Edge of Field Phosphorus Export via Tile Drainage and Overland Flow from Reduced Tillage Systems in Ontario

by

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Author’s Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions accepted by my examiners. I understand that my thesis may be made electronically available to the public.

Abstract

This study examined the role of tile drainage and overland flow in field-scale phosphorus (P) export from reduced tillage systems, as well as the influence of event type and antecedent conditions on P export during major runoff events. Three field-scale sites representing a range of soil, climatic, and management conditions, were monitored intensely for an 18 month period. Annual P export from the sites ranged between 0.267 and 0.419 kg ha^{-1}. The non-growing season (NGS) was an important period for P export due to the volume of discharge during the period. Tile drainage contributed the majority of combined annual discharge at all sites (78-83%). Tile drainage was an equal or dominant contributor to annual total P (TP) export. Overland flow was the dominant transport pathway for soluble reactive P (SRP) at two of the three sites. The nature of the discharge events (e.g. rain on soil, rain on snow, and radiation melt) influenced P speciation in runoff. Particulate P + soluble unreactive P (PP+SUP) concentrations were highest during events where rain fell on bare soil. The proportion of TP as SRP in major events appeared to decline over the NGS. Understanding the seasonality of P export, the relative role of tile drainage and overland flow, and the influence of event type will improve our ability to manage non-point source P export.

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1.1 Introduction

The environmental impacts of agriculture have long been a topic of public concern (Skinner, et al. 1997). Common agricultural practices such as subsurface tile drainage, tillage and nutrient applications have been associated with a range of problems including; habitat destruction, soil degradation and water quality issues (Foley, et al. 2011, Stoate, et al. 2009). There has been a concerted effort among agricultural producers, public organizations, academia and various levels of government to address these concerns, leading to the recommendation of Beneficial Management Practices (BMPs). However, the management of agricultural impacts is complicated, and although there have been notable improvements, there is need for a continued effort to improve or refine BMPs (McRae, et al. 2000). The re-emergence of algal blooms in Lake Erie has raised concern about phosphorus (P) pollution in the Great Lakes and St. Lawrence River basins (Joosse and Baker 2011). Previous conservation efforts in the 1970's, which addressed point and non-point sources of P were successful in reducing total P (TP) export (Joosse and Baker 2011, Michalak, et al. 2013, Scavia, et al. 2014). Improving urban waste water treatment was a major factor, but, on the rural side, the adoption of no-till (NT) farming, and reduction of P inputs to fields was credited with part of the success (Joosse and Baker 2011, Michalak, et al. 2013, Scavia, et al. 2014). However, in the mid-1990's levels of dissolved P exported to Lake Erie began to increase and have since exceeded 1970's levels (Great Lakes Commission Phosphorus Reduction Task Force 2012, Scavia, et al. 2014). Of particular importance, the increase in P has come from an increase in the bioavailable form of P, known as soluble (or dissolved) reactive phosphorus (SRP) (Great Lakes Commission Phosphorus Reduction Task Force 2012, Scavia, et al. 2014), which has the potential to significantly increase algal production in aquatic systems.

The occurrence of algal blooms along the shores of the Great Lakes is being driven by the interaction of a number of factors including ecosystem changes, changing land management practices, and climate change (Auer, et al. 2010, Joosse and Baker 2011, Scavia, et al. 2014). Since the mid-1990's, the population of invasive dreissenid mussels have increased in the Great Lakes. This has resulted in increased clarity of the water, and has sped up the recycling of P found in particulate forms in the near shore environment. These ecosystem changes have contributed to excessive *Cladophora* growth in areas of the Great Lakes because the system is now more sensitive to P loading (Auer, et al. 2010).

The Phosphorus Reduction Task Force has identified changes in tillage practices as a likely contributor to the increase in dissolved P loading (Great Lakes Commission Phosphorus Reduction Task Force 2012). Reducing the intensity and frequency of tillage is often promoted as a BMP for reducing erosion and improving soil health. However, the use of these systems has also been shown to increase SRP and TP losses at many sites (Tiessen, et al. 2010, Gaynor and Findlay 1995, Hansen, et al. 2000, Puustinen, et al. 2007), both in surface overland flow (Tiessen, et al. 2010) and tile drain effluent (Gaynor and Findlay 1995). The percentage of the total cropland acres seeded using NT and reduced tillage (RT) practices has been increasing in Ontario, moving from 21% in 1996, to 56% in 2006 and 63% in 2011 (Statistics Canada 2007, Statistics Canada 2013). Similar trends have been observed in watersheds on the USA side of the Great Lakes. This change in land management coincides with the increase in SRP export to Lake Erie and has caused managers to question the use of NT or RT as a BMP (Kleinman, et al. 2011, Michalak, et al. 2013, Smith, et al. 2014) The undesired consequence of RT demonstrates the complexity of managing the environmental impacts of agriculture. It is accepted that controlling soil erosion and conserving soil should remain a priority, as erosion can lead to severe soil degradation and nutrient losses if not addressed (Hansen, et al. 2002). However, it is also recognized that there is further need to improve conservation-type tillage systems (NT and RT) to reduce P loading, while maintaining the demonstrated benefits of these systems.

The Great Lakes Commission Phosphorus Reduction Task Force has also noted changes to fertilization practices as a contributing factor. Although there is limited data on the subject, regional observations are that some larger farms have shifted to applying fertilizer in the fall instead of during the spring time at planting, as is common for smaller operations (Great Lakes Commission Phosphorus Reduction Task Force 2012). In these NT/RT systems, the fall applied P is surface broadcast and often not incorporated. This apparent shift in management practices is potentially significant in terms of P export for two reasons: 1) surface applied applications tend to stratify nutrients near the surface and increase the long-term risk of losses (Sharpley 2003), and 2) applications made prior to periods which are more susceptible to large runoff events can significantly increase the risk of P losses associated with fertilizer on the surface (Allen and Mallarino 2008, Smith, et al. 2007). The timing and method of P applications used in some RT systems may be a problem from a P export perspective. The Phosphorus Reduction Task Force has identified the need to improve fertilization practices as one of the management priorities for reducing P loading. They state that in priority watersheds, all P applications should be made below the surface, or immediately incorporated in a “non-erosive manner”. This priority recognizes the benefits of reduced tillage practices in controlling erosion, yet identifies the need to further improve the management practice by reducing the stratification of nutrients near the surface.

Changes to the Great Lake’s climate have been identified as another potential source of the observed increases in P export. Increased frequency of large-magnitude (or extreme) events, in particular more rain on snow and melt events during the non-growing season (NGS) could be contributing to changes in P loading (Burkitt, et al. 2011). Major flow events have the capacity to export large amounts of P from fields. If large events occur shortly after nutrient applications, there is an elevated risk of SRP losses (Gentry, et al. 2007). It is probable then, that the interaction between changing hydro-climatic drivers, (e.g. more extreme events during the NGS), and the evolution of management practices (i.e., no-till and reduced tillage, and more fall, non-incorporated applications) could lead to increases in SRP loading (Great Lakes Commission Phosphorus Reduction Task Force 2012, Scavia, et al. 2014).

In cold regions, annual P export is typically dominated by NGS losses (Ontario Ministry of the Environment: Environmental Monitoring and Reporting Branch 2012, Penn, et al. 2010, Macrae, et al. 2007). Thus the NGS is a critical period in which to evaluate the efficacy of BMPs. Despite this, there has been limited evaluation of BMPs over the NGS, in part, due to the difficulty in monitoring during this period (Penn, et al. 2010). Improving our understanding of this period will become increasingly important as resource managers plan to adapt BMPs for climate change. As the climate in Ontario changes, so will the NGS runoff patterns, ground conditions, and the resulting P export. In order to improve the performance of reduced tillage systems, we need to monitor these systems year round to understand how they perform under a range of NGS conditions.

One additional major area of uncertainty is the pathway through which runoff and P leave fields. P export was historically considered a problem related to overland flow, though numerous studies have now shown that tiles can be an important source as well (e.g., (Gaynor and Findlay 1995, Dils and Heathwaite 1999, Smith, et al. 2014). Since tiles increase connectivity within a watershed, not understanding the relative contribution of these pathways makes it difficult to identify key contributing areas within a watershed. P-Indexes are a widely used tool for identifying the risk of P losses at the field scale throughout the world including Ontario (Buczko and Kuchenbuch 2007, Reid, et al. 2012). However, the current Ontario P-index does not account for tile drainage in a meaningful way. In order to include tile drainage, which may account for a considerable proportion of field runoff in Ontario (e.g. Macrae et al., 2007) there is need for increased understanding of the relative contribution of tile drainage to total site losses (Reid, et al. 2012).

In summary, there are several key gaps in our understanding of P export from agricultural systems and these affect our ability to strategically target appropriate P management efforts. Firstly, we need to understand when P is leaving fields, and the transport pathway it leaves in (Reid, et al. 2012). To do this properly, surface and subsurface P transport pathways must be monitored simultaneously, and furthermore, monitoring must continue year round to accurately estimate export (Miles, et al. 2013). Another complicating factor is the influence of tillage systems on P export, as this has been shown to vary based on region (Tiessen, et al. 2010), and is further complicated by the impact the chosen system can have on nutrient management practices. So despite a significant body of research on P export, questions remain about the role of different transport pathways and the influence of seasonality and land management practices. Further study will help improve our ability to assess the risk of P loss at the field-scale, and thus better target conservation efforts to priority areas (Pionke, et al. 2000).

1.1.1 Objectives

This study has the following objectives:

1) Using three field scale sites, examine runoff and P export (Total, Soluble Reactive, and Particulate+Soluble Unreactive) in tile drainage and overland flow from reduced tillage fields in Ontario, for a minimum of one full year (12 month period) to:
    a. Quantify seasonal and annual discharge from tile drainage and overland flow at the field scale;
    b. Quantify the seasonal and annual speciation and mass of P loss in tile drainage and overland flow; and,
2) Demonstrate the significance of peak discharge events in P export for a 12 month period at two sites, through the
    a. Identification of the seasonal distribution of peak discharge events;
    b. Determination of the contribution of peak events to annual P export;
    c. Determination of how event type (climate drivers and pre-event (antecedent) ground conditions) influence the event export and speciation (Total, Soluble

Reactive, and Particulate+Soluble Unreactive) during successive peak discharge events in the NGS.

1.1.2 Thesis Organization

This thesis is organized into six chapters. Chapter 1 provides an introduction to the study and presents the thesis objectives. Chapter 2 provides a literature review focused on P cycling and describes factors that influence P export. Chapter 3 outlines the methodology used for the entire study including, study overview, instrumentation, sampling procedures, lab analysis and data analysis. Chapters 4 and 5 address the separate objectives and are intended to form the basis of two separate manuscripts. Each of these chapters has a separate literature review and methods section where further explanation is required. Chapter 4 addresses Objective 1, using three field scale sites for an 18 month period. Chapter 5 addresses Objective 2, and focuses event variability in P export observed over the NGS from two sites. Chapter 6 provides the overall summary and conclusions. The Appendix provides supplementary data including rating curves and photographs.

2 Literature Review

The purpose of this literature review is to provide an overview of P cycling in agricultural systems, explain how P is exported from fields via tile drainage and overland flow, and identify hydro-climatic drivers, and management factors which influence P export from these systems.

2.1 Phosphorus Dynamics in Landscapes

The cycling of P occurs at different temporal scales. At the broadest timescale, the cycling of P begins with the dissolution of P from rocks containing phosphate, and ends when P is transferred by water to the ocean where it is deposited. The uplifting of rock begins the cycle again (Schlesinger and Bernhardt 2013). Given the length of this cycle, P is not considered to be a renewable resource. However, en route to oceans, P is cycled between soil, water and organisms on much shorter timescales (Schlesinger and Bernhardt 2013). Understanding P cycling at these shorter timescales is important to agricultural production and in understanding P export to surface water. The following is review of field-scale P cycling including the movement of P between different pools, P inputs, and export pathways (Figure 2-1).

2.2 Phosphorus Pools and Sources

Topsoil contains between 100 to 3000 mg P kg^{-1} (Frossard, et al. 1995, Frossard, et al. 2000). Phosphorus exists in two broad pools within soil, solid state soil P and soil solution P, with the majority of P being found within the solid state. Phosphorus within the solid state is found in both organic and inorganic species. Inorganic P can be found in several different pools of varying plant availability: surface sorbed P, as secondary P minerals including calcium, iron, or aluminum phosphates, or as primary P minerals such as Apatites (Zaimes and Schultz 2002). The organic P component can make up between 30-65% of total P within topsoil P (Frossard, et al. 1995), and is found within soil biomass, organic matter and decaying plant residues. Microbial P, which is included in soil biomass, accounts for 3-24% of total organic P (Frossard, et al. 1995). Organic P in the soil matrix is found as phytic acid or inositol hexaphosphates, phosphate diesters, phosphonates, and polyphosphanates (Zaimes and Schultz 2002) and this P is not considered immediately available for plants. Solid state P in soil is found in both inorganic and organic pools, with varying degrees of availability to plants.

A smaller portion of P is found in the soil solution, yet this is an important pool as plants can only obtain P from the soil solution. Phosphorus in the soil solution is found mostly as orthophosphate anions. The species of orthophosphate present in solution is pH dependant, with $H_2PO_4^-$ (pH 4-7.2) and $HP0_4^{2-}$ (pH>7.2) being the most prevalent in pH 4-9 (Holtan, et al. 1988, Pierzynski, et al. 2005). The concentration of P in the soil solution is low because orthophosphate anions are easily removed from solution into solid state forms (sorbed P and secondary P) (Pierzynski, et al. 2005). The soil solution P, is a small percentage (<1%) of total soil P and must be replenished to supply crops with their P requirements (Frossard, et al. 1995, Frossard, et al. 2000). P in the soil can be described as being in a dynamic equilibrium, where P can move between various pools, rather than being in static, isolated pools (Frossard, et al. 2000). The processes controlling these transfers will be discussed later in this chapter.

There are internal and external sources of P to soil. P from these sources can be retained by soil, used by vegetation or be lost in runoff (Figure 2-1). One internal source of P is P that originates from the dissolution of primary and secondary P minerals (Pierzynski, et al. 2005). Another internal source is P in plant material as it can be leached from plant residues, or released during decomposition or through mineralization. Following dissolution, leaching or mineralization, the released P is available for plant uptake or can quickly react with other soil components (Pierzynski, et al. 2005).

In agricultural systems, there are also external sources of P (i.e., fertilizers, manures, and other soil applied amendments) used to maintain P levels sufficient for crop production. The sources available vary in total P content, inorganic P content and solubility (Shigaki and Sharpley 2011, Smil 2000). The most commonly applied form of commercial P fertilizer in Ontario is Monoammonium phosphate (MAP). MAP contains 90-100% water soluble P, and is 100% inorganic. Livestock manure is also used as a P source. A large portion of the P in livestock manures is present as inorganic P, while a portion is organic P and must be mineralized before it can be taken up by crops (Bundy, et al. 2005). The rate of mineralization depends on many factors including soil moisture, temperature and the C:N ratio of the manure (Bundy, et al. 2005). This variability makes it more difficult to match the plant availability of P to plant uptake when using manure as a P source (Bundy, et al. 2005). There are a range of external P sources used in agricultural systems with varying properties which influence their management and crop benefit.

Soil P fractions respond to inputs of P from external sources. Initially applications result in increased P concentrations in the solution P pool. These levels decline as P is removed from solution into reactive and more stable forms of solid state P (Pierzynski, et al. 2005). If P applications are in excess of crop removal, soil test phosphorus (STP) can be built up above levels required for crop production (Pote, et al. 1996). If built up, it can take many years without P applications to lower STP to acceptable levels (Dodd and Mallarino 2005). The annual reduction of STP is less than the crop removal rate because of the movement from less available forms of P to more available P. Applications of P must be managed according to crop removal rates otherwise P levels can become elevated (Sharpley, et al. 2001).

2.3 Internal Cycling of Phosphorus

The movement of inorganic and organic P between the solution and solid phase, as well as between various solid phases, is seen as a continuum and is controlled by both abiotic and biotic processes (Frossard, et al. 2000).

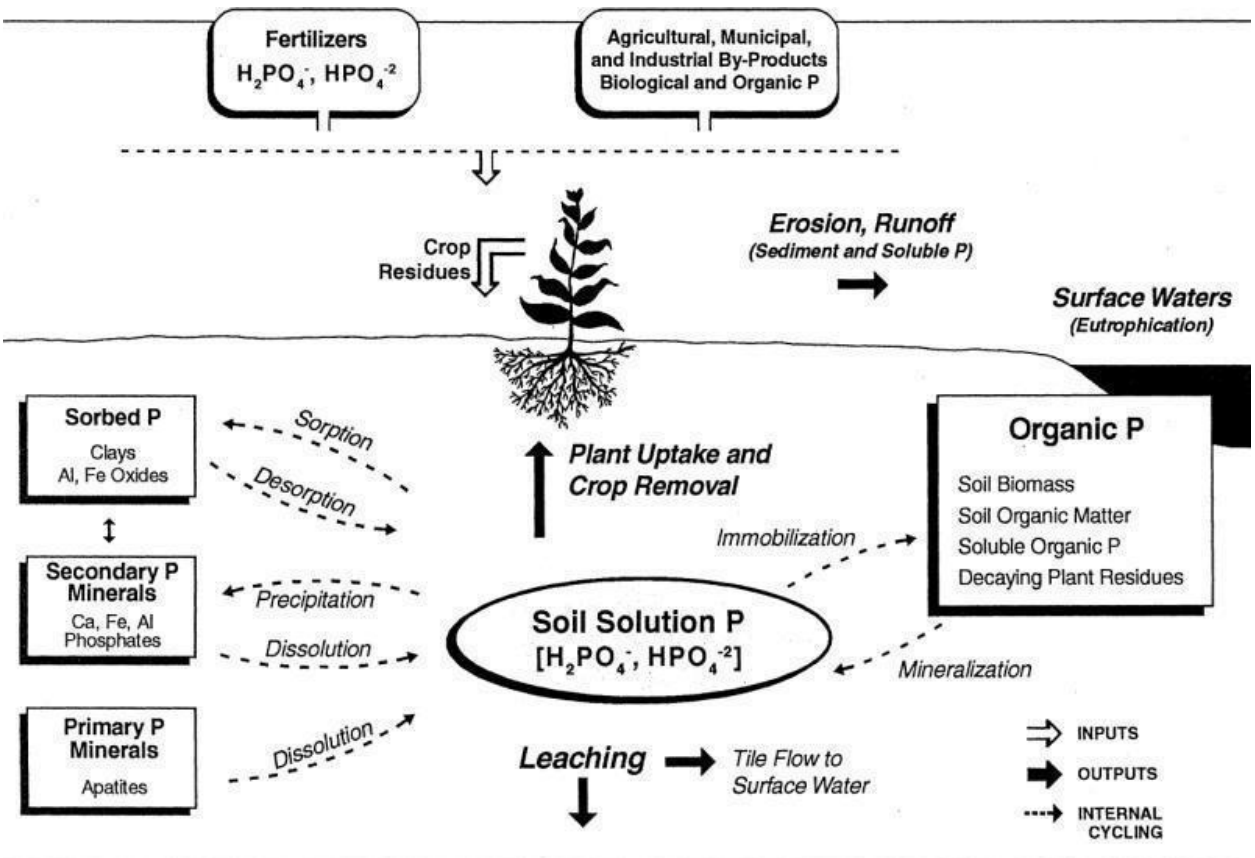

Figure 2-1 The phosphorus cycle (Pierzynski, et al. 2005).

2.3.1 Abiotic Processes

Abiotic processes are responsible for the transfers between soil state P and soil solution P, as well as transfers between various solid state pools. These transfers are governed by sorption – desorption and precipitation – dissolution processes (Figure 2-1).

2.3.1.1 Sorption and Desorption

Sorption removes phosphate from the solution pool, limiting P available to plants. Sorption is a general term referring to the short and long-term removal of P from soil solution and includes both the adsorption of P molecules to surfaces of soil constituents, and the absorption of P molecules into a substance by solid or liquid state diffusion (Frossard, et al. 1995, Holtan, et al. 1988, Pierzynski, et al. 2005). P in solution sorbs to Iron (Fe) and Aluminum (Al) oxides/hydroxides, clay minerals, calcium carbonate, magnesium carbonate and humic compounds (Holtan, et al. 1988, Frossard, et al. 1995). The initial sorption reaction occurs by ligand exchange where an OH^- or a H_2O molecule in the solid phase is replaced by a phosphate molecule forming a phosphate surface complex (Frossard, et al. 1995). This can be followed by subsequent absorption reactions rendering the P less available. The amount and rate of sorption is influenced by multiple factors including Fe, Al, Ca, Mg content, soil pH, clay content, organic matter, and phosphorus saturation percentage which are discussed further below.

The ability of soil to buffer inorganic P is strongly related to Fe and Al oxide content (Frossard, et al. 2000, Gburek, et al. 2005). Fe and Al oxides have the strongest influence on P sorption in low pH soils, while Ca and Mg become important in neutral and alkaline soils (Frossard et al. 1995). Fe and Al oxides do maintain a strong influence on initial sorption reactions in calcareous soils, but calcium carbonates govern the long-term sorption (Frossard et al. 1995).

Solution pH determines the charge of the Al and Fe hydroxides which affects P sorption. When the pH is high, H^+ is disassociated from the Al and Fe oxides resulting in a negative surface charge. In contrast, at low pH, Fe and Al oxides bind H^+ resulting in a positive surface charge. Since orthophosphate is an anion, there is decreasing P sorption with increasing pH, thus higher sorption at lower pH (Mulder, et al. 1994, Holtan, et al. 1988, Frossard, et al. 1995)

The sorption capacity of a soil is influenced by soil texture. Clay content is positively correlated with increasing P sorption capacity (Holtan, et al. 1988). There are more binding sites in clay soils, as phosphate will sorb to either the negatively charged surface of clays, or to the edge of an Al layer (Frossard, et al. 1995). In low concentrations P is sorbed to the broken edge of clay lattices by replacing water molecules, but at higher concentrations P can exchange with hydroxyl groups or displace structural silicate (Holtan, et al. 1988).

The influence of organic matter on P sorption is not always clear. Organic matter can sorb phosphate when associated with Fe, Al and Ca cations. However, it can also block available sorption sites (Holtan, et al. 1988, Frossard, et al. 1995). Organic compounds can also prevent reaction between metals and phosphate by chelating metals (Frossard, et al. 1995). Not surprisingly, the effect of organic anions differs depending on soil conditions. For example, Braschi et al. (2003) found that the addition of organic matter reduced the amount of P sorbed to soil at high solution P in calcareous soils.

The rate of P sorption, or the ability of soils to buffer P in solution, changes with increasing P sorption or higher P saturation. As the amount of adsorbed P increases, the rate of sorption decreases because the adsorption of P anions increases the negative charge of the adsorbing surface (Barrow 1978).

Although orthophosphate anions are readily removed from solution, these sorption reactions are reversible (Barrow 1983). Phosphorus can re-enter soil solution through desorption if there is a reduction in the soil solution P concentration or if other competing anions are introduced into solution (Hinsinger 2001). The quantity of desorbable P is related to the amount of amorphous Fe and Al oxides in soil and their P saturation (Frossard, et al. 2000). At higher P saturations, there is a greater rate of P desorption. Lookman et al. (1995) described P desorption kinetics using two pools, an ‘easily desorbable pool’ and a ‘slowly desorbable pool’. The slowly desorbing pool was larger than the quickly desorbing pool, and continued to release for a substantial amount of time suggesting all the P in this pool is desorbable. The reversibility of the initial sorption reaction means that soil P is not static, and it can move from less available pools to available pools throughout a growing season (Frossard, et al. 2000).

2.3.1.2 Precipitation - Dissolution

Phosphates are also removed from solution by precipitation processes. Phosphate can form precipitates with metal cations in solution, primarily Ca, Fe and Al (Frossard, et al. 1995, Hinsinger 2001). Soil pH regulates which cations will be dissolved in solution and available to form precipitates (Hinsinger 2001). In calcareous soils, calcium phosphates will form after phosphate sorption to calcite. After the initial sorption to calcite a series of reactions takes place leading to the development of hydroxyapatite (Frossard, et al. 1995). In acid soil Al and Fe phosphates are precipitated, usually as amorphous Al and Fe – P compounds (Frossard, et al. 1995). Organic acids found in fertilized soils can inhibit crystal growth which works to keep phosphate in more available forms (Frossard, et al. 1995).

P in secondary and primary minerals can re-enter solution following dissolution under the appropriate conditions. The dissolution of calcium phosphate (e.g. Apatite) requires a source of H^+ and sinks for P and Ca (Frossard, et al. 1995). The rate of dissolution of primary and secondary minerals is influenced by the morphology and rate of substitution within the crystal lattice (Frossard, et al. 1995). Phosphate released during dissolution is then available for other reactions within the environment and is often quickly immobilized into insoluble forms (Smil 2000).

2.3.2 Biotic Processes

The cycling of P between inorganic and organic forms is controlled by biotic processes of immobilization and mineralization (Figure 2-1) (Condron, et al. 2005). One way inorganic P in solution can be immobilized is through plant uptake (Condron, et al. 2005). When this occurs a portion of P removed from solution is synthesised into organic P; however, a portion can remain as inorganic P. As mentioned above, crop residues which are left on fields following harvest are an internal source of P to the system. Some of this P is transferred quickly to solid state inorganic P pools, while some may remain within the organic P pool for a greater length of time. As plants decay, soluble inorganic P, which can make up between 15-50% of total plant P, will quickly sorb to soil (Frossard, et al. 1995). Organic P within decaying plant material can also be released to soil solution, but this transfer involves the process of mineralization by microorganisms. Microorganisms play an important role in the internal P cycle by moving organic P from soil pools into plant available forms (Frossard, et al. 2000, Richardson, et al. 2009). Though microorganisms can release P, they also require P for their growth and development and can compete with plants for P (Richardson and Simpson 2011). For example, inorganic P is also removed from solution during microbial decomposition of residues with higher C:P ratios (Condron, et al. 2005). The majority of organic P in residues is quickly taken up by decomposers and is retained in microbial biomass for over one year in temperate cultivated systems (Frossard, et al. 1995). The turnover of P through immobilization-mineralization processes can be an important part of P requirements for crops (Condron, et al. 2005).

2.4 Forms and Pathways of Phosphorus in Runoff

In most cases, the largest flux of P from the system occurs when P in harvested grain is removed from the site. Mallarino et al. (2011) found that over an 11 year period, the average 10.67 mt ha^{-1} corn harvest removed 30 P kg ha^{-1} y^{-1} from the system. The removal of P in harvested crops demonstrates the need to fertilize with external sources of P to replenish removed P. While crop removal of P mentioned above is a significant removal pathway, the greatest risk to water quality is the P which is removed via subsurface and overland flow (Withers, et al. 2005). These pathways will be the focus of this chapter.

2.4.1 Forms of Phosphorus in Runoff

P in runoff is found in a variety of forms. Inorganic and organic species of P are found in both particulate and soluble forms in water (Jarvie, et al. 2002). The different fractions of P measured in runoff are operationally defined, and actually measure a combination of P species (Jarvie, et al. 2002). There are a range of fractions reported in the literature, which vary somewhat depending on the study's goals. Common fractions include: Total P (TP), Total Dissolved P (TDP), Soluble Reactive P (SRP), Soluble Unreactive P (SUP) and Particulate P (PP). The distinction between particulate and dissolved/soluble fractions is based on filtration, commonly using a 0.45 µm membrane filter, though filter sizes can vary (Jarvie, et al. 2002). However, it should be noted that P is found associated with a range of colloids which would not be removed with the standard filter size (Jarvie, Withers and Neal 2002). The Total Dissolved Phosphorus fraction (TDP or DP), is a combination of dissolved hydrolysable phosphorus, also called Soluble Unreactive Phosphorus (SUP), and SRP (TDP=SRP+SUP). SUP is commonly considered an estimate of soluble organic P (Condron, et al. 2005), however others have noted that this is an over simplification (Anderson and Magdoff 2005). SRP is considered orthophosphate in solution. The PP fraction is calculated as the difference between TP and TDP (Jarvie, et al. 2002). Some studies report PP as the difference between TP and SRP. Total P consists of all the above mentioned fractions and species of P. The different fractions vary in their bio-availability. SRP is considered to be readily available to plants. While a portion, typically 20%-30%, of the PP fraction is considered bio-available (Hansen, et al. 2002, Poirier, et al. 2012). Biologically Available Phosphorus (BAP) is estimated separately from the above fractions as it is composed of a variety of P species from dissolved and particulate fractions (Jarvie, et al. 2002). P forms in water are measured using operationally defined fractions. It is important to clearly define how the fractions reported are measured as authors often use different estimations (e.g. PP=TP-SRP vs PP=TP-TDP).

2.4.2 Pathways for Phosphorus Export

Tile drainage and overland flow are two primary export pathways for P. The pathways function differently in terms of hydrology which influences P export. The following is a review of subsurface (including tile drains) and overland flow hydrology and their role in P export. These are described below.

In the subsurface, P can move down through the soil in matrix flow and preferential flow. Concentrations of P fractions in solution vary between the two pathways (Jarvis 2007). Matrix flow refers to the typically slow, uniform flow through the soil matrix, while preferential flow, or non-equilibrium flow, occurs when water close to atmospheric pressure quickly bypasses a drier soil matrix (Jarvis 2007). There are several types of preferential flow, though the most influential in terms of P export is flow through macropores, as it results in the least amount of soil-water interaction. The two main types of macropores include bio-pores and structural cracks or fissures. Bio-pores are created by living or dead organisms, while structural cracks and fissures are created by the shrinking and swelling of clays (Beven and Germann 1982). These pores interact to form complex networks within the soil and provide a conduit where water can bypass the soil matrix (Shipitalo, et al. 2004). Macropore flow can occur in pores with a cylindrical equivalent diameter greater than 0.3-0.5mm (Jarvis 2007). The initiation of macropore flow depends on multiple factors including antecedent moisture conditions, the amount, intensity and duration of rainfall, and the saturated hydraulic conductivity of the soil (Heppell, et al. 2002, Jarvis 2007). Macropore flow is initiated when the pressure potential exceeds the water entry pressure, which will only occur at near saturated conditions. However, this does not mean the entire soil column must be saturated, as macropore flow can be initiated from small isolated saturated areas (Jarvis, 2007). The presence of macropore flow enhances P export because this flow bypasses more of the soil matrix, and thus has less opportunity to react with soil constituents (Jarvis, 2007). Concentrations of PP and SRP are typically higher in macropore flow relative to matrix flow. Concentrations of soluble organic P are often higher than soluble reactive P further down in the soil profile because SRP is quickly removed from solution while some organic P compounds are more mobile (Condron, et al. 2005, Anderson and Magdoff 2005).

The subsurface movement of P is altered by the presence of tile drains which are common practice in Ontario. It is estimated that 63% of non-pasture, agricultural land is drained in Ontario (Wang, et al. 2012). The drainage systems remove excess water from the rooting zone, improving growing conditions and offering more flexibility to managers during the spring and fall periods (Skaggs, et al. 1994). 25% of agricultural land in US and Canada does not have sufficient natural drainage for agricultural production (Skaggs, et al. 1994). Tile drains are necessary for agricultural production on many soil types found in Ontario and are a common practice.

The presence of tile drains alters field scale and watershed scale hydrology and P export. At the watershed level, tiles can account for 40-50% of discharge, and their role varies seasonally (Macrae, et al. 2007, King, et al. 2014). In terms of annual runoff, land drained with tile drains tends to have slightly higher (10%) total discharge when compared to land which is naturally drained (Blann, et al. 2009). Drains increase the water storage capacity of the soil, often reducing the instances of overland flow by as much as 60% (Blann, et al. 2009), and reducing surface erosion (Skaggs, et al. 1994). These changes have been reported to reduce overall site TP export (McDowell, et al. 2001, Ball Coelho, et al. 2012). Reductions of 30-36% in TP export have been reported (R. W. McDowell, et al. 2001). This reduction may not be observed in all situations, for example, in soils that are more permeable and have significant water storage initially, tile drains can increase the rate at which tile water drains and thus increase peak flow downstream (Blann, et al. 2009), which could increase downstream erosion and P export. Another reason that tiles may contribute to P losses is that installing tiles can increase the amount of land directly connected to surface drainage (Dils and Heathwaite 1999). Furthermore, since macropores provide a conduit for surface water to enter tile drains (Reid, et al. 2012), installing tile drains could enhance export by reducing interaction between runoff and the soil matrix. Macropore flow is often cited as the reason for elevated concentrations observed in tile drains (e.g. Gaynor and Findlay, 1995; Eastman et al., 2010)

Overland flow has long been recognized as an important P export pathway. There has been a strong effort to reduce P export in overland flow by promoting practices that reduced erosion, such as NT and erosion control structures (Sims, et al. 1998). Overland flow is generated as hortonian overland flow, or as saturated overland flow (Parlance, et al. 1999).

Hortonian overland flow is generated when the infiltration capacity of a soil is exceeded, which can occur in intense rainfall events, or where an ice lens has developed restricting infiltration near the surface. Saturated overland flow is initiated after soil has become saturated, at which point excess water will flow overland (Parlance, et al. 1999). Once it is generated, overland flow is capable of exporting PP through erosion of soil particles, as well as exporting DP. Soil erosion that occurs in overland flow selectively removes finer soil particles which tend to have higher concentrations of P relative to the bulk soil (Sharpley 1980). Soluble P in overland flow originates from desorption of reactive P in soil, as well as leaching from crop residues (McDowell, et al. 2001). Concentrations of PP and SRP are typically greater in overland flow than in tile drainage (McDowell, et al. 2001). Factors which influence P losses in overland flow and tile drainage are discussed in the following section.

2.5 Factors Influencing Phosphorus Export

There are a variety of other factors which can influence the amount and speciation of P in runoff. The review below is organized based on soil properties, land management practices, and hydro-climatic drivers but it is important to note that ultimately, P export results from the combined influence of all these factors.

2.5.1 Soil and Management Factors

Soil test phosphorus (STP) and soil P saturation are good indicators of DP concentrations in runoff. Generally, as STP and P saturation increase, so does the concentration of DP in runoff. Several agronomic measures have been shown to be good predictors of SRP in runoff, including the Olsen P test (Wang, et al. 2012, Wang, et al. 2010). The relationship between STP and DP in runoff varies between different soil types. For example, soils with higher clay content will release less DRP to runoff relative to sandy soils (Cox and Hendricks 2000). Several studies have noted rapid increases in the loss of DP beyond a certain STP, referred to as a change point (McDowell and Sharpley 2001). Change points can vary depending on the soil type so a range of change points have been reported (e.g. 33 mg Olsen P kg-1 (McDowell and Sharpley 2001), 47.8 mg Olsen P kg-1 (Wang, et al. 2012) and 60 mg Olsen P kg-1 (Heckrath, et al. 1995)).

Given that STP is a good predictor of DP concentrations in runoff, it is important to manage soil fertility in ways that do not build STP beyond levels required for agricultural production. STP is often built up in areas that receive annual manure applications. If these applications are made based on N requirements, then they can provide P above the crop requirements and thus P is built up in the soil (Hooda, et al. 2000).

In addition to avoiding the long term buildup of STP levels, particular caution must be taken to limit the direct losses from recent P applications (Hansen, et al. 2002, Hart, et al. 2004). Elevated TP and DP losses are seen in rain and melt events that closely follow the P applications (Macrae, et al. 2007, Gentry, et al. 2007, Hart, et al. 2004, Eastman, et al. 2010). The losses are referred to as direct fertilizer losses, event specific losses, or incidental losses, and are important in terms of annual P losses (Hart, et al. 2004). Concentrations in runoff decrease rapidly with time following application (Hart, et al. 2004). These losses are further influenced by the source, rate, application method and seasonal timing and frequency.

Source properties influence the amount and speciation of P losses. Nichols, et al. (1994) compared surface runoff from soils fertilized with manure or commercial fertilizer and found that concentration of TP in runoff from manure fertilized soil was lower (15.4 mg L^{-1}) than commercial fertilizer fertilized fields (26.2 mg L^{-1}) 7 days after fertilization. Heathwaite et al. (1998) also found TP concentrations were higher in runoff from commercial fertilizer soil (15.3 mg L^{-1}) compared to soils fertilized with farm yard manure (1.76 mg L^{-1}). If heavy rainfall is avoided following application and manure is incorporated, manure can pose a lower risk to P losses than commercial fertilizers (D. R. Smith, et al. 2007). Inositol phosphates, a form of soluble organic P found in high amounts in pig and poultry manure, can displace inorganic P and release it into solution by competing for binding sites (Condron, et al. 2005, Anderson and Magdoff 2005). Other organic P compounds are more mobile than inorganic orthophosphate, this has resulted in greater concentrations of organic P in tile drainage relative to overland flow due to the fact that inorganic P is quickly sorbed by soil (Condron, et al. 2005).

Higher application rates lead to higher P losses for both incorporated and non-incorporated sources. Allen and Mallarino (2008) found that P concentrations in surface runoff from plots fertilized with liquid swine manure increased with increasing application rate in both incorporated and non-incorporated plots. The rate of increase was less when manure was incorporated (Allen and Mallarino 2008).

Phosphorus sources are applied in a variety of ways: surface broadcast and incorporated, surface broadcast and not incorporated, surface banded, subsurface banded, or applied as a starter fertilizer (Baute 2002). The incorporation or subsurface banding of P fertilizer reduces losses of DP in surface runoff (Kleinman, et al. 2002, Allen and Mallarino 2008). Allen and Mallarino (2008) found that runoff concentrations of SRP, BAP, and TP from non-incorporated manure were 3.3, 7.7 and 3.6 times higher than incorporated manure plots respectively. Incorporation of manure can reduce the portion of TP present as SRP and BAP (Allen and Mallarino 2008). However, if incorporation is not below the depth of surface interaction incorporation may have no effect (Nichols, et al. 1994).

The presence or absence of vegetation or residue cover influences soil erosion, runoff and P export. In Ontario, maintaining residue cover year round is promoted as a BMP (Baute 2002). This residue cover is achieved using a range of systems referred to as reduced tillage (RT) systems. In reduced tillage systems, soil is worked less aggressively, and less often relative to conventional systems. These systems have been adopted by farmers for a variety of reasons including reduced operating cost, benefits to soil integrity, improved infiltration, water retention and erosion prevention. The adoption of reduced tillage practices has often reduced soil erosion and PP losses (e.g., Chichester and Richardson, 1992).

However, studies have also shown that NT and RT systems can increase the risk of DP and TP losses (Gaynor and Findlay 1995, Hansen, et al. 2000, Kleinman, Sharpley and McDowell, et al. 2011). The influence of tillage systems on DP losses was recognized as early as 1973, by using simulated rainfall (Römkens, et al. 1973). This increase is attributed the stratification of P near the surface if nutrients are broadcasted and not incorporated (Delgado and Scalenghe 2008, Kleinman, et al. 2011). In NT systems, P levels in the top 5cm can be 10-20 times greater than soil from 5-20cm (Sharpley 2003). Stratification of P allows runoff to interact with potentially P saturated soils near the surface, increasing the risk of DP losses (Puustinen, et al. 2007, Kleinman, et al. 2011). Puustinen et al. (2007) found that DP concentrations in runoff were higher in RT methods relative to fall conventional tillage methods. Lui et al. (2014) found that TDP and TP concentrations and export in surface runoff could be reduced by changing conservation tillage systems to rotational tillage system (tilled aggressively every other year). They attributed the reduction to reductions in Olsen P at the surface, P loss from residue and in the duration of runoff (Liu, et al. 2014).

There is also seasonal variation in the effectiveness of RT as a BMP due to different runoff patterns between tillage systems, and the effect of event type and ground conditions on P export. Uusitalo, et al. (2008) found that the reduced tillage systems exported more PP (TP-DRP (0.2µm filter)) relative to plowed systems because of the greater amount of overland flow volume during the winter in reduced tillage systems. Tan, et al. (2002) found that RT fields had greater surface runoff relative to conventional tilled fields during the NGS and attributed this to greater snow retention and less surface roughness. Hansen et al. (2000) found that over two NGS's RT systems had greater surface runoff and TP losses than conventional tillage plots. Reduced tillage systems are effective at reducing PP export on thawed soils, however on frozen soils, DP losses are more important. In both systems, NGS losses are often dominated by DP, however the RT system had greater DP losses (Hansen, et al. 2000). RT systems increase the supply of nutrients near the surface (STP and residue P), and the amount of runoff during the NGS which appears to increase the risk of P losses from reduced tillage systems in cold climates.

2.5.2 Hydro-Climatic Variables and Event Types

Rainfall intensity affects the speciation of P in runoff by affecting export mechanisms (erosion and transfer of soluble P). Rainfall impact increases soil erosion and as a result tends to increase PP concentrations in runoff (Fraser, et al. 1999, Su, et al. 2011). Increased rainfall intensity is also related to increases DRP concentrations in surface runoff (Shigaki, Sharpley and Prochnow 2007).

Differences in speciation between winter snow melt and summer time events are often noted (Ball Coelho, et al. 2012, Hansen, et al. 2000, Macrae, et al. 2007). These differences are related to the influence of soil conditions and rainfall impact. Higher SRP and TP concentrations are observed during summer storms when soil is exposed to rainfall impact than during winter snowmelts (Ball Coelho, et al. 2012). Furthermore, DP represents a larger portion of TP during snowmelt events relative to rain events due to the absence of raindrop impact and the presence of frozen soil (Hansen, et al. 2000). P speciation and export is influenced by hydro-climatic drivers and soil conditions.

The volume of runoff is a major driver in overall P losses (Liu, et al. 2013). Due to the responsiveness of the flow pathways to rainfall and snow melt, the export of P from agricultural land is highly episodic. Furthermore, annual totals are often dominated by a small number of large events (B. Ulén 1995). For example, in colder climates, snowmelt events can be particularly important in terms of their contribution both to annual runoff and annual P export (Jamieson, et al. 2003, Tiessen, et al. 2010). In cold climates, runoff is often greater during the non-growing season (Hirt, et al. 2011) and P export tends to have a seasonal distribution, related to the seasonal distribution of flow. Antecedent moisture and rainfall characteristics influence the volume of runoff generated, which ultimately affects P export. Hirt et al. (2001) found that tiles responded to 70% of rain events throughout the year, but responded to a higher percentage during the non-growing season (84%) when conditions were wetter, compared to the growing season (56%). Hirt et al. (2011) also found that in the summer, it generally required a greater rainfall intensity to initiate a response, as compared to the winter months. The same is true for overland flow, where greater rainfall or melting snow is required to generate overland flow under dry antecedent conditions than is needed under wetter conditions. The partitioning of discharge is influenced in part by precipitation characteristics. Higher rainfall intensities increase the likelihood of overland flow events, and thus influence the partitioning of flow between export pathways (Kleinman, et al. 2006). Although concentrations of P are lower in tile drainage relative to overland flow, tile drains have been found to make important contributions to overall site losses due to the high volume of water leaving through this flow path (Ball Coelho, et al. 2012, Eastman, et al. 2010, Gaynor and Findlay 1995).

The area within a watershed directly contributing to runoff generation can expand and contract seasonally as well and during individual events. This is referred to as variable source area concept (Ward 1984). This concept is particularly important when it comes to identifying areas likely to contribute to P loading within a watershed, as it may only be necessary to identify areas where a source of P coincides with an area contributing to runoff generation.

2.6 Critical Source Areas and Management Approaches

P export is controlled by the interaction of export pathways and P sources (Gburek and Sharpley 1998, Kleinman, et al. 2006). Areas with high STP, or with soils that are susceptible to erosion are not necessarily contributors to P loading at the watershed scale. These areas only contribute to P loading if there is hydrological connectivity between this P source and a receiving water body. Pionke et al. (1997) investigated the spatial variability contribution to P export and found that the >90% of P export to the Chesapeake Basin originated from <10% of area within the watershed. Due to the overwhelming influence of flow volume on transport, even modest sources of P may contribute a large percentage of loading within a watershed because of the site's hydrology (Sharpley, et al. 2013). Targeting critical source areas has become a common approach for addressing P export at the watershed scale (Kleinman, et al. 2011). The lack of targeting has been cited as reason for the benefits of conservation practices not being observed at the watershed scale (Tomer and Locke 2011).

There is a recognized need for site specific management approaches for dealing with P losses (e.g. Ulén and Jakobsson, 2005). Even though sites may have similar STP, soils and topography, if they generate runoff differently, they may require different management approaches to control P losses (Daniel, Sharpley and Lemunyon 1998). The recognized need for site specific assessments, and targeted management efforts, has led to the development of numerous regional P indices (Ulén, et al. 2011). These P-indexes account for the risk of P losses based on site specific data and management practices. Unfortunately, existing P Indexes do not properly account for P losses through tile drainage (Reid, et al. 2012).

2.7 Summary

Managing P in agricultural systems offers several challenges. P exists in soil in varying degrees of availability to plants. Since the supply of P in the soil solution is low, P must be cycled between various pools to provide enough P to sustain crop growth. Applications of external sources of P are used in agriculture to maintain adequate levels of P in soil and to replenish P removed via crop removal. Soluble and particulate P can be exported from the site in overland flow and tile drainage. Export of P through these pathways is episodic and has a seasonal distribution related to the distribution of runoff. Concentrations of P in runoff are influenced by the interaction of hydro-climatic drivers, site properties and management practices. These amounts are usually small from an agronomic stand point but can be significant from an environmental stand point. Thus, there is a need to better manage these losses to mitigate environmental risk. One of the widely accepted management tools is the P-Index, however the current Ontario Index does not accurately account for tile drainage. To include tile drains we must understand the relative contribution of tiles and overland flow to P export. Other factors specific to Ontario must also be investigated further, including the performance of tillage systems common in Ontario, and the effect of seasonality in the Ontario setting.

3 Site Descriptions and Methods

3.1 Site Descriptions

Overland flow and tile drainage P export from agricultural fields was monitored at three field-scale sites, one in eastern Ontario (Site 1: UTM 18T 547572m E, 5003684m N) and two sites in southern Ontario (Site 2: UTM 17T 472219m E, 4767583m N and Site 3: UTM 17T 466689m E, 4832203m N) (Figure 3-1). The sites were selected because they were managed with *reduced* tillage (RT) systems, represented a range of climate conditions, and had isolated overlaying overland and tile catchments within a field.

At each site, hydraulically isolated catchment areas (tile drainage and surface drainage) with uniform cropping history were selected. The tile drainage catchments were similar in size (Site 1: 4.266 ha; Site 2: 8.655 ha; and Site 3: 7.773 ha). Each field had existing systematic tile systems (~75cm depth), though the tile spacing varied between the sites (Site 1 ~11.5m; Site 2 ~9m; Site 3 ~14m). The diameter of the main outlet drains for the drainage systems varied between the sites (Site 1: 6”; Site 2: 8”; Site 3: 6”). Surface drainage catchment areas were, 2.520 ha, 7.787 ha, 8.139 ha for Site 1, Site 2 and Site 3, respectively. These sites were selected because overland flow exits the fields predominantly in one location in each field rather than many diffuse exit points, allowing the measurement of surface runoff. However, hydrology at two of the sites was complicated because of features discovered after the sites had been selected. Site 1 had several French drains in the fields which allowed for surface water to enter the tile drainage system. There was some uncertainty about the surface drainage catchment area at Site 1 because there was one additional outlet that was not monitored, thus not all surface runoff was captured by the monitoring equipment. The catchment area used in all calculations represents the largest potential catchment area. Site 2 also had complicating factors, as the tile drainage system was connected to two catch basins that were located in the homestead’s driveway. Although these features result in some uncertainties, they are common practice, and the sites are considered fair representation of real working farms in Ontario.

Figure 3-1 Locations of monitored sites within Ontario.

3.1.1 Climate

Generally, the three sites are located in a similar climate type; however, there are some notable differences with regards to average winter temperatures and precipitation (Figure 3-2). Long-term average daily temperatures are similar at Sites 1 (7.0 ˚C)(St-Anicet 1) and Site 3 (7.2 ˚C)(Blyth), but slightly warmer at Site 2 (8.2 ˚C)(Bear Creek) (Figure 3-2). Monthly temperatures demonstrate significant seasonality across the year, with cold winters and warm summers. Temperatures are generally below freezing from December through March in this region. However, daily maximum and minimum temperatures can be highly variable. Due to the climate, spring planted crops are seeded in April-June and harvest is typically over by November. Average annual precipitation amounts are 1004 mm (St-Anicet 1), 1024 mm (Bear Creek) and 1247 mm(Blyth) for Site 1, Site 2 and Site 3, respectively (Figure 3-2), a portion of which falls as snow (Site 1: 16%, Site 2: 17% and Site 3: 30%). Precipitation distributions throughout the year are similar among the sites, except for higher amounts of winter precipitation at Site 3 (Figure 3-2).

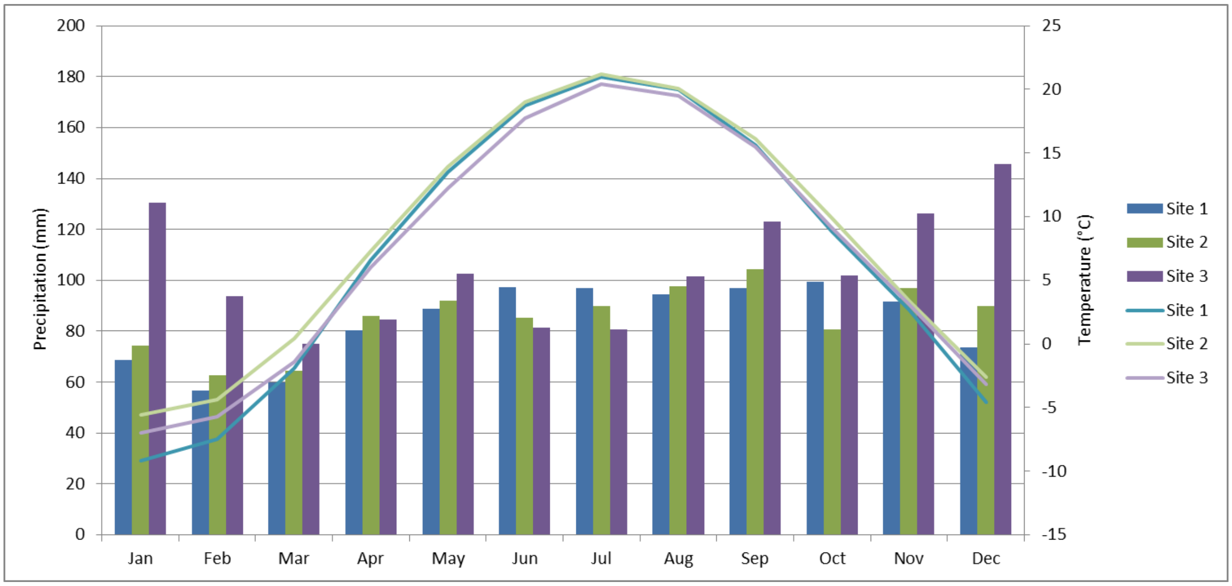

Figure 3-2 Long-term (30 year) mean monthly precipitation totals (bars) and mean monthly temperatures (lines) for the three study sites.

3.1.2 Soil Physical and Chemical Properties

The sites represent a range of soil types and topographies. A description of each site is provided below, including topography, soil type, texture, organic matter, pH and soil test phosphorus levels. Descriptions in this section include a combination of results from soil testing completed in 2011 for this study, and descriptions from historical county soil surveys. A description of Olsen P stratification and field variability according to the 2011 sampling results is also provided.

Site 1 has unique soil physical and chemical properties relative to the other sites. The area was flat and the soils are poorly drained (Matthews, Richards and Wicklund 1957). Based on the topographic survey completed at the site, slopes in the field range from 0.5 to 3.6% (Table 3-1). The dominant soil at Site 1 is Bainsville Silt Loam, which is part of the Dark Grey Gleysolic great soil group (Matthews, et al. 1957). This soil was developed on water deposited sands and silts, which overlay glacial till (Matthews, et al. 1957). The underlying clay layer begins approximately 1 meter below the surface (Matthews, et al. 1957). Based on soil sampling completed in 2011, the average texture in the top 15 cm is Silt Loam (1.2 ± 0.7 % Clay, 58.4 ± 4.6 % Silt, 40.4 ± 3.9 % Sand) (Table 3-1). In the top 15 cm, soil organic matter is 4.6 ± 0.2 %, and pH was 6.1 ± 0.2. Average Olsen P in the top 15 cm is 15 ± 5 mg kg^{-1}. STP is vertically stratified with higher concentrations occurring at the surface (Average Olsen P at 0-2.5

cm: 17 ± 3 mg kg^{-1}) than at greater depths (Average Olsen P at 60-75 cm: 5 ± 1 mg kg^{-1}) (Figure 3-3). Olsen P levels in the top 2.5 cm of soil varied across the site (12-22 mg kg^{-1}) (Figure 3-3).

Site 2 has undulating topography (Hagerty and Kingston 1992), with slopes ranging from 0.3 to 3% based on a topographic survey completed at the site (Table 3-1). The field at Site 2 consists of soils from the Bryanston association (Bryanston and Thorndale soils) (Hagerty and Kingston 1992). The Bryanston and Thorndale soils developed on loamy-textured glacial till (Hagerty and Kingston 1992). Bryanston soil is well drained, while Thorndale soil has imperfect drainage (Hagerty and Kingston 1992). Soil samples taken from the site indicate the average texture in the top 15 cm is Silt (Clay 0.5 ± 0.6%, Silt 84.4 ± 3.3%, Sand 15.1 ± 3.4%) (Table 3-1). Due to the surface texture, these soils are susceptible to erosion (Hagerty and Kingston 1992). 50-58 cm below the surface the soil is a loam texture (Hagerty and Kingston 1992). In the top 15cm the average soil organic matter is 4.1 ± 0.7%, and the average pH is 7.7 ± 0.3 (Table 3-1). Average Olsen P in the top 15 cm is 15 ± 3 mg kg^{-1}. STP concentrations are higher at the surface (Average Olsen P at 0-2.5 cm: 27 ± 6 mg kg^{-1}) than at greater depths (Average Olsen P at 60-75 cm: 3 ± 1 mg kg^{-1}) (Figure 3-3). There is spatial variability in Olsen P levels in the top 2.5 cm, with levels ranging between 21-36 mg kg^{-1} (Figure 3-3).

Site 3 is located on gently undulating terrain (Hoffman, et al. 1952), with slopes ranging from 0.2 to 3.5% (Table 3-1). The site is mapped as Perth Clay Loam, which developed on limestone glacial till, and is in the Grey Brown Podsolic great group (Hoffman, et al. 1952). These soils have imperfect drainage (Hoffman, et al. 1952). The average texture in the top 15 cm is Silt (Clay 0.9 ± 1.3%, Silt 75.7 ± 2.1%, Sand 23.4 ± 3.3%) (Table 3-1), whereas the till 40cm below the surface is clay loam textured (Hoffman, et al. 1952). In the top 15 cm the average soil organic matter is 4.1 ± 0.5 % and the average pH is 7.7 ± 0.3. Average Olsen P in the top 15 cm is 12 ± 2 mg kg^{-1}. Concentrations of STP are higher near the soil surface (Average Olsen P at 0-2.5 cm: 22 ± 6 mg kg^{-1}), than at greater depths (Average Olsen P at 60-75cm: 4 mg kg^{-1} ± 2). The Olsen P levels in the top 2.5 cm ranged between 18-30 mg kg^{-1} at the site (Figure 3-3).

These sites represent a range of soil properties, however some similarities exists. The sites have topography ranging from flat to undulating, but none of the sites have steep slopes. Another similarity is that all sites have soil with poor to imperfect drainage and thus benefit from tile drainage.

Table 3-1 Soil properties for the top 15 cm (mean and standard deviation) and slope ranges for all sites.

	Site 1	Site 2	Site 3
Sand (%)	40.4 ± 3.9	15.1 ± 3.4	23.4 ± 3.3
Silt (%)	58.4 ± 4.6	84.4 ± 3.3	74.7 ± 2.1
Clay (%)	1.2 ± 0.7	0.5 ± 0.6	0.9 ± 1.3
Organic Matter (%)	4.6 ± 0.2	4.1 ± 0.7	4.1 ± 0.5
pH	6.1 ± 0.2	7.7 ± 0.3	7.7 ± 0.3
Olsen-P (mg kg^{-1})	15 ± 5	15 ± 3	12 ± 2
Slope	0.5-3.6%	0.3-3.0%	0.2-3.5%

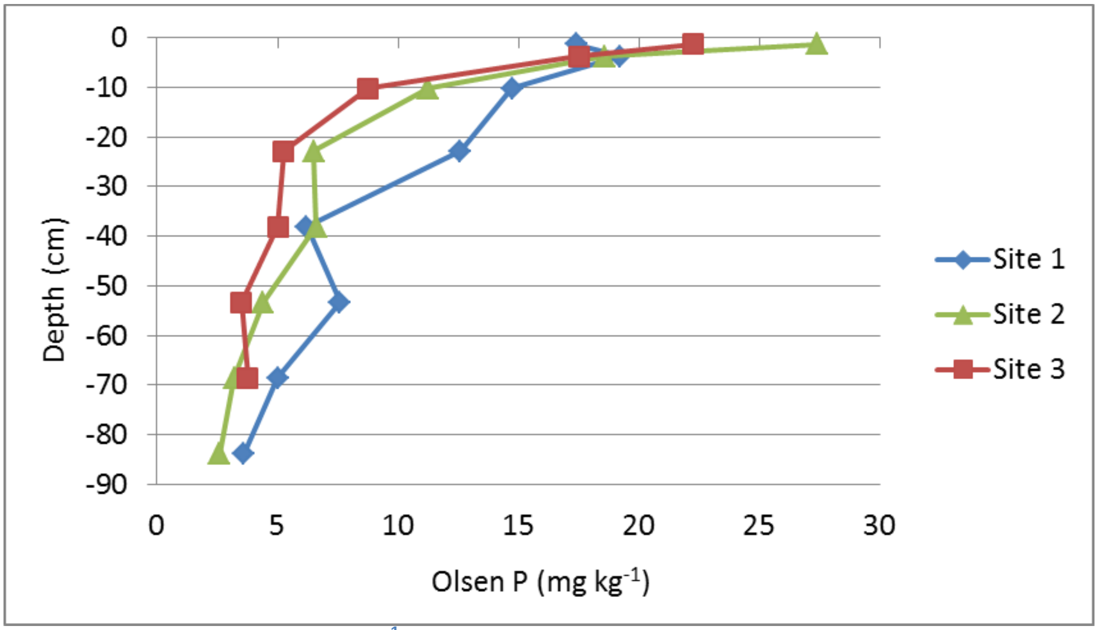

Figure 3-3 Average Olsen P mg kg^{-1} vs depth for each site.

3.1.3 Management Practices

All sites were located on working farms and thus all management decisions were made by the farm operators. The tillage systems, fertilization methods, and crop rotations differed between the sites (Table 3-2). Three different types of RT systems were used (i.e., Ridge, Site 1; Zone, Site 2; and Vertical, Site 3). In this study, RT refers to a range of tillage systems, including rotational tillage systems, as well as systems where light tillage occurs for each crop. The sites had different fertilization strategies, application rates and application methods. At all sites, the long term strategy was to apply P to maintain adequate STP. In each system, individual applications were intended to supply the needs of multiple crops. Manure was the only source of P used at Site 1 prior to the study period. Liquid manure (laying hen) and mineral P fertilizer were used at Site 2. At Site 3 mineral P fertilizer was the only source of P used.

Within the study period no applications of P were made at Site 1, but fall applications of P were made at Site 2 and Site 3 in 2011. Starter P fertilizer was applied in 2012 at Site 3. Site 2 incorporated mineral P fertilizer into the tilled zone, and broadcasted liquid laying hen manure (not incorporated). At Site 3, P fertilizer was broadcast and incorporated with shallow tillage. P applications at the sites exceeded OMAFRA fertilizer recommendations for the Olsen P levels observed (Baute 2002); however the rates were intended to match crop removal which is an accepted agronomic practice. Both Site 1 and Site 3 used a cover crop following wheat in 2011, while the following winter no cover crops were used at the sites. The length of rotation, as well as the number and type of crops within rotations differed between the sites (Table 3-2). Site 1 uses a five year rotation of cereal, grain and legume crops. The rotation at Site 2 included, corn, soybeans, wheat and azuki beans. Site 3's rotation is corn, soybeans, wheat. Corn was grown at all sites during the first growing season of the study. These sites are carefully managed with respect to tillage and P application, all decisions made within this study period were accepted agronomic practices (Baute 2002).

Table 3-2 Farming practices for all sites. Crop, tillage, application rate and method, and use of cover crops between October 2011 and April 2013.

		Site 1	Site 2	Site 3
Crop	2011	Wheat	Soybeans	Wheat
	2012	Corn	Corn	Corn
	2013	Study period ends in April before planting		
Tillage	2011	Ridge Tillage	Post Harvest: Zone Tillage	Post Harvest: Vertical Tillage
	2012	Pre-Plant: Disc Harrow		Pre-Plant: Vertical Tillage Post Harvest: Vertical Tillage
P Application Rate (P_2O_5 kg ha^{-1})	2011	No Applications	*Laying Hen Manure: 83 (Total Manure P_2O_5: 207) MAP: 87	MAP :172
	2012	No Applications	No Applications	MAP: 32
Application Method	2011	NA	Manure Surface Broadcast; MAP Incorporated in Zone	MAP Incorporated with Tillage
	2012	NA	NA	Incorporated with Planter
Cover Crops	2011	Oil Seed Radish after Wheat	None	Red Clover
	2012	None	None	None

*P_2O_5 in Manure represents the estimated value available to the next crop (40% of total) (Brown, 2013). Total P_2O_5 content is also provided.

Residue cover varied across sites and years because of different crop rotations and tillage practices. No direct field measurements of residue cover were taken during NGS1; however, based on visual observations, Site 2 (soybeans) had less soil cover than Sites 1 and 3 (wheat) primarily because of the crops grown that year. Residue cover was lowest after planting at Site 1 and Site 3 because of pre-plant tillage. The residue survey conducted following planting indicated there was 20-34% residue cover at the sites (Table 3-3). No measurements were taken during the NGS2, but based on visual observation, all sites had a high level of corn residue, with some variances because of minimal fall tillage.

Table 3-3 Percent residue cover from measurements completed after planting, May 2012.

Site	% Cover
Site 1	21 ± 9
Site 2	34 ± 6
Site 3	20 ± 3

3.2 Methods

Three isolated watersheds were used for this study. Each site was equipped with a weather station to collect meteorological variables, as well as flow monitoring and water sampling equipment to monitor flow and collect samples for water quality analysis (Figure 3-4, Figure 3-5, and Figure 3-6). The study period began in October 2011 and ended in April 2013. The exact start dates varied between the sites. Where meteorological equipment was installed before flow monitoring equipment, it was assumed that no flow was missed based on what had been observed at other monitoring sites in the province. Monitoring at the sites ended between April 17 and April 31, 2013. The study period is broken into three periods: the first non-growing season (NGS1) (October, 2011-April 2012), the first growing season (GS1) (May 2012-September, 2012), and the second non-growing season (NGS2) (October 2012-April 2013). This allows for meaningful comparisons of tile drainage discharge and P export between the two NGSs, as well an accurate partitioning of overland and tile drainage for one 12 month period (May 2012-April 2013, inclusive). The instrumentation and data collection methods are described in the following sections.

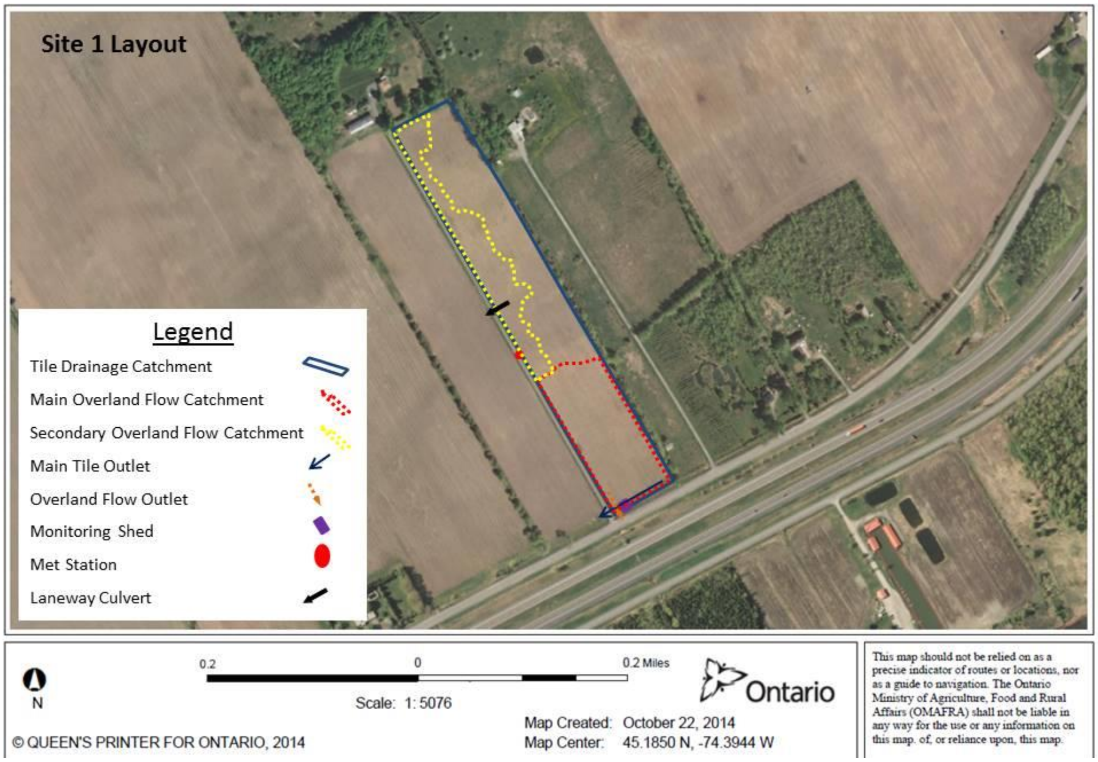

Figure 3-4 Site 1 Catchment area and site instrumentation.

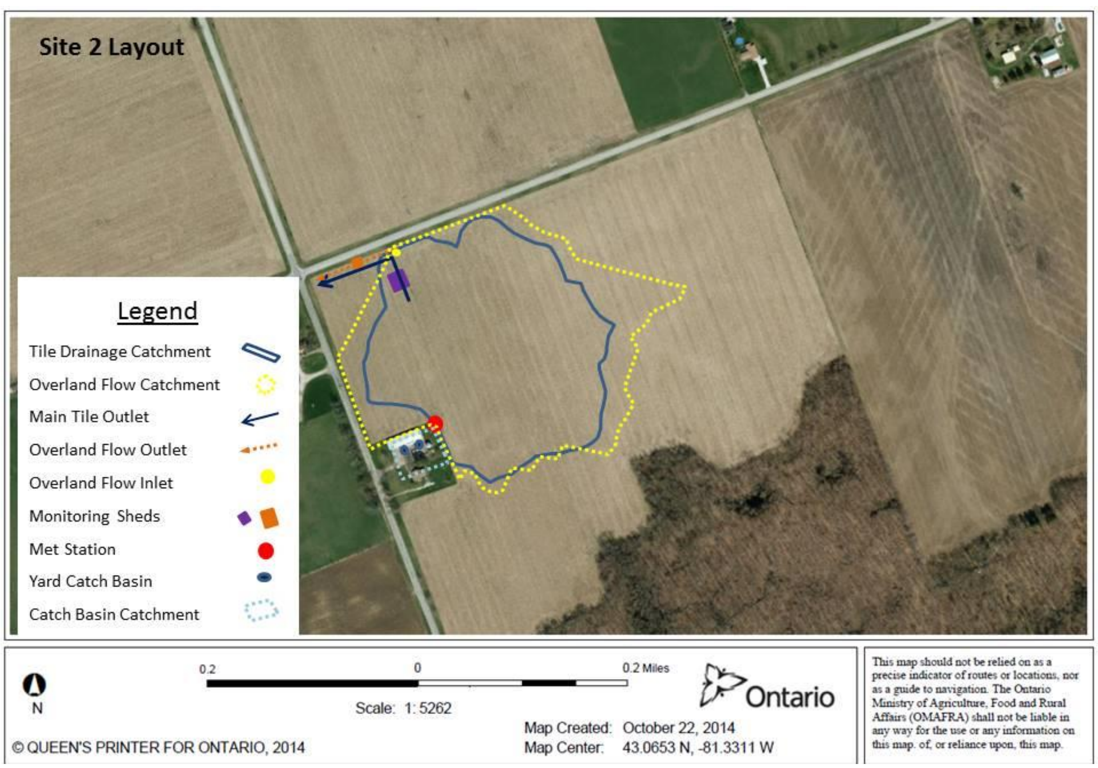

Figure 3-5 Site 2 Catchment area and site instrumentation.

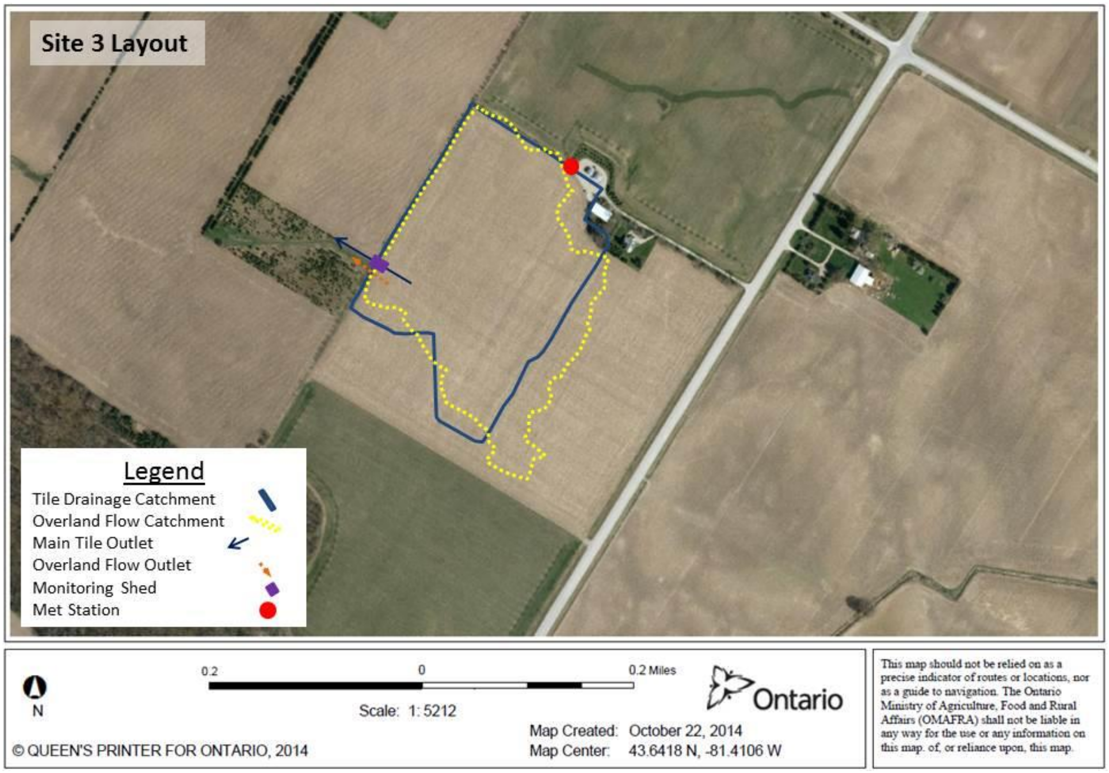

Figure 3-6 Site 3 Catchment area and site instrumentation.

3.3 Meteorological and Soil Environmental Variables

Meteorological data at each site was measured using automated weather stations equipped with the following sensors (Onset Corporation HOBO Weather Station): tipping bucket rain gauge (.2mm Rainfall Smart Sensor - S-RGB-M002), air temperature and relative humidity (12-bit Temperature/RH Smart Sensor - S-THB-M002) with a solar radiation shield, incoming solar radiation (Solar Radiation Sensor (Silicon Pyranometer) - S-LIB-M003), wind speed/direction (Wind Smart Sensor Set - S-WSET-A), soil moisture (EC-5 Soil Moisture Smart Sensor) and soil temperature (12-Bit Temp Smart Sensor - S-TMB-M002) (installed at 10cm, 30cm and 50cm depths). Data was logged at 15 minute intervals using a Hobo U30 GSM logger (Onset Corporation). Stations were located on the field boundary so they did not interfere with regular farming operations (Figure 3-4,Figure 3-5, and Figure 3-6). Water table elevation was monitored using Hobo Level Loggers (U20) at one well (2m depth, 2” ID) per site. Wells were located midway between tile runs at Site 1 and Site 3 (Figure 3-4, Figure 3-6). The well at Site 2 was located 2m from the main drain (Figure 3-5).

3.4 Snow Water Equivalent Estimation

Snowfall data from nearby Environment Canada Weather Stations were used to estimate monthly precipitation as snowfall at each site. Snowfall at Site 1 was estimated using data from the St-Anicet 1 station in Quebec. Snowfall at Site 2 was estimated using data from Strathroy-Mullifarry station in Ontario. Site 3 was estimated using data from Wroxeter station in Ontario. These stations were the closest station with daily snowfall data for the study period. Snow depth of the snow cover at the sites was monitored throughout the second NGS (NGS2). Snow surveys were completed between March 7 and March 11, 2013 to determine the SWE prior to the final melt. To complete this survey, six snow cores were taken from random locations in the field for determinations of snow density using a 67mm diameter snow tube. Six snow depth measurements were taken randomly around the location of each snow sample. The measured density and average snow depth at each location were used to calculate the SWE at each location. The average SWE of the six sampling locations was used to estimate the SWE at the site. Snow depth was also recorded prior to and following major events. On ground snow measurements at the Environment Canada weather stations were consistent with field measurements before and after events. Precipitation in periods prior to snowmelt events, snow depth measurements, and snow surveys were used to estimate SWE available prior to melt events.

3.5 Tile Drainage Monitoring

At each site a tile drainage monitoring station was constructed near the field edge, above the main drain exiting the field. Tile drain pipes were accessed using a backhoe, and a section of the main tile drain was removed and replaced with a custom built piece that allowed direct access to the tile drain through two riser pipes (Figure 3-7). The custom built piece had the same diameter as the existing tile at Site 1 and Site 2, but a greater diameter than the existing tile at Site 3. Excavated soil was subsequently backfilled around the tile drain and riser pipes. The upstream riser pipe was used for the flow monitoring equipment and the downstream pipe was used for water sampling. At the downstream end of the custom insert there was a cipolletti weir located just prior to the tile emptying into a built-in a sampling basin (Figure 3-7). This weir permitted the estimation of flow during low-flow periods based on measurements of water depth in the weir (using the same Hach Flo-tote3 sensor) as reliable estimates of velocity are not possible with the flowmeter unit at flow depths below 50 cm (discussed further below). Plastic sheds (10x10ft) were built above the riser pipes to protect the equipment from freezing or contamination.

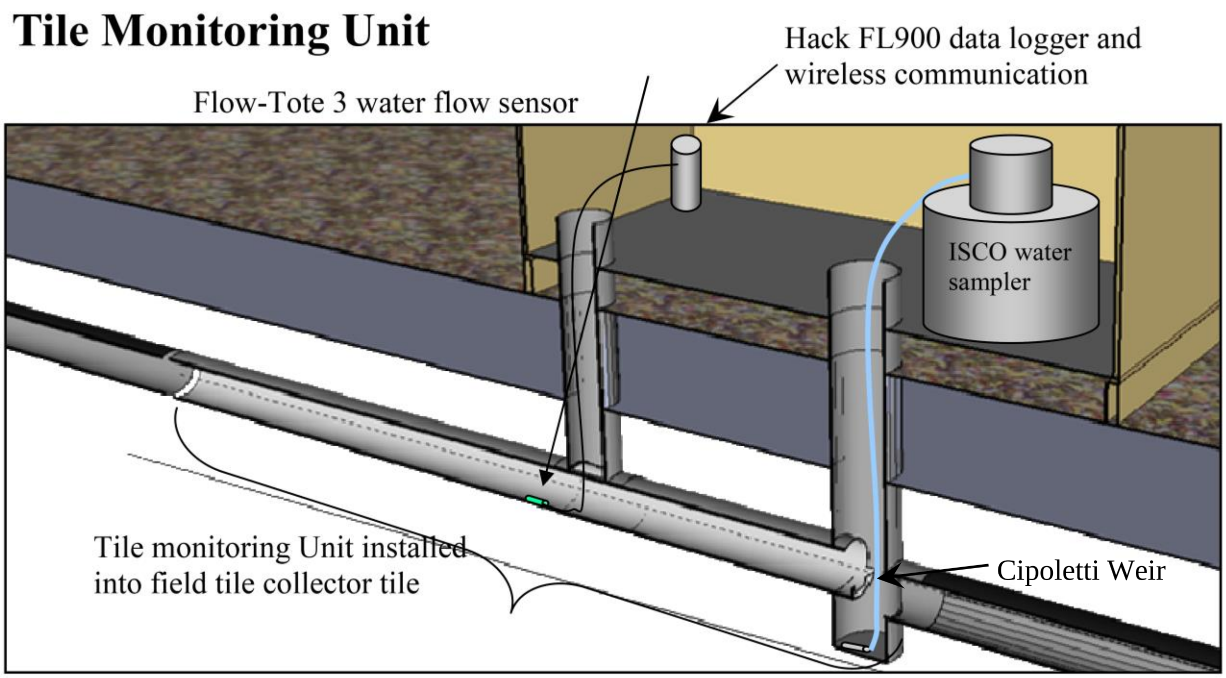

Figure 3-7 Schematic of the tile drainage monitoring station (R. Brunke, unpublished).

The method used to measure flow within the tile drain was changed mid-way through the study period. Between Oct 2011 and May 2012 the depth of water in the tile was measured using a Hobo Water Level Data Logger which was lowered into the drain through the upstream riser pipe. This logger measured water depth at 15 minute intervals. Between April and May 2012, the Hobo Water Level Data Logger was removed and a Hach Flo-Tote3 sensor was installed to monitor flow (depth and velocity). Data from the Flo-Tote3 was logged using a Hach FL-900 at 15 minute intervals. The Flo-tote3 measured depth and velocity and calculated discharge as:

$$Q = AV$$

Where Q is discharge, A is cross-sectional area and V is velocity.

Low flow (Site 2: < 80 mm water depth, Site 3: <82 mm water depth) in the tile drain at Site 2 and Site 3 was calculated using a cipolletti, or trapezoidal, weir because the sensor could not achieve a reliable velocity reading at these flow rates. At Site 1 all flow was determined using the Flo-tote because a section of the tile downstream had a negative grade which caused there to be standing water at the weir for long periods, even though no flow was occurring. The weir crest was 25 mm above the bottom of the tile, with a 60 mm crest length, and the sides of the weir opening were inclined at a 4:1 slope. Total height of the weir was 52.5 mm.

The cipoletti weir equation is:

$$Q = 0.000588Lh^{3/2}$$

Where Q is discharge in l s^{-1}, 0.000588 is the discharge coefficient; L is the length of the weir crest in cm; and h is the depth of water above the crest of the weir in mm (United States Department of the Interior Bureau of Reclamation 2001) (Figure 3-8). Water depth measured by the Flo-Tote3 was used to determine h. Field measurements were taken to determine the depth of water, at which water began to flow over the crest of the weir. If the level dropped below this depth, discharge was zero. Above a defined depth, discharge measurements obtained directly from the Flo-Tote3 was used instead of the weir calculation (Table 3-4). For example, at Site 3, the weir was used to calculate flow between Flot-Tote3 depth readings > 55 m and ≤ 82 mm, while flow readings directly from the Flo-Tote3 were used at depths >82 mm (Figure 3-9).

Table 3-4 Tile depths used to determine flow calculation method (No Flow, Weir or Flo-Tote) at each site.

Site	Assumed No Flow (mm)	Weir Calculation (mm)	Flo-Tote Sensor (mm)
Site 1	Flo-tote Only	Flo-tote Only	Flo-tote Only
Site 2	≤50	>50 and ≤ 80	80>
Site 3	≤55	>55 and ≤ 82	82>

Rating curves were created using discharge data collected between May 2012 and January 2013. These relationships were used to determine discharge for the period prior to the installation of the Flo-Tote3 sensors (Oct 2011-May 2012). Due to variability within the stage discharge relationships at Site 1 and Site 2, separate rating curves for the rising and falling limbs of the hydrograph were developed to better represent major discharge events. These additional relationships were used to estimate flow for a small number of major flow events at Site 1 and Site 2. All rating curves that were developed are included in Appendix A.

Figure 3-8 Picture of the cipoletti weir from inside the tile (picture taken looking upstream).

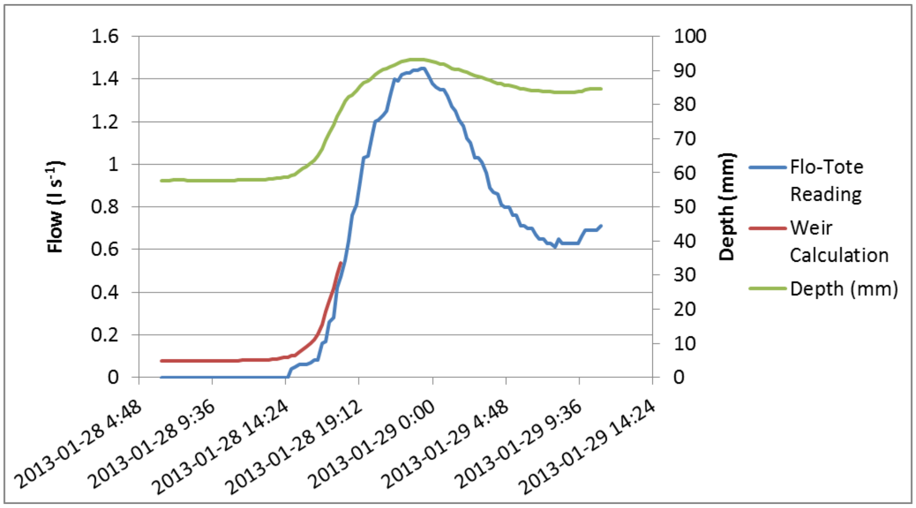

Figure 3-9 Switching from the cipoletti weir calculated flow to the Hach Flo-Tote3 calculated flow at Site 2 on January 28, 2013.

3.6 Overland Flow Monitoring

Overland flow was measured for the 12 month period. With the exception of the one unmonitored culvert outlet at Site 1, surface overland flow left each of the study sites via a single outlet rather than many diffuse points, permitting the measurement of the volume of overland flow exiting the field. The specific monitoring setup differed slightly between the sites due to topographic features of the sites; however, the same monitoring instrumentation and general monitoring approach was used at all three sites. At Site 1 and Site 3 overland flow from fields was confined using berms at the field edge, and directed into an outlet pipe (0.20m in diameter at Site 1, 0.46m in diameter at Site 3) where flow could be measured (Figure 3-4, Figure 3-6 and Figure 3-10). At Site 2 overland flow was collected in a hickenbottom inlet and measured in a non-perforated pipe (0.20m dia.) installed below ground. The pipe was accessed through vertical riser pipes identical to those used in the tile drainage design (Figure 3-5). Overland flow was measured using depth and velocity measurements (Hach Flo-Tote3 sensor) taken at 15 minute intervals, and logged using a Hach FL900 data logger. If the flow sensor malfunctioned, discharge was estimated using rating curves established from captured events combined with water depth measurements from Hobo Water Level Loggers which were installed as a backup for the flowmeters.

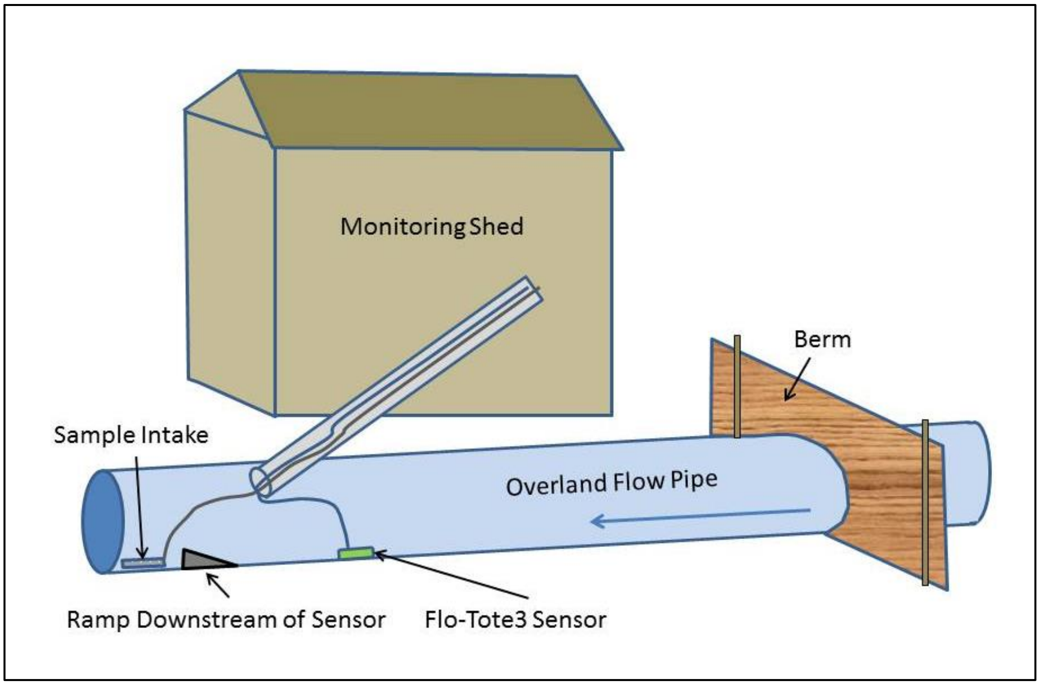

Figure 3-10 Schematic of overland flow monitoring station (Site 1 and Site 3).

3.7 Tile and Overland Flow Water Sample Collection

Each tile/overland flow monitoring station was equipped with an automated water sampler (AWS) (ISCO 6712, Teledyne). For tile monitoring, the sampling hose was fed through the tile monitoring insert's downstream riser pipe into a basin integrated into the tile monitoring pipe located just below the cipoletti weir (Figure 3-7). The inlet of the sample hose was positioned so that the sample was pulled from the center of the collection basin, rather than the bottom where sediment could accumulate. For overland flow monitoring the sampling hose was positioned slightly above the bottom of the pipe (Figure 3-10).

Sample collection was a collaborative effort between farm operators and researchers. The sampling strategy was to capture a range of hydrologic events at high frequency sample intervals throughout the year when tiles were flowing, including the winter months. In addition, individual grab samples were taken between events to characterize baseflow concentrations. Composite samples were used for tile drainage samples between October 2011 and May 2012. The decision to switch from composite samples to discrete samples was made in May 2012 so that P export dynamics within events could be carefully studied.

Sampling programs were adjusted throughout the study period to get coverage of the hydrograph for each sampled event. This involved adjusting the threshold water depth used to trigger or enable the AWS, and modifying water quality sampling intervals seasonally, and in some cases for individual storms. Different mechanisms were used to enable/trigger the AWS. During the first 10 months of the study period the AWSs were triggered manually prior to events. Ten (10) months into the study, a circuit was installed that allowed the Hach Fl900 to enable the AWS when a defined flow depth was reached. The triggering depth was adjusted seasonally to ensure the AWS would be triggered for all significant flow events. Following periods of zero flow, the trigger depth was lowered to 50 mm in the tile drain in order to capture the first flow events during the wet up. During the autumn, winter and spring seasons, the trigger depth was increased to approximately 75 mm in the tile drain because of the occurrence of baseflow during these periods.

Water quality sampling intervals were changed as needed to provide adequate coverage of discharge events (i.e. the entire hydrograph). In general, summer events were shorter in duration than autumn or winter events due to differences in storage in the soil profile. Consequently, sampling intervals were shorter during summer storms to provide full coverage of the storm hydrograph. Typically, the sampling interval used for tile flow events in summer rainstorms was between 1.5-3 hours, while longer drawn out events such as winter melts were sampled at 4-8 hour intervals. Two part sampling programs (stratified sampling) were used to collect samples at a higher frequency during the rising limb of the hydrograph, when pulses of P are known to occur (e.g., Dils and Heathwaite, 1999). Overland flow events were anticipated to have a shorter duration and were programmed to sample at a higher frequency 0.5-1.5 hours during the summer, and 1-4 hours in the winter. Two part programs (reduced frequency at later stages in the event) were also used for overland flow events if events were expected to be of shorter duration (<24 hours).

During the summer months samples were retrieved from the field within 24 hours of being collected to minimize the opportunity for bacterial growth within sample bottles in the AWS. However, samples were left in the AWSs for the length of the sampling program during colder months of the year (up to one week). This was possible because AWS were housed in sheds and thus shaded from the sun (and could not be warmed) and the cool air temperatures 'refrigerated' or 'froze' samples. All samples were frozen upon retrieval until processing and analysis.

The procedure used to collect, store and ship samples also changed within the study period to improve the efficiency of the process. Prior to January 2013, the procedure was as follows: Samples were transferred from 1000ml Isco bottles into 250ml acid-washed Nalgene bottles, then stored in a freezer prior to being shipped to the laboratory for processing and analysis. The 1000ml plastic bottles were then triple rinsed with distilled water before being reinstalled in the AWSs.

In January 2013, new bottle trays were installed to improve the efficiency of the retrieval procedure. The bottle trays held 24, 230 ml acid washed bottles. These bottles could be capped by the cooperators, removed from the AWSs, stored in a freezer prior to shipping to the laboratory. Cooperators would reload the AWS with a new set of acid washed bottles immediately following removal of collected samples. These improvements eliminated field washing of bottles and transferring of samples.

3.8 Processing Samples and Laboratory Analysis

At the laboratory, each sample was thawed and immediately processed. A 50 ml aliquot was filtered through a 0.45 µm cellulose acetate filter (FlipMate, Delta Scientific) and stored in the dark at 4^oC. A second 50 ml unfiltered aliquot was acidified with H_2SO_4 (0.2% H_2SO_4 final concentration). Unfiltered samples were digested (acid) for the analysis of Total Kjeldahl P. Both soluble reactive P (SRP) and Total P (TP) were determined using colorimetric analysis (ammonium-molybdate ascorbic-acid (Bran Luebbe AA3, Seal Analytical Ltd., G-175-96 Rev. 13 for SRP, Method No. G-188-097 for TP). Five percent of all samples were analyzed in duplicate and found to be within 6% of reported values. The detection limit for SRP analysis was 1 µg L^{-1} and 10 µg L^{-1} for TP analysis.

3.9 Data Analysis

3.9.1 Hydrograph Analysis

Tile drainage flow was separated into event based and non-event based flow. The start of an event was defined as a noticeable rise in the hydrograph following a rain or melt event. The end of the event was determined graphically using a simple two line graphical method described by Stuntebeck et al. (2008). Briefly, a straight line was drawn on the hydrograph during non-event flow following the event. A second line was drawn following the hydrograph from the point where the first straight line diverges from the hydrograph. The point where the second line diverges from the hydrograph was determined to be the end of an event. Events with multiple peaks were separated at the lowest flow rate between events, or, combined into a single event if separation could not be achieved reliably. Each event and non-event flow period was given a unique ID (e.g., Event1, or BF1). Overland flow events were identified based on the tile drainage event in which they occurred. Multiple peaks within overland flow events were considered one event.

3.9.2 Phosphorus Load Calculations

During the 18-month period, a number of individual events were captured, representing between 22-77% of the event-based flow among the sites (Site 1: n=14 (22%), Site 2: n=22 (77%), Site 3: n=17 (60%)). Loading for the captured tile and overland flow events was calculated by multiplying the concentration of a discrete or composite sample, by the average flow rate for the sample interval, multiplied by the length of time the sample represented:

$$Event\ Load = \left(\sum_{i=1}^{n} C_i\ \bar{Q}_p T_i\right)$$

Where C_i is the concentration (mg l^{-1}) of a discrete or composite sample. $\bar{Q}_p$ is the mean discharge (l s^{-1}) for the interval between samples. T_i is the time interval (s) represented by the sample. Following the determination of event loading, the Flow-Weighted Mean Concentrations for events were calculated as:

$$FWMC = \frac{Event\ Load}{\text{Event Discahrge}}$$

To complete the event load and FWMC calculations, it was necessary to estimate concentrations for certain intervals where samples were missing. These concentrations were estimated using one of the following interpolation methods: 1) using flow-concentration relationships from the event or events with similar conditions 2) using the mean concentrations of the nearest two samples, or 3) the concentration from the closest sample interval if the flow rates of the two intervals were similar.

Loads for uncaptured tile events were estimated using FWMCs from captured events that had similar event properties to the event. The approach taken in this study was based on the methodology used at the Discovery Farms in Wisconsin, described in Stuntebeck et al. (2008). Uncaptured tile events represented 23-78% of the data set and were not used in statistical analyses. If there were no captured events with similar properties (season, peak flow, total flow, and event type), then relationships between FWMC and total discharge, or cumulative flow since fertilization were used to estimate the event's FWMC. This method was appropriate due to the variability between events related to event type, ground conditions, the presence of overland flow, and the effect of fertilization, which resulted in a poor relationship between discharge and concentration. The load from uncaptured events was calculated as:

$$Event\ Load = Estimated\ FWMC * \mathrm{Event}\ Discharge$$

No overland flow events were missed entirely, although some sample intervals were missed due to equipment malfunctions.

Non-event flow (baseflow) loading was estimated differently because fewer samples were taken and flow was less variable. Periodic grab samples and samples taken at the tail end of events were used to calculate the average concentration for each non-event flow period when tiles were flowing. This average concentration was multiplied by the total flow for the period to determine the loading. Loading for the entire study period was calculated by adding the loads of individual events (tile and overland flow), and non-event-based loading.

3.10 Additional Surveys

3.10.1 Soil Survey

Soil samples were collected for chemical and textural analysis. Soil samples were collected from Site 3 and Site 1 sites in August 2011 following wheat harvest. Samples were collected from Site 2 in October 2011 following soybean harvest. Stratified random sampling was used to form composite samples representing five subsections within each catchment area, which were defined based on elevation and slope. Five sample locations were randomly chosen within each subsection. At each sample location, soil samples were collected at 8 depth intervals to capture P stratification (0-2.5cm, 2.5-5cm, 5-15cm, 15-30cm, 30-45cm, 45-60cm, 60-75cm, 75-90cm). Composite samples of each depth were created for each subsection in the field. The soil core was used to extract the first three depth intervals. Then a 2 1/4" diameter soil auger was used to sample to greater depths. Subsamples from specified depth intervals were thoroughly mixed in clean plastic pails and subsequently divided into two plastic bags in the field. One bag was sent to A & L Laboratories (London, Ontario, Canada) for the analysis of soil chemistry including soil test P (Olsen P) and the second subsample was kept for particle size analysis at Wilfrid Laurier University (Horiba LA950, ATS Scientific Inc., Canada).

3.10.2 Residue Cover

A residue cover survey was conducted at the start of the first growing season. Residue cover was measured using the line transect method described in Laflen et al.(1981). A minimum of five random 30m transects were selected in the field and the presence of crop residue was measured at 100 points (*i.e.*, residue identified at 60 points would mean %60 residue cover) The average percent cover calculated from all transects provided an estimate of percent residue cover for each field. In addition, regular field photos were taken to further characterize seasonal changes in residue cover.

4 Field-Scale Phosphorus Export via Tile Drainage and Overland Flow from Three Reduced Tillage Sites in Ontario

4.1 Introduction

Harmful algal blooms (HABs) and eutrophication are problematic globally (Norton, et al. 2012, Schindler, et al. 2012), and have been identified as a priority research area in the lower Great Lakes region of Ontario, Canada, particularly for Lake Erie (Great Lakes Commission Phosphorus Reduction Task Force 2012). The number of reported algal blooms in lakes in Ontario has increased since the mid-1990's (Winter, et al. 2011), and has been attributed in part to increased soluble reactive phosphorus (SRP) loading from tributaries to the Great Lakes (Great Lakes Commission Phosphorus Reduction Task Force 2012). There has been considerable effort to reduce phosphorus (P) export to the Great Lakes (Great Lakes Commission Phosphorus Reduction Task Force 2012, Scavia, et al. 2014) as well as other large water bodies both within North America (Schindler, et al. 2012) and Europe (Helin, et al. 2008). Agricultural fields have been recognized as an important non-point source of P to water bodies (Carpenter, et al. 1998) and a likely contributor to the increase in algal blooms observed (Great Lakes Commission Phosphorus Reduction Task Force 2012, Scavia, et al. 2014), which has resulted in public pressure to identify ways to manage these agricultural related losses.

Strategies to reduce P loading have evolved in recent years. Historically, overland flow was thought to be the primary export pathway for P leaving farm fields because of its capacity to erode soil (Sims, et al. 1998). However, it is now accepted that tile drainage is also an important pathway for P export (Sims, et al. 1998, Smith, et al. 2014). In fact, studies have shown that tiles can be a significant pathway for P loss (e.g. Sharpley and Syers, 1979; Smith, et al. 2014), and can be the dominant pathway in some situations (Gaynor and Findlay 1995).

Beneficial Management Practices (BMPs) such as no-till (NT) have been promoted extensively to reduce soil erosion via overland flow (Sims, Simard and Joern 1998). However, NT systems may be problematic for the loss of SRP both in surface runoff (Tiessen, et al. 2010, Elliott 2013, Hansen, et al. 2000) and in tile drainage (Michalak, et al. 2013) due to the stratification of P in surface soil (e.g. Sharpley, 2003) and/or increased connectivity between surface soils and tile drains due to enhanced macropore development (Sims, et al. 1998, Stamm, et al. 1998). In fact, increases in the loading of SRP to the Western end of Lake Erie has been attributed to the increased use of conservation tillage strategies (Kleinman, et al.

2011, Michalak, et al. 2013) as well as the surface broadcasting of fertilizers and autumn spreading (Michalak, et al. 2013). More recently there is an understanding that P export may be most effectively addressed by focusing management efforts on critical source areas of P, which are areas where there is an elevated source of P that is connected to surface water via some export pathway (Pionke, et al. 2000). Managing P export requires understanding how management practices increase the likelihood of P movement and the pathways where P is moved.

Gaps in our current understanding of P export are limiting our ability to manage P loss in Ontario. To properly identify critical source areas, we must understand the role of the export pathways. Research from other regions suggests export from tiles will also need to be addressed to reduce P export (e.g. Smith, et al. 2014). Tile drainage is used extensively in Ontario, yet the current tool used to assess the risk of P losses at the field scale, the Ontario P Index, does not account for tile drains as a pathway for P (Reid, et al. 2012). It is important to understand the relative contribution of overland flow and tile drainage to P export because this allows managers to properly account for the risks of P losses (Reid, et al. 2012). Another gap is our limited understanding of how reduced tillage (RT) systems perform as a BMP in Ontario year round. Much of the previous work on NT and tile drained systems (e.g. Smith et al. 2014) has not included sampling through the winter months. However, tillage systems do perform differently in certain climates. In Manitoba, where the flow regime is dominated by spring runoff on frozen soils, Tiessen et al. (2010) found that NT systems increased P losses relative to conventional. Soluble P losses were the dominant form during snowmelt, which accounts for the majority of annual site discharge, and using conservation tillage increased concentrations of soluble P and the volume of runoff in that environment. On the same sites, Lui, et al. (2014) later showed that converting no-till systems to rotational tillage systems (aggressively tilled every other year) reduced P export significantly. However, the work done by Tiessen et al. (2010) and Lui et al. (2014) focussed on surface runoff and did not include tile drain effluent. Understanding how export differs seasonally will help us understand which seasons have greater potential for P export, and will help identify opportunities to manage P applications and tillage to reduce overall P losses.

In this study, runoff and P export (Total, Soluble Reactive, and Particulate+Soluble Unreactive) in tile drainage and overland flow from three reduced tillage fields in Ontario were examined over an 18-month period. The specific objectives of the study were to:

a. Quantify seasonal and annual discharge from tile drainage and overland flow at the field scale.
b. Quantify the seasonal and annual speciation and mass of P loss in tile drainage and overland flow.

4.2 Methods

The methods used for this chapter were described fully in Chapter 3.

4.3 Results

4.3.1 Meteorological Conditions

Conditions experienced over the study period varied between sites and compared to long-term (30 year) mean conditions recorded at nearby weather stations (Environment Canada, 2014) (Figure 4-1). At all sites, the daily mean temperatures were warmer during the winter months than the long-term means (Figure 4-1). Site 1 remained cooler in winter than Sites 2 and 3, which is typical for the three study regions (Figure 4-1).The early part of the summer of 2012 was also warmer than the long-term 30-year mean at all the sites (Figure 4-1), although temperatures did not differ as much between sites in the summer months as they did during the winter.

Precipitation varied from long-term means and among sites. All sites had below average precipitation in the 2012-2013 year (Figure 4-1), with 923, 721, and 833 mm of precipitation falling at Sites 1,2 and 3, respectively, compared to long term means of 1004, 1024 and 1243 mm. The driest period occurred between March and August 2012, when Site 1, Site 2 and Site 3 had just 81%, 56% and 71% of the normal precipitation for the period, respectively. In contrast, precipitation in the 2012-2013 winter-spring period was near normal or above normal at all sites (Figure 4-1). Overall the sites experienced similar differences from the long-term means for their respective areas.

The snow water equivalent (SWE) of the snowpack prior to major melts varied among the sites and between years. During the first non-growing season (NGS), Site 1 had greater snowfall and a greater snowpack prior to the spring freshet than Site 2 and Site 3 (visual observation). Frequent thaw events resulted in little snow accumulation over the first NGS at Site 2 and Site 3. Site 1 had greater snow accumulation than Sites 2 and 3 in the second NGS, and, all 3 sites had greater snow accumulation in the second NGS relative to the first NGS due to cooler temperatures. The snow survey completed in March 2013 at all sites confirmed the difference in SWE prior to the final melt at the sites (Table 4-1). Total snowfall at the sites differed (Site 1 > Site 3 > Site 2) (Table 4-1). However, due to the thaw events during the second NGS, both Site 2 and Site 3 had even less SWE present prior to the final melt (Table 4-1). Generally, the southwestern sites (Site 2 and Site 3) experienced warmer winters with more frequent thaw events relative to the eastern Ontario site (Site 1).

Table 4-1 Results of snow surveys conducted at all sites between March 8-11, 2013 prior to the final spring snow melt. Snowfall prior to survey was calculated from local Environment Canada Weather Station data.

Site	Date	Snow Fall Prior to Survey SWE (mm)	Survey SWE (mm)
1	2013-03-11	253	148
2	2013-03-08	100	42
3	2013-03-08	214	69

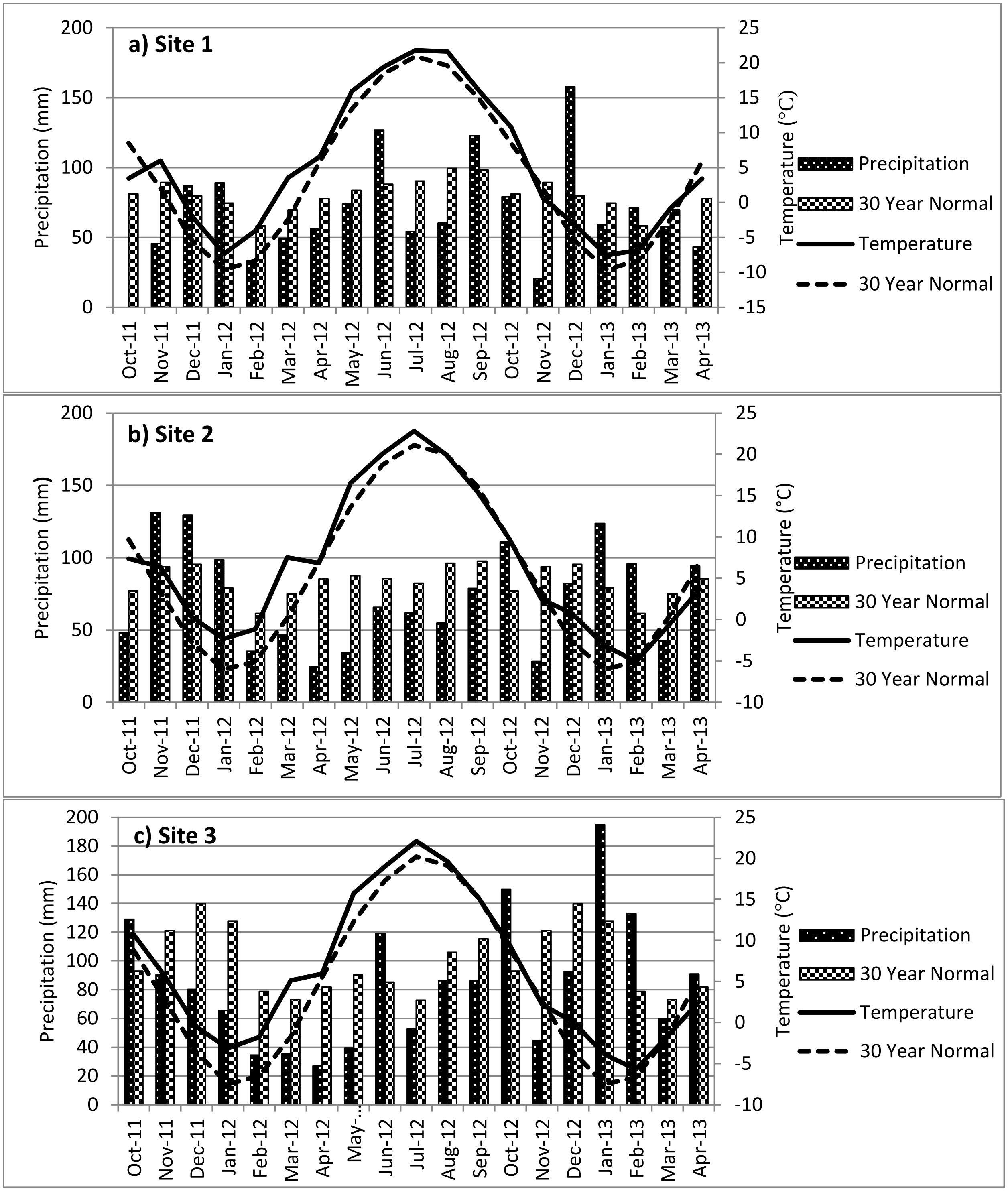

Figure 4-1 Monthly precipitation and average temperature for Study Site 1 (a), Site 2 (b) and Site 3 (c) compared to 30 year normal (means). Note that monitoring at Site 1 did not begin until October 25, 2011, thus observed data represents a partial month.

4.3.2 Runoff Generation

4.3.2.1 Temporal Variability in Runoff Generation

Site discharge and responsiveness of hydrologic pathways varied with time. Tiles responded to precipitation and melt events throughout the year, but were much less responsive during the summer months (Figure 4-2, Figure 4-3 and Figure 4-4) when water table position was lower (Figure 4-5). Tile discharge typically responded quickly following rain events, increasing from dry conditions or low flow conditions to as high as 9-21 l s^{-1} in larger events. Tiles would sustain baseflow for long periods during the NGS, but would dry up between events during the GS (Figure 4-2, Figure 4-3 and Figure 4-4). Overland flow events were rare (Average: 18% of all events included overland flow) and were restricted to large rainfall events or snowmelts. These events occurred primarily in the NGS, with just one overland flow event occurring during the GS (Site 1) (Figure 4-2, Figure 4-3 and Figure 4-4). The overland flow during the GS at Site 1 occurred following 46 mm of intense rainfall over a period of 75 minutes. Peak overland flow rates (20-110 l s^{-1}) were more variable between sites relative to tile drainage. The highest rates of overland flow were seen during the summer storm at Site 1 (21 l s^{-1}), a rain on frozen soil event at Site 2 (20 l s^{-1}), and rain on snow event at Site 3 (110 l s^{-1}). All events that included overland flow also included a large tile drainage response (Figure 4-2, Figure 4-3 and Figure 4-4).

Discharge totals and overall runoff ratios differed between the sites during the study period. Annual discharge (May 2012-April 2013) was 577, 277, and 375 mm for Site 1, Site 2 and Site 3, respectively (Table 4-4). Annual (May 2012-April 2013) runoff ratios were 0.62, 0.32 and 0.33 for Site 1, Site 2 and Site 3, respectively (Figure 4-6). Site 1 had greater combined annual discharge and a higher runoff ratio than the other sites.

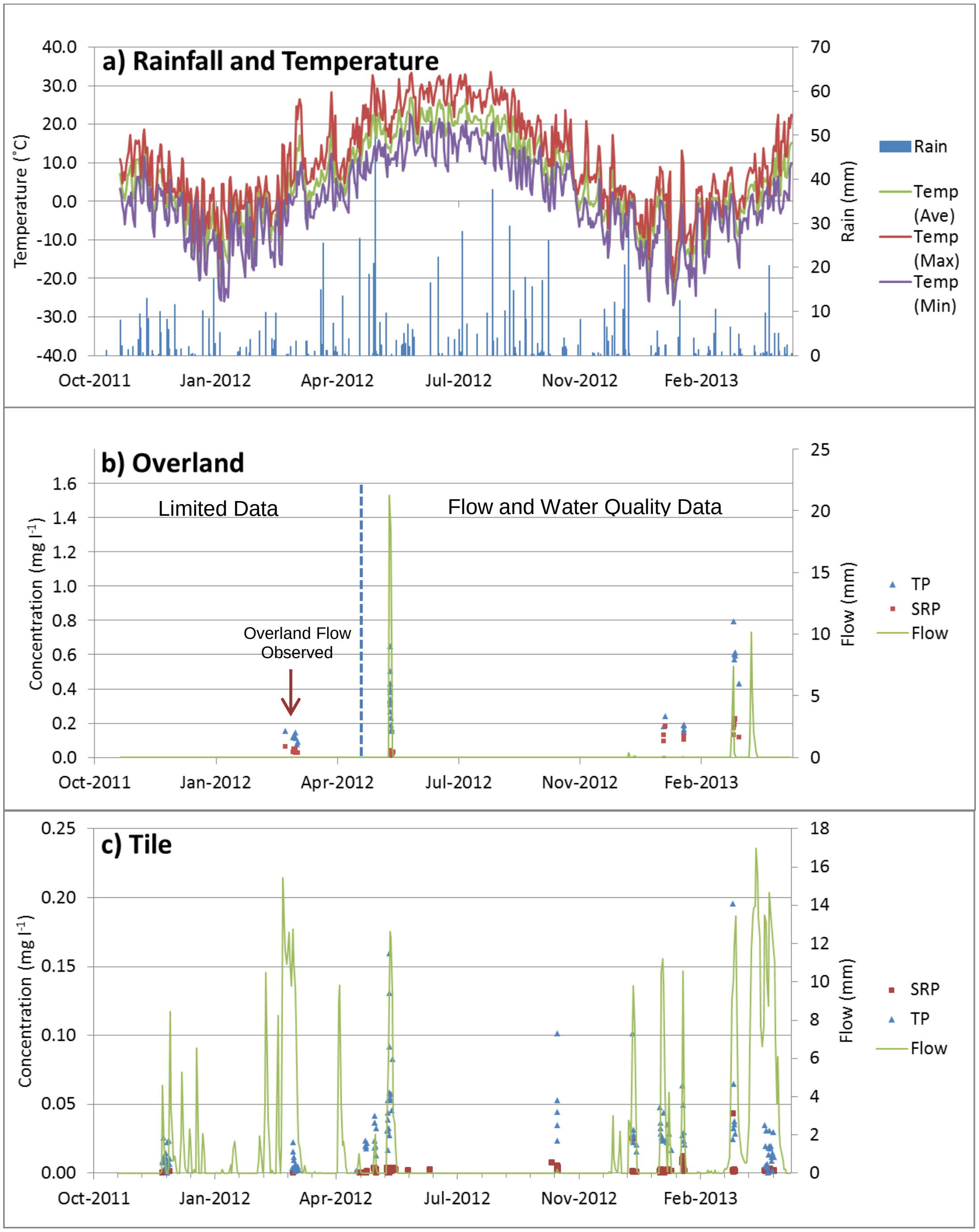

Figure 4-2 Site 1 Daily Temperature (Average, Minimum and Maximum) and Rainfall (a), Overland Flow (b) and Tile Drainage (c) Discharge and TP (Blue Triangles) and SRP (Red Squares) Concentrations. Values below detection limit are not shown.

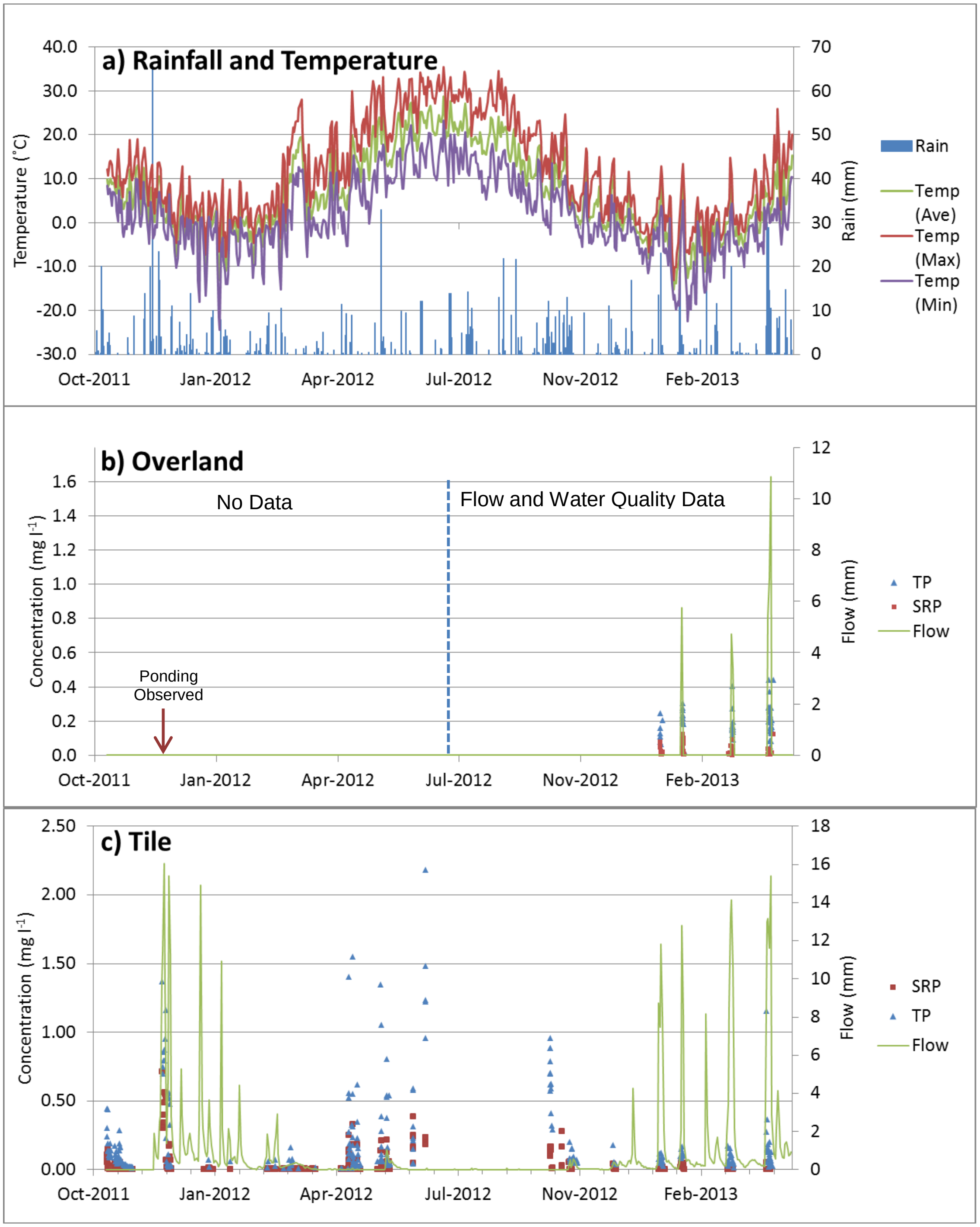

Figure 4-3 Site 2 Daily Temperature (Average, Minimum and Maximum) and Rainfall (a), Overland Flow (b) and Tile Drainage (c) Discharge and TP (Blue Triangles) and SRP (Red Squares) Concentrations. Values below detection limit are not shown.

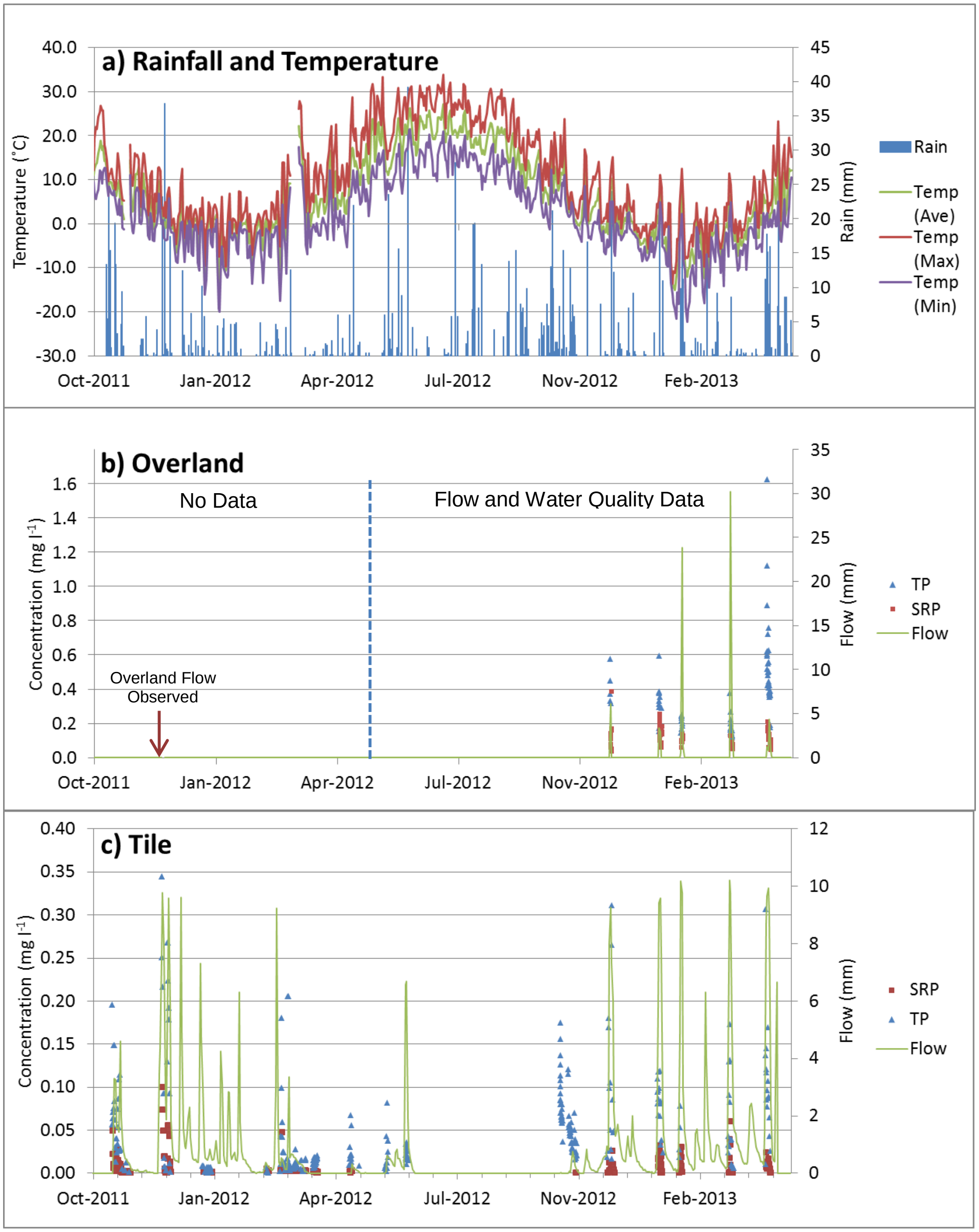

Figure 4-4 Site 3 Daily Temperature(Average, Maximum and Minimum) and Rainfall (a), Overland Flow (b) and Tile Drainage (c) Discharge and TP (Blue Triangles) and SRP (Red Squares) Concentrations. Values below detection limit are not shown.

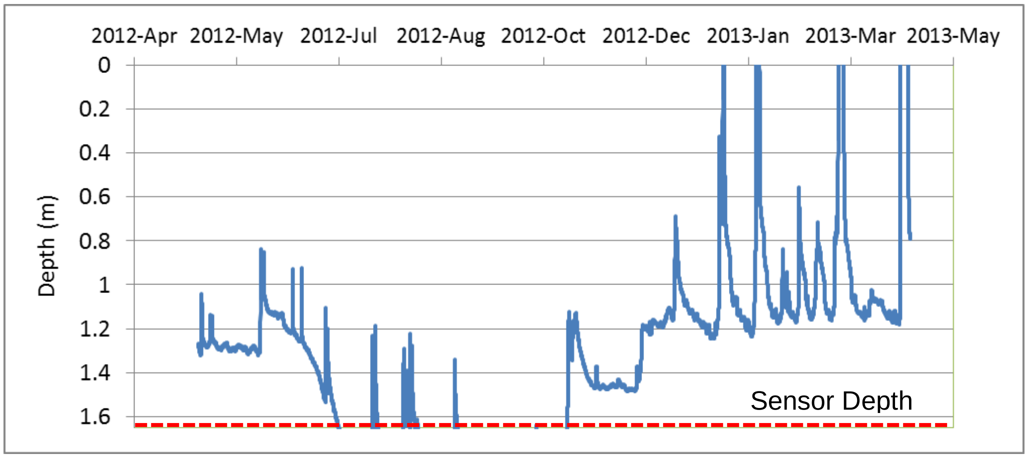

Figure 4-5 Water table depth from the surface for Site 2. A similar seasonal pattern was observed at all the sites. The sensor was 1.68 m below the surface so no readings were available below this depth.

Figure 4-6 Seasonal (NGS and GS) and annual, runoff ratios for all sites.

4.3.2.2 Seasonal Tile Drainage and Overland Flow Runoff

Generally, the sites had similar seasonal runoff patterns. Between May 2012 and April 2013, the majority of combined runoff occurred during the NGS (83-98%), while a smaller amount of flow occurred during the GS (2-17%). In terms of the separate pathways, only 2-11% of annual tile flow occurred during the GS period. Overland flow occurred exclusively during the NGS at Site 2 and Site 3. Whereas the single overland flow event at Site 1 during the GS accounted for 56% of the annual overland flow at the site. In general, runoff precipitation ratios were greater during the NGSs (>0.46) relative to the GS (0.02 – 0.22) (Figure 4-6). Site 1 had greater NGS and GS runoff:precipitation ratios (GS: 0.22, NGS: 0.99) relative to Sites 2 (GS: 0.02, NGS: 0.47) and 3 (GS: 0.06, NGS: 0.46) (Figure 4-6).

The distribution of runoff (tile and overland) within seasons differed between the sites (Figure 4-7, Figure 4-8 and Figure 4-9). In both NGS's, site discharge at Site 1 was dominated by spring snowmelt; however, different distributions of runoff occurred during the NGSs at Site 2 and Site 3. During NGS1 discharge at Site 2 and Site 3 was dominated by November-December events, while during NGS2, discharge at these sites came from a series of rain and thaw events between December and April. GS monthly discharge totals were relatively low at all sites, with the exception of June 2012 at Site 1 (Figure 4-7, Figure 4-8 and Figure 4-9). Overall, the flow regime at Site 1, which was dominated by spring melts, differed from the flow distributions observed at the southwestern Ontario sites (Site 2 and Site 3).

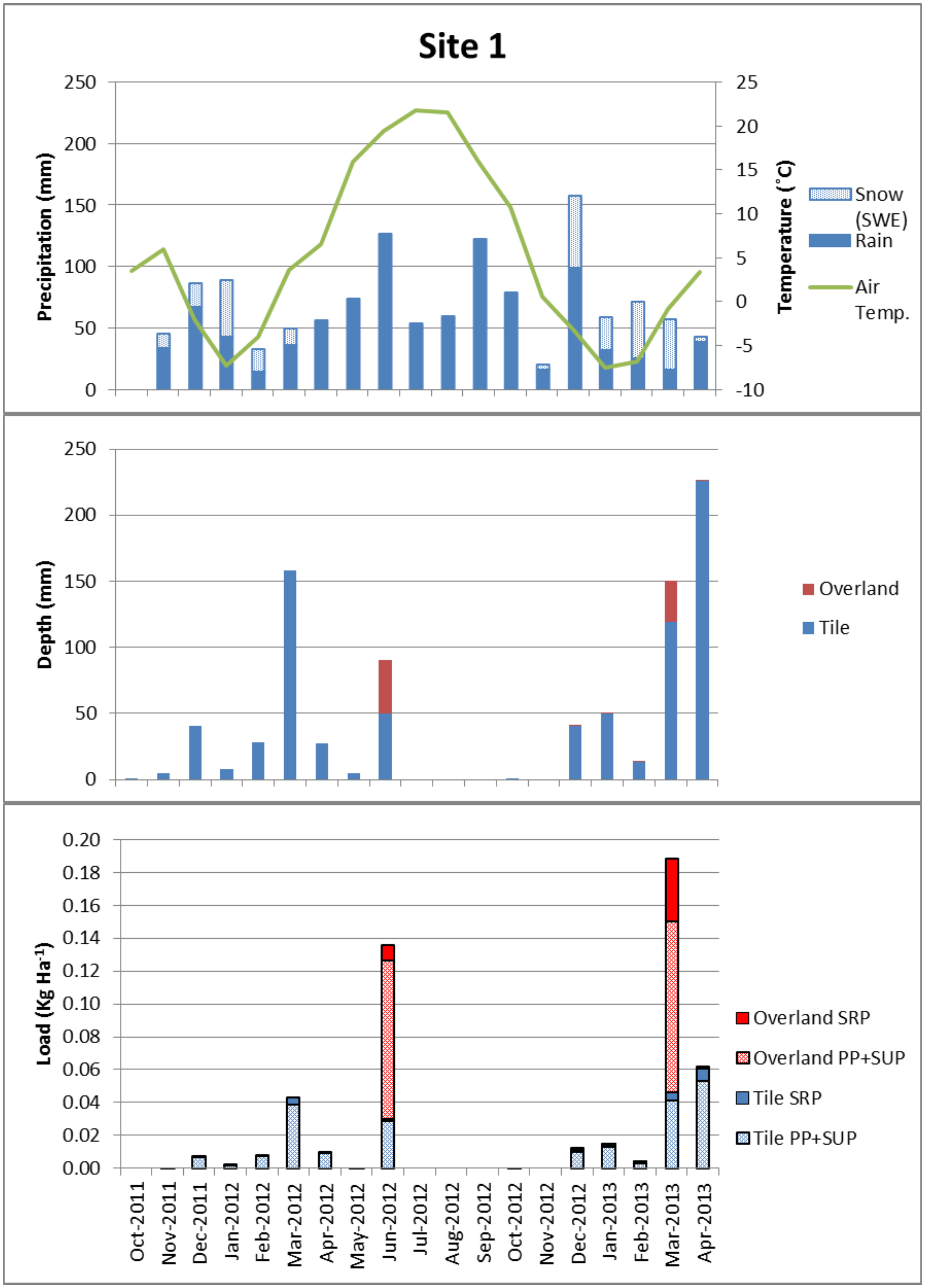

Figure 4-7 Site 1 Monthly Precipitation and Temperature, Tile and Overland Flow Discharge, and Phosphorus Loading.

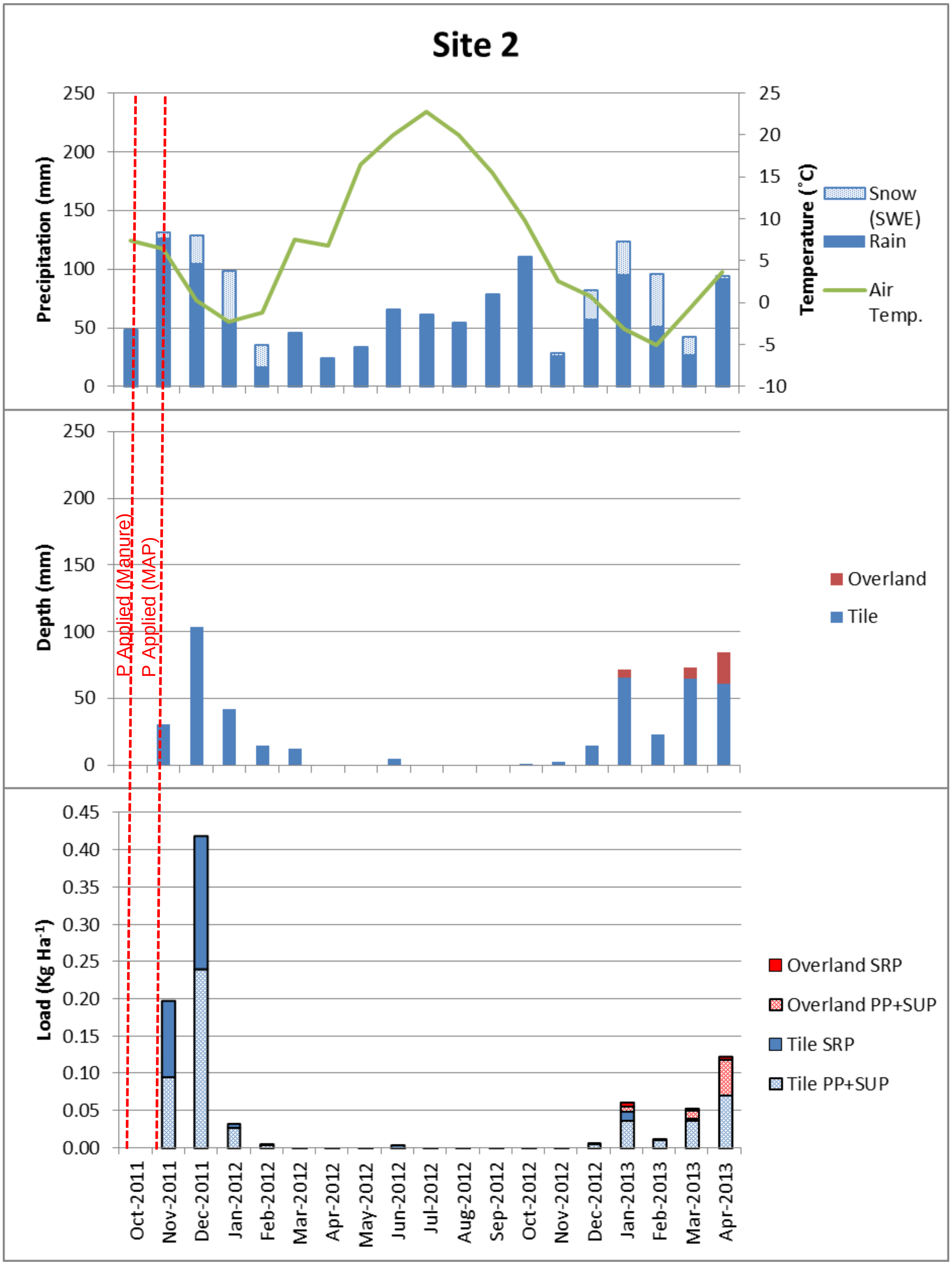

Figure 4-8 Site 2 Monthly Precipitation and Temperature, Tile and Overland Flow Discharge, and Phosphorus Loading.

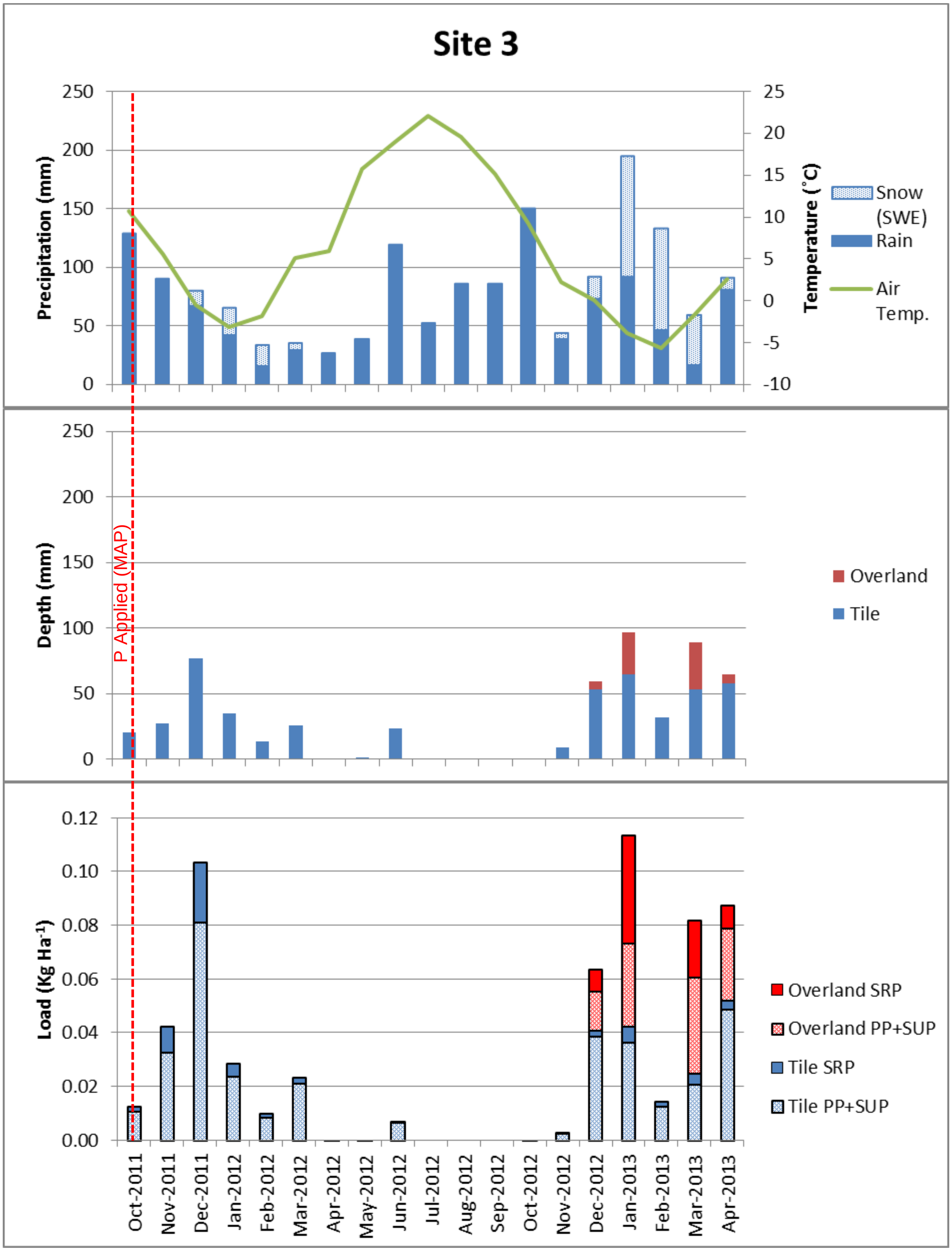

Figure 4-9 Site 3 Monthly Precipitation and Temperature, Tile and Overland Flow Discharge, and Phosphorus Loading.

4.3.3 Phosphorus Concentrations and Export in Tile Drain Effluent and Overland Flow

Tile drainage samples were collected from all three sites for an 18 month study period. The number of samples and number of captured events varied between sites due to event opportunities, equipment failure, as well as cooperator and researcher availability. On average there were 47 possible runoff events at each site over the study period. On average, 19 events were captured at each site. The captured events ranged in magnitude and antecedent conditions, and no large events were missed. No events with overland flow were missed entirely, although there were some equipment malfunctions within some of these events. The sampling program captured a range of event types. The variation between event types is discussed in Chapter 5.

Over the study period, concentrations of TP and SRP in tile drainage showed temporal and spatial variability (Table 4-2) (Figure 4-2, Figure 4-3 and Figure 4-4). SRP concentrations were variable at the sites (median, range: Site 1: 0.002, 0.001-0.044 mg l^{-1} ; Site 2: 0.046, 0.001-0.712; Site 3: 0.002, 0.001-0.100 mg l^{-1}). A range of TP concentrations were observed at the sites (median, range: Site 1: 0.02, 0.01-0.20 mg l^{-1} ; Site 2: 0.05, 0.01-2.17; Site 3: 0.02, 0.01-0.35 mg l^{-1})(Table 4-2). P concentrations were typically lower in baseflow conditions (0.001-0.002 mg l^{-1} SRP and 0.01-0.02 mg $^{l-1}$ TP) and higher and more variable during event based flow (0.003-0.086 mg l^{-1} SRP and 0.032-0.25 mg l^{-1} TP) (Table 4-2). The greatest discrete tile SRP concentrations occurred during a significant rainfall event in late November 2011 at Site 2 (0.712 mg/l) and Site 3 (0.1 mg/l). These elevated concentrations occurred within major flow events (November 30, 2011) that happened between 2.5 and 8 weeks after applications of P were made at the two sites (Liquid laying hen manure and MAP at Site 2; MAP at Site 3) (Figure 4-3 and Figure 4-4). TP concentrations were also elevated following fertilization as well as during major flow events (Figure 4-2, Figure 4-3 and Figure 4-4). TP concentrations were elevated at all sites during the initial wet up events in the fall of 2012 (Figure 4-2, Figure 4-3 and Figure 4-4). Concentrations of TP and SRP at Site 2 were elevated during most GS events (May-July), this is assumed to be related to site specific contamination from the farmstead's driveway as there is a surface inlet from the driveway that discharges into the tile system. The flow weighted mean concentrations (FWMCs) of TP and SRP for captured events varied across all the sites (0.001-0.442 mg l^{-1} SRP and 0.01-1.52 mg l^{-1} TP) (Table 4-2). Overall, Site 2 had the highest event flow FWMCs for SRP and TP, while Site 1 had the lowest (Table 4-2).

Overland flow samples were not collected at the same intensity throughout the study period because flow monitoring equipment was not installed for overland flow in NGS1. Several grab samples were collected in NGS1, but no flow data was available for this time period. Sampling equipment was installed in May 2012, allowing for events to be sampled intensively.

Concentrations in overland flow were higher than concentrations in tile, and like tiles, showed temporal variability within and among events (Table 4-2) (Figure 4-2, Figure 4-3 and Figure 4-4). In general, discrete overland flow concentrations ranged 0.001-0.720 mg l^{-1} SRP and 0.021-1.60 mg l^{-1} TP (Table 4-2). The highest concentrations of SRP in overland flow were from two grab samples taken in NGS1, following P fertilization at Site 3 (SRP 0.72 mg l^{-1}). TP concentrations were elevated in the grab samples following fertilization (Site 3: 1.35 mg l^{-1}), but also in events where rain fell on bare soil in April (e.g., Site 3: 1.6 mg l^{-1}) (Figure 4-4). Event P FWMCs for the sites ranged between 0.015-0.151 mg l^{-1} SRP and 0.16-0.58 mg l^{-1} TP (Table 4-2).

Table 4-2 Median and range of soluble reactive (SRP) and total phosphorus (TP) concentrations in discrete samples from tile drainage and overland flow over the study period. Mean and range of flow weighted mean concentrations during non-event flow (tile only) and event-based flow for all sites (tile and overland flow). Overland flow grab samples taken before May 2012 are included below the table.

Tile Drainage										
	Discrete Samples				Non-Event FWMC		Overall Event FWMC		Range of Event FWMC	
Site	SRP mg l^{-1}		TP mg l^{-1}		SRP mg l^{-1}	TP mg l^{-1}	SRP mg l^{-1}	TP mg l^{-1}	SRP mg l^{-1}	TP mg l^{-1}
	Median	Range	Median	Range						
1	0.002	0.001-0.044	0.02	0.01-0.20	0.001	0.01	0.003	0.03	0.001-0.006	0.01-0.07
2	0.004	0.001-0.712	0.05	0.01-2.17	0.002	0.01	0.086	0.25	0.001-0.442	0.01-1.51
3	0.002	0.001-0.100	0.02	0.01-0.35	0.001	0.02	0.017	0.11	0.001-0.054	0.01-0.21
Overland Flow										
	Discrete Samples						Overall Event FWMC		Range of Event FWMC	
Site	SRP mg l^{-1}		TP mg l^{-1}				SRP mg l^{-1}	TP mg l^{-1}	SRP mg l^{-1}	TP mg l^{-1}
	Median	Range	Median	Range						
1	0.035	0.011-0.224	0.233	0.15-0.79			0.066	0.35	0.023-0.151	0.17-0.58
2	0.01	0.001-0.12	0.187	0.021-0.44			0.031	0.21	0.015-0.107	0.16-0.23
3	0.123	0.042-0.252	0.351	0.07-0.1.60			0.096	0.23	0.059-0.151	0.16-0.50

* Grab samples from March 2012 melt at Site 1 (Mean: 0.031 mg l^{-1} SRP, 0.12 mg l^{-1} TP)

* Grab samples from November 2011 event at Site 3 (0.72 mg l^{-1} SRP, 1.353 mg l^{-1} TP)

Seasonal (NGS1, GS1, NGS2) and annual (May 2012-April 2013) Tile P FWMC's were determined for the study period. Tile seasonal FWMCs differed across years, and across sites (Table 4-3). There were no seasonal trends consistent between the sites. At Sites 2 and 3, the tile FWMC of SRP and TP was higher in NGS1, than the subsequent GS1 and NGS2. This trend was not seen at Site 1. The 2012-2013 annual P FWMCs for the sites ranged between 0.003-0.007 mg l^{-1} SRP and 0.03-0.08 mg l^{-1} TP (Table 4-3).

Overland flow annual and seasonal P FWMC were calculated for the period when sites were equipped to monitor overland flow. Overall, annual (2012-2013) overland flow P FWMC ranged between 0.031-0.096 mg l^{-1} SRP and 0.21-0.34 mg l^{-1} TP (Table 4-3). There was little data to compare seasonal values as there was only one overland flow event during the GS. Based on the one summer event at Site 1, the GS had lower FWMCs of SRP and TP compared to the NGS (Table 4-3). In NGS2, Site 2 (0.031 mg l^{-1}) had a lower SRP FWMC relative to Site 1 (0.122 mg l^{-1}) and Site 3 (0.096 mg l^{-1}). The NGS2 TP FWMC ranged between 0.21-0.45 mg l^{-1} (Table 4-3). Overall, there was limited overland flow data to compare seasonal differences.

Table 4-3 Summary of tile drainage and overland flow seasonal (NGS1, GS1 and NGS2) flow weighted mean concentrations (FWMC) of SRP and TP (mg l^{-1}), and SRP:TP ratio. Overland flow was not monitored fully in NGS1. No flow occurred during GS1 at Site 2 and Site 3.

Site	Tile Drainage				Overland Flow			
	NGS1	GS1	NGS2	Annual	NGS1	GS1	NGS2	Annual
				(FWMC SRP mg l^{-1})				
Site 1	0.002	0.003	0.003	0.003	NA	0.023	0.122	0.067
Site 2	0.139	0.025	0.007	0.007	NA	No Flow	0.031	0.031
Site 3	0.021	0.001	0.007	0.006	NA	No Flow	0.096	0.096
				(FWMC TP mg l^{-1})				
Site 1	0.03	0.06	0.03	0.03	NA	0.26	0.45	0.34
Site 2	0.32	0.14	0.08	0.08	NA	No Flow	0.21	0.21
Site 3	0.11	0.03	0.07	0.06	NA	No Flow	0.23	0.23
				(SRP:TP)				
Site 1	0.08	0.05	0.11	0.10	NA	0.09	0.27	0.19
Site 2	0.43	0.17	0.09	0.09	NA	No Flow	0.15	0.15
Site 3	0.19	0.05	0.10	0.10	NA	No Flow	0.42	0.42

P export from tile drainage was calculated for an 18 month period, while overland export was only calculated for the final 12 months of the study period. P export was episodic at the sites. At all sites, the majority of P was exported during event-based flow. For example, at Site 1 during the 2012-2013 water year, 97% of annual TP export occurred in event-based flow. Furthermore, a large percentage of annual export typically came from a small number of major discharge events. For example, at Site 3, 63% of 2012-2013 annual TP export came from three events (a January rain on snow event, a March melt, and an April rain).

Seasonal variability in P export was observed at the sites (Figure 4-7, Figure 4-8 and Figure 4-9). P export was distributed similarly to discharge which was discussed earlier. Tile P export occurred primarily in the NGS, except for one significant event at Site 1 in June 2012 (Figure 4-7, Figure 4-8 and Figure 4-9). Site 1 was the only site to experience overland flow export during the GS (Figure 4-7, Figure 4-8 and Figure 4-9). Similar to discharge, P export occurred at different times during the NGS, with the spring melt playing a greater role at Site 1 compared to Site 2 and Site 3 which had more frequent rain and melt events through the winter (Figure 4-7, Figure 4-8 and Figure 4-9).

Seasonal (NGS1, GS1 and NGS2) and annual (May 2012-April 2013) totals for tile and overland P export were calculated for the study period. There were obvious differences in tile P export between NGS1, and the following hydrologic year. In the first NGS, tile P export at the sites for SRP and TP ranged between 0.006-0.285 kg SRP ha^{-1} and 0.072-0.656 kg TP ha^{-1}, respectively. In the following year, annual tile P export totals (GS1 + NGS2) for SRP and TP ranged between 0.017-0.018 kg SRP ha^{-1} and 0.169-0.190 kg TP ha^{-1}, respectively (Table 4-4). Annual overland flow SRP and TP export at the sites ranged between 0.011-0.078 kg SRP ha^{-1} and 0.077-0.250 kg TP ha^{-1} (Table 4-4). At Site 1, the only site with GS P export, GS TP export (0.105 kg ha^{-1}) was similar to NGS2 P export (0.144 kg ha^{-1}). NGS2 export at the three sites ranged between 0.077 – 0.186 kg TP ha^{-1}, with the greatest export occurring at Site 3. NGS2 losses of SRP were between 0.011-0.078 kg ha^{-1}, again with the greatest export occurring at Site 3 (Table 4-4). Overall, combined annual losses (tile + overland) based on the May 2012 – April 2013 water year, were greatest at Site 1. The annual export totals from the three sites were each less than the NGS1 total from Site 2.

Seasonal tile discharge was plotted against tile P export totals to examine the relationship between seasonal tile discharge and seasonal P export. There is a positive trend between Seasonal P Export and Seasonal Discharge at all sites when seasons affected by fertilization are ignored (NGS 1 at Site 2 and Site 3) (Figure 4-10). The influence of fertilization is more obvious at Site 2, where the loads of SRP (0.285 kg ha^{-1}) and TP (0.656 kg ha^{-1}) during NGS1 were the highest observed at any site during the study period (Table 4-4). SRP and TP loads during NGS2 were less than loads during NGS1, despite having higher total discharge in NGS2 (Table 4-4). In contrast, Site 1, which was not fertilized during the study period, had greater SRP and TP losses in NGS2, corresponding to the higher discharge for the season (Table 4-4). This plot shows that fertilization can cause seasonal variability in P loads beyond what is explained by discharge alone.

Table 4-4 Seasonal (NGS1, GS1 and NGS2) and Annual (GS1+NGS2) runoff, phosphorus (SRP, PP+SUP and TP) loading totals and partitioning.

Site	Tile Drainage					Overland Flow					Combined		
	NGS1	GS1	NGS2	Annual	%	NGS1	GS1	NGS2	Annual	%	GS1	NGS2	Annual
						Runoff (mm)							
Site 1	267	55	450	505	**87%**	NA	40	32	72	**13%**	95	482	577
Site 2	205	6	234	239	**87%**	NA	0	37	37	**13%**	6	271	277
Site 3	200	25	269	294	**78%**	NA	0	81	81	**22%**	25	350	375
						SRP (kg ha^{-1})							
Site 1	0.006	0.001	0.015	0.017	**26%**	NA	0.009	0.039	0.048	**74%**	0.011	0.055	0.065
Site 2	0.285	0.001	0.016	0.017	**60%**	NA	0.000	0.011	0.011	**40%**	0.001	0.027	0.029
Site 3	0.043	0.000	0.018	0.018	**19%**	NA	0.000	0.078	0.078	**81%**	0.000	0.095	0.096
						PP+SUP (kg ha^{-1})							
Site 1	0.066	0.030	0.122	0.152	**43%**	NA	0.096	0.105	0.201	**57%**	0.126	0.227	0.354
Site 2	0.371	0.007	0.166	0.172	**72%**	NA	0.000	0.066	0.066	**28%**	0.007	0.231	0.238
Site 3	0.179	0.007	0.161	0.167	**61%**	NA	0.000	0.109	0.109	**39%**	0.007	0.269	0.276
						TP (kg ha^{-1})							
Site 1	0.072	0.032	0.138	0.169	**40%**	NA	0.105	0.144	0.250	**60%**	0.137	0.282	0.419
Site 2	0.656	0.008	0.181	0.190	**71%**	NA	0.000	0.077	0.077	**29%**	0.008	0.259	0.267
Site 3	0.221	0.007	0.178	0.185	**50%**	NA	0.000	0.186	0.186	**50%**	0.007	0.364	0.371

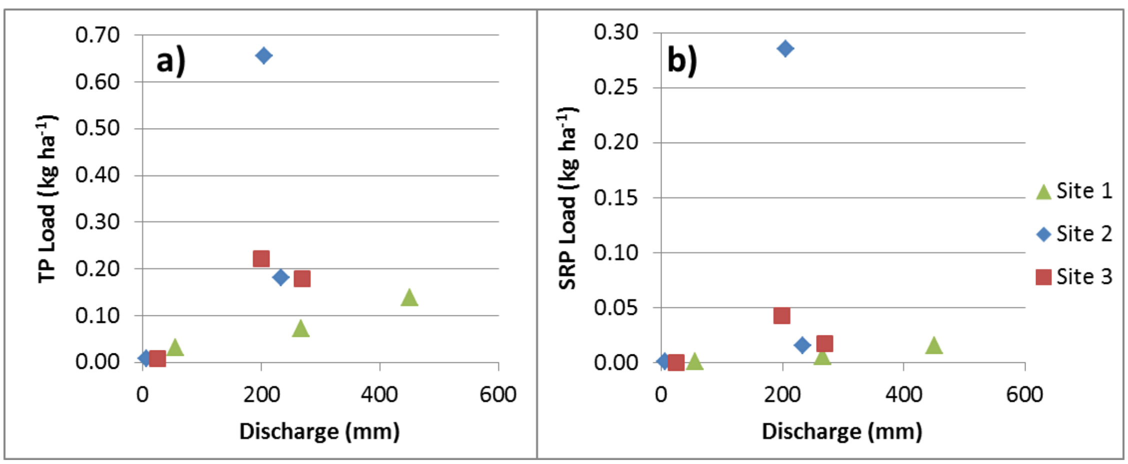

Figure 4-10 Seasonal Tile TP Load vs Seasonal Tile Discharge (a) and Seasonal Tile SRP Load vs Seasonal Tile Discharge (b) for Site 1 (green triangle), Site 2 (blue diamond) and Site 3 (red square).

4.3.4 Partitioning of Runoff and Phosphorus Export between Tile Drainage and Overland Flow

The partitioning of runoff between overland flow and tile drainage was calculated for the May 2012-April 2013 water year (Table 4-4). Tile drainage was the primary flow pathway at all sites. Between May 2012 and April 2013, tile drainage accounted for 78%-87% of annual runoff (Table 4-4). Thus, annually overland flow contributed 13%-22% to total flow from the sites. Even though overland flow events were rare during the study period (i.e.18% of all runoff events had overland flow), they made notable contributions to annual flow at all sites. In terms of annual discharge, tile drains were the dominant pathway for water discharge from fields between May 2012 and April 2013 at all sites.

Total site P export was calculated as the sum of tile drainage and overland flow export. The partitioning of total site losses was calculated for the period between May 2012 and April 2013. During this period total site losses were similar among the sites (0.267-0.419 kg TP ha^{-1}) (Table 4-4). The partitioning of SRP and TP export between tile and overland flow pathways (2012-2013) was different than the partitioning of discharge, and also varied between the sites (Table 4-4). Annually, tile drainage was the primary export pathway for TP export at Site 2 (71%), an equal contributor at Site 3 (50%) and Site 1 (40%) (Table 4-4). In terms of SRP, overland flow was the dominant export pathway at Site 1 (74%) and Site 3 (81%), while contributing less to annual export at Site 2 (40%) (Table 4-4). Tiles were a major contributor to TP export at all sites, but overland flow was more important in terms of SRP export at two of the three sites.

The speciation of P varied seasonally and between flow pathways (Figure 4-11). PP+SUP was the dominant fraction of P in tile drainage at all sites, with some seasonal variation (Figure 4-11). The SRP:TP ratios in tiles at Site 2 and Site 3 during NGS1 (Site 2 (0.43) and Site 3 (0.19)) were greater than the ratios found in the following year, (0.10 at both sites) (Table 4-3). SRP:TP ratios in tiles at Site 1 and Site 3 during the second NGS were greater than in ratios at the sites during the GS, but the opposite trend was observed at Site 2 (Table 4-3). PP+SUP was the dominant fraction in overland flow as well. However, the portion found as SRP was greater relative to tile drainage (Figure 4-11). The SRP fraction was greater in the NGS2 than in the GS at Site 1 (Figure 4-11). Overall, the speciation of P differed between the pathways, with higher SRP content in overland flow relative to tile drainage.

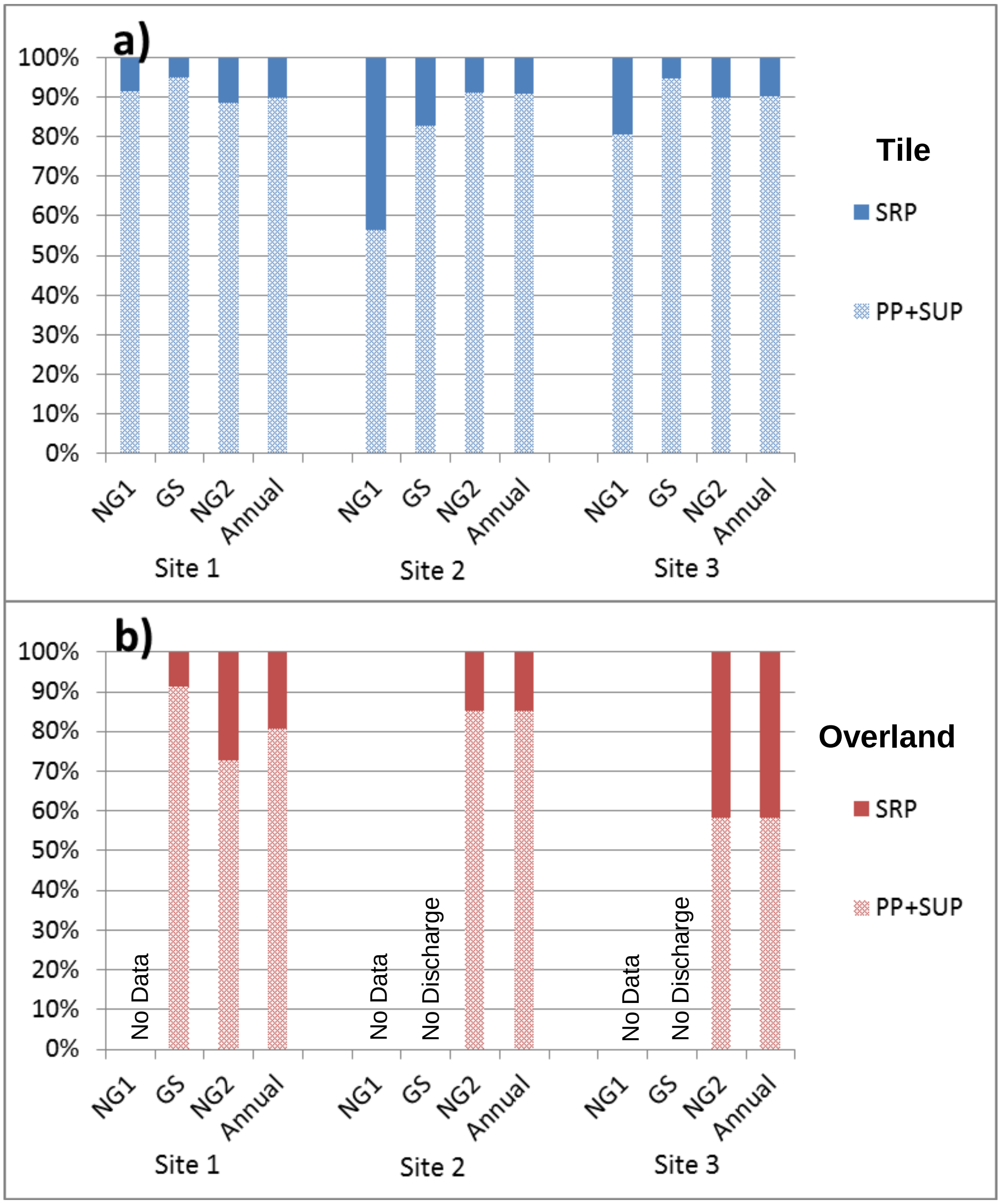

Figure 4-11 Seasonal and Annual speciation of phosphorus (SRP and PP+SUP) in Tile Drainage (a) and Overland Flow (b) for Site 1, Site 2 and Site 3. Note there was no overland flow analysis for NGS1, and no overland flow occurred during GS at Site 2 and Site 3.

4.4 Discussion

4.4.1 Spatial and Temporal Differences in Runoff from Fields

The seasonal distribution of flow was consistent with other studies in similar climates. The seasonal distribution of flow was representative of the average distribution for the area (Ontario Ministry of the Environment: Environmental Monitoring and Reporting Branch 2012), even though it was an atypical period (low GS precipitation) (Figure 4-1). While there was evidence that major flow events can occur during the GS (i.e., June 2012 at Site 1), overall, the flow pathways were more active during the NGS. Annual discharge is often dominated by the NGS in colder temperate regions as lower evapotranspiration and higher antecedent moisture conditions result in greater runoff ratios during this period (Hirt, et al. 2011). This was the pattern observed at all sites (Figure 4-6), and it is consistent with other studies from cold climates (e.g. Eastman, et al. 2010). This study confirms that export pathways are most active in the NGS, but also that extreme GS events can also have an influence on annual discharge in this region.

The differences in the seasonal runoff ratios among the sites can be explained by hydro-climatic drivers and/or physical site differences. The NGS runoff ratio at Site 1 was quite high relative to the other sites (Figure 4-6). This could be explained by the timing and magnitude of the snowmelt event as there was more SWE in the snow pack prior to melts at Site 1 than there was at the other sites in southern Ontario. Eastman et al. (2010) found a similarly high ratio (0.88) at their clay site in one year of their study. In that case, the higher ratio resulted from a very wet spring period where large rains fell on saturated soils. Alternatively, physical site differences may have been the cause, as surface water would have been able to rapidly enter the drainage system through French drains within the field at Site 1. However, more field observations are necessary to confirm this. Another explanation is that the estimated snowfall, taken from the closest Environment Canada weather station, underestimated the precipitation as snow fall at the site. In addition to the above factors, there is uncertainty about the actual contributing area for overland flow discharge. If the contributing area is larger than the area used to calculate runoff ratios, then the runoff ratios would be lower than reported.

The distribution of discharge within the NGS also differed among the sites. The defined snowmelt periods at Site 1 were distinctly different from the snowmelt observed at Site 2 and Site 3 (Figure 4-7, Figure 4-8 and Figure 4-9). Defined spring snowmelt periods, such as the ones observed at Site 1, often dominate discharge in the NGS (Jamieson, et al. 2003, Tiessen, et al. 2010); however, during warmer winters, or in lower latitudes, there may be a series major discharge events throughout the NGS as was observed at Site 2 and Site 3 (Ontario Ministry of the Environment: Environmental Monitoring and Reporting Branch 2012). The distribution of discharge did differ spatially in the study period. In colder years, flow distributions between the sites would likely be similar due to fewer snow melts at the more southwestern sites. This study shows that the distribution of discharge within the NGS can differ across a range of sites in Ontario.

The study provides field-scale examples of the partitioning of overland flow and tile drainage. The results confirmed that tile drainage is often the dominant flow path in terms of total water discharge, as has been found in several studies (Eastman, et al. 2010, Gaynor and Bissonnette 1992). It should be noted that inter-annual variability is expected. In wetter years, a higher percentage of flow may come from overland flow pathways, while in the absence of major rainfall events or rapid snow melts, there may be less overland flow than was observed during this study period.

There was variation in partitioning of flow between the sites, with a greater contribution of overland flow occurring at Site 3. The variability between sites may be explained by the amount of SWE in snowpack prior to melt events, and the rate at which the snow melted due to weather patterns. Weather conditions were favorable for the initiation of overland flow at Site 3. Another explanation is that the peak discharge at Site 3 (10 mm day^{-1}), was lower than rates observed at Site 1 (17 mm day^{-1}) and Site 2 (16 mm day^{-1}), suggesting that the tile drainage system had less capacity than the other systems. There were also other site specific conditions that may have reduced overland flow at these sites. At Site 1, the outlet of the overland flow monitoring pipe discharges into an open ditch that receives overland flow from neighbouring areas. During the NGS this ditch was observed to be full of ice and snow, which resulted in the outlet pipe being submerged under water and ice. Snow and ice in the ditch likely reduced runoff leaving the field through the ditch. There are also French drains within low lying areas in the fields, which would limit the contribution of overland flow to total discharge measured at the monitored outlet. Again, there is uncertainty around the size of the overland flow contributing area. If this contributing area is smaller than the area used in the calculations, the role of overland flow in total discharge would be greater than what has been reported. At Site 2, overland flow is drained through a hickenbottom, but not all of the ponded water is able to exit the field via the hickenbottom as some does not reach the outlet and must instead infiltrate into the soil due to the hummocky nature of the site (Figure 4-12). These physical site differences may explain some of the site variability in flow partitioning. The variability in the partitioning of discharge may be explained by hydro-climatic drivers and site conditions, but more observations are required.

Figure 4-12 Ponding area upstream of hickenbottom outlet (bottom center of the picture) for overland flow at Site 2.

4.4.2 Phosphorus Concentrations

Tile P concentrations were higher during event flow than in baseflow conditions. This has been noted in other studies, for example, Dils and Heathwaite (1999) found subsurface TP concentrations in storm flow greater than 1 mg l^{-1}, while baseflow concentrations were generally >0.001 mg l^{-1} . The variability in tile P concentrations observed during various events was consistent with variability noted in other studies (Dils and Heathwaite 1999). Concentrations were elevated during peak flow events throughout the year. The surface ponding observed during these events suggests that macropore flow had occurred, and thus would have contributed to the elevated concentrations (Jarvis 2007). The elevated TP concentrations observed during the initial wet up following dry periods observed in this study have also been reported elsewhere (Dils and Heathwaite 1999). Over the course of the 18 month study period, the greatest fluctuations in concentrations were related to fertilizer and manure applications at Site 2 and Site 3. Increases in P concentrations following P applications have also been widely reported (Djodjic, et al. 2000, Gentry, et al. 2007, Macrae, et al. 2007, Hart, et al. 2004). Overall, the variability in discrete tile P concentrations was consistent with findings from similar studies.

P concentrations in overland flow were greater than concentrations in tile drainage, which is consistent with the literature (Gaynor and Findlay 1995, Heathwaite and Dils 2000, Algoazany, et al. 2007, Eastman, et al. 2010). Gaynor and Findlay (1995) found that average orthophosphate concentrations in overland flow (0.59 mg l^{-1}) were greater than tile drainage (0.37 mg l^{-1}). Similarly, in large field plots in Southern Ontario, Tan and Zhang (2011) had average DRP concentrations of 0.034mg l^{-1} and 0.057 mg l^{-1} for tile drainage and overland flow, respectfully. In the same study, TP concentrations were 0.480 mg l^{-1} and 0.627 mg l^{-1} for tile drainage and overland flow, respectively.

Olsen P levels have been shown to be related to SRP concentrations in both tile drainage and overland flow (Heckrath, et al. 1995, Wang, et al. 2010, Wang, et al. 2012). STP levels at each site were much lower than levels reported in other studies (Wang, et al. 2012), and are considered a low-moderate risk of SRP losses (Hilborn and Stone 2005). Generally, higher STP levels can result in higher SRP concentrations in runoff (Wang, et al. 2012, Wang, et al. 2010). Studies that have looked at the relationship between STP and soluble P in runoff have often identified change points, a point at which soluble P concentrations begin to increase at a higher rate per increase in STP. Even the highest STP levels found in the upper most 2.5cm across the sites were below critical change points reported in other studies (33 mg Olsen P kg^{-1} (McDowell and Sharpley, Approximating phosphorus release from soils to surface runoff and subsurface drainage 2001), 47.8 mg Olsen P kg^{-1} (Wang, et al. 2012) and 60 mg Olsen P kg^{-1} (Heckrath, et al. 1995)). It is noteworthy that even though STP levels were acceptable from an agronomic standpoint, the concentrations of P in tile drainage and overland flow regularly exceeded the Ontario Provincial Water Quality Objective for TP of 0.03 mg l^{-1}. It was not appropriate to explain differences in SRP concentrations at the sites based on STP differences for several reasons. Soil tests were taken at different times of the year and at different times relative to when fertilization that took place. Even though there are differences between the sites in Olsen P values, the differences are low relative to the range of Olsen P values observed in other studies.

4.4.3 Critical Periods for Phosphorus Export

The NGS was a critical period for P export at all sites, in both years. The seasonal distribution of P export was consistent with studies from similar climates (Withers, et al. 1999, Gentry, et al. 2007, Ball Coelho, et al. 2012, Eastman, et al. 2010, Gaynor and Findlay 1995). Ball et al. (2012) noted that despite higher concentrations observed in the GS, greater losses occurred during the NGS due to higher flow volume. This study also provided evidence that extreme summer events can produce high losses if they generate a large amount of runoff (i.e., June 2012 at Site 1). Eastman et al. (2010) also reported high loadings from summer events, particularly from overland flow. Identifying periods when P is exported, helps to find opportunities to improve management practices. This study confirmed that the NGS is a critical period for P export in Ontario reduced tillage systems.

Overall, the annual P losses calculated in this study were similar to values reported in the literature. Ulen and Persson (1999) reported average tile drainage TP export of 0.29 kg ha^{-1} yr^{-1}, Gentry et al. (2007) found TP export between 0.13-1.31 kg ha^{-1} yr^{-1}, and King et al., (2014) average tile export of 0.48 kg ha^{-1} yr^{-1} , which are comparable to NGS1 tile export of 0.072-0.656 kg ha^{-1}, and annual losses (GS1 + NGS2) of 0.169-0.190 kg ha^{-1} yr^{-1} at the sites (Table 4-4). Overland flow losses were also comparable to other studies. Eastman et al. (2010) reported annual P loads from overland flow ranging from 0.4-1.9 kg ha^{-1}. These values were higher than overland totals during this study (0.077-0.250 kg ha^{-1} yr^{-1}). The higher exports reported by Eastman et al. (2010) occurred on conventionally tilled sites and during higher runoff years.

Combined losses (tile+overland flow) from all three sites in this study were also comparable to other studies in the literature. Eastman et al. (2010) found combined losses from overland and tile drainage from their clay and sandy sites ranged between 1.6-4.2 kg ha^{-1} yr^{-1} and 0.8-0.9 kg ha^{-1} yr^{-1}, respectively. The sites in the Eastman et al. (2010) study had greater amounts of annual runoff, particularly from overland flow, which may explain why annual totals were higher at their sites. Losses were within values reported in the literature, but notably lower than values reported in some studies.

Inter-annual variability can be driven by hydro-climatic drivers and management practices. Wet years/seasons have been shown to increase P export (Gentry, et al. 2007). Since this study was only 18 months, little can be said about inter-annual variability based on these drivers, but since the study period included a dry year, greater losses should be expected in years with greater runoff. This study revealed inter-annual variability related to management practices. P losses from tile drainage at Site 2 during NGS1 (0.656 kg ha^{-1}) were much higher than annual losses from combined pathways the following year at all the sites (0.267-0.419 kg ha^{-1} yr^{-1}), despite there being less flow volume in NGS1. Though not as great of a difference between the NGSs, Site 3 losses for tile drainage were also greater in NGS1 (0.221 kg ha^{-1}) than in NGS2 (0.178 kg ha^{-1}), despite greater flow volumes in NGS2. The losses of P from Site 2 and Site 3 during the November 2011 event were enhanced because of direct fertilizer losses. These types of losses do not always show up in studies because the magnitude of losses depends largely on the timing of major rainfall or melt events relative to application (Hansen, et al. 2002, Smith, et al. 2007). At Site 2 and Site 3, applications of P were made prior to large rainfall events which occurred between Nov 28-Dec. 4, 2011 (Site 2: manure surface broadcast 7 weeks prior and MAP incorporated into a band using a zone tillage unit, 2.5 weeks prior; Site 3: MAP incorporated with vertical tillage, 8 weeks prior). These events resulted in the majority of NGS discharge, and consequently had a large influence on NGS P export. At Site 2, the tile export from the two major discharge events following manure and fertilizer application, accounted for 80% of SRP and 50% of TP export during the entire 18 month study period. Increases in soluble P concentrations are often associated with recent fertilizer applications (Withers, et al. 2003). Withers et al. (2003) reported that if direct fertilizer losses occur, they can account for 50-98% of losses measured at the field edge.

The difference between export totals at the two sites was likely attributed to the timing and method of application used. At Site 2, the MAP was incorporated into soil, but the manure was surface applied and not incorporated. In contrast, at Site 3, MAP was broadcast and then incorporated with a single pass of vertical tillage (depth ~5cm). Tarkalson and Mikkelsen (2004) found that incorporating broiler manure reduced the P load in surface runoff by 88% in their simulated rainfall study. Losses from incorporated manure and fertilizer plots were not different than the control in their study. The source of P can also influence losses, as Tarkalson and Mikkelsen (2004) found that losses from surface applied manure were significantly greater than surface applied fertilizers, although only at higher application rates (Tarkalson and Mikkelsen 2004). The difference in application methods used explains the difference in P losses between Site 2 and Site 3. Incorporation of manure and fertilizer at Site 2, rather than leaving it on the surface, would likely have reduced P losses in the large discharge events following application, particularly soluble P losses. However, it is important that incorporation be done in a non-erosive manner (Great Lakes Commission Phosphorus Reduction Task Force 2012). The timing of MAP applications at the site may also have contributed to site differences observed. The MAP application at Site 2 was incorporated similar to MAP at Site 3, but the Site 2 application was made much closer to the rainfall event and thus the soil and fertilizer would have had less time to interact. This study confirms the importance of direct fertilizer P losses, highlighting the susceptibility of unincorporated P sources to losses in systems where full-width tillage cannot be used for incorporation.

4.4.4 Partitioning of Runoff and Phosphorus Export between Tile Drainage and Overland Flow

This study aimed to improve understanding of the relative contribution of tile drainage and overland flow to P export. Tile drainage has been found to make significant contributions to field-scale losses of P in many studies (Eastman, et al. 2010, Ball Coelho, et al. 2012, Gaynor and Findlay 1995, Gentry, et al. 2007, Smith, et al. 2014). Although P concentrations are typically lower in tile drainage relative to overland flow, the greater flow volume leaving through tile drainage results in significant P export. This study confirmed the importance of tile drainage in field-scale P export. Despite lower SRP and TP concentrations found in tile drainage, this pathway contributed a large percentage of annual export. Tile drainage accounted for 40-71% of total site TP losses between May 2012 and April 2013. Higher SRP concentrations made overland flow an important export pathway for SRP at all the sites. Gaynor and Findlay (1995) found that tile drainage contributed 55-68% of ortho-P exported. In a plot study in Ontario, Ball et al., (2012) reported that tile drains accounted for 34% of combined annual SRP and 24% of combined annual TP losses. Eastman et al. (2010) found that overland and tile drainage contributed equally to TP export in both years at their clay soil site. However at their sandy site, results were mixed, with overland flow dominating TP export in the first year, then contributing equally in the second year of the study. Algoazany et al. (2007) found that tile drainage had higher average soluble P losses than overland, but there was also variability in the partitioning between sites and between years. In reduced tillage plots, Uusitalo, et al. (2007) found that overland flow contributed significantly more SRP than subsurface drainage, and subsurface drainage contributed significantly more PP to combined export (TP- SRP (0.2μm filter)). In a large plot study Tan and Zhang (2011) found that tile drainage was the dominant export pathway for SRP, SUP, PP and TP, largely because of flow volume. In the Maumee River watershed, Smith et al. (2014) found that tiles contributed 49% of soluble P and 48% of TP losses measured at the field edge. They concluded that addressing P loss from tile drainage is likely necessary to meet the reduction targets for that region (Smith, et al. 2014). Though the relative contribution of tile drainage to soluble and PP export varies, the results of this study are consistent with other studies in the literature and confirm that tiles can be an important export pathway.

The speciation of P in runoff differed between the pathways. The PP+SUP fraction dominated both tile drainage and overland flow at all sites in this study, with a greater portion of SRP occurring in overland flow relative to tile drainage. There are no consistent results reported in the literature as far as dominant forms for particular flow paths. In some studies, the dominant form differs between pathways. For example, Heathwaite and Dils (2000) found that DP was the dominant fraction in overland flow, while PP was dominated in tile drain flow. While in other studies, the dominant form was consistent between the pathways and was influenced by site specific conditions. For instance, Eastman et al. (2010) reported that soil type influenced speciation. They found that in clay soil, PP was the dominant form in both overland flow and tile drainage, while in sandy soil soluble P was the dominant form in both pathways. Ulen and Persson (1999) found that PP was the dominant form in tile drainage in clay soils in Sweden. Several studies reported PP fraction to be dominant in both pathways. In a plot study in Southwest Finland on clay soil, Uusitalo, et al. (2007) found PP to be the dominant fraction in overland flow and tile drainage, in the conventional and no-tillage plots. Tan and Zhang (2011) reported speciation in tile drainage and overland flow similar to results of this study. Tan and Zhang (2011) reported SRP to be 7% of TP in tile drainage and 8% of TP in overland flow over 5 years in their large plot study. PP was the dominant fraction in both pathways relative to TDP in their study. SUP was the dominant soluble form relative to SRP in their study. Based on their results, it is likely that the SUP fraction was a notable portion of PP+SUP measured in this study. One reason there may have been less SRP in tile drains than in overland flow is because soluble inorganic P is removed from solution by soil constituents more readily than some forms of soluble organic P found in the SUP fraction (Condron, et al. 2005, Anderson and Magdoff 2005). Although P fractions measured are not always directly comparable to other studies, speciation differences between pathways were similar other results reported in the literature.

The speciation did vary throughout the study period and appeared to be influenced by fertilization and seasonal ground conditions. The greater portion of TP as SRP in runoff observed following fertilization in this study is consistent with the literature (Hart, et al. 2004). In addition to management influences on speciation, seasonal trends in speciation are often linked to ground conditions (Hansen, et al. 2000, Macrae, et al. 2007, Ball-Coelho, et al. 2012). Macrae et al. 2007 noted that SRP comprised a greater portion of TP during winter events relative to summer events, a trend which was observed at Site 1 and Site 3, outside of the period directly affected by fertilization.

4.4.5 Opportunities to Manage Phosphorus Export

This study confirmed that both export pathways are more active during the NGS. In the study period major flow events in both fall, mid-winter, and spring made important contributions to P export. This highlights the importance of selecting BMPs that will address P losses during these periods.

Field-scale studies make it possible to capture the impact of land management practices that may be lost at the watershed scale. In this study, the influence of direct fertilizer losses was apparent. In systems that are managed carefully as the ones in this study, it appears there may be further opportunity to reduce field-edge P losses by reducing the effect of direct fertilizer losses.

Given that direct fertilizer losses can be substantial, the frequency and seasonal timing of applications is another important management consideration. Since higher application rates carry a greater risk of P losses, splitting applications, thus making multiple smaller applications has been considered as an alternative management strategy to reduce P losses. However, Burkitt, et al. (2011) found that there was less risk associated with larger single applications made in dry periods because there was less probability that applications would align with major discharge events. In this study, the two fertilized sites, each received a single fall application, intended to meet P requirements for the following three crops.

It is recognized that fall applications of P carry increased risk because this season, and the period following, is more conducive to major flow events. McDowell and Catto (2005) found that direct P losses were less when fertilizer was applied during the dry season as opposed to the wet season. The risk of applying P prior to wet seasons was apparent in this study. However, to properly assess the risk associated with multi-crop fall applications, P losses must be measured for a least one crop rotation/fertilization cycle. Although relatively large losses occurred in NGS1, there will be no further P applications made for two years, and therefore no direct fertilizer losses are likely to occur. Based on Burkitt et al., (2011), it is possible that greater losses could occur if smaller annual P applications were made.

As direct fertilizer losses can be substantial, an increase in high risk application methods in high risk periods is a potential source of increased P losses observed in tributaries to the Great Lakes. This study provides direct evidence of this management problem at the field scale. Although aspects of the fertilization strategies appear to be high risk, these methods are an accepted agronomic practice and a part of the production system in place. These single fall applications reduce equipment passes, saving time and reducing input costs. Proposing changes to these operations would need to balance other priorities, such as soil health benefits, and management constraints.

4.5 Conclusion

This field-scale study has demonstrated the influence of seasonality and land management practices on P export from RT systems, as well as the relative contribution of runoff pathways to annual P export. It was demonstrated that periods of high runoff generation, such as the NGS, are capable of exporting relatively large amounts of P. Therefore, particular attention should be given to management practices prior to these periods. The study provided two field-scale examples of enhanced P losses as a result of P applications occurring prior to large runoff events, which highlighted the risks associated with fall applications of P. Both tiles and overland flow made important contributions to overall TP export. Overland flow contributed the majority of SRP export at two of the sites. Even though overland flow contributed a small portion of total runoff, the high SRP concentrations observed in overland flow resulted in disproportionately higher SRP export. P export from reduced tillage systems measured at the edge of field occurs primarily during periods of high runoff generation and can be influenced by P application practices. Improving management of P applications by avoiding fall applications and incorporating below the surface will likely reduce the risk of losses in reduce tillage systems in Ontario; however, the benefits of such changes to these systems should be weighed against other conservation goals.

5 Variability in surface and subsurface phosphorus export from agricultural fields during peak flow events over the non-growing season

5.1 Introduction

In southern Ontario, Canada, discharge from agricultural watersheds occurs primarily during the non-growing season (NGS) (Ball-Coelho, et al. 2012, Macrae, et al. 2007, Ontario Ministry of the Environment: Environmental Monitoring and Reporting Branch 2012). Runoff during this period is often characterized by several major snowmelt or rain events (Ontario Ministry of the Environment: Environmental Monitoring and Reporting Branch 2012), rather than a single spring freshet, which is typical in colder regions at higher latitudes (Jamieson, et al. 2003, Tiessen, et al. 2010). In addition to supplying a considerable portion of annual runoff, NGS events contribute significantly to annual phosphorus (P) loads from agricultural watersheds (Macrae, et al. 2007, Puustinen, et al. 2007, Jamieson, et al. 2003) and it is now generally accepted that monitoring hydrology and water quality through the NGS is important for providing more accurate estimates of P loading in regions that experience “winter” and “snowmelt” conditions (Miles, et al. 2013). However, relatively few studies accomplish this due to logistical challenges and equipment failure. P export through the NGS has been reported on at the plot (Ball-Coelho, et al. 2012) and watershed scales (Macrae, et al. 2007, Su, et al. 2011, Dils and Heathwaite 1999) but fewer field-scale studies have been completed. Although, seasonal loadings, or spring snow melt loadings (Su, et al. 2011, Jamieson, et al. 2003, Puustinen, et al. 2007), are often reported, inter-event variability in the partitioning and speciation of P export through the NGS is reported less often (Penn, et al. 2010). Some studies have used multi-year datasets to look at factors that influence variability in winter losses at the watershed scale (Danz, et al. 2013, Su, et al. 2011); however this has not been reported for field-scale studies that monitored both tile drainage and overland flow simultaneously. Consequently, gaps remain in our understanding of how P losses are influenced by hydro-climatic drivers, ground conditions and management practices. Such information is important for improving models, which currently do not function properly under winter conditions. This chapter characterizes differences in runoff and P export during major flow events occurring throughout the NGS.

NGS events are important under the current climate conditions in Ontario; however, it is unclear how this may change under future climate change scenarios. Climate change predictions include warmer winter temperatures, more frequent winter thaw events, more precipitation as rain, more days with unfrozen soils and more extreme precipitation events (Colombo, et al. 2007, Melillo, et al. 2014). If these predictions are accurate, the changing climate will likely result in different runoff patterns and pre-event ground conditions during the NGS. Thus, monitoring runoff and P load responses under a range of NGS conditions and event types will improve our understanding of this critical period, and, will assist us in predicting how water quality issues may change with climate change.

This chapter builds on findings from chapter 4, and addresses objective 2 of this thesis, to demonstrate the significance of peak discharge events in P export for a 12 month period at two sites, by

a. Identification of the seasonal distribution of peak discharge events;
b. Demonstration of the contribution of peak events to annual P export;
c. Demonstration of how event type (climate drivers and pre-event (antecedent) ground conditions) influence the event export and speciation (Total, Soluble Reactive, and Particulate + Soluble Unreactive) during successive peak discharge events in the NGS.

5.2 Study Approach and Methods

This chapter focusses on the period where both overland flow and tile drainage pathways were monitored (May 2012 – April 2013). Only Site 2 and Site 3 were used in this Chapter because they experienced comparable events (driven by similar weather systems) over the course of the second NGS (NGS2), making direct comparisons of specific events possible. This Chapter has a greater focus on inter-event variability (and pre-event conditions leading to this variability), as opposed to Chapter 4 where more general seasonal trends were described. The site description, general site setup and data collection methods are described in Chapter 3. The following describes methods specific to this chapter.

5.2.1 Definition of Flow Events and Classification of Peak Flow Periods

Individual events were characterized based on pre-event conditions, event properties and event discharge and loading. Field-scale discharge and loading were calculated as the sum of tile and overland flow losses for individual events, and deeper groundwater fluxes are not considered. Events were considered to have commenced when there was a hydrograph response for overland flow or tile drainage, and concluded when the tile hydrograph returned to baseflow conditions. In all events, overland flow ceased to flow prior to the cessation of tile discharge. Overland flow often had several defined peaks over the same period as a single response in tile drainage (e.g. over multiple days). Such occurrences were combined and considered a single event for the purpose of this study. After identifying and characterizing all events during the study period, the distribution and contribution of peak discharge events to annual runoff and P losses were determined.

Peak events were defined as events where overland flow occurred, or significant surface ponding was observed at the site. For the purposes of this chapter, peak events have been assigned unique identifiers based on the date at which the event began (i.e., December 1, 2012 is identified as D_1). At both sites, peak events occurred as the result of regional weather systems, and thus the temporal distribution of peak events was similar between the two sites. Similarly timed events are aligned within tables to allow for comparison of these events between sites. However, due to slight differences between the sites, not all events are paired between the sites, specifically there were no December events at Site 2, and no January 12th event (J_{12}) at Site 3. If there was no comparable event, cells in tables comparing events were left blank.

5.2.2 Characterization of Event Types

This chapter evaluates the influence of event type on P losses. Event types were classified based on two criteria: (a) event climate driver (i.e. rainfall vs melting snow or a mixture of the two, described in more detail below) and (b) antecedent ground conditions (i.e. presence/absence of frozen ground; presence of snow cover).

Classifying the event driver required rainfall and snowfall data. Rainfall was measured from the tipping bucket rain gauge and SWE was estimated based on nearby weather station data, infield observations of snow depth, and SWE surveys as described in Chapter 3. Events were classified as being driven by Rain, Rain and Snow (SWE), or Snow Melt. The relative contribution of rain to total inputs was also calculated for each event using the ratio of Rain:Rain+SWE, though it was not used specifically to classify event types.

Two antecedent ground conditions were monitored: (1) the presence of snow cover and (2) soil temperature. Snow cover was classified as present or not present. Soil temperature data from the 10 cm depth temperature probe on-site (Onset Systems: 12-Bit Temp Smart Sensor -

S-TMB-M002) was used to assess pre-event soil temperature status. This data is used to classify events into three categories based on soil temperature: Unfrozen (≥1 ˚C), Partially Frozen (0-1 ˚C), and Frozen (< 0 ˚C). When soil is ≥1 ˚C, soil is presumed to be unfrozen, when soil temperature is 0-1 ˚C it is accepted that there is a possibility of some frozen soil between 10cm and the surface, finally, when soil temperature is < 0 ˚C, soil is expected to be frozen in a large portion of the field. The presence of frozen soil at the surface was also confirmed with field visits prior to and during each event.

Based on the classifications above, specific event types were defined (e.g. Rain and SWE –Unfrozen Soil) (Table 5-1). In this example, the event was driven by rain and melting snow, and the event occurred on unfrozen soil.

Table 5-1 Event type, properties, and ground conditions, for all peak events from Site 2 and Site 3.

	Site 2						
			J_{11}	J_{13}	J_{28}	M_{10}	A_9
Event Type (Driver - Soil Status)			Rain and SWE - Unfrozen	Rain - Unfrozen	Rain and SWE - Frozen	Rain and SWE - Partially Frozen	Rain - Unfrozen
Rain (mm)			18	25	53	21	91
Rain + SWE (mm)			43	25	73	51	91
Rain:Rain+SWE			0.42	1.00	0.72	0.41	1.00
Max mm/Hr			5	5.8	9.6	2.2	14.8
Soil Temperature (˚C)			1.2	5.0	-0.9	0.4	5.1
Snow Remained?			No	NA	No	No	NA
	Site 3						
	D_1	D_4	J_{11}		J_{28}	M_{10}	A_9
Event Type	Rain and SWE - Unfrozen	Rain - Unfrozen	Rain and SWE - Unfrozen		Rain and SWE - Partially Frozen	SWE - Partially Frozen	Rain - Unfrozen
Rain (mm)	25	14	33		44	9	69
Rain + SWE (mm)	30.2	14.4	60.2		89.2	64.4	69.0
Rain:Rain+SWE	0.83	1.00	0.55		0.49	0.14	1.00
Max mm/Hr	5	6	4.2		5.4	2	6.4
Soil Temperature (˚C)	1.9	6.7	1.0		0.3	0.8	4.6
Snow Remained?	No	NA	No		No	Yes	NA

5.2.3 Additional Site Description

Sites were described in Chapter 3, the following is supplementary description necessary for this chapter.

5.2.4 Residue Cover

There was minor variation in residue cover between the sites, which was influenced by tillage operations. Generally, all sites maintained a high level of residue cover (>30%) during the NGS. However there were slight differences in residue management following corn harvest. At

Site 3 corn residue was worked once with a vertical tillage implement to incorporate and size the residue, though the majority of corn stalks remained upright. While at Site 2, the corn residue was not tilled following harvest (Figure 5.1).

Site 2: October 14, 2013

Site 3: November 20, 2013

Figure 5-1 Picture of corn residue cover at Site 2 and Site 3 taken following corn harvest and prior to snow accumulation.

5.3 Results:

5.3.1 Temporal Variability in Runoff Generation

Rain or melting snow resulted in numerous hydrologic responses in tile drains throughout the year (Site 2: 46, Site 3: 19) as well as a small number of overland flow events (Site 2: 3, Site 3: 5) (Figure 5-2 a, c and Figure 5-3a, c). Generally, greater total atmospheric inputs (Rain+SWE) leading to an event resulted in greater total event discharge; however, this relationship was influenced by antecedent moisture conditions (Figure 5-4). For example, when soil was drier as experienced during the growing season (GS), high rainfall events did not necessarily yield a large flow response (Figure 5-4).

A series of peak discharge events occurred over the NGS at both sites. A total of five peak discharge events were identified at Site 2 (Figure 5-2) with six peak discharge events at Site 3 (Figure 5-3). The combined discharge from overland flow and tile drainage for the peak events resulted in the majority of annual discharge from each site (65%-72%, Table 5-1). The temporal distribution of peak discharge events was similar between the two sites because the sites generally experienced the same regional weather patterns (Figure 5-2 and Figure 5-3). Notable exceptions were the two early NGS events at Site 3(D_1 and D_4), which occurred because of a greater snowfall accumulation at the site prior to a warm weather system moving through the area. Site 2 did not receive the same snowfall, and as such, the warmer temperatures did not result in a peak discharge event. Following these initial thaw events at Site 3, the distribution between the sites was similar, although the ensuing peak events differed both in their event drivers (Rain, Rain and Snow, Radiation Melt), total discharge, and flow path partitioning (Figure 5-2 and Figure 5-3, Table 5-1).

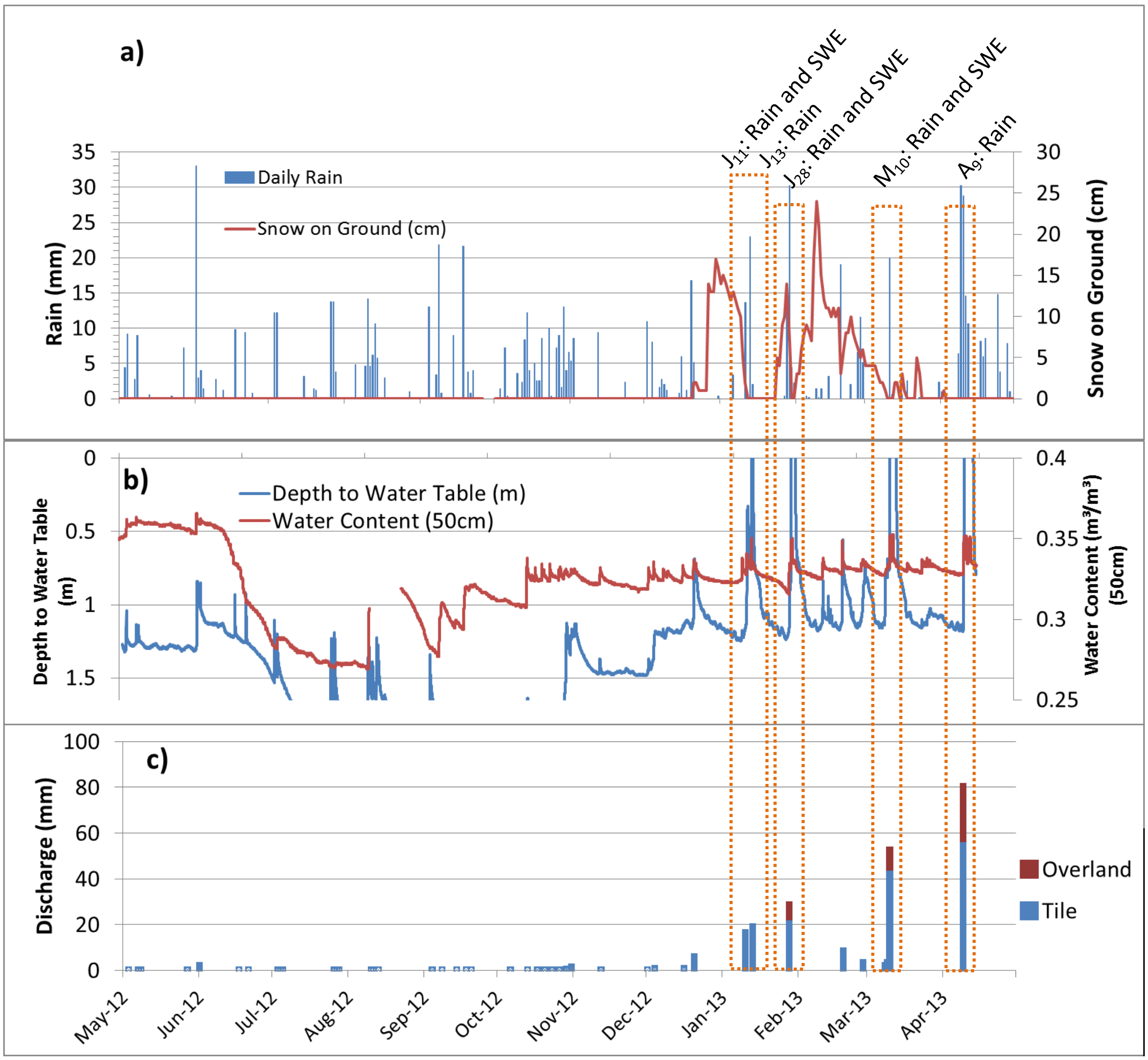

Figure 5-2 Site 2 hydro-climatic drivers and total event discharge. Daily rainfall and snow depth (a). Water table depth and soil water content at 30cm (b), total event discharge for subsurface tile drainage (blue) and overland flow (red). Peak discharge events are outlined with dashed orange boxes and labeled near the top of the figure. Water table below 1.45m was not detectable. Gaps in water table data occurred because sensors were temporarily removed from the well. Snow on ground was estimated using Environment Canada Weather Station Data.

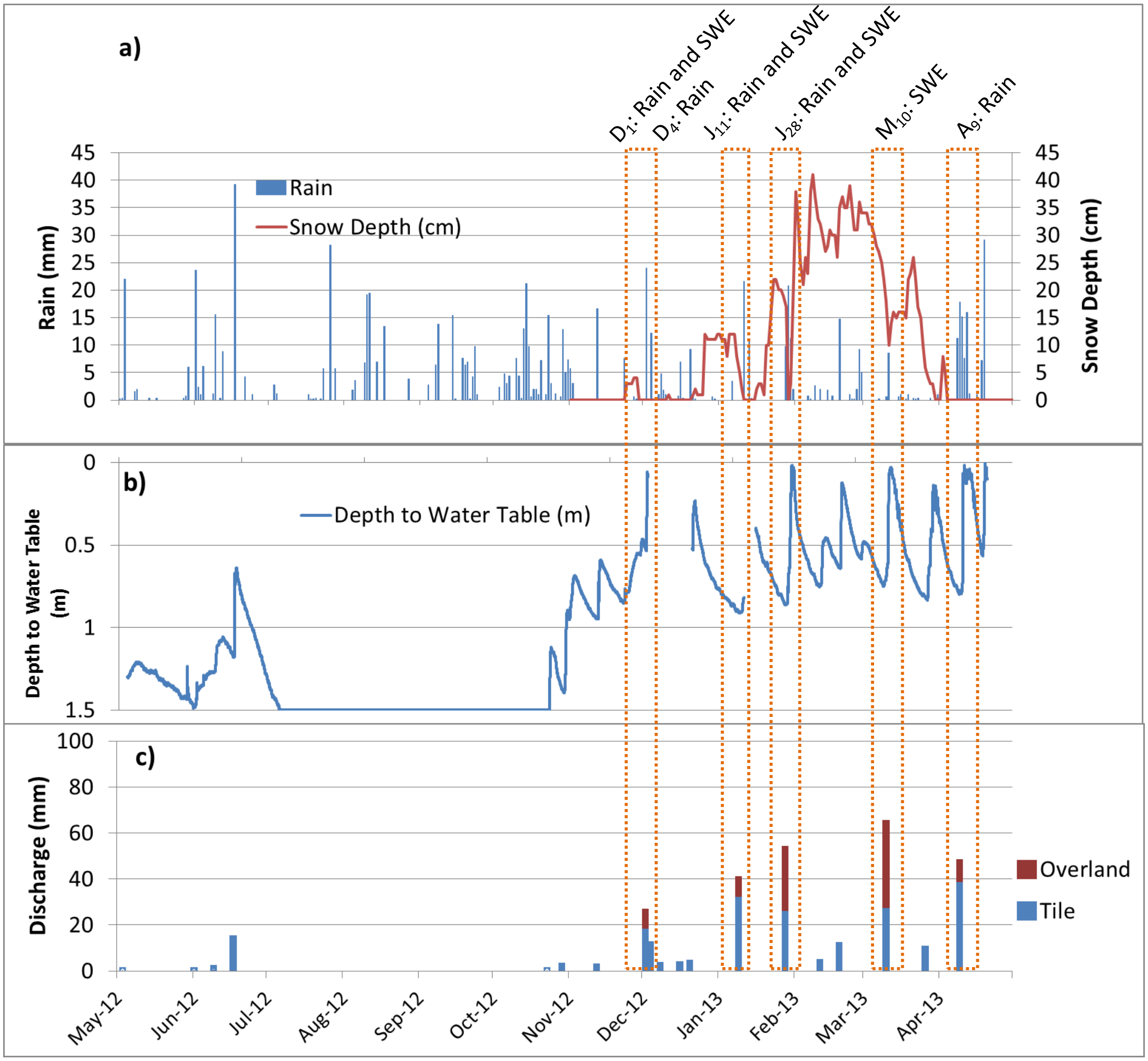

Figure 5-3 Site 3 hydro-climatic drivers and event discharge. Daily rainfall and snow depth (a). Water table depth and soil water content at 30cm (b), total event discharge for subsurface tile drainage (blue) and overland flow (red). Peak discharge events are outlined with dashed orange boxes and labeled near the top of the figure. Water table below 1.45m was not detectable. Gaps in water table data occurred because sensors were temporarily removed from the well. Snow on ground was estimated using Environment Canada Weather Station Data.

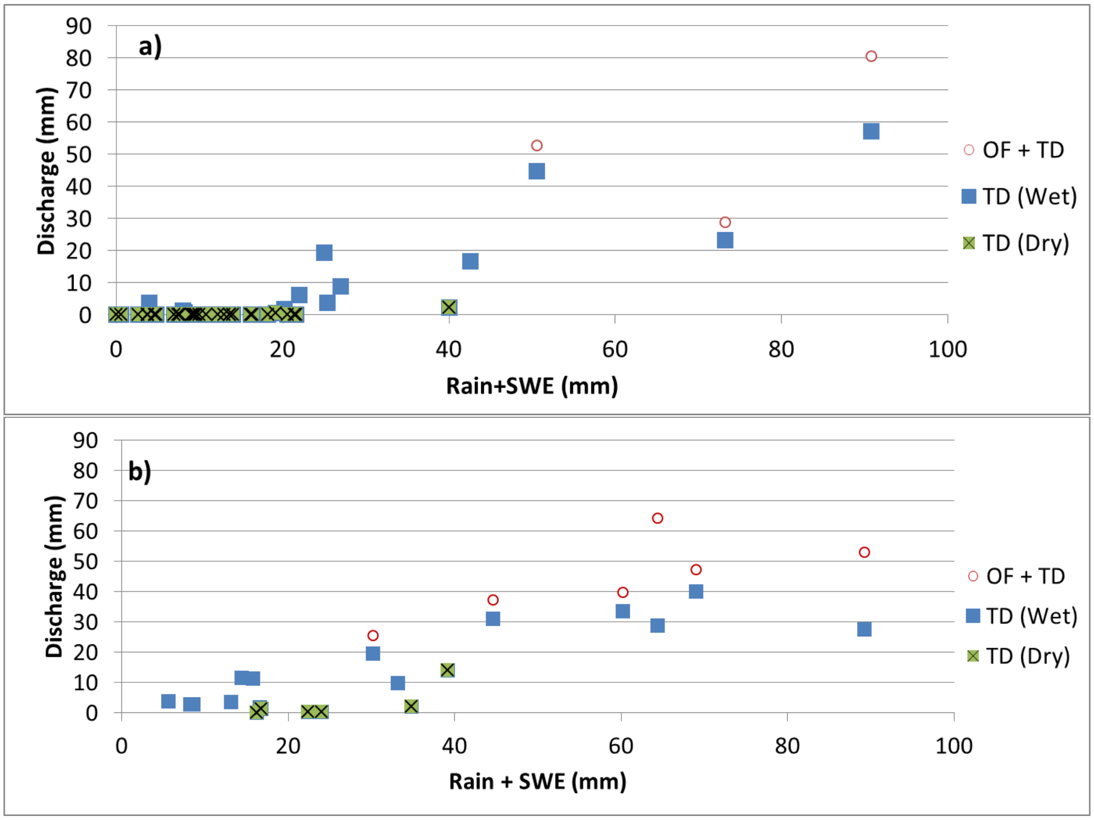

Figure 5-4 Site 2 (a) and Site 3 (b) Total event inputs of Rain + SWE in mm vs Total Event Discharge in mm (Subsurface Tile (blue box), Overland Flow + Subsurface Tile Drainage (red circle)) during dry (green box) and wet (blue box and red circle) pre-event conditions.

5.3.2 Variability in Pre-Event Ground Conditions for Peak Events

Ground conditions (snow cover and soil temperature) prior to and during peak discharge events were variable (Figure 5-5 and Figure 5-6). Snow cover was present at the onset of three out of five peak events at Site 2 (J_{11}, J_{28}, and M_{10}) and four out of six peak events at Site 3 (D_1, J_{11}, J_{28} and M_{10}) (Figure 5-5 and Figure 5-6). The snowpack melted completely in all peak events (if present at the event's onset), except for one event at Site 3 (M_{10}). Soil temperature was also variable between these peak discharge events (Figure 5-5 and Figure 5-6, Table 5-1). The most notable difference was the presence of frozen soil(<0˚C) to at least 10 cm depth at Site 2 during Event J_{28}, as all other events occurred on unfrozen soil (Table 5-1). Patches of frozen soil were observed prior to Event J_{28} at Site 3; however, soils were not frozen to the degree observed at Site 2 (Table 5-1). The combinations of event drivers, discussed in Section 5.3.1(e.g. rain, rain and snow melt and radiation melt) and variable ground conditions, discussed in Section 5.3.2 (snow cover and soil temperature status) led to several distinct event types over the NGS (Table 5-1). The contribution of these events to site discharge and P export is discussed below.

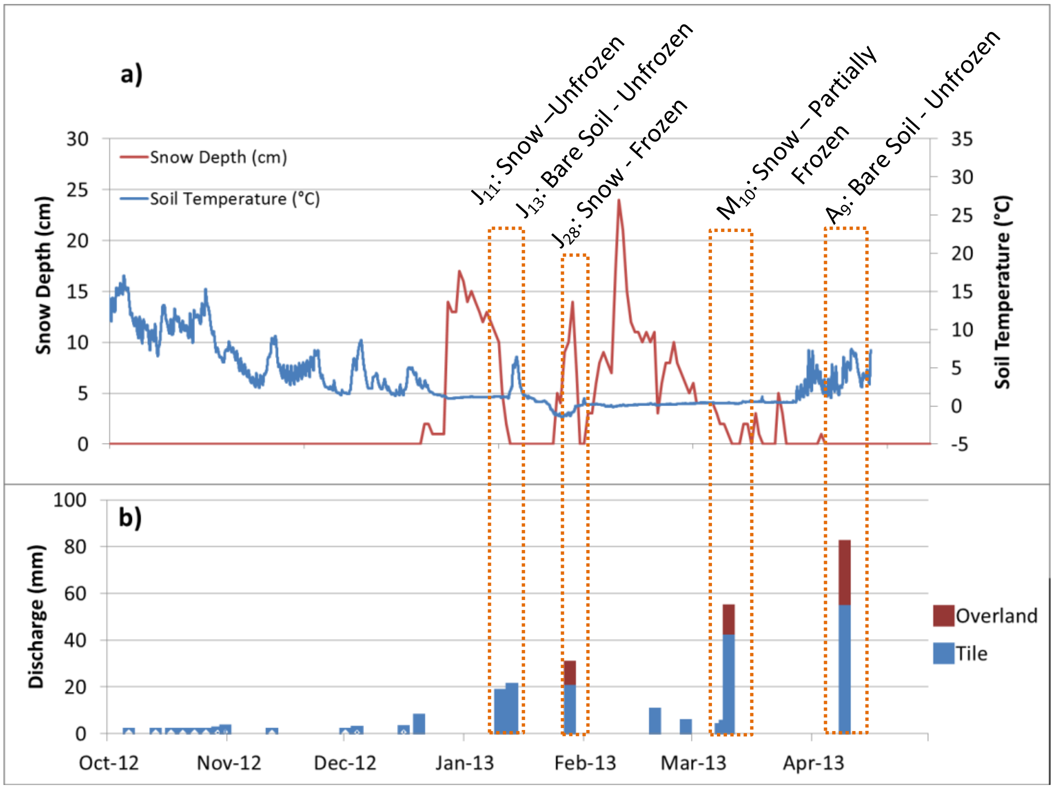

Figure 5-5 Site 2 ground conditions for peak discharge events. Daily snow depth (red) and soil temperature (10cm) (blue)(a). Total event discharge from tile (blue) and overland flow (red) (b).

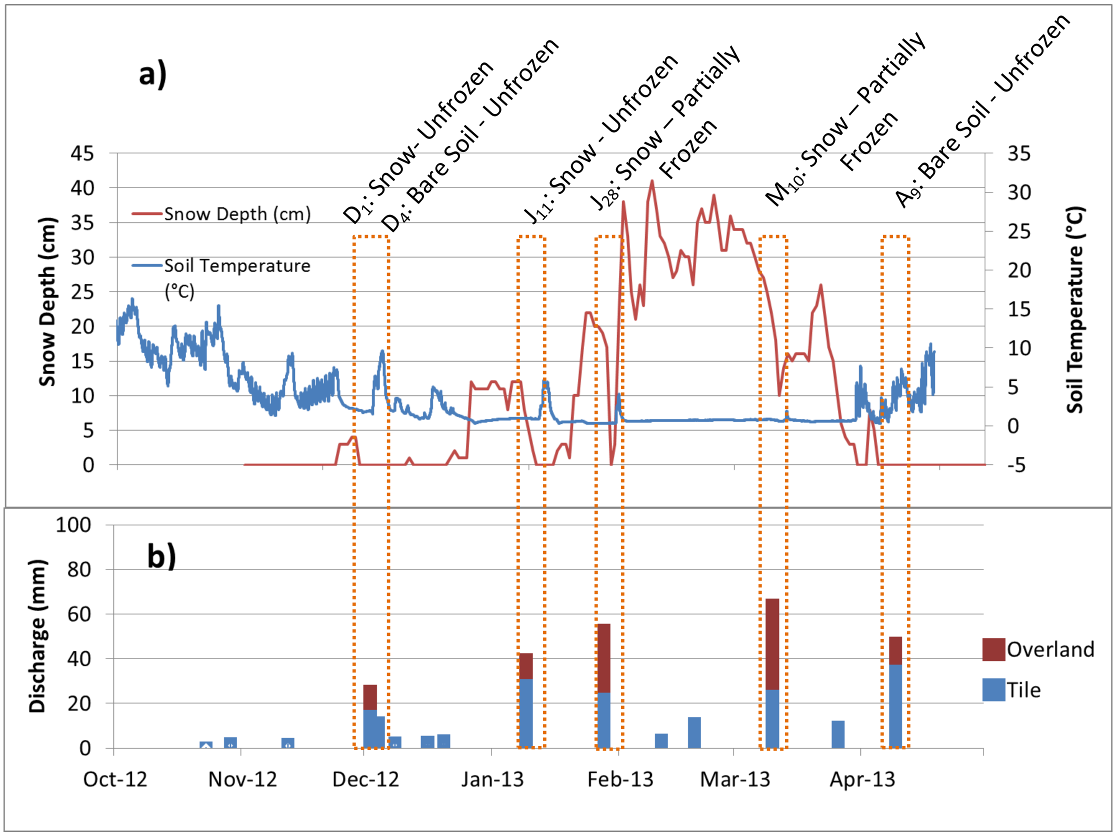

Figure 5-6 Site 3 ground conditions for peak discharge events. Daily snow depth and soil temperature (10cm) (a). Total event discharge (b).

5.3.3 Contribution of Peak Discharge Events to Annual Phosphorus Export

The majority of annual P losses came from the series of peak discharge events occurring within the NGS (Figure 5-7 and Figure 5-8, Table 5-1). At Site 2, the combined tile and overland flow losses from the five peak events contributed 86% of TP, 85% of PP+SUP and 90% of SRP annual losses. At Site 3, combined losses during six events contributed 90% of TP, 88% PP+SUP and 96% of SRP annual losses (Table 5-1). Although all peak events contributed to annual P losses, there was some variability in the quantity and speciation of P losses among these events (Figure 5-7 and Figure 5-8). For example, April rain events (A_9) contributed disproportionately higher amounts of PP+SUP than all other peak discharge events. SRP export was greatest in late January events at both sites (J_{28}), and SRP losses subsequently decreased over the remaining NGS (Figure 5-7 and Figure 5-8), despite increases in event discharge. In contrast, TP loading generally increased over the NGS (Figure 5-5 and 5-6), and was positively related to total site discharge (Figure 5-9 and Figure 5-10). The relationship between SRP event losses and discharge was not as strong as it was for TP (Figure 5-9 and Figure 5-10). Events with overland flow resulted in the greatest event discharge and consequently greater P losses (Figure 5-9 and Figure 5-10). In one year of data, an approximate threshold appeared showing that events with discharge > 20mm were likely to be high loss events in terms of TP (Figure 5-9 and Figure 5-10).

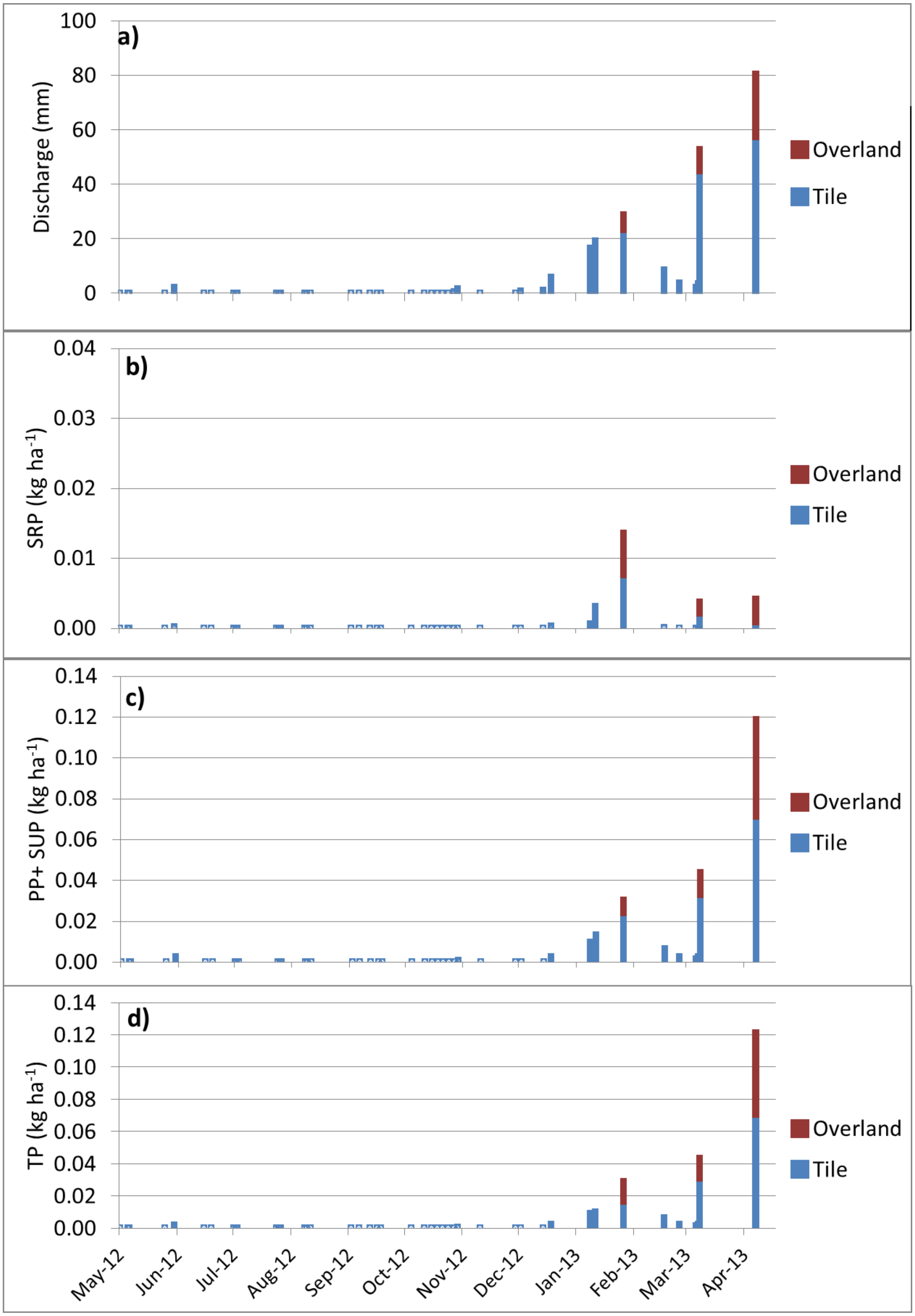

Figure 5-7 Event Discharge (a), and phosphorus losses of SRP (b), PP (c) and TP (d) from tile drainage (blue) and overland flow (red) from Site 2.

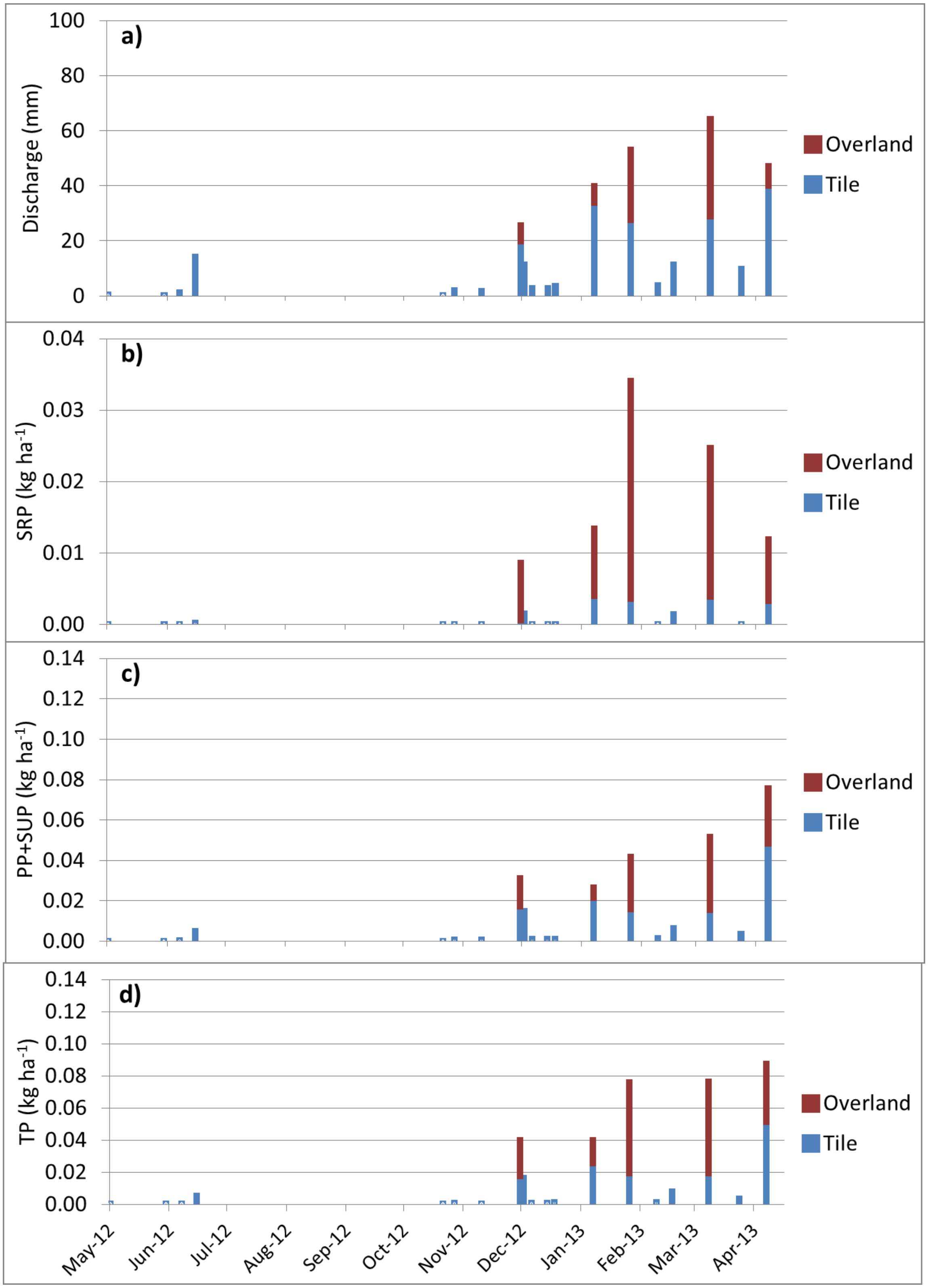

Figure 5-8 Event Discharge (a), and phosphorus losses of SRP (b), PP (c) and TP (d) from tile drainage (blue) and overland flow (red) from Site 3.

Table 5-2 Peak event phosphorus loading from tile drainage, overland flow, and combined flow for Site 2 and Site3.

		Site 2				Site 3			
				Combined Flow				Combined Flow	
	Event	Tile Drainage	Overland Flow	Combined Flow	% of Annual	Tile Drainage	Overland Flow	Combined Flow	% of Annual
Q (mm)	D_1					20	6	26	7%
	D_4					12	0	12	3%
	J_{11}	17	0	17	6%	34	6	40	11%
	J_{13}	19	0	19	7%				
	J_{28}	23	6	29	10%	27	26	53	14%
	M_{10}	45	8	53	19%	29	36	64	17%
	A_9	57	23	80	29%	40	7	47	13%
	∑	161	37	198	72%	161	81	242	65%
SRP (kg ha^{-1})	D_1					0.001	0.008	0.009	9%
	D_4					0.002	0.000	0.002	2%
	J_{11}	0.001	0.000	0.001	3%	0.004	0.010	0.013	14%
	J_{13}	0.003	0.000	0.003	11%				
	J_{28}	0.008	0.006	0.014	48%	0.002	0.031	0.033	34%
	M_{10}	0.002	0.002	0.004	14%	0.004	0.021	0.025	25%
	A_9	0.001	0.003	0.004	15%	0.003	0.009	0.012	12%
	∑	0.014	0.011	0.026	90%	0.015	0.078	0.093	96%
TP (kg ha^{-1})	D_1					0.018	0.022	0.040	11%
	D_4					0.017	0.000	0.017	4%
	J_{11}	0.010	0.000	0.010	4%	0.025	0.015	0.040	11%
	J_{13}	0.014	0.000	0.014	5%				
	J_{28}	0.024	0.013	0.037	14%	0.019	0.057	0.076	20%
	M_{10}	0.033	0.013	0.046	17%	0.019	0.057	0.076	20%
	A_9	0.071	0.051	0.123	46%	0.051	0.036	0.088	23%
	∑	0.152	0.077	0.229	86%	0.149	0.188	0.337	90%
PP+SUP (kg ha^{-1})	D_1					0.017	0.014	0.031	11%
	D_4					0.015	0.000	0.015	5%
	J_{11}	0.009	0.000	0.009	4%	0.021	0.005	0.027	10%
	J_{13}	0.011	0.000	0.011	4%				
	J_{28}	0.016	0.007	0.023	10%	0.017	0.026	0.043	16%
	M_{10}	0.031	0.011	0.042	18%	0.015	0.037	0.052	19%
	A_9	0.070	0.048	0.118	50%	0.048	0.028	0.076	27%
	∑	0.138	0.066	0.204	85%	0.134	0.110	0.244	88%

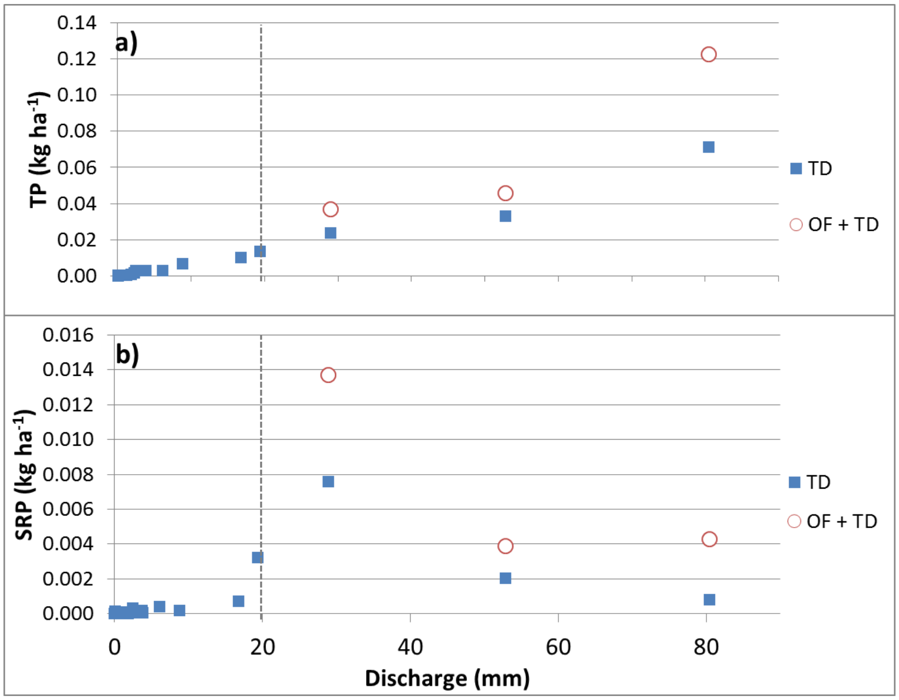

Figure 5-9 Site 2 Event TP load (a) and SRP load (b) vs. total event discharge. Blue squares indicated tile events losses. Red circles indicated combined overland flow and tile losses. The gray dashed line shows an apparent threshold at both sites.

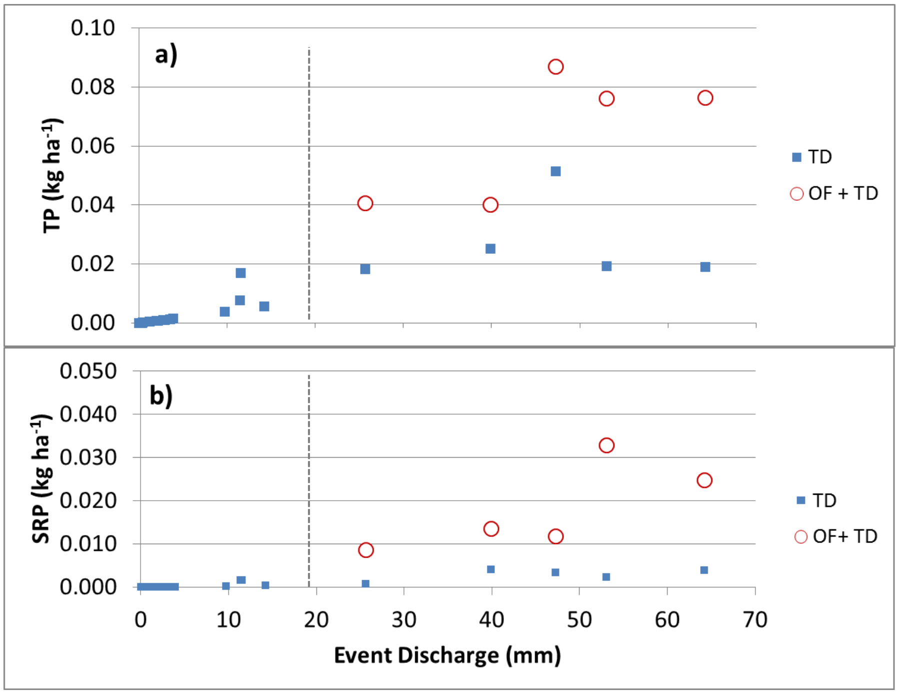

Figure 5-10 Site 3 Event TP load (a) and SRP load (b) vs. total event discharge. Blue squares indicated tile events losses. Red circles indicated combined overland flow and tile losses. The gray dashed line shows an apparent threshold at both sites.

5.3.4 Influence of Event Type on Phosphorus Export during Peak Events

Event hydrographs, P concentrations, air temperature and rainfall were plotted to show the temporal variability within individual peak events related to event type. Plots of four events are included in this chapter to demonstrate the general trends observed, as well as examples of variable responses related to differing event types (Figure 5-11-Figure 5-14).

Tile drainage was the first flow path to respond in all peak events. However, there was variability in the lag time between the initial rise in tile flow and the generation of overland flow (See Figure 5-11, Figure 5-12, Figure 5-14). For example, Event J_{28} at Site 2 (Figure 5-13) which resulted from intense rainfall on frozen soil, lead to a more rapid generation of overland flow in comparison to Event M_{10} (Figure 5-14) which was caused by a combination of less intense rain and snowmelt on unfrozen soil.

P concentrations in tile drainage and overland flow varied temporally throughout all events. Generally, in tile drainage both PP+SUP and SRP reached maximum concentrations near the onset of the event, either at or prior to peak discharge. However, following the initial peak in concentrations, there were also subsequent elevated concentrations associated with additional rainfall, or the onset of overland flow. The response to additional rainfall can be seen in event A_9 at Site 3 (Figure 5-11). The relation between tile concentrations and the onset of overland flow is clearly seen in the M_{10} event at Site 3 (Figure 5-12). Overland flow concentrations were also variable within and among events. In general, concentrations of PP+SUP and SRP in overland flow were elevated near the onset of each event and in response to further rainfall. Following peak concentrations, there was a general decline in P (SRP and PP+SUP) concentrations (Figure 5-11, Figure 5-12, Figure 5-13, Figure 5-14).

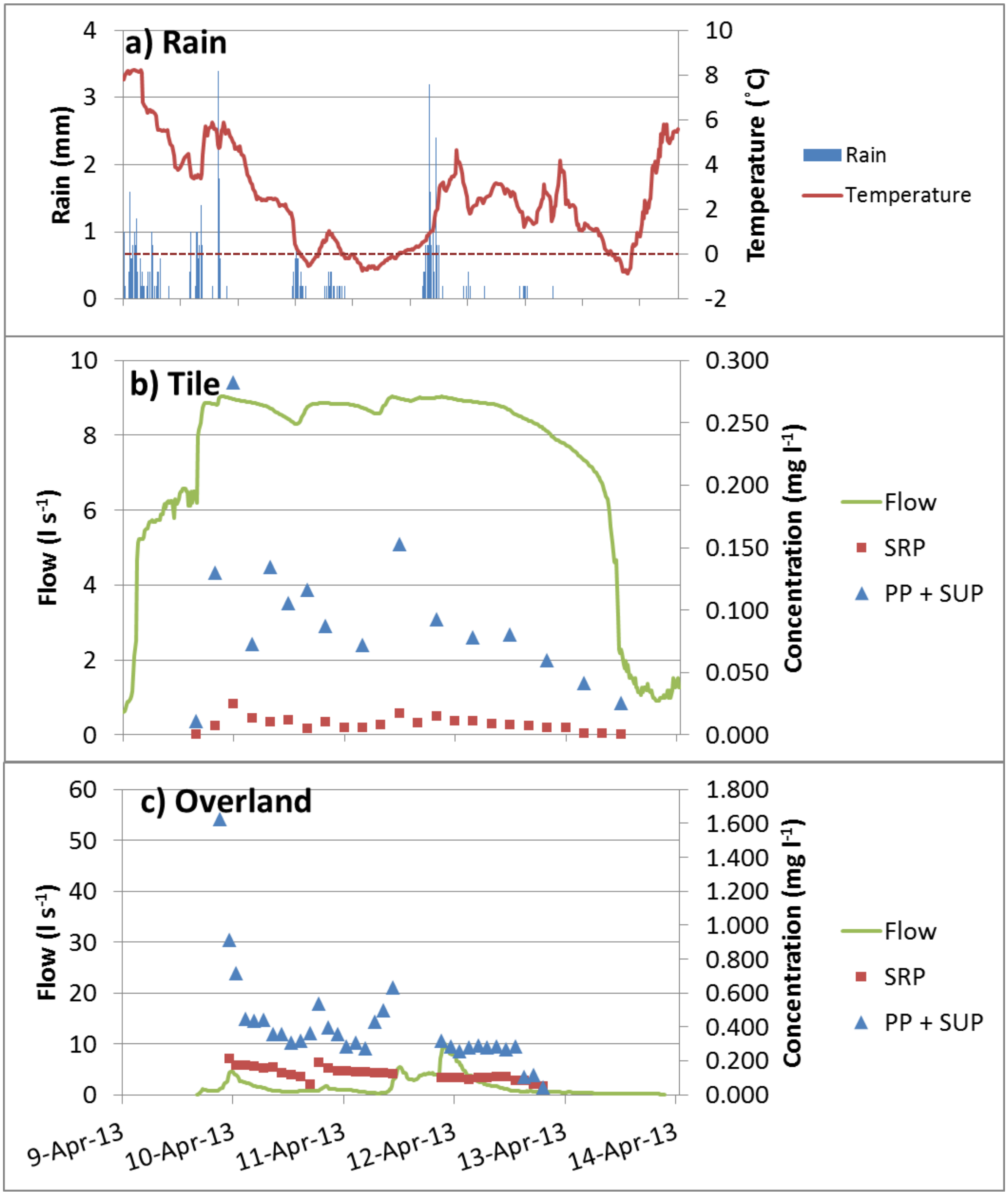

Figure 5-11 Site 3 Event A_9 Plots including Rain and Air Temperature (a), Tile Flow and Chemistry (SRP and PP+SUP) (b), Overland Flow and Chemistry (SRP and PP+SUP) (c). The event occurred on unfrozen soil and was driven by rainfall. Flow, Temperature and Rain are plotted at 15 minute intervals.

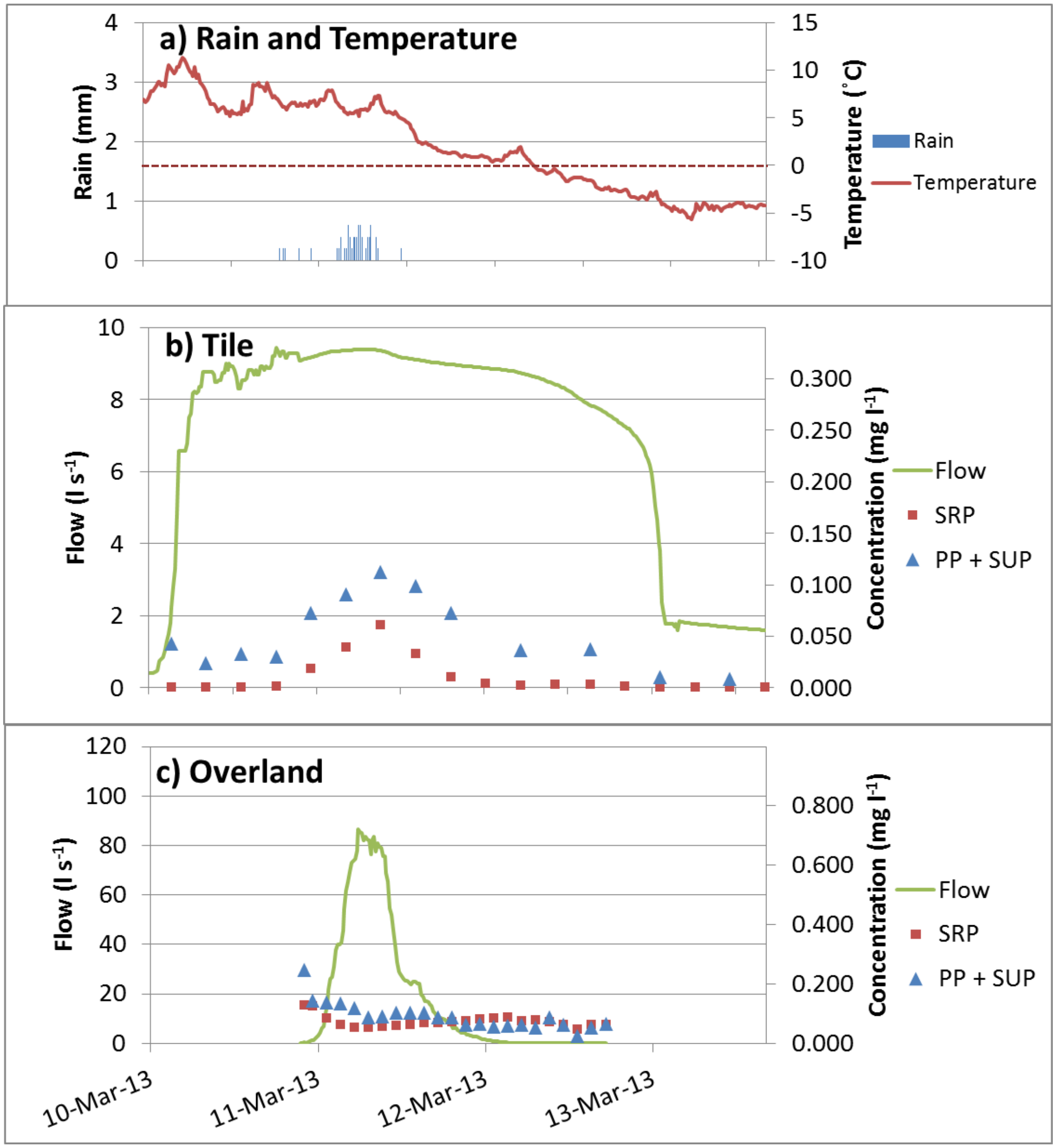

Figure 5-12 Site 3 Event M10 Plots including Rain and Air Temperature (a), Tile Flow and Chemistry (SRP and PP+SUP) (b), Overland Flow and Chemistry (SRP and PP+SUP) (c). Flow, Temperature and Rain are plotted at 15 minute intervals.

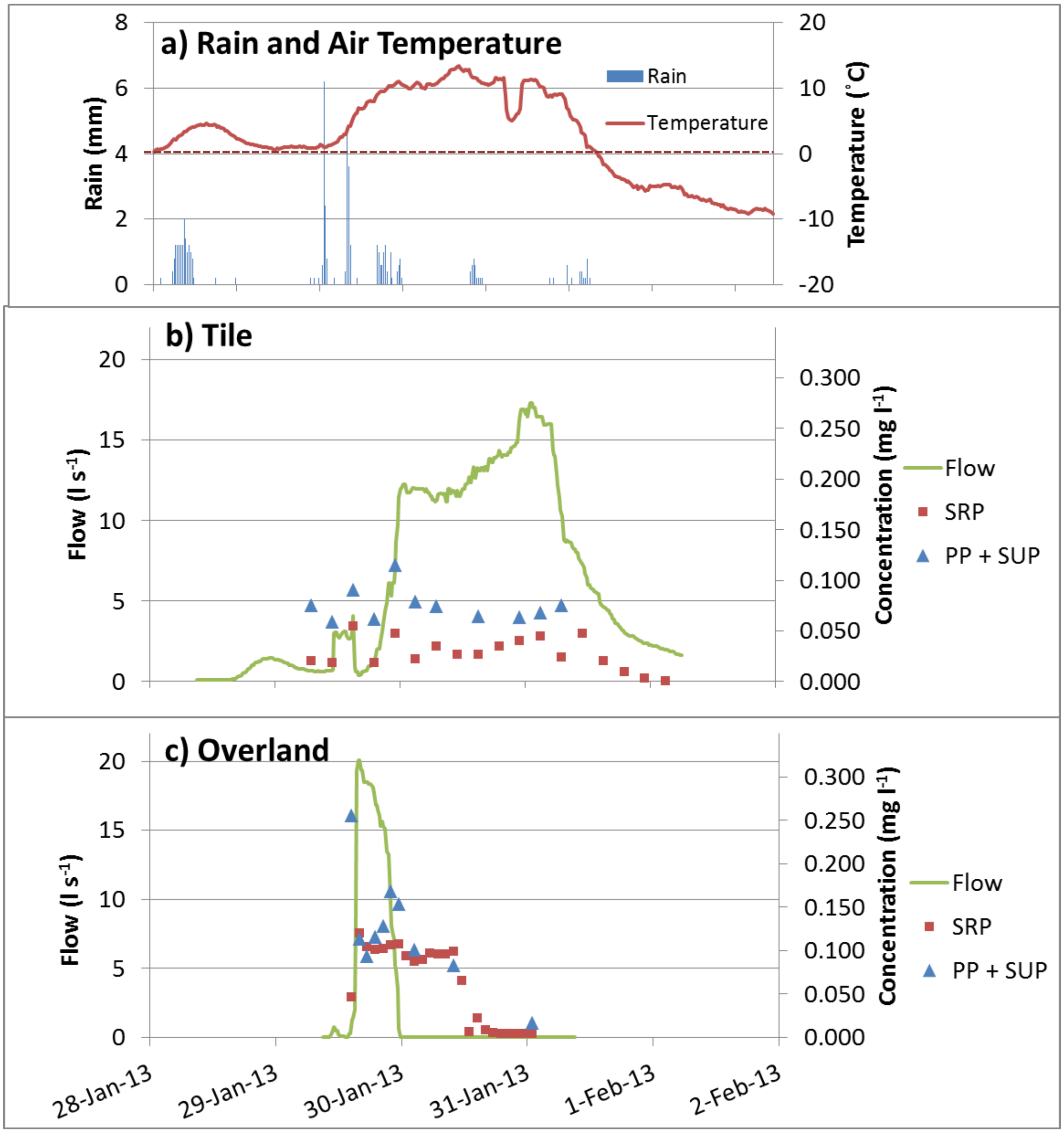

Figure 5-13 Site 2 Event J_{28} Plots including Rain and Air Temperature (a), Tile Flow and Chemistry (SRP and PP+SUP) (b), Overland Flow and Chemistry (SRP and PP+SUP) (c). Flow, Temperature and Rain are plotted at 15 minute intervals. The event occurred on frozen soil and was driven primarily by rain.

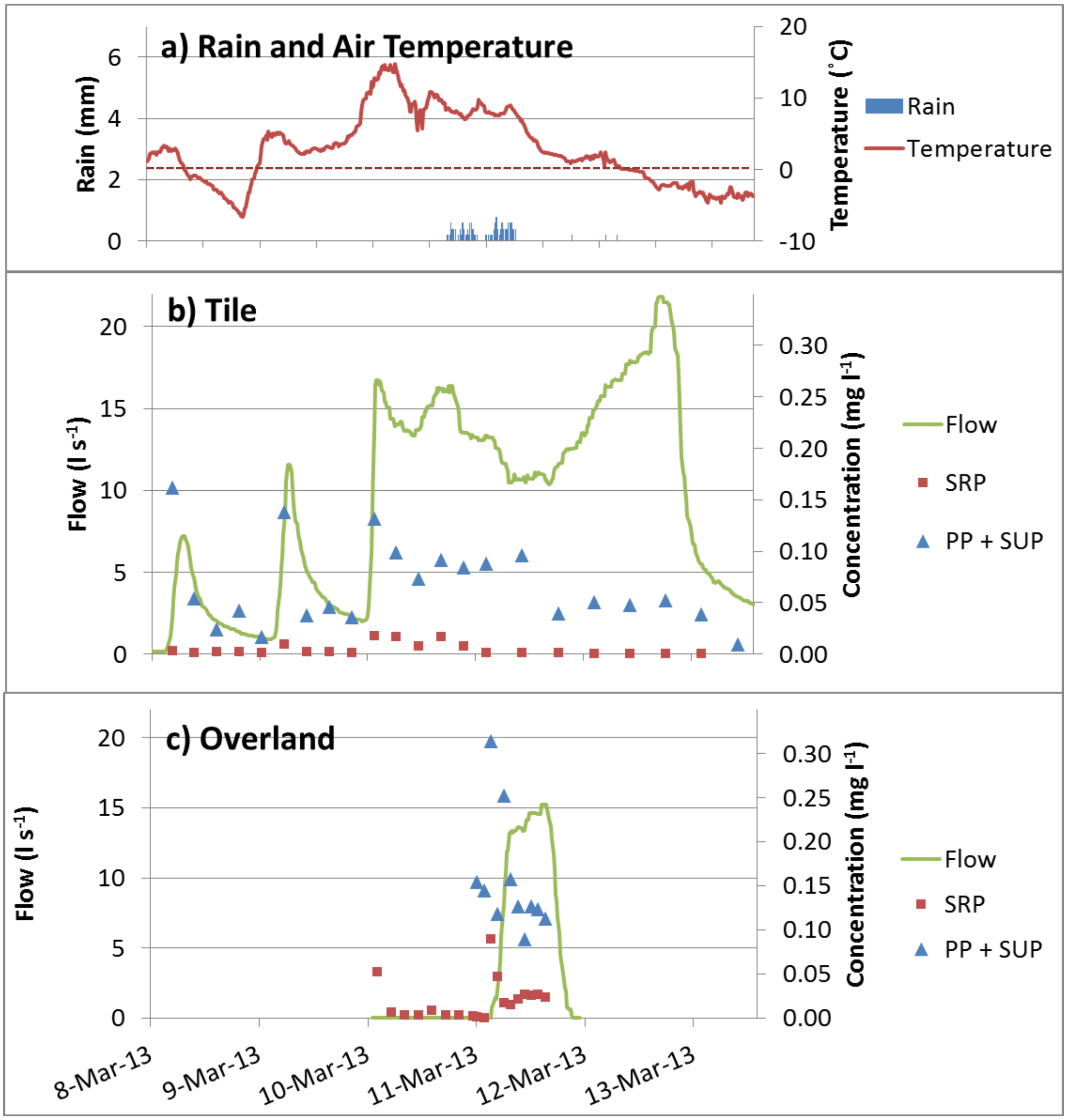

Figure 5-14 Figure 5 14 Site 2 Event M10 Plots including Rain and Air Temperature (a), Tile Flow and Chemistry (SRP and PP+SUP) (b), Overland Flow and Chemistry (SRP and PP+SUP) (c). Flow, Temperature and Rain are plotted at 15 minute intervals. The event occurred on frozen soil and was driven primarily by Rain.

Event PP+SUP flow-weighted mean concentrations (FWMCs) in both overland flow and tile drainage varied among the series of NGS peak events (Table-5-3). In tile drainage, the range in PP+SUP FWMC for all peak events was small relative to what was observed in overland flow, and was similar between the sites (0.07-0.12 mg/l for Site 2 and 0.05-0.13 mg/l for Site 3). In comparison, overland flow PP+SUP FWMCs were higher and more variable (Site 2: 0.12-0.22 mg/l, Site 3: 0.08-0.38 mg/l) (Table 5-3). A comparison of PP+SUP FWMC in overland flow and tile drainage for individual events showed a general positive trend, and thus the two pathways showed similar temporal trends over the study period (Figure 5-15) (Table-5-3). The highest event PP+SUP FWMC for both overland and tile drainage occurred during events where rain fell on bare, unfrozen soil (e.g. Site 2: A_9, Site 3: $D_{4,}$ A_9). In contrast, rain on snow and radiation melt events (e.g. Site 3: J_{11}, J_{28} and M_{10}) had lower PP+SUP FWMC (Table 5-3). The ratio of Rain:Rain+SWE was plotted against event FWMC's to see if event driver influenced speciation (Figure 5-16). At Site 3, Rain:Rain+SWE was positively related to overland flow PP+SUP FWMCs (Figure 5-16 b). Ground conditions and event drivers appear to influence PP+SUP concentrations.

Event SRP FWMC varied due to event conditions and showed a general seasonal decline. SRP FWMCs in tile drainage and overland flow were not as clearly related as PP+SUP FWMCs (Figure 5-15); however, some similarities in temporal patterns were observed. In both overland flow and tile drain effluent, the highest SRP FWMCs and SRP:TP ratios occurred between December-January (D_1-J_{28}) (Figure 5-17, Figure 5-18). Following this mid-NGS peak, the SRP:TP ratio in overland flow declined over the NGS (Figure 5-19). This was driven by changes in both SRP and TP concentrations as the season progressed. Generally, this decline did not appear to vary with event type, although a few notable exceptions. For example, Event J_{28} at Site 2 was the only event with frozen soil (Table 5-1) and had notably greater SRP FWMC than subsequent events. M_{10} at Site 3 had a notably lower SRP FWMC. This event had the highest overland discharge and was the only radiation-driven melt event (Figure 5-18). The SRP:TP ratio of M_{10} was consistent with the seasonal decline (Figure 5-19). SRP FWMCs appear to be influenced by event conditions, and seasonal fluctuations. Time since fertilization was not considered as applications of P occurred in the previous fall period (2011).

Table 5-3 Site 2 and Site 3 event tile drainage and overland flow flow-weighted mean concentrations (FWMC) for SRP, TP and PP.

	Event	Site 2 Tile Drainage	Site 2 Overland Flow	Site 3 Tile Drainage	Site 3 Overland Flow
SRP FWMC (mg l^{-1})	D_1			0.003	0.131
	D_4			0.014	
	J_{11}	0.004		0.012	0.153
	J_{13}	0.017			
	J_{28}	0.033	0.106	0.008	0.120
	M_{10}	0.005	0.023	0.013	0.059
	A_9	0.001	0.015	0.008	0.117
TP FWMC (mg l^{-1})	D_1			0.09	0.37
	D_4			0.15	
	J_{11}	0.06		0.07	0.24
	J_{13}	0.07			
	J_{28}	0.10	0.23	NA	0.22
	M_{10}	0.07	0.16	0.07	0.16
	A_9	0.12	0.22	0.13	0.49
PP+SUP FWMC (mg l^{-1})	D_1			0.09	0.24
	D_4			0.13	
	J_{11}	0.06		0.06	0.09
	J_{13}	0.05			
	J_{28}	0.07	0.12	NA	0.10
	M_{10}	0.07	0.14	0.05	0.10
	A_9	0.12	0.20	0.12	0.38

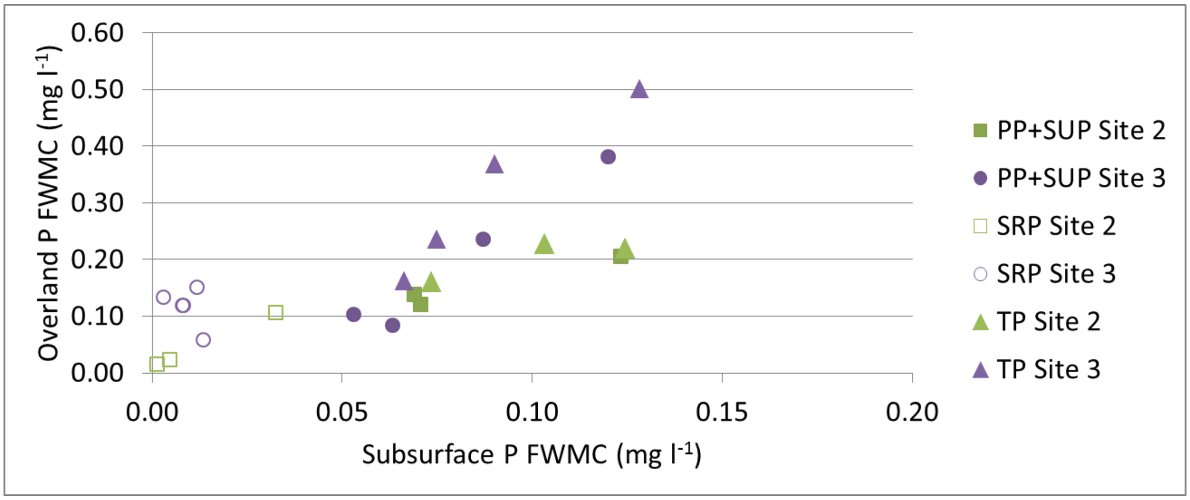

Figure 5-15 PP+SUP and SRP flow weighted mean concentration (FWMC) of overland flow versus tile drainage for Site 2 and Site 3.

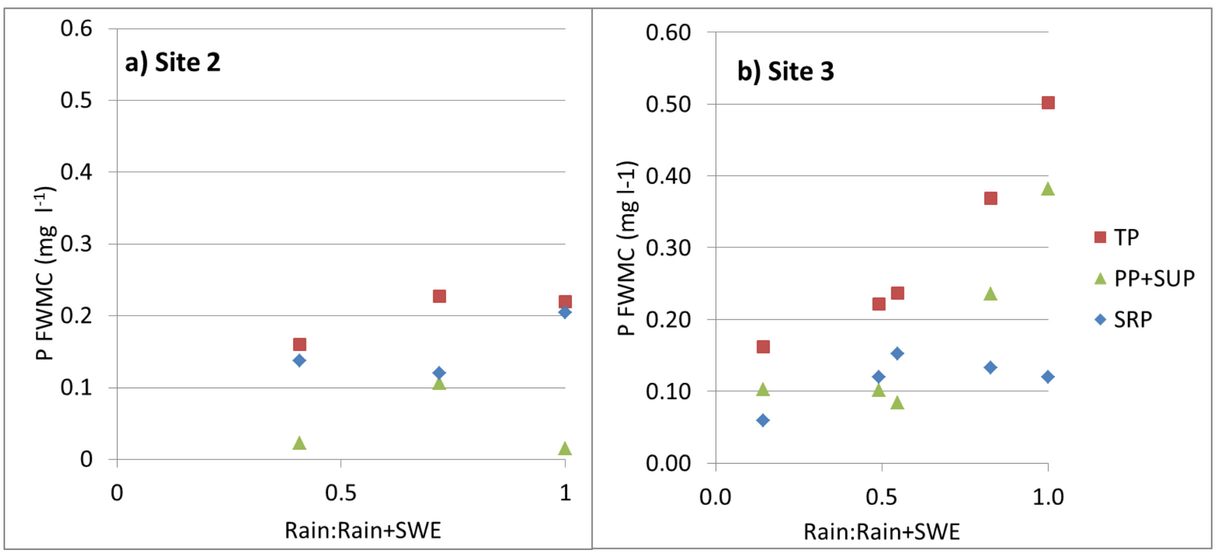

Figure 5-16 Rain to Rain+SWE ratio vs Overland flow TP (red), PP (green) and SRP (blue) FWMC for Site 2 (a) and Site 3 (b).

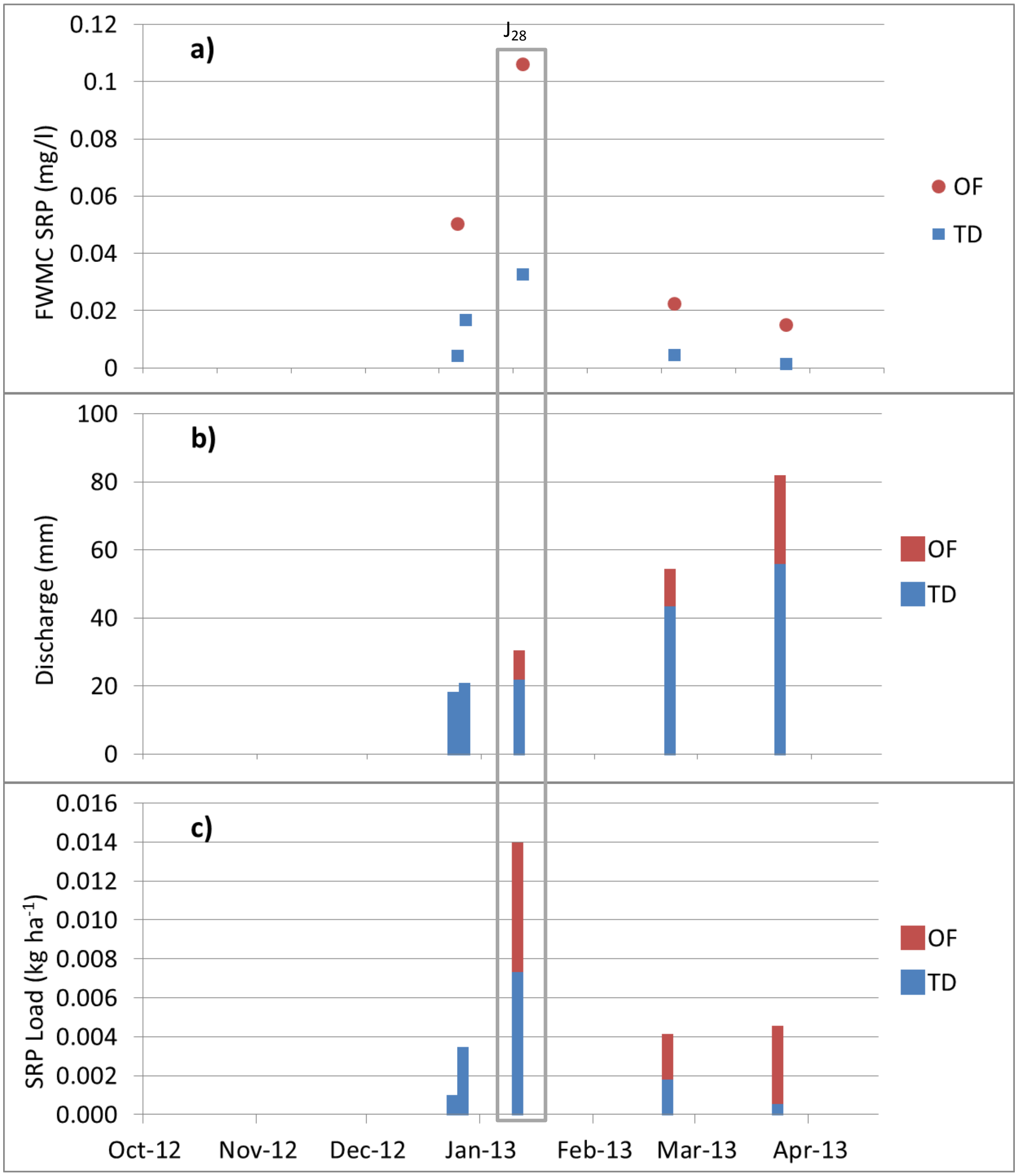

Figure 5-17 Site 2 temporal changes in SRP flow weighted mean concentration (FWMC) (a), Discharge (b) and SRP load (c) for high P loss event for tile (blue) and overland flow (red). Event J_{28} is highlighted as an atypical event.

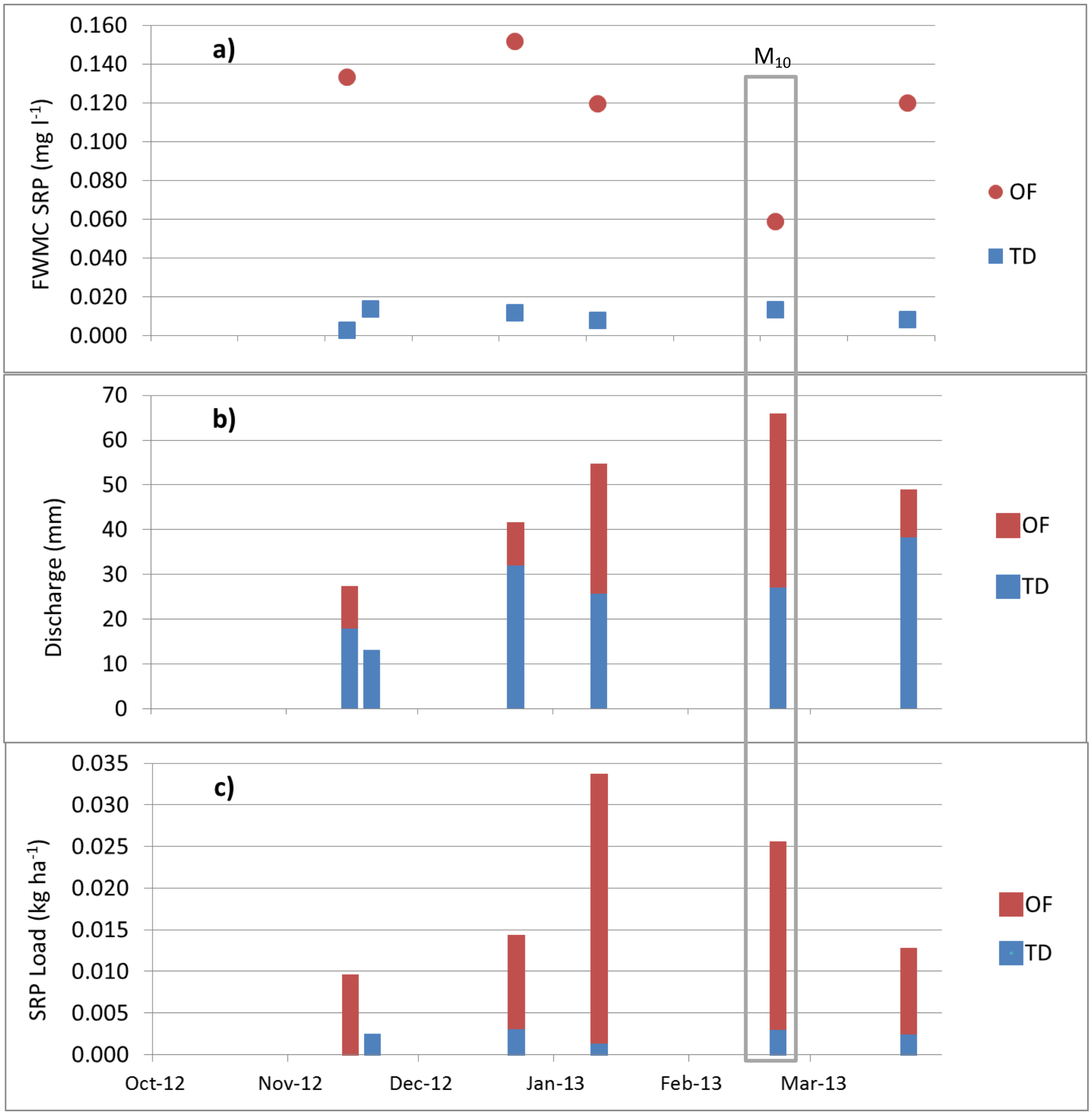

Figure 5-18 Site 3 temporal changes in SRP flow weighted mean concentration (FWMC) (a), Discharge (b) and SRP load (c) for high P loss event for tile (blue) and overland flow (red). All events occurred on unfrozen soil. Event M10 is highlighted as an atypical event.

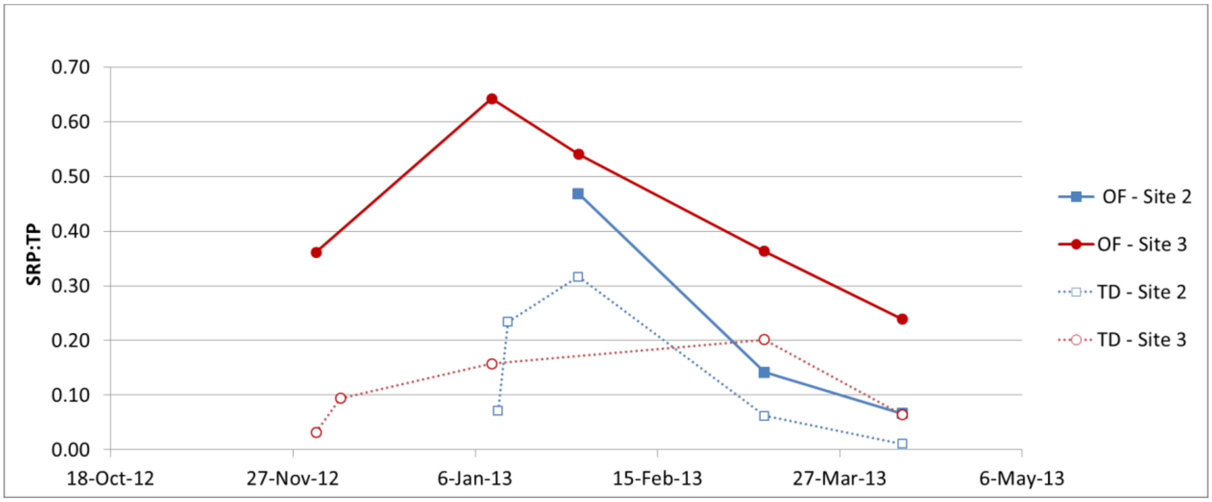

Figure 5-19 Temporal variability in SRP:TP over the NGS for tile drainage and overland flow during peak events at Site 2 and Site 3.

TP FWMCs in peak events varied over the NGS (Figure 5-20 and Figure 5-21) due to event type (driver and pre-event ground conditions) and seasonal fluctuations. TP FWMCs of peak events were higher in the first and last events of the NGS, while the lowest TP FWMCs occurred during the mid-march event at both sites (M_{10}) (Table 5-3, Figure 5-20, Figure 5-21). The TP FWMCs in the initial events were elevated because of two factors: a higher SRP loss (Figure 5-17, Figure 5-18), and rain on bare soil event types that resulted in higher PP+SUP FWMCs (discussed above). In contrast, the higher TP FWMC in the A9 event was driven solely by the flushing of PP+SUP, likely due to rain on bare soil as SRP concentrations were not elevated during this event.

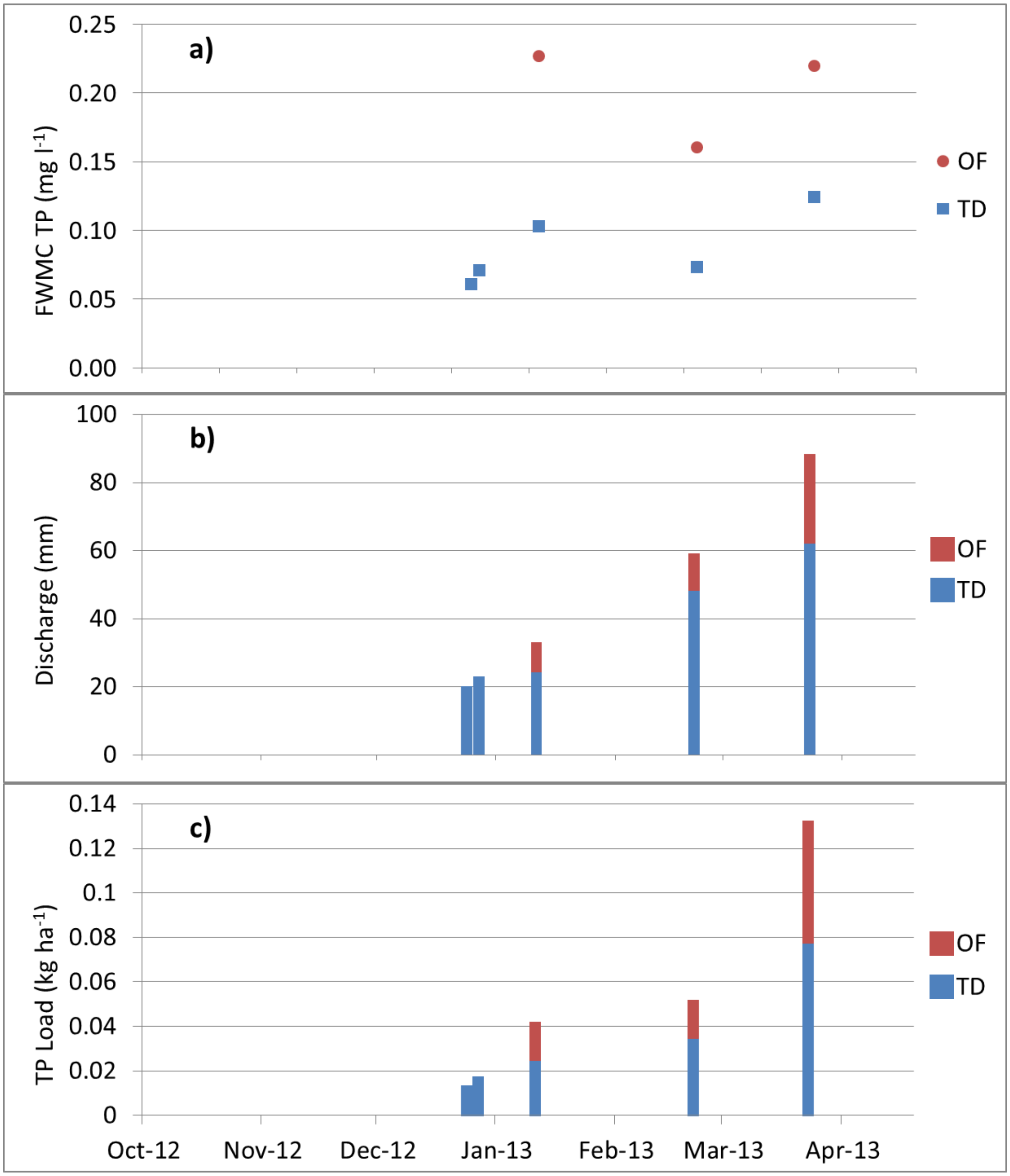

Figure 5-20 Site 2 TP flow weighted mean concentration (FWMC) (a), Discharge (b), and TP Load (c) for peak discharge events.

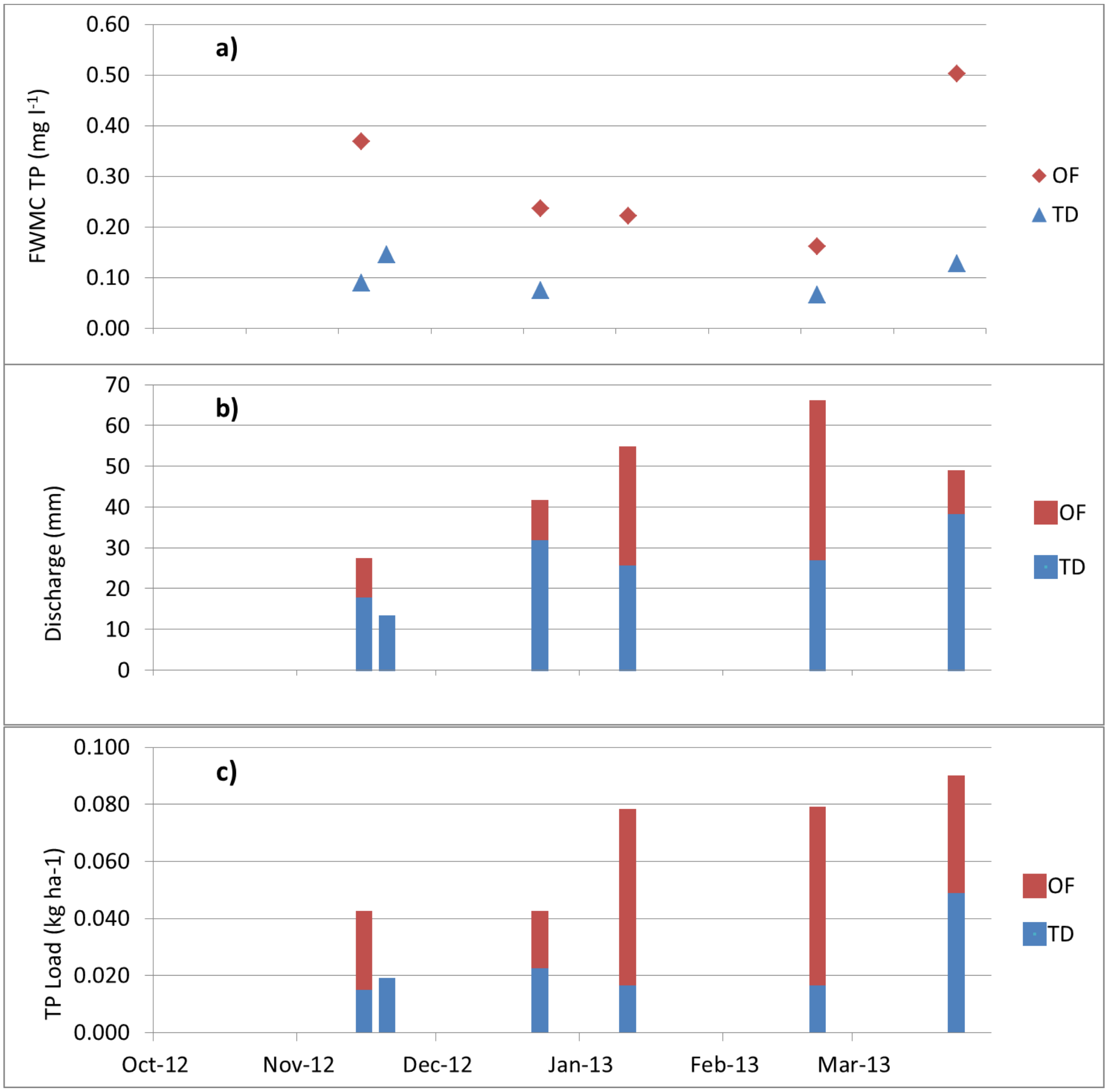

Figure 5-21 Site 3 TP flow weighted mean concentration (FWMC) (a), Discharge (b), and TP Load (c) for peak discharge events.

5.4 Discussion

5.4.1 Seasonality in Runoff Distributions and the Prevalence of Peak Flow Events during the Non-Growing Season (NGS)

The seasonal flow pattern observed in this study is similar to the stream discharge pattern observed in agricultural watersheds across southern Ontario (Ontario Ministry of the Environment: Environmental Monitoring and Reporting Branch 2012). The Ontario Ministry of the Environment (2012) studied 15 southern Ontario streams over a five year period between

2004 and 2009 and found that a large percentage of annual flow came from the winter months. They also noted that flow in this period was typically dominated by multiple high discharge events, driven by melting snow and rainfall (Figure 5-22). The winter experienced during the current study was somewhat atypical based on the high number of complete snow melts. These conditions provided an opportunity to evaluate the effect of event type on P export. Furthermore, the NGS demonstrated conditions that may be representative of future NGS weather based on climate change predictions.

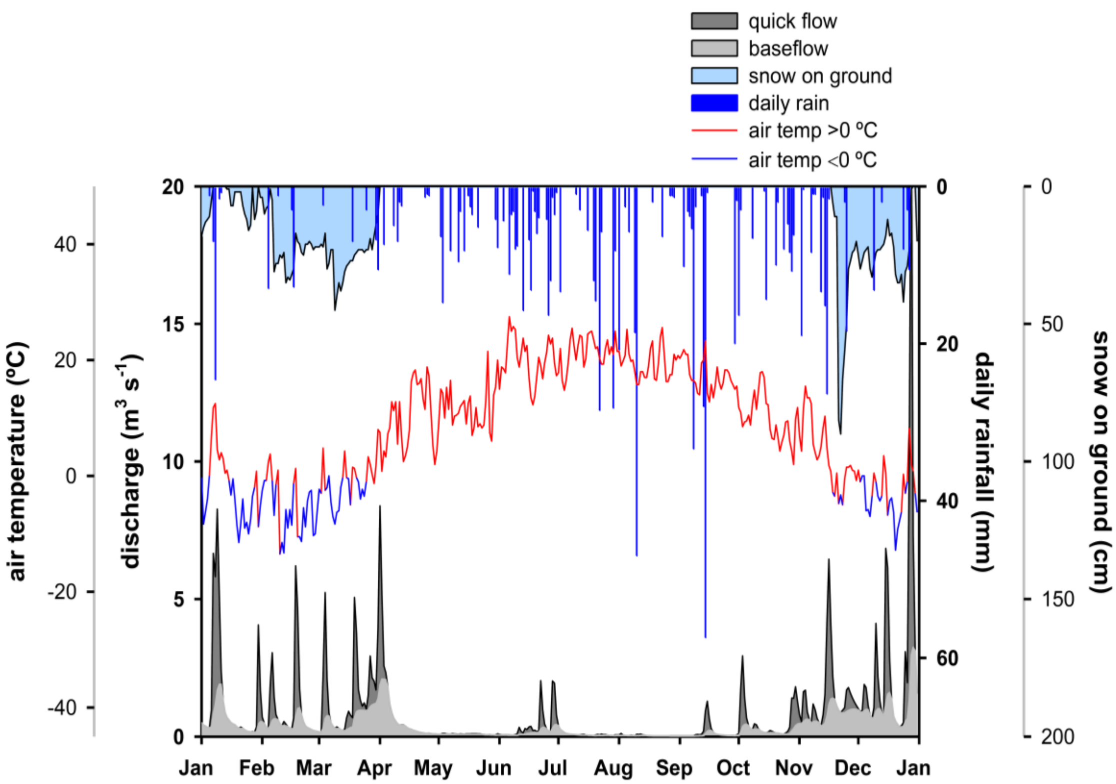

Figure 5-22 Stream discharge, rainfall, air temperature and snow on ground for the Little Ausable River in 2008, a small agricultural watershed located between Site 2 and 3. This plot demonstrates the typical seasonal discharge pattern for that area in southern Ontario. Separation between baseflow and quick flow shows events where overland flow was probable. Note majority of flow and peak events occur between October and April (Ontario Ministry of the Environment: Environmental Monitoring and Reporting Branch 2012).

An objective of this study was to determine the contribution of peak discharge events to annual P mass export. The dominance of peak flow events to annual P losses has been demonstrated in numerous studies conducted globally (Ulén and Persson 1999, Macrae, et al. 2007, Gentry, et al. 2007). In the current study, the series of peak discharge events over the NGS at both sites were responsible for the majority of site discharge, and the majority of annual P export. Other studies have also found that major events during the NGS contribute a large portion of annual P export (Tiessen, et al. 2010, Eastman, et al. 2010, Macrae, et al. 2007, Ball Coelho, et al. 2012, Jamieson, et al. 2003). However, in slight contrast to studies that were carried out in Manitoba and Quebec, which typically attributed P loads to one or two dominant events, P export in the current study was derived from multiple peak flow events taking place over the course of the NGS. Furthermore, peak events in April driven entirely by rain made significant contributions both to discharge and P export. This is reflective of the different climatic conditions experienced in southern Ontario relative to the above-mentioned studies.

5.4.2 Inter-Event Variability and the Significance of Antecedent Conditions

Another objective of this study was to determine the influence of event type (event driver and ground conditions) on P export (P load and P FWMC) during peak discharge events in the NGS. Lui et al. (2013) looked specifically at antecedent conditions and event properties during snowmelt events in western Canada and found that several factors had significant relationships with snowmelt P loading (i.e., duration of runoff, degree days during runoff, cumulative rain, maximum temperature, cumulative snow, and peak flow rate) and snowmelt P FWMCs (i.e., SWE, average flow rate, volume of runoff and precipitation in October). In the Lui et al. (2013) study, flow volume was the most important factor in determining P load. Interestingly, flow volume had a negative relationship with P FWMC due to the effect of dilution (Liu, et al. 2013). The Lui et al. (2013) study also found temperature during the melt, SWE and the duration of runoff to be critical factors influencing P load and P FWMC which suggests that ground conditions, and event drivers are also important factors. In this study, the different combinations of event drivers and ground conditions influenced event export dynamics, and thus, were a potential source of variability in the relationship between TP loading and discharge (Figure 5-9, Figure 5-10). In this study, the greatest PP+SUP concentrations occurred in events which were primarily driven by rain and that occurred on unfrozen soil. These results are consistent with findings in the literature. Su et al. (2011) reported that more PP export occurred during the snowmelt period with higher amounts of rainfall, a finding which agrees with the relationship found in this study between Rain: Rain+SWE and PP+SUP FWMC (Figure 5-16). In a long-term study in Manitoba, Liu et al. (2013) found that PP in overland flow was higher in the second half of the snowmelt, and attributed this increase to the reduction in frozen soil, snow cover and increased susceptibility to soil erosion. Although surface frozen soils are less susceptible to erosion, as soils thaw there is a greater risk to erosion if subsurface soils remain frozen (Rudra, Dickinson and Wall 1989). Subsurface soils that are frozen can reduce infiltration (Kane and Stein 1983), causing increased runoff and erosion of surface soils (Su, et al. 2011) .

Due to the timing of overland flow relative to thawing soil, and the relatively shallow frost layer during the study period, this was not observed in this study. The higher PP+SUP concentrations observed during rain on bare soil events relative to rain on snow events in this study is consistent with the literature.

The presence of frozen surface soil during a runoff event can increase the rate of soluble P export. In Manitoba, Tiessen et al. (2010) found that soluble P concentrations were higher in snowmelt generated overland flow than during rainfall initiated overland flow. They further suggested that when soils are frozen, crop residue on the soil surface may be a significant contributor to soluble P losses. The crops in their study were more prone to leaching relative to corn residue (Lupwayi, et al. 2007); however, corn has also been shown to be a significant contributor of soluble P to runoff under certain conditions (Cermak, et al. 2004, Gilley, et al. 1997). Cermak et al. (2004) found that, when submerged for a one day period, corn and soybean residue leached almost all of the extractable PO_4 within the residue. Furthermore, when P is leached from residue sitting on frozen soil, there is little opportunity for P to be removed from solution. Therefore, it is possible that the elevated SRP concentrations during the event J_{28} were the result of residue losses.

In addition to the variability related to event drivers and ground conditions, there was a seasonal trend in SRP concentrations. At both sites, SRP FWMC peaked in January and declined for the remainder of the NGS. The peak may have been caused by an increase in the available P pool near the surface caused by the leaching of P from corn residue. Messiga et al. (2009) reported that, following several freeze thaw cycles, the soil water extractable P was higher from soils with crop residue applied than soils without. Following the apparent buildup of P at the surface, concentrations declined, which is consistent with the concept of depletion. Depletion is more commonly reported following successive storms after fertilization. Although several studies have noted differences in seasonal speciation in winter events relative to summer events (Ball Coelho, et al. 2012, Macrae, et al. 2007), no studies have reported a decline in SRP FWCM over the NGS. Since concentrations of soluble P typically return to background levels within several months of application (Hart, et al. 2004), the decline of SRP concentrations during the second NGS is not thought to be related to fertilizer application timing.

The M_{10} event at Site 3 had a notably lower SRP FWMCs than others during the NGS. The unique event type, or the higher volume of runoff during the event may have caused this. M_{10} was the only event where the snowpack remained following the event. This could have resulted in less interaction between runoff and the soil, as runoff was seen moving over the snow pack. Another explanation is that the higher volume of runoff during M_{10} caused dilution of the limited P available to runoff. Lui et al. (2013) observed a negative relationship between runoff volume and TP FWMC and attributed this to dilution. More data would be required to establish this relationship at the sites in this study.

5.4.3 Relevance to Climate Change Impact Studies

The series of peak discharge events captured in this study show how pre-event conditions and event drivers can influence P losses. This information is necessary to better predict how beneficial management practices (BMPs) will perform in climate change scenarios. Climate change predictions for southern Ontario involve warmer winter temperatures, similar amounts of precipitation but with more of the precipitation as rainfall (Colombo, et al. 2007). Based on these predictions, it is reasonable to expect changes to winter flow regimes. For instance, changes in the hydro-climatic drivers (rain and SWE) may influence the frequency of overland flow events. For example, if there is less snow accumulation prior to rain on snow or rapid snow melt events, total event inputs may be below the threshold required to generate overland flow, which would potentially result in fewer overland flow events throughout the NGS. Another potential outcome is more frequent melt events over the NGS, and thus the potential for increased frequency of discharge events over the NGS, much like the NGS experienced in this study. Another effect of climate change is the increase in the frequency of extreme precipitation events. This would have an obvious effect on the occurrence of peak discharge events especially if there is an increased frequency of these events in the NGS. Given the role of these events in NGS export, changes to the typical flow regime as a result of climate change will have an effect on P export. P export within peak discharge events is largely determined by flow volume, and thus export is likely to increase or decrease in response to increases or decreases in site discharge.

Climate change in southern Ontario not only means potential changes in the flow regime, but also changes to event drivers and ground conditions within this critical period. Warmer soil temperatures and less snow cover mean that soil will be more vulnerable to erosion over the winter than historically has been the case, which will have implications for P export. If a peak discharge event is generated under these pre-event conditions, PP export may resemble export typical of the rain on soil events observed in this study (i.e., A_9). If PP export is the primary concern, maintaining higher levels of residue cover, as is currently done at both sites, would be a beneficial practice. Some have argued that reduced tillage may be contributing to increased export in the NGS, and thus may not be a suitable BMP (Tiessen, et al. 2010, Hansen, et al. 2000, Puustinen, et al. 2007, Liu, et al. 2014). However, this argument is based on P export alone, and does not weigh the other benefits of the practice. The increased frequency of freeze thaw cycles predicted as part of climate change may also have impacts of SRP losses, particularly in systems where crop residue or cover crops remain on the surface. Different studies have shown an increased risk of SRP losses from soils with residues or cover crops following multiple freeze thaw cycles (Bechmann, et al. 2005, Messiga, et al. 2010). So an increase in freeze thaw cycles could result in greater P losses. High SRP losses have been reported during events that occur on frozen soil (Tiessen, et al. 2010). Warmer winter temperatures would mean fewer days with frozen soil which could reduce winter losses of SRP. Phosphorus export is related to numerous factors, so predicting the effect of climate change on P export is problematic.

5.4.4 Implications for Managing Field-Scale Phosphorus Export

One difficulty in selecting BMPs is that the appropriate management practices may actually vary between regions. In cold regions, the flow regime is often dominated by NGS events, and thus greater P losses can also occur in this period. BMPs that are effective during the GS, are not necessarily effective in the NGS (Hansen, et al. 2000, Puustinen, et al. 2007, Tiessen, et al. 2010). Tiessen et al. (2010) concluded that BMPs for areas with frozen winter soils are likely different than BMPs for areas with unfrozen soils. Both Tieseen et al. (2010) and Hansen et al. (2000) showed P losses from snowmelt driven events are dominated by DP losses, and that the greatest losses occurred in reduced tillage systems. This study has shown that in southern Ontario, the problem for producers and resource managers is that a range of event types are likely to occur between harvest, and next season's planting. In this way, designing BMPs for the southern Ontario flow regime is even more complex than for a flow regime dominated by a distinct snow melt event, or where snowmelt is not a factor. Furthermore, the spatial variability in snow accumulation and overland flow frequency between the two sites alone shows that BMP decisions may vary within the region of southern Ontario as well.

The inter-annual variability at the sites discussed in Chapter 4 adds further complexity to the issue. In years when fertilizer is applied, there is an increased risk of P export, especially if applications are made prior to, or within, periods likely to see overland flow. The risks, however, are reduced if nutrients are incorporated into the soil (Hart, et al. 2004). In these circumstances, tillage to incorporate nutrients may be the best practice, even with the increased risk of erosion following fall tillage. Residue could still be left on the surface over the winter in years where fertilization does not take place. The solutions appear quite complex as P export is the result of the combined influence of site characteristics, events drivers, ground conditions, and land management practices. Appropriate strategies to reduce losses may not only vary depending on the type of events likely to be experienced, but the timing within the producer's planned crop rotation and nutrient applications.

5.5 Conclusions

This study demonstrated the inter-event variability of P export related to event drivers and antecedent ground conditions over the NGS. P speciation during peak events was variable over the NGS. Some of this variability appears related to the effect of ground conditions and event type on PP+SUP FWMCs. Higher PP+SUP and TP FWMCs during rain on bare soil events shows the importance of protecting soil from erosion by leaving residue in place. SRP export was enhanced during events where soil was frozen. There was also a seasonal decline in SRP export through the NGS, suggesting the supply of SRP was limited. This study confirmed soil conditions and event drivers impact P export, and demonstrated the relative risk of different NGS event types which are likely to occur more frequently as a result of climate change. Designing suitable BMPs to be implemented ahead of the NGS in Ontario is problematic, as conditions are less predictable than other regions. Furthermore, the appropriate BMP may differ in years when P applications are made, which further complicates the task. This study has provided an improved understanding of the processes driving runoff and P transport during critical periods in Ontario

6 Final Discussion and Conclusion

This study improves our understanding of edge of field phosphorus (P) export from reduced tillage (RT) systems in Ontario. The year-round monitoring of overland flow and tile drainage using an event-based sampling strategy provided insight into the relative contribution of these export pathways and the influence of hydro-climatic drivers, ground conditions, and land management practices on the seasonality of P export. This is one of the first studies to demonstrate this using multiple sites and multiple seasons. Chapter 4 demonstrated the seasonality of discharge and P export over the study period and confirmed that event based export dominated total annual export. While it may have been an atypical year, the distribution of discharge and export was representative of general trends for the region as the non-growing season (NGS) accounted for a greater percentage of annual discharge and P export at all sites. Chapter 4 also demonstrated that inter-annual variability in P export can be driven by hydro-climatic drivers and land management practices. In-terms of discharge and export partitioning, Chapter 4 showed that tiles were the dominant water discharge pathway and an important P export pathway, but overland flow also contributed a large amount of total P (TP) and soluble reactive P (SRP) to annual losses. This has important implications for our understanding of the role of tile drains in P transport. Lastly, although export from the sites were low relative to fertilizer inputs and crop removal, the concentrations of P leaving these systems during event flow was elevated above levels associated with eutrophication. Therefore, management opportunities to further improve efficiencies in these systems should be investigated.

Building on the findings from Chapter 4, Chapter 5 aimed to improve our understanding of how event type (event driver and ground conditions) influenced P export within peak discharge events. The second NGS experienced during the study period provided an excellent opportunity to evaluate the influence of these factors on P export. It was evident that event driver and ground conditions influenced P speciation within events. In addition to the inter-event variability, caused by pre-event conditions and event drivers, there was an apparent seasonal trend of SRP build up and decline. Due to the observed variability in speciation, Chapters 4 and 5 showed the importance of year round monitoring as well as the importance of capturing major flow events. This thesis has provided critical information and field data regarding processes occurring over the NGS, including the winter season, which is poorly represented in the current scientific literature.

Both chapters highlighted aspects that make managing P export and selecting BMPs complicated. As other authors have stated, managing P losses requires a suite of BMPs (Tomer, et al. 2013, Sheng, et al. 2011). Reducing tillage intensity should be paired with application methods which reduce the stratification of P, and the vulnerability to export during higher risk periods. Steps taken to manage P losses must also be balanced with other conservation goals. So although some studies have shown RT systems may increase soluble P losses (e.g. Tiessen et al. 2010), this undesired consequence must be weighed against the other benefits. No-till and RT systems have well documented benefits for soil health (Margulies 2012), which should remain a top priority of overall conservation efforts in all environments.

There are several opportunities to reduce P losses at the field scale: 1) avoid soil erosion 2) avoid a buildup of P by maintaining an appropriate level of soil test phosphorus (STP) in soils, 3) minimize export directly related to P applications by applying P below the surface during periods less susceptible to high runoff events, 4) managing overland flow volume with structures, and lastly 5) P filtration or removal. All systems in this study are already taking steps to reduce erosion and have avoided unnecessary buildup of STP. Despite these efforts, concentrations and loading from these well managed systems are still above levels associated with eutrophication. Thus there is need for further improvements to these systems. There may be opportunities to further reduce P export by addressing opportunities #3,#4 and #5 mentioned above. As discussed in Chapter 4, P export directly related to recent P applications can be mitigated be using alternative application methods and making applications in periods with lower risk. However, application during the lower risk period may not be practical because crops are growing in the field. Certain crops which are harvested earlier in the GS afford the opportunity to apply P before periods of elevated risk; however, this can increase the risk of nitrogen leaching. If applications were limited to this time period, the producer may have to increase the use of these crops in their rotation (e.g. wheat), or sell manure to nearby operations where application is possible. The window for low risk application could be extended if an appropriate application method was used.

Alternatively, P export could be reduced by reducing the amount of runoff, by capturing it in water and sediment control basins, or ponds for irrigation, or by increasing infiltration. Control basins have been shown to reduce annual P loading (Tiessen, et al. 2011). These structures are costly to construct, may require land to be taken out of production, as well as potentially increase the hydrologic connectivity of the landscape. Lastly, efforts can be made to remove P from runoff using a variety of approaches. The issue with treatment is dealing with the quantity of water during peak flows. The ability of soil to buffer P concentrations was apparent in this study, based on the observed differences in concentrations between the two pathways. Blind inlets that include a layer of soil may provide a solution. Smith and Livingston (2013) found that using blind inlets reduced TP loading by 78% relative to hickenbottom inlets. However, the long-term perform of this BMP is not known, and these systems do not drain water as quickly.

When addressing the challenge of reducing P export we must remember that although there is potential to reduce losses with on-farm BMPs, P loss from even the best managed systems is inevitable. To meet water quality targets a suite of BMPs may be required. Tomer et al. (2013) proposed a conceptual framework for agricultural watershed planning. They suggested that improving nutrient and water efficiencies in fields should be the first priority. Practices that improve soil health, such as reducing tillage and using cover crops help to avoid water quality problems in the first place. The second focus is then on controlling water and nutrients movement at various scales; in the field, below the field and at the riparian area. However, all control and treat approaches are more effective and easier to maintain after steps to improve water and nutrient efficiency are taken at the field-scale (Tomer, et al. 2013).

Efforts at the field and sub-watershed scales may be lost at the larger watershed scale. In a review of watershed scale benefits of conservation practices, Tomer and Locke (2011) found that although there were measureable benefits of BMPs at the field-scale, the benefits were not always observed at the large watershed scale for numerous reasons. They noted the importance of targeting conservation efforts in critical source pathways. They also noted that natural stream processes contributed the majority of sediment export rather than in-field erosion, and further stressed that unless in-field erosion control is accompanied by a reduction in flow volume, natural stream processes will likely result in enhanced in-channel erosion. The benefits of BMPs were masked by the changing climate, and legacies of previous poor management. Lastly, they noted the complexity of addressing numerous water quality parameters, as there are often trade-offs associated with reducing a particular contaminant (e.g., implementing no-till to reduce erosion and nitrogen losses can lead to increased P losses in some environments) (Tomer and Locke 2011).

Future efforts to mitigate P export must outweigh the value of different practices and conservation targets. The P export from the systems in this study has highlighted that there is opportunity for further improvement even in operations currently implementing many BMPs. The complexity in selecting appropriate BMP was also apparent. If further reductions in P losses are required on the farms monitored, then careful consideration should be given to adjusting the timing of P applications, and application methods. Adjustments to these practices would reduce both tile drainage and overland flow losses. If these adjustments are not suitable to the operation or not balanced with other conservation priorities, additional improvements could be made with complementary BMPs to treat runoff at the field edge with water and sediment control basins (Tiessen, et al. 2011) utilizing blind inlets (Smith and Livingston 2013), or by capturing runoff and using it for irrigation (Sheng, et al. 2011). Unfortunately, these require a significant investment, and often require land to be taken out of production. It is clear that P export from agricultural land results from the combined influence of multiple factors, and that effectively addressing P export issues will likely require strategically implementing a suite of site specific BMPs.

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Appendix A

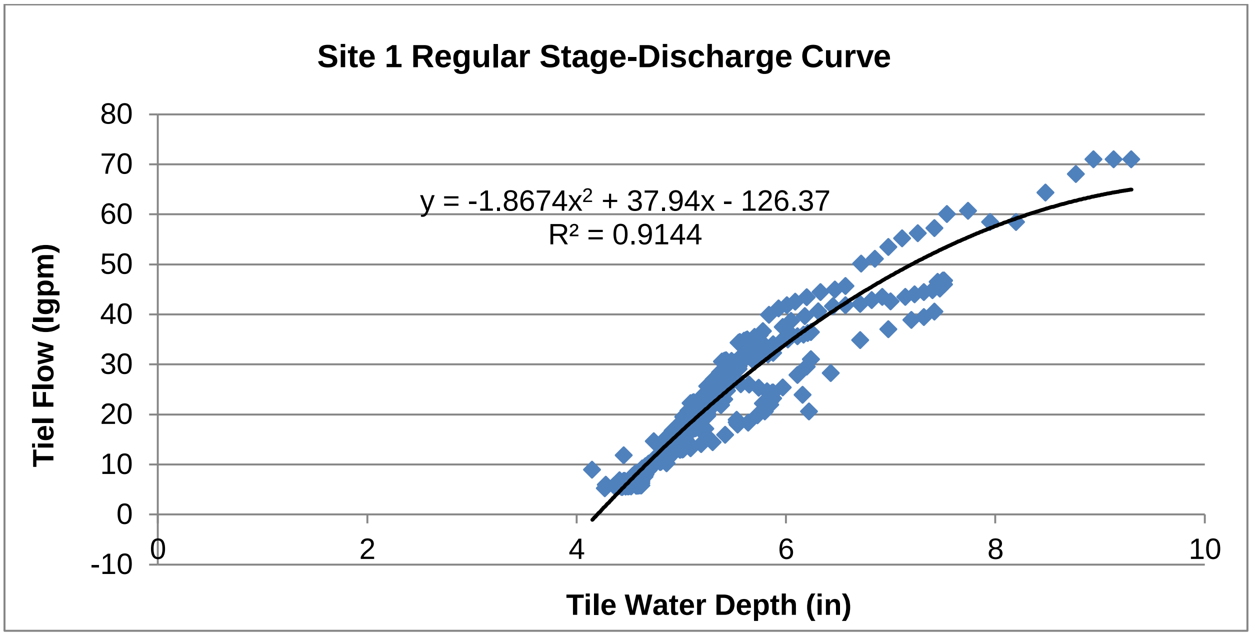

Figure A-0-1 Site 1 Tile stage-discharge curve used between 4.2 and 9.3 inches.

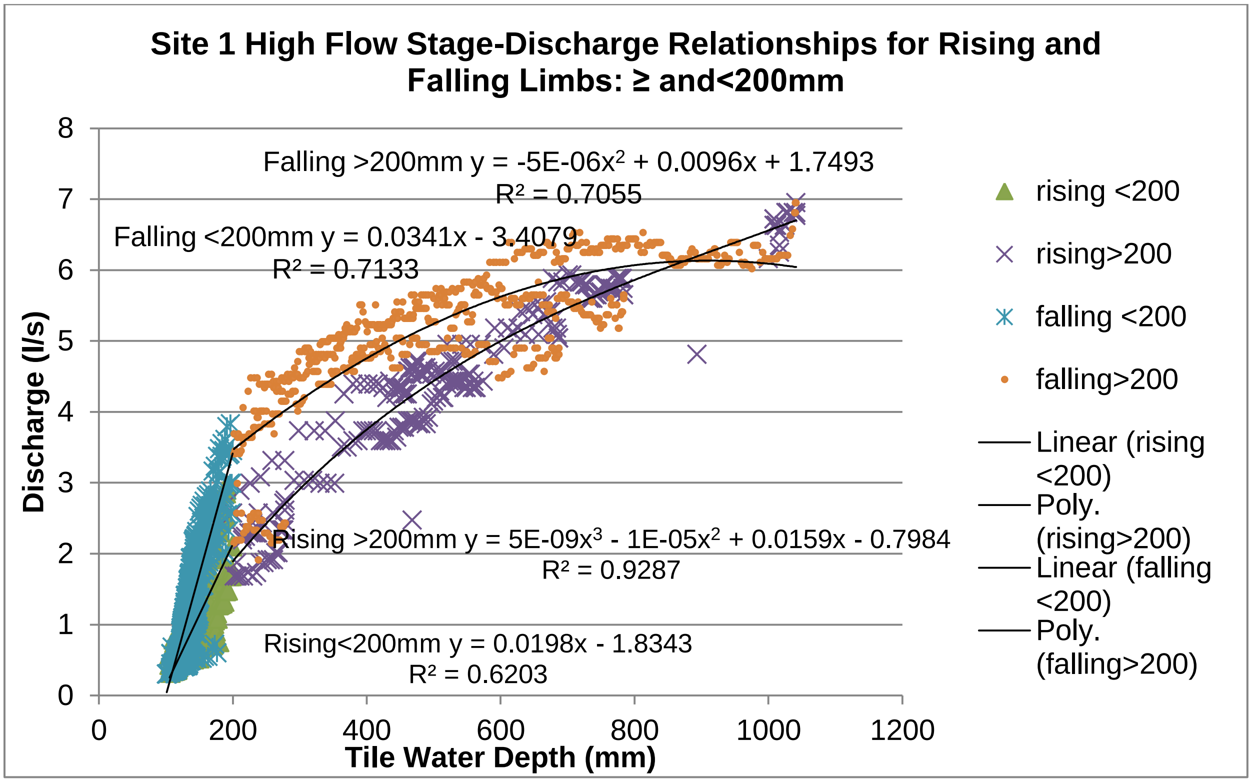

Figure A-0-2 Site 1 High Flow Stage-Discharge Curves for events that peaked over 200mm. Multiple curves were used depending if the stage was on the rising or falling limb of the hydrograph and whether the stage was greater or less than 200mm.

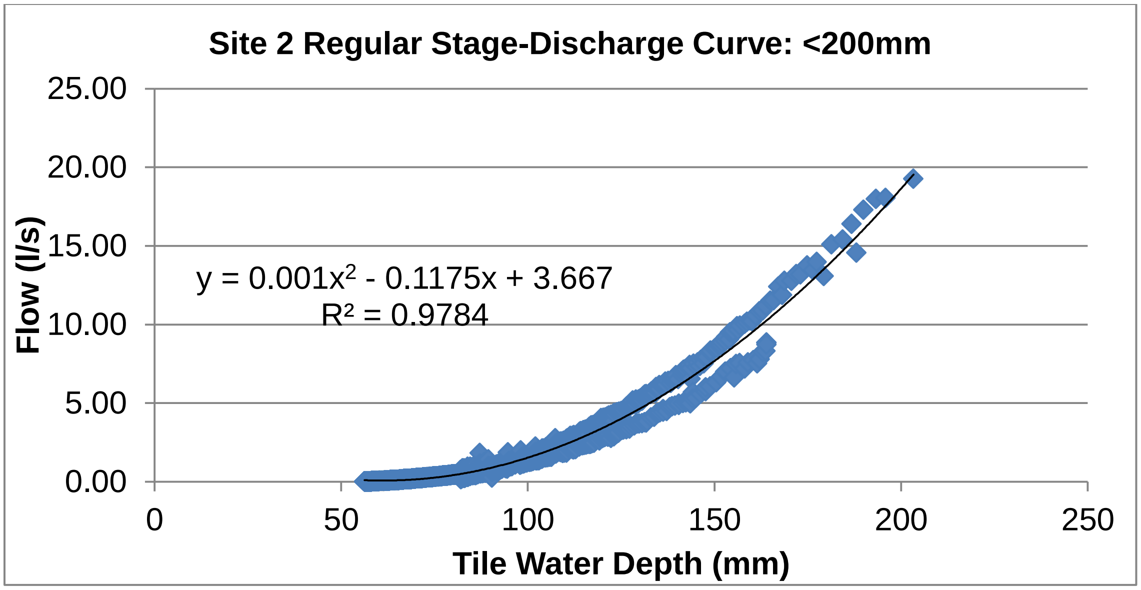

Figure A-0-3 Site 2 Stage -discharge curve for stages <200mm.

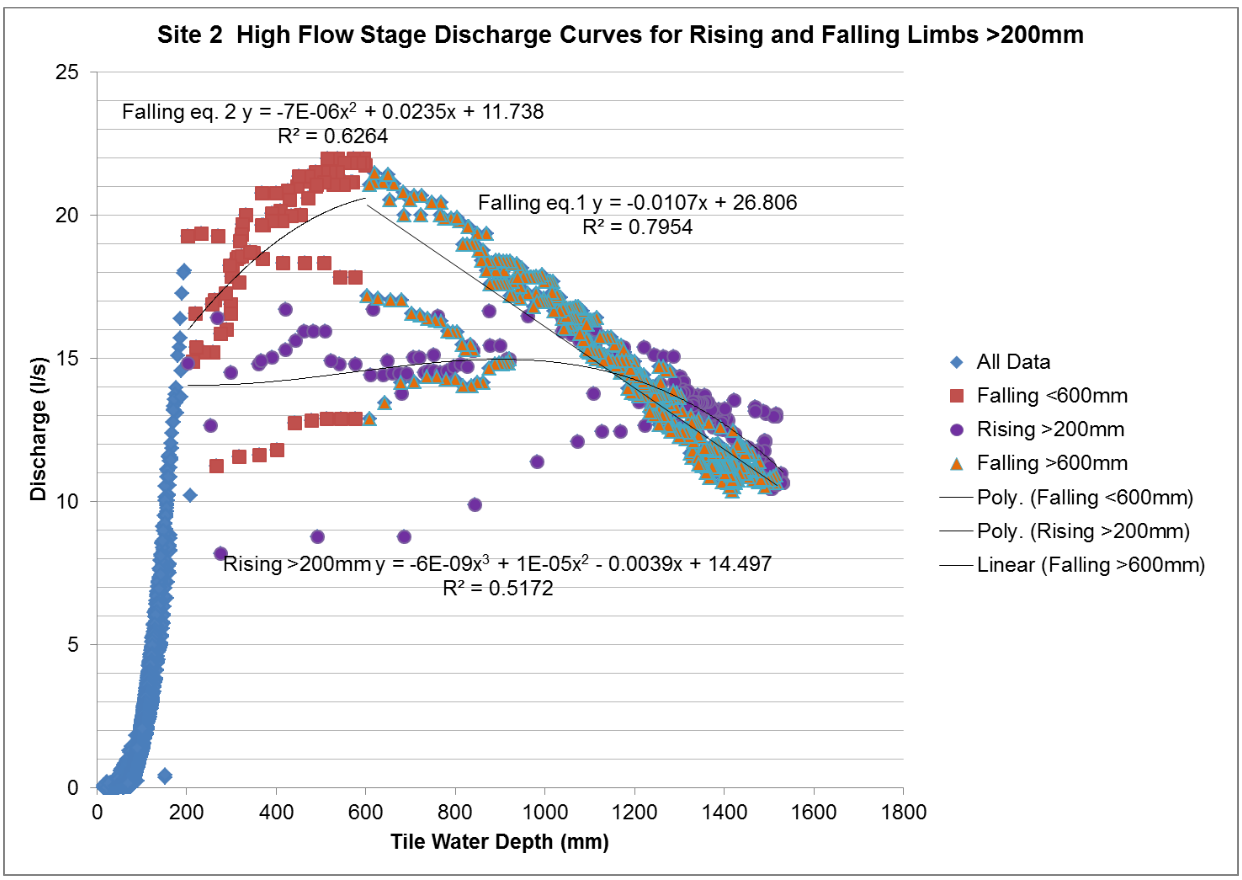

Figure A-0-4 Site 2 High Flow Stage-Discharge Curves for events that peaked over 200mm. Multiple curves were used depending on if the stage was on the rising or falling limb of the hydrograph, and two different curves were used for the falling limb. Below 200mm, the regular curve was used

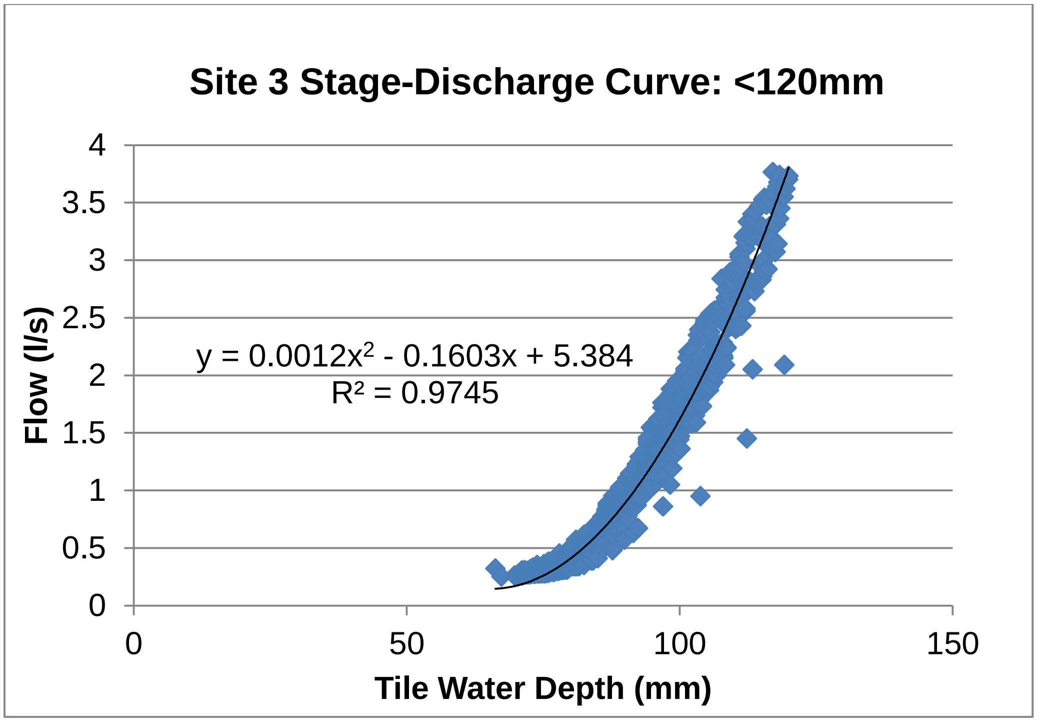

Figure A-0-5Site 3 Tile stage-discharge curve for depths <120mm.

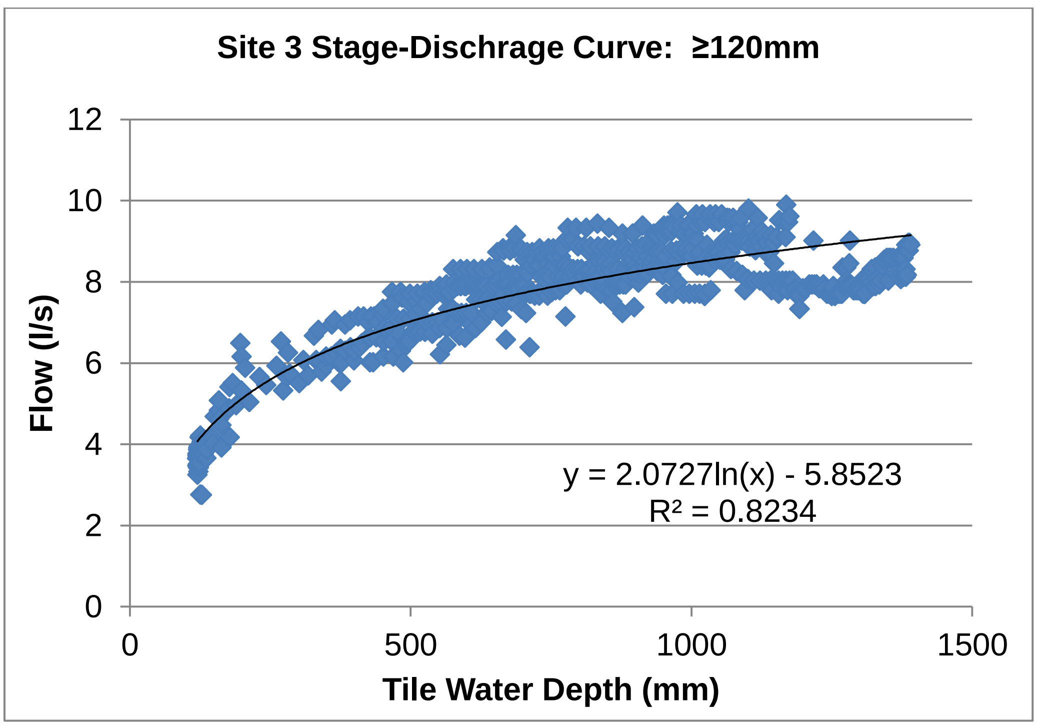

Figure A-0-6 Site 3 Tile stage-discharge curve for depths ≥120mm.