



Effects of fire on dissolved carbon and greenhouse gas production rates and their temperature sensitivity in subalpine wetlands

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Received: 15 April 2025 / Accepted: 19 January 2026
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Abstract Wetlands comprise only 5–8% of land surface but hold 20–30% of estimated soil carbon globally. However, wetlands are also significant sources of greenhouse gases such as methane (CH₄) and nitrous oxide (N₂O). Disturbances such as wildfires can alter the balance between carbon storage and greenhouse gas production in wetland systems; therefore, it is

crucial to understand wetland response and recovery after wildfires. While wildfires are known to significantly impact ecosystem function through changes in soil properties, nutrient cycling, and hydrology, subalpine wetlands remain understudied, with the exception of organic matter-rich peatlands. Though temperature fluctuations regulate microbial processes, it is unclear how seasonal temperature patterns influence wildfire effects. We investigated these interactions in burned subalpine wetland soils in the Medicine Bow National Forest, Wyoming, USA, 1 year after the 2020 Mullen fire. We measured potential rates of carbon dioxide (CO₂), CH₄, N₂O, and DOC production using slurry experiments and flow-through experiments with soil collected from two depths (0–2 and 15–17 cm). Both experiments were conducted at local minimum, mean, and maximum July air temperatures (9, 18, and 27 °C). In situ porewater measurements showed that burned wetland areas had higher dissolved organic carbon (84–105 mg/L vs. 65 mg/L), sulfate (2.8–3.3 mg/L vs. 1.4 mg/L), and nitrate concentrations (1.3–1.9 mg/L vs. 0.5 mg/L) compared to unburned wetland areas, particularly in shallow depths (0–12 cm). Slurry experiments revealed approximately 1.3 times higher potential CO₂ production rates and fivefold higher N₂O production rates, but 2.9 times lower CH₄ production rates in burned compared to unburned wetland soils. Flow-through reactor experiments corroborated these findings, showing higher DOC (2–4×), Fe(II) (1.5–2×), and DIC (1.3–1.8×) potential production rates but

Supplementary Information The online version contains supplementary material available at <https://doi.org/10.1007/s10533-026-01307-1>.

Responsible Editor: Brian Branfireun.

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lower CH₄ production rates (0.4–0.8×) in burned wetland soils. The suppression of methanogenesis and enhancement of Fe(III) reduction in these soils suggest altered redox conditions, potentially resulting from changes in organic matter composition, soil exposure, and hydrology following fire. Temperature sensitivity analysis revealed higher Q₁₀ values for Fe(II) production in burned wetland soils (1.60–2.90 vs. 1.57), indicating that fires enhance the temperature response of Fe(III) reduction pathways. These findings provide insights into post-fire biogeochemistry of sensitive subalpine wetland systems, with implications for the global carbon cycle and drinking water quality.

Keywords Wetland biogeochemistry · Post-fire water quality · Greenhouse gas emissions · Reductive Fe(III) dissolution · Wildfire inputs · Temperature sensitivity

Introduction

Wildfires are natural disturbances that play important roles in ecosystem functioning (Pausas and Keeley 2019). While fires are often associated with upland ecosystems, their impacts extend to adjacent wetland systems, which are critical yet vulnerable components of the global carbon cycle. Wetlands comprise only 5–8% of the Earth's land surface but store an estimated 20–30% of global soil carbon, making them crucial sinks in the global carbon cycle (Bridgman et al. 2006; Nahlik and Fennessy 2016). However, wetlands are also significant sources of greenhouse gases (GHGs), particularly methane (CH₄) and nitrous oxide (N₂O) and are vulnerable to disturbances. Despite this significance, their response to disturbances such as wildfires remains poorly understood, especially in mountainous regions, since most previous research studying this interaction has been conducted in organic matter-rich peatlands (Smith et al. 2001; Nakano et al. 2006; Medvedeff et al. 2013). Fires can influence wetland systems through multiple pathways, including: the formation of ash and pyrogenic C, defined as thermally altered chemically stable C (Licata and Sanford 2012; Buma et al. 2014; Reisser et al. 2016; Pingree and DeLuca 2017); changes in vegetation cover; disruption of soil aggregates (Inbar et al. 2014); exposure of mineral soil

(Doerr et al. 2009); altered hydrologic connectivity (Keesstra et al. 2014); changes in microbial communities (Nelson et al. 2022); and shifts in nutrient/trace metal availability (Abraham et al. 2017; McCarter et al. 2024). Recent studies have shown short-term increases in water-extractable organic carbon and higher carbon dioxide (CO₂) fluxes in burned forest soils compared to unburned ones (VanderRoest et al. 2024). In wetland environments, pyC derived from historical fires can act as an electron shuttle, accelerating the reduction of Fe(III) oxides and influencing the production of CO₂, CH₄, and N₂O (Klöpfer et al. 2014; S. Wu et al. 2021a, b).

The effects of fire on dissolved organic carbon (DOC) concentrations in aquatic systems have been widely documented, though findings are often contradictory. Studies have reported increases (Schindler et al. 1997; Minshall et al. 2001; Rhoades et al. 2018), decreases (Shibata et al. 2011; X. Wei et al. 2021a, b), or no change (Mast and Clow 2008; Smith et al. 2011) in DOC concentrations following fires. Post-fire decreases in DOC have been attributed to combustion of soil organic matter, reduced root exudates following vegetation mortality, decreased canopy throughfall, and suppressed microbial activity (Hart et al. 2005; Holden et al. 2015). Conversely, increases in DOC have been linked to erosion of organic carbon, decreased sorptive capacity of soils due to higher pH, and storm-driven hydrologic changes (Shibata et al. 2003; Aaltonen et al. 2019). These highlight the complexity of post-fire biogeochemical responses, which can vary depending on ecosystem type, topography, fire severity, and hydrological conditions (Bladon et al. 2014). In subalpine wetlands, which can have complex hydrology including both areas of water retention and fast-flowing streams, fire-induced changes in DOC and nutrient dynamics may persist for extended periods (Rhoades et al. 2018; Chow et al. 2019; Marcotte et al. 2024, 2025), or be overshadowed by hydrogeologic settings with no discernible fire effect on DOC concentrations (Davidson et al. 2019a), which highlights the importance of this work for carbon cycling and downstream water quality.

Since wetlands contain disproportionately high amounts of carbon and are located in hydrologically, biologically, and biogeochemically sensitive areas, also referred to as “hotspots” (McClain et al. 2003; Bernhardt et al. 2017) the impacts of fire can be

disproportionate. While wetlands are typically considered fire-resistant due to their high water content, prolonged drought periods can lower water tables, making organic soils susceptible to ignition (Watts and Kobziar 2013; Turetsky et al. 2015). Wetland fires can smolder for extended periods, consuming significant amounts of organic matter and creating distinctive pyrogenic signatures in soil profiles (Watts et al. 2015; Sulwiński et al. 2020). Studies in boreal peatlands and subtropical marshes have shown that wetland fires can dramatically alter carbon cycling, temporarily transforming these systems from carbon sinks to source (Pitkänen et al. 1999; Medvedeff et al. 2013). However, fire effects on subalpine wetlands remain understudied compared to other wetland types, creating critical knowledge gaps in understanding post-fire biogeochemical processes in these high-elevation systems (Grosse et al. 2011). This is why stewards and scientists are concerted in their efforts to restore wetlands after wildfires and there is growing interest in understanding wetlands' role in both mitigating wildfire responses and promoting post-fire ecosystem recovery.

While the biogeochemical impacts of fire have received increasing attention (Bixby et al. 2015), less is known about how these fire-induced changes interact with temperature variability. Subalpine wetlands are particularly sensitive to climate change, which is expected to increase both wildfire frequency and seasonal temperature variability (Parker et al. 2008; Grabherr et al. 2010). These ecosystems are characterized by substantial monthly temperature fluctuations, which can significantly influence microbial activity, organic matter decomposition, and GHG production. This interaction is particularly relevant in subalpine ecosystems, where monthly temperature fluctuations are pronounced and expected to intensify with climate change (IPCC 2022). Temperature is a key driver of microbial activity and organic matter decomposition (Crowther et al. 2016; García-Palacios et al. 2021), and its interaction with fire could amplify or mitigate post-fire biogeochemical changes. Understanding these dynamics is essential for predicting wetland carbon cycling under current variable conditions and future warming scenarios, and for assessing their continued role as carbon sinks.

In this study, we hypothesize that fire will increase the production of CO₂, N₂O, and DOC while suppressing CH₄ production due to shifts in microbial

respiration pathways, and that these effects will be amplified by higher temperatures. To test these hypotheses, we investigated differences in production rates of gaseous and dissolved carbon and nitrogen from burned wetland soils compared to unburned wetland soils. Specifically, we measured potential rates of CO₂, CH₄, N₂O, and DOC production using soils sampled approximately 7 months post-fire using controlled laboratory experiments with systems of increasing complexity (slurry experiments on homogenized soil and flow-through experiments on intact soil cores). Experiments were conducted at three temperatures (9, 18, and 27 °C) to capture the range of thermal conditions experienced in July when sampling occurred. Our findings indicate that fire has significant effects on wetland biogeochemical cycling of carbon and nitrogen, with implications for GHG production and temperature sensitivity under a changing climate.

Methods

The 2020 Mullen fire burned 715 km² of lodgepole pine (*Pinus contorta*) on the Colorado–Wyoming border in the Medicine Bow National Forest starting in September, 2020 and was 97% contained as of January, 2021 (InciWeb, US Forest Service), roughly 7 months prior to sampling. The Mullen Fire burned through forests and subalpine wetlands, thereby impacting the headwaters of the North Platte River watershed, which contributes significant water for agricultural, municipal and recreational use in Wyoming and Nebraska. The fire created a mosaic of burned and unburned areas within its perimeter, with varying burn severities across the landscape (Rodman et al. 2022). We utilized this patchiness to scout wetlands appropriate for this study. The burned wetland is located on Road 517/Dry Park Road on Lake Creek (41.148244, −106.155794) in the Medicine Bow–Routt National Forest (Fig. 1), with the unburned wetland on the other side of this road being drained by an unnamed tributary of Lake Creek. To limit spatial heterogeneity in soil and porewater biogeochemical properties, adjacent burned and unburned sections of the same wetland were sampled in July 2021. Burned sites #1 and #3 were separated by 150 m, and the unburned site #4 was situated 300 m away from site #1 on the other side of the access road. The burned



Fig. 1 Location of the 2020 Mullen fire. Inset shows sampling locations for the burned section of the wetland (sites 1–3, red) and for the unburned section of the wetland (site 4, blue). Inset shows Wetland site 1 in July 2021, approximately 7 months

after the 2020 Mullen fire in the Medicine Bow National Forest, Wyoming. The inset shows recovery of willows and herbaceous wetland vegetation with some charring and burned forest in the background

wetland was characterized by patches of charred and singed vegetation, and samples were taken in proximity to these patches. Three locations on the burned wetland were sampled to account for differences in post-fire vegetation growth, ranging from reed grass (*Calamagrostis canadensis*) and willow (*Salix alba*) dominant sampling site #1 (Fig. 1), tall cottongrass (*Eriophorum angustifolium*) dominant site #2, and water sedge (*Carex aquatilis*) dominant site #3. All of the sites were waterlogged and on similar aspect and slope to minimize variation in hydrological positions within the wetland complex.

Sampling and characterization of the wetland porewater and soil

Wetland porewater samples were collected using porewater diffusion chambers, also known as ‘peepers’ (Hesslein 1976; Mayer 1976). These sampling devices were constructed to be 36 cm long, 25 cm wide, and 5 cm thick, with a detachable handle for retrieval and a triangular bottom for easy insertion into sediments. Constructed from light polyethylene,

they contained 22 60 mL chambers ($1 \times 20 \times 3$ cm) spaced 0.5 cm apart, which were filled with deoxygenated, deionized water initially and covered with a $0.45 \mu\text{m}$ nitrocellulose membrane. To maintain anoxic conditions, these were then placed in large Ziploc bags flushed with N_2 when transported to the field. The peepers were installed in duplicate in each of the three burned and single unburned locations (down to 30 cm) in July 2021, for a total of eight peepers, and left to equilibrate with the surrounding porewater for 21 days. At each site, the peepers were quickly inserted into the unvegetated part of the wetland a meter away from the edge, and the top row of the chambers was flush with the sediment surface. Upon retrieval, the peepers were immediately placed in a N_2 -flushed bag to prevent oxidation of water samples in the peeper chambers and transported to the laboratory where they were transferred in a N_2/H_2 (96%/4%) glove bag. Porewater samples were extracted from the chambers, composited between the replicate peepers by depth, filtered, and subsampled for chemical analyses within 6 h of being retrieved from the field. The pH of porewater samples was measured, and

the water was subsequently acidified with dilute (2% w/v) HNO_3 to preserve chemical speciation of dissolved Fe and N cations/anions. Cation (Ca^{2+} , K^+ , NH_4^+ , Mg^{2+}) and anion (SO_4^{2-} , NO_3^- , Cl^-) concentrations were measured using a Dionex ion chromatograph (IC, electrochemical detector, ED40; analytical column, AS23/AG23; anion self-regenerating suppressor, ASRS-I, 4 mm; mean detection limit 30 $\mu\text{g/L}$, error 5 $\mu\text{g/L}$) supplied with a carbonate eluent (7.2 mM Na_2CO_3 and 1.28 mM NaHCO_3) at a flow rate of 1.0 mL/min. DOC concentrations were measured using a total inorganic and total organic C analyzer (Model 1010 Wet Oxidation Total Organic/Total Inorganic Carbon Analyzer OI Analytical, College Station, TX, USA) with reagents 50 g/L H_3PO_4 for total inorganic C and 100 g/L $\text{Na}_2\text{S}_2\text{O}_8$ for total organic C with heat catalysis at 100 °C (MDL 12 $\mu\text{g/L}$, AE 3.5%).

Duplicate soil cores were collected from each peeper location to a depth of approximately 24 cm. After the cores were transported back to the lab, the organic layer was removed, approximately 2–5 cm across cores, and only the mineral soil was analyzed, since we are interested in coupled Fe(III) reduction and greenhouse gas production in deeper soil layers. Mineral soil is a better baseline for comparison as it is less affected by fire than the organic horizons. Visible pieces of charred plant material were retained in these mineral soil horizons. One core was separated into two depth intervals (0–12 cm and 12–24 cm) which were used for soil physical and chemical characterization, while the second core was used for flow-through reactor experiments (see "Statistical analyses" section). To characterize the physical and chemical properties of the wetland soils, we conducted a series of standard laboratory analyses carried out in triplicates. We determined soil dry bulk density by drying a known volume of soil for 24 h at 105 °C, providing insight into soil compaction and porosity that could influence water and gas movement. Soil pH was measured by suspending 5 g of dried soil in a 10 mM CaCl_2 solution following methods described by Thomas (1996). For soil organic carbon and nitrogen content assessment, we first pre-treated soil samples with dilute HCl to remove inorganic carbon before combusting 30 mg of dried soil using a NC 2100 Elemental Analyzer (CE Instruments, USA). To evaluate the soils' potential for redox reactions that might influence greenhouse gas production, we quantified

oxalate-extractable iron using Schwertmann's protocol (1973). This extraction provides a reliable proxy for poorly-ordered iron oxides, which are highly reactive and likely to participate in electron transfer processes in anaerobic environments. Finally, we measured the mass of ash remaining after combusting dried soil samples at 450 °C in a muffle furnace according to ASTM D2794-20e1 protocol (American Society for Testing and Materials 2020). The ash remaining after 'loss-on-ignition' (LOI) is a combination of the ash resulting from combustion of pre-fire soil organic matter, material charred during the fire and wildfire ash. We assume that potential differences in LOI-ash between burned and unburned soils with similar pre-fire soil organic matter is attributable to wildfire ash.

Slurry experiments

To estimate maximum potential rates of CO_2 , CH_4 , and N_2O production at each study site, soil slurry experiments, also referred to as batch incubations, were carried out at 27 °C (maximum air temperature in July at our site). Duplicate soil slurry experiments were carried out in 125 mL serum bottles, using a 1:4 (w/w) wet soil: deionized water mixture (Burdige 1989), with an initial pH of 7.5 based on the pore-water pH measured for shallow soils (Table 1). Shallow soil (0–12 cm depth interval) and deeper soil (12–24 cm depth interval) were analyzed separately. At the beginning of the experiments, the serum bottles were evacuated, and the sterile deionized water was flushed with N_2 to maintain oxygen-free experimental conditions. The slurries were incubated in anoxic conditions under gentle orbital shaking to maintain the soil in suspension and homogenous conditions. Slurries were sub-sampled for liquid and gas analyses every hour for the first 6 h, and subsequently daily, for a period of 7 days. Liquid samples were filtered through a 0.22 μm PTFE membrane syringe filter (Whatman; 25 mm filter size) prior to chemical analyses for concentrations of total Fe, Fe(II), NO_3^- , SO_4^{2-} , NH_4^+ , and DOC. Liquid samples were analyzed as described in "Slurry experiments" section.

An equal volume of N_2 was injected into the bottles to maintain vapor pressure during gas sampling. Concentrations of CO_2 , CH_4 , and N_2O in the gaseous sub-samples were analyzed on an Agilent 7890B GC (Agilent, Santa Clara, CA, mean detection limit

Table 1 Soil and porewater physical and chemical characteristics for the four study sites and two depth intervals (0–12 and 12–24 cm)

Soil characteristics	Burned 1 Depth interval (0–12 cm)	Burned 2	Burned 3	Unburned
Sand content (%)	32.1 ± 2.5	29.7 ± 1.5	24.4 ± 3.7	12.3 ± 1.9
Silt content (%)	25.3 ± 2.5	22 ± 2.5	30.6 ± 4.1	36.5 ± 2.7
Clay content (%)	42.3 ± 6.5	47 ± 4.6	44.9 ± 2.5	50.9 ± 2.5
Dry bulk density (g/cm ³)	0.31 ± 0.07	0.30 ± 0.05	0.29 ± 0.03	0.34 ± 0.08
pH	6.17 ± 0.61	6.36 ± 0.42	6.19 ± 0.33	5.28 ± 0.53
LOI ash (wt%)	8.83 ± 1.71	7.19 ± 2.51	6.55 ± 1.5	1.12 ± 0.25
C _{org} soil (%)	23.6 ± 2.1	21.3 ± 3.5	22.5 ± 2.0	30.7 ± 2.3
Total N content (%)	1.17 ± 0.11	1.26 ± 0.35	0.97 ± 0.25	2.28 ± 0.57
Molar C _{org} :N ratio	22.54 ± 2.06	19.73 ± 3.15	22.6 ± 1.87	14.39 ± 2.16
Extractable Fe content (mg/g soil)	22.6 ± 4.5	23.9 ± 3.6	22.9 ± 5.5	19.4 ± 3.7
Extractable Ca content (mg/g soil)	344.7 ± 42.5	281.5 ± 31.6	266.7 ± 37.1	126.3 ± 12.9
Extractable Mg content (mg/g soil)	27.12 ± 3.92	24.80 ± 2.33	23.9 ± 4.91	19.86 ± 2.55
Soil characteristics	Depth interval (12–24 cm)			
Sand content (%)	12.1 ± 3.6	9.2 ± 2.7	14.2 ± 1.9	7.4 ± 1.2
Silt content (%)	35.3 ± 4.2	32.6 ± 5.2	32.6 ± 4.6	38.6 ± 6.1
Clay content (%)	52.6 ± 4.9	57.6 ± 6.7	54.9 ± 4.8	55.9 ± 7.6
Dry bulk density (g/cm ³)	0.51 ± 0.13	0.48 ± 0.11	0.53 ± 0.07	0.52 ± 0.09
pH	6.28 ± 0.35	6.33 ± 0.47	6.11 ± 0.37	6.09 ± 0.42
LOI ash (%)	2.43 ± 0.57	3.19 ± 0.36	2.55 ± 0.44	0.73 ± 0.17
C _{org} content soil (%)	10.56 ± 1.38	9.73 ± 2.51	10.51 ± 1.67	12.67 ± 3.45
Total N content (%)	0.33 ± 0.06	0.32 ± 0.07	0.34 ± 0.09	0.41 ± 0.11
Molar C _{org} :N ratio	26.54 ± 1.37	30.31 ± 2.15	30.91 ± 1.16	28.39 ± 3.36
Extractable Fe content (mg/g soil)	28.2 ± 4.6	24.2 ± 5.5	29.9 ± 6.2	25.4 ± 4.8
Extractable Ca content (mg/g soil)	284.2 ± 32.8	270.4 ± 42.2	226.7 ± 39.2	116.4 ± 12.7
Extractable Mg content (mg/g soil)	28.3 ± 2.7	26.0 ± 4.6	27.9 ± 4.0	22.2 ± 1.9
Porewater concentrations	Depth interval (0–12 cm)			
pH	7.36 ± 0.12	7.55 ± 0.19	7.67 ± 0.22	7.11 ± 0.17
DOC (mg/L)	84.15 ± 10.37	103.70 ± 2.5	104.82 ± 2.5	65.12 ± 2.5
Sulfate (mg/L)	3.33 ± 0.55	2.83 ± 0.47	3.12 ± 0.39	1.38 ± 0.33
Nitrate (mg/L)	1.88 ± 0.36	1.39 ± 0.25	1.30 ± 0.17	0.47 ± 0.08
Fe(II) (mg/L)	3.72 ± 0.68	3.19 ± 0.44	3.45 ± 0.47	2.57 ± 0.29
Ca (mg/L)	45.58 ± 8.32	62.22 ± 15.11	50.07 ± 7.49	26.67 ± 4.55
Mg (mg/L)	3.72 ± 0.33	6.21 ± 0.46	5.57 ± 0.59	2.11 ± 0.32
Porewater concentrations	Depth interval (12–24 cm)			
pH	6.88 ± 0.09	6.75 ± 0.11	6.83 ± 0.07	6.82 ± 0.12
DOC (mg/L)	90.27 ± 17.15	108.93 ± 22.7	104.23 ± 12.4	60.27 ± 12.33
Sulfate (mg/L)	1.11 ± 0.27	1.43 ± 0.36	1.22 ± 0.27	0.78 ± 0.15
Nitrate (mg/L)	0.12 ± 0.02	0.15 ± .05	0.10 ± 0.01	0.11 ± 0.03
Fe(II) (mg/L)	3.59 ± 0.28	4.55 ± 0.39	4.19 ± 0.56	3.88 ± 0.63
Ca (mg/L)	50.22 ± 4.91	48.89 ± 9.46	47.43 ± 8.55	37.72 ± 5.4
Mg (mg/L)	6.37 ± 1.28	6.58 ± 2.09	7.07 ± 1.43	4.45 ± 0.96

Porewater concentrations were obtained via averaging by depth for values shown in Fig. S1. Values reported are means obtained from triplicate samples

10 ppb, error 3 ppb) using thermal conductivity, flame ionization, and nitrogen chemiluminescence detectors (TCD, FID, and NCD) respectively. Concentrations of CO₂, CH₄, N₂O, and Fe(II) were then used to calculate production rates for each according to:

$$R = \frac{\Delta C}{\Delta T}, \quad (1)$$

where ΔC ($\mu\text{g C/cm}^3$) is the change in concentration of the gas/solute over a time increment ΔT (h^{-1}) of the slurry being shaken. Gas concentrations from initial kinetic phase (0–6 h) were considered for these calculations due to a stronger linear slope, following best practices from Hutchinson et al. (2000). Later reaction conditions represent conditions where substrates become limiting and reactants start accumulating, inhibiting mineralization, and thus were not considered for these measurements (Colores et al. 1996).

Flow-through reactor experiments

Flow-through reactor (FTR) experiments on intact soil cores were performed to measure the potential rates of Fe(II), DOC, DIC, CO₂, N₂O and CH₄ production. These differ from rates calculated from slurry experiments as they are more representative of physical and chemical heterogeneity of the soil matrix, which influences the availability of substrates to microbes compared to well-mixed slurry experiments that overestimate reaction rates (Pallud and Van Cappellen 2006). Other advantages of FTR experiments are that there is no accumulation of reaction byproducts which can influence reaction rates, and that these rates are only calculated when steady state is achieved in the reactor (Pallud and Van Cappellen 2006). The reaction rates obtained with such systems, more specifically Fe(II), DOC, DIC, and CH₄ production rates, are referred to as “potential” rates, as they refer to conditions where Fe(III) oxides are the dominant terminal electron acceptor accessible to microorganisms, and no electron acceptor or donor are added to the system. Experiments were performed in duplicate using soils from two depth intervals, a shallow soil (0–2 cm depth) and a deeper soil (15–17 cm depth). Each reactor cell contained a soil core within a 2 cm long and 4.2 cm diameter Plexiglass ring, with 0.2 μm pore size nitrocellulose filters

and glass fiber filters at each end. The FTR were sealed on either side with Plexiglass caps tightened with screws, with O-rings to prevent leaks. Input/output channels allow for uniform flow throughout the cross-section of the soil slice (Pallud and Van Cappellen 2006). The experiments were carried out in a glove bag under anoxic conditions (4.5% H₂, 95% N₂) and the FTR were placed in a temperature-controlled water bath to ensure constant temperatures of 9, 18, and 27 °C, simulating minimum, mean, and maximum air temperatures for the month of July for this area, using data from the Elk Mountain Weather Station, located at a similar elevation (2225 m) (National Weather Service, 2021). Via the input channel, a solution was supplied to the reactors at a constant volumetric flow rate (1.5 ± 0.1 mL/h) maintained by a peristaltic pump, corresponding to residence times of 12–14 h, similar to residence times for wetland geographies at similar elevation and slope (Jepsen et al. 2019; Daugherty et al. 2019; Pallud et al. 2020). The input solution used mimicked the chemical composition of porewater samples in the field. It contained an average of 5 mg/L Na⁺ (as Na₂HPO₄), 30 mg/L Ca²⁺ (as CaCl₂·2H₂O), 15 mg/L K⁺ (as K₂HPO₄), and 0.15 mg/L NH₄⁺ (as NH₄Cl) with a pH of 7.5 and was purged with N₂ to remove any oxygen. Outflow liquid samples were collected every day and analyzed for Fe(II), DOC and DIC concentrations. Fe(II) was measured spectrophotometrically using the ferrozine method at 562 nm (Stookey 1970). Concentrations of DOC and DIC were measured using a wet oxidation total inorganic and total organic C analyzer (Model 1010 Wet Oxidation Total Organic/Total Inorganic Carbon Analyzer OI Analytical) with 50 g/L H₃PO₄ for total inorganic C and 100 g/L Na₂S₂O₈ for total organic C with heat catalysis at 100 °C (detection limit 12 $\mu\text{g/L}$, standard error 3.5%). Outflow gas samples were subsampled and sparged with N₂ gas and the headspace sampled to purge the water of CH₄. Gas samples were analyzed on an Agilent 7890B GC (Agilent, Santa Clara, CA) for CH₄ using flame ionization detector (FID).

Potential steady-state rates of production of Fe(II), DOC, DIC, and dissolved CH₄ were calculated according to Eq. (2) for steady-state conditions when chemical composition of the outflow was no longer fluctuating significantly (about 18 ± 3 data points) (Figs. S3–S6), according to:

$$R = \frac{\Delta C * Q}{V}, \quad (2)$$

where Q is the volumetric flow rate (mL/h), V is the reactor volume (cm³), and ΔC is the difference in concentration of the solute between the outflow and inflow solutions (mM) at steady-state (Richards and Pallud 2016). It took around 12 days after the start of the experiment to reach steady-state for Fe(II), CH₄, DIC and DOC concentrations. Fe(II) production rates provide a lower limit for Fe(III) reduction rates due to secondary precipitation of Fe(II) to siderite and lepidocrocite (Pallud et al. 2020), sorption onto clay minerals, and/or binding to native organic matter sorbed onto mineral surfaces (Daugherty et al. 2017).

Reaction rates were used to calculate apparent temperature coefficients, Q_{10} (Atkin et al. 2000), as:

$$Q_{10} = 10^{(10 * \text{coef}_{\log_{10}(R)})}, \quad (3)$$

where R indicates rates for the three temperatures tested in the experiment, and $\text{coef}_{\log_{10}(R)}$ is the slope obtained from the plot between $\ln(R)$, plotted against T (in Kelvin), Activation energy, E_a , was calculated by multiplying the slope from the plot between $\ln(R)$ and T^{-1} with the universal gas constant R (8.314 J/mol/K) (Atkin et al. 2000).

Statistical analyses

All statistical analyses were carried out in R (2025). Data was tested for normality using the Shapiro–Wilk test and log-transformed when necessary. Given the constraints of spatial sampling in this study, and the spatial autocorrelation between depth-resolved soil and porewater measurements, these field measurements are presented descriptively without formal statistical testing. Porewater depth profiles (Fig. S1) are presented to visualize vertical gradients in biogeochemical properties. For slurry and flow-through reactor experiments, production rates (measured after ~12 days for FTR reactors; Figs. S3–S6) were analyzed using linear mixed-effects models with burn status, depth, and temperature as fixed effects and random effects accounting for site-level variation (1site), spatial autocorrelation between depths sampled from the same core (1site:depth), and temporal autocorrelation in repeated measurements during steady-state (1time). The random effects structure was specified

to account for the hierarchical nature of our data: site-level variation, spatial autocorrelation between depths from the same core, and temporal autocorrelation in repeated measurements. This approach helps address concerns about non-independence in our experimental design. These mixed effects models were implemented using lme4 (Bates et al. 2025) and lmerTest (Kuznetsova et al. 2020) packages. Table S3 displays F statistics, degrees of freedom and p-values for all effects tested. Tukey’s honest significance difference test was used to determine which variables were found to be different across sites, depths, and temperatures, if found significant ($\alpha=0.05$). Pearson’s correlation coefficients were calculated between measured water, soil, and gas variables from slurry experiments and visualized using a correlation matrix using the “corrplot” package (T. Wei et al. 2021a, b). Figures were generated using the ‘ggplot2’ package (Wickham et al. 2025).

Results

Soil from four sites possessed a clay texture with similar bulk density (0.29–0.34 and 0.48–0.53 g/cm³ for the shallow and deep soils, respectively). All sites were characterized by high clay content (42.3–50.9 and 52.6–57.6% for the shallow and deep soils, respectively). The pH of all soils was slightly acidic, with the pH of the shallow soil in the burned sites (6.17–6.36) was about one unit higher than for the unburned site (5.28). However, there were no differences in pH between sites for the deeper soil. The LOI ash content differed between study sites and soil depths. Shallow soil from burned sites contained substantially higher LOI ash (6.6–8.8%) compared to the unburned site (1.1%). This pattern persisted in deeper soils, though with lower overall values: burned sites contained 2.4–3.2% LOI ash while the unburned site had only 0.73%.

Carbon and nitrogen content showed the opposite pattern. All soils contained high organic carbon, but burned shallow soils had less carbon (21.3–23.6%) than unburned shallow soil (30.7%). Nitrogen followed a similar trend, with burned shallow soils containing 0.97–1.26% compared to 2.28% in unburned soil. This pattern extended to deeper soil layers, where burned sites had lower carbon (9.7–10.5% versus 12.6%) and nitrogen (0.32–0.35% versus

0.41%) than unburned sites. The soil C:N ratios increased with depth for all sites and were smaller in the unburned site than the burned ones for the shallow soils only. Extractable Fe content did not differ between the burned and unburned soils but increased with depth. The unburned soil was characterized by significantly lower extractable Ca and Mg content than the unburned soils at both the depths investigated. The porewater was neutral to slightly alkaline (pH 7.1–7.7) in the shallow soil and closer to neutral (pH 7.0–7.2) in the deeper soils. The porewater from the burned soils was characterized by higher DOC, sulfate, nitrate, Fe(II), Ca and Mg concentrations compared to the unburned soils. These trends did not persist with porewater from deeper soils.

Evolution of concentrations and production rates of CO_2 , CH_4 and N_2O in slurry experiments

Figure 2a shows the initial CO_2 production rates calculated for the first 6 h of the slurry experiments. Fire did not significantly affect CO_2 production rates when accounting for repeated measures within experimental units, though burned soils had numerically higher mean production rates ($150.1 \mu\text{g C/cm}^3/\text{h}$) compared to unburned soils ($112.3 \mu\text{g C/cm}^3/\text{h}$). Concentrations of CO_2 increased throughout the course of the slurry experiments across all but one of the eight experimental conditions (Fig. S2). For all experimental conditions, CO_2 concentrations rapidly increased in the first 6 h of the experiment followed by a slower increase for the remaining time points for most experimental conditions (Fig. S2a). Over the course of the entire experiment, burned shallow soils produced on

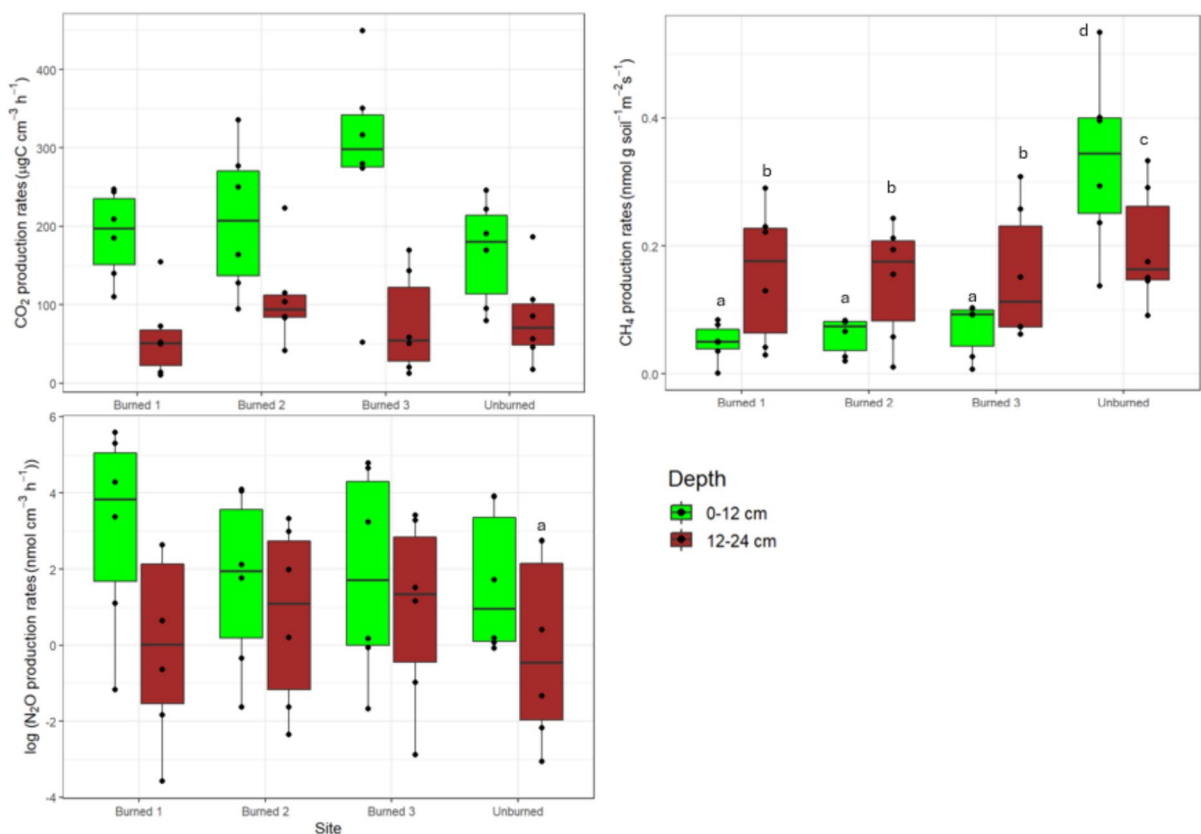


Fig. 2 Average **a** CO_2 production rates, **b** CH_4 production rates, and **c** N_2O production rates calculated for the initial rapid evolution phase of the slurry experiments, using measurements taken during the first 6 h of the experiments. Green and red

boxplots represent soil depth intervals of 0–12 and 12–24 cm, respectively. Different letters indicate significant differences at $p \leq 0.05$

average as much as 25% more CO_2 than unburned soils; however, there was no significant difference in CO_2 production between unburned and burned soils for the deeper soils. Deeper soils cumulatively produced 53% less CO_2 than shallower soils through the course of the experiment.

Figure 2b shows production rates of CH_4 calculated for the first 6 h of the slurry experiments. CH_4 production rates were significantly higher from the unburned soil than the burned ones (burn effect: $F_{1,4}=42.11$, $p=0.003$), with a significant burn $\times$ depth interaction ($F_{1,4}=14.63$, $p=0.019$). Unburned shallow soils produced $0.34 \mu\text{g C/cm}^3/\text{h}^1$ compared to an average of $0.07 \mu\text{g C/cm}^3/\text{h}$ for burned shallow soils. The depth effect approached significance ($F_{1,4}=6.89$, $p=0.058$). Concentrations of CH_4 increased over time for the whole duration of the slurry experiments for unburned soils but stabilized within 1–3 days for the burned soils (Fig. S2b). Averaged over all time points, unburned soils produced, on average, 2.9 times more CH_4 compared to soils from burned sites. Shallow soils at the unburned site produced more CH_4 than the deep soils, while on the other hand, shallow soils at the burned site produced less methane than the deep soils. ($p=0.042$).

The temporal evolution of N_2O concentrations in the slurry experiments was characterized by a strong initial pulse around 1–2 h after the start of the experiment, followed by a steep decline for the remainder of the experiments, indicating consumption of the N_2O (Fig. S2c). N_2O concentrations were higher in experiments with burned soils compared to unburned soils (Fig. S2c).

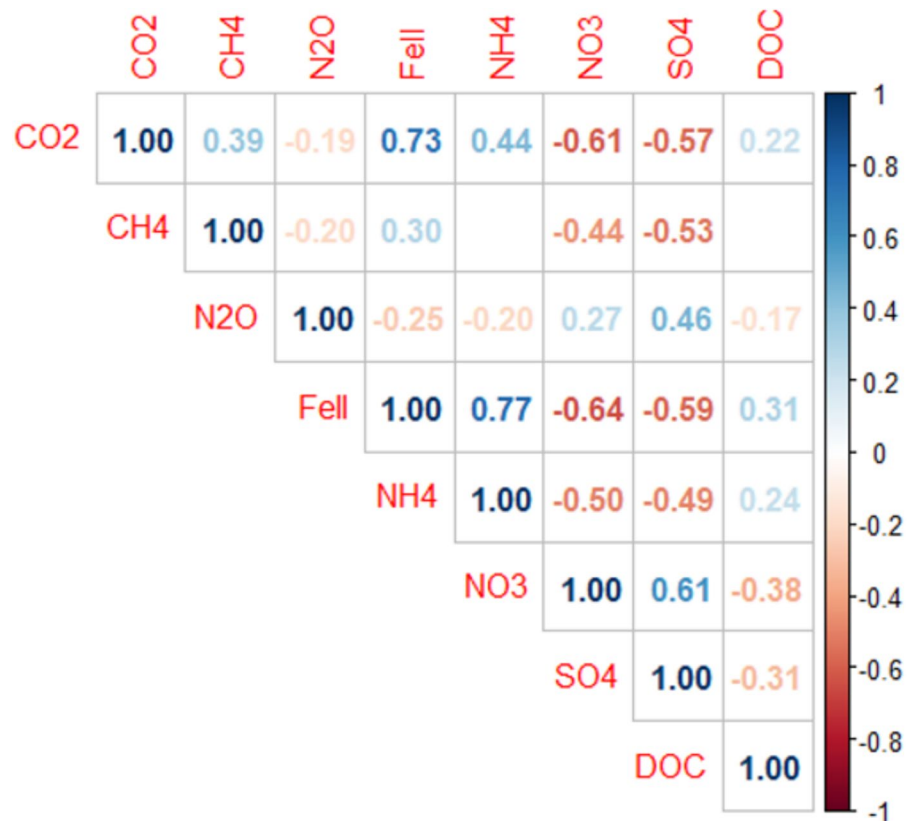
Figure 2c shows production rates of N_2O calculated for the first 6 h of the slurry experiments plotted as log-transformed for ease of visual representation. Neither burn status, depth, nor their interaction significantly affected N_2O production rates when accounting for repeated measures, though numerically higher rates were observed in some burned shallow soils. The temporal evolution of N_2O concentrations in the slurry experiments was characterized by a strong initial pulse around 1–2 h after the start of the experiment, followed by a steep decline for the remainder of the experiments, indicating consumption of the N_2O (Fig. S2c). N_2O concentrations were higher in experiments with burned soils compared to unburned soils (Fig. S2c).

Building on our analysis of individual gas production rates, we next examined the relationships between different biogeochemical parameters to better understand the underlying processes driving these patterns. Correlation analysis of measured chemical parameters from the slurry experiments (Fig. 3) revealed several significant associations between gaseous and aqueous chemical parameters. Carbon dioxide concentrations were strongly positively correlated with porewater Fe(II) concentrations and NH_4^+ concentrations, whereas associations between CO_2 and CH_4 concentrations and DOC were weaker with lower correlation coefficients ($r=0.39$ and 0.22) respectively. Concentrations of CO_2 were also strongly negatively associated with porewater NO_3^- and SO_4^{2-} concentrations, and weakly negatively associated with N_2O concentrations. Methane concentrations were negatively associated with porewater NO_3^- and SO_4^{2-} concentrations, while being weakly positively correlated with porewater Fe(II) concentrations. Nitrous oxide concentrations were weakly positively correlated to NO_3^- and SO_4^{2-} concentrations. There were other notable strong associations, with NO_3^- and SO_4^{2-} concentrations positively correlated to each other, whilst both were negatively correlated with porewater NH_4^+ and Fe(II) concentrations.

Potential rates and temperature sensitivity of reduced iron, dissolved inorganic/organic carbon, and methane production measured using flow-through reactor experiments

Figure 4 shows steady-state potential production rates of Fe(II), DIC, and CH_4 and steady-state potential export rates of DOC, measured for all sites and depths, and in response to three temperatures, while Table 2 summarizes the apparent temperature coefficient, Q_{10} , and activation energy, E_a , calculated for all production rates. On average, soils from burned sites were characterized by higher Fe(II) production rates compared to the unburned site ($F_{1,12}=3.07$, $p=0.078$). Fe(II) production rates were higher ($F_{1,12}=2.66$, $p=0.092$) for deeper soils than shallow soils across sites and temperature regimes, except for Burned site 2. This trend was reflected in Q_{10} values obtained for shallow and deep soils from burned and unburned sections (Table 2). On average, Q_{10} values for Fe(II) production were found to be higher for burned sites compared to the unburned site and

Fig. 3 Correlation plot of all measured response variables from the slurry experiments for all time points. CO₂, CH₄, and N₂O represent concentrations measured in gas samples taken at each time point. Fe(II), NH₄⁺, NO₃⁻, SO₄²⁻ and DOC represent concentrations measured in aqueous samples post filtering at every time point. Correlation coefficients are reported at a threshold of $p \leq 0.05$. Values represent correlation coefficients ranging from -1 (deep red) to 1 (deep blue)



ranged from 1.57 to 2.90 for shallow soils compared to 1.4 to 1.85 for deeper soils. For the shallow soils, E_a values for Fe(II) production were significantly higher for burned soils than unburned soil, however, E_a values for Fe(II) production were similar for burned and unburned sites for deeper soils. The range of E_a for Fe(II) production rates was 32.1–73.2 kJ/mol for shallow soils, compared to 30.8–43.3 kJ/mol for deeper soils (Table 2).

Across all temperature treatments, burned sites exhibited higher DOC export rates compared to unburned sites ($F_{1,12}=6.42$, $p=0.025$). On average, shallower soils produced higher DOC than deeper soils, however this effect was not statistically significant. DOC rates on average were higher for 18 °C across most sites and depths, with the exception of the deeper unburned soil.

DIC production rates showed a strong temperature effect ($F_{2,12}=31.45$, $p<0.001$), with rates increasing substantially from 9 to 27 °C across all sites. DIC rates were similar at 9 °C across sites and depths ($<0.5 \mu\text{g C/cm}^3/\text{h}$), however rates were numerically

higher at 18 and 27 °C for shallow compared to deep soils. The Q_{10} values for DIC production were tightly distributed, ranging from 4.90 to 5.85 across and unburned shallow and deep soils (Table 2). On average, values were slightly higher for deep compared to shallow soils, with no differences between burned and unburned soils from either depth. E_a values for DIC production varied between 114.3 and 129.0 kJ/mol for shallow soils compared to 116.2 to 132.4 kJ/mol for deeper soils, with the lower values reported for unburned soils for both shallow and deep depths (Table 2).

CH₄ production rates showed significant effects of burn status ($F_{1,12}=31.08$, $p<0.001$), depth ($F_{1,12}=15.58$, $p=0.002$), and temperature ($F_{2,12}=65.12$, $p<0.001$). The three-way interaction was also significant ($F_{2,12}=4.05$, $p=0.045$), indicating that fire effects on CH₄ production varied with depth and temperature. Unburned soils consistently exhibited higher CH₄ production rates compared to burned soils, particularly at 27 °C (Fig. 4g, h). Q_{10} values for CH₄ production ranged from 1.21 to 1.42

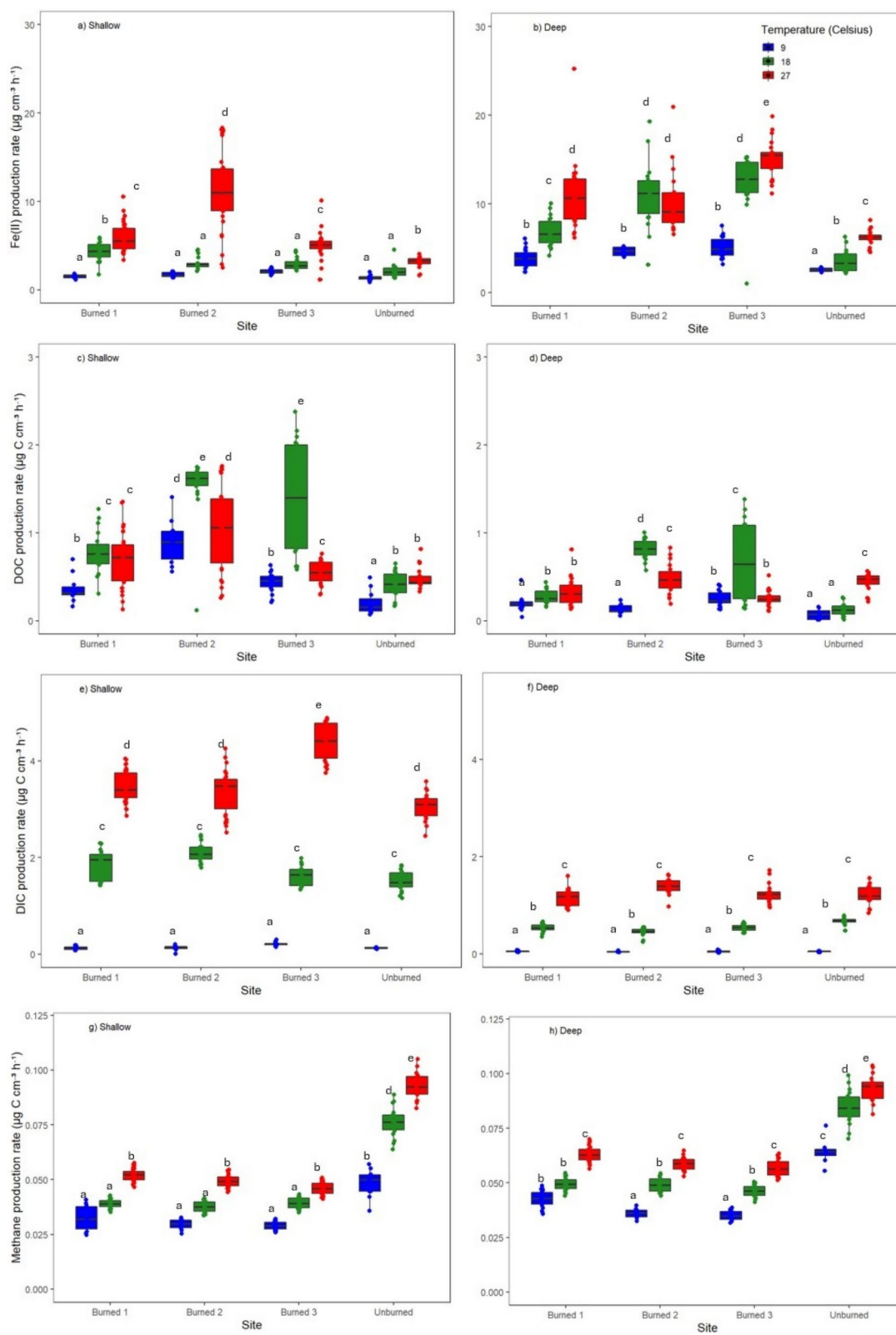


Fig. 4 Steady-state production rates of **a, b** Fe(II), **c, d** DOC, **e, f** DIC, and **g, h** dissolved methane (CH_4) calculated using continuous flow-through reactor experiments performed on intact soil cores for the four sampling sites in response to temperatures of 9 (blue), 18 (green), and 27 °C (red), for **a, c, e, g** the shallow soil depth interval (0–2 cm) and **b, d, f, h** the deeper soil depth interval (15–17 cm). Different letters denote significant differences at $p \leq 0.1$. Each boxplot represents 18 ± 3 production rates for the specific temperature and depth

across all conditions and sites tested. E_a values for CH_4 production ranged from 18.1 to 25.6 kJ/mol for shallow soils compared to 14.6 to 19.1 kJ/mol for deep soils.

Discussion

Our results reveal significant alterations in biogeochemical cycling following fire, with pronounced effects on greenhouse gas dynamics and iron–carbon coupling in subalpine wetlands. Burned wetland soils possessed higher Fe(III) reduction rates while producing significantly less methane than unburned soils, possibly indicating an increase in soil redox potential. In fact, pyC can increase soil redox potential due to its electron storage capacity (Klöpfer et al. 2014). Acting as “electron snorkels”, pyC can accelerate reduction and dissolution of Fe(III) oxides, leading to desorption and subsequent mineralization of organic matter, which would explain higher CO_2 and Fe(II) export rates for burned soils, also confirmed by higher dissolved Fe(II) concentrations in porewaters from burned soils. In fact, the highest Fe(II), DOC, and CO_2 production rates were observed for Burned site 2, which also possessed the highest concentrations of oxalate-extractable iron oxides, a proxy for reactive iron oxides known to stabilize up to 22% of all carbon in sediments (Lalonde et al. 2012; Mu et al. 2016). This elevated Fe–C coupled cycling has been attributed to the electron-shuttling capacity of pyrrolic and pyridinic nitrogen present in fire-derived organic matter (Wu et al. 2021a, b). Additionally, this increase in soil redox potential in conjunction with higher LOI ash in the upper depth of burned wetland soils, also may have altered CH_4 and N_2O production rates we measured. Burned soils produced considerably less CH_4 compared to unburned soils due to the promotion of thermodynamically favorable alternate respiration strategies such as nitrate, sulfate, and Fe(III)

reduction. Furthermore, the decrease in $C_{org}/\text{nitrate}$ ratios following the combustion of organic matter and subsequent processing of bioavailable nitrate has been associated with a higher tendency for denitrification compared to dissimilatory nitrate reduction (DNRA; Rahman et al. 2019; Palacin-Lizarbe et al. 2019) which would explain the higher potential nitrous oxide production rates for burned soils compared to unburned wetland soils. Our results strongly support the idea that fire inputs enhance nitrate and Fe(III) oxide reduction in soils, resulting in higher CO_2 , DOC, and N_2O production while suppressing CH_4 emissions, the last of which has been reported previously for wetland peat amended with artificially produced biochars (Sun et al. 2021) and a boreal fen affected by a wildfire (Davidson et al. 2019b), although Davidson et al. attribute it more to the loss of organic matter substrate for methanogenesis.

Our results showed that burned and unburned wetland soils were characterized by significantly different soil and porewater chemistry, particularly in shallow depths. Porewater chemistry is influenced by fire due to combustion and loss of native organic matter and the lateral transport of pyrolyzed carbon from adjacent forest sections, resulting in lower C and N content but higher ash content observed in burned soils. Wildfire ash increases pH, extractable Ca and Mg (Ohno and Susan Erich 1990; Kotze 2013), which is consistent with our porewater concentrations measurements (Fig. S2). Higher soil pH and soil temperatures induced by fires are also known to increase mineralization of organic matter, which would explain elevated DOC, nitrate, sulfate, and Fe(III) in burned soil porewaters. The persistence of the fire signal in porewater chemistry seven months after the event could be explained by reduced plant demand for these nutrients as vegetation recovers and regenerates post snowmelt. This is supported by observations from a previous study showing elevated nitrate and DOC in streams and watersheds for up to two years post-fire (Rhoades et al. 2018). Our field observations are supported by measurements from slurry incubations and flow-through reactor experiments. The higher potential CO_2 and DOC production rates observed in burned soils both in slurry and flow-through reactor experiments suggest that fire-derived ash (i.e., LOI-ash) combined with a warmer, drier soil microclimate may stimulate short-term microbial mineralization of organic carbon, possibly through priming and

Table 2 Apparent temperature coefficient (Q_{10}) and activation energy (E_a) values for production rates of Fe(II), DIC, and methane (CH_4) calculated for each site and depth combination

Values are reported pairwise for temperatures at which flow-through reactor experiments were performed

Site + depth	Fe(II) production rate		DIC production rate		CH ₄ production rate	
	Q_{10}	E_a (kJ/mol)	Q_{10}	E_a (kJ/mol)	Q_{10}	E_a (kJ/mol)
Burned 1 (0–2 cm)	1.95	53.08	5.16	129.02	1.32	18.51
Burned 2 (0–2 cm)	2.90	73.21	5.11	127.41	1.33	20.01
Burned 3 (0–2 cm)	1.60	35.05	5.37	119.60	1.29	18.11
Unburned (0–2 cm)	1.57	32.09	5.25	124.31	1.42	25.55
Burned 1 (15–17 cm)	1.79	41.58	5.30	120.08	1.25	15.37
Burned 2 (15–17 cm)	1.40	30.83	5.85	131.09	1.30	19.09
Burned 3 (15–17 cm)	1.85	43.25	5.90	124.91	1.31	18.89
Unburned (15–17 cm)	1.68	34.41	4.90	126.58	1.21	14.56

co-metabolism effects. Wildfire inputs are known to increase soil pH and mineralization of organic N and S, providing readily available nutrients for microbes (Whitman et al. 2021). Recent research by VanderRoest et al. (2024) has highlighted the presence of labile organic moieties in pyrogenic carbon itself, which could enhance native microbial activity and increase post-fire CO_2 fluxes and production rates (Maestrini et al. 2015). Post-fire microbial community shifts favoring Actinobacteria, and genes associated with heat tolerance, rapid growth, and substrate utilization could also explain the higher mineralization rates in burned sites (Nelson et al. 2022; VanderRoest et al. 2024).

Temperature sensitivity of measured production rates

Q_{10} values for Fe(II) production in burned soils (1.60–2.90) consistently exceeded those in unburned soils (1.57) in shallow depths. This pattern aligns with the electron shuttle capabilities of fire inputs, particularly pyC, which can catalyze Fe(III) reduction processes (Klöpfer et al. 2014; Sun et al. 2017). Our data show a gradient of temperature sensitivity that correlates with ash content—shallow burned soils with higher ash content (6.6–8.8%) exhibited greater Q_{10} values than deeper burned soils (2.43–3.19% ash), suggesting a direct relationship between pyrogenic inputs and temperature sensitivity. The activation energies we measured for Fe(III) reduction (32.1–73.2 kJ/mol) are comparable to those reported by Schilling et al. (2019) for subalpine wetland soils, but substantially higher than the 10–50 kJ/mol range reported for temperate wetlands and lake sediments (Roden and Wetzel 1996; Meier et al. 2005). This difference in temperature sensitivity likely reflects

adaptations of microbial communities to the colder temperature regime in subalpine environments, where organisms often require higher activation energy to initiate biochemical processes (German et al. 2012). The increased temperature sensitivity in burned soils may have important implications for Fe and carbon cycling under future warming scenarios, potentially accelerating carbon mineralization during seasonal warm periods.

The temperature response of DOC production showed a non-linear pattern, with highest rates observed at 18 °C rather than 27 °C across all treatments. This decrease in DOC production rates between 18 and 27 °C, particularly pronounced in burned soils, can be attributed to increased microbial consumption of DOC at higher temperatures, as previously observed in subalpine wetland soils (Pallud et al. 2020). This temperature-dependent balance between DOC production and consumption made traditional temperature sensitivity calculations (Q_{10} , E_a) inappropriate for this process. Instead, the data suggests that warming could lead to more rapid DOC turnover rather than accumulation, particularly in shallow, fire-affected soils where microbial activity appears enhanced.

The Q_{10} values for DIC production rates (4.9–5.9), which serve as a proxy for soil respiration rates, were substantially higher in our subalpine wetland soils than those reported for coastal wetland and tropical forest-soil incubations (1.1–2.4) by Chari et al. (2021), despite their similar incubation temperature ranges (8–35 °C). However, our values are consistent with wetland soils Q_{10} estimates from modeling approaches where values as high as 16.4 were measured (Q. Wu et al. 2021a, b), and with a global dataset spanning different ecosystems, where values could

go as high as 300 for tundra soils and around 30 for peat and mineral wetlands (Hamdi et al. 2013). One reason which could explain the similarity in Q_{10} values from our experiments is that % ash remains stable throughout the FTR experiments, which suggests that most of the carbon mineralized during the experiment is primarily native and non-pyrogenic. This has also been observed in microbial communities from fire-affected beaver ponds, where a proclivity for labile C was found in comparison to pyrogenic C (Roth et al. 2022). Indeed, carbon quality has been known to affect soil respiration sensitivity and Q_{10} values (Wang et al. 2018; Zhang et al. 2024). Lastly, our measured E_a values fell on the higher end of estimates from an analysis of data from 42 widely distributed wetland sites in the FLUXNET-CH₄ database (Li et al. 2023), indicating that microbial respiration in these subalpine wetland soils requires more energy and shows greater temperature dependence compared to wetlands in warmer climates.

Both Q_{10} and E_a values (14.5–25.5 kJ/mol) for CH₄ production fall on the lower end of values reported in a global meta-analysis of wetland, rice paddies, and aquatic ecosystem methane emissions (15–85 kJ/mol) (Yvon-Durocher et al. 2014). This could reflect site-specific differences in temperature regimes, as many wetlands in the meta-analysis experience warmer mean annual temperatures compared to our subalpine system. For instance, boreal and temperate wetlands in the meta-analysis showed higher Q_{10} values (2.3–5.8) compared to our measurements (1.2–1.4), suggesting that methanogens in our system may be adapted to function under wider temperature fluctuations typical of subalpine environments. The similarity in Q_{10} values between burned and unburned soils, and between shallow and deep depths, suggests that fire did not fundamentally alter the temperature sensitivity of the methanogenic communities, despite affecting overall CH₄ production rates.

Several methodological considerations should be noted when interpreting our results. The field sampling design, while capturing the natural mosaic of burned and unburned areas, included limited replication at the burn-status level, particularly with only one unburned reference site. While our mixed-effects modeling approach accounted for spatial and temporal dependencies, some findings should be interpreted as preliminary given these constraints. Future studies would benefit from greater spatial replication across

multiple independent fire events to strengthen statistical inference. Despite these challenges, these results have significant implications for ecosystem services under projected increases in wildfire frequency and intensity in forests and uplands (Abatzoglou and Williams 2016). Higher Q_{10} values for Fe(III) reduction for burned soils suggest that warming could disproportionately accelerate carbon mineralization in fire-affected wetlands, potentially shifting these ecosystems from net carbon sinks to sources during freeze–thaw cycles (not part of this study!) and higher annual temperatures in the future (Turetsky et al. 2020). Higher DOC export rates, as documented in our study, pose a challenge to downstream water treatment plants, particularly as subalpine, forested wetlands provide critical drinking water across the western United States (Emelko et al. 2011; Writer et al. 2014). Observed suppression of methanogenesis coupled with enhanced CO₂ production for burned soils in our experiments represents a shift in wetland greenhouse gas dynamics that could alter regional carbon budgets and influence climate feedback mechanisms (Bridgman et al. 2013). As wildfire–wetland interactions become increasingly common, more research is needed in northern boreal and arctic wetlands during freeze–thaw cycles (Schoor and Mack 2018), longitudinal studies to determine the persistence of fire-induced changes in carbon fluxes (Kotze 2013), and sampling across different fire severity gradients and inputs, which can vary significantly in chemical composition (Sánchez-García et al. 2023).

Conclusions

Wildfires are known to increase emissions of GHGs and lead to elevated dissolved organic carbon concentrations due to high-temperature oxidation of accumulated organic matter in wetland soils (Osborne et al. 2013). It is less clear whether these emissions persist post-fire, both on shorter and longer temporal scales. Our experiments fill a critical gap in literature by shedding light on short-term greenhouse gas dynamics via slurry and flow-through reactor soil experiments. Broadly, it appears that the fire in wetland soils alters pore-water chemistry and increases redox potential leading to increased mineralization of carbon, dissolved ferrous iron, nitrous oxide, and suppression

of methane compared to unburned soils. Additionally, temperature variation interacts with wildfire inputs to accelerate the dissolution of ferric oxides, leading to higher production rates of dissolved carbon and GHGs. This underscores the importance of fire-derived organic matter in shaping biogeochemical processes and microbial activity in these ecosystems.

Overall, this analysis highlights the intricate coupling between carbon and iron cycling in wetland soils and how this relationship is influenced by fire events. Additionally, high DOC production rates post-fire poses a challenge to environmental engineers and water filtration systems for cities dependent on subalpine wetlands for their drinking water. It also underscores the importance of post-fire sampling and consideration of short-term GHG production in response to wildfires. Subalpine wetlands have been considered carbon sinks due to their lower temperatures and high gross primary productivity (Heikkinen et al. 2002; Scott and Wohl 2017), which makes this work important both for the continued ability of these systems to sequester carbon, and broadly for the global carbon cycle.

Acknowledgements We would like to thank Timothy Fegel and the Biogeochemistry Lab at the US Forest Service Rocky Mountain Research Station for stream and porewater analyses. We would like to acknowledge the Silver Lab at UC Berkeley for their generosity with the usage of their elemental analyzer and gas chromatograph.

Funding The research was funded by T.B. from the National Science Foundation under Grant Number 2114868 and the United States Department of Agriculture (USDA) National Institute of Food and Agriculture through AFRI Grant Number 2021-67019-34608.

Data availability Data and code can be downloaded and accessible at <https://github.com/pranjaldwiv38/Wetland-biogeochem-data>.

Declarations

Conflict of interest The authors have no relevant financial interests implicated by this work.

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