 m of anthracite on top. To the filtered water, chlorine is added for disinfection, zinc/ortho phosphate is added for corrosion control, sodium hydroxide is added to adjust the pH to about 7.4 and hydrofluosilic acid is added to prevent tooth decay (Halifax Water 2006).

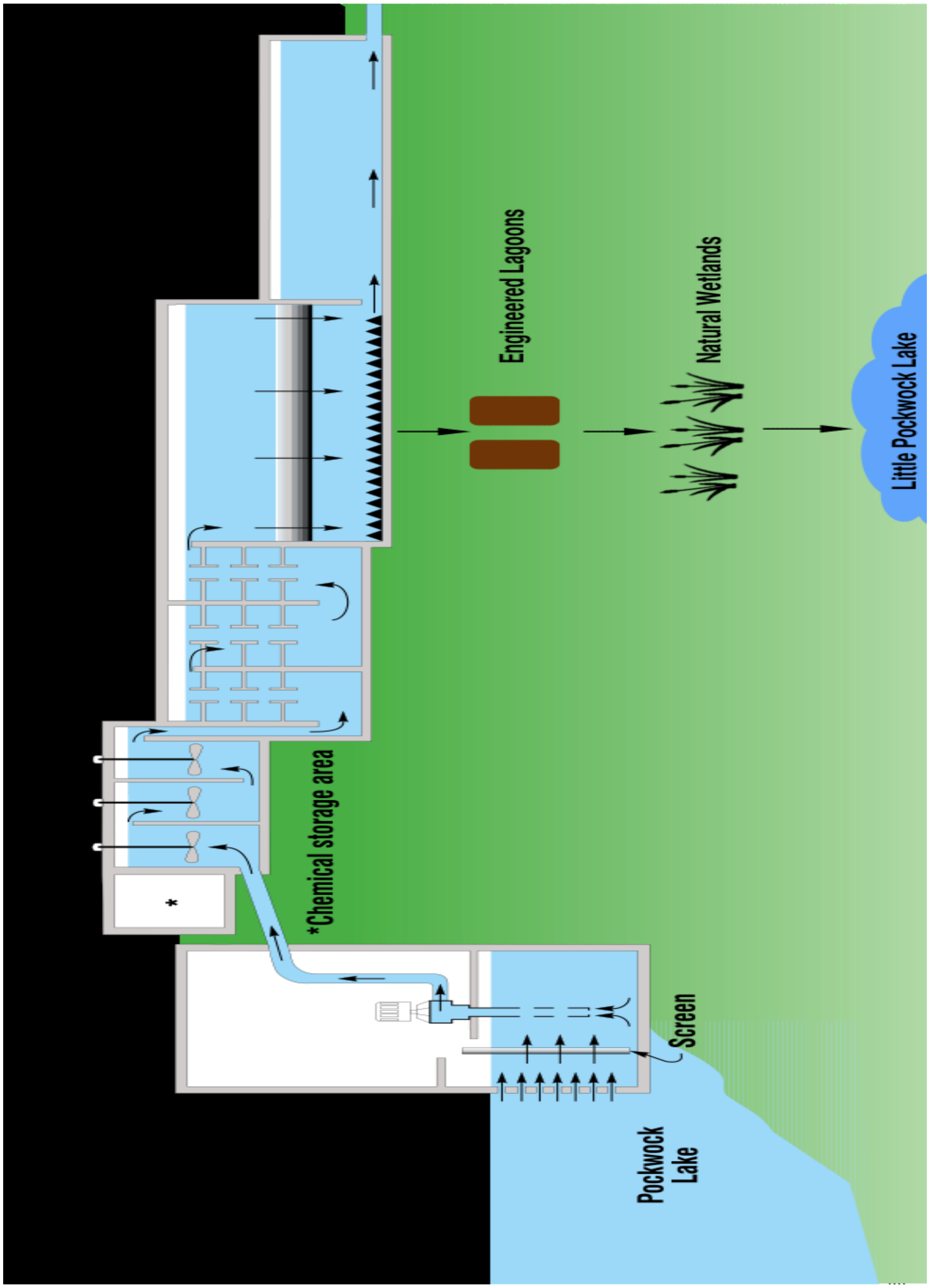

FIGURE 3.1 PROCESS DIAGRAM FOR THE JDKWTP (ADAPTED FROM HALIFAX WATER, 2005).

3.1.2 FILTER BACKWASHING PROCESS AT JDKWTP

The eight dual media filters are backwashed based on three parameters; either turbidity greater than 0.2 NTU, an 80 hour run time, or a head loss of 2.15 m. Before backwashing, the filters are drained. Each filter has eight backwash troughs that run the width of the filter. The gates for the troughs are opened prior to beginning the backwash releasing approximately 25,000 L of water. Then the backwash pumps are turned on and approximately 700,000 L of water is pumped in the reverse direction to flow through the filters to release the particles trapped in the filter's pores by fluidizing the filter media. Skimmers on top of the filter loosen particles on the tops of the filters. All of this water is gravity fed outside of JDKWTP to the FBW treatment area.

3.1.3 FILTER BACKWASH WATER TREATMENT AT JDKWTP

The backwash water treatment at JDKWTP consists of two-engineered lagoons in parallel followed by a natural wetland (Figure 3.2). The water from the backwash is gravity fed through a three foot main down to the two-engineered 14,000 m^3 lagoons located southwest of the plant. The engineered lagoons have clay liners to minimize seepage into the ground. The flow into the two lagoons splits at the gatehouse; the openings can be controlled with stop logs in the gatehouse. The water is fed into the lagoons via a one-meter (3 ft) in diameter vertical inlet located at the opposite end of the lagoons from the outlet. The outlets for both lagoons are overflow weirs and the size of the openings can be controlled in the gatehouse. The overflow from the lagoons combines at the outlet chamber and from there the water is gravity fed north into a natural wetland. The natural wetland is fed with lagoon supernatant from a 1 m (3 ft) main and runoff from the north drying bed. The natural wetlands have a 1.5 km stream running through Hamilton Pond, which then cross the JDKWTP driveway and discharges into Little Pockwock Lake (LPL). The wetlands were not an initially intended part of the FBW treatment; however, they have since been included.

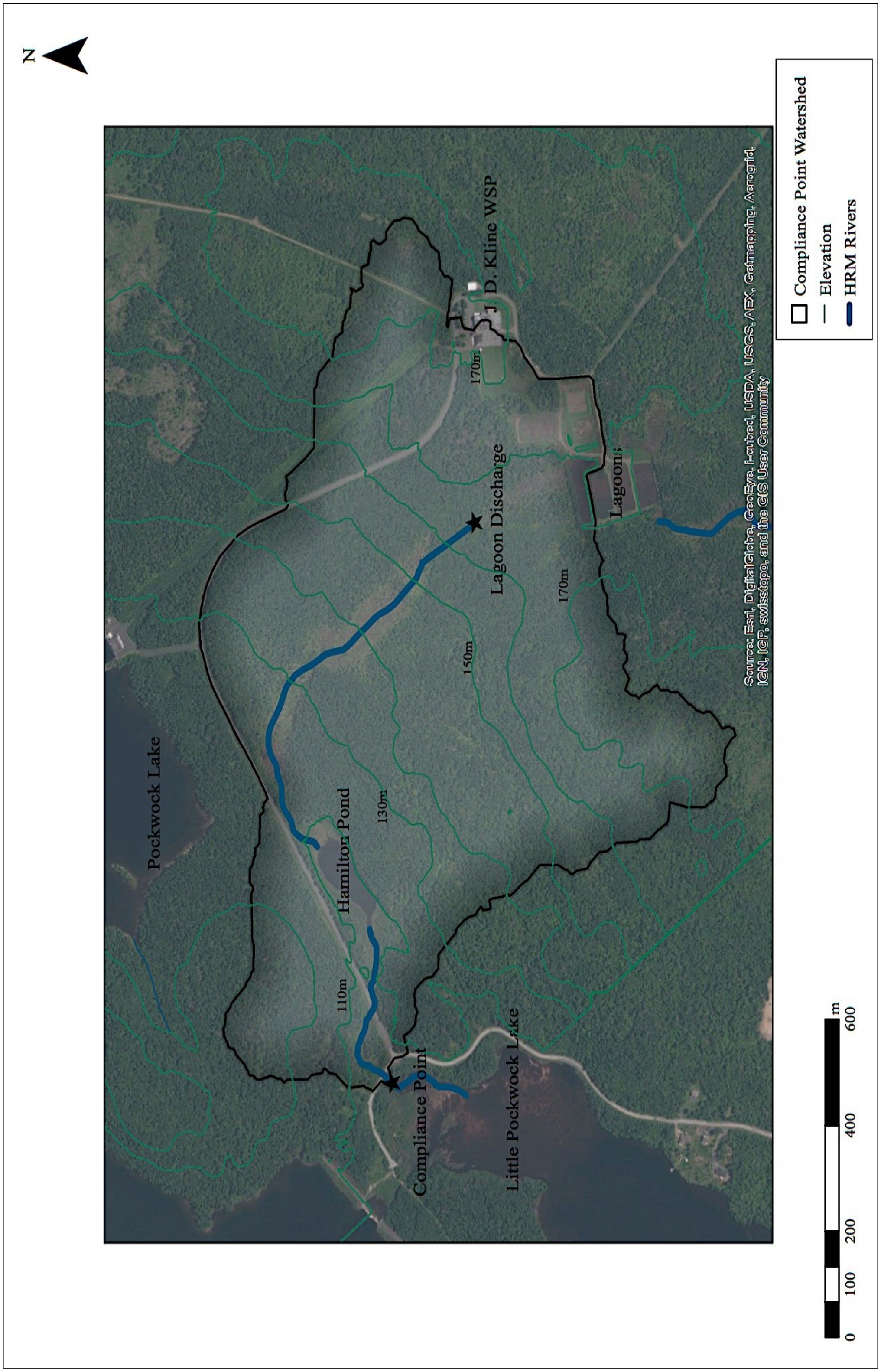

FIGURE 3.2 WATERSHED FOR THE COMPLIANCE POINT OF THE FBW TREATMENT AT THE JDKWTP.

FIGURE 3.3 FBW HANDLING AREA AT JDKWTP

Little Pockwock Lake is downstream of Pockwock Lake; a controlled dam separates them. Where the wetlands discharge into LPL at the edge of Halifax Water's property, the discharge criteria is implemented (the Compliance Point). The engineered lagoons are drained and the settled material is pumped out once a year. The material is pumped over the man-made berm into the either the north or south drying beds (Figure 3.3). Once the material has solidified, the drying beds are dredged with a backhoe and moved to one of the three-engineered drying beds. The engineered drying beds were designed with liners to minimize seepage.

3.2 WATERSHED DELINEATION

The watershed boundaries for the compliance point at LPL were delineated using Light Detection and Ranging (LiDAR) data provided by Dalhousie GISciences Center and ESRI ArcGIS ArcMap10© (ESRI, Redlands, CA, USA) software. The LiDAR data was converted to a 2 m digital elevation model (DEM) from which the watershed was delineated through following the method created by ERSI (ESRI, 2007). The results of the delineation are depicted in Figure 3.2.

3.3 DISCHARGE CRITERIA

Nova Scotia Environment (NSE) has set the discharge criteria into LPL based on the Canadian Water Quality Guidelines (CWQG) for the Protection of Aquatic Life (CCME 1999) except for the aluminum, which was set based on a Site Specific Guideline (CCME 2003). The water being discharged from the backwash treatment process shall meet the following criteria:

- Total suspended solids (TSS) must be below 25 mg/L
- Chlorine residual must be below 0.002 mg/L
- pH must be between 6.5 and 9
- Total aluminum must be below 184 μg/L
- Discharge must be nontoxic to aquatic life

These regulations are a slight variation of the Nova Scotia Treatment Standards for Municipal Drinking Water Systems. The site-specific aluminum regulation was used because the background concentration of aluminum in LPL was higher than the Canadian Water Quality Guidelines as mentioned in Chapter 1. The average aluminum levels in LPL from August 2008 and December 2011 were used to set the discharge criteria at 184 μg/L of total aluminum.

The pre-filter chlorine was no longer added as of March 2013. Therefore, there is no longer chlorine being discharged with the FBW. Nova Scotia Treatment Standards for Municipal Drinking Water Systems sets these standards at 'end of pipe limits', for 95% of the samples taken at least monthly. However, for compliance purposes at JDKWTP, the natural wetlands downstream of the discharge pipe have been included in the treatment process and water quality is monitored at the discharge from the wetlands.

3.4 Retention Time Distribution Analysis

Retention time distribution (RTD) curves are probability distribution functions for residence times in a tank (Kadlec 1994). In this study RTD curves were developed for the natural wetland, engineered lagoons and CFD modeled lagoons. Typical tracer RTD curves can be analyzed based on the moments which define key parameters including mean residence time ($\bar{t}$), the equivalent number of completely stirred tank reactors (N_{CSTR}), equivalent tank volume, and variance (σ^2) (Kadlec 1994).

The mean residence, was calculated with the following equations respectively (AWWA 1996; Levenspiel 1999):

$$\bar{t} = \frac{\sum_{i=1}^{i=n} t_i c_{p,i} dt}{\sum_{i=1}^{i=n} c_{p,i} dt} \qquad [4.3]$$

Where:

t is the time at which sample i was taken

c is the concentration of the tracer

n is the number of samples

dt is the size of the time step

Calculating such mean retention time allows for comparison between tracer studies.

3.5 Analytical Methods for the Water Samples

Water samples were analyzed using analytical techniques for various parameters. All metals were analyzed using an X-Series 2 Inductively Coupled Mass Spectrometry (Thermo Fisher Scientific, Beverly, Massachusetts, United States). Dissolved metals and organic carbon samples were run through a preconditioned 0.45 µm filter. Organic carbon was analyzed using a Shimadzu ASI-V TOC analyzer (Kyoto, Japan). The method detection limits for these methods are located in Table 3.1.

The pH of the samples was measured using an Accumet Excel XL60 dual channel probe (Fisher Scientific, MA, USA). The pH probe was calibrated daily with a three point calibration pH 4, 7, and 10.

The total suspended solids (TSS) represent the mass of the solids remaining on 1.5µm pore size Whatman™ 934-AH™glass fiber filter (Maidstone, UK). A known volume of sample was passed through filters that were baked at 105°C and weighed. Once the sample was filtered the samples were then baked and weighed again.

TABLE 3.1 METHOD DETECTION LIMITS FOR THE WATER QUALITY PARAMETERS

Element	Detection Limit (µg/L)
Magnesium	10
Aluminum	4
Phosphorus	10
Potassium	10
Calcium	10
Chromium	0.4
Manganese	0.8
Iron	7
Nickel	0.4
Copper	0.7
Zinc	0.6
Arsenic	0.4
Lead	0.4
TOC	0.6

3.6 Data Analysis

The error bars throughout the text represent one standard deviation about the mean. The data was analyzed to determine if there was a statistically significant difference with a 95% confidence interval using Prism 6.0c (Graphpad Software Inc., San Diego, CA).

Chapter 4: SITE CHARACTERIZATION AND ANALYSIS OF EXISTING WASTE TREATMENT AT THE JDKWTP

4.1 INTRODUCTION

Wetlands and lagoons can provide a high degree of pollutant removal through natural methods. Where land area is not a restriction, treatment wetlands can provide a viable low cost and energy efficient water treatment option. Pollutants are removed by wetlands through various processes including settlement of suspended particles, uptake by vegetation and uptake by microorganisms (Kadlec and Wallace 1996). As noted by the literature in Chapter 2, these treatment mechanisms in wetlands can also provide substantial aluminum removal. The natural wetlands provide treatment through plant uptake of dissolved aluminum. Plant root networks have been shown to also provide treatment through capture of particulate metals, which are then sloughed to the sediments (Tanner and Headley 2011; Booth *et al.,* 1988; Wood *et al.,* 1990; Driscoll 1985). Roots can also contain significant bacterial growth for the uptake of dissolved metals (Borne *et al.,* 2013). Lagoons, in contrast treat the FBW by providing quiescent flow conditions ideal for settling of particulate aluminum (Shilton and Harrison 2003).

The wetland at the JDKWTP has previously gone unmonitored aside from a weekly effluent water sample. Water quality sampling can determine changes in water quality as it moves through the different stages of treatment. Knowing how the water quality changes as it moves through the treatment system could provide insight into a possible retrofit solution to the inadequate aluminum removal occurring at the JDKWTP FBW treatment site.

The removal efficiency in wetlands and lagoons is often largely determined through the hydraulics of the system and contact time (Kadlec and Wallace 1996). A hydraulic characterization of the treatment area will determine retention times. Tracer studies are the most common way of determining hydraulics of lagoons and wetlands (Mangleson and Watters 1972; Kadlec and Wallace 1996; Kadlec 1994; NRC 2004).

In this chapter the JDKWTP wetlands and engineered lagoons are analyzed to determine treatment efficiency of the FBW. The hydraulics of the system are also characterized to determine the flow patterns and retention times that are providing the treatment. Tracer studies and a hydrograph are used for the hydraulic analysis.

4.2 METHODOLOGY

The current treatment of the FBW occurring at JDKWTP was characterized through flow monitoring, tracer studies and water sampling. This data was collected to provide insights into possible causes of the shortcomings of treatment of the FBW.

4.2.1 HYDRAULIC CHARACTERIZATION

The volumetric flow rate of water being discharged from the lagoon and the wetland was determined with the use of a HOBO® U20 water level loggers (Onset® Computer Corporation, Bourne, Massachusetts, United States) and a 625DF2N digital pygmy meter (Gurley Precision Instruments, Troy, New York, United States). Two data level loggers were used; one at the lagoon discharge, and the other at the wetland discharge. The third data logger was used to measure air pressure. All three loggers were logging temperature, time and pressure every ten minutes, which was used to calculate water depth. The following equations were used to convert pressure and temperature to water depths (Maidment *et al.,* 1993):

$$\rho_{H_2O} = 1000 * \frac{((1-(T_{H_2O}+288.9414))}{(508{,}929.2*(T_{H_2O}+68.1296))} * (T_{H_2O} - 3.986)^2 \quad [4.1]$$

$$d_{H_2O} = \frac{1000*(P_{H_2O}-P_{air})}{9.81*\rho_{H_2O}} \quad [4.2]$$

Where:

ρ_{H_2O} is the density of water in kg/m^3;

T_{H_2O} is the temperature of the water in °C;

d_{H_2O} is the depth of the water in meters; and

P_{H_2O} and P_{air} are the pressure of air and temperature respectively.

A pygmy meter was used to determine velocity and calculate the flow rate using the cross sectional area of the flow channel. A rating curve for the inlet and outlet of the wetlands was created using the flow rates at a range of depths (Appendix A). These rating curves were applied to determine flow discharges.

4.2.2 TRACER STUDIES

Tracer studies were conducted by injecting a dissolved inert measureable substance into the inlet of a reactor and measuring it at the outlet (Baawain *et al.,* 2006). Rhodamine water tracer (RWT) ($C_{29}H_{29}N_2O_5Na_2Cl$) a pink fluorescent dye 20% w/w (Keystone Aniline Corporation, Inman, South Carolina, United States) was used during this thesis. A pulse input tracer study was used for both the lagoon and wetland tracer studies to determine the retention time of the system. The RWT was measured using the YSI 6130 RWT sensor on the YSI 6920 mutli-parameter sonde (YSI., Yellow Springs, Ohio, States). The sensor was set to take readings of RWT every 10 minutes. A two point, (0 and 100 µg/L of RWT) true dye calibration was performed according to the YSI 6-Series Multiparameter Water Quality Sondes User Manual (YSI Inc. 2009). The multi-parameter sonde was put into place before the study began so that a backwash with no rhodamine present could be measured. Density stratification of the RWT, which would distort the RTD, was prevented by using a prolonged injection period and turbulent waters (Headley and Kadlec 2007). The mass of the tracer injected was calculated based on an estimated goal effluent concentration of less than 100 µg/L.

The wetland tracer study used 200 mL of RWT, diluted into 40 L of lagoon effluent discharged over 30 minutes where the lagoons discharge into the wetlands. The RWT concentration was measured where the wetlands discharge into LPL at the compliance point. The study was conducted in-sync with a three-filter backwash that produced a volume of 2,000 m^3 of backwash water.

A lagoon tracer study was performed in July 2013. The lagoons had last been pumped of alum sludge 10 months prior and are usually pumped annually. The study was performed with 500-mL of RWT diluted in 23-L of water from the third flocculation tank and injected over 30 minutes. FBW was not used to dilute the RWT because it is difficult to obtain; instead water from the third flocculation tank was used as it is filter influent. A two-filter backwash of 1,400 m^3 was used for this study. The diluted RWT was injected at the gatehouse from a manhole that provides access to the influent pipe of the south lagoon. The RWT concentration was measured at the lagoon effluent with samples every minute to capture the short retention time. The tracer studies were both described in terms of time from the backwash pumps turning on as a consistent equalization method.

4.2.3 Choice in Tracer Chemical

An ideal tracer will not interact with the environment, will be easily measured and will have a low background concentration (Templeton *et al.,* 2006). Many different chemicals could be used depending on the specifics of the situation in which they are being used. Salts, fluorescent dyes and active isotopes are the commonly used tracer types. RWT was chosen because it is easily measured, has low natural background levels, low adsorption and low degradation rates when being used in systems with a HRT of less than one week (Holland *et al.,* 2004). Another concern is the potential toxicity of RWT, which has also been found to be a genotoxin (Behrens *et al.,* 2001). Approval to use the RWT was given by NSE before conducting the tracer studies by C. Curley (personal communication, October 2012).

4.2.4 Water Sampling

To quantify the water treatment efficiency occurring across the system samples were analyzed for metals, solids, pH and organics. The samples were taken in 1-L bottles. The low flow conditions were regularly used because high flow regularly occurs at night with the nightly backwash.

4.3 RESULTS

4.3.1 HYDRAULIC CHARACTERIZATION

4.3.1a DISCHARGE PATTERNS

The inflow and outflow of the wetlands monitoring displayed very distinct surges in flow with every backwash (Figure 4.1). With each nightly backwash, there is a 95% increase in the flow into the wetland from normal base flow of about 0.3 L/s to a maximum of 10 L/s which quickly begins to taper off within ten minutes. The flow rates may actually be larger than calculated as the flow monitoring failed to capture the fastest velocity discharging from the lagoons. This suggests that the lagoons are not providing flow equalization before discharging into the wetlands.

The wetland effluent flow sees a similar diurnal flow; however, the surge has been dampened out greatly. The flow increases by 20% from the average minimum of 75 L/s to 100 L/s. There is a three to five hour delay between the peak discharge into the wetlands and the peak discharge out of the wetlands. The drastic difference in flow volumes and the relatively short delay in between the peak flows at the inlet and outlet would suggest that any removal occurring may be from of dilution.

Staff at the JDKWTP collects rainfall data twice daily. The effects of rainfall on the flow rates were looked at using this data. Hurricane Sandy, which hit the Halifax area around October 30th 2012, is depicted to the right of the dotted vertical line on Figure 4.1. Despite an increase in precipitation during Hurricane Sandy, the peak flows into the wetlands were not affected and the base flow had a slight increase from 0.3 L/s to about 1.0 L/s. However, the wetland discharge was affected by the large amount precipitation and the flow increased to approximately 200 L/s.

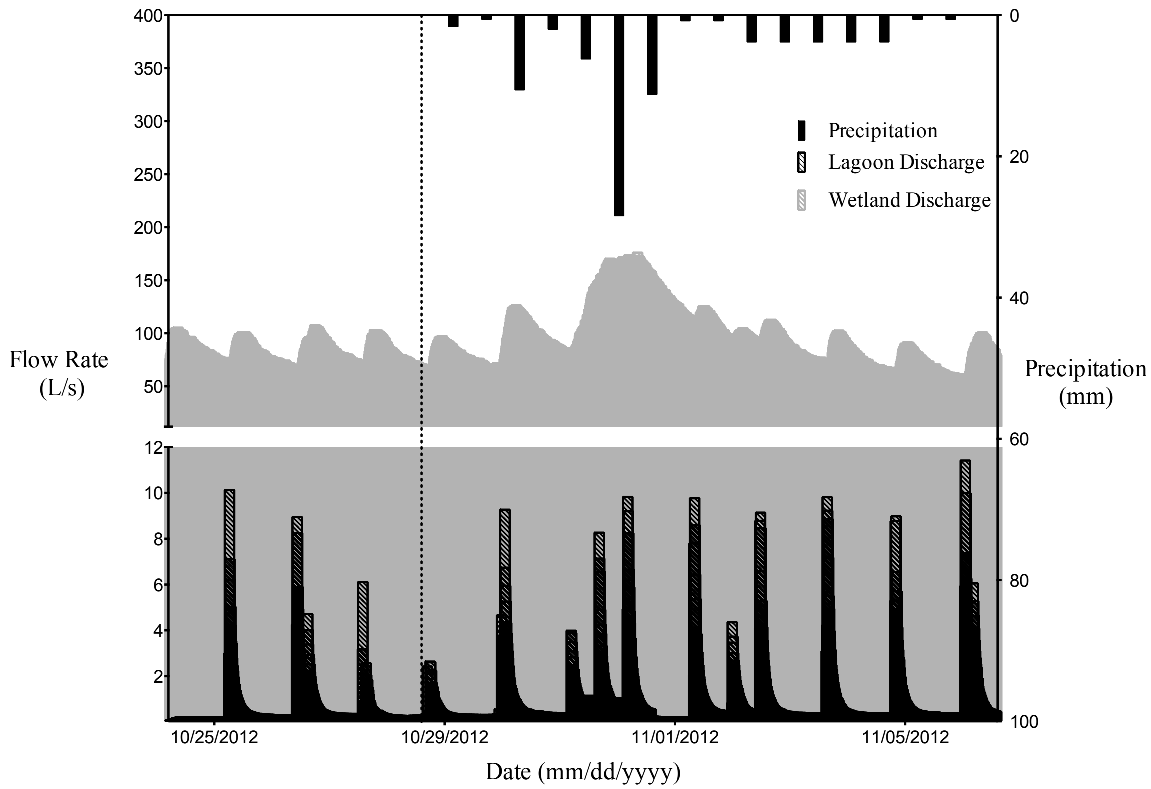

FIGURE 4.1 HYDROGRAPH FOR THE NATURAL WETLANDS. THE RIGHT SIDE OF THE DOTTED LINE REPRESENTS NORMAL FLOWS AND THE LEFT SIDE REPRESENTS HIGH FLOWS DURING HURRICANE SANDY.

4.3.1b TRACER STUDIES

Tracer studies can be used to determine the internal hydraulics of a wetland such as the preferential flow path, dead zones and retention times. The retention time of the wetlands has a significant influence on the pollutant removal efficiency of wetlands (Kadlec and Wallace 1996). The two tracer studies conducted through the FBW treatment area at the JDKWTP are analyzed to determine the retention time of the system.

LAGOON TRACER STUDY

The RWT study from the lagoons displayed the outflow pattern of a two-filter backwash (1,400 m^3). The RTD curve for the tracer study is displayed in Figure 4.2. It took 25 minutes from the time the backwash pumps were on until the RWT was first detected at the lagoon discharge point into the wetlands. The FBW covers approximately a kilometer in these 25 minutes. The pink colour of the RWT could be seen at the overflow weir of the outlet (Figure 4.3) within 25 minutes of the backwash pumps turning on. The FBW continues to discharge for 6.5 hours and about 65% of the tracer is discharged in this time. The mean retention time of the first backwash was calculated at 3 hours and 50 minutes as depicted by the dotted lines on Figure 4.2. About nine hours later, the JDKWTP staff backwashed two more filters; this is depicted by the second surge of RWT.

Both surges in RWT are characterized by an initial peak, which tapers off followed by a final peak. This could be caused by the unsteady nature of the backwash cycle. The water held in the backwash troughs is first released and discharges into the lagoons with a much higher velocity than the rest of the backwash water which comes up through the filter media.

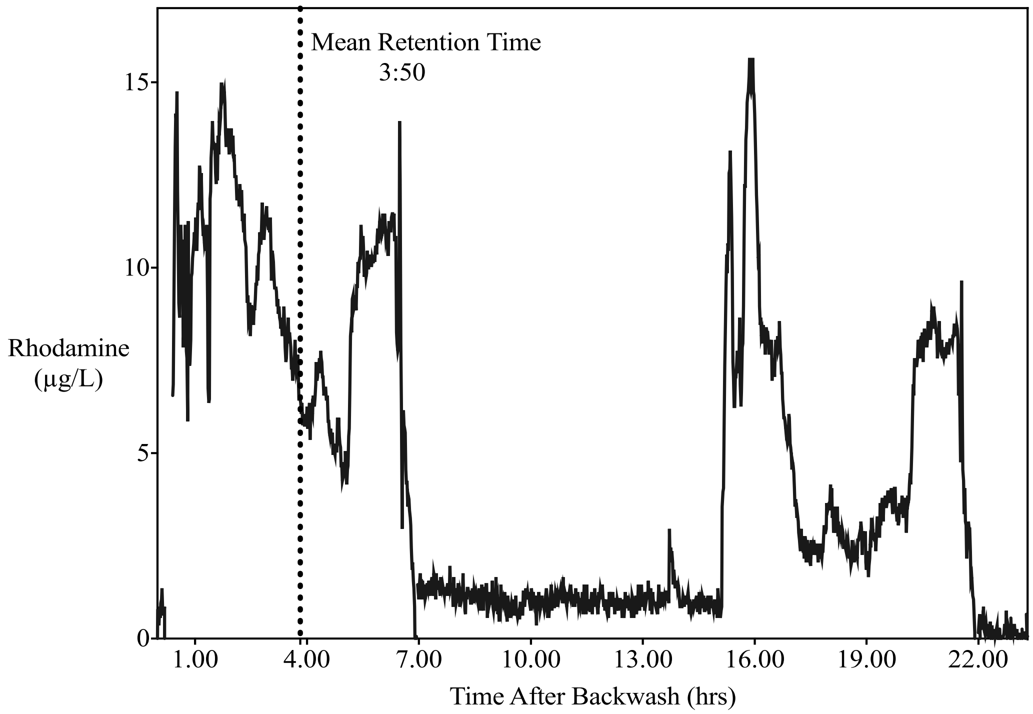

FIGURE 4.2 TRACER RESPONSE CURVE FOR THE SOUTH LAGOON RWT STUDY DISCHARGED FROM THE SOUTH LAGOON IN SYNC WITH A TWO-FILTER BACKWASH.

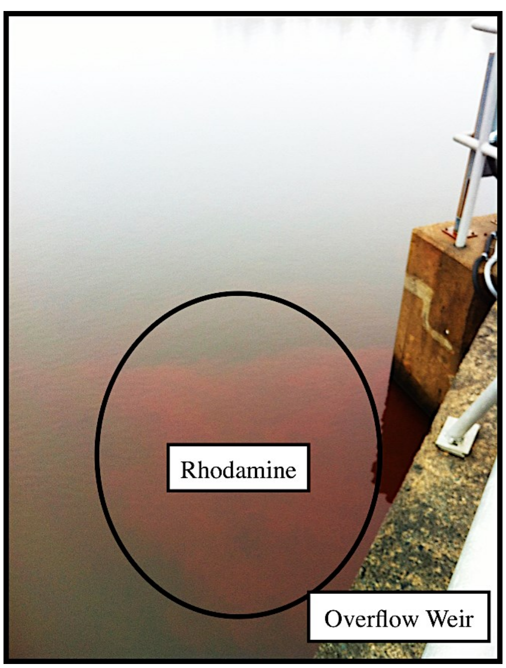

FIGURE 4.3 THE PINK OF THE RWT AT THE OVERFLOW WEIR 25-MINUTES AFTER THE BACKWASH PUMPS TURNED ON.

Throughout the lagoons, there was clear short-circuiting around the perimeter of the lagoon (Figure 4.4). The short-circuiting around the perimeters of the lagoon has been found in previous studies of WSP using a vertical inlet and no method of dissipating the velocity (Shilton and Harrison 2003).

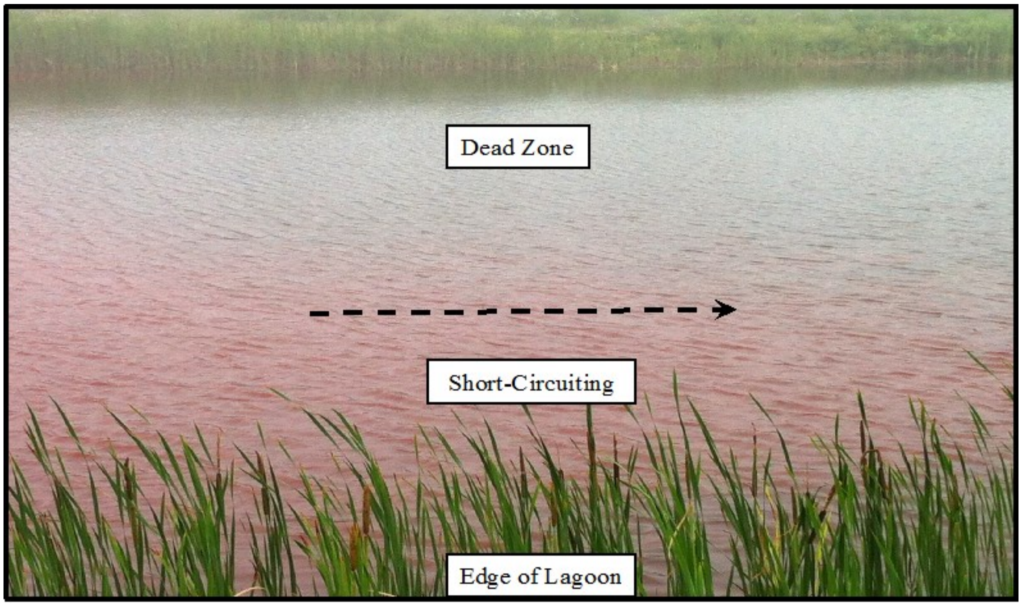

FIGURE 4.4 RWT SHORT CIRCUITING AROUND THE PERIMETER OF THE LAGOON. THE PHOTO WAS TAKEN FROM THE BERM RUNNING BETWEEN THE TWO-ENGINEERED LAGOONS.

WETLAND TRACER STUDY

The RWT movement through the natural wetlands highlighted the very direct route from the lagoon discharge through the center of Hamilton Pond and discharging at the compliance point at LPL. The skewed bell shaped curve depicted in the tracer response data of Figure 4.5 is the most typical tracer response curve (Kadlec and Wallace 1996). The RWT was first detected 5 hours and 24 minutes after the backwash pumps were turned on with peak in concentration at 6 hours and 48 minutes. The mean retention time of the wetlands was calculated to be 12 hours and 51 minutes. The relatively short retention time can be attributed to sudden surge of FBW and the very distinct channel running directly to LPL that has developed as the JDKWTP has been discharging the

FBW in this manner since 1977. The lagoon discharge is not being contained for a period in the wetlands or in Hamilton Pond as previously assumed.

Although tracer response curves of different wetlands cannot be directly compared, Kadlec and Wallace (2005) compiled 34 tracer studies conducted in FWS wetlands. The shortest mean retention time of the 34 wetlands was calculated to be 16.8 hours. The main difference between the JDKWTP wetland and the wetlands looked at by Kadlec and Wallace (2005) is that they are for the most part designed as opposed to the natural wetlands at the JDKWTP.

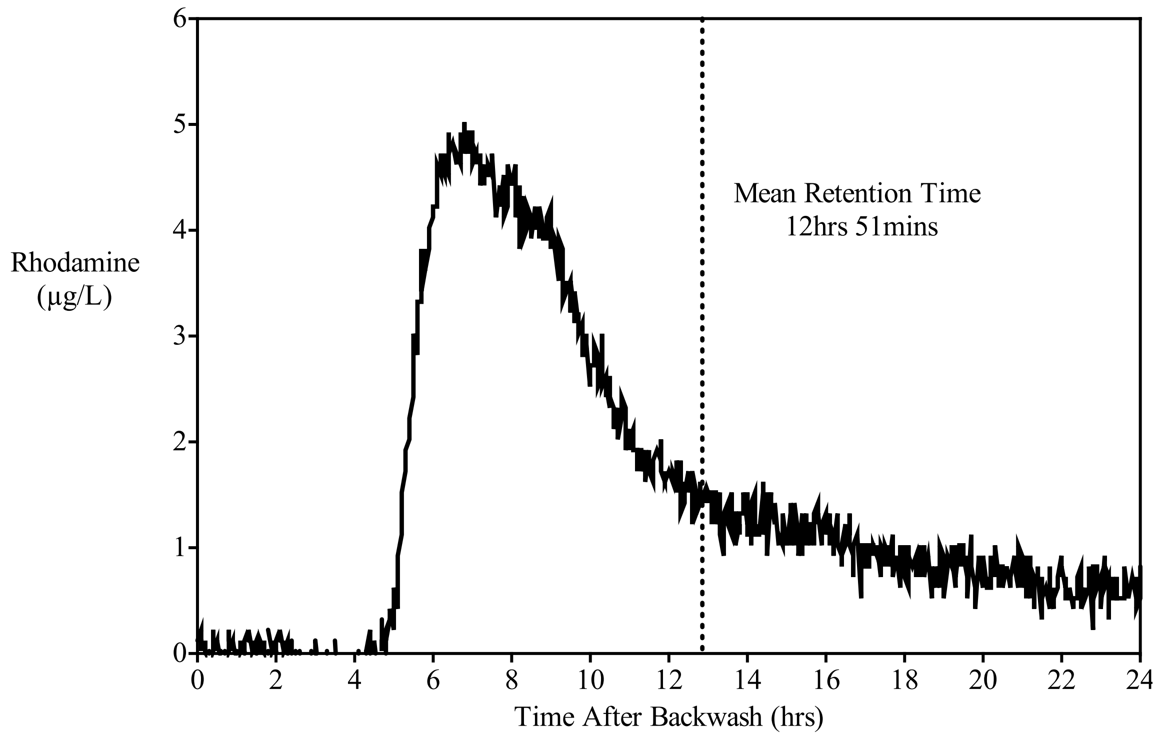

FIGURE 4.5 RWT RESPONSE CURVE FOR THE WETLANDS DURING A THREE-FILTER BACKWASH AT THE WETLAND DISCHARGE POINT IN LITTLE POCKWOCK LAKE.

4.3.2 Water Quality

Water was sampled at three points along the FBW treatment site. The three sample sites include, the FBW flowing into the lagoons, the lagoon discharge and the wetland discharge for various metals, pH, organic carbon and TSS. The lagoon discharge and wetland discharge were sampled during low flow conditions to represent a more typical discharge. This sampling was conducted to quantify the treatment occurring throughout the FBW treatment site.

The change in water quality across the two-step treatment process is documented in Table 4.1. The samples in Table 4.1 represent samples taken over the period between 2012 and 2013 during regular low flow conditions. Overall the concentrations of the various metals are highly variable as denoted by the high standard deviations in Table 4.1. This could be a sign of seasonal changes in water quality or that sample timing was a larger determinate in water quality than originally thought.

At the discharge into LPL, the copper concentrations were found to be in violation of the CWQG with an average concentration of 4.80 ± 3.80 μg/L compared to the guideline of 2 μg/L. The source of the copper may be the JDKWTP's plumbing as copper is often released into water because of corrosion of pipes (MWH 2005). The pH of the water discharging into LPL fell just below the lower limit of the CWQG of 6.5. However, the pH of the water in this area is typically low, Pockwock Lake water averages at a pH of 5.6 (Halifax Water 2013). Therefore, the pH guideline may not be the most suitable for this area.

The engineered lagoons demonstrated adequate lead and zinc removal. The FBW contained lead and zinc levels above the CWQG (1 μg/L and 30 μg/L respectively), and the lagoon discharge fell below the guidelines (ND and 13.9±36.1 μg/L respectively). The wetlands combined with the engineering lagoons showed sufficient iron removal to below the CWQG (300 μg/L) from 634±273 μg/L to 282±76.0 μg/L.

It was noted that on average the dissolved aluminum concentration does not change within the lagoon; this is reasonable as the lagoons generally provide settlement of particulate matter. However, the dissolved aluminum increases throughout the wetlands, which is counter to the customary treatment method in a natural wetland that involves plant uptake of dissolved constituents. It is also inconsistent with the pH of ~6.4 within the wetland, which is the range at which aluminum usually precipitates in water. This increase in dissolved aluminum in the wetland may be caused by anaerobic conditions. During sampling odors indicative of anaerobic digestion were noted however, more studies would be required to confirm this hypothesis. Previous studies have shown significant metals removal though anaerobic digestion; these conditions also increased the pH of the water (Eger 1994; Shilton 2005).

Despite the increase in dissolved aluminum within the wetland the levels remain relatively low. This is a good indicator that the aluminum concentrations may not be toxic to aquatic life as it is the dissolved aluminum that biologically available (Gundersen *et al.,* 1994). However, if the increasing trend continues downstream there may be a cause for concern. As described in Chapter 2, other water quality parameters often determine aluminum toxicity rather than the aluminum concentration. The calcium, DOC and pH discharging at LPL are relatively high (Table 4.1). These conditions in previous studies mitigated the toxic effects of the high aluminum waters (Gensemer and Playle 1999; Ingersoll *et al.,* 1990; Parent and Campbell 1994; Gensemer 2000; Pettersson *et al.,* 1988).

If the CWQG were being applied to the aluminum in this situation using the water quality discharging from the wetlands (Table 4.1), the calcium, 6.5 mg/L, and DOC, 5.0 mg/L, would put the limit at 100 μg/L. However, the pH, 6.3, would put it at the 5 μg/L aluminum limit. Regardless, assuming the regulations are based on dissolved aluminum the 100 μg/L is being exceeded at the LPL discharge at 108 μg/L. Conversely, at the lagoon discharge (the designed end of the treatment system) the dissolved aluminum falls under the upper, 100 μg/L limit at 45 μg/L, along with the calcium at 13.8 mg/L. However, the other water quality parameters fall within the 5 μg/L limit with DOC at 4.8 mg/L just under the 5 mg/L limit and the pH at 6.4 just under the 6.5 limit, thus the ambiguity in the guideline.

TABLE 4.1 WATER QUALITY THROUGHOUT THE FBW TREATMENT DURING REGULAR, LOW FLOW CONDITION TAKE DURING 2012 TO 2013 (AVERAGE ± STANDARD DEVIATION) (NON-DETECT, ND) (N ≥ 6, UNLESS ND N ≥ 2).

Water Quality Parameter (µg/L)	CWQG	FBW	Lagoon Discharge	Wetland Discharge
Aluminum	5	48,500±23,500	1,550±1,120	669±471
Dissolved Aluminum	NA	45.0±21.0	43.0±23.0	108±93.0
Arsenic	5	2.80±1.90	0.80±0.50	ND
Dissolved Arsenic	NA	ND	ND	ND
Calcium	NA	5,180±692	13,800±16,700	6,560±2,610
Dissolved Calcium	NA	4,280±475	10,100±2,500	5,770±1,596
Chromium	1	NA	0.7	0.60
Copper	2	9.10	4.80±4.40	4.80± 3.80
Dissolved Copper	NA	8.10±4.70	8.20±8.00	11.6±9.60
Iron	300	3,420± 3,420	634±273	282±76.0
Dissolved Iron	NA	ND	132.0±159.0	37.9±30.5
Lead	1	1.20±1.20	ND	ND
Dissolved Lead	NA	ND	ND	ND
Magnesium	NA	670±96.6	1,060±340	711±272
Dissolved Magnesium	NA	502±45.3	1,010±345	603±184
Manganese	NA	5,140±2,550	2,160±996	288±153
Dissolved Manganese	NA	410±228	2,170±981	213±124
Phosphorus	NA	107.5	293±373	99.6±104
Potassium	NA	635±165	915±331	1,010±746
Dissolved Potassium	NA	550±114	1,040±475	833±354
Zinc	30	39.0±18.0	13.9±36.1	12.9±20.7
Dissolved Zinc	NA	17.7±17.7	7.50±3.60	9.90±9.20
pH	6.5-9.0	5.50±0.20	6.40±0.20	6.30±0.30
TSS (mg/L)	NA	255±190	7.90±6.50	2.70±2.30
TOC (mg/L)	NA	94.5±48.6	6.10±1.80	5.80±2.20
DOC (mg/L)	NA	2.80±0.30	4.80±1.00	5.00±1.80

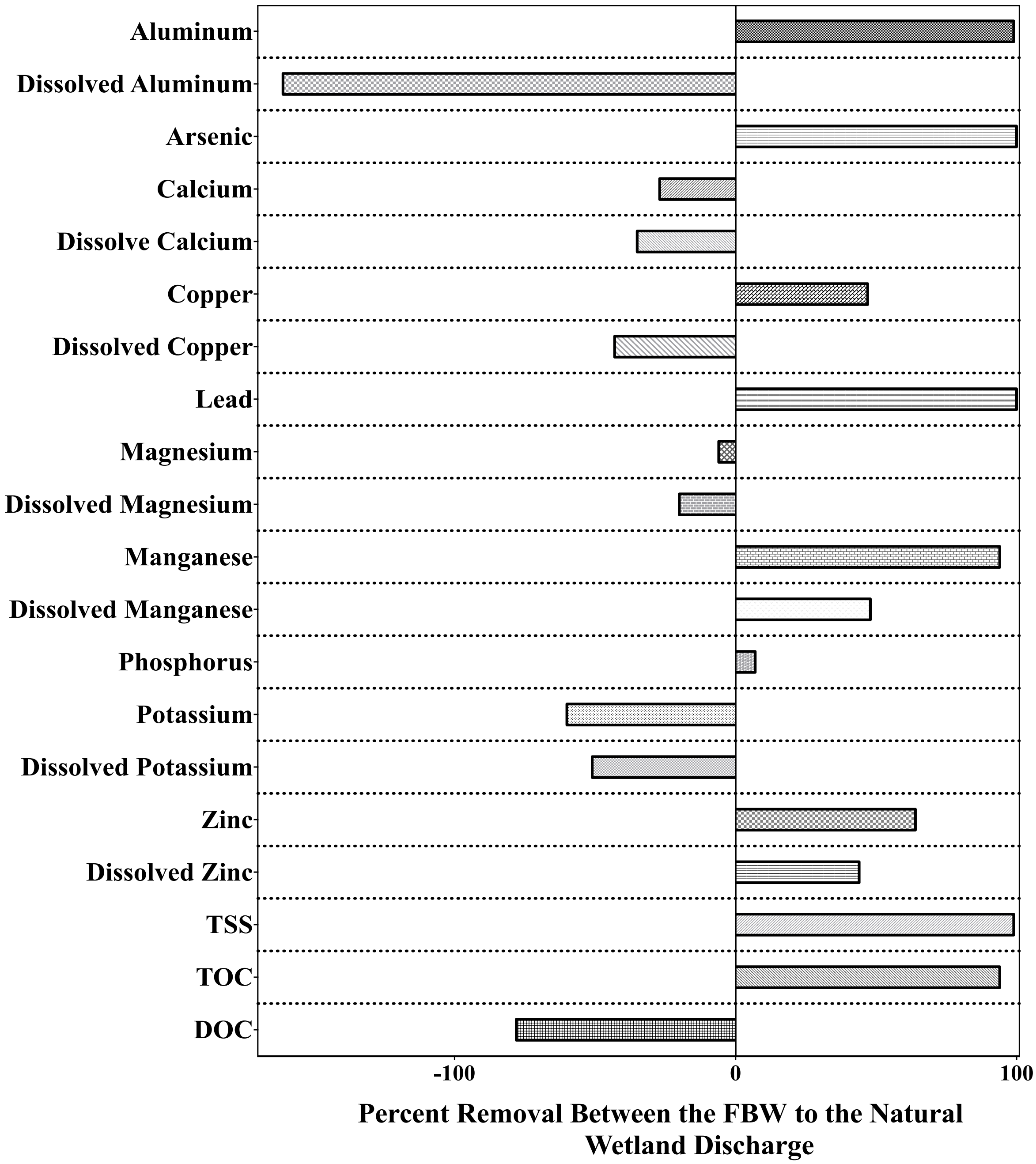

FIGURE 4.6 PERCENT REMOVALS ACROSS THE FBW TREATMENT SYSTEM, FROM THE FBW DISCHARGE INTO THE LAGOONS TO THE WETLAND DISCHARGE INTO LPL

The percent change in water quality from the start of the FBW treatment process to the discharge into LPL was calculated and documented in Figure 4.6. Most of the water quality constituents showed high removal throughout the system. TSS, total aluminum, total arsenic, total lead, and total manganese had close to 100% removal. The aluminum removal is consistent with literature, Kaggwa *et al.,* 2001 looked at a wetland for FBW treatment that was achieving 97% removal of aluminum. Kadlec and Wallace (2009) compiled a list of 25 wetlands used for the treatment of high aluminum waters and found a range from -16% to 100% removal aluminum. The dissolved constituents in comparison in some cases including aluminum, copper, organic carbon, calcium and potassium increased throughout the treatment system.

4.3.2a ALUMINUM TRENDS

Aluminum trends were analyzed more in depth, as aluminum was the original water quality parameter of concern. The aluminum concentrations discharging from the lagoons and natural wetlands were analyzed against the three flow conditions described in section 4.3.1. The timing of the sampling of the high flows versus low flows was determined based on the tracer studies described also in section 4.3.1. The high flow from the lagoons was sampled 25 minutes after backwash and high flow samples from the wetland were sampled five hours and 24 minutes after the backwash. The changes in aluminum concentrations with changes in flow rates suggest that aluminum concentrations may be time dependent, based on the time of the last backwash and amount of rainfall. Figure 4.7 showed that the highest aluminum concentrations out of the lagoons occurred during a backwash and a rain event. The rain stirring up the settled debris in the lagoons may cause the elevation in aluminum concentration. However, at the wetland discharge, the aluminum concentration during a backwash and rain event was actually lower than during a backwash without rainfall. This may be caused by the size of the watershed (1 km^2) discharging at the compliance point creating a dilution factor. However, regardless of the flow conditions during sampling, the aluminum concentrations were always in excess of the 184 µg/L regulation.

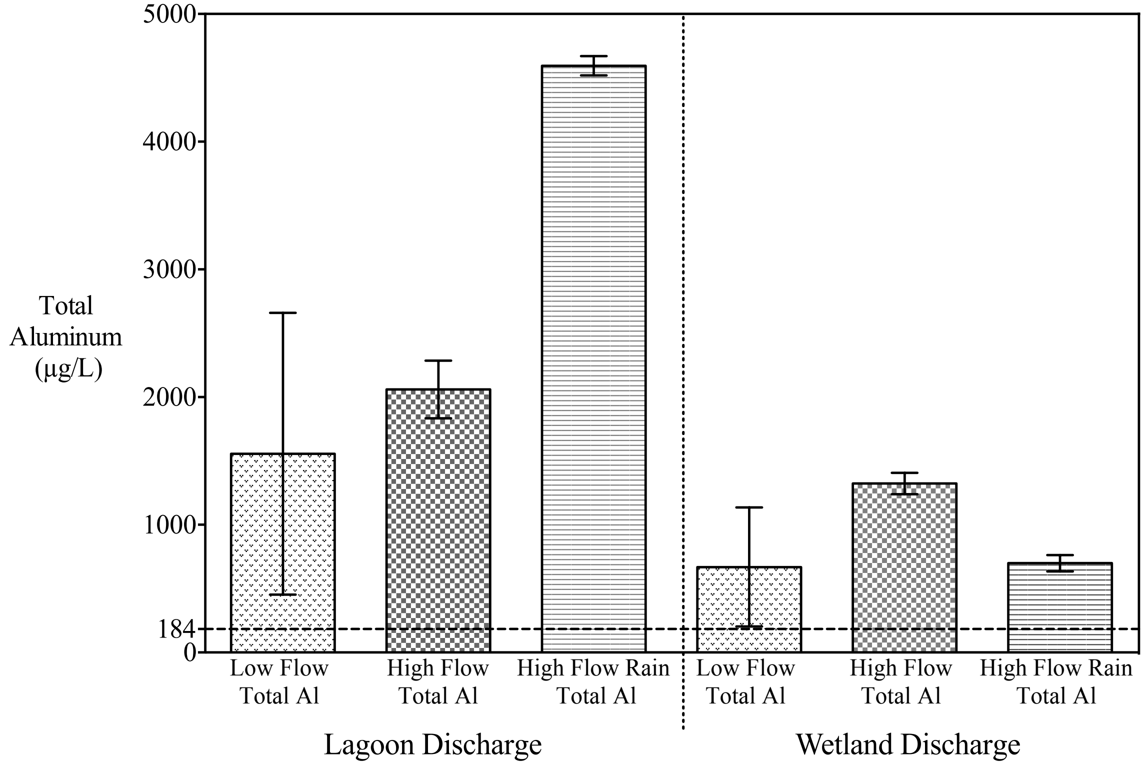

FIGURE 4.7 ALUMINUM CONCENTRATIONS AT VARIOUS FLOW CONDITIONS AT THE LAGOONS DISCHARGE INTO THE WETLAND AND THE WETLAND DISCHARGE INTO LITTLE POCKWOCK LAKE.

Over the course of a year, weekly sample results from the compliance point at LPL are depicted in Figure 4.8. The samples were taken around 11 am and are representative of low flow sampling. Of the 48 samples taken, no samples had aluminum measurements below the 184 µg/L site-specific regulation. At the JDKWTP, raw and finished water aluminum concentrations are documented; trends show increasing aluminum through the winter months to a peak in March and decreases until August (Appendix C). This trend was not emulated in the compliance point samples, which may be caused by additional factors affecting aluminum discharge including timing of lagoon pumping. The lagoons were pumped in October of 2012, at which time the procedure may stir up settled solids and discharge from the lagoon. This is depicted in Figure 4.8 by a sudden spike in aluminum concentration in October.

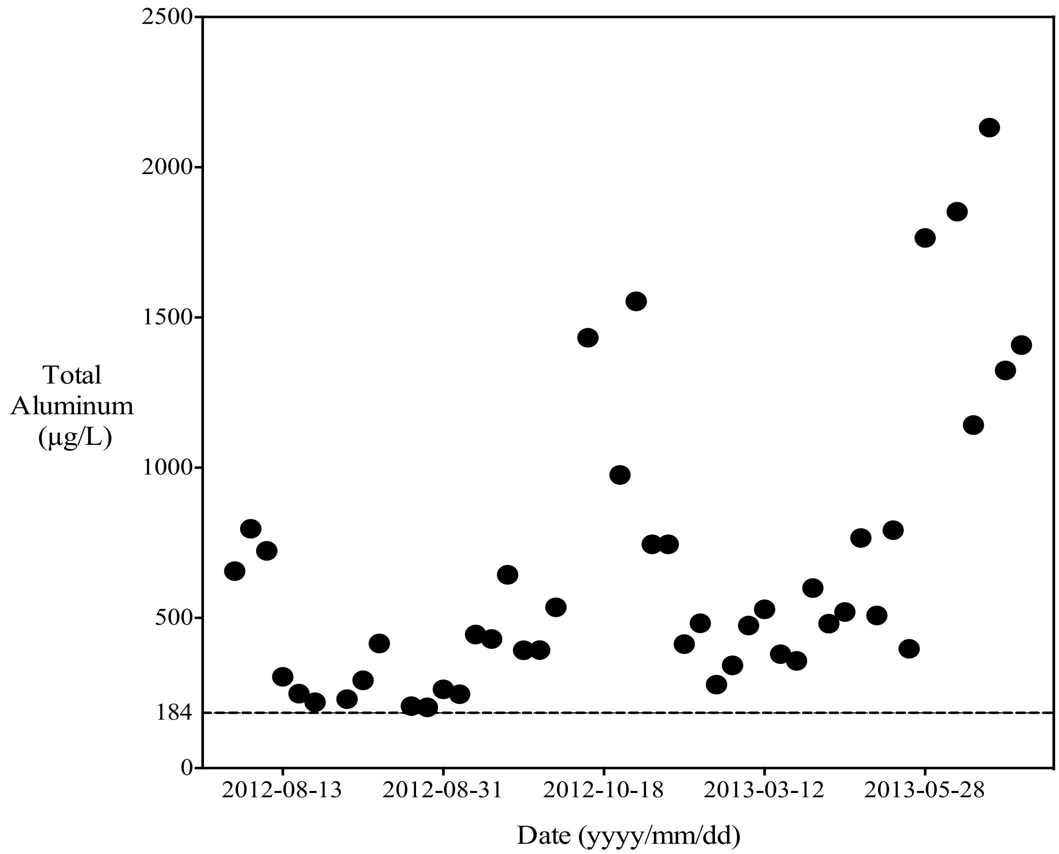

FIGURE 4.8 TIME SERIES OF ALUMINUM CONCENTRATIONS FOR ONE YEAR AT THE WETLAND DISCHARGE INTO LITTLE POCKWOCK LAKE.

4.3.2b FBW Treatment Using Lagoons in Nova Scotia

Similar studies have been conducted at various WTPs in Nova Scotia also due to concerns with failing to meet CWQG with respect to aluminum. The Shelburne, Yarmouth and Glace Bay WTPs also use lagoons to treat their FBW (CBCL 2010; CBCL 2008; CBCL 2006). The results from the JDKWTP lagoon effluent were compared with these three plants' lagoon effluents (Table 4.3) to put the results into context. These plants were specifically looked at because of concern of toxic discharge; therefore, this is a comparison in the worst cases.

Compared to the lagoon effluents from the three other WTPs, the JDKWTP lagoon discharge had comparable aluminum concentrations. The dissolved aluminum concentration at the JDKWTP was slightly lower than the other plants' lagoon discharge. Though TSS is difficult to compare as it was only measured at the Shelburn WTP, the JDKWTP TSS discharge was quite a bit higher than Shelburn. The pH of the water discharging from the JDKWTP is approximately one log lower than the other three plants. This could be caused by differences in lagoon influent water quality.

TABLE 4.2 A COMPARISON WATER QUALITY DISCHARGING FROM LAGOONS USED FOR FBW SETTLING IN NOVA SCOTIA

	JDKWTP	Shelburn	Yarmouth	Glace Bay
Aluminum (µg/L)	1,550	889	4562	445
Dissolved Aluminum (µg/L)	43.0	108	407	47.0
TSS (mg/L)	7.90	3.50	NA	NA
pH	6.40	7.50	7.20	7.20

4.4 DISCUSSION

A hydraulic analysis of the FBW treatment area at the JDKWTP was conducted. The hydrograph developed for the wetland showed a significant peak in flow within minutes of a backwash event at the lagoon discharge increasing in flow rate from 0.3 L/s to 10 L/s. A less extreme peak is noted at the wetland discharge into LPL within a few hours of a backwash increasing from 75 L/s to 100 L/s.

The rhodamine tracer studies deduced that there is significant short-circuiting occurring through the entire FBW treatment system. The tracer study in the lagoons displayed an initial discharge from the lagoons 25 minutes after the backwash pumps turn on. The wetland tracer study exposed similar short-circuiting through the wetland with the tracer first being detected at the wetland discharge five hours and 24 minutes after the backwash pumps were turned on. This results in a minimum treatment time of five hours and 50 minutes. Visually the RWT displayed a very perimeter based flow within the lagoons and within the wetlands a very distinct channel.

Weekly grab samples from the inlet and the outlet of the natural wetland showed that despite the high removal efficiency of the system with regards to aluminum and TSS (99%), the aluminum concentration never met the site-specific guideline of 184 μg/L with an average total aluminum discharge from the wetlands at 669±447 μg/L. The dissolved aluminum was quite a bit lower at 108±93 μg/L but showed an increasing trend. Sampling also showed that aluminum concentrations were time dependent based on the time of the previous backwash. These water quality results were consistent with other FBW treatment lagoons in Nova Scotia.

Chapter 5: Optimization of FBW Treatment: An Application of CFD and Bench-scale Experiments

5.1 Introduction

Alum based filter backwash water (FBW) is typically gelatinous in nature and more specifically in Atlantic Canada it is typically high in organics and lightweight (CBCL 2004; Gruninger 1975). Therefore, FBW can be difficult to dewater and treatment through settling alone does not typically provide sufficient treatment (CBCL 2004; Gruninger 1975). Settling occurring within lagoons can be enhanced by various methods including lengthening the retention time to allow for more time for settling also by chemically enhancing the FBW to create denser flocs, which will settle faster. Retention time can be extended in a pond by physically improving the hydraulics of the system through optimized baffle arrangement, or by changing the inlet or outlet design as described in Chapter 2.

Short-circuiting caused by improper placement of inlets and outlets is the most common cause of poorly performing lagoons (NRC 2004). When an optimized baffle arrangement is used, the baffles can act to dissipate the inlet energy; this will reduce the short-circuiting and dead zones in the pond. Dissipating the inlet energy may act to reduce the peak discharge occurring at the JDKWTP lagoons during a filter backwash (Figure 4.1). Reducing peak flows may reduce the associated peak aluminum concentration occurring during high flow (Figure 4.7). Experiments with pond geometry are commonly done using fluid dynamics models or at a pilot-scale (Shilton and Harrison 2003; Abbas *et al.,* 2006; Vega *et al.,* 2003). In this chapter, computational fluid dynamics (CFD) is employed to model various baffle layouts based on the JDKWTP lagoon's geometry.

Chemically enhancing FBW with the use of a polymer has been shown to significantly increase the settling rate for FBW (CBCL 2004). The optimal polymer type used for sludge conditioning is dependent on sludge properties and bench-scale experiments are required to determine (MWH 2005). In this chapter, four polymer types at five different doses are tested to determine if using a polymer to condition the FBW would improve settling rate in the lagoons.

5.2 Materials and Methodology

5.2.1 CFD Model of the Engineered Lagoons

Numerical simulations of the engineered lagoons were conducted using Ansys Fluent 14.0® (Ansys, Lebanon NH). Ansys Fluent was chosen as it is commonly used for CFD modeling in the water treatment industry (Kennedy *et al.,* 2006; Goula *et al.,* 2008; Kim *et al.,* 2010) and it had previously been used in studies at the JDKWTP (Vadasarukkai *et al.,* 2011; Vadasarukkai and Gagnon 2010). Fluent uses the finite volume method to model flow, turbulence, heat transfer and reactions within user-defined boundary conditions. It works as a part of Ansys' Workbench platform. The Workbench is an easy to use interface that links the Ansys' suit of software used in the CFD process together. To simulate the flow within the engineered lagoons Ansys' DesignModeler® and Meshing applications were used for preprocessing to develop the fluid domain. Fluent 14.0 was used for the processing step for the simulation.

5.2.1a Preprocessing

The DesignModeler and Meshing applications were using during the preprocessing steps of the CFD modeling. DesignModeler was used to create the geometry of the flow domain; it uses a parametric approach to solid modeling. The original construction drawings, provided by the JDKWTP staff, were used to create a three-dimensional representation of the engineered lagoons. A three-dimensional model was chosen as opposed to the more common and computationally efficient 2D option because of the 3D vertical inlet that could not be accurately represent in only 2-dimensions (Wood *et al*., 1998; Salter *et al*., 2000). The sketch tool was used to create a two-dimensional outlet of the lagoons in the XY-plane. Using the sketches as an outline, the three-dimensional lagoon and its details were created with Boolean Operators, the Modify Toolbox and Body Operations.

The geometry was then exported to the Meshing application for discretization. Tetrahedral cell volumes (CV) were used for the entirety of the geometry as they are best suited for irregular geometries (Kennedy *et al.,* 2006). Non-uniform CVs were used to account for the large range of velocities throughout the flow domain. Such that, finer CVs were used in regions with high velocities (pipes, inlets and outlets) to capture the rapid changes in the velocity, and coarser CVs were used throughout the volume of the lagoon where the velocities are much slower and changing less rapidly. The non-uniform CVs were optimized to ensure minimal computational time with maximum accuracy.

The quality of the mesh was evaluated against various mesh metrics including skewness, aspect ratio and element quality. Skewness is a measure from zero to one of how close the cell face is to an equilateral triangle (Ansys Inc. 2010). The skewness of the mesh was used as the principal metric for determining the quality of the mesh. This was because the tetrahedrons have a tendency to skew when over refined by moving away from the equilateral toward to the obtuse triangles. The mesh always had less than 100 cells with a skewness of 0.8 or greater, as greater than 0.8 is considered ‘very skewed’ (Ansys Inc. 2006).

Boundary conditions for the inlet, outlet, interior faces and wall were defined. The inlet was defined as a velocity inlet and set to represent the velocity of a three-filter backwash. The turbulent intensity was estimated using equation 5.1; it typically ranges from 1 to 10% for fully developed pipe flow (Kennedy *et al.,* 2006; Ansys Inc. 2009).

$$I = 0.16Re^{-\frac{1}{8}} \qquad [5.1]$$

The outlet was defined as a pressure outlet with a zero gauge pressure. The walls of the lagoons were defined as a no-slip boundary, which represents a zero velocity at the wall (Ansys Inc. 2010). For the near-wall boundary condition, a standard wall function was used which decreases the turbulent instability as the wall is approached. Finally, an interface boundary condition was used to ensure continuous flow between various portions of the geometry.

5.2.1b PROCESSING

Ansys' Fluent application was used for the CFD modeling. The modeling was completed in an iterative three-step process, which is depicted in Figure 5.1. This process sets the flow field for the discretized domain. The model was run with first order upwind spatial discretization for 2000 iterations and the mesh was refined in areas of the 10% highest velocity. Residuals from velocity in the x, y, and z directions, k, ε, and continuity were monitored for stability and a value below $1x10^{-3}$. The mesh was refined until the difference between the inlet mass flow rate and outlet mass flow rate was less than 0.01 kg/s or the difference started to increase, and the volume weighted average velocity no longer fluctuated more than 0.001 m/s. Once these criterions were met, the model was switched to second order upwind spatial discretization and again run for 2000 iterations and checked against the same criteria. If the model still met the criteria, the model was further complexed by using a transient simulation. The transient simulation incorporates virtual time and this allows for a virtual tracer study to be completed.

TABLE 5.1 DISCRETIZATION SCHEMES APPLIED TO SOLVING THE GOVERNING FLOW EQUATIONS

Flow Equation	Discretization Scheme
Pressure-Velocity Coupling	SIMPLE
Pressure	Standard
Momentum	Second Order Upwind
Turbulent Kinetic Energy	Second Order Upwind
Turbulent Dissipation	Second Order Upwind
Tracer	QUICK

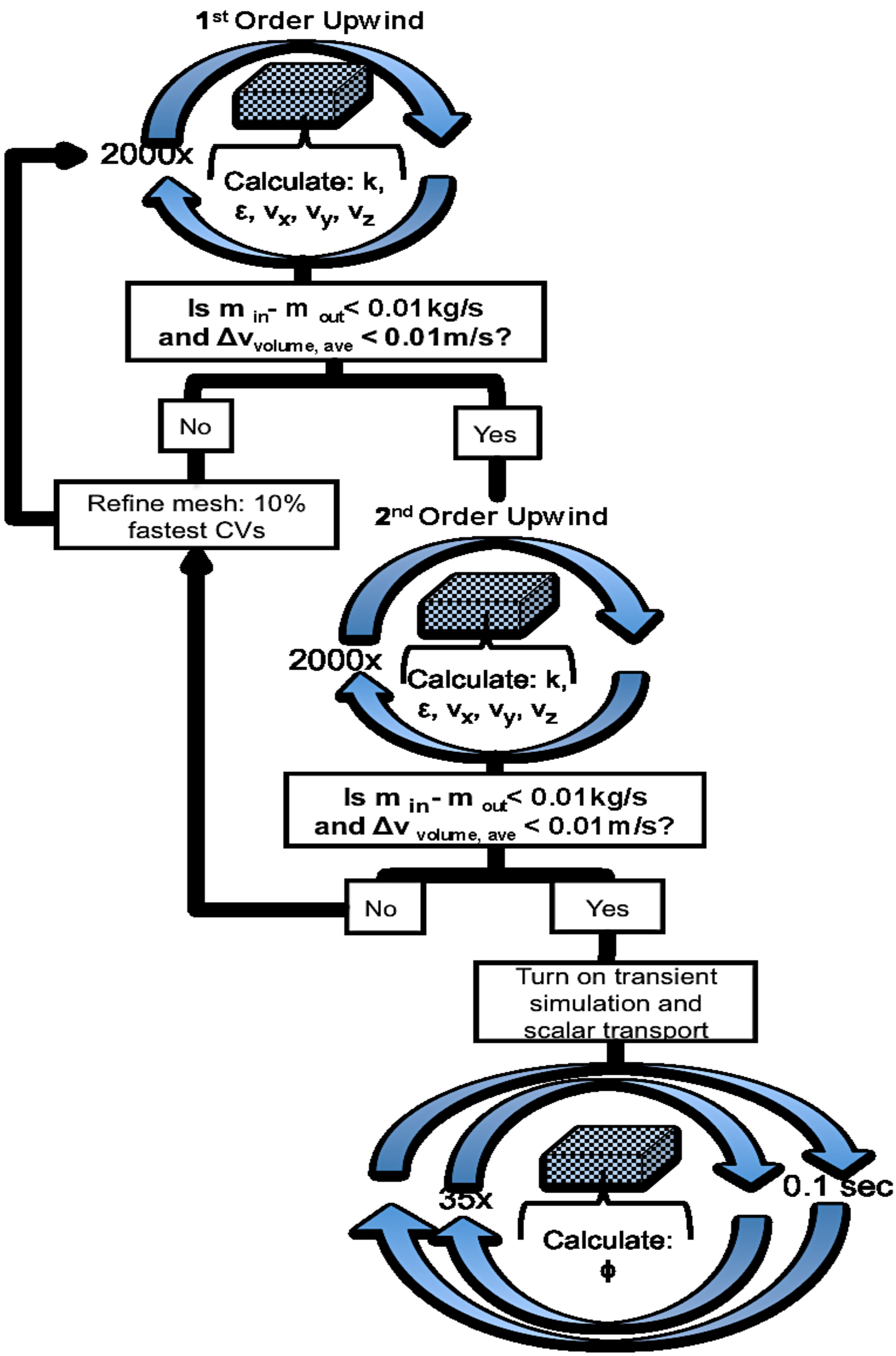

FIGURE 5.1 CFD MODELING METHOD EMPLOYED USING ANSYS FLUENT 14.0®

5.2.1c MODELED TRACER STUDY

Modeled tracer studies were performed to determine the mean retention time of the lagoons with various baffle arrangements. Once the flow field (velocity and turbulence) was established in steady state, a transient simulation was run with a scalar transport equation (Wood *et al.,* 1998; Choi *et al.,* 2004; Zhang 2006). A fluid with the same properties as water was defined with a molecular diffusivity of 1×10^{-10} m^2/s and a mass fraction of one to represent the tracer. The tracer was injected at the inlet face for 30 minutes to represent the approximate length of time the backwash flow lasts for, and monitored at the outlet face of the lagoon. A time step of 0.1 second was used initially and gradually increased to 1 second (so as to optimize computational time and space) as the model stabilized. A maximum of 35 iterations per time step was set with the same convergence criteria as above. To solve the scalar transport equation, the quadratic upwind interpolation for convective kinetics (QUICK) scheme was used. The QUICK scheme was chosen because an upwind scheme is needed to accurately approximate the advection term (Wang and Falconer 1997) and when compared to other interpolation schemes (backward implicit, central implicit, and upwind implicit) QUICK was able to estimate the PDF curve in Falconer *et al.,* (1988) more accurately.

5.2.1d GOVERNING EQUATIONS

There are three main principles employed in CFD models, conservation of mass, conservation of energy, and Newton's second law (F=ma) (Anderson 1995; Versteeg and Malalasekera 2007). The following equations represent the general form applicable to most situations.

$$\frac{\partial \bar{\rho}}{\partial t} + div(\rho \bar{U}) = 0 \qquad [5.2]$$

$$\frac{\partial \bar{\rho}\tilde{U}}{\partial t} + \mathrm{div}(\bar{\rho}\tilde{U}\tilde{\mathbf{U}}) = -\frac{\partial \bar{P}}{\partial x} + div(\mu \; grad \; \tilde{U}) + \left[-\frac{\partial(\overline{\bar{\rho}u'^2})}{\partial x} - \frac{\partial(\overline{\bar{\rho}u'v'})}{\partial y} - \frac{\partial(\overline{\bar{\rho}u'w'})}{\partial z} \right] + S_{M_x} \qquad [5.3]$$

$$\frac{\partial \bar{\rho}\tilde{V}}{\partial t} + \mathrm{div}(\bar{\rho}\tilde{V}\tilde{\mathbf{U}}) = -\frac{\partial \bar{P}}{\partial y} + div(\mu \; grad \; \tilde{V}) + \left[-\frac{\partial(\overline{\bar{\rho}u'v'})}{\partial x} - \frac{\partial(\overline{\bar{\rho}v'^2})}{\partial y} - \frac{\partial(\overline{\bar{\rho}v'w'})}{\partial z} \right] + S_{M_y} \qquad [5.4]$$

$$\frac{\partial \bar{\rho}\widetilde{W}}{\partial t} + \mathrm{div}(\bar{\rho}\widetilde{W}\widetilde{\mathbf{U}}) = -\frac{\partial \bar{P}}{\partial z} + div(\mu\ grad\ \widetilde{W}) + \left[-\frac{\partial(\overline{\rho u'w'})}{\partial x} - \frac{\partial(\overline{\rho v'w'})}{\partial y} - \frac{\partial\left(\overline{\rho w'^2}\right)}{\partial z} \right] + S_{M_z} \quad [5.5]$$

$$\frac{\partial \bar{\rho}\tilde{\phi}}{\partial t} + \mathrm{div}(\bar{\rho}\tilde{\phi}\widetilde{\mathbf{U}}) = div(\Gamma_\phi\ grad\ \tilde{\Phi}) + \left[-\frac{\partial(\overline{\rho u'\phi'})}{\partial x} - \frac{\partial(\overline{\rho v'\phi'})}{\partial y} - \frac{\partial(\overline{\rho w'\phi'})}{\partial z} \right] + S_\phi \quad [5.6]$$

Equation 5.2 represents the conservation of mass for an unsteady compressible fluid with sum of the rate change of density and the net flow of mass out of the CV equaling to zero (Versteeg and Malalasekera 2007). Equations 5.3, 5.4, and 5.5 represent the conservation of momentum or Newton's second law in the x, y, and z directions respectively where the forces applied to a CV are made up of body forces and surfaces forces (pressure, shear stress, normal stress and turbulent) are equal to the acceleration times the mass of the CV (Anderson 1995). Finally, equation 5.6 is the scalar transport equation for any species ϕ. In these equations, the overbar represents terms that are averaged over time and the tilde represents terms that are density weighted.

TURBULENCE MODELING

Turbulence modeling is required for flow regimes exhibiting a high Reynolds's number or for flows with velocity and other flow parameters varying in a chaotic manor (Versteeg and Malalasekera 2007). Reynolds's number is the ratio of inertial forces to viscous forces in a fluid and calculated with the following equation.

$$Re = \frac{QD\rho}{A\mu} \quad [5.7]$$

Turbulent fluids experience Reynolds stresses, which are shear and normal stresses caused by the layers moving at different rates chaotically (Versteeg and Malalasekera 2007). These stresses are represented by additional terms in the conservation of momentum equations ($\tau_{xx} = -\overline{\rho u'^2}, \tau_{yy} = -\overline{\rho v'^2}$, $\tau_{zz} = -\overline{\rho w'^2}$, $\tau_{xy} = \tau_{yx} = -\overline{\rho u'v'}$, $\tau_{xz} = \tau_{zx} = -\overline{\rho u'w'}$ and $\tau_{yz} = \tau_{zy} = -\overline{\rho v'w'}$) (5.3-5.5). When the six terms are added the equations are referred to as Reynolds-Averaged Navier-Stokes (RANS). The terms are averaged over a time scale rather than solving the detailed turbulent fluctuations to reduce computational effort (Versteeg and Malalasekera 2007).

The additional nonlinear terms create a closure problem and require a turbulence model to solve the equations. The standard k- ε turbulence model was used, as it is the most widely used and validated for fluid modeling (Vadasarukkai 2010; Hadiyanto *et al.,* 2013; Zhang 2006; Templeton *et al.,* 2006; Peterson *et al.,* 2000; Ansys Inc. 2011; Versteeg and Malalasekera 2007). It is a popular choice, since it requires a relatively lower computational effort for an accurate calculation (Versteeg and Malalasekera 2007).

Launder and Spalding first proposed the standard k- ε model in 1974 this model solves for eddy viscosity (μ_t) (Equation 5.8) by introducing a two transport equations one to calculate the turbulent kinetic energy (k) (Equation 5.9) and another for the turbulent dissipation energy (ε) (Equation 5.10).

$$\mu_t = \rho C_\mu \frac{k^2}{\varepsilon} \quad [5.8]$$

$$\frac{\partial \rho\varepsilon}{\partial t} + div(\rho\varepsilon \boldsymbol{U}) = div\left[\frac{\mu_t}{\sigma_\varepsilon} grad\ \varepsilon\right] + \frac{\varepsilon}{k}\mu_t C_{1\varepsilon} \cdot S_{ij} - C_{2\varepsilon}\rho\frac{\varepsilon^2}{k} \quad [5.9]$$

$$\frac{\partial \rho k}{\partial t} + div(\rho k \boldsymbol{U}) = div\left[\frac{\mu_t}{\sigma_k} gradk\right] + 2\mu_t S_{ij} \cdot S_{ij} - \rho\varepsilon \quad [5.10]$$

5.2.2 BAFFLE ANALYSIS AND IMPLEMENTATION

Various baffle placements were studied in an attempt to improve the hydraulics of the engineered lagoons at the JDKWTP. All baffles were placed at 70% width as deemed optimal by previous research discussed in Chapter 2. Three baffle arrangements are discussed in this study, one baffle at the outlet, two baffles and three baffles.

5.2.3 BENCH-SCALE SETTLING EXPERIMENT

To experiment with settling trends, a standard six-paddle jar test apparatus (Phipps & Bird, Fisher Scientific) was used. Polymer used to increase settling was prepared 24 hours before use at a 1% solution, using Milli-Q water for the dilution. In a similar settling test developed by Zhao (2004) for alum sludge, it was found that settling results are independent of vessel size. One-litre sample of FBW was used for each of the six jars due to the difficulty of getting the FBW.

In an attempt to simulate the mixing occurring in the lagoons, G-values from the CFD model were used to determine the mixing speeds of the paddles. The G-value is a measure of the mixing energy input. It was calculated as the root mean turbulent energy dissipation per unit of mass for each CV using equation [5.11] (MWH 2005).

$$G = \sqrt{\frac{\varepsilon}{\nu}} \qquad [5.11]$$

Where,
ε is the local turbulent dissipation energy (m^2/s^3);
and ν is the kinematic viscosity of water (m^2/s).

The G-value within the center of the lagoon was averaged at 0.21/sec and at inlet 400/sec. The conversion curve provided by Phipps and Bird was used to convert G-values to a paddle speed 300 RPM for one min and left to settle at 1 RPM.

All FBW samples were provided in 20 L totes by the JDKWTP staff. It was collected from a backwash trough during the beginning of a backwash. Various people provided the polymers that were tested as described in Table 5.2. Each polymer was tested in five doses, 3, 5, 10, 12 and 25 mg/L, based on optimal results found in literature as described in Chapter 2.

TABLE 5.2 POLYMERS TESTED TO PROMOTE SETTLING IN THE LAGOONS

Name	Source	Charge	Molecular Weight
Magnafloc LT20	JDKWTP Staff	Cationic	High
Flopam EMF 140 PWG	Benery Lake WTP Staff	Cationic	Medium
EM533 PWG	Millennium Water Staff	Anionic	Medium
EM630 PWG	Millennium Water Staff	Anionic	High

5.3 JDKWTP Lagoons Compared to Atlantic Canada Guidelines for Residual Lagoons

Hydraulic design is said to be one of the greatest determinates in the performance of WSPs (Persson 2000). The design of the engineered lagoons at the JDKWTP were compared with the Atlantic Canada Guidelines (CBCL 2004) (Table 5.3). This comparison shows that the lagoons were designed three times deeper than required (4.5 m as compared to the required 1.5 m). The HRT falls within the 15 to 30 day range and the total volume of the two lagoons is 8,000 m^3 larger than required. The theoretical retention time in this case is calculated by the daily incoming flow rate of 1,500 m^3 into the two 14,000 m^3 lagoons. However, the length to width ratio is half of what is required according to the guidelines (2:1 as compared to 4:1).

TABLE 5.3 THE DESIGN OF THE ENGINEERED LAGOONS AT THE JDKWTP WERE COMPARED WITH THE ATLANTIC CANADA DESIGN GUIDELINES FOR LAGOONS FOR THE TREATMENT OF WTP FBW.

Design Parameter and Guideline	JDKWTP Engineered Lagoons	Guideline Met
Theoretical HRT 15 to 30 days	18.7 days	Yes
Sludge Storage 2 Years	Infinite years of storage on site	Yes
Minimum of Two Parallel Lagoons	Two	Yes
Located in an area free of flooding		Yes
Minimum depth 1.5 m	4.5 m	Yes
Adjustable decanting device		Yes
Low permeability liner		Yes
Effluent sampling location		Yes
Outlets and inlets located to minimize short circuiting		No
Volume of ten times the volume of water discharged during a 24 hr period	28,000 m^3 compared to the required 20,000 m^3	Yes
Minimum length to width ratio 4:1	2:1	No
Minimum width to depth ratio 3:1	10:1	Yes
Velocity dissipated at the inlet		No

The length to width ratio is of particular interest because it is the design parameter that has the highest correlation to plug flow conditions (Persson 2000). Perfect plug flow conditions are supposed to represent the maximum hydraulic efficiency (Holland *et al.,* 2004; Shilton and Harrison 2003). To increase the hydraulic efficiency or the length to width ratio baffles can be added to the pond (Shilton and Harrison 2003; Broughton and Shilton 2012).

5.4 RESULTS AND DISCUSSIONS

5.4.1 HYDRAULIC CHARACTERIZATION OF ENGINEERED LAGOONS USING CFD

A snapshot of the velocity vectors is depicted in Figure 5.2 taken at approximately 7 virtual hours after the start of the modeled flow. Seven hours was chosen based on the estimated theoretical HRT calculated from dividing the total volume of the lagoons by incoming flow rate for a continuous flow (Equation 2.2). It was assumed that the actual lagoons were performing much closer to theoretical values. The velocity of the lagoon averages at 0.02 m/s, which is consistent with the mean retention time of the CFD modeled lagoons 4 hours and 57 minutes and the rhodamine water tracer study mean retention time of 3 hours and 50 minutes.

The velocity streamline diagram of Figure 5.2 shows the areas of highest velocity in red and lowest in blue. The velocity vectors highlight the short-circuiting route around the perimeter of the lagoon (in green). The short-circuiting appears to be caused by the high incoming velocity at the inlet, which causes the water to bounce off the back wall of the lagoon and continue to flow towards the outlet. This is similar to the flow pattern noted during the rhodamine tracer study in Chapter 4. Shitlon *et al.,* (2003) also noted a perimeter bound flow in a CFD model with a vertical inlet and no baffles.

The inlet flow rate at the JDKWTP lagoons varies greatly. The biggest determinate in the flow rate is the number of filters backwashed at a time. Each filter backwashed produces approximately 0.70 ML of FBW. However, this number varies, as the process is not automated and can range between 0.66 ML and 0.78 ML depending on a number of factors (Follett 2012). The CFD lagoons were modeled to represent 1.5 ML of backwash water, which is approximately equal to a two-filter backwash, the most common.

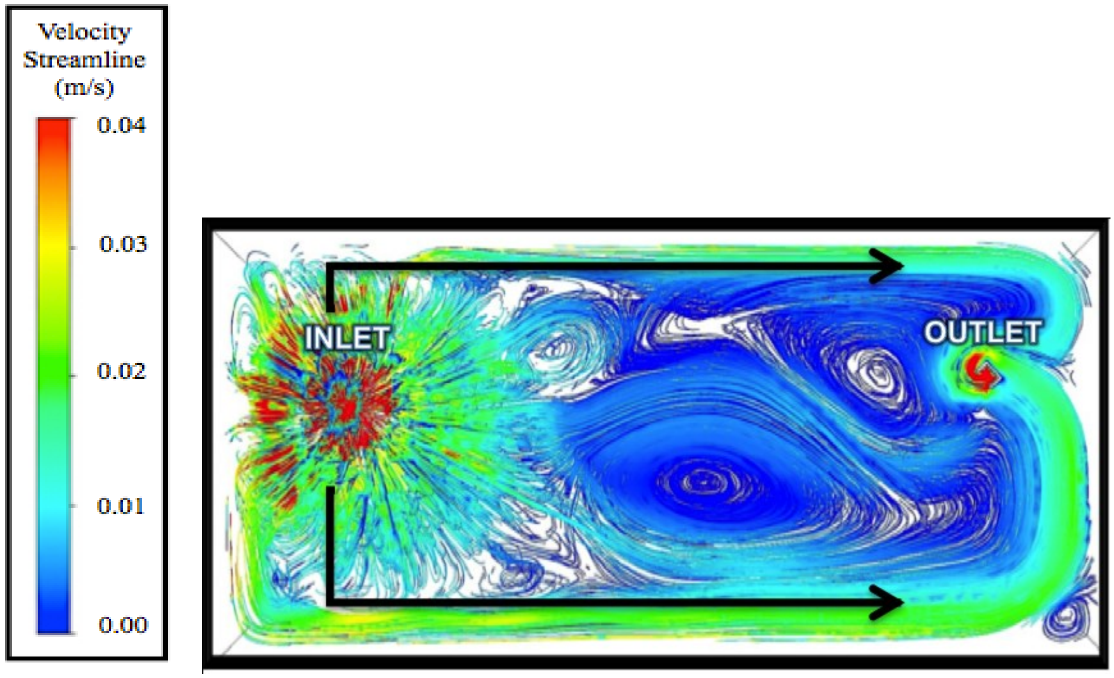

FIGURE 5.2 VELOCITY STREAMLINE DIAGRAM TO DEPICT AREAS OF SHORT-CIRCUITING AND DEAD ZONES, THE FLOW DIRECTION IS FROM LEFT TO RIGHT.

The CFD tracer study further indicates that the flow is largely perimeter based. Even after five hours of continuous flow the mass fraction of the tracer within the center of the lagoon is still zero this is as shown as the blue colour in Figure 5.3. This indicates a dead zone within the center of the lagoon and a large volume of the lagoon that is not being utilized for treatment. Based on the number of mesh cells with a mass fraction of less than 0.1 there is approximately 400m^3 not being used. This number is a low estimate as the mesh cells vary in size and the mesh within the center of the lagoon is significantly coarser than at the inlet and outlets where the velocity is significantly higher.

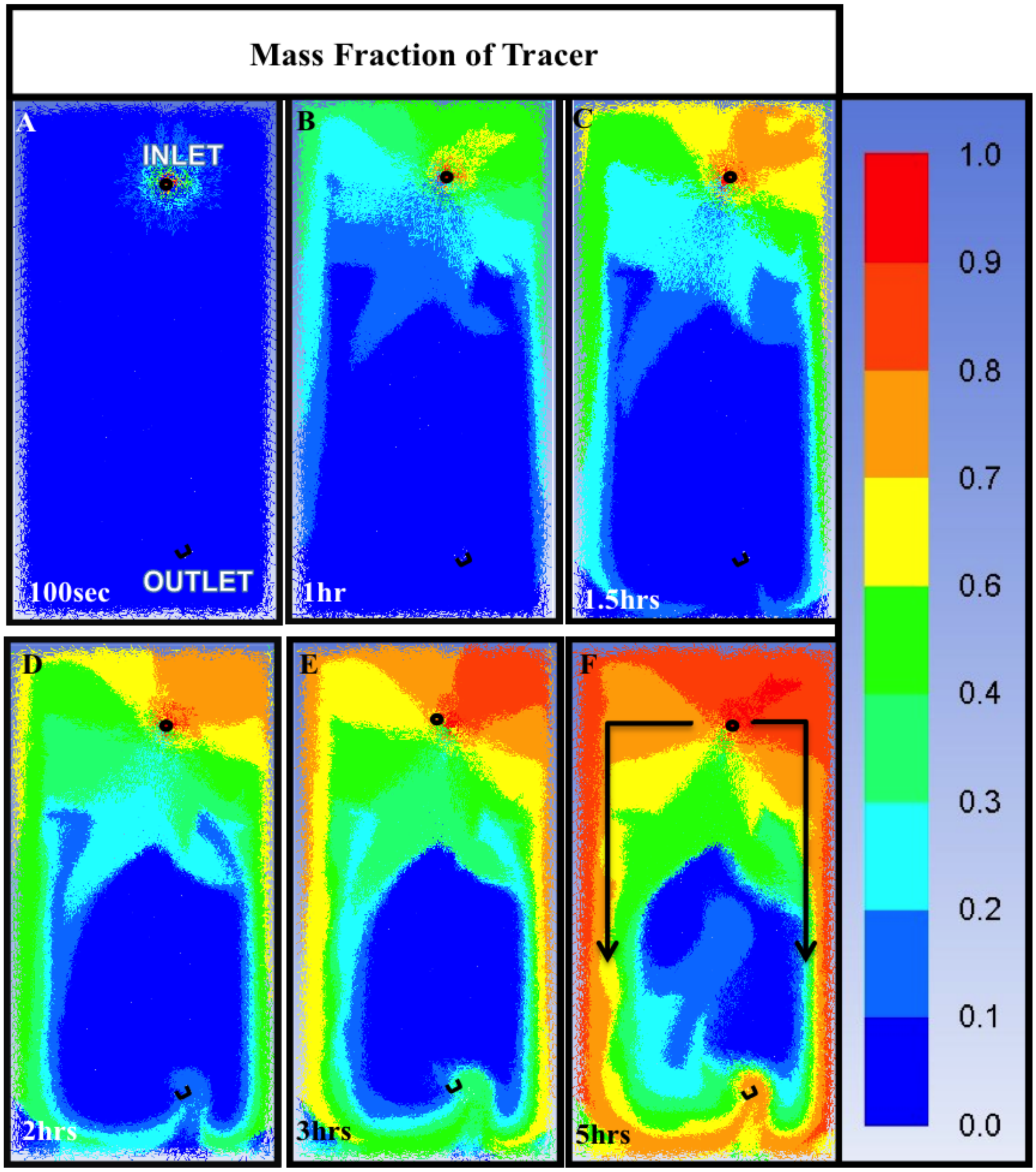

FIGURE 5.3 CFD TRACER STUDY SHOWING CUMULATIVE MASS FRACTION OF TRACER OVER TIME.

5.4.2 Hydraulic Characterization of Engineered Lagoons with Baffles Using CFD

Three baffle combinations were modeled using the same CFD method as the original lagoon discussed previously. The baffles were placed in an attempt to minimize the short-circuiting and dead zones. One baffle at the outlet, two and three baffles evenly spaced were deemed most likely to have the greatest improvement on the pond hydraulics based on initial CFD models and previous studies (Shilton and Harrison 2003).

The velocity vectors in Figures 5.4 to Figure 5.6 depict improved mixing patterns over the unbaffled pond (Figure 5.2). The dead zones (blue areas) are reduced, caused by the baffles forcing the short-circuiting along the perimeter of the lagoon more into the center of the lagoon. In the case of three baffles (Figure 5.6) it appears that flow has become channelized along the baffles.

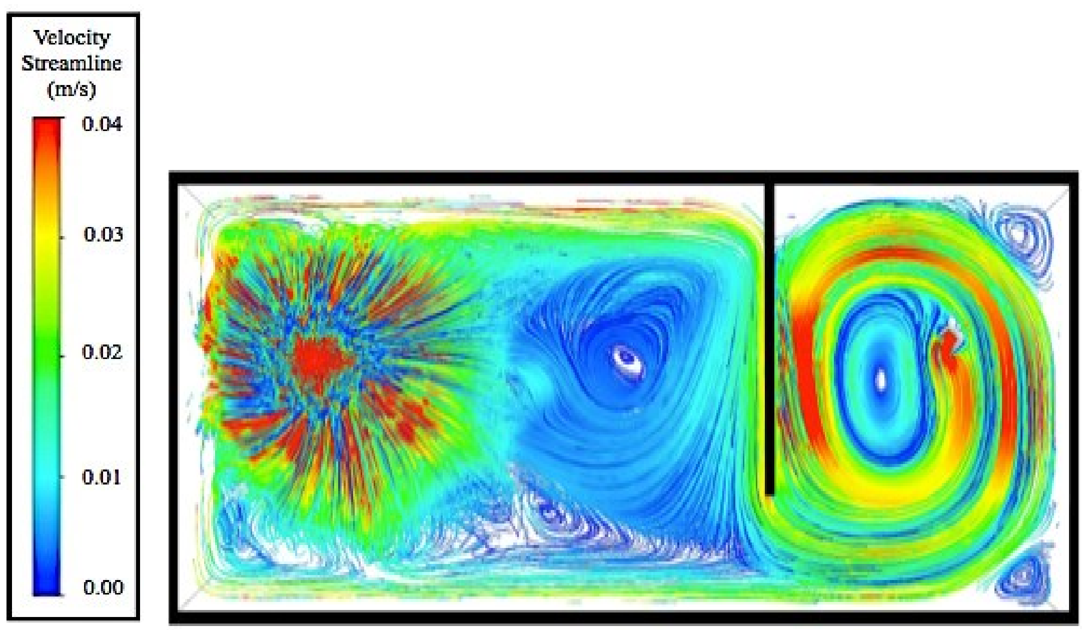

FIGURE 5.4 CFD MODELED LAGOON WITH ONE BAFFLE AT THE OUTLET

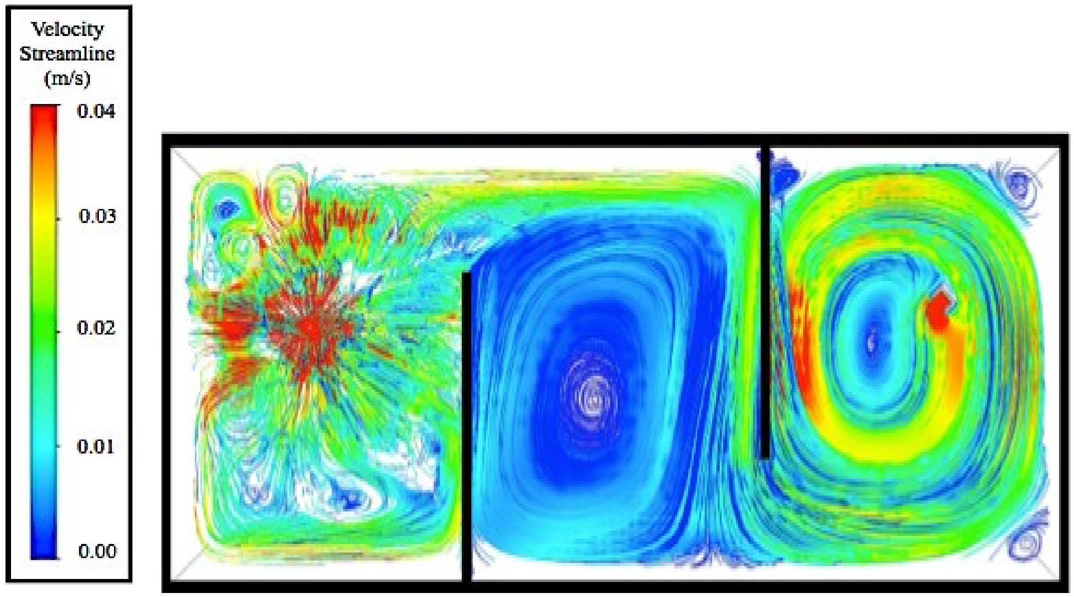

FIGURE 5.5 CFD MODELED LAGOONS WITH TWO BAFFLES

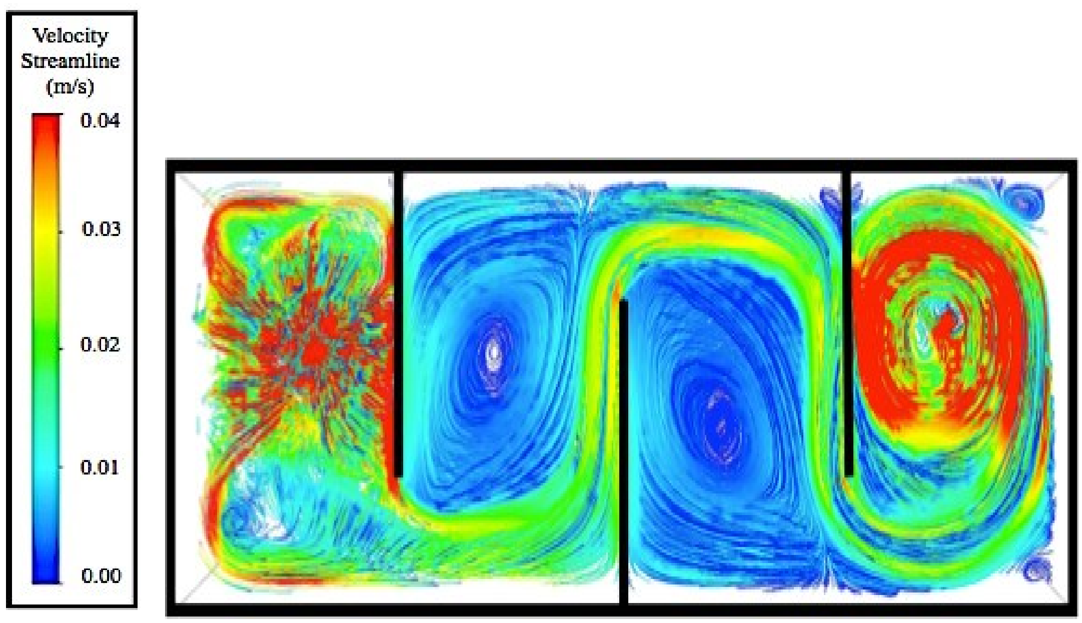

FIGURE 5.6 CFD MODELED LAGOON WITH THREE BAFFLES

5.4.3 RTD CURVES AND ANALYSIS

All the baffle arrangements showed improvements in mean retention time over the unbaffled lagoon (Table 5.3). The greatest improvement came from the two-baffle arrangement with an hour and 26-minute increase in mean retention time over the unbaffled CFD lagoon. The three baffles showed the least improvement in retention time. In previous studies, two baffles have also been found to be the optimal number of baffles in both pilot-scale and CFD modeled ponds (Shilton and Harrison 2003; Abbas *et al.,* 2006; Vega *et al.,* 2003). This may be caused by the channelized flow created by using too many baffles. Figure 5.6 depicts the channelized flow with a hirer velocity along the baffles. Similarly, Pedahzur *et al.,* 1993 compared two and four baffles at full-scale and found no improvement in pond hydraulics or microbial removal when using four baffles over two baffles.

The RTD curves for each of the baffle combinations showed the same typical tracer response of a skewed bell shape curve (Figure 5.7). All of the baffle combinations proved to dampen the peak out flow over the unbaffled CFD lagoon.

TABLE 5.3 KEY TIMES FOR THE RTD CURVES OF THE VARIOUS BAFFLE ARRANGEMENTS.

Setup	Initial Discharge (hrs)	Time of Peak (hrs)	Mean Retention Time (hrs)
JDKWTP	0:25	NA	3:50
CFD No Baffles	1:07	2:17	4:54
CFD One Baffle at Outlet	1:16	3:19	6:11
CFD Two Baffles	1:31	4:00	6:22
CFD Three Baffles	1:28	3:35	5:47

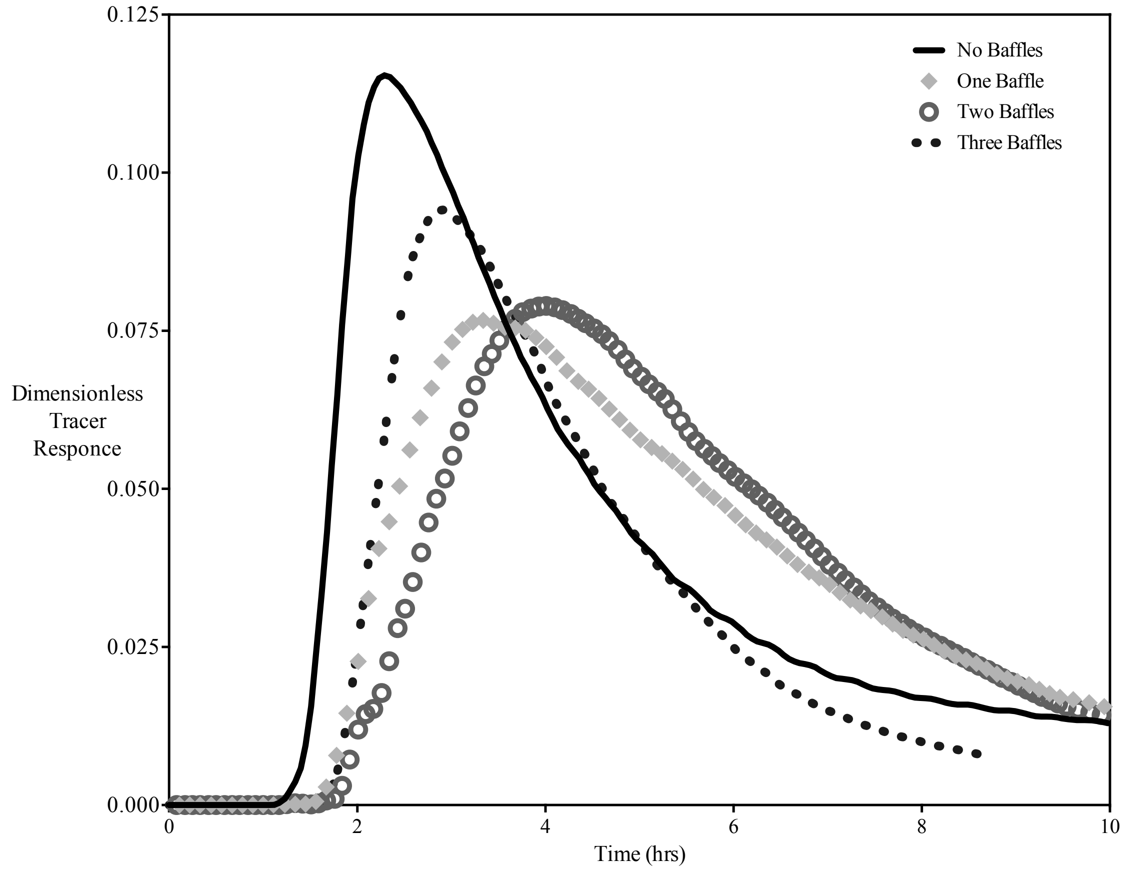

FIGURE 5.7 RTD CURVES FOR THE MODELED LAGOONS WITH NO BAFFLES, ONE, TWO AND THREE BAFFLE ARRANGEMENTS.

5.4.4 CFD IMPROVEMENTS

Due to computational limitations (Table 5.4), the CFD tracer response curves do not resemble the tracer response curve from the tracer study conducted at the JDKWTP lagoons (Figure 4.2). The limitations include the inability to model the unsteady flows that occur at the JDKWTP with every backwash. Also, the sludge accumulation was not modeled with the CFD models. Therefore, possible improvements in the lagoon's hydraulics from the use of baffles cannot be explicitly stated and the retention time between the JDKWTP lagoons and the CFD modeled lagoons are not directly comparable. However, the curves can be used to compare the benefits of baffles between the different

CFD models and this information can be used as a guide for the optimal situation to direct future pilot-scale experiments.

TABLE 5.4 MAIN CAUSES FOR DIFFERENCES IN RTD CURVES BETWEEN THE JDKWTP LAGOONS AND THE MODELED LAGOONS

	CFD Lagoon	JDKWTP Lagoon
Flow Conditions	Steady	Unsteady
Solids Accumulation	No solids accumulation	Significant solids accumulation

To create a more similar flow pattern to the one seen at JDKWTP a user defined function (UDF) was created (Appendix E). This function was written in the language of C and fed into Fluent® at the inlet boundary to set the inlet velocity condition. However due to computational limitations and the initial conditions applied the UDF was unable to simulate the pulse nature of the flows occurring the at the JDKWTP lagoons. The model was run as incompressible to eliminate the pressure term from the Navier-Stokes equations this is common practice in the water treatment industry as water can be assumed to an incompressible fluid (Peterson *et al.,* 2000; Wu and Chen 2011). Modeling the fluid as incompressible although improved computational performance limited the performance of the UDF. This resulted in RTD curve that also did not represent the actual lagoon discharge of 6.5 hours (Figure 4.2). Instead it resulted in a lagoon discharge of 30 minutes the total time of the modeled backwash. This is consistent with the law of conservation of mass. Therefore the use of the UDF was not continued.

To better represent the free surface of the lagoons a volume of fluid (VOF) model may have been more appropriate. The VOF model allows for the addition of different phases or immiscible fluids (Ansys 2006). Therefore the air layer above the water as it is with the lagoons could be modeled. This would create a better representation of the water being discharged at the inlet and at the overflow weir by allowing for the modeling of the change in water depth with the addition of the backwash water. This model is not readily used in the water treatment industry due to computational limitations; more commonly a single-phase model is used (Peterson *et al.,* 2000; Templeton *et al.,* 2006; Khan *et al.,* 2012). However, He *et al.,* (2004) used a VOF it has been used to model a combined sewer overflow (CSO). The VOF may have been more necessary in the situation of the CSO due to the large change in water depths. The VOF model was attempted during this study for a more accurate representation of the lagoon's flow pattern. Due to computational limitations the VOF model was not continued.

5.4.5 FBW SETTLING

Settling of the FBW was analyzed with no polymer addition to determine an optimal retention time (Figure 5.8). The FBW was left to settle for up to 19 hours and sampled regularly. It was found that 96% of the aluminum settled out within the first ten minutes. Gruninger (1975) and Ma *et al.,* (2007) found a similar trend for alum FBW settling except their experiments were conducted in terms of depth of solids. The initial discharge time of the two baffle CFD modeled lagoons was one hour and 30 minutes, which would result in an aluminum concentration of 2520.0±1070.0 μg/L. This is compared to the actual initial discharge time of 30 minutes based on the JDKWTP tracer study which would result in an aluminum concentration of 3040.0±1230.0 μg/L for a 21% decrease in aluminum concentration. There was no statistically significant ($p>0.05$) difference between the 30-minute (3040 ± 1230 μg/L) aluminum sample and the samples that were allowed to settle for a longer period. Not until 19 hours of settling was there was a significant difference ($p=0.05$) between the 30-minute (3040 ± 1230 μg/L) and the 19 hour aluminum sample (1580.0±811.0 μg/L), the aluminum concentration was still greatly in excess of the 184 μg/L regulation and a 19 hour HRT is most likely an unreasonable improvement in HRT.

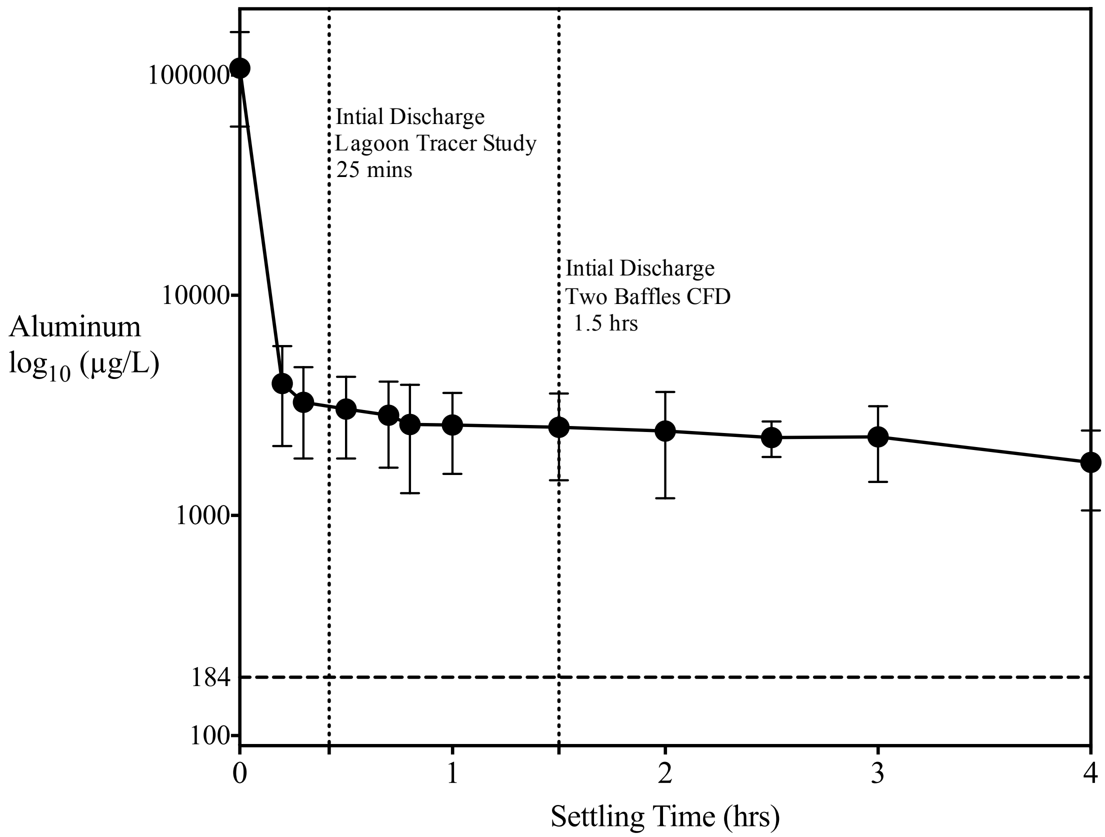

FIGURE 5.8 SETTLING OF FBW IN TERMS OF ALUMINUM WITH NO POLYMER ADDITION COMPARED WITH THE 184 µg/L REGULATION. (ERROR BARS REPRESENT ONE SD AROUND THE MEAN).

5.4.6FBW SETTLING WITH POLYMER ADDITION

A very similar settling trend was noted when the polymer was used to enhance the settling as depicted in Figure 5.9. With the polymer, the settling was not quite as fast as without and did not stabilize until around an hour as opposed to without where the settling was essentially complete within 10-minutes. When comparing the two initial discharge times, the 30-minutes found during the JDKWTP tracer study and the 1.5-hours found when two baffles are added there was an improvement in aluminum concentration with the increased retention time. Therefore, if polymer conditioning is being employed it may be advantageous to also lengthened retention time. In Figure 5.9, the results from medium molecular weight (MW) anionic polymer are shown. Regardless of the polymer type or dose, the aluminum concentration was improved over no polymer addition. However, settling improved with decreasing polymer dose.

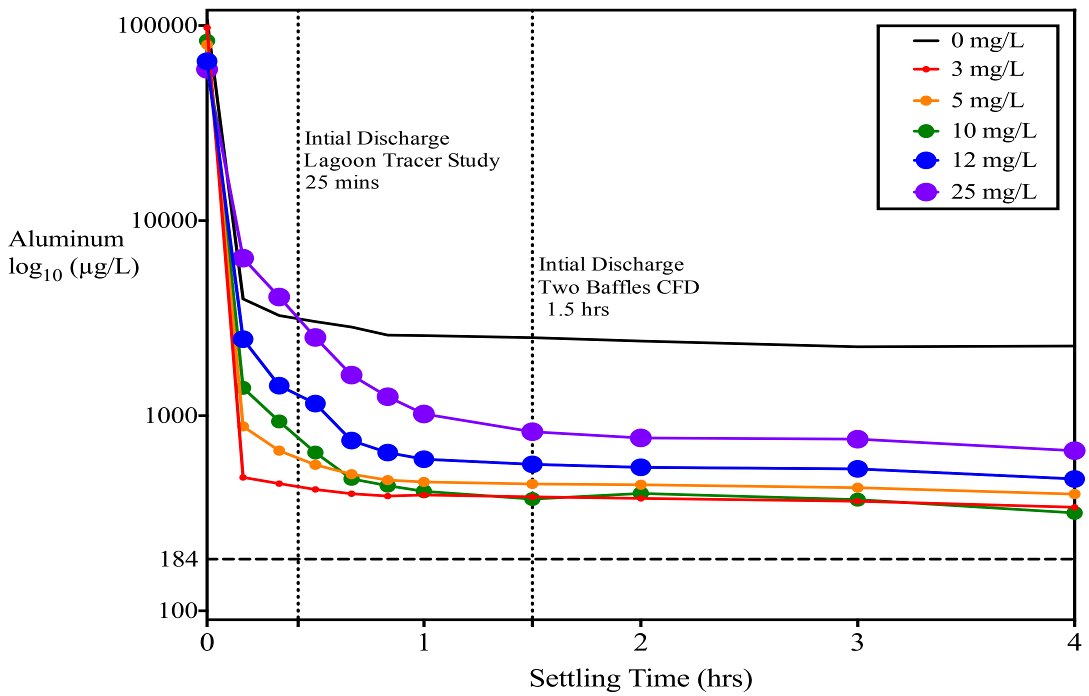

FIGURE 5.9 ALUMINUM SETTLING TREND FOR THE MEDIUM MOLECULAR WEIGHT ANIONIC POLYMER.

The different types of polymer showed similar trends with the low doses typically performing better than the higher doses (Figure 5.10). The 5 mg/L cationic high MW polymer with one hour 30 minutes of settling showed optimal the best aluminum settling with an aluminum concentration of 101 µg/L of aluminum.

The high MW polymer typically settled faster than the medium MW as the 30-minute samples are more consistently lower in aluminum. However, after 1.5 hours of settling, the medium MW cationic showed the best results with all of the doses except the 25 mg/L having an aluminum concentration below 250 µg/L. The lower doses of polymer tend to have better results because at higher doses the colloidal particles will gain a charge and re-stabilize in the water (Droste 1997).

Optimal doses found in literature vary significantly, Cornwell and Lee (1994) found an optimal polymer dose of 0.1 mg/L in contrast Ma *et al.,* (2007) found an optimal dose between 30 and 50 mg/L for FBW settling. This wide range in optimal doses is likely caused by the differences in FBW characteristics and thus bench-scale experiments are needed for optimization.

Studies have shown that adding polymer to the backwash water before backwashing could improve filter performance in addition to improving FBW settling in settling ponds (Yapijakis 1982). A study conducted on backwash water quality at the JDKWTP at a bench-scale looked at adding polymer at a dose of 10 mg/L to the backwash to improve filter performance. This study showed that the filter ripening time in the winter was bettered; however, filter run time was shortened (Follett 2012).

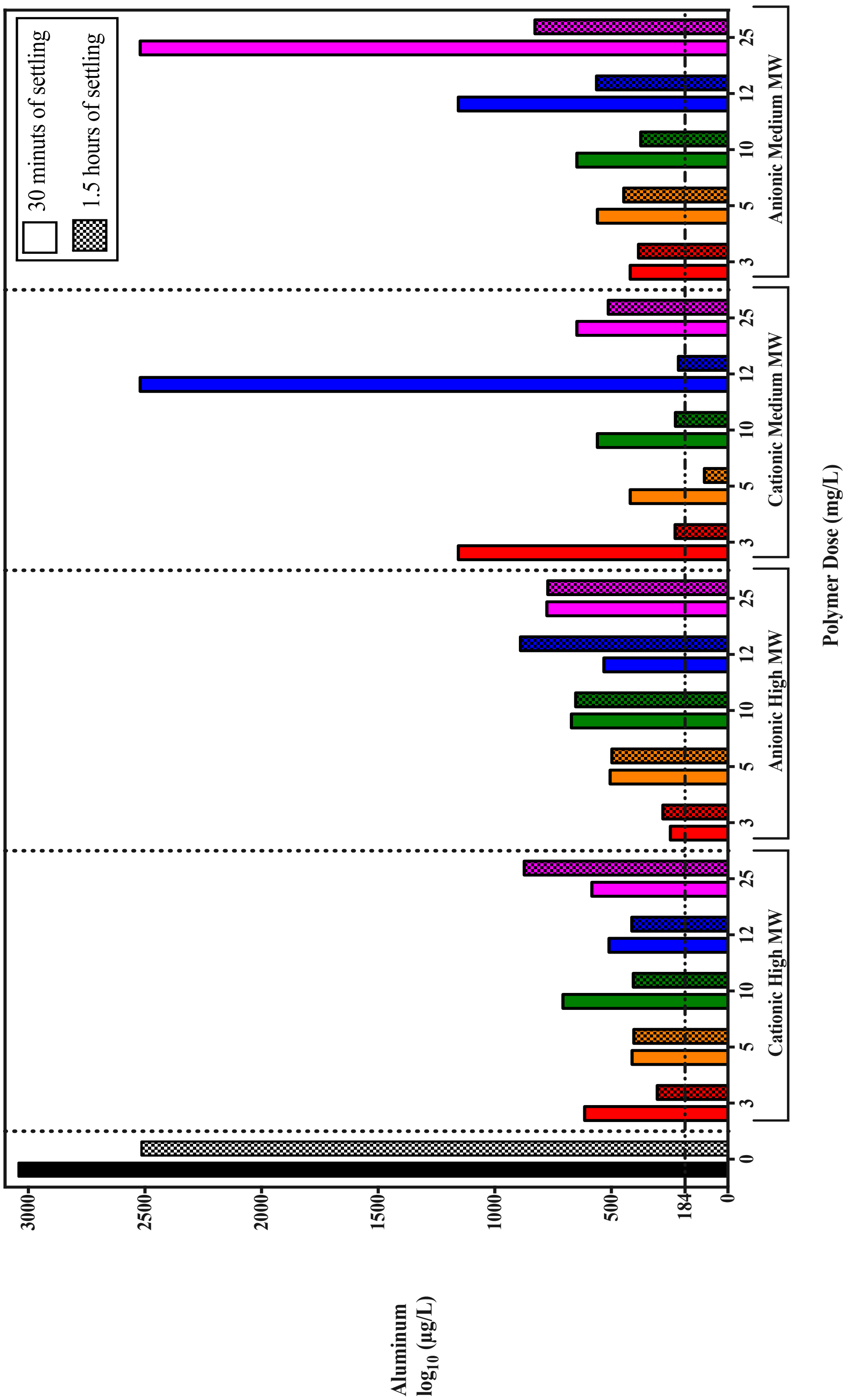

FIGURE 5.10 BENCH-SCALE SETTLING AFTER 30 MINUTES AND 1.5 HRS FOR DIFFERENT TYPES AND DOSES OF POLYMER.

5.5 DISCUSSION

A 3D CFD model using Ansys Fluent was created using the geometry of the JDKWTP FBW lagoons. It was shown by the modeled tracer studies that the flow tends to short-circuit around the perimeter of the lagoon leaving a large dead-zone in the center of the lagoon. This flow pattern is thought to be caused by the vertical inlet with no velocity dissipation mechanism. The water appears to hit the back wall of the lagoon and follow the wall around the perimeter to the outlet at the other end.

Due to computational limitations, the unsteady nature of the flow with the nightly backwashes into the lagoons at the JDKWTP was not modeled. Because of this limitation, the RTD curves of the CFD and the JDKWTP lagoons do not depict the same discharge pattern and it is difficult to directly compare them.

Adding baffles improved retention time in the modeled lagoons. The baffles acted to dissipate the velocity of the water at the inlet forcing flow into what was previously a dead zone at the center of the lagoons. All the baffle combinations improved the retention time of the lagoons; however, two evenly spaced baffles at 70% width noted the biggest improvement to the retention time. The average retention time increased from four hours and 57 minutes with no baffles to six hours and 22 minutes with two baffles.

At a bench-scale, polymer addition greatly reduced aluminum concentration in the settled FBW. Five doses of four different types of polymer were tested using a jar test apparatus. Each of the polymer doses and types improved aluminum concentration in the settled even with only 30 minutes of settling. The optimal results were found for a 5 mg/L dose of cationic medium molecular weight polymer with a settling time of 1.5 hours, which is representative of the initial discharge of the two-baffle CFD model.

Comparing a bench-scale 1 L experiment to a 14,000 m^3 lagoon is unrealistic. However, these setting tests act as a starting point for a pilot-scale study performed in conditions more similar to the lagoons at the JDKWTP.

Chapter 6: CONCLUSIONS

6.1 SYNTHESIS

The purpose of this study was to develop an understanding of the current treatment of the FBW at the JDKWTP and develop an easily implemented strategy to improve FBW treatment. The JDKWTP uses alum as a coagulant in the water treatment process. The aluminum then ends up in the FBW and is not adequately removed during the FBW treatment process. This process employs two engineered lagoons in parallel, followed by a natural wetland. This study characterized the treatment process through a hydraulic characterization and water quality analysis. Two methods of improving the treatment occurring in the system were studied. To improve the treatment hydraulically, a fluid dynamics model was created to determine benefits of adding baffles to the engineered lagoons to improve retention times. To improve the treatment chemically, polymer conditioning the FBW to increase the rate of settling was also studied.

6.1.1 HYDRAULIC CHARACTERIZATION OF THE FBW TREATMENT SITE

A hydrograph showed that backwash results in a significant increase in discharge into and out of the wetlands. Within minutes of each backwash, the flow from the lagoons into the wetland increased from 0.3 L/s to 10 L/s. Within an hour of the backwash, the flow increased from 75 L/s to 100 L/s.

Rhodamine tracer studies conducted in the lagoons and in the natural wetland displayed similar results as the hydrograph. The lagoon tracer study determined that the flow within the lagoons short-circuits around the perimeter of the lagoon as a result of the high velocity at the vertical inlet. The RWT was first detected 25-minutes after the start of the backwash and continued to discharge for 6.5-hours. The wetland tracer study determined an initial discharge time of 5 hours and 24 minutes after the start of the backwash. This results in a minimum retention time of 5 hours and 49 minutes.

6.1.2 WATER QUALITY ANALYSIS

Water quality monitoring conducted determined that there was 99% removal of aluminum and TSS across the entire system. Monitoring also determined that the average aluminum concentration where the wetlands discharge into LPL was 669±471 µg/L and never met the site-specific guideline of 184 µg/L. Of the other water quality parameters measured, the copper concentration (4.80±3.80 µg/L) discharging into Little Pockwock Lake was above the CWQG of 2 µg/L. Sampling during different flow conditions identified that water quality is dependent on the time of the previous backwash.

6.1.3 FLUID DYNAMICS MODEL

The CFD model created using Ansys Fluent 14.0® based on the geometry of the lagoons at the JDKWTP showed that the flow was primarily perimeter based. The perimeter based flow results in a large dead-zone within the center of the lagoons.

Three baffle placements were modeled; one baffle at the outlet, two and three baffles all at 70% width. All three of the baffle arrangements showed improvements in retention times over the unbaffled lagoon. The greatest improvement was noted in the two-baffle lagoon with an increase in the mean retention time of 4 hours and 54 minutes for the unbaffled CFD lagoon to 6 hours and 22 minutes.

6.1.4 BENCH-SCALE FBW SETTLING

Settling tests at a bench scale using the mixing patterns determined from the CFD model showed that the FBW settles within 10-minutes and no significant change in aluminum occurs until 19-hours of settling. The same settling tests with polymer conditioning of the FBW showed that regardless of dose or type of polymer that significant improvements in aluminum concentration could be achieved. A dose of 5 mg/L of medium molecular weight cationic polymer with 1.5-hours of settling (initial discharge time of 2-baffle arrangement) resulted in aluminum concentration of 101 µg/L, which is below the 184 µg/L discharge guideline.

6.2 AREAS OF FUTURE RESEARCH

Future areas of research should focus on scaling up the treatment methods experimented within this study. Bench-scale and CFD experiments act as a cost effective first step to determining a solution to reduce aluminum concentrations from the FWB treatment wetland at the JDKWTP. Many factors would affect the results of baffling the lagoons or using polymer to conditioning the FBW that were not or could not be considered through modeling and bench-scale experiments. Therefore, these studies act as an initial step in determining a possible solution.

Performing similar studies at a pilot-scale would allow for the incorporation of some of the conditions that could not be experimented with through modeling and bench-scale experiments to better emulate the actual conditions of the JDKWTP lagoons. Flow conditions that are more representative of the flows at the JDKWTP lagoons could be created at a pilot-scale. This would determine how the baffling of the lagoons would affect retention times with discontinuous flows. Piloting also lends itself to a longer time scale for settling tests to determine if the effects of the polymer are sustained. Gruninger (1974) showed promising results with polymer enhanced FBW settling at a pilot-scale.

References

Abdulla, M., Baba, I. and Erdawati, S. (1995) "Distribution of Aluminum From Alum Sludge in Water and Sediment." *Marine and Freshwater Research* 46: 159-165.

Anderson, J. D., Degroote, J., Degrez, G., Dick, E., Grundmann, R. and Vierendeels, J. (2009). *Computational Fluid Dynamics*. Edited by John F Wendt. Third Ed. Berlin: Springer.

Anderson, J. D. (1995). *Computational fluid dynamics: The basics with applications*. New York: McGraw-Hill.

Ansys Inc. (2010). *ANSYS Meshing User's Guide*. Lebanon NH.

Ansys Inc. (2006) User's manual, ANSYS Inc., Lebanon NH.

AWWA. (1996). "Tracer Studies in Water Treatment Facilities: a Protocol and Case Studies."

Baawain, M., El-Din, G. and Smith, D., (2006). "Computational Fluid Dynamics Application in Modeling and Improving the Performance of a Storage Reservoir Used as a Contact Chamber for Microorganism Inactivation." *Journal of Environmental Engineering and Science* 5 (2): 151–62. doi:10.1139/s05-023.

Bache, D. and Zhao. Y. (2001). "Optimizing Polymer Use in Alum Sludge Conditioning: an Ad Hoc Test." *Journal of Water Supply: Research and Technology—AQUA*: 29–39.

Baldigo, B. P., Lawrence, G. and Simonin, H. (2007). "Persistent Mortality of Brook Trout in Episodically Acidified Streams of the Southwestern Adirondack Mountains, New York." *Transactions of the American Fisheries Society* 136 (1): 121–34. doi:10.1577/T06-043.1.

Behrens, H, Beims, U., Dieter, H., Dietzer, G., Eikmann, T., Grummt, T., Hanisch, H., *et al., (*2001). "Toxicological and Ecotoxicological Assessment of Water Tracers." *Hydrogeology Journal* 9 (3): 321–25. doi:10.1007/s100400100126.

Birchall, J. D, Exley, C., Chappell, J. and Phillips, M. (1989). "Acute Toxicity of Aluminum to Fish Eliminated in Silicon-Rich Acid Waters." *Letters to Nature* 338: 146–48.

Böhm, N., and Kulicke, W. M. (1997). "Optimization of the use of polyelectrolytes for dewatering industrial sludges of various origins." *Colloid & Polymer Science, 275* (1): 73-81.

Bolto, B. and Gregory. J., (2007). “Organic Polyelectrolytes in Water Treatment.” *Water Research* 41 (11): 2301–24. doi:10.1016/j.watres.2007.03.012.

Booth, C. E, McDonald, D. G., Simons, B. P. and Wood. C. M. (1988). “Effect of Aluminum and Low pH on Net Ion Fluxes and Ion Balance in Brook Trout (Salvelinus Fontinalis).” *Canadian Journal of Fisheries and Aquatic Sciences* 45: 1563–74.

Borne, K. E., Fassman, Elizabeth A. and Tanner, Chris C. (2013). “Ecological Engineering.” *Ecological Engineering* 54: 173–82. doi:10.1016/j.ecoleng.2013.01.031.

Bourgeois, J. C, Walsh, M. E. and Gagnon. G. A. (2004). “Comparison of Process Options of Treatment of Water Treatment Residual Streams.” *Journal Environmental Engineering* 3: 477–84.

Broughton, A. and, Shilton, A. (2012). “Tracer Studies on an Aerated Lagoon.” *Water Science & Technology* 65 (4): 611-17. doi:10.2166/wst.2012.906.

Camnasio, E., Orsi, E. and Schleiss, A. J. (2011). “Experimental Study of Velocity Fields in Rectangular Shallow Reservoirs.” *Journal of Hydraulic Research* 49 (3): 352–58. doi:10.1080/00221686.2011.574387.

CBCL. (2004). *Atlantic Canada Guidelines for the Supply, Treatment, Storage, Distribution, and Operation of Drinking Water Supply Systems*.

CBCL. (2010). “Aluminum Discharge Study”: 1–36.

CBCL. (2008). “Glace Bay WTP Wastewater Lagoon Assessment”: 1–43.

CBCL. (2006). “Yarmouth Lagoons.”: 1-16.

CCME. (2005). “Application and Testing of the Water Quality Index in Atlantic Canada Report Summary.”

CCME. (1999). *Canadian Water Quality Guidelines for the Protection of Aquatic Life. Canadian Council of Ministers of the Environment*, Winnipeg. Available online at: http://ceqg- rcqe.ccme.ca/

Choi, B. S., Wan, B., Philyaw, S., Dhanasekharan, K. and Ring T. (2004). "Residence Time Distributions in a Stirred Tank: Comparison of CFD Predictions with Experiment." *Industrial & Engineering Chemistry Research* 43 (20): 6548–56. doi:10.1021/ie0308240.

Cornwell, D. A. and Ramon G Lee. (1994). "Waste Stream Recycling: Its Effect on Water Quality." *Journal AWWA*: 50–63.

Cornwell. D., MacPhee. M., McTigue, N., Arora, H., DiGiovanni, G., LeChevallier. M. and Taylor. J. (2001). Treatment options for *Giardia, cryptosporidium* and other contaminants in recycled backwash water. *American Water Works Research Foundation (AWWARF)*, Denver. Colo.

Crittenden, J. C. and Montgomery Watson Harza (Firm). (2005). *Water treatment principles and design*. Hoboken, N.J: J. Wiley.

Dennis, I. and Clair, T. (2012). "The Distribution of Dissolved Aluminum in Atlantic Salmon (Salmo Salar) Rivers of Atlantic Canada and Its Potential Effect on Aquatic Populations": 1174–83. doi:10.1139/F2012-053.

Droste R. L. (1997). *Theory and practice of water and wastewater treatment*. New York: J. Wiley.

Driscoll, C. T. (1985). "Aluminum in Acidic Surface Waters: Chemistry, Transport and Effects." *Environmental Health Perspectives* 63: 93–104.

Driscoll, C. T., Baker, J. T., Bisogni, J. J. and Schofield. C. L. (1980). "Effect of Aluminum Speciation on Fish in Dilute Acidified Waters." *Nature* 284: 161–64.

Ecuru, J., Okot-Okumu, J. and Okurut, T. (2011). "Monitoring Residual Chlorine Decay and Coliform Contamination in Water Distribution Network of Kampala, Uganda." *Journal of Applied Science and Environmental Management* 15: 167–73.

Eger, P. (1994). "Wetland Treatment for Trace Metal Removal From Mine Drainage: the Importance of Aerobic and Anaerobic Processes." *Water Science & Technology* 29: 249–56.

ESRI. (2007). "Delineating watersheds." *ArcGIS 9.2 Desktop Help*. Retrieved from http://webhelp.esri.com/arcgisdesktop/9.2/index.cfm?TopicName=Delineating%20watersheds

Falconer, R. and Liu, S. (1988). “Modeling Solute Transport Using Quick Scheme.” *Journal of Environmental Engineering* 114 (1): 3-20.

Flanagan, N. E., Mitsch, W. J. and Beach. K. (1994). “Predicting Metals Retention in a Constructed Mine Drainage Wetland.” *Ecological Engineering*: 135–59.

Fogler, S. (2006). Elements of Chemical Reaction Engineering. Fourth Edition. Prentice Hall. Upper Saddle River, New Jersey, United States.

Follett, M. H. (2012). *Assessment of Filter Ripening at a Direct Filtration Plant, Halifax, NS.* (Masters dissertation). Dalhousie University, Canada.

Frecker, M. F. (1991). “Dementia in Newfoundland: Identification of Geographical Isolate.” *Journal of Epidemiology and Community Health* 45: 307–11.

Gauthier, E., Frontier, I., Courchesn, F., Pepin, P., Mortimer, P. and Gauvreau. D. (2000). “Aluminum Forms in Drinking Water and Risk of Alzheimer's Disease.” *Environmental Research* 84 (3): 234–46. doi:10.1006/enrs.2000.4101.

Gensemer, R. W. (2000). “The Effects of pH and Aluminum on the Growth of Acidophilic Diatom.” *The American Society of Limnology and Oceanography* 36: 123–31.

Gensemer, R. W. and Playle, R. (1999). “The Bioavailability and Toxicity of Aluminum in Aquatic Environments.” *Critical Reviews in Environmental Science and Technology* 29 (4): 315–450. doi:10.1080/10643389991259245.

Goula, A. M., Kostoglou, M., Karapantsios, T. D. and Zouboulis. A. (2008). “A CFD Methodology for the Design of Sedimentation Tanks in Potable Water Treatment.” *Chemical Engineering Journal* 140 (1-3): 110–21. doi:10.1016/j.cej.2007.09.022.

Goulet, R. R., Lalonde, J., Munger, C., Dupuis, S., Dumont-Frenette, G., Prémont, S. and Campbell. P. (2005). “Phytoremediation of Effluents From Aluminum Smelters: a Study of Al Retention in Mesocosms Containing Aquatic Plants.” *Water Research* 39 (11): 2291–2300. doi:10.1016/j.watres.2005.04.029.

Grayman, W. M., Deininger, R., Green, A., Boulos, P. F., Bowcock, R. and Godwin. C. (1996). “Water Quality and Mixing Models for Tanks and Reservoirs.” *Journal AWWA*: 60–74.

Gundersen, B.T., Bustarnan, S., Seim, W. K. and Curtis. R. L. (1994). "pH, hardness, and humic acid influence aluminum toxicity to rainbow trout (Qncsshynchus mykiss) in weakly alkaline waters." *Can. j. Fish. Aquat. Sci.* 51: 1345-1 355.

Gruninger, R. M. (1975). "Disposal of Waste Alum Sludge from Water Treatment Plants." *Journal (Water Pollution Control Federation),* 47(3): 543-552.

Hadiyanto, H, Elmore, S., Van Gerven, T. and Stankiewicz, A. (2013). “Hydrodynamic Evaluations in High Rate Algae Pond (HRAP) Design.” *Chemical Engineering Journal* 217: 231–39. doi:10.1016/j.cej.2012.12.015.

Halifax Water. (2013). “Typical Analysis of Pockwock and Lake Major Water.”

Hamzah, A., Abdullah, M. P., Sarmani, S. and Johari, M. A. (1997). “Chemical and Bacteriological Monitoring of Drinking Water From an Urban Water Catchment Drainage Basin.” *Environmental Monitoring and Assessment* 44: 327–38.

He, C., Marsalek, J. and Rochfort, Q. (2004). “Numerical Modelling of Enhancing Suspended Solids Removal in a CSO Facility.” *Water Quality Research Journal of Canada* 39: 457–65.

Headley, T. and Kadlec, R., (2007). “Conducting Hydraulic Tracer Studies of Constructed Wetlands a Practical Guide.” *Ecohydrology & Hydrobiology* 7: 269–82.

Health Canada. (2010). Guidelines for Canadian Drinking Water Quality: Summary Table. *Water, Air and Climate Change Bureau, Healthy Environments and Consumer Safety Branch,* Ottawa, Canada.

Holland, J. F., Martin, J., Granata, T., Bouchard, V. and Quigley, M. (2004). “Effects of Wetland Depth and Flow Rate on Residence Time Distribution Characteristics.” *Ecological Engineering* 23 (3): 189–203. doi:10.1016/j.ecoleng.2004.09.003.

Ingersoll, C. G., Mount, R. D., Gulley, D. D., La Point, H. W. and Bergman, H. L. (1990). “Effects of pH, Aluminum and Calcium on Survival and Growth of Eggs and Fry of Brook Trout (Salvelinus Frontinalis).” *Canadian Journal Fisheries and Aquatic Sciences* 47 (8): 1580–92.

Kadlec, R. H. (1994). “Detention and Mixing in Free Water Wetlands.” *Ecological Engineering* 3: 345-380.

Kadlec, R. H. and Wallace, S. D. (2009). *Treatment Wetlands*. Boca Raton, FL: CRC Press.

Kaggwa, R., Mulalelo, C., Denny, P. and Okurut, T. (2001). “The Impact of Alum Discharges on a Natural Tropical Wetland in Uganda.” *Water Resources Management* 35 (3): 795–807.

Kennedy, M. G., Ahlfeld, D. P., Schmidt, D. P. and Tobiason. J. E. (2006). “Three-Dimensional Modeling for Estimation of Hydraulic Retention Time in a Reservoir.” *Journal of Environmental Engineering* 132 (9): 976-84. doi:10.1061/(ASCE)0733-9372(2006)132:9(976).

Kingston Utilities. (2012). 2012 Annual Report on Drinking Water Quality, King St. Water Treatment Plant. utilitieskingston.com 20 September 2013. <http://www.utilitieskingston.com/pdf_downloads/2012%20King%20St.%20WTP%20Annual%20Report.pdf>

Kim, D., Kim, D., Kim, J. and Stoesser, T. (2010). “Large Eddy Simulation of Flow and Tracer Transport in Multichamber Ozone Contactors.” *Journal of Environment Engineering* 136: 22–31. doi:10.1061/ASCE.EE.1943-7870.0000118.

Kozłowski, H., Brown, D. R. and Valensin, G. (2006). *Metallochemistry of neurodegeneration biological, chemical, and genetic aspects*. Cambridge: Royal Society of Chemistry. http://dx.doi.org/10.1039/9781847555311.

James M. Montgomery, Consulting Engineers. (1985).*Water treatment principles and design*. New York: Wiley.

Launder, B. E. and Spalding, D. B. (1974), "The numerical computation of turbulent flows", *Computer Methods in Applied Mechanics and Engineering*, 3(2): 269-289.

Levenspiel, O. (1999). Chemical Reaction Engineering, 3rd Ed, Wiley & Sons, NJ, USA.

Ma, W., Zhao, Y. Q. and Kearney, P. (2007). "A study of dual polymer conditioning of aluminum-based drinking water treatment residual" *Journal of Environmental Science and Health, Part A: Toxic/Hazardous Substances and Environmental Engineering* 42:7, 961-8.

Maidment, D. R. (1993). *Handbook of hydrology*. New York: McGraw-Hill.

Mangleson, K. and Watters, G. (1972). "Treatment Efficiency of Waste Stabilization Ponds." *Journal of the Sanitary Engineering Division*: 407–25.

McLachaln, D., Bergeron, C., Smith, J. E., Boomer, D. and Rifat, J. L. (1996). "Risk for Neuropathologically Confirmed Alzheimer's Disease and Residual Aluminum in Municipal Drinking Water Employing Weighted Residential Histories." *Neurology* 46: 401–5.

Mitsch, W. J. and Wise, K. M. (1998). "Water Quality, Fate of Metals, and Predictive Model Validation of a Constructed Wetland Treating Acid Mine Drainage." *Water Research* 32: 1888–1900.

Moody, G. M. (2012). "Handbook of Industrial Water Soluble Polymer": 1–40.

Muisa, N., Hoko, Z. and Chifamba, P. (2011). "Impacts of Alum Residues From Morton Jaffray Water Works on Water Quality and Fish, Harare, Zimbabwe." *Physics and Chemistry of the Earth* 36 (14-15): 853–64. doi:10.1016/j.pce.2011.07.047.

Muttamara, S., and Puetpaiboon, U. (1997). Roles of Baffles in Waste Stabilization Ponds. *Water Science and Technology: a Journal of the International Association on Water Pollution Research* 35 (8): 275-83.

MWH. (2005). Water treatment principles and design. Hoboken, NJ, USA: John Wiley and Sons.

Nalekwajko, C. and Paul, B. (1985). "Effects of Manipulations of Aluminum Concentration and pH on Phosphate Uptake and Photosynthesis of Planktonic Communities in Two Precambrian Shield Lakes." *Canadian Journal Fisheries and Aquatic Sciences* 42: 1946–53.

Novak, J. and Langford, M. (1977). "The Use of Polymers for Improving Chemical Sludge Dewatering on Sand Beds." *Journal AWWA*: 106–11.

Nova Scotia Environment. (1994). Interpretation: c1. s3(bg). Available online at: http://nslegislature.ca/legc/statutes/envromnt.htm

Nova Scotia Environment. (2011). "Nova Scotia Wetland Conservation Policy": 1–26.

Nova Scotia Environment. (2012). "Nova Scotia Treatment Standards for Municipal

Drinking Water Systems”: 1–104.

NRC. (2004). “Optimization of Lagoon Operation.”

Parent, L. and Campbell, P. (1994). “Aluminum Bioavailability to the Green Alga Chlorella Pyrenoidosa in Acidified Synthetic Soft Water.” *Environmental Toxicology and Chemistry* 13: 587–98.

Pedahzur, R., Nasser, A. M., Dor, I., Fattal, B., and Shuval, H. I. (1993). The effect of baffle installation on the performance of a single-cell stabilization pond. *Water Science & Technology*, *27*(7-8), 45-52.

Persson, J. (2000). “The Hydraulic Performance of Ponds of Various Layouts.” *Urban Water* 2 (3): 243–50. doi:10.1016/S1462-0758(00)00059-5.

Peterson, E. L, Harris, J. A. and Wadhaw, L. C. (2000). “CFD Modelling Pond Dynamic Processes.” *Aquacultural Engineering* 23: 61–93.

Pettersson, A., Hallbom, L. and Bergman, B. (1988). “Aluminum Effects on Uptake and Metabolism of Phosphorus by Cyanobacterium Anabaene Cylindrica.” *Plant Physiology* 86: 112–16.

Poleo, A. (1995). “Aluminum Polymerization- a Mechanism of Acute Toxicity of Aqueous Alumimiun to Fish.” *Aquatic Toxicology* 31: 347–56.

Sah, L., Rousseau, D., Hooijmans, C. and Lens, P. (2011). “3D Model for a Secondary Facultative Pond.” *Ecological Modelling* 222 (9): 1592–1603. doi:10.1016/j.ecolmodel.2011.02.021.

Salter, H., Ta, C., Ouki, S. K. and Williams, S. C. (2000). “Three-Dimensional Computational Fluid Dynamic Modelling of a Facultative Lagoon.” *Water Science & Technology* 42: 335–42.

Shilton, A. N. and Harrison, J. (2003). “Guidelines for the Hydraulic Design of Waste Stabilization Ponds”: 1–78.

Shilton A. N. (2005). *Pond treatment technology*. London: IWA Pub.

Sookhak L. K. (2013). “A Note on Baffle Orientation in Long Ponds.” *Journal of Environmental Informatics* 2: 136–41.

Sposito, G. (1996). *The environmental chemistry of aluminum*. Boca Raton, FL: Lewis Publishers.

Stamou, A. I. (2008). "Improving the Hydraulic Efficiency of Water Process Tanks Using CFD Models." *Chemical Engineering and Processing: Process Intensification* 47 (8): 1179–89. doi:10.1016/j.cep.2007.02.033.

Tanner, C. C. and Headley, T. (2011). "Ecological Engineering." *Ecological Engineering* 37 (3). Elsevier B.V.: 474–86. doi:10.1016/j.ecoleng.2010.12.012.

Teixeira, E. C., and Do, N. S. R. (2008). Performance assessment of hydraulic efficiency indexes. *Journal of Environmental Engineering, 134,* 10, 851-859.

Templeton, M., Hofmann, R. and Andrews, R. (2006). "Case Study Comparisons of Computational Fluid Dynamics (CFD) Modeling Versus Tracer Testing for Determining Clearwell Residence Times in Drinking Water Treatment." *Journal of Environmental Engineering and Science* 5 (6): 529–36. doi:10.1139/s06-007.

Thackston, Edward L, Shields, F. D. and Schroader, P. R. (1987). "Residence Time Distributions of Shallow Basins." *Journal of Environment Engineering* 113: 1319–32.

Thirumurthi, D. (1969). "A Break-Through in the Tracer Studies of Sedimentation Tanks" *Journal (Water Pollution Control Federation)* 41: 405–17.

Thirumurthi, D. (1974). "Design Criteria for Waste Stabilization" *Water Pollution Control Federation* 46: 2094–2106.

USEPA. (2001). "Filter Backwash Recycling Rule." http://water.epa.gov/lawsregs/rulesregs/sdwa/mdbp/upload/2003_04_25_mdbp_filter backwash_fbrr_techguidance.pdf.

USEPA. (2005). Current National Recommended Water Quality Criteria. http://www.epa.gov/waterscience/criteria/wqcriteria.html#nonpriority Accessed March 12, 2014.

Vadasarukkai, Y. S. (2010). "Assessment of Hydraulic Performance of Flocculation Processes Using Computational Fluid Dynamics (CFD)." Dalhousie University.

Vadasarukkai, Y. S. and Gagnon, G. A. (2010). "Determination of Conventional Velocity Gradient (G) Using CFD Technique for a Pilot-Scale Flocculation System." *Journal of Water Supply: Research and Technology—AQUA* 8: 459–70.

Vadasarukkai, Y. S., Gagnon, G. A., Campbell, R. and Clark. S. (2011). "Assessment of Hydraulic Flocculation Processes Using CFD." *Journal AWWA* 103 (11): 66–80.

Vega, G. P., Pena, M. R., Ramirez, C., and Mara, D. D. (2003). Application of CFD modelling to study the hydrodynamics of various anaerobic pond configurations.*Water Science and Technology : a Journal of the International Association on Water Pollution Research, 48,* 2, 163-71.

Versteeg, H. K., and Malalasekera, W. (2007). *An introduction to computational fluid dynamics: The finite volume method*. 2nd Edition. Harlow, Essex, England: New York.

Walt, J. and Haarhoff, J. (2000). "Is CFD an Engineering Tool or Toy?." WISA 2000, 1–19. Sun City.

Wang, H. and Falconer, R. A. (1997). "Simulating Disinfection Process in Chlorine Contact Tanks Using Various Turbulence Models and High-Order Accurate Difference Schemes." *Water Research* 32: 1529–43.

Watters, G. (1972). "The Hydraulics of Waste Stabilization Ponds." 18. http://digitalcommons.usu.edu/water_rep/18.

Wauer, G. and Teien, H. C. (2010). "Risk of Acute Toxicity for Fish During Aluminum Application to Hardwater Lakes." *Science of the Total Environment* 408 (19): 4020–25. doi:10.1016/j.scitotenv.2010.05.033.

Wauer, G., Heckemann, H. J. and Koschel, R. (2004). "Analysis of Toxic Aluminum Species in Natural Waters." *Microchimica Acta* 146 (2): 149–54. doi:10.1007/s00604-004-0198-2.

Wieder, R. K. (1989). "A Survey of Constructed Wetlands for Acid Coal Mine Drainage Treatment in the Eastern United States." *Wetlands* 9: 299–315.

Wieder, R. K., Linton, M. N. and Heston. K. P. (1990). "Laboratory mesocosm studies of Fe, Al, Mn, Ca, and Mg Dynamics in wetlands exposed to synthetic acid Coalmine drainage." *Water, Air, & Soil Pollution* 51: 181–96.

Williams, P. A. (2007). *Handbook of industrial water-soluble polymers*. Oxford: Blackwell Pub.

Wood, C. M., McDonald, D. G., Ingersoll, G. C., Mount, R. D., Johannsson, O. E., Landsberger, S. and Berman. H. L. (1990). "Effects of Water Acidity, Calcium, and Aluminum on Whole Ions of Brook Trout (Salveinus Fontinalis) Continously Exposed From Fertilization to Swim-Up: a Study by Instrumental Neutron Activation Analysis." *Canadian Journal Fisheries and Aquatic Sciences* 47: 1593–1603.

Wood, C. M., Playle, R. C., Simons, B. P., Goss, G. G. and McDonald, D. G. (1988). "Blood Gases, Acid-Base Status, Ions, and Hematology in Adult Brook Trout (Salvelinus Fontinalis) Under Acid/Aluminum Exposure." *Canadian Journal Fisheries and Aquatic Sciences* 45: 1575–86.

Wood, M. G., Howes T., Keller, J. and Johns, M. R. (1998). "Two Dimensional Computational Fluid Dynamic Models for Waste Stabilization Ponds." *Water Research* 32 (3). Elsevier: 958–63.

Wu, B. and Chen, Z. (2011). "An integrated physical and biological model for anaerobic lagoons." *Bioresource Technology* 102: 5032-5038.

Yapijakis, C., (1982). "Direct Filtration: Polymer in Backwash Serves Dual Purpose." *Journal AWWA* 74 (8): 426-428.

YSI Inc. (2009). "YSI 6-Series Multiparameter Water Quality Sondes User Manual": 1–377.

Zhang, J. (2006). "An Integrated Design Approach for Improving Drinking Water Ozone Disinfection Treatment Based on Computational Fluid Dynamics." University of Waterloo.

Zhao, Y. Q. (2003). Correlations between floc physical properties and optimum polymer dosage in alum sludge conditioning and dewatering. *Chemical Engineering Journal* 92: 227-35.

Zhao, Y Q. (2004). "Settling Behaviour of Polymer Flocculated Water-Treatment Sludge II: Effects of Floc Structure and Floc Packing." *Separation and Purification Technology* 35 (3): 175–83. doi:10.1016/S1383-5866(03)00133-3.

Appendix A. RATING CURVES

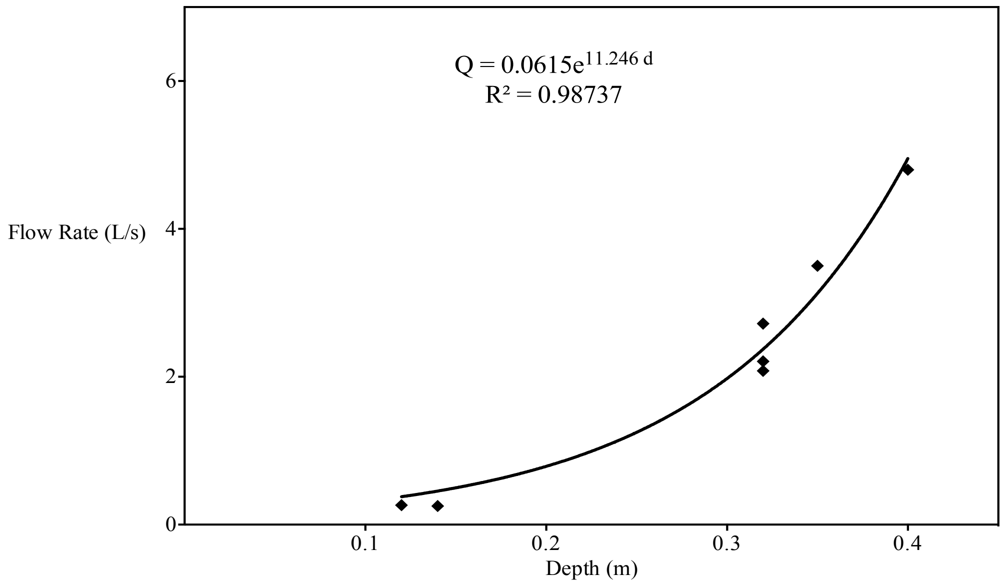

FIGURE A-1 DISCHARGE RATING CURVE FOR THE NATURAL WETLAND INLET AT THE LAGOON DISCHARGE

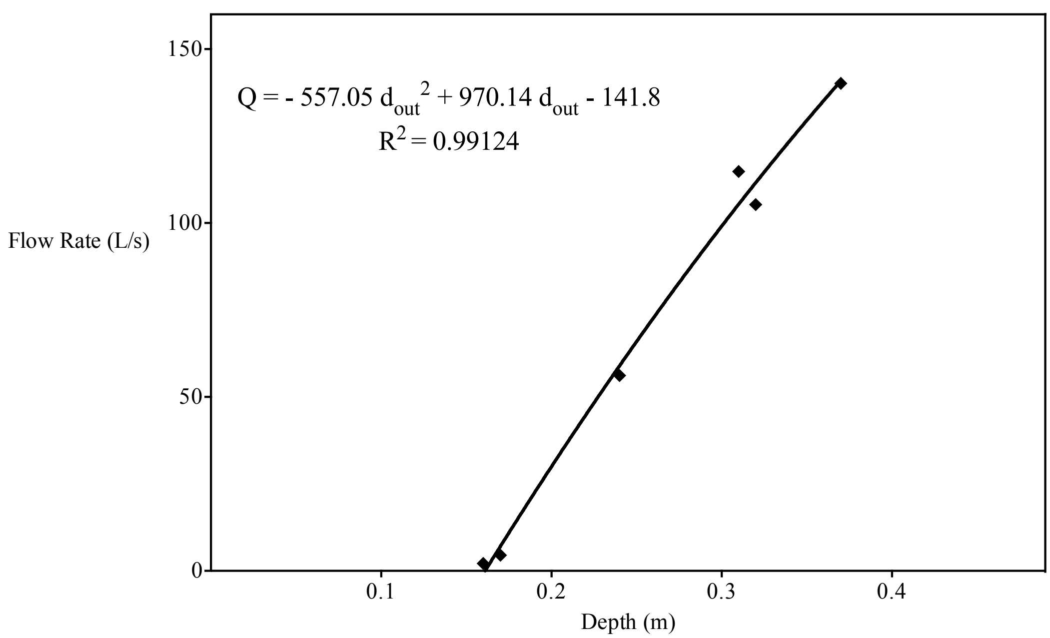

FIGURE A-2 DISCHARGE RATING CURVE FOR THE WETLAND OUTLET INTO LITTLE POCKWOCK LAKE.

Appendix B. LAGOON DISCHARGE

The following two figures show the difference in flows between the high flow, Figure B-2, during a backwash event, and low flow, Figure B-1 the regular discharge during the rest of the day.

FIGURE B-1 LAGOON DISCHARGE INTO THE NATURAL WETLANDS DURING TYPICAL LOW FLOW CONDITIONS.

FIGURE B-2 LAGOON DISCHARGE INTO THE NATURAL WETLANDS DURING A BACKWASH EVENT

Appendix C. POCKWOCK LAKE ALUMINUM SEASONAL TREND

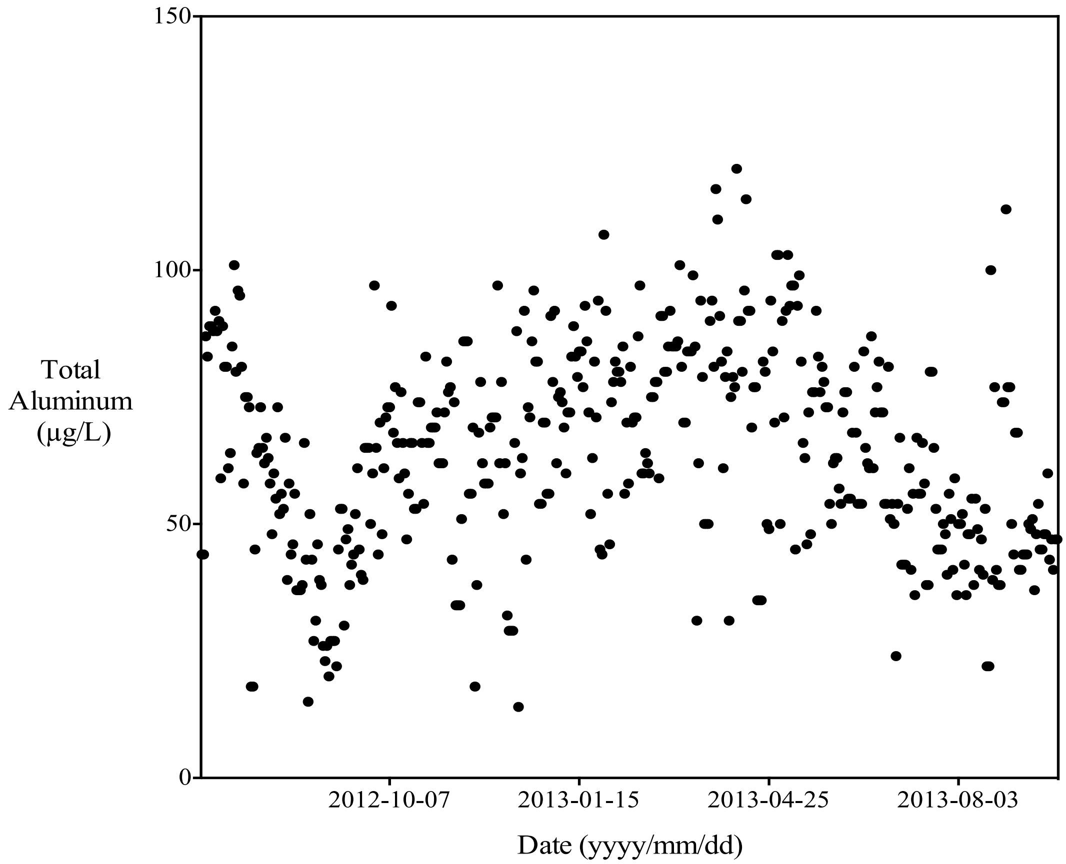

FIGURE C-1 POCKWOCK LAKE ALUMINUM CONCENTRATIONS ANALYZED BY JDKWTP STAFF

Appendix D. MESH DENSITY INDEPENDENCE

Using the exact same model setup and varying the grid sizes various parameters were compared to determine the independence of the model. The outlet velocity, the volume averaged kinetic and dissipation energies were compared against three grid sizes (see Figure C-1 and C-2). As there was little difference between the flow parameters the grid size with 337508 cells was chosen for the experiments.

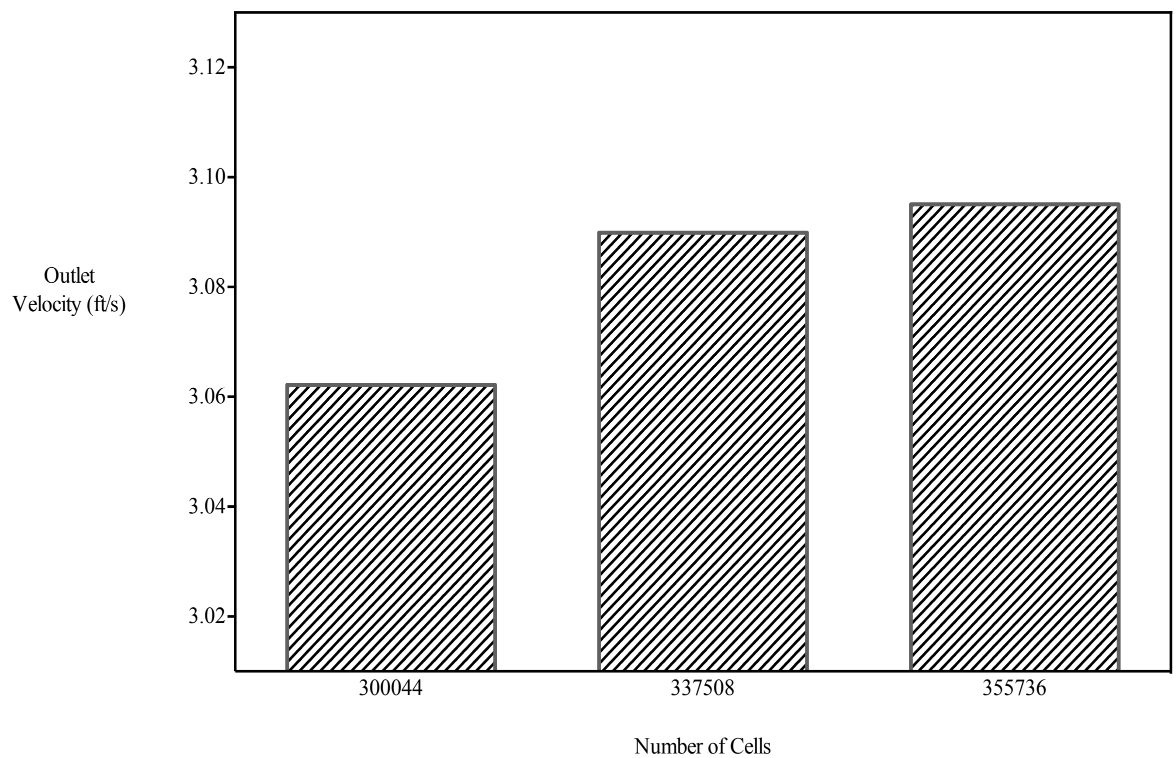

FIGURE D-1 OUTLET VELOCITY FOR VARIOUS GRID SIZES.

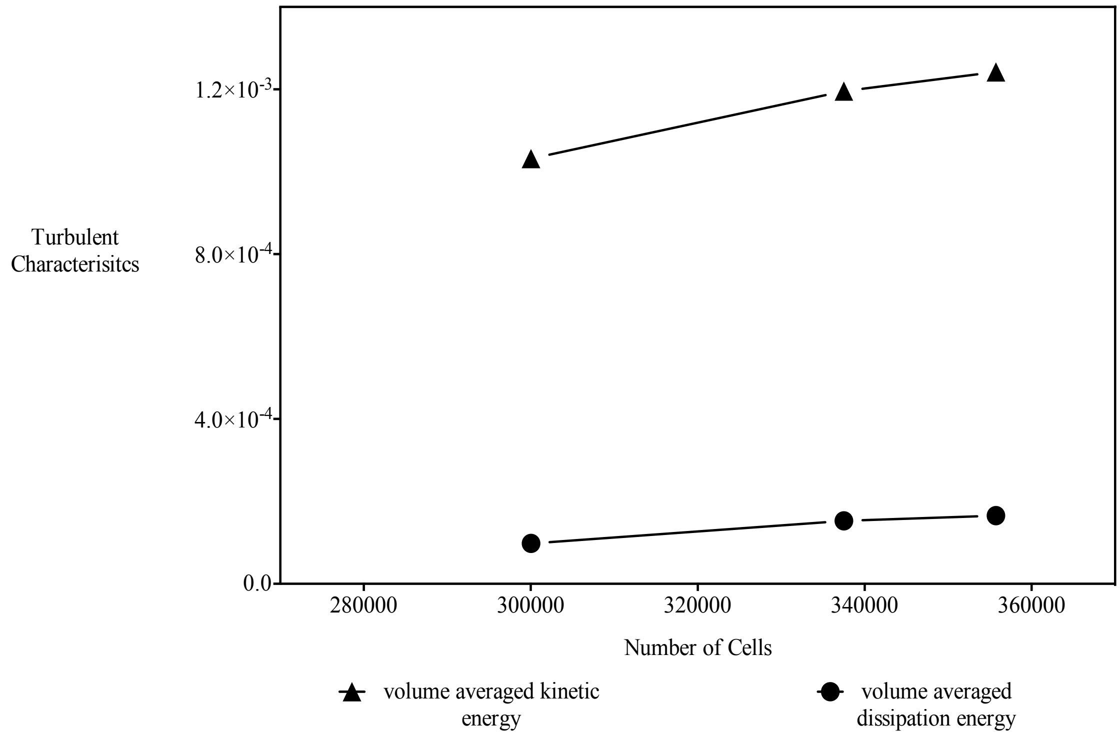

FIGURE D-2 VOLUME AVERAGED KINETIC AND DISSIPATION ENERGY FOR THE VARIOUS GRID SIZES.

For the various baffle arrangements the mesh-density sensitivity analysis was performed and the optimal grid sizes are summarized in Table D-1. The number of cells is proportional with the velocity of the flow in each model. The two-baffle model had the longest retention time or slowest flow and the least number of cells.

TABLE D-1 OPTIMIZED GRID SIZE FOR EACH BAFFLE ARRANGEMENT

Number of Baffles	Number of Cells	Cell Volume (m^3)
No Baffles	337,508	0.0415
One Baffle	343,963	0.0407
Two Baffles	231,115	0.0606
Three Baffles	301,544	0.0464

Appendix E. USER DEFINED FUNCTION FOR UNSTEADY CFD FLOWS

```
#include "udf.h"
DEFINE_PROFILE (velocity_magnitude, t, i)
{
real velocity;                                  /*Inlet velocity condition*/
real the_current_time;
face_t f;
the_current_time = CURRENT_TIME;                /*High flow conditions to represent
if ((the_current_time>=100) &&                  a 30 minute backwash*/
(the_current_time<1900))
{
velocity=0.8924544;
}
else if ((the_current_time>=7048) &&
(the_current_time<8848))
{
velocity=0.8924544;
}
else if ((the_current_time>=12448) &&
(the_current_time<14248))
{
velocity=0.8924544;
}
else if ((the_current_time>=17848) &&
(the_current_time<19648))
{
velocity=0.8924544;
}
else if ((the_current_time>=23248) &&
(the_current_time<25048))
{
velocity=0.8924544;
}
else
{                                               /*Low flow conditions to represent
velocity=0;                                     when there is no backwash
}                                               occurring*/
begin_f_loop(f,t)
{
F_PROFILE(f,t,i) = velocity;
}
end_f_loop(f,t)
}
```