Effects of environmental factors and agronomic practices on greenhouse gas emissions

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Abstract

Identifying environmental factors that enhance the efficacy of best management practices (BMPs) in mitigating cropland nitrous oxide (N_2O) emissions is crucial for reducing emissions. The research reported in this thesis quantitatively assessed BMPs for their efficacy in reducing emissions by focusing on variations in soil conditions, season, or climate. Practices of water table and nutrient management were investigated through field studies on sandy loam and silty clay soil field sites in southwestern Quebec, Canada. Three non-growing season practices including cover crops (CC), nitrification and urease inhibitors (NI + UI) and tillage were investigated using meta-analysis. In addition, the effects of changing climate on winter N_2O emissions were evaluated using the Denitrification and Decomposition (DNDC) model.

Long-term effects of controlled drainage with sub-irrigation (CDS) on crop yield and N_2O emissions revealed that CDS could improve grain yield compared to regular free tile drainage (FD), depending on the growing season rainfall and its temporal distribution. CDS positively affected grain yield based on data collected over 12 years at one of the study sites. Lower yields under CDS were observed when excessive monthly rainfall (230 mm) occurred during the crop's vegetative period. In three of six years, N_2O fluxes under CDS treatments were greater by 49% than those under FD, but 45% lower in the remaining years, implying that – notwithstanding the quantity of growing season rainfall – CDS does not necessarily always produce greater fluxes than FD. N_2O fluxes coincided more with fertilizer application.

The effects of soil type (sandy loam vs. silty clay) on GHG emissions were examined by investigating three nitrogen fertilization rates (140, 180, 220 kg N ha^{-1}) on yield-scaled N_2O emissions. Grain yields increased with N fertilization rate in both soils. Yields were greater on the sandy loam than on the silty clay. Yield-scaled N_2O emissions from the silty clay soil were lowest at the 180 kg N ha^{-1} fertilization rate, compared to 140 kg N ha^{-1} for the sandy loam. Under grain corn production, yield-scaled emissions from the poorly drained fine-textured silty clay soil were five-fold greater than well-drained medium-textured sandy loam soil.

A meta-analysis study focusing on over-winter cropland N_2O emissions showed that the non-growing season emissions ratio to full-year N_2O emissions ranged between 5% to 91%. No-till significantly reduced N_2O emissions by 28% compared to conventional tillage, and this effect was more pronounced in drier climates. NI + UI also significantly reduced over-winter emissions by

23% compared to conventional fertilizers, and this effect was more evident in medium-textured soils than coarse soils. CC showed an overall reduction potential of 18%; however, this effect was not significant. Under the CC practice, N_2O emissions were reduced in humid climates but increased in drier climates, while no-till and NI + UI practices effectively reduced over-winter emissions in dry and humid winter regions on all soil types.

The DNDC model was used to simulate historical and future winter emissions over 30 years for intensive grain corn production in southwestern Quebec. A historical analysis showed that the greatest average winter N_2O emissions occurred in warm and wet years. Future scenario [2038 - 2067] analysis showed a 10% rise in winter N_2O emissions, associated with an increase of winter soil temperature of 1°C, soil moisture (WFPS) increase of 8%, and snow water equivalent decrease of -1 mm yr^{-1}. These simulations highlighted the need to focus on mitigation measures for winter N_2O emissions from agricultural soils.

Farm practices' effectiveness in mitigating GHGs varied substantially due to differences in soil types and climate patterns. These results will be useful to stakeholders and policymakers seeking to make decisions regarding promoting and adopting BMPs for climate change mitigation solutions.

Résumé

Afin de réduire les émissions, il devient crucial d'identifier les facteurs environnementaux réhaussant l'efficacité des pratiques exemplaires de gestion (PEG) qui atténuant les émissions de protoxyde d'azote (N_2O) provenant de terres cultivées. La présente recherche évalua quantitativement l'efficacité des PEGs à réduire ces émissions selon les variations saisonnières, climatiques et pédologiques. Les pratiques de gestion de la nappe phréatique et des éléments nutritifs du sol furent étudiées au moyen d'études sur le terrain dans le sud-ouest du Québec (Canada), ayant lieu sur des loams sablonneux et limons argileux. Trois pratiques hors saison (soit, cultures de couverture (CC), ajout d'inhibiteurs de nitrification et d'uréase (NI + UI), et travail du sol), furent étudiées à l'aide d'une méta-analyse. De plus, les effets du changement climatique sur les émissions hivernales (saison morte) de N_2O furent évalués à l'aide d'un modèle de dénitrification et de décomposition (DNDC).

Les effets à long terme d'un drainage contrôlé avec irrigation souterraine (DCIS) sur le rendement des cultures et les émissions de N_2O démontrèrent que le DCIS pouvait, selon la pluviosité et sa distribution temporelle lors de la saison de culture, améliorer le rendement en grain par rapport à un drainage conventionnel (DC). Vu douze ans de données provenant d'un seul site, le DCIS eut un effet positif sur le rendement en grain. En revanche, des rendements inférieurs furent observes sous DCIS lorsque la précipitation excéda 230 mm pendant la phase végétative de la culture. Trois ans sur six, les flux en N_2O sous le DCIS furent 49% plus élevés que sous le DC; cependant, durant les autres années ils furent 45% moins élevés, laissant entendre que – nonobstant la quantité de précipitation durant la période de culture – le DCIS ne génère pas nécessairement des flux plus élevés que ceux notés sous le DC. Les flux de N_2O les plus élevées coïncidèrent plutôt avec les applications de fertilisant.

L'effet du type de sol (loam sablonneux *vs.* limon argileux) sur les émissions de gaz à effet de serre (GES) fut examiné grâce à une étude menée sur l'effet de trois taux de fertilisation (140, 180, et 220 kg N ha^{-1}) sur les émissions en N_2O ajustées au rendement. Sur les deux types de sol, le rendement en grain augmenta avec le taux de fertilisation, avec des rendements plus élevés sur le loam sablonneux que sur le limon argileux. Les émissions en N_2O ajustés au rendement sur le limon argileux furent les plus basses au taux de fertilisation de 180 kg N ha^{-1}, compare à 140 kg N ha^{-1} sur le loam sablonneux. En culture de maïs-grain, les émissions en N_2O ajustées au rendement provenant loam limono-argileux à texture fine mal drainé furent

cinq fois plus élevés que celles provenant d'un sol sablo-limoneux à texture moyenne bien drainé.

Une méta-analyse d'études sur les émissions de N_2O provenant de champs agricoles en saison morte (hiver), montra que celles-ci représentèrent de 5% à 91% des émissions annuelles. La culture sans labour réduisit de manière significative les émissions en N_2O (28%) par rapport au labour conventionnel, et cet effet était plus prononcé sous un climat sec. De même, l'ajout d'inhibiteurs NI + UI réduisit les émissions en saison hivernale par 23% comparé à une fertilisation conventionnelle, et cet effet fut plus grand pour les sols à texture moyenne que les sols grossiers. Les cultures de couverture offrirent une réduction potentielle de 18%; cependant cet effet ne fut pas significatif. Sous une pratique de culture de couverture, les émissions en N_2O furent réduites sous les climats humides, mais augmentèrent sous des climats secs. De plus, les pratiques de culture sans labour, et l'ajout d'inhibiteurs NI + UI se montrèrent efficaces dans la réduction des émissions en saison hivernale dans les régions avec des hivers secs ou humides, sur tout type de sol.

Le modèle DNDC servit à simuler les émissions en période hivernale lors de 30 ans de production intensive de maïs grain dans le sud-ouest du Québec, à la fois dans une période historique et une période future. L'analyse de la période historique indiqua que les émissions les plus élevées en saison hivernale eurent lieu lors des années chaudes et humides. Quant aux scenarios futurs [2038 - 2067] l'analyse indiqua une augmentation de 10% en émissions de N_2O en saison hivernale, associée à une augmentation de 1°C en température hivernale, une augmentation de 8% de l'humidité des sols (part de porosité occupée par l'eau), et un déclin de 1 mm yr^{-1} dans l'équivalent en eau de la neige. Ces simulations soulignent la nécessité de mettre l'accent sur les mesures d'atténuation des émissions hivernales de N_2O provenant de sols agricoles.

L'efficacité des mesures d'atténuation appliquées à la ferme, à réduire les émissions de gaz à effet de serre, varia sensiblement selon les différents types de sol et conditions climatiques. Les présents résultats s'avéreront utiles aux parties prenantes et décideurs visant à prendre des décisions quant à la promotion et l'adoption de PEGs s'adressant au changement climatique.

Dedication

This thesis is dedicated to my late father, Mr. R.I Ekwunife.

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Contributions of the Authors

This thesis was written in manuscript-based format. For each manuscript, Kosoluchukwu Ekwunife, Ph.D. candidate, carried out field experiments, curated data from other studies, performed all the analyses, wrote first drafts of the papers, and undertook the revisions made by the co-authors. Dr. Chandra A. Madramootoo, who co-authored all manuscripts, provided guidance and supervision of the work reported in the papers, outlining the study's scope and obtaining the funding to do the research. Dr. Naeem A. Abbasi, a co-author of Chapter 5, curated data and provided technical comments. Dr. Qianjing Jiang, a co-author of Chapter 6, calibrated the DNDC model and provided the downscaled data used for the future climate scenario study. All the manuscripts were reviewed by the authors involved.

A. Thesis components that have been or will be submitted for publication in peer-review journals

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B. Thesis components that have been presented at scientific conferences

- **Ekwunife K.**, 2021. "Greenhouse gas emissions from frozen agricultural soils." McGill International Symposium – Mitigating Agricultural Greenhouse Gases and Increasing Carbon Sequestration in a Circular Economy, September 29, 2021.

- **Ekwunife K.**, Madramootoo C.A., Abbasi N.A., 2021. "Assessment of the Impacts of Tillage, Nitrification Inhibitors and Cover Crops on Nitrous Oxide Emissions over Winter and Early Spring." ASABE 2021 Annual International Meeting, Virtual and On-Demand, July 12-16, 2021.
- **Ekwunife K.**, Madramootoo C.A., Abbasi N.A., 2021. "Effects of Fertilizer Rates on Nitrous Oxide Emissions on Two Soil Types." ASABE 2021 Annual International Meeting, Virtual and On-Demand, July 12-16, 2021.
- **Ekwunife K.**, Madramootoo C. A., 2021. "Estimating greenhouse gas emissions from frozen agricultural soils in the Canada-US North Atlantic Region." 5th CIGR International Conference 2021, Virtual, May 11-14, 2021.
- **Ekwunife K.**, 2018. "Mitigating Greenhouse gas emissions under agricultural water management systems in Eastern Canada – A synopsis of findings at St Emmanuel experimental site, Quebec." Agricultural Greenhouse Gas Program meeting, McGill University, Macdonald Campus, Ste Anne de Bellevue, Quebec, September 25th, 2018.

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List of Abbreviations and Symbols

ANOVA	Analysis of variance
C	carbon
CC	cover crops
CDS	controlled drainage systems with sub-irrigation
CO_2	carbon dioxide
CT	conventional tillage
d	index of agreement
DNDC	denitrification - decomposition
FD	regular tile drainage
FTC	freeze-thaw cycles
GHG	greenhouse gas
IPCC	Intergovernmental panel on climate change
KCl	potassium chloride
N	nitrogen
N_2	nitrogen gas
N_2O	nitrous oxide
NI + UI	nitrification and urease inhibitors
NO_3^-	nitrate
NT/RT	no-till or reduced tillage
NH_4^+	ammonium
θ_{fc}	field capacity
PBIAS	percent bias
r	Pearson's correlation coefficient
R^2	coefficient of determination
SOC	soil carbon content
SOM	soil organic matter
SWE	snow water equivalent
WFPS	water filled pore space
WTM	water table management
WTD	water table depths

Chapter I

General Introduction

1.1 Background

The agricultural sector emits significant amounts of greenhouse gases such as carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O), which contribute to global climate change. In 2017, the agricultural sector was responsible for 8.4 % of total GHG emissions in Canada, while cultivated soils accounted for approximately 40% of that proportion (ECCC, 2019). The increasing use of fertilizer N inputs to meet the growing food demand has caused a significant increase in N_2O emissions, a major greenhouse gas with about 300 times more warming potential than CO_2. For instance, in 2017, the sector accounted for 77% of Canada's N_2O emissions, mainly linked to the 71% rise in fertilizer use relative to 2005 levels (ECCC, 2019). This environmental concern calls for a thorough understanding of the factors that can influence these emissions and potential agronomic practices that can be applied to reduce emissions.

Fertilizer inputs and the early spring periods have been identified as the two major periods of N_2O emissions occurrence on croplands. Fertilizer N inputs, mainly inorganic fertilizer, will usually result in heightened emissions occurring two to four weeks after application. Elevated N_2O emissions occurring over winter can be as much as 90% of the total annual emissions (Risk et al. 2014) and thus are also a major concern for agriculture. Significant factors responsible for the emissions include the availability of substrates and microbes, soil moisture, soil temperature and cumulative soil freezing days (Wagner-Riddle et al. 2017). Many cropland practices can potentially mitigate greenhouse gas (GHG) emissions, the most prominent of which include nutrient and water management, tillage, and cropping systems. However, there have been mixed research results on the effectiveness of some of these practices in mitigating cropland emissions. Understanding how these management practices can best be utilized to reduce N_2O fluxes is crucial in mitigating emissions. Thus, knowledge of the mechanisms as well as soil and climate conditions that influence the emission of N_2O is essential.

1.2 Research objectives

The overarching goal of this research is to investigate the influence of location-specific environmental and soil factors on N_2O responses to agronomic practices. Specific objectives of this study are as follows:

i. To investigate the influence of seasonal climate on the impact of CDS on yields and N_2O fluxes.
ii. To study the effects of fertilizer rate on yield-scaled nitrous oxide emissions from two soils (sandy loam and silty clay).
iii. To assess the impacts of tillage, cover crops, nitrification and urease inhibitors on over-winter N_2O across soil types and aridity zones (cold-dry vs. cold-humid areas).
iv. To investigate the relationship between winter N_2O emissions from intensive crop production under changing winter and spring conditions and climate variables (e.g., snowpack, soil temperature and soil moisture).

1.3 Synopsis of the research methods and significance of the study

The study focused on two management practices (water table management (WTM) and fertilizer rate management) that influence growing season N_2O emissions; and three management practices (cover crops, nitrification and urease inhibitors, and tillage) that influence over-winter N_2O emissions. The WTM practice was investigated through a one-year field study conducted in 2018 in addition to historical data collected in previous years on a sandy loam field site in southwestern Quebec, Canada. The fertilizer rate management practice was investigated through field studies conducted in 2019 on sandy loam and silty clay soil sites in southwestern Quebec, Canada. It also aggregated data points from previous field studies conducted on grain corn production sites in North America. Three non-growing season practices were investigated using meta-analysis. In addition, the effects of changing climate on winter N_2O emissions were evaluated using the Denitrification and Decomposition (DNDC) model.

Aggregation of historical data, meta-analysis, and modeling allow for extrapolation over space and time to supplement data points from short-term field studies to look at data on a larger scale with broader impacts and implications. These methods offer a more robust understanding of factors and conditions that influence emissions and can help draw conclusions on the effectiveness of management practices in reducing emissions. The information from this study will be invaluable in selecting agricultural practices aimed at mitigating cropland emissions by ensuring that the practices are carefully managed to avoid significant emissions while not hampering crop yields.

1.4 Thesis structure

This thesis is presented in a manuscript format in which an introduction is presented in Chapter 1 including research background, knowledge gaps, objectives, and thesis structure. Chapter 2 presents an overview of relevant literature. Chapters 3, 4, 5, and 6 provide the research studies corresponding to objectives 1 to 4 and are written in the format of journal manuscripts. Chapter 7 summarizes the original contributions and conclusions from this work.

1.5 References

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Chapter II

Literature Review

2.1 Nitrous oxide emissions from soils

Greenhouse gas emissions are formed via the biochemical metabolism of soluble carbon substrates and reactive nitrogen species (NH_4^+ and NO_3^-) in the soil (van Groenigen et al. 2010). N_2O is produced in the soil mainly through nitrification and denitrification processes. Nitrification is an aerobic process in which ammonium is converted to nitrite and nitrate by aerobic bacteria, while denitrification is an anaerobic process that involves the conversion of nitrate to gaseous forms (NO, N_2O, N_2)(Liebig et al. 2012). Of the two processes, the denitrification process is considered the dominant source of N_2O emissions in the relatively moist soil condition predominant in parts of US and Canada (Liebig et al. 2012; Elmi et al. 2005). For instance, the process of denitrification is more prominent for producing N_2O emissions than nitrification in eastern Canada but perhaps not in the semi-arid prairies of western Canada. N_2O emissions from fertilized croplands have ranged from 0.7 to 51.8 mgN_2O-N m^{-2} d^{-1} (Molodovskaya et al., 2012). The contribution of each process to N_2O emissions varies and depends on environmental characteristics, including weather, climate, soil and agronomic factors that impact N_2O emissions. N_2O can be emitted from cropland either directly or indirectly. Direct emissions are cropland fluxes resulting from nitrogen additions or nitrogen mineralized from soil organic matter. Indirect emissions occur when reactive nitrogen is volatilized as ammonia (NH_3) or transported via surface runoff or leaching from cropland (a primary location), leading to N_2O emissions in another location.

2.2 Drivers of N_2O emissions

2.2.1 Substrates and microbes

Soil factors that impact N_2O emissions include substrate availability, microbes, soil type, moisture, and temperature (Liebig et al., 2012; Signor & Cerri, 2013). Adding fertilizers and manure supplies N and organic matter to the soil, the essential substrates for nitrification and denitrification processes (Gregorich et al. 2015). The timing of the fertilizer addition also affect the fluxes significantly (Crézé & Madramootoo, 2019). Soil organic carbon, which has been shown to positively correlate with soil clay content, is an important parameter that affects emissions (Rochette et al. 2018; Rochette et al. 2008; Uzoma et al., 2015). For instance, Rochette et al. (2008) suggested that large emissions from clay soils with high SOC in the fall long after spring fertilizer N application result from the denitrification sustained from the decomposition of large quantities of organic matter rather than from the fertilizer N inputs. Clay particles possess a strong adsorption capacity for organic carbon due to their large specific surface area and surface charge (Singh et al., 2018). It is noteworthy that clay soils generally have lower permeability with lower drainage resulting in higher soil water content which can result in higher denitrification rates (Rochette et al. 2008). Simple models such as the Intergovernmental Panel on Climate Change (IPCC) model that estimates N_2O emissions using a 1% emission factor for every N fertilizer amount applied (Bell et al., 2016) exclude these other important explanatory variables including soil types, substrates, temperature and moisture.

2.2.2 Soil moisture

Soil moisture has the second most effect on N_2O fluxes after substrate availability (Fisher et al., 2018). Linn & Doran (1984) proposed a model to demonstrate the effect of WFPS on N_2O production in which they suggested that 60% WFPS was the critical point above which anaerobic processes take effect. High rates of N_2O emissions from soils often occur when soil mineral N is available and moisture is above 60%WFPS but below saturation (Liebig et al. 2012). Optimum N_2O emissions for European soils have been observed under wetter conditions than 80%WFPS (Butterbach-Bahl et al., 2013). High-intensity N_2O fluxes responding to the availability of soil N and abrupt weather events such as changes in soil moisture and temperature are referred to as hot moments (Molodovskaya et al., 2012; Wagner-Riddle et al., 2020). Pulse emissions from hot moments could account for 59-84% of annual fluxes (Molodovskaya et al., 2012). High frequent flux measurements have been recommended to capture the sporadic peak fluxes associated with N_2O emissions (Creze, 2015).

2.2.3 Soil texture

Soil texture is an important soil parameter as it influences water infiltration rates through the soil. It is determined by the relative sizes of soil particles (sand, silt, and clay). Soil texture also influences how much water is available in the soil. For instance, coarse-textured soils have a lower water holding capacity than fine-textured clay soil (Oertel et al. 2016). Various soils influence N_2O emissions differently due to their soil hydraulic properties (Giltrap, Li, & Saggar, 2010). For instance, freely drained soils emitted lower rates of N_2O, possibly due to their physical attributes, including higher water infiltration rate and lower denitrification rate (Li and Kelliher, 2005). On the contrary, the high N_2O emissions observed in poorly drained soils with high clay content were attributed to the high moisture content (Rochette et al. 2008). However, with a WFPS often at 100%, significantly lower N_2O fluxes could also occur in poorly drained soil. Such low fluxes occurring at saturation were attributed to possible complete denitrification, favoring the production of N_2 over N_2O (Weerden & Styles, 2012).

2.2.4 Soil temperature

An increase in soil temperature triggers increased microbial metabolism in the soil (Oertel et al. 2016). Several studies have linked the increase in N_2O emissions with temperature change (Rezaei Rashti et al. 2015; Fisher et al., 2018; Schindlbacher et al. 2004). In their study, Rezaei Rashti et al. (2015) observed a positive relationship between emissions and seasonal mean temperature from 7^oC to 14^oC, but this relationship was not significant above 14^oC. The authors implied that the role of rising temperature in enhancing N_2O emissions is more evident in colder seasons and colder regions. Fisher et al., (2018) observed that temperature controlled the magnitude of the fluxes in their study. Generally, fluxes were < 0.14 mg N_2O-N m^{-2} h^{-1} at low temperatures usually outside the growing season and > 0.14 mg N_2O-N m^{-2} h^{-1} during the growing season. Schindlbacher et al. (2004) showed that soil temperature and soil moisture combined could explain 86% of the variations in N_2O fluxes. These studies indicate that significant changes in these soil factors, including substrate availability, soil moisture and soil temperature, could lead to increased N_2O emissions although the relationship might not be clear cut due to other confounding factors.

2.2.5 Weather and climate

Environmental and climate factors that occur naturally and are generally not under management control can potentially influence N_2O emissions. Environmental factors such as precipitation and temperature directly influence soil moisture and temperature respectively, which stimulates the production of N_2O (Fisher et al., 2018; Signor & Cerri, 2013). For instance, high N_2O emissions have been observed during periods of heavy precipitation, particularly when soil N is available (Creze, 2015; Flessa et al., 2002). Similarly, using stepwise regression analysis on metadata, Rochette et al. (2018) showed that cumulative N_2O fluxes are mostly influenced by growing season precipitation. Wind speed and solar radiation can also strongly influence evapotranspiration which effects soil water status and growth/crop N uptake.

Freeze-thaw cycles that occur in temperate regions in late fall to early spring likely result in a range in N_2O emissions that can be high – from 6.6 % of annual emissions to 90% (Risk et al., 2014; Li et al., 2012). However, studies investigating greenhouse gas (GHG) emissions from agricultural soils tend to focus on the growing season. The rapid increase in moisture and temperature over winter, particularly during the spring thaw, creates a flush of nutrients, alters the bacterial community, enhances microbial activities, and finally increases N_2O fluxes (Chen et al., 2020; Smith et al., 2010). Substrates are released when soil is frozen for long periods, likely due to the death of microbes and other soil fauna, making the duration of freezing potentially the largest driver of emissions during the spring thaw (Del Grosso et al., 2022; Wagner-Riddle et al., 2017). Soils subject to freeze-thaw cycles are likely to emit more N_2O (Chen et al., 2020). The contribution of freeze-thaw emissions to annual emissions varies widely, likely due to differences in substrate availability, soil type, snow cover, freeze-thaw dynamics, soil temperature, and moisture conditions (Chen et al., 2016; Matzner & Borken, 2008; Smith et al., 2010; Zhe et al., 2018). This variation highlights the need to review the magnitude of reported freeze-thaw emissions and identify what factors might be responsible for these differences. The complex interactions among these factors regulating the production and consumption of N_2O and the temporal and spatial variability associated with N_2O fluxes sometimes make it difficult to generalize how management impacts emissions (Liebig et al. 2012).

2.3 Gas flux measurements and emission estimations

2.3.1 Flux measurements

Quantifying greenhouse gas emissions precisely helps in determining the effectiveness of proposed management practices. Emissions are quantified using the gas measurements from agricultural fields by employing either the micrometeorological or chamber techniques: automatic or manual chamber. Chamber techniques are preferred for their suitability for scientific studies as they allow for observing direct emission changes from multiple plots by varying field treatments (Wang et al., 2013). Unlike the manual static closed flux chambers with which gas measurements are typically carried out weekly or biweekly, automated chambers allow for sub-daily measurements. Despite the supposed benefits of the automatic chamber, static closed flux chambers remain a popular method in measuring GHG emissions as this equipment is low cost and requires no electrical power. However, the low sampling frequency for closed static chambers leads to gaps in the dataset.

2.3.2 Chamber-based flux estimation models

After gas measurements are taken, the gas flux is then calculated using flux estimation models. Usually, bias and uncertainties are associated with a given chamber design and flux estimation method (Oertel et al., 2016; Myrgiotis et al., 2018). An adequate flux calculation method thus accounts for these sources of variability and minimizes the flux estimation error. Recently, improved methods have been developed to address some of the factors causing bias in static chambers. Chamber-based estimation models are generally classified as linear and non-linear models (Yan et al., 2015). During chamber deployment, the gas build-up in the chamber headspace reduces the diffusive flux, leading to flux data that are mostly non-linear (Parkin, Venterea, & Hargreaves, 2012). As a result, some researchers argue that applying linear regression on chamber-based measurement leads to underestimating trace gas fluxes (Kutzbach et al., 2007; T. B. Parkin, Venterea, & Hargreaves, 2012; Pedersen, Petersen, & Schelde, 2010). To address the underestimation problem associated with linear models, a couple of non-linear mathematical models have been proposed. For instance, a quadratic procedure (Quad model) described by Wagner-Riddle et al. (1997) involves fitting a quadratic equation rather than a linear equation to the concentration vs. time data. The non-steady-state diffusive flux estimator (NDFE) model uses an exact solution to a partial differential equation describing the gas diffusion (Livingston, Hutchinson, & Spartalian, 2006). The R-based Hutchison and Mosier method (HMR) model accounts for the diffusivity theory for fluxes from the soil into the chamber, resulting in differential equations that describe an exponential curve of limited flux growth (Parkin et al., 2012; Pedersen et al., 2010; Venterea, 2013). The HMR procedure also allows for the manual selection of flux estimation models in cases where linear models might be appropriate. Comparing flux estimation models over a range of sampling conditions, Parkin et al. (2012) showed that linear models had a lower detection limit than non-linear models, while the HMR model had the highest detection limit compared to the other estimation methods.

2.3.3 Gap-filling methods

Estimating total growing season emissions requires a dataset with average daily gas fluxes measured throughout the observation period. Although gas measurement techniques such as the micrometeorological or automatic closed chamber methods are capable of near-continuous observations, it is not always possible to ensure perfect continuity due to likely device malfunction. Also, researchers who utilize the manual chambers resort to collecting gas samples a few times a week due to the cost associated with gas sampling. As such, the final time series of fluxes will unavoidably have periods with gaps that will need to be gap-filled. Some general interpolation approaches utilized for gap-filling GHG time series include linear interpolation and Piecewise Cubic Hermite Interpolating Polynomial (PCHIP) (Pelster et al., 2011; Abbasi et al., 2020).

2.4 Cropland management practices

Strategies for mitigating soil N_2O emissions generally focus on limiting the activities of nitrifying and denitrifying bacteria (Liebig et al. 2012). Cropland management practices such as fertilizer management, crop rotation including, cover crops, tillage and water management, could potentially reduce emissions by altering the nutrient availability, porosity and moisture in croplands.

2.4.1 Nutrient management

Conventional cropland practices entail using inorganic fertilizer inputs in addition to tillage. Inorganic fertilizer inputs lead to a significant loss of emissions over the growing season, one argument being that their high inputs select for a greater abundance of nitrifiers and denitrifiers. Inorganic fertilizer applied in the previous year has been suggested to have no residual effect on the emissions (Gagnon et al. 2011). Adopting the alternative management practice of adding organic manure improves soil's health and reduces growing season emissions relative to inorganic fertilizer (Chantigny et al., 2010; Halvorson et al., 2016; Abalos et al., 2016; Abbasi et al., 2020). However, this effect has been shown to be inconsistent, with some studies showing contradictory effects of using inorganic fertilizers vs. manure on growing season N_2O emissions. A review study covering the US Midwest showed manure application to result in greater N_2O emissions than inorganic fertilizers (Decock, 2014). Manure contains both inorganic N, which is readily available for plants roots to absorb, and an organic N component which needs to be mineralized before plants can use it. The slow mineralization of the organic component could be responsible for the lower emissions observed with manures (Halvorson et al. 2016; Abbasi et al., 2020). Manures can, however, result in significant emissions during the winter period (Chantigny et al., 2016; Halvorson et al., 2016; Wagner-Riddle & Thurtell, 1998).

Nutrient management is necessary since not all fertilizers applied in most cases can be taken up by plants (Oertel et al., 2016), and the excess could cause water pollution or get emitted as N_2O. Although the IPCC suggests a 1% emission factor for every N fertilizer amount applied (Bell et al., 2016), N_2O emissions have increased exponentially as N inputs increase to exceed crop needs (Iurii Shcherbak, Millar, & Robertson, 2014). The four Rs of nutrient management entail utilizing the right fertilizer source, type, rate and timing. For instance, applying nitrification inhibitors to prevent oxidation of NH_4^+ to NO_3^- has been suggested as an effective method for reducing N losses, especially from fall-applied manure, which bears its N mainly in the form of NH_3 or organic N (VanderZaag et al., 2011). Similarly, reducing the fertilizer rate is a popularly recommended way to mitigate N_2O emissions (Yan et al., 2015). Fertilizers and manures contribute to emissions since they make available substrates (C and N) required for decomposition, nitrification, and denitrification. However, reducing fertilizer or manure rate alone is insufficient since it could lower crop yield (Guangxuan 2015). A better alternative for reducing N_2O emissions entails matching N fertilizer application rates to crop requirements to mitigate N_2O emissions (Yan et al. 2015) (Oertel et al., 2016). The use of yield scaled emissions is a preffered way to quantify optimum fertilizer rate compared to using the direct relationship between applied N rate and N_2O that indirectly suggests low N application rate to mitigate N_2O emission (van Groenigen et al., 2010). Yield scaled emission is a term used for expressing N_2O in relation to crop yield (van

Groenigen et al., 2010). It determines the N_2O efficiency of the cropland while factoring in need to maximize yield.

Recommendations on optimum rates are available, although they are generally based on aggregated data with no distinction for various soil types. For instance, in Quebec, the recommendations for grain corn range from 120 to 170 kg N/ha from urea-based fertilizers, of which 30 to 50 kg ha $^{-1}$ should be side dressed at seeding (CRAAQ, 2010). These general N recommendations are broad compromises envisioned for regional use and may not be suitable for site-specific N management (Miao et al. 2006). However, there are suggestions that the responses of N_2O losses to fertilizer rates can vary with soil, environmental, and climate factors (Eagle et al. 2017). As a commonly used fertilizer in Canada, urea fertilizers which account for about 45% of N fertilizer production (Pelster et al., 2019), are ideal for such studies. It is also necessary to compare N_2O emissions across various N fertilization rates and soil/environmental conditions by incorporating findings from other studies.

2.4.2 Water table management

Subsurface drainage systems that remove excess water in agricultural fields have other benefits, including aeration of the soil profile, improved soil properties, better field machine trafficability, and increased crop yield (Madramootoo, et al., 2007). The controlled drainage system (CDS), a management practice that regulates water table depth, can reduce the export of nutrients to nearby water bodies by increasing the soil water table. During the spring thaw, the tile drainage can freely drain for a while to facilitate field operations (Crézé and Madramootoo, 2019). However, differently than the free tile drainage (FD), after this period, water flow control structures can be set to retain the water in the subsoil and keep the water table at a target level (Cicek et al., 2010). In addition to regulating the water table, the CDS can be used as a subsurface-irrigation system by pumping water through the drainage system during critical drought events. Thus CDS increases water table levels and soil water contents in the field, allowing water and nutrients to be retained at required periods which subsequently leads to improved crop yields (Nangia et al. 2013; Madramootoo et al. 2007; Cicek et al., 2010). However, several studies have suggested that since CDS retains water and nutrients in the field, the practice may enhance N_2O production (Elmi et al., 2000; Van Zandvoort et al., 2017). For instance, CDS plots have been shown to contain less NO_3-N after harvest, suggesting a possibility of more denitrification in these plots (Elmi et al., 2000). A contrary observation by Nangia et al. (2013) suggests that N_2O emissions were more related to total N concentration in the topsoil and not the physical factors directly influenced by CDS practice, such as water table depth and soil water content.

Controlled drainage systems have also been shown to impact crop yield compared to the conventional tile drainage system. For example, Delbecq et al. (2012) reported an increased corn yield of 5.8% to 9.8% under CDS compared to FD, while Ng et al. (2002) recorded an increased yield of about 64%. However, some other studies have reported no evident effect of CDS on crop yields (Drury et al., 2009; Schott et al., 2017). Some researchers have suggested that the potential of CDS to improve crop productivity may be more evident in dry growing seasons or peak growing months (Ballantine & Tanner, 2013; Mejia et al., 2000; Ramoska et al., 2011). For instance, Ramoska et al. (2011) observed up to 10% greater crop yield in CDS treatment plots than in the FD plots. Whereas under moderate conditions, the yield advantage of CDS over FD was insignificant. Conversely, (Tan et al., 1998) observed a slight difference in two years yield between CDS and FD, which was also attributed to dry growing seasons. Observed yields have also been correlated with drought stress intensity occurring at the early and middle growth stages (Cicek et al., 2010; Wang et al. 2016). For example, corn responded most to water and nutrient deficiencies at V12-R1 (Cicek et al. 2010). Cicek et al. (2010) further suggesting that the capacity of CDS to reduce nutrient and water stress in the critical growth stages will depend on weather conditions. In addition to several other factors, including genetics, soil characteristics and water management, weather conditions remain a major factor that influences crop development. This is also the reason weather-based indices such as temperature and water stress indicators are used for crop yield predictions (Mathieu & Aires, 2018). Together, these studies outline the likelihood that the relative impact of CDS on yield will vary with the weather. Exploring the effects of weather variables such as growing season rainfall over a long period could help gauge the efficacy of CDS on yield.

2.4.3 Cropping systems and tillage practices

Incorporating over-winter cover crops such as rye, wheat and oats as agricultural field vegetation covers can protect and improve the soil conditions over the winter. The use of vegetative cover as a mitigation practice for N_2O emissions is centered on the hypothesis that cover crops will absorb more moisture and NO_3^- from the soil, lowering their levels and resulting in reduced N_2O emissions (Shackelford et al., 2019). Having cover crops atop the soil can also slow down the thawing of frozen soils and lower the soil moisture, thereby influencing emission rates. Some studies show no consensus as to the effects of cover crops on either spring thaw or growing season emissions (Iqbal et al., 2015; Parkin et al., 2016). There may be greater emissions from legume cover crops but less from non-legumes (Abdalla et al., 2019). Although cover crops did not show any significant ($p > 0.05$) effects on direct N_2O emissions, they do reduce N leaching which lowers indirect N_2O emissions (Abdalla et al., 2019).

Tillage could affect emissions through its influence on soil aeration, structure, microbial activity, temperature and moisture content, as well as affects gas diffusion through the subsoil to the surface. (Gregorich et al., 2015; Signor & Cerri, 2013). Tillage increases rate of mineralization of soil N, therefore, NO3-N loss can be reduced with zero tillage (Nyborg et al., 1997). Review studies focusing on growing season N_2O losses have shown contradiction regarding the effect of tillage on emissions (Gregorich et al., 2015; Oertel et al., 2016). Several of the studies reviewed showed no significant differences in growing season N_2O emissions between conventional tillage (CT) and no-till (NT) fields (Six et al. (2004).

The efficacy of mitigative interventions on cropland could vary with environmental and soil conditions. For instance, in studying the effect of tillage practices on N_2O emissions, Rochette et al. (2008) showed N_2O losses were higher under minimum tillage (vs. standard tillage) in heavy clay soil but not in loam soil. The higher rates of N_2O flux from minimum tillage practice were attributed to the high-water content and reduced aeration in high clay soils. Similarly, Gagnon et al. (2011) observed greater emissions in soils with relatively low N additions, thus suggesting that other soil condition variables other than N additions also influenced emissions.

Agricultural practices and resulting soil conditions prevalent in the fall prior to a series of over-winter freeze-thaw cycles will influence nutrient availability and moisture conditions. They can, therefore, have a significant impact on the magnitude of emissions. For instance, use of cover crops and reduced tillage may also reduce soil freezing as residue is left for insulation. In addition to incorporating cover crops and reducing tillage, some of the mitigation practices that could be used to lessen the emissions include spring vs. fall manure application and use of nitrogen inhibitors (Wagner-Riddle & Thurtell, 1998; Fan et al., 2022). However, the efficacy of some of these mitigation approaches (e.g., tillage, cover crops) remains under debate (Behnke et al. 2019; Thomas et al., 2017). A comprehensive knowledge of the soil and environmental conditions that drive, or limit emissions is crucial for developing and recommending best N_2O mitigation strategies.

2.5 Overview of analytical methods

The large inter-annual variations in N_2O emissions from field measurements highlight the importance of multiple-year continuous observations (Liu et al., 2014). In many cases, the extent to which these environmental factors, soil properties together with agricultural practices drive, or limit emissions is uncertain. This is in part due to the lack of extensive historical and large GHG flux and environmental datasets which if available could be used to uncover relationships that might not be obvious at first. Some of the analytical approaches that can be used on such datasets are discussed in the following sub sections.

2.5.1 Historical analysis

Historical datasets contain complex relationships and patterns that can help explain causal mechanisms when analysed. Long term studies are increasingly being analysed in agricultural research as large datasets become available. These analyses investigate temporal trends and percentage changes in emissions when an intervention is applied. For instance, long-term studies employing historical data have been used by Basche et al. (2016) to study the impact of cover crops and climate change on crop production. Similarly, Ussiri et al. (2009) assessed long-term impact of tillage on N_2O emissions under intensive corn production. Long term studies have also been used to ascertain the predictive prowess of process models used to study hydrological processes, nutrient cycling and transport, crop phenology and growth, soil freezing and thawing. These models ranged from very simplistic models that target one outcome to complex models that target multiple outcomes (Brilli et al., 2017; Zhang & Niu, 2016).

2.5.2 Meta-analysis

Aggregated datasets from other studies could be analysed using meta-analysis. Meta-analysis is a technique to quantitatively synthesize the findings of various individual studies and it focuses on the direction and magnitude of the studies' effects. It synthesizes results of wide range of studies to determine the effect sizes of interventions. Random effect meta-analysis have been used to quantitatively evaluate the effect of some mitigation strategies on N_2O fluxes (Hu et al., 2019; Shakoor et al., 2021; Basche et al., 2014; Eagle et al., 2017; Shcherbak et al., 2014; Thapa et al., 2016). In addition to assessing the overall impact of unique N_2O mitigation practices, meta-analysis studies have also been used to investigate the impact of other factors such as temperature, moisture, substrates, texture and pH as potential contributing factors to the efficacy of the practices in reducing emissions.

Meta-analysis approach usually includes a literature search, using various research databases, conducted by applying a particular mitigation strategy (e.g.fertilizer, tillage, cover crop) one at a time as a keyword. Other relevant keywords can also be included depending on the topic researched. To be included in the meta-analysis, studies usually met the following criteria: be a field study, have treatments replicated, provide information on the number of daily flux samples collected and the daily mean flux, measure of variance (including P value, standard error, standard deviation (SD), confidence interval) included. When the measure of variance is not provided, an average of the SD from other studies in the same meta-analysis could be applied as suggested in the Cochrane Interactive Learning website (CIL, 2022). A plot digitizer is often used to obtain data where graphs were provided.

The standardized mean difference (SMD), an expression of the size of a treatment effect relative to the study's variability, was used to analyse the data. Also referred to as 'effect size', SMD is used for quantifying the effectiveness of a treatment option relative to another alternative.

$$\mathrm{SMD} = \frac{(<\mathrm{trt}> \; - <\mathrm{alt}>)}{\mathrm{stdev}}$$

where <trt> and <alt> are the mean of the cumulative winter N_2O emissions for the treatment and alternative groups respectively, stdev is the pooled standard deviation of the reported mean emission values from the two groups. The result of the analysis is visualized using a forest plot which shows the estimated effects of individual studies together with their 95% confidence interval. The result summary shows the summary estimate with centre corresponding to the estimate and the edges denoting the confidence interval limits.

2.5.3 Simulating N_2O fluxes using the DNDC model

Process based models attempt to represent the key processes that govern agricultural outcomes such as crop growth and decomposition, soil climate, nitrification, denitrification and fermentation. The use of process models has proven effective for estimating greenhouse gas emissions and assessing potential hydrological and gas fluxes changes. Some of these models include DayCent (Robertson et al., 2018); Root Zone Water Quality Model 2 (RZWQM2) (Jiang et al., 2019); DeNitrification- DeComposition (DNDC) (Jarecki et al., 2018). Among nine carbon and nitrogen cycling models compared in a study, the DNDC was the only model that simulated all N and C gases considered (Brilli et al., 2017). In another previous work (Jiang et al., 2020), RZWQM (root zone water quality model) and DNDC models were compared to ascertain their ability to simulate crop growth and biogeochemical processes occuring in the soil. Both accurately predicted soil temperature and N_2O emissions. RZWQM and DNDC had comparable performance in simulating N_2O emissions. However, both simulated significantly different patterns in the emissions. DNDC simulated more high peaks including freeze-thaw N_2O peaks whereas RZWQM predictions were more constant and did not capture freeze-thaw peak emissions. Jiang's work suggested that DNDC is better situated for freeze-thaw simulations. In the same vein, Dutta et al. (2018) noted the improvements to the mechanisms describing the thermal insulation of snow, residue cover and heat transfer throughout the soil profile.

The Denitrification - Decomposition (DNDC) model is a biogeochemical model of carbon (C) and nitrogen (N) processes in agricultural systems. It was developed mainly for quantifying carbon sequestration and emissions of greenhouse gases (GHG) from agricultural fields (Gilhespy et al., 2014; Giltrap et al., 2010; Li, 2007; Li et al.,1994). The GHG predictions comprise microbe-mediated biogeochemical processes including decomposition, nitrification, denitrification, fermentation and methanogenesis. The rates of processes are simulated by tracking the activities of the groups of microbes whose activation depend on environmental conditions including temperature, moisture, pH, redox potential (E_h) and available substrates in the soil. The model comprises two components. The first is made up of soil, climate, crop growth and decomposition sub-models. These sub models simulate temporal soil temperature, soil moisture and soil water fluxes, pH, E_h and substrate concentration profiles using ecological data (soil, climate, vegetation and anthropogenic activities). The second component consists of the denitrification, decomposition and the fermentation sub models based on the soil environmental variables. It predicts C and N gas fluxes such as CO_2, N_2O, CH_4 and NH_3. The denitrification sub model which simulates N_2O production is an hourly scale model which is activated after any irrigation or rainfall event brings the WFPS to 40%, shown to be the soil water condition that activates the denitrification process in the soil.

Model calibration involves finding one or more optimal values in the model parameter space for achieving goodness of fit of a model's output to a set of data or hypothesis. Model validation involves comparing a new set of experimental data (different from the data set used for the model calibration) against model prediction to see if they agree. When there is an agreement between the modelled result and the experimental data, it implies that the model is correctly simulating the underlying processes. On the other hand, if there is a poor or no agreement, it means that the model's performance in simulating the underlying process is poor. Stopping conditions are a set of criteria that when met by the results of model, a stop in the calibration process is made to avoid overfitting the model. The model is validated after calibration is finished. Some model performance measures for determining if a model can be judged "satisfactory" as suggested in the literature (Jiang, 2018; Moriasi et al., 2015) include the following conditions:

Statistics	Name	Acceptance range	Equation
PBIAS	Percent bias	$-15\% < PBIAS < 15\%$	$\frac{\sum_{i=1}^{n}(O_i - P_i)100}{\sum_{i=1}^{n}(O_i)}$
NSE	Nash-Sutcliffe model efficiency	> 0.5	$1 - \frac{\sum_{i=1}^{n}(O_i - P_i)^2}{\sum_{i=1}^{n}(O_i - O_{avg})^2}$
R^2	Coefficient of determination	> 0.5	$1 - \frac{\sum_{i=1}^{n}(O_i - O_{avg})(P_i - P_{avg})^2}{\sum_{i=1}^{n}(O_i - O_{avg})^2 \sum_{i=1}^{n}(P_i - P_{avg})^2}$
d	Index of agreement	> 0.7	$1 - \frac{\sum_{i=1}^{n}(P_i - O_i)^2}{\sum_{i=1}^{n}(\lvert P_i - O_{avg}\rvert + \lvert O_i - O_{avg}\rvert)^2}$
RMSE	Root Mean Square Error	–	$\sqrt{\frac{\sum_{i=1}^{n}(O_i - P_i)^2}{n}}$
RRMSE	Relative RMSE	$< 30\%$	$\frac{RMSE}{O_{avg}}$

Sensitivity analysis entails testing the model output as various inputs are changed to determine which inputs have the greatest influence on the predicted results. Identifying the input parameters that greatly influence model output can be used to quantify and/or reduce the uncertainty in the predictions arising from the input parameter (Giltrap et al., 2010). Depending on the output of interest, such knowledge could help modelers focus effort on the most important parameters during data collection to reduce uncertainty in the prediction. Focusing attention on such sensitive parameter could help model developers enhance the model's robustness perhaps by further research. Sensitivity analyses is also used for quantifying the degree of uncertainty in the model prediction assuming an imperfect knowledge of the input parameters (Giltrap et al., 2010). A study (Y. Zhang & Niu, 2016) utilizing the DNDC model has shown that N_2O emissions in agricultural soils and pasture systems are most sensitive to clay content, bulk density and SOC.

2.6 Summary and recommendations

Overall, the studies presented so far provide evidence that substrate availability and rapid increase in moisture and temperature are the main regulators of N_2O emissions. There is evidence that freeze-thaw emissions that occur over-winter can be significant although studies have majorly focused on growing season emissions. The studies also showed that agricultural practices such as nutrient management, irrigation and drainage water management, cropping systems and tillage practices could potentially be used in achieving reduction in direct and indirect sources of N_2O emissions.

The extent to which these agricultural practices drive, or limit emissions is uncertain as some literature offer contradictory findings about the efficacy of the practices highlighted. It is still not very clear under what soil and environmental conditions these practices are effective in mitigating emissions. This is in part due to the lack of extensive historical flux, environmental and soil data which if available could be used to investigate the trends and the drivers of the fluxes. Joint studies of various environmental and soil conditions and their responses to management practices are thus crucial for effective N_2O mitigation. They also help in understanding the potential trade-offs or prospects inherent with the practices. Another interesting research focus is to understand how future climate projections could impact over-winter N_2O emissions. In conclusion, there is an opportunity to present robust evidence of the practices benefit by drawing conclusions from historical data analysis, modelling and aggregating secondary data from other studies.

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Connecting Text

Chapter 2 provided a review of literature, including the drivers of cropland N_2O emissions, mitigation practices, and their knowledge gaps. Controlled drainage systems (CDS) have potential agronomic and environmental benefits. However, there have been inconsistent reports regarding these benefits. For instance, some studies have suggested that CDS elevates growing season N_2O emissions with little impact on crop yields. In Chapter 3, the long-term impacts of water table management (WTM) on N_2O emissions and corn yields were assessed by investigating the potential influence of seasonal weather variability (relatively dry vs. normal vs. wet years) on CDS' effectiveness over regular tile drainage (FD). This study includes a one-year field experiment at the St Emmanuel site in Coteau Du Lac, carried out by PhD Candidate Kosoluchukwu Ekwunife, along with thirteen years of historical data (N_2O emissions and corn yields) reported at the same site.

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Contribution of authors:

Ekwunife K.C., the author of this thesis, was responsible for conceptualization, methodology, fieldwork, analysis, data curation, and writing the original draft, followed by reviews and editing. Dr. Madramootoo acquired the funds for the project, provided supervision, and assisted in conceptualizing, reviewing, and editing the manuscript.

Chapter III

Influence of seasonal climate and water table management on corn yield and nitrous oxide emissions

3.1 Abstract

Compared to regular tile drainage (FD), controlled drainage systems with sub-irrigation (CDS) can increase crop yield but could potentially heighten the production and subsequent emission of nitrous oxide (N_2O). Accounting for growing season rainfall categorized under wet, dry and normal, the long-term effects of CDS on crop yield and nitrous oxide (N_2O) emissions were investigated at a commercial corn farm in southwestern Quebec. Based on yield data collected over 12 years at the study site, CDS improved grain yield compared to regular tile drainage (FD), depending on the quantity and temporal distribution of rainfall during the growing season. On average, CDS positively affected grain yield by 17.7% and 3.4% in dry and normal years, but reduced yields by 25% in a wet year. The lower yield under CDS were particularly observed when excessive monthly rainfall (230 mm) occurred during the crop's vegetative period. In three of six years, N_2O fluxes under CDS treatments were greater by 49% than those under FD but 45% lower in the remaining years, implying that – notwithstanding the quantity of growing season rainfall – CDS does not necessarily always produce greater fluxes than FD. N_2O fluxes coincided more with fertilizer application, particularly when associated with a rainfall event. Given that CDS tends to increase crop yield and does not heighten N_2O emissions, it remains a beneficial management practice to be adopted in subsurface drained croplands, where appropriate.

Keywords: controlled drainage, N_2O, crop yield, water-table depth, Southern Quebec

3.2 Introduction

The increasing use of fertilizer N inputs to meet the growing food demand has caused a significant rise in nitrous oxide (N_2O) emissions, a major greenhouse gas that contributes to the greenhouse effect. N losses in the form of nitrates and N_2O emissions are expected to further increase under future climate scenarios (Jiang et al., 2020; Scherger et al., 2022).This important global concern calls for the adoption of farm management practices that can help feed an increasing population without posing further risk to the environment. The use of water table management practices such as controlled drainage structures providing sub-irrigation (CDS) promises increased yields and other environmental benefits, including a reduction of N losses through leaching. Drainage intensity varies over time: during spring field operations and fall harvest, the water table is lowered for machine trafficability, whereas, for the growing season, it is raised to ensure adequate water supply for crop growth. The regulation of water table under CDS can potentially enhance water and nutrient uptake by the crop, thereby increasing yields (Delbecq et al., 2012). The implementation of CDS also increases yield by conserving drainage water in the field, where it can supply crop water needs during dry periods. Studies have reported 35% (Stämpfli and Madramootoo, 2006) and 64% (Ng et al., 2002) greater yields under CDS than FD.

Studies suggest that the potential of CDS to improve crop yield tends to be more evident in dry growing seasons (Ballantine and Tanner 2013; Ramoska et al., 2011; Mejia et al., 2000). For instance, implementation of CDS in a dry year (annual rainfall of 465mm) led to crop yield increases of up to 5.6% compared to FD plots; however, under a moderate year (annual rainfall of 563 mm), the yield advantage of CDS over FD disappeared (Ramoska et al., 2011). Few studies besides that of Skaggs et al., (2012) have drawn upon extensive environmental data to investigate the influence of seasonal weather variability (*i.e.*, wet, normal, or dry growing seasons) on the effectiveness of CDS over FD. Skaggs et al., (2012) showed that CDS did not always have a positive or consistent influence on crop yields. CDS (*vs*. FD) might show little or no impact on yield in unusually dry periods as there might be no drainage water to conserve. Similarly, in wet years when rainfall occurs at the right time to meet crop water requirements, CDS might have little effect on yields. More extensive studies using observed long-term data are needed to clarify the influence of CDS on crop yields in relatively dry *vs.* wet years.

In addition to its effects on yields, CDS provides environmental benefits, including improved water quality (Madramootoo et al., 2007; Drury et al., 2014). By raising the soil water table, CDS can reduce drainage outflow and the export of the nutrients it bears, into nearby water bodies. Skaggs et al. (2012) also reported a roughly 75% reduction in N losses through leaching upon implementing CDS. However, since CDS retains water and nutrients — including nitrate (NO_3^-) — in the field, it could enable the production and subsequent emission of N_2O. Indeed, CDS plots have been shown to contain less NO_3^- after harvest, suggesting that the greater soil moisture in these plots could have enhanced denitrification compared to FD plots (Elmi et al., 2000). Given that the denitrification process in the soil is one of the main pathways leading to N_2O release, environmental concerns have been raised regarding the use of CDS.

Few studies have investigated the effects of water table management (WTM) practices on N_2O emissions (Van Zandvoort et al., 2017; Nangia et al., 2013; Elmi et al., 2005). These studies reporting WTM practices' impact on yield or N_2O emissions were conducted in the short term, sometimes covering three growing seasons or less. Van Zandvoort et al. (2017) and Nangia et al. (2013) found no systematic significant difference in N_2O between FD and CDS field drainage management practices. Elmi et al. (2005) also observed a lack of difference in N_2O emissions between CDS and FD plots. They found that although the N_2O+ N_2 evolution rates were greater in CDS than in FD plots, lower percentages of N_2O relative to overall N_2O+N_2 evolution were observed under CDS plots. Elmi et al. (2005) further suggested that the soil moisture conditions under CDS (water-filled pore space (WFPS) >70%) compared to that of FD (WFPS <50%) allowed complete denitrification of N_2O to N_2. However, to our knowledge, no study has specifically studied how climate (dry *vs.* normal *vs.* wet years) influences the effect of CDS on N_2O emissions.

The potential of CDS to emit greater N_2O fluxes than FD might occur in a normal year rather than a wet year, given the relatively less shallow water table of CDS (*vs*. FD) plots that would not necessitate the complete denitrification to N_2. High N_2O emission rates have been observed at field capacity (≥60% WFPS), which is the critical point for activating anaerobic processes in most arable soils (Linn & Doran, 1984), although a complete denitrification of N_2O to N_2 could occur at more elevated soil moisture condition (WFPS >70%) (Elmi et al. 2005). Conversely, in a relatively dry year, when a drier soil matrix (<60% WFPS) would be expected, conditions would be unfavourable for denitrification, the major process of N_2O production. The current long-term study comprises a one-year field study at a site and 13 years of previously reported data from the same site. The aim is to build upon previous studies to further our knowledge of the impact of WTM practices on N_2O emissions and crop yields in wet, normal and dry years. It is vital to adopt a long-term perspective regarding the effects of WTM practices on N_2O emissions and crop yields to uncover conditions leading to positive or negative impacts of CDS systems.

3.3 Materials and methods

3.3.1 The study site

The study was based on field trials over 14 years (between 1993 and 2018) on a Soulanges sandy loam soil of the Gleysol order planted to a corn (*Zea mays* L.) and located in St. Emmanuel (Côteau-du-Lac, 45°19'N, 74°9'W) Quebec, Canada. The site was arranged in 3 blocks (B1, B2, and B3) with 30 m wide buffer separations also planted to corn (Fig. 3.1). Each block was subdivided into eight 15 m × 75 m plots, separated by vertical plastic sheets to a depth of 1.5 m. Subsurface drainage pipes (76 mm diam.) were laid at a depth of 1.0 m along the centre of each plot. Tait et al. (1995) provide more details regarding design of the controlled drainage system.

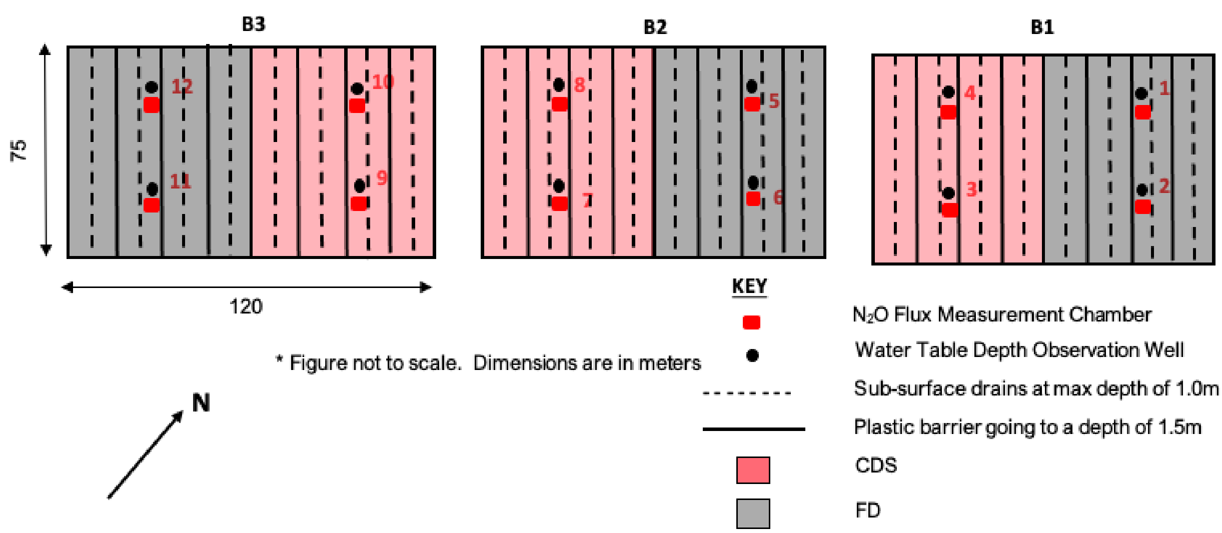

Figure 3.1: Layout of the experimental plots at the St. Emmanuel site

The field experiments consisted of two water table management (WTM) treatments: regular tile drainage (FD) and controlled drainage with sub-irrigation (CDS). The water table depth was maintained by water control tanks located in two 5 by 5 m buildings near the experimental blocks. Details on water-table control facility have been well documented in previous reports (Madramootoo et al., 2001; Zhou et al., 2000). Corn was usually seeded in May with about a 30 kg N ha^{-1} starter N fertiliser banded at seeding. At the corn crop's V6 stage, the rest of Urea fertilizer was broadcast and incorporated into the topsoil. Information on the total N fertilizer applied (kg N ha^{-1}) and mean growing season water table depth (WTD) are shown in Table 3.1.

Table 3.1: Growing season (May - Sep) rainfall (mm), fertilizer applied (kg N ha^{-1}), mean water table depth (m), dry grain yields (mg ha^{-1}) and N_2O fluxes (mg m^{-2} hr^{-1}) in CDS and FD treatments.

Year	Rainfall (mm)	Rainfall category	Fertilizer applied (KgNha-1)	WTD (m)		Yield (Mg ha^{-1})		Difference in yields (%)	N_2O emissions		Source
				CDS	FD	CDS	FD		CDS	FD	
1993	482	Normal	270	0.7	1	8.2	8	2.50%			Zhou et al. (2000)
1994	444	Normal	270	0.7	1	8.9	9.4	-5.30%			
1995	479	Normal	130	1	1.3	11.4	11	2.70%			Mejia et al. (2000)
1996	511	Normal	140	0.9	1.2	7.3	6.8	7.40%			
1998	618	Wet	200	0.6	1.1	6.6	8.8	-25.00%	0.04	0.048	Madramootoo et al. (2001); Elmi et al. (2005)
1999	482	Normal	200	0.7	1.2	9.5	9.7	-2.10%	0.05	0.024	
2000	554	Wet	200	0.7	1.1				0.08	0.047	Elmi et al. (2005)
2001	365	Dry	199	0.8	1.4	9.4	6.9	36.20%			Stämpfli and Madramootoo (2006)
2002	476	Normal	253	0.8	1.4	10.1	7.6	32.90%			
2008	433	Normal	186	0.6	1	12.3	13	-1.60%			Singh et al. (2014)
2009	465	Normal	179	0.8	1.3	10.4	11	-8.00%			
2014	499	Normal	204	0.9	1	9.7	9.6	1.00%	0.13	0.055	Crézé and Madramootoo (2019)
2015	508	Normal	228	0.9	1				0.04	0.074	
2018	360	Dry	222	1	1.2	10.9	11	-0.90%	0.03	0.035	Current study

3.3.2 Crop yield, N_2O flux and other measurements

Grain yields were measured in 12 of the total 14 years reported while N_2O fluxes were measured in six years (Table 3.1). Samples for corn yields were usually taken before harvest. For each treatment (FD or CDS), yield sampling was done along three east-west rows, equally spaced, within each of the three blocks on the site. Along each line, corn plants were cut off at ground level along a 2.5 m row length, and stalks tied together. The fresh weight (f.w.) of the stalk bundles and cobs placed in separate paper bags and labelled by sampling location was measured. The sub-samples of shelled cobs and stalks were then dried at 50°C in a propane dryer. The dry weight (d.w.) of the sub-samples was taken to calculate the total dry biomass of the stalks and dry grain weight for each sampling location.

N_2O and denitrification rates were determined from the topsoil (0–0.15 m depth) using the intact core method with and without acetylene (C_2H_2) inhibition (Elmi et al., 2005) in 1998,1999 and 2000. In 2014, 2015 and 2018, N_2O fluxes was determined using the vented non-steady-state closed chamber method (Crézé and Madramootoo, 2019). The N_2O fluxes were measured weekly from May to a month after fertilizer application and then biweekly for the rest of the growing season. Soil samples were collected at the end of the 1998, 1999, 2000, 2018 growing seasons for soil nitrate (NO_3^-) analysis. Water table depths were measured from the observation wells throughout the growing season using observation pipes of about 1.3 meters. Soil moisture content (WFPS) were measured from the topsoil (0–0.15m depth) alongside gas samples.

3.3.3 Statistical analysis

A rainfall category was used to distinguish between wet, normal or dry years using the ratio between the growing season (May-September) rainfall amount in the year of study and the 30-year (1989-2018) historical average rainfall (479mm). A ratio between 0.9 and 1.1 signifies a normal year while a ratio above 1.1 and below 0.9 signifies a wet year and a dry year, respectively. The yield averages for wet, normal and dry years were calculated for CDS and FD treatments. The yield difference (%) between CDS and FD was also calculated. The N_2O flux averages for wet, normal, and dry years were calculated for CDS and FD treatments. A pairwise t-test comparison of means was used to test significant mean differences among mean N_2O flux, yields, water table depths (WTD) and WFPS in the treatment plots across the years. The statistical analyses were performed using the statistical software JMP®, version 14 (SAS Institute Inc., Cary, NC, 1989-2021) and treatments were deemed significant at $P<0.05$.

3.4 Results and discussion

3.4.1 Effect of water table management on corn yields

We compared grain yields at the study site to investigate the impact of CDS on yield compared to FD, especially relative to wet, normal, and dry years (Table 3.1). While CDS showed greater grain yields in some years, in others, FD showed greater grain yields. The t-test showed that the difference in means between the treatments was not significant ($p = 0.67$). On average, CDS resulted in 3.3% (±16.7%) greater yields than FD over the years at our site. In dry and normal years, the impact of CDS on yield compared to FD was +17.7% (SD = 26.2%) and +3.4% (SD = 12.0%) respectively, whereas in a wet year, its impact was -25%. A positive average impact of CDS over FD was also observed when other studies within North America were reviewed (Table 3.2). On average, a 10 ± 16%) increase in yield was observed. In relatively wet and normal years, CDS increased yield by 11.6 (SD = 1.9%) and 17.1 (SD = 24.1%) respectively, while in relatively dry years, the increase was 7.3 (SD = 9.0%).

Table 3.2: Effect of controlled drainage (CDS) on crop yields, compared to free drainage (FD) in relatively dry, normal and wet years, from other North American studies.

Source	Location	Crop	Year	Rainfall Ratio (yr/avg)	Effects of CDS on crop yield
Drury et al. 2009	Ontario	Soybean	1995	0.8	-11.00%
		Corn	1996	1	-5.90%
		Soybean	1997	1	10.00%
		Corn	1998	0.7	0.60%
Schott et al. 2017	Iowa	Corn and Soybean	2011	0.8	No effect
			2012	0.8	No effect
			2013	1	No effect
			2014	1	No effect
			2015	1	No effect
Poole 2011	North Carolina	Corn	1991	1.2	5.80%
			1993	0.4	17.30%
			2007	0.7	10.80%
			2008	0.6	4.70%
			2009	1.2	3.80%
			2010	0.8	18.70%
			2011	0.7	6.50%
		Soybean	1990	1.1	17.10%
			1992	1	5.50%
			1994	1.1	11.70%
			1998	0.8	11.40%
			2008	0.7	6.50%
			2009	1.6	2.00%
Tan et al. 1998	Ontario	Soybean	1995	0.7	No effect
			1996	0.6	No effect
			2003	1.3	No effect
Ng et al. 2002	Ontario	Corn	1996	1	64.20%

Our study results showed that the water table depth (WTD) obtained in CDS (vs. FD) plots was shallower in all observed years ($p < 0.05$) and is likely responsible for the average positive impact on yield (Table 3.1). Plots under CDS are believed to retain some of the N loads that FD plots would otherwise discharge to surrounding water bodies, therefore, the potential for higher yields (Skaggs et al., 2012). Madramootoo et al. (2001) supported this idea when they showed the overall seasonal NO_3^--N concentration in drainage water from CDS to be up to 80.3% lower than that from FD — an indication that a portion of the soil NO_3-N was retained in the CDS plots. However, Madramootoo et al. (2001) showed that though soil NO_3^--N could potentially be higher in CDS (*vs.* FD) plots, this did not necessarily lead to greater yield in the CDS plots. A possible explanation for this is that the applied N fertilizer was sufficient to cater to crop N requirements; hence, any extra N resulting from CDS treatment might have only contributed to luxury consumption by the crop, and not led to a noticeable increase in yield over FD.

Although previous studies suggest that yield increases due to CDS could be more evident in dry growing seasons (Ramoska et al., 2011; Mejia et al., 2000), the two arid years (approx. 365 mm of rain) experienced at this study site (2018 and 2001) showed contrasting grain yield values. Marginally (0.1%) greater yields occurred under FD (*vs.* CDS) in 2018, whereas in 2001, 36.2% greater yields were obtained under CDS (*vs.* FD). The mean water table depths (WTD) in the CDS treatment plots in 2018 (target depth of 0.80 m) and 2001 (target depth of 0.6 m) were 1.04 m and 0.77 respectively while the FD plots were >1.18m. Setting the target water table depth to 0.80m was to avoid flooding the crop root zone should heavy rainfall occur. Sub-irrigation events in 2018 did not elevate the water table level in the CDS plots closer to the target depth as in 2001.

Since it was unclear what could have brought about such disparity, we further evaluated the seasonal rainfall distribution to assess the impact of CDS on yield (Fig. 3.2). The within-season rainfall variability was evaluated by summing up daily rainfall for each month in the growing season (May to September) within each year. Seasonal rainfall distribution could result in periods with excessive rainfall or droughts that could hamper crop growth, grain yield and CDS' ability to positively impact yield. Water and nutrient demands vary with the crop growth stage. Corn responds most to water and nutrient deficiencies at the vegetative and early reproductive stages (Cicek et al., 2010). Looking at the two dry years with significant contrasting grain yields, the rainfall in May 2001 was 116 mm compared to 52 mm in May 2018. Water table controls for CDS were activated in early May, immediately after the fields were drained in early spring. High rainfall amounts occurring after activating the controlled drainage imply that CDS could retain water in the root zone for the vegetative corn stage. This could explain why the rainfall and sub-irrigation events in 2018 did not elevate the water table levels in the CDS plots close to the target depth like in 2001. This finding agrees with Skaggs et al. (2012) who also suggested that there might not be available drainage water to conserve in CDS plots in unusually dry periods. The difference in May rainfall amounts could thus explain the considerable variations between 2018 and 2001 yields.

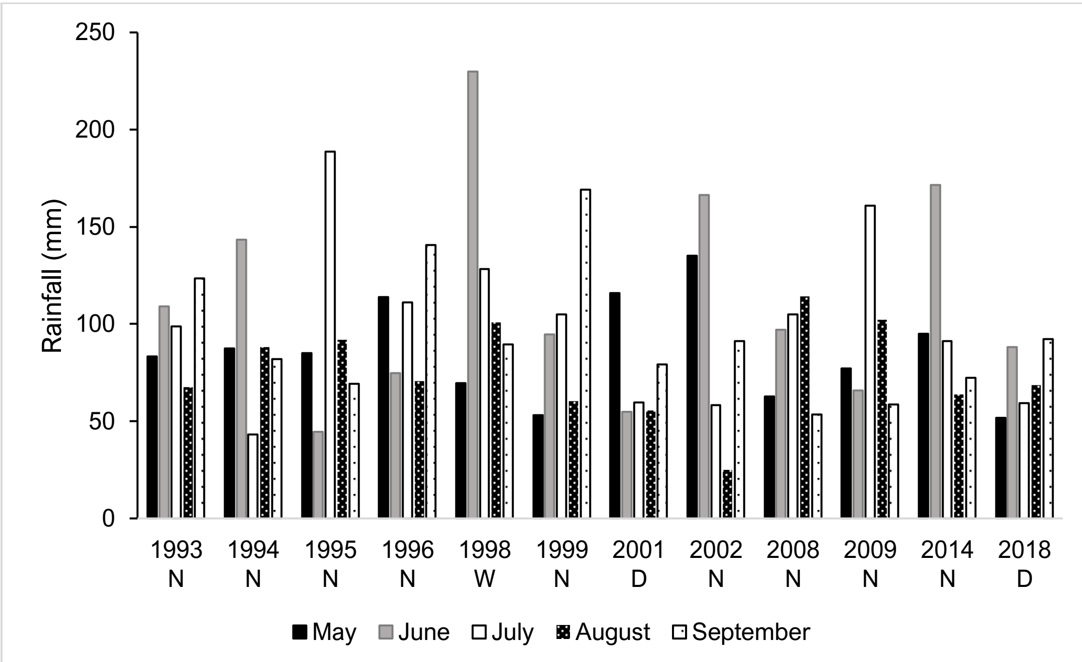

Figure 3.2: Seasonal rainfall distribution for studies that reported grain yields at the study site. N, D, and W represent normal, relatively dry, and relatively wet years, respectively.

Contrary to expectations, our results showed a significantly lower yield (-25%) under CDS in the wet year of 1998. However, this reduced yield likely resulted from the excessively high (229.8 mm) rainfall in June (Madramootoo et al., 2001) as shown in Fig. 3.2. In studying the impact of rainfall extremes on corn yield in the United States, Li et al. (2019) showed excessive rainfall to have the most negative impact in June, the corn crop's vegetative period. Madramootoo et al. (2001) suggested that while a static water table depth may suffice in a dry year, better control of the water table would be required in wet years to prevent yield loss. Setting the water table at a shallower depth requires that sub-irrigation be switched off more frequently to prevent flooding in the crop root zone.

Although there were years when there were no apparent effects of CDS on yields, CDS clearly has the potential to increase yield under certain conditions. This potential depends on the seasonal rainfall distribution. It is possible that year-to-year variability in the effect of CDS implementation on yields could result from other factors, including the farmer's management of the CDS systems, variations in total weekly rainfall, and differing corn variety heat units. However, these factors are beyond the scope of this study.

3.4.2 Effect of water table management on N_2O fluxes

The mean N_2O fluxes under CDS and FD treatments for the six reported years are presented in Fig. 3.3. In three of the six years, N_2O fluxes under CDS treatments were greater by 49% than those under FD but 45% lower in the remaining years. The seasonal mean WTD were significantly higher under CDS in the six years ($p < 0.05$) (Fig. 3.3). Likewise, the seasonal mean soil water content expressed in percentage of water-filled pore space (WFPS) were higher under CDS in those years except 2018 (dry year) (Fig. 3.3). The mean fluxes were assessed under wet, normal and dry years. Mean N_2O fluxes were (0.06 and 0.05 mg m^2 hr^{-1} for CDS and FD, respectively) in wet years, (0.07 and 0.05 mg m^2 hr^{-1} for CDS and FD, respectively) in normal years, and (0.03 and 0.04 mg m^2 hr^{-1} for CDS and FD, respectively) in a dry year. Average N_2O fluxes were lowest in the dry year (2018) (Fig. 3.3).

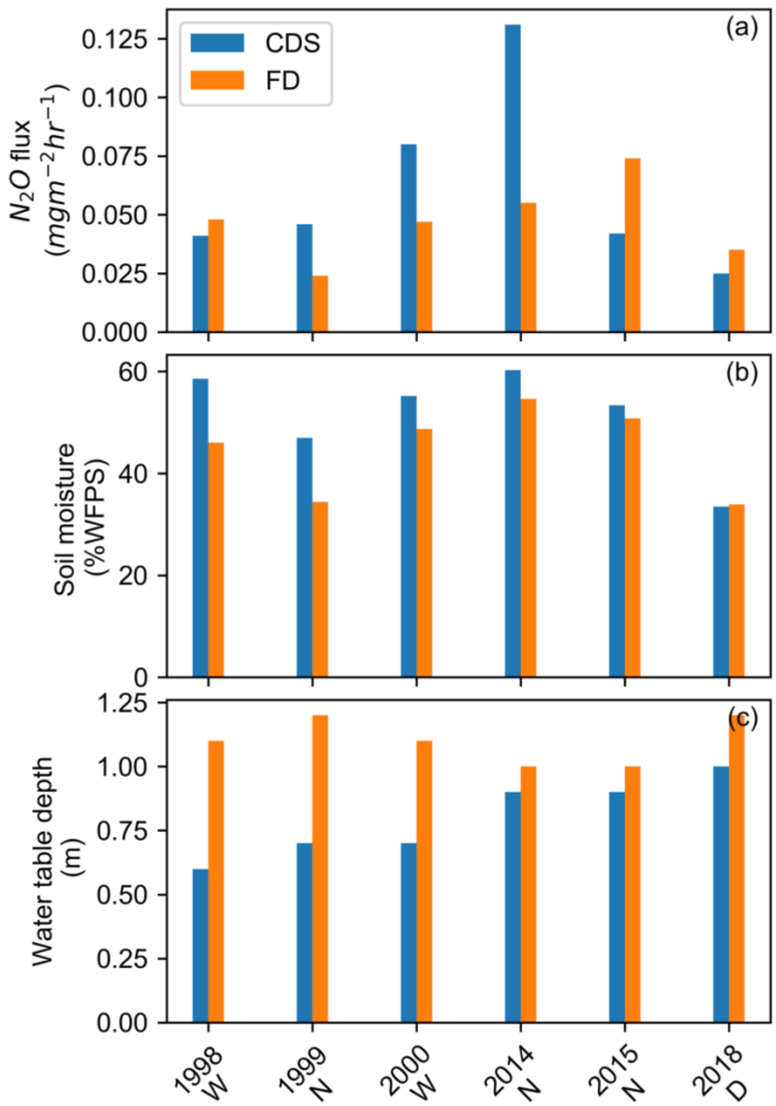

Figure 3.3: Growing season (May - Sep) mean (a) N_2O fluxes (mg m^{-2} hr^{-1}), (b) soil moisture (%WFPS) and (c) water table depth (m) under CDS and FD treatments at the study site. N, D, and W represent normal, dry, and wet years, respectively. The target water table depths for 1998 -2000 study was 0.60 m and 0.80m for 2014, 2015 and 2018.

The pairwise t-test revealed that the differences between mean N_2O fluxes from CDS and FD was not statistically significant ($p = 0.42$). No systematic pattern in the differences in N_2O fluxes was observed between CDS and FD plots in both normal and wet years. Based on individual years, the seasonal mean of daily N_2O fluxes in 2018 for the CDS and FD treatments were 0.025 ± 0.020 and 0.035 ± 0.030 mg m^{-2} hr^{-1}, respectively. Mean N_2O flux from the CDS (*vs.* FD) plots was greater in 2014 (0.131 and 0.055 mg m^2 hr^{-1} for CDS and FD, respectively) but lower in 2015 (0.042 and 0.074 mg m^2 hr^{-1} for CDS and FD, respectively) (Fig. 3.3). Similar mixed results were also reported for 1998 – 2000 (Elmi et al., 2005). Microbial activities that cause the release of N_2O fluxes in elevated soil moisture conditions occur predominantly in the topsoil. The shallower water table of CDS plots (*vs*. FD) can increase the WFPS of the topsoil which could subsequently affect N_2O emissions. For instance, in 2014 peak N_2O fluxes in CDS and FD plots occurred when soil WFPS were 91% and 75%, respectively (Crézé and Madramootoo, 2019). In that same year, CDS increased topsoil water-filled pore space by 4% on average while the average seasonal fluxes of CDS plots were 2.3 times that of FD plots. Our study shows that this higher WFPS observed in the CDS plots does not always lead to greater N_2O emissions. For instance, in 1998, Elmi et al. (2005) observed that the peak N_2O flux was measured on FD plots on a day when soil moisture under CDS was higher (WFPS >70%) than FD (<50%). Although a greater denitrification rate was observed under CDS than FD on that day, the quantity of N_2O fluxes was similar for the two WTM treatments due to a possible complete denitrification process of N_2O to N_2 (Elmi et al., 2005). Similarly for 2015, Crézé and Madramootoo (2019) suggested that although CDS may create anaerobic soil conditions that could trigger increased denitrification rates, the N_2O fluxes from the CDS treatment plot may not be greater. No significant WFPS difference was observed between the two treatments in 2018 (a dry year), even though the mean water table level in the CDS plots was more than 0.14 m shallower than in the FD plots. It is possible that the rate of upward flux did not differ sufficiently between the two treatments to cause a significant difference in the topsoil moisture content between the CDS and FD treatment plots.

Application of mineral N releases NO_3^--N into the soil, a substrate for N_2O production (Van Groenigen et al., 2010; Gregorich et al., 2015) and CDS can reduce N losses through leaching by retaining them within the plots (Madramootoo et al., 2007). Topsoil NO_3^--N measured at the end of the growing season in 1998,1999, 2000 and 2018 is shown in Table 3.3. The average topsoil NO_3^--N under CDS was lower than under FD in 2018, though this difference was not significant ($p = 0.23$). Elmi et al. (2005), who also reported topsoil NO_3^--N tending to be lower under CDS than FD in 1998 – 2000, observed denitrification (N_2O+N_2) rates to be twice as high under CDS than FD. However, N_2O emissions in their study were not consistently higher under CDS (Fig. 3.3), denoting a greater reduction of N_2O to N_2 in the CDS plots. Notably, the three years of Elmi et al. (2005) study were either wet or normal, so the potential for complete denitrification was greater than in a dry year (2018). It can thus be suggested that the difference in topsoil NO_3^--N between FD and CDS may not be adequate to explain the differences in observed N_2O fluxes.

Table 3.3: End of growing season soil (0–0.15m) NO_3^--N (kg ha-[1])

Year	NO_3^--N (kg ha $^{-1}$)	
	CDS	**FD**
1998	3.0	13.6
1999	17.1	31.3
2000	4.6	7.6
2018	7.1	8.0

The greater mean N_2O flux observed under FD in 2018 is explained by a peak N_2O flux observed under FD but not under CDS at the beginning of the growing season (Fig. 3.4). A possible explanation for such short-term peak flux is that it originated from microbial hotspots, where process rates are ten- to a hundred-fold greater than in the surrounding bulk soil (Kuzyakov and Blagodatskaya, 2015). These findings altogether show that although the shallower water table of CDS plots increase the topsoil WFPS, particularly in wet and normal years, the potential for a complete denitrification process could lead to a lack of treatment differences in the N_2O fluxes. In contrast, in a dry year, the WTD could reach such depths that upward flux to the surface would be minimal, resulting in no distinct topsoil moisture or treatment-driven N_2O flux differences being observed between the treatments. The current study's findings, which show no systematic differences in N_2O fluxes between WTM treatments, agree with those of Nangia et al. (2013) and Van Zandvoort et al. (2017) that indicated that CDS does not pose an environmental menace with regards to N_2O emissions.

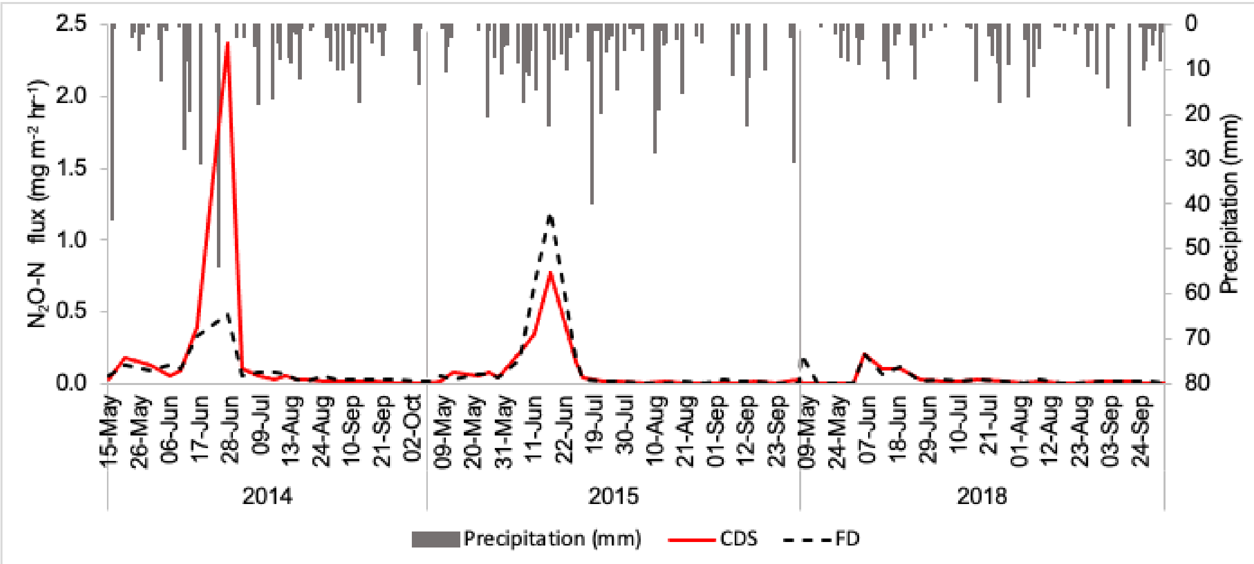

Figure 3.4: N_2O fluxes from free drainage (FD) and controlled drainage (CDS) treatment plots over the 2014, 2015 and 2018 growing seasons.

Our study shows that the rainfall category (wet vs. normal vs. dry) could impact CDS' control on topsoil WFPS. For instance, contrary to the wetter years, sub-irrigation events in 2018 (dry year) did not elevate the water table level in the CDS plots closer to the target depth, so the topsoil WFPS did not vary. However, using the rainfall category alone is not enough to fully understand WTM impact on N_2O fluxes. Most major peaks in N_2O fluxes in CDS and FD occurred after fertilizer was applied in June. These elevated fluxes continued for about two weeks before returning to baseline values. Crézé and Madramootoo (2019) observed that these peaks coincided with high rainfall events that made easily decomposable substrates available and renewed mineralization by microbes. For instance, after a single rainfall of >24mm in 2014, N_2O production rates increased and were 100 times greater than the maximum value from the CDS plots. Crézé and Madramootoo (2019) further showed that while sub-irrigation increased WFPS by 4% in 2014, rainfall of about 30mm raised WFPS by 30%. However, this association between elevated fluxes and high rainfall events was not observed in 2018, probably because 2018 was a dry year with relatively lower growing season rainfall, compared to a normal or wet year (Fig. 3.4). Since the occurrence of N_2O fluxes in the field may depend more strongly on soil nitrogen availability, particularly when associated with rainfall events, future studies should consider rainfall events, which can also trigger N_2O fluxes.

3.5 Conclusions

The central point of this study was to quantify WTM impacts on yield and N_2O fluxes in a temporal context while accounting for the growing season rainfall. Our study has shown that CDS can either negatively or positively affect grain yield for a given growing season, depending on rainfall distribution and water table management practices. On average, CDS resulted in 3.3% greater yields than FD over the years at our site. While positive average yields under CDS were observed in dry and normal years, a negative average yield observed in the wet year was mainly due to the excessive monthly rainfall (230 mm) that occurred during the crop's vegetative period. The second major finding was that there was no clear pattern of increase in N_2O fluxes associated with CDS (vs. FD) plots. In three of six years, fluxes from CDS plots were higher but lower in the other three years. This study showed that in wetter years where CDS plots increase the topsoil WFPS, the potential for complete denitrification of N_2O to N_2 could lead to a lack of treatment differences in the N_2O fluxes. In contrast, in a dry year (360 mm growing season rainfall), low WTD caused upward water flux to the surface be minimal, resulting in no distinct moisture or N_2O flux differences being observed between the treatments. In this study, peak N_2O fluxes from both treatments mainly coincided with high rainfall amounts following fertilizer application, indicating that more N_2O emissions could be expected in relatively wet years than in dry years. Taken together, these findings suggest that CDS has the potential to increase corn yields but does not necessarily result in elevated N_2O emissions, thus making it a beneficial management practice.

3.6 References

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Connecting Text

In Chapter 3, the long-term impacts of water table management (WTM) on corn yield and N_2O emissions were assessed. In Chapter 4, fertilizer management was assessed as another N_2O mitigation practice focused on the growing season since substrate availability has the most effect on N_2O fluxes. Adding fertilizers supplies N to the soil, an essential substrate for N_2O emissions. However, fertilizer needs vary across fields, indicating that soil available N required for crop growth as well as N_2O emissions is affected by soil texture. Therefore, Chapter 4 of this thesis investigated the impact of soil texture on optimal fertilizer rates for achieving low yield-scaled N_2O emissions. This study comprises a one-year experimental study carried out by PhD Candidate Kosoluchukwu Ekwunife and findings from other studies carried out in North America.

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Contribution of authors:

Ekwunife K.C., the author of this thesis, was responsible for conceptualization, methodology, fieldwork, analysis, data curation, and writing the original draft, followed by reviews and editing. Dr. Madramootoo acquired the funds for the project, provided supervision, and assisted in conceptualizing, reviewing, and editing the manuscript.

Chapter IV

Effects of fertilizer rate on yield-scaled nitrous oxide emissions from two soil types

4.1 Abstract

Mineral N fertilizer application has increased dramatically to boost crop yields and meet the growing food demand. This has led to increasing N_2O emissions from cultivated fields. Best management practices have been proposed to mitigate these emissions, including the 4Rs (right source, right rate, right time, right place) of nutrient management. However, fertilization rates vary depending on soil type and cropping system. Accordingly, these factors must be considered when developing N_2O emission reduction scenarios. A field study was undertaken to evaluate the influence of three nitrogen fertilization rates (140, 180 and 220 kg N ha^{-1}) on N_2O emissions and grain corn (*Zea mays* L.) yield from sandy loam and silty clay soil field sites situated in southwestern Quebec, Canada. Cumulative N_2O emissions from the sandy loam soil were up to three-fold greater than those from the silty clay soil. Grain yields increased with the N fertilization rate. Crop nitrogen uptake and yields were greater on the sandy loam than on the silty clay. Yield-scaled N_2O emissions from the silty clay soil were lowest at the 180 kg N ha^{-1} fertilization rate, compared to 140 kg N ha^{-1} for the sandy loam. Incorporating results from five other studies, we found that under corn production, yield-scaled emissions from the poorly drained fine-textured soils were five-fold greater than well-drained (coarse- and medium-textured) soils. These results demonstrate the need to consider soil textural differences when recommending fertilizer rates to reduce N_2O emissions.

4.2 Introduction

Nitrous oxide (N_2O) is a major greenhouse gas (GHG) with approximately 300-fold greater warming potential than carbon dioxide (CO_2). Assuming a business-as-usual scenario, global anthropogenic N_2O emissions are projected to be 83% greater in 2050 than they were in 2005 (Davidson and Kanter, 2014). The agricultural sector currently accounts for roughly two-thirds of anthropogenic N_2O emissions, but the increasing use of mineral N fertilizer to raise crop yields and meet a growing food demand, risks worsening the situation (Davidson and Kanter, 2014).

Increased application of mineral N releases reactive N forms ($NH_4^+ + NO_3^-$) into the soil, the substrates for N_2O production. If these forms exceed crop requirements, arable soils will emit more N_2O (Van Groenigen et al., 2010; Gregorich et al., 2015). Management practices that maximize plant N use efficiency (NUE), while minimizing N_2O emissions, are therefore crucial. One management approach to reducing cropland N_2O emissions, is to moderate N inputs to arable soils by reducing fertilizer rates and minimizing excess reactive N in the soil (Van Groenigen et al., 2010). Minimizing excess soil N not only reduces direct N_2O emissions, but also indirect losses such as N leaching and ammonia volatilization.

In order to achieve global food security, any reduction in N fertilization rate implemented to minimize N_2O emissions must still ensure economically optimum yields (Nasielski et al., 2020). A yield-scaled emissions approach, in which N_2O emissions are expressed in relation to crop yield, could inform crop growers, agronomy advisors, and policymakers of the optimal quantity of fertilizer to apply for specific crops and soil types. In a meta-analysis of 19 studies, Van Groenigen et al. (2010) showed that yield-scaled N_2O emissions for non-leguminous field crops were lower (8.4 g N_2O-N kg^{-1}) at N application rates ranging from 180–190 kg N ha^{-1} than at lower rates, and significantly greater (219%) at an elevated N fertilization rate (301 kg N ha^{-1}). Similarly, Zhao et al. (2017) found yield-scaled emissions below the average economic optimum yield, were achieved with an N fertilization rate of 171 kg N ha^{-1} (0.18 g N_2O-N kg^{-1}), and increased by 67% at 250 kg N ha^{-1}. The magnitude of differences in yield-scaled values reported in both studies likely resulted from the effect of variations in regional climate, crops, and fertilizer management practices.

As recommended fertilizer rates for various crops are based on aggregated data, they are not representative of site-specific conditions. Fertilizer needs vary across fields due to spatial variability of soil chemical and physical properties within fields, highlighting the fact that both soil available N required for crop growth and N_2O emissions are affected by soil texture (Schimel and Bennett, 2004; Mamo et al., 2003; Inman et al., 2005). Fine-textured soils with smaller pore sizes have a greater surface area for absorbing substrates particularly NH_4^+, while soils with larger pore sizes allow for more aerobic conditions. Soil texture influences N_2O emissions by affecting the substrates available for microbial metabolism and redox conditions.

Previous studies have assessed the influence of soil texture on N_2O emissions. Fine-textured soils with a greater clay content have been reported to emit at least twice as much N_2O than coarse-textured soils (Rochette et al., 2008; Li and Kelliher, 2005). In contrast, other studies have shown a negative correlation between N_2O emissions and clay content, with three-fold lesser N_2O emissions from high clay content soils (including clay-amended and native clay soils) compared to soils with less clay (Pratt et al., 2016; Wang et al., 2018). While the reason for these contrasting results remains unclear, it could be related to variations in soil and environmental variables (e.g., soil texture, temperature, moisture), and the availability of nitrogen and carbon (Rochette et al., 2018). For instance, in low precipitation regions, clayey soils may not have low soil oxygen (redox potential is not in a range favorable for N_2O) but will adsorb NH_4 which can result in lower nitrification and thus less N_2O from nitrification. Also, in very wet climates clay can cause complete denitrification to N_2. Further studies are thus required to clarify these inconsistencies.

For proper site-specific N rate management and subsequent N_2O emission reduction, studies on the effect of fertilizer N rates across soil textural types are essential. Accordingly, the present study's objectives were to (*i*) compare N_2O emissions, N uptake, and corn yields from two soils as affected by N fertilizer rate (*ii*) assess the influence of soil reactive nitrogen, soil temperature and soil moisture on N_2O fluxes, and (*iii*) quantify and compare yield-scaled emissions across various N fertilization rates and soils with varying clay content, by incorporating findings from other studies. Our results will inform policy directives on appropriate mitigation strategies for crop growers to adopt and improve emissions accounting by highlighting factors that could affect emissions.

4.3 Materials and methods

4.3.1 Experimental site

A field study was conducted in Southern Quebec at two different arable fields, one situated in Saint-Emmanuel-du-Côteau-du-Lac (45°19'N, 74°9'W) and another in Sainte-Anne-de-Bellevue (45°26'N, 73°56'W), during the 2019 growing season (April-October). The soil at the St. Emmanuel site is classified as sandy loam, while the Ste-Anne-de-Bellevue site is classified as a silty clay soil. The surface layer of the soils (0-0.20 m) were sampled early in May and analysed for particle size distribution, SOC and pH (Table 4.1). Corn was seeded on the sandy loam on May 18, and the silty clay on June 10. The delay in planting at the silty clay site was due to wet field conditions early in the growing season. Both fields were chisel-plowed after harvest in the previous year and harrow-disked in the spring prior to planting. Each experimental field comprised three blocks, with a buffer separation of 3 m between blocks. Each block was divided into 3 plots of 15 m × 12 m at the sandy loam site and 5 m × 12 m at the silty clay site. Fields were drained with subsurface tile drainage buried at a depth of 1 m. Urea was applied at three different rates: low (140 kg N ha^{-1}), medium (180 kg N ha^{-1}) and high (220 kg N ha^{-1}). For each treatment, 35 kg N ha^{-1} was banded at seeding, and the remainder was applied at the 6-leaf stage of corn growth. These three fertilizer treatments were replicated three times in a randomized complete block design at each field.

Table 4.1: Soil properties over the sampling depth 0-0.20 m

	sandy loam	**silty clay**
Particle size classes		
% sand	55	11
% silt	38	41
% clay	7	48
Field capacity (%WFPS)	37.7	77.8
pH	7.1	7.5
SOC (% weight)	1.4	2.7

4.3.2 Crop biomass and plant N uptake

The crop (grain and stalks) yield and plant N uptake measurements were undertaken on October 8 and October 21 at the sandy loam and silty clay fields, respectively. Sampling was done in both fields along three 1 m rows for each fertilizer treatment plot. Stalks and cobs from each row were harvested separately. The wet mass of the stalks and cobs was measured per sampling row. Stalks were then chopped, and a weighed subsample was collected. The wet stalk and cob samples were then dried in an electric oven for five days at 50°C. The cobs were shelled, and the dry weight of the grains and stalks was measured to calculate the water content and dry mass of the stalks and grains. Representative seed and cob samples were finely ground and analyzed for total nitrogen (mg N g^{-1}) using a sulfuric acid and hydrogen peroxide oxidative digestion carried out at 340°C.

4.3.3 Gas measurements

N_2O fluxes were measured from each field during the spring thaw period (April) and growing season (May to October) using non-steady-state soil chambers (Crézé and Madramootoo, 2019). Spring thaw flux measurements were taken between April 2 and May 8 and growing season flux measurements were conducted between May 9 to October 2. Samples were collected three times per week from snowmelt in April to a month after fertilizer application, and then once every two weeks for the rest of the growing season. Sampling was focused around emission trigger events such as fertilizer application, tillage and rainfall events when gas fluxes were expected to be high. Gas measurements were taken from the middle of each plot between 1000H and 1300H to minimize diurnal variability (Parkin et al., 2010). Gas samples were drawn from the chamber headspace using 20 ml polypropylene syringes. The samples were injected into pre-evacuated 12 ml exetainers and transported to the laboratory for greenhouse gas concentration analysis. Employing the R-based non-linear Hutchison and Mosier (HMR) model, the N_2O-N flux was calculated as the rate of change in N_2O-N concentration over time (Pedersen et al., 2010). Cumulative fluxes over the growing season were calculated by plotting daily fluxes over time, gap-filling daily fluxes for days when samples were not taken and summing the fluxes. The gap-filling was performed using linear interpolation. The yield-scaled N_2O emissions were calculated by dividing N_2O-N emissions by grain yield (Mg ha^{-1}).

4.3.4 Soil and meteorological data

Soil reactive nitrogen (nitrate and ammonium) concentrations ($[NO_3^-]_{soil}$ and $[NH_4^+]_{soil}$, respectively) were measured every month of the growing season. During each sampling event, three soil samples were collected from each plot at two depths: 0-0.20 m and 0.20-0.50 m. The samples were taken close to the chambers, moving clockwise to avoid resampling at the same spot. On the same day, soil samples were extracted with 2M KCl and analyzed for NO_3^--N and NH_4^+-N, using a multi-channel Lachat autoanalyzer (Maynard et al., 2009). For total soil organic carbon, soil samples were collected during the pre-growing season in May from the experimental plots at depths 0–0.20 m and determined by the loss-on-ignition method. Soil carbon content was estimated by dividing organic matter (%) by two (Pribyl, 2010). Volumetric moisture content was measured during gas sampling using a Delta-T soil moisture Theta Probe Model ML2x, vertically inserted into the soil to a depth of 0.10 m. The probe was calibrated using soil samples from both sites. Water-filled pore space WFPS (%) was calculated as the ratio of volumetric water content and soil porosity. Soil temperature was obtained during gas sampling for the top 0.10m using a handheld thermometer (Habor 022). Rainfall and air temperature data for both sites were collected from weather stations close to the sites: Côteau-du-Lac and Sainte-Anne-de-Bellevue.

4.3.5 Statistical analysis

Using JMP statistical software (version 14.1, SAS Institute, Cary, NC), ANOVAs were used to test the differences in N_2O-N emissions, yield and N uptake for the different soil types and N rates. It also served to test differences in soil inorganic N between two ranges of soil depth and by sampling month. Regressions were carried out to test if the N_2O-N fluxes correlated with soil temperature and moisture. A statistical probability of $p \leq 0.05$ was considered significant for all comparisons of means.

4.4 Results

4.4.1 Sandy loam

4.4.1.1 Weather, soil chemical characteristics and crop biomass

Total seasonal rainfall during the growing season (May to October) was 635.7 mm. Eleven rainfall events of over 20 mm contributed roughly 57% of total growing season rainfall (Fig. 4.1); however, these heavy rainfall events did not seem to cause a corresponding significant change in WFPS. The WFPS at this site ranged from 31% to 88%. Soil temperatures at a 0.10 m depth ranged from 0.1°C to 24°C. The $[NO_3^-]_{soil}$ and $[NH_4^+]_{soil}$ for the months of May through September, measured at depths of 0-0.20 m and 0.20-0.50 m are shown in Fig. 4.2. The residual pre-planting $[NO_3^-]_{soil}$ at 0-0.20 m depths were 2.7, 2.4 and 3.3 mg NO_3^--N kg^{-1} for high, medium and low N-fertilization plots, respectively, whereas after the bulk of the fertilizer was applied in June, they increased sharply to 14.4, 11.6 and 10.8 mg NO_3^--N kg^{-1}. This increase in $[NO_3^-]_{soil}$ was followed by a gradual decline, which started in July. Over the growing season, June and July showed the greatest $[NO_3^-]_{soil}$, and these were significantly different from the $[NO_3^-]_{soil}$ for the remaining months. As expected, greater $[NO_3^-]_{soil}$ and $[NH_4^+]_{soil}$ was found at the 0-0.20 m depth than the 0.20-0.50 m depth. No statistical difference was observed for $[NH_4^+]_{soil}$ among the N fertilizer treatments. The $[NH_4^+]_{soil}$ was greatest in June, and significantly different statistically ($p \leq 0.05$) from $[NH_4^+]_{soil}$ in the remaining months. The total N uptake in the plants in the sandy loam field ranged from 15.3 mg N g^{-1} (low-rate fertilizer plots) to 17.8 mg N g^{-1} (high-rate fertilizer plots) (Table 4.2). The difference in total dry yields of corn under the high, medium and low fertilization rates were 19.2, 19.1 and 18.6 Mg ha^{-1} (Table 4.2) and were not statistically significant.

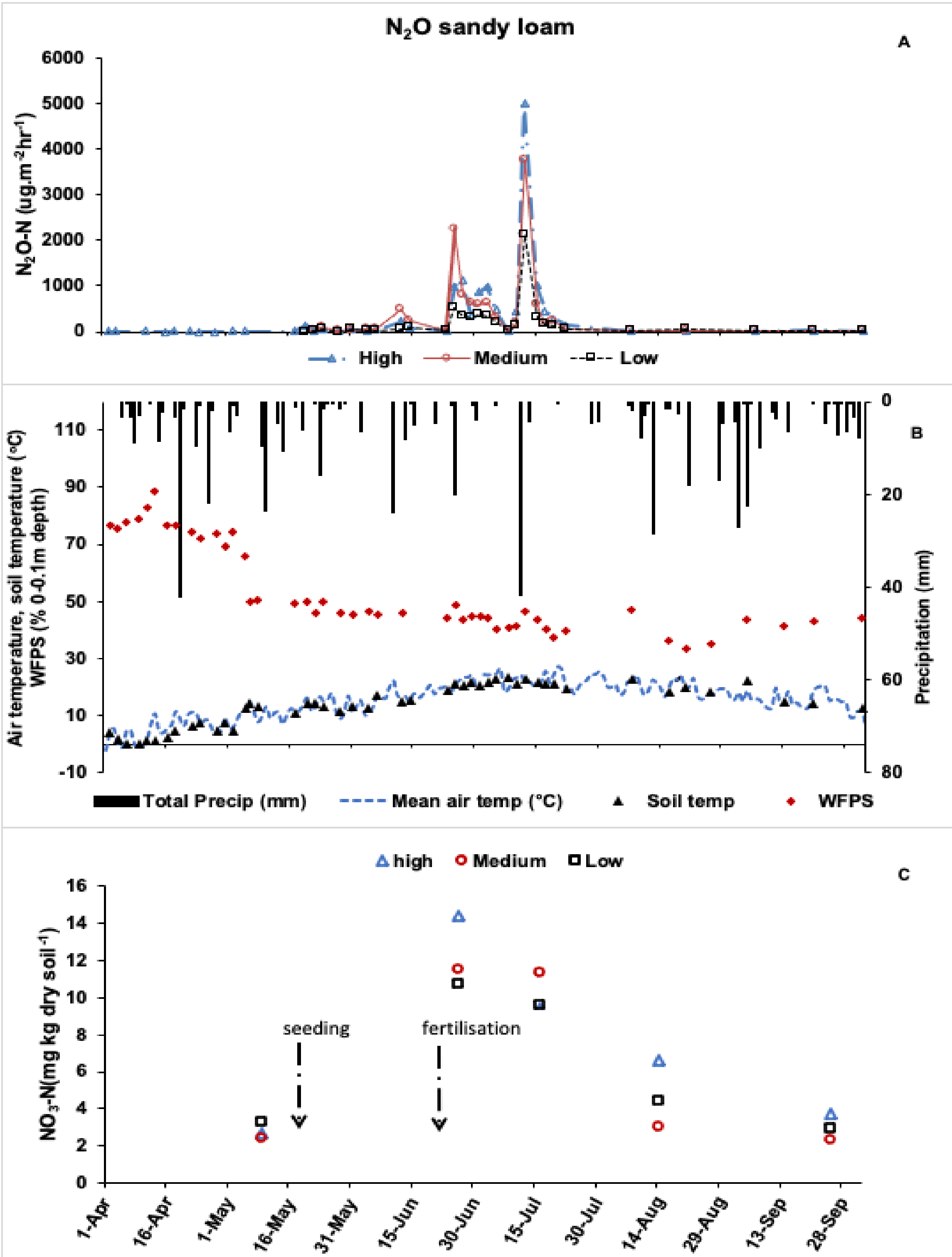

Figure 4.1: N_2O fluxes under different N fertilization rates (high - 220 kg N ha-1; medium - 180 kg N ha-1; low - 140 kg N ha-1) along with weather and soil characteristics for the sandy loam site. (A) N_2O fluxes; (B) air temperature, soil temperature, soil water content and daily precipitation; (C) Soil NO_3^--N (top 0.20 m).

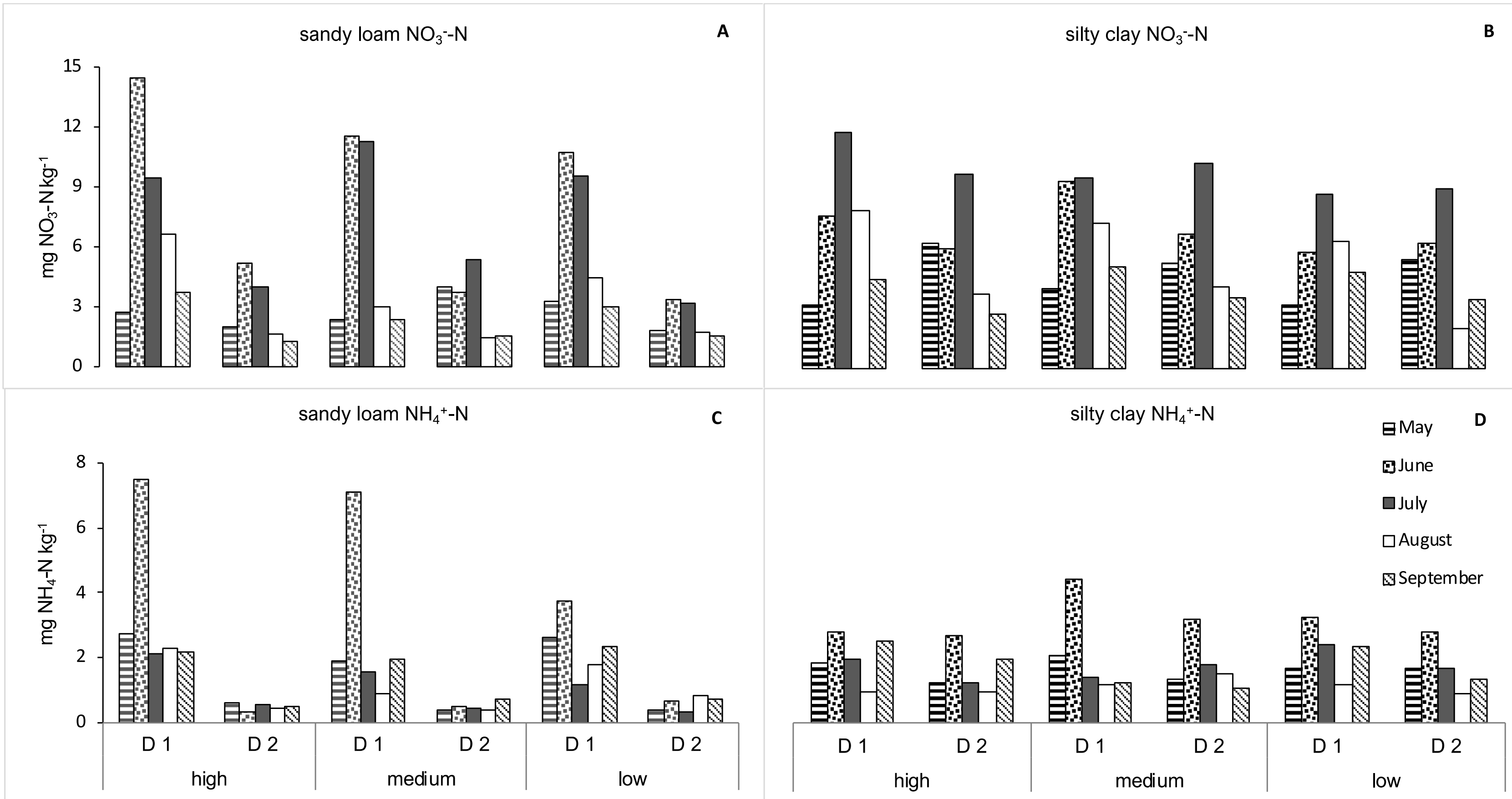

Figure 4.2: Growing season $[NO_3^-\text{-N}]_{soil}$ (mg NO_3^--N kg^{-1}) and $[NH_4^+]_{soil}$ (mg NH_4^+-N kg^{-1}) for the two soils at two depth ranges: 0-0.20 m (D 1) and 0.20- 0.40 m (D 2). A. $[NO_3^-]_{soil}$ in sandy loam; B. $[NO_3^-]_{soil}$ in silty clay; C. $[NH_4^+]_{soil}$ in sandy loam; D. $[NH_4^+]_{soil}$ in silty clay.

Table 4.2: N-uptake and corn yield for each soil type and fertilizer rate treatment

Soil type	N rate (kg ha^{-1})	Total N uptake (mg N g^{-1})			Corn yield (Mg ha^{-1})		
		Grain	Stalk	Total N uptake	Grain	Stalk	Total dry yields
Sandy loam	220	10.5	7.3	17.8	9.1	10.1	19.2
	180	10.2	5.4	15.6	9.1	10.0	19.1
	140	10.2	5.1	15.3	8.8	9.8	18.6
Silty clay	220	10.3	4.8	15.1	7.8	7.6	15.3
	180	9.8	4.2	14.0	7.4	7.1	14.5
	140	9.8	4.2	14.0	6.8	6.7	13.5

4.4.1.2 N_2O fluxes

Mean hourly N_2O fluxes from the sandy loam plots ranged from 0.1 to 5017 µg N_2O-N m^{-2} hr^{-1} (Fig. 4.1). The total spring thaw (April) emissions were 0.02 kg N_2O-N ha^{-1}, accounting for less than 0.2% of the total growing season emissions. Cumulative growing season N_2O emissions from the sandy loam soil ranged from 3.7 kg ha^{-1} from the plots receiving low rates of fertilization, to 8.7 kg ha^{-1} from the plots receiving medium rates (Table 4.3). However, the differences between the emissions from the N fertilizer treatments were not statistically significant ($p \le 0.05$). Peak N_2O fluxes were observed between June and July following fertilizer application at the seeding and V6 growth stages. The greatest flux occurred 3 weeks after fertilizer application (July 11), following a heavy rainfall event (41.6 mm; Fig. 4.1). This peak N_2O-N flux occurred at 49% WFPS. N_2O fluxes and $[NO_3^-]_{soil}$ were found to be positively correlated r (n = 4) = 0.99, $p \le 0.05$ (Table 4.4). No significant correlation was found between N_2O and WFPS (Table 4.4). Growing season N_2O fluxes correlated positively with soil temperature, r (n = 27) = 0.4, $p \le 0.05$ (Table 4.4). Yield-scaled emissions were 0.40, 0.93 and 0.87 kg N_2O-N Mg^{-1} from the low, medium and higher fertilizer N plots, respectively (Fig. 4.3).

Table 4.3: N_2O emissions from the sandy loam and silty clay soils subject to three rates of N fertilization.

Measured parameter	Spring thaw	Growing season N-fertilization rate (kg ha^{-1})		
		220 (High)	180 (Medium)	140 (Low)

N_2O emissions (kg N ha^{-1})				
Sandy loam	0.02 (0.01)	8.3 (6.1)	8.7 (5.1)	3.7 (3.1)
Silty clay	0.01 (0.04)	2.3 (0.8)	1.9 (0.6)	2.5 (0.8)

Note: Mean emission from plots with same treatment and SD in brackets.
Estimates based on observed and interpolated data
Spring thaw (April 2nd – May 8th). Growing season (May 9th – October 2nd)

Table 4.4: Correlation analysis between N_2O emissions and soil variables from the sandy loam and silty clay soils.

Soil type	WFPS	Temperature	NO_3^--N	NH_4^+-N
Sandy loam	0.20	0.40*	0.99*	0.60
Silty clay	0.26	0.29	0.50	-0.35

* statistically significant at 0.05 probability level

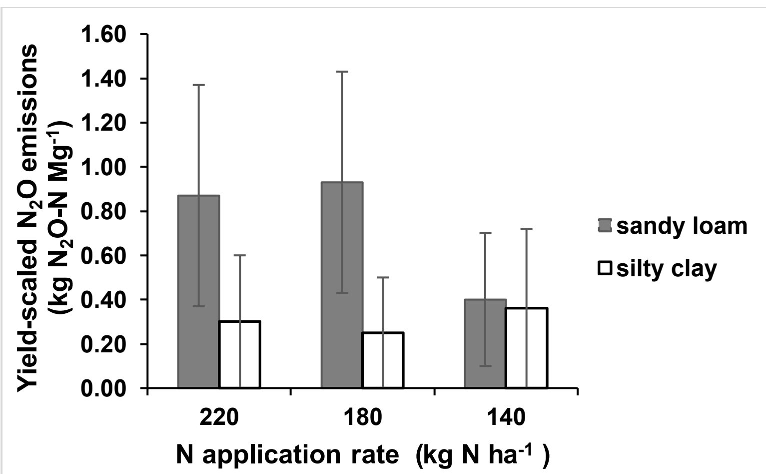

Figure 4.3: Yield-scaled N_2O emissions for high, medium, and low fertilizer N-rates for sandy loam and silty clay soils.

4.4.2 Silty clay

4.4.2.1 Weather, soil chemical characteristics and crop biomass

The seasonal (May-October) total rainfall at the silty clay site was 621 mm. Similar to the sandy loam site, 53% of the total rainfall could be attributed to nine rainfall events, which accounted for a total of 331 mm (Fig. 4.4). The soil WFPS ranged from 33% to 89%, while the soil temperatures ranged from -0.2^oC to 26^oC (Fig. 4.4). The residual pre-planting $[NO_3^-]_{soil}$ in the 0-0.20 m soil layer were 3.12, 3.95 and 3.18 mg NO_3^--N kg^{-1} for high, medium, and low N-fertilizer plots, respectively, increasing to 11.54, 9.36 and 8.60 mg NO_3^--N kg^{-1} after the bulk of the fertilizer was applied on July 8th (Fig. 4.2). After that, a gradual decline in the quantity of NO_3^--N in the soil was generally observed. $[NO_3^-]_{soil}$ at the 0-0.20 m depth were significantly greater than $[NO_3^-]_{soil}$ at the 0.20-0.50 m depth. However, no significant difference was observed in either $[NO_3^-]_{soil}$ or $[NH_4^+]_{soil}$ among the three N-fertilization rates over the growing season. In the silty clay field, N uptake ranged from 14.0 mg N g^{-1} (medium N-rate plots) to 15.1 mg N g^{-1} (high N-rate plots) (Table 4.2). Total dry yields of corn under the high, medium, and low N-fertilization rates (15.3, 14.5 and 13.5 Mg ha^{-1}, respectively; Table 4.2), were not significantly different.

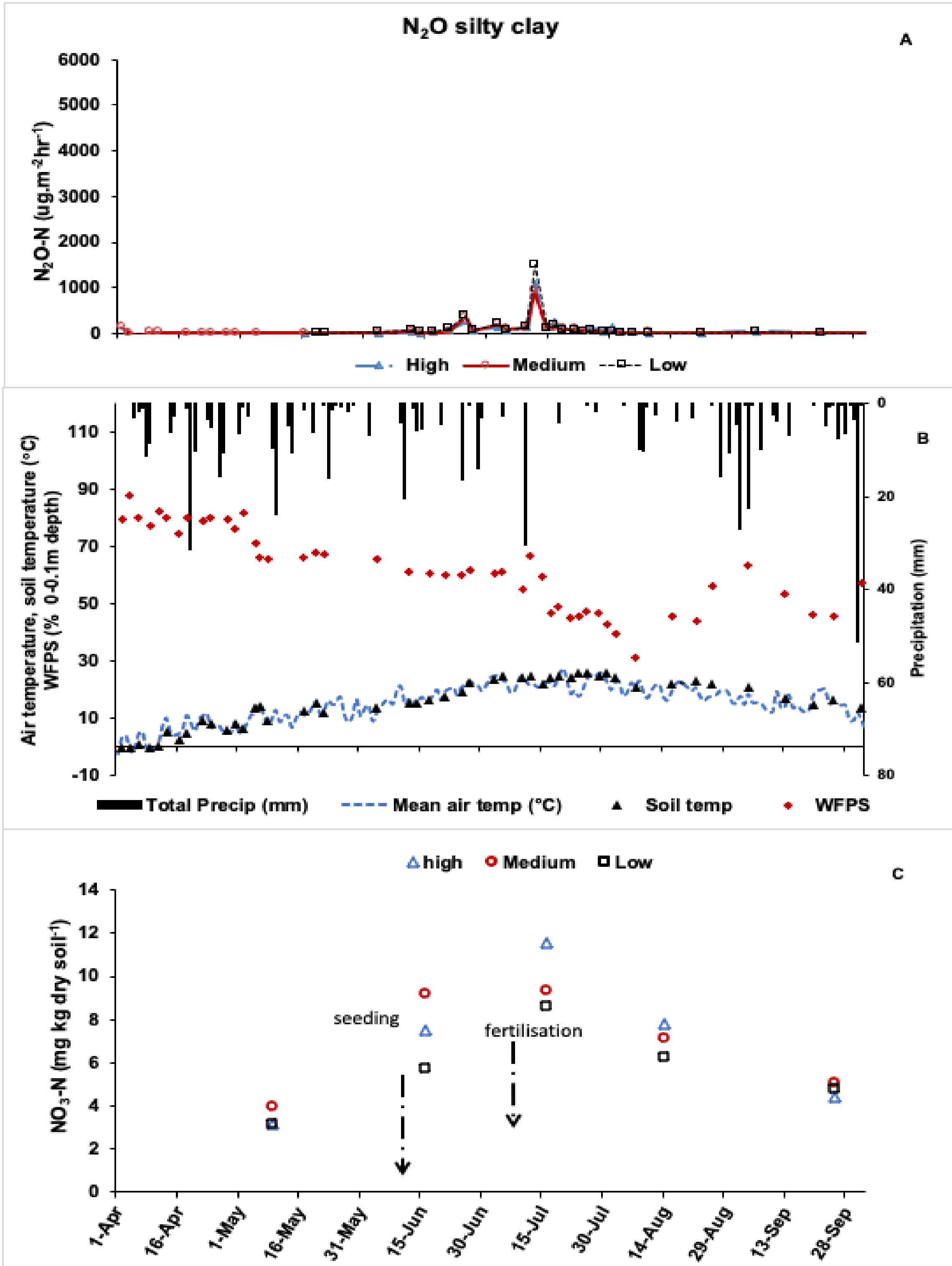

Figure 4.4: N2O fluxes under different N fertilization rates (high - 220 kg N ha-1; medium – 180 kg N ha-1; low - 140 kg N ha-1) along with weather and soil characteristics for the silty clay site. (A) N2O fluxes; (B) air temperature, soil temperature, soil water content and daily precipitation; (C) Soil NO_3^--N (top 0.20 m)

4.4.2.2 N_2O fluxes

Mean hourly N_2O fluxes from the silty clay plots ranged from 0.65 to 1489 μg N_2O-N m^{-2} hr^{-1} (Fig. 4.4). Cumulative spring thaw emissions were 0.01 kg N_2O-N ha^{-1}, accounting for about 0.5% of the annual growing season emissions. Cumulative growing season N_2O emissions ranged from 1.9 kg ha^{-1} from the plots with medium N fertilizer rates to 2.5 kg ha^{-1} from the plots with low fertilizer rates (Table 4.3). The maximum N_2O flux was observed in July after fertilizer application and a rainfall event of 30.5 mm. This peak N_2O flux occurred at 67% WFPS. There was no significant correlation between N_2O-N fluxes and $[NO_3^-\text{-N}]_{soil}$, r (n = 4) = 0.72, p = 0.17 (Table 4.4). The N_2O-N fluxes did not correlate with either soil temperature or WFPS (Table 4.4). Yield-scaled N_2O emissions were 0.36, 0.25 and 0.30 kg N_2O-N Mg^{-1} from the low, medium and higher N fertilization plots, respectively (Fig 4.3).

4.5 Discussion

4.5.1 N_2O fluxes

N_2O fluxes changed significantly over the measurement period (April 2 - October 2). The spring thaw fluxes' magnitude was low, accounting for only about 0.5% of the growing season N_2O fluxes, and similar to average fluxes observed during periods not associated with fertilizer application or rainfall. Studies have reported spring thaw emissions ranging from 6.6% to 70% of annual emissions (Li *et al.*, 2012; Lemke *et al.*, 1998). Spring thaw emissions consist of gas fluxes from the release of the built-up N_2O emissions trapped in the soil during the winter, and *de novo* production of N_2O gases stimulated due to increased availability of denitrifying substrates at the time of snowmelt (Risk et al., 2014). Such fluxes usually occur in cold and temperate climates, between mid-March and early April (Wagner-Riddle *et al.*, 2008; Chantigny *et al.*, 2017). While we did not observe significant fluxes during the spring-thaw period, the saturated soil conditions occurring for several days could have led to complete denitrification in the soil causing the release N_2 instead of N_2O. Similar low spring thaw emissions ranging from 0.03 to 0.66 kg N_2O-N ha^{-1} have also been reported in Eastern Canada (Hung *et al.*, 2021; Pattey *et al.*, 2008).

The N_2O fluxes remained relatively low until 4-6 days after fertilizer application, when spikes in N_2O fluxes started to emerge. After about four weeks from fertilizer application, the fluxes returned to low background values. This trend in fluxes is comparable with Crézé and Madramootoo (2019), who under similar climate conditions, recorded peak fluxes 20 days after applying bulk fertilizer. The N_2O fluxes were not significantly affected by the N rates in either soil during the growing season. Although not significant, emissions increased with fertilizer application rate from low to high and medium for the sandy loam soil. However, in the silty clay soil, the plots with the lowest N rates showed the greatest cumulative fluxes, followed by plots with the highest N and the medium N fertilizer rates. This behaviour of N_2O emissions from the silty clay is uncommon as increases in fertilizer N rates generally tend to trigger greater N_2O fluxes (Van Groenigen et al., 2010). It is possible that soil N was not limiting for N_2O emissions across the N-rate treatments. For future studies it may be best to start with lower rates to ensure N stress occurs with the lowest N treatment.

Interestingly, the magnitude of the average fluxes emitted from the silty clay soil was up to three times lower than that of fluxes from the sandy loam soil. These results are consistent with previous results (Wang et al., 2018) that showed that N_2O emissions from denitrification are negatively correlated with clay content. As reported by Hu et al., (2019), N_2O emissions were not significantly stimulated by crop residue returning in soils with > 25% clay content compared to the positive and negative effects observed at 15-22% and ≤15% clay content respectively. Similarly, Księżopolska et al., (2017) showed that adding clay to soil creates a reducing environment that increases the rate of complete denitrification, thereby lowering N_2O emissions. However, other studies that investigated the influence of soil texture on N_2O emissions found greater levels of N_2O were emitted from poorly drained soils than well-drained soils (Rochette et al., 2008; Li and Kelliher, 2005). Li and Kelliher (2005) related the greater N_2O fluxes from poorly drained soils to the higher observed WFPS of such soils, which could have resulted in conditions promoting denitrification. Rochette et al., (2008) related the elevated N_2O emissions from poorly drained soils in their study more to mineralization of soil organic matter than applied N fertilizer. This inconsistency with respect to N_2O emissions from poorly drained soils may be explained by the variability in soil clay content and drainage characteristics impacting soil moisture, which affects the availability of soil reactive nitrogen and microbial activities.

4.5.2 Soil and environmental variables as related to N_2O emissions

Following fertilizer application, combined N soil levels ($[NO_3^-]_{soil} + [NH_4^+]_{soil}$) at the 0-0.20 m soil depth in June and July were greater in the sandy loam than in the silty clay. Since N_2O

fluxes released from the soils result from the nitrification and denitrification processes occurring in the topsoil, the high $[NO_3^-]_{soil}$ and $[NH_4^+]_{soil}$ observed at the 0-0.20 m depth of the sandy loam field would have contributed to the high fluxes observed in the sandy loam soils. The $[NO_3^-]_{soil}$ decreased with decreasing fertilizer rates in the sandy loam soil, which most likely resulted in lower N_2O-N fluxes at the low fertilizer rate treatment. However, in the silty clay soil, no significant difference was observed in $[NO_3^-]_{soil}$ among the N-fertilizer rates. This is likely because of adsorption of NH_4^+onto clay, which would limit nitrification to NO_3^-, which may explain the absence of significant differences between N_2O fluxes from the different N fertilizer treatments. Additionally, the strong correlation between N_2O fluxes and soil available NO_3^- in the sandy loam, but not in the silty clay, (Table 4.4), could explain why fluxes were higher in the sandy loam. Soil N ($[NO_3^-]_{soil}$ and $[NH_4^+]_{soil}$) were mostly increased after the addition of fertilizer, except for NH_4^+ in the silty clay (Fig 4.2). Although low NH_4^+ levels were expected given the greater adsorption of NH_4^+ to clay particles, such low NH_4^+ levels were also observed in a study by Pelster et al., (2019). Their study showed that soils with high clay content (29.9% to 56.8%) tend to have lower levels of NH_4^+ than low clay content (7.9% to 20.4%) soils during the first week after fertilizer N application. However, in their study, $[NH_4^+]_{soil}$ in high clay soils eventually peaked about two weeks after N fertilization. The monthly frequency of soil sampling in our study did not allow for the confirmation of this trend. At the lower (0.20-0.50 m) depth of the silty clay soil, the $[NO_3^-]_{soil}$ and $[NH_4^+]_{soil}$ were greater than at the same depth in the sandy loam, confirming the potential of clay soils, such as the silty clay in our study with 48% clay content, to adsorb soil nutrients (Fig 4.2). Moreover, the heavy rainfall event (30.5 mm) of July 11, three days after fertilizer application, could have aided in leaching soil N to the lower depths. The N_2O fluxes from such lower depths may have been converted to N_2 when they reached the soil surface, resulting in lower N_2O fluxes.

The measured WFPS varied from 33% to 89% in the sandy loam, compared to 31% to 88% in silty clay. However, the silty clay showed a greater mean WFPS (approx. 7.42%) than the sandy loam. This elevated WFPS was expected, given the fine texture of the silty clay soil and its greater ability to retain moisture. It has been shown that, due to their higher WFPS values, poorly drained soils emit more N_2O compared to freely drained soils (Chantigny et al., 2010). In our study, the higher WFPS observed in the silty clay did not result in relatively greater N_2O emissions than in the sandy loam. Moreover, WFPS did not significantly influence N_2O fluxes in either soil (Table

4.4). Our results are similar to those of Sänger et al. (2011) who showed a lack of influence of WFPS on N_2O emissions. The soils' field capacities (θ_{fc}) may help explain how WFPS influences N_2O fluxes from soils. The model proposed by Linn and Doran (1984) suggests that at a 60% WFPS, which represents the θ_{fc} and critical point for activating anaerobic processes in most arable soils with loamy texture, high rates of N_2O emissions could occur. Since farming operations on the silty clay site in this study were delayed because of the wet field conditions experienced early in the season, fertilizer application was done in June, a relatively dry month compared to earlier months. Field capacity, θ_{fc} (expressed as WFPS) and maximum post-fertilization WFPS during gas sampling at our sites were 37.7% and 49%, respectively, for the sandy loam; and 77.8% and 67% for the silty clay. Thus, lower N_2O fluxes from the silty clay could be attributed to the post-fertilization WFPS always being below the θ_{fc}, resulting in microbial activities being limited by available water. Our study suggests that the WFPS should be used relative to the θ_{fc} when assessing the influence of soil moisture on N_2O fluxes. Since θ_{fc} specifies the soil moisture point for N_2O production: WFPS values below the θ_{fc} could indicate a potential for low N_2O emissions as observed in our study for the silty clay.

While N_2O fluxes did not show any correlation with WFPS, an influence of precipitation on the fluxes was observed in the early growing season. At both sites, high fluxes occurred on the day of the precipitation event or a few days later (*e.g.*, July 11, >30 mm of rain). The elevated N_2O fluxes caused by rainfall after extended dry periods, diminished over time with more wet-dry cycles and lower soil mineral N levels. Our finding on the impact of rainfall on the N_2O fluxes supports previous findings that N_2O emissions increased significantly following fertilizer application coupled with irrigation or rainfall events (Yan et al., 2015; Rochette et al., 2018).

In our study the soil temperature correlated strongly (R^2 =0.8) with the mean air temperature in the two fields indicating that air temperature could be used as a proxy for estimating soil temperature. The influence of temperature on N_2O from the two soils was inconsistent. A moderately significant positive relationship (p <0.05) was observed between growing season N_2O fluxes and soil temperature in the sandy loam, but not in the silty clay (Table 4.4). Even for the sandy loam where the positive correlation was significant (p <0.05), the effect was very small ($R^2 = 0.40$). This absence of strong relationships between soil temperature and N_2O emissions may be due to soil dryness at higher temperatures, resulting in sub-optimal soil conditions for N_2O production (Zak et al., 1999) or due to a limitation in substrates in the silty clay.

4.5.3 Plant N uptake and yields

N uptake and total yields were generally greater in the sandy loam than the silty clay (Table 4.2). Some researchers (Gregorich et al., 2014) have adduced that the compacted nature of clay soils and less oxygen in the root zone limits the ease with which the plant roots can access nutrients deep in the soil, explaining why we observed lower yields even with the high N levels seen at the 0.20-0.50 m depth (Fig. 4.2). Furthermore, in a more compacted and poorly aerated soil such as silty clay, soil oxygen requirements for root growth are greater since the roots require greater energy to penetrate the soil (Abuarab et al., 2019). This greater oxygen requirement, coupled with the inherent low oxygen levels in such compacted soil, reduces the soil's yield potential. On the contrary, there is a high potential for the plant roots to easily access nutrients in the coarse-grained sandy loam, as corroborated by the higher N uptake and yield values.

Although yields mostly increased with increasing N fertilizer rates in both soils, the yields in the sandy loam at medium (180 kg N ha^{-1}) and high fertilizer rates (220 kg N ha^{-1}) were similar. In the sandy loam, fertilizer added above the optimum fertilizer rate of 180 kg N ha^{-1} led to more N uptake by the crops without any significant yield increment, representing an extra cost that could be prevented. Not only did this excess have a cost implication, it also led to greater N_2O fluxes. This result is similar to the findings of the meta-study by Van Groenigen et al., (2010) on the effects of fertilizer rates on N_2O emissions, namely that above the optimum N-fertilization rate of 180 kg N ha^{-1}, yields did not change significantly, but N_2O fluxes increased exponentially.

4.5.4 Yield-scaled N_2O emissions

Yield-scaled N_2O emissions increased with fertilizer rates in the sandy loam plots ranging between 0.40 to 0.93 kg N_2O-N Mg^{-1} (Fig 4.3). Emissions were lowest at 140 kg N ha^{-1} but increased sharply at 180 kg N ha^{-1}. A further increase to 220 kg N ha^{-1} did not substantially increase emissions. The silty clay soil showed that yield-scaled emissions ranged between 0.25 and 0.36 kg N_2O-N Mg^{-1}. The lowest yield-scaled emissions occurred at 180 kg N ha^{-1} while the highest occurred at 140 kg N ha^{-1}. Our results indicate that the optimal fertilizer N rates for minimizing emissions at two sites are the medium fertilizer N rates in the fine-textured soils and the low rate in coarse-textured soil. Based on this study's findings, N fertilizer recommendations for our soils specifically are 140 kg N ha^{-1} and 180 kg N ha^{-1} for the sandy loam and silty clay, respectively. The yield-scaled N_2O emissions for the two sites in our study fall within the ranges recommended in other studies. In a synthesis of 19 studies, Van Groenigen et al. (2010) showed overall yield-scaled emissions to be lowest at an average N fertilizer rate of 187 kg N ha^{-1} but triple at an average N rate of 301 kg N ha^{-1}. A similar meta-analysis focusing on maize farms (Zhao et al., 2017) suggested a rate of 171 kg N ha^{-1} for achieving the lowest yield-scaled emissions required for ensuring economic returns in the US Midwest. In Quebec, Canada, the recommended rate is 120 to 170 k N ha^{-1} for grain-corn, of which 20 to 50 kg N ha^{-1} should be side-dressed at seeding (CRAAQ, 2010).

Yield-scaled N_2O emissions from other studies were incorporated to compare yield-scaled emissions across various N fertilization rates and soils with varying clay content (Fig. 4.5, supplementary data Table 4.S1). These studies, including our study, had a total of 85 observations with varying N-rates (0-235 kg N ha^{-1}) and were conducted on rainfed or irrigated cornfields in North America. Applying regression to test the association between the explanatory variables (clay content and N-rate) and response variable (yield-scaled N_2O emissions), we found that only clay content significantly explained yield-scaled emissions (Table 4.S2). Coarse-textured soils with very low clay content (7%), such as the sandy loam, resulted in yield-scaled emissions varying from 0.42 - 0.91 kg N_2O-N Mg^{-1}. Medium-textured soils with clay content ranging from 20-38% generally emitted much lower yield-scaled emissions (0.06 - 0.84 kg N_2O-N Mg^{-1}). Soils with high clay content (>40%) exhibited a wider range (low and high values) of yield-scaled emissions (0.24-3.6 kg N_2O-N Mg^{-1}) at varying N-rates (0-235 kg N ha^{-1}). Over the observed ranges, yield-scaled emissions from the poorly drained soils were 5-fold greater than well-drained (coarse- and medium-textured) soils. Such high emissions are likely due to the enhanced denitrification rates that occur in poorly drained soils (Gagnon et al., 2011). This finding differs from the one-year result from our silty clay site and indicates a potential for fine-textured soils to emit high yield-scaled emissions (Fig. 4.S1). This potential shows the importance of not relying on only a year's experiment when determining optimal N rates since variation in weather conditions could affect the results. Overall, our study shows that soil texture is a vital factor to be considered in fertilizer recommendations, as it influences yield-scaled emissions.

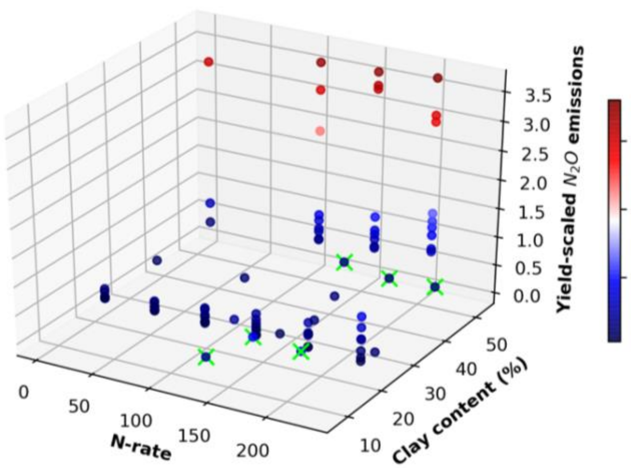

Figure 4.5: Yield-scaled N_2O emissions (kg N_2O N Mg^{-1}) from various studies including five peer reviewed studies (85 observations in total), across gradients of N-rate (kg N ha^{-1}) and clay content (%). Data points with green crosses represent values from the current study.

4.6 Conclusions

This study was undertaken to investigate optimum fertilizer N rates on two distinct soils (sandy loam and silty clay) with the goal of mitigating N_2O losses without negatively impacting crop productivity. Topsoil N, N-uptake (biomass), and crop yield were greater on the sandy loam than the silty clay. By comparing soil cumulative N_2O emissions from the two soils, we found that in the year of study, N_2O emissions were up to three times greater on the sandy loam than the silty clay. A common generalization is that poorly drained soils such as silty clay may lose more N through denitrification (N_2O losses) than well-drained soils, given their potentially elevated WFPS. However, our study suggests that poorly drained soils could produce lower N_2O emissions due to lower denitrification rates under suboptimal moisture conditions in the topsoil.

This investigation shows that soil texture is an essential factor to consider when determining the optimal N rates. Overall, yield-scaled emissions from silty clay were lower than sandy-loam. Yield-scaled emissions were lowest under a 140 kg N ha^{-1} fertilization rate for sandy loam and 180 kg N ha^{-1} for silty clay. Thus, applying fertilizer at these rates to the respective study sites improves crop yield and does not pose an environmental threat regarding N_2O emissions, compared to the other rates investigated. We note that the delayed planting that occurred on the silty clay would have influenced these findings about fine-textured soils. Despite our finding regarding lower yield-scaled N_2O emissions from fine-textured soils, a review of previous studies conducted in North America indicates that fine-textured soils could emit 5-fold greater yield-scaled emissions than well-drained soils. This observation highlights the likelihood of fine-textured soils in rainfed or irrigated cornfields in North America to result in high yield-scaled N_2O emissions. A viable mitigation strategy for such soil includes reducing the frequency of high N demanding crops such as corn and incorporating crops that require low fertilizer N (e.g., non-legume cover crops) into crop rotations.

4.7 References

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4.8 Appendix: Supplementary data

Table 4.S1: Description of location, crop, fertilizer type and rates, and soil types reported in yield-scaled emissions review of this study.

Country	Location	No of observations	No of years	Form of N input	Range of fertilizer N input	Soil type	Clay content (%)	Reference
Canada	Quebec	30	3	UAN/CAN/AA	0 -200	Clay	54	Gagnon et al., 2011; Gagnon & Ziadi, 2010
USA	Michigan	42	2	Urea	0-225	Fine Loam	23	Hoben et al., 2011
USA	Iowa	2	2	AA	125-168	Silty clay loam	20,29	Parkin & Hatfield, 2010
Canada	Quebec	3	1	AN	0-160	Clay loam	38	Pelster et al., 2011
USA	Nebraska	2	2	AA/UAN/N	153-234	Silt loam	24	Qian et al., 1997

UAN - Urea ammonium nitrate; CAN - Calcium ammonium nitrate; AA – Anhydrous ammonia; AN - Ammonium nitrate; N – Nitrate

Table 4.S2: Model estimates and ordinary least square regression results for a quadratic regression model.

Dep. Variable:	Yield-scaled N_2O emissions	R-squared:	0.38
Model:	OLS	F-statistic:	12.78
Method:	Least Squares	Prob (F-statistic):	4.33E-08
No. Observations:	85	AIC:	197.7
Df Residuals:	80	BIC:	210
Df Model:	4		
Covariance Type:	Non-robust		

	coef	**std err**	**t**	**P>|t|**
Intercept	1.3834	0.726	1.907	0.06
Clay content^2	0.0017	0.001	2.987	0.004
Clay	-0.095	0.043	-2.198	0.031
N-rate^2	4.39E-06	1.70E-05	0.258	0.797
N-rate	-2.66E-06	0.004	-0.001	0.999

We first used regression to test the association between the explanatory variables and response variable. The non-linear (quadratic regression) model is presented as: $y = b_o + b_1x_1 + b_2x_1^2 + b_1x_2 + b_2x_2^2$; where y is the response variable, b_o is the intercept and x_1, x_2 are the explanatory variables representing clay content and N-rate respectively.
Results (Table 4.S2) show that a quadratic regression model depicting the association of clay content and N-rate explained 38% of the variability in yield scaled N_2O emissions. Unlike clay-content variables that significantly explained the yield scaled emissions, N rate variables were not significant. These significant values of the clay content variables show that the curvilinear pattern observed in the scatter plot is statistically significant.

Since not significant in predicting the response variable, N-rate variables were removed from the prediction model. The final prediction model after removing the N-rate variables was thus: $Yield\ scaled\ N_2O\ emissions = 1.5542 - 0.1004x_1 + 0.0018x_1^2$;
where x_1 is the explanatory variable representing clay content (%). A negative linear coefficient and a positive quadratic term indicate that the best fit curve is convex (Fig. 4.S1).

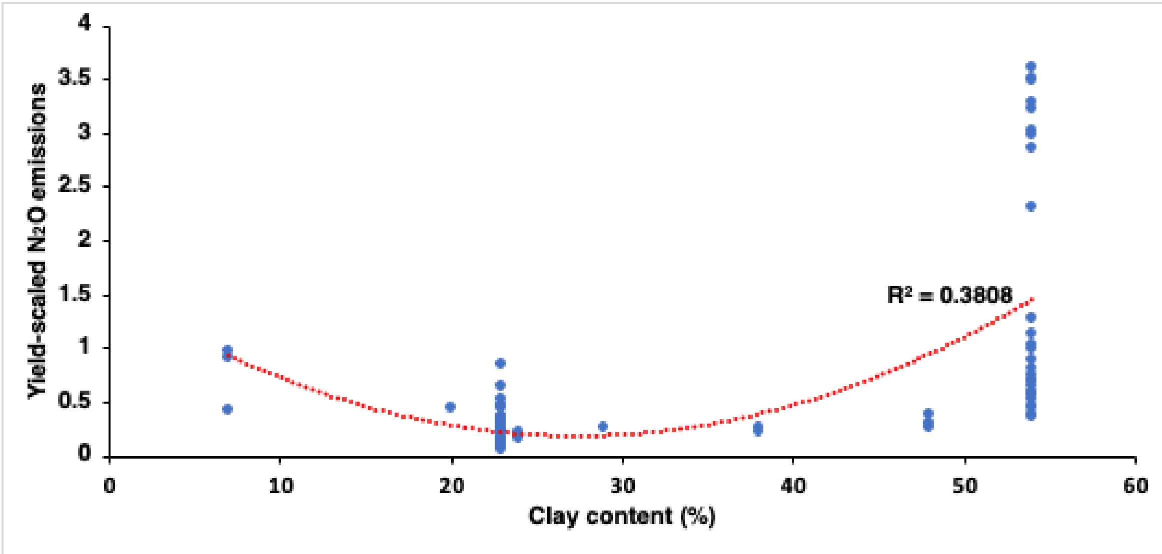

Figure 4.S1: Model fit showing association between clay content (%) and yield-scaled N_2O emissions (kg N_2O N Mg^{-1})

Connecting Text

Chapters 3 and 4 assessed WTM and fertilizer management as mitigation practices for growing season N_2O emissions. Nonetheless, a significant portion of N_2O emissions, which are not always accounted for, occur over winter in cold regions during soil's freezing and thawing. Identifying viable mitigation strategies for lowering freeze-thaw emissions is essential. Chapter 5 investigated winter and early spring N_2O emissions. This chapter used meta-analysis to clarify the effectiveness of tillage, cover crops, nitrification, and urease inhibitors in mitigating the considerable N_2O losses that occur over winter. The efficacy of these mitigation practices across soil types and aridity zones (cold-dry vs. cold-humid areas) was also assessed.

The following manuscript, co-authored by Dr. Chandra Madramootoo and Dr. Abbasi N.A., has been published in the Journal of Biology and Fertility of Soils:

Ekwunife, K. C., Madramootoo, C. A., & Abbasi, N. A. (2022). Assessing the Impacts of Tillage, Cover Crops, Nitrification, and Urease Inhibitors on Nitrous Oxide Emissions over Winter and Early Spring. Biology and Fertility of Soils, *58*(3), 195–206. https://doi.org/10.1007/s00374-021-01605-w.

Contribution of authors:

Ekwunife, K.C. curated data, performed the analysis, and wrote the paper. Madramootoo, C.A outlined scope of study and reviewed and edited all drafts. Abbasi, N.A. curated data and provided technical comments. All authors read and approved the final manuscript.

Chapter V

Assessing the impacts of tillage, cover crops, nitrification, and urease inhibitors on nitrous oxide emissions over winter and early spring

5.1 Abstract

There are increasing demands to reduce greenhouse gas emissions from agricultural soils worldwide. A significant portion of these emissions occur in cold regions during soil's freezing and thawing. Focusing on over-winter cropland nitrous oxide (N_2O) emissions, a review of 21 relevant peer-reviewed studies with a total of 88 comparisons was conducted to quantify the efficacy of field management practices (no-till, cover crops (CC), nitrification, and urease inhibitors (NI + UI)) in reducing emissions. We also assessed these mitigation practices' efficacy across soil types and between cold humid and cold dry areas. The non-growing season emissions ratio to full-year N_2O emissions reported in the studies used in this review ranged between 5% to 91%. No-till significantly reduced N_2O emissions by 28%, and this effect was more pronounced in drier climates. NI + UI also significantly reduced over-winter emissions by 23% compared to conventional fertilizers, and this effect was more evident in medium-textured soils than coarse soils. CCs showed an overall reduction potential of 18%; however, this effect was not significant. This review showed that under the CC practice, N_2O emissions were reduced overall in humid climates but increased in drier climates, while no-till and NI + UI practices effectively reduced over-winter emissions in both dry and humid winter regions and all soil types.

5.2 Introduction

Agricultural soils emit trace gases that increase atmospheric concentrations of nitrous oxide (N_2O), a greenhouse gas that contributes to global climate change (Davidson and Kanter 2014). Studies investigating N_2O emissions from agricultural soils have vastly focused on growing season. Nonetheless, the contribution of over-winter N_2O emissions to annual emissions that occur in mid-high to high latitude regions, such as Canada and the Mid-West US where soil freezing and thawing occurs over a 5 month period, could reach up to 90% (Risk et al. 2014). The bulk of over-winter emissions mostly occurs in the early spring partly due to the anaerobic conditions created during the thawing of soils enhancing N_2O production (Chantigny et al. 2016; Ejack and Whalen 2021; Risk et al. 2014; Wagner-Riddle et al. 2007). Understanding the factors including substrate availability, snow cover, freeze-thaw dynamics, soil temperature, soil type and moisture conditions that impact the magnitude of over-winter emissions is becoming the focus of recent studies (Chen et al. 2016; Smith et al. 2010; Zhe et al. 2018).

Appropriate agronomic management practices could help achieve a reduction in over-winter emissions. These practices include those tailored towards minimizing the substrates available for emission during the spring thaw such as the use of nitrogen inhibitors during fall manure application and incorporation of cover crops. The use of nitrification inhibitors to prevent oxidation of NH_4^+ to NO_3^- has been suggested as an effective method for reducing N_2O losses, especially from fall-applied manure, which carries N mostly in the form of ammonium (NH_4) or organic N (Dong et al. 2018; Vallejo et al. 2001; VanderZaag et al. 2011; Zhang et al. 2018). Apart from lowering substrate availability, practices that can alter soil temperature or moisture, such as the presence cover crop and no tillage have been shown to lessen over-winter emissions (Congreves et al. 2017; Preza-Fontes et al. 2020; Wagner-Riddle et al. 2007). The use of non-legume cover crops (CC) as a mitigation practice for N_2O emissions is centered on the hypothesis that cover-crops will absorb more moisture and NO_3^- from the soil, lowering their levels and resulting in reduced N_2O emissions (Shackelford et al. 2019). The major difference between legume CC and non-legume CC is that while legume CC can fix nitrogen from the atmosphere to the soil, non-legume CC cannot, but can absorb large quantities of N from the soil. Having a vegetation cover atop the soil can also slow down the thawing of frozen soils and lower the freeze-thaw cycles, thereby reducing emission rates (Chen et al. 2020). Tillage could affect emissions through its influence on soil aeration, structure, temperature, moisture content, microbial activity, and gas diffusion through the subsoil to the surface (Gregorich et al. 2015; Signor and Cerri 2013). No-till field management could reduce N_2O emissions by lowering freezing intensity and soil moisture (Wagner-Riddle et al. 2007). However, the efficacy of no-till and the use of non-legume cover crops remains under debate as some studies have also reported no consistent effects of these practices (Behnke and Villamil 2019; Elmi et al. 2003; Thomas et al. 2017). It is essential to clarify the controversies since farmers' and policymakers' willingness to adopt and implement recommended strategies will depend on data-driven evidence of reduced emissions, without hampering farm productivity.

Meta-analysis is a review technique to synthesize the findings of various individual studies quantitatively, and it focuses on the direction and magnitude of the studies' effects. Meta-analysis has been used to investigate the efficacy of agronomic practices including tillage, cover crops, and nitrification and urease inhibitors in mitigating N_2O emissions. Earlier meta-analysis studies focusing majorly on growing season N_2O losses showed no significant differences in N_2O emissions between CT and no-till or reduced tillage (NT/RT) (Decock 2014; Feng et al. 2018; Six et al. 2004; van Kessel et al. 2013). Other studies reported significantly greater emissions by 12% (Shakoor et al. 2021) and 10% (Huang et al. 2018) in NT/RT treatments compared to CT. Concerning nitrification and urease inhibitors, previous meta-analysis studies have shown that the use of the inhibitors could reduce N_2O losses by up to 44% (Feng et al. 2016; Qiao et al. 2015; Thapa et al. 2016; Yang et al. 2016). Investigating the overall effect of cover crops on N_2O emissions, Basche et al. (2014) showed that 60% of the 106 observations utilized in their meta-analysis study positively affected N_2O emissions, while 40% indicated negative effects.

Although these meta-analysis studies quantified the above management practices' effectiveness on N_2O emissions, they mainly included individual studies that reported growing season emissions and have not adequately accounted for the sizeable N_2O losses that occur over winter. For instance, only 14% of the observations used by Decock (2014) were all year round measurement. Since meta-analysis studies that have focused on winter and early spring N_2O emissions are scarce, we assess these strategies' effectiveness in mitigating the considerable N_2O losses that occur over-winter. Accordingly, this work is aimed to review studies to quantify the effectiveness of nitrification and urease inhibitors, tillage, and cover crops in reducing non-growing season N_2O emissions in cold temperate regions. We also assessed the efficacy of these mitigation practices across soil types and aridity zones (cold-dry *vs*. cold-humid areas).

5.3 Mechanism of over-winter emissions

The potential mechanism of over-winter emissions in this study is discussed for two distinct periods: the freezing period and the spring-thaw period. The frozen soil reduces the microbial activities it harbors, although it does not completely inhibit them. After the initial lysis of some microbial cells that may occur at subzero temperatures (Maljanen et al. 2007), soil microorganisms gradually acclimatize to low temperatures (Smith et al. 2010). The relatively warm soil under snow cover further supports a level of their activities. For instance, decomposition of soil organic matter occurs during the freezing period, evidenced by the low rates of CO_2 accumulation (Maljanen et al. 2007). However, the CO_2 fluxes were lower compared to the fluxes observed during pre-freezing period. Likewise, mineralization of organic N also takes place, as shown by high levels of ammonium (NH_4^+) in these same soils (Maljanen et al. 2007). However, a significant concomitant increase in NO_3^- is not always observed following the soil thaw (Chen et al. 2016; Maljanen et al. 2007). This condition has been attributed to low levels of nitrifying bacteria found in the soil, as a result of their greater susceptibility to cold temperatures (Smith et al. 2010) or the possible rapid denitrification of NO_3^- — generated from a portion of the accumulated NH_4^+ — to N_2O and N_2 (Maljanen et al. 2007; Zhe et al. 2018). Contrary to the effect of cold temperature on nitrifier populations that made their detection in mid-winter difficult, cold temperatures have a lesser effect on the denitrifier community and did not hamper their detection (Smith et al. 2010). Gases produced in the freezing period are trapped in the soil, given that ice blocks the pores of soil surface, resulting in increased concentrations of N_2O and CO_2 in the soil over time. However, partial release of such trapped gases can sometimes occur during the freezing period through cracks caused by increases in temperature and slight thawing of the soil (Teepe et al. 2001).

In early spring during the soil thaw, the ice barriers that trapped gases begin to give away, causing the physical release of these gases (Burton and Beauchamp, 1994; Teepe et al. 2001). As the soil thaws further, the frozen subsoil melts, enhancing soil drainage, thereby creating the aerobic topsoil required for nitrification of available NH_4^+. Although more N_2O is produced in the deeper soil layers over the freezing period, as evidenced by its greater concentrations at these depths, as well as the depletion of NH_4^+ and organic N, a substantial proportion of these N_2O molecules are converted to N_2 by the time they reach the surface (Wagner-Riddle et al. 2008). One study by Van Groenigen et al. (2005) showed that while the subsoil is characterized by depleted δ15N indicating N_2O production, the topsoil consumed N_2O, as shown by a greater δ15N. Consumption of N_2O is generally typical for soils which are anoxic and rich in OM, but have depleted NO_3^-. This depletion of NO_3^- in the subsoil usually occurs at the onset of thawing, resulting in greater N_2 than N_2O fluxes (Ludwig et al. 2006). In addition, more recent studies suggest the release of trapped gases and newly produced N_2O occurs at the onset of thawing. Such newly produced N_2O fluxes could be up to five times the amount observed during the release of trapped gases. (Risk et al. 2014). *De novo* production results from favorable conditions for nitrification or denitrification processes during thawing. Identified as the principal source of N_2O emitted during the thawing process, the denitrification process requires anaerobic conditions, nitrates and simple C compounds (Priemé and Christensen, 2001; Sehy et al. 2004; van Groenigen et al. 2005). Anaerobic conditions are enhanced by a frozen subsoil which hinders drainage while creating saturated soil conditions (Nyborg et al. 1997). Additionally, the thaw period, characterized with high soil moisture from the snow melt, also enhances anaerobic conditions. The C and N required for denitrification could arise from the death of microbes due to freeze-thaw stresses, as well as residual C and N available in the soil (Chen et al. 2020; Pelster et al. 2013; Teepe et al. 2001).

A few studies have evaluated the role of nitrifiers and denitrifiers on over-winter emissions. Using PCR-DGGE analysis, (Smith et al. 2010) observed significant changes in the diversity of the nitrifier and denitrifier populations observed between the pre- and post- spring thaw period. The authors alluded that the variations in the microbial community structure, influenced mainly by soil temperature, available soil moisture and nutrient levels, suggest that microbial organisms must be active immediately after spring thaw. Significant N_2O flushes that occurred during the thaw in their study were primarily attributed to denitrification. Similarly, Ludwig et al. (2004) used 15N as a tracer to show that denitrification contributed 83% of N_2O production in organic soils immediately after the soil began to thaw. A microcosm study (Sharma, Szele, Schilling, Munch, & Schloter, 2006) observed an apparent increase of denitrifying bacteria and higher expression levels for nitrate reductase (*napA*) and nitrite reductase (*nirS*) occurring soon after the soil thaw began, followed by a decrease in expression. Their study thus concluded that the release of spring N_2O flushes was influenced by increased microbial activities and the expression of denitrifying genes. Wagner-Riddle et al. (2017) have shown that longer periods of soil freezing release greater amounts of substrates for denitrification. However, a recent study (Yin, Gao, Tenuta, Gui, & Zeng, 2019) has suggested that although changes in the nitrifier and denitrifier populations were observed over the winter period, these variations in the abundance of functional genes were not correlated with the changes in over-winter N_2O fluxes. Their study showed that winter N_2O fluxes were rather correlated with denitrifying enzyme activity (the rate of N_2O production) and soil environmental factors, including soil temperature and water-filled pore space. There seems to be no consensus yet on the role of the denitrifier populations on spring thaw N_2O fluxes, and this could result from the detection limits of conventional amplicon sequencing methods. Future investigations could thus utilize an improved PCR primer set (B. Zhang et al., 2021) developed for better detection of denitrifying genes. Further studies could use 15N analysis of site preference to understand the role of denitrifying and nitrifying genes on over-winter N_2O emissions.

5.4 Magnitude of over-winter N_2O emissions

Table (5.1) provides the summary of the studies used for the review in the present study. It shows the magnitude of the non-growing N_2O emissions from the various studies, also expressed as a percentage of total annual emissions. Additionally, it includes the major treatments of the studies and the measurement period of the over-winter fluxes. More details of the data and methodology can be found in the supplementary information section. In this review (21 studies with a total of 88 comparisons), the ratio of the non-growing season to full-year N_2O emissions ranged between 5% to 91%, highlighting high variation and the significance of these emissions. The over-winter flux measurement varied among studies. For example, in the North Atlantic region of the US and Canada, and the mid-west US, snow cover appears in early December, peaks between February and early March, with the spring melt occurring within 2-3 weeks between late March and early April (Chantigny et al. 2016). Some of the studies started earlier or ended later to capture the entire winter and early spring period. As snow accumulation varies from year to year, the timing and intensity of winter-time soil fluxes may vary, as would the temporal occurrence and magnitude of peak fluxes.

Table 5.1: Summary of the studies used in the meta-analysis

Treatment			N_2O emissions (kg N ha^{-1})					
Tillage	Crop cover	Nitrification Inhibitors	Annual	Non-growing season	$100 \cdot \frac{Over-winter}{Annual}$	Measurement period	Number of comparisons	Source
✓	✓		–	1.8	–	Dec - May	4	Behnke and Villamil (2019)
✓			1.9	1.4	77	Jan - April	2	Congreves et al. (2017)
✓			3.3	2.9	88	Nov - April	5	Wagner-Riddle et al. (2007)
✓			2.8	2.6	91	Mar - April	12	Lemke et al. (1999)
✓			0.9	0.3	30	Dec - March	2	Ussiri et al. (2009)
✓			0.8	0.2	22	Nov - April	1	Mutegi et al. (2010)

⍰	⍰		–	3.9	–	Sep - May	2	Petersen et al. (2011)
⍰			–	0.6	–	Oct - April	4	Ferrari Machado et al. (2021)
	⍰		12.1	7.2	59	Oct - May	2	Jarecki et al. (2009)
	⍰			0.03		Mar - April	8	Hung et al. (2021)
	⍰		1.9	1.1	57	Jul - April	4	Preza-Fontes et al. (2020)
	⍰		–	0.4	–	Sep - June	4	Thomas et al. (2017)
	⍰		–	27.1	–	Feb - May	4	Dietzel et al. (2011)
		⍰	6.2	5.2	84	Mar - April	4	Lin et al. (2017)
		⍰	8.8	4.1	47	Nov - April	2	Pfab et al. (2012)
		⍰	5.3	2.5	48	Nov - April	2	Parkin and Hatfield (2010)
		⍰	0.8	0.2	22.1	Oct - March	4	Halvorson and Del Gross (2012)
		⍰	2.6	0.1	5.4	April - May	1	Asgedom et al. (2014)
		⍰	1.1	0.8	72.2	Oct - April	6	Dong et al. (2018)
		⍰	2.3	1.3	56.4	Nov - April	6	Ferrari Machado et al. (2020)
		⍰	–	0.5	–	Oct - June	3	Ding et al. (2015)

5.5 Mitigation potential of no-till/reduced tillage

Using the meta-analysis approach, we quantified the effectiveness of mitigation practices in reducing non-growing season N_2O emissions in cold temperate regions (more details of the method can be found in the supplementary information). In the winter and early spring, the effect of no-till/reduced tillage (NT/RT) compared with conventional tillage (CT) was -28% (95% confidence interval = -39% to -16%), indicating a significant reduction of N_2O emissions (Fig. 5.1, Table 5.2). Under the tillage practice, dry areas emitted significantly lower emissions than the humid areas (35% less), while soil texture did not significantly affect N_2O emissions (Fig. 5.2, Table 5.S2). Although among the studies there were years when no-till did not reduce N_2O emissions over winter, in some other cases, studies reported that mean N_2O fluxes under NT were one to six times lower than in CT plots (Congreves et al. 2017; Lemke et al. 1999; Wagner-Riddle et al. 2007). The micrometeorological method used in gas emission measurements in some studies (Congreves et al. 2017; Wagner-Riddle et al. 2007), allowing for more frequent sampling, would have accorded the researchers a greater chance of capturing distinct peak fluxes.

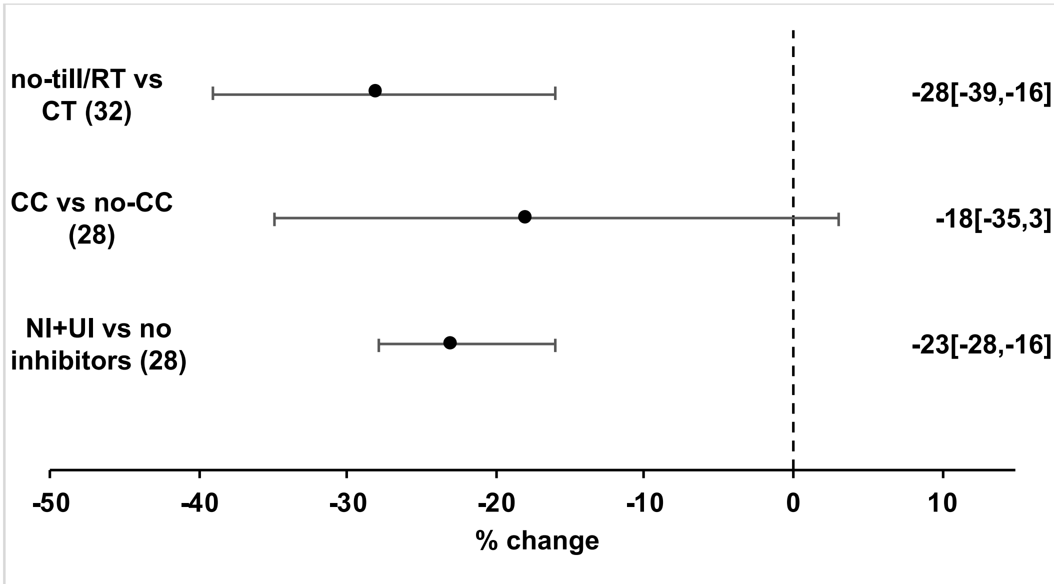

Figure 5.1: Forest plot of the random effect meta-analysis indicating the overall effect size estimates of management practices: no till/reduced tillage (RT) vs. tillage treatments; cover crops (CC) vs. no cover crops (no-CC); nitrification, and urease inhibitors (NI + UI) vs. no inhibitors, on over-winter N_2O emissions. Numbers in the bracket indicate the number of observations utilized in the analysis. The center circle represents the summary estimate. Confidence intervals are also reported. CI crossing the null (zero) line indicates no significant change in effect or that both treatments show equivalent effects.

Table 5.2: List of management practices, environmental conditions, and percent change in N_2O emissions relative to the alternative treatment.

Practice	Conditions	% change over-winter
No-till	Overall	▼*28%
	Climate	
	humid	▼*18%
	dry	▼*47%
	Soil type	
	coarse	▼*41%
	medium	▼*19%
	fine	-
Cover crop	Overall	▼18%
	Climate	
	humid	▼26%
	dry	△41%
	Soil type	
	coarse	▼43%

	medium	▼2%
	fine	-
NI + UI inhibitors	Overall	▼*23%
	Climate	
	humid	▼*31%
	dry	▼*18%
	Soil type	
	coarse	▼*12%
	medium	▼*28%
	fine	-

Arrow pointing downwards indicates a decreasing change in N_2O emissions due to the treatment effect, while the arrow pointing upwards shows an increasing change.
* Statistically significant change

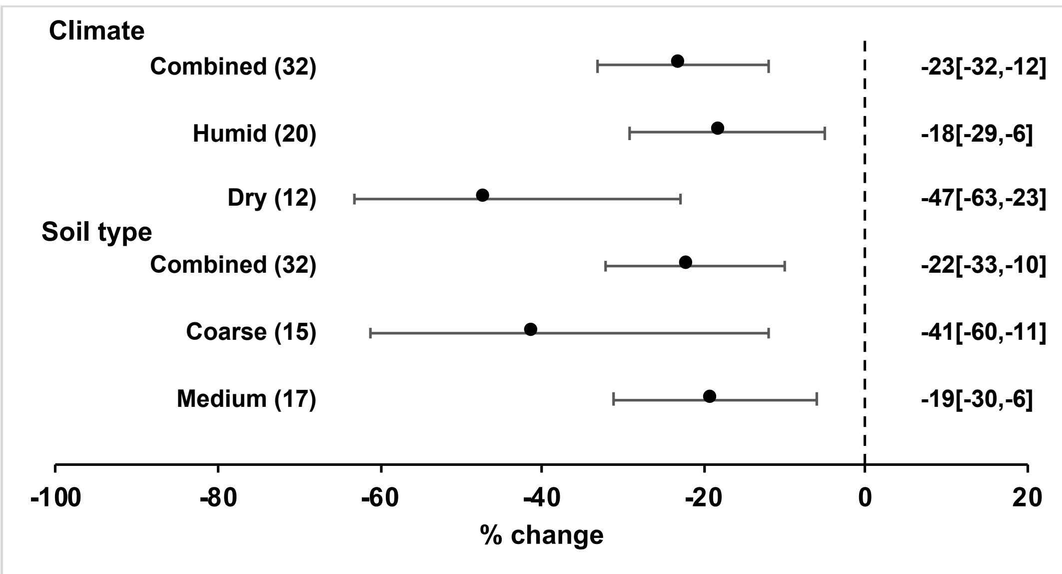

Figure 5.2: Forest plot of the random effect meta-analysis indicating subgroup (climate and soil type) effects of no tillage/reduced tillage (no-till/RT) vs. tillage treatments (CT). Confidence intervals are also reported. CI crossing the null (zero) line indicates no significant change in effect or that both treatments show equivalent effects.

The lower N_2O emissions under NT/RT can be attributed to the combined insulating effects of crop residues and snow cover that lowers the cumulative degree hours below 0°C, an indicator of freezing intensity and duration (Wagner-Riddle et al. 2007). Wagner-Riddle et al. (2007) also suggested that crop residues that are retained on NT/RT plots can effectively trap snow. Due to their low heat of diffusivity, snow and residues insulate the soil and decrease the transfer of heat from the soil to the atmosphere, resulting in higher soil temperature in RT/NT than CT (Wagner-

Riddle et al. 2007). Under CT, soil temperatures varied over a wider range, increasing the degree of freezing than under NT/RT plots. The magnitude of over-winter N_2O emissions have been linked to the number of degree hours below 0°C. An exponential to plateau relationship model defined this link and showed that N_2O emissions increased exponentially with cumulative freezing degree days until a plateau was reached (Wagner-Riddle et al. 2017). The wider temperature variation reported for CT (*vs.* NT/RT) soils (Wagner-Riddle et al. 2007; Smith et al. 2010) could lead to a greater microbes' die-off. When freezing stresses kill off some soil microbes, microbial polymers return to the soil C pool, while amino acids and other organic monomers are returned to the dissolved organic nitrogen (DON) pool (Schimel and Bennett 2004). Living microbes in the frozen soil's liquid film then utilize these substances to stimulate N_2O emissions (Chen et al. 2020; Pelster et al. 2013). Additionally, Ferrari Machado et al. (2021) suggested that higher N_2O emission under CT may have been induced by mechanisms potentially resulting from lower temperatures such as the breakdown of soil aggregates and subsequent release of previously protected soil organic C. Lower temperatures could therefore cause an exponential rise in N_2O fluxes for a few days resulting from the death of microbes and subsequent release of NO_3^- (Larsen et al. 2002; Maljanen et al. 2007; Sharma et al. 2006). This observation was also confirmed by Smith et al. (2010) who suggested that the soil's lower nutrient availability in NT/RT plots due to lower freezing intensity could lower N_2O emissions than CT plots. Smith et al. (2010) also implied that the significant differences in the composition of nitrifier and denitrifier communities observed soon after spring thaw could explain the differences in emissions between the CT and NT/RT treatments.

The difference in the establishment period of NT/RT from CT is an important factor to consider when investigating the effect size of NT/RT practice on N_2O emissions. (Six et al. 2004) showed that the potential to reduce N_2O emissions with NT is only achieved with long-term NT practice. Elevated N_2O emissions from newly adopted NT/RT plots slowly diminish due to the development of macropores, improved soil structure and aeration status, and decreased formation of anaerobic microsites. Studies have shown that the elevated N_2O emissions observed under NT/RT only occurred within the first ten years following the shift from CT to NT/RT, after which a significant reduction in N_2O emissions was observed under NT/RT (Six et al. 2004; Feng et al. 2018; van Kessel et al. 2013). In our meta-analysis, 62% of the tillage studies had ≥10 years of

NT/RT and this could be one of the reasons for the significant reduction in N_2O emissions observed under NT/RT.

This review revealed that relatively drier areas such as the low annual precipitation areas in Western Canada or China's moist continental climate with dry winter periods emitted significantly lower emissions than the humid winter areas (35% less). Elevated soil moisture during the winter and especially in the spring thaw period favors the denitrification process (Maljanen et al. 2007), the main process contributing to freeze-thaw N_2O emissions. Antecedent fall moisture conditions have also been shown to affect spring thaw N_2O emissions, with a greater fall water-filled pore space (WFPS) leading to greater spring fluxes (Chen et al. 2016; Li et al. 2012; Priemé and Christensen 2001). Following a distinctly wet field growing season, Chen et al. (2016) observed relatively high subsequent spring thaw N_2O fluxes. These elevated spring emissions, accounting for up to 49% of the non-growing season fluxes, were attributed to the soil's high moisture condition, which led to a rise in soil WFPS to 54%, compared to the 28% measured in a previous drier year when no significant thaw emissions were observed. A confirmatory lab experiment within the same study showed that freeze-thaw cycles did not induce distinct fluxes at a 20% WFPS, but that emissions increased by an order of magnitude at a WFPS of 80% (Chen et al. 2016). Subsequent to a dry antecedent fall and low winter air temperatures, Li et al. (2012) measured limited spring thaw emissions — 6.6% of annual emissions, in contrast to about 70% from the previous wetter and warmer preceding fall and winter. Dry fall soil conditions and the presence of less unfrozen water during the winter would have limited denitrification. Therefore, dry areas which are likely to have lower antecedent (fall) soil moisture prior to the soil freezing up in the winter could emit significantly lower spring thaw emissions than humid areas.

Although our meta-analysis showed that soil texture did not significantly impact the emissions (Fig 5.2, Table 5.S2), coarse soils tended to lower emissions more than medium soils under tillage practices. This observation reveals that the lower N_2O levels observed may be partially associated with soil texture influence on soil moisture since coarse soils have lower WFPS than medium soils. Soil texture, which influences water-filled pore space and soil moisture, affects growing season emissions. For instance, Linn and Doran (1984) showed NT/RT fields to have a greater average water-filled pore space (WFPS = 62%) than CT fields (WFPS = 44%), leading them to relate the greater emissions to anaerobic metabolism occurring in the NT/RT plots.

However, this impact of soil texture under tillage practices is not observed over the winter period as soils would tend to be saturated, no matter the soil type.

5.6 Mitigation potential of cover crops

In our meta-analysis, the effect of cover crop treatment across the trials was -18% (95% CI = -35% to 3%) (Fig. 5.1, Table 5.2). This result indicates that although cover crops showed a tendency to reduce emissions overall (-18%), the change in their effect on over-winter N_2O emissions compared to the absence of a cover crop was not significantly different. In humid regions, the use of CCs significantly resulted in lower emissions (89% less) than in the dry areas, while soil texture did not significantly affect N_2O emissions (Fig. 5.3, Table 5.S2). This finding concurs with Basche et al. (2014) findings that cover crops did not significantly affect direct N_2O emissions during the winter period. Their study also showed a significantly positive impact for data points measured during the decomposition of the cover crops compared to the impact during the cover crop growth period, which showed no significant difference. Although CC can assimilate NO_3^- over winter, they have also been observed to produce elevated N_2O concentrations in the rhizosphere by drawing on a supply of labile C and N generated by decomposing tissues and living roots (Thomas et al. 2017; Wertz et al. 2016). Therefore, while CC have a better potential to mitigate N_2O emissions by depleting NO_3^-, this potential could be countered by the production of denitrification substrates in the cover crops' root zone.

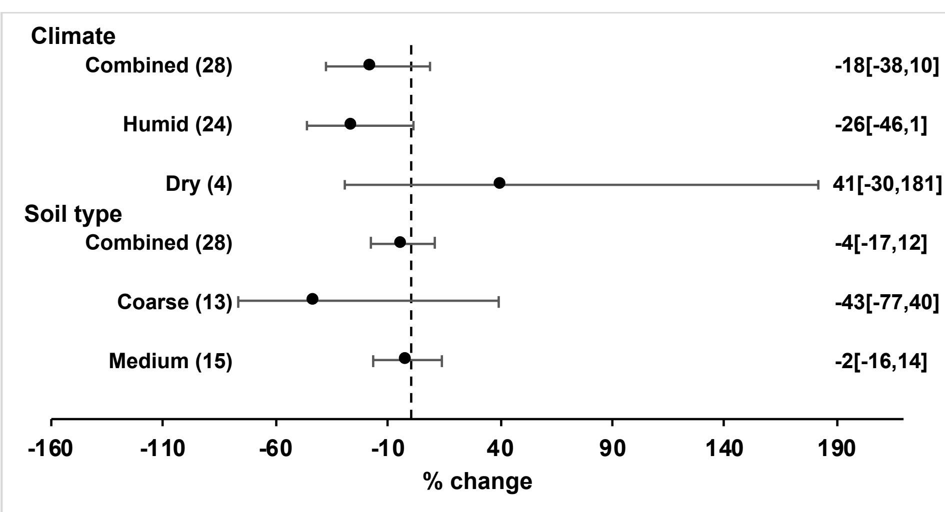

Figure 5.3: Forest plot of the random effect meta-analysis indicating subgroup (climate and soil type) effects of cover crop (CC) vs. no cover crop treatments (no-CC). Confidence intervals are

also reported. CI crossing the null (zero) line indicates no significant change in effect or that both treatments show equivalent effects

The capacity of cover crops to reduce over-winter N_2O emissions through the assimilation of NO_3^- largely depends on whether the cover crop is adequately established before the winter. This could be why a clear distinction was observed between the humid and dry climates in our meta-analysis. In cold and dry areas, cover crops sometimes develop poorly, suffer winter-kill, and cannot produce spring biomass. This situation was the case in a study by Behnke and Villamil (2019) where N_2O emissions (1.64 kg N ha^{-1}) in the first two years of study, when the cover crop was poorly established, were significantly larger than emissions (0.32 kg N ha^{-1}) observed in the last two years, which had warmer temperatures and early November precipitation. Similarly, Dietzel et al. (2011) found that after a harsh winter, a winter rye cover crop (*vs.* the absence of a cover crop) had no significant effect on over-winter N_2O emissions; however, after the following year's milder winter, the cover crop resulted in lower N_2O fluxes. When simulated in growth chambers compared to field conditions where CC did not reduce over-winter N_2O emissions, Jarecki et al. (2009) observed that cover crops resulted in N_2O reduction due to better growing conditions in the growth chambers, which allowed for the development of roots, faster growths, and N uptake. Our analysis showed that although not significantly different, emissions under the CC practice were lower in coarse soils than in medium soils. We adduce that the ease of plant growth in coarse soils, which facilitates good plant establishment, may explain why lower N_2O emissions were observed in coarse soils. These observations point to the potential influence of CC developmental status on their overall impact on over-winter emissions.

Although cover crops tend to reduce spring thaw emissions in humid climates, and when their roots are well established, they have been observed to cause a significant increase in the growing season N_2O emissions after the following crop has been planted and fertilizer applied (Mitchell et al. 2013; Preza-Fontes et al. 2020). This increase in N_2O emissions is likely due to the denitrification substrates produced in the cover crops' rhizosphere, as well as the decomposition and subsequent release of N_2O when CCs are incorporated into the soil. Consequently, to improve N efficiency in CC plots, fertilizer inputs should be reduced after plowing (Guardia et al. 2016). The selected studies show no consensus about the cover crops' effects on either spring thaw or growing season N_2O emissions (Iqbal et al. 2015; Parkin et al. 2016). However, cover crops have other benefits, including reducing NH_3, NO_3^- , DOC, and suspended solids in leachate (Gillette et al. 2018; Smukler, O'Geen, and Jackson 2012). Cover crops also increase soil organic C sequestration (Basche et al. 2014) and enhance microbial abundance (Kim et al. 2020).

5.7 Mitigation potential of nitrification and urease inhibitors

Our analysis on nitrification and urease inhibitors (NI + UI's) effect on over-winter N_2O emissions shows a summary result of -23% (95% CI = - 28% to -16%) (Fig. 5.1, Table 5.2), indicating that on average, NI + UI significantly reduced N_2O emissions. Under NI+ UI, no significant difference was observed in N_2O emissions between humid and dry climates, although emissions were lower in humid areas. However, soil texture significantly impacted emissions under NI+ UI (Fig. 5.4, Table 5.S2). This effect was more evident in medium-textured soils than coarse soils. This meta-analysis result concurs with a modelling study (Grant et al. 2020) which showed that nitrification inhibitors (NI) could reduce N_2O emissions from fall-applied slurry by up to 33%. Our review included various types of NI and UI — 3,4-dimethyl pyrazole phosphate (DMPP), Nitrapyrin, dicyandiamide (DCD), N-(n-butyl) thiophosphoric triamide (NBPT), hydroquinone (HQ). It has been shown that some inhibitor types could be more effective in reducing N_2O emissions than others. For instance, Lin et al. (2017) revealed that the reduction coefficients for DMPP and Nitrapyrin were 81% and 57%, respectively. Ding et al. (2015) also showed that DCD achieved more N_2O reduction (78.6%) than NBPT (50%).

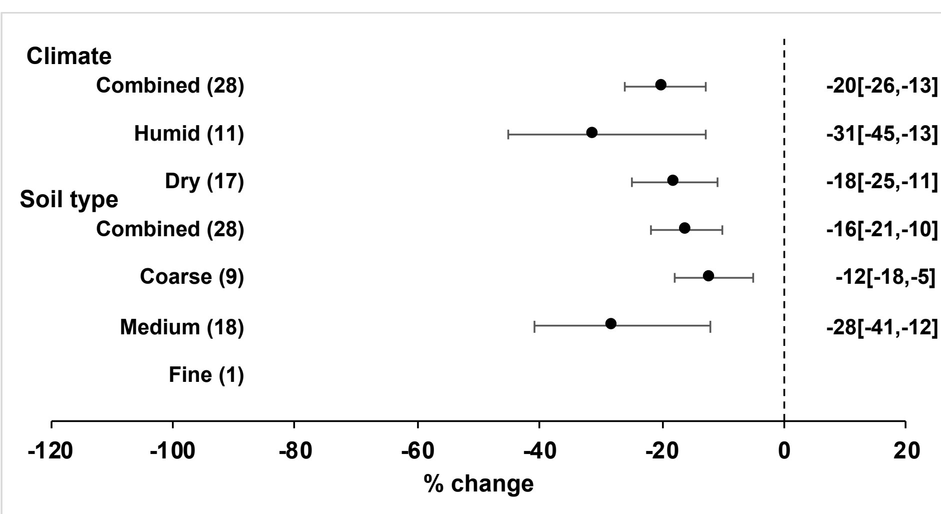

Figure 5.4: Forest plot of the random effect meta-analysis indicating subgroup (climate and soil type) effects of nitrification and urease inhibitors (NI + UI) vs. no inhibitors. Confidence intervals

are also reported. CI crossing the null (zero) line indicates no significant change in effect or that both treatments show equivalent effects. Only data for groups with more than one study is presented.

Our finding together with earlier meta-analysis that focused majorly on the growing season (Feng et al. 2016; Qiao et al. 2015; Thapa et al. 2016; Yang et al. 2016) all agree that the use of NI+ UI to prevent oxidation of NH_4^+ to N_2O is an effective method for reducing N_2O losses. The significantly lower over-winter N_2O emissions observed with NI + UI application could result from low soil temperatures during winter since the efficacy of NI + UI for reducing N_2O emissions has been shown to depend on temperature. For instance, in their study of UK soils, McGeough et al. (2016) measured 89-, 37- and 18-day DCD inhibitors half-lives ($t_{½}$) in soils at 5°C, 15 °C and 25 °C, respectively. Similarly, Kelliher et al. (2008) found DCD inhibitors' $t_{½}$ values in the soil to be 110 days at 5 °C, compared to 20 days at 25 °C. NI + UI's more prolonged efficacy at colder temperatures ensures the NI stays effective, particularly during the spring thaw period when N loss potential is typically high.

We further separated fall fertilizer + inhibitor application studies (Lin et al. 2017; Parkin and Hatfield 2010; Ding et al. 2015) from the rest of the inhibitor studies to investigate the effect size of fall application vs. prior spring application. We found that fall and prior spring application of inhibitors significantly reduced over-winter emissions by 12 and 14 %, respectively. Fall applied inhibitors likely remained active by the following spring given their shorter duration and the low temperature during winter, explaining their effectiveness in significantly lowering over-winter N_2O emissions. However, the reason for the effectiveness of inhibitors applied in prior spring remains unclear given that the inhibitors should have degraded by the winter season. Additionally, it is hypothesized that the use of inhibitors, causing more N uptake by plants, could lower the C:N ratio in residues after harvest (Grant et al. 2020; Pfab et al. 2012). Ferrari Machado et al. (2020) suggested that the carryover effect of inhibitor treatments increasing soil N levels could be negligible after observing low N_2O fluxes during both the growing and non-growing seasons compared to the conventional N sources. Other researchers have attributed lower over-winter emissions in NI + UI plots to the reduction of denitrifying bacteria rather than NO_3^- availability, since NO_3^- levels were the same as those found in conventional plots (Pfab et al. 2012). However, a contradictory finding shows a NI such as DMPP to have significantly reduced NH_4^+ oxidation on grasslands, though it showed no adverse effect on the denitrifier community's growth and activity

(Duan et al. 2017). More studies are needed to understand the carryover effect of spring-applied inhibitors on over-winter N_2O emissions. Further research could focus on the effect of inhibitors on the microbial community.

The result of our meta-analysis showed that soils with medium texture significantly influenced the efficacy of NI + UI in reducing N_2O emissions more than coarse soils (Fig 5.4). Medium-textured soils have larger surface area which allows the soil to hold more water. NI + UI's effectiveness has been tied to WFPS, Lin et al. (2017) showing a wetter site to exhibit a noticeable beneficial effect of NI compared to a less wet site. This scenario was due to increased N_2O production in the control plots resulting from high soil WFPS compared with the lower N_2O production in NI plots where fall added NIs prevented the oxidation of manures. This impact of WFPS on NI + UI's efficacy may also be the reason for the lower N_2O emissions observed in humid areas compared to dry areas. Similarly, Lin and Hernandez-Ramirez (2020) observed a greater NI efficacy in soils with increasing moisture content when the potential for production was high. However, after a few days, when WFPS surpassed field capacity, NI's potential to reduce N_2O emissions from the soil diminished. This reduced efficiency was possibly due to the complete denitrification of NO_3^- to N_2 in control plots leading to lower N_2O emissions or the lapsing of NI effectiveness in NI treatment plots, giving rise to the possible oxidation of NH_4^+ to N_2O. Increasing NI rates has been recommended to ensure NI's continued efficacy in soils with high moisture content. For instance, the most efficient NI rates were achieved by increasing the Nitrapyrin rate by 25% as soil moisture increased from 60 to 80% WFPS (Lin and Hernandez-Ramirez 2020).

However, caution must be applied interpreting these results as NI + UI efficacy to inhibit NO_3^- production may also be affected by other soil properties, including organic matter, clay content, soil Cu, and total N (McGeough et al. 2016). For instance, NI's lower inhibitory efficiency has been reported to result from high rates of adsorption by clay and SOC (Zhu et al. 2019). Therefore, it may be the case that there are variations in environmental and soil factors such as moisture, temperature, substrate availability, and management decisions such as timing, quantity, and placement of the NI + UI, that could affect the efficacy of NI + UI. Further studies investigating how the combination of these factors interact with various NI rates over winter are needed. Given the potential trade-offs of NI application, including an increase in NH_3 emissions and possible NI contamination, it is essential that studies also account for volatilization and N-leaching in addition to N_2O measurements. Overall, NI + UI practice significantly reduces both over-winter and growing season N_2O emissions. NI + UIs have also been shown to increase NUE use by 13% and boost yield by an average of 7.5%, and by adopting NI + UI practice, farmers could potentially gain up to US$6 per ha (Kanter and Searchinger 2018).

5.8 Conclusion

Occurring predominantly at the onset of the winter freezing period and in early spring when the soil begins to thaw, over-winter N_2O emissions make up a significant portion (up to 91%) of annual cropland emissions. We discussed the overall efficacy of nitrification and urease inhibitors (NI + UI), no-till and cover crops to mitigate emissions. We show that while no-till and the use of NI + UI significantly reduced over-winter N_2O emissions by 28% and 23%, respectively, the use of cover crops overall did not cause any significant change in the emissions. Although CCs have the potential to mitigate N_2O emissions by assimilating NO_3^- over winter, the production of denitrification substrates in the cover crops' root zone could hinder this potential. This review revealed that cold-dry soil conditions, such as in Western Canada, could prevent cover crops from being fully established in some years, hampering this management practice's effectiveness for reducing N_2O emissions in these regions. To effectively mitigate the significant freeze-thaw emissions, we recommend the enactment of policies that will encourage farmers' adoption of NI + UI in addition to other management practices such as no-till and the use of cover crops. We highlight the need for more extensive research focusing on the effect of the interaction of soil and climate conditions with field practices on the magnitude of emissions. Further studies could focus on the roles of denitrifying and nitrifying genes on over-winter N_2O emissions.

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5.10 Appendix: Supplementary Information

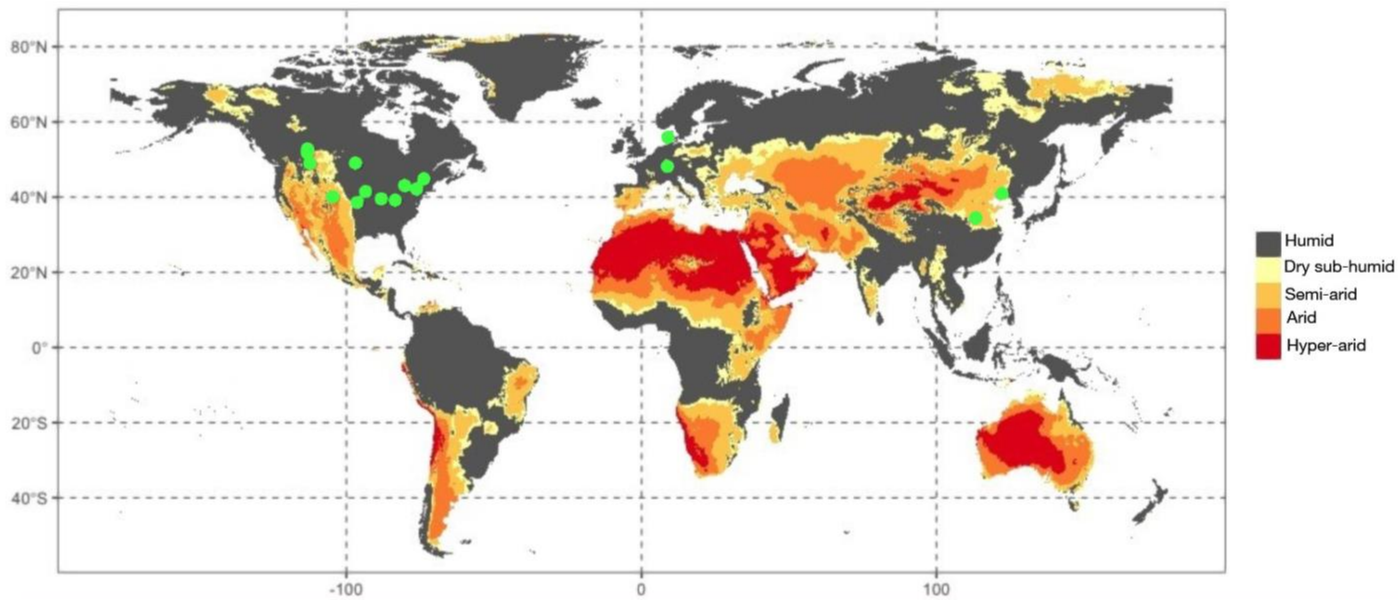

Figure 5.S1: The distribution of study sites utilized in this meta-analysis

Table 5.S1: Description of location, crop, climate, and soil types reported in this meta-analysis

Treatment	Country	Location	Crop	NI, UI type	Climate class	Soil type	Soil class	Reference
Tillage/CC	USA	Illinois	corn, soybean (CC: rape, cereal rye, radish, ryegrass, spring oats)	–	humid	silty clay loam (70%), silty loam (20%)	medium	Behnke and Villamil (2019)
Tillage	Canada	Ontario	corn, soybean	–	humid	silt loam	medium	Congreves et al. (2017)
Tillage	Canada	Ontario	corn-winter wheat	–	humid	silt loam	medium	Wagner-riddle et al. (2007)
Tillage	Canada	Alberta	wheat	–	dry	loamy sand	coarse	Lemke et al. (1999)
Tillage	USA	Ohio	corn	–	humid	silt loam	medium	Ussiri et al. (2009)
Tillage	Denmark	Foulum	winter barley	–	humid	coarse sandy loam	coarse	Mutegi et al. (2010)
Tillage/CC	Denmark	Foulum	fodder radish	–	humid	coarse sandy loam	coarse	Petersen et al. (2011)
Tillage	Canada	Ontario	corn, soybean and winter wheat	–	humid	silt loam	medium	Ferrari Machado et al. (2021)
Cover crop	USA	Iowa	rye and oat (70/30 mix)	–	humid	silty clay loam, fine loamy	medium	Jarecki et al. (2009)
Cover crop	USA	Kansas	daikon Radish and Sorghum Sudangrass	–	humid	silty clay loam	medium	Preza-Fontes et al. (2020)
Cover crop	Canada	Alberta	Rye and oilseed radish	–	dry	clay loam	medium	Thomas et al. (2017)

Cover crop	USA	New York	Rye	–	humid	gravelly loam	coarse	Dietzel et al. (2011)
Cover crop	Canada	Quebec	rye grass, hairy vetch	–	humid	sandy loam	coarse	Hung et al. (2021)
Nitrification inhibitor	Canada	Alberta	barley for silage	DMPP and Nitrapyrin	dry	(sandy/clay loam)	medium and fine	Lin et al. (2017)
Nitrification inhibitor	Germany	Stuttgart	lettuce - cauliflower	DMPP	humid	silty clay loam	medium	Pfab et al. (2012)
Nitrification inhibitor	USA	Iowa	corn	Nitrapyrin	humid	silty clay loam	medium	Parkin and Hatfield (2010)
NI + UI	USA	Colorado	corn	SuperU and AgrotainPlus	dry	clay loam	medium	Halvorson and Del Grosso (2012)
NI + UI	Canada	Manitoba	barley, wheat, rapeseed	SuperU (ie DCD and NBPT)	humid	clay	fine	Asgedom et al. (2014)
NI + UI	China	Shenyang	corn	DCD, HQ	dry	sandy loam	coarse	Dong et al. (2018)
NI + UI	Canada	Ontario	corn	DCD and NBPT	humid	silt loam	medium	Ferrari Machado et al. (2020)
NI + UI	China	Henan	wheat	NBPT and DCD	dry	sandy loam	coarse	Ding et al. (2015)

CC = Cover crop; NI = Nitrification Inhibitor; UI = Urease Inhibitor; DCD = Dicyandiamide; HQ =Hydroquinone; DMDP = 3,4-dimethyl pyrazole phosphate

5.10.1 Data selection

Three pairs of agronomic strategies were selected and quantitatively evaluated for their effect on winter N_2O fluxes: (*i*) no tillage *vs*. conventional tillage, (*ii*) cover crop *vs.* no cover crop, and (*iii*) fertilizer applied with urease and nitrification inhibitors *vs*. fertilizer application without inhibitors. A literature search of the Scopus and Web of Science databases was undertaken, applying — one-at-a-time — a particular mitigation strategy (tillage, cover crop, nitrification and urease inhibitors), combined with other keywords (thaw, winter, non-growing season, emissions). The search was limited to studies with agriculture and the environment as subject areas. To be included in the meta-analysis, studies had to meet the following criteria: (*i*) be a field study, (*ii*) have replicated treatments, (*iii*) provide information on the daily mean flux or cumulative emissions, and (*iv*) provide a measure of the magnitude of variance (including P-value, standard error, standard deviation (SD), confidence interval). When the variance was not provided, an average of the SD from other studies in the same meta-analysis was applied. Plot digitizer 2.6.9 served in gathering data when graphs were provided.

Studies were grouped based on climatic conditions and soil types. For grouping the studies according to climate, we used the Köppen-Geiger climate classification to select only temperate or snow regions with annual minimum threshold temperatures of $\leq -3^{o}C$ (Kottek et al. 2006). The climate aridity index (the ratio of the rainfall to potential evapotranspiration) of the study area was used to further classify the regions as either humid or dry (van Kessel et al. 2013). Study areas with an aridity index of > 0.65 were classified as humid, and those below this threshold were classified as dry. The soils were classified as coarse (sandy loam, loamy sand, sandy clay loam), medium (loam, silt loam, clay loam, silty clay loam, silt), or fine (clay, silty clay, sandy clay) (Shakoor et al. 2021).

5.10.2 Meta-analysis

A random effect meta-analysis was conducted to compare effect sizes between each pair of treatments. The natural log of the response ratio was used for quantifying the effectiveness of a treatment option relative to the alternative (or control).

$$\ln RR = \ln\left(\frac{X_t}{X_a}\right) = \ln X_t - \ln X_a$$

where X_t and X_a are the averages of the cumulative winter N_2O emissions for the treatment and alternative groups respectively. Data were extracted for respective years and (or) treatment pairs. The variance in each study was calculated by

$$v = \frac{SDt^2}{nX_t^2} + \frac{SDa^2}{nX_a^2}$$

where SDt and SDa are the standard deviations for the treatment and the alternative, and n is the number of replicates in treatment and the alternative. The weight of the individual effect sizes was calculated as the inverse of the variance:

$$w = \frac{1}{v}$$

The effect sizes were weighted to assign greater weights to experiments that were well replicated, such that the mean effect size was calculated as:

$$\overline{\ln RR} = \frac{\sum(\ln RR_i \times w_i)}{\sum(w_i)}$$

where RR_i and w_i are the effect size and weight, respectively, from the i^{th} comparison.

The response ratio analyses were conducted using the R package, *metafor* 2.4-0. Analysis results were visualized using forest plots displaying the estimated mean effects of treatment pairs together with their 95% confidence interval. For straightforward interpretation, the results are presented as percentage changes between compared variables, calculated as follows:

$$\left(e^{\overline{\ln RR}} - 1\right) \times 100\%$$

The studies' subgroups were compared using the Q-test for heterogeneity to assess the dispersion of the mean effects of the subgroups about the combined effect. In cases where Q-test was significant, the magnitude of the difference was quantified.

Table 5.S2: Results of the statistical analysis

Treatment	*p*-val (Z-test)	Q	df	*p*-val	I^2	Estimate	CI.lb	CI.ub
Tillage	< .0001	990.7	31	< .0001	96.87	0.72	0.61	0.84
Climate	0.000	990.7	31	2.75E-188				
Humid	0.006	518.4	19	7.651E-98	96.33	0.82	0.71	0.94
Dry	0.001	79.8	11	1.58E-12	86.22	0.53	0.37	0.68
Effect difference*				0.032		0.65	0.44	0.96
Soil type	0.001	990.7	31	2.75E-188				

Coarse	0.011	213.2	14	1.10E-37	93.43	0.59	0.4	0.89
Medium	0.007	506.0	16	1.77E-97	96.84	0.81	0.7	0.94
Effect difference				0.153		1.37	0.89	2.11
Cover crop	0.089	218.8	27	<0.0001	87.66	0.82	0.65	1.03
Climate	0.356	179.9	27	1.58E-24				
Humid	0.062	170.2	23	1.95E-24	86.49	0.74	0.54	1.01
Dry	0.226	8.1	3	4.32E-02	63.15	1.41	0.7	2.81
Effect difference*				0.043		1.9	0.89	4.06
Soil type	0.630	218.8	27	6.31E-32				
Coarse	0.219	169.8	12	5.26E-30	92.93	0.57	0.23	1.4
Medium	0.777	31.4	14	4.86E-03	55.42	0.98	0.84	1.14
Effect difference				0.243		1.72	0.69	4.25
NI + UI		206.0	27	<0.0001	86.90	0.77	0.72	0.84
Climate	<0.0001	206.0	27	1.75E-29				
Humid	0.002	44.7	10	2.42E-06	77.65	0.69	0.55	0.87
Dry	<0.0001	131.1	16	3.90E-20	87.80	0.82	0.75	0.89
Effect difference				0.172		1.18	0.93	1.52
Soil type	0.000	206.0	27	1.75E-29				
Coarse	0.001	66.4	8	2.50E-11	87.96	0.88	0.82	0.95
Medium	0.001	78.6	17	6.67E-10	78.38	0.72	0.59	0.88
+Fine	<0.0001	0.0	0	1	0.00	0.24	0.14	0.42
Effect difference (coarse *vs.* medium)*				0.0547		0.81	0.66	1.01

Note:

Estimates and confidence intervals (CI) are in raw formats (i.e. not converted to %)

p-val = test of difference; df= degree of freedom; Q = Cochran's Q-statistic; I^2 = Higgin's and Thompson's percentage of variability

+ only sub-groups with df >1 were further discussed in the main text

* significant at P<0.1

Connecting Text

The findings from the meta-analysis in Chapter 5 suggest that N_2O emissions that occur over the winter period can be significant. The meta-analysis clarified the effectiveness of no-till, cover crops, nitrification, and urease inhibitors in mitigating the considerable N_2O losses over winter. While recent efforts have characterized N_2O emissions over the winter period, they have not adequately addressed the impact of climate variables, including soil temperature, soil moisture and snowpack on the emissions, particularly given the changing climate. This understanding is essential to better target emission reduction efforts. Chapter 6 used DNDC to investigate the historical and future effects of the changing weather conditions on winter N_2O emissions. The DNDC model used in this study had earlier been calibrated by Dr. Jiang Qianjing using experimental measurements of soil moisture, soil temperature, GHG flux, crop yield, and drainage at the St Emmanuel study site. In this study, the model was validated using observed field data from the study site, including the 2019 early spring N_2O emissions data.

The following manuscript co-authored by Dr. Jiang Qianjing and Dr. Chandra Madramootoo has been prepared for submission in the Environmental Research Letters journal:
Ekwunife, K.C., Madramootoo, C.A and Jiang, Q. 2022. Nitrous oxide emissions from intensive crop production under changing winter and spring conditions.

Contribution of authors:

Ekwunife K.C., was responsible for conceptualization, methodology, fieldwork (for 2018 and 2019), analysis, data curation, and writing the original draft, followed by reviews and editing. Dr. Jiang Qianjing calibrated the DNDC model and provided the dataset for simulating future scenarios. Dr. Madramootoo acquired the funds for the project, provided supervision, and assisted in conceptualizing, reviewing, and editing the manuscript.

Chapter VI

Nitrous oxide emissions from intensive crop production under changing winter and spring conditions

6.1 Abstract

Nitrous oxide (N_2O) emissions in the non-growing season (December-April) in cold temperate regions could be quite significant. The extent of emissions is further influenced by rapid soil freezing and thawing in the late winter-spring. Using the Denitrification and Decomposition (DNDC) model, N_2O fluxes and climatic triggers during winter and early spring were assessed by simulating historical and future winter emissions over a 30-year period for intensive grain corn production in Southern Quebec. A historical analysis showed the greatest average winter N_2O emissions to occur in warm-wet years. Winter N_2O emissions for the historical period [1990-2019] increased with lower snow water equivalent (SWE). Future scenario simulations [2038 - 2067] projected a 10% rise in winter N_2O emissions, associated with an average winter soil temperature increase of 1°C and soil moisture (WFPS) increase of 8%. The SWE trend magnitude projected for the future period is -1.03mmyr^{-1}, implying a snowpack reduction of about 12cm over the 30 years period (assuming a 250kgm^{-3} average snowpack density). These results imply that as SWE gradually declines and soil temperature and moisture increase in the future, greater winter N_2O emissions will occur. This study points to the need to focus on mitigation measures for winter N_2O emissions from agricultural soils.

6.2 Introduction

In cold regions, such as the North Atlantic region of the US and Canada, nitrous oxide (N_2O), a major greenhouse gas, is often emitted in substantial quantities from arable lands during the non-growing season (December - April). This occurs mainly in response to freeze-thaw cycles (FTC) that occur during the winter freezing period and the early spring thawing period (Chen et al. 2020; (Teepe et al. 2001). In this cold region, winter N_2O emissions from croplands can account for 5% to 91% of total annual N_2O emissions (Ekwunife, Madramootoo, and Abbasi 2022). Prior studies on winter cropland emissions have linked the variability in their magnitude to differences in substrate availability, soil temperature and moisture conditions (Matzner and Borken 2008; Chen et al. 2016; Smith, Wagner-Riddle, and Dunfield 2010; Zhe et al. 2018). Previous research has indicated that soil environmental factors including moisture and temperature play a more critical role in modulating winter emissions than does substrate availability (Yin et al. 2019). Historical climate data show that trends in climate variables have changed over the years. For instance, the likelihood of precipitation falling as rain rather than snow is heightened with increasing temperatures (Trenberth, 2011). Decreasing quantity of snow and earlier spring snowmelt associated with rising temperatures are known impacts of climate change observed in eastern Canada and in northeastern North America (Qian et al. 2011; Grogan et al. 2020). Freeze-thaw cycles are projected to increase under warming winter conditions (Hatami & Nazemi, 2022) and this can significantly impact over-winter N_2O emissions. However, studies investigating the historical and future effect of changing climatic conditions on winter N_2O emissions remain scarce. Given the complex interplay of soil parameters and climatic influences on winter emissions, an understanding of the way these might interact to effect emissions may offer insights into how winter N_2O emissions would vary under a changing climate and what type of mitigation practices could be implemented.

Soil temperature, snowpack and soil moisture affect FTC-induced N_2O emission. Higher N_2O emissions have been observed with lower soil freezing temperature (–15 ^oC vs. –1.5 ^oC)(Koponen et al., 2006) and longer freeze durations (Wagner-Riddle et al., 2017). The lower soil freezing temperature and longer freeze durations result in freeze stresses that can kill some microbes, returning microbial polymers and other organic monomers to soil carbon and dissolved organic nitrogen (DON) pools (Schimel and Bennett 2004; Larsen, Jonasson, and Michelsen 2002; Sharma et al. 2006). Surviving microbes that gradually acclimatize to subzero temperatures can then draw upon these substances to stimulate N_2O emissions (Chen et al. 2020; Pelster et al. 2013). Snowpack also influences N_2O fluxes, as it provides insulation for the soil, thereby reducing soil temperature variation (Maljanen et al. 2007). In other words, snowpack prevents soil freezing or allows only freezing to very shallow depths. In the presence of an impermeable ice barrier in the snow layers above the soil surface, gases are trapped in the soil, causing soil N_2O concentrations to increase over time. Partial release of such trapped gases can occur through cracks caused by increased temperature and slight thawing of the soil (Teepe et al., 2001). Greater N_2O emissions occur during the snowmelt periods than in the snow-covered period (Zhe et al. 2018; Chen *et al.*, 2020). The emission of N_2O, mainly attributable to anaerobic conditions prevailing at the onset of thawing, has also been observed (Risk *et al*., 2014). The anaerobic conditions required for the denitrification process are enhanced by high soil moisture resulting from snowmelt, a frozen subsoil and the availability of nitrates and simple carbon compounds (Nyborg et al., 1997; Priemé & Christensen, 2001; Sehy et al., 2004; Jan Willem Van Groenigen, Georgius, et al., 2005). Therefore, in addition to snowpack and soil temperature, soil moisture could impact on the magnitude of winter N_2O emissions.

Given the changes in climate, it is necessary to investigate whether these changes would lead to a change in winter N_2O emissions. Understanding how these variables would likely impact emissions, especially under a changing climate, is essential to better target emission reduction efforts. The denitrification and decomposition model (DNDC) model can be utilized to investigate trends of winter N_2O fluxes and to understand the freeze-thaw processes influencing the emissions. The DNDC model predicts daily N_2O gas fluxes for the growing and winter seasons. Although initially developed to estimate N_2O emissions by simulating soil biogeochemical cycling under various climate and management conditions, the model has been expanded to simulate soil C and N dynamics, crop growth, soil temperature, soil moisture, and other trace gases. More details on the model have been documented in previous reports (Gilhespy et al., 2014; Giltrap, Li, & Saggar, 2010). The DNDC model has been used to predict N_2O emissions at various locations, including Canada (Gilhespy et al., 2014). The DNDC model's applicability in simulating freeze-thaw N_2O emissions has been assessed by various authors (Norman et al. 2008; Kariyapperuma et al. 2011; Smith et al. 2002). Rather than the model's initially-implemented mechanism of ice-trapped N_2O fluxes, recent improvements account for the newly considered N_2O emissions resulting from anaerobic soil conditions that allow the primary mechanisms for freeze-thaw emissions in cold regions to prevail, (Kariyapperuma et al., 2011). DNDC's heat transfer routine has also been recently improved to simulate soil temperatures for soils in cold areas under various management practices (Dutta et al., 2018).

We conducted DNDC model simulations to determine the impact of changing precipitation and temperature on historical and future winter emissions. Accordingly, the objectives of this study were to: (i) analyze historical climate data for winter (Dec-Apr) precipitation, snowfall, and minimum temperature trends and assess the impact of weather scenarios (*e.g*., cold-wet, cold-dry, warm-wet, and warm-dry) on winter N_2O emissions using model simulations; (ii) investigate the relationship between winter N_2O emissions and climate variables (e.g., snowpack, soil temperature, and soil moisture); and (iii) analyze how winter emissions might change over a 30-year future scenario (2038-2067).

6.3 Materials and Methods

6.3.1 Study site

A field experiment to measure N_2O fluxes was conducted in 2018 and 2019 on a 4.2 ha corn-soybean farm in Saint-Emmanuel-du-Côteau-du-Lac (45°19'N, 74°9'W) Quebec, Canada. The crop planted in the two years was corn (*Zea mays* L.). The soil is classified as Soulanges very fine sandy loam with 5.0% organic matter in the top layer (0-0.25 m), underlain by a sandy clay loam (0.25-0.55 m) and clay (0.55-1.00 m) layers. The field was drained with subsurface tile drainage buried at a depth of 1 m. Growing season gas emission measurements were carried out in 2018 and 2019 using a vented non-steady-state closed chamber method as previously reported by Crézé and Madramootoo (2019). In 2019 gas sampling was extended to capture the fluxes that occurred during the spring thaw period. Gas sampling along with soil moisture and temperature measurements were performed three times per week during the spring thaw and nitrogen application period to capture anticipated peak fluxes, then once every 1-2 weeks for the remainder of the growing season.

6.3.2 The Denitrification – Decomposition (DNDC) model

An earlier calibrated denitrification and decomposition model (DNDC) (Jiang et al. 2022) was validated using the observed two years' (2018 and 2019) N_2O fluxes. The DNDC model was used to investigate trends of winter N_2O fluxes and to understand the freeze-thaw processes influencing the emissions. The DNDC model predicts daily N_2O gas fluxes for both the growing and winter seasons. Although initially developed to estimate N_2O emissions by simulating soil biogeochemical cycling under various climate and management conditions, the model has been expanded to simulate soil C and N dynamics, crop growth, soil temperature, soil moisture, and other trace gases. More details on the model have been documented in previous reports (Gilhespy et al., 2014; Giltrap, Li, & Saggar, 2010). The DNDC model has been used to predict N_2O emissions at various locations, including Canada (Gilhespy et al., 2014). The model's applicability in simulating freeze-thaw N_2O emissions has been assessed by multiple authors (Norman et al. 2008; Kariyapperuma et al. 2011; Smith et al. 2002).

The DNDC model was used to assess freeze-thaw processes by predicting soil temperature, water content, and snowpack changes. Snowpack changes were measured from snow water equivalent (SWE), the amount of water that results when snowpack completely melts. Previous studies ascertaining changes in snowpacks have also been based on snow water equivalent (Dutta et al., 2018; Gan et al., 2013). The daily weather data, collected from a nearby weather station (Coteau-du-Lac, elev. 74.6 m, 45.29°N, 74.2°W) situated 500 m from the research plots, included six variables: maximum and minimum air temperature, precipitation, solar radiation, wind speed and relative humidity. Gaps in the weather data were filled using data from NASA (https://power.larc.nasa.gov/).

The DNDC model was run for the historical 30-year period (1990–2019) and the 30-year (2038-2067) future period to simulate the interactions between winter N_2O fluxes and climate-influenced soil variables (e.g., snowpack, soil water content and temperature). Soil water content and temperature were considered at the 0.10m depth. For each simulated year, the winter and early spring period covered a total period of 151 days (December 1 to April 30) representing DOY 335-366 of the specified year and DOY 1-120 of the following year, while annual emissions spanned from May 1 of the specified year to April 30 of the following year.

6.3.3 Scenario analysis

6.3.3.1 Historical scenarios: cold-wet, cold-dry, warm-wet, and warm-dry

The DNDC model was applied to achieve two goals. The first goal was to investigate the effect of four historical weather scenarios on the magnitude of winter N_2O emissions. This knowledge is crucial to understand how climate variables and associated N_2O emissions have changed over the recent years. The impact of historical climate conditions on N_2O emissions was investigated by exploring weather for the study site from 1990-2019, separating individual years into four different weather scenario categories (*e.g.*, cold-wet, cold-dry, warm-wet, and warm-dry), according to the year's total precipitation and average minimum air temperature during the winter season (Fig 6.1). We then used the validated DNDC model to estimate total winter emissions and averages of the climate variables under these weather scenarios and identify distinct patterns. Given that future climate projection points to a wetter and warmer climate, this analysis could also point to how the changing climate could affect future N_2O emissions.

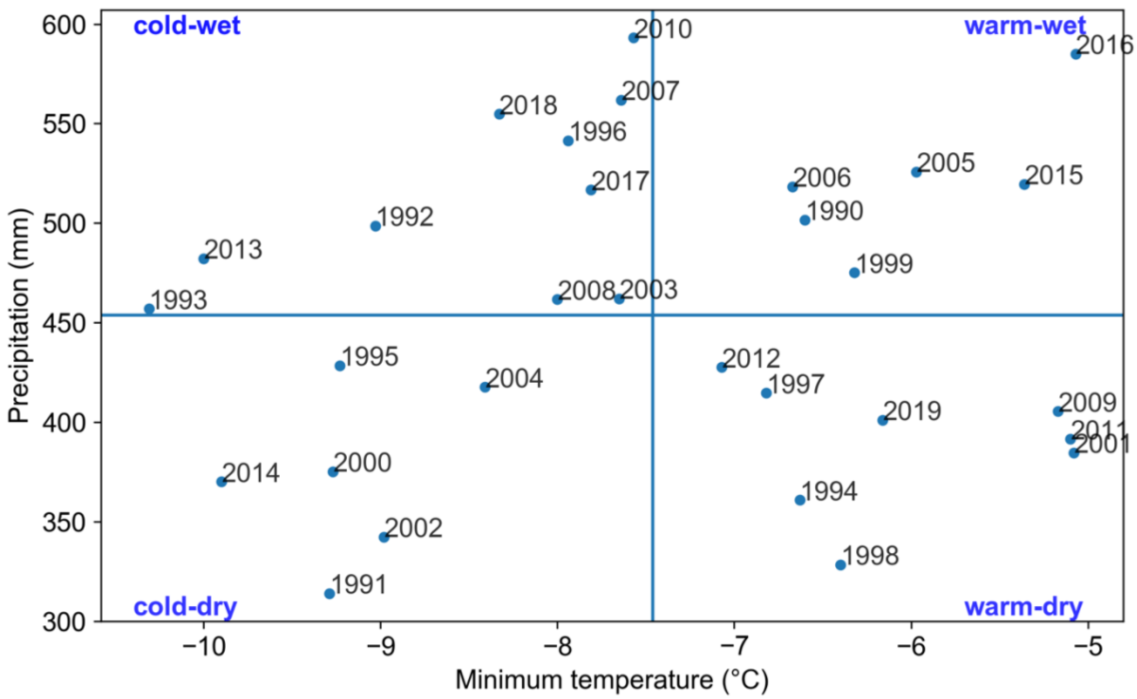

Figure 6.1:Winter season total precipitation and average minimum air temperature from 1990-2019 at the main study site. The lines crossing the figure represent the 30-year mean cumulative precipitation (454 mm) and mean minimum temperature (-7.5 ^{o}C).

6.3.3.2 Future climate scenario

The second goal of the DNDC model application was to study the impact of climate change on future winter N_2O emissions at the site by predicting winter climate variables and associated N_2O emissions under high emission IPCC climate change scenario. This analysis of potential future conditions was done to validate the insights gained from the historical scenarios analysis. N_2O emissions were simulated for the future period with 30-year (2038-2067) projected climate scenarios at a 548 ppm atmospheric CO_2 concentration (Qianjing Jiang et al., 2020). We assumed similar model parameters and cropping practices for historical and future climate scenarios. Future climate data from 10 climate models were obtained from the North American Climate Change Assessment Program (NARCCAP), an international program to produce high-resolution climate change simulations to investigate uncertainties in regional scale projections of future climate and to generate climate change scenarios for use in impact research. The NARCCAP data are derived from a combination of General Circulation Models and Regional Climate Models (GCMs-RCMs). The regional climate models (RCMs) driven by a set of atmosphere-ocean general circulation models (GCMs) over a domain covering the conterminous United States and most of Canada. More information on NARCCAP can be found on its website (www.narccap.ucar.edu). Ten coupled General Circulation Model and Regional Climate Model (GCM-RCM) were used to generate ten different future climatic scenarios. Annual winter (Dec-Apr) average climate variables for historical and future scenarios are presented in Table 6.1. Under these scenarios, future maximum and minimum temperatures for the winter period are projected to rise by 2°C and 3.8°C, respectively. Precipitation could increase by 9.6%, while relative humidity could decrease by 18%. Wind speed and solar radiation for the winter period are projected to increase by 25.7% and 32.3%, respectively.

Table 6.1: Annual winter average climate variables for historical (1990-2019) and different future scenarios (2038-2067)

	Climate variable	**Tmax (°C)**	**Tmin (°C)**	**Precipitation (cm d^{-1})**	**Windspeed (ms^{-1})**	**Solar Radiation (MJ m^{-2} d^{-1})**	**Relative Humidity %**
Winter (Dec - Apr)	Historical	0.3	-8.8	0.25	3.5	9.5	86
	CRCM_ccsm	2.3	-4.8	0.28	4.6	12.7	73
	ECP2_gfdl	2.5	-4.5	0.32	4.6	11.5	70
	HRM3_gfdl	3.1	-4.9	0.25	4.5	13.0	70
	HRM3_hadcm3	2.3	-5.6	0.25	4.5	13.0	69
	MM5I_ccsm	2.0	-5.4	0.27	4.2	12.8	70
	MM5I_hadcm3	3.0	-3.9	0.26	3.7	12.5	70
	RCM3_cgcm3	2.3	-4.8	0.28	4.6	12.7	73
	RCM3_gfdl	1.5	-6.0	0.30	4.4	13.0	71
	WRFG_ccsm	2.4	-4.6	0.24	4.2	13.1	70
	WRFG_cgcm3	1.3	-6.0	0.29	4.9	12.6	71
	Future average	2.3	-5.0	0.27	4.4	12.7	71
	Difference	1.9	3.8	9.6%	25.9%	33.7%	-18%
Annual	Historical	11.4	1.7	0.29	3.0	12.8	81
	Future average	13.5	5.0	0.29	3.9	15.5	71
	Difference	2.1	3.4	-0.3%	29.9%	21.1%	-12%

6.3.4 Statistical analysis

The Pearson correlation coefficient was used to assess the relationship between predicted and forecasted winter N_2O emissions in relation to climate variables (soil temperature, SWE and soil moisture). An analysis of variance (ANOVA) was used to analyse the difference among means of all the climate variables and associated N_2O emissions assessed under the weather scenarios. We also applied the *post hoc* tests (Tukey) to identify differences between compared pairs of means for significance, a $p \leq 0.05$. The statistical analysis of modelled results to assess the DNDC model's 'goodness of fit' was carried out using three statistical equations:

$$PBIAS = \frac{\sum_{i=1}^{n}(O_i - P_i)100}{\sum_{i=1}^{n}(O_i)} \quad [1]$$

$$R^2 = 1 - \frac{\sum_{i=1}^{n}\left(O_i - O_{avg}\right)\left(P_i - P_{avg}\right)^2}{\sum_{i=1}^{n}\left(O_i - O_{avg}\right)^2 \sum_{i=1}^{n}\left(P_i - P_{avg}\right)^2} \quad [2]$$

$$d = 1 - \frac{\sum_{i=1}^{n}(P_i - O_i)^2}{\sum_{i=1}^{n}\left(|P_i - O_{avg}| + |O_i - O_{avg}|\right)^2} \quad [3]$$

where O_i is the observed value, P_i is the predicted value, n is the number of observations, O_{avg} is the average of the observed values, PBIAS is the percent bias, R^2 is the correlation coefficient and d is index of agreement. The model performance in predicting N_2O emissions was judged to be satisfactory based on the test statistics (Table 6.2; Fig 6.S1).

Table 6.2: Statistical evaluation of DNDC model for N_2O emissions ($kgNha^{-1}day^{-1}$), for 2018/2019 at the St Emmanuel Experimental Site, Coteau du Lac, QC, Canada

Statistics	Value	Acceptance range
PBIAS	13.9	$-15\% < PBIAS < 15\%$
R^2	0.63	> 0.5
d	0.73	> 0.7

6.4 Results and Discussion

6.4.1 Observed temporal changes in climate variables

We first investigated trends in time series of winter precipitation, snowfall and minimum air temperature. Our results show that in our study location in Eastern Canada, increasing trends in both winter precipitation (r = 0.55 p <0.05) and minimum air temperature (r = 0.34 p <0.05) have been observed between 1980 and 2020 (Fig 6.2a and 6.2b). The snowpack trend in our study remained stable (Fig 6.2c). It does, however, look like less snow was accumulated per precipitation event. This must mean there was greater snow-melt or more of the winter precipitation came as rain rather than snow. Some previously published assessments show a long-term decreasing snow depth trend over most parts of Canada (Brown & Braaten, 1998; Dyer & Mote, 2006). For instance, Dyer and Mote (2006) found that decreasing snow depth observed from the daily U.S and Canadian surface observations (1960–2000) began in late January and grew in intensity and extent through March and into April, indicating an earlier onset of spring melt. Brown and Braaten (1998) highlighted that snow depth decreases over much of Canada were characterized by an abrupt transition to lower snow depths in the mid-1970s. Our study's historical observations, which started in 1980, did not capture this abrupt transition and could explain why we did not observe a decrease in the historical snowpack trend.

We also assessed the potential impact of rising minimum winter temperatures on winter precipitation (snowfall and winter rainfall) by investigating the correlation between them. Our result reveals that rising minimum temperatures resulted in an inverse trend in snowfall (r = –0.35 p <0.05) but an increasing trend in winter rainfall (r = 0.4 p <0.05) (Fig 6.2d and 6.2e), indicating a warmer and wetter winter in the future. This finding agrees with that of Mekis and Vincent (2011), who also reported a wetter climate in eastern Canada under the 20th-century warming trend.

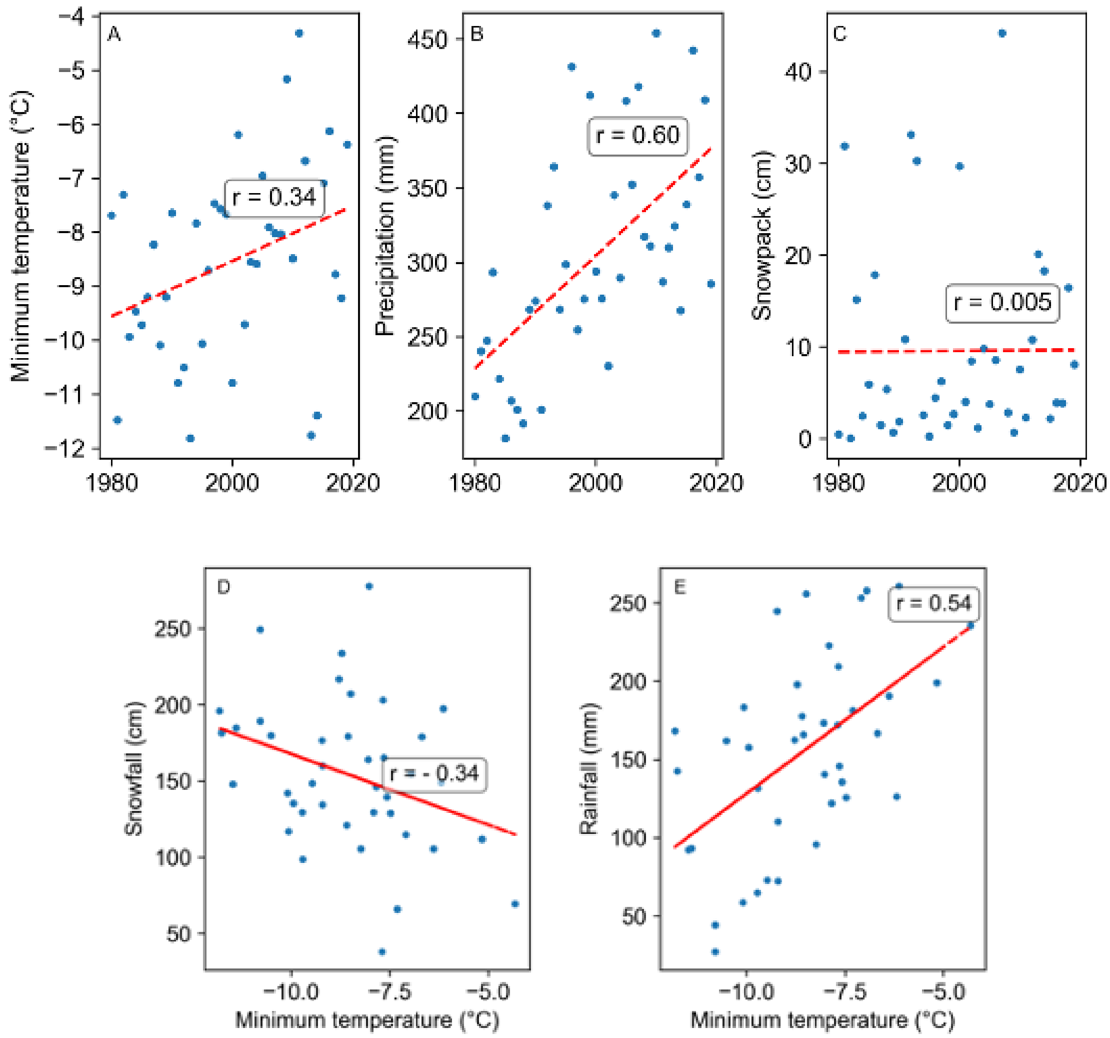

Figure 6.2: The trend of historical winter (Dec-Apr) (a) precipitation, (b) minimum temperature, (c)snowpack, (d) snowfall versus minimum temperature, and (e) winter rainfall versus minimum temperature at the study site for the historical period of 1980-2019. The trendline is indicated in red, and *r* represents the Pearson correlation coefficient.

6.4.2 Winter N_2O emissions under the four weather scenarios

The second goal of this study was to analyse historical weather data to determine the effect of four weather scenarios (*e.g*., cold-wet, cold-dry, warm-wet, and warm-dry) on winter N_2O emissions. The results of the analysis are shown in Fig 6.3 and Table 6.3. Simulated winter N_2O emissions ranged from 0.002 to 6.2 kg N ha^{-1}, representing 0.1–72.8% of annual N_2O emissions. The magnitude of emissions generated through our simulation proved similar to those from other studies (Ekwunife et al. 2022; Risk, Wagner-riddle, and Blodau 2014; Wagner-Riddle et al., 2017;

Asgedom et al. 2014). Mean N_2O emissions were greatest in the warm-wet years, although the ANOVA test revealed that the differences were only marginally significant (p-value ≤ 0.10).

Climate variables were also compared under the four scenarios using pairwise comparison of means. Among the groups compared, there were significant differences in SWE and soil temperature (p-value ≤ 0.05) but not in soil water content (p-value > 0.05). Our findings show that the SWE is significantly reduced (p-value ≤ 0.05) in warmer climates and that the average (Dec-Apr) soil temperature increased as SWE decreased. These findings are consistent with those of Qian et al. (2011) who reported that declining snowpack associated with increasing air temperatures commonly led to elevated soil temperatures during the early spring, which was not often observed during the winter. In our historical scenario analysis, soil water content within the groups did not differ significantly, probably because the soils tended towards saturation irrespective of the scenario.

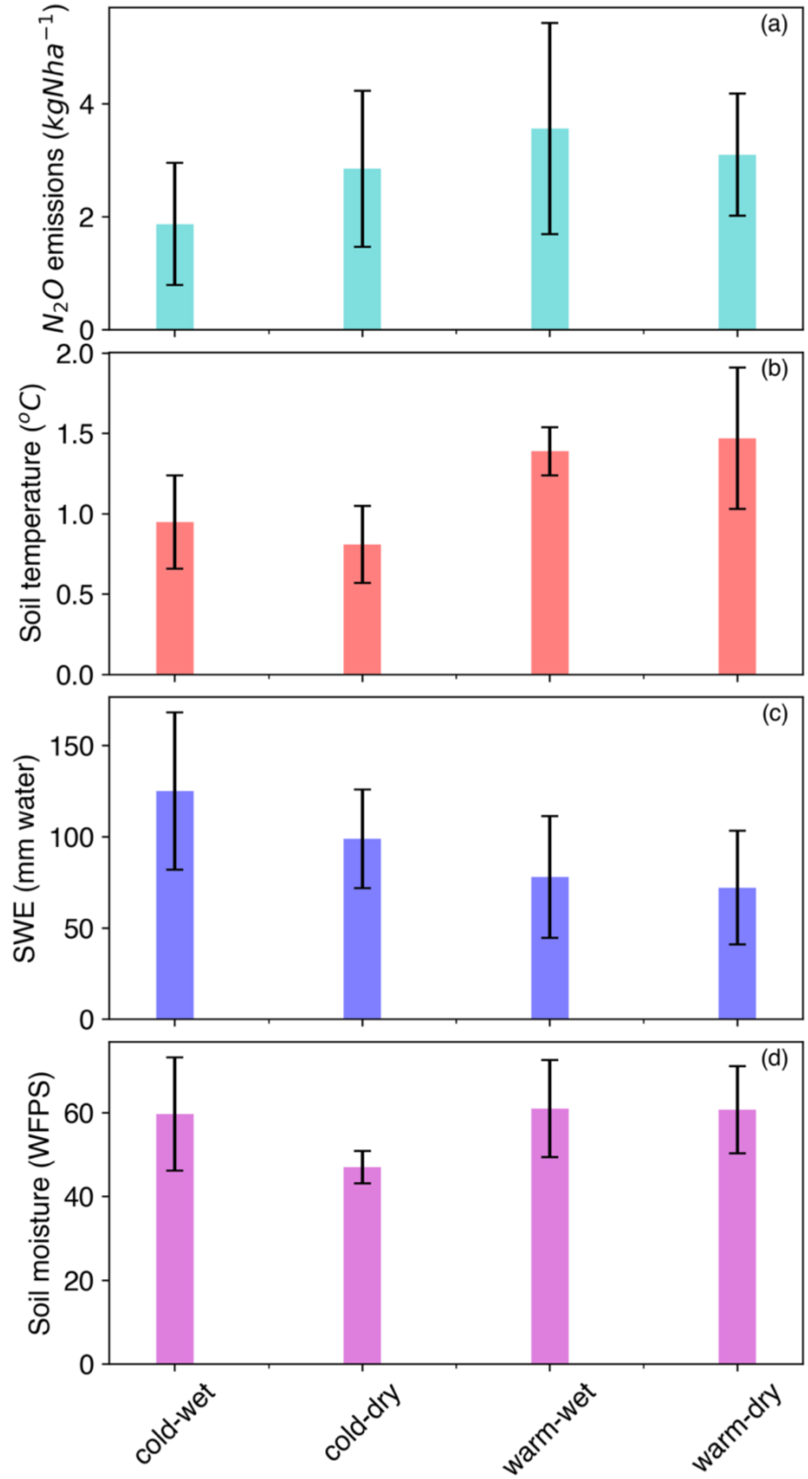

Figure 6.3: Simulated mean (a) winter N_2O emissions, (b) soil temperature, (c) SWE (mm water), (d) soil water content (WFPS) of the selected years under the four scenarios investigated from 1991-2019 at the main study site location. Error bars represent the standard deviations.

Table 6.3: Adjusted p-values for the multiple comparison of means -Tukey HSD for the variables compared under the four scenarios [Abbreviations represent: cd, cold dry; cw, cold wet; wd, wet dry; ww, wet warm seasons]

Variable	ANOVA test (p value = 0.05)	Multiple comparison of means - Tukey HSD (p value = 0.05)					
		cd-cw	cd-wd	cd-ww	cd-wd	cw-ww	wd-ww
Winter N_2O emissions	0.09	0.4949	0.9	0.7674	0.2335	0.0892	0.9
Soil temperature (°C)	0.007*	0.8301	0.0031*	0.0179*	0.0074*	0.0485*	0.9
SWE (mm water)	0.019*	0.4949	0.5116	0.7152	0.0203*	0.0719	0.9
Soil water content (WFPS)	0.09	0.139	0.1233	0.1495	0.9	0.9	0.9

6.4.3 Relationship between historical winter N_2O emissions and climate variables

The third goal of this study was to investigate the relationship between simulated historical winter N_2O emissions and climate variables, including snowpack, soil temperature and soil moisture. For illustrative purposes, the two years with greatest winter emissions (1999 and 2006) and lowest emissions (2007 and 2018) are presented (Figs 6.4 and 6.5), respectively. The figures also show the changes in snow water equivalent (SWE), air and soil temperature, and soil water content over the winter period. Winter N_2O fluxes were mostly observed in periods with reduced or no snowpack (e.g. Fig 6.4 and 6.5). Fluxes that occurred in the presence of snowpack could have resulted from cracks in the frozen soil when snowpack was not inhibiting gas fluxes. The correlation analysis showed a strong negative correlation between N_2O emissions and snowpack, indicating that winter N_2O emissions increased with decreasing snowpack (Table 6.4). For instance, in the two years with the greatest winter emissions (1999 and 2006), the SWE averaged 46.1 mm and 35.4 mm water, respectively. In contrast, in the two years with the lowest winter emissions (2007 and 2018), the average SWE were 206.5 mm and 199.3 mm water, respectively. There are several explanations for this result. Snowpack can impact winter emissions, given its impacts on soil temperature and moisture dynamics. Snowpack insulates the soil, thereby reducing variation in soil temperature in the latter portion of the winter season. In contrast, bare soils or soils with reduced snowpack showed comparatively lower minimum temperatures (Flerchinger 1991; Maljanen *et al.*, 2007). For instance, under snowpack, the minimum soil temperature of a boreal agricultural soil at a 0.05 m depth was –0.8°C, compared to –15°C for bare soil (Maljanen *et al.*, 2007). Maljanen et al. (2007) suggested that the insulation brought about by snowpack provides a warm environment that supports microbial activities, such that soil microbial activities are reduced but not completely inhibited. However, in their study, soils without snowpack triggered higher N_2O emissions during freezing and thawing compared to snow-covered soils. The researchers related the elevated emissions observed in bare soils to a more severe soil frost, causing the death of some microbes and the release of substrates available for microbial denitrification. We observed that the minimum soil temperatures for the two years with the highest emissions (1996 and 2006) were –5.9 and –4.1 °C compared to –2.1 and –1.3 °C observed in the two years with the lowest emissions (2007 and 2018). These findings are consistent with the ideas of (Larsen et al 2002; Wu et al. 2020; Zhe et al. 2018; Schimel and Bennett 2004) who suggest that following the death of some microbes due to severe frost, living microbes utilize the released compounds to stimulate N_2O emissions. Thus, a low snowpack results in lower minimum soil temperatures, which tend to cause greater N_2O emissions than high snowpack.

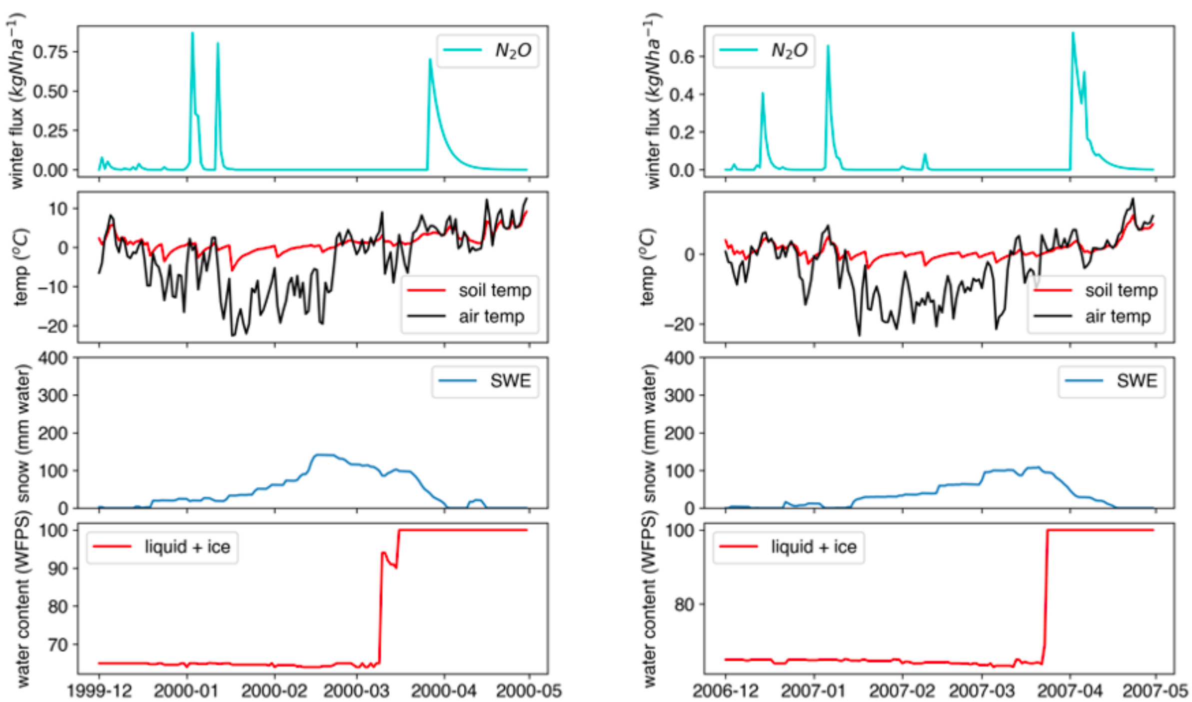

Figure 6.4: Simulated winter (a) N_2O fluxes, (b) soil and minimum air temperatures, (c) snow water equivalent (SWE), (d) soil moisture for the two historical years with greatest winter emissions (1999 and 2006)

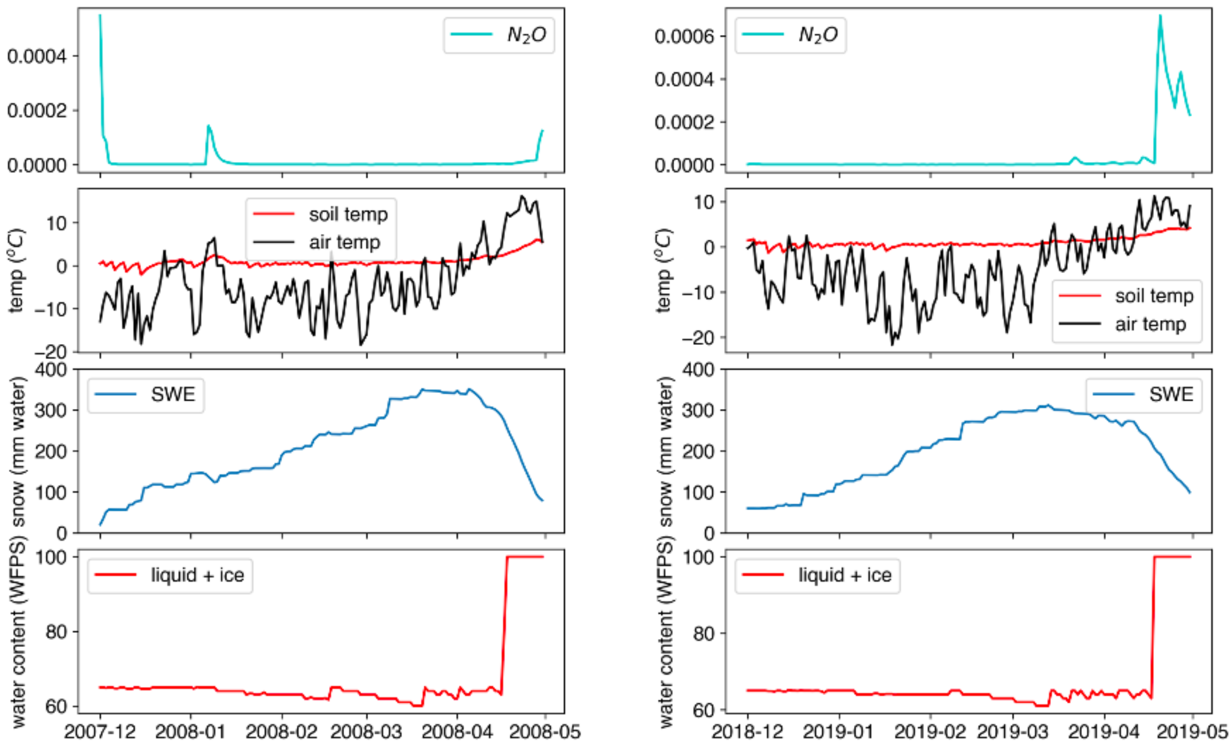

Figure 6.5: Simulated winter (a) N_2O fluxes, (b) soil and minimum air temperatures, (c) snow water equivalent (SWE), (d) soil moisture for the two historical years with lowest emissions (2007 and 2018)

Our simulation results show that the soil temperature correlated with air temperature before and after snow accumulation. However, with the snowpack atop the soil, soil temperature showed little variation (1.1 to -5.9°C) compared with air temperature that was sometimes lower than -20°C. Similar small fluctuations in soil temperature (0 to -2°C) were also reported by Chantigny et al. (2016) after snowpack developed in the field due to the insulating effect of snowpack. The results showed that the fluxes mainly occurred at a point, usually above 0°C, when the air temperature increased above the soil temperature. The spring thaw period which is often characterized by soil saturation and soil temperatures above 0 °C resulted in about 58% of the total winter fluxes. This finding agrees with those of Chantigny et al. (2016) which showed that high fluxes could occur over the winter period, but the peak flux occurred during the spring thaw following the rise in soil temperature and moisture. High soil moisture from snowmelt and a frozen subsoil that hinders downward (*vs*. lateral surface runoff) drainage could create anaerobic conditions that favor N_2O production through denitrification (Nyborg *et al*., 1997). The frozen subsoil melts as the soil thaws further, enhancing downward soil drainage, thereby creating the aerobic topsoil required for nitrification of available NH_4^+. Although winter N_2O emissions are influenced by soil temperature and moisture, the results of our correlation analysis showed that the variability in winter N_2O emissions could not be attributed to the averaged (Dec-Apr) year-to-year soil temperature and moisture variability (Table 6.4).

Table 6.4: Pearson correlation coefficient (r) of historical and future winter N_2O emissions versus December-April averages of climate variables

Variable	**Historical**		**Future**	
	r	p value	**r**	p value
Soil temperature (°C)	0.12	0.54	-0.05	0.79
SWE (mm water)	-0.82	< 0.05	-0.62	< 0.05
Soil water content (WFPS)	0.07	0.71	0.22	0.24

6.4.4 Projected winter N_2O emissions under future scenarios

Under future high emissions scenarios (2038-2067), winter N_2O emissions ranged between 0.45 and 6.2 kg N ha^{-1} representing 14.4 and 77.6% of the annual N_2O emissions (Table 6.4). Our simulations showed that winter N_2O emissions were projected to increase by 10% in the future relative to historical years. This increase in winter N_2O emissions is associated with an average winter soil temperature increase of 1°C and soil moisture increase of 8% WFPS (Fig 6.6, Tables 6.S1 & 6.S2). Another important finding was that winter precipitation and air temperature are projected to rise higher than annual values (Table 6.1). Average winter air temperature is projected to increase by 2.9°C compared to the average annual change of 2.7°C. Likewise, the average annual winter precipitation is projected to increase by 9.6% compared to the annual precipitation that remained relatively stable. Similar predictions indicate that temperature and precipitation increase in the Lake Simcoe region, Eastern Canada, would be more pronounced in winter as reported by the LSRCA (2020). The annual and winter air temperatures are projected to increase by up 5.5 °C and 6.5 °C, respectively, by 2080 (RCP 8.5 scenario), as LSRCA (2020) reported, while the annual and winter precipitation are projected to increase by 10.1% and 23%, respectively.

A major aspect of changes to climate variables under climate change is the significant decrease in SWE. The SWE mean trend magnitude for the future (2038-2067) high emissions scenario is –1.03mmyr^{-1}, implying a snow depth reduction of about 12cm over the 30 years period (assuming a 250kgm^{-3} average snowpack density). The increasing future winter N_2O emissions also correlated moderately with a decreasing SWE (r = –0.6, p <0.05) (Table 6.4). Decreases in SWE are likely driven by the greater proportion of winter precipitation occurring as rain than snow, and an increase in air temperature causing snowmelt (Grogan et al., 2020; Shi & Wang, 2015). For instance, Shi and Wang (2015) projected a reduction in snowfall due to warming and suggested that the largest SWE decreases would occur in the spring period, where SWE strongly negatively correlated with air temperature. These observations accord with our earlier observation from the historical climate trends that rising minimum temperatures resulted in an increasing trend in winter rainfall, but an inverse trend in snowfall. Our results suggest that rising winter N_2O emissions can be anticipated in the future under climate change. These findings are consistent with Griffis et al. (2017), who indicated that greater N_2O emissions occur in warmer and wetter years, including the early spring. The future scenario analysis confirmed our earlier findings based on the historical data that the future warmer and wetter climate will increase winter N_2O emissions.

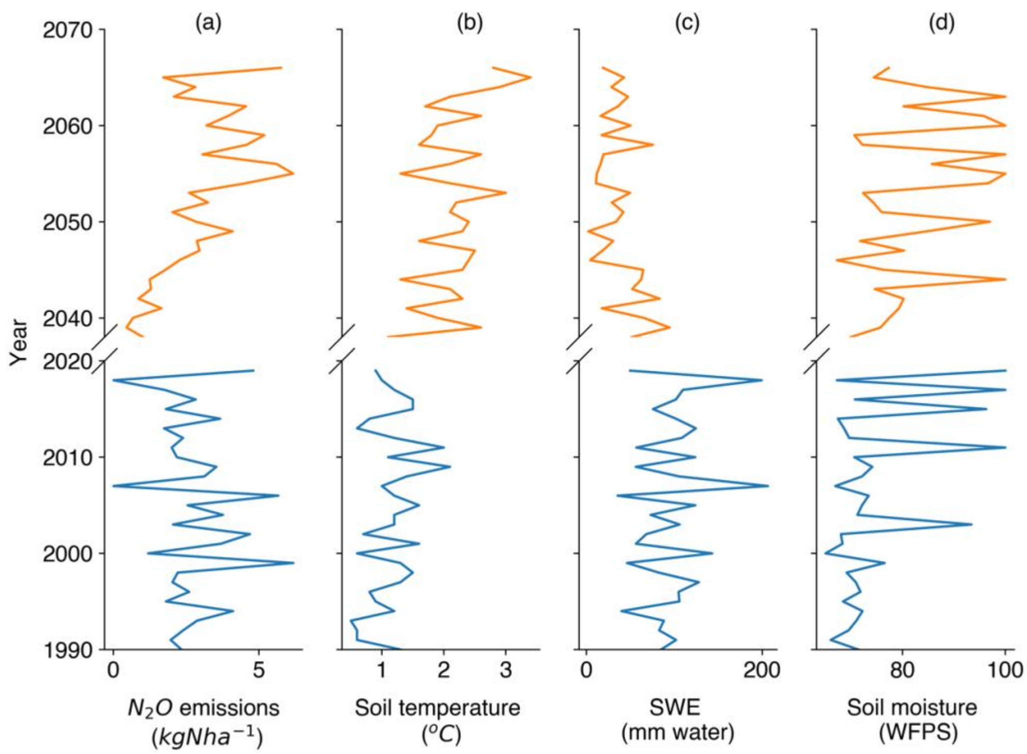

Figure 6.6: Simulated (a) winter N_2O emissions (kg N ha^{-1}) and averages (Dec – April) of winter climate variables: (b) soil temperature (^oC), (c) SWE (mm water) and (d) soil water (WFPS) for 30-year historical (blue lines) and future (orange lines) period.

6.5 Conclusions

Potential climate change impacts on winter N_2O emissions were assessed using the DNDC model, validated with data from a site in Southern Quebec. We identified major climate triggers of winter N_2O fluxes by examining climate variables' effects on historical and future N_2O emissions. Our analysis showed that warmer and wetter climates increase winter N_2O emissions. Average winter N_2O emissions for the future period (2038 - 2077) increased by 10% relative to winter N_2O emissions over the historical years (1990 - 2019). The increase in historical and future winter N_2O emissions correlated strongly with a decreasing snow water equivalent (SWE). Although our historical analysis showed that changes in snowpack have remained stable, the SWE mean trend magnitude for the future (2038-2067) high emissions scenario is -1.03mmyr^{-1}. This knowledge of snowpack impact on winter N_2O emissions would aid in adopting effective mitigation approaches. For example, planting winter cover crops, leaving crop residue in the field, and no fall tillage could help accumulate snow and reduce N_2O emissions. Other agronomic practices that can mitigate winter N_2O emissions, including nitrification inhibitors, and applying fertilizers in spring rather the fall season need to be adopted by farmers. Overall, our results show that the increasing air temperature and winter rainfall trends and decreasing snowfall anticipated in the future will result in higher winter N_2O emissions. Future studies could investigate how changes in soil freeze events correlate with winter N_2O emissions.

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6.7 Appendix: Supplementary Information

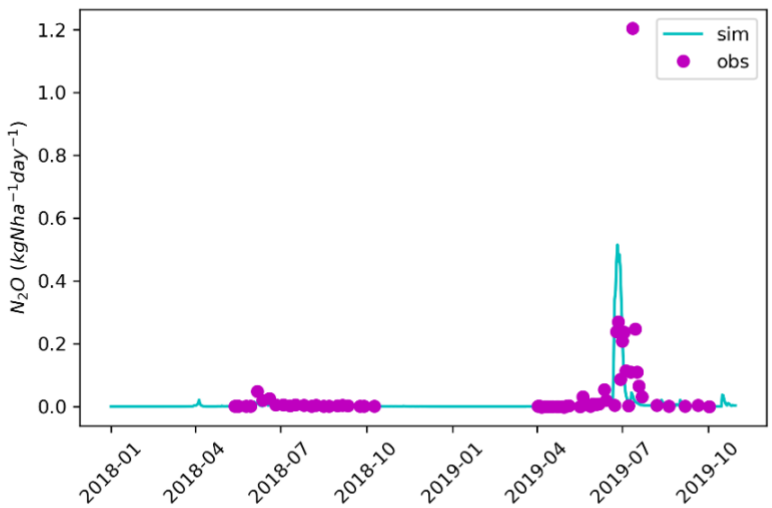

Figure 6.S1: Simulated and observed daily N_2O fluxes for 2018 and 2019

Table 6.S1: Simulated winter N_2O emissions (kg N ha^{-1}) and averages of winter climate variables: soil precipitation (mm), air temperature (°C), soil temperature (°C), snowpack (mm water) and soil moisture (WFPS) for 30-year historic period.

Year	Winter N_2O emissions (kg N ha^{-1})	Proportion of annual emission (%)	Winter climate variables (Overwinter averages)		
			Soil temperature (°C)	Snowpack (mm water)	Soil moisture (WFPS)
1990	2.34	37.1	1.3	84.9	71.4
1991	1.969	50.4	0.6	101.9	66.0
1992	2.399	38.5	0.6	82.7	69.5
1993	2.874	36.7	0.5	88.1	71.0
1994	4.115	48.8	1.2	39.9	72.2
1995	1.805	35.4	0.9	105.7	68.4
1996	2.603	44.4	0.8	104.8	71.8
1997	2.029	27.8	1.3	127.9	70.9
1998	2.21	36.2	1.5	82.4	69.1
1999	6.185	72.8	1.3	46.1	76.5
2000	1.193	16.9	0.6	143.1	65.0
2001	3.695	43.2	1.6	56.3	68.3
2002	4.695	44.7	0.7	68.1	68.0
2003	2.049	28.4	1.2	106.0	93.4
2004	3.77	51.9	1.2	73.0	71.2
2005	2.549	40.3	1.6	123.9	72.1
2006	5.671	65.7	1.2	35.4	73.4
2007	0.002	0.1	1.0	206.5	66.9
2008	3.127	35.1	1.4	105.4	72.1
2009	3.539	40.9	2.1	56.1	74.1
2010	2.182	56.9	1.1	123.8	70.6
2011	2.004	44.1	2.0	56.7	100.0
2012	2.394	56.6	1.2	108.4	69.6
2013	1.741	26.2	0.6	124.4	68.6
2014	3.67	35.8	0.8	101.8	67.4
2015	1.802	27.9	1.5	75.7	96.3
2016	2.829	49.4	1.5	101.8	70.7
2017	1.758	12.7	1.2	109.7	100.0
2018	0.005	0.4	1.0	199.3	67.2
2019	4.805	42.4	0.9	49.8	100.0
Average	2.7	38.3	1.1	96.3	74.7

Table 6.S2: Simulated winter N_2O emissions (kg N ha^{-1}) and averages of winter climate variables: soil precipitation (mm), air temperature (°C), soil temperature (°C), snowpack (mm water), and soil moisture (WFPS) for the 30-year future period.

Year	Winter N_2O emissions (kg N ha^{-1})	Proportion of annual emission (%)	Winter climate variables (Overwinter averages)		
			Soil temperature (°C)	Snowpack (mm water)	Soil moisture (WFPS)
2038	1.02	34.1	1.1	51.5	69.9
2039	0.45	18.5	2.6	94.6	75.7
2040	0.67	14.4	1.9	65.4	77.4
2041	1.65	40.2	1.4	17.4	79.3
2042	0.86	25.9	2.3	83.4	80.2
2043	1.29	40.1	2.1	52.0	74.6
2044	1.25	35.5	1.3	62.3	100.0
2045	1.79	50.2	2.3	64.7	76.3
2046	2.27	51.3	2.4	4.3	67.2
2047	2.96	47.2	2.5	18.4	80.3
2048	2.87	64.8	1.6	30.5	71.7
2049	4.10	67.6	2.3	2.0	84.7
2050	2.84	56.3	2.4	33.9	97.0
2051	2.03	41.6	2.1	42.1	75.9
2052	3.25	52.2	2.2	28.9	74.3
2053	2.60	60.2	3.0	49.7	72.3
2054	4.55	63.9	2.1	10.8	96.7
2055	6.18	77.6	1.3	11.8	100.0
2056	5.61	63.6	2.1	16.1	85.7
2057	3.06	51.5	2.6	19.4	100.0
2058	4.58	69.5	1.6	75.6	72.2
2059	5.19	71.8	1.8	17.1	70.6
2060	3.20	42.7	1.9	50.3	100.0
2061	3.95	61.4	2.6	16.2	95.8
2062	4.56	56.7	1.7	36.0	80.2
2063	2.08	40.8	2.1	47.3	100.0
2064	2.82	42.3	2.9	28.4	84.7
2065	1.71	42.4	3.4	42.7	74.4
2066	5.76	67.2	2.8	18.9	77.3
Average	2.9	50.1	2.1	37.6	82.6

Chapter VII

General Summary and Conclusion

7.1 Summary of results

In this thesis, cropland management practices, including water table management, fertilizer management, use of nitrification and urease inhibitors, no-till, and cover cropping, were investigated for their efficacy in mitigating growing season and winter nitrous oxide emissions by focusing on variations in soil types, season, or climate. The summary of the findings on the mitigative effects of various practices and their optimal soil conditions is presented in Table 7.1. Specific findings for each of the objectives are summarized below

1. Controlled drainage increases corn yields but does not necessarily cause elevated N_2O emissions. Our analysis indicated that overall, controlled drainage with sub-irrigation (CDS) had a positive influence on grain yields. The effect of CDS on grain yield depended on rainfall distribution and water table controls during the growing season. There was no clear pattern of increase in N_2O fluxes associated with CDS (vs. FD) treatment. Fluxes from CDS plots were higher in some years but lower in others, regardless of the growing season precipitation. N_2O fluxes depended more strongly on the availability of soil nitrogen and rainfall events than WTM treatments.

2. Optimal N rates are impacted by soil texture. We show that N fertilizer rates are 140 kg N ha^{-1} and 180 kg N ha^{-1} for the sandy loam and silty clay, respectively. A common generalization is that poorly drained soils such as silty clay may lose more N through denitrification than well-drained soils, given their potentially elevated WFPS. However, our study suggests that poorly drained soils could also produce lower N_2O emissions due to lower denitrification rates under suboptimal moisture conditions in the topsoil. Therefore, poorly drained soils exhibit wider range (low and high values) of yield-scaled emissions compared to well-drained soils.

3. No-till and the use of NI + UI significantly reduced over-winter N_2O emissions by 28% and 23%, respectively while the use of cover crops overall does not cause any significant change in the emissions. Our meta-analysis showed that cold-dry soil conditions, such as in

Western Canada, could prevent cover crops from being fully established in some years, hampering their effectiveness for reducing N_2O emissions in these regions.

4. Average winter N_2O emissions under future (2038 - 2067) warmer and wetter climates may increase by 10% relative to winter N_2O emissions over historical years (1990 - 2019). Agronomic practices including nutrient management (NI + UI and applying fertilizers in spring rather the fall season), no-till practice and cover crops are potential effective strategies for mitigating future winter N_2O emissions.

Table 7.1: Summary of agronomic practices, their mitigative capacity and comments on impacts of environmental factors on the efficacy of the practices.

Period	Practice	Factors assessed	Comments on impact of environmental factors on the efficacy of the practice
Growing Season	Water table management	rainfall variability	There was no clear pattern in the differences in N_2O fluxes between CDS and FD plots, regardless of rainfall amount. CDS has the potential to increase corn yields. However, a significant difference in rainfall amounts during corn vegetative stage could lead to considerable variations in yields. For instance, lower yields under CDS were observed when excessive monthly rainfall (230 mm) occurred in June 1998.
	Fertilizer Management (lowering N rates)	soil texture (clay content)	N fertilizer recommendations for our soils specifically based on the experimental result are 140 kg N ha-1 and 180 kg N ha-1 for the sandy loam and silty clay. Aggregated data show that poorly drained soils with high clay content could emit 5-fold greater yield-scaled emissions than well-drained soils. Soils with high clay content (>40%) exhibited a wider range (low and high values) of yield-scaled emissions at varying N-rates (0-235 kg N ha-1). In other words, yield scaled emissions from poorly drained fields could be high even with low N inputs.
Winter	Nitrification & Urease Inhibitors	soil texture and aridity zones	Overall, NI + UI significantly reduced over-winter N_2O emissions by 23%. Soils with medium texture significantly influenced the efficacy of NI + UI in reducing N_2O emissions more than coarse soils. Under NI+ UI, no significant difference was observed in N_2O emissions between humid and dry climates, although emissions were lower in humid areas.

No Tillage	soil texture and aridity zones	Overall, no-till significantly reduced over-winter N_2O emissions by 28% . Relatively drier areas emitted significantly lower emissions than the humid winter areas (35% less). Soil texture did not significantly impact the emissions. However, coarse soils tended to lower emissions more than medium soils under tillage practices.
Cover crops	soil texture and aridity zones	CC showed an overall reduction potential of 18%; however, this effect was not significant. Under the CC practice, N_2O emissions were reduced in humid climates but increased in drier climates

7.2 Contribution of knowledge

The original contributions of this thesis include:

i. This thesis demonstrated that controlled drainage with sub-irrigation has potential yield benefits compared to regular tile drains and does not lead to increased N_2O emissions as suggested by earlier studies.
ii. This thesis developed a regression model to show the association of clay content and N-rates in predicting growing season yield scaled N_2O emissions. The knowledge that sites with higher clay (>40%) exhibit a wider range of yield-scaled emissions indicates that such soils are good targets for implementing improved N management strategies to mitigate N_2O emissions.
iii. This thesis demonstrated for the first time that no-till and NI + UI practices effectively reduced over-winter emissions in dry and humid winter regions on all soil types. For cover crop practice however, N_2O emissions were reduced in humid climates but increased in drier climates. This is a significant contribution to knowledge given that earlier meta-analysis studies have focused majorly on growing season N_2O losses. Also, assessing the efficacy of these mitigation practices across soil types and aridity zones is useful for identifying appropriate agronomic management practices that could mitigate winter and early spring N_2O emissions.
iv. This thesis estimated a 10% rise in future (2038 - 2077) winter emissions relative to historical years (1990 - 2019), mainly associated with a decrease in snow water equivalent. The recommendations in this study including adopting agronomic practices such as planting winter cover crops, leaving crop residue in the field, and no fall tillage to help accumulate snow and subsequently reduce N_2O emissions.

7.3 Recommendations for future research

Recommendations for future research as identified in this thesis are listed as follows:

i. High ammonia volatilization that is expected with urea fertilizers and in soils with low clay content (e.g., the sandy loam) may have resulted in significant N loss that was not measured in our study. Future studies could adopt a more holistic approach and measure other indirect forms of N losses.
ii. The meta-analysis study highlighted a lack of consensus on the role of denitrifiers in over-winter N_2O fluxes. The incongruity stems from the detection limits of these sequencing methods used in the previous studies. Also, some studies relied on gene copy numbers (which do not directly translate to expression) as a parameter for denitrifying activity. Focus can now be placed on utilizing an improved PCR primer set developed for better detection of denitrifying genes and metatranscriptomics to determine denitrifying activity.
iii. The adoption of nitrification and urease (NI + UI) inhibitors by farmers to reduce over-winter emissions has been proposed in this work. Future studies could also account for volatilization and N-leaching in addition to N_2O measurements to capture the potential trade-offs of NI + UI application, including an increase in NH_3 emissions and possible NI contamination.
iv. The climate change impacts on winter N_2O emissions were simulated with data from the North American Regional Climate Change Assessment Program, which uses the SRES A2 emissions scenario capped at 2070. Further work could use more recent scenarios with robust projections (e.g., AR6 IPCC shared socioeconomic pathways (SSP)) to investigate this query. There could be more freezing with lower snow cover. Future studies could also investigate if soil freeze events explain the higher winter N_2O emissions simulated under future climate scenarios.