Nitrous oxide uptake in rewetted wetlands with contrasting soil organic carbon contents

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Abstract: Rewetting drained peatlands creates favorable reduced conditions for denitrification, but, the potential of restored peatlands to act as N₂O sink is poorly characterized. We monitored N₂O emissions in rewetted peatlands converted to rice paddies with a range of soil organic carbon (SOC) (6%, 11%, and 23%) during the growing season. Negative N₂O emissions were repeatedly observed in all sites, which were not affected by nitrogen fertilization at 80 kg N ha⁻¹. The highest consumption rates amounted to 8.2 tones CO₂ eq ha⁻¹ yr⁻¹, 32% of the average CO₂ emissions from drained peat soils. Porewater N₂O concentrations were frequently less than the calculated *in situ* N₂O equilibrium concentrations, suggesting diffusional constraints on atmospheric N₂O into pore water. Redox potentials were correlated to N₂O emission rates ($r^2 = 0.40$, $p < 0.01$). However, relatively higher frequencies and magnitudes of negative N₂O emissions were mostly observed during the period of water draw down for rice harvest activities, especially in the field with the highest SOC. It is likely that reducing water depth by drainage reduced the diffusional barrier for atmospheric N₂O into the soil. In all, the capacity of rewetted peatlands to act as atmospheric N₂O sinks can be significant and seemingly can be managed through manipulating water depths.

Keyword: N₂O consumption, restored peatland, soil organic carbon, redox potential

1. Introduction

Nitrous oxide (N_2O) is a potent greenhouse gas (GHG) and also plays a significant role in the destruction of stratospheric ozone, making it one of the most reactive forms of nitrogen (N) pollution (Ravishankara et al., 2009; Stocker et al., 2013). Soils are the largest contributors to atmospheric N_2O , accounting for approximately 65% of the total annual emissions with 4.1 Tg N_2O -N yr^{-1} from agricultural and 6.6 Tg N_2O -N yr^{-1} from natural sources (Davidson and Kanter, 2014). Less understood, however, is the role of soils as a N_2O sink.

The main sink for N_2O in soils is its enzymatic reduction to dinitrogen gas (N_2) by N_2O reductase (Richardson et al., 2009). However, unlike the other two major GHGs, CO_2 and methane (CH_4), the underlying controls over N_2O consumption and the capacity of soils to act as sinks have received less attention. High soil moisture, low mineral N, and high soil C have been shown to favor N_2O consumption (Majumdar, 2013). Other factors affecting N_2O consumption include soil pH (Čuhel et al., 2010; Stevens et al., 1998), redox potential (Włodarczyk et al., 2005; Yu and Patrick, 2003), and the ease of diffusion of N_2O within the soil profile (Chapuis-Lardy et al., 2007; Ryden, 1981). However, there are no clear lines of evidence for any factor or combination of factors that can reduce the uncertainty in predicting *in situ* N_2O consumption (Chapuis-Lardy et al., 2007).

Nitrous oxide consumption has been observed under various soil conditions (Audet et al., 2014; Chapuis-Lardy et al., 2007; Warneke et al., 2015). In a recent survey of various natural ecosystems, Schlesinger (2013) concluded that global sinks for N_2O in soils is less than 2% of the current estimated sources of N_2O and is likely not important, largely due to the fact that most observations and measurements of net N_2O uptake were sporadic with a frequency of < 10% of

the measurements or the measurements were at the detection limits of analysis. However, in a review of soil N₂O uptake, Chapuis-Lardy et al. (2007) suggested that N₂O consumption might be important and should be taken into account, which is also supported by others believing that N₂O consumption can have a larger than realized role in regulating atmospheric N₂O (Jones et al., 2014; Richardson et al., 2009; Ryden, 1981). Despite this disagreement, studies have shown large variations in the rates of negative N₂O emission, but the highest negative values were mostly observed in wetlands and peatlands, suggesting N₂O uptake and consumption is more efficient under reduced conditions (Chapuis-Lardy et al., 2007; Schlesinger, 2013).

Peatlands cover only 3% of the world's surface, but contain 30% of soil C stocks (Parish, 2008). The formation and maintenance of peatlands requires a water table close to the surface (Belyea and Baird, 2006), and thus a substantial portion of the soil profile normally undergoes anaerobic conditions. Naturally, water saturated conditions and high soil organic C (SOC) content create favorable environments for denitrification and N₂O reduction (Reddy and DeLaune, 2004), which may at least partially explain why highest negative N₂O emissions were frequently observed in these systems (Schlesinger, 2013). However, drainage and cultivation of these organic soils has modified the oxidation-reduction status, resulting in extensive degradation and GHG emissions, primarily as carbon dioxide (CO₂) and N₂O (IPCC, 2014). Restoring drained peatlands is considered a climate mitigation action in the United Nations Framework Convention on Climate change (UNFCCC) and Inter-Governmental Panel for Climate Change (IPCC) (IPCC, 2014). Rewetting allows for the re-establishments of reduced soil conditions, which is expected to retard SOC decomposition and CO₂ emissions (IPCC, 2014; Joosten et al., 2012). A less considered outcome, however, is the potential of these organic soils to act as a N₂O sink.

The Sacramento-San Joaquin Delta (hereafter the Delta) was once a 1400 km² tidal marsh influenced by runoff from the Sierra Nevada Mountains, but drained for agriculture in the mid-1800s, resulting in extensive soil C loss and subsidence and large continuous CO₂ and N₂O emissions (Deverel and Leighton, 2010; Drexler et al., 2009). Restoration of these degraded agriculture systems by rewetting has been increasingly recognized as a regional solution to reduce or reverse soil subsidence and mitigate GHGs emissions (Knox et al., 2015). We have recently observed significant net N₂O uptakes in a rewetted drained peatland (containing 15% SOC) during a water draw-down period before rice harvest (Morris et al., in preparation). The negative emission rates ranged from -17 to -102 with a median of -38 μg m⁻² h⁻¹ and the frequency of negative emissions observed was 30% of the measurements. The median and frequency are 9.5 and 3 times of those reported by Schlesinger (2013) in a survey of various natural ecosystems, demonstrating the potential of rewetted peatlands to serve as N₂O sinks. We therefore explored this potential with three more rewetted degraded peatland sites containing contrasting SOC contents (6%, 11%, and 23%). It is expected that similar frequencies and magnitudes of net N₂O uptake would be observed in these rewetted peatlands, where soils are highly organic, while porewater inorganic N were minimal (Kirk et al., 2015), both of which are seemingly favorable for denitrification and N₂O reduction (Chapuis-Lardy et al., 2007).

2. Materials and methods

2.1. Site description

Field studies were conducted on Twitchell Island (38.1053 N, 121.6542 W) in the Sacramento-San Joaquin Delta in 2013. Mean annual precipitation for this region is 380 mm and

over 80% occurs from November to March. Yearly average high and low temperature is 22 °C and 9 °C. The areas were once expansive wetlands, but drained and converted to agricultural production in late 1800's, which resulted in severe soil oxidation and subsidence (Deverel and Leighton, 2010). The soils are Rindge silt loams classified as Eutric Rheic Sapric Histosol (Hyperorganic) (FAO WRB) or euic, thermic Typic Haplosaprists (USDA) with organic C contents ranging from less than 5 to greater than 25% in concentration depending on the degree of soil oxidation. Rice (*Oryza sativa* L.) has been grown on the site since 2009 to test its potential as a regional solution to reduce soil subsidence, mitigate GHG emissions, and improve Delta water quality.

2.2. Experimental design and field setup

The experiment was embedded in an N rate trial across a gradient of SOC (Espe et al., 2015). Three sites with contrasting SOC (6, 11, and 23%) were selected for this study with the distance between each site less than 1.25 km. The SOC gradients cover most of the SOC ranges across the Delta (Deverel and Leighton, 2010). A total of five N rates (0, 40, 80, 120, 160 kg N ha⁻¹) were conducted as a randomized complete block design with four replicates in each site. Only two rates (0 and 80 kg N ha⁻¹) were chosen for this study. The 80 kg N ha⁻¹ was chosen as it was the highest recommended N rate for rice production in this range of SOC contents (Espe et al., 2015). All sites were managed the same throughout the entire season. Each experimental plot was 4 × 5 m². Rice variety 'M-206' was planted on mid-April in 2013 in all fields, except for the 23% C field, where 'M104' was planted by accident. Both varieties are medium grain having similar yield potential (data available online at <http://www.carrb.com/Variety/M-206.htm>) and N requirements (Mutters et al., 2013). In statewide yield tests, 'M-206' flowered three days later than 'M104', had improved lodging resistance, and improved resistance for blanking caused

by temperatures. Following N fertilization as urea and recommended amounts of phosphorus and potassium, fields were flooded on May 15 for plant growth and drained on August 10 for harvest in 2013.

2.3. N₂O emissions

N₂O fluxes were monitored weekly during the growing season and every other day following paddy drainage prior to harvest. Emissions were measured with close vented chambers from 10:00 am to 3:00 pm (Parkin et al., 2003). The chambers have an inner diameter of 25 cm and a height of 9, 69, and 99 cm, depending on the height of the rice plants. A PVC ring (25 cm in diameter and 15 cm in depth) was driven into soils one week prior to seasonal measurements and protruded 2-3 cm above the soil surface. The rings were removed at harvest then reinstalled. Before measurements, chambers were placed on rings and sealed with a wide rubber strip made from a cross cut section of a tire inner tube that overlapped the chamber and ring by 5 cm. A fan was used to mix the chamber headspace for 10 or 60 seconds (depending on chamber height) to avoid gas gradients (Parkin et al., 2003), followed by extracting 20 mL of air from the chamber every 10 min during a 30 min period for a total of 4 samples including time 0. Air samples were transferred completely to 12 mL pre-vacuumed vial and subsequently analyzed for N₂O on a GC-2014 gas chromatograph (Shimadzu, Columbia, Maryland). N₂O emission rates were calculated with a linear regression model of their concentrations against time and expressed as $\mu\text{g m}^{-2} \text{ hr}^{-1}$ after adjusting for soil temperature and chamber volumes (Pittelkow et al., 2013). If the determinant coefficient (R^2) for N₂O was < 0.9 , emissions were set to zero, which were mostly observed when NH_4^+ concentration was less than 0.5 mg L^{-1} and no NO_3^- was detected ($< 0.01 \text{ mg L}^{-1}$). We assumed that N₂O emission was minimal when these substrates for nitrification and denitrification were both limited (Baggs, 2011; Reddy and DeLaune, 2004).

2.4. Interstitial water chemistry and other field measurements

Porewater was collected with a custom sampler. The sampler was a 15 cm long and 5 mm in diameter polyethylene tube perforated with a row of small holes (2 mm in diameter) equally spaced every 1 cm, covered with a nylon mesh (30 μm) (Genesee Scientific Corp., San Diego, CA) to exclude large plant and soil particles, and connected to a 2-way valve via Tygon tubing, through which a 5 μm syringe filter was attached. The sampler was inserted vertically into the soil (0-15 cm) inside the chamber ring after it was installed and remained there during the seasonal sampling periods. Water was collected by attaching a pre-evacuated 12 mL extainer onto the syringe filter with attached needle. The initial porewater sample was discarded and the second sample was transported on ice to the laboratory and stored at 4°C until used.

Porewater NH_4^+ (Verdouw et al., 1978) and NO_3^- (Doane and Horwath, 2003) was determined colorimetrically. Porewater N_2O was determined according to the method described by Bossio et al. (1999). In brief, after the removal of half of porewater for NH_4^+ , NO_3^- and other analysis, the headspace of the extainer with the remaining porewater was replenished with helium (overpressurized by 2 mL helium) and shaken vigorously with a vortex mixer for 3 min, followed by the extraction of the headspace gas for N_2O analysis. *In situ* N_2O equilibrium concentrations in water were calculated with measured field atmospheric N_2O and the solubility coefficient for N_2O (Clough et al., 2007; Weiss and Price, 1980).

Soil temperature during sampling was measured inside the chamber ring with a T type thermocouple at 5 cm depth. Redox potential was determined using an Orion combination ORP probe (Thermo Scientific Orion, USA) at 10 cm below soil surface, which was inserted into soils

adjacent to the chamber before each sampling campaign started. In each experimental field ($16 \times 20 \text{ m}^2$) two probes were deployed and average values of the two probes were reported.

2.5. Laboratory incubation

After harvest, all the surface soils (0-15cm) within each gas chamber ring were collected, air dried, and sieved (2 mm) in the laboratory after the removal of visible plant residues and roots. Soils from the same field were composited, homogenized, and stored at room temperature ($20 \pm 1 \text{ }^\circ\text{C}$) until used.

To investigate the water level effect on N_2O emissions, the 11% and 23% C soils were selected for a water manipulation experiment with 4 replicates. Fifty grams of soils were placed in a 1000 mL Mason jar, followed by addition of deionized water to 10 cm above soil surface. The jars were capped with lids equipped with a foam plug to minimize water evaporation while allowing air exchange and incubated at $20 \pm 1 \text{ }^\circ\text{C}$ in dark for 7 days. After the pre-incubation N_2O emissions were quantified daily for 8 days. However, on day 3, following the measurements of N_2O emissions, two thirds of the samples were randomly selected and surface water inside the jars were completely removed for half of the selected samples, while only 5 cm water was removed for the other half. Gas measurements were resumed on the next day. Prior to gas measurement, the lids of the jars were removed and replaced with a lid with a butyl septum in its center. Gas sampling, analyses, and flux calculations essentially followed the same procedures for field measurements, except that headspaces of Mason jars were replenished with ambient air after each removal of 20 mL headspace for gas analysis. Ambient air was also collected and analyzed for N_2O concentration and used to correct the changes of headspace N_2O concentration during the close period.

2.6. Data analysis

Repeated measures one-way ANOVA were performed with mixed-model (SAS 9.3, SAS Institute) to detect the main effect of N fertilization on emission rates and porewater N_2O and NH_4^+ . The N rate is the fixed-effect and field site is treated as random-effect block. Similar ANOVA analysis was also conducted to determine time effects on redox potential with field site treated as a random-effect block. Pairwise correlations between measured field variables were tested by the Multivariate function of JMP pro 10 (SAS Institute). For the lab incubation, repeated measures two-way ANOVA were performed to test the main effects of water level and soil type and their interactions. Significance level was set at $\alpha = 0.05$ for all the analysis.

3. Results

3.1. N_2O field emission rates

Negative emissions were repeatedly observed in all fields and not affected by N application at 80 kg N ha^{-1} (Fig. 1, Table 1). The magnitudes of net N_2O uptake increased after the onset of drainage, especially in 23% C field, where the most negative emission values were observed, ranging from -2 to $-199 \mu\text{g N}_2\text{O-N m}^{-2} \text{ h}^{-1}$ with a median of $-40 \mu\text{g N}_2\text{O-N m}^{-2} \text{ h}^{-1}$. When surface water was completely drained, N_2O emission rates started to increase and became more and more positive towards the end of the growing season in both 6% and 11% fields, while during the same period of time the rates were persistently negative in 23% C field.

3.2. Redox potentials

In general, Eh decreased gradually during the flooded period, but remained relatively stable prior to drainage (Fig. 1, Table 1). Water-level draw down had no effect on Eh if surface water still existed. However, like N₂O emission rates, Eh started to increase towards the end of the experiment in 6% and 11% C fields, right after the complete drainage of surface water, while it remained mostly less than -386 mV in 23% C field during the same period of time. Moreover, Eh was correlated ($r^2 = 0.40$, $p < 0.01$) to N₂O emission rates (Fig. 2).

3.3. Porewater N₂O concentrations

Overall, calculated *in situ* N₂O equilibrium concentrations in water ranged from 0.21 to 0.38 µg L⁻¹ with a median of 0.31 µg L⁻¹ (Fig. 3). They were not different between fields, but fluctuated slightly during the flooded period. Observed N₂O concentrations in porewater were not changed by N fertilization (Fig. 3, Table 1). In the 6% C field, porewater N₂O was highest at the beginning of the flooded period, averaging 20 µg L⁻¹, but decreased gradually during the first month flooding, after which it was repeatedly observed at concentrations below *in situ* the equilibrium concentration and ranged from < 0.10 to 2.53 µg L⁻¹ with a median of 0.32 µg L⁻¹. Similarly, in 11% and 23% C fields, porewater N₂O concentrations were also frequently less than their *in situ* equilibrium concentrations, ranging from < 0.10 to 2.03 µg L⁻¹ with a median of 0.13 µg L⁻¹ and from < 0.10 to 2.40 µg L⁻¹ with a median of 0.23 µg L⁻¹, respectively.

3.4. Porewater NH₄⁺ and NO₃⁻ concentrations

Nitrogen application resulted in higher NH₄⁺ concentrations in porewater (Fig. 4, Table 1). This effect diminished gradually in all fields and disappeared in one week in 6% and 11% C fields, but lasted for a month in 23% C fields following flooding. However, in all fields, porewater NH₄⁺ concentrations were mostly less than 0.5 mg L⁻¹ after June, while no NO₃⁻ was

detected ($< 0.01 \text{ mg L}^{-1}$) one week after the flooding. Like porewater N_2O ($r^2 = 0.65$, $p < 0.01$), NH_4^+ ($r^2 = 0.22$, $p < 0.01$) was correlated to N_2O emission rates.

3.5. Water level effect on N_2O emission rates

Emissions of N_2O were not different between 11% and 23% C soils and mostly less than $3 \text{ ng g}^{-1} \text{ d}^{-1}$ when water level was at 10 cm above the soil surface (Fig. 5). However, upon the onset of water manipulation, N_2O emissions varied across soils and time (Table 2). In the 11% C soils, adjusting water level from 10 to 5 cm did not affect the emission rates, but caused more than a 13 fold increase when surface water was completely removed (Fig. 5). In contrast, in the 23% C soils, adjusting water level to eight 5 or 0 cm resulted in negative N_2O emissions, ranging from -7 to -22 $\text{ng g}^{-1} \text{ d}^{-1}$. However, this effect disappeared in 3 days, after which N_2O emissions were minimal at 5 cm, but became positive at 0 cm.

4. Discussion

Negative emissions of N_2O have been frequently observed in various ecosystems with the maximal rates varying widely from -0.0014 to $-484 \text{ } \mu\text{g N}_2\text{O-N m}^{-2} \text{ h}^{-1}$, suggesting that soils can act as sinks for N_2O under many circumstances (Audet et al., 2014; Chapuis-Lardy et al., 2007; Majumdar, 2013). However, most studies remain focused on its production and net emission, attributing consumption mostly to variability or issues with detection limits (Chapuis-Lardy et al., 2007). In a recent survey of studies that reported N_2O uptake in soils of natural ecosystems, Schlesinger (2013) showed that most of the reported values range from 1 to $10 \text{ } \mu\text{g N}_2\text{O-N m}^{-2} \text{ h}^{-1}$ with a median of $4 \text{ } \mu\text{g N}_2\text{O-N m}^{-2} \text{ h}^{-1}$. Moreover, most of these observations were sporadic and

represented <10% of the measurements (Schlesinger, 2013). It was therefore concluded that projected global N₂O sinks of soils is not significant, although others believe that N₂O consumption can have a larger than realized role in regulating atmospheric N₂O dynamics (Chapuis-Lardy et al., 2007; Richardson et al., 2009; Ryden, 1981). In the present study, the most negative emission values observed occurred in soil with the highest SOC content, ranging from -2 to -199 µg N₂O-N m⁻² h⁻¹ with a median of -40 µg N₂O-N m⁻² h⁻¹. These values are substantially higher (10 times the median values) than those summarized by Schlesinger (2013). So is the frequency of occurrences (> 90% vs. < 10% of the measurements). Similar results were also found in our previous study in an adjacent field containing 15% SOC, in which observed negative emissions ranged from -17 to -102 with a median of -38 µg m⁻² h⁻¹, while observed frequency was 30% of the measurements (Morris et al., in preparation). The upper limit of the sink potential (-199 µg N₂O-N m⁻² h⁻¹ or 8.2 tones CO₂ eq ha⁻¹ yr⁻¹) equals up to 32% of the average CO₂ emissions from drained peat soils globally (Joosten, 2011). Our results suggest that high organic matter soils can serve as significant sinks for atmospheric N₂O, at least under some circumstances.

Many edaphic and environmental factors are able to influence N₂O reduction and their interactions make it difficult to identify a set of conditions conducive for *in situ* N₂O consumption (Flechard et al., 2005; Majumdar, 2013). Given that denitrification is the primary process responsible for both production and consumption of N₂O, one may expect its reduction more likely to occur in soils with high water content, low NO₃⁻ concentration, and high C availability. Indeed, Schlesinger (2013) reported that the highest negative N₂O emissions were mostly attributable to wetlands and peatlands, environments in which SOC provides a source of electrons to create reduced soil condition favorable for denitrification. It is therefore not

surprising that we observed significant negative N₂O emissions across the rewetted peatland sites (Fig. 1), where porewater inorganic NO₃⁻ was minimal. Moreover, redox potentials were found positively correlated with N₂O emission rates (Fig. 2). Redox potential is a variable describing the intensity of soil redox transformation, i.e. denitrification (Reddy and DeLaune, 2004). Previous studies have shown redox potential to affect N₂O emissions by regulating the balance between its production and consumption (Włodarczyk et al., 2005; Yu et al., 2008; Yu and Patrick, 2003).

In our study, relatively higher frequencies and magnitudes of negative N₂O emissions were observed during the draining period, especially in the 23% C site. During the draining period prior to enable rice harvesting, water levels drop considerably, resulting in a diminished water column close to the soil surface that maintains low Eh levels even after the fields were completely drained (Fig. 1). The diminished water column removed the diffusional constraint for N₂O, which is 10,000 times slower in water than in air, allowing greater diffusion into the soil. In the absence of NO₃⁻, porewater and atmospheric N₂O are the primary electron acceptors remaining for denitrification (Chapuis-Lardy et al., 2007; Reddy and DeLaune, 2004). Our results showed that porewater N₂O was frequently considerably below its expected air-water *in situ* equilibrium concentration in all fields (Fig. 3), showing the soils were an active sink for N₂O (Fig. 1). Our laboratory incubations confirmed that lowering the water level induced negative N₂O emissions in 23% C soils (Fig. 5). The negative emission rates observed in the present study were much lower than some similar laboratory studies (Hénault et al., 2001; Warneke et al., 2015), but comparable to others (Clough et al., 2006). However, this inducing effect was not observed in 11% C soils. It is likely that we were not able to establish redox potentials at levels allowing N₂O reduction to occur at rates that overwhelm its production during the laboratory

incubation (Włodarczyk et al., 2005; Yu and Patrick, 2003). Despite this uncertainty, our results suggested that N₂O consumption in reduced peat soils, where N₂O reduction potential is expectedly high (i.e. low redox potentials), is likely limited by the ease of diffusion of atmospheric N₂O in the water into soils.

Mitigating N₂O emissions by enhancing its reduction to N₂ has been proposed, but the exploration of management options to increase N₂O consumption has not been adequately pursued (Richardson et al., 2009). Peatlands are degraded ecosystems that are frequently managed for restoration efforts (Joosten, 2011), and therefore can be managed simultaneously to serve as atmospheric N₂O sinks, likely by tightly controlling water levels and hence the redox potentials as suggested by our results and others (Regina et al., 1999; Włodarczyk et al., 2005; Yu et al., 2008). Our results also show that optimum N inputs in agriculturally restored wetlands have little influence on N₂O consumption potential. Systematic studies that focus on microbial processes, biogeochemical reactions, and related environmental factors are needed to increase our understanding of the mechanisms controlling N₂O uptake in restored agricultural wetlands and beyond to develop effective mitigation strategies to increase N₂O sinks.

5. Summary

We explored the potential of restored organic soils to be N₂O sinks in three rewetted degraded peatland sites with contrasting SOC contents (6%, 11%, and 23%). The potential was negatively correlated to redox potentials and appeared to be limited by diffusional constraints of atmospheric N₂O into soils. Results suggested that these rewetted degraded peatlands can serve as significant N₂O sinks, at least temporally as influenced by water level management, with the maximal potential amounting to 8.2 tones CO₂ eq ha⁻¹ yr⁻¹, 32% of the average CO₂ emissions

304 from drained peat soils. It is seemingly possible that drained peatlands can be restored to
305 become N₂O sinks by controlling water level (i.e. enhancing the diffusion of atmospheric N₂O
306 into soils) and redox potentials (i.e. promote N₂O reduction). However, more research is
307 required to better understand the mechanisms underlying N₂O consumption and their responses
308 to environmental and climate variations, such that effective mitigation strategies can be informed
309 and developed.

310

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419 **Figure Legend:**

420 Fig.1. N₂O emissions, redox potentials, and surface water level in rewetted drained peatlands
421 with contrasting soil organic carbon (6%, 11%, 23%) during rice growing season. Bars represent
422 standard error of means (n = 4), while 0 and 80 in the legend is fertilization rate at 0 and 80 kg N
423 ha⁻¹ as urea. The arrows show when drainage started for harvest.

424 Fig.2. Relationships between N₂O emission rates and redox potentials. The rates are averages
425 across the nitrogen treatment.

426 Fig. 3. Porewater N₂O concentrations in three rewetted drained peatlands used for rice
427 cultivation. Bars represent standard error of means (n = 4), while 0 and 80 in the legend is
428 fertilization rate at 0 and 80 kg N ha⁻¹ as urea. The “6%”, “11%”, and “23%” in the legend
429 represent the SOC content. Dash lines indicate the calculated *in situ* N₂O equilibrium
430 concentrations in surface water.

431 Fig. 4. Porewater NH₄⁺ concentrations in three rewetted drained peatlands used for rice
432 cultivation. Bars represent standard error of means (n = 4), while 0 and 80 in the legend is
433 fertilization rate at 0 and 80 kg N ha⁻¹ as urea. The “6%”, “11%”, and “23%” in the legend
434 represent the SOC content.

435 Fig. 5. N₂O emission rates in 11% and 23% C soils incubated at three water levels (0, 5, 10 cm
436 above soil surface). Bars represent standard error of means (n = 4). Shade area indicates the
437 period, during which all samples were incubated at 10cm.

Table 1. Results of ANOVA analysis of nitrogen fertilization (N rate) effect on N₂O emission rates and the concentrations of porewater N₂O and NH₄⁺, and time effect on redox potential.

Variables	Source	df	F	<i>p</i>
N ₂ O emission	N rate	1	3.07	0.08
	Time	30	4.91	< 0.0001
	N rate * Time	30	0.67	0.91
Porewater N ₂ O	N rate	1	0.26	0.6122
	Time	17	7.64	< 0.0001
	N rate * Time	17	0.24	0.9994
Porewater NH ₄ ⁺	N rate	1	29.26	< 0.0001
	Time	16	12.87	< 0.0001
	N rate * Time	16	5.03	< 0.0001
Redox potential	Time	23	4.98	< 0.0001

Table 2. Results of ANOVA analysis of water level, soil, and time effects on N₂O emission rates.

Source	df	F	<i>p</i>
Water level	2	37.58	< 0.0001
Soil	1	60.10	< 0.0001
Time	3	10.54	< 0.0001
Water level * Soil	2	36.66	< 0.0001
Water level * Time	6	7.41	< 0.0001
Soil * Time	3	1.39	0.262
Water level*Soil * Time	6	1.18	0.3364

Figure

Fig. 1

Fig. 2

Fig. 3

Fig. 4

Fig. 5