

PHOSPHORUS RETENTION IN ON-SITE WASTEWATER SYSTEMS
IN NOVA SCOTIA

by

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ABSTRACT

The use of residential on-site wastewater systems (OWS) is one of the main non-point sources of phosphorus (P) loading to aquatic systems, which is the primary cause of accelerated blue-green algal growth and eutrophication of surrounding aquatic systems. Research is lacking on the long term treatment performance of the OWSs currently used in Nova Scotia. This study examines how P is removed and retained within OWS disposal fields to gain a better understanding of how to limit P loading to surrounding aquatic systems and aid in the management of OWSs across Nova Scotia in regards to P loading. The first phase utilized an existing set of field scale OWSs at the Bio-Environmental Engineering Center in Truro, Nova Scotia. Two types of disposal fields, which had been receiving wastewater for 4-8 years, were sampled. These included C2 raised contour trench systems (C2R) and lateral flow sand filter (LFSF) systems. The second phase of the sampling was conducted in the Thomas Brook Watershed (TBW) located in Berwick, Nova Scotia. Three full-scale on-site wastewater systems that range in age from 10-20 years old were selected for sampling. Soil samples were taken at various depths, and at various spatial locations, within each disposal field to assess the spatial distribution of the P in the disposal field. The three full-scale systems in Berwick each had 6 piezometers installed in the disposal field for the collection of water samples on a monthly basis for one year to capture seasonal variations in P concentrations. Electron microprobe analysis and back scattered electron images were used to identify P-rich solids on soil grains to determine if mineral precipitation reactions involving P could be occurring within the disposal field as a possible long term treatment mechanism. This study highlights the importance of the physical and chemical characteristics of the sand media used in the disposal fields to increase P treatment. The soil media used in the OWSs observed in this study showed a minimal amount of P treatment as they mature, which should be taken into account particularly in watersheds that are currently experiencing water quality issues and are at risk for eutrophication of surrounding surface water.

LIST OF ABBREVIATIONS USED

Al	Aluminum
AP	Available Phosphorus
BEEC	Bio-Environmental Engineering Center
BNQ	Bureau de Normalisation du Quebec
Ca	Calcium
CaCO ₃	Calcium Carbonate
C2R	C2 Raised Contour Trench
EDS	Electron Dispersice Spectroscopy
Fe	Iron
FePO ₄	Iron (III) Phosphate
H	Hydrogen
H ₂ O	Water
HPO ₄ ²⁻	Monohydrofen Phosphate
H ₂ PO ₄ ⁻	Dihydrogen Phosphate
H ₃ PO ₄	Phosphoric Acid
<i>i</i>	Hydraulic Gradient
<i>k</i>	Hydraulic Conductivity / Coefficient of permeability
K ₂ S ₂ O ₈	Potassium Persulfate
LFSF	Lateral Flow Sand Filter
Mg	Magnesium
Mn	Manganese
NS	Nova Scotia
O ₂	Oxygen
OH ⁻	Hydroxide
ORP	Oxidation Reduction Potential
OWS	On-Site Wastewater System
%	Percentage
P	Phosphorus
PO ₄ ³⁻	Phosphate
P ₂ O ₅	Phosphorus Pentoxide
PVC	Perforated Polyvinyl Chloride
STE	Septic Tank Effluent
TBW	Thomas Brook Watershed
TN	Total Nitrogen
TOC	Total Organic Carbon
TP	Total Phosphorus
USEPA	United States Environmental Protection Agency

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CHAPTER 1: INTRODUCTION

1.1 Project Context

Across Canada many aquatic systems are experiencing water quality issues caused by algae blooms, which have negative impacts on aquatic habitat (Chambers et al, 2001). Phosphorus (P) loading is of concern because it is the primary cause of accelerated blue-green algal growth and eutrophication of freshwater aquatic systems (Dillon and Rigler, 1974; Schindler, 1977). One of the main non-point sources of P loading to aquatic systems is the use of residential on-site wastewater systems (OWS) (Schindler, 1977). A typical OWS is designed to pre-treat the wastewater from a single or cluster of households using a septic tank. The effluent is then filtered through a disposal field, which is constructed using either native or imported material. Once the effluent has percolated through the disposal field, the treated water is discharged into the surrounding soil and water systems (McCray et al, 2008). The P from the effluent can be transported into the surrounding surface water by two main pathways: (i) a hydraulic failure of the disposal field as a result of improper drainage or clogging, and (ii) the disposal field is unable to remove or retain P (USEPA, 2002).

Currently in Nova Scotia, approximately 45% of residents rely on OWSs to treat and dispose of their wastewater (Nova Scotia Environment, 2011). The majority of the OWSs installed in NS are designed to allow the effluent to flow laterally within imported sand media which is typically used to treat the effluent instead of native soils due to low permeability in the native soils (Havard et al, 2008). Many of these systems have been loaded with P over an extended period of time and as these systems age, hydraulic failure can occur allowing P to be transported into surrounding aquatic systems. In general, the geology of Nova Scotia provides an adverse environment for disposal field construction due to the shallow depth to bedrock (See Appendix A), high water tables and soils with low permeability (Havard et al, 2008). Nova Scotia also contains a large amount of developed lakeshore area, which is at risk of contamination from on-site wastewater systems.

1.2 Research Objectives

The main research goal of this thesis is to determine how P is removed and retained within the disposal field of OWSs to gain a better understanding of how to limit P loading to surrounding aquatic systems. This research could aid in the management of OWSs across Nova Scotia with regards to P loading and was undertaken in recognition of the research gaps identified in the following literature review. To achieve this main research goal, the following research objectives were developed:

- i. Characterize the forms of P present within OWSs that have been operational for several years to determine if the primary mechanism of retention is reversible adsorption or secondary mineralization.
- ii. Evaluate the spatial distribution of P within selected OWSs to determine if current OWS designs are effectively using the total mass of treatment media in the system.
- iii. Assess the long term treatment performance of OWSs and characterize the potential for P loading to aquatic systems associated with systems currently used in Nova Scotia

CHAPTER 2: LITERATURE REVIEW

2.1 Phosphorus Management in Nova Scotia

With increasing demands on the development of watersheds across Nova Scotia, it is necessary to implement methods to ensure that development projects do not negatively impact watersheds and surrounding aquatic systems. One of the major impacts from development on aquatic systems is P loading from residential OWSs, as P is a limiting nutrient for algal growth in aquatic environments. Some jurisdictions have implemented the use of lakeshore development capacity models that require P-loading coefficients in an effort to quantify potential P inputs to aquatic systems prior to development (Dillon et al., 1986; Brylinsky, 2004). In areas that have a large amount of P inputs due to OWSs, lakeshore development is limited to protect the aquatic environment. To ensure excessive P loading does not occur, Nova Scotia has implemented a model for determining the extent to which a watershed can be developed before aquatic systems begin to have impaired water quality due in part to excessive P loading (Brylinsky, 2004).

The equation used in the Nova Scotia model to predict the P concentration in a lake is based on the previous work developed by Vollenweider (1968; 1975). The Vollenweider model uses the theory that the change in P in the lake over time is equal to the amount of P that is entering the lake minus the P that is lost to sediments and outflow. The major assumptions of this model are that the P entering the lake is mixed throughout the lake; the outflow concentration of the P is equal to the lake concentrations; the amount of P that is lost due to sedimentation on the lake bottom is directly proportional to the lake concentration; and any seasonal fluctuations of P and hydraulic loading are neglected (Brylinsky, 2004).

The major input in this model is the amount of P loading to the lake as a result of development in the watershed. This value is based on the amount of P produced per capita and the proportion of that P which would enter the lake (Brylinsky, 2004). The model also takes into account the adsorption capacity of the native soils where the septic systems are located. There are many factors that could affect this value such as whether

native soil or imported soil was used for the construction of the disposal field, the age of the disposal field, the maintenance history, the physical and chemical characteristics of the soil and the interactions of the disposal field with the water table (Brylinsky, 2004). Data are minimal on the retention of P within OWSs typically used in Nova Scotia and the degree to which P from OWSs can impact the nearby aquatic systems. Therefore major assumptions have been made when calculating the P loading due to development, which is a major component in the model. Currently, a majority of the estimates are based on research conducted by Robertson (2012).

2.2 Phosphorus Dynamics in Soil and Water Systems

Phosphorus is an essential nutrient for plants, animals and humans. The P cycle (Figure 2.1) is strongly influenced by biological activity, the chemistry of the soil solution such as pH, ionic strength, redox potential, and many environmental factors including soil moisture content and temperature (Bünemann et al. 2011).

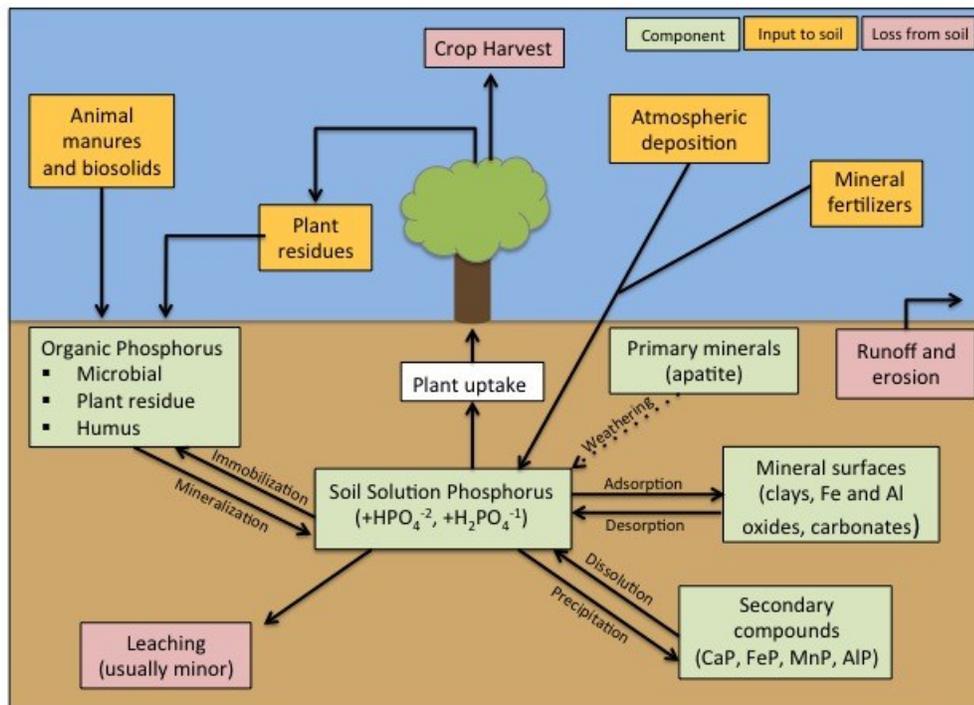


Figure 2.1 A schematic of the dynamics involved in the phosphorus cycle.

Under natural conditions, aquatic systems have low concentrations of P. However excessive loading of P into aquatic systems can be problematic resulting in algae growth, especially in lakes. When P is found in low concentrations, it is a limiting nutrient but as the P concentration increases the nutrient and algae level increases creating a eutrophic environment (Zurawsky et al., 2004; Zanini et al, 1998; Dillon and Rigler, 1974; Schindler, 1977).

2.2.1 Phosphorus Sources

The P found in soil originates primarily from two sources: the weathering and erosion of rocks containing P from minerals such as apatite or from non-point sources such as runoff from agricultural areas or residential OWSs. The weathering of P containing rocks and minerals is not a major contributor as these are slow dissolution processes that rarely release enough P into the soil to meet plant requirements (Walker and Syers, 1976).

2.2.2 Forms of Phosphorus in Soil

Phosphorus exists in various forms within soil. It is most commonly differentiated within the soil as plant available inorganic P and three forms that are not plant available: organic P, absorbed P and mineral P. The transformation processes of P are: weathering and precipitation; mineralization and immobilization; and adsorption and desorption (Bünemann et al. 2011). Inorganic P is increased by weathering, mineralization and desorption whereas it is decreased by immobilization, precipitation and adsorption. Both inorganic P and organic P are controlled by precipitation, weathering and sorption (Bünemann et al. 2011).

A large portion of P is retained in soil due to adsorption. The soil has a limit to the amount of P it can retain through adsorption and once that capacity is reached the excess dissolves and moves freely in the water, and is typically present as the orthophosphate ion (PO_4^{3-}) (Domagalski and Johnson, 2012). As the soil reaches saturation, the dissolved P migrates into unsaturated zones and into underlying aquifers.

2.2.3 Forms of Phosphorus in Water

There are two main forms of P that exist in water: dissolved and particulate. Dissolved P occurs primarily as inorganic P in the form of orthophosphates (PO_4^{3-} , HPO_4^{2-} , H_2PO_4^- , H_3PO_4), which is soluble and is readily available to plants and algae. Phosphorus can also adsorb onto soil particles when clays and reactive metal oxides are present (Domagalski and Johnson, 2012). Controlling factors that affect the adsorption of orthophosphate are pH (Figure 2.2) and dissolved oxygen. When soil and water have a pH greater than 7 and there is no dissolved oxygen, the soil becomes saturated with orthophosphate more quickly compared to when the pH of the soil and water are less than 7. When there is an absence of oxygen in the soil, iron oxides will dissolve and release the adsorbed P back into the water, but this process relies on the presence of specific bacteria (Domagalski and Johnson, 2012).

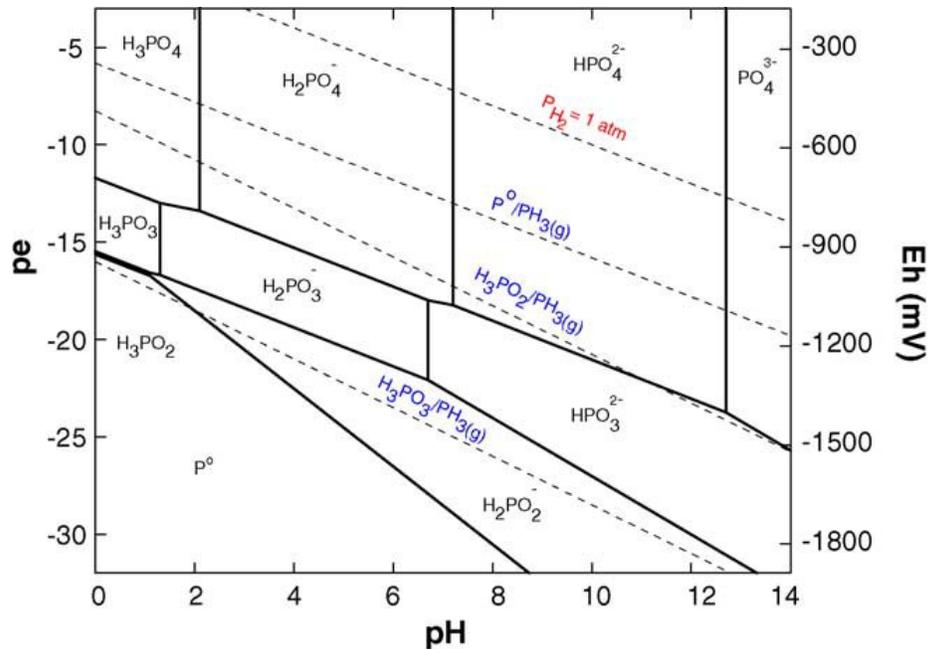


Figure 2.2 Eh/pe-pH diagram for phosphorus species in water, 10^{-6}M , with the dashed line $\text{P}_{\text{H}_2} = 1 \text{ atm}$ shown for reference for a reducing environment on Earth (Hanrahan et al., 2005).

2.2.4 Phosphorus Mineralization and Immobilization

The transformation of organic P to forms of plant available P, known as orthophosphates, in the soil is primarily determined by the rates of mineralization and immobilization. The process of mineralization is the microbial conversion of organic P to orthophosphates, which are inorganic P forms. The immobilization of P is the biological conversion of inorganic P to organic P (Sims et al., 2005; Bünemann et al. 2011). This process occurs when the microbes consume the inorganic P turning the P into the organic form, which is not available to plants. As the microbes die, the P returns to its inorganic form and becomes available for the plants to uptake. The accumulation rate of organic P declines as equilibrium is reached between mineralization and the organic inputs (Sims et al., 2005; Bünemann et al. 2011).

2.2.5 Phosphorus Precipitation and Dissolution

The natural P levels found in most soil originate from the weathering of apatite. When the source of P is from weathering of native materials, the prominent forms of P found in the soil are aluminum phosphates, iron phosphates and organic P (Bünemann et al. 2011). The typical P concentration in soil solution ranges from <0.01 mg/L to 1 mg/L, but can be as high as 7 to 8 mg/L when there is organic waste present in the soil (Dillon and Rigler, 1974; Bünemann et al. 2011).

Orthophosphates have a tendency to act similar to a hard Lewis base within the soil (Bünemann et al. 2011). Due to this tendency the orthophosphates form soluble inner and outer sphere complexes with hard Lewis acids (H^+ , Ca^{2+} , Mg^{2+} and Fe^{3+}). Within the normal pH range of 4.0 – 9.0, the majority of the orthophosphates present are HPO_4^{2-} (pH >7.2) or $H_2PO_4^-$ (pH 4.2 – 7.2) (Figure 2.2). At higher pH values the soluble complexes such as $CaHPO_2$ or $CaHPO_4^-$ represent a significant portion of the P present in the soil. In some cases these soluble complexes chemically disassociate converting them to orthophosphates (Bünemann et al. 2011).

The process of precipitation occurs when P ions react with cations such as aluminum, iron, calcium, and magnesium (Nelson and Parsons, 2007). The reaction forms an amorphous solid precipitate. Precipitation of P ions has a tendency to occur at

high concentrations of phosphate in combination with a high concentration of cations. When the soil solution is acidic, P has a tendency to precipitate with either aluminum or iron, if there is a supply of these cations. When the soil solution has a high alkalinity, P will precipitate if there is a high calcium or magnesium concentration present (McGechan and Lewis, 2002; Bünemann et al. 2011).

The process of dissolution is the solubilisation of the precipitate. Dissolution occurs when the concentration of either the phosphate or the metallic cation decreases below the solubility product of the compound (Bünemann et al. 2011). The solubility of P is influenced by the pH and Eh of the soil solution. P retention in acidic soils is regulated by iron and aluminum oxides and in alkaline soils it is regulated by calcium-based minerals (McGechan and Lewis, 2002; Bünemann et al. 2011). Soils that are high in iron-minerals associated with P are stable under aerobic or drained soil conditions. The decrease in Eh increases P solubility through the reduction of FePO_4 . When the pH ranges from 5 – 8 the P solubility is low resulting in a low P concentration in the soil solution. When the Eh value decreases from 300 mv to -250 mv, the solubility of P increases at all pH values resulting in high P concentrations in the soil solution. The solubility of P is at its highest when the environment has low pH and low Eh. Under these conditions, an increase in the P solubility is a result of the reduction of ferric ion in ferric phosphate (Bünemann et al. 2011).

When soil has high clay content and also a high amount of iron and aluminum oxides it has the ability to strongly retain P. The iron and aluminum oxides provide surfaces for the P retention (Robertson, 1995; Bünemann et al. 2011). High organic content in the soil can also provide surfaces for P sorption. Alkaline soils containing calcium carbonate also have the ability to retain P through sorption of P on grain surfaces and precipitation of calcium ions with P (Bünemann et al. 2011).

Precipitation and adsorption rates are affected by temperature (Bünemann et al. 2011). Both rates are accelerated when the temperature increases. Under high temperatures the time required for the formation of crystalline forms is also decreased (Bünemann et al. 2011).

2.2.6 Phosphorus Sorption and Desorption

The process of sorption involves two steps: adsorption, which is the accumulation of P on the surfaces of soil constituents and absorption, which is the diffusion of P into soil constituents (Bray and Weil, 2008; Bünemann et al. 2011). Orthophosphates have the ability to adsorb to the surfaces of hydrous oxides, clay minerals and carbonates by replacing H₂O or OH⁻. The sorption process is controlled by the concentration of P in the soil. When the soil particles become saturated with P, the P concentration in the soil solution is high (Bünemann et al. 2011). When the solubility product constants are exceeded, secondary P solid phases can proceed to precipitation out of saturated solutions. The primary cations involved in this reaction are calcium, aluminum, and iron (Bünemann et al. 2011).

Adsorption differs from precipitation in that adsorption is a reversible chemical binding of P to the soil constituents while precipitation is a more permanent change in the chemical properties of the P as it is removed from the soil solution (Bünemann et al. 2011). Soils that have higher concentrations of metal oxide minerals have the potential to adsorb more P than other soil solutions. When the pH of the soil solution is between 6 and 7 the P is in its most plant available form (Bünemann et al. 2011; Robertson, 1995; Parfitt et al., 1975; Rajan, 1975).

The process of desorption is when there is a release of the adsorbed P from its bound state in the soil solution. Desorption is usually a slow process unlike adsorption which occurs quickly (Bünemann et al. 2011). The depletion of P in the soil solution releases P from the mineral surfaces until a new equilibrium is reached. Phosphate buffering is a phenomenon that occurs between P adsorption and desorption in order to maintain equilibrium between the solid phase and P in the soil solution. Similarly to sorption, desorption has a two-step process: the rapid release of P present on the particle surfaces followed by the slow release of P that is retained within the solid phase (Bünemann et al. 2011).

2.2.7 Phosphorus Transport Mechanisms

The identification of the transport mechanisms of P is important when evaluating potential management solutions. The transport of P from soil is primarily due to surface flow and erosion (Sharpley et al. 2001). When water travels across the soil surface it can transport the soluble P or erode and transport the particulate P. Erosional processes transport particulate P through the movement of soil particles such as clays that contain a high level of P compared to larger grain sizes (Sparks, 2003). Subsurface flow predominately transports dissolved P but can also contribute to the movement of particulate P (Gentry et al. 2007; Kinley et al. 2007).

2.2.8 Analytical Methods for Phosphorus

The characterization of the various forms of P present in soil and water is important to gain a better understanding of the transformation processes that are taking place in the OWSs. The different forms of P differ in their bioavailability and their potential to become mobile resulting in potential P loading to surrounding surface water systems.

The two main forms of P that are identified and discussed in this study are available P and total P. The loosely bound form of P is referred to as available P (AP) and is assumed to be reversibly adsorbed and dissolved inorganic plant available P. The acid extractable P concentrations in the sediment are subsequently referred to as the measured total P concentrations (TP) in the sediment, which would include: inorganic P, organic P, adsorbed P, and mineral P.

2.2.8.1 Analytical Methods to Estimate Available Phosphorus (AP) in Soil

There are four main methods that are used in the United States and Canada to determine the amount of available P (AP) in soil: Bray and Kurtz P-1, Mehlich 1, Mehlich 3, and Olsen P.

The Bray and Kurtz P-1 method was developed by Roger Bray and Touby Kurtz in 1945 and uses an extraction solution that contains fluoride that enhances the P release from aluminum phosphates by forming aluminum-fluoride complexes (Bray and Kurtz, 1945). The extraction solution is acidic with a pH of 2.6, which aids in the dissolution of available P from aluminum, iron and calcium forms in the soil. This method is not suitable for soils that have a pH greater than 6.8, soils that have a high clay content with a high degree of base saturation, soils that have a calcium carbonate equivalent to greater than 7% of the base saturation, or soils that have a large amount of lime (Kovar and Pierzynski (eds), 2009).

The Mehlich 1 method was developed in the 1950s by Mehlich and his co-workers (Mehlich, 1953; Nelson et al. 1953) and is a multi-element extractant for P, potassium, calcium, magnesium, copper, iron, manganese and zinc. This method is best used on soils that have a low cation exchange capacity, organic matter that is <5% and acidic soils with a pH of less than 6.5 (Kovar and Pierzynski (eds), 2009).

The Mehlich 3 method was developed by Mehlich in 1984 and was the improved version of the Mehlich 1 (Mehlich, 1984). It is similar to the Mehlich 1 in that it is a multi-element extractant but it is more effective when predicting AP in neutral and alkaline soils (Kovar and Pierzynski (eds), 2009). The Mehlich 3 is widely used across the US and Canada and is used for various soil types that are both acidic and basic. This method is comparable to the Bray and Kurtz P-1 test because the acidic extraction solution contains ammonium fluoride. Mehlich 3 also contains acetic acid in the extraction, which allows for the release of available P in most soils (Kovar and Pierzynski (eds), 2009).

The Olsen P method was developed by Sterling Olsen in 1954 (Olsen et al., 1954) and is suited for calcareous soils with >2% calcium carbonate. This method increases the solubility of P by decreasing the concentration of soluble calcium, aluminum and iron by the formation of CaCO_3 and iron and aluminum oxyhydroxides. The extraction also increases the desorption of available P in the solution (Kovar and Pierzynski (eds), 2009).

2.2.8.2 Analytical Methods to Estimate Bound Phosphorus in Soil

There are various extractions that can be used to determine particular forms of P in the soil. The most widely used methods are sodium carbonate fusion, perchloric acid digestion and aqua regia extraction (Crossland et al., 1995). The most commonly used method to distinguish between inorganic and organic P in soil is the aqua regia extraction. The aqua regia extraction involves digesting the sample using a strong acid at a high temperature to remove both organic P and inorganic forms of P in the sample resulting in the total acid extractable P concentration (Bünemann et al. 2011; Kovar and Pierzynski (eds), 2009). Knowing the inorganic and total P concentration, the estimated organic concentration can be determined. This method can overestimate the organic P concentration when inorganic polyphosphates are present because they do not react with the reagent therefore the inorganic polyphosphate are included in the organic fraction of the soil (Bünemann et al. 2011; Kovar and Pierzynski (eds), 2009).

2.2.8.3 Analytical Methods to Estimate Phosphorus in Water

In water, the two main forms of P that are measured are dissolved P and total P. In order to measure the total P content of a water sample, the particulate P must be converted to orthophosphate by digesting in a strong acid at a high temperature. The digestion oxidizes the organic matter and allows for the release of P as orthophosphate (Kovar and Pierzynski (eds), 2009). Once the digestion is complete the samples can either be tested using colorimetric analysis or by inductively coupled plasma (ICP) spectrometry. When determining the total dissolved P fraction, the sample must be filtered using a 0.45 µm filter prior to digestion. To determine the total P concentration, which includes dissolved and particulate, an unfiltered sample is used for the digestion and is shaken prior to analysis to suspend the particulate matter in the sample (Kovar and Pierzynski, 2009).

2.3 Phosphorus Transport in On-site Wastewater Systems

2.3.1 Phosphorus Mobility and Retention Mechanisms in On-site Wastewater Systems

OWSs are known to be a main source of P loading to aquatic systems; therefore P mobility in septic system plumes can present significant environmental risks. Research has established that P has the ability to be mobile in aquifers at high concentrations over short distances (Jones and Lee, 1979; Robertson et al, 1991; Whelan, 1988; Wilhelm et al., 1994). Distinct P plumes in groundwater that have exceeded 15 meters distance from the source of a mature septic system have been observed (Walter et al., 1996; Harmon et al., 1996; Robertson et al., 1998).

The attenuation of P in septic systems is controlled by soil adsorption and mineral precipitation reactions (Reneau et al, 1976; Lance, 1977; DeCamargo et al, 1979; Jones and Lee, 1979; Bolt, 1982; Whelan and Barrow, 1984; Robertson et al., 1998; Harmon et al., 1996). The adsorption of P in soil is a limited treatment mechanism affected by the soil composition, available sorption sites, pH, redox conditions, groundwater velocity and the amount and duration of loading (Cogger et al., 1988; McCray et al., 2009). The soil that has the highest capacity for P sorption contains metal-oxide minerals such as iron, aluminum and manganese and is fine grained (Brady and Weil, 2008). The process of mineral precipitation of P is a slower reaction than adsorption but may provide an unlimited solution for a long-term treatment mechanism if the soil chemistry is conducive to the reaction (Jones and Lee, 1979; Harmon et al., 1996). The precipitation of P can occur from P in the solution as well as P that has been sorbed (Hill and Sawhney, 1981). Mineral precipitation reactions with P can occur with metal cations that include iron, aluminum, manganese, magnesium and calcium (Nriagu and Dell, 1974; Stumm and Morgan, 1981; Nelson and Parsons, 2007). The pH of the soil and the nature of the available cations dictate what cation the P will react and precipitate with; under basic conditions calcium and magnesium phosphates will precipitate and acidic conditions favor the precipitation of iron and aluminum phosphates (Bünemann et al., 2011). The precipitation of P is perceived to be an irreversible process (Robertson et al., 1998) but P

is vulnerable to be remobilized if the redox conditions change (Zurawsky et al., 2004). To study P solid phases various methodologies can be used: x-ray diffraction, electron microscopy, x-ray absorption fine structure spectroscopy, or x-ray absorption near edge spectroscopy. The most common P solid phases are fluorapatite, hydroxyapatite, tricalcium phosphate, octacalcium phosphate, dicalcium phosphate, and dicalcium phosphate dehydrate which are listed as being increasingly soluble under high pH conditions (Bünemann et al. 2011). Under low pH conditions, the most common P solid phases are variscite and strengite and are possible controls of inorganic P solubility (Lindsay, 1979 as cited by Bünemann et al. 2011).

Studies of attenuation of P in septic systems in Ontario have shown that mineral precipitation reactions are occurring (Zurawsky et al., 2004). Zanini et al. (1998) also observed P -enriched layers located near the infiltration pipes in disposal fields, which represented the zone with the highest oxidation. Using electron microprobe imaging secondary P-rich solids were observed as either coatings or as discrete grains occurring with aluminum and iron as the dominant cations (Zanini et al., 1998). This research suggested that two attenuation reactions could be occurring: (1) noncalcareous soils under acidic conditions lead to the dissolution of gibbsite [$\text{Al}(\text{OH})_3$] subsequently leading to the precipitation of variscite [$\text{AlPO}_4 \cdot 2\text{H}_2\text{O}$] and (2) the conversion of ferrous iron [$\text{Fe}(\text{II})$] to ferric [$\text{Fe}(\text{III})$] causing the precipitation of strengite [$\text{FePO}_4 \cdot 2\text{H}_2\text{O}$] (Zanini et al., 1998). Robertson (2012) also found the presence of strengite ($\text{FePO}_4 \cdot 2\text{H}_2\text{O}$) and variscite ($\text{AlPO}_4 \cdot 2\text{H}_2\text{O}$) in noncalcareous sand in disposal fields in Ontario indicating attenuation of P through mineral precipitation and not adsorption. The study by Robertson (2012) indicated that noncalcareous sand and wastewater effluent oxidation reactions created acidic conditions, which are favorable to the precipitation of iron and aluminum with P.

2.3.2 On-site Wastewater Systems in Nova Scotia

In Nova Scotia, OWSs are used to dispose of sewage from residential dwellings in rural areas (Figure 2.2). OWSs that are installed in Nova Scotia must take into account the unique geology and topography that are present across the province. The topography is generally hilly, the depth to bedrock is largely shallow, water tables are high and soils with low permeability are generally present (See Appendix A). These conditions provide

an adverse environment for disposal fields and this must be taken into account when designing and installing an OWS (Nova Scotia Environment, 2009).

Typical OWSs start with a building sewer, which is a pipe that connects the building to the disposal system, largely a septic/holding tank (Figure 2.3). The septic tank holds the sewage to allow for settling of solids. Once the sewage has settled, septic effluent leaves the septic tank through an effluent line, which is connected to the disposal field. The disposal field distributes the septic tank effluent into the soil and provides treatment to the effluent. The disposal field typically consists of a gravity distribution pipe that is perforated and installed on a slope to ensure the effluent is evenly distributed in the disposal field (Nova Scotia Environment, 2009). The distribution pipe is installed in a distribution trench that consists of crushed rock to provide protection of the distribution pipe. The disposal field is designed using a filter sand material that is used to filter the effluent as it passes through the disposal field removing all harmful bacteria and organisms and retaining them in the field. The bottom of the disposal field is typically a native or installed material that has low permeability to ensure that the effluent travels laterally through the disposal field and not into the material vertically below the disposal field prior to being filtered (Nova Scotia Environment, 2009).

There is a requirement that all systems must have a minimum soil thickness of one meter between the bottom of the disposal field and the groundwater and bedrock in order to avoid the contamination of the groundwater from the effluent prior to filtration. If the local soil material onsite does not meet this criteria, a built-up disposal field using imported material is designed to ensure the protection of the groundwater (Nova Scotia Environment, 2009).

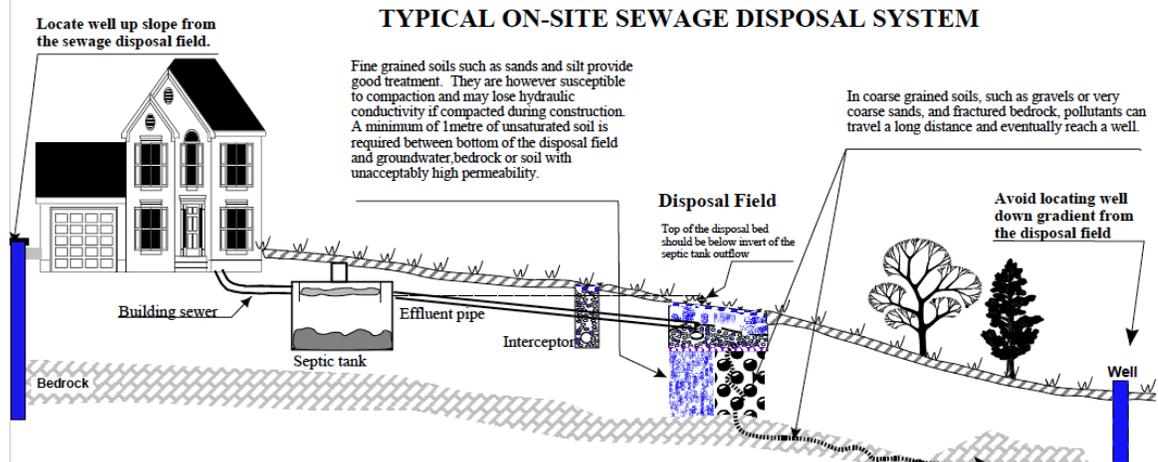


Figure 2.3 Typical on-site sewage disposal system (Nova Scotia Environment, 2009).

There are several different types of OWSs that can be installed in Nova Scotia. Most designs use imported sand media and rely on lateral flow of wastewater through the media. The types of OWSs used in Nova Scotia are the C1 contour trench, C2 contour trench (standard and raised), C3 contour trench, mound, area bed, multiple trench, and lateral flow sand filters (Table 2.1) (See Appendix B for disposal system cross-sections).

Table 2.1 Disposal system selection options (Nova Scotia Environment, 2009).

DISPOSAL SYSTEM SELECTION OPTIONS*

* *Not all options provided are suitable for each property being assessed, separation distances to water table and bedrock etc. must be taken into consideration.*

Permeable Soil Depth (mm)	Slope less than 3 %	Slope of 3% or greater and less than 30%	Slope over 30%
0 to less than 300	Mound	C2(r), C3	Not Acceptable For Selection. Design Only By A Level 1 Qualified Person
300 to less than 600	Mound	C2, C2(r), C3	
600 to less than 750	Mound	C2, C2(r), C3	
750 to over 1300	Mound	C1, C1(r), C2, C2(r), C3	
780 and over	Area Bed		
875 and over	Multiple Trench		

The contour trench disposal field design consists of a single distribution trench, which is installed along a contour. A trench is excavated along the contour to a certain depth and the width that is required for the site. The bottom of the trench is kept level and imported sand media is installed in the bottom of the trench to a depth of 75 mm. Crushed rock is then installed in the trench surrounding the distribution pipe. The sand and crush rock layers are installed level along the contour, with the gravity distribution pipe sloped at 50-100 mm per 30 meters. The maximum depth for a C2, C3, or mound system from ground surface to finished surface is approximately 1600 mm (Nova Scotia Environment, 2009).

The C1 and C2 systems are gravity distribution systems. A C2 trench is used as an alternative to the C1 trench. There are two types of C2 systems, the standard C2 and the raised C2. A raised C2 system may be used when there is groundwater, rock or soil with a high permeability directly under the C2 trench. Both the C1 and the C2 systems are designed to allow the effluent to move laterally through the imported sand material below the organic surface layer (Nova Scotia Environment, 2009).

A C3 contour trench design is selected if a raised C2 contour trench cannot be installed, additional depth of imported sand is required in order to protect the groundwater, the surface slope is at least 3% and if the site is uneven due to the presence of boulder fields or wooded areas. The effluent in the C3 contour trench is designed to move vertically through the imported sand filter until it reached the natural soil under the system (Nova Scotia Environment, 2009).

A mound system is selected when the site has a slope of less than 3%, the depth of unsaturated soil is less than is required for all other on-site wastewater systems, and additional depth is required to provide protection over underlying groundwater, bedrock or soil with high permeability. The mound systems are constructed above ground and are designed to allow the effluent to move vertically through the imported sand material into the natural soil if the natural soil's permeability will allow. When the natural soil has a low permeability, the effluent will move laterally through imported sand toward natural soil with adequate permeability (Nova Scotia Environment, 2009).

Area bed and multiple trench systems are designed to distribute the effluent over a horizontal area then the effluent travels vertically throughout the subsurface soils. The sizing of these systems is determined based on loading rates. These systems are only selected when the contour trench systems are not suitable to the site where there is little slope and deep permeable soil (Nova Scotia Environment, 2009).

The lateral flow sand filter (LFSF) design, also known as a sloping sand filter, varies from the guidelines that outline the installation of the previous systems. The LFSF is currently being approved as a replacement to remediate a malfunctioning system. The LFSFs are designed using a different imported sand material than used in the other systems, as it must have a permeability of 1×10^{-4} to 5×10^{-4} m/sec. The system has a distribution trench that consists of crushed gravel where the septic effluent enters from the septic tank. The system allows the effluent to flow vertically into the imported sand material until it reaches the impermeable layer below the system and then travels laterally through the sand filter (Nova Scotia Environment, 2009; Havard et al, 2008).

2.3.3 Phosphorus Transport in On-site Wastewater Systems in Nova Scotia

In Nova Scotia, contour trenches and LFSFs have been used as OWSs for more than 30 years, but until recently very little was known about their in-field treatment performance. Research has recently been conducted by installing six field-scaled LFSFs at varying slopes and with different sand grain-size to evaluate their performance (Havard et al., 2008). Havard et al. (2008) initially studied the systems for a period of one year and determined that P removal was greatest in the fine grain sand filters, presumably due to enhanced adsorption processes, but suggested that the short term study was not sufficient to evaluate the LFSF capacity to treat P. Wilson et al. (2011) completed a longer term study to further investigate the P treatment of the same LFSFs and to confirm previous research suggesting that the removal of P decreases with time due to saturation of the disposal fields (Check et al., 1994). The study showed that there was a decrease in the treatment of P compared to the first year, showing that P treatment declines over time (Wilson et al., 2011). Continuous monitoring of the P concentrations in the LFSFs has shown a steady decline in the P removal within the sites from 2004 to 2011. After 7

years, the P removal in all of the LFSFs had diminished to 20% removal and has maintained that rate. The P retention mechanism contributing to the consistent removal rate of 20% after 7 years has not been identified. However, it could potentially be attributed to secondary P precipitation.

A typical OWS that is used in Nova Scotia is the C2 contour trench design. Research was conducted by installing two field-scale C2 contour trench systems with varying locations at which the effluent entered the distribution trench as well as different loading methods to evaluate their performance in the treatment of wastewater. The study was conducted over a period of three years and also found that the removal of P decreased in both systems over time (Bridson-Pateman et al., 2013)

2.4 Knowledge Gaps and Research Needs

The retention of P in the various study sites has suggested that disposal fields have varying capacity for retention based on the physical and chemical composition of the soil media in the disposal field (Robertson, 2012). The precipitation of P in disposal fields involves sufficient cation and orthophosphate concentrations and is controlled by pH and redox conditions. However Robertson (2003) found that non-calcareous sand provided more acidic conditions, which appears to enhance the P retention within the disposal field. Research into the processes surrounding the mechanism of P retention has not been conducted with the soil materials commonly used in OWSs designs in Nova Scotia. Therefore further investigation into the varying soil compositions of the OWSs used in Nova Scotia could provide a solution to recommend specific sand types when installing disposal fields to improve the attenuation of P within the disposal field to reduce the risk of P loading to surrounding aquatic systems.

CHAPTER 3: METHODOLOGY

3.1 Pilot Scale Studies

The research in this project was divided into two phases. The first phase consisted of utilizing an existing set of pilot scale on-site wastewater systems (OWS); the methods used during this data collection program are outlined below.

3.1.1 Site Description

The pilot scale study site was located at the Bio-Environmental Engineering Center (BEEC) in the AgriPark located in Bible Hill, Nova Scotia (45° 23' N, 63° 14' W). The BEEC site has various field scale OWSs that have been constructed according to the *Nova Scotia Environment On-site Sewage Disposal Technical Guidelines* (Nova Scotia Environment, 2009) and has been used for research purposes. The two most commonly used septic fields installed in Nova Scotia were selected for this project, the C2 raised contour trench (C2R) system and lateral flow sand filter (LFSF) system. The systems receive wastewater through a flow splitter and septic tank that is fed from the Bible Hill municipal sewer system. The layout and design of this facility has been presented previously by Wilson et al. (2011) and Bridson-Pateman et al. (2013), but is described in brief here.

The BEEC site has two C2R systems that were installed in the summer of 2007 and adhere to the specifications outlined in the *Nova Scotia Environment On-site Sewage Disposal Technical Guidelines* (Nova Scotia Environment, 2009). The two systems were installed on a 10% slope with a medium grained imported sand fill used as the filter media. Both C2R systems are a total of 10.5 m long and 8 m wide with a 5 m gravel distribution trench at the head of the disposal field (Figure 3.1). Within the gravel trench is a perforated polyvinyl chloride (PVC) pipe that is connected to the septic tanks. The C2R systems were constructed with two different loading methods for the effluent to enter the distribution trench. One of the C2R systems uses a dosing system and has wastewater entering the gravel trench in the middle of the distribution pipe. The other

C2R system is gravity fed with the wastewater entering the gravel trench and distribution pipe on the end of the distribution pipe. Both systems consist of 1.5 m of gravel trench followed laterally by 6 m of sand filter then another 0.5 m gravel trench that is used as a collection trench.

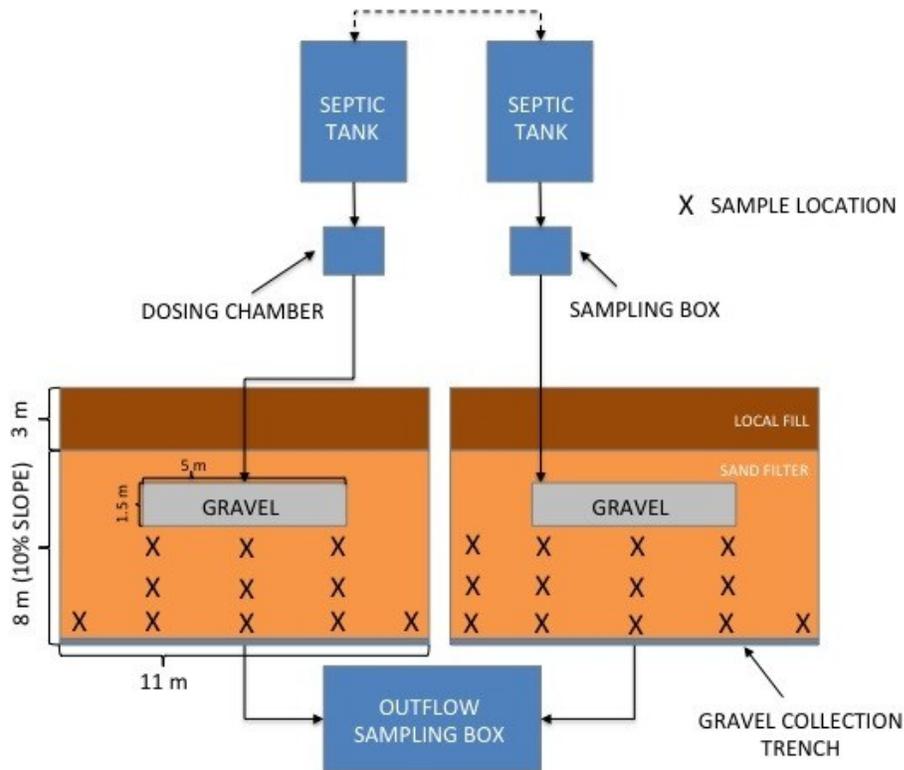


Figure 3.1 Plan view of the C2 raised contour trenches experimental facility installed in 2007 at the Bio-Environmental Engineering Center in Truro, NS. Sample locations are shown and were selected to capture wastewater flow throughout the disposal field (Adapted from Bridson-Pateman et al., 2013).

Both systems have a liner that was installed beneath the system ensuring that all subsurface flow is directed towards the collection trench at the base of each system. The two systems are hydraulically isolated from one another and in order to prevent upslope drainage from entering the systems, a diversion swale was installed. Wastewater has been loaded to each system since June 2008. The gravity fed system has been receiving approximately 300 L of wastewater per day, while the dosed system has been receiving approximately two doses of 150 L of wastewater per day.

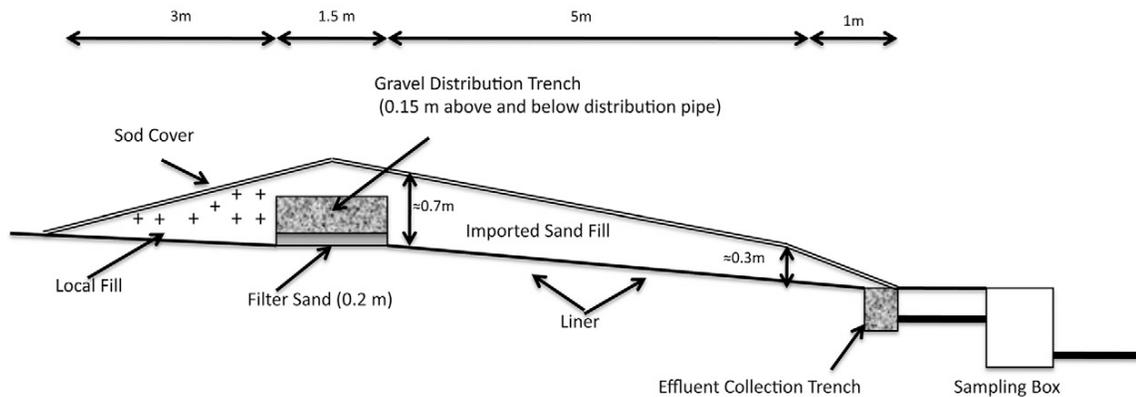


Figure 3.2 Cross-sectional schematic of the C2 raised contour trenches installed at the Bio-Environmental Engineering Center in Truro, NS (Bridson-Pateson et al., 2013).

There are also six LFSFs that were installed on the BEEC site in 2004 and were designed according to the specifications outlined in the *Nova Scotia Environment On-site Sewage Disposal Technical Guidelines* (Nova Scotia Environment, 2009). Currently the specifications regarding imported sand material does not include ideal chemical compositions, therefore the imported sand media was selected based on grain size and from various sources. Three LFSFs were installed on a 5% slope to represent the low slope limit (LFSF 1-3), one has fine grained sand (LFSF 1), one has medium grained sand (LFSF 2), and one has coarse sand (LFSF 3). Three additional LFSFs were installed on a 30% slope to represent the high slope limit (LFSF 4-6) one has fine grained sand (LFSF 4), one has medium grained sand (LFSF 5), and one has coarse sand (LFSF 6). For this study the fine grained and medium grained filters on the 5% and 30% slopes (LFSF 1, LFSF 2, LFSF 4, LFSF 5) were sampled (Figure 3.3).

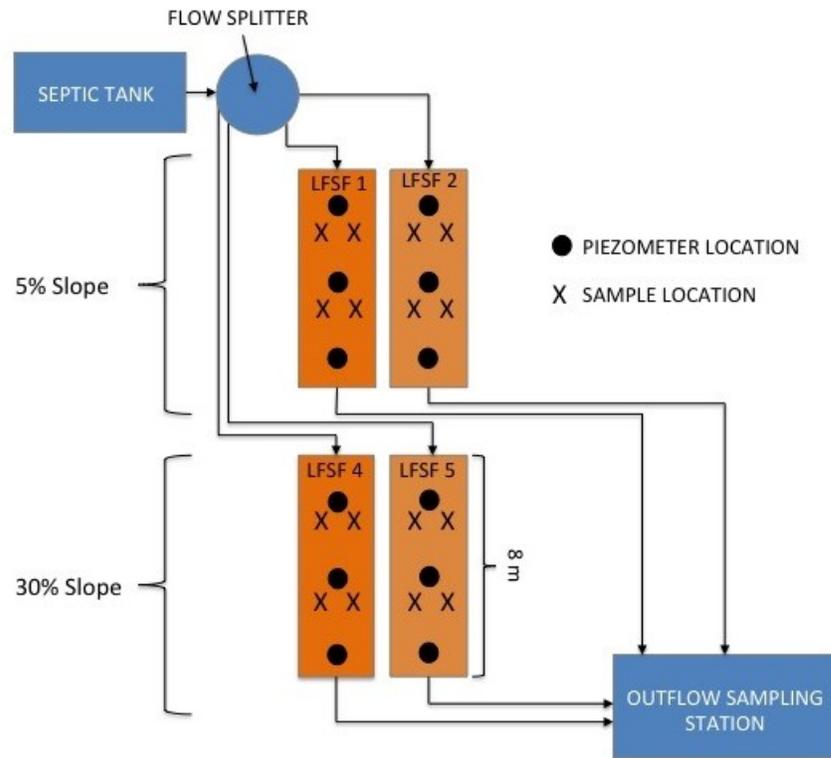


Figure 3.3 Plan view of the fine grained (LFSF 1 and 4) and medium grained (LFSF 2 and 5) lateral flow sand filter field experimental facility located at the Bio-Environmental Engineering Center in Truro, NS. Sample locations are shown and were selected to capture wastewater flow throughout the system (Adapted from Havard et al., 2008).

Each sand filter was constructed in an 8 m long by 1.5 m wide lined plywood box. At the entry point of each LFSF there is a gravel distribution trench to evenly distribute the wastewater across the system. The sand filter layer is approximately 1 m thick and tapers to 0.45 m thick at the end of the filter system. The top of the sand filter is covered with a geotextile material to separate the sand filter layer from the soil layer that covers the disposal field (Figure 3.4). Wastewater has been loaded to each sand filter since September 2004 with each filter receiving approximately 100 L of wastewater per day. In January 2007, the wastewater loading was increased to approximately 175 L of wastewater per day to assess the performance of the systems at a higher loading rate (Wilson et al., 2011). The influent and effluent of the systems were sampled on a monthly basis using ISCO 6712 auto-samplers (ISCO Inc., Lincoln, Nebraska), which

collected samples over a 24 hour period and the samples were tested for a suite of water quality parameters at the Nova Scotia Agricultural College Environmental Research Laboratory.

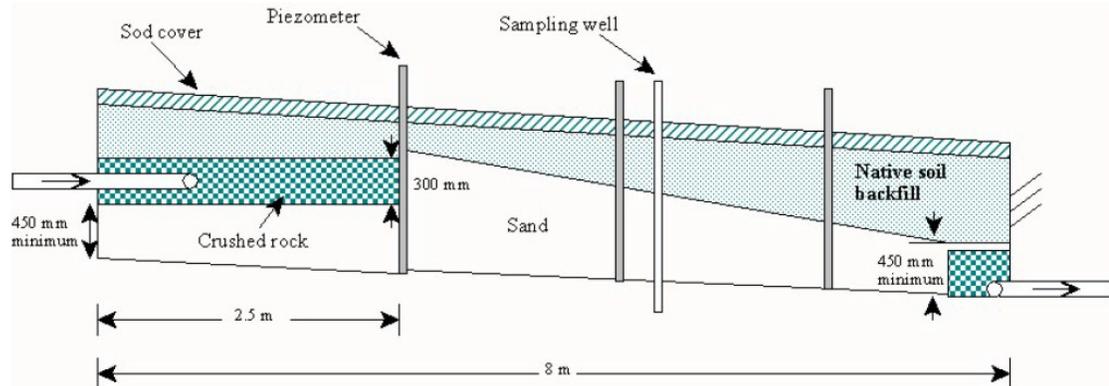


Figure 3.4 Cross-sectional schematic of the lateral flow sand filters installed at the Bio-Environmental Engineering Center in Truro, NS (Havard et al., 2008).

3.1.2 Soil Sampling Methods

Soil samples were collected from the selected OWSs in July 2012. A sampling grid was designed for each OWS and a dutch auger was used to collect the soil samples at varying depths throughout each system. The soil samples were collected in a sampling box provided by the Nova Scotia Department of Agriculture’s Analytical Services Laboratory along with a plastic sampling bag for additional analysis that were conducted at Dalhousie University. The samples were stored at 4°C until analysis could be completed.

3.1.2.1 C2 Raised Contour Trench Systems

A sampling grid was designed for each disposal field based on the specifications of the OWS in order to capture the distribution of wastewater throughout the disposal field. The disposal fields for the C2R systems were approximately 6 m long and 8 m wide. The sample locations were spaced every 2.5 m for a total of 12 sampling locations (Figure 3.1). Two to three samples were collected at each location to assess the vertical distribution of soil chemistry within the disposal field. The C2R systems were installed

with a liner, which aided in identifying the bottom of the system when sampling. Approximately 60 soil samples were collected in total from the two C2R systems along with 6 duplicate samples.

3.1.2.2 Lateral Flow Sand Filters

A sampling grid of 0.5 m across and 2.5 m in length was implemented on each LFSF for a total of 4 sampling locations per filter (Figure 3.3). Two samples were taken at each location, one capturing the top of the sand filter and one capturing the middle to the base of the filter. Since the lateral sand filters are constructed with plywood, it was evident that the base of the system was reached when augur reached the plywood base. Approximately 30 samples were taken in total from the four LFSFs along with 3 duplicate samples.

3.2 Field Scale

The second phase of the data collection program consisted of selecting existing full scale OWS that had been operational for over 10 years and were located within the Thomas Brook Watershed; the methods used during this data collection program are outlined below.

3.2.1 Site Description

The Thomas Brook Watershed is located north of Berwick within Kings County, Nova Scotia (45° 2' 51.17" N, 64° 44' 9.42" W). The Thomas Brook Watershed is approximately 750 ha and is a small upper subcatchment of the Cornwallis River Watershed, which is one of the largest watersheds within the Annapolis Valley with an approximate area of 26,000 ha (Figure 3.5). The Thomas Brook headwaters originate on the North Mountain with the outlet discharging into the Cornwallis River near the town of Berwick. The stream in the watershed has been identified as having two upper branches draining from the North Mountain into a single channel covering one third of the distance of the watershed. The stream is approximately 5.8 km and rarely exceeds a width greater than 2 m. The average slope of the stream network is 3.5% with steeper slopes reaching up to 9% in the upper section on the North Mountain and the lower

sections of the stream network having a slope of 0.5-3%. The general soil type within the watershed has been characterized as reddish brown sandy loam (Jamieson et al, 2003).

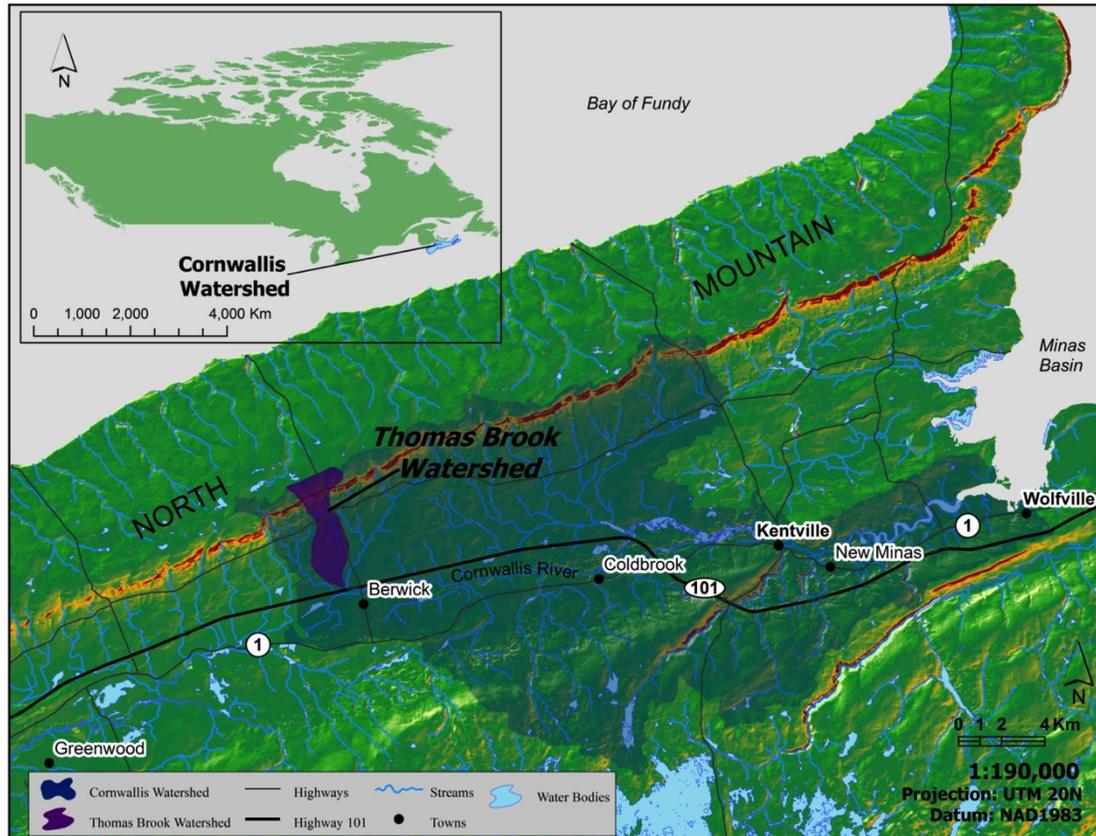


Figure 3.5 Watershed location map for Thomas Brook Watershed within the Cornwallis River Watershed located in Nova Scotia, Canada (Sinclair et al., 2014).

The Thomas Brook Watershed consists of mixed land-use comprised of agricultural, residential, and forested area. Agricultural is the dominant land-use within the watershed and consists of approximately 60% of the total watershed area. The main crops grown within the watershed are strawberries, corn and grain. The residential area consists of approximately 89 residents while the forested area comprises approximately 33% of the total area and is predominately located in the upper portions of the watershed. The Thomas Brook Watershed has been under continuous monitoring since 2004 as apart of the Agriculture and Agrifood Canada’s watershed evaluation of beneficial management practices (WEB’s) program. The results of the monitoring program showed elevated concentrations of nutrients within the watershed (Jamieson et al., 2003) and

OWSs have been shown to be a potentially significant source of P to the stream (Sinclair et al., 2014)

For this phase of the study, three OWS within the Thomas Brook Watershed that had been operational for over 10 years were selected for monitoring. The three systems that were selected vary in age and type of system and are identified as Site 1, Site 2, and Site 3.

3.2.1.1 Site 1 Description

The Site 1 OWS was constructed in 2002 to service a permanent two-person residence. The system is a raised C2 consisting of a septic tank and a gravity-fed filter bed with a distribution trench and is 39 m in length and 1.2 m in width. The disposal field consists of imported sand fill and is situated on dense clayey subsoil material. The general slope of the system is 15%.

3.2.1.2 Site 2 Description

The Site 2 OWS was constructed in the early 1990's to service a single-family residence; two people currently live at the residence. The system is a raised C2 consisting of a septic tank and a gravity-fed filter bed with a distribution trench and is approximately 30 m in length and approximately 1 m in width. The disposal field consists of imported sand fill and is situated on dense clayey subsoil material. The general slope of the system is 10%.

3.2.1.3 Site 3 Description

The Site 3 OWS was constructed in the late 1990's to service a single-family residence, which is currently the permanent residence of a two-person household. The system is a raised C2 consisting of a septic tank, a dosing chamber and a filter bed with a distribution trench and is approximately 40 m in length and approximately 1 m in width. The disposal field consists of imported sand fill and is situated on dense clayey subsoil material. The general slope of the system is 24%.

It was assumed that the water consumption and wastewater generation for all three sites are typical and approximately 292 L/person/day (Environment Canada, 2011).

3.2.2 Soil Sampling Methods

Soil samples were collected from the three OWS in August 2012 using a regular grid-based sampling strategy (Figure 3.6). A hand ditch auger was used to collect the soil samples at varying depths throughout the disposal field. Approximately two to three samples were taken at each sample location to determine the vertical distribution of the wastewater within the disposal field. All three disposal fields were situated on dense clayey subsoils, which aided in identifying the bottom of the system when sampling.

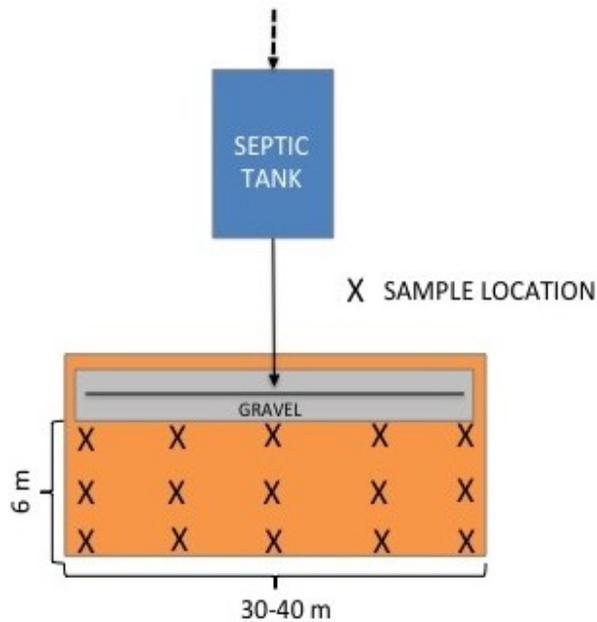


Figure 3.6 Layout of sampling grid used when sampling the three on-site wastewater systems located within the Thomas Brook Watershed.

The OWS located at Site 1 is a raised C2 with a disposal field that is 39 m in length and 6 m in width. A sampling grid of 10 m by 3 m was implemented for a total of 15 sample locations. Twenty-seven soil samples were collected along with 3 duplicate samples.

The OWS located at Site 2 is a raised C2 with a disposal field that is 30 m in length and approximately 5.5 m in width. The sample locations were spaced every 10 m by 2.5 m for a total of 12 sample locations. Twenty-five soil samples were collected along with 3 duplicate samples.

The OWS located at Site 3 is a raised C2 with a disposal field that is 40 m in length and approximately 5.7 m in width. The sample locations were spaced every 10 m by 2.7-3 m for a total of 15 sample locations. Thirty-four soil samples were collected along with 3 duplicate samples.

The soil samples were collected in a sampling box provided by the Nova Scotia Department of Agriculture's Analytical Services Laboratory along with a plastic sampling bag for additional analysis that were conducted at Dalhousie University. The samples were stored at 4°C until analysis could be completed.

3.2.3 Wastewater Effluent Sampling Methods

Six monitoring wells were installed throughout the disposal field at each site in the Thomas Brook Watershed at the time of soil sampling. The monitoring wells were installed to help characterize effluent flow throughout the disposal field. Water levels were measured in the wells and samples were collected to assess seasonal variations of in effluent water quality. Boreholes were constructed using a hand auger during soil sample collection, allowing the monitoring wells to be installed. The selection of boreholes as monitoring wells was determined based on: (i) the presence of water in the borehole, (ii) proximity to the distribution trench, and (iii) the presence of lush vegetation indicating the presence of effluent flow in that area of the disposal field. The well depths were based on the depth of the disposal field, which was indicated by the presence of dense subsoil. The depths of the wells ranged from 45 cm to 105 cm. The wells were constructed using 2.5 cm PVC casing and screen sections. The bottom of the monitoring well was fitted with a pointed PVC end cap that securely attached to the bottom of the screen section. The length of the screen section was selected to assure that the water level would fall within that section of the well. The solid PVC casing was threaded into the screen section and a minimum of 15 cm of casing was used to prevent surface water flow from entering

the well. The well was secured using a watertight locking nylon expansion plug. The remaining borehole was backfilled with silica sand and a bentonite clay was used as a seal around the solid casing. The monitoring well was protected with a PVC well cover.

Samples were collected on a monthly basis from all three sites from September 2012 until October 2013, with the exception of August 2013 because of lack of water in the wells. Sampling consisted of measuring the static water levels in all 18 monitoring wells, purging the wells of three to five well volumes and allowing the wells to recharge prior to collecting the samples. Bailers were used to purge and sample the wells and were rinsed with distilled water following sampling (Figure 3.7). Water samples were stored at 4°C during transport to the laboratory at Dalhousie University.



Figure 3.7 Water samples were collected using bailers from each monitoring well installed in the three on-site wastewater systems located within the Thomas Brook Watershed.

The monitoring wells were decommissioned following the completion of the groundwater-sampling program in October 2013. The monitoring wells were removed from the disposal field and the boreholes were backfilled with silica sand and covered with topsoil.

3.3 Soil Physical Characterization

3.3.1 Sand Media Classification

Textural classification was completed for each of the three filter sands from the pilot scale septic systems at the BEEC location in addition to the three field scale disposal fields located in the Thomas Brook Watershed. A grain size analysis was used to classify each sand and was conducted in accordance to the ASTM Standard Test Method D422 – 63 (ASTM Committee D-18 on Soil and Rock et al., 2007). A composite sample from soil throughout the disposal field was collected at the time of sampling to represent the entire disposal field and stored in a plastic sampling bag pending analysis. Each sample was oven dried at 105°C and the dry weight of the sample was recorded. Approximately 1000g of sample was shaken through a series of sieves which included 1 cm, 5 mm, 2.5 mm, 1.25 mm, 0.63 mm, 0.315 mm, 0.160 mm, 0.075 mm and a bottom collection pan. The sample was shaken for 20 minutes to separate the particle sizes and the mass retained in each sieve was weighed and recorded. The sample was also washed over a 0.075 mm sieve multiple times to ensure the finer particles were separated. The sample was dried and the weighed again after the removal of the finer particles. The distribution of the grain size for each sample was plotted and the percent finer (%) for each sieve size was calculated. The grain size distribution for each sample was then used to classify the soil type in accordance to the Unified Soil Classification System (USCS) ASTM Standard Practice D-2487-11 (ASTM Committee D18 on Soil and Rock et al., 2011).

At the time of sampling, additional soil samples were collected in plastic sample bags and transported to Dalhousie University and stored at 4°C pending analysis. Soil moisture content for each soil sample was calculated following the ASTM Standard Practice D-2216-10 (ASTM Committee D18 on Soil and Rock et al., 2010). This method involves removing the moisture from the soil by oven drying approximately 10 g of the soil sample at 105°C until the weight of the sample remains constant. The moisture content of the sample is calculated using the weight before and after the drying process.

3.3.2 Hydraulic Conductivity

The hydraulic conductivity (k) was determined in the laboratory in accordance with ASTM Method D2434 – 68 (ASTM Committee D18 on Soil and Rock et al., 2006). This procedure determines k using constant head conditions producing laminar flow through saturated granular soil. The method assumes that Darcy's Law is valid and tests were repeated at different heads to increase the accuracy of the analysis. The test was performed a total of three times for each soil sample to verify the results, and produce a precision estimate. The analysis was completed on a composite samples from each disposal field. The k was calculated using the following equation:

$$k = \frac{QL}{Ath} \quad (3.1)$$

Where:

k= hydraulic conductivity/ coefficient of permeability

Q= quantity of water discharged

L= distance between manometers

A= cross-sectional area of specimen

t = total time of discharge

h = difference in head of manometers

3.3.3 Hydraulic Gradient

The hydraulic gradient (i) within the disposal field was measured at each of the three sites in the Thomas Brook Watershed. Hydraulic head measurement, based on the relative elevation of the well in relation to the water table in each monitoring well, was determined on a monthly basis from September 2012 until October 2013. The hydraulic head in each well was then used to estimate the monthly hydraulic gradient at each sampling site. The average hydraulic gradient value was then calculated for each site. The following equation was used when calculating the hydraulic gradient:

$$i = \frac{dh}{dl} \quad (3.2)$$

Where:

dh = the difference in hydraulic head between two monitoring wells (m).

dl = the horizontal distance between two monitoring wells (m).

3.4 Soil Analysis

3.4.1 Soil Chemistry Parameters

All soil samples were analyzed by the Nova Scotia Department of Agriculture Analytical Services Laboratory for a suite of parameters including: pH, organic matter, phosphorus, potassium, calcium, magnesium, sodium, aluminum, iron, manganese, sulfur, copper, zinc, boron, and cation exchange content. The laboratory follows the standard operating procedures for Mehlich 3 extraction (Mehlich, 1984) of soil samples followed by analysis using an Inductively Coupled Plasma Optical-Emission Spectrometry (ICP-OES) with a radial view. The Mehlich 3 method was developed in 1984 and is widely used across North America as a multi element soil extraction procedure. This procedure provides accurate determination of elements such as phosphorus, potassium, iron, manganese, copper, zinc, calcium, sodium, magnesium, boron and sulphur. It is effective when predicting phosphorus in neutral and alkaline soils than previous methods and the extraction solution contains acetic acid, which allows for the release of available phosphorus in most soils (Kovar and Pierzynski (eds), 2009). The automated AS3010D pH analyzer was used to measure pH and organic matter in the soil was determined by loss on ignition.

3.4.2 Total Phosphorus Extraction Method

Subsamples from all soil samples were digested in acid using the Aqua Regia method (ISO 11466, 1995) in order to extract the total phosphorus from each sample (Figure 3.8). Acid digestion is one of the most common and recognizable methods of extracting phosphorus from soil. The Aqua Regia technique liberates phosphorus associated with hydroxide, carbonate and sulfide mineral and was also utilized in previous research conducted by Robertson (2008).

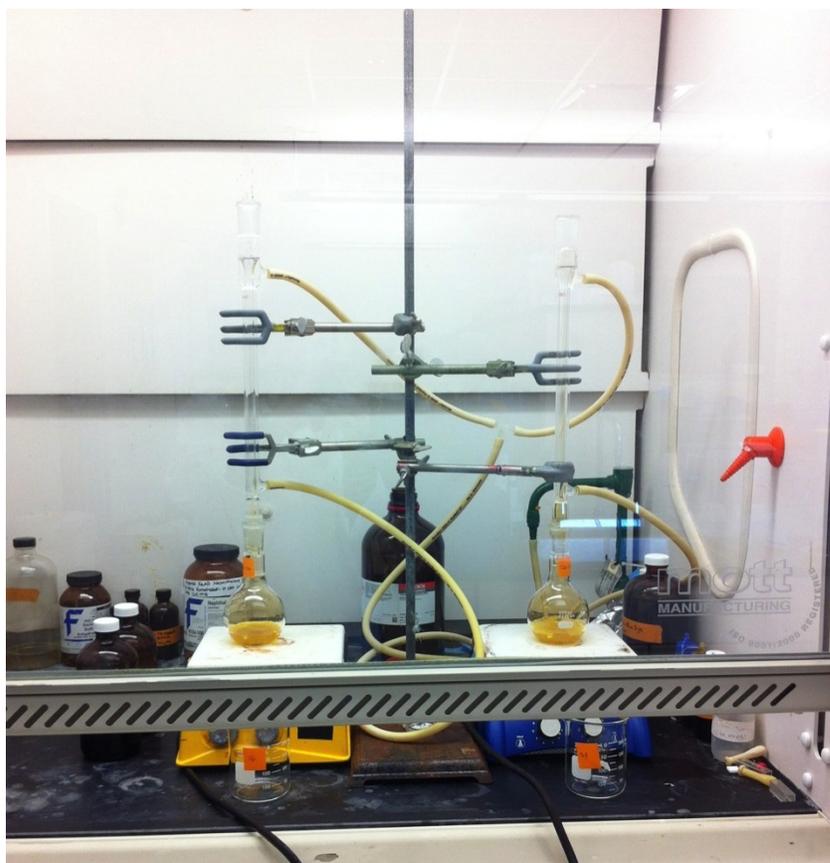


Figure 3.8 Soil samples being digested using the aqua regia method to extract the total phosphorus from the sample.

A reflux condenser attached to a reaction vessel was placed on a heat source. Soil weighing approximately 3 g, to the nearest 0.001 g, was transferred into the reaction vessel. The soil was moistened with 1 ml of distilled water. While swirling the reaction vessel, hydrochloric acid, followed by nitric acid, was slowly added to reduce frothing. Boiling aids that were pre-washed in nitric acid were transferred into the reaction vessel. The reaction vessel was then connected back to the reflux condenser. The mixture remained at room temperature until frothing ceased to allow for slow oxidation of the organic mass of the sample. Cold water was used as a coolant passing in between the inner glass tube and the outer glass tube to cool hot vapors as they entered the condenser. The temperature was slowly raised on the heating device until the mixture was boiling and the condensation zone was lower than 1/3 of the height of the reflux condenser and was maintained for two hours. Once heated for two hours, the mixture was removed from

the heat source and allowed to cool. The reflux condenser was rinsed using distilled water and the mixture was filtered using 0.45 µm filter paper. The extraction solutions were stored at 4°C pending analysis and diluted using distilled water prior to being analyzed for total phosphorus. Approximately 10% of the total number of samples were randomly selected for duplication.

3.4.3 Electron Microprobe

To obtain a more detailed characterization of the behavior of phosphorus in the disposal fields, soil samples were selected from both the fine grained and medium grained LFSFs located at the BEEC. The samples were selected from the LFSFs based on previous studies (Sinclair et al., 2014) which characterized the flow of wastewater in these systems to capture both high flow areas and low flow areas for comparison. Samples were prepared in an epoxy matrix and were examined prior to analyses using the Nikon Eclipse 50i POL (Nikon Instruments Inc., Melville, NY 11747-3064 USA) polarizing microscope to identify areas of interest. The analysis was conducted using a Jeol JXA-8200 Superprobe located in the Earth Science Department at Dalhousie University. Back Scattered Electron images were generated to locate the phosphorus-rich grains and grain coatings. The electron microprobe uses a 15 kV, 20 nA beam to conduct quantitative and semi-quantitative Electron Dispersive Spectroscopy (EDS) to determine the elemental concentration of the P-rich solids. In some cases the target areas were smaller than the diameter of the electron microprobe beam therefore the analyses may include minerals surrounding the target area resulting in more qualitative than quantitative elemental concentrations. The results are expressed as percentage by weight therefore they may overestimate of the concentration of elements that are present in the target area. However, the proportions of the elemental results are considered accurate due to the consistency between multiple analyses of the same sample.

3.5 Water Quality Analysis

Monthly water samples were collected from all three sites in the Thomas Brook Watershed from September 2012 until October 2013, with the exception of August 2013 due to lack of water in the monitoring wells. Samples were analyzed in the laboratory at

Dalhousie University for the following parameters: Total Organic Carbon (TOC), Total Phosphorus (TP), Total Nitrogen (TN), oxidation reduction potential (ORP) and pH. All samples were collected in sterile bottles that were transported from site to Dalhousie University in a refrigerated cooler. Samples were either stored at 4°C pending analysis or preserved immediately upon return to the laboratory until analysis was to be completed. All laboratory results were completed at Dalhousie University.

3.5.1 Total Organic Carbon Analysis

Samples analyzed for TOC were collected in 250 mL bottles and stored at 4°C for a maximum of 24 hours. Samples were then decanted into 40 mL vials and the pH was lowered to below 2 using phosphoric acid (H_3PO_4) for preservation. The acidified samples were stored at 4°C pending analysis. The samples were analyzed using a Shimadzu ASI-V (Kyoto, Japan) with a TOC-V analyzer. All total organic carbon results are reported in mg/L.

3.5.2 Total Phosphorus Analysis

Water samples analyzed for TP were collected in 250 mL bottles and immediately frozen upon return to the laboratory until analysis. Samples were thawed using a water bath prior to analysis. Soil extraction solutions were stored at 4°C and diluted using distilled water prior to analysis. Samples were digested using potassium persulfate ($K_2S_2O_8$) to account for the high turbidity of the samples and the possibility of adsorption of phosphorus to the organic matter present in the water sample. The TP analysis was completed using a LKB Biochrom Ultrospec 4051 spectrophotometer (Cambridge, UK) using the Ascorbic Acid Method 4500-P E (Clesceri et al., 1998). All total phosphorus results are reported in mg/L.

3.5.3 Total Nitrogen Analysis

Samples analyzed for TN were collected in 250 mL bottles and immediately frozen upon returning to the laboratory until analysis. Samples were thawed using a water bath prior to preparation for analysis. Samples were digested using potassium persulfate ($K_2S_2O_8$) and decanted into 40 mL vials. The pH of the samples were reduced to below 2

using phosphoric acid (H_3PO_4) for preservation and the acidified samples were stored at 4°C pending analysis. The samples were analyzed using a Shimadzu ASI-V (Kyoto, Japan) with aTNM-1 analyzer. All TN results are reported in mg/L.

3.5.4 Oxidation Reduction Potential

Effluent ORP was analyzed *in-situ* using the Thermo Scientific Orion Star A121 Portable ORP Meter (Thermo Fisher Scientific, Beverly, MA USA, 01915). The meter was calibrated in conjunction with the Thermo Scientific Orion Star Manual. The water was analyzed according to the method specified in the Thermo Scientific User Guide for the meter.

3.5.5 pH

Effluent pH was analyzed *in-situ* using the Denver Instruments UltraBasic Portable pH and Temperature Meter (Denver Instrument Company, Arvada, Colorado USA, 80004). The meter was calibrated prior to use and the wastewater was analyzed for pH and temperature in accordance with the Denver Instrument UltraBasic Operation Manual.

CHAPTER 4: RESULTS AND DISCUSSION

4.1 Physical and Chemical Characteristics of the Treatment Media

4.1.1 Lateral Flow Sand Filters

In this study, soil samples were collected from four LFSFs that were installed during the summer of 2004. The samples were collected in 2012, after the systems had been receiving wastewater for approximately eight years. The physical characteristics of the sand used are summarized in Table 4.1. Two filters contained fine grained sand (LFSF1 and LFSF4) and two filters contained medium grained sand (LFSF2 and LFSF5), as defined by Havard et al. (2008). Those characteristics were compared to the limits outlined in the Nova Scotia On-Site Technical Guidelines and both sands fall within the acceptable permeability and soil type criteria.

Table 4.1 Physical characteristics of lateral flow sand filters

Sand Type	d_{10} (mm)	Uniformity coefficient (d_{60}/d_{10})	Hydraulic Conductivity (m/s)
Fine	0.15	8	1.5×10^{-4}
Medium	0.17	5.6	5.0×10^{-4}

(Adapted from Havard et al., 2008)

Soil samples collected from the LFSFs were analyzed for various parameters to provide information about the long-term treatment performance; key parameters are summarized in Table 4.2.

Table 4.2 Statistical summary of soil chemistry for lateral flow sand filters. There were eight soil samples along with one duplicate sample taken from each filter.

Parameter		LFSF1	LFSF2	LFSF4	LFSF5
		Fine (5% Slope)	Medium (5% Slope)	Fine (30% Slope)	Medium (30% Slope)
pH	Mean	6.0	5.6	5.5	5.7
	Maximum	6.5	6.1	5.8	6.1
	Minimum	5.3	5.1	5.3	5.4
Soil Moisture (%)	Mean	6.4	4.4	4.1	4.6
	Maximum	11.5	6.7	4.9	6.0
	Minimum	3.0	2.7	3.3	3.8
Organic Matter (%)	Mean	0.9	0.8	0.9	0.9
	Maximum	1.0	1.0	1.0	1.2
	Minimum	0.9	0.7	0.9	0.8
AP (ppm)	Mean	100.8	59.3	51.5	69.8
	Maximum	164.7	106.6	79.8	120.2
	Minimum	23.2	31.2	25.2	35.5
TP (ppm)	Mean	114.3	535.2	84.2	545.7
	Maximum	173.8	680.0	105.6	643.5
	Minimum	64.6	460.1	70.4	476.6
Ca (ppm)	Mean	173.7	180.5	86.8	201.3
	Maximum	268.5	219.0	113.5	253.0
	Minimum	56	139.0	61.5	145.5
Mg (ppm)	Mean	15.2	24.3	16.2	29.4
	Maximum	20	39.0	26.0	44.0
	Minimum	12.5	20.5	10.0	23.5
Al (ppm)	Mean	546.2	756.6	601.2	788.5
	Maximum	635.6	997.5	674.9	972.2
	Minimum	450.5	613.4	517.4	672.0
Fe (ppm)	Mean	163.4	156.9	81.0	145.4
	Maximum	287	263.0	106.0	175.0
	Minimum	62	113.0	61.0	132.0

LFSF1 and LFSF4 consist of fine grained sand. The average values of organic matter (0.9%) are consistent throughout the systems with little variation. Maximum organic matter levels (1%) were observed at greater depths in LFSF4. Elevated loosely bound or AP concentrations were observed in LFSF1 in the soil near the infiltration pipe (102 ppm – 162 ppm), up to four times higher than the minimum which is assumed to be comparable to the background concentration (25 ppm). This zone of enriched AP values was not observed in LFSF4, with concentrations ranging from 58 ppm to 79 ppm near the infiltration pipe. The overall pH levels of the systems range from 5.3 to 6.5, suggesting moderately to slightly acidic conditions with lower pH values observed at greater depths. Elevated TP concentrations (173 ppm) were observed in LFSF1, which are up to two

times greater than the background levels (64 ppm). The zone of enriched TP levels was observed in the soil directly below the infiltration pipe in both LFSF1 and LFSF4, with higher concentrations in LFSF1. The Al and Fe concentrations range from 450 ppm to 674 ppm and 61 ppm to 287 ppm, respectively. With such a small variation in concentration it is difficult to establish if elevated Al and Fe are occurring in the same zone as elevated P concentrations. The distribution of Fe in LFSF1 suggests that the highest concentrations of Fe are occurring near the infiltration pipe with similar high values occurring at greater depths. This increase in Fe corresponds to an enriched zone of elevated AP and TP. The overall concentrations of TP, Al, and Fe in both systems were not as high as those reported in other studies (Zanini et al.,1998; Robertson, 2012) suggesting the chemistry of the soil may not be conducive to the retention of phosphorus.

To assess the ability of the LFSFs to utilize the entire sand media to treat the influent, contour maps for the systems illustrating the distribution of TP throughout the disposal field are shown in Figures 4.1 and Figure 4.2. The contour maps give a general overview of the distributions of TP in each disposal field using the sample locations collected in the summer of 2012. These contour maps have limitations, as the concentrations between sampling locations is not known. Similar results were found in the same type of sand media, therefore a representation of each sand media was selected.

The TP results for the samples collected within the LFSF1 disposal field are shown in Figure 4.1 to illustrate the distribution of TP within fine grained LFSF systems. The wastewater enters the system through an infiltration pipe at the entry point in the center of the disposal field and the system has enriched P concentrations near the infiltration pipe. Though the P concentrations seem evenly distributed throughout the top portion of the system near the infiltration pipe, the P concentration is not evenly distributed through the entire system and the bottom portion has low P concentrations indicating that the system is not utilizing the entire sand media for the removal of P.

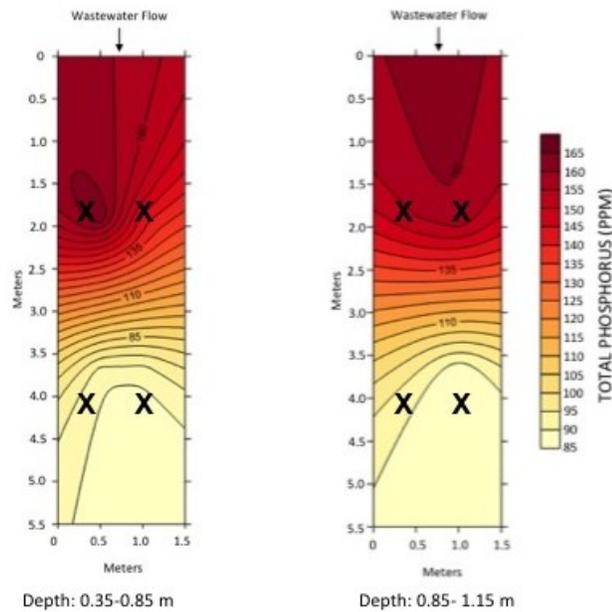


Figure 4.1 Planar view of the disposal field depicting the distribution of total phosphorus for throughout the LFSF1 containing fine grained sand. Sample locations that were used in creating the contour maps are indicated using an “X”.

LFSF2 and LFSF5 consist of medium grained sand with organic matter in the systems ranging from 0.7% - 1.2%. There were no identifiable areas of elevated values of organic matter in either system. Zones of enriched AP (106 ppm in LFSF2; 120 ppm in LFSF5) were observed in both systems in the soil below the infiltration pipe at the same depth (0.88 m - 1.16 m). Concentrations decreased in the soil with increasing distance from the infiltration pipe. The highest AP concentrations occurred in LFSF5 (120 ppm), up to three times higher than the minimum levels (35 ppm). The pH levels in both systems ranged from 5.1 to 6.1 indicating strong to slightly acidic conditions with higher pH levels (6.1) occurring below the infiltration pipe at greater depth. In LFSF2 the TP concentrations ranged from 460 ppm to 680 ppm with an enriched zone occurring at the infiltration pipe at a depth of 0.46 m - 0.82 m and continuing further away from the infiltration pipe at a depth of 0.90 m - 1.17 m. The highest concentration was observed along the bottom left side of the system showing the wastewater is preferentially flowing to that side of the system. The TP concentrations in LFSF5 ranged from 476 ppm to 643 ppm with the highest concentration located below the infiltration pipe at a depth of 0.88 m - 1.50 m. Similar TP levels (616 ppm) were also observed in the overlying soil

layer (0.64 m – 1.05 m). The Al and Fe concentrations range from 613 ppm to 997 ppm and 113 ppm to 263 ppm, respectively with marginal variation indicating there is no defined zone of enrichment in either system. Based on visual association observation there does not seem to be a relationship between Al or Fe levels in the area of elevated AP or TP concentrations.

The AP concentrations are comparable in values between the fine grain and medium grain LFSF systems. However, the medium grained systems (LFSF2 and LFSF5) have up to four times the levels of TP. These results suggest that the medium grained sand filters are able to retain more TP under similar flow conditions compared to the fine grained sand filters, and that secondary mineralization of P could be occurring in these systems. The fine grained and medium grained sand media consisted of different soil chemistry (Table 4.2), which could be the primary factor in the differing ability of the sand media to retain P. This is further suggested when comparing the retention ability of the LFSFs and previous studies (Zanini et al., 1998; Robertson, 2012) whose disposal field media has a much greater ability to retain P possibly due to the chemistry of native soils that were used in the disposal field. Zanini (1998) reported acid extractable P, Fe, and Al results for all 4 study sites; overall the results for Fe ranged from 1410 –10,740 µg/g, P results ranged from 239 –2186 µg/g, and Al results ranged from 645 –10,900 µg/g.

The distribution of TP concentrations for the samples collected within the LFSF2 disposal field are shown in Figure 4.2. The wastewater enters the system through an infiltration pipe at the entry point in the center of the disposal field. Similar to the fine grained LFSFs, this system has an area of enriched P concentrations near the infiltration pipe but only for the shallower depth (0.35 m-0.85 m). The samples collected below the infiltration pipe at a depth of 0.85 m – 1.15 m does not have elevated P as seen in LFSF1. The P is remaining in the shallow soils near the infiltration pipe then migrating to the deeper soils on the left side on the system. This indicates that there is preferential flow of the wastewater in the system and it is not functioning to its full potential as there is a large portion of sand media that is not being utilized for the removal of P.

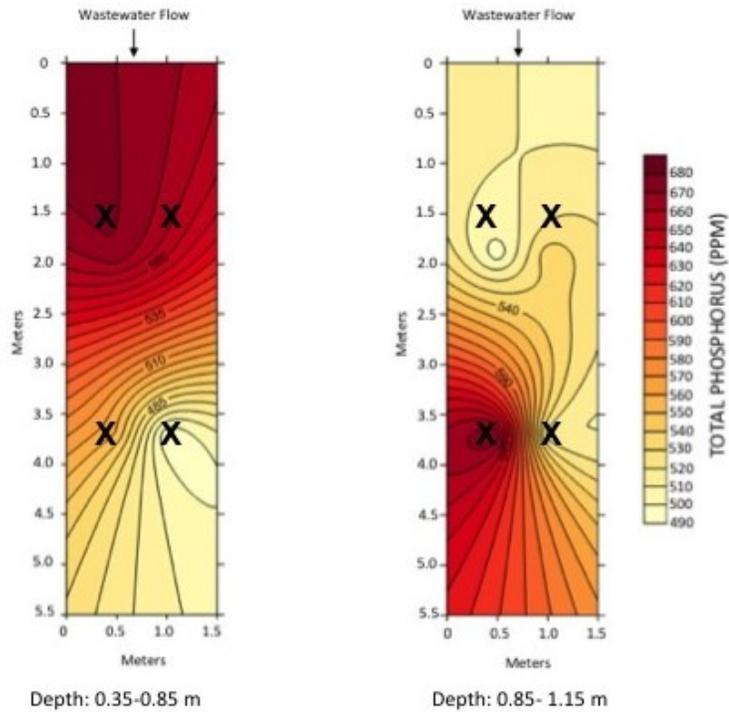


Figure 4.2 Planar view of the disposal field depicting the distribution of total phosphorus for throughout the LFSF2 containing medium grained sand. Sample locations that were used in creating the contour maps are indicated using an “X”.

4.1.2 C2 Raised Contour Trenches

Soil samples were collected from the two C2R systems that were installed in 2007. Both systems were installed with medium grained sand. The physical characteristics of the sand are summarized in Table 4. 3. When compared to the standards outlined in the Nova Scotia On-Site Technical Guidelines the sand is within the acceptable permeability and soil type limits.

Table 4.3 Physical characteristics of C2 raised contour trenches

Sand Type	d ₁₀ (mm)	Uniformity coefficient (d ₆₀ /d ₁₀)	Hydraulic Conductivity (m/s)
Medium	0.27	5.2	4.22 x 10 ⁻⁴

Key parameters are summarized in Table 4.4. The organic matter values are similar in both C2R systems with an overall level of 1.1%. The pH of the both systems range from 7.3 to 8.3 indicating the soil in the systems are slightly to moderately alkaline.

The pH of the soil is variable throughout the disposal field with no indication of changes in pH values near the distribution trench or at greater depths. The AP concentrations varied between the gravity fed and dosed systems. The gravity fed C2R system had an elevated AP concentration of 237 ppm, up to forty-seven times the minimum concentration in the system. The minimum concentration of the system was assumed to be comparable to the background level of the sand media. The zone of enriched AP was observed at a depth of 0.35 m – 0.54 m and extends for the entire 8 m along the distribution trench. In the soil samples 2 m downgradient from the distribution trench, the enriched zone was only observed at a depth of 0.30 m – 0.43 m in the sample collected directly down gradient from where the infiltration pipe enters the distribution trench. The dosed C2R system had an elevated AP concentration of 117 ppm, up to eleven times the minimum values. The zone of enriched AP was observed in the soil near the right side of the distribution trench at a depth of 0.37 m – 0.48 m. The AP concentrations were significantly reduced throughout the remaining disposal field.

Both systems had comparable concentrations of TP, which ranged from 91 ppm – 355 ppm. The dosed system had two areas where enriched TP was observed, along the left side of the distribution trench at a depth of 0.16 m – 0.47 m and along the right side of the system extending from the distribution trench down gradient 4 m to the edge of the disposal field. There was no enriched TP zone observed in the center of the field. The area of enriched TP in the gravity fed system corresponds to the zone of enriched AP extending for the entire 8 m of the distribution trench at a depth of 0.35 m – 0.54 m and in the soil 2 m down gradient from where the infiltration pipe enters the distribution trench at a depth of 0.30 m – 0.43 m. The Ca concentrations in both systems range from 1472 ppm – 2520 ppm, with higher concentrations occurring at greater depths. The entire disposal field contains elevated levels of Ca with no indication that the Ca levels correspond to the enriched available and TP zones in the disposal field. The elevated Ca levels are up to nine times higher than the maximum levels occurring in the LFSFs. These elevated levels contribute to the alkaline conditions of the soil. The Al concentrations ranged from 6 ppm – 759 ppm; the distribution of the Al indicates that the elevated Al concentrations are present at shallower depths (0.16 m – 0.27 m) and decrease with depth. The range in Fe levels in the gravity fed system are slightly higher than in the

dosed system, with the higher concentrations in the gravity system corresponding to the zone of enriched TP and available P.

Table 4.4 Statistical summary of soil chemistry for C2 raised contour trench systems. There were twenty-seven soil samples collected from C2R dosed and thirty-one soil samples collected from C2R gravity along with three duplicate samples taken from each system.

Parameter		C2R Gravity (10% Slope)	C2R Dosed (10% Slope)
pH	Mean	7.9	7.8
	Maximum	8.3	8.2
	Minimum	7.3	7.3
Soil Moisture (%)	Mean	8.9	7.6
	Maximum	25.0	11.9
	Minimum	3.7	3.6
Organic Matter (%)	Mean	1.1	1.1
	Maximum	1.4	1.7
	Minimum	0.9	0.9
AP (ppm)	Mean	63.4	33.8
	Maximum	237.1	117.6
	Minimum	5.2	10.5
TP (ppm)	Mean	145.3	138.3
	Maximum	320.1	355.9
	Minimum	91.8	97.1
Ca (ppm)	Mean	1987.1	2004.5
	Maximum	2384.5	2520.5
	Minimum	1577	1472.5
Mg (ppm)	Mean	38	33.0
	Maximum	73	49.5
	Minimum	29.5	28.5
Al (ppm)	Mean	214.9	227.9
	Maximum	687.8	759.7
	Minimum	6.5	16.5
Fe (ppm)	Mean	277.6	158.8
	Maximum	834	259.0
	Minimum	144	130.0

The overall concentrations of TP were lower in the C2R systems compared to the levels observed in the LFSF that contain similar sized medium grain sand. Though they have similar grain size, the varying geochemistry in LFSF2 and LFSF5 along with differing age of the systems could be contributing factor in the lower TP concentrations. The soil in the C2R systems contained elevated concentrations of Ca with alkaline soil conditions, while the LFSFs contained elevated Al concentrations and more acidic soil conditions.

To assess the ability of the C2Rs to utilize the entire sand media to treat the influent, contour maps for the systems illustrating the distribution of TP throughout the disposal field were created. Similar results were found in the dosed and gravity systems, therefore a representation of the systems was selected and is presented in Figure 4.3.

The distribution of TP concentrations throughout the dosed C2R disposal field is shown in Figure 4.3. The wastewater enters the system with the use of a dosing chamber through an infiltration pipe at the center of the disposal field and is dispersed through a distribution pipe perpendicular to the infiltration pipe. The distribution of TP suggests that there may be an issue disrupting the flow of effluent directly at the infiltration pipe as the concentration of TP is reduced near the infiltration pipe compared to the area surrounding the distribution pipe. Though there is an enriched area of P near the distribution pipe, the higher P concentrations are observed at the deeper depth of 0.30 m - 0.50 m with a highest levels being observed on the right side of the disposal field. The mapping also indicates that the P is migrating along the right side of the disposal field within the shallower soils (0.16 m - 0.30 m) and there are consistently elevated P concentrations from the distribution pipe to the end of the disposal field. There is a large portion of the disposal field that has low P concentrations and the P is not evenly distributed throughout the disposal field, as there is a preferential pathway. Therefore there is a large portion of sand media that is not being utilized for the removal of P.

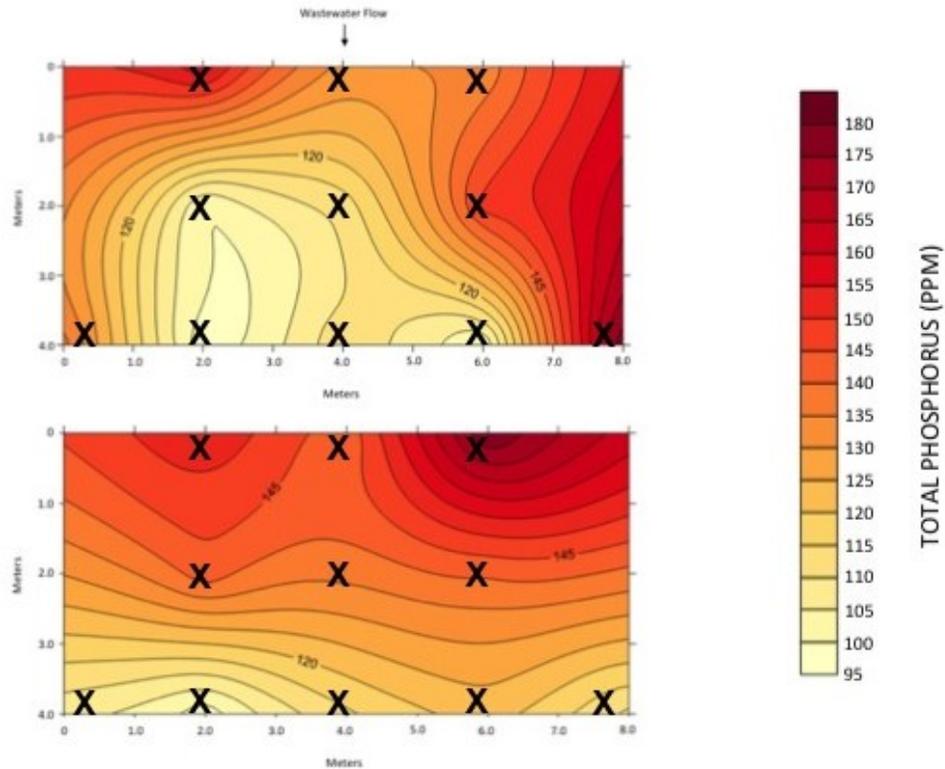


Figure 4.3 Planar view of the disposal field depicting the distribution of total phosphorus for the dosed C2 raised contour trench system. Sample locations that were used in creating the contour maps are indicated using an “X”. The top planar view represents samples collected at a depth ranging from 0.16 m – 0.30 m, the bottom planar view represents a depth of 0.30 m – 0.50 m.

4.1.3 Thomas Brook Watershed On-Site Wastewater Systems

Soil samples that were collected from three sites in the Thomas Brook Watershed were analyzed for various parameters to assess their physical characteristics as well as to evaluate the long-term treatment performance of mature full scale systems. The key parameters are summarized in Table 4.5 and Table 4.6.

Table 4.5 Physical characteristics of on-site wastewater systems located within the Thomas Brook Watershed

On-site Wastewater System	d_{10} (mm)	Uniformity coefficient (d_{60}/d_{10})	Hydraulic Conductivity (m/s)	Hydraulic Gradient (m/m)
Site 1	0.26	2.88	1.29×10^{-4}	
Site 2	0.14	5.07	3.69×10^{-5}	- 0.04
Site 3	0.19	4.95	4.68×10^{-5}	

Table 4.6 Statistical summary of soil chemistry for on-site wastewater systems located within the Thomas Brook Watershed. There were twenty-seven soil samples collected from Site 1, twenty-five soil samples collected from Site 2 and thirty-four soil samples collected from Site 3 along with three duplicate samples taken from each site.

Parameter		Site 1	Site 2	Site 3
pH	Mean	6.3	5.5	6.6
	Maximum	6.9	5.8	7.1
	Minimum	4.3	5.2	5.9
Soil Moisture (%)	Mean	5.0	9.3	6.0
	Maximum	14.4	26.6	22.7
	Minimum	1.4	2.3	2.0
Organic Matter (%)	Mean	1.3	1.5	1.4
	Maximum	2.3	3.5	2.9
	Minimum	0.9	1.0	1.1
AP (ppm)	Mean	63.7	52.2	67.2
	Maximum	346.2	112.0	300.8
	Minimum	28.0	12.3	29.7
TP (ppm)	Mean	163.8	166.7	194.4
	Maximum	377.4	298.0	386.5
	Minimum	105.6	110.9	123.7
Ca (ppm)	Mean	582.9	524.1	931.0
	Maximum	1106	1392.0	1804.0
	Minimum	213	296.0	229.5
Mg (ppm)	Mean	168.3	82.9	159.5
	Maximum	351	336.5	289.5
	Minimum	104	38.0	32.5
Al (ppm)	Mean	762.2	866.4	709.8
	Maximum	1481.7	1163.7	1530.1
	Minimum	318.1	460.4	452.5
Fe (ppm)	Mean	181.2	293.9	213.0
	Maximum	362	575.0	556.0
	Minimum	111	104.0	135.0

Site 1 is comprised of medium grained poorly graded sand ($d_{50} \sim 0.50$ mm). The hydraulic gradient could not be calculated due to the lack of water in the monitoring wells during the monitoring period. Elevated organic matter levels were not observed in the soil below the infiltration pipe (1.2%). Maximum organic matter levels (2.3%) were observed in the soil 3 m downgradient from the infiltration pipe at a depth of 0.45 m - 0.54 m. This area in the disposal field had a lower pH (5.7) comparable to the soil below the infiltration pipe, which varies from 5.8 to 4.3 at depth. These two areas have more

acidic conditions compared to the remaining disposal field, with an overall pH of 6.3. Elevated TP concentrations (377 ppm), up to two times higher than the average TP concentration levels in the remaining disposal field, were observed in the zone of lower pH and higher organic matter, down gradient from the infiltration pipe. Soil sampled directly below the infiltration pipe did not have the expected elevated concentrations (126 ppm - 209 ppm) with TP values at similar depths ranging from 125 ppm – 218 ppm. The zone of enriched P observed in this system is less pronounced than the enriched zones identified in the pilot scale systems located at BEEC, which was surprising given the age of the system and the amount of P that would have been loaded into the disposal field over the last ten years. The Al and Fe concentrations range from 318 ppm to 1481 ppm and 111 ppm to 362 ppm, respectively with the highest concentrations of both Fe and Al occurring in the same zone of elevated P concentrations, lower pH and higher organic matter levels suggesting this area is the preferential pathway for the wastewater.

Site 2 is comprised of medium grained poorly graded sand ($d_{50} \sim 0.40$ mm). The mean hydraulic gradient over the duration of the monitoring period was -0.04 m/m. The organic matter levels were not elevated directly below the infiltration pipe (1.5% at a depth of 0.37 m - 0.50 m) compared to the mean value for the entire disposal field despite the presence of discolored black soil. An elevated organic matter value was observed 5m directly down gradient from the infiltration pipe at a depth of 0.14 m – 0.34 m with a value of 3.5%. The pH in the soil below the infiltration pipe did not vary from the overall values observed in the disposal field. Elevated TP concentrations (298 ppm), up to two times higher than the minimum, which is assumed to be comparable to the background level (110 ppm) were observed in the disposal field. The soil directly below the infiltration pipe did not have elevated concentrations (139 ppm – 196 ppm) with the higher concentration occurring at a depth of 0.37 m – 0.50 m. The zone of enriched TP (207 ppm – 298 ppm) in the disposal field was observed along the edge of the system ranging in depth from 0.14 m – 0.70 m. The Al and Fe concentrations range from 460 ppm to 1163 ppm and 104 ppm to 575 ppm respectively, with the highest concentrations of Al occurring in the same zone as the elevated TP concentrations and higher organic matter levels, along the bottom edge of the disposal field. The Fe is

elevated in this area, but the highest concentrations occur 2.5 m up gradient from the zone of enrichment, closer to the infiltration pipe.

The distribution of TP throughout the Site 2 system located within the Thomas Brook Watershed is shown in Figure 4.4. Based on homeowner's account and soil observations, it is assumed the wastewater is entering the system through an infiltration pipe on the right side of the system. Typical disposal fields, such as those reported by Zanini et al (1998) in central Canada along with systems at the BEEC site, have an area of enriched P concentration near the infiltration pipe, but this was not observed in this system. The area of enriched P concentrations was observed at the outflow of the disposal field. As seen in the other disposal fields in this study, a large portion of the sand media in this disposal field is not being utilized for the removal of P, as the P concentrations are not evenly distributed throughout the disposal field.

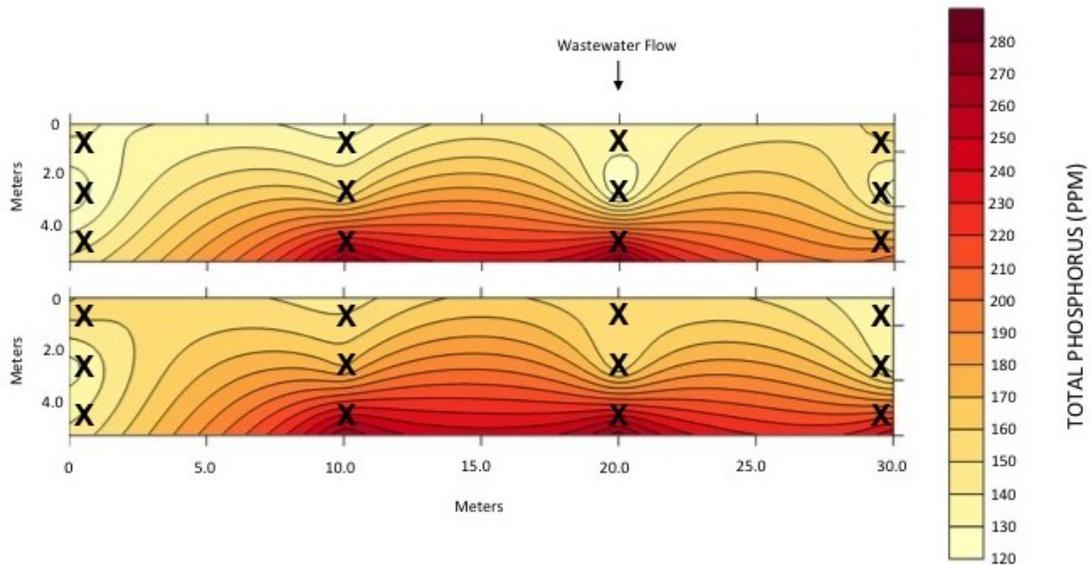


Figure 4.4 Planar view of the disposal field depicting the distribution of total phosphorus for the Site 2 on-site wastewater system located in the Thomas Brook Watershed. Sample locations that were used in creating the contour maps are indicated using an “X”. The top planar view represents samples collected at an approximate depth of 0.18 m – 0.35 m, the bottom planar view represents an approximate depth of 0.35 m – 0.70 m.

Site 3 was comprised of a coarse grained well graded sand ($d_{50} \sim 0.60$ mm). The hydraulic gradient could not be calculated as effluent was only observed in one monitoring well over the duration of the study and two hydraulic head measurements are required in calculating hydraulic gradient. The soil near the infiltration pipe did not have elevated organic matter levels (1.1-1.2%) compared to the mean values for the disposal field (1.4%). The pH of the soil ranged from 5.9 in the saturated soils where free effluent was present to 7.1, which is more representative of the pH values throughout the remaining disposal field that was not saturated. Elevated TP concentrations (386 ppm), up to three times higher than the minimum, which is assumed to be comparable to the background level (123 ppm) were observed in the disposal field. The soil directly below the infiltration pipe did not have elevated concentrations (142 ppm – 163 ppm) with the higher concentration occurring directly below the infiltration pipe at a depth of 0.49 m – 0.60 m. The zone of enriched TP (246 ppm – 386 ppm) in the disposal field was observed in the area of free effluent that is collecting on the left side of the disposal field near the distribution trench. This enriched zone ranges in depth from 0.17 m – 0.89 m with the highest concentrations occurring in the top layer (0.17 m - 0.39 m). The Al and Fe concentrations range from 452 ppm to 1530 ppm and 135 ppm to 556 ppm, respectively. The disposal field also has a slightly elevated level of Ca compared to Site 1 and 2 with concentrations ranging from 229 ppm to 1804 ppm. The highest concentrations of Fe were observed in the same zone as the elevated TP concentrations and lower pH values with the presence of free effluent. There was no obvious area of increased Al or Ca with elevated values occurring throughout the disposal field.

The distribution of TP throughout the Site 3 system located within the Thomas Brook Watershed is shown in Figure 4.5. The wastewater is entering the system with the use of a dosing chamber through an infiltration pipe on the right side of the system. The mapping of the TP soil results indicates that the system is not functioning as intended as shown by the TP concentrations being poorly distributed and a large portion of the sand media in the disposal field is not being utilized in the removal of P. The mapping suggests that the wastewater is entering the infiltration pipe but is being directed in the distribution pipe towards the left side of the disposal field. Similar to the findings in

Site 2, the enriched P concentrations near the infiltration pipe was not observed, in contrast to previous studies (Zanini et al., 1998).

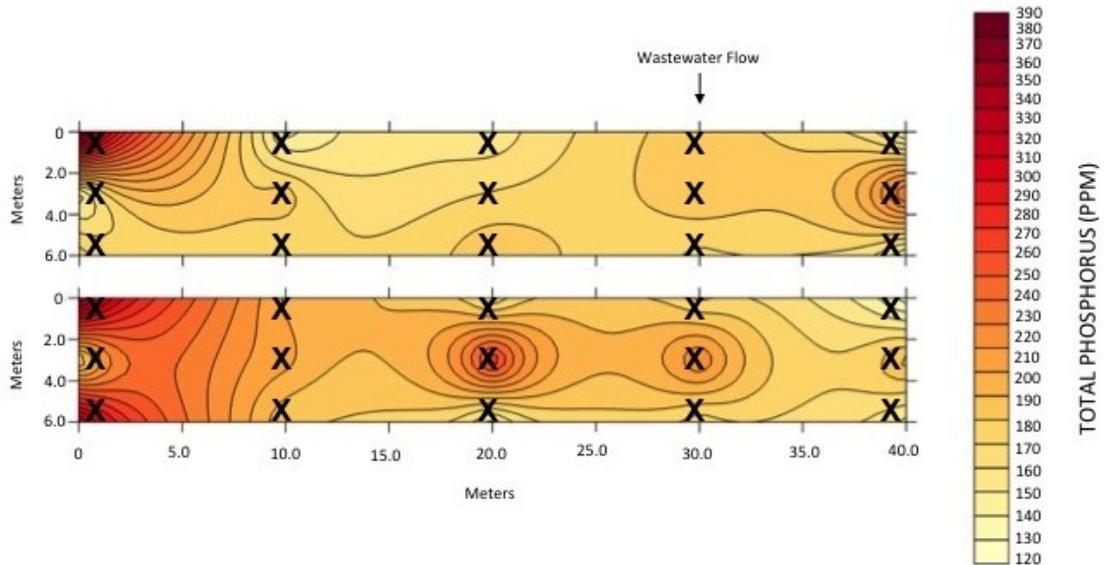


Figure 4.5 Planar view of the disposal field depicting the distribution of total phosphorus for the Site 3 on-site wastewater system located in the Thomas Brook Watershed. Sample locations that were used in creating the contour maps are indicated using an “X”. The top planar view represents samples collected at an approximate depth of 0.30 m – 0.50 m, the bottom planar view represents an approximate depth of 0.50 m – 1.10 m.

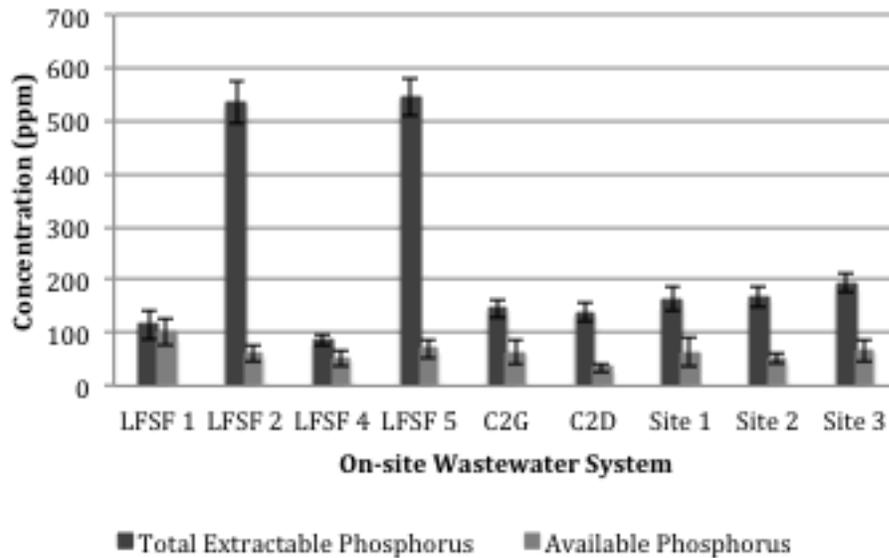
Zanini et al. (1998) determined that in the four sites he studied, the soil directly below the infiltration pipe was enriched in TP concentrations by a factor of 2 to 5 times the background. The magnitude of TP concentrations, ranging from 1200 ppm – 1800 ppm (Zanini et al., 1998), along with the location of the enriched zones was not observed in the TBW sites. The soil chemistry of the imported sand media used in the TBW sites also contained overall lower Al and Fe concentrations with the overall pH of the sand media consistently lower in the TBW systems compared to those observed in the OWSs studied by Zanini et al. (1998) and Robertson (2012). The difference between the soil chemistry observed in the NS disposal field sand media and the soil in the OWSs studied by Zanini et al. (1998) and Robertson (2012) could be contributed to the disposal fields in NS being installed with imported sand media from local quarries, while the systems studied by Zanini et al. (1998) and Robertson (2012) were installed using native soils. Considering

the age of the systems in the TBW, it was predicted that the TP results would have been significantly higher in the disposal fields, suggesting the imported sand media used in the disposal fields is not ideal for retaining P long-term.

4.2 Phosphorus Treatment Mechanisms in the Studied On-Site Wastewater Systems

Phosphorus can undergo various reactions once it enters the disposal field and interacts with the soil components. The most common forms of phosphorus found within the disposal field are organic P and inorganic P. Inorganic P exists primarily as soluble orthophosphates (PO_4^{3-} , HPO_4^{2-} , H_2PO_4^- , H_3PO_4), which is generally available P that is loosely bound and readily available for plant uptake. The majority of P found in wastewater is in the form of orthophosphates and is a large portion of the TP in typical on-site wastewater systems (McCray et al., 2005). There are two main treatment processes that can occur in the disposal field, the adsorption of P to soil particles and the precipitation of P with available mineral cations. The fate of the P in the disposal field is depended on the pH, redox conditions, availability of adsorption sites and availability of mineral components to precipitate forming a phosphorus compound.

In order to further explore the treatment of P in the on-site wastewater systems monitored during this study, the soil was measured for the AP and the acid extractable TP. The mean values comparing the AP and TP concentrations for each disposal field are summarized in Figure 4.6.



Note: Error bars indicate 95% confidence intervals.

Figure 4.6 Phosphorus soil concentration summaries for each on-site wastewater system.

The AP that is represented is the P that was loosely bound and easily removed from the soil. The difference between the AP and TP results indicates the potential amount of inorganic orthophosphates in the system that is not easily removed and therefore is strongly adsorbed or precipitated within the disposal field. The amount of P that can be removed by a particular soil is dependent on the pH and redox conditions (McCray et al., 2009). Adsorption primarily occurs with metal-oxide or clay minerals particles in near-neutral pH conditions but P can also bind on the surface of calcium carbonate or organic matter. The pH of the soil controls the amount of H⁺ protons that are available for adsorption sites (McCray et al., 2009). Soils with lower pH have a higher potential for adsorption due to the higher concentration of H⁺ protons available in the media. The potential for precipitation reactions to occur in the disposal field is also dependent on the pH conditions in the systems.

Soil with a higher pH, such as those present in the C2R systems, have less potential for sorption sites due to the amount of H⁺ protons available at the surface media (McCray et al., 2009). In addition, metal-oxide minerals, such as Al, Fe, and Mn, contain the highest capacity for P adsorption (Brady and Weil, 2008), however these minerals were not found in sufficient concentrations in the C2R systems, reducing the potential capacity for P adsorption in these disposal fields. Though the adsorption potential for P is

seemingly low in the C2R systems, the elevated Ca and basic conditions have the potential for Ca-P precipitation reactions forming such minerals as hydroxyapatite $[\text{Ca}_5(\text{PO}_4)_3\text{OH}]$ and fluorapatite $[\text{Ca}_{10}(\text{PO}_4)_6\text{F}_2]$ (Robertson et al., 1998; Robertson, 2008; McCray et al., 2009).

The P concentrations in the medium grained sand filters (LFSF2 and LFSF5) showed the greatest difference between the loosely bound AP and the TP concentration level of the system. Therefore the LFSF systems have the highest concentrations of P that is remaining in the soil potentially through precipitation and/or strong adsorption mechanisms onto oxides. The medium grained LFSFs have a lower pH and elevated Al concentrations; these conditions are favorable to both adsorption and Al-P precipitation reactions, forming minerals such as variscite $[\text{AlPO}_4 \cdot 2\text{H}_2\text{O}]$ (Robertson et al., 1998; Robertson, 2008; McCray et al., 2009). The precipitation of variscite compounds in OWS was observed by Robertson (2012) in a 20 year old system that contained acidic noncalcareous sand enriched with Al (0.59 +/- 0.09 %w/w) and Fe (2.1 +/- 0.4 % w/w). The systems observed by Robertson (2012) had higher concentrations of Al in the soil and given the marginal difference between the minimum and maximum Al concentration observed in the LFSF medium grained sand, it appears that the longevity of the system to retain P through precipitation with Al would depend on a continual source of Al in the wastewater as the concentrations in the sand media are not sufficient for P precipitation to be the main mechanism for P retention (Zanini et al., 1998).

The bulk of the P in the fine grained LFSFs (1 and 4) was in the form of AP. This suggests that the majority of the P in the system is weakly adsorbed to the soil particles and that precipitation of P in these systems does not seem to be a major treatment process in retaining P: adsorption appears to play a greater role. Even though the conditions are favorable with an overall low pH and the presence of Al and Fe, the sand media was not capable of adequately retaining the P that was entering the system. Though concentrations of Fe in the system are not as elevated as the levels found by Robertson (2012), the zone of enrichment does correlate with the area of enriched P which is favorable for the Fe-P precipitation reactions to occur forming minerals such as strengite $[\text{FePO}_4 \cdot 2\text{H}_2\text{O}]$ and vivianite $[\text{Fe}_3(\text{PO}_4)_2 \cdot 8\text{H}_2\text{O}]$ (Robertson et al., 1998; Robertson, 2008;

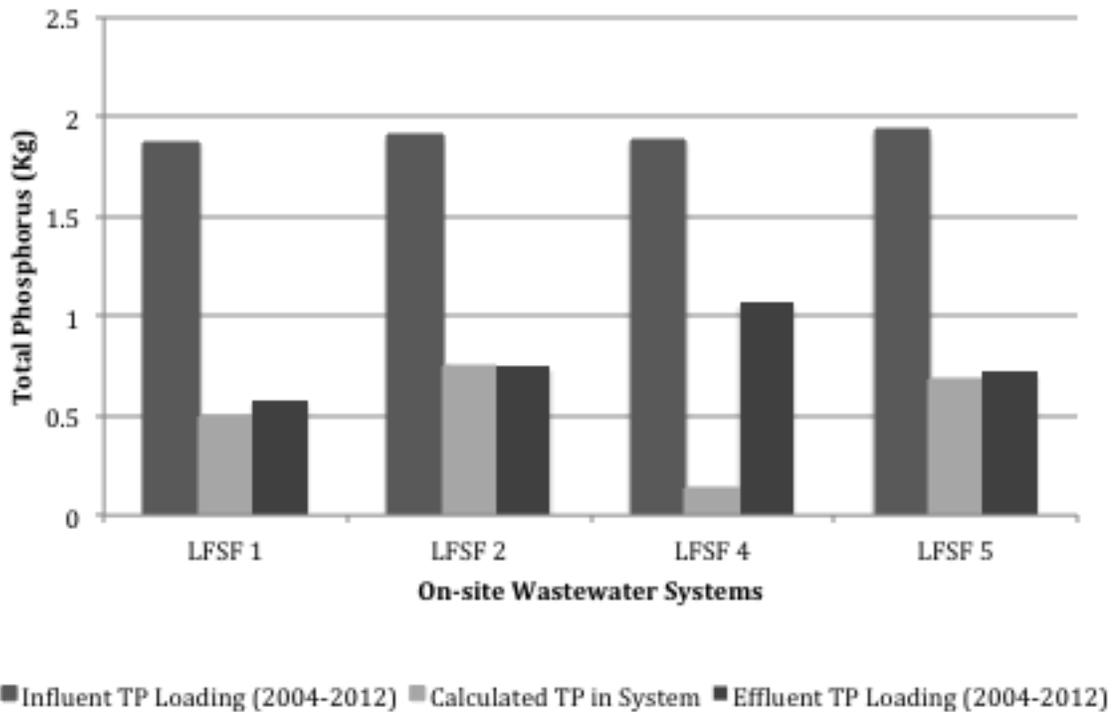
McCray et al., 2009). Robertson (2012) found that strengite formed within the disposal field of a 20 year old OWS. Similar results could be observed in the LFSF as they mature as long as the wastewater provides a source of the Fe for precipitation, therefore increasing the longevity of the system. Further evidence supporting the potential for Fe precipitation in the fine grained sand media is presented in Section 4.4.1.

The OWS located in TBW have been receiving wastewater for 10-15 years longer than the LFSFs but the overall TP concentrations in the treatment media did not reflect this additional wastewater loading. When compared to systems of similar age studied by Robertson (2012), the bulk of P has not been retained in these disposal fields suggesting the P is leaving the systems and migrating into surrounding soil and groundwater. Though the TP concentrations are low, approximately 35% of the TP is in the form of available P; therefore the majority of the P that is in the system is not loosely bound. All three sites have a low pH and contain elevated metal-oxides, which are favorable for both adsorption and precipitation. Site 2 has elevated Al along with a low pH occurring in the enriched zone of P creating the potential for Al-P compounds such as variscite $[\text{AlPO}_4 \cdot 2\text{H}_2\text{O}]$ to form (Robertson et al., 1998; Robertson, 2008; McCray et al., 2009). Site 3 has a low pH where the wastewater is being directed in the disposal field and has elevated Fe occurring in the enriched zone of P producing the potential for Fe-P compounds to form such as strengite $[\text{FePO}_4 \cdot 2\text{H}_2\text{O}]$ and vivianite $[\text{Fe}_3(\text{PO}_4)_2 \cdot 8\text{H}_2\text{O}]$ (Robertson et al., 1998; Robertson, 2008; McCray et al., 2009). However, even though these systems have favorable conditions, precipitation of P compounds does not appear to be a long-term treatment mechanism for retaining P in these systems as the majority of the P has presumably left the disposal field.

4.3 Observed Lateral Flow Sand Filter Phosphorus Retention

To assess the treatment performance of the four LFSFs the cumulative phosphorus loading was calculated using the influent and effluent loading results from previous studies on the LFSFs completed by Havard et al. (2008), Wilson et al. (2011) and Sinclair et al. (2014). Using the these loading rates from the previous studies, the amount of phosphorus (kg) that has entered and left each system since 2004 was

calculated and compared to the amount of phosphorus present in the system at the time of sampling in July 2012 as illustrated in Figure 4.7.



Note: 2004-2011 loading periods adapted from Sinclair et al., 2014.

Figure 4.7 Comparison of cumulative total phosphorus influent loading and the calculated total phosphorus within the on-site wastewater system at time of soil sampling

In Section 4.2 we discussed how the main P treatment process in the LFSF systems is adsorption, which is a limited treatment process as the adsorption sites are filled over time. Therefore the treatment performances of the systems have consistently declined with each monitoring period (Sinclair et al., 2014), as highlighted in Figure 4.7.

The TP loading amount for the four LFSFs were similar with an estimated value of 1.9 kg of TP entering the disposal field over the life of the system. When compared to the amount of TP remaining in the system, the medium grained sand has a greater ability to retain P as shown by LFSF2 and LFSF5 systems retaining 39% and 36% of TP entering the system, respectively. The worst treatment performance was observed in LFSF4 with 93% of the TP not being retained in the system. To determine the amount of

TP not accounted for, the effluent and the calculated TP loading were compared to the TP loading from the influent. The fine grained sand filters had a higher percentage (36% - 42%) of the TP not represented in the disposal field or as effluent leaving the system compared to the medium grained sand filters (22% - 27%). The additional TP could be located directly under the distribution trench in the biomat and sand media, which was not sampled during this study.

Robertson (2012) observed that the majority of the P loading was retained in the disposal field of the 20 year old systems he studied. However, this was not observed in the LFSFs and as the results demonstrate, these systems are providing limited P removal, suggesting the media used in these systems are not effective for long term P treatment.

4.4 Geochemistry of Phosphorus Solids in Lateral Flow Sand Filters

Microprobe analysis indicated that there was a presence of phosphorus-rich solid phases on the sand grains in the LFSF systems and the results are represented using back scattered images and compositional analysis (Figures 4.8 to 4.10). The P-rich solids were observed as both P-coatings and discrete unattached P-grains, which were irregularly shaped and found on composite soil grains, often on multiple locations on the same grain. The microprobe analyses indicated that the P-rich solids did not occur on any particular primary mineral and were found on various lithic fragments that contained plagioclase, K-feldspar and quartz. Background soils were also analyzed and no grain coatings were evident in these samples for either LFSF. Primary apatite was observed in both LFSF soils but was observed in higher frequency in the LFSF2 soil and had similar morphology, with high P and Ca content.

Microprobe analytical totals of the P-solids are relatively low in both LFSF systems, this could be attributed to the following factors: (1) the location of the P-solids are along the edge of the grain and the electron beam is larger than the material that is being measured, therefore the percentage could include the epoxy matrix material; (2) the P-solids contain water or hydroxide components; (3) the microprobe is unable to distinguish between P-solids and associated organic matter (Zanini et al, 1998).

P-coating

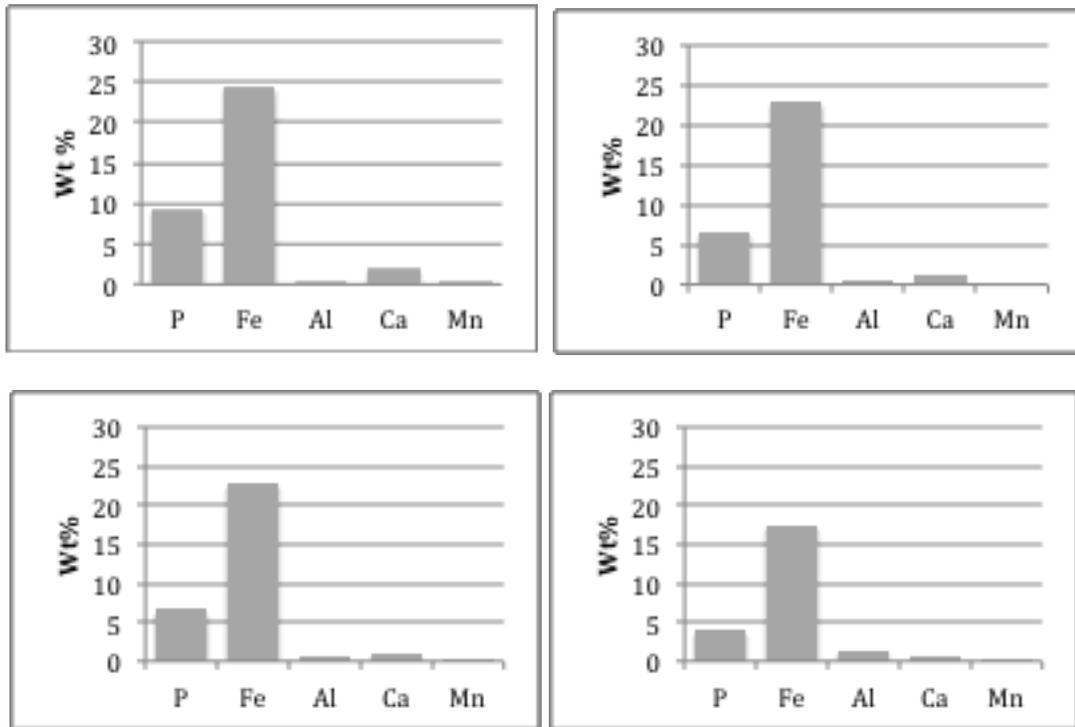
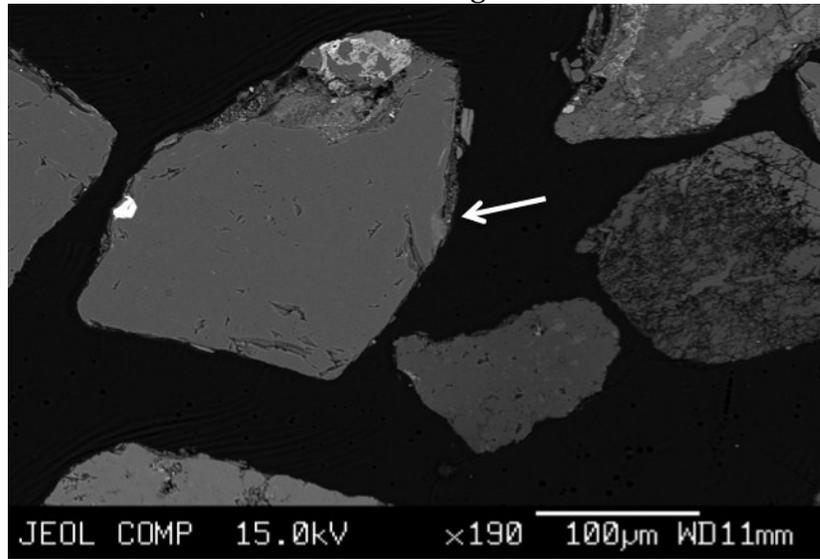
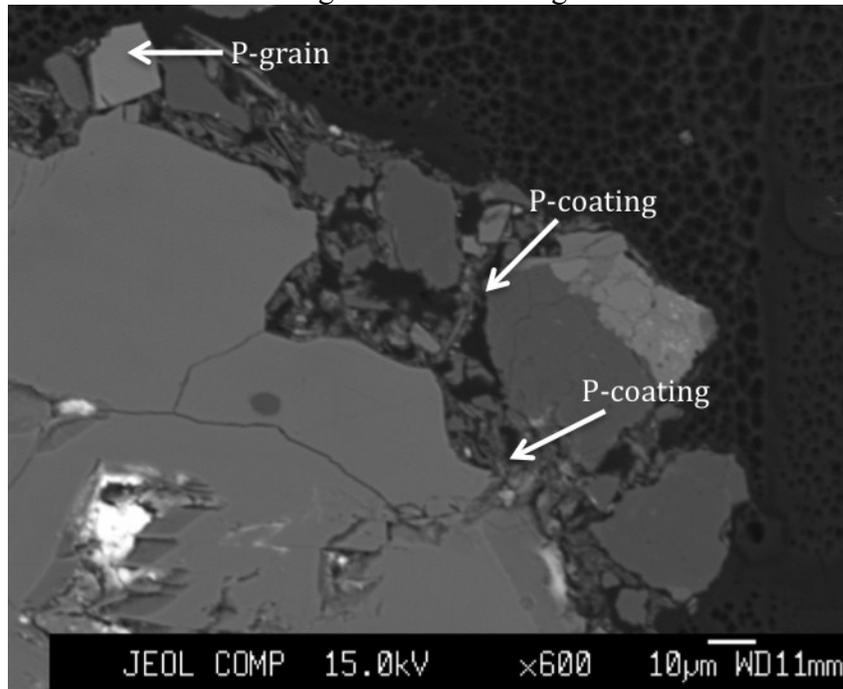
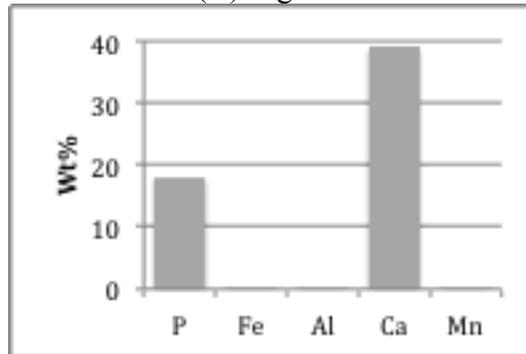


Figure 4.8 Back scattered electron image of representative phosphorus solids with four representative microprobe compositional analyses from LFSF1 (fine grained) from near the infiltration pipe at a depth of 0.35 m – 0.85 m.

P-grain and P-coating



(A) P-grain:



(B) P-coating:

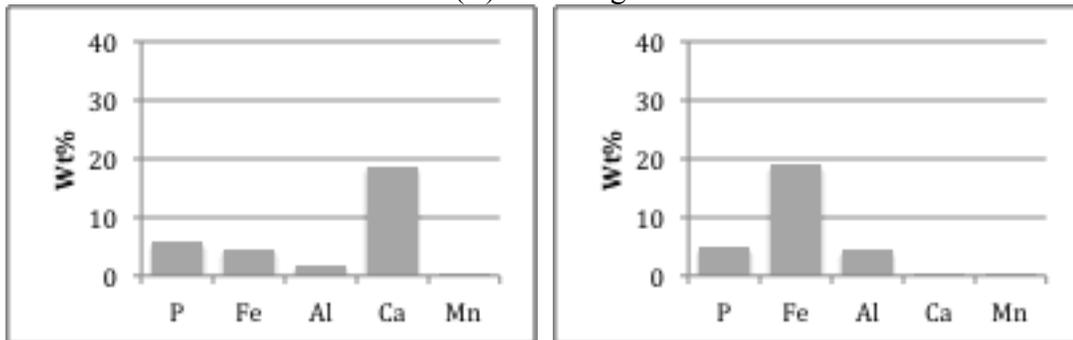


Figure 4.9 Back scattered electron images of representative phosphorus solids with five representative microprobe compositional analyses from LFSF2 (medium grained) from near the infiltration pipe at a depth of 0.82 m – 1.18 m: (a) unattached phosphorus grain with one analysis; (b) grain coating with two representative analyses.

Primary apatite grains

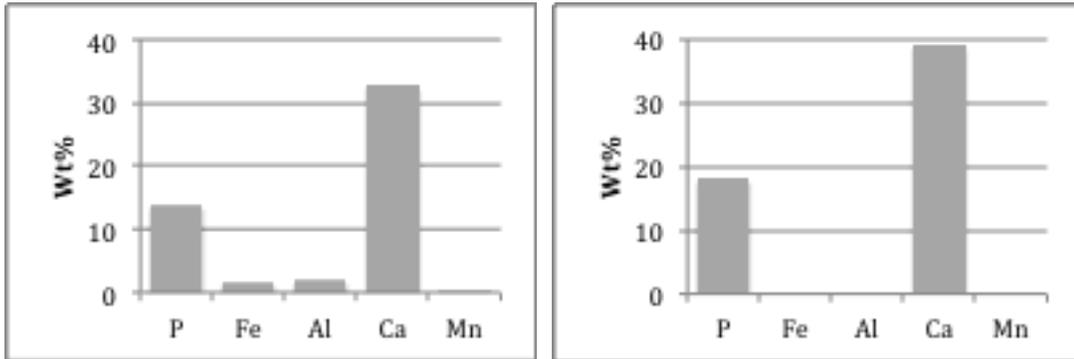
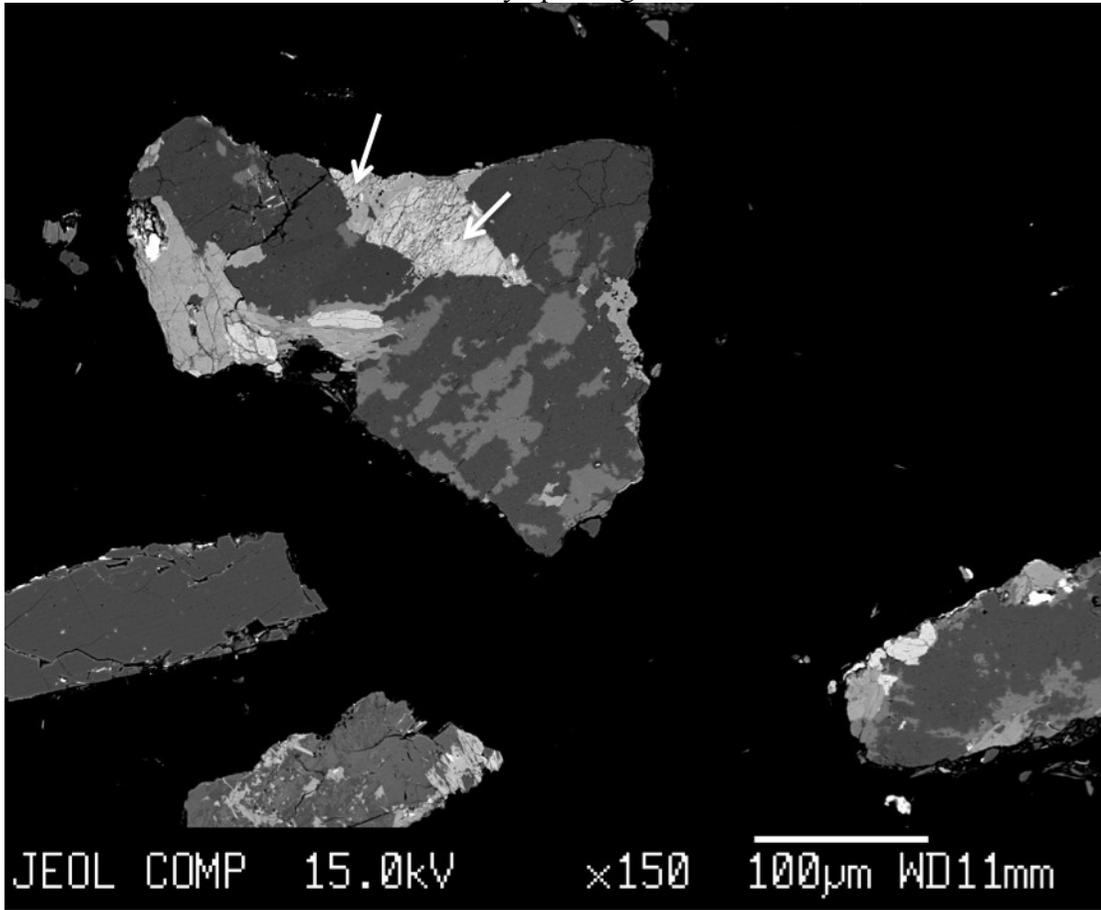


Figure 4.10 Back scattered electron image of a primary apatite grain with visible alterations found with garnet in a predominately albite grain with two representative microprobe compositional analyses from LFSF2 (medium grained) 2.5 m away from the infiltration pipe at a depth of 0.90 m – 1.17 m.

4.4.1 Fine Grained Lateral Flow Sand Filter

The P-solids that were observed in LFSF1 occurred as P-coatings along the grain edges and crevices and did not surround the entire surface of the host grain. Due to the size of the P-coatings, only qualitative elemental analyses were obtained. The P-coatings had P contents ranging from 0.14 wt% to 9.4 wt% with Fe predominately occurring as the dominant cation (2% - 26%) with a lesser amount of Al (0.4 - 11%) and Ca (0.3 – 2%) present. The compositions of the coatings that were analyzed varied slightly, but the Fe:P cation proportion for the coatings was 1.7:1. However, it is known that Fe has the potential to exchange with cations such as Ca and Mn causing impurities in the P-precipitates (Moore, 1970). Therefore when the cation contents of Ca and Mn were included in the ratio, the total cation (Ca, Mn, Fe):P ratio increased slightly to an average of 1.9:1.

The occurrence of the P solids in close proximity to the infiltration pipe, the presence of the coatings on non-specific host grains and the similar composition and texture of these coatings suggest that the P-solids have formed due to precipitation reactions and are therefore of secondary origin. To further establish that the P-solids are secondary in origin, a sample at a greater distance from the infiltration pipe was examined and no P-solids were observed. Primary apatite grains that are non-wastewater derived P minerals were found embedded in the host grains throughout the disposal field sand media indicating that primary P minerals are native to this soil.

The observation of elevated Fe concentrations in LFSF1 corresponds to the enriched areas of elevated TP concentrations and suggests that the P precipitation reactions occurring in this system are associated with this cation. The Fe:P ratios for the P-coatings are an average of 1.7:1, which are higher than what would be expected for the ferric mineral strengite (Fe:P = 1:1) and the ferrous mineral vivianite (Fe:P = 1.5:1) suggesting the mineral could potentially be rockbridgite (Fe:P = 1.67:1) which has a similar ratio to the P-solids found in the system. Rockbridgite is found in lower pH environments (See Appendix D), similar to what is found in the LFSF1 soil, and is derived from the oxidation of vivianite (Nrigu and Dell, 1974). Other secondary P minerals that have similar ratios are hydroapatite (Ca:P = 1.67:1) and fluoroapatite (Ca:P =

1.67:1). When the possibility of Fe exchange with other cations, such as Ca, is taken into consideration, the ratio increases to 1.9:1, but with such a low Ca content in the P-coatings, it is unlikely that these secondary minerals are precipitating. The majority of the ferric and ferrosferric secondary PO₄ minerals containing Fe as the dominant cation have ratios of about 1.5:1 which are lower than what was observed in LFSF1, but with qualitative results, it is possible that these minerals could be precipitating. Zanini (1998), found slightly lower ratios (Fe:P = 1.2:1) in the soil he observed but taking into account the possibility of Fe exchange the ratio increased to 1.5:1. This suggested that the secondary PO₄ minerals that were precipitating in the systems were potentially ferric and ferrous minerals consisting of a Fe:P ratio of 1.5:1.

4.4.2 Medium Grained Lateral Flow Sand Filter

The P-solids in the medium grained LFSF2 predominantly occurred as coatings found along the grain edges and within crevices on the surface of the host grain. The qualitative elemental analyses identified varying composition between the coatings. Several Fe-P rich coatings had average compositions of 18.6 wt.% Fe, 4.8 wt.% Al and 4.5 wt.% P (standard deviation <0.6 wt.%). The Fe:P cation proportion for the coatings is 2.3:1. Several Ca-P rich coatings contained average compositions of 18.2 wt.% Ca, 3.2 wt.% Fe, 2.7 wt.% Al and 7.3 wt.% P (standard deviation <1.6 wt.%). The Ca:P cation proportion for the coating is 2:1. The P-coating that did not have a Ca or Fe as the dominant cation had a total cation (Fe, Ca, Mn):P proportion of 1.9:1, suggesting there may be impurities in these coatings between the exchange of Fe and Ca.

The P-coatings were observed in samples in close proximity to the infiltration pipe and showed no preference in the composition of the host grain, suggesting that the P-coatings are of secondary origin and have formed due to precipitation reactions. Samples farther from the infiltration pipe were analyzed and no secondary P-solids were observed, further indicating that the P-solids near the infiltration pipe are likely secondary.

The Fe:P and Ca:P ratios observed are higher than expected given the results observed by Zanini (1998). The Fe:P ratios for the P-coatings are an average of 2.3:1,

which are higher than the ferric mineral strengite (Fe:P = 1:1) and the ferrous mineral vivianite (Fe:P = 1.5:1) but is similar to the mineral fouchерite (Fe,Ca:P = 2.5:1) which has a similar ratio to the P-solids found in the system especially given the presence of a higher Ca content. Fouchерite is found in lower pH environments (See Appendix D), similar to what is found in the LFSF2 soil, and is derived from the oxidation of hydroxyapatite (Nrigu and Dell, 1974). The Ca:P ratios have an average of 2:1 which is between the ratio of fouchерite (2.5:1) and hydroxyapatite (Ca:P = 1.67:1) suggesting the sample could be near the redox boundary of the system.

The P-grains that were observed had an average P content of 18 wt.% and Ca is the dominant cation with 39.7 wt.% (standard deviation <0.9 wt.%). The Ca:P cation proportion for the P-grains is 1.7:1. This ratio is consistent with hydroxyapatite (Ca:P = 1.67:1), which is similar to the ratios found in the P-coatings where Ca is the dominant cation. The P-grains that were observed contained no pitting, no alterations, and no inclusions unlike the primary apatite grains that were observed. The composition of these P-grains is similar to the Ca-P rich coatings along with the texture of these unattached grains; this suggests that they may be secondary in origin. Also, unattached P-grains were only found in the sample that is from near the infiltration pipe again suggesting they are wastewater-derived.

Primary apatite grains, that were non-wastewater derived P minerals, were commonly found throughout the sand media of this system including the sample selected at greater distance from the infiltration pipe. The grains contained average compositions of 17.5 wt.% P and 38.2 wt.% Ca (standard deviation <1.7 wt.%). The Ca:P cation proportion for the primary P-grains is 2.2:1, which is similar to the secondary grains that were observed near the infiltration pipe. However, the primary P-grains were observed within composite grains unlike the secondary P-grains. Several primary Ca-P grains were altered in appearance, suggesting that the P could be removed during the acid extraction process contributing to the elevated TP results observed in this soil. Numerous primary apatites occurred with Ca-rich garnet, however the Ca associated with garnet is not available for P precipitation as it is tightly bound.

4.5 Water Quality

Effluent monitoring results of key parameters such as TOC, TP, TN, pH, Eh and water level over time provides essential information about the long term performance of the mature systems in TBW. The monitoring period extended from September 2012 to October 2013. The locations of the monitoring wells are shown in Figure 4.11.

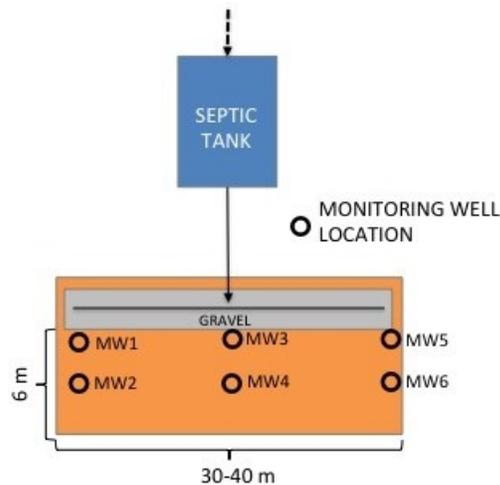
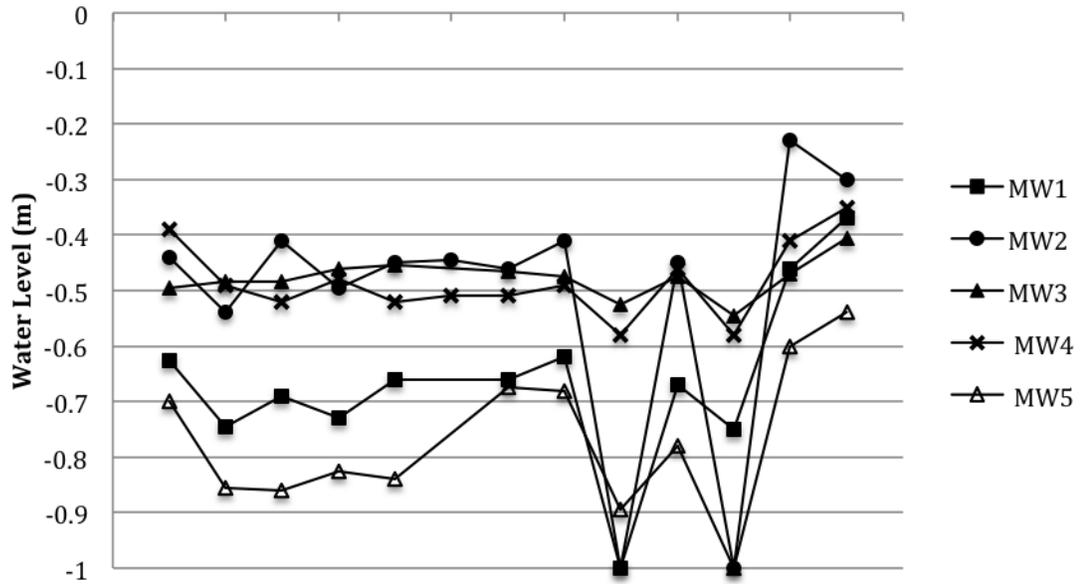


Figure 4.11 Plan view of the monitoring well locations installed in each of the three on-site wastewater systems located in the Thomas Brook Watershed.

4.5.1 Groundwater Conditions

The monthly water table levels for Site 2 and Site 3 during the study periods are illustrated in Figure 4.12. Monthly water levels were not collected for Site 1 due to lack of free water in the monitoring wells during the monitoring period. In addition, only one monitoring well contained effluent at Site 3, whereas five monitoring wells regularly contained effluent at Site 2.

Site 2



Note: Monitoring wells that did not contain water during sampling events are set at a value of -1. MW3, MW1 and MW5 were inaccessible due to ice during the February sampling event.

Site 3

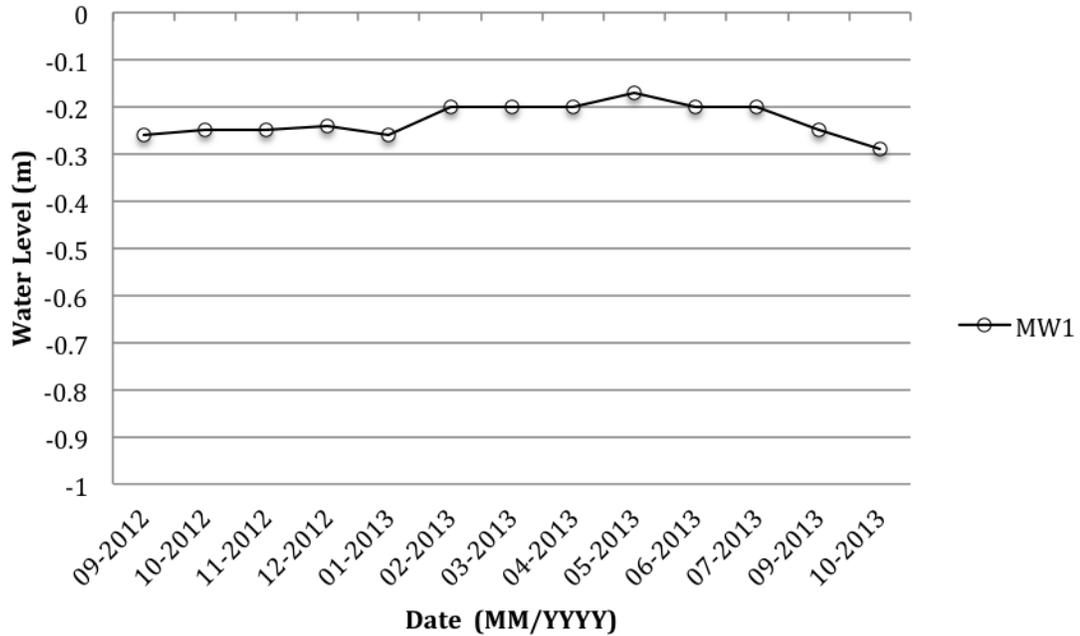


Figure 4.12 The water level for Site 2 and Site 3 shown as the depth below ground surface level for the monitoring period of September 2012 to October 2013.

During the monitoring period, there was no water present in the monitoring wells at Site 1 therefore groundwater condition could not be assessed. However, the hydraulic conductivity of the sand media was estimated to be 0.0129 cm/s with a d_{10} grain size diameter of 0.026 cm (Table 4.5).

The water table within the disposal field at Site 2 had seasonal variation with the observed water table ranging in depths from 0.23 m to 0.9 m below grade. During May 2013 and July 2013, MW1 and MW2 did not contain any effluent and overall the effluent levels were low during these two sampling events. The highest water levels were observed in September 2013. Groundwater flow direction was toward the downgradient slope of the disposal field. The mean hydraulic gradient calculated for the sampling events was estimated to be 0.04. The hydraulic conductivity of the sand media was estimated to be 3.69×10^{-3} cm/s with a d_{10} grain size diameter of 0.014 cm (Table 4.5).

The water table within the disposal field at Site 3 did not vary greatly seasonally, but remained consistent throughout the monitoring period with depths ranging from 0.17 m to 0.29 m below grade. Higher water table conditions were observed in the spring (May 2013) with the lowest water table conditions occurring in the fall (October 2013). Even after purging the well, water levels remained constant, suggesting a rapid flow from the distribution pipe towards the location of the monitoring well. Groundwater flow was unable to be calculated with only one monitoring well containing effluent so it is unknown where the effluent is flowing from MW1. The hydraulic conductivity of the disposal field was estimated to be 4.68×10^{-3} cm/s with a d_{10} grain size diameter of 0.019 cm (Table 4.5).

4.5.2 Redox Conditions

To determine the redox conditions in the disposal field, Eh measurements conducted to identify areas of oxidizing or reducing zones in the disposal field. Oxidizing zones are identified by a presence of elevated Eh values (>200 mV), while reducing zones have a lower Eh value (<200 mV) (Robertson et al., 1998). The effluent *in-situ* water quality results are shown in Table 4.7.

Table 4.7 Effluent *in-situ* water quality summary for on-site wastewater systems located within the Thomas Brook Watershed

Parameter	Site 1		Site 2					Site 3	
	MW2	MW4	MW1	MW2	MW3	MW4	MW5	MW1	MW3
pH			5.7	5.4	6.3	5.6	5.6	6.6	
ORP (mV)	194.1	220.6	84.6	71.6	-201.3	-62.5	-98.2	-236.6	45.8
Eh (RmV)	393.9	420.4	284.4	271.4	1.5	137.3	101.6	36.8	245.6
Temperature (°C)	13.9	13.6	12.8	11.4	13.1	11.8	12.3	13.3	13.7

During the October 2013 sampling event, there was enough effluent present (>5cm) in MW2 and MW4 at Site 1 to capture an *in-situ* measurement to determine the redox conditions in the disposal field. The measurements indicate an oxidizing environment in both monitoring wells however; limited samples were obtained, as the disposal field remained unsaturated during the entire study period with surprisingly no effluent present in the monitoring well located near where the infiltration pipe enters the disposal field.

At Site 2, a reducing zone is identified near the infiltration pipe where Eh values are lower (<200 mV) in MW3, MW4 and MW5. The mean Eh values range from 1.5 RmV to 137.3 RmV, with the lowest measurement found in MW3, which is within close proximity of the infiltration pipe. The disposal field also contains an oxidizing zone farther away from the infiltration pipe with higher Eh values being observed in MW1 and MW2. The overall pH of the effluent is acidic with the highest pH (6.3) being observed in MW3, which corresponds to the lowest Eh measurement. The reducing zone in the disposal field is presumably receiving greater organic loading due to its proximity to the infiltration pipe and a phosphorus-rich plume (2.02 mg/L to 5.45 mg/L) occurs in this area extending downgradient from the distribution trench. Lower phosphorus concentrations are observed in the effluent in the oxidizing zone (0.05 mg/L to 2.33 mg/L).

The presence of effluent in MW1 throughout the study period at Site 3 identified the area as a reducing zone where Eh values are on average 36.8 RmV. This corresponds

with the enriched TP zone that was observed in the soil chemistry along with effluent TP concentrations ranging from 2.11 mg/L to 6.82 mg/L. The mean pH of the effluent is 6.6, which is similar to the Eh and pH results observed in MW3 at Site 2. This further suggests that the effluent present in MW1 is flowing to this area of the disposal field directly from the infiltration pipe and not being evenly distributed through the distribution trench. Minimal effluent was present in MW3 at Site 3 during the October 2013 sampling event, but was not a sufficient amount to enable collection by a bailer for additional sampling. An *in-situ* measurement was obtained indicating an oxidizing zone however, limited samples were obtained, as this area of the disposal field remained unsaturated during the study period.

When reducing environments are present at close proximity to the infiltration pipe, the liberation of additional Fe^{2+} occurs as the results of the dissolution of ferric (oxy) hydroxides that are present in the soil which may promote the precipitation of ferric (oxy) hydroxides and ferric or ferrosferric P solids (Zanini et al., 1998). P removal could be attributed to the precipitation of P solids or the adsorption of P on the newly formed ferric (oxy) hydroxides. Zanini (1998) found, through chemical equilibrium modeling, that as the pH nears neutral Ca- PO_4 solids have the potential to form however the reaction rate for the formation of these precipitates is too slow to retain the P in close proximity to the infiltration pipe. This could potentially be what was observed in the Site 2 disposal field that had an enriched zone of TP downgradient from the infiltration pipe along the edge of the disposal field. When the sand media in the disposal field contains favorable conditions for the precipitation of Ca-P solids, the disposal field width and slope should be adjusted to allow for the slow reaction to take place within the field. Precipitation reactions with P near the infiltration pipes are more favorable involving Fe and Al especially with acidic conditions (Zanini et al., 1998). These conditions were observed in the LFSFs which exhibited Fe-P solids in the sand media in close proximity to the infiltration pipe however, not to the extent to be the main treatment mechanism for retaining P in the disposal fields.

4.5.3 Treatment Performance Assessment

The treatment performances of the OWSs in the TBW are assessed in the following sections. The statistical water quality summaries for Site 2 and Site 3 are presented in Table 4.8 with the seasonal variation illustrated for the monitoring period in Figure 4.13.

Table 4.8 Effluent water quality summary for on-site wastewater systems located within the Thomas Brook Watershed.

Parameter		Site 2					Site 3
		MW1	MW2	MW3	MW4	MW5	MW1
TP (mg/L)	Mean	1.1	0.1	3.5	0.2	0.5	3.9
	Maximum	2.3	0.3	5.5	0.7	1.2	6.8
	Minimum	0.1	0.1	2.0	0.0	0.2	2.1
TOC (mg/L)	Mean	15.1	9.9	25.2	14.5	17.8	37.4
	Maximum	27.7	15.1	46.7	33.6	32.2	79.3
	Minimum	7.9	6.1	11.6	7.6	7.9	14.2
TN (mg/L)	Mean	61.3	67.7	80.9	60.0	53.7	77.0
	Maximum	88.9	180.4	114.9	78.0	73.5	97.2
	Minimum	40.2	39.4	45.3	39.7	32.8	61.8

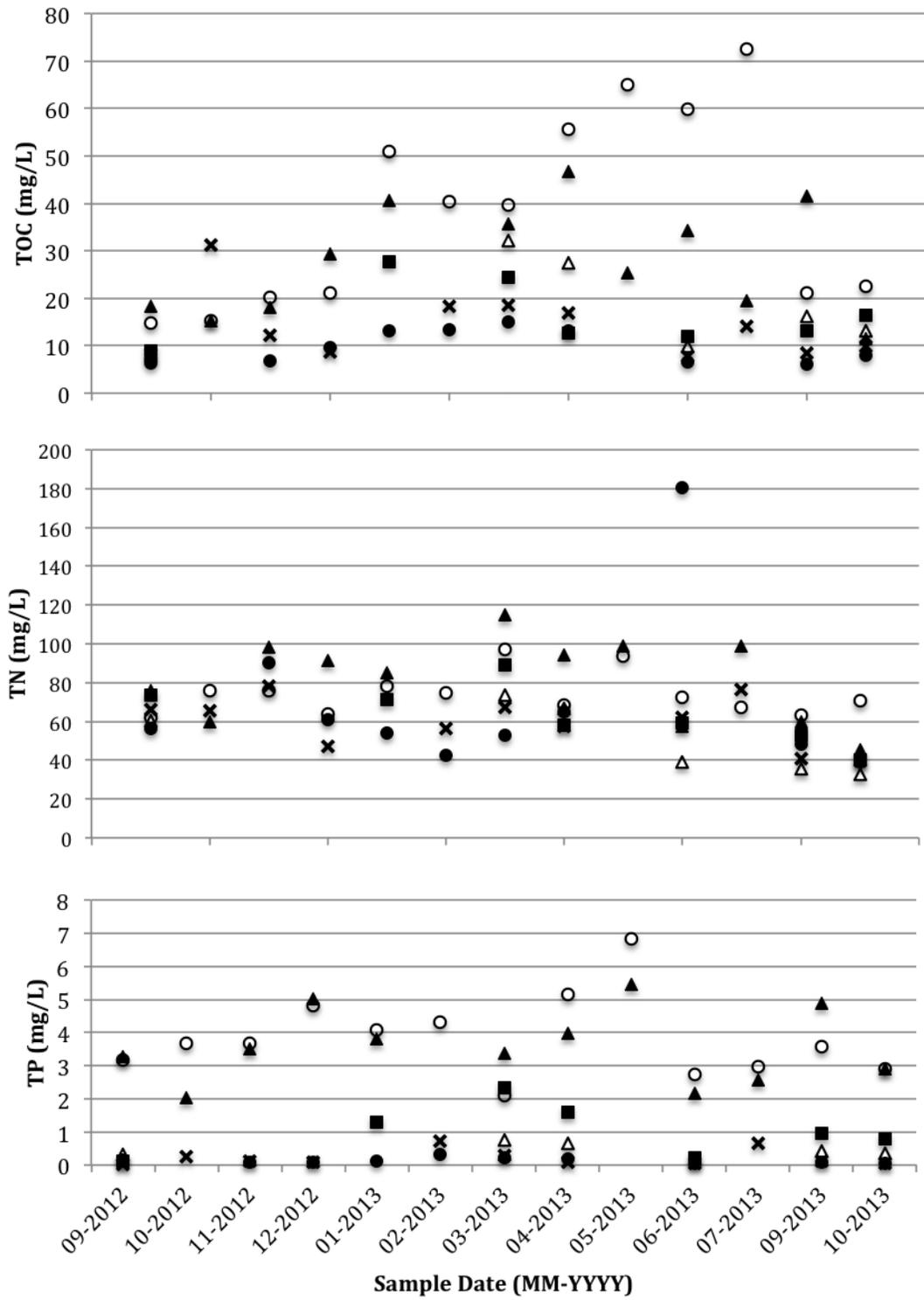


Figure 4.13 Monthly effluent water quality results for Site 2 and Site 3 monitoring wells within the Thomas Brook Watershed.

4.5.3.1 Total Nitrogen

Total Nitrogen (TN) is the combination of all forms of nitrogen: nitrite, nitrate, and Total Kjeldahl Nitrogen (ammonia nitrogen and organic nitrogen). Removal of TN within an OWS is due to sorption, ammonia volatilization, nitrification and denitrification (McCray et al., 2009).

For Site 2, the overall TN concentrations in the monitoring wells were the highest during the early spring in March 2013 with concentrations ranging from 52 mg/L to 115 mg/L, whereas the lowest overall TN concentrations were observed during the fall in October 2013 with concentrations ranging from 40 mg/L to 45 mg/L (Table 4.8, Figure 4.13). The highest TN concentration was 180 mg/L, which was observed in MW2 during the June 2013 sampling event, but elevated results were not measured in the remaining effluent samples for that sampling event. The highest TN concentration is much higher than the mean concentration (67.7 mg/L) observed over the duration of the sampling period for that monitoring well. A small amount of TN reduction was observed between the wells closer to the distribution trench and the wells nearing the edge of the disposal field as shown in Table 4.8. The mean TN concentration in MW3 near the infiltration pipe is 81 mg/L whereas the mean concentration in the monitoring well directly downgradient (MW4) is 60 mg/L, which is a reduction of 26%. A TN reduction of 41.6% was observed between MW3 and MW4 during the March 2013 sampling event, whereas a reduction of 12.4% was observed during the October 2013 event. Similar results were observed when comparing MW1 and MW2, where the 40.6% reduction was observed during the spring and a 2% reduction was observed during the fall with an overall mean reduction of 10.5%.

The reduction of TN from the C2R sites at the BEEC location ranged from 67–80% during a 2008 to 2011 sampling program, which is significantly higher than the TN reduction observed in the effluent sampled at Site 2, which ranged from 2 – 41.6%. The average effluent TN concentrations for the C2R systems were also significantly lower with maximum concentrations of approximately 12mg/L (Bridson-Pateman et al., 2013). Wilson et al. (2011) found that LFSFs capacity for TN reduction declined over time and during the 2007 to 2008 sampling program, the removal rates ranged from 40.9% to

54.3% with maximum outlet concentrations of 18.6 mg/L. The removal rates observed in the Site 2 results appears to be similar to vertical flow sand filters which literature reports having TN removal efficiencies of less than 50% (Brandes, 1980; Kristiansen, 1981b; Pell and Nyberg, 1989b).

Throughout the monitoring period for Site 3, free water was only observed in MW1. The TN concentrations show similar seasonal variation in concentrations as seen in Site 2 (Figure 4.13). The highest TN concentration was observed during the early spring in March 2013 with a concentration of 97.2 mg/L, whereas the lowest concentration was observed during the fall in September 2012 with a concentration of 61.8 mg/L. Though effluent is observed to be 0.15 m below grade in MW1 it is unknown where the effluent is flowing as the downgradient monitoring well in the disposal field does not contain any effluent, therefore the concentration reduction of TN could not be calculated.

4.5.3.2 Total Phosphorus

During the monitoring period, the highest TP concentrations at Site 2 was observed in MW3, closest to the infiltration pipe, during the December 2012 and May 2013 sampling events with concentrations of 4.8 mg/L and 5.5 mg/L, respectively (Table 4.8, Figure 4.13). Effluent was present in the downgradient well in December 2012, with a concentration of 0.1 mg/L, which is a reduction of 98% between MW3 and MW4. There was no effluent present in the downgradient well during the May 2013 sampling event so the reduction between MW3 and MW4 could not be calculated. The overall lowest ranges in TP concentrations were observed during the fall in October 2013 with concentrations ranging from 0.2 mg/L to 2.0 mg/L. The highest concentrations were consistently observed in MW3 with concentrations ranging from 2.0 mg/L to 5.5 mg/L throughout the monitoring period. These values were reduced as the effluent filtered through the disposal field as shown by the reduction in concentration in the downgradient monitoring well (MW4), which had an average TP reduction of 87%. Similar results were observed between MW1 and MW2, where an average TP reduction of 86% was found. The effluent TP concentrations in the monitoring wells at the each of the disposal field (MW2 and MW4) were compared to the Bureau de Normalisation du Quebec (BNQ)

guidelines for P treatment and found that during the monitoring period, the TP concentration achieved the minimum requirement of 1 mg/L (BNQ, 2009).

The C2R systems at the BEEC location had comparable results with both systems achieving average reductions of greater than 90% during the sampling program from 2008 to 2011, as reported by Bridson-Pateman et al. (2013). The mean effluent outlet TP concentrations ranged from 0.1 mg/L to 0.3 mg/L, however the effluent TP concentrations did increase over the course of the study while still maintaining the treatment guideline set by BNQ.

Sinclair et al. (2014) compared the TP influent and effluent concentrations from the BEEC LFSFs for three study periods: 2004-2006, 2007-2008, and 2009-2011. The TP removal rates in 2004-2006 ranged from 72-93%, whereas those rates had declined to 8-44% during the 2009-2011 study period. The latest results are significantly lower than the removal rates observed at Site 2 and the C2R systems at BEEC. The average TP effluent concentration reported by Sinclair et al. (2014) indicated that each LFSF exceeded the BNQ guidelines requirement for the 2009-2011 sampling period. Therefore, all three types of sand media used in the LFSFs are not effective for long-term P treatment.

Although the treatment performance of these systems are shown to decrease with respect to TP, when comparing the C2R systems monitored at the BEEC (>5 years) and the mature system (<20 years) at Site 2, there is not a significant reduction in treatment performance, as would be expected as the sand media reached sorption capacity. This could be due to the differing soil chemistry in the sand media used in each system along with the variable amount of P loading to the Site 2 system in comparison to the controlled P loading to the C2R systems.

Unlike Site 2, the highest TP concentration at Site 3 was observed during the spring in May 2013 with a concentration of 6.8 mg/L, whereas the lowest TP concentration was observed during the early spring in March 2013 with a concentration of 2.1 mg/L. The lowest TP concentration measured during the course of the sampling period exceeds the minimum guideline requirement of 1 mg/L established by BNQ and it is unknown if the TP concentrations are further reduced through the disposal field, as the

flow of groundwater is unknown. Therefore it is not possible to determine if the disposal field is achieving the minimum treatment requirement for TP, however based on the results the assumption is that this system is not meeting the minimum treatment requirement.

4.5.3.3 Total Organic Carbon

During the monitoring period, TOC concentrations in the monitoring wells at Site 2 were the highest during the spring in April 2013 with concentrations ranging from 12.7 mg/L to 46.7 mg/L, whereas the lowest TOC concentrations were observed during the fall in October 2013 with concentrations ranging from 7.9 mg/L to 16.4 mg/L; similar seasonal variation was observed in the TN concentrations (Table 4.8, Figure 4.13). The highest TOC concentration was 46.7 mg/L, which was observed in MW3 during the April 2013 sampling event. This TOC concentration is much higher than the mean concentration (25.1 mg/L) observed over the duration of the sampling period for that monitoring well and the overall mean for the disposal field (16.5 mg/L). The reduction of TOC varied seasonally and spatially in the disposal field between the wells closer to the distribution trench and the wells nearing the edge of the disposal field (Table 4.8). The mean TOC concentration in MW3 near the infiltration pipe is 25.2 mg/L whereas the mean concentration in the monitoring well directly downgradient (MW4) is 14.5 mg/L, which is a reduction of 42.6%. A TOC reduction of 64.9% was observed between MW3 and MW4 during the April 2013 sampling event, whereas a reduction of 15.8% was observed during the October 2013 sampling event. Varying results were observed when comparing MW1 and MW2, where an increase of 4.7% was observed during the April 2013 sampling event and a 51.7% reduction was observed during the October 2013 sampling event with an overall mean reduction of 34.6%.

The TOC concentrations in MW1 at Site 3 varied throughout the monitoring period. The highest TOC concentration was observed during the summer in July 2013 with a concentration of 72.6 mg/L, whereas the lowest TOC concentration was observed during the fall in September 2012 with a concentration of 14.9 mg/L (Table 4.8, Figure 4.13).

CHAPTER 5: CONCLUSIONS

5.1 Conclusion

5.1.1 Phosphorus Retention in On-Site Wastewater Systems in Nova Scotia

This study evaluated nine OWS utilizing six different soil materials to better understand long term P treatment capabilities and determine what characteristics promote the retention of P in OWS.

For the medium grained LFSFs, the observed TP concentrations were up to four times the levels compared to the fine grained LFSFs under similar flow conditions. However, microprobe analysis indicated the presence of non-wastewater derived P-rich apatite grains in the medium grained material, which could be contributing to the overall TP concentration. When comparing the cumulative P treatment over the lifetime of the LFSFs, both the fine grained and medium grained soil media did not have the ability to retain the majority of the estimated P loading. The majority of the P in the fine grained LFSFs was also in the form of AP, which is weakly adsorbed to the soil particles indicating that precipitation of P in these systems does not seem to be a major treatment process in retaining P. Microprobe analysis identified P-rich solids on the soil grains in both soil media materials installed in the LFSF systems. However, the low amount of the secondary mineralization in the LFSF systems indicates it is not an effective solution for the long term treatment of P with the current materials being used. The differing results between the LFSF systems indicates the importance of the soil geochemistry of the materials being used in the disposal fields as a major factor in the systems' effectiveness for long term treatment of P.

The overall TP concentrations in the C2R systems were lower compared to the level observed in the LFSFs that contained similar sized medium grain sand. This could be the result of differing ages of the systems as well as further indication of the importance in the soil geochemistry in the disposal field and the systems ability to retain P.

The OWS located in TBW have been receiving wastewater for 10-20 years, however the overall TP concentrations in the treatment media did not reflect the amount of P loading the systems have received as they matured. After comparing to systems of similar age studied by Robertson (2012), the majority of P is presumed to have left the systems and migrated into surrounding soil and groundwater suggesting the imported sand media used in the disposal fields is not ideal for retaining P long-term.

This research indicates that the physical and chemical characteristics of the sand media used in the installation of disposal fields plays a key role in the ability to retain P. Overall, this study suggests that current materials being used in OWS in Nova Scotia are not effective for long term P treatment. In the systems that were evaluated, precipitation of P was not the major treatment mechanism, and the BEEC systems have shown a decline in P retention as they mature. The lack of ability of the OWS in this study to retain P was different than mature systems studied by Robertson (2012) in Ontario. This could be attributed to the type of media used in the disposal fields. In Nova Scotia, the material used is imported sand to maximize the hydraulic capabilities of the disposal field, unlike systems in Ontario that utilize native materials.

In addition to the chemical characteristics of the sand media, this research suggests that the ideal conditions for maximizing P attenuation within disposal fields would include the use of fine grained material for greater adsorption of P on the grain surface. The results from this study also indicate that pH and redox conditions will also influence retention mechanisms. The presence of high water content in the disposal field would also promote reducing conditions leading to the dissolution of ferric (oxy) hydroxides in turn increasing the chances of Fe-P precipitation occurring. Lower pH conditions in the disposal field tend to further promote the leaching of Fe and Al from the sand material providing additional cations for precipitation with P along with enhancing the adsorption of P to (oxy) hydroxide minerals. Elevated Fe and Al concentrations in the sand would be beneficial, with continuing source being supplied from the effluent.

It was also found that none of the nine OWS designs were effective in spatially distributing the wastewater to the entire disposal field bed. In both the LFSF and C2R designs, preferential effluent flow was observed with the majority of the wastewater

flowing along the base of the systems. Therefore a large portion of the sand media in the disposal field is not active in the wastewater treatment process.

The ability to predict the degree of P attenuation in a disposal field is difficult as the controlling factors for P immobilization are site specific. However, implementing new regulations for the physical and chemical characteristics of the sand material used in the installation of a disposal field may be required to increase P treatment, particularly in watersheds that are currently experiencing water quality issues and are at risk for eutrophication of surface water.

5.1.2 Phosphorus Management in Nova Scotia

In Nova Scotia, a relatively simple mass balance model is used to determine the amount and type of development that can occur in the watershed of a lake without causing eutrophication issues due to P loading. A major input parameter in the model is P retention within OWS in the watershed. Due to the limited research on the retention of P in typical OWS in Nova Scotia, a majority of the estimates are based on the research conducted by Robertson (2012). The systems in Robertson's (2012) research have the ability to retain a large portion of the P in the OWS disposal fields as they mature, which was not observed in the material in the nine OWS studied during this research. Based on the results of this research, the material used in the OWS in Ontario is physically and chemically different than the sand media used in Nova Scotia OWS, resulting in an over-estimation in the material's ability to retain P. The material observed in the nine OWS predicted a minimal amount of P treatment in the systems as they mature. This should be taken into account when predicting the amount of P loading when developing a watershed. A conservative input parameter of 10-20% of P being retained in the OWSs is suggested based on this research, especially in watersheds that are at risk for eutrophication of surrounding surface water.

5.2 Recommendations for Future Research

- The sand media studied in this research is not conducive to the long term treatment of P. Therefore OWS across Nova Scotia that contain similar material have minimal ability to treat P after approximately 8 years. Nova Scotia Environment should implement regulations to restrict what material is installed in OWS to promote the treatment of P in the disposal fields. Also, an inspection and sand media replacement program should be implemented in watersheds at risk for P loading and eutrophication of surrounding surface water.
- A survey of the physical and chemical characteristics of OWS sands should be completed on the distributors of sand material used for the installation of disposal fields in Nova Scotia to determine if the material is capable of long term P treatment. Further testing should be completed to identify a source of sand material that contains the optimal physical and chemical characteristics to facilitate long term treatment of P in the disposal fields installed in Nova Scotia.
- Research should be conducted to determine if sand media could be amended with a coating to better retain P.
- Within the LFSFs and C2R systems assessed in this study, the wastewater is not distributed efficiently to the entire disposal field, leaving the majority of the sand material not active in the treatment process. Further assessment into the distribution of wastewater in the disposal field should be conducted in order to determine how to modify the system to optimize the use of the disposal field to increase the treatment capacity of the systems.
- Additional research should be conducted to better assess the P loading parameter that is currently used in the Nova Scotia Phosphorus Loading Model. Currently there are no guidelines for the selection of the P retention parameter.

Based on the results of this study, minimal P was retained in the OWS as they matured, and indicates that a conservative estimate of P retention would be zero.

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APPENDIX A Nova Scotia Geology Maps

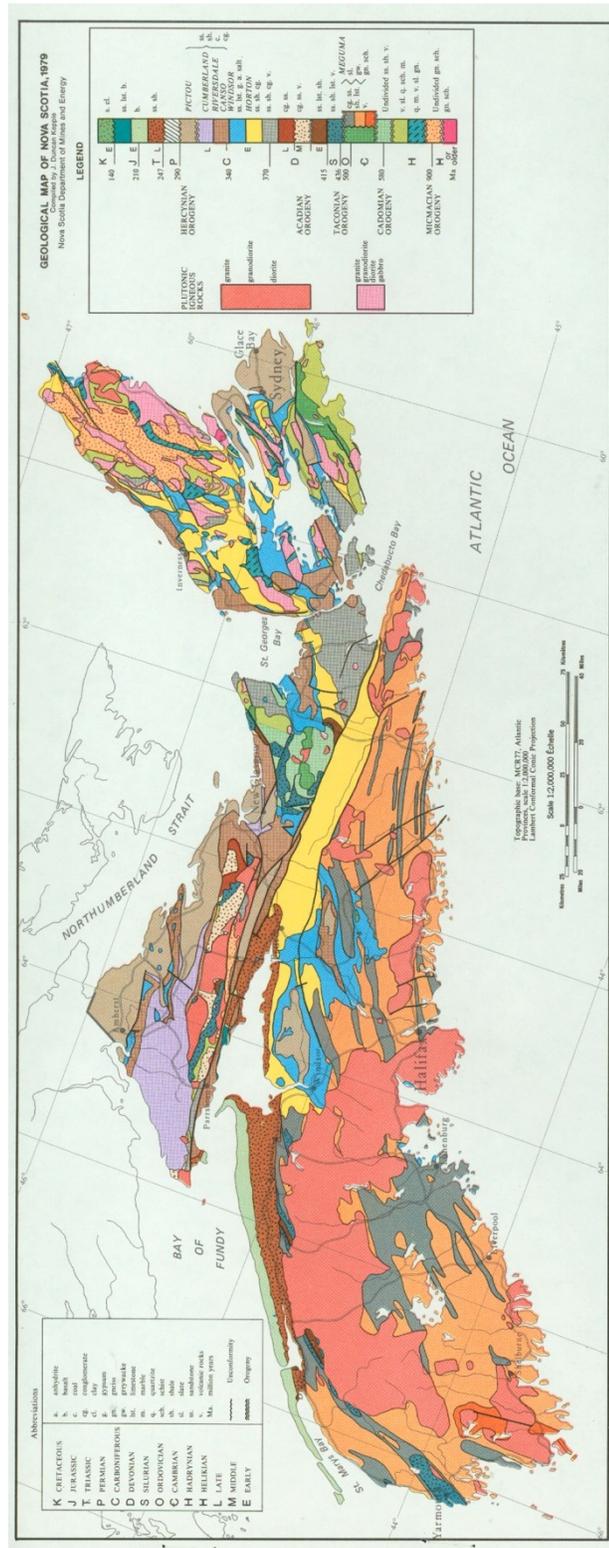


Figure A-1 Geological map of Nova Scotia (Keppie, 1979).

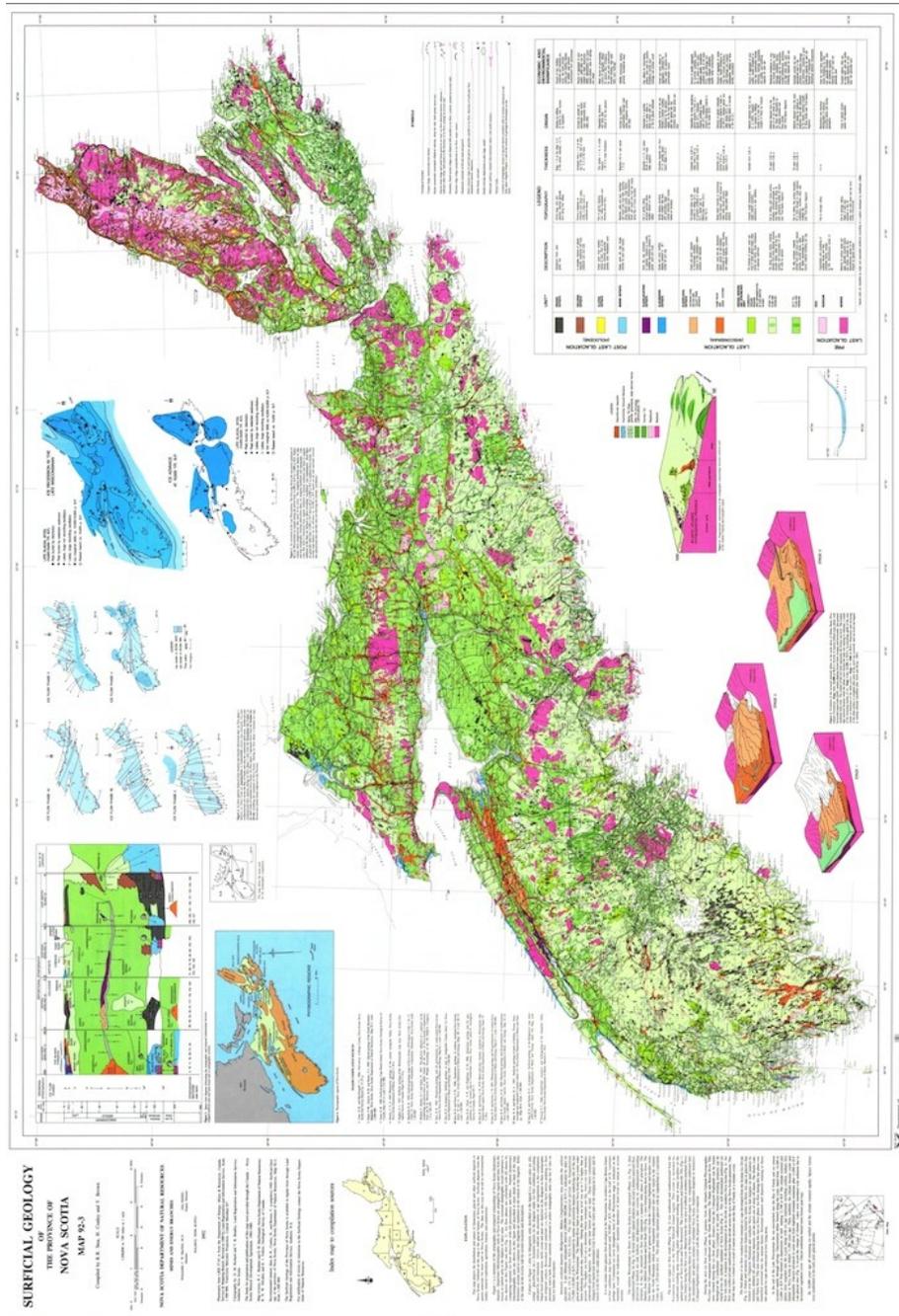
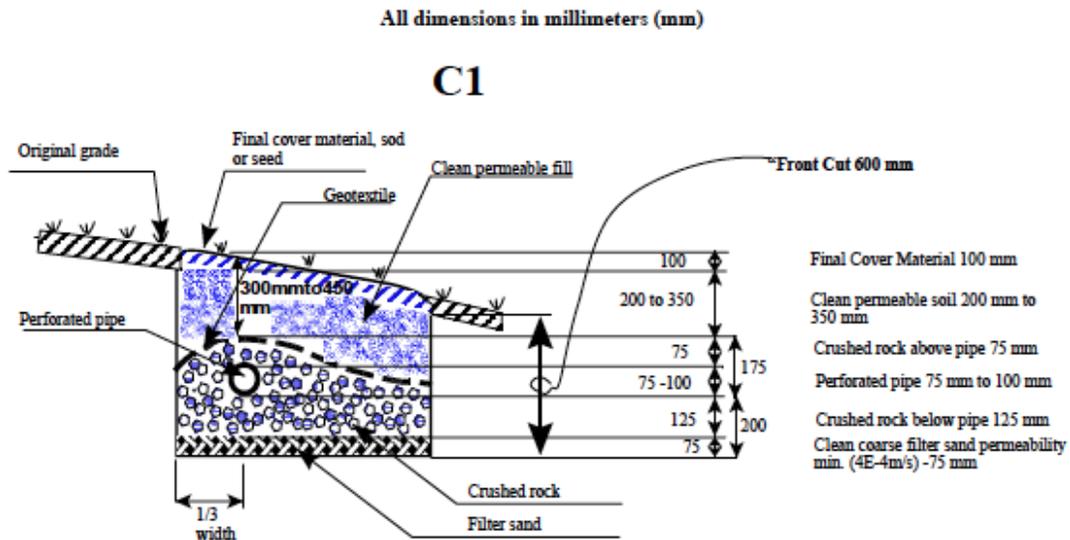


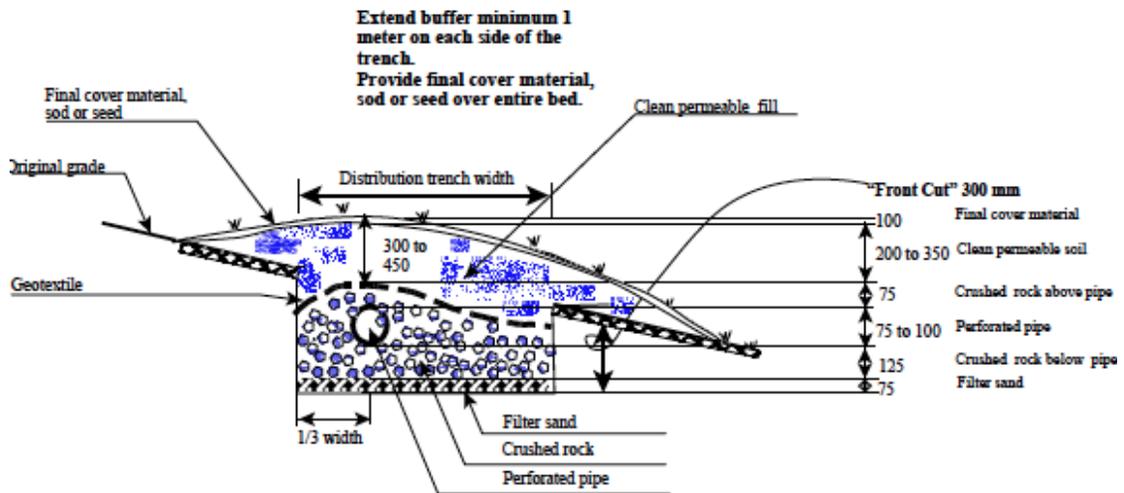
Figure A-2 Surficial Geology map of Nova Scotia (Stea et al., 1992).

APPENDIX B On-site Wastewater Systems used in Nova Scotia



Minimum 1 meter separation from bedrock, water or too permeable soil

C1 raised



Minimum 1 meter separation from bedrock, water or too permeable soil

Figure B-1 Typical C1 contour trench system cross-section (Nova Scotia Environment, 2009).

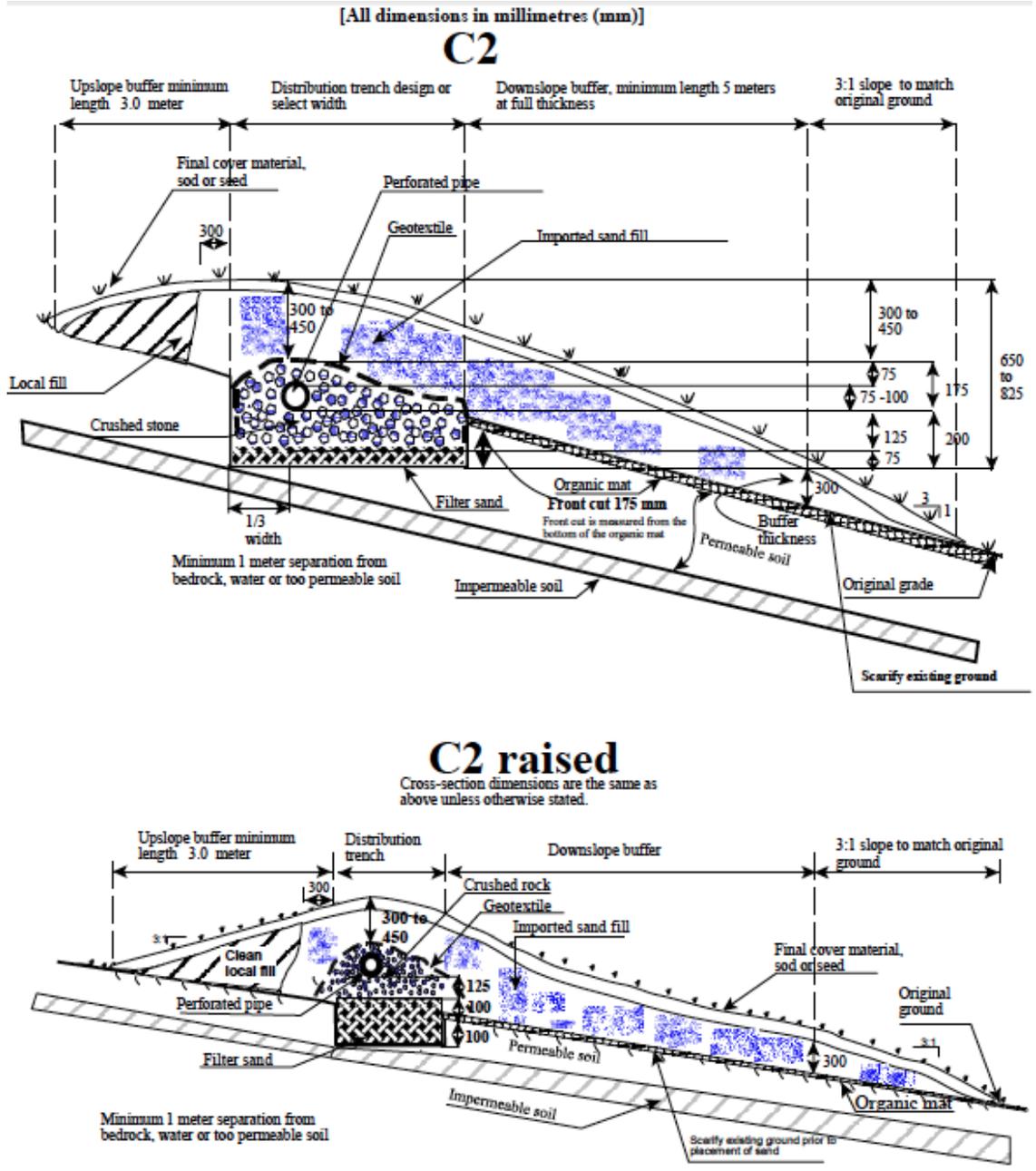


Figure B-2 Typical C2 contour trench system cross-section (Nova Scotia Environment, 2009).

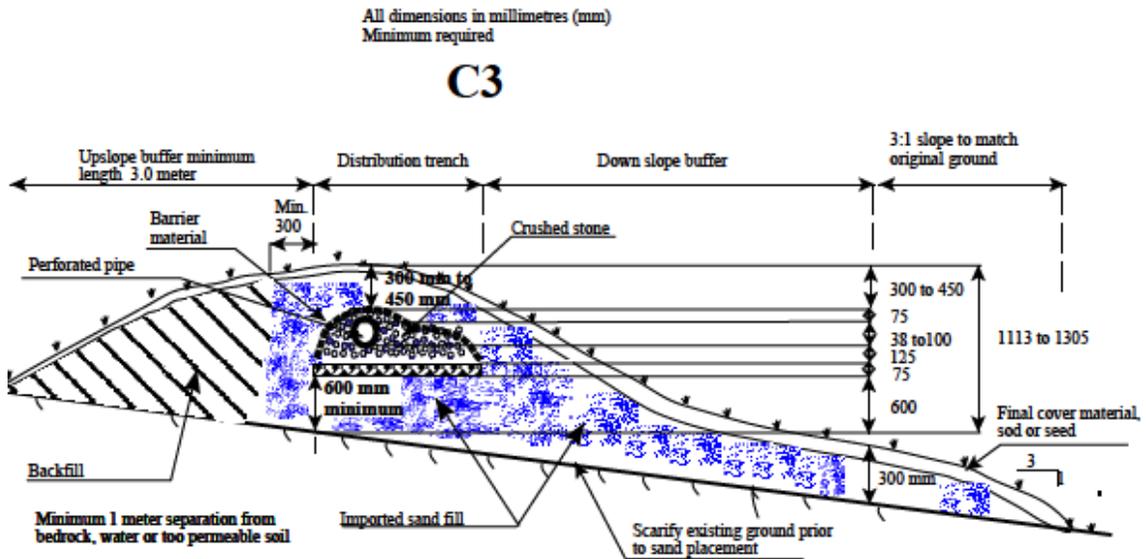


Figure B-3 Typical C3 contour trench system cross-section (Nova Scotia Environment, 2009).

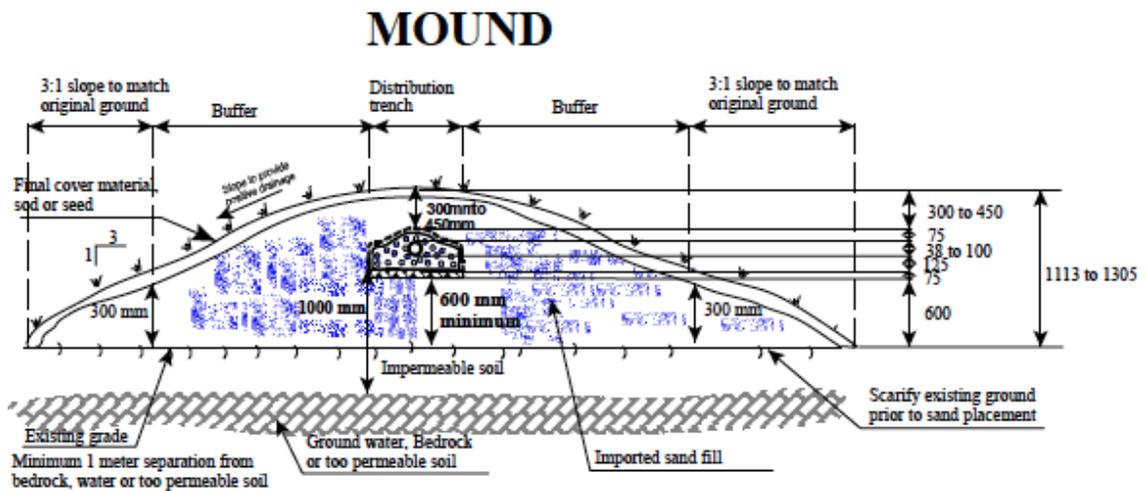


Figure B-4 Typical mound system cross-section (Nova Scotia Environment, 2009).

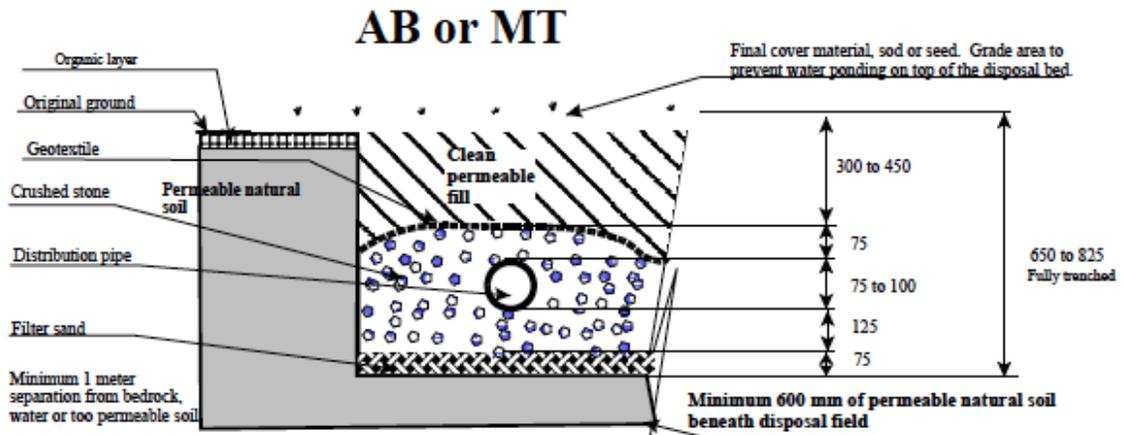
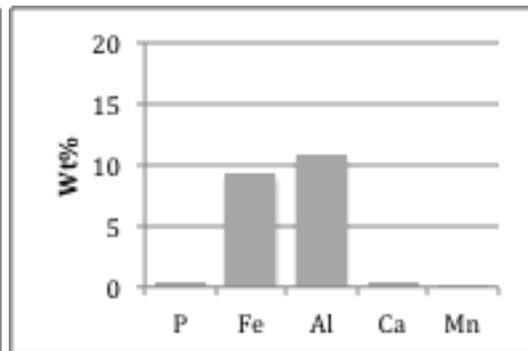
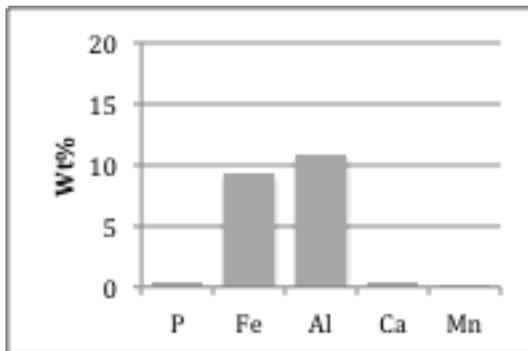
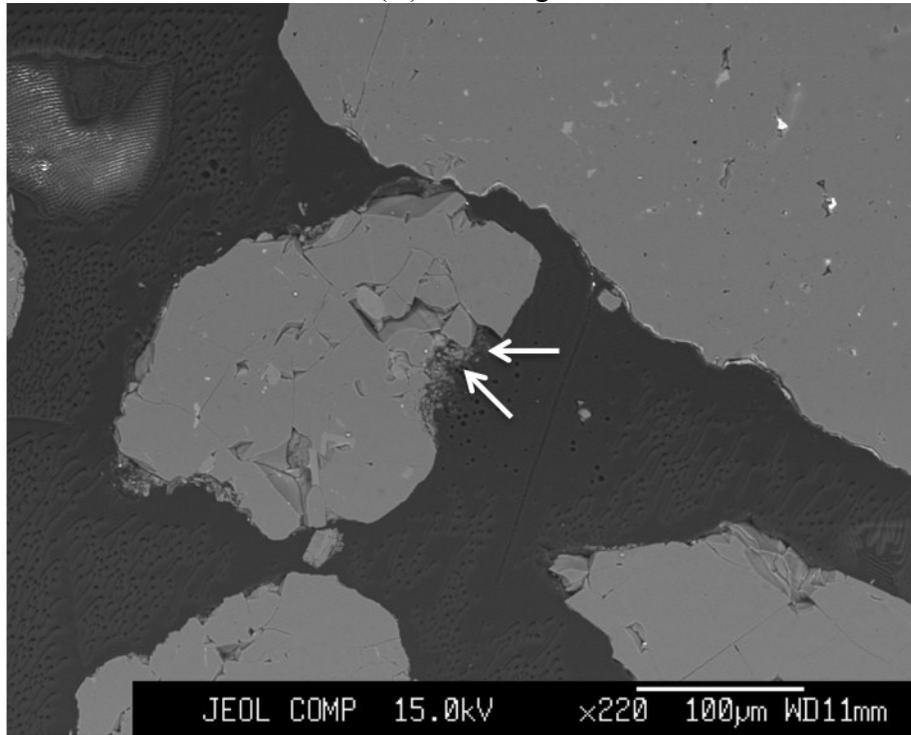


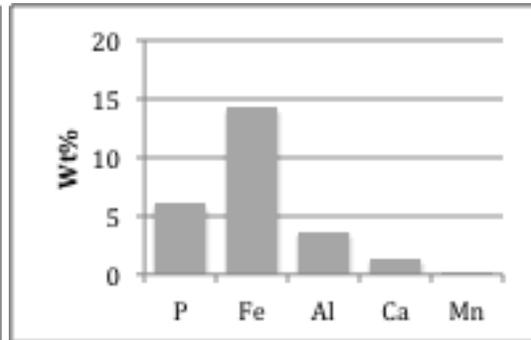
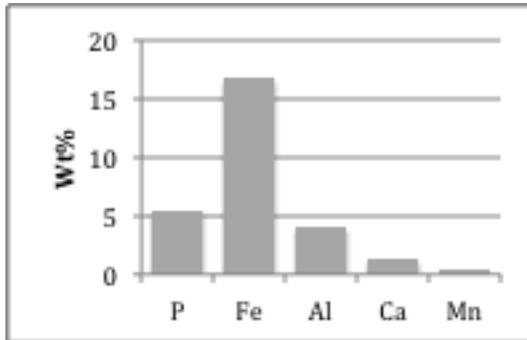
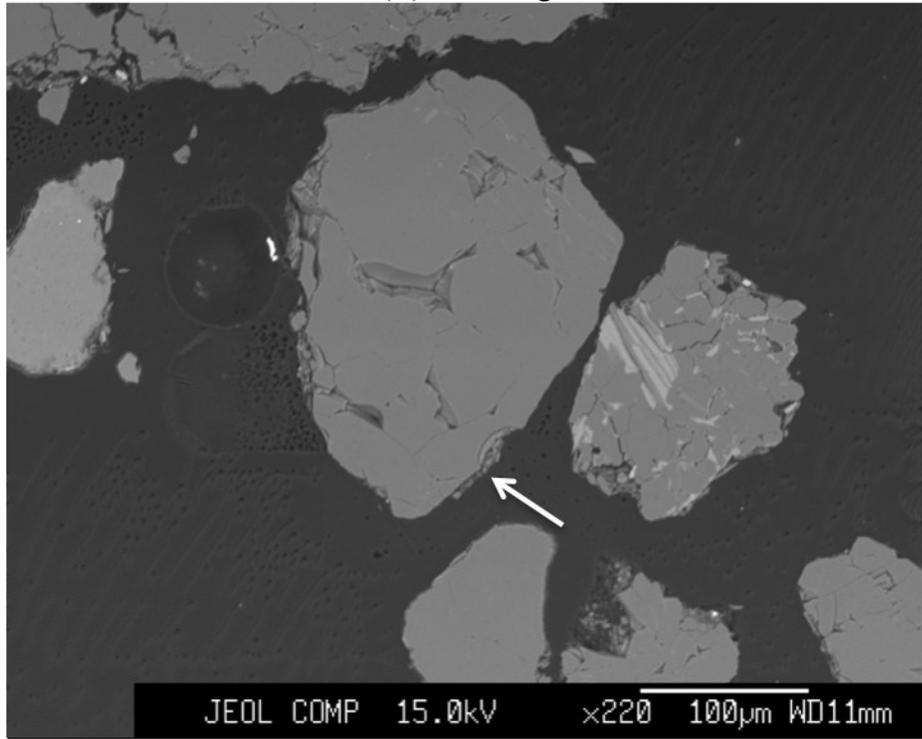
Figure B-5 Typical area bed or multiple trench system cross-section (Nova Scotia Environment, 2009).

APPENDIX C Additional Electron Microscope Analysis

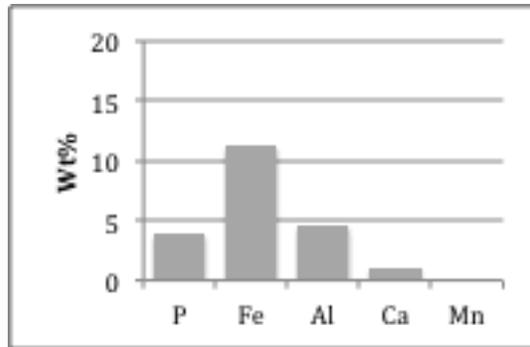
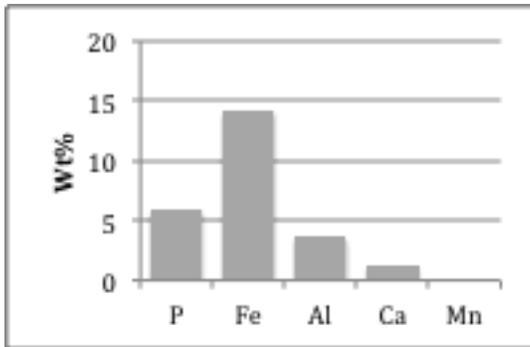
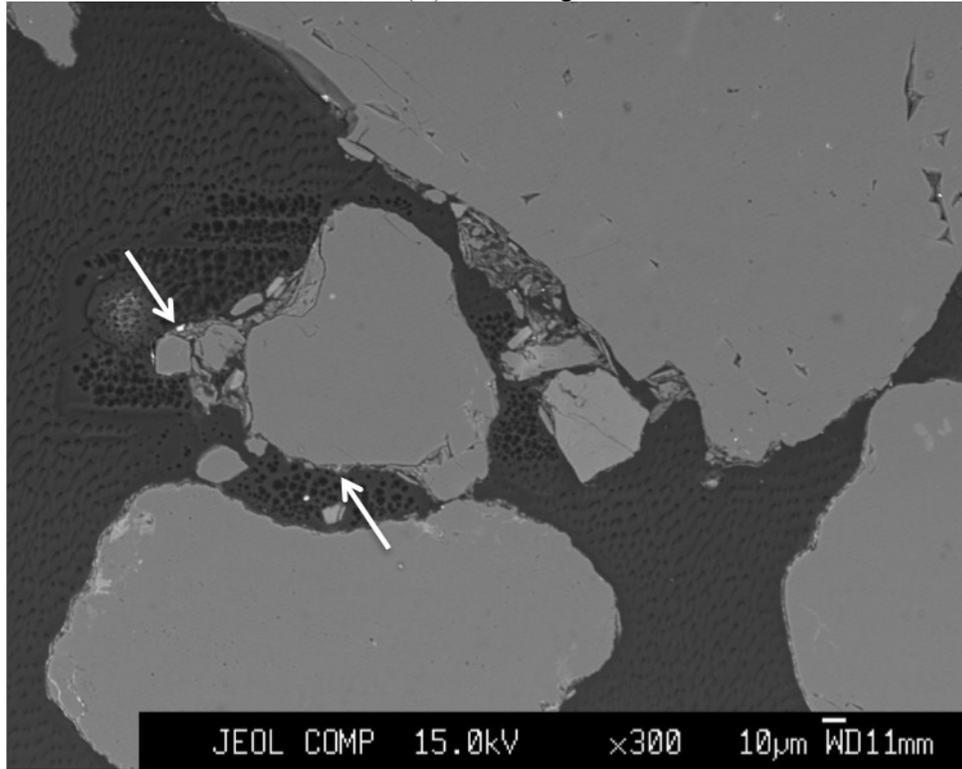
(A) P-coating



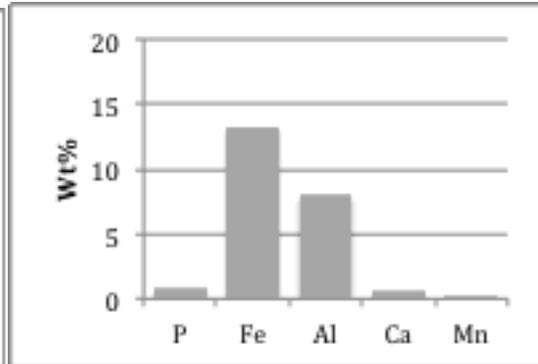
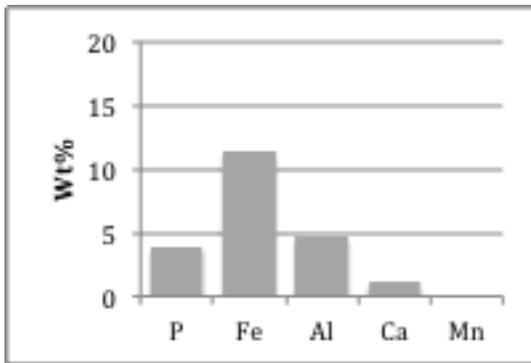
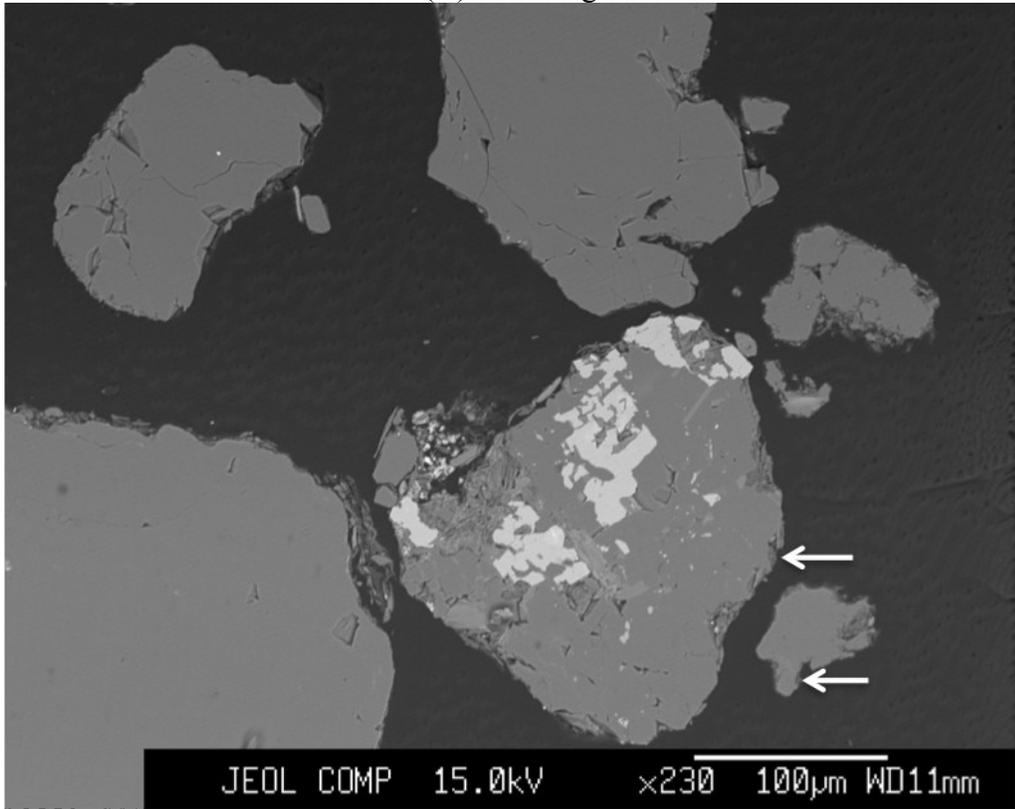
(B) P-coating



(C) P-coating



(D) P-coating



(E) P-coating

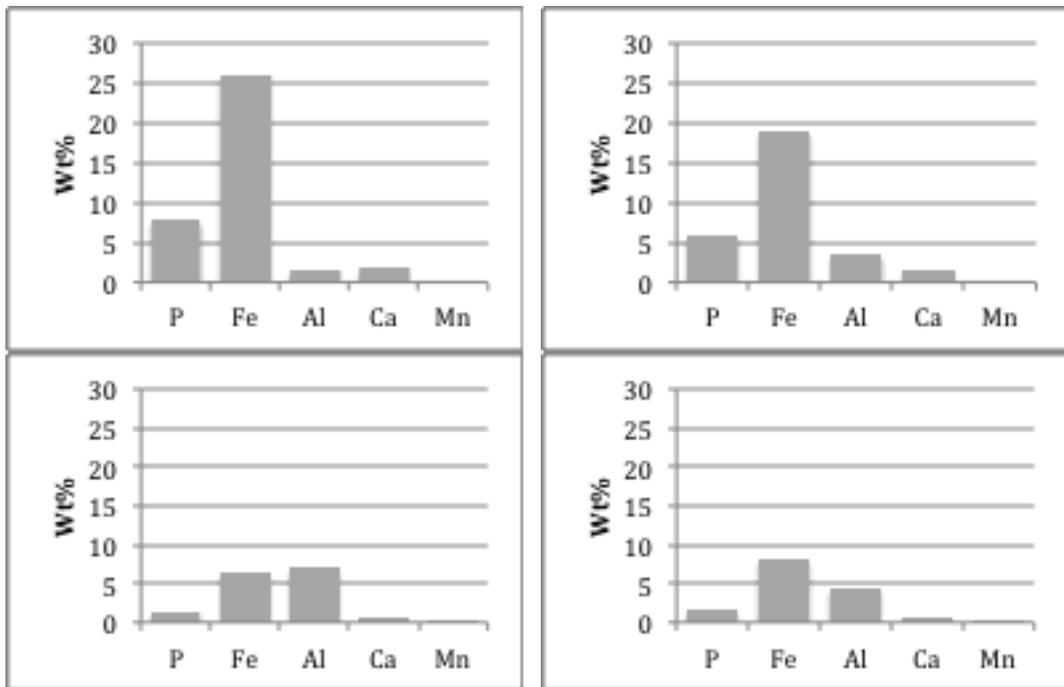
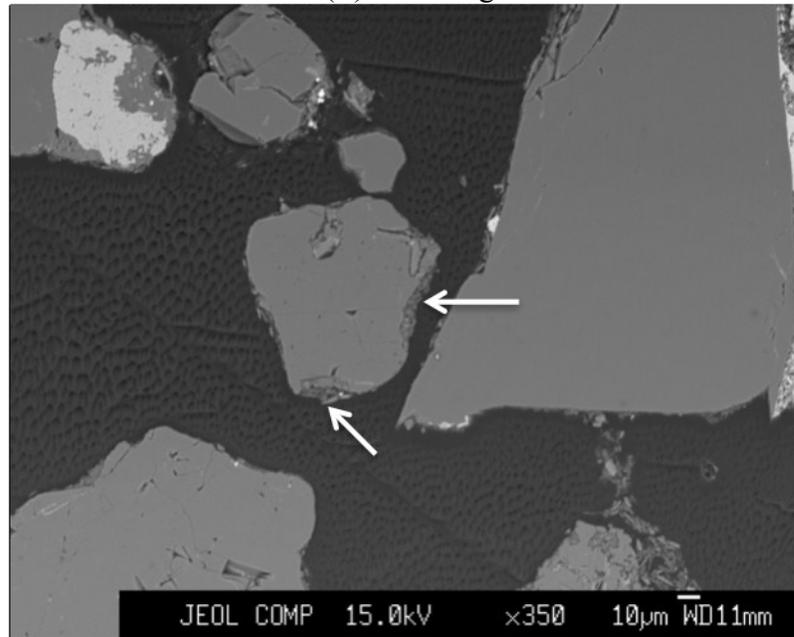
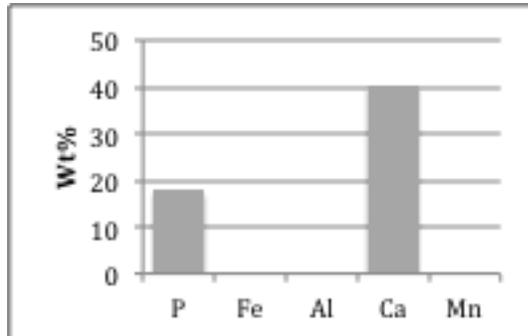
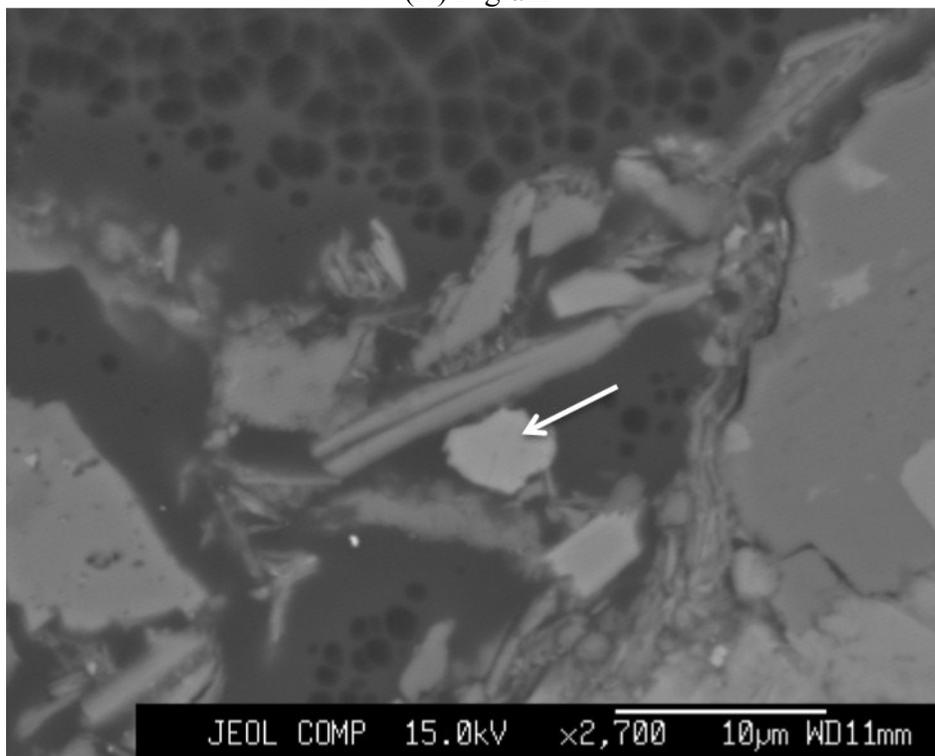


Figure C-1 Back scattered electron images and microprobe compositional analysis of representative P solids from LFSF1 (fine grained) from near the infiltration pipe at a depth of 0.35 – 0.85 m: (a) grain coating with two representative analysis, one for each location shown; (b) grain coating with two representative analysis in same location; (c) grain coating with two representative analysis, one for each location shown; (d) grain coating with two representative analysis, one from each grain as shown; (e) grain coating with two representative analysis from each location shown

(A) P-grain



(B) P-coating

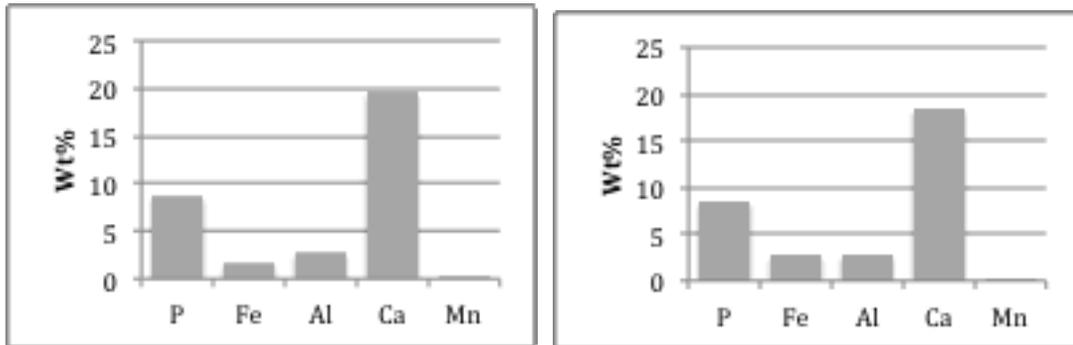
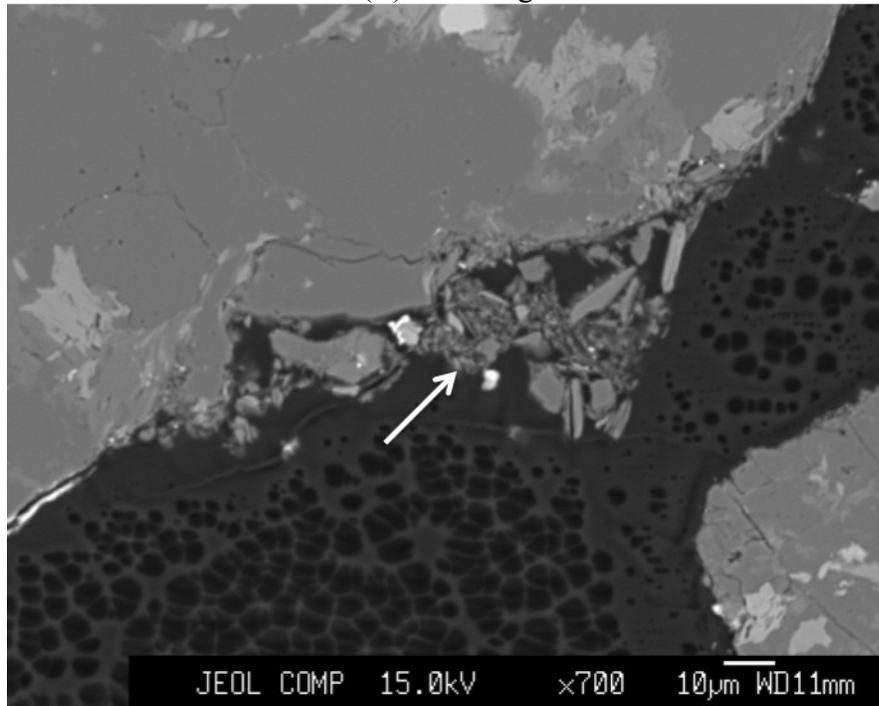
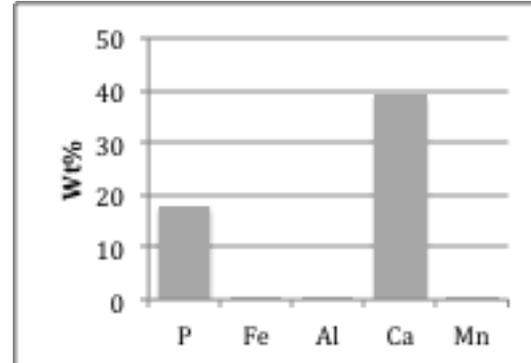
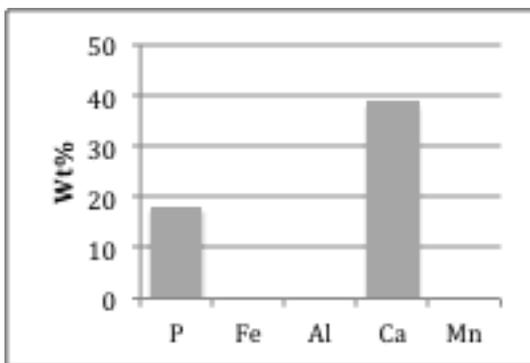
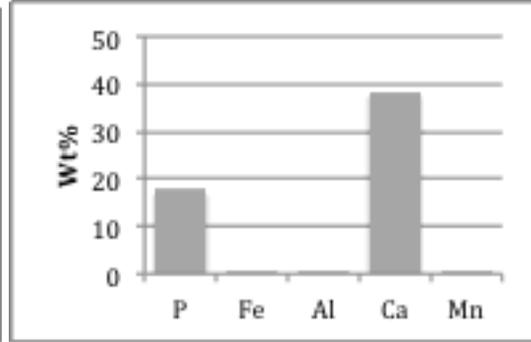
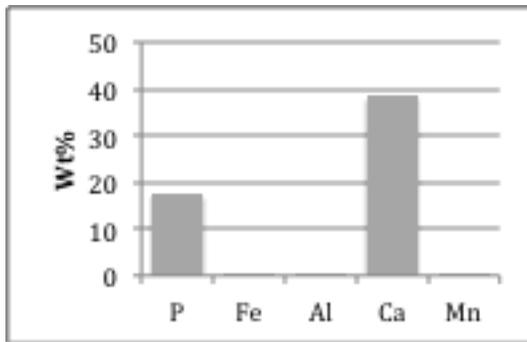
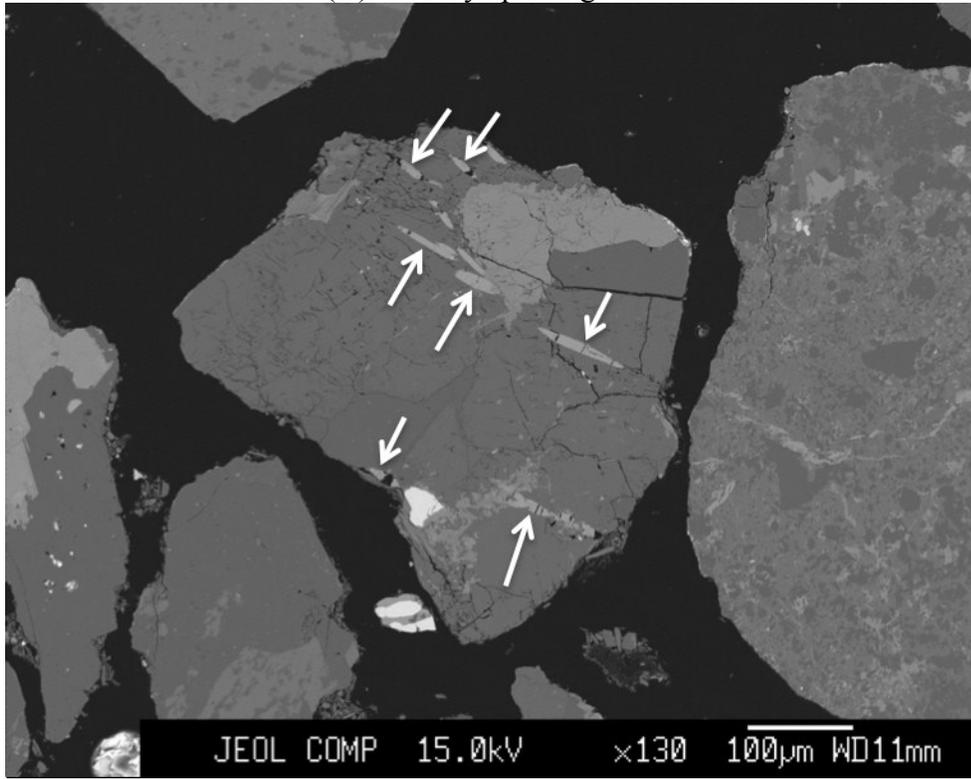


Figure C-2 Back scattered electron images and microprobe compositional analysis of representative P solids from LFSF2 (medium grained) from near the infiltration pipe at a depth of 0.82 – 1.18 m: (a) unattached P-grain with one analysis; (b) grain coating with two representative analysis

(A) Primary apatite grains



(B) Primary apatite grains

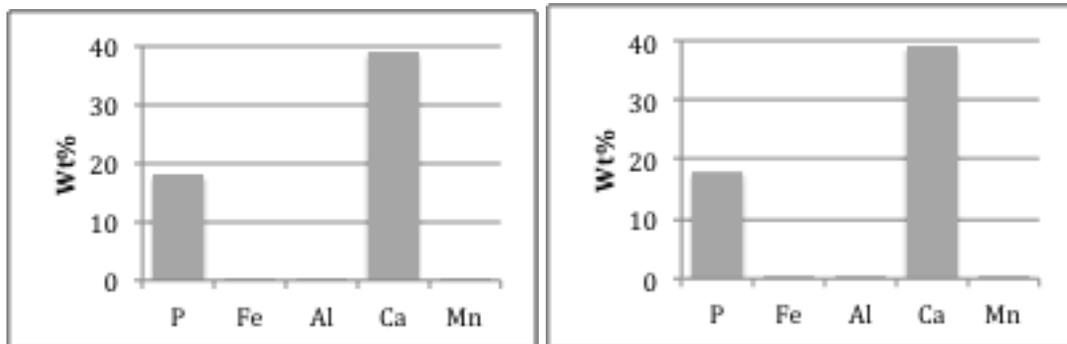
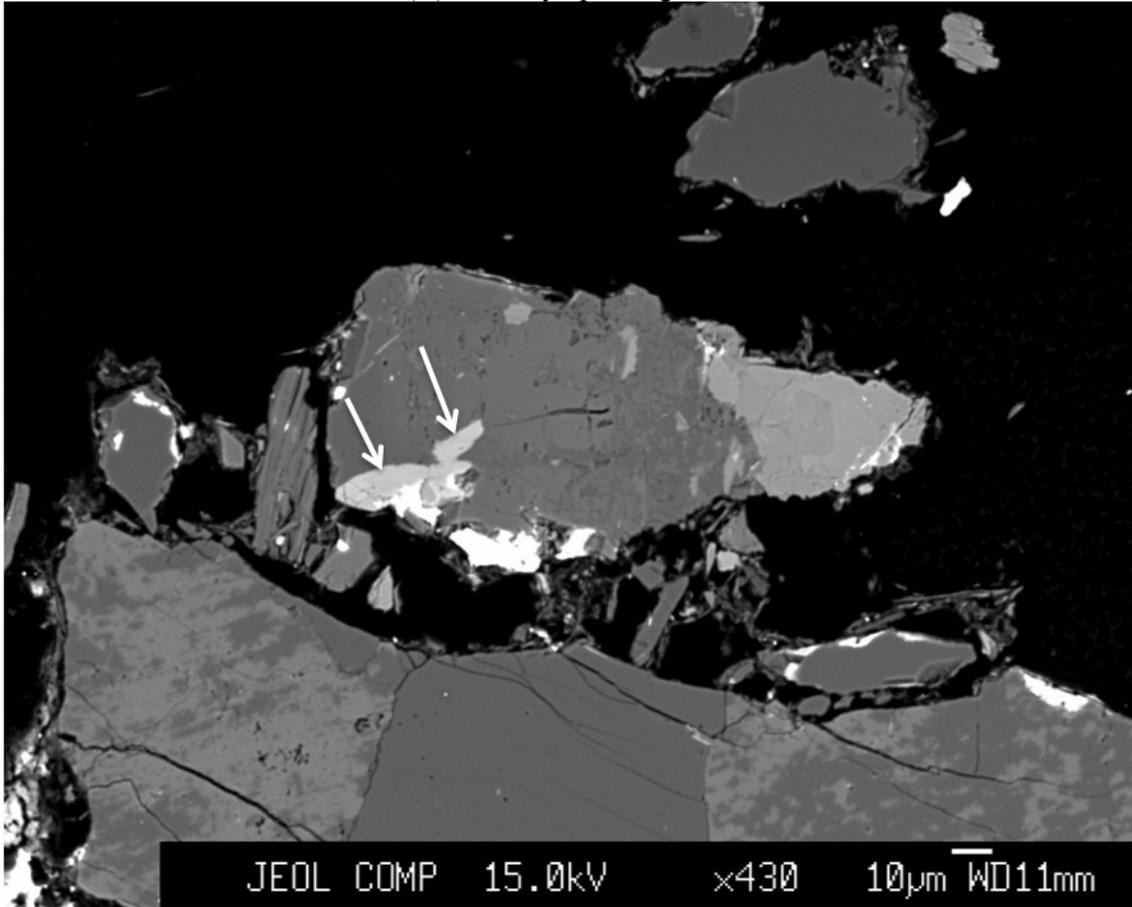


Figure C-3 Back scattered electron images and microprobe compositional analysis of representative primary P grains from LFSF2 (medium grained) 2.5m away from the infiltration pipe at a depth of 0.90 – 1.17 m: (a) primary apatite grains with four representative analysis; (b) primary apatite grain with two representative analysis, one in each location

APPENDIX D Eh/pe-pH Diagrams of Phosphates

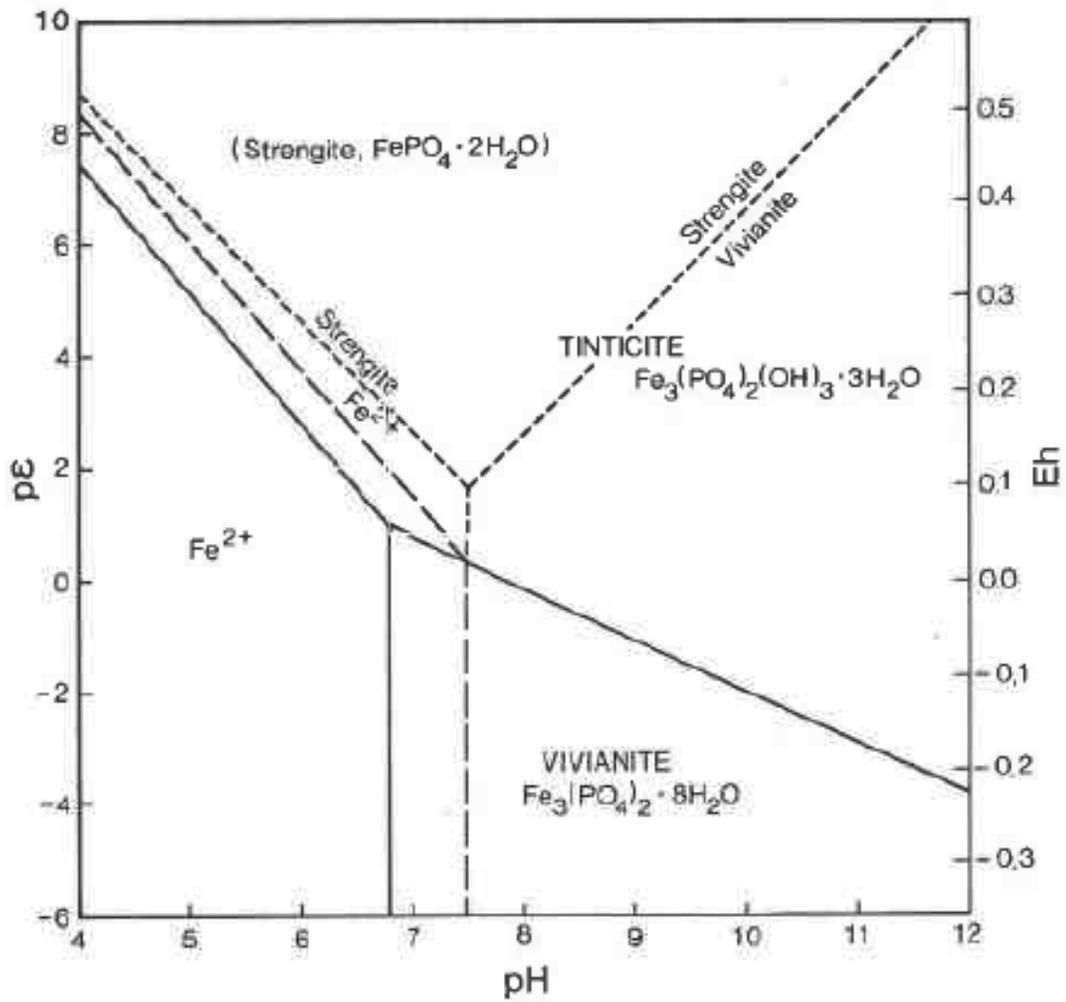


Figure D-1 Eh/pe-pH diagram for ferric and ferrous phosphates at an activity of dissolved phosphate of 10^{-6} . Dashed line represents the stability field for strngite. Solid and broken lines represent the boundaries at activities of dissolved iron species of 10^{-4} and 10^{-5} , respectively (Nrigu and Dell, 1974).

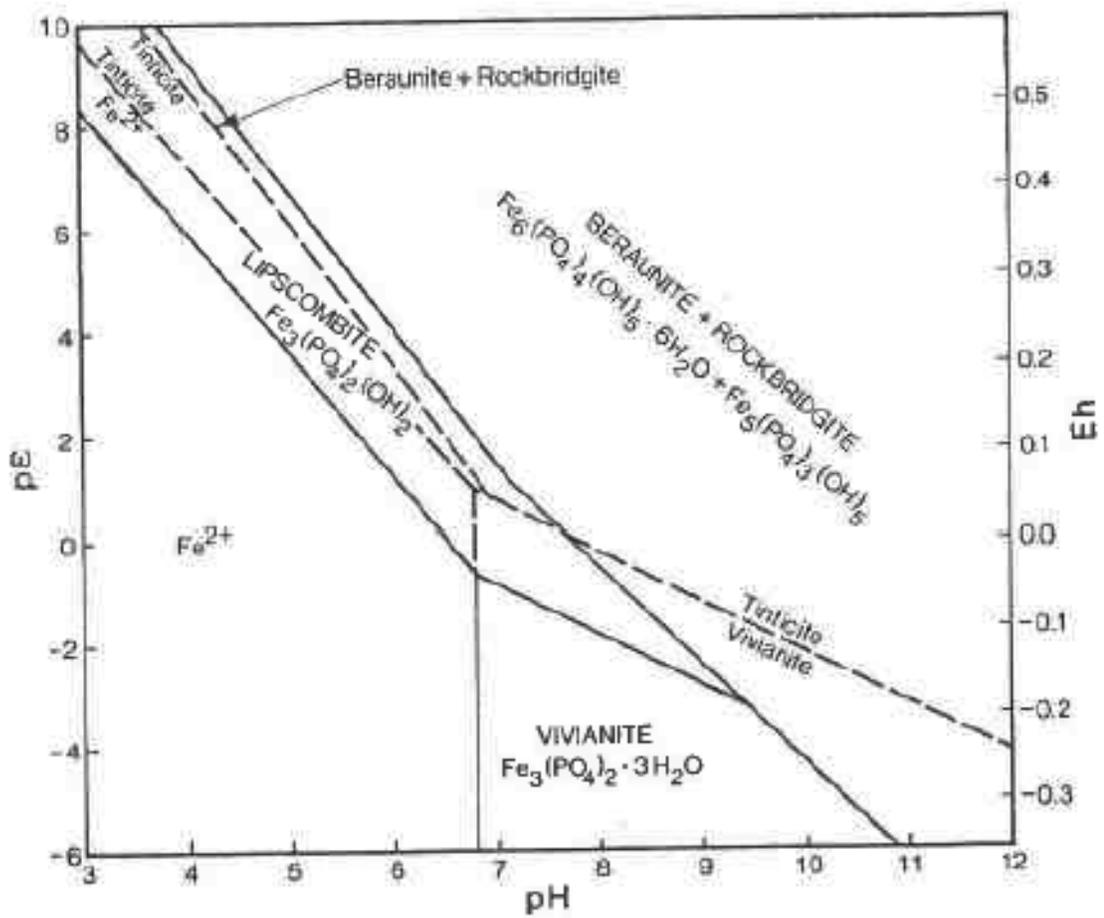


Figure D-2 Eh/pe-pH diagram for ferrosferric hydroxyphosphates at phosphate and dissolved iron activities of 10^{-6} and 10^{-4} , respectively. Dashed line represents the boundaries for tenticite or cacoenite (Nriagu and Dell, 1974).

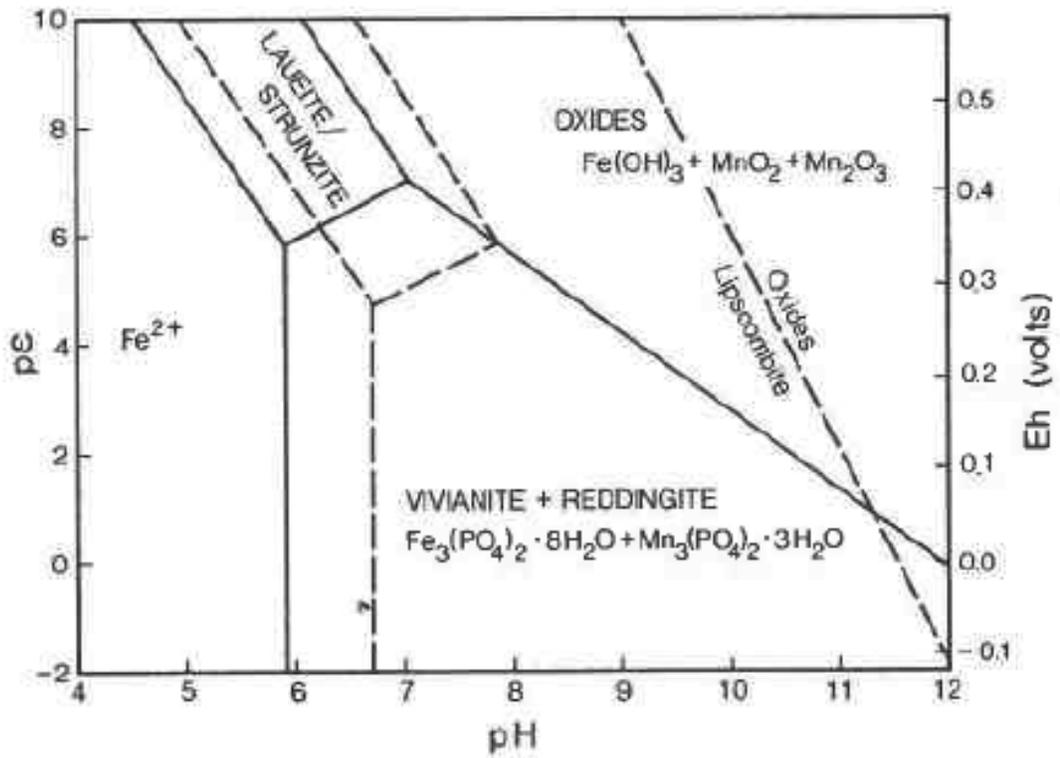


Figure D-3 Eh/pe-pH diagram for ferromanganese phosphates at dissolved phosphate and iron activities of 10^{-6} and 10^{-4} , respectively. Dashed line represents the boundaries for dissolved manganese activity of 10^{-4} (Nriagu and Dell, 1974).

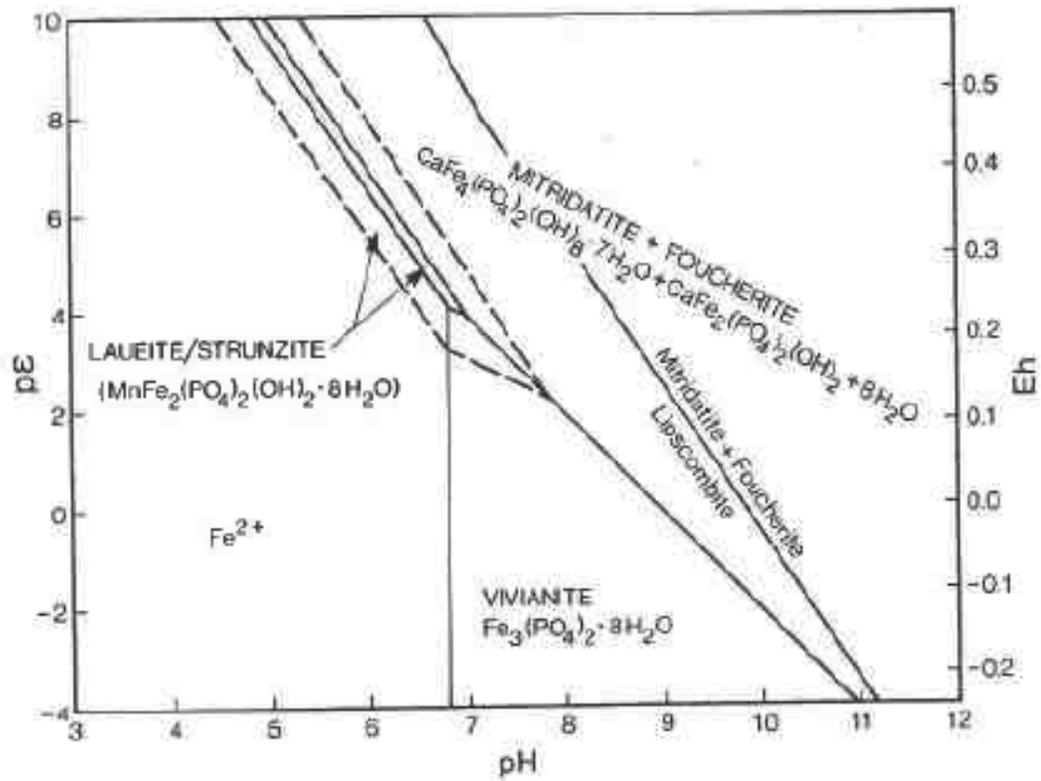


Figure D-4 Eh/pE-pH diagram for ferromanganese and calcite-iron phosphates at dissolved phosphate, iron, and calcium activities of 10^{-6} , 10^{-4} and 10^{-3} , respectively. Dashed line represents the boundaries for dissolved manganese activity of 10^{-2} (Nriagu and Dell, 1974).

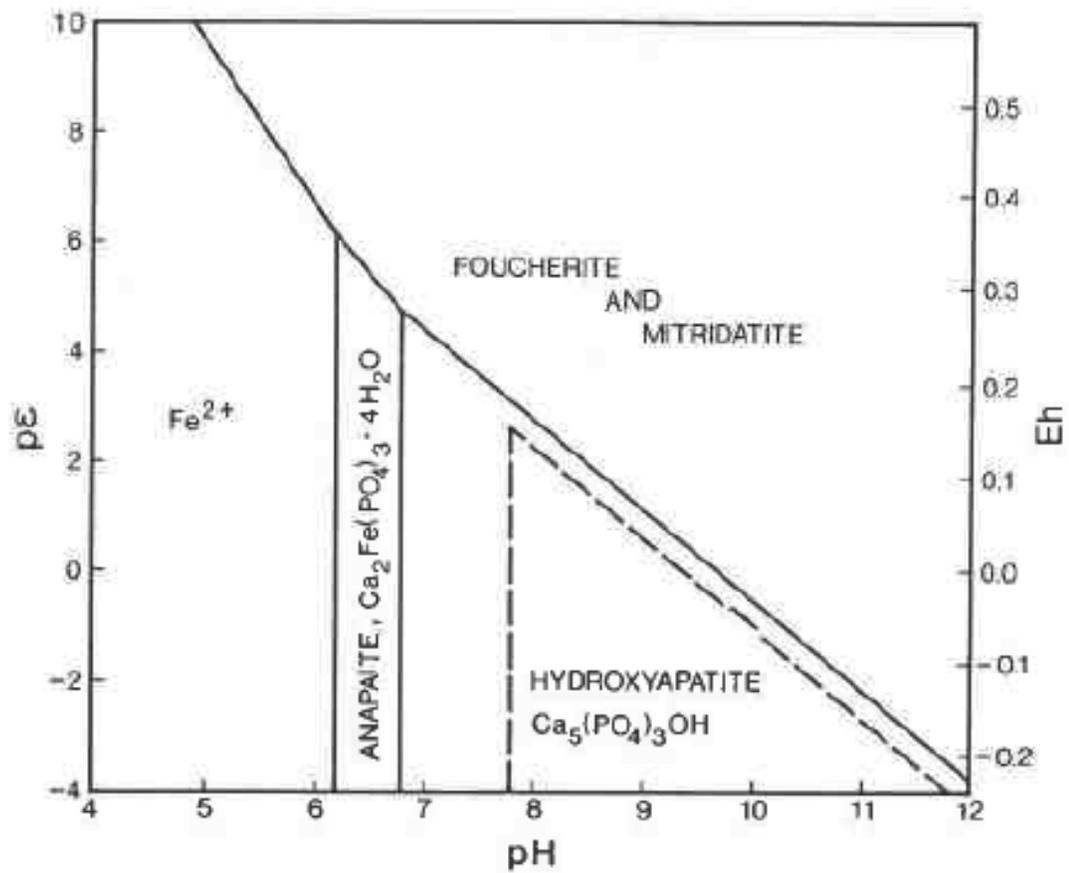


Figure D-5 Eh/pe-pH diagram for calcium and iron phosphates at dissolved phosphate, iron, and calcium activities of 10^{-6} , 10^{-4} and 10^{-3} . Dashed line represents the boundaries for hydroxyapatite when $A_{Ca^{3+}}=10^{-4}$ (Nrigu and Dell, 1974).