

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: http://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 awesole@clemson.edu.

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

3

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

6

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

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

11 ^c Department of Computer, Anhui Vocational College of Grain Engineering, Hefei, Anhui Province,
12 230011, China

13 ^d Department of Environmental Engineering and Earth Sciences, Clemson University, South Carolina
14 29631, United States

15

16

17 *Corresponding Author: Tel.: + 8435461013-232

18 Email address: achow@clemson.edu.

19

20

21

22

23

24

25

26

27

28 **Abstract**

29 Forested wetlands dominated by baldcypress (*Taxodium distichum*) and water tupelo (*Nyssa aquatica*) are
30 commonly found in coastal regions of the southeastern United States. Global climate change and in
31 particular sea level rise will alter the frequency and magnitude of wet/dry periods and salinity levels in
32 these ecosystems. Soil microcosm experiments were set up to identify the effects of water level variations
33 (0.4-3.0 g-water g-soil⁻¹) and salinity changes (0, 1 and 5 ppt of NaCl) on greenhouse gas emissions (CH₄,
34 CO₂ and N₂O) and dissolved organic carbon (DOC) characteristics from forested wetland soils. Our results
35 indicate that, the effect of water level was much greater than salt intrusion on C and N cycling.
36 Wet-dry treatments significantly decreased DOC production and total CH₄-C loss, aromatic and
37 humic-like substance compounds in DOC were increased in both flooding and wet-dry treatments
38 after 60-d incubation. The molecular weight (MW) of DOC after flooding treatments was higher
39 than that in wet-dry treatments. A first order kinetic model showed there was a positive linear
40 correlation ($r^2=0.73$) between CO₂ emission rate and DOC concentration which indicated that CO₂
41 was mainly generated from DOC. An exponential kinetic model was applied to describe the
42 correlation between CH₄ emission rate and DOC concentration ($r^2=0.41$). This study demonstrates
43 that an increase in salinity, and in particular variations in wet-dry cycles, will lead to changes in the
44 formation of climate-relevant greenhouse gases, such as CH₄, CO₂, and N₂O.

45

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

47

48

49

50

51

52

53

54

55 **Introduction**

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

69 Previous studies have demonstrated that decomposition rate of DOC (Fierer and Schimel, 2002) and
70 the quantity, as well as, the chemical characteristics of DOC are affected by water level fluctuations and
71 wet–dry cycles (Lundquist et al., 1999; Kalbitz et al., 2000; Chow et al., 2003). Positive correlations
72 between C mineralization rate and DOC concentration were found in coastal wetlands (Cook and Allan,
73 1992; Chow et al., 2006). The impact on greenhouse gas formation however, remains poorly understood.
74 While Shi and Marschner (2014) reported an increase in GHG formation as a consequence of varying water
75 levels, Morillas et al. (2015) reported a decline in GHG formation based on a lower microbial biomass and
76 microbial activity.

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

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

94 As the changes in C turnover occur, it can be expected that the turnover of N, and in particular the
95 formation of the greenhouse gas nitrous oxide (N_2O) will be affected, too. Previously, it has been
96 demonstrated that emission of N_2O is the primary pathway of N loss from wetland soils into the atmosphere,
97 and that variations in water levels and conductivity affect N_2O emissions from an alkaline soil (Silva et al.,
98 2008; Kraus and Whitbeck, 2012). The impacts of different salinities in combination with varying water
99 levels on N_2O formation has, to the best of our knowledge, not yet been investigated. The above mentioned
100 aspects illustrate the necessity of a thorough quantification of greenhouse gas emissions under varying
101 water levels and salinities to gain a better understanding of potential alterations in atmospheric levels of
102 these climate-relevant gases. In particular, the contribution of different C and N fractions (DOC, soil
103 organic carbon (SOC), total nitrogen (TN), and soil nitrogen (SN) as precursors for greenhouse gas
104 formation is also of interest. Additionally, the identification of DOM optical properties is essential as it
105 serves as a C source for microorganisms in the soils and is crucial for biogeochemical turnover of C and
106 N.

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

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

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

121

122 **2. Materials and experimental methods**

123 **2.1. Field site and sampling procedure**

124 The study was conducted with soil samples from a healthy forested wetland on Hobcaw Barony (33°21'N,
125 79°12'W), near Georgetown, South Carolina, which is currently not affected by saltwater intrusions. The
126 highly productive (average annual litterfall input of 620 g m⁻² yr⁻¹; Conner, unpublished), seasonally
127 flooded 24-ha (2.4×10⁵ m²) wetland is dominated by baldcypress (*Taxodium distichum* (L.) Rich.), water
128 tupelo (*Nyssa aquatica* L.), and swamp tupelo (*Nyssa biflora* Walt.) trees. The surrounding forest
129 community included longleaf pine (*Pinus palustris* Mill.) and turkey oak (*Quercus laevis* Walt.) (Busbee
130 et al., 2003). The predominant soil type in this wetland has been characterized as a fine-loamy, siliceous,
131 thermic, and Yypic Ultisol (USDA Soil Taxonomy) (Stuckey, 1982). Detailed characterizations of this site,
132 including tree composition, stem density, aboveground productivity, litterfall, nutrient dynamics, and
133 hydrology have been published previously (Busbee et al., 2003; Chow et al., 2013). Surface soil samples
134 were collected from 0–10 cm after removal of litter from the surface during the wetland's dry period in the
135 summer. The soils were transported to the laboratory, air-dried at room temperature (22 ± 1°C) until
136 constant weight, sieved to a particle size < 2 mm for homogenization, and stored at room temperature until
137 further analysis and start of the soil incubation experiments. This homogenization might have impacted the
138 microbial activity or microbial community. Therefore, results from the laboratory microcosm incubations
139 presented here are not directly transferable to results that can be expected from the field. Relevant soil
140 characteristics are summarized in Table 1.

141

142 2.2. Setup of microcosm experiments and sampling procedure

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

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

170

171 2.3. Analysis

172 Concentrations of C and N in the original soils (0 day) and after 60-d incubation (oven dried at 70 °C for
173 48 h) were quantified using a Thermo Flash EA 2000 element analyzer. CO₂, CH₄, and N₂O were analyzed
174 by a Shimadzu Greenhouse Gas analyzer (GC 2014). All filtered solution samples were analyzed for pH,
175 EC, DOC, and DN. The pH and EC of filtered solution samples were determined using an Accumet XL60
176 dual channel pH/ion/conductivity meter. EC was corrected to EC at 25 °C (EC₂₅) using the temperature
177 correction equation: $EC_{25} = EC_t / [1 + 0.021 (t-25)]$ (Hayashi, 2004). The DOC and DN were measured
178 using a Shimadzu TOC/TN analyzer (SM 5310B).

179 DOM optical properties were further characterized by UV-VIS spectrophotometry (Shimadzu UV-
180 1800) and spectrofluorometry (Shimadzu Spectrofluorometer RF5301). All DOC samples also were
181 analyzed for Specific ultraviolet absorbance (SUVA), and one of the 3 replicates was used for
182 spectrofluorometry. Specific ultraviolet absorbance, spectral slope ratio (S_R), and E₂/E₃ ratio were
183 determined as described previously (Chow et al., 2008; Helms et al., 2008; Wang et al., 2015). Based on
184 SUVA and E₂/E₃ ratio, selected samples were analyzed using fluorescence emission-excitation matrix
185 (EEM) (Zhou et al., 2013). Several spectrofluoroscopic indices including the fluorescence index (FI), the
186 freshness index (β/α), and humification index (HIX) were calculated as shown previously (Fellman et al.,
187 2010; Wang et al., 2015; Cory and McKnight, 2005).

188

189 2.4. Calculation of gas fluxes and statistical analysis

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

200

201 **2.5 Development of kinetic model**

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

215

216 One of the reasons for that is, that variations in water content most likely affect DOC availability and its
 217 transport processes in soil which cannot be described by the proposed reaction kinetic model. In the kinetic
 218 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],
 219 where the constants k_{SOC} and k_{DOC} are the reaction rates of SOC and DOC in forming CO₂. The first-order
 220 kinetics we used to describe the reaction among d[CO₂-C]/dt, and SOC, DOC was:

$$221 \quad d[CO_2 - C]/dt = k_{SOC}[SOC] + k_{DOC}[DOC]$$

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

$$226 \quad d[CH_4 - C]/dt = K_{SOC} e^{K_{DOC}[DOC]}$$

227 **3. Results**228 **3.1 Kinetic models for C and N cycles**

229 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)
 230 after 60-d incubation exceeded initial DOC concentrations (1.0 – 1.2 mg-C g-soil⁻¹) for all treatments
 231 suggesting both SOC and DOC could be substrates for CO₂ and CH₄ production (Boyer and Groffman,

232 1996). However, SOC was considered a relatively large C pool and the change in SOC content within the
233 60-d incubation should be minimal in comparison to the C changes in gas and water phases. In order to
234 delineate the relationships of the dynamic variables, CO₂-C or CH₄-C was plotted against DOC
235 concentration, as shown in Figure 1a and 1b, respectively. A linear correlation with ($R^2 = 0.73$) between
236 CO₂-C and DOC was observed (Fig. 1a), whereas an exponential correlation ($R^2 = 0.41$) between CH₄-C
237 and DOC was obtained (Fig. 1-b). DOC concentration is the same between CO₂ and CH₄ during the
238 incubation, but there is no CH₄ emitted at the beginning of incubation, so we deleted the high DOC with
239 no CH₄ emission in figure 1-b. We applied a kinetic model to demonstrate the contribution of different
240 carbon fractions and concentrations on formation rates of CO₂ (Chow et al., 2006). Wet-dry treatments are
241 not modeled by this approach because the variation in water content could change DOC availability as well
242 as its transport processes, which cannot be simply described by the reaction kinetic model. In the kinetic
243 model, total available organic carbon (TAOC) is the sum of degradable SOC and DOC, which are available
244 and accessible to microbes during the 60-d incubation. TAOC was not necessarily equal to total organic C
245 in soils because not all of the C would be involved in the reactions occurring during the 60-d period. In
246 addition, DOC could be produced at an independent reaction rate constant k_{SD} by microbes utilizing SOC
247 as C source. The reaction rate constant k_{SD} is not equal to k_{SOC} because the mechanism producing CO₂ is
248 probably different from that producing DOC (Moore and Dalva, 2001; Chow et al., 2006).

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

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

262 In contrast to CO_2 emission, the linear equation did not fit well when CH_4 emission rate ($d[\text{CH}_4\text{-C}]/dt$)
 263 was plotted against DOC concentration. Instead, the $\text{CH}_4\text{-C}$ emission rate had a relatively close fit ($r^2 =$
 264 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,
 265 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.
 266 $[\text{DOC}] = 0$), $d[\text{CH}_4\text{-C}]/dt$ is equal to K_{SOC} with unit of concentration over time. Therefore, K_{SOC} is the
 267 reaction rate constant or C mineralization rate directly from SOC to CH_4 in our system. In addition, K_{DOC}
 268 has a unit of an invert of DOC concentration because a value of exponential function should be
 269 dimensionless. Therefore, K_{DOC} , with a unit of $\text{g-soil} / \text{ug-C}$ in our case, is the solubility or equilibrium
 270 constant of DOC from SOC. Overall $\text{CH}_4\text{-C}$ emission rate is a factor of K_{SOC} with an exponential function
 271 that depends on the SOC to DOC production. If K_{DOC} is small, the function flattens out, suggesting a slower
 272 rate of $\text{CH}_4\text{-C}$ emission. If K_{DOC} is large, the curve increase rapidly, suggesting a rapid production of $\text{CH}_4\text{-C}$.
 273 In this study, although we only found a rather weak correlation between DOC and CH_4 ($r^2 = 0.41$), it is
 274 conceivable that the activity, and potentially also population size, of Archaea responsible for CH_4 formation
 275 increased with increasing DOC concentration, explaining at least to some extent the observed relation
 276 between CH_4 and DOC. Nevertheless, it has to be considered that many other factors such as redox
 277 conditions play a crucial role for CH_4 formation that are not considered in the equation above and that a
 278 good correlation between variables does not imply causality.

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

283

284 **3.2 GHG dynamics under salinity and wet-dry treatments**

285 Figure 2 shows formation of CO_2 , CH_4 , and N_2O from our soil microcosm incubations over 60-d under
 286 different salinities and water levels. Results demonstrated that water level significantly altered the patterns
 287 of CO_2 emissions but salinity did not (Figs. 2a-c). In the three salinity treatments under permanent flooding
 288 (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

289 464.7 (± 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
290 time. Considering the highest CO_2 emission rate, an increase in salinity apparently increased the CO_2
291 emission rate but the differences were not statistically significant ($p > 0.05$). Differences in total $\text{CO}_2\text{-C}$
292 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
293 and 5 ppt of NaCl, respectively, and found to be statistically not significant (Table 2). Under the wet–dry
294 cycle treatment, CO_2 flux followed closely with soil water content. The highest CO_2 flux in each cycle
295 occurred 8 days after each re-flooding, and then the CO_2 mineralization rate declined and was lowest when
296 the soils had the lowest θ_g . Moreover, the highest CO_2 flux decreased with wet–dry cycles during the first
297 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
298 (± 0.7), and 11.6 (± 0.5) mg-C g-soil^{-1} at 0 ppt, 1 ppt and 5 ppt NaCl treatments, respectively. Generally,
299 total $\text{CO}_2\text{-C}$ loss under the wet-dry treatment was statistically equal to flooding condition in the 60-d
300 incubations ($p > 0.05$).

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

309 In contrast, both water level and salinity significantly influenced the emission dynamics of N_2O (Figs.
310 2g-i). Under flooding conditions, high salinity reduced the peak of N_2O emission rate from 9.0 ng-N gsoil^{-1}
311 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 $\text{N}_2\text{O-}$
312 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
313 closely with soil water content, especially in the later incubation. Highest N_2O emissions were quantified
314 immediately after re-flooding for each wet-dry treatment and decreased with decreasing soil water content.
315 Interestingly, N_2O emission rate apparently increased at the later cycles of wet–dry incubation. Total $\text{N}_2\text{O-}$
316 N losses after 60-d of incubation during wet–dry treatments were 0.22 (± 0.03), 0.17 (± 0.01), and 0.15 (\pm
317 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

318 flooding ($p < 0.05$). Compared to the initial concentrations, TC, TN, and DN in the soil actually decreased
319 for all treatments after 60-d incubation, but changes were not statistically significant. Total C loss ($\text{CO}_2\text{-C}$
320 + $\text{CH}_4\text{-C}$) 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
321 original TN.

322

323 **3.3 Water quality**

324 The original forested wetland soil pH was slightly acidic with an average of 5.0 ± 0.1 ($n = 3$), but
325 increased to (6.0 - 7.5) during the 60-d incubation. No significant difference was found among the salinity
326 treatments ($p > 0.05$) although the average pH decreased with salinity. The pH under flooding conditions
327 (6.2 - 7.5) was significantly higher than under wet-dry treatment conditions (5.9 - 6.1) ($p < 0.05$). An
328 increase of pH in soils with increasing salinity is possibly due to the carbonate and a relatively
329 high pH of sea water (pH ~ 8). In our study, we focus on the effect of NaCl. Without the carbonate
330 as buffer reagents, the soil pH could be easily altered. Moreover, a decrease of pH is commonly
331 observed in submerged wetland soils. Organic soil is often acidic during submergence through the
332 slow oxidation of sulfur compounds, producing sulfuric acid, and the production of humic acids
333 (Mitsch and Gosselink, 1993). In fact, we did not observe a decrease in pH in the wet-dry
334 treatment. This suggests that the presence of NaCl is not the main driving force in the observed
335 pH changes.

336 Figure 3 shows the temporal variations of DOC and DN concentrations as well as DOC/DN ratios
337 over 60-d incubation. Original DOC concentrations in all cases were the lowest with 1.2 ± 0.0 mg-C g-
338 soil⁻¹ (Figs. 3a-c). After 60-d of incubation DOC concentrations increased significantly under flooding
339 treatments (1.7 ± 0.1 - 2.2 ± 0.3 mg-C g-soil⁻¹) and 5 ppt in the wet-dry treatments (1.6 ± 0.1 mg-C g-soil⁻¹).
340 DOC concentrations were highest after 12 days of incubation and increased 3.4 - 4.5 times under
341 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
342 with varying salinities. However, DOC concentrations were significantly higher in the permanent
343 flooding incubation compared to incubation under wet-dry conditions ($p < 0.05$). Although DOC
344 concentrations at 0 ppt and 1ppt NaCl during flooding exceeded DOC concentrations in the 5ppt NaCl

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

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

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

355

356 **3.4. Optical characterization of DOM**

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

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

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

374 65% of DOM, and protein-like DOM was the second most abundant fraction, comprising 20 to 41% of
375 DOM in all treatments. Although the proportion of various components of DOM fluctuated during the
376 incubation under various salinity and water level conditions, the observed structural changes were
377 statistically not significant ($p > 0.05$) (Fig. 6). Compared to original DOM, fulvic-acid like and humic-acid
378 like DOM are the most dominant fractions after 60 days.

379

380 **4. Discussion**

381 **4.1 Contribution of different C fractions to GHG formation**

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

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

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

404

405 **4.2 Carbon –salinity, water level (flooding and wet-dry)**

406 **4.2a Carbon – salinity vs water level**

407 Rising sea level has increased the hydroperiod and salinity in low-lying coastal freshwater forested
408 wetlands (Krauss and Whitbeck, 2012). These environmental changes alter the C and N biogeochemical
409 processes in wetland soils in different ways. Our experiments suggest that an increase in salinity under
410 oligohaline conditions (≤ 5 ppt) did not significantly alter C cycling in terms of CO₂/CH₄ emission and
411 DOC production (Figs. 5 a and c). For gas emissions, our results are in line with a previous study showing
412 no effect on CO₂ or CH₄ emission with mean pore water salinity ranging from 0.2 to 4.7 ppt in the same
413 type of forested wetland (Krauss and Whitbeck, 2012). Results suggest the impacts of low levels of
414 saltwater intrusion on CO₂ and CH₄ productions from coastal wetland soils were negligible. However, it
415 has been demonstrated that fluctuations in salinity affect DOC production in coastal wetlands (Olsen et
416 al., 1996; Ardón et al., 2016; Chambers et al., 2014). Olsen et al. (1996) and Ardón et al. (2016)
417 demonstrated that saltwater intrusions could reduce DOC in leachate while Chambers et al. (2014) showed
418 an increase in DOC concentration in soil pore water. Our study did not observe any differences among
419 salinity treatments after 60-d incubation. While CO₂ and CH₄ emissions are mainly controlled by microbial
420 processes, DOC concentrations and exports are the result of a combination of environmental biotic and
421 abiotic factors (Chow et al., 2003). The inconsistencies among studies are probably due to the involvement
422 of several physical (e.g., coagulation and adsorption), chemical (e.g., photochemical and redox processes),
423 and biological (e.g., microbial production and decomposition) processes during DOC production in soil
424 water (Zsolnay, 2003). In this study, we specifically investigated the effects of NaCl and hydroperiod on
425 freshwater wetland soils under low salinity levels (≤ 5 ppt) for a period of 60 days. An increase in DOC
426 during the incubation was mainly due to the effects of water level (see section 4.2b). Results demonstrated
427 that the effect of salinity on C cycling in oligohaline environments in a short period of time is negligible.
428 Water level or the hydroperiod is the driving force on C cycling under oligohaline conditions. However, it
429 has to be considered that results from a simplified laboratory incubation cannot directly be transferred to
430 conditions in the field which are much more complex. In particular, the choice of salt can be of relevance.

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

434

435 **4.2b Carbon – wet-dry vs flooding**

436 In contrast to the salinity treatments, water level treatments had profound effects on C cycles under
437 oligohaline conditions. Our experiments demonstrated no significant increase in CO_2 emission but
438 significant decreases in CH_4 and DOC productions when comparing flooding to wet-dry treatments ($p <$
439 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
440 wet-dry cycle, there was no significant difference in total $\text{CO}_2\text{-C}$ emission between wet-dry and flooding
441 treatments after 60-d incubations. Highest emissions of CO_2 were quantified immediately after rewetting.
442 This results from an increase in C which most likely was caused by microbial biomass from cells death
443 during the drying cycle as the decrease of soluble microbial byproduct-like from EEMs shown in Figs 6 a
444 and b. Also, wet-dry treatments raised the exposure of organic residues (Denef et al., 2001a, 2001b) due to
445 the grinding of soil structure for exchange of wetting and drying process (Fierer and Schimel, 2002;
446 Lundquist et al., 1999). These findings have been reported in laboratory and field experiment in peatland
447 soil (Chow et al., 2006) and forest soils (Jarvis et al., 2007). However, the wet-dry treatment did not affect
448 total $\text{CO}_2\text{-C}$ loss in our study because of the offset of highest CO_2 emission rate at re-flooding and lowest
449 CO_2 emission rate during the dry period. In sum, the wet-dry treatment resulted in 4.6-5.1% ($\text{CO}_2\text{-C} / \text{TC}$)
450 C loss and flooding resulted in 4.4-4.9% C loss.

451 The drying cycle in the wet-dry treatments probably led to oxic conditions, and consequently might
452 have inhibited the formation of CH_4 (Olsson et al., 2015). Only 0.1-0.2% of soil C was converted to $\text{CH}_4\text{-}$
453 C in wet-dry treatments, and 1.0-1.2% in flooding treatments. Total C release from wet-dry treatments was
454 9.1-15.9% lower compared to flooding treatments. Our results are in line with previous results showing a
455 decrease in CH_4 emissions in peatland soils and freshwater marsh when water table fluctuates (Moore and
456 Knowles, 1989; Yang et al., 2013). Although wet-dry treatments did not affect total $\text{CO}_2\text{-C}$ loss, there was
457 a decrease in $\text{CH}_4\text{-C}$ loss. Therefore, our data suggest that wet-dry treatments significantly decreased total
458 C emissions ($\text{CO}_2 + \text{CH}_4$). These results are consistent with previous studies where fluctuating water tables
459 reduced C emissions (Bass et al., 2014; Olsson et al., 2015).

460 Our study demonstrated that wet-dry treatments increased DOC production 1.1-1.6 fold after 60-d
461 incubation. This was significantly lower compared to flooded treatments (1.7-1.9 fold) ($p < 0.05$).
462 Researchers have shown that short heterotrophic microbial processes such as respiration and
463 denitrification could lower DOC concentrations in wet-dry conditions more than that in flooding
464 environments (Blodau and Moore, 2003; Burford and Bremner, 1975). Enhanced penetration of oxygen
465 in wet-dry treatments resulted in an increase of both aerobic and anaerobic microbes (Fierer and Schimel,
466 2003; Lundquist et al., 1999). Also, the decrease in DOC concentration in wet-dry treatments could be
467 attributed to altered microbial community structures during wet-drying process enhancing labile DOC
468 consumption.

469

470 **4.3 Nitrogen – salinity, water level (flooding and wet-dry)**

471 In contrast to C biogeochemistry, significant changes were observed in N cycling in both saline and
472 flooding treatments. An increase in salinity increased DN release but decreased N₂O emission from soils
473 (Figs. 5 a, c). The increase in DN in water can be explained by ion exchange with NH₄⁺ cations in soil with
474 sodium cations (Na⁺) in water (Wang and Sun, 2013). However, the increased availability of DN did not
475 necessarily increase N₂O emission because high levels of salt can suppress both nitrification and
476 denitrification (Osborne et al., 2015; Li et al., 2013). In addition to salinity, water level, which affects the
477 availability of O₂ and redox status, could affect denitrification and N₂O production processes (Weitz et al.,
478 2001; Lu et al., 2014). Continuous flooding limits the availability of oxygen in soils, reducing nitrification
479 (Pezeshki and DeLaune, 2012). In contrast, wet–dry cycles allow O₂ penetration into the soil during dry
480 periods, influencing microbial processes (i.e., substrate availability and microbial cell physiology) and soil
481 physical properties, thus enhancing N₂O emissions (Stark and Firestone 1995; Burger et al., 2005; Kim et
482 al., 2012) and increasing soil N losses (Borken and Matzner, 2009). Similar observations showing the
483 effects of salt on N₂O and DN in wetlands have been reported previously (Azam and Ifzal, 2006; Chambers
484 et al., 2013; Ardon et al., 2013).

485

486 **4.4 Quality of DOM**

487 SUVA is widely used as an indicator of aromatic C in soil and aquatic humic substances (Novak et
488 al., 1992; Wang et al., 2015a; Yu et al., 2010). The E₂/E₃ ratio is inversely correlated with MW of DOM

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

498

499 **Conclusions and Outlook**

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

516

517 **Acknowledgements**

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

524

525 **References**

526 Allen, J.A., Pezeshki, S.R., Chambers, J.L., 1996. Interaction of flooding and salinity stress on baldcypress
527 (*Taxodium distichum*). *Tree Physiology* 16, 307–313. doi:10.1093/treephys/16.1-2.307

528 Ardón, M., Helton, A.M., Bernhardt, E.S., 2016. Drought and saltwater incursion synergistically reduce
529 dissolved organic carbon export from coastal freshwater wetlands. *Biogeochemistry* 127, 411–426.
530 doi:10.1007/s10533-016-0189-5

531 Ardón, M., Morse, J.L., Colman, B.P., Bernhardt, E.S., 2013. Drought-induced saltwater incursion leads
532 to increased wetland nitrogen export. *Glob. Chang. Biol.* 19, 2976–2985. doi:10.1111/gcb.12287

533 Azam, F., Ifzal, M., 2006. Microbial populations immobilizing NH_4^+ -N and NO_3^- -N differ in their
534 sensitivity to sodium chloride salinity in soil. *Soil Biology and Biochemistry* 38, 2491–2494.
535 doi:10.1016/j.soilbio.2006.01.028

536 Bai, J., Gao, H., Xiao, R., Wang, J., Huang, C., 2012. A review of soil nitrogen mineralization as affected
537 by water and salt in coastal wetlands: issues and methods. *Clean - Soil, Air, Water* 40, 1099–1105.
538 doi:10.1002/clen.201200055

539 Baldwin, D.S., Rees, G.N., Mitchell, A.M., Watson, G., Williams, J., 2006. The short-term effects of
540 salinization on anaerobic nutrient cycling and microbial community structure in sediment from a
541 freshwater wetland. *Wetlands* 26, 455–464.
542 doi:10.1672/0277-5212(2006)26[455:TSEOSO]2.0.CO;2

543 Bass, A.M., O'Grady, D., Leblanc, M., Tweed, S., Nelson, P.N., Bird, M.I., 2014. Carbon dioxide and
544 methane emissions from a wet-dry tropical floodplain in northern Australia. *Wetlands* 34, 619–627.
545 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 ST–Soil carbon sequestration impacts. 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., Kansiime, 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.

- 714 doi:10.1126/science.1135456
- 715 Renaud, F.G., Le, T.T.H., Lindener, C., Guong, V.T., Sebesvari, Z., 2015. Resilience and shifts in agro-
716 ecosystems facing increasing sea-level rise and salinity intrusion in Ben Tre Province, Mekong Delta.
717 *Climatic Change* 133, 69–84. doi:10.1007/s10584-014-1113-4
- 718 Schofield, R. V., 2003. Application of salinization indicators and initial development of potential global
719 soil salinization scenario under climatic change. *Global Biogeochem. Cycles* 17, 1–13.
720 doi:10.1029/2002GB001935
- 721 Sholkovitz, E.R., 1976. Flocculation of dissolved organic and inorganic matter during the mixing of river
722 water and seawater. *Geochim. Cosmochim. Acta* 40, 831–845. doi:10.1016/0016-7037(76)90035-1
- 723 Setia, R., Gottschalk, P., Smith, P., Marschner, P., Baldock, J., Setia, D., Smith, J., 2013. Soil salinity
724 decreases global soil organic carbon stocks. *The Science of the Total Environment* 465, 267–72.
725 doi:10.1016/j.scitotenv.2012.08.028
- 726 Shi, A., Marschner, P., 2014. Changes in microbial biomass C, extractable C and available N during the
727 early stages of decomposition of residue mixtures. *Soil Research* 52, 366–372. doi:10.1071/SR13128
- 728 Silva, C.C., Guido, M.L., Ceballos, J.M., Marsch, R., Dendooven, L., 2008. Production of carbon dioxide
729 and nitrous oxide in alkaline saline soil of Texcoco at different water contents amended with urea: A
730 laboratory study. *Soil Biology and Biochemistry* 40, 1813–1822. doi:10.1016/j.soilbio.2008.03.004
- 731 Stuckey, B.N. 1982. Soil survey of Georgetown County South Carolina. Rep. Prepared for the Soil
732 Conservation Service, USDA, Washington, DC.
- 733 Wang, J.J., Dahlgren, R.A., Chow, A.T., 2015a. Controlled burning of forest detritus altering spectroscopic
734 characteristics and chlorine reactivity of dissolved organic matter: Effects of temperature and oxygen
735 availability. *Environmental Science and Technology* 49, 14019–14027. doi:10.1021/acs.est.5b03961
- 736 Wang, J.J., Dahlgren, R.A., Erşan, M.S., Karanfil, T., Chow, A.T., 2015b. Wildfire altering terrestrial
737 precursors of disinfection byproducts in forest detritus. *Environmental Science and Technology* 49,
738 5921–5929. doi:10.1021/es505836m
- 739 Wang, L., Sun, H., 2013. Prediction of $\text{Na}^+ / \text{NH}_4^+$ exchange in faujasite zeolite by molecular dynamics
740 simulation and thermodynamic integration method. *The Journal of Physical Chemistry C* 117, 14051–
741 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

770 Zhu, B., Cheng, W., 2013. Impacts of drying-wetting cycles on rhizosphere respiration and soil organic
771 matter decomposition. *Soil Biology and Biochemistry* 63, 89–96. doi:10.1016/j.soilbio.2013.03.027

772 Zsolnay, Á., 2003. Dissolved organic matter: Artefacts, definitions, and functions. *Geoderma* 113, 187–
773 209. doi:10.1016/S0016-7061(02)00361-0

774

775

776

777

778

779

780

781

782

783

784

785

786

787

788

789

790

791

792 **Figure Captions**

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

795 **Fig. 2** Formation rates of CO₂ (a-c), CH₄ (d-f), and N₂O (g-i) in surface forested wetland soil incubated at
796 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
797 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
798 cumulative emission over 60 days. Error bars represent the standard deviations of triplicate measurements.
799 The vertical lines in each diagram indicate the 12-day wet-dry treatments. a-c variations are rate and total
800 of CO₂-C emission among salinity, d-f variations are rate and total of CH₄-C emission among salinity; g-i
801 are rate and total of N₂O-N emission among salinity during 60-d incubation

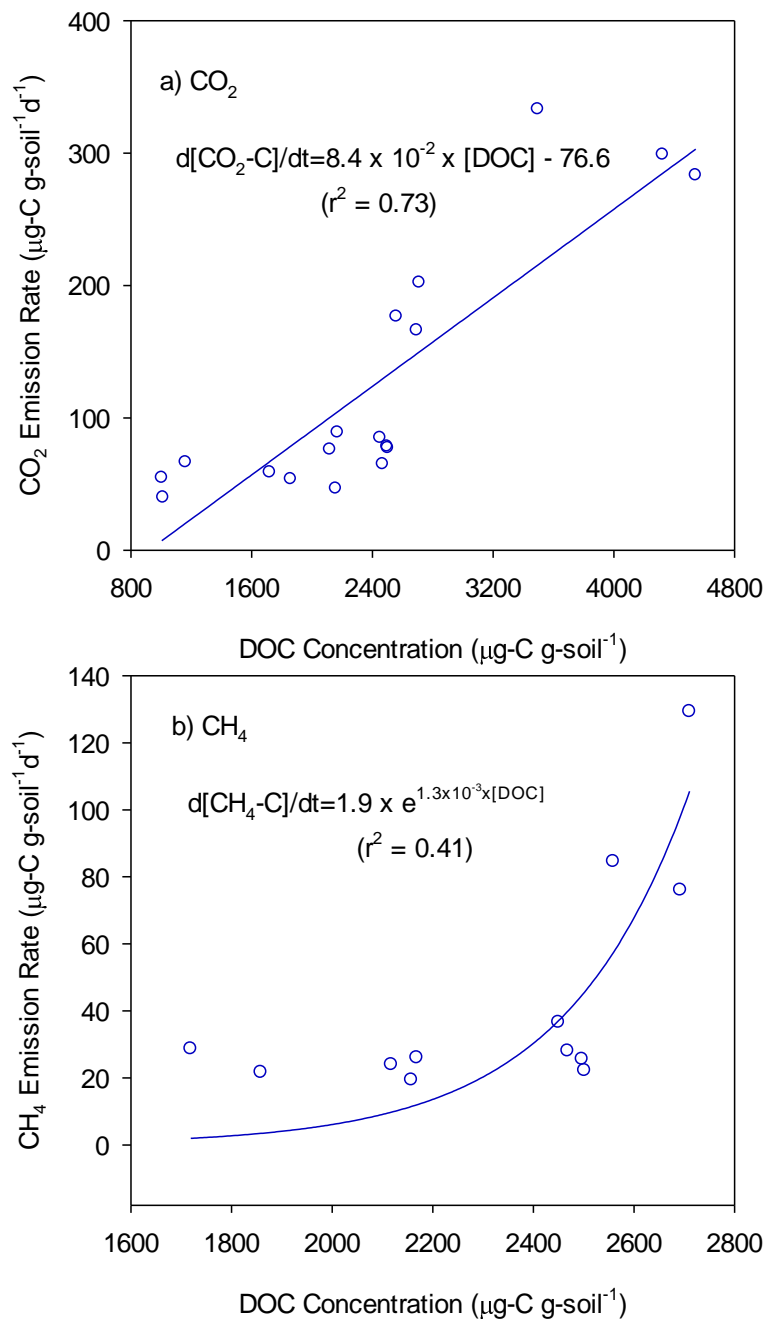
802 **Fig.3** Variations in DOC (a-c), DN (d-f) and DOC/DN (g-i) at 0, 1, and 5 ppt NaCl in surface forested
803 wetland soil under flooding and wet-dry treatments during 60-d incubation. Error bar represents the
804 standard deviation from triplicate measurements

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

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

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

819



820

821

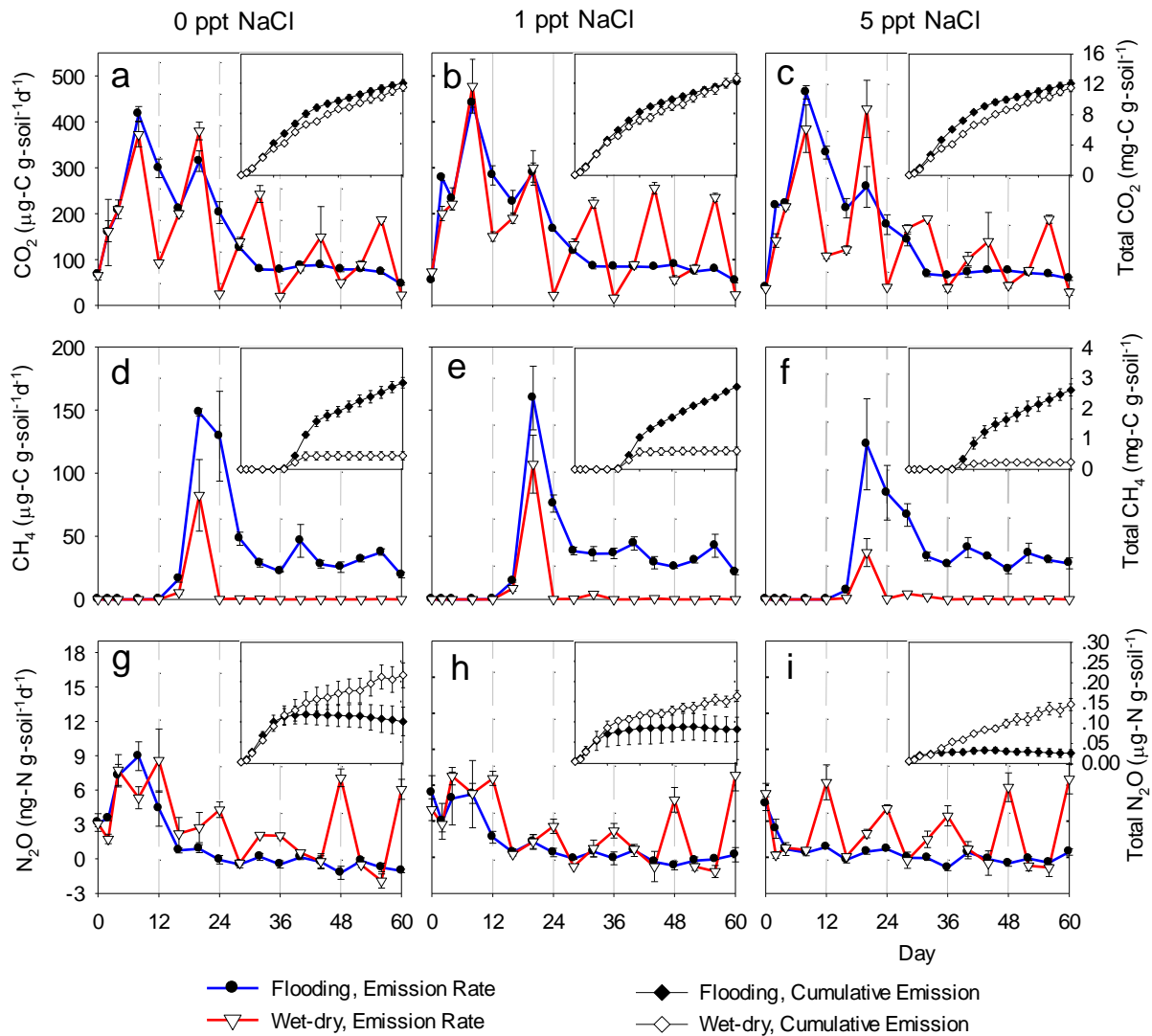
822 Fig. 1 The relationship between (a) DOC- CO_2 and (b) DOC- CH_4 in surface forested wetland soil under
823 permanent flooding.

824

825

826

827

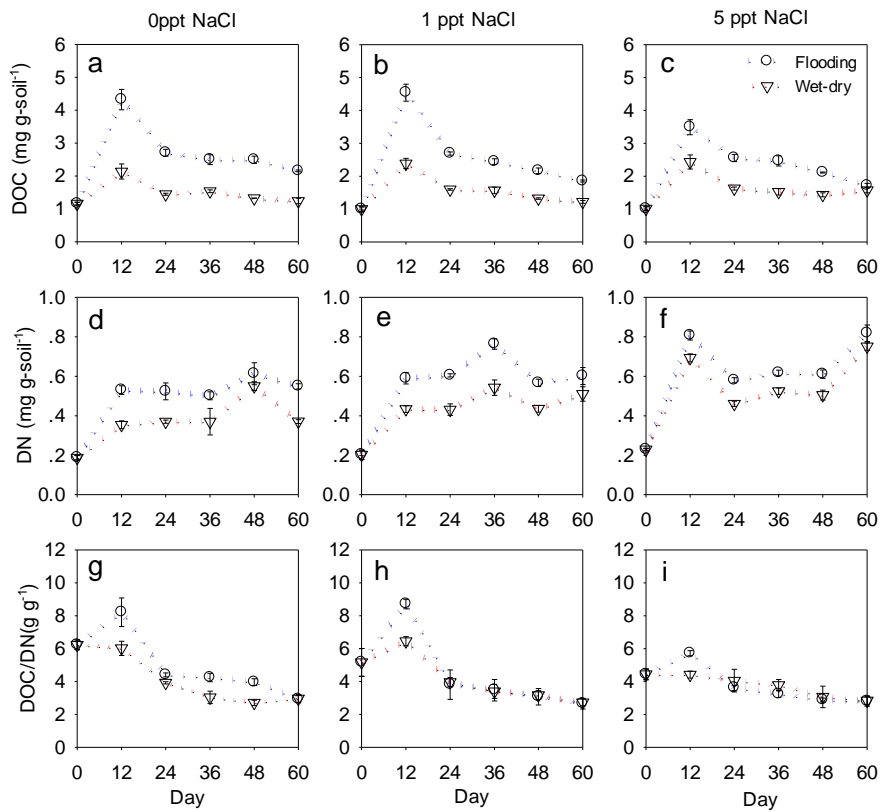


828

829 Fig. 2 Formation rates of CO₂ (a-c), CH₄ (d-f), and N₂O (g-i) in surface forested wetland soil incubated at
 830 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
 831 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
 832 cumulative emission over 60 days. Error bars represent the standard deviations of triplicate measurements.
 833 The vertical lines in each diagram indicate the 12-day wet-dry treatments. a-c variations are rate and total
 834 of CO₂-C emission among salinity, d-f variations are rate and total of CH₄-C emission among salinity; g-i
 835 are rate and total of N₂O-N emission among salinity during 60-d incubation

836

837



838

839

840 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
 841 wetland soil under flooding and wet-dry treatments during 60-d incubation. Error bar represents the
 842 standard deviation from triplicate measurements

843

844

845

846

847

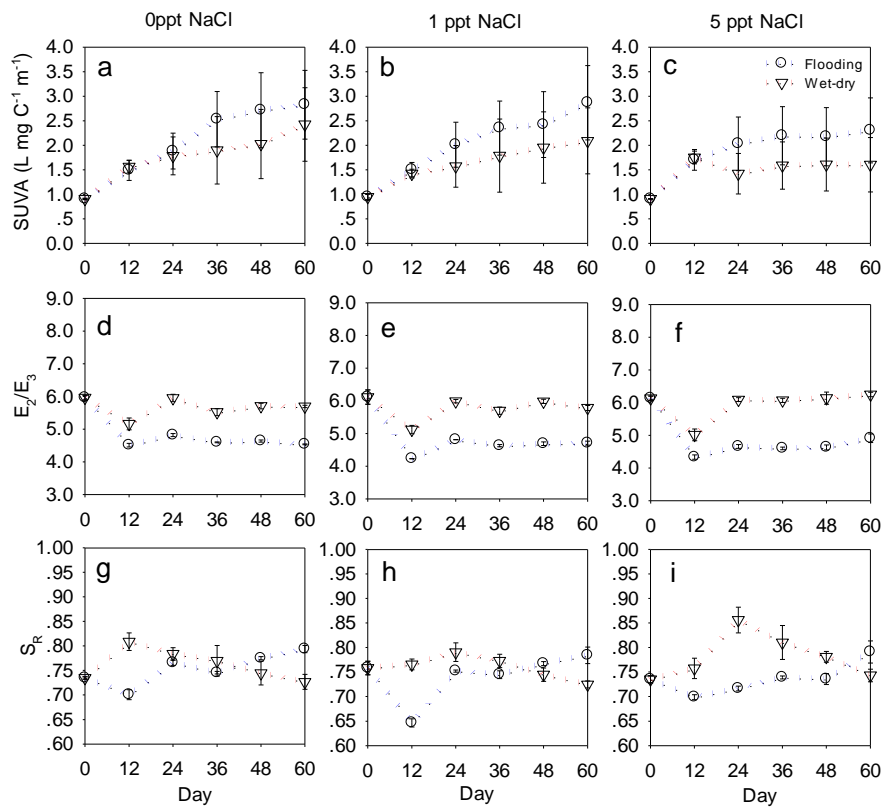
848

849

850

851

852



853

854

855 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
 856 salinities (0, 1, and 5 ppt NaCl) under flooding and wet-dry conditions over 60 days. Error bar represents
 857 the standard deviation from triplicate measurements

858

859

860

861

862

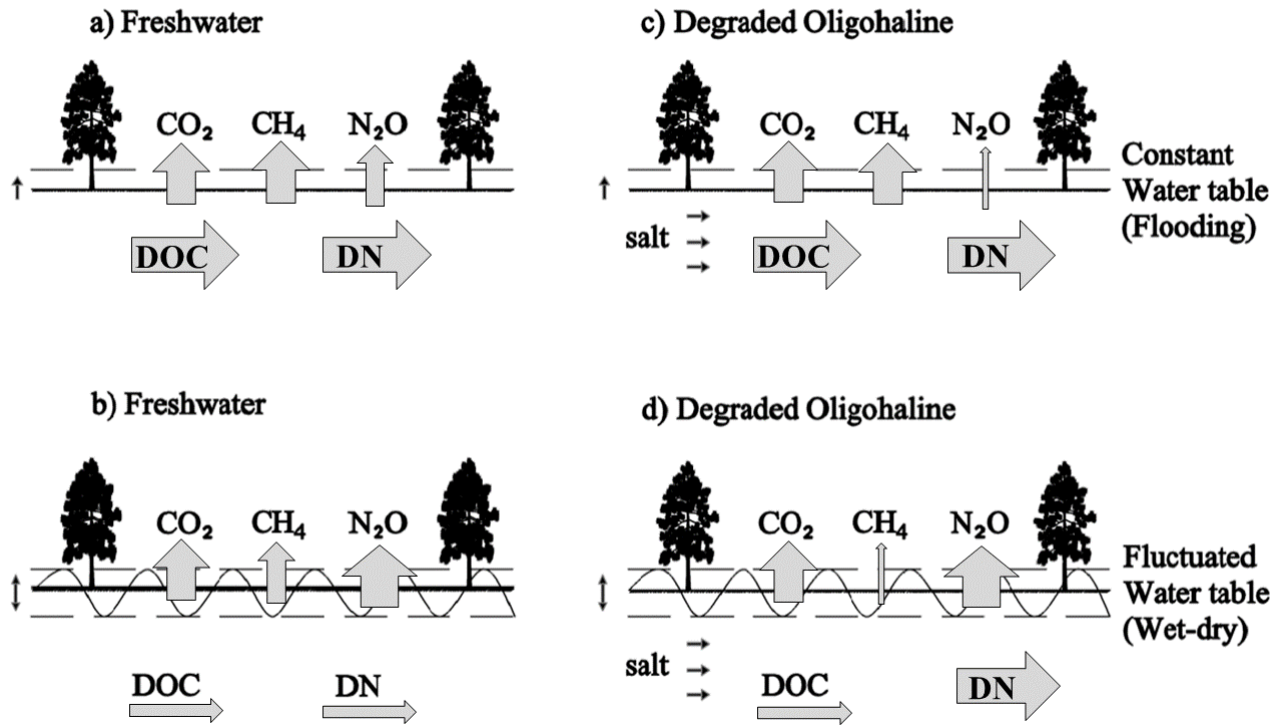
863

864

865

866

867



868

869

870 Fig. 5 A conceptual model for describing effect of water level and salinity on freshwater forested wetlands.

871 a and b are soil incubation with freshwater (0 ppt) experiencing flooding and wet-dry treatments,

872 respectively; c and d are soil incubation with degraded oligohaline (5 ppt) experiencing flooding and wet-

873 dry treatments, respectively.

874

875

876

877

878

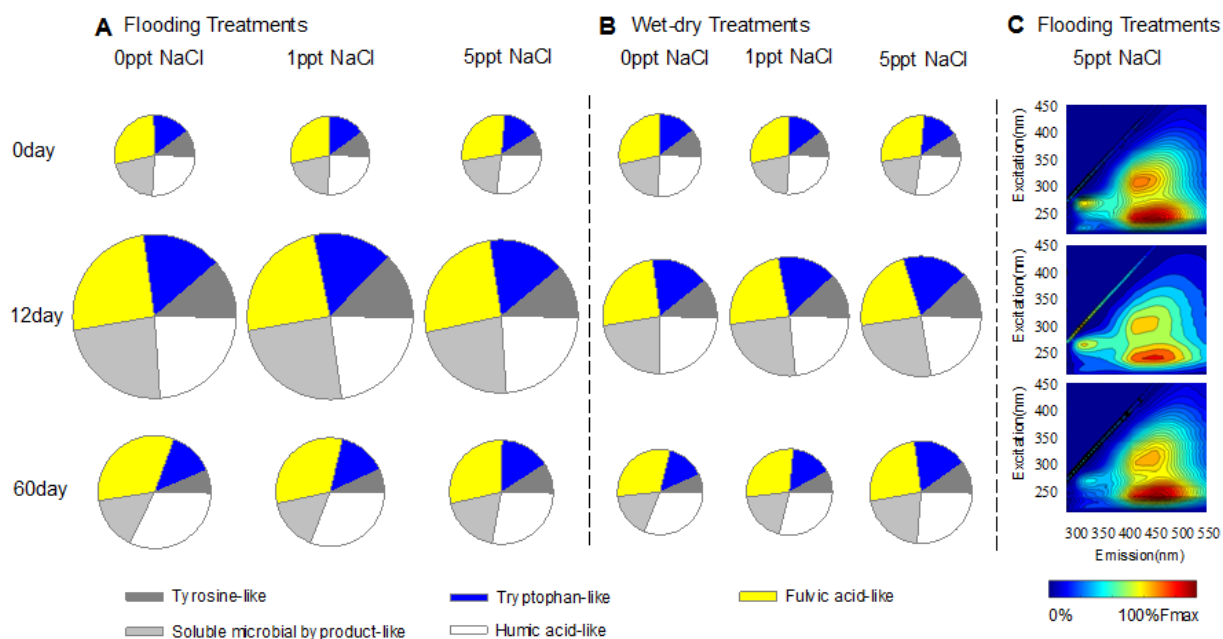
879

880

881

882

883



884

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

892

893

894

895

896

897

898

899 **Tables**900 **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

901

902

903 **Table 2. Total emissions of carbon dioxide, methane and nitrous oxide with salinity and soil water**904 **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}

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

906

907

908

909

910

911

912

913

914

915

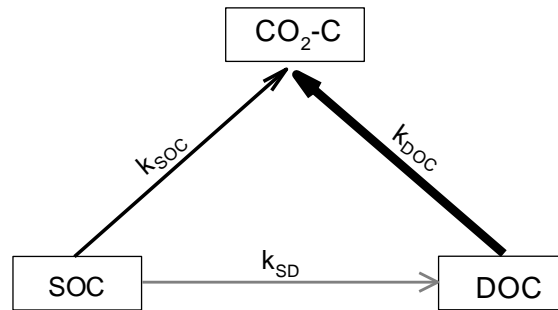
916

917

Supplementary Information

918

919



Box

CO₂-C:

[1] $d[\text{CO}_2\text{-C}]/dt = k_{\text{SOC}}[\text{SOC}] + k_{\text{DOC}}[\text{DOC}]$

[2] $d[\text{CO}_2\text{-C}]/dt = k_{\text{SOC}}([\text{TAOC}] - [\text{DOC}]) + k_{\text{DOC}}[\text{DOC}]$

[3] $d[\text{CO}_2\text{-C}]/dt = (k_{\text{DOC}} - k_{\text{SOC}})[\text{DOC}] + k_{\text{SOC}}[\text{TAOC}]$

[4] $d[\text{CO}_2\text{-C}]/dt = k_{\text{app}}[\text{DOC}] + k_{\text{SOC}}[\text{TAOC}]$

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

920

921

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

923

924

925

926

927

928

929

930

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

938 **Table s2. The pH and EC (mean ± standard deviation; n=3) of forested wetland soil during 60-d**
 939 **incubation**

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

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

941

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