REWETTING POCOSIN WETLANDS LOWERS GREENHOUSE GAS EMISSIONS

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Abstract

Restored peatlands are known to be highly efficient carbon sinks and wetland restoration efforts take advantage of this efficiency to use as a climate change mitigation strategy. Evaluating the carbon sequestration capacity of peatland requires a greater understanding of how the seasonal fluxes of greenhouse gases (GHG) change, both in magnitude and direction, during and after peatland restoration. One aspect that has not received much attention is how carbon dioxide, methane and nitrous oxide emissions change during peatland restoration involving a hydrologic manipulation (i.e. raising water table depth). This study involved field data collection of GHG fluxes, soil temperature, soil moisture, and water table depths before and after hydrologic manipulation. I investigated the role of water table depth and soil temperature on changes in GHG fluxes pre- and post-hydrologic manipulation of a pocosin wetland (a shrub/scrub bog containing low nutrient, acidic soils). The study revealed that higher water table depths correlated to lower carbon dioxide (CO_2), higher methane (CH_4) and nitrous oxide (N_2O) emissions. I found that the reduction in CO_2 outpaced the increases in CH_4 and N_2O by a factor of 100. In addition, rewetting the pocosin wetlands also decreased the susceptibility to fire as evidenced by the decreased smoldering potential of soils undergoing hydrologic restoration. Taken together, the decreased CO_2 emissions and reduced smoldering potential of rewetted soils

indicates that hydrologic restoration of a pocosin wetland may be an effective mechanism to lower soil greenhouse gas emissions.

REWETTING POCOSIN WETLANDS LOWERS GREENHOUSE GAS EMISSIONS

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REWETTING POCOSIN WETLANDS LOWERS GREENHOUSE GAS EMISSIONS

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INTRODUCTION

From 2002 to 2011 atmospheric concentrations of carbon dioxide (CO_2) increased 2.0 ppm yr^{-1}, and marks the fastest decadal increase recorded (IPCC 2015). This increase in CO_2 is predicted to lead to a change in climate such that average annual air temperatures will be higher, sea levels will rise, and extreme weather events will become more frequent and intense beyond the year 2100 (IPCC 2015). Realization of the ecological impacts associated with climate change have led to an increased interest in understanding how natural systems could help mitigate rising greenhouse gas emissions (Griscom et al. 2017).

Despite being relatively rare landscapes, peatlands play an important role in the global carbon cycle. It is estimated that 350-550 GT of carbon (20-25% of the total global stored organic soil carbon) are stored in peatlands which cover just 3% of the total global land area (IPCC 2006). This ability of peatlands to store large amounts of carbon relative to acreage means that they are capable of providing climate change mitigation (Bridgham et al. 2008). This large carbon pool stored in the world's peatlands is being depleted due to anthropogenic disturbances such as draining for agricultural production, extreme wildfire events, and peat mining. Countries around the world are making a priority of restoring peatlands with the dual goals of reversing the previous destruction as well as increasing the acres of peatland soils to mitigate the effects of climate change.

In peatlands, the rate of decomposition, which is slowed by an anoxic environment, is dependent upon several biogeochemical processes; temperature, type of electron acceptors available, and moisture levels. Additional study of peatlands soils is necessary to better understand mechanisms controlling greenhouse gas (GHG) emissions as well as calculations into if and how much carbon is sequestered post-restoration. This research will aid in informing government agencies, stakeholders, land owners, and the public if rewetting is effective in sequestering carbon in a pocosin wetland.

Causes and consequences of increasing greenhouse gas emissions

Negative effects of climate change include, increases in air temperature, changes in plant community, and alteration of weather patterns. Atmospheric temperature is controlled by GHGs holding and reflecting the sun's radiative energy back to the earth's surface. The atmosphere's rising radiative energy is the result of the increased concentration of GHGs that began with the industrial revolution, of which the primary causes can be traced to use of fossil fuels and alteration of land use. The rate at which each GHG traps radiative energy depends upon molecule size, residence time, and reaction rate with oxygen. (Forster and Ramaswamy 2007). These chemical behaviors cause radiative forcing resulting in a change in the energy of the atmosphere. Radiative forcing is unequal for individual GHGs, and to more accurately predict climate change outcomes, each GHG is converted to the metric referred to as global warming potential (GWP). The GWP is a measure of how much energy the emissions of one ton of a gas will absorb over a given time period relative to the emissions of one ton of carbon dioxide (IPCC 2015). The GWP is calculated on a pulse emission event. However, wetland GHG emissions are constant rather than pulse events and new metrics are being developed to more precisely report wetlands GHG emissions with some scientists suggesting the use of sustained-flux global warming potential (SGWP) as a more accurate methodology (Neubauer and Megonigal 2015). Based on the SGWP method, methane and nitrous oxide reach steady states after 50 years and 480 years, respectively, while the radiative forcing of increasing carbon dioxide will affect the temperature of the atmosphere continuously.

Between years 1750 – 2011, the concentrations of atmospheric GHGs dramatically increased: CO_2 increased 40% (278 ppm to 390.5 ppm), CH_4 increased 150% (722 ppb to 1803 ppb) and N_2O increased 20% (271 ppb to 324.2 ppb) (IPCC 2015). Historic fluctuations of a similar magnitude developed over a period of 100,000 years. In addition to increased GHG concentrations, GHG emission rates have also increased; from 0.4GtCO_2-eq. per year from 1970-2000 to 1.0GtCO_2-eq. per year from 2000-2010 (IPCC 2015). The human-induced increases in the magnitude and rate of increase of GHG emissions have prompted governments around the world to research methods that both reduce the amount of GHG emissions and remove GHG from the atmosphere.

Peatland soil as carbon storage

Peatlands cover 3% of the earths terrestrial environments and store 25-30% of soil carbon (IPCC 2015). As climate change causes increased air temperatures in the northern latitudes, peatlands may experience decreases in water table levels. These lower water tables allow for increased rates of decomposition, the result of heterotrophic and autotrophic respiration, increasing emissions of greenhouse gases from the soil which results in many wetlands becoming a carbon source. The anoxic environment of flooded wetlands supports slow decomposition of organic material and the wetlands become a carbon sink. This soil carbon (455 Pg) is the result of an imbalance between photosynthesis and decomposition and peatlands biogeochemistry contributes to the slow rate of decomposition. The ability of peatlands to store carbon comes about as the lack of oxygen forces the microbial community to seek alternative sources of fuel for ATP production by consuming terminal electron acceptors, such as sulfate and nitrate, with low redox potentials. The lower redox potential significantly slows the rate at which microbes can consume organic material and over a time frame of millennia, deep layers of high carbon content peat soil is formed (Moore 1989). The presence of methanogens and methanotrophs (CH_4 producers and consumers, respectively) are heavily influence by the oxic/anoxic horizon in peatlands. Methanogens (archaea), obligate anaerobes, consume carbon dioxide, hydrogen, and acetate compounds in anoxic soils to produce methane. Oxygen is toxic to these organisms. Methanotrophs are methane loving bacteria, strictly aerobic, consuming methane as an energy source. In addition to the microbial community present in the soil, plant communities also affect rate of decomposition through the chemical makeup of leaf litter and root exudates. In addition, changes in soil temperature also influences enzymatic decomposition of organic matter as low temperatures slow enzyme activity and high temperatures denature enzymes. (Mitsch and Gosselink 2007). Historically, biogeochemical processes have allowed wetlands to serve as long term carbon storage pools but climate change and land use changes are causing wetland carbon losses at alarming rates (IPCC 2015).

The conterminous United States contains ~1500 Mg C ha^{-1} stored in 93,000 km^2 of peatland (Bridgham et al. 2006), with the Eastern United States comprising a swath of peat soils from Virginia down to Florida (Wells 1942) (Figure 1.1). In North Carolina this land includes pocosin wetlands, an ombrotrophic bog composed of peat soil (Richardson 1981). Specifically, the Albemarle Pamlico peninsula contains approximately 190 million metric tons of peat (Ingram and Otte 1981) (Figure1.2). At one time this area contained 10,000 km^2, but due to draining for forestry, agriculture and peat mining only 2,810 km^2 remained by 1980 (Richardson 1983). This major loss in peatlands have promoted pocosin restoration efforts such as the Pocosin Lakes National Wildlife Refuge (PLNWR), established in (Phillips 2007).

Peatlands hydrology

Water tables in North Carolina peatlands fluctuate seasonally in response to rainfall amounts: water tables are low in summer months and higher in spring and fall. Drainage of the land for agricultural production alters the fluctuations of the water table, introducing oxygen into deeper soil layers. This electron acceptor (high redox potential) increases the rate of decomposition thereby increasing CO_2 emissions. Facultative anaerobic activity is suppressed leading to a decrease in CH_4 emissions (Turetsky, Donahue, and Benscoter 2011). Studies have shown wetlands restored by rewetting result in a reduction in CO_2 and an increase in CH_4 emissions (Dinsmore et al. 2009; Wilson et al. 2009; Olefeldt et al. 2017). If the wetland restoration goal is to enhance carbon storage capacity then the primary goal should be to decrease GHG fluxes from the soil. If the overall goal is to reduce atmospheric GHG levels then the primary goal should be to decrease CO_2 fluxes at a pace that will offset increases in CH_4 fluxes.

Not only does water level influence GHG emissions, but soil temperature and plant community structure indirectly influence GHG emissions (Krauss and Whitbeck 2012; Olefeldt et al. 2017). For example, denitrification results in two gaseous end products; nitrous oxide and dinitrogen. Water levels, soil temperature and pH influence how much nitrous oxide compared to dinitrogen gas is produced. A recent study observed higher N_2O fluxes during winter than summer (Mcnicol et al. 2017), while another study reported that N_2O flux switched from gas production to consumption between ~7.5°C and 8.5°C (Dinsmore et al. 2009). In addition, the plant community influences the rates of CO_2 and CH_4 emissions by controlling rates of evaporation, oxygen to the roots, root exudates influencing soil enzyme activity and phenols in leaf litter. Tall pocosin (dominated by *Pinus serotina* (pond pine)) has greater soil CO_2

respiration rate than that of short pocosin (dominated by shrub/scrub) under similar water level conditions (Bridgham and Richardson 1992). Herbaceous peatlands produced higher levels of CH_4 fluxes three years post restoration compared to shrub and moss peatlands (Waddington and Day 2007). The phenolic compounds introduced to the soil, root exudates and leaf litter compositions caused reduced microbial CH_4 production pathways resulting in a lower CH_4 fluxes. This plant-induced reduction in methane appears to be unique to pocosin wetlands (Bridgham and Richardson 1992; Fenner and Freeman 2011).

Objectives, Purpose, Questions, and Hypotheses

The Pocosin Lakes National Wildlife Refuge (PLNWR) Clayton Blocks restoration project was initiated in a pocosin wetland in 2015 as a pilot project to investigate the possibility of developing carbon credits for wetland restoration. My study investigated the role of water table depth, soil temperature, and plant community structure on GHG emissions. The overarching goals of the project were to quantify carbon storage of pocosin soil and develop proxies for direct measures of GHG emissions. I addressed the following questions: (1) How does hydrologic restoration influence GHG flux rate? (2) What are the GHG fluxes before and during rewetting? and (3) How do water table depth, soil temperature and plant community relate to GHG fluxes? **I hypothesize that: (1) raising the water table level in a pocosin wetland will result in decreased CO_2 fluxes and increased CH_4 and N_2O fluxes as soils switch from oxic to anoxic conditions, and (2) restoring a pocosin wetland by rewetting (raising the water table level to more closely resemble natural seasonal cycles) will result in net carbon storage as increases in CH_4 and N_2O carbon equivalents will be offset by decreases in CO_2 emissions.**

MATERIALS AND METHODS

Site Description

This study was conducted in the Pocosin Lakes National Wildlife Refuge (PLNWR) (110,000 acres of pocosin habitat) located in the Pamilco-Albemarle peninsulas in the counties of Hyde, Tyrrell, and Washington, North Carolina, USA (35.7510°N, 76.5102°W), (Figure 1.2). Pocosin is a swamp habitat dominated by dense shrubs and deep, nutrient poor, organic soils (Richardson 1981). These soils, located throughout the coastal plain of North Carolina USA, are categorized as Belhaven muck (loamy, mixed, dysic, thermic, Terric Haplosaprist) (National Cooperative Soil Survey 2008). This region of North Carolina experiences a humid, sub-tropical climate where the average annual rainfall is 1320 mm and the average temperature range is 0.6°C in January to 31.9°C in July (NCSU 2017).

During the latter half of the 20th century this area of North Carolina was ditched and drained. Agriculture proved to be unprofitable and in 1990 the PLNWR was created restore the previous agricultural land back into wetlands (Phillips 2007). To keep expenses down, the PLNWR was partitioned into blocks separated by dykes and canals utilizing existing farm canals. The primary hydrologic input is rainfall and water can also be moved onto the refuge through the operation of water control structures installed in many of the canals. Hydrologic output is seepage and evapotranspiration. There is an elevation gradient of approximately 1 meter, high point at the northeast corner of the study area and low point at the southwest corner of the study area. The study sites west of DeHoog Road, are typical pocosin hummock and hollow topography with ferns, dense shrubs (typically less than 2 meters in height), and sumac trees. The study east of DeHoog Road, are dominated by fern and canebrake with areas of dense shrubs growing taller than 3 meters, and cypress saplings. All sites have pond pine growing near the canals and along depressions left by agricultural use.

Wetland Restoration

A primary goal of wetland restoration projects is to create an environment that will allow for the return of ecosystem functions. The Clayton Road Blocks Project is seeking to restore the carbon storage capacity of PLNWR. This restoration project involved building 2.5 miles of new dike to restore 536 hectares of the PLNWR while protecting the surrounding farm land from seepage (Figure 1.3). Four study plots were placed within the Clayton Blocks: two study plots were placed within site D16 and served as the nonmanipulated controls, one study plot each were placed within sites C13 and C14 and served as the manipulated experimental sites (Figure 1.4). The newly installed dike and water control structures allowed the water table to be raised and lowered to mimic historic seasonal fluctuations. Rewetting will enhance the carbon storage capability of the peat soils by counteracting the drained state. Decomposition rates will decrease post-rewetting as the microbial community reacts to the loss of the oxygen as a favored electron acceptor and must search out new electron acceptors. This should result in an increase in soil organic matter, thus increased carbon storage.

Data Collection

Sampling Methods – Field

Greenhouse Gas Sampling

The goal of the study was to estimate total GHG fluxes, compare pre-rewetting flux rate to post-rewetting flux rate, and determine whether the pocosin wetland experiences net carbon storage post-restoration. I measured carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) fluxes via static chamber method over two years (2016, 2017) at monthly intervals that occurred during eight months each year for a total of 16 GHG collections. The 2016 GHG collection from each chamber consisted of 3 samples taken at 30 minute intervals; minute 0, minute 30 and minute 60. The 2017 GHG collection from each chamber consisted of 4 samples taken at 20 minute intervals; minute 0, minute 20, minute 40, and minute 60. An experiment was conducted to assess the validity of 3 vs 4 time points.

Soil Temperature

To assess the effect of soil temperature and air temperature had on GHG fluxed, I installed Hobo™ Pro V2 Onset soil temp/external temp (Bourne MA, USA) data loggers at each study site. The probe was installed at a depth of 10 cm and recorded temperatures hour. I downloaded data monthly through April 2017. However, data collection ended prior to study completion due to wildlife disrupting monitoring devices. Therefore, no post-manipulation data are available. To obtain the soil temperature data used for GHG flux rate calculations I collected single point measures of soil temperature during each individual GHG collection Taylor Green Living model 5976N Soil Thermometer (Oak Brooke IL, USA) by placing the thermometer 10 cm into the ground next to each chamber base.

Water Table Depth

To assess a possible WTD effect on GHG fluxes, continuous water table depth (WTD) was monitored using In-Situ Inc©, Level Troll 500 Data Logger (Fort Collins CO, USA). In August of 2015 I installed water wells (4 inch pvc pipe) in duplicate at each of the five study sites and I permanently installed data loggers inside the water wells. The loggers recorded data hourly and I downloaded the data monthly via a handheld device In-Situ Rugged Reader (Fort Collins CO, USA). Data were reported as meters below the soil surface.

Soil Moisture

To assess the effect of soil moisture upon GHG fluxes, I installed Decagon Devices Inc© model Em5b (Pullman WA, USA) soil moisture data loggers at each site and the probes were installed at a depth of 15 cm. The Em5b internal circuitry calculated soil moisture content by taking a measurement every second, averaging across the hour and saving that single value as volumetric water content m^3 (water) m^{-3} (soil). Since I downloaded the data monthly beginning in April 2017, there was no pre-manipulation soil moisture data.

Greenhouse Gas Chamber Design

Greenhouse gas fluxes were collected using the static chamber method. The GHG chamber design was based on established USGS protocols (Cormier and Moss 2016). At each study site I permanently installed four bases in the ground to a depth of 5 cm. Chamber tops opaque lexan plastic coil chambers 29.4 x 29.4 x 30.5-cm and were fitted with septa for gas sample extraction. To create an air tight seal during sampling, each base was designed with a 2 cm (W) x 2 cm (D) channel which was filled with water. Tops were installed just prior to sampling and removed immediately after.

Greenhouse Gas Sampling

To estimate total GHG fluxes, four study sites were established; two manipulated experimental sites and two nonmanipulated control sites. I sampled pre-rewetting (January 2016 thru March 2017) and post-rewetting (April 2017 thru October 2017) to assess GHG fluxes. To examine how plant community composition, affect GHG fluxes a fifth set of chambers was placed inside a shrub canopy. This shrub canopy was located within the nonmanipulated control site. However, I only sampled this set of chambers during 2016 due to wildlife disrupting monitoring site.

Greenhouse gas sampling followed USGS protocols (Cormier and Moss 2016). When possible, sampling efforts were separated by a minimum of 4 weeks. Prior to placing the chamber top, vegetation inside the chamber base was identified (species when possible), counted (stem), and if necessary either cut to 9-inch height or bent over (dependent upon density). The action taken was recorded in field notes and transcribed into the GHG data sheets. Plant growth within the chambers was categorized as present/absent, action taken (none, bent or cut) and amount of plant coverage (none, low, medium, high). These data were incorporated into the GHG flux rate linear models.

Proper assessment of greenhouse gas fluxes required the stirring of the headspace as gases can stratify over time within the sealed chamber (Parkin and Venterea 2010). To ensure a well-mixed headspace prior to gas collection, I used a 30-mL plastic syringe equipped with a 25 gauge precision glide hypodermic needle (Becton Dickinson & Co Franklin Lakes NJ, USA) to manually stir the headspace gases. Air was pulled into the syringe and pushed back into the chamber headspace by 3 pumps (2016) and 4 pumps (2017). (Supplemental material) Optimal chamber deployment time is dependent upon chamber system design. Both size and shape of chamber affects the linearity of the selected time points to provide and accurate rate of soil GHG fluxes (Collier et al. 2014). Gas samples were transferred from the headspace of the static chamber using a 10-mL plastic syringe/hypodermic needle at intervals of 0, 30, 60 minutes (2016) and intervals of 0, 20, 40, 60 minutes (2017). (Supplemental material).

I transferred the individual 10 mL gas samples into 3.7 mL Labco Exetainers® fitted with double septa caps (Lampeter, United Kingdom) and stored in whirlpak bags inside of ziplock bags for transport. Double wadded septa provide gas-tight seal to ensure sample integrity.

Methods – Laboratory analysis

Greenhouse Gas Analysis

To determine the flux rate of GHGs from the study sites, CO_2, CH_4 and N_2O concentration of the gas samples were analyzed using a Shimadzu GC2014 (Durham, North Carolina, USA) gas chromatography with a single point manual injection port. Gas samples are delivered into injection port and analyzed for CH_4 and CO_2 using a flame ionization detector (FID) and N_2O using an electron capture device (ECD). Gas concentrations were analyzed within 48 hours when possible. Six-point calibration curves were used to determine GHG sample concentrations. Calibration curves were made using calibrated Airgas Specialy Gases (Plumsteadville, PA, USA) gas standard containing 600 ppm-v CO_2, 5 ppm-v CH_4, 1 ppm-v N_2O (nitrogen = 99.9394%) and 3000 ppm-v CO_2, 50 ppm-v CH_4, 50 ppm-v N_2O (nitrogen=99.69%).

I used GCSolution software (version 5.81 SP1 Shimadzu Corporation) to integrate the gas peaks for quantifying GHG concentrations. Fluxes were calculated from the linear change in gas concentrations as a function of time, chamber volume, collar area, and air temperature. To first determine the GHG emission levels from the pocosin study sites, Excel® spreadsheets were developed to tabulate fluxes. The ideal gas law was used to obtain mols of the gas. The slope of the line generated from the relationship of gas concentration over time for each chamber represented greenhouse gas flux rate per hour. Carbon dioxide equivalents used for calculation of global warming potential are 1 kg CO_2 = 32 kg CH_4 = 270 kg N_2O over a 100 year period (Neubauer and Megonigal 2015).

Statistical Analyses

To examine whether the manipulation of the water table depth was independent of plot location, I used a t-test. Linear models tested the fixed effects of water table depth, soil temperature, plot location and season on GHG fluxes. Water table depths used in the models were obtained by averaging reported depths of the 24 hours prior to each removing of GHG samples.

Since I sampled 11 sample points pre-rewetting and sampled 5 sample points post-rewetting, I compared GHG fluxes from similar months pre- and post-restoration (April-October 2016 and 2017). I used linear models to investigate how GHG emissions varied as a function of water table depth and soil temperature. Statistical analyses were conducted in R© Statistical Computing environment (version 3.3.2).

RESULTS

Environmental variation

The study area is in a temperate climate zone in which extreme weather and are temperatures seldom occur. The average temperature for the area ranges from 0.6 °C (January) to 32.3 °C (July). Soil temperature at all sites followed seasonal patterns with 4-10 °C recorded during winter and 25-30 °C recorded during summer (Figure 1.6). Soil temperatures did not vary significantly among were similar across study site locations (Figure 1.6). During the study period, 3 major hurricanes affected PLNWR study site: Hermine (September 2016) Matthew (October 2016) and Harvey (August 2017). These hurricanes increased the monthly rainfall totals: 414.5 mm (218% above normal), 301 mm (207% above normal), 215 mm (51% above normal) respectively (NCSU 2017).

The sole hydrologic manipulation in my study involved raising the water table through water control structures in adjacent canals, to more closely resemble natural seasonal fluctuation. Water table depth (WTD) at the manipulated sites ranged from 0.161 to -1.127 m over the course of the study. Maximum post-manipulation WTDs were 0.057 and 0.161 (meters above soil horizon) at sites C13 and C14 respectively, occurring in June 2017. Observations of mean daily pre- and post-manipulation (WTD) varied significantly at the hydrologically manipulated sites but not at the nonmanipulated sites $t = -5.35$, $p = 0.002$ $t = -6.62$, $p = 0.0005$ resulting in 64% and 66% (C13 and C14, respectively) decrease in average WTD post-manipulation (C13 and C14 respectively) (Figure 1.5). High WTDs recorded in September and October 2016 were a direct result of hurricanes Hermine and Matthew.

Trends

Greenhouse gas fluxes varied widely over the course of the study. Carbon dioxide (CO_2) fluxes ranged from -29,325 µg m^{-2} h^{-1} (site C14) to a maximum of 1,466,828 µg m^{-2} h^{-1} (site C13). Methane (CH_4) fluxes ranged from -173.5 µg m^{-2} h^{-1} (site D16B January 2016) to a maximum of 49.36 µg m^{-2} h^{-1} (site D16A). Nitrous oxide (N_2O) fluxes ranged from -1280 µg m^{-2} h^{-1} (site C14) to a maximum of 969 µg m^{-2} h^{-1} (site D16B). There was no effect of study site location on CH_4 flux rates however, sites C13 and C14 had significant increases in CH_4 fluxes post-manipulation indicating that the manipulation of the water table influenced CH_4 flux rates. Pre-hurricane water table depths were ~0.95 m (C14) and ~0.76 m(C13) below the soil surface. Post-hurricane high water table depths were ~0.12 m (C14 – October 24, 2016) and ~0.04m (C13 – October 9, 2016) above the soil surface. Water table depth had returned to pre-hurricane seasonal depths by December 2, 2016. Post-hurricane high water table depths were similar to the spring 2017 manipulation of the water table depths 0.15m above soil surface (C14 – June 21, 2017) and 0.04m above the soil surface (C13 – June 6, 2017)

GHG fluxes varied widely over the course of the study with lowest fluxes of all 3 GHGs measured in the nonmanipulated sites. There were trends among GHG fluxes and season. Lowest CO_2 and CH_4 fluxes were measured in winter for all study sites. Manipulated sites CO_2

flux rates were highest in summer and CH_4 flux rates were highest in the fall. The observed trends were the following: (1) an increase in WTD resulted in a decrease in CO_2 fluxes (Figure 1.7), an increase in CH_4 fluxes (Figure 1.7), and no influence on N_2O fluxes: (2) an increase in soil temperature resulted in an increase in CO_2 fluxes (Figure 1.7), a slight increase CH_4 fluxes (Figure 1.8), but had no effect on N_2O fluxes.

Abiotic impacts on GHG fluxes

The manipulation of the water table, soil temperature (ST), season, plot location, and time of day influenced GHG fluxes in different ways. Over the course of the study the variability in CO_2 fluxes was explained by WTD (Figure 1.7) (F=11.76, df=1,59, p=0.001), ST (Figure 1.7) (F=123.78, df=1,63, p=0.0006), season (F=7.395, df=3,66, p=0.0002), and WTD * Season (Figure 1.8) (F=14.88, df=7,53, p=1.473e-10). The variability in CH_4 was explained by WTD*Plot (F=4.898, df=9,51, p=9.691e-05) (Figure 1.9).

PRE- VS POST-MANIPULATION GHG FLUXES

Trends

In general CO_2 fluxes decreased post-manipulation. Manipulated sites C13 (713.15 mg m^{-2}, hr^{-1} to 305.26 mg m^{-2}, hr^{-1}), and C14 (1173.30 mg m^{-2}, hr^{-1} to 286.78 mg m^{-2}, hr-^{1}) experienced a 57% and 75% decline in CO_2 flux rates, respectively, while nonmanipulated site D16A (30.9 – 28.6 µg $m^{-2,}$ hr^{-1}) declined by 20%. Over the same time frames CO_2 fluxes in the nonmanipulated site D16B (263 – 328 mg $m^{-2,}$ hr^{-1}) increased 25%. (Figure 1.10). In general, CH_4 fluxes increased post-manipulation. Manipulated sites C13 (8.31 - 21.9 µg $m^{-2,}$ hr^{-1}) and C14 (5.59 - 27.1 µg m^{-2} hr^{-1}) experienced a 164% and 385% increase in CH_4 flux rates, respectively, while nonmanipulated site D16A (4.38 – 12.5 µg m^{-2} hr^{-1}) increased 190% (Figure 1.10). Over the same time frames CH_4 fluxes in the nonmanipulated site D16B (-6.25 to -8.3) declined by 25% (Figure 1.10). In general, N_2O fluxes increased post-manipulation. Manipulated sites C13 (122 - 134 µg $m^{-2,}$ hr^{-1}) and C14 (101 - 226 µg $m^{-2,}$ hr^{-1}) experienced a 9% and 123% increase, respectively, while nonmanipulated site D16A (10.1 – 26.8 µg $m^{-2,}$ hr^{-1}) increased 160% (Figure 1.10). Over the same time frame N_2O fluxes in the nonmanipulated site D16B (673 – 43.9 µg $m^{-2,}$ hr^{-1}) decreased 35%. GHG flux rates across all sites for September 2016, 2017 and October 2016, 2017 were compared to assess the effect the hurricanes may have had. The September model explained 41% of the variance of CO_2 fluxes and the October model shows no variation between 2016 and 2017 indicating the effect of the raised water tables post-hurricane (Figure 1.11).

Abiotic impacts on GHG fluxes

The manipulation of the water table, soil temperature, season, influenced GHG fluxes. Water table depth explained 42% of the variance in CO_2 fluxes (Figure 1.12) (F= 26.27, df = 1,34, p=1.181e-05), 16% of the variance in CH_4 fluxes (Figure 1.12) (F=7.187, df=1,31, p=0.01), and 16% of the variance of N_2O fluxes (Figure 1.12) (F=6.573, df=1,29, p=0.01). In addition, WTD*season explained 62% of the variance of CO_2 fluxes (F=12.73, df= 7,43, p=1.022e-8) (Table 1.1) and 18% of the variance of CH_4 fluxes (F=2.63, df=7,43, p=0.02) (Table 1.2) while WTD*plot location explained 44% of the variance of CH_4 fluxes (F=7.173, df=4,28, p=0.0004) (Figure 1.16, Table 1.3). There was no measured effect of soil temperature on GHG fluxes.

DISCUSSION

The high carbon storage capacity of peatland occurs only when carbon fixation outpaces decomposition rates. While previous studies suggest pocosin wetlands becoming carbon sinks under high water table conditions these data come from laboratory mesocosm experiments and may not be applicable to field studies (Bridgham et al. 2008; Wang, Richardson, and Ho 2015;

Dinsmore et al. 2009; Bridgham and Richardson 1992). My field study focused on GHG fluxes post-rewetting, where the sole experimental treatment being manipulation of the water table, has begun to fill this gap in knowledge. My findings showed that restoration in the form of re-wetting by raising the water table reduces net carbon emissions from the wetland. Post rewetting CO_2 fluxes from manipulated sites decreased on average ~647000 μg m^{-2} h^{-1}, and CH_4 and N_2O increased on average ~17 and ~68 μg m^{-2} h^{-1} respectively. Using the global warming potential of 32 for methane and comparing that to the decrease in CO_2, my results indicated that CO_2 decreases are offsetting the methane increases. Thus the CO_2 decrease far outpaced the increases in CH_4 and N_2O emissions. The caveat is that this CO_2 decrease must continue for the next 100 years to be considered a carbon sink.

Edaphic factors individual effects on GHG emissions

While I measured decreased CO_2 emissions under raised water table conditions, past studies (Dinsmore et al. 2009 and Krauss and Whitbeck 2012) found that the highest CO_2 fluxes related highest to lowest WTD. The Krauss and Whitbeck (2012) study found an interaction between lowest WTD and ST increased the reduction in CO_2 emissions. However, in my study, the interaction of season and WTD influenced the CO_2 fluxes which can be explained by plant growth and increased rainfall typically occurring in late summer and early fall. At the manipulated sites, the increased rainfall due to two major hurricanes resulted in a short term decrease in CO_2 fluxes. Similar results following a pulse event have been attributed to rewetting and an interaction with the plant community (Dinsmore et al. 2009). However, my study did not explicitly link this change in GHG fluxes to the plant community and more investigation is needed into plant community and GHG flux rate.

The increased CH_4 emissions were lower in the present study compared to those reported for boreal peatland studies. For example, a study conducted in a restored peatland (former site of peat harvesting for fuel) in Ireland linked WTD to CH_4 flux rates and measured a maximum CH_4 flux rate 16 mg CH_4 m^{-2} hr^{-1} (Wilson et al. 2009), which is far higher than recorded fluxes in the post-manipulation result of maximum of 0.048 mg CH_4 m^{-2} hr^{-1} in this study. This result, in accordance with previous studies, found that the increase in CH_4 emissions is lower in pocosin wetlands compared to a boreal peatlands (Bridgham and Richardson 1992). In a former peat mining field left untouched for 20 years Waddington and Day (2007) reported mean CH_4 range of fluxes to be -170 to 137.5 µg CH_4 m^{-2} h^{-1}, which is higher by a factor of 100 above the measured CH_4 fluxes of -1.735e-02 - 4.936e-03 µg m^{-2} hr^{-1} that I measured in the nonmanipulated sites, where pocosin fields were left untouched for 27 years. These extremely low CH_4 fluxes may be a result of specific biogeochemical processes unique to these pocosin peat soils. Future investigation focused on the role that microbial communities play within the rewetted pocosin soils such as methanotroph community structure and abundance may provide insight into why methane emissions are lower than expected in the pocosin wetlands. Large methanotroph communities living in aerobic soil layers may be able to consume the methane produced by the facultative anaerobes residing in the anoxic soil conditions created by raising the water table (Ran et al. 2017).

Past studies report a strong interaction between plant community and WTD (Dinsmore et al. 2009; Wilson et al. 2009). The authors attribute lower CH_4 emissions to the sedge dominant plant community compared to *Sphagnum* dominated communities. Labile vs recalcitrant carbon (Wilson et al. 2009) and amount of lignin present (Hartmann 1999) influenced CH_4 emissions. This may explain the differences found in the nonmanipulated site D16B and why D16B closely resembles manipulated sites. Phenolic levels in the soil and leaf litter have been linked to lower CH_4 production (Wang, Richardson, and Ho 2015). However, more studies focused on identifying specific phenolic compounds and their relationship with heterotrophic respiration.

Previous GHG studies in other types of wetlands have found that N_2O fluxes increase with increasing soil temperatures (Dinsmore et al. 2009) and as a function of season (Mcnicol et al. 2017). An examination of the data over the course of my study did not link N_2O fluxes to water table depths, soil temperatures, season or site location. An examination of the subset data revealed increases in N_2O fluxes post-manipulation as an effect WTD * site location as well as an interaction effect between WTD and season. The interactions between edaphic factors exerts more influence over N_2O emissions than any single factor.

Conclusions

In conclusion, I found consistent results that raising the water table increased N_2O and CH_4 fluxes and decreased CO_2 fluxes. However, the rise in N2O and CH_4 is offset by decreased CO_2 fluxes. The pocosin ecosystem responded quickly to extreme pulse events (major hurricane) that increased the water table levels and seasons influence CO_2 emissions with the highest emissions occurring in summer and lowest in winter regardless of plot location or water table manipulation. The non-manipulated plots provided some evidence that plant community may be a factor in GHG emissions and merits further investigation. Overall my results suggest that rewetting may be an effective short-term mechanism to lower overall GHG emissions in a previously drained pocosin wetland. However, more data are needed to determine whether restoration in the form of rewetting creates a long-term carbon sink.

Future directions

Long-term data is needed to determine whether the restoration of pocosin wetland and subsequent decreases in carbon dioxide emissions will continue to outpace the increases in methane and nitrous oxide emissions. Future studies investigating the role of the plant community in GHG emissions can provide information as to which plant species are most effective at increasing soil carbon storage rates. This knowledge can be used by land managers to develop enhanced soil carbon storage plans through replanting acreage with specific plant species with potential to mitigate soil carbon release.

CHAPTER 2: SMOLDERING POTENTIAL

INTRODUCTION

Peatlands are 3% of global land mass yet store 25-30% of soil carbon approximately 560 Gt. Much of this carbon is the product of slowly decomposing organic plant matter deposited over millennia where primary productivity outpaced decomposition resulting in carbon storage. The northern latitudes contain approximately 4 million km^2 of peatland (Yu 2012), of which 68,000 km^2 is located in the contiguous United States (Mickler, Welch, and Bailey 2017). The United States forested peatland carbon pool is estimated to be 14 Gt (Johnson and Kern 2003). Specifically, North Carolina's peatland carbon pool is estimated to be 0.327 Gt (Ingram and Otte 1981). Despite the carbon storage capacity, (peat depth of up to 6 meters) human activity (drainage for agriculture, peat harvesting for fuel) and natural events (fire, drought) are responsible for major declines in the peatland carbon pool. The loss to fire events can be substantial as seen by comparing the lost carbon of 0.009 Gt from the Evans Road fire (2008 Albemarle peninsula NC) and it would require 600 years to replace this lost carbon.

The main controls of carbon cycling rates in peatlands are temperature, water table depth and plant community composition. Severe drought, which is expected to become more prevalent under the effects of climate change, and draining introduce oxygen deeper into the peat layer threatening decomposition of carbon as oxygen enables enzymes such as phenol oxidases to mineralize organic matter more rapidly (Fenner and Freeman 2011). Subsequent changes in plant community composition occur because of lowering of the water table and increasing fire pressures. For example, a long term research study in western Canada reported a decline in water table increased tree productivity, led to increased organic matter accumulation and a 9-fold increase in combustion of soil organic matter during wildfire (Turetsky, Donahue, and Benscoter

2011). Approximately 365,160 km^2 of peatlands, 44,100 km^2 have been damaged by fire, 2630 km^2 affected by permafrost melt, 37 km^2 subjected to peat extraction and 16 km^2 damaged by mining (Turetsky et al. 2002). Extreme fire events have also led to the destruction of several meters of peat which released carbon to the atmosphere (Page et al. 2002; Mickler, Welch, and Bailey 2017; Turetsky et al. 2002).

Retaining this carbon is a primary goal of the restoration of peatlands worldwide as it is well studied that when this carbon is released to the atmosphere during wildfire, significant amounts of CO_2 can be added to the already high levels of GHG's (Table 2.2). Wildfires intensity and severity is increasing due to changes in land use and the effects of climate change. Lowered water table levels, caused by changes in land use, resulted in smoldering peat fires. This smoldering, consists of low temperature, little to no flame, high smoke and can remove a large amount of carbon. Once the peat is ignited, burning can continue both vertically downward and laterally. (Hungerford, Frandsen, and Ryan 1995). Smoldering is a three-stage process: (1) sufficient heat to dry the surface layer of peat (2) chemical reactions of char created by pyrolysis (endothermic), and (3)oxidation of the char, which produces the heat for pyrolysis and creates a smoldering front that moves both horizontally and vertically (Huang and Rein 2014). For example, Indonesia (1997) released approximately 0.95 Gt of carbon to the atmosphere (Page et al. 2002). These fires (May 1997 – March 1998) spread across the forested peatlands located in Central Kalimantan, Borneo and resulted in damage to both natural forested peatlands and land converted for other uses. The wildfires developed from out of control burns by farmers to clear land and were more intense due to the dry weather pattern El Niño. In addition to increased fire intensity, altered weather patterns resulting from climate change are causing an increase in primary production in several ecosystems (Poulter et al. 2014) resulting in large fuel loads for wildfire. These same altered weather patterns may be implicated in an increase of wildfires in Canadian peatlands which is released approximately 0.005 Gt annually (Turetsky et al. 2002). The Evans Road fire (2008) in Albermarle/Pamlico peninsulas of North Carolina, an area ditched and drained for agricultural production, released 0.009 Gt of carbon (Mickler, Welch, and Bailey 2017). During a fire event in peatlands, approximately 3% of carbon emissions are from above ground biomass and approximately 97% are from peat smoldering (Mickler, Welch, and Bailey 2017). As important as is the direct loss of carbon during a fire event, the time it takes to replace this lost carbon must be included in any calculations of carbon sequestration capabilities.

Soil factors of moisture content (MC) and mineral content (MnC) exert the most influence on the probability that smoldering will be sustained (referred to here after as smoldering potential). Studies show that smoldering will propagate when soil MC is insufficient to prevent ignition of the organic soil layers typical of peatlands and appears in shallow depths due available oxygen and high heat which develops during above ground fire fueled by highly flammable leaf litter (Turetsky, Donahue, and Benscoter 2011; Reardon, Hungerford, and Ryan 2007). Carbon accumulation rates in drained Canadian peatlands are twice that of pristine peatlands (68.7±15.1 g C m^{-2} yr^{-1} and 33.9±10.8 g C m^{-2} yr^{-1} respectively) (Turetsky, Donahue, and Benscoter 2011). A post-fire study on these Canadian peatlands revealed that the drained plots experienced carbon losses nine times higher than the pristine plots (16.8±0.2 kg C m^{-2} and 2.0±0.5 kg C m^{-2} respectively) (Turetsky, Donahue, and Benscoter 2011). Although a drained forested peatland is more effective at sequestering carbon, mainly due to higher primary productivity creating more litter than the microbial community can consume, severe burning resulted in complete loss of post-drainage carbon accumulation plus 450 years of peat accumulation (the adjacent pristine peatland lost 58 years of peat accumulation in the fire)

(Turetsky, Donahue, and Benscoter 2011). Both natural and restored peatlands have high water tables, high MC, and low MnC. (Granath et al. 2016). The wet, dense organic layers found deeper in the peat profile typically present a barrier to smoldering (Turetsky et al. 2015). At MC's below 150% horizontal fire spread rates of between 1 and 9 cm h^{-1} have been measured while moisture distributions in the top 5 cm of peat determine the spread of smoldering fires with a MC of >250% found to inhibit smoldering and MC of >160% will constrain smoldering to a lateral distance of <10 cm (Prat-Guitart et al. 2016). Smoldering potential decreases 19.3% with a 5% increases in moisture content and increases 155.9% with a 1% increase in mineral content (Reardon, Hungerford, and Ryan 2007). In the Pocosin Lakes National Wildlife Refuge (PLNWR) MnC changes across the landscape both vertically and horizontally with the higher levels located on the edges of the plots (mean MnC of 7.4% (Ingram 1987). However, in boreal peatlands MnC has been shown to vary among plant communities; *Sphagnum fuscum* hummocks (2.4±1.0%), feather moss hummocks (4.6±2.7%), mixed species hollows (6.6±2.7%) but not with soil depth (Benscoter et al. 2011). The fire adapted plants of the pocosin -- *Andropogon virginicus* (L.) (broomsedge bluestem), *Woodwardia virginica* (L.) Sm. (Virginia chainfern), *Arundinaria gigantea* (Walter) Muhl. (giant cane) *Ilex glabra* (L.) (inkberry), *Cyrilla racimiflora* (L.) (swamp titi) -- increase in density post-fire (Wells 1928). Leaves rich in aromatic compounds increase both flammability and fire intensity (Christensen et al. 1981). In boreal peatlands MnC has been shown to vary among plant communities; *Sphagnum fuscum* hummocks (2.4±1.0%), feather moss hummocks (4.6±2.7%), mixed species hollows (6.6±2.7%) but not with soil depth (Benscoter et al. 2011). The fire adapted plants of the pocosin -- *Andropogon Virginicus* (L.) (broomsedge bluestem), *Woodwardia virginica* (L.) Sm. (Virginia chainfern), *Arundinaria gigantea* (Walter) Muhl. (giant cane) *Ilex glabra* (L.) (inkberry), *Cyrilla racimiflora*

(L.) (swamp titi) -- increase in density post-fire (Wells 1928). Leaves rich in aromatic compounds increase both flammability and fire intensity (Christensen et al. 1981).

Plant communities may influence smoldering by retaining soil moisture, altering MnC and increasing the intensity of the above ground fire. In a past study, undisturbed moss-dominated peatlands recover in the first decade post-fire due to *Sphagnum* mosses maintaining a high moisture content in the surface soil thus reducing the smoldering potential (Kettridge et al. 2015). Draining of moss-dominated peatlands led to plant succession to a deciduous plant community and resulted in losses of up to 77% of *Sphagnum* creating an environment where smoldering potential is high (Kettridge et al. 2015). Although this plant community increases fire intensity, adequate soil moisture will extinguish smoldering (Turetsky et al. 2015).

The Pocosin Lakes National Wildlife Refuge is currently undergoing a restoration of 752 previously drained hectares in the southern area of the refuge. It is necessary to monitor smoldering potential post-restoration to document loss of newly stored carbon during a fire event. The objectives of this study were to measure current smoldering potential at the study sites and to assess how long it might take to counteract carbon loss due to fire based on GHG flux rates measured during this study. To address this objective, I (i) to measured soil moisture and mineral content in the GHG study plots across time, (ii) determined smoldering potentials of the soils in the GHG study plots, and (iii) compared smoldering potential in the hydrologically manipulated plots compared to the unmanipulated plots. I hypothesized that due to rewetting in the form of raising the water table the manipulated plots will have a significantly lower smoldering potential compared to unmanipulated plots.

To achieve my objectives, I sampled 4 months during summer and autumn of 2017. At each GHG study plot replicate root mat (soil layer composed of moderately decomposed organic matter with small to fine roots) soil cores were removed from 0-5 cm and 5-10 cm depth, dried to determine gravimetric moisture content and ashed to determine soil mineral content. These data were then used to calculate smoldering potential. Linear models were used to assess smoldering potential within and among plots. The relationship between seasonal changes in water table and smoldering potential has been investigated however, our study seeks to add to this knowledge by investigating smoldering potential during a hydrological restoration (raising the water table) of pocosin wetlands. My study assessed the time to restore the lost carbon from the Evans Road fire, should a similar event occur, using currently measured greenhouse gase flux rates. My study seeks to add to the knowledge base of mineral contents of pocosin soils as current information is limited.

MATERIALS AND METHODS

Site Description

Pocosin Lake National Wildlife Refuge (PLNWR) is located in the counties Hyde, Tyrrell, and Washington North Carolina (35.7510° N, 76.5102° W). This area of North Carolina commonly referred to as the Pamlimarle (Pamlico and Abelmarle peninsulas), contains 44,500 hectares of pocosin habitat (Figure 3.1). Pocosin soils are composed of peat, a product of slowly decomposing organic plant matter deposited over millennia which accumulates as primary productivity outpaces decomposition. To facilitate agricultural production in the latter half of the 20th century the pocosin in the Pamlimarle area of North Carolina was ditched and drained. Established in 1990, PLNWR was created to protect and restore wetlands (Phillips 2007). To keep expenses down the PLNWR was partitioned into blocks utilizing existing agricultural ditches. Replanting was not conducted at the study sites and the plant community naturally propagated. This wetland habitat is dominated by dense shrubs and deep organic soils which requires fire to maintain natural ecological functions (Richardson 1981).

A goal of this wetland restoration is the return to a natural fire regime, prevention of smoldering, and preservation of carbon stores. Through the analysis of soil cores, I compared the smoldering potential within and among the four GHG study sites: two hydrologically manipulated sites and two non-manipulated sites.

Field Sampling and Processing

Continuous Soil Moisture

I used Decagon Devices Inc© data collection system Em5b loggers for continuous soil moisture measurement. Data collection began in April 2017, and loggers recorded moisture hourly, and I downloaded the data monthly. The Em5b internal circuitry calculated soil moisture content by taking a soil moisture measurement each second and then averaging the values across the hour and saving the average hourly soil moisture content. Data were reported as VWC (%).

Soil Sampling and Gravimetric Soil Moisture and Mineral Content

To measure soil moisture and mineral content I collected soil cores 3.1 cm diameter to a 10 cm depth from the study site in the summer (June and July, 2017) and autumn (September and October 2017). The 10 cm depth was based on the laboratory smoldering studies conducted by Reardon et al (2007) and Frandsen (1987). For replication, I collected three individual soil cores per site and, each core was separated into 0-5 cm and 5-10 cm and stored individually prior to laboratory analysis. When possible, samples were processed within 24 hours.

I determined soil moisture by weighing out ~25 grams of the fresh soil (Genemate model 500 balance) in an aluminum tin and recording this value in lab notebook. After removal from convection oven the sample was reweighed and this weight was recording in lab notebook. These two weights were used to determine gravimetric moisture content using equation 2.2. I determined mineral content by subtracting the post-ashing eight from the pre-ashing weight.

Equation 2.2.1 %soil moisture (fw) = 100 * ((1- dry weight)/fresh weight)

I determined mineral content by subtracting the final weight from the oven dried weight + tin weight. Twenty-five samples (0-5 cm range) and six samples (5-10 cm range) did not meet the 25-gram weight but as this calculation is based on a ratio, the lower weight will still provide valid analysis.

Smoldering Potential

Samples were a minimum of 25g (where possible) and weighed to the nearest 0.01 gram. Moisture content was determined by drying samples in a Fisher Scientific model OV702F Isotemp Oven for 24 hours at 105°C. Mineral content was determined by ashing at 500°C under a 3-hour soak in a Fisher Scientific Isotemp® Programmable Muffle Furnace 650 series. The weights of the three replicates were averaged for overall moisture and mineral content.

Statistical Analysis

To determine smoldering potential of the individual sites, I used values of gravimetric moisture and mineral content in equation 3.1 to calculate smoldering potential. To investigate my hypothesis that the smoldering potentials would be lower in the manipulated sites compared to the nonmanipulated sites I used a linear model. This linear model investigated smoldering potentials (derived from equation 3.1) as a function of site location and site status (manipulated or nonmanipulated).

Equation 3.1: Smoldering potential = $1/(1 + e^{-(2.033-0.43\times\text{moisture content}+0.44\times\text{mineral content})})$

To test the influence of hydrologic manipulation to smoldering potentials I used a linear model to examine the relationship between smoldering potentials (derived from equation 3.1) and the environmental factors of site location and according to hydrologic site status (manipulated, nonmanipulated).

RESULTS

Soil Moisture and Mineral Content

Soil moisture content (MC) was greater in the hydrologically manipulated sites than the unmanipulated sites (Figure 2.1a). Specifically, MC was 33% ± 7% (SE) lower at site C-13 and 31% ± 20% (SE) lower at site D16B compared to site D16A (Figure 2.2a). During this study, the range of MC differed across study sites: 87% to 200% at site C-13, 157% to 325% at site C-14, 112% to 228% at site D16A and 90% to 123% at site D16B (Table 2.1). There was no relationship between soil moisture content and water table depth.

During this study, the range of MnC differed across study sites: 1.9% to 8.7% at site C-13, 4% to 12.7 % at site C-14, 3.34% to 35.8% at site D16A, and 11.4% to 42.7% at site D16B (Table 3.1). Laboratory studies of ignition probability and smoldering propagation typically hold the MnC constant at ~2.5% (Prat-Guitart et al. 2016; Frandsen 1997). Using this % MnC may lead to low estimates of smoldering potential as in my study only 1 in 8 data points had MnC <2.5% and studies have shown that smoldering propagation increased 155% for every 1% increase in MnC (Reardon, Hungerford, and Ryan 2007). These results were similar to a past study where MC alone did not control smoldering potential and sustained smoldering was dependent upon MnC (Reardon, Hungerford, and Ryan 2007).

Estimated Smoldering Potential

Smoldering potential, the probability that sustained smoldering will occur, was low at all study sites. While the majority of samples recorded smoldering potentials of <4% sites C13 and D16B were significantly higher in June, July and October. Maximum estimated smoldering potentials were measured at sites C13 and D16B: D16B = 24% ±4 (June 0-10cm), 17.2% (October 0-5cm), 4.7% (July), C-13 = 15.3% (July 0-5cm), 12.1% (October 0-5cm), 5.8% (June). Smoldering potential was influence by site location but not month or the manipulation of the water table. Site location explained 27% of the variation in SP ($F=4.748$, $df=3,28$, $p=0.008$) (Figure 2.2c) (Table 2.3).

DISCUSSION

Determining the moisture content to mineral content ratio may allow us to measure an increase or decrease in the likelihood of a smoldering during a fire event. If soil moisture content is high relative to mineral content smoldering propagation will not occur or will quickly extinguish. My study found that the moisture content to mineral content ratio was sufficient at three of the study sites and it is unlikely that a smoldering event would occur at these sites.

Based on previous studies, high moisture reduced smoldering potential. In my study smoldering in July may be possible at all study sites as the mean MC was <160% at each site. Furthermore, MCs at nonmanipulated site D16B averaged <160% across all months thus this site may be highly susceptible to smoldering. Frandsen (1997) proposed that smoldering would not be sustained in peat with a MC > 160%. Prat-Guitart et al. (2016) found that MC <160%, smoldering propagation across horizontal distances of more than 10cm is possible.

My results linking a plant community to low MC were similar to those found in previous studies. Nonmanipulated site D16B, site with lowest MC, has experienced an influx of the canebrake (*Arundinaria gigantean)* over the two years of the GHG study (Figure 2.3). Moisture content of soil was influenced by plant community with communities such as *Sphagnum* creating high soil moisture environments and grasses creating low soil moisture environments.

My measurements indicated that the study sites have higher minimum MC than those reported by Reardon et al. (2009) -- 64%-167%. I found the minimum MC ranges were 112% to 228%, 87% to 200%, 90% to 143%, 157% to 325%, at the nonmanipulated sites (D16A, D16B), and manipulated sites (C13, C14) respectively. The water table depth (WTD) at manipulated site C13 (20-30 cm below soil surface) was consistently higher than the WTD at nonmanipulated site D16A (30-40 cm below soil surface) however, WTD has not been clearly linked to MC (Reardon, Curcio, and Bartlette 2009) and statistical analysis of my data indicate there is not a linear relationship between WTD and MC. It is likely that location may be the dominant influence on MC at site C13.

Conclusions

In conclusion, I found that moisture content and mineral content are highly variable among sites and across time within sites which underscores the need to monitor mineral content and mineral content in real time. This information is necessary to accurately assess smoldering potential and that setting artificial mineral content as *in situ* experiments may underestimate smoldering potential. The high smoldering potential calculated for nonmanipulated site D16B may be a result of the unique plant community. This plant community may have influenced soil moisture and mineral contents through mechanisms that were not be teased out from my study and needs further exploration.

Future directions

Overall, this wetland restoration project in the form of raising the water table level has reduced the smoldering potential as well in addition to lowering net soil greenhouse gas emissions. Unfortunately, the water table data cannot be used as a proxy for soil moisture content thus, real time data of soil moisture content should be collected at consistent intervals to monitor the smoldering potential. Additionally, fires beginning in unrestored sites have the potential to spread into restored sections of the refuge so smoldering potentials should be monitored at non-restored sites.

TABLES AND FIGURES

Figure 1. 1 Shaded area indicates peatlands along the eastern coast of the United States.

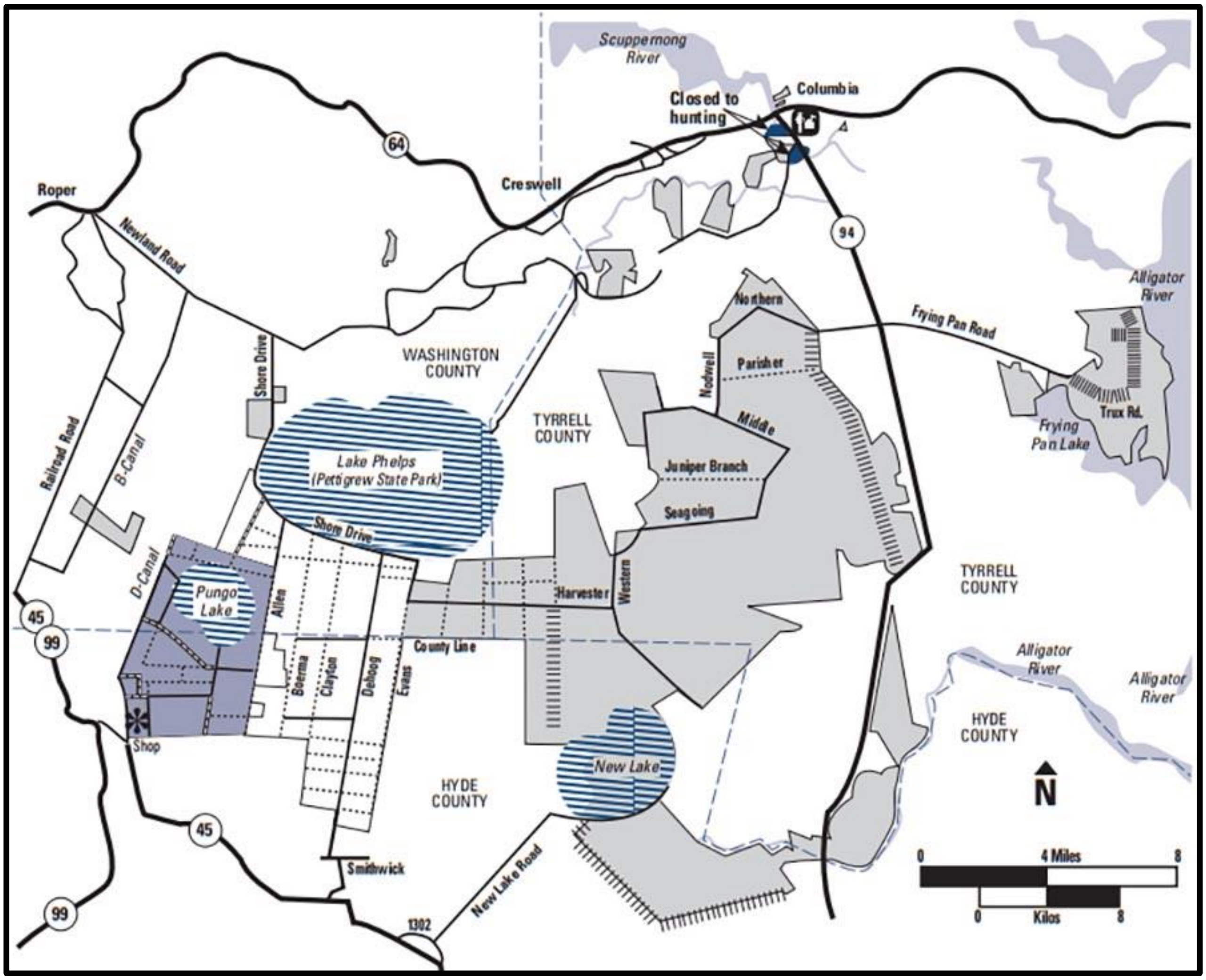

Figure 1. 2 Pocosin lake national wildlife refuge.

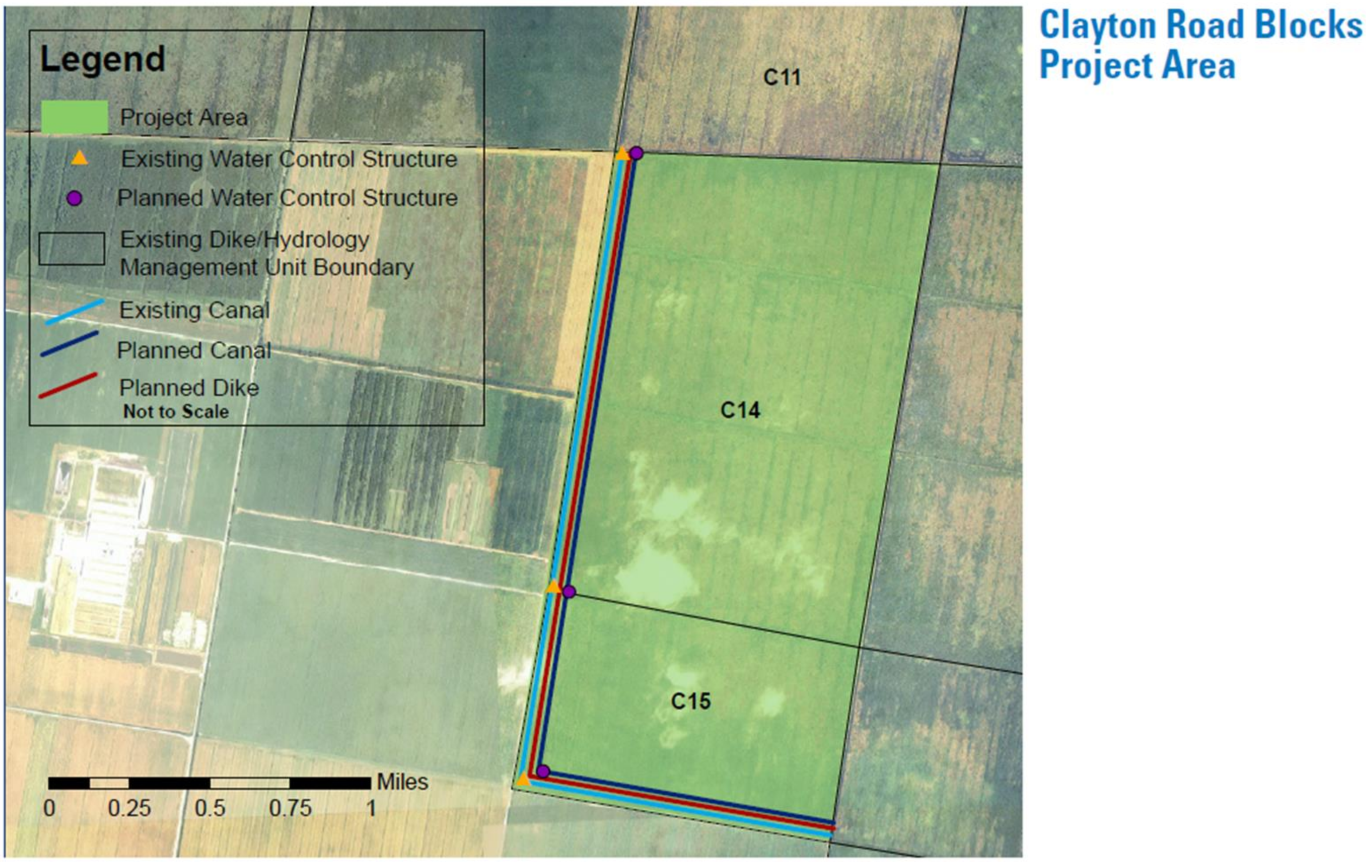

Figure 1. 3 Map showing location of dyke and canal constructed to facilitate water table manipulation. Map courtesy of the Nature Conservancy.

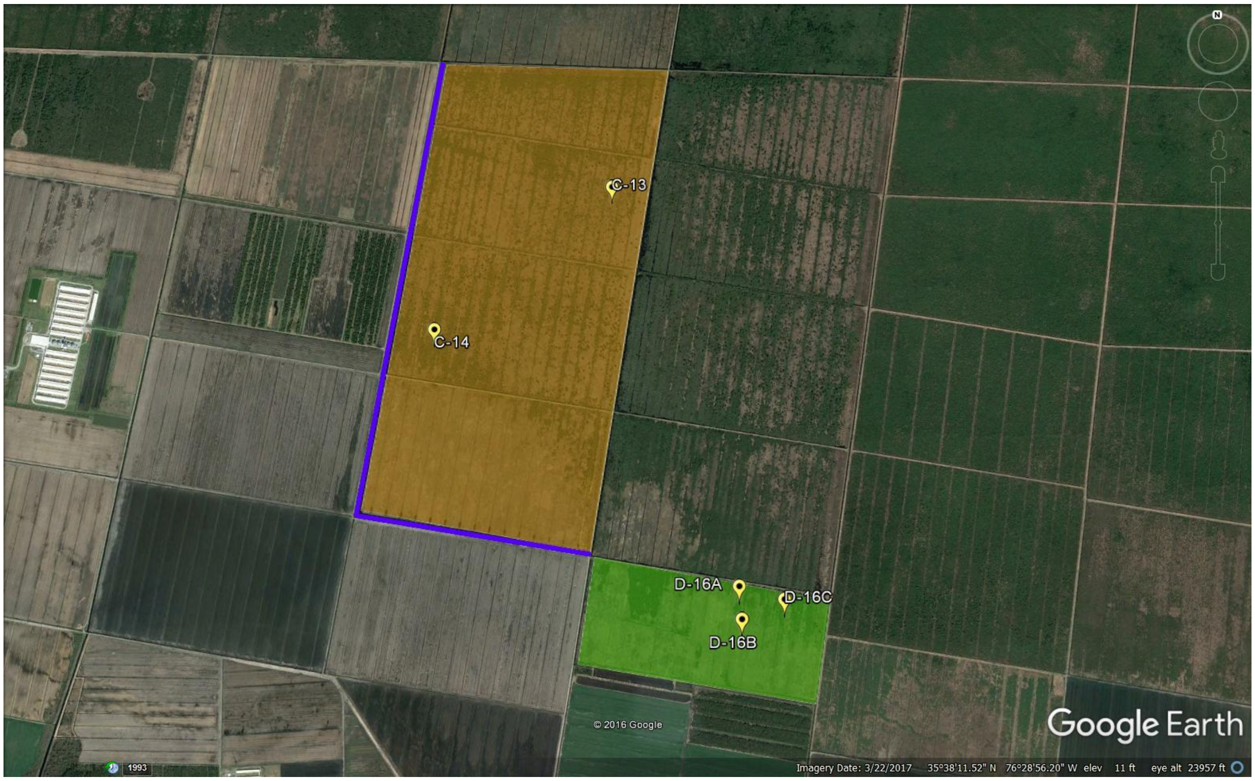

Figure 1. 4 Map describing the location of the four study plots. The manipulated sites are shaded yellow. Nonmanipulated shaded green. Blue line represents canal installed to facilitate water table manipulation of the study sites.

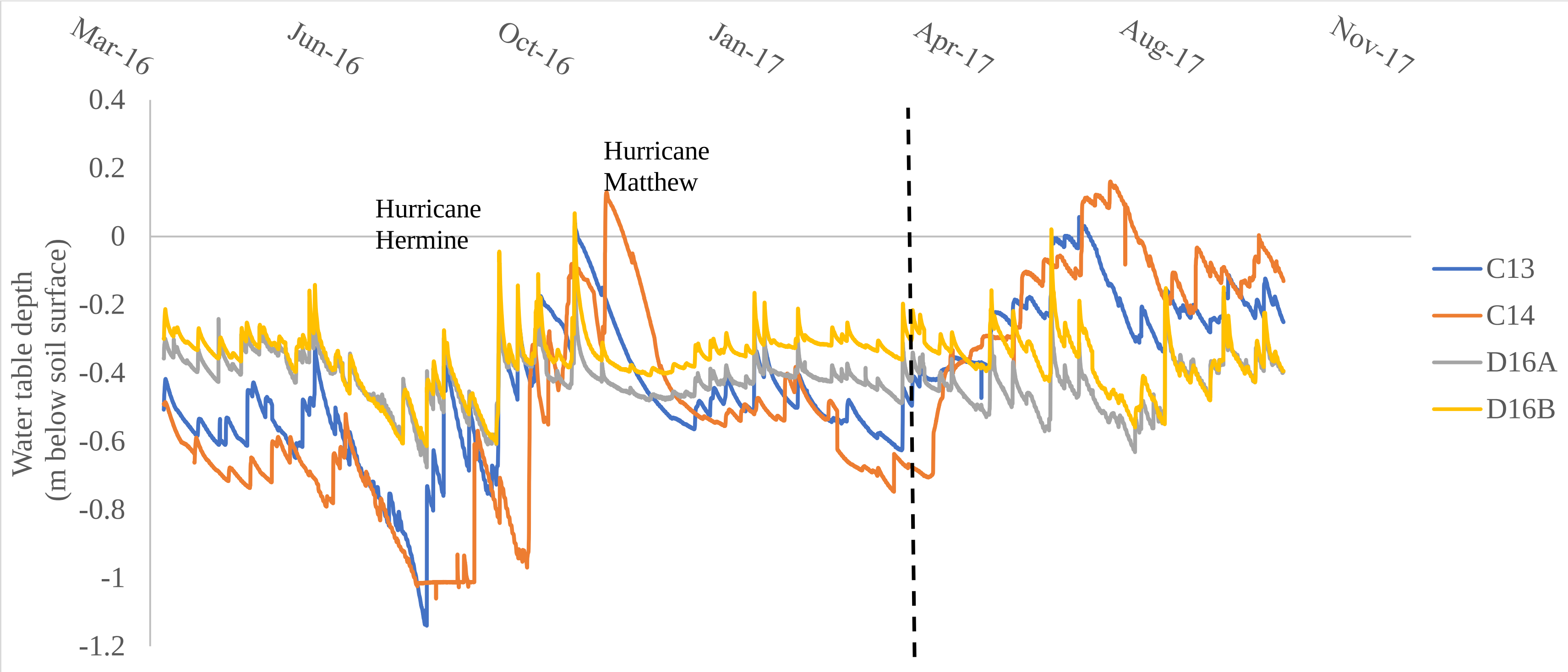

Figure 1. 5 Water table depth shown as meters above/below the soil surface. Data recorded hourly by continuous data logger. Dashed line indicates date of water table manipulation.

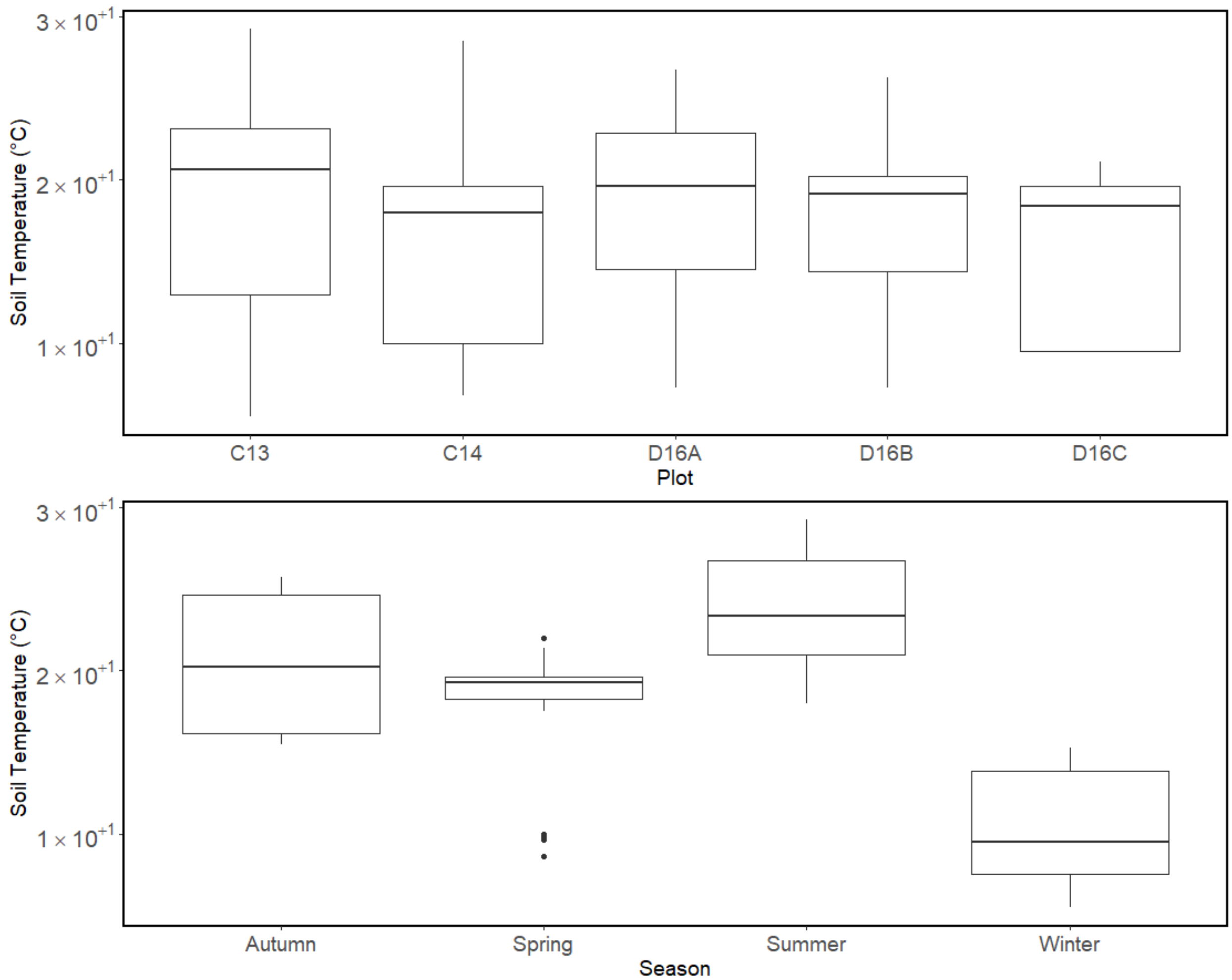

Figure 1. 6 Box plots of soil temperatures over the course of the study across (upper) plot location, (lower) season.

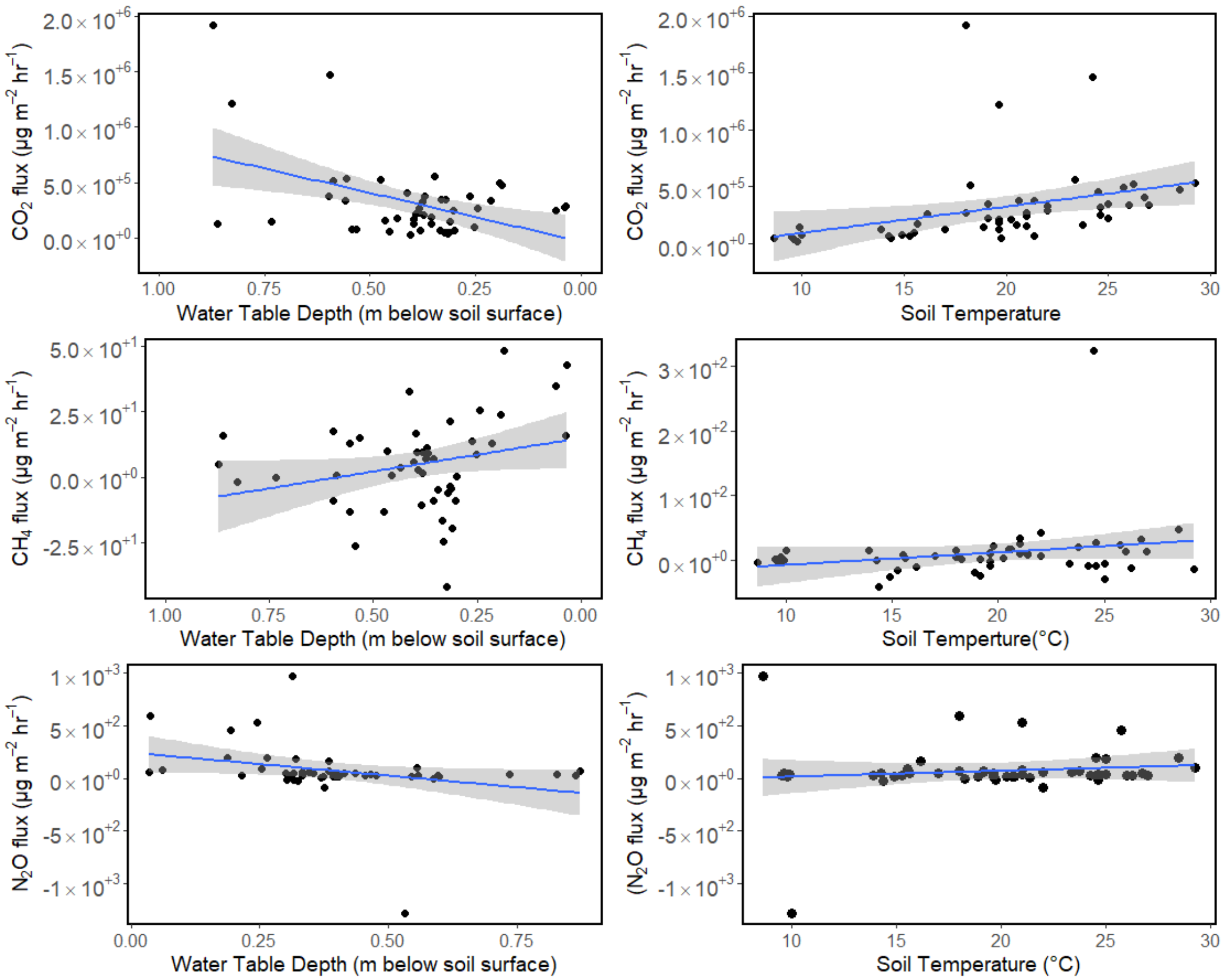

Figure 1. 7 Relationship between GHG flux rate and water table depth (left panels), GHG flux rat and soil temperature (right panels). Shading represents 95% confidence intervals.

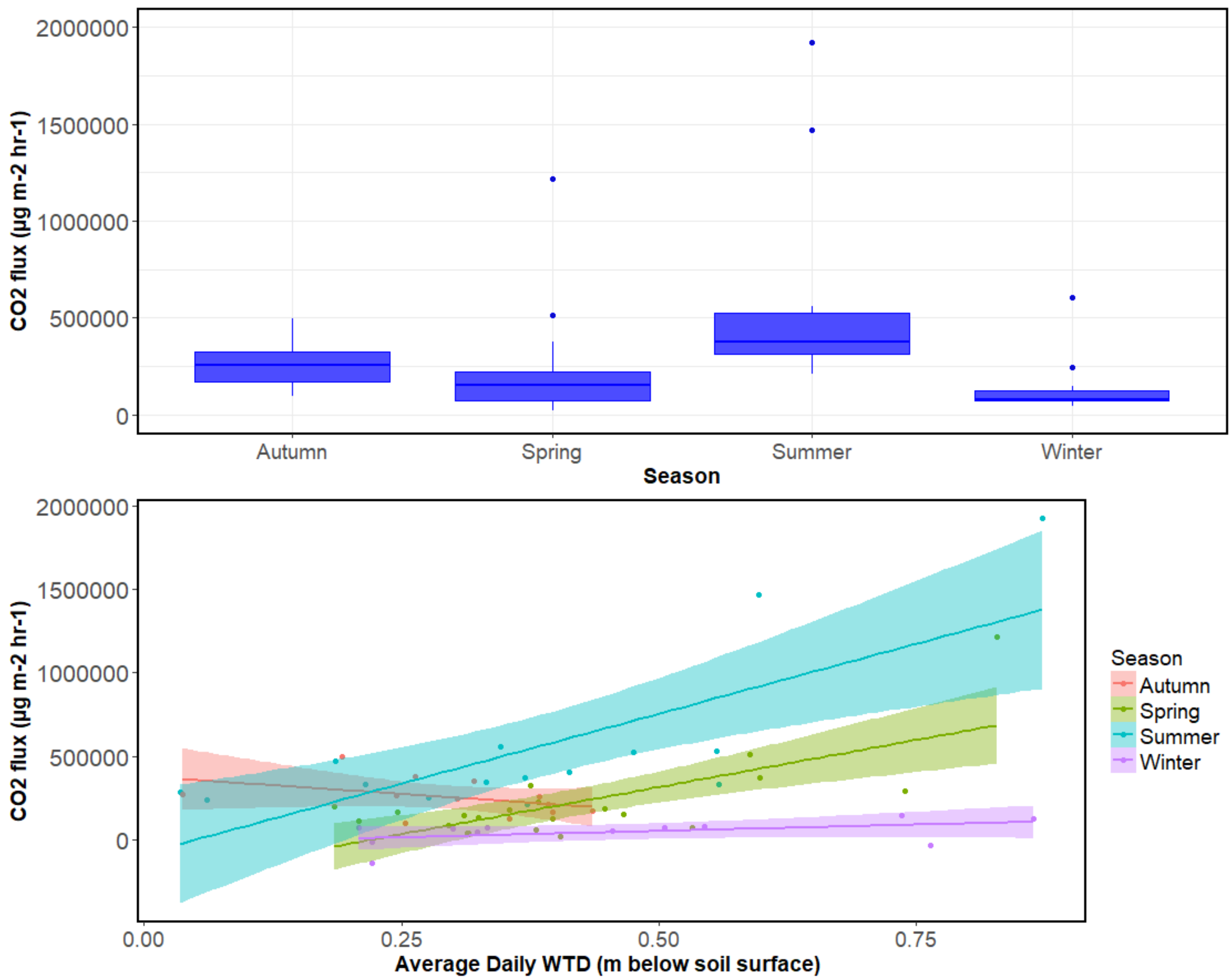

Figure 1. 8 Carbon dioxide flux rate over course of study as it relates to (upper panel) season, (lower panel) water table depth and season. Shading represents 95% confidence intervals.

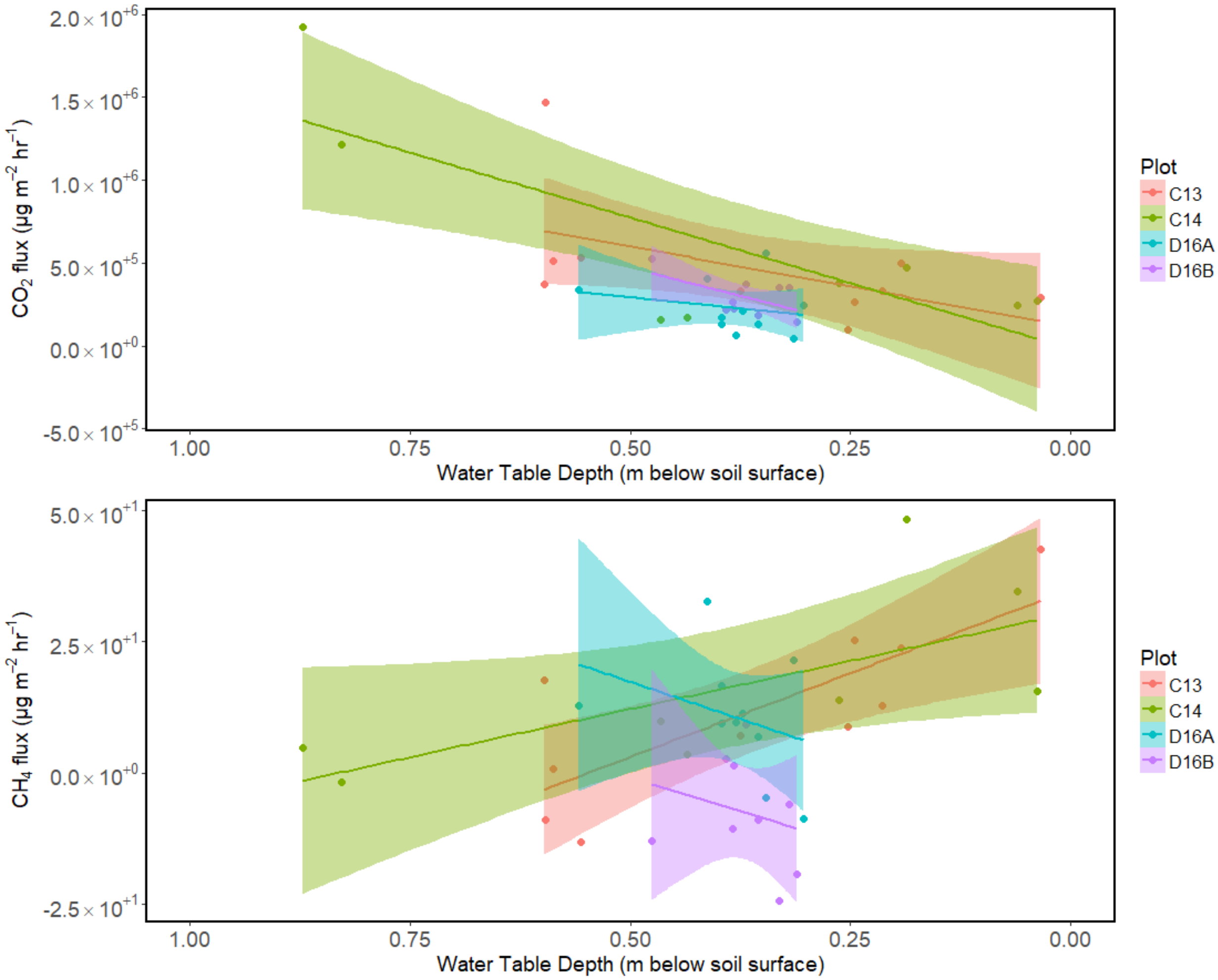

Figure 1. 9 Carbon dioxide flux rate as it relates to water table depth and plot location. Shading represents 95% confidence intervals. Figure 1.16 Methane flux rate as it relates to water table depth and plot location. Shading represents 95% confidence intervals.

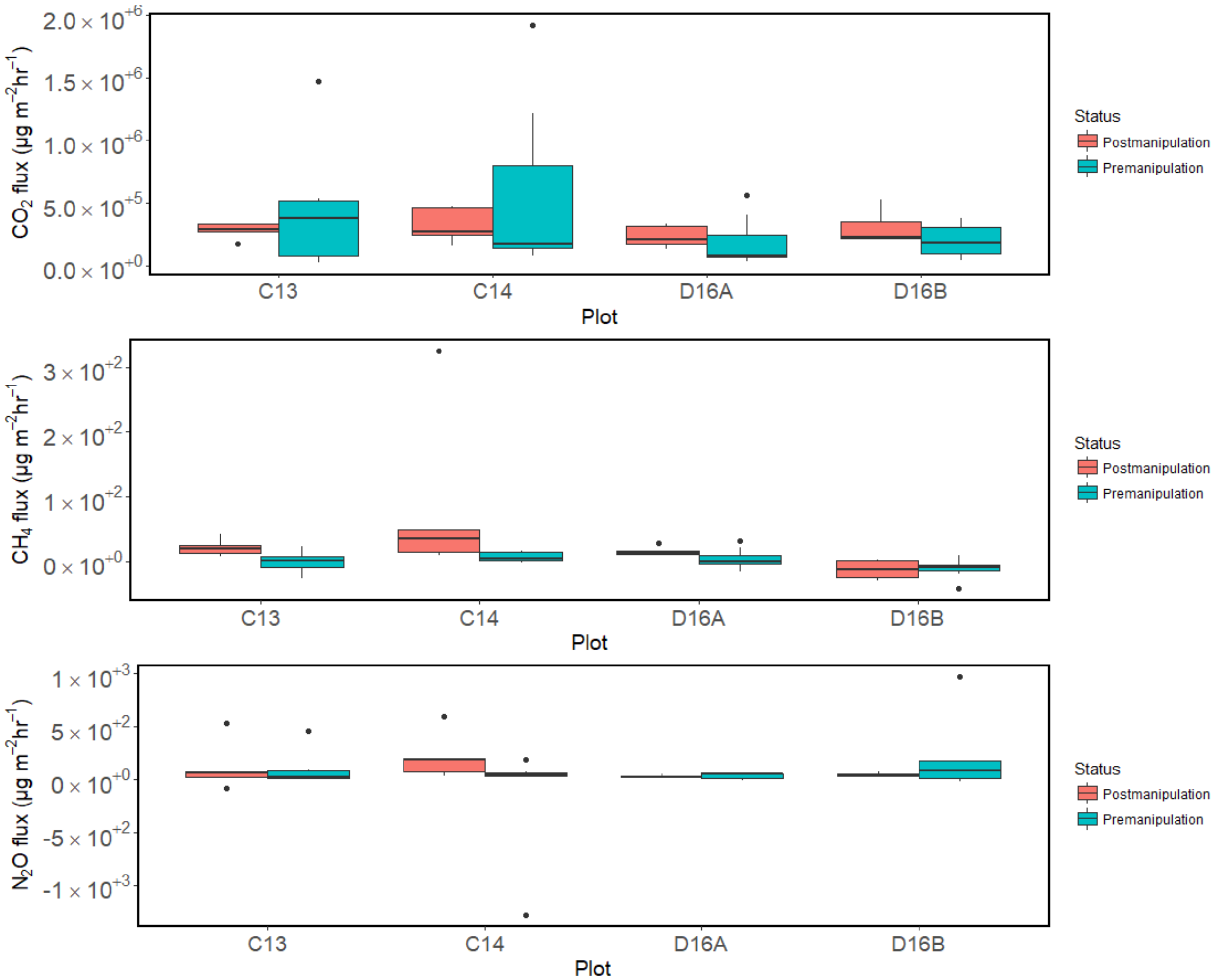

Figure 1. 10 Pre- and post-manipulation comparison of carbon dioxide (top panel), methane (center panel), nitrous oxide (bottom panel) flux rate as it relates to plot location.

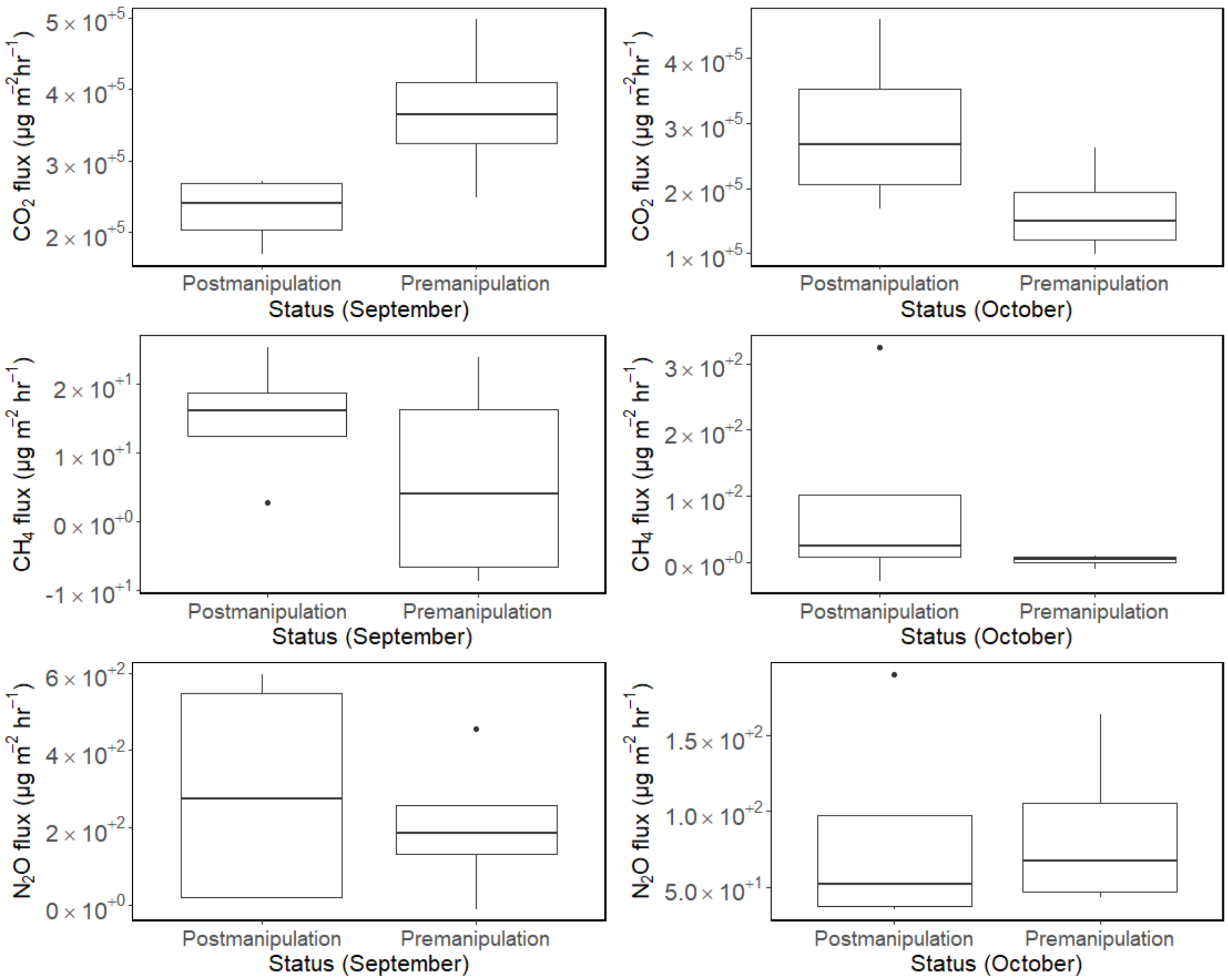

Figure 1. 11 Hurricane influence on GHG fluxes. Fluxes are averaged from all study sites. Premanipulation fluxes raised water table conditions due to rainfall occurring during hurricanes. Postmanipulation fluxes raised water table conditions due to manipulation.

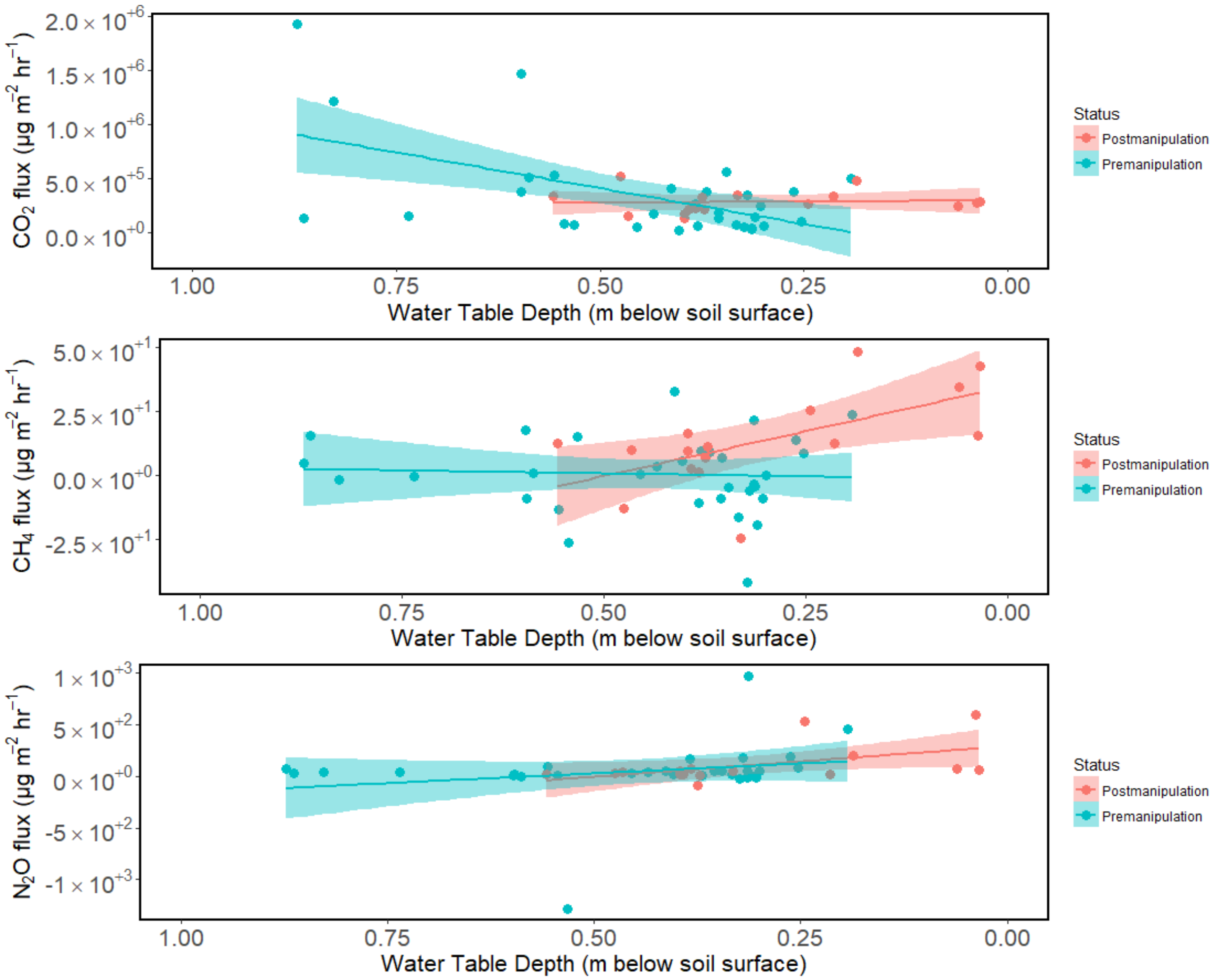

Figure 1. 12 Carbon dioxide (top panel), methane (center panel), nitrous oxide (bottom panel) flux rate as it relates to water table depth and manipulation status. Shading represents 95% confidence intervals. Data represents pre- and post- manipulation subset data.

Table 1. 1 ANOVA results of the relationship of average CO_2 flux and water table depth and season

Factor	Sum of Squares	d.f.	*F*-ratio	*P*-value
Avg Daily WTD	7.7e+11	1	27.7012	4.2e-6***
Season	2.3e+12	3	14.3675	1.2e-6***
Avg Daily WTD:Season	1.3e+12	3	6.1011	0.001**
Residuals	2.2e+12	43		

***Significant at the 0.001 level **significant at the 0.01 level

Table 1. 2 ANOVA results of the relationship of average CH_4 flux and water table depth and season

Factor	Sum of Squares	d.f.	*F*-ratio	*P*-value
Avg Daily WTD	897.8	1	3.0209	0.089
Season	1544.1	3	1.7319	0.017
Avg Daily WTD:Season	3039	3	3.4087	0.025*
Residuals	12778.6	43		

*Significant at the 0.05 level

Table 1.3 ANOVA results of the relationship of average CH_4 flux and water table depth and plot location

Table 1. 3 ANOVA results of the relationship of average CH_4 flux and water table depth and plot location

Factor	Sum of Squares	d.f.	*F*-ratio	*P*-value
Avg Daily WTD	897.8	1	4.3260	0.043*
Plot	4690.4	4	5.6504	0.001**
Avg Daily WTD:Plot	4162.7	4	5.0146	0.002**
Residuals	8508.6	41		

**Significant at the 0.01 level, *significant at the 0.05 level

Table 2. 1 Moisture and mineral content of peat soils at PLNWR across two seasons: Summer and Autumn 2017. Three soil cores per site were obtained each sampling period and averaged to calculate moisture and mineral content at each site.

	Average of %Moisture				**Average of %Mineral**			
	June	**July**	**September**	**October**	**June**	**July**	**September**	**October**
C13 (manipulated)	232±44	106±26	193±5.3	159±47	8.4±0.4	7.3±1.6	1.6±0.34	2.7±0.8
C14 (manipulated)	332±9.6	172±20	312±29	210±49	6.6±4.8	14.4±1.7	3.38±0.82	7.1±8
D16A (unmanipulated)	229±1.2	115±3.8	242±24.4	208±17	22±20	8.5±0.5	2.5±1.2	2.7±1.5
D16B (unmanipulated)	116±3.6	111±30	150±9.6	137±7	20±3.3	15.1±8.5	10.6±1.18	10.9±6.5

Table 2. 2 Reported carbon losses from fire events with large smoldering propagation

Source	Fire Location	Carbon Loss	CO2 emissions from fossil fuels	Time to replace
Page et al. (2002)	Indonesia 1997	0.81 to 2.57 Gt From 730000 ha	13 to 40%	…
Mickler et al. (2017)	Evans Road Fire 2008 North Carolina	0.0095 Gt from 16814 ha	<1%	…
Lukenback et al. (2015)	Utikuma Complex Alberta Cananda 2011	0.018 Gt from 90000 ha	<1%	600 years[1]

Table 2. 3 ANOVA result of the relationship of smoldering potential and plot location

Factor	Sum of Squares	d.f.	*F*-ratio	*P*-value
Site	0.047090	3	4.7484	0.008439**
Residuals	0.092559	28		

**Significant at the 0.01 level

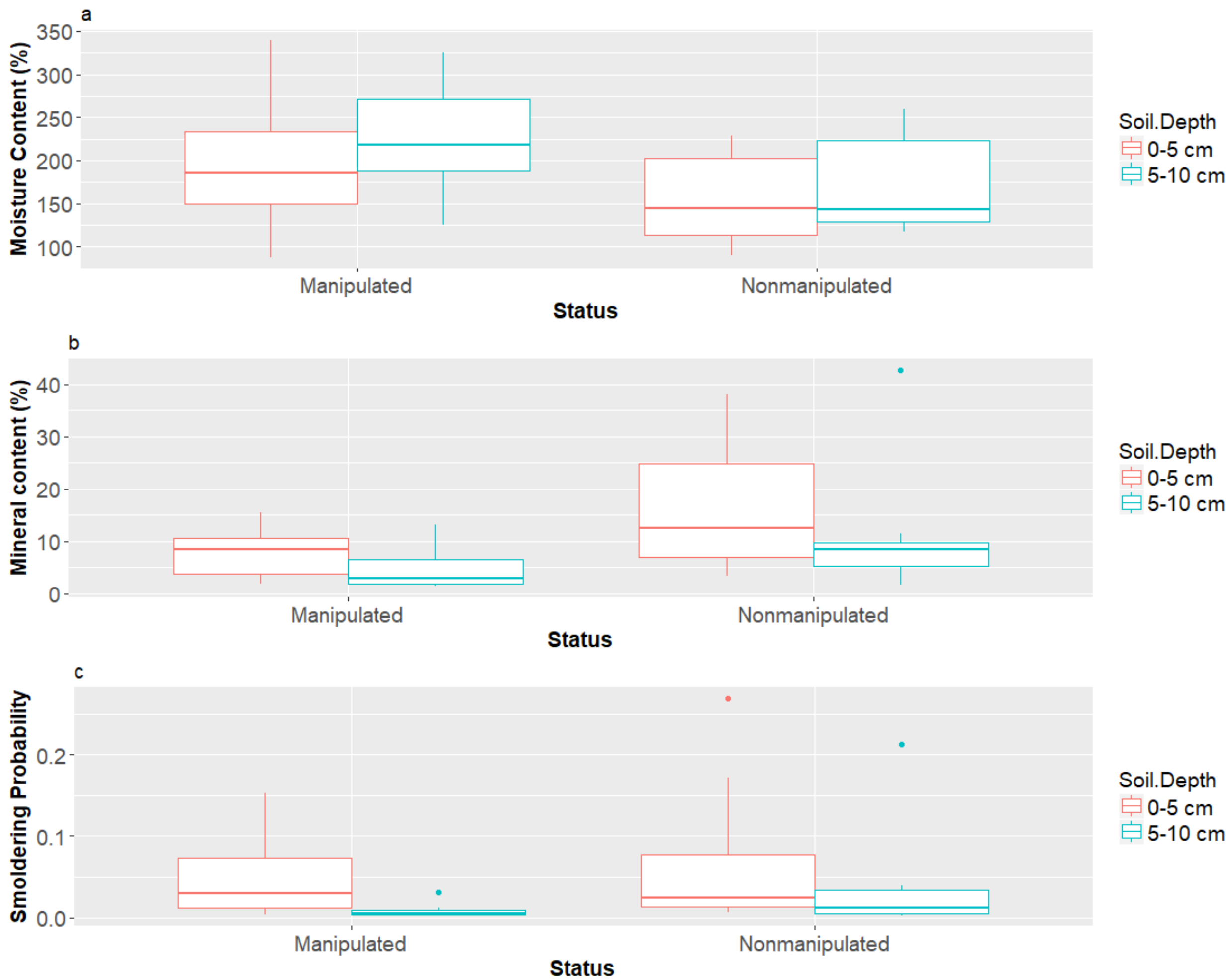

Figure 2. 1 Box plots of (a) moisture content (%), (b) mineral content (%) and (c) smoldering potential at manipulated and nonmanipulated sites for June, July, September and October according to soil depths 0-5 and 5-10 cm.

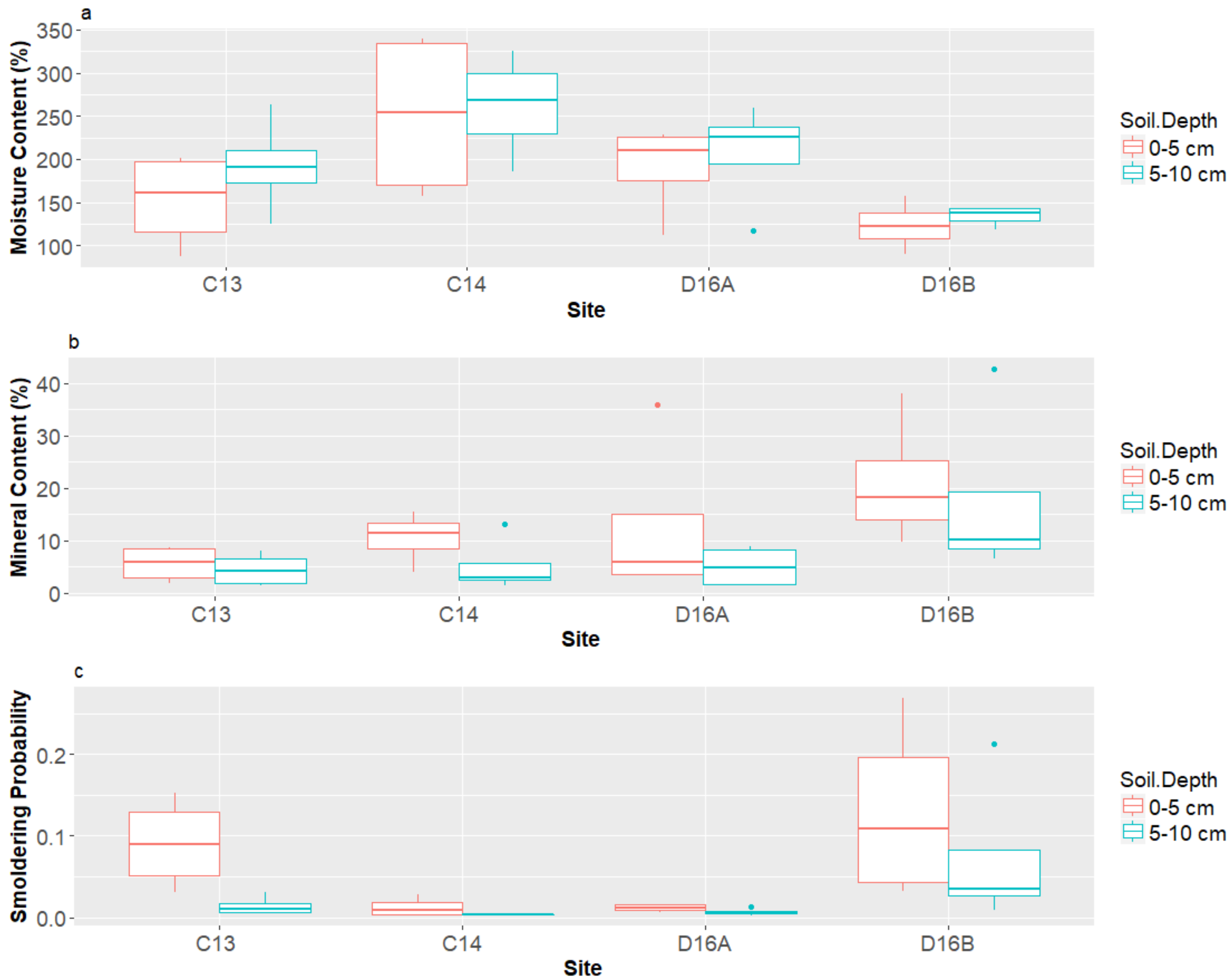

Figure 2. 2 Box plots of (a) moisture content (%), (b) mineral content (%) and (c) smoldering potential of individual study plots for June, July, September and October according to soil depths 0-5 and 5-10 cm.

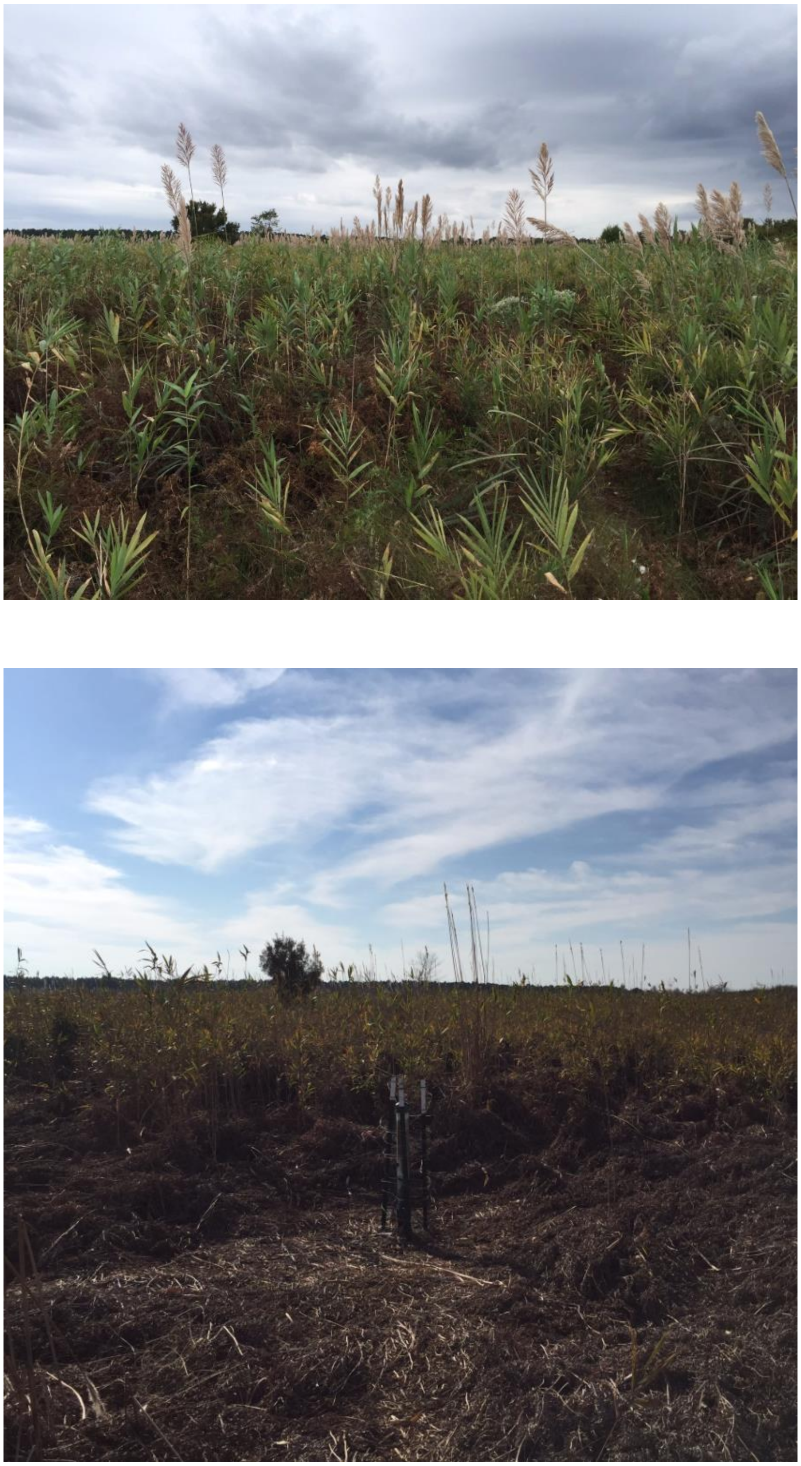

Figure 2. 3 Photos of nonmanipulated site D16B Top taken September 29, 2017 (top) and December 15, 2015 (bottom)

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