Elevated salinity and water table drawdown significantly affect greenhouse gas emissions in soils from contrasting land-use practices in the prairie pothole region

Shayeb Shahariar (shayeb.shahariar@usask.ca)
Univeristy of Saskatchewan https://orcid.org/0000-0001-6559-0548
Richard Farrell
University of Saskatchewan
Raju Soolanayakanahally
Agriculture and Agri-Food Canada
Angela Bedard-Haughn
University of Saskatchewan

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Author Names and Institutional Affiliations:

Shayeb Shahariar[1]*, Richard Farrell[1], Raju Soolanayakanahally[2], Angela Bedard-Haughn[1]

1) Department of Soil Science, College of Agriculture and Bioresources, University of Saskatchewan, 51 Campus Drive, Saskatoon SK S7N 5A8 Canada
2) Indian Head Research Farm, Agriculture and Agri-Food Canada, Indian Head SK S0G 2K0 Canada

Corresponding Author Name and Address:

Shayeb Shahariar

Department of Soil Science, College of Agriculture and Bioresources, University of Saskatchewan, 51 Campus Drive, Saskatoon SK S7N 5A8 Canada

E-mail: shayeb.shahariar@usask.ca; Phone: +1 306 966 4299

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Author Contributions:

- Shayeb Shahariar conceptualized, designed, and performed the experiments, analyzed the data, prepared all figures and tables, authored and reviewed drafts of the manuscript, and made final editorial decisions regarding all text and graphs, and approved the final draft.
- Richard Farrell contributed reagents and instrument support for GHG analyses, made critical comments, reviewed drafts of the manuscript, and approved the final draft.
- Raju Soolanayakanahally provided technical supports for soil core samples collection, reviewed drafts of the manuscript, made critical comments, and approved the final draft.

- Angela Bedard-Haughn provided all necessary financial support and supplies required for the greenhouse experiment, contributed and guided the fundamental ideas for this work, reviewed drafts of the manuscript, made critical comments, and approved the final draft.

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Highlights

- Studied GHG emissions under elevated salinity and lowered water tables in PPR soils
- GHG emissions were higher in pasture soil than annual crop or short rotation willow
- Increased groundwater salinity significantly increased N_2O, but decreased CO_2 and CH_4
- Lowering the water table decreased CH_4 and CO_2, but increased N_2O up to week four
- Global warming potential was significantly lower in soils from short rotation willow

Abstract

Land-use practices can alter shallow groundwater and salinity, further impacting greenhouse gas (GHG) emissions, particularly in the hydrologically dynamic riparian zones of wetlands. Emissions of CO_2, CH_4, and N_2O were estimated in soil cores collected from two prairie pothole region (PPR) sites with three adjacent land-use practices (i.e., annual crop = AC, pasture = PA, and short rotation willow = SRW) and treated with declining water table depths (2 to 26 cm), and salinity (S0 = control, S1 = 6 mS cm^{-1}, and S2 = 12 mS cm^{-1}) in a microcosm experiment. Land-use practices significantly ($p < 0.001$) affected GHG emissions in soils from both sites in the order of PA > AC = SRW. Compared to the control, emissions of CO_2 and CH_4 were significantly lower under higher salinity treatments (i.e., S1 and S2), while N_2O was significantly higher ($p < 0.05$). Emissions under declining groundwater table depths were significantly ($p <$

0.001) variable and specific to each gas, indicating the impacts of shifted soil moisture regime. Overall, the CO_2 and CH_4 emissions increased up to week four and then decreased with declining water table depths, whereas N_2O emission increased up to a maximum at week six. The soils from SRW had considerably lower global warming potential compared to AC and PA. Groundwater salinity in soils from contrasting land-use in the PPR has significant impacts on GHG emissions with potential for crucial climate feedback; however, the magnitude and direction of the impacts depend on hydrology.

Keywords: greenhouse gas (GHG) emission**,** land-use practice, shallow groundwater table, salinity, wetland soil, prairie pothole region (PPR)

1. Introduction

Agroecosystem soil C and N cycling contribute significantly to atmospheric carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) emissions and, thus, global warming (IPCC 2018; Smith et al. 2008). The North American prairie pothole region (PPR) is characterized by relatively small and highly productive wetlands embedded within an agriculture-dominated landscape. The PPR delivers essential ecosystem services such as improving soil and water quality, storing water, reducing soil erosion, and providing habitat for wildlife, especially waterfowl (Gleason et al. 2008). Salt dynamics within the PPR wetlands are driven by hydrology, which cycles seasonally and responds to land-use practice changes (Nachshon et al. 2013). Vegetation in the riparian zone pulls water from the soil and transfers it to the atmosphere via transpiration (Millar 1971), resulting in a gradual decline in the groundwater table (GWT). In turn, a declining GWT can increase soil salinity and deposit soluble salts at the soil surface (Arndt & Richardson 1989). Land-use affects both the production and consumption of GHG through its influence on wetland soil hydrology. For instance, wetland-riparian zones that are drained and cropped likely would

have minimal CH_4 production because this practice promotes aerobic conditions that do not favor methanogenesis (Smith et al. 2003). Conversely, the same catchment would have a higher likelihood of emitting N_2O due to a combination of N-fertilizer additions and moist, but unsaturated soil moisture conditions (Davidson et al. 2000). Hence, changes in land-use practice can increase the potential for greenhouse gas (GHG) emissions and diminish the capacity of PPR wetlands to deliver ecosystem services (Gleason et al. 2009).

Land-use practice can strongly influence soil-derived GHG emissions (Liebig et al. 2005; Schaufler et al. 2010; Tangen et al. 2015). In general, wetlands have a greater GHG emission potential than forestlands, croplands, and grasslands (Oertel et al. 2016); however, the amount of CO_2, CH_4, and N_2O emitted vary depending on the type of vegetation and environmental conditions (Kayranli et al. 2009). The production of GHG in wetlands is controlled by highly variable abiotic factors that are themselves affected by land-use; these include the soil moisture and groundwater regime, the period of inundation, redox conditions, and groundwater salinity (Marton et al. 2012). The land-use practice also affects soil biological processes that regulate GHG emissions by influencing the composition of soil microbial and plant communities and the availability of organic substrates (Tangen et al. 2015). Moreover, riparian land-use practice affects the microclimate and soil properties that can influence the production/consumption and GHG emission (Moore et al. 2017). Consequently, land-use practices that affect dynamic wetland riparian zones can significantly alter the amount of GHG released into the atmosphere (Vidon 2010).

Land-use practice can alter soil organic carbon (SOC) dynamics and, in turn, GHG emissions (Kooch et al. 2016; Lang et al. 2010; Merino et al. 2004). Agroforestry is a promising land-use practice that can increase above- and below-ground C stocks, mitigate N_2O and CO_2 emissions, and increase the CH_4 sink potential when compared to cropland; unlike cropland, agroforestry

has lower losses of aboveground biomass via harvest and lower CO_2 emissions from soil organic matter (SOM) decomposition (Mutuo et al. 2005). One study (Baah-Acheamfour et al. 2016) recommended that incorporating agroforestry and grassland cover into agricultural lands can reduce CH_4 and N_2O emissions. Parmar et al. (2015) also found that "short-rotation" forestry can contribute to GHG savings via reduced soil respiration losses. Thus, establishing perennial agroforestry systems such as short rotation willow (SRW) in the riparian zones of PPR wetlands may deliver GHG mitigation benefits. However, the effects of agroforestry practices on soil N_2O and CH_4 emissions are poorly understood (Albrecht & Kandji 2003). It has also been suggested that SOC could be sequestered by re-establishing permanent vegetation (i.e., grass) in PPR wetlands (Bedard-Haughn et al. 2006). However, it is unclear how the establishment of perennial SRW vegetation in the marginal riparian zones of the semi-arid PPR wetlands affects GHG emissions under dynamic soil hydrology (e.g., GWT) and salinity.

The effects of a fluctuating GWT on GHG emissions from peatlands (Berglund & Berglund 2011; Blodau et al. 2004; Updegraff et al. 2001) and riparian mineral wetlands (Mander et al. 2015) have been well studied. The effects of salinity on GHG emissions associated with a land-use change (Martin & Moseman-Valtierra 2015; Sheng et al. 2015), or depth to the GWT (Ardón et al. 2018; Mander et al. 2011) have also been studied, albeit mainly in coastal wetlands. These studies have variable results. For instance, in one microcosm experiment, artificial salinity treatments suppressed CO_2 emissions under both drought and flooded conditions, CH_4 emissions increased in flooded conditions only, and the impacts of salinity were conditional on hydrologic treatments for N_2O (Ardón et al. 2018). In contrast, in a tidal forest soil, salinity inhibited CH_4 production but increased CO_2 and N_2O emissions (Marton et al. 2012). In another microcosm study using semi-arid cropland soil from Australia, salinity increased N_2O emissions and reduced CO_2 and CH_4

emissions; however, increasing soil moisture increased CO_2, increased CH_4 – but only up to 75% water-holding capacity, and had no effect on N_2O emissions (Maucieri et al. 2017). In contrast, in the riparian zones of mineral wetlands, flooding increased CH_4 emissions, and CO_2 and N_2O emissions increased as the depth to GWT decreases (Mander et al. 2015). In constructed wetlands, CH_4 emissions were reduced, and N_2O emissions amplified at high salinity (>10 ‰), whereas the CO_2 emissions were greatest at intermediate salinity, i.e., ~5 ‰ (Sheng et al. 2015). Nevertheless, studies on the combined effects of GWT and salinity on GHG emissions under contrasting land-use practices within mineral wetlands in the PPR are scarce.

Depending on various factors, wetland soils can either be a source or sink for GHG (Beetz et al. 2013). Examining GHG emissions under the combined effects of fluctuating water table and salinity in the context of contrasting land-use practices will improve our ability to develop best management practices and mitigation strategies while advancing agricultural sustainability in the PPR. Therefore, the objective of this microcosm study was to examine the effect of a declining groundwater water table—with different groundwater salinity levels—on GHG emissions from riparian zone soils collected from different land-use practices in the PPR.

2. Materials and Methods

2.1. Site Description and Collection of Intact Soil Cores

A controlled microcosm experiment was conducted to determine the influence of groundwater salinity and declining water table level on soil-derived emissions of CO_2, CH_4, and N_2O. Soils were collected from sites managed under three different land-use practices at two sites in the PPR. Both sites (Site A and Site B) were located near the Agriculture and Agri-Food Canada Indian Head Agroforestry Development Centre at Indian Head, Saskatchewan, Canada (N 50° 30.605'; W 103° 43.011') (Supplementary Figure 1). Soils at both sites were classified as Oxbow

Association, non-calcareous Black Chernozems developed on loamy glacial till in a landscape with level to gentle rolling (0–10% slope) topography (Saskatchewan Soil Survey Staff 1986). At both sites, the SRW treatments (*Salix dasyclados* Wimm, popularly known as 'India') were established in June 2013 in the marginal fallow riparian zones. The pasture treatment (PA) comprised of a mix of alfalfa (*Medicago sativa*) and bromegrass (*Bromus madritensis*) that had been established in 2001–2003. Both SRW and PA areas were located (Supplementary Figure 1) adjacent to the cropped area that was seeded with oat (*Avena sativa*).

The soils at Site A were non-saline, with ECs ranging from 0.6 to 1.9 mS cm^{-1}; soils at Site B were non- to slightly saline, with ECs ranging from 1.0 to 2.6 mS cm^{-1} (see Supplementary Table 1). Intact soil cores (n = 3) were collected from each of the three land-use treatments at Sites A and B (i.e., annual crop [AC], pasture [PA], and short-rotation willow [SRW]) in mid-August 2015. Intact soil cores were used to avoid the disturbance produced by sieving (Reichstein et al. 2005). The soil cores were collected using a truck-mounted hydraulic punch (Giddings Machine Company Ltd., Windsor, CO, USA) fitted with cylindrical (30-cm tall × 9-cm i.d.) PVC sleeves. Cores were collected three years after SRW plantation (i.e., at the end of the first rotation cycle of SRW) to capture land-use practice effects on soil. The overlying litter-fibric-humic layer and grasses were removed before collecting the soil cores from the field. All soil cores were collected from the riparian zones. For the SRW, all soil cores were collected within a 1-m radius of the root zone between two planted rows. In total, 54 soil cores (2 sites × 3 land-use practices × 9 reps) were collected and transported in coolers to the University of Saskatchewan where they were preserved frozen (at -20°C) until the start of the incubation study. Additional soil cores (0–30 cm depth; 9-cm i.d.) from each sampling location were collected and analyzed to determine

soil physical and chemical properties (see Supplementary Table 1). Bulk density samples were collected using a hand-held core sampler (3-cm tall × 5.4-cm i.d.).

2.2. Initial Soil Characterization

Soil physiochemical properties were determined prior to starting the microcosm experiment. Each soil was divided into three subsamples, which were processed as follows: (1) one subsample was air-dried, ground, passed through a 2-mm sieve, and analyzed for particle size distribution, cation exchange capacity (CEC), pH, electrical conductivity (EC), and ammonium acetate extractable N and P; (2) the second subsample was air-dried, finely ground with a ball mill, and analyzed for organic- and total-C and total-N; and (3) the third subsample was frozen until it was analyzed for water-extractable organic carbon (WEOC) and water-extractable organic nitrogen (WEON). Samples collected for bulk density measurement were weighed, oven-dried at 105°C for 24 h, cooled to room temperature in a desiccator, and reweighed. Bulk density was determined by dividing the oven-dry weight of the soil by the volume (74.7 cm^3) of the core sampler.

Soil physiochemical analyses were carried out using the procedures described in *Soil Sampling and Methods of Analysis* (Carter and Gregorich (2008). The modified pipette method (Kroetsch & Wang 2008) was used to determine soil particle size distribution. Cation exchange capacity was determined using ammonium acetate at pH 7, followed by colorimetric analysis using a Technicon Auto-Analyzer (Technicon Industrial Systems; Tarrytown, NY, USA) (Hendershot et al. 2008a). Soil pH was determined in a 1:2 (w/v) soil:deionized-water suspension using a digital pH meter (Oakton™ PC700 pH/mV/conductivity meter; Oakton Instruments, Vernon Hills, IL, USA) (Hendershot et al. 2008b). EC was determined in a same extract after 1 hour shaking with

an end-over-end shaker; filtrate (No. 42, Whatman Inc., Piscataway, NJ) was measured using a digital EC meter (PC700 pH/mV/conductivity, Oakton, Vernon Hills, IL, USA) (Miller & Curtin 2008). Ammonium (NH_4^+-N), nitrate (NO_3^--N), phosphate (PO_4^{3-}-P), and sulfate (SO_4^{2-}-S) were measured using a 1M ammonium acetate (buffered at pH 7) extraction followed by colorimetric analysis for NH_4^+-N, NO_3^--N, PO_4^{3-}-P via Technicon Auto-Analyzer (Technicon Industrial Systems, Tarrytown, NY, USA), and SO_4^{2-}-S via Microwave Plasma-Atomic Emission Spectrometer (Model 4100, Agilent Technologies, Santa Clara, CA, USA) (Simard 1993). Total soil carbon (TSC) and soil organic carbon (SOC) were determined by dry combustion—following HCl fumigation to remove carbonates—using a Leco-2000 CNS analyzer (Leco Corporation, St. Joseph, MI, USA) (Skjemstad & Baldock 2008). Total nitrogen (TN) was determined using dry combustion with a Leco C632 CNS analyzer (Leco Corporation, St. Joseph, MI, USA) (Rutherford et al. 2008). Water extractable organic C and WEON were determined by gently mixing defrosted soil (20 ± 1 g) with 30-mL of 5 mM $CaCl_2$, filtering the suspension through a 0.45-µm polycarbonate membrane filter (Whatman Inc., Piscataway, NJ, USA), and measuring total C and N in the filtrate using a TOC-VCPN analyzer (Shimadzu Scientific Instruments, Kyoto, Japan) (Chantigny et al. 2008).

2.3. Experimental Design

The microcosm incubation experiment was set up in the greenhouse at the University of Saskatchewan using a nested experimental design (Krzywinski et al. 2014; Schielzeth et al. 2013); the experiment was conducted over nine weeks. The 54 soil cores were arranged into following: 2 sites × 3 land-use practices × 3 groundwater salinity treatments (control = 0.3 mS cm^{-1}, S1 = 6 mS cm^{-1}, and S2 = 16 mS cm^{-1}) × 3 replicates (Supplementary Figure 2). Each experimental unit consisted of a 19-L plastic (PVC) bucket (38.1 cm tall × 30.48 cm i.d.) containing a 2.5-cm thick

layer of gravel, 17-L of synthetic groundwater, and a single intact soil core—the bottom of which was wrapped in 1-mm mesh fiberglass screen to hold the soil securely (Figure 1). The PVC cylinders housing the soil cores were drilled with a uniform series of 3-mm holes which allowed for movement of the synthetic groundwater into and out of the soil core.

The dominant salts present in the soil and groundwater in the Prairie region of Canada and the northern United States are Na_2SO_4, KCl, $CaCl_2$, and $MgSO_4$ (Last & Ginn 2005). Thus, the synthetic groundwater treatments were prepared using a 5:2:12:14 mix of Na_2SO_4:KCl: $CaCl_2$:$MgSO_4$ salts (by weight) in distilled water; the quantity (g) of salts in S2 was double that of the S1 treatment. The control (no added salts) salinity treatment consisted of distilled water alone. Initially, the synthetic groundwater was maintained level with the surface of the soil cores (Figure 1), with subsequent GWT drawdown achieved by manually lowering the water level by 2 cm at the end of the first week and then by 3 cm at the end of each of the next nine weeks (Figure 1).

The EC of the synthetic groundwater was checked weekly to ensure that salinity remained constant. The volumetric soil water content (VSWC) and EC of the experimental soil cores were measured using a digital soil moisture meter (HydroSense II, Campbell Scientific Inc., Logan, UT, USA) at the time of GHG flux measurements. The temperature of the greenhouse chamber was maintained at 20 ± 1°C; relative humidity in the greenhouse ranged from 37.73% to 67.05% (average 50.33%) during the first seven weeks of the experiment, and then from 16.05% to 43.26% (average 29.53%) during the last three weeks of the experiment (Supplementary Figure 3).

2.4. GHG Flux Measurements

249 Greenhouse gas flux measurements were done using non-vented, static (i.e., non-steady-state)
250 chambers (Collier et al. 2014; Rochette & Bertrand 2008) constructed using an ABS cleanout
251 adapter (model # RLN105-030) and male plug (model # RLN106R-030) fitted with a sampling
252 port sealed using a gas-impermeable, grey butyl rubber septum (Supelco, USA) (see Figure 1).
253 Gas flux measurements were made seven days after each GWT adjustment by attaching the
254 sampling chamber to the top of the cores using a flexible coupling (model # FC-33) and
255 sampling the headspace atmosphere immediately after the chamber was attached (t_0) and again
256 after 30 min (t_{30}). The cores remained open to the atmosphere during the period between GWT
257 adjustments.

258 Headspace gas samples were collected at t_0 and t_{30} using a 20-mL polypropylene syringe
259 (Monoject^{TM}, Luer lock fitting) fitted with a 25-gauge needle; samples were injected
260 immediately into pre-evacuated 12-mL Exetainer® vials (LabCo Inc., High Wycombe, UK).
261 Ambient air samples—used as a check on the t_0 samples— were also collected on each sampling
262 day. The gas samples were then brought to the Prairie Environmental Agronomy Laboratory in
263 the Department of Soil Science at the University of Saskatchewan for analysis. The
264 concentrations of CO_2, CH_4, and N_2O in each gas sample were determined using gas
265 chromatography (Farrell & Elliott 2008). Sample analyses were performed using a Bruker 450 GC
266 (Bruker Biosciences Corporation, USA) equipped with a thermal conductivity detector (TCD),
267 flame ionization detector (FID), and electron capture detector (ECD) for the detection and
268 quantification of CO_2, CH_4, and N_2O, respectively. Samples were introduced into the GC using a
269 CombiPAL auto-sampler (CTC Analytics AG, Switzerland); data processing was completed
270 using Varian Star Chromatography Workstation (ver. 6.2) software. The GHG fluxes were
271 calculated from the change in concentration measured during the 30-min chamber deployment

using Equation 1:

$$F = \Delta C \frac{V.k_t}{A} \quad (1)$$

where F is the GHG flux at time zero (mg m^{-2} d^{-1}); ΔC is the change in concentration (mg CO_2, CH_4, or N_2O L^{-1} min^{-1}) measured during the 30-min deployment period; V is the volume of the chamber headspace (0.6089 L); A is the surface area of the soil cores (0.0064 m^2); and k_t is the time constant (1440 min d^{-1}). For the correction of potential gas losses through leaks and sampling removal, net GHG fluxes were calculated by subtracting the respective blank (sample from ambient air collected at the time of GHG sampling) values from the values for the soil cores.

Cumulative GHG emissions for each land-use were calculated using linear interpolation (Equation 2) as described in Pennock et al. (2010). This assumes that emissions were constant both throughout the day of the measurement and during the seven days since the previous water table adjustment.

$$CF = (F_{w1} \times 1) + (F_{w2} \times 7) + (F_{w3} \times 7) + (F_{w4} \times 7) + (F_{w5} \times 7) + (F_{w6} \times 7) + (F_{w7} \times 7) + (F_{w8} \times 7) + (F_{w9} \times 7) \quad (2)$$

Where CF is the cumulative GHG flux (mg m^{-2}); F_W is the daily flux rate measured at the end of each week (weeks 1 through 9; total 57 days of incubation); 7 is the number of days in a week. For the first week (F_{w1}), the calculation only includes GHG emissions for the day of sampling. Global warming potential (GWP) for each land-use practice were calculated for a 100-year time scale using conversion factors for CO_2 = 1, CH_4 = 25, N_2O = 298 after adjusting by mass to obtain carbon dioxide equivalents (i.e., CO_{2e}) flux (Myhre et al. 2013; Wang et al. 2017b).

2.5. Statistical Analyses

Soil GHG emission data were statistically analyzed and visualized using R version 3.4.4 for Windows (R Core Team 2018). The Shapiro-Wilk test and histogram were used to assess the normality and Levene's test was used to check the homogeneity of variances using "car" package. The relationships among CO_2, CH_4, and N_2O emissions, VSWC, EC, and initial soil parameters were measured by Spearman rank-order correlation and visualized using "corrplot" package. Assumptions of both univariate and multivariate analysis of variance (ANOVA) normality were fulfilled by adding a positive constant number (+2) during the transformation (Logarithmic with base 10) to manage negative CH_4 and N_2O values in the dataset. Significant differences among land-use practices, groundwater salinity treatments, and water table depths were compared parametrically by univariate ANOVA with nested design and linear mixed-effects models (Zuur et al. 2009) using " lmerTest". Pairwise multiple comparison procedures (Tukey's HSD method) were used as a post-hoc test. The permutation multivariate ANOVA (PERMANOVA) and analysis of similarities (ANOSIM) were used to assess significant differences (multivariate hypothesis testing) in GHG emission among land-use practices, groundwater salinity, and GWT depths. ANOSIM was also used to calculate a matrix of dissimilarity ranks after converting the scores to find the ratio between within-group and between-group similarities. Unconstrained ordination with a non-metric multidimensional scale (NMDS) was used to plot the position in multidimensional space with a reduced number of dimensions to visualize the difference among groundwater salinity treatments, GWT depths, and land-use practices. The variation partitioning analysis (VPA) was used to determine the proportional contribution of land-use practices, groundwater salinity, and water table depth in the variation of GHG emissions. Constrained ordination with redundancy analysis (RDA) was performed to summarize the variation explained by measured soil physiochemical

317 characteristics. The PERMANOVA, ANOSIM, NMDS, VPA, and RDA analyses were
318 performed using the "vegan" package (Oksanen et al. 2017). All differences were considered
319 significant at *p*-values ≤ 0.05 (95% confidence interval or alpha level = 0.05).

320 ## 3. Results

321 ### 3.1. Emissions of GHG in Soils from Contrasting Land-use Practices, Elevated
322 Groundwater Salinity, and Declining Groundwater Table

323 The soils from PA land-use in both sites showed significantly ($p < 0.001$) higher CO_2 emissions;
324 CO_2 emissions followed consistent land-use patterns in the order of PA > AC = SRW (Table 1).
325 Cumulative CO_2 emissions were higher in soils from site A than site B (Figure 2). The CO_2
326 emissions were significantly ($p < 0.05$) higher in the control (S0) compared to the elevated
327 salinity treatments (i.e., S1 and S2) across all land-use practices from both sites (Table 1 and 2).
328 A significant ($p < 0.001$) difference in CO_2 emissions was observed among the depth to GWT
329 (Table 2). The CO_2 flux initially showed an increasing trend with declining GWT depths (i.e.,
330 weeks of measurements) in soils across all land-use practices from both sites and showed a
331 decreasing trend after 4 weeks (Table 1 and Supplementary Figure 4). The mean emissions of
332 CO_2 were highest ($p > 0.05$) at GWT = 11-cm (week 4) in both sites, and lowest at GWT = 26-
333 cm (week 9) in site A, and at GWT = 20-cm (week 7) in site B (Table 1 and 2).

334 The CH_4 emissions significantly ($p < 0.001$) differed among the soil from all land-use practices
335 in both sites (Table 2). The mean CH_4 emissions were significantly higher ($p < 0.001$) in soils
336 from PA and showed a consistent pattern among land-use practices (PA > AC = SRW) from both
337 sites (Table 1). Cumulative CH_4 emissions were relatively low, and variable between sites;
338 emissions were negligible in site B compared to site A (Figure 2). Groundwater salinity
339 treatments (both S1 and S2) reduced CH_4 emission compared to the control (i.e., S0) in site A (*p*

340 = 0.012); however, the effect was not significant ($p = 0.069$) in site B (Table 2). The CH_4 flux in
341 soils from all land-use practices from both sites showed a slightly increasing trend up to week
342 four and then decreased with a further decline in GWT depths (Table 1 and Supplementary
343 Figure 4). Significantly ($p < 0.001$) higher mean CH_4 emissions were observed at GWT = 11-cm
344 (week 4) in site A and GWT = 8-cm (week 3) at site B; CH4 emissions from both sites were
345 lowest at GWT = 26-cm (week 9; Table 1 and 2).

346 The N_2O emissions were significantly ($p < 0.001$) higher in soils from PA and followed a similar
347 land-use pattern to the other GHGs (PA > AC = SRW) for both sites (Table 1 and 2). The
348 cumulative N_2O emissions were higher ($p < 0.01$) under both groundwater salinity treatments
349 (i.e., S1 and S2) compared to the control in soils from both sites (Table 1 and 2). Overall, the
350 cumulative N_2O emission was relatively low and variable between sites; however, higher in soils
351 from site A than site B (Figure 2). The lowest mean N_2O emission was observed at GWT = 2-cm
352 (week 1) in both sites, whereas the highest emission was at GWT = 17-cm (week 6) in site A and
353 GWT = 23-cm (week 8) in site B (Table 1 and 2).

354 Multivariate unconstrained ordination (NMDS analysis) of soil GHG emission data (stress value
355 for site A is 0.0670, and site B is 0.0724) differed considerably among land-use practices in both
356 sites, indicating the land-use practice type was a key factor driving the variability (Figure 3A, B,
357 D, and E). The NMDS ordination also showed a distinct clustering of GHG emissions based on
358 land-use practices in both sites (stress values below 0.10 provide a fair representation of data in
359 reduced dimension), indicating a robust land-use effect of PA soil.

360 The multivariate permutation analysis of variance (PERMANOVA) test confirmed the
361 significant difference in GHG emissions among land-use practices ($p = 0.001$), salinity ($p =$
362 0.001) and depth to GWT ($p = 0.001$) in both sites (Table 3). The VPA test exhibited that the

363 land-use practice alone has the highest contribution to the variation of soil GHG emissions in
364 both sites (site A = 79.3% and site B = 69.6%), followed by depth to GWT (i.e., measurement
365 week) and salinity treatments (Supplementary Figure 5).

366 3.2. Soil Physiochemical Characteristics and their Relationships with GHG

367 3.2.1. Physiochemical Characteristics of Experimental Soil

368 The physiochemical properties of soils used for the microcosm experiment are presented in
369 Table 4. No significant differences were observed in soil physiochemical properties among land-
370 use practices and between sites except SOC, TN and SO_4^{2-} content (ANOVA results are not
371 shown here). The SOC and TN were significantly ($p < 0.05$) higher in soils from PA compared to
372 other land-use practices in the order of PA > SRW = AC in both sites (Table 4). No significant
373 differences ($p > 0.05$) were found in SOC and TN content between sites. The SO_4^{2-} content was
374 approximately eight times higher in soils from site B than site A (Supplementary Table 1). By
375 land-use practice, the soil SO_4^{2-} contents were SRW > PA = AC in site A and SRW = AC > PA
376 in site B, suggesting no consistent land-use patterns between sites.

377 3.2.2. Relationships of GHG with Soil Physiochemical Characteristics

378 Overall, the relationships between soil GHG (CO_2, CH_4, and N_2O) and soil clay content, SOC,
379 TN, and C/N ratio were positive, whereas bulk density, initial EC, WEOC, and SO_4^{2-}-S were
380 negative (Figure 4). Significant positive relationships ($p < 0.05$) between soil GHG and clay
381 content, SOC, and C/N ratio were observed; however, SOC vs CH_4, and C/N vs N_2O were non-
382 significant ($p > 0.05$) The relationships between all GHG emissions and bulk density were
383 negative ($p < 0.05$) (Figure 4). Correlations between soil PO_4^{3-}-P, SO_4^{2-}-S, and WEOC content
384 with CO_2 and CH_4 were negative ($p < 0.05$) except for N_2O ($p > 0.05$). None of the correlations

between other initial soil physiochemical properties and soil GHG emissions were statistically significant ($p > 0.05$) (Figure 4).

3.2.3. Redundancy Analysis (RDA) between Soil Physiochemical Characteristics and GHG

Redundancy analysis (RDA) was performed to determine the relationships among soil physiochemical properties and GHG emissions, as shown in Figure 3C and F. The first two component axes explained 86.23% and 12.43% of site A (Figure 3C), 85.87% and 12.83% (Figure 3F) of site B of soil GHG. The vector lines of SOC, TN, VSWC, EC from site A and site B were statistically significant ($p < 0.05$), showing that SOC and TN played a crucial role in explaining soil GHG emissions in both sites. There was a significant positive correlation ($p < 0.05$) between SOC, TN, and soil GHG emissions in both sites A and B (Figure 3C and F).

3.2.4. Relationships of GHG with VSWC and EC measured during the microcosm experiment

Groundwater salinity manipulation resulted in a statistically significant difference ($p < 0.05$) in soil EC among different salinity treatment levels (in S1 and S2 compared to control) in both sites (Table 1 and 2). Similarly, water table manipulation resulted in a significant difference ($p < 0.05$) in observed VSWC among groundwater table depths in both sites. We did not find any significant difference ($p > 0.05$) in VSWC or EC among land-use practices from site A ($p > 0.05$). However, we found a significant difference ($p < 0.05$) in soil EC and VSWC in both sites because of groundwater salinity and water table manipulation (Table 1 and 2). We also observed a significant ($p < 0.05$) positive relationship between soil EC and VSWC in both sites (Figure 5 and Supplementary Figure 7) during the incubation experiment.

3.3. Global Warming Potential

The effects of different land-use practices from two sites on the GWP of CO_2, CH_4, and N_2O were calculated (Table 4) based on CO_2e during the incubation period. The GWP was significantly affected ($p < 0.05$) by the origin of the soil from three different land-use practices and sites. The GWP was significantly higher in soils from PA, followed by AC and SRW land-use practices in both sites, whereas site A showed significantly higher GWP than site B.

4. Discussion

4.1. Effects of Land-use, Salinity and Groundwater Table on GHG Emissions

4.1.1. Land-use Effects

In our study, CO_2 emissions were significantly affected by contrasting land-use practices, suggesting that land-use was a significant driver of CO_2 emission by influencing the heterotrophic respiration of SOC (Oertel et al. 2016). The highest mean and cumulative CO_2 emissions in our experiment was seen from PA soils, followed by AC and SRW, respectively. Enrichment of SOC can trigger microbial activities that result in the emission of CO_2, CH_4, N_2O; microbial activities are themselves subject to various proximal and distal drivers in soil (Oertel et al. 2016). Land-use practices control SOC accumulation due to the influence of tillage (or lack thereof) and the morphological and biochemical traits of the vegetation; therefore, any changes in land-use practices can change the potential for GHG emissions (Liebig et al. 2005).

We observed a significant positive relationship between CO_2 and both SOC and TN in both sites. Hence, elevated CO_2 emissions were perhaps triggered by higher SOC content and turnover rates from root biomass in PA soils from both sites. Like this study, others have found the C/N ratio was positively correlated with CO_2 and CH_4 emissions (Shi et al. 2014; Weslien et al. 2009).

Likewise, Lang et al. (2010) found that the SOC and C/N ratio dominate CO_2 and N_2O emissions from soil. Restoring cropland to pasture has been shown to increase SOC in the PPR; however, quantifying SOC associated with land-use can be difficult in the short term, given the high degree of variability of both biotic and abiotic factors controlling SOC sequestration over time (Tangen et al. 2015). For example, Follett et al. (2012) found that PA soil can be a significant source or sink of C and N. Similar to our experiment, Parmar et al. (2015) found reduced GHG emissions in soil cores collected from short rotation forestry. In contrast, Lang et al. (2010) found significantly higher CO_2 emissions in the forest than in PA soils, with the reverse relationship for N_2O emissions.

Several factors may have contributed to the variable CH_4 emissions in our experimental soils. Soil CH_4 emissions are generally related to moist environments where methanogenesis can occur (Bridgham et al. 2013; Levy et al. 2012), although the C/N ratio can also influence the CH_4 emission (Gundersen et al. 2012). Using stable C isotope, Wu et al. (2018) observed high CH_4 uptake following afforestation, which they attributed to increased SOC and microbial biomass carbon, lower C/N ratio and less inorganic N. In a meta-analysis of 5000 chamber measurements collected from a range of land-use types, Levy et al. (2012) observed low emissions or a lower rate of net uptake of CH_4 in mineral soils and high emissions from organic soils; SOC, VSWC, and pH were the best sub-set of explanatory variables. Hence, higher SOC in our soils from PA land-use perhaps caused higher CH_4 emissions compared to AC and SRW. Similarly, Lang et al. (2010) found that PA soils were a weak source of CH_4 emission, whereas forest soils were a weak sink of CH_4.. However, we also found higher background SO_4^{2-} content in site B than site A and higher SO_4^{2-} content under SRW compared to PA land-use practices; SO_4^{2-} content was negatively correlated with CO_2 and CH_2 fluxes in soil. Conceivably the high SO_4^{2-} content

451 inhibited the CH_4 emission even under wet conditions (Ardón et al. 2018), resulting in the lower
452 CH_4 emissions from SRW.

453 In a controlled laboratory experiment to assess the effects of land-use and climate (particularly
454 soil temperature and moisture) on the potential GHG emission from intact soil cores collected
455 from 13 European sites, Schaufler et al. (2010) found higher N_2O emission from grasslands
456 compared to croplands, forests, and wetlands. Similarly, we found significantly higher N_2O
457 emissions from PA soil, followed by AC and SRW in both sites. High available C and N content
458 in our PA soil likely stimulated microbial activity leading to high N_2O emission, as observed by
459 Follett et al. (2012). Research has shown that heterotrophic nitrifying bacteria can denitrify with
460 low NO_3-N under aerobic conditions given sufficiently high SOC content (Wrage-Mönnig et al.
461 2018). The quality and availability of SOM input from different land-use practices are likely a
462 key driver (Chantigny 2003) because most of the dissolved organic matter is directly involved in
463 many soil microbial processes (Bolan et al. 2011). In a pot experiment under field conditions, Qiu
464 et al. (2015) observed that management practices that were adding plant-derived dissolved
465 organic matter to the soil increased microbial biomass and were responsible for a significant
466 increase in CO_2 and N_2O emissions. Hence, Wu et al. (2019) suggested that GHG fluxes from the
467 soils are rigorously controlled by the labile components of SOM, such as dissolved organic C
468 and N, as well as inorganic N.

469 In a field-scale study within a Canadian prairie agroecosystem, Baah-Acheamfour et al. (2016)
470 observed that agroforestry could reduce CH_4 and N_2O emissions to a greater extent than
471 grassland, providing potential to mitigate climate change. In a field-scale study in a humid
472 temperate region of southern Europe, Merino et al. (2004) also observed that afforestation could
473 significantly increase SOC content relative to annual cropland, while also decreasing N_2O

474 emission and increasing CH_4 uptake. Likewise, we observed significantly lower GWP (CO_2-e) in
475 soils from SRW than AC and PA in our experiment. Hence, SRW can be a promising land-use
476 practice in the fallow marginal riparian zones of the PPR agroecosystem (Amichev et al. 2014).
477 However, it is often challenging to generalize the impact of agroforestry on the GHG budget
478 without a better understanding of the plant types, soil, and climatic drivers that control the GHG
479 emissions (Benanti et al. 2014).

480 **4.1.2. Salinity Effects**

481 Salinity treatments significantly decreased CO_2 and CH_4 emissions except for the CH_4 emission
482 from site B ($p = 0.069$); however, it significantly increased the N_2O emission in soils from both
483 sites. A microcosm experiment performed on the coastal forested wetlands (Ardón et al. 2018)
484 found that salinity can suppress CO_2 emission under both flooded and drought conditions.
485 Similar to our experiment, an incubation study by Maucieri et al. (2017) examined short-term
486 effects of irrigation water salinity on soil GHG emissions from semi-arid Australian soil; CO_2
487 emissions were reduced by 19% at 5-mS cm^{-1} and 28% at 10-mS cm^{-1}, whereas N_2O emissions
488 increased 60%, and CH_4 emissions were not affected by increased salinity, only by soil water.
489 Setia et al. (2011) also found a significant decrease in CO_2 emission with increasing salinity
490 ranged from 1 to 5 mS cm^{-1}.

491 The salt concentration and water content regulate the osmotic potential in the soil; at both high
492 salinity and low water content, soil microorganisms can tolerate the high osmotic potentials by
493 synthesizing osmolytes, which lets them continue metabolism (Yan & Marschner 2013).
494 Consequently, perhaps both salinity and hydrology controlled N_2O emissions during our
495 experiment. Fluctuating aerobic-anaerobic conditions and environments low in oxygen can
496 promote N_2O production by nitrifier denitrification (Wrage-Mönnig et al. 2018). Dang et al. (2017)

likewise observed enhanced N_2O production in an incubation experiment under higher soil salinity and suggested that the addition of available carbon (glucose) and nitrogen (nitrate) created favorable conditions for denitrification. Comparably, in our experimental soil, SOC and TN acts as significant drivers that controlled the GHG emission. Increased N_2O emission through denitrification with increased salinity is likely (Marton et al. 2012). Tsuneda et al. (2005) observed that increased salt concentrations could substantially influence N_2O emission by inhibiting nitrous oxide reductase activity. Enhanced N_2O emissions may be triggered by inhibited nitrous oxide reductase impeding the kinetic balance between N_2O production and consumption under salt stress conditions (Han et al. 2019)

Increased soil NH_4^+ and dissolved organic carbon were observed with increased salinity in a laboratory incubation experiment with core soils collected from freshwater tidal marshes in southeast China (Wang et al. 2017a). In contrast to our experiment, (Wang et al. 2017a) observed stimulated CO_2 emission at intermediate salinities (i.e., 5 to 7.5‰) but inhibited at ≥15‰, CH_4 emissions were unaffected up to 7.5‰ but declined substantially at salinity ≥10‰, whereas salinity did not affect the N_2O emission. Similarly, in a review, Poffenbarger et al. (2011) observed that the CH_4 emission decreased with increasing salinity in tidal marshes. We got a very low CH_4 emission or some cases uptake in site B and site A except under PA land-use practice. In these soils, higher salinity increases SO_4^{2-} availability; SO_4^{2-} acts as an alternative terminal electron acceptor under anaerobic conditions and can shift microbial metabolism towards more energetically favorable processes (Bridgham et al. 2013). A significant inverse correlation has been observed between the CH_4 emission and SO_4^{2-} content in PPR wetland soils (Pennock et al. 2010). Our experimental soil from site B had very high SO_4^{2-} content and had a highly negative correlation with CH_4; the lowest CH_4 emissions were observed from those soils with high SO_4^{2-}

520 content in the soil. Similarly, the presence of high SO_4^{2-} in soil inhibited the CH_4 emission in an
521 incubation experiment (Ardón et al. 2018) and riparian areas of PPR wetlands (Dunmola et al.
522 2010).

523 **4.1.3. Water Table Effects**

524 The GWT significantly controlled the soil GHG emissions from our experimental soil cores. Soil
525 water content controls microbial activity and processes and is the single most crucial soil
526 parameter that regulates GHG emissions (Oertel et al. 2016). Overall, higher CO_2 emission can
527 occur from the rapid decomposition of C in well-drained areas (Freeman et al. 2001), N_2O
528 emissions are most likely between strict aerobic and anaerobic conditions (Davidson et al. 2000),
529 and CH_4 is produced via the reduction of CO_2 in a strictly anaerobic microbial process known as
530 methanogenesis (Bridgham et al. 2013). Conversely, soils can also be a sink of atmospheric CH_4
531 through microbial oxidation under aerobic conditions (Thangarajan et al. 2013).

532 In our experiment, the CO_2 emission rate was variable and, to some extent, dependent on VSWC
533 in both sites, whereas CH_4 emissions decreased with declining water table depths and as the
534 VSWC decreased. Similarly, an incubation experiment with peat cores from central and eastern
535 Canada (Blodau et al. 2004) found a lower water table depth increased CO_2 production through
536 soil respiration and microbial biomass, whereas CH_4 production and emissions decreased.
537 Additionally, in a lysimeter experiment using undisturbed peat soil columns, higher CO_2
538 emissions were observed at the low water table depth (40 cm below surface) compared to the
539 greater depths (80 cm below surface), CH_4 emissions were very low or negative (Berglund &
540 Berglund 2011). Studies of flooding effects on GHG observed significant CH_4 emission in both
541 forested and non-forested soils (Mander et al. 2015; Wang & Bettany 1997); however, Wang and

Bettany (1997) also found that 80-90% of the CH_4 was taken up within a week under non-flooded conditions.

We observed low N_2O emissions under the higher water table level as the experimental soil cores were near-saturated; however, emissions increased once the moisture condition became ideal. This occurred at the midpoint of groundwater table treatments, similar to what was observed by (Berglund & Berglund 2011) in their peat core experiment. Similarly, a laboratory incubation study found high N_2O emission with adequate but unsaturated soil water availability, indicating that water-filled pore spaces and C availability primarily controlled the denitrification process and thus N_2O emissions (Gillam et al. 2008). Substantially higher N_2O emissions can occur when intermittently flooded soils are exposed to air, enhancing combined nitrification-denitrification; however, drying also initiates suboptimal conditions for complete denitrification as enhanced oxygen supply inhibits N_2O reductase (Knowles 1982). Therefore, as the GWT is lowered (in the absence of standing water), the N_2O reductase enzyme that catalyzes the reduction of N_2O to N_2 is inhibited by oxygen under suboxic conditions; N_2O emissions are most likely the by-product of denitrification, and N_2O diffusion to the atmosphere is unrestricted by porewater, increasing N_2O fluxes (Pinto et al. 2021).

A study within the PPR agricultural landscape observed that the hotspots of GHG are predominantly driven by soil moisture and SOC availability (Dunmola et al. 2010). We collected intact soil cores from the annual crop, pasture, and short rotation willow plantation; however, we did not measure GHG emissions directly in the field. Although we might not specifically compare our results with that of field-scale studies, we did observe that the VSWC in our experimental cores largely controlled the GHG emissions. When soils dry out, the substrate supply becomes increasingly limited for microbes as the water drains out from soil pores, and

565 water films around the soil aggregates become thinner and disconnected (Yan et al. 2015).
566 However, we should consider that lowering the water table may also expose new layers in soil
567 containing substrate for microbial decomposition and that soil physical properties at depth might
568 have distinct impacts on the emissions rate (Berglund & Berglund 2011).

569 **5. Conclusions**

570 Our results showed that adjacent contrasting riparian land-use practices significantly influenced
571 GHG emissions within the PPR agroecosystems. We observed significantly higher CO_2, CH_4,
572 and N_2O emissions from PA land-use practice. Changes in soil properties, particularly organic C
573 and N, evidently shaped the observed difference in soil GHG emissions because of contrasting
574 land-use practices. Conceivably, high background SO_4^{2-} concentration in soils collected from
575 SRW land-use practices cut the CH_4 emission and subsequently contributed less towards the
576 GWP.

577 We saw that lowering the water table decreased CH_4 emissions with the reduction of VSWC but
578 resulted in higher N_2O emissions under an intermediate water table position when VSWC in the
579 cores reached suitable conditions for denitrification. We also noticed variable CO_2 emissions,
580 with an initial increase with the lowering of the water table followed by a decrease as the VSWC
581 diminished. With elevated groundwater salinity, we observed a decrease in CO_2 and CH_4
582 emissions, but a significant escalation in N_2O emissions.

583 The GWP of SRW was significantly lower than AC and PA, suggesting this is a potentially
584 promising land-use practice in those fallow marginal riparian zones of the PPR that are not
585 suitable for crop production due to the higher salinity. Overall, our experiment showed a

586 decrease in GHG emissions with increasing salinity and varying responses to GWT based on
587 GWT depth and the GHG in question.

588 **References**

589 Albrecht A, Kandji ST (2003) Carbon sequestration in tropical agroforestry systems. Agriculture,
590 Ecosystems & Environment 99(1): 15-27

591 Amichev BY, Hangs RD, Konecsni SM, Stadnyk CN, Volk TA, Bélanger N, Vujanovic V,
592 Schoenau JJ, Moukoumi J, Van Rees KCJ (2014) Willow Short-Rotation Production Systems in
593 Canada and Northern United States: A Review. Soil Science Society of America Journal 78(S1):
594 S168-S182

595 Ardón M, Helton AM, Bernhardt ES (2018) Salinity effects on greenhouse gas emissions from
596 wetland soils are contingent upon hydrologic setting: a microcosm experiment. Biogeochemistry
597 140(2): 217-232

598 Arndt JL, Richardson JL (1989) Geochemistry of Hydric Soil Salinity in a Recharge-
599 Throughflow-Discharge Prairie-Pothole Wetland System. Soil Science Society of America
600 Journal 53(3): 848-855

601 Baah-Acheamfour M, Carlyle CN, Lim SS, Bork EW, Chang SX (2016) Forest and grassland
602 cover types reduce net greenhouse gas emissions from agricultural soils. Science of the Total
603 Environment 571: 1115-1127

604 Bedard-Haughn A, Jongbloed F, Akkerman J, Uijl A, de Jong E, Yates T, Pennock D (2006) The
605 effects of erosional and management history on soil organic carbon stores in ephemeral wetlands
606 of hummocky agricultural landscapes. Geoderma 135: 296-306

607 Beetz S, Liebersbach H, Glatzel S, Jurasinski G, Buczko U, Höper H (2013) Effects of land use
608 intensity on the full greenhouse gas balance in an Atlantic peat bog. Biogeosciences 10(2): 1067-
609 1082

610 Benanti G, Saunders M, Tobin B, Osborne B (2014) Contrasting impacts of afforestation on
611 nitrous oxide and methane emissions. Agricultural and Forest Meteorology 198-199: 82-93

612 Berglund Ö, Berglund K (2011) Influence of water table level and soil properties on emissions of
613 greenhouse gases from cultivated peat soil. Soil Biology and Biochemistry 43(5): 923-931

614 Blodau C, Basiliko N, Moore TR (2004) Carbon turnover in peatland mesocosms exposed to
615 different water table levels. Biogeochemistry 67(3): 331-351

616 Bolan NS, Adriano DC, Kunhikrishnan A, James T, McDowell R, Senesi N (2011) Dissolved
617 Organic Matter. 110: 1-75

618 Bridgham SD, Cadillo-Quiroz H, Keller JK, Zhuang Q (2013) Methane emissions from
619 wetlands: biogeochemical, microbial, and modeling perspectives from local to global scales.
620 Global Change Biology 19(5): 1325-1346

621 Carter MR, Gregorich EG (Eds) (2008) Soil Sampling and Methods of Analysis. CRC Press and
622 Taylor & Francis Group, Boca Raton, FL, USA

623 Chantigny MH (2003) Dissolved and water-extractable organic matter in soils: a review on the
624 influence of land use and management practices. Geoderma 113(3-4): 357-380

625 Chantigny MH, Angers DA, Kaiser K, Kalbitz K (2008) Extraction and Characterization of
626 Dissolved Organic Matter. In: Carter MR & Gregorich EG (eds) Soil Sampling and Methods of
627 Analysis, Second Edition. CRC Press.

628 Collier SM, Ruark MD, Oates LG, Jokela WE, Dell CJ (2014) Measurement of Greenhouse Gas
629 Flux from Agricultural Soils Using Static Chambers. Journal of Visualized Experiments : JoVE
630 (90): 52110

631 Dang DM, Macdonald B, Warneke Sr, White I (2017) Available carbon and nitrate increase
632 greenhouse gas emissions from soils affected by salinity. Soil Research 55(1): 47

633 Davidson EA, Keller M, Erickson HE, Verchot LV, Veldkamp E (2000) Testing a Conceptual
634 Model of Soil Emissions of Nitrous and Nitric Oxides. BioScience 50(8): 667

635 Dunmola AS, Tenuta M, Moulin AP, Yapa P, Lobb DA (2010) Pattern of greenhouse gas
636 emission from a Prairie Pothole agricultural landscape in Manitoba, Canada. Canadian Journal of
637 Soil Science 90(2): 243-256

638 Farrell RE, Elliott JA (2008) Soil Air. In: Carter MR & Gregorich EG (eds) Soil Sampling and
639 Methods of Analysis, Second Edition. CRC Press.

640 Follett RF, Stewart CE, Pruessner EG, Kimble JM (2012) Effects of climate change on soil
641 carbon and nitrogen storage in the US Great Plains. Journal of Soil and Water Conservation
642 67(5): 331-342

643 Freeman C, Ostle N, Kang H (2001) An enzymic 'latch' on a global carbon store. Nature 409:
644 149

645 Gillam KM, Zebarth BJ, Burton DL (2008) Nitrous oxide emissions from denitrification and the
646 partitioning of gaseous losses as affected by nitrate and carbon addition and soil aeration.
647 Canadian Journal of Soil Science 88(2): 133-143

648 Gleason RA, Laubhan MK, Euliss Jr NH, Tangen BA, Kermes KE (2008) Ecosystem services
649 derived from wetland conservation practices in the United States Prairie Pothole Region with an
650 emphasis on the US Department of Agriculture Conservation Reserve and Wetlands Reserve
651 Programs. In. US Geological Survey, Reston, Virginia.

652 Gleason RA, Tangen BA, Browne BA, Euliss Jr NH (2009) Greenhouse gas flux from cropland
653 and restored wetlands in the Prairie Pothole Region. Soil Biology and Biochemistry 41(12):
654 2501-2507

655 Gundersen P, Christiansen JR, Alberti G, Brüggemann N, Castaldi S, Gasche R, Kitzler B,
656 Klemedtsson L, Lobo-do-Vale R, Moldan F, Rütting T, Schleppi P, Weslien P, Zechmeister-
657 Boltenstern S (2012) The response of methane and nitrous oxide fluxes to forest change in
658 Europe. Biogeosciences 9(10): 3999-4012

659 Han H, Song B, Song MJ, Yoon S (2019) Enhanced Nitrous Oxide Production in Denitrifying
660 Dechloromonas aromatica Strain RCB Under Salt or Alkaline Stress Conditions. Frontiers in
661 microbiology 10: 1203

662 Hendershot WH, Lalande H, Duquette M (2008a) Ion Exchange and Exchangeable Cations. In:
663 Carter MR & Gregorich EG (eds) Soil Sampling and Methods of Analysis, Second Edition. CRC
664 Press.

665 Hendershot WH, Lalande H, Duquette M (2008b) Soil Reaction and Exchangeable Acidity. In:
666 Carter MR & Gregorich EG (eds) Soil Sampling and Methods of Analysis, Second Edition. CRC
667 Press.

668 IPCC (2018) Climate Change 2013: The Physical Science Basis In: Stocker TF, D. Qin, G.-K.
669 Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (ed)
670 Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel
671 on Climate Change. IPCC, Cambridge, United Kingdom and New York, NY, USA. p 1-1535

672 Kayranli B, Scholz M, Mustafa A, Hedmark Å (2009) Carbon Storage and Fluxes within
673 Freshwater Wetlands: a Critical Review. Wetlands 30(1): 111-124

674 Knowles R (1982) Denitrification. Microbiology and Molecular Biology Reviews-1982-
675 Knowles-43.full.pdf> 46(1): 43-70

676 Kooch Y, Moghimian N, Bayranvand M, Alberti G (2016) Changes of soil carbon dioxide,
677 methane, and nitrous oxide fluxes in relation to land use/cover management. Environmental
678 Monitoring and Assessment 188: 346

679 Kroetsch D, Wang C (2008) Particle Size Distribution. In: Carter MR & Gregorich EG (eds) Soil
680 Sampling and Methods of Analysis, Second Edition. CRC Press.

681 Krzywinski M, Altman N, Blainey P (2014) Nested designs. Nature Methods 11(10): 977-978

682 Lang M, Cai Z, Chang SX (2010) Effects of land use type and incubation temperature on
683 greenhouse gas emissions from Chinese and Canadian soils. Journal of Soils and Sediments
684 11(1): 15-24

685 Last WM, Ginn FM (2005) Saline systems of the Great Plains of western Canada: an overview
686 of the limnogeology and paleolimnology. Saline systems 1: 10

687 Levy PE, Burden A, Cooper MDA, Dinsmore KJ, Drewer J, Evans C, Fowler D, Gaiawyn J,
688 Gray A, Jones SK, Jones T, McNamara NP, Mills R, Ostle N, Sheppard LJ, Skiba U, Sowerby A,
689 Ward SE, Zieliński P (2012) Methane emissions from soils: synthesis and analysis of a large UK
690 data set. Global Change Biology 18(5): 1657-1669

691 Liebig M, Morgan J, Reeder J, Ellert B, Gollany H, Schuman G (2005) Greenhouse gas
692 contributions and mitigation potential of agricultural practices in northwestern USA and western
693 Canada. Soil & Tillage Research 83(1): 25-52

694 Mander Ü, Maddison M, Soosaar K, Karabelnik K (2011) The Impact of Pulsing Hydrology and
695 Fluctuating Water Table on Greenhouse Gas Emissions from Constructed Wetlands. Wetlands
696 31(6): 1023-1032

697 Mander Ü, Maddison M, Soosaar K, Teemusk A, Kanal A, Uri V, Truu J (2015) The impact of a
698 pulsing groundwater table on greenhouse gas emissions in riparian grey alder stands.
699 Environmental Science and Pollution Research 22(4): 2360-2371

700 Martin RM, Moseman-Valtierra S (2015) Greenhouse Gas Fluxes Vary Between Phragmites
701 Australis and Native Vegetation Zones in Coastal Wetlands Along a Salinity Gradient. Wetlands
702 35(6): 1021-1031

703 Marton JM, Herbert ER, Craft CB (2012) Effects of Salinity on Denitrification and Greenhouse
704 Gas Production from Laboratory-incubated Tidal Forest Soils. Wetlands 32(2): 347-357

705 Maucieri C, Zhang Y, McDaniel MD, Borin M, Adams MA (2017) Short-term effects of biochar
706 and salinity on soil greenhouse gas emissions from a semi-arid Australian soil after re-wetting.
707 Geoderma 307: 267-276

708 Merino An, Pérez-Batallón P, Macías F (2004) Responses of soil organic matter and greenhouse
709 gas fluxes to soil management and land use changes in a humid temperate region of southern
710 Europe. Soil Biology and Biochemistry 36(6): 917-925

711 Millar JB (1971) Shoreline-area ratio as a factor in rate of water loss from small sloughs. Journal
712 of Hydrology 14(3): 259-284

713 Miller JJ, Curtin D (2008) Electrical Conductivity and Soluble Ions. In: Carter MR & Gregorich
714 EG (eds) Soil Sampling and Methods of Analysis, Second Edition. CRC Press.

715 Moore BD, Kaur G, Motavalli PP, Zurweller BA, Svoma BM (2017) Soil greenhouse gas
716 emissions from agroforestry and other land uses under different moisture regimes in lower
717 Missouri River Floodplain soils: a laboratory approach. Agroforestry Systems 92: 335-348

718 Mutuo PK, Cadisch G, Albrecht A, Palm CA, Verchot L (2005) Potential of agroforestry for
719 carbon sequestration and mitigation of greenhouse gas emissions from soils in the tropics.
720 Nutrient Cycling in Agroecosystems 71(1): 43-54

721 Myhre G, Shindell D, Bréon FM, Collins W, Fuglestvedt J, Huang J, Koch D, Lamarque JF, Lee
722 D, Mendoza B, Nakajima T, Robock A, Stephens G, Takemura T, Zhang H (2013)
723 Anthropogenic and Natural Radiative Forcing. In: T. F. Stocker, D. Qin, G. K. Plattner, M.
724 Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex & Midgley PM (eds) Climate
725 Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth

726 Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University
727 Press Cambridge, United Kingdom and New York, NY, USA.

728 Nachshon U, Ireson A, van der Kamp G, Wheater H (2013) Sulfate salt dynamics in the
729 glaciated plains of North America. Journal of Hydrology 499: 188-199

730 Oertel C, Matschullat J, Zurba K, Zimmermann F, Erasmi S (2016) Greenhouse gas emissions
731 from soils—A review. Chemie der Erde - Geochemistry 76(3): 327-352

732 Oksanen J, Blanchet FG, Friendly M, Kindt R, Legendre P, McGlinn D, Minchin PR, O'Hara
733 RB, Simpson GL, Solymos P, Stevens MHH, Szoecs E, Wagner H (2017) R Package "vegan":
734 Community Ecology Package.

735 Parmar K, Keith AM, Rowe RL, Sohi SP, Moeckel C, Pereira MG, McNamara NP (2015)
736 Bioenergy driven land use change impacts on soil greenhouse gas regulation under Short
737 Rotation Forestry. Biomass and Bioenergy 82: 40-48

738 Pennock D, Yates T, Bedard-Haughn A, Phipps K, Farrell R, McDougal R (2010) Landscape
739 controls on N2O and CH4 emissions from freshwater mineral soil wetlands of the Canadian
740 Prairie Pothole region. Geoderma 155(3-4): 308-319

741 Pinto R, Weigelhofer G, Brito AG, Hein T (2021) Effects of dry-wet cycles on nitrous oxide
742 emissions in freshwater sediments: a synthesis. PeerJ 9: e10767

743 Poffenbarger HJ, Needelman BA, Megonigal JP (2011) Salinity influence on methane emissions
744 from tidal marshes. Wetlands 31(5): 831-842

745 Qiu Q, Wu L, Ouyang Z, Li B, Xu Y, Wu S, Gregorich EG (2015) Effects of plant-derived
746 dissolved organic matter (DOM) on soil CO 2 and N 2 O emissions and soil carbon and nitrogen
747 sequestrations. Applied Soil Ecology 96: 122-130

748 R Core Team (2018) R: A Language and Environment for Statistical Computing. In. R
749 Foundation for Statistical Computing, Vienna, Austria.

750 Reichstein M, Subke J-A, Angeli AC, Tenhunen JD (2005) Does the temperature sensitivity of
751 decomposition of soil organic matter depend upon water content, soil horizon, or incubation
752 time? Global Change Biology 11(10): 1754-1767

753 Rochette P, Bertrand N (2008) Soil-Surface Gas Emissions. In: Carter MR & Gregorich EG (eds)
754 Soil Sampling and Methods of Analysis, Second Edition. CRC Press.

755 Rutherford PM, McGill WB, Arocena JM, Figueiredo CT (2008) Total Nitrogen. In: Carter MR
756 & Gregorich EG (eds) Soil Sampling and Methods of Analysis, Second Edition. CRC Press.

757 Saskatchewan Soil Survey Staff (1986) Soil Survey Reports for Saskatchewan: The Soils of
758 Indian Head, Rural Municipalty no 156 Saskatchewan. In. Saskatchewan Institute of Pedology
759 Publication S202, Saskatoon, Saskatchewan.

760 Schaufler G, Kitzler B, Schindlbacher A, Skiba U, Sutton M, Zechmeister-Boltenstern S (2010)
761 Greenhouse gas emissions from European soils under different land use: effects of soil moisture
762 and temperature. European Journal of Soil Science 61(5): 683-696

763 Schielzeth H, Nakagawa S, Freckleton R (2013) Nested by design: model fitting and
764 interpretation in a mixed model era. Methods in Ecology and Evolution 4(1): 14-24

765 Setia R, Marschner P, Baldock J, Chittleborough D, Smith P, Smith J (2011) Salinity effects on
766 carbon mineralization in soils of varying texture. Soil Biology and Biochemistry 43(9): 1908-
767 1916

768 Sheng Q, Wang L, Wu J (2015) Vegetation alters the effects of salinity on greenhouse gas
769 emissions and carbon sequestration in a newly created wetland. Ecological Engineering 84: 542-
770 550

771 Shi W-Y, Yan M-J, Zhang J-G, Guan J-H, Du S (2014) Soil CO2 emissions from five different
772 types of land use on the semiarid Loess Plateau of China, with emphasis on the contribution of
773 winter soil respiration. Atmospheric Environment 88: 74-82

774 Simard RR (1993) Ammonium Acetate-Extractable Elements. In: Carter MR (ed) Soil Sampling
775 and Methods of Analysis. Lewis Publishers, Boca raton, FL, USA.

776 Skjemstad JO, Baldock JA (2008) Total and Organic Carbon. In: Carter MR & Gregorich EG
777 (eds) Soil Sampling and Methods of Analysis, Second Edition. CRC Press.

778 Smith KA, Ball T, Conen F, Dobbie KE, Massheder J, Rey A (2003) Exchange of greenhouse
779 gases between soil and atmosphere: interactions of soil physical factors and biological processes.
780 European Journal of Soil Science 54: 779–791

781 Smith P, Martino D, Cai Z, Gwary D, Janzen H, Kumar P, McCarl B, Ogle S, O'Mara F, Rice C,
782 Scholes B, Sirotenko O, Howden M, McAllister T, Pan G, Romanenkov V, Schneider U,
783 Towprayoon S, Wattenbach M, Smith J (2008) Greenhouse gas mitigation in agriculture.
784 Philosophical Transactions of the Royal Society B: Biological Sciences 363(1492): 789-813

785 Tangen BA, Finocchiaro RG, Gleason RA (2015) Effects of land use on greenhouse gas fluxes
786 and soil properties of wetland catchments in the Prairie Pothole Region of North America.
787 Science of the Total Environment 533: 391-409

788 Thangarajan R, Bolan NS, Tian G, Naidu R, Kunhikrishnan A (2013) Role of organic
789 amendment application on greenhouse gas emission from soil. Science of the Total Environment
790 465: 72-96

791 Tsuneda S, Mikami M, Kimochi Y, Hirata A (2005) Effect of salinity on nitrous oxide emission
792 in the biological nitrogen removal process for industrial wastewater. J Hazard Mater 119(1-3):
793 93-98

794 Updegraff K, Bridgham SD, Pastor J, Weishampel P, Harth C (2001) Response of CO2 and CH4
795 emissions from peatlands to warming and water table manipulation. Ecological Applications
796 11(2): 311-326

797 Vidon P (2010) Riparian zone management and environmental quality: a multi-contaminant
798 challenge. Hydrological Processes 24(11): 1532-1535

799 Wang C, Tong C, Chambers LG, Liu X (2017a) Identifying the Salinity Thresholds that Impact
800 Greenhouse Gas Production in Subtropical Tidal Freshwater Marsh Soils. Wetlands 37(3): 559-
801 571

802 Wang FL, Bettany JR (1997) Methane emission from Canadian prairie and forest soils under
803 short term flooding conditions. Nutrient Cycling in Agroecosystems 49(1): 197-202

804 Wang H, Yu L, Zhang Z, Liu W, Chen L, Cao G, Yue H, Zhou J, Yang Y, Tang Y, He JS
805 (2017b) Molecular mechanisms of water table lowering and nitrogen deposition in affecting
806 greenhouse gas emissions from a Tibetan alpine wetland. Global Change Biology 23(2): 815-829

807 Weslien P, Kasimir Klemedtsson Å, Börjesson G, Klemedtsson L (2009) Strong pH influence on
808 N2O and CH4fluxes from forested organic soils. European Journal of Soil Science 60(3): 311-
809 320

810 Wrage-Mönnig N, Horn MA, Well R, Müller C, Velthof G, Oenema O (2018) The role of
811 nitrifier denitrification in the production of nitrous oxide revisited. Soil Biology and
812 Biochemistry 123: A3-A16

813 Wu H, Xingkai X, Cheng W, Lin H (2019) Dissolved organic matter and inorganic N jointly
814 regulate greenhouse gases fluxes from forest soils with different moistures during a freeze-thaw
815 period. Soil Science and Plant Nutrition 66(1): 163-176

816 Wu J, Li Q, Chen J, Lei Y, Zhang Q, Yang F, Zhang D, Zhang Q, Cheng X (2018) Afforestation
817 enhanced soil CH4 uptake rate in subtropical China: Evidence from carbon stable isotope
818 experiments. Soil Biology and Biochemistry 118: 199-206

819 Yan N, Marschner P (2013) Response of soil respiration and microbial biomass to changing EC
820 in saline soils. Soil Biology and Biochemistry 65: 322-328

821 Yan N, Marschner P, Cao W, Zuo C, Qin W (2015) Influence of salinity and water content on
822 soil microorganisms. International Soil and Water Conservation Research 3(4): 316-323

823 Zuur AF, Ieno EN, Walker NJ, Saveliev AA, Smith GM (2009) Mixed Effect Models and
824 Extensions in Ecology with R. Springer, New York, USA

825

Figures

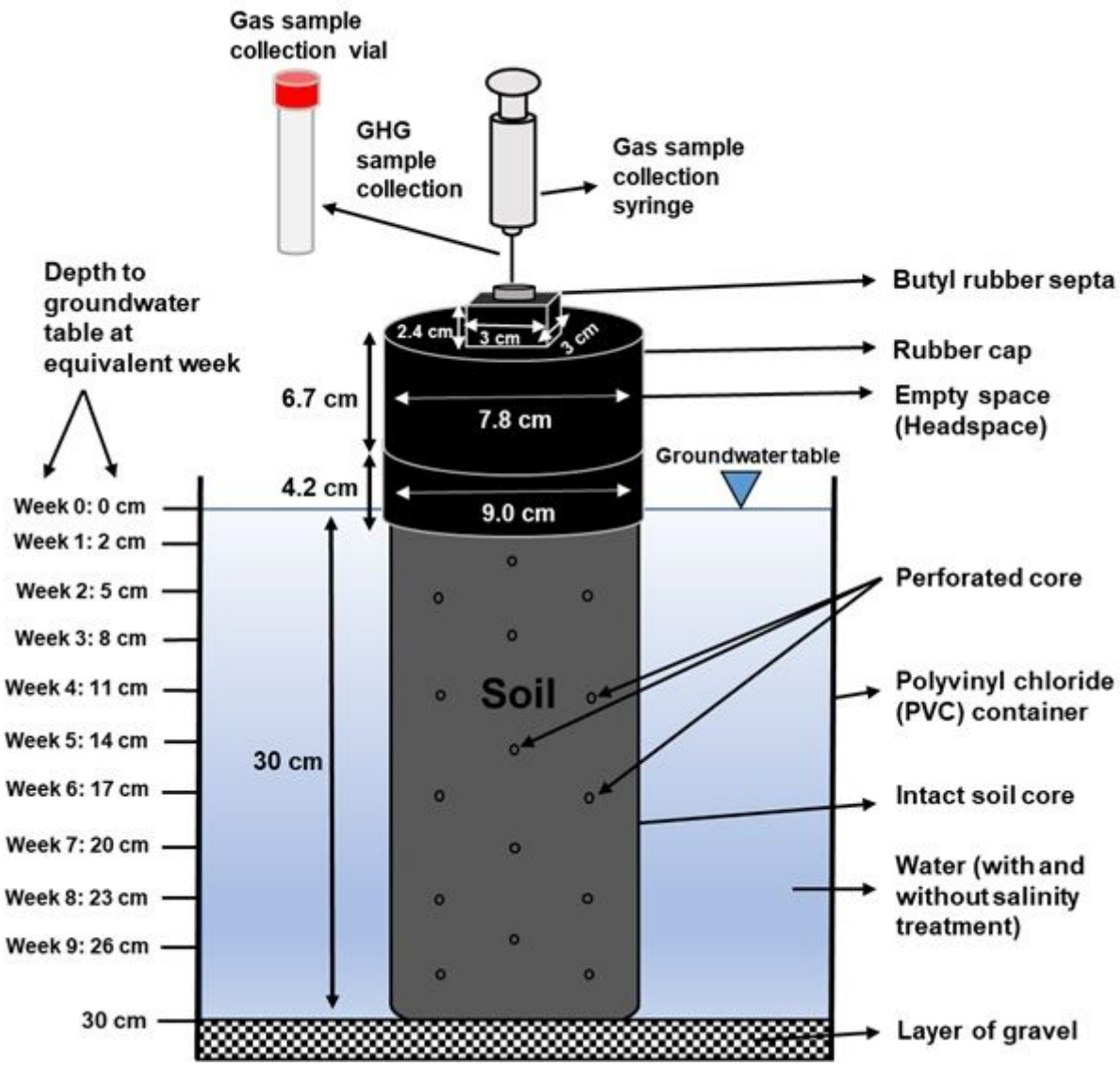

Figure 1

An individual experimental unit with intact soil core and greenhouse gas chamber used for microcosm experiment (Note: diagram is not to scale).

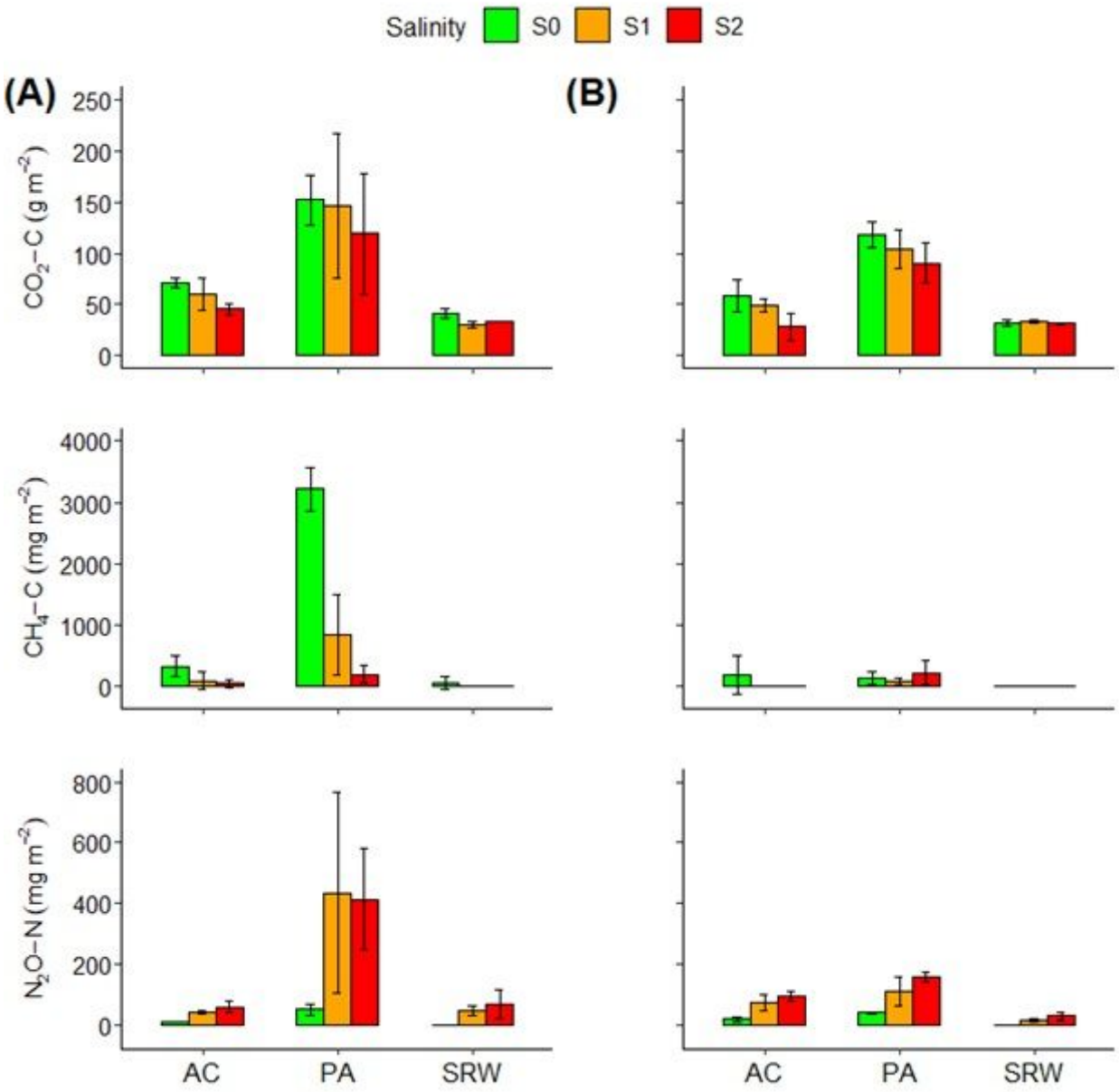

Figure 2

Cumulative GHG emissions from core soils with different groundwater salinity treatments from soils collected from three land-use practices from A) site A, and B) site B. † Error bar stands for standard deviations (± SD). †† GHG = greenhouse gas, S0 = control, S1 = 6 mS cm-1, S2 = 12 mS cm-1, AC = annual crop, PA = pasture, SRW = short rotation willow.

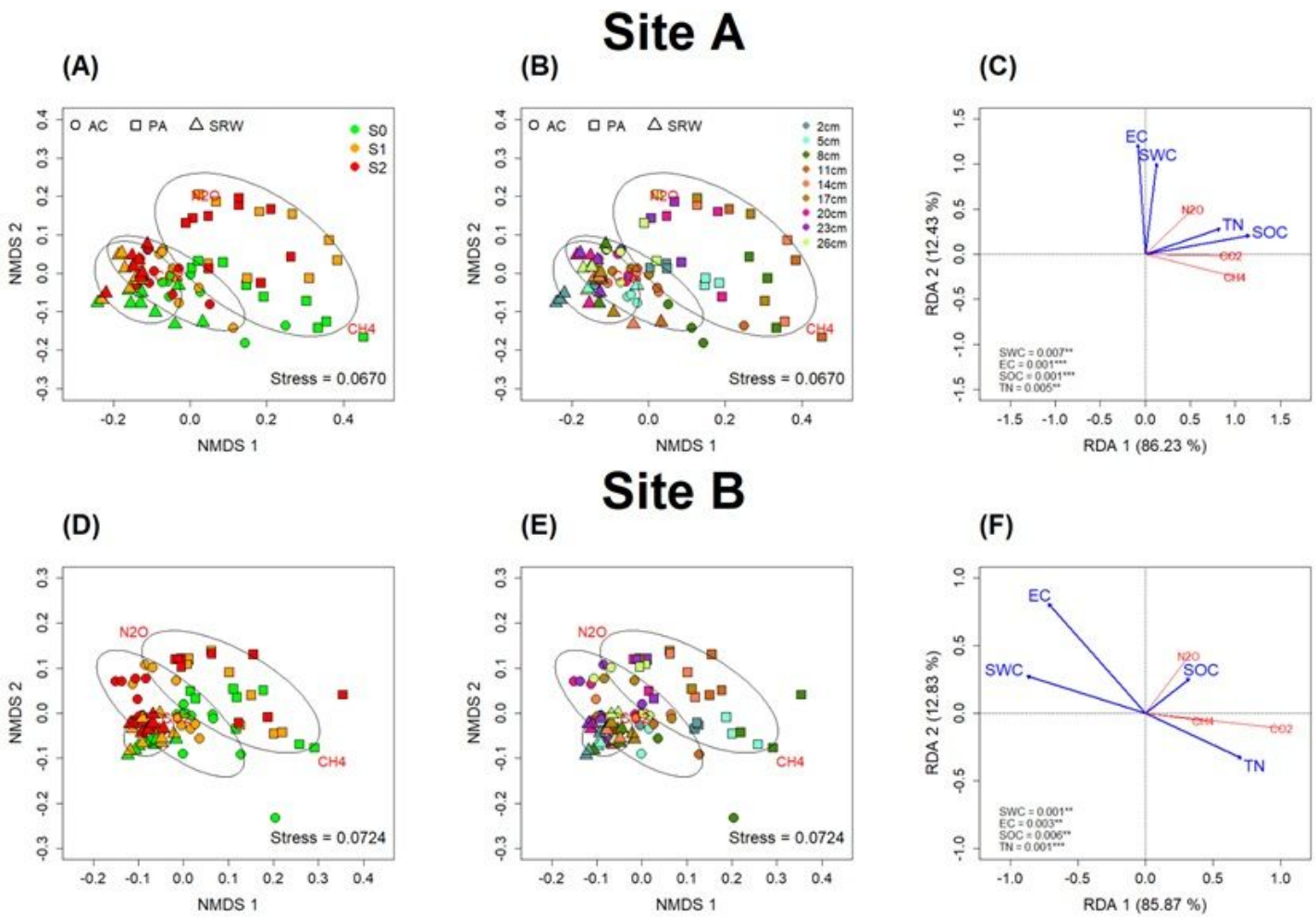

Figure 3

Non-metric multidimensional scaling (NMDS) test of soil GHG emissions visualized with land-use and groundwater salinity treatments, land-use and groundwater table depth treatments, and redundancy analysis (RDA) from site A (A, B, C) and site B (D, E, F). † Blue vectors indicate linear correlations between the ordination and soil physiochemical properties. Directions and lengths of the vectors indicate the strength of correlations between variables and the ordination. The angles between vectors reflect their correlations (i.e., a vector pair with an angle of 20° have strong positive correlation as cos(20) = 0.94, and with an angle of 90° are uncorrelated as cos(90) = 0). †† *, **, *** Indicate there is a statistically significant difference at $p \leq 0.05$, $p \leq 0.01$, and $p \leq 0.001$ level of significance, respectively; ns, is not significantly different ($p > 0.05$). ††† GHG = greenhouse gas, AC = annual crop, PA = pasture, SRW = short rotation willow, S0 = control, S1 = 6 mS cm-1, S2 = 12 mS cm-1, EC = electrical conductivity, SOC

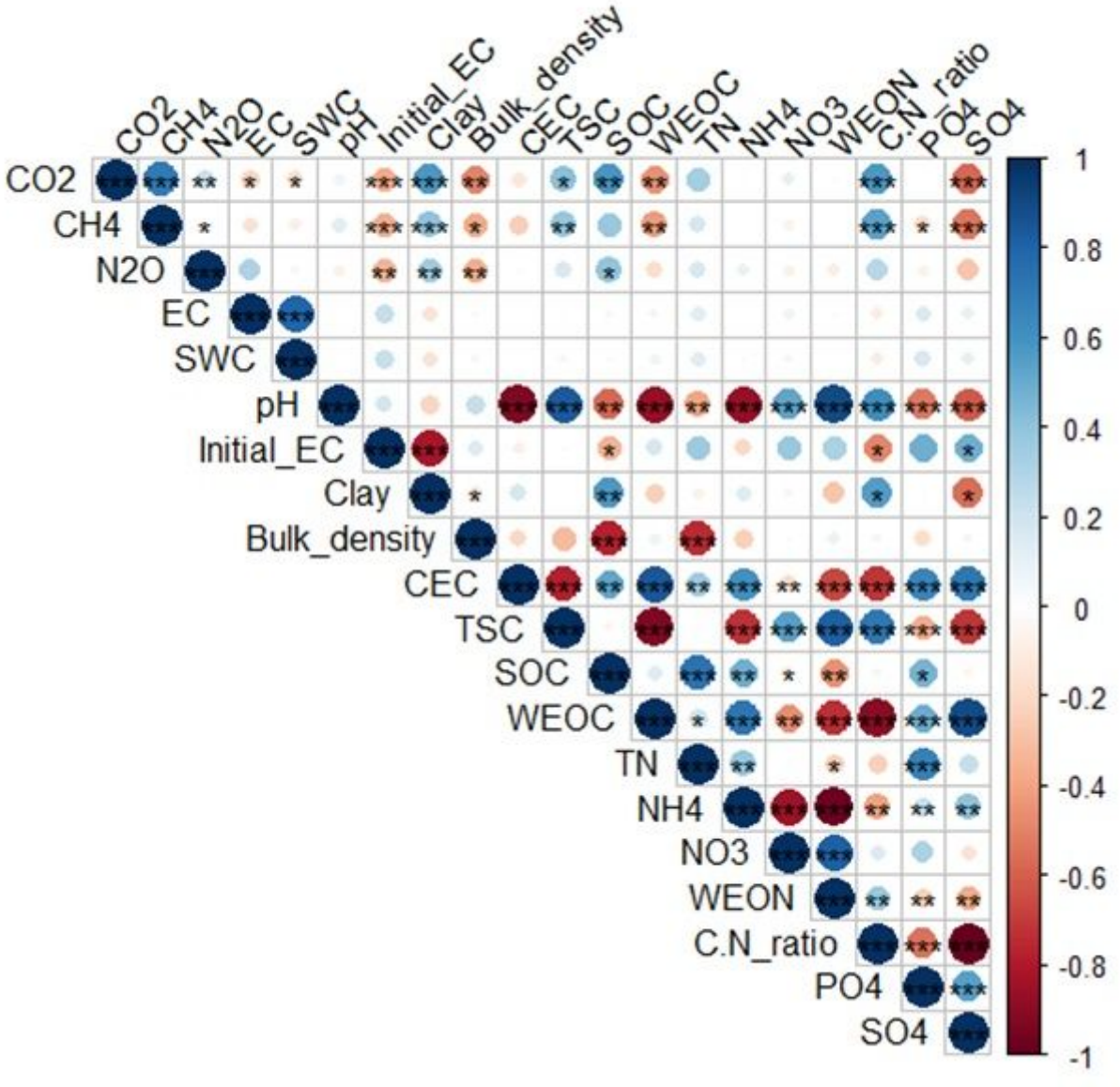

Figure 4

Relationship (Spearman rank-order correlation) among GHG, VSWC, EC, and physicochemical characteristics of the experimental soils. † Blue circles indicate positive and red circles indicate a negative relationship. Larger circles and deeper colors indicate stronger relationships. †† *, **, *** indicate there is a statistically significant relationship at $p \leq 0.05$, $p \leq 0.01$, and $p \leq 0.001$ level of significance, respectively; and the remainder are not significant ($p > 0.05$). ††† GHG = greenhouse gas, VSWC = soil water content, EC = electrical conductivity, CEC = cation exchange capacity, TSC = total soil carbon, SOC = soil organic carbon, WEOC = water-extractable organic carbon, TN = total nitrogen, WEON = water-extractable organic nitrogen, C/N ratio = carbon and nitrogen ratio.

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