ARTICLE | Article in Press

Low methane emissions following experimental bog rewetting

Rodney A. Chimner · Jason K. Keller · Randy K. Kolka · Erik A. Lilleskov · Julia Stuart · Rachel Tarpey · Colin L. Tucker · Cassandra A. Zalman · Louis J. Lamit · Evan S. Kane

Received: 12 July 2026 / Accepted: 21 October 2026
© The Author(s) 2026

Abstract
Peatlands cover only 3–4% of the world's land surface, but store one-third of global soil carbon. Undisturbed peatlands act as long-term net carbon sinks, but draining peatlands can convert them into net sources of greenhouse gases. Rewetting drained peatlands can improve CO_2 sequestration; however, there is concern that rewetting produces additional CH_4. Our experimental bog restoration project found that rewetting drained peat resulted in only a modest increase in CH_4 emissions, which was still lower than undrained natural peat at similar water table levels. We also saw an increase in CO_2 uptake. We hypothesize that peat chemistry is limiting CH_4 emissions after rewetting, with strong differences between the peat's physical and chemical characteristics in the rewetted vs. reference treatments. Our results, and others, indicate that rewetting drained peatlands will likely increase CH_4 efflux, but post-restoration emissions typically remain below those of naturally occurring peatlands, suggesting that CH_4 concerns are not necessarily a barrier to restoration if done correctly.

Keywords Restoration · Methane · Peatlands · Hydrology · Peat Chemistry · Rewetting

Introduction

Peatlands cover 3–4% of the world's land surface but store one-third of global soil carbon, estimated at 450,000 to 650,000 million tons (Mt)[1]. Undisturbed peatlands act as long-term net carbon sinks, but degradation can convert them into net sources of greenhouse gases (GHG), estimated at 2,000 Mt of carbon dioxide equivalent of greenhouse gases annually, because of microbial oxidation of soil carbon. This is 4% of all non-fire anthropogenic emissions[1]. Drying out peatlands by ditching is one of the most common disturbances to peatlands globally, representing ~ 12% of all global peatlands[1]. Ditches are used to lower the water table, allowing peatlands to be used for agriculture, peat mining, horticulture, grazing, forestry, and other uses. The lowered water table causes the peat surface to subside and the peat to oxidize[2]. Draining peatlands can also increase dissolved organic carbon export and shift vegetation composition[3], while also increasing peatland susceptibility to fire and subsequent carbon loss[4].

This Article in Press is shared early to give you faster access to new research. It is citable and carries a permanent DOI. The final edited version will replace it automatically.

https://doi.org/10.1038/s41598-026-77546-x

Published online: 27 October 2026

scientific reports

Because of the GHG emissions associated with peatland disturbance, peatland restoration is a critical component of Natural Climate Solutions[5–7]. Restoration through rewetting can significantly reduce CO_2 emissions by re-establishing anoxic conditions that preserve soil C and improve hydrologic conditions[8]. However, rewetting may have the unintended consequence of increasing methane (CH_4) emissions, a greenhouse gas with a global warming potential 28–36 times greater than CO_2 over a 100-year time horizon[8]. Methane production occurs under anoxic conditions typical of peatlands and, thus, generally increases at higher water table levels[9]. Consequently, there is concern that rewetting peatlands during restoration could lead to significant CH_4 emissions, potentially offsetting the climate benefits of restoration[10]. However, hydrology is not the sole determinant of CH_4 emissions; peat chemistry, vegetation, and microbial communities also play key roles. Many peatland restoration projects target sites drained decades ago, which have altered the peat's physical and chemical properties, and altered microbial communities, which may, in turn, affect CH_4 emissions following rewetting[11].

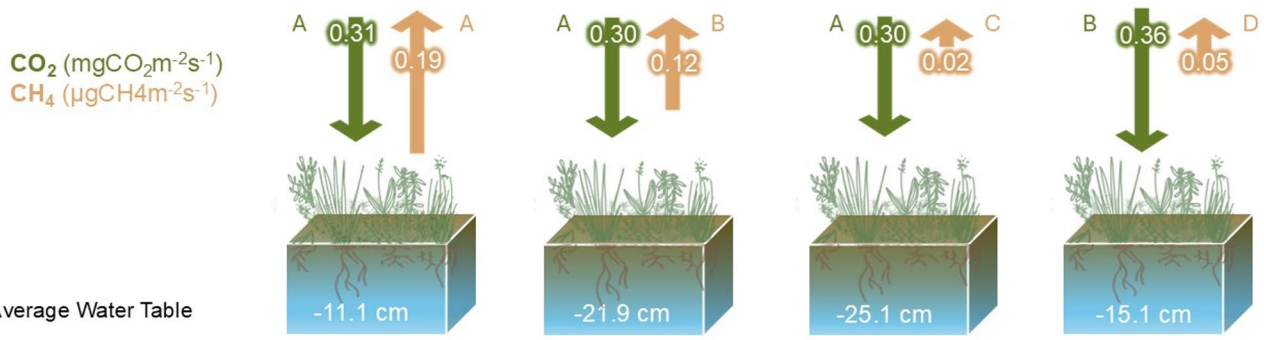
Long-term drainage alters the biochemical composition of peat substrate by converting it from labile, easily fermentable compounds such as polysaccharides toward more recalcitrant, aromatic compounds including lignin, phenolics, and lipid/wax fractions[12]. This shift is compounded when drained peatlands become colonized by trees, since the dominant plant functional type is a strong driver of peat quality. Tree-derived peat has lower quality than *Sphagnum*-derived peat, with higher carbonyl/lignin and humification indices and lower lipid/polysaccharide ratios compared to moss-derived peat[13]. Because methanogens are strict anaerobes reliant on simple substrates supplied by fermentative and syntrophic bacteria breaking down labile organic matter this compounded decline in substrate lability can constrain methanogenesis independent of redox status alone[14]. As a result, rewetting may restore anoxic conditions favorable to methanogens without restoring a substrate of comparable quality to the original *Sphagnum* peat, which could cause a low CH_4 response following rewetting of drained peat.

We tested how rewetting degraded peat affects GHG emissions in a mesocosm experiment with 24 *Sphagnum* bog monoliths, half collected from an open-canopy undrained reference bog and half from adjacent forested drained peat. We manipulated water tables to create four treatments ($n = 6$ each; Fig. 1): *Undrained-Reference* (reference peat, high water table matching the reference site), *Undrained-Dry* (reference peat, low water table), *Drained-Control* (drained peat, low water table), and *Drained-Rewetted* (drained peat, water table raised to reference levels to mimic restoration rewetting).

Results and discussion

The *Drained-Rewetted* treatment raised the average snow free water table to only 4 cm lower than the *Undrained-Reference* treatment (Fig. 1), depths that are within our target range of the hydrologic reference bog[15], and the range of many boreal peatland restoration projects[16]. The water tables were not identical even though they received the same input of water, partially because of the bulk density and pore space differences between the drained and undrained peat (Fig. 1). The average snow free water table of the *Undrained-Dry* and *Drained-Control* treatments were 10–15 cm lower than the *Undrained-Reference* and were 20–30 cm lower during dry periods. Overall, the range of water table levels across treatments and seasons varied between 10 above the soil surface to ~80 cm below the surface (Fig. 2).

Log-transformed CH_4 emissions differed significantly among all four treatments (mixed model, $p < 0.001$: Tukey, $p \leq 0.003$). Peat type, water table treatment, and the interaction all had significant ($p < 0.001$) effects on CH_4 emissions. The *Undrained-Reference* treatment had the highest mean CH_4 emissions, followed by *Undrained-Dry* (Fig. 1). Methane emissions increased in the *Drained-Rewetted* treatment compared to the *Drained-Control* treatment, however the mean CH_4 was still ~4–5 times lower than the *Undrained-Reference* or *Undrained-Dry* treatments (Fig. 1). Emissions of CH_4 varied with water table levels within the drained and undrained peat types, with greater variation in the undrained peat (Fig. 2). The mixed model for mid-day CO_2 flux indicated that the *Drained-Rewetted* treatment had significantly higher mean uptake of CO_2 flux (8.47 ± 5.64) than all other treatments (Fig. 1: Tukey, $p \leq 0.017$) with no significant interaction terms ($p = 0.087$).



	Undrained-Reference	Undrained-Dry	Drained-Control	Drained-Rewetted
Bulk Density (0-40 cm)	0.09[a]	0.09[a]	0.08[a]	0.1[a]
Bulk Density (60-80 cm)	0.11[a]	0.13[a]	0.11[a]	0.13[a]
C:N Ratio (0-40 cm)	39.9[a]	42.3[a]	34.9[b]	35.0[b]
C:N Ratio (60-80 cm)	32.8[a]	32.3[a]	31.2[a]	33.1[a]
von Post Scale (20-40 cm)	2.0[a]	2.1[a]	2.6[b]	2.7[b]
Polysaccharides:lignin (0-40)	1.22		1.07	
CH_4 Potential (0-40 cm)	2.72[a]	3.65[a]	0.62[a]	1.41[a]
CH_4 Potential (40-80 cm)	1.39[a]	1.88[a]	0.99[a]	1.68[a]
CO_2 Potential (0-40 cm)	175.53[a]	174.15[a]	87.16[b]	109.02[ab]
CO_2 Potential (40-80 cm)	55.60[a]	67.24[a]	70.75[a]	66.51[a]
% Methanogen (0-30 cm)	1.12	0.79	0.29	2.65
% Methanogen (30-70 cm)	2.44	2.52	0.70	0.85
Mgen/Mtroph (0-30 cm)	1.49	1.80	1.60	7.70
Mgen/Mtroph (30-70 cm)	0.95	1.35	0.34	0.57
DOM redox index (40 cm)	0.18[a]	0.16[a]	0.21[b]	0.20[b]

Fig. 1 Conceptual and quantitative summary of greenhouse gas fluxes, radiative forcing, hydrology, and peat biogeochemical properties across peatland degradation and restoration treatments. Panels represent four peatland conditions: *Undrained-Reference*, *Undrained-Dry*, *Drained-Control*, and *Drained-Rewetted*. For each treatment, arrows indicate the direction and magnitude of 5-year average mean surface–atmosphere fluxes of mid-day CO_2 (mg CO_2 m^{-2} s^{-1}; green), and CH_4 (µg CH_4 m^{-2} s^{-1}; orange), with numeric values shown above each panel. Letters denote significant differences $p<0.05$). Boxes illustrate 5-yr mean growing-season water table depth (cm relative to peat surface), with deeper water tables shown as larger unsaturated zones. Table below summarize key peat physical, chemical, and microbial indicators by depth, including bulk density (g cm^{-3}), C: N ratios, von Post humification index, polysaccharide: lignin ratios, CO_2 and CH_4 production potentials (mmol C $gram^{-1}$), methanogen abundance (%), methanogen: methanotroph (Mgen/Mtroph) ratios, and DOM redox index.

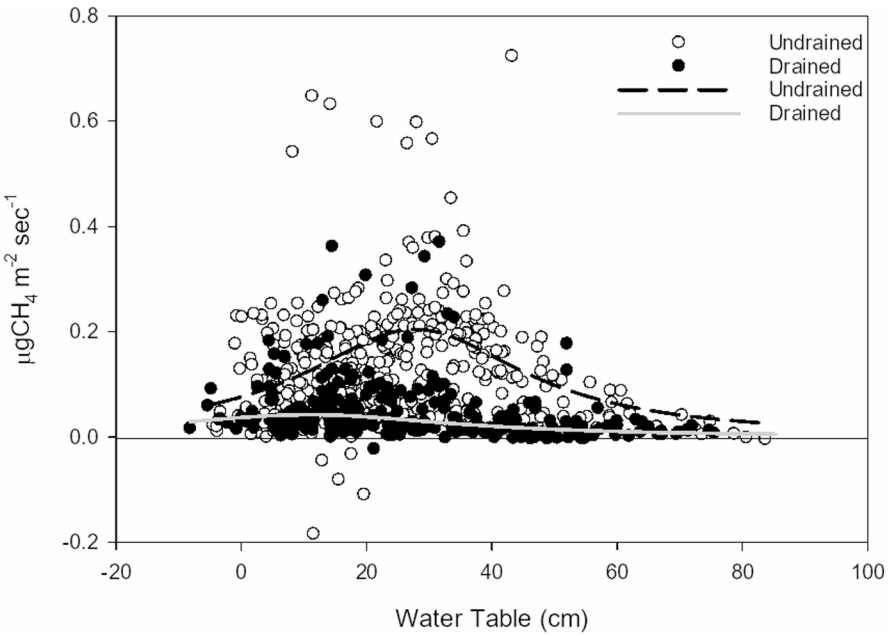


Fig. 2 Five-year record of temperature-corrected (15 °C) methane (CH_4) flux by water table level (negative values indicate water table is above ground surface) grouped by drained and undrained treatments. There are overall significant differences in CH_4 emissions between *Undrained* and *Drained* peat types, but not CH_4 emissions between treatments within the same peat type (e.g., *Undrained-Reference* vs. *Undrained-Dry*, *Drained-Control* vs. *Drained-Rewetted*) (Fig. 1).

We hypothesize that the difference in peat chemistry is partly responsible for the low CH_4 emissions after rewetting. The peat's physical and chemical characteristics were similar between the *Undrained-Reference* (undegraded) and *Drained-Control* (ditched and degraded) peats at the 40–80 cm depth (Fig. 1), but in the top 40 cm, drained peat was more modified, exhibiting greater von Post humification and a lower C: N ratio. Fourier Transform Infrared (FTIR) Spectroscopy further indicated that drained peat was more humified and recalcitrant, with a lower polysaccharide-to-lignin ratio (Fig. 1). The Q_{10} temperature coefficient for net CH_4 emissions also differed between peat types: *Undrained-Reference* peat had a Q_{10} of 1.88 compared to 1.16 for *Drained-Control* peat, indicating that CH_4 emissions increase more rapidly with temperature in undrained peat. This pattern is consistent with electron donor limitation with reduced availability of labile substrates such as H_2 and acetate in the more humified drained peat likely constrains methanogenic activity even after rewetting. Incubations support this interpretation as a mixed model revealed significant overall differences in CH_4 and CO_2 production potential between surface undrained and drained peats ($p < 0.02$), but not water treatments, though few pairwise treatment differences reached significance in post-hoc comparisons (Fig. 1). The porewater DOM redox index (RI) was also significantly higher in drained peat (0.20–0.21) than in undrained peat (0.16–0.18), with rewetted sites remaining elevated (0.20), consistent with greater oxidation of porewater DOM under drainage and incomplete chemical recovery following rewetting (Fig. 1).

The lack of correlation between methanogen relative abundance at the surface and methane emissions suggests the importance of other factors than relative abundance of methane producers and consumers in regulating efflux. It is plausible that peat quality, rather than the methanogen community, limits CH4 production after rewetting.[17–19]. The *Drained-Rewetted* surface soils had the highest proportion of methanogen reads and the highest methanogen: methanotroph ratio (Fig. 1), yet this treatment combination had a very low CH_4 production potential compared with the undrained-control. Methanogens depend on fermentation products derived largely from labile polysaccharides. A plausible hypothesis is that the more humified, polysaccharide-poor drained peat (Fig. 1) lacked sufficient labile electron donors, restricting CH_4 production, and leaving an abundant but substrate-limited methanogen community. By contrast, in undrained peat, methanogens were most abundant in deeper, less-degraded peat, suggesting this zone supplies most of the higher CH_4 efflux. Similar substrate limitation has been reported for restored cutover bogs[20,21], whereas fresh labile inputs after rewetting can drive high emissions[22]. Changes in plant community composition under long-term drainage, particularly the loss of *Carex oligosperma* and its dense aerenchymatous root system, may also contribute to the suppressed methane efflux, though separating this effect from substrate-driven controls was beyond the scope of this study.

Our findings suggest that peat quality could play a key role in CH_4 emissions after rewettimg[22], as our *Drained-Rewetted* treatment exhibited CH_4 emissions 4–5 times lower than our *Undrained-Reference* treatment, at similar water table levels. However, in our experiment, water tables in the rewetted treatment remained below the peat surface, maintaining a shallow oxic surface layer, so our results may not represent flooded conditions. Although the surface drained peat in this study was degraded relative to the reference peat due to being drained for ~80 years, with greater humification, lower C: N ratios, and a lower polysaccharide-to-lignin ratio, the degradation reflects drainage and tree encroachment rather than more intensive land uses. Peat that is more severely degraded, or nutrient-enriched, such as in former agricultural sites, may respond differently to rewetting.

Peatland restoration is a complex process, encompassing a wide range of peatland types, disturbances, and restoration techniques. Rewetting drained peatlands by blocking or filling ditches is among the most common peatland restoration approaches[23]; however, its effects on greenhouse gas (GHG) dynamics are influenced by multiple interacting factors. A body of research, including this study, is beginning to unravel some patterns that can be used to establish general principles to inform peatland restoration projects from a GHG perspective. Rewetting drained peatlands typically enhances CO_2 uptake[24,25], but the overall short-term climate forcing is largely dictated by the extent of CH_4 emissions following rewetting[8,26], which can vary significantly[27–29]. Site wetness is one of the strongest predictors of CH_4 emissions post-restoration[25,30], with standing water often associated with elevated CH_4 fluxes[31,32]. Consequently, restoration strategies should aim to maintain saturated peat without excessive flooding[33], which can be challenging when blocking ditches in subsided areas or in degraded peat with low pore



Fig. 3 Mesocosm Facility at the USDA Forest Service Northern Research Station Forestry Sciences Laboratory in Houghton, MI, USA.

space[34]. Notably, several studies, including this one, have found that restored peatlands rewetted to a near-surface water table can emit less CH_4 than natural reference peatlands[20,35], whereas rewetting that flood the surface can emit CH_4 well above natural levels[22,29,36]. These findings suggest that CH_4 emissions following rewetting are likely to be of limited concern where drainage has reduced substrate quality, but water tables are held near, but below, the surface.

Methods

Peat harvesting and project design

In August 2017, we harvested 24 intact 1 m · 1 m · 0.9 m peat monoliths (Fig. 3) near Meadowlands, MN (47.116195° N, -92.796530° W). We used a custom-designed steel peat cutter that minimized compaction and disturbance to transfer peat monoliths to the mesocosm bins for prompt transportation to the USDA Forest Service Mesocosm Facility in Houghton, MI. Half of the harvested monoliths came from a forested site adjacent to a drainage ditch created before 1939, while the other half came from a reference open canopy, undrained site, located more than 300 m from the drainage ditch. The open canopy, undrained site was dominated by *Sphagnum* mosses, ericaceous shrubs (*Chamaedaphne calyculata* L.,Moench., *Kalmia polifolia* Wangenh., and *Vaccinium oxycoccus* L.), sedges (*Carex oligosperma* Michx.), and small, widely scattered black spruce (*Picea mariana* (Mill.) Britton, Sterns & Poggenb.) and tamarack (*Larix laricina* (Du Roi) K. Koch) trees. The forested, drained site was an open peatland before the 1930s, but by the 1980s experienced tree encroachment. The forested, drained site was vegetated by a dense overstory of black spruce and tamarack, with a minor understory of shrubs

(mainly *Vaccinium myrtilloides* Michx. and *Rhododendron groenlandicum* (Oeder) Kron & Judd), and a moss layer dominated by *Hylocomium splendens*. The peat from the forested, drained site has experienced long-term (~80 years) impacts of drainage and subsequent tree encroachment, thus we refer to that peat as "drained", and that from the open canopy, undrained site as "undrained". For further information on characteristics of the original harvest sites, see[37,38].

Following installation in the mesocosm facility, we manipulated water table levels to either replicate the average water table depth at the open canopy, undrained site (a target of 15 cm depth at mid-growing season) or to drop the water table level to replicate hydrological conditions at the forested, drained site (target of 35 cm depth at mid-growing season)[15]. Specifically, in drained peat, water table levels were either kept low (*Drained-Control* treatment) or raised (*Drained-Rewetted* treatment*)* to meet average water table levels at the open canopy, undrained site (*Undrained-Reference*), mimicking rewetting in restoration practices[39]. In undrained peat, water tables were either lowered (*Undrained-Dry* treatment) or kept at the average level (*Undrained-Reference* treatment). We achieved these four treatment categories (*Undrained-Reference*, *Undrained-Dry*, *Drained-Control*, and *Drained-Rewetted*) by strategically draining water from the acrotelm-catotelm boundary area (primarily in the spring), placing clear corrugated fiberglass rain-out shelters during precipitation events, or adding artificial rainwater that resembles local precipitation based on averages from the nearby National Atmospheric Deposition Program site (https://nadp.slh.wisc.edu/networks/national-trends-network/, 2018–2024). We measured water table depths co ntinuously throughout the growing season using submersible vented pressure transducers (Grainger 2HMC7) in wells in each bin from 2018 to 2022[40]. We manually measured water levels in the wells with a blow tube surface detection method along a meter stick from 2023 to 2024. Water additions and rainouts were applied uniformly to the wet (*Undrained-Reference*, *Drained-Rewetted*) and dry (*Undrained-Dry*, *Drained-Control*), resulting in some variation in emergent water table based on differences in bulk density and/or evapotranspiration.

Flux measurements

Carbon dioxide and CH_4 gas flux measurements occurred at least biweekly during the growing seasons of 2020–2024. Carbon dioxide and CH_4 were measured simultaneously using a clear acrylic chamber (6280 cm^3) constructed to fit tightly to the edge of the mesocosm bins, with closed-cell foam around the bottom edge creating an air-tight seal. A hatch was built into the top of the chamber to allow ventilation between sequential measurements on the same bin. Two computer CPU fans mixed the air in the chamber during measurements. The change in CO_2 and CH_4 concentration in the chamber was measured for 2 min using the Li-Cor 7810 CH_4/ CO_2/H_2O Trace Gas Analyzer (Li-Cor Environmental, Lincoln, NE, USA). All measurements were conducted between 9 a.m. and 3 p.m. during sunny conditions to ensure maximum sunlight conditions. Both CH_4 and CO_2 flux estimates were converted to an aerial basis (mg CO_2 m^{-2} s^{-1} or µg CH_4 m^{-2} s^{-1}. To compare CH_4 emissions with water table levels, CH_4 fluxes were normalized to a common temperature (15 °C) using a Q_{10} correction for each peat type (*Drained* and *Undrained)* to account for the exponential temperature dependence of CH_4 production.

Treatment effects on CO_2 uptake and CH_4 emissions were analyzed separately using linear mixed-effects models with peat type (undrained, drained), water table treatment (reference, drought), and their two-way interaction as fixed effects, and mesocosm bin as a random effect to account for repeated measurements over time. CH_4 emission data were log-transformed prior to analysis to improve normality. Pairwise comparisons among treatment cells were conducted using Tukey's honestly significant difference (HSD) post-hoc test. All analyses were conducted in SYSTAT 13.2 (Systat Software Inc. (2009), San Jose, CA, USA, https://grafiti.com/systat) with statistical significance assessed at $\alpha = 0.05$.

Soil and molecular methods

In August 2023, we collected a peat core (3.5 × 70 cm) from each of the 24 experimental bins. Upon collection, cores were cut in half lengthwise and sectioned into 10-cm-depth increments (0–10, 10–20, 20–30, 30–40, and

60–70 cm). From each section, a representative subsample (5 g) was collected by hand using sterile gloves, wrapped in plastic and foil, stored in liquid nitrogen for 1–3 d, and stored at -80 °C for 4 months until processing. A subsample from the remaining peat from each section was analyzed for total carbon and nitrogen using a Thermo Scientific EA IsoLink connected to a Thermo Scientific Conflo IV Interface and a Delta V Continuous Flow-Stable Isotope Ratio Mass Spectrometer (Thermo Fisher Scientific, Waltham, MA, USA). Bulk density was calculated as the dry weight of peat divided by the volume. The degree of decomposition was determined using the von Post scale of humification[41]. The von Post scale consists of a H1-10 scale, with H1–3 corresponding to fibric peat, H4–5 to hemic peat, and H7–10 to sapric peat. For more detailed methods refer to [38]. Treatment effects on bulk density and von Post were analyzed separately using linear mixed-effects models with peat type (undrained, drained), water table treatment (reference, drought), and their two-way interaction as fixed effects, and mesocosm bin as a random effect.

Fourier Transform Infrared (FTIR) Spectrometry was performed on peat samples collected from 0 to 40 cm in summer of 2022[42]. We obtained FTIR spectra for each sample using a Thermo Scientific Nicolet iS5 Spectrometer from 400 to 4000 cm^{-1} range with a resolution of 4 cm^{-1} and a data interval of 0.5 cm^{-1} by averaging 64 scans[42]. Following application of automatic baseline in the proprietary software, we used a custom Python 3.0 code to correct and standardize the spectra as previously described[42]. The ratio of relative absorbance at 1060/1620 (polysaccharides/lignin)[43] was used to calculate an index of peat quality. Peak height calculations are strongly correlated to peak areas ($r^2 > 0.99$), providing a reliable semiquantitative interpretation of constituent concentrations[44].

DNA extraction methods were performed as described by[38]. Peat for the focal dataset presented here was collected in 2024 and processed from the 0–10, 10–20, 20–30, 30–40, and 60-70-cm depth increments (118 samples total: 24 bins x 5 depths and 2 missing sample depths). Frozen peat samples were thawed and ground using a mini bead beater (Biospec Products, Inc., Bartlesville, OK USA). Samples were kept frozen until DNA extraction. DNA was extracted from 250 mg of frozen ground peat using the DNeasy PowerSoil Pro Kit (Qiagen) according to the manufacturer's instruction, and quantified using a Qubit Fluorometer (Model Q32857, Invitrogen). The V4 region of the bacterial and archaeal 16 S gene was PCR amplified using primers 515 F (Parada) and 806R (Apprill)[45,46] following[47,48]. Multiplexed libraries were sequenced on an Illumina MiSeq platform (San Diego, CA USA) using 2 × 250 bp chemistry (MiSeq Reagent Kit V2, MS-102-2003; spiked with 10% PhiX) by theEnvironmental Sample Preparation and Sequencing Facility at Argonne National Laboratory (https://www.anl.gov/bio/environmental-sample-preparation-and-sequencing-facility).

Bioinformatic processing of 16 S DNA sequences focused on extracting methanogen and methanotroph relative abundances to link to ecosystem processes. First, BBDuk was used to remove Illumina adapter and PhiX 174 contamination (https://sourceforge.net/projects/bbmap/), and PCR primers (5′ artifacts and 3′ readthrough) were trimmed with cutadapt 3.2[49]. The resulting sequences were imported into Qiime2[50], were the DADA2[51] plug-in was used for joining forward and reverse reads, quality filtering (expected error rate = 2), chimera removal and ASV (Amplicon Sequence Variant) generation. Taxonomy was assigned with the SIlVA-138-99 dataset[52] containing all eukaryotes and prokaryote 18 S/16S sequences using the Qiime2 naïve Bayes feature-classifier (Bokulich et al. 2018); reads matching mitochondria, chloroplasts and eukaryotes were filtered from the dataset. Sequence read depth was then equalized among observations to 9978 sequences by scaling with ranked subsampling[53]. Methanogenic archaea and aerobic methanotrophic bacteria were identified using FAPROTAX 1.2.11[54], then sequences from ASV were summed by functional group and expressed as relative proportions of the observation total (i.e., 9978). Treatment effects on methanogens were analyzed separately using linear mixed-effects models with peat type (undrained, drained), water table treatment (reference, drought), and their two-way interaction as fixed effects, and mesocosm bin as a random effect using JMP version 18 (JMP Statistical Discovery LLC, Cary, NC, 2024).

In September 2019, we collected pore water samples from 40 to 70 cm below the peat surface from previously installed monitoring wells into airtight 20 mL syringes. Monitoring wells were enmeshed with 120-µm Nitex mesh to prevent intake of coarse particulate matter. Ultraviolet-visible absorbance and fluorescence spectra (EEMs, Excitation-Emission Matrices) were measured using a Horiba-Jobin Yvon Aqualog C (Horiba Corp.)

with the following run parameters - Excitation: 240 to 600 nm by 3-nm increments; Emission: 212 to 608 nm by 3 nm bandpass; integration time = 0.25 s. Samples with absorbance greater than 0.6 at λ_{254} (A_{254}> 0.6) were diluted with DI H_2O to satisfy the assumption of detector linearity. Resulting UV-Vis data were corrected using the inverse of diluted concentration for data analysis. EEM fluorescence intensities were normalized to the integral of the water Raman peak (Ex 350/Em 371–428) of the sample and then normalized the resulting fluorescence intensities to an external, ultrapure Raman peak water standard (Starna Cells, Atascadero, CA, USA). Excitation-emissions matrices were analyzed for an index of DOM redox status (RI) using the area in a 10 nm × 10 nm window under the component peak maxima (as a % total EEM fluorescence). For more details on these methods see[55]. Treatment effects on RI were analyzed using linear mixed-effects models with peat type (undrained, drained), water table treatment (reference, drought), and their two-way interaction as fixed effects, and mesocosm bin as a random effect.

CO_2 and CH_4 production potential

Peat cores were collected from the mesocosms using a 2-cm diameter coring tube attached to an electric drill on August 3–4 of 2020. In each mesocosm, a 0–40 cm core was collected first, followed by a second 40–80 cm core from the same location. The coring tube was washed with tap water and then rinsed with distilled water between each collection. Upon collection, the peat was extruded from the coring tube and when compaction occurred (common in the surface depths), the peat core was 'stretched' to the full 40 cm range of the core. All cores were subsequently cut into 10-cm depth increments (i.e., 0–10, 10–20, 20–30, 30–40, 40–50, 50–60, 60–70, and 70–80 cm increments from all mesocosms), wrapped in plastic wrap and then foil before being frozen in liquid nitrogen and transferred to a -80 °C freezer. On September 22 of 2020, frozen samples were shipped overnight to Chapman University (Orange, California) on ice. Upon arrival, they were stored at -20 °C until processing.

For logistical reasons, samples were processed and analyzed in batches, with each batch containing all samples from a given depth increment in the order 20–30, 60–70, 40–50, 0–10, 10–20, 30–40, 50–60, 70–80. For each depth increment, samples were removed from the freezer and allowed to thaw and oxidize for 24 h in the dark at 5 °C. After thawing, samples were quickly processed to remove large roots, living vegetation, and woody material and allowed to continue oxidizing in the dark at 5 °C for an additional 24 h. After this second day of oxidation, 3 g of homogenized, field-moist peat were added to 72-mL serum bottles and slurried with 20 mL of deionized water. Due to low density peat in the 0–10 cm depth increment, slurries were created with 2 g of homogenized, field-moist peat. Additional subsamples of field-moist peat were dried to constant mass at 60 °C to calculate percent moisture. Serum bottles were capped with gray, butyl septa and flushed with N_2 for 15 min to establish anaerobic conditions before being incubated at 15 °C in the dark for 8 weeks.

At the end of the 8-week incubations, headspace concentrations of CO_2 and CH_4 were measured using a gas chromatograph equipped with a flame ionization detector and an in-line methanizer to convert CO_2 to CH_4 (SRI 8610 C, SRI Instruments, Torrance, California). Dissolved CO_2 and CH_4 were calculated using Henry's Law adjusting for headspace pressure, temperature, and pH[56]. Greenhouse gas production is expressed as mmol C $gram^{-1}$ dry weight of peat.

Data presented in Fig. 1 are the average of all available samples between 0 and 40 cm (i.e., 0–10, 10–20, 20–30, and 30–40 cm depth increments) and 40–80 cm (i.e., 40–50, 50–60, 60–70, and 70–80 cm depth increments). As a result of incomplete peat recovery at depth upon coring and limited sample mass in the low density 0–10 and 10–20 cm depth increments, not all 48 potential samples (12 bins x 4 depths) were available for analysis. Undrained bins included 46 samples in the 0–40 cm range and 44 samples in the 40–80 cm range whereas drained bins included 38 samples in the 0–40 cm range and 47 in the 40–80 cm range. Treatment effects on CO_2 to CH_4 potential at 0–40 cm and 40–80 cm were analyzed separately using linear mixed-effects models with peat type (undrained, drained), water table treatment (reference, drought), and their two-way interaction as fixed effects, and mesocosm bin as a random effect.

Acknowledgements We appreciate the assistance of John Hribljan, Maxwell Wegener, Monica Heirman, Camille DeFrenne, Dominic Uhelski, Ally O'Neill, Kaitlyn Dodge, Jesse Barta, Matthew Hennessey, Liam Krause, Lukasz Spiewla, Maddie Peterson, Nora Sullivan and many undergraduate students.

Author contributions The study was conceived by E.A.L., E.S.K., R.A.C., and R.K.K. C.L.T. and R.T. were responsible for the collection and analysis of flux data, with additional analysis by R.A.C. E.A.S., R.T., C.L.T, and E.S.K. led the collection and analysis of peat biogeochemical properties, while J.K.K. and C.A.Z. led the analysis of CO2 and CH4 production potential. E.A.L. led the collection and analysis of the microbial community data, with bioinformatic help from L.J.L. R.A.C. led the paper with contributions from all authors. All authors read and approved of the final manuscript.

Funding Funding for this research was provided by NSFDEB grants 2031076 and 2031085. There was in-kind support from the USDA Forest Service, Northern Research Station.

Data availability Data incorporated in this study are available in the public domain through Pangaea (https://doi.org/10.1594/PANGAEA.960334). Sequence data are archived with the National Center for Biotechnology Information (NCBI) Sequence Read Archive under accession number PRJNA1497992.

Declarations

Competing interests The authors declare no competing interests.

Open Access This article is licensed under a Creative Commons Attribution-NonCommercial-NoDerivatives 4.0 International License, which permits any non-commercial use, sharing, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons licence, and indicate if you modified the licensed material. You do not have permission under this licence to share adapted material derived from this article or parts of it. The images or other third party material in this article are included in the article's Creative Commons licence, unless indicated otherwise in a credit line to the material. If material is not included in the article's Creative Commons licence and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this licence, visit http://creativecommons.org/licenses/by-nc-nd/4.0/.

References

1. UNEP. Global Peatlands Assessment – The State of the World's Peatlands: Evidence for Action toward the Conservation, Restoration, and Sustainable Management of Peatlands. (2022).
2. Krause, L. et al. Impacts of historical ditching on peat volume and carbon in northern Minnesota USA peatlands. *J. Environ. Manage.* **296**, 113090 (2021).
3. Holden, J., Evans, M. G., Burt, T. P. & Horton, M. Impact of land drainage on peatland hydrology. *J. Environ. Qual.* **35**, 1764–1778 (2006).
4. Benscoter, B. W., Greenacre, D. & Turetsky, M. R. Wildfire as a key determinant of peatland microtopography. *Can. J. For. Res.* **45**, 1132–1136 (2015).
5. Griscom, B. W. et al. Natural climate solutions. *Proc. Natl. Acad. Sci. U S A*. **114**, 11645–11650 (2017).
6. Leifeld, J. & Menichetti, L. The underappreciated potential of peatlands in global climate change mitigation strategies. *Nat. Commun.* **9**, 1071 (2018).
7. Ury, E. A. et al. Strategic restoration of global inland peatlands can maximize local and global co-benefits. *Ecological Applications* **36**, (2026).
8. Günther, A. et al. Prompt rewetting of drained peatlands reduces climate warming despite methane emissions. *Nat. Commun.* **11**, 1644 (2020).
9. Turetsky, M. R. et al. A synthesis of methane emissions from 71 northern, temperate, and subtropical wetlands. *Glob Chang. Biol.* **20**, 2183–2197 (2014).
10. Milner, A. M. et al. Priority research questions in global peatland science. *Commun. Earth Environ.* **7**, 349 (2026).
11. Loisel, J. & Gallego-Sala, A. Ecological resilience of restored peatlands to climate change. *Commun. Earth Environ.* **3**, 1–8 (2022).
12. Hribljan, J., Kane, E. & Chimner, R. Implications of Altered Hydrology for Substrate Quality and Trace Gas Production in a Poor Fen Peatland Wetland Soils. *Soil Sci. Soc. Am. J.* https://doi.org/10.2136/sssaj2016.10.0322 (2017).

13. Uhelski, D. M., Kane, E. S. & Chimner, R. A. Plant functional types drive Peat Quality differences. *Wetlands* **42**, 51 (2022).
14. White, J., Shannon, R., Weltzin, J., Pastor, J. & Bridgham, S. Effects of soil warming and drying on methane cycling in a northern peatland mesocosm study. *J Geophys. Res* **113**, (2008).
15. Sebestyen, S. D. et al. Long-Term Monitoring Sites and Trends at the Marcell Experimental Forest. in Peatland biogeochemistry and watershed hydrology at the Marcell experimental forest (eds Kolka, R., Sebestyen, S., Verry, E. & Brooks, K.) 15–71 (CRC, Boca Raton, (2011).
16. Karimi, S., Maher Hasselquist, E., Salimi, S., Järveoja, J. & Laudon, H. Rewetting impact on the hydrological function of a drained peatland in the boreal landscape. *J. Hydrol. (Amst)*. **641**, 131729 (2024).
17. Reumer, M. et al. Impact of peat mining and restoration on methane turnover potential and methane-cycling microorganisms in a northern bog. *Appl Environ. Microbiol* **84**, (2018).
18. Pärn, J. et al. Soil moisture and microbiome explain greenhouse gas exchange in global peatlands. *Sci. Rep.* **15**, 10153 (2025).
19. Stuart, J. E. M. et al. Evidence for older carbon loss with lowered water tables and changing plant functional groups in peatlands. *Glob Chang. Biol.* **29**, 780–793 (2023).
20. Nugent, K. A. et al. Cutover Peat Limits Methane Production Causing Low Emission at a Restored Peatland. *J Geophys. Res. Biogeosci* **126**, (2021).
21. Urbanová, Z., Bárta, J. & Picek, T. Methane Emissions and Methanogenic Archaea on Pristine, Drained and Restored Mountain Peatlands, Central Europe. *Ecosystems* **16**, 664–677 (2013).
22. Hahn-Schöfl, M. et al. Organic sediment formed during inundation of a degraded fen grassland emits large fluxes of CH_4 and CO_2. *Biogeosciences* 8, 1539–1550 (2011).
23. Chimner, R. A., Cooper, D. J., Wurster, F. C. & Rochefort, L. An overview of peatland restoration in North America: where are we after 25 years? *Restor. Ecol.* **25**, 283–292 (2017).
24. Darusman, T., Murdiyarso, D., Anas, I. & Impron & Effect of rewetting degraded peatlands on carbon fluxes: a meta-analysis. *Mitig Adapt. Strateg Glob Chang.* **28**, 10 (2023).
25. Xu, S. et al. Responses of methane emissions to global wetland restoration and influencing factors. *Agric Meteorol* **364**, (2025).
26. Nugent, K. A. et al. Prompt active restoration of peatlands substantially reduces climate impact. *Environ. Res. Lett.* **14**, 287–295 (2019).
27. Zou, J. et al. Rewetting global wetlands effectively reduces major greenhouse gas emissions. *Nat. Geosci.* **15**, 627–632 (2022).
28. Wilson, S. et al. Greenhouse gas emission factors associated with rewetting of organic soils. *Mires Peat*. **17**, 1–28 (2016).
29. Abdalla, M. et al. Emissions of methane from northern peatlands: a review of management impacts and implications for future management options. *Ecology and Evolution* vol. 6 7080–7102 Preprint at (2016). https://doi.org/10.1002/ece3.2469
30. Yang, T. et al. Impact of Water Table on Methane Emission Dynamics in Terrestrial Wetlands and Implications on Strategies for Wetland Management and Restoration. *Wetlands* **42**, 120 (2022).
31. Rosentreter, J. A. et al. Half of global methane emissions come from highly variable aquatic ecosystem sources. *Nat. Geosci.* **14**, 225–230 (2021).
32. Ury, E. A., Zhang, Z. & Buma, B. Addressing methane emission feedbacks from global wetlands. *Nat. Sustain.* **8**, 1115–1118 (2025).
33. Zou, J. et al. Rewetting global wetlands effectively reduces major greenhouse gas emissions. *Nat. Geosci.* **15**, 627–632 (2022).
34. Schimelpfenig, D. W., Cooper, D. J. & Chimner, R. A. Effectiveness of Ditch Blockage for Restoring Hydrologic and Soil Processes in Mountain Peatlands. *Restor. Ecol.* **22**, 257–265 (2014).
35. Urbanová, Z., Bárta, J. & Picek, T. Methane Emissions and Methanogenic Archaea on Pristine, Drained and Restored Mountain Peatlands, Central Europe. *Ecosystems* **16**, 664–677 (2013).
36. Evans, C. D. et al. Overriding water table control on managed peatland greenhouse gas emissions. *Nature* https://doi.org/10.1038/s41586-021-03523-1 (2021).
37. Hupperts, S. F. & Lilleskov, E. A. Predictors of taxonomic and functional composition of black spruce seedling ectomycorrhizal fungal communities along peatland drainage gradients. *Mycorrhiza* **32**, 67–81 (2022).
38. Defrenne, C. E. et al. Peat loss collocates with a threshold in plant–mycorrhizal associations in drained peatlands encroached by trees. *New Phytol.* **240**, 412–425 (2023).
39. González, E., Henstra, S. W., Rochefort, L., Bradfield, G. E. & Poulin, M. Is rewetting enough to recover Sphagnum and associated peat-accumulating species in traditionally exploited bogs? *Wetl Ecol. Manag*. **22**, 49–62 (2014).
40. Romanowicz, K. J. et al. Understanding drivers of peatland extracellular enzyme activity in the PEATcosm experiment: mixed evidence for enzymic latch hypothesis. *Plant. Soil.* **397**, 371–386 (2015).
41. Clymo, R. A. & Peat in Gore, AJP (ed) Mires, swamp and fen: ecosystems of the world 4a, 159–224. Preprint at (1983).

42. Uhelski, D. M. et al. Pyrogenic carbon content of Sphagnum peat soils estimated using diffuse reflectance FTIR spectrometry. *Mires Peat*. **28**, 1–22 (2022).
43. Basiliko, N., Blodau, C., Roehm, C., Bengtson, P. & Moore, T. R. Regulation of Decomposition and Methane Dynamics across Natural, Commercially Mined, and Restored Northern Peatlands. *Ecosystems* **10**, 1148–1165 (2007).
44. Hodgkins, S. B. et al. Changes in peat chemistry associated with permafrost thaw increase greenhouse gas production. *Proc. Natl. Acad. Sci.* **111**, 5819–5824 (2014).
45. Parada, A. E., Needham, D. M. & Fuhrman, J. A. Every base matters: assessing small subunit<scp>rRNA primers for marine microbiomes with mock communities, time series and global field samples. *Environ. Microbiol.* **18**, 1403–1414 (2016).
46. Apprill, A., McNally, S., Parsons, R. & Weber, L. Minor revision to V4 region SSU rRNA 806R gene primer greatly increases detection of SAR11 bacterioplankton. *Aquat. Microb. Ecol.* **75**, 129–137 (2015).
47. Caporaso, J. G. et al. Ultra-high-throughput microbial community analysis on the Illumina HiSeq and MiSeq platforms. *ISME J.* **6**, 1621–1624 (2012).
48. Caporaso, J. G. et al. Global patterns of 16S rRNA diversity at a depth of millions of sequences per sample. *Proc. Natl. Acad. Sci.* **108**, 4516–4522 (2011).
49. Martin, M. Cutadapt removes adapter sequences from high-throughput sequencing reads. *EMBnet J.* **17**, 10 (2011).
50. Bolyen, E. et al. Reproducible, interactive, scalable and extensible microbiome data science using QIIME 2. *Nat. Biotechnol.* **37**, 852–857 (2019).
51. Callahan, B. J. et al. DADA2: High-resolution sample inference from Illumina amplicon data. *Nat. Methods*. **13**, 581–583 (2016).
52. Quast, C. et al. The SILVA ribosomal RNA gene database project: improved data processing and web-based tools. *Nucleic Acids Res.* **41**, D590–D596 (2012).
53. Beule, L. & Karlovsky, P. Improved normalization of species count data in ecology by scaling with ranked subsampling (SRS): application to microbial communities. *PeerJ* **8**, e9593 (2020).
54. Louca, S., Parfrey, L. W. & Doebeli, M. Decoupling function and taxonomy in the global ocean microbiome. *Sci. (1979)*. **353**, 1272–1277 (2016).
55. Kane, E. S. et al. Reduction-oxidation potential and dissolved organic matter composition in northern peat soil: interactive controls of water table position and plant functional groups. *J. Geophys. Res. Biogeosci*. https://doi.org/10.1029/2019jg005339 (2019).
56. Bridgham, S. D. et al. Organic Matter Mineralization and Decomposition. in (2013). https://doi.org/10.2136/sssabookser10.c20

Publisher's Note Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.

Authors and Affiliations

Rodney A. Chimner[1] · Jason K. Keller[2,3] · Randy K. Kolka[4] · Erik A. Lilleskov[5] · Julia Stuart[5,6] · Rachel Tarpey[5] · Colin L. Tucker[5,7] · Cassandra A. Zalman[3] · Louis J. Lamit[8] · Evan S. Kane[1,5]

✉ Rodney A. Chimner
rchimner@mtu.edu

[1] College of Forest Resources and Environmental Science, Michigan Technological University, Houghton, MI, USA

[2] Kravis Department of Integrated Sciences, Claremont McKenna College, Claremont, CA 91711, USA

[3] Schmid College of Science and Technology, Chapman University, Orange, CA 92866, USA

[4] USDA Forest Service-Northern Research Station, Grand Rapids, Minnesota 55744, USA

[5] USDA Forest Service-Northern Research Station, Houghton, MI 49931, USA

[6] USDA Forest Service-Mountain Planning Service Group, Lakewood, CO 80401, USA

[7] Mountain Studies Institute, 162 Stewart St. Unit A, Durango, Durango, CO 81303, USA

[8] College of Arts and Sciences, Syracuse University, 120 Crouse Drive, Syracuse, NY 13244, USA