TITLE

A Millimeter-Scale Change in Leaf Litter Placement Within Soil-Water Interfaces Alters Carbon Dioxide and Methane Emission

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STATEMENT

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Highlights

- The placement of leaf litter at SWI significantly impacts on CO_2 and CH_4 emissions from wetland soils.
- A thin soil layer covering leaf litter led to doubled soil CH_4 emissions and a 25% reduction in CO_2 emissions, and an overall lower GWP20, compared to leaf litter without soil covering.
- The presence or absence of a thin soil covering on leaf litter triggers distinct biogeochemical processes, influencing CO_2 and CH_4 emissions
- Geographic patterns, particularly the spatial distribution of plant litters along the soil redox gradient, are crucial factors in controlling carbon fate

Title

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Abstract

Flooded soils play a critical role in global carbon cycling, serving as significant reservoirs of soil organic carbon and sources of carbon emissions. Leaf litter, particularly from local vegetation, is a major contributor to soil organic carbon formation in these ecosystems, with its decomposition driving the production of carbon dioxide and methane. While numerous studies have investigated the factors influencing leaf litter decomposition and the associated greenhouse gas emissions, the impact of millimeter-scale variations in leaf litter placement

within the soil-water interface (SWI) remains underexplored. This study hypothesizes that such minor changes in burial depth can significantly alter the emission patterns of CO_2 and CH_4. To test this, a microcosm experiment was conducted, monitoring gas fluxes and profiles of physicochemical properties in treatments with leaf litter closely placed at two depths within the SWI. Results revealed that a sub-centimeter difference in leaf litter placement could lead to a substantial shift in CO_2 and CH_4 emissions, with important implications for modeling wetland carbon dynamics and predicting their climate impact. These findings underscore the sensitivity of greenhouse gas emissions to small-scale environmental variations, highlighting the need for more precise models in estimating wetland contributions to global carbon fluxes.

1. Introduction

Wetlands are important reservoirs of soil organic carbon as well as sources of carbon emission (Xiao et al., 2019). While wetlands cover only 5-8% of the global terrestrial area, wetland soils contain about 20-25% of global soil carbon stocks (Mitra, 2005; Mitsch et al., 2012; Nahlik and Fennessy, 2016). The plant litters from local vegetation contribute significantly to the soil organic carbon (SOC) formation (Cao et al., 2020; Scharlemann et al., 2014) while the leaf litter have greater impacts on SOC accumulation than stem and root litter in forested wetlands (Ding et al., 2023; Ji et al., 2020). Leaf litter is an important carbon source for forested wetland ecosystems, and its decomposition process affects the entire ecosystem (Stoler and Relyea, 2010; Stoler and Relyea, 2013; Watkins et al., 2011). The carbon and nutrients released from leaf litter fuels the respiration of microbes and methanogenesis in wetlands, which converts the labile carbon released from leaf litter into carbon dioxide and methane (Corteselli et al., 2017; Dušek et al., 2020). The linkage between leaf litter decomposition and greenhouse gases emission have been extensively studied for wetlands. The factors like leaf species (Yakimovich et al., 2018; Yavitt and Williams, 2015), hydroperiod (Batson et al., 2015; Battle and Golladay, 2001; Neiff et al., 2006; Peng et al., 2022), temperature (Bridgham et al., 2006; Ma et al., 2022), oxygen availability (Dušek et al., 2020; Neckles and Neill, 1994), water chemistry (Chiapponi et al., 2024) are found to collectively influence decomposition rate of leaf litter in various types of wetlands, therefore

46 controlling the CO_2 and CH_4 emission. However, to date the positioning of fresh leaf litter has been rarely
47 discussed and knowledge of its impact on wetland CO_2 and CH_4 emission remains scarce.

48 The placement of leaf litter in soil significantly influences decomposition rates and the emission of CO_2 and CH_4.
49 Factors such as burial depth, soil-litter mixing, and the removal of aboveground litter affect leaf litter
50 processing in wetlands (Gong et al., 2020; Hu et al., 2021; Ma et al., 2022). Decomposition is most rapid near
51 the wetland surface, where freshly deposited litter and newly synthesized labile organic matter are abundant
52 (Schiff et al., 1998). This surface oxic layer is also the primary site for CO_2 and CH_4 production (Kayranli et al.,
53 2009). However, the variable positions of leaf litter and dynamic conditions in natural wetlands create
54 challenges in accurately determining CO_2 and CH_4 emissions from litter decomposition at different locations
55 along the soil-water interface (SWI).

56 The ratio of CO_2 and CH_4 emitted from wetlands during leaf litter decomposition are primarily influenced by
57 redox condition (Dušek et al., 2020; Galera et al., 2023; Nilsson and Öquist, 2009). CO_2 emission dominates the
58 wetland emission in dry period and aerobic layer of wetland soils whereas CH_4 emission becomes more
59 significant in the oxygen-depleted cases like high water table and anaerobic soil layer (Bridgham et al., 2013;
60 Ding et al., 2002; Liu et al., 2021). Nonetheless, the transition from oxic to anoxic zone usually occurs over a
61 depth ranging from several millimeters to centimeters below SWIs (Glud et al., 2007; Roy et al., 2004). The
62 sharp redox gradient suggests that a minor displacement of leaf litter at the SWIs may change the patterns of
63 CO_2 and CH_4 fluxes a lot, but this effect has not to the best of the authors knowledge been previously described.
64 Hereby, we propose that the partitioning of CO_2 and CH_4 is sensitive to a millimeter scale variation in leaf litter
65 burial depth across the SWIs.

66 This study hypothesizes that the positioning of leaf litter in the SWI significantly affects the fate of dissolved
67 organic carbon. We think that even slight variations in burial depth, less than ten millimeters, can alter CO_2 and
68 CH_4 emission patterns from wetlands, complicating model predictions. To investigate this, we conducted a
69 microcosm experiment, monitoring CO_2 and CH_4 emissions after adding leaf litter at two closely positioned
70 locations. Concurrently, we measured the profiles of dissolved oxygen (DO), redox potential (Eh), fluorescent

dissolved organic matter (fDOM) and microbial communities across different treatments. The results could improve the accuracy of models estimating CO_2 and CH_4 emissions from wetlands.

2. Materials and methods

2.1. Collection of soil, water, and leaf litter

A rice paddy soil was collected from a farmland in Kunshan, Jiangsu Province, China. The soil was air-dried under room temperature. The gravel and plant litter were removed from the soil with a one-millimeter mesh sieve. The surface water was collected from a local pond. The water was filtered with 0.45 μm filter cartridge and then autoclaved. The sterile surface water was stored in 1-L Schott bottle < 4˚C before use. The fresh leaf litter from *Liriodendron Chinense* was collected in campus and stored in a sterile plastic bag before use.

2.2. Microcosm experiment

Open-top microcosms were constructed using collected paddy soil and surface water. Each microcosm consisted of a soil column measuring 60 mm in height and a water column of 30 mm. These structures were then incubated at room temperature (~20 ˚C). To offset evaporation, ultrapure water was added daily throughout the incubation and experimental period.

To test the hypothesis, three experimental groups were established. The first group received dry uncovered (denoted as LU), with the leaf litter placed at the soil-water interface (SWI). The second group was treated with covered leaf litter (LC), where the leaf litter was covered with a 10-mm-thick soil layer. A control group with leaf litter addition (NL) was included. Each group was replicated twice. For both the LU and LC groups, 0.8 g of dry leaf litter were added to the water column, and after ~4 h, the leaf litter settled at the SWI. In the LC group, a stainless steel spatula was used to gently till the topsoil, covering the leaf litter with a roughly 10-mm thick layer.

Gas fluxes of carbon dioxide and methane were monitored at 0, 5, and 15 h post-leaf litter addition, and

subsequently, measurements were continued on a daily basis. DO, Eh, fluorescent fDOM, and nitrogen species were analyzed on days 3, 14, and prior to the treatment. The porewater samples were collected using the Integrated Porwater Injection (IPI) samplers with a depth resolution of 2 mm (Yuan et al., 2019; Zhang et al., 2023).

2.3. Measurement of CO_2 and CH_4 fluxes

The real time concentration of carbon dioxide and methane was monitored using the LI-7810 trace gas analyzer (LI-COR, USA) and the closed dynamic chamber on daily basis. Every measurement takes 90 s, and the readings are fitted with a linear line. The slope of linear fitting line is the approximate emission rate and used to calculate the gas fluxes. A more detailed description of the chamber measurement is provided in the supplementary information. The global warming potential over a 20-year timescale (GWP20) is calculated after the AR6, with a factor of 81 chosen for biogenic methane (IPCC, 2021).

2.4. fDOM characterization

The excitation-emission matrix (EEM) of the sterile and filtered surface water and leaf litter leachate was characterized with Cary Eclipse fluorescence spectrometer (Agilent Technologies, USA). The EEM program parameters are as follows: scanned excitation from 200 to 450 nm every 5 nm at a rate of 1200 nm/min; detected emission from 300 to 550 nm with a slit width of 10 nm; the voltage of photomultiplier at 700 V.

The fluorescent components of dissolved organic matters in overlying water and porewater samples was analyzed using the 1260 Infinity high performance liquid chromatography (HPLC) system (Agilent Technologies, USA) equipped with a fluorescence detector (FLD) and a diode array detector (DAD). An autosampler injected a 5 µL aliquot of each sample. A gradient elution program of acetonitrile and 10 mM sodium acetate aqueous buffer is modified from a previous work (Li et al., 2013). Based on the EEM results, the parameters of FLD and DAD were optimized as follows: for FLD excitation at 207, 230, 315 nm, emission at zero order; for DAD absorption at 254 nm as proxy of aromatic compounds.

2.5. *In situ* measurement of DO and Eh

The dissolved oxygen and redox potential were profiled with commercially available (EasySensor, China) and self-made microelectrodes. The microelectrodes are mounted on the coordinate measuring system (see Section 2.4) and connected to CHI 1040 potentiostat (Chenhua Instrument, China). Each profile starts at 20 mm above the soil-water interface and ends at forty-millimeter below the interface. The readings were logged every 2 mm with a 30 s cycle.

2.6. Nucleic acid extraction

By end of the incubation, the water layer was collected into 50 mL sterile centrifuge tubes. The soil cores were collected in headless 5 mL syringe using the core sampling tool (see Section 2.5) and then sealed with parafilm. Both water samples and soil cores were frozen at –40 ˚C immediately after collection. For each replicate, 150 mm water samples were thawed and filtered through the sterile 0.22 μm mixed cellulose esters membranes. The soil cores were segmented in 3 mm thick aliquot. Each soil aliquot was directly transferred into a 2 mm cryotube and weighed. The DNA from water and soil were extracted with the FastDNA Spin Kit (MP Biomedical, USA). The size of genomic DNA was verified using agarose gel electrophoresis. The DNA quality and concentration was determined using Nanodrop and Qubit (Thermo Fisher Scientific, USA).

2.7. qPCR

Each PCR reaction was carried out in 20.0 μL mixture and contained 1.0 μL DNA template, 10.0 μL TB Green Premix Ex Taq (Tli RNaseH Plus) (Takara Bio, RR420A), 1.0 μL of each primer (10 μM), and nuclease-free water. The primer pairs were chosen for the V4 region of 16S rRNA gene (515F: 5’GTGCCAGCMGCCGCGGTAA3’, 806R: 5’GGACTACHVGGGTWTCTAAT3’). The qPCR reactions were performed in triplicate under thermal cycler conditions of 15 min at 95 °C, and 39 cycles of 10 s at 95 °C, 30 s at 55 °C and 32 s at 72 °C in a 96-well format LightCycler 480 Real-Time PCR System (Roche). All results were normalized and calculated using the ΔCt method.

2.8. Amplicon sequencing and analysis

The V3-V4 region of 16S ribosomal RNA gene was amplified using the primer pairs 341F (5'CCTACGGGAGGCAGCAG3') and 806R (5'GGACTACHVGGGTWTCTAAT3'). The length of amplified region is roughly 470 bp. The sequences were obtained on the NovaSeq 6000 platform (Illumina, USA) with a 2x250 bp paired-end run. The raw sequence files will be later deposited in an online archive.

The raw sequencing data was denoised and merged using the DADA2 pipeline (Callahan et al., 2016). The taxonomy of the amplicon sequence variants was assigned using QIIME2 and the latest SILVA database (Bolyen et al., 2019; Quast et al., 2013). The metabolic functional potential of the microbial communities was predicted using PICRUSt2 pipeline (Barbera et al., 2019; Douglas et al., 2020; Louca and Doebeli, 2018; Ye and Doak, 2009).

3. Results

3.1. Temporal variation in CH_4 and CO_2 fluxes

The CH_4 fluxes from the flooded soils were monitored from the treatments with or without the leaf litter amendment (Fig. 1a). Clear distinctions in CH_4 flux patterns were observed between the LU and LC groups, while the NL group released nearly no CH_4. In the LC group, CH_4 fluxes increased immediately after amendment and peaked at an emission rate of 0.52-0.76 $gC/m^2/y$ on day 5. These fluxes remained around 0.6 $gC/m^2/y$ until day 8 and then diminished to almost zero by day 21. In contrast, the CH_4 flux curve for the LU group exhibited a more gradual increase, reaching a maximum intensity of 0.2 $gC/m^2/y$ on day 8, approximately one-fourth of the LC group's peak.

The CO_2 fluxes for the LU and LC groups exhibited similar trends (Fig. 1b). Both groups experienced an initial peak in CO_2 fluxes, reaching levels between 157 and 221 $gC/m^2/y$ in day 1. From day 2 to day 11, the intensity of CO_2 fluxes declined and fluctuated between 101 and 192 $gC/m^2/y$, followed by a subsequent gradual decrease. Eventually, CO_2 fluxes leveled off within a range of 71 to 120 $gC/m^2/y$, remaining stable until the 24-

day incubation period terminated. During this stage, the CO_2 fluxes in the LU group were, on average, approximately 20% greater than those in the LC group. These distinct patterns in CH_4 fluxes and similar trends in CO_2 fluxes were previously confirmed by replicated findings from an independent experiment (Fig. S1).

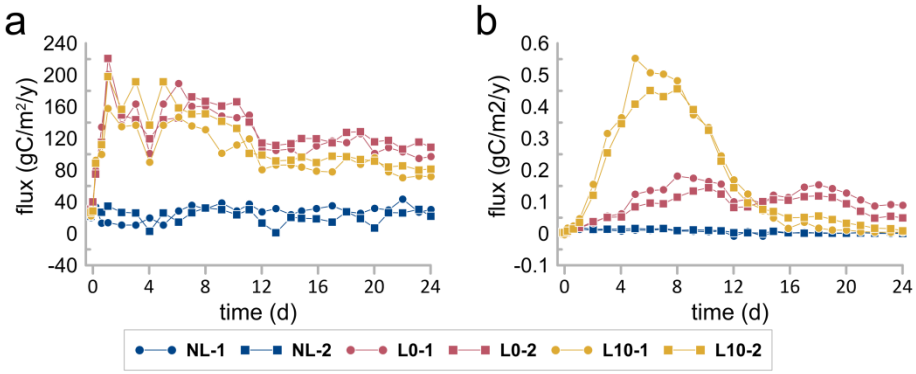

Fig. 1. (a) Temporal variations in CO_2 fluxes. (b) Temporal variations in CH_4 fluxes. The black arrows indicate the points of profiling DO, Eh, fDOM in time. NL = control groups; LU = leaf litter placed at the soil-water interface; LC = leaf litter was covered with a 10-mm-thick soil layer.

The overall cumulative CH_4 and CO_2 emissions are presented in Fig. 2. The LU and LC groups emitted 42.5 µgC and 115.3 µgC of CH_4, respectively (Fig. 2b). The LC group's CH_4 emissions were more than double those of the LU group. Notably, the NL group did not emit any CH_4, suggesting that CH_4 production is surely attributed to the litter amendment. CO_2 emissions from the LU and LC groups were 103 mgC and 77 mgC, respectively, by the end of the experiment (Fig. 3a). One centimeter repositioning of leaf litter resulted in a 25% reduction in overall CO_2 emissions, reflecting the lower CO_2 fluxes during the late stage of incubation in the LC group. The potential impact on climate change was assessed using the GWP20 metric (Fig. 3c) The GWP20 values for the LU and LC groups were 323.3 mgCO_2e and 280.7 mgCO_2e, respectively. Despite the doubling of CH_4 emissions due to leaf litter perturbation, the GWP20 of the LC group was still 13.2% lower than that of the LU group, due to the reduced CO_2 release.

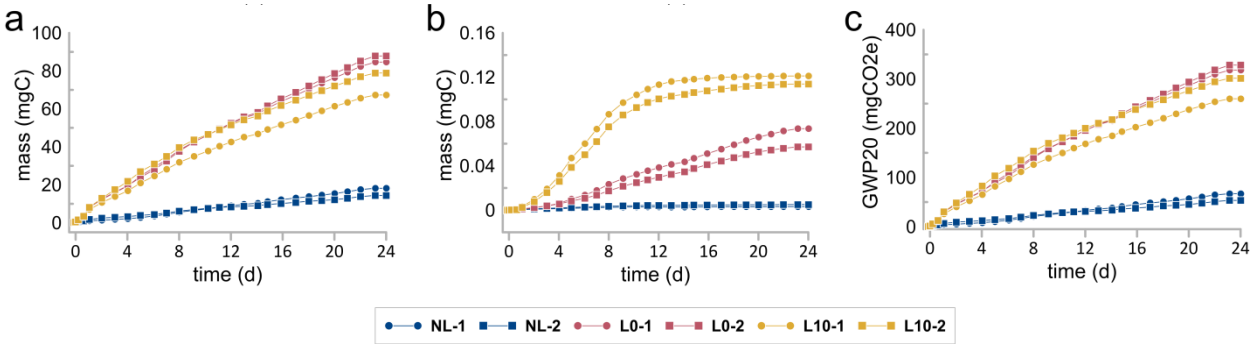

Fig. 2. (a) Cumulative carbon emission in the form of CO_2. (b) Cumulative carbon emission in the form of CH_4. (c) The combined GWP20. NL = control groups; LU = leaf litter placed at the soil-water interface; LC = leaf litter was covered with a 10-mm-thick soil layer.

3.2. Depth profiles of DO and Eh

The depth profiles of DO were monitored on days 0, 3, and 14 (Fig. 3a-c). Prior to the leaf litter amendment, all groups exhibited consistent vertical patterns of DO (Fig. 3a). The DO concentration ranged from 3.1 to 3.8 mg/L in the oxic water layer, gradually declining at the soil-water interfaces. Aerobic conditions were noted at 20 mm below the soil-water interface (SWI). On day 3, the vertical patterns of DO differentiate in both the LU and LC groups, while the DO profiles in the NL group remained relatively unchanged. In both treatment groups, the DO concentration in the overlying water decreased, with the LC group exhibiting a slightly higher concentration than the LU group (Fig. 2b). Additionally, the gradient in DO profiles became more pronounced, with an elevated front of downward-going DO flux. The thickness of the redox transition zone was reduced to 5 mm and 10 mm in the LU and LC groups, respectively. Similar vertical patterns of DO were observed on day 14 (Fig. 3c), indicating the persistence of the treatment's effects.

The depth profiles of Eh were also monitored on days 0, 3, and 14 (Fig. 3d-f). Before treatment, all groups displayed similar vertical Eh patterns (Fig. 3d); the Eh values remained constant at approximately 220 mV in the oxic water layer and the top soil layer, and drop roughly -80 mV at 40 mm depth. By day 3, both treatment groups exhibited reduced Eh at the water and top soil layers. In the LU group, there was a slight decrease of around 30 mV at the interface where the leaf litter was placed, as well as in the upper soil layer. Similar Eh

drop at where the leaf litter was placed was also observed in the LC group. On day 14, further decreases in Eh values were noted. The LU group experienced an enhanced decline of up to 100 mV in the topsoil layer relative to the NL group. In the LC group, the maximum decrease of about 150 mV occurred at 15 mm below the SWI. However, the uniformity of the Eh profiles on day 14 within the same group was not as consistent as at earlier time points. This variability suggests that the heterogeneity developing in the surface soil layer due to the uneven distribution of degraded leaf litter may have influenced the results. Overall, the Eh profiles indicate that the surface soil layer quickly became reducing following the addition of leaf litter, with the most pronounced effects occurring near the litter material.

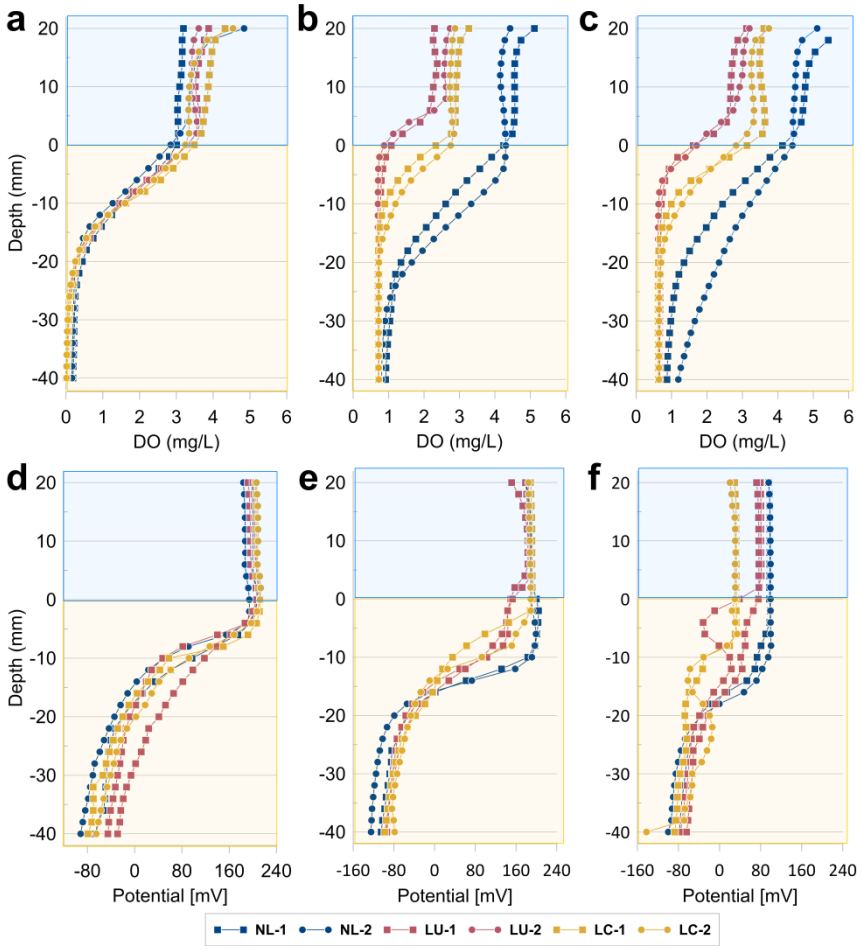

Fig. 3. Spatial distribution of DO and Eh across the soil-water interfaces. Depth profiles of DO on day 0 (a), day 3 (b), and day 14 (c). Depth profiles of Eh on day 0 (d), day 3 (e), and day 14 (f). The blue and yellow background denotes the water column and soil column, respectively. NL = control groups; LU = leaf litter placed at the soil-

water interface; LC = leaf litter was covered with a 10-mm-thick soil layer.

3.3. Depth profiles of fDOM

Fluorescent dissolved organic matters (fDOM) levels were measured to represent the degraded products from leave litter. The depth profiles of fDOM were examined on days 0, 3, and 14 (Fig. 4). Prior to treatment, the background fluorescence was generally low, with a few exceptions noted in the soil layer deeper than 10 mm below the sediment-water interfaces (SWIs) (Fig. 4a). Following the addition of leaf litter, there was a rapid release of fDOM (Fig. 4b) at day 3. In the LU group, fDOM levels in the overlying water rose to between 50 and 60 arbitrary units (a.u.), while the LC group (containing a water overlay, due to the leaf litter settling at the SWI) experienced an increase of approximately 37 a.u. The variation in fDOM signals suggests that the labile DOM was rapidly diffused from where leaf litter buried to the overlaying top water and the soil layer. By day 14, the fDOM signal intensity had diminished in both treatment groups where the leaf litter was at the SWI (LO in Fig. 4c) or buried by 10 mm (L10 in Fig. 4c), although it remained significantly higher than that of the control group. The fDOM signals in the LU and LC groups persisted at around 25 a.u. and 20 a.u., respectively, slightly higher than the DOM level in the LU group.

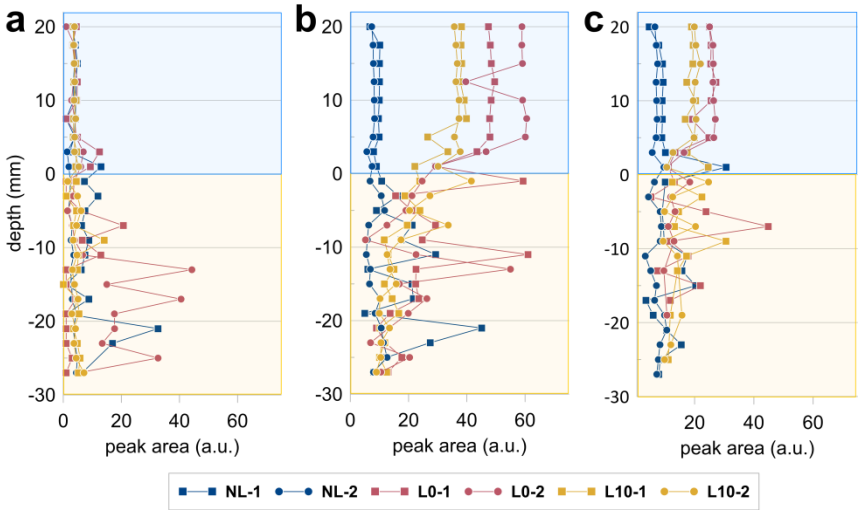

Fig. 4. Spatial distribution of fDOM across the soil-water interfaces. Depth profiles of fDOM on day 0 (a), day 3 (b), and day 14 (c). NL = control groups; L0 = leaf litter placed at the soil-water interface; L10 = leaf litter was covered with a 10-mm-thick soil layer.

3.4 Abundance and distribution of microbial communities

The abundance of microbial organisms along the SWI were measured using 16S rRNA sequencing with quantitative PCR and displayed in Fig. 5.

The addition of leaf litter stimulated the growth of microbial organisms in the water layer and surrounding soil. In the LU group, the abundance of 16S rRNA gene copies was significantly higher at the top of SWI, reaching ~$1.4x10^7$ copies per gram of soil. This abundance then decreased to levels comparable to the control group at a depth of about 7.5 mm below the SWI. In the LC group, microbial growth was also enhanced at approximately 13.5 mm below the SWI, where the number of 16S rRNA gene copies was nearly as abundant as at the surface water.

The abundance of the acetoclastic methanogenesis pathway in both the water column and soil column was depicted in Fig. 5b & 5e, respectively. In the overlying water, the acetoclastic pathway was only detected in the LU group. In the soil, the LU group and the control group showed similar trends in the abundance of the acetoclastic pathway. However, the LC group demonstrated a notable increase in the abundance of this pathway between 10 to 15 mm below the SWI. The abundances of the hydrogenotrophic methanogenesis pathway were consistently at least one order of magnitude lower compared to the acetoclastic methanogenesis pathway in both compartments (Fig. 5c & 5f).

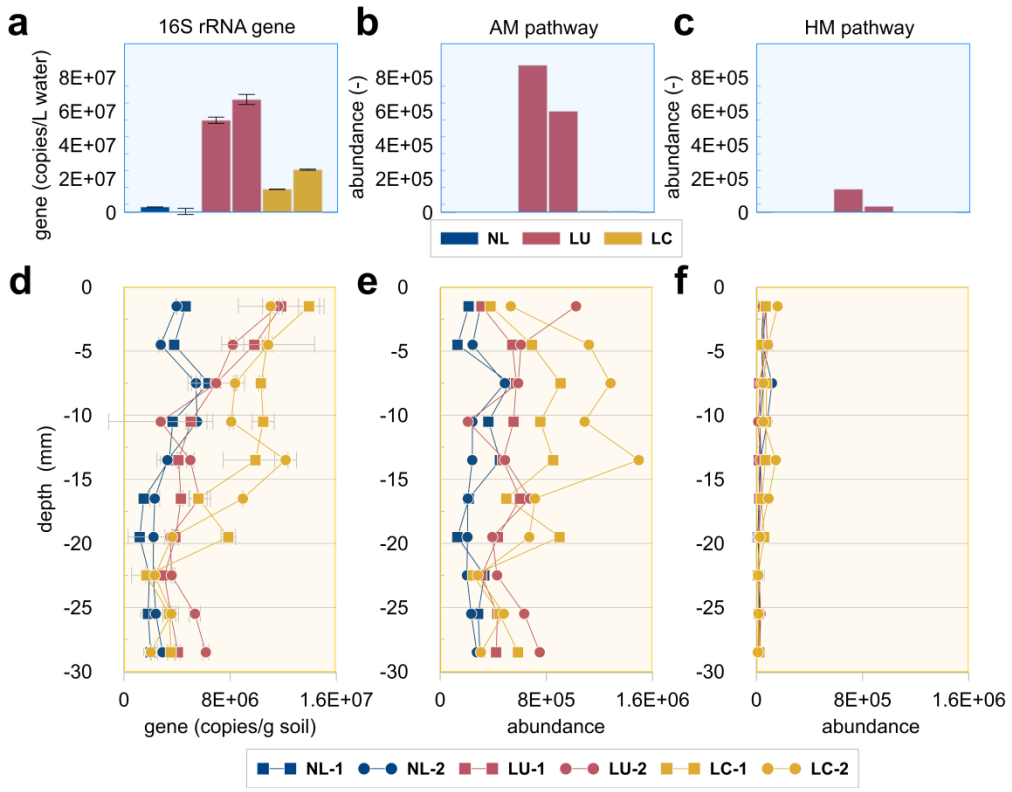

Fig. 5. Spatial patterns of microbial communities. 16S rRNA gene copies in water column (a) and soil column (d) from technical replicates *n*=3; averages and standard deviation error bars are shown. The predicted pathway abundances of acetoclastic methanogenesis in water column (b) and soil column (e). Predicted pathway abundance of hydrogenotrophic methanogenesis in water column (c) and soil column (f). AM denotes acetoclastic methanogenesis pathway. HM denotes hydrogenotrophic methanogenesis pathway. NL = control groups; LU = leaf litter placed at the soil-water interface; LC = leaf litter was covered with a 10-mm-thick soil layer.

4. Discussion

4.1. Distinct patterns in CH_4 and CO_2 emissions

Our findings show that the addition of leaf litter significantly induces CH_4 and CO_2 emissions, but the time-dependent variations are distinct with the placement of leaf litter. When leaf litter was placed on the top of SWI, the leaf litter was rapid mineralized, resulting in high CO_2 emissions. At the same time, the CH_4 emission was enhanced too due to two processes, first, the methane oxidation is inhibited because the diffusion of

oxygen to deep soils is blocked (Elberling et al., 2011), as indicated of the Eh profile; second, the labile organic matter in leaf litter diffused to the deep soils and fueled the acetoclastic methanogens (Fu et al., 2018; Ji et al., 2018), which is supported by the functional analysis and fDOM profile. Thus, a prolonged methane emission peak was observed in the LU treatment.

When the leaf litter was covered by a thin layer of soil, the soil layer becomes an efficient barrier to slow down the diffusion of oxygen in the overlying water to the leaf litter (Lorke et al., 2003; Gundersen and Jørgensen, 1990). At the same time, the labile organic matter released during leaf litter degradation diffused upward to the overlying water layer. Therefore, less but significant CO_2 emission was seen. The addition of leaf litter stimulated the growth of anaerobic microbial communities, including the methanogens (Nottingham et al., 2009). The methane emission was high but short-lived, compared to the treatment with leaf litter on the top, which implies that the supply of organic substrate to the methanogens is hindered in the anaerobic condition. The prolonged or short-lived methane emission peaks were also observed in other studies. For example, Gong *et al*., (2020) investigated the response of methane emissions to litter input in a marsh in two years, and found a prolonged methane peak in the first year, and a short-lived peak in the second year. The underlying mechanism of methane emission patterns were usually ignored by other researchers. Our study indicated that the patterns may be explained by the slight change of litter position during the experiments.

Our findings suggest that minor disturbances, on the scale of millimeters, to the place of leaf litter can substantially alter CH_4 and CO_2 emissions in systems such as wetlands and rice paddies. This raises the possibility that overlooking the effects of hydrological processes and bioturbation could lead to amplification of uncertainties in wetland methane budgets and their feedback mechanisms with climate change (Jones et al., 2020).

4.2. Implications for wetland carbon budget

Although the accumulated CH_4 emission doubled when the leaf litter move from the top of the SWI to 10 mm below, the GWP20 is lower in the treatment with buried leaf litter, since much less CO_2 was emitted. This

finding showed the importance of whole-life analysis of the fate of plant litters.

Farmers often supplement the soil with organic matter, mainly straw. This practice enhances soil fertility by increasing the reservoir of organic matter and essential nutrients for crop growth. However, the strategy of straw incorporation in paddy systems has sparked debates due to the possible amplification of methane emissions ensuing from the decay of straw under the anaerobic conditions prevalent in waterlogged rice fields. Based on the findings, we conclude that burying of plant litters, even with a thin soil layer, can mitigate its impact on greenhouse emission, at the same time, store more carbon in the anoxic environments. Many studies showed litter addition, for example, straw returning, induced strong methane emission using the soils without straw as the control (Ye et al., 2015; Yuan et al., 2018; Jiang et al., 2019). A more reasonable way to evaluate the consequence of straw returning is to consider the fate of straws, as most of the straws were eventually transformed to CO_2 or other greenhouse gases through burning, feeding to animals, and/or composting.

Our findings showed that small-scale spatial alternation influences the fate of organic carbon in wetlands. The existing modeling frameworks usually considered the microbial processes (e.g., methanogens and methane oxidizers), and the redox-active compounds (e.g., nitrate, sulfate, and organic substrates), in the environment. Here, we provided evidence showing the geographic patterns, especially the vertical distribution of plant litters along the redox gradient at SWI, is another vital factors controlling carbon fate. The trade-off between CO_2 and CH_4 is very sensitive for the leaf litter introduced to SWI. When moving along the depth, leaf litter produce less CO_2 emissions and more CH_4, and the ratio may change rapidly at the millimeter scale. Our experiment revealed that rearranging leaf litter near the soil-water interface can significantly alter the balance of CO_2 and CH_4 emissions, as well as the GWP20. Therefore, we recommend accounting for the effect of leaf litter placement on carbon dynamics in modeling scenarios.

5. Conclusion

This study provides compelling evidence that millimeter-scale variations in the placement of leaf litter within

the SWI of wetlands can significantly impact greenhouse gas emissions, particularly CO_2 and CH_4. Our microcosm experiments demonstrated that even minor changes in burial depth can alter the pathways and rates of organic matter decomposition, leading to considerable differences in gas fluxes. These findings suggest that existing models of wetland carbon dynamics, which often assume uniform conditions at the SWI, may overlook critical small-scale heterogeneities that influence carbon cycling and greenhouse gas emissions. The implications of this research extend to both the scientific understanding of wetland ecosystems and the strategies employed to mitigate climate change. As wetlands are recognized as key players in global carbon storage and emissions, accounting for micro-scale variations in leaf litter decomposition could improve the accuracy of carbon flux predictions and the effectiveness of wetland management practices. Furthermore, this study highlights the need for high-resolution environmental data and more nuanced modeling approaches that consider the complexity of natural systems. Future research should focus on exploring the mechanisms driving the observed effects of leaf litter placement on gas emissions and extending these findings to different types of wetlands and environmental conditions. By deepening our understanding of these processes, we can better predict the role of wetlands in the global carbon cycle and develop more informed conservation and management strategies to harness their potential in mitigating climate change.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Declaration of interests

☒The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

☐The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: