9-2017

Effects of salinity and wet–dry treatments on C and N dynamics in coastal-forested wetland soils: Implications of sea level rise

Xijun Liu

Anhui Agricultural University

Alexander Ruecker

Clemson University

Bo Song

Clemson University

William H. Conner

Clemson University

Alex T. Chow

Clemson University, achow@clemson.edu

Follow this and additional works at: https://tigerprints.clemson.edu/forestry_env_pub

Part of the [Forest Sciences Commons](#)

Recommended Citation

Please use the publisher's recommended citation. <http://www.sciencedirect.com/science/article/pii/S0038071717304649>

This Article is brought to you for free and open access by the Forestry & Environmental Conservation at TigerPrints. It has been accepted for inclusion in Publications by an authorized administrator of TigerPrints. For more information, please contact kokeefe@clemson.edu.

**Effects of Salinity and Wet–dry Treatments on C and N Dynamics in Coastal-Forested Wetland
Soils: Implications of Sea Level Rise**

XIJUN LIU^{a,b}, ALEXANDER RUECKER^b, BO SONG^b, JING XING^c, WILLIAM H. CONNER^b, ALEX
T. CHOW^{b,d*}

^a Key Lab of Silviculture, School of Forestry and Landscape Architecture, Anhui Agricultural University,
Hefei, Anhui Province 230036, China

^b Baruch Institute of Coastal Ecology and Forest Science, Clemson University, South Carolina 29442,
United States

^c Department of Computer, Anhui Vocational College of Grain Engineering, Hefei, Anhui Province,
230011, China

^d Department of Environmental Engineering and Earth Sciences, Clemson University, South Carolina
29631, United States

*Corresponding Author: Tel.: + 8435461013-232

Email address: achow@clemson.edu.

Abstract

Forested wetlands dominated by baldcypress (*Taxodium distichum*) and water tupelo (*Nyssa aquatica*) are commonly found in coastal regions of the southeastern United States. Global climate change and in particular sea level rise will alter the frequency and magnitude of wet/dry periods and salinity levels in these ecosystems. Soil microcosm experiments were set up to identify the effects of water level variations (0.4-3.0 g-water g-soil⁻¹) and salinity changes (0, 1 and 5 ppt of NaCl) on greenhouse gas emissions (CH₄, CO₂ and N₂O) and dissolved organic carbon (DOC) characteristics from forested wetland soils. Our results indicate that, the effect of water level was much greater than salt intrusion on C and N cycling. Wet-dry treatments significantly decreased DOC production and total CH₄-C loss, aromatic and humic-like substance compounds in DOC were increased in both flooding and wet-dry treatments after 60-d incubation. The molecular weight (MW) of DOC after flooding treatments was higher than that in wet-dry treatments. A first order kinetic model showed there was a positive linear correlation ($r^2=0.73$) between CO₂ emission rate and DOC concentration which indicated that CO₂ was mainly generated from DOC. An exponential kinetic model was applied to describe the correlation between CH₄ emission rate and DOC concentration ($r^2=0.41$). This study demonstrates that an increase in salinity, and in particular variations in wet-dry cycles, will lead to changes in the formation of climate-relevant greenhouse gases, such as CH₄, CO₂, and N₂O.

Keywords: Dissolved Organic Carbon; Greenhouse Gases; SUVA; Tidal Wetlands

Introduction

Climate change, and in particular sea level rise, is impacting coastal forest and wetland ecosystems. The mean rate of sea level rise in the past 10 years has been exceeding the mean rate of global sea level rise of 0.19 m, reported from 1901 to 2010 (IPCC, 2014). Sea level rise leads to an increase in salt water intrusions (Donato et al., 2011; Breithaupt et al., 2012; Chow et al., 2013) and soil inundation and moisture, which implies greater risks and frequency of flooding (Zhu and Cheng, 2013; Woodruff et al., 2013) and increased soil drying–rewetting cycles (Morillas et al., 2015). Coastal forested wetlands will be among the environments most likely impacted by a combination of alterations in soil biogeochemistry experiencing saltwater intrusions in the next 50-100 years (Schofield, 2003; Allen et al., 1996; Renaud et al., 2015; Donato et al., 2011; Breithaupt et al., 2012; Chow et al., 2013) and hydrology due to changed wet-dry dynamics. Of particular interest are alterations in SOM (soil organic matter) levels and DOC (dissolved organic carbon) structure, which will affect carbon sequestration, soil microbial activity, and consequently biogeochemical element cycling and soil to air gas exchange (Moyano et al., 2013; Chow et al., 2003, 2005; Lal, 2004; Moseman-Valtierra et al., 2011).

Previous studies have demonstrated that decomposition rate of DOC (Fierer and Schimel, 2002) and the quantity, as well as, the chemical characteristics of DOC are affected by water level fluctuations and wet–dry cycles (Lundquist et al., 1999; Kalbitz et al., 2000; Chow et al., 2003). Positive correlations between C mineralization rate and DOC concentration were found in coastal wetlands (Cook and Allan, 1992; Chow et al., 2006). The impact on greenhouse gas formation however, remains poorly understood. While Shi and Marschner (2014) reported an increase in GHG formation as a consequence of varying water levels, Morillas et al. (2015) reported a decline in GHG formation based on a lower microbial biomass and microbial activity.

In addition to fluctuations in water levels, the intrusions of sea water into coastal freshwater environments is of particular interest as higher salt concentrations will contribute to changes in biogeochemical turnover of C and N (Chow et al., 2013; Lewis et al., 2014). Recently the consequences of freshwater wetland salinization have been summarized in a review by Herbert and colleagues (Herbert et al., 2015). Therefore, we will only mention the most important impacts on N and C dynamics and refer the reader to the work by Herbert et al. (2015) for an in-depth discussion on the impacts of wetland salinization. The combination of an enhanced N mineralization and dissimilatory nitrate-reduction coupled to

ammonium, a reduced nitrification-denitrification, and an increase in NH_4^+ are the direct consequences of saltwater intrusions on N dynamics in wetlands (Ardón et al., 2013). Interestingly it has been demonstrated that an increase in salinity can also have a stimulating effect on nitrification in saline soils and that an inhibition only occurred when electric conductivity (EC) exceeded $16,000 \mu\text{S cm}^{-1}$ (Ardón et al., 2013). The main consequence of increasing salinities on carbon dynamics is a decrease in plant productivity, and hence carbon inputs to the soil. Additionally, it will cause a decrease in microbial activity and consequently slower DOC decomposition rates (Setia et al., 2013). In particular, salinization affects solubility and mobility of DOC and potentially the formation of CH_4 (Ardón et al., 2016; Mulholland, 1981; Sholkovitz, 1976). Therefore, an increase in salinity as it occurs in coastal saline wetlands may lead to lower CH_4 emission rates compared to freshwater wetlands (Baldwin et al., 2006; Marton et al., 2012).

As the changes in C turnover occur, it can be expected that the turnover of N, and in particular the formation of the greenhouse gas nitrous oxide (N_2O) will be affected, too. Previously, it has been demonstrated that emission of N_2O is the primary pathway of N loss from wetland soils into the atmosphere, and that variations in water levels and conductivity affect N_2O emissions from an alkaline soil (Silva et al., 2008; Kraus and Whitbeck, 2012). The impacts of different salinities in combination with varying water levels on N_2O formation has, to the best of our knowledge, not yet been investigated. The above mentioned aspects illustrate the necessity of a thorough quantification of greenhouse gas emissions under varying water levels and salinities to gain a better understanding of potential alterations in atmospheric levels of these climate-relevant gases. In particular, the contribution of different C and N fractions (DOC, soil organic carbon (SOC), total nitrogen (TN), and soil nitrogen (SN) as precursors for greenhouse gas formation is also of interest. Additionally, the identification of DOM optical properties is essential as it serves as a C source for microorganisms in the soils and is crucial for biogeochemical turnover of C and N.

The flow of C and N through soil is inherently complex. Many thousands of different chemical transformations and physical processes occur simultaneously within the soil matrix. Instead of tracking individual reactions and processes, kinetic models based on empirical and experimental data can predict the flows of C and N even if the reaction mechanisms are not known (Glanville et al., 2016). These approaches have been used to describe different nutrient cycling in wetland soils (Chen and Rudolf, 2016; Chow et al. 2006; Chow et al., 2004). Results of the kinetic model could also provide insights about the

biogeochemical processes within soil matrix. For example, Chow et al. (2006) used a 1st order kinetic model to demonstrate that CO₂ was mainly produced from DOC in soil pore water under oxidized condition, whereas CO₂ was produced from SOC under reduced conditions. However, the relationships of CH₄ and N₂O emissions on DOC and DTN, as well as impacts of salinity have not examined before.

Therefore, the overall goals of this study were (I) to quantify greenhouse gas emissions (CO₂, N₂O, and CH₄) under varying salinities, and water levels, (II) to demonstrate the effects of varying salinities and water levels on DOM optical properties and (III) to develop a kinetic model to elucidate the contribution of different C fractions to CO₂ and CH₄ formations in our microcosm experiments.

2. Materials and experimental methods

2.1. Field site and sampling procedure

The study was conducted with soil samples from a healthy forested wetland on Hobcaw Barony (33°21'N, 79°12'W), near Georgetown, South Carolina, which is currently not affected by saltwater intrusions. The highly productive (average annual litterfall input of 620 g m⁻² yr⁻¹; Conner, unpublished), seasonally flooded 24-ha (2.4×10⁵ m²) wetland is dominated by baldcypress (*Taxodium distichum* (L.) Rich.), water tupelo (*Nyssa aquatica* L.), and swamp tupelo (*Nyssa biflora* Walt.) trees. The surrounding forest community included longleaf pine (*Pinus palustris* Mill.) and turkey oak (*Quercus laevis* Walt.) (Busbee et al., 2003). The predominant soil type in this wetland has been characterized as a fine-loamy, siliceous, thermic, and Yypic Ultisol (USDA Soil Taxonomy) (Stuckey, 1982). Detailed characterizations of this site, including tree composition, stem density, aboveground productivity, litterfall, nutrient dynamics, and hydrology have been published previously (Busbee et al., 2003; Chow et al., 2013). Surface soil samples were collected from 0–10 cm after removal of litter from the surface during the wetland's dry period in the summer. The soils were transported to the laboratory, air-dried at room temperature (22 ± 1°C) until constant weight, sieved to a particle size < 2 mm for homogenization, and stored at room temperature until further analysis and start of the soil incubation experiments. This homogenization might have impacted the microbial activity or microbial community. Therefore, results from the laboratory microcosm incubations presented here are not directly transferable to results that can be expected from the field. Relevant soil characteristics are summarized in Table 1.

2.2. Setup of microcosm experiments and sampling procedure

Microcosm experiments for the quantification of greenhouse gas formation in varying approaches were set up using one-liter mason jars. Each jar contained 50 g of dried ground surface soil and 150 g of deionized water or solutions of 1 ppt and 5 ppt NaCl, to simulate freshwater and two levels of oligohaline-degraded wetlands with the dieback of some trees in the canopy layer (Cormier et al., 2013). Hereafter the 3 setups are labelled as 0, 1, and 5 ppt. Soils were incubated either under constant flooding conditions or wet-dry cycles of 12-d for a total of 60 days at room temperature. Besides the 3 different salinities, we applied two different water levels (permanent flooding, and wet-dry cycles). A total of 120 jars were used (18 jars x 6 treatments + 12 blank controls). All jars were incubated at room temperature in the dark. In the flooding experiment, each jar was covered by a lid with a 2-mm opening to allow for gas exchange. The θ_g in each jar was monitored regularly by gravimetric measurement. Water content was readjusted with deionized water when a 5% or more change in θ_g occurred. In the wet–dry cycle incubation experiment, surface soils were initially flooded with a θ_g of 3.0 g-water g-soil⁻¹. All jars were incubated at room temperature without lids so that soil samples were allowed to dry naturally through evaporation. When the soils reached the desired θ_g values, soils were re-flooded to their initial θ_g with deionized water for continued incubation.

For pH, electrical conductivity (EC), DOC, dissolved nitrogen (DN), UV-VIS, and spectrofluorometry, three replicates of each incubation condition were terminated at 12, 24, 36, 48, and 60-d. To obtain water extracts, deionized water was added to the original water level (150 g of water to 50 g soil) and shaken rigorously for 5 minutes and then filtered through a 0.45µm membrane filter (Millipore Express) prior to analysis. All samples were kept at 4 °C and analyses were completed within a week. In addition to the DOC extraction, three replicates of each treatment were used to quantify CO₂, CH₄, and N₂O emissions during the 60-d incubation. Eighteen jars for gas measurement were sealed with a gastight lid equipped with a removable rubber septum. The jars were capped every 3 days and gas samples were collected after 24 hours. A septum was used to seal the opening for 24 h before sampling and analysis. Twenty-four ml of gas was withdrawn from the sealed jar for quantification of CO₂, CH₄, and N₂O, expressed as emissions in µg g-soil⁻¹ d⁻¹. In addition, three empty jars were used to determine CO₂, CH₄, and N₂O background levels. CO₂, CH₄, and N₂O production from the soils was determined by subtracting the CO₂, CH₄, and N₂O concentrations of the empty jars.

2.3. Analysis

Concentrations of C and N in the original soils (0 day) and after 60-d incubation (oven dried at 70 °C for 48 h) were quantified using a Thermo Flash EA 2000 element analyzer. CO₂, CH₄, and N₂O were analyzed by a Shimadzu Greenhouse Gas analyzer (GC 2014). All filtered solution samples were analyzed for pH, EC, DOC, and DN. The pH and EC of filtered solution samples were determined using an Accumet XL60 dual channel pH/ion/conductivity meter. EC was corrected to EC at 25 °C (EC₂₅) using the temperature correction equation: $EC_{25} = EC_t / [1 + 0.021 (t-25)]$ (Hayashi, 2004). The DOC and DN were measured using a Shimadzu TOC/TN analyzer (SM 5310B).

DOM optical properties were further characterized by UV-VIS spectrophotometry (Shimadzu UV-1800) and spectrofluorometry (Shimadzu Spectrofluorometer RF5301). All DOC samples also were analyzed for Specific ultraviolet absorbance (SUVA), and one of the 3 replicates was used for spectrofluorometry. Specific ultraviolet absorbance, spectral slope ratio (S_R), and E₂/E₃ ratio were determined as described previously (Chow et al., 2008; Helms et al., 2008; Wang et al., 2015). Based on SUVA and E₂/E₃ ratio, selected samples were analyzed using fluorescence emission-excitation matrix (EEM) (Zhou et al., 2013). Several spectrofluoroscopic indices including the fluorescence index (FI), the freshness index (β/α), and humification index (HIX) were calculated as shown previously (Fellman et al., 2010; Wang et al., 2015; Cory and McKnight, 2005).

2.4. Calculation of gas fluxes and statistical analysis

Fluxes of CO₂, CH₄, and N₂O were calculated on the basis of daily data using the ideal gas law. As CO₂, CH₄, and N₂O concentrations were determined every four days, their mineralization rates during sampling intervals were assumed equal to the average of the two mineralization rates from the two sampling events when calculating the cumulative production (Chow et al., 2006). Greenhouse gas fluxes and concentrations of DOC and DN were used to examine the relationship among CO₂-C, CH₄-C and DOC, N₂O-N, and DN. The least squares method was used to construct the best fit between formation of CH₄, and CO₂ and DOC concentration, as well as between N₂O formation, and DN concentration. The slope of each linear regression line was equal to the reaction rate constant for the specific incubation condition. One-way ANOVA (SPSS, 20.0) was used to detect statistically significant differences in the effects of salinity and wet-dry treatments on C and N cycles, respectively.

2.5 Development of kinetic model

The above mentioned reaction rates were used to develop a first-order or an exponential kinetic model demonstrating the importance of different carbon fractions (SOM and DOM) in the formation of the greenhouse gases (CO₂, and CH₄) under flooding conditions. Interestingly, our first order or exponential kinetic model was suitable to detect differences in the precursors of CO₂ or CH₄ in the present study. This demonstrates the importance of a thorough DOM characterization and how changes in the salinity can affect C turnover. Unfortunately, we were not able to expand our model to the wet-dry treatment. There are possible two reasons accounting for the limitation using the model in wet-dry treatment. First, the variations in water content most likely affects DOC availability and GHG transport processes in soil. The kinetic model here focuses on the chemical reactions and it does not consider other transportation processes. During the wet-dry processes, water evaporation and other gas transport, which were not accounted in the proposed reaction kinetic model, could be factors affecting DOC concentration and GHG emission. Besides, microbes might be stressed and microbial communities might be altered during the wet-dry processes. The model could not predict the lag phases of GHG or DOC productions.

One of the reasons for that is, that variations in water content most likely affect DOC availability and its transport processes in soil which cannot be described by the proposed reaction kinetic model. In the kinetic model, daily CO₂-C emission rates (i.e., d[CO₂-C]/dt) were equal to the sum of k_{SOC} [SOC] and k_{DOC} [DOC], where the constants k_{SOC} and k_{DOC} are the reaction rates of SOC and DOC in forming CO₂. The first-order kinetics we used to describe the reaction among d[CO₂-C]/dt, and SOC, DOC was:

$$d[CO_2 - C]/dt = k_{SOC}[SOC] + k_{DOC}[DOC]$$

In the exponential model, daily CH₄-C emission rate (i.e., d[CH₄-C]/dt) was equal to K_{SOC}exp(K_{DOC}[DOC]), where K_{SOC} is the reaction rate constant or C mineralization rate from SOC to CH₄, and K_{DOC} is solubility or production of DOC from SOC. The correlation equation we used to describe the reaction among d[CH₄-C]/dt and DOC was:

$$d[CH_4 - C]/dt = K_{SOC} e^{K_{DOC}[DOC]}$$

3. Results

3.1 Kinetic models for C and N cycles

We observed that total CO₂-C (12.1-12.4 mg-C g-soil⁻¹) and CH₄-C losses (2.6 - 2.9 mg-C g-soil⁻¹) after 60-d incubation exceeded initial DOC concentrations (1.0 – 1.2 mg-C g-soil⁻¹) for all treatments suggesting both SOC and DOC could be substrates for CO₂ and CH₄ production (Boyer and Groffman,

1996). However, SOC was considered a relatively large C pool and the change in SOC content within the 60-d incubation should be minimal in comparison to the C changes in gas and water phases. In order to delineate the relationships of the dynamic variables, CO₂-C or CH₄-C was plotted against DOC concentration, as shown in Figure 1a and 1b, respectively. A linear correlation with ($R^2 = 0.73$) between CO₂-C and DOC was observed (Fig. 1a), whereas an exponential correlation ($R^2 = 0.41$) between CH₄-C and DOC was obtained (Fig. 1-b). DOC concentration is the same between CO₂ and CH₄ during the incubation, but there is no CH₄ emitted at the beginning of incubation, so we deleted the high DOC with no CH₄ emission in figure 1-b. We applied a kinetic model to demonstrate the contribution of different carbon fractions and concentrations on formation rates of CO₂ (Chow et al., 2006). Wet-dry treatments are not modeled by this approach because the variation in water content could change DOC availability as well as its transport processes, which cannot be simply described by the reaction kinetic model. In the kinetic model, total available organic carbon (TAOC) is the sum of degradable SOC and DOC, which are available and accessible to microbes during the 60-d incubation. TAOC was not necessarily equal to total organic C in soils because not all of the C would be involved in the reactions occurring during the 60-d period. In addition, DOC could be produced at an independent reaction rate constant k_{SD} by microbes utilizing SOC as C source. The reaction rate constant k_{SD} is not equal to k_{SOC} because the mechanism producing CO₂ is probably different from that producing DOC (Moore and Dalva, 2001; Chow et al., 2006).

The CO₂-C emission rate was linearly proportional to DOC concentration; therefore, we applied the first order kinetic model to describe the relationship. CO₂ emission rate ($d[CO_2-C]/dt$) is equal to CO₂ production from SOC and DOC, written as $k_{SOC}[SOC]$ and $k_{DOC1}[DOC]$ in equation [1] in Figure s2. After substituting and manipulating the variables as shown in Box in Figure s2, a linear relationship ($y = ax + b$) was obtained as shown in equation [4], where y is the CO₂ mineralization rate ($d[CO_2-C]/dt$), a is the difference between reaction rate constants ($k_{DOC1}-k_{SOC}$) alternatively called an apparent reaction rate constant (k_{app}), x is the DOC concentration, and b is the y-intercept and is equal to TAOC concentration with a factor of k_{SOC} . The linear equation [4] in Figure s2 can be used to predict the sources of CO₂ emission. Based on this model, the correlation between CO₂ emission rate and DOC concentrations depends on k_{app} , which is a function of k_{DOC1} , and k_{SOC} . The slope of the linear relationship between CO₂-C emission rate and DOC is positive, indicating that the rate constant of k_{DOC1} representing microbes mineralizing DOC is

greater than the rate constant of k_{SOC} representing mineralizing SOC (Chow et al., 2006). This suggests that microorganisms preferentially utilized DOC to produce CO_2 over SOC.

In contrast to CO_2 emission, the linear equation did not fit well when CH_4 emission rate ($d[\text{CH}_4\text{-C}]/dt$) was plotted against DOC concentration. Instead, the $\text{CH}_4\text{-C}$ emission rate had a relatively close fit ($r^2 = 0.41$) with an exponential function, as expressed by $d[\text{CH}_4\text{-C}]/dt = K_{\text{SOC}} \times \exp(K_{\text{DOC}}[\text{DOC}])$ (Fig. 1b). Here, y is the $\text{CH}_4\text{-C}$ emission rate and x is DOC concentration. If DOC concentration of the system is zero (i.e. $[\text{DOC}] = 0$), $d[\text{CH}_4\text{-C}]/dt$ is equal to K_{SOC} with unit of concentration over time. Therefore, K_{SOC} is the reaction rate constant or C mineralization rate directly from SOC to CH_4 in our system. In addition, K_{DOC} has a unit of an invert of DOC concentration because a value of exponential function should be dimensionless. Therefore, K_{DOC} , with a unit of $\text{g-soil} / \mu\text{g-C}$ in our case, is the solubility or equilibrium constant of DOC from SOC. Overall $\text{CH}_4\text{-C}$ emission rate is a factor of K_{SOC} with an exponential function that depends on the SOC to DOC production. If K_{DOC} is small, the function flattens out, suggesting a slower rate of $\text{CH}_4\text{-C}$ emission. If K_{DOC} is large, the curve increase rapidly, suggesting a rapid production of $\text{CH}_4\text{-C}$. In this study, although we only found a rather weak correlation between DOC and CH_4 ($r^2 = 0.41$), it is conceivable that the activity, and potentially also population size, of Archaea responsible for CH_4 formation increased with increasing DOC concentration, explaining at least to some extent the observed relation between CH_4 and DOC. Nevertheless, it has to be considered that many other factors such as redox conditions play a crucial role for CH_4 formation that are not considered in the equation above and that a good correlation between variables does not imply causality.

The same correlation analyses were done for N_2O formation and DN. The experimental data suggest a negative linear correlation between N_2O and DN ($R^2 = 0.45$; data not shown). However, the data set showed two distinct data clouds and thus we can only speculate about the actual contribution of DN to N_2O formation and the factors that are relevant for N_2O emissions from our setups.

3.2 GHG dynamics under salinity and wet-dry treatments

Figure 2 shows formation of CO_2 , CH_4 , and N_2O from our soil microcosm incubations over 60-d under different salinities and water levels. Results demonstrated that water level significantly altered the patterns of CO_2 emissions but salinity did not (Figs. 2a-c). In the three salinity treatments under permanent flooding (blue line), CO_2 emissions were highest on day 9, with an average of $417.5 (\pm 15.8)$, $440.6 (\pm 8.7)$, and

464.7 (± 14.6) $\mu\text{g-C g-soil}^{-1} \text{ d}^{-1}$ at 0 ppt, 1 ppt and 5 ppt of NaCl, respectively, and gradually decreased over time. Considering the highest CO_2 emission rate, an increase in salinity apparently increased the CO_2 emission rate but the differences were not statistically significant ($p > 0.05$). Differences in total $\text{CO}_2\text{-C}$ losses over 60-d of incubation were 12.2 (± 0.5), 12.4 (± 0.4), and 12.1 (± 0.5) mg-C g-soil^{-1} at 0 ppt, 1 ppt and 5 ppt of NaCl, respectively, and found to be statistically not significant (Table 2). Under the wet–dry cycle treatment, CO_2 flux followed closely with soil water content. The highest CO_2 flux in each cycle occurred 8 days after each re-flooding, and then the CO_2 mineralization rate declined and was lowest when the soils had the lowest θ_g . Moreover, the highest CO_2 flux decreased with wet–dry cycles during the first 8 days of incubation. Total $\text{CO}_2\text{-C}$ loss during the 60d incubation in wet–dry cycle was 11.7 (± 0.5), 12.8 (± 0.7), and 11.6 (± 0.5) mg-C g-soil^{-1} at 0 ppt, 1 ppt and 5 ppt NaCl treatments, respectively. Generally, total $\text{CO}_2\text{-C}$ loss under the wet-dry treatment was statistically equal to flooding condition in the 60-d incubations ($p > 0.05$).

CH_4 was not detected until the 12th day of incubation in both wet-dry and flooding treatments. Our results demonstrated that CH_4 formation was mainly influenced by fluctuations in water level, whereas effects of different salinities on CH_4 formation were negligible (Figs. 2d-f). In all salinity and wet-dry treatments, the highest CH_4 fluxes occurred after 21 days of incubation, then decreased to 19%-28% at flooding and 1%-6% at wet-dry contrast to highest fluxes and remained stable until the end of the experiment. Highest CH_4 fluxes from 0, 1, and 5 ppt NaCl were 148.3, 159.7 to 123.0 $\mu\text{g-C g-soil}^{-1} \text{ d}^{-1}$, respectively. Wet-dry treatments significantly reduced CH_4 emission rates and total $\text{CH}_4\text{-C}$ losses from all treatments (Fig. 2).

In contrast, both water level and salinity significantly influenced the emission dynamics of N_2O (Figs. 2g-i). Under flooding conditions, high salinity reduced the peak of N_2O emission rate from 9.0 $\text{ng-N gsoil}^{-1} \text{ d}^{-1}$ (0 ppt NaCl), 5.3 $\text{ng-N g-soil}^{-1} \text{ d}^{-1}$ (1 ppt NaCl) to 0.4 $\text{ng-N g-soil}^{-1} \text{ d}^{-1}$ (5 ppt NaCl), with a total $\text{N}_2\text{O-N}$ loss of 0.10, 0.08, and 0.02 $\mu\text{g-N g-soil}^{-1}$, respectively. Under wet–dry cycles, N_2O emissions followed closely with soil water content, especially in the later incubation. Highest N_2O emissions were quantified immediately after re-flooding for each wet-dry treatment and decreased with decreasing soil water content. Interestingly, N_2O emission rate apparently increased at the later cycles of wet–dry incubation. Total $\text{N}_2\text{O-N}$ losses after 60-d of incubation during wet–dry treatments were 0.22 (± 0.03), 0.17 (± 0.01), and 0.15 (± 0.02) $\mu\text{g-N g-soil}^{-1}$ at 0 ppt, 1 ppt, and 5 ppt NaCl, respectively, which was significantly higher than during

flooding ($p < 0.05$). Compared to the initial concentrations, TC, TN, and DN in the soil actually decreased for all treatments after 60-d incubation, but changes were not statistically significant. Total C loss ($\text{CO}_2\text{-C} + \text{CH}_4\text{-C}$) after 60-d incubation was 6.2% of original TC, and total $\text{N}_2\text{O-N}$ loss after 60 days was 0.1% of original TN.

3.3 Water quality

The original forested wetland soil pH was slightly acidic with an average of 5.0 ± 0.1 ($n = 3$), but increased to (6.0 - 7.5) during the 60-d incubation. No significant difference was found among the salinity treatments ($p > 0.05$) although the average pH decreased with salinity. The pH under flooding conditions (6.2 - 7.5) was significantly higher than under wet-dry treatment conditions (5.9 - 6.1) ($p < 0.05$). An increase of pH in soils with increasing salinity is possibly due to the carbonate and a relatively high pH of sea water (pH ~ 8). In our study, we focus on the effect of NaCl. Without the carbonate as buffer reagents, the soil pH could be easily altered. Moreover, a decrease of pH is commonly observed in submerged wetland soils. Organic soil is often acidic during submergence through the slow oxidation of sulfur compounds, producing sulfuric acid, and the production of humic acids (Mitsch and Gosselink, 1993). In fact, we did not observe a decrease in pH in the wet-dry treatment. This suggests that the presence of NaCl is not the main driving force in the observed pH changes.

Figure 3 shows the temporal variations of DOC and DN concentrations as well as DOC/DN ratios over 60-d incubation. Original DOC concentrations in all cases were the lowest with $1.2 \pm 0.0 \text{ mg-C g-soil}^{-1}$ (Figs. 3a-c). After 60-d of incubation DOC concentrations increased significantly under flooding treatments ($1.7 \pm 0.1 - 2.2 \pm 0.3 \text{ mg-C g-soil}^{-1}$) and 5 ppt in the wet-dry treatments ($1.6 \pm 0.1 \text{ mg-C g-soil}^{-1}$). DOC concentrations were highest after 12 days of incubation and increased 3.4 - 4.5 times under flooding ($\theta_g = 3.0$) and 1.8 - 2.4 times under wet-dry cycles ($\theta_g = 0.4 - 3.0$). The same trend was observed with varying salinities. However, DOC concentrations were significantly higher in the permanent flooding incubation compared to incubation under wet-dry conditions ($p < 0.05$). Although DOC concentrations at 0 ppt and 1ppt NaCl during flooding exceeded DOC concentrations in the 5ppt NaCl

treatment, no statistically significant difference in mean DOC concentrations were observed between the salinity treatments ($p > 0.05$).

DN concentration had a similar trend over time with varying salinity under both flooding and wet-dry treatments (Figs. 3d-f). DN concentrations increased rapidly from the lowest concentration in original soil (0 day), then fluctuated during the remainder of incubation. DN concentrations under flooding increased 2.9 - 3.6 times after 60-d incubation and more than 2.0 - 3.3 times under wet-dry treatments.

The dynamics of DOC/DN ratios were similar to DOC concentration starting at 4.4 - 6.2 and reaching its highest level of 5.7 - 8.7 after 12 days of incubation, then decreasing gradually to 2.5 - 3.0 (Figs. 3g-i). The range of DOC/DN ratios was greater with flooding than that in wet-dry treatments, and with increasing salinity, the peak of DOC/DN ratios was reduced, but ratios after the 60th day were similar.

3.4. Optical characterization of DOM

Samples for characterization of DOM optical properties were collected every 12 days and included analyses of SUVA, S_R , E_2/E_3 ratio, and Fluorescence EEM. Similar to the other parameters described above, there were no obvious differences between salinity treatments ($p > 0.05$), but differences were observed between flooding and water level treatments ($p > 0.05$) (Fig. 4). Higher salinity treatments lowered SUVA, but the difference was not statistically significant ($p > 0.05$) during the 60-d incubation (Figs. 4a-c).

E_2/E_3 ratios and S_R showed similar temporal trends at different salinities under both permanent flooding and wet-dry treatments, however no statistically significant difference among salinity was found ($p > 0.05$) (Figs. 4d-i). E_2/E_3 ratios under wet-dry treatments were significantly higher than under flooding ($p < 0.05$). In the same incubation period, DOC concentration increased gradually, while E_2/E_3 dropped to its minimum on day 12 in flooding and wet-dry treatments. SUVA and S_R increased and E_2/E_3 ratios decreased over time when comparing values at the beginning of the experiment and after 60 days of permanent flooding. Furthermore, lower SUVA but higher E_2/E_3 ratios of DOC were found at 60-d during wet-dry cycling.

Results from 3D fluorescence regional integration and emission-excitation matrix are shown in Fig. 5. From fluorescence index (FI), freshness index (β/α), and Fluorescence regional integration, we can conclude that DOM has higher MW in both flooding and wet-dry treatments after 60-d incubation compared to its original properties. Humic-like DOM was the largest component, comprising about 34-

65% of DOM, and protein-like DOM was the second most abundant fraction, comprising 20 to 41% of DOM in all treatments. Although the proportion of various components of DOM fluctuated during the incubation under various salinity and water level conditions, the observed structural changes were statistically not significant ($p > 0.05$) (Fig. 6). Compared to original DOM, fulvic-acid like and humic-acid like DOM are the most dominant fractions after 60 days.

4. Discussion

4.1 Contribution of different C fractions to GHG formation

The linear relation between DOC and CO₂ emissions suggests that CO₂ is mainly formed from DOC and that SOC only contributes to a minor extent to CO₂ emissions from our microcosm experiments. In contrast to that, an exponential function was applied to fit CH₄ formation in our setups. Initially we also applied a linear model to our CH₄ data, however we did not find a correlation between DOC and CH₄ (data not shown). In a next step we used a second order quadratic model which showed a relatively good correlation between DOC and CH₄ ($R^2=0.70$) but this model does not make sense from a biogeochemical point of view as it is based on an initial decrease in DOC with a simultaneous increase in the emissions of CH₄. Therefore, we developed an exponential model to describe the contribution of DOC to CH₄ formation in our microcosms under permanent flooding conditions. Although the correlation was considerably weaker ($R^2=0.41$) than observed for the quadratic model, we believe that it better describes the actual relationship between DOC and CH₄.

However, it also demonstrates that the formation of CH₄ in our setups is more complex than the formation of CO₂ and that many other factors besides the availability of DOC play an important role for CH₄ formation. Some factors that might be of importance are redox conditions but also the competition between methanogens and other strictly anaerobic microorganisms, such as sulfate reducers and Fe(III)-reducers might play a role. Nevertheless, we can only speculate about the underlying formation mechanism and the contribution of DOC to CH₄ formation as correlations are only a statistical tool and do not imply causality. The observed differences in the precursor substances of CO₂ and CH₄ are still of great relevance in regard to global climate change. Rising mean temperatures will affect soil C decomposition rates and thus carbon structure which might lead to alterations in the formation rates of the greenhouse gases CO₂ and CH₄.

(Davidson and Janssens, 2006). However, these conclusions are based on laboratory soil incubations and in how far these results are transferable to conditions in the environment is currently unknown.

4.2 Carbon –salinity, water level (flooding and wet-dry)

4.2a Carbon – salinity vs water level

Rising sea level has increased the hydroperiod and salinity in low-lying coastal freshwater forested wetlands (Krauss and Whitbeck, 2012). These environmental changes alter the C and N biogeochemical processes in wetland soils in different ways. Our experiments suggest that an increase in salinity under oligohaline conditions (≤ 5 ppt) did not significantly alter C cycling in terms of CO_2/CH_4 emission and DOC production (Figs. 5 a and c). For gas emissions, our results are in line with a previous study showing no effect on CO_2 or CH_4 emission with mean pore water salinity ranging from 0.2 to 4.7 ppt in the same type of forested wetland (Krauss and Whitbeck, 2012). Results suggest the impacts of low levels of saltwater intrusion on CO_2 and CH_4 productions from coastal wetland soils were negligible. However, it has been demonstrated that fluctuations in salinity affect DOC production in coastal wetlands (Olsen et al., 1996; Ardón et al., 2016; Chambers et al., 2014). Olsen et al. (1996) and Ardón et al. (2016) demonstrated that saltwater intrusions could reduce DOC in leachate while Chambers et al. (2014) showed an increase in DOC concentration in soil pore water. Our study did not observe any differences among salinity treatments after 60-d incubation. While CO_2 and CH_4 emissions are mainly controlled by microbial processes, DOC concentrations and exports are the result of a combination of environmental biotic and abiotic factors (Chow et al., 2003). The inconsistencies among studies are probably due to the involvement of several physical (e.g., coagulation and adsorption), chemical (e.g., photochemical and redox processes), and biological (e.g., microbial production and decomposition) processes during DOC production in soil water (Zsolnay, 2003). In this study, we specifically investigated the effects of NaCl and hydroperiod on freshwater wetland soils under low salinity levels (≤ 5 ppt) for a period of 60 days. An increase in DOC during the incubation was mainly due to the effects of water level (see section 4.2b). Results demonstrated that the effect of salinity on C cycling in oligohaline environments in a short period of time is negligible. Water level or the hydroperiod is the driving force on C cycling under oligohaline conditions. However, it has to be considered that results from a simplified laboratory incubation cannot directly be transferred to conditions in the field which are much more complex. In particular, the choice of salt can be of relevance.

Although NaCl is the predominant salt in sea water it also contains divalent ions such as Ca^{2+} or Mg^{2+} which could lead to aggregate formation and flocculation of the sediment and thereby impacting C cycling (Grace et al., 1997).

4.2b Carbon – wet-dry vs flooding

In contrast to the salinity treatments, water level treatments had profound effects on C cycles under oligohaline conditions. Our experiments demonstrated no significant increase in CO_2 emission but significant decreases in CH_4 and DOC productions when comparing flooding to wet-dry treatments ($p < 0.05$) (Figs 3 a and b; c and d). Although re-flooding led to the highest CO_2 emission rate from soils at each wet-dry cycle, there was no significant difference in total CO_2 -C emission between wet-dry and flooding treatments after 60-d incubations. Highest emissions of CO_2 were quantified immediately after rewetting. This results from an increase in C which most likely was caused by microbial biomass from cells death during the drying cycle as the decrease of soluble microbial byproduct-like from EEMs shown in Figs 6 a and b. Also, wet-dry treatments raised the exposure of organic residues (Denef et al., 2001a, 2001b) due to the grinding of soil structure for exchange of wetting and drying process (Fierer and Schimel, 2002; Lundquist et al., 1999). These findings have been reported in laboratory and field experiment in peatland soil (Chow et al., 2006) and forest soils (Jarvis et al., 2007). However, the wet-dry treatment did not affect total CO_2 -C loss in our study because of the offset of highest CO_2 emission rate at re-flooding and lowest CO_2 emission rate during the dry period. In sum, the wet-dry treatment resulted in 4.6-5.1% (CO_2 -C /TC) C loss and flooding resulted in 4.4-4.9% C loss.

The drying cycle in the wet-dry treatments probably led to oxic conditions, and consequently might have inhibited the formation of CH_4 (Olsson et al., 2015). Only 0.1-0.2% of soil C was converted to CH_4 -C in wet-dry treatments, and 1.0-1.2% in flooding treatments. Total C release from wet-dry treatments was 9.1-15.9% lower compared to flooding treatments. Our results are in line with previous results showing a decrease in CH_4 emissions in peatland soils and freshwater marsh when water table fluctuates (Moore and Knowles, 1989; Yang et al., 2013). Although wet-dry treatments did not affect total CO_2 -C loss, there was a decrease in CH_4 -C loss. Therefore, our data suggest that wet-dry treatments significantly decreased total C emissions ($\text{CO}_2 + \text{CH}_4$). These results are consistent with previous studies where fluctuating water tables reduced C emissions (Bass et al., 2014; Olsson et al., 2015).

Our study demonstrated that wet-dry treatments increased DOC production 1.1-1.6 fold after 60-d incubation. This was significantly lower compared to flooded treatments (1.7-1.9 fold) ($p < 0.05$). Researchers have shown that short heterotrophic microbial processes such as respiration and denitrification could lower DOC concentrations in wet-dry conditions more than that in flooding environments (Blodau and Moore, 2003; Burford and Bremner, 1975). Enhanced penetration of oxygen in wet-dry treatments resulted in an increase of both aerobic and anaerobic microbes (Fierer and Schimel, 2003; Lundquist et al., 1999). Also, the decrease in DOC concentration in wet-dry treatments could be attributed to altered microbial community structures during wet-drying process enhancing labile DOC consumption.

4.3 Nitrogen – salinity, water level (flooding and wet-dry)

In contrast to C biogeochemistry, significant changes were observed in N cycling in both saline and flooding treatments. An increase in salinity increased DN release but decreased N_2O emission from soils (Figs. 5 a, c). The increase in DN in water can be explained by ion exchange with NH_4^+ cations in soil with sodium cations (Na^+) in water (Wang and Sun, 2013). However, the increased availability of DN did not necessarily increase N_2O emission because high levels of salt can suppress both nitrification and denitrification (Osborne et al., 2015; Li et al., 2013). In addition to salinity, water level, which affects the availability of O_2 and redox status, could affect denitrification and N_2O production processes (Weitz et al., 2001; Lu et al., 2014). Continuous flooding limits the availability of oxygen in soils, reducing nitrification (Pezeshki and DeLaune, 2012). In contrast, wet–dry cycles allow O_2 penetration into the soil during dry periods, influencing microbial processes (i.e., substrate availability and microbial cell physiology) and soil physical properties, thus enhancing N_2O emissions (Stark and Firestone 1995; Burger et al., 2005; Kim et al., 2012) and increasing soil N losses (Borken and Matzner, 2009). Similar observations showing the effects of salt on N_2O and DN in wetlands have been reported previously (Azam and Ifzal, 2006; Chambers et al., 2013; Ardon et al., 2013).

4.4 Quality of DOM

SUVA is widely used as an indicator of aromatic C in soil and aquatic humic substances (Novak et al., 1992; Wang et al., 2015a; Yu et al., 2010). The E_2/E_3 ratio is inversely correlated with MW of DOM

(Hunt and Ohno, 2007; Wang et al., 2015b). In our study, not only did both, flooding and wet–dry cycle treatments increase DOC concentrations, they also increased the relative proportion of aromatic and humic-like compounds in soils (Fig.6), as indicated by a higher SUVA, FI and humic substance peak in region V in EEM and lower Freshness index (β/α) after 60-d incubation. In addition, a lower E_2/E_3 ratio in flooding treatments compared to wet-dry treatments after 60-d incubation suggests that DOM produced under flooding might mainly consist of compounds of higher molecular weight than that originating from wet-dry cycles. Such differences were attributed to the breakdown of large molecules by microorganisms under oxic conditions during the drying period (Chow et al., 2006; Krupa et al., 2012). Thus, increased MW, or recalcitrant DOM, limited microbial activities, resulting in the decrease of CO_2 and CH_4 .

Conclusions and Outlook

Coastal wetland soils represent a large reservoir for global C, but the stability of this important C pool is endangered by rising sea level, which will cause seawater intrusion and alter the wetland hydroperiod. Our study demonstrated that low salt concentrations (<5 ppt) and changes in water contents can significantly affect GHG formation and DOM optical properties in coastal wetland soils, confirming that saltwater intrusion and water level fluctuation in coastal wetlands, due to sea level rise, can impact C and N cycles in this ecosystem. Noticeably, the impacts from fluctuating water level on C cycles are greater than from salt intrusion in oligohaline areas. In the low-lying coastal areas of the southeastern US, large inland freshwater forested wetlands have been experiencing changing water levels or hydroperiods due to rising water tables caused by sea level rise. The changes in C and N cycles in these wetlands could be significant and might lead to changes in the global budgets of climate-relevant greenhouse gases such as CH_4 , CO_2 , and N_2O . In combination with the presented kinetic model it might be possible to predict how formation of greenhouse gases (CO_2 , CH_4) might be impacted by alterations in the carbon fractions in soil as a consequence of global climate change. However, in how far these results are also representative for other coastal wetlands or for processes in the environment is currently unknown. In addition, future studies and management efforts should also consider inland freshwater tidal wetlands, not just the salt-impacted areas, to evaluate how these environments might change as a consequence of sea level rise.

Acknowledgements

We gratefully acknowledge the Baruch Foundation for access to their land during the study. This research was partially supported by the NSF awards 1529927 and 1617040 and the U.S. Geological Survey, Climate and Land Use Change Research and Development Program, and was supported in part by the National Institute of Food and Agriculture, U.S. Department of Agriculture, under award numbers SC-1700489 and SC-1700424. Technical Contribution No. 6626 of the Clemson University Experiment Station. Dr. Liu was supported by the Chinese State Scholarship Fund from the China Scholarship Council (CSC).

References

- Allen, J.A., Pezeshki, S.R., Chambers, J.L., 1996. Interaction of flooding and salinity stress on baldcypress (*Taxodium distichum*). *Tree Physiology* 16, 307–313. doi:10.1093/treephys/16.1-2.307
- Ardón, M., Helton, A.M., Bernhardt, E.S., 2016. Drought and saltwater incursion synergistically reduce dissolved organic carbon export from coastal freshwater wetlands. *Biogeochemistry* 127, 411–426. doi:10.1007/s10533-016-0189-5
- Ardón, M., Morse, J.L., Colman, B.P., Bernhardt, E.S., 2013. Drought-induced saltwater incursion leads to increased wetland nitrogen export. *Glob. Chang. Biol.* 19, 2976–2985. doi:10.1111/gcb.12287
- Azam, F., Ifzal, M., 2006. Microbial populations immobilizing NH_4^+ -N and NO_3^- -N differ in their sensitivity to sodium chloride salinity in soil. *Soil Biology and Biochemistry* 38, 2491–2494. doi:10.1016/j.soilbio.2006.01.028
- Bai, J., Gao, H., Xiao, R., Wang, J., Huang, C., 2012. A review of soil nitrogen mineralization as affected by water and salt in coastal wetlands: issues and methods. *Clean - Soil, Air, Water* 40, 1099–1105. doi:10.1002/clen.201200055
- Baldwin, D.S., Rees, G.N., Mitchell, A.M., Watson, G., Williams, J., 2006. The short-term effects of salinization on anaerobic nutrient cycling and microbial community structure in sediment from a freshwater wetland. *Wetlands* 26, 455–464. doi:10.1672/0277-5212(2006)26[455:TSEOSO]2.0.CO;2
- Bass, A.M., O’Grady, D., Leblanc, M., Tweed, S., Nelson, P.N., Bird, M.I., 2014. Carbon dioxide and methane emissions from a wet-dry tropical floodplain in northern Australia. *Wetlands* 34, 619–627. doi:10.1007/s13157-014-0522-5

- 546 Blodau, C., Moore, T.R., 2003. Micro-scale CO₂ and CH₄ dynamics in a peat soil during a water fluctuation
 547 and sulfate pulse. *Soil Biology and Biochemistry* 35, 535–547.
 548 doi:10.1016/S0038-0717(03)00008-7
- 549 Borken, W., Matzner, E., 2009. Reappraisal of drying and wetting effects on C and N mineralization and
 550 fluxes in soils. *Global Change Biology* 15, 808–824. doi:10.1111/j.1365-2486.2008.01681.x
- 551 Bottner, P., 1985. Response of microbial biomass to alternate moist and dry conditions in a soil incubated
 552 with ¹⁴C- and ¹⁵N-labelled plant material. . *Soil Biology & Biochemistry* 17, 329–337.
 553 doi:10.1016/0038-0717(85)90070-7
- 554 Boyer, J.N., Groffman, P.M., 1996. Bioavailability of water extractable organic carbon fractions in forest
 555 and agricultural soil profiles. *Soil Biology & Biochemistry* 28, 783–790.
 556 doi:10.1016/0038-0717(96)00015-6
- 557 Breithaupt, J.L., Smoak, J.M., Smith, T.J., Sanders, C.J., Hoare, A., 2012. Organic carbon burial rates in
 558 mangrove sediments: Strengthening the global budget. *Global Biogeochemical Cycles* 26,156–169.
 559 doi:10.1029/2012GB004375
- 560 Burford, J.R., Bremner, J.M., 1975. Relationships between the denitrification capacities of soils and total,
 561 water-soluble and readily decomposable soil organic matter. *Soil Biology & Biochemistry* 7, 389–
 562 394. doi:10.1016/0038-0717(75)90055-3
- 563 Burger, M., Jackson, L.E., Lundquist, E.J., Louie, D.T., Miller, R.L., Rolston, D.E., Scow, K.M., 2005.
 564 Microbial responses and nitrous oxide emissions during wetting and drying of organically and
 565 conventionally managed soil under tomatoes. *Biology and Fertility of Soils* 42, 109–118.
 566 doi:10.1007/s00374-005-0007-z
- 567 Busbee, W.S., Conner, W.H., Allen, D.M., Lanham, J.D., 2003. Composition and aboveground
 568 productivity of three seasonally flooded depressional forested wetlands in coastal South Carolina.
 569 *Southeastern Naturalist* 2, 335–346. doi:10.1656/1528-7092(2003)002[0335:CAAPOT]2.0.CO;2
- 570 Cayuela, M.L., Sánchez-Monedero, M.A., Roig, A., Sinicco, T., Mondini, C., 2012. Biochemical changes
 571 and GHG emissions during composting of lignocellulosic residues with different N-rich by-products.
 572 *Chemosphere* 88, 196–203. doi:10.1016/j.chemosphere.2012.03.001

- 573 Chambers, L.G., Davis, S.E., Troxler, T., Boyer, J.N., Downey-Wall, A., Scinto, L.J., 2014.
 574 Biogeochemical effects of simulated sea level rise on carbon loss in an Everglades mangrove peat
 575 soil. *Hydrobiologia* 726, 195–211. doi:10.1007/s10750-013-1764-6
- 576 Chambers, L.G., Osborne, T.Z., Reddy, K.R., 2013. Effect of salinity-altering pulsing events on soil
 577 organic carbon loss along an intertidal wetland gradient: A laboratory experiment. *Biogeochemistry*
 578 115, 363–383. doi:10.1007/s10533-013-9841-5
- 579 Chambers, L.G., Reddy, K.R., Osborne, T.Z., 2011. Short-Term Response of Carbon Cycling to Salinity
 580 Pulses in a Freshwater Wetland. *Soil Science Society of America Journal* 75, 2000–2007.
 581 doi:10.2136/sssaj2011.0026
- 582 Chow, A.T., Dahlgren, R.A., Zhang, Q., Wong, P.K., 2008. Relationships between specific ultraviolet
 583 absorbance and trihalomethane precursors of different carbon sources. *Journal of Water Supply:
 584 Research and Technology - AQUA* 57, 471–480. doi:10.2166/aqua.2008.064
- 585 Chow, A.T., Dai, J., Conner, W.H., Hitchcock, D.R., Wang, J.J., 2013. Dissolved organic matter and
 586 nutrient dynamics of a coastal freshwater forested wetland in Winyah Bay, South Carolina.
 587 *Biogeochemistry* 112, 571–587. doi:10.1007/s10533-012-9750-z
- 588 Chow, A.T., Tanji, K.K., Gao, S., 2003. Production of dissolved organic carbon (DOC) and trihalomethane
 589 (THM) precursor from peat soils. *Water Research* 37, 4475–4485. doi:10.1016/S0043-
 590 1354(03)00437-8
- 591 Chow, A.T., Tanji, K.K., Gao, S., Dahlgren, R.A., 2006. Temperature, water content and wet-dry cycle
 592 effects on DOC production and carbon mineralization in agricultural peat soils. *Soil Biology &
 593 Biochemistry* 38, 477–488. doi:10.1016/j.soilbio.2005.06.005
- 594 Cook, B.D., Allan, D.L., 1992. Dissolved organic carbon in old field soils - Compositional changes during
 595 the biodegradation of soil organic matter. *Soil Biology & Biochemistry* 24, 595–600.
 596 doi:10.1016/0038-0717(92)90085-C
- 597 Cormier, N., Krauss, K.W., Conner, W.H., 2013. Periodicity in stem growth and litterfall in tidal freshwater
 598 forested wetlands: Influence of salinity and drought on nitrogen recycling. *Estuaries and Coasts* 36,
 599 533–546. doi:10.1007/s12237-012-9505-z

- 600 Cory, R.M., McKnight, D.M., 2005. Fluorescence spectroscopy reveals ubiquitous presence of oxidized
 601 and reduced quinones in dissolved organic matter. *Environmental Science and Technology* 39, 8142–
 602 8149. doi:10.1021/es0506962
- 603 Davidson, E.A., Janssens, I.A., 2006. Temperature sensitivity of soil carbon decomposition and feedbacks
 604 to climate change. *Nature* 440, 165–173. doi:10.1038/nature04514
- 605 Donato, D.C., Kauffman, J.B., Murdiyarso, D., Kurnianto, S., Stidham, M., Kanninen, M., 2011.
 606 Mangroves among the most carbon-rich forests in the tropics. *Nature Geoscience* 4, 293–297.
 607 doi:10.1038/ngeo1123
- 608 Fellman, J.B., Hood, E., Spencer, R.G.M., 2010. Fluorescence spectroscopy opens new windows into
 609 dissolved organic matter dynamics in freshwater ecosystems: A review. *Limnology and*
 610 *Oceanography* 55, 2452–2462. doi:10.4319/lo.2010.55.6.2452
- 611 Fierer, N., Allen, A.S., Schimel, J.P., Holden, P.A., 2003. Controls on microbial CO₂ production: A
 612 comparison of surface and subsurface soil horizons. *Global Change Biology* 9, 1322–1332.
 613 doi:10.1046/j.1365-2486.2003.00663.x
- 614 Fierer, N., Schimel, J.P., 2002. Effects of drying-rewetting frequency on soil carbon and nitrogen
 615 transformations. *Soil Biology and Biochemistry* 34, 777–787. doi:10.1016/S0038-0717(02)00007-X
- 616 Goni, M.A., Gardner, L.R., 2003. Seasonal dynamics in dissolved organic carbon concentrations in a
 617 coastal water-table aquifer at the forest-marsh interface. *Aquatic Geochemistry* 9, 209–232.
 618 doi:10.1023/B:AQUA.0000022955.82700.ed
- 619 Gorban, A.N., Yablonsky, G.S., 2015. Three Waves of Chemical Dynamics, *Mathematical Modelling of*
 620 *Natural Phenomena* 10, 1–5.
- 621 Grace, M.R., Hislop, T.M., Hart, B.T., Beckett, R., 1997. Effect of saline groundwater on the
 622 aggregation and settling of suspended particles in a turbid Australian river. *Colloids Surfaces*
 623 *A Physicochem. Eng. Asp.* 120, 123–141. doi:10.1016/S0927-7757(96)03863-0
- 624 Guha, H., Panday, S., 2012. Impact of sea level rise on groundwater salinity in a coastal community of
 625 South Florida. *Journal of the American Water Resources Association* 48, 510–529.
- 626 Guldberg, C.M., Waage, P., 1864. "Studies Concerning Affinity" C. M. Forhandlinger: Videnskabs-
 627 Selskabet i Christiana, 35.

- 628 Hayashi, M., 2004. Temperature-electrical conductivity relation of water for environmental monitoring and
 629 geophysical data inversion. *Environmental Monitoring and Assessment* 96, 119–128.
 630 doi:10.1023/B:EMAS.0000031719.83065.68
- 631 Helms, J.R., Stubbins, A., Ritchie, J.D., Minor, E.C., Kieber, D.J., Mopper, K., 2008. Absorption spectral
 632 slopes and slope ratios as indicators of molecular weight, source, and photobleaching of
 633 chromophoric dissolved organic matter. *Limnology and Oceanography* 53, 955–969.
 634 doi:10.4319/lo.2008.53.3.0955
- 635 Herbert, E.R., Boon, P., Burgin, A.J., Neubauer, S.C., Franklin, R.B., Ardón, M., Hopfensperger, K.N.,
 636 Lamers, L.P.M., Gell, P., 2015. A global perspective on wetland salinization: ecological
 637 consequences of a growing threat to freshwater wetlands. *Ecosphere* 6, art206.
 638 doi:10.1890/ES14-00534.1
- 639 Hoff, J. H. van't (Jacobus Henricus van't), Cohen, E., Thomas, E. (1896-01-01), 1884. *Studies in chemical*
 640 *dynamics*. Amsterdam: F. Muller; London: Williams & Norgate.
- 641 Hunt, J.F., Ohno, T., 2007. Characterization of fresh and decomposed dissolved organic matter using
 642 excitation-emission matrix fluorescence spectroscopy and multiway analysis. *Journal of Agricultural*
 643 *and Food Chemistry* 55, 2121–2128. doi:10.1021/jf063336m
- 644 IPCC, 2014. *Climate Change 2014. Synthesis Report Summary Chapter for Policymakers*. Ipcc.
- 645 Jarvis, P., Rey, A., Petsikos, C., Wingate, L., Rayment, M., Pereira, J., Banza, J., David, J., Miglietta, F.,
 646 Borghetti, M., Manca, G., Valentini, R., 2007. Drying and wetting of Mediterranean soils stimulates
 647 decomposition and carbon dioxide emission: the “Birch effect.” *Tree Physiology* 27, 929–940.
 648 doi:10.1093/treephys/27.7.929
- 649 Kaiser, K., Guggenberger, G., Zech, W., 1996. Sorption of DOM and DOM fractions to forest soils.
 650 *Geoderma* 74, 281–303. doi:10.1016/S0016-7061(96)00071-7
- 651 Kalbitz, K., Solinger, S., Park, J.-H., Michalzik, B., Matzner, E., 2000. Controls on the dynamics of
 652 dissolved organic matter: A review. *Soil Science* 165, 277–304.
 653 doi:10.1097/00010694-200004000-00001
- 654 Kim, D.-G., Vargas, R., Bond-Lamberty, B., Turetsky, M.R., 2012. Effects of soil rewetting and thawing
 655 on soil gas fluxes: A review of current literature and suggestions for future research. *Biogeosciences*
 656 9, 2459–2483. doi:10.5194/bg-9-2459-2012

- 657 Krauss, K.W., Whitbeck, J.L., 2012. Soil greenhouse gas fluxes during wetland forest retreat along the
658 lower Savannah River, Georgia (USA). *Wetlands* 32, 73–81. doi:10.1007/s13157-011-0246-8
- 659 Krupa, M., Spencer, R.G.M., Tate, K.W., Six, J., van Kessel, C., Linnquist, B.A., 2012. Controls on
660 dissolved organic carbon composition and export from rice-dominated systems. *Biogeochemistry*
661 108, 447–466. doi:10.1007/s10533-011-9610-2
- 662 Lal, R., 2004. Soil carbon sequestration impacts on global climate change and food security. *Science* (80-
663 .). 304, 1623–1627. doi:10.1126/science.1097396 [doi]
- 664 Lewis, D.B., Brown, J.A., Jimenez, K.L., 2014. Effects of flooding and warming on soil organic matter
665 mineralization in *Avicennia germinans* mangrove forests and *Juncus roemerianus* salt marshes.
666 *Estuarine, Coastal and Shelf Science* 139, 11–19. doi:10.1016/j.ecss.2013.12.032
- 667 Liu, W.C., Liu, H.M., 2014. Assessing the impacts of sea level rise on salinity intrusion and transport time
668 scales in a tidal estuary, Taiwan. *Water (Switzerland)* 6, 324–344. doi:10.3390/w6020324
- 669 Lundquist, E.J., Jackson, L.E., Scow, K.M., 1999. Wet-dry cycles affect dissolved organic carbon in two
670 California agricultural soils. *Soil Biology and Biochemistry* 31, 1031–1038.
671 doi:10.1016/S0038-0717(99)00017-6
- 672 Marton, J.M., Herbert, E.R., Craft, C.B., 2012. Effects of salinity on denitrification and greenhouse gas
673 production from laboratory-incubated tidal forest soils. *Wetlands* 32, 347–357.
674 doi:10.1007/s13157-012-0270-3
- 675 Mitsch, W.J., Gosselink, J.G., 1993. *Wetland*. John Wiley & Sons, Inc. New York.
- 676 Montaña, N.M., García-Oliva, F., Jaramillo, V.J., 2007. Dissolved organic carbon affects soil microbial
677 activity and nitrogen dynamics in a Mexican tropical deciduous forest. *Plant Soil* 295, 265–277.
678 doi:10.1007/s11104-007-9281-x
- 679 Moore, T.R., Dalva, M., 2001. Some controls on the release of dissolved organic carbon by plant tissues
680 and soils. *Soil Science* 166, 38–47. doi:10.1097/00010694-200101000-00007
- 681 Moore, T.R., Knowles, R., 1989. The influence of water table levels on methane and carbon dioxide
682 emissions from peatland soils. *Canadian Journal of Soil Science* 69, 33–38. doi:10.4141/cjss89-004
- 683 Morillas, L., Durán, J., Rodríguez, A., Roales, J., Gallardo, A., Lovett, G.M., Groffman, P.M., 2015.
684 Nitrogen supply modulates the effect of changes in drying-rewetting frequency on soil C and N

- 685 cycling and greenhouse gas exchange. *Global Change Biology* 21, 3854–3863.
 686 doi:10.1111/gcb.12956
- 687 Moseman-Valtierra, S., Gonzalez, R., Kroeger, K.D., Tang, J., Chao, W.C., Crusius, J., Bratton, J., Green,
 688 A., Shelton, J., 2011. Short-term nitrogen additions can shift a coastal wetland from a sink to a source
 689 of N₂O. *Atmospheric Environment* 45, 4390–4397. doi:10.1016/j.atmosenv.2011.05.046
- 690 Mosier, A.R., 1994. Nitrous oxide emissions from agricultural soils. *Fertilizer Research* 37, 191–200.
 691 doi:10.1007/BF00748937
- 692 Moyano, F.E., Manzoni, S., Chenu, C., 2013. Responses of soil heterotrophic respiration to moisture
 693 availability: An exploration of processes and models. *Soil Biology & Biochemistry* 59, 72–85.
 694 doi:10.1016/j.soilbio.2013.01.002
- 695 Mulholland, P.J., 1981. Organic carbon flow in a swamp-stream ecosystem. *Ecol. Monogr.* 51,
 696 307–322. doi:10.2307/2937276
- 697 Novak, J.M., Mills, G.L., Bertsch, P.M., 1992. Estimating the percent aromatic carbon in soil and aquatic
 698 humic substances using ultraviolet absorbency spectrometry. *Journal of Environmental Quality* 21,
 699 144–147.
- 700 Ohno, T., 2002. Fluorescence inner-filtering correction for determining the humification index of dissolved
 701 organic matter. *Environmental Science and Technology* 36, 742–746. doi:10.1021/es0155276
- 702 Olsson, L., Ye, S., Yu, X., Wei, M., Krauss, K.W., Brix, H., 2015. Factors influencing CO₂ and CH₄
 703 emissions from coastal wetlands in the Liaohe Delta, Northeast China. *Biogeosciences* 12, 4965–
 704 4977. doi:10.5194/bg-12-4965-2015
- 705 Opio, A., Jones, M.B., Kansime, F., Otiti, T., 2015. Dissolved organic carbon in a tropical wetland
 706 dominated by *Cyperus papyrus*. *Wetlands Ecology and Management* 23, 1033–1038.
 707 doi:10.1007/s11273-015-9437-z
- 708 Pathak, H., Rao, D.L.N., 1998. Carbon and nitrogen mineralization from added organic matter in saline
 709 and alkali soils. *Soil Biol. Biochem.* 30, 695–702.
 710 doi:http://dx.doi.org/10.1016/S0038-0717(97)00208-3
- 711 Pezeshki, S.R., DeLaune, R.D., 2012. Soil Oxidation-Reduction in wetlands and its impact on plant
 712 functioning. *Biology* 1, 196–221. doi:10.3390/biology1020196
- 713 Rahmstorf, S., 2007. A semi empirical approach to projecting future sea-level rise. *Science* 315, 368–370.

doi:10.1126/science.1135456

Renaud, F.G., Le, T.T.H., Lindener, C., Guong, V.T., Sebesvari, Z., 2015. Resilience and shifts in agro-ecosystems facing increasing sea-level rise and salinity intrusion in Ben Tre Province, Mekong Delta.

Climatic Change 133, 69–84. doi:10.1007/s10584-014-1113-4

Schofield, R. V., 2003. Application of salinization indicators and initial development of potential global soil salinization scenario under climatic change. Global Biogeochem. Cycles 17, 1–13.

doi:10.1029/2002GB001935

Sholkovitz, E.R., 1976. Flocculation of dissolved organic and inorganic matter during the mixing of river water and seawater. Geochim. Cosmochim. Acta 40, 831–845. doi:10.1016/0016-7037(76)90035-1

Setia, R., Gottschalk, P., Smith, P., Marschner, P., Baldock, J., Setia, D., Smith, J., 2013. Soil salinity decreases global soil organic carbon stocks. The Science of the Total Environment 465, 267–72.

doi:10.1016/j.scitotenv.2012.08.028

Shi, A., Marschner, P., 2014. Changes in microbial biomass C, extractable C and available N during the early stages of decomposition of residue mixtures. Soil Research 52, 366–372. doi:10.1071/SR13128

Silva, C.C., Guido, M.L., Ceballos, J.M., Marsch, R., Dendooven, L., 2008. Production of carbon dioxide and nitrous oxide in alkaline saline soil of Texcoco at different water contents amended with urea: A

laboratory study. Soil Biology and Biochemistry 40, 1813–1822. doi:10.1016/j.soilbio.2008.03.004

Stuckey, B.N. 1982. Soil survey of Georgetown County South Carolina. Rep. Prepared for the Soil Conservation Service, USDA, Washington, DC.

Wang, J.J., Dahlgren, R.A., Chow, A.T., 2015a. Controlled burning of forest detritus altering spectroscopic characteristics and chlorine reactivity of dissolved organic matter: Effects of temperature and oxygen

availability. Environmental Science and Technology 49, 14019–14027. doi:10.1021/acs.est.5b03961

Wang, J.J., Dahlgren, R.A., Erşan, M.S., Karanfil, T., Chow, A.T., 2015b. Wildfire altering terrestrial precursors of disinfection byproducts in forest detritus. Environmental Science and Technology 49,

5921–5929. doi:10.1021/es505836m

Wang, L., Sun, H., 2013. Prediction of $\text{Na}^+/\text{NH}_4^+$ exchange in faujasite zeolite by molecular dynamics simulation and thermodynamic integration method. The Journal of Physical Chemistry C 117, 14051–

14060. doi:10.1021/jp403326n

- 742 Wang, Y.L., Yang, C.M., Zou, L.M., Cui, H.Z., 2015. Spatial distribution and fluorescence properties of
 743 soil dissolved organic carbon across a riparian buffer wetland in Chongming Island, China.
 744 *Pedosphere* 25, 220–229. doi:10.1016/S1002-0160(15)60007-8
- 745 Weier, K.L., Doran, J.W., Power, J.F., Walters, D.T., 1993. Denitrification and the dinitrogen/nitrous oxide
 746 ratio as affected by soil water, available carbon, and nitrate. *Soil Science Society of America Journal*
 747 57, 66–72. doi:10.2136/sssaj1993.03615995005700010013x
- 748 Weitz, A., Linder, E., Frolking, S., Crill, P., Keller, M., 2001. N₂O emissions from humid tropical
 749 agricultural soils: Effects of soil moisture, texture and nitrogen availability. *Soil Biology &*
 750 *Biochemistry* 33, 1077–1093. doi:10.1016/S0038-0717(01)00013-X
- 751 Weston, N.B., Vile, M.A., Neubauer, S.C., Velinsky, D.J., 2011. Accelerated microbial organic matter
 752 mineralization following salt-water intrusion into tidal freshwater marsh soils. *Biogeochemistry* 102,
 753 135–151. doi:10.1007/s10533-010-9427-4
- 754 Woodruff, J.D., Irish, J.L., Camargo, S.J., 2013. Coastal flooding by tropical cyclones and sea-level rise.
 755 *Nature* 504, 44–52. doi:10.1038/nature12855
- 756 Wu, C.Y., Liu, J.K., Cheng, S.H., Surampalli, D.E., Chen, C.W., Kao, C.M., 2010. Constructed wetland
 757 for water quality improvement: A case study from Taiwan. *Water Science and Technology* 62, 2408–
 758 2418. doi:10.2166/wst.2010.492
- 759 Xenopoulos, H.F.W. & M.A., 2009. Effects of agricultural land use on the composition of fluvial dissolved
 760 organic matter. *Nature Geoscience*.
- 761 Yang, J., Liu, J., Hu, X., Li, X., Wang, Y., Li, H., 2013. Effect of water table level on CO₂, CH₄ and N₂O
 762 emissions in a freshwater marsh of Northeast China. *Soil Biology & Biochemistry* 61, 52–60.
 763 doi:10.1016/j.soilbio.2013.02.009
- 764 Yu, H., Xi, B., Su, J., Ma, W., Wei, Z., He, X., Guo, X., 2010. Spectroscopic properties of dissolved fulvic
 765 acids: An indicator for soil salinization in arid and semiarid regions in China. *Soil Science* 175, 240–
 766 245. doi:10.1097/SS.0b013e3181e055b4
- 767 Zemke, K., Liebscher, A., Wandrey, M., 2010. Petrophysical analysis to investigate the effects of carbon
 768 dioxide storage in a subsurface saline aquifer at Ketzin, Germany (CO₂SINK). *International Journal*
 769 *of Greenhouse Gas Control* 4, 990–999. doi:10.1016/j.ijggc.2010.04.008

- Zhu, B., Cheng, W., 2013. Impacts of drying-wetting cycles on rhizosphere respiration and soil organic matter decomposition. *Soil Biology and Biochemistry* 63, 89–96. doi:10.1016/j.soilbio.2013.03.027
- Zsolnay, Á., 2003. Dissolved organic matter: Artefacts, definitions, and functions. *Geoderma* 113, 187–209. doi:10.1016/S0016-7061(02)00361-0

Figure Captions

Fig. 1 The relationship between (a) DOC-CO₂ and (b) DOC-CH₄ in surface forested wetland soil under permanent flooding.

Fig. 2 Formation rates of CO₂ (a-c), CH₄ (d-f), and N₂O (g-i) in surface forested wetland soil incubated at 0 ppt, 1 ppt, and 5 ppt sodium chloride under flooding with θ_g of 3.0 g-water g-soil⁻¹ and wet-dry cycles with θ_g of 0.4~3.0 g-water g-soil⁻¹. The small diagram in the upper right hand corner of each figure is the cumulative emission over 60 days. Error bars represent the standard deviations of triplicate measurements. The vertical lines in each diagram indicate the 12-day wet-dry treatments. a-c variations are rate and total of CO₂-C emission among salinity, d-f variations are rate and total of CH₄-C emission among salinity; g-i are rate and total of N₂O-N emission among salinity during 60-d incubation

Fig.3 Variations in DOC (a-c), DN (d-f) and DOC/DN (g-i) at 0, 1, and 5 ppt NaCl in surface forested wetland soil under flooding and wet-dry treatments during 60-d incubation. Error bar represents the standard deviation from triplicate measurements

Fig. 4 Variations in SUVA₂₅₄ (a-c), E₂/E₃ ratio (d-f) and S_R (g-i) in surface soils incubated at various salinities (0, 1, and 5 ppt NaCl) under flooding and wet-dry conditions over 60 days. Error bar represents the standard deviation from triplicate measurements

Fig. 5 A conceptual model for describing effect of water level and salinity on freshwater forested wetland soils. a and b are soil incubation with freshwater (0 ppt) experiencing flooding and wet-dry treatments, respectively; c and d are soil incubation with salt water (5 ppt) experiencing flooding and wet-dry treatments, respectively

Fig. 6 The quantity, quality and 3D fluorescence excitation emission matrices of DOC in forested wetland soil with salinity under flooding (A) and wet-dry (B) during 60-d incubation. The size of the pie chart represents the quantity of DOC, and the quality of DOC shows in each pie chart. A and B represent flooding and wet-dry treatment, respectively. Typical EEM figures are shown in Fig.6 C. Fluorescence regional integration can be used to quantify the fluorescent DOM by dividing EEM into five operationally defined regions: I) tyrosine-like; II) tryptophan-like; III) fulvic acid-like; IV) soluble microbial byproduct-like; and V), humic acid-like

Fig. 1 The relationship between (a) DOC- CO_2 and (b) DOC- CH_4 in surface forested wetland soil under permanent flooding.

Fig. 2 Formation rates of CO₂ (a-c), CH₄ (d-f), and N₂O (g-i) in surface forested wetland soil incubated at 0 ppt, 1 ppt, and 5 ppt sodium chloride under flooding with θ_g of 3.0 g-water g-soil⁻¹ and wet-dry cycles with θ_g of 0.4~3.0 g-water g-soil⁻¹. The small diagram in the upper right hand corner of each figure is the cumulative emission over 60 days. Error bars represent the standard deviations of triplicate measurements. The vertical lines in each diagram indicate the 12-day wet-dry treatments. a-c variations are rate and total of CO₂-C emission among salinity, d-f variations are rate and total of CH₄-C emission among salinity; g-i are rate and total of N₂O-N emission among salinity during 60-d incubation

Fig.3 Variations in DOC (a-c), DN (d-f) and DOC/DN g-i) at 0, 1, and 5 ppt NaCl in surface forested wetland soil under flooding and wet-dry treatments during 60-d incubation. Error bar represents the standard deviation from triplicate measurements

Fig. 4 Variations in $SUVA_{254}$ (a-c). E_2/E_3 ratio (d-f) and S_R (g-i) in surface soils incubated at various salinities (0, 1, and 5 ppt NaCl) under flooding and wet-dry conditions over 60 days. Error bar represents the standard deviation from triplicate measurements

Fig. 5 A conceptual model for describing effect of water level and salinity on freshwater forested wetlands. a and b are soil incubation with freshwater (0 ppt) experiencing flooding and wet-dry treatments, respectively; c and d are soil incubation with degraded oligohaline (5 ppt) experiencing flooding and wet-dry treatments, respectively.

Fig. 6 The quantity, quality and 3D fluorescence excitation emission matrices of DOM in forested wetland soil with salinity under flooding (A) and wet-dry (B) during 60-d incubation. The size of the pie chart represents the quantity of DOM, and the quality of DOM shows in each pie chart. A and B represent flooding and wet-dry treatment, respectively. Typical EEM figures are shown in Fig.6 C. Fluorescence regional integration can be used to quantify the fluorescent DOM by dividing EEM into five operationally defined regions: I) tyrosine-like; II) tryptophan-like; III) fulvic acid-like; IV) soluble microbial byproduct-like; and V), humic acid-like.

Tables**Table 1. Soil properties of surface forested wetland soils (mean $\pm$ standard deviation, n=3)**

Parameters	Original soil
pH	5.0 $\pm$ 0.1
Electrical conductivity (EC ₂₅) (mS cm ⁻¹)	7.6 $\pm$ 3.1
Dissolved organic carbon (mg g-soil ⁻¹)	1.2 $\pm$ 0.0
Total dissolved nitrogen (mg g-soil ⁻¹)	0.2 $\pm$ 0.0
Total carbon (TC) (mg g-soil ⁻¹)	240.9 $\pm$ 4.0
Total nitrogen (TN) (mg g-soil ⁻¹)	17.3 $\pm$ 1.0
C:N ratio	13.9 $\pm$ 0.3

Table 2. Total emissions of carbon dioxide, methane and nitrous oxide with salinity and soil water content during 60-d incubation. (mean $\pm$ standard deviation, n=3)

Water level	Salinity	CO ₂ -C loss (mg-C g-soil ⁻¹)	CH ₄ -C loss (mg-C g-soil ⁻¹)	Total loss (CO ₂ -C + CH ₄ -C) (mg-C g-soil ⁻¹)	N ₂ O-N loss (μ g-N g-soil ⁻¹)
flooding	0ppt	12.2 $\pm$ 0.5 ^a	2.9 $\pm$ 0.2 ^a	15.1 $\pm$ 0.4 ^a	0.10 $\pm$ 0.04 ^{ad}
	1ppt	12.4 $\pm$ 0.4 ^a	2.7 $\pm$ 0.1 ^a	15.2 $\pm$ 0.3 ^a	0.08 $\pm$ 0.03 ^{ab}
	5ppt	12.2 $\pm$ 0.5 ^a	2.6 $\pm$ 0.2 ^a	14.8 $\pm$ 0.7 ^a	0.02 $\pm$ 0.01 ^b
wet-dry	0ppt	11.7 $\pm$ 0.5 ^a	0.5 $\pm$ 0.1 ^b	12.1 $\pm$ 0.6 ^b	0.22 $\pm$ 0.03 ^c
	1ppt	12.8 $\pm$ 0.7 ^a	0.6 $\pm$ 0.1 ^b	13.4 $\pm$ 0.5 ^b	0.17 $\pm$ 0.01 ^{cd}
	5ppt	11.6 $\pm$ 0.5 ^a	0.2 $\pm$ 0.1 ^b	11.8 $\pm$ 0.5 ^b	0.15 $\pm$ 0.02 ^{ac}

Different letters represent significantly different means ($p < 0.05$) based on one-way ANOVA.

Supplementary Information

Box

CO₂-C:

[1] $d[CO_2-C]/dt=k_{SOC}[SOC]+k_{DOC}[DOC]$

[2] $d[CO_2-C]/dt=k_{SOC}([TAOC]-[DOC])+k_{DOC}[DOC]$

[3] $d[CO_2-C]/dt=(k_{DOC}-k_{SOC})[DOC]+k_{SOC}[TAOC]$

[4] $d[CO_2-C]/dt=k_{app}[DOC]+k_{SOC}[TAOC]$

SOC: Soil organic carbon (μg-C g-soil⁻¹)

DOC: Dissolved organic carbon (μg-C g-soil⁻¹)

TAOC: Total available organic carbon (μg-C g-soil⁻¹)

TAOC = SOC + DOC

k_{SOC}: reaction rate constant for C mineralization (d⁻¹)
utilizing SOC to produce CO₂-C

k_{DOC}: reaction rate constant for C mineralization (d⁻¹)
utilizing DOC to produce CO₂-C

k_{SD}: reaction rate constant for C mineralization (d⁻¹)
utilizing SOC to produce DOC

k_{app}: apparent rate constant for overall C (d⁻¹)
transfer to CO₂-C

Fig.s1 A conceptual model describing CO₂ and CH₄ production in freshwater forested wetland soils

931 **Table s1 Salinity and soil water content used in the incubation experiments**

Soil used	Treatment	Incubation conditions	
		Variables	Constant parameters
Surface soil	Salinity effect	$S=0, 1, 5$	$\theta_g=3.0$
	Wet-dry cycles	$\theta_g=0.4-3.0$	$S=0, 1, 5$

932 *θ_g is soil water content in g-water g-soil⁻¹ and S is salinity in ppt (1 part in one trillion parts of water*
 933 *solution).*

934

935

936

937

Table s2. The pH and EC (mean $\pm$ standard deviation; n=3) of forested wetland soil during 60-d incubation

Parameter s	water level	salinity (NaCl)	incubation(day)					
			0	12	24	36	48	60
pH	flooding	0ppt	5.0 $\pm$ 0.1 ^a	6.4 $\pm$ 0.4 ^{ad}	7.9 $\pm$ 0.2 ^a	7.4 $\pm$ 0.1 ^a	7.3 $\pm$ 0.1 ^a	7.5 $\pm$ 0.0 ^a
		1ppt	4.8 $\pm$ 0.1 ^a	5.9 $\pm$ 0.1 ^{ac}	7.6 $\pm$ 0.0 ^a	7.2 $\pm$ 0.0 ^a	7.3 $\pm$ 0.0 ^a	7.2 $\pm$ 0.6 ^a
		5ppt	4.8 $\pm$ 0.1 ^a	6.2 $\pm$ 0.3 ^{ad} _c	6.6 $\pm$ 0.2 ^b	6.2 $\pm$ 0.2 ^b	5.7 $\pm$ 0.1 ^b	6.2 $\pm$ 0.4 ^b
	wet-dry	0ppt	5.0 $\pm$ 0.1 ^a	6.3 $\pm$ 0.2 ^{ad} _c	6.5 $\pm$ 0.3 ^b	6.2 $\pm$ 0.1 ^b	5.9 $\pm$ 0.0 ^b	6.1 $\pm$ 0.1 ^b
		1ppt	4.8 $\pm$ 0.1 ^a	6.3 $\pm$ 0.2 ^{ad}	6.7 $\pm$ 0.1 ^b	6.2 $\pm$ 0.3 ^b	5.2 $\pm$ 0.7 ^b	5.9 $\pm$ 0.1 ^b
		5ppt	4.8 $\pm$ 0.1 ^a	5.7 $\pm$ 0.1 ^c	6.4 $\pm$ 0.2 ^b	6.1 $\pm$ 0.1 ^b	5.5 $\pm$ 0.2 ^b	5.9 $\pm$ 0.1 ^b
EC _{25°C} (mS/cm)	flooding	0ppt	4.6 $\pm$ 3.1 ^a	4.1 $\pm$ 0.1 ^{ac}	4.0 $\pm$ 0.4 ^a	3.5 $\pm$ 1.0 ^a	3.6 $\pm$ 0.3 ^a	3.9 $\pm$ 0.0 ^a
		1ppt	6.4 $\pm$ 3.1 ^a	5.6 $\pm$ 0.2 ^a	5.1 $\pm$ 0.0 ^a	4.8 $\pm$ 0.1 ^{ac}	5.7 $\pm$ 0.0 ^{ac}	6.0 $\pm$ 0.1 ^a
		5ppt	13.5 $\pm$ 0.1 _b	11.5 $\pm$ 0.2 _b	11.4 $\pm$ 0.4 _b	12.7 $\pm$ 0.1 _b	14.0 $\pm$ 0.0 _b	14.1 $\pm$ 0.1 _b
	wet-dry	0ppt	4.6 $\pm$ 0.3 ^a	2.9 $\pm$ 0.6 ^c	3.1 $\pm$ 0.4 ^a	3.4 $\pm$ 0.2 ^a	3.8 $\pm$ 0.1 ^{ac}	4.0 $\pm$ 0.1 ^a
		1ppt	6.4 $\pm$ 0.3 ^a	4.2 $\pm$ 0.7 ^{ac}	4.3 $\pm$ 0.7 ^a	5.2 $\pm$ 2.4 ^{ac}	6.0 $\pm$ 0.0 ^c	6.1 $\pm$ 0.1 ^a
		5ppt	13.5 $\pm$ 0.1 _b	10.7 $\pm$ 0.7 _b	10.5 $\pm$ 0.7 _b	9.1 $\pm$ 2.4 ^d	13.4 $\pm$ 0.4 _b	13.8 $\pm$ 0.1 _b

Different letters represent significantly different means ($p < 0.05$) based on a one-way ANOVA.

942 **Table s3 Fluorescence Index and Freshness Index of DOC during 60-d incubation**

Parameters	Water level	Salinity	Incubation(day)					
			0	12	24	36	48	60
Fluorescence Index	Flooding	0ppt	1.51	1.44	1.46	1.45	1.44	1.46
		1pp	1.50	1.47	1.45	1.46	1.47	1.45
		5ppt	1.55	1.45	1.49	1.47	1.51	1.51
	Wet-dry	0ppt	1.51	1.47	1.47	1.51	1.47	1.47
		1ppt	1.50	1.47	1.49	1.5	1.49	1.48
		5ppt	1.55	1.52	1.52	1.5	1.53	1.48
Freshness Index	Flooding	0ppt	0.56	0.55	0.52	0.51	0.52	0.50
		1pp	0.57	0.56	0.53	0.53	0.52	0.53
		5ppt	0.58	0.58	0.57	0.56	0.57	0.57
	Wet-dry	0ppt	0.56	0.56	0.6	0.61	0.63	0.63
		1ppt	0.57	0.56	0.61	0.61	0.63	0.65
		5ppt	0.58	0.60	0.66	0.64	0.64	0.66

943