Effects of throughfall reduction and snow removal on soil physical and biogeochemical properties in aspen forests of northern Minnesota, USA

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Dedicated to my best friend of eleven years.

Abstract

Climate change is projected to alter precipitation patterns across northern latitudes, with decreased snow accumulation and summer rainfall predicted. These changes may alter soil physical and biogeochemical properties, which would have implications for the operability and productivity of forest soils. Reductions in summer and winter precipitation were simulated using a paired-plot design with throughfall reduction and snow removal across four drainage classes at each of three locations in northern Minnesota, USA. Soil temperature and moisture were measured every fifteen minutes to a depth of 60 cm, and soil frost depth (winter) and soil strength (summer) were monitored for two years. Soil respiration and extractable nitrogen were measured during two growing seasons, and a laboratory incubation was performed to test the response of carbon and nitrogen fluxes under controlled conditions. Soil temperature and moisture increased from well-drained to poorly-drained soils during the winter and growing season, respectively. Snow removal caused large declines in soil temperature and significantly deeper penetration of frost that varied by drainage class, and there were strong relationships between frost depth and freezing degree days. Throughfall reduction had no effect on soil strength, soil respiration, or extractable nitrogen concentrations. Drainage class was a significant, although limited, indicator of soil strength, soil respiration, and extractable nitrogen concentrations. The laboratory incubation confirmed the lack of treatment effect on soil carbon and nitrogen fluxes, and instead showed that drainage class and soil moisture controlled these fluxes. These findings show that the dominant response of forest soils to reduced seasonal precipitation will occur during the

winter with decreased soil temperatures and increased frost depth across drainage classes, which has implications for seasonal timber harvesting in northern latitudes under a changing climate.

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List of Abbreviations and Chemical Formulas

AIC: Akaike information criterion

ANOVA: Analysis of variance

C: Carbon, element

CH_4: Methane gas

CO_2: Carbon dioxide gas

FDD: Freezing degree days

KCl: Potassium chloride

MWD: Moderately-well drained

N: Nitrogen, element

N_2O: Nitrous oxide gas

NH_4^+: Ammonium cation

NO_3^-/NO_2^-: Sum of nitrate and nitrite anions

PD: Poorly drained

SPD: Somewhat-poorly drained

SWC: Soil water content

TN: Total nitrogen

WD: Well-drained

Introduction

Forest soils have a large impact on a host of ecosystem processes, such as the cycling of nutrients and water, and serve as a foundation for sustainable forest management. Changing climatic conditions may alter the operability and productivity of forest soils, in particular via fluctuations in the timing and amount of precipitation (rain and snow). In northern latitudes, summer precipitation is predicted to decrease overall (with an increase in extreme precipitation events), and winter snowpack is also expected to decrease (Handler et al., 2014). These changes may increase the likelihood of impacts to soils during forest operations. For example, soil compaction can negatively impact forest health by reducing macropore space and increasing bulk density, which can decrease water availability, gas exchange, and root growth (Grigal, 2000). Long-term effects of soil compaction have major implications for stand growth, since compaction limits the supply of oxygen and water to roots and microbes and reduces tree regeneration (Cambi et al., 2015). Avoiding compaction is crucial in maintaining long-term stand growth since forest soils are unlikely to recover from compaction in the short-term (Greacen & Sands, 1980; von Wilpert & Schäffer, 2006). Since soil moisture influences soil strength, changing climatic conditions may increase the likelihood of compaction in forest soils.

Soil operability is defined as the ability of a soil to withstand the physical stresses from forest operations with limited impacts on soil properties (NCASI, 2004). Forest management relies on soil operability to ensure that negative impacts to the soil during harvesting are avoided. In practice, a large portion of timber is harvested in Minnesota in the winter (Blinn et al. 2015) to reduce the risk of soil compaction and maintain operability. Frozen soils can withstand higher shear stresses (*e.g.*, sustain heavy loads like harvesting equipment) compared to non-frozen soils of the same texture (Shoop, 1995). Warming winter temperatures and reductions in winter snowpack may affect soil frost and may alter the operability of forest soils during the winter (Handler et al., 2014). Similarly, the feasibility of harvesting during the summer on non-frozen soils will also likely be impacted by the timing and amount of annual precipitation (Uusitalo et al., 2019). High bulk density soils with low water contents typically have higher soil strength, and thus are less prone to soil compaction (Uusitalo et al., 2019; McNabb et al., 2001). As a result, summer harvesting in Minnesota typically occurs on drier, coarse- or medium- textured soils, which have the lowest compaction risk due to the dominance of sand and silt-sized particles (Greacen & Sands, 1980). Thus, changes in summer precipitation patterns may also alter the moisture dynamics of these soils, subsequently affecting the operability of forest stands in the summer.

Changes in long term precipitation patterns may also influence the productivity of forest soils through impacts on soil carbon and nitrogen turnover rates (Borken et al., 2006a & 2006b; Fitzhugh et al., 2001). Soil moisture and temperature have been widely shown to control microbial processes in soil, such as soil respiration and nitrogen mineralization. (Homyak et al, 2017; Öquist & Laudon, 2008; Borken et al. 2006a & 2006b). The study of the effects of moisture and frost dynamics on biogeochemical processes within forest soils provides applicable insight on the long-term productivity of forest ecosystems under a changing climate. Additionally, understanding the response of forest carbon dynamics to changes in precipitation may improve mechanistic models of carbon pools and fluxes in these ecosystems under different climate scenarios.

Soil drainage class may be an important factor when considering fluctuations in soil moisture and frost with climate change. In the field, soil texture and landscape position create differences in soil drainage (*i.e*., relative soil wetness), which is reflected in the depth to redoximorphic features (*i.e.*, mottling, gleying) used to define drainage classes (Venemen et al, 1998). Soil aeration, relative moisture supply, and potential rooting depth are all influenced by drainage class (Briggs & Lemin 1994). Therefore, the effects of reduced rainfall and snowpack will likely vary by drainage class in forest soils due to the inherent differences in porosity and moisture-holding capacity. Drainage class is also a useful metric when measuring soil strength since drainage class is easily measured in the field, conceptually understandable, and readily mapped.

There has been little research that pairs snowfall removal and throughfall reduction treatments to study the effects of altered precipitation inputs on soil physical and biogeochemical properties. Snow removal treatments simulate a lack of snowpack, as predicted under warmer winter conditions (Friesen et al., 2021; Monson et al., 2006; Muhr et al., 2009; Öquist & Laudon, 2008). Throughfall reduction experiments simulate reduced precipitation in forest ecosystems via shelters that intercept some of the precipitation that occurs as throughfall (Borken et al., 2006a; Homyak et al., 2017). No such studies have been conducted in Minnesota upland forests, particularly not in quaking aspen (*Populus tremuloides*, Michx.) stands, which are of high economic importance in forests within the northern parts of the state and compose 53% of timber harvests in Minnesota (Domke et al., 2008; Handler et al., 2014).

To address this knowledge gap, I utilized a paired-plot design that implemented both summer throughfall reduction and winter snow removal treatments across a soil drainage class gradient to study the influence of soil moisture and frost dynamics on soil physical and biogeochemical properties. Soil strength and carbon and nitrogen cycling are critical measures of forest operability and productivity in the long term. My research builds on past studies by pairing both throughfall reduction and snowfall removal in the study of forest soil physical and biogeochemical properties under a changing climate. The first chapter addresses how reduction of rainfall and snow removal influence soil physical characteristics, specifically soil strength during summer and frost depth during winter. The second chapter then addresses how throughfall reduction and snow removal affect carbon and nitrogen fluxes and pools in forest soils. The research presented in this thesis will be extremely relevant and applicable to sustainable forest management in northern latitudes under a changing climate.

Chapter 1

The effects of throughfall reduction and snow removal on soil physical properties across a drainage gradient in aspen forests of northern Minnesota, USA

Abstract

Climate change is projected to alter precipitation patterns across northern latitudes, with decreased snow accumulation and summer rainfall predicted. These changes may alter soil physical properties such as soil strength, which would have implications for the feasibility of forest management activities under a changing climate. Reductions in summer and winter precipitation were simulated using a paired-plot design with throughfall reduction and snow removal as treatments across four drainage classes (well, moderately well, somewhat poor, and poorly drained) at each of three locations in northern Minnesota, USA. Soil temperature and water content were measured every fifteen minutes to a depth of 60 cm, and soil frost depth (winter) and soil strength (summer) were monitored for two years. As expected, soil temperature and water content increased from well-drained to poorly-drained soils. Snow removal caused large declines in soil temperature and significantly deeper penetration of frost that varied by drainage class, where frost depth decreased with decreasing (wetter) drainage. There was a strong positive relationship between cumulative freezing degree days and frost depth when snow was removed, and a positive yet weaker relationship in the control plots (Treatment: r^2 = 0.74 in 2019/20, r^2 = 0.82 in 2020/21; Control: r^2 = 0.02 in 2019/20, r^2 = 0.38 in 2020/21). Throughfall reduction had limited effects on soil water content and inconsistent

effects on soil strength. Based on these findings, changes in soil physical properties with altered precipitation are likely to manifest primarily in winter, but the direction and magnitude of any effect is uncertain. During winters with limited snow cover, relationships between freezing degree days and soil frost may be useful to predict when sufficient frost is present for forest management activities to occur without detrimental effects to soil functions. Drainage class may also be used to predict the depth and development of frost. Taken together, our findings may be useful for managers to predict optimal harvesting conditions under a changing climate.

Introduction

Soil strength, the ability of a soil to resist shear stresses, determines the operability of the soil in the context of forest management (Grigal, 2000). Soil operability is defined as the ability of a soil to withstand the physical stresses from forest operations (e.g., harvest equipment) with limited impacts on soil properties (NCASI, 2004). For example, soil compaction can negatively impact forest health by reducing macropore space and increasing bulk density, which can decrease water availability, gas exchange, and root growth (Grigal, 2000). Long-term effects of soil compaction have major implications for stand growth, since compaction limits the supply of oxygen and water to roots and microbes and reduces tree regeneration (Cambi et al., 2015). Thus, avoiding compaction is crucial in maintaining long-term productivity since forest soils are unlikely to recover from compaction in the short-term (Greacen & Sands, 1980; von Wilpert & Schäffer, 2006; Powers et al., 1990).

Current climate change models for northern latitudes predict an overall decrease in summer precipitation but with more extreme precipitation events (Handler et al., 2014). More winter precipitation will occur as freezing rain rather than snow due to warmer winter temperatures, resulting in an overall decrease in snowpack depth (Handler et al., 2014). Since soil strength is influenced by soil moisture and frost depth, future changes in precipitation will likely affect forest soil operability during the summer and winter harvesting seasons, which would have major economic and ecological implications (Uusitalo et al., 2019; McNabb et al., 2001; Shoop, 1995; Horn et al., 2007; Kok & McCool, 1990).

The feasibility of harvesting on soils during the summer will likely be impacted by the timing and amount of annual precipitation (Uusitalo et al., 2019). High bulk densities and low water contents are characteristics of high strength soils, which have a low compaction risk (Uusitalo et al., 2019; McNabb et al., 2001). As a result, summer harvesting in the United States typically occurs on drier, coarse- or medium- textured soils, which have the lowest compaction risk due to the dominance of sand and silt-sized particles (Greacen & Sands, 1980). Soil strength has also been shown to be strongly controlled by soil water content (SWC), clay content, and bulk density (Uusitalo et al., 2019). Given this, altered soil moisture dynamics arising from changes in summer precipitation patterns may affect summer operability of forest stands. For example, a study by McNabb et al. (2001), which investigated the effects of skidding and SWC on compaction, found that decreases in SWC were directly related to increases in effective shear strength. Given this, soil water content is a determining factor in forest operability due to its strong correlation to soil strength, soil type, and harvesting intensity.

A large portion of timber harvesting in northern latitudes is conducted in the winter to reduce the risk of soil compaction and maintain operability (Blinn et al. 2015). Frozen soils can withstand higher shear stresses (*e.g.,* heavy harvesting equipment) compared to non-frozen soils of the same texture (Kok & McCool, 1990; Shoop, 1995). However, changes in winter precipitation and frost dynamics may also affect the compaction risk of forest soils due to the role of snowpack in frost development. Snowpack acts as an insulative layer over the soil surface due to its high albedo and low thermal conductivity, so frost does not develop under a thick snowpack to the same extent as a thin snowpack (Zhang, 2005). Stone (2002) recommends at least 15 cm of soil frost to support heavy harvesting equipment with limited impacts to the soil. Changes in winter precipitation may decrease the period between soil freeze and thaw when operators may harvest forest stands with minimal soil disturbance. Thus, the in-situ study of frost dynamics in forest soils under a changing precipitation regime can help forest managers and operators understand how soil operability will change in the future

Drainage class, which can be easily measured in the field and mapped, may be an important indicator of soil strength. Soil water content, texture, and porosity are all related to drainage class, meaning that drainage class may be useful when categorizing a site's compaction risk (Briggs & Lemin, 1994; McNabb et al., 2001; Uusitalo et al., 2019; Veneman et al., 1998). For example, soil water content increases as drainage worsens due to change in landscape position and increase in clay content (Veneman et al., 1998). Soil temperature also tends to be higher during the winter in poorly-drained soils due to the low thermal diffusivity of soils with high soil water contents (Arkhangelskaya & Lukyashchenko, 2018). As a result, soil drainage class can be used to infer relative soil water content and temperature of site during winter, which will influence soil strength and frost development.

Paired-plot experiments using throughfall reduction or snow removal have been used to investigate the effects of altered precipitation in forest ecosystems. Yang et al. (2019) constructed throughfall reduction shelters (20 x 20m) below a tropical forest canopy and measured the size, distribution, and stability of soil aggregates during a four-year study period. In that case, the stability and size of soil aggregates was significantly reduced by the throughfall reduction treatment (Yang et al., 2019). Snow removal experiments at the Hubbard Brook Experimental Forest (HBEF) in New Hampshire, USA found that the removal strongly increased frost depth and frost persistence in the treatment plots, even during a "mild" winter with relatively low amounts of snowfall (Hardy et al., 2001). That study serves as an example of a paired-plot experiment to study the effects of reduced snow cover on frost depth in northern climates.

I sought a better understanding of soil moisture and frost responses in forest soils to changing climatic conditions, specifically seasonal precipitation. To do so, I investigated the influence of throughfall reduction and snow removal on soil strength, frost depth, moisture, and temperature via a paired-plot design. I aimed to quantify the effect of the treatments and drainage class on soil water content and soil strength during the growing season, and soil temperature and frost during the winter. The findings of this study may be insightful for forest managers and operators when planning timber harvests with the goal to maintain operability while limiting soil compaction.

Methods

Study area

I studied three sites in northern Minnesota in Aitkin, Itasca, and St. Louis counties in the Laurentian Mixed Forest Province (LMFP) of northeastern MN (Figure 1.1). The Aitkin and Itasca sites were located within state-managed forests (Solana and George Washington State Forests, respectively), and the St. Louis County site was located on county-owned land. The study region is located on the ancestral, traditional, and contemporary lands of the Anishinaabe, Oceti Sakowin, and Mdewakanton. These study sites specifically reside on land ceded in the Chippewa Treaty of 1839 and the Chippewa Treaty of 1854.

Soils in this region are typically medium to coarse textured with glacial parent material from the last glacial retreat 12,000 years ago (Handler et al., 2014). Quaking aspen is a large part of the LMFP, composing 30% of Minnesota's forest land and is most concentrated in the LMFP (Handler et al., 2014).

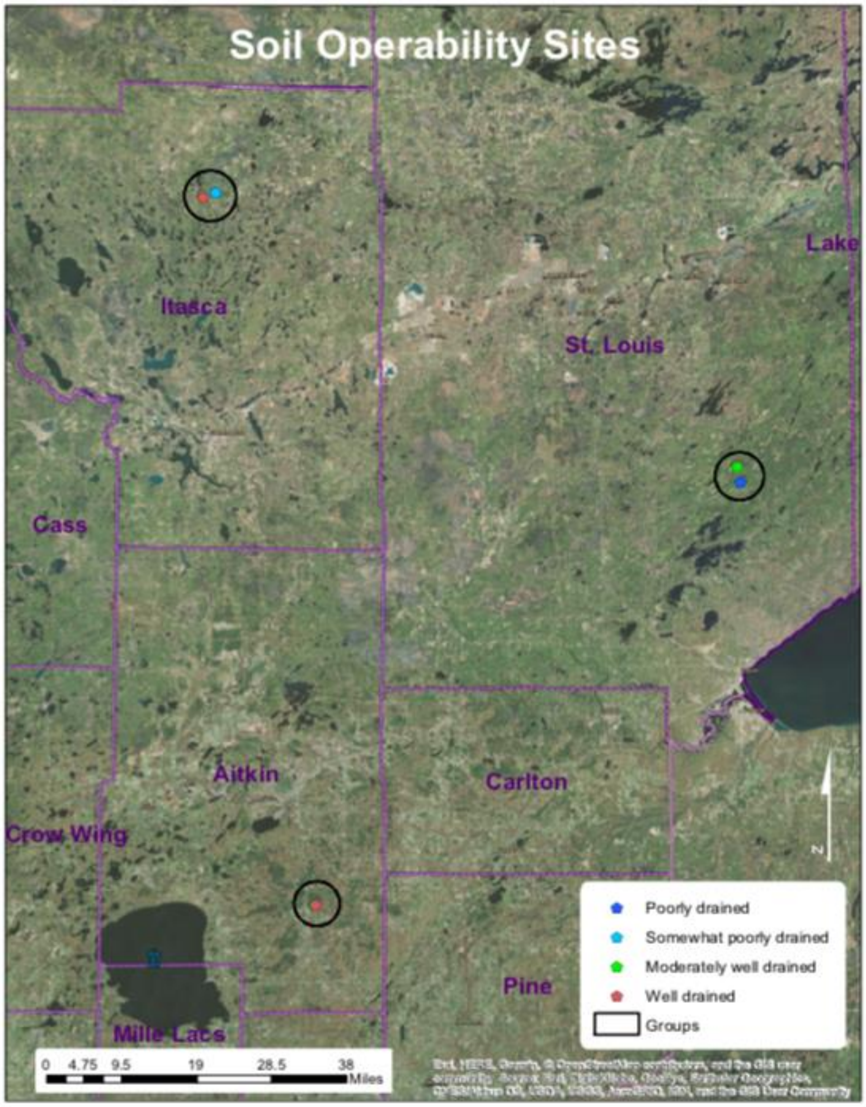

Figure 1.1: Map of study site locations. Black circles indicate site groupings. Each site contains paired plots across the four drainage classes.

All sites were dominated by upland quaking aspen in the forest canopy with beaked hazel (*Corylus cornuta*), willow (*Salix* spp.), or speckled alder (*Alnus incana*) in the understory. Mean summer (June – August) temperatures for this region is 18˚C and winter temperatures average -12˚C (Handler et al., 2014). Average precipitation during the summer is 305 mm, and average accumulated snowfall ranges from 1,016 mm to 1,778 mm (Handler et al, 2014).

Site characteristics

Mature quaking aspen (40-60 years) was the dominant tree species at all sites. Soils at each site were predominantly loams occurring on relatively flat topography (less than 10% slope). Plot locations with the target drainage classes (well-drained through poorly drained) were identified based on depth to redoximorphic features (Table 1.1).

Table 1.1: Description of soil series and textures for each drainage class within the three sites (county) determined from soil survey information. Soil survey information from National Cooperative Soil Survey (NRCS)**.**

<table>
<tr><th>Site (County)</th><th>Drainage class</th><th>Soil unit/taxonomy</th><th>Soil texture</th><th>Depth to redoximorphic features</th></tr>
<tr><td rowspan="4">Aitkin</td><td>Well</td><td rowspan="4">738B: Milaca-Millward complex, 2-8% slopes

Coarse-loamy, mixed, superactive, frigid Oxyaquic Glossudalfs

Coarse-loamy, mixed, superactive, frigid Typic Hapludalfs</td><td rowspan="4">Fine sandy loam</td><td>> 102 cm</td></tr>
<tr><td>Moderately well</td><td>51 – 102 cm</td></tr>
<tr><td>Somewhat poor</td><td>25 – 51 cm</td></tr>
<tr><td>Poor</td><td>0 – 25cm</td></tr>
<tr><td rowspan="3">Itasca</td><td>Well</td><td rowspan="2">803B: Warba-Menahga complex, 1-8% slopes

Fine-loamy, mixed, superactive, frigid Haplic Glossudalfs

Mixed, frigid Typic Udipsamments</td><td rowspan="2">Fine sandy loam</td><td>> 102 cm</td></tr>
<tr><td>Moderately well</td><td>51 – 102 cm</td></tr>
<tr><td>Somewhat poor</td><td>621: Morph very fine sandy loam

Fine-loamy, mixed, superactive, frigid Typic Glossaqualfs</td><td>Very fine sandy loam</td><td>25 – 51 cm</td></tr>
</table>

	Poor	167B: Baudette silt loam, 0-5% slopes Fine-silty, mixed, superactive frigid Oxyaquic Hapludalfs	Silt loam	0 – 25cm
St. Louis	Well	F122B: Aldenlake-Pequaywan complex, pitted, 0-8% slopes Coarse-loamy, isotic, frigid Dystric Eutrudepts Coarse-loamy, isotic, frigid Aquic Dystric Eutrudepts	Sandy loam	> 102 cm
	Moderately well			51 – 102 cm
	Somewhat poor	F103B: Brimson stony fine sandy loam, 2-5% slopes, very stony Coarse-loamy, isotic, frigid Aquic Dystric Eutrudepts	Stony fine sandy loam	25 – 51 cm
	Poor			0 – 25cm

Snow removal treatment

Manual snow removal was conducted on treatment plots during the winter (November – May) according to the method presented in Friesen et al. (2021). To allow for snow removal without impacting the soil surface, gray aluminum window screening (Phifer Incorporated, Tuscaloosa AL) was placed over the entire treatment plot area during the winter prior to the first snowfall (Figure 1.2). All woody stems were trimmed prior to placement of screening over the treatment area, in addition to all woody stems in the control plot. The color of the screening mimicked the albedo of the forest floor. The screening also allowed for gas exchange without completely shading the soil surface. Screening was divided into three panels and anchored on the side of the plot opposite to the control plot to allow for ease in removing snow via lifting the panels. Snow was cleared manually by a shovel or leaf-blower, or by lifting the panels and moving the snow off the far edge of the plot away from the control plot. Snow was always cleared and deposited away from the control plot to limit any possible disturbance of the experimental control. Snow was cleared after every storm of 2-inches or more, or at least weekly.

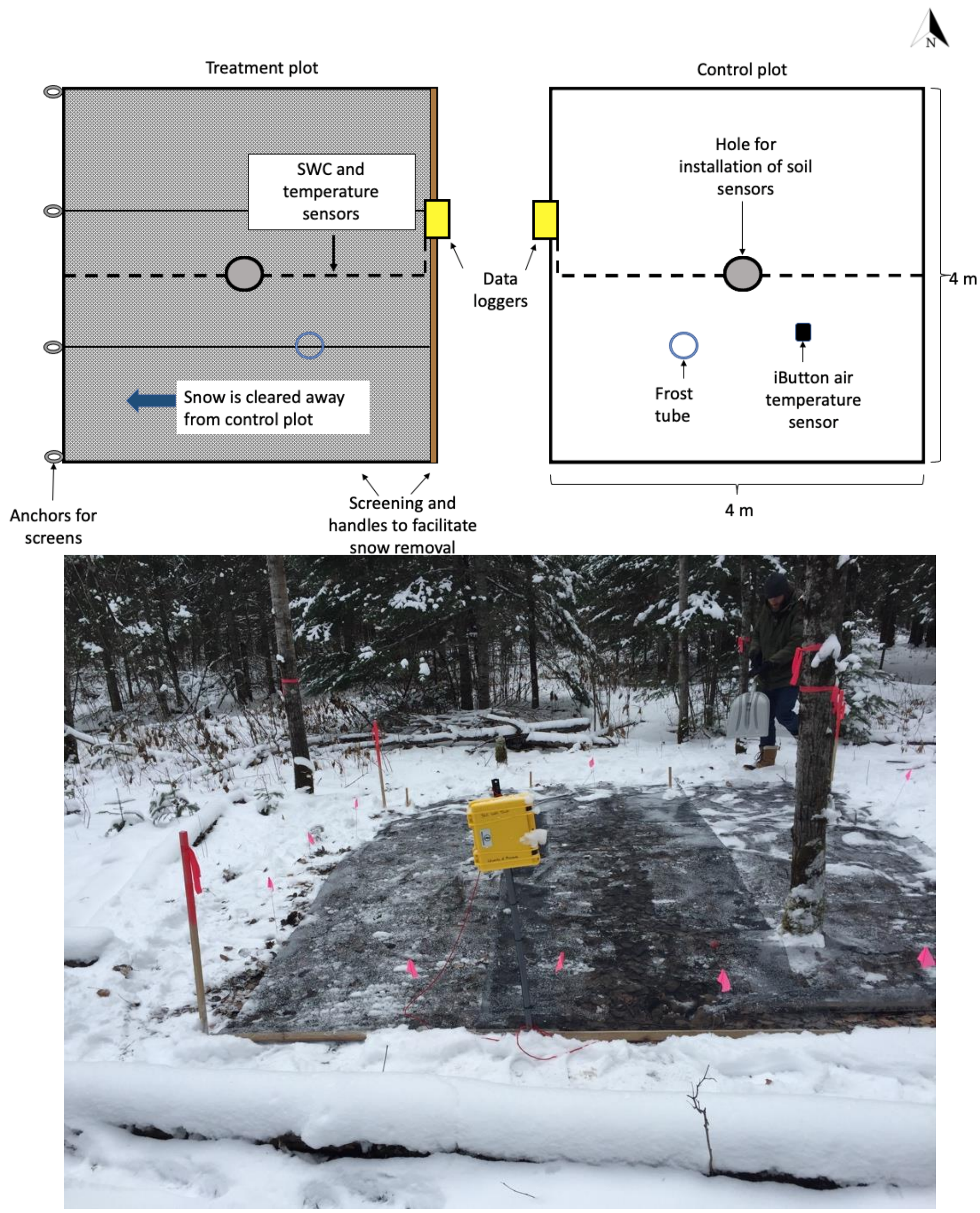

Figure 1.2: Paired-plot design schematic and field photo with snow removal treatment during the winter. (Photo credit: Alan Toczydlowski, University of Minnesota)

Throughfall reduction treatment

Throughfall reduction shelters were installed during the growing season to simulate a 50% reduction in throughfall (Figure 1.3). The shelters were constructed of 6-inch wide Greca clear Amerilux polycarbonate roof panels (Amerilux International LLC, De Pere WI). Roof panels were placed on shelters after soil frost was completely thawed each spring. Each shelter contained four panels, with a total area of 8 x 8 feet (2.4 x 2.4 m) and was framed with 1.5 in (3.81 cm) PVC pipe and connector fittings. The shelter framing was constructed from 2 x 4 in (5.08 x 10.16 cm) and 4 x 4 in (10.16 x 10.16 cm) dimensional lumber. The shelters were guttered with 4-inch (10.16 cm) wide, U-shaped white vinyl gutters that extended 40 centimeters past the 4 x 4-meter plot boundary and away from the control plot to drain guttered water away from the plots. The peak height of the shelter was 7-feet (2.13 m) at the ridgeline, and 2-feet (0.61 m) at the gutter edge. The ridgeline of the A-frame shelter ran along a north-south transect so that panels were situated on an east-west transect to avoid greenhouse effects created by a south-facing panel. Control plots were left as an experimental reference and did not receive any precipitation reduction treatment.

To assess treatment efficacy, the volume of throughfall in plots was measured biweekly during the growing season of 2021 using 20.3 cm funnels attached to glass jars that were placed in each quadrant of moderately drained plots at each site (n = 4 collectors per plot and site). The biweekly average throughfall volume for control plots was 649 mL (± 54.4 mL) and was 306 mL (± 108 mL) for treatment (exclusion) plots.

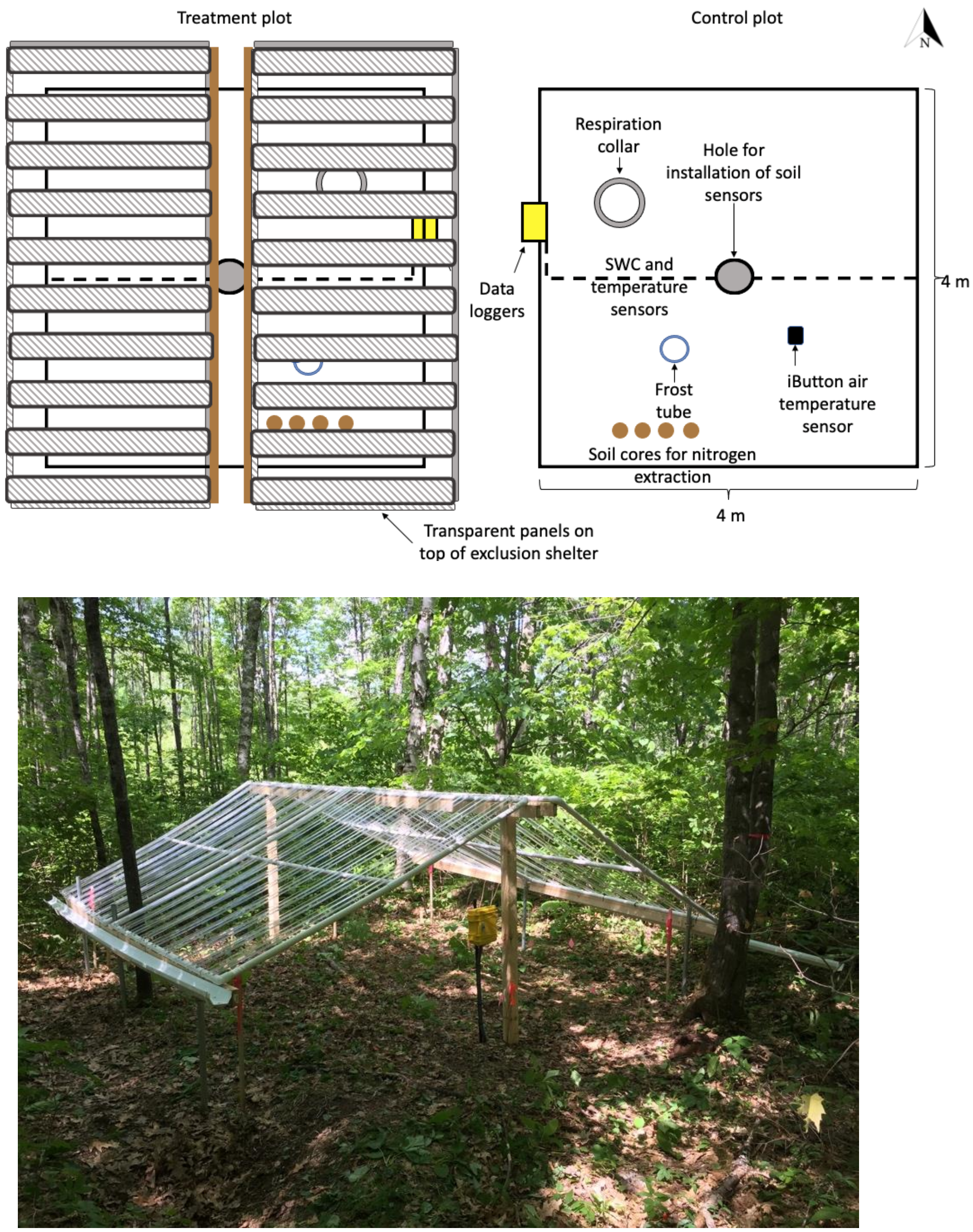

Figure 1.3: Paired-plot design with precipitation reduction treatment during the growing season. All plots and transparent roof panels were oriented on an east-west transect, with the ridgeline of the shelter running north-south. Precipitation reduction shelters were designed to exclude 50% of throughfall. Plots that received treatment were randomized in each pair. The bottom photo shows the throughfall reduction shelter on a treatment plot during the growing season. (Photo credit: Alan Toczydlowski)

Soil water content, soil temperature, and air temperature measurements

Soil temperature and moisture were measured every 15 minutes at depths of 10, 20, 30, 40, and 60 cm via Decagon 5TM sensors (± 0.1˚C, ± 0.08% SWC; METER

Group, Pullman WA). Sensors were installed in a cluster at the center of each plot (Figure 1.2, 1.3) and connected to EM50 data loggers (METER Group, Pullman WA). Air temperature was recorded at control plots every 90 minutes (starting at 00:00) by Thermochron iButton sensors (± 0.5˚C; Maxim Integrated Products, Inc., Sunnyvale CA) enclosed in a PVC solar shield.

Soil frost measurements

Frost depth was measured weekly for three years during the winters of 2019/20, 2020/21, and 2021/22 in all plots. Frost tubes were constructed by Northern Frost Tubes (Brian Hahn, Oconomowoc WI). Frost tubes were installed to a depth of 1.5 meters in the soil profile and were filled with a solution of water and color-changing indicator dye. The solution turned clear when frozen, indicating the depth of frost. Frost depth was measured to the nearest 2.5 cm in all plots.

Soil strength measurements

Soil strength measurements were collected biweekly between June and September of 2020, and monthly between May and September of 2021. Soil strength was measured via a dual-mass dynamic cone penetrometer (Humboldt Mfg. Co., Elgin IL). Strength measurements followed the protocol of the Minnesota Department of Transportation (MNDOT, n.d.). At least two full penetrometer runs of blows to a depth of 45cm were conducted per plot in two random quadrants.

Data analysis

Analyses focused on soil water content during the growing season (May – September/October 2019 – 2021), and soil temperature during the winter (October/November – April/May 2018 – 2021). Soil water content and temperature, as well as air temperature were first averaged by day and then by week using the "lubridate" package in R (Grolemund & Wickham, 2011). Frost depths were grouped into time periods (week) based on measurement dates from each site, since observations occurred at different days across sites.

Repeated measures, linear mixed effect models were used to evaluate the influence of drainage class, treatment, and time on soil strength, frost depth, moisture, and temperature. Site (block) was included as a random effect in all models, and each year of measurement was run independently. A mixed effects model with year and drainage class modeled as fixed effects, and site as a random effect, was used to analyze differences in snow depth among years and drainage classes. The R package "nlme" (Pinheiro et al., 2021) was used to run the models. Autocorrelation matrices (corAR1 function) were included in models to account for temporal correlation in the data (Pinheiro et al., 2021). Least square means analysis with the Tukey p-value adjustment was performed when significant effects were found by using the "lsmeans" (previously "emmeans") R package (Lenth, 2021).

Plots of standardized residuals and quantile-quantile plots were used to validate the assumptions of normality, linearity, constant variance, and independence. Soil strength was transformed using a natural logarithm to correct for non-normality. Frost depth was transformed as the logarithm of frost depth + 1 to avoid taking the logarithm of zero in 2020 to correct for non-normality. Quantile-quantile plots and plots of standardized residuals were used to identify the best transformation of the dependent variable. The Akaike Information Criterion (AIC) was used to determine the inclusion of interactions and random effects in the model, with the lower AIC value indicating the better model (Arnold, 2010). All least square means and confidence intervals were presented in original, non-transformed units for interpretation in figures.

Linear regression was used to determine correlation between frost depth and cumulative freezing degree days (FDD) for control and treatment plots. Cumulative degree days were calculated as the cumulative number of days where the mean daily air temperature for each plot was below freezing (0˚C). Regression lines were compared to assess the effect of drainage class on the relationship between cumulative FDD and frost depth in control and treatment plots. Analysis of covariance (ANCOVA) was used to test alternative models (variable intercepts and slopes between drainage classes, variable intercepts between drainage classes, or no difference in intercepts or slopes), and AIC was used to identify the best model, with a low AIC value indicating the better model.

Regression line comparisons were used to assess differences between drainage classes in the relationship between cumulative FDD and frost depth for control and treatment plots in the winters of 2019/20 and 2020/21. The variable intercept model was determined to be the best model compared to the global model based on the lower AIC value during both years. Pairwise comparisons of the least square means were then conducted between the drainage classes for control and treatment plots during both winters.

Linear regressions were used to determine relationships between soil strength (bearing capacity) and soil water content (%). Distance between blows (DPB) was used to calculate the California Bearing Ratio (Equation 1.1; Black, 1962) and bearing capacity (Equation 1.2) in pounds per square inch (psi). Runs for each plot were averaged to create a plot-level soil strength estimate.

Equation 1.1

$$CBR\ (\%) = \frac{292}{DPB^{1.12}}$$

Equation 1.2

$$BC\ (psi) = 4.5915 \times CBR^{0.6105}$$

Results

Effects of snow removal

There were significant differences in air temperature and ambient snow depth between 2019/20 and 2020/21 (Table 1.2). Mean air temperature was significantly higher and snow depth significantly lower in 2020/21 compared to 2019/20.

Table 1.2: Least square mean weekly air temperature and snow depth during the winters of 2019/20 and 2020/21. Superscript letters indicate significant differences in air temperature or snow depth between the years, with "a" being significantly different from "b". Standard error is indicated by SE.

2019/20		**2020/21**	
Air temperature (˚C)			
Mean	-5.4^{a}	Mean	-2.9^{b}
SE	0.45	SE	0.42
Maximum	5.7	Maximum	11.1
Minimum	-15.3	Minimum	-27.6
Snow depth (cm)			
Mean	37^{a}	Mean	13^{b}
SE	2.3	SE	2.2

There was a significant three-way interaction between drainage, treatment, and week for soil temperature in all three years (Table 1.3; Figure 1.4). Soil temperature increased from well-drained to poorly-drained, likely a result of the higher water content of the poorly-drained plots (Figure 1.4, A.1, A.2). Additionally, the well-drained plots experienced more rapid freeze and thaw durations compared to the poorly-drained plots, which showed slower warming during the spring period.

Soil temperature was consistently and significantly lower in the snow removal plots throughout the three winters (Figure 1.4). Minimum soil temperature in snow removal plots across drainage classes was reached during late February or early March, depending on the year, with minimum soil temperatures of -13.4˚C, -8.9˚C, and -14.9˚C in the winters of 2018/19, 2019/20, and 2020/21, respectively. There was a significant three-way interaction between treatment, week, and depth during 2018/19 and 2020/21 (Figure 1.5). The interaction manifested as more pronounced differences between treatments at shallow depths with decreasing differences as depth increased. For example, for the winter of 2019/20, mean soil temperatures in the snow removal treatment were lower than ambient conditions by 1.7˚C, 1.6˚C, 1.6˚C, 1.4˚C, and 1.3˚C for depths 10cm, 20cm, 30cm, 40cm, and 60cm ($p < 0.001$), respectively.

Soil temperature increased as depth increased, with soil temperature at 60cm rarely reaching sub-freezing temperatures and showing little variability, compared to 10-40cm depths, which reached sub-freezing soil temperatures during all three winters with high temporal variability that mirrored changes in air temperature (Figures A.1, A.2). Under ambient conditions, mean soil temperature at 10cm was significantly lower than 60cm by 0.9˚C ($p < 0.001$), 0.7˚C ($p < 0.001$), 0.7˚C ($p < 0.001$) during the winters of

2018/19, 2019/20, and 2020/21, respectively. In the snow removal treatment, mean soil temperature at 10cm was lower than at 60cm by 1.3˚C ($p < 0.001$), 1.1˚C ($p < 0.001$), 1.0˚C ($p < 0.001$) during the winters of 2018/19, 2019/20, and 2020/21, respectively.

Table 1.3: ANOVA summary for soil temperature models for the winters of 2018/19, 2019/20 and 2020/21. Numerator degrees of freedom and model coefficient p-values are shown. Bolded values indicate a significant result (p-value < 0.05).

	2018/2019		**2019/2020**		**2020/2021**	
	2018/11/04 - 2019/05/26		2019/11/14 - 2020/04/15		2020/10/23 - 2021/05/04	
Model term	**Degrees of freedom**	**p-value**	**Degrees of freedom**	**p-value**	**Degrees of freedom**	**p-value**
Intercept	1	**<0.001**	1	**<0.001**	1	**<0.001**
Drainage	3	**<0.001**	3	**<0.001**	3	**<0.001**
Treatment	1	**<0.001**	1	**<0.001**	1	**<0.001**
Week	29	**<0.001**	30	**<0.001**	30	**<0.001**
Depth	4	**<0.001**	4	**<0.001**	4	**<0.001**
Drainage:Treatment	3	**<0.001**	3	**<0.001**	3	**<0.001**
Drainage:Week	87	**<0.001**	90	**<0.001**	90	**<0.001**
Treatment:Week	29	**<0.001**	30	**<0.001**	30	**<0.001**
Drainage:Depth	12	0.276	12	**0.008**	12	**0.002**
Treatment:Depth	4	**<0.001**	4	**<0.001**	4	**<0.001**
Week:Depth	116	**<0.001**	120	**<0.001**	120	**<0.001**
Drainage:Treatment:Week	87	**<0.001**	90	**<0.001**	90	**<0.001**
Drainage:Treatment:Depth	12	0.897	12	0.050	12	0.649
Drainage:Week:Depth	348	1.000	360	1.000	360	1.000
Treatment:Week:Depth	116	**0.010**	120	0.272	120	**<0.001**
Drainage:Treatment:Week:Depth	348	1.000	360	1.000	360	1.000

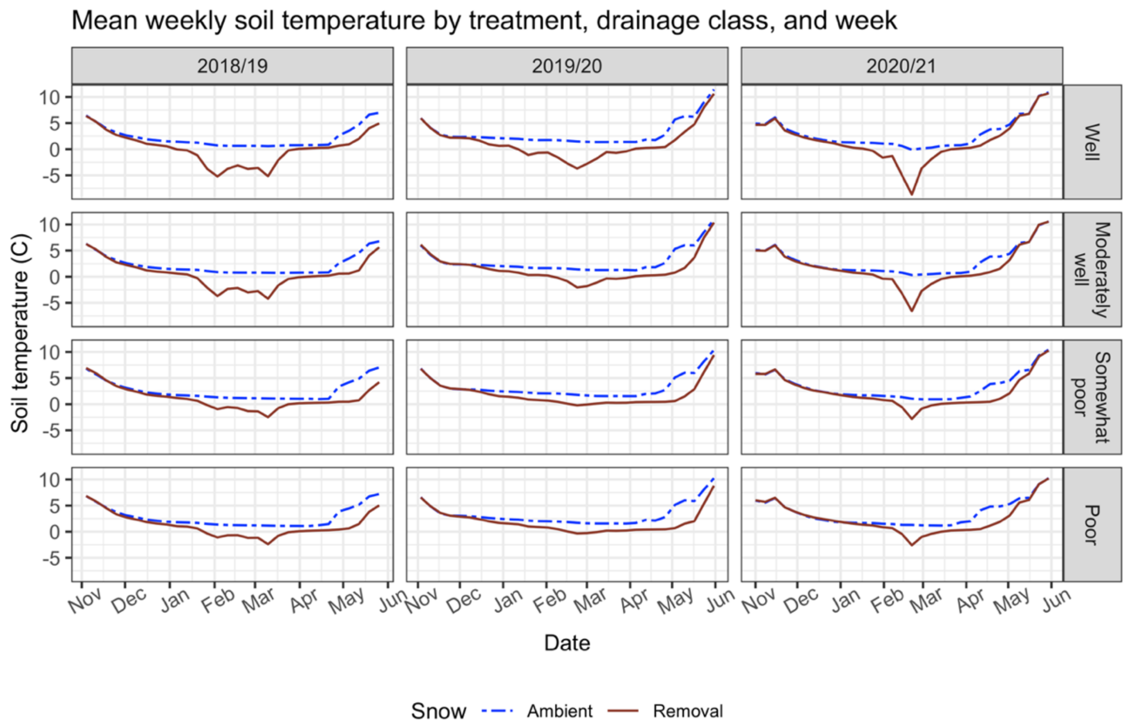

Figure 1.4: Mean weekly soil temperatures across drainage classes during the winters of 2018/19, 2019/20 and 2020/21 for the three-way interaction of treatment, drainage class, and week.

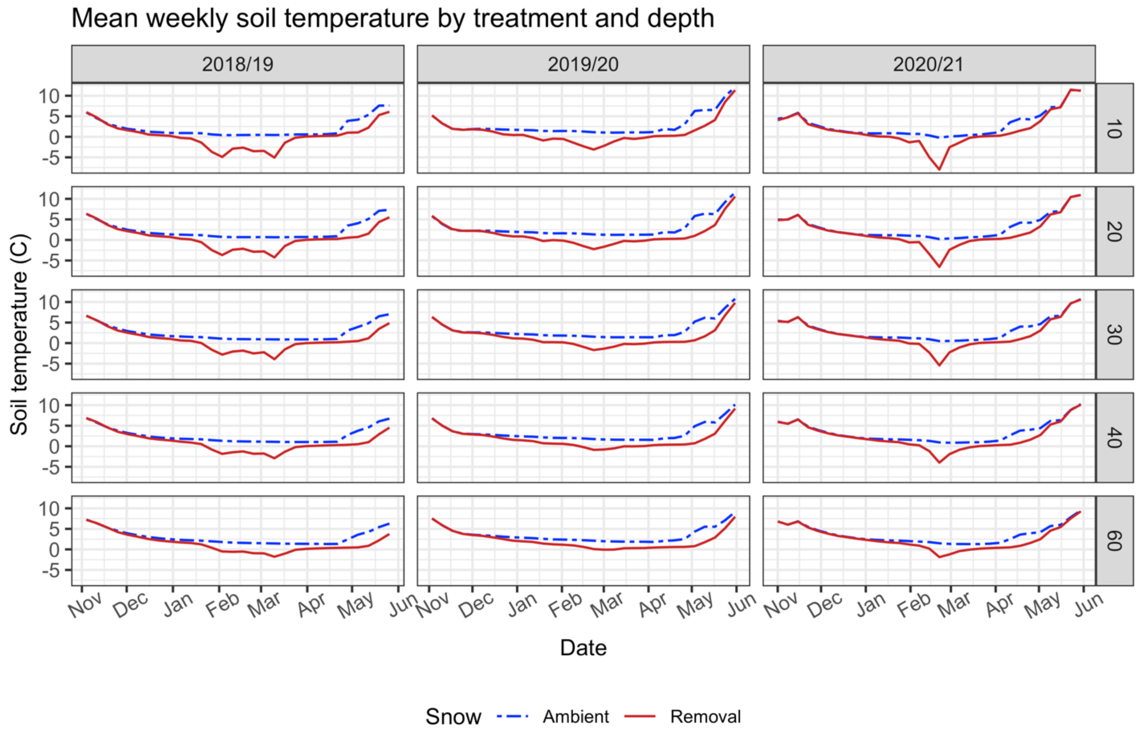

Figure 1.5: Mean weekly soil temperature for each year for the three-way interaction between treatment, depth, and week. Sensor depth is shown in centimeters on the right y-axis.

Soil frost was measured weekly between November and April of the winter of 2019/20, and between October and May of the winter of 2019/20. There was a significant interaction between treatment and week ($p < 0.001$) in both years (Table 1.4). Frost depth in the snow removal treatment across all drainage classes was 23cm ($p < 0.001$) and 25cm ($p < 0.001$) deeper compared to the control in 2019/20 and 2020/21, respectively. There was a significant interaction between drainage and treatment ($p = 0.001$) in 2019/20 for the effect on soil frost. Snow removal caused significantly deeper penetration of frost across drainage classes (Figure 1.6b). The difference between treatments decreased as drainage class became progressively wetter (*e.g.*, 31.0cm in the WD class versus 19.2cm in the PD class).

There was a significant main effect of drainage class in 2020/21 ($p < 0.001$). The drier drainage classes froze to a deeper depth compared to the wetter drainage classes (Figure 1.6b). For example, mean frost depth in the WD class was 17cm deeper compared to the poorly-drained class ($p < 0.001$). Additionally, visual examination of drainage-treatment effects by week indicated the drier drainage classes experienced a faster rate of thaw than the wetter drainage classes (data not shown).

Table 1.4: Three-way ANOVA results summary for the soil frost models for winters 2019/20 and 2020/21. Numerator degrees of freedom and model coefficient p-values are shown. Bolded values indicate a significant result (p-value < 0.05).

	2019/2020 2019/11/14 - 2020/04/15		**2020/2021** 2020/10/23 - 2021/05/04	
Model term	**Degrees of freedom**	**p-value**	**Degrees of freedom**	**p-value**
Intercept	1	**<0.001**	1	**<0.001**
Drainage	3	**<0.001**	3	**<0.001**
Treatment	1	**<0.001**	1	**<0.001**
Date	13	**<0.001**	20	**<0.001**
Drainage:Treatment	3	**0.001**	3	0.183
Drainage:Date	39	0.985	60	0.230
Treatment:Date	13	**<0.001**	20	**<0.001**
Drainage:Treatment:Date	39	0.994	60	0.963

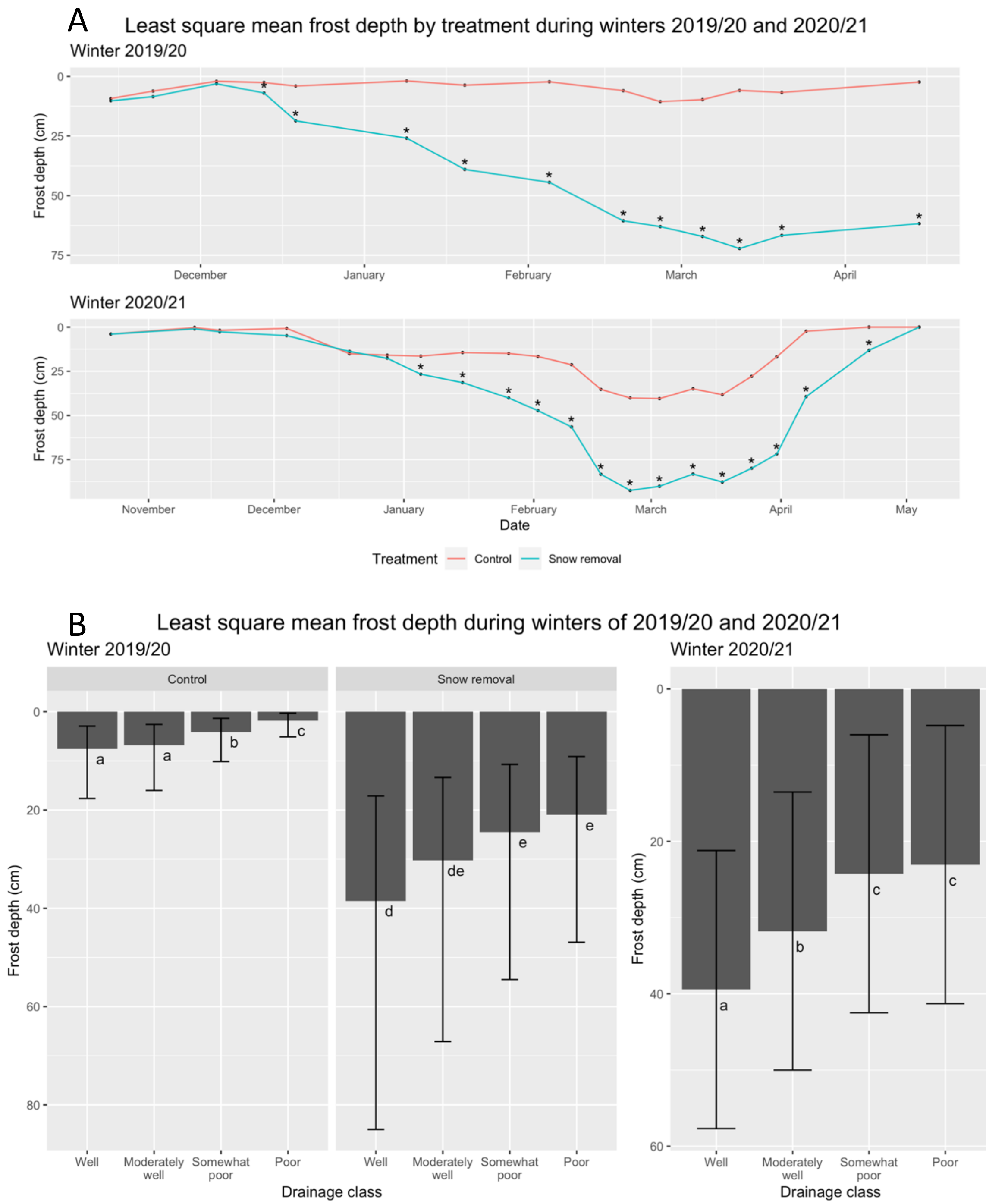

Figure 1.6: Least square means of soil frost depth during the winters of 2019/20 and 2020/21. The top panel (A) shows the significant treatment:date interaction for both years. The bottom panel (B) shows the drainage:treatment interaction for 2019/20, and the significant drainage effect during 2020/21. Asterisks indicates time periods where there was a significant difference in soil frost depth between treatments in panel A. In panel B, treatments with different letters indicate significant differences between means.

There was a significant positive relationship between FDD and frost depth for both control and treatment plots during both winters, except for the control during winter

2019/20 (see Figure 1.7 for p-values). However, the relationship was stronger in the treatment plots during both years ($r^2 = 0.74$ in 2019/20, $r^2 = 0.82$ in 2020/21; Figure 1.7) compared to the control plots (insignificant - $p = 0.07$ in 2019/20, $r^2 = 0.38$ in 2020/21; Figure 1.7). Comparison of the regression coefficients indicated that the rate of frost development was approximately 70% higher in the treatment plots compared to the control plots.

The slope of the regression line for the control plots in 2020/21 was notably higher than that of the control plots in 2019/20 (Figure 1.7), indicating more rapid frost development in the control plots during this year that was likely caused by lower snow depth in that year (Table 1.2).

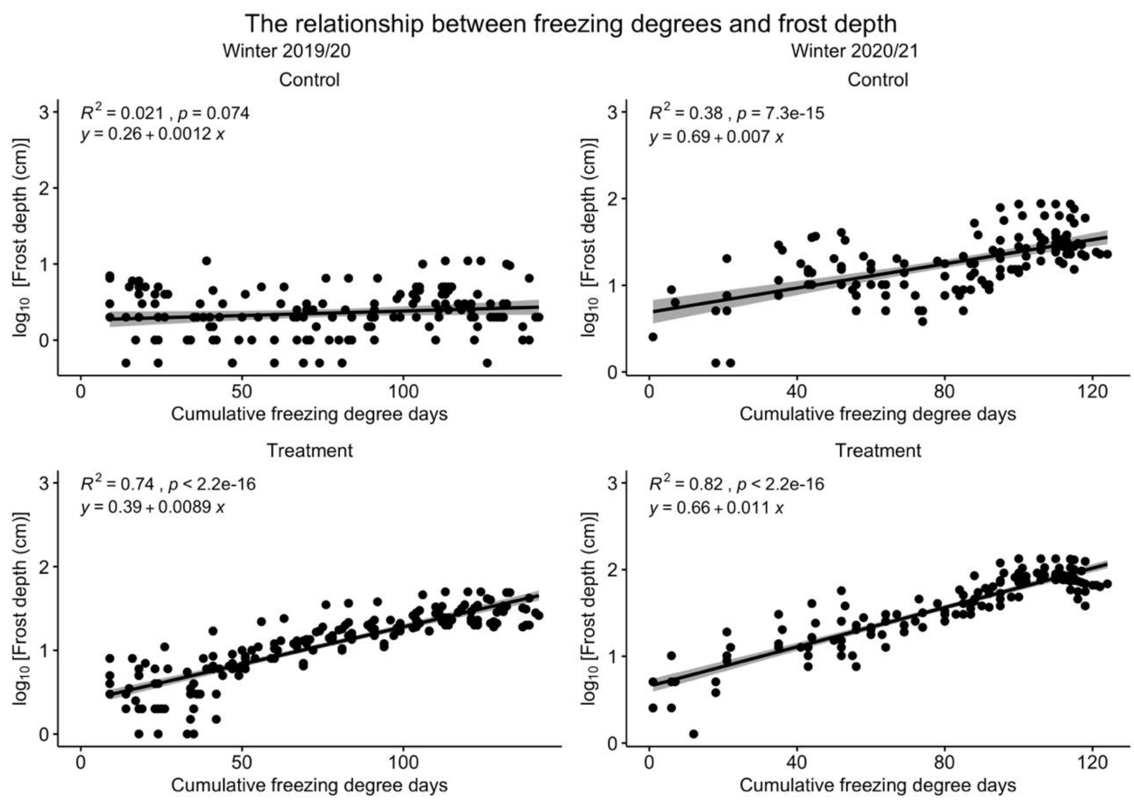

Figure 1.7: Linear regressions between cumulative freezing degree days and log-transformed frost depth in control and treatment plots during the winter of 2019/20 and 2020/21. Confidence limits are 95% confidence intervals.

The pairwise comparison of the estimated intercepts by drainage class indicated that the intercepts decreased as drainage decreased (*e.g.*, well-drained had the highest intercept, followed by MWD, SPD, and PD; Tables 1.5, 1.6). WD and MWD were significantly different than SWP and P in the control plots during winter 2019/20 (Table 1.5). In the winter of 2020/21, there were significant differences between MWD and PD, and between WD, SPD, and PD in the control plots (Table 1.5). For the treatment plots, WD differed from MWD, SPD, and PD in 2019/20 (Table 1.6). Finally, only WD significantly differed from MWD, SPD, and PD in treatment plots during the winter of 2020/21 (Table 1.6).

Table 1.5: Results of pairwise-comparisons of regression intercepts in control plots. Superscript letters indicate significant differences between means within each year and treatment (p-value < 0.05). Intercepts are in units of depth (cm). Values are back-transformed estimates.

	Control			
	2019/20		2020/21	
Drainage class	**Intercept**	**95% confidence interval**	**Intercept**	**95% confidence interval**
WD	3.35^{a}	2.73 - 4.10	25.7^{a}	20.8 - 30.9
MWD	2.74^{a}	2.25 - 3.35	20.0^{ab}	16.4 - 24.5
SPD	1.69^{b}	1.38 - 2.08	17.0^{bc}	13.7 - 20.9
PD	1.41^{b}	1.09 - 4.10	12.3^{c}	9.93 - 15.1

Table 1.6: Results of pairwise-comparisons of regression intercepts in treatment plots. Superscript letters indicate significant differences between means within each year and treatment (p-value < 0.05). Intercepts are in units of depth (cm). Values are back-transformed estimates.

	Treatment			
	2019/20		2020/21	
Drainage class	**Coefficient**	**95% confidence interval**	**Coefficient**	**95% confidence interval**
WD	16.1^{a}	14.2 - 18.2	50.1^{a}	44.7 - 56.2
MWD	11.9^{b}	10.5 - 13.5	37.2^{b}	33.1 - 41.7
SPD	10.6^{b}	9.37 - 12.0	33.1^{b}	29.5 - 38.0
PD	9.51^{b}	8.38 - 10.7	30.9^{b}	26.9 - 34.7

Effects of throughfall reduction

There was a significant interaction among drainage class, treatment, and depth on SWC in all three years (Table 1.7, Figure 1.8). Individual effects of treatment were inconsistent since the direction of the treatment effect varied across drainage classes (Figure 1.8). There were no significant differences in SWC between treatments at the soil surface (10cm, no effect at 20cm except for PD during 2020 and 2021). Significant differences in SWC between control and treatment within a drainage class at a given depth were mainly present for depths 30-60cm for the WD, MWD, SPD, and PD classes during all three years (except there was a significant difference between control and treatment for PD during 2020 and 2021) (Figure 1.8). However, the treatment plots were not consistently drier than the control plots. For example, the treatment plots were drier than the control for the WD class at 40cm during 2019 and 2020 (difference of -0.05, -0.04; $p < 0.001$, $p = 0.01$, respectively), and there was no significant difference ($p = 0.15$) during 2021. The SPD class followed a similar trend at 30cm and 60cm during 2019 ($p < 0.001$, $p = 0.04$, respectively). The treatment plots in the MWD class had significantly higher SWC, however, than the control plots at 30 and 60cm during 2019 ($p = 0.03$, $p = 0.004$), 2020 ($p = 0.007$, $p < 0.001$), and 2021 ($p = 0.02$, $p < 0.001$) with differences ranging from 0.04 to 0.06. The PD class showed a similar trend at 60cm during 2019 ($p < 0.001$), 20cm and 60cm during 2020 ($p < 0.001$, $p < 0.001$), and 20cm during 2021 ($p = 0.03$).

Table 1.7: Four-way ANOVA summary for soil water content models for the growing seasons of 2019, 2020, and 2021. Model coefficient p-values are shown. Bolded values indicate a significant result (p-value < 0.05).

	2019	**2020**	**2021**
Model term	**p-value**	**p-value**	**p-value**
Intercept	**<0.001**	**<0.001**	**<0.001**
Drainage	**<0.001**	**<0.001**	**<0.001**
Treatment	0.976	0.338	**<0.001**
Week	**<0.001**	**<0.001**	**<0.001**
Depth	**<0.001**	**<0.001**	0.220
Drainage:Treatment	**<0.001**	**<0.001**	**<0.001**
Drainage:Week	0.114	0.941	**<0.001**
Treatment:Week	0.001	0.650	0.399
Drainage:Depth	**<0.001**	**<0.001**	**<0.001**
Treatment:Depth	**0.002**	**<0.001**	**<0.001**
Week:Depth	1.000	1.000	0.974
Drainage:Treatment:Week	1.000	1.000	1.000
Drainage:Treatment:Depth	**<0.001**	**<0.001**	**<0.001**
Drainage:Week:Depth	1.000	1.000	1.000
Treatment:Week:Depth	1.000	1.000	1.000
Drainage:Treatment:Week:Depth	1.000	1.000	1.000

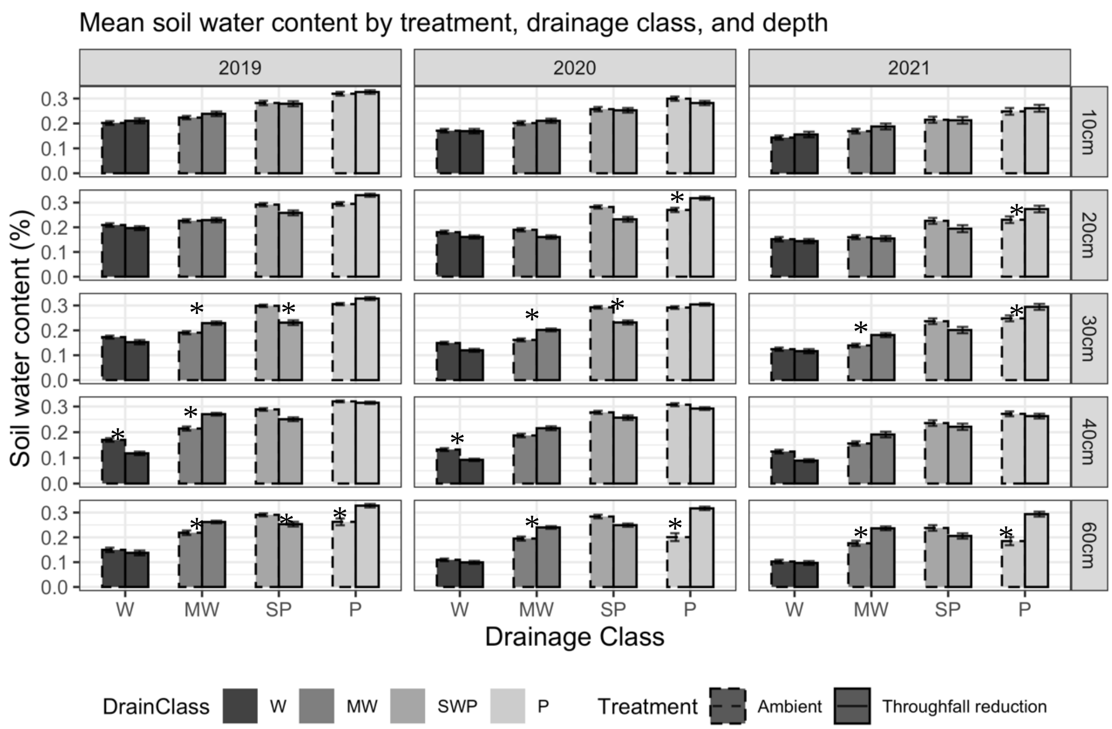

Figure 1.8: Three-way interaction between treatment, drainage class, and sensor depth for the three years of soil water content models. Sensor depth in centimeters in shown on the right y-axis. Asterisks indicate significant differences (p-value < 0.05) between control and treatment within a drainage class for a given depth.

Treatment effects on soil strength (bearing capacity) were limited. There was a significant interaction between drainage class and treatment, but only at 60cm during the growing season of 2020 (see Table 1.8 for p-values, Figure 1.9). Measurement date had no effect on soil strength, and percent clay was not a significant covariate in the models. Pairwise comparisons of drainage class by treatment means in 2020 indicated that the mean bearing capacity for the SPD class in the treatment plots (SPD-T) was significantly lower than the WD-T (p = 0.02, 16.17 psi) and MWD-T (p = 0.005, 18.37 psi difference) (Figure 1.9). SPD-T was also significantly lower than MWD-C (p = 0.005, 18.49 psi) and SPD-C (p = 0.04, 13.23 psi; Figure 1.9).

Table 1.8: Three-way ANOVA results summary for the soil strength models for 2020 and 2021. Bolded values indicate a significant result (p-value < 0.05).

	2020		**2021**	
	30 cm depth	60 cm depth	30 cm depth	60 cm depth
Model term	**p-value**	**p-value**	**p-value**	**p-value**
Intercept	**<0.001**	**<0.001**	**<0.001**	**<0.001**
Drainage	0.888	**0.034**	0.891	0.716
Treatment	0.262	0.253	0.647	0.351
Date	0.779	0.943	0.551	1.000
Percent clay	0.245	0.502	0.025	0.183
Drainage:Treatment	0.242	**0.014**	0.688	0.555
Drainage:Date	0.990	0.999	0.864	0.915
Treatment:Date	0.872	0.999	0.778	0.636
Drainage:Treatment:Date	0.997	1.000	0.967	0.942

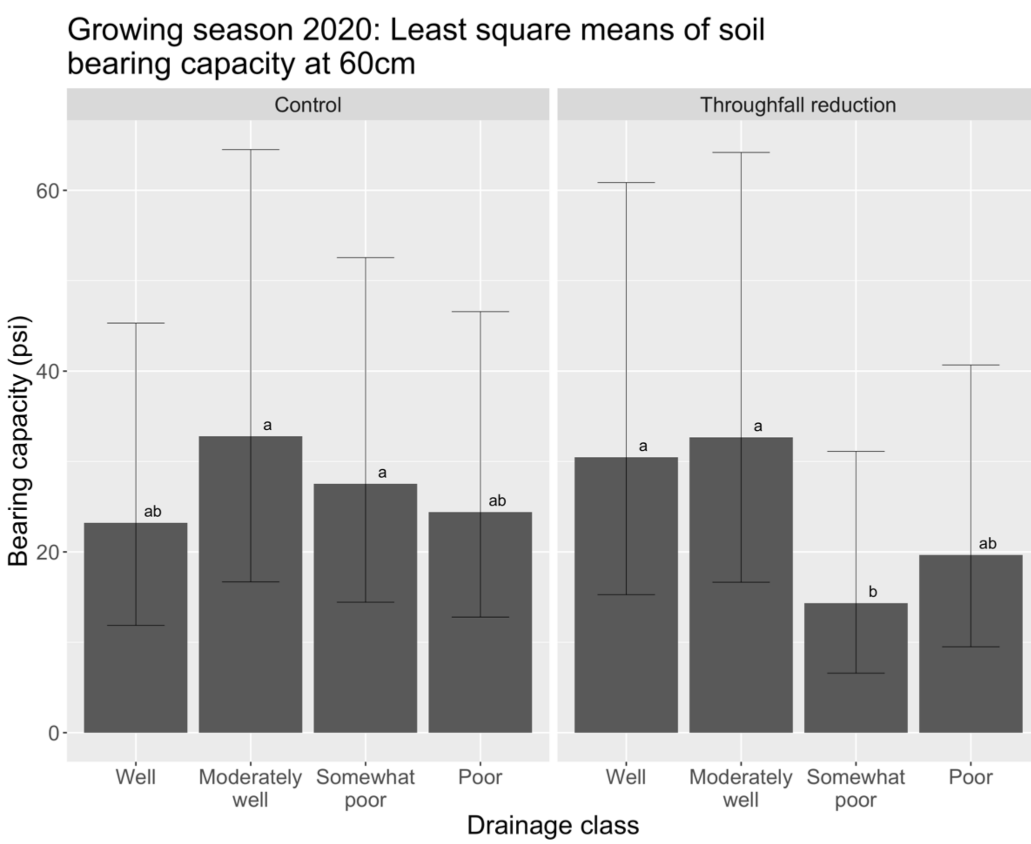

Figure 1.9: Least square means of soil strength (bearing capacity) across drainage classes and treatments at 60cm. Letters indicate significant differences as a pairwise comparison between drainage class and treatment (p-value < 0.05). Error bars represent 95% confidence intervals.

Linear regression indicated relationships between soils strength (bearing capacity) and soil water content were also limited. All relationships (r^2) were weak and inconsistent in direction across drainage classes (Figures B.1, B.2, B.3, B.4). For example, there was a significant positive relationship between soil strength and SWC in the well-drained class at 30cm in 2020 ($r^2 = 0.25$, $p = 0.002$), as well as the poorly-drained class at 60cm in 2021 ($r^2 = 0.30$, $p = 0.005$). On the other hand, there was a significant negative relationship between soil strength and SWC in the moderately well-drained class at 30cm in 2020 ($r^2 = 0.18$, $p = 0.011$) and the poorly-drained class at 60cm in 2020 ($r^2 = 0.17$, $p = 0.0013$).

Discussion

Changes in winter and summer precipitation under climate change will have implications for forest soil operability, since frost depth (as influenced by changes in snow cover) and soil moisture have been shown to influence soil strength (Greacen & Sands, 1980; McNabb et al., 2001; Uusitalo et al., 2019). In this study, I manipulated winter snow cover and summer throughfall across a soil drainage class gradient to assess the effect of reduced precipitation on soil physical properties. Drainage class was a strong indicator of soil temperature and soil moisture throughout the period of study. The snow removal treatment significantly increased frost depth which varied by drainage class and year. Associated with that effect was a strong relationship across drainage classes between frost depth and cumulative freezing degree days when snow was removed. In contrast, there were limited effects of throughfall reduction on soil moisture during the growing season and no effect on soil strength. Relationships between soil strength and soil moisture were generally weak and inconsistent across and within drainage classes.

Effects of snow removal

In terms of soil temperature, these results show that snow removal significantly decreased soil temperature (Figure 1.4) and increased frost depth. These results are consistent with previous literature that has shown soil temperature is significantly decreased under snow removal treatments (Decker et al., 2003; Groffman et al., 2001; Hardy et al., 2001). Decker et al. (2003) found similar trends in soil temperature under snow removal versus ambient snow treatments, where the temperature variation in soil decreased with depth and when snow was retained. Additionally, snow cover was found to be a strong regulator of soil temperature during the winter by Hardy et al. (2001). Soil temperature was attenuated as drainage worsened, which mirrors the soil moisture results in that warmer soil temperatures correlate with higher soil water contents due to the low thermal diffusivity of wet soils (Arkhangelskaya & Lukyashchenko, 2018). In this study, soil temperature increased from well-drained to poorly-drained in both the control and snow removal treatment during winter months. Even when snow was removed, temperature effects did not manifest at the same depth in the wetter drainage classes compared to the drier drainage classes, and wetter drainage classes demonstrated a slower rate of warming due to low thermal diffusivity. A drainage class gradient has not been utilized in previous snow removal studies, so these results add novel insight on frost development under changing precipitation regimes across a range of conditions. Additionally, the observed increases in soil temperature during winter with depth across both treatments and drainage classes has been observed in a number of other studies

(Decker et al., 2003; Friesen et al., 2021; Pavelka et al., 2007; Singh & Sharma, 2017). The depth dependence of soil temperature has been shown to be universal across sites, where soil temperature fluctuates in the top two meters of soil but remains constant as depth increases due to the heat capacity of the soil and thermal gradient from the land surface (Selker & Or, 2019).

Soil frost development was also dependent on treatment and drainage class, where snow removal caused significantly deeper frost development that was further associated with drainage class (Figure 1.6). The soil frost results mirror the temperature findings from this study and much of the previous literature on soil frost and snow removal. Snow removal studies at the Hubbard Brook Experimental Forest (HBEF) showed that snow removal can cause deeper frost penetration across a range of landscape positions and aspects (Cleavitt et al., 2008; Hardy et al., 2001). The frost depths observed under snow removal in this study, however, were deeper than those observed during snow removal studies at the HBEF due to the colder climate of northern MN compared to NH (Cleavitt et al., 2008; Hardy et al., 2001). Drainage class regulated soil frost depth since soil temperature was highly dependent on drainage class as well as the snow removal treatment. From WD to PD (increasing wetness), soil temperature became less variable and remained warmer throughout the winter. As a result, frost did not develop to the same degree in wetter drainage classes since soil temperature did not reach sub-freezing temperatures at the same depths as drier drainage classes. Importantly, the use of a drainage class gradient in this study has not been explored in previous snow removal studies, thus significantly contributing to our knowledge of frost development in mineral soils across a landscape with varying soil drainage classes.

Frost depth increased with the cumulative number of freezing degree days (FDD), and the slope of this relationship was higher in the snow removal treatment compared to the control. Even when the coldest air temperatures were reached (maximum FDD), frost depth in the control remained relatively shallow compared to the snow removal treatment (Figure 1.7). For example, approximately 40 FDD were required to reach 30cm of frost with snow removal, compared to approximately 70 FDD in the control. The differences in these relationships across drainage classes reflect the influence of drainage class on soil moisture and how that affects the change in soil temperature. The well-drained class, under both snow removal and ambient conditions, had the highest estimated intercept in the regression of frost depth on FDD. Estimated intercepts decreased from well-drained to poorly-drained, representing the decline in frost depth in wetter drainage classes. There were also differences between years in the rate of frost development due to differences in air temperature and snow depth. Mean air temperature was significantly higher and mean snow depth was significantly lower in the winter of 2020/21 compared to 2019/20 (Table 1.2). Given that air temperature was still relatively cold in the winter of 2019/20 (Table 1.2), the differences in frost development rate and depth were likely caused by reduced snow cover in the treatment. Differences in frost between the control and treatment emphasize the importance of snow cover as a regulator of soil temperature and frost depth in mineral soils.

The results of the regression-line comparison support the findings of the soil temperature and frost depth models, which also reflect the strong regulation of temperature and frost depth by drainage class (in a three-way interaction with treatment and week, as well as another three-way interaction with treatment and depth). Across the drainage classes, however, snow removal caused an increase in the rate of frost development with the number of cumulative freezing degree days. The positive relationship between frost depth and FDD suggests that mineral soils across drainage classes will respond relatively consistently to a decrease in winter snowpack, as predicted by current climate change models (Handler et al., 2014). A drought index (*e.g.*, U.S. Drought Monitor, Palmer Drought Severity Index) may be useful to identify antecedent moisture of sites during the fall in order to predict the approximate maximum depth of frost and the rate of its development based on a site.

Effects of throughfall reduction

Effects of throughfall reduction, drainage class, and depth on soil moisture were often inconsistent and unexpected. Notably, there was no difference in soil water content at the soil surface (10cm) between the control and throughfall reduction treatments, which is where a reduction effect should be most apparent. Additionally, some of the treatment plots often had higher SWC than the control plots across the drainage classes even though the treatment plots were receiving less than half the volume of throughfall compared to the control during 2021 (Figure 1.7; see methods for throughfall volume measurements). For example, SWC was higher in the throughfall reduction treatment compared to the control treatment in MWD at 30cm, 40cm, and 60cm (2019-2021), and PD at 60cm (2019-2021). In contrast, SWC was higher in the control in WD at 40cm (2019, 2020) and SPD at 30cm (2019, 2020) and 60cm (2019). This trend in soil moisture, which was inconsistent with our expectations, suggests that either the treatment was not modifying soil moisture or that another variable was negating the throughfall reduction.

Potential artifacts exist when designing and implementing throughfall reduction treatments, especially in forested ecosystems with one level of precipitation manipulation (Beier et al., 2012; Hoover et al., 2018). For example, the relatively small plot size (16 m^2) may have limited the ability of the throughfall reduction shelters to modify the soil microenvironment. As plot size decreases, the risk of edge effects increases, meaning that precipitation could enter the plot via other routes other than vertical interception (Beier et al., 2012; Fay et al., 2000). Additionally, the plots in this study were not trenched, which may have resulted in lateral flow or influence from tree roots outside the plot boundaries. Increased gradients in total water potential in treatment plots may have caused differences in capillary rise, which may have also contributed to the unclear trends in soil moisture (Romero-Saltos et al., 2005). Manipulations to precipitation may also alter evapotranspiration (ET) from the soil surface, which could reduce the amount of water infiltrating into the soil (Beier et al., 2012). Finally, heterogeneity in sensor placement may have resulted in inherent differences between control and treatment plots within a drainage class. Although the cause of the inconsistent treatment effect is unclear, the results highlight the need to give careful thought in the design of throughfall reduction studies.

Drainage class was also a clear indicator of soil moisture in the context of the interaction between treatment, drainage class, and depth. Soil water content consistently increased from well-drained to poorly-drained (Figure 1.7, A.3). This trend agrees with previous studies of the relationship between drainage class and soil moisture (Briggs &

Lemin, 1994; Henninger et al., 1976; Veneman et al., 2008). The consistent trends in soil water content across the drainage class confirms the correct identification of drainage classes in the field during this study.

The lack of any effect of throughfall reduction on soil strength aligns with the lack of treatment effect on SWC (Table 1.8). There were also inconsistent effects of drainage class on soil strength (Figure 1.8, Table 1.8). The lack of significant differences among drainage classes may have been due to differences in soil texture, since the dynamic penetrometer is sensitive to texture (Herrick & Jones, 2002). However, the results overall contrast with many studies that have shown that soil strength decreases as soil water content increases (Cambi et al., 2015; Greacen & Sands, 1980; McNabb et al., 2001; Uusitalo et al., 2019). Few studies, however, have investigated the effect of experimental throughfall reduction on soil strength in forest ecosystems. Yang et al. (2019) constructed throughfall reduction shelters over 20 x 20 m plots in subtropical planted forests in China. That study found that the throughfall reduction treatment significantly reduced soil water content and soil aggregate stability due to an increase in porosity and decline in free Al oxides (Yang et al. 2019). Notably, however, the soils in the study by Yang et al. (2019) were sampled during the rainy season, possibly indicating that pre-wetted soils may be more at risk of aggregate disruption with subsequent drying. The authors suggest that decreases in precipitation under a changing climate may result in more extreme wetting and drying cycles, which have the potential to disrupt soil aggregate stability in the long-term. While Yang et al. (2019) studied a different climate than that of northern Minnesota (subtropical versus temperate) and there may have been site-specific differences such as ET, it serves as an example of a study in which soil moisture and soil strength (in this case, aggregate stability) were modified by throughfall reduction. The plot size (20 x 20 m) was much larger than the size of the plots in our study (4 x 4 m), thus potentially reducing edge effects.

Compared to laboratory measurements, the in-situ measurement of soil strength has the potential for measurement error, especially in soils with glacial parent material like those in Minnesota. There were inconsistent and often insignificant relationships between soil strength and SWC in this study (Figure B.1, B.2, B.3, B.4), which may have been due to measurement error with the use of the dynamic penetrometer in rocky, heterogenous soils. For example, plots in Aitkin and St. Louis counties were extremely rocky, and large boulders were present within plots in the St. Louis County site. Contact with a belowground root or coarse fragment could alter the angle of the dynamic penetrometer, which reduces the accuracy of the measurement (Minnesota Department of Transportation, n.d.). Previous studies have suggested that the dynamic penetrometer is sensitive to differences in soil moisture and texture, especially in heterogenous soils (Herrick & Jones, 2002). Therefore, much difficulty still exists when using a dynamic penetrometer in highly heterogenous soils with a high concentration of tree roots and coarse fragments.

The dynamic penetrometer, however, is not the only method to measure in-situ soil strength in forest soils. The US Forest Service Forest Inventory and Analysis (USFS FIA) program has used pocket penetrometers to measure in-situ soil strength. Kolka et al. (2012) found significant relationships between soil strength and harvest history, harvest season, and landscape position. They found that soil strength was lowest in areas with a high soil water content (*i.e.,* toeslope position with fine-textured soils; Kolka et al., 2012). The study by Kolka et al. (2012) showed that soil strength varied across landscape position, which is similar to a drainage class gradient. The pocket penetrometer avoids many of the issues of the dynamic penetrometer, since it only measures surface soil strength, in contrast to a depth of 45 cm with the dynamic penetrometer. The pocket penetrometer may be more logistically feasible (*e.g.*, more affordable, smaller, and easier to transport) compared to the dynamic penetrometer, and further study would be required to test the accuracy of the pocket penetrometer versus other methods of measuring in-situ soil strength.

Implications for management

The increase in frost development that occurred with snow removal may have implications for future accessibility of forest stands during the winter, potentially increasing the period in which those stands could be harvested with limited impacts to the soil if predicted reduction in snowfall occur. The maximum frost depths reached in the snow removal treatment would sufficiently support harvesting equipment since previous work has recommended at least 15 cm of frost for heavy equipment, though this estimate may be inaccurate to the increase in equipment weight since the early 2000s (Stone, 2002). This overall increase in accessibility during the winter would have major economic implications, especially for stands with a large quaking aspen component which reproduces clonally via root suckers, such as in this study. Aspen (including quaking aspen and balsam poplar, *Populus balsamifera*) comprises 30% of Minnesota's forested lands and 53% of timber harvests (Domke et al., 2008; Handler et al., 2014). Aspen is mainly utilized for pulp and paper, which is the dominant product manufactured by forest industry in Minnesota, as well as oriented strand board and engineered wood products (Division of Forestry, 2021). Therefore, an increase in accessibility to winter harvest sites would potentially increase the utilization of aspen and other species with minimal impacts to soil.

Current climate change models have simulated warming winter temperatures, which would result in a decline in the total number of frost days and possibly negate the effect of reduced snow cover (Handler et al., 2014). While this study suggests that winter frost depths will increase with reduced snow cover, there will be interactions between the effects of reduced snow cover and warmer winter temperatures on frost development in mineral soils of northern Minnesota. Current climate change models predict the mean winter temperatures in northern MN will increase by 2100 (PCM B1: 2.2˚C; GFDL A1FI: 3.0˚C), though mean winter temperatures are not expected to rise above freezing (Handler et al., 2014). So even with the predicted warming, sub-freezing temperatures with reduced snowpack would likely still result in increased frost development. Regardless, future research on frost regimes under a changing climate could include the addition of a warming treatment to simulate warmer winter air temperatures.

Further study is required to quantify the effects of reduced precipitation on soil strength in forest ecosystems. Understanding the operability of forest soils under climate change is crucial in maintaining sufficient yield from summer timber harvests with minimal impacts to the soil. The relationship between soil strength and soil moisture has been reported in previous studies, so the predicted declines in summer precipitation (in addition to an increase in extreme precipitation events) will likely have a tangible effect on forest operations in northern Minnesota (Greacen & Sands, 1980; Handler et al., 2014;

McNabb et al., 2001; Uusitalo et al., 2019). Future studies should aim to quantify soil strength under reduced precipitation scenarios across cover types and drainage classes for improved prediction of the operability of forest soils under climate change.

A major takeaway from this study is that drainage class was a strong predictor of soil moisture, temperature, and frost development. Drainage class is easily mapped and measured, and thus may be an important metric for forest managers when determining the feasibility of harvesting in the winter. Managers may be able to rely on drainage class, and the relationship between FDD and frost, to determine the harvesting periods of certain stands with minimal monitoring. Relationships between FDD and frost depth by soil type could be established to approximate the winter operability of a site, based on the approximate required frost depth based on the harvesting equipment. Drainage class can help managers to identify sites that may take longer to freeze and to determine approximately how many days would be required to reach sufficient frost depth to sustain heavy equipment. Alternatively, operators should also be encouraged to pack snow several days prior to harvesting to encourage increased frost development across drainage classes.

Conclusions

The results of this study will provide critical insight to managers on the long-term operability of forest soils under a changing climate. I applied a novel methodology by combining throughfall reduction and snow removal treatments across a gradient of drainage class in aspen forests of northern Minnesota, USA. However, the effects of winter warming on frost development were not investigated in this study, and thus would be an important addition to future studies to accurately understand the future of forest soil operability under climate change.

Based on current climate change models, northern latitudes are expected to experience decreased growing season precipitation and winters with reduced snow cover, which would have major implications for the operability of forest soils. I found that throughfall reduction during the growing season had minimal impacts on soil moisture and soil strength. The snow removal treatment during the winter significantly increased frost development and decreased soil temperature across drainage classes. Drainage class was a strong indicator of soil moisture, temperature, strength, and frost development. These results demonstrate the utility of using drainage class as a metric when inferring soil moisture and temperature and determining harvesting periods.

Chapter 2

No effect of throughfall reduction and snow removal on carbon dioxide emission and extractable nitrogen concentrations across forest soil drainage classes in northern Minnesota, USA

Abstract

Understanding the effects of moisture and frost dynamics on biogeochemical processes in forest soils provides insight on the long-term response of forest soils to changing climate, particularly changes in summer precipitation and winter frost influence on the storage and cycling of carbon and nitrogen. This study simulated predicted climate change conditions with growing season throughfall reduction and winter snow removal using a paired-plot design across a soil drainage class gradient in upland forest soils in northern Minnesota, USA. *In situ* bulk soil respiration and concentrations of extractable soil nitrogen species (ammonium, nitrate/nitrite, and total nitrogen) were measured between June and September 2020, and April and September 2021. Soil respiration was not significantly affected by throughfall reduction and snow removal. Drainage class was a significant factor only during the spring thaw period, in which the poorly drained plots had a significantly lower respiration rate compared to the well-drained plots which was associated with drainage class effects on soil temperature. Results of the laboratory incubation corroborate no significant effect of treatment on soil respiration, and that drainage class and moisture content were stronger indicators of respiration. Similarly, nitrogen concentrations were not significantly different between treatments, and few nitrogen species differed by drainage class. Due to the lack of responses to the treatments

both in the field and the incubation, nitrogen and carbon fluxes in northern Minnesota forest soils may not be sensitive to reductions in summer soil moisture and deeper winter frost.

Introduction

Forest soils are a major component of carbon and nitrogen cycling both within upland forest ecosystems and globally (Bernal et al., 2012; Schlesinger & Andrews, 2000; Wei & Shaopeng, 2010). Soil respiration serves as the main pathway for the release of plant-fixed carbon dioxide (CO_2) back to the atmosphere either through root (autotrophic) respiration or decomposition of plant biomass by microbes (heterotrophic respiration) (Högberg & Read, 2006; Schlesinger & Andrews, 2000; Wei & Shaopeng, 2010). Nitrogen commonly limits growth in upland forest ecosystems because it is an essential macronutrient for plant and forest health (Bernal et al., 2012; Rennenberg et al., 2009). Plant available forms of nitrogen enter the forest soil environment via the decomposition of biomass in the organic horizon and transformation by microbes (Rennenberg et al., 2009). Microbes and fungi transform of carbon and nitrogen in soils, and these reactions are largely controlled by climate, specifically soil moisture and temperature (Bernal et al., 2012; Haaf et al., 2021; Knoepp & Swank, 2002; Monson et al., 2006). Since forest soils store large amounts of carbon (as well as nitrogen), understanding forest biogeochemical cycling under a changing climate is critical when managing forests as long-term carbon sinks (Bernal et al., 2012; Schlesinger & Andrews, 2000).

Carbon and nitrogen dynamics in northern latitude forests are likely to be impacted by changes in summer and winter precipitation, as well as changes to winter frost dynamics (Borken et al., 2006a; Schindlbacher et al., 2012; Monson et al., 2006; Muhr et al., 2009). For the Laurentian Mixed Forest Province in northern Minnesota, climate modeling by Handler et al. (2014) projects a slight decrease in total precipitation by 2100 with the largest decline (40 percent) occurring during the summer, although precipitation events may become more extreme with greater total precipitation per event (Handler et al., 2014). Additionally, by the end of the century, winter temperatures are projected to increase by 2.8˚C and 6.7˚C by low and high emission scenarios climate scenarios, respectively, and more winter precipitation will occur as rain instead of snow (Handler et al., 2014). Winter soil temperatures are predicted to increase and depth of snowpack to decrease (Collins et al. 2013), with concurrent reductions in total annual soil frost days (Handler et al., 2014). These regional climate change predictions are similar to those for northern latitudes not only in north America, but also northern latitudes globally.

In northern ecosystems, snowpack serves as an insulative layer over the soil surface, maintaining warmer soil temperatures and shallower soil frost (Campbell et al., 2010). Decreased snowpack depth, and thus greater frost depth, have been correlated with decreased net heterotrophic respiration from forest soils (*i.e.,* carbon dioxide flux) (Hardy et al., 2001; Monson et al., 2006; Muhr et al., 2009). Large fluxes of carbon dioxide, however, often coincide with the thawing period due to the release of stored carbon dioxide or increase in microbial activity following thaw (Öquist & Laudon, 2008). Increased soil frost during the winter has also been shown to control soil respiration into the growing season due to changes in microbial activity and root respiration associated with altered soil temperature (Öquist & Laudon, 2008). Similarly, Fitzhugh et al. (2001) found that inorganic nitrogen concentrations increased following freezing events, and nitrogen leaching subsequently increased. Increased frost depth may increase nitrogen mineralization and nitrification rates due to higher amounts of microbial and root mortality and the disruption of soil aggregates (Fitzhugh et al., 2001).

Climate change associated declines in soil moisture during the growing season may cause decreases in microbial activity and decreases in root respiration (Borken et al., 2006a; Schindlbacher et al., 2012). Past studies have used throughfall reduction, where a percentage of precipitation below the canopy is diverted from experimental plots, to approximate the effects of reduced precipitation in the future. In a study that examined the effect of reduced precipitation on bulk soil respiration, Borken et al. (2006a) found that complete throughfall exclusion significantly decreased bulk soil respiration in temperate forest soils in Massachusetts, USA. Additionally, Schindlbacher et al. (2012) found that reductions in bulk soil respiration due to complete throughfall exclusion offset the increases in bulk soil respiration due to soil warming. In that study, both throughfall exclusion and soil warming reduced soil moisture, with declines greatest when throughfall exclusion and warming were combined. The influence of throughfall reduction on nitrogen dynamics, however, is not as clearly understood due to the complexity of nitrogen cycling in soils. For example, throughfall reduction has been shown to increase extractable ammonium concentrations but decrease extractable nitrate concentrations, with no discernable effect on total nitrogen supply (Homyak et al., 2017). It is possible that nitrogen supply in soil may not be as sensitive to changes in soil moisture as compared to soil carbon fluxes. Deng et al. (2021) conducted a meta-analysis of global nitrogen dynamics and found that drought had no significant effect on total nitrogen concentrations in forest ecosystems. However, as with the above, extractable ammonium significantly increased under drought conditions, and extractable nitrogen significantly decreased (Deng et al., 2021).

Drainage class may be an important factor when considering fluctuations in soil moisture and frost with climate change and related effects on carbon and nitrogen dynamics. In the field, soil texture and landscape position create differences in soil drainage, which is classified by depth to redoximorphic features (*i.e.*, mottling, gleying; Veneman et al, 1998). Soil aeration, relative moisture supply, and potential rooting depth are all influenced by drainage class (Briggs & Lemin 1994). As a result of variation in soil moisture, microbial communities and their activity may also vary with drainage class. For example, rates of soil respiration versus methanogenesis differ with drainage class due to variations in moisture levels (Davidson et al., 1998). Effects of reduced rainfall and snowpack may vary by drainage class in forest soils, but I am not aware of such studies.

Assessment of the effects of moisture and frost dynamics on biogeochemical processes provide applicable insight on the long-term response of forest soils under a changing climate. Paired-plot experiments with either snow removal or throughfall exclusion treatments have been used to investigate the response of forest soil carbon and nitrogen fluxes to changes in seasonal precipitation (Friesen et al., 2021; Monson et al., 2006; Muhr et al., 2009; Öquist & Laudon, 2008; Borken et al., 2006a; Schindlbacher et al., 2012; Homyak et al., 2017). Snow removal treatments increase soil frost depth in forest ecosystems and alter soil temperature, which influences microbial transformations of carbon and nitrogen. Few studies have combined snow removal and throughfall reduction treatments in one experiment, with none known of in the Lake States. Applying the two treatments seasonally on the same plot allows for the simulation of future precipitation patterns, as projected by climate models. Use of a paired-plot design directly compares the treatment to ambient conditions to evaluate the effect of the two treatments. This study aims to bridge the gap between existing snow removal and throughfall reduction experiments to better understand the future of forests in Minnesota under a changing climate. To do so, I quantified the influence of combined throughfall exclusion and snow removal on soil respiration and nitrogen cycling, in conjunction with a laboratory incubation under controlled conditions.

Methods

Study area

The study region covered three sites located in three counties in northern Minnesota: Aitkin, Itasca, and St. Louis counties (Figure 2.1). The Aitkin and Itasca sites were located within state forests (Solana and George Washington State Forests, respectively). The St. Louis County site was located on county-owned land. Our study region was located on the ancestral, traditional, and contemporary lands of the Anishinaabe, Oceti Sakowin, and Mdewakanton. These study sites specifically reside on land ceded in the Chippewa Treaty of 1839 and the Chippewa Treaty of 1854.

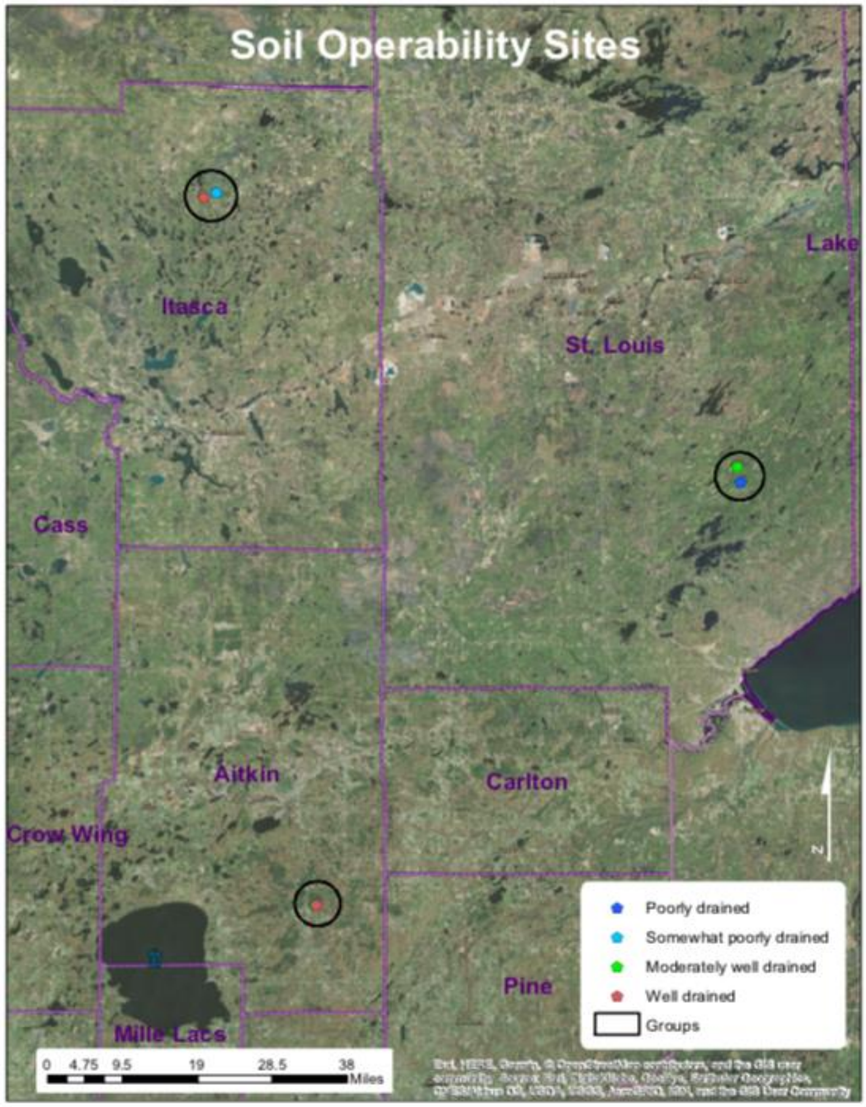

Figure 2.1: Map of study sites. Black circles indicate site groupings. Each site contains paired plots across four drainage classes.

All sites were dominated by upland quaking aspen in the forest canopy with beaked hazel (*Corylus cornuta*), willow (*Salix* spp.), or speckled alder (*Alnus incana*) in the understory in the Laurentian Mixed Forest Province. Mean summer (June – August) temperatures for this region is 18 ˚C and winter temperatures average -12.28 ˚C (Handler et al., 2014). Average precipitation in LMFP during the summer was 305 mm, and average snowfall ranges from 1,016 mm to 1,778 mm (Handler et al, 2014).

Site characteristics

Mature quaking aspen (40-60 years) was the dominant tree species at all sites. Soils at each site were predominantly loams occurring on relatively flat topography (less than 10% slope). Plot locations within the target drainage classes (well-drained through poorly drained) were identified based on depth to redoximorphic features (Table 2.1).

Differences in mean percentages of pre-treatment carbon and nitrogen are shown in Table E.1. Generally, percentages of carbon and nitrogen increased from the well-drained to poorly-drained class, and were highest in the St. Louis county site.

Table 2.1: Description of soil series and textures for each drainage class within the three sites (county) determined from soil survey information. Soil survey information from National Cooperative Soil Survey (NRCS)**.**

Site (County)	Drainage class	Soil unit/taxonomy	Soil texture	Depth to redoximorphic features
Aitkin	Well	738B: Milaca-Millward complex, 2-8% slopes Coarse-loamy, mixed, superactive, frigid Oxyaquic Glossudalfs Coarse-loamy, mixed, superactive, frigid Typic Hapludalfs	Fine sandy loam	> 102 cm
	Moderately well			51 – 102 cm
	Somewhat poor			25 – 51 cm
	Poor			0 – 25cm
Itasca	Well	803B: Warba-Menahga complex, 1-8% slopes Fine-loamy, mixed, superactive, frigid Haplic Glossudalfs Mixed, frigid Typic Udipsamments	Fine sandy loam	> 102 cm
	Moderately well			51 – 102 cm
	Somewhat poor	621: Morph very fine sandy loam Fine-loamy, mixed, superactive, frigid Typic Glossaqualfs	Very fine sandy loam	25 – 51 cm

	Poor	167B: Baudette silt loam, 0-5% slopes Fine-silty, mixed, superactive frigid Oxyaquic Hapludalfs	Silt loam	0 – 25cm
St. Louis	Well	F122B: Aldenlake-Pequaywan complex, pitted, 0-8% slopes Coarse-loamy, isotic, frigid Dystric Eutrudepts Coarse-loamy, isotic, frigid Aquic Dystric Eutrudepts	Sandy loam	> 102 cm
	Moderately well			51 – 102 cm
	Somewhat poor	F103B: Brimson stony fine sandy loam, 2-5% slopes, very stony Coarse-loamy, isotic, frigid Aquic Dystric Eutrudepts	Stony fine sandy loam	25 – 51 cm
	Poor			0 – 25cm

Experimental design

The study occurred from May 2018 until May 2022. I used a paired-plot, factorial (4 x 2) experimental design with Factor 1 being drainage class and Factor 2 being treatment. Treatments were replicated across sites (n = 3), with each site containing eight plots across four drainage classes (well, moderately well, somewhat poorly, or poorly drained), for a total of twenty-four plots across all three sites (three replications per drainage class x treatment combination). Paired treatment and control plots (ambient conditions) were located adjacent to each other within each drainage class, creating a gradient across drainage conditions. Snow was removed from treatment plots during the winter (Figure 2.2), and throughfall was reduced during the growing season (Figure 2.3). More detail on each treatment is outlined below.

Snow removal treatment

Snow was removed from treatment plots during the winter according to the method defined by Friesen et al. (2021). All plots were oriented on an east-west transect. Plots that received treatment were randomized in each pair. To allow for snow removal without impacting the soil surface, gray aluminum window screening (Phifer Incorporated, Tuscaloosa AL) was placed over the entire treatment plot area prior to the first snowfall (Figure 2.2). Screens were not placed within the control plots. The color of the screening mimicked the albedo of bare soil. The screening also allowed for gas exchange without shading the soil surface. Screening was divided into three panels and anchored on the side of the plot opposite to the control plot and cut to fit around the boles of trees. Shrubs and other woody stems were cut prior to screen placement in both the control and treatment. Snow was cleared manually by a shovel or leaf-blower, or by lifting the panel and moving the snow off the far edge of the plot. Snow was always cleared and deposited away from the control plot to limit any possible disturbance of the experimental control. Snow was cleared after every storm of 2-inches or more, or at least weekly.

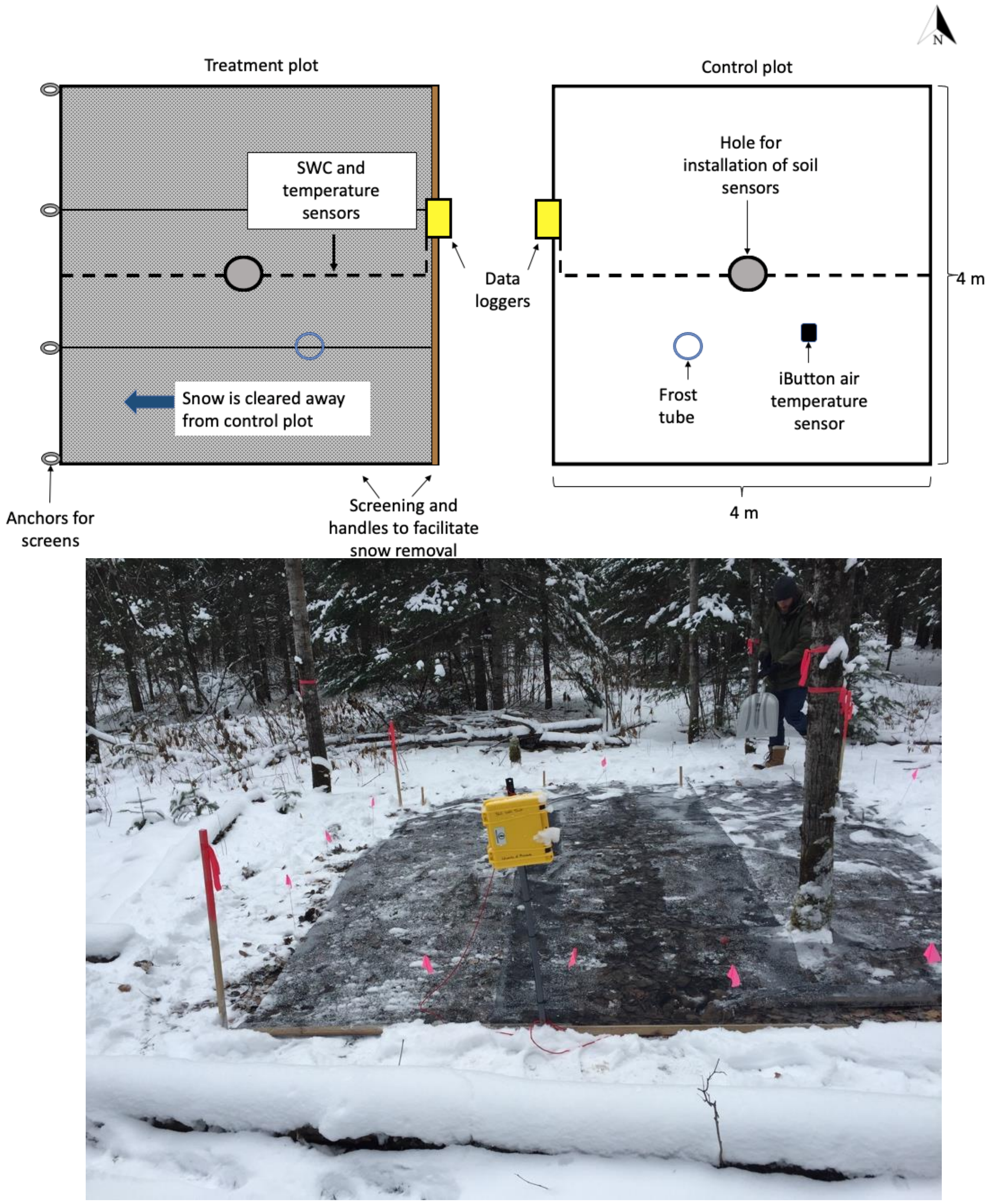

Figure 2.2: Paired-plot design schematic and field photo with snow removal treatment during the winter. (Photo credit: Alan Toczydlowski, University of Minnesota)

Throughfall reduction treatment

Throughfall reduction shelters were installed during the growing season to simulate a 50% reduction in throughfall (Figure 2.3). The shelters were constructed of 6-inch wide Greca clear Amerilux polycarbonate roof panels (Amerilux International LLC, De Pere, WI). Each shelter contained four panels, with a total area of 8 x 8 feet (2.4 x 2.4

m) and was framed with 1.5 in (3.81 cm) PVC pipe and connector fittings. The shelter framing was constructed from 2 x 4 in (5.08 x 10.16 cm) and 4 x 4 in (10.16 x 10.16 cm) dimensional lumber. The shelters were guttered with 4-inch (10.16 cm) wide, U-shaped white vinyl gutters that extended 40 centimeters past the 4 x 4-meter plot boundary. The peak height of the shelter was 7-feet (2.13 m) at the ridgeline, and 2-feet (0.61 m) at the gutter edge. The ridgeline of the A-frame shelter ran along a north-south transect so that panels were situated on an east-west transect to avoid greenhouse effects created by a south-facing panel. Control plots were left as an experimental reference and did not receive any precipitation reduction treatment. Plots that received treatment were randomized in each pair.

To assess treatment efficacy, the volume of throughfall in plots was measured biweekly during the growing season of 2021 using 20.3 cm funnels attached to glass jars that were placed in each quadrant of moderately drained plots at each site (n = 4 collectors per plot and site). The biweekly average throughfall volume for control plots was 648.6 mL (± 54.44 mL). and was 305.5 mL (± 108.41 mL) for treatment (exclusion) plots.

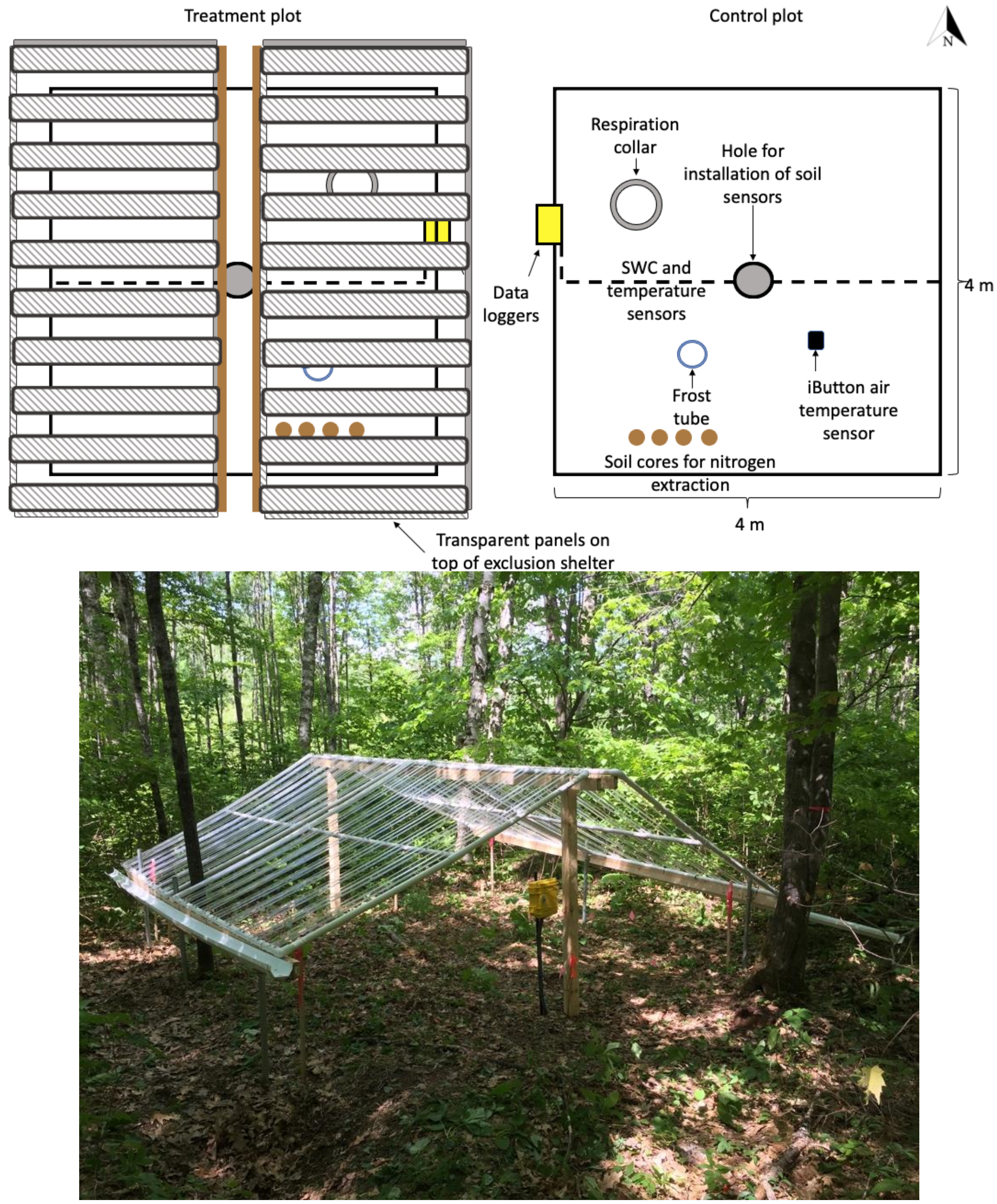

Figure 2.3: Paired-plot design with throughfall reduction treatment during the growing season. All plots and transparent roof panels are oriented on an east-west transect, with the ridgeline of shelter running north-south. Precipitation reduction shelters are designed to exclude 50% of throughfall. Plots that received treatment were randomized in each pair. The bottom photo shows the throughfall exclusion shelter on a treatment plot during the growing season. (Photo credit: Alan Toczydlowski)

Soil moisture, soil temperature, and air temperature measurements

Soil temperature and moisture were measured every 15 minutes at depths of 10, 20, 30, 40, and 60 cm via Decagon 5TM sensors (± 0.1˚C, ± 0.08% SWC; METER Group, Pullman, WA). Sensors were installed in a cluster at the center of plots (Figure

2.2, 2.3) and connected to EM50 data loggers (METER Group, Pullman, WA). Air temperature was recorded at control plots every 90 minutes (starting at 00:00) by Thermachron iButton sensors (± 0.5˚C; Maxim Integrated Products, Inc., Sunnyvale, CA) enclosed in a PVC solar shield. Soil moisture, soil temperature, and air temperature results are described in *Chapter 1*.

Soil frost measurements

Frost depth was measured weekly during the winters of 2019/20, 2020/21, and 2021/22 in all plots. Frost tubes were constructed by Northern Frost Tubes (Brian Hahn, Oconomowoc WI). Frost tubes were installed to a depth of 1.5 meters in the soil profile and were filled with a solution of water and color-changing indicator dye. The solution turned clear when frozen, indicating soil frost depth. Frost depth was measured to the nearest 2.5 cm in all plots. Soil frost results are described in *Chapter 1*.

In-situ measurements of bulk soil respiration

Bulk soil respiration (μmol m^{-2} s^{-1}) was measured biweekly during the growing season of 2020 (June – September 2020) and 2021 (April – September 2021). Gaseous carbon dioxide flux was measured with a LI-COR LI-8100 Automated Soil CO_2 Flux System (LI-COR Biosciences, Lincoln NE). Collars made of PVC with a diameter of 20 cm were constructed for use with the gas flux analyzer. Collars in each plot were inserted into soil to a depth of 2.5 cm during June 2020. Measurements of CO_2 concentrations within the chamber were taken over a two-minute period with a forty-five second post-purge. One measurement was taken per plot. Over successive days to accommodate travel among sites, all measurements were taken between 10:00am and 1:00pm to limit temporal variations in CO_2 flux. The LI-8100 was calibrated once in November 2020 and twice during the 2021 field season using certified pure, zero-grade nitrogen (N_2) gas and 1000 ppm CO_2 standard. A single soil temperature and moisture measurement was taken next to the soil collar at the time of the soil respiration measurement using a Decagon PROCheck soil temperature and moisture probe (model 5TM, METER Group, Pullman WA).

Extractable soil nitrogen

A sequential core technique was used to assess N availability during the growing season of 2020 and 2021, with cores being deployed at the same time each growing season and then sequentially extracted over consecutive months. Four PVC tubes (25 cm long and 5 cm diameter) were hammered 20 cm into the soil along a transect in each plot. One core was removed from each plot each month. An extra set of cores were taken in September 2020 and September 2021 to assess the effect of vegetation on nitrogen cycling. The extra set of cores were installed on the same day as removal to serve as a contrast to the cores that had been installed in the soil since June.

Cores were processed in the USFS Northern Research Station Marcell Research Center laboratory and separated by depth (0-5 cm, 5.01-20 cm). Ten-gram samples from each depth were separated after cutting crosswise and homogenizing, and then stored in a refrigerator overnight prior to extraction. The remaining soil was oven-dried at 105˚C for twenty-four hours to calculate gravimetric water content in the samples. The 10g samples were then combined with 40.0 mL of a 2.0 mol/L potassium chloride (KCl) solution, shaken for one hour (via shaker table), and chilled for one hour at 1.7 – 3.9˚C to limit any additional reactions within the slurry. Soil slurries were then filtered (Whatman 42 filter paper) using gravity filtration into plastic 20 mL sample vials, and frozen until analysis at -12 to -17˚C. Samples were analyzed for ammonium, nitrate/nitrite, and total nitrogen concentrations (ppm) using a Lachat Quickchem 8500 Flow Injection Analysis System (Hach, Loveland CO) in the USFS Northern Research Station chemistry laboratory in Grand Rapids, MN. Following analysis, concentrations were corrected for soil mass (adjusted based on oven dried mass) and converted to units of milligrams per kilogram of soil.

Laboratory incubation

A laboratory incubation using field soils was performed to determine the effect of varying moisture levels on heterotrophic respiration under a controlled environment. Four subsamples of soil were taken from each plot to a depth of 15 cm and combined to produce a bulk soil sample for each of the twenty-four plots. The bulk soil samples were air-dried for one month and then sieved through a 2mm mesh. Three 10.0g subsamples of soil were taken from each bulk soil sample (n = 72). To each subsample, 2.5 mL, 5.0 mL, and 7.5 mL of deionized water was added, respectively.

Soils were then incubated for fourteen days in 237 mL glass jars inside a 20˚C growth chamber in the absence of light. The lids remained sealed during the incubation, but the jars were opened every three days for three minutes to maintain an aerobic environment. Two HOBO U23-002 temperature loggers (±0.2˚C; Onset Computer Corporation, Bourne MA) were placed within the growth chamber and recorded air temperature every fifteen minutes (one on the top shelf and one on the bottom shelf). The average temperature for the top shelf was 20.4˚C ± 0.05˚C and 19.9˚C ± 0.144˚C for the bottom shelf.

Three days prior to sampling, the vials were evacuated using ultra high purity helium (He). Gas samples (12 mL) were collected on the seventh and fourteenth days of the incubation. To begin, jars were opened and allowed to equilibrate with the atmosphere. After 1 minute and 30 seconds, a time-zero (T0) gas sample was taken to represent ambient conditions. Gas samples were immediately transferred with a needle from the syringe to 9mL glass vials sealed with butyl rubber septa. Following the T0 measurement, jars were resealed with a lid and septa and allowed to incubate for 1 hour. Gas samples were then taken from each jar after 1 hour (T1) through the septa and immediately transferred to the vials. Gas samples were analyzed within 24 hours using a gas chromatograph (Model 5890, Agilent/Hewlett-Packard, Santa Clara, CA) in conjunction with an autosampler (Tekmar 7000, Teledyne Tekmar, Mason, OH). The gas chromatograph was equipped with a thermal conductivity detector for CO_2, flame ionization detector for methane (CH_4), and electron capture detector for nitrous oxide (N_2O). Fluxes ($\mu g\ g^{-1}\ h^{-1}$) of CO_2, N_2O, and CH_4 were calculated from the T0 and T1 measurements.

Vegetation community surveys

The vegetation community within each plot (n = 24) were quantified in July 2021. Cover to the nearest 5% of herbaceous species and count of tree species were recorded in

0.25 m^2 subplots in each quadrant of all plots. Cover values were then averaged to obtain a plot-level average for cover of herbaceous species.

Species richness was calculated for all plots as the total number of species per plot. Shannon's Diversity Index was also calculated for each plot, where H' is Shannon's Diversity Index, n_i is the number of individuals per species in each plot, and N is the total number of species per plot (Equation 2.1).

Equation 2.1

$$H' = -\sum\left(\frac{n_i}{N} \times \ln\frac{n_i}{N}\right)$$

Data analysis

Soil respiration fluxes (µmol CO_2 m^{-2} s^{-1}) were calculated from *in situ* CO_2 point measurements using Soil Flux Pro (Version 4.2.1, LI-COR Biosciences, Lincoln NE). Extractable nitrogen concentrations were converted to a mass-per-mass basis (mg N/kg soil) and corrected for water content from oven-dried samples.

The datasets were checked for outliers using plots of residuals and boxplots. The only dataset with outliers which skewed the distribution were in the extractable nitrogen dataset from 2021. Extreme outliers were assumed to be from sample contamination and were removed from the dataset.

Repeated measures, linear mixed effect models were used to evaluate the influence of drainage class, treatment, and time on soil respiration and extractable nitrogen concentrations. For soil water content, analysis was constrained to the growing season (May – September/October 2019 – 2021) in order to focus on effects of the throughfall exclusion treatment, which was expected to have the most influence on the response of soil water content. For the soil incubation, drainage class, treatment, moisture content, and time were used as factors in repeated measures, mixed effect models. Site (block) was included as a random effect in all models. Soil temperature, pre-treatment carbon (%), and clay content (%) were included as covariates in the model of soil respiration. Pre-treatment nitrogen (%) and clay content were also included as covariates in the ammonium, nitrate/nitrite, and total nitrogen models.

The mixed effect model analysis with repeated measures was performed using the R package "nlme" (Pinheiro et al., 2021). Autocorrelation matrices (corAR1 function) were included in models to account for temporal correlation in the data (Pinheiro et al., 2021). For all analyses, each year was run separately. Least square means analysis with the Tukey p-value adjustment was performed when significant effects were found by using the "lsmeans" package in R (Lenth, 2021).

Plots of standardized residuals and quantile-quantile (Q-Q) plots were used to visually validate the assumptions of normality, linearity, constant variance, and independence. Respiration fluxes were transformed using a natural logarithm to correct for non-normality. Carbon dioxide and nitrous oxide fluxes from the incubation were also log-transformed to correct for non-normality. Quantile-quantile plots and plots of standardized residuals were used to identify the best transformation of the dependent variable. The Akaike Information Criterion (AIC) was used to determine the inclusion of interactions and random effects in the model, with the lower AIC value indicating the stronger model (Arnold, 2010). If transformed to meet the assumptions of linear models, least square means and confidence intervals were presented in original, back-transformed units for interpretation in figures.

Richness and Shannon Diversity Index values were compared using a two-sample test. Equality of variances were checked using the F-test to compare two variances. Linear models and mixed effect models were then used to assess the effect of site, drainage, and treatment on species richness and Shannon's Diversity Index. Site was included as a random effect in the mixed model.

All statistical analyses and data visualizations were performed using R statistical software in RStudio (Version 1.1.463). The level of significance (alpha) was defined as p-value < 0.05.

Results

In-situ bulk soil respiration

There was no effect of treatment on in-situ bulk soil respiration in either year, but there was a significant effect of drainage class in the 2021 study period (Table 2.2). In that year, the well-drained class had a higher respiration rate compared to the other drainage classes that ranged from 3.6 – 4.2 μmol m^{-2} s^{-1} (Figure 2.4).

Pre-treatment carbon and soil temperature were significant predictors of bulk soil respiration only in 2020 (Table 2.2). There was a notable decline by 2.7 μmol m^{-2} s^{-1} in soil respiration rates (CO_2 flux) from 2020 to 2021 (Figure 2.4; Table C.1). Across treatments and drainage class, bulk soil respiration rates decreased by roughly 25% from 2020 to 2021 during the growing season (Table C.1, Figure 2.4). During the same period, there was no significant effect of throughfall reduction on soil water content in the surface soil (Figure 2.5).

Visual examination of the data indicated that there was some evidence of differences in bulk soil respiration during the thaw period (Figure C.1). Bulk soil respiration was suppressed in the wetter drainage classes (SPD and PD) during this period, compared to the WD and MWD classes (significant difference of 1.38 μmol m^{-2} s^{-1} between WD and PD). Notably, soil temperature during early spring/summer (May – June) was lower in the treatment compared to ambient conditions in all three years (difference of 3.3 – 4.4˚C during first week of May, 2019-2021; Figure A.4). There were also significant differences in bulk soil temperature between drainage classes, with the wetter drainage classes having higher soil temperatures compared to the drier drainage classes (difference of 0.21 – 0.75˚C in mean soil temperature during summer 2019-2021; Figure A.5). This pattern suggests that soil temperature may have been influencing bulk soil respiration across the drainage classes during this period. The marginal difference in soil temperature between treatments, however, was not large enough to produce significant differences in bulk soil respiration between treatments during this period ($p = 0.81$).

Table 2.2: Three-way ANOVA results summary for the field bulk soil respiration model. Numerator degrees of freedom and model coefficient p-values are shown. Bolded values indicate a significant result (p-value < 0.05).

	2020		**2021**	
	06/23 - 09/01		04/07 - 08/24	
Model term	**Degrees of freedom**	**p-value**	**Degrees of freedom**	**p-value**
Intercept	1	**<0.001**	1	**<0.001**
Drainage	3	0.091	3	**0.006**
Treatment	1	0.207	1	0.893
Date	5	**<0.001**	10	**<0.001**
Temperature	1	**0.003**	1	0.599
Percent clay	1	0.477	1	0.670
Pretreatment carbon	1	**0.001**	1	0.350
Drainage:Treatment	3	0.054	3	0.661
Drainage:Date	15	0.375	30	0.087
Treatment:Date	5	0.745	10	0.359
Drainage:Treatment:Date	15	0.991	30	0.940

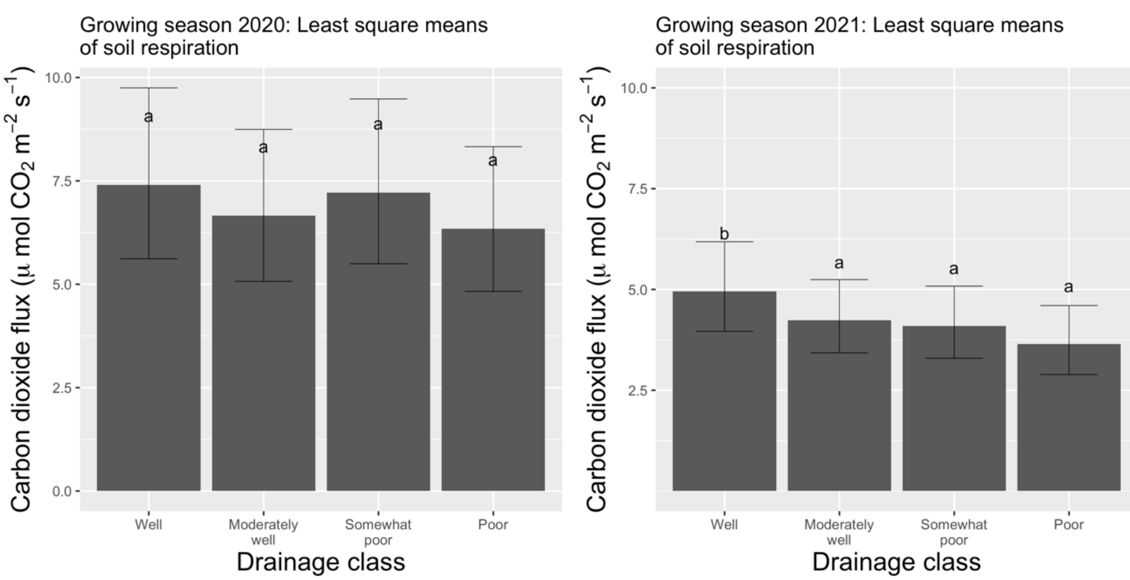

Figure 2.4: Least square mean values of bulk soil respiration by drainage class for 2020 and 2021. Letters represent significant differences between drainage classes within a specific year (p-value < 0.05). Error bars represent 95% confidence intervals.

Notably, there were limited effects of throughfall reduction across drainage classes on soil moisture in the surface horizons (Figure 2.5). Trends between treatments within a drainage class were inconsistent, and differences between treatments were often insignificant.

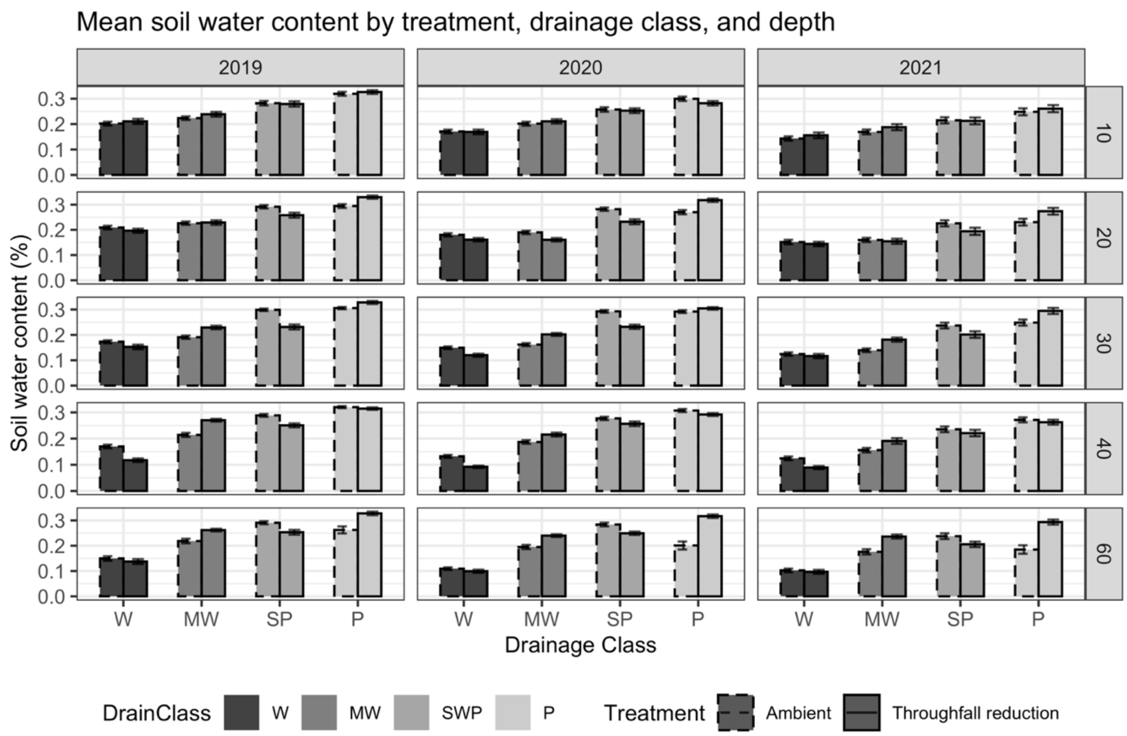

Figure 2.5: Three-way interaction between treatment, drainage class, and sensor depth for the three years of soil water content models. Sensor depth in centimeters in shown on the right y-axis.

Extractable nitrogen

Concentrations of extractable nitrogen were mainly influenced by drainage class (statistics presented in Table 2.3). Drainage class significantly affected total nitrogen (TN) concentrations during 2020 and 2021 ($p < 0.001$ for both years). Drainage class was a significant factor in the ammonium (NH_4^+) model during 2020 and was not significant in 2021 ($p = 0.014$, $p = 0.372$ respectively). Nitrate/nitrite (NO_3^-—NO_2^-) was affected by drainage class in both 2020 and 2021 ($p < 0.001$ for both years).

Treatment was not a significant factor in any of the models in either year (Table 2.3). There were no significant interactions between drainage class and treatment or any other factors in the models. (Table 2.3).

Table 2.3: Three-way ANOVA results for total nitrogen, ammonium, and nitrate/nitrite models. Pre-treatment nitrogen and percent clay were included as covariates in the models. Bolded values indicate a significant result (p-value < 0.05).

	Total nitrogen		**Ammonium**		**Nitrate/nitrite**	
	2020	**2021**	**2020**	**2021**	**2020**	**2021**
Model term	**p-value**	**p-value**	**p-value**	**p-value**	**p-value**	**p-value**
Intercept	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
Drainage	**<0.001**	**<0.001**	**0.014**	0.372	**<0.001**	**<0.001**
Treatment	0.096	0.954	0.782	0.364	0.170	0.080
Date	0.139	**<0.001**	**<0.001**	**<0.001**	**<0.001**	**<0.001**
lnpretrtN	**0.008**	0.094	0.548	0.081	0.984	0.322
percentClay	0.797	0.545	0.159	0.831	0.458	**0.016**
Drainage:Treatment	0.222	0.053	0.632	0.569	0.068	0.495
Drainage:Date	0.662	0.067	0.423	**0.025**	0.296	0.413
Treatment:Date	0.331	0.347	0.918	0.460	0.651	0.888
Drainage:Treatment:Date	0.818	0.311	0.915	0.754	0.162	0.763

The main differences in TN concentrations in 2020 and 2021 were driven by higher values in the somewhat poorly drained (SWP) class (Figure 2.6). In 2020, TN concentrations in the SWP class were higher than the W, MW, and P classes by 6.5 mg/kg ($p < 0.001$), 5.6 mg/kg ($p = 0.002$), and 4.2 mg/kg ($p = 0.02$), respectively. In 2021, W was different from SWP (6.9 mg/kg; $p < 0.001$) and P (-5.0 mg/kg; $p = 0.01$). SWP was different from MW (6.0 mg/kg; $p = 0.002$)).

There was a significant increase in TN concentrations from 2020 to 2021 (Table C.1). TN concentrations increased across drainage classes from the growing season of 2020 to 2021 (Table C.1; Figure 2.6). The same pattern was maintained across years, with SWP having the highest mean concentration of TN, then P, MW, and W having the lowest mean concentration (Figure 2.6).

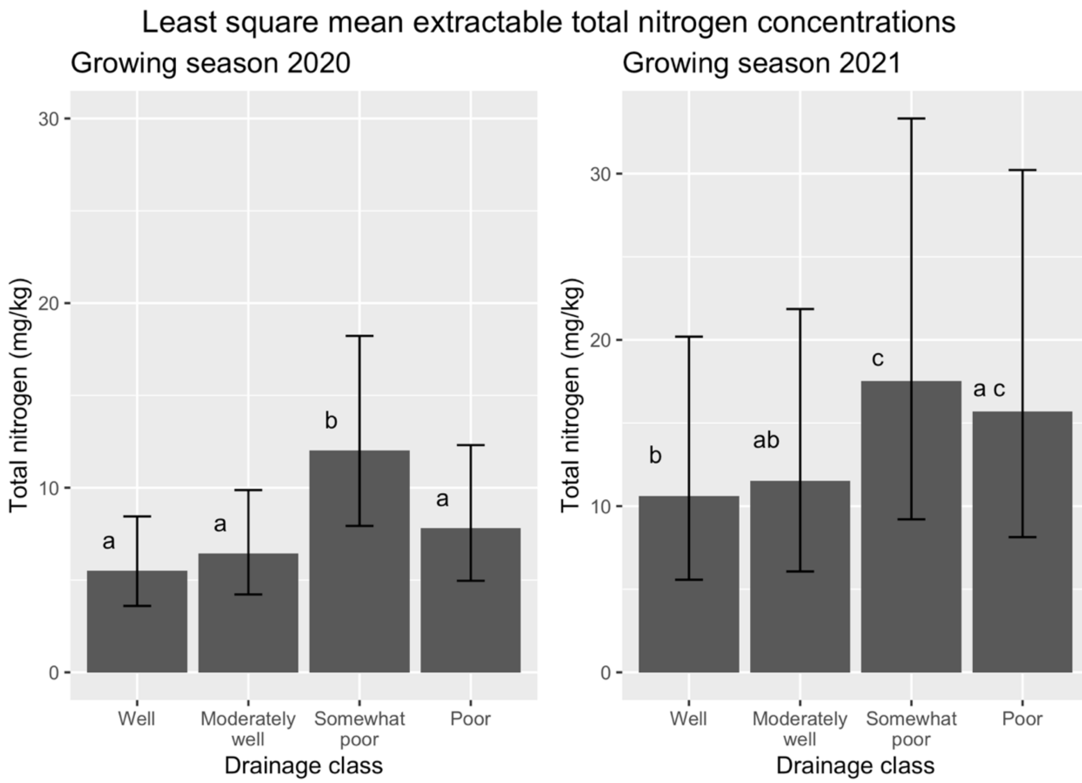

Figure 2.6: Least square mean values for concentrations of extractable total nitrogen by drainage class during growing seasons 2020 and 2021. Letters represent significant differences between drainage classes within each year (p-value < 0.05). Error bars represent 95% confidence intervals.

In 2020, the only significant difference in NH_4^+ concentrations was between the SWP and P class (-6.0 mg/kg; $p = 0.02$) (Figure 2.7). There were no other significant differences between drainage classes. In the significant interaction between drainage and date in 2021 ($p = 0.025$), the only significant differences in NH_4^+ concentrations were between WD 2021-05-06 and WD 2021-06-03 (-18 mg/kg; $p < 0.001$), and WD 2021-05-06 and WD 2021-08-26 (-6.96 mg/kg; $p = 0.02$)

There was a significant increase in NH_4^+ concentrations from 2020 to 2021 (Table C.1). The increase in NH_4^+ concentrations from 2020 to 2021 reflects the increase in TN concentrations between the two growing seasons.

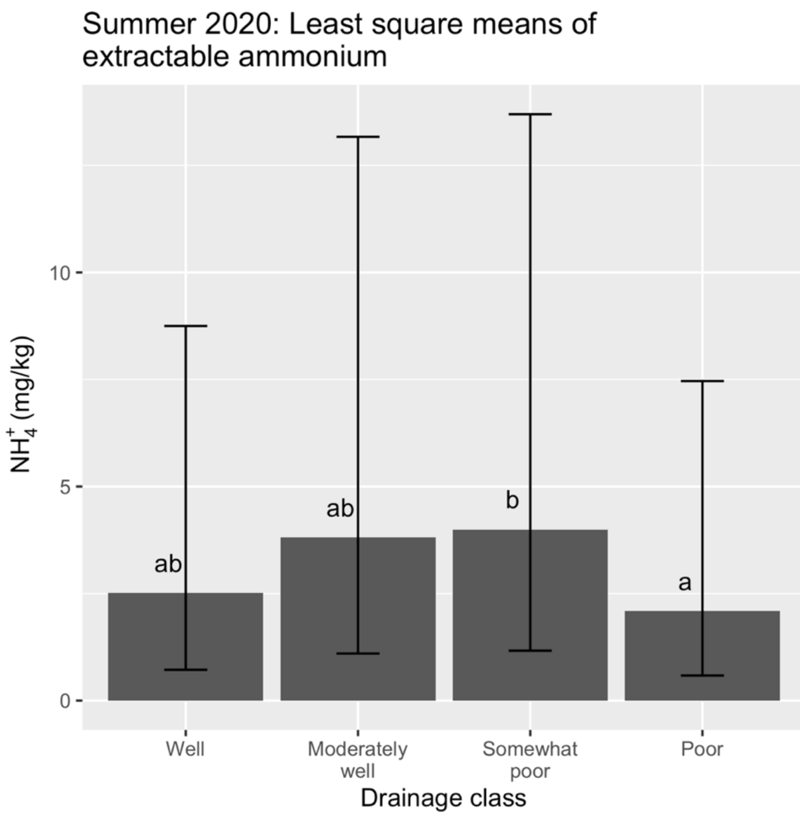

Figure 2.7: Least square mean values for concentrations of extractable ammonium by drainage class during growing season 2020. Letters represent significant differences between drainage classes (p-value < 0.05). Error bars represent 95% confidence intervals.

In 2020, only the well-drained class was significantly different and had the lowest concentration of NO_3^-+NO_2^--N compared to all other drainage classes (1.2 – 1.6 mg/kg). Similarly in 2021, the well-drained class also had a significantly lower concentration of NO_3^-+NO_2^--N (compared to 0.7 – 3.9 mg/kg), though the mean is much smaller in 2021 than in 2021 (Figure 2.8). Additionally, NO_3^-+NO_2^--N concentrations in the SWP class during 2021 were significantly higher than the MW class (3.1 mg/kg; $p < 0.001$), but not different than P. The pattern in NO_3^-+NO_2^--N concentrations during 2021 mirrors that of TN concentrations (Figure 2.6), with SWP having the highest concentration of NO_3^-+NO_2^--N, then P, MW, and W having the lowest concentration.

There was a significant increase in nitrate/nitrite concentrations from 2020 to 2021 (Table C.1).

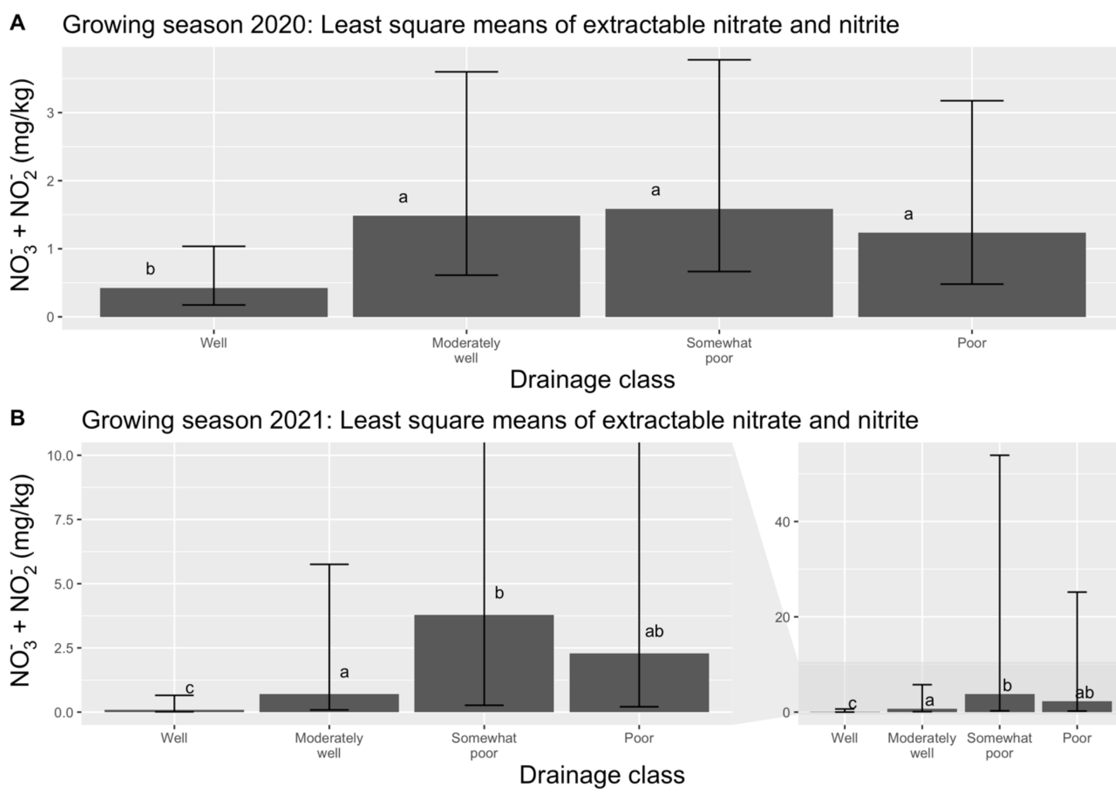

Figure 2.8: Least square mean values for concentrations of nitrate/nitrite by drainage class during growing seasons 2020 (panel A) and 2021 (panel B). Letters represent significant differences between drainage classes within each year (p-value < 0.05). Error bars represent 95% confidence intervals. In the bottom panel, the figure was expanded (original scale 0-60 mg/kg) due to the large error bar making it difficult to discern the differences between means.

Vegetation communities

There were few clear patterns in species richness. Control plots had a higher species richness than treatment plots for all but the St. Louis County sites (Table 2.5).

The somewhat-poorly drained class had the highest species richness, followed by WD, MWD, and then PD (Table 2.4). There were also few consistent trends in Shannon's Diversity Index (SDI) across the plots (Table 2.5). There were no significant differences in species richness or diversity between treatments, or among drainage classes or sites (Table D.1).

Table 2.4: Species richness for all plots.

	Aitkin		Itasca		St. Louis	
	Control	**Treatment**	**Control**	**Treatment**	**Control**	**Treatment**
W	13	12	10	11	8	13
MW	9	11	13	11	7	9
SWP	10	9	15	15	11	10
P	12	6	10	9	11	11
Total	**44**	**38**	**48**	**46**	**37**	**43**

Table 2.5: Shannon's Diversity Index for all plots.

	Aitkin		Itasca		St. Louis	
	Control	**Treatment**	**Control**	**Treatment**	**Control**	**Treatment**
W	2.87	2.83	2.93	2.69	2.27	2.92
MW	2.04	2.95	2.96	2.52	2.07	2.65
SWP	2.88	2.62	3.16	3.25	2.85	2.29
P	2.76	1.93	2.80	2.47	2.80	2.63

Laboratory incubation

The results of the soil incubation were generally consistent with the field results with respect to soil respiration (CO_2 flux). Drainage class, the amount of water added (H_2O added), and sample date (week one or week two of gas sampling) were significant main effects in the model of CO_2 fluxes. There was a significant interaction between the amount of water added and sample date. As with the field results, there was no significant effect of treatment on CO_2 flux (Table 2.6).

Only the moderately well drained class had a significantly lower CO_2 flux compared to the other drainage classes (Figure 2.9; $p = 0.01$ MW – SWP, $p = 0.01$ MW - W). The mean CO_2 fluxes during the incubation were similar to those observed in the field during 2020 and 2021, although the incubation confidence intervals were larger than those from the *in situ* respiration means (Figures 2.4, 2.9).

CO_2 fluxes significantly increased with the volume of water added, and this pattern was seen during both sampling periods (Figure 2.9; Table 2.6). However, during the second week of the two-week incubation, each level of water added was significantly different (the 5.0 mL and 7.5 mL levels were not significantly different during week one). CO_2 fluxes decreased significantly from the first to second week of the incubation ($p < 0.001$; >50% reduction).

Table 2.6: Four-way ANOVA results for carbon dioxide, methane, and nitrous oxide models. Numerator degrees of freedom and model coefficient p-values are shown. Bolded values indicate a significant result (p-value < 0.05).

		Carbon dioxide	Methane	Nitrous oxide
Model term	**Degrees of freedom**	**p-value**	**p-value**	**p-value**
Intercept	1	**<0.001**	0.150	**0.007**
Drainage	3	**0.004**	0.791	0.057
Treatment	1	0.438	0.073	0.642
H2O added	2	**<0.001**	0.311	**<0.001**
Sample date	1	**<0.001**	**0.004**	**<0.001**
Drainage:Treatment	3	0.740	0.370	0.259
Drainage:H2O added	6	0.949	0.908	0.117
Treatment:H2O added	2	0.240	0.671	0.740
Drainage:Sample Date	3	0.189	0.424	**0.034**
Treatment:Sample Date	1	0.956	0.560	0.466
H2O added:Sample Date	2	**<0.001**	0.585	**<0.001**
Drainage:Treatment:H2O added	6	0.917	0.568	0.645
Drainage:Treatment:Sample Date	3	0.942	0.635	0.168
Drainage:H2O added:Sample Date	6	0.974	0.812	**0.013**
Treatment:H2O_added:Sample Date	2	0.561	0.984	0.676
Drainage:Treatment:H2O added: Sample Date	6	0.918	0.850	0.876

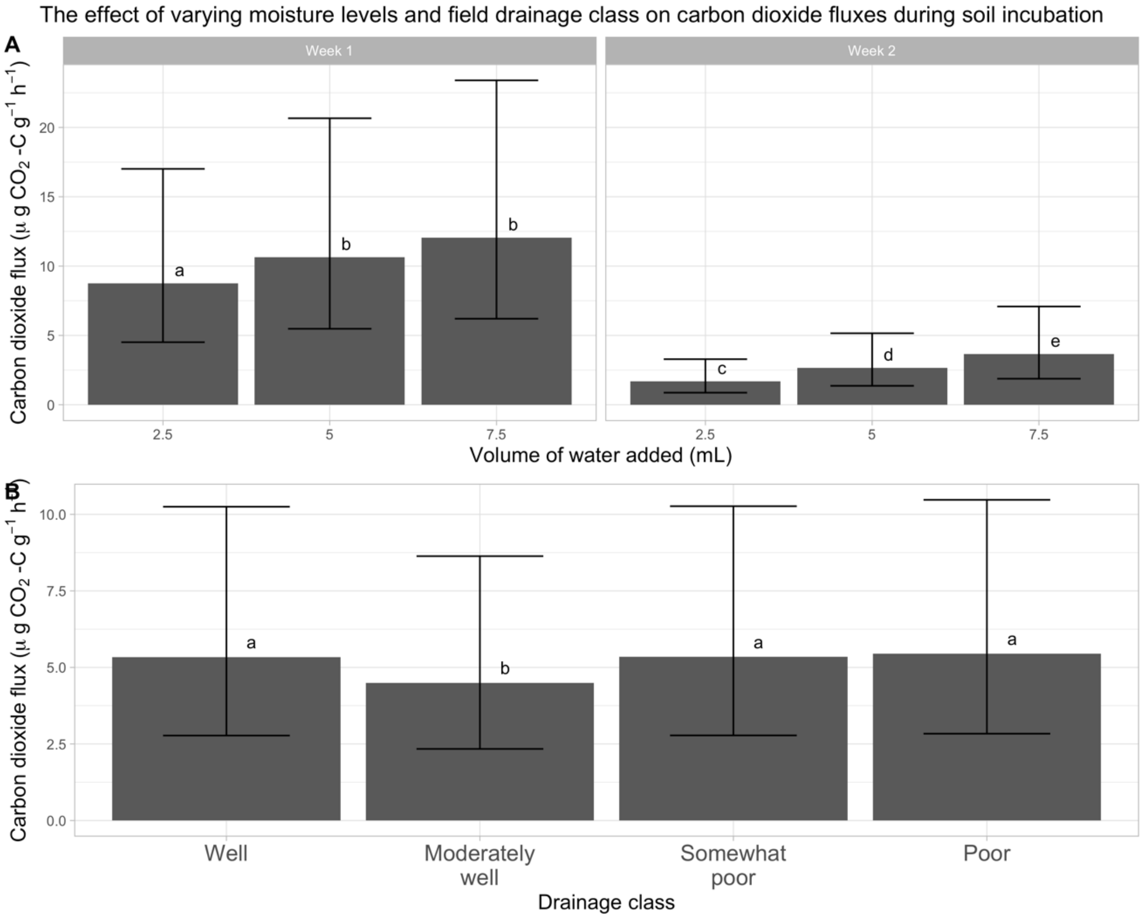

Figure 2.9: Least square mean values of CO_2 flux for pairwise comparisons of volume of water added by week are shown in panel A. Least square means of CO_2 flux by drainage class are shown in panel B. Letters represent significant differences (p-value < 0.05). Error bars represent 95% confidence intervals.

Methane fluxes differed by sample week and there were no significant interactions with any of the main effects (Table 2.6). CH_4 fluxes were extremely small, which was expected due to the aerobic field conditions of these soils, and likely the lack of a dominant methanogen community. The CH_4 flux changed from CH_4 production (positive flux) to consumption (negative flux) from the first to second week of the incubation (Figure 2.10). This pattern mirrors the overall decrease in CO_2 fluxes during the two-week incubation.

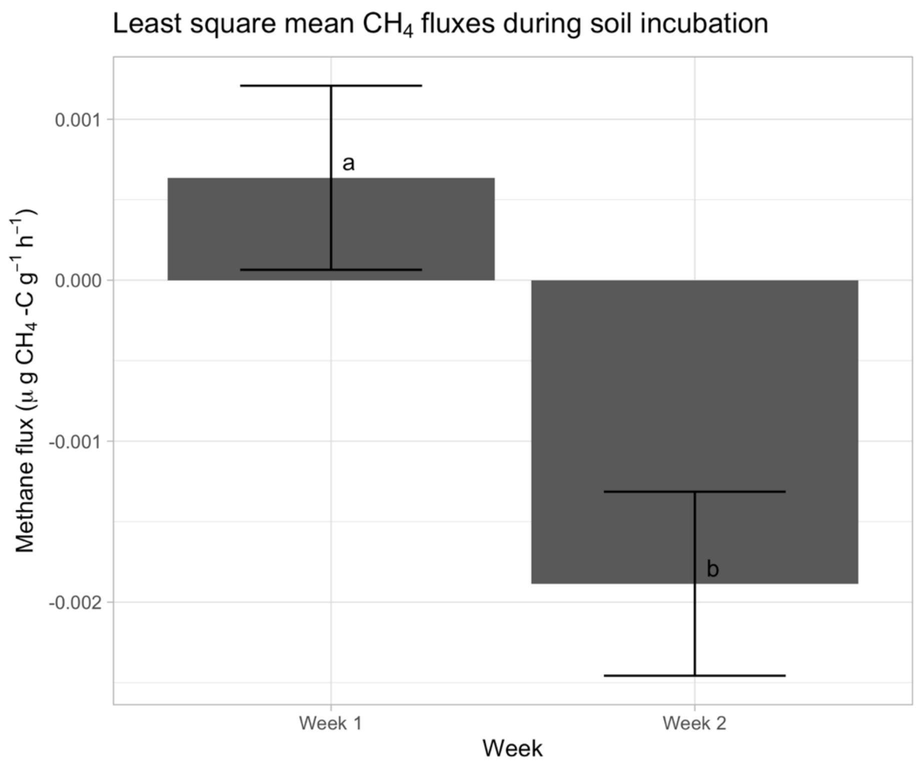

Figure 2.10: Least square mean values of CH4 flux by sample date. Letters represent significant differences between the two sample weeks (p-value < 0.05). Error bars represent 95% confidence intervals.

In terms of N_2O fluxes, there was a significant three-way interaction between drainage class, volume of water added, and sample date (Table 2.6). Like CH_4, fluxes of N_2O were extremely low (Figure 2.11). There were no significant differences in week one. In week two, the highest N_2O fluxes occurred in the wetter drainage classes with the highest volume of water added (7.5 mL) (Figure 2.11). Overall, N_2O fluxes increased from week one to week two, which contrasts to the pattern seen in the CO_2 and CH_4 models (Figures 2.9, 2.11).

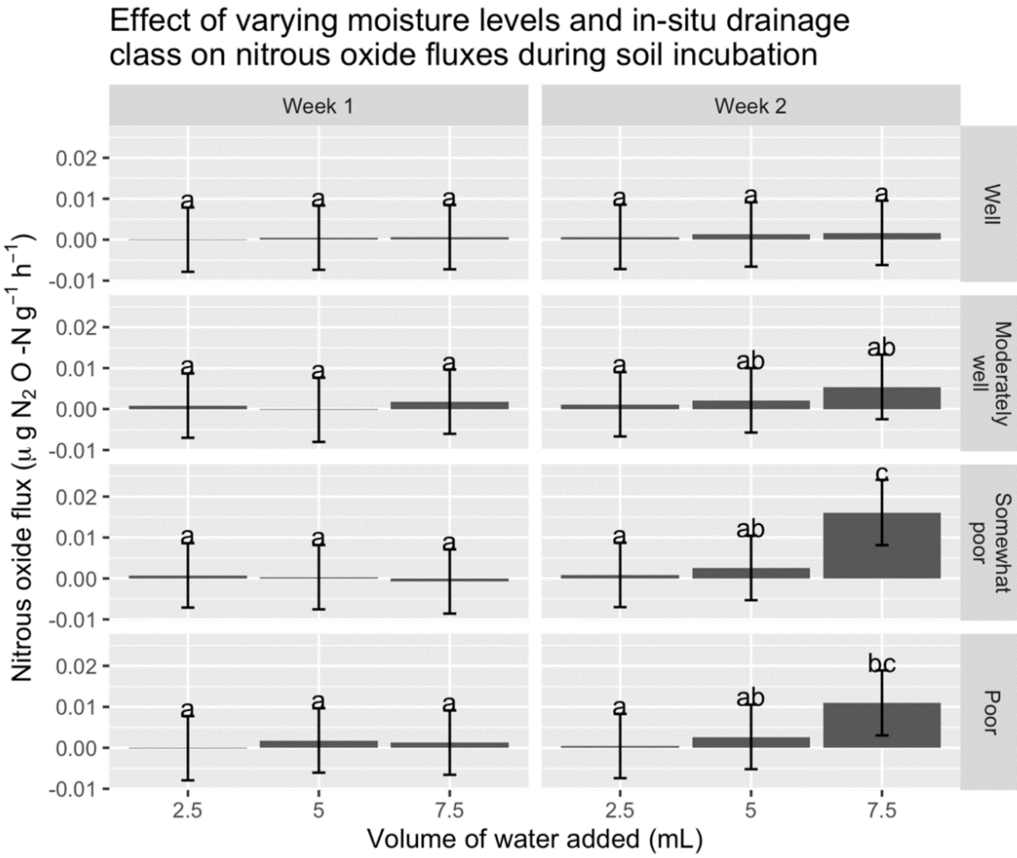

Figure 2.11: Least square mean values of N_2O flux for the three-way interaction of drainage class, volume of water added, and sample date (week). Letters represent significant differences in the three-way interaction (p-value < 0.05). Error bars represent 95% confidence intervals.

Discussion

Due to the fundamental influence of soil moisture and temperature on biogeochemical processes in forest soils, evaluating the response of those processes to changes in precipitation and temperature is crucial for understanding the long-term response of forest soils under a changing climate. I used a field study to simulate reduced summer and winter precipitation and found minimal effects of throughfall reduction and snow removal on soil respiration and extractable nitrogen concentrations. Drainage class was a stronger indicator of soil respiration and extractable nitrogen concentrations than treatment, but the effects were still limited. The responses are likely due to the limited effect of the throughfall reduction treatment on soil temperature and soil moisture. Results from the laboratory incubation support field measurements as there were significant effects of soil moisture content on carbon dioxide and nitrous oxide fluxes, but no effect of treatment under controlled conditions during the incubation. Finally, the lack of any differences in vegetation communities (richness, diversity) between the drainage class or treatments indicates that differences in vegetation did not mask or offset observed responses.

Bulk soil respiration and incubation

The treatment (summer throughfall reduction followed by winter snow removal) did not result in significant differences in *in situ* bulk soil respiration during either year of the study. These results contrast with previous studies that have shown a decline in soil respiration with throughfall reduction (Borken et al., 2006b; Schindlbacher et al., 2012). However, those studies excluded all throughfall from entering the plots (similar in size to plots in the current study) to simulate severe drought, resulting in SWC reductions in surface soils (Borken et al., 2006b; Schindlbacher et al., 2012). In contrast, our 50% reduction in throughfall in the current study had inconsistent effects on SWC across drainage classes, with the treatment plots being wetter at times compared to the control plots (Figure 2.5). Regardless of the mechanism, the lack of a consistent effect of treatment on SWC is likely why there was no significant effect of treatment on soil respiration. Experimental artifacts may have contributed to the insignificant treatment effect on SWC. For example, the small size of the plots may have limited treatment efficacy to modify SWC within the plots or created an edge effect from precipitation entering via the sides of the shelter (Beier et al., 2012; Fay et al., 2000). Additionally, the 50% reduction of throughfall may have not been sufficient to produce a significant effect on respiration, in contrast to studies with complete throughfall exclusion (Borken et al., 2006b; Schindlbacher et al., 2012).

Snow removal resulted in a temperature lag, where soil temperature was significantly lower in treatment plots compared to the control from May – June in all three years (Figure A.4). However, there were no significant differences in soil respiration between ambient and treated conditions during this period, despite the strong relationship between soil temperature and soil respiration (Davidson et al., 1998; Muhr et al., 2009; Schindlbacher et al., 2012; Wei et al., 2010). For example, in early May 2021, mean soil temperature was significantly lower in the treatment plots (3.0˚C) compared to the control (4.7˚C) (Figure A.4). There may have been confounding effects with SWC during this period, such as high SWC due to snowmelt in the plots may have suppressed soil respiration and negated any treatment effect. Standing water was present due to inundated soils during the 2021 spring thaw period in the SPD and PD classes in the Aitkin and St. Louis sites (personal observation). During the spring thaw period, the wetter drainage classes experienced delayed warming due to high SWC, which supports the suppression of soil respiration observed during the thaw period in the SPD and PD classes during the early spring in 2021 (Figure C.1).

The effect of drainage class on soil respiration in 2021was minimal as only the well-drained class had a higher respiration rate compared to the other three drainage classes (Figure 2.4). Davidson et al. (1998) showed that soil respiration generally decreased with increasing SWC (from well-drained to poorly-drained), but that soil respiration was also suppressed at low values of SWC ($\theta_v < 0.12$) under drought conditions. This effect generally agrees with the patterns in soil respiration shown in this study. The lack of more pronounced and consistent effects of drainage class may be due to interactions between SWC and soil temperature on soil respiration which may confound any response, though was surprising since pre-treatment carbon content significantly increased from WD to PD (Davidson et al., 1998; Figure E.1). Since I aimed to not only manipulate SWC (throughfall reduction), but also soil temperature and frost (snow removal), there may have been confounding effects between these two treatments on soil respiration. For example, the snow removal treatment caused significantly deeper frost penetration, which resulted in a longer thaw period compared to the control and may have suppressed soil respiration during the spring, which has been reported by previous studies (Chapter 1; Monson et al., 2006; Oquist & Laudon, 2008). Previous research has shown that reductions in SWC decreases the sensitivity of soil respiration to soil temperature, while wetting increases the temperature sensitivity of soil respiration (Guamont-Guay et al., 2006). It is possible that in this study, rapid wetting due to snowmelt during the spring may have offset the effects of drying due to throughfall reduction during the growing season.

The comparison of the laboratory incubation to the field experiment removes the effects of the field microclimate on the response, which allows the direct observation of heterotrophic respiration. The lack of treatment effect under controlled laboratory conditions agrees with the field observations. The lack of treatment effect in the incubation suggests that the treatment did not significantly modify the soil microbial community or other factors that influence C and N efflux (*e.g.*, microbial community composition, litter quality, enzyme activity, etc.) resulting in the lack of significant differences in CO_2, CH_4, and N_2O between the control and treatment (Högberg & Read, 2006; Rennenberg et al., 2009; Schlesinger & Andrews, 2000). The results of the incubation confirm that the lack of treatment effect on *in situ* soil respiration was not due to variation among the plots. Since microbial transformations of C and N are strongly affected by soil moisture and temperature, a treatment effect should have appeared under controlled incubation conditions and without the influence of vegetation (Barba et al., 2016; Davidson et al., 1998; Schlesinger et al., 2016). Therefore, the lack of treatment effect during the incubation is consistent with the results of the field study and suggests that the treatment did not alter the soil microenvironment in such a way that would offset any effect associated with the soil microbial community.

The observed effect of drainage class and volume of water added showed that microbial activity or community composition likely differed by drainage class and responded to changes in SWC (Table 2.9; Figure 2.10). The drainage effect during the incubation was minimal, since only MWD had a significantly lower CO_2 flux which could be due to lower microbial biomass or substrate availability compared to the other drainage classes. In contrast, WD may have had the highest *in situ* respiration rate during 2021 due to higher autotrophic respiration. There was variation in both pre-treatment carbon and nitrogen content by drainage class, with carbon and content increasing from WD to PD, but no differences between the control and treatment plots (Table E.1). However, there were no significant differences in vegetation communities or basal area that may have contributed to variations in autotrophic respiration between treatments and/or among drainage classes. Regardless, the investigation of the differences in autotrophic versus heterotrophic respiration is beyond the scope of this study, but it emphasizes the complexity of soil-microbe-plant interactions across a drainage gradient in forest soils. Additionally, soil respiration increased with the amount of water added, indicating that a limiting water content was not reached in the incubation. This finding may explain why there were limited effects of drainage class observed in the field: the maximum moisture content during the incubation ($\theta_{g\text{-incubation}} = 0.75$, average bulk density of in-situ soils = 1.05 g cm^{-3}, $\therefore \theta_{v\text{-incubation}} \approx 0.71$) was higher than the maximum moisture content observed in the field across all three years ($\theta_{v\text{-field}} = 0.47$). It is important to note, however, that the soils used in the incubation were sieved (2mm), so the water-holding capacity of the sieved soils may have been greater than the in-situ soils.

Soil extractable nitrogen

The lack of any treatment effects on extractable nitrogen concentrations is also likely due to the insignificant effect of throughfall reduction on the SWC of the surface horizons. Since the throughfall reduction treatment did not modify SWC of the surface horizons, the productivity and community composition of N-transforming microbes was likely not affected since microbial transformations of nitrogen are dependent on soil moisture and temperature (Deng et al., 2021; Homyak et al., 2017; Knoepp & Swank, 2002). The response of extractable nitrogen concentrations to reduced precipitation has been shown to be complex and the direction of change is variable. For example, drying may limit the transport of substrates and enzymes via changes in water potential or alter soil structure by disrupting soil aggregates (Borken & Matzner, 2009). A study by Homyak et al. (2017) found that precipitation reduction decreased extractable NH_4^+ but NO_3^- was not affected, potentially due to the increase in microbial mortality and decline in plant uptake of NH_4^+ and declines in the production and consumption of NO_3^- (Homyak et al., 2017). In contrast, Deng et al. (2021) found that both NH_4^+ and NO_3^- concentrations increased with drought in forest ecosystems, potentially due to decreased uptake of N by plants and reduced NO_3^- leaching due to reductions in SWC (Deng et al., 2021). This study differs from past throughfall reduction studies due to the addition of the snow removal treatment and a drainage class gradient. Therefore, there may be confounding effects between the throughfall reduction and snow removal treatments, since increased frost depth due to snow removal may have resulted in an increase in microbial and fine root mortality (Fitzhugh et al., 2001; Groffman et al., 2001; Monson et al., 2006; Muhr et al., 2009). Still, the lack of treatment effect in this study was most likely due to the insignificant effect of throughfall reduction on the SWC of the surface horizons. However, even if there were confounding effects between the treatments, the lack of biogeochemical responses observed in this study still represents the future response of C and N dynamics to reduced precipitation in the future under a changing climate.

There were some effects of drainage class on soil extractable nitrogen concentrations, but these effects were small and not consistent. Even though transformations of N via microbes are dependent on soil moisture, and drainage class clearly influenced SWC (Figure 2.5), the absolute differences in SWC were apparently insufficient to influence extractable nitrogen concentrations. In aerobic mineral soils, concentrations of NH_4^+ typically increase with increasing soil moisture, but mineralization is limited at high moisture levels due to anoxic conditions and at extremely low moisture conditions due to reduced substrate supply and enzyme activity (Porporato et al., 2003). Under favorable conditions (warm and moderate water content), nitrification may occur quickly and thus shift the mineral N balance towards NO_3^- dominance, but denitrification may occur at high soil moisture levels (Porporato et al., 2003). Thus, in this study, NH_4^+ concentrations were expected to increase from WD to PD, assuming that soil moisture in the PD class was not limiting. Since the soil temperature and moisture regimes of these soils are frigid and udic, respectively, one could also expect NO_3^-/NO_2^- to be a smaller component of the mineral N balance due to less ideal conditions for rapid nitrification (Table 2.1; Porporato et al., 2003). This pattern was not observed in the field, but NH_4^+ was a larger component of the mineral N balance compared to $NO_3^-+NO_2^{-=}$-N. Typically, the SPD class had the highest concentrations of TN, NH_4^+, and $NO_3^-+NO_2^{-=}$-N, and the lowest concentrations were observed in WD. This pattern may have been due to an optimal moisture content for microbial activity in SPD, but moisture may have been limiting in PD (too wet) and WD (too dry). The high concentration of $NO_3^-+NO_2^{-=}$-N in the SPD class may also be due to error in the sampling or processing of soil cores, such as sample contamination (Figure 2.5). Although the effect of drainage class was not clear, the use of a drainage class gradient in a combined throughfall reduction and snow removal study has not been implemented in prior studies, and thus provides valuable insight in the context of experimental design and C and N dynamics across varying moisture conditions in a forested landscape.

Future directions

I found an overall decrease in soil respiration and an increase in extractable nitrogen concentrations from 2020 to 2021 (Table C.1). It is important to note that extreme drought occurred in northern Minnesota during the summer of 2021 (U.S. Drought Monitor, 2021). Though the throughfall reduction treatment did not affect the SWC of the surface horizons, the drought may have influenced SWC within the plots since mean August SWC significantly decreased from 0.22 in 2020 to 0.13 in 2021. For example, soil respiration may have decreased from 2020 to 2021 potentially due to drought stress on vegetation causing a decline in root respiration (and potentially heterotrophic respiration as well). Soil extractable N may have experienced an overall increase from 2020 to 2021 due to reductions in plant uptake of N and increase in dead plant and fungal biomass, resulting in higher concentrations of extractable N on the soil matrix. Even though the throughfall reduction treatment did not influence C and N dynamics, the effect of the 2021 drought agrees with previous research on the effect of climate on respiration and soil N content (Bernal et al., 2012; Deng et al., 2021; Rennenberg et al., 2009; Schindlbacher et al., 2012; Wei et al., 2010).

Long-term reductions in precipitation will have an impact on C and N dynamics in northern forest ecosystems, although the magnitude and direction of these effects are still unclear (Bernal et al., 2012; Deng et al., 2021; Rennenberg et al., 2009; Schindlbacher et al., 2012; Wei et al., 2010). For example, concurrent warming will likely modify the responses of C and N dynamics to changes in precipitation. Changes in soil respiration rates may alter the carbon balance of forest ecosystems in northern latitudes, which depends on subsequent changes in photosynthesis due to reduced precipitation and warming (Hoberg & Read, 2006; Schlesinger & Andrews, 2000). Since nitrogen is often the limiting nutrient in forest ecosystems, changes in mineral N could result in changes in competition for N between microbial and plant communities (Rennenberg et al., 2009). Therefore, these effects are unclear due to the complex response of N dynamics to reduced precipitation, as well as the predicted concurrent warming.

These results suggest that the response of soil respiration and N dynamics under reduced precipitation scenarios is complex but may not be greatly affected by changes in throughfall, assuming no additional changes occur. Additional effects of climate change, such as warming, may alter the response of C and N dynamics. Further studies are required to identify the mechanisms that control C and N dynamics under reduced precipitation scenarios across drainage classes in forest ecosystems, as well as in combination with warming. Understanding these processes is crucial when predicting the future of forest biogeochemical cycling and productivity under a changing climate.

Conclusions

Carbon and N dynamics in northern forest ecosystems are likely to be influenced by projected reductions in summer and winter precipitation since microbial transformation is influenced by both soil moisture and temperature, although the magnitude and direction of this effect remain unclear and will likely vary by drainage class. I found that the combined treatment of throughfall reduction and snow removal had limited impacts on soil respiration and extractable N dynamics in mineral forest soils of northern Minnesota, likely due to the limited effect of throughfall reduction on the SWC of the surface soils. Finally, further studies are needed to evaluate the influence of concurrent warming with reduced precipitation, since warming would likely confound the response of C and N dynamics to reduced summer and winter precipitation over time.

Conclusions

Changes in precipitation, in addition to temperature, in northern latitudes are expected to have implications for the operability and biogeochemical cycling of forest soils. In Chapter 1, I found that throughfall reduction had no impact on the SWC of the surface soil, and the effects of treatment were inconsistent across drainage classes and increasing depth. Snow removal during the winter significantly decreased soil temperature and thus increased frost development in treatment plots compared to the control. There was a strong correlation between frost depth and freezing degree days (FDD) when snow was removed, and this relationship was weaker, although still significant, under ambient conditions. Soil temperature increased and frost development decreased as drainage decreased under both the treatment and control. There were weak and inconsistent relationships between soil strength and SWC across drainage classes, resulting in no significant effect of treatment on soil strength.

In chapter 2, I found a lack of treatment effect on soil respiration and extractable N concentrations during the growing season. There were minimal effects of drainage class on soil respiration and extractable N concentrations. A laboratory incubation with SWC manipulation suggested that throughfall reduction/snow removal potentially did not modify the microbial community, resulting in no significant treatment effect during the incubation. Drainage class and level of SWC manipulation significantly influenced soil respiration. Only the MWD class had a significantly lower soil respiration rate, and soil respiration increased with the amount of water added at the beginning of the incubation. Fluxes of CH_4 and N_2O were extremely small compared to CO_2, and only varied by incubation time (CH_4) and amount of drainage class/water added/incubation time (N_2O). Finally, there were no significant differences in vegetation communities between sites, drainage classes, or treatments, indicating that the lack of significant effects was not due to differences in vegetation communities.

The results of Chapter 1 and 2 suggest that the largest impact of reduced precipitation in northern latitudes will occur during the winter with decreased soil temperature and increased frost development across drainage classes in mineral soils. Increased frost development could improve the accessibility and operability of these soils during the winter but may also lead to a delay in operability during the spring due to a prolonged thaw period. Drainage class could be used to identify which sites may require more time to freeze to a depth sufficient for operation with limited impacts to the soil. Since the relationship between FDD and frost depth is independent of climate change, this relationship can be used to predict the period of time necessary to reach a particular frost depth under future winter climate conditions.

Further study is required, however, since concurrent warming is also predicted alongside reductions in winter and growing season precipitation. Increased extreme precipitation and drought severity are projected using climate change models (Handler et al., 2014). Therefore, climate change projections may be improved through study of warming with extreme wetting. My research also highlights a need for larger plot sizes for throughfall reduction experiments, as well as potentially separating out the two treatments, to reduce the influence of experimental artifact (edge effects) and confounding effects between treatments. As stated previously, warming may alter the responses observed in this study, but these results still demonstrate the effect of reduced precipitation on soil physical and biogeochemical properties (Handler et al., 2014).

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Appendices

Supplementary material not needed in the body of the thesis has been included in the appendix. This includes results that do not necessarily contribute to the research objectives but still provide insight on observed trends and patterns. The following appendices include supplementary results for soil water content and temperature (Appendix A), soil strength (Appendix B), soil respiration and extractable nitrogen concentrations (Appendix C), vegetation communities (Appendix D), and pre-treatment carbon and nitrogen (Appendix E).

Appendix A: Soil water content and temperature

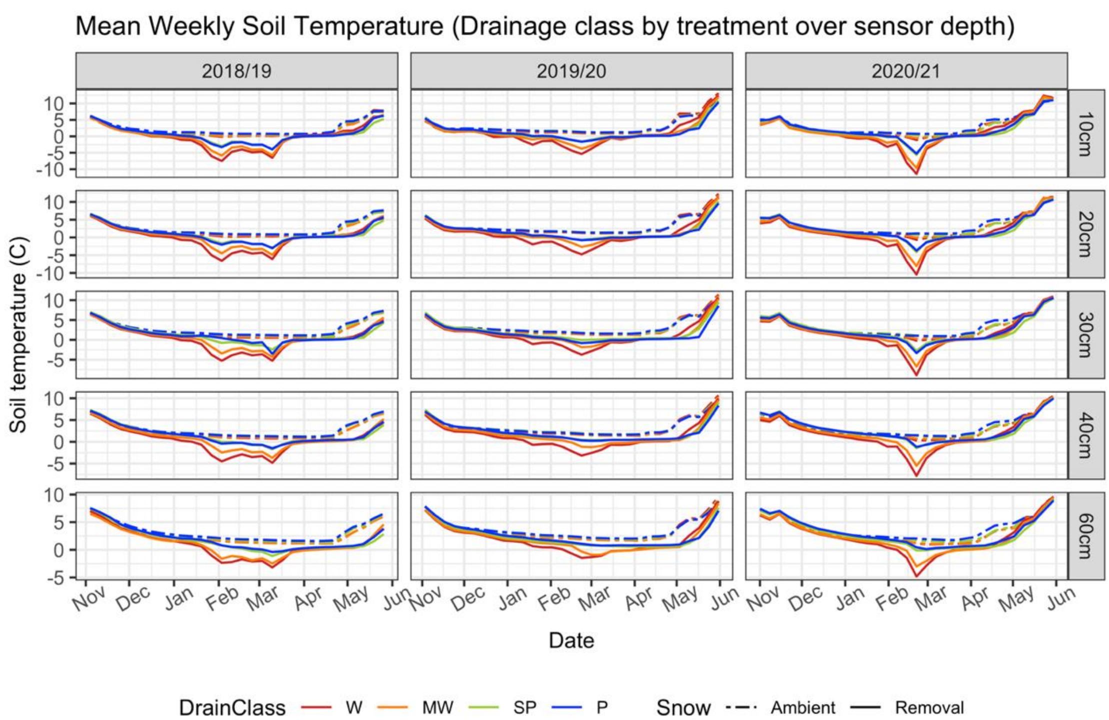

Figure A.1: Mean weekly soil temperature during the winters of 2018/19, 2019/20, and 2020/21 across drainage class, treatment, depth, and week.

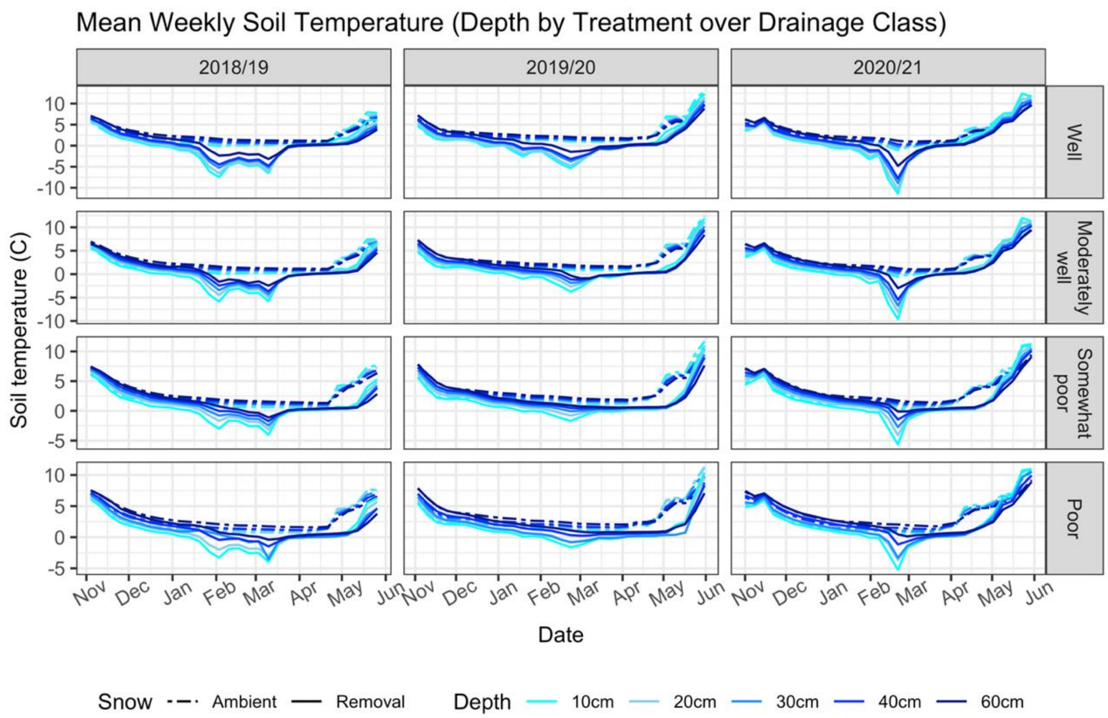

Figure A.2: Mean weekly soil temperature during the winters of 2018/19, 2019/20, and 2020/21 across treatment, drainage class, depth, and time.

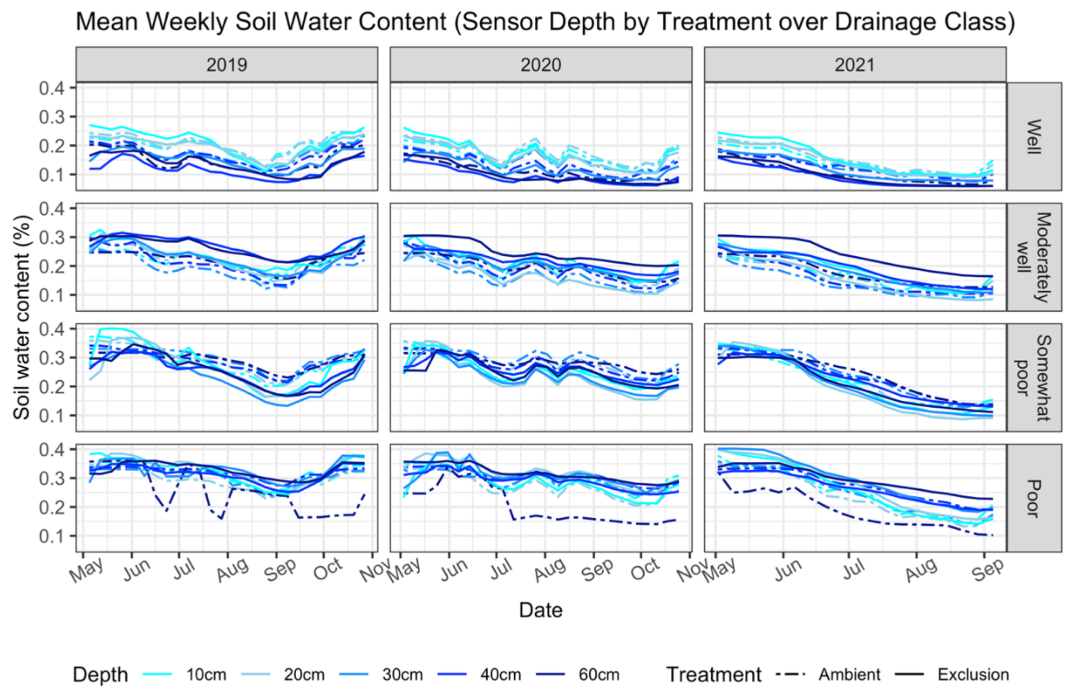

Figure A.3: Mean weekly soil water content during the growing seasons of 2019, 2020, and 2021 by drainage class, treatment, and depth.

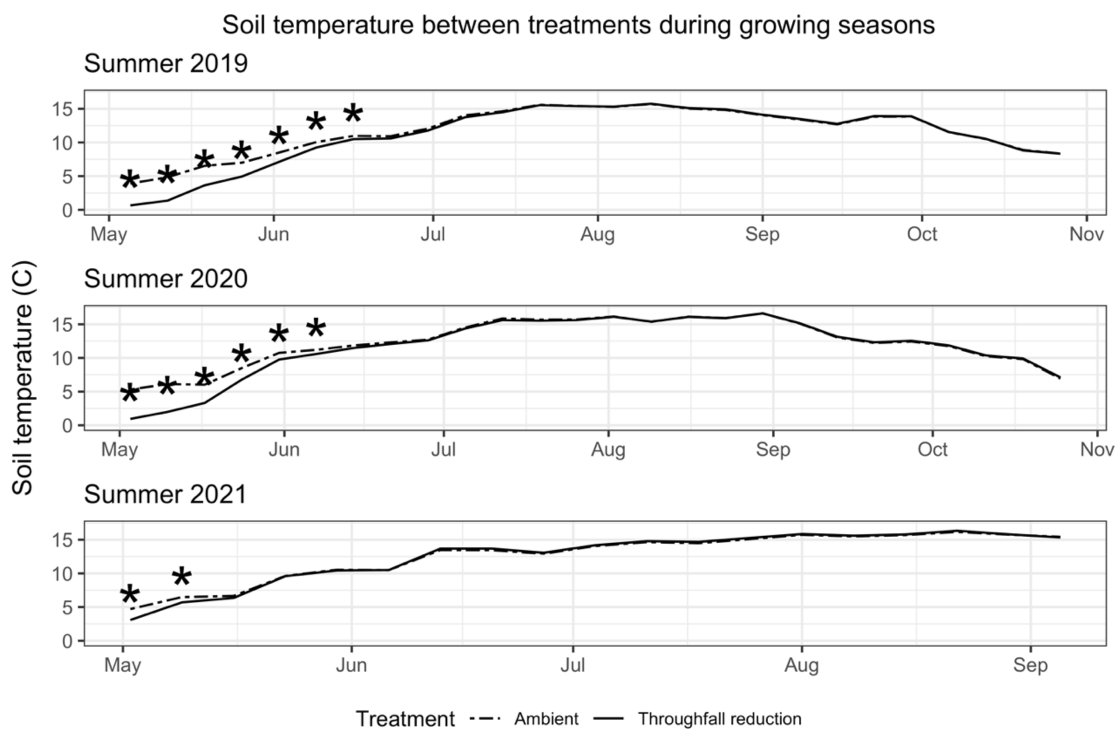

Figure A.4: Mean weekly soil temperature across time during growing seasons of 2019, 2020, and 2021 by treatment. Asterisks indicate weeks with significant difference between means of control and treatment.

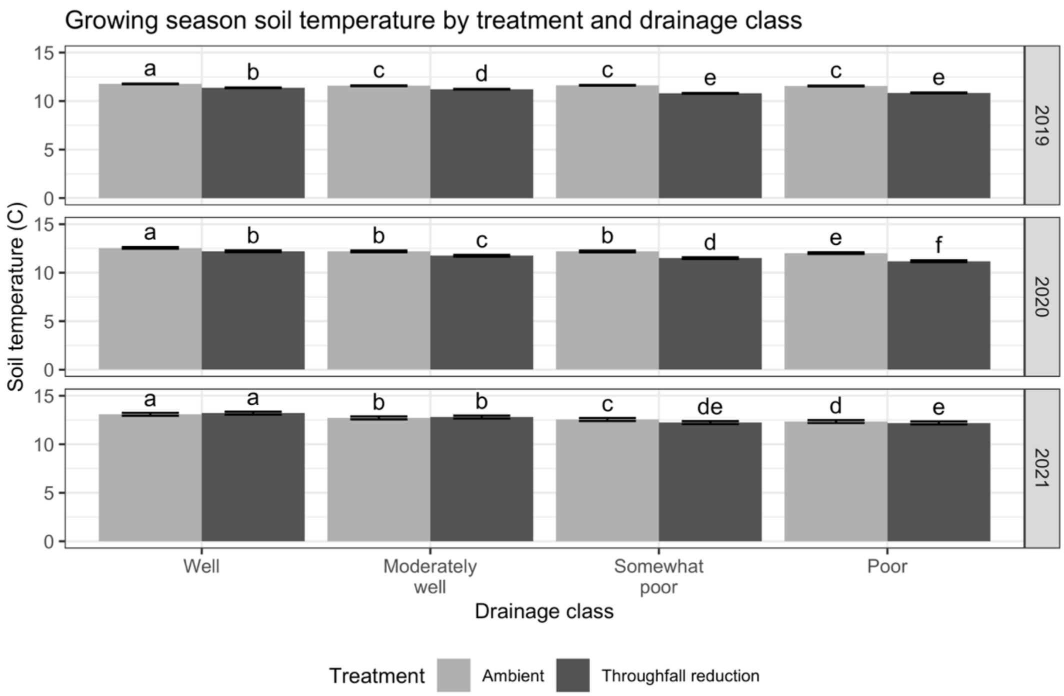

Figure A.5: Mean soil temperature by drainage class and treatment for growing seasons 2019, 2020, and 2021. Letters indicate significant differences within each year.

Appendix B: Soil strength (bearing capacity) on soil water content linear regressions

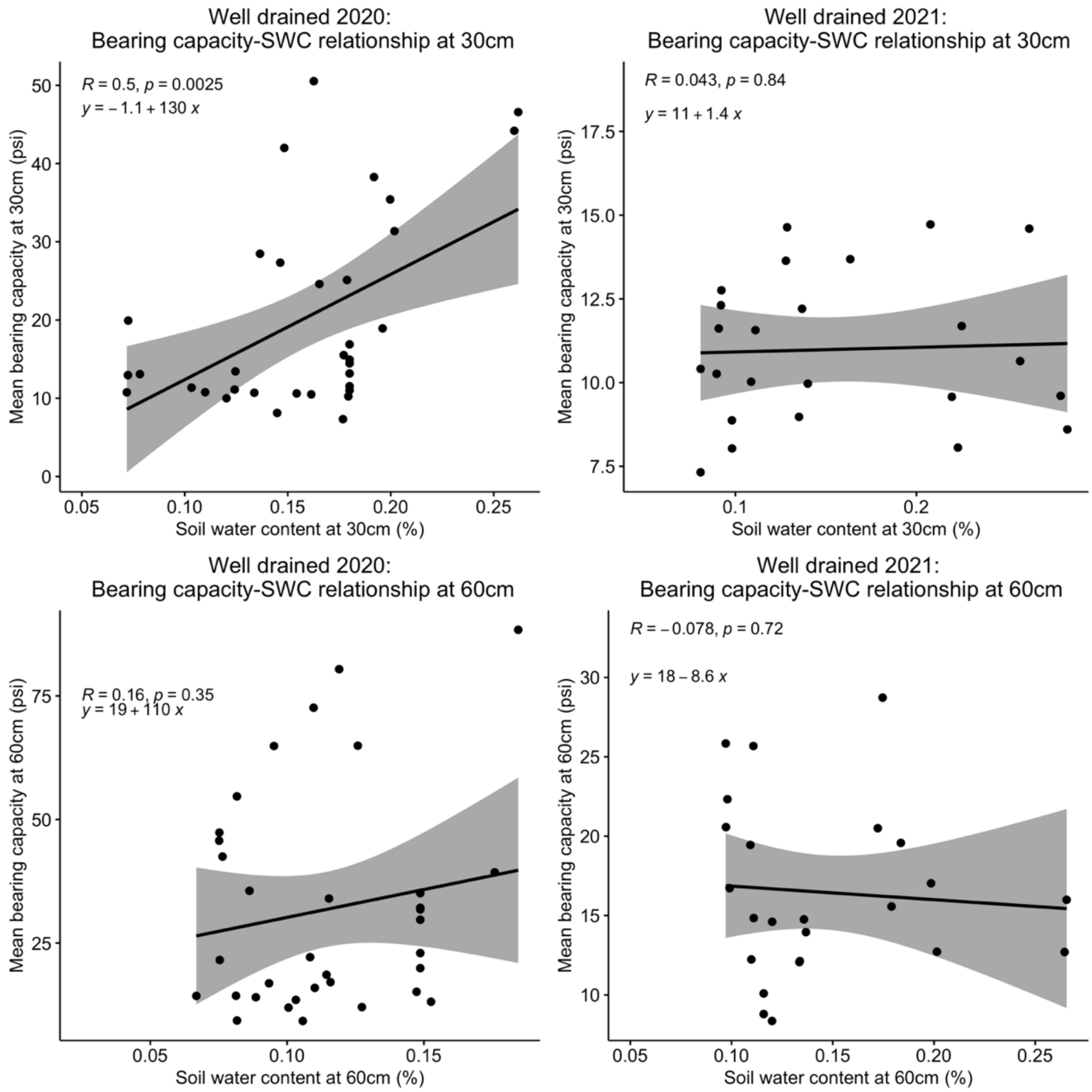

Figure B.1. Linear regressions between soil water content and mean bearing capacity for the well-drained class. Confidence intervals are 95% and level of significance is equal to 0.05.

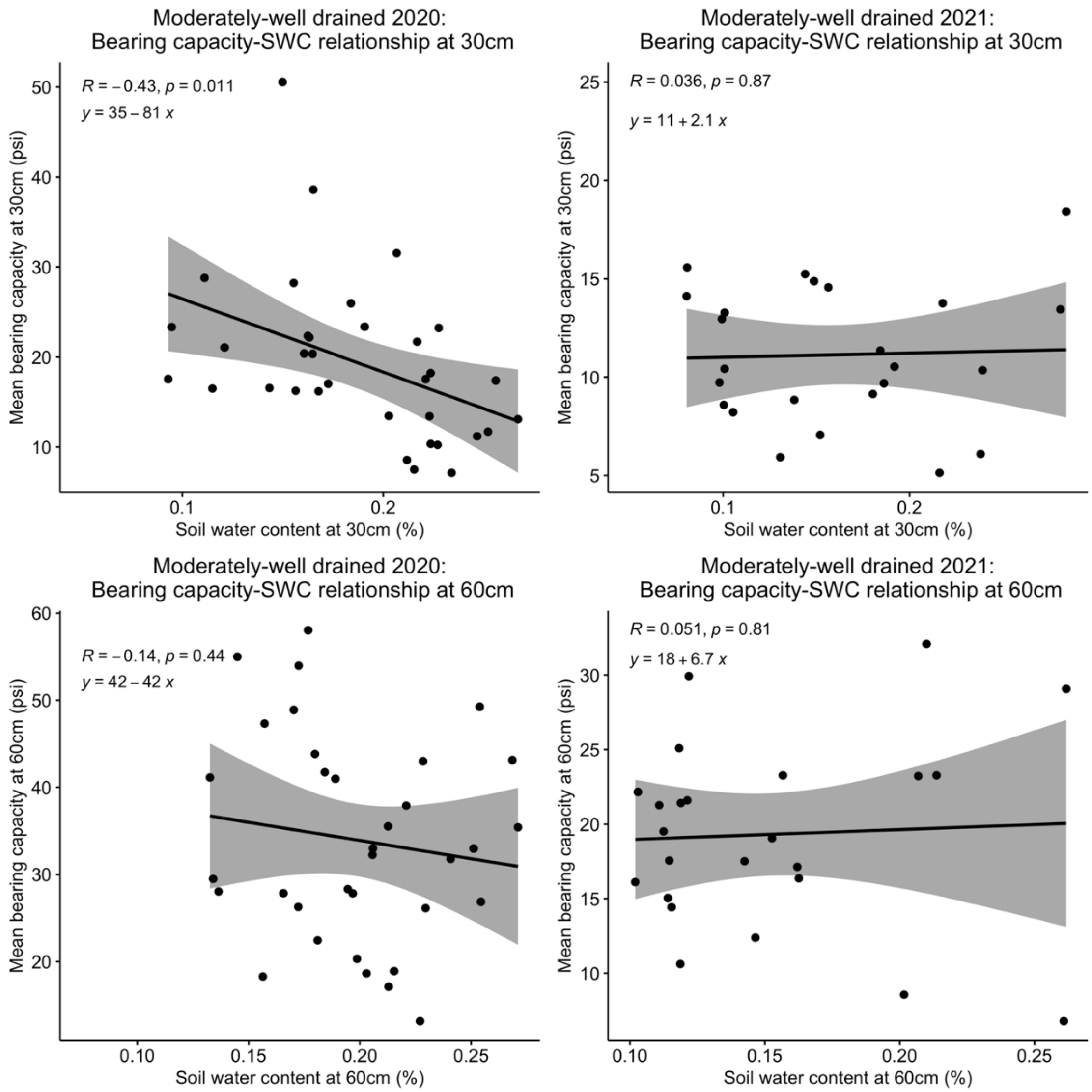

Figure B.2: Linear regressions between soil water content and mean bearing capacity for the moderately well-drained class. Confidence intervals are 95% and level of significance is equal to 0.05.

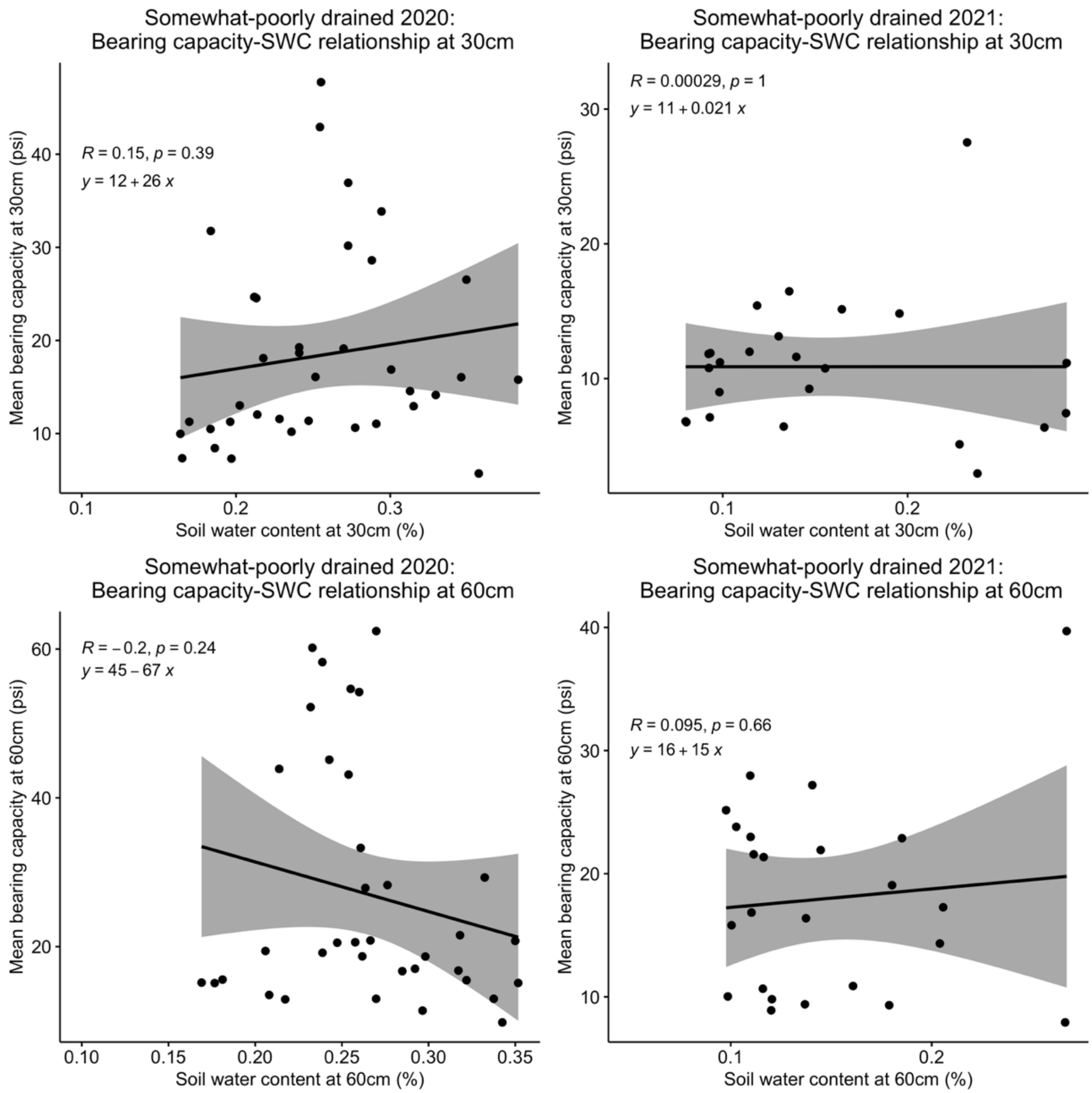

Figure B.3: Linear regressions between soil water content and mean bearing capacity for the somewhat poorly-drained class. Confidence intervals are 95% and level of significance is equal to 0.05.

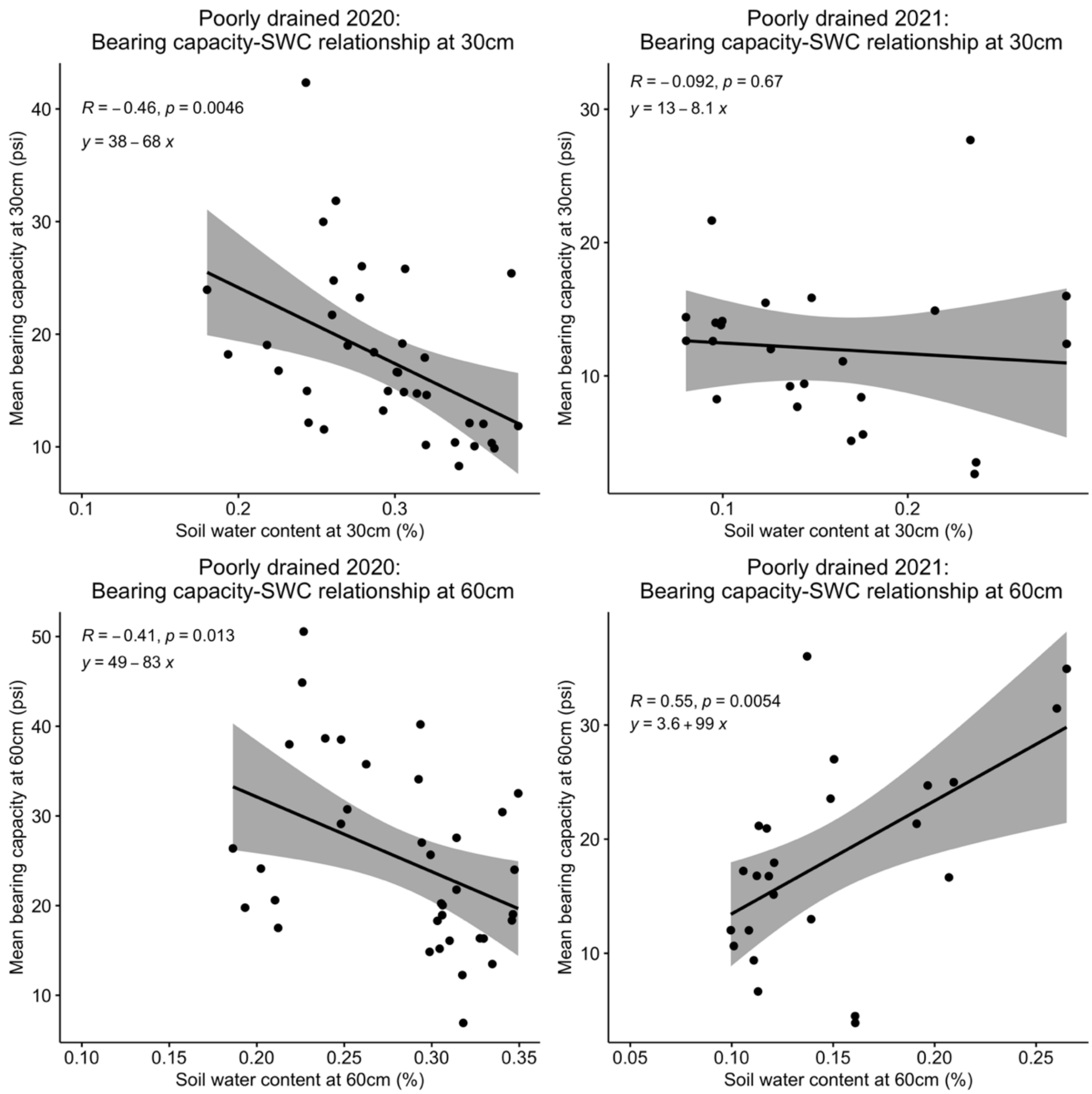

Figure B.4: Linear regressions between soil water content and mean bearing capacity for the poorly-drained class. Confidence intervals are 95% and level of significance is equal to 0.05.

Appendix C: Soil respiration and extractable nitrogen concentrations

Table C.1: Mean soil respiration, total nitrogen, ammonium, and nitrate/nitrite for summers of 2020 and 2021. Superscript letters indicate significant differences between means within a given year. Level of significance (alpha) is 0.05. Confidence intervals are 95% confidence.

		2020		2021	
	Units	Mean	Confidence interval	Mean	Confidence interval
Respiration	μmol m^{-2} s^{-1}	6.96^{a}	5.75 -8.41	4.26^{b}	3.71 - 4.90
Total nitrogen	mg^{-1} kg^{-1}	7.61^{a}	5.31 - 10.8	13.3^{b}	9.49 - 18.7
Ammonium	mg^{-1} kg^{-1}	2.97^{a}	1.17 - 7.61	12.9^{b}	5.05 - 33.1
Nitrate/nitrite	mg^{-1} kg^{-1}	3.03^{a}	1.19 - 7.69	5.10^{b}	2.05 - 12.8

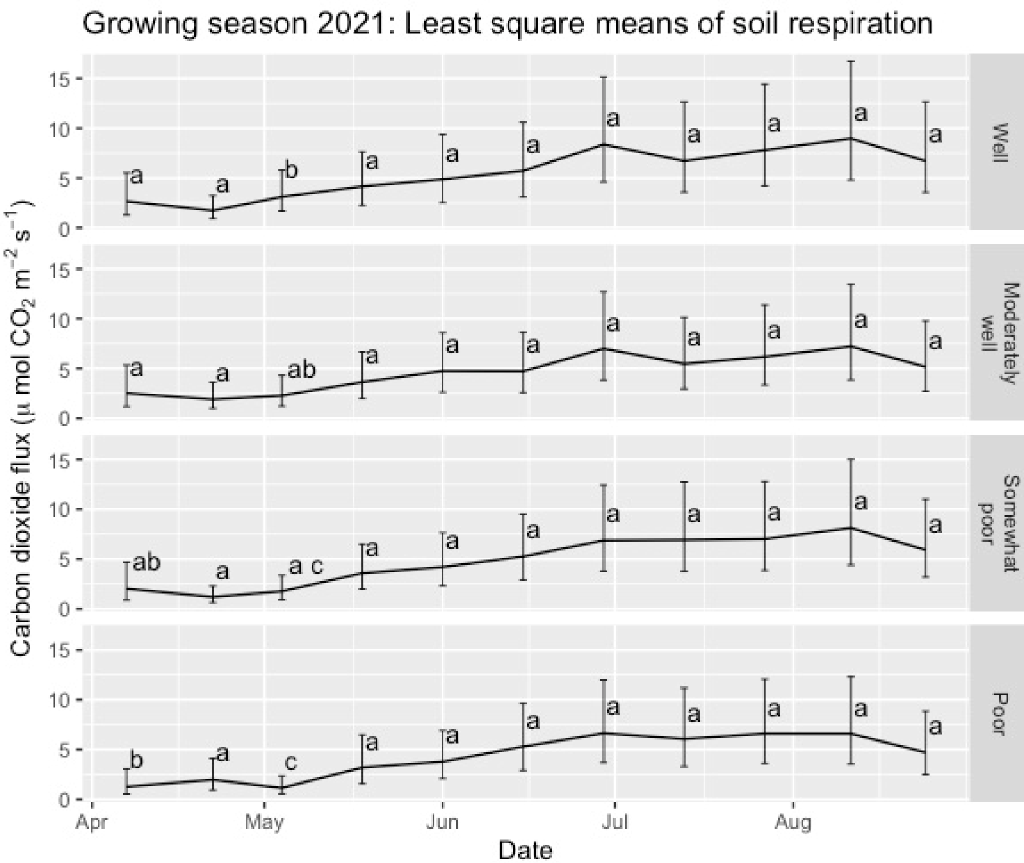

Figure C.1: Soil respiration during 2021 across time and drainage classes. Letters indicator significant differences (p-value < 0.05).

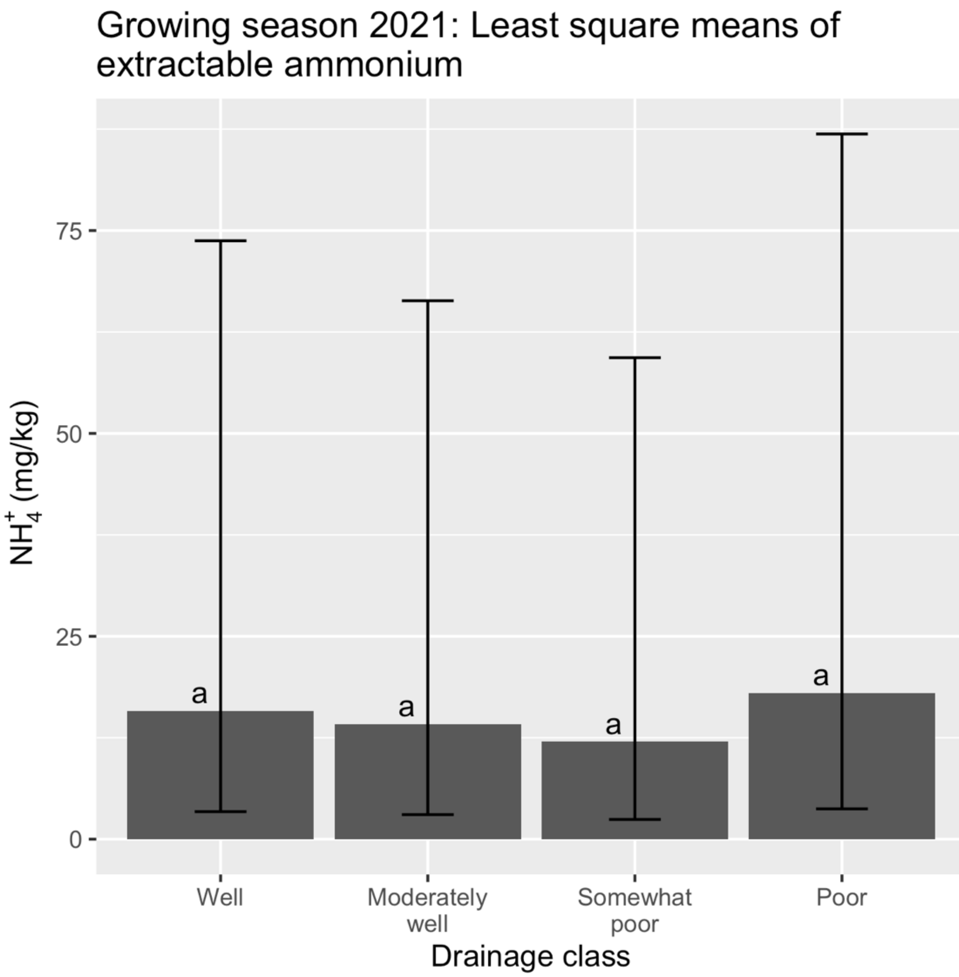

Figure C.2: Least square mean values of extractable ammonium across drainage classes for 2021. Letters indicate significant differences between drainage classes (p-value < 0.05). Error bars represent 95% confidence intervals.

Appendix D: Vegetation community surveys

Table D.1.: Two-way ANOVA summaries of mixed models of species richness and Shannon's Diversity Index for 2021 vegetation community surveys. Site was included as a random variable in the models. Numerator degrees of freedom are shown. Bolded values denote significant effect (p-value < 0.05).

		Species richness	**Shannon's Diversity Index**
Model Term	**Degrees of freedom**	**p-value**	**p-value**
Intercept	1	**<0.001**	**<0.001**
Drainage	3	0.4467	0.2953
Treatment	1	0.8587	0.6819
Drainage:Treatment	3	0.4716	0.1663

Appendix E: Pre-treatment carbon and nitrogen percentages

Table E.1: Mean percentages of pre-treatment carbon and nitrogen by site and drainage class.

	Aitkin		**Itasca**		**St. Louis**	
Drainage class	**%C**	**%N**	**%C**	**%N**	**%C**	**%N**
WD	1.27	0.07	0.73	0.04	1.24	0.08
MWD	1.24	0.07	1.07	0.06	1.90	0.13
SPD	1.62	0.10	0.82	0.05	3.25	0.22
PD	1.94	0.14	1.12	0.08	5.25	0.39