PASSIVE TREATMENT OF ACID MINE DRAINAGE FROM A SULPHIDIC COALFIELD

by

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

at

Dalhousie University
Halifax, Nova Scotia
August 2023

Dalhousie University is located in Mi'kma'ki, the
ancestral and unceded territory of the Mi'kmaq.
We are all Treaty people.

DEDICATION PAGE

This thesis is dedicated to my husband, Justin, and our children, Nyanza, Lachlan, and baby to be whose unconditional love, absolute support, and personal sacrifices have enabled me to pursue and complete this research.

This work is also dedicated to my late sister, Mandy, whose unwavering heart, spunk, and positive attitude shaped and inspired me. I miss you so much.

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ABSTRACT

An ongoing concern is acid mine drainage (AMD) from abandoned sulphidic mines in the Sydney coalfield, which threatens local water bodies and fishing industries if left untreated. This work focuses on evaluating the performance of two semi-passive treatment systems at the Neville Street facility that treat this AMD, which is acidic and contains heavy metals such as iron, manganese, and aluminum. These treatment systems include caustic soda addition, aeration cascades, settling ponds, and constructed wetlands. Current monthly sampling by a local government agency may or may not represent worst-case scenarios as sampling may not coincide with conditions that could impact performance, such as peak loading. This work was divided into three research objectives: (1) characterize system performance during peak loading, (2) statistically evaluate treatment performance, and (3) develop a treatment performance model of the constructed wetlands.

Analysis of sampling data during peak loading events indicates that both systems are effectively removing iron from the mine water with effluent readings meeting federal and provincial guidelines. However, inlet iron concentrations have been increasing over time for both systems; thus, exceedances of guideline limits could become a concern in the future. Due to newly adopted federal and provincial guidelines for manganese and sulphate, exceedances of these guideline limits have been or are likely to become a concern in the near future. Modeling of wetland treatment performance using a modified tanks-in-series (TIS) model revealed the necessity for site-specific calibration of model constants. Areal rate constants at 20 °C for iron were developed for both systems based on design and average operating conditions. The design areal rate constants at 20 °C for the two systems were 2,325 m/year and 1,380 m/year; the average areal rate constants at 20 °C were 1,930 m/year and 560 m/year.

LIST OF ABBREVIATIONS AND SYMBOLS USED

&	And
℃	Degree Celsius
%	Percent
α	Transpiration Fraction
ΔP	Static Pressure
Θ	Temperature Correction Factor
µg/L	Microgram per Liter
ρ	Density
τ	Mean Residence Time or Mean Detention Time
A	Area
a.k.a.	Also Known As
Al^{3+}	Trivalent Aluminium Cation
$Al(OH)_3$	Aluminium Hydroxide
AMD	Acid Mine Drainage
ANOVA	Analysis of Variance
A_t	Area of Tank
b	Channel Width
BCMOECCS	British Columbia Ministry of Environment and Climate Change Strategy
C	Effluent Concentration
C^*	Background Concentration
$CaCO_3$	Calcium Carbonate
CAD	Canadian Dollars
CB	Cadegan Brook
CBDC	Cape Breton Development Corporation
CBU	Cape Breton University
CCME	Canadian Council of Ministers of the Environment
C_i	Influent Concentration
C_{in}	Inlet Concentration
cm	Centimeter
cm/day	Centimeter per Day
CO_2	Carbon Dioxide
C_{out}	Outlet Concentration
CPI	Cascade Pond Inlet or Treatment System Inlet
CWQG	Canadian Water Quality Guidelines
DEVCO	Cape Breton Development Corporation
d_w	Average Wetland Depth
E	Evaporation
ECBC	Enterprise Cape Breton Corporation
ECCC	Environment and Climate Change Canada
e.g.	*Exempli gratia*
ET	Evapotranspiration
et al.	*Et alia*
etc.	*Et cetera*

EQS	Environmental Quality Standards
ES	Event Sampling
Fe	Iron
Fe (II), Fe^{2+}	Ferrous Iron
Fe (III), Fe^{3+}	Ferric Iron
$Fe(OH)_x$	Iron Hydroxide
FeS_2	Pyrite
FNU	Formazin Nephelometric Unit
FWAL	Freshwater Aquatic Life
FWS	Free Water Surface
Ft	Feet
g	Gravitational Constant
h	Average Water Depth
H^+	Hydrogen Cation
H_2O	Water
HCO_3^-	Bicarbonate
hr	Hour
HRT	Hydraulic Retention Time
HSSF	Horizontal Subsurface Flow
I	Infiltration
IAP	Ion Activity Product
i.e.	*Id est*
k	First Order Areal Rate Constant
k_{20}	First Order Areal Rate Constant at 20 °C
kg/m^3	Kilogram per Meter Cubed
km	Kilometer
km^2	Kilometer Squared
k_T	First Order Areal Rate Constant Corrected to T °C
L	Liter
L	Length
L/min	Liter per Minute
Ltd.	Limited
m	Meter
m^2	Meter Squared
m^3	Meter Cubed
m^3/day	Meter Cubed per Day
m^3/s	Meter Cubed per Second
m/day	Meter per Day
m/s	Meter per Second
m/s^2	Meter per Second Squared
m/year	Meter per Year
mg	Milligram
mg/L	Milligram per Liter
MinWaReP	Mine Water Research Program
mL	Milliliter
Mn^{2+}	Divalent Manganese Cation

Mn^{4+}	Tetravalent Manganese Cation
MnO_2	Manganese Dioxide
mV	Millivolt
N	Number of Tanks-in-Series
N/m^2	Newton per Meter Squared
NA	Not Applicable
NaOH	Sodium Hydroxide or Caustic Soda
NG	No Guidelines
NRCan	Natural Resources Canada
NS	Nova Scotia
NSDOE	Nova Scotia Department of Energy
NSECC	Nova Scotia Department of Environment and Climate Change
NSE	Nova Scotia Environment
NSW	Neville St. Wellfield
NTU	Nephelometric Turbidity Unit
O_2	Oxygen
OWL	Outlet of Constructed Wetland
ORP	Oxidation-Reduction Potential
OSP	Outlet of Settling Pond
P	Precipitation
P_{atm}	Atmospheric Pressure
pH	$-\log_{10}[H^+]$
Pt/Co	Platinum-Cobalt Scale
PTS	Passive Treatment System
PS&PC	Public Services and Procurement Canada
P_w	Water Pressure at Channel Bottom
PWGSC	Public Works and Government Services Canada
Q	Flow Rate
Q_{in}	Inlet Flow Rate
Q_{out}	Outlet Flow Rate
Q_{ws}	Watershed Flow Rate
RDL	Reliable Detection Limit
SHE	Standard Hydrogen Electrode
SO_4^{2-}	Sulphate
SP1	System 1
SP2	System 2
SPD	Settling Pond Discharge
SSF	Subsurface Flow
St.	Street
T	Temperature
t	Transpiration
TCU	True Colour Unit
TIS	Tanks-in-Series
US EPA	United States Environmental Protection Agency
usgpm	United States Gallon per Minute
W	Width

WMO	World Meteorological Organization
v	Velocity
V	Volume
VF	Vertical Flow

ACKNOWLEDGEMENTS

First and foremost, I would like to sincerely thank my supervisors, Dr. Rob Jamieson and Dr. Allison Mackie. From the initial thesis idea to final submission, you both have been unwavering in your support and guidance.

Rob: Thank you for your understanding and flexibility, which allowed me to be able to pursue my master's degree from the other end of the province, through multiple pregnancies, with small children, and while working full time. You were always there to answer any question or address any concern I might have. Your guidance allowed me to navigate the master process with ease and enabled me to be able to pivot my research in response to barriers created by the pandemic. I admire your even-keeled approach to supervising, which empowers your students, and it has inspired me to change my own approach to teaching. It was a pleasure to work with you.

Allison: Thank you for providing me with the opportunity to do this work by helping me make the necessary connections at Dalhousie University and by providing funding for my research. Thank you for your endless support throughout the entire process from training me on lab equipment to being out in the proverbial trenches with me so that I could complete my field work. You were my steadfast sounding board and my lighthouse in the fog, helping me navigate the field of hands-on research. I am deeply grateful for your friendship and mentorship.

Additionally, I would like to express my gratitude to the members of my supervisory committee, Dr. Martin Mkandawire and Dr. Margaret Walsh, for your committed support and for providing me with new insights on my research. Martin, my research would not have been possible without you inspiring the topic and sharing your vast knowledge and experience on the topic. Thank you for going out of your way to help me so many times.

I would like to acknowledge the funding sources that made this research possible: Cape Breton University (CBU) 2019 Start-Up Research Grant and CBU 2019 Research, Innovation, Scholarship, and Exploration Grant. Furthermore, I want to thank Kathy Isaac for her work as a student researcher including helping me in the field, in the lab,

and with data processing. You are going to become an incredible engineer and I cannot wait to see what you accomplish in your career.

This work would not have been possible without the support from Joseph MacPhee at Public Services and Procurement Canada. Thank you for allowing me access to the site for sampling and for sharing critical system information, operational data, and technical insight.

Last, but not least, I want to thank my family for their unfailing support. A special thank you goes out to my mother, Louise D’Orsay, who helped take care of my children countless times so that I could work on my research. The most important thank you goes out to my husband, Justin Spillman. You are my biggest supporter. From easing my workload by taking care of our children and doing more than your share of domestic chores to helping me in the field by helping me install sensors and by taking field photos with a drone, you have been consistently there for me. You are amazing and I am so lucky to have you. Thank you for your constant encouragement, endless support, and generous sacrifices. I would not have been able to do this without you.

CHAPTER 1 INTRODUCTION

With the largest coal reserves in eastern Canada located underneath the ocean in the Cape Breton area (Hacquebard, 1993) and after nearly three hundred years of coal mining in the region, the closure of the last underground mine in the Sydney coalfield in 2001 (Natural Resources Canada [NRCan], 2020) had a huge impact on the local economy. In addition to a loss of a vital industry in the area, another concern in the form of acid mine drainage (AMD) rose up from the depths of the mines, threatening the local water bodies and fishing industry.

1.1 Background

Remediation of the coal mining operations in Cape Breton was and continues to be a massive endeavor. The effluent from the Sydney coalfield mines is acidic and contains heavy metals, which are persistent toxins that bioaccumulate and biomagnify. The remediation effort entails the continuous treatment of six billion liters per year of AMD with four passive treatment plants and one active treatment plant in the Sydney area (Government of Canada, 2016).

This thesis focused on the Neville St. Passive Treatment System (PTS), which is located in Reserve Mines, Nova Scotia, and treats AMD from 10 interconnected, abandoned coal mines in the Sydney coalfield, which are commonly referred to as the 1B hydraulic system. This facility includes two semi-passive treatment systems, the second of which was added when the facility was expanded in 2016 (Government of Canada, 2016). This expansion was necessary as it was determined that the initial configuration of the Neville St. PTS was inadequate to fully treat the AMD during peak loading.

Peak loading occurs in the late fall and early spring when there is a tendency for higher rainfall and snowmelt. In most passive treatment systems, additional water inputs have a diluting effect on contaminant concentrations, which counteracts the low residence time associated with higher hydraulic loading; the resulting overall effect is an improvement in system performance as lower contaminant concentrations in the system effluent are generally achieved. However, in this case, the additional water input results in increased

AMD generation, which causes the system to experience the upper range of contaminant concentrations simultaneously with short residence times. Thus, the contaminant removal performance of the Neville St. PTS is the poorest during peak loading. The goal of this work was to evaluate the performance of this facility post-expansion to ensure adequate treatment of raw mine water during peak loading events.

1.2 Thesis Objectives

The aim of this work was to conduct an evaluation of the post-expansion performance of the Neville St. PTS in treating AMD during peak loading events to ensure that contaminant removal meets federal and provincial guidelines and regulations prior to release to the environment. An initial review of historical sampling data collected by Public Services and Procurement Canada (PS&PC) was conducted. The purpose of analyzing this data was to identify contaminants of concern for which the facility has a history of poor removal, to determine overarching performance trends, and to identify gaps in the current sampling program, which were key to informing the development of the framework for this thesis. The work in this thesis was divided into three distinct research objectives: (1) characterize system performance during peak loading, (2) statistically evaluate treatment performance, and (3) develop a treatment performance model of wetlands. These three objectives were progressive in nature, meaning that outputs from earlier objectives were incorporated into later objectives as inputs.

1) Characterize system performance during peak loading

   Contaminant reduction performance at the Neville St. PTS is the weakest during peak loading. Due to the low frequency of sampling, PS&PC sampling data may not represent the performance of the facility during peak loading events. Event sampling was conducted during peak loading events and these samples were analyzed for key performance indicators in order to evaluate the performance of both systems at the Neville St. PTS, with a particular emphasis on contaminant removal.

2) Statistically evaluate treatment performance

Sample readings of various key variables at locations across each system were compared using statistical analysis to determine if significant removal of contaminants is taking place within both systems at the Neville St. PTS. In addition, sample values of key performance parameters from multiple points in both systems were compared to evaluate if there are significant differences in treatment performance between operational years. Historical data provided by PS&PC and data collected as part of this project were incorporated into the statistical analyses.

3) Develop a treatment performance model of wetlands

   Based on historical data collected by PS&PC and event sampling data, a modified tanks-in-series (TIS) model with first order rate constants to incorporate contaminant attenuation mechanisms was developed and applied to simulate the removal performance of the two constructed wetlands at the Neville St. PTS for iron, a key contaminant of concern.

Collectively, the data and the insights gathered from the above three steps were used to evaluate if the Neville St. PTS is able to remove sufficient amounts of contaminants during all operational conditions, including peak loading. This information was also employed in the development of recommendations stemming from this thesis.

1.3 Thesis Structure

This thesis is organized according to the following structure:

Chapter 1 Introduction (1) establishes the topic of the thesis with a high-level summary, (2) describes the scope of the thesis by explaining the objectives, and (3) presents an overview of the thesis structure while summarizing each chapter in a manner that demonstrates how it contributes to the main objective of the thesis.

Chapter 2 Literature Review explores the current state of understanding of AMD treatment, with a particular focus on the application of constructed wetlands to treat AMD similar to that generated by the Sydney coalfield, while supplying necessary background information to provide context and relevance. The literature review includes a breakdown of why AMD is an issue, a brief history of the treatment of AMD from the Sydney coalfield, an explanation of the Neville St. PTS process and operation, key details about of the current monitoring program, and a review of modern modelling approaches for wetland treatment performance. The purpose of this literature review is to ensure the scope of this thesis was informed by industry best practices and to provide a theoretical framework for the thesis based on identified performance gaps within the Neville St. PTS process.

Chapter 3 Methodology is divided into three research objectives: (1) characterize system performance during peak loading, (2) statistically evaluate treatment performance, and (3) develop a treatment performance model of wetlands. The methodologies of each of the three research objectives are addressed separately, including a breakdown of what work was carried out for each objective and why.

Chapter 4 Results and Discussion begins with a review of historical sampling data from PS&PC to provide context for identified monitoring gaps, which was the basis for the work undergone in this thesis. This section is similarly divided into three sequential sections based on the same three research objectives discussed in the methodology section. Each section reviews the results of the associated objective in detail, providing context and relevance.

Chapter 5 Conclusion summarizes the findings of the entire thesis and provides recommendations derived based on the work completed in this study.

CHAPTER 2 LITERATURE REVIEW

Acid mine drainage is a legacy issue in the former coal mining towns throughout Cape Breton as a direct result of nearly three centuries of coal mining in the region with the first coal mine opening in Port Morien in 1720 (Nova Scotia Archives, 2023). With the largest coal reserves in eastern Canada located underneath the ocean and littoral land in the Cape Breton area (Hacquebard, 1993), the Sydney coalfield was an essential part of the Canadian economy for centuries, contributing 40 % of the Canadian coal production in 1912 (Miner's Museum, 2023). During this time, coal from Nova Scotia supplied the raw energy that allowed for the development of modern industry not only locally, but in Central Canada as well (Miner's Museum, 2023). Outside of a couple of small project exceptions, such as the seven year Surface Coal Mine and Reclamation Project at the Prince Mine site that was approved in 2006 (Nova Scotia Department of Environment and Climate Change [NSECC], 2017a) and the recent opening of Donkin Mine in 2017 (NSECC, 2017b) with its subsequent closure in March of 2020 (Morien Resources Corperation, 2023) and reopening in December of 2022 (Government of Nova Scotia, 2022), there has been no coal production from mines in Cape Breton since the closure of the last Cape Breton Development Corporation mine in 2001 (NRCan, 2020). Even though the coal industry is a shadow of its former glory, the environmental impacts persist as a concern in the form of AMD.

This chapter (1) explains why AMD is a concern, (2) provides a brief overview of the treatment history of AMD in the Sydney coalfield area, (3) describes the AMD treatment process at Neville St. PTS including current performance gaps, (4) provides details about the existing monitoring program including possible deficiencies, and (5) discusses how wetlands can be used to treat AMD and reviews current methods of modelling wetland treatment performance.

2.1 Acid Mine Drainage (AMD)

Acid mine drainage is characterized by low pH and high concentrations of sulphates and heavy metals. Acid mine drainage is effluent released from mines located largely in areas that contain sulphide minerals; iron sulphides are the prevalent source of AMD, but other metals sulphides can also generate AMD (Skousen et al., 2017). These mineral deposits are inert until exposed to oxygen through the mining process (Kefeni et al., 2017). The exposure of this ore to oxygen causes these sulphide minerals to become oxidized. These oxidized minerals come in contact with water as the mines naturally fill with water. Upon the exposure to water, the oxidized ore dissolves into the water, resulting in the formation of sulphuric acid, sulphates, and metal ions (Skousen et al., 2019). The associated rise in water acidity increases the solubility of the sulphide minerals, releasing more acid and metal contaminants (Dold, 2014). This cycle of reactions repeats quickly and is self-perpetuating until the oxidized sulphide minerals are consumed (Ford, 2003). The mines continue to fill with water until it overflows, discharging AMD which contaminates local surface and groundwater if not properly treated before release. Water levels in mines fluctuate depending on weather patterns, with the rising and falling water level exposing new ore to oxygen and water, continuously generating AMD as a result.

The chemical reactions associated with AMD can be a naturally occurring event through the normal weathering of sulphide minerals; however, mining activities augment the amount of AMD generation by increasing the amount of sulphide minerals being exposed to the elements (Muhammad et al., 2015). Acid mine drainage is generated in and discharged by active, inactive, and abandoned mines (Acharya & Kharel, 2020). Active mines typically have dewatering pumps, and this reduces AMD generation, but does not eliminate it completely. Dewatering activities generally cease once mines are no longer active. Contaminated and acidic effluent is also produced by mining waste rock piles upon exposure to water and oxygen (Dold, 2014). This form of AMD discharge is likewise a concern and is known as acid rock drainage (ARD). The release of acidic effluent, whether generated by mines or waste rock, is a consequence of land being disturbed by anthropogenic activities and can continue indefinitely, posing risk to the environment for many years. Some experts estimate that AMD can be produced for centuries (or even millennia) post termination of the activity that first caused the AMD formation (Ramasamy et al., 2018).

2.1.1 Chemistry of AMD Formation

AMD can have a pH which varies widely but it typically ranges from 2 to 8 (Skousen et al., 2019). The most common dissolved cations found in AMD are Group II and transition metals such as iron ($Fe^{2+,3+}$) and Aluminum (Al^{3+}); the most common dissolved anions in AMD are sulphate (SO_4^{2-}) and bicarbonate (HCO_3^-) (Skousen et al., 2019). The alkalinity of AMD can vary quite considerably depending on the amount of acid neutralizing minerals (generally carbonates) present within local rock. Thus, some AMD is alkaline in nature, but is still referred to as AMD due to its means of formation and its high concentration of contaminants (Skousen et al., 2019).

Iron sulphide is the most common sulphide mineral involved with AMD formation; thus, it is common for AMD to contain dissolved iron (Akcil & Koldas, 2006). Pyrite, also often known as fool's gold, is a common form of iron sulphide and has the chemical formula of FeS_2 [iron (II) disulphide]. The oxidation of pyrite results in the formation of ferrous iron (Fe^{2+}), sulphates (SO_4^{2-}), and hydrogen cations as shown in Equation 2.1:

$$FeS_{2(s)} + {}^{7}/_{2}\,O_2 + H_2O \rightarrow Fe^{2+} + 2SO_4^{2-} + 2H^+ \qquad (2.1)$$

The above reaction results in a reduction in pH, which promotes the dissolution of other minerals, many of which contain metals. In sufficiently oxidizing environments, ferrous iron (Fe^{2+}) can also react with an oxidizer, such as oxygen, to form ferric iron (Fe^{3+}), which releases more hydrogen cations and decreases pH further. The production of AMD often involves the oxidation of other metal sulphides, which forms metal cations and sulphates in a similar manner. Thus, AMD generation involves numerous chemical reactions and can result in various AMD compositions and chemistries. Accordingly, the amount and severity of AMD production varies with time and from case to case, depending upon the local climate and minerology (Amos et al., 2015). Factors that have the most significant effect on AMD formation include pH, temperature, aqueous oxygen concentration, and mineral surface area; some of the other physical, chemical, and biological factors that can contribute to rate of AMD generation include rock permeability, alkalinity, and bacterial activity (Akcil & Koldas, 2006). Hence, assessing the quantity/quality of AMD generation from a location throughout its lifetime is complex, making treatment of said AMD costly and challenging (Acharya & Kharel, 2020). Moreover, there are numerous methods for treating AMD and the best approach varies from site to site (Akcil & Koldas, 2006).

Although sulphate is a major constituent of AMD, it was not a concern for treatment until recently as there were no federal or provincial guidelines for fresh or marine water until September of 2021 when Nova Scotia adopted the fresh water guideline of 128 mg/L for sulphate from the British Columbia Ministry of Environment and Climate Change Strategy or BCMOECCS (NSECC, 2021). BCMOECCS adopted this limit due to concerns for aquatic toxicity (British Columbia Ministry of Environment and Climate Change Strategy [BCMOECCS], 2013). Sulphate is a polyatomic anion with the chemical formula of SO_4^{2-}, and it is highly soluble in water. Sulphate is part of the sulphur cycle in wetlands, which involves multiple biological and geological processes that transform sulphur into different species in oxidized and reduced states (Kadlec & Wallace, 2009). Oxidized forms of sulphur (such as sulphite, sulphate, and thiosulphate) are found in the water column where oxygen is present and redox potential is high; reduced forms of sulphur (including sulphide, bisulphide, and elemental sulphur) are found in the sediments where oxygen is deficient and redox potential is low (Kadlec & Wallace, 2009). From a treatment perspective, sulphur has a critical role in the formation / storage of metal sulphides and the majority of sulphur removal will normally produce organic, elemental, and metal sulphide molecules in the wetland sediments (Kadlec & Wallace, 2009). Sulphate can reduce the growth rate of some macrophytes due to phytotoxicity and eutrophication by releasing phosphorus (Kadlec & Wallace, 2009). In general, settling ponds and wetlands are not effective at removing sulphates. For wetlands treating AMD, the inlet concentrations of sulphates typically exceed the treatment capacity; for example, only an average of 14 % reduction in sulphates was seen across 32 free water surface wetlands from multiple studies (Kadlec & Wallace, 2009, p. 417).

2.1.2 Economic, Environmental, and Health Impacts of AMD

Acid mine drainage is not only a concern in Cape Breton, but throughout Canada with more than 10,000 abandoned mines (Coumans, 2003). Abandoned mines require continuous treatment that can last indefinitely, ultimately requiring the Canadian Federal and Provincial Governments to pay the reclamation costs (Coumans, 2003); estimated government spending in 1994 associated with AMD related expenditures was $5.25 billion CAD (Mining Watch Canada, 2000). In the United States, recent estimates indicate that more than 20,000 km of streams are polluted by AMD (Skousen et al., 2019) even though any release of AMD in the United States should be governed through regulations at multiple levels (local, regional, national, and international), such as the Clean Water Act (CWA) and the Surface Mining Control and Reclamation Act (SMCRA) (Acharya & Kharel, 2020). On a global level, "…governments and private companies spend millions of dollars each year in capital and operational costs to treat AMD, meet effluent limits, and minimize environmental risks" (Acharya & Kharel, 2020). According to Skousen et al. (2019), a combination of reclamation and treatment of AMD discharge over the last 40 years has significantly reduced the amount of AMD released into the environment globally. However, AMD continues to be a worldwide problem with continued costs and environmental impacts (Muhammad et al., 2015).

Unless the AMD is gathered and treated before release, then the acidic effluent from the abandoned mines will make its way out of the mine and into local water bodies, which poses risk to the environment and the species that inhabit the area (Mine Environment Neutral Drainage, 1994). Due to its high acidity and its high metal content, AMD has serious environmental consequences that not only affect local water bodies and aquatic organisms, but that also travel up the food chain ensuing in greater ramifications both temporally and spatially (Kefeni et al., 2017). Environmental concerns for AMD include contamination of drinking water, reduction of aquatic life in affected water bodies, accumulation of metals in food chain, and deterioration of ecosystems (Ochieng et al., 2010). "When AMD enters surface water bodies, the effects include biotic impacts on stream and lake organisms through direct toxicity, habitat alteration by metal precipitates, visual changes from orange or yellow staining of stream sediments, nutrient cycle disruptions," etc., rendering the water unsuitable for other uses (Skousen et al., 2019).

The heavy metals associated with coal mining effluent, such as iron and arsenic, are not degradable and become concentrated as a result of the mining process (Dutta et al., 2017). Heavy metals are persistent toxins that have the potential to bioaccumulate and biomagnify in food chains (Ali et al., 2019). Elevated heavy metals have a negative impact on plants and animals; in humans, high amounts of heavy metals can form complex toxic compounds that negatively affect biological functionality (Dutta et al., 2017). In general, health concerns associated with effluent leached from abandoned mines that penetrate into local water supplies include elevated metals in body fat due to contaminated food supplies, resulting in higher cancer and heart disease rates (Mining Watch Canada, 2000).

2.2 Treatment of AMD from Sydney Coalfield

Acid mine drainage is an inevitable consequence of coal mining (Dutta et al., 2017). The environmental impacts of AMD can be reduced in three ways: (1) prevention of the production of acidic effluent, (2) prevention of the drainage of acidic effluent, and (3) the collection and subsequent treatment of the acidic effluent (Akcil & Koldas, 2006).

Preventing AMD formation is considered the most effective option (Johnson & Hallberg, 2005) and generally involves preventing sulphide minerals coming in contact with oxygen, water, and/or certain bacteria (Kefeni et al., 2017). Some prevention methods include backfilling mines to reduce oxygen ingress and/or increase alkalinity (Kefeni et al., 2017), controlling post-mining hydrology at site by diverting water from site or though pretreatment to increase water alkalinity, and employing oxygen barriers such as impermeable membranes and hydraulic seals (Skousen et al., 2019).

However, in the case of the Sydney coalfields region, the hydraulic systems that have formed in the former coal mines are too vast and complex for prevention of AMD formation to be considered as the main reclamation strategy (M. Mkandawire, personal communication, February 13, 2020). Remediation of the Sydney coalfields was further complicated by inaccurate mine documentation (due to the long and complicated history of mining in the area which includes bootleg mines) and changes in mine structure/ hydrology post mine closures (due to erosion and substructure collapses). Even though prevention methods have been employed on a smaller scale in specific circumstances within the Sydney area, e.g., redirecting surface / ground waters and managing the water level in the hydraulic systems by adjusting pumping rates (J. MacPhee, personal communication, August 5, 2021), collection and treatment is the primary method used for addressing the environmental impacts of AMD generated by the abandoned underground mines in the Sydney region (Government of Canada, 2013).

2.2.1 Scope of AMD Issue in Cape Breton

The majority of the coal in the Sydney coalfield is considered submarine, with 5/6 of the reserve being classified as thermal coal and the remainder considered to be metallurgical (Hacquebard, 1993). Mining started along the shoreline and extended out underneath the Atlantic Ocean (Zodrow, 2005), forming large, interconnected labyrinths of mine workings (Shea, 2010). The coal in this area is sulphurous, containing 2.5 % to 6.2 % sulphur depending on the coal seam (Hacquebard, 1993), and pyrite rich (Zodrow, 2005). The AMD from the Sydney coalfield can be typically characterized by low pH and the contaminants of concern include sulphates and heavy metals such as iron, manganese, and aluminum (Morykot, 2014). Contaminated receptors include sediment, ground water, and surface water (Government of Canada, 2023a-d).

The mines in the Sydney coalfield were operated by Cape Breton Development Corporation (CBDC or DEVCO) since 1967; CBDC was a federal crown corporation and the proponent eventually responsible for the operation / monitoring of the AMD treatment (Wilson et al., 2011). Enterprise Cape Breton Corporation (ECBC) assumed the AMD legacy upon CBDC ceasing operation in 2009 (Meiers et al., 2015). Responsibility was then transferred to Public Works and Government Services Canada (PWGSC) in 2014 when ECBC disbanded (Government of Canada, 2016). PWGSC is now known as Public Services and Procurement Canada (PS&PC) and continues to be the party responsible for AMD treatment in Cape Breton.

CBDC performed audits in 1998/1999 and 2004/2005 to assess the extent of assets and properties for which CBDC would be responsible to close and remediate (Wiatzka, 2006). The scope of the environmental remediation program (see Figure 2.1) consists of more than fifty underground coal mines, which produced approximately 500 million tons of coal and waste rock (Shea, 2009). Remediation of the coal mining operations in Cape Breton entailed addressing more than 720 affected properties and 95 coal associated operations encompassing over 1,000 km^2 (Meiers et al., 2015) with a wide range of different types of facilities related to the coal mining industry including (but not limited to) mines, an international pier, rail systems, loading facilities, waste rock disposal sites, and wash plants (Wiatzka, 2006). Following federal and provincial guidelines [including the Soil Remediation Petroleum Products Policy from Nova Scotia Department of Energy (NSDOE), Mine Closure Guidelines from Ontario, and federal guidelines from Canadian Council of Ministers of the Environment (CCME)], CBDC developed a strategy to address the closure and remediation of all CBDC sites after considering multiple options including cleanup, removal, passive / active treatment systems, etc. (Wiatzka, 2006).

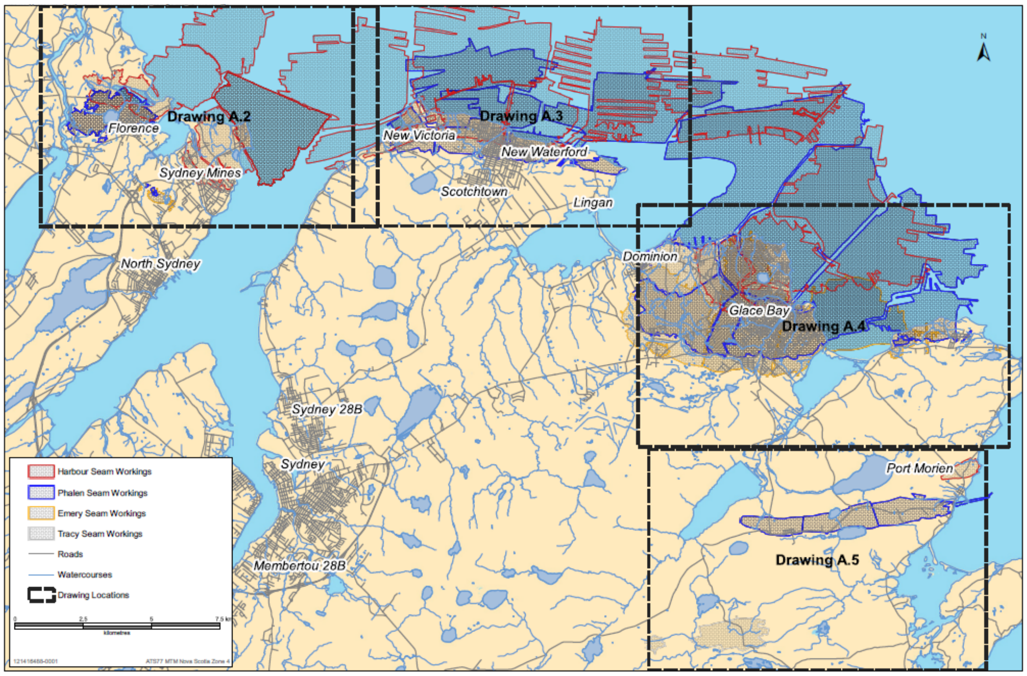

Figure 2.1 Former coal mines in Sydney coalfield (Stantec Consulting Ltd., 2020).

Currently, there exists five treatments sites for AMD in the Sydney area that jointly treat six billion liters per year of polluted mine water, including an active treatment plant in New Victoria, and four passive treatment plants: (1) Neville St. PTS in Reserve Mines, (2) No. 11 mine in Glace Bay, (3) the Beaver mine in Albert Bridge, and (4) the Franklin mine in Big Bras d’Or (Government of Canada, 2016).

2.2.2 Treatment Methods for AMD

When considering the treatment methods of AMD, there are a plethora of options, which are generally grouped under two main categories having many subcategory permutations: active treatment (requires continuous operation and maintenance support) and passive treatment (requires little support once in operation) (Johnson & Hallberg, 2005). Whether passive or active, treatments can use chemical, physical, and/or biological means to neutralize the acidic effluent and remove the metals from the AMD (Johnson & Hallberg, 2005). Typically, passive treatment uses naturally occurring sources of energy to treat less polluted waters and active treatment uses chemicals and powered systems to treat more heavily polluted water (Wolkersdorfer, 2011).

Treatment of AMD requires neutralization of acidity and removal of heavy metals (Johnson & Hallberg, 2005). The chemicals typically utilized to neutralize acidic mine effluent include limestone, hydrated lime, caustic soda, soda ash, ammonia, calcium peroxide, etc. (Akcil & Koldas, 2006). Acidity can also be reduced by aeration if CO_2 is a major contributor of acidity (Atkins International Ltd., 2008). Increases in acidity also increase the dissolution of heavy metals in mine effluent; thus, heavy metals will start to precipitate with increases of pH. Aeration also helps the precipitation of metals as oxidized metals drop out of solution given sufficient residence time (Johnson & Hallberg, 2005). In addition, microorganisms can be used to increase alkalinity and precipitate metals (Johnson & Hallberg, 2005). Acid mine drainage varies in chemistry based on location and thus requires site-specific treatment strategies that treat AMD both effectively and economically (Skousen et al., 2019).

2.2.3 Passive treatment of AMD

Passive treatment is an effective option for AMD treatment that is low maintenance, easy to operate, cost-effective, and does not require continuous chemical inputs. Passive treatment is suitable for treating AMD with either high contamination concentrations or high throughput, but the other has to remain low to moderate (Skousen et al., 2019). In comparison to active treatment methods, passive treatment methods generally require longer retentions times and greater land areas (Skousen et al., 2017). Passive treatment systems often use multiple treatment strategies in series to achieve effective treatment. The monitoring and maintenance required by passive treatment systems are typically lower than that of active treatments systems, but some passive systems can need more extensive maintenance every five to 10 years, such as replacement of an alkaline substrate, removal of accumulated precipitates, or replacement of aquatic plants (Skousen et al., 2017).

Passive treatment technology employs natural biological and / or geochemical processes to neutralize acidity and remove metal contaminants (Skousen et al., 2017). Biological treatments depend on bacterial activity to treat AMD; examples include aerobic and anaerobic constructed wetlands, vertical flow wetlands, and bioreactors (Skousen et al., 2017). Geochemical treatments use alkaline materials to increase the buffering capacity of AMD in order to neutralize acidity; examples of geochemical treatments include anoxic limestone drains, open limestone channels, limestone leach beds, steel slag leach beds, diversion wells, limestone sand, and low pH Fe oxidation channels (Skousen et al., 2017).

Four of five treatment sites for AMD in the Sydney area are passive treatment plants including the Neville St. PTS, which is the focus of this work. The Neville St. PTS has undergone upgrades and an expansion over the years it has been in operation, and is now technically a semi-passive treatment system. Passive treatment technologies employed at this facility include settling ponds, constructed wetlands, and a natural wetland.

2.3 Neville St. Passive Treatment System (PTS)

At a high level, there are three basic approaches to reduce the environmental impact of AMD: (1) primary control which consists of methods that prevent of the formation of AMD, (2) secondary control which consists of methods that prevents AMD migration, and (3) tertiary control which consists of methods that collect and subsequently treat AMD (Akcil & Koldas, 2006). At the Neville St. PTS, both primary and tertiary controls are incorporated into the design. Primary control is employed through the use of pumps to maintain the mine water level at a relatively constant height in the 1B hydraulic system to minimize the generation of AMD. Tertiary control at the Neville St. PTS involves the chemical treatment of the AMD followed by removal of contaminants by several process steps, including aeration cascades, settling ponds, and constructed / natural wetlands.

2.3.1 History of Neville St. PTS

In Cape Breton, the mining industry was located around the city of Sydney and the surrounding towns and villages (Sydney Mines, New Waterford, Lingan, Dominion, Reserve Mines, Glace Bay, Port Morien, etc.) as depicted in Figure 2.2. Over the centuries, the mining process formed large, interconnected systems of mine workings (Shea, 2010). When the mines were allowed to flood, an estimated 50 billion gallons of water filled the tunnels, creating hydraulic systems with varying inputs of ground and surface water and intermittent outputs of acidic effluent (Shea, 2009). The network of mines in the Sydney coalfield created three major hydraulic systems that were identified to require discharge treatment: (1) 1B hydraulic system, (2) New Waterford hydraulic system, and (3) Sydney Mines hydraulic system (Shea, 2010).

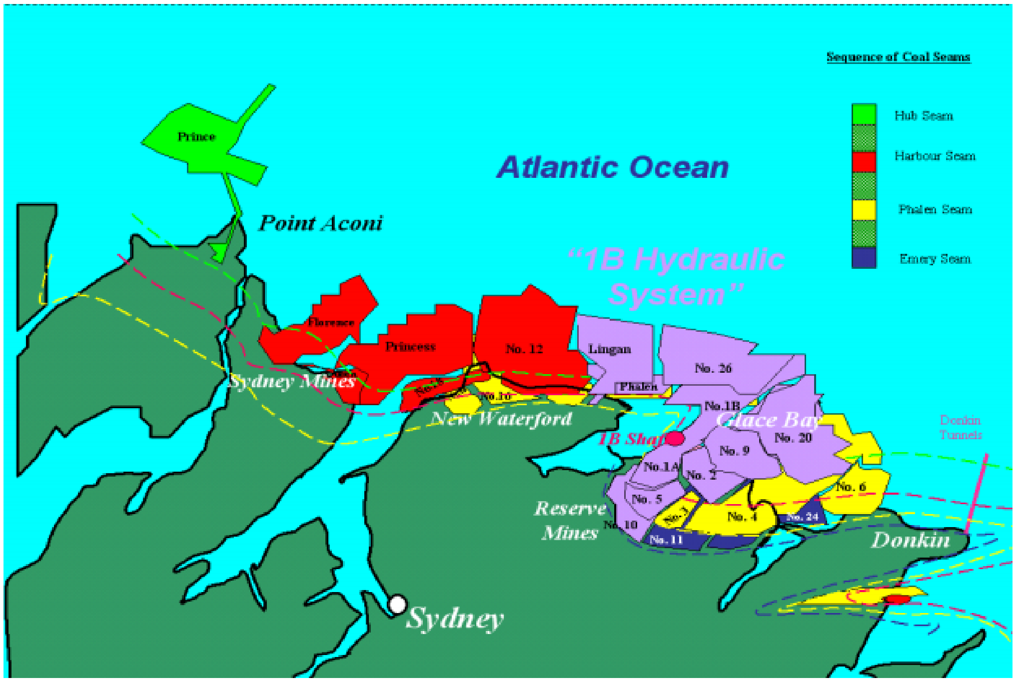

Figure 2.2 Former collieries in the Sydney coalfield (Shea, 2009).

The 1B hydraulic system is situated beneath the communities of Glace Bay, Dominion, Reserve Mines, and Donkin and includes 10 abandoned underground, interconnected mines that were created through 127 years of mining, that have a depth range from 125 ft above sea level to 2,700 ft below sea level, and that span three different coal seams (Shea, 2009). Different sections of the mines in the 1B hydraulic system were allowed to fill up with water as dewatering pumps were decommissioned as part of a staggered closure process that started in 1938; some mines sections continued to remain active while new mines continued to be opened until 1985 (Shea, 2009). After the final colliery closure in 1999 (Shea, 2009) and once the last dewatering pumps were ceased, the mine water levels continued to slowly rise. Presently, the 1B hydraulic system alone is estimated to contain 16 billion gallons of AMD (Stantec Consulting Ltd., 2020).

As part of the management of mine water levels, water was pumped from the mines in the 1B hydraulic system and discharged to the ocean using the 1B Shaft pumps for more than 50 years until 1985 “without any noticeable discoloration to the ocean” (Shea, 2009, p. 292). Figure 2.3 shows an incident in 1992 where an inrush of water from a closed colliery due to a failed underwater dam triggered the reuse of the 1B Shaft outfall to the Atlantic Ocean to try to manage water levels in operating collieries; mine water discharge to the ocean post incident was stopped after mine water quality dropped drastically (Shea, 2009). Boreholes were drilled in 2001 to monitor rising mine water levels, and analysis of mine water samples from these boreholes in 2002 showed high acidity and metal concentrations (Shea, 2010). Shortly thereafter, it was determined that the rate of rising mine waters coupled with the declining water chemistry posed a severe risk to the inshore fishing industry if released to the ocean, which would occur in a short time if no action was taken (Shea, 2010). In response, a mine water working group of international experts was formed to inform the development of management strategies for mine water levels in the 1B hydraulic system and an alternate discharge site was established adjacent to Neville St. in Reserve Mines (Government of Canada, 2013).

The Neville St. Wellfield (NSW) was developed in 2003 to pump better quality AMD from the 1B hydraulic system to avoid discharging worse quality AMD to the Atlantic Ocean via the 1B Shaft pumps (Wolkersdorfer, 2011). The NSW consisted of multiple submersible pumps installed in cased boreholes to transfer AMD from the 1B hydraulic system to Cadegan Brook (Stantec Consulting Ltd., 2020), which eventually flows to the

Atlantic Ocean. Since put in operation, the NSW has stabilized the water levels in the 1B hydraulic system (Stantec Consulting Ltd., 2020). However, between 2003 and 2008, the mine water discharge from the NSW continued to worsen with average iron concentration increasing from 0.36 mg/L to 5.22 mg/L (more than five times greater than the target of 1.00 mg/L) and average aluminum concentration increasing from 0.09 mg/L to 0.79 mg/L (Shea, 2009). Hence, the quality of the mine water effluent at the NSW had deteriorated such that it required treatment before release.

Figure 2.3 Release of AMD from 1B hydraulic system into Atlantic Ocean (Shea, 2009).

In 2008, the pumps at the NSW were automated to improve water level control to help minimize AMD generation (Stantec Consulting Ltd., 2020). The original PTS was completed in 2009 at the NSW location to treat 7,000 L/min of AMD. Tracer tests were conducted in 2009 and 2010 at the Neville St. PTS by the Mine Water Remediation Industrial Research Chair at CBU through the federally funded 2008 – 2018 Mine Water Research Program (MinWaReP) and showed that the effectiveness of PTS treatment was being diminished due to insufficient residence time in the settling pond (Wolkersdorfer, 2011). Baffle sheets were installed in the settling pond in 2010 to increase residence time, which is shown in Figure 2.4; but little improvements in performance were seen as it is suspected that the baffles created higher flow rates in the settling pond (Atkins International Ltd., 2013). Overall, the original treatment system was able to reduce average iron content from 8.39 mg/L at the settling pond inlet to 0.67 mg/L at the wetland outlet, meeting the discharge target of less than 1.00 mg/L (Shea, 2010).

Figure 2.4 Neville St. passive treatment system in 2010 (Wolkersdorfer, 2011); OWL = outlet of constructed wetland, OSP = outlet of settling pond, SPD = settling pond discharge, CPI = cascade pond inlet.

Through continued performance analyses jointly conducted by MinWaReP at CBU and by PS&PC, it was determined that the original facility was unable to handle the additional loading during high rainfall and runoff seasons in the spring and fall (Atkins International Ltd., 2013). In May of 2014, construction of a second, larger semi-passive system began to increase AMD treatment capacity (Government of Canada, 2016). The facility was brought online at reduced rates in June of 2015 as the constructed wetland was not fully developed; it should be noted that this is when sampling data for the second system began to be collected. By April of 2016, the expanded facility was operating at full capacity, with the ability to treat 19,000 L/min (see Figure 2.5); the expansion costed $6.1 million CAD (Government of Canada, 2016). During the same period (from late 2013 until late 2016), five of the wells with the poorest mine water quality were abandoned and sealed (CBCL Limited Consulting Engineers, 2016a). As part of the expansion, improvements were made to the original system including chemical dosing and additional aeration cascades, making the original system also semi-passive (Stantec Consulting Ltd., 2020). This new system increased the combined capacity to treat AMD such that regulatory discharge guidelines should be met for all operating regimes, including peak flow (Stantec Consulting Ltd., 2020).

Figure 2.5 Current configuration of Neville St. PTS (Google Maps, 2023).

2.3.2 Process Description of Current Neville St. PTS

The current Neville St. PTS consists of two parallel treatment systems, the original facility (System 1) and the expansion (System 2). A process flow diagram of the current layout of these two systems is provided in Figure 2.6. The semi-passive treatment systems at the Neville St. PTS include caustic soda (NaOH) addition, aeration cascades, settling ponds, and wetlands. The pumps supplying mine water to both treatment systems are automated in order to be able to maintain the mine water level at approximately 5.5 m ± 0.3 m (18 ft ± 1 ft) below sea level (Stantec Consulting Ltd., 2020).

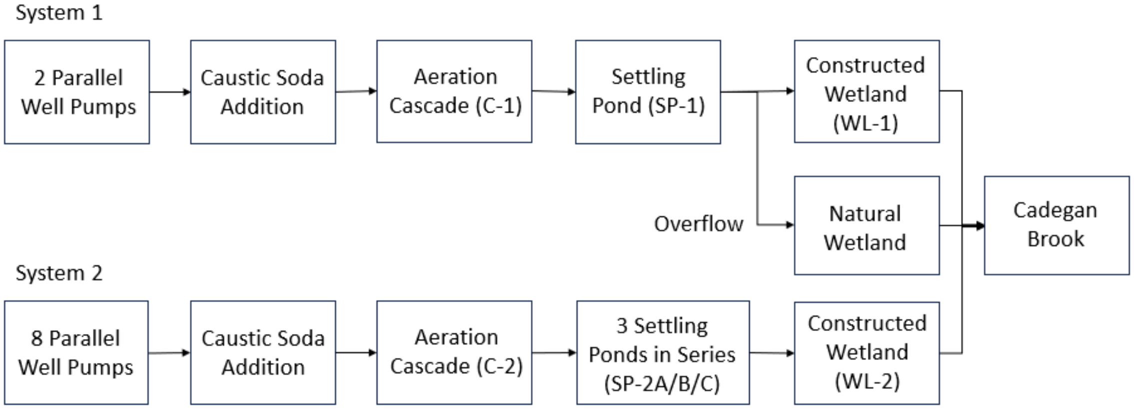

Figure 2.6 Process flow diagram for Neville St. PTS post-expansion.

Figure 2.7 depicts the current process treatment scheme in more detail. The original treatment facility was completed in 2009 to treat 7,000 L/min of AMD (Government of Canada, 2016) and is located on the east side of the access road. After several improvements over the years of operation, System 1 is currently semi-passive and treats the mine water in four stages: (1) caustic soda addition neutralizes mine water acidity, (2) aeration cascades (C-1) allow pumped mine water to vent off gases (CO_2) and absorb oxygen, (3) the majority of oxidized metal drops out in the settling pond (SP-1), and (4) the remaining metals are reduced in a constructed wetland (WL-1). Treated effluent is released to Cadegan Brook, which is east of the entire facility and travels northward to the Atlantic Ocean. Directly east of System 1 is a natural wetland. In the event of high throughput, an overflow weir located between the settling pond and the constructed wetland directs overflow towards the natural wetland; after treatment via the natural wetland, the overflow eventually ends up in Cadegan Brook as well. The 2016 expansion included construction of a second, larger PTS located on the west side of the access road, again consisting of chemical dosage, aeration cascades (C-2), settling ponds (SP-2A, SP-2B, & SP-2C), and a constructed wetland (WL-2). The combined treated effluent streams are still released to Cadegan Brook. Pictures of the process at Neville St. PTS can be found in Appendix A.

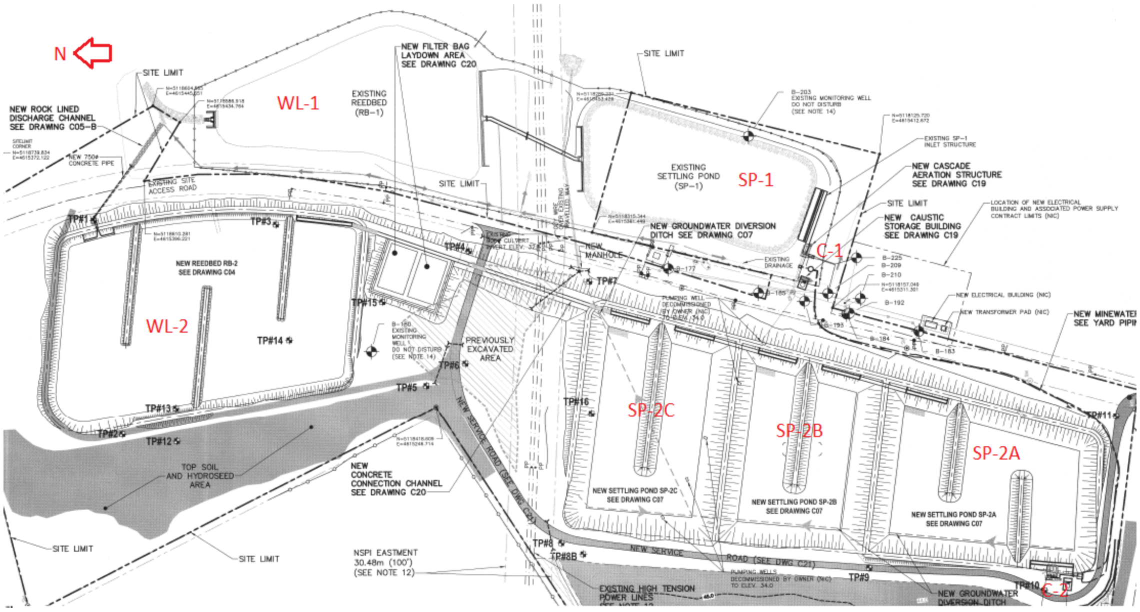

Figure 2.7 Neville St. PTS post-expansion process treatment scheme (CBCL Limited Consulting Engineers, 2016b).

Although the expansion of the Neville St. PTS was needed because additional performance analysis found that the original system was unable to treat AMD to meet guideline standards during peak loads, there was no additional performance analysis of the expanded system other than regular monitoring by PS&PC, which may or may not represent peak loading performance, due to the end of the MinWaReP funding at CBU in 2018.

2.4 Current Monitoring Program

A monitoring program for the Neville St PTS has been conducted by PS&PC and its predecessors since System 1 was put in operation in 2009. Prior to this, the location was a well field that had a separate monitoring program and released effluent without treatment.

Sampling has taken place twice a month on System 1 (SP1) since September of 2009. Sampling on the expansion, System 2 (SP2), has taken place twice a month since October of 2015. Sampling was reduced to once a month for both systems in April of 2019.

The performance of the Neville St PTS is monitored based on three key parameters: pH increase, acidity removal, and metals removal. These parameters have been measured at three points in each system: (1) treatment system inlet (CPI), (2) settling pond outlet (OSP), and (3) constructed wetland outlet (OWL). However, sampling at the settling pond outlets ceased in April of 2019; thus, after this date, samples were only taken at the treatment system inlet and wetland outlet for both systems. Samples are also taken monthly since 2009 at Cadegan Brook (CB) upstream and downstream of the release point of the Neville St. PTS combined effluent. For a more detailed description of each sampling point including its sampling point code, see Table 2.1.

Table 2.1 Sampling point names and codes at Neville St. PTS.

Sampling Point Description	**System 1 Sampling Point Code**	**System 2 Sampling Point Code**	**Sampling Point Description**	**Cadegan Brook Sampling Point Code**
Treatment system influent sampled at the upper portion of the steel cascade from the inlet pipe	SP1-CPI	SP2-CPI	Cadegan Brook water sampled upstream of the PTS inflow point	CB-3S
Effluent sampled at the outflow of the settling pond	SP1-OSP	SP2-OSP	Cadegan Brook water sampled downstream of the PTS inflow point	CB-5S
Effluent sampled at the outflow of the wetland	SP1-OWL	SP2-OWL		

Figure 2.8 is a picture of the Neville St. PTS taken in 2015 (post-expansion) with sampling points labeled on System 1, System 2, and Cadegan Brook. Samples are

processed by Bureau Veritas (formerly Maxxam Analytics) in Sydney. Samples are analyzed with respect to meteorological data (precipitation, temperature, etc.) and federal guidelines and provincial standards. See Appendix B for a detailed list of parameters measured regularly as part of the ongoing monitoring program.

Figure 2.8 Sampling Program by PS&PC at Neville St. PTS (PS&PC, 2021).

Since sampling only occurs once a month (previously twice a month), sampling results may or may not represent peak load conditions. Thus, any potential performance deficiencies that occur during peak loading may not be detected by the current monitoring program due to sampling infrequency.

2.4.1 Contaminants of Concern

The effluent from the Sydney coalfield mines is acidic and contains heavy metals such as iron, manganese, and aluminum (Morykot, 2014). Atkins Industrial Limited (2008), (2013) designed and sized the original system (System 1) and the expansion (System 2) based primarily on the attenuation of iron with secondary consideration to manganese and aluminum removal. Prior to the expansion, the original system had difficulty meeting the intended iron removal performance during higher flow rates due to insufficient residence times. Manganese removal rate was poor as well prior to the expansion and this deficiency was attributed to the low iron removal rate as iron oxidizes more readily than manganese; as such, manganese will only be attenuated by treatment systems once iron is removed or after iron concentration becomes less than manganese concentration (Atkins International Ltd., 2013). On the other hand, aluminum oxides will form oxyhydroxides as long as the acidity is neutralized first; thus, aluminum was removed to acceptable concentrations early in the process regardless of flow rates (Atkins International Ltd., 2013). Thus, the goal of the expansion was to increase the treatment capacity of the Neville St. PTS so that sufficient iron would be removed for all flow regimes.

Heavy metals are constituents of concern that need to be removed from wastewater before being released into the environment. Some trace metals – e.g., iron, manganese, zinc, etc. – act as essential micronutrients as long as they are at low concentrations (United States National Library of Medicine, 2015). However, in many wastewaters such as industrial effluent and landfill leachates, the concentration of trace metals can well exceed acceptable toxicity (Kadlec & Wallace, 2009). Heavy metals are a concern as they are persistent toxins that have the potential to bioaccumulate and biomagnify in food chains (Ali et al., 2019). In passive treatment systems that receive heavy metals in their influent, most metals are seen to accumulate at higher concentrations in soils and biological tissue whereas surface waters typically see relatively lower concentrations of heavy metals (Kadlec & Wallace, 2009). In Canada, the treated wastewater from treatment wetlands must meet certain metal guidelines prior to being released into natural ecosystems.

2.4.2 Regulating Bodies and Limits

The provincial government of Nova Scotia and the federal government of Canada work together to regulate mining activities in Nova Scotia (Canadian Council of Ministers of the Environment [CCME], 2023). At the federal level, Section 36 of the Fisheries Act states that discharging “deleterious substance[s] of any type in water frequented by fish is prohibited” (Government of Canada, 2023e, p. 33). Environment Canada’s Toxic Substances Management Policy regulates the creation, usage, and disposal of substances considered to be toxic, predominantly those which are anthropogenic, persistent, and/or bioaccumulative (Environment and Climate Change Canada [ECCC], 2013). The CCME provides water quality guidelines for both freshwater and marine environments, which are available for selected parameters in Table 2.2. Constructed wetlands typically discharge into water bodies that fall under the CCME purview, and therefore are often indirectly regulated by these guidelines through their operating permits based on the wetland effluent quality (CCME, 2003). As the treated mine water from the Neville St. PTS is first released to Cadegan Brook (fresh water) and eventually flows to the Atlantic Ocean (marine water), both fresh and marine water limits are presented.

Table 2.2 CCME water quality guidelines for the protection of aquatic life for selected parameters (CCME, 2023).

Parameter	**Water Quality Guidelines for the Protection of Aquatic Life**					
	Fresh Water			**Marine Water**		
	Short Term Concentration (µg/L)	**Long Term Concentration (µg/L)**	**Date**	**Short Term Concentration (µg/L)**	**Long Term Concentration (µg/L)**	**Date**
Colour	-	Narrative	1999	-	Narrative	1999
Iron	-	300	1987	-	-	-
Dissolved Iron	-	-	-	-	-	-
Ferrous Iron (Fe^{3+})	-	-	-	-	-	-
Ferric Iron (Fe^{2+})	-	-	-	-	-	-
Aluminium	-	Variable	1987	-	-	-
Manganese	Equation	Variable	2019	-	-	-
Oxidation-Reduction Potential	-	-	-	-	-	-
pH	-	6.5 to 9.0	1987	-	7.0 to 8.7 & Narrative	1996
Sulphate	-	-	-	-	-	-
Turbidity	-	Narrative	1999	-	Narrative	1999

To simplify Table 2.2, the guidelines that vary or that are calculated are noted in the table, with details discussed further presently. According to CCME (2023) for both fresh and marine waters, the long-term concentration guideline for apparent colour is “the mean percent transmission of white light per meter shall not be significantly less than the seasonally adjusted expected value for the system under consideration.” The corresponding long-term concentration guideline for true colour states that “the mean absorbance of filtered water samples at 456 nm shall not be significantly higher than the seasonally adjusted expected value for the system under consideration” (CCME, 2023).

The long-term concentration guideline limit provided by CCME for aluminum in fresh water is dependent upon pH with a limit of 5 µg/L when pH is less than 6.5 and 100 µg/L when pH is greater than 6.5 (CCME, 2023). CCME’s short-term concentration guideline limit for manganese in fresh water (in µg/L) is determined by Equation 2.2, which is a function of water hardness measured in $CaCO_3$ equivalents in mg/L (CCME, 2023):

$$Guideline\ benchmark = exp\big(0.878\big(\ln(hardness)\big) + 4.76\big) \tag{2.2}$$

The equivalent long-term concentration guideline limit for manganese is determined by employing the Canadian Water Quality Guidelines (CWQG) calculator, which is a function of water hardness and pH (CCME, 2023).

According to CCME (2023), “The pH of marine and estuarine waters should fall within the range of 7.0 – 8.7 units unless it can be demonstrated that such a pH is a result of natural processes. Within this range, pH should not vary by more than 0.2 pH units from the natural pH expected at that time.” For turbidity of both fresh and marine waters, CCME’s concentration guideline limits are different for clear flow and for high flow or turbid waters (CCME, 2023). For clear flow, the short-term turbidity guideline limit (i.e., 24 hour period) is 8 NTUs above background levels while the short-term turbidity guideline limit (i.e., 30 day period) is 2 NTUs above background levels. For high flow or turbid waters, the turbidity guideline limit is 8 NTUs above background levels at any time when background levels are between 8 and 80 NTUs or 10 % above background levels when background levels are greater than 80 NTUs.

At a provincial level, mineral rights, mineral leasing, and reclamation of mining operations in Nova Scotia are regulated through the *Mineral Resources Act* (Government of Nova Scotia, 2019). Through the *Environment Act*, the Nova Scotian Department of Environment and Climate Change regulates environmental assessments, the process for industrial approvals, and reclamation for mining operations (Government of Nova Scotia, 2018). Nova Scotia Environment (NSE) Tier 1 Environmental Quality Standards (EQS) for Surface Water provides water quality guidelines for surface water and groundwater discharging to surface water, which are available for selected parameters in Table 2.3. Many of the NSE Tier 1 EQS are adopted from the CCME guidelines. Again, in order to simplify Table 2.3, the guidelines that vary are noted in the table without detailed description. Colour is the only parameter for which this is the case and the NSE Tier 1 EQS is the same as the CCME guideline.

Table 2.3 NSE Tier 1 EQS for surface water for selected parameters (NSECC, 2021).

Parameter	**NSE Tier 1 EQS for Surface Water**					
	Fresh Water			**Marine Water**		
	Limit	**Reference**	**Unit**	**Limit**	**Reference**	**Unit**
Colour	Narrative	CCME	TCU	Narrative	CCME	TCU
Iron	300	CCME	µg/L	-	-	-
Dissolved Iron	-	-	-	-	-	-
Ferrous Iron (Fe^{3+})	-	-	-	-	-	-
Ferric Iron (Fe^{2+})	-	-	-	-	-	-
Aluminium	5	CCME; at pH <6.5	µg/L	-	-	-
Manganese	430	CCME	µg/L	-	-	-
Oxidation-Reduction Potential	-	-	-	-	-	-
pH	6.5 to 9.0	CCME	-	7.0 to 8.7	CCME	-
Sulphate	128,000	BCMOECCS	µg/L	-	-	-
Turbidity	-	-	-	-	-	-

The selected parameters in both Table 2.2 and Table 2.3 include heavy metals associated with the local ore (i.e., iron, manganese, and aluminum), parameters associated with pyritic ore oxidation (i.e., sulphates, pH, and oxidation-reduction potential (ORP)), and general water quality parameters (i.e., colour and turbidity). The significance of these parameters will be discussed in more detail in the methodology section.

PS&PC compares Neville St. PTS effluent concentrations to CCME water quality guidelines and NSE Tier 1 EQS. See Appendix B for a detailed list of parameters measured regularly as part of the ongoing monitoring program. If effluent from the Neville St. PTS does not meet the guidelines set out by the provincial and federal governments, then steps must be taken to treat the effluent before release (Shea, 2010).

Additionally, PS&PC commissioned a report with the objective of characterizing the background surface water quality of the Sydney coalfield area; a summary of said report for selected parameters is presented as Table 2.4. The results presented in this report were based on data collected at 17 locations within the Sydney coalfield area; these sampling sites were selected such that they have as little anthropogenic influence as possible (Wood Environment & Infrastructure Solutions, 2019). Sampling was conducted four times in 2018, with each sampling event timed to take place in a different season (Wood Environment & Infrastructure Solutions, 2019). Samples were analyzed by Maxxam Analytics (currently known as Bureau Veritas) for metals and general chemistry parameters (Wood Environment & Infrastructure Solutions, 2019).

The value of background surface water quality data is that it can be compared against sampling data from the Neville St. PTS and from other remediation sites in the Sydney coalfield area. Moreover, the results of this report can be further used as "preliminary or secondary screening criteria" for contaminants of concern that regularly exceeded federal and provincial limits (Wood Environment & Infrastructure Solutions, 2019, p. 3). Note that for key metals (iron, aluminum, and manganese), mean background values exceed current CCME and NSE guidelines. Based on the CCME long-term freshwater guidelines at the time, iron concentrations exceeded the limit for iron (0.3 mg/L) for 124 of 179 samples and aluminum concentrations exceeded the limit for aluminum (0.1 mg/L) for 140 of 179 samples (Wood Environment & Infrastructure Solutions, 2019); there was no CCME long-term freshwater guideline limit for manganese at that time.

Table 2.4 Background surface water quality for Sydney coalfield area (Wood Environment & Infrastructure Solutions, 2019).

Item	Aluminum (μg/L)	Cadmium (μg/L)	Copper (μg/L)	Iron (μg/L)	Manganese (μg/L)	Zinc (μg/L)	Lead (μg/L)
Mean	232.44	0.0308	1.60	1,377	480.4	10.35	0.77
Standard Error	23.35	0.0031	0.10	170.9	66.64	0.90	0.11
Median	140	0.021	1	630	161	6.8	0.25
Mode	130	0.005	1	1,100	220	2.5	0.25
Standard Deviation	312.44	0.042	1.24	2,287	884.1	11.24	1.39
Minimum	16	0.005	1	25	2.3	2.5	0.25
Maximum	3,090	0.48	8	15,000	8,000	71	12.5
Count	179	179	167	179	176	157	147
Preliminary Screening Criteria[1]	5	0.017	2	300	700	30	1
Secondary Screening Criteria[1]	3,290	2	8.6	7,500	730	60	28
CCME FWAL[2]	5 / 100	0.04 - 0.37	2 - 4	300	NG	30	1 - 7
NSE EQS for Surface Water[2]	5	0.01	2	300	820	30	1

[1] Criteria prepared by Risk Assessment and Regulatory Framework Committee, CBDC Mine Closure Program, June 2008.

[2] CCME and NSE guideline limits are from March of 2019 and may have changed (e.g., limits for manganese).

2.5 Treatment Performance Modeling of Wetlands

Natural wetlands are often referred to as nature's kidneys due to the fact that they absorb and purify large volumes of water that eventually flow into larger bodies of water, such as lakes, rivers, and oceans (United States Environmental Protection Agency [US EPA], 2004). Wetlands are a type of ecosystem characterized by containing huge quantities of water, either on a seasonal or permanent basis, and by having a vast range of plants and animals, which is why wetlands are considered to be one of the most biologically diverse environments on Earth (National Geographic, 2020). Wetlands cover approximately 14 % of land area in Canada (Kennedy & Mayer, 2002, p. 295). Examples of wetlands include swamps, marshes, bogs, fens, peatlands, and sloughs; the water in wetlands can be categorized as freshwater, brackish, or saltwater (National Geographic, 2020).

Plants in wetlands are unique because they have adapted to environments with high volumes of water and low quantities of essential nutrients, including oxygen (Kadlec & Wallace, 2009). Wetland hydrological conditions consist of shallow water depths and slow moving water, which provides ample time to allow for many physical and chemical processes to transpire (US EPA, 1995). Wetlands are vital ecosystems as they store large quantities of water and regulate watersheds, limiting erosion and flooding (National Geographic, 2020). Moreover, wetlands have the inherent capacity to improve water quality by trapping and removing contaminants, often converting these constituents into benign compounds or even into nutrients that can be used by wetland organisms (Kadlec & Wallace, 2009). Lastly, wetlands are reliable and “self-adjusting” systems (US EPA, 1995, p. 7). Hence, wetlands are a viable treatment option for many different applications.

2.5.1 Treatment Wetlands

Due to increasing legislature around the protection of wetlands, the use of natural wetlands to treat wastewater has diminished, and the application of constructed wetlands, also known as engineered or artificial wetlands, has become more prominent (Kadlec & Wallace, 2009). Constructed wetlands are “complex, integrated systems of water, plants, animals, microorganisms, and the environment” (US EPA, 1995, p. 7). Research on constructed wetlands started in Germany in the early 1950s, spread to America in the 1970s, and increased in interest worldwide in the mid-1980s (Kadlec & Wallace, 2009). Constructed wetlands are predominantly passive treatment systems based on “mechanically simple” designs (Kadlec & Wallace, 2009, p. 20) that utilize naturally occurring physical / chemical / biological processes to treat water to high quality standards (US EPA, 2020).

Design variables vary tremendously from one engineered wetland to another in order to effectively treat the vast variety of wastewater qualities in different environmental conditions to achieve a range of treatment objectives (Kennedy & Mayer, 2002). These treatment systems have gained in popularity in recent decades because they are economical, low maintenance, and environmentally sustainable (Kennedy & Mayer, 2002). Constructed wetlands can be the only treatment for a wastewater or one step in a series of treatment processes (US EPA, 1995). Often, wetland influent is pretreated; furthermore, constructed wetlands are predominantly used as a mode of secondary treatment or higher (Kadlec & Wallace, 2009).

Engineered wetlands have been built to mimic natural wetlands and to augment certain advantageous features of natural wetlands as part of the design to increase the treatment capability of the wetland (Kadlec & Wallace, 2009). Wetlands can improve water quality by removing different constituents of concern from the water that they are treating and by sequestering these constituents within the wetland through multiple mechanisms.

Constructed wetlands are usually divided into two main categories: surface flow (also known as free water surface, FWS) and subsurface flow, SSF (US EPA, 1995, p. 12). Kadlec and Wallace (2009) divide SSF wetlands into two sub-categories: horizontal subsurface flow (HSSF) and vertical flow (VF). The wetlands at the Neville St. PTS are FWS wetlands.

2.5.2 Constructed Free Water Surface (FWS) Wetlands

FWS wetlands have an open water appearance like natural wetlands and the flow moves horizontally over the bed as shown in Figure 2.9. FWS wetlands can be operated in systems that have continuous or intermittent flow (Kadlec & Wallace, 2009). Constructed FWS wetlands normally entail one or multiple shallow basins with a subsurface barrier (i.e., clay and/or geosynthetic liner(s)) to prevent seepage between the wetland and surrounding groundwater systems, and an underwater layer of soil substrate to support macrophytes (US EPA, 2000). Typical macrophytes planted in constructed FWS wetlands are species of cattail, bulrush, and reeds (US EPA, 2000).

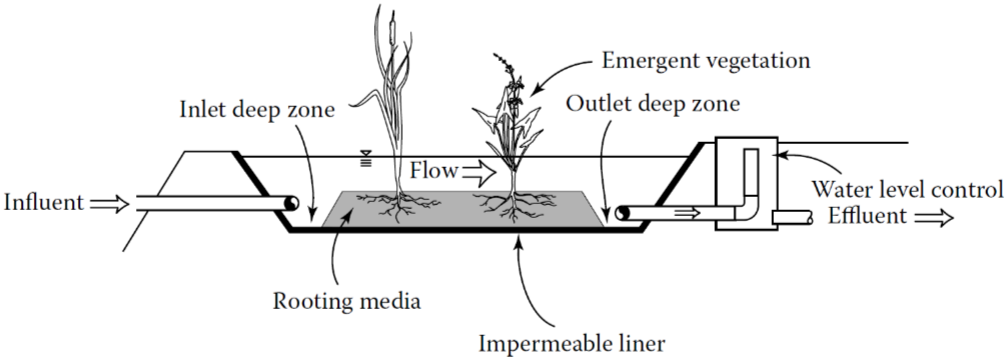

Figure 2.9 Basic configuration of FWS constructed wetlands (Kadlec & Wallace, 2009, p. 5).

Flow rates and storage volume in wetlands are important hydraulic parameters as they "determine the length of time that water spends in the wetland, and thus the opportunity for interactions between waterborne substances and the wetland ecosystem" (Kadlec & Wallace, 2009, p. 21). The flow of water in wetlands is not uniform and is prone to stagnant areas and channeling, which impacts mixing and performance of contaminant removal (Kadlec & Wallace, 2009). Currents or channeling can be created by friction, topography, wind, seepage, variation in vegetation density, etc. (Kadlec & Wallace, 2009). Mixing occurs in all directions between regions with different flow directions, velocities, and concentrations (Kadlec & Wallace, 2009). Inlet and outlet structures are employed to promote uniform dispersal and gathering of water and to minimize preferential flow / short-circuiting within the wetlands (US EPA, 2000).

2.5.3 Treatment of Acidity

High acidity in mine water can be caused by (1) proton acidity, which is a measure of protons (i.e., hydrogen cations) freely available in the water, (2) mineral acidity, which is a potential acidity that is caused by dissolved metals in the mine water that can form protons, and (3) carbonic acidity, which is a temporary acidity that is caused by dissolved carbon dioxide in the mine water that can form carbonic acid (Atkins International Ltd., 2008).

Atkins International Limited (2008) performed a comprehensive anaylsis for PS&PC on the mine water chemistry at the Neville St. location prior to construction of System 1 to inform the design process. The mine water was found to have high concentrations of dissolved carbon dioxide, which increases carbonic acidity, and low concentrations of dissolved oxygen, which means that dissolved metals are in their reduced form. As the mine water is discharged to the surface, the water becomes aerated and these metals become oxidized and hydrolyzed (Atkins International Ltd., 2008). These two chemical processes generate mineral acidiy. Equations 2.3 to 2.7, which are contained below in Table 2.5, are the oxidation and hydrolysis reactions for iron, aluminum, and manganese, which are the metals of concern at the Neville St. PTS and the combined reactions for each metal generates 2, 3, and 2 protons respectively (Atkins International Ltd., 2008).

Table 2.5 Oxidation and hydrolysis reactions for iron, aluminum, and manganese (Atkins International Ltd., 2008).

Metal	Chemical Process	Reaction	Equation
Iron	Oxidation of Ferrous Iron	$Fe^{2+} + \frac{1}{4} O_2 + H^+ \leftrightarrow Fe^{3+} + \frac{1}{2} H_2O$	2.3
	Hydrolysis of Ferric Iron	$Fe^{3+} + 2H_2O \leftrightarrow Fe(OH)_x + 3H^+$	2.4
Aluminium	Hydrolysis	$Al^{3+} + 3H_2O \leftrightarrow Al(OH)_3 + 3H^+$	2.5
Manganese	Oxidation	$Mn^{2+} + \frac{1}{2} O_2 + 2H^+ \leftrightarrow Mn^{4+} + H_2O$	2.6
	Hydrolysis	$Mn^{4+} + 2H_2O \leftrightarrow MnO_2 + 4H^+$	2.7

Through further testing, it was found that the acidity of the mine water was predominantly due to carbonic acidity, and an average of 77 % of acidity could be removed by aeration of the mine water at the beginning of the process, which effectively degasses the mine water of CO_2 (Atkins International Ltd., 2008). Furthermore, Atkins International Limited (2013) performed another assessment for PS&PC on the Neville St. location, this time post construction of System 1 to assess its performance and to inform the design process for System 2. System 1 was found to be struggling with removing iron by the settling pond as intended, especially during higher flow rates, due to insufficient oxidation and/or hydrolysis of iron, resulting in the overloading of the constucted wetland with iron and in insufficient removal of iron by the entire system (Atkins International Ltd., 2013). Likewise, manganese was not being consistently removed by System 1 in either the settling pond nor the constructed wetland; this was also due to the inadequate uptake of oxygen by mine water, which is required in order to acheive sufficient oxidation of manganese (Atkins International Ltd., 2013). Thus, a metal cascade was placed at the beginning of each system at the Neville St. PTS during the 2016 expansion to improve the uptake of oxygen, which improves the oxidation and hydrolysis of iron and manganese in order to form insoluble constituents, thereby increasing the settling rates of these metals in the settling pond(s) (Atkins International Ltd., 2013). Prior to this, the original cascade for System 1 only consisted of a concrete staircase (CBCL Limited Consulting Engineers, 2016a). Moreover, caustic soda addition was added to both systems during the 2016 expansion (CBCL Limited Consulting Engineers, 2016b), providing a way for the remaining acidity in both systems to be neutralized.

Degassing of mine water at the Neville St. location was also found to remove approximately 30 % of the alkalinity (Atkins International Ltd., 2008). This occurs because the increase in pH due to degassing causes the alkalinity to convert to carbonate, which is then removed through precipitation. Nevertheless, the degassed water was found to be net alkaline (Atkins International Ltd., 2008). However, historical mine water data indicated that acidy was increasing and alkalinity was decreasing, with anticipated conversion to net acidic occurring between June of 2014 and December of 2015 (Atkins International Ltd., 2008). Conversely, this trend was not observed when evaluating well field chemistry five years later as part of the designing process for the expansion; well chemistry including acidity and alkalinity seemed to have stabilized (Atkins International Ltd., 2013). Nevertheless, caustic soda addition can be employed to compensate for changes in acidity and alkalinity over time.

Historically, the pH within the Neville St. PTS was seen to increase from just below 7 to approximately one pH unit higher on average between the treatment system influent sampling point (which is measured before aeration occurs but after caustic soda addition) and the settling pond effluent sampling point (Atkins International Ltd., 2013). Thus, this overall increase in pH within the settling pond is due to three chemical processes: (1) neutralization of protons with hydroxide ions from caustic soda (which decreases acidity), (2) degassing of the mine water (which reduces carbonic acidity), and (2) metals removal through oxidation and hydrolysis (which increases mineral acidity). Since the net change in pH across the settling pond is positive, it is understood that the reduction in acidity from degassing and caustic soda addition is greater than the increase in acidity generated through metals removal (Atkins International Ltd., 2013). No further significant changes in acidity were observed in the wetland, which indicates that the vast majority of degassing, oxidation, and hydrolysis occurs in the settling pond (Atkins International Ltd., 2013).

Acidity is managed at the Neville St. PTS through a two-pronged approach. The first part of the approach is to try to manage mine water acidity by minimizing mineral acidity. The second part is to treat the acidity with degassing and chemical dosage. With regards to the first part of the approach, mineral acidity in the feed to the PTS is affected by pump sequencing and by managing water level in the mines. The average acidity (and alkalinity for that matter) of the untreated mine water at the PTS inlet can be reduced by operating pumps with better quality mine water more frequently. Pumps located in areas with poorer quality mine water are only operated to help maintain water level in the mines during high flow events, which are associated with high rainfall and/or rapid snowmelt events. Additionally, System 2 has more treatment capacity as it was intended to handle approximately 70 % of design flow (Atkins International Ltd., 2013). As such, poorer quality mine water is typically directed to System 2. Managing water level in the mines also affects water quality as rising and falling water level generates more mineral acidy as more ore is oxidized and dissolved into the mine water; thus, water levels have been managed to stay within a small range of 5.2 m to 5.8 m (18 ft ± 1 ft) below sea level whenever possible by an automated pumping system since 2008 (Stantec Consulting Ltd., 2020). This is more difficult to achieve during heavy rainfall and/or rapid snowmelt events. Hence, the poorest feed water quality to the Neville St. PTS is generally seen during high flow events.

2.5.4 Metal Contaminant Attenuation Mechanisms

Constructed wetlands are an effective, low cost, and sustainable option for improving wastewater quality and have become a more common option for treating metals in wastewaters. Metals are persistent toxins that bioaccumulate and biomagnify, which is why wastewaters need to be treated to ensure that metals are below federal and provincial water quality guidelines before being released into natural water bodies. Contaminants are removed in wetlands by several mechanisms, as shown in Table 2.6; metals are sequestered by some of these mechanisms. Moreover, the speciation of metals in water affects how metals can be attenuated. Metals in wetlands predominantly partition into solid and solution phases and the fraction of a metal in each phase varies over time (Roberts et al., 2005). Precipitated metals species can be removed by physical separation processes such as filtration and sedimentation (Allende et al., 2011). Dissolved metals can be removed by organisms and by soil via sorption and precipitation (Roberts et al., 2005).

Table 2.6 Mechanisms for contaminant removal by treatment wetlands (Kennedy & Mayer, 2002, p. 314).

Process		**Effluent Parameters**
Physical	Straining and Sedimentation	Suspended solids, particulate organic carbon, nitrogen, and phosphorus
Chemical	Adsorption	Dissolved organic compounds, anions (phosphate), and cations (metals)
	Precipitation	Inorganic phosphorus, sulphides, and metals
	Volatilization	Ammonia and volatile organic compounds
Biological, Microbial	Respiration	Biological oxygen demand, oxygen, nitrate, sulphate, bicarbonate, and volatile fatty acids
	Nitrification	Ammonium
	Denitrification	Nitrate nitrogen and nitrite nitrogen
	Mineralization	Organic nitrogen and phosphorus
	Assimilation	Nutrients
Biological, Plants	Growth and Uptake	Nutrients
	Gas Transport	Oxygen and related reactions

Specifically looking at iron as an example since it is the primary metal of concern for AMD from the Neville St. wellfield, it can be observed that iron is sequestered in wetlands by several mechanisms. Like other metals, iron in wetlands can be found in solid and solution phases. Wetlands can contain oxidized or reduced forms of iron depending on environmental conditions such as redox potential and pH (Kadlec & Wallace, 2009). Ferric iron, Fe^{3+}, is the dominant form in oxidized conditions; ferrous iron, Fe^{2+}, in reduced conditions. Ferric iron tends to form stable, insoluble compounds whereas ferrous iron tends to be soluble. However, ferrous iron can react with sulphide under certain conditions to form precipitates (Kadlec & Wallace, 2009).

The rate of metal contaminant removal is affected by the pH of the wastewater because the solubility of metals is dependent upon pH. High acidity in wastewaters results in the dissolution of many metals often to toxic levels (Ness et al., 2014), which reduces the effectiveness of physical separation processes that require metals to be undissolved. Acid mine drainage is very acidic and contains high concentrations of metals, and most metals are not attenuated in acidic conditions (Lottermoser, 2010). As such, acid mine drainage often needs to be pretreated in order to neutralize the acidity before metal removal can take place. For a variety of metal contaminants, such as aluminum or iron, raising water pH to neutrality is usually sufficient to decrease their concentrations to acceptable levels (Ness et al., 2014). At the Neville St. PTS, wastewater is dosed with caustic soda and aerated at the beginning of the process to reduce acidy. Once the acidy is reduced, many metal contaminants will precipitate, which allows them to be removed more easily by the settling ponds and treatment wetlands.

Metal attenuation mechanisms are discussed in the following subsections in order of decreasing significance when possible. Note that there are multiple ways to categorize these mechanisms and different authors arrange these topics in different groupings. This is largely due to the overlapping nature of many of these mechanisms, especially when applied to real world scenarios where interactions are complex and interdependent. Mechanisms that attenuate metals in treatment wetlands are covered in the following order: sedimentation, uptake by plants (osmosis), sorption, precipitation, and filtration.

2.5.4.1 Sedimentation

A significant portion of metals in treatment wetlands are present as a solid suspended in solution (Kadlec & Wallace, 2009), which explains why sedimentation is the dominant mechanism contributing to metals attenuation in treatment wetlands (Allende et al., 2011). Sedimentation is a physical process whereby suspended solids entrained in a fluid settle out of said fluid and deposit on a surface. The main driving force behind sedimentation is gravity, which causes a particle in suspension to settle out due to the higher density of that particle in comparison to the density of water. Sedimentation can be modeled by using a linear relationship with area; however, mass balances need to incorporate factors other than just sedimentation, such as generation of precipitates and resuspension of sediments (see Figure 2.10), to accurately model the attenuation of constituents of concern including metals (Kadlec & Wallace, 2009).

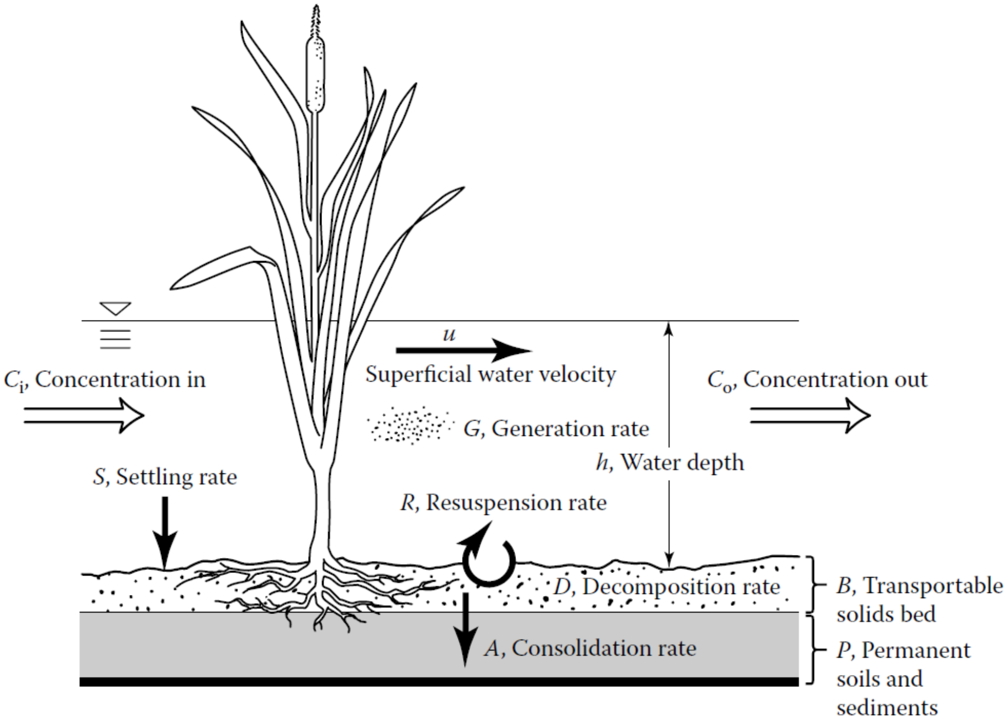

Figure 2.10 Diagram of wetland sedimentation with parameters applied in mass balances (Kadlec & Wallace, 2009, p. 217).

Settling of suspended particles in wetlands is promoted by low water velocities, shallow water depths, and high surface area (US EPA, 1995). Surface area can be augmented in FWS systems by improving plant growth and in SSF systems by bed media selection (Kadlec & Wallace, 2009). Sedimentation efficiency in FWS wetlands is hampered by the resuspension of settled particles by geological and biological interactions, such as turbulence or animal activity (Kadlec & Wallace, 2009). Resuspension is less of a factor in SSF wetlands because much of the sediment is deposited within the bed; hence, SSF systems are prone to fouling and losses in hydraulic conductivity (Kadlec & Wallace, 2009).

The sequestering of metals in soils by sedimentation (and by other mechanisms such as sorption) exposes organisms that live in the soil to higher metal concentrations. It should be noted that dissolved metals are more bioavailable than those in solid phase, and thus dissolved metals pose the most risk (Baker et al., 2012). Organisms living in the metal rich soil, such as microorganism who facilitate the transformation of some compounds into other chemical species, can be affected by metal toxicity and this can impact conversion performance of microbial reactions (US EPA, 1995).

At the Neville St. PTS, sedimentation is the dominant metal attenuation mechanism in the settling ponds. The pretreatment of the wastewater reduces acidity and results in net alkaline influent to the settling pond, which allows metals such as iron to precipitate. The iron precipitates are in the form of insoluble iron oxides and oxyhydroxides, which predominantly settle out in the settling pond, but can continue to settle out in the subsequent constructed wetland when throughput is higher (Atkins International Ltd., 2013).

2.5.4.2 Macrophytes (Aquatic Plants)

Wetland vegetation removes contaminants including metals from wastewater via osmosis. In fact, aquatic plants, also known as macrophytes, provide the second biggest sink for metals after sedimentation (Kadlec & Wallace, 2009). Macrophytes in the Neville St. PTS wetland would likely play a significant role in reducing metals concentrations to their final discharge concentrations. For example, aquatic plants likely help reduce iron concentrations up to 10 mg/L at the wetland inlets to the outlet target of less than 1.0 mg/L (Atkins International Ltd., 2013).

Aquatic plants in wetlands absorb metals that are dissolved in the sediment pore water (Baker et al., 2012). The dominant mechanism by which plants sequester metals is through uptake by their roots; this is demonstrated in Table 2.7 as the data contained therein establishes that the majority of metals attenuated by plants are found in their roots. Uptake of metals by stems / leaves in the water column by some plants does occur, but to a much lesser extent (Kadlec & Wallace, 2009).

Table 2.7 Metal distribution in wetland plants after five years of operation at a regional wastewater treatment plant in Sacramento (Kadlec & Wallace, 2009, p. 424).

Metal	Aboveground (%)	Belowground (%)
Arsenic	0.6	10.1
Cadmium	0.0	13.3
Chromium	2.2	16.8
Copper	0.6	5.5
Lead	2.0	11.8
Mercury	0.0	6.7
Nickel	0.3	4.7
Silver	0.0	2.0
Zinc	0.4	6.1

Furthermore, macrophytes can indirectly promote metal attenuation. Aquatic plants stabilize bed media and limit channeling in the bed, which increases residence time, enabling physical and chemical reactions to occur (US EPA, 1995). Macrophytes elevate dissolved oxygen content in the water, which can react with metals to form precipitates (US EPA, 1995). Aquatic plants can act as a filter and remove suspended solids; nevertheless, filtration by macrophytes is, for all intents and purposes, inconsequential due to the high void space between plants (Kadlec & Wallace, 2009). However, the presence of aquatic vegetation does significantly reduce the likelihood of particle resuspension by two mechanisms: dampening wind/wave induced motion and enhanced trapping of settled particles (Kadlec & Wallace, 2009).

Wetland vegetation, wildlife, and bacteria can only survive within their particular pH regimes (although some have specifically adapted to pH extremes), which is why wetland effluents are typically regulated to be maintained within a pH regime of 6.5 to 9.0

(Kadlec & Wallace, 2009). Macrophytes are more prone to phytotoxicity at low pH values as they tend to uptake more metals; this phenomenon occurs because metals tend to dissolve in acidic environments and are therefore more available to be absorbed by aquatic plants (Roberts et al., 2005).

2.5.4.3 Sorption

Sorption is a mechanism by which dissolved metal species are largely attenuated. Metal cations suspended in the wetland water interact with the surface of the substrate (bed media), which results with the lessening of dissolved metal content in the fluid and the deposition of said metal on the substrate surface (Ugwu & Igbokwe, 2019). Thus, the bulk of sorption occurs within the substrate media. Sorption is a general term that encompasses multiple mechanisms including diffusion and cation exchange reactions (Roberts et al., 2005). Sorption plays a bigger role in SSF wetlands than FWS wetlands due to the grater interaction of wastewater with substrate. Since the Neville St. PTS has FWS wetlands, sorption does take place, but to a lesser extent than other metal attenuation mechanisms.

In wetlands, the concentration of a pollutant in the bulk water can vary significantly from the concentration of said constituent in the substrate pore water (Kadlec & Wallace, 2009). Sorption sites can be considered renewable in the sense that more sites are added through the accretion of newly deposited sediments; however, sorption is also irreversible in the sense that sorption sites can be permanently engaged when strong bonds are formed between the sorbate and the substrate surface (Kadlec & Wallace, 2009).

Sorption isotherms developed using empirical correlations are commonly used to model the amount of sorbate that is sorbed to a surface based on the concentration of sorbate available in the fluid at a constant temperature (Ugwu & Igbokwe, 2019). Metal sorption can be modeled by three sorption isotherms (Linear, Freundlich, and Langmuir) depending on pollutant and wetland substrate (Kadlec & Wallace, 2009).

Different medias have different sorption capacities. Iron and manganese oxides (such as goethite, magnetite, and birnessite) perform well in terms of sorption capacity of metals and anions. (Ugwu & Igbokwe, 2019). A sorption substrate used in wetlands may require replacement during the lifetime of the wetland depending on pollutant loadings of the wastewater and the sorption capacity of the substrate; substrate replacement is costly and requires the removal / disposal of the used substrate (Kadlec & Wallace, 2009). Sorption capacity ranges of substrates can vary significantly and can contain a large amount of uncertainty; for example, a sorption capacity given by Kadlec and Wallace (2009) was 20 - 780 years.

2.5.4.4 Precipitation

Precipitation occurs when a dissolved chemical in solution turns into a solid. Precipitation is a mechanism for metal sequestering applied in treatment wetlands. Roberts et al. (2005) divided precipitation into two categories: (1) precipitation in the pore water, and (2) precipitation on or near the soil surface. Coprecipitation occurs when trace metals are precipitated as impurities in a mineral; for example, goethite and jarosite can contain various metals including aluminum, chromium, nickel, etc. and zinc, lead, copper, etc. respectively (Lottermoser, 2010).

There are different methods for determining whether a chemical species will precipitate or dissolve. One such method is to calculate the saturation index, which compares the ion activity product (IAP), which is a measurement of the ionic strength of a solution, to the theoretical thermodynamic solubility product constant; a positive saturation index indicates that the chemical species will tend to precipitate because it is supersaturated whereas a negative saturation index indicates that the chemical species will tend to dissolve because it is undersaturated (Roberts et al., 2005). Roberts et al. (2005) list the limitations of using this approach. Firstly, the validity of this approach depends on the availability of thermodynamic solubility data of all species. This approach also assumes that soils are at equilibrium, which is never the case as soils are continuously undergoing weathering. Moreover, this approach does not perform well with low metal concentrations. Lastly, this approach does not take into account the impact of the solid surface involved. Solid surfaces in soil, such as clay minerals, interact with the metals in solution and affect precipitation. Solid surface properties that could impact precipitation include solubility, reactivity, diffuse double layer thickness, and cation exchange capacity (Roberts et al., 2005).

Many metals (cadmium, lead, mercury, zinc, etc.) form precipitates when reacted with sulphides, which is a common anion in acid mine drainage, effectively sequestering these metals (Kadlec & Wallace, 2009). At the Neville St. PTS, precipitation is a prevalent metal attenuation mechanism in the settling pond. The pretreatment of the wastewater reduces acidity and results in net alkaline influent to the settling pond, which allows metals such as iron to precipitate into insoluble oxyhydroxides predominantly (Atkins International Ltd., 2013). Chemical analysis show that most precipitation occurs in the settling pond during normal flow regimes (Atkins International Ltd., 2013). Sedimentation of these precipitates would then occur primarily in the settling ponds, but could continue to occur in the wetlands, particularly during higher flow regimes.

2.5.4.5 Filtration

Filtration occurs when particles are removed from a fluid when passed through a multilayer lattice and become trapped by the lattice. In wetlands, the bed media filters suspended metals from the wastewater. In FWS wetlands, granular bed filtration occurs via three mechanisms: impaction, diffusion, and flow line inception (Kadlec & Wallace, 2009). For SSF wetlands, the bed media size can vary. Thus, for coarse-grained media such as gravel, flow line inception and settling will be the dominant modes for metal sequestering; for fine-grained media, impaction and diffusion will play a bigger role (Kadlec & Wallace, 2009). Filtration by macrophytes does occur in treatment wetlands. In most cases, filtration of suspended particles by aquatic plants is minor as the void fraction in this “nonhomogeneous” media is too high to be an effective filter (Kadlec & Wallace, 2009, p. 210). Since the Neville St. PTS has FWS wetlands and there is less interaction between the wastewater and the bed media, filtration does take place, but to a much lesser extent than other metal attenuation mechanisms.

2.5.5 Modeling Contaminant Removal in Treatment Wetlands

Modeling contaminant removal in wetlands is not a simple task and needs to incorporate multiple complicating factors. Wetlands are considered to be “open” systems because they are greatly impacted by environmental factors (Kadlec & Wallace, 2009). As a result, wetlands are considered to be “more complex than other types of biological treatment reactors ([such as] activated sludge, trickling filters)” that can be isolated from the environment (Kadlec & Wallace, 2009, p. 163). Another difference between wetlands and other biological reactors is that wetlands have large “biomass storage compartments” when compared to pollutant concertation (Kadlec & Wallace, 2009, p. 163). Processes that contribute to (or inhibit) treatment wetland removal performance include microbial action, chemical network reactions, volatilization, sedimentation, sorption, photodegradation, plant uptake, vertical diffusion in sediments, transpiration, accretion, etc. (Kadlec & Wallace, 2009).

When modeling treatment wetlands, one has to take into consideration two types of variability: internal (intrasystem) and intersystem variability (Kadlec & Wallace, 2009). Internal variability includes performance changes that occur as the seasons change and over the lifetime of the wetland, and are typically caused by changes in the macrophytes, hydraulic loading, and/or weather (Kadlec & Wallace, 2009). Intersystem variability is the performance variances between comparable wetlands that result due to differences in macrophytes, system geometry, and climate (Kadlec & Wallace, 2009).

There are multiple approaches to predicting contaminant removal by a treatment wetland, each with its own advantages and disadvantages. These approaches can be applied individually or in conjunction with each other. These approaches include graphical representations, mass balances, reaction rate models, and models that incorporate internal hydraulics such as plug flow models, single stirred tank (well mixed) models, and tanks-in-series (TIS) models (Kadlec & Wallace, 2009).

2.5.5.1 Graphical Representations

Treatment wetland performance can be modeled using graphical representations and are typically evaluated by comparing the outlet concentration to the inlet concentration (which is useful in determining percent removal) or by comparing the outlet concentration to areal loading, i.e., mass flow rate per area (which is useful in determining detention time or hydraulic loading) (Kadlec & Wallace, 2009). Although linear regression of data based on input and output concentrations can give an average percent removal with the associated amount of standard deviation, this method does not allow a way to understand the reasons behind the performance variability nor does it allow prediction of performance changes based on internal characteristics (Kadlec & Wallace, 2009). Nonetheless, comparing inlet and outlet concentrations using statistical analysis does allow one to determine if significant removal of contaminants is taking place within a system.

2.5.5.2 Mass Balances

Mass balances can also be used to determine percent removal and predict treatment wetland performance. Mass balances can be difficult to apply due to the numerous inputs / outputs and due to the fact that these flows are unsteady (Kadlec & Wallace, 2009). In addition, hydraulic retention times must be taken into consideration. Even though samples at the inlet and outlet are often taken at the same time, calculations based on synchronous sampling do not adequately reflect the removal performance of wetlands (Kadlec & Wallace, 2009). To employ mass balances to model treatment wetland performance, flows and concentrations are often averaged over allotted time periods (usually multiple times larger than the hydraulic retention time) and removal rates are typically averaged over the entire wetland area (Kadlec & Wallace, 2009). Similar to graphical representations, this method also does not take internal characteristics of the wetland (such as operational and environmental conditions) into consideration when predicting performance (Kadlec & Wallace, 2009).

2.5.5.3 Reaction Rate Models

Chemical reactor theory is often applied to wetlands to model hydraulic behaviour (Jamieson, 2014). Rate constants (k) are applied to model the rate by which contaminants are removed from the bulk fluid in a wetland through various processes (Kadlec & Wallace, 2009). Contaminant removal behaviour in wetlands may be zero order, first order, or second order, with most contaminants being represented by first order reactions (Jamieson, 2014). Rate constant values vary between contaminants, wetlands, and over time. Operational conditions, such as temperature, also affect rate constant values. Some rate constants at 20 ˚C can be found in literature and can be corrected to the field temperature using the Arrhenius equation (Kadlec & Wallace, 2009). Most rate constants found in literature do not consider water losses nor gains (Kadlec & Wallace, 2009).

2.5.5.4 Ideal Flow Models

Flow through lakes is often modeled as a single, well mixed tank and flow through rivers is often modeled by plug flow (Kadlec & Wallace, 2009). Notwithstanding, ideal flow models (see Figure 2.11), such as completely unmixed (plug flow reactor) or well mixed (i.e., continuously stirred tank reactor), do not adequately predict hydraulic behaviour and contaminant removal within treatment wetlands (Kadlec & Wallace, 2009). Nevertheless, initial models of wetlands assumed plug flow and have been widely used since; thus, there is much literature and data based on this overly simplistic model (Kadlec & Wallace, 2009). The simplicity of this model, along with the abundance of data available for this model, make further use of the model appealing (Jamieson, 2014). It should be noted that synchronous sampling data should not be employed to plug flow models because it does not incorporate transport delay (Kadlec & Wallace, 2009).

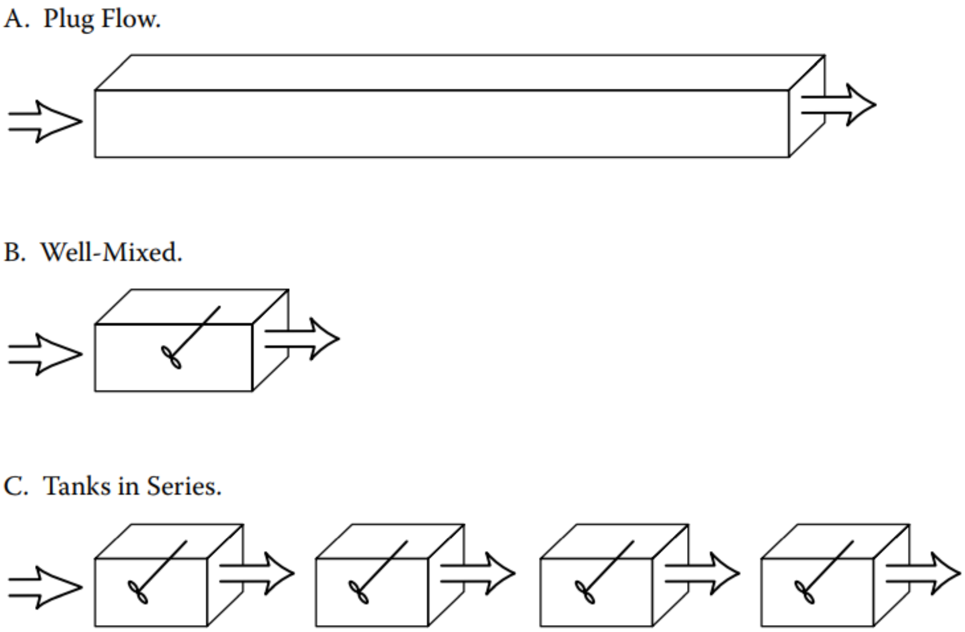

Figure 2.11 Hydraulic flow models: (A) plug flow, (B) well-mixed, and (C) tanks-in-series (Kadlec & Wallace, 2009, p. 185).

A widely used model based on plug flow is the *k*-*C** model where *k* is the first order rate constant and *C** is background concentration (Hayward & Jamieson, 2015). Although originally assumed to produce conservative estimates, plug flow models tend to predict lower effluent concentration than actual (Kadlec & Wallace, 2009). Plug flow can be used to interpolate within known data sets without adding too much error but should not be employed for extrapolation (Kadlec & Wallace, 2009).

The plug flow model can be coupled with dispersion processes, which assumes mixing occurs in a convective diffusion manner, i.e., there is no axial dispersion or back mixing (Kadlec & Wallace, 2009). Although some SSF systems have limited back mixing and thus can be modeled well by plug flow with dispersion, most wetlands (particularly FWS systems) experience axial mixing, which makes this model not a good fit (Kadlec & Wallace, 2009). Hence, neither ideal flow models, plug flow or well mixed, were applied to the Neville St. PTS wetlands, which are FWS wetlands.

2.5.5.5 Tanks-in-Series (TIS) Models

Non-ideal chemical reactor models are now considered to be the most accurate and favourable model for wetland hydraulic behaviour and contaminant removal (Hayward & Jamieson, 2015). Non-ideal chemical reactor models require first order coefficients and curve fitting using regression to determine parameters such as areal rate constant, *k* (Jamieson, 2014). A commonly used example is the tank-in-series (TIS) model (see Figure 2.12).

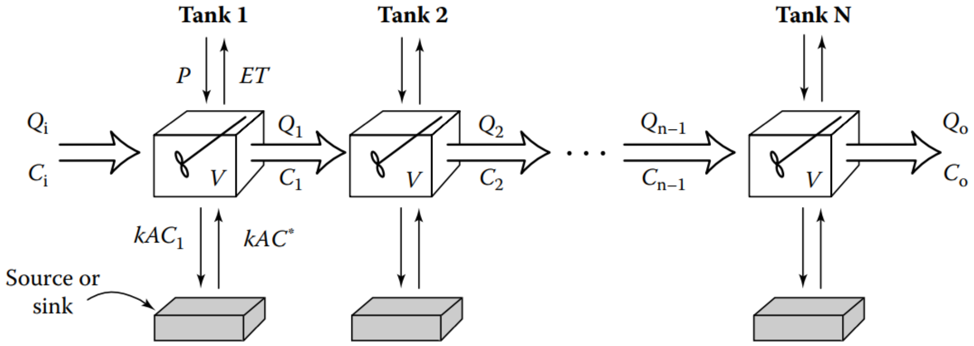

Figure 2.12 TIS model for wetland hydraulics and contaminant removal (Kadlec & Wallace, 2009, p. 189).

The TIS model accurately replicates detention time distribution with "skewed bell-shaped responses" (Kadlec & Wallace, 2009, p. 179). "The TIS hydraulic model is flexible enough to describe both mixing and preferential flow paths for a wide range of hydraulic efficiencies" (Kadlec & Wallace, 2009, p. 186). The TIS model is dependent upon the number of tanks and the mean retention time (a.k.a. hydraulic retention time, HRT, or mean detention time). A tracer test is generally conducted to determine the number of tanks, which is typically 4.1 ± 0.4 tanks for FWS wetlands (Kadlec & Wallace, 2009). Equation 2.8 is the TIS model as presented by Kadlec and Wallace (2009, p. 189):

$$\frac{(C-C^*)}{(C_i-C^*)} = \left(1+\frac{k\tau}{Nh}\right)^{-N} \tag{2.8}$$

Where:

C = effluent concentration, (mass/volume)

C^* = background concentration, (mass/volume)

C_i = influent concentration, (mass/volume)

k = first order areal rate constant, m/day

τ = actual HRT as determined from tracer tests, d

h = average water depth, m

N = number of tanks, dimensionless

The units for concentration can be any unit of mass per volume as long as the units are used consistently throughout the formula. As N becomes very large, the hydraulic behaviour becomes plug flow (Kadlec & Wallace, 2009).

Employing the TIS model to simulate treatment wetlands behaviour has numerous benefits. Firstly, the model accounts for internal and external hydraulic factors. Furthermore, the model requires sensible inputs and is simple to apply. Hayward & Jamieson (2015) developed a modified version of the TIS model to incorporate external hydrologic impacts from the watershed incrementally throughout the wetland. The wetland hydraulics are modeled as a series of well mixed tanks, each with its own hydraulic gains and losses. However, the HRT remains constant for all tanks. For this modified TIS model, the general mass balance for each tank is provided by Equation 2.9:

$$Q_{out}C_{out} = \left[Q_{in}C_{in} + \left(\frac{Q_{ws}}{N}\right)C^*\right] - \frac{k\tau Q_{out}}{Nd_w}(C_{out} - C^*) \tag{2.9}$$

Where:

Q_{out} = flow out of tank N, m^3/day

C_{out} = concentration out of tank N, kg/m^3

Q_{in} = flow into tank N, m^3/day

C_{in} = concentration into tank N, kg/m^3

Q_{ws} = watershed flow into tank N, m^3/day

C^* = background concentration, kg/m^3

N = number of tanks, dimensionless

k = first order areal rate constant, m/day

τ = actual HRT determined from the tracer tests, d

d_w = average wetland depth, m

Theoretically, the modified TIS model includes progressive watershed additions in equal amounts for each tank. A steady-state mass balance is applied to each tank to calculate the outlet flow by incorporating external hydraulic factors such as precipitation, evapotranspiration, and infiltration. Furthermore, for each tank, a relative mass addition of the modeled contaminant is associated with each hydraulic input, including wastewater influent and external hydraulic factors. The contaminant is then captured by first order sequestering processes that are constrained by the HRT (Hayward & Jamieson, 2015). The outgoing concentration of contaminant then undergoes the same process in each tank. As such, the sequential tank effluent concentrations are further reduced by dilution due to hydraulic additions and by attenuation processes in each subsequent tank. Rearranging Equation 2.9 to solve for outlet contaminant concentration, C_{out}, of each tank is shown below as Equation 2.10:

$$C_{out} = \frac{\left(\frac{Q_{in}}{Q_{out}}\right)C_{in} + \left(\frac{Q_{ws}}{NQ_{out}}\right)C^* + \left(\frac{k\tau C^*}{Nd_w}\right)}{1 + \frac{k\tau}{Nd_w}} \tag{2.10}$$

Equation 2.10 was further adapted to incorporate infiltration and evapotranspiration, which is incorporated in Equation 2.11 based on the pollutant mass balance for first order areal TIS models assuming steady state, nonuniform flow as proposed by Kadlec &Wallace (2009, p. 629):

$$C_{out} = \frac{\left(\frac{Q_{in}}{Q_{out}}\right)C_{in}+\left(\frac{Q_{WS}}{NQ_{out}}\right)C^{*}+\left(\frac{k\tau C^{*}}{Nd_{W}}\right)}{1+\frac{\tau}{Nd_{W}}(I+\propto ET+k)} \tag{2.11}$$

Where I is infiltration (m/day), α is transpiration fraction (dimensionless), and ET is evapotranspiration (m/day). Although transpiration fraction is not applicable for all contaminants, it is relevant for trace metals as macrophytes do retain metal contaminants.

2.5.5.5.1 *Hydraulic Retention Times for Modified TIS Model*

Wetland hydraulic retention time (HRL) or mean detention time, τ (d), is the average amount of time needed for water (and the constituents contained therein) to move through a wetland (Jamieson, 2014) and can be calculated according to Kadlec & Wallace (2009, p. 23) as presented by Equation 2.12:

$$\tau = \frac{V}{Q} = \frac{LWh}{Q} = \frac{Ah}{Q} \tag{2.12}$$

Where V is wetland volume (m^3), Q is flow rate (m^3/day), L is wetland length (m), W is wetland width (m), h is wetland depth (m), and A is wetland area (m^2). The recommended hydraulic retention time for FWS Wetlands is seven to 10 days (Jamieson, 2014). Wetland detention time can be categorized as actual or nominal, with nominal detention time generally being greater than actual due to inefficiencies introduced by channeling, stagnant areas, depth variations, lost volume to plant life, etc. (Kadlec & Wallace, 2009).

2.5.5.5.2 *Water Balance for Modified TIS Model*

The steady-state mass balance applied to each tank in the modified TIS model factors in watershed flow (Q_{ws}, m^3/day), precipitation (P, m/day), evapotranspiration (ET, m/day), and infiltration (I, m/day). This correlation, which is slightly modified from the water balance provided by Kadlec and Wallace (2009, p. 628) to incorporate watershed flow, is given by Equation 2.13:

$$Q_{out} = Q_{in} + Q_{ws} + A_t(P - ET - I) \tag{2.13}$$

Where Q_{out} is flow out of tank N (m^3/day), Q_{in} is flow into tank N (m^3/day), and A_t is area of tank N (m^2). Each parameter in Equation 2.13 can be calculated in theory; however, in practice, measurements are not precise, and assumptions / estimates need to be made, producing results with typical error of ± 5 – 10 % (Kadlec & Wallace, 2009).

The following discusses considerations that need to be taken into account while calculating selected parameters for the above steady-state mass balance:

- Precipitation includes rainfall and snowmelt and has two impacts on wetland performance: shortening of detention time and dilution of contaminants (Kadlec & Wallace, 2009). Rainfall is the most substantial positive contributor to the mass balance and can be estimated by using historical rainfall rates and catchment area (Kadlec & Wallace, 2009). This would include flow from surrounding land that is directed into the wetland, which is typically an additional 20 - 25 % of the wetland area (Kadlec & Wallace, 2009), unless the watershed from the surrounding area is directed away, which is common for constructed wetlands. In cooler climates, seasonal snowmelt (especially in springtime) can add a sudden load increase to the wetland mass balance.

- Evapotranspiration is the loss of water to the atmosphere through evaporation of surface water and through the transpiration of emergent plants. Evapotranspiration rates, *ET*, can be calculated based on evaporation rates, *E* (or transpiration rates, *t*), and transpiration fraction, α, as shown by Equation 2.14 which originates from Kadlec and Wallace (2009, p. 628):

  $$ET = \frac{t}{\alpha} = \frac{E}{1-\alpha} \tag{2.14}$$

  Transpiration fraction, α, is the portion of evapotranspiration that can be attributed to transpiration (Kadlec & Wallace, 2009). Evapotranspiration of a FWS wetland is similar to that of a lake or 80 % of pan evaporation (Kadlec & Wallace, 2009). In wetlands, evapotranspiration lengthens the detention time and concentrates constituents of concern. Evapotranspiration is the greatest sink affecting the mass balance (Kadlec & Wallace, 2009). In cooler climates, evapotranspiration only occurs during the warmer half of the year (Kadlec & Wallace, 2009). Evapotranspiration rates vary cyclically based on the time of year and based on the time of the day because this process is driven by heat from the sun; hence, evapotranspiration occurs at the highest rates in the afternoon and during the summer season, and at the lowest rates at night and during the winter season (Kadlec & Wallace, 2009).

- Infiltration refers to seepage losses and gains. Infiltration is more of an occurrence for natural wetlands, as constructed wetlands typically have liners to minimize infiltration. Whether there are seepage losses or gains is dependent upon whether the wetland water or the ground water has greater head pressure (Kadlec & Wallace, 2009). Magnitude of infiltration is based on the difference between the wetland water or the ground water head pressures. Infiltration rates in constructed wetlands are also affected by the permeability of the liner. If a liner is involved, a leak test can be used to determine if the liner is performing as designed.

2.5.5.5.3 First Order Areal Rate Constant for Modified TIS Model

First order areal rate constants (*k*) are applied to the modified TIS model to integrate contaminant sequestering processes. Areal rate constant values vary between pollutants, locations, and over time due to changes in operational conditions. Values for some areal rate constants at 20 ˚C can be found published in literature and can be corrected to field temperature using the modified Arrhenius equation (Kadlec & Wallace, 2009, p. 196) as seen in Equation 2.15:

$$k_T = k_{20}\theta^{(T-20)} \tag{2.15}$$

where k_T is the areal rate constant at temperature T = T °C, k_{20} is the areal rate constant at 20 °C, and Θ is the temperature correction factor. Some values for temperature correction factors are available in literature or can be approximated based on operational data.

CHAPTER 3 METHODOLOGY

The purpose of this thesis was to evaluate the post-expansion performance of the Neville St. PTS with a particular focus on peak loading events. During peak loading events, the residence time is reduced due to the higher flow rates in the system and the mine water quality is the poorest due to higher rates of AMD generation. An initial review of past sampling data collected by PS&PC was completed to help define the scope of this work and to inform later steps. This thesis was divided into three research objectives: (1) characterize system performance during peak loading events using event sampling, (2) conduct a statistical evaluation of treatment performance, and (3) develop a treatment performance model of wetlands. These three tasks were sequential, as the results from earlier tasks were needed to complete the later steps.

3.1 Event Sampling

The first research objective of this thesis was to characterize system performance during peak loading. Peak loading events are associated with conditions that result in the poorest performance of the Neville St. PTS. Peak loading events occur during the spring and fall when precipitation and snowmelt flowrates are the highest. High throughput results in low residence times, which reduces the effectiveness of contaminant removal. This is compounded by an increase in AMD production by the mine hydraulic systems that occurs simultaneously due to the increase of water runoff into the mines. Thus, these treatment systems see the worst quality feed at the same time that their residence times are at the lowest. The aim of event sampling was to gather samples during peak loading events at the Neville St. PTS and to subsequently analyze these samples to characterize the system performance during worst-case performance conditions, with a special focus on contaminant removal.

3.1.1 Location and Timing of Sampling

Locations for event sampling include all the sample points on System 1 and System 2 regularly monitored by PS&PC and a few additional sample points including the outlets of sampling pond SP-2A and SP-2B to understand how each settling pond performs individually, the diverted groundwater and overflow from System 1 entering the natural wetlands, and the combined wetland effluent from both systems. Twelve event sampling points, as depicted in Figure 3.1, were collected on twelve occasions between November of 2020 and July of 2021. System 2 cascade inlet sample (Sample #9) was not taken on several occasions due to no inlet flow; samples were taken at the base of the cascade instead. The samples at the inlet (Sample #8) and outlet (Sample #7) of the natural wetland were not taken on several occasions due to inaccessibility caused by high brook levels or high snowdrifts.

Figure 3.1 Event sampling locations.

Although some event samples were taken to become familiarized with the site/ sampling / processing samples and / or to train student researchers, most samples were taken with the intent to obtain data during peak loading events. With no way to monitor current flow rates through the Neville St. PTS, sample timing (see Table 3.1) was based on observing weather (precipitation, temperature, etc.) to identify when flowrates could possibly be high.

Table 3.1 Summary of event sampling dates.

Date	Start Time	Total Time (hr)	Description of Weather and Flow
09-Nov-20	12:00	2.5	7 ˚C, sunny, no flow through either cascade, flow bypassing old cascade (coming out underneath)
29-Dec-20	10:00	2.0	2 ˚C, overcast and showers, high flow through both cascades
17-Feb-21	10:00	1.5	-1 ˚C, cloudy, freezing rain the day and night before, snow up to knees, high flow through both cascades
23-Mar-21	14:30	1.0	12 ˚C, sunny, some snow left, high flow through both systems
01-Apr-21	09:30	1.5	4 ˚C, sunny, no snow left, very high flow through both systems
19-Apr-21	14:00	1.5	5 ˚C, sunny, low flow through both systems
26-Apr-21	13:00	1.5	10 ˚C, rainy, very high flow through both systems
17-Jun-21	11:00	2.0	21 ˚C, sunny, rained day before, low flow through old system & no flow through new system
24-Jun-21	12:00	2.0	22 ˚C, sunny, rained day before, low flow through old system & no flow through new system
02-Jul-21	09:00	1.0	11 ˚C, overcast and raining, raining all week, moderate flow through both systems
06-Jul-21	09:30	1.0	12 ˚C, sunny, no flow through either cascade, flow bypassing old cascade (coming out underneath)
23-Jul-21	09:30	1.0	16 ˚C, sunny, rained last four days, low flow through old system & no flow through new system

Sampling was completed by two people (for safety) and generally took 1.5 hours (1.0 – 2.5 hours) starting in the morning or early afternoon. Sampling in heavy snow and / or rain generally required more time simply due to accessibility being reduced by the weather and movement being restricted by additional clothing. Samples would be taken to the lab directly afterwards, where samples would be processed immediately or be stored in the refrigerator for processing the next day. Flowrates associated with each sampling date were determined after the fact.

3.1.2 Procedures for Sampling and Sample Processing

Preparation for sampling included bottle preparation. This was generally done after processing the previous batch of samples, but could be done at least 24 hours in advance. Bottles were washed and placed in an acid bath (10 % Nitric Acid). After the acid bath, bottles were rinsed with distilled water and air dried (if time allowed). Caps were placed on bottles after drying to prevent contamination. Bottles were labeled and packed in a cooler for transport to the field.

During sampling in the field, samples were kept cool using two coolers: a smaller, soft cooler with ice packs for transporting bottles in the field and a hard-sided cooler filled with ice that remained inside the vehicle. Gloves, rubber boots, and safety vests were worn in the field. Samples were only collected if it were safe to do so. Information about sampling was recorded in the field, including time and location. Each bottle was rinsed two to three times with the water to be sampled before collecting the sample. One liter samples were taken without head space whenever possible.

Velocity was measured at sampling locations using a Swoffer Open Stream Current Velocity Meter (Model 2100). Field velocity measurements were used to roughly gage whether flow was relatively high or low until actual flow rate could be verified, but were not precise enough to be used for more than relative reference. Velocity measurements were prone to high variation and were highly dependent upon where within the cross-sectional area the measurement was taken. Samples were taken at constricted points of the system, which reduced channeling and stagnation effects, but also increased turbulence and consequently augmented measurement error.

For more details about the field sampling process, see Appendix C for PTS sampling and sample processing procedures.

3.1.3 Sample Analyses

The selection of parameters analyzed for each sample (see Figure 3.2 for a photo of samples collected in one day) were based on key contaminants of concern related with AMD in the Sydney coalfield area, on parameters that have previously been inadequately removed by the Neville St. PTS, and on resources available for analysis at CBU. The selected parameters include heavy metals associated with the local ore (i.e., total iron, ferrous iron, and dissolved iron), parameters related to pyritic ore oxidation (i.e., sulphates, pH, and oxidation-reduction potential (ORP)), and general water quality parameters (i.e., true colour, apparent colour, and turbidity). The apparatuses employed to process the samples include a Themo Scientific pH meter, a Hach turbidimeter, an Hach spectrophotometer, and filtration (refer to Table 3.2 for more details).

Figure 3.2 Twelve 1 L samples collected at the Neville St. PTS on April 1st, 2021.

Acidity removal and increases in pH are primary performance markers of AMD treatment and are also essential for metals removal. As iron is a major contaminant of concern for the Neville St. PTS, total iron, ferrous iron, and dissolved iron are key indicators of treatment performance. That is to say that the performance is directly related to its ability to convert iron from its ferrous form (which is soluble) to its ferric form (which is insoluble). Figure 3.3 shows samples being analyzed for total iron. Figure 3.4 contains the 0.45 µm filters used during the filtering process required before measuring dissolved iron (and true colour). Since the ferrous form primarily occurs in reduced conditions and the ferric form primarily occurs in oxidized conditions, redox potential is another gauge of performance. Sulphates are produced through AMD generation and is a measure of original mine water quality even post treatment as settling ponds and wetlands are not effective in sulphate removal (Kadlec & Wallace, 2009).

Table 3.2 Parameters measured on event samples and the associated apparatus(es) for measurement.

Apparatus(es)	Parameter(s)
Themo Scientific Orion Star A211 pH Benchtop Meter and Orion Redox/ORP/Temp Electrodes probe with Ag/AgCl reference	pH, ORP
Hach 2100Q Portable Turbidimeter	Turbidity
Hach DR6000 Benchtop Spectrophotometer	Apparent colour, total iron, ferrous iron, sulphate
Filtration (0.45 µm) & HACH DR6000 Benchtop Spectrophotometer	True colour, dissolved iron

For measuring total iron, ferrous iron, and sulphates, 10 mL, 25 mL, and 10 mL samples were analyzed following Hach methods 8008 (Hach Company, 2014a), 8146 (Hach Company, 2019a), and 8051 (Hach Company, 2019b) respectively. For colour (apparent and true), Hach methods 8025 (Hach Company, 2014b) was utilized to process 10 mL samples. ORP data was converted to E_H values to normalize the data to a standard hydrogen electrode (SHE) by adding 200 mV to the original ORP reading (Thermo Fisher Scientific Inc., 2007).

For more details about the sample processing, see Appendix C for PTS sampling and sample processing procedures.

Figure 3.3	Analysis of samples taken on April 1st, 2021, for total iron.

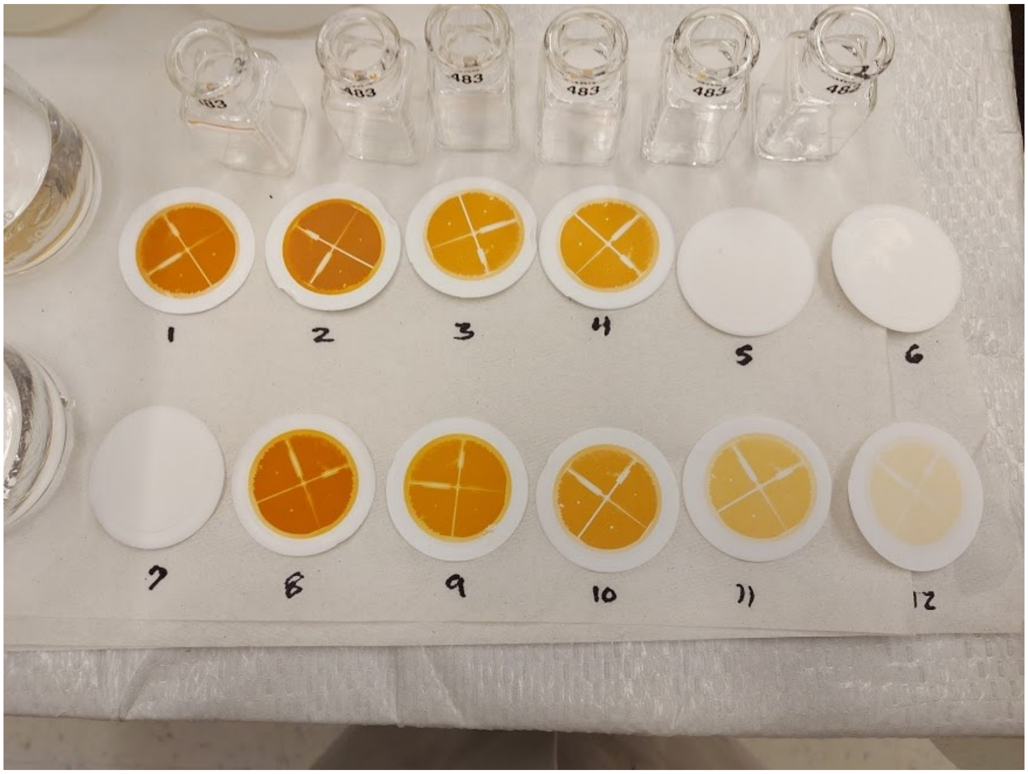

Figure 3.4	Filtering of samples taken on April 1st, 2021, for dissolved iron.

3.1.4 HOBO Pressure Transducers

HOBO pressure transducers are data loggers commonly used in water systems for continuous monitoring. HOBO pressure transducers provide information at variable frequencies about temperature and pressure (water level). HOBO pressure transducers were installed on both systems to help gauge flow rate through each system to verify if sampling took place during peak flow events and if sampling data is therefore representative of worst-case scenarios.

Two HOBO pressure transducers were placed on each system, one to monitor atmospheric pressure conditions and one to monitor water level. However, the atmospheric HOBO pressure transducer for System 2 stopped working halfway through the monitoring period and was sent off for maintenance; therefore, data collected by the atmospheric HOBO pressure transducer for System 1 was used for both systems from that point forward. For monitoring atmospheric pressure conditions, plastic models of the data logger were used while titanium versions were employed instream due to the harsher conditions. The HOBO pressure transducers were installed in the channel located downstream of the settling pond(s) and upstream of the constructed wetland for both systems. For System 1, the HOBO pressure transducers were anchored to a bridge crossing the channel (see Figure 3.5), located about 200 ft upstream of the sluice gate. For System 2, there was no good anchor point; so cinder blocks were used as anchors and the HOBO pressure transducers were placed directly before the sluice gate as depicted in Figure 3.6.

Pressure was converted to water depth at the HOBO pressure transducer location using the formula for static pressure as shown in Equation 3.1.

$$h = \frac{\Delta P}{\rho g} = \frac{P_w - P_{atm}}{\rho g} \tag{3.1}$$

Where h is water depth, ΔP is static pressure (N/m^2), P_w is water pressure at the bottom of the channel (N/m^2), P_{atm} is atmospheric pressure (N/m^2), ρ is water density (kg/m^3), and g is the gravitational constant (9.81 m/s^2). Flow rate through the channel can then be calculated based on water depth. First, the water depth upstream of the sluice gate is calculated by adjusting the water depth at the HOBO pressure transducer location to include the incline slope between the two points. For System 2, there was very little adjustment in depth between the HOBO pressure transducer location and upstream of the sluice gate; for System 1, there was a more significant adjustment due to the near 200 ft of channel length between the points. Subsequently, Bernoulli's Equation needs to be applied upstream (point 1) and downstream (point 2) of the sluice gate as demonstrated in Equation 3.2:

$$\frac{1}{2}\rho {v_1}^2 + \rho g h_1 = \frac{1}{2}\rho {v_2}^2 + \rho g h_2 \tag{3.2}$$

Where *v* is velocity (m/s) and can be related to flow rate (*Q*, m^3/s) through Equation 3.3 if *b* is channel width (m):

$$Q = v_1 h_1 b = v_2 h_2 b \tag{3.3}$$

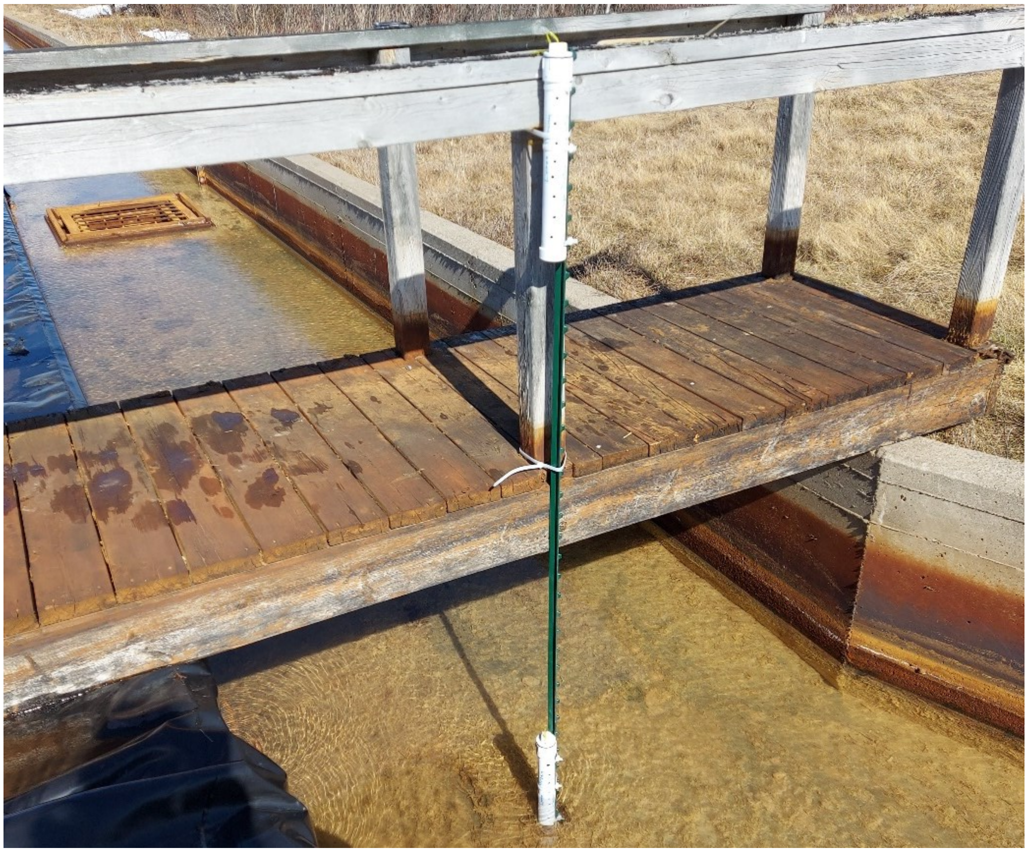

Figure 3.5 HOBO pressure transducers with low flow in channel between settling pond and constructed wetland for System 1.

Thus, flow rate can be determined by incorporating Equation 3.3 into Equation 3.2 and solving for Q as shown in Equation 3.4:

$$Q = h_2 b \left[\frac{2g(h_1 - h_2)}{1 - \frac{h_2}{h_1}}\right]^{\frac{1}{2}} \tag{3.4}$$

Figure 3.6 HOBO pressure transducers with high flow in channel between settling ponds and constructed wetland for System 2.

Lastly, for System 1 only, flow rate needed to be adjusted to incorporate a 50 cm high weir located immediately downstream of the sluice gate as seen in Figure 3.7. There is no such weir for System 2. Approximately 3 - 5 % of the cross-sectional area of the channel actually goes over the weir. This area percentage was factored into the flow rate calculation for System 1.

Note that these simple calculations are to provide relative estimates rather than absolute values, i.e., the value of these estimates is to identify peak loading events rather than to calculate exact water depth or flow rate in the channel. Discharge coefficients are often incorporated into flow calculations to account for error generated by friction, turbulence, gate opening shape, contraction coefficient, etc. For different sluice gate designs, this coefficient is usually determined empirically using different operational flow regimes. Furthermore, sedimentation and biomass due to plant growth (which was observed around the sluice gate in System 1) are other sources of error. None of these factors were taken into account in the flow rate calculations.

Figure 3.7 Sluice gate in channel between settling pond and constructed wetland for System 1.

3.1.5 Pump Flow Rate Data

In addition to using the HOBO pressure transducers to verify if event sampling occurred at peak loading rates, pump flow rate data was used as well. Event sampling was conducted between November of 2020 and July of 2021. After event sampling was completed, PS&PC provided pump run times from August of 2020 to July of 2021 and several pump calibration readings taken during this time period. Using this data, flow rates were calculated for System 1 and System 2. Then the sampling dates were compared to the calculated flow rates to determine whether samples were taken during peak loading events.

System 1 is supplied by two pumps with a design flow of 7,000 L/min; System 2 by eight pumps with a design flow of 12,000 L/min (Government of Canada, 2016). PS&PC stated that typical combined flow rates experienced by the Neville St. PTS is around 2,500 usgpm or 9,460 L/min. According to CBCL Limited Consulting Engineers (2017), average pump flow rate data from 2013 until 2017 shows that four pumps ran 50 % of the time at a rate of 2,100 usgpm or less, six pumps ran 35 % of the time at a rate of 3,400 usgpm, and nine pumps ran 14 % of the time at a rate of 5,100 usgpm (a 10th pump has been added since this analysis).

Pumps are brought online sequentially to maintain mine water level as steady mine level reduces AMD generation and maintains better mine water quality feeding the Neville St. PTS. However, during high water inputs to the mine from high precipitation or snowmelt, the level control in the 1B hydraulic system is more difficult to maintain. The first pump in the sequence directs AMD to System 1. Then several pumps direct flow to System 2. Then a second pump supplying System 1 is started. The remaining pumps are turned on as needed, feeding AMD to System 2. The flow distribution is divided such that approximately 30 % of the flow is directed to System 1 and 70 % of the flow is directed towards System 2 on average (Atkins International Ltd., 2013).

3.2 Statistical Analysis of Data

The second research objective of this thesis was to statistically evaluate treatment performance of the two parallel semi-passive treatment systems at the Neville St. PTS. Statistical analyses were applied to data following methods outlined by Berthouex and Brown (2002). Analysis of Variance (ANOVA), which is a widely used method employed across numerous fields of study (Berthouex & Brown, 2002), was the main technique utilized.

One-way ANOVAs were performed on the event sampling data for each parameter measured to determine if there was statistically significance difference between mean concentrations across both System 1 and System 2. In other words, ANOVAs were performed on each parameter measured by the event sampling to determine if each system was effectively reducing that parameter. The parameters analyzed include total iron, ferrous iron, dissolved iron, sulphates, pH, ORP, turbidity, apparent colour, and true colour. For System 1, means were compared among data collected from the settling pond inlet, the settling pond outlet, the wetland inlet, and the wetland outlet. For System 2, means were compared among data collected from the settling pond inlet, the settling pond outlets for all three successive settling ponds, and the wetland outlet.

Moreover, one-way ANOVAs were completed on total iron, ferrous iron, manganese, aluminum, sulphates, apparent colour, turbidity, pH (field and lab), ORP, alkalinity, dissolved iron, and dissolved ferrous iron to compare mean values across each year of operation for both systems at three points: the settling pond inlets, the settling pond outlets (of just the final settling pond for System 2), and the wetland outlets. This was to determine if there were statistically significance differences in operational performance between years, with a particular focus on changes in performance due to the introduction of the second system. For each system, every year of operation was included, i.e., 2009 – 2021 for System 1 and 2015 – 2021 for System 2. Additional ANOVAs were performed on only pre-expansion data and only post-expansion data for System 1 to better understand the impact on performance by the expansion. Furthermore, these analyses included data from PS&PC sampling and from event sampling for all parameters except for four. For manganese, aluminum, field pH, and alkalinity, event sampling was not performed and thus only data from PS&PC sampling was included. Data after March of 2019 from PS&PC sampling was not made available for settling pond outlets; thus, statistical analysis at these locations do not include PS&PC data from 2020 and early 2021 like the other four locations do. For any measurements that were found to be below the reliable detection limit (RDL), then half of the RDL was assumed as per common practice (Berthouex & Brown, 2002).

3.3 Treatment Performance Modeling of Wetland

The third research objective of this thesis was to develop a treatment performance model of the two constructed wetlands at the Neville St. PTS. This was done by employing a modified TIS model with first order rate constants to incorporate contaminant attenuation mechanisms (see Equation 2.11). This model was applied to data collected by PS&PC and by event sampling to quantify the iron removal performance. First, the modified TIS model was constructed by selecting values for all parameters based on historical environmental / performance data and industry best practices. Subsequently, sensitivity analyses were performed to determine model inputs for selected parameters that were not available in literature.

3.3.1 Tanks-in-Series (TIS) Model Construction

Construction of the modified TIS model involved selection of values or a range of values for each model parameter. Some parameters were determined based on historical environmental data and some parameters were selected based on assumptions aligning with industry best practices. Other parameters were chosen by taking into consideration the specific performance attributes of the two constructed wetlands in question. Two general types of parameters were included in the modified TIS model: water balance components and treatment performance components. In the following two subsections, the reasoning and assumptions behind the selection of all parameters for both water balance components and treatment performance components are explained. Afterward, the model variations for these parameters are presented to show how the model provides outputs for a range of operating conditions including worst-case, average case, and best-case performance scenarios.

3.3.1.1 Water Balance Components

Parameters necessary for incorporating water balance information into the modified TIS model (as shown in Equations 2.11 and 2.13) included design inflow, design outflow, external hydraulic contributions from watershed, HRT, total wetland area (or area per tank), water depth, precipitation rates, evapotranspiration rates, transpiration fraction, infiltration rates, and the number of tanks being used to model each constructed wetland. Explanations are provided in the below subsections as to how values were chosen for each of the above parameters.

3.3.1.1.1 Design Inflow, Q_{in} (m^3/day)

Two different inlet flow rates were chosen for both systems to represent average flow rates and maximum flow rates. Minimum flow rates were not modeled as minimum flow takes place when there is no flow through either system. As such, there is no time constraint for contaminant removal and contaminants would be reduced to the assigned background concentration. For average flow rate, the average flow was calculated based on pump flow rate data taken during the event sampling window, i.e., from August 1st, 2020, until July 31st, 2021. For maximum flow rate, both design flow rate and maximum flow rate observed from pump flow rate data were considered. As the maximum flow observed during the event sampling window was slightly higher than the design flow, the former was selected. Flow rates incorporated into the modified TIS model for both systems are available in Table 3.3.

Table 3.3 Flow rate values selected for modified TIS model for both systems.

Flow Category	System 1		System 2	
	Rate (m^3/day)	Source	Rate (m^3/day)	Source
Average Observed	5,011	PS&PC Pump Flow Rate Data	8,640	PS&PC Pump Flow Rate Data
Maximum Observed	10,714	PS&PC Pump Flow Rate Data	18,403	PS&PC Pump Flow Rate Data
Design	10,080	Government of Canada (2016)	17,280	Government of Canada (2016)

In addition, the 2009 pump maximum capacity flow was used to model a worst-case scenario that would occur if all pumps were directed into System 1 at maximum capacity, which is equivalent to 31,363 m^3/day. As this is the configuration before expansion, this gives some insight as to how the Neville St. PTS would be performing if no expansion had taken place; however, it was deemed that this scenario is currently not plausible due to configuration changes and current operational procedures.

3.3.1.1.2 Design Outflow, Q_{out} (m^3/day)

It was assumed that the design outlet flow was the same as the design inlet flow as there is only one inlet and one outlet for both systems. Also, no infiltration or leakage was assumed due to geosynthetic clay liners that were installed during construction of the settling ponds and wetlands (CBCL Limited Consulting Engineers, 2016b; Shea, 2010). Other inputs and outputs, such as precipitation and evapotranspiration, were incorporated into the model outflow by using the water balance formula as shown by Equation 2.13.

It should be noted that there is an overflow weir for System 1, located in the channel between the settling pond and constructed wetland, which is used solely during high flow events and redirects some flow from the channel to the adjacent natural wetland. This helps to control flow to the constructed wetland. High flow events sufficient to cause spillage over the overflow weir do not occur often and would be difficult to incorporate into the model as there is no way to monitor when these events happen with the current configuration. This was a known source of error in the model.

3.3.1.1.3 External Hydrologic Contribution from Watershed, Q_{ws} (m^3/day)

External hydrologic contributions from the surrounding watersheds were assumed to be negligible as the constructed settling ponds and wetlands included groundwater diversion ditches (CBCL Limited Consulting Engineers, 2016b). Due to pyritic outcrops close to surface and local bootleg mines formerly located where the new system is placed, the local ground water can have high levels of contaminants and acidity (Wood Environment & Infrastructure Solutions, 2019) which could affect the operation of the system if the watershed was allowed to flow into the Neville St. PTS. Thus, groundwater is redirected to the adjacent natural wetland. The aforementioned overflow weir for System 1 flows into the culvert redirecting the groundwater to the natural wetland.

3.3.1.1.4 Hydraulic Retention Time, HRT (d)

Although HRT was calculated for the settling pond(s) of both systems in their design phases (Atkins International Ltd., 2008; Atkins International Ltd., 2013), and measured using tracer tests for the settling pond of System 1 (Wolkersdorfer, 2011), no HRT was provided in design documentation for the constructed wetland of either system. Thus, HRT was calculated using Equation 2.12.

3.3.1.1.5 Total Area, A (m^2)

For System 1, the wetland area is approximately 12,200 m^2 (Wolkersdorfer, 2011). This value was confirmed using Google Maps and Google Earth area estimates. For System 2, wetland area, which is approximately 21,000 m^2, was determined using Google Maps (Google Maps, 2023) and Google Earth (Google Earth, 2023) area estimates.

3.3.1.1.6 Wetland Depth, d_w (m)

For System 1, wetland depth could be calculated using a wetland area of 12,200 m^2 and a wetland volume of 550 m^3 (Wolkersdorfer, 2011). However, the depth calculated from these values is extremely shallow and does not align with observations in the field. For System 2, no documentation was available that stated the depth of the constructed wetlands. However, both the System 1 design report (Atkins International Ltd., 2008) and the System 2 design report (Atkins International Ltd., 2013) refered to PIRAMID guidelines for sizing wetlands. According to these guidelines, the maximum water depths for the recommended vegetation is less than 0.25 m (PIRAMID Consortium, 2003, p. 66). The recommended plant species were confirmed to be employed in both systems, with all five species in System 1 (Wolkersdorfer, 2011) and at least four species in System 2 (Public Works and Government Services Canada, 2015). At least one species in each of each of the wetlands requires less than 0.25 m water depth. This depth also better aligns with observations made in the field. Thus, 0.25 m was the water depth assumed for both System 1 and System 2.

3.3.1.1.7 Precipitation, P (cm/day)

Minimum, average, and maximum daily precipitation rates, including both rainfall and snow fall, for all four seasons were calculated using data from “Daily Data Reports,” which was collected at meteorological station Sydney A from January of 2011 until July of 2022 (ECCC, 2023a). This station meets the World Meteorological Organization’s (WMO’s) highest standard for temperature and precipitation data collection for more than 30 years, i.e., the "3/5” rule, which states no more than three consecutive and no more than five total missing data points for each month (ECCC, 2023b).

Alternatively, a larger but older data set could have been used, i.e., the “Canadian Climate Normals 1981 - 2010,” which was also collected at meteorological station Sydney A (ECCC, 2023c). However, it was decided to use the smaller, more current data set as it overlaps with the operating window of both systems at the Neville St. PTS; hence, these precipitation rates would more accurately represent actual conditions.

Precipitation volumes (see Table 3.4) were calculated based on wetland surface area; catchment area was not included as the watershed from the surrounding area is directed away from the systems (CBCL Limited Consulting Engineers, 2016b).

Table 3.4 Daily precipitation rates for all four seasons based on “Daily Data Reports” collected by Station Sydney A (ECCC, 2023a).

Season	**Months**	**Daily Precipitation (cm/day)**		
		Minimum	**Average**	**Maximum**
Winter	Jan/Feb/Mar	0.0	5.1	9.5
Spring	Apr/May/Jun	0.0	4.0	9.1
Summer	Jul/Aug/Sep	0.0	3.0	7.7
Fall	Oct/Nov/Dec	0.0	5.3	13.6

3.3.1.1.8 Evapotranspiration, ET (cm/day)

Evapotranspiration rates were calculated based on historical evaporation rates in Nova Scotia and transpiration fraction, α, as shown by Equation 2.14. Minimum, average, and maximum daily evaporation rates for all four seasons were calculated using "Canadian Climate Normals 1981 - 2010," which are based on pan evaporation data (ECCC, 2020). In Nova Scotia, only two meteorological stations collect evaporation data, which are located in Truro and Kentville. The Truro Station meets WMO C standard with at least 20 years of complete months in the 1981 - 2010 period (ECCC, 2023d), and the Kentville Station CDA meets WMO D standard with at least 15 years of complete months in the 1981 - 2010 period (ECCC, 2023e). The maximum daily evaporation rate observed based on monthly averages was 0.4 cm/day (recorded in Kentville); the minimum daily evaporation rate based on monthly averages was 0.0 cm/day (recorded in both Truro and Kentville). Average daily evaporation rates were calculated based on the seasonal averages between the two locations and are available for each season in Table 3.5.

Table 3.5 Average daily evaporation and evapotranspiration rates for all four seasons based on "Canadian Climate Normals" data collected by the Truro and Kentville meteorological stations.

Season	**Months**	**Truro Lake Evaporation (cm/day)**	**Kentville Lake Evaporation (cm/day)**	**Average Lake Evaporation (cm/day)**	**Average Evapo-transpiration (cm/day)**
Winter	Jan/Feb/Mar	0.0	0.0	0.0	0.00
Spring	Apr/May/Jun	0.21	0.22	0.22	0.43
Summer	Jul/Aug/Sep	0.31	0.33	0.32	0.64
Fall	Oct/Nov/Dec	0.04	0.08	0.06	0.12

For transpiration fraction, a conservative value of 0.5 (i.e., it results in lower contaminant removal) was assumed to account for the impact that macrophytes have on metals attenuation. This is based on recommendations by Kadlec & Wallace (2009, p. 628), who state that for a fully vegetated wetland, "transpiration is typically about one half to two thirds of evapotranspiration." Based on this transpiration value, average daily evapotranspiration rates were calculated and are likewise available for each season in Table 3.5. The maximum daily evapotranspiration rate was calculated to be 0.8 cm/day;

the minimum daily evapotranspiration rate was calculated to be 0.0 cm/day.

3.3.1.1.9 Infiltration, I (cm/day)

No infiltration was assumed because geosynthetic clay liners were incorporated into the design of the settling pond(s) and constructed wetland of both systems (CBCL Limited Consulting Engineers, 2016b; Shea, 2010).

3.3.1.1.10 Number of Tanks, N (dimensionless)

For FWS wetlands, the number of tanks used in the TIS model is typically 4.1 ± 0.4 tanks (Kadlec & Wallace, 2009). As per industry best practice, the number of tanks used to model wetlands is normally determined using tracer tests, but tracer tests were out of scope for this project. This did contribute some uncertainty into the model. Hence, the number of tanks were estimated based on the number of passes in each wetland to incorporate non-ideal flow behaviour into the model. Systems were modeled with one cell per pass and with three tanks per cell. System 1 has one pass in its wetland (i.e., the flow travels directly from inlet to outlet with no directional changes); thus one cell with three tanks was assumed. System 2 has four passes in its wetland (i.e., the wetland has three internal berms to create sinuosity and flow direction changes three times between inlet and outlet); thus four cells with three tanks each (or 12 tanks total) were assumed.

3.3.1.2. Treatment Performance Components

Parameters necessary for incorporating treatment performance information into the modified TIS model (as shown in Equations 2.11 and 2.15) included influent concentration, target effluent concentration, background concentration, areal rate constant, temperature correction factor, and field water temperature. Descriptions can be found below as to how values were selected for each of the above variables.

3.3.1.2.1 Influent Concentration, C_{in} (mg/L)

Four different types of wetland inlet concentration values were employed in the model for both systems: (1) maximum design inlet concentration of 10 mg/L (Atkins International

Ltd., 2008; Atkins International Ltd., 2013), (2) minimum actual inlet concentrations, (3) average actual inlet concentrations, and (4) maximum actual inlet concentrations. Actual inlet concentration values for the later three categories (see Table 3.6) were calculated for each season. Event sampling data included in calculations for determining actual minimum, average, and maximum inlet concentrations was collected between November of 2020 and July of 2021. Historical sampling data collected by PSPC was also included in said calculations. For System 1, only post-expansion sampling data was used as this data represents the current operation. Therefore, PSPC data included for both systems was collected between June of 2015 and March of 2019.

Table 3.6 Constructed wetland influent iron concentrations for all four seasons based on data collected by PSPC and event sampling.

Season	**Months**	**System 1 Influent Concentration (mg/L)**		
		Minimum	**Average**	**Maximum**
Winter	Jan/Feb/Mar	0.10	1.8	5.1
Spring	Apr/May/Jun	0.10	1.4	8.6
Summer	Jul/Aug/Sep	0.05	0.31	1.3
Fall	Oct/Nov/Dec	0.11	1.0	2.6
Season	**Months**	**System 2 Influent Concentration (mg/L)**		
		Minimum	**Average**	**Maximum**
Winter	Jan/Feb/Mar	0.077	0.69	3.0
Spring	Apr/May/Jun	0.025	0.23	1.1
Summer	Jul/Aug/Sep	0.025	0.050	0.30
Fall	Oct/Nov/Dec	0.025	0.23	0.76

For System 1, maximum inlet concentrations were also determined for the entire period of System 1 operation, i.e., using pre-expansion and post-expansion data (from January of 2009 until March of 2019). This data was used to model a worst-case scenario that would occur if all pumps were directed into System 1 at maximum capacity. As this is the arrangement that existed before the facility was expanded, this provides some understanding as to how the Neville St. PTS would be performing if no expansion had taken place. For winter, spring, summer and fall, these maximum inlet concentrations values for System 1 were found to be 25 mg/L, 22 mg/L, 18 mg/L, and 24 mg/L respectively.

3.3.1.2.2 Target Effluent Concentration, C_e (mg/L)

The CCME FWAL and NSE EQS guideline values were used for target effluent concentration, which are 0.3 mg/L for iron (CCME, 2023; NSECC, 2021). This target value was compared to the output effluent concentration values generated by the modified TIS model for each constructed wetland.

3.3.1.2.3 Background Concentration, C^* (mg/L)

In selecting the value for background concentration, one must understand the purpose of this variable in the modified TIS model. "The concentration C^* is achieved when there is no net uptake or conversion of the chemical in question" (Kadlec & Wallace, 2009, p. 187). As per Kadlec and Wallace (2009), multiple factors could contribute to the presence of a nonzero background concentration depending on the contaminant: the constituent is resistant to wetland attenuation mechanisms, the chemical is associated with another nonzero background constituent such as particulate, the wetland may generate contaminant inputs (perhaps through biochemical cycles, decomposition, leaching, etc.), the constituent could be reintroduced through seasonally associated drying and wetting of wetland sediments resulting in oxidation and subsequent dissolution of minerals, and internal concentration gradients could cause recontamination of treated water when short circuiting occurs (which is more prevalent with higher loading rates). Background iron concentrations are seen in wetlands treating influent containing iron. Iron removal efficiencies vary between wetlands and increase with higher inlet loading rates at the expense of higher outlet concentrations (Kadlec & Wallace, 2009). Kadlec and Wallace (2009, p. 429) compared the iron removal performance of constructed FWS wetlands at 22 locations and found an average 78 % iron attenuation efficiency, with the higher removal rates (up to 98 %) occurring at sites with greater average inlet iron loading.

Furthermore, there is a practical consideration to incorporating a nonzero background concentration when it comes to wetland sizing. Without incorporating a nonzero background concentration, the model would size the wetlands so to remove nearly all contaminants, i.e., reach near zero concentrations, which would result in unrealistically large wetlands. So, it is generally a more reasonable, cost-effective approach to try to size treatment lagoons and wetlands in order to achieve background levels (Kadlec & Wallace, 2009).

Although surface water concentration values (given in Table 2.4) would seem to be the most appropriate selection at face value as this data was collected at 17 sites within the Sydney coalfield area that have as little anthropogenic influence as possible, surface water concentrations for constituents of concern in this region (e.g., 1.377 mg/L surface water concentration for iron) are well above limits set by guidelines. In order for our model to reflect the wetland design, the target background concentration for iron needed to be below provincial and federal guidelines of 0.3 mg/L. A lower background concentration also reflected the wetland field performances of System 1 and System 2 more adequately as the treatment facility effluent is predominantly below the guideline limits. As such, local lake chemistry data (i.e., data from Cape Breton County only) was used for reference to select a background concentration value of 0.05 mg/L (Government of Nova Scotia, 2021) as it better aligns with design limits and historical constructed wetland performances. The Nova Scotia Lake Chemistry Data for Cape Breton County was collected through lake sampling conducted from 1956 until 2018 and is funded through multiple provincial government departments to develop baseline information in regard to water quality (Government of Nova Scotia, 2021).

3.3.1.2.4 Areal Rate Constant at 20 °C, k_{20} (m/day)

Areal rate constants at 20 °C are more or less readily available depending on the constituent of concern. However, for iron, areal rate constants were very difficult to find in literature. Kadlec and Wallace (2009, p. 422) provided two areal rate constants at 20 °C for iron from two different sources: 106 m/year (or 0.29 m/day) based on data from 35 wetlands (Tarutis et al., 1999) and 38 m/year (or 0.105 m/day) based on data from one wetland system (Younger et al., 2002). As the value from Tarutis et al. was determined using a larger data set from multiple wetlands, it was selected as the original value for areal rate constant at 20 °C. Moreover, the wetlands used by Tarutis et al. (1999) to determine this value were all treating coal mine drainage. However, initial modeling results showed that modeled outlet iron concentrations generated with the Tarutis et al. value of 106 m/year were much greater than design target and historical wetland performances. Thus, a larger areal rate constant would be required to accurately represent actual wetland performances; this criterion also further eliminated the second areal rate constant given by Kadlec and Wallace (2009) of 38 m/year as this areal rate constant would result in even less iron being removed by the model. Hence, a sensitivity analysis was performed to determine areal rate constants for each system at different operating regimes. More information on the sensitivity analysis methodology for areal rate constants at 20 °C is provided in the section on sensitivity analysis of model inputs.

3.3.1.2.5 Temperature Correction Factor, θ (dimensionless)

Temperature correction factor for some constituents of concern can be difficult to find in literature as well. This was the case for iron as no temperature correction factor could be found. Therefore, a value of one was assumed for modeling all scenarios and then a sensitivity analysis was performed on multiple scenarios to assess the impact on model output. More details about the sensitivity analysis methodology for temperature correction factor are discussed in the section on sensitivity analysis of model inputs.

3.3.1.2.6 Field Water Temperature, T (°C)

Minimum, average, and maximum field water temperature for all four seasons were calculated using data from “Daily Data Reports,” which was collected at meteorological station Sydney A from January of 2011 until July of 2022 (ECCC, 2023a). This is the same source from which precipitation data was obtained. Similarly to precipitation data, an alternate data set was available that was larger but older, i.e., the “Canadian Climate Normals 1981 - 2010,” which was likewise collected at meteorological station Sydney A (ECCC, 2023c). However, it was decided to use the smaller, more current data set as it overlaps with the operating window of both systems at the Neville St. PTS; thus, these temperatures would more accurately represent actual conditions.

In addition, 2020 – 2021 field water and atmospheric temperature data was collected using HOBO pressure transducers in the channels upstream of both wetland systems. This was another alternative source for temperature data. There was no significant variation between seasonal average values for ambient and water temperatures; however, the minimum and maximum temperature values were more extreme for ambient temperatures than field water temperatures. This aligns with the described seasonal effects on water temperature as described by Kadlec & Reddy (2001). Water has a higher heat capacity than air, which has a stabilizing effect on temperature. Nevertheless, the data from the HOBO pressure transducers was only for one year and did not include the impacts of annual variation. It was decided to use annual atmospheric temperature data from "Daily Data Reports" in the performance model over the field collected data by the HOBO pressure transducers to ensure the model would more accurately consider temperature variations between years. Moreover, ambient temperature data would produce more conservative values in the model than field water temperature data. Atmospheric temperature data had more extreme minimum and maximum values, and thus would give a slighter better best-case scenario and a slighter worse worst-case scenario. Wetland field water temperatures never drop below 0 ˚C, even in northern climates with ice cover (Kadlec & Reddy, 2001, p. 544). Thus, any negative ambient values were replaced with 0 ˚C, as presented in Table 3.7.

Table 3.7 Field water temperature for all four seasons based on "Daily Data Reports" collected by Station Sydney A (ECCC, 2023a).

Season	**Months**	**Field water temperature, T (°C)**		
		Minimum	**Average**	**Maximum**
Winter	Jan/Feb/Mar	0.0[1]	0.0[1]	0.6
Spring	Apr/May/Jun	3.0	8.0	12.9
Summer	Jul/Aug/Sep	12.1	17.4	22.6
Fall	Oct/Nov/Dec	0.3	4.3	8.3

[1] Negative atmospheric temperature value was replaced with 0 ˚C.

3.3.1.3 Model Variations

While modeling iron removal across both wetlands, five parameters were treated as multiple-value variables: inlet flow, daily precipitation rates, daily evapotranspiration rates, temperature, and inlet concentrations. Two flow rates were used in the modified TIS model for each system: average and maximum flow rate as calculated based on pump flow rate data. Minimum, average, and maximum values were determined for all four seasons for the later four multiple-value variables. Using combinations of these variables, 38 scenarios were modeled for System 1 and 33 scenarios were modeled for System 2.

When developing scenarios, three multiple-value variables were matched together to form combinations that are associated with lowest contaminant removal rates, average contaminant removal rates, and highest contaminant removal rates for realistic environmental conditions. These three multiple-value variables included daily precipitation rates, daily evapotranspiration rates, and temperature, and are collectively referred to as “environmental conditions.” For average contaminant removal rates (or average environmental conditions), average values were used in the model for all three parameters. Maximum precipitation rates, minimum evapotranspiration rates, and minimum temperature were combined as they are associated with the lowest contaminant removal rates (or worst environmental conditions). Note that the worst environmental conditions replicate peak loading conditions. Minimum precipitation rates, maximum evapotranspiration rates, and maximum temperature were combined as they are associated with the highest contaminant removal rates (or best environmental conditions).

Four scenarios were developed by matching environmental conditions and inlet concentrations: (1) average environmental conditions and average inlet concentrations, (2) best environmental conditions and average inlet concentrations, (3) worst environmental conditions and maximum inlet concentrations, and (4) average environmental conditions and design inlet concentrations (i.e., 10 mg/L). Note for the second scenario that average inlet concentrations were used instead of minimum inlet concentrations as the observed minimum inlet concentrations were well below the effluent target of 0.3 mg/L; thus, there is no value in modeling these conditions as the goal is already achieved. These four scenarios were modeled using both flow rates for all four seasons: winter (January to March), spring (April to June), summer (July to September), and fall (October to December). These combinations make up the first 32 scenarios listed in Table 3.8. Scenario 33 is a best-case scenario, which combined average flow, best environmental conditions, and minimum inlet concentrations; this is the only scenario that has minimum inlet concentrations as these values are well below effluent target for all seasons.

Table 3.8 Modified TIS model configurations of multiple-value variables for all modeled scenarios.

Description	Scenario Number	Q_{in}	Environmental Conditions			C_{in}
			P	ET	T	
Avg flow / Avg env. conditions / Avg inlet conc.	1 - 4	Avg	Avg	Avg	Avg	Avg
Avg flow / Best env. conditions / Avg inlet conc.	5 - 8	Avg	Min	Max	Max	Avg
Avg flow / Worst env. conditions / Worst inlet conc.	9 - 12	Avg	Max	Min	Min	Max
Avg flow / Avg env. conditions / Design inlet conc.	13 - 16	Avg	Avg	Avg	Avg	Design
Max flow / Avg env. conditions / Avg inlet conc.	17 - 20	Max	Avg	Avg	Avg	Avg
Max flow / Best env. conditions / Avg inlet conc.	21 - 24	Max	Min	Max	Max	Avg
Max flow / Worst env. conditions / Worst inlet conc.	25 - 28	Max	Max	Min	Min	Max
Max flow / Avg env. conditions / Design inlet conc.	29 - 32	Max	Avg	Avg	Avg	Design
Best-case scenario: Avg flow / Best env. conditions / Min inlet conc.	33	Avg	Min	Max	Max	Min
Worst-case scenario: Pre-expansion max flow / Worst env. conditions / Pre-expansion max inlet conc.[1]	34	PE Max	Max	Min	Min	PE Max
Max flow / Avg env. conditions / Pre-expansion max inlet conc.[1]	35 - 38	Max	Avg	Avg	Avg	PE Max

[1] Only applicable to System 1

Avg = Average, Conc. = Concentrations, Env. = Environmental, Min = Minimum, Max = Maximum

Scenarios 34 through 38 are applicable to System 1 only and include not just post-expansion data, but pre-expansion data as well. The inclusion of these scenarios provides insight to how the Neville St. PTS would be performing if the expansion never took place. Scenario 34 is a worst-case scenario, which combined pre-expansion maximum flow (i.e., all 10 pumps were directed into System 1 at maximum capacity), worst environmental conditions, and maximum inlet concentrations. Scenarios 35 through 38 depict how the current configuration would perform for maximum flow in average environmental conditions if inlet concentrations rose to pre-expansion levels for all four seasons.

3.3.2 Sensitivity Analysis of Model Inputs

Sensitivity analyses are performed on an input variable to determine how the output parameter (outlet iron concentration) is impacted based on changes to said input variable. Sensitivity analyses were performed for two parameters that were either not available in literature or the available values did not adequately represent the performance of the wetlands at the Neville St. PTS. Parameters on which a sensitivity analysis was performed included areal rate constant at 20 °C and temperature correction factor.

3.3.2.1 Areal Rate Constant at 20 °C, k_{20} (m/day)

The areal rate constants at 20 °C that were available for iron did not adequately represent the contaminant removal performances of the two wetlands modeled and produced outlet iron concentrations much higher than historically observed for similar operating conditions. Hence, a sensitivity analysis was performed using the modified TIS model to calibrate areal rate constants at 20 °C for each system for different operating regimes. Six scenarios were modeled, each at average and maximum flow rates, based on inlet and outlet concentration combinations: (1) average observed inlet concentrations and less than target concentration of 0.3 mg/L, (2) maximum observed inlet concentrations and less than target concentration of 0.3 mg/L, (3) inlet design concentration of 10 mg/L and less than target concentration of 0.3 mg/L, (4) average observed inlet concentrations and less than average observed outlet concentration, (5) maximum observed inlet concentration and less than average observed outlet concentration, and (6) inlet design concentration of 10 mg/L and less than average observed outlet concentration. Average and maximum observed inlet concentrations can be found in Table 3.6; average observed outlet concentrations are available below in Table 3.9. Note that the background concentration value in the model had to be adjusted from 0.05 mg/L to 0.02 mg/L for the last three combinations to allow the model to simulate outlet concentrations below the target values (average effluent concentrations). The areal rate constants at 20 °C selected for each scenario listed above had to reach the specified outlet concentrations for all four seasons.

Table 3.9 Constructed wetland average effluent iron concentrations for all four seasons based on data collected by PSPC and event sampling.

Season	**Months**	**Average Effluent Concentration (mg/L)**	
		System 1	**System 2**
Winter	Jan/Feb/Mar	0.032	0.047
Spring	Apr/May/Jun	0.030	0.031
Summer	Jul/Aug/Sep	0.025	0.355
Fall	Oct/Nov/Dec	0.059	0.036

Average effluent iron concentrations for both systems were calculated based on the same data sources and using the same methodology as influent iron concentrations. Event sampling data and historical sampling data collected by PSPC were both included in the calculations. For System 1, only post-expansion sampling data was included in the calculations as this data represents the current process. PSPC data was collected between June of 2015 and February of 2021. Event sampling data was collected between November of 2020 and July of 2021.

3.3.2.2 Temperature Correction Factor, θ (dimensionless)

No temperature correction factor for iron was available in literature. Therefore, a value of one was assumed for the modeling of all scenarios and then a sensitivity analysis was performed on different scenarios to assess the impact. As most median values for temperature correction factor fall between 0.95 and 1.05 (Kadlec & Wallace, 2009), the temperature correction factor was varied between these two values to determine how outlet concentration would be affected by varying this input value. It was speculated that temperature would have little impact on iron removal performance as iron is removed more through physical processes such as sedimentation rather than more temperature dependent mechanisms like microbial action. This is similar to phosphorus, which does have temperature correction factors readily available in literature that are not greatly affected by temperature and that typically fall between 0.995 and 1.020 (Kadlec & Reddy, 2001). More research and development of temperature correction factors are needed to confirm how sensitive iron removal performance is to temperature.

CHAPTER 4 RESULTS AND DISCUSSION

Originally, a review of sampling data collected at various points throughout the Neville St. PTS process by PS&PC was conducted in order to define the scope of this thesis. This chapter will review the results of said historical sampling data review and the results of the three research objectives of this work: (1) system performance characterization during peak loading through event sampling, (2) statistical analysis of the Neville St. PTS treatment performance, and (3) performance modeling of the Neville St. PTS wetlands using the modified TIS Model.

4.1 Historical Sampling Data Review

PS&PC has been collecting and processing samples from the two systems at Neville St. PTS twice a month from multiple locations since each began to operate. The sampling frequency was reduced to once per month in April of 2019. The purpose of reviewing historical data was to refine the scope of this thesis by becoming familiarized with existing performance trends, by targeting contaminants of concern for which the facility has a record of poor removal, and by identifying gaps in the existing sampling regime.

The review included data collected by PS&PC from April of 2009 until February of 2021. Contaminants related to AMD generation include acidity, sulphates, and heavy metals. Heavy metals associated with AMD from the Sydney coalfield are predominantly iron, manganese, and aluminum. In general, the performance of the Neville St PTS is judged based on three main indicators: increase in pH, removal of acidity, and removal of heavy metals. The expansion of the Neville St. PTS was required because the original system failed to remove sufficient amounts of iron during higher flow rates due to insufficient residence times (Atkins International Ltd., 2013). Manganese removal rate was also poor, but is dependent on iron removal; aluminum removal in the original system was adequate to meet environmental guidelines (Atkins International Ltd., 2013).

Figure 4.1 contains iron concentration data for settling pond inlets and constructed wetland outlets for System 1 and System 2, which was collected by PS&PC from Jan 6th, 2011, until February 10th, 2021. The first half of the graph shows data from System 1 only as this was the only system in operation during this period. Before System 2 was brought online, the average and maximum iron concentrations at the inlet of System 1 were steadily increasing. Note multiple exceedances for outlet iron concentration much above the 0.3 mg/L target during the first half of the graph; these outlet iron concentration exceedances correspond to high inlet iron concentrations.

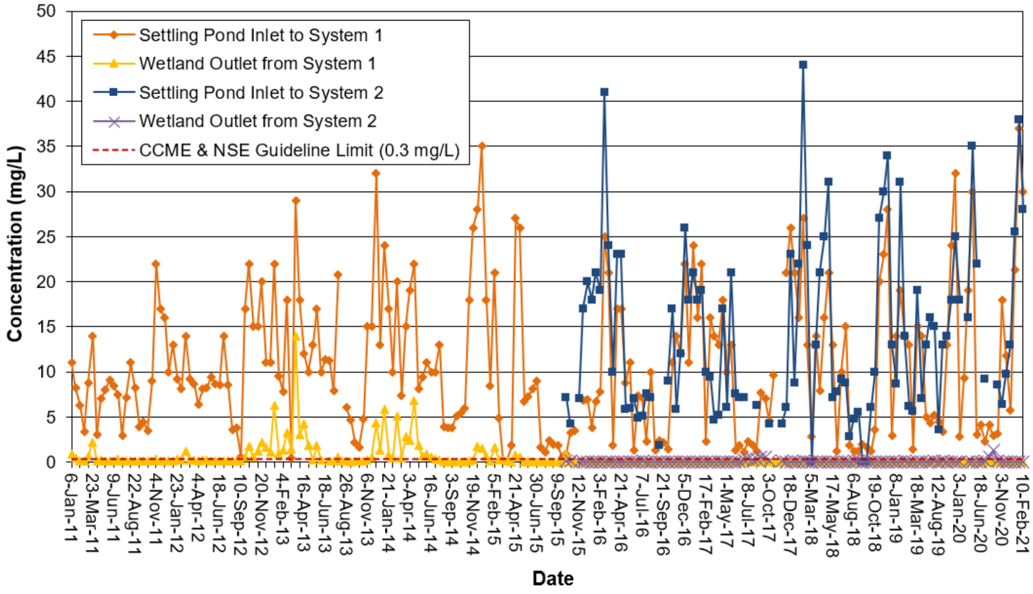

Figure 4.1 Total iron concentration (mg/L) at settling pond inlets and constructed wetland outlets.

When System 2 was brought online (at reduced rates in June of 2015 and at full rates by April of 2016), which occurs midway through the graph, there was a step change in overall performance. There was a notable, immediate drop in the average iron concentration at the inlet, but the average iron concentration still continued to increase after this step change; however, the maximum iron concentration at inlet continued to increase with no obvious impact by the expansion. Furthermore, exceedances for iron concentration at the outlets of both systems rarely happen post-expansion, i.e., the second half of the graph.

Therefore, the expansion seems to have helped reduce the inlet iron concentrations on average, possibly by providing more operational flexibility and treatment capacity, which allowed for improved management of mine water levels and thereby resulted in less

AMD generation. Other potential contributing factors to this sudden change in mine water quality include three events that occurred around the same time as the start-up of the expansion: the decommissioning of five pumping wells associated with poor mine water quality (and therefore were only used during high flow events) sometime between 2013 and 2016, the reduction of mine water generation as a result of pump sequence optimization post fall 2016 mine water quality study, and possible general improvements in mine water quality between 2013 and 2016 (CBCL Limited Consulting Engineers, 2016a).

However, maximum inlet iron concentrations continue to rise unabated, which indicate that high inlet concentrations associated with peak loading operating regimes still pose a risk. Average inlet iron concentration continues to deteriorate post-expansion. Thus, as the inlet mine water quality declines, there is a potential that it may surpass the treatment capacity of the expanded system. There is also the potential that this already occurs on occasion, but it may not be captured by the current monitoring program.

4.2 Event Sampling

Event sampling was conducted to accomplish the first research objective of this thesis: characterize system performance during peak loading. Event sampling at the Neville St. PTS took place on twelve occasions between November of 2020 and July of 2021.

Trends observed during the preliminary review of sampling data collected by PS&PC were also found to hold true for the event sampling data, as illustrated in Figure 4.2 (which is an overlay of event sampling data on Figure 4.1). Post-expansion average and maximum iron concentrations at the settling pond inlet to each system continue to rise over time while outlet iron concentrations remain consistently below the 0.3 mg/L target set by CCME and NSE guidelines.

The maximum iron concentration at the settling pond inlet for both systems during event sampling was observed on April 1st, 2021 (30.8 mg/L for System 1 and 46.3 mg/L for System 2). Samples were collected on this day because it was noted during a site visit conducted the day prior that the flow rates through both systems had significantly increased since the last sampling date, which had taken place nine days earlier on March 23rd. Although very little precipitation had occurred during the interim period, there had been snowmelt. It was also observed that the water leaving the settling ponds and entering the wetlands of both systems (depicted in Figure 4.3) was quite opaque and orange (indicating high iron content) in comparison to earlier sampling dates; however, treated water leaving the wetlands looked clear as usual.

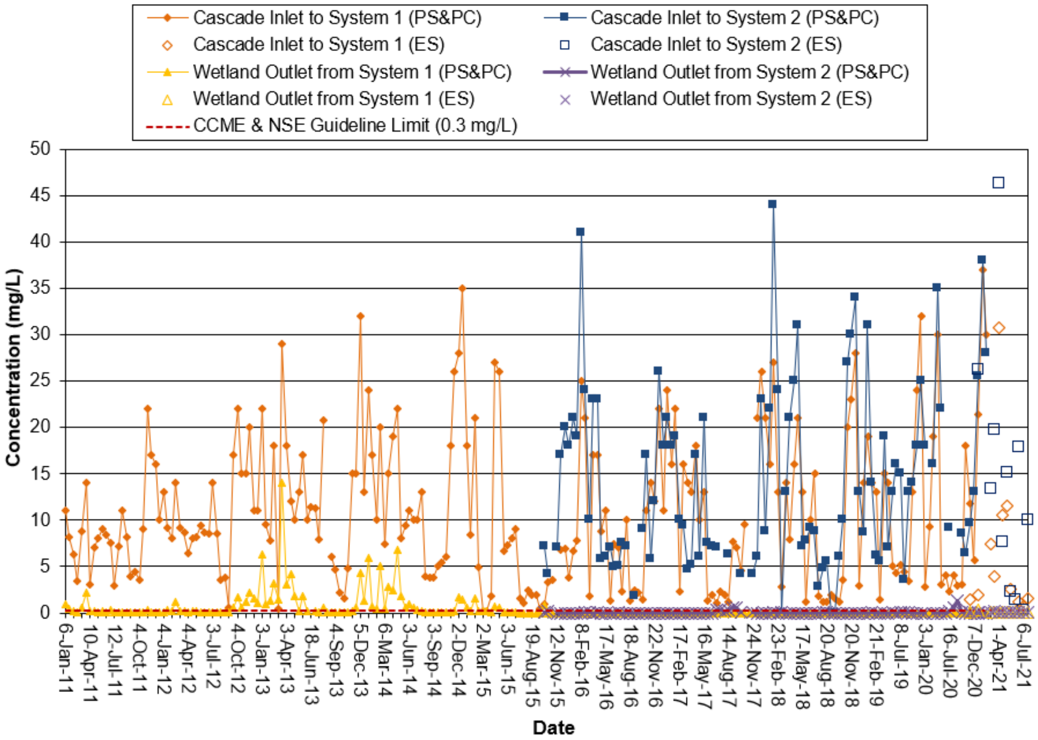

Figure 4.2 Total iron concentration (mg/L) at settling pond inlets and constructed wetland outlets, including event sampling (ES) data.

The settling pond inlet iron concentration to System 2 observed on April 1st, 2021 (46.3 mg/L) was the highest inlet concentration observed throughout the sampling history of the site. The settling pond inlet iron concentration to System 1 observed that day (30.8 mg/L) was the third highest inlet concentration observed throughout the sampling history of System 1. Wetland outlet iron concentrations observed on April 1st, 2021, remained well below the guideline maximum of 0.3 mg/L (0.01 mg/L for System 1 and 0.06 mg/L for System 2).

Figure 4.3 Wetland inlet of System 1 with high iron concentrations on April 1st, 2021.

The results of event sampling are presented by covering the following subtopics: 1) water level data, 2) pump flow rate data, and 3) physicochemical analysis data of event samples. The water level data and pump flow rate data are examined first to identify possible peak loading events. Then, event sampling analysis results are reviewed to determine if contaminants are sufficiently removed during peak loading events.

4.2.1 Water Level Data

HOBO pressure transducers were installed on both systems to estimate flow rates and to thereby identify peak loading events. Sampling dates were then cross verified with flow rates to determine if sampling data is representative of worst-case scenarios. The transducers were installed for System 1 and System 2 on March 23rd and March 30th, 2021, respectively.

Figure 4.4 shows four months of water level and flow rate data plotted as hourly moving averages for 24 consecutive hours to reduce noise as per the moving average smoothing method (Berthouex & Brown, 2002). The data was collected between March 23rd and July 31st and includes nine sampling dates: March 23rd, April 1st, April 19th, April 26th, June 17th, June 24th, July 2nd, July 6th, and July 23rd. The three samples that were taken on November 9th, December 29th, and February 17th, happened prior to the HOBO pressure transducer installations, and are therefore not included in Figure 4.4. Based on HOBO pressure transducer data, two potential sample dates could have occurred during peak loading events: April 1st and April 26th.

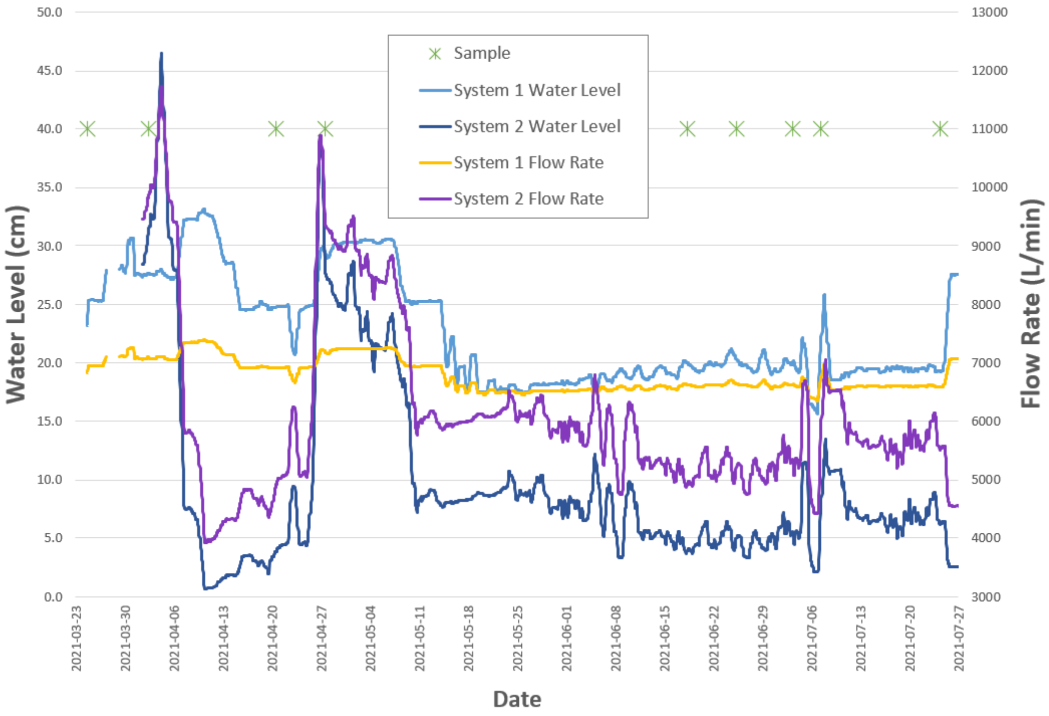

Figure 4.4 Channel water level and flow rate data from HOBO pressure transducers over four months (March 23rd – July 31st, 2021).

Figure 4.5 plots the six weeks of data from March 23rd until May 3rd, which is the period with the highest recorded throughputs through both systems. For this figure, the data is plotted as hourly moving averages for twelve consecutive hours to reduce noise. Samples were collected on March 23rd, April 1st, April 19th, and April 26th. This figure also includes ambient temperature data recorded by the atmospheric HOBO pressure transducer and precipitation data from ECCC (2023f). In regard to the two potential sampling dates occurring during peak loading events, April 1st and April 26th, the former corresponds to high snowmelt rates due to rising ambient temperatures and the later corresponds to a high rainfall event, as depicted in Figure 4.6. Nevertheless, processing of samples only showed high contaminant concentrations being experienced by the Neville

St. PTS on the April 1st sampling date.

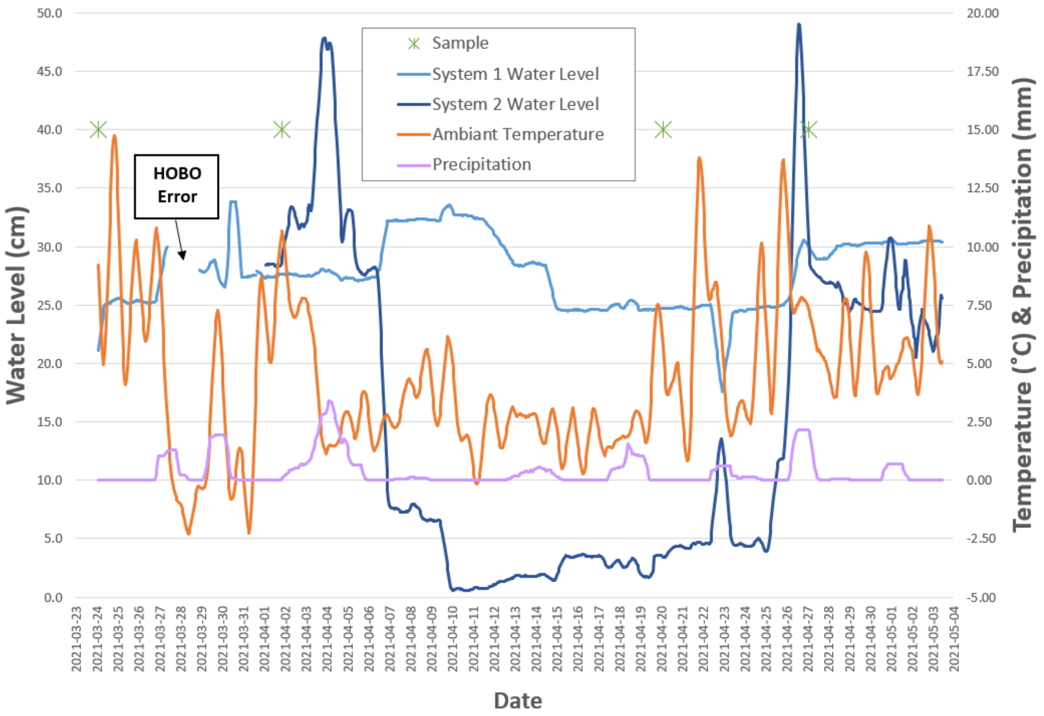

Figure 4.5 Channel water level from HOBO pressure transducers over six weeks with precipitation and temperature data (March 23rd – May 3rd, 2021).

Figure 4.6 Velocity measurement using flow meter at wetland outlet of System 1 on April 26th, 2021.

No PS&PC sampling data was received after February of 2021; hence, PS&PC data could not be compared to water levels and estimated flow rates.

4.2.2 Pump Flow Rate Data

Flow rates were also estimated for both systems using pump flow rate data from August of 2020 to July of 2021. Event sampling dates, which all fall between November of 2020 and July of 2021, were checked against flow rates to see if they happened during peak loading events.

Figure 4.7 plots a year of pump flow rate data and includes twelve event sampling dates (November 9th, December 29th, February 17th, March 23rd, April 1st, April 19th, April 26th, June 17th, June 24th, July 2nd, July 6th, and July 23rd) and seven PS&PC sampling dates (August 12th, September 14th, October 15th, November 3rd, December 7th, January 11th, and February 10th). The same two possible sample dates were identified by pump flow rate data to have potentially coincided with peak loading events: April 1st (91.6% of maximum flow rate) and April 26th. (98.0 % of maximum flow rate). Based on sample data, only the April 1st sampling date corresponded to high contaminant concentrations.

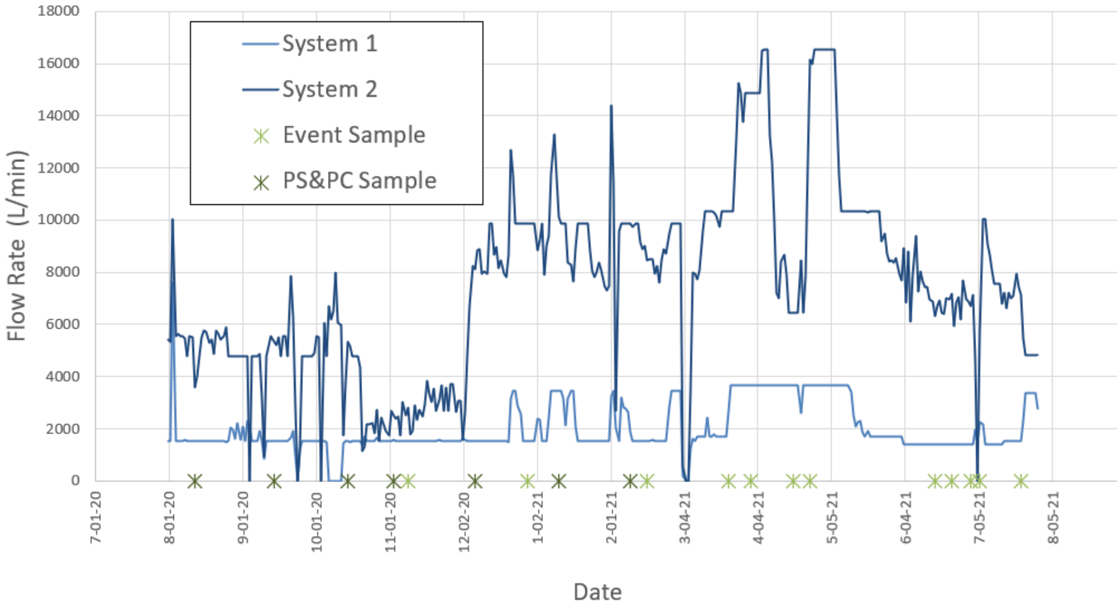

Figure 4.7 Flow rates through System 1 and System 2 based on pump flow rate data from August of 2020 to July of 2021.

4.2.3 Physicochemical Analysis Data of Event Samples

Between November of 2020 and July of 2021, event sampling was conducted on twelve occasions at the Neville St. PTS. The objective of this sampling was to measure key physiochemical parameters during high loading events to evaluate the performance of both System 1 and System 2. Table 4.1 shows the average values and standard deviations for the data collected through event sampling at three locations for each system: settling pond inlet, settling pond outlet, and wetland outlet.

Table 4.1 Event sampling data averages for both systems.

Parameter	**Units**	**System 1 Averages**			**System 2 Averages**		
		Settling Pond Inlet	**Settling Pond Outlet**	**Wetland Outlet**	**Settling Pond Inlet**	**Settling Pond Outlet**	**Wetland Outlet**
Total Iron	mg/L	6.69 ± 8.87	1.79 ± 2.59	0.05 ± 0.07	14.6 ± 13.4	0.33 ± 0.37	0.07 ± 0.06
Ferrous Iron	mg/L	2.61 ± 2.85	0.38 ± 0.49	0.21 ± 0.15	4.87 ± 5.29	0.23 ± 0.14	0.22 ± 0.12
Dissolved Iron	mg/L	4.44 ± 7.99	0.14 ± 0.28	0.03 ± 0.03	7.87 ± 7.41	0.05 ± 0.06	0.06 ± 0.05
Sulphate	mg/L	476 ± 295	471 ± 258	515 ± 230	706 ± 283	832 ± 325	798 ± 620
pH	NA	7.05 ± 0.50	8.01 ± 0.41	7.57 ± 0.27	7.28 ± 0.64	8.15 ± 0.39	7.65 ± 0.25
ORP (E_H)	mV	460 ± 122	465 ± 87.4	497 ± 108	446 ± 73.5	446 ± 66.6	487 ± 75.4
Turbidity	NTU	34.0 ± 70.3	14.4 ± 22.8	0.25 ± 0.34	31.4 ± 66.9	2.10 ± 2.47	0.30 ± 0.27
Apparent Colour	Pt/Co	129 ± 88.2	96.9 ± 105	15.8 ± 26.6	275 ± 162	49.6 ± 67.0	42.9 ± 63.8
True Colour	Pt/Co	190 ± 188	51.4 ± 103	23.8 ± 33.4	289 ± 247	19.5 ± 33.2	61.7 ± 98.6

NA = Not applicable

Physicochemical parameters analyzed include metals inherent to local mineralogy (i.e., total iron, ferrous iron, and dissolved iron), variables associated with pyritic mineral oxidation (i.e., pH, oxidation-reduction potential (E_H), and sulphates), and common water quality parameters (i.e., true colour, apparent colour, and turbidity). The data collected for each parameter in Table 4.1 is discussed in the following paragraphs, including observed trends and inconsistencies. See Appendix D for event sampling data.

Total iron concentrations measured at the settling pond inlets by event sampling ranged from 0.69 mg/L to 30.8 mg/L for System 1 and from 0.26 mg/L to 46.3 mg/L for System 2; wetland outlet iron concentrations were all below the CCME and NSE EQS limits of 0.3 mg/L with average values of 0.05 ± 0.07 mg/L for System 1, 0.07 ± 0.06 mg/L for System 2, and 0.06 ± 0.06 mg/L for the combined wetland outlet. Even on April 1st of 2021, when the highest total iron concentrations were measured at the settling pond inlets of both systems (30.80 mg/L for System 1 and 46.33 mg/L for System 2), outlet iron concentrations (0.01 mg/L for System 1 and 0.06 mg/L for System 2) were well below the federal and provincial guideline limits. Thus, based on event sampling results, the two systems are effectively reducing total iron concentrations as shown in Figure 4.8, even during the highest recorded loading rates.

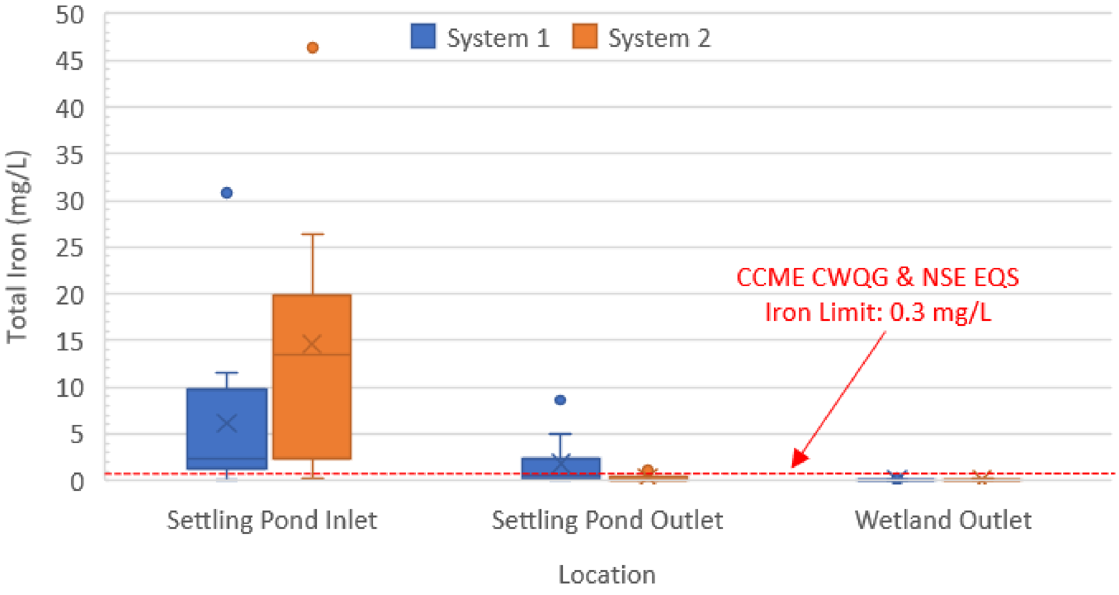

Figure 4.8 Total iron concentration reduction across both systems based on event sampling.

Figure 4.8 is a box and whisker plot which provides the following information about total iron concentrations at the three locations within the two systems: the mean value (indicated by "x" in the box), the median value (denoted by the horizontal line in the box), the interquartile range (encompassed by box), the lower and upper data points excluding outliers (designated by whiskers), and outliers (represented by circles). Interquartile range is the distance between the upper and lower quartile. Whisker length is determined using 1.5 times the interquartile range less (greater) than the lower quartile (upper quartile). Outliers are values found outside the boundaries of the whiskers.

Figure 4.9 and Figure 4.10 illustrate the decrease of iron (total, ferrous, and dissolved) as mine water traveled though System 1 and System 2 on April 1st, 2021. All three parameters are predominantly captured in the settling pond(s) with some attenuation taking place in the constructed wetland. Based on event sampling results shown in Table 4.1, the average reduction of total iron by the System 1 settling pond was from 6.69 ± 8.87 mg/L to 1.79 ± 2.59 mg/L; by the System 2 settling ponds, from 14.6 ± 13.4 mg/L to 0.33 ± 0.37 mg/L. The total iron concentrations at the settling pond outlets of both systems were always less than the design target of 10 mg/L, with the highest total iron concentration at the settling pond outlets being recorded on April 1st at 8.60 mg/L for System 1 and 5.37 mg/L for System 2. Note that although System 2 generally receives poorer quality mine water (as shown in Figures 4.8, 4.9, and 4.10) and at higher flow rates (as shown in Figure 4.7) than System 1, the settling ponds of System 2 are more effective at reducing iron content than System 1; this is because the area of the System 2 settling ponds is approximately three times greater than that of System 1.

Event sampling data in Table 4.1 shows that dissolved iron is almost completely removed by the time mine water is exiting the settling pond(s), with the average dissolved iron concentrations at settling pond outlets being 0.14 ± 0.28 mg/L and 0.05 ± 0.06 mg/L for System 1 and System 2 respectively. This indicates that soluble iron is converted to insoluble iron, which is easily removed by the settling pond through sedimentation. Event sampling data also reveals that ferrous iron behaves similarly to dissolved iron, which is expected as ferrous iron is soluble. Ferrous iron removal principally occurs in the settling pond(s) of each system with little to no change in ferrous iron concentration within the constructed wetland. Ferrous iron values were observed to be higher than total iron values on occasion, particularly for low value readings such as seen at the constructed wetland outlets. The total iron and ferrous iron concentration readings measured at the wetland outlets are close to the lower limit (0.02 mg/L) of the applicable testing range for the two analysis methods employed; this could be a contributing factor to the noted error. Incongruities between ferrous iron and total iron were also observed in PS&PC data; PS&PC attributed these inconsistencies to the different analytical methodologies and considered the error to be acceptable as long as the results were within 25 % percent difference.

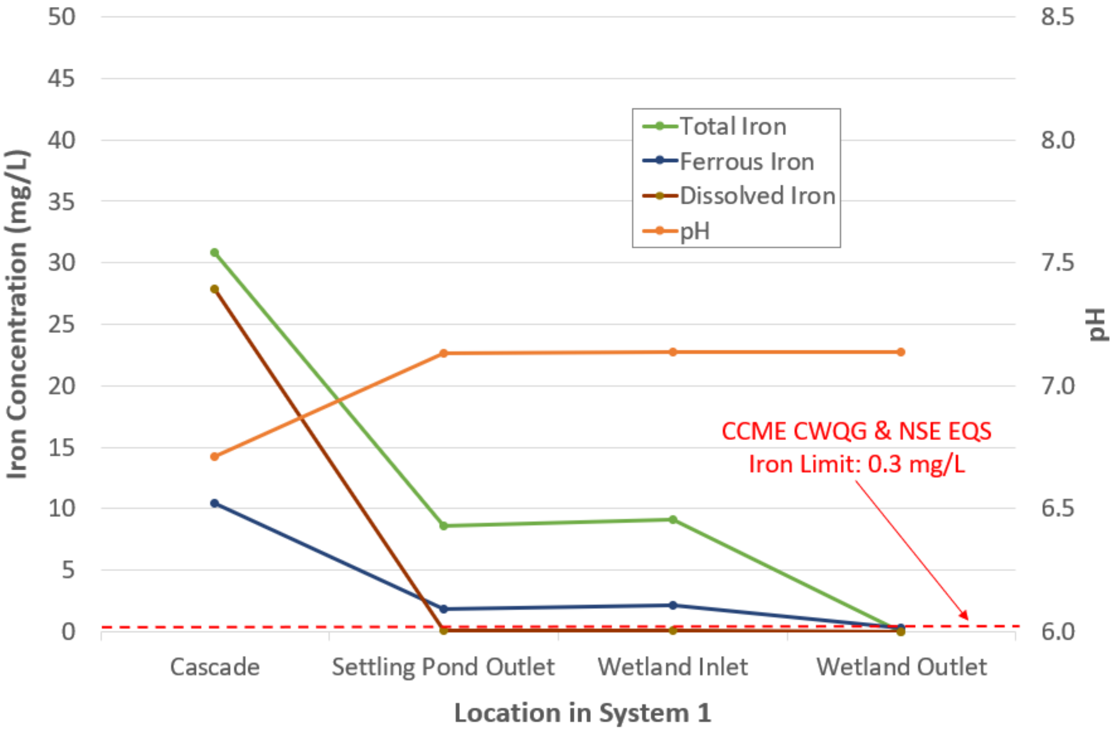

Figure 4.9 Performance of System 1 on April 1st, 2021.

Figure 4.9 and Figure 4.10 also demonstrate how pH on April 1st, 2021, initially increases after entering the settling pond(s) of each system, but then stays relatively constant after leaving the settling pond(s). The settling pond inlet sample is taken just after caustic soda addition and aeration from the cascade; hence, the pH is still adjusting due to addition of caustic soda (which decreases acidity), due to degassing of the mine water (which reduces carbonic acidity), and due to metals removal through oxidation and hydrolysis (which increases mineral acidity). Figure 4.11 shows the average pH change across the settling pond(s) observed through event sampling as a result of all these reactions, which is approximately one pH unit increase (refer to Table 4.1). This behaviour aligns with a previous performance analysis performed by Atkins International Ltd. (2013), which also observed an increase of one pH unit across the settling pond of System 1. In the System 1 settling pond, the average pH rises from 7.05 ± 0.50 to 8.01 ± 0.41; in the System 2 settling ponds, from 7.28 ± 0.64 to 8.15 ± 0.39. The pH is relatively more stable across the wetland with a comparatively smaller reduction in pH, indicating that some oxidation and hydrolysis reactions occur in the wetland of each system. This could potentially be due to macrophytes increasing dissolved oxygen content in the water, which promotes oxidation reactions. Since iron data from event sampling indicates that the bulk of the iron removal occurs in the settling pond(s) of each system, this reduction in pH across the wetland of each system could be due to oxidation and hydrolysis reactions of other metals such as aluminum and manganese. Federal and provincial guideline limits for pH are 6.5 to 9.0 for fresh water (7.0 – 8.7 for marine waters); the average wetland outlet pH readings measured by event sampling were 7.57 ± 0.27 for System 1, 7.65 ± 0.25 for System 2, and 7.58 ± 0.44 for the combined effluent from both systems. Thus, wetland outlet pH readings were consistently within guideline limits for both systems.

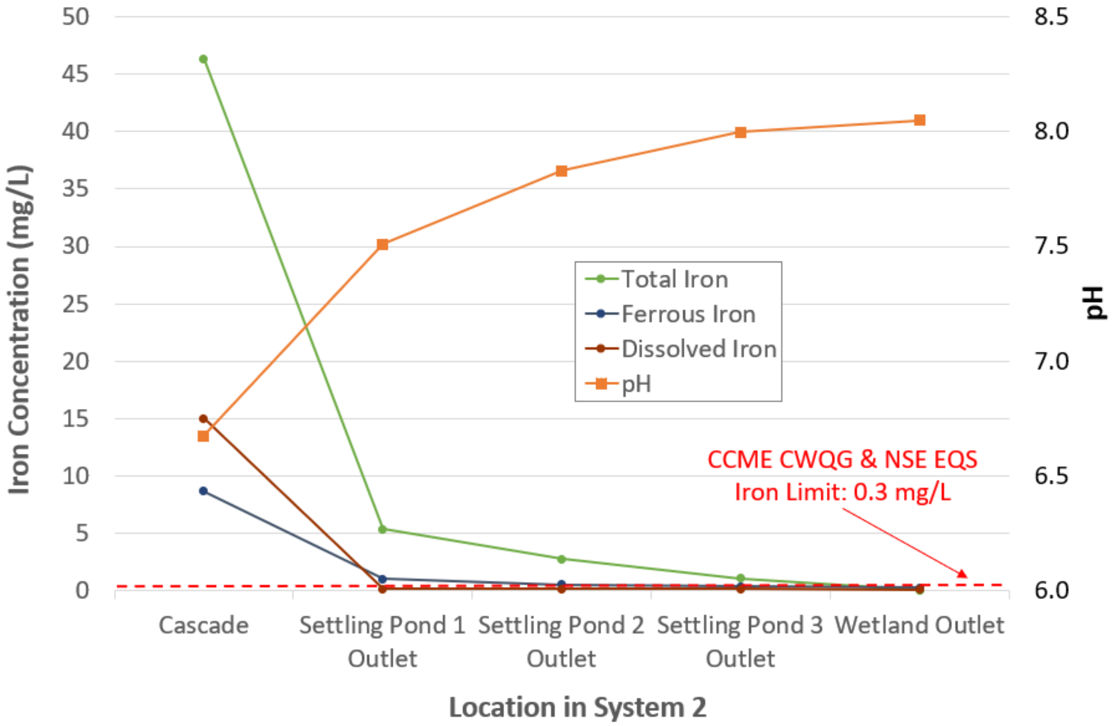

Figure 4.10 Performance of System 2 on April 1st, 2021.

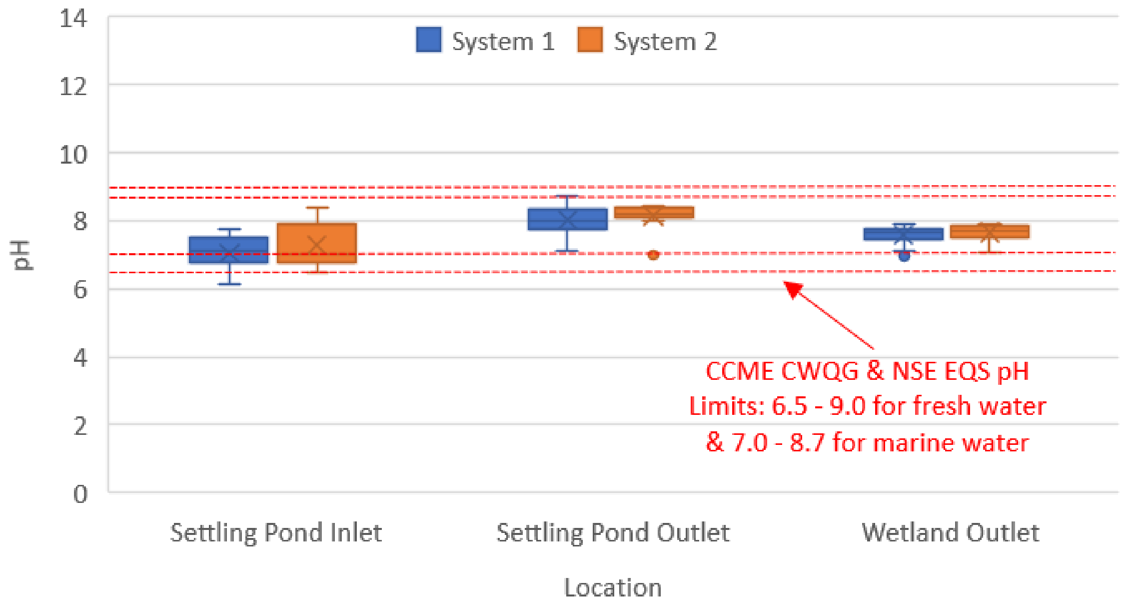

Figure 4.11 pH changes across both systems based on event sampling.

Based on event sampling (refer to Table 4.1), ORP (E_H) falls within the expected range for insoluble iron precipitates based on the iron Pourbaix diagram (Pourbaix diagrams, 2023), indicating an oxidizing environment which is beneficial for iron attenuation through precipitation and sedimentation. ORP does not vary much across both systems, which is not surprising as all samples are taken downstream of caustic addition and aeration, both of which contribute to creating said oxidizing environment.

Sulphates are a byproduct of AMD generation, but historically PTSs have not been designed to reduce this parameter. Settling ponds and wetlands are not effective at removing sulphate, as evidenced in Figure 4.12 by the lack of significant reduction. Average sulphate concentrations at the wetland outlets for System 1 and System 2 were 515 ± 230 mg/L and 798 ± 620 mg/L respectively. However, sulphate concentration is an indication of the severity of contamination of the mine water through AMD production. Figure 4.12 also demonstrates that System 2 generally treats poorer quality mine water than System 1. As System 2 is larger and has greater treating capacity, the poorest quality mine water is deliberately directed towards System 2 as part of the operational strategy.

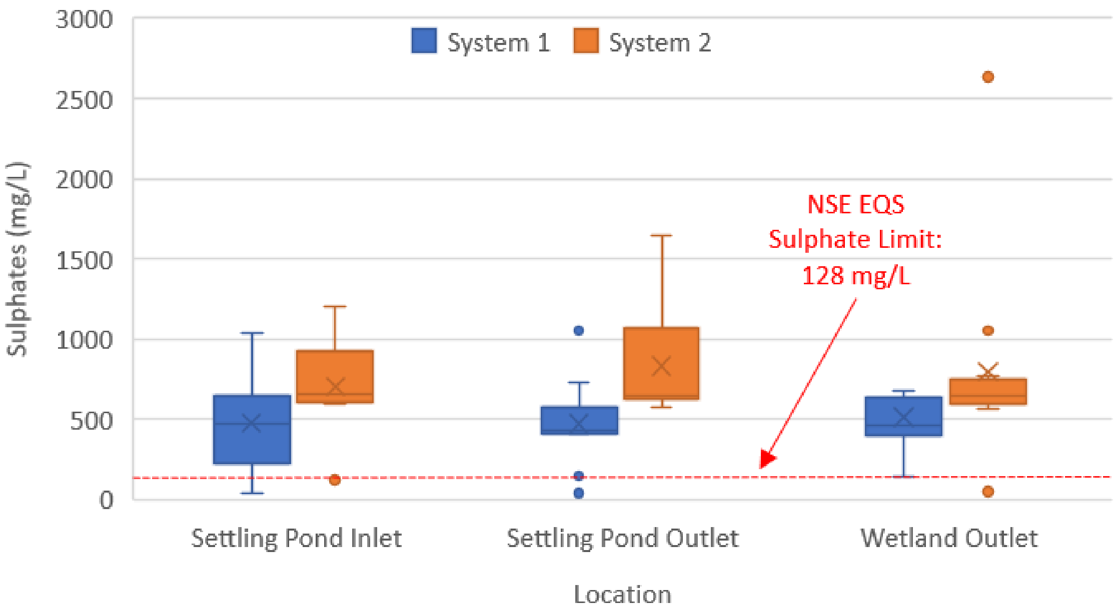

Figure 4.12 Sulphate concentration across both systems based on event sampling.

In September of 2021, NSE Tier 1 EQS adopted a new limit for sulphates of 128 mg/L whereas prior to this there was no limit (NSECC, 2021). As all event samples were taken before this revision, the sulphate concentrations technically do not exceed any limits, but this could be a potential performance issue in the future. Nonetheless, the guideline published by British Columbia Ministry of Environment and Climate Change Strategy (BCMOECCS), from which the NSE Tier 1 EQS limit was adopted, does take into account water hardness. For example, the 128 mg/L limit for sulphate adopted by Nova Scotia is for very soft water, i.e., 0 – 30 mg/L, but there is also a 429 mg/L limit for sulphate for very hard water, i.e., 181 – 250 mg/L (BCMOECCS, 2013). As the hardness of the treated water from Neville St. PTS (based on PS&PC sampling data) is typically much greater than the “very hard water” range from BCMOECCS guideline, then future evaluations may consider water hardness and deem sulphates concentrations to still be within acceptable range.

Turbidity, apparent colour, and true colour are all indicators of general contaminant concentration. All three parameters show a significant decrease across both systems (see Table 4.1), indicating a general decrease in contaminants and an overall improvement in mine water quality. These parameters are correlated to iron concentration as iron content increases opacity and colours the water bright orange. This phenomenon can be visually observed on location as the entrance of the settling ponds is generally opaque and orange with iron deposits and the outlets of the wetlands are always clear. The high opacity and orange colour permeate further into both systems with higher throughput. Both the CCME and NSE Tier 1 EQS for fresh and marine waters are the same for apparent and true colour and state that the value should not be "significantly higher than the seasonally adjusted expected value for the system" (CCME, 2023). Without specific background data for colour for comparison, the wetland outlet colour readings were not compared to a particular reference target, but were compared to settling pond inlet values to determine if colour is being improved by the Neville S. PTS. For turbidity of fresh and marine waters, the only applicable guidelines are from CCME and are also based on background concentrations. Again, specific background data is not available for reference. However, the guideline states that the value cannot be 8 NTUs above the background value for the short term and 2 NTUs above the background value for the long term. Moreover, as per Table 4.1, the wetland outland turbidity readings from System 1 and System 2 are 0.25 ± 0.34 NTUs and 0.30 ± 0.27 NTUs respectively. Hence, the turbidity of treated mine waters from the Neville St. PTS is well below guideline limits according to event sampling.

Event sampling was also conducted at the inlet of the natural wetland (seven data points collected) and at the outlet of the natural wetland (six data points collected). The natural wetland inlet is a culvert that continuously receives diverted groundwater from around System 2; periodically during high flow events, partially treated overflow from System 1 is also diverted to the natural wetland through this culvert via an overflow weir located in the channel between the settling pond outlet and the constructed wetland inlet. Five of the seven sample data points were taken when there was no overflow from System 1; the two other data points were taken when there was overflow from System 1. For all data collected during event sampling, the natural wetland inlet readings for total iron (3.72 ± 1.55 mg/L) were greater than those recorded at the settling pond outlet (1.79 ± 2.59 mg/L). Thus, the diverted groundwater had higher total iron concentrations than the partially treated mine water from the overflow weir in System 1, and the partially treated mine water had a diluting effect. This observation aligns with the noted high background concentrations for certain heavy metals such as iron in the Sydney coalfield area (refer to Table 2.4). It should also be noted that the total iron readings collected at the natural wetland outlet (0.51 ± 0.20 mg/L) exceeded the guideline limit of 0.3 mg/L five of six times; in fact, the only time that the reading was below the limit was the one time that the natural wetland outlet sample was collected while there was overflow from System 1.

4.3 Statistical Analysis of Data

Statistical analysis was completed to accomplish the second research objective of this thesis: statistically evaluate treatment performance of the two parallel semi-passive treatment systems at the Neville ST. PST, with a special focus on contaminant removal. Statistical analysis of sampling data is subdivided into two sections: the statistical analysis of event sampling and the statistical analysis of all sampling data, including data from PS&PC and from event sampling. First, event sampling data was analyzed using one-way ANOVAs to determine if parameters are being impacted by treatment by comparing means at different locations along each system. Secondly, one-way ANOVAs were performed on both systems using all data to compare variance between yearly means.

4.3.1 Statistical Analysis of Event Sampling Data

Performance parameters measured by event sampling across both systems at the Neville St. PTS were compared using statistical analysis to determine if the AMD is being sufficiently treated. One-way ANOVAs were performed on event sampling data to evaluate if there was statistically significance difference between mean concentrations across System 1 and System 2. The parameters analyzed include total iron, ferrous iron, dissolved iron, sulphates, pH, ORP, turbidity, apparent colour, and true colour. The results are presented in Table 4.2 and are discussed in the following paragraphs, including noted trends and irregularities. For System 1, data was collected from four points along the process: the settling pond inlet, the settling pond outlet, the wetland inlet, and the wetland outlet; for System 2, data was collected from five points along the system: the settling pond inlet, the settling pond outlets for all three successive settling ponds, and the wetland outlet.

Table 4.2 P-values for event sampling parameters across each system.

Parameter	**System 1**		**System 2**	
	P-Value	**Statistical Significance**	**P-Value**	**Statistical Significance**
Total Iron	0.011	Yes	<0.001	Yes
Ferrous Iron	<0.001	Yes	<0.001	Yes
Dissolved Iron	0.024	Yes	<0.001	Yes
Sulphates	0.975	No	0.853	No
pH	<0.001	Yes	<0.001	Yes
ORP (E_H)	0.814	No	0.500	No
Turbidity	0.232	No	0.115	No
Apparent Colour	0.023	Yes	<0.001	Yes
True Colour	0.003	Yes	<0.001	Yes

The two major indicators of performance for the Neville St. PTS are pH correction and contaminants removal. The one-way ANOVAs determined that there was a statistically significant increase in pH and statistically significant removal of all forms of iron (total, ferrous, and dissolved). Turbidity, apparent colour, and true colour are gages of overall contaminant concentrations, and the latter two are reduced in a statistically significant manner. Turbidity averages do show a decrease across each system, with a total reduction from 34.0 ± 70.3 NTU to 0.25 ± 0.34 NTU for System 1 and from 31.4 ± 66.9 NTU to 0.30 ± 0.27 NTU for System 2, but the difference was determined to be statistically insignificant.

Sulphate concentrations, which act as a surrogate parameter for estimating the original severity of AMD production, did not show statistically significant reduction across both systems. This behaviour was anticipated as settling ponds and wetlands are ineffective at removing sulphates. As the Neville St. PTS is not intended to remove sulphates, this has not been a concern in the past. However, Nova Scotia adopted a sulphate guideline from British Columbia of 128 mg/L in September of 2021 (NSECC, 2021), i.e., after event sampling was completed. Moving forward, sulphate concentrations of treated water from Neville St. PTS may or may not be a concern as discussed in the previous section on event sampling analysis results.

Likewise, ORP (E_H) does not change across both systems in a statistically significant manner. This can be attributed to all the samples being taken after caustic addition and aeration, which are the treatments that affect ORP. Thus, due to the nature of the sampling point locations, the event sampling data does not reflect the full effect of said treatments on mine water ORP.

4.3.2 Statistical Analysis of All Sampling Data

One-way ANOVAs were applied to performance parameters to compare mean values across each year of operation for both systems at three locations: the settling pond inlets, the settling pond outlets, and the wetland outlets. Years of operation for System 1 consist of 2009 to 2021; System 2, 2015 to 2021. Parameters analyzed include total iron, ferrous iron, dissolved iron, dissolved ferrous iron, manganese, aluminum, sulphate, pH (field and lab), alkalinity, ORP, turbidity, and apparent colour. Data incorporated into the statistical analysis comparing between operational years includes that provided by PS&PC sampling and that by event sampling, with the exception of four parameters (manganese, aluminum, field pH, and alkalinity), which only include data from PS&PC sampling.

It should be noted that some years had fewer data points than others, which could be a source of error. For the System 1 settling pond inlet only, the data for the latter half of 2013 was not included in the data shared by PS&PC for an unknown reason. Thus, 2013 only has about half the data points. In addition, the collection of settling pond outlet samples ceased for both systems in April of 2019. Therefore, 2019 only has a quarter of the sampling data for these locations. Moreover, 2020 and 2021 data for these locations is solely from event sampling with only two and 10 data points respectively; thus, 2020 data for these locations was excluded from analysis to avoid introducing error due to insufficient data points.

Furthermore, sampling frequency at all locations was reduced from twice a month to once a month in April of 2019. Hence, 2019 and 2020 only have about half the sampling data.

The last data supplied by PS&PC was for February of 2021; therefore, 2021 only has two data points from PS&PC. However, 2020 and 2021 also have event sampling data. Consequently, these two years have about half the data points for most parameters evaluated by one-way ANOVAs. For parameters that have only two data points for 2021 (i.e., parameters based on PS&PC data alone as indicated in Table 4.3), 2021 was excluded from analysis to prevent the error it would have introduced due to lack of data points. Lastly, for several parameters, the readings from the settling pond and wetland outlets were often below the RDL. In these cases, half of the RDL was used. This assumption could have introduced error as well.

4.3.2.1 Statistical Analysis of All Sampling Data for System 1

One-way ANOVAs were conducted on both systems across all years of operation to compare yearly mean values of performance variables at three locations. The results of these one-way ANOVAs for System 1 are available in Table 4.3 and are discussed in the following paragraphs, including observed trends and incongruencies. There are two distinct phases of operation for System 1: prior to expansion and post-expansion. Additional one-way ANOVAs were performed for each parameter on pre-expansion data only (before June of 2015) and on post-expansion data only (after June of 2015) to better understand the impact of the expansion on operational performance.

For all forms of iron (total, ferrous, dissolved, and dissolved ferrous), one-way ANOVAs on System 1 showed no statistically significant difference between mean values across years of operation at the settling pond inlet, but did show statistically significant difference at the settling pond outlet and the wetland outlet. For these four parameters, box and whisker plots of sampling data were created and illustrate similar behaviour at all three locations in System 1. Prior to expansion, the average and maximum iron concentrations at the settling pond inlet (illustrated in Figure 4.13 for total iron) and at the settling pond outlet of System 1 were steadily increasing. Moreover, the wetland outlet of System 1 experienced the highest average and maximum iron concentrations for the pre-expansion years in the last two full years of operation (depicted in Figure 4.14 for total iron).

Table 4.3 P-values for selected parameters across each year of operation for System 1.

Parameter	**System 1**					
	Settling Pond Inlet		**Settling Pond Outlet**		**Wetland Outlet**	
	P-Value	**Statistical Significance**	**P-Value**	**Statistical Significance**	**P-Value**	**Statistical Significance**
Total Iron	0.116	No	<0.001	Yes	<0.001	Yes
Ferrous Iron	0.195	No	<0.001	Yes	<0.001	Yes
Dissolved Iron	0.194	No	<0.001	Yes	<0.001	Yes
Dissolved Ferrous Iron	0.136	No	<0.001	Yes	<0.001	Yes
Manganese[1]	<0.001	Yes	<0.001	Yes	<0.001	Yes
Aluminum[1]	<0.001	Yes	0.278	No	<0.001	Yes
Sulphate	0.002	Yes	<0.001	Yes	0.317	No
pH	0.017	Yes	<0.001	Yes	<0.001	Yes
Field pH[1]	<0.001	Yes	<0.001	Yes	<0.001	Yes
Alkalinity[1]	<0.001	Yes	<0.001	Yes	<0.001	Yes
ORP (E_H)	<0.001	Yes	<0.001	Yes	<0.001	Yes
Turbidity	0.260	No	<0.001	Yes	<0.001	Yes
Apparent Colour	0.077	No	<0.001	Yes	<0.001	Yes

[1] Parameter based on PS&PC data alone.

Upon the addition of System 2, the average and maximum iron concentrations decreased immediately at all locations in System 1 and then either continued to increase (i.e., at the settling pond inlet) or remained relatively constant (i.e., at the settling pond and wetland outlets). In regard to the wetland outlet of System 1 post-expansion, most of the PS&PC data for all forms of iron was below the RDL and all total iron data was below the federal and provincial limit of 0.3 mg/L (except one data point in 2015 when System 2 was still only partially online as per the staggered commissioning process employed), which was not the case prior to expansion. The average total iron at the wetland outlet of System 1 pre-expansion was 0.75 ± 1.63 mg/L; post-expansion, 0.04 ± 0.08 mg/L.

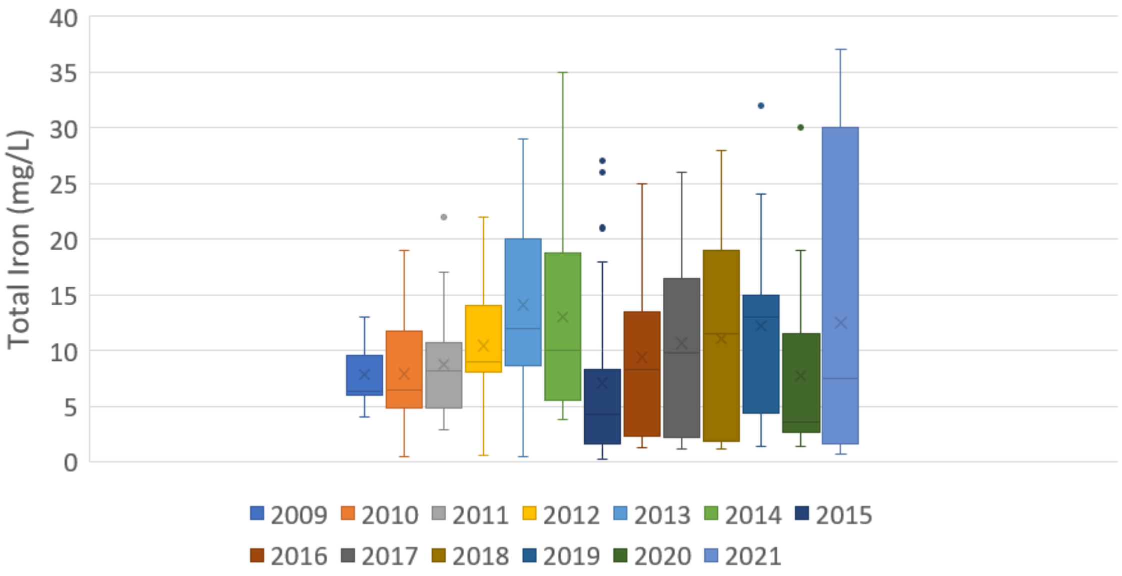

Figure 4.13 Total iron at settling pond inlet of System 1 from 2009 until 2021.

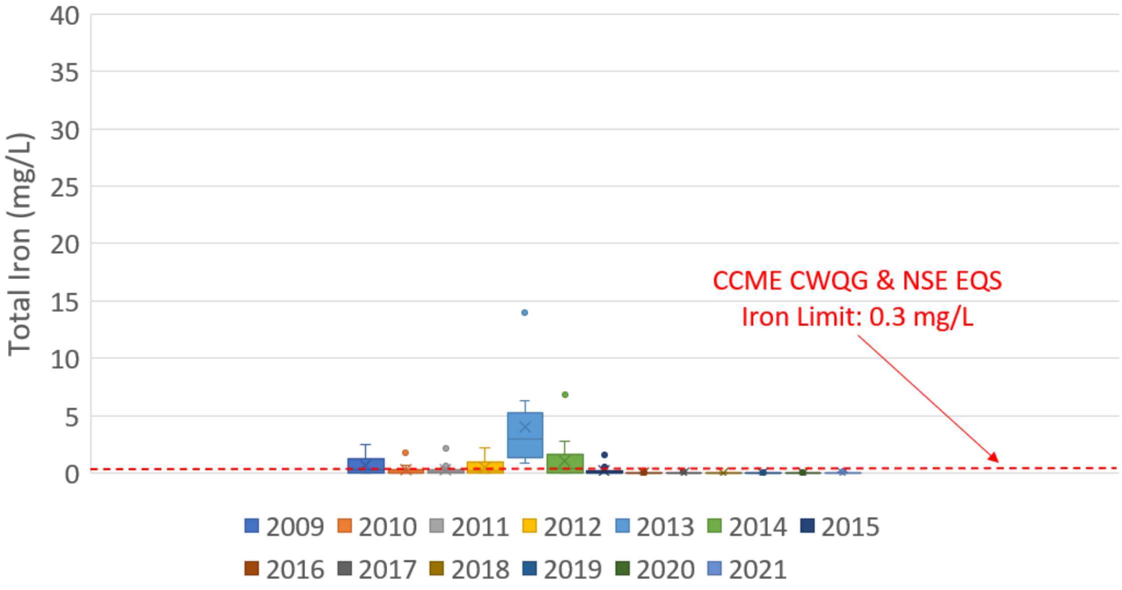

Figure 4.14 Total iron at wetland outlet of System 1 from 2009 until 2021.

Thus, at the settling pond inlet of System 1, although the concentration of iron seems to be increasing over the years (in terms of both average and maximum values) with a temporary improvement upon adding the expansion as shown by the box and whisker plot in Figure 4.13, one-way ANOVAs show no statistically significant difference between yearly mean values (see Table 4.3). Interestingly, when one-way ANOVAs were applied solely to pre-expansion data and solely to post-expansion data, it was determined that the increase in iron concentration at the settling pond inlet of System 1 over the years was statistically significant for pre-expansion years, but statistically insignificant post-expansion years. Thus, it can be argued that the addition of System 2 improved feed quality in terms of iron concentration by providing greater operational flexibility and treatment capacity, which helped manage mine water levels and consequently reduced AMD generation. Furthermore, there is statistical significance between yearly mean values for all forms of iron at the settling pond outlet and the wetland outlet of System 1, with improvements in iron removal being noted post-expansion as shown in Figure 4.14. Again, one-way ANOVAs for these two locations applied to exclusively pre-expansion data show statistically significant differences between years of operation whereas those applied exclusively to post-expansion data show statistically insignificant differences. The majority of iron data post-expansion collected at the settling pond outlet and the wetland outlet of System 1 is close to or below the RDL. In summary, the iron concentrations at the settling pond outlet and wetland outlet were increasing in a statistically significant manner prior to expansion and are consistently being reduced to below the detectable limit post-expansion.

Manganese and aluminum show statistically significant differences in yearly means at all locations (except for aluminum at the settling pond outlet). This can easily be explained upon comparing pre-expansion and post-expansion averages for both metals. Manganese and aluminum are being attenuated across System 1 with a post-expansion total average manganese reduction from 6.41 ± 2.55 mg/L to 0.19 ± 0.46 mg/L and a total average aluminum reduction from 0.011 ± 0.013 mg/L to 0.005 ± 0.008 mg/L. This is a notable improvement when compared to pre-expansion data where the total average manganese reduction was from 8.70 ± 3.41 mg/L to 4.93 ± 3.61 mg/L and the total average aluminum reduction was from 1.27 ± 1.84 mg/L to 0.048 ± 0.042 mg/L. In addition, post-expansion data shows no significant difference between yearly means for both metals at all locations. Thus, post-expansion performance has been consistently reducing manganese (demonstrated in Figure 4.15) and aluminum (depicted in Figure 4.16). Guidelines for Manganese were introduced by CCME in 2019 and by NSE in 2021. The NSE Tier 1 EQS for fresh water is 0.43 mg/L. Since guidelines have come into effect for manganese, the wetland outlet averages for System 1 have been 0.062 ± 0.104 mg/L (2019) and 0.052 ± 0.087 mg/L (2020) and thus well under the limit. Since the pH of the system is consistently above 6.5, the CCME limit for aluminum is 0.1 mg/L; post-expansion wetland outlet aluminum concentrations (with an average value of 0.005 ± 0.008 mg/L) have been well below said limit.

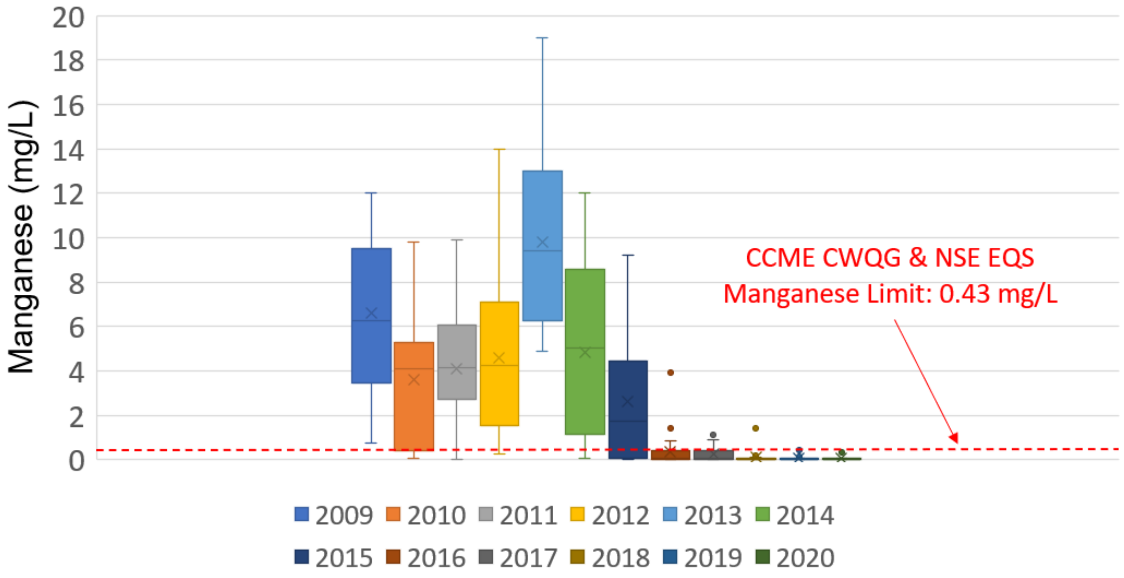

Figure 4.15 Manganese at wetland outlet of System 1 from 2009 until 2020.

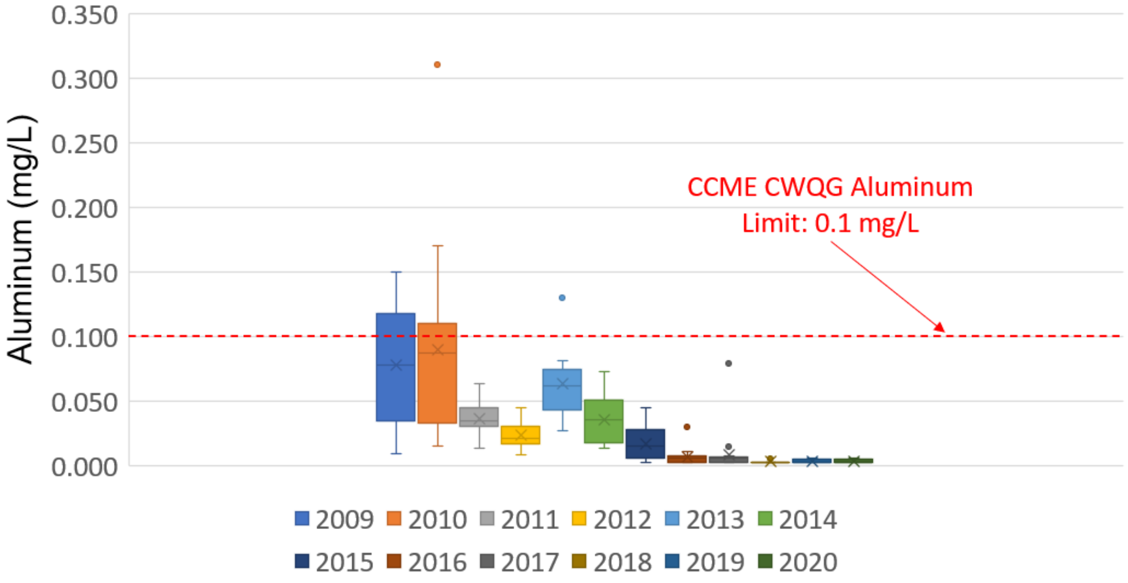

Figure 4.16 Aluminum at wetland outlet of System 1 from 2009 until 2020.

Sulphates concentrations show statistically significant variance between yearly means for the settling pond inlet and settling pond outlet, but not for the wetland outlet. Sulphate concentration can be treated as a surrogate parameter for approximating AMD generation and is not greatly influenced by the Neville St. PTS treatment, as FWS wetlands typically only reduce sulphates by 14 % on average based on 32 studies on treatment wetlands for various applications (Kadlec & Wallace, 2009, p. 417). Thus, variations in sulphates between years can more likely be attributed to yearly variations in weather, such as precipitation, snowmelt, and evaporation. Pump sequencing can attribute to variations in sulphates as mine water quality varies from pump to pump, but the pump sequencing strategy has stayed relatively constant with pumps being put in operation in sequence of best quality mine water to worst with the number of pumps online at a time determined by the flow rate necessary to maintain water level in the 1B hydraulic system.

Parameters associated with pyritic ore oxidation, i.e., pH, field pH, alkalinity, and ORP (E_H), show statistically significant variance between yearly means for all locations. In comparing average values for all these parameters at the settling pond inlet for pre-expansion data (6.56 ± 0.63, 6.36 ± 0.46, 135 ± 47.1 mg/L as CaCO3, and 431 ± 114 mV) to post-expansion data (7.02 ± 0.26, 7.25 ± 0.59, 203 ± 82.4 mg/L as CaCO3, and 450 ± 129 mV), a slight increase in all values can be observed. The same increase in average reading values was observed for all four parameters at the two other locations in System 1. Box and Whisker plots of operational data for all these parameters show that pH (lab and field) and alkalinity are generally increasing across all years of operation, with a notable step change increase coinciding with the expansion start-up. However, for ORP, average readings are increasing pre-expansion, with a step change increase during expansion construction, and then decreasing post-expansion. Caustic soda addition and aeration, which were incorporated into System 1 as part of the expansion, would account for the step change increase observed in all four parameters (aeration and caustic soda increase pH, caustic soda increases alkalinity, aeration increases ORP). The increasing pH and alkalinity and decreasing ORP post-expansion could result from optimizing caustic soda addition over the years. Caustic soda addition is manually adjusted by operators based on field readings. As the site is remote and operators visit periodically, there can be a response lag between mine water pH changes and caustic soda addition rate changes. Rising pH averages over the years could indicate increased caustic soda addition to maintain a larger buffer zone. Both lab and field pH at the wetland outlet (7.73 ± 0.17 and 7.55 ± 0.57) are consistently between federal and provincial limits for both fresh water (6.5 to 9.0) and marine water (7.0 to 8.7) after the expansion.

For turbidity and apparent colour, which are indicators of general water quality, there is no statistically significant difference between yearly means at the settling pond inlet, but there is statistically significant difference between yearly means for the settling pond outlet and wetland outlets. This statistical behaviour is similar to that of all forms of iron. Box and whisker plots of data also depict similar trends as iron with little variation between pre-expansion data (52.3 ± 41.8 NTU and 107 ± 118 Pt/Co) and post-expansion (53.9 ± 46.3 NTU and 81.8 ± 89.6 Pt/Co) data at the settling pond inlet, but with a notable improvement at the settling pond and wetland outlets, with pre-expansion data averages being 3.00 ± 4.37 NTU and 6.20 ± 7.65 Pt/Co and post-expansion data averages being 0.22 ± 0.23 NTU and 4.87 ± 11.9 Pt/Co at the wetland outlet.

4.3.2.2 Statistical Analysis of All Sampling Data for System 2

One-way ANOVAs were applied to performance parameters to compare mean values across each year of operation for System 2. The results of these one-way ANOVAs are provided in Table 4.4 and are discussed in the following paragraphs, including noted trends and inconsistencies.

For all forms of iron (total, ferrous, dissolved, and dissolved ferrous), manganese, aluminum, and sulphate, there is no observed statistically significant difference between yearly means at the settling ponds inlet and outlet of System 2, but there is statistically significant variance between means at the wetland outlet for all these parameters except manganese. Thus, concentrations of these parameters are consistent over the years at the settling ponds inlet and outlet of System 2. In regard to the statistically significant difference between yearly means at the wetland outlet of System 2, the variation between yearly means seems to be caused by different reasons for different parameters.

For all forms of iron, the wetland outlet concentration readings are more often than not below the RDL, and hence half the RDL was assumed for these values. In fact, the dissolved ferrous iron P-value for the settling pond outlet of System 2 could not be calculated because all readings were below the RDL. Thus, readings above the RDL are causing variation between yearly means at the wetland outlet. In addition, several years had exceedances above the federal and provincial limit for total iron of 0.3 mg/L, which could have contributed to the variation between yearly means: 2015 had four exceedances; 2017, three exceedances; and 2020, two exceedances. The average total iron concentration at the wetland outlet of System 2 is 0.12 ± 0.42 mg/L.

Table 4.4 P-values for selected parameters across each year of operation for System 2.

Parameter	**System 2**					
	Settling Pond Inlet		**Settling Pond Outlet**		**Wetland Outlet**	
	P-Value	**Statistical Significance**	**P-Value**	**Statistical Significance**	**P-Value**	**Statistical Significance**
Total Iron	0.270	No	0.729	No	0.003	Yes
Ferrous Iron	0.320	No	0.541	No	0.029	Yes
Dissolved Iron	0.317	No	0.154	No	0.019	Yes
Dissolved Ferrous Iron	0.241	No	NA	No	0.025	Yes
Manganese[1]	0.243	No	0.052	No	0.074	No
Aluminum[1]	0.055	No	0.103	No	0.005	Yes
Sulphate	0.069	No	0.062	No	0.007	Yes
pH	<0.001	Yes	0.011	Yes	0.246	No
Field pH[1]	0.526	No	0.022	Yes	0.041	Yes
Alkalinity[1]	0.007	Yes	0.009	Yes	<0.001	Yes
ORP (E_H)	0.017	Yes	0.180	No	<0.001	Yes
Turbidity	0.512	No	0.982	No	<0.001	Yes
Apparent Colour	0.002	Yes	<0.001	Yes	0.004	Yes

[1] Parameter based on PS&PC data alone.
NA = Not applicable

Manganese concentrations at the wetland outlet showed no statistical variation. The average manganese concentration at the settling pond inlet and wetland outlet of System 2 are 7.52 ± 3.05 mg/L and 1.66 ± 3.40 mg/L. The CCME CWQL for Manganese in fresh water is 0.43 mg/L. Since the adoption of the limit in 2019, there were three exceedances in both 2019 and 2020; this means approximately one out of four samples from System 2 have exceeded the manganese guideline limit since adoption. Aluminum concentrations at the wetland outlet of System 2 improve after the first year of operation, which could be due to process optimization. The NSE Tier 1 EQS for Aluminum in fresh water is 0.1 mg/L. All wetland outlet readings were well below said limit., with the average aluminum concentration being reduced by System 2 from 1.65 ± 0.96 mg/L to 0.012 ± 0.047 mg/L.

Lastly, sulphate concentrations at the wetland outlet of System 2 seem to be increasing over the years with 2015 and 2021 average values being 516 ± 190 mg/L and 889 ± 569 mg/L respectively. This indicates a deterioration of mine water quality over time. Note that corresponding sulphate concentrations for System 1 show no statistically significant variation. This aligns with the pump sequencing strategy as the worst quality AMD is sent to System 2 during peak loading because of its greater treatment capacity.

Parameters associated with pyritic ore oxidation, such as alkalinity, pH (lab and field), and ORP, show statistically significant variation between yearly means for most or all locations of System 2. Based on Box and Whisker plots and yearly averages, pH and alkalinity generally increase and ORP generally decreases through the years, which aligns with System 1 post-expansion data. Again, this could be due to optimizing caustic soda addition over the years. When comparing post-expansion averages for these four variables at different locations within System 2, each parameter (pH, field pH, alkalinity, and ORP) increases between settling pond inlet (6.89 ± 0.28, 7.29 ± 0.67, 115 ± 27.2 mg/L as CaCO3, and 456 ± 130 mV) and the wetland outlet (7.69 ± 0.16, 7.41 ± 0.70, 172 ± 130 mg/L as CaCO3, and 464 ± 119 mV). System 2 pH data at the wetland outlet is generally between federal and provincial limits for both fresh water (6.5 to 9.0) and marine water (7.0 to 8.7). Lab pH drops below the marine water lower limit once; field pH exceeds freshwater upper limit twice (marine upper limit four times) and drops below freshwater lower limit 10 times (marine lower limit 21 times) between 2015 and 2021.

Fresh water limits are the true concern as the downstream receiver, Cadegan Brook, is freshwater, but marine water limits are included as well as Cadegan Brook flows into the Atlantic Ocean.

Finally, turbidity only shows statistically significant variation between yearly means at the wetland outlet (just like iron, manganese, aluminum, and sulphates) whereas apparent colour data shows statistically significant variation between yearly means at all three locations across System 2. The variation for turbidity at the wetland outlet is likely due to several high readings in the first couple of years of operation as both averages and variances are relatively low and constant afterwards. Apparent colour readings from event sampling on high flow days seem to be the source of variance at all three locations as the years that include this data have much greater averages and variance. Also, the first year of operation also has some high readings, which causes this year's average and variance to be higher than most other years. However, significant reduction can be observed across System 2 when comparing settling pond inlet averages (58.9 ± 50.3 NTU and 105 ± 122 Pt/Co) to wetland outlet averages (0.75 ± 1.77 NTU and 13.3 ± 39.9 Pt/Co), indicating that general water quality is improving due to treatment.

4.4 Treatment Performance Modeling of Wetland

Treatment performance modeling of the wetlands at the Neville St. PTS was carried out to achieve the third research objective of this thesis. The initial attempt at modeling the iron removal performance of the wetlands using the modified TIS model revealed that the areal rate constants at 20 °C that were available in literature for iron did not appropriately represent the contaminant attenuation performance of the two wetlands modeled; as a result, the outlet iron concentrations produced by the model were much higher than historically observed values. Thus, a sensitivity analysis was conducted to determine areal rate constants at 20 °C for iron for each system. The areal rate constants at 20 °C developed using the sensitivity analysis were then used as inputs to the modified TIS model to calculate outlet iron concentrations for different operating regimes. Lastly, a sensitivity analysis was performed to see how varying temperature correction factor impacted the model's output as no temperature correction factor for iron could be found.

4.4.1 Sensitivity Analysis for Areal Rate Constant at 20 °C, k_{20}

A sensitivity analysis was performed using the modified TIS model to calibrate areal rate constants at 20 °C for iron for both System 1 and System 2 for various operating regimes. Six scenarios were modeled based on inlet and outlet iron concentration combinations for both average flow and maximum flow as presented in Table 4.5 for System 1 and Table 4.6 for System 2. Therefore, twelve areal rate constants at 20 °C in total were determined for each system. Each calculated areal rate constant at 20 °C had to allow the model to reach the outlet concentration criteria for all four seasons.

Table 4.5 Iron areal rate constants at 20 °C for System 1 wetland as determined by sensitivity analysis.

Criteria	**System 1**					
Combination	**1**	**2**	**3**	**4**	**5**	**6**
C_{in} (mg/L)	**Average C_{in}**	**Maximum C_{in}**	**10**	**Average C_{in}**	**Maximum C_{in}**	**10**
C_{out} (mg/L)	**0.3**	**0.3**	**0.3**	**Average C_{out}**	**Average C_{out}**	**Average C_{out}**
k_{20} for Average Flow (m/year)	410	915	1,090	1,930	3,540	5,250
k_{20} for Maximum Flow (m/year)	865	2,060	2,325	4,100	7,875	11,205

The first inlet and outlet concentration combination, with average inlet concentrations (for each season) and an outlet concentration maximum of 0.3 mg/L, did not realistically represent the actual operation of the two wetlands modeled as both wetlands perform much better than this in reality. At average flow rates for this inlet/outlet concentration combination, both wetlands effectively reduce average inlet concentrations to well below the provincial and federal guideline of 0.3 mg/L. At maximum flow rates, inlet and outlet concentrations would be higher than average in reality, rendering this scenario implausible. Thus, the areal rate constants at 20 °C determined by this combination are conservative and provide lower limits for the areal rate constants at 20 °C.

Table 4.6 Iron areal rate constants at 20 °C for System 2 wetland as determined by sensitivity analysis.

Criteria	**System 2**					
Combination	**1**	**2**	**3**	**4**	**5**	**6**
C_{in} **(mg/L)**	**Average** C_{in}	**Maximum** C_{in}	**10**	**Average** C_{in}	**Maximum** C_{in}	**10**
C_{out} **(mg/L)**	**0.3**	**0.3**	**0.3**	**Average** C_{out}	**Average** C_{out}	**Average** C_{out}
k_{20} for Average Flow (m/year)	150	375	650	560	815	1380
k_{20} for Maximum Flow (m/year)	315	835	1380	1185	1780	2935

The second inlet and outlet concentration combination, with maximum inlet concentrations (for each season) and an outlet concentration maximum of 0.3 mg/L, is a more realistic scenario. The whole objective of event sampling was to collect samples during peak loading, which occurs at maximum flow rates with maximum inlet concentrations, and then observe whether outlet concentrations stayed below the guideline of 0.3 mg/L. Hence, the areal rate constants at 20 °C produced by this combination provide the performance range required to meet provincial and federal guidelines based on maximum observed inlet concentrations.

The third inlet and outlet concentration combination, with an inlet concentration of 10 mg/L and an outlet concentration maximum of 0.3 mg/L, is the design scenario on which the wetlands were sized. Thus, the areal rate constant at 20 °C calculated by this combination for maximum flow is replicating design conditions and will be referred to as the "design" areal rate constant at 20 °C. It provides an upper limit for the areal rate constants at 20 °C, and any areal rate constants at 20 °C above this value exceeds the intended design performance and would be questionable. The design areal rate constant at

20 °C for System 1 developed by the sensitivity analysis was 2,325 m/year; System 2 was 1,380 m/year.

The last three inlet/outlet concentration combinations have average seasonal outlet concentrations as the maximum limit for outlet concentration. Note that for these three combinations, the background concentration input to the model had to be adjusted from 0.05 mg/L to 0.02 mg/L to allow the model to simulate outlet concentrations below the target values.

The fourth inlet and outlet concentration combination, with average inlet concentrations (for each season) and average outlet concentrations (for each season), is much more representative of the actual performance of both wetlands. The areal rate constant at 20 °C determined by this combination for average flow represents average operating conditions and will be referred to as the “average” areal rate constant at 20 °C. The average areal rate constant at 20 °C for System 1 and System 2 produced by the sensitivity analysis were 1,930 m/year and 560 m/year respectively.

The final two combinations are scenarios not necessarily expected to be representative of actual performance, but provide areal rate constants at 20 °C for comparison to gauge whether these scenarios are plausible or not. If not plausible, comparison of these constant values can help determine how unrealistic these scenarios are.

For the fifth inlet and outlet concentration combination, with maximum inlet concentrations (for each season) and average outlet concentrations (for each season), comparison of these areal rate constants at 20 °C for System 1 reveals that maximum inlet concentrations decreasing down to average outlet concentrations is not plausible even at average flow rates. This is a result of the areal rate constant at 20 °C for this scenario being much greater than that for design. However, for System 2, the same areal rate constants at 20 °C is not only less than the design value, but also falls in the range of areal rate constants at 20 °C deemed plausible by the second inlet/outlet concentration combination. Hence, the areal rate constants at 20 °C produced for average flow rates through System 2 for the fifth inlet/outlet concentration combination would suggest that this is a plausible scenario. In other words, System 2 could be capable of reducing maximum inlet concentrations to average outlet concentrations at average flow rate, but System 1 should not be.

For the sixth and final inlet and outlet concentration combination, with an inlet concentration of 10 mg/L and average outlet concentrations (for each season), comparison of these areal rate constants at 20 °C for both systems reveals that reducing iron concentrations from the design value of 10 mg/L to the average outlet concentration is not plausible for either system for both flow regimes. This is due to the fact that the areal rate constants at 20 °C for this scenario are at or much greater than the design upper limit.

Appendix E shows the modified TIS model results (i.e., iron effluent concentration) of the sensitivity analysis for areal rate constant at 20 °C for both systems. For each scenario, the appropriate areal rate constant at 20 °C from Table 4.5 or Table 4.6 is matched to the corresponding flow rate, inlet concentration, and target outlet concentration. All model results are below the target effluent concentration (of either 0.3 mg/L or average outlet concentration) for all seasons, with the exception of some results for scenario 34 through 38 for System 1. This is due to the fact that these scenarios use the areal rate constant at 20 °C derived from post-expansion data, but the inlet concentrations are based on pre-expansion data and are significantly greater than design inlet concentrations. Hence, exceedances are to be expected. For target outlet concentration of 0.3 mg/L, all five scenarios (i.e., 34 -38) exceeded; for target outlet concentration of average outlet concentration, only one scenario (i.e., 35) exceeded. These results align with the established fact that pre-expansion inlet iron concentrations were too high during high flow events for System 1 to provide proper treatment, which is why the system was expanded.

Comparing the areal rate constants at 20 °C developed for different operating regimes for System 1 (Table 4.5) and for System 2 (Table 4.6) to those found in literature (i.e., 106 m/year from Tarutis et al. (1999) and 38 m/year from Younger et al. (2002)), it can be observed that the constant values calibrated for these two systems are much greater than those from literature. These findings demonstrate that areal rate constants at 20 °C vary between systems and site-specific calibration is often required; thus, caution is warranted when applying values obtained from literature for this modeling constant. Moreover, Table 4.5 and Table 4.6 show that areal rate constants at 20 °C can have intrasystem variance dependent upon operating conditions such as contaminant concentration and flow rate. This aligns with the fact that the rate of many attenuation mechanisms within a system can also vary with contaminant concentration and flow rate.

4.4.2 Tanks-in-Series (TIS) Model

An initial attempt at modeling the iron removal performance of the wetlands at the Neville St. PTS was completed using the modified TIS model with an areal rate constant at 20 °C of 106 m/year as supplied by Tarutis et al. (1999). The results of this modeling are displayed in Table 4.7 for System 1 and in Table 4.8 for System 2. For both systems, the outlet iron concentrations produced by the Tarutis et al. areal rate constant at 20 °C were much greater than actual outlet concentration readings observed for similar operating regimes. In fact, only five of 38 scenarios for System 1 did not exceed the provincial and federal guideline limit of 0.3 mg/L. These five scenarios either occurred in the summer when flow rates are minimal and inlet iron concentrations are low (scenarios 3, 7, 19, and 23) or for the best-case scenario, which was incorporated to model the circumstances that would produce the lowest outlet concentrations (scenario 33). For System 2, 18 of 33 scenarios exceeded the guideline limit. These scenarios with exceedances include some with average inlet concentrations, most with maximum inlet concentrations, and all with design inlet concentrations. Thus, the model using an areal rate constant at 20 °C of 106 m/year was deemed to be unrepresentative of the actual performance of the wetlands at the Neville St. PTS.

Therefore, a sensitivity analysis was performed using the modified TIS model to determine iron areal rate constants at 20 °C for both systems for various operating scenarios. Table 4.7 and Table 4.8 provide the results of the modeling conducted for System 1 and System 2 respectively using two iron areal rate constants at 20 °C produced by the sensitivity analysis: (1) the iron areal rate constants at 20 °C for design conditions and (2) the iron areal rate constants at 20 °C for average conditions.

Table 4.7 Modified TIS model results for iron effluent concentration for System 1 for different operating scenarios based on various areal rate constants at 20 °C.

Scenario Number for System 1	**Tarutis et al.** (1999)		**Design Conditions**		**Average Conditions**	
	k_{20} **(m/year)**	C_{out} **(mg/L)**	k_{20} **(m/year)**	C_{out} **(mg/L**	k_{20} **(m/year)**	C_{out} **(mg/L**
1	106	0.937	2,325	0.057	1,930	0.031
2	106	0.776	2,325	0.056	1,930	0.029
3	106	0.188	2,325	0.051	1,930	0.022
4	106	0.525	2,325	0.054	1,930	0.026
5	106	0.960	2,325	0.057	1,930	0.032
6	106	0.787	2,325	0.056	1,930	0.030
7	106	0.190	2,325	0.051	1,930	0.022
8	106	0.536	2,325	0.054	1,930	0.026
9	106	2.218	2,325	0.067	1,930	0.048
10	106	3.757	2,325	0.079	1,930	0.067
11	106	0.603	2,325	0.054	1,930	0.027
12	106	1.040	2,325	0.057	1,930	0.032
13	106	5.259	2,325	0.092	1,930	0.087
14	106	5.305	2,325	0.092	1,930	0.087
15	106	5.334	2,325	0.093	1,930	0.088
16	106	5.265	2,325	0.092	1,930	0.087
17	106	1.282	2,325	0.092	1,930	0.083
18	106	1.052	2,325	0.084	1,930	0.072
19	106	0.240	2,325	0.057	1,930	0.031
20	106	0.709	2,325	0.073	1,930	0.054
21	106	1.296	2,325	0.093	1,930	0.084
22	106	1.060	2,325	0.085	1,930	0.072
23	106	0.241	2,325	0.057	1,930	0.031
24	106	0.716	2,325	0.073	1,930	0.055
25	106	3.377	2,325	0.164	1,930	0.188
26	106	5.710	2,325	0.243	1,930	0.306

Scenario Number for System 1	**Tarutis et al.** (1999)		**Design Conditions**		**Average Conditions**	
	k_{20} (m/year)	C_{out} (mg/L)	k_{20} (m/year)	C_{out} (mg/L	k_{20} (m/year)	C_{out} (mg/L
27	106	0.887	2,325	0.078	1,930	0.063
28	106	1.632	2,325	0.104	1,930	0.100
29	106	7.279	2,325	0.298	1,930	0.385
30	106	7.307	2,325	0.299	1,930	0.387
31	106	7.325	2,325	0.300	1,930	0.388
32	106	7.282	2,325	0.298	1,930	0.385
33	106	0.051	2,325	0.050	1,930	0.020
34	106	21.254	2,325	3.941	1,930	4.973
35	106	18.176	2,325	0.671	1,930	0.933
36	106	16.060	2,325	0.600	1,930	0.828
37	106	13.174	2,325	0.501	1,930	0.683
38	106	17.459	2,325	0.647	1,930	0.898

As per Table 4.7 for System 1, the guideline limit of 0.3 mg/L was exceeded in five scenarios for the design areal rate constants at 20 °C and in 10 scenarios for the average areal rate constants at 20 °C. For both of these areal rate constants at 20 °C, five of the scenarios that exceeded the guideline limit were scenarios 34 to 38, which use pre-expansion maximum observed inlet concentrations that are much higher than the design concentration of 10 mg/L. Hence, exceedances were expected for these cases. For the five other scenarios that exceeded the guideline limit for average areal rate constants at 20 °C, the flow rates were at maximum rates and the inlet concentrations were either at maximum values (scenario 26) or were the design value of 10 mg/L (scenarios 29 – 32). These exceedances for an areal rate constant at 20 °C that represents average performance indicate that the System 1 wetland may not always be able to reduce maximum or design concentrations at maximum flow rate to below the guideline limit of 0.3 mg/L; nonetheless, the lack of exceedances for the same scenarios for the design areal rate constant at 20 °C indicate that the System 1 wetland could reduce iron concentrations to below the guideline limit of 0.3 mg/L if System 1 is achieving design performance.

Table 4.8 Modified TIS model results for iron effluent concentration for System 2 for different operating scenarios based on various areal rate constants at 20 °C.

Scenario Number for System 2	**Tarutis et al.** (1999)		**Design Conditions**		**Average Conditions**	
	k_{20} **(m/year)**	C_{out} **(mg/L)**	k_{20} **(m/year)**	C_{out} **(mg/L)**	k_{20} **(m/year)**	C_{out} **(mg/L)**
1	106	0.368	1,380	0.051	560	0.046
2	106	0.141	1,380	0.050	560	0.028
3	106	0.050	1,380	0.050	560	0.021
4	106	0.137	1,380	0.050	560	0.028
5	106	0.377	1,380	0.051	560	0.046
6	106	0.143	1,380	0.050	560	0.028
7	106	0.050	1,380	0.050	560	0.021
8	106	0.140	1,380	0.050	560	0.028
9	106	1.251	1,380	0.052	560	0.113
10	106	0.465	1,380	0.050	560	0.053
11	106	0.151	1,380	0.049	560	0.028
12	106	0.310	1,380	0.049	560	0.040
13	106	4.999	1,380	0.061	560	0.403
14	106	5.040	1,380	0.061	560	0.407
15	106	5.066	1,380	0.061	560	0.409
16	106	5.005	1,380	0.061	560	0.404
17	106	0.509	1,380	0.066	560	0.150
18	106	0.181	1,380	0.055	560	0.061
19	106	0.050	1,380	0.050	560	0.026
20	106	0.176	1,380	0.054	560	0.060
21	106	0.515	1,380	0.066	560	0.152
22	106	0.182	1,380	0.055	560	0.062
23	106	0.050	1,380	0.050	560	0.026
24	106	0.178	1,380	0.054	560	0.060
25	106	1.966	1,380	0.116	560	0.544
26	106	0.713	1,380	0.072	560	0.205

Scenario Number for System 2	**Tarutis et al.** (1999)		**Design Conditions**		**Average Conditions**	
	k_{20} (m/year)	C_{out} (mg/L)	k_{20} (m/year)	C_{out} (mg/L)	k_{20} (m/year)	C_{out} (mg/L)
27	106	0.212	1,380	0.055	560	0.070
28	106	0.487	1,380	0.064	560	0.144
29	106	7.184	1,380	0.298	560	1.956
30	106	7.211	1,380	0.299	560	1.964
31	106	7.228	1,380	0.300	560	1.968
32	106	7.187	1,380	0.298	560	1.957
33	106	0.038	1,380	0.050	560	0.020

As per Table 4.8 for System 2, the guideline limit was exceeded in no scenarios for the design areal rate constants at 20 °C and in nine scenarios for the average areal rate constants at 20 °C. Eight of these exceedances are for inlet concentrations set at the design concentration of 10 mg/L (scenarios 13 – 16, 29 – 32). The last exceedance is for an inlet concentration of maximum value (scenario 25). This indicates that design and maximum inlet concentrations between average and maximum flow may not always be reduced to below the guideline limit of 0.3 mg/L by the System 2 wetland. Since there were no exceedances for the design areal rate constant at 20 °C, then System 2 could reduce iron concentrations to below the guideline limit if the wetland performs as per design. The System 2 wetland performing more poorly than the System 1 wetland (i.e. experiencing more exceedances for the first 33 scenarios when modeled using the average areal rate constant at 20 °C), could be attributed to the larger gap between average and design values for areal rate constants at 20 °C seen for System 2.

4.4.3 Sensitivity Analysis for Temperature Correction Factor, θ

As there was no temperature correction factor available for iron in literature, a value of one was assumed for the modeling of all scenarios and then a sensitivity analysis was performed on both systems using design and average areal rate constants at 20 °C. The wetland outlet iron concentration results produced by this analysis are presented in Table 4.9. The temperature correction factor input value was varied between 0.95 and 1.05 to determine how the wetland outlet iron concentration would be affected. The results shown in Table 4.9 for both systems are for scenarios 1 to 33; scenarios 34 to 38 for System 1 were not included as these scenarios use pre-expansion data and therefore are not representative of the current configuration.

Table 4.9 Sensitivity analysis for temperature correction factor using design areal rate constants at 20 °C and average areal rate constants at 20 °C for both systems.

System 1	**Design k_{20} = 2325 m/year**			**Average k_{20} = 1930 m/year**		
θ	**0.95**	**1.00**	**1.05**	**0.95**	**1.00**	**1.05**
Minimum Iron C_{out} (mg/L)	0.050	0.050	0.050	0.050	0.050	0.050
Maximum Iron C_{out} (mg/L)	0.237	0.300	1.467	0.329	0.417	1.875
System 2	**Design k_{20} = 1380 m/year**			**Average k_{20} = 560 m/year**		
θ	**0.95**	**1.00**	**1.05**	**0.95**	**1.00**	**1.05**
Minimum Iron C_{out} (mg/L)	0.049	0.049	0.050	0.049	0.049	0.049
Maximum Iron C_{out} (mg/L)	0.210	0.300	2.203	1.615	1.993	5.254

For design areal rate constants at 20 °C for both systems, the only exceedances of the guideline limit of 0.3 mg/L were observed when the input value for temperature correction factor was set to 1.05. These exceedances took place for all seasons of the worst-case scenario, i.e., when flow rates were set at the maximum observed values and inlet iron concentrations were set at the design value of 10 mg/L (scenarios 29 – 32). Exceedances were also observed for some seasons when flow rates were set at the average observed values and inlet iron concentrations were set at the design value of 10 mg/L (scenario 13 for both systems and scenario 16 for System 2 only) or when flow rates were set at the maximum observed values and inlet iron concentrations were set at the maximum observed values (scenario 25 for both systems and scenarios 26 and 28 for System 1 only).

For average areal rate constants at 20 °C for both systems, exceedances of the guideline limit of 0.3 mg/L were noted for all three input values for temperature correction factor (0.95, 1.00. and 1.05) with the fewest number of exceedances being observed for 0.95 and the greatest number of exceedances occurring for 1.05. In general, more exceedances were observed for scenarios with higher flow rates and/or higher inlet iron concentrations. More exceedances are to be expected for the average areal rate constants at 20 °C than the design areal rate constants at 20 °C as the design areal rate constants at 20 °C values are larger numbers than the average areal rate constants at 20 °C values (and are therefore more effective at reducing iron concentrations). Moreover, for the worst-case scenarios with the highest observed flow rates and design inlet concentration of 10 mg/L, exceedances occurred for all seasons for the temperature correction factors of 1.00 and 1.05 and for some seasons for the temperature correction factor of 0.95. Exceedances were also noted when temperature correction factor was set to 1.00 or 1.05 for some seasons of scenarios where flow rates were set at the maximum observed values or inlet concentrations were set at the maximum observed values or at the design inlet concentration of 10 mg/L.

Overall, approximately 20 % of the scenarios for System 1 and 28 % of the scenarios for System 2 experienced exceedances of the guideline limit for outlet iron concentration when using the average areal rate constants at 20 °C. Furthermore, System 2 generally produced higher maximum outlet iron concentrations than System 1, as demonstrated by Table 4.9. This can be attributed to the fact that the average areal rate constant at 20 °C is closer to the design areal rate constant at 20 °C for System 1 than for System 2, i.e., System 1 is performing closer to design expectations than System 2 based on average iron removal performance. Hence, the exceedances on System 2 are taking place for more scenarios and producing higher maximum outlet iron concentrations.

For iron and other metals, there is a noted lack of values for the constants required by the modified TIS model such as temperature correction factor. Although the lower temperature dependance of dominant attenuation mechanisms for iron would suggest that iron removal would be less sensitive to temperature, more research and development of temperature correction factor values for iron (and other metals) are needed to confirm how sensitive attenuation performance is on temperature. If iron sequestration mechanisms are dependent on temperature, then based on the sensitivity analysis conducted for this work, significant outlet concentration variation would be expected. However, due to the number of parameters incorporated into the model and the various sources of error, variations in outlet concentrations cannot be definitively correlated with temperature changes nor any other source without further analysis. Site-specific calibration of temperature correction factor values would be required for both systems.

CHAPTER 5 CONCLUSION

The following chapter will summarize the findings from this thesis on passive treatment of AMD at the Neville St. PTS, including the results from the performance characterization and modeling components. Subsequently, recommendations developed from this work are listed and explained.

5.1 Summary

The summary of the results of this work evaluating the performance of the Neville St. PTS in treating AMD after expansion is subdivided into two sections: (1) physicochemical and statistical analyses, and (2) treatment performance modeling of wetland.

5.1.1 Physicochemical and Statistical Analyses

Statistical analysis of event sampling data determined that there was statistically significant removal of all forms of iron (total, ferrous, and dissolved) across the two systems at the Neville St. PTS. Dissolved iron and ferrous iron were primarily reduced in the settling pond(s), indicating that most soluble iron is converted to insoluble iron by the time the mine water reaches the settling pond outlet. The wetland effluent total iron concentration readings recorded during event sampling were all well below the federal and provincial guideline limits of 0.3 mg/L, even during the highest recorded loading rates, with average values of 0.05 ± 0.07 mg/L for System 1 and 0.07 ± 0.06 mg/L for System 2.

Statistical analysis of System 1 comparing pre-expansion and post-expansion concentration data for all forms of iron (total, ferrous, dissolved, and dissolved ferrous) showed a marked improvement in iron attenuation performance. Wetland outlet iron concentrations were increasing in a statistically significant manner prior to expansion (with an average pre-expansion total iron concentration of 0.75 ± 1.63 mg/L) and are currently showing no statistically significant variance between post-expansion years while being consistently reduced to below the RDL (with an average post-expansion total iron concentration of 0.04 ± 0.08 mg/L). Although the yearly average and maximum values for total iron concentration at the settling pond inlet have been increasing since the expansion, all total iron readings taken post-expansion at the wetland outlet (except one during the first year of operation) have been below the federal and provincial limit of 0.3 mg/L. For System 2, average and maximum concentrations of total iron over the years have also been increasing at the settling pond inlet. The average total iron concentration at the wetland outlet of System 2 is 0.12 ± 0.42 mg/L with most readings below the RDL and nine exceedances since beginning operation. Hence, it appears that the addition of System 2 has thus far improved iron removal performance as intended by providing greater treatment capacity and operational flexibility.

The manganese attenuation performance of System 1 improved in a statistically significant manner since it began operating with pre-expansion average concentration being reduced from 8.70 ± 3.41 mg/L to 4.93 ± 3.61 mg/L and with post-expansion average concentration being reduced from 6.41 ± 2.55 mg/L to 0.19 ± 0.46 mg/L. System 2 performance and System 1 post-expansion performance are consistent in terms of manganese sequestration with no statistically significant difference between yearly means at all locations. System 2 manganese reduction performance has been reducing the average concentration from 7.52 ± 3.05 mg/L to 1.66 ± 3.40 mg/L. The CCME CWQL for Manganese in fresh water is 0.43 mg/L. Since the adoption of the limit in 2019, there have been no exceedances by System 1 and three exceedances per year in 2019 and 2020 by System 2 (which equates to approximately a quarter of the samples having exceedances).

The aluminum removal performance of System 1 also showed statistically significant differences in yearly means across its total operational history. Prior to expansion, the total average aluminum reduction was from 1.27 ± 1.84 mg/L to 0.048 ± 0.042 mg/L; post-expansion; from 0.011 ± 0.013 mg/L to 0.005 ± 0.008 mg/L. For System 2, the average aluminum concentration is being reduced from 1.65 ± 0.96 mg/L to 0.012 ± 0.047 mg/L. The CCME limit for aluminum for the applicable pH range is 0.1 mg/L; System 1 (post-expansion) and System 2 wetland outlet aluminum concentrations have been well below this limit.

Statistical analysis on event sampling data showed no significant reduction of sulphate concentrations across the two systems at the Neville St. PTS. Average sulphate concentrations at the wetland outlets for System 1 and System 2 were 515 ± 230 mg/L and 798 ± 620 mg/L respectively. As settling ponds and wetlands are not effective at removing sulphate, the Neville St. PTS is not intended to remove sulphates; this has not been an issue in the past as there was no guideline limit for sulphates until recently. NSE Tier 1 EQS adopted a new limit for sulphates of 128 mg/L in 2021. As event sampling was conducted before this revision, no sulphate exceedances were observed, but this could be a potential performance issue moving forward. Adjusting the limit to account for hardness as done by BCMOECCS (from whence the limit was adopted) could mitigate exceedance concerns.

Wetland outlet average sulphate concentrations for System 2 were noted to be increasing in statistically significant manner over the years since coming online, which is indicative of mine water quality deterioration over time. Note that corresponding post-expansion average sulphate concentrations for System 1 do not exhibit any statistically significant variation. This can be explained by the pump sequencing strategy, which preferentially directs poorer quality mine water to System 2 during peak loading.

The average pH across the settling pond(s) increases in a statistically significant manner by one pH unit according to data obtained through event sampling. In the System 1 settling pond, the average pH rises from 7.05 ± 0.50 to 8.01 ± 0.41; in the System 2 settling ponds, from 7.28 ± 0.64 to 8.15 ± 0.39. Federal and provincial guideline limits for pH are 6.5 to 9.0 for fresh water (7.0 – 8.7 for marine waters); the average wetland outlet pH readings measured by event sampling (7.57 ± 0.27 for System 1 and 7.65 ± 0.25 for System 2) were within these limits.

Statistical analysis of System 1 comparing pre-expansion and post-expansion data for pH shows statistically significant variance between yearly means for all locations, with inlet settling pond average pH increasing from 6.56 ± 0.63 for pre- expansion data to 7.02 ± 0.26 for post-expansion data. Caustic soda addition and aeration was incorporated into both systems as part of the expansion and accounts for this change in pH behaviour in

System 1. System 1 post-expansion and System 2 pH values at the wetland outlet (7.73 ± 0.17 and 7.69 ± 0.16 respectively) are consistently between federal and provincial limits for both fresh water (6.5 to 9.0) and marine water (7.0 to 8.7).

5.1.2 Treatment Performance Modeling of Wetland

An attempt at modeling the wetland iron attenuation performance of both systems at the Neville St. PTS was performed using the modified TIS model with an areal rate constant at 20 °C as provided by literature of 106 m/year. The outlet iron concentrations produced by the model were much higher than those seen in actuality, rendering the results unrepresentative. Consequently, a sensitivity analysis was conducted using the modified TIS model to develop areal rate constants at 20 °C for iron for both systems for various operating scenarios. The resulting design areal rate constants at 20 °C were 2,325 m/year for System 1 and 1,380 m/year for System 2; the resulting average areal rate constants at 20 °C were 1,930 m/year and 560 m/year respectively. Modeling performed with these areal rate constants at 20 °C for iron was successful for both System 1 and System 2 and produced outputs that better aligned with actual performance. Furthermore, it should be noted that areal rate constants at 20 °C are site-specific and can even vary within a system depending on operating conditions; thus, caution is warranted when using values obtained from literature for this modeling constant.

In response to there being no temperature correction factor available for iron in literature, an input value of one was assumed for the modeling of both systems. Subsequently, a sensitivity analysis was completed using design and average areal rate constants at 20 °C by varying the temperature correction factor between 0.95 and 1.05 to determine how iron concentration at the wetland outlet would be affected. In general, more exceedances on the wetland outlet iron concentration were experienced for higher temperature correction factor values and for scenarios with greater flow rates and/or greater inlet iron concentrations. For both systems, more exceedances were observed for the average areal rate constants at 20 °C than for the design areal rate constants at 20 °C, because the design values are greater than the average values, which results in better attenuation performance. Moreover, System 2 experienced more exceedances than System 1 because the difference between design and average values for the areal rate constant at 20 °C is smaller for System 1 than it is for System 2. Accordingly, System 1 is performing closer to design expectations than System 2 in terms of average iron sequestration performance.

5.2 Recommendations

The recommendations stemming from this work evaluating the performance of the Neville St. PTS in treating AMD after expansion is subdivided into two sections: (1) recommendations for PS&PC, and (2) recommendations for future research.

5.2.1 Recommendations for PS&PC

Due to the infrequency of sampling within the current PS&PC monitoring program (i.e., once per month), it is unlikely that sampling will take place during peak loading events. Therefore, there is a strong likelihood that sampling is not representative of worst-case performance and exceedances could be missed. In addition to sampling on a periodic basis, PS&PC should sample during peak flow events. This would require upgrading the monitoring system to include the capability for remote monitoring of flow and for alerting operators of peak flow events. Alternatively, autosamplers triggered by rainfall or high flow rates could be installed and incorporated into the sampling protocol.

Automation of the caustic addition based on pH and inlet flow rate is another recommendation. This addition would reduce the need for operator intervention, lessening the demands on operators, and improve treatment by decreasing the system response time.

Given the new limit that was adopted in 2021 by NSE Tier 1 EQS for sulphates, a strategy needs to be developed to address potential exceedances of said limit by the treated mine water effluent. This strategy could include a grandfather clause or adjusting the limit to account for hardness as done by BCMOECCS (from whence the limit was adopted).

5.2.2 Recommendations for Future Research

There were limits to the scope carried out by this study of the Neville St. PTS performance. More sampling during peak flow events should be conducted and analyzed for aluminum and manganese to evaluate the attenuation performance of these metals by the Neville St. PTS.

Another recommendation is to conduct tracer studies for various flow scenarios to help understand the performance of the Neville St. PTS post-expansion. Data from tracer tests would provide insight on hydraulic behaviour such as HRT, which could be employed to improve sampling protocols by incorporating the time required for mine water to travel through each system during high flow events. Moreover, these tracer studies could be used to improve accuracy of the wetland performance modeling by providing a more accurate estimate of input values for HRT and number of tanks.

For metals, there is a marked lack of values for the constants required by the modified TIS model (most notably for the areal rate constant at 20 °C and temperature correction factor). More research and development of model constant values for metals are needed. Furthermore, caution should be used when employing these values from literature in models as values can vary greatly between wetlands. Site-specific calibration of model constant values may be required for accurate modeling.

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APPENDIX A Photos of Neville St. PTS

Figure A.1 Aeration cascade with high flow at settling pond inlet for System 1.

Figure A.2 Aeration cascade with no flow at settling pond inlet for System 1.

Figure A.3 Caustic building and aeration cascade at settling pond inlet for System 2.

Figure A.4 Inlet distribution structure after aeration cascade with low flow at settling pond inlet for System 1.

Figure A.5 Inlet and outlet structures of third settling pond for System 2 (foreground), settling pond for System 1 (middle ground), and natural wetland (background).

Figure A.6 Three settling ponds for System 2.

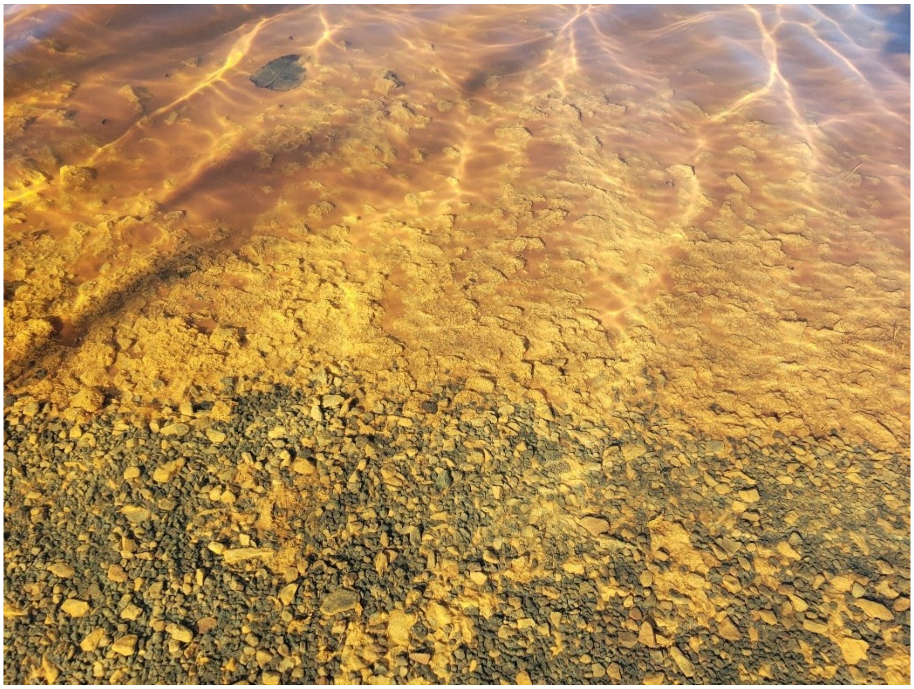

Figure A.7 Sedimentation of iron oxyhydroxides in settling pond.

Figure A.8 Sampling between settling ponds for System 2.

Figure A.9 Outlet structure of settling pond for System 1.

Figure A.10 HOBO pressure transducers with high flow in channel between settling pond and constructed wetland for System 1.

Figure A.11 Overflow weir to natural wetlands with high flow in channel between settling pond and constructed wetland for System 1.

Figure A.12 Sluice gate in channel between settling pond and constructed wetland for System 1.

Figure A.13 Sluice gate in channel between settling ponds and constructed wetland for System 2.

Figure A.14 Inlet of constructed wetland for System 1.

Figure A.15 Constructed wetland for System 2.

Figure A.16 Outlet structure of constructed wetland for System 1.

Figure A.17 Combining of discharge channels for System 1 and System 2.

Figure A.18 Combined discharge channel for System 1 and System 2 flowing to Cadegan Brook.

Figure A.19 Neville St. PTS.

APPENDIX B Parameters Monitored by PS&PC

Table B.1 Parameters monitored at Neville St. PTS by PS&PC through sampling.

	PARAMETER	CCME FWAL	NSE EQS	Units	Unfiltered Samples	Filtered Samples
Major Ions	Sodium (Na)	NG	NG	mg/L	Regularly	Regularly
	Potassium (K)	NG	NG	mg/L	Regularly	Regularly
	Calcium (Ca)	NG	NG	mg/L	Regularly	Regularly
	Magnesium (Mg)	NG	NG	mg/L	Regularly	Regularly
	Alkalinity ($CaCO_3$)	NG	NG	mg/L	Regularly	NA
	Sulphate (SO_4)	NG	NG	mg/L	Regularly	NA
	Chloride (Cl)	120	NG	mg/L	Regularly	NA
	Silica (SiO_2)	NG	NG	mg/L	Regularly	NA
Nutrients	Orthophosphate (P)	NG	NG	mg/L	Regularly	NA
	Phosphorus (P)	NG	NG	mg/L	Regularly	Regularly
	Nitrite + Nitrate	NG	NG	mg/L	Regularly	NA
	Nitrate (N)	13	NG	mg/L	Regularly	NA
	Nitrite (N)	0.06	NG	mg/L	Regularly	NA
	TKN	NG	NG	mg/L	Regularly	NA
	Nitrogen (Ammonia Nitrogen)	Narrative	NG	mg/L	Regularly	NA
	Organic Carbon (C)	NG	NG	mg/L	Regularly	NA
Physical Parameters	Hardness ($CaCO_3$)	NG	NG	mg/L	Regularly	NA
	Bicarb. Alkalinity ($CaCO_3$)	NG	NG	mg/L	Regularly	NA
	Carb. Alkalinity ($CaCO_3$)	NG	NG	mg/L	Regularly	NA
	Colour	Narrative	NG	TCU	Regularly	NA
	Turbidity	Narrative	NG	NTU	Regularly	NA
	Conductivity	NG	NG	umho/cm	Regularly	NA
	pH	6.5 - 9.0	NG	pH Units	Regularly	NA

	PARAMETER	CCME FWAL	NSE EQS	Units	Unfiltered Samples	Filtered Samples
Physical Parameters	Total Acidity	NG	NG	mg/L	Regularly	NA
	Modified Acidity ($CaCO_3$)	NG	NG	mg/L	Regularly	NA
Calculated Values	Acidity Calc. [6]	NG	NG	mg/L	Regularly	NA
	Acidity Net [6]	NG	NG	mg/L	Regularly	NA
	Calculated TDS	NG	NG	mg/L	Regularly	NA
	Cation Sum	NG	NG	meq/L	Regularly	NA
	Anion Sum	NG	NG	meq/L	Regularly	NA
	Ion Sum	NG	NG	meq/L	NA	NA
	Ion Balance (% Difference)	NG	NG	%	Regularly	NA
Biological Parameters	Total Coliforms	NG	NG	MPN/ 100mL	Regularly	NA
	Fecal Coliforms (E.Coli)	NG	NG	MPN/ 100mL	Regularly	NA
	Chemical Oxygen Demand	NG	NG	mg/L	Regularly	NA
	Biochemical Oxygen Demand	NG	NG	mg/L	Regularly	NA
Metals	Iron (Fe)	0.3	0.3	mg/L	Regularly	Regularly
	Iron (ferrous) [6]	NG	NG	mg/L	Regularly	Regularly
	Iron (ferric)	NG	NG	mg/L	Regularly	Regularly
	Manganese (Mn)	NG	0.82	mg/L	Regularly	Regularly
	Copper (Cu)	0.002 - 0.004 [1]	0.002	mg/L	Regularly	Regularly
	Zinc (Zn)	0.007	0.03	mg/L	Regularly	Regularly
	Aluminum (Al)	0.005 - 0.1 [2]	0.005	mg/L	Regularly	Regularly
	Antimony (Sb)	NG	0.02	mg/L	Regularly	Regularly
	Arsenic (As)	0.005	0.005	mg/L	Regularly	Regularly
	Barium (Ba)	NG	1	mg/L	Regularly	Regularly
	Beryllium (Be)	NG	0.0053	mg/L	Regularly	Regularly
	Bismuth (Bi)	NG	NG	mg/L	Regularly	Regularly
	Boron (B)	1.5	1.2	mg/L	Regularly	Regularly

	PARAMETER	**CCME FWAL**	**NSE EQS**	**Units**	**Unfiltered Samples**	**Filtered Samples**
Metals	Cadmium (Cd)	0.00004 - 0.00037 [3]	0.00001	mg/L	Regularly	Regularly
	Chromium (Cr)	0.0089	NG	mg/L	Regularly	Regularly
	Cobalt (Co)	NG	0.01	mg/L	Regularly	Regularly
	Lead (Pb)	0.001 - 0.007 [4]	0.001	mg/L	Regularly	Regularly
	Lithium (Li)	NG	NG	mg/L	Regularly	Regularly
	Mercury (hg)	0.000026	0.000026	mg/L	NA	NA
	Molybdenum (Mo)	0.073	0.073	mg/L	Regularly	Regularly
	Nickel (Ni)	0.025 - 0.150 [5]	0.025	mg/L	Regularly	Regularly
	Selenium (Se)	0.001	0.001	mg/L	Regularly	Regularly
	Silver (Ag)	0.00025	0.0001	mg/L	Regularly	Regularly
	Strontium (Sr)	NG	21	mg/L	Regularly	Regularly
	Sulphur (S)	NG	NG	mg/L	Regularly	Regularly
	Thallium (Tl)	0.0008	0.0008	mg/L	Regularly	Regularly
	Tin (Sn)	NG	NG	mg/L	Regularly	Regularly
	Titanium (Ti)	NG	NG	mg/L	Regularly	Regularly
	Uranium (U)	0.015	0.3	mg/L	Regularly	Regularly
	Vanadium (V)	NG	0.006	mg/L	Regularly	Regularly
Field Parameters	Temperature	Narrative	NG	˚C	Periodically	NA
	Dissolved Oxygen	NG	NG	%	Periodically	NA
	Dissolved Oxygen	5.5 - 9.5	NG	mg/L	Periodically	NA
	pH	6.5 - 9.0	NG	units	Periodically	NA
	ORP	NG	NG	mV	Periodically	NA
	TDS	NG	NG	mg/L	Periodically	NA
	Conductivity	NG	NG	us/cm	Periodically	NA
	Specific Conductance	NG	NG	us/cm	Periodically	NA
	Flow Rate	NG	NG	USGPM	NA	NA
	Wells Pumping	NG	NG	-	NA	NA
	Mine Water Elevation	NG	NG	Feet	NA	NA

Blue font indicates value below RDL, 50% of RDL is shown

🟨	**Exceeds CCME FWAL**	🟩	**Exceeds NSE EQS**	🟧	**Exceeds Both Guidelines**	🟦	**Field Parameter Exceeds Guideline**

[1] Copper: 2ug/L at [CaCO^3] = 0 to 120mg/L; 3ug/L at [CaCO^3] = 120 to 180mg/L; 4ug/L at [CaCO^3] > 180mg/L

[2] Aluminium: 5ug/L at pH < 6.5; or 100ug/L at pH > 6.5.

[3] Cadmium: 10E{0.86[log(hardness)]-3.2}

[4] Lead: 1ug/L at [CaCO^3] = 0 to 60 mg/L; 2ug/L at [CaCO^3] = 60 to 120mg/L; 4ug/L at [CaCO^3] = 120 - 180mg/L; 7ug/L at [CaCO^3] = 180mg/L

[5] Nickel: 25ug/L at [CaCO3] = 0 to 60mg/L; 65ug/L at [CaCO3] = 60 to 120mg/L; 110ug/L at [CaCO3] = 120 - 180mg/L; 150ug/L at [CaCO3] = 180mg/L

[6] Iron (Ferrous) results were noted at times to be higher than Total Iron. Variations between Iron (Ferrous) and Total Iron are likely due to different analytical methodologies and is considered to be acceptable if the results are within 25% relative percent difference.

NG = No guideline, NA = Not applicable

APPENDIX C Sampling and Sample Processing Procedures

Passive Treatment System Sampling and Sample Processing Procedures

Authored by: Allison Mackie **Revision Date: September 1, 2020**
Revised by: **Revision Date:**

Field Measurements:

Collect 1 L of water from each sample location in cleaned, acid-washed, and rinsed bottles.
Using field probe, record measurements from each sample location in the field by placing the probe in the water and allowing it to equilibrate in place for at least 30 seconds.
Using a flowmeter, measure the flow rate at several heights in the water at the sample location and at several distances from the shoreline (if applicable).

Lab Procedures:

Samples must be processed ASAP upon returning from the field. Samples that cannot be processed immediately must be refrigerated. **Ferrous iron, pH, and ORP must be measured immediately** upon returning to the lab and cannot be held for later.

A. Tests to be run on unfiltered water samples

1. Orion Meter Methods

   1.1. pH

Ensure pH probe is plugged into the meter. Select the pH option on the meter. Rinse the probe with nanopure water and calibrate the probe using the pH 4, 7, and 10 buffers following the meter's procedures.

- Obtain the water sample.
- Place the precalibrated probe into the sample.
- Gently stir the sample container or move the probe to ensure "fresh" sample water at the face of the probe. Ensure that the glass electrode remains submerged in the sample.
- The instrument will indicate by beeping and flashing ***READY*** when the pH being measured has stabilized. Note that too slow a mixing will give lower inaccurate readings. Should the ***READY*** sign not appear, watch until the pH stabilizes and perhaps fluctuates around two or three numbers.
- Record the pH.

   1.2. ORP

Ensure ORP probe is plugged into the meter. Select the ORP option on the meter. Rinse the probe with nanopure water then check the probe's calibration using the ORP standard. Record the result in your notebook.

- Rinse the probe with nanopure water and place into a small portion of sample.
- Allow probe to equilibrate, moving the probe around gently if necessary.

- Record the reading (units are mV). This result needs to be normalized to the standard solution used.

2. Turbidity (HACH Turbidimeter)

Turn on the turbidimeter and check its calibration following the procedure outlined on the instrument's screen. Using nanopure water, rinse out a 20-mL round sample cell and fill to the line indicated. Zero the instrument and record the reading of the sample blank.

- Obtain the water sample. Rinse the turbidity sample cell two or three times with a small portion of sample or milli-Q water.
- Pour the sample into the turbidity sample cell up to the fill line.
- Using a KimWipe, wipe the entire cell free of fingerprints or other marks. Holding the cell up to a source of natural light is useful for this.
- Cap and gently invert the cell a couple of times to mix the contents. Do not shake as tiny air bubbles can alter readings.
- Place the cell into the turbidimeter, close the lid and press ***READ***. The sample cell normally has a mark that shows which way to orient the sample cell for consecutive readings.
- Record the turbidity (units are NTU).

3. Apparent Colour (HACH Spectrophotometer)

- Power on the HACH spectrophotometer and find the colour program (465nm).
- Rinse and fill a 10ml sample cell with milli-Q water.
- Wipe sample cell with a Kim Wipe.
- Place blank sample cell into cell holder, shut lid and press Zero.
- Fill sample cell with sample (you will be doing both filtered (true) and unfiltered (apparent) colour).
- Wipe down sample cell, place into HACH, shut lid, press READ, and record color (units are PtCo)

4. HACH Reagent Methods

4.1. Total Fe (FerroVer)

- Detection range is 0.02 to 3.00 mg Fe/L
- See HACH method 8008 for more information.
- To preserve samples for later analysis, adjust the sample pH to less than 2 with concentrated nitric acid (approximately 2 mL per liter). No acid addition is necessary if the sample is tested immediately.
- Keep the preserved samples at room temperature for a maximum of six months.
- Before analysis, adjust the pH to 3–5 with 5 N sodium hydroxide solution.
- Correct the test result for the dilution caused by the volume additions.

4.2. Fe^{2+} (Ferrous Iron)

- **Samples must be measured immediately and cannot be held for later testing**
- 0.02 to 3.00 mg Fe^{2+}/L.
- See HACH method 8146 for more information.

4.3. Sulphate $SO_4{}^{2-}$ (SulfaVer)

- Hold time is 28 days at 4 °C unfiltered.
- Range is 0 to 70 mg SO_4^{2-}/L.
- See HACH method 8051 for more information.

B. To run on 0.45 µm filtered water samples:

5. <u>Filtration</u>

- Place filter apparatus on top of Erlenmeyer flask that has a tube coming out of the side that can be connected to a vacuum pump.
- Place a 0.45µm filter on the filter apparatus using tweezers and place "cup" over apparatus.
- Fill cup to top with deionized water.
- Draw water through filter by turning on vacuum pump.
- Empty the deionized water from flask into sink by detaching the filter surface and cup from the flask.
- Fill the flask with at least 40 ml of the water sample. If TSS was performed, use filtrate saved from the 1.5 um filter used for TSS analysis.
- Draw sample through filter by turning on vacuum pump.

6. <u>True Colour</u>

See above procedure for apparent colour.

7. <u>Dissolved Iron</u> (FerroVer)

See above procedure for total iron.

APPENDIX D Event Sampling Data

Table D.1 Total iron data (mg/L) from event sampling.

Sampling Date	System 1				System 2				
	Settling Pond Inlet	**Settling Pond Outlet**	**Wetland Inlet**	**Wetland Outlet**	**Settling Pond Inlet**	**Settling Pond 1 Outlet**	**Settling Pond 2 Outlet**	**Settling Pond 3 Outlet**	**Wetland Outlet**
09-Nov-20	1.38[1]	0.11	0.08	0.02	[2]	0.45	0.02	0.08	0.03
29-Dec-20	1.97	2.56	2.62	0.26	26.30	1.52	1.12	0.38	0.01
17-Feb-21	7.47	1.24	1.28	0.12	13.40	1.90	1.22	1.05	0.13
23-Mar-21	3.93	0.10	0.09	0.01	19.77	1.15	0.58	0.47	0.22
01-Apr-21	30.80	8.60	9.07	0.01	46.33	5.37	2.74	1.07	0.06
19-Apr-21	10.53	1.87	2.46	0.02	7.70	0.32	0.18	0.29	0.03
26-Apr-21	11.50	5.07	5.00	0.06	15.20	2.76	0.30	0.09	0.03
17-Jun-21	2.56	0.27	0.25	0.02	2.35	0.23	0.11	0.07	0.04
24-Jun-21		0.31	0.22	0.08	1.42	0.30	0.21	0.14	0.11
02-Jul-21	1.17	0.36	0.25	0.02	17.90	0.65	0.16	0.09	0.07
06-Jul-21	0.69[1]	0.33	0.29	0.02	0.26[2]	0.14	0.23	0.09	0.08
23-Jul-21	1.62	0.61	0.55	0.02	10.00[2]	0.37	0.21	0.11	0.08

[1] Cascade bypassed, sample taken under cascade

[2] No flow through cascade, sample taken at bottom of cascade

Table D.2 Ferrous iron data (mg/L) from event sampling.

Sampling Date	System 1				System 2				
	Settling Pond Inlet	**Settling Pond Outlet**	**Wetland Inlet**	**Wetland Outlet**	**Settling Pond Inlet**	**Settling Pond 1 Outlet**	**Settling Pond 2 Outlet**	**Settling Pond 3 Outlet**	**Wetland Outlet**
09-Nov-20	0.87[1]	0.32	0.28	0.29	[2]	0.30	0.30	0.28	0.27
29-Dec-20	2.58	0.43	0.50	0.30	2.68	0.38	0.42	0.30	0.34
17-Feb-21	5.30	0.40	0.41	0.30	10.33	0.39	0.33	0.42	0.39
23-Mar-21	1.22	0.03	0.01	0.01	14.42	0.37	0.29	0.35	0.29
01-Apr-21	10.48	1.83	2.18	0.33	8.63	0.99	0.53	0.34	0.24
19-Apr-21	1.38	0.41	0.39	0.37	1.48	0.38	0.32	0.31	0.28
26-Apr-21	3.73	0.54	0.50	0.30	4.35	0.49	0.29	0.32	0.35
17-Jun-21	1.93	0.15	0.32	0.41	0.29	0.12	0.08	0.09	0.04
24-Jun-21	1.79	0.16	0.09	0.10	0.01	0.16	0.14	0.20	0.09
02-Jul-21	0.80	0.15	0.07	0.10	11.10	0.01	0.03	0.03	0.15
06-Jul-21	0.42[1]	0.03	0.00	0.05	0.15[2]	0.03	0.06	0.09	0.16
23-Jul-21	0.77	0.07	0.02	0.00	0.14[2]	0.00	0.08	0.03	0.04

[1] Cascade bypassed, sample taken under cascade

[2] No flow through cascade, sample taken at bottom of cascade

Table D.3 Dissolved iron data (mg/L) from event sampling.

Sampling Date	System 1				System 2				
	Settling Pond Inlet	Settling Pond Outlet	Wetland Inlet	Wetland Outlet	Settling Pond Inlet	Settling Pond 1 Outlet	Settling Pond 2 Outlet	Settling Pond 3 Outlet	Wetland Outlet
09-Nov-20	0.59[1]	0.03	0.02	0.03	[2]	0.04	0.04	0.02	0.03
29-Dec-20	0.34	0.98	0.03	0.02	16.50	0.71	0.81	0.03	[3]
17-Feb-21	4.30	0.15	0.14	0.12	18.27	0.17	0.14	0.15	0.13
23-Mar-21	2.23	0.05	[4]	0.00	12.53	0.13	[4]	0.08	[4]
01-Apr-21	27.90	0.08	0.10	0.02	15.00	0.16	0.15	0.17	0.11
19-Apr-21	7.17	0.29	0.22	0.07	2.56	0.01	0.10	0.02	0.12
26-Apr-21	9.13	0.02	0.01	0.02	10.00	0.02	0.02	0.01	0.02
17-Jun-21	0.01	0.02	0.02	0.02	0.01	0.02	0.04	0.02	0.02
24-Jun-21	0.02	0.01	0.02	0.01	0.01	0.01	0.01	0.01	0.02
02-Jul-21	0.67	0.02	0.02	0.02	11.60	0.02	0.01	0.03	0.04
06-Jul-21	0.10[1]	0.02	0.03	0.01	0.02[2]	0.01	0.02	0.01	0.03
23-Jul-21	0.80	0.03	0.03	0.02	0.02[2]	0.01	0.03	0.02	0.04

[1] Cascade bypassed, sample taken under cascade

[2] No flow through cascade, sample taken at bottom of cascade

[3] Erroneous result, deleted

[4] Short of reagent for testing

Table D.4 Sulphate data (mg/L) from event sampling.

Sampling Date	System 1				System 2				
	Settling Pond Inlet	Settling Pond Outlet	Wetland Inlet	Wetland Outlet	Settling Pond Inlet	Settling Pond 1 Outlet	Settling Pond 2 Outlet	Settling Pond 3 Outlet	Wetland Outlet
09-Nov-20	340[1]	407	400	480	[2]	680	660	650	643
29-Dec-20	35	42	45	[3]	[3]	[3]	[3]	[3]	48
17-Feb-21	680	733	600	633	933	1007	1033	1073	773
23-Mar-21	540	547	577	680	927	813	813	847	677
01-Apr-21	1040	1053	1167	1047	1200	1387	1300	1080	1053
19-Apr-21	138	148	150	148	123	138	2000	1650	2633
26-Apr-21	875	583	592	608	733	550	558	742	700
17-Jun-21	517	410	393	400	683	643	613	630	647
24-Jun-21	557	450	447	460	610	653	660	650	643
02-Jul-21	363	410	433	397	630	557	627	610	623
06-Jul-21	187[1]	410	380	343	597[2]	580	577	580	560
23-Jul-21	437	457	473	467	627[2]	660	643	640	580

[1] Cascade bypassed, sample taken under cascade

[2] No flow through cascade, sample taken at bottom of cascade

[3] Sulphate concentration above detectable limit

Table D.5 pH data from event sampling.

Sampling Date	System 1				System 2				
	Settling Pond Inlet	Settling Pond Outlet	Wetland Inlet	Wetland Outlet	Settling Pond Inlet	Settling Pond 1 Outlet	Settling Pond 2 Outlet	Settling Pond 3 Outlet	Wetland Outlet
09-Nov-20	6.12[1]	7.74	7.70	7.40	[2]	8.48	8.39	8.42	7.55
29-Dec-20	7.06	7.76	7.68	6.97	6.49	7.57	8.01	6.99	7.41
17-Feb-21	6.85	8.37	8.08	7.74	6.78	8.55	8.57	8.44	7.84
23-Mar-21	7.21	8.74	8.75	7.73	6.95	8.16	8.37	8.37	7.81
01-Apr-21	6.71	7.13	7.14	7.14	6.67	7.51	7.83	8.00	7.48
19-Apr-21	7.02	7.80	7.86	7.64	7.08	8.65	8.41	8.34	7.64
26-Apr-21	7.13	7.85	7.83	7.71	7.02	8.35	8.37	8.20	7.86
17-Jun-21	7.53	8.23	8.27	7.58	7.77	8.15	8.38	8.10	7.66
24-Jun-21	7.37	8.05	7.87	7.75	7.87	8.04	8.05	8.15	7.07
02-Jul-21	7.74	7.97	8.05	7.90	6.93	7.75	8.06	8.22	7.73
06-Jul-21	6.23[1]	8.39	8.19	7.60	8.38[2]	8.28	8.34	8.42	7.90
23-Jul-21	7.57	8.09	8.10	7.72	8.13[2]	8.19	8.34	8.20	7.89

[1] Cascade bypassed, sample taken under cascade
[2] No flow through cascade, sample taken at bottom of cascade

Table D.6 Oxidation-reduction potential, ORP (E_H), data (mV) from event sampling.

Sampling Date	System 1				System 2				
	Settling Pond Inlet	Settling Pond Outlet	Wetland Inlet	Wetland Outlet	Settling Pond Inlet	Settling Pond 1 Outlet	Settling Pond 2 Outlet	Settling Pond 3 Outlet	Wetland Outlet
09-Nov-20	496[1]	501	505	504	[2]	492	492	490	506
29-Dec-20	397	425	429	441	407	428	425	449	435
17-Feb-21	420	465	492	598	362	420	431	431	597
23-Mar-21	447	432	434	459	445	420	404	408	464
01-Apr-21	346	372	372	387	372	359	350	348	390
19-Apr-21	412	491	500	555	494	436	439	439	543
26-Apr-21	383	377	379	390	385	368	385	378	385
17-Jun-21	390	400	398	394	443	436	433	441	442
24-Jun-21	421	417	420	430	439	425	426	422	461
02-Jul-21	439	435	429	446	409	424	417	416	448
06-Jul-21	795[1]	632	623	710	568[2]	564	559	559	590
23-Jul-21	582	630	624	649	579[2]	573	571	574	582

[1] Cascade bypassed, sample taken under cascade
[2] No flow through cascade, sample taken at bottom of cascade

Table D.7 Velocity data (m/s) from event sampling.

Sampling Date	System 1				System 2				
	Settling Pond Inlet	Settling Pond Outlet	Wetland Inlet	Wetland Outlet	Settling Pond Inlet	Settling Pond 1 Outlet	Settling Pond 2 Outlet	Settling Pond 3 Outlet	Wetland Outlet
09-Nov-20	0.60[1]	0.10	0.05	0.35	0.00[2]	0.00	0.00	0.00	1.53
29-Dec-20	0.07	0.07	0.02	0.02	0.13	[3]	[3]	[3]	0.39
17-Feb-21	[3]	[3]	[3]	[3]	[3]	[3]	[3]	[3]	[3]
23-Mar-21	[3]	[3]	[3]	[3]	[3]	[3]	[3]	[3]	[3]
01-Apr-21	0.50	0.14	0.13	0.54	0.27	0.34	0.38	0.39	0.28
19-Apr-21	1.03	0.17	0.23	0.42	0.12	0.17	0.12	0.17	1.07
26-Apr-21	1.26	0.21	0.23	2.52	0.59	0.35	0.40	0.38	1.57
17-Jun-21	0.30	0.07	0.17	0.21	0.00	0.08	0.12	0.16	0.89
24-Jun-21	0.24	0.07	0.11	0.23	0.00	0.00	0.00	0.00	0.50
02-Jul-21	0.35	0.05	0.03	0.22	0.36	0.31	0.30	0.30	0.14
06-Jul-21	0.00[1]	0.00	0.02	0.18	0.00[2]	0.00	0.00	0.00	0.13
23-Jul-21	0.38	0.03	0.09	0.10	0.00[2]	0.16	0.27	0.30	0.36

[1] Cascade bypassed, sample taken under cascade
[2] No flow through cascade, sample taken at bottom of cascade
[3] Impeller for flow probe lost and pending replacement

Table D.8 Turbidity data (FNU) from event sampling.

Sampling Date	System 1				System 2				
	Settling Pond Inlet	Settling Pond Outlet	Wetland Inlet	Wetland Outlet	Settling Pond Inlet	Settling Pond 1 Outlet	Settling Pond 2 Outlet	Settling Pond 3 Outlet	Wetland Outlet
09-Nov-20	15.8[1]	1.24	1.22	0.11	[2]	3.05	1.58	0.69	0.15
29-Dec-20	3.1	28.00	28.54	0.42	6.0	10.04	6.58	2.18	0.62
17-Feb-21	1.7	6.08	6.33	0.22	4.6	5.65	3.57	3.62	0.51
23-Mar-21	13.5	9.17	7.84	0.16	24.1	2.16	1.14	1.14	0.15
01-Apr-21	254.9	79.96	93.53	0.50	231.3	67.46	35.03	9.30	0.10
19-Apr-21	17.4	11.81	14.91	0.07	22.2	1.67	2.86	2.59	0.15
26-Apr-21	19.2	27.21	29.27	1.20	21.0	17.41	1.82	2.18	0.44
17-Jun-21	18.7	1.67	1.65	0.14	4.3	0.57	0.63	0.57	0.16
24-Jun-21	35.0	1.68	1.22	0.05	2.8	0.61	0.51	0.57	0.19
02-Jul-21	1.9	1.59	1.11	0.07	8.6	1.28	1.47	0.72	0.11
06-Jul-21	23.6[1]	2.49	2.16	0.03	0.8[2]	0.42	0.62	0.62	0.94
23-Jul-21	2.5	2.29	2.60	0.05	19.3[2]	1.13	0.93	1.04	0.13

[1] Cascade bypassed, sample taken under cascade
[2] No flow through cascade, sample taken at bottom of cascade

Table D.9 Apparent colour data (Pt/Co) from event sampling.

Sampling Date	System 1				System 2				
	Settling Pond Inlet	Settling Pond Outlet	Wetland Inlet	Wetland Outlet	Settling Pond Inlet	Settling Pond 1 Outlet	Settling Pond 2 Outlet	Settling Pond 3 Outlet	Wetland Outlet
09-Nov-20	55.0[1]	9.7	5.7	2.0	[2]	34.0	17.7	12.0	8.0
29-Dec-20	258.0	153.7	135.3	8.3	500.0	98.3	69.3	18.3	1.0
17-Feb-21	121.7	128.0	157.3	75.0	180.0	173.3	309.7	167.0	75.0
23-Mar-21	242.3	79.0	53.3	69.0	277.3	161.0	54.7	18.7	150.7
01-Apr-21	83.3	348.0	363.7	0.0	327.7	332.3	299.3	88.7	28.0
19-Apr-21	134.0	120.3	149.3	3.0	401.0	34.3	24.3	29.3	1.3
26-Apr-21	239.3	224.3	224.0	12.3	500.0	95.0	16.7	8.0	6.0
17-Jun-21	38.7	13.7	16.0	8.7	327.7	226.7	216.7	201.7	191.7
24-Jun-21	178.7	16.7	10.0	0.0	43.3	7.7	3.7	7.3	4.0
02-Jul-21	16.0	17.3	13.0	2.7	203.0	23.3	15.0	15.0	13.0
06-Jul-21	156.7[1]	23.7	22.3	4.0	9.0[2]	6.0	10.0	11.7	25.7
23-Jul-21	19.0	29.0	27.3	4.0	259.0[2]	24.0	18.0	17.0	10.7

[1] Cascade bypassed, sample taken under cascade

[2] No flow through cascade, sample taken at bottom of cascade

Table D.10 True colour data (Pt/Co) from event sampling.

Sampling Date	System 1				System 2				
	Settling Pond Inlet	Settling Pond Outlet	Wetland Inlet	Wetland Outlet	Settling Pond Inlet	Settling Pond 1 Outlet	Settling Pond 2 Outlet	Settling Pond 3 Outlet	Wetland Outlet
09-Nov-20	20.0[1]	12.7	4.0	8.3	[2]	8.0	5.3	6.7	7.0
29-Dec-20	302.3	64.3	6.0	8.3	500.0	50.7	52.3	12.0	124.7
17-Feb-21	119.3	54.0	138.7	104.7	500.0	52.0	18.0	48.0	86.3
23-Mar-21	205.7	73.0	61.3	74.0	500.0	17.0	46.7	15.0	343.3
01-Apr-21	500.0	35.3	59.7	11.0	500.0	87.3	95.3	117.0	57.0
19-Apr-21	500.0	365.7	181.3	45.7	169.7	4.0	85.7	13.0	91.0
26-Apr-21	398.0	4.0	0.7	9.3	500.0	1.0	7.0	1.0	3.0
17-Jun-21	43.0	3.7	2.0	4.3	2.0	19.7	9.0	6.3	5.3
24-Jun-21	98.3	2.0	2.7	1.0	1.3	2.0	3.0	4.7	4.0
02-Jul-21	39.0	1.3	4.0	4.0	500.0	3.3	3.3	5.0	7.0
06-Jul-21	2.7[1]	1.3	4.0	4.0	1.0[2]	1.0	3.0	1.0	6.0
23-Jul-21	48.3	4.0	4.0	11.0	2.0[2]	2.0	16.0	4.3	6.0

[1] Cascade bypassed, sample taken under cascade

[2] No flow through cascade, sample taken at bottom of cascade

APPENDIX E Modified TIS Model Results for Sensitivity Analysis

Table E.1 Modified TIS model results (i.e., iron effluent concentration) for different operating scenarios based on various areal rate constants at 20 °C produced by sensitivity analysis.

Scenario Number	System 1				System 2			
	C_{out} Target = 0.3 mg/L		C_{out} Target = Average C_{out}		C_{out} Target = 0.3 mg/L		C_{out} Target = Average C_{out}	
	k_{20} (m/year)	C_{out} (mg/L)	k_{20} *(m/year)*	C_{out} (mg/L)	k_{20} (m/year)	C_{out} (mg/L)	k_{20} (m/year)	C_{out} (mg/L)
1	410	0.290	1,930	0.0315	150	0.292	560	0.0457
2	410	0.246	1,930	0.0295	150	0.119	560	0.0282
3	410	0.087	1,930	0.0220	150	0.050	560	0.0212
4	410	0.178	1,930	0.0262	150	0.116	560	0.0279
5	410	0.296	1,930	0.0319	150	0.299	560	0.0465
6	410	0.250	1,930	0.0297	150	0.121	560	0.0284
7	410	0.088	1,930	0.0220	150	0.050	560	0.0212
8	410	0.182	1,930	0.0265	150	0.118	560	0.0282
9	915	0.195	3,540	0.0258	375	0.295	815	0.0468
10	915	0.299	3,540	0.0299	375	0.133	815	0.0291
11	915	0.086	3,540	0.0214	375	0.069	815	0.0221
12	915	0.115	3,540	0.0225	375	0.101	815	0.0254
13	1,090	0.295	5,250	0.0248	650	0.294	1,380	0.0307
14	1,090	0.298	5,250	0.0249	650	0.296	1,380	0.0308
15	1,090	0.299	5,250	0.0249	650	0.298	1,380	0.0309
16	1,090	0.295	5,250	0.0248	650	0.294	1,380	0.0307
17	865	0.296	4,100	0.0318	315	0.297	1,185	0.0464
18	865	0.250	4,100	0.0296	315	0.120	1,185	0.0284
19	865	0.088	4,100	0.0220	315	0.050	1,185	0.0212
20	865	0.181	4,100	0.0264	315	0.118	1,185	0.0281
21	865	0.299	4,100	0.0319	315	0.300	1,185	0.0468
22	865	0.252	4,100	0.0297	315	0.121	1,185	0.0285
23	865	0.088	4,100	0.0220	315	0.050	1,185	0.0212
24	865	0.183	4,100	0.0265	315	0.119	1,185	0.0282
25	2,060	0.196	7,875	0.0258	835	0.299	1,780	0.0475

Scenario Number	**System 1**				**System 2**			
	C_{out} Target = 0.3 mg/L		**C_{out} Target = Average C_{out}**		**C_{out} Target = 0.3 mg/L**		**C_{out} Target = Average C_{out}**	
	k_{20} (m/year)	**C_{out} (mg/L)**	**k_{20} *(m/year)***	**C_{out} (mg/L)**	**k_{20} (m/year)**	**C_{out} (mg/L)**	**k_{20} (m/year)**	**C_{out} (mg/L)**
26	2,060	0.299	7,875	0.0299	835	0.135	1,780	0.0295
27	2,060	0.086	7,875	0.0215	835	0.070	1,780	0.0224
28	2,060	0.119	7,875	0.0227	835	0.106	1,780	0.0261
29	2,325	0.298	11,205	0.0249	1,380	0.298	2,935	0.0309
30	2,325	0.299	11,205	0.0249	1,380	0.299	2,935	0.0309
31	2,325	0.300	11,205	0.0249	1,380	0.300	2,935	0.0310
32	2,325	0.298	11,205	0.0249	1,380	0.298	2,935	0.0309
33	410	0.050	1,930	0.0200	150	0.041	560	0.0202
34	2,325	3.941	11,205	0.2119	NA	NA	NA	NA
35	2,325	8.598	11,205	0.9333	NA	NA	NA	NA
36	2,325	0.600	11,205	0.0309	NA	NA	NA	NA
37	2,325	0.501	11,205	0.0289	NA	NA	NA	NA
38	2,325	0.647	11,205	0.0318	NA	NA	NA	NA

NA = Not applicable