THE EFFECTS OF DIVERSION TERRACES, GRASSED WATERWAYS, AND TILE DRAINAGE ON SELECTED SOIL PROPERTIES AND GREENHOUSE GAS EMISSIONS

by

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Abstract

New Brunswick is one of the largest potato producing regions in Canada, which has experienced soil erosion and a decline in soil health due to intensive tillage from potato production. To address these concerns and the uncertainties of temperature and precipitation pattern changes from climate change, a new system has been proposed for soil and water conservation in Atlantic Canada. The landscape integrated soil and water conservation (LISWC) system combines diversion terraces, grassed waterways, tile drainage, water retention structures, supplemental irrigation, conservation tillage, and soil-landscape restoration. The objective was to evaluate the effects of diversion terraces, grassed waterways, and tile drainage relative to contour tillage on selected soil properties and greenhouse gas emissions over the initial three seasons of management implementation. The study site consisted of three land management practices (LMPs): contour tillage (CT), diversion terraces and grassed waterways (DTGW), and diversion terraces, grassed waterways, and tile drainage (DTGW+TD). Aggregate stability decreased in all LMPs, and LMP had a significant effect on labile carbon (POXC). CT led to a decrease of 11.2% POXC over the study period, while DTGW increased 19.8% and DTGW+TD increased 50.6%. Mineral associated organic matter (MAOM) carbon was affected by LMP: CT carbon concentrations decreased 3.75%, while DTGW+TD rose 9.7% and DTGW rose 15.8%.

I also conducted a short-term incubation experiment in February of 2021 to quantify greenhouse gas fluxes in response to volumetric water content (20%, 27%, 35%, and 40%), nitrogen fertilizer application rates (0 and 170 kg N ha^{-1}), and temperature (10°C and 25°C).

Overall, both DTGW and DTGW+TD offer potential for improvement of soil health, relative to CT, in the second and third years of implementation, with increases in the labile C fraction POXC, a fraction significantly related to SOC, and increases in MAOM. This is the first

study in the area measuring GHG emissions from erosion control structures. This study demonstrated that DTGW+TD significantly reduced the loss of CO_2 relative to DTGW, while also not emitting significantly more than CT, in the three years after implementation. Results were similar with respect to loss of nitrogen as N_2O.

Lay Summary

Potatoes are an economically important crop in New Brunswick, but they require intensive management. The objective of my study was to assess how soil properties and greenhouse gas emissions are affected by combinations of best management practices – including diversion terraces, grassed waterways, and tile drainage – to form two new land management practices (LMPs) for potato production. No LMP was able to protect against a decline in aggregate stability during the second and third years after establishment, but active carbon increased in both new LMPs. I also found that carbon in the mineral fraction was greater in the new LMPs relative to contour tillage. This represents a more stable form of carbon and could lead to carbon sequestration. More CO_2 and N_2O was emitted from the LMP that excluded tile drainage. This research supports development of new management practices that could improve soil health and resiliency to climate change in the potato industry in this region.

Preface

This thesis represents unpublished work, which I conducted with assistance from undergraduate students, research technicians, and advisors. I was the lead investigator in the studies included in chapters 2, 3, and 4. The work hereafter is the result of a collaboration between me and the Fredericton Research and Development Centre (FRDC). Initial design of the experiment site was done by Dr. Sheng Li of the FRDC, and design of the current experiment was done by Dr. Louis-Pierre Comeau.

Data in chapters 2, 3, and 4 were collected by me and the research team at the FRDC, including Kayli McGarrigle, Kyle MacKinley, Yulia Kupriyanovich, and Dr. Louis-Pierre Comeau. Additional assistance with lab work was provided by Allysia Murphy from the University of New Brunswick and from Camilla Menhardt and Paula Porto from the University of British Columbia.

Dr. Louis-Pierre Comeau was the supervisory author on this project and was closely involved on all aspects of the study and thesis. The study was completed in collaboration with Dr. Maja Kržić and Dr. Bianca Eskelson, who aided in the design of the experiment in Chapter 4 and interpretation of all chapter results. All three were instrumental in providing thesis edits.

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List of Abbreviations and Symbols

Ap – An A horizon that has been disturbed by human activity
BMP – Best management practice
C – Carbon
CH_4 – Methane
CO_2 – Carbon dioxide
CO_2 eq – Carbon dioxide equivalent
DOC – Dissolved organic carbon
DTGW – Diversion terrace and grassed waterway
fPOM – Free light particulate organic matter
FRDC – Fredericton Research and Development Centre
GHG – Greenhouse gases
GWC – Gravimetric water content
LMP – Land management practice
MAOM – Mineral associated organic matter
MWC – Maximum water capacity
NaN – Not a number
NB – New Brunswick
NH_4^+ – Ammonium
NO_3^- – Nitrate
N_2O – Nitrous oxide
oPOM – Occluded particulate organic matter
PPM – Parts per million
POM – Particulate organic matter
POX-C – Permanganate oxidizable carbon
SAS – Soil aggregate stability
SCE – Soil CO_2 efflux
SOC – Soil organic carbon
SOM – Soil organic matter
SWC – Soil water content
TD – Tile drainage
VWC – Volumetric water content
WFPS – Water-filled pore space

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I would like to sincerely thank the members of my graduate committee for their support and guidance during my degree. Dr. Maja Kržić was my academic advisor and, along with Dr. Sandra Brown, was responsible for inspiring me during undergraduate studies to pursue soil science further. Dr. Louis Pierre Comeau was my research supervisor in Fredericton. Unfortunately, due to the Covid-19 pandemic we did not get to see each other much in person. But the times we had sampling and potato harvesting were fun. Thanks for giving me the opportunity to work at the FRDC and explore New Brunswick for two years. Finally, thank you to Dr. Bianca Eskelson for your help with stats and making the stats course manageable and fun.

Perhaps the biggest thanks go out to the two field technicians at the FRDC who were immensely helpful and supportive in my summers in Fredericton. Thank you, Kyle MacKinley. Thank you, Yulia Kupriyanovich. Every grad student should have a Kyle and a Yulia!

Thanks go out to Raelani Kesler, who started the same program in 2019 with me. It was a pleasure to travel this road with you. The bike rides, the runs, and the potluck dinners were all a cherry on top. We did it!

Special thanks are owed to my parents, who have supported me throughout my years of education. Once farmers, always farmers. Love you ♥

I would like to acknowledge that my research in Fredericton, New Brunswick took place on the traditional territories of the Wolastoqiyik, Mi'kmaq, and Peskotomuhkati peoples.

Dedication

This thesis is dedicated to my late grandmother, Ina Brownell, who generously offered her financial support when I first decided to go to university. Thanks, Gram!

Chapter 1: General Introduction

1.1 Background

Soil and water are fundamental natural resources for potato (*Solanum tuberosum* L.) production in Atlantic Canada, where Prince Edward Island and New Brunswick alone accounted for about one third of all Canadian land seeded with potato in 2019 (Agriculture and Agri-Food Canada, 2020). Potatoes are the main crop grown in the province of New Brunswick. There were over 21,000 hectares of land in the province seeded with potato in 2020, representing 166 million dollars in farm cash receipts (Statistics Canada, 2021).

Potato production requires intensive management, causing high soil disturbance from numerous tillage operations like disking, harrowing, planting, hilling, and harvesting (Wilson et al., 2018). It also requires numerous spray applications of fungicide and insecticide to battle blight and Colorado potato beetle, respectively, as well as a topkill application two weeks before harvest (Govaere et al., 2019). Furthermore, the amount of crop residue left after harvest is low (Zebarth et al., 2022). This means there is little to protect the soil surface from erosive wind and water forces, as well as little in the way of organic matter to be incorporated back into the soil. Potatoes are also a high nutrient uptake crop, and, because of all these factors, are known as a soil-depleting crop (Government of New Brunswick, n.d. -a).

It is expected that climate change will bring increases in precipitation, weather variability, and occurrence of extreme weather events to Atlantic Canada (Stocker et al., 2013; Hoegh-Guldberg et al., 2019). Potato fields in this region are subject to heightened soil erosion from water runoff and further intensified by tillage (Chow et al., 1990; 2000). This, in turn, leads not only to loss of fertile soil, but groundwater pollution by leached chemicals, pathogens, and nutrients (Li, 2021). To effectively address these challenges and to conserve soil and water resources in this region, there is a need to develop and refine best management practices (BMPs).

1.2 Soil Quality Concerns in New Brunswick

1.2.1 Soil organic carbon

New Brunswick is entirely situated within the Appalachian Mountain range – a chain of old, eroded mountains that has given rise to river valleys and low, gently rolling hills. Much of the potato production in the province happens to the northwest, in an area known as the 'potato belt', close to the border with Maine. Soils are generally characterized by poor structure and high bulk density, which is even more prominent at lower depths (Chow, 1999). The slopes in this area generally range from 2% to 9%, but in rare cases can be above 15% (Stuart, 2017). This makes it a topographically complex landscape for farming – particularly for intensive crops such as potatoes that require constant field activity.

Soil health and soil loss have been of great concern in the region for decades; soil organic matter (SOM) content in intensive potato production fields in New Brunswick has been declining since the 1960s (Eilers et al., 2010; Rees et al., 2014). Over the past 40 years or more, economic pressures in North America have resulted in increased crop specialization and greater potato production intensity (Stark & Porter, 2005). The soil disturbances that accompany potato production often break down soil aggregates, oxidize organic matter, and increase erosion (Lal et al., 1994; Unger, 1992). The result is a decline in SOM and soil organic carbon (SOC) – often-used indicators of soil health.

Over the years, crop rotations have gotten shorter, and the frequency of potato production has increased; a typical rotation is potato-potato-grain or potato-grain (Rees et al., 1999). These rotations do not provide adequate organic carbon (C) inputs to maintain or improve SOC levels

(Zielke & Christenson, 1986). For example, between 1989 and 1999, Rees et al. (2007) reported an 8% reduction from a long-term potato benchmark site under intensive up-and-down slope cultivation in northwestern New Brunswick – near Grand Falls. In addition, Rees et al. (2008) also reported a 4% reduction in SOC in a paired benchmark site under intensive potato production in a variable grade diversion and grassed waterway system in the same region. In 1984, Wang et al. reported an average of just 17.1 g SOC kg $soil^{-1}$ in loam-textured plow layers of intensively cropped potato belt soils. Prior to the intensification of production, Langmaid et al. (1976) found 26.3 to 47.8 g C kg $soil^{-1}$ in the Ap horizons of similar potato belt soils. An SOC concentration of 20.0 to 29.0 g kg^{-1} has previously been reported as the ideal range for Atlantic Canadian potato production (Hinds, 1989). Organic matter contributes significantly to the soil's ability to store and exchange nutrients. With a decline in SOM, the likelihood of lowered nutrient retention is to be expected (Stark & Porter, 2005).

1.2.2 Erosion

1.2.2.1 Soil Water Erosion

Soil erosion, a process which removes topsoil and deposits it elsewhere, is also a major concern for Atlantic Canada. Because of high levels of precipitation, potato crops are predominantly rain-fed in Atlantic Canada (Li, 2021). The area receives high-intensity rainfall events during the summer months and freeze-thaw cycles during winter months. Earlier research has identified water erosion as a serious problem (Chow et al., 1999; Edwards et al., 1998; Saini & Grant, 1980; Wang et al., 1984). Coote et al. (1981) reported that soil water erosion in northwestern New Brunswick was among the most severe seen in all of Canada. The risk of water runoff and runoff-induced soil erosion are high when row crops are located on hilly landscapes with shallow glacial till soils and in areas with high levels of precipitation. Annual soil loss from runoff erosion in the mid-1990s has exceeded 24 Mg ha^{-1} in up-and-down slope cultivated study plots in drainage basins of the Upper Saint John River Valley (Chow et al., 2010). The degree of erosion will be dependent on water infiltration rate, water holding capacity, structural stability, and intensity and duration of precipitation (Government of New Brunswick, n.d. -b). Because New Brunswick is close to large bodies of water such as the Bay of Fundy and the Atlantic Ocean, erosion can lead to permanent loss of soil, whereas in other regions, like the Prairies, soil erosion leads to soil redistribution.

1.2.2.2 Tillage Erosion

In the last two decades, much research has been done on the effects of tillage erosion. Studies have linked tillage operations to soil erosion and demonstrated that they are significant (Govers et al., 1994; Lindstrom et al., 1990; Lobb et al., 1995). In some studies, tillage erosion has been the dominant form of erosion, exceeding effects of soil redistribution by water (Govers et al., 1996; Quine et al., 1997; Van Oost et al., 2005). In a study at a soil quality monitoring benchmark site, set up by Agriculture and Agri-Food Canada in the New Brunswick potato belt, Tiessen et al. (2009) reported that the "directional movement of soil by tillage operations had been the dominant soil redistribution process" from 1990 to 2005. They estimated the annual average soil loss to be 13.6 Mg ha^{-1} yr^{-1}, with some field areas exceeding 25 Mg ha^{-1} yr^{-1}. It is estimated that over half of the cropland in New Brunswick and 75% of the cropland across Atlantic Canada is at risk of unsustainable levels of soil loss (>6 Mg ha^{-1} yr^{-1}) by tillage erosion (Government of Canada, n.d. -c).

The ability to retain soil is key to also maintaining the productivity and fertility of soil, as erosion can lead to loss of nutrients and organic matter as well as declines in crop yield (Ochuodho et al., 2013). Bakker et al. (2004; 2005) found that on average, a 4% loss in crop productivity occurred for each 10 cm of soil lost in mechanized agricultural systems. Design and type of plowing instrument, depth and speed of tillage, number of tillage passes, and tillage direction and pattern can all influence the degree and severity of erosion. Researchers in New Brunswick's potato belt have found that a combination of diversion terraces and grassed waterways can help address the control of erosion in the Atlantic region (Chow, 1999).

1.3 Diversion Terraces and Grassed Waterways

Diversion terraces are an engineering means to reducing runoff and soil loss (Agriculture and Agri-Food Canada, 2012; Li, 2021). They are meant to break up long, sloping fields into shorter field segments, thus reducing the kinetic energy of runoff water and, in turn, its erosive power. In the context of Atlantic Canada, the term 'terrace' typically involves a ~1 m high, grassed berm that runs across a field, following the natural contours of that field. A series of berms will be spaced at appropriate intervals (minimum 36 m) and run parallel to each other despite any natural hills or depressions within the field. Diversion terraces (DT) and grassed waterways (GW) are generally used in conjunction with each other. Grassed waterways are usually constructed in natural depressions where water would normally flow or collect. Grassed waterways are built on the upslope side of the berm. The waterways – with a reduced grade – would intentionally be seeded with grasses to slow down surface water runoff, allow increased water infiltration, filter out eroded soil particles, and reduce erosion (Huffman et al., 2013; Government of New Brunswick, n.d. -c). Water is channeled safely out of the field, usually via another grassed waterway running downslope to the base of the field; when the speed of surface runoff is reduced, more water can infiltrate the soil, thereby increasing surface water retention via storage of precipitation and snow melt water (Agriculture and Agri-Food Canada, 2012). It can also increase water availability due to the storage of water in the frontal channel of the terrace, conserving water for plant growth (Baryla & Pierzgalski, 2008). Research has shown that DTs have been able to reduce runoff and soil loss by 87% and 95%, respectively (Chow et al., 2010).

These BMPs have become more prevalent in recent decades. In the 1990s there was an incentive under the Land Management and Conservation Program to help fund 66% of the costs of implementing diversion terraces and grassed waterways (Rolfe, 1993). The Black Brook Watershed, northeast of the city of Grand Falls, New Brunswick encompasses 1,450 hectares of land. Roughly two thirds of that land is agricultural, and more than 50% of the agricultural land (~460 ha) is now farmed using diversion terraces and grassed waterways (Yang et al., 2009). Statistics for the rest of the province are not as easy to find, but, anecdotally, terraces can be seen all along the Trans-Canada Highway driving between Fredericton and Grand Falls.

Costs to introduce a conservation practice are of considerable importance for farmers. The typical DT takes up 10% of the field being protected, while the typical grassed waterway takes up 4% of the field it protects (Agriculture and Agri-Food Canada, 2012). While terraces will guard against excessive soil loss, it will mean a considerable amount of land being taken out of production. Expected increases in yield may not offset the costs of construction and maintenance. In 2008, annual maintenance costs were pegged at $84 and $70 CAD per hectare for DT and GW, respectively. Short term analysis of yield determined no consistent improvements on yield that would offset maintenance costs.

While the Chow et al. (2010) study proved terracing can be extremely effective in preventing soil loss, terraces are not without shortcomings. In the same study, for instance, the researchers found SOC decreased a steady 0.36% each year over a 10-year period – though still better than the 0.7% drop in the same period from paired up-and-down slope cultivated sites where potato rows were oriented perpendicular to the slope.

Because of the increased water infiltration, there are concerns of fields being too wet in years with high precipitation (Chow et al., 2010). This raises issues with trafficability of the field, which could delay planting and tillage operations or potentially affect yield (Madramootoo et al., 2007). There is also the concern that terraces could induce the movement of chemicals, nutrients, and sediments into surface or groundwater systems, as a greater proportion of typical overland flow would be directed downwards into the soil profile (Li, 2021; Yang et al., 2009).

1.4 Tile Drainage

Tile drains (TD) are an engineering means for removing the excess water after high precipitation events and for managing high water tables (Huffman et al., 2013). Wet field conditions can lead to reduced field trafficability, delayed tillage operations, and, occasionally, yield losses (Madramootoo et al., 2007). In New Brunswick, soils tend to be shallow and are underlain frequently with compact subsoil or low-permeability parent materials (Burton et al., 2012). This can lead to slow subsurface water drainage and waterlogged soil. Though tile drainage is usually installed in low-lying areas with high water tables, they are increasingly being installed in areas that have poor internal drainage (Li, 2021). Adoption in New Brunswick has been rising since the 1970s (Milburn & Gartley, 1988; Ritter et al., 1995). There are many benefits that arise from TD, including increased soil temperature, water infiltration, field trafficability, and crop yield. It can also reduce soil erosion, waterlogging, soil compaction, and phosphorus and potassium losses (Fraser & Fleming, 2001).

The main drawback from TD arises from its role in transporting substances into the soil subsurface. Tile drainage accelerates and provides a shortcut for water flowing from fields to surface water systems (Li, 2021). The result is a higher risk of contaminants ending up in surface water including, but not limited to, nitrate (NO_3^-), dissolved organic carbon, dissolved nitrous oxide (N_2O), pesticides, and soil sediment (Burton et al., 2012; Milburn et al., 1990).

1.5 Contour Tillage

Contour tillage is one example of a conservation tillage practice that is intended to reduce negative impacts of tillage on soil (Li, 2021). It is an easily implemented practice that orients the ridges and furrows associated with potato production along the natural contours of the field, generally perpendicular to the slope. Conventional tillage for New Brunswick potatoes involves orienting rows up-and-down slope. While this is generally easier and faster to manage with tractors, it increases surface water flow in channels and enhances water erosion. Contour tillage creates barriers that reduce this surface water runoff and water erosion. If a field has drainage problems, this can, however, be an issue.

1.6 Landscape Integrated Soil and Water Conservation (LISWC)

The Landscape Integrated Soil and Water Conservation (LISWC) system was designed in 2017 by Dr. Sheng Li of the FRDC to help conserve soil and water resources on sloping landscapes in Atlantic Canada. The LISWC system is a holistic land management practice (LMP) that includes several types of best management practices (BMPs) (Li, 2021). Individual BMPs included in the LISWC system are DT, GW, TD, water retention structures, supplemental irrigation, conservation tillage practices, and soil-landscape restoration. Each individual BMP has a particular targeted effect and is usually studied on its own. While each BMP may be successful in its own regard, none alone is sufficient to address all soil and water conservation issues facing potato farming. The LISWC strategy is to offset the shortcomings of one BMP by integrating several of them into a single LMP, with the idea that drawbacks from one BMP are complemented by benefits from another BMP. For example, water runoff and erosion would be reduced by the diversion terraces and grassed waterways, in addition to the implementation of contour tillage. Saturated soil conditions would be addressed by tile drainage. Tile-drained water and runoff would be stored in the water retention structure for potential use later via supplemental irrigation in drought conditions. The hope is that the LISWC system boosts soil health that has been declining for decades in New Brunswick (Eilers et al., 2010; Rees et al., 2014) and increases potato yields that have been stagnant for years (Statistics Canada, 2021). The economic and environmental outcomes of the LISWC system are, however, uncertain since this is a relatively new concept. The aim of my study was to assess how certain soil health indicators (i.e., soil aggregate stability and labile soil carbon fractions) respond to the implementation of the LISWC system. In addition, an aim was to evaluate how higher amounts of rainfall being directed through the soil in diversion terraces affect greenhouse gas (GHG) emissions.

1.7 Research Objectives and Hypotheses

Study objective 1 was to evaluate effects of two LMPs (each comprised of several individual BMPs) relative to contour tillage (CT) on soil aggregate stability across three slope positions in the second and third years after LMP implementation. The two LMPs included the following combinations of BMPs: (i) DT, GW, and CT [from now on labeled DTGW] and (ii) DT, GW, CT, and TD [from now on labeled DTGW+TD].

> **Hypothesis 1**: Soil aggregate stability will decline in DTGW and DTGW+TD, relative to CT due to the increased soil moisture. The effects will be more severe in DTGW, since there is no tile drainage to remove excess water after a heavy rainfall.
>
> ***Objective 1 is addressed in Chapter 2 of this thesis.***

Study objective 2 was to evaluate effects of two LMPs (each comprised of several individual BMPs), relative to CT on total carbon and labile soil organic matter fractions [permanganate-oxidizable C (POXC) and density fractionation of physically un-complexed light fraction organic matter] across three slope positions in the second and third years after LMP implementation. The two LMPs included the following combinations of BMPs: (i) DT, GW, and CT [from now on labeled DTGW] and (ii) DT, GW, CT, and TD [from now on labeled DTGW+TD].

Hypothesis 2: Total carbon will not change in DTGW and DTGW+TD relative to CT over the course of the study. Total carbon changes slowly and therefore will not change over two seasons.

Hypothesis 3: POXC fractions will be lower in DTGW and DTGW+TD relative to CT where soil water content will be higher, thus leading to enhanced decomposition rates.

Hypothesis 4: Free light particulate organic matter (fPOM) will be lower in DTGW and DTGW+TD, relative to CT, as higher amounts of soil water in CT will lead to more rapid decomposition rates.

Hypothesis 5: Mineral-associated organic matter (MAOM) will be adversely affected by DTGW and DTGW+TD, relative to CT. DTGW and DTGW+TD divert more water down into the soil profile. Since MAOM is made up of single molecules and microscopic fragments of OM, more of this OM will be leached out of the soil profile.

Objective 2 is addressed in Chapter 2 of this thesis.

Study objective 3 was to evaluate the effects of DTGW and DTGW+TD, relative to CT, on GHG emissions and nitrate supply rates across three slope positions from the soil during the summers of the initial three seasons after LMP establishment.

Hypothesis 6: GHG emissions will be higher in DTGW and DTGW+TD than in CT, due to the increased soil moisture from diversion terraces.

Hypothesis 7: Soil nitrate will be higher in DTGW and DTGW+TD than in CT, since nitrate leaching declines with declining soil water content.

Objective 3 is addressed in Chapter 3 of this thesis.

Study objective 4 was to quantify the GHG fluxes in response to volumetric water content (20%, 27%, 35%, and 40% VWC), nitrogen fertilizer application rates (0 and 170 kg N ha^{-1}), and temperature (10 and 25°C) with a 6-day soil incubation experiment.

Hypothesis 8: CO_2 fluxes will decline at 35% VWC as this equates to 69% water-filled pore space (WFPS).

Hypothesis 9: N_2O fluxes will be highest with 35 and 40% VWC and 25°C temperature. These VWC values equate to 69% and 79% WFPS. Nitrous oxide is generally produced in soils with WFPS > 50%.

Hypothesis 10: CH_4 will begin to act as a source (i.e., positive fluxes) at a VWC of 40%, due to anaerobic conditions caused by displacement of air, by water, in pore spaces.

Objective 4 is addressed in Chapter 4 of this thesis.

Chapter 2: Short-term Response of Soil Aggregate Stability and Labile Carbon to Contour Tillage, Diversion Terrace, Grassed Waterway, and Tile Drainage Implementation

2.1 Introduction

Soil degradation has been of great concern for New Brunswick's potato farmers for several decades. With over 21,000 hectares under potato production, New Brunswick farmers are contributing 40% of Canada's potatoes (Statistics Canada, 2021). Potato production requires frequent tillage operations that inevitably break down aggregates and enhance organic matter decomposition (Grandy et al., 2002), in turn leading to enhanced erosion susceptibility (Le Bissonnais, 1988). Several studies have shown that soil organic matter (SOM) content in intensive potato production fields in New Brunswick has been declining since the 1960s (Eilers et al., 2010; Rees et al., 2014). As SOM levels decline, soil functioning is adversely affected, leading to reduction of biological activity and further degradation of soil structure (Haynes & Tregurtha, 1999; Gagnon et al., 2001).

Negative effects of intensive potato production are particularly noticeable in regions with shallow soils and hilly landscapes, as is the case in northwest New Brunswick (Langmaid et al., 1980; Saini & Grant, 1980; Coote et al., 1981; Wang et al., 1984; Edwards et al., 1998; Chow et al., 1999). Hence, many potato farmers in New Brunswick have begun using diversion terraces (DT) and grassed waterways (GW) as means to combat those negative effects associated with this intensive crop production system. While DT and GW have been extremely effective at reducing runoff and soil loss during a decade-long study in northwest New Brunswick outside the city of Grand Falls, these structures were not able to stop the decline of soil carbon (Rees et al., 2008; Chow et al., 2010). This emphasized the need to combine several management practices to address all the negative issues associated with the intensive potato production.

In 2017, the Landscape Integrated Soil and Water Conservation (LISWC) was proposed by Dr. Sheng Li of the Fredericton Research and Development Centre (FRDC) as a new, holistic system designed to conserve soil and water on sloping landscapes by integrating multiple best management practices (BMPs) and studying their interactions (Li, 2021). Individual BMPs included in the LISWC system are diversion terraces (DT), grassed waterways (GW), tile drainage (TD), water retention structures, supplemental irrigation, contour tillage (CT), and soil-landscape restoration. At present, only one study has been completed at the LISWC site. It focused on the removal of excess water from the tile-drained LISWC fields relative to undrained fields that otherwise had the same combination of BMPs (Dobson, 2022). This study found the tile-drained LISWC fields improved field-scale drying, relative to non-tile drained fields, within the first 48 hours after heavy rainfall events. This could have implications on soil quality indicators that are affected by changes in soil moisture.

No studies in the area have investigated the effects of these combined BMPs on aggregate stability, which is key in stabilizing soil and organic matter, regulating air and water infiltration, and protecting against erosion (Carter, 2002). Because soil aggregate stability generally correlates negatively with soil water content (Caron et al., 1992; Rasiah et al., 1992), the changes in hydrology when TD is added to DT and GW may affect aggregates and the organic matter occluded within. Generally, tile drainage may increase aggregate stability, especially of the largest aggregates (Abid & Lal, 2009; Kumar et al., 2014), but there have been no studies focused on the effects of combined BMPs on aggregate stability and SOM fractions. Typically, DT and GW studies in Atlantic Canada focused on soil hydrology or potato yield, with SOC as the main indicator of soil health (Chow et al., 1990; Chow et al., 1999; Rees et al., 2008). None have investigated labile carbon (C) fractions, which are sensitive indicators of changes in total SOM (Carter, 2002). Permanganate oxidizable carbon (POXC) and particulate organic carbon (POC) are key labile C fractions that reflect processes such as soil aggregation and soil C accrual (Wardle, 1992; Six et al., 1998; Wander, 2004). A study in Missouri investigated the combination of terraces and underground tile outlets in cover and no-cover cropping systems on total organic C (TOC), labile C (POXC) and water stable aggregates – finding that two years after construction of the terraces, labile C, TOC, and water stable aggregates all declined (Adler et al., 2020).

The objective of this study was to evaluate effects of two land management practices (LMPs) (each comprised of several individual BMPs) on SOC, aggregate stability, permanganate oxidizable C (POXC), and particulate organic C (POC), relative to CT, across three slope positions in the second and third years after LMP implementation. The two LMPs included the following combinations of BMPs: (i) DT, GW, and CT [from now on labeled DTGW] and (ii) DT, GW, CT, and TD [from now on labeled DTGW+TD].

2.2 Materials and Methods

2.2.1 Site Description

The study was conducted between May 2020 and October 2021 on a field experimental site maintained by the FRDC of Agriculture and Agri-Food Canada in New Brunswick. The study was located on a northeast-facing slope facing the Saint John River (45°55'33.5"N 66°36'55.5"W). The whole Fredericton area, including the study site location, has been subject to glaciation, and most upland soils have been developed from weathered glacial till parent materials (Zelazny, 2007). The study site is located on soils that were part of the Research

Station Association and classified as Orthic Humo-Ferric Podzols. They developed in areas with loose ablation till over compact lodgment till (Rees & Fahmy, 1984). The soils are moderately permeable sandy loam with 10 to 25% angular cobbles and gravels, averaging 55% sand, 34% silt, and 10% clay (Appendix B), with a pH of 5-6. The Fredericton area receives a mean annual rainfall of 885.9 mm and a mean annual snowfall of 214.8 cm (Government of Canada, 2019).

Construction of the LISWC system on the study site began in 2018 and finished in the same year. Prior to construction, the site was under grass for two seasons. Before those two seasons, several strips were plowed along the slope for a rainfall simulation experiment, but nothing was planted. When construction began in 2018, old clay drainage tiles (dating to the mid-20th century) were found. The tiles were removed from the site after excavation. They were no longer functional and would have had marginal impact on soil moisture and water flow. In the initial 2019 season the study site was seeded to oat (*Avena sativa* L.), but this season was not factored into this study. In 2020, the site was again seeded to oat, while in 2021, a potato (*Solanum tuberosum* L.) crop was planted.

The study consisted of three land management practices (LMPs) constructed on a slope with a gradient of 10% (Figure 2-1). Each LMP took up a total of 1400 m^2 (70 × 20 m). Each LMP represents a different combination of individual BMPs. The 1st LMP acted as a control and consisted only of contour tillage – it will be referred to from here on as CT. The only permanent BMP installed in CT was the water retention structure at the base of the slope. There were 6 plots within CT.

The 2nd LMP was subdivided into 3 segments, each with 2 plots. Diversion terraces and grassed waterways were constructed below each of the three 20 × 20 m blocks. A water retention structure was constructed below the lowest block. This LMP is referred to as DTGW. The 3rd

LMP, DTGW+TD, was a repeat of DTGW but with tile drainage installed, which is representative of the full LISWC system. Two polyvinyl chloride (PVC) tile drains, measuring 10 cm in diameter, were installed per block, 10 m apart. Edges of the LMPs were lined with an impermeable plastic barrier extending to a depth of 1 m, to isolate each LMP from potential inflows from other LMPs. Contour tillage was implemented in both DTGW and DTGW+TD.

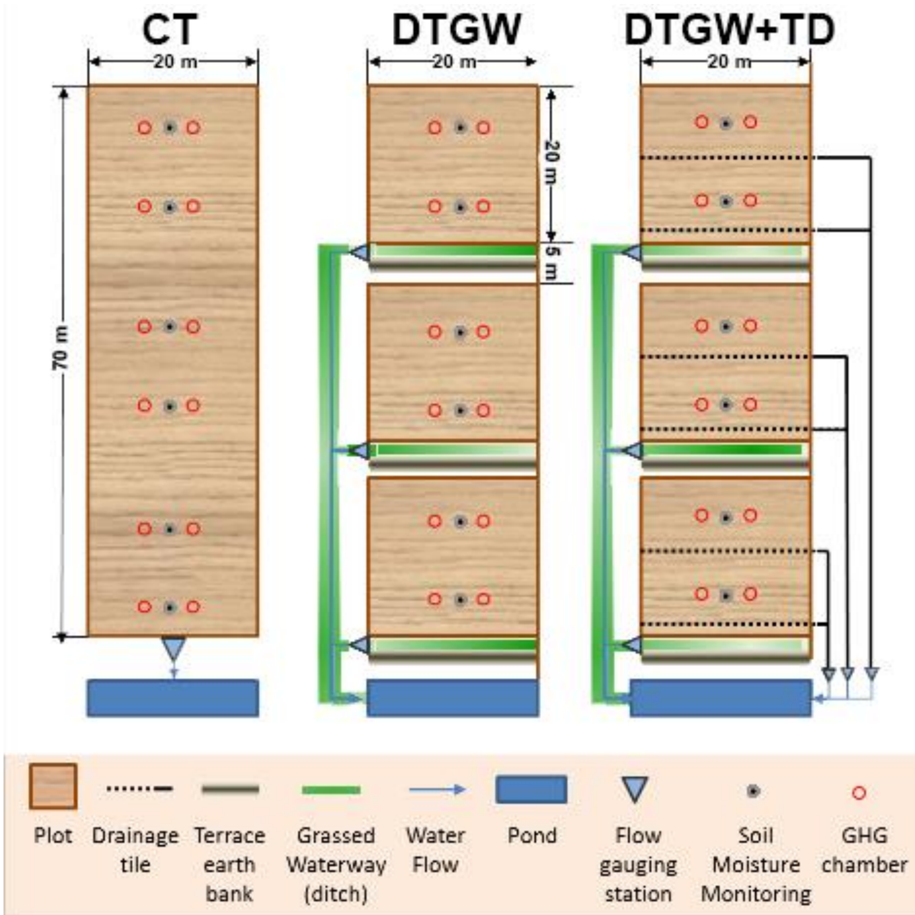

Figure 2-1 Schematic of the study site layout and the 3 land management practices (LMP) – CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage).

2.2.2 Field Seasons

2.2.2.1 2020 Field Season

Timeline of events

The experiment site was disked on June 5th (Appendix C), followed by additional disking and harrowing on June 15th, and oats were planted. On the same day as planting, a 17-17-17 urea-based fertilizer was applied at a rate of 337 kg ha^{-1} (57 kg N ha^{-1}). An herbicide was sprayed on July 6th. Oats were harvested on September 22nd to prevent them from seeding in subsequent field seasons, the straw was mowed and left on the ground. No data was collected on oat yield. A final plowing with a moldboard plow took place on October 1st.

2.2.2.2 2021 Field Season

The CT management practice consisted of 68 total potato rows. In both DTGW and DTGW+TD, each plot had a total of 16 rows, for a total of 48 rows per LMP. Oats were grown at the base and top of CT, as well as at the bottom and top of each of the 3 plots in both DTGW and DTGW+TD to prevent soil from washing out at the flumes into the water retention pond.

Timeline of events

On June 7th Russet Burbank potatoes were planted with rows spaced at 0.91 m and plants spaced at 0.38 m within rows. A 17-17-17 urea-based fertilizer was applied during planting at a rate of 1120 kg ha^{-1} (190.5 kg N ha^{-1}). This was slightly lower than the 208 kg N ha^{-1} fertilizer rate that is the typical maximum fertilization rate for potato farming in New Brunswick (Government of New Brunswick, n.d. -d). Potato rows were hilled on July 20th. Insecticide was sprayed on both July 23rd and August 11th to suppress Colorado potato beetle (*Leptinotarsa decemlineata*) activity. Fungicide (Admire) spraying was scheduled every Friday (weather permitting) but sometimes took place a day earlier or later depending on rain events. Spray dates and a full timeline of events appear in Appendix D. Potato harvest took place on October 6th.

The 2021 field season was considerably wetter than 2020 and without as many peak temperatures in the mid-30°C. In June, July, and August of 2020, a total of 168 mm of precipitation fell, whereas 258.4 mm fell in the same three months of 2021 (Government of Canada, 2021). The weather station on the study site recorded 51.2 mm or rain on July 9th during the remnants of Hurricane Elsa. Minor rill erosion was observed in all three land management practices the following day, causing exposed potato plant roots and small channels oriented up-and-down slope (Appendix E).

2.2.3 Sampling and Analysis

In 2020, composite soil samples were collected (Appendix C) on July 2nd and September 24th for all analyses except bulk density, which was done on August 19th (A and B subplots) and August 26th (C subplots).

In 2021, composite soil samples were collected on May 27th and October 7th. All samples from both years were air dried and passed through a 2-mm sieve. Each composite sample was comprised of 5 to 6 cores (0-15 cm depth) collected by Dutch auger using a random sampling strategy within each subplot.

Each soil sampling date had a total of three samples per plot, for a total of 54 (A, B, and C subplots) samples per date. The C subplots were added late and had to be located in the middle to match inventories and sampling patterns. For aggregate stability, POXC, and total C, all 54 samples were used. For time-consuming analyses like density fractionation of POC, only 36 samples were used (A and B subplots).

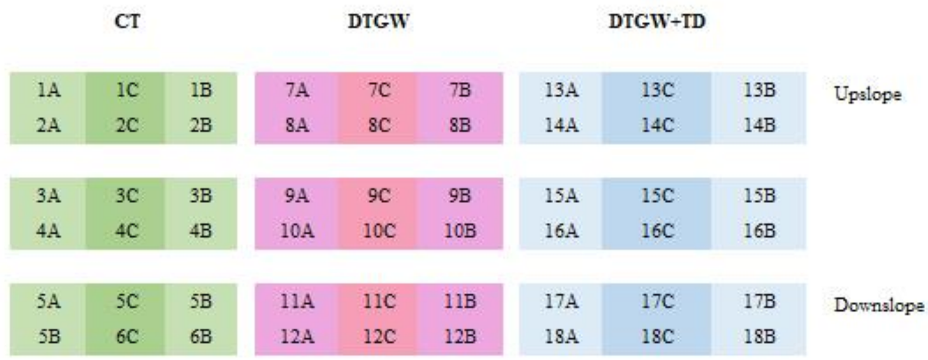

Figure 2-2 Sampling locations for the 3 land management practices – CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage). Each LMP contained 3 blocks, which each contained 2 plots. Each plot was then further subdivided into 3 subplots containing the A, B, and C sampling locations.

2.2.3.1 Soil Organic Carbon

Soil organic carbon (SOC) concentrations were determined by dry combustion on 0.2-g subsamples using an Elementar varioMACRO apparatus (Elementar Americas Inc., Mt. Laurel, NJ).

2.2.3.2 Soil Aggregate Stability

Soil aggregate stability was determined using a 08.13 Wet Sieving Apparatus (Eijkelkamp, 2008) on soil samples sieved to 1-2 mm collected on July 2nd, 2020 and October 7th, 2021. Results were determined according to modified Eijkelkamp operating instructions (Wilson et al., 2018; Zebarth et al., 2022). Soil aggregate stability (SAS) is calculated using the following equation:

Equation 2.1

$$SAS = ((Sa\text{-}Sw)/(T\text{-}Sw)) * 100$$

where *Sa* is the stable aggregate weight after H_2O (dry stable aggregate portion + sand portion), *Sw* is the dry sand weight, and *T* is the total dry weight (loose + stable + sand portions).

2.2.3.3 Permanganate Oxidizable Carbon

Permanganate oxidizable carbon was determined on samples collected from the following sampling dates: July 2nd and September 24th, 2020; and May 27th and October 7th, 2021. Results were analyzed according to Weil et al. (2003), with procedures described by Culman et al. (2012) (Appendix H).

Samples were analyzed at 550 nm using a Biochrom Libra spectrophotometer (Biochrom Ltd., Cambridge, United Kingdom) and calculated using the following equation:

Equation 2.2

$$POXC\ (mg\ kg^{-1}) = [0.02\ mol/L - (a+b*Abs)] * (9000\ mg\ C/mol) * (0.02\ L\ solution/Wt)$$

where 0.02 mol/L is the initial solution concentration, *a* is the intercept of the standard curve, *b* is the slope of the standard curve, *Abs* is the Absorbance of Unknown, 9000 is the milligrams of carbon oxidized by 1 mole of MnO_4 changing from Mn^{7+} to Mn^{2+}, 0.02 L is the volume of stock solution reacted, and *Wt* is the weight of air-dried soil sample in kg.

2.2.3.4 Particulate Organic Carbon

Density fractionation used to determine particulate organic carbon was performed on soil samples collected on: July 2nd, 2020; September 24th, 2020; May 27th, 2021; and October 7th, 2021. The protocol was based on the method described in Poeplau et al. (2018) with one alteration – removal of the sieving at 0.2 mm (Appendix J). Hence, the protocol included the following three specific fractions: (i) fPOM (free light POM separated from oPOM); (ii) oPOM (sand-sized and occluded POM, 53-2000 μm); and (iii) MAOM (silt- and clay-sized mineral associated organic matter, < 53 μm rinsed from catch pan). Final samples were analyzed for total

C concentrations using an Elementar varioMACRO apparatus (Elementar Americas Inc., Mt. Laurel, NJ).

2.2.4 Statistical Analysis

A linear model was used to assess the *change* in SOC, aggregate stability, POXC, and the C concentration of the three fractionated pools (each fraction assessed individually). Change was assessed from July 2nd, 2020, to October 7th, 2021. Land management practice (CT, DTGW, and DTGW+TD) and slope position (upper, middle, and lower) were considered as fixed main effects. If there was no interaction between LMP and slope position, the interaction was dropped from the model. A Tukey post-hoc HSD test was performed using the *emmeans* function (Lenth, 2018) when ANOVA results showed significant differences between main effects ($\alpha = 0.05$). The normality and variance homogeneity assumptions of ANOVA were confirmed using Shapiro-Wilks test and Bartlett's test, respectively, in addition to residual diagnostic plots. Statistical analyses were conducted using R software version 4.0.3 (R Core Team, 2020).

2.3 Results

2.3.1 Soil Organic Carbon

Soil organic carbon did not change substantially from post-planting 2020 to post-harvest 2021 across LMP or slope position (Figure 2-3). At the study period beginning, total SOC in CT, DTGW, and DTGW+TD averaged 18.7 ± 0.5, 18.5 ± 0.4, and 16.3 ± 0.7 g C kg soil^{-1}, respectively. At the end of the study, SOC values averaged 18.0 ± 0.5, 18.5 ± 0.4, and 15.9 ± 0.8 g C kg soil^{-1}, respectively. There was approximately 21% more C in the lower slopes than in the upper or middle slopes. Across the entire study period, SOC in the upper, middle, and lower slopes averaged 16.4 ± 0.4, 16.6 ± 0.3, and 20.0 ± 0.3 g C kg soil^{-1}, respectively.

In the model assessing change in SOC through the length of the study period, LMP ($p = 0.79$) and LMP × slope position interaction ($p = 0.50$) was not significant at a statistical threshold of $\alpha = 0.5$ (Figure 2-4). Slope position ($p = 0.06$) was suggestive but inconclusive.

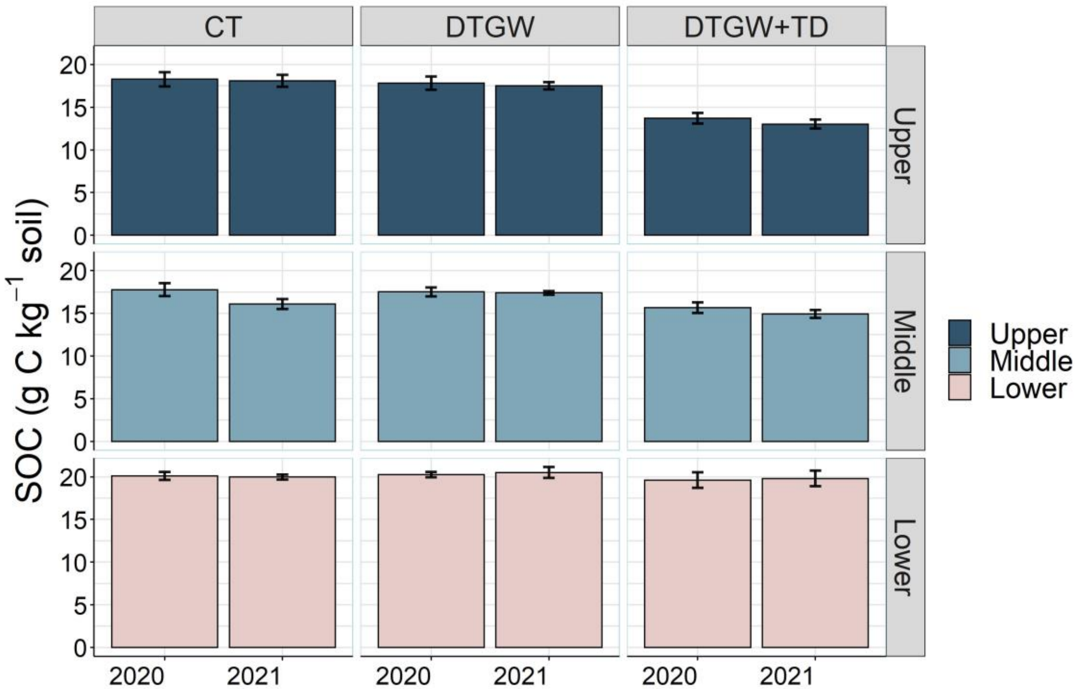

Figure 2-3 Soil organic carbon concentration within each land management practice (LMP) at post-planting 2020 and post-harvest 2021 for contour tillage - CT, diversion terraces and grassed waterways - DTGW, and diversion terraces, grassed waterways, and tile drainage - DTGW+TD, shaded by slope position. Error bars represent the standard error of the mean (n = 6).

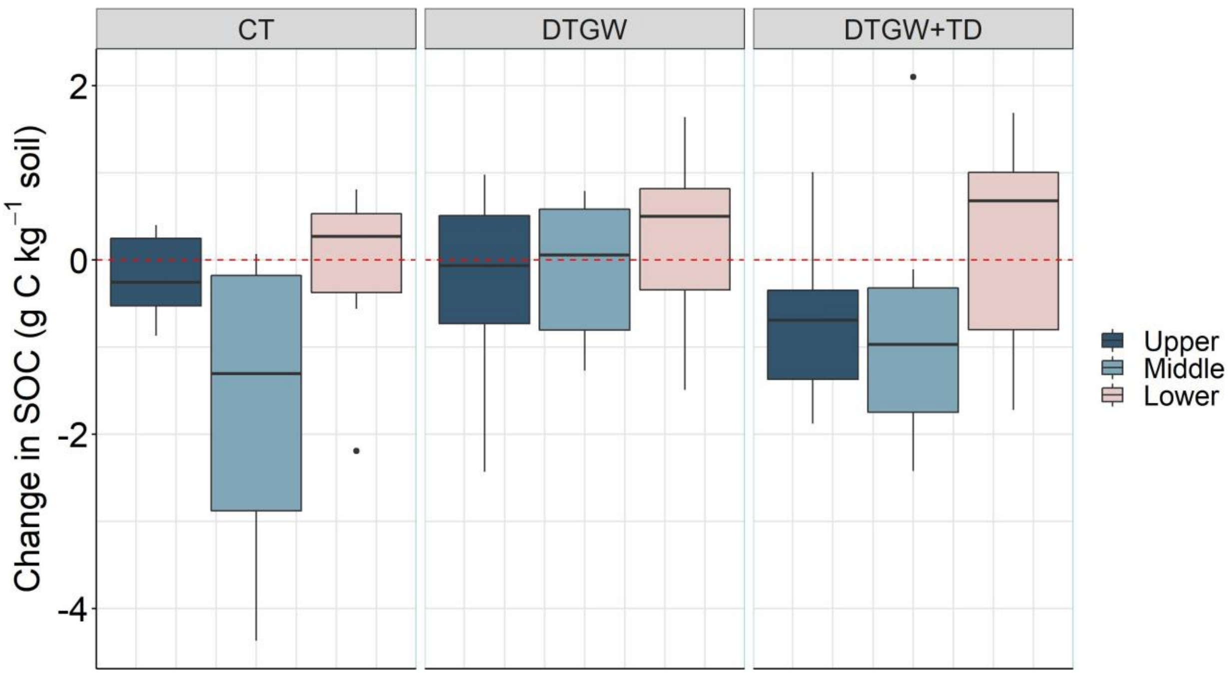

Figure 2-4 Change in soil organic carbon (SOC, g C kg soil^{-1}) between post-planting 2020 and post-harvest 2021. Boxplots represent one slope position (upper, middle, or lower) per land management practice – CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage). Neither treatment nor slope position were significant ($\alpha = 0.05$). Boxplots display the median, 25th and 75th percentiles, whiskers, and individual outliers (dots), for this and all subsequent boxplots. Dashed red line equals 0 change.

2.3.2 Soil Aggregate Stability

No significant differences were found in aggregate stability among LMPs ($p = 0.62$) or slope positions ($p = 0.84$), likely due to the large variability in terraced management practices (Figure 2-5). There was no interaction between LMP and slope position ($p = 0.61$).

Mean aggregate stability percentages in CT, DTGW, and DTGW+TD were 92.1 ± 0.4, 93.2 ± 0.3, and 88.6 ± 0.6 in 2020 and 87.7 ± 0.7, 89.1 ± 0.6, and 85.3 ± 1.1 in 2021, respectively. Soil aggregate stability percentages across all LMPs were between 83.7 and 95.9% in July 2020 and from between 77.53 and 97.07% in October 2021.

Mean aggregate stability percentages in upper, middle, and lower slopes were 91.6 ± 0.7, 90.2 ± 0.8, and 92.1 ± 0.4 in 2020 and 87.7 ± 0.6, 86.6 ± 0.8, and 87.9 ± 1.1 in 2021, respectively.

The only LMP that showed any increase in aggregate stability was DTGW+TD. However, only 3 of 18 samples showed an increase: 1 from the middle slope position and 2 from the lower slope. The most variability overall was seen in DTGW+TD, while CT had the least variability. Although aggregate stability declined in all LMPs, the measured aggregate stability values were all very high (Appendix G).

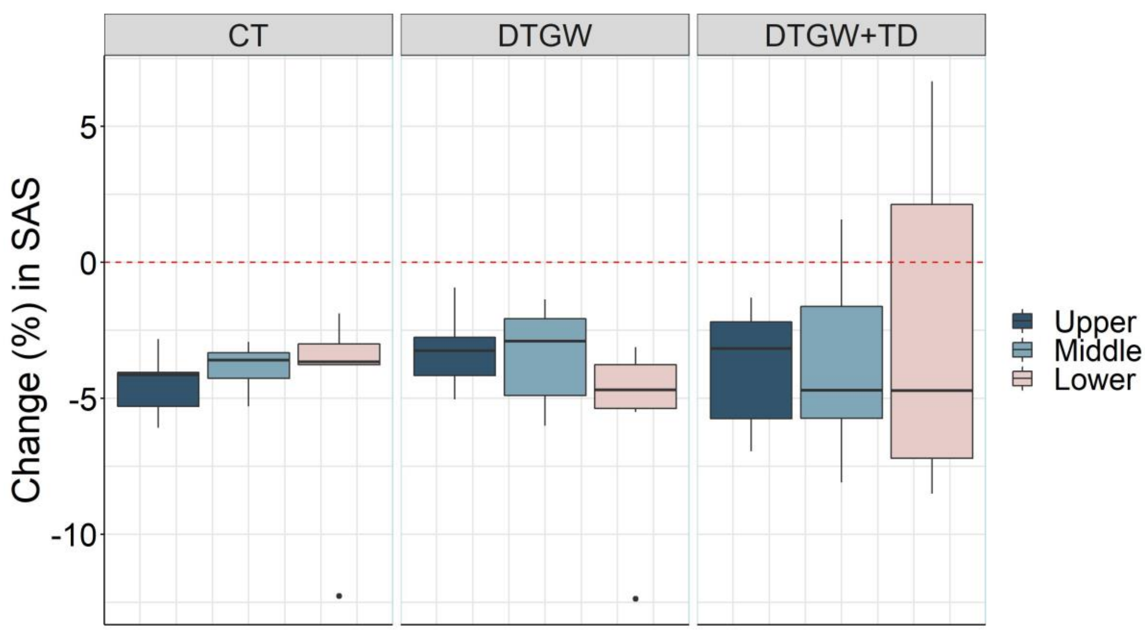

Figure 2-5 Percentage change (%) in soil aggregate stability (SAS) between post-planting 2020 and post-harvest 2021 in CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage).

2.3.3 Permanganate Oxidizable Carbon

Clear trends across LMPs were evident with POXC data. In CT, POXC values declined over the sampling period (Figure 2-6). In CT, average POXC values began at 319.71 ± 8.5 mg kg^{-1} soil on July 2nd, 2020 and measured 282.2 ± 9.1 mg kg^{-1} soil on October 7th, 2021. Meanwhile, in the terraced LMPs, POXC trended upward over time. In DTGW and DTGW+TD, POXC values measured 265.7 ± 8.3 and 191 ± 12.1 mg kg^{-1} soil, respectively, on July 2nd, 2020, and measured 312.6 ± 9.8 and 278.3 ± 11.7 mg kg^{-1} soil, respectively, on October 7th, 2021.

As shown in Figure 2-7, CT mostly lost labile C over the 16-month study period, averaging a loss of 11.2% across LMP and slope. It is possible that C in this form washed away each growing season. Terraced LMPs, on the other hand, gained POXC. The more that water was regulated, the more POXC accumulated; DTGW gained an average of 19.8%, while DTGW+TD rose an average of 50.6%.

Land management practice ($p < 0.001$) and the interaction between LMP and slope position ($p < 0.01$) were significant in the model, while slope position itself was not ($p = 0.19$).

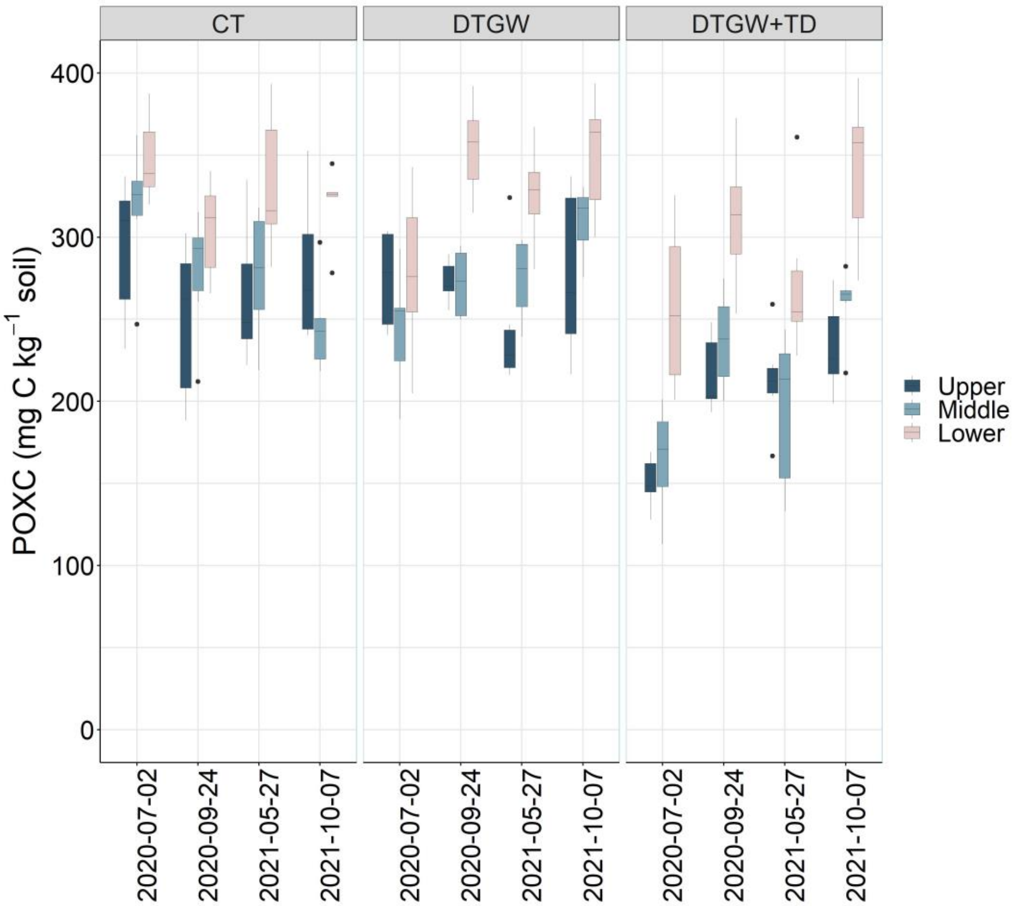

Figure 2-6 Permanganate oxidizable carbon (POXC) values in CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage). Soil measurements were taken on July 2nd and September 24th, 2020 and May 27th and October 7th, 2021. Boxplots represent one slope position (upper, middle, or lower) per LMP.

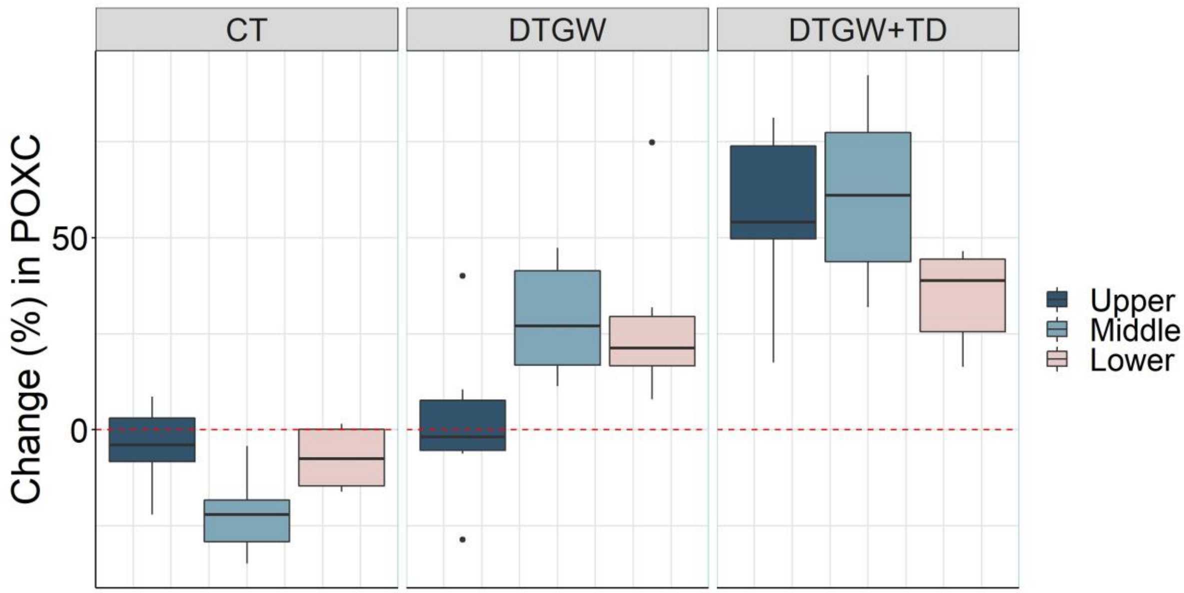

Figure 2-7 Percentage change (%) in permanganate oxidizable carbon (POXC) between post-planting 2020 and post-harvest 2021 in CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage). There was a significant LMP × slope position interaction ($\alpha = 0.05$).

2.3.4 Particulate Organic Carbon

The concentrations of C in each LMP changed over the study period (Figure 2-8). Overall, CT lost an average of 0.58 ± 1.4 g C kg^{-1} soil (-3.75%) in the MAOM fraction, while DTGW+TD gained an average 1.7 ± 0.9 g C kg^{-1} soil (+9.7%). The biggest change was seen in DTGW, where C in the MAOM fraction rose by an average of 2.5 ± 1.03 g C kg^{-1} soil, or roughly +15.8%. There was considerably more C in the MAOM fraction than in other fractions. Concentrations, across all LMPs, slope positions, and dates, averaged 2.0 ± 0.04, 1.65 ± 0.1, and 13.6 ± 0.3 g C kg $soil^{-1}$ in fPOM, oPOM, and MAOM, respectively. What is interesting about the gains made in terraced LMPs is that they occurred while the whole mass of the MAOM fractions recovered declined (Figure 2-9). An average decline of 4.85% to 417 g kg $soil^{-1}$ occurred in DTGW, while DTGW+TD declined an average of 10.6% to 415 g kg $soil^{-1}$. There was an increase of MAOM C in 75% of DTGW subplots and in 66% of DTGW+TD subplots, particularly in the middle and lower plots.

The fPOM C concentration trended slightly lower with increasing LMP water management (i.e., CT>DTGW>DTGW+TD), though differences were minimal. The fPOM concentrations in the CT treatment had an average 2.14 ± 0.1, while DTGW had an average 2.07 ± 0.1, and DTGW+TD had an average 1.89 ± 0.1 g C kg^{-1} soil. However, CT lost the most g C kg^{-1} soil over the 16 months of this study (-0.34 ± 0.1), while DTGW+TD lost the least (-0.09 ± 0.1) (Figure 2-5).

The oPOM trends were very similar to those of fPOM. Contour tillage had the highest average g C kg^{-1} soil throughout the two years (1.97 ± 0.1) but lost the most C over the 16 months (-0.32 ± 0.1), whereas DTGW+TD had the lowest average (1.39 ± 0.04) but gained C (+ 0.22 ± 0.1).

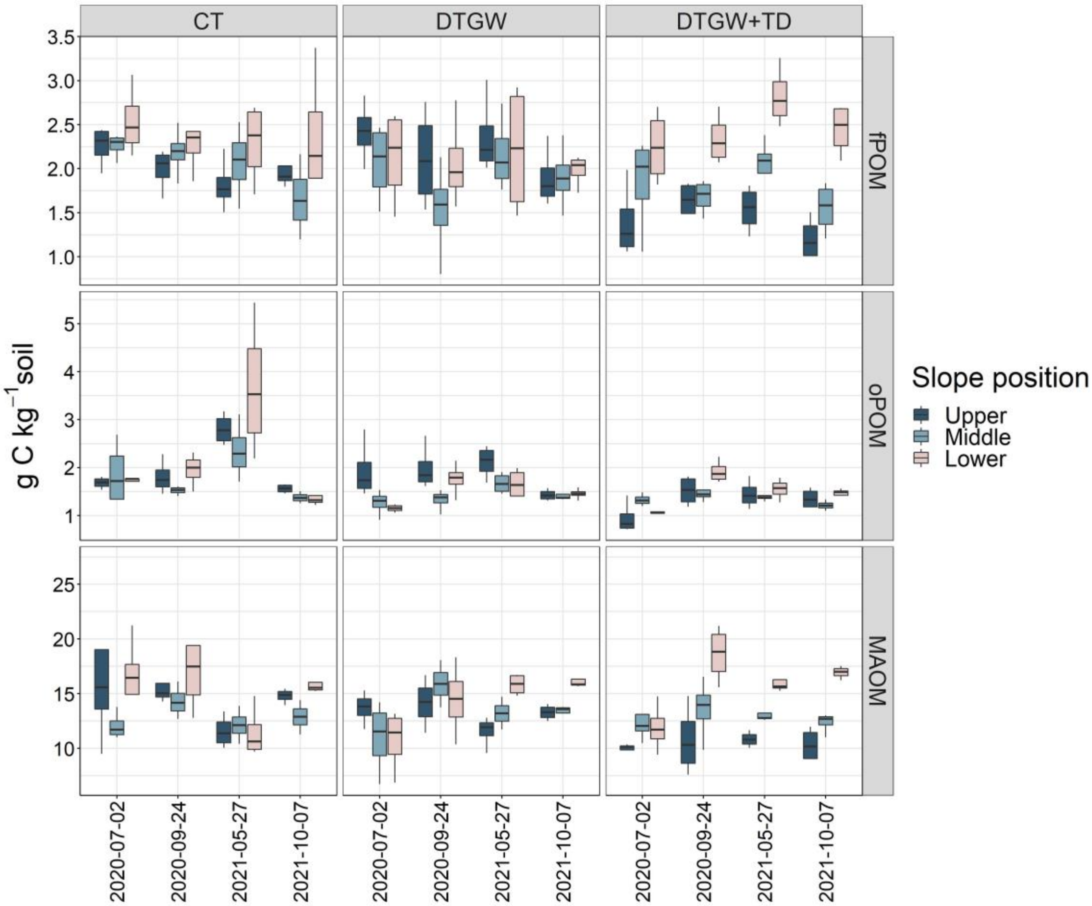

Figure 2-8 Organic C concentrations per kg of soil between post-planting 2020 and post-harvest 2021 in CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage).

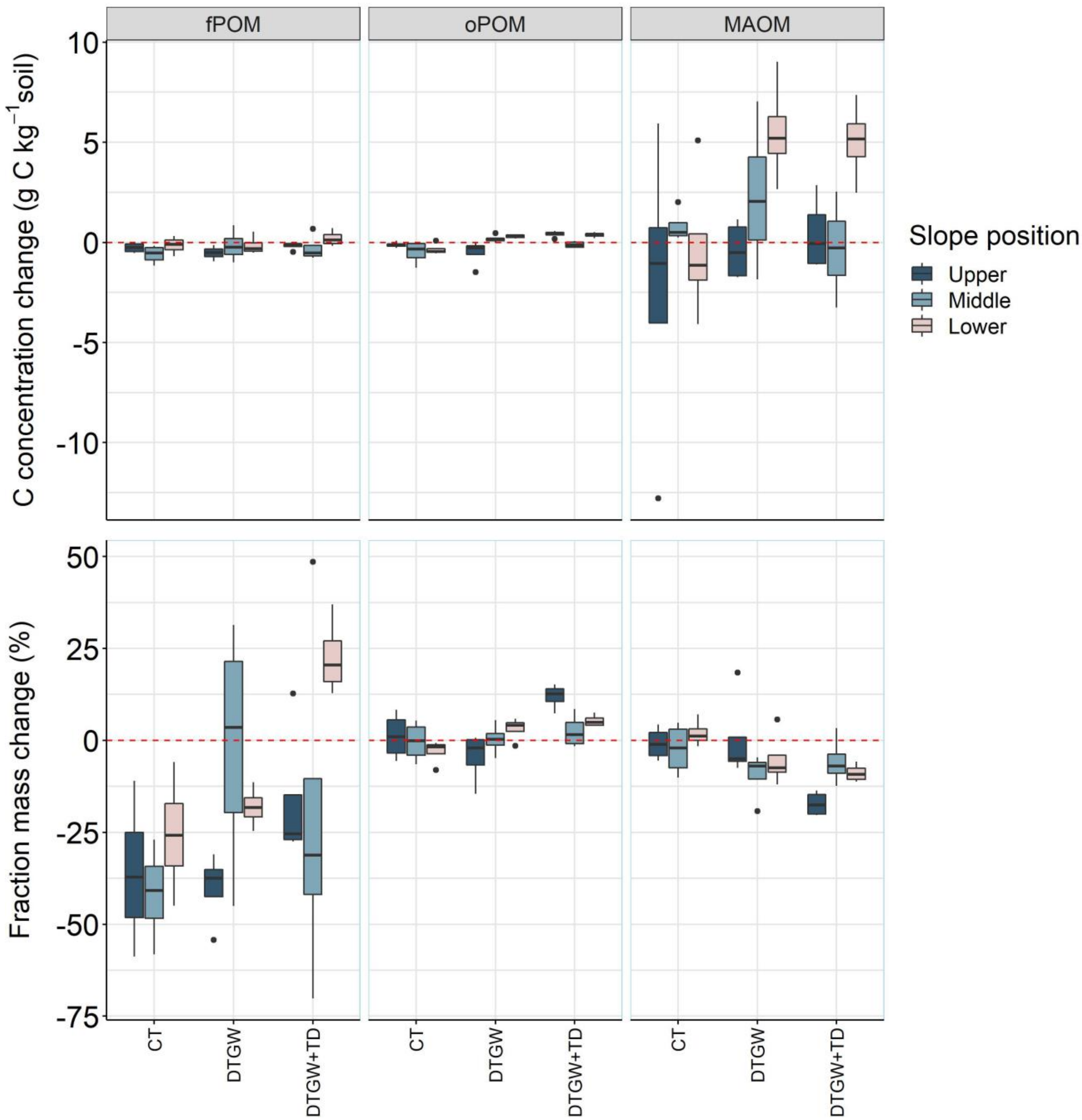

Figure 2-9 Top: Change in organic C concentration per kg of soil between post-planting 2020 and post-harvest 2021 in CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage). Bottom: Percentage change in the mass of each fraction between the same dates.

2.4 Discussion

2.4.1 Soil Organic Carbon

Soil organic carbon (SOC) was not expected to change significantly over the course of the second and third years after LMP establishment. The turnover of SOC is generally slow, even in cases where intensive potato production takes place. In the Chow et al. (2010) study on impacts of long-term intensive potato production and terraces on runoff hydrology and soil quality, for instance, SOC only declined 3.6% over the course of an entire decade. Once a new management practice is implemented, measurable differences in SOC arising from altered soil management can often take years to be detected and may not be picked up in a short-term study (Sikora et al., 1996). This is the impetus for many of the methods that quantify labile soil C rapidly and inexpensively, such as particulate organic carbon and POXC, that are included in this study.

2.4.2 Soil Aggregate Stability

In the second and third year after LMP establishment, the CT management practice showed a small but clear trend where aggregate stability decreased more in the elevated slope position and less in the lowest slope position (Figure 2-5). The opposite trend appeared in the terraced LMPs, where aggregate stability decreased more on lower slopes.

It is not surprising that soils, which were grass pasture before site construction, showed high aggregate stability values above 80%. The LISWC system had only been built in 2018, and no potato cultivation had taken place there in the past decade, so the soil had become well-aggregated. The oat crop from 2020 also required very few tillage events.

Also, 2020 was warmer and drier than typical for this region, which could have further contributed to better aggregate stability. Soil aggregate stability is generally negatively correlated with soil water content, since aggregates tend to become stronger with drying (Caron et al., 1992; Rasiah et al., 1992). Percentages in the 80 to 90% range are on the higher end for potato cropping systems in the area. A study on similar Fredericton-area soils with compost amendments to potato crops had values as low as 67.7% (Wilson et al., 2018), and a biochar-amended soil at the FRDC in 2015 reported values in the 70% range (Delisle, 2018). Aggregate stability averaged 68% in a province-wide study of commercial fields between 2013 and 2017, with values as low 32% (Zebarth et al., 2022). Longer term monitoring of this site would reveal whether or not diversion terraces could assist in retaining high aggregate stability relative to contour tillage with repeated potato cropping.

Other research in Atlantic Canada has demonstrated that certain BMPs can increase water-stable soil aggregates either through conservation tillage (Carter & Sanderson, 2001; Carter et al., 2009) or crop rotation (Angers et al., 1999), but those were 6-year studies. There have been few studies in the area specifically focused on terraces and/or tile drainage effects on aggregate stability. There is evidence elsewhere that tile drainage can increase macroaggregation but negatively affect other aggregate size classes, as seen in a study comparing tile-drained and non-drained Ohio soils (Kumar et al., 2014). Increasing water-stable aggregates is a slow process (Rees et al., 2014), and it is possible that over a longer study period that patterns may emerge, given the differences in variation between the three land management practices.

2.4.3 Permanganate Oxidizable Carbon

Soil organic matter was determined directly through changes in SOC and through changes in POXC and POC. These labile SOM fractions – POXC and POC – have been identified in other studies to be sensitive to management practices (Bongiorno et al., 2019; Culman et al., 2012; Haynes, 2005; Hurisso et al., 2016). By monitoring trends in these pools of carbon, it is possible to get an earlier indication of the consequences of different land management practices than monitoring total organic matter (Dalal & Mayer, 1986a,b). Little is known about how SOC pools respond to the effects of soil water redistribution from the implementation of diversion terraces, grassed waterways, and tile drainage.

In each of the two study years, the following trends were evident between planting and post-harvest – a decrease in labile POXC in CT and an increase in the terraced LMPs. However, between fall of 2020 and spring of 2021, the trend in all three LMPs reversed – but did not completely revert to starting values – before resuming their respective trends (Figure 2-6). This suggests a seasonal effect on POXC. Culman et al. (2012) found POXC to be significantly related to microbial biomass C and closely related to smaller-sized (53–250 μm) particulate organic carbon (POC) fractions. As microbial biomass C might decline in the freezing conditions of a Maritime winter, and small POC fractions should be washed away in the spring snow melt, I would expect that POXC would decrease in all LMPs over winter – perhaps more in CT where there are no terraces to slow water runoff. But this was not the case in CT, where it increased over winter. One potential explanation might be that the presence of diversion terraces increased the soil moisture heading into winter and therefore also decreased the soil temperature. This could slow down the microbial processing of this carbon fraction. However, tile drainage has been shown to increase soil temperature (Fraser & Fleming, 2001), and since there was no difference in POXC trends between DTGW and DTGW+TD over winter, soil temperature is not likely the reason POXC values went up over winter in CT.

An Ohio study that compared tile-drained and undrained cropland found that the loss of readily mineralizable C was roughly 4 times greater in tile-drained land (Jacinthe et al., 2001). Not only did the tile-drained soil in our study gain labile C instead, but the tile-drained terraced soil gained more labile C than the undrained terraced LMP. Other studies also document higher depletion rates of labile C pools when native ecosystems are converted to agriculture. (Elliott, 1986; Carter et al., 1998).

Studies have shown a close, positive relationship between POXC and SOC, and POXC is considered to be a relatively processed fraction of labile C (Culman et al., 2012). Study by Culman et al. (2012) specifically showed the fraction was more closely related to heavier than lighter particulate organic C (POC) and could thus be considered a good indicator of organic matter stabilization. The POXC trends in land uses were in line with the heavier POC pools fractioned in this study (oPOM, MAOM). Short-term changes in POXC reflect long-term changes in SOM (Weil et al., 2003), and though total C was statistically unchanged between years in any LMP, these results indicate an increase in C stabilization in terraced LMPs over fields managed with contour tillage.

2.4.4 Particulate Organic Carbon

The majority of C (~78.7%) in my study was found in the MAOM fraction of the soil – very similar to the lower end of results from a study by Gregorich et al. (2009) on long-term tillage and crop rotations in Prince Edward Island, Québec, and Ontario.

With the C increasing in both terraced LMPs over the two-season study period, it is plausible to expect that those LMPs might eventually hold more C in the MAOM fraction if the trend continued. This is because, as Haddix et al. (2020) point out, the MAOM pool forms quickly from any labile C in the soil and, in soils with relatively low C content (< 3%), as in the soil in our study, there is positive feedback between new SOM stabilization and soil C. Both terraced LMPs saw large gaps between the gains made in upper and lower plots. It is possible there was some downslope movement of C in terraced LMPs, but it is unlikely for several reasons. First, diversion terraces have proven effective at preventing erosion (Chow et al., 2010), so it is not likely that POM C or MAOM C were translocated downhill by way of soil erosion. The POM C changes were also minimal and would not be large enough to account for the change. Although dissolved organic carbon (DOC) may have moved through the soil profile, it generally makes up less than 2% of SOM (von Lützow et al., 2007). The DOC (data not shown) was generally equal in each LMP and slowly (and equally) declined throughout the study period across all treatments.

The MAOM fraction represents a more stable form of soil C due to its association with soil mineral particles and consequent protection from decomposition. From the perspective of climate change mitigation and soil health on potato fields in Atlantic Canada, it might be advantageous that MAOM increases over POM. Recently, Lavallee et al. (2020) proposed a simplified framework to assess SOM with only two components, with fundamental differences in formation, persistence, and function – particulate organic matter (POM) and mineral-associated organic matter (MAOM). The defining difference between the two being that POM is physically unprotected from decomposition processes, while MAOM is protected through association with silt- and clay-sized soil mineral particles. Such differentiation does not hold for my study, since recent results from numerous New Brunswick locations (unpublished data), including agricultural sites, have shown that the oPOM fraction contains the oldest C (L.P. Comeau, personal communication, March 30, 2022). If the oPOM C is able to persist the longest in New Brunswick agricultural soils, increasing its relative proportion would be more conducive to storing C long term. The fPOM fraction is not protected by organo-mineral complexes and is therefore a readily available substrate and nutrient source for micro-organisms (Gregorich et al., 1994). This fraction will turn over in Canada in 1-8 years (Carter, 2002) and is still important to have in the soil so that SOC consistently turns over, stimulating plant productivity and releasing nutrients.

The changes in POM (fPOM + oPOM) C were small, given that the expectation was that with cultivation comes a decrease in POM. The persistence of POM is controlled mainly by microbial and enzymatic inhibition (Lavallee et al., 2020). Due to fPOM being less protected from the activity of microbes and enzymes, environmental changes that may decrease microbial inhibition or disrupt aggregates (e.g., tillage) can immediately increase the POM decomposition rates.

Because oats were seeded in 2020, I was not able to observe any follow-up effect of the 2021 potato crop on POM during my study period. The oats were harvested, but the straw was left to be incorporated into the soil in the fall. There is potential that in a season following potato, POM C (particularly fPOM C) might decrease significantly more, since so little organic matter inputs occur following potato harvest.

2.5 Conclusions

This study examined effects of three LMPs on indices of soil quality. It was hypothesized that aggregate stability would decrease the most in terraced LMPs, but there was no difference in the 2nd and 3rd seasons after LMP establishment. Because the diversion terrace and grassed waterway system diverts more of the surface water downwards into the soil profile, this was the rationale for the hypothesis. Perhaps soil was not wet enough in either growing season to induce greater loss in aggregate stability in terraced treatments over CT, or perhaps the effects of higher soil water content were masked in the short term when intensive tillage was taking place.

Contour tillage (CT) averaged a loss of 11.2% POXC, while terraced LMPs – DTGW and DTGW+TD – gained 19.8% and 50.6%, respectively. The more water was regulated, the higher labile C rose. The data from this study show that, at least in the short term, labile C can rise shortly after the construction of BMPs that include diversion terraces and grassed waterways, with or without tile drainage.

There was an increase in MAOM C in the terraced LMPs, despite the high amount of tillage events that took place during potato cultivation. I also saw an increase in MAOM C concentrations in the terraced LMPs despite the mass of the whole fraction in each LMP declining.

Overall, there is evidence that the LISWC system has positive impacts on the soil properties evaluated. Though both terraced LMPs saw a decline in aggregate stability, neither were any worse than the CT. The diversion terraces and grassed waterways increased the fraction of POXC relative to CT. The further addition of tile drainage contributed to even higher POXC increases. Diversion terraces and grassed waterways also increased the fraction of organic matter associated with mineral particles, which represents a type of carbon that tends to have a slower turnover rate. This indicates that carbon is being stabilized and SOC may begin to increase due to the change in soil management.

Chapter 3: Greenhouse Gas Emissions and Nitrate Supply Rate Response to Initial Three Years of Diversion Terrace, Grassed Waterway, and Tile Drainage Implementation

3.1 Introduction

The Landscape Integrated Soil and Water Conservation (LISWC) is a new system, designed five years ago, to conserve soil and water resources on sloping landscapes by integrating multiple BMPs (Li, 2021). The system combines diversion terraces, grassed waterways, tile drainage, water retention structures, supplemental irrigation, conservation tillage practices, and soil-landscape restoration. There are concerns, however, that in wet years the diversion terraces can lead to fields being too wet (Chow et al., 2010) since diversion terraces slow down surface drainage and direct more water into the soil profile. In turn, this can also lead to cooler soil temperatures. Soil water and temperature are the two most influential parameters for soil gas emissions (Oertel et al., 2016). Soil moisture controls microbial activity and related microbial processes by solubilizing substrate compounds and, through diffusion, increases substrate availability to active microbial sites (Sierra et al., 2017). Heightened microbial activity will then lead to higher decomposition of organic matter and higher microbial respiration rates that produce GHGs in soils. Rising soil temperatures also increase microbial metabolic rates, increasing respiration further (Larcher, 2003).With too much water in the system (i.e., saturated soil), anaerobic conditions foster the production of CH_4 and N_2O from methanogenesis and nitrification/denitrification, respectively. The dominant process in humid climates such as in New Brunswick is denitrification (Mosier, 1998).

Crop management practices can affect GHG emissions. Destructive management like the frequent tillage in potato farming can break down aggregates and expose organic matter to oxidation or decomposition processes. In general, carbon (C) sequestration will be more successful in land management systems that minimize erosion and soil disturbances, maximize return of crop residues, or maximize the efficiency of crop water and nutrient use (Paustian et al., 2004). Because of the intensive nature of most cropping systems – particularly potato production systems – we could mitigate GHG production and store more C and nitrogen (N) in the soil by modifying soil management practices (Smith et al., 2008).

Because nitrate (NO_3^-) is a soluble substance prone to leaching, the water redistribution from diversion terraces, grassed waterways, and tile drainage may affect the NO_3^- concentrations within the soil. Emissions of N_2O are a direct function of nitrogen (N) application rate, and the NO_3^- supply rate within the soil to plants, after fertilization, may be affected by the different management practices studied – contour tillage (CT), diversion terraces and grassed waterways (DTGW); and diversion terraces, grassed waterways, and tile drainage (DTGW+TD).

The objective of this study was to evaluate the effects of diversion terraces, grassed waterways, and tile drainage, relative to contour tillage, on GHG emissions from the soil and on nitrate supply rates that may relate to N_2O production during the initial three years of management implementation. Information gathered will help understand how GHG fluxes respond to altered water regimes under the differing land management practices. There have been no studies in the area that quantify GHG emissions from fields with diversion terraces and grassed waterways.

3.2 Materials and Methods

3.2.1 Site Description

The field experiment was conducted between May 2020 and October 2021, at the Fredericton Research and Development Centre (FRDC) of Agriculture and Agri-Food Canada. The FRDC is located on the south-east edge of the city of Fredericton, New Brunswick, while the study site was located on a northeast facing slope facing the Saint John River (45°55'33.5"N 66°36'55.5"W). The topography of the Fredericton area ranges from 6 to 215 m above mean sea level. The whole area has been subject to glaciation and most upland soils have been developed from weathered glacial till parent materials (Zelazny, 2007). The soils were part of the Research Station Association, classified as Orthic Humo-Ferric Podzols, and developed in areas with loose ablation till over compact lodgment till (Rees & Fahmy, 1984). They are moderately permeable sandy loam with 10 to 25% angular cobbles and gravels and a pH of 5-6. The soil texture is classified as a sandy loam, averaging 55% sand, 34% silt, and 10% clay (Appendix B). The Fredericton area receives a mean annual rainfall of 885.9 mm and a mean annual snowfall of 214.8 cm (Government of Canada, 2019).

Construction of the LISWC system began in 2018. Prior to construction, the site was under grass for two seasons. Before those two seasons, several strips were plowed along the slope for a rainfall simulation experiment, but nothing was planted. When construction began in 2018, old clay drainage tiles (dating to the mid-20th century) were found. The tiles were removed from the site after excavation. They were no longer functional and would have had marginal impact on soil moisture and water flow. In the initial 2019 season the study site was seeded to oat (*Avena sativa* L.). In 2020, the site was again seeded to oat, while in 2021 a potato (*Solanum tuberosum* L.) crop was planted.

The study consisted of three land management practices (LMPs) constructed on a slope with a gradient of 10% (Figure 2-1). Each LMP took up a total of 1400 m^2 (70 × 20 m). Each LMP represents a different combination of individual best management practices. The 1st LMP acted as a control and consisted only of contour tillage on each year's crop – it will be referred to from here on as CT. The only permanent BMP installed in CT was the water retention structure at the base of the slope. There were 6 plots within CT.

The 2nd LMP was subdivided into 3 segments, each with 2 plots. Diversion terraces and grassed waterways were constructed below each of the three 20 × 20 m blocks. A water retention structure was constructed below the lowest block. This LMP is referred to as DTGW. The 3rd LMP, DTGW+TD, was a repeat of DTGW but with tile drainage installed, which is representative of the full LISWC system. Two polyvinyl chloride (PVC) tile drains, measuring 10 cm in diameter, were installed per block, 10 m apart. Edges of the LMPs were lined with an impermeable plastic barrier extending to a depth of 1 m, to isolate each LMP from potential inflows from other LMPs. Contour tillage was implemented in both DTGW and DTGW+TD.

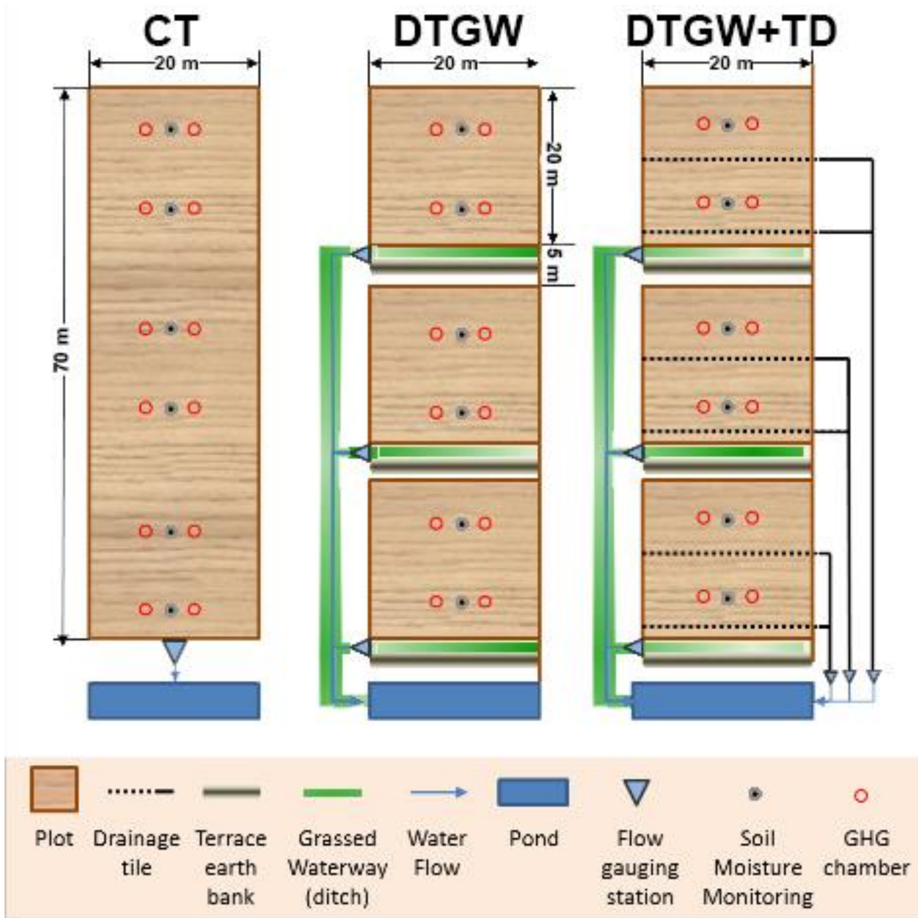

Figure 3-1 Schematic of the study site layout and the 3 land management practices (LMP) – CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage).

3.2.2 Sampling and Analysis

3.2.2.1 CO_2 Readings

To sample CO_2 fluxes, an EGM-5 Portable CO_2 Gas Analyzer (PP Systems, Amesbury, MA, USA) with an SRC-2 Soil Respiration Chamber was used throughout the growing seasons of 2020 and 2021. A HydraProbe sensor (Stevens Water Monitoring Systems, Portland, OR, USA) was used with the EGM gas analyzer to gather surface soil moisture and temperature data with each flux measurement.

Flux measurements were taken twice a week, weather dependent, and began at 10:00 am to coincide with daily mean soil temperatures. Fluxes were measured either for 60 seconds or until the threshold for change in CO_2 concentration was reached. Measurements in 2020 went from May 28th to September 22nd. Measurements in 2021 were able to begin earlier, ranging from April 29th to September 28th. Once collars for N_2O and CH_4 were installed in the field, measurements were taken inside the collars, which stayed free of oat or potato.

The linear respiration rate was calculated by the EGM with the following formula:

$$R = \frac{(Cn - C_0)}{Tn} * \frac{V}{A}$$

where R is the respiration/assimilation rate (CO_2 flux, or moles of CO_2 unit area^{-1} unit time^{-1}), C_0 is the CO_2 concentration at T=0, and Cn is the concentration at finishing time Tn. The area of soil exposed is represented by A, and V is the total system volume (PP Systems, 2018).

3.2.2.2 N_2O and CH_4 Readings

Closed static dark chambers were used to make measurements of CH_4 and N_2O in the growing seasons of 2019, 2020, and 2021. A total of 36, cylindrical collars, measuring 38 cm tall and 30 cm in diameter were installed in the field at each subplot, at least 72 hours before the first measurement of the season (Appendix K). In 2019 and 2020, the site was seeded to oat, and any visible plant matter inside the collar was removed immediately after installation. In 2021, the collars were installed directly into potato rows on July 21st and 22nd. A single potato plant was pulled where each collar was to be installed, to ensure no seed potato or potato plant was inside the collar. This restricted the emissions generated from plant root respiration. In 2019, measurements were taken on August 28th and September 10th and 26th. In 2020, measurements were taken on July 29th, August 12th and 26th, and September 4th. In 2021, measurements were taken on July 28th, August 10th and 25th, and September 8th and 21st.

On each day of measurement, plastic lids 30 cm in diameter were secured to the top of the collar with a tube at a diameter small enough to ensure the collar would be an air-tight, closed system. Lids from plastic, 5-gallon buckets were custom fit with 2 one-touch fittings (SMC Corporation, Tokyo, Japan). Each fitting had placed inside of it a ten-inch length of 6mm tubing with a one-way stopcock (Cole-Parmer, Montréal, Canada) at the halfway point. Stopcocks were used to ensure venting, so that buildup of any gas was prevented before the time 0 measurement. Lids also had a rubber septum in the middle where the syringe would draw a 20 mL gas sample into a pre-evacuated 12 mL exetainer.

Once all collars were secured with lids, measuring could begin. Four measurements were made at time 0, 30, 60, and 90 minutes. Measurements were staggered by 60 seconds so that there was time to draw a syringe sample and move to the next sampling location. Two people each took 18 samples at each of the 4 intervals, leaving 12 minutes between the end of one interval and the beginning of the next.

All gas samples from exetainers were analyzed at the University of British Columbia using a 7890A gas chromatograph (Agilent Technologies Inc., CA, USA) equipped with a flame ionization detector and electron capture detector. It was also equipped with a PAL auto-sampler (Agilent Technologies Inc., CA, USA). Fluxes were calculated with R, version 4.0.3, in R Studio with the *gasfluxes* package (Fuss et al., 2020). The *lin.fit* method from the package was used to fit a linear model via R's *lm* function, using concentration and time data.

Continuous soil moisture and temperature data were taken from the 5TE sensors (METER Group). Data was taken from the 11:00 am reading – halfway between start and end of gas sampling – and from the 15 cm depth reading. During this study, measurements were only taken on bare soil – intentionally excluding plant root respiration and focusing solely on heterotrophic respiration.

3.2.2.3 Nitrate Supply Rate

The availability of nitrate was determined using Plant Root Simulator (PRS™-Probes) technology from Western Ag Innovations Inc., Saskatoon (SK), Canada. The probes are ion exchange membranes (adsorbing surface area of 17.5 cm^2/probe) encapsulated in a thin plastic probe. They were inserted into the 0–10 cm layer of the mineral soil where they adsorbed nutrients in their available forms. The probes also adsorbed ammonium ions, but a majority of the results were below method detection limits and were therefore not included. Probes were placed adjacent to the greenhouse gas sampling chambers in each of 2020 and 2021. In 2021, because the fields were seeded with potato, probes were only inserted into potato rows and not furrows. Probes were sent back to Western Ag for analysis after the removal of each round of probes.

The four sampling periods in 2020 were: (1) June 17 – July 12; (2) July 2 – July 28; (3) July 28 – August 18; and (4) August 19 – September 8. The three sampling periods in 2021 were: (1) July 26 – August 16; (2) August 16 – September 7; and (3) September 7 – September 22.

3.2.3 Statistical Analysis

The differences in GHG fluxes were analyzed using ANOVA and Tukey post-hoc tests. A linear mixed effects model was used with LMP (CT, DTGW, DTGW+TD) and slope position (upper, middle, lower) as fixed effects, subplot as a random effect to account for repeated measurements, and soil water content and temperature as continuous explanatory variables. A model was fit for each gas separately. Soil temperature was not included in the N_2O model, as it was not significant in the model. A linear mixed effects model was also used to assess volumetric water content (VWC); only LMP, slope position, and their interaction were included in the model as fixed effects. For CO_2, soil moisture and temperature values from the HydraProbe sensor were included as continuous explanatory variables. A linear mixed effects model was fit with LMP to assess total greenhouse gases emitted as CO_2 equivalents, with 298 and 25 used as the conversion factors for N_2O and CH_4, respectively (IPCC, 2019). Finally, for nitrate supply rate, a linear mixed effects model was fit with LMP and slope position as fixed effects and subplot as a random effect.

Type III ANOVA was used to evaluate significance of interaction terms in the models. A Tukey post-hoc HSD test was performed using the *emmeans* function (Lenth, 2022) if the main effects LMP or slope position were significant without a significant interaction between the two. The ANOVA assumptions of normality and homogeneity of variance were assessed using Shapiro-Wilks test and Bartlett's test, respectively, in addition to diagnostic plots. Data transformations were made when necessary to ensure conditions of ANOVA were met. A statistical significance threshold of $\alpha = 0.05$ was used for hypothesis testing. Statistical analyses were conducted using R software version 4.0.3 (R Core Team 2020).

3.3 Results

3.3.1 Carbon Dioxide

Soil temperature, soil water, and LMP were all significant in each year. Slope position was not significant in either year, nor was the interaction between LMP and slope position. Figures 3-5 and 3-6 show the CO_2 fluxes from 2020 and 2021.

Mean daily emissions (kg CO_2-C ha^{-1}) in 2020 were 56.6 ± 3.6, 68.8 ± 4.8, and 56.7 ± 3.7 in CT, DTGW, and DTGW+TD, respectively (Figure 3-7). In 2021 they averaged 30.1 ± 1.4, 42.2 ± 4.5, and 28.8 ± 1.2. Daily emissions across the entire study period averaged 43.1 ± 2.0, 55.3 ± 3.3, and 42.5 ± 2.0.

There were significant differences in total CO_2 emissions in both 2020 ($p = 0.018$) and 2021 ($p < 0.001$) (Figure 3-5). In 2020, total C emitted from CO_2 was 1640 ± 82.5, 1996 ± 112, and 1644 ± 74.7 kg C ha^{-1}, while in 2021 total C emitted was 903 ± 28.5, 1265 ± 105, and 863 ± 26 kg C ha^{-1} in CT, DTGW, and DTGW+TD, respectively.

Emissions were correlated to soil temperature slightly more than soil moisture. Emissions were positively correlated to soil temperature at 15 cm in 2020 ($r = 0.27$, $p < 0.001$, $n = 803$) and 2021 ($r = 0.10$, $p < 0.05$, $n = 468$). In 2020, there was no correlation between emissions and soil moisture, while in 2021 fluxes were only weakly correlated to VWC ($r = 0.10$, $p < 0.05$, $n = 468$).

In 2020, VWC averaged 0.158 ± 0.003, 0.169 ± 0.003, and 0.200 ± 0.005 $m^3 m^{-3}$, while in 2021, VWC averaged 0.282 ± 0.005, 0.313 ± 0.005, and 0.277 ± 0.001 in CT, DTGW, and DTGW+TD, respectively. In 2020, soil temperature averaged 22.9 ± 0.3, 23.2 ± 0.3, and 24.1 ± 0.3°C, while in 2021, temperature averaged 21.6 ± 0.2, 21.2 ± 0.2, and 21.8 ± 0.2°C in CT, DTGW, and DTGW+TD, respectively.

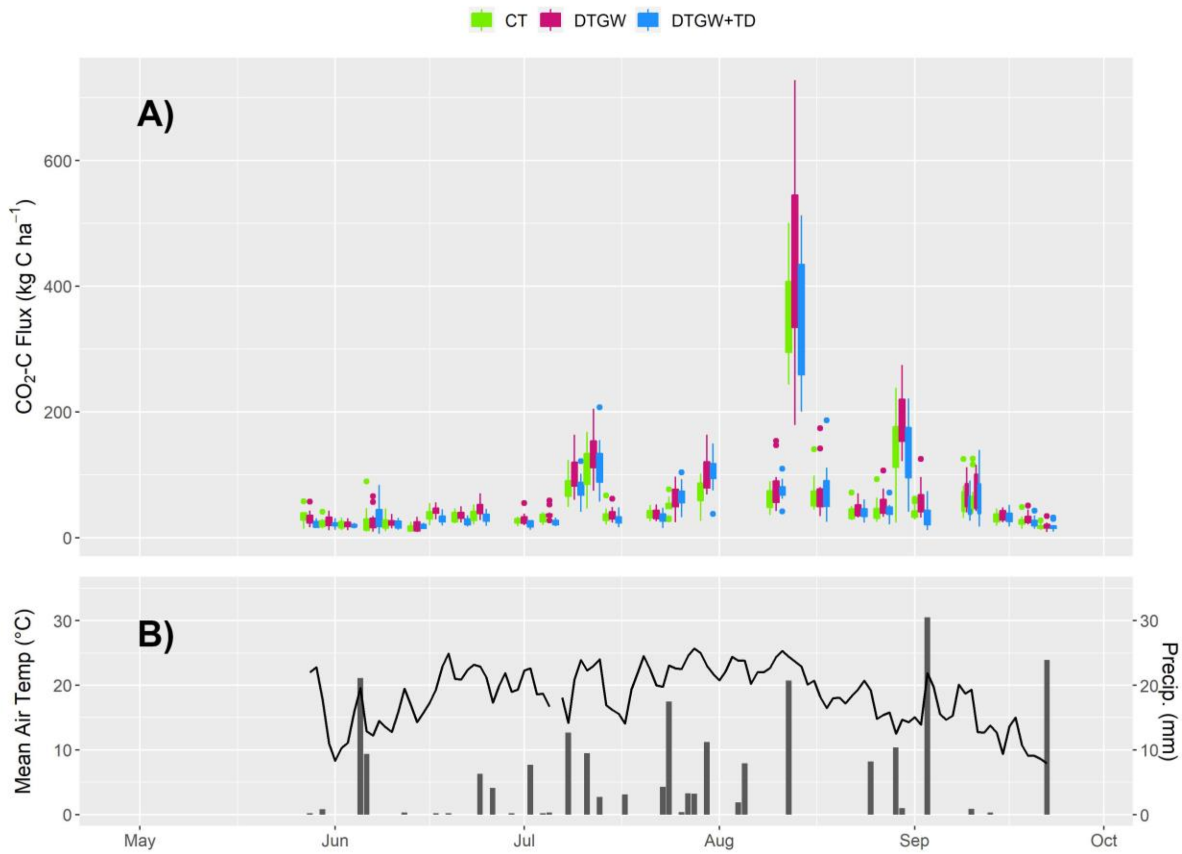

Figure 3-2 A) CO_2 fluxes from May 28th to September 22nd, 2020 in the 3 land management practices: CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage). B) Mean daily temperatures and daily precipitation levels.

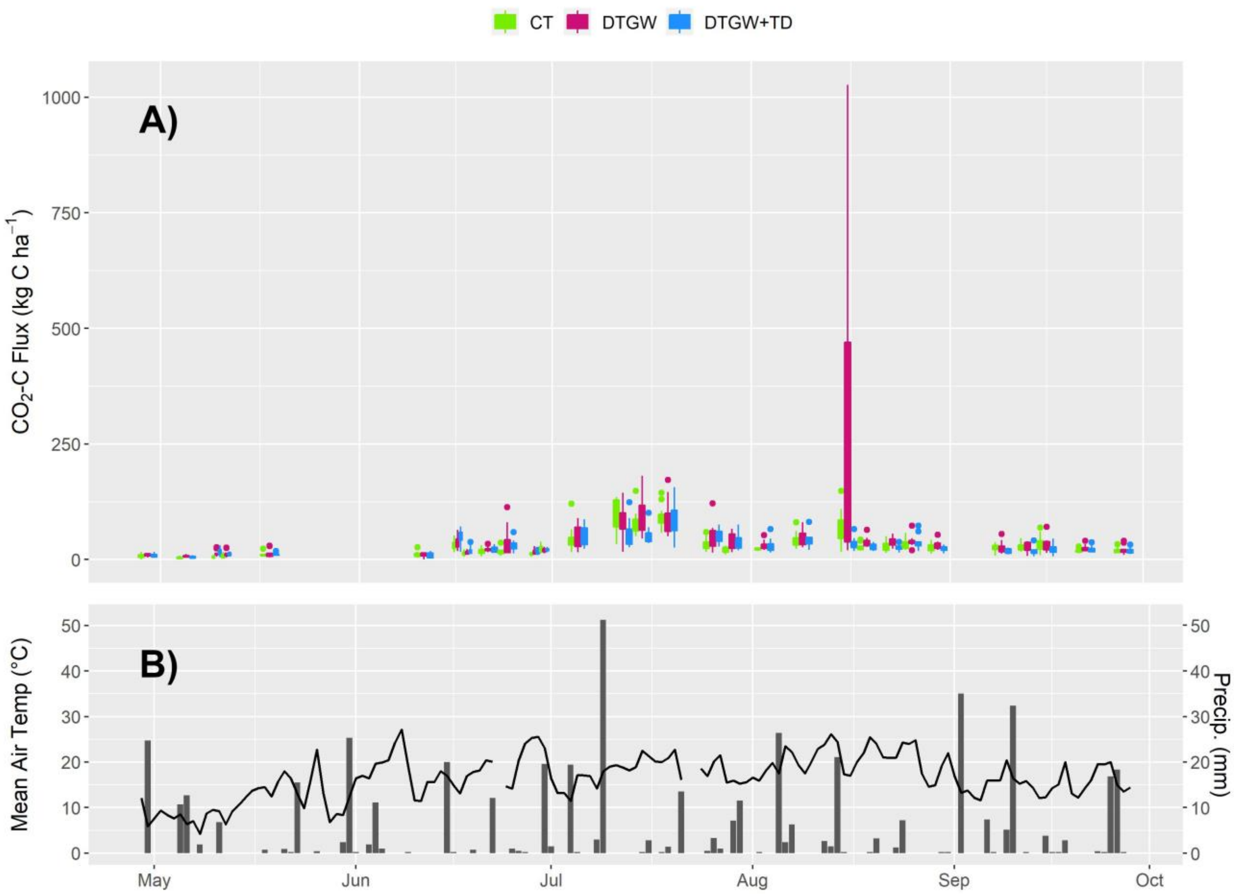

Figure 3-3 A) CO_2 fluxes from April 29th to September 28th, 2021 in the 3 land management practices: CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage). B) Mean daily temperatures and daily precipitation levels.

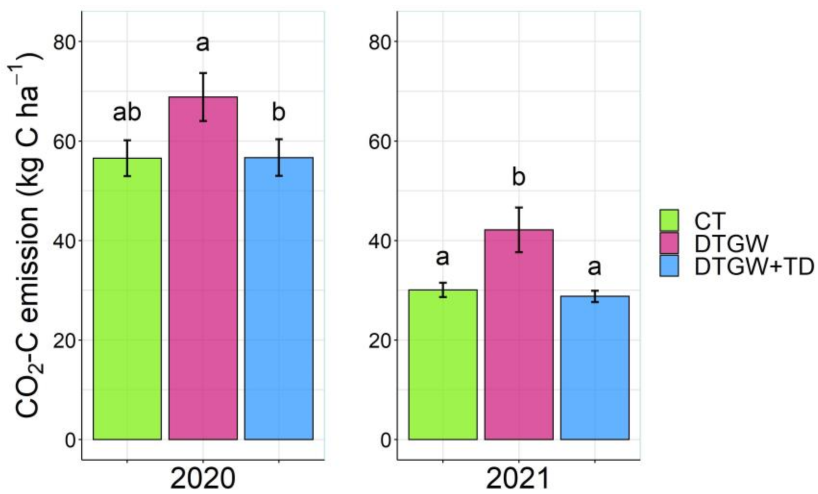

Figure 3-4 Mean daily emissions (kg CO_2-C ha^{-1}) for each LMP – CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage) – in each of the 2020 and 2021 seasons. Error bars represent the standard error of the mean (2020, n = 348; 2021, n = 360). Different letters indicate a statistically significant difference between LMPs using Tukey's HSD (α = 0.05).

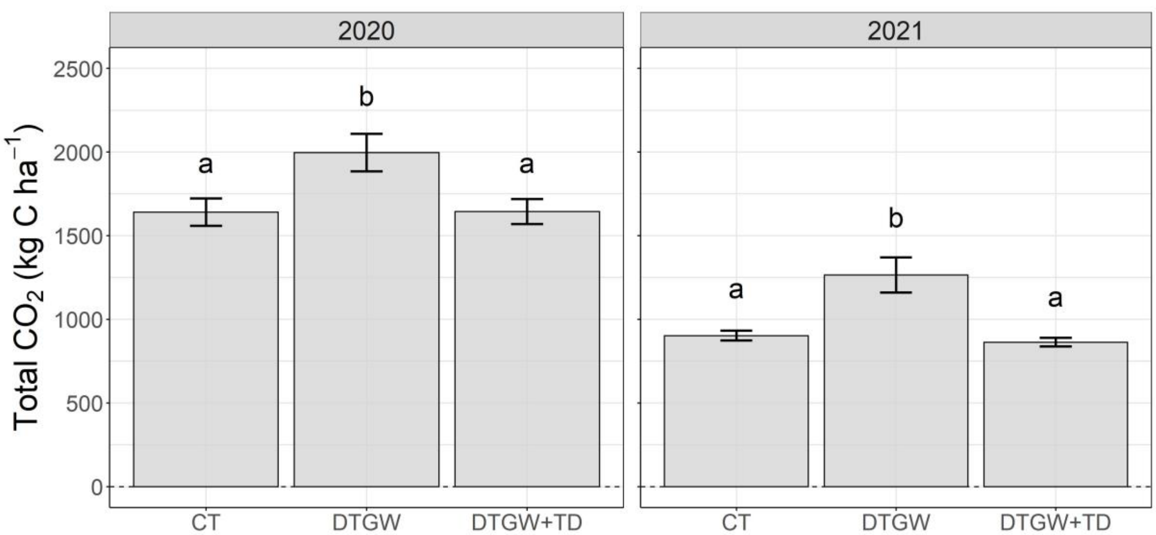

Figure 3-5 Total CO_2 emissions (kg C ha^{-1}) in 2020 and 2021 in each LMP – CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage). Error bars represent the standard error of the mean (n = 12). Different letters indicate a statistically significant difference between LMPs (α = 0.05).

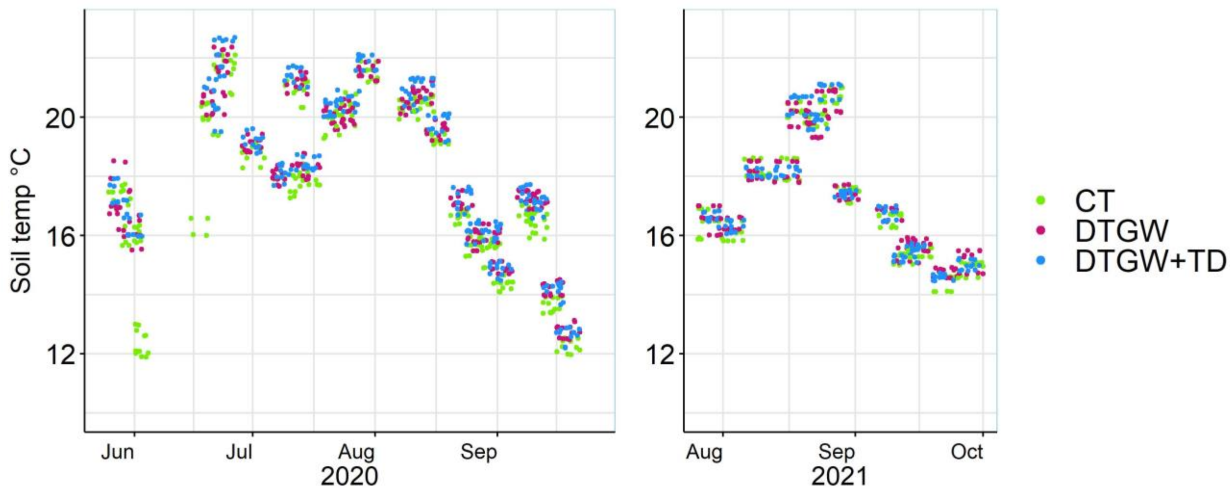

Figure 3-6 Soil temperature data in 2020 and 2021for each LMP – CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage).

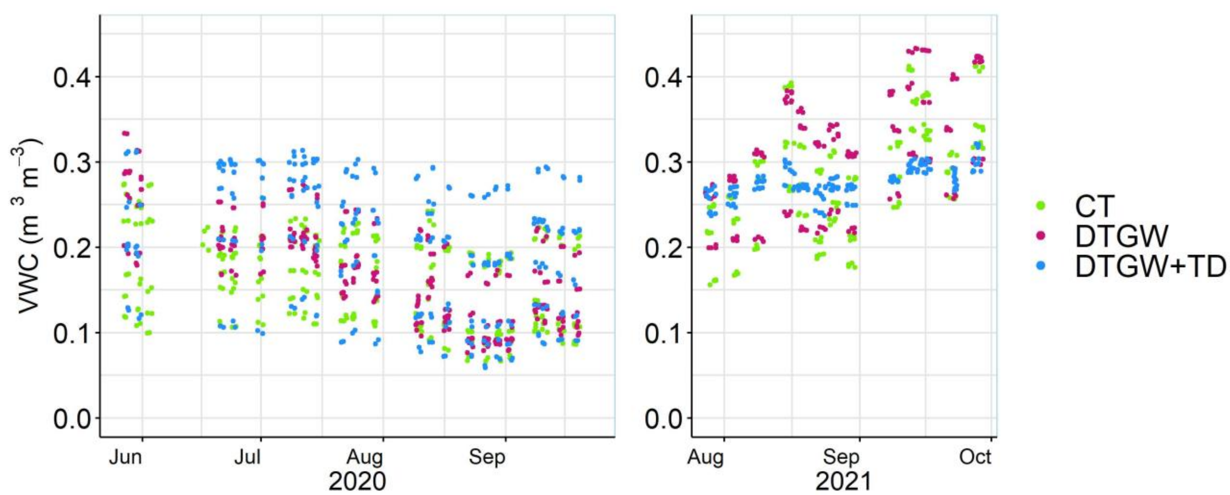

Figure 3-7 Soil volumetric water content data in 2020 and 2021 for each LMP – CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage).

3.3.2 Nitrous Oxide

Land management practice ($p < 0.01$), slope position ($p < 0.05$), and soil moisture ($p < 0.001$) were all significant in the model for N_2O. There was also a significant interaction ($p < 0.01$) between LMP and slope position.

In the VWC model, LMP was not significant ($p = 0.71$). However, slope position was significant ($p < 0.01$), and the lower slope position was significantly different from the mid ($p < 0.01$) and upper ($p < 0.001$) positions. N_2O was positively correlated with soil moisture ($r = 0.23$, $p < 0.001$) across all three seasons but was not, however, significantly correlated with soil temperature ($r = -0.01$, $p = 0.74$).

The N_2O fluxes in DTGW were higher than in CT or DTGW+TD (Figure 3-2). The mean daily efflux (kg N_2O-N ha $^{-1}$) for CT, DTGW, and DTGW+TD across all 3 years was 0.0124 ± 0.002, 0.015 ± 0.0019, and 0.011 ± 0.0016, respectively (Figure 3-3). Mean VWC measurements ($m^3\ m^{-3}$) in CT, DTGW, and DTGW+TD were 0.23 ± 0.01, 0.25 ± 0.01, and 0.24 ± 0.01, respectively.

The only significant difference between LMPs in total CO_2e emission of N_2O was seen in 2019 ($p < 0.001$) (Figure 3-11).

Mean daily effluxes from upper to lower slope positions were 0.0151 ± 0.002, 0.0130 ± 0.002, and 0.0106 ± 0.001 kg N_2O-N ha^{-1} day^{-1}, respectively. The upper slopes were surprisingly the wettest and the lower slopes the driest. Soil VWC averaged 0.26 ± 0.01, 0.25 ± 0.01, and 0.22 ± 0.01 $m^3\ m^{-3}$ top to bottom. The CT emissions increased from top to bottom. In terraced LMPs emissions increased bottom to top.

3.3.3 Methane

Mean CH_4-C oxidation rates across all years for CT, DTGW, and DTGW+TD were 1.17 ± 0.2, 0.42 ± 0.5, and 1.56 ± 0.2 g ha^{-1} day^{-1}, respectively. Methane oxidation rates were positively correlated with soil moisture ($r = 0.26$, $p < 0.001$, $n = 425$) and negatively correlated with soil temperature ($r = -0.24$, $p < 0.001$, $n = 425$). There was a significant interaction between LMP and slope position ($p = 0.048$). Soil temperature and moisture were both significant in the model ($p < 0.01$). There were no significant differences between LMPs seen in 2019 ($p = 0.08$), 2020 ($p = 0.86$) or 2021 ($p = 0.17$) (Figure 3-12).

The highest CH_4 uptake by slope position (-1.6 ± 0.2 g ha^{-1} day^{-1}) occurred in the lower slopes, which were ~16% drier than the upper slopes. This is in line with expectations for the driest soils. The upper and middle slopes oxidized 0.9 ± 0.2 and 0.7 ± 0.5 g ha^{-1} day^{-1}.

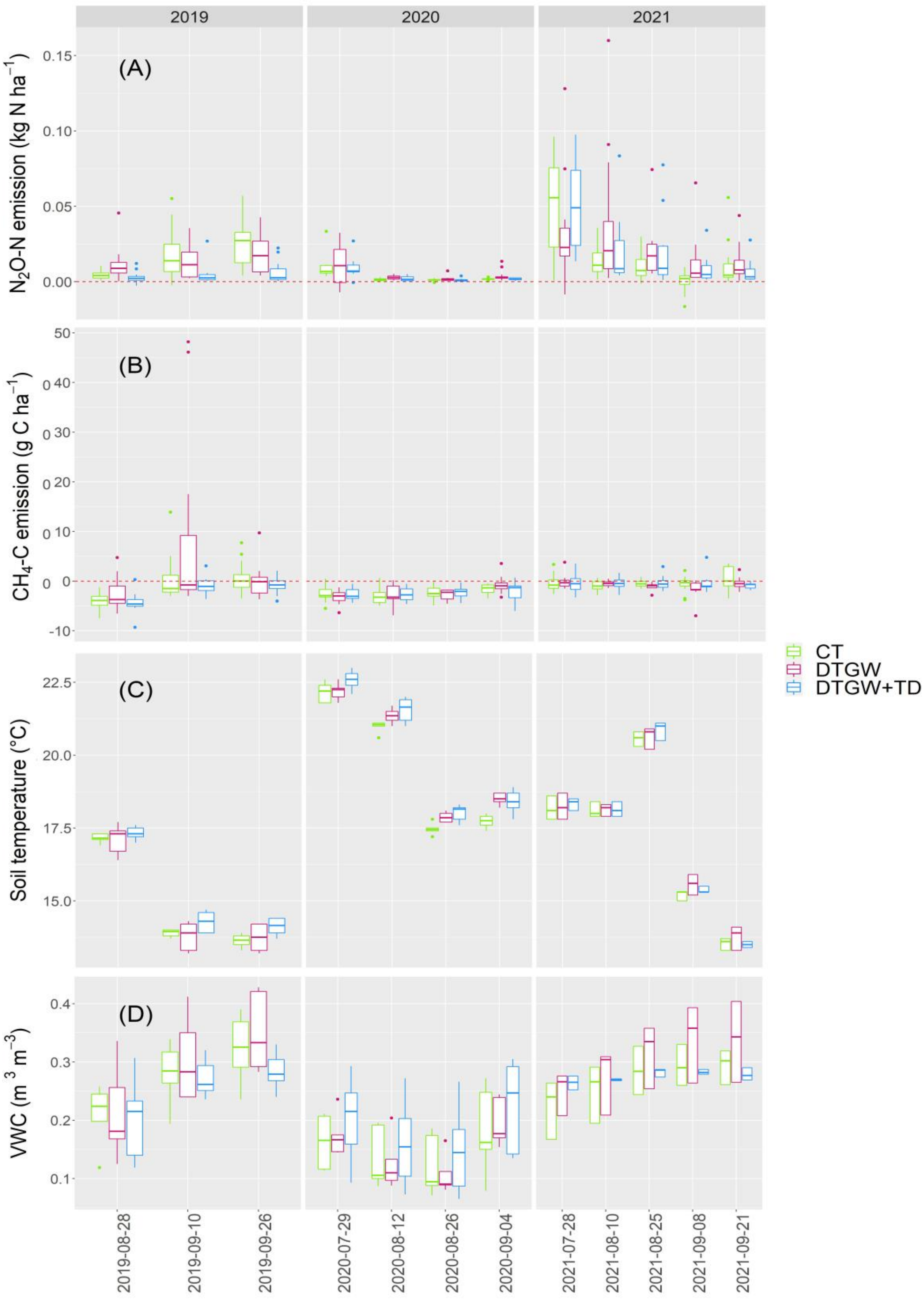
2019
2020
2021
(A)
(B)
(C)
(D)
N_2O-N emission (kg N ha^{-1})
CH_4-C emission (g C ha^{-1})
Soil temperature (°C)
VWC (m^3 m^{-3})
CT
DTGW
DTGW+TD
2019-08-28
2019-09-10
2019-09-26
2020-07-29
2020-08-12
2020-08-26
2020-09-04
2021-07-28
2021-08-10
2021-08-25
2021-09-08
2021-09-21

Figure 3-8 Mean daily emissions of N_2O (A) and CH_4 (B) in 2019, 2020, and 2021, with soil temperature (C) and soil moisture (D), and coloured by LMP – CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage).

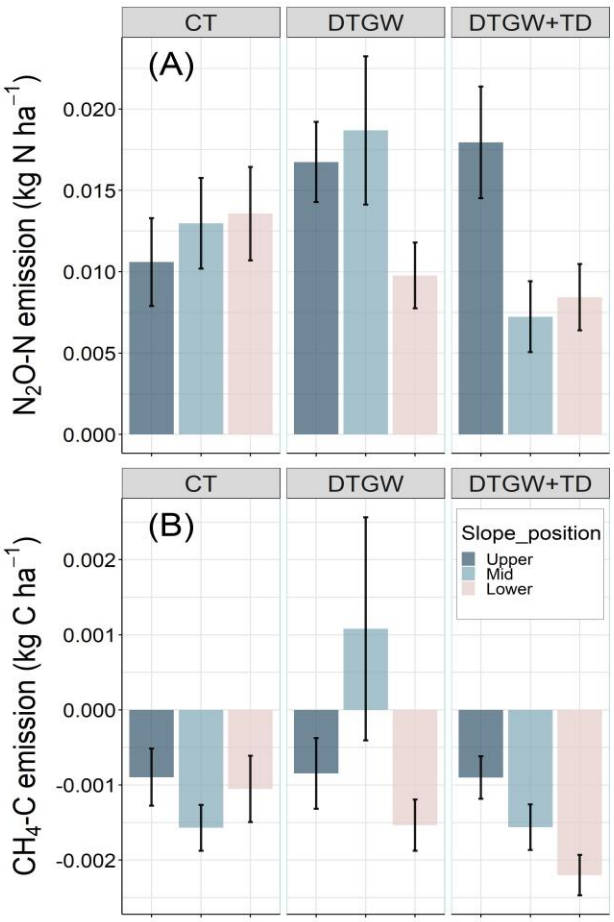

Figure 3-9 Effect of land management practice (LMP) – CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage) – on emission of N_2O (A) and CH_4 (B) across 2019, 2020, and 2021 growing seasons. The bars of the columns represent standard errors of the mean.

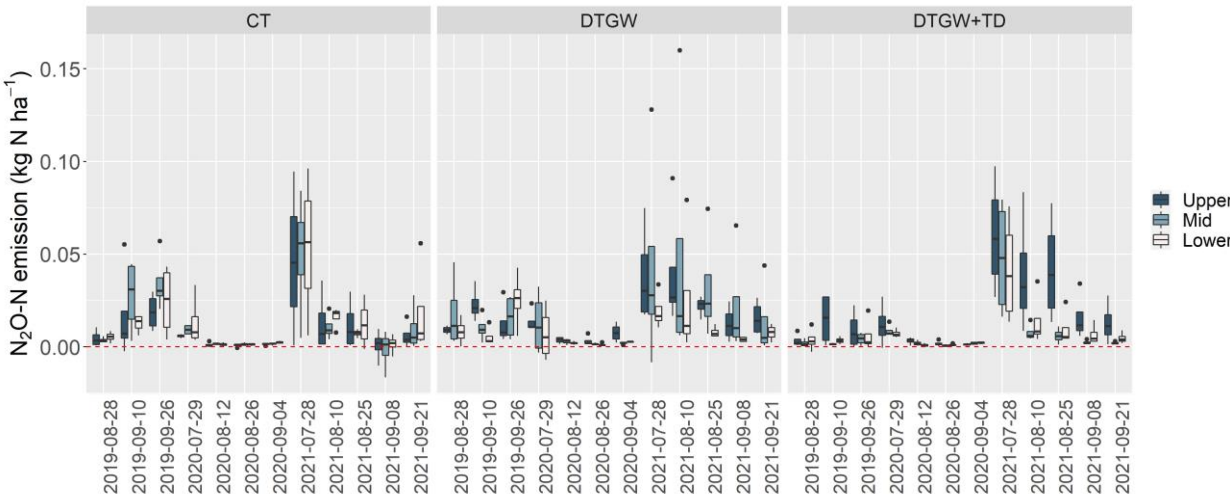

Figure 3-10 Emission of N_2O by slope position in each land management practice – CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage).

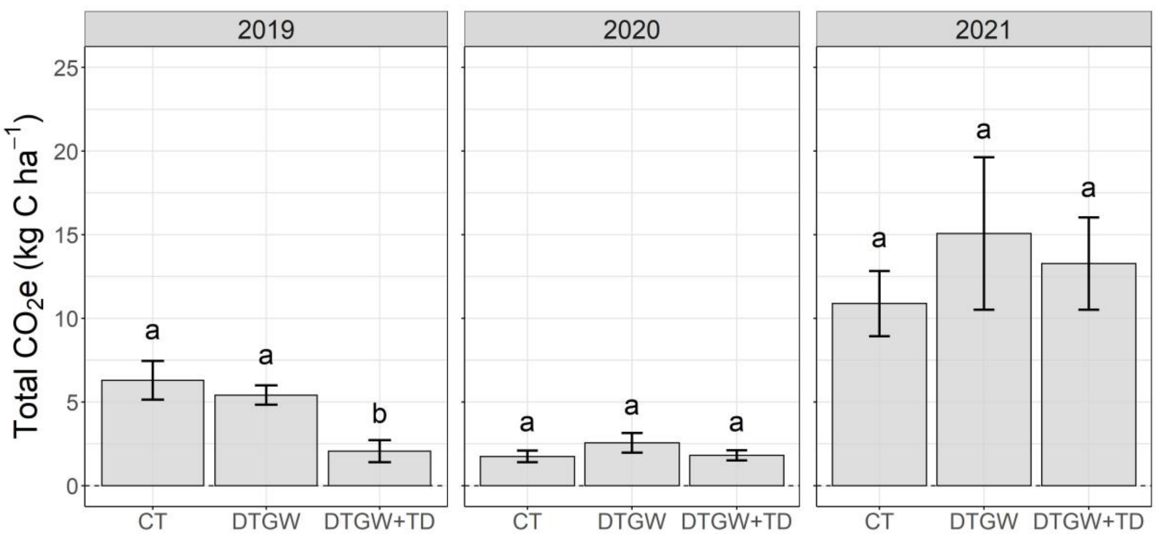

Figure 3-11 Total N_2O emission for the 2019, 2020, and 2021 seasons expressed in CO_2 equivalents (CO_2e) (kg C ha^{-1}) for each LMP – CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage). Error bars represent the standard error of the mean (n = 12). Different letters indicate a statistically significant difference between LMPs ($\alpha = 0.05$), but comparisons can only be made within year.

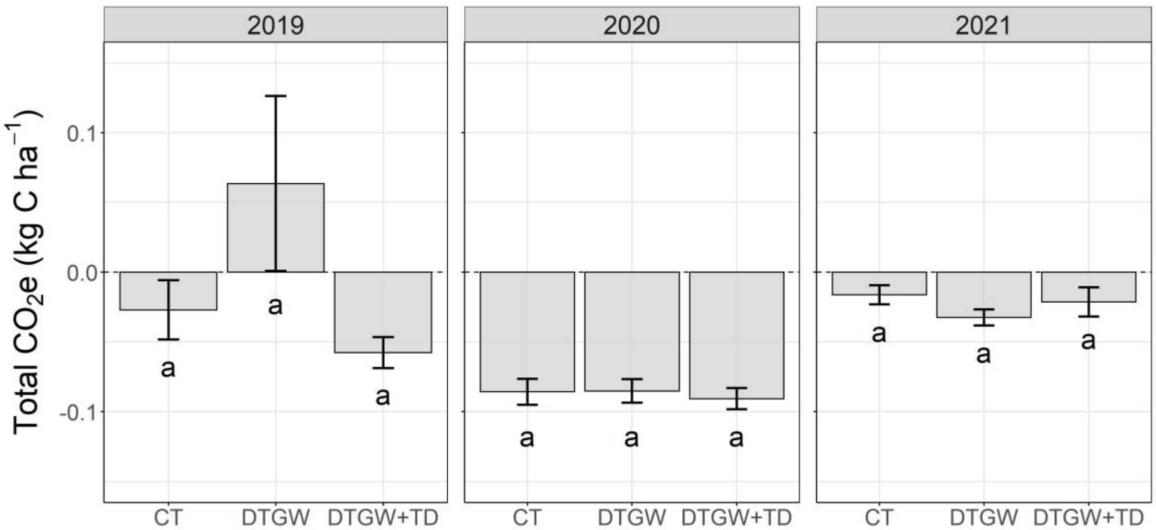

Figure 3-12 Total CH_4 emission for the 2019, 2020, and 2021 seasons expressed in CO_2 equivalents (CO_2e) (kg C ha^{-1}) for each LMP – CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage). Error bars represent the standard error of the mean (n = 12). Different letters indicate a statistically significant difference between LMPs ($\alpha = 0.05$), but comparisons can only be made within year.

3.3.4 Nitrate Supply Rate

In 2020, there was a statistically significant interaction between LMP and slope position ($p = 0.017$). In 2021, not one of LMP (0.65), slope position (0.33), or their interaction (0.41) was significant.

In 2020, the highest supply rates were seen in DTGW between July 2nd and July 28th at 404 ± 50.5 µg/cm^2 (Figure 3-13). Rates in the July 26th to August 16th period of 2021 were 198% higher than between July 28th and August 18th of 2020. Rates in the August 16th to September 7th period of 2021 were 128% higher than between August 19th to September 8th, 2020.

The only significant correlation in 2020 between nitrate supply rate and N_2O fluxes occurred between the burial period of August 19th to September 8th and the N_2O fluxes from September 4th. No nitrate supply rates significantly correlated to N_2O fluxes in 2021.

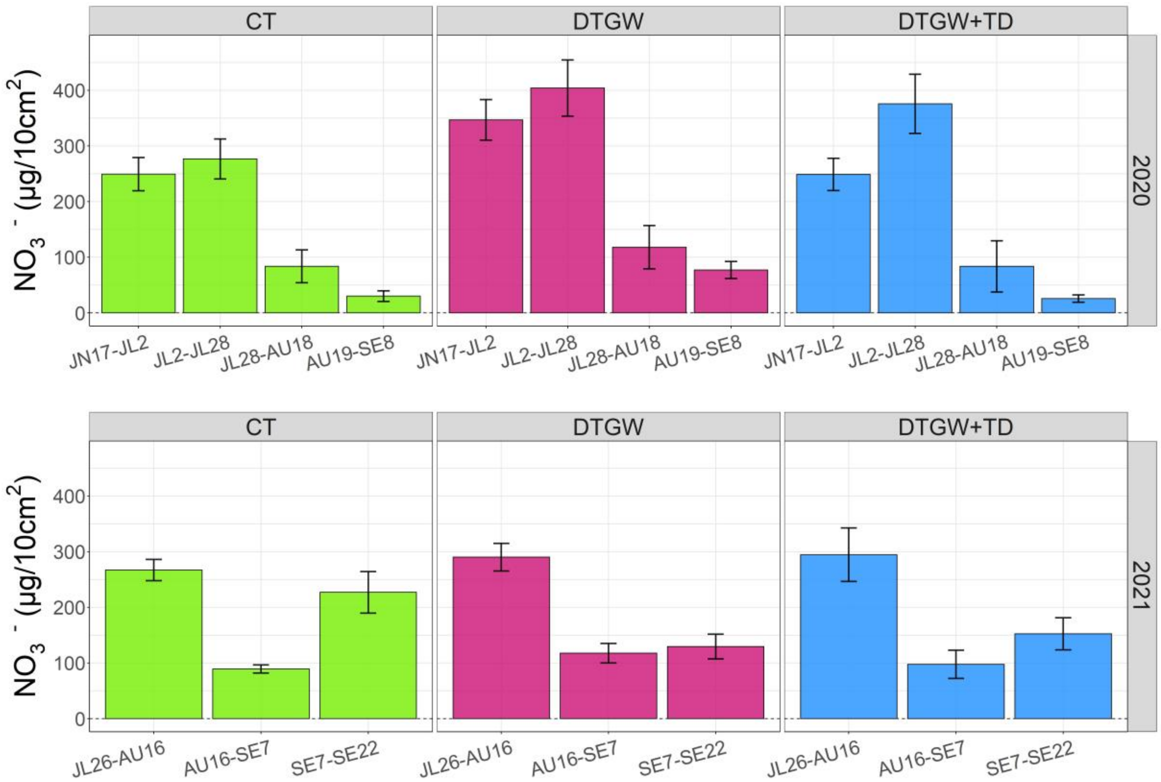

Figure 3-13 Effect of land management practice – CT (contour tillage), DTGW (diversion terraces and grassed waterways), and DTGW+TD (diversion terraces, grassed waterways, and tile drainage) on nitrate supply rates measured in (top) 2020 and (bottom) 2021. The four sampling periods in 2020 were: (1) June 17 – July 12; (2) July 2 – July 28; (3) July 28 – August 18; and (4) August 19 – September 8. The three sampling periods in 2021 were: (1) July 26 – August 16; (2) August 16 – September 7; and (3) September 7 – September 22. Error bars represent the standard error of the mean (n = 12).

3.4 Discussion

3.4.1 Carbon Dioxide

The general emissions trend between years was different than that of N_2O. While N_2O emissions were minimal in 2020 and higher in 2019 and 2021, CO_2 emissions were, on average, higher in the hotter and drier 2020 than in 2021. Rising temperatures increase metabolic rates, leading to increased microbial respiration (Larcher, 2003). Temperature rises can increase CO_2 emissions exponentially up to a certain point, before decreasing due to enzyme denaturation or thermodynamic limitations (Fang & Moncrieff, 2001; Schipper et al., 2014). Temperature likely partially explains why emissions were higher in 2020 than 2021. However, potatoes are associated with lower agricultural emissions relative to other major crops (Haile-Mariam et al., 2008; Jennings et al., 2020) and could also help explain the lowered emissions in 2021.

The frequent soil disturbances caused by tillage events in potato production cause a breakdown of soil aggregates, which aerates the soil and exposes organic matter to microbial decomposition (Grandy et al., 2002). In this study, we saw a decline in aggregate stability over the course of the two seasons. However, there were no statistically significant differences among LMPs and therefore likely does not contribute to explaining why CO_2 emissions in DTGW were significantly highest. Total carbon was higher in DTGW than DTGW+TD, but on par with CT. So, while this could contribute, it is likely not the only explanation. Soil temperature was similar in each of the terraced LMPs in each year (Figure 3-6), but CT had slightly lower soil temperatures in 2020. Soil VWC in 2020 was higher in DTGW+TD than in DTGW, but the results were reversed in 2021 (Figure 3-7). There is no obvious explanation, since the VWC levels reversing in each year is confounding, but the results are likely a combination of soil temperature, soil water, and total carbon.

3.4.2 Nitrous Oxide

The effects of the hot and dry summer of 2020 were evident: soil temperatures were high and soil moisture was low. This resulted in very low N_2O emissions. Conversely, in the 2019 and 2021 seasons, where soil water content was higher, N_2O emissions were higher. The N_2O emissions in this study are in line with emissions from conventional New Brunswick potato farming experiments that took place at the FRDC and ranged between 0 and 0.04 kg N_2O-N ha^{-1} day^{-1} (Burton et al., 2008).

Emissions by LMP did not follow the soil moisture trend. While DTGW had both the highest effluxes and soil moisture, CT had the lowest soil moisture, and DTGW+TD had the lowest average efflux – slightly below the conventional contour tillage and ~26% below DTGW.

The tile drainage appeared to help regulate soil water relative to its terraced counterpart (Figure 3-2). In both 2019 and 2021, soil moisture in DTGW+TD trended below the other LMPs. In 2021, specifically, VWC was nearly flat with no variability for nearly two months, while soil moisture consistently rose in the other LMPs. This water regulation from tile drainage likely aided in lowering N_2O effluxes. Denitrification is the dominant source of N_2O in humid climates such as Atlantic Canada (Mosier, 1998). When soil VWC approaches saturation, denitrification processes start to release more N_2O to the atmosphere under anaerobic conditions (Ball et al., 2008; Ruser et al., 2006). This commonly takes place at a water-filled pore space (WFPS) of 0.7 $m^3\,m^{-3}$ (Bateman & Baggs, 2005) – or roughly a VWC of 0.36, assuming a bulk density of 1.3 $g{\cdot}cm^3$. VWC in DTGW+TD never eclipsed this value, while on 4 occasions VWC in DTGW approached or passed 0.4. The hypothesis was that we would see increased water content in both terraced LMPs, leading to higher N_2O effluxes. But while both LMPs had higher average VWC than CT, DTGW+TD emitted the least amount of N_2O.

Burton et al. (2012) stated that lower landscape positions have greater water contents for greater periods of time, leading to potentially greater denitrification and N_2O emissions. This was in contrast with results from this study – overall emissions were highest in upper slopes and lowest in lower slopes. This may be due to the fact that the hydrology was more complicated than initially thought (S. Li, personal communication, June 6, 2022). There may be water feed into the upper slope areas from above since the location of the site was not completely at the top of the knoll.

3.4.3 Methane

While there was slightly higher variability in 2019, CH_4 fluxes remained mostly negative through all seasons, regardless of soil moisture, resulting in all LMPs acting as C sinks in the growing period. Eastern Canadian soils are typically weak CH_4 sinks (Gregorich et al., 2005).

While the mean VWC values in DTGW and DTGW+TD only differed by ~3%, mean CH_4-C oxidation in DTGW+TD was roughly triple that of DTGW. This was, much like with N_2O, perhaps due to the variation seen in the VWC values in the wetter 2019 and 2021; DTGW+TD values were less variable than in DTGW and also had lower maximum values (Figure 3-2). Since CH_4-C uptake and soil moisture were positively correlated, it could be that the tile drainage – keeping the VWC more constant – was aiding in increasing the CH_4-C oxidation rates. Oxidation rates in the drier 2020 were almost identical in each LMP.

3.4.4 Nitrogen Supply Rate

Land management practice and slope position only had an effect on nitrate supply rates in 2020 and not 2021. There were two main differences between the growing seasons. First, 2020 was much hotter and drier than 2021 (Figure 3-2; 3-3). The crops were also different, with oats seeded in 2020 and potato seeded in 2021. Research has shown that soil ion fluxes measured with PRS-probes often decline with declining soil moisture (Qian & Schoenau, 2002). This aligns with the higher fluxes seen in the wetter 2021, but higher nitrogen (N) fertilization rates likely contributed as well. The N fertilization rate in 2021 after potato planting was 233% higher than after 2020 oat planting. This does not explain why there was no management effect in 2021.

It could be that since probes were placed in potato hills in 2021 that there was no effect seen. The potato rows might be more homogenous across LMPs since they are raised up above the furrow and the normal ground height. The differences in soil water between LMPs may not be as pronounced within the rows as they are 15 cm belowground where VWC was measured. The one curious result was in 2021 when, in the final burial period, the supply rate jumped considerably in CT but less so in the terraced LMPs. This does not align with the soil moisture trend, where DTGW was wettest and rates there should have risen the most if the rise was caused by an increase in soil moisture from rainfall.

Rates were initially high in 2020 and continued for six weeks, very likely due to the fertilizing that took place two days prior to the first installation of probes. The hot and dry summer was likely the source of the large drop in supply rate after July 28th, 2020. Supply rates from the first burial period in 2021 were nearly as high as from the first period of 2020, despite beginning 49 days after potato planting and fertilization and 39 days later relative to the first burial period in 2020. The N fertilization rates are high in potato farming, and the 190.5 kg N ha^{-1} used on June 7th likely contributed to sustained nitrate supply through June and July. Higher supply rates were seen in 2021 in burial periods corresponding to similar dates in 2020.

3.5 Conclusions

DTGW+TD did not emit significantly more CO_2 in either year than CT – which served as the control LMP – despite soil temperature and moisture being slightly higher in 2020. DTGW+TD also emitted the least amount of N_2O on average and consumed the most CH_4 through the 3 summer seasons. In 2020, CT and DTGW+TD N_2O emissions were similar. But in 2021, where DTGW+TD had visibly more regulated VWC, emissions were even lower than in CT. DTGW emitted significantly more CO_2 in each year, despite all three LMPs having similar soil temperature and moisture in 2021.

The findings from this study show that soil moisture in DTGW+TD was more regulated than in CT or DTGW. In 2020, when the summer was hot and dry, tile drained plots surprisingly had some of the highest VWC values. In 2021, when the field received more regular rainfall, DTGW+TD had the least amount of variability, and VWC did not trend upwards through the summer as much as in the other two LMPs. In 2019, 2020, and 2021, VWC in DTGW+TD was capped at roughly 0.3 m^3 m^{-3}, while the other two LMPs had values approaching 0.45 m^3 m^{-3}. Soil temperatures in 2020 were minimally cooler in CT, while in 2021 they were similar across all LMPs.

The hypothesis was that because of increased soil moisture from rainwater being directed downward into the soil and creating wetter soil environments, all GHG emissions would be elevated in the terraced LMPs. This was not the case, as DTGW+TD emitted less N_2O and CO_2, and consumed more CH_4, relative to DTGW. DTGW+TD also emitted less N_2O and consumed more CH_4 than CT. Carbon dioxide levels were on par between CT and DTGW+TD.

Chapter 4: Combined Effects of Nitrogen, Temperature, and Soil Water Content on GHG Emissions in a Short-Term Incubation Experiment

4.1 Introduction

The fundamental idea behind the LISWC system is to integrate several BMPs together such that they complement each other; the drawbacks of one BMP are offset by the benefits of another (Li, 2021). By implementing diversion terraces and grassed waterways, it essentially forces the usage of tillage conducted along the field contour. Up-and-down-slope oriented tillage becomes inefficient when the field segments are considerably wider than they are from top to bottom. The combination of diversion terraces, grassed waterways, and contour tillage reduces water erosion, but it can also lead to excessive amounts of water in the soil. This is then offset by the addition of tile drainage, which quickly removes excess water.

The pattern of greenhouse gas (GHG) emissions was anticipated to change based on the altered water regimes (and subsequent soil temperature changes) in the three land management practices (i.e., contour tillage; diversion terraces and grassed waterways; and diversion terraces, grassed waterways, and tile drainage). Soil water content and temperature are the two most influential parameters affecting soil gas emissions (Oertel et al., 2016; Raich & Schlesinger, 1992; Vicca et al., 2014). To study the effects of altered water regimes on greenhouse gas fluxes, the system must be experiencing regular rainfall. The summer of 2019, winter of 2019/2020, and summer of 2020 were all abnormally dry. The thought at the end of the 2020 growing season was that it might be difficult to detect differences with the *in-situ* field conditions; hence, a decision was made to complement the field seasons with an incubation experiment, in case the 2021 season mimicked the same hot and dry conditions. The objective of this study was to quantify the

GHG fluxes in response to volumetric water content (20%, 27%, 35%, 40% VWC), nitrogen fertilizer application rates (0, 170 kg N ha^{-1}), and temperature (10°C, 25°C) with a 6-day soil incubation experiment.

4.2 Materials and Methods

4.2.1 Experimental Design

In October 2020, composite bulk samples of a sandy loam texture were collected from a study site located at the Fredericton Research and Development Centre, with an average 55.3% sand, 34.5% silt, and 10.2% clay (Appendix B). The soils were taken using a Dutch auger (0-15 cm depth) from a northeast facing slope that faced the Saint John River (45°55'33.5"N 66°36'55.5"W). The soil was sieved through an 8-mm sieve to keep larger aggregates intact and remove large, organic debris or coarse fragments and was stored at 4°C until incubation was ready.

The incubation experiment was set up as a randomized complete block design. There was a total of 8 experimental units. Volumetric water content (VWC) had 4 factor levels: 20%, 27%, 35%, and 40%; equivalent to 15.5%, 20.8%, 27%, and 30.8% gravimetric water content (GWC) at an average bulk density of 1.3 g cm^{-3}. Nitrogen fertilizer had 2 levels: 0 and 170 kg N ha^{-1}. The VWC levels were chosen to represent the range of soil moisture levels expected in an average growing season in central New Brunswick, as well as to examine higher moisture levels (40% VWC) that are less common. The N levels were chosen to represent a common fertilizer application rate used for potatoes in the region (Government of New Brunswick, n.d. -d). The two temperatures, 10°C and 25°C, were chosen to represent shoulder season temperatures and growing season temperatures, respectively.

The experiment was replicated 9 times for each temperature; all replicates for each separate temperature were run at the same time. Therefore, there were a total of 8 experimental units per replicate, and a total of 72 experimental units per temperature.

Microcosm preparation

The experiment used a method that assesses heterotrophic respiration using soil microcosm cores (Comeau et al., 2018). Each microcosm and filter paper, which covered the drainage holes, was weighed separately. Each microcosm was packed with field-moist soil to a bulk density of 1.3 $g{\cdot}cm^{-3}$, based on the respective gravimetric water contents. The microcosms were packed in two stages – half the soil mass was packed up to the 2.5 cm mark, followed by the remaining half, up to the 5 cm mark. This ensured optimized bulk density uniformity.

Each microcosm required either removal or addition of water to reach the desired VWC level. The microcosms which required drying were dried in the oven (max 30°C) until they reached the desired mass. All microcosms were dried a further 0.5 g to account for the future addition of either 0.5 mL water or 0.5 mL of N solution. Dry filter masses were recorded for all experimental units, whereas a wet filter mass correction was made uniformly across all 72 final microcosm masses based on an average of 5 wet filters. This was to balance out the water which will leave the soil and enter the dry filter.

Pre-incubation

To avoid a rewetting burst of greenhouse gases, soil moisture and temperature were established in advance. The samples were pre-incubated in a growth chamber (Conviron CMP 6050) at their respective temperatures for 3 days to allow soils to equilibrate. The microcosms were covered in parafilm for this time to prevent water evaporation. At the end of the pre-incubation, if any microcosm masses were significantly lowered, a readjustment of water content levels occurred, and any sprouted seedlings were removed.

Nitrogen addition

Nitrogen solution was added to half the microcosms after the pre-incubation was performed. A nitrate-based 20-20-20 NPK fertilizer (TerraLink Horticulture Inc., Abbotsford, BC, Canada) was dissolved in distilled water to a concentration of 38.56 mg $N \cdot mL^{-1}$. A total of 0.5 mL solution was added, which delivered the 19.28 mg N required. Each microcosm, regardless of treatment, was then adjusted individually to reach target soil moisture level. Microcosms were placed individually in hermetically sealed 2.9 L plastic containers. The containers were randomly located within the growth chamber.

4.2.2 Sampling and analysis

Soil GHG flux was measured 4 times over the course of the incubation, at time 0, 2, 4, and 6 days. Before the time 0 measurement, the 0.5 mL solutions of distilled water or fertilizer solution were added to respective treatments. Containers were flushed prior to the time 0 measurement to remove initial bursts of gases. 40 mL of ambient air was added immediately after flushing to balance out removing 20 mL of gas during each of the 4 samplings. When sampling, headspace mixing took place for 15 s to generate air flow and prevent stratification. After all samples were completed on a given day, their locations were randomized within the growth chamber again. All gas samples were analyzed at the University of British Columbia using a 7890A gas chromatograph (Agilent Technologies Inc., CA, USA) equipped with a flame ionization detector and electron capture detector and a PAL auto-sampler (Agilent Technologies Inc., CA, USA). The Ideal Gas Law was used to determine fluxes (Lang et al. 2011). The greenhouse warming potentials (GWPs) of 298 for N_2O and 25 for CH_4 on a 100-year time scale were used to convert emissions to CO_2 equivalents (IPCC, 2019).

4.2.3 Statistical Analysis

The differences in GHG emissions and CO_2 equivalents between treatments were analyzed using ANOVA and Tukey post-hoc tests. A linear mixed-effects model was used and included VWC and N fertilization rate as fixed effects. Block was included as a random effect. Type III ANOVA was used to assess significance of the model interaction terms. A Tukey post-hoc test using the *emmeans* function (Lenth, 2022) tested for differences in VWC and N fertilization rate. Analyses were conducted for each temperature separately. The normality and variance homogeneity assumptions of ANOVA were assessed using Shapiro-Wilks test and Bartlett's test, respectively, in addition to diagnostic plots. A statistical significance threshold of $\alpha = 0.05$ was used for hypothesis testing. Statistical analyses were conducted using R software version 4.0.3 (R Core Team 2020). Transformations were made on N_2O fluxes to meet model assumptions.

4.3 Results

Carbon dioxide

Only N fertilization rate had significant effects on CO_2 fluxes at each temperature (Figure 4-1). There was no interaction between VWC and N fertilizer rate at either temperature. No significant differences were observed between VWC levels at 10°C. At 25°C, there were significant differences between: 20% and 27%; 20% and 35%; 20% and 40%; and 27% and 40% VWC (Table 4-1). Nitrogen fertilizer levels significantly differed at each temperature ($p < 0.001$). At 25°C, the increase in VWC linearly increased CO_2 fluxes up to 34%, after which it plateaued.

Nitrous oxide

For N_2O, there were significant interactions between VWC and N fertilization rate at both temperatures ($p < 0.001$). There were significant ($p < 0.001$) differences in N_2O fluxes between fertilized and non-fertilized treatments in 40% VWC treatments at 10°C and 27%, 35%, and 40% VWC treatments at 25°C (Figure 4-2). All VWC treatments below 40% with 0 N fertilizer had slightly negative fluxes, regardless of temperature. The only 170 kg N ha^{-1} treatment that had negative fluxes was at 20% VWC and 10°C. The combination of 40% VWC and fertilizer addition resulted in the largest positive fluxes at each temperature.

Methane

For CH_4, at 10°C, there was moderate evidence of a significant interaction between VWC and N fertilization rate ($p = 0.037$). However, no treatments significantly differed (Figure 4-3). At 25°C, there was no significant VWC × N fertilizer rate interaction, VWC was not significant, and N fertilizer rate significance was suggestive but not conclusive ($p = 0.08$). All CH_4 mean fluxes were negative, regardless of temperature or treatment.

CO_2 equivalents

When comparing CO_2 equivalents of all three gases, there was no significant interaction ($p > 0.4$) between VWC and N fertilization rate at either temperature (Figure 4-4). Neither fertilization rate ($p > 0.12$) nor VWC ($p > 0.3$) were significant at either temperature. Though not statistically significant, the CO_2 equivalents emitted at 25°C and 40% VWC were 199% higher than the next highest emitting treatment (25°C and 35%).

Table 4-1 Pairwise comparisons of volumetric water content (VWC) groups, averaged over N fertilization level, using Tukey adjustment.

	10°C			25°C		
Contrast	df (degrees of freedom)	t	p-value	df	t	p-value
20-27	56	0.42	0.975	56	-2.62	0.053
20-35	56	-1.85	0.264	56	-4.18	<0.001
20-40	56	-1.79	0.289	56	-5.59	<.0001
27-35	56	-1.92	0.233	56	-1.97	0.213
27-40	56	-1.88	0.250	56	-2.68	0.046
35-40	56	0.01	1.000	56	0.02	1.000

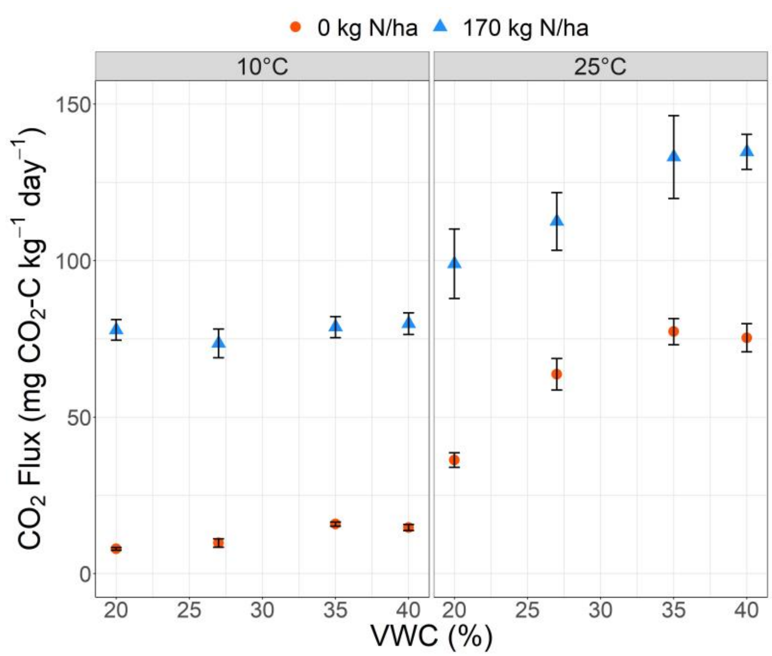

Figure 4-1 Mean CO_2 flux calculated at the end of the 6-day incubation experiment. Error bars represent the standard error of the mean (n = 9).

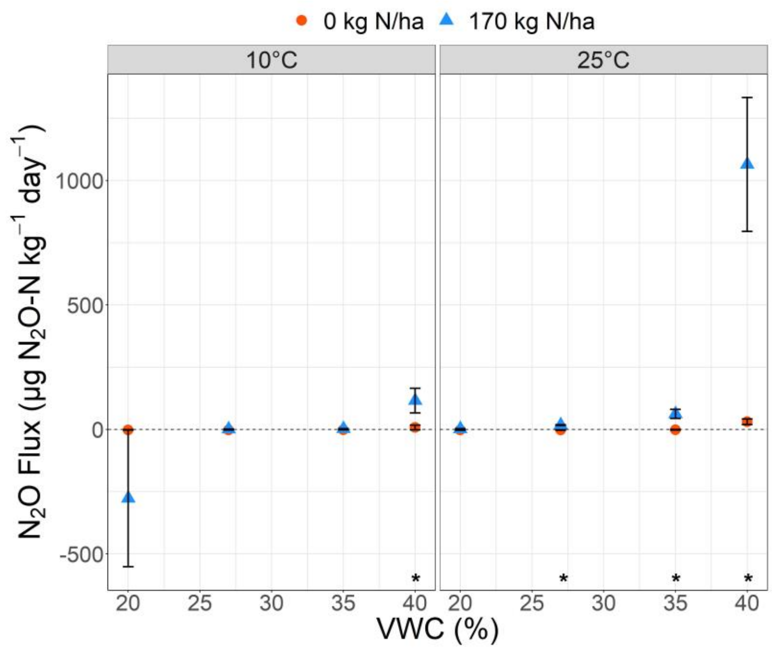

Figure 4-2 Mean N_2O flux calculated at the end of the 6-day incubation experiment. Error bars represent the standard error of the mean (n = 9). For each volumetric water content (VWC) within a temperature, asterisks denote a significant difference between nitrogen fertilizer rate × VWC treatments (α = 0.05).

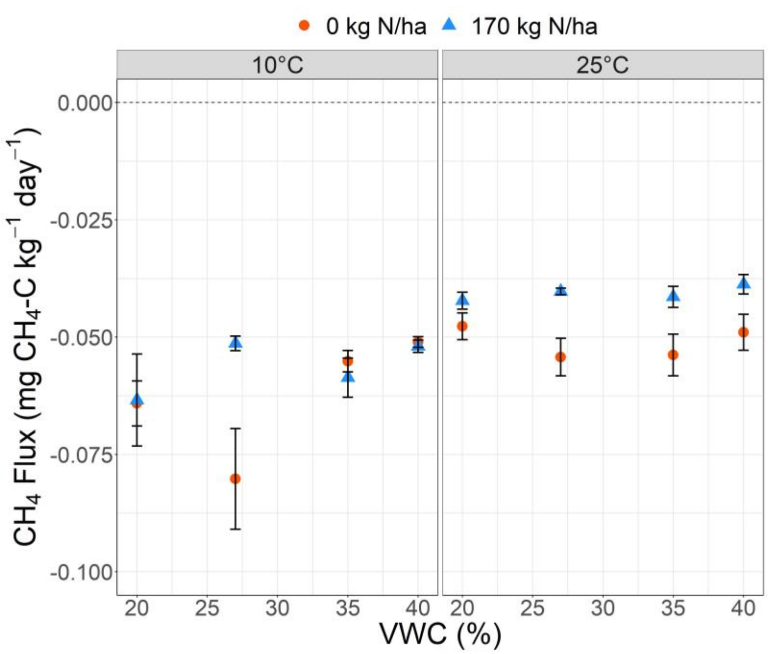

Figure 4-3 Mean CH_4 flux calculated at the end of the 6-day incubation experiment. Error bars represent the standard error of the mean (n = 9).

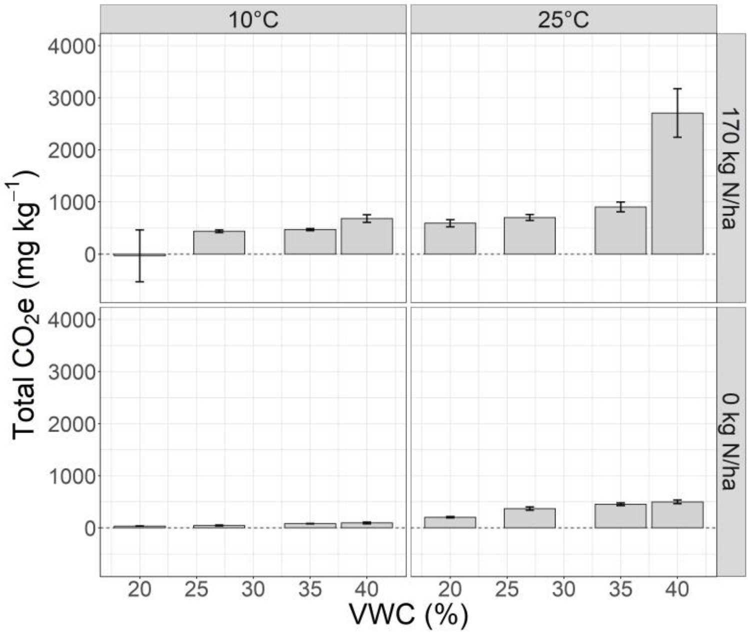

Figure 4-4 Total CO_2 equivalent emissions over the 6-day incubation experiment. Error bars represent the standard error of the mean (n = 9).

4.4 Discussion

Carbon dioxide

Volumetric water content did not have an effect on CO_2 fluxes at 10°C, where temperature clearly had more of an effect on soil respiration than soil water content. It appears that soil respiration was not sensitive to changes in VWC at this temperature. This suggests that in shoulder seasons there may be little change to soil CO_2 fluxes in between 20 and 40% VWC levels. However, CO_2 flux could drop near saturation, where VWC would approach 50.7% (100% WFPS), or below 20%, where respiration would be water-limited.

Water content did have a significant effect on fluxes at 25°C; as VWC increased, CO_2 fluxes rose steadily up to 35% and peaked. It is possible fluxes would drop at a higher VWC than 40%. A study of an Ontario soil with similar soil particle distribution found fluxes to be maximal at 65% WFPS (Fairbairn, 2020) – equivalent to 33% VWC in our soils. In my study, fluxes did not decline significantly between 35 and 40%. It can be very near saturation that some soils finally begin to show a decline in CO_2 fluxes. An incubation study on a fine loamy soil in Germany found that CO_2 emissions only became restricted past 98% WFPS (Ruser et al., 2006). Our soil was a well-aggregated sandy loam with 55% sand. In the field, this soil would have difficulty reaching VWC values greater than 40% in summer temperatures, especially on sloping land, as observed in 2021.

The addition of N fertilizer had a significant effect on CO_2 fluxes at both temperatures. Nitrogen addition raised CO_2 fluxes by approximately 422% at 10°C and 55% at 25°C, on average, across all treatments. Nitrogen addition raised CO_2 fluxes by an average 892% between temperatures. Increases in soil N content will generally lead to increases in soil respiration, as long as C is not a limiting factor (Niu et al., 2010; Peng et al., 2011). This N addition can lead to higher soil respiration sensitivity toward soil moisture and to lowered sensitivity toward temperature (Peng et al., 2011). This holds true at 25°C; however, at 10°C there was no sensitivity toward VWC increasing.

Nitrous Oxide

Three quarters of the N_2O fluxes at 0 kg N ha^{-1} were slightly negative. Only 40% VWC treatments were positive, where fluxes increased from 7.6 ± 9.6 µg N_2O-N kg^{-1} day^{-1} without N addition to 30.7 ± 11.2 µg N_2O-N kg^{-1} day^{-1} with N addition. Negative N_2O fluxes are uncommon and often research centers on net N_2O production (Chapuis-Lardy et al., 2007). However, there is a wide range of evidence of net negative fluxes measured in various conditions from temperate to tropical areas and from natural to agricultural systems (Chapuis-Lardy et al., 2007). For 20, 27, and 35% VWC levels at 0 kg N ha^{-1} and both temperatures, the slightly negative fluxes were likely a result of the 6-day incubation length. The N_2O concentration inside the container headspace dropped in most treatment combinations from day 0 to day 2, before steadily increasing upward. For these six specific treatments, concentrations were approaching the starting concentration at day 6, and, given a slightly longer incubation, would have flipped positive. The fertilized treatment at 10°C and 20% VWC, however, consumed substantially more N_2O in the 6 days. The same concentration pattern occurred in this treatment, but it would have taken considerably longer than 6 days to flip positive. The initial reduction in N_2O concentration between day 0 and 2 could have been caused by a number of factors. Consumption of N_2O is often due to low mineral N availability or large moisture contents, but there are other factors such as nitrifier denitrification, aerobic denitrification, soil pH, and O_2 content (Chapuis-Lardy et al., 2007).

The greatest N_2O emissions were observed in the treatment with highest temperature, water content, and N fertilization rate. Fluxes reached 1065 ± 269 μg N_2O-N kg^{-1} day^{-1}. The dominant N_2O production process in humid climates is denitrification (Mosier, 1998), so it was not surprising to see highest emissions in near-saturated soil conditions of both temperatures.

Methane

All treatments led to uptake of methane, and no CH_4 emissions were observed. Methane in soils is produced by methanotrophic microorganisms under strictly anaerobic conditions (Dutaur & Verchot, 2007; Topp & Pattey, 1997). Even at the maximum 40% VWC studied, no CH_4 was emitted. I had hypothesized that we would see emissions beginning at 40%. At 25°C there was suggestive evidence of lowered uptake of CH_4 in the presence of N fertilizer. Addition of inorganic N fertilizers has the ability to reduce CH_4 uptake or increase CH_4 emission, and this could explain the potential increase in fertilized treatments (Bodelier & Laanbroek, 2004).

CO_2 equivalents

Carbon dioxide was primarily responsible for total GHG emissions in CO_2 equivalent units when total emissions were positive in a treatment over the 6 day period. Approximately 75% of the total equivalent in the treatment consisting of 170 kg N ha^{-1} and 40% VWC at 25°C was due to N_2O emissions. The one negative occurrence of total CO_2 equivalents occurred in the 20% VWC and 0 kg N ha^{-1} treatment at 10°C; the negative totals from N_2O and CH_4 slightly offset the positive CO_2 emissions. Overall, CO_2 equivalents generally rose with increasing soil water content.

4.5 Conclusions

The 6-day soil incubation under different VWC, N fertilizer levels, and temperatures had varying effects on GHG emissions. Only at 25°C did VWC have a significant effect on CO_2 fluxes, while N addition had a significant effect at both temperatures. The addition of N had a significant effect on N_2O fluxes. At the lowest VWC and temperature, there was a net uptake in N_2O over the 6 days. At each temperature, the treatments of 40% VWC and N addition resulted in the highest fluxes. There was suggestive evidence that N addition reduced CH_4 uptake at 25°C. Otherwise, there were no significant treatment effects on CH_4 fluxes.

There is interest in the results that correspond to periods where field measurements were not able to be collected – that is, in the early spring and later in the fall where soil is likely to be wetter due to reduced evaporation. This is potentially a critical period for N_2O and CH_4, when saturated soil, and consequently anoxic conditions, can produce higher emissions. In the 10°C incubation, only N_2O showed higher sensitivity toward water content. There is no evidence that at cooler soil temperatures, CO_2 or CH_4 emissions would rise significantly. Since incubation studies are uncommon at 10°C, results from this incubation study provide insights into how temperature, soil water content, and nitrogen fertilizer affect GHG emissions from typical agricultural soil in Atlantic Canada.

Chapter 5: General Conclusions and Recommendations for Future Research

To address soil erosion and soil health concerns, as well as to address the uncertainties of temperature and precipitation pattern changes in the wake of a changing climate, the LISWC system has been proposed in the last five years as a new paradigm for soil and water conservation in agricultural landscapes. The LISWC system combines diversion terraces, grassed waterways, tile drainage, water retention structures, supplemental irrigation, conservation tillage practices, and soil-landscape restoration, with an overarching goal of integrating these BMPs to obtain all the benefits while minimizing the drawbacks. The objective of my thesis study was to evaluate the effects of terraces, grassed waterways, and tile drainage relative to contour tillage on various soil quality indicators and greenhouse gas (GHG) emissions during the initial years of management implementation. I also conducted a short-term incubation experiment to quantify GHG fluxes in response to volumetric water content, nitrogen fertilizer application rates, and temperature. Diversion terraces should have an effect on soil water content, thereby affecting GHG emissions, but a continuous range of soil water content is difficult to produce in the field and therefore justifies an incubation study.

5.1 Soil Aggregate Stability and Labile Carbon Response to Second and Third Years of Contour Tillage, Diversion Terrace, Grassed Waterway, and Tile Drainage Implementation

There was no statistically significant change in total soil organic carbon between the second and third years since management implementation. No change was expected, as total carbon typically changes slowly. This justifies the use of indicators that are more sensitive to changes in land management that may show early changes in the soil ecosystem.

Aggregate stability declined between post-planting 2020 and post-harvest 2021 in all three LMPs, but there was no significant difference between the three LMPs. However, DTGW+TD saw much more variability than the other two, and this was the only LMP showing any evidence of increasing aggregate stability. The increases in labile C in both DTGW and DTGW+TD should improve the physical structure of the soil and improve aggregation. It is possible that there hasn't been enough time since the initial construction of the study site to see any improvements in aggregate stability.

Land management practice had a significant effect on POXC. The contour tillage saw a decrease in POXC over the study period, dropping 11.2% from post-planting 2020 to post-harvest 2021, while DTGW increased an average of 19.8% and DTGW+TD increased an average of 50.6%. POXC is considered to be a more processed fraction of labile C, closely correlating with SOC. It is possible that over time DTGW and DTGW+TD will begin to see a significant increase in SOC storage compared to before the BMPs were implemented.

The density fractionation of particulate organic matter led to interesting results, especially in the mineral-associated organic matter (MAOM) fraction. The contour tillage lost an average of 0.6 ± 1.4 g C kg^{-1} soil (-3.75%) in the MAOM fraction, while DTGW+TD gained an average 1.7 $\pm$ 0.9 g C kg^{-1} soil (+9.7%). The largest change was seen in DTGW, where C in the MAOM fraction rose by an average of 2.5 ± 1.0 g C kg^{-1} soil, or about +15.8%. Almost 80% of the C in this soil was found in this fraction. The rise in MAOM C in both terraced LMPs was not likely due to any breakdown of aggregates since the pattern of aggregate stability decrease was mostly similar in all LMPs. The breakdown of fresh organic material (fPOM) may be being immediately captured within this fraction; the pattern was similar between LMPs, but the terraced LMPs may have been more efficient at capturing these low molecular weight compounds. The MAOM fraction represents a more stable form of soil C due to its association with soil mineral particles and protection from decomposition. It is possible that the LISWC system is able to increase stabilized C, thereby increasing the health of the soil, since that fraction constitutes the largest amount of C and also had the largest C increases.

As expected, the fPOM fraction, consisting of the freshest, least decomposed material, declined in all LMPs between post-planting 2020 and post-harvest 2021. CT lost the most C from this fraction. This is likely due to small, free light particles washing out of the field in large rainfalls and during the spring snow thaw because of the lack of soil berms. The tile drained LMP, DTGW+TD, lost the least amount of C in the fPOM fraction. This may be due to the fact that quicker removal of subsurface water kept the levels of microbial decomposition lower relative to the other LMPs.

The greatest amount of C lost from the oPOM fraction was also observed in CT. This fraction of most New Brunswick soils has the oldest C (unpublished data) and is therefore important to protect from decreasing (L.P. Comeau, personal communication, March 30, 2022). There was a small gain of 0.22 ± 0.1 g C kg soil^{-1} in DTGW+TD, perhaps due to a small increase in microaggregates that remained on top of the 53 µm sieve; the occluded particulate organic matter is controlled mostly by physical protection from decomposition in aggregates. The only aggregate size fraction I analyzed was 1-2 mm, and there may have been an increase in smaller-sized micro-aggregates.

Overall, both drained and undrained terraced LMPs offered more promising results for the improvement of soil health, relative to contour tillage, in the second and third years of implementation. Both DTGW and DTGW+TD saw increases in the labile C fraction POXC – a fraction that is significantly related to SOC. Both also saw large increases in MAOM – a fraction that has shown positive feedback between new SOM stabilization from labile C and total soil C. These results occurred despite all three LMPs undergoing a decrease in aggregate stability. An increase in SOM would be positive for degraded New Brunswick soils and would improve soil functioning by improving soil physical structure and increasing biological activity.

5.2 Greenhouse Gas Emissions and Nitrate Supply Rate Response to Initial Three Years of Diversion Terrace, Grassed Waterway, and Tile Drainage Implementation

In this study, average CO_2 fluxes were highest in the undrained DTGW in both 2020 and 2021, though not statistically different than CT in 2020. However, cumulative CO_2 emissions were greater in DTGW than in either of the other LMPs, true for both years. CT and DTGW+TD did not significantly differ in either year in terms of CO_2 fluxes. In 2020, CO_2 fluxes were 63% higher than in 2021. This was likely caused by a much hotter summer of 2020 that increased microbial metabolism and stimulated the decomposition of organic matter.

Land management practice had a significant effect on N_2O fluxes. In 2019, DTGW+TD fluxes were significantly lower than either CT or DTGW. Fluxes were higher in DTGW in 2020 and 2021, though not statistically significant. This likely corresponds to the higher VWC in DTGW, decreasing the oxygen levels within the soil and increasing denitrification rates. Tile drainage kept soil moisture levels lower than in the other LMPs. This was particularly clear in 2021, where VWC stayed relatively constant while VWC increased each time we measured gas fluxes using the collars.

There were no significant differences between LMPs for CH_4 fluxes or cumulative CO_2 equivalents of CH_4 in 2019, 2020, or 2021. Aside from DTGW in 2019, which was skewed by one large spike, all CH_4 fluxes were negative in any LMP each year. It is common in the area for soils to act as methane sinks.

This is the first study in the area measuring GHG emissions from erosion control structures that include diversion terraces, grassed waterways, and tile drainage. Soil organic carbon in agricultural ecosystems in New Brunswick has been declining for decades. This study demonstrated that the addition of tile drainage to diversion terraces and grassed waterways significantly reduced the loss of stored carbon (in the form of CO_2) relative to undrained diversion terraces and grassed waterways, while also not emitting significantly more carbon than contour tillage, in the three years after implementation. The results were similar with respect to the loss of nitrogen in the form of harmful N_2O, where undrained diversion terraces and grassed waterways generally emitted more N_2O in the first three years after implementation. This has implications for farmers with respect to the efficiency of added inputs and may allow for the reduction of nitrogen fertilizer.

5.3 Combined Effects of Nitrogen, Temperature, and Soil Water Content on GHG Emissions in a Short-Term Incubation

To supplement the field experiment, a 6-day soil incubation experiment was conducted to investigate how volumetric water content, temperature, and nitrogen fertilization rate interact to affect greenhouse gas emissions (GHG) in a sandy loam soil from the FRDC research station. Nitrogen fertilization rate and VWC had a significant effect on CO_2 fluxes at 25°C, where they rose until 35% VWC, then plateaued. However, at 10°C, only N fertilization had an effect on CO_2 fluxes, and VWC did not alter CO_2 fluxes whatsoever. This suggests that in colder periods in spring or fall, soil moisture is not the dominant factor that controls CO_2. A field consisting of diversion terraces that happen to increase soil moisture would most likely not see increased CO_2 fluxes relative to a field without erosion control structures.

There were significant effects on N_2O fluxes. At 10°C, the only significant difference occurred at 40% VWC, where fluxes increased between fertilized and non-fertilized soils. At 25°C, fluxes increased significantly between fertilized and non-fertilized soils at 27%, 35%, and 40%. The combination of 40% VWC and fertilizer addition resulted in a large spike in emissions above the unfertilized 40% VWC treatment, rising to 1065 ± 269 from 30.7 ± 11.1 μg N_2O-N kg^{-1} day^{-1} – an increase of 3369%. Approximately 75% of the total CO_2e in the treatment consisting of 170 kg N ha^{-1} and 40% VWC at 25°C was due to N_2O emissions. This has implications for the timing of fertilization in the field, as emissions can be greatly reduced if avoiding this combination of conditions. To minimize the risk of a large burst of N_2O, fertilizer application should not occur when soil VWC is at or above 40%.

5.4 Recommendations for Future Research

The results of this study do not consider the spring snow melt period, which is often the wettest period of the year in the study region. It would be useful to examine periods during which soils approach saturation, to determine whether these terraced systems affect spring GHG emissions. Emissions can be significant outside the growing season in eastern Canada, often exceeding in-season emissions (Ruser et al., 2006; Wagner-Riddle & Thurtell, 1998). No studies have examined GHG emission response to snow melt periods under diversion terrace and grassed waterway management systems.

Another possibility would be to assess the GHG emissions coming from the furrow in the seasons in which potato is grown. I placed collars directly into the potato ridges, but water is not uniformly distributed in the potato ridge-furrow complex (Burton et al., 2012). Water application from rainfall will tend to accumulate in the furrow, leading to higher water-filled pore space and decreased O_2 diffusion into the soil. Combined with a higher bulk density, this leads to higher denitrification and N_2O emissions. The results in this study could be underestimating the overall emissions by not accounting for furrows.

The CH_4 oxidation rates in CT and DTGW+TD were similar to those seen in potato ridges in a German experiment (Ruser et al., 1998). Another study (Flessa et al., 2002) found ~86% of the CH_4-C uptake occurred in the potato ridges. This suggests most of the uptake was likely captured in our study but, again, that furrows are unaccounted for. The same study found that uncompacted tractor rows were a C sink but that compacted tractor rows were a source of CH_4, so, like N_2O, future research in this site should attempt to study differences between ridges and compacted/uncompacted furrows to avoid over- or underestimation of fluxes.

There is evidence that tile drainage can increase macroaggregation but negatively affect other aggregate size classes (Kumar et al., 2014). As we only studied aggregates in the 1 to 2 mm size class, it might be interesting to investigate smaller size classes and their response to a tile-drained LISWC system. This might help to explain why there was no significant difference in our study between the effect of LMPs on aggregate stability, but the distribution of carbon was changing (i.e., MAOM increasing in terraced LMPs).

Research has also shown that tile drainage can lead to high levels of nitrate loss in potato fields (Madramootoo et al., 1992). Indirect N_2O emissions from the water retention pond could add to the fluxes emitted from soil and push emissions higher. This could negate the observed benefits from the LISWC system on N_2O emissions relative to a contour-tilled field.

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Appendices

Appendix A Photos of the LISWC system at the FRDC

A.1 CT (left), DTGW (middle), DTGW+TD (right) with oats in the field in 2020.

A.2 View from the side of one of the grassed waterways with the soil berm on the downslope edge (left).

A.3 View of DTGW+TD plots (foreground) from the side.

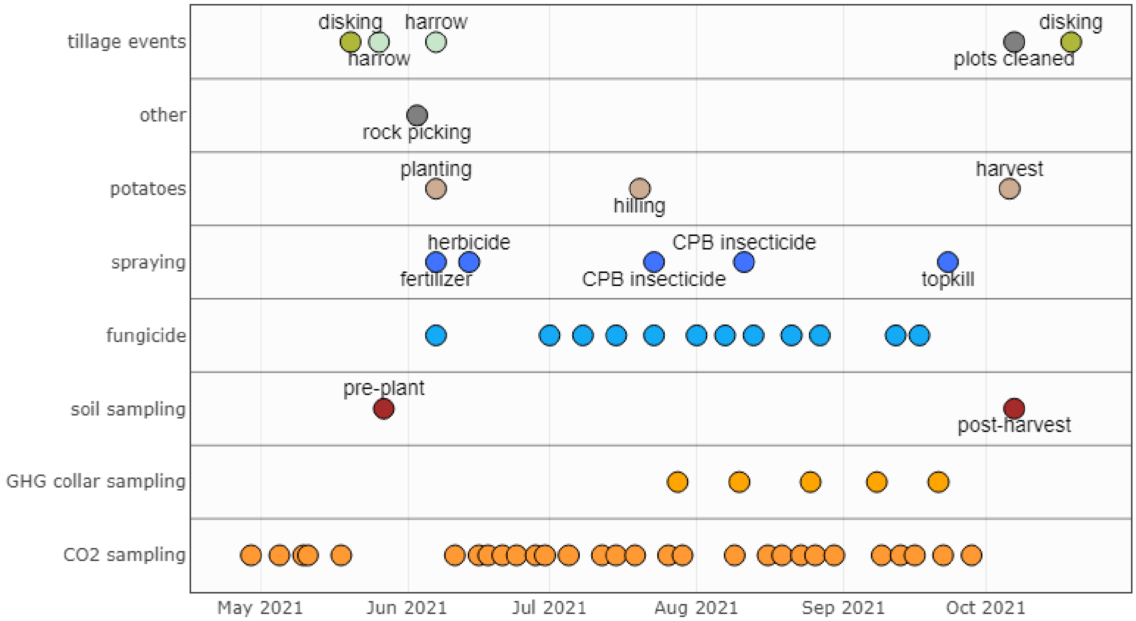
A.4 View of CT from the bottom looking upward after a heavy rainfall in which visible signs of water erosion appear.

Appendix B List of soil properties for each LMP and slope position. Bulk density was determined August 19th and 26th, 2020. All others determined July 2nd, 2020.

		Soil property						
		pH	CEC	Sand	Silt	Clay	Bulk density	C
			meq/100g	%	%	%	g.cm^3	%
LMP	Slope position							
CT	Upper	5.43 (0.056)	10.3 (0.21)	52.2 (0.6)	37.7 (0.33)	10.3 (0.33)	1.26 (0.048)	1.83 (0.084)
CT	Mid	5.45 (0.043)	10 (0.37)	57.3 (0.42)	33.2 (0.4)	9.33 (0.21)	1.23 (0.037)	1.78 (0.076)
CT	Lower	5.52 (0.04)	10.2 (0.17)	57.2 (0.4)	32.2 (0.31)	10.8 (0.17)	1.35 (0.055)	2.01 (0.046)
DTGW	Upper	5.58 (0.07)	9.5 (0.22)	53.8 (0.79)	36.3 (0.67)	9.67 (0.49)	1.28 (0.055)	1.78 (0.078)
DTGW	Mid	5.48 (0.054)	10 (0.26)	55 (0.37)	34.2 (0.4)	10.8 (0.17)	1.4 (0.038)	1.75 (0.052)
DTGW	Lower	5.45 (0.056)	11 (0.37)	55.3 (0.61)	32.8 (0.7)	11.8 (0.17)	1.22 (0.032)	2.03 (0.031)
DTGW+TD	Upper	5.65 (0.072)	8.17 (0.31)	55.2 (0.48)	36 (0.68)	8.83 (0.48)	1.25 (0.057)	1.37 (0.063)
DTGW+TD	Mid	5.07 (0.042)	6.67 (0.33)	55.7 (0.67)	34.5 (0.5)	9.83 (0.31)	1.21 (0.044)	1.57 (0.063)
DTGW+TD	Lower	5.17 (0.042)	7.67 (0.42)	55.7 (0.76)	33.3 (0.49)	10.7 (0.21)	1.35 (0.096)	1.96 (0.092)

Appendix C Timeline of events in 2020

Appendix D Timeline of events in 2021



Disking took place on May 20th. Rocks were picked following spring disking by approximately a dozen people on June 3rd. A first harrowing took place on May 26th. A final harrowing took place on June 7th, followed by a Russet Burbank potato planting. Rows were spaced at 0.91 m, and plants were spaced at 0.38 m within rows. An application of 17-17-17 urea-based fertilizer was applied during planting at a rate of 1000 lbs acre^{-1} (190.5 kg N ha^{-1}). This was slightly lower than the 208 kg N ha^{-1} fertilizer rate that is the typical maximum fertilization rate for potato farming in New Brunswick (Government of New Brunswick, n.d. -d).

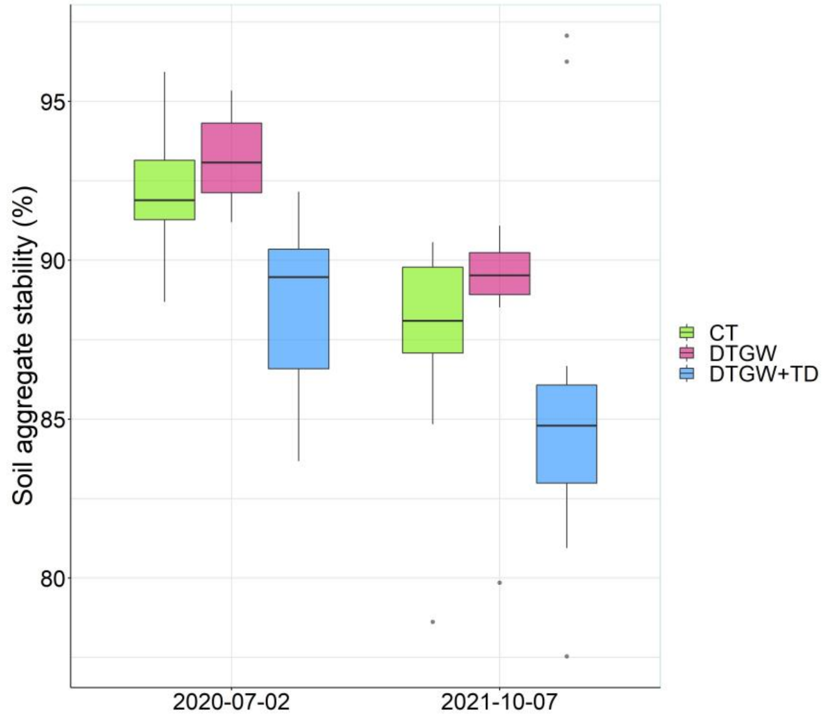
On June 14th, a pre-emergent herbicide (Sencor 1.5 kg/ha and Dual II 1.75 L/ha) was sprayed to suppress weed activity. Potato rows were hilled on July 20th. Insecticide was sprayed on both July 23rd and August 11th to suppress Colorado Potato Beetle (*Leptinotarsa decemlineata*) activity. Fungicide (Admire) spraying was scheduled every Friday (weather permitting) but sometimes took place a day earlier or later depending on rains. Topkill (Reglon) was applied on September 23rd. Top flail took place on October 4th, and potato harvest finally took place on October 6th. All plots were cleaned with a rock picker on October 7th. The final disking took place on October 19th.

Appendix E Rill erosion after Hurricane Elsa.

Appendix F Soil Aggregate Stability Analysis Procedure

The method separates the soil sample into loose aggregate, stable aggregate, and sand portions. A total of 4g of soil was used per sample, and samples were run in duplicate. Loose aggregate portion was determined by first placing the soil in the sieve nests of the apparatus, lowering them into the metal cans below that were filled with distilled water, and allowing to rest for 5 minutes. The sieves were then lowered into the starting position, and a 3-minute process begins, whereby the sieves are oscillated up and down gently to allow loose aggregate portion to collect in the metal can below. Water and soil were then vacuum filtered over pre-weighed 90mm filters in a Buchner funnel. Paper and soil were washed into a pre-weighed and labeled tin to dry in the oven at 110°C. At this point, stable aggregate and sand portions remain in the sieve. For the stable aggregate portion, the metal cans were filled with a dispersing solution (sodium hydroxide) and the sieves were oscillated in the apparatus for 5 minutes. After the 5 minutes, the sieve was raised, and a rubber-tipped policeman was used to gently break up aggregates. These oscillation and physical agitation steps were repeated twice more so that the aggregates were fully dispersed. A 55mm glass filter was used to vacuum filter the stable aggregate portion into pre-weighed and numbered tins to dry in the oven. Remaining sand portion in the sieve was washed into a final tin for oven-drying. If the difference between the final aggregate stability percentage values of reps 1 and 2 was greater than 7%, a third rep was required.

Appendix G Soil Aggregate Stability by LMP in 2020 and 2021.



Appendix H Permanganate Oxidizable Carbon Analysis Procedure

A 0.2 *M* stock solution of potassium permanganate ($KMNO_4$) was first prepared from a solution of $KMNO_4$ in solution of calcium chloride ($CaCl_2$). This was then used to create the 0.02 *M* working solution for the procedure. It should be noted that the creation of a working solution differs from the initial protocol developed by Weil (2003) and Culman (2013). This was to help minimize variation between samples caused by instrument or human error.

Four standard concentrations of $KMnO_4$ were prepared from the 0.2 *M* stock solution (0.005 M, 0.01 M, 0.015 M, and 0.02 M). These standards, and their resulting absorbance values (once diluted to final working standards), are then used to build a standardized curve. The intercept and slope values from this curve are then used in the final calculation of permanganate oxidizable carbon.

A total of 2.5 g of air-dried soil was sub-sampled from each of the 54 subplot samples and placed in a centrifuge tube. Working solution (20ml) was added to each tube and placed in an oscillating shaker for two minutes at 240 rpm. Each tube was then placed in the dark for 10 minutes to settle. Once settled, 0.5 ml of supernatant was pipetted into a glass test tube with 49.5 ml DDH_2O, capped, and inverted to mix thoroughly. A total of ~1.5 ml of the final sample solution was pipetted into disposable cuvettes for absorbance analysis in a Biochrom Libra spectrophotometer (Biochrom Ltd., Cambridge, United Kingdom). Samples were analyzed at 550 nm, and calculated using the following formula:

$$POXC\ (mg\ kg^{-1}) = [0.02\ mol/L - (a+b*Abs)] * (9000\ mg\ C/mol) * (0.02\ L\ solution/Wt)$$

where 0.02 mol/L is the initial solution concentration, a is the intercept of the standard curve, b is the slope of the standard curve, Abs is the Absorbance of Unknown, 9000 is the milligrams of carbon oxidized by 1 mole of MnO_4 changing from Mn^{7+} to Mn^{2+}, 0.02 L is the volume of stock solution reacted, and Wt is the weight of air-dried soil sample in kg.

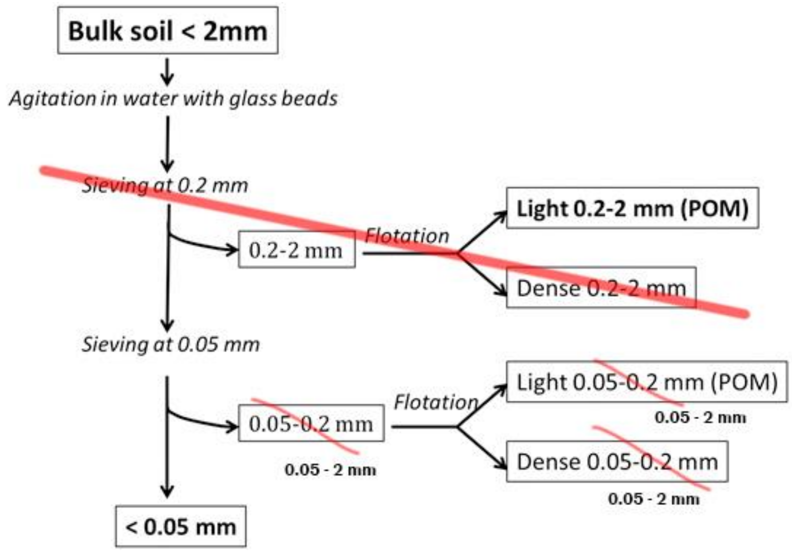
Appendix I Density Fractionation Analysis Procedure

Density fractionation was performed on soil samples from: July 2nd, 2020; September 24th, 2020; May 27th, 2021; and October 7th, 2021. The protocol was based on the method described in Peoplau et al. (2018) with one alteration – removal of the sieving at 0.2 mm (Appendix G). The result of the protocol was three specific fractions: fPOM (free light POM separated from oPOM); oPOM (sand-sized and occluded POM, 50-200 μm); and MAOM (silt- and clay-sized mineral associated organic matter, < 50 μm rinsed from catch pan).

A total of 100 g of air-dried, 2mm-sieved soils were first shaken for 16 hours at 225 RPM on an orbital shaker in plastic bottles with 200 mL distilled water and seven 5mm glass beads. When shaking was finished, contents were wet-sieved at 50 μm. The MAOM fraction passed below to the catch pan after rinsing with distilled water. The remaining portion on top of the sieve were the combined fPOM and oPOM fractions. The MAOM fraction was rinsed into one beaker, while the remaining fractions were rinsed into a second beaker. All beakers were placed in the oven at 55°C overnight to fully dry.

A total of between 50 and 100mL of sodium iodide (specific gravity of 1.7 g/mL) is added to the fPOM and oPOM fractions. In this case, vigorous stirring for 1 minute was substituted for shaking. After a roughly 2-minute settling time, the floating material was fPOM and the sunken material was the mineral oPOM fraction. A spoon was used to scoop as much floating fPOM as possible onto a filter paper inside a Buchner funnel, before carefully emptying remaining fPOM and sodium iodide (NaI) onto the filter paper, without allowing any mineral fraction from the bottom of the beaker to transfer into the funnel. Pure sodium iodide was then collected with vacuum filtration for re-use. The fPOM was rinsed with distilled water and vacuum filtered to ensure no NaI remained in the fraction. The fPOM was then rinsed off the filter paper into a 100mL beaker. The rinsing and vacuum filtering was repeated with the remaining mineral oPOM fraction. Both beakers with respective fractions were oven dried at 55°C overnight. Once dry, the MAOM and oPOM fractions were ground in a Retsch ball mill grinder for 3 minutes. The fPOM fraction was hand-ground using a mortar and pestle. Final samples were analyzed at the FRDC for total C concentrations using an Elementar varioMACRO apparatus (Elementar Americas Inc., Mt. Laurel, NJ).

Appendix J Diagram of modified protocol for density fractionation



Appendix K Collar set up for collecting GHG samples in 2019, 2020, and 2021

Appendix L COVID-19 notes from my workbook

Below are the notes I wrote down in my masters workbook beginning in early 2020. I was slated to fly to Ottawa at the end of April to pick up my parents' car for the summer and drive out to Fredericton, but of course the pandemic shutdown happened in March. Maja joked about keeping a journal, and I figured it could be an interesting thing to look back on decades from now. March and April were uncertain times, and many people were scrambling to figure out what to do with growing season arriving quickly.

These are random, unorganized thoughts and notes of mine, with no particular structure – only what I found notable at any given time.

NBA shutdown - 1st realization this was going to start affecting people outside Wuhan and Washington

University of Washington shutdown - still wasn't thinking too much about it affecting us. Chinese community was aware and malls in Richmond began to empty long before Canadian shutdown.

Classes moved online. Fortunately, no exam in FRST545. Exam cancelled in FRST530. Assignment grade only. Very lucky! Many others having to adapt to online for 6 long weeks. TA'ing online a bit more annoying. No more labs :(

The Great Toilet Paper Shortage of 2020 :/

Began dating online over 'Zoom'. Had two first dates on Zoom followed by bike ride second dates. Dated Johanna for about 3 weeks before leaving. Gave me a kickstand for my bike the day before I left. Bittersweet. Hope to see her in fall. Will see if I date anyone in Fredericton. What a time to be dating!

Counting myself very fortunate at this time. Research maybe delated. But still on way to New Brunswick where things are in very good shape. Only 118 cases and no deaths. Many friends having to make contingency plans. Raelani was thinking about quitting grad school :(but is still subletting my place. Amy Mays is basically having to throw away a year. People are having to apply for exemptions to conduct research at UBC. Rumours that September classes will be <u>only</u> online. Wow. Hard to fathom.

Think I will apply to the BCIA in the next couple weeks. Economy is in the tank. Thinking this could last potentially years, so it will coincide with graduation from UBC. Might need the leg up on folks. Have 2 weeks in Freddy to do my best quarantine.

30 people on the flight heading east!! There are more people on our Friday trivia zoom gatherings (40+ people) than on this flight.

The Drive to NB.

Everything was closed along the way so had to get inventive with where to go… #2. Added a little extra time driving around looking for forests to use.

Went through a checkpoint somewhere in Quebec. Somewhere between Quebec City and NB border. Pretty quick. Checkpoint at NB border was about 10 mins. They were pretty thorough. Took roommate's info, Louis's info, job info, etc.

Had a laugh passing a sign for a potato museum.

Immediately passes another sign "Town of Florenceville: French fry capital of the world" … funny, must be in NB now.

5 days into quarantine. Had to run outside today. Feeling antsy.

Roommate brought home Covered Bridge chips and Chicken Bones – two NB delicacies.

After over 2 weeks of no new cases there were apparently 2 new ones today. Still, we are very lucky here. Saw a tent set up downtown yesterday with a few people dressed head to toe in PPE.

Struggling mightily with motivation for school. Did my taxes yesterday but schoolwork has been absent for weeks on end. Moving was one thing. Covid doesn't help. There is certainly time. But finding self-discipline right now is hard. Gone are the routines. The workplace environment. Etc.

May 20th. Had orientation meeting today over Zoom with new students. There is talk of a regular social meeting on Zoom which would go a long way to making things more exciting around here.

Was supposed to go out to the field for the 1st time today, but it got cancelled at the last minute

May 26th. A glimmer of hope today. The email from Josée Owen seems to suggest I'll be able to at least take EGM samples soon.

June 18th. Have been able to gas sample and plant probes. No sign of collars or soil sampling. But have gone for beers and met people which helps.

June 22nd. Louis said early July I might have access to the lab. Great news! Time to get busy!

July 9th. Lab access pushed back…but have government email and laptop now. Access coming soon. Have received orientation docs and WHMIS docs. Moving along :)

This is where my notes on Covid end. I wouldn't end up getting into the lab until nearly September. By mid-October I was on the road back to Ontario to return the car and spend some time with the family before flying back. Thankfully I met a few girls through the AAFC virtual meetings. Isabelle, Mackenzie, Jess, Sarah, and I had some good times with Fredericton and the 'Atlantic bubble' being less restrictive than the rest of Canada. We went camping in PEI. We took weekend trips to Halifax, where I was able to visit my aunt. I was in 6 different provinces in 2020. Wild, considering what was going on.

Over the winter, things changed. When I returned in May of 2021, New Brunswick was requiring 14 day quarantines, but unlike early in 2020, they were requiring people (if they didn't already live there and were able to quarantine privately) to stay in hotels for at least the first 7 days. Fortunately, UBC was able to help with these costs. But I was stuck in that room for 7 days, doing jumping jacks and pushups to keep sane and fit. Of course, I watched a lot of movies. The craziest thing was getting my first Covid test. Protocol was to sit in a chair inside the door. They would open the door, all dressed in clothes that reminded me of those scenes in E.T. They swabbed me and left. As far as I know, I never got Covid. The hotel got a few cases of Covid inside, shortly after I left quarantine. Then the whole program was shut down. Had I arrived two weeks later, I wouldn't have had to quarantine. It was an utter gong show. I played a lot of phone tag those two years trying to figure out how to get negative tests to the right people. The check-ins I was supposed to have in 2021 never happened. Nobody ever showed up to the house to check on me after arrival. I could have been anywhere. Both provincial and federal governments were unbelievably unprepared for this to happen. All in all, I was very fortunate to be able to travel so much and continue with my program.