Effects of Tillage and Fertilizer Placement on Subsurface Phosphorus Loss Following Fall Manure Application over the Non-Growing Season

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, as accepted by my examiners.

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Abstract

Increased phosphorus (P) loadings from agricultural runoff into the Great Lakes can lead to eutrophication, resulting in harmful algal blooms and hypoxic conditions. Many studies have demonstrated that subsurface tile drains contribute to total P loss, particularly under no-till. However, most studies have been conducted on soils receiving synthetic fertilizers, and less is understood regarding P loss in tile drains following manure application and if and how tillage and/or manure placement can impact these losses. The goal of this study was to determine if different management practices i.e., conservation till, conventional till, and incorporation, mitigates P loss through tile drains following fall manure application over the Non-Growing Season (NGS). The objectives of this field-based study were to: 1) quantify annual runoff, and P loss from tile drains in a silt loam soil throughout the NGS; 2) investigate if losses differ between conventional and conservation tillage; and 3) determine if incorporation of manure impacts P loss in tile runoff. Tile discharge was monitored from 3 adjacent tile drains with different management treatments (annual till without incorporation, conservation till (with and without manure incorporation) over the span of 8 years (2011-2018), with water samples collected during runoff events for most years during this period. Two years that followed fall manure application (2014-15, 2017-18) were selected for more intensive study. Most P loss occurred during discrete hydrologic events over the NGS, predominantly during the first large discharge event. During this event deep annual tillage increased P loss compared to conservation tillage, with manure incorporation further reducing P loss resulting in differences in cumulative P loss in the tiles over the NGS. This study highlights the importance of in-field long term monitoring in order to capture temporal and spatial variability within a system and recommends that fall manure is incorporated to reduce P losses in tile drains.

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Chapter 1
Introduction and Literature Review

1.1 Problem Statement

Agriculture provides an essential product for the human population. Canadian agriculture not only supports our population but also makes significant contributions to the economy. In 2016 Canadian agriculture and agri-food systems made up 6.7% of the Canadian Gross Domestic Product (GDP) generating $111.9 billion with 12.5% of the Canadian population being employed within the sector (Agriculture and Agri-Food Canada 2017). The development of agriculture over time has been well documented with society's transitioning from hunting and gathering to agriculture and settlement being recorded as early as circa 10,000BC (Weisdorf, 2005). As a result, anthropogenic impacts from agriculture on our environment have been evident as production has intensified. These observed impacts include deforestation, soil degradation, and an increase in fossil fuels, water use, and pollutants (Clay, 2013). Farmers face constant challenges in trying to maximize yield while reducing cost, which is further complicated by the potential impacts on the environment.

The accumulation of excess nutrients in our freshwater bodies has remained a top environmental concern, with nutrients from agriculture largely contributing to this problem. This enrichment of nutrients within a water body is also known as eutrophication. Since the majority of algal growth is nutrient limited by phosphorous (P) in the water, excess amounts of P is of the most concern. When algae growth is no longer limited by the amount of P in the water, there is the potential to develop harmful algal blooms and hypoxic conditions (Hallegraen 1993). Harmful algal blooms can be categorized into planktonic or benthic blooms and are composed of various chlorophytes and cyanobacterium with their growth being controlled by the amount of P in the water. When these blooms reach a certain threshold they are able to deplete the oxygen in the water due to uncontrolled growth and also have the ability to produce toxins when the cyanobacterium *Microcystis* is present. It is still unclear what causes these blooms to produce toxins; however, the size and species makeup of each bloom is determined by a number of environmental factors such as temperature, pH, and ecosystem biodiversity (Scavia et al., 2014). Additionally, shallow littoral zones are at the greatest risk for developing hypoxic conditions resulting in devastating impacts on the invertebrate community (Scavia et al., 2014). Eutrophication not only has environmental impacts such as the ones discussed above but also has severe social and economic consequences. For example, in the Lake Erie basin this includes affecting the drinking water to over 11 million consumers in addition to affecting the tourism, shipping, and fishing industries (Watson et al., 2016). Therefore, maintaining good water quality in our great lakes is vital not only for our environment, but also our economy.

Irrespective of societal concern and the economic impacts, eutrophication continues to be an environmental issue with no simple solution. “Wicked” problems are defined as unique complex problems involving humans with no single solution or end point (Thorton et al., 2013). As a result, due to the multi-level stakeholders in addition to the many geochemical and hydrological cycles involved, eutrophication falls under this category of being a “wicked” problem. Unfortunately, this means that no simple step-wise solution exists to tackle this issue and that extensive research is necessary in order to understand all aspects of this problem.

The agricultural industry has been active in trying to find effective Best Management Practices (BMPs) to minimize P loading to our freshwater bodies while not impacting productivity. However, numerous gaps within the literature remain regarding the effectiveness of certain agricultural BMPs due to the multitude of environmental factors that impact P transport within a system. Uncontrollable factors such as climate and soil type in addition to controllable factors such as fertilizer management can all influence P transport within a system. Furthermore, our understanding of current systems is currently changing due to the effects of climate change. In light of these challenges, we need to increase our understanding of P in order to find effective BMPs to mitigate these challenges and better manage P.

No-till is a popular BMP that was once proposed to reduce P losses from agricultural fields. However, recent work has determined that no-till can lead to unintended consequences, where it may increase P losses in tile drainage (Jarvie et al., 2017). However, much of the evidence of this has been generated using synthetic fertilizers in clay soils throughout the growing season and less has been done in coarser textured soils with more pronounced winter climates (Lam et al., 2016). Thus, this thesis examines the impacts of tillage and manure application on losses of P in tile drainage throughout the non-growing season in a silt loam soil in Ontario, Canada.

1.2 Introduction to Terrestrial Phosphorus

1.2.1 Origin and Uses of Phosphorus

P is a nutrient and essential for all biota to survive due to its role in many biological functions. This includes being a major structural component of many cellular structures, in addition to a key component in photosynthesis and metabolism (Compton et al., 2000). For example, phosphate ester bridges make up a main building block of DNA and RNA and hydrozylapatite is essential for the development of bones and teeth (Filippelli, 2008). Additionally, adenosine triphosphate (ATP) plays a crucial role in energy transport in plants, animals, and microbes (Filippelli, 2008). In terrestrial systems P exists in many forms and is stored within the bedrock, soil, or biota and does not exist in a gaseous state within the atmosphere (Compton et al., 2000). P cycles between these stores through various biological and geochemical reactions which will be discussed below.

1.2.2 Forms of Phosphorus and the P cycle

The P cycle is a very slow process taking millions of years to complete, starting from rock deposits and eventually making its way to water and sediments (Holtan et al., 1988). P enters the system through the weathering of this bedrock rich in apatite. Apatite is a phosphate mineral and can be categorized into 3 groups including hydroxylapatite, fluroapatite, and chloropatite frequently written as ($Ca_5(PO_4, CO_3)_3$ (F, Cl, OH) (Holton 1988). These apatite deposits can be further subdivided into two types; igneous and sedimentary. Igneous deposits are usually low grade and contain apatites of fluroapatite [$Ca_{10}(PO_4)_6F_2$] and hydroxyapatite [$Ca_{10}(PO_4)_6OH_2$] and is relatively unreactive making it not well suited for direct application as fertilizer. Sedimentary deposits make up approximately 80% of apatite bedrock with many containing carbonate-fluorapatite. Sedimentary apatite that contains a higher proportion of carbonate substitution is highly reactive and therefore a great source of fertilizer (Stewart et al., 2005). Regardless of the deposit origin, once P has been weathered from the bedrock it then moves to the soil and biota or is lost from the system.

P in the soil can be categorized into 4 groups consisting of plant available inorganic P in addition to non-plant available P including organic P, inorganic adsorbed P, and mineral P (Ruttenburg, 2003). P can cycle between these forms through various geochemical processes such as weathering, precipitation, adsorption, desorption, mineralization, and immobilization. Firstly, mineral P (apatite) as discussed above can become plant available inorganic P through weathering. However, inorganic P can return to mineral form through precipitation when reacting with dissolved iron, aluminum, manganese, and calcium (Ruttenburg, 2003). Inorganic P is the only form available to be taken up by plants and is found primarily as orthophosphates including HPO_4 and H_2PO_4. This can be made available through mineralization from organic sources or made unavailable through immobilization where inorganic P is taken up by soil microbes and converted into organic forms. Organic P can also be held in the soil in plant residue and humus (Compton et al., 2000). Furthermore, inorganic soil P can be made unavailable to plants when adsorbed to the surface of soil particles or mineral surfaces such as iron, aluminum oxides, cations, and carbonates. However, this adsorbed P can be made plant available through desorption (Ruttenburg, 2003). The amount of inorganic plant available P is highly dependent on the pH of the soil (Richardson, 1985). Nevertheless, all soils have an adsorption capacity resulting in P being lost from the system through leaching if the P saturation level is reached (Richardson, 1985).

P in Water can be categorized into either dissolved (soluble) ($<0.45\mu m$) or particulate ($>0.45\mu m$) P existing as both organic and inorganic compounds (Compton et al., 2000). Dissolved P can be further subdivided into dissolved reactive P (DRP) and dissolved non-reactive P (DNRP). DRP is available for the uptake by plants whereas, DNRP and particulate

P are not biologically available. Dissolved and particulate P is able to be carried in and out of the system through water through three main pathways. This includes surface runoff in addition to subsurface losses through tile drains and leaching to groundwater.

1.3 Phosphorus Transport in Agricultural Systems

1.3.1 Overview

In agricultural systems P is mainly lost from the field via surface runoff through soil erosion and subsurface pathways through tile drains (Sharpley et al., 1992). Traditionally, research has been focused on surface pathways due to tile drains historically not being thought of as a significant source of P. This was believed owing to the geochemistry of P and its tight bond to the soil. Nevertheless, as discussed below today tile drains and these subsurface transport pathways are now being seen as a significant source of P to freshwater bodies due to the high amount of runoff through tile drains during the winter and spring months (King et al., 2015). Recent studies have been completed demonstrating how both surface and subsurface pathways contribute to total P entering our freshwater bodies with Plach et al., (2019) finding that surface runoff was more limited by hydrologic transport, whereas tile drains were more limited by supply. The majority of annual runoff and P loss in both the surface and subsurface occur during the non-growing season (NGS, discussed in Section 1.4).

1.3.2 Phosphorus Loss and Runoff through Surface Pathways

Surface pathways of P runoff can either include the direct “wash-off” of fertilizer or the transportation of P through soil erosion. These “wash-off” events typically occur following snowmelt or heavy rain events in the winter and spring (Daniel et al., 1994), and are at greater risk when fertilizer is applied on frozen or snow covered ground during the winter months, or during flash flood events during the summer (Sharpley et al., 1994). Nevertheless, these events are preventable with proper year-round fertilizer management. Soil erosion, the other common surface pathway for P runoff is more prevalent during the spring months due to heavy rainfall events. Sharpley et al., 1994 found a direct correlation with rainfall intensity on bare ground and P loss in surface runoff. Therefore, management strategies such as cover crops to reduce the total amount of particulate P and help to stabilize the soil are seen as a BMP to reduce surface runoff.

1.3.3 Phosphorus Loss and Runoff through Subsurface Pathways

A common agricultural management practice is the use of tile drainage which refers to the removal of excess water through a subsurface drain. A report by Vander Veen (2011) stated that approximately 45% of farms in Southern Ontario have tile drains with an additional 30 million meters being installed every year. Tile drains are perforated pipes placed underneath poorly draining agricultural fields to lower the water table and improve drainage allowing for an extended growing season (Madramootoo et al., 1997). They have several agronomic benefits due to the removal of excess water resulting in increased soil aeration, porosity, and temperature (Hill, 1976). Drainage characteristically occurs episodically on an event basis in response to hydrologic events such as precipitation or snowmelt. As a result, there is a greater amount of tile discharge in the cooler months when greater amounts of precipitation and snowmelt occur (Lam et al., 2016). Nevertheless, there is still a lack of data with year round field monitoring to determine which management practices play the most significant role in mitigating P loss from tile drains (King et al., 2015).

Previous research has focused heavily on surface pathways as the main contributor of P concentrations in agricultural runoff. However, more recent research has highlighted the importance of P loss through tile drains and subsurface pathways (Osterholz et al., 2020; Van Esbroeck et al., 2016; Qi and Qi, 2017). P is transported to tile drains by matrix flow or preferential flow through macropores (Haygarth and Jarvis, 1999). Matrix flow occurs in the small pores within the soil matrix resulting in increased P retention through slow water transport and high contact between the soil matrix and water (Grant et al., 2018). Conversely, preferential flow is rapid flow through larger soil pores also known as macropores. These macropores can be created through physical processes such as drying and rewetting of the soil creating cracks/fissures or through biological activities such as root growth and earthworm borrowing. Macropores with a diameter of 0.3mm or greater promote non-equilibrium flow allowing solute transfer of relatively immobile solutes such as P with movement occurring in the direction of progressively larger macropores as the soil wets (Geohring et al., 2001; Jarvis 2007). Furthermore, Goehring et al., 2001 found that macropores with a diameter greater than 1mm had insignificant P sorption on the pore wall resulting in increased concentrations of P in tile drains. Over time P saturation can occur on the macropore wall due to limited matrix-macropore interaction causing a greater risk for subsurface P loss (Beven and Germann, 2013). Furthermore, it has been shown that fields with accumulated P in the surface layer (also known as legacy P) have increased concentrations of DRP in tile drains due to preferential flow through macropores (Baker et al., 2017). Therefore, land management practices that breakdown these macropores and redistribute nutrients throughout the top soil profile such as tillage and manure incorporation should prevent higher concentrations of dissolved P in tile drain runoff.

1.3.4 P Stratification in the Soil Profile

Stratification of P in the top layer of the soil can occur due to breakdown of crop residue in addition to when high concentrations of fertilizer are applied and plants are unable to take up all of the applied nutrients (Baker et al., 2017). If the soil column is not inverted as seen in no-till management systems, an accumulation of P in the top layer of soil will be observed resulting in P stratification (Cook and Trlica, 2016). This creates a high risk for surface runoff of P through wash-off from large rain events or snowmelt in addition to soil erosion (Haygarth et al., 2014). These high runoff periods can also be associated with an increase in preferential flow through tile drains causing increased concentrations of DRP within the tiles due to limited interaction between the soil particles and P (Van Esbroeck et al., 2016). Jarvie et al., (2017) found that conservation practices such as no-till with surface application of fertilizer and no incorporation showed an increase in labile P in the top fraction of the soil resulting in an increase in soluble P through both surface and subsurface pathways.

1.4 Phosphorus Management in Agriculture

In light of the fact that P can be lost in the surface and subsurface in response to the application of P on fields, managers have struggled to develop management activities that mitigate P losses in runoff and drainage. Best/better Management practices have been around for decades and refer to management practices that have been proven through scientific research and tested in the field for the best crop uptake and environmental sustainability (Roberts, 2007). Although a BMP ‘approach’ has been around for some time, the various recommended BMPs vary in their efficacy. Indeed, both no-till and riparian buffer strips are examples of BMPs that were recommended to reduce P loads to tributaries; however, it is now understood that their efficacy varies significantly in space and time (e.g. Kieta et al., 2018) and they may even lead to unintended consequences (e.g. Jarvie et al., 2017; Vanrobaeys et al., 2019). Thus, researchers are now focussing on more prescriptive recommendations for different regions or land uses. These recommendations may control P "transport" or "supply" or both. Efforts are being made to improve the efficiency of nutrient application strategies to mitigate P losses.

1.4.1 Summary of 4R Nutrient Stewardship

In order to address the mismanagement of fertilizer the 4R nutrient stewardship program was created worldwide to improve nutrient use efficiency (Johnston and Bruulsema 2014). This concept focuses on applying the Right source of nutrients, at the Right rate, at the Right time, and in the Right place resulting in improved environmental, economical, and social outcomes. However, the majority of research under the 4R nutrient stewardship has involved the use of synthetic fertilizers, and not a lot of consideration has been undertaken with the use of animal fertilizers. Currently the main source of controlling nutrient levels in manure is through feed management because treatment and transport of manure can be very costly (Ribaudo et al., 2003). Therefore, using different land management techniques under the 4R nutrient stewardship guidelines for both inorganic and organic sources of fertilizer to better control nutrient runoff could be more sustainable both environmentally and economically.

1.4.2 Right Source

It has been shown that the use of organic (manure) versus inorganic (synthetic/mineral) fertilizer can have an impact on P concentrations in both the surface and subsurface runoff. However, there are mixed findings on which source is more susceptible to increased P concentrations in runoff.

Several studies have examined the differences in DRP and total P (TP) in surface runoff following the application of organic compared to inorganic fertilizer. However, findings are mixed as to which source results in higher concentrations of DRP. For example, Nichols et al., (1994) using a treatment of poultry manure and inorganic fertilizer found that the inorganic fertilizer had statistically significant higher concentrations of TP in surface runoff compared to the manure amended soil. Yet of the TP, the inorganic fertilizer had a smaller proportion of DRP compared to the manure. Withers et al., (2001) found a similar result when comparing inorganic P fertilizer triplesuperphosphate with liquid cattle manure, and found that inorganic fertilizer had the highest TP in surface runoff compared to the cattle manure due to the P in the inorganic fertilizer having increased solubility in water. In contrast, Kleinman et al., (2002) compared the inorganic P fertilizer diammonium phosphate with different types of manure including dairy, poultry, and swine with results showing no significant difference in TP. Nevertheless, there was a very strong linear relationship with the amount of water-soluble P and the amount of DRP found in the runoff depending on the manure source. This is further supported in a review completed by Hart et al., (2004) where it was concluded that differences in the concentration of TP found in surface runoff were correlated with the amount of water soluble P present. Results of these studies indicate that inorganic sources of fertilizer are slightly more susceptible to discharging higher concentrations of TP in surface runoff with concentration of DRP being highly correlated to the amount of water soluble P. This demonstrates the need for more research needing to be completed on inorganic and organic sources of P on their respective solubility and soil chemistry to categorize their risk to runoff.

In contrast to surface runoff, the majority of studies have shown that organic sources are more susceptible to producing higher concentrations of P in subsurface runoff (King et al., 2015b). This includes a study by Nayak et al., (2009) which found that when applying swine manure at nitrogen based rates compared to inorganic fertilizer, there were statistically significant higher concentrations of DRP in tile runoff. Additionally, McDowell et al., (2005) found that when comparing soils with long term dairy manure application against soils amended with similar rates of superphosphate, manure amended soils had higher concentrations of DRP further down the soil profile resulting in 3.7 times greater DRP concentrations found in tile drain runoff. This is further supported by Eghball et al., (1996), where they compared manure versus inorganic fertilizer plots from 1953 to 1993 and found that available P moved further down the soil profile with manure compared to those with inorganic fertilizer. This supports the finding of Macrae et al., (2007) that soluble reactive P, TP, and DRP in subsurface drainage were higher with fields receiving exclusively manure compared to inorganic fertilizers. Nevertheless, there are studies that contrast this result including a study completed by Carefoot and Whalen (2003) that compared cattle manure against the inorganic fertilizer triple superphosphate and found that subsurface water collected from piezometers did not differ by source and that particulate P was the most dominant form. However, total and particulate P concentrations in subsurface runoff were strongly correlated with the Mehlich-3 soil P concentration. Furthermore, McDowell and Sharpley (2004) found that at both low and high flow conditions dairy manure and the inorganic fertilizer mineral superphosphate showed no significant differences in the amount of P leached with organic applied plot leachate concentrations able to be estimated by the initial Mehlich-3 P soil concentration. Although there are mixed findings on whether organic or inorganic sources are more susceptible to subsurface runoff, several studies have indicated that source chemical characteristics determine the extent of P losses to subsurface pathways (King et al., 2015b). In the McDowell et al., (2005) study mentioned above, they found that manure amended soils contained higher levels of organic P as phytate, the major P form in plants, which competed with P sorption sites. Therefore, the organic sources of P sorb less strongly compared to the inorganic resulting in increased leaching. In a study completed by Kinley et al., (2007), they found that organic sources had higher levels of soluble reactive P resulting in consistently higher levels of soil test P and TP in drainage waters with poultry and swine manure having the highest concentrations of P runoff. Furthermore, it is also suggested that the amount of water-extractable P increases subsurface P. Sharpley and Moyer (2000) found that when comparing different manure sources the total P leached was strongly correlated to the amount of water-extractable P, suggesting that it is a good predictor estimating the potential of P leachate and runoff. Nevertheless, although it has been established that water-extractable P is correlated with P concentration in runoff, Kleinman et al., 2002 demonstrated the experimental variability in measuring water-extractable P in different manure sources and suggest that one standardized method should be used to reduce error and improve P runoff predictability. Therefore, source characteristics such as P sorption, soluble reactive P, and water-extractable P are all good indicators of potential P leaching from soils and suggest why organic sources are more susceptible to subsurface runoff. Regardless of the contradicting literature, there is a consensus that initial soil characteristics will contribute to the amount of subsurface P leached. Consequently, more research is needed to be done to understand the soil and source chemical characteristics and their interactions to better understand P leaching potential to subsurface pathways.

Due to the differences in TP runoff risk associated with different fertilizers it is important to consider chemical characteristics of all fertilizer applied. Synthetic fertilizer can be customized to specific nutrient ratios depending on the soil conditions and crop rotation. However, organic fertilizer sources are much more difficult to manage. Differences in organic sources of fertilizer are highly species specific, and even within a species can vary depending on the operation. The nitrogen to P ratios varies greatly by species with dairy cattle having 7:1 followed by poultry at 6:1 then swine at 4:1 (Eghball 2002). Therefore, operations that apply swine manure long term will be at the highest risk for P accumulation and runoff because manure application rates will need to be doubled in order to meet the crop nitrogen requirement of 8:1, resulting in the over application of P. Kleinman and Sharpley (2003) compared dairy (*Bos Taurus*), poultry (*Gallus gallus*), and swine (*Sus scrofa*) manure and found that differences in the water extractable P between the sources correlated between the differences in dissolved reactive P concentrations found in runoff. Dairy manure had the lowest concentration of dissolved reactive P in runoff (0.4-2.2mg L^{-1}) when compared to poultry (0.3-32.5mg L^{-1}) and swine (0.3-22.7mg L^{-1}). Therefore, consideration on the type of organic source used is important due to differences is their nutrient composition resulting in some being more susceptible to P runoff than others.

As discussed above, the source of fertilizer can strongly influence the susceptibility of TP concentrations in runoff. The literature suggests that inorganic sources are more susceptible to surface runoff whereas organic sources are more susceptible to subsurface runoff through tile drains. Nevertheless, many studies have demonstrated exceptions to these general trends and emphasize the importance of further research needing to be completed to better understand source chemistry and how this interacts with the soil.

1.4.3 Right Time

The right time refers to when fertilizer is applied relative to planting and onset of precipitation. The timing of fertilizer applications can depend on a number of factors such as weather conditions, crop rotations, and manure storage capacity. Fertilizer applications typically occur in the spring and fall. For synthetic fertilizer, approximately 95% of fertilizer is applied in the spring with approximately 3% being applied in fall and 2% being applied in summer (Korol 2004). However, the type and timing of application is highly dependent on the crop. For example, in Southern Ontario spring application of an N-P-K fertilizer is common with the planting of corn and alfalfa, whereas wheat is applied in the fall during planting and contains a higher concentration of P to stimulate root growth. It is also common to see N broadcast in the spring after planting of both corn and wheat. For manure application in Canada, the proportion applied throughout the year is 33.2, 25.9, 35.4, and 5.5% for spring, summer, fall, and winter respectively (Beaulieu 2004). Therefore, it is important to consider the risks of P runoff relative to the time applied seasonally in addition to the time between application and precipitation.

Using the P-risk index there is greater risk associated with fall and winter applications compared to the spring and summer due to the lack of plant uptake in addition to increased runoff including snow melt (Sharpley et al., 2003). Due to fall fertilizer applications typically being manure applications, the majority of fall/winter runoff studies examine the effects of manure application over inorganic sources. In a study by Van Es et al., (2004) they observed greater amounts of runoff, TP, and DRP in the fall applications of liquid dairy manure compared to the spring. This is also true regarding subsurface runoff where tile drains have shown to have a greater amount of discharge in cooler months due to the accumulation of precipitation and freeze-thaw cycles over the non-growing season (Macrae et al., 2007). This is further supported by Tan et al., (2002) who found that over half of the tile drainage produced in a year happened in the non-growing season (NGS). Nevertheless, some studies have found opposite results with greater P concentrations in the spring and summer relative to the fall and winter. For example, King et al., (2015a) looked at discharge and P concentrations from tile drains from the Upper Big Walnut Creek watershed in Ohio and found that although discharge was greatest in the fall, winter, and spring, P concentrations were greatest in the summer most likely due to surface runoff. Nevertheless, King et al., (2015a) conclude that the majority of P loading occurs in the fall and spring, and therefore it is at this time that best management practices should be undertaken to reduce P loading to water bodies. Additionally, there are also mixed findings as to what the dominant form of P is at different seasons. Chapman et al., (2001) found that particulate P concentrations were greater in the winter compared to Macrae et al., (2007) where they found that DRP concentrations were greater in the winter. Due to the impacts freeze-thaw cycles (FTC) and snow melt have on P loss, the NGS still remains an understudied time period (King et al., 2015b).

FTCs can result in the solubilisation of P due to the death of the microbial community; however, the extent of microbial biomass affected is dependent on a number of factors including the community composition, soil type, and the degree of freezing and thawing, as microbes have shown to be able to survive and actively decompose organic matter up to -5ºC (Blackwell et al., 2010; Calcott, 1978). Additionally, FTCs can affect soil aggregate stability, with soils that retain higher moisture contents such as clay being more susceptible to destabilization of soil aggregates resulting in a greater risk of surface runoff (Oztas and Fayertorbay 2003). Messiga et al., (2010) found that water extractable P and Mehlich-3 P in the soil increased with the number of FTCs under controlled conditions. Specific to manure applications, Bechmann et al., (2005) looked at P runoff and leaching with soil columns comparing bare soil, mixed dairy manure, and the cover crop ryegrass (*Lolium multiflorum* L.) under repeated FTCs. Results indicated that manure did not have a significant impact on increasing P runoff. Nevertheless, very few studies examining the effects of freeze-thaw on manure have been completed. Therefore, more studies in this area need to be completed in order to determine if the addition of manure has a significant impact on P concentration in runoff if repeated FTCs occur. Overall, manure application in the fall and winter tend to be a higher risk for P runoff however there are still uncertainties on the dominant form of P in runoff depending on season in addition to the impact of manure over the NGS.

There is also a greater risk of P runoff when the timing of fertilizer application is followed immediately by a precipitation event (Sharpley et al., 2003). Smith et al., (2007) found that when comparing inorganic fertilizer, swine manure, and poultry litter following rain events ranging from 1 to 29 days, swine manure one day after manure application had the greatest TP concentration. Nevertheless, as the amount of time between application and precipitation increased the risk of P runoff decreased. This is supported by a study completed by Allen and Mallarino (2008) where they looked at surface runoff following simulated rainfall events at 1, 10, and 16 days after liquid swine manure application and found that dissolved P significantly decreased with increased time between application and precipitation. Therefore, increasing the time between the application of manure and the onset of precipitation will significantly reduce the risk of P runoff.

Overall, the majority of fertilizer in Canada is applied in the spring and fall. However, there are greater risks associated with fall and winter fertilizer applications compared to the spring. Additionally, as the time between application and precipitation increases the risk of P runoff decreases.

1.4.4 Right Place

The correct placement of fertilizer can improve the crop response, decrease the application rate, and reduce the amount of surface and subsurface nutrient runoff (Johnston and Bruulsema 2014). Proper management including fertilizer application methods and tillage practices can greatly impact transport pathways resulting in change in P concentrations in runoff.

Fertilizer application can vary greatly depending on the crop and operation. Inorganic fertilizer can be applied in multiple ways with the most common being broadcast or subsurface banding. Broadcast application refers to when the fertilizer is applied across the surface of the whole field. In contrast, subsurface banding refers to when fertilizer is placed within the soil next to the seed to optimally supply the plant with nutrients. It has been well established that subsurface banding greatly reduces P concentrations in surface and subsurface runoff (Grant et al., 2019; Lamba et al., 2013; Watts et al., 2011). Watts et al., (2011) found that surface losses from broadcast synthetic fertilizer had the greatest concentration of P with subsurface banding reducing DRP and TP by 86%. Additionally, Grant et al., (2019) found with column experiments that subsurface banding reduced DRP concentrations by 60% in leachate relative to surface broadcasting of inorganic fertilizer. These results have been consistent with in-lab and field-based studies, and as a result subsurface banding of fertilizer is a highly recommended BMP to reduce P concentrations in both surface and subsurface runoff.

Manure/organic fertilizer can be applied to the field in a variety of ways including solid/liquid spreaders, liquid injectors, and irrigation systems. It is recommended by BMPs that manure should be incorporated within 24 hours to minimize environmental risk and improve nutrient use efficiency (Beaulieu 2004). It has been shown in multiple studies that the incorporation of manure can greatly reduce P runoff (Feyereisen et al., 2010; Kleinman et al., 2008; Osei et al., 2003). In a study by Osei et al., (2003) it was found through modeling that incorporation following dairy manure application decreased P edge-of-field runoff by up to 45% depending on the application rate. Additionally, due to the destruction of preferential pathways following incorporation, a study by Kleinman et al., (2008) showed that sub-surface P loss was significantly reduced following incorporation. To further support that P travels through preferential pathways a study by Feyereisen et al., (2010) found that direct injection of poultry manure had the highest subsurface P runoff followed by surface broadcast and that manure incorporated with disking was found to have the lowest P subsurface runoff. However, as stated by Beaulieu (2004) 52.4% of farms producing livestock in Canada only surface applied their manure or incorporated it more than a week after it was applied, with only 15% of producers injecting or incorporating their manure the same day it was applied. It is thought to be that the most common reason for not incorporating manure would be a lack of time, especially in the fall when harvesting is a priority over the spring when farmers are getting their fields ready for planting. In a review completed by King et al., (2015b) it was noted that the majority of studies find that differences in P runoff concentrations between incorporated and surface applied fertilizer are greatest following the first precipitation events with the difference decreasing over time. Additionally, most of these studies are completed in the lab through leaching experiments or from spring to fall without considering the impacts of winter. Therefore, there is a gap in knowledge in whether incorporating manure in the fall truly makes a significant impact in decreasing subsurface P runoff.

In North America there are three common tillage practices including conventional tillage, no-till, and reduced tillage. Conventional tillage refers to the complete workup of the topsoil by inverting the soil and removing crop residue through ploughing, followed by a secondary tillage to level and prepare the seed bed. This type of land management system destroys soil structure, reduces soil organic matter, and makes the soil more susceptible to erosion (Lam et al., 2016). In contrast, no-till systems are not ploughed and the soil structure and organic matter is preserved. However, no-till fields have shown to have increased preferential flow and P stratification (Bertol et al., 2007). Finally, in reduced tillage practices the soil usually receives less intensive secondary tillage such as disking and is never completely inverted like conventional tillage. A common practice of reduced tillage is a rotational tillage system where the soil is worked on average every three years (Ulen et al., 2010). Typically, these three tillage systems are classified by conventional tillage leaving less than 15% of crop residue, reduced tillage leaving 15-30% crop residue, and no-till leaving greater than 30% crop residue (West and Marland 2002). The popularity of each tillage practice differs from area to area, and as shown from the 2016 Statistics Canada census data in provinces like Ontario and Manitoba the total number of no-till acres has decreased from 2011, whereas the total number of reduced till and conventional till has increased. In contrast, Saskatchewan has shown the opposite trend, with the number of no-till and reduced till acres increasing from 2011 and the number of conventional till acres decreasing. Nevertheless, the land management practice of no-till/reduced-till on increasing subsurface P flow in tile drains has had very mixed results in the literature depending on the location and soil characteristics (King et al., 2015b). For example in studies both completed in Southwestern Ontario, Lam et al., (2016) found that TP and DRP loads in tiles were not affected by reduced tillage versus annual disk till in a sandy loam soil. Whereas in finer textured soils it was found that total soluble P and total P increased using no-till versus conventional tillage (Gaynor and Findlay, 1995; Williams et al., 2018). Due to these mixed findings, more studies should be completed using no-till, reduced till, and conventional till treatments on a range of soil textures to determine their susceptibility to surface runoff and preferential flow through macropores.

As discussed above, subsurface banding of inorganic fertilizer and incorporation of organic fertilizer can greatly reduce DRP and TP concentrations in surface and subsurface runoff. This is done by reducing the risk of manure to be “washed off” the surface in addition to breaking up preferential pathways in the subsurface. However, it is important to consider how the soil was worked prior to manure application and the impact of incorporation on surface runoff as results have varied depending on soil texture.

1.4.5 Right Rate

Using the right rate of fertilizer is a common management practice to reduce P concentrations in runoff. Nevertheless, some farmers see over-applying fertilizer as an inexpensive “insurance” to make sure crops have all of the required nutrients for optimal yields (Santos, 2011). Additionally, due to some manure application rates being determined on crop nitrogen requirements, livestock rich areas with long term manure amendments have shown to have P accumulation in soils (Whalen and Chang 2001). Therefore, applying the right rate of fertilizer is very important in order to prevent a buildup of P in the soil in addition to reducing the potential for P runoff. Several studies have demonstrated that application rate has a significant impact on P concentrations in surface and subsurface runoff (Hart et al., 2004; King et al., 2015b). Whalen and Chang (2001) found that under cultivated soils of barley (*Hordeum vulgare* L.), irrigated plots with a cattle manure application rate of >60Mg Ha^{-1} had as much as 1.4Mg of P per hectare per year lost most likely through leaching over the 16 year period of the study. McDowell and Sharpley (2000) demonstrated the significance of manure application on soils that are intensively farmed with high levels of soil P, and found that the addition of manure moved P further down the soil profile making it more susceptible to P leaching despite the planting of fast growing grass immediately following manure application. Therefore, soils that already have a buildup of P in the soil are more susceptible to runoff. However, even in soil with low soil test P initially are at risk to P runoff if the incorrect rate is applied. Allen and Mallarino (2008) found that increasing manure rates increased P concentrations in surface runoff linearly. This is further supported by Ball-Coelho et al., (2007) which found that following the application of liquid swine manure onto corn (*Zea mays* L.) with rates varying from 0 to 94 m^3 per hectare that the application rate was an essential driver of preferential flow resulting in greater concentrations of DRP. Therefore, a higher application rate of manure increases P runoff in the surface and subsurface. Overall, all the literature concludes that a higher P application rate leads to higher P runoff. Nevertheless, because the majority of manure application rates are based on Nitrogen, P is often applied in excess. As a result, fields that already have an accumulation of soil P should alter P application rates accordingly in addition to applying manure to nitrogen fixing crops due to their decreased risk of P runoff and further buildup of soil P.

1.5 Thesis Rationale and Objectives

Understanding the impacts of manure application, particularly during the high risk runoff period of the NGS is important in order to create more robust evidence-based management solutions. Currently the existing literature has found mixed findings on whether tillage impacts P concentrations and loads in tile drains. Of these studies, very few use year round in-field monitoring and even fewer have been completed using manure as the fertilizer source. Although the BMP of incorporation has strong evidence to support the effectiveness at reducing P, very few studies have looked at the timing and placement of fall manure application and the long term impacts. Therefore, due to these existing gaps within the literature, this thesis aims to provide evidence to farmers and policy makers on which BMPs will have the most significant impact on reducing P loads to our freshwater bodies. The goal of this study was to determine if tillage and incorporation impacted subsurface P loads and concentrations from fall applied dairy manure fields over the NGS. Because tillage is known to break up subsurface preferential pathways it was predicted that plots with a deep conventional tillage would have lower P loads and concentrations compared to plots under conservation tillage management. Furthermore, since incorporation helps to combine organic P within the soil profile, it was predicted that plots receiving manure incorporation would have reduced P loads and concentrations compared to plots that received surface applied manure.

The specific objectives of this thesis are to:

1) Quantify SRP and TP losses from tile drains over the NGS following fall manure application under the treatment of deep till (DT), conservation till with incorporation (CT-I), and conservation till (CT).

2) Determine if conventional tillage versus conservation tillage impacts subsurface P loads and concentrations over the NGS following fall applied manure.
3) Determine if incorporation following fall applied manure impacts subsurface P loads, concentrations or speciation over the NGS.

Chapter 2
Materials and Methods

2.1 Site

Research for this study was conducted on a 35Ha field with 3 adjacent plots in St Marys Ontario (Figure 2.1). St Marys is located in Perth South Township in Perth County and is part of the subwatershed in the Upper Thames River watershed. The site is located on imperfectly drained Perth silt loam soil that exhibit Grey-Brown Podzolic characteristics, and is managed with a corn-soybean-wheat crop rotation (Hoffman and Richards 1952). From the Environment Canada 30-year climate data, the annual precipitation for Perth County is approximately 1,069mm with snowfall making up 20% of the total annual precipitation. Average daily temperature in January is -6.0ºC compared to 20.2ºC in July supporting that this region experiences the typical 4 season climate common in Southern Ontario with warm summers, and cold snowy winters prone to freeze-thaw cycles (FTC) (Environment Canada 2020).

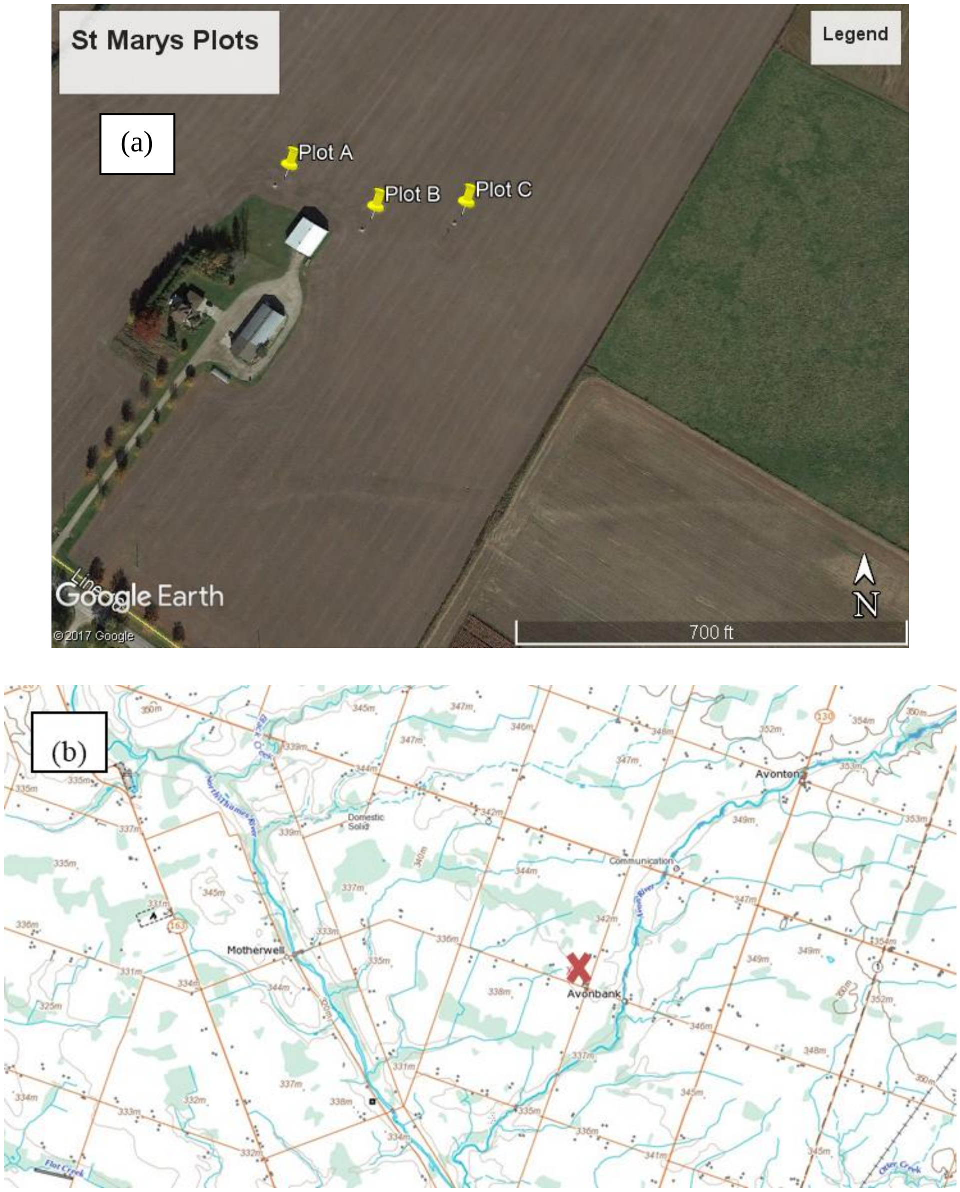

Figure 2.1. Google Earth image of the St Marys field site indicating each plot's tile sampling location (a). Canadian Ministry of Natural Resources topographic map of the region with x marking the approximate location of the field site (b).

Three adjacent 24.4m wide plots were situated within a field centered along a single tile line. Tiles within the field were spaced 12.2m apart and were therefore assumed that 50% of the area on either side of the tile was contributing to that tile line (Figure 2.2). The site was established in 2010 with all plots initially receiving annual minimal disc tillage to a depth of approximately 8cm (table 2.1). Tillage treatments began in fall 2011 with a deep disc till (DT) plot to simulate conventional tillage methods and two conservation tillage (CT) plots that received shallower and less frequent disc tillage. This tillage treatment breakdown remained consistent throughout the study period with the exception of fall 2017 where one of the CT plots received incorporation (CT-I) of 5cm disc till following the fall manure application.

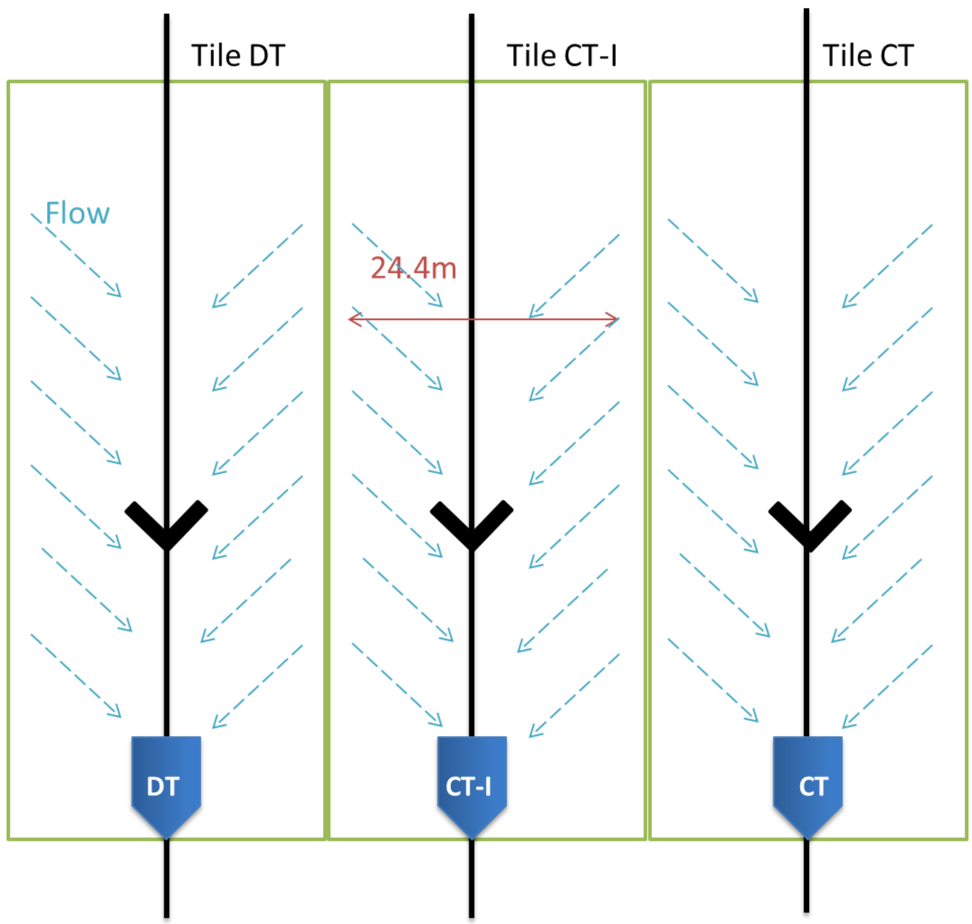

Figure 2.2. Simplified schematic of St Marys field set up with blue pentagons indicating sampling location.

Table 2.1 Agronomic history of the St. Marys study site between 2010 and 2018

Year	**Parameter**	**Conventional Plot (DT)**	**Conservation Plots (CT-I & CT)**
2010	*Crop*	Winter Wheat	
	Fertilizer	April: 100kg/Ha of N	
		Broadcast	
	Tillage	Minimum Tillage	
		Disked to 8cm following mid-September	
2011	*Crop*	Soybean	
	Fertilizer	None	
	Tillage	Chisel Plow to approx. 30cm	No Till
2012	*Crop*	Corn	
	Fertilizer	May: 45kg/Ha of N, 16kg/Ha of P	
		Mineral Banded with seed	
		June: 90kg/Ha of N	
		Banded between rows	
		November: 114-33-111 N-P-K	
		Liquid dairy manure with dragline	
	Tillage	Disk till to 8cm	No Till
2013	*Crop*	Soybean	
	Fertilizer	September: 5kg/Ha of N, 24kg/Ha of P	
		Banded with seed	
		September: 129kg/Ha of N	
		Broadcast	
	Tillage	Disk Till to 8cm	No Till
2014	*Crop*	Wheat	
	Fertilizer	October: 60-25-100 N-P-K	
		Liquid dairy manure with dragline	
		Disk till to 8cm	
	Tillage	Chisel plow to 18cm	
2015	*Crop*	Soybean	
	Fertilizer	November: 120kg/Ha of K_2O, 27kg/Ha of S	
		Broadcast	
	Tillage	Disk till to 8cm	Shallow disk till

		Chisel plow to 30cm	
2016	*Crop*	Soybean	
	Fertilizer	May: 85kg/Ha of K Broadcast October: 6Kg/Ha of N, 30kg/Ha of P Banded with seed	
	Tillage	Disk Till to 10cm	No Till
2017	*Crop*	Wheat	
	Fertilizer	September: 17-19-84 N-P-K Dairy manure with dragline	
	Tillage	Disk till to 10cm	Disk till to 6cm Additional Disk till of 6cm to CT-I plot following manure application
2018	*Crop*	Corn	
	Fertilizer	May: 13kg/Ha of N, 45 kg/H of P, 30Kg/Ha of K Banded with seed May: 170kg/Ha of N Broadcast	
	Tillage	Disk till to 10cm	No Till

2.2 Field Methods

To directly capture edge-of-field losses from individual plots, tile lines were intercepted below ground at the end of the plot. A 10° v-notch weir box (50cm x 50 cm in area, with a 1.5 m riser that permitted instruments to access tile discharge) was installed within the 6” tile laterals to allow water to move freely within the tile (Figure 2.3). Water depths were recorded for the duration of the 8-year study period at 15-minute intervals using pressure transducers in each weir box ((Onset HOBO-U20-001-04), +/- 0.3cm accuracy) and corrected with barometric pressure. Rating curves for weirs were calibrated using Hach FL900 loggers equipped with depth-velocity sensors for a three-month period for each tile line. Using an automated water sampler (6710, Teledyne ISCO) discrete water samples were taken on an event basis at 2-8hr intervals (adjusted seasonally) to catch the entire hydrograph within a hydrologic event. Hydrologic events were determined using precipitation and temperature data in addition to in-field observations. Samples were collected within 1-4 days of collection (or earlier depending on the length of the hydrologic event) and were taken directly to the lab to be processed. To protect the field equipment from weather and freezing, the flow and water quality monitoring equipment was kept within an enclosed structure.

Figure 2.3. Schematic drawing depicting 10° v-notch weir setup within tile line and photographs of the structure.

Soil samples were obtained throughout the 8-year study period, including the fall of 2011, 2014, and 2017 in addition to the spring of 2018. Soil cores were taken at a depth of 1-2.5, 2.5-5, 5-7.5, and 7.5-15 cm across all three plots and brought back to the lab and oven dried at $30^{\circ}C$ for the determination of soil test P (Olsen).

On site meteorological conditions were monitored at 30min intervals using a Campbell Scientific CR-1000 data logger. Air temperature, relative humidity (Vaisala HMP45C), net radiation (Kipp & Zonen NR-LITE), wind speed and direction (R.M. Young 05103), and rain (Texas Electronics TE525M) were all recorded. Snow depth measurements were taken periodically throughout the 8-year study period across all plots.

2.3 Laboratory Methods

Water samples were immediately processed for P analysis. For soluble reactive P (SRP) a subsample of 50mL was filtered through a 0.45µm cellulose acetate filter under a vacuum that was then refrigerated up to a week before analysis. To determine total P (TP) a second subsample of 50mL was acidified using 0.2% H_2SO_4 for storing. To prepare the sample for analysis, an acid digestion using the persulfate digestion method B protocol from Kovar and Pierzynski (2009) was used. To quantify SRP and TP an ammonium molybdate/ascorbic acid colormetric method was used (Bran-Luebbe AutoAnalyzer lll system, Seal Analytical, Methods G-103-93, detection limit 0.001mg/L P). Similarly for TDP the filtered samples were digested by UV and acid persulfate using an inline UV-digester, and subsequently analyzed colourmetrically (Bran-Luebbe AutoAnalyzer lll system, Seal Analytical, Methods G-092-93, detection limit 0.001mg/L P). Duplicates were randomly selected at a rate of 5% of samples per run to monitor any technical errors that may occur during runs.

Soils were analyzed for water-extractable (WEP) and Olsen P. Dried soils were sieved to a particle size of 2mm. Water-extractable soil P was determined using the Kovar and Pierzynski (2009) protocol following the deionized water extraction procedure with a centrifuge at 5,000rpm for 5 minutes and 0.45µm Whatman filter papers. A molybdate/ascorbic acid colormetric method analysis with the Bran-Luebbe AutoAnalyzer lll system, Seal Analytical was used to analyze WEP. Olsen P was determined using the Amacher et al., (2003) protocol and analysed with the molybdate/ascorbic acid colormetric method using a spectrometer at 880nm.

2.4 Flow Correction for Backpressure

In the event that back pressure occurred within the tiles the flow data was corrected. A manual method identifying and correcting events where backpressure occurred was used. This approach was compared to alternative methods including one used by Martin (2015) (Appendix A). To identify potential hydrologic events experiencing backpressure, any tiles where the event reached the maximum flow rate were examined. In this study, observations of sensor depth data were used to identify inflection points for where backpressure was estimated to start and end for each individual hydrologic event (apparent from distinct rapid increases in water levels within the weirs in which water levels stagnated or plateaued in the weirs). During this estimated period of backpressure, flow was set to 0L/s. Flow resumed when the 'plateau' ceased and water levels began to drop, indicating that drainage water was moving.

2.5 Data and Statistical Analysis

To delineate flow in individual "events", tile drain baseflow was separated from total tile drain flow using a hydrograph slope with the R software package EcoHydRology (Nathan and McMahon 1990; Fuka et al., 2014). Events were deemed to have commenced when tile flow rose above baseflow and ended when flow returned to baseflow or ceased altogether. The separation of baseflow was only done to assist in the identification of events. In the quantification of event flow or P loads, total tile flow (including baseflow) for a given event was determined. Flow-weighted mean concentrations (FWMC) of SRP and TP were calculated for individual events over the study period (Williams et al., 2015). Flow-weighted means for events that were missed by samplers were estimated using linear interpolation. Event P loads (mg or kg) were calculated by multiplying event tile discharge (L) by event FWMCs (mg/L) which were then normalized using estimated contributing areas for each tile (Ha) to generate a load estimate in kg/ha. Normality of the data was checked using quantile-quantile plots. To statistically analyze discharge the data was log transformed in order to meet the assumption of normality. Discharge of different tillage treatments was plotted with regression analysis to determine treatment effects (Chatterjee and Hadi 2006). Log transformed discharge data was then plotted against log transformed event P loads and analyzed with linear regressions to determine if the system was chemostatic or chemodynamic driven (Basu et al., 2010). Autocorrelation was completed to check if correlation existed within the lagged residuals (Helsel and Hirsch 2002). Treatment comparisons including P FWMCs, loads, and snow surveys were completed using an unpaired Wilcoxon test or one-way analysis of variance depending if data met the assumption of normality (Gehan 1965; Chatterjee and Hadi 2006). Statistical tests were deemed to be significant with a probability level of 0.05.

Chapter 3
Results

3.1 Meteorological and Environmental Conditions over the Eight-Year Study Period

Day-to-day weather and seasonal patterns varied over the 8-year study period, but generally fell within what was normal for the region based on 30-year long-term data (Environment Canada, 2020, Figure 3.1). Precipitation over the 8-year study period fell mainly in the NGS with 67 +/- 5% of annual precipitation occurring between September and April in any given year. Given the dominance of the NGS in annual hydrological budgets, the NGS periods are focused on in this chapter.

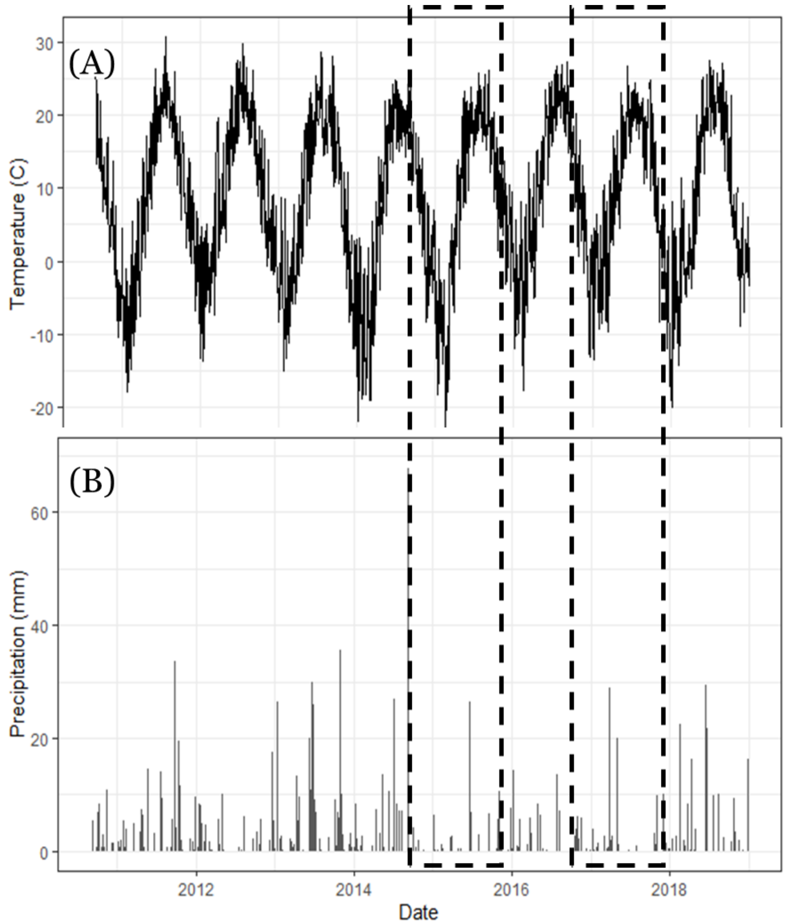

Figure 3.1 (A) Mean daily temperature and (B) Daily total precipitation (rain and snow) of the St Mary's site over the study period (2011-18). Dashed boxes highlight the 2014-15 and 2017-18 NGS focused on in this study.

Some notable seasonal differences were observed among years. For example, of the two years that were selected for intensive study, the 2014-15 water year was considerably drier than normal, with the exception of September and June (Figure 3.2(A)). The 2017-18 water year more closely resembled a typical year of precipitation compared to the 30-year average with the exception of September which was noticeably drier (Figure 3.2(C)).

Air temperature varied seasonally over the study period with the minima and maxima observed in the winter and summer respectively. The 2014-15 water year experienced extremely cold temperatures in the winter months compared to the 30-year average, with February's minima being significantly cooler (Figure 3.2(B)). Aside from this difference, the spring, summer, and fall closely resembled temperatures to that of the 30-year average. The 2017-18 water year was generally warmer in the fall and summer months and cooler in the winter and spring months compared to the 30-year average with the exception of February being slightly warmer (Figure 3.2(D)).

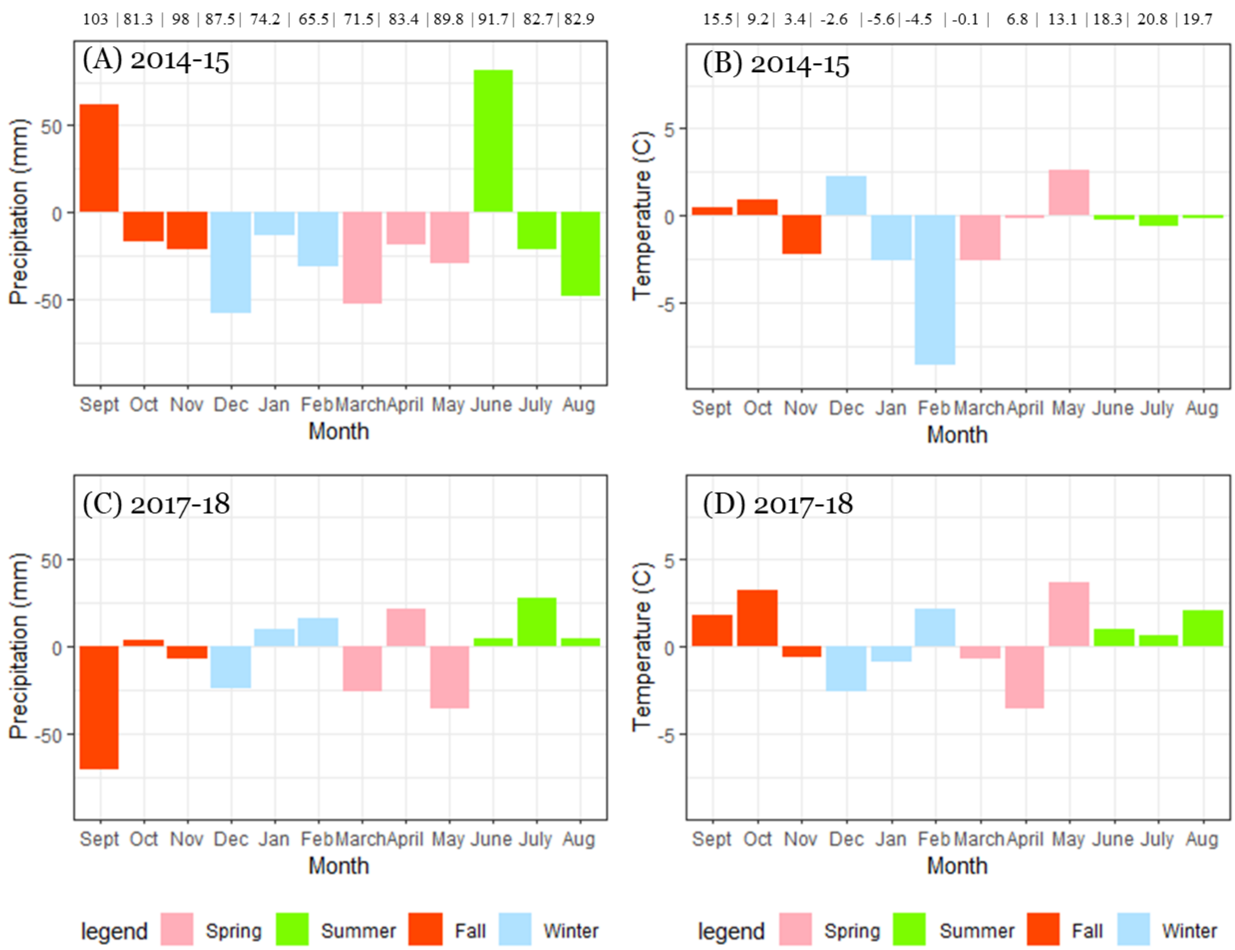

Figure 3.2 Monthly precipitation (A and C) and temperature (B and D) anomalies for 2014-15 and 2017-18 from the 30-year environment Canada climate average (top bar). Colours indicate seasons. The numbers on top of the figures indicate the 30-year normal precipitation (mm) for a given month.

Each NGS experienced a different number of FTCs including both melt and rain on snow events when the temperature exceeded 0ºC. For the 2014-15 NGS, FTCs were characterized as quick and brief with the temperature dropping back below 0ºC less than two days. In contrast, the 2017-18 NGS FTCs were characterized as long and slow with the temperature not dropping back below 0ºC for up to seven days.

Annual and seasonal soil temperatures within the soil profile closely resembled the recorded air temperature. The highest amount of variance was observed in the top 10cm of the soil profile. Nevertheless, temperatures remained above 0ºC with the exception of brief periods during the winter of 2015 (Figure 3.3) during which exceptionally cold air temperatures resulted in freezing within the top 10cm of the soil profile (Figure 3.3).

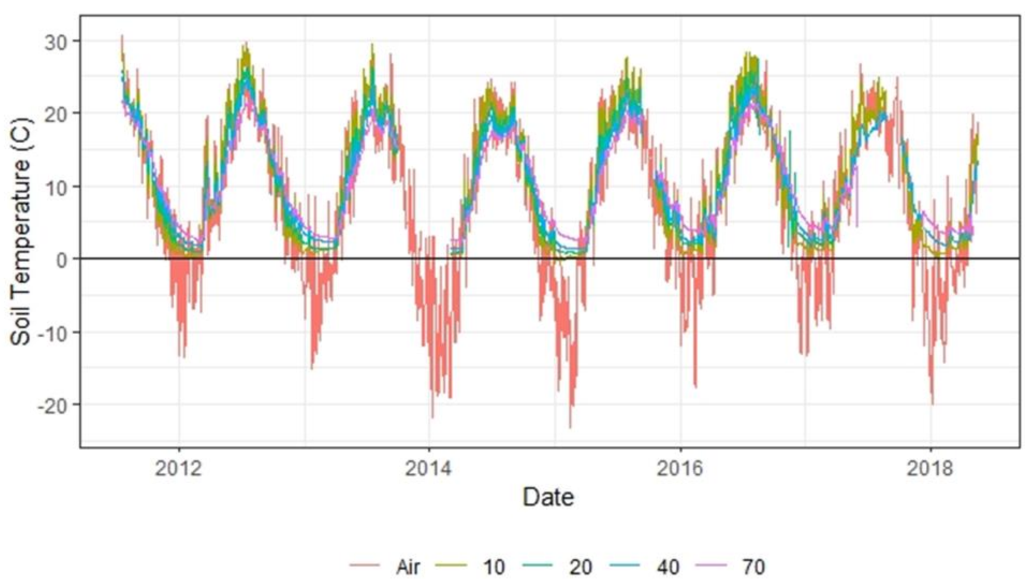

Figure 3.3 Soil Temperature of the DT plot at the depths of 10, 20, 40, and 70 cm for the entire study period (2011-2018).

Given that the study plots were located within the same field, rainfall amounts did not differ between them. A detailed snow survey in 2014 showed no statistically significant difference among plots (Figure 3.4; p-value > 0.05). Periodic, less detailed snow surveys done over the study period also did not demonstrate any differences in snow accumulation across the plots.

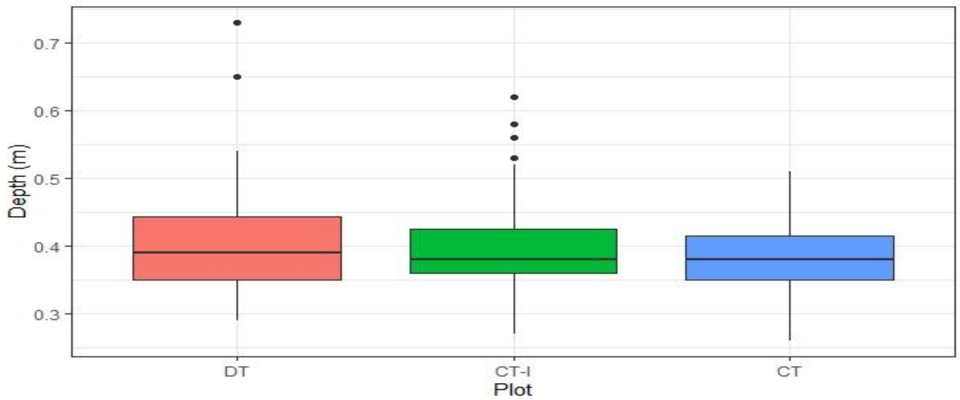

Figure 3.4 In field snow depth measurements from 2014 winter across the DT (n=40), CT-I (n=40), and CT (n=43) plots. Boxplot illustrates the median (solid horizontal line), the interquartile range of the 25th to 75th percentile values (box outline), values within 1.5 times the interquartile range (line whiskers), and outlier values that are 1.5 times greater than the interquartile range (circles)

Although precipitation did not differ among plots, soil moisture showed a variety of trends depending on depth and treatment. Previous to any tillage treatments, the CT-I plot exhibited higher levels of soil moisture at the surface (10cm) of the soil profile. However, following tillage treatment this trend is reversed with DT having consistently higher levels of soil moisture at the surface (Figure 3.5). Regardless of tillage treatment, CT-I exhibited greater levels of soil moisture compared to DT deep in the subsurface (Figure 3.5).

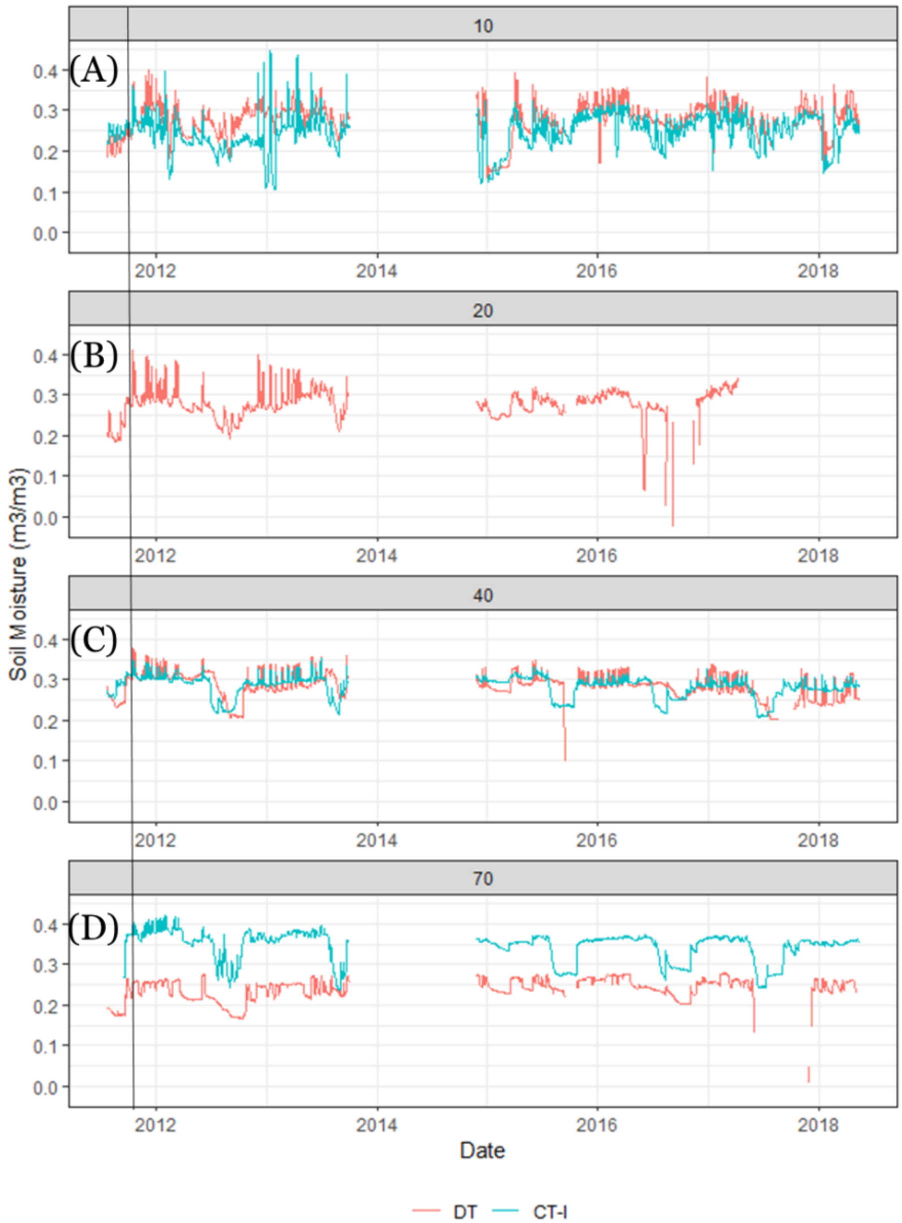

Figure 3.5 Soil Moisture of the DT and CT-I plots at the depths of 10 (A), 20 (B), 40 (C), and 70 (D) cm. Vertical line indicates the start of tillage treatments.

3.2 Annual and Seasonal Variability in Tile Drainage over the Eight-Year Study

Although annual precipitation inputs were similar across study years, runoff output from the tiles exhibited variability across study years (Figure 3.6). Runoff ratios varied across study years, seasons, and tiles (Table 3.1). The majority of annual runoff from the tiles followed discrete hydrological events. However, responses varied depending on individual events and seasonality. Some event runoff ratios for the two intensive study years were greater than 1, indicating the presence of backpressure. Flow data for the 2014-15 and 2017-18 NGS was corrected accordingly with corrected data being used for the remainder of this thesis (Appendix B).

Figure 3.6 Hydrograph of the DT, CT-I, and CT tiles across all study years. Dashed box highlights the NGS in addition to brackets highlighting the two NGS with fall applied dairy manure.

Table 3.1. Seasonal precipitation, discharge, and runoff ratios (discharge/precipitation) for the 8-year study period for all tiles (DT,CT-I, and CT). [ ] indicates corrected discharge and runoff ratios using the manual backpressure correction method.

	Precip (mm)	Discharge			Runoff Ratio		
		DT	CT-I	CT	DT	CT-I	CT
Fall	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Winter	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Spring	353.8	65.27	240.63	208.59	0.18	0.68	0.59
Summer	230.9	11.37	26.45	90.29	0.05	0.11	0.39
2010-11	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Fall	360.2	40.10	175.53	128.28	0.11	0.49	0.36
Winter	212.6	62.10	147.16	97.79	0.29	0.69	0.46
Spring	103	43.99	48.34	47.46	0.43	0.47	0.46
Summer	179.6	16.17	14.66	16.68	0.09	0.08	0.09
2011-12	855.4	162.36	385.69	290.20	0.19	0.45	0.34
Fall	237.2	13.70	10.23	17.13	0.06	0.04	0.07
Winter	246	84.10	243.51	182.41	0.34	0.99	0.74
Spring	261.7	82.47	350.05	207.01	0.32	1.34	0.79
Summer	233.8	54.03	101.48	91.87	0.23	0.43	0.39
2012-13	978.7	234.31	705.27	498.41	0.24	0.72	0.51
Fall	370.8	86.10	282.71	170.82	0.23	0.76	0.46
Winter	102.2	60.66	86.00	67.71	0.59	0.84	0.66
Spring	182.4	80.92	153.95	94.78	0.44	0.84	0.52
Summer	233.6	9.56	0.00	4.53	0.04	0.00	0.02
2013-14	889	237.24	522.66	337.83	0.27	0.59	0.38
Fall	291.1	108.85	151.75	152.57	0.37	0.52	0.52
		[108.85]	[125.2]	[143.92]	[0.37]	[0.43]	[0.49]
Winter	164.4	20.61	34.16	26.63	0.13	0.21	0.16
		[20.61]	[34.16]	[26.63]	[0.13]	[0.21]	[0.16]
Spring	146.7	27.53	48.19	33.60	0.19	0.33	0.23

		[27.53]	[48.19]	[33.60]	[0.19]	[0.33]	[0.23]
Summer	274.6	25.85	39.24	40.35	0.09	0.14	0.15
		[25.85]	[39.24]	[40.35]	[0.09]	[0.14]	[0.15]
2014-15	876.8	182.83	273.85	253.16	0.21	0.31	0.29
		[182.83]	[246.8]	[244.51]	[0.21]	[0.28]	[0.28]
Fall	187.1	15.29	18.22	15.21	0.08	0.10	0.08
Winter	207.9	62.88	78.84	98.31	0.30	0.38	0.47
Spring	222.3	81.94	90.27	57.45	0.37	0.41	0.26
Summer	324.6	13.73	5.13	3.52	0.04	0.02	0.01
2015-16	941.9	173.84	192.46	174.48	0.18	0.20	0.19
Fall	175.1	0.09	26.14	0.00	0.00	0.15	0.00
Winter	214.5	49.18	72.13	76.22	0.23	0.34	0.36
Spring	332.1	45.39	53.99	121.35	0.14	0.16	0.37
Summer	159.6	0.00	0.00	0.00	0.00	0.00	0.00
2016-17	881.3	94.66	152.26	197.57	0.11	0.17	0.22
Fall	207.7	38.76	192.46	96.83	0.19	0.93	0.47
		[38.76]	[192.46]	[96.83]	[0.19]	[0.93]	[0.47]
Winter	229.4	101.66	114.75	123.56	0.44	0.50	0.54
		[101.66]	[68.58]	[76.74]	[0.44]	[0.30]	[0.33]
Spring	205.5	32.00	69.36	58.76	0.16	0.34	0.29
		[32.00]	[56.96]	[58.76]	[0.16]	[0.28]	[0.29]
Summer	295.1	N/A	N/A	N/A	N/A	N/A	N/A
2017-18	937.7	172.42	376.57	279.15	0.18	0.40	0.30
		[172.42]	[318.0]	[232.32]	[0.18]	[0.34]	[0.25]

The majority of the runoff occurred during the NGS with periods of time when the tiles dried up in the summer months. Runoff ratios exhibited high seasonal variability ranging from 0.21-0.97 across tiles (Table 3.1). During the 2014-15 water year the largest proportion of tile runoff occurred during the fall months. This was not seen in the 2017-18 water year, where the largest proportion of tile runoff occurred during the winter months.

All tiles responded to hydrologic events corresponding to rain, melt, or rain on snow events. During the two intensive study years a total of 11 and 12 hydrologic events (up to 2.8L/s in 6” diameter tiles) were recorded during the 2014-15 and 2017-18 NGS respectively (table 3.2). In-between hydrologic events base flow (approximately 0.05L/s) occurred. During the summer months events were triggered by storms with no-flow conditions in-between. The first hydrologic event to reach a discharge threshold larger than 20mm was the first rain on snow event which occurred in late November and January for the 2014-15 and 2017-18 NGS respectively (Figure 3.7). Snowmelt contributed to the largest hydrologic events in any given year. All events for the 2014-15 and 2017-18 NGS were highly correlated to level of precipitation and number of days above 0ºC when the ground was snow covered. This resulted in the 2017-18 NGS experiencing a greater number of rain on snow events compared to the 2014-15 NGS.

Table 3.2. Event delineation of the 2014-15 and 2017-18 NGS including the start and end date and event type (R=rain, M=melt, and M+R=melt and rain).

NGS	Event	Start Date	End Date	Event Type
2014-15	1	9/5/2014	9/7/2014	R
	2	9/10/2014	9/11/2014	R
	3	9/21/2014	9/21/2014	R
	4	10/6/2014	10/7/2014	R
	5	11/22/2014	11/24/2014	M+R
	6	12/24/2014	12/25/2014	R
	7	3/30/2015	3/31/2015	M

	8	3/31/3015	4/1/2015	M
	9	4/2/2015	4/3/2015	M
	10	4/8/2015	4/9/2015	R
	11	4/9/2015	4/10/2015	R
2017-18	1	11/2/2017	11/3/2017	R
	2	11/15/2017	11/17/2017	R
	3	11/18/2017	11/20/2017	R
	4	12/18/2017	12/20/2017	M
	5	1/10/2018	1/17/2018	M+R
	6	1/22/2018	1/23/2018	R
	7	2/20/2018	2/21/2018	M+R
	8	3/29/2018	3/30/2018	M+R
	9	4/3/2018	4/5/2018	R
	10	4/14/2018	4/15/2018	R
	11	4/15/2018	4/18/2018	R
	12	5/4/2018	5/4/2018	R

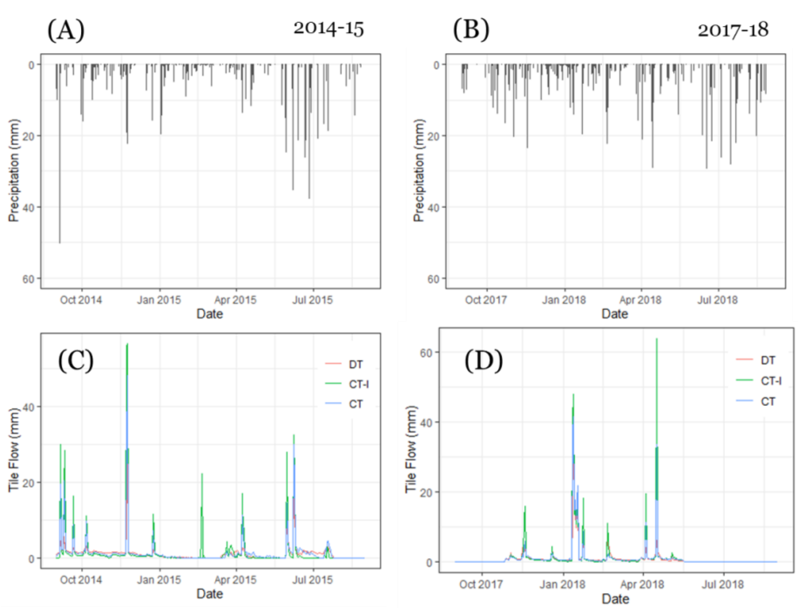

Figure 3.7 Daily precipitation of the 2014-15 (A) and 2017-18 (B) water year (snow and rain) in addition to the hydrograph of the 2014-15 (C) and 2017-18 (D) for the three tiles.

3.3 Spatial Variability in Tile Drain Hydrology over the Eight-Year Period: Impacts of Tillage Treatments

Naturally occurring hydrological variability across the tiles was observed prior to any tillage treatment (Feb-Sept 2011; Figure 3.6). The CT-I tile generally exhibited a larger runoff response relative to all tiles with the DT tile typically having a smaller response. These observed differences among the tiles were very dependent on the type and size of hydrologic event with some events having all 3 tiles behave very similarly. This trend remains consistent across the tiles following the treatment of tillage and is also supported by in field observations (Figure 3.8).

Figure 3.8 In field observations of each tile. Pictures taken on April 16 2018.

Regardless of the natural variability observed among the tiles, in field hydrologic differences in tile runoff were detected with tillage treatments. It was found that discharge was similar between the CT and DT plots during some periods, but different during others. Closer inspection of the relationship between the two plots revealed that the relationship between the two plots changed over time following tillage (Figure 3.9). Discharge from the CT and DT plots closely followed a 1:1 line (R^2=0.70, $p<0.001$) for up to 5 months following tillage. However, after this threshold of 5 months since tillage was reached, CT discharge was much more responsive compared to the DT plot (R^2=0.60, $p<0.001$). Overall, treatment effects of CT showed to have increased hydrologic activity compared to the DT following the time threshold of 5 months post tillage regardless of no observed differences between soil temperature and moisture.

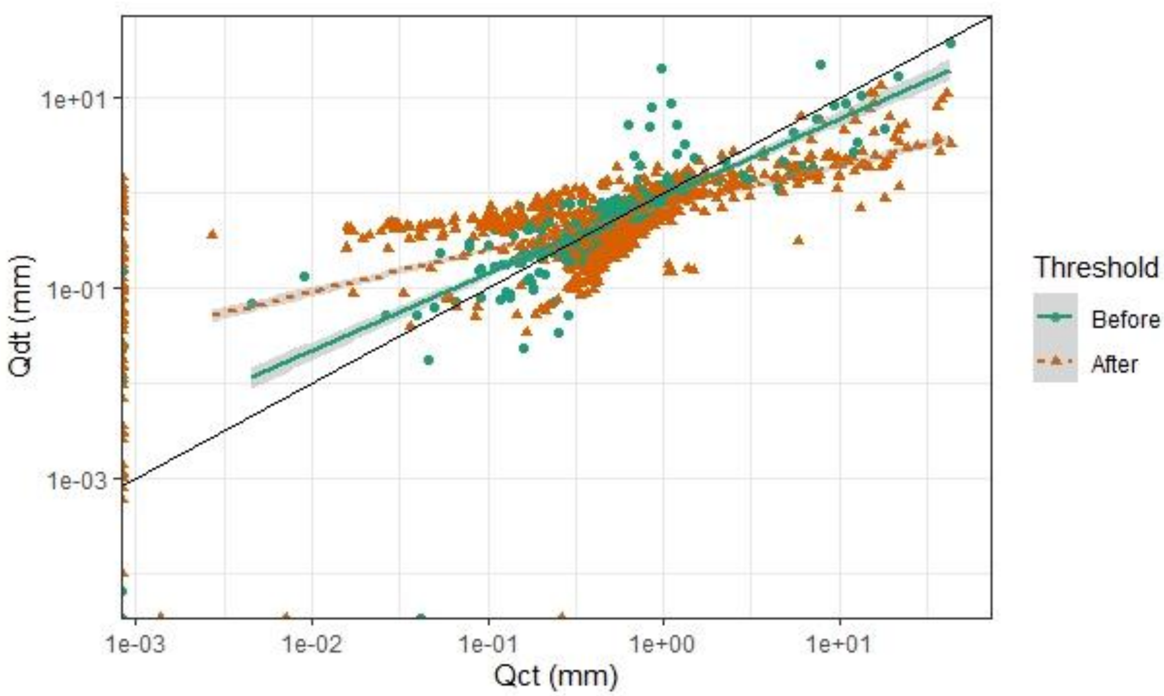

Figure 3.9 Daily discharge (mm) of DT against CT across all study years (2011-2018). Colour indicates time since tillage whether the point is before (green) or after (orange) the 5 month tillage threshold. Linear regression of before (solid line) and after (dashed line) this 5 month tillage threshold is displayed.

3.4 Differences in P Concentrations and Loads Following Manure Application

The 2014-15 and 2017-18 NGS instantaneous water samples collected on a hydrological event basis showed variability within and among events. During periods of base flow, TP and SRP concentrations were found to be less than 0.01mg/L and as a result did not significantly contribute to P loads or FWMCs. Therefore, P loads and FWMCs were determined for individual hydrologic events and compared.

3.4.1 Impacts of Tillage Following Manure Application on Phosphorus Losses in Tile Drainage

On an event basis, FWMC in the DT plot were often higher and showed greater variability compared to the other tiles (Figure 3.10). However, no statistically significant differences with the exception of DT compared to CT for SRP and TP 2014 and 2017 respectively were found (figure 3.10).

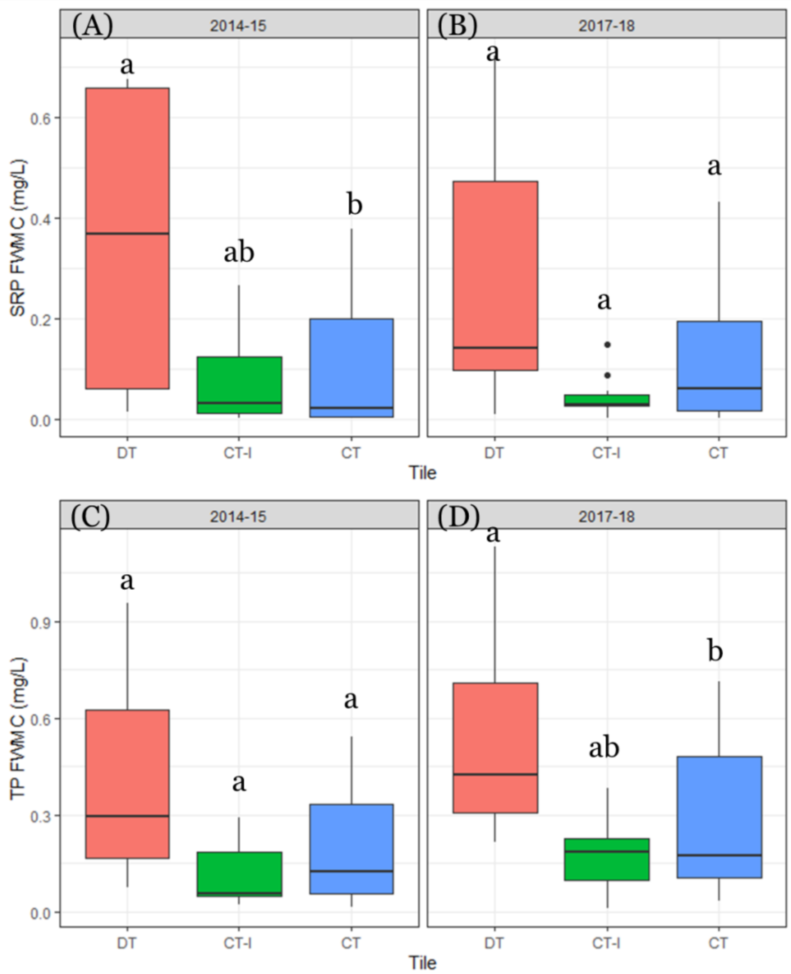

Figure 3.10 Calculated event FWMCs for SRP (A, B) and TP (C, D) for individual tiles and NGS (n=11 and 12 for the 2014-15 and 2017-18 NGS respectively). Boxplot illustrates the median (solid horizontal line), the interquartile range of the 25th to 75th percentile values (box outline), values within 1.5 times the interquartile range (line whiskers), and outlier values that are 1.5 times greater than the interquartile range (circles) Letters indicate statistical significance using an unpaired Wilcoxon test at a probability level of 0.05.

Some differences between the tiles were observed when FWMC was combined with the event flow to calculate P loads per event (Figure 3.11). These observations were more prevalent in the events leading up to and including the first large melt event. For the 2014-15 and 2017-18 NGS the DT plot generally exhibited higher loads compared to the CT-I and CT plots. However, when all of the event P loads were compared for the tiles, the variability initially observed was lost and showed no statistical differences (Figure 3.12). Outliers observed in Figure 3.12 can be attributed to the first large melt event 5 for both the 2014-15 and 2017-18 NGS. In other words, for most events, P loads did not differ among the tiles; however, for the largest events in a given year, differences were observed, with greater loads from the DT tile.

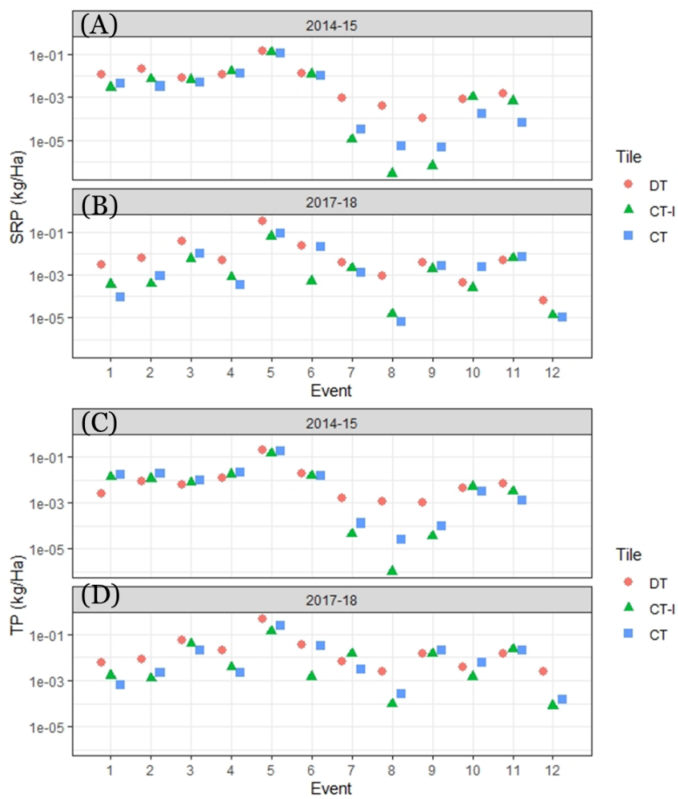

Figure 3.11 Event Loads for SRP (A, B) and TP (C, D) for all tiles.

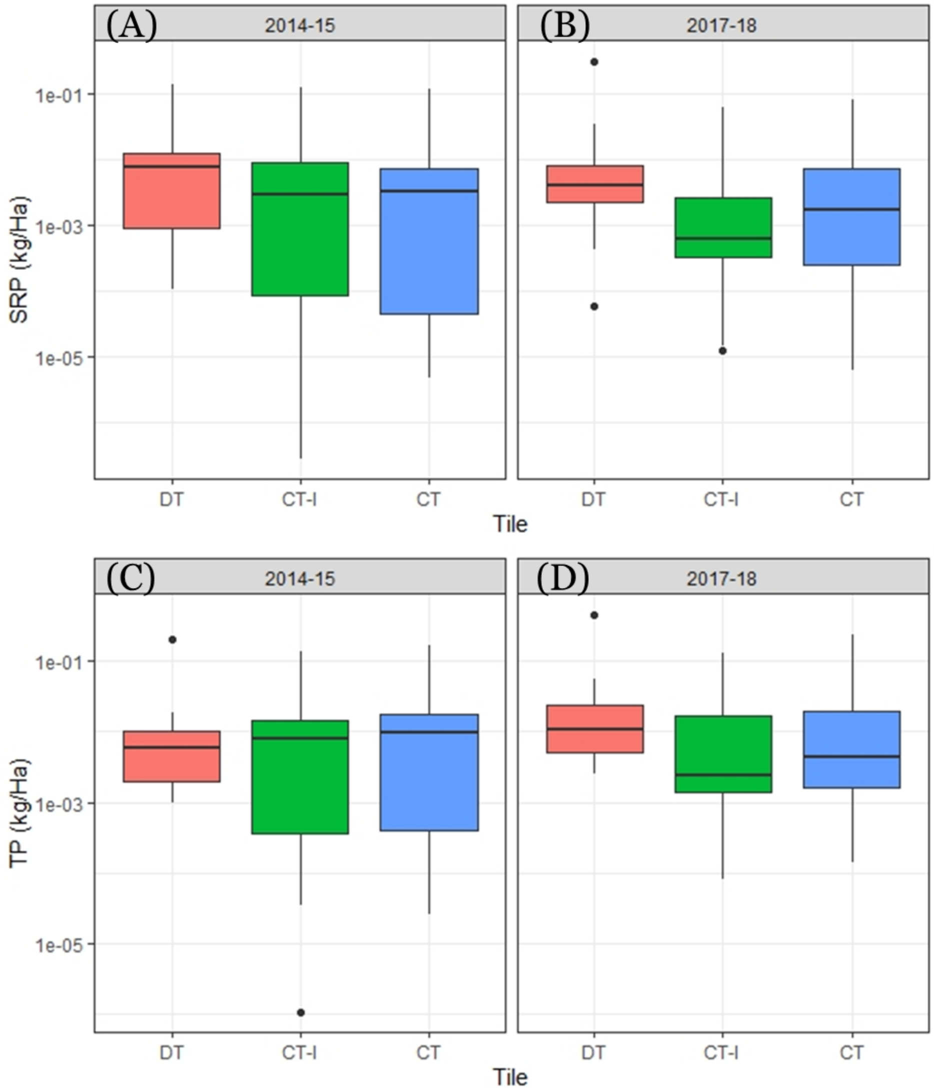

Figure 3.12 Calculated event loads for SRP (A, B) and TP (C, D) for individual tiles and NGS (n=11 and 12 for the 2014-15 and 2017-18 NGS respectively). Boxplot illustrates the median (solid horizontal line), the interquartile range of the 25th to 75th percentile values (box outline), values within 1.5 times the interquartile range (line whiskers), and outlier values that are 1.5 times greater than the interquartile range (circles)

Given the relevance of the large events to annual P loads, when cumulative loads were determined, the spatial difference between tiles re-emerged (Figure 3.13). The pattern in cumulative SRP and TP loads across events strongly reflected the observed temporal trends in cumulative discharge, which were consistent among the tiles. Divergence in P loads among tiles was observed following the outlier event 5, which represented the first large melt event in both the 2014-15 and 2017-18 NGS. During the 2014-15 event 5, very little differences between the tiles are observed, with DT showing slightly higher SRP loads. In the 2017-18 event 5 a greater divergence is observed with the DT plot having consistently higher SRP and TP loads followed by CT then CT-I. Succeeding this event, minimal changes in cumulative P loads occurred.

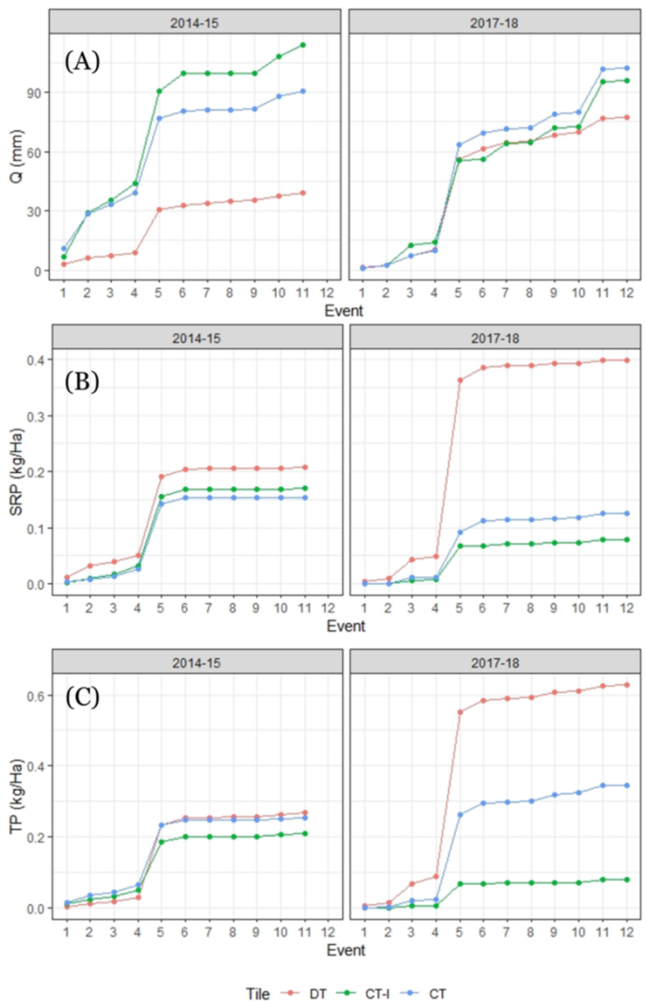

Figure 3.13 Cumulative discharge across recorded hydrologic events for the 2014-15 (A) and 2017-18 (B) NGS of all tiles. In addition to cumulative P loads across events for the 2014-15 (C and E) and 2017-18 (D and F) for SRP and TP respectively for all tiles.

Due to the significance of the first large discharge event on cumulative loads, an exploration of the relationships between P and discharge was explored further. When SRP and TP FWMCs were plotted against discharge, no patterns emerged (Figure 3.14). However, when cumulative SRP and TP loads against cumulative discharge were plotted, the spatial pattern between tiles still emerged with variance amongst tiles being driven by event 5 and differences in discharge (Figure 3.15). In order to investigate the power of event 5 on the system and whether the system was transport or supply driven, P load and discharge (L-Q) across all tiles were plotted (Figure 3.16). A linear relationship was detected and following linear regressions, the L-Q plots were found to be statistically significant (Table 3.3) in addition to no autocorrelation being detected across the tiles (p-value >0.05). SRP consistently had weaker regressions compared with TP. Therefore, with the exception of CT-I and CT SRP, all R^2 values exceeded 0.8 supporting a chemostatic (transport) driven system among all tiles for the manure application years (Basu et al., 2010).

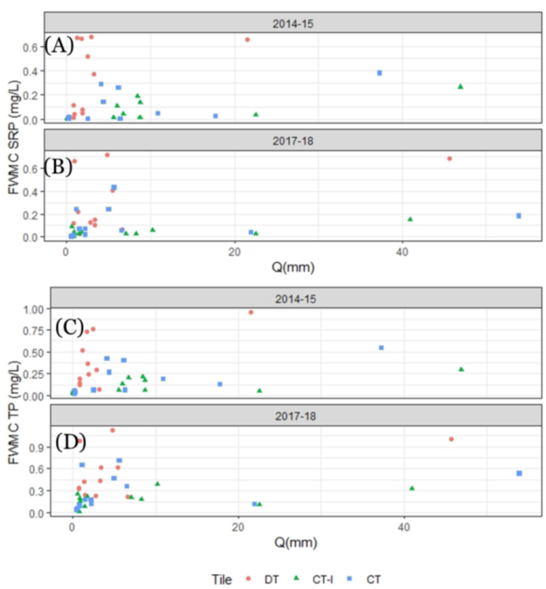

Figure 3.14 Scatter plot of SRP (A, B) and TP (C, D) FWMCs against discharge across tiles and the 2014-15 and 2017-18 NGS.

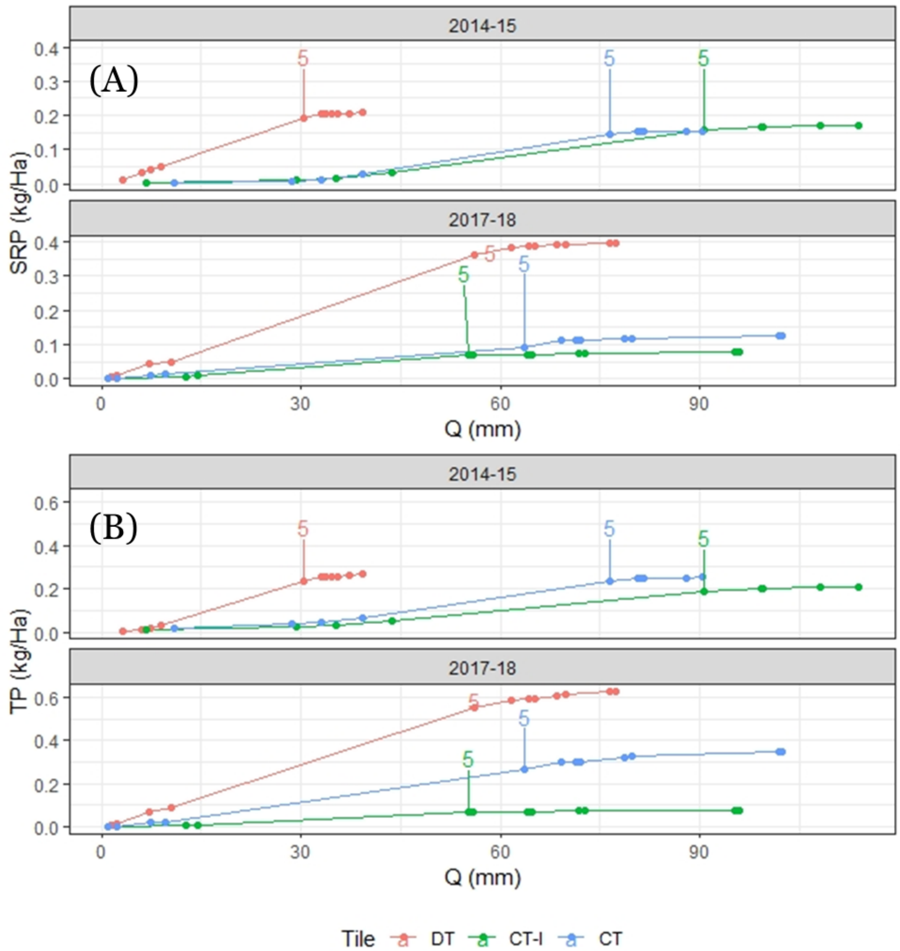

Figure 3.15 Cumulative SRP (A) and TP (B) loads against Cumulative discharge across all tiles. Points indicate successive recorded hydrologic events with particular emphasis on the large hydrologic event 5.

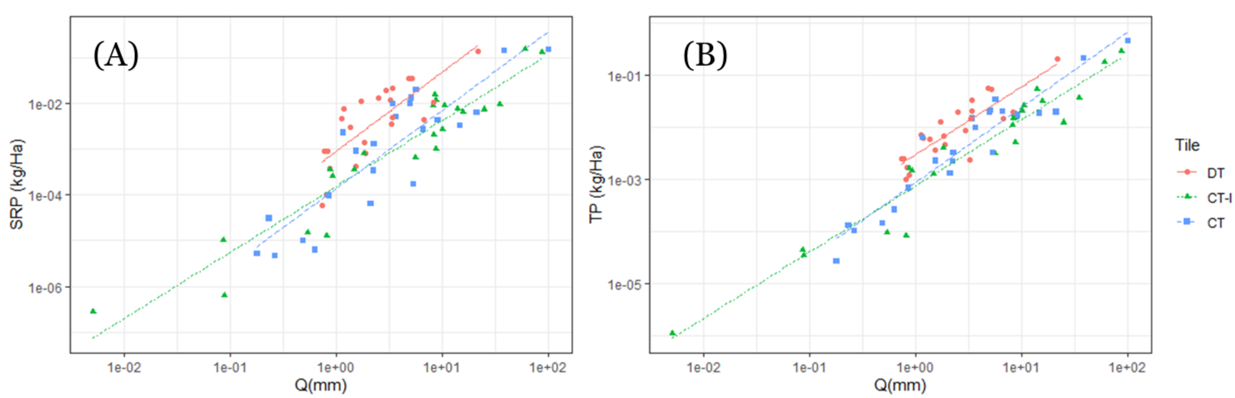

Figure 3.16 Event loads of SRP (A) and TP (B) against discharge on a log-log scale with regression lines per tile for both 2014-15 and 2017-18 NGS.

Table 3.3 Metric values for linear regressions of event SRP and TP loads (kg/Ha) with discharge (mm) for the 2014-15 and 2017-18 NGS. * indicating statistical significance at a probability level of 0.05.

P form	Tile	n	R^2	P-value
SRP	DT	23	0.98	<0.001*
SRP	CT-I	23	0.74	<0.001*
SRP	CT	24	0.70	<0.001*
TP	DT	23	0.99	<0.001*
TP	CT-I	23	0.84	<0.001*
TP	CT	24	0.88	<0.001*

3.4.2 Impacts of Incorporation Following Manure Application on Phosphorus Losses in Tile Drainage

The impact of incorporation resulted in decreased P loss in the tile over the NGS. Instantaneous SRP and TP concentrations showed no observed differences between treatments of the control NGS of 2014-15 (Figure 3.17). However, when looking at SRP, TDP, and TP FWMCs during the 2017-2018 water year, differences across events were found. The CT tile generally had higher levels of FWMCs although this was not always consistent (Figure 3.18). Notably, when calculating P loads, no differences were observed between events with the exception of the first large discharge event (event 5). At this first melt event, the CT-I tile displayed lower levels of SRP, TDP, and TP compared the CT tile even though both tiles experienced very similar levels of discharge. The impact of this melt event drove the cumulative P loss trend over the NGS. Although there were differences in discharge between the two plots, the differences in P loss between the plots were greater, suggesting that incorporation had an impact on reducing P loss in tile drainage.

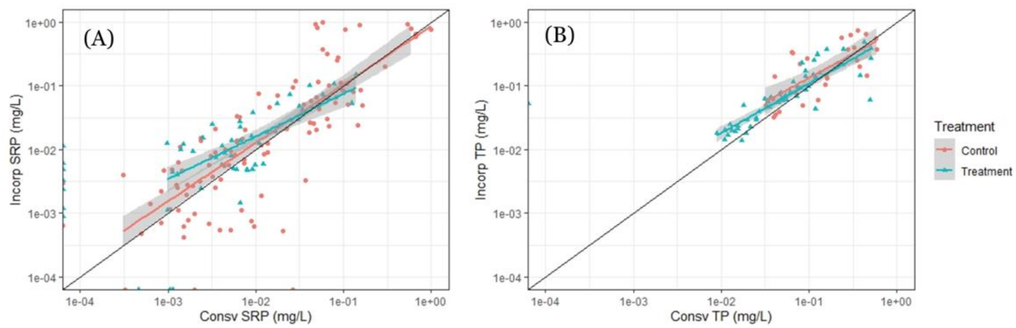

Figure 3.17 Instantaneous SRP (A) and TP (B) concentrations of incorporation (CT-I) against conservation tillage (CT) before (2014-15) and after (2017-18) treatment on a log-log scale with lines of best fit.

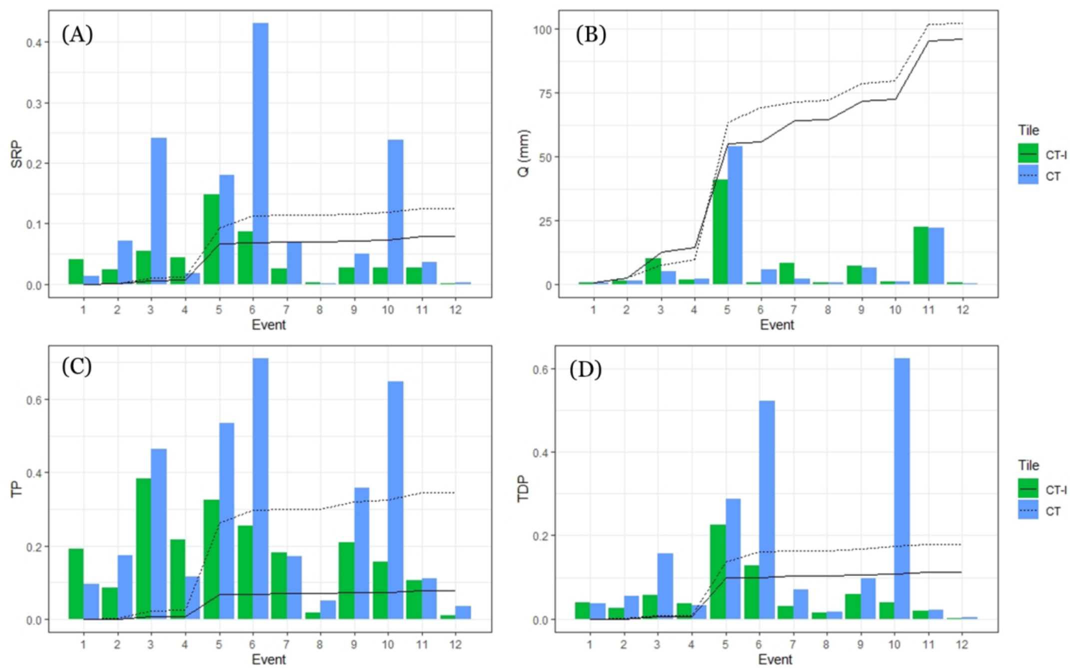

Figure 3.18 Cumulative SRP (A), TP (C), TDP (D) loads (kg/Ha) and discharge (mm) (B) across events for the CT-I (solid line) and CT (dashed line) tiles. In addition to SRP (A), TP (C), TDP (D) event FWMCs (mg/L) and discharge (mm) (B) for the CT-I (green bar) and CT (blue bar) tiles.

The speciation of P exhibited no consistent trends across events or between plots for either the 2014-15 or 2017-18 NGS (Figure 3.19). During the treatment year of 2017-18 a greater proportion of DRP and DNRP were observed in the CT tile, which is particularly evident during the large discharge event 5. However, regardless of treatment or NGS a larger proportion of particulate P loss was observed in the CT tile.

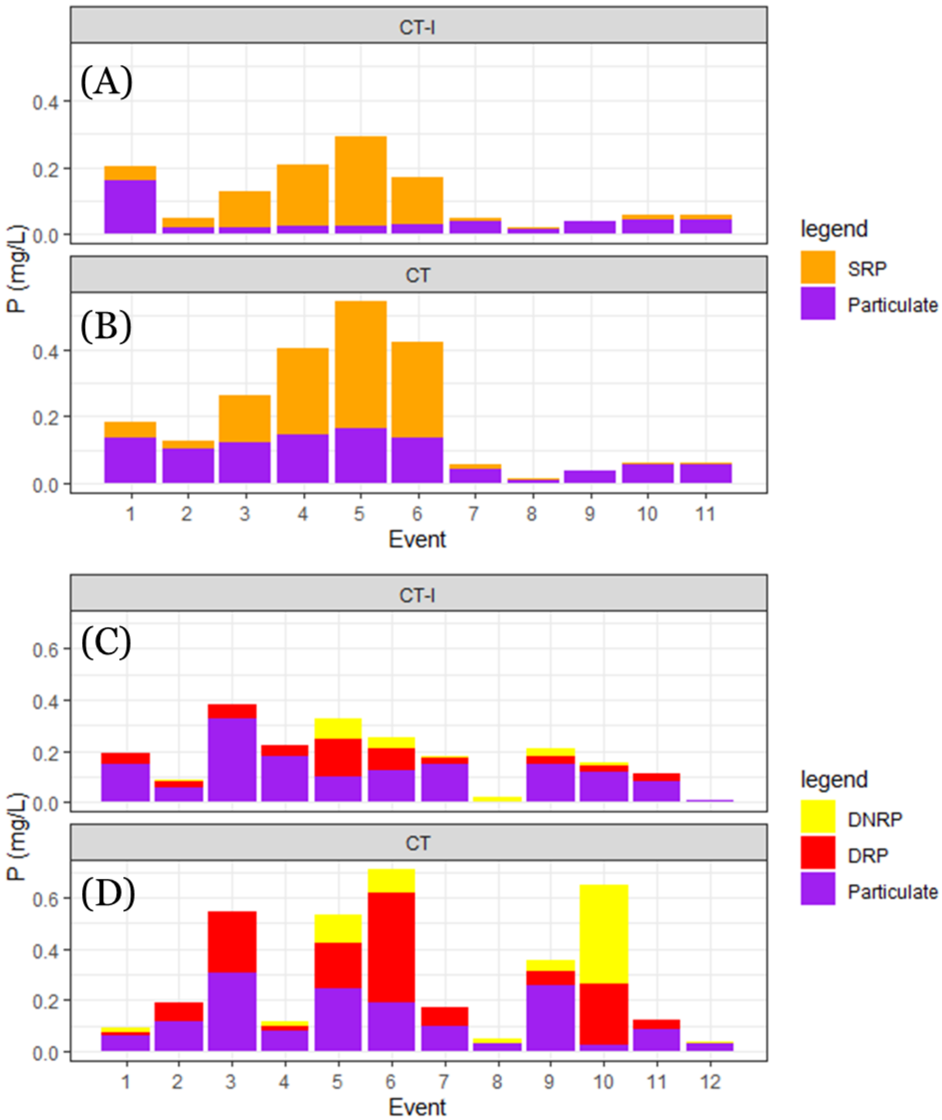

Figure 3.19 P speciation of the 2014-15 (A, B) and 2017-18 (C, D) NGS across events for the CT-I and CT tiles.

3.5 Soil Test Phosphorus Distributions at Study Site

Observations of the silt-loam soil profile varied depending on climate conditions, sampling time, and fertilizer application. There were no observed differences in soil profile over the 8-

year study period, which showed an overall decreasing trend of Olsen P with increasing depth (Figure 3.20). This trend remained consistent across all plots. When examining P stratification more closely within the NGS, changes in Olsen P and water extractable P (WEP) from fall to spring during the 2017-18 NGS were observed. During fall 2017 differences in Olsen P stratification between plots was evident with DT having higher P values from 2.5 to 7.5cm compared to CT and CT-I (Figure 3.21(A)). However, these differences in P stratification disappeared over time with spring Olsen P values demonstrating no significant stratification with depth (Figure 3.21(B)). In contrast, WEP showed the opposite trend over the NGS with fall WEP demonstrating no stratification with depth compared to spring WEP which showed slight stratification (Figure 3.22). Slight differences across the tiles were detected, with the most notable being the DT plot having higher Olsen P levels further down in the soil profile.

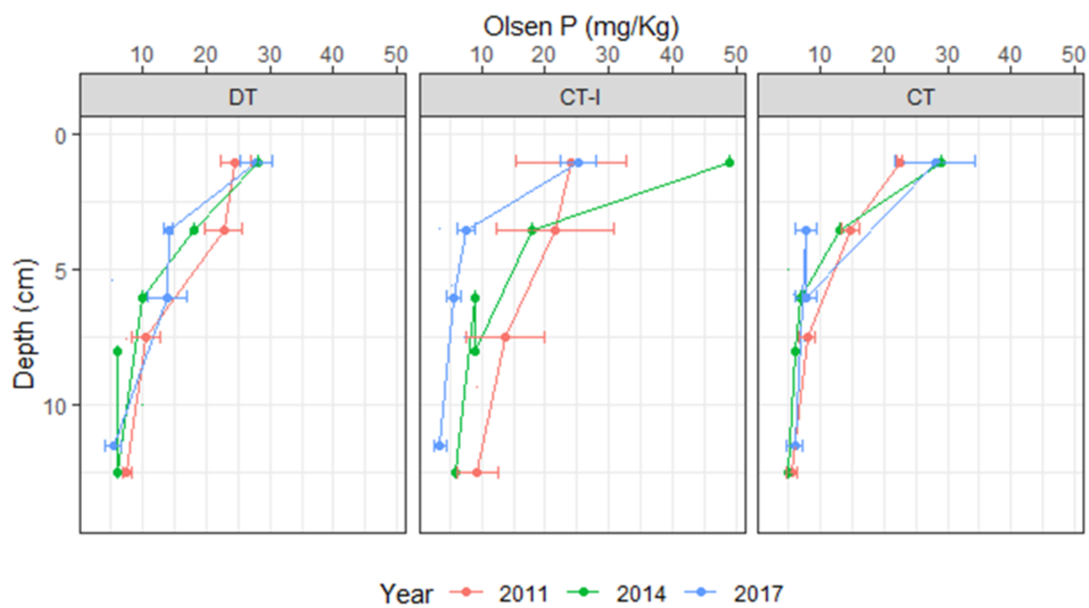

Figure 3.20 Fall sampled Olsen P against depth across tiles for 2011, 2014, and 2017. Horizontal whiskers indicate calculated standard error.

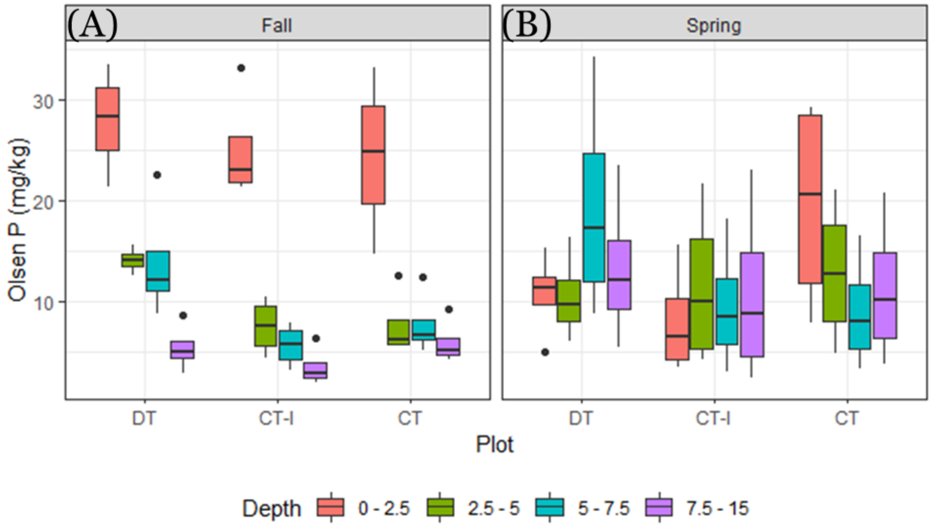

Figure 3.21 Olsen P at the depths of 0-2.5, 2.5-5, 5-7.5, and 7.5-15 cm across tiles for fall (A) and spring (B) of the 2017-18 NGS. Boxplot illustrates the median (solid horizontal line), the interquartile range of the 25th to 75th percentile values (box outline), values within 1.5 times the interquartile range (line whiskers), and outlier values that are 1.5 times greater than the interquartile range (circles)

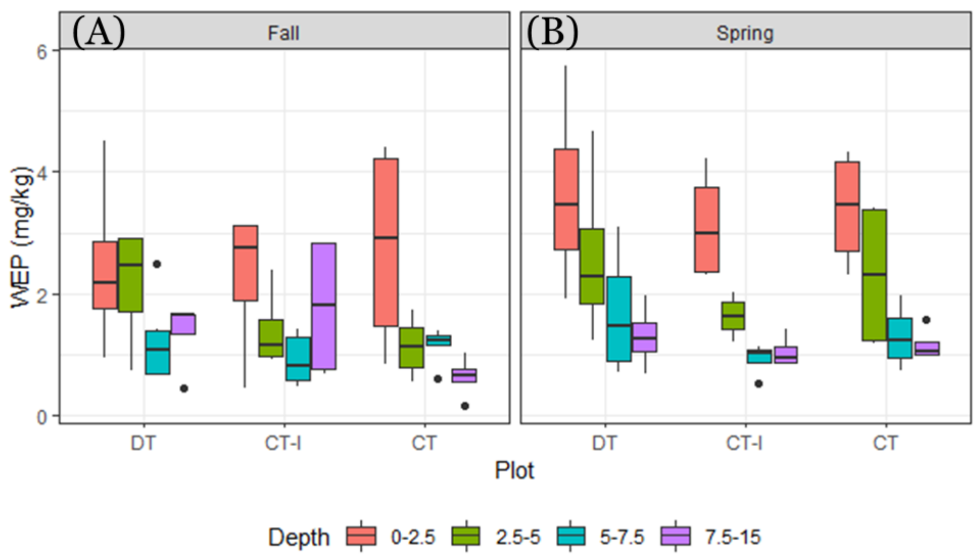

Figure 3.22 WEP at the depths of 0-2.5, 2.5-5, 5-7.5, and 7.5-15 cm across tiles for fall (A) and spring (B) of the 2017-18 NGS. Boxplot illustrates the median (solid horizontal line), the interquartile range of the 25th to 75th percentile values (box outline), values within 1.5 times the interquartile range (line whiskers), and outlier values that are 1.5 times greater than the interquartile range (circles)

Chapter 4
Discussion

The goal of this study was to determine the impacts of tillage and incorporation on subsurface P loads and concentrations from fall applied dairy manure fields over the NGS. It was predicted that due to tillage breaking up preferential pathways, plots with a deep till treatment would have overall lower P loads and concentrations compared to those with a conservation tillage treatment. Additionally, the incorporation of fall applied dairy manure was predicted to reduce subsurface P loads and concentrations due to incorporation combining organic P within the soil profile. The objectives of this study were to: 1) quantify SRP and TP losses from tile drains following the fall application of manure over the NGS; 2) investigate if conventional or conservation tillage impacts subsurface P loss; and 3) determine if incorporation of manure impacts subsurface P loss in runoff. The results of this study were variable and were not always consistent with the above predictions. This study demonstrates the complexity of P release and transport within the system.

4.1 Comparison of the Study to Existing Literature

Due to the long-term monitoring of this study, the data gathered encompassed a variety of conditions to make it comparable to other similar studies. Across all study years the NGS contributed a greater proportion of annual precipitation resulting in greater tile discharge in the NGS. This supports what has been observed in the literature and highlights the importance of tile drain data collection over the NGS (Coelho et al., 2012; King et al., 2016; Lam et al., 2016). Macrae et al., (2007) found that approximately 42% of annual precipitation inputs were exported through tile drains with the greatest contribution occurring in the winter and spring. The precipitation lost via tile drains varied greatly over the 8-year study period. Annual runoff ratios for the 2014-15 and 2017-18 water year ranged from 20-40% across tiles demonstrating comparability to other studies. Nevertheless, the range of the tile precipitation coefficients indicates seasonal and within field variability. It is important to note that there was no consistent pattern observed amongst tiles across study years. This suggests natural variability within the site and should therefore be taken into consideration when making inferences on the management treatments and their overall impact. Grant et al., 2019 found that in a silt-loam soil, matrix flow dominated compared to preferential flow pathways which supports a greater amount of P retention within the soil due to the higher soil-matrix and water contact. The type of melt event will also contribute to the amount of discharge observed in the tiles. Slow radiation melt events promote tile flow versus rain or rain on snow events which will promote higher proportions of surface overland flow (Plach et al., 2019). Overall, due to the multi-year nature of this study, comparisons between study years can be used to explain within field and seasonal variability while still being comparable to similar studies conducted in Southern Ontario.

P losses from the study site were relatively comparable to what has been observed in the literature within Ontario. NGS P losses ranged from 0.08 to 0.40kg/Ha and 0.21 to 0.62kg/Ha for SRP and TP respectively. Van Esbroeck et al., (2016) conducted a similar study where P tile losses were quantified from 3 Ontario reduced till sites and found P values ranging from 0.006 to 0.078kg/Ha and 0.072 to 0.881kg/Ha for SRP and TP respectively over 2 NGSs. This can also be compared to annual losses from a multi-year/site study in Ontario which showed P values ranging from 0.03 to 0.35kg/Ha/yr and 0.18 to 1.93kg/Ha/yr for SRP and TP respectively (Plach et al., 2019). Furthermore, P FWMCs were also comparable to what has been observed elsewhere in the literature (King et al., 2015; Macrae et al., 2007; Smith et al., 2015). These observed P losses can be attributed to the initial low soil P values and the general management of the study site. The mean Olsen P observed was 14.3+/-2.7mg/kg which is low when compared to a study by Wang et al., (2015a) that looked at 60 Ontario soils with mean Olsen P values ranging from 7.2-60.1mg/kg. Wang et al., (2010b) found that soils with Olsen P <30mg/kg had significant P sorption resulting in minimal soluble P loss in runoff which further supports the low P observed within the tile drains at the study site.

Low P losses observed in the tile drains can also be attributed to the type of flow in addition to the solute transport and retention experienced within the soil profile to the tiles. The bulk of P losses observed in the tile occurred during event flow. This highlights the importance of capturing storm events year round and supports the existing literature that has observed significant P loss in tiles during storm events (King et al., 2015; Macrae et al., 2007; Van Esbroeck et al., 2016; Vidon and Cuadra 2011). Within events, the majority of P loss was associated with the rising limb of the hydrograph with relatively low P concentrations during the latter portion. There were no patterns of FWMCs with total amount of discharge or across events. These results further support the transport of solutes through matrix flow rather than preferential flow (Macrae et al., 2007 and Kleinman et al., 2009). This dominance in matrix flow is largely in part due to the silt loam texture of the soil, which has a lower risk of developing defined macropores supporting preferential solute transport resulting in great P retention within the soil (Grant et al. 2019).

The overall study parameters of discharge and P loss were comparable to what has been observed elsewhere in the literature. Yet at the same time demonstrates the variability that can be observed among seasons and within the field. The chemistry data supports the general dominance of matrix flow within the field. Therefore, capturing event flow is crucial in order to understand P losses within the tile drains in addition to the type of event (e.g. radiation melt, rain, or rain on snow) to explain the proportion of discharge observed within the tiles. As a result, understanding mechanisms that manipulate these processes is crucial in order to grasp how different management strategies may affect P loads and concentrations.

4.2 Impacts of Tillage Practices on P Transport and Loss via Tile Drains

This study has shown mixed results regarding the role of tillage practices on P mobilization in tile drains, demonstrating the complexity of influencing factors within the system. The impact of tillage on P concentrations and loads in tile drains is still debated today due to the conflicting results within the literature (Lam et al., 2016; Williams et al., 2016; Zhang et al., 2017). It was predicted that the CT treatment would support the development of macropores resulting in increased preferential solute transport to the tiles.

4.2.1 Impact of Tillage on Tile Drain Hydrology

In the current study, tillage practices were compared among plots within a field. However, such a comparison relies on the assumption that there are no within-field differences among study plots. Thus, a comparison of plots (baseline) prior to the treatments is essential. When examining the comparison in discharge between DT and CT before and after tillage treatments it was observed that CT is naturally more hydrologically reactive regardless of tillage treatments. Vadose zone hydrology is still a very difficult area to study and therefore differences in the underlying soil structure may be at play. Nevertheless, challenges in understanding and mapping accurate water movement within the vadose zone still remains difficult (Liu and Yeh, 2004; Van Schaik, 2009; Vereecken et al., 2008). Lam et al., (2016) found that natural variability in tile discharge and P loss within a field occurred regardless of tillage treatments. This is understandable considering the heterogeneity of landscapes. United States soil mapping typically has 5-10 soil mapping units within a average 160Ha farm, with the variability in soil properties being largely attributed to small changes in topography that impacts storage and transport throughout the soil profile (Mulla and McBratney, 2002). Nevertheless, closer examination of tillage impacts on tile drain hydrology is necessary to determine if tillage does in fact influence local hydrology.

Responsiveness of the tiles in relation to time since tillage showed slight differences between the DT and CT treatments. The impact of the tillage treatment breaking up macropores appears to be only effective until it reaches the threshold of 5 months. Following this 5 month threshold the tile becomes increasingly more responsive suggesting the development of a more defined macropore network supporting preferential flow. The DT treatment is done bi-annually making the plot unable to recover and therefore acts as a good treatment comparison to the CT plot. It is well documented within the literature that tillage impacts the physical properties of soil (Bosch et al., 2005; Cassel et al., 1995; and Hussain et al., 1998). Management practices such as conservation tillage and no-till have shown to increase soil compaction, bulk density, and infiltration (Cassel et al., 1995). Williams et al., (2016) found that disk tillage substantially reduced event discharge in tiles with 10-50% attributed to preferential flow. Understanding this relationship between the soil matrix and preferential flow is necessary to grasp how tillage can impact water transport to the vadose zone and subsequently into the tiles. Klaus et al., (2013) found that up to 80% of tile drain effluent originated from the soil matrix. However, many studies have found that in no-till systems the soil matrix has a greater likelihood to be bypassed resulting in greater proportions of preferential flow through macropores (Grant et al., 2019; Shipitalo and Edwards, 1993; Shipitalo et al., 2000). Understanding water transport of the vadose zone is very difficult due to unpredictable movement of water through macropores which is exacerbated by the difficulty in quantifying the factors involved. Factors influencing macropore development such as root growth, worm holes, and clay desiccation is understood to have an impact, however very little research has been completed exploring the extent of macropore development over time. Impacts on soil properties over time since tillage treatment has been explored at the surface using soil erodibility as an indicator (Auerswald, 1993; Knapen et al., 2008). Nevertheless, understanding macropore development over time and how it pertains to water and solute transport in tiles needs to be investigated further. A better understanding of the mechanisms involved in addition to the rate of macropore network development has the potential to be very important for BMP tillage recommendations.

Types of tillage are classified by the percentage of crop residue left at the surface with no-till having greater than 30% crop residue and conventional tillage practices have less than 15% (West and Marland 2002). Increased proportions of crop residue left on the soil can create a buildup of residual P in addition to increasing snow capture resulting in higher levels of surface runoff (Benoit et al., 1986; Zhang et al., 2015; Hansen et al., 2000). From a snow survey conducted in 2014, this study showed no statistical difference in snow cover across plots. Yet, soil moisture at the surface was on average higher at DT compared to CT-I, suggesting a higher amount of overland flow occurring at the DT plot. This is in contrast to before tillage treatments where DT had lower soil moisture at the surface compared to CT-I. The soil moisture data indicates that tillage influences overland flow which could be due to the disc till creating divots in the soil capturing water and snow melt providing horizontal preferential pathways along the surface of the field. The influence of disk tillage on soil moisture appears to be only effective up to the depth of the tillage since both the depth of the disk till and differences in soil moisture were observed at 10cm. This soil moisture results from this study contradicts the existing literature that states that due to a decrease in evaporation and a greater capacity to hold soil moisture, no-till generally has higher soil moisture compared to tilled plots (Blevins et al., 1971; Blevins et al., 1983; and Kovac et al., 2005). However, this conclusion is based on studies completed during the growing season. Maule and Chanasky (1990) examined the effects of tillage on snow capture and found that plots with chiselled stubble and standing stubble had significantly greater snow depths and soil water gains over the NGS compared to the disked stubble and the fallow field. This suggests that disk tillage can reduce snow capture and increase overland flow. Studies like Van Esbroeck et al., (2016) and Lam et al., (2016) have demonstrated the importance of gathering snow data and monitoring hydrology over the NGS. Nevertheless, the NGS remains to be an understudied time period within agricultural studies and the impacts of tillage on snow capture and soil moisture should be investigated further.

Overall, the results of this study support that tillage does impact the hydrology and flow to the tiles. The type and depth of tillage needs to be explored further as these can both significantly impact surface and subsurface flow pathways. Although tillage does appear to influence local hydrology, variability that naturally occurs within the field is still present.

4.2.2 Impact of Tillage on Tile Drain Chemistry

The results of this study showed mixed findings as to whether tillage impacted P loads and concentrations in the tiles. Values across events demonstrated an overall trend of DT having a higher P FWMC and loads compared to CT. This trend contradicts the initial prediction that DT would have lower P loads and concentrations. Indeed, several studies have shown the increase in hydrologic connectivity at no-till sites which resulted in greater P loss (Jarvie et al., 2017; Rahm and Huffman, 1984; Ulen et al., 2010; and Williams et al., 2018). Notably, the differences observed in this study can be attributed to one large event that drives the overall trend. The increase of P loss observed in the DT treatment can be attributed to a combination of factors. Firstly, the type of tillage used to simulate conventional tillage was a disk till, which is sometimes classified as a secondary tillage method due to its lack of inversion of the soil profile. Goehring *et al.*, (2001) compared incorporation methods and their effect on P in subsurface drains and found that incorporation using a moldboard plough at 15cm had significantly lower P concentrations compared to incorporation by disking to a depth of 5cm. This study suggests that without the inversion of the top soil layer as seen in traditional moldboard ploughing, macropores are still present to allow to preferential flow and solute transport to tile drains. Olsen P values from this study displayed stratification among all treatments indicating that inversion of the top soil layer before or after manure application never occurred. Consequently, the DT plot potentially still possessed a network of macropores to promote preferential flow resulting in the increased P FWMCs and loads. The Olsen P values also indicate higher concentrations further down the soil profile of the DT plot. This could be due to the disk till creating points of entry for surface applied dairy manure further down the soil profile creating greater connectivity to preferential pathways. Katsvairo et al., (2002) found that different types of tillage including moldboard plow, chisel tillage, and ridge tillage all had different infiltration rates. Furthermore, the direction of runoff can be strongly influenced by factors such as surface roughness and tillage direction (Souchere et al., 1998; and Takken et al., 2001). Most tillage comparison studies look at the comparison between conventional and no-till. Yet, the different type of tillage between these two extremes needs to be investigated further in order to make stronger conclusions on their impacts on local hydrology and chemistry. Moreover of the studies completed, very few have considered subsurface impacts and the NGS. Finally, the soil texture can also greatly influence the impact of tillage and macropore development. This study was conducted in a silt-loam soil and as a result macropore development can be mainly attributed to biological activity compared to cracking commonly seen in clays. Therefore, the CT plot could be more sensitive to the rotational tillage it receives and is unable to have as well defined of a macropore network. Unlike no-till studies completed in clays, this study most likely experienced greater levels of matrix flow within the CT plot resulting in lower levels of P loss.

4.3 Impacts of Manure Incorporation on P Loss via Tile Drains

The results of this study support the existing literature stating that incorporation helps to reduce subsurface P loads due to the breaking up of preferential pathways in addition to the mixing of organic P within the soil profile (Geohring et al., 2001; King et al., 2015; Kleinman et al., 2009). There were no observed differences in FWMCs between the CT-I and CT plots in the control year of 2014-15 or the treatment year of 2017-18. However, there was an observed trend when considering cumulative loads. Because the plots experienced similar levels of discharge, differences observed can be more strongly attributed to treatment effects rather than differences in hydrology. Using 2014-15 as a control we observe very little difference between the CT-I and CT plot cumulative load, which remains true when examining individual event P loads. In contrast, during the treatment NGS of 2017-18 we observe CT-I having a lower cumulative P load compared to CT, which is particularly evident in TP compared to SRP. Individual P loads show no statistical differences between CT-I and CT. Nevertheless, the difference observed in cumulative loads can be attributed to the outliers associated with the first large discharge event occurring in the tiles. This observation suggests that the treatment of incorporation helps to reduce overall P loads in tiles.

When breaking down the speciation of P in the tiles and across 2014-15 and 2017-18 NGS we observe the CT plot exhibiting larger losses of particulate P. During the treatment NGS of 2017-18 this trend remains true and is exacerbated by the strong observed particulate P difference in the first large discharge event, further supporting the overall impact of this event. Few studies have extensively examined the impact of incorporation and conservation tillage on P speciation as there is greater importance placed on the bioavailable SRP. Furthermore, particulate P loss is strongly correlated to particle distribution size which is more widely studied for surface loss pathways such as overland flow (River and Richardson 2018). Williams et al., (2018) found that the degree of soil-fertilizer contact acted as the main mechanism for P transport when comparing leachate of plots with differing fertilizer placement and tillage. This supports the findings from this study where the treatment of incorporation would have increased this contact reducing dissolved and particulate P losses. Particulate P remains the most dominant form of P in the soil (Sharpley et al., 1992), and as this study demonstrates there is strong evidence to support particulate P loss through subsurface pathways even in minimal soil disturbance management practices such as conservation tillage. The particulate fraction of P should not be ignored, as it can transform and impact in-stream P retention (Withers and Jarvie 2008). Therefore, management practices that increase the soil-fertilizer contact such as incorporation should be considered an effective BMP to reduce particulate and dissolved fractions of P in runoff. These BMPs would allow for greater P retention within the soil and therefore farmers can reduce subsequent fertilizer applications to prevent P from building up within the soil profile.

4.4 Impact of the "First Flush" and Timing of Manure Application

In order to better manage manure application and its resulting impact on water quality, it is important to understand how and when P is transported through the system. Both 2014-15 and 2017-18 NGS had a large discharge event that produced an outlier P load compared to the mean. This event dictates the overall trend by mobilizing the most P and creating differences in cumulative loads. Using the L-Q plots, the results indicate that TP is driven by a chemostatic (transport) response whereas SRP is more variable with CT-I and CT SRP driven by a chemodynamic (supply) response. Therefore, the amount of P lost in the tile is driven by the amount of discharge rather than the amount of P within the soil. This is a novel finding for this study, as previous literature has suggested that loam soils are typically driven by chemodynamic processes due to the soil buffering capacity (Beauchemin and Simard 1999; Jalali and Jalali 2016; Plach et al., 2018). Notably, most of these studies have looked at annual discharge and P loads whereas this study differed by focusing solely on the NGS. Due to the high levels of discharge and event flow experienced within the NGS, this could explain why this study was found to be more chemostatically driven. Plach et al., (2018) conducted an analysis using the same sites from this study and found that plots receiving annual tillage were transport limited and plots receiving conservation tillage were supply limited. This study supports these findings regarding SRP, however suggests all plots were transport limited for TP. Plach et al., (2018) also found that concentrations of soil test P acted as a good indicator of the system, with higher concentrations of soil test P being associated with chemostatic controls and lower concentrations of soil test P being linked with chemodynamic controls. Although this study site had relatively low Olsen P and WEP, dairy manure was applied in the fall. This fall fertilizer application may have mimicked high soil test P resulting in a chemostatically driven system in the NGS. These results suggest that tillage and fall fertilizer application can both impact the chemodynamic and chemostatic response of the system.

The examination of manure application time is typically done with the comparison of fall vs. spring application, with results indicating greater P losses during the fall (Grande et al., 2005 and Liu et al. 2017). Few studies have been completed that examine the timing of fall manure application even though it is a common management practice for livestock operations. A study completed by Van Es et al., 2004 found that early fall manure application increased subsurface P loss. Furthermore, a consensus exists within the literature that an increase in time between manure application and the first rainfall event will decrease P loss in runoff (Allen and Mallarino 2008; Kleinman and Sharpley 2003; Vadas et al., 2011). While this remains true for P surface loss, this study suggests that the same consensus is not true for subsurface pathways. The results of this study demonstrated that total P loss in tiles over the NGS was not driven by the timing of the fall manure application or time between application and the first rainfall event. Alternatively, the first large hydrologic event to reach a minimum discharge threshold of 20mm largely determined the overall P loss for the NGS. Over the two study years with fall manure application this hydrologic event transpired at very different times, occurring in late-November and January for the 2014-15 and 2017-18 NGS respectively. Minimal losses in P were observed pre and post event, suggesting this large event acts as a flush of the system. The results of this study highlight the importance of in-field and long-term monitoring in addition to targeting BMPs that help to reduce the impact of the "first flush", such as incorporation.

4.5 Uncertainty within the Study

Field based studies are excellent in capturing naturally occurring spatial and temporal variability in the environment. However, due to heterogeneity within a landscape levels of uncertainty with results do occur. Regardless of the tiles being within the same field in addition to the tiles being side-by-side, flow variances between the tiles were observed.

These differences could be explained by changes of topography at the field scale in addition to slight differences in soil texture resulting in a variance of available water capacity. It was also assumed that contributing area to the tiles is 50% of the area on either side of the tile. However, due to the before mentioned factors this may not be the case.

Techniques and equipment for in-field monitoring must also be considered as areas of uncertainty with the potential to create bias within a dataset. This study used the method of pressure transducers corrected with barometric pressure to record water depths to then calculate flow. However, if there is backpressure in the tile the hydrograph will be skewed in addition to increasing the base flow volume. Backpressure can skew the tile precipitation coefficient showing higher levels of tile output compared to precipitation input (Martin 2015). In-field observations supported the likelihood of backpressure occurring within the tiles.

In the event that monitoring equipment is recording inaccurate field data, corrections to the dataset must be completed. In the case of correcting for backpressure, the goal is to correct to best estimate true flow. Because this is an estimate, it is impossible to know what the true flow is. This inevitably results in a certain degree of uncertainty during these impacted events. The correction method used creates different levels of uncertainty (Appendix B).

4.6 Bridging Science with Current Policy and Agricultural Conservation Programs

As discussed by Holmes and Clark (2008) there has been a movement by governments globally to make a push towards evidence-based policy making, particularly in the environmental sector. Nevertheless, disconnects between policy-makers and scientists still exist including the need for better question development, research communication, and analytical capacity (Holmes and Clark 2008). Regardless of previous research on knowledge mobilization for science and technology (Bonds 2011; Cash et al., 2003; Michaels 2009), policy decisions for environmental issues remain to be heavily based on political agendas and stake-holder influence rather than scientific evidence.

An industry where politics and stake-holders maintain a lot of influence is the agricultural sector. Unlike other industries that mainly deal with point-source pollution such as textiles and manufacturing, agriculture deals with diffuse sources of pollution making it more difficult to manage. Agriculture not only provides an essential service to our population but also makes up an important part of many countries' economies. Agricultural businesses can vastly range in their size and are often multi-generational. Therefore, a long-standing history of management practices stems from experience and familial knowledge. Arbuckle et al., (2015) examined farmer's perceived beliefs and risk on climate change and found that beliefs varied with trust of the source which had a significant direct effect on perceived risk of climate change. Unfortunately, government is not always seen as a trusted source for farmers, who are hesitant to change their existing management practices. Conner et al., (2016) completed a study in Vermont, USA examining farmer's willingness to adopt BMPs with government incentives and found that only a small proportion of farmer's participated in conservation programs concluding that incentive levels needed to be higher for more complex and lesser known BMPs.

There are currently several programs and legislative acts within Ontario that have the overarching goal of reducing phosphorus and improving water quality. These include the Nutrient Management Act (2002), the Clean Water Act (2006), the Great Lakes Protection Act (2015), and the Canada-Ontario Lake Erie Action Plan with the latter having a 40% P load reduction goal from 2008 levels by 2025 (Ministry of Agriculture Food and Rural Affairs 2020; Government of Canada 2018). An adaptive management strategy is a strong part of the Canada-Ontario Lake Erie Action Plan in order to reach the 40% reduction target. Therefore, the inclusion of the adaptation of agricultural BMPs is a very important component. Some of the recommended BMPs include cover crops, riparian buffer strips, tillage and nutrient application equipment modifications, and organic amendments to the soil. All of these BMPs are included in the Lake Erie Agriculture Demonstrating Sustainability (LEADS) cost-share program provided by the Ontario government (Ontario Soil and Crop 2020). The promotion and adaptation of these BMPs is an important piece to improving environmental sustainability within the agricultural sector. Nevertheless, most agricultural environmental programs and policies in Ontario remain under a guideline and voluntary framework which results in limited participation (Conner et al., 2016). This is of particular concern when dealing with factors or time periods that cause a greater risk of nutrient runoff to our freshwater bodies such as the NGS.

As demonstrated in this study in addition to the existing literature, manure applied during the winter has the greatest risk of P runoff compared to manure applied at other times of year due to snow melt and lack of plant uptake (King et al., 2015; Sharpley et al., 2003; Van Es et al., 2004). Thus, it is no surprise that many countries have an array of regulations and guidelines to manage winter manure spreading. However, these regulations and guidelines vary greatly between and even within a country (Liu et al., 2018). Following the European Nitrates Directive, there are “closed periods” throughout Europe that place a ban on winter manure spreading, with the Scandinavian countries having the most restrictive regulations (Liu et al., 2018). On the other hand, the United States, Canada, Australia, and New Zealand are controlled by their respective states/provinces/regions (Liu et al., 2018). Within Canada, Manitoba and Quebec ban manure spreading from Nov 10 to April 10 and Oct 1 to March 31 respectively. To make the transition of banning winter manure spreading easier, Manitoba provided government incentives to increase storage, and Quebec allows temporary field stockpiling of manure (Agriculture, Pecheries et Alimentation Quebec 2017; Liu et al., 2018). Nevertheless, many provinces and states only provide recommendations to be executed on a voluntary basis, including Ontario. A method used to make these recommendations is the P-index (PI) with the original PI being developed by Lemunyon and Gilbert (1993) to be a straight forward tool to assess P runoff risk at the field scale. This original PI is referred to as the additive approach, where the source and transport factors are graded then multiplied by a weighting factor and summed (Buckzo and Kuchenbuch 2007). Nevertheless, many versions of this original PI exist to include new research and specific factors of that area. Furthermore, a multiplicative approach along with an additive-multiplicative approach was developed to better account for interactions between factors (Buckzo and Kuchenbuch 2007). For example, the Pennsylvania PI was developed to use the multiplicative approach by multiplying source and transport factors. Many other countries have built on this Pennsylvania PI and added factors that are specific to their region. For example, the Norwegian PI takes into account precipitation, plant P (based on P release from freeze-thaw studies), subsurface drainage, leaching potential, and flooding frequency (Buckzo and Kuchenbuch 2007). However, it should be noted that as these PIs become more complicated the data becomes more difficult to collect. In contrast, Ontario's PI is based on the original PI from Lemunyon and Gilbert (1993) and follows an additive approach with the factors of soil erosion, P soil test, water runoff class, and fertilizer application rate, type, and method being considered (Hilborn and Stone 2015). Consequently, many factors such as subsurface tile flow, snowmelt, and plant P that make winter such a high risk P runoff period are not considered. Furthermore, only operations that require a Nutrient Management Strategy (NMS) which are farms that have livestock numbers greater than 300 nutrient units or farms located within 100m of a municipal well are required to use a PI (Doris et al., 2015). The current strategy to control winter manure spreading in Ontario is through a recommendation of no spreading from Dec 1 to March 31 in addition to a "peer-pressure" system where neighbours can call into Ontario Ministry of Agriculture and Rural Affairs (OMAFRA) when they spot someone spreading on frozen ground or snow. However, as demonstrated by Ontario's surrounding provinces as well as Europe, it is time to enforce stricter regulations on winter spreading within the province. This study along with the existing literature shows extensive support for the NGS being a high risk period. The current Ontario policies/guidelines are dated and highlight the lack of evidence-based policy decisions. If government environmental targets are to be reached including the 40% P reduction in Lake Erie, updating Ontario's PI to gear it towards risk factors associated with winter conditions is crucial. The existing research conducted within the province allows for customized evidence-based policy to mitigate Ontario's changing climate and risk factors. For example, this could include using an updated PI as a tool to create a zoned system where high and low risk areas are identified to regulate manure spreading in addition to the consideration of current climatic conditions. Of course with any new/updated policy, economic and social factors must be considered. The overwhelming scientific evidence including this study for the high risk period of the NGS cannot be ignored, and highlights the need for the continuation for these types of studies to support government in their evidence-based policy development.

Chapter 5
Conclusion

This study quantified P concentrations and loads in tile drains over the 2014-15 and 2017-18 NGS following fall applied dairy manure under a conventional deep till, conservation till, and conservation till with manure incorporation. The findings of this study have identified the fact that tillage and manure placement both impact P loss in tile drains. Tillage decreases runoff through tile drains, but only for the first 5 months following tillage. In general, for most events, tillage did not impact P loads in tile drains; however, for exceptionally large events in a given year, annual-till seems to increase P loss from tile drains, leading to an overall difference in annual losses. Thus, no-till can reduce P losses in tile drains. The incorporation of manure was found to lessen these losses further. However, the timing of when manure is incorporated does not appear to be as important since the impacts of this BMP do not occur until the first large flush event. This study highlighted the impact of the first large flush event on tile drain losses and demonstrates that the management of fields prior to the first flush is most relevant to annual P losses. This finding supports further investigation in order to build better management practices around this high risk hydrologic event.

Although differences were shown among the plots, this study has also shown the importance of natural differences in within-field hydrology that can complicate our ability to demonstrate treatment differences between adjacent plots. It was found that although the study plots were side by side within the same field, in-field hydrology still varied. Thus, the results of this study demonstrate the importance of in-field experiments in order to grasp and understand the complexity of naturally occurring spatial and temporal variance within a field, particularly over the NGS.

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Appendices

Appendix A

Correction Methods for Backpressure

Method 1 – Runoff Ratio > 1

Theoretically, input cannot be greater than output. This method provides a quick solution using runoff ratios (tile discharge/precipitation). When the event runoff ratio is greater than 1, this method's protocol is to make it equal to 1. Using contributing areas it is then possible to calculate flow. This method is quick however it does not take antecedent moisture conditions into consideration and also eliminates spatial heterogeneity among the tiles.

Method 2 – Max Flow equal to 0

By calculating flow using a rating curve a max flow rate is set using field slope in addition to tile roughness and diameter. Consequently, when backpressure occurs the flow data will plateau at this max flow rate. This method assumes when the max flow rate is reached backpressure starts and ends when the flow starts to decline. Therefore, the protocol for this method is to set flow equal to zero when max flow is occurring (Figure A1). This method is relatively quick and keeps the naturally occurring spatial heterogeneity among the tiles. However, it does remove true occurring max flow rates and does not consider the rising and falling limb that are affected by backpressure. Additionally, it assumes when back pressure occurs flow is equal to zero. Although flow would not be zero during backpressure it would be minimal.

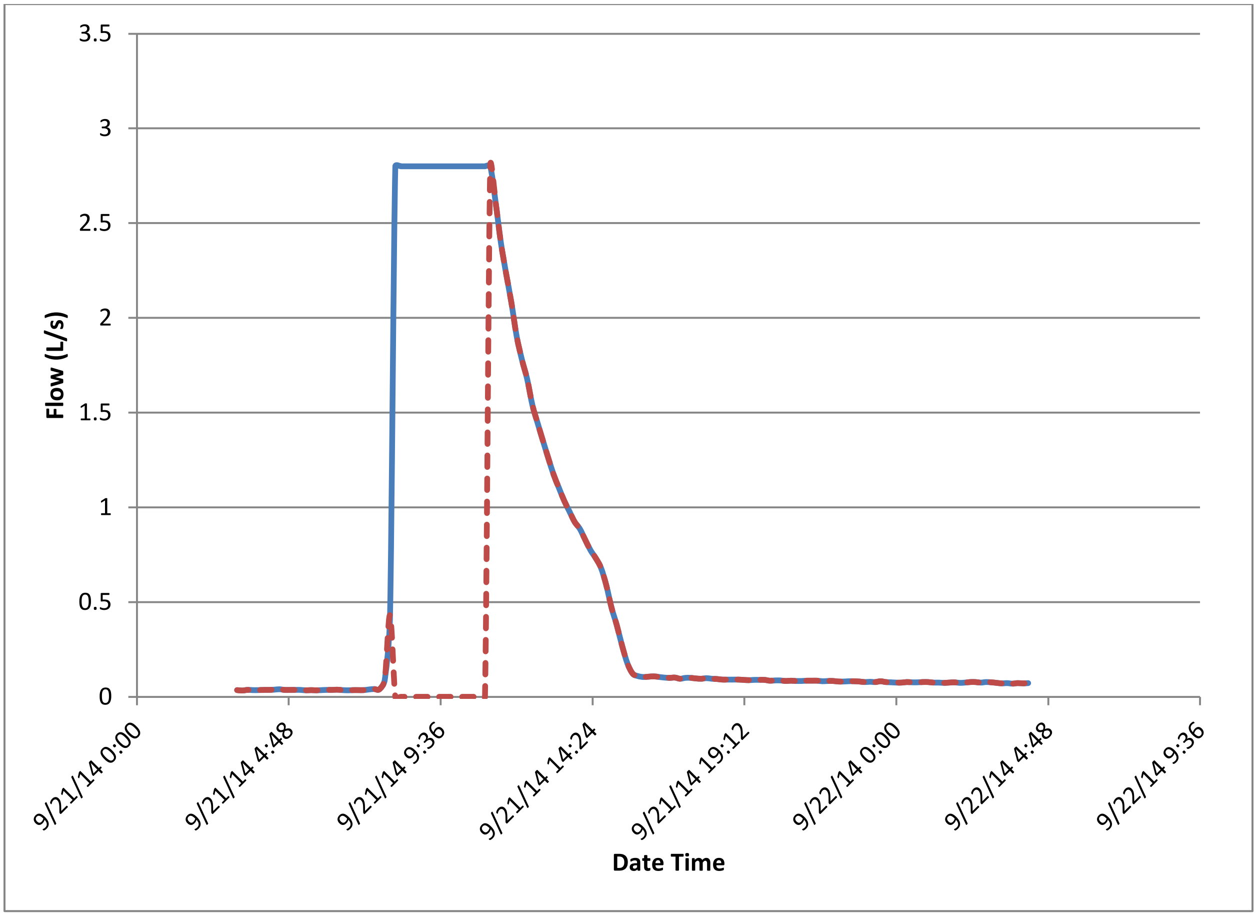

Figure A1. Original flow (blue line) and corrected flow (red dashed line) using the max flow equal to 0 backpressure correction method for event 3 of the CT-I tile during the 2014-15 NGS.

Method 3 – Manual Correction Using Sensor Depth Data

This method was developed from the previous max flow equal to 0 correction method. Like the previous method it uses the max flow rate to identify hydrologic events that may be experiencing backpressure. Once these events are identified it uses the raw sensor depth data to identify inflection points when backpressure starts and ends to make a more accurate estimate. The start inflection point is identified when sensor depth exhibits a sharp incline and the end inflection point is identified at the end of the plateau. It assumes that once the sensor depth has begun to fall true flow is occurring. During this identified period of backpressure, flow is made equal to zero (Figure A2). This method does a better job capturing the variability occurring among the tiles. However, still assumes when backpressure is occurring that flow is equal to zero.

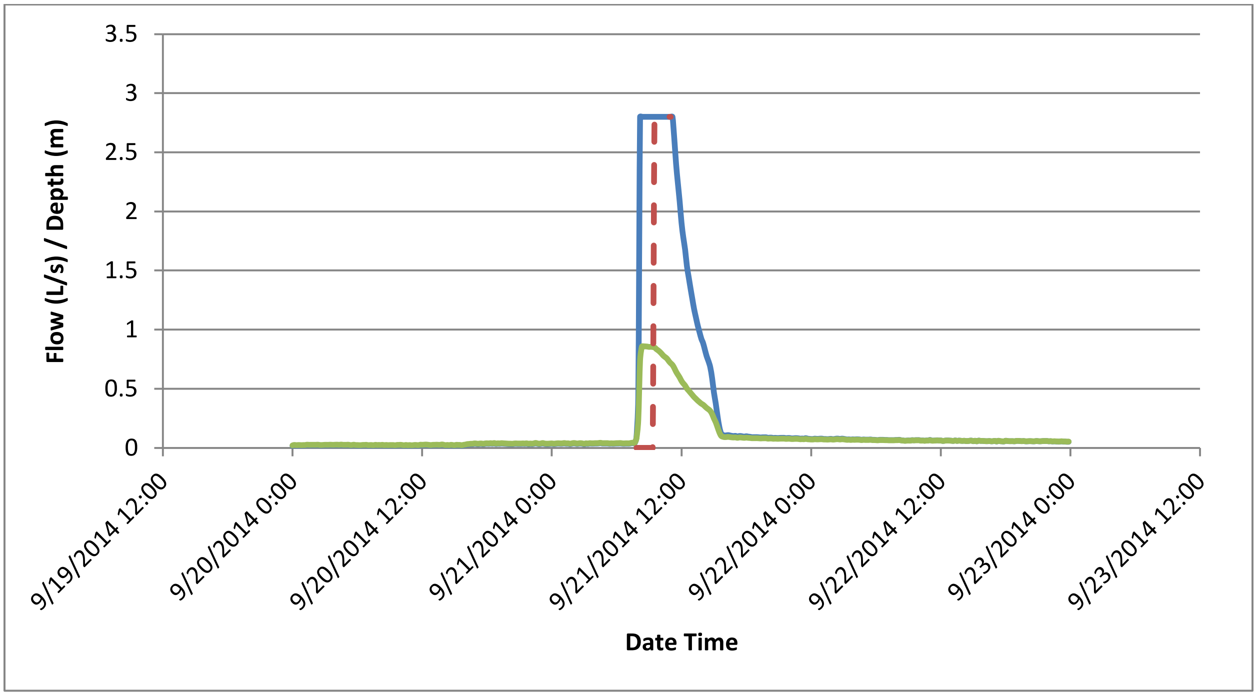

Figure A2. Original flow (blue line), sensor depth (green line), and corrected flow (red dashed line) using the manual backpressure correction method for event 3 of the CT-I tile during the 2014-15 NGS.

Method 4 – Linear Interpolation

The method is based on the procedure developed by Martin (2015). Because the DT tile rarely experiences backpressure, a ratio between the tiles can be used as the foundation to estimate peak flow in the tiles that experience back pressure. To determine this ratio, total volumes per hydrologic event are calculated and graphed between DT:CT-I and DT:CT. A linear trend line is added to determine the relationship between the tiles (Figure A3). Max flow rates are used to identify events that potentially experience backpressure. Events that display a plateau at max flow are designated as backpressure events. Using the peak flow of the DT tile that is not experiencing backpressure, the ratio is used to determine the corrected peak flow of the tile experiencing backpressure. Start and end inflection points are determined using flow data. The start inflection point is identified when flow exhibits a sharp rise to max flow and the end inflection point is identified when a more steady flow is observed. Corrected flow data was determined using a linear interpolation between the inflection points and the peak flow (Figure A4). This method creates a synthetic rising and falling limb. However, the ratio between the tiles is based on an insignificant linear relationship (Figure A3). Furthermore, it also assumes that the tiles behave similarly for small and large hydrologic events. Therefore, it does not reflect spatial and temporal variability among the tiles. Additionally, this method is unable to be used when all 3 tiles experience backpressure.

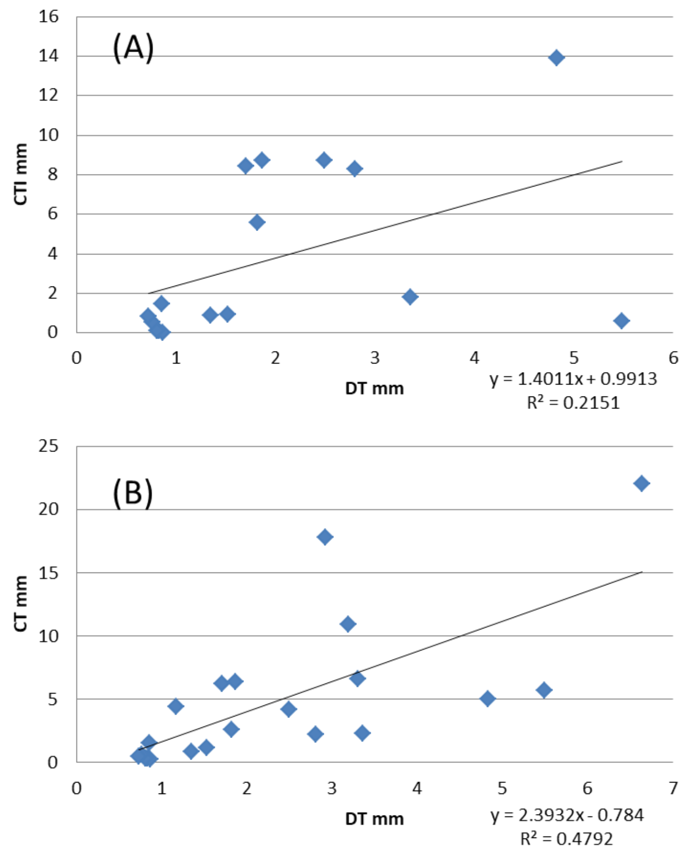

Figure A3. 2014-15 and 2017-18 DT against CT-I (A) and CT against CT (B) tile discharge during events that experience no backpressure. Linear trend line and equation displayed.

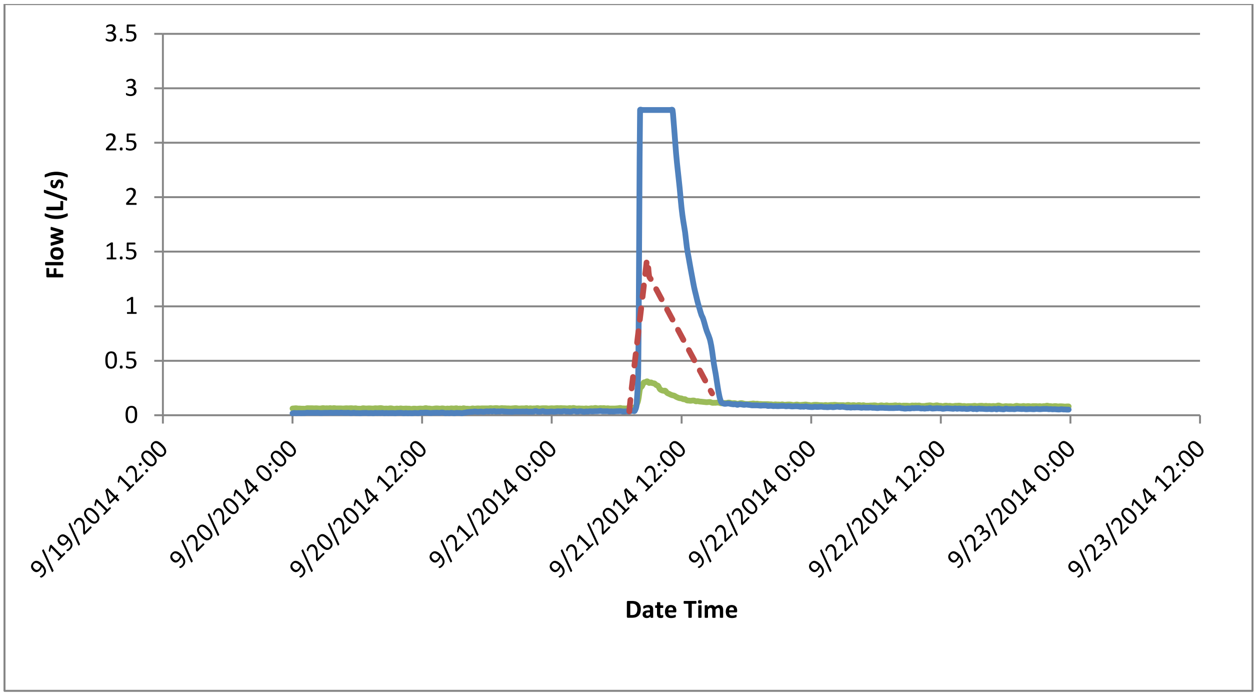

Figure A4. Original CT-I flow (blue line), original DT flow (green line), and corrected flow (red dashed line) using the linear interpolation backpressure correction method for event 3 of the CT-I tile during the 2014-15 NGS.

Appendix B

Backpressure Correction Method Discharge and Chemistry Comparison

Table B1. Event discharge volume (mm) using each backpressure correction method for all tiles during the 2014-15 and 2017-18 NGS. Events with underline indicate a change in discharge from the uncorrected data. (*) indicates the manual correction method was used due to all 3 tiles experiencing backpressure.

			Discharge														
			DT					CT-I					CT				
Season	**Event**	**Precip (mm)**	**Original**	**1:1**	**2.8 = 0**	**Manual**	**Linear**	**Original**	**1:1**	**2.8 = 0**	**Manual**	**Linear**	**Original**	**1:1**	**2.8 = 0**	**Manual**	**Linear**
2014-15	*1*	53	3	3	3	3	3	16	16	***6***	***7***	***7***	11	11	***5***	11	11
	2	69	3	3	3	3	3	25	25	***8***	***23***	***17***	18	18	***6***	18	18
	3	3	1	1	1	1	1	8	***3***	***3***	***6***	***3***	4	***3***	***4***	4	4
	4	3	2	2	2	2	2	8	***3***	***4***	***8***	***8***	6	***3***	***4***	6	6
	5	20	21	***20***	21	21	21	60	***20***	***18***	***47***	***55***	46	***20***	***26***	***37***	***57***
	6	16	2	2	2	2	2	9	9	***6***	9	9	4	4	***3***	4	4
	7	2	1	1	1	1	1	0.1	0.1	0.1	0.1	0.1	0.3	0.3	***0.2***	0.3	0.3
	8	2	1	1	1	1	1	0.01	0.01	0.01	0.01	0.01	0.2	0.2	***0.2***	0.2	0.2
	9	6	1	1	1	1	1	0.1	0.1	0.1	0.1	0.1	0.3	0.3	***0.2***	0.3	0.3
	10	12	2	2	2	2	2	9	9	***4***	9	9	6	6	***5***	6	6
	11	8	2	2	2	2	2	6	6	6	6	6	3	3	***2***	3	3
	SUM	193	39	***38***	39	39	39	140	***90***	***55***	***114***	***113***	99	***69***	***57***	***90***	***110***
2017-18	*1*	21	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
	2	13	1	1	1	1	1	1	1	1	1	1	2	2	2	2	2
	3	26	5	5	5	5	5	14	14	***10***	14	14	5	5	5	5	5
	4	2	3	2	3	3	3	2	2	2	2	2	2	***2***	2	2	2
	5	30	70	***30***	***61***	***46***	***46****	87	***30***	***46***	***41***	***41****	101	***30***	***71***	***54***	***54****
	6	25	5	5	5	5	5	1	1	1	1	1	6	6	6	6	6
	7	35	3	3	3	3	3	8	8	8	8	8	2	2	2	2	2
	8	16	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1

9	24	3	3	3	3	3	10	10	***4***	***7***	***4***	7	7	7	7	7
10	31	2	2	2	2	2	1	1	1	1	1	1	1	1	1	1
11	27	7	7	7	7	7	35	***27***	***7***	***23***	***17***	22	22	22	22	22
12	4	1	1	1	1	1	1	1	1	1	1	0.5	0.5	0.5	0.5	0.5
SUM	254	102	***61***	***93***	***77***	***77***	161	***96***	***83***	***100***	***91***	149	***79***	***120***	***102***	***102***

Table B2. Event runoff ratios using each backpressure correction method for all tiles during the 2014-15 and 2017-18 NGS. Events with underline indicate a change in runoff ratios from the uncorrected data. (*) indicates the manual correction method was used due to all tiles experiencing backpressure.

		Runoff Ratio														
		DT					CT-I					CT				
Season	**Event**	**Original**	**1:1**	**2.8 = 0**	**Manual**	**Linear**	**Original**	**1:1**	**2.8 = 0**	**Manual**	**Linear**	**Original**	**1:1**	**2.8 = 0**	**Manual**	**Linear**
	1	0.06	0.06	0.06	0.06	0.06	0.29	0.29	***0.11***	***0.13***	***0.12***	0.21	0.21	***0.10***	0.21	0.21
	2	0.04	0.04	0.04	0.04	0.04	0.36	0.36	***0.11***	***0.33***	***0.24***	0.26	0.26	***0.09***	0.26	0.26
	3	0.38	0.38	0.38	0.38	0.38	2.65	***1.00***	***1.09***	***1.94***	***1.09***	1.42	***1.00***	***1.18***	1.42	1.42
	4	0.50	0.50	0.50	0.50	0.50	2.49	***1.00***	***1.14***	***2.49***	***2.49***	1.82	***1.00***	***1.26***	1.82	1.82
	5	1.05	***1.00***	1.05	1.05	1.05	2.94	***1.00***	***0.89***	***2.30***	***2.70***	2.25	***1.00***	***1.27***	***1.83***	***2.79***
2014-15	*6*	0.16	0.16	0.16	0.16	0.16	0.55	0.55	***0.37***	0.55	0.55	0.26	0.26	***0.22***	0.26	0.26
	7	0.56	0.56	0.56	0.56	0.56	0.06	0.06	0.06	0.06	0.06	0.19	0.19	***0.15***	0.19	0.19
	8	0.58	0.58	0.58	0.58	0.58	0.00	0.00	0.00	0.00	0.00	0.14	0.14	***0.12***	0.14	0.14
	9	0.13	0.13	0.13	0.13	0.13	0.01	0.01	0.01	0.01	0.01	0.05	0.05	***0.04***	0.05	0.05
	10	0.16	0.16	0.16	0.16	0.16	0.76	0.76	***0.36***	0.76	0.76	0.56	0.56	***0.46***	0.56	0.56
	11	0.23	0.23	0.23	0.23	0.23	0.71	0.71	0.71	0.71	0.71	0.32	0.32	***0.27***	0.32	0.32
	SUM	0.20	***0.20***	0.20	0.20	0.20	0.72	***0.47***	***0.28***	***0.59***	***0.59***	0.51	***0.36***	***0.29***	***0.47***	***0.57***
	1	0.06	0.06	0.06	0.06	0.06	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
2017-18	*2*	0.07	0.07	0.07	0.07	0.07	0.11	0.11	0.11	0.11	0.11	0.12	0.12	0.12	0.12	0.12
	3	0.19	0.19	0.19	0.19	0.19	0.54	0.54	***0.40***	0.54	0.54	0.19	0.19	0.19	0.19	0.19
	4	1.60	1.00	1.60	1.60	1.60	0.85	0.85	0.85	0.85	0.85	1.08	***1.00***	1.08	1.08	1.08

5	2.32	***1.00***	***2.03***	***1.51***	***1.51****	2.88	***1.00***	***1.53***	***1.35***	***1.35****	3.33	***1.00***	***2.36***	***1.78***	***1.78****
6	0.22	0.22	0.22	0.22	0.22	0.02	0.02	0.02	0.02	0.02	0.22	0.22	0.22	0.22	0.22
7	0.08	0.08	0.08	0.08	0.08	0.24	0.24	0.24	0.24	0.24	0.06	0.06	0.06	0.06	0.06
8	0.05	0.05	0.05	0.05	0.05	0.03	0.03	0.03	0.03	0.03	0.04	0.04	0.04	0.04	0.04
9	0.14	0.14	0.14	0.14	0.14	0.42	0.42	***0.17***	***0.30***	***0.16***	0.28	0.28	0.28	0.28	0.28
10	0.05	0.05	0.05	0.05	0.05	0.03	0.03	0.03	0.03	0.03	0.04	0.04	0.04	0.04	0.04
11	0.25	0.25	0.25	0.25	0.25	1.30	***1.00***	***0.28***	***0.84***	***0.65***	0.82	0.82	0.82	0.82	0.82
12	0.18	0.18	0.18	0.18	0.18	0.20	0.20	0.20	0.20	0.20	0.12	0.12	0.12	0.12	0.12
SUM	0.40	***0.24***	***0.37***	***0.30***	***0.30***	0.64	***0.38***	***0.33***	***0.39***	***0.36***	0.59	***0.31***	***0.47***	***0.40***	***0.40***

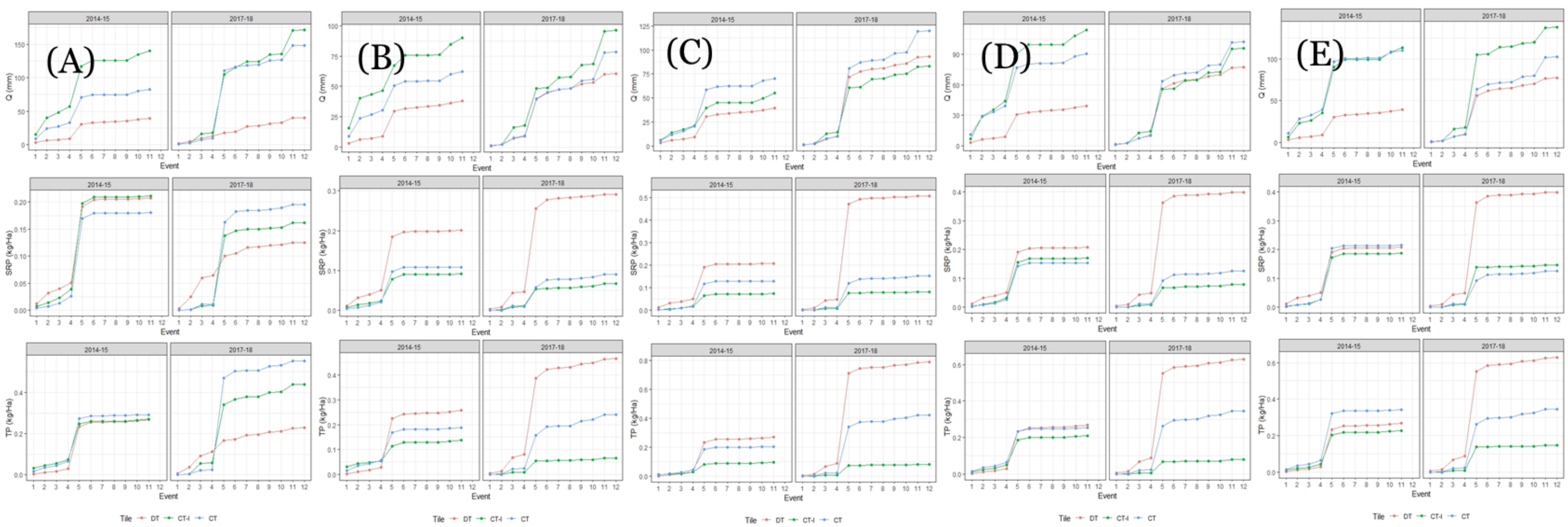

Figure B1. Cumulative event discharge, SRP load, and TP load for the original (A), 1:1 (B), 2.8=0 (C), manual (D), and linear interpolation (E) backpressure flow correction methods.