Runoff Response and Nutrient Loading in Vertisolic Clay Soils of Near-Level Artificially Drained Southern Manitoban Landscapes

by

Kokulan Vivekananthan

A thesis
presented to the University of Waterloo
in fulfillment of the
thesis requirement for the degree of
Doctor of Philosophy
in
Geography

Waterloo, Ontario, Canada, 2020

Examining Committee Membership

The following will serve on the Examining Committee for this thesis. The decision of the Examining Committee is by majority vote.

External Examiner: Dr. Christopher Spence, Watershed Hydrology and Ecology Division, Environment and Climate Change Canada, Saskatoon, Saskatchewan, Canada.

Supervisor: Dr. Merrin Macrae, Professor, Department of Geography and Environmental Management, University of Waterloo, Canada.

Internal Member: Dr. Mike Stone, Professor, Department of Geography and Environmental Management, University of Waterloo, Canada.

Internal-External Member: Dr. David Lobb, Professor, Department of Soil Science, University of Manitoba, Canada.

Internal-External Member: Dr. Genevieve Ali, Associate Professor, School of Environmental Sciences, University of Guelph, Canada.

Author's Declaration

This thesis consists of material all of which I authored or co-authored: see Statement of Contributions included in the thesis.

This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Statement of Contributions

This thesis is organized in accordance with the manuscript option and Chapters 2 through 5 either have been published (Chapters 2 and 3) or will be submitted (Chapters 4 and 5) for publication in a peer-reviewed journal. Appendix 1 is a review article that has been written by Kokulan Vivekananthan for the Canadian Agri-Food Policy Institute (CAPI) by incorporating the results of the current study into the broader literature. The published papers may vary from the chapters presented in the thesis based on the comments from the peer review process and advisory committee. Due to the manuscript format, some repetition between chapters in background and methodology may occur. Some discrepancies in figure formatting and abbreviations may also be possible as per the requirements of individual journals.

Kokulan Vivekananthan conceptualized, designed and executed the field and laboratory work, and wrote and edited the thesis. Dr. Merrin Macrae worked with Kokulan Vivekananthan in the conceptualization of the research, advised on methodological protocols, and edited the thesis chapters. Drs. Merrin Macrae, David Lobb, and Genevieve Ali all provided editorial comments on Chapters 2, 3, 4 and 5 and are listed as co-authors for the published manuscripts (Chapters 2 and 3). They will also be listed as the co-authors for the future submissions (Chapters 4 and 5) from this thesis. Dr. Matthew Morison and Brendan Brooks assisted with fieldwork and provided editorial comments for Chapter 4. Dr. Janina Plach and Dr. Matthew Morison assisted with laboratory work and provided editorial comments for Chapter 5. They will be listed as co-authors for the eventual submission of manuscripts arising from chapters 4 and 5 to scientific journals.

Chapter two is published as:

Kokulan, V., Macrae, M. L., Lobb, D. A., & Ali, G. A. (2019). Contribution of overland and tile flow to runoff and nutrient losses from Vertisols in Manitoba, Canada. *Journal of Environmental Quality*, 48(4), 959-965. DOI: 10.2134/jeq2019.03.0103

Chapter three is published as:

Kokulan, V., Macrae, M. L., Ali, G. A., & Lobb, D. A. (2019). Hydroclimatic controls on runoff activation in an artificially drained, near-level vertisolic clay landscape in a Prairie climate. *Hydrological processes*, *33*(4), 602-615. DOI: 10.1002/hyp.13347

Appendix A is published as:

Kokulan, V. (2019). Environmental and economic consequences of tile drainage systems in Canada. *Canadian Agri-food Policy Institute*. https://capi-icpa.ca/wp-content/uploads/2019/06/2019-06-14-CAPI-Vivekananthan-Kokulan-Paper-WEB.pdf

Abstract

The installation of tile drainage systems in Southern Manitoba has been accelerating over the past two decades to improve crop production. Given current environmental and political concerns related to agricultural pollution and the eutrophication of Lake Winnipeg, the role that tiles may play in both runoff and nutrient loading from agricultural fields must be evaluated because tiles can also have environmental consequences due to their capacity to export significant quantities of pollutants such as phosphorus (P) and nitrogen (N) from croplands by acting as subsurface lateral conduit pathways. This study examined surface and subsurface runoff from tiled and non-tiled fields on a farm in Elm Creek, Manitoba from 2015 to 2017 to quantify edge of field runoff and nutrient losses, to characterize surface-tile connectivity through the vadose zone, and to characterize ditch-overland flow dynamics at the edge-of-field. Water samples were collected from field surfaces, tile drainage, groundwater and roadside ditches during runoff events that occurred throughout the open water season. In addition, soil samples were collected in 2017 and analyzed for inorganic P fractions and P availability. This thesis has shown that overland flow was the major pathway for runoff and nutrient (P and N) edge of field losses, and the presence of the tile drainage did not decrease the frequent occurrence of the overland flow due to the prevailing climate conditions and vertisolic clays in the Red River Valley. Tile drains were responsible for 11-28% annual runoff losses, < 5% annual P losses and 40-50% annual nitrate N losses. Thus, although tile drainage did not exacerbate the edge of field P losses, it has the potential elevate N losses. Tile drainage was often activated from top-down water front movement and tile flow activation was hastened by higher rainfall intensities and wetter antecedent moisture conditions. Significant tile drainage predominantly occurred in late spring under wet antecedent conditions when the water

table was elevated. During such periods, the chemistry of tile drain effluent was similar to that of groundwater, which was low in P and high in N. In contrast, tile drainage in both early spring (snowmelt) and summer was small, although for different reasons. During snowmelt, when most runoff occurs in the Prairies, tile drainage was impeded by the presence of frozen ground and most runoff left fields as overland flow. Tile chemistry during this period reflected surface runoff, which was rich in P, indicating the presence of preferential flow through frozen ground. The chemistry of tile drainage was also rich in P and reflected surface runoff in summer, when rain fell on dry soils, also indicating preferential flow. Thus, although preferential flow between the surface and tile drains appears to have occurred in the vertisolic clays of the Red River Valley, it was associated with very small flow volumes and therefore small loads, whereas tile drain chemistry during periods when the majority of tile flow occurred resembled that of groundwater. This thesis has shown that tile drains will do little to modify water volume or chemistry during the snowmelt period, which dominates annual water cycles, due to the presence of frozen ground, and surface runoff will remain the greatest source of P loss from agricultural fields. This thesis has shown that some of the P loss from fields is due to direct losses from fields, but some may be mobilized during flooding due to water backing up in roadside ditches during snowmelt runoff, spring storms and massive thunderstorms. This suggests that an improved understanding of the role of ditch management on agricultural P loss is needed. This thesis has produced a comprehensive view of edge-of-field and in-field hydrochemical losses in tile drained fields in the Red River Valley. The outcomes of this thesis have implications for both water and nutrient management perspectives for farmers and policymakers.

Acknowledgements

The Ph.D. journey is a roller coaster ride. For me, it was challenging at times when I tried to balance my family, financial and academic responsibilities. This accomplishment could not have been possible if not the support of my family, friends, supervisor and the advisory committee and I thank every one of them from the bottom of my heart.

I would like to thank my best friend and beloved wife, Karthika for enormous support and encouragement throughout my post-graduate life. Karthika took additional responsibility in running the family especially during the later parts of my Ph.D. despite her own Ph.D. workload. This was not an easy task for two international graduate students in a foreign country and I am so glad that we have made to this point so far. I would also like to convey my heartiest thanks to my loving parents Vivekananthan Ponnuthurai and Yasothara Vivekananthan, sister Jeevika Vivekananthan and brother-in-law Jeyakumaran Chandrasekaram for your continuous support throughout this journey.

I would like to extend my deepest gratitude and sincere thanks to my adviser, Dr. Merrin Macrae, for her invaluable encouragement and thoughtful advice for the successful completion of my research. Her continuous evaluation, feedback and mentorship have not only helped me to successfully execute the tasks but also have improved my academic and professional skills.

I thank my advisory committee members, Dr. David Lobb and Dr. Genevieve Ali, for their support and guidance in the completion of research. I really appreciate Dr. David Lobb for recommending this Ph.D. opportunity to me. I also thank Dr. David Lobb and Dr. Genevieve Ali for providing field and laboratory support during my stay in Winnipeg for site establishment and sample collection.

This five-year-long journey has also helped me to earn some excellent friends. I really appreciate the invaluable support that I received from Dr. Matthew Morison throughout my Ph.D. program. I also thank my other lab mates Dr. Janina Plach, Dr. Will Pluer, Dr. Mazda Kompanizare, Vito Lam, Kirsten Grant, James Cober, Ryan Carlow, Karl Hanke, Stephanie Higgins, Dylan Price and Meredith Marshall for their timely help in the lab. I also appreciate the support of Brandon Brooks, Rob Ellis and Bo Pan, which enabled me to complete my laboratory analysis and fieldwork successfully. I am grateful to Anthony Buckley and Eva Slavicek for analyzing some of my water samples for chemical properties. My sincere thanks go to the summer students who had helped me in the field during the data collection period.

This Ph.D. is also an opportunity for me to familiarize with the Canadian farming practices. I thank Carl Classen and Joel Classen for not only allowing me to carry out my research on their farm but also educating me on various agricultural practices and operations.

Finally, I thank the Canadian Agri-food Policy Institute (CAPI) for selecting me as one of their Ph.D. scholars and providing an opportunity to produce two scientific articles on tile drainage and wetlands.

Land Acknowledgements

I acknowledge the traditional territory of Ontario's Aboriginal peoples, upon which we live and carry out our studies at the University of Waterloo. Specific to southwestern Ontario, I acknowledge the Anishinabek and Haudenosaunee people. I also acknowledge the territory in which this field research was conducted, in Winnipeg, Manitoba, the traditional territory of Anishinaabeg, Cree, Oji-Cree, Dakota, and Dene Peoples, and the homeland of the Métis Nation.

Dedication

I dedicate this thesis to my beloved daughter,

Aaranya Kokulan

Table of Contents

List of Figures

List of Tables

Chapter 1: Introduction

1.1. Introductory literature review

Agricultural runoff transports nutrients such as phosphorus (P) and nitrogen (N) that act as environmental pollutants in fresh and coastal marine ecosystems that cause eutrophication and subsequent algal blooms. Eutrophication is a current global and North American environmental problem as various large freshwater bodies, and coastal marine systems are affected (Smith, 2003; Schindler et al., 2012; Bol et al., 2018). In North America, freshwater lakes like Lake Erie of the Great Lakes, Lake Winnipeg in Manitoba and Lake Champlain in Quebec are affected by eutrophication, where agricultural runoff is a key driving factor (Levine et al., 2018; Watson et al., 2016; Schindler et al., 2012). Enhanced drainage may also increase downstream flooding risks (Rahman et al., 2014) and both overland flow (OF) and subsurface tile drainage systems contribute to high P and N edge of field losses (Christianson et al., 2016; Christianson and Harmel, 2015; King et al., 2015). However, the proportion of their contribution to runoff and nutrient losses vary depending on regional climate, soil and managemental factors. Understanding how agricultural nutrients travel through runoff pathways is essential for the successful implementation of countermeasures to mitigate the pollution of water bodies without compromising agricultural productivity.

Agricultural drainage is an inherent farm component which is essential to maintain crop productivity by removing excess moisture from the vadose zone (unsaturated soil profile). Natural farm drainage is facilitated by in-field surface swales which remove runoff water from upslope to downstream areas. However, a substantial proportion of croplands lack natural drainage due to lack of relief. In addition, existing natural drainage is also inadequate to drain excess water during major precipitation events that are expected to occur more frequently in the future due to climate change. Inadequate natural drainage prompts farmers to adopt artificial drainage systems in their croplands (Ritzema et al., 2006). Artificial farm drainage includes enhanced surface drainage through artificial swales and ephemeral drainage ditches. Artificial subsurface drainage systems include tile drains where perforated plastic or clay tubes are installed in the vadose zone. Tile drains reduce waterlogged conditions and the occurrence of overland flow by removing excess water from the vadose zone (thus enhancing water infiltration), improve soil aeration by keeping the water table at desired depth, facilitate enhanced crop growth, and extend the cropping and grazing seasons (King et al., 2015). However, enhanced farm drainage can also cause environmental problems despite increasing agricultural productivity. Thus, an improved understanding of the tile drainage and nutrient losses in different regions is needed.

1.1.2. Runoff pathways from fields

In general, runoff occurs either as overland flow (OF) or subsurface runoff, where OF can be partitioned into two components. Infiltration excess OF, also known as Hortonion overland flow (HOF), is initiated when the water input rate is higher than the soil infiltration rate (Horton, 1933). In contrast, saturation excess overland flow (SOF) occurs when the water table rises and exfiltrates at the soil surface. The water table position can be classified as either a permanent groundwater table (close to streams and in valley bottoms) or a perched water table that develops on a relatively impervious subsoil layer (such as fragipan or argillic layer) or bedrock (McDonnell, 2013).

Subsurface stormflow (SSSF) initiates when water moves through shallow permeable soil layers or on a soil bedrock interface from the upslope to downslope (Weiler et al., 2006). Like SOF, SSSF also occurs when the water input rate is lower than soil infiltration capacity. This subsurface stormflow may be amplified when there is a rise in lateral hydraulic gradient or an increase in lateral flow area or when the groundwater table rises to more conductive soil layers (Weiler et al., 2006).

Subsurface stormflow occurs as either homogeneous matrix flow or preferential flow (Weiler et al., 2006). Preferential flow may occur through lateral macropores (Beven and Germann, 2013) and soil pipes that form due to internal erosion (Weiler et al., 2006). In fine-textured soils, preferential flow pathways are dominated by macropores. Macropores are further classified as biopores and desiccation cracks, depending on their origin. Biopores include animal burrows, earthworm burrows and root channels, whereas desiccation cracks form due to an abundance of smectites (Beven and Germann, 2013). The diameter of macropores varies from a few millimetres in biopores (Heppell et al., 2002) to a few centimetres in desiccation cracks (Brierley et al., 2011). These wider openings allow water to rapidly percolate in large quantities (relative to capillary flow) with minimal interaction with soil matrix (Bishop et al., 2015). Therefore, preferential flow pathways act as conduits for water and pollutants to bypass the soil matrix. Nearly vertical earthworm channels, hydrophobic macropore walls and preferential movement of water towards the micro-depressions (where macropore openings could be found) can further amplify preferential flow through macropores (Edwards et al., 1992).

It is generally assumed that runoff is frequently generated from a small part of a watershed or landscape. This area contributing to runoff usually varies according to soil moisture storage (Betson, 1964). However, recent research suggests that these active runoff initiating areas must be hydrologically connected in order to contribute to a macroscale runoff response (Ambroise, 2004; McDonnell, 2013). This hydrological connectivity is controlled by climatic factors such as ecoregion and precipitation, landscape factors (e.g. slope), hillslope runoff potential, runoff delivery pathways, and lateral buffering (Bracken and Croke, 2007).

Agricultural landscapes differ from natural landscapes due to various human modifications like the usage of heavy farm equipment, intensive animal grazing, artificial irrigation and drainage systems and land management activities such as tillage and channelling. Therefore, runoff activation, generation and the factors that control hydraulic connectivity are distinct in agricultural landscapes from their natural counterparts (Deasy et al., 2009; Knox, 2001; Rittenburg et al., 2015; Skaggs et al., 1994; Zumr et al., 2015).

1.1.2.1. Hydroclimatic controls on runoff activation and generation from agricultural landscapes

1.1.2.1.1. Precipitation characteristics

Precipitation plays a critical role in runoff generation by providing the necessary input for the runoff process. In tropical regions, almost all the precipitation is received as rainfall, whereas in temperate regions, precipitation inputs are maybe either rainfall or snowfall. Runoff responses during rain and snowmelt are different, as rain provides higher kinetic energy than snowmelt (Bracken and Croke, 2007). Higher rainfall intensities can impact runoff response primarily in two ways. Hortonion OF is initiated when the rainfall intensity exceeds the infiltration capacity of surface soil. In general, maximum infiltration rates are observed in dry soils with water content values closer to the permanent wilting point (Dingman, 2015). Infiltration rates decrease with increasing soil moisture content and become minimal when the soil is saturated (Dingman, 2015). As such, for a similar intensity storm, an initially wet soil will initiate a quicker HOF response than if it were in a drier state (Zehe et al., 2005). Saturated (K_{sat}) and unsaturated soil hydraulic conductivities will vary with soil texture (Zehe et al., 2005), where sandy soils have a higher K_{sat} than clay soils. Threshold rainfall intensity values to generate HOF will also vary with evapotranspiration (ET). Dominance of HOF during high intensity, short duration summer thunderstorms have been observed in semi-arid regions such as the Canadian Prairies (Shook and

Pomeroy, 2012), in humid regions such as Southern Ontario (Van Esbroeck et al., 2016), and in semi-humid regions (Zehe et al., 2005). In farmlands that use heavy machinery, HOF may be initiated at the wheel tracks for low-intensity storms due to a decrease in soil infiltrability due to soil compaction (Deasy et al., 2009).

High-intensity rainfall also activates preferential flow pathways in macroporous soils (Edwards et al., 1992; Heppell et al., 2002; Stone and Wilson, 2006). High water input rates also could have allowed more macropore flow at the rate of rainfall intensity, thus limiting the interaction with soil matrix (Heathwaite and Dils, 2000; Heppell et al., 2002). In general, the relative contribution of macropore flow decreases with a wetting of soil column, at which point matrix flow also becomes substantial (Edwards et al., 1992; Heppell et al., 2002). As such, a quick tile response can be expected in a dry macroporous soil during a high-intensity storm event (Vidon and Cuadra, 2010; Stillman et al., 2006; Stone and Wilson, 2006).

Long duration, low-intensity rainfall events seldom contribute to the generation of HOF as their intensity is often lower than the K_{sat} of most soils. However, rainfall duration plays a vital role in the generation of SOF and SSSF. In general, SSSF pathways are activated prior to SOF. Saturation excess OF is initiated once the hydraulic capacity of SSSF been reached (Zumr et al., 2015) or when the water table rises to or above the surface. Significant contributions of matrix flow are observed during low-intensity, long-duration events, as the duration, provides enough time for the wetting front to move along the soil matrix (Vidon and Cuadra, 2010; Vidon et al., 2012). Contact time between water percolating along macropore walls and the soil matrix is also extended in long-duration events compared to high-intensity events, which eventually reduces the water conducting efficiency of macropores. Tiles may be activated depending on the magnitude of macropore flow and may continue to flow at sub-maximum capacity until the water table rises to tile depth from below (bottom-up) or the wetting front (matrix flow) reaches the tile depth (top-down). Tiles significantly limit the initiation of SOF during low-intensity events by removing excess water from the vadose zone and by prohibiting the water table from rising to the surface (Deasy et al., 2009; Zumr et al., 2015).

1.1.2.1.2. Antecedent moisture conditions

While some work has found rainfall characteristics as dominant as runoff generating factors, other studies have identified antecedent soil moisture as the crucial condition for the generation of different runoff responses for a single or for a series of rain events (Hardie et al., 2011; Macrae et al., 2007 and 2010; Vidon et al., 2012; Weiler et al., 2006). Many of these hydrologic studies have used surrogate measurements such as antecedent stream or tile discharge (Heppell et al., 2002; Macrae et al., 2010), antecedent precipitation (Macrae et al., 2010; Vidon and Cuadra, 2010), antecedent water table depth (Heppell et al., 2002; Macrae et al., 2010; Vidon and Cuadra, 2010) and season (Macrae et al., 2010) instead of direct soil moisture measurements. However, most of the time, runoff response to antecedent moisture conditions was nonlinear in nature (Macrae et al., 2010) and varied between different storms (Macrae et al., 2007), seasons (Macrae et al., 2007) and years (Vidon et al., 2012). For example, tile discharge was positively increased with stream discharge during wet conditions in a Southern Ontario agricultural catchment ($r^2 = 0.58$; Macrae et al., 2007). However, this relationship was not observed (poor) when the basin was in a dry state. Results of these studies indicate the existence of a specific threshold antecedent moisture value for every runoff generating field or watershed (Macrae et al., 2010; Zehe et al., 2005). This could also be related to the saturation of the soil matrix, rise of the water table into more conductive soil layers, or to the activation of new preferential flow pathways (Macrae et al., 2010; Weiler et al., 2006; Zehe et al., 2005). However, current research is still inconclusive on which, when, where and how these threshold controls impact runoff generation.

Uncertainty also exists regarding the activation and function of macropore pathways in various antecedent soil moisture conditions (Nimmo, 2012). While some studies suggest greater water flow through biopores (Beven and Germann, 1982) and desiccation cracks (Heathwaite and Dils, 2000) during wet antecedent moisture conditions, other studies have reported higher infiltration in desiccation cracks during dry antecedent soil moisture conditions (Hardie et al., 2011; Grant et al., 2019b). Infiltration in cracked clays was observed to be decreasing during wet antecedent moisture conditions as the desiccation cracks close due to swelling (Hardie et al., 2011). However, the occurrence of macropore flow through cracks at deeper depths whilst surface cracks were closed has also been reported (Baram et al., 2012; Greve et al., 2010).

1.1.2.1.3. Winter runoff processes

Hydroclimatic drivers also play a crucial role in determining the magnitude of winter and spring snowmelt runoff, which is common in temperate regions. For example, rain on snow events accelerate snowmelt either by providing additional energy to melt the snowpack or by releasing latent heat while freezing. Once the snowpack begins to melt, a thawing front will move through the soil profile. The rate of this thawing front advance is determined by supply factors such as snow available to melt (SWE), incoming radiation (Hayashi et al., 2003), and soil conditions such as the infiltrability of soil, K_{sat} of soil, depth of the frozen layer, and soil water/ice content (Gray et al., 2001).

In general, the initial meltwater infiltration rate is controlled first by the pace of the thawing front advance, and then by the infiltration capacity of subsoil (Hayashi et al., 2003). Hortonion OF is initiated if the melting rate is higher than the thawing rate of the frozen soil layer (Shanley et al., 2002). This is a common scenario in the Canadian Prairies, where the soil can freeze down to 1m depth (Hayashi et al., 2003). Rapid infiltration through frozen macropores was also observed in frozen soils despite limited surface infiltration where surface water was able to bypass shallow soil-ice layers, which ranged from 15 to 30-cm (Demand et al., 2019; Grant et al., 2019a). Percolated water was further transported to deeper layers through open earthworm burrows or root channels in the subsurface (Demand et al., 2019). In contrast, seasonally frozen soils like vertisols of Red River Valley usually develop a thicker soil-ice layer (75-100-cm) and often lack biopores. However, water movement through seasonally frozen vertisols has yet to be assessed.

In humid temperate regions such as Southern Ontario, initial snowmelt in the early thaw season saturate the soil profile (Van EsBroeck et al., 2016). Subsequent snowmelt events favour the activation of SOF pathways. If the site is tile-drained, initiation of SOF will be delayed until tiles reach their maximum discharge (Van EsBroeck et al., 2016). Continuous tile discharge has been observed during winter months in humid temperate regions (Heathwaite and Dils, 2000; Macrae et al., 2007). In contrast, snow only melts during late winter or early spring in much colder regions such as Red River Valley of the North. There, HOF over frozen soils is favoured during annual snowmelt. There is also a possibility for rapid infiltration through macropores during snowmelt in regions like the Red River Valley. This flow through frozen macropores may also contribute to substantial early spring tile flow. However, this has not been assessed yet.

Snowmelt infiltration in macroporous soil also depends on antecedent moisture conditions before freezing. In antecedent saturated conditions, macropores could be filled with water that eventually turns into ice following the progression of the freezing front into the soil. This requires higher energy inputs to melt the ice, which could delay thawing front propagation. However, an early breakthrough in the snowmelt infiltration rate could be observed through initially dry macropores filled with air (Hayashi et al., 2003; Nimmo, 2012). In the meantime, rain on frozen ground may generate large OF response due to reduced infiltrability of surface soil (Shanley et al., 2002).

1.1.3. Agricultural nutrients in runoff

As described above, surface and subsurface runoff partitioning is determined by various hydroclimatic drivers, soil types and management factors. The activation of different runoff pathways also determines the quantity of sediments, agrochemicals, and microbial organisms in the runoff from agricultural landscapes. In addition, the amount of pollutants in runoff is also determined by the interaction between soil and contaminants. Agricultural runoff has been a source of sediments (Uusitalo et al., 2001), agricultural nutrients such as phosphorus (P) and nitrogen (N) (Pease et al., 2018a), insecticides (Challis et al., 2018), herbicides (Stone and Wilson, 2006) and micro-organisms such as *E. coli* (Clarke et al., 2017) to downstream systems. The following sections will discuss the dynamics of agricultural nutrients (P and N) in agricultural soils and runoff.

1.1.3.1. Phosphorus

In agricultural fields, P is mobilized from its source material via various biological and chemical mechanisms in either particulate or soluble forms (Withers and Haygrath, 2007). Mobilized P will be transported from field-scale delivery mechanisms such as overland flow (OF), subsurface stormflow (SSSF), and tile flow, to the edge of the field, and subsequently to a first order stream or to a drainage ditch. From there, it will be further transported through larger order streams before terminating at a larger waterbody (Withers and Jarvie, 2008). Along transport pathways, P is subjected to various exchange processes such as deposition, adsorption and desorption by streambed sediments, uptake and release by benthic algae and vegetation, and mineralization of organic matter before becoming available to phytoplankton (Withers and Jarvie, 2008). Hydrologic controls on P mobility and transport vary depending on soil characteristics and land management practices.

1.1.3.1.1. Hydroclimatic drivers on runoff P losses

Phosphorus in OF and tile flow varies depending on precipitation characteristics, landscape and land management. In general, higher intensity rainfall events lead to substantial total phosphorus (TP) losses in OF (Panuska and Karthikeyan, 2010). Dominance of particulate P (PP) during OF producing rainfall events has been reported (Panuska and Karthikeyan, 2010; Uusitalo et al., 2003). In addition, the type of OF also influences P speciation in runoff. For example, Hortonion overland flow (HOF), which is generated relatively quicker than saturation excess overland flow (SOF). As such, HOF may transport more soil colloids and sediments that are mainly associated with PP and are dispatched by the rain splash kinetic energy. Saturation OF generation is relatively slow, and the contact time with soil matrix and soil surface is longer. This leads to lesser detachment of sediments and colloids, thus resulting in more dissolved P fractions and less PP. When tile drainage is present, much of this dissolved P could be re-routed into tiles.

Large rainfall events also generate higher tile discharge and have been associated with greater TP exports (Gentry et al., 2007; Lam et al., 2016a). Greater TP concentrations were observed in tile discharge during summer compared to winter (Heathwaite and Dils, 2000; Macrae et al., 2007). This could have associated with relatively low flows in summer, fertilizer applications, more crack openings, and animal grazing. Antecedent soil moisture is also likely to be a determinant of event-scale and seasonal-scale variations of tile P discharge, by mediating increase/decreases in hydraulic connectivity, the activation and deactivation of preferential flow paths, and the availability of source P by influencing microbial mineralization rates (Macrae et al., 2010).

Substantial runoff P losses occur during the non-growing season in temperate regions with lengthy winter periods (Pease et al., 2018b; Plach et al., 2019; Tiessen et al., 2010). However, partitioning of runoff between surface and subsurface pathways depends on winter precipitation characteristics and soil-ice layer development. For example, tile drainage has been shown to be responsible for the majority of runoff and P losses during non-growing seasons in low latitude North American regions, such as Ohio, which experience mild winter conditions (Pease et al., 2018b). Tile drainage was also responsible for the majority of the non-growing season runoff losses in mid-latitude regions like Southern Ontario, where substantial snowfall occurs, but the development of a thick soil-ice layer is unlikely (Plach et al., 2019). However, the majority of non-growing season TP losses in regions like Southern Ontario have often been associated with episodic winter melt runoff events with substantial overland flow. In areas like the Red River Valley, further north, overland flow has been the major pathway for both runoff and P losses due to thick soil-ice layer during snowmelt runoff and the absence of tile drainage in general (Tiessen et al., 2010). How tile drains will alter the hydrology and biogeochemistry of winter runoff processes is yet to be explored in the higher latitude agricultural regions like Red River Valley.

1.1.3.1.2. Soil characteristics on runoff P losses

Soil texture affects runoff P concentrations by influencing P sorption capacity and soil permeability. Clay soils retain more P due to their high surface area compared to sandy soils (Berg and Joern, 2006; Pizzeghello et al., 2011; Ulen et al., 2011). In addition, higher soil organic matter content of clay soils further facilitates P retention. Also, clays have lower hydraulic conductivities than sandy textured soils, resulting in the potential for delayed tile responses in clay soils relative to sandy soils.

Overland flow runoff and P losses could be substantial in clayey soils due to their lower hydraulic conductivities. In addition, preferential flow through macropores could also contribute to higher P losses from the finer textured soils (Addiscott and Thomas, 2000; Grant et al., 2019a). Washed out P sediments and colloids quickly reach tiles through macropores. Various studies have reported substantial P losses in tile flow in a range of different fine-textured soils such as loams (Jameison et al., 2003), clay loams (Heathwaite and Dils, 2000), silt loams (Vidon and Cuadra, 2011), clays (Djodjic et al., 2000), silty clay (Hooda et al., 1999) and heavy clays (Uusitalo et al., 2001; Turtola and Jaakola, 1995) with greater contributions of PP as a fraction of TP. However, P losses through tile drainage in vertisolic soils with dynamic shrinking and swelling potential have not been assessed yet.

Flow through the soil matrix is generally dominated by dissolved P forms. As such, equal or greater contributions of dissolved P to initial tile flow can be observed in a macroporous soil when the antecedent moisture content is high (Gentry et al., 2007). This also explains relatively higher soluble reactive P (SRP) concentrations in winter and spring tile flow when compared to summer (Macrae et al., 2007). Dominance of dissolved P fractions also has been observed in tile flow from the coarse textured soils such as sand (Liu et al., 2012) and sandy loam (Eastman et al., 2010; Lam et al., 2016a) where macropore activity is minimal.

1.1.3.1.3. Management aspects on runoff P losses

Apart from hydroclimatic drivers and soil characteristics, runoff P losses are also influenced by management practices such as fertilization and tillage operations. Fertilizer source, rate, timing and placement have a significant impact on runoff P losses. For example, animal manure application has resulted in elevated overland and tile flow P losses when compared to inorganic fertilizers (Sweeney et al., 2012; Kinley et al., 2007). In general, manure is applied according to crop N requirement. This results in the over application of P, far exceeding crop requirements (Randall et al., 2000). In addition, some farmers apply manure during the fall or off-season due to insufficient storage (Sharpley et al., 2001a). This may lead to more P runoff loses during non-growing seasons. Also, organic P in manure is less readily adsorbable by soil retention sites than inorganic P, and thus more available for leaching. Increased tile dissolved P losses with continuous pig manure (Kinley et al., 2007) cattle manure (Macrae et al., 2007; Randall et al., 2000) and poultry manure (Kinley et al., 2007) application have been reported. Likewise, increased fertilizer application rates could also result in elevated runoff P losses. Fertilizer timing is also critically important as major rainstorms following fertilizer application could contribute to greater P losses (Vadas et al., 2011). Phosphorus application methods also impact incidental P losses. For example, broadcasting is found to cause more P leaching than incorporation (Kleinman et al., 2009).

Greater P runoff losses also occur due to elevated soil test P (STP) contents of topsoil, which is primarily influenced by land management activities (Duncan et al., 2017). Overland flow P concentrations gradually increase with surface STP content (Wilson et al., 2019). However, elevated P losses occur once the P saturation capacity of soils is overwhelmed (Kleinman et al., 2000; Vadas et al., 2005). Even though the P loading in tile does not increase with manure application at lower STP values, it rapidly increases once soils have exceeded a particular STP threshold (Klatt et al., 2002; McDowell and Sharpley, 2001).

Tillage is a farm operation that is carried out to loosen soil aggregates in order to improve root growth and soil aeration. However, conventional tillage methods have yielded substantial overland flow P losses (Tiessen et al., 2010). Therefore, reduced tillage (conservation tillage) techniques were adopted to offset the soil and P loss from overland flow during conventional tillage. However, conservation tillage has been found to increase subsurface P losses by preserving surface macropore networks, especially when the crop residues were left after the harvest and soil was not plowed (Djodjic et al., 2000; Kleinman et al., 2009). Frequent freeze and thaw cycles increase P losses from crop residues that are left in the field throughout the winter season and favour the development of cracks (Djodjic et al., 2000; Liu et al., 2012). However, preferential flow related tile P losses in no-till systems can be reduced by incorporating P fertilizers at the subsurface (Williams et al., 2018a).

In addition to fertilization and tillage, crop type (King et al., 2016; Turtola and Jaakola, 1995), crop rotation (Kinley et al., 2007), cropping systems (Oquist et al., 2007), tile dimensions such as tile depth and spacing (Morrison et al., 2013) and surface inlets (Coelho et al., 2012) can also affect edge of field P losses and surface-subsurface P partitioning.

1.1.3.2. Nitrogen

Nitrogen (N) is another essential nutrient that controls the primary productivity of ecosystems. Natural N cycle involves atmospheric deposition, N fixation, nitrification and denitrification processes through atmosphere, land and oceans where N is transformed in between its reactive (NH_3, N_2O and NO_3-N) and un-reactive forms (N_2) (Coskun et al., 2017; Gruber and Galloway, 2008). However, addition of synthetic N fertilizers to croplands in large volumes, and emission of N gases through fossil fuel burning, have substantially altered this natural cycle and have caused negative consequences like poor drinking water quality, freshwater eutrophication, coastal ecosystem destruction, adverse air quality, and climate change (Erisman et al., 2013; Gruber and Galloway, 2008). Similar to P, several soil, hydroclimatic and managemental factors determine the amount of N leached or lost though runoff from agricultural lands (Cameron et al., 2013; Christianson and Harmel, 2015)

1.1.3.2.1. Hydroclimatic drivers on runoff N losses

Greater N surface runoff losses have been observed during higher intensity rainfall on wetter soil conditions (Kleinman et al., 2006; Liu et al., 2014). However, NO_3-N concentrations were found to be diluted during greater surface runoff volumes potentially due to source exhaustion (Kleinman et al., 2006). Similarly, dilution of NO_3-N concentrations was also observed in tile drainage during major rainfall events (Cuadra and Vidon, 2011).

Climate also affects the edge of field N losses by affecting crop-soil interactions. Like P, more N is available for runoff losses during non-growing seasons, where there is no crop uptake and more drainage (Cameron et al., 2013). Likewise, severe rainstorms or snowmelt following a dry summer also elevate N losses due to poor crop growth and uptake.

1.1.3.2.2. Soil characteristics on runoff N losses

Nitrate ions poorly interact with soil particles (Cameron et al., 2013). Therefore, NO_3-N movement within a soil profile is governed by solute transport mechanisms. Nitrates are either transported through convection following mass movement of water, diffusion through NO_3-N concentration gradient or through hydrodynamic dispersion. It is generally assumed that clayey soils are less prone to N leachate losses than sandy soils due to slower water movement and denitrification processes. Influence of preferential flow paths on NO_3-N leaching largely depends on the location of nitrates (Cameron et al., 2013). For example, in soils with macropores, NO_3-N can also be preferentially transported if they are present in infiltrating water. However, NO_3-N may not be available to be preferentially transported if they are present within soil aggregates. In addition, soils higher in organic matter were also found to be elevating runoff N through increased mineralization (Randall and Mulla, 2001). When present, tile drains exacerbate nitrate losses by intercepting subsurface leaching.

1.1.3.2.3. Management aspects on runoff N losses

Runoff N losses are also related to fertilization. For example, increased N fertilizer application rates have been positively associated with runoff N concentrations and N losses (Christianson and Harmel, 2015). Poor crop N use efficiency, as well as not accounting for soil N mineralization, also impact fertilizer rate related N losses (Coskun et al., 2017; Dinnes et al., 2002). In addition, N runoff losses can be exacerbated if manure or fertilizer is applied during wetter periods (i.e. early spring) or on frozen soils (i.e. winter) when the crop soil N utilization rate is generally lower (Dinnes et al., 2002).

Considering the gravity of agricultural water quality issues and heterogeneity in agricultural systems with respect to regional climate, soils and management, various studies have assessed different aspects of runoff-nutrient dynamics from artificially drained agricultural landscapes at different scales ranging from research plots to climatic regions (Christianson et al., 2016; Christianson and Harmel, 2015). However, these aspects have been sparsely looked at in regions like Red River Valley of the North where historical weather patterns have been changing, and new water management practices such as tile drainage are expanding.

1.2. Research Problem, objectives and approach

1.2.1. Water quality problems in the Red River Valley of the North

The Red River Valley (RRV) occupies a vast area of central North America, including the provinces of Manitoba and Saskatchewan of Canada, as well as States of Minnesota, North Dakota and South Dakota of United States, from which the Red River of the North flows northward to eventually draining into Lake Winnipeg. The climate of the RRV is generally characterized by short warm summers and long cold winters with the development of a thick soil-ice layer (Stoner et al., 1993). Precipitation varies from winter snowfall to low-intensity long-duration spring storms to high-intensity short duration summer thunderstorms. Snowmelt occurs in early spring, and mid-winter snowmelt runoff events are rare.

There are two distinct types of terrain that can be seen in the RRV, nearly flat plain and rolling landscapes, respectively (Rahman et al., 2014). Nearly flat terrain with glaciolacustrine lake sediments with high clay contents dominates the majority of the RRV basin where the ancient Lake Agassiz existed (Stoner at al., 1993). The occurrence of frequent wetting and drying cycles with a net moisture deficit in a nearly flat terrain with fine, clay-rich glaciolacustrine deposits favour the development of vertisolic soils (Brierley et al., 2011). As such, a substantial proportion of this flat terrain is dominated with vertisolic clays, which are characterized by slickensides, high clay content (> 60%) and formation of desiccation cracks (Brierley et al., 2011).

Historically, the RRV landscape favoured runoff generation mainly as overland flow due to its nearly-flat terrain and clayey soil. The majority of annual overland flow occurs as snowmelt runoff over frozen or partially frozen soils in early spring (Gray et al., 2001; Shook et al., 2015). The spring freshet has been the major annual runoff event, accounting for the majority of annual P and N losses (Salvano et al., 2009; Tiessen et al., 2010; Rattan et al., 2019). Although Hortonion overland flow is often initiated during high-intensity summer storms, contributions of summer runoff to annual runoff and groundwater recharge are minimal (Chen et al., 2004). However, climatic change within this region is resulting in increasing frequency of multiday spring and summer rainfall events (Dumanski et al., 2015; Shook and Pomeroy, 2012). In addition, the frequency of mid-winter/early spring rain on snow events is also expected to increase (Jeong and

Sushama, 2018). Several water bodies in the region, including Lake Winnipeg, the 10^{th} largest freshwater lake in the world, have been severely affected by algal blooming and eutrophication.

Multiday spring and summer storms often leave excess moisture on agricultural farm fields and can damage the emerging crops. Expansion of tile drainage is occurring within the RRV region as a means to remove this excess moisture from agricultural lands. However, the potential of tile systems to de-water the soil profile has not been assessed in the vertisolic soils of the RRV (Kokulan, 2019). Studies conducted on cracking soils outside the RRV imply that flow through desiccation cracks could exacerbate P losses to surface bodies (Grant et al., 2019a; Turtola and Jaakkola, 1995; Uusitalo et al., 2001). However, this aspect of tile drainage as a pathway for nutrient losses has not been evaluated in the RRV.

In the RRV, rain on snow events have the potential to induce spring floods with multiday flooding of agricultural fields (Schindler et al., 2012). These flooding events can further exacerbate the edge of field losses either by increased contact time with soils (Tiessen et al., 2010) or by releasing P into floodwater through reductive dissolution reactions in potential anoxic conditions (Amarawansha et al., 2015). However, the tendency of these flooding events on runoff P dynamic has not been assessed in field settings.

1.2.2. Objectives

The objective of this thesis is to understand the hydroclimatic and biogeochemical processes that govern runoff and nutrient dynamics from artificially drained agricultural fields of RRV with vertisolic clays. The specific objectives are to:

1) evaluate the relative contributions of tile drains and surface overland flow to edge-of-field runoff and nutrient losses

2) assess the hydroclimatic controls (season, antecedent conditions and rainfall characteristics) on runoff generation and flow paths (overland flow, tile drainage and groundwater table) in a near-flat vertisolic clay landscape

3) characterize the activation of preferential and matrix transport in the vadose zone in vertisolic Prairie soils and determine if and how this may vary with soil antecedent moisture

and to

4) characterize the occurrence and frequency of flooding events and to determine if they contribute to increased edge of field P losses

This thesis is organized as chapters to address the overview of the thesis topic and specific sections. Chapter 1 provides an overview of the processes that drive runoff and biogeochemical processes and the general approach and methodology of this study. Chapters 2-5 address the specific objectives of this thesis (1 to 4, respectively). Chapter 6 summarizes the main conclusions of the thesis. Appendix 1 incorporates findings from the current study with existing literature and provides an overview of runoff processes and nutrient dynamics in Canada and the RRV region.

1.2.3. General approach and methodology

1.2.3.1. Site description

This study was conducted in two adjacent farm fields (Figure 1.1) in Elm Creek, Manitoba. Both fields are underlain by Gleyed Humic Vertisols of the Red River Series (U.S taxonomy: Gleyic humicryerts), and the topography is characterized by nearly flat terrain (0-2 % slope). Both fields drain into adjacent roadside ditches through in-field surface swales. However, flow in the roadside ditches become retarded and temporarily backs up into fields during major snowmelt and runoff events.

One of the fields was both surface and tile-drained, whereas the other field was only surface drained. In the tile-drained field, tiles were installed in 2012 using Real-Time Kinematic (RTK) and differential global positioning system (DGPS) and a soil-max gold digger stealth ZD tile drainage plow. Tile drain laterals (10 cm diameter) were systematically placed at ~ 1 m depth, with 13 m spacing and were connected to a 37.5 cm diameter header tile. Tiles do not drain into the roadside ditches in this site. Instead, the header tile drained first into an initial “receiving unit” in the middle of the field, where water was subsequently pumped into a larger retention pond (75 × 75 × 3 m), located adjacent to the receiving unit. In 2015, the initial ‘receiving unit’ was an open collection pond that was pumped manually into the retention pond, managed by the landowner. In 2016, this collection pond was replaced with an automatic lifting station that was a cylindrical storage unit that collected water from the tile and then pumped it into the retention pond. However, manual pumping was also used in 2016 and 2017 when there was a lifting station failure.

Both fields were managed similarly in 2015, 2016 and 2017 with respect to crops, fertilizer application and tillage. The farm follows a canola (*Brassica napus L.*, (2015)), spring wheat (*Triticum aestivum L.*, (2016)) and soybean (*Glycine max L. (Merr)*, (2017)) crop rotation and is annually tilled to 15 cm depth in the fall season. Mineral fertilizers are applied in spring. Phosphorus is subsurface seed placed as mono ammonium phosphate (40 kg ha^{-1} in 2015, 45 kg ha^{-1} in 2016, and 20 kg ha^{-1} in 2017), whereas the N is surface broadcast as urea (127 kg ha^{-1} in 2015 and 123 kg ha^{-1} in 2016) and ammonium sulphate (22 kg ha^{-1} in 2015 and 2016) . Nitrogen fertilizers were seed placed in 2017 (7 kg ha^{-1} urea and 9 kg ha^{-1} ammonium phosphate).

1.2.3.2. Sample collection and analyses

Overland flow was monitored on each field at the outlets of two swales at 15-minute intervals (Figure 1.1). V-notch weirs equipped with capacitance sensors (Odyssey, Dataflow Systems Ltd.)

and ultrasonic depth sensors (50A, Campbell Scientific, Edmonton, Canada) were used for this purpose. However, overland flow was not monitored on the non-tiled field in 2017 due to logistical issues.

Stream stage (water levels) in the roadside ditches adjacent to each field were also monitored, both upstream and downstream of the surface drains, at 15-minute intervals using capacitance sensors. In 2016, a depth-velocity sensor (Flo-tote 3 and FL 900 series logger, Hach Ltd.) was deployed at the downstream sampling location of the roadside ditch adjacent to the tiled field to develop a rating curve for converting water depths to flow values, and to estimate the volume of water that was leaving the field during lateral flow reversal conditions from ditch to field. However, this was not done for the non-tiled field due to logistical issues. Ditch flow was not monitored for the non-tiled field in 2017.

The groundwater table (GWT) was also monitored on each field using PVC wells (3.75 cm inside diameter, 1.5 m depth) mounted with capacitance level loggers. Wells were installed along two transects (west to east direction) to characterize the downfield, midfield and upfield section of each field. The groundwater table was not monitored for non-tiled field in 2017.

Tile drain discharge was measured at 15-minute intervals at the tile drain main outlet using a Flo-tote 3 and FL 900 series logger (Hach Ltd.). A pressure transducer (HOBO U20, Onset Corp.) was also placed into the tile to monitor water levels during periods when the velocity and level sensor did not function. During periods when the tile outlet was submerged in the collection pond or the lifting station did not work, manual pumping was used to transfer the collected tile water to the on-site retention pond. In 2016, when the lifting station was added, an automatic water level logger was deployed in the station to record water levels for comparison with measured tile discharge rates (Mini Orpheus, Campbell Scientific Ltd.). A standard meteorological station

(CR10x, Campbell Scientific Ltd.) was established at the site to take hourly measurements of precipitation (TE525M, Texas Electronics) and air temperature (HMC45C, Campbell Scientific Ltd.).

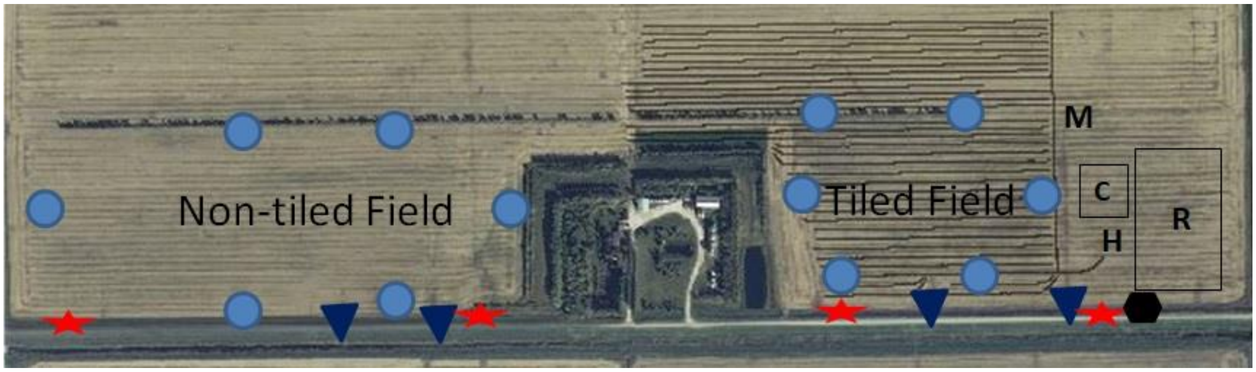


Figure 1.1. Schematic diagram of the study site. Inverted triangles represent the V-notch weirs. Circles represent the groundwater wells. Stars represent the ditch monitors. The hexagon represents the ditch flow meter in the culvert. The header tile, collection pond (lifting station), meteorological station and the retention pond were indicated by the letters H, C, M and R respectively

1.2.3.3. Flow estimation

Due to the flat landscape, there were periods during which the adjacent ditches were full, and fields was flooded. During such periods, surface runoff can become stagnant (backwater effects). Such periods were differentiated from more conventional flow periods (i.e., when surface water ran freely into ditch) using the difference between water levels at the weirs and the culvert in the ditch exit. When flow resumed or initiated from inundation, overland flow was assumed to be equal to ditch flow until ditch and surface weirs were disconnected. It was safe to make this assumption at our site as the roadside ditch essentially received its runoff water from our study site, and there was a drainage divide between the two fields where the farm is found, with the two ditches flowing away from the farmyard. All of the water in each ditch flows through culverts located at the roads adjacent to the farm. In the culvert where the flow meter was installed, two different rating curves were developed for spring (early spring snowmelt and rainstorm events) (r^2 =0.96) and summer (r^2 = 0.78) when estimating flows for 2015 and 2017 events with backwater effects/stagnation. This method may have slightly overestimated runoff volumes by accounting already available ditch water as overland flow. Error estimates, considering the water available prior to flow resumption, were 2.6, 3.5 and 4.5% of annual overland flow estimations for 2015, 2016 and 2017. We also considered the possibility of overestimation from subsurface flow that may have seeped into the ditch. Possibilities for subsurface dilution are lower in this landscape during snowmelt runoff due to minimal subsurface flow due to frozen soils. Moreover, ion concentrations in the ditch water (not shown) did not show higher concentrations during snowmelt backflow periods, despite the fact that subsurface water in fields was found to be ion-rich, suggesting that minimal subsurface seepage was received by the ditch. Currently, there are no established methods to measure flow during inundation periods, nor are there estimates of acceptable error ranges to enable a direct comparison with existing hydrometric literature (Water Survey of Canada, 2012; Kiang et al., 2018). This is an area where additional research is needed. For overland flow events without stagnation, the Kindsvater–Shen equation was used to estimate water flow through V-notch weirs (US Department of the Interior, Bureau of Reclamation, 1997).

Readings from a Hach Flo-tote 3 and a Hach FL 900 series logger were used to estimate tile flow when the tiles were flowing unobstructed. However, during periods when the tile outlet was fully submerged, and equipment could not be accessed, tile flow was estimated from a manual pump (flow rate of 7 L s^{-1}) that transferred tile water from the collection zone to a larger on-farm retention pond. The manual pump rate combined with the dimensions of the lifting station and its reservoir were used to estimate the tile flow volume pumped following the lifting station failure.

The Manning equation was used to estimate the tile flow volume when the lifting station reservoir and lateral tiles were partially full.

Chapter 2: Contribution of Overland and Tile Flow to Runoff and Nutrient Losses from Vertisols in Manitoba, Canada

2.1. Introduction

The eutrophication of large inland lakes is a major global environmental issue. In the northern Great Plains of North America, Lake Winnipeg experiences frequent algal blooms as a consequence of nutrient loading from agricultural fields with smaller inputs from urban centers (Schindler et al., 2012). Thus, reducing nutrient loads—particularly P—to the lake is a priority (Schindler et al., 2016). At present, there is little tile drainage in this region (Council of Canadian Academies, 2013). However, the expansion of tile drainage has been occurring to lengthen the cropping season and protect crops from waterlogged conditions (Cordeiro and Ranjan, 2012; Rahman et al., 2014), which have historically been present in spring but are now becoming more prevalent in summer due to increasing summer rainfall (Dumanski et al., 2015). Given the potential for tile drains to exacerbate P loads (Kleinman et al., 2015), there is a concern that increased tile drainage in the northern Great Plains will enhance nutrient loads to Lake Winnipeg. However, little is known about the role of tile drainage in northern climates.

The Red River valley of North America is a major subregion of the Lake Winnipeg watershed. In the absence of subsurface drainage, the landscapes of the Red River valley favour runoff export via infiltration excess overland flow due to their relatively flat topography, long cold winters, and the presence of clay-rich soils with low hydraulic conductivities and soil-ice in winter and spring (Fang et al., 2007). The majority of the annual runoff and nutrient (P and N) losses from agricultural fields in the Red River valley typically occur in spring via snowmelt runoff over frozen soils (Tiessen et al., 2010), whereas summer and fall months are dry with little runoff (Rattan et al., 2019). With ongoing climatic change, however, the magnitude of spring runoff in the region is exacerbated by the increasing frequency of rain-on-snow events, which are replacing the more gradual radiation snowmelt events that have historically occurred (Jeong and Sushama, 2018). In addition, an increased frequency of multiday summer storms in the northern Great Plains causes enhanced surface runoff and subsurface flow generation in warmer periods when soil frost is absent (Shook and Pomeroy, 2012). It is unclear if, and to what extent, tile drainage may affect runoff generation and the partitioning of runoff between surface and subsurface pathways in the Red River valley, or how this may vary seasonally.

In more temperate regions in North America (i.e., Quebec and Ontario), a substantial volume of runoff is routed into subsurface tile drainage throughout the year, thus decreasing the magnitude of overland flow (Jamieson et al., 2003; Van Esbroeck et al., 2016; Plach et al., 2018a). It is possible that tile drains may reduce edge-of-field P losses in the Red River valley through the suppression of overland flow. Alternatively, tile drains may increase edge-of-field losses of P, exacerbating water quality issues in the Red River valley. In other clayey landscapes, increased runoff and nutrient losses through tile drains have been linked to preferential flow through macropores and desiccation cracks, which are active during frozen as well as thawed conditions (Smith et al., 2015; Grant et al., 2019a). However, this has not been assessed in the Vertisolic soils found within the Red River valley. The dynamic swelling and shrinking nature of the Vertisols, due to their high ratio of montmorillonitic clay minerals and lack of permanent biopores, may lead to variable subsurface responses in comparison with other clay-rich soils (Kurtzman et al., 2016; Kokulan et al., 2019a).

Little is known about the impacts of tile drainage on runoff generation and water quality in the Red River valley, or in cold regions with Vertisolic soils. An improved understanding of the relative contributions of overland flow and tile drainage to runoff and nutrient losses is crucial to determine if the expansion of tile drainage will exacerbate existing water quality issues in the Red River valley. In the current study, runoff, soluble reactive P (SRP), total P (TP), and NO_3–N losses were monitored in overland flow and tile drainage at the edge of the field on a farm located in the Red River valley during three consecutive years (2015–2017). The objectives of this study were (i) to quantify the relative contributions of overland and tile flow to edge-of-field runoff and nutrient loads, and (ii) to characterize seasonal variability in nutrient concentrations and loads in overland and tile flow.

2.2. Methods

2.2.1. Study site

This study was conducted on a farm in Manitoba, Canada. Soils at the field site are montmorillonitic Gleyed Humic Vertisols of the Red River series (Dystric Vertisols, FAO classification). The climate of the region is characterized by long and cold winters with minimal snow cover and a deep seasonal soil-ice layer, contrasted with short, warm, and relatively dry summers (Rahman et al., 2014). The mean annual temperatures at the field site are 2.8°C with annual maxima in July (17°C) and minima in January (−16°C).

The 25-ha study field is nearly flat, with an average slope of 0.3%. Surface runoff from the field flows into nearby roadside ditches via shallow in-field surface swales (drains) (Kokulan et al., 2019a). Beneath the field, lateral tiles (10-cm diam. at ~85- to 120-cm depth) are installed at 13-m intervals, which is typical for the province of Manitoba. More detailed information about tile drainage at this site is provided in Kokulan et al. (2019a).

The field has been under a long-term canola (*Brassica napus* L.)–spring wheat (*Triticum aestivum* L.)–soybean [*Glycine max* (L.) Merr.] rotation, with canola in 2015, spring wheat in 2016, and soybeans in 2017. The field is tilled annually in fall to a depth of ~15 cm, and mineral fertilizers are applied in spring. Monoammonium phosphate is applied in the subsurface at the time of seeding (40 kg ha^{-1} in 2015, 45 kg ha^{-1} in 2016, and 20 kg ha^{-1} in 2017). Nitrogen is surface broadcast as both urea (127 kg ha^{-1} in 2015, and 123 kg ha^{-1} in 2016) and ammonium sulphate (22 kg ha^{-1} in 2015 and 2016). However, in 2017, N fertilizers were seed placed (7 kg ha^{-1} urea + 9 kg ha^{-1} ammonium phosphate).

2.2.2. Field Instrumentation and data collection

Overland flow depths were monitored at 15-minute intervals at the outlets of two surface swales using V-notch weirs, each equipped with an ultrasonic depth sensor (50A, Campbell Scientific Ltd.) and a capacitance sensor (Odyssey, Dataflow Systems Ltd.). Sensor estimates were validated using manual measurements. Within the roadside ditch adjacent to the field, capacitance sensors were installed in stilling wells both upstream and downstream of the field to record water levels at 15-minute intervals (Odyssey, Dataflow Systems Ltd.). In 2016, a depth-velocity sensor (Flo-tote 3 and FL 900 series logger, Hach Ltd.) was deployed at the downstream sampling location of the ephemeral ditch to develop a rating curve for converting water depths to flow values, and to estimate the volume of water that was leaving the field during lateral flow reversal conditions from ditch to field, which are common in the region (Cordeiro et al., 2017). Tile drain discharge was measured at 15-minute intervals at the tile drain main outlet using a Flo-tote 3 and FL 900 series logger (Hach Ltd.). A pressure transducer (HOBO U20, Onset Corp.) was also placed into the tile to monitor water levels during periods when the velocity and level sensor did not function. During periods when the tile outlet was submerged in the collection pond or the lifting station did not work, manual pumping was used to transfer the collected tile water to the on-site retention pond. In 2016, when the lifting station was added, an automatic water level logger was deployed in the station to record water levels for comparison with measured tile discharge rates (Mini Orpheus, Campbell Scientific Ltd.). A standard meteorological station (CR10x, Campbell Scientific Ltd.) was established at the site to take hourly measurements of precipitation (TE525M, Texas Electronics) and air temperature (HMC45C, Campbell Scientific Ltd.).

Tile and overland flow water samples were collected with programmable autosamplers (AS950, Hach Ltd.) from March 1st to September 30th in each of the study years (2015 to 2017), capturing the annual spring freshet (snowmelt), spring storms and summer thunderstorms. However, events in September 2015 and 2016 were not sampled for water chemistry due to logistical issues. No significant rain events occurred between late September and early March in any of the study years. Indeed, daily precipitation was below <10 mm and mean air temperatures were below 0^{o}C from mid-November through the onset of spring snowmelt runoff (early-March in 2015, mid-March in 2016 and late-March in 2017) (Fig. 2.1). Thus, although sampling was not conducted during these periods, no flow at the site occurred. Periodic field visits were done to validate this. Between snowmelt and freeze up in each of the study years, a 2 to 4-hour sampling interval was employed to capture the rising and recession limbs of storm hydrographs, while 6 to 12-hour sampling intervals were used during periods of water stagnation or impeded flow. Autosamplers collected samples in acid-washed 1 L polyethylene bottles. Manual grab samples were collected in acid-washed 0.5 L polyethylene bottles when the flow was too low for autosamplers to capture.

Upon return to the laboratory, a 200 mL subsample was promptly filtered through a 0.45 μm filter paper (Whatman, 47 mm), and stored at 4 ^{o}C. Samples were analyzed for soluble reactive phosphorus (SRP) (QuikChem Method 10-115-01-1-A) and nitrate-nitrogen (NO_3-N) (QuikChem Method 10-107-04-4-C) with a QuikChem 8500 series 2 FIA system (Lachat instruments) through flow injection analysis colorimetry and flow injection analysis within 48 hours of collection. A final 200 ml subsample was frozen at -18 ^{o}C and subsequently processed and analyzed for total phosphorus (TP) (QuikChem Method 10-115-01-4-C) with QuikChem 8500 series 2 FIA system. Standard checks were used per 20 samples as a mean for quality control. Samples that exceeded the standard spectrum were diluted and re-analyzed.

2.2.3. Data analyses

Due to the flat landscape, there were periods during which the adjacent ditch was full, and the field was flooded. During such periods, surface runoff can become stagnant (backwater effects). Such periods were differentiated from more conventional flow periods (i.e., when surface water ran freely into ditch) using the difference between water levels at the weirs and the culvert in the ditch exit. When flow resumed or initiated from inundation, overland flow was assumed to be equal to ditch flow until ditch and surface weirs were disconnected. It was safe to make this assumption at our site as the roadside ditch essentially received its runoff water from our study site (Kokulan et al., 2019a) and all of this water flows through the culvert where the flow meter was installed. Two different rating curves were developed for spring (r^2 =0.96) and summer (r^2 = 0.78) when estimating flows for 2015 and 2017 events with backwater effects/stagnation. This method might have slightly overestimated runoff volumes by accounting already available ditch water as overland flow. Over error estimates, considering the water available prior to flow resumption, were 2.6, 3.5 and 4.5% of annual overland flow estimations for 2015, 2016 and 2017. We also considered the possibility of overestimation from subsurface flow that may have seeped in the ditch. Possibilities for subsurface dilution are lower in this landscape during snowmelt runoff due to minimal subsurface flow due to frozen soils. Moreover, ion concentrations in the ditch water

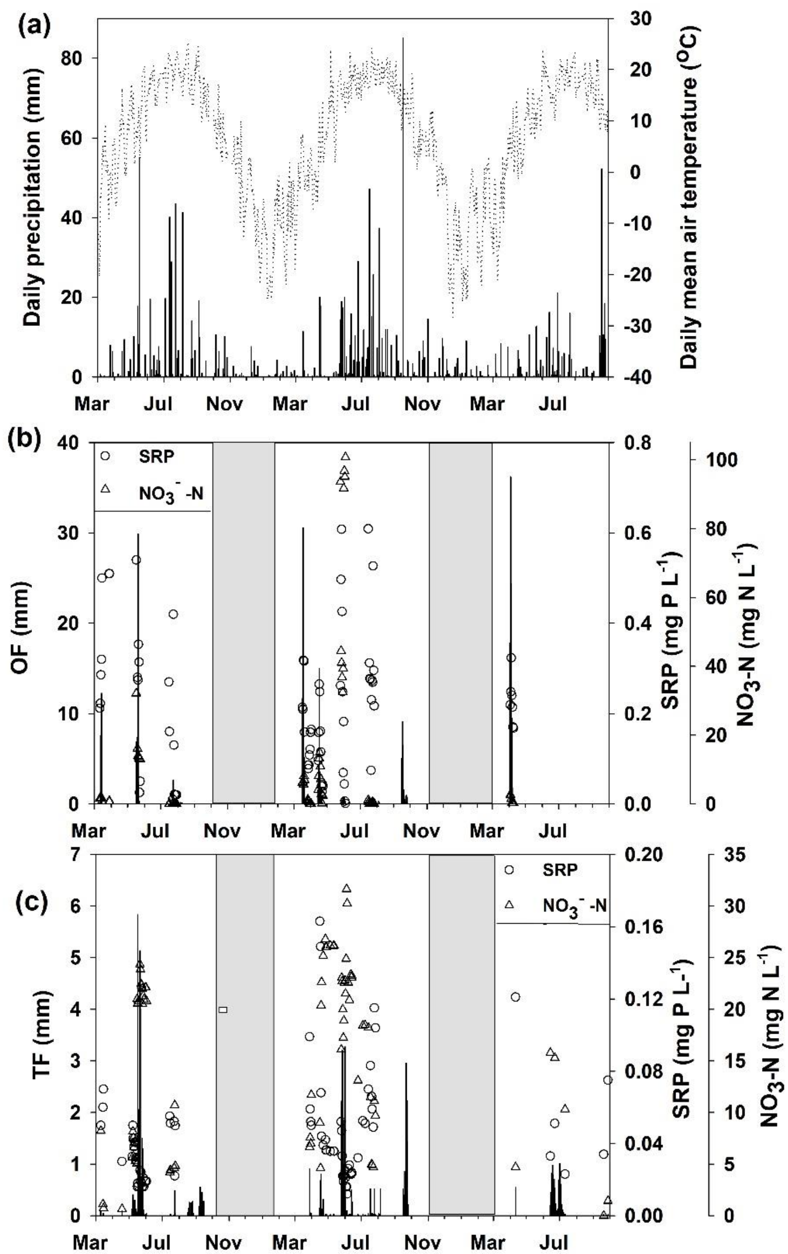


Figure 2.1. (a) Daily precipitation (vertical bars) and daily mean air temperature (dashed lines), (b) daily overland flow (OF), and (c) daily tile flow (TF) with daily flow-weighted mean concentration of soluble reactive P (SRP) and NO3–N. Different scales have been used for Panels b and c to highlight minor variations. Runoff was not monitored during the shaded periods.

Currently, there are no established methods to measure flow during inundation periods, nor are there estimates of acceptable error ranges to enable a direct comparison with existing hydrometric literature (Water Survey of Canada, 2012; Kiang et al., 2018). This is an area where additional research is needed. For overland flow events without stagnation, the Kindsvater–Shen equation was used to estimate water flow through V-notch weirs (US Department of the Interior, Bureau of Reclamation, 1997).

Readings from a Hach Flo-tote 3 and a Hach FL 900 series logger were used to estimate tile flow, when the tiles were flowing unobstructed. However, during periods when the tile outlet was fully submerged, and equipment could not be accessed, tile flow was estimated from a manual pump (flow rate of 7 L s^{-1}) that transferred tile water from the collection zone to a larger on-farm retention pond.

Data were divided into early spring (snowmelt dominated), late spring (rain dominated, commencing after melt was over until 31 May), summer (1 June to 31 August), and fall (September) to characterize seasonality in runoff volumes and water chemistry. To compare water chemistry between overland and tile flow, daily flow-weighted mean concentrations (FWMCs) of SRP, TP, and NO_3–N were estimated using a time proportional compositing strategy (Williams et al., 2015). Daily nutrient loads were estimated using daily discharge and daily FWMC and subsequently combined to estimate seasonal and annual loads. Annual median nutrient concentration values were used to estimate the chemistry of the events in September 2015 and 2016 that were missed by our autosamplers.

Flow-weighted mean concentrations of SRP, TP, and NO_3–N for daily water samples and seasonal averages were compared with Mann–Whitney rank sum tests (pairwise comparisons), and one-way Kruskal–Wallis and Dunn's tests (group comparisons). Spearman's rank correlations were used to explore relationships between nutrient concentrations.

2.3. Results

2.3.1. Interannual Variability in Precipitation and Runoff Patterns

Substantial variability in precipitation types and depths was observed among the study years, permitting the examination of responses under a range of conditions (Fig. 2.1). Compared with long-term climate normals (1981–2010; ECCC, 2010), annual precipitation was normal in 2015 (534 mm, compared with the 30-yr mean of 545 mm), whereas 2016 (637 mm) was 17% wetter, and 2017 (425 mm) was 22% drier. Runoff volumes differed among and within the study years (Table 2.1), although annual runoff coefficients (total runoff [sum of overland and tile flow]/precipitation) were similar for the 3 yrs. (19, 24, and 24% for 2015, 2016, and 2017, respectively). The percentage of annual runoff that occurred as snowmelt in early spring was 22 (2015), 40 (2016), and 91% (2017), respectively (Table 2.1). Snowmelt runoff in 2015 was entirely the result of radiation melt with no rainfall, whereas snowmelt runoff in 2016 and 2017 was triggered by rain-on-snow events. Low-intensity (<8 mm h^{-1}) multiday spring storms were responsible for major spring runoff events in 2015 and 2016, whereas high-intensity convective thunderstorms were responsible for the other major runoff events (>10 mm h^{-1}) in the summer months.

2.3.2. Annual and Seasonal Differences in Runoff Responses and Pathways

In all 3 yr, daily air temperatures remained below zero from mid-November until the onset of spring snowmelt (March), and rainfall or melt events were not observed within those winter months (Fig. 2.1). Consequently, runoff was restricted to the March to October period, and no runoff occurred during the winter periods when our equipment was removed. On an annual basis, the contribution of overland flow to total runoff was 72 to 89%, thus significantly greater than that of tile flow (Fig. 2.2). However, the relative contributions of the two pathways differed by season. For example, in early spring, when considerable annual runoff occurred, overland flow contributed to ~99% of total runoff, and little flow passed through drainage tiles. In contrast, in late spring, summer, and fall, a substantial proportion of total runoff was contributed by tile drains (contribution to total runoff between May and September of 30% in 2015, 45% in 2016, and 100% in 2017). Tiles were more active in response to multiday late-spring storms, whereas rapid overland flow responses were observed during high-intensity convective storms. The annual contribution of overland flow relative to tile flow was greater in years during which early-spring snowmelt represented a greater fraction of total runoff (Table 2.1).

2.3.3. Nutrient Concentrations and Loads along Flow Pathways

Phosphorus concentrations and loads were substantially greater in overland flow than tile flow. Median daily SRP and TP concentrations were significantly greater in overland flow than in tile flow (600% greater for SRP, 500% greater for TP, $p < 0.001$), irrespective of years or seasons (Fig. 2.3a and b). Concentrations of SRP and TP in overland flow did not significantly vary across the seasons (median daily FWMCs were 0.214, 0.161, and 0.25 mg P L^{-1} for SRP and 0.306, 0.301, and 0.326 mg P L^{-1} for TP during early spring, late spring, and summer, respectively). However, P concentrations in tile flow were significantly ($p < 0.001$) larger in early spring (median daily FWMCs of 0.056 mg P L^{-1} for SRP and 0.108 mg P L^{-1} for TP) and summer (median daily FWMCs of 0.051 mg P L^{-1} for SRP and 0.13 mg P L^{-1} for TP), compared with late spring (median daily FWMCs of 0.028 mg P L^{-1} for SRP and 0.039 mg P L^{-1} for TP).

Table 2.1. Annual and seasonal precipitation, overland flow (OF) and tile flow (TF) depths, and nutrient losses.

Period	Precipitation		Runoff			SRP†			TP‡			NO_3–N		
	Snow	Rainfall	OF	TF	Total	OF	TF	Total	OF	TF	Total	OF	TF	Total
	mm					kg ha^{-1}								
Annual														
2015	27	507	72.9	28.6	101.5	0.227	0.007	0.234	0.332	0.014	0.346	8.3	5.3	13.6
2016	46	591	121.6	33.2	154.8	0.276	0.014	0.29	0.376	0.02	0.396	10.6	6.7	17.3
2017	94	331	89.8	10.9	100.7	0.234	0.007	0.241	0.374	0.007	0.381	1.7	1.7	3.4
Early spring														
2015	N/A§	0	21.9	0.2	22.1	0.067	<0.001	0.067	0.081	<0.001	0.081	0.4	<0.1	0.4
2016	N/A	19	60.7	1.1	61.8	0.142	<0.001	0.142	0.189	0.002	0.191	3.7	0.1	3.8
2017	N/A	12	89.8	0.6	90.4	0.234	<0.001	0.234	0.374	0.001	0.375	1.7	<0.1	1.7
Late spring														
2015	N/A	147	46.9	23.3	70.2	0.149	0.005	0.154	0.232	0.01	0.242	7.9	4.9	12.8
2016	N/A	127	31.5	17.7	49.2	0.067	0.006	0.073	0.096	0.008	0.104	6.2	3.8	10
2017	N/A	50	N/O¶	N/O	–	–	–	–	–	–	–	–	–	–
Summer														
2015	N/A	286	4.2	2.3	6.5	0.011	0.001	0.012	0.018	0.003	0.021	<0.1	0.3	0.3
2016	N/A	303	3.8	2.7	6.5	0.018	0.002	0.02	0.022	0.004	0.026	<0.1	0.4	0.4
2017	N/A	136	N/O	10.3	10.3	–	0.006	0.006	–	0.006	0.006	–	1.7	1.7
Fall														
2015	N/A	74	N/O	2.87	2.87	–	<0.001	<0.001	–	0.001	0.001	–	0.3	0.3
2016	N/A	142	25.7	11.8	37.5	0.051	0.004	0.055	0.07	0.007	0.077	0.7	2.4	3.1
2017	N/A	133	N/O	<0.1	<0.1	–	<0.001	<0.001	–	<0.001	<0.001	–	<0.1	<0.1

† SRP, soluble reactive P.

‡ TP, total P.

§ N/A, not applicable. ¶ N/O, not observed

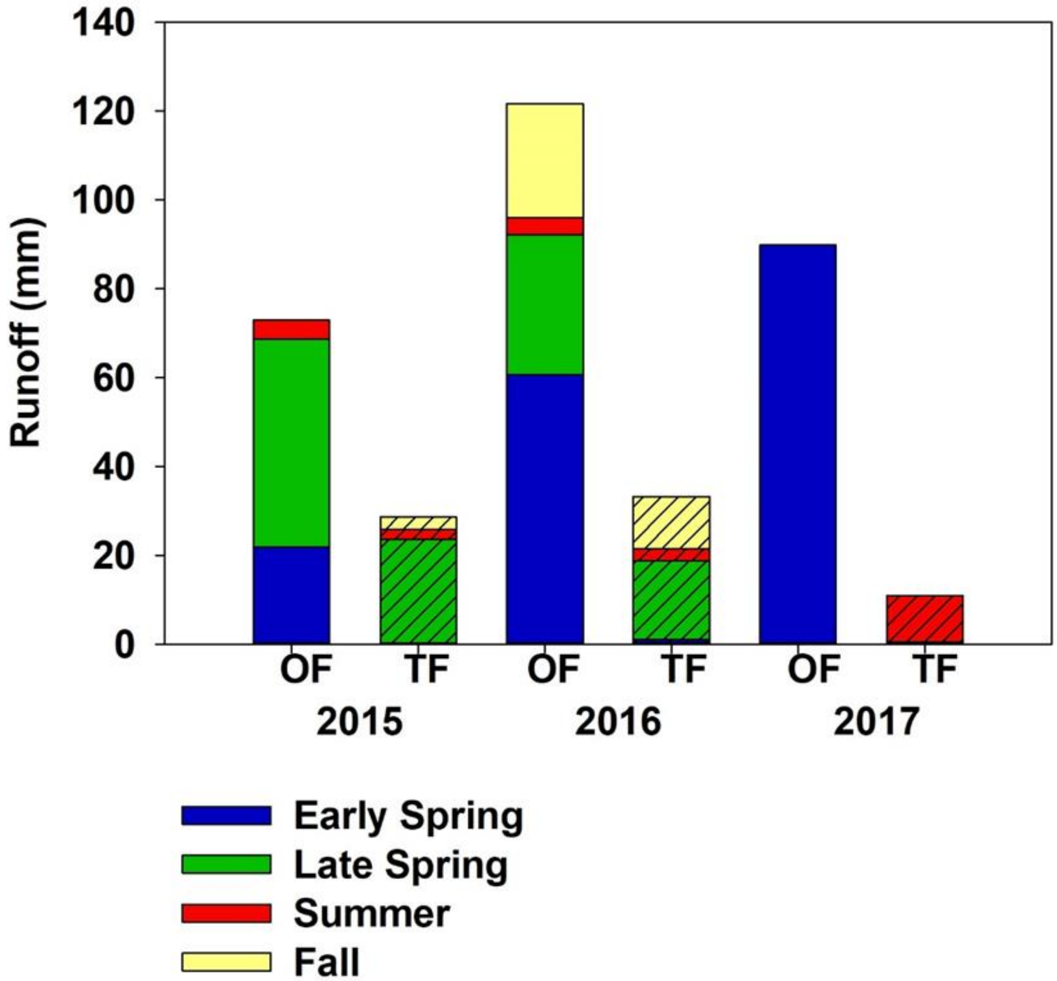


Figure 2.2. Comparison of annual and seasonal overland flow (OF) depths and tile flow (TF) depths during the study years.

Nitrate concentrations varied among years and seasonally within years (Fig. 2.3c). During early spring and in summer, concentrations of NO_3–N were significantly larger ($p < 0.05$) in tile flow (median daily FWMCs of 5.7 and 11.5 mg N L^{-1}) than in overland flow (median daily FWMCs of 1.09 and 0.01 mg N L^{-1}). In late spring, NO_3–N concentrations were elevated in both tile (median daily FWMC of 21.4 mg N L^{-1}) and overland flow (median daily FWMC of 13.9 mg

N L^{-1}), compared with other seasons ($p < 0.001$) (Fig. 2.3c). A few extreme overland flow NO_3–N values (>90 mg N L^{-1}) were observed during low flows that happened in late spring when rainfall occurred shortly after fertilizer application (Fig. 2.1.b).

In tile discharge, SRP and TP were negatively correlated with NO_3–N (SRP: $r^2 = 0.4$; TP: $r^2 = 0.62$; $p < 0.001$) (Fig. 2.4). The relationship between tile flow TP and NO_3–N was statistically significant in late spring and summer ($p < 0.001$), but not in early spring. Overland flow NO_3–N and TP were also positively correlated ($p < 0.05$).

Overland flow contributed to the majority of P losses from the study site (95–97% of annual SRP, 95–98% of annual TP) in all years (Fig. 2.2, Table 2.1), and the majority of P losses occurred during early and late spring (74–97% of annual SRP losses, 74–98% of annual TP losses). In contrast, tile flow contributed disproportionately more to NO_3–N losses, accounting for 39 to 50% of yearly NO_3–N losses during the study years. Despite its lower NO_3–N concentrations, overland flow contributed to >50% of annual NO_3–N losses due to its larger runoff volumes. In years with normal or above-normal precipitation, substantial NO_3–N losses were attributed to late spring runoff coinciding with fertilizer application and increased tile activity. For example, 93% of the annual NO_3–N losses occurred in late spring 2015 and 64% in late spring 2016.

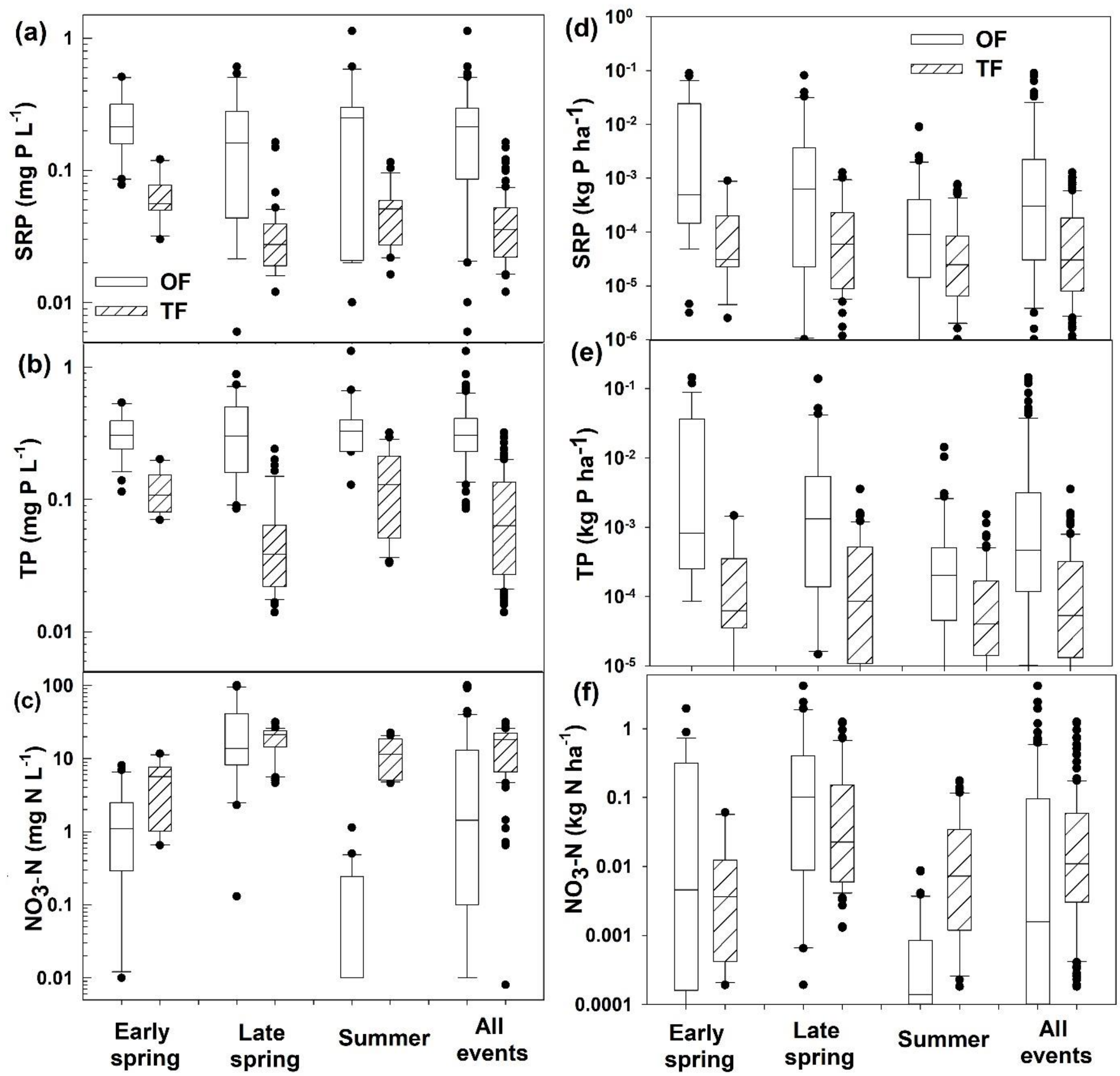


Figure 2.3. Box and whisker plots showing seasonal and overall daily flow-weighted mean concentrations of (a) soluble reactive P (SRP), (b) total P (TP), and (c) NO_3–N and daily loads of (d) SRP, (e) TP, and (f) NO_3–N in overland flow (OF) and tile flow (TF) during the study period. Fall concentrations were not provided due to lack of sampling. The center line of the boxes indicates median values and the boxes are bound by the 25th and 75th percentiles. Whiskers represent the 10th and 90th percentiles, and the black circles represent statistical outliers.

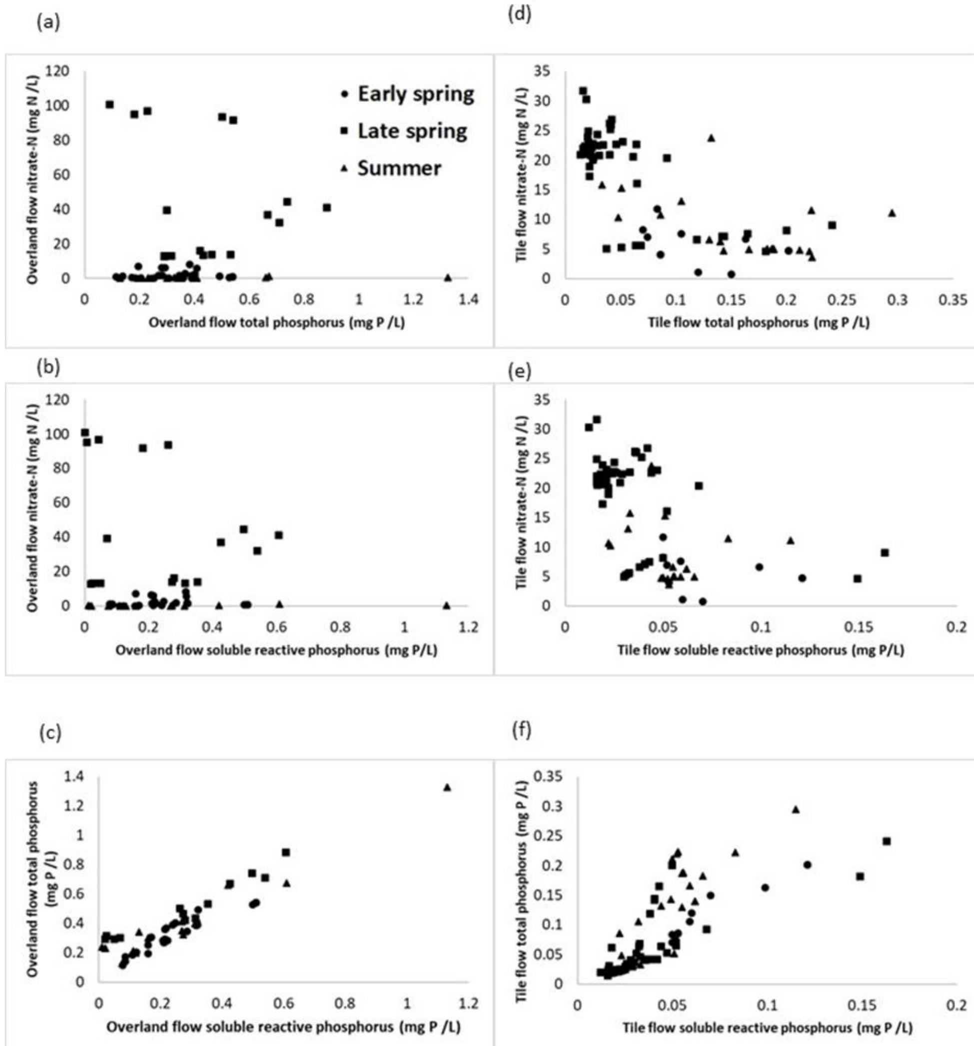


Figure 2.4. Relationships between (a) total phosphorus and nitrate-N in overland flow (OF); (b) soluble reactive phosphorus and nitrate-N in OF; (c) soluble reactive phosphorus and total phosphorus in OF; (d) total phosphorus and nitrate-N in tile flow (TF); (e) soluble reactive phosphorus and nitrate-N in TF; and (f) soluble reactive phosphorus and total phosphorus in TF.

2.4. Discussion

2.4.1. Runoff Export Pathways

The current study has shown that in the Red River valley, which is underlain by clay-rich soils, overland flow prevails as the major runoff pathway, despite the presence of tile drainage. This is due to the fact that most of the annual runoff occurs during the snowmelt period when tile drains are decoupled from the surface by seasonal frost. The development of soil-ice layers as deep as 0.75 to 1 m during winter months can impede water movement through soil profile during snowmelt (Kahimba et al., 2009) and may also enhance the activation of rapid overland flow by creating a perched water table over tile drains (Kokulan et al., 2019a). Manual probing of the soil during routine site visits in each of the study years suggest that the soil-ice melted by the first week of May in 2015 and 2017, and in mid-April in 2016 (during a rainstorm).

In contrast with the early spring period, the relative contribution of tile drains during other seasons was substantial, demonstrating that when tile drainage is present, and soil-ice is not, runoff can be rerouted into the subsurface (Table 2.1). Although tile flow can represent a substantial volume of discharge during the ice-free period, the annual contribution of tile drains was small. However, given the predicted increase in the frequency of multiday spring and summer storms in the Canadian Prairies (Shook and Pomeroy, 2012), tiles may become more active in the future. Indeed, tiles have the potential to be a tool for farmers to tackle the uncertainty associated with increased summer rainfall, although this may increase edge-of-field runoff (Rahman et al., 2014). The generation of subsurface runoff through tile drains will, however, depend on the nature of the precipitation. Indeed, tiles substantially contributed to runoff during low-intensity, multiday late-spring events on thawed soils, but not after high-intensity rainfall events. After convective thunderstorms, rapid overland flow responses occur, and tile flow responses lag behind overland flow (Kokulan et al., 2019a). Under such scenarios, tile responses were small, in contrast with the more rapid tile flow activation that has been observed in other clay-rich soils (Smith et al., 2015). Unlike macroporous soils with stable biopores, desiccation cracks in Vertisolic soils swell and seal preferential pathways once they become moist (Hardie et al., 2011), which can result in slower flow. Moreover, the desiccation cracks that are prevalent at the study site may not be deep enough to intersect tile drains typically located 85 to 120 cm below the surface (Ali et al., 2018; Kokulan et al., 2019a), which may also reduce the potential for preferential flow to route surface water into tile drains rapidly.

Our results suggest that given the current precipitation characteristics in the Red River valley (i.e., dominant snowmelt processes in early spring and convective storms in late spring and summer), tile drains will do little to reduce the occurrence of overland flow in the region. However, tile drains may increase edge-of-field nutrient losses if low-intensity summer storms occur more frequently, or if the presence of seasonal frost is lessened under a warmer future climate. Runoff was not monitored in October and November 2015 and 2016. However, none of the precipitation events observed in these periods exceeded maximum intensity and/or antecedent moisture thresholds to generate a runoff response, as reported by Kokulan et al. (2019a).

2.4.2. Differences in Water Chemistry between Overland Flow and Tile Drainage

Although P and N were lost in both overland flow and tile drainage, nutrient losses in overland flow were substantially larger due to the dominance of overland flow hydrologically. Phosphorus losses in overland flow were particularly great relative to tile flow, as a result of the significantly larger P concentrations in surface runoff relative to tile drainage. The P concentrations and loads in overland flow from our study are comparable with other studies that have evaluated overland flow P losses in the region, notably in the Red River valley (Tiessen et al., 2010; Liu et al., 2013).

The majority of the P losses were associated with major runoff events with substantial overland flow contributions, which generally occurred with spring snowmelt (Fig. 2.2). Snowmelt runoff from the fields is impeded in the Red River valley due to the nearly flat terrain, coupled with frozen ditches or ice-jammed culverts and provincial channels (Schindler et al., 2012; Cordeiro et al., 2017). Consequently, runoff water remains on fields for several days, risking nutrient release from soils or plant residues (Tiessen et al., 2010; Amarawansha et al., 2015). The results of this study show that tile drainage will do little to reduce the P loads associated with overland flow during snowmelt, largely due to the presence of frozen ground, which impedes tile drainage. However, increased contribution of tiles to winter or early-spring edge-of-field P losses under a future warmer climate with frequent melts and less ground-ice may affect the flow path partitioning of runoff and P loss.

Major storms in late spring can also contribute to substantial nutrient losses in overland flow as late spring is associated with fertilizer application (i.e., 2015). Although NO_3–N concentrations in overland flow were generally <10 mg N L^{-1} over the study period, they were elevated during the late spring season, coinciding with fertilizer application. Previous studies have listed N fertilizer application rate (Liu et al., 2013) and timing (Smith et al., 2007), as well as tillage (Liu et al., 2013), as some of the reasons for increased N concentrations in runoff. At our site, N was applied in the form of fertilizers in spring, no manure was applied, and the land was tilled in fall. The fact that N concentrations in runoff were small prior to the application of fertilizers but increased substantially in the first event after fertilizer application suggests that fertilizer is the source of this N at our site. The fact that N concentrations declined in events that occurred later in the season suggests that source exhaustion was occurring.

In contrast with overland flow, tile drainage P concentrations were much smaller, whereas NO_3–N concentrations were larger. Greater P concentrations in surface runoff have been shown in selected studies in flat landscapes with fine-textured soils (Algoazany et al., 2007; Pease et al., 2018b), whereas other studies have observed similar or higher tile P concentrations compared with surface runoff (Turtola and Jaakkola, 1995; Uusitalo et al., 2003). In contrast with P, elevated NO_3–N concentrations are frequently observed in tile drainage because flow through the soil matrix often favours NO_3^- anion leaching (Zhao et al., 2001). The higher NO_3–N concentrations compared with overland flow suggest that tile drainage has the potential to enhance NO_3–N loading in the Red River valley.

Elevated loads of P into tile drains have been linked to preferential flow pathways (Vidon and Cuadra, 2011; Reid et al., 2012), whereas flow through the soil matrix is a source for NO_3–N (Cuadra and Vidon, 2011). The lower P and higher NO_3–N concentrations and the inverse relationship between P and NO_3–N concentrations in tile drainage at our study site suggest that matrix flow intercepted by tile drains is a major transport mechanism in this region (Cuadra and Vidon, 2011; Vidon and Cuadra, 2011; Pease et al., 2018a). Indeed, the majority of tile flow occurred during late spring when low-intensity, long-duration storms and wet antecedent conditions prevailed, thus favouring matrix flow instead of rapid preferential flow (Kokulan et al., 2019a). Phosphorus in this slowly propagating matrix flow is likely being adsorbed by calcareous clays, resulting in smaller P concentrations (Ige et al., 2005), whereas NO_3–N would be mobile. However, it must be noted that P concentrations in tile flow during frozen (early spring) and dry (summer) conditions were higher than late-spring concentrations, when vertical preferential flow is more likely (Hardie et al., 2011; Grant et al., 2019a). Although greater than in spring, winter and summer tile P concentrations were still lower than the overland flow P concentrations at that time.

Therefore, although they were likely active in early spring and summer, vertical preferential flow paths were not directly connected to tile drains, and a mixing between preferential flow and matrix flow may have occurred somewhere in the upper soil profile, as noted by Ali et al. (2018) and Kokulan et al. (2019a). In addition, farm activities such as conservation tillage and subsurface banding of P fertilizers also could have contributed to lower tile P concentrations by sealing surface preferential flow paths (Williams et al., 2016; Grant et al., 2019a). Both our tile P and NO_3–N loads were substantially lower than those reported in other North American studies outside of the Red River valley (Christianson and Harmel, 2015; Christianson et al., 2016), a discrepancy that confirms that dominant flow pathways and nutrient mobilization mechanisms are strongly influenced by regional climate and management practices. However, it must be noted that we did not collect water samples during fall 2015 and 2016 and used the annual medians to estimate the daily nutrient loads. Therefore, there may be uncertainties associated with this estimation, even though our annual medians were similar to fall nutrient concentrations from other North American studies (Pease et al., 2018b).

2.5. Conclusions

We evaluated edge-of-field runoff and nutrient losses via overland and tile flow from an annually tilled agricultural farm in the northern Great Plains of North America. Despite the presence of tile drainage, overland flow was still the dominant pathway contributing to the majority of total annual runoff, P, and NO_3–N losses. This was mainly because tiles are decoupled from the surface by frozen ground when the majority of yearly runoff occurs, but also because tile flow, when it happens, appears to be driven mainly by matrix flow or mixed flow rather than primarily as preferential flow, which is in contrast with what was expected in a Vertisolic soil. Our findings suggest that tile drainage will do little to modify edge-of-field P losses in the Vertisolic soils of the Red River valley, but it may exacerbate NO_3–N losses. These findings provide valuable insight into the potential role of tile drains in cold regions, which has implications for both agronomic and environmental-based management decisions in the clayey landscapes of the northern Great Plains.

2.6. Acknowledgements

This project was funded by the Manitoba Agriculture, Food and Rural Development (MAFRD), Manitoba Conservative Districts Association (MCDA), Environment and Climate Change Canada's (ECCC) Lake Winnipeg Basin Stewardship Fund, the Natural Sciences and Engineering Research Council of Canada (Macrae–DG and Lobb–DG), the Canada First Excellence Research Fund (Agricultural Water Futures) and Canadian Agri-food Policy Institute (CAPI). Northern Plains Drainage Systems, Brendan Brooks, Anthony Buckley, Eva Slavicek, Vito Lam, Bo Pan, Kui Liu, and Reza Habibiandehkordi are thanked for the field and laboratory assistance.

Chapter 3: Hydroclimatic controls on runoff activation in an artificially drained, near-level vertisolic clay landscape in a Prairie climate

3.1. Introduction

Approximately 194 million hectares of arable land are currently drained globally for agriculture (International Commission on Irrigation and Drainage, 2017). Artificial drainage systems, which can be either surface drains or subsurface tile or mole drains, may reduce waterlogged soil conditions and the occurrence of ponding and overland flow (OF) by removing excess water from the vadose zone, thereby enhancing water infiltration. This improves soil aeration by maintaining a desired water table depth and promotes enhanced crop growth while extending cropping and grazing seasons (Dils and Heathwaite, 1999; King et al., 2015). Although tile drains have clear agronomic benefits, they can also have negative environmental consequences, notably because they may act as large, laterally-oriented preferential subsurface conduits that can export significant quantities of nutrients such as phosphorus (P) and nitrogen (N) from crop lands (Dils and Heathwaite, 1999; King et al., 2015). Although this potential exists, the impacts of tile drainage on water chemistry are variable throughout North America (Beauchemin et al., 1998; Jamieson et al., 2003; King et al., 2016; Macrae et al., 2007; Smith et al., 2015; Van Esbroeck et al., 2016). The activation of tile flow and its associated nutrient export are largely influenced by soil characteristics, the design of the drainage system, management practices such as tillage/no-till and nutrient application strategies (form, placement, rate and timing), climate and site-specific hydrology (King et al., 2015). Mechanistic and process-based research is therefore needed to identify situations in which tile drainage poses a greater or smaller risk of exporting nutrients to surface water bodies, and to identify the flow generation mechanisms behind this risk.

It is known that artificial drainage modifies precipitation-runoff relationships (Runoff here is defined as the discharge from the field where surface runoff primarily occurs as overland flow and subsurface runoff primarily occurs as tile flow); however, significant knowledge gaps remain regarding the control of hydroclimatic drivers on flow generation in tiled landscapes, and the relative contributions of tile flow and other flow pathways (i.e., overland, non-tile groundwater) in regions underlain by soils with high water-holding capacity such as clay-rich soils. A process-based understanding of these drivers is critical to understanding the efficacy of tile drains in present and future, both with climate change and with expansion of tile drainage. This understanding also has implications for the understanding and management of nutrient transport into tile drains. Climate change notably has the potential to influence flow activation in agricultural systems by affecting precipitation characteristics such as form, amount, duration and intensity, as well as soil antecedent moisture and temperature status (Bosch et al., 2014). Past studies conducted in humid temperate regions have shown that tiles significantly limit the initiation of saturation-excess overland flow (SOF) by removing water from the vadose zone and by suppressing water table rise to the surface (Deasy et al., 2009; Zumr et al., 2015). However, Hortonian (or infiltration-excess) overland flow (HOF) can still be activated when rainfall intensity exceeds the infiltration capacity of soil. High-intensity rainfall may also activate preferential flow pathways in macroporous soils (Edwards et al., 1992; Heppell et al., 2002; Stone and Wilson, 2006), thus allowing vertical connectivity between the surface and subsurface tiles drains: this can produce rapid tile flow responses and lead to the rapid mobilization of nutrients in tile drain effluent (Smith et al., 2015; Stillman et al., 2006; Stone and Wilson, 2006; Vidon and Cuadra, 2010). Antecedent moisture conditions (AMC) have also been identified as important drivers of runoff generation, both in the presence and in the absence of tile drains (Hardie et al., 2011; Macrae et al., 2007; Macrae et al., 2010; Vidon et al., 2012), as they control effective porosity for water flow in unsaturated soil conditions as well as portion of soil volume occupied by ice formed during seasonal freezing (Cordeiro et al., 2017; Gray et al., 2001). In temperate regions, runoff responses to AMC have been shown to be non-linear in nature (Macrae et al., 2010), and vary temporally (Lam et al., 2016a; Macrae et al., 2007; Vidon et al., 2012). However, the relative importance of controls such as rainfall intensity and antecedent conditions has not been assessed across different types of tile drained landscapes.

Tile drainage is rapidly expanding in the Northern Great Plains of North America (Council of Canadian Academies, 2013; Cordeiro and Ranjan, 2012; Jia et al., 2017; Rahman et al., 2014). However, the potential impacts of tile drainage on precipitation-runoff dynamics in this region are poorly understood. Although a few studies on tile drainage have been undertaken in this region (e.g. Cordeiro and Ranjan, 2012; Rahman et al, 2014; Satchithanantham and Ranjan, 2015), these were not done in clay-rich soils such as those in the Red River Valley (RRV) basin. Although studies have been conducted on tile drainage in clay soils in other regions (e.g. King et al., 2015), these regions do not have cold climates such as those in the Northern Great Plains. The RRV basin is a generally flat landscape that hosts predominantly vertisolic clay soils prone to shrinking and swelling and with low hydraulic conductivities (Cordeiro et al., 2017), characterized by a sub-humid climate with long and cold winters having minimal snow cover and a deep seasonal ice layer, contrasted with short, warm and relatively dry summers (Fang et al., 2007; Pomeroy et al., 2013). Runoff typically occurs as OF, and the risk of flooding on frozen, saturated soils in spring is considerable (Fang et al., 2007). At present, there is considerable surface drainage in the RRV but comparatively little subsurface (i.e., tile) drainage. However, increased frequencies of spring floods and intense summer thunderstorms are predicted under future climates (Dumanski et al., 2015), which will likely further increase the rate of tile installations in the future in response to increasing uncertainty in moisture in fields (Council of Canadian Academies, 2013). Although several studies in the RRV have attempted to evaluate the hydrologic function of tiles through modeling approaches (e.g., Cordeiro and Ranjan, 2015; Rahman et al., 2014; Satchithanantham and Ranjan, 2015), there remain several unknowns regarding the seasonal character of flow activation and runoff generation patterns in this landscape and climatic setting. Notably, the presence of frozen ground may impede infiltration, increasing the potential for HOF (Gray et al., 2001) and decreasing the potential for tile flow in spring. Under summer and fall conditions, threshold runoff responses to rainfall characteristics and AMC have been shown in the Canadian Prairies (Ross et al., 2017; Shook et al., 2015), but it is unclear if and how tile drainage may modify these relationships.

Vertisolic clay soils are prevalent in several parts of the world including substantial proportions of RRV (Brierley et al., 2011). Vertisols differ from other clayey soils by their ability to shrink and swell in response to moisture differences (Kurtzman et al., 2016). Dry conditions can lead to the formation of desiccation cracks that develop vertically in summer, offering preferential pathways for the rapid delivery of rainfall into tile drains. This has important implications for mobilizing phosphorus-rich overland flow into the subsurface (Kurtzman et al., 2016; Turtola and Jaakkola, 1995; Uusitalo et al., 2001), which could exacerbate ongoing water quality issues in large lakes within the region (Schindler et al., 2012). However, empirical evidence of the importance of such a mechanism – compared to that of other flow pathways – is lacking in the RRV.

An improved understanding of flow pathway activation is needed to evaluate the efficiency of tile drains from both agronomic and environmental perspectives. This is especially important in landscapes such as the RRV of the Northern Great Plains, which is an example representative of flat landscapes with poorly draining soils that require on-farm drainage infrastructure to allow agricultural production. Given current environmental and political concerns related to agricultural pollution and the eutrophication of surface water bodies, the role that tiles may play in both runoff and nutrient loading from agricultural fields must be evaluated. Studies that explicitly focus on the presence/absence of different flow pathways, as well as the temporal sequence of their activation, are needed to assess whether tile drainage systems can truly impact groundwater rise and surface runoff from occurring in near-level, clay-rich soils. The objectives of the current study were, therefore, to: 1) link the activation of different flow pathways to hydroclimatic drivers such as season, precipitation characteristics and antecedent moisture conditions, and 2) determine the timing of activation of different flow pathways (i.e., overland flow, tile flow, non-tile shallow subsurface flow as indicated by water table rise) relative to one another in a vertisolic clay landscape under a Prairie climate. The focus was on two adjacent fields (one with and one without tile drains) located in the Northern Great Plains as well as two different years of study to explicitly address issues of spatial and temporal variability in flow pathway activation.

3.2. Materials and methods

3.2.1. Site description

This study was conducted on a farm located near Elm Creek, ~70 km southwest from the City of Winnipeg, Manitoba, Canada (Figure 3.1a), within the RRV of Northern Great Plains. The 880 km-long Red River of the North flows northward to drain into Lake Winnipeg (Stoner et al., 1993) and its basin spans a vast area (116,000 km^2), partly covering the province of Manitoba in Canada and the states of North Dakota and South Dakota in the United States of America. The long-term mean annual temperature near the study site is 2.8^{o}C (1981-2010), with the mean annual maximum recorded in July (17°C) and mean annual minimum in January (-16°C). The seasonal soil-ice layer typically extends to ~0.75 to 1 m depth below the ground surface (He et al., 2015). Mean annual precipitation, which exceeds annual evapotranspiration, is 579 mm, with around 22% falling as snow (Environment Canada, 2017). Soils on the farm belong to the Gleyed Humic Vertisols of the Red River Series (US taxonomy: Gleyic humicryerts), and the slope of the fields is near-level (0-2 % slope) (Land Resource Unit, 1999).

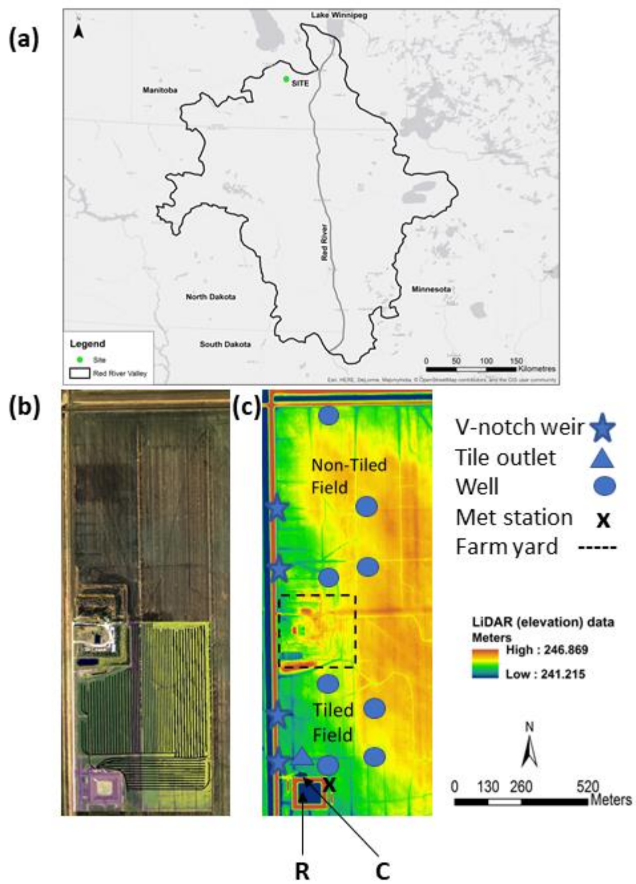


Figure 3.1. (a) Study site location. (b) Aerial image and experimental set up. (c) LiDar image. C indicates collection pond (2015) and lifting station (2016) and R denotes retention pond.

On the farm, two adjacent fields (500 × 500 m each), one tiled and one non-tiled, were instrumented (Figure 3.1b and Figure 3.1c). Although the fields are adjacent, they are separated by a farmyard and are not hydrologically connected. In the tile-drained field, 10 cm diameter tiles (Ideal Pipe) were installed in 2012 using Real-Time Kinematic (RTK) and differential global positioning system (DGPS) and a soil-max gold digger stealth ZD tile drainage plow. Soil from the same field was used to backfill trenches. Tiles have a spacing of 13 m, are at 0.2 % grade and approximately 85 - 120 cm depth and are connected to a 37.5 cm inner diameter header tile. In 2015, the header tile drained into a small collection pond (40 m diameter, ~4 m depth) (Figure 1c). However, in 2016, the collection pond was replaced with an automatic lifting station (6 m depth, 2 m diameter) (Jemco – Maxair, West Fargo, North Dakota). In both years, the collection pond/lifting station was equipped with a pumping system that transferred water to a larger retention pond (75 x 75 m x 3 m). Natural depressions and swales have been enhanced on both fields to drain into small, roadside ditches adjacent to the fields. The ditch adjacent to the non-tiled field drains to the northwest while the ditch adjacent to the tiled field drains to the southwest. Both fields were under a canola *(Brassica napus)* -spring wheat *(Triticum aestivum)* -soybeans *(Glycine max)* rotation (canola in 2015, spring wheat in 2016) and tilled to approximately 15 cm annually, in the fall.

3.2.2. Field Instrumentation and data collection

On each field, overland flow (OF) was monitored at 15-minute intervals at the outlets of two swales using V-notch weirs equipped with ultrasonic depth sensors (50A, Campbell Scientific, Edmonton, Canada) (Figure 3.1c). The surface runoff catchment areas were 20 ha in the tiled field and 15 ha in the non-tiled field. Although there were slight differences in the contributing areas for OF between the field, this is not anticipated to substantially impact the timing of OF activation.

Ultrasonic depth measurements were validated using capacitance sensors (Odyssey, Dataflow Systems Ltd.) attached to the V-notch weirs and periodic manual measurements. On the tiled field, both stage and velocity were measured at 15-minute intervals at the tile drain outlet using a Hach Flo-tote 3 and Hach FL 900 series logger (Hach, Ltd., accuracy ± 2% of reading). A pressure transducer (HOBO U20, Onset Corp., accuracy ± 0.5 cm), was placed into the tile drain outlet to monitor water levels at 15-minute intervals during periods when the other sensor did not function. A second pressure transducer was placed adjacent to the tile drain for barometric correction. A meteorological station (CR10x, Campbell Scientific Ltd., Edmonton, Canada) was established at the site to take hourly measurements of rainfall (TE525M, Texas Electronics) and air temperature (HMC45C, Campbell Scientific). In 2016, when the lifting station was added, an automatic water level logger (Mini Orpheus, Campbell Scientific, Edmonton, Canada) was deployed in the station to record water levels as a check against tile discharge rates. Groundwater table (GWT) levels were recorded using Odyssey capacitance water level loggers (Dataflow Systems Ltd) fitted inside slotted PVC wells screened with nylon (3.75 cm inside diameter, 1.5 m depth). Wells were installed along two transects (west to east direction) to characterize the upfield, midfield and downfield sections of each field (Figure 3.1c). However, downfield GWT data were found to strongly reflect the influence of nearby roadside ditches where water used to pond longer than the fields; consequently, these wells were excluded from the analysis and only upfield and midfield GWT data were considered further.

3.2.3. Runoff response and event characterization

Overland flow and tile flow (TF) were assumed to have been activated when water levels > 1 cm were detected by the sensors (on fields for OF, in the tile for TF). The 1-cm value was chosen to ensure that the detected changes were only deemed significant when they were noticeably larger than the resolution of the water depth measuring equipment. This one cm level at V-notch gauges roughly translates to 0.05 mm water depth in the field. A rise in groundwater levels of 5 cm or more in the wells was assumed to reflect subsurface flow (GWT) response to surface recharge. This translates to 1 mm of recharge if a specific yield of 2% is assumed (Johnson, 1967), and was chosen to ensure that only substantial GWT responses were identified as subsurface flow response after a rainfall event. For each event and flow pathway, the activation time was estimated as the delay between the start of rainfall and the initial detection (activation) of flow response. The precise estimation of flow pathway activation times during spring snowmelt was difficult due to uncertainties associated with the timing of widespread ground thaw, the existence of flow reversals from the ephemeral ditches into the field, and the submergence of tile outlet. Therefore, only runoff events that were triggered by rainfall events between April 1st and September 30th were considered. These events captured roughly 60 – 80% of the annual runoff in each year (Appendix B1). A runoff event was identified when OF, TF or GWT (or a combination of two or all) response was detected after a rainfall event. As such, 23 runoff events were identified during the 2015-2016 study period. Season-specific event dynamics were also examined, with seasons delineated based on calendar dates as well as general weather and soil conditions (e.g., presence/absence of soil-ice) and crop growth stage (i.e., pre-plant, growing, post-harvest) (Rattan et al., 2017) (Table 3.1). Spring (April 1st to May 31) events therefore included rain events occurring on wholly or partially frozen soils with cool air temperatures, no crop growth and wet antecedent conditions following snowmelt. Summer events occurred between June 1st and August 31st when soils were thawed and crops were actively growing. Summer storms were generally high-intensity, short duration thunderstorms which are typical of the Prairies (Dumanski et al., 2015). Fall events occurred in September 1st and September 30th on thawed, generally dry soils. Of the 23 rain events captured in this study, 12 occurred in 2015 and 11 in 2016. Overall, 8 were categorized as spring events, 11 as summer thunderstorms, and 4 as fall events.

Table 3.1. Summary statistics for hydroclimatic variables determined for the studied rainfall events in 2015-2016. AP: antecedent precipitation. Ant.:Antecedent GWT: groundwater table; GWT positions are expressed in meters below ground.

	Spring			**Summer**			**Fall**		
	Min	Max	Median	Min	Max	Median	Min	Max	Median
Rainfall characteristics									
Duration (hr)	5	41	17.5	1	12	2	2	33	6.5
Size (mm)	16.8	63.3	18.8	15.2	44.7	29.1	10	85	19.7
Maximum rainfall intensity (mm/hr)	3.2	8.6	7.2	10.8	36.3	19	4.1	30.2	6.8
Average rainfall intensity (mm/hr)	0.9	3.4	1.5	3.3	36.3	10.4	0.6	9.4	4.9
Antecedent moisture conditions based on AP									
7-day AP (mm)	0.7	48.2	11.7	0	52	14.2	0.3	19.3	1.7
14-day AP (mm)	3.4	66	22.2	15.3	86.1	49.1	1.9	37.6	24
Antecedent moisture conditions based on GWT									
Tiled field									
Upfield ant. GWT position (m)	0.55	1.3	1.05	1.13	1.37	1.31	1.22	1.4	1.39
Midfield ant. GWT position (m)	0.37	1.28	0.83	0.79	1.28	1.23	1.04	1.25	1.2
Non-tiled field									
Upfield ant. GWT position (m)	0.44	1.22	1.05	0.93	1.37	1.31	1.13	1.39	1.37
Midfield ant. GWT position (m)	0.12	1.01	0.9	0.85	1.17	1.03	0.81	1	0.95

For each event, a range of parameters were determined to understand the drivers behind the activation of individual pathways, and their activation relative to each other. Rainfall event characteristics such as duration, size, average and maximum intensity are summarized on a seasonal basis in Table 3.1. As studies have shown that antecedent precipitation (AP) is a reasonable proxy for AMC (Kim et al., 2014; Vidon and Cuadra, 2010; Williams et al., 2018b), cumulative rainfall amounts over the seven and fourteen days prior to each event, hereafter refer to as 7-day AP and 14-day AP, were also determined (Table 3.1). To characterize AMC more broadly and independently from rainfall data, several GWT-derived parameters were estimated, including antecedent (i.e., pre-event) GWT position and GWT position when TF was activated.

3.2.4. Data analysis and synthesis

Relationships between the runoff parameters and hydroclimatic drivers or antecedent GWT were evaluated using Pearson correlations. To evaluate the influence of the combination of rainfall characteristics and AMC on OF and TF activation times and/or timing parameters, multiple linear regression analysis was performed. To confirm normality according to Shapiro-Wilk test ($p > 0.05$), data used in the correlation and regression analysis were had to be log-transformed prior to analysis. All statistical analyses were performed using SAS 9.4 software (SAS Institute Inc. 2013) and SigmaPlot 12.5 software (Systat software, San Jose, CA).

In this study, seasonal (i.e., spring versus summer versus fall) and treatment (tile-drained versus non-tile drained field) differences were explored in: (1) the activation of OF, TF and GWT (i.e., presence/absence of individual pathways); (2) the activation of OF, TF and GWT, not only in absolute terms but also relative to one another; and (3) the sequence of flow pathway activation as a function of depth. The latter was notably done in the tiled field to gain insights on water movement through the soil profile. Potential activation schemes for the tiled field included: (i) a single pathway activated (and hence no depth-related sequence); (ii) "in-sequence" activation when activation times increased monotonically from the bottom up or the top down; (iii) "out-of-sequence" activation when activation times did not increase or decrease monotonically with depth; and (iv) “nonsequential” activation when two or more flow pathways were concurrently activated. For all three flow pathways (i.e., OF, TF and GWT), activation was inferred when the criteria outlined in section 3.2.3 were met. Activation times were estimated as the delay between the start of event rainfall and the time when activation criteria were first met or exceeded. In addition to GWT activation times, event-specific GWT dynamics were also quantified via the peak event GWT position and the maximum event GWT rise. As for TF dynamics, they were further characterized using timing parameters such as the time of rise (i.e., time lag between flow activation and peak flow) and the lag time (i.e., time interval between peak rainfall and peak flow). It should be noted that timing parameters were not estimated for the OF pathway given the data uncertainty associated with frequent back flow from the ditches into the fields.

3.3. Results

3.3.1. Flow pathway activation

Precipitation varied slightly during the study years (2015 = 508 mm, 2016 = 591 mm precipitation from March to September (Appendix B1). The largest event for 2015 was a spring storm that occurred in May with substantial OF and TF, whereas snowmelt runoff was the biggest event for 2016. Very little TF was observed during spring snowmelt events when compared to the activation of OF (Appendix B1). Multiday rain events dominated the springs of 2015, 2016 and fall 2015 whereas high intensity thunderstorms dominated the summers of 2015, 2016 and fall 2016 (Appendix B1, B2).

The responses of individual flow pathways differed both seasonally and between the tiled and non-tiled fields. For example, in spring, OF responses were observed for some but not all events (5 of 8 events in both the tiled and non-tiled fields). GWT responses were observed in all 8 spring events in the non-tiled field, but only 6 in the tiled field. TF responses were observed in 7 of the 8 spring events. OF was more prevalent in summer, as it occurred during all 11 summer events in both fields. In contrast, GWT and TF responses were not always observed in summer (GWT: 8 of 11 events in tiled field, 8 of 11 events in non-tiled field; TF: 9 of 11 events). OF response was observed during single fall event. While TF and GWT responses were observed across all fall events in the tiled field, GWT response was observed for only one of the 4 events in the non-tiled field.

3.3.2. Relationships between activation times and hydroclimatic conditions

Pathway-specific activation times were correlated with factors related to rainfall characteristics and AMC, but the identified relationships differed with both season and pathway (Table 3.2). For example, OF activation times were positively related to rainfall duration and negatively related to rainfall intensity (i.e., shorter times under higher intensity rainfall) when all events were considered together (Table 3.2, Figure 3.2a). When spring and summer events were considered independently, however, similar significant relationships were only observed in the non-tiled field during summer. Although OF activation times were not correlated with AP measures (Figure 3.2b), they were negatively correlated with peak water table position (i.e., shorter OF activation times with higher water tables, Table 3.2). These patterns were consistent between the tiled and non-tiled fields.

In contrast to OF, the relationship between GWT activation times, climate drivers and AMC varied both between the tiled and non-tiled fields and seasonally (Table 3.2, Figure 3.2). When all events were pooled together in the tiled field, upfield GWT activation times were correlated with average rainfall intensity and duration but not with AP measures or antecedent GWT position. When events were assessed by season, upfield GWT activation times in both fields were driven by rainfall intensity in spring (Table 3.2). Statistically significant negative correlations were, however, observed during spring between the non-tiled upfield GWT activation times and peak GWT (i.e., shorter GWT activation times with higher water tables).

TF response times – which include activation times, times of rise and lag times – were significantly related to both rainfall characteristics and various indicators of AMC (Table 3.2). For example, longer activation times were observed with longer-duration rainfall, whereas shorter activation times were observed with greater rainfall intensity as well as greater AP values (Table 3.2). For TF, relationships between activation times and maximum rainfall intensity were stronger than those between activation times and AP (Table 3.2, Figure 3.2 and 3.2). It should be noted, however, that rainfall intensities were greater for summer events than for spring events (Table 3.1, Figure 3.2), which may explain the seasonal differences in activation times seen in Figure 3.2. In terms of seasonal differences, during spring events, faster tile responses were observed with greater-intensity rainfall, higher AP values and shallower antecedent and peak GWT positions (Table 3.2). In summer, statistically significant relationships were found between TF response times rainfall duration, average rainfall intensity, 7-day antecedent rainfall, midfield antecedent GWT position and upfield and midfield antecedent and peak GWT positions.

Table 3.2. Pearson correlation coefficients between flow pathway activation times or timing parameters and hydroclimatic variables for the monitored events.

	Duration	**Rainfall intensity**		**AP**		**Upfield GWT**		**Midfield GWT**	
		Maximum	**Average**	**7-day**	**14-day**	**Ant.**	**Peak**	**Ant.**	**Peak**
(a) OF activation time									
Tiled-All events (n=17)	0.6**	-0.63**	-0.63**				-0.46		-0.52*
Non-Tiled-All events (n=17)	0.71***	-0.8***	-0.8***			-0.52*	-0.8***		-0.57*
Non-Tiled-Spring (n=5)								0.9*	
Non-Tiled-Summer (n=11)	0.63*		-0.61*						
(b) Upfield GWT activation time									
Tiled-All events (n=16)	0.67**	-0.48	-0.55*		-0.43				
Tiled-Spring (n=5)		-0.89*							
Tiled-Summer (n=8)									0.63
Non-Tiled-All events (n=14)	0.65**	-0.81***	-0.65**						
Non-Tiled-Spring (n=6)		-0.81*					0.89*	0.79	0.75
Non-Tiled-Summer (n=8)						-0.68			
(c) TF activation time									
All events (n=19)	0.81***	-0.83***	-0.86***	-0.47*	-0.73***				
Spring (n=7)		-0.91**		-0.72	-0.71		0.7	0.71	0.82*
Summer (n=9)	0.7*		-0.7*	-0.57			0.76*	0.76*	0.83**
(d) TF other timing parameters									
Time of rise-All events(n=19)	0.57**	-0.41	-0.49*				-0.48*		
Time of rise-Summer(n=9)				-0.63					
Lag time-All events (n=19)	-0.53*	-0.58**	-0.63**	-0.51*	-0.52*				
Lag time-Spring(n=7)					-0.72				
Lag time-Summer(n=9)	0.67*	-0.6	-0.65*			0.62		0.76*	0.77*

** $p < 0.001$; ** $p < 0.01$; * $p < 0.05$.
Blank cells signal correlation coefficients with $p > 0.1$. OF: Overland flow; TF: Tile flow; GWT: groundwater table. AP: antecedent precipitation; Ant.: antecedent

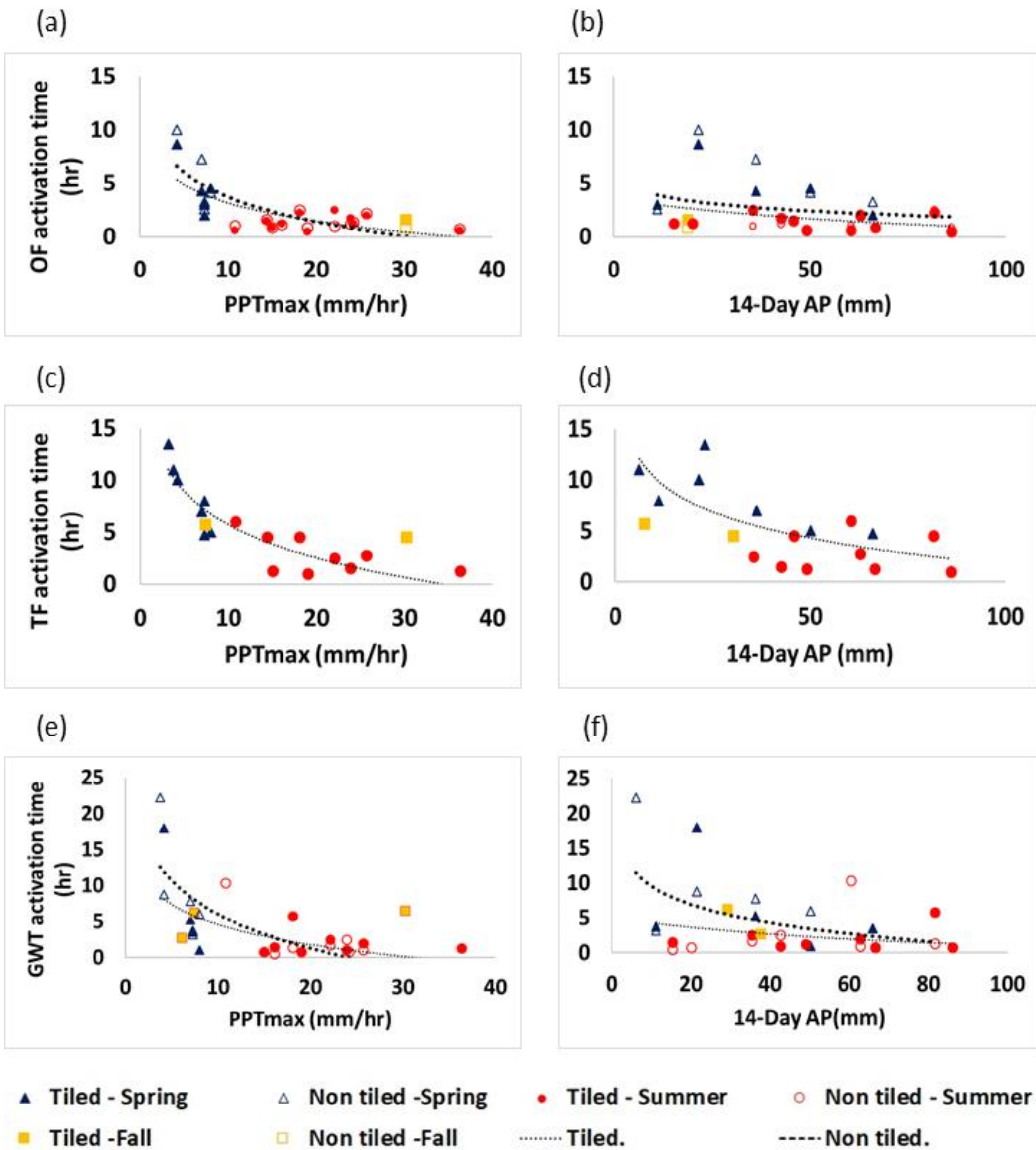


Figure 3.2. Relationships between the OF, TF and GWT (upfield) activation times, maximum event rainfall intensity (PPTmax) and 14-day antecedent precipitation (AP). OF: overland flow; GWT: groundwater table; TF: tile flow

In light of the statistically significant relationships between flow pathway activation times and rainfall intensity and AP, these climate drivers were subsequently regressed to evaluate the combined effects of rainfall intensity and antecedent precipitation on flow path activation times. The multiple regression results showed that OF response times could be better predicted with maximum rainfall intensity alone, without incorporating 14-day AP (Table 3.3). Relationships (r^2 and P values) did not improve appreciably when GWT activation times were regressed against both maximum rainfall intensity and AP. On the other hand, significant relationships were found between TF activation times and rainfall intensity (r^2 = 0.68) and AP (r^2 = 0.57) (Table 3.3). Using both maximum rainfall intensity and 14-day AP as explanatory variables also improved the strength of the relationship (r^2 = 0.78).

Finally, TF activation was compared with GWT positions (Figure 3.3) to determine if TF was activated when a threshold water table position had been reached. The water was above or close to the deepest tile drain (located 120 cm below ground) when the tiles were activated during spring and summer events, and the GWT positions were highly variable during the spring season. Positive seasonal relationships were observed between TF activation times and GWT position indicating rapid TF during shallow GWT positions (Spring, r^2 = 0.33; Summer, r^2 = 0.67) However, during some summer and fall events, TF was activated even though the GWT position was lower than the location of the deepest tile (Figure 3.3).

Table 3.3. Results of single and multiple linear regression analyses relating OF, TF and upfield GWT activation times to climatic variables. PPTmax: maximum event rainfall intensity; AP14: 14-day antecedent precipitation. The subscripts "Tiled" and "Non-Tiled" refer to the studied fields.

Activation time	Driver/s	Equation	r^2	*p* value
TF	PPT_{Max}	$Log(TF) = 1.6-(0.95*log(PPT_{Max}))$	0.68	<0.001
	AP_{14}	$Log(TF) = 1.6-(0.7*log(AP_{14}))$	0.57	<0.001
	PPT_{Max} & AP_{14}	$Log(TF) = 1.9 - (0.7 *log(PPTMax)) - (0.4 * log(AP_{14}))$	0.78	<0.001
OF_{Tiled}	PPT_{Max}	$Log(OF) = 1.1-(0.8*log(PPT_{Max}))$	0.39	<0.01
	AP_{14}	$Log(OF) = 0.9-(0.4*log(AP_{14}))$	0.13	
	PPT_{Max} & AP_{14}	$Log(OF) = 1.6- (0.7 *log(PPT_{Max})) - (0.3 * log(AP_{14}))$	0.5	0.01
$OF_{Non-Tiled}$	PPT_{Max}	$Log(OF) = 1.4-(log(PPT_{Max}))$	0.65	<0.001
	AP_{14}	$Log(OF) = 0.5-(0.2*log(AP_{14}))$	0.02	
	PPT_{Max} & AP_{14}	$Log(OF) = 1.4-(log(PPT_{Max})) - (0.001*log(AP_{14}))$	0.65	<0.001
GWT_{Tiled}	PPT_{Max}	$Log(GWT) = 1.1-(0.7*log(PPT_{Max}))$	0.23	0.06
	AP_{14}	$Log(GWT) = 1.4-(0.7log(*AP_{14}))$	0.19	0.09
	PPT_{Max} & AP_{14}	$Log(GWT) = 1.8-(0.5*log(PPT_{Max}))- (0.5*log(AP_{14}))$	0.33	0.07
$GWT_{Non-Tiled}$	PPT_{Max}	$Log(GWT) = 2-(1.4*log(PPT_{Max}))$	0.65	<0.001
	AP_{14}	$Log(GWT) = 1-(0.4*log(AP_{14}))$	0.07	0.36
	PPT_{Max} & AP_{14}	$Log(GWT) = 1.8-(1.5*log(PPT_{Max}))+ (0.2*log(AP_{14}))$	0.66	0.03

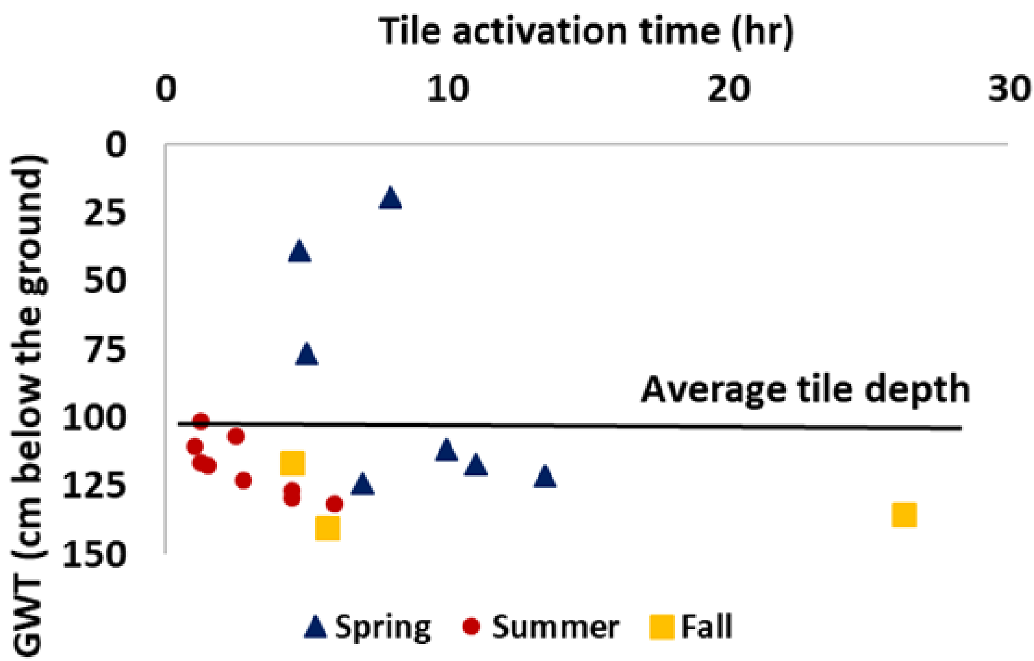


Figure 3.3. TF activation times as a function of antecedent upfield groundwater table (GWT) position.

3.3.3. Flow path activation sequence

Relative activation times: In general, OF was activated prior to the GWT (Figure 3.4a), irrespective of whether the event was in spring or summer, or on the tiled versus non-tiled field. In the tiled field, OF was also activated prior to TF in most cases (14 of 18 events, Figure 3.4b), although simultaneous OF and TF activation were observed during two summer events. While simultaneous OF-TF and OF-GWT responses were observed, they were uncommon and only occurred in response to summer storms in the tiled field (Figure 3.4). Simultaneous GWT-TF responses were observed for 3 summer events (Figure 3.4c), but it was more common for one pathway to respond before the other: there were 9 instances of TF being activated before the GWT and also 9 instances of TF rather being activated after the GWT.

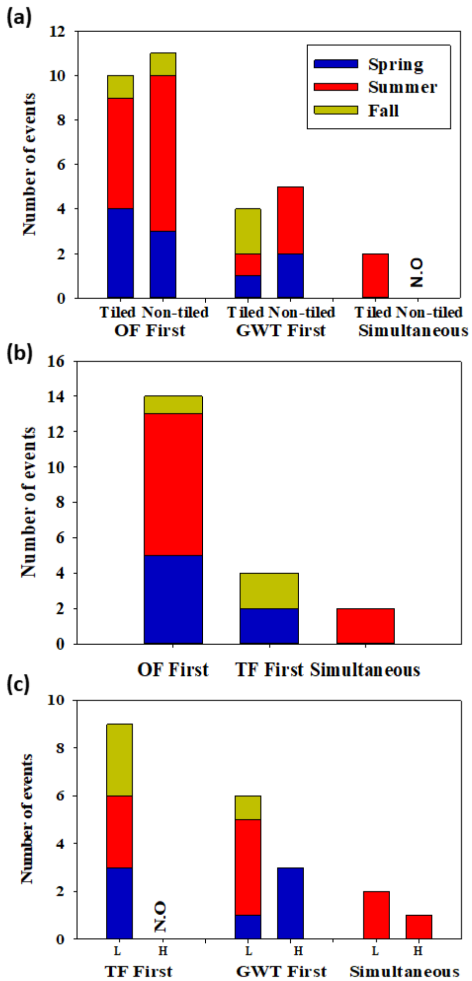


Figure 3.4. Relative activation of: (a) OF and upfield GWT in the tiled and non-tiled fields; (b) TF and OF in the tiled field; and (c) TF and upfield GWT in the tiled field across the monitored events. In panel (c), "L" stands for events with antecedent GWT > 1.2 m below ground, while "H" stands for events with antecedent GWT < 1.2 m below ground. OF: overland flow; GWT: groundwater table; TF: tile flow; N.O: TO not observed.

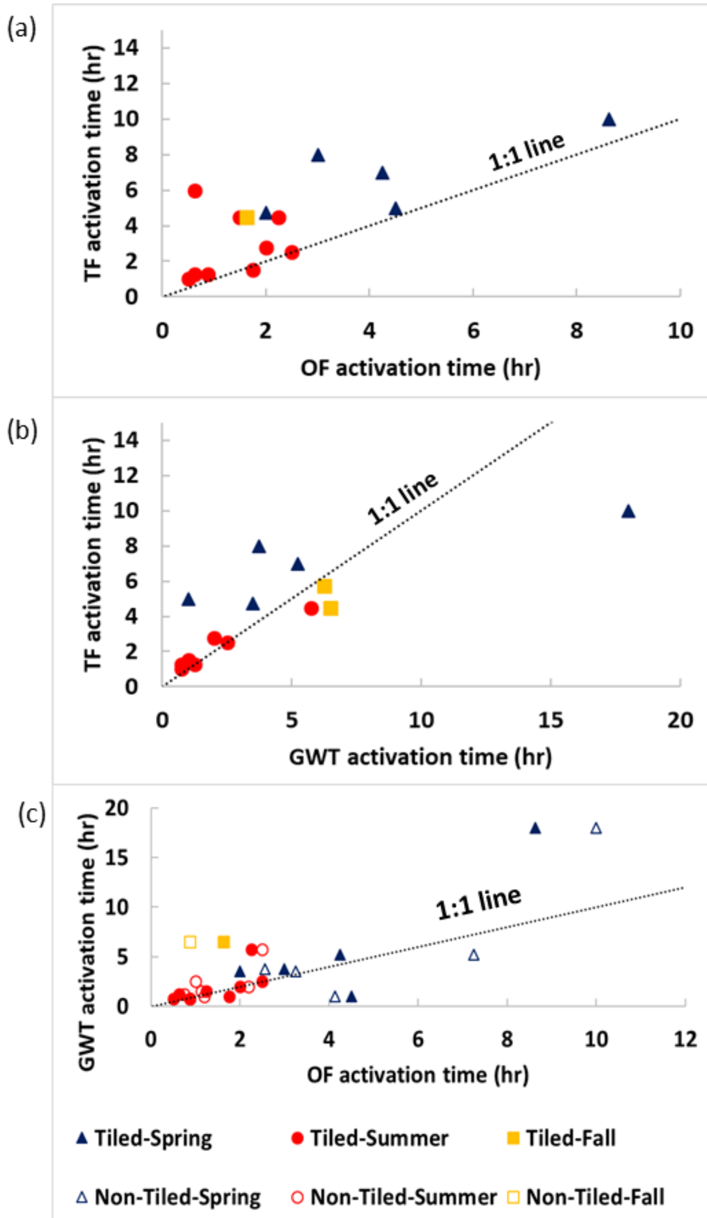


Figure 3.5. Comparison of flow pathway activation times on a seasonal basis. (a) TF versus OF in the tiled field; (b) TF versus GWT in the tiled field; (c) OF versus GWT in the tiled and non-tiled fields. OF: overland flow; GWT: groundwater table; TF: tile flow.

Irrespective of the flow pathway considered, shorter activation times were observed in summer than in spring or fall (Figure 3.5). Statistically significant positive relationships were found between the activation times of the different flow pathways for spring events (r^2 = 0.56 – 0.95, Figure 3.5). In summer, statistically significant relationships could not be found between the OF and GWT activation times, but a strong linear relationship was found between TF and GWT activation times (r^2 = 0.94). Although TF occurred prior to GWT in a few events, TF and GWT responses were generally simultaneous during summer events (1:1 line in Figure 3.5b).

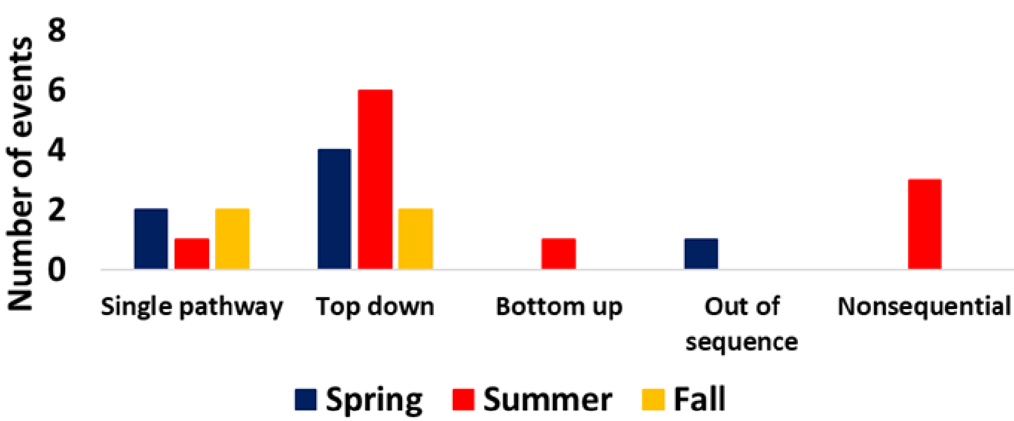


Figure 3.6. Vertical sequence of flow pathway activation in the tiled field across the monitored events. Refer to text for details regarding the activation sequence labels used in panel.

Activation sequence as a function of depth: Flow activation in the tiled field generally appeared to be from top to bottom, where OF activation was followed by either TF or GWT activation (Figure 3.6). For many spring events, the GWT was activated prior to TF, but specifically when GWT was above the deepest tile level (< 1.2 m below ground), suggesting that the flow pathway activation sequence was still from the top down (rather than from the bottom up). While there were events during which flow was activated from the bottom up (i.e., the GWT was activated ahead of TF, or prior to TF and OF (Figure 3.6), they were uncommon during the study period. There were also three summer events with nonsequential pathway activation in the tiled field where two or more pathways were activated simultaneously.

3.4. Discussion

3.4.1. Hydroclimatic controls on individual flow pathway activation

Rainfall intensity appeared to be the most important driver that influenced the activation of OF, TF and GWT at our study site, which is consistent with observations made in other sub-humid and semi-arid environments (Castillo et al., 2003; Siteur et al., 2014; Wainwright and Parsons, 2002). Within our data set, we observed distinct temporal runoff activation patterns for rainfall events less than 4 mm/hr (low intensity), 4-9 mm/hr (medium intensity) and > 10 mm/hr (high intensity) (Figure 3.2). In summer, HOF dominated in response to convective storms as rain intensities often exceeded the infiltration capacity of the clays. The relationship between rainfall intensity and OF activation times for summer events was however poor, likely due to canopy interception. A few rapid OF responses were also seen in early spring which were likely attributed to impeded infiltration due to partially frozen soils or due to a perched water table over the soil-ice layer (Figure 3.2a and 3.2b). Rapid OF responses were observed in spring at lower rainfall intensities than those necessary to activate OF in summer (Figure 3.2a and 3.2b). For instance, under spring conditions when infiltration was impeded, low or medium rain intensities could lead to HOF generation. Conversely, under fully thawed conditions, low-intensity, long-duration rainfall events allow more infiltration and saturation of the clay soils (Liu et al., 2011), thus favoring SOF (Castillo et al., 2003).

TF activation was also related to rainfall intensity. During prolonged low-intensity spring rains, the lack of relief and availability of more water for infiltration may have activated the tiles after a lengthy lag time by thawing the ice layer. Medium- to high-intensity rainfall events in late spring may have yielded a faster tile activation by raising the water table to or above the tile position, or by satisfying the water storage potential of the soil profile. During medium-intensity rainfall events, runoff may have begun as HOF but then changed to SOF when the water table rose to the ground level (Ross et al., 2017). High-intensity, short-duration summer thunderstorms likely triggered tiles by providing enough water for infiltration and vertical percolation. However, as noted earlier, TF activation times in summer were longer than OF activation times except for two summer events where simultaneous OF and TF activations were observed (Figure 3.4). This was likely attributable to a slowly propagating wetting front, as opposed to the rapid transmission of water into tile drains through preferential flow pathways. This is further evidenced by the fact that the GWT activation (when below the tile drains) also lagged behind OF. The impact of rainfall intensity on tile activation may have been suppressed by crop interception (Evrard et al., 2008). It should, however, be noted that rainfall data was recorded at an hourly time step at the field site, which may have been insufficient to capture the true effects of rain intensity on TF activation given that many Prairie thunderstorms deliver substantial amounts of rain within very short periods. GWT activation times were also significantly negatively related to rainfall intensity. However, this relationship was stronger in non-tiled field than the tiled field (Table 3.2). It is possible that the presence of tile drains in the field impacted this relationship by intercepting the matrix flow.

Relationships between indicators of AMC and flow activation times were weaker, compared to those with rainfall intensity. Although several studies in semi-arid areas have reported strong relationships between AMC and OF characteristics in general (e.g., Brocca et al., 2008; Li and Chibber, 2008) as well as seasonally (e.g., Castillo et al., 2003; Jarihani et al., 2017), such was not consistently the case here. This is likely because OF occurred primarily as HOF at our site. There was a negative relationship between OF activation and AP in the spring (Figure 3.2b), but it was not significant due to a single event where an early OF activation was observed after rain fell on partially frozen soils. In contrast to OF, TF activation showed a negative relationship with AP (Figure 3.2d) even though it was not as strong as the relationship between TF activation and rainfall intensity. The activation of tiles or subsurface flow pathways during wet antecedent conditions has been reported in mid-latitude temperate regions (Lam et al., 2016a; Macrae et al., 2010; Musolff et al., 2016; Outram et al., 2016). The activation of TF was most rapid when soils were wet and the water table was high. Studies have also suggested the existence of antecedent moisture thresholds for TF activation (Lam et al., 2016a; Macrae et al., 2007). At our site, this threshold for tile activation appeared to be achieved by the propagation of the wetting front or by elevating the water table (potentially via the capillary fringe).

When both rainfall intensity and AMC indicators were used in regression analysis as explanatory variables, the strength of the negative relationship of TF and GWT activation times against maximum rainfall intensity and AP improved in the tiled field, suggesting that the shortest TF or GWT activation times occur during high-intensity storms in relatively wet antecedent conditions (Table 3.3). This is comparable to the findings of Blaen et al (2017) who observed rapid TF with higher rainfall intensity and wetter antecedent conditions in a tiled UK agricultural landscape. In contrast, both OF and GWT activation times in the non-tiled field were mostly influenced by rainfall intensity, where predictability was not improved by the combination of rainfall intensity and AMC through regression analysis. This is consistent with the findings of Ziadat and Taimeh (2013) who showed rainfall intensity to be more influential than AMC for runoff generation in a non-tiled semi-arid landscape. Surface inundation and the lack of relief in the non-tiled field may have led to water ponding and affected the relationships between AMC, OF and GWT activation times, whereas additional moisture had been drained in the tiled field, yielding relatively stronger relationships. Our results show that hydroclimatic factors affected flow activation differently between the tiled and non-tiled fields, but such differences also varied temporally, as seen by the correlation coefficients that varied across seasons. Although hydrological models have been developed for vertisolic clays (e.g. Kurtzman et al, 2016), the performance of runoff models that strive to capture runoff dynamics in tiled landscapes with cracking clay soils can be improved. Our data suggest that models should include temporal variability in the activation of hydrological pathways in the region, and the impact of hydroclimatic drivers on activation patterns.

3.4.2. Process connectivity between natural and human-made flow pathways

Our focus on OF, GWT and TF pathways in the vertisolic soils of the RRV was motivated by the need to assess the extent to which human-made flow pathways might influence natural ones, notably by determining the moisture status of the vadose zone and thus affect “process connectivity” by coupling or decoupling surface and subsurface runoff generation mechanisms. Our results showed that flow pathway activation patterns varied not only seasonally but also among events. OF was activated ahead of TF and GWT during the majority of events (Figure 3.4), and the combined examination of OF and GWT data suggest that OF was due to infiltration excess rather than saturation excess. While the dominance of OF in the RRV had been extensively demonstrated before (Cordeiro et al., 2017; Dumanski et al., 2015; Tiessen et al., 2010), OF occurrence had not previously been examined in conjunction with TF and GWT on an event basis. The current study demonstrates that the activation times OF were comparable between tiled and non-tiled fields, indicating that tile drains do little to delay the activation of surface runoff atop vertisolic clays in the Prairies.

The data presented here also suggest that tiles are somewhat decoupled from the surface, despite the presence of preferential flow pathways in the vertisolic clays. This is contrary to previous studies that have observed simultaneous OF and TF responses in clay soils and attributed this to rapid connectivity between the soil surface and tile drains (e.g., Smith et al., 2015). Such dynamics are expected to be most prevalent during intense summer rainstorms on dry soils when desiccation cracks are maximized, as the montmorillonitic clays of the RRV form extensive networks of vertical and lateral cracks when their moisture is below field capacity (Brierley et al., 2011). Conversely, such dynamics are expected to be less prevalent under wetter conditions when desiccation cracks close due to swelling (Hardie et al., 2011; Robinson and Beven, 1983). However, simultaneous responses of OF and TF were seldom observed in the current study (2 of 12 summer events), even during especially dry conditions when cracks were present. Indeed, the activation of flow pathways mostly followed a “top down” sequence (Figure 3.6), where an approximate 2:1 relationship was found between OF activation times and TF activation times during summer storms (Figure 5a). Correlations between TF activation times and AP suggested that tile drain activation occurred most rapidly when soils were wet, which is when desiccation cracks would be expected to be least pronounced (Table 3.2). Some studies have reported the occurrence of preferential flow through cracks in deeper soils even when surface cracks are closed (Baram et al., 2012; Greve et al., 2010). At our study site, water flow from the surface to tile systems may have been impacted by cracks to some extent during the summer (faster TF activation in summer), but not entirely as the clays swell and may seal cracks once they are moist. Other preferential flow paths such as biopores may have been active at our site, irrespective of AMC;

however, they did not appear to be contributing to the rapid delivery of OF into tile drains at our site. Although the current study is investigating hydrologic connectivity, these findings have implications for nutrient transport via tile drains. Rapid tile drain responses are assumed to be related to high P concentrations in TF due to increased surface connectivity in clayey soils through preferential flow pathways (Turtola and Jaakkola, 1995; Uusitalo et al., 2001). Consequently, the expansion of tile drains in regions where there is potential for such connectivity (such as the clays in the RRV basin that desiccate and crack under the Prairie climate) is controversial. Given the hydrological behaviors or tiles in the current study, we do not anticipate that P concentrations will be highly elevated in TF. In contrast, longer lag times may increase nitrate losses in TF due to increased interaction with soil matrix. Future studies are needed to evaluate the chemistry of OF and TF and their relative contributions to edge of field nutrient losses given their environmental and policy significance.

In the current study, there were periods during which the water table was elevated (above the tile) but the tiles were not flowing and hence, GWT activation preceded TF activation (Figure 3.3). This was notably the case when a perched water table atop frozen ground existed above the tiles. Such occurrences were, therefore, restricted to spring events. Rapid GWT activation was also observed in summer but appeared to be related to a combination of precipitation intensity/soil infiltration capacity and AMC (either as indicated by antecedent precipitation or antecedent GWT position) as the water table is most frequently below the tile drains during the dry Prairie summers. Early GWT activation in this landscape could have been attributed to groundwater table ridging (reverse Wieringermeer effect) or the Lisse effect in the summer, where the water table rises sharply after medium- to high-intensity rainfall events due to the rapid addition and transfer of extra pressure head into the capillary fringe (Khaled et al., 2011; Waswa and Lorentz, 2015). This is often followed by a sharp decline in water table position. The initiation of OF was observed during only one high intensity fall event, likely because low-intensity rain on thawed soil may have favored more infiltration.

It should be noted that the current study was conducted over two years, and 2016 received more seasonal precipitation than 2015. There is a possibility that variability induced by large weather patterns may not be recognized by such short-term studies (Koivusalo et al., 2017). Given the paucity of field data on tile drainage in the Northern Great Plains, additional longer-term field studies are needed to capture climatic variability. In addition, the tile drainage system at our site was 3 to 4 years old when these measurements were taken. Given that the age of tile drainage systems may influence soil and hydrologic behavior (Messing and Wesstrom, 2006), continued observations are required to evaluate if and how field hydrology and soil structure may evolve in future.

3.5. Conclusions

The main goal of this study was to add to the body of knowledge on dominant runoff generation processes, in the specific context of near-level, sub-humid, clay-rich, agricultural landscapes where natural and human-made flow pathways co-exist. Specific research objectives targeted the quantification of activation times for OF, TF and GWT, as well as the identification of hydroclimatic controls on flow pathway activation. Field data collected across two years and about two dozen runoff events revealed that both rainfall characteristics and antecedent moisture conditions control pathway activation, although to a different degree based on pathway type and season. OF and GWT activation was notably driven by rainfall intensity whereas TF was, rather, under the combined influence of rainfall intensity and antecedent moisture conditions. During the study period, the presence of tile drainage did not lead to the anticipated reduction in OF frequency.

TF activation through direct preferential flow was uncommon in the vertisolic soils studied, despite the dry Prairie summers. Rather, unsaturated water flow through the soil into tiles occurred either via a slowly propagating wetting front or wetting front advancement with partial coupling with OF, often resulting in delayed TF and GWT activation. These findings have practical implications for the management of water resources in this portion of the Canadian Prairies. This study suggests that tile drains will do little to offset the rapid runoff generation of surface runoff and its associated mobilization of nutrients such as phosphorus. The fact that tile drains did not exhibit the rapid flow responses associated with significant preferential flow may also have implications for the impacts of tile drainage on nutrient transport. Future studies should evaluate the impacts of tile drainage on edge-of-field runoff and nutrient losses.

3.6. Acknowledgements

This project was funded by the Manitoba Conservative Districts Association (MCDA), Environment Canada's Lake Winnipeg Basin Stewardship Fund and the Natural Sciences and Engineering Research Council of Canada (Macrae – DG). CM Classen Farm and Northern Plains Drainage Systems Ltd., Brendan Brooks, Bo Pan and Kui Liu are thanked for the field assistance.

Chapter 4: Temporal variability in water and nutrient movement through vertisols into agricultural tile drains

4.1. Introduction

Impairment of surface water bodies from agricultural runoff is a global environmental problem (Le Moal et al., 2019). In freshwater systems, this is driven by phosphorus (P) loads from fertilizers and livestock, whereas in marine ecosystems, nitrate (NO_3-N) is of greater concern (Paerl, 2009). Agricultural runoff can occur either as surface or subsurface (tile drain) runoff; however, the relative contributions of these two pathways can vary. In the Great Lakes Region of North America, the majority of agricultural runoff travels via subsurface flow in tile drainage (Macrae et al., 2019; Plach et al., 2019; King et al., 2015). In contrast, in colder regions, such as the Northern Great Plains of North America, most runoff occurs as overland flow due to the presence of deep seasonal frost that restricts infiltration when the majority of annual runoff occurs (Baulch et al., 2019; Dumanski et al., 2015; Kokulan et al., 2019b). Although most annual runoff has historically been associated with snowmelt runoff in the Northern Great Plains, there has been an increase in the occurrence of large spring and summer events (Shook and Pomeroy, 2012).

With the uncertainty in modified precipitation regimes under a changing climate and the potential for more frequent major storm events, there has been an expansion of tile drainage into areas like the Red River Valley in the Northern Great Plains (Cordeiro and Ranjan, 2015; Council of Canadian Academics, 2013; Kokulan et al., 2019a). Runoff and nutrient loss via tiles are currently a small proportion of the annual budget due to the presence of seasonally frozen ground during snowmelt (Kokulan et al., 2019b) and the prevalence of high-intensity thunderstorms in summer that generate infiltration-excess overland flow (Kokulan et al., 2019a). However, runoff into tile drains can be considerable during the longer-duration, low-intensity rain events, (Cordeiro and Ranjan, 2015; Kokulan et al., 2019b). Given the water quality issues associated with tile drainage in warmer regions (e.g. King et al., 2015; Jarvie et al., 2017), agricultural tile drainage has the potential to become more problematic in the future within the Northern Great Plains. An improved understanding of subsurface hydrological and biogeochemical processes in tile drained systems in the Northern Great Plains region is needed to assess the potential for the expansion of tile drainage to exacerbate current water quality issues.

Soil type is a critical factor that influences subsurface water and nutrient dynamics. For example, sandy textured soils are less effective at retaining water and agricultural nutrients (e.g. P and nitrogen (N)) within the vadose zone (Gaines and Gaines, 1994). In contrast, in loam soils, there is significant retention of P (e.g. Beauchemin et al., 1998; Plach et al., 2018b). However, although clay soils possess higher nutrient retention capacities, these systems can also contribute to elevated nutrient losses because of preferential flow pathways (Simard et al., 2000; King et al., 2015). Indeed, preferential flow pathways facilitate the rapid delivery of water and nutrients into subsurface drainage systems and deep groundwater reserves, bypassing the natural buffering ability of subsoils (Jarvis, 2007). Preferential flow paths can form because of biological activities (biopores) such as earthworm burrows and root channels or because of desiccation cracks that appear due to the presence of 2:1 clay mineral such as smectites (Jarvis, 2007).

Vertisolic clay soils occupy a substantial proportion of the global agricultural lands, including North America (Kurtzman et al., 2016). In North America, the vertisols are often found in regions with historically drier climates such as the Red River Valley basin of the Northern Great Plains (Southern Manitoba in Canada, North Dakota and South Dakota and Minnesota in the United States) and in parts of the Southern Great Plains (e.g. Texas, USA) (Harmel et al., 2019; Kokulan et al., 2019a). Vertisols shrink during drier periods and swell during moist conditions because of smectite clay minerals (Brierly et al., 2011). When vertisols shrink, desiccation cracks form that can act as potential preferential flow pathways for water and nutrients. Under dry conditions, the widths of these cracks can be greater than 7.5 cm, and they can extend down to 1–1.2 m depth (Brierley et al., 2011; Bagnall et al., 2019). During such periods, the rapid transport of infiltrating water into deeper soil layers can occur (Bagnall et al., 2019); however, this rapid infiltration is substantially reduced once soils become moist (Kokulan et al., 2019a; Bagnall et al., 2019). In addition to antecedent soil moisture, the activation of artificial subsurface drainage is influenced by precipitation characteristics (e.g. Vidon and Cuadra, 2010; Kokulan et al., 2019a), with increased rapid transport during high-intensity rainfall. However, if and to what extent these differences impact subsurface nutrient chemistry are unclear.

An additional complicating factor in assessing the impact of subsurface nutrient losses in the cold regions is the presence of seasonally frozen ground. Due to the severe Prairie climate during winter, the soils of the Northern Great Plains develop a thick soil-ice layer that extends to 0.75 to 1 m depth (Kahimba et al., 2009). The influence of this frost layer on infiltration and subsurface water movement depends on factors such as the thickness of the frost layer, the soil texture, the antecedent soil moisture prior to freeze up (or the occurrence of freeze-thaw events over winter that can refreeze in pores), and, the macropore network (Demand et al., 2019; Fouli et al., 2013; Grant et al., 2019a; Gray et al., 2001; Mohammed et al., 2018). For example, in a laboratory study, Grant et al. (2019a) found that seasonal frost enhanced nutrient transport in the vadose zone due to the presence of macropores in the frozen soil. In vertisols of the Red River Valley, Kokulan et al., (2019b) found that the contribution of tile drains to total runoff (Overland flow + tile flow) and nutrient loss was small relative to overland flow; however, less is known about the seasonal-scale differences in nutrient transport into tile drains.

Researchers have deployed various physical, chemical, and modelling approaches to capture the dynamic nature of preferential flow pathways, such as lysimeters (Greve et al., 2010), tension infiltrometers (Watson and Luxmoore, 1986), dye tracers (Ali et al., 2018; Demand et al., 2019; Grant et al., 2019b), stable isotopes (Klaus et al., 2013; Williams et al., 2016), conservative tracers such as Cl^- and Br^- (Grant et al., 2019a), and semi-conservative tracers such as electrical conductivity (EC; Heppell et al., 2002; Michaud et al., 2019). The outcome from these studies has led to an improved understanding of the homogenous and heterogenous water and nutrient movement through the vadose zone from the pedon to the catchment scale (Jarvis et al., 2016). However, many of these approaches provide insight into spatial patterns, or, provide insight over short timescales, and few have linked water movement through vadose zone into tile drains to nutrient transport into tile drains at the event and seasonal scales. Furthermore, challenges remain in assessing the macroporous nature of certain soils such as vertisolic clays because of the dynamic nature of desiccation cracks (Kurtzman et al., 2016). An improved understanding of water quality in tile drainage in vertisolic clays that experience seasonal frost is needed to assess the potential for the expansion of tile drainage in these regions to exacerbate water quality issues. Therefore, the specific objectives of this study are to 1) characterize seasonal patterns in tile flow and chemistry under variable hydroclimatic conditions, and 2) to characterize temporal variability in soil water movement in the vadose zone under different climatic conditions using direct field-scale measurements.

4.2. Methods

4.2.1. Study site

This work was conducted on a 25-ha tile-drained farm in Elm Creek, Manitoba. The site is located within the Red River Valley basin, roughly 70 km southwest of the City of Winnipeg, Canada.

Long term mean annual temperature of the Elm Creek area is about 2.8°C (1981-2010), with January being the coldest month (mean temperature of -16°C) and July being the warmest month (17°C). During the long cold winter season, a thick soil-ice layer develops to 0.75-1.00 m depth in the region (Kahimba et al., 2009). The mean annual precipitation is approximately 580 mm, of which 130 mm falls as snow (Environment Canada, 2017). Soils of this farm belong to the Humic Gleyed Vertisols of the Red River Series (US taxonomy: Gleyic humicryets). The topography of the site is nearly flat with a 0.3% average slope, which is typical for the Red River Valley.

The farm follows a canola (*Brassica napus L.*, (2015)), spring wheat (*Triticum aestivum L.*, (2016)) and soybean (*Glycine max L. (Merr)*, (2017)) crop rotation and is annually tilled to 15 cm depth in the fall season. Mineral fertilizers are applied in spring. Phosphorus is subsurface seed placed as mono ammonium phosphate (40 kg ha^{-1} in 2015, 45 kg ha^{-1} in 2016, and 20 kg ha^{-1} in 2017), whereas the N is surface broadcast as urea (127 kg ha^{-1} in 2015 and 123 kg ha^{-1} in 2016) and ammonium sulphate (22 kg ha^{-1} in 2015 and 2016) . Nitrogen fertilizers were seed placed in 2017 (7 kg ha^{-1} urea and 9 kg ha^{-1} ammonium phosphate).

In 2012, 10 cm diameter lateral tile drains were installed at 85-120 cm depth at 13 m (40 feet) intervals across the entire field. These lateral drains passed into a 37.5 cm diameter main tile that drained into a collection pond until 2015. Water was subsequently pumped from the collection pond into a larger retention pond at the site. In 2016, the collection pond was replaced by an automatic lifting station that pumped water from the lifting station into the retention pond.

4.2.2. Experimental Design

Tile drainage discharge and water chemistry were monitored over a three-year period (2015-2017 open water seasons) and compared with hydroclimatic drivers of runoff quality and quantity. In 2017, a series of infiltration and slug tests were done under a range of antecedent temperature and moisture conditions in the tile drained field to better understand vadose zone processes in the vertisolic clay and explain the observed patterns in tile drain chemistry by exploring hydraulic conductivity at different depths in the soil.

4.2.2.1. Field Monitoring Work

Tile flow and sample collection

Tile flow was monitored at the outlet with a Hach Flo-tote 3 and Hach FL 900 series logger (Hach, Ltd.,) at 15-minute intervals. A HOBO U20 (Onset Corp.) was also placed in the header tile to record the water level as a backup. In 2016, another automatic water level logger (Mini Orpheus, Campbell Scientific Ltd.) was installed inside the tile lifting station to compare tile flow with water levels at the lifting station.

Tile flow samples were collected in 1 L acid-washed polyethylene bottles at 2 to 6-hour intervals with programmable AS 950 (Hach Ltd.) autosamplers. Samples were collected to span the entire event hydrograph. Manual grab samples were also collected during low flow periods and during autosampler failures. A 200 mL sub-sample was filtered through a pre-weighed 0.45 µm Whatman glass microfiber filter paper. Once filtered, filter papers with sediments were dried at 105 ^{o}C for 24 hours and weighed for total suspended solids (TSS). A 100 mL sub-sample was used to measure pH and EC with an Accumet Basic AB 15 pH meter and an Accumet Basic AB 30 EC meter respectively (Fisher Scientific.). The filtered sub-sample was kept in a refrigerator at 4 ^{o}C and analyzed for soluble reactive phosphorus (SRP) (QuikChem Method 10-115-01-1-A) and nitrate (NO_3-N) (QuikChem Method 10-107-04-4-C) within 48 hours from collection with a QuikChem 8500 series 2 FIA system (Lachat instruments) through flow injection colorimetry (Lachat Applications Group) and flow injection analysis (Lachat Applications Group) respectfully. A non-filtered sub-sample (200 mL) was stored at -18^{o}C and was analyzed for total phosphorus

(TP) in a later day with QuikChem 8500 series 2 FIA system (QuikChem Method 10-115-01-4-C, Lachat Applications Group).

Overland flow and sample collection

Overland flow was monitored from two v-notch weirs at the edge of the field. Runoff water samples were collected in 1 L acid-washed polyethylene bottles at 2 to 8-hour intervals with programmable AS 950 (Hach Ltd.) autosamplers during overland flow occurring events. Upon bringing to the laboratory, a 100 mL sub-sample was used to measure EC with the Accumet Basic AB 30 EC meter.

Hydroclimatic variables

Hourly precipitation and air temperature measurements were recorded by an on-site station (precipitation: TE525M, Texas Electronics; air temperature: HMC45C, Campbell Scientific Ltd.) connected to a CR10x (Campbell Scientific Ltd.) data logger. The water table was monitored at 6 locations throughout the field using Odyssey capacitance water level loggers (Dataflow systems Ltd.) at 15-minute intervals to understand tile-groundwater interactions. Polyvinyl chloride pipes (3.75 cm inner diameter; 1.5 m depth) screened with nylon were used to house the groundwater level loggers. Frequent groundwater samples were collected during late spring 2015 and throughout 2016 open water season. A small plastic bucket attached to a metal wire was used to collect groundwater samples from the wells. The bucket and wire were thoroughly washed with reverse osmosis water before collecting water samples from a particular well. Groundwater was not sampled in early spring and summer of 2015 and throughout 2017 due to logistical issues. Upon bringing to the laboratory, the EC of the groundwater samples was measured with the Accumet Basic AB 30 EC meter.

4.2.2.2. Experimental Work

Soil Hydraulic Conductivity

A series of piezometers were installed in a randomized complete block design in fall 2016 (October) to study the temporal variability in vertical and horizontal water movement in vertisolic clay profile (Figure 4.1). Piezometers (n=15) were made with 1.5” ABS pipes and installed along a transect at 3 m intervals. Each nest (also spaced at 3m intervals) had piezometers screened at 3 different depths, 25-35 cm (shallow), 65-75 cm (medium) and 90-100 cm (deep). The 90-100 cm depth represented the depth of the tile drains in the particular area of the field. The transect of piezometers was placed both above (rows 1,2) and between (rows 3-5) tile laterals.

Soil hydraulic conductivity was measured at each of the three depths using the falling head slug test method (Hvorslev, 1951). Four applications were carried out throughout the year of 2017 under different antecedent moisture conditions, including frozen/partially frozen (April 12th, early spring snowmelt runoff), wet (June 27th, following a series of rainstorms), dry-summer (July 21st) and dry-fall (October 14th). For each test, a known quantity of water (500 mL in 25-35 cm piezometers; 1 L in 65-75 and 90-100 cm piezometers) was rapidly added to the piezometer, and changes in water levels were monitored at 30 s intervals using Hobo U20 water level data loggers (Onset Ltd). Subsurface hydraulic conductivity (K) was estimated with the Hvorslev (1951) method as follows,

$$K = r^2 \ln(L/R)/2LT_0$$

where r is the inner radius of the ABS pipe (m), L is screen length (m), and R is borehole radius (m). T_0 was estimated from 0.37 Ts from H/H_0 semi-log graph where H is the water level (m) at time T (s), and H_0 is the reference water level (m). In addition to the experimental work in 2016, piezometers in the first and second rows were emptied prior to the onset of snowmelt runoff in 2017 and Hobo U20 water level data loggers (Onset computer corporation) were added to monitor water percolation at 25-35, 65-75 and 90-100 cm depths at 15-minute intervals throughout the 2017 spring snowmelt runoff period.

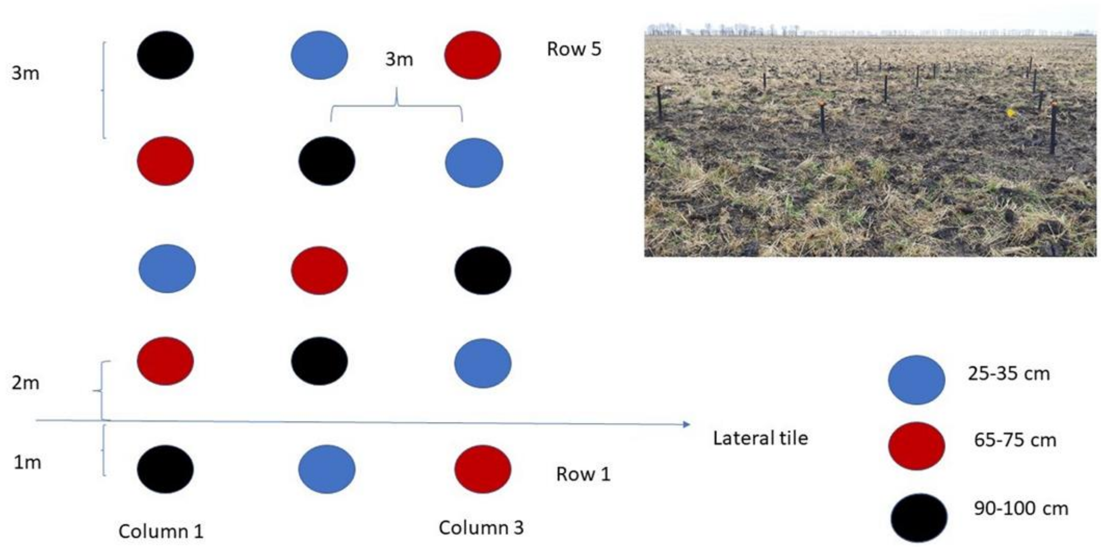


Figure 4.1. Layout of piezometer installation for slug tests and for monitoring snowmelt water movement.

Surface infiltration measurements

Surface infiltration was measured using the falling head method in 3 pairs of double-ring infiltrometers (Eijkelkamp Soil & Water) on the same day as the slug tests. The inner and outer ring diameters of the double-ring infiltrometer pairs were ~30 (inner) and ~55 cm (outer), and the height of the infiltrometers was 25 cm. A driving plate and shock absorbing hammer were used to drive infiltrometers into the soil to ~ 5 cm depth. The outer ring was filled with water before the inner ring. Water levels were measured with Hobo U20 water level loggers at 5-minute intervals for the April and June applications and at 30 s intervals for the July and October applications.

Surface saturated hydraulic conductivity was estimated from the method described by Reynolds (2008).

4.2.3. Data analyses

One-way repeated measures ANOVA test was used to assess the differences in seasonal water movement in the ring infiltrometers and piezometers (Sigmaplot, Version 12.5). The normality of the data was assessed with the Shapiro-Wilk test, where differences were deemed to be significant at the arbitrary $p < 0.05$ level.

Tile flow was estimated from the readings from a Hach Flo-tote 3 and a Hach FL 900 series logger (Kokulan et al., 2019b). During periods when the tile and collection pond were fully submerged and thus inaccessible, tile flow was estimated using observed water levels in the collection pond or lifting station and the flow rate (7 L s^{-1}) of the manual pump that transferred tile water from the collection zone to the larger on-farm retention pond. Daily flow-weighted mean concentrations (FWMCs) of SRP, TP, EC, TSS and NO_3–N were estimated using the discrete high-frequency water chemistry samples and the continuous flow data (after Williams et al., 2015). The same protocol was followed to determine the EC of the overland flow samples. The daily mean EC was used to estimate the EC of the groundwater.

Spearman correlations were used to identify significant relationships among tile flow and the various water quality parameters. To assess seasonality in water chemistry, the data were divided into early spring (snowmelt runoff (March 10th to 16th in 2015, March 10th to April 2nd in 2016 and March 20th to April 12th in 2017)), late spring (runoff commencing after melt was over until May 31st) and summer (June 1st to August 31st). Electrical conductivity was used to explore potential sources of tile water, and EC was also compared with GWT levels to characterize surface-subsurface connectivity.

4.3. Results

4.3.1. Seasonal patterns in tile drainage under variable hydroclimatic conditions

At the study site, there was strong seasonality in tile flow, precipitation patterns, and groundwater table position (GWT), with some of these patterns consistent across all study years as well as some subtle differences. During the snowmelt periods of all three years, there was minimal flow (< 1 mm day^{-1}) from tiles despite the presence of a shallow GWT in all three years throughout this period (Figure 4.2). Water levels and temperatures on the surface and in the subsurface (in shallow piezometers) recorded during the snowmelt period of 2017 confirmed the presence of frozen ground at 75 and 100 cm depth (Figure 4.3), which likely impeded tile flow during this time. The shallow GWT gradually lowered following snowmelt before it rose once again above tile levels (~1m) during subsequent multi day spring storms in the late spring, following the thaw of ground frost. Tiles responded to these late spring rainstorms, yielding substantial flows where 82% of annual tile flow in 2015 and 54% in 2016 occurred during the late spring. In contrast, tiles did not flow in the late spring of 2017 due to a lack of precipitation and a deeper GWT. In summer, tile flow and GWT responses to thunderstorms were small despite the greater magnitudes and intensities of the storms during that period. Tile responses during the fall season differed with antecedent moisture conditions. For example, substantial tile flow was observed in September 2016 following a large thunderstorm (85 mm) on moderately wet antecedent conditions (2016 was 17% wetter than 30-year climate normal).

In contrast, tiles did not respond to a storm that was similar in magnitude (60 mm in size) in September 2017 that fell on dry antecedent conditions (2017 was 22% drier than the 30-year climate normal, Environment Canada, 2017).

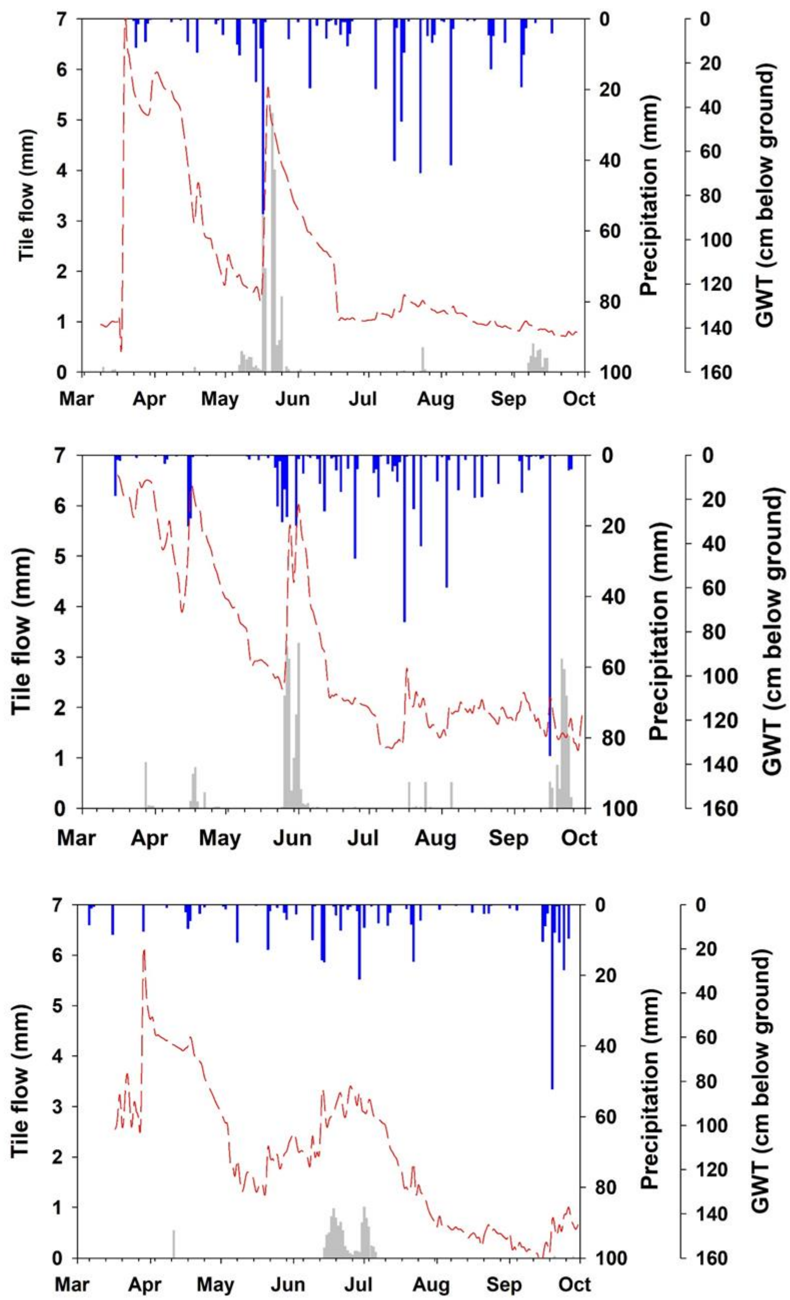


Figure 4.2. Daily tile flow (vertical gray bars), precipitation (inverted blue bars) and groundwater table (GWT) levels for 2015 (a), 2016 (b) and 2017 (c) study periods.

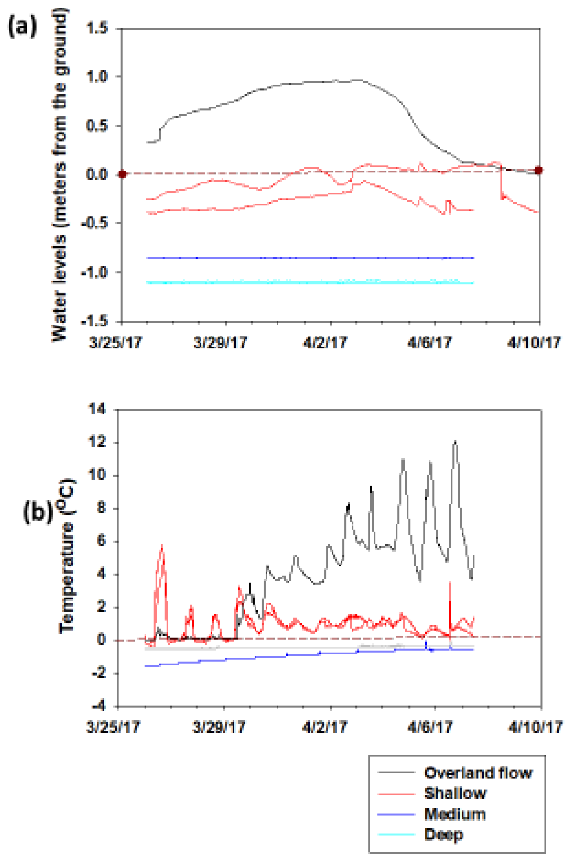


Figure 4.3. Water levels (a) and temperatures of (b) of overland flow and piezometers (two at each depth) during 2017 snowmelt runoff period. Screened depths of the piezometers below ground were 25-35 cm (shallow), 65-75 cm (medium) and 90-100 cm(deep). The dashed horizontal line shows the ground levels (a) and 0oC (b).

4.3.2. Infiltration capacities and hydraulic conductivities under different antecedent conditions

Infiltration capacities differed with both antecedent moisture conditions and the presence/absence of frozen ground. Infiltration capacities were lesser under wet antecedent conditions (April, June), and greater and more variable under dry antecedent conditions (July, October, Figure 4.4, $p < 0.1$,

Appendix C1). Infiltration capacities on moist, frozen ground (April) exceeded those on thawed, wet soils (June, Figure 4.4). Field infiltration capacities were two orders of magnitude higher in double-ring infiltrometers during drier antecedent conditions (July 2017) and one order of magnitude higher during frozen conditions (April 2017) when compared to wetter antecedent conditions (June 2017).

Hydraulic conductivities of subsoils, experimentally determined from slug tests in piezometers, were also variable under different antecedent moisture and temperature conditions. Hydraulic conductivities were lowest during frozen conditions in all shallow, medium, and deep piezometers (Figure 4.5). During frozen conditions, hydraulic conductivity was greatest in shallow piezometers when compared to medium and deep piezometers. This is consistent with field observations of water level fluctuations in field piezometers during the snowmelt period (Figure 4.3). Hydraulic conductivities increased in the medium and deep piezometers after soils thawed in June, and continued to rise in July and October as conditions became progressively drier (Figures 4.2, 4.5). However, this was not observed in shallow piezometers as a reduction in hydraulic conductivity was observed in summer (July) when compared to June and October.

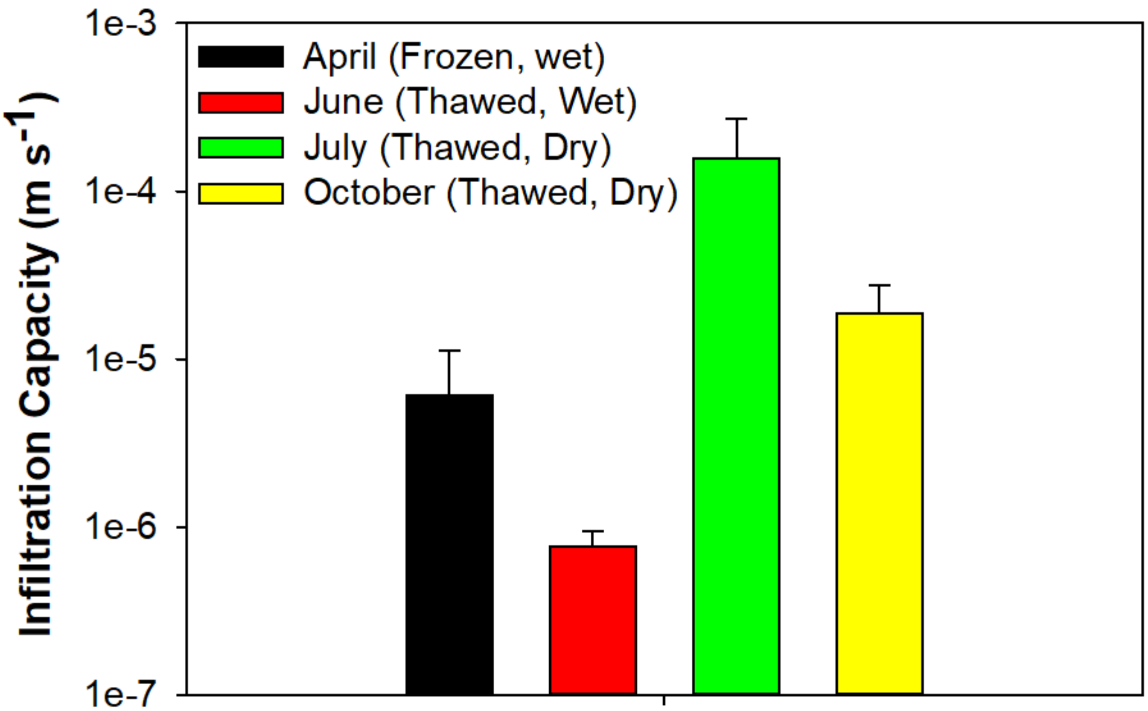


Figure 4.4. Surface infiltration capacities determined experimentally in 2017 under variable antecedent moisture and temperature conditions. Error bars represent standard error of the mean estimated with generalized mean squares.

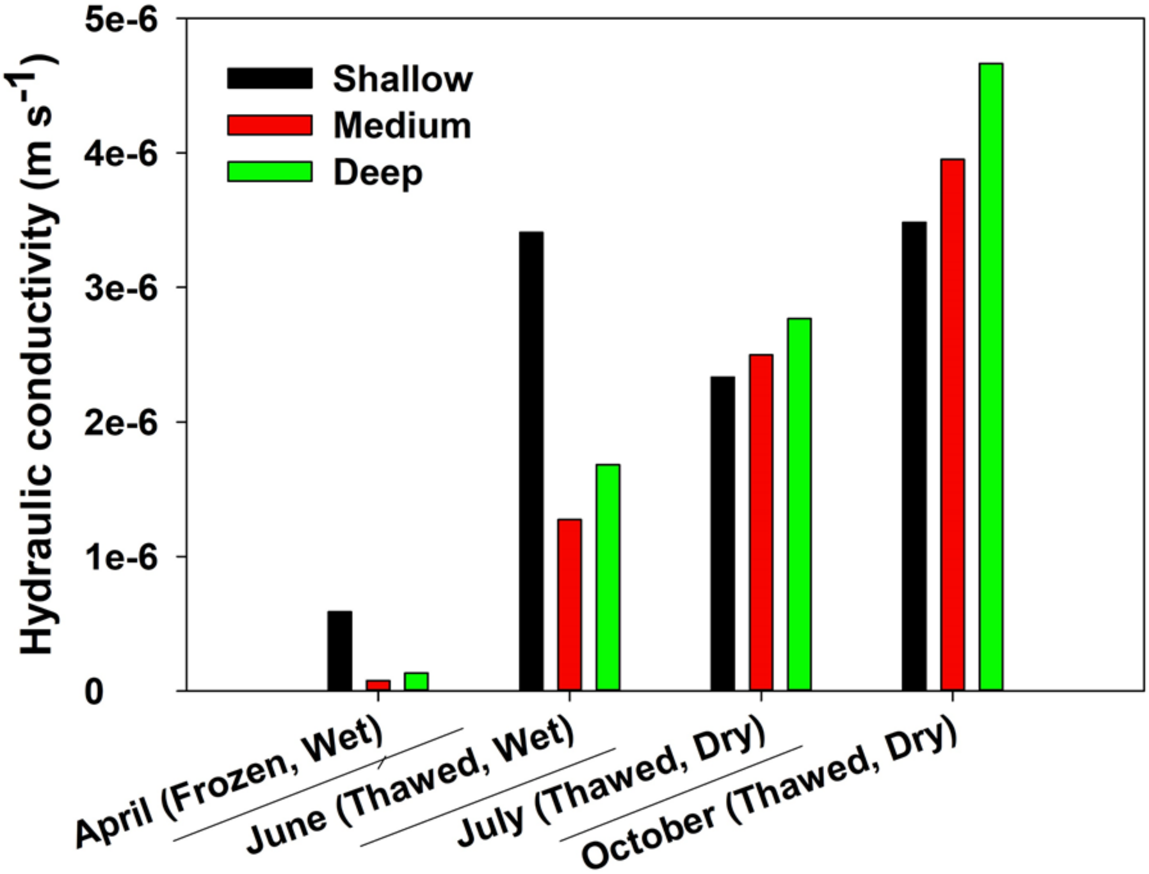


Figure 4.5. Subsurface hydraulic conductivities observed from slug tests during periods of 2017 with different antecedent moisture conditions. Screened depths of the piezometers below ground were 25-35 cm (shallow), 65-75 cm (medium), 90-100 cm (deep).

The chemistry of tile water showed strong temporal variability coinciding with precipitation and antecedent moisture characteristics (Figure 4.6). For example, lower tile EC concentrations (< 1000 µS cm^{-1}) were seen in early spring (Figure 4.6), coinciding with the existence of the soil-ice layer (Figure 4.3). Although a shallow GWT was present during the March-April months when the lower EC values were observed, this appeared to be a perched water table on frozen ground (Figure 4.3). Electrical conductivity concentrations in tile drainage were similar to those in surface runoff over the same time period (Figure 4.7). Electrical conductivity concentrations in tile drainage increased during late spring when the soil profile was fully thawed and wetter and subsequently decreased during the summer months. The increase in EC concentrations in tile drainage during May and June of each year coincided with a shallower GWT whereas the decrease in EC concentrations in summer coincided with a deeper GWT (Figure 4.7). During the late spring storms, the EC in tile drainage closely resembled the EC of the groundwater (Figure 4.7), and both tile drainage and GW EC concentrations were greater than EC concentrations in overland flow. In contrast, EC concentrations in tile drainage in summer were intermediate between the EC concentrations in overland flow and groundwater and thus appeared to be a combination of both sources (Figure 4.7).

There was also temporal variability in tile P concentrations. Greater SRP and TP concentrations in tile drainage were observed during frozen (or partially frozen) and drier conditions (Figure 4.6d-i). However, tile flow during such periods was smaller and was associated with elevated suspended sediments (Figure 4.8, Table 4.1). These greater SRP and TP concentrations often corresponded to lower EC values (Figures 4.6, 4.8). In contrast, P concentrations were smaller in late spring, coinciding with wetter conditions when substantial tile flow prevailed (Figure 4.6). These smaller P concentrations during late spring were also associated with greater EC and smaller TSS concentrations. Correlations between tile flow and P concentrations were not significant, and patterns showed distinct threshold behavior, with different responses between the spring events and events at other times of year (Figure 4.8).

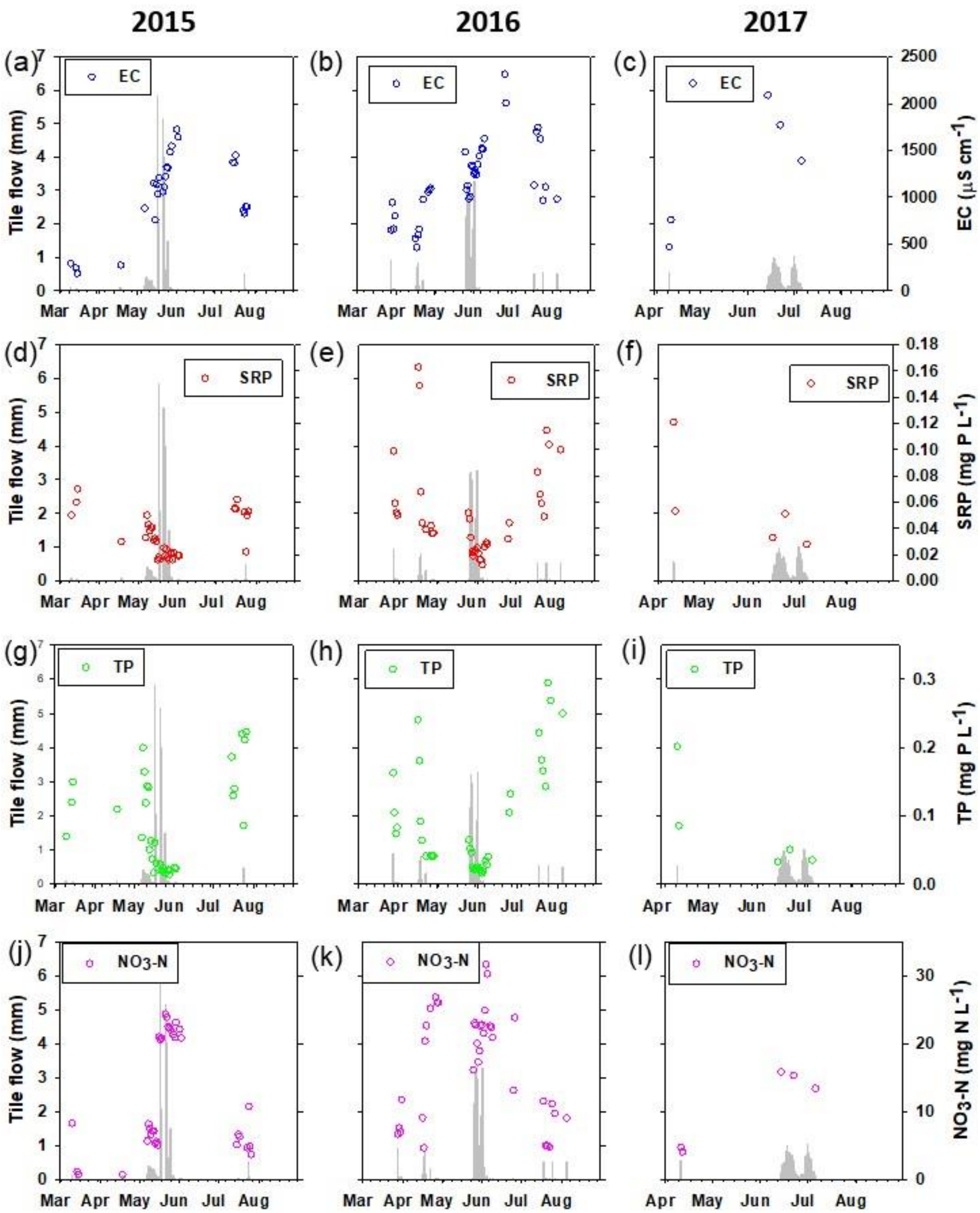


Figure 4.6. Daily electrical conductivity (EC), soluble reactive phosphorus (SRP), total phosphorus (TP) and NO_3-N during the study period. Tile flow was shown as vertical gray bars.

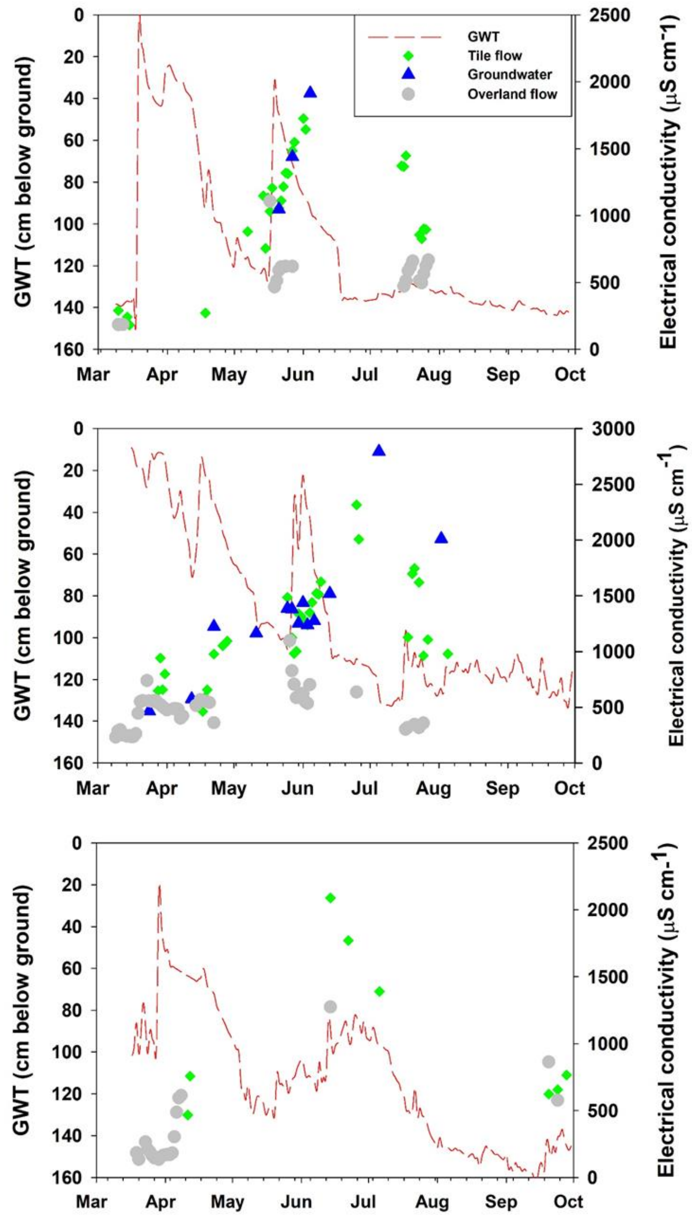


Figure 4.7. Daily groundwater table (GWT) (red dashed lines), with electrical conductivity (EC) in tile drainage (green diamonds) and groundwater (blue triangles) and overland flow (grey circles) piezometers during 2015 (a), 2016 (b) and 2017 (c) study period.

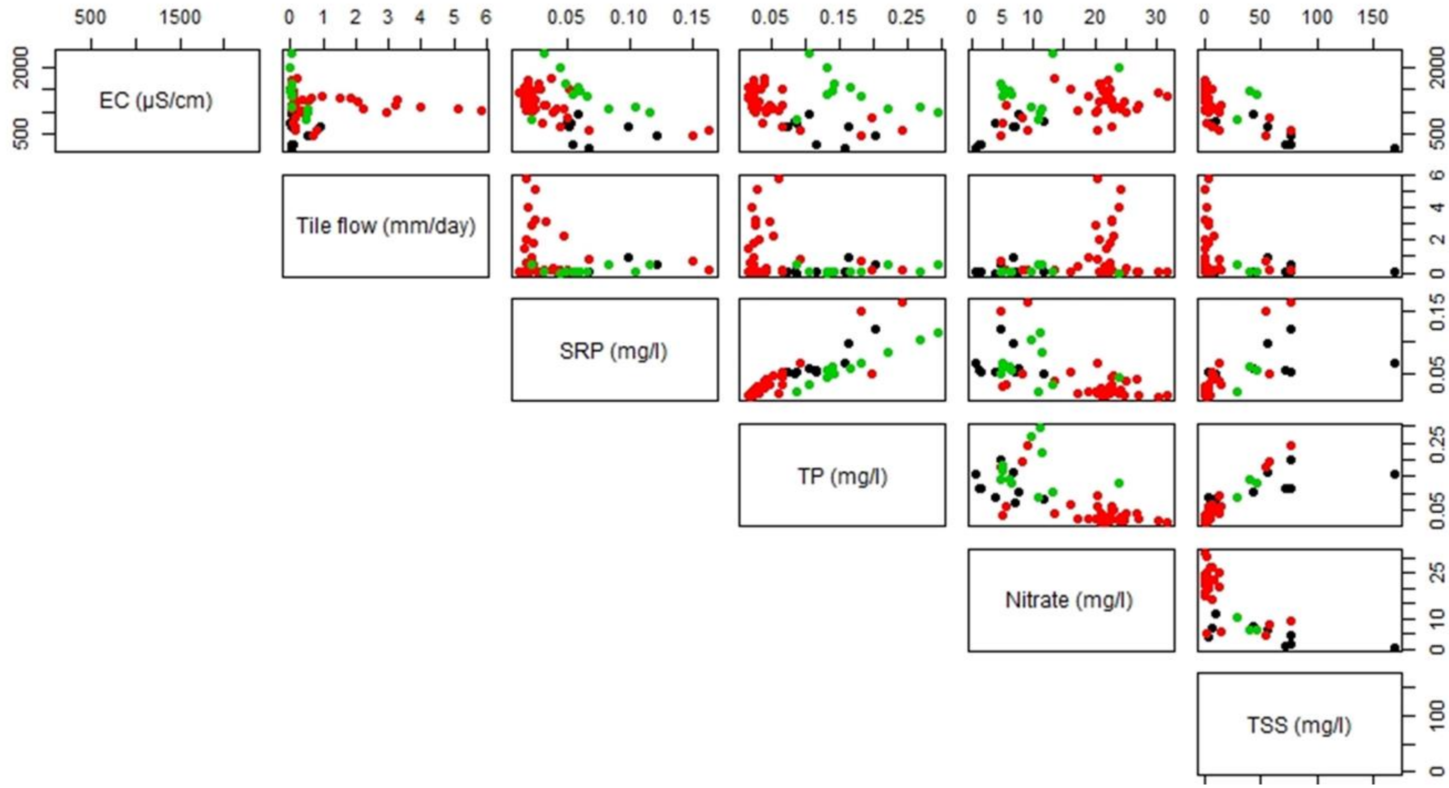


Figure 4.8. Scatter plots showing correlations between the water quality parameters. Black dots represent the early spring, red dots show the late spring and green dots represent summer period.

Table 4.1. Spearman correlation coefficients between daily tile flow and water quality parameters (n= 57). Note: EC: Electrical conductivity; SRP: Soluble reactive phosphorus; TP: Total phosphorus; TSS: Total suspended solids.

	Tile flow	SRP	TP	Nitrate	TSS
EC	-0.36**	-0.42***	-0.35**	0.34**	-0.68***
Tile flow		-0.22	-0.24	0.22	-0.14
SRP			0.92***	-0.67***	0.75***
TP				-0.7***	0.83***
Nitrate					-0.68***

*** $p < 0.001$; ** $p < 0.01$; * $p < 0.05$

In contrast to P concentrations, elevated NO_3-N concentrations were observed during the late spring (Figure 4.6, Table 4.1), with smaller NO_3-N concentrations during the snowmelt and summer events. Elevated NO_3-N in tiles tended to be associated with greater EC ($p < 0.001$). Nitrate concentrations in tile drainage were negatively related to P and TSS concentrations in tile drainage ($p < 0.001$). However, the relationships between the tile flow and water quality parameters were often non-linear, displaying threshold-like behavior between the seasons (Figure 4.8). For example, the relationship between the tile flow and EC showed a variable collapse where late spring EC values remained almost constant despite variable flow. This implies that a single source could be driving the substantial (>1 mm day^{-1}) late spring tile flows. However, this was not observed in early spring and summer seasons.

4.4. Discussion

Tile flow and chemistry at this study site showed strong seasonal patterns that were influenced by the presence and development of the soil-ice layer, antecedent moisture conditions and precipitation characteristics. Overall, little tile flow was observed in the early spring (snowmelt) or summer and fall, whereas the tile drains flowed substantially in late spring. The lack of flow in summer is consistent with what has been observed in more temperate regions (e.g. Ford et al., 2017, Lam et al., 2016a). However, the lack of tile drainage during the snowmelt period in the current study is in contrast to what has been observed in warmer regions (e.g. Macrae et al., 2007).

4.4.1 Impacts of frozen ground on tile flow and chemistry

The tiles experienced minimal flow (< 1 mm day^{-1}) during the early spring snowmelt period due to the presence of soil frost that interrupted flow from the surface and shallow subsurface to tiles. The impacts of frozen ground on subsurface runoff have been described previously in this region, where infiltration is impeded and overland flow is favoured (e.g. Cordeiro et al., 2017; Dumanski et al., 2015). Kokulan et al. (2019b) reported that surface runoff dominated during the snowmelt period, irrespective of the presence of tile drains and hypothesized that frozen ground decoupled tiles from the surface. Results from the current study confirm the presence of frozen ground deep in the subsurface (Figure 4.3) and demonstrate that a shallow GWT was observed during early spring despite smaller tile flow volumes (Figure 4.2). Tiles are expected to flow when a shallow GWT is present (Cordeiro and Ranjan, 2015). The shallow GWT in the early spring likely resulted from the formation of water table mounds in the near-surface zone due to frozen subsoil (Hayashi et al., 2003), and this perched GWT over the frozen subsurface layer likely did little to influence the tile flow during the snowmelt runoff. It should be noted, however, that although percolation to the tile drains was impeded, there was some tile flow, indicating that some water passed through the soil into tile drains.

In addition to impacting flow, the soil-ice layer also appeared to influence the chemistry of tile drainage. Examinations of the EC in tile drainage water and overland flow during the snowmelt period and groundwater (when present) suggest that the small amount of tile drainage observed during the snowmelt period was likely primarily due to preferential transport. Indeed, macropores or desiccation cracks that remain air-filled before freezing can contribute to rapid water infiltration during the subsequent snowmelt (Grant et al., 2019a; Mohammed et al., 2018). Tile P concentrations during snowmelt were elevated. These greater P concentrations were often associated with greater TSS loads. In contrast, snowmelt NO_3-N concentrations were smaller. These observations imply that in early spring flow at this site was delivered to tile drains via preferential flow through frozen soils. Although previous studies undertaken during the snowmelt period have reported greater P concentrations in tile drainage during snowmelt (e.g. Van Esbroeck et al., 2016; Lam et al., 2016a; Macrae et al., 2007), these concentrations were considerably smaller than concentrations observed in surface runoff, indicating either subsurface buffering or dilution by groundwater. Although P concentrations in tile water were large during snowmelt, it should be noted that gross nutrient loads in tile drainage were small due to the small volume of tile flow (Kokulan et al., 2019b).

4.4.2 Impacts of antecedent moisture conditions and precipitation characteristics on tile flow and chemistry

Following the thaw of the seasonal soil frost layer, tile flow was influenced by both antecedent soil moisture conditions and precipitation characteristics. Significant tile flows were observed in the late-spring, following low intensity, long-duration spring storms in two of the study years.

Infiltration capacities of the wet, thawed soils were considerably lower than those observed on frozen soils, (Figure 4.4), likely due to the swelling that typically occurs in vertisolic clays (Bagnall et al., 2019; Kokulan et al., 2019a). However, greater subsurface hydraulic conductivities in the late spring (when compared with the early spring) implied increased subsurface percolation. Moreover, spring storms also activated and raised the GWT, which sustained tile flow during the wet late spring season. The GWT was above the tile level (approximately 1 m depth) during the major late spring tile flowing events in 2015 and 2016, as well as during the early summer tile flow in 2017 (Figure 4.2). These observations agree with previous research in this region that showed the occurrence of tile flow under shallow GWT conditions (Cordeiro and Ranjan, 2015). A shift in the EC of tile effluent in the late spring season, where tile EC was similar to the EC of the groundwater, and both were considerably greater than the EC of surface runoff, also suggests that groundwater was the major contributor to tile flow during this period (Figure 4.7). This is consistent with work done in warmer regions, where groundwater contributes substantially to tile drainage in the late spring following rain storms on wet soils (e.g. Macrae et al., 2007; Macrae et al., 2010; Ford et al., 2017).

Tile P concentrations were small during the late spring, even though the field had been fertilized in early May during each of the three study years. In contrast, NO_3-N concentrations were large (20-25 mg N L^{-1}). Fertilization has been identified as a critical factor leading to elevated tile P losses (King et al., 2015) as well as NO_3-N losses (Macrae et al., 2010). Thus, the small P concentrations observed during the spring were surprising. In vertisolic clays, surface cracks swell and seal preferential flow pathways once soils become moist (Bagnall et al., 2019; Hardie et al., 2011). This prompts water to move through the soil matrix until it is either intercepted by the tiles or contributing to groundwater recharge. Slower infiltration and percolation allow the P in the runoff water to interact with soil, allowing more removal by the calcareous clays. Conversely, NO_3-N ions are mobile and therefore transported through soil matrix (Zhao et al., 2001). Tile nutrient loads during the late spring period were significant due to the large volume of water flowing through the tiles (Kolulan et al., 2019b).

Flow through tiles under dry antecedent conditions differed substantially from what was observed under wet antecedent conditions. Little tile flow was observed in summer despite the frequent occurrence of high-intensity summer storms. Despite greater seasonal precipitation, the average soil moisture in summer is lower in Southern Manitoba, due to greater potential evapotranspiration rates and reduced groundwater recharge (Chen et al., 2004). Drier antecedent conditions increase the extent of shrinkage cracks and facilitate rapid infiltration and subsurface percolation (Bagnall et al., 2019; Greve et al., 2010). Moreover, high intensity rainfall events activate macropore flow through desiccation cracks (Bagnall et al., 2019). The generally higher SRP and TP concentrations, lower NO_3-N and smaller EC in tile flow (relative to the late spring) suggest that preferential flow likely led to the rapid activation of tiles in summer. The fact that the GWT was below the tile depth during these events supports this. In fact, the majority of the summer tile flow activations occurred from the top-down water front movement, as reported by Kokulan et al. (2019a). Although P concentrations were high during these periods, loads were small due to the low flow volumes that occurred. Although summer storms in the region appear to raise the water table through a groundwater ridging effect (Kokulan et al., 2019a), they do not have the capacity to recharge the groundwater table to sustain large tile outflows (Chen et al., 2004).

Precipitation characteristics were also important in driving tile flow in the fall; however, the effects of precipitation were also impacted by antecedent conditions. Similar to the summer periods, substantial tile flows (> 1 mm day^{-1}) were not observed in the falls of either 2015 or 2017 despite the occurrence of multiple storms. In contrast, tiles responded to a major thunderstorm (85 mm) that occurred in fall 2016 on wet antecedent conditions. Unfortunately, this study did not monitor the tile nutrient dynamic in the falls of 2015 and 2016. Additional research is required to assess the impact of autumnal management practices (i.e. tillage) on tile flow and chemistry and the impact of fall moisture conditions on the development of soil-ice layer during the following winter season.

4.5. Conclusions

This study assessed water movement and tile nutrient dynamics from an agricultural field in the Red River Valley of the Northern Great Plains. Water infiltration and percolation during snowmelt runoff was restricted to the near-surface due to a thick subsurface soil-ice layer resulting in smaller tile outflows. In open water periods (late spring, summer and fall), substantial tile flows (>1 mm day^{-1}) were largely limited to the periods with wetter surface moisture conditions and shallower GWT. Tiles in this region have the potential to desaturate the soil profile and maintain the GWT beneath the tile depth during the late-spring season. Decoupling of preferential flow pathways during wetter conditions also reduce the risk of elevated P loss through tile drainage in the late spring season. However, this late spring tile flow could exacerbate tile NO_3-N losses.

Infiltration test and EC data suggest that tile flow during the early spring and summer were dominated by preferential flow through the frozen and dry shrinkage cracks. These outflows contained greater P but smaller NO_3-N concentrations. On the other hand, tile flow during late spring was largely driven by a groundwater hydrochemical signature. Phosphorus concentrations during the late spring were lower, while late-spring tile flow showed elevated NO_3-N concentrations.

The findings from this study provide insights on the potential of tile drainage as a runoff pathway and as a subsurface conduit for environmental pollutants in the clay soils of the Red River Valley. Future research should assess this potential with other management practices such as fertilization, tillage and variable tile configuration to obtain a complete picture of the role of tile drainage in hydrochemical export in this region.

4.6. Acknowledgements

This project was funded by the Manitoba Agriculture, Food and Rural Development (MAFRD), Manitoba Conservative Districts Association (MCDA), Environment and Climate Change Canada's (ECCC) Lake Winnipeg Basin Stewardship Fund, the Natural Sciences and Engineering Research Council of Canada (Macrae–DG and Lobb–DG) and the Canada First Excellence Research Fund (Agricultural Water Futures). Northern Plains Drainage Systems, Anthony Buckley, Eva Slavicek, Vito Lam, Bo Pan, Kui Liu, Lindsey McKenty, Julie DePauw and Reza Habibiandehkordi are thanked for the field and laboratory assistance.

Chapter 5: Phosphorus dynamics in surface runoff during overbank flooding of field-edge ditches in vertisolic clay agricultural land in the Red River Valley

5.1. Introduction

Freshwater bodies in North America, such as Lake Winnipeg in Manitoba, Canada, have been affected by harmful and nuisance algal blooms. Agricultural runoff is considered to be the primary contributor to the current deterioration of the health of these freshwater bodies (Schindler et al., 2012). Elevated edge of field phosphorus (P) and nitrogen (N) losses typically occur upstream of Lake Winnipeg, in the Red River Valley, during major spring events, which are often associated with spring snowmelt (Tiessen et al., 2010). However, an increasing frequency in the rain on snow events and multiday spring and summer storms has been observed in this region, which is projected to increase (Jeong and Sushama, 2018; Shook and Pomeroy, 2012). A higher frequency of large storms in spring and summer has the potential to exacerbate runoff related nutrient losses to water bodies in the future. Considering the potential for such large events to contribute substantial runoff and nutrient losses from agricultural fields (Kokulan et al., 2019b), field studies are required to explore the sources and processes that may drive P mobilization during these large events. This knowledge can assist land managers in the selection of appropriate Best Management Practices under both contemporary and future climates.

Much of the impaired water quality in Lake Winnipeg, which receives runoff from the Red River, is thought to originate from surface runoff from agricultural fields throughout the watershed (Schindler et al., 2012). The Red River Valley is naturally prone to flooding conditions as a result of both landscape and climate drivers. The soil in the Red River Valley of the Northern Great Plains is composed mainly of clayey soils with low infiltration capacities (Schindler et al., 2012).

In addition to a little natural tendency for infiltration to occur, runoff in the region is also significantly impacted by climate drivers. In early spring, the infiltration of meltwater is impeded by a frozen soil-ice layer that extends to 0.75-1 m depth (Gray at al. 2001), causing most runoff to leave fields as overland flow (Kokulan et al., 2019b; Dumanski et al., 2015). Unlike other North American agricultural landscapes such as the Great Lakes-St. Lawrence region, where frequent melt events are common during the winter due to the milder climate (Plach et al., 2019; Jamieson et al., 2003), snow in the Northern portion of the Red River Valley generally melts during late winter or early spring as a large ‘freshet,’ favouring greater magnitudes of spring runoff (Kokulan et al., 2019b). When snowmelt is accompanied by rain-on-snow events, runoff occurs more rapidly, leading to larger runoff peaks (Kokulan et al., 2019b). Although summer is typically dry due to the Prairie climate, convective rainstorms can rapidly deliver large volumes of rainfall to dry soils, which exceed the infiltration capacity of the clay and consequently leave fields as overland flow (Kokulan et al., 2019a).

The rapid addition of water as snowmelt or convective rain events to soils with low and/or impeded infiltration capacities, combined with the naturally flat topography of the region, leads to frequent flooding events. Such occurrences are exacerbated by ice jams (in early spring) and the blockage of drainage ditches and culverts (year-round) by regional municipalities to manage the risk of flooding in the downstream areas of the Red River. This retarded drainage prompts roadside ditches to fill and back up into adjacent fields, causing periods of “inundation.” The inundation, which usually lasts from several days to weeks, may have implications for P release and immobilization processes between the sediment and the water column from the fields and the adjacent ditches (Schindler et al., 2012). Although it is known that much of the P loss into ditches and tributaries in the Red River Valley are associated with overland flow, it is not clear if this is overland flow contributed by fields, or, if it is the prolonged inundation of the ditches and adjacent riparian zone.

Naturally, runoff water from the agricultural fields in the Red River Valley region is drained through naturally existing or man-made surface swales (or surface drains). These surface swales drain into the roadside ditches adjacent to the farm. Roadside ditches are an integral part of the farm drainage system connecting the farms with regional streams (Buchanan et al., 2013). Ditches also receive sediments, dust and salt mixture from the nearby roads via erosion processes (Cade-Menun et al., 2017). Historically, ditches have been examined from a water conveyance standpoint, but their role in water quality standpoint is less well understood and appears to differ regionally (Smith et al., 2005; Van Nguyen and Maeda, 2016). For example, studies have shown P in runoff being reduced through retention by ditch sediments and vegetation buffers in the warmer temperate regions (Haggard et al., 2007). In contrast, recent work has shown that vegetated buffers are less effective in retaining P in colder regions like Northern Great Plains of North America (Kieta et al., 2018). However, the role of ditch sediments in P mobilization or immobilization in this region has not been studied in detail.

Biogeochemical processes that operate at the soil-solution interface determine the mobility of P; however, these processes may be impacted by flooding conditions. The supply of P to runoff from soils is governed by adsorption-desorption, precipitation-dissolution reactions, biological uptake and organic matter decomposition. In acidic environments, these exchange processes may be dominated by adsorption-desorption reactions with positively charged amorphous and crystalline oxides and hydroxides of Al, Fe and Mn (Hinsinger, 2001). Adsorbed P may diffuse into the solid phase or may diffuse out of the solid phase depending on the sediment Equilibrium Phosphorus Concentration (EPC) (Withers and Jarvie, 2008). In contrast, P precipitates as various

Ca-phosphate forms (e.g., hydroxyapatite, octacalcium and dicalcium phosphates) and may form different Ca-P and Mg-P complexes in alkaline environments (Ige et al., 2005; Plach et al., 2018b). The flooding of ditches, adjacent riparian areas and surface swales has the potential to elevate P losses in the Red River Valley. First, flooding increases the contact time between P-rich soil pools and the more dilute snowmelt or rainwater, likely leading to the desorption of P to runoff. Prolonged inundation of soils also affects the EPC by decreasing the redox potential and subsequently altering the pH of the environment. In alkaline environments such as in Red River Valley, a decrease of redox potential favours the dissolution of CO_2, thereby shifting the pH towards neutrality (Amarawansha et al., 2015). This, in turn, affects the stability of P complexes and elevates P release from sediments to the water column. In fact, previous laboratory incubation studies suggest that the majority of soils in Manitoba are prone to increased P loss under periods of prolonged inundation (Amarawansha et al., 2015). Although this potential exists, this has not been explored in a field setting under different types of flooding events spanning a range of seasons.

Surface runoff from two agricultural fields and their adjacent roadside ditches was monitored between March 1st and September 31st, 2015, 2016 and 2017 in Manitoba, Canada. The objectives of this study were (1) to observe and characterize the occurrence and frequency of periods in which ditches flood over their banks, (2) to monitor the evolution of overland flow P concentrations during different stages of the overland flow hydrograph throughout events to determine if P concentrations increase throughout periods when overbank flooding occurs, and (3) to use soil and water quality data from the fields and the ditches to infer mechanisms that may potentially govern P dynamics during the inundation.

5.2. Materials and methods

5.2.1. Site description

Two adjacent farm fields (25 ha each) and nearby roadside ditches in Elm Creek, Manitoba, were instrumented. Both fields are underlain by Gleyed Humic Vertisols of the Red River Series (U.S taxonomy: Gleyic humicryerts), and topography is characterized by nearly flat terrain (0-2 % slope). Both fields drain into adjacent roadside ditches through in-field surface swales. Field A was also tile-drained, whereas Field B was only surface drained (Figure 5.1). Tile drain laterals (4" diameter) in Field A are systematically placed at ~ 1 m depth, with 13 m spacing. The laterals drain into a large 37.5 cm main that discharges to a collection pond and adjacent retention pond (described in Kokulan et al., 2019a). Both fields were managed similarly with respect to crops, fertilizer application, seeding and tillage operations. Canola (Brassica napus L.) was grown in 2015, which was followed by spring wheat (Triticum aestivum L.) and soybeans (Glycine max L. Merr.) in 2016 and 2017, respectively. Phosphorus was subsurface seed placed as monoammonium phosphate (40 kg ha^{-1} in 2015, 45 kg ha^{-1} in 2016, and 20 kg ha^{-1} in 2017), and the fields were annually tilled to 15 cm depth in fall. Additional information on the regional climate and tile configuration are described by Kokulan et al. (2019a), and other information on crops and fertilizer application are described in Kokulan et al. (2019b).

The roadside ditches adjacent to both fields receive their runoff water from the fields and from direct precipitation (snow or rain) (Figure 5.1). The two ditches are separated by a farmstead, and flow in the two ditches drains in opposite directions (the ditch associated with field A flows in north-south direction while field B ditch flows in south-north direction). Both ditches subsequently drain into provincial drainage channels through culverts. Ditches are approximately 2 m wide and are sloped 0.3 % towards the culverts that they drain into. Ditch vegetation, which is generally comprised of bromegrass (*Bromus riparius*), quack grass (*Elymus repens*) and alfalfa

(*Medicago sativa L.*), was not managed during the study periods. Ditches and the adjacent fields are separated by a thin (1 m width) grass riparian buffer, which is also composed of the ditch vegetation.

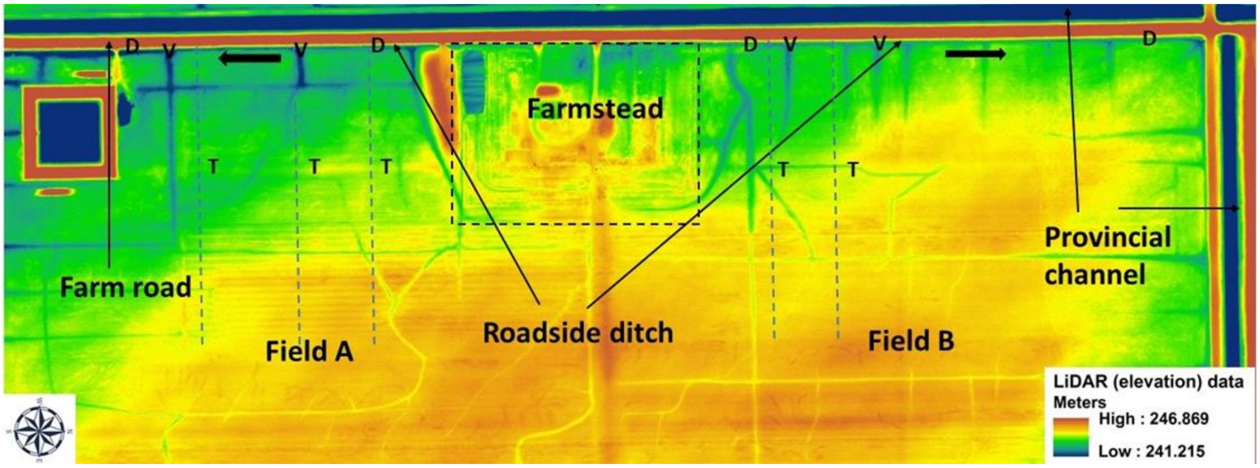


Figure 5.1. LiDAR image of the study site showing fields A and B, location of roadside ditches, farm road and provincial channel and the directions of ditch flow. The locations of the V-notch weirs and the ditch monitors were shown by “V” and “D” respectively. Dashed blue lines (T) show the transects for the soil sampling. Farmstead was delineated by a black dashed rectangle.

5.2.2. Instrumentation and data collection

Surface runoff was monitored in both fields and adjacent ditches from 2015 to 2017 from March until September, covering annual spring snowmelt and several spring and summer storms. The flow from each field was monitored at two surface swales. A V-notch weir was established in each surface swale to monitor overland flow. Each weir was equipped with an SR-50 A ultrasonic sensor (Campbell Scientific Ltd) and a capacitance sensor (Odyssey, Dataflow Systems Ltd.) to measure water levels (Figure 5.1). Capacitance sensors (Odyssey, Dataflow Systems Ltd.) were installed in the ditches, both upstream and downstream of the ditches, to measure ditch water levels. Water levels in weirs and ditches were recorded at 15 minutes intervals, and frequent manual water level measurements were taken to validate sensor readings during runoff periods. A meteorological station (CR10x, Campbell Scientific Ltd) was installed at the site to take hourly measures of rainfall (TE525M, Texas Electronics) and air temperature (HMC45C, Campbell Scientific Ltd.). In 2016, a Flo-tote 3 and FL 900 series logger (Hach Ltd.) was installed in the downstream culvert of Field A to estimate the flow through the ditch (15-minute intervals). However, flow meter was not installed in the downstream culvert of Field B due to logistical issues.

Eighteen runoff events were monitored during 2015 – 2017 March to September months (Figure 5.2, Table 5.1). These included 3 early spring snowmelt events, 5 late spring rainstorms and 10 summer thunderstorms. Field B and its adjacent ditch were not monitored in 2017 due to logistical issues.

Overland flow samples from the swales were collected at each weir with programmable water samplers (AS950, Hach Ltd.) and acid-washed 1 L polyethylene bottles. Samples were collected at two to four-hour sampling intervals during the rising limbs of the overland flow hydrographs, and six to 12-hour sampling intervals during the peak and recession limbs of the hydrographs. Additional overland flow grab samples were collected at a daily time step during small flow periods (< 1 L s^{-1}). Ditch water samples were collected manually in 500 mL polyethylene bottles both upstream and downstream of the ditches at 24 - 48 hour intervals during runoff events. Runoff samples from events 14 and 17 were not collected at both fields due to autosampler failures.

Table 5.1. Dates, types, rainfall sizes, maximum daily rainfall and occurrence of inundation in the monitored events

Event	Year	Date	Type	Rainfall size (mm)	Maximum daily rainfall (mm)	Flooding	Overland[a] Flow (mm)	Mean SRP concentration[a] (mg P L^{-1})
1	2015	Mar 10-16	Snowmelt	N/A	N/A	No	21	0.31
2	2015	May 16-25	Spring storm	64.1	55.1	Yes	46	0.32
3	2015	Jul 16-17	Thunderstorm	38.3	28.9	No	<1	0.27
4	2015	Jul 23-26	Thunderstorm	43.7	43.5	No	3	0.29
5	2016	Mar 10-Apr 2	Snowmelt	14.1	11.4	Yes	58	0.24
6	2016	Apr 15-23	Spring storm	38.2	20	Yes	28	0.23
7	2016	May 25-26	Spring storm	45.9	18.9	No	1	0.33
8	2016	May 31-Jun 4	Spring storm	26.9	19.9	No	3	0.08
9	2016	Jun 25	Thunderstorm	32.9	29.1	No	<0.1	0.05
10	2016	Jul 16	Thunderstorm	47.4	47.1	No	1	0.83
11	2016	Jul 20	Thunderstorm	15.2	15.2	No	<1	0.16
12	2016	Jul 23-24	Thunderstorm	25.7	25.7	No	2	0.35
13	2016	Aug 3-5	Thunderstorm	38.6	37.3	No	<1	0.19
14	2016	Sep 16-23	Thunderstorm	85	85	Yes	24	NS
15	2017	Mar 20 – Apr 9	Snowmelt	16	8.4	Yes	89	0.26
16	2017	Jun 13-14	Spring storm	31.8	16.2	No	<0.1	0.22
17	2017	Jun 28	Thunderstorm	22.8	22.1	No	<0.1	NS
18	2017	Sep 18	Thunderstorm	60	52.5	No	<0.1	0.08

a- Field A only; N/A- Not available;

NS- No water samples were collected

Soil samples were collected from the ditches, downfield, midfield and upfield in fall 2017 following the harvest (before fall till) along 225 m-long transects. In Field A, surface soil samples (0-15 cm) were collected along three transects with a gouge auger. In Field B, surface soil samples (0-15 cm) were collected along two transects. Each sample was a composite of 5 subsamples taken within 1 m radius. Collected soil samples were delineated upon their depth (0-6 cm and 6-15 cm). Collected samples were air-dried, ground and passed through a 2-mm sieve prior to chemical analysis.

5.2.3. Laboratory analyses

Upon return to the laboratory, the pH and electrical conductivity (EC) of the water samples were measured with an Accumet Basic AB 15 pH meter and an Accumet Basic AB 30 EC meter, respectively (Fisher Scientific.) A subsample was filtered through a Whatman glass microfiber filter paper (47 mm, 0.45 μm pore size, GE life sciences) and refrigerated at 4^{o}C. This sample was analyzed for soluble reactive phosphorus (SRP) (QuikChem Method 10-115-01-1-A) with a QuikChem 8500 series 2 FIA system (Lachat instruments) through flow injection analysis colorimetry within 48 hours of collection. Once filtered, filter papers with sediments were dried at 105 ^{o}C for 24 hours and weighed for total suspended solids (TSS). A non-filtered sub-sample (200 mL) was stored at -18^{o}C and was analyzed for total phosphorus (TP) in a later day with QuikChem 8500 series 2 FIA system (QuikChem Method 10-115-01-4-C, Lachat Applications Group). Samples collected between March 1st, 2016 and April 30th, 2016, were further analyzed for dissolved Fe using the FerroVer HACH method (Shimadzu 1800, UV/visible spectrophotometer) and for dissolved Ca^{2+} and Mg^{2+} ions (ICS3000 Ion Chromatography System, Dionex Corporation). This period between March 1st and April 30th, 2016, was responsible for 73% of the overland flow runoff and 74% overland flow related SRP losses from Field A in 2016 (Kokulan et al., 2019b). Standard checks were used per 20 samples as a mean for quality control. Samples that exceeded the standard spectrum were diluted and re-analyzed.

Soil P concentrations and solid-phase inorganic P partitioning were examined within each field to determine what form the P is held in (to infer mobility) and to determine if this varied with landscape position (i.e. within ditches, at the downfield location (field-buffer boundary), midfield and upfield positions. Soil test P content of the collected samples was determined by Olsen-P colorimetry method (Sims, 2009a). One gram of soil was shaken with 20 ml pH adjusted (8.5) 0.5 M $NaHCO_3$ extracting solution in a mechanical shaker for 30 minutes at room temperature. The solution was then filtered through a Whatman no-42 filter paper and was analyzed for P with a spectrophotometer. For the subsequent analysis, samples that were taken from a particular landscape position (i.e. ditch, downfield, midfield and upfield) were bulked to form a composite sample considering the time and resource limitation. Phosphorus sorption index (PSI) was determined from a modified single point sorption isotherm method, as described by Sims (2009b). For PSI analysis, 1 g of soil sample was shaken with 20 ml P sorption solution (75 mg P/L) in a mechanical shaker for 18 hours. The solution was then filtered with 0.45 µm pore size syringe filters and analyzed for SRP with an autoanalyzer. Loosely bound and moderately soluble (Sol-P_i), reducible (CBD-P_i), and acid-soluble (HCl-P_i) P fractions of the soil samples were estimated according to the procedure described by Zhang and Kovar (2009) for calcareous soils. For Sol-P_i, 0.5 g of soil was shaken with 25 ml of 0.1 M NaOH + 1M NaCl solution for 17 hours in a mechanical shaker. For CBD-P_i, soil residue was then heated in a water bath with 20 ml 0.3 M $Na_3C_6H_5O_7.2H_2O$, 2.5 ml of 1 M $NaHCO_3$ and 0.5 g $Na_2S_2O_7$ in a water bath at 85 ^oC. Soil residue was then shaken with a 0.5 M HCl solution for 1 hour to extract the HCl-P_i. At the end of every extraction step, the soil was centrifuged at 10 000 rpm for 5 minutes (11,180 x g) and washed with saturated NaCl. The wash was added to each extract prior to analysis. Reducible and HCl-P_i P extracts were also neutralized with either 2 M NaOH or 2 M HCl with p-nitrophenol indicator prior to P determination by an autoanalyzer. The extract from CBD-Pi was also analyzed for soil Fe with colorimetry (510 nm) with a UV spectrometer (Hach Ltd.). Soil organic matter (SOM) and calcium carbonate equivalent (CCE) of soils were determined using the loss-on-ignition method (Dean, 1974). For soil analyses, 20 % of samples were analyzed in duplicates.

5.2.4. Data analyses and interpretation

Event overland and ditch flow hydrographs were delineated into 5 classes to assess the evolution of P concentrations based on their flow and stage characteristics (Figure 5.2). The period of water ponding on fields but not leaving fields, before the onset of significant flow in the swales (< 1 L s^{-1}), was classified as "ponding." The hydrograph stage classified as "rising" refers to the rising limb of the hydrograph (overland flow > 1 L s^{-1}), where overland flow exited fields and entered the ditch. During the third stage classification, referred to as "merged," ditch stage was at its peak, but the flow was retarded in both fields and ditches, and the ditch water and overland flow were connected based on the water level and flow data. The "falling" classification defined the period when flow resumed in the swales following the "merged," coinciding with the falling limb of a hydrograph. "Drying" indicates periods with low flow (< 1 L s^{-1}) or low-level stationary water (< 15 cm) in the swales at the end of the runoff hydrograph. Flow estimation during at Field A is described in Kokulan et al. (2019b). Unfortunately, flow from field B could not be successfully estimated using a rating curve as the ditch flow was not measured during high flow periods. However, considering the close proximity of both fields and similar overland flow activation times

(Kokulan et al., 2019a), overland flow responses from Field B were assumed to be similar to Field A.

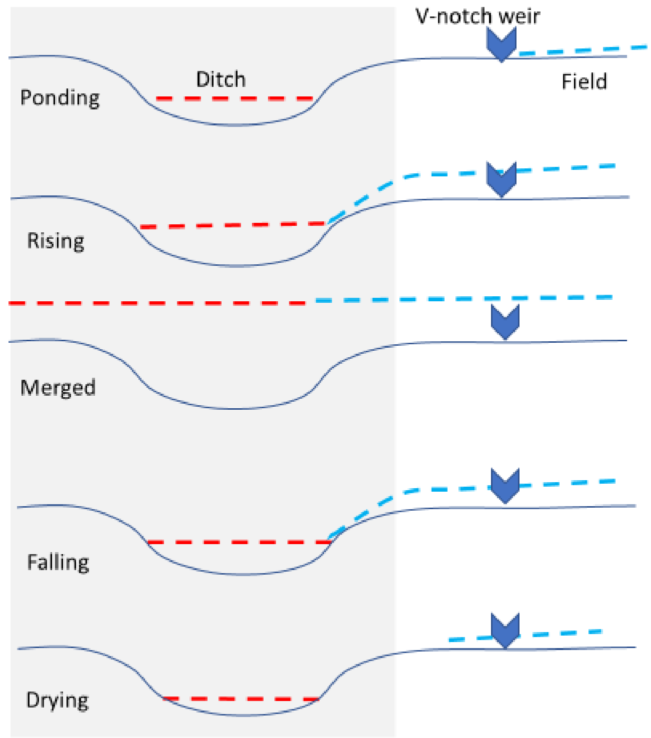


Figure 5.2 Schematic diagram that shows different stages of flooding event. Red dashed lines represent the weir water levels and blue dashed lines show the weir water levels in the field. Diagrams were not drawn to the proportions.

As flow weighted mean concentrations could not be precisely determined for periods of “merged” with retarded flow and periods of drying (due to low flow rates), daily median concentrations were used to compare P concentrations between two fields and between fields and ditches using Kruskal-Wallis test by ranks (one-way ANOVA on ranks test). Student t-tests were performed to compare soil chemical parameters between two fields. Kruskal-Wallis test on ranks was performed to compare Olsen-P contents between landscape positions within a field.

5.3. Results

5.3.1. Frequency of overbank flooding of ditches

Multi-day events during which ditches filled and merged with surface runoff in swales were observed during spring snowmelt and also during late spring rainstorms and a fall thunderstorm event (Figure 5.3). During these periods of flooding ("merged"), substantial portions of the fields were submerged (Appendix D1), lengthening contact time between surface soil and runoff. In general, periods of "merged" during the spring snowmelt period followed rapid snowmelt associated with rain on snow events. This occurred in two out of the three study years. The flooding events that occurred in late spring followed multiday spring storms. A flooding event was also observed in September 2016, following a high-intensity short duration thunderstorm (85 mm rain in two hours).

Even though events experiencing flooding represented only a small fraction of the observed events (5 out of 18), such events contributed to the majority of annual runoff and P losses (Table 5.1). On Field A, these five events, during which prolonged periods of flooding were observed, accounted for 70% of the edge of field runoff losses (overland flow + tile flow), 83% of the SRP losses, and 84% of the TP losses over the entire study period. Overland flow was particularly important during these events, as these events accounted for 87% of the total overland flow hydrologic losses, 86% of the SRP losses in overland flow, and 88% of the TP losses in overland flow over the three-year period. Although flooding was observed within these larger events, the duration of flooding ("merged") within events was relatively short. For example, the events typically occurred over 10-14 days, and "merged" conditions occurred throughout approximately 10-50% (1-7 days) of each event.

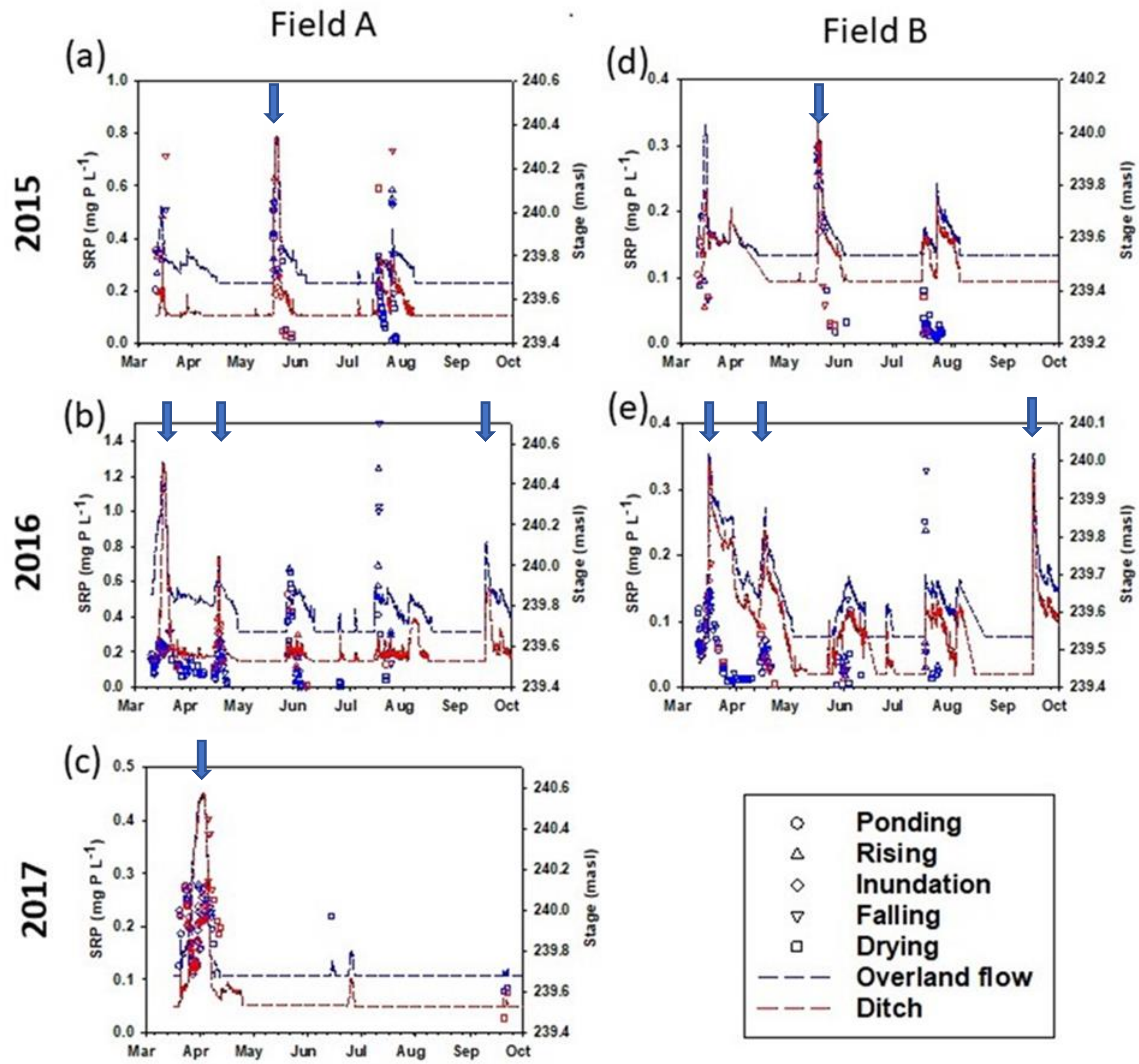


Figure 5.3 Field and ditch SRP concentrations against the stage during 2015-2017 study years. Field SRP concentrations and stage were indicated by blue whereas ditch SRP concentrations and stage were indicated by red. Field B data for 2017 was not shown due to lack of sampling. Events with inundation are shown with blue arrows.

Given that flooding generally occurred when ditches were blocked by managers to prevent flooding downstream, or flow was impeded downstream, flow was generally minimal during periods of "merged." The majority of the flow (75-99%) exiting fields via ditches during these large events occurred after the merging of the swales and ditches had occurred (falling limb of hydrograph). Periods of stagnation were also observed during “drying” periods that occurred after events, where ditches and swales were disconnected at the weirs. During such periods, prolonged contact occurred between the ditch water and sediments, but little ditch flow occurred. This period of “drying” lasted approximately 14-21 days following snowmelt, and for about 7 days in late spring (Figure 5.3). Stagnant water in the swales was also observed in summer between frequent summer thunderstorms (July 15th to August 5th, 2015 and 2016). However, this stagnant water was primarily limited to the downfield, and water levels were generally < 10 cm at the weirs.

5.3.2. Evolution of P concentrations during the events with inundation

When SRP concentrations during the specific “merged” period of inundation are compared to other periods throughout each event, SRP concentrations were greatest during “merged” for two events (Snowmelt 2016 (Event 5) and spring storm 2016 (Event 6)) but not for the other two events (Spring storm 2015 (Event 2) and snowmelt 2017 (Event 15)) (Figure 5.4). A dilution in SRP concentrations was observed on the rising limb for Field A during snowmelt events (Events 5 and 15). However, this was not observed in Field B during Event 5, as SRP concentrations continued to rise. In contrast, during Event 2 (spring storm 2015), median SRP concentrations were similar between the ponding and rising stages in both fields (although slightly greater on the rising limb for Field B), and subsequently decreased once merging had occurred. Soluble reactive P concentrations on the falling limb were smaller than the “merged” periods for Field B in all

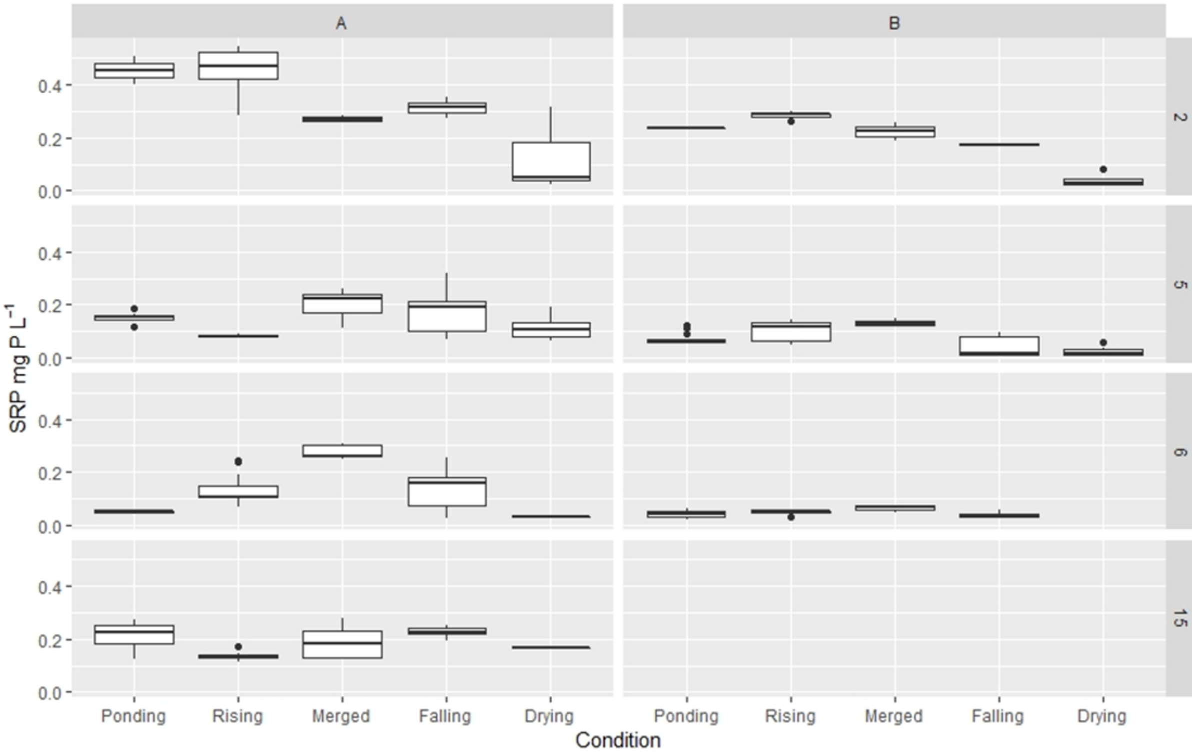


Figure 5.4 Box plots showing soluble reactive phosphorus (SRP) concentrations of collected overland flow samples from both fields (A and B) during various stages of runoff during events with flooding. Numbers on the right-hand side indicate the event numbers. Event 2 and 6 were runoff events from spring storms whereas Events 5 and 15 were spring snowmelt runoff events. Runoff was not monitored at Field B for event 15.

sampled events. However, this was only observed in two events (Event 5 and 6) for Field A. In general, SRP concentrations were smaller during the drying stage when compared to other stages.

The evolution of SRP concentrations throughout individual events with the stage (level) of swale (weir) water levels was explored (hysteresis loops) to determine whether increased SRP was

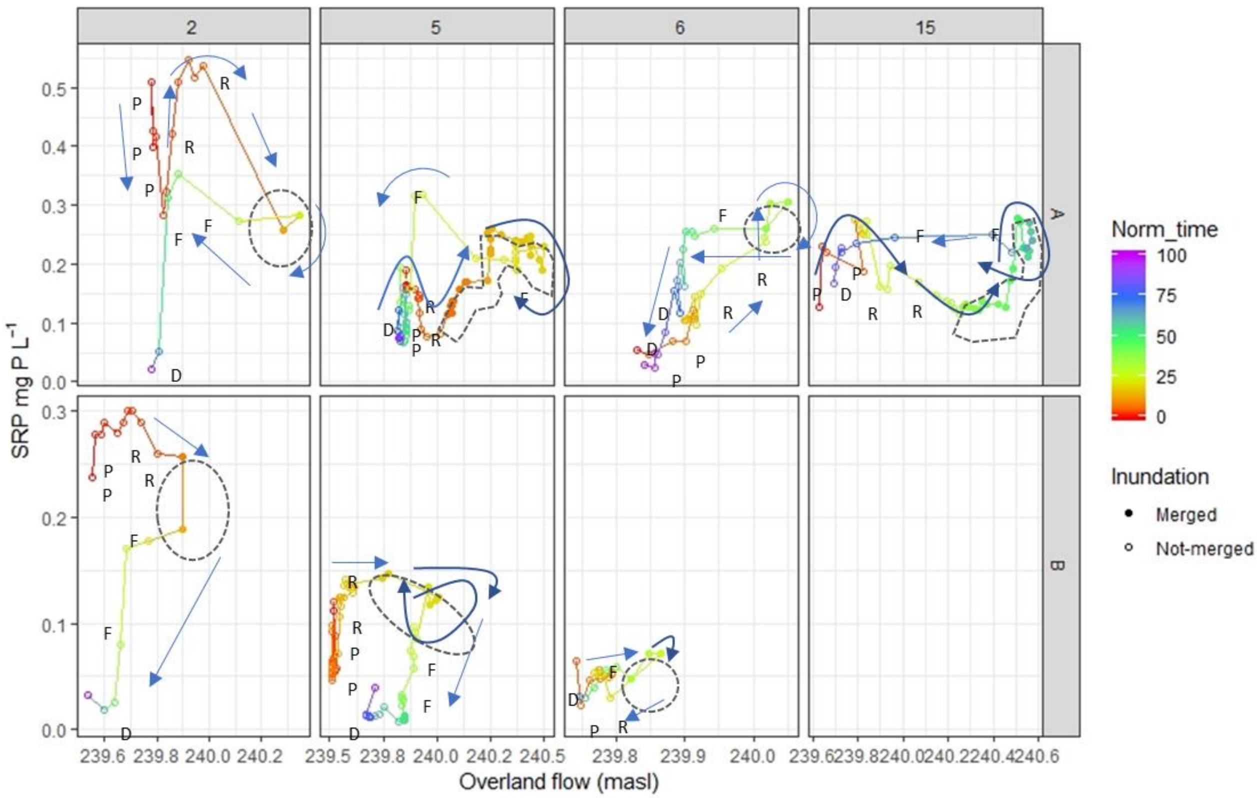


Figure 5.5. Hysteresis loops demonstrating the evolution of soluble reactive phosphorus (SRP) concentrations under different stages (overland flow water levels) during events. Time throughout a given event, expressed as a percentage of the total event duration, is shown by colour. Periods of "merged" are differentiated from other periods of the event using solid symbols. Other stages were represented by letters. "P" indicates ponding, "R" indicates rising, "F" indicates falling and "D" indicates drying, respectively. Arrows show the direction of the hysteresis loop. A clockwise loop indicates that SRP concentrations are decreasing throughout the event (supply exhaustion or dilution effect), whereas an anti-clockwise loop indicates that SRP concentrations are increasing. Field B data for event 15 is not shown as it was not sampled.

being mobilized throughout an event (anti-clockwise loops), or, if an apparent supply exhaustion or dilution effect was occurring (clockwise loops), and, to determine how flooding affected this.

Soluble reactive P concentrations showed an incomplete "eight" shaped hysteresis loop in most events (All events for Field A and Event 5 for Field B) indicating switching controls in the mobilization/immobilization of P. Early dilutions in SRP concentrations were observed during the ponding stages of several of the monitored events (e.g., Events 5, 6 and 15 for Field A and Event 5 and 6 For Field B, Figure 5.5). Further dilutions in SRP concentrations were observed during the rising stage of the monitored snowmelt events in Field A. However, SRP concentrations increased during the rising stage of the monitored rainstorm events and the snowmelt event in Field B. Although SRP concentrations increased during the early "merged" stages, they began to decline towards the later "merged" stages in all events except Event 2, forming a clockwise loop within the "merged" stage. In general, SRP concentrations during the falling limb of the overland stage changed very little and decreased during the drying phase.

5.3.3 Relationships between SRP concentrations and other geochemical parameters during events

The SRP/TP ratios were explored to understand the dominant flow processes and potential P sources (Figure 5.6, Appendix D2, D3). Greater SRP/TP ratios were observed during the “merged” periods for all events except event 2 (Appendix D2), indicating the dominance of the dissolved P

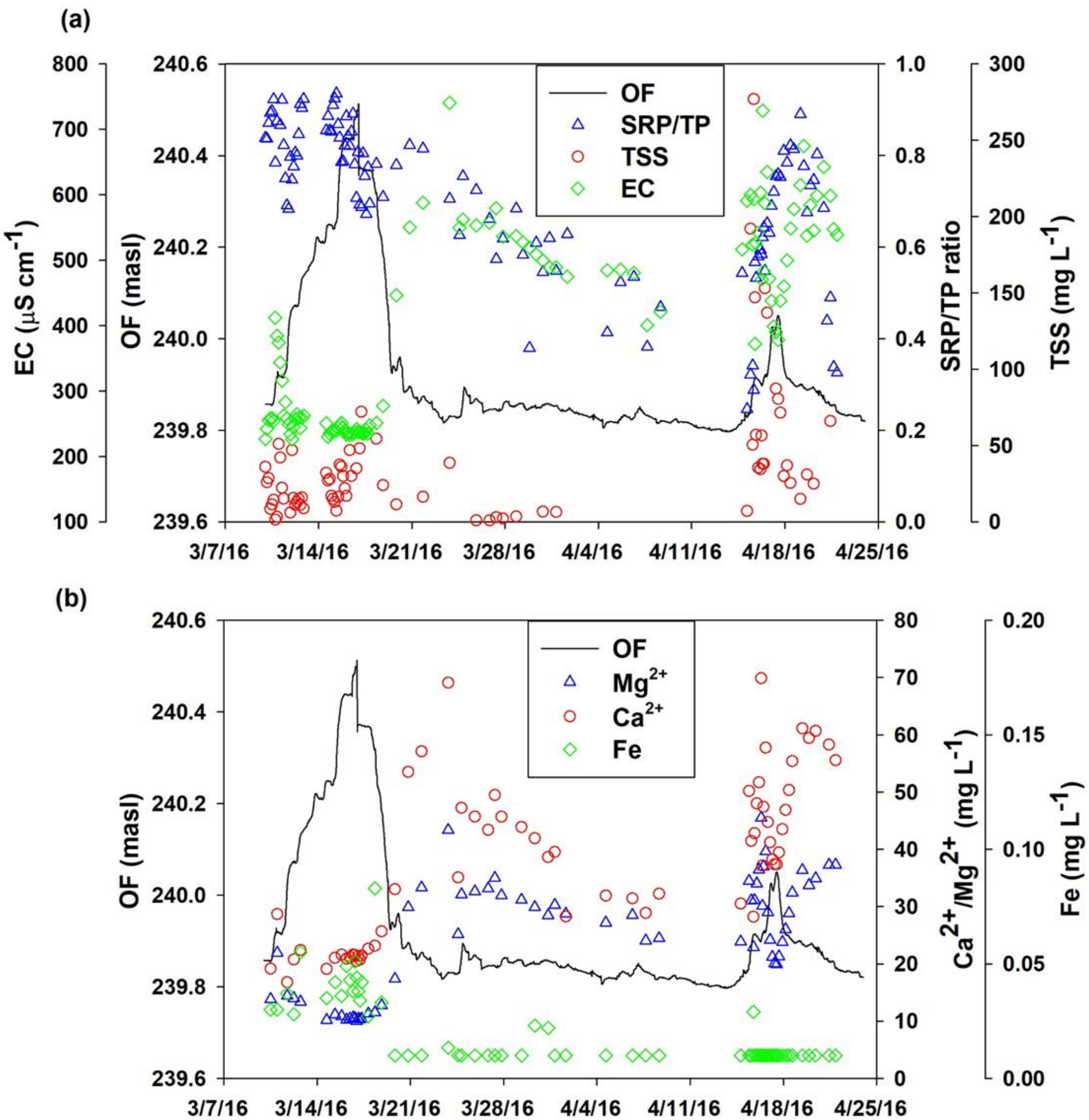


Figure 5.6. Overland flow (OF) soluble reactive phosphorus (SRP)/total phosphorus (TP) ratios, total suspended solids (TSS), electrical conductivity (EC) (a) and the concentrations of dissolved Mg^{2+}, Ca^{2+} and Fe ions concentrations (b) in Field A during Events 5 and 6.

fraction during "merged" conditions. In Field A, the TSS concentrations were negatively related to SRP/TP ratio and were smaller during the "merged" conditions indicating potential retarded flow or settling of particulates (Figure 5.6a).

The concentrations of the EC and dissolved Ca^{2+} and Mg^{2+} ions were also assessed to explore the potential sources of the runoff water. In general, lower EC concentrations indicate the dominance of the ion poor snowmelt or rainwater, whereas greater EC values indicate the dominance of the pore or groundwater sources. The EC concentrations were lower during the "merged" periods for most of the events (Figure 5.6, Appendix D3). The EC concentrations were also lower during the ponding and rising limps of the snowmelt flooding events (Figure 5.6). Calcium and Mg^{2+} ion concentrations positively related to EC concentrations. Concentrations of Ca^{2+} and Mg^{2+} were low during the snowmelt period (Event 5) when merging occurred but increased once the water began to drain in both fields (Figure 5.6). A brief decline in Ca^{2+} and Mg^{2+} concentrations was also observed during the short "merged" period that occurred during the subsequent spring storm (April 2016). Dissolved Ca and Mg ions showed a negative relationship with SRP concentrations for monitored events 5 and 6 (Figure 5.7).

Relationships between SRP and pH, EC, Fe, Ca^{2+} and Mg^{2+} concentrations were explored during merged periods to infer potential mechanisms of P retention or release. pH and SRP were negatively related during the "merging" periods in both fields for Events 2 (snowmelt 2015), 5 (snowmelt 2016) and 6 (spring storm 2016) (Figure 5.8). However, this was not observed during the snowmelt runoff of 2017 (Event 15). Clear relationships were not apparent between EC and SRP (Figure 5.8).

Dissolved Fe concentrations were generally small and were only detected during the ponding and "merged" periods of Event 5, during which they were generally less than 0.05 mg Fe L^{-1} (Figure 5.6).

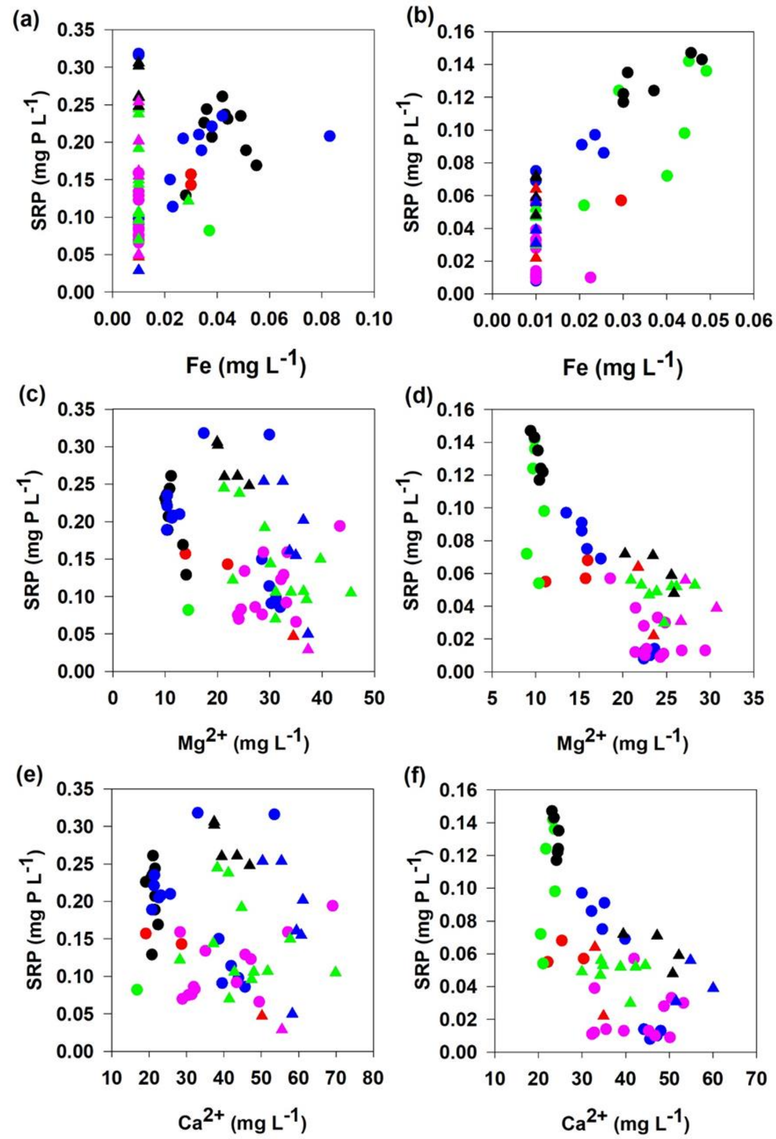


Figure 5.7. Relationships between soluble reactive phosphorus (SRP) concentrations and dissolved Fe, Mg^{2+} and Ca^{2+} concentrations in Field A (a,c and e) and Field B (b, d and f) during Events 5 and 6. Event 5 is represented by circles and Event 6 is shown with triangles. Red colour indicates ponding, green indicates rising, black indicates merged, blue indicates falling and pink indicates drying, respectively.

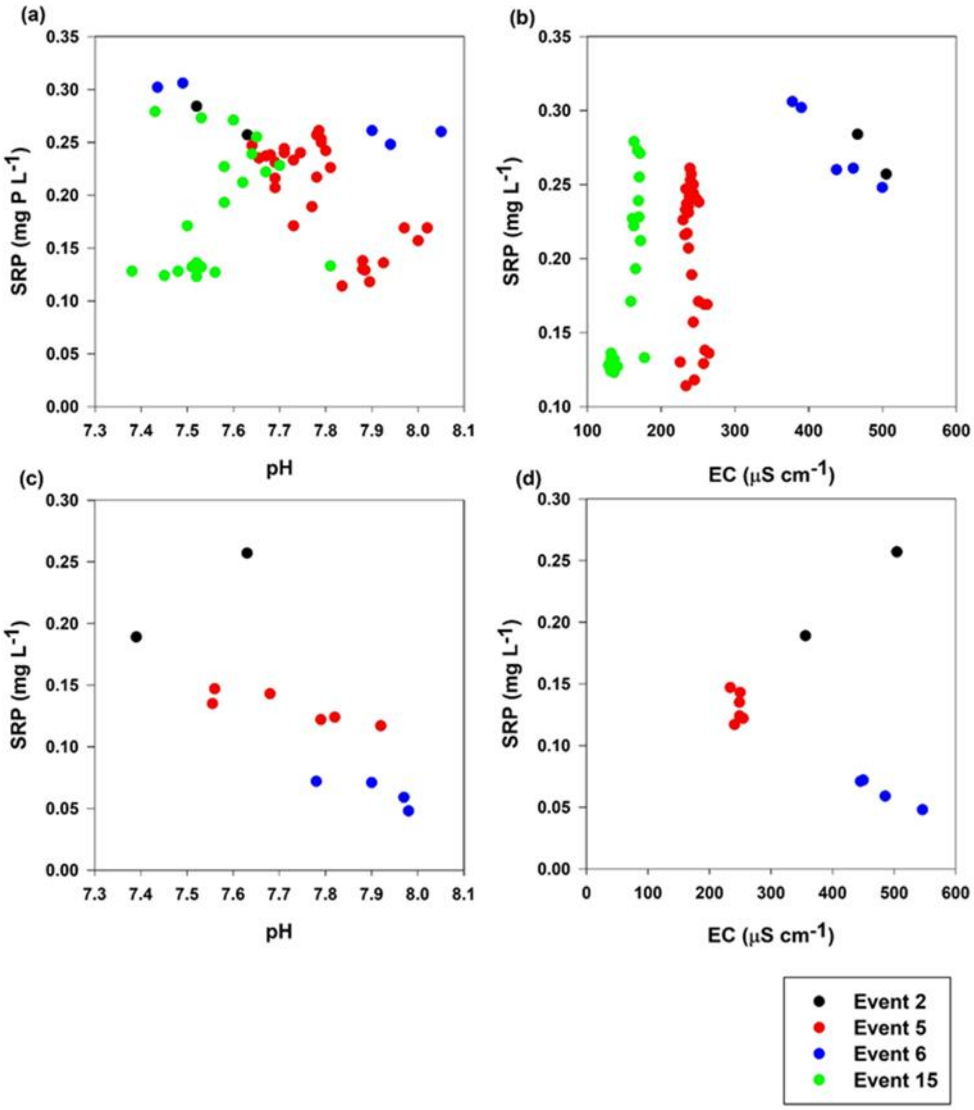


Figure 5.8. Relationships between soluble reactive phosphorus (SRP), electrical conductivity (EC) and pH during the merged periods. Plots a and b represent Field A and plots c and d represent Field B, respectively.

Dissolved Fe concentrations were below the detection limit (0.02 mg Fe L^{-1}) during falling limb and drying period of Event 5 and throughout event 6 in both fields (Figure 5.8).

Daily SRP concentrations between fields and their respective ditches did not significantly vary ($p > 0.05$). Comparisons were not made for 2017 due to lack of water sampling from Field B.

5.3.4. Soil chemistry along the field to ditch continuum

Differences in soil P concentrations and solid-phase partitioning were observed along transects from the field to the ditch; however, these differed between the two fields (Table 5.2, Figure 5.9). Within the fields, Olsen-P concentrations within the top 6 cm of soil were 27 ± 11 mg kg^{-1} for Field A and 26 ± 5 mg kg^{-1} for Field B (median ± standard deviation). However, Olsen P concentrations declined in the 6-15 cm layer (10 ± 3 mg kg^{-1} for Field A and 11 ± 4 mg kg^{-1} for Field B, respectfully. Within the fields, Olsen P concentrations were generally greatest in the midfield location. In contrast, Olsen-P concentrations in ditch sediments were considerably lower (19 ± 7 mg kg^{-1} for Field A and 4 ± 1 mg kg^{-1} for Field B in the top 6 cm). Phosphorus sorption indices were similar within the two fields (Median=306, range 274-381 l kg^{-1}) and did not vary with position within the field but were greater in ditch sediments (Median=390, range 372-408 l kg^{-1}).

Differences in the inorganic forms of P (P_i) explored using sequential extractions showed variability with landscape position (Table 5.2, Figure 5.9). For example, Sol-P_i was found to be enriched at the midfield locations of the fields, similar to Olsen-P (Table 5.1). However, Sol-P_i was a much smaller fraction of the total Pi (~ 10%) than the other P_i fractions.

The acid-soluble P fraction (HCl-P_i) was the dominant P fraction in the soils from both fields (Figure 5.9). The HCl-P_i concentrations within the top 6 cm of soil were 219 ± 12 mg kg^{-1} for Field

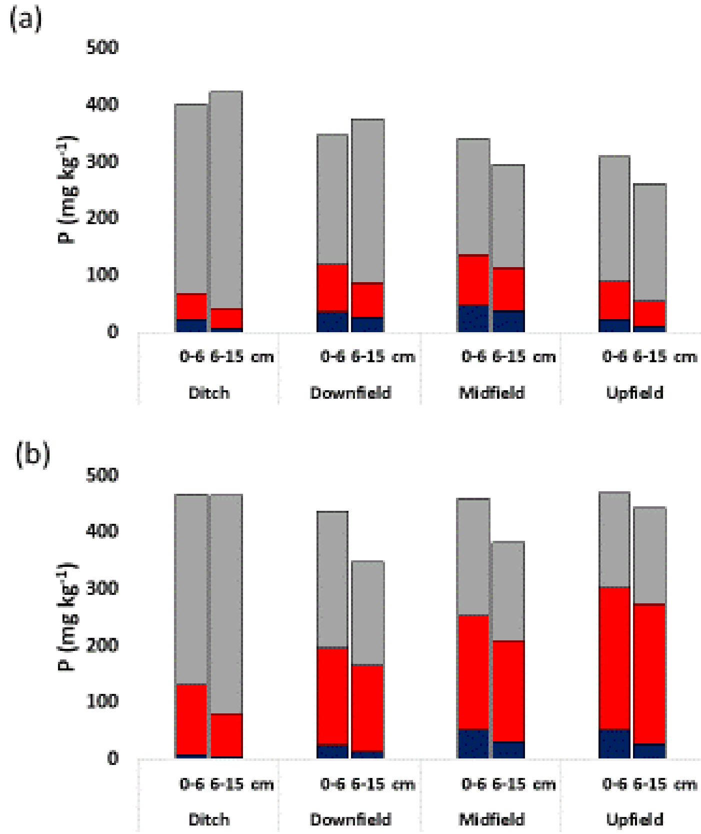


Figure 5.9. Soluble (blue), reducible (red) and acid soluble phosphorus (grey) (P) contents of the surface (0-6 m) and subsurface (6-15 cm) soils from the ditches and different landscape positions of field A (a) and field B (b).

A and 206 ± 40 mg kg^{-1} for Field B (median ± standard deviation). This was more apparent in Field A than in Field B. The HCl-P_i fraction was greater in the ditch sediments in both fields compared to soils in fields. These differences coincided with greater carbonate contents in ditch sediments (Table 5.2). The reducible P (CBD-P_i) content was found to be significantly higher in Field B and its ditch when compared to Field A and its corresponding ditch (p < 0.01) (Figure 5.9)

Table 5.2. Measured soil parameters. P: Phosphorus; PSI: Phosphorus sorption index.

Parameters	Field	Depth	Ditch	Downfield	Midfield	Upfield
Olsen P (mg/kg)	A	0-6 cm	15.98	28.86	32.85	15.23
		6-15 cm	9.54	13.37	10.15	7.64
	B	0-6 cm	3.65	22.95	29.88	25.85
		6-15 cm	1.29	4.76	11.67	11.25
PSI (l/kg)	A	0-6 cm	407.81	274.39	276.24	362.97
		6-15 cm	390.03	329.23	381.82	363.43
	B	0-6 cm	371.92	298.05	313.46	322.76
		6-15 cm	399.7	333.79	324.91	340.03
P sequential extraction						
Soluble P (mg/kg)	A	0-6 cm	22.42	35.78	47.42	21.49
		6-15 cm	7.53	25.08	37.24	11.31
	B	0-6 cm	7.97	24.49	50.19	50.56
		6-15 cm	4.13	12.79	30.97	27.3
Reducible P (mg/kg)	A	0-6 cm	45.76	83.43	88.95	67.57
		6-15 cm	32.95	62.00	75.38	43.34
	B	0-6 cm	122.55	171.25	202.45	250.61
		6-15 cm	74.69	152.28	177.25	244.9
Acid soluble P (mg/kg)	A	0-6 cm	334.1	227.04	203.69	219.26
		6-15 cm	381.98	286.97	182.59	207.38
	B	0-6 cm	335.77	241.10	205.84	167.22
		6-15 cm	385.24	181.75	172.55	169.11
Soil organic matter (%)	A	0-6 cm	7.22	8.83	9.79	8.75
		6-15 cm	4.55	7.15	9.5	7.88
	B	0-6 cm	5.97	7.48	9.3	11.35
		6-15 cm	4.88	6.61	7.99	9.97
Soil Fe content (mg/kg)	A	0-6 cm	4148	3613	4309	6257
		6-15 cm	4363	4479	5278	6945
	B	0-6 cm	5862	5837	5854	6490
		6-15 cm	6919	5765	5797	6483
Soil carbonate content (%)	A	0-6 cm	4.97	2.83	2.30	2.8
		6-15 cm	5.57	3.64	2	2.46
	B	0-6 cm	4.1	2.42	2.11	1.75
		6-15 cm	5.38	2.17	1.93	2.04

but was smaller in the ditches relative to the upland soils in both fields. These spatial differences did not coincide with the total Fe content of the soils, as Fe concentrations were greatest in the upfield positions of both fields and did not differ between the ditch sediments and upfield fields.

5.4. Discussion

5.4.1. Frequency of inundation

This study observed events with prolonged flooding periods (> 1 day) in all 3 study years, with 1-3 flooding events occurring in a given year. Of the 5 runoff events that experienced flooding, two were associated with early spring snowmelt runoff. In all 3 years, early spring snowmelt began as radiation melt. However, in 2016 and 2017, the snowmelt was hastened by rain on snow events, which triggered rapid melts. Several soil, weather and anthropogenic factors influence the progression of spring runoff and its tendency of turning into a multiday inundation event in the Red River Valley. For example, a wetter fall season followed by a severe winter substantially decreases the infiltrability of meltwater through wet, frozen soils, thus increases the chance for spring snowmelt flooding (Wazney and Clark, 2016). This flooding risk is further exacerbated by rain on snow events, which not only add additional water for runoff but also rapidly melt/ripen the snowpacks (Jeong and Sushama, 2018). In general, meltwater exited the monitored fields once the adjacent ditches begin to thaw. However, during rapid melt such as rain on snow events, the ditch water often flowed back into fields potentially due to frozen culverts and ice jams in the nearby provincial channel where the ditches drained. This eventually led to prolonged flooding periods during snowmelt (1-7 days of the 7-14-day event).

This study also observed three major flooding events following major late spring and thunderstorms in 2015 and 2016. Previous studies that have discussed the occurrence of flooding events in the Red River Valley have related them only to early spring snowmelt runoff (e.g.

Cordeiro et al., 2017; Schindler et al., 2012) and less is known about flooding during late spring or summer events. Our study shows that sizeable late spring multi-day rain events and thunderstorms also have the potential to flood the agricultural fields in this region, which may have implications in the future as frequent multiday spring and summer storms are forecasted for the Prairies in the future (Shook and Pomeroy, 2012). Major thunderstorms also occurred during the summer (Table 5.1). However, they did not trigger field scale flooding potentially due to higher evapotranspiration and crop demand.

5.4.2. Evolution of phosphorus concentrations during flooding events

Observations from this study suggest that there is potential for P release into floodwater during “merged” conditions. In these moderate P soils, there is a finite amount of P available to be mobilized in runoff (Table 5.2), and consequently, supply exhaustion or a dilution effect was expected (Liu et al., 2013). Indeed, an initial dilution in SRP was observed during the ponding and rising phases of the snowmelt flooding events (Figure 5.5, 5.6). It is unclear whether this is supply exhaustion of P released from soils or, dilution by the melting snowpack in the ditches and riparian areas for the snowmelt flooding events. For rainstorm events, SRP concentrations increased during ponding and rising stages, potentially due to erosion forces as indicated by lower SRP/TP ratios and greater TSS concentrations.

For Events 5, 6 and 15, the SRP concentrations subsequently increased during the initial “merged” stage, indicating P may have been released to P poor floodwater. In addition, increasing SRP/TP ratio and decreasing TSS concentrations during the “merged” stage also imply that this released P might have been mobilized primarily in its dissolved form. However, SRP concentrations began to decline towards the end of the “merged” stage for most occasions potentially due to dilution or supply exhaustion. During Event 2, the SRP concentrations were higher in the rising limb and declined during the "merged" stage. The fields were fertilized only a few days before the rainstorm that triggered the flooding for Event 2.

However, SRP concentrations increased again during the falling limb due to increased flow, as indicated by the decreasing SRP/TP ratio and increasing TSS concentrations (Figure 5.6, Appendix D2). Decreasing SRP concentrations and increasing EC, Ca^{2+}, and Mg^{2+} concentrations suggest potential precipitation of P during the drying stage as its Ca-P or Mg-P forms (Chow and Eanes, 2001).

5.4.3. Potential mechanisms influencing P dynamics during "merged" periods

Observations from the current study show potential P mobilization during the monitored flooding events (Figure 5.5). During flooding periods, P could have been mobilized through desorption through increased soil-water contact periods, pH mediated dissolution or redox-mediated dissolution reactions (Liu et al., 2013; Chow and Eanes, 2001; Amarawansha et al., 2015). However, P concentrations during the "merged" periods were not substantially higher when compared to other stages of the runoff, especially rising and falling limbs. It is possible that P concentrations during the rising and falling limb of the runoff could have been affected by other potential drivers like fertilization (as observed in Event 2) and erosion processes, as indicated by the elevated TSS concentrations (Sharpley and Kleinman, 2003).

Soil P concentrations (Olsen) and forms from the current study indicate that there is a potential for P to be lost during inundation through desorption through increased soil-water contact periods. Surface soils with high soil test P (STP) risk losing P during runoff events (Sharpley et al., 2001b). Olsen P contents at the study sites were moderate for Prairie soils and comparable to soil test P that was observed in other studies from Southern Manitoba (Tiessen et al., 2010; Wilson et al., 2019). Runoff P concentrations observed in this study were also comparable to the abovementioned Prairie studies (Kokulan et al., 2019b; Tiessen et al., 2010). However, the methods adopted in this study were not adequate to determine the amount of this soil test P that was released to floodwater during the "merged" periods.

A substantial amount of soil P at the study sites was held in a reducible P phase (~ 37%), particularly in Field B (Figure 5.6). Prolonged soil inundation can elevate P release due to reductive dissolution of P binding Mn^{4+} and Fe^{3+} oxides. In fact, the majority of Manitoban soils were found to be prone to redox P losses (Amarawansha et al., 2015). The present study observed an increase in dissolved Fe concentrations during the ponding, rising and "merged" stages of the monitored 2016 snowmelt (Figure 5.6). This increase in Fe was positively related to the rise in SRP concentrations (Figure 5.7). This shows some of the CBD-P_i could have been mobilized during the progression of snowmelt runoff. However, dissolved Fe was not detected during the subsequent spring storm. Further research is required to understand the drivers behind these contrasting observations.

The majority of the soil P in this study was bound to relatively unavailable Ca and Mg phosphates (~ 54%). In calcareous Manitoban soils, the formation of carbonic acid due to increased partial pressure of CO_2 could reduce the pH towards neutrality during flooding (Amarawansha et al., 2015). In addition, melting snowpack and rain also could reduce the pH (Casson et al., 2014). This pH alteration could destabilize P bearing metal oxides and, increasing P concentrations in the water column. The current study also found negative relationships between runoff P concentrations and pH for the majority of the monitored events (Figure 5.7). In calcareous environments, increases in P with decreasing pH are often related to increases in Ca^{2+} concentrations (Chow and Eanes, 2001). However, this study did not find an increase in dissolved Ca and Mg ion concentrations during snowmelt "merged" stage (Event 5), which implies the HCl-P_i was unlikely to be solubilized during this brief reduction in pH (Figure 5.7, 5.8). Electrical conductivity also did not change during the "merged" period and often remained below 200 μS cm^{-1} during snowmelt, indicating the absence of the ions (Figure 5.8).

In contrast to "merged," an increase in the dissolved Mg and Ca concentrations was observed during the falling limbs of events, post-merged and when the drying phase ensues (Figure. 5.7, 5.8). This increase was associated with a decrease in SRP concentrations (Figure 5.3, 5.4, 5.7). Therefore, the data suggests the operation of P immobilizing mechanisms following the flooding events. The southern Manitoba region is underlain by Ca and Mg-rich parent materials (Ige et al., 2005). Therefore, subsurface flow in this region is usually higher in EC and enriched by dissolved Ca^{2+} and Mg^{2+} ions. Calcium and Mg precipitate as their respective phosphates in high pH environments (Dharmakeerthi et al., 2019; Plach et al., 2018b). This could explain the decline in SRP concentrations during the later stages of the runoff.

Roadside ditches are unlikely to be elevating runoff P losses during flooding events in this study. Ditch and swale SRP concentrations were statistically similar ($p > 0.05$). In fact, a substantial quantity of HCl-P_i was found in the ditch sediments. In addition, roadside ditches showed higher amounts of carbonates and possessed greater P sorption capacities (Table 5.2). Thus, it appears that Ca and Mg from the ditches may be retaining the P in this study. Given that roads are graded with limestone, this material may be assisting in the reduction of P loss in the region. However, our results have to be interpreted carefully considering the limited water and soil sampling of roadside ditches. Future studies should address this potential role that roadside ditches could play in P dynamics in the Red River Valley region.

In addition to STP, pH, redox reactions and roadside ditches, some other factors also could have influenced the P dynamics in this agricultural landscape. Potential of plant materials as P sources for snowmelt runoff is well documented in cold environments (Liu et al., 2019). Even though crop residue was hayed and removed from the fields, the ditch vegetation was not managed. Future research should consider assessing the influence of ditch vegetation to P mobilization (early spring snowmelt) and immobilization (intake by ditch vegetation during late spring). Likewise, P derived from sediment and dust from the adjacent farm road into ditches was also not assessed in this research but may be necessary. Future research should devise means and methods to get a complete understanding of the P dynamics in the ditch-field interface during inundation events.

5.5. Conclusions

This study documented the occurrence of inundation in the agricultural landscape of Manitoba during early spring snowmelt, late spring rainstorms and thunderstorms. Even though events with inundation periods were few compared to total runoff events, they contributed to higher annual runoff and edge of field P losses. Soil and runoff water sample analyses suggest that P might have been mobilized from the available soil P pool or due to redox dissolution reactions during periods of flooding in the fields. Future research should consider characterizing and quantifying specific contributions from each source. Future studies should also consider the role of subsurface water in immobilizing runoff P especially in the latter stage of runoff.

Reducing the occurrence of inundation and P losses by improving the farm drainage is controversial, considering the nearly flat Prairie landscape and the unpredictable nature of flooding in cold environments. Improving farm drainage may also increase downstream flooding risk. Enhancing subsurface drainage such as tiles also have limited advantages as soils are frozen or partially frozen during cold periods where the majority of runoff occurs (Kokulan et al., 2019b). Therefore, re-routing surface floodwater to on-farm retention structures such as retention ponds or wetlands and reusing during the water demand periods might have potential economic and environmental benefits.

5.6. Acknowledgements

This project was funded by the Manitoba Agriculture, Food and Rural Development (MAFRD), Manitoba Conservative Districts Association (MCDA), Environment and Climate Change Canada's (ECCC) Lake Winnipeg Basin Stewardship Fund, the Natural Sciences and Engineering Research Council of Canada (Macrae–DG and Lobb–DG), the Canada First Excellence Research Fund (Agricultural Water Futures), and the Canadian Agri-Food Policy Institute (CAPI). Northern Plains Drainage Systems, Brendan Brooks, Anthony Buckley, Eva Slavicek, Vito Lam, Bo Pan and Reza Habibiandehkordi are thanked for the field and laboratory assistance.

Chapter 6: Overall synthesis and conclusions

There has been an increase in tile drainage system installation in Southern Manitoba over the past two decades (Council of Canadian Academies, 2013) in order to improve crop production. Predictions of climatic changes in the region have shown likely increases in the frequencies of spring floods and intense summer thunderstorms, which will likely accelerate the rate of tile installations to address increasing uncertainty in periods of excess moisture on fields. Given current environmental and political concerns related to agricultural pollution and the eutrophication of Lake Winnipeg (Schindler et al., 2012), the role that tiles may play in both runoff and nutrient loading from agricultural fields must be evaluated due to the capacity for tiles to export significant quantities of nutrients such as phosphorus (P) and nitrogen (N) from croplands by acting as subsurface lateral conduit pathways (King et al., 2015). In Canada, much of the existing research on runoff from artificially drained agricultural landscapes has originated from the Great Lakes Region and St. Lawrence River Lowlands (King et al., 2015). However, hydrologic and biogeochemical processes in tile-drained systems in Southern Manitoba are relatively understudied.

This thesis was the first study that simultaneously monitored overland and tile flow responses in the Red River Valley basin on an annual basis, including annual spring snowmelt, spring storms and thunderstorms. In this thesis, I monitored overland flow, tile flow, groundwater table and ditch flow from two agricultural farms (one tile-drained; one surface drained) from 2015 to 2017 to quantify edge of field runoff and nutrient losses, to characterize surface-tile connectivity through the vadose zone, and to characterize ditch-overland flow dynamics at the edge-of-field. In addition, a series of infiltration and percolation tests were also performed under different antecedent soil moisture conditions to characterize in-field hydrochemical processes. Soil samples were also collected from the fields and adjacent roadside ditches and analyzed for different P fractions and P adsorption capacities. Findings from the current study have implications on the present expansion of the tile drainage and water quality issues in the Southern Manitoba region.

Chapter 2 of this thesis suggests that overland flow will prevail as the primary pathway for runoff and nutrient (N and P) losses as substantial annual runoff occurs during the snowmelt period when tile flow is restricted due to the frozen subsurface (Figure 6.1). Therefore, current conservation strategies that are aimed at reducing overland flow-related nutrient losses in this region should be continued regardless of the presence or absence of tile drainage. Although tiles did not flow readily during the peak snowmelt runoff period, they did flow in late spring following large rainstorms on wet antecedent conditions. Tile drainage is useful in lowering the groundwater table under wet conditions on thawed ground, and therefore tiles likely to have the potential to protect field crops from the adverse effects of extreme moisture during the open water periods. This thesis has shown that the potential crop benefits of tiles are not accompanied by a significant risk for P loss via tile drainage. Indeed, this study did not find tile drainage as the primary pathway of the substantial edge of field losses (Chapter 2). In fact, tiles were responsible for less than 5% of the annual P losses. These findings contrast other research that has found that a substantial amount of P is preferentially transported to tile drains in clayey textured soils on an annual basis (King et al., 2015). The potential reasons for the small losses of P in tile drainage were explored in Chapters 3 and 4 through characterizations of in-field hydrochemical processes.

Chapter 3 demonstrated the rapid activation of the overland flow in this landscape despite the presence of tile drainage, and used the novel approach of response times to better understand surface-tile connectivity. Activations of both overland flow and groundwater table were hastened by rainfall intensity, whereas the activation of tile flow was influenced by the combined effects of rainfall intensity and antecedent moisture conditions. The infiltration capacities of vertisolic soils were substantially reduced once the soils become wetter due to closure of the desiccation cracks through clay swelling during periods of sufficient moisture inputs. Retarded infiltration favoured the activation of the overland flow following moderate to high-intensity rainstorms in wetter conditions. Outside of these conditions, overland flow was also immediately activated following the high-intensity thunderstorms. The activation of tile drainage often lagged behind overland flow, indicating the absence of direct surface-tile connectivity (preferential flow pathways), despite the presence of surface cracks. Instead, the soil profile was often wetted from the top through a slowly propagating wetting front or wetting front advancement with partial coupling with overland flow. These observations showed that suppressing the frequent activation of overland flow pathways in this near-flat vertisolic clay landscape with tile drainage may not be effective.

Chapter 4 explored tile drain chemistry under different antecedent conditions and with differing event drivers and demonstrated that although tile drainage could respond significantly during shallower groundwater table conditions in the thawed soils (Chapter 4; Figure 6.1), tile water chemistry during this significant period resembled that of groundwater and showed smaller P concentrations and elevated N concentrations. In contrast, tiles did not significantly flow in early spring, despite the presence of a shallower groundwater table. Field monitoring has shown this groundwater table in the early spring was a perched water table over a deep subsurface frozen layer. This layer largely restricted the water percolation through subsurface, thus limiting the tile flow. However, given that the electrical conductivity (EC) of the tile flow closely resembled overland flow during this period, this indicates that the minimal tile flow that occurred in the early spring would have occurred through few cracks (preferential flowpaths) in the frozen ground. Preferential flow was also observed in summer on dry soils, although not always. Indeed, the EC of tile flow in summer appeared to be a mixture of both overland flow and groundwater. Rapid activation of tile drains was observed in the summer months following high-intensity storms. In addition, infiltration and slug tests also showed rapid water movement in drier conditions. However, the summer tile outflows were small, potentially due to the closure of desiccation cracks and the lack of a shallower groundwater table. Thus, although preferential flow was observed in tile drains, leading to elevated P concentrations in tile drainage but smaller N concentrations, this was restricted to the spring (melt) and summer periods when tile flow was minimal, and most of the seasonal tile flow (i.e. late spring) had low P concentrations and elevated N concentrations that resembled groundwater, suggesting that a considerable quantity of the tile flow volume was matrix flow.

Chapters 3 and 4 have provided an improved understanding of surface-subsurface (tile) connectivity and nutrient dynamics in the vertisolic clays of the Red River Valley. Although P dynamics in vertisolic clays have been studied previously (Harmel et al., 2019), and, P dynamics in tile-drained clay soils have also been studied (Turtola and Jaakola, 1995; Smith et al., 2015), vertisolic clays in the Northern Great Plains have not been studied. Work done in this thesis is novel as it provides an improved understanding of the combined roles of antecedent temperatures (i.e. thawed or frozen ground), antecedent soil moisture and event characteristics in determining P dynamics both within a field (surface-tile P dynamics) and at the edge-of-field. Such information

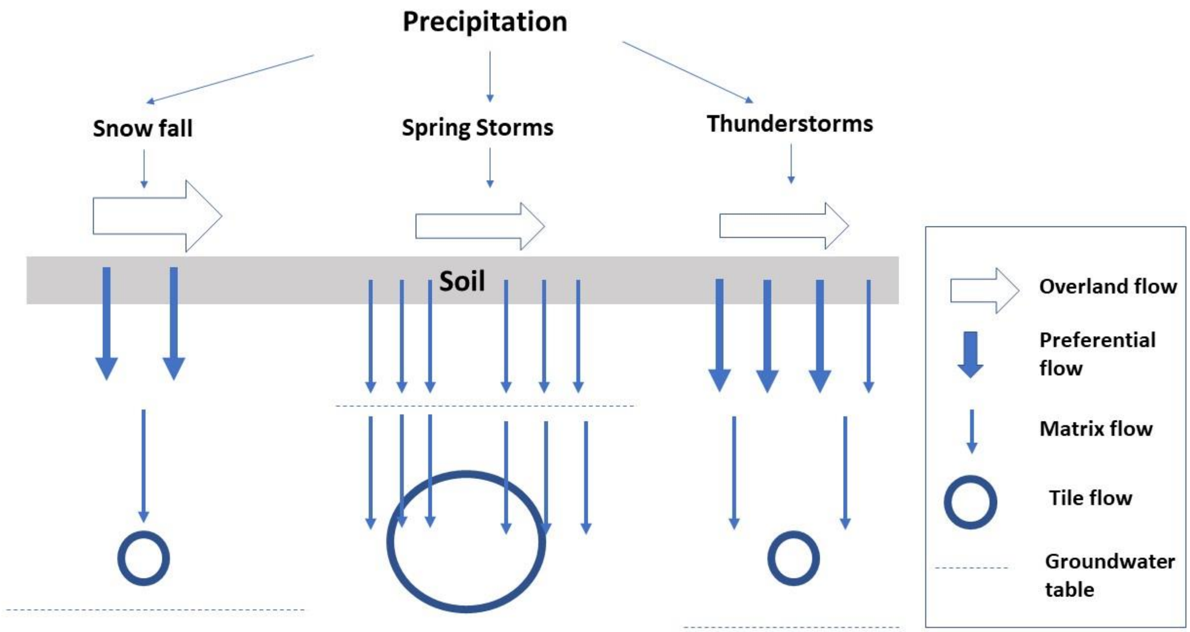


Figure 6.1. Schematic diagram showing the overland flow and tile response in the vertisolic clay soils of the Red River Valley. Early spring snowmelt runoff was dominated by overland flow due to frozen ground. Water movement at near-surface was observed. However, this has resulted in little tile flow. Low intensity, long-duration spring rains favoured substantial tile outflows due to wetter soil and groundwater conditions. However, activation of overland flow was not entirely offset by tile drains. Both tile and overland flows were lower during summer due to drier conditions despite high-intensity storms. Direct surface-tile connectivity was not observed in any conditions. Partial connectivity between surface and tile was seen in early spring and summer whereas the matrix flow dominated the tile flow in late spring.

on the process can improve our understanding of the processes driving tile drain hydrological and biogeochemical processes and can assist in the development of predictive models.

Although annual runoff and P loss are primarily driven by overland flow on frozen ground during snowmelt (e.g. Chapter 2 of this thesis; Cordeiro et al., 2017), the magnitude of P loss in overland flow may also be influenced by the overbank flooding of the roadside ditches. Previous research in this region has reported overbank flooding of fields during the annual spring snowmelt runoff period (e.g. Cordeiro et al., 2017). However, I also observed overbank flooding during multiday spring storms and large thunderstorms. Chapter 5 provides insight on water samples collected during flooding events, which indicate potential P release to floodwater, and on water and soil analyses, which show the potential for P mobilization from available and reducible soil P fractions. Although this potential exists, sediments within ditches also appeared to be retaining P rather than mobilizing it into floodwater. Future research should deploy a mass balance approach to understand the P dynamics during flooding events. An improved understanding of ditch management on P loss from fields is needed.

There were several uncertainties at the beginning of this study about the potential roles of tile drainage on the edge of field runoff and nutrient losses in the Red River Valley region. The findings from this study suggest that tile drainage will not increase edge of field runoff and P losses in this region. Although crop performance was not examined in the current study, tile drainage may have agronomic benefits by dewatering the soil profile during late-spring and summer, which may reduce crop losses that typically occur when low-relief vertisolic soils are saturated. Given the predicted increase in the frequency of multiday spring and summer storms in the Prairies with climate change (Shook and Pomeroy, 2012), tiles have the potential to be a tool for farmers to tackle this uncertainty. However, these potential benefits must be carefully approached considering the inefficiency of the tile drainage to significantly reduce overland flow runoff and its associated nutrient losses in this region. In addition, tiles may also exacerbate the nitrogen problem in this region. In the current study, tile water was re-routed to a larger in-farm retention pond rather than being discharged to the adjacent drainage ditch. One reason this was done was to ensure that tiles did not exacerbate the P problem, as has been shown in other regions such as Ohio, USA. However, this strategy was unlikely to have made an impact on the edge of field P losses as > 95% annual P losses were associated with overland flow, and tile drains did little to change the occurrence of surface runoff. Indeed, this study found that the majority of the overland flow P losses were related to events that were associated with flooding of fields and adjacent roadside ditches. Consequently, an improved management strategy that could make a significant difference to edge-of-field P loss would be to route the overland flow and/or the ditch water to the retention pond during flooding periods, and either allow this water to evaporate or use it to irrigate crops during subsequent dry periods. This is an area where subsequent work is needed.

The role of tile drainage systems as pathways for runoff and nutrient losses could become prominent in this region under a future warming climate. At present, tile flow during spring snowmelt is mostly restricted because of a thick soil-ice layer. However, this thickness of the soil-ice layer could be reduced by warmer winter air temperatures and frequent rain-on-snow events. In such conditions, the contribution of tile flow to winter runoff and nutrient losses could be significant (Plach et al., 2019). Similarly, drier summers could also lead to deeper desiccation cracks that could potentially extend to the tile depth (1 m). We may see increased summer tile outflows and nutrient concentrations through a direct surface-tile connectivity during major storms that follow drier periods. The duration (3 years) of this study may not be adequate to assess the trends in hydrologic responses to a changing climate. Therefore, future observations are recommended.

Limitations

Although runoff was monitored, nutrient losses during the fall were not assessed in the fall of 2015 and 2016 in this study. Future research should include the contributions of autumnal runoff processes, particularly as they may have implications on the development of the soil-ice layer in the following winter, which is vital in driving P losses at the edge of field the next spring. In addition, only mineral fertilizers have been used in this study. Future research should also consider assessing the impact of manure on tile related runoff and nutrient losses in this region. Although this study has monitored the occurrence of overbank flooding of roadside ditches and their implications for P concentrations, monitoring strategies were not designed to pinpoint all hydrobiogeochemical mechanisms (e.g. redox potential) for the elevated P release during merged conditions in individual events. Future research is required in this topic to identify the sources and processes that could exacerbate P losses to runoff during the inundation events.

To summarize, the major conclusions from this study are listed below;

- ✓ Overland flow was the major pathway for runoff and nutrient losses in this near-level Southern Manitoban landscape with vertisolic soils despite the presence of the tile drainage.
- ✓ Tile drainage did not exacerbate edge of field P losses.
- ✓ Tile drainage could elevate the edge of field N losses.
- ✓ Presence of tile drainage did not significantly reduce the occurrence of the overland flow
- ✓ The activation of overland flow was influenced by rainfall intensity.
- ✓ The activation of the tile drainage was influenced by rainfall intensity and antecedent moisture conditions
- ✓ Substantial tile flow was observed during periods with a shallow groundwater table
- ✓ Tiles barely flowed during early spring snowmelt due to the presence of a thick soil-ice layer

- ✓ Water movement into tile drainage was influenced by matrix flow pathways during wetter antecedent conditions. Preferential flow pathways dominated during the frozen and drier conditions.
- ✓ Overbank flooding was observed not only during snowmelt runoff but also during multiday spring storms and massive thunderstorms.
- ✓ Substantial reduction in P concentrations was not observed during the “merged” stage, which indicates the potential release of P to floodwater during the inundation events.
- ✓ Roadside ditches were likely to retain more P due to their higher P retention capacities.

Bibliography

Addiscott, T. M., & Thomas, D. (2000). Tillage, mineralization and leaching: phosphate. *Soil and Tillage Research*, *53*(3-4), 255-273.

Algoazany, A. S., Kalita, P. K., Czapar, G. F., & Mitchell, J. K. (2007). Phosphorus transport through subsurface drainage and surface runoff from a flat watershed in east central Illinois, USA. *Journal of Environmental Quality*, *36*(3), 681-693.

Ali, G., Macrae, M., Walker, M., Laing, J., & Lobb, D. (2018). Preferential flow in Vertisolic soils with and without organic amendments. *Agricultural & Environmental Letters*, *3*(1). doi:10.2134/ael2018.04.0018

Amarawansha, E. A. G. S., Kumaragamage, D., Flaten, D., Zvomuya, F., & Tenuta, M. (2015). Phosphorus mobilization from manure-amended and unamended alkaline soils to overlying water during simulated flooding. *Journal of Environmental Quality*, *44*(4), 1252-1262.

Ambroise, B. (2004). Variable 'active'versus 'contributing'areas or periods: a necessary distinction. *Hydrological Processes*, *18*(6), 1149-1155.

Bagnall, D., Morgan, C. L., Molling, C. C., Heilman, J. L., & Moore, G. W. (2019). Testing a Water Redistribution Model in a Cracked Vertisol at Two Scales. *Vadose Zone Journal*, *18*(1). doi:10.2136/vzj2018.09.0173

Baram, S., Kurtzman, D., & Dahan, O. (2012). Water percolation through a clayey vadose zone. *Journal of Hydrology*, *424*, 165-171.

Baulch, H. M., Elliott, J. A., Cordeiro, M. R., Flaten, D. N., Lobb, D. A., & Wilson, H. F. (2019). Soil and water management practices: Opportunities to mitigate nutrient losses to surface waters in the northern Great Plains. *Environmental Reviews*. doi.org/10.1139/er-2018-0101

Beauchemin, S., Simard, R. R., & Cluis, D. (1998). Forms and concentration of phosphorus in drainage water of twenty-seven tile-drained soils. *Journal of Environmental Quality*, *27*(3), 721-728.

Berg, A. S., & Joern, B. C. (2006). Sorption dynamics of organic and inorganic phosphorus compounds in soil. *Journal of Environmental Quality*, *35*(5), 1855-1862.

Betson, R. P. (1964). What is watershed runoff? *Journal of Geophysical Research*, *69*(8), 1541-1552.

Beven, K., & Germann, P. (1982). Macropores and water flow in soils. *Water Resources Research*, *18*(5), 1311-1325.

Beven, K., & Germann, P. (2013). Macropores and water flow in soils revisited. *Water Resources Research*, *49*(6), 3071-3092.

Bishop, J. M., Callaghan, M. V., Cey, E. E., & Bentley, L. R. (2015). Measurement and simulation of subsurface tracer migration to tile drains in low permeability, macroporous soil. *Water Resources Research*, *51*(6), 3956-3981.

Blaen, P. J., Khamis, K., Lloyd, C., Comer-Warner, S., Ciocca, F., Thomas, R. M., ... & Krause, S. (2017). High-frequency monitoring of catchment nutrient exports reveals highly variable storm event responses and dynamic source zone activation. *Journal of Geophysical Research: Biogeosciences*, *122*(9), 2265-2281.

Bol, R., Gruau, G., Mellander, P. E., Dupas, R., Bechmann, M., Skarbøvik, E., ... & Van der Grift, B. (2018). Challenges of reducing phosphorus based water eutrophication in the agricultural landscapes of Northwest Europe. *Frontiers in Marine Science*, *5*(276), 1-16.

Bolton, E. F., Aylesworth, J. W., & Hore, F. R. (1970). Nutrient losses through tile drains under three cropping systems and two fertility levels on a Brookston clay soil. Canadian Journal of Soil Science, 50(3), 275-279.

Bosch, N. S., Evans, M. A., Scavia, D., & Allan, J. D. (2014). Interacting effects of climate change and agricultural BMPs on nutrient runoff entering Lake Erie. *Journal of Great Lakes Research*, *40*(3), 581-589.

Bracken, L. J., & Croke, J. (2007). The concept of hydrological connectivity and its contribution to understanding runoff-dominated geomorphic systems. *Hydrological Processes*, *21*(13), 1749-1763.

Brierley, J. A., Stonehouse, H. B., & Mermut, A. R. (2011). Vertisolic soils of Canada: Genesis, distribution, and classification. *Canadian Journal of Soil Science*, *91*(5), 903-916.

Brocca, L., Melone, F., & Moramarco, T. (2008). On the estimation of antecedent wetness conditions in rainfall–runoff modelling. *Hydrological Processes*, *22*(5), 629-642.

Broughton, R., & Jutras, P., Farm Drainage (2013). In *The Canadian Encyclopedia*. Retrieved from https://www.thecanadianencyclopedia.ca/en/article/farm-drainage. (Accessed on February 14th, 2019).

Buchanan, B. P., Archibald, J. A., Easton, Z. M., Shaw, S. B., Schneider, R. L., & Walter, M. T. (2013). A phosphorus index that combines critical source areas and transport pathways using a travel time approach. *Journal of Hydrology*, *486*, 123-135.

Cade-Menun, B. J., Bainard, L. D., LaForge, K., Schellenberg, M., Houston, B., & Hamel, C. (2017). Long-term agricultural land use affects chemical and physical properties of soils from southwest Saskatchewan. *Canadian Journal of Soil Science*, *97*(4), 650-666.

Cameron, K. C., Di, H. J., & Moir, J. L. (2013). Nitrogen losses from the soil/plant system: a review. *Annals of Applied Biology*, *162*(2), 145-173.

Casson, N. J., Eimers, M. C., & Watmough, S. A. (2014). Sources of nitrate export during rain-on-snow events at forested catchments. *Biogeochemistry*, *120*(1-3), 23-36.

Castillo, V. M., Gomez-Plaza, A., & Martınez-Mena, M. (2003). The role of antecedent soil water content in the runoff response of semiarid catchments: a simulation approach. *Journal of Hydrology*, *284*(1-4), 114-130.

Challis, J. K., Cuscito, L. D., Joudan, S., Luong, K. H., Knapp, C. W., Hanson, M. L., & Wong, C. S. (2018). Inputs, source apportionment, and transboundary transport of pesticides and other polar organic contaminants along the lower Red River, Manitoba, Canada. *Science of the Total Environment*, *635*, 803-816.

Chen, Z., Grasby, S. E., & Osadetz, K. G. (2004). Relation between climate variability and groundwater levels in the upper carbonate aquifer, southern Manitoba, Canada. *Journal of Hydrology*, *290*(1-2), 43-62.

Chen, X., & Hu, Q. (2004). Groundwater influences on soil moisture and surface evaporation. *Journal of Hydrology*, *297*(1-4), 285-300.

Chow, L. C., & Eanes, E. D. (2001). Solubility of calcium phosphates. *Monographs in Oral Science*, *18*, 94-111.

Christianson, L. E., & Harmel, R. D. (2015). 4R Water quality impacts: An assessment and synthesis of forty years of drainage nitrogen losses. *Journal of Environmental Quality*, *44*(6), 1852-1860.

Christianson, L. E., Harmel, R. D., Smith, D., Williams, M. R., & King, K. (2016). Assessment and synthesis of 50 years of published drainage phosphorus losses. *Journal of Environmental Quality*, *45*(5), 1467-1477.

Clarke, R., Peyton, D., Healy, M. G., Fenton, O., & Cummins, E. (2017). A quantitative microbial risk assessment model for total coliforms and E. coli in surface runoff following application of biosolids to grassland. *Environmental Pollution*, *224*, 739-750.

Coelho, B. B., Murray, R., Lapen, D., Topp, E., & Bruin, A. (2012). Phosphorus and sediment loading to surface waters from liquid swine manure application under different drainage and tillage practices. *Agricultural Water Management*, *104*, 51-61.

Cordeiro, M. R. C., & Ranjan, R. S. (2012). Corn yield response to drainage and subirrigation in the Canadian Prairies. *Transactions of the ASABE*, *55*(5), 1771-1780.

Cordeiro, M. R., Ranjan, R. S., Ferguson, I. J., & Cicek, N. (2014). Nitrate, phosphorus, and salt export through subsurface drainage from corn fields in the Canadian Prairies. *Transactions of the ASABE*, 57(1), 43-50.

Cordeiro, M. R., Wilson, H. F., Vanrobaeys, J., Pomeroy, J. W., & Fang, X. (2017). Simulating cold-region hydrology in an intensively drained agricultural watershed in Manitoba, Canada, using the Cold Regions Hydrological Model. *Hydrology and Earth System Sciences*, *21*(7), 3483-3506.

Cordeiro, M. R., & Ranjan, R. S. (2015). DRAINMOD simulation of corn yield under different tile drain spacing in the Canadian Prairies. *Transactions of the ASABE*, *58*(6), 1481-1491.

Coskun, D., Britto, D. T., Shi, W., & Kronzucker, H. J. (2017). Nitrogen transformations in modern agriculture and the role of biological nitrification inhibition. *Nature Plants*, *3*(6), 17074.

Council of Canadian Academies. (2013). Water and agriculture in Canada: Towards sustainable management of water resources. The Expert Panel on Sustainable Management of Water in the Agricultural Landscapes of Canada, Council of Canadian Academics.

Crabbé, P., Lapen, D. R., Clark, H., Sunohara, M., & Liu, Y. (2012). Economic benefits of controlled tile drainage: Watershed evaluation of beneficial management practices, South Nation river basin, Ontario. *Water Quality Research Journal*, 47(1), 30-41.

Cuadra, P. E., & Vidon, P. (2011). Storm nitrogen dynamics in tile-drain flow in the US Midwest. *Biogeochemistry*, 104(1-3), 293-308.

Deasy, C., Brazier, R. E., Heathwaite, A. L., & Hodgkinson, R. (2009). Pathways of runoff and sediment transfer in small agricultural catchments. *Hydrological Processes*, *23*(9), 1349-1358.

Demand, D., Selker, J. S., & Weiler, M. (2019). Influences of Macropores on Infiltration into Seasonally Frozen Soil. *Vadose Zone Journal*, *18*(1). doi:10.2136/vzj2018.08.0147

Dean, W. E. Jr., 1974. Determination of carbonate and organic matter in calcareous sediments and sedimentary rocks by loss on ignition: Comparison with other methods. *Journal of Sedimentary Petrology*, 44, 242–248.

Dharmakeerthi, R. S., Kumaragamage, D., Goltz, D., & Indraratne, S. P. (2019). Phosphorus Release from Unamended and Gypsum-or Biochar-Amended Soils under Simulated Snowmelt and Summer Flooding Conditions. *Journal of Environmental Quality*, 48(4), 822-830.

Dils, R. M., & Heathwaite, A. L. (1999). The controversial role of tile drainage in phosphorus export from agricultural land. *Water Science and Technology*, *39*(12), 55.

Dingman, S. L. (2015). *Physical hydrology*. 3rd edition. Waveland pres, Long Grove, III.

Dinnes, D. L., Karlen, D. L., Jaynes, D. B., Kaspar, T. C., Hatfield, J. L., Colvin, T. S., & Cambardella, C. A. (2002). Nitrogen management strategies to reduce nitrate leaching in tile-drained Midwestern soils. *Agronomy Journal*, *94*(1), 153-171.

Djodjic, F., Ulén, B., & Bergström, L. (2000). Temporal and spatial variations of phosphorus losses and drainage in a structured clay soil. *Water Research*, *34*(5), 1687-1695.

Drury, C. F., Tan, C. S., Gaynor, J. D., Oloya, T. O., & Welacky, T. W. (1996). Influence of controlled drainage-subirrigation on surface and tile drainage nitrate loss. *Journal of Environmental Quality*, *25*(2), 317-324.

Drury, C. F., Tan, C. S., Welacky, T. W., Reynolds, W. D., Zhang, T. Q., Oloya, T. O., ... & Gaynor, J. D. (2014). Reducing nitrate loss in tile drainage water with cover crops and water-table management systems. *Journal of Environmental Quality*, *43*(2), 587-598.

Dumanski, S., Pomeroy, J. W., & Westbrook, C. J. (2015). Hydrological regime changes in a Canadian Prairie basin. *Hydrological Processes*, *29*(18), 3893-3904.

Duncan, E. W., King, K. W., Williams, M. R., LaBarge, G., Pease, L. A., Smith, D. R., & Fausey, N. R. (2017). Linking soil phosphorus to dissolved phosphorus losses in the Midwest. *Agricultural & Environmental Letters*, *2*(1). doi:10.2134/ael2017.02.0004

Eastman, M., Gollamudi, A., Stämpfli, N., Madramootoo, C. A., & Sarangi, A. (2010). Comparative evaluation of phosphorus losses from subsurface and naturally drained agricultural fields in the Pike River watershed of Quebec, Canada. *Agricultural Water Management*, *97*(5), 596-604.

Environment Canada. (2017). Canadian climate normals 1981–2010 station data. http://climate.weather.gc.ca/climate_normals/results_1981_2010_e.html?stnID=3582&autofwd=1 (accessed 30th March 2018).

Edwards, W. M., Shipitalo, M. J., Owens, L. B., & Dick, W. A. (1992). Rainfall intensity affects transport of water and chemicals through macropores in no-till soil. *Soil Science Society of America Journal*, *56*(1), 52-58.

Erisman, J. W., Galloway, J. N., Seitzinger, S., Bleeker, A., Dise, N. B., Petrescu, A. R., ... & de Vries, W. (2013). Consequences of human modification of the global nitrogen cycle. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *368*(1621), 20130116.

Evrard, O., Vandaele, K., Bielders, C., & Wesemael, B. V. (2008). Seasonal evolution of runoff generation on agricultural land in the Belgian loess belt and implications for muddy flood triggering. *Earth Surface Processes and Landforms: The Journal of the British Geomorphological Research Group*, *33*(8), 1285-1301.

Fang, X., Minke, A., Pomeroy, J., Brown, T., Westbrook, C., Guo, X., & Guangul, S. (2007). A review of Canadian Prairie hydrology: Principles, modelling and response to land use and drainage change. *Center for Hydrology Report*, (2). Centre for Hydrology, Saskatoon, SK, Canada.

Ford, W. I., King, K. W., Williams, M. R., & Confesor, R. B. (2017). Modified APEX model for simulating macropore phosphorus contributions to tile drains. *Journal of Environmental Quality*, *46*(6), 1413-1423.

Fouli, Y., Cade-Menun, B. J., & Cutforth, H. W. (2013). Freeze–thaw cycles and soil water content effects on infiltration rate of three Saskatchewan soils. *Canadian Journal of Soil Science*, *93*(4), 485-496.

Frey, S. K., Topp, E., Khan, I. U., Ball, B. R., Edwards, M., Gottschall, N., ... & Lapen, D. R. (2015). Quantitative Campylobacter spp., antibiotic resistance genes, and veterinary antibiotics in surface and ground water following manure application: influence of tile drainage control. *Science of the Total Environment*, *532*, 138-153.

Fuller, K. D., Gordon, R., Grimmett, M., Fillmore, S., Madani, A., VanRoestel, J., ... & George, E. S. (2010). Seasonal and crop rotational effects of manure management on nitrate–nitrogen leaching in Nova Scotia. *Agriculture, Ecosystems & Environment*, *137*(3-4), 267-275.

Gaines, T. P., & Gaines, S. T. (1994). Soil texture effect on nitrate leaching in soil percolates. *Communications in Soil Science and Plant Analysis*, *25*(13-14), 2561-2570.

Gentry, L. E., David, M. B., Royer, T. V., Mitchell, C. A., & Starks, K. M. (2007). Phosphorus transport pathways to streams in tile-drained agricultural watersheds. *Journal of Environmental Quality*, *36*(2), 408-415.

Gottschall, N., Edwards, M., Craiovan, E., Frey, S. K., Sunohara, M., Ball, B., ... & Lapen, D. R. (2016). Amending woodchip bioreactors with water treatment plant residuals to treat nitrogen, phosphorus, and veterinary antibiotic compounds in tile drainage. *Ecological Engineering*, *95*, 852-864.

Grant, K. N., Macrae, M. L., Rezanezhad, F., & Lam, W. V. (2019a). Nutrient leaching in soil affected by fertilizer application and frozen ground. *Vadose Zone Journal*, *18*(1). doi:10.2136/vzj2018.08.0150

Grant, K. N., Macrae, M. L., & Ali, G. A. (2019b). Differences in preferential flow with antecedent moisture conditions and soil texture: Implications for subsurface P transport. *Hydrological Processes*, *33*(15), 2068-2079.

Gray, D. M., Toth, B., Zhao, L., Pomeroy, J. W., & Granger, R. J. (2001). Estimating areal snowmelt infiltration into frozen soils. *Hydrological Processes*, *15*(16), 3095-3111.

Greve, A., Andersen, M. S., & Acworth, R. I. (2010). Investigations of soil cracking and preferential flow in a weighing lysimeter filled with cracking clay soil. *Journal of Hydrology*, *393*(1-2), 105-113.

Gruber, N., & Galloway, J. N. (2008). An Earth-system perspective of the global nitrogen cycle. *Nature*, *451*(7176), 293.

Haggard, B. E., Smith, D. R., & Brye, K. R. (2007). Variations in stream water and sediment phosphorus among select Ozark catchments. *Journal of Environmental Quality*, *36*(6), 1725-1734.

Hardie, M. A., Cotching, W. E., Doyle, R. B., Holz, G., Lisson, S., & Mattern, K. (2011). Effect of antecedent soil moisture on preferential flow in a texture-contrast soil. *Journal of Hydrology*, *398*(3-4), 191-201.

Harmel, R. D., Smith, D. R., Haney, R. L., & Allen, P. M. (2019). Comparison of nutrient loss pathways: Run-off and seepage flow in Vertisols. *Hydrological Processes*, *33*(18), 2384-2393.

Haverstock, M. J., Madani, A., Baldé, H., VanderZaag, A. C., & Gordon, R. J. (2017). Performance of an Agricultural Wetland-Reservoir-Irrigation Management System. *Water, 9*(7), 472.

Hayashi, M., van der Kamp, G., & Schmidt, R. (2003). Focused infiltration of snowmelt water in partially frozen soil under small depressions. *Journal of Hydrology*, *270*(3-4), 214-229.

He, H., Dyck, M. F., Si, B. C., Zhang, T., Lv, J., & Wang, J. (2015). Soil freezing–thawing characteristics and snowmelt infiltration in Cryalfs of Alberta, Canada. *Geoderma Regional*, *5*, 198-208.

Heathwaite, A. L., & Dils, R. M. (2000). Characterising phosphorus loss in surface and subsurface hydrological pathways. *Science of the Total Environment*, *251*, 523-538.

Heppell, C. M., Worrall, F., Burt, T. P., & Williams, R. J. (2002). A classification of drainage and macropore flow in an agricultural catchment. *Hydrological Processes*, *16*(1), 27-46.

Hinsinger, P. (2001). Bioavailability of soil inorganic P in the rhizosphere as affected by root-induced chemical changes: a review. *Plant and Soil, 237*(2), 173-195.

Horton, R. E. (1933). The role of infiltration in the hydrologic cycle. *Eos, Transactions American Geophysical Union*, *14*(1), 446-460.

Hooda, P. S., Moynagh, M., Svoboda, I. F., Edwards, A. C., Anderson, H. A., & Sym, G. (1999). Phosphorus loss in drainflow from intensively managed grassland soils. *Journal of Environmental Quality*, *28*(4), 1235-1242.

Husk, B. R., Sanchez, J. S., Anderson, B. C., Whalen, J. K., & Wootton, B. C. (2018). Removal of phosphorus from agricultural subsurface drainage water with woodchip and mixed-media bioreactors. *Journal of Soil and Water Conservation, 73*(3), 265-275.

Hvorslev, M. J. (1951). Time Lag and Soil Permeability in Ground-Water Observations, Bull. No. 36, Waterways Exper. Sta. Corps of Engrs, US Army, Vicksburg, Mississippi, 1–50.

Ige, D. V., Akinremi, O. O., Flaten, D. N., Ajiboye, B., & Kashem, M. A. (2005). Phosphorus sorption capacity of alkaline Manitoba soils and its relationship to soil properties. *Canadian Journal of Soil Science*, *85*(3), 417-426.

International Commission on Irrigation and Drainage. (2017). World drained areas. International Commission on Irrigation and Drainage, New Delhi, India. http://www.icid.org/world-drained-area.pdf (accessed 30th March, 2018).

Jamieson, A., Madramootoo, C. A., & Enright, P. (2003). Phosphorus losses in surface and subsurface runoff from a snowmelt event on an agricultural field in Quebec. *Canadian Biosystems Engineering*, *45*, 1-1-1-7.

Jarihani, B., Sidle, R., Bartley, R., Roth, C., & Wilkinson, S. (2017). Characterisation of hydrological response to rainfall at multi spatio-temporal scales in savannas of semi-arid Australia. *Water*, *9*(7), 540.

Jarvie, H. P., Johnson, L. T., Sharpley, A. N., Smith, D. R., Baker, D. B., Bruulsema, T. W., & Confesor, R. (2017). Increased soluble phosphorus loads to Lake Erie: Unintended consequences of conservation practices?. *Journal of Environmental Quality*, *46*(1), 123-132.

Jarvis, N. J. (2007). A review of non-equilibrium water flow and solute transport in soil macropores: Principles, controlling factors and consequences for water quality. *European Journal of Soil Science*, *58*(3), 523-546.

Jarvis, N., Koestel, J., & Larsbo, M. (2016). Understanding preferential flow in the vadose zone: Recent advances and future prospects. *Vadose Zone Journal*, *15*(12).

Jeong, D. I., & Sushama, L. (2018). Rain-on-snow events over North America based on two Canadian regional climate models. *Climate Dynamics*, *50*(1-2), 303-316.

Jia, X., Scherer, T. F., Steele, D. D., & DeSutter, T. M. (2017). Subirrigation system performance and evaluation in the Red River Valley of the North. *Applied Engineering in Agriculture*, *33*(6), 811-818.

Johnson, A. I. (1967). Specific yield: compilation of specific yields for various materials. Geological survey water supply paper 1662-D. Washington: United States Government Printing Office.

Kahimba, F. C., Ranjan, R. S., & Mann, D. D. (2009). Modeling soil temperature, frost depth, and soil moisture redistribution in seasonally frozen agricultural soils. *Applied Engineering in Agriculture*, *25*(6), 871-882.

Khaled, I. M., Tsuyoshi, M., Kohei, N., Taku, N., & Hiromi, I. (2011). Experimental and modeling investigation of shallow water table fluctuations in relation to reverse Wieringermeer effect. *Open Journal of Soil Science*, *1*(02), 17.

Kiang, J. E., Gazoorian, C., McMillan, H., Coxon, G., Le Coz, J., Westerberg, I. K., ... & Reitan, T. (2018). A comparison of methods for streamflow uncertainty estimation. *Water Resources Research*, *54*(10), 7149-7176.

Kieta, K. A., Owens, P. N., Lobb, D. A., Vanrobaeys, J. A., & Flaten, D. N. (2018). Phosphorus dynamics in vegetated buffer strips in cold climates: A review. *Environmental Reviews*, *26*(3), 255-272.

Kim, K., Whelan, G., Purucker, S. T., Bohrmann, T. F., Cyterski, M. J., Molina, M., ... & Franklin, D. H. (2014). Rainfall–runoff model parameter estimation and uncertainty evaluation on small plots. *Hydrological processes*, *28*(20), 5220-5235.

King, K. W., Williams, M. R., LaBarge, G. A., Smith, D. R., Reutter, J. M., Duncan, E. W., & Pease, L. A. (2018). Addressing agricultural phosphorus loss in artificially drained landscapes with 4R nutrient management practices. *Journal of Soil and Water Conservation*, *73*(1), 35-47.

King, K. W., Williams, M. R., Macrae, M. L., Fausey, N. R., Frankenberger, J., Smith, D. R., ... & Brown, L. C. (2015). Phosphorus transport in agricultural subsurface drainage: A review. *Journal of Environmental Quality*, *44*(2), 467-485.

King, K. W., Williams, M. R., & Fausey, N. R. (2016). Effect of crop type and season on nutrient leaching to tile drainage under a corn–soybean rotation. *Journal of Soil and Water Conservation*, *71*(1), 56-68.

Kinley, R. D., Gordon, R. J., Stratton, G. W., Patterson, G. T., & Hoyle, J. (2007). Phosphorus losses through agricultural tile drainage in Nova Scotia, Canada. *Journal of Environmental Quality*, *36*(2), 469-477.

Kinley, R. D., Gordon, R. J., & Stratton, G. W. (2010). Soil Test Phosphorus as an Indicator of Nitrate–Nitrogen Leaching Risk in Tile Drainage Water. *Bulletin of Environmental Contamination and Toxicology*, *84*(4), 413-417.

Kladivko, E. J., Brown, L. C., & Baker, J. L. (2001). Pesticide transport to subsurface tile drains in humid regions of North America. *Critical Reviews in Environmental Science and Technology*, *31*(1), 1-62.

Klatt, J. G., Mallarino, A. P., & Allen, B. L. (2002). Relationship between soil P and P in surface runoff and subsurface drainage: An overview of ongoing research. In *North Central Extension-Industry Soil Fertility Conference* (Vol. 18, pp. 183-189).

Klaus, J., & McDonnell, J. J. (2013). Hydrograph separation using stable isotopes: Review and evaluation. *Journal of Hydrology*, *505*, 47-64.

Kleinman, P. J., Bryant, R. B., Reid, W. S., Sharpley, A. N., & Pimentel, D. (2000). Using soil phosphorus behavior to identify environmental thresholds. *Soil Science*, *165*(12), 943-950.

Kleinman, P. J., Smith, D. R., Bolster, C. H., & Easton, Z. M. (2015). Phosphorus fate, management, and modeling in artificially drained systems. *Journal of Environmental Quality*, *44*(2), 460-466.

Kleinman, P. J., Srinivasan, M. S., Dell, C. J., Schmidt, J. P., Sharpley, A. N., & Bryant, R. B. (2006). Role of rainfall intensity and hydrology in nutrient transport via surface runoff. *Journal of Environmental Quality*, *35*(4), 1248-1259.

Kleinman, P. J., Sharpley, A. N., Saporito, L. S., Buda, A. R., & Bryant, R. B. (2009). Application of manure to no-till soils: phosphorus losses by sub-surface and surface pathways. *Nutrient Cycling in Agroecosystems*, *84*(3), 215-227.

Knox, J. C. (2001). Agricultural influence on landscape sensitivity in the Upper Mississippi River Valley. *Catena*, *42*(2-4), 193-224.

Koivusalo, H., Turunen, M., Salo, H., Haahti, K., Nousiainen, R., & Warsta, L. (2017). Analysis of water balance and runoff generation in high latitude agricultural fields during mild and cold winters. *Hydrology Research*, *48*(4), 957-968.

Kokulan, V., Macrae, M. L., Ali, G. A., & Lobb, D. A. (2019a). Hydroclimatic controls on runoff activation in an artificially drained, near-level vertisolic clay landscape in a Prairie climate. *Hydrological Processes*, *33*(4), 602-615.

Kokulan, V., Macrae, M. L., Lobb, D. A., & Ali, G. A. (2019b). Contribution of overland and tile flow to runoff and nutrient losses from Vertisols in Manitoba, Canada. *Journal of Environmental Quality*, 48(4), 959-965.

Kokulan, V. (2019). Environmental and economic consequences of tile drainage systems in Canada. *Canadian Agri-food Policy Institute*. https://capi-icpa.ca/wp-content/uploads/2019/06/2019-06-14-CAPI-Vivekananthan-Kokulan-Paper-WEB.pdf

Kroeger, A. C., Madramootoo, C. A., Enright, P., & Laflamme, C. (2007). Efficiency of a small constructed wetland in southern Québec for treatment of agricultural runoff waters. In IWA specialist conference: Wastewater biosolids sustainability-Technical, managerial, and public synergy, pp. 1057-1062.

Kumaragamage, D., & Akinremi, O. O. (2018). Manure Phosphorus: Mobility in Soils and Management Strategies to Minimize Losses. *Current Pollution Reports*, *4*(2), 162-174.

Kurtzman, D., Baram, S., & Dahan, O. (2016). Soil–aquifer phenomena affecting groundwater under vertisols: a review. *Hydrology and Earth System Sciences*, *20*(1), 1-12.

Lam, W. V., Macrae, M. L., English, M. C., O'halloran, I. P., Plach, J. M., & Wang, Y. (2016a). Seasonal and event-based drivers of runoff and phosphorus export through agricultural tile drains under sandy loam soil in a cool temperate region. *Hydrological Processes*, *30*(15), 2644-2656.

Lam, W. V., Macrae, M. L., English, M. C., O'Halloran, I. P., & Wang, Y. T. (2016b). Effects of tillage practices on phosphorus transport in tile drain effluent under sandy loam agricultural soils in Ontario, Canada. *Journal of Great Lakes Research*, *42*(6), 1260-1270.

Land Resource Unit (1999). Soils and terrain. An introduction to the land resource. Rural Municipality of Grey. Information Bulletin 97-21 (Revised), Brandon Research Centre, Research Branch, Agriculture and Agri-Food Canada.

Le Moal, M., Gascuel-Odoux, C., Ménesguen, A., Souchon, Y., Étrillard, C., Levain, A., ... & Pinay, G. (2019). Eutrophication: A new wine in an old bottle? *Science of the Total Environment*, *651*, 1-11.

Levine, S. N., Lini, A., Ostrofsky, M. L., Burgess-Grant, H., Lami, A., Collyer-Gilles, E., ... & Kamman, N. (2018). The relative roles of point and nonpoint phosphorus sources in the eutrophication of Lake Champlain as recorded in sediment cores. *Journal of Great Lakes Research*, *44*(5), 1043-1056.

Li, M. H., & Chibber, P. (2008). Overland flow time of concentration on very flat terrains. *Transportation Research Record*, *2060*(1), 133-140.

Liu, J., Aronsson, H., Bergström, L., & Sharpley, A. (2012). Phosphorus leaching from loamy sand and clay loam topsoils after application of pig slurry. *SpringerPlus*, *1*(1), 53. doi:10.1186/2193-1801-1-53

Liu, K., Elliott, J. A., Lobb, D. A., Flaten, D. N., & Yarotski, J. (2013). Critical factors affecting field-scale losses of nitrogen and phosphorus in spring snowmelt runoff in the Canadian prairies. *Journal of Environmental Quality*, *42*(2), 484-496.

Liu, H., Lei, T. W., Zhao, J., Yuan, C. P., Fan, Y. T., & Qu, L. Q. (2011). Effects of rainfall intensity and antecedent soil water content on soil infiltrability under rainfall conditions using the run off-on-out method. *Journal of Hydrology*, *396*(1-2), 24-32.

Liu, Q., Li, Z., & Li, P. (2014). Effect of runoff dynamic on sediment and nitrogen losses in an agricultural watershed of the southern Shaanxi region, China. *CLEAN–Soil, Air, Water*, *42*(1), 56-63.

Liu, J., Macrae, M. L., Elliott, J. A., Baulch, H. M., Wilson, H. F., & Kleinman, P. J. (2019). Impacts of cover crops and crop residues on phosphorus losses in cold climates: A review. *Journal of Environmental Quality*, 48(4), 850-868.

Macrae, M. L., Ali, G. A., King, K. W., Plach, J. M., Pluer, W. T., Williams, M., ... & Tang, W. (2019). Evaluating Hydrologic Response in Tile-Drained Landscapes: Implications for Phosphorus Transport. *Journal of Environmental Quality*, 48(5), 1347-1355.

Macrae, M. L., English, M. C., Schiff, S. L., & Stone, M. (2007). Intra-annual variability in the contribution of tile drains to basin discharge and phosphorus export in a first-order agricultural catchment. *Agricultural Water Management*, *92*(3), 171-182.

Macrae, M. L., English, M. C., Schiff, S. L., & Stone, M. (2010). Influence of antecedent hydrologic conditions on patterns of hydrochemical export from a first-order agricultural watershed in Southern Ontario, Canada. *Journal of Hydrology*, *389*(1-2), 101-110.

McDonnell, J. J. (2013). Are all runoff processes the same? *Hydrological Processes*, *27*(26), 4103-4111.

McDowell, R. W., & Sharpley, A. N. (2001). Approximating phosphorus release from soils to surface runoff and subsurface drainage. *Journal of Environmental Quality*, *30*(2), 508-520.

Messing, I., & Wesström, I. (2006). Efficiency of old tile drain systems in soils with high clay content: Differences in the trench backfill zone versus the zone midway between trenches. *Irrigation and Drainage: The journal of the International Commission on Irrigation and Drainage*, *55*(5), 523-531.

Michaud, A. R., Poirier, S. C., & Whalen, J. K. (2019). Tile Drainage as a Hydrologic Pathway for Phosphorus Export from an Agricultural Subwatershed. *Journal of Environmental Quality*, *48*(1), 64-72.

Milburn, P., MacLeod, J. A., & Sanderson, B. (1997). Control of fall nitrate leaching from early harvested potatoes on Prince Edward Island. *Canadian Agricultural Engineering*, *39*(4), 263-272.

Morrison, J., Madramootoo, C. A., & Chikhaoui, M. (2013). Modeling the influence of tile drainage flow and tile spacing on phosphorus losses from two agricultural fields in southern Québec. *Water Quality Research Journal*, *48*(3), 279-293.

Mohammed, A. A., Kurylyk, B. L., Cey, E. E., & Hayashi, M. (2018). Snowmelt infiltration and macropore flow in frozen soils: Overview, knowledge gaps, and a conceptual framework. *Vadose Zone Journal*, *17*(1).

Musolff, A., Schmidt, C., Rode, M., Lischeid, G., Weise, S. M., & Fleckenstein, J. H. (2016). Groundwater head controls nitrate export from an agricultural lowland catchment. *Advances in Water Resources*, *96*, 95-107.

Ng, H. Y. F., Tan, C. S., Drury, C. F., & Gaynor, J. D. (2002). Controlled drainage and subirrigation influences tile nitrate loss and corn yields in a sandy loam soil in Southwestern Ontario. *Agriculture, Ecosystems & Environment*, *90*(1), 81-88.

Nimmo, J. R. (2012). Preferential flow occurs in unsaturated conditions. *Hydrological Processes*, *26*(5), 786-789.

Oquist, K. A., Strock, J. S., & Mulla, D. J. (2007). Influence of alternative and conventional farming practices on subsurface drainage and water quality. *Journal of Environmental Quality*, *36*(4), 1194-1204.

Outram, F. N., Cooper, R. J., Sünnenberg, G., Hiscock, K. M., & Lovett, A. A. (2016). Antecedent conditions, hydrological connectivity and anthropogenic inputs: Factors affecting nitrate and phosphorus transfers to agricultural headwater streams. *Science of the Total Environment*, *545*, 184-199.

Panuska, J. C., & Karthikeyan, K. G. (2010). Phosphorus and organic matter enrichment in snowmelt and rainfall–runoff from three corn management systems. *Geoderma*, *154*(3-4), 253-260.

Paerl, H. W. (2009). Controlling eutrophication along the freshwater–marine continuum: dual nutrient (N and P) reductions are essential. *Estuaries and Coasts*, *32*(4), 593-601.

Pease, L. A., Fausey, N. R., Martin, J. F., & Brown, L. C. (2018a). Weather, landscape, and management effects on nitrate and soluble phosphorus concentrations in subsurface drainage in the western Lake Erie basin. *Transactions of the ASABE*, *61*, 223-232.

Pease, L. A., King, K. W., Williams, M. R., LaBarge, G. A., Duncan, E. W., & Fausey, N. R. (2018b). Phosphorus export from artificially drained fields across the Eastern Corn Belt. *Journal of Great Lakes Research*, *44*(1), 43-53.

Phillips, P. A., Culley, J. L. B., Hore, F. R., & Patni, N. K. (1982). Dissolved inorganic nitrogen and phosphate concentrations in discharge from two agricultural catchments in eastern Ontario. *Agricultural Water Management*, *5*(1), 29-40.

Pizzeghello, D., Berti, A., Nardi, S., & Morari, F. (2011). Phosphorus forms and P-sorption properties in three alkaline soils after long-term mineral and manure applications in north-eastern Italy. *Agriculture, Ecosystems & Environment*, *141*(1-2), 58-66.

Plach, J. M., Macrae, M. L., Ali, G. A., Brunke, R. R., English, M. C., Ferguson, G., ... & Opolko, G. (2018a). Supply and transport limitations on phosphorus losses from agricultural fields in the Lower Great Lakes Region, Canada. *Journal of Environmental Quality*, *47*(1), 96-105.

Plach, J. M., Macrae, M. L., Williams, M. R., Lee, B. D., & King, K. W. (2018b). Dominant glacial landforms of the lower Great Lakes region exhibit different soil phosphorus chemistry and potential risk for phosphorus loss. *Journal of Great Lakes Research*, *44*(5), 1057-1067.

Plach, J., Pluer, W., Macrae, M., Kompanizare, M., McKague, K., Carlow, R., & Brunke, R. (2019). Agricultural Edge-of-Field Phosphorus Losses in Ontario, Canada: Importance of the Nongrowing Season in Cold Regions. *Journal of Environmental Quality*, 48(4), 813-821.

Pomeroy, J., Shook, K., Fang, X., & Brown, T. (2013). Predicting spatial patterns of inter-annual runoff variability in the Canadian Prairies. In G. Bloschl, M. Sivapalan, T. Wagener, A. Viglione, & H. Savenije (Eds.), *Runoff prediction in ungauged basins: Synthesis across* *processes, places and scales.* Cambridge: Cambridge University Press. https://doi.org/10.1017/CBO9781139235761, 14.

Qi, H., Qi, Z., Zhang, T. Q., Tan, C. S., & Sadhukhan, D. (2018). Modeling Phosphorus Losses through Surface Runoff and Subsurface Drainage Using ICECREAM. *Journal of Environmental Quality*, *47*(2), 203-211.

Rahman, M. M., Lin, Z., Jia, X., Steele, D. D., & DeSutter, T. M. (2014). Impact of subsurface drainage on streamflows in the Red River of the North basin. *Journal of Hydrology*, *511*, 474-483.

Randall, G. W., Iragavarapu, T. K., & Schmitt, M. A. (2000). Nutrient losses in subsurface drainage water from dairy manure and urea applied for corn. *Journal of Environmental Quality*, *29*(4), 1244-1252.

Randall, G. W., & Mulla, D. J. (2001). Nitrate nitrogen in surface waters as influenced by climatic conditions and agricultural practices. *Journal of Environmental Quality*, *30*(2), 337-344.

Rattan, K. J., Blukacz-Richards, E. A., Yates, A. G., Culp, J. M., & Chambers, P. A. (2019). Hydrological variability affects particulate nitrogen and phosphorus in streams of the Northern Great Plains. *Journal of Hydrology: Regional Studies*, *21*, 110-125.

Rattan, K. J., Corriveau, J. C., Brua, R. B., Culp, J. M., Yates, A. G., & Chambers, P. A. (2017). Quantifying seasonal variation in total phosphorus and nitrogen from prairie streams in the Red River Basin, Manitoba Canada. *Science of the Total Environment, 575,* 649-659.

Reid, D. K., Ball, B., & Zhang, T. Q. (2012). Accounting for the risks of phosphorus losses through tile drains in a phosphorus index. *Journal of Environmental Quality*, *41*(6), 1720-1729.

Reynolds WD. (2008). Chapter 77: Saturated hydraulic properties: Ring infiltrometer. In Soil Sampling and Methods of Analysis, Carter MR, Gregorich EG (eds). Canadian Society of Soil Science, CRC Press: Boca Raton; 1043–1056

Rittenburg, R. A., Squires, A. L., Boll, J., Brooks, E. S., Easton, Z. M., & Steenhuis, T. S. (2015). Agricultural BMP effectiveness and dominant hydrological flow paths: concepts and a review. *Journal of the American Water Resources Association*, *51*(2), 305-329.

Ritzema, H. P., Nijland, H. J., & Croon, F. W. (2006). Subsurface drainage practices: From manual installation to large-scale implementation. *Agricultural Water Management*, *86*(1-2), 60-71.

Robinson, M., & Beven, K. J. (1983). The effect of mole drainage on the hydrological response of a swelling clay soil. *Journal of Hydrology*, *64*(1-4), 205-223.

Ross, C. A., Ali, G., Bansah, S., & Laing, J. R. (2017). Evaluating the relative importance of shallow subsurface flow in a prairie landscape. *Vadose Zone Journal*, *16*(5), 1-20. https://doi.org/10.2136/vzj2016.10.0096

Salvano, E., Flaten, D. N., Rousseau, A. N., & Quilbe, R. (2009). Are current phosphorus risk indicators useful to predict the quality of surface waters in southern Manitoba, Canada? *Journal of Environmental Quality*, *38*(5), 2096-2105.

Satchithanantham, S., & Ranjan, R. S. (2015). Evaluation of DRAINMOD for potato crop under cold conditions in the Canadian Prairies. *Transactions of the ASABE*, *58*(2), 307-317.

Satchithanantham, S., Ranjan, R. S., & Shewfelt, B. (2012). Effect of water table management and irrigation on potato yield. *Transactions of the ASABE*, *55*(6), 2175-2184.

Schindler, D. W., Carpenter, S. R., Chapra, S. C., Hecky, S. R., & Orihel, D. M. (2016). Reducing Phosphorus to Curb Lake Eutrophication is a Success. *Environmental Science & Technology*, 50 (17), 8923-8929.

Schindler, D. W., Hecky, R. E., & McCullough, G. K. (2012). The rapid eutrophication of Lake Winnipeg: Greening under global change. *Journal of Great Lakes Research*, *38*, 6-13.

Shanley, J. B., Schuster, P. F., Reddy, M. M., Roth, D. A., Taylor, H. E., & Aiken, G. R. (2002). Mercury on the move during snowmelt in Vermont. *Eos, Transactions American Geophysical Union*, *83*(5), 45-48.

Sharpley, A., & Kleinman, P. (2003). Effect of rainfall simulator and plot scale on overland flow and phosphorus transport. *Journal of environmental quality*, *32*(6), 2172-2179.

Sharpley, A. N., McDowell, R. W., and Kleinman, P. J. (2001a). Phosphorus loss from land to water: integrating agricultural and environmental management. *Plant Soil*, *237*, 287-307.

Sharpley, A. N., McDowell, R. W., Weld, J. L., & Kleinman, P. J. (2001b). Assessing site vulnerability to phosphorus loss in an agricultural watershed. *Journal of Environmental Quality*, *30*(6), 2026-2036.

Shook, K., & Pomeroy, J. (2012). Changes in the hydrological character of rainfall on the Canadian prairies. *Hydrological Processes*, *26*(12), 1752-1766.

Shook, K., Pomeroy, J., & van der Kamp, G. (2015). The transformation of frequency distributions of winter precipitation to spring streamflow probabilities in cold regions; case studies from the Canadian Prairies. *Journal of Hydrology*, *521*, 395-409.

Simard, R. R., Beauchemin, S., & Haygarth, P. M. (2000). Potential for preferential pathways of phosphorus transport. *Journal of Environmental Quality*, *29*(1), 97-105.

Sims, J. T., (2009a). Soil test phosphorus: Principles and methods. In: Kovar, J.L., Pierzynski, G.M. (Eds.), Methods of Phosphorus Analysis for Soils, Sediments, Residuals, and Waters, 2nd ed. Virginia Tech University, pp. 9–20.

Sims, J. T., (2009b). A phosphorus sorption index. In: Kovar, J.L., Pierzynski, G.M. (Eds.), Methods of Phosphorus Analysis for Soils, Sediments, Residuals, and Waters, 2nd ed. Virginia Tech University, pp. 20–22.

Siteur, K., Eppinga, M. B., Karssenberg, D., Baudena, M., Bierkens, M. F., & Rietkerk, M. (2014). How will increases in rainfall intensity affect semiarid ecosystems? *Water Resources Research*, *50*(7), 5980-6001.

Skaggs, R. W., Breve, M. A., & Gilliam, J. W. (1994). Hydrologic and water quality impacts of agricultural drainage∗. *Critical Reviews in Environmental Science and Technology*, *24*(1), 1-32.

Smith, V. H. (2003). Eutrophication of freshwater and coastal marine ecosystems a global problem. *Environmental Science and Pollution Research*, *10*(2), 126-139.

Smith, D. R., Haggard, B. E., Warnemuende, E. A., & Huang, C. (2005). Sediment phosphorus dynamics for three tile fed drainage ditches in Northeast Indiana. *Agricultural Water Management*, *71*(1), 19-32.

Smith, E. L., & Kellman, L. M. (2011). Nitrate loading and isotopic signatures in subsurface agricultural drainage systems. *Journal of Environmental Quality*, *40*(4), 1257-1265.

Smith, D. R., King, K. W., Johnson, L., Francesconi, W., Richards, P., Baker, D., & Sharpley, A. N. (2015). Surface runoff and tile drainage transport of phosphorus in the midwestern United States. *Journal of Environmental Quality*, *44*(2), 495-502.

Smith, D. R., Owens, P. R., Leytem, A. B., & Warnemuende, E. A. (2007). Nutrient losses from manure and fertilizer applications as impacted by time to first runoff event. *Environmental Pollution*, *147*(1), 131-137.

Stillman, J. S., Haws, N. W., Govindaraju, R. S., & Rao, P. S. C. (2006). A semi-analytical model for transient flow to a subsurface tile drain. *Journal of Hydrology*, *317*(1-2), 49-62.

Stone, W. W., & Wilson, J. T. (2006). Preferential flow estimates to an agricultural tile drain with implications for glyphosate transport. *Journal of Environmental Quality*, *35*(5), 1825-1835.

Stoner, J. D., Lorenz, D. L., Wiche, G. J., & Goldstein, R. M. (1993). Red river of the North Basin, Minnesota, North Dakota, and South Dakota. *Journal of the American Water Resources Association*, *29*(4), 575-615.

Sunohara, M. D., Craiovan, E., Topp, E., Gottschall, N., Drury, C. F., & Lapen, D. R. (2014). Comprehensive nitrogen budgets for controlled tile drainage fields in eastern Ontario, Canada. *Journal of environmental quality*, *43*(2), 617-630.

Sunohara, M. D., Gottschall, N., Craiovan, E., Wilkes, G., Topp, E., Frey, S. K., & Lapen, D. R. (2016). Controlling tile drainage during the growing season in Eastern Canada to reduce nitrogen, phosphorus, and bacteria loading to surface water. *Agricultural Water Management*, *178*, 159-170.

Sweeney, D. W., Pierzynski, G. M., & Barnes, P. L. (2012). Nutrient losses in field-scale surface runoff from claypan soil receiving turkey litter and fertilizer. *Agriculture, Ecosystems & Environment*, *150*, 19-26.

Tan, C. S., Drury, C. F., Reynolds, W. D., Groenevelt, P. H., & Dadfar, H. (2002). Water and nitrate loss through tiles under a clay loam soil in Ontario after 42 years of consistent fertilization and crop rotation. *Agriculture, Ecosystems & Environment, 93*(1-3), 121-130.

Tan, C. S., Zhang, T. Q., Drury, C. F., Reynolds, W. D., Oloya, T., & Gaynor, J. D. (2007). Water quality and crop production improvement using a wetland-reservoir and draining/subsurface irrigation system. *Canadian Water Resources Journal*, *32*(2), 129-136.

Tan, C. S., & Zhang, T. Q. (2011). Surface runoff and sub-surface drainage phosphorus losses under regular free drainage and controlled drainage with sub-irrigation systems in southern Ontario. *Canadian Journal of Soil Science, 91*(3), 349-359.

Tiessen, K. H. D., Elliott, J. A., Yarotski, J., Lobb, D. A., Flaten, D. N., & Glozier, N. E. (2010). Conventional and conservation tillage: Influence on seasonal runoff, sediment, and nutrient losses in the Canadian prairies. *Journal of Environmental Quality*, *39*(3), 964-980.

Turtola, E., & Jaakkola, A. (1995). Loss of phosphorus by surface runoff and leaching from a heavy clay soil under barley and grass ley in Finland. *Acta Agriculturae Scandinavica B-Plant Soil Sciences*, *45*(3), 159-165.

Ulén, B., Djodjic, F., Etana, A., Johansson, G., & Lindström, J. (2011). The need for an improved risk index for phosphorus losses to water from tile-drained agricultural land. *Journal of Hydrology*, *400*(1-2), 234-243.

US Department of the Interior, Bureau of Reclamation. (1997). Water measurement manual. 3rd ed. US Dep. Interior. https://www.usbr.gov/tsc/techreferences/mands/wmm/WMM_3rd_2001.pdf (accessed 31 May 2019).

Uusitalo, R., Turtola, E., Kauppila, T., & Lilja, T. (2001). Particulate phosphorus and sediment in surface runoff and drainflow from clayey soils. *Journal of Environmental Quality*, *30*(2), 589-595.

Uusitalo, R., Turtola, E., Puustinen, M., Paasonen-Kivekäs, M., & Uusi-Kämppä, J. (2003). Contribution of particulate phosphorus to runoff phosphorus bioavailability. *Journal of Environmental Quality*, *32*(6), 2007-2016.

Vadas, P. A., Jokela, W. E., Franklin, D. H., & Endale, D. M. (2011). The effect of rain and runoff when assessing timing of manure application and dissolved phosphorus loss in Runoff. *Journal of the American Water Resources Association*, *47*(4), 877-886.

Vadas, P. A., Kleinman, P. J., Sharpley, A. N., & Turner, B. L. (2005). Relating soil phosphorus to dissolved phosphorus in runoff. *Journal of Environmental Quality*, *34*(2), 572-580.

Valero, C. S., Madramootoo, C. A., & Stämpfli, N. (2007). Water table management impacts on phosphorus loads in tile drainage. *Agricultural water management*, *89*(1-2), 71-80.

Van Esbroeck, C. J., Macrae, M. L., Brunke, R. I., & McKague, K. (2016). Annual and seasonal phosphorus export in surface runoff and tile drainage from agricultural fields with cold temperate climates. *Journal of Great Lakes Research*, *42*(6), 1271-1280.

Van Nguyen, H., & Maeda, M. (2016). Effects of pH and oxygen on phosphorus release from agricultural drainage ditch sediment in reclaimed land, Kasaoka Bay, Japan. *Journal of Water and Environment Technology*, *14*(4), 228-235.

Vidon, P., & Cuadra, P. E. (2010). Impact of precipitation characteristics on soil hydrology in tile-drained landscapes. *Hydrological Processes*, *24*(13), 1821-1833.

Vidon, P., & Cuadra, P. E. (2011). Phosphorus dynamics in tile-drain flow during storms in the US Midwest. *Agricultural Water Management*, *98*(4), 532-540.

Vidon, P., Hubbard, H., Cuadra, P. E., & Hennessy, M. (2012). Storm flow generation in artificially drained landscapes of the US Midwest: Matrix flow, macropore flow, or overland flow. *Water*, *4*, 90-111.

Wainwright, J., & Parsons, A. J. (2002). The effect of temporal variations in rainfall on scale dependency in runoff coefficients. *Water Resources Research*, *38*(12), 1271-7-10.

Wang, Z., Zhang, T. Q., Tan, C. S., Vadas, P., Qi, Z. M., & Wellen, C. (2018). Modeling phosphorus losses from soils amended with cattle manures and chemical fertilizers. *Science of The Total Environment*, *639*, 580-587.

Waswa, G. W., & Lorentz, S. A. (2015). Energy considerations in groundwater-ridging mechanism of streamflow generation. *Hydrological Processes*, *29*(23), 4932-4946.

Water Survey of Canada. (2012). Hydrometric manual: Data computations. Water survey of Canada, Weather and environmental monitoring directorate, Meteorological Service of Canada. (accessed 5th January 2020).

Watson, K. W., & Luxmoore, R. J. (1986). Estimating Macroporosity in a Forest Watershed by use of a Tension Infiltrometer 1. *Soil Science Society of America Journal*, *50*(3), 578-582.

Watson, S. B., Miller, C., Arhonditsis, G., Boyer, G. L., Carmichael, W., Charlton, M. N., ... & Matisoff, G. (2016). The re-eutrophication of Lake Erie: Harmful algal blooms and hypoxia. *Harmful algae*, *56*, 44-66.

Wazney, L., & Clark, S. P. (2016). The 2009 flood event in the Red River Basin: Causes, assessment and damages. *Canadian Water Resources Journal/Revue Canadienne des Ressources Hydriques*, *41*(1-2), 56-64.

Wilkes, G., Sunohara, M. D., Topp, E., Gottschall, N., Craiovan, E., Frey, S. K., & Lapen, D. R. (2019). Do reductions in agricultural field drainage during the growing season impact bacterial densities and loads in small tile-fed watersheds?. *Water Research*, *151*, 423-438.

Wilson, H., Elliott, J., Macrae, M., & Glenn, A. (2019). Near-surface soils as a source of phosphorus in snowmelt runoff from cropland. *Journal of Environmental Quality*, *48*(4), 921-930.

Williams, M. R., King, K. W., Ford, W., Buda, A. R., & Kennedy, C. D. (2016). Effect of tillage on macropore flow and phosphorus transport to tile drains. *Water Resources Research*, *52*(4), 2868-2882.

Williams, M. R., King, K. W., Duncan, E. W., Pease, L. A., & Penn, C. J. (2018a). Fertilizer placement and tillage effects on phosphorus concentration in leachate from fine-textured soils. *Soil and Tillage Research*, *178*, 130-138.

Williams, M. R., King, K. W., Macrae, M. L., Ford, W., Van Esbroeck, C., Brunke, R. I., ... & Schiff, S. L. (2015). Uncertainty in nutrient loads from tile-drained landscapes: Effect of sampling frequency, calculation algorithm, and compositing strategy. *Journal of Hydrology*, *530*, 306-316.

Williams, M. R., Livingston, S. J., Penn, C. J., Smith, D. R., King, K. W., & Huang, C. H. (2018b). Controls of event-based nutrient transport within nested headwater agricultural watersheds of the western Lake Erie basin. *Journal of hydrology*, *559*, 749-761.

Withers, P. J. A., & Haygarth, P. M. (2007). Agriculture, phosphorus and eutrophication: a European perspective. *Soil Use and Management*, *23*, 1-4.

Withers, P. J. A., & Jarvie, H. P. (2008). Delivery and cycling of phosphorus in rivers: a review. *Science of the Total Environment*, *400*(1-3), 379-395.

Weiler, M., McDonnell, J. J., Tromp-van Meerveld, I., & Uchida, T. (2006). Subsurface stormflow. In: Anderson, M.G. (Ed.), *Encyclopedia of Hydrological Sciences*. John Wiley & Sons, Inc.

Woodley, A. L., Drury, C. F., Reynolds, W. D., Tan, C. S., Yang, X. M., & Oloya, T. O. (2018). Long-term Cropping Effects on Partitioning of Water Flow and Nitrate Loss between Surface Runoff and Tile Drainage. *Journal of Environmental Quality*, *47*(4), 820-829.

Zehe, E., Becker, R., Bárdossy, A., & Plate, E. (2005). Uncertainty of simulated catchment runoff response in the presence of threshold processes: Role of initial soil moisture and precipitation. *Journal of Hydrology*, *315*(1-4), 183-202.

Zhang, H., Kovar, J. L., (2009). Fractionation of soil phosphorus. In: Kovar, J.L., Pierzynski, G.M. (Eds.), Methods of Phosphorus Analysis for Soils, Sediments, Residuals, and Waters, 2nd ed. Virginia Tech University, pp. 50–54.

Zhang, T. Q., Tan, C. S., Wang, Y. T., Ma, B. L., & Welacky, T. (2017). Soil phosphorus loss in tile drainage water from long-term conventional-and non-tillage soils of Ontario with and without compost addition. *Science of the Total Environment, 580*, 9-16.

Zhang, T. Q., Tan, C. S., Zheng, Z. M., & Drury, C. F. (2015). Tile drainage phosphorus loss with long-term consistent cropping systems and fertilization. *Journal of Environmental Quality*, *44*(2), 503-511.

Zhao, S. L., Gupta, S. C., Huggins, D. R., & Moncrief, J. F. (2001). Tillage and nutrient source effects on surface and subsurface water quality at corn planting. *Journal of Environmental Quality*, *30*(3), 998-1008.

Ziadat, F. M., & Taimeh, A. Y. (2013). Effect of rainfall intensity, slope, land use and antecedent soil moisture on soil erosion in an arid environment. *Land Degradation & Development*, *24*(6), 582-590.

Zumr, D., Dostál, T., & Devátý, J. (2015). Identification of prevailing storm runoff generation mechanisms in an intensively cultivated catchment. *Journal of Hydrology and Hydromechanics*, *63*(3), 246-254.

Appendix A

Environmental and economic consequences of tile drainage systems in Canada

1. Agricultural drainage in a changing climate

Drainage is a natural process that occurs in any landscapes and is a key component of water cycling. This process is crucial in cropped systems as it reduces waterlogging conditions and facilitating plant growth. However, natural drainage is ineffective in many farmlands due to soils with lower hydraulic conductivities, soil compaction and poor relief. Therefore, a substantial proportion of agricultural lands, which, in most cases are vulnerable to waterlogged conditions, rely on artificial drainage systems. Around 11 % of the world's agricultural cropland is artificially drained. In North America, 27 % of the agricultural lands in the United States and 14 % of Canadian croplands are artificially drained (International Commission on Irrigation and Drainage, 2018). The demand for agricultural drainage has increased recently to tackle uncertainties in precipitation patterns that are anticipated under a changing climate.

Agricultural drainage can either be surface or subsurface. Surface drainage is often facilitated by using naturally existing in-farm swales and depressions to reroute or store water. Surface drainage can further be enhanced by improving the conditions of near-farm ditches. In contrast, subsurface drainage is enhanced through the installation of tile drains, which are perforated plastic or clay tubes installed in the vadose zone (unsaturated soil profile). Other artificial subsurface drainage systems are mole drainage, interceptor drains and ground water pumps. In general, tile drains reduce waterlogged conditions and the occurrence of overland flow by removing excess water from the vadose zone (thus enhancing water infiltration), improve soil aeration by keeping water table at desired depth, and facilitate improved crop growth and extended cropping and grazing seasons (King et al., 2015).

1.1. Subsurface drainage as a cause for environmental issues

Even though agronomically effective, tiles can also be the cause for several environmental problems. Enhancing drainage tiles may increase the edge of field runoff leading to increased risk of downstream flooding (Rahman et al., 2014). In addition, tile drainage can function as subsurface conduits for agricultural nutrients such as nitrogen (N) and phosphorus (P), which are crucial sources for algal blooms and subsequent eutrophication in surface water bodies (King et al., 2015). The scientific literature has also reported occurrences where tiles have been the sources for pathogens and other chemicals such as pesticides, veterinary antibiotics and heavy metals (Kladivko et al., 2010). The benefits and risks of tile drainage substantially vary according to regional climate, soils, management and tile configuration (e.g., King et al., 2015; Plach et al., 2018b). Therefore, regional and field scale studies must evaluate the impact of tile drains on both agronomy and environment relative to soils, management and tile configuration.

In Canada, extensive research on tile drainage has been done in Ontario and Quebec whereas little literature is available from other provinces (Christianson et al., 2015, 2016). Currently there is no literature that exclusively looks at the impacts of tile drainage from a pan-Canadian perspective. This paper has three objectives. The first part of this study reviews the impact of tile drainage on edge of field runoff and agrochemical pollution in Canada. The second part details the best management practices that can reduce tile nutrient losses without compromising the productivity. The last part of this study identifies and outlines research gaps and their practical importance in a changing climate from policy perspectives. Outcomes of this study will be useful for Canadian farmers, researchers and policy makers in identifying and adopting tools to increase the efficiency of subsurface drainage while minimizing its negative impacts.

2. Tile drainage in Canadian agriculture

Tile drains have been adopted by Canadian farmers since the mid-19th century. Initially, clay pipes were installed in hand dug trenches. However, tile drains are now installed with tile plows assisted by advanced surveying techniques such as Real Time Kinematics Global Navigation Satellite Systems (RTK GNSS). Historically, tile drainage systems were most extensively installed in the Canadian regions of Southern Ontario and Southwestern Quebec, in intensively farmed fields. For example, around 45 % of the Southern Ontario crop lands have been tiled. In Western Canada particularly in Alberta, tile drains have been historically used to overcome the salinity issue via flushing through tile drains (Broughton and Jutras, 2013). Substantial proportions of British Colombian and Nova Scotian farmlands have also been tile-drained. Although tile drainage has not historically been used in the Canadian Prairies, an increasing frequency of multiday spring and summer storms in these regions (Shook and Pomeroy, 2012) has caused farmers in provinces such as Manitoba and Saskatchewan to install tile drains at an accelerated rate to tackle the unprecedented waterlogging conditions in their crop fields (Cordeiro and Ranjan, 2012; Kokulan et al., 2019a). Installation of tile drains effectively modifies the vadose zone physical, chemical and biological properties, thus modifying field hydrology and biogeochemical processes.

2.1. Edge of field runoff

Tile drains can modify both the volume and pathways of runoff at the edge-of-field. The removal of excess vadose zone water by tile drainage increases the effective soil hydraulic conductivity resulting in suppression of surface runoff. However, this suppression of overland flow often depends on the regional climate and soil types. For example, tile flow accounted 73 % of the total flow in a clay loam soil and 86 % of the total flow in a sandy loam soil in a two-year study conducted in Quebec (Eastman et al., 2010). In Southern Ontario, Canada, Plach et al. (2019)

reported that ~80% of annual runoff at the edge of field occurred through tile drains in a range of soil textures. In contrast, little flow travelled through tile drains in the nearly flat Southern Manitoban agricultural landscapes underlain by clay rich soils, and overland flow dominated instead (Kokulan et al., 2019b). Indeed, 72-89 % of annual runoff occurred as overland flow.

The timing of runoff through tile drains also differs across Canada. For example, larger tile outflows have been observed in Ontario throughout the non-growing season and tile drains often do not flow during the growing season (Van Esbroeck et al., 2016; Woodley et al., 2018). In contrast, tiles rarely flowed in summer and never in winter in the Canadian prairies (Kokulan et al., 2019b) because conditions were either too dry or too cold. Substantial tile flow was only observed after the thawing of soil-ice layer in the Canadian Prairies.

It is not clear if, and to what extent, tile drainage may impact the volume of runoff exiting fields as few studies actively compared tiled and non-tiled fields. There is a possibility of increased edge of field runoff due to tile drains which in turn will increase the downstream flooding; however, none of the reviewed studies have evaluated this concept. This in an area where additional research is needed.

2.2. Phosphorus losses

Despite the dominance of tile drains as a flow path in some landscapes, overland flow appears to be the major runoff pathway for edge-of-field P losses in Canadian landscapes where both overland and tile flow prevail. Overland flow was responsible for more than 90 % of annual P losses in the nearly flat Southern Manitoban landscapes with clay rich soils over a three-year period (Kokulan et al., 2019b). In a five-year study in Ontario, substantial amounts of P were lost via overland flow even though tile drainage was responsible for the majority of annual runoff (Van Esbroeck et al., 2016; Plach et al., 2019). Nevertheless, tile drains can be the crucial edge of field P pathway in landscapes with minimal overland flow (Tan and Zhang, 2011; Plach et al., 2019).

The tendency of tile drains to be significant exporters of P depends in a range of factors. Soils with higher soil test P are more likely to desorb P to runoff and drainage water (e.g. Plach et al., 2018a; Duncan et al., 2017). However, the threshold P value varies between soils depending on their P sorption capacity and P saturation. For example, clayey soils can retain more P over sandy soils due to their higher P sorption capacity. In addition, the tendency of P release by soils also depends on soil P pool in which most of the P is retained. For example, P retained in reducible form (higher oxides of Fe and Mn) may become available during anoxic conditions through reductive dissolution reactions. On the other hand, P is often stable and rarely released to runoff water when bound to calcium and magnesium. Recent work (Plach et al., 2018b) has shown that soils across Ontario retain P in different forms, which has implications for the vulnerability of those fields to lose P via tile drains. However, the existence of preferential flow pathways in soils often overrides the natural tendency of the soils to retain P in their matrix. Tile P losses through preferential flow paths are critical in clay-rich soils due to the existence of macropores (Grant et al., 2019), which can be present as either desiccation cracks or biopores. Preferential flow tile P losses have been reported in Ontario and Quebec clayey and loamy soils at times accounting for majority of tile P losses (Michaud et al., 2018; Tan and Zhang, 2011). On the contrary, Kokulan et al (2019a, b) did not find direct surface-tile connectivity in Manitoban vertisols despite their tendency to form cracks.

Farm operations such as tillage methods and nutrient application can affect P in tile runoff. In fine textured soils, conservation practices such as no-till operations have been found to increase preferential flow-attributed tile P losses by preserving the macropore network (King et al., 2015;

Jarvie et al., 2017). In such cases, practices such as minimum till are recommended to disrupt the macropore network and to reduce subsequent P losses (Zhang et al., 2017). However, the effects of no-till are not consistent across the landscape. For example, no significant difference was observed in tile P loads between annual disk till and minimum till in non-macroporous soils like sandy loams (Lam et al., 2016b) and silt loams (M. Macrae, University of Waterloo. Unpublished data). Fertilizer and manure application methods such as broadcasting can also result in higher tile P losses mainly though preferential flow, whereas the subsurface placement of fertilizer can reduce P losses (Grant et al., 2019; Qi et al., 2018).

The form of nutrient application (manure vs fertilizer) may also influence tile P losses (Macrae et al., 2007). In general, manure applications contain more P than mineral fertilizers as manure requirements are calculated based on N requirements. Furthermore, in clayey soils, increased preferential flow connectivity has been observed in manure applied plots, which may further increase the possibility of tile P losses (Ali et al., 2018). Indeed, tile P losses may be further exacerbated, especially when manure application is accompanied with zero tillage (Zhang et al., 2017). Kinley et al (2007) observed higher tile P losses from several Nova Scotian fields that received swine and poultry manure. On the contrary, Wang et al (2018) have found solid cattle manure more resistant to P losses when compared to inorganic fertilizers and liquid cattle manure. Continuous fertilization could further increase tile P losses (Zhang et al., 2015). Improved knowledge of the relative contributions of manure and inorganic P fertilizer to edge-of-field losses are needed. Moreover, research is needed on whether periodic assessments of soil test P and variable rate P application (avoid applying on P enriched areas) may considerably decrease tile P losses.

2.3. Nitrogen losses

Nitrate (NO_3^-) ions are not only sensitive to crop management practices but also have the potential to leach through the soil matrix. Therefore, tile drainage often contributes to increased N losses. For example, 40-50 % of annual nitrate losses were attributed to tile drainage in vertisols of Southern Manitoba where tiles were responsible for only 11-28% of annual flow (Kokulan et al., 2019b). Tile drainage (54-58 %) and groundwater resurgence (39-45 %) were responsible for the majority of the nitrate loads in an agricultural catchment in Quebec whereas surface runoff was only responsible for 3 % of the total nitrate loads (Michaud et al., 2018).

Like phosphorus, fertilization is one of the primary reasons for increased tile N losses. Elevated tile N concentrations were seen even with recommended N rates (Philips et al., 1982). Farms that receive organic inputs like poultry and swine manure are likely to lose more N through tile drains (Kinley et al., 2010; Smith and Kellman, 2011). Fields that receive herbicides such as glyphosate also have resulted in elevated tile N losses potentially due to increased mineralization (Fuller et al., 2010). Rainfall events following fertilizer application may also exacerbate tile N losses (Kokulan et al., 2019b). Unlike P, tile N has been shown to be lower in no till cultivation as increased volatilization and denitrification result in aerial N losses (Fuller et al., 2010).

Nitrate concentrations in tile water often exceed the recommended thresholds for drinking water (< 10mg N/ L) rotation and continuous cropping systems with corn (Bolton et al., 1970; Woodley et al., 2018) potentially due to poor crop N use efficiencies (Tan et al., 2002). These losses can further be exacerbated in cropping systems where a leguminous cover crop is being ploughed in addition to fertilization (Woodley et al., 2018). Therefore, N fertilizer rates should be adjusted when a legume crop is cultivated. Management practices like straw mulching may also reduce nitrate leaching into tile for a certain period (Milburn et al., 1997). However, long term studies are absent.

3. Ways to minimize tile nutrient losses to environment without compromising productivity

3.1. 4Rs

The 4R principle is centered around adapting fertilizer application to optimize the productivity with little environmental impact. The 4 Rs stand for right fertilizer source, right rate, right time and right place. For example, a high proportion of soluble P fraction leads to elevated P losses from raw manure application. Therefore, generating low P manure or processed manure with lower available P fractions might be a solution (Kumaragamage and Akinremi, 2018). Following P fertilizer application rates based on soil test P levels not only reduces runoff P, but also boosts economic returns. Timing of fertilization is also a major concern as fall application of manure often results in increased nutrient losses during winter runoff in Ontario, Quebec and Nova Scotia. Therefore, applying fertilizers in spring is recommended. Even if they are applied in spring, a subsequent rainfall could lead to major runoff nutrient losses. Therefore, the fertilizers have to be mixed (right place) with soil for better retention and crop growth. Subsurface banding of P fertilizers or applying as P fertilizers liquids has the potential to reduce tile P losses by limiting the contact between P and preferential flowpaths (Grant et al., 2019). However, these conditions may vary with region and management options. Therefore, formulating regional 4R strategies considering managemental options could aid in better crop production with minimal environmental impacts (King et al., 2018).

3.2. Controlled drainage

Controlled drainage (CD) is a best management practice where water table depths are regulated by an adjustable raised structure at the tile outlet. Maintaining desirable water depths through CD has agronomic and environmental advantages (Crabbe et al., 2012; Sunohara et al., 2016). However, these benefits may vary with regional climate, soils and management. For example, increased soybean and corn yields were reported under CD when compared to free drainage (FD) in Ontario (Crabbe et al., 2012; Ng et al., 2002; Sunohara et al., 2016). On the contrary, higher yields were observed in FD in Manitoban sandy loams over CD in potato cultivation potentially due to deeper tile depths (Satchithanantham et al., 2012). However, CD can significantly reduce P loads due to controlled flow (Corderio et al., 2014; Sunohara et al., 2016; Tan and Zhang, 2011). Controlled drainage also reduces P losses when combined with a wetland reservoir (Tan et al., 2007) and wood chip bioreactors with alum-based drinking water plant residues (Gottschall et., 2016). In contrast, Valero et al (2007) observed high P losses in CD, which they attributed to increased P solubility due to higher water table positions.

Controlled drainage with sub-irrigation (CDS) was found to be effective in reducing nitrate concentrations and loads when compared to FD in a variety of soil textures ranging from sandy loams to clay loams (Cordeiro et al., 2014; Drury et al., 1996; Ng et al., 2002; Sunohara et al., 2016). Further reductions in nitrates were seen when CDS was combined with another best management practice such as conservation tillage (Drury et al., 1996), cover crops (Drury et al., 2014), recycling through a wetland reservoir (Tan et al., 2007), woodchip denitrification bioreactors (Husk et al., 2018) and woodchip bioreactors with alum-based drinking water treatment plant residues (Gottschall et al., 2016). However, reduction of nitrate in CDS may increase leaching of nitrate into groundwater, lateral seepage into drainage ditches and gaseous N emissions (Sunohara et al., 2014). An increase in surface runoff nitrate loads was also observed with CDS but the combined losses (surface runoff +CDS) were still smaller when compared to FD (Drury et al., 1996). In addition, increased antibiotic concentrations were observed in CDS effluent (Frey et al., 2015) and in the ditches that received CDS outflows (Wilkes et al., 2019) potentially due to the reduction of flow and absence of dilution. However, antibiotics in the CDS water may be reduced with a woodchip bioreactor with 10% alum-based drinking water plant residues (Gottschall et al., 2016).

3.3. Recycling drainage water

Re-routing tile effluent to retention structures like in-farm retention ponds or constructed wetlands is also a viable alternative to control edge of field tile nutrient losses. This retained water can be recycled for irrigation during water demanding periods. Phosphorus and N losses were reduced and yields were boosted during dry years when CDS effluent was recycled through a wetland reservoir in Ontario (Tan et al., 2007) and Quebec (Kroeger et al., 2007). However, re-irrigating constructed wetland water to raw crops should be done with caution as another study observed increased E. coli in a retention pond during warmer days (Havestock et al., 2017). This is another area that requires further research.

3.4. Using caution with no till cultivation

Increased P losses in macroporous soils through preferential pathways are an issue especially in no till systems where the soil structural development is enhanced (Williams et al., 2018a; Zhang et al., 2017). Conversely, conventional tillage may loosen up soil particles and encourage nutrient and sediment loss through overland flow. Therefore, current research recommends minimum conservation tillage or reduced tillage where only the top soil is tilled annually or bi-annually to disturb surface-tile connectivity. If not, subsurface fertilizer placement methods such as subsurface fertilizer injection may help in reducing P losses in no tillage fields (Williams et al., 2018a).

4. Suggestions for future research and policy making

Despite wide use of tile drainage in Canadian agriculture and research being done regarding its agronomic potential and environmental impacts, there are some important gaps and questions still remain.

4.1. Need for region-specific research

Factors that influence tile flow and nutrient losses vary with regional climate, soils and management. Significant work has been done on drainage tiles from Ontario and Quebec. These works are important considering the wide adaptation of tile drains in these provinces and their locations close to the Great Lakes. However, certain findings of these studies cannot be applied to areas like Canadian Prairies where tile drainage is expanding. For example, larger tile outflows that are frequently observed in Ontario during winter months were not seen in Manitoba due to frozen soils (Kokulan et al., 2019b; Plach et al., 2019). Therefore, further research is needed especially from the regions where tile drainage is expanding to correctly assess their economic and environmental feasibility.

4.2. Need for long term water quality monitoring programs

In a changing climate with increasing weather uncertainty, short-term monitoring of hydrologic responses often fails to capture extreme events and long-term trends in hydrologic responses. Currently, long term tile drainage studies are lacking. Long term monitoring programs also aid in identifying changes in runoff composition with adaptation or modification of a particular management practice.

4.3. Simultaneous monitoring of agrochemicals in tile effluent

Even though both P and N are responsible for water quality issues, only a handful of studies have simultaneously monitored the trends of both nutrients in tile water. In soil profile, majority of P is moved through preferential flow pathways whereas nitrate-N is mainly transported through soil matrix. Therefore, strategies that are taken to reduce one nutrient in tile flow may exacerbate the loses of another nutrient. Therefore, studies that include simultaneous measurements of both N and P nutrients are recommended to evaluate the efficiency of management strategies.

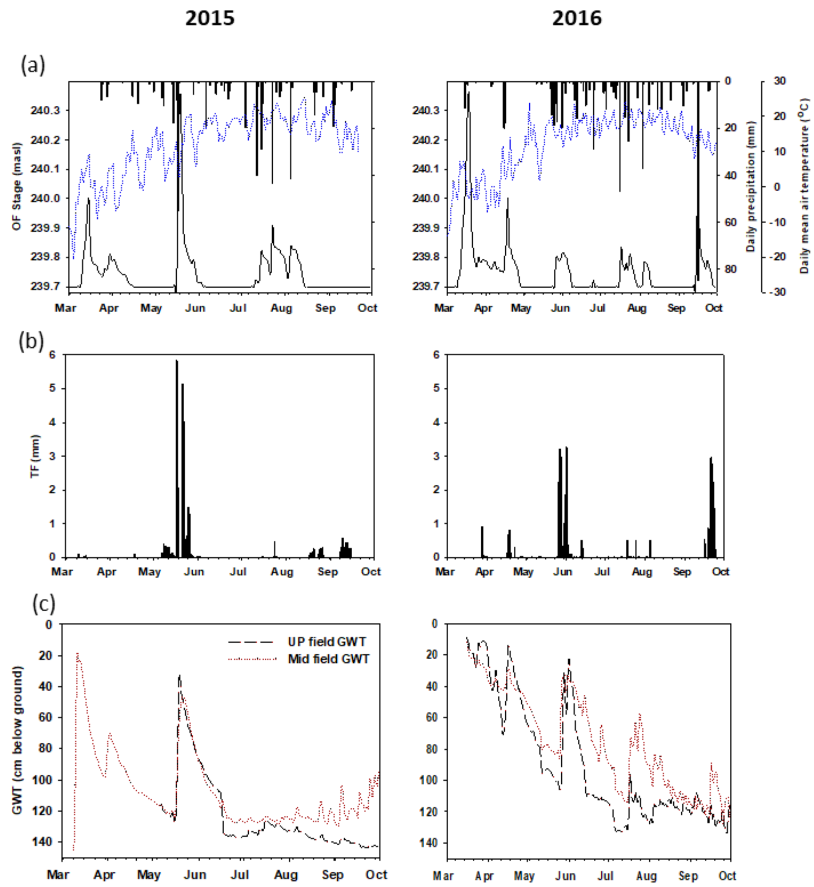
Studies focusing on other agrochemicals (e.g. pesticide residue) and antibiotics in agricultural drainage are also encouraged as there are not many Canadian studies have addressed this.

In addition, overland flow is the greater contributor for P loss and its importance as a major hydrologic pathway cannot be understated. Therefore, studies that look into tile and control drainage should also monitor the overland flow for runoff and nutrient losses.

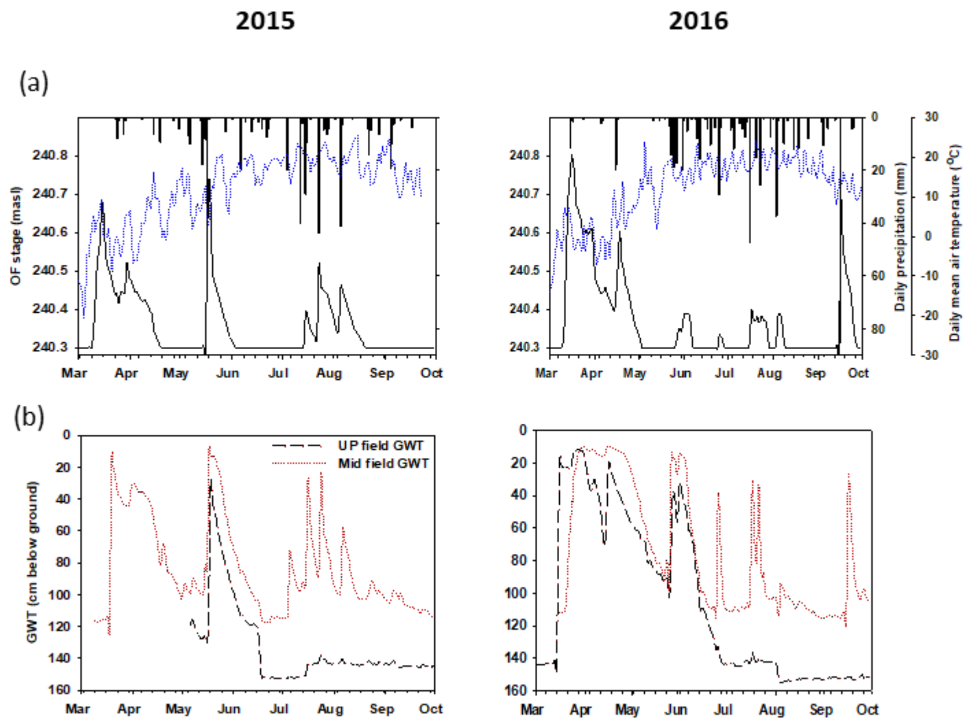
4.4. Controlled drainage and water recycling

Research done in Canada shows controlled drainage combined with another best management practice such as bioreactors or sub-irrigation could reduce edge of field nutrient loads without compromising crop yield. However, the suitability of controlled drainage in agricultural regions underlain by groundwater aquifers should be evaluated due to nitrate leaching concerns. Automated lifting stations are increasing in popularity in regions with little slope such as Southern Manitoba. Their efficacy on water management and quality should be evaluated. In addition, the efficiency of retention and re-cycling facilities such as retention ponds and constructed wetlands in reducing edge of field nutrient losses should also be evaluated.

Appendix B



Appendix B1. Daily overland flow stage (masl), daily mean air temperature (dashed line) and daily precipitation (vertical bars) (a); daily tile flow (TF) (b) and daily GWT positions (c) of the tiled field for 2015 and 2016 study years.



Appendix B2. Daily overland flow stage (masl), daily mean air temperature (dashed line) and daily precipitation (vertical bars) (a) and daily GWT positions (b) of the non-tiled field for 2015 and 2016 study years.

Appendix C

Infiltration capacities under different antecedent moisture conditions (Sigmaplot version 12.5)

a) Frozen/Wet conditions vs dry conditions

Normality Test (Shapiro-Wilk) Passed (P = 0.686)

Equal Variance Test: Passed (P = 1.000)

Treatment Name	N	Missing	Mean	Std Dev	SEM
Frozen/Wet	6	0	0.00000344	0.00000435	0.00000178
Dry	6	0	0.0000900	0.000106	0.0000435

Source of Variation	DF	SS	MS	F	P
Between Subjects	5	0.0000000307	0.00000000614		
Between Treatments	1	0.0000000225	0.0000000225	4.312	0.092
Residual	5	0.0000000261	0.00000000522		
Total	11	0.0000000793			

All Pairwise Multiple Comparison Procedures (Holm-Sidak method):

Comparisons for factor:

Comparison	Diff of Means	t	P
Dry vs. Frozen/Wet	0.0000866	2.077	0.092

Appendix D

Event 2, Field A (May 18th, 2016,1200 CDT

Event 2, Field B (May 18th, 2015,1200 CDT)

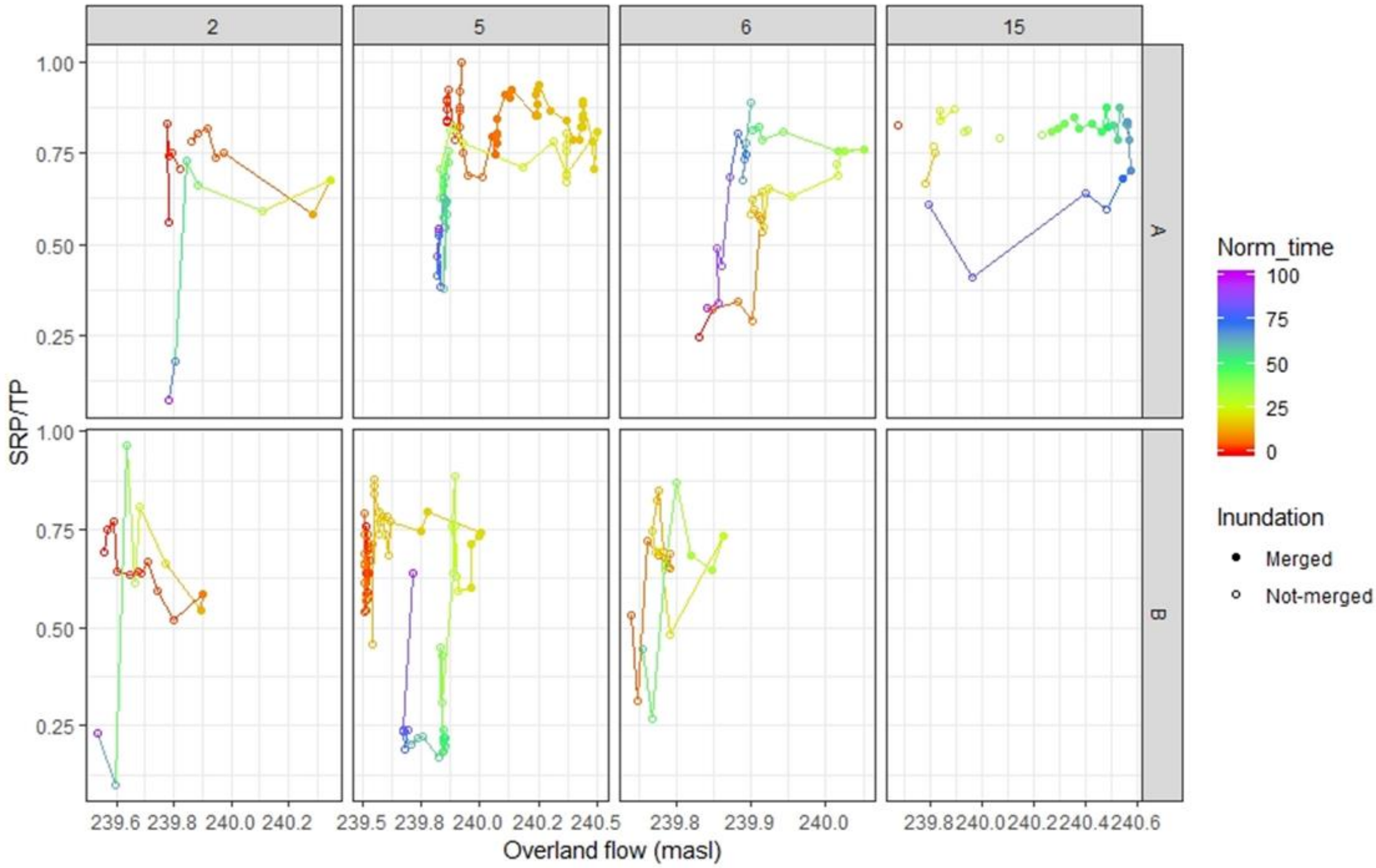
Event 5, Field A (March 17th, 2016,1300 CDT

Event 5, Field B (March 17th, 2016,1400 CDT)

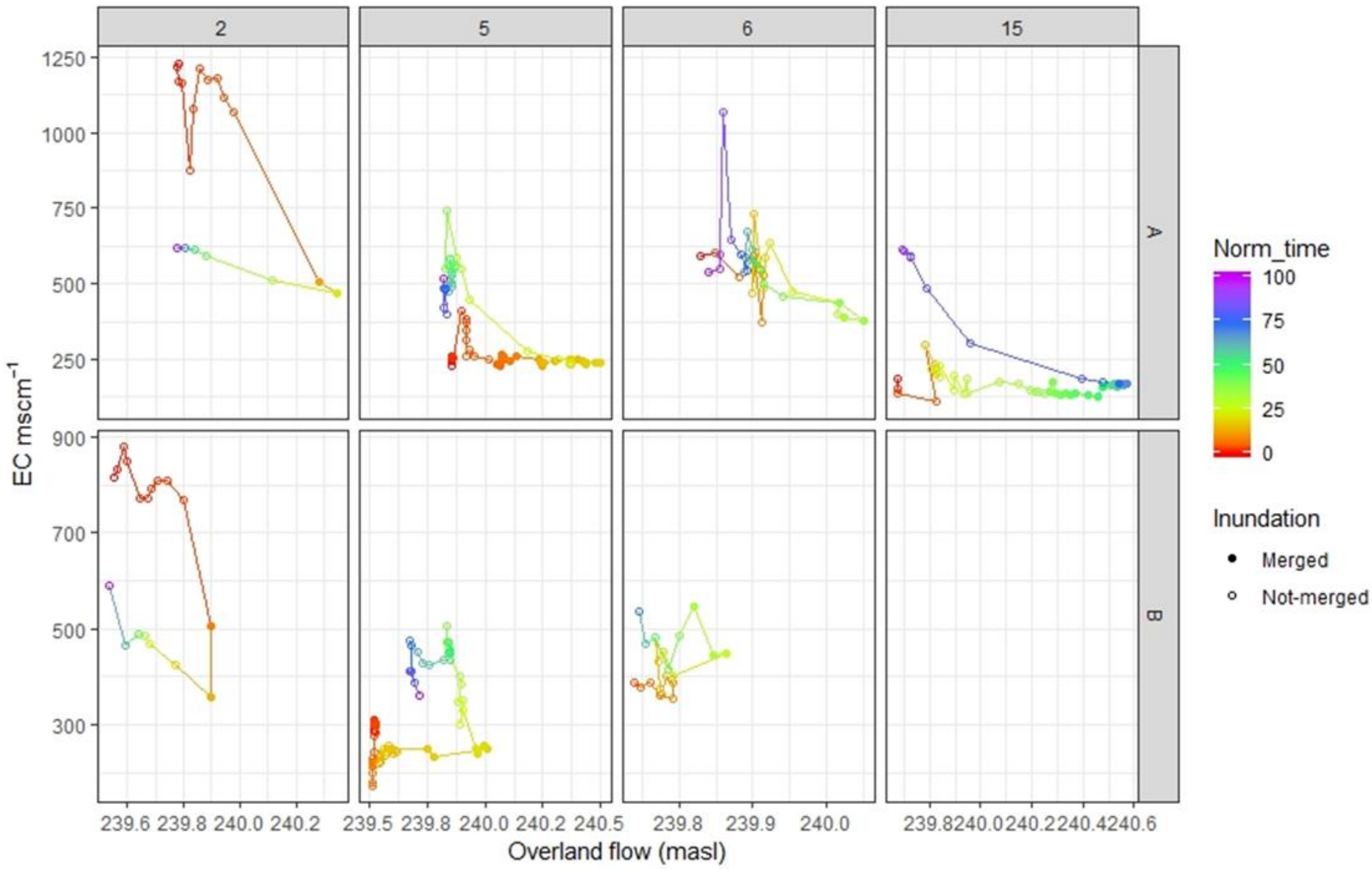
Event 6, Field A (April 17st, 2017, 1530 CDT)

Event 15, Field A (April 1st, 2017, 1330 CDT)

Appendix D1. Images showing the occurrence of flooding during the monitored events.



Appendix D2. Soluble reactive phosphorus (SRP)/ total phosphorus (TP) ratios in the overland flow for the monitored flooding events.



Appendix D3. Electrical conductivity values in the overland flow for the monitored flooding events.