Effects of nitrogen and phosphorus amendments on CO_2 and CH_4 production rates in peat soils of Scotty Creek, Northwest Territories, Canada: exploratory incubation results highlight a potential impact of ~~Complexity of nutrient enrichment on subarctic peatland soil CO_2 and CH_4 production under increasing~~ wildfires and permafrost thaw on peatland carbon exchanges

Eunji Byun[1], Fereidoun Rezanezhad[2], Stephanie Slowinski[2], Christina Lam[2], Saraswati Saraswati[2,3], Stephanie Wright[4], William L. Quinton[5], Kara L. Webster[6], Philippe van Cappellen[2]

[1]Department of Earth System Sciences, Yonsei University, Seoul, 03722, Korea
[2]Ecohydrology Research Group, Water Institute and Department of Earth and Environmental Sciences, University of Waterloo, Waterloo, Ontario, N2L 3G1, Canada
[3]Department of Geography and Geospatial Sciences, South Dakota State University, Brookings, SD 57007, USA
[4]Department of Civil Engineering, Queen's University, Kingston, ON K7L 3N6, Canada
[5]Cold Regions Research Centre, Wilfrid Laurier University, Waterloo, ON N2L 3C5, Canada
[6]Great Lakes Forestry Centre, Canadian Forest Service, Natural Resources Canada, Sault Ste. Marie, ON P6A 2E5, Canada

Correspondence to: Eunji Byun (eb@yonsei.ac.kr)

Abstract. ~~The adverse impacts of excessive soil nutrients on water quality and carbon sequestration have been recognized in tropical and temperate regions, with already widespread industrial farming and urbanization, but rarely in subarctic regions. However, recent studies have shown significant increases in porewater nitrogen (N) and phosphorus (P) concentrations in burned subarctic peatlands and downstream waters, which is a growing concern as climate change leads to increasing wildfires, permafrost thaws, and waterlogged peatlands. In this study, we present the results of a short-term incubation experiment conducted on soils from subarctic bogs and fens, aimed at evaluating the effects of high levels of nutrients on carbon gas production rates. We divided aliquots of the peatland soil samples into separate containers and added artificial porewater to each, enriching them with dissolved inorganic nitrogen (N), phosphorus (P), both, or none for controls. Overall, the fen samples showed higher carbon dioxide (CO_2) and methane (CH_4) production rates at 1, 5, 15, and 25°C compared to the bog samples, which we attributed to differences in soil properties and initial microbial biomass. The bog sample with added N produced more CO_2 compared to its control, while the fen sample with added P produced more CO_2 compared to its control. It was unexpected that the addition of both N and P reduced CO_2 but increased CH_4 production in both soils compared to their controls. After a month, the pore water C, N, and P stochiometric ratios approached the initial soil microbial biomass ratios, suggesting microbial nutrient recycling in an inherently nutrient-poor soil environment. These preliminary results imply a complex response of carbon turnover in peatland soils to nutrient enrichment.~~ Impacts of nutrient enrichment on soil carbon cycling have been extensively studied in temperate and tropical regions where intensive agriculture and land development have led to large increases in anthropogenic emissions of nitrogen (N) and phosphorous (P). However, how soil carbon sequestration and soil-atmosphere gas exchanges in cold regions respond to greater inputs of N and P remains poorly known, despite recent observations showing significant increases in porewater N and P in burned subarctic peatlands and downstream waters. Wildfires plus enhanced hydrological connectivity due to permafrost thaw have therefore the potential to change carbon turnover and gas emissions in soils of northern peatlands. To start exploring the sensitivity of peatland soil biogeochemistry to variations in N and P availability, we measured the carbon dioxide (CO$_2$) and methane (CH$_4$) production rates during a month-long incubation experiment with soils from a bog and fen collected at the long-term Scotty Creek research station in the Northwest Territories, Canada. Subsamples of the peatland soils were divided into a series of containers to which artificial porewater solutions were added. These solutions were amended with either dissolved inorganic N, dissolved inorganic P, or both N and P together. Unamended controls were run in parallel. The containers cycled through pre-set temperature steps of 1, 5, 15, and 25°C. Overall, the fen soil yielded higher CO$_2$ and CH$_4$ production rates than the bog soil. Amendment of N to the bog soil produced more CO$_2$ compared to its control, while amendment of P increased CO$_2$ production in the fen soil. Amendment of N and P together reduced CO$_2$ production but increased that of CH$_4$ in both the fen and bog soil incubations. Porewater chemistry at the end of the 30-day experiment showed aqueous C, N, and P stoichiometric ratios that trended toward those of the soil microbial biomasses, hence, implying that the initial microbial nutrient status played a crucial role in determining the responses to the different nutrient amendments. Our preliminary results show that the effects of nutrient enrichment on peatland soil biogeochemistry requires further investigation.

1 Introduction

Thawing permafrost and collapsing peatlands pose a threat to the stability of the net ecosystem carbon sequestration in subarctic regions (Treat et al., 2019; Schuur et al., 2022). While the extent of permafrost thaw and peatland collapse is rapidly expanding across the regions (Porter et al., 2019; Quinton et al., 2019; Hugelius et al., 2020), wildfires have also been increasing in western Canada (Gibson et al., 2018), Siberia (Talucci et al., 2022), and Alaska (Mekonnen et al., 2022). Increases in the number and extent of fire events result not only in the immediate carbon loss through biomass burning but also an increase of downstream export of particulate and dissolved organic carbon (DOC) from the burned areas ~~as summarized in Table 1~~ (Burd et al., 2018; Burd et al., 2020; Ackley et al., 2021; Koch et al., 2022; Mekonnen et al., 2022).

Increasing ~~nutrient~~ N and P inputs to peatland soils~~ecosystems~~ have the potential to perturb soil ~~is a rising concern for the large storage of~~ carbon cycling in peat soils. For instance, by relieving nutrient limitations on the soil microbial community, additional N and P may accelerate the decomposition of soil organic matter. Recent observations in western Canada reported that the surface water and shallow groundwater in burned areas were substantially enriched in dissolved nitrogen (N) and phosphorus (P), and even the high P concentrations persisted for several years after a fire as summarized in Table 1 (Emelko et al., 2016; Van Beest et al., 2019; Emmerton et al., 2020; Orlova et al., 2020). Thawing permafrost may release~~s~~ additional dissolved nutrients previously bound in frozen sediments (Treat et al., 2019; Schuur et al., 2022; Wright et al., 2022)~~,~~. ~~while~~ Furthermore, at ~~a~~the landscape level, permafrost thaw-induced ground subsidence increases the hydrological and geochemical connectivity among landscape components, including forested areas, fens, thermokarst bogs, and adjacent peatlands (Connon et al., 2014; Gibson et al., 2018; Haynes et al., 2018; Post et al., 2019; Carpino et al., 2021). Thus, there is growing probability for organic carbon stored in peat soils to experience enhanced microbial decomposition driven by nutrient enrichment~~ such as forests, fens, thermokarst bogs and other peatlands~~.

Table 1: Nitrogen (N) and phosphorus (P) concentrations in wildfire-impacted water samples from recent studies in western boreal and subarctic Canada.

Site	Disturbance	Sampling and analysis methods	N reported (mg L^{-1})	P reported (mg L^{-1})	Reference
Scotty Creek and Notawohka Creek catchments, NWT	Wildfire 3 years ago (the 2013 Notawohka fire)	Burned catchment downstream water sampling	0.5-0.9	0.06	Burd et al. (2018)
Scotty Creek and Notawohka Creek catchments, NWT	Wildfire 3 years ago (the 2013 Notawohka fire)	Burned peatland porewater field sampling and light exposal incubations	1.4-2.8	0.6	Burd et al. (2020)
Fort McMurray, Alberta	Wildfire in the same year (the May 2016 Fort McMurray wildfire)	Surface water samples collected from high- to low-order rivers	2-3	0.5-1	Emmerton et al. (2020)
Pelican Mountain, Alberta	Prescribed burn at research site in the same year	Shallow groundwater samples from monitoring wells	1-3	0.7	Orlova et al. (2020)
Scotty Creek watershed, NWT	Wildfire 2 years ago (2014 low-severity fire in the headwater areas)	Burned peatland porewater, field collected; comparison with unburned area	1-7	0.5-6	Ackley et al. (2021)
Pelican Mountain, Alberta	Laboratory simulated burning of research site samples at 250 and 300°C	5 g of burned peat leached by 1 litre water for 2 days	220-420	12-26	Wu et al. (2022)

Peatlands are generally poor-nutrient ecosystems with respect to both plants and soil microbial communities, while the magnitude of nutrient limitation varies with landscape position, peatland type, and groundwater connectivity (Hill et al., 2014; Lin et al., 2014; Moore et al., 2019). For example, isolated ombrotrophic ~~isolated~~ bogs rely on ~~the~~ direct atmospheric deposition of N and P ~~and~~ plus microbial N fixation~~ by N-fixing microbes, ~~. Comparatively~~generally most poor in soil nutrients.~~, ~~Minerotrophic~~ minerotrophic channel fens capture~~receive~~ ~~more~~ nutrients from surrounding water bodies through groundwater and surface water pathways~~connections~~. ~~Although the impacts of changing nutrient levels in temperate and tropical peatland soils have been studied in relation to the agricultural expansion and fossil fuel-driven atmospheric deposition (Amador and Jones, 1993; Qualls and Richardson, 2000; Lin et al., 2014; Hoyos-Santillan et al., 2018; Moore et al., 2019; Schillereff et al., 2021), the results are considered not directly applicable for the remote subarctic peatlands.~~ Rapid increases in nutrient inputs to peatlands have long been studied in temperate and tropical regions, where there is a higher chance of fertilizer spillover and growing agriculture or fossil fuel-driven atmosphereic deposition (Amador and Jones, 1993; Hoyos-Santillan et al. 2018; Lin et al., 2014; Moore et al., 2019; Qualls and Richardson, 2000; Schillereff et al., 2021). Early research on subtropical peatlands in the Everglades, Florida, showed that experimentally P-enriched soils released significantly more CO_2 than controls, seemingly because the added nutrient P stimulated soil heterotrophic respiration (Amador and Jones, 1993; Qualls and Richardson, 2000). Recently, however, a study suggested that nutrient availability is not always a rate-determining factor as microbial communities in tropical peatland soils have developed strategies to process site-specific plant litter even without adequate nutrient supply (Hoyos-Santillan et al., 2018). Still, other studies support the important control of nutrient increases in peat decomposition rates. For example, a data synthesis across temperate regional bogs revealed that a long-term P inputs to surface peat enhanced microbial decomposition activities and reduced net organic C burial to deeper peat layers (Schillereff et al., 2021). Another study of a 12-year P fertilization field experiment for temperate mountainous peatlands showed that the extra P supply increased soil respiration and CO_2 release, reducing the overall carbon sink function of these peatland ecosystem there (Lu et al., 2022).

Given the ~~large amounts of OC in~~high accumulation of plant organic materials in peatlands~~ soils~~, the absolute amount of ~~OC~~carbon as substrates for microbial activities is ~~likely~~ assumed not ~~at all~~ limiting in peat~~land~~ soils. Microbial strategies to overcome relative nutrient imbalances of N, P or other nutrients relative to ~~each other and/or relative to OC~~carbon may fall into three broad categories: (1) increasing the production of extracellular enzymes to acquire necessary nutrient elements from enzyme-facilitated organic matter degradation, (2) recycling of the nutrient elements assimilated in dead microbial cells (i.e., necromass recycling), and (3) releasing some extra carbon~~substrate C~~ in relation to the other major nutrients via auxiliary respiration without biomass assimilation (i.e., overflow respiration) (Giesler et al., 2011; Manzoni et al., 2012; Lin et al., 2014; Hoyos-Santillan et al., 2018; Schillereff et al., 2021; Lu et al., 2022). Laboratory soil incubation experiment to measure the changes in carbon gas production rates with net microbial biomass change can help address which strategy is activated.

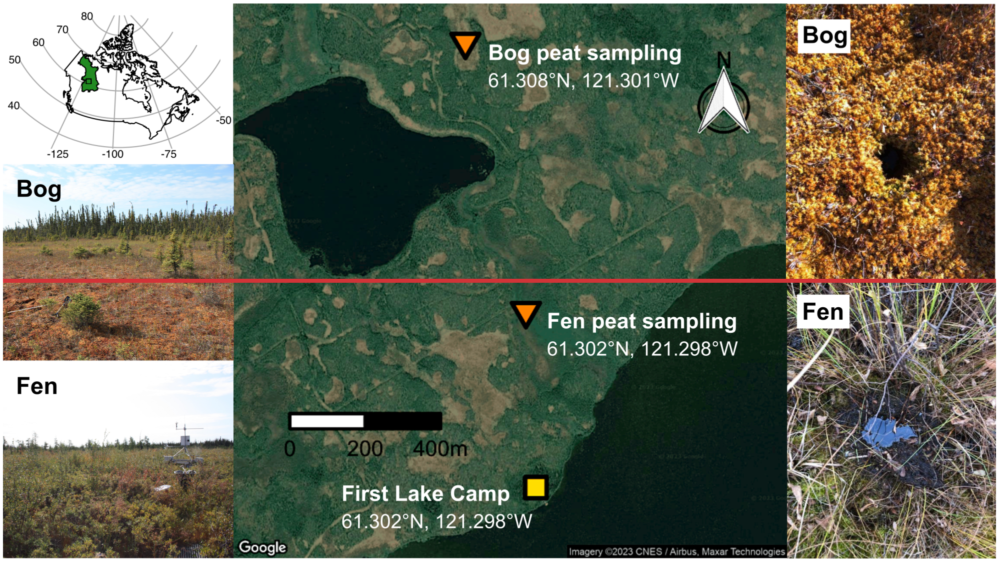
The Scotty Creek watershed, Northwest Territories, Canada, hosts one of the main research stations in the subarctic region, with active monitoring and field investigations on rapidly changing discontinuous permafrost landscapes and potential biogeochemical climate feedbacks (http://www.scottycreek.com/). Recent observations ~~pointed to~~highlighted the increase of wildfire impacts, the acceleration of permafrost thaw and the degradation of permafrost stability~~,~~. As a result, ~~with~~ peatland ecosystems and their associated hydrological processes and pathways are rapidly transforming (Gibson et al., 2018; Ackley et al., 2021; Wright et al., 2022)~~,~~. ~~but~~ However, the potential effects of sudden nutrient inputs to ~~the~~ peatland soils have not been explored for subarctic regions, despite the possible increase of organic carbon decomposition and greenhouse gas emissions suggested by studies in other climate zones~~remain poorly understood~~. In this study, our aim was to investigate ~~whether~~ the potential acceleration of ~~addition of dissolved inorganic N and P facilitates~~ organic matter decomposition and ~~nutrient recycling~~carbon gas production in subarctic peatland soils following the addition of readily bioavailable dissolved inorganic N and P. Also, we anticipated that variations in initial nutrient ratios (N:P, C:N, C:P) and the existing adaptations of soil microbial communities would contribute to understanding the short-term responses of different types of peatlands to additional

N or P (Hill et al., 2014; Hoyos-Santillan et al., 2018). A laboratory soil incubation experiment was conducted to compare the CO_2 and CH_4 production rates in the field-sampled bog or fen soils under various treatments: control, N only, P only, and both N and P (NP hereafter) addition. ~~The main incubation lasted for 30 days , subject to f~~Fluctuating temperatures~~,~~ were imposed during the month-long incubation to analyze~~to assess~~ the temperature sensitivity of the gas production rates in the different nutrient treatments~~underlying biogeochemical processes~~.

2 Methods

2.1 Scotty Creek field sites and peat coring

The sampling locations were within the Scotty Creek drainage basin, Northwest Territories, Canada, which lies within the Taiga Plains Ecozone (Fig. 1). Approximately 25% of this ecozone is covered by wetlands (Mahdianpari et al., 2021) with an estimate of 71,600 km^2 bog and 5,100 km^2 fen peatlands (Webster et al., 2018). The Scotty Creek basin is underlain by discontinuous permafrost that supports a peatland complex with peat plateau forests, thermokarst bogs, channel fens, and open water ponds. In October 2020, two duplicate shallow peat cores (0-25 cm) were taken from each of two sites: a thermokarst bog covered by *Sphagnum* mosses (hereafter referred to as ‘bog peat’ or ‘bog soil’) and a channel fen covered by herbaceous plants (hereafter referred to as ‘fen peat’ or ‘fen soil’) as shown in Fig. 1. These cores were then transported to the University of Waterloo, Waterloo, Canada, and stored in a -20°C freezer until being thawed to start the experiment.



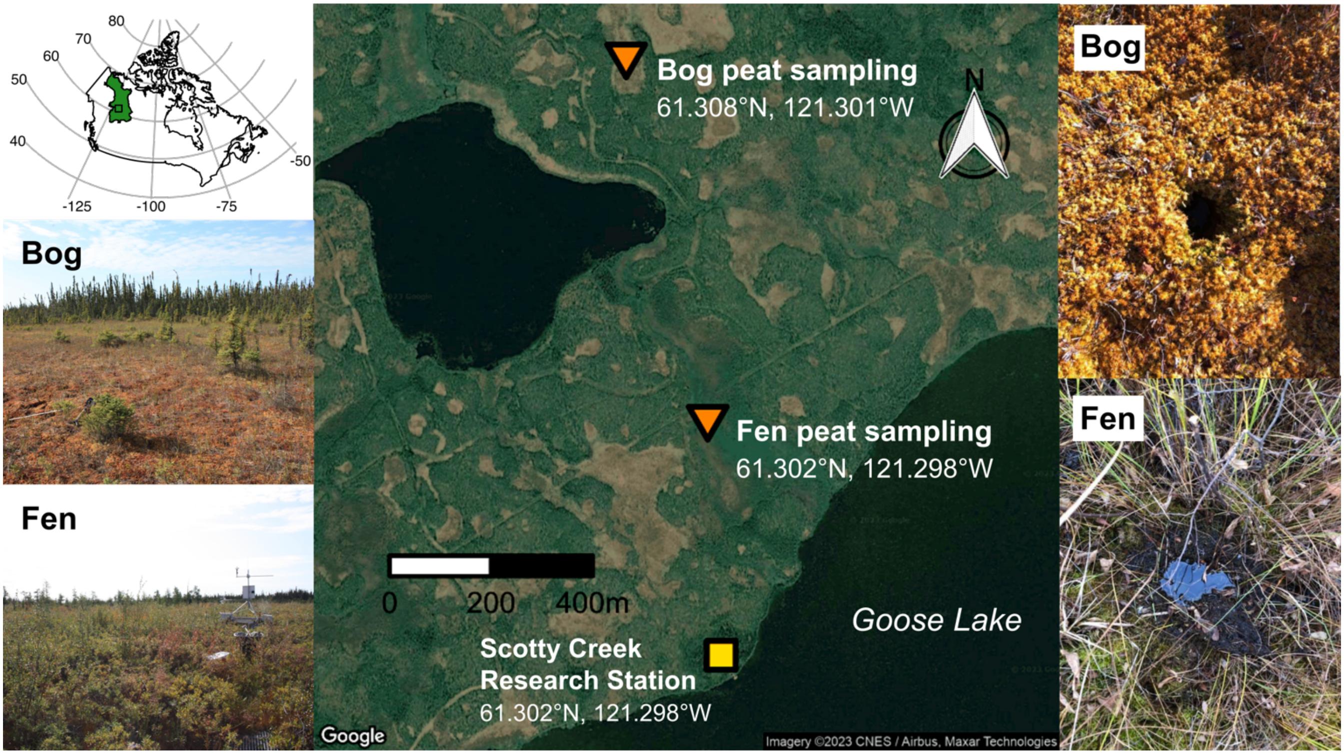


Figure 1: Scotty Creek watershed bog and fen peat sampling site locations (Map data: Google ©2023 CNES / Airbus, Maxar Technologies) and photos (credit: Mason Dominico). The location and extent of the Taiga Plains Ecozone of Canada (National Ecological Framework for Canada, 2017) is shown in the inset map with a small box indicator for the site locations. ~~Please replace "First Lake Camp" with "Scotty Creek Research Station". I would also label the larger lake "Goose Lake" because it appears as such on most maps.~~

2.2 Sample preparation and pre-incubation

The frozen peat cores were thawed at room temperature (22˚C). During the thawing, each peat core was sealed inside its original liner (from the field sampling) to preserve initial porewater release upon its thawing. After 24 hours, the liners were opened carefully, the released porewater from each core was collected and aliquots were saved for chemical analyses. The top layer containing fresh, large plant debris was removed from each peat core. The rest of the wet peat was placed in a clean plastic container and gently hand mixed to take small aliquots for determining approximate values of initial peat bulk density (approximate as the peat was de-structured), and moisture content (by oven-drying at 80˚C for 48 hours). In addition, a sub-sample ~~some~~ of each of the peat soils was freeze-dried for analysis of the total organic carbon (TOC) and total nitrogen (TN) and total phosphorus (TP) concentrations. ~~contents~~ TOC and TN were determined using a CHNS analyzer (Carlo Erba NA-1500 Elemental Analyzer; detection limit of 1% by mass for both TOC and TN~~each parameter~~). TP was determined following the method of Aspila et al. (1976): an aliquot of soil was burned to create ash ~~and for total phosphorus content by magnesium nitrate ($MgNO_3$) digestion, ashing~~ in a muffle furnace at 550˚C~~,~~ with magnesium nitrate added as an oxidant. The ashed soil was subsequently extracted by mixing with 1 M~~and~~ hydrochloric acid on a shaker for 16 hours~~extraction (Aspila et al., 1976)~~. The extract was then analyzed for total dissolved P by ICP-OES (Thermo Scientific iCAP 6300 Duo). ~~However, we did not attempt to interpret the total mass balance results for total pools due to the addition of extra salts and mass for nutrient treatment.~~

The containers with bulk peat samples were placed inside an environmental chamber at a constant temperature of 5˚C for two weeks. During this pre-incubation, the peat containers were uncovered inside the chamber, allowing for aerobic conditions. The pre-incubation caused some reduction of the peat moisture contents. ~~of peat were reduced~~ We therefore re-measured the moisture contents at the end of the pre-incubation period by the oven-drying method to determine how much porewater solutions needed to be added to return the peat moisture contents to the selected levels. Note that the latter were selected to be close to average ~~but not desiccated (i.e., peat was visually moist, but no water comes out when it was gently squeezed). The moisture contents by oven-drying were repeated after this pre-incubation period, to decide how much porewater solutions to add to return the peat to the~~ field conditions.

After the pre-incubation, each of the bog and fen peat samples was subsampled for fumigation to measure microbial biomass C, N and P concentrations. These concentrations were considered the baseline values against which changes in microbial C, N, and P accompanying the various nutrient treatments were evaluated. Then, the rest of the bulk peat was allocated into 250 mL mason glass jars (bog: 55 g peat and fen: 40 g peat). Porewater base solutions were prepared by diluting the initial thaw-collected porewater around 10:1, to make the required volume of the base solution while accounting for pH and initial nutrient levels, rather than using pure water. ~~The total amount of the porewater base solutions needed for the incubation jars was estimated based on the desired peat moisture levels during the incubation (volumetric water content of 80% for bog peat and 100% for fen peat as approximate growing-season averages measured in adjacent meteorological stations).~~The moisture conditions for the incubation experiment were selected using field-based growing-season averages measured in adjacent meteorological stations equipped with soil sensors. The selected moisture conditions were a volumetric water content of 80% for the bog peat and 100% for the fen peat. These values were then used to calculate the volumes of artificial porewater to add to the experimental containers.

The artificial porewater solutions were prepared, to adjust the peat moisture content and different treatments, by adding different amounts of salts including monobasic sodium phosphate (NaH_2PO_4), ammonium nitrate (NH_4NO_3), and ammonium chloride (NH_4Cl) for the N and/or P porewater enrichment scenarios. This experiment was aimed to test high levels of porewater nitrogen (7 mg-N L^{-1}) and phosphorus (6 mg-P L-1) from previous study measurements (Table 1). For the N amendment, the mix of NH_4NO_3 and NH_4Cl was used to introduce a 3:1 ratio for NH_4-N to NO_3-N following the measurement from a burned peat surface (Table 1). The solutions were added to bog and fen peat samples for control, N added, P added, and both N and P added, all prepared in duplicate (4 treatments × 2 sites × 2 samples = 16 total incubation jars). After adding the solutions, the headspace was 100 mL for bog samples and 150 mL for fen samples. The jars were placed back in the environmental chamber at 5˚C and pre-incubated for four weeks until stable CO_2 efflux rates were measured.

2.3. Incubation experiment and subsampling

The incubation jars were left open inside the environmental chamber to maintain aerobic headspace conditions, except during the closed headspace gas sampling. To maintain the moisture contents in the peat samples, the jars were placed together in a partially opened plastic bag with a small beaker containing a Milli-Q water-soaked sponge. The potential evaporative water loss was monitored by measuring the weights of two additional jars containing the same quantity of bog and fen peat, going through the same temperature course. Using this information, the peat moisture content was maintained within 10% of the initial condition by spraying some Milli-Q water inside the plastic bags to compensate for evaporation loss throughout the incubation experiment.

During the one-month incubation experiment, the headspace gas sampling was performed every third day once the chamber was set to a new temperature. Stepwise changes in the chamber temperature from 1˚C, 5˚C, 15˚C, 25˚C, 15˚C, 25˚C, 15˚C, 5˚C, 1˚C, 5˚C and back to 1˚C were imposed every 3-day interval to roughly mimic field situations where the ground temperature fluctuates during the non-growing to growing season transition. This non-growing to growing season transition, which includes snow melt, was thought to be a plausible scenario for the sudden contact of nutrient enriched water for distant peatlands after a watershed fire event. The comparison of flux rates at various temperatures was also aimed to test potential variation in nutrient interaction with microbial activities.

2.3.1 Porewater chemistry

Approximately 20 mL of porewater was extracted from bog and fen peat samples during the pre-incubation phase, and from each incubation jar during the post-incubation phase. The chemical properties of the porewater collected from the initial thawing of the peat cores were used for determining the initial porewater conditions and preparation of the artificial porewater solutions. The post-incubation porewater was collected by gently squeezing the saturated peat inside each incubation jar. The porewater pH was measured by a calibrated electrode (Orion™ Economy Series pH Combination Electrode, ThermoScientific) before filtering. The rest of the porewater samples were filtered through a 0.45 µm pore size membrane filter (nylon membrane syringe filters, VWR Scientific) and stored at 4˚C in the refrigerator if not analyzed immediately.

Concentrations of dissolved organic carbon (DOC), total dissolved nitrogen (TDN), and dissolved inorganic carbon (DIC) in the filtered porewater were measured using a total organic carbon analyzer (TOC-LCPH/CPN, Shimadzu; method detection limit 3 µmol L^{-1}). For the measurement, 1 mL of porewater was filtered through a 0.2 µm pore size membrane filter (Polyethersulfone membrane syringe filters, Thermo Scientific) and frozen for subsequent analysis of major anion concentrations including chloride (Cl^-), nitrite (NO_2^-), nitrate (NO_3^-), and sulfate (SO_4^{2-}) using ion chromatography (IC, Dionex ICS-5000 with a capillary IonPac® AS18 column; ± 3.0% error and ± 1.6% precision; method detection limit 0.59, 1.29, 1.13, 1.47 µmol L-1, respectively). Porewater ammonium (NH_4^+) and dissolved reactive P (DRP) concentrations were measured spectrophotometrically on a Thermo Scientific™ Gallery™ Discrete Analyzer (±10% error and ±3% precision). The concentrations of major cations including dissolved calcium, iron, potassium, magnesium, manganese, sodium, sulfur, and silicon concentrations were measured using ~~Inductively Coupled Plasma – Optical Emission Spectrometry (~~ICP-OES~~,~~ (Thermo Scientific iCAP 6300 Duo).

2.3.2. Microbial biomass fumigation

The microbial biomass carbon (MBC), nitrogen (MBN) and phosphorus (MBP) concentrations of the peat samples were measured using the chloroform fumigation method (Brookes et al., 1984; Vance et al., 1987; Joergensen, 1996; Jenkinson et al., 2004). For both the extractions of biomass P and biomass C and N, 4 g subsamples of the peat were sampled in parallel. One set of subsamples was treated with chloroform for 24 hours to fumigate the biomass in a vacuum desiccator, while a parallel set of subsamples was not fumigated. ~~Approximately 4 g of~~ Both the fumigated and non-fumigated peat samples ~~was used~~ were then extracted for OC and N ~~extraction with~~using 0.5 M potassium sulfate~~, or~~ and for P ~~extraction with~~using 0.5 M sodium bicarbonate ($NaHCO_3$ solution with pH adjusted to 8.5). ~~The subsamples were treated either with chloroform for 24 hours to fumigate in a vacuum desiccator, or with no fumigation.~~ ~~The~~ Both the potassium sulfate extracts and the sodium bicarbonate extracts ~~was~~ were then filtered through a 0.45 µm pore size membrane filter (nylon membrane syringe filters, VWR Scientific), and the extracts ~~were measured~~ analyzed, respectively, for DOC and TDN by TOC-L and for total dissolved phosphorus by ICP-OES. ~~The sodium bicarbonate extract was also filtered through the 0.45 µm filter, and the extracts were measured for total dissolved phosphorus (TDP) using ICP-OES.~~ The difference in DOC, TDN, and TDP concentrations between the fumigated and non-fumigated samples represented the C, N, and P present in the microbial biomass. ~~An e~~Extraction efficiencies~~y~~ of 0.45, 0.54 and 0.4 were assumed for MBC, MBN and MBP, respectively ~~was assumed for MBC and 0.54 for MBN~~ (Joergensen, 1996~~;~~) ~~, and 0.4 for MBP (~~Jenkinson et al., 2004). The resulting values from these sample measurements were compiled into a calculation spreadsheet to derive MBC, MBN and MBP values for each peat sample (Byun et al., 2024).

2.3.3. Headspace CH_4 and CO_2 flux measurements

The changes in CO_2 and CH_4 concentrations in the headspace of the jars during the incubation were measured by closing the jars for 15 or 20 minutes. Up to 10 mL headspace gas samples were taken from each jar at the end of the closed incubation through the custom lid with three-way gastight valves and a 10- or 20-mL plastic syringe. The CO_2 and CH_4 concentrations were analyzed by direct sample injection into a Gas Chromatograph (Shimadzu, Model GC-2014) equipped with a flame ionization detector and methanizer. The method detection limits for CH_4 and CO_2 were 0.384 and 17.145 ppm, respectively. For gas samples with CO_2 concentrations greater than 1,000 ppm, the samples were diluted with 15-30 mL of helium gas and allowed to mix for 20 minutes before analysis on the Gas Chromatograph. Gas efflux rates (F_{gas}, µmol g-1 h-1) were calculated following Eq. (1):

$$F_{gas} = \frac{PV_H(C_{t1} - C_{t0})}{RTm\Delta t} \qquad (1)$$

where P is the headspace pressure (atm), V_H is the headspace volume (L), R is the gas constant (0.0821 L atm mol^{-1} K^{-1}) and T is the temperature (K). Gas efflux was estimated using the relative headspace concentration changes inside the closed jar environments, as $C_{t1}–C_{t0}$ (ppm or 10-6) for each gas species during the time, $\Delta t=t_1\text{-}t_0$ (h). The gas efflux rates were then normalized per dry mass of peat (m, g dry peat) determined at the beginning of the incubation experiment (the mass loss was assumed minor and not considered in the flux rate calculations).

3 Results and Discussion

3.1 Transient changes in the soil CO_2 and CH_4 production

The fen soil incubation resulted in higher CO_2 and CH_4 production rates than the bog soil incubation as the incubation temperature was cycling from 1 to 25°C (Fig. 2). The CO_2 production rates of the bog samples were about 50% of those of the fen samples and the CH_4 production rates of the bog samples were an order of magnitude smaller than those of the fen samples. The higher CO_2 and CH_4 production rates in the fen soil are consistent with ~~its~~ the higher initial biomass of the fen soil~~microbial biomass~~ (MBC 4.3 mg C g^{-1}) ~~than~~ compared to that of the bog soil (MBC 0.7 mg C g^{-1}) ~~initially~~. Not accounting for primary production from living plants in this experiment, this result of higher fen CO_2 production rates are inconsistent with previous Soctty Creek field measurements by Chasmer et al. (2012) which showed higher net CO_2 emissions in bogs than in fens.

~~The temperature effects complicated the interpretation of the resulting gas production rates. The bog soil with added P and NP resulted distinctively lower CO_2 production rates at 25°C compared to the other treatments, while the same P additions resulted higher CO_2 production rates at 15°C compared to the N only addition (Fig. 2a). The fen peat with added N and NP resulted lower CO_2 production rates at 25°C, while the same N additions showed no apparent divergence at 15°C compared to the control, but instead the P only addition having distinctively higher rates than the other three (Fig. 2b). The CH_4 production rates of the fen soil also presented noticeable differences by temperature as well as nutrient (Fig. 2b) without a clear trend. The complex responses of soil carbon gas production may reflect inherently different limiting nutrient in either type of peatland (more discussed in 3.3).~~Temperature variably affected the gas production rates in the different nutrient treatments. The bog peat soil with added P and NP (thus, P amendment effects in general) yielded distinctively lower CO_2 production rates at 25°C but higher CO_2 production rates at 15°C, compared to the no-nutrient control (Fig. 2a). The fen peat soil with added N and NP (thus, N amendment effects in general) resulted in lower CO_2 production rates at 25°C but no apparent difference at 15°C, compared to the no-nutrient control. For the fen peat soil, the P only amendment (i.e., without N) resulted in distinctively higher CO_2 rates than the N amended treatments (N only and NP) and the control (Fig. 2b). The CH_4 production rates of the fen soil also showed noticeable temperature effects with the greatest differences between the various nutrient treatments observed at 25°C (Fig. 2b).

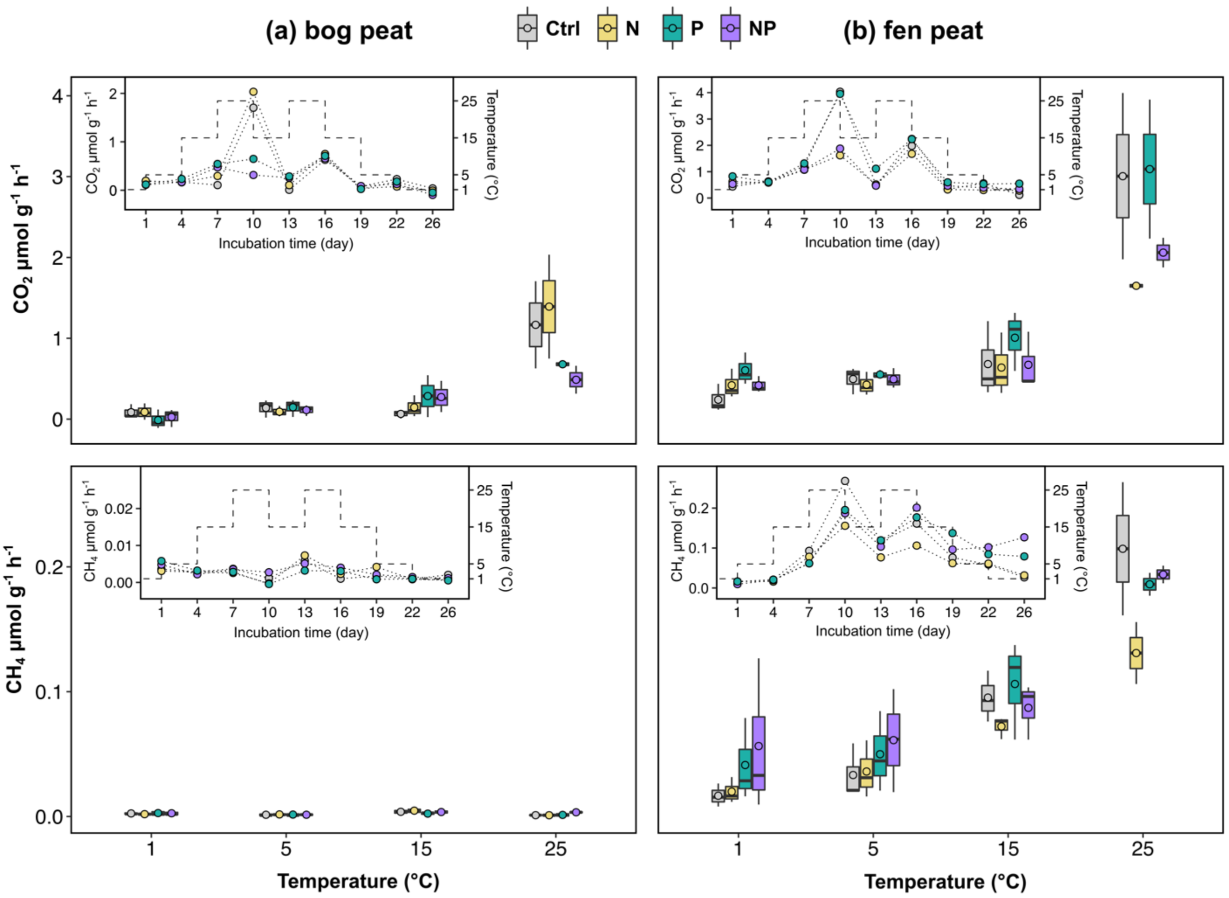


Figure 2: The CO$_2$ and CH$_4$ flux rates from the laboratory incubation measurements for subarctic (a) bog and (b) fen peat samples. Inset figures show the rate changes (connected dots) for each nutrient treatment (Ctrl: control; N: nitrogen addition; P: phosphorus addition; NP: nitrogen and phosphorus addition) over the incubation period with temperatures shown as dashed step lines. Box plots show the summary of rates at each temperature (median line and mean circle in each box) for each treatment.

3.2 Variations in the temperature sensitivity of CO$_2$ and CH$_4$ productions

In a broad sense, the CO$_2$ and CH$_4$ production rates followed the anticipated exponential increase with temperature (Fig. 2). However, the CH$_4$ production rates in the bog peat containers were very low and yielded too little variation to derive meaningful Q_{10} values for CH$_4$ production. Based on the average gas production rates at 5°C and 25°C, the temperature sensitivity parameter Q_{10} was calculated using an exponential rate increase following Eq. (2)

$$Q_{10} = \left(\frac{R2}{R1}\right)^{\left(\frac{10}{T2-T1}\right)} \qquad (2)$$

where $R2$ is the rate at the incubation temperature 25°C ($T2$) and $R1$ at the 5°C ($T1$) ~~(Table 2)~~. The Q_{10} values shows that the additional nutrients reduced the temperature sensitivity of the CO$_2$ emissions (i.e., the Q_{10} values were lower), except for the only N addition to the bog soil. According to the flux rate summary in Fig. 2, the low Q_{10} values correspond to relatively slow flux rate increases with temperature, while the higher Q_{10} represents the relatively fast flux rate increase with temperature rise to 25°C. For example, the NP addition resulted in the lowest CO_2 production rates in the bog soil at 25°C, contributing to the apparently lowest Q_{10} of 2.09 (Table 2); in contrast, the N only addition resulted in the highest CO_2 production rate in the bog soil at 25°C, contributing to the highest Q_{10} of 3.89 (Table 2).

Table 2: Estimated temperature sensitivity based on the average gas production rates at 5°C and 25°C.

Treatments	**Bog CO_2**	**Fen CO_2**	**Fen CH_4**
Control	2.9~~89~~	2.~~46~~5	2.5~~4~~
N addition	3.9~~0~~	~~1~~2.~~97~~0	1.9~~1~~
P addition	2.2~~15~~	2.4~~37~~	1.9~~3~~
N P addition	2.1~~09~~	2.0~~4~~	1.~~78~~8

The ~~estimated~~ temperature ~~sensitivity~~ sensitivities estimated here suggest~~s~~ that the alleviation of ~~relatively high CO_2 production rates at low temperature from alleviated~~ nutrient ~~limitations~~ limitation may result in either relatively ~~(as in low Q_{10} values of both NP, bog P, and fen N),~~ high CO_2 production rates at the lower temperatures tested (according to low Q_{10} values for both NP, bog P, and fen N), or high CO_2 production rates at the higher-end temperature~~s~~ ~~by effective mitigation of lacking nutrients~~ (according to the~~s in~~ high Q_{10} for~~of~~ bog N and fen P), or some combination of both trends ~~combined~~. Recently, Liu et al. (2022) suggested that the catalytic efficiency of hydrolytic enzymes ~~to facilitate~~involved in soil N and P recycling ~~largely~~ significantly increase~~d~~s with increasing temperature~~,~~. This~~which~~ would imply ~~supports~~ that ~~the second suggestion of effective mitigation of lacking nutrients (i.e., soil~~microbial respiration at warmer temperature would become~~s~~ less dependent on the initial soil and N and P stores~~nutrient shortage)~~. However, as shown by our results, the coupled effects of nutrient limitation and temperature may be more complex than currently recognized. A better predictive understanding of the variations in ~~Determining~~ Q_{10} values across ~~parameter for varying~~ peatland classes remains a crucial ~~has been an important~~ task for the calibration of process-based C cycle models of peatlands (Bona et al., 2020).~~,~~ For example, ~~and~~ the recently observed~~found~~ latitudinal increase of Q_{10} ~~temperature sensitivity~~values for CO_2 production in peat soils (Byun et al., 2021) ~~might~~ could point to an ~~actually reflect the~~ increasing severity of nutrient limitation for peatlands ~~towards~~at high latitude~~ peatland~~s.

3.3 Cumulative carbon release by nutrient addition

The stoichiometric ratios of primarily accessible substrates are important for soil microbial C cycling. The nutrient element ratios in the soil solution relative to the microbial biomass ratios can determine the use efficiency of DOC for respiration or biomass storage. This process has been conceptualized as microbial carbon use efficiency (CUE), which is expressed by the fraction of substrate OC that is taken up into microbial biomass of the total carbon that is used for both biomass uptake and cell metabolism (Manzoni et al., 2012; Sinsabaugh et al., 2013; Sinsabaugh et al., 2016; Geyer et al., 2016). If microbial cells are substantially limited by available nutrient N and/or P, they may allocate less of the DOC for biomass growth but still respire more (producing more CO_2 from DOC) for alleviating the nutrient limitation (thus, low CUE). To discuss this effect, the comparison chart for the cumulative emission of CO_2, CH_4 and the total of both gases (Gas C) is shown in Fig. 3, as the cumulative emissions were considered more indicative of the resulting net changes by the initial nutrient additions in peat carbon cycling over incubation.

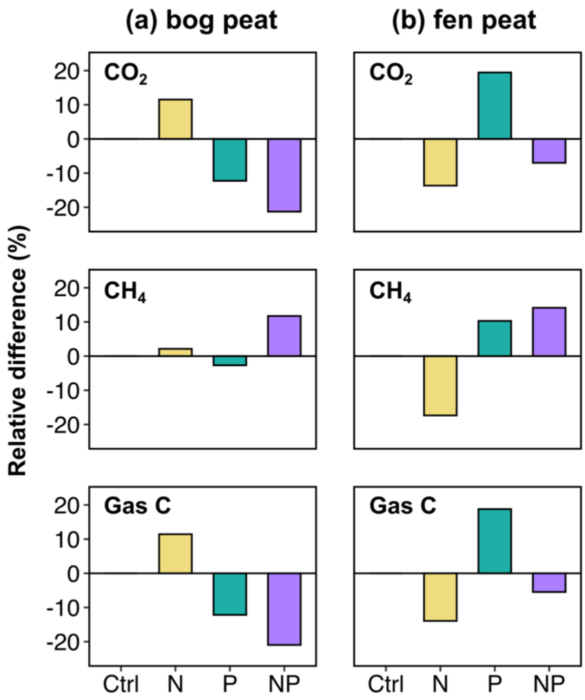


Figure 3: Relative differences (%) in ~~the~~ cumulative CO_2, CH_4, and total carbon gas~~ carbon~~ (Gas C) emissions integrated over the entire incubation experiment: comparing the unamended control (Ctrl) and the nutrient amended treatments N (N only), P (P only) and NP (both N and P together)~~observed throughout the incubation~~.

The microbial CUE was estimated for the bog soil by fumigation extraction method at the end of the incubation (Table 3). The fen soil fumigation experiment was not successful yielding negative microbial biomass calculations based on the sample weight and concentration measurements (Byun et al. 2024). It is difficult to imagine that the observed CO_2 and CH_4 production throughout the incubation period could have occurred without microbes in fen soil samples. Thus, the inferred negative biomass for the fen soil was considered a methodological artefact. Then, the microbial CUE of bog soil was approximated by the gross MBC acquisition, divided by the total of the MBC acquisition and the cumulative gas C release. As a result, the lower CUE by the N addition was attributed to the largest C gas release with the smallest gross MBC acquisition. The highest CUE value with the larger MBC acquisition and the smaller cumulative C gas release was found in the case of NP addition. This partially corresponds to the above interpretation of temperature sensitivity trend that the microbial respiration activities increased to alleviate relatively lacking nutrient P in the bog soil in response to the imbalance intensified by N addition.

Table 3: Microbial biomass changes and carbon use efficiency (CUE) of bog peat.

Extraction	**Before**			**After**				
Bog	**MBC (mg C g^{-1} dry soil)**	**MBN (mg N g^{-1} dry soil)**	**MBC:MBN (atomic ratio)**	**MBC (mg C g^{-1} dry soil)**	**MBN (mg N g^{-1} dry soil)**	**MBC:MBN (atomic ratio)**	**Gas C (mg C g^{-1} dry soil)**	**Approx. CUE**
Control	0.69	0.10	8.1	8.5	0.97	10.2	4.0	0.49
N addition	0.69	0.10	8.1	6.8	0.79	10.0	4.5	0.26
P addition	0.69	0.10	8.1	7.7	0.86	10.4	3.5	0.50
NP addition	0.69	0.10	8.1	9.8	1.2	9.7	3.2	0.65

~~The amount of OC substrates is likely not at all limiting in peatland soils, and microbial strategies to overcome relative nutrient imbalances may fall into three broad categories: (1) increase the production of extracellular enzymes to acquire necessary nutrient elements from enzyme-facilitated organic matter degradation, (2) recycle the nutrient elements assimilated in dead microbial cells (i.e., necromass recycling), and (3) release extra substrate C via respiration without biomass assimilation (i.e., overflow respiration) (Giesler et al., 2011; Manzoni et al., 2012; Lin et al., 2014; Hoyos-Santillan et al., 2018; Schillereff et al., 2021; Lu et al., 2022). The quality or degradability of C substrates may have a determining effect on which strategy to be activated by soil microbes, which in turn accelerates or decelerates the overall decomposition rate of soil organic matter (SOM).~~

3.4 Stoichiometric changes in soil solution

We conducted the incubations in ~~Given~~ the dark ~~chamber environment used for the incubation and no~~without adding~~tional~~ any organic carbon substrate~~s,~~. Thus, we interpret the observed changes in the porewater pools of C, N and P to be the result of the processing of existing C, N and P soil pools ~~were considered fully the~~coupled to the net mineralization of soil organic matter by ~~heterotrophic~~ the resident microbial community~~metabolism of existing pools~~. The changes in the pore water DOC concentration~~s~~ and the ~~with~~ stoichiometric ratios of DOC to dissolved N and P, respectively, ~~before and after the incubation~~ are shown in Fig. 4. The initial microbial biomass stoichiometry is added as a dotted line in each panel of Fig. 4. Overall, the DOC decreased from the initial concentrations, but the remnants were more than the estimated consumption from the cumulative carbon gas emissions, indicating the use of solid organic particles. The final concentration of DOC in the bog with NP addition (78.2 ± 6.88 mg C L^{-1}, mean ± SD of duplicates) was smaller than the control (101.6 ± 9.97 mg C L^{-1}), while the

N-only or P-only addition was not meaningfully smaller than the control (97.0 ± 0.69 mg C L-1 for the N addition; 94.3 ± 1.19 mg C L-1 for the P addition) (Fig. 4a). In the fen, the final DOC concentrations were all around the control result (Fig. 4b).

The TDN by the end of the incubation was around approximately 40-60% of NH_4^+ in both peat types (e.g., NH_4^+ 2.3-3.8 mg-N L-1 and TDN from 5.2-6.1 mg-N L-1 in bog control to N added; NH_4^+ 5.2-7.7 mg-N L-1 and TDN 13.3-15 mg-N L-1 in fen control to N added). In theory, the NH_4^+ would have been taken up by plants under field conditions. The majority of the remaining TDN was dissolved organic N with minor contributions by NO_3^- or NO_2^- to the TDN. These porewater patterns may provide some explanation for the reported water quality changes in burned catchments, lasting high phosphate and high dissolved organic N concentrations (Ackley et al., 2021; Burd et al., 2020; Wright et al., 2022).

The stoichiometric ratios of DOC to TDN and TDP in the porewater approached the initial microbial biomass ratios by the end of the incubation of both bog and fen peat. In the bog peat, as the relative N shortage by the initial P addition reduced both MBC growth and cumulative respiration, the CUE was estimated close to the control while the N addition resulted in a much smaller CUE (Table 3). This could be due to the microbial community of bog needing relatively less energy for the necessary N acquisition than for the P acquisition from the peat environment. The bog peat microbial community may have specifically adapted to the constant shortage of available P in peat, allowing them to release as much bioavailable P as required to overcome the nutrient imbalance for C cycling, even with a high energy cost (Hoyos-Santillan et al., 2018). In the fen peat, P release seemed to be an efficient process for the fen adapted microbial community, given the prompt enzymatic DRP release to meet the increased need by the N addition.

In the bog peat, the largest net microbial biomass growth occurred when NP were added, possibly by more direct assimilation of available DOC instead of auxiliary respiration (Giesler et al., 2011; Manzoni et al., 2012; Sinsabaugh et al., 2013; Lin et al., 2014). It seems that the NP addition could best enhance the microbial CUE while reducing the auxiliary CO_2 production, while also more quickly consuming available DOC and further degradation of solid carbon, which under the anaerobic soil condition resulted in the higher cumulative CH_4 production (Fig.3).

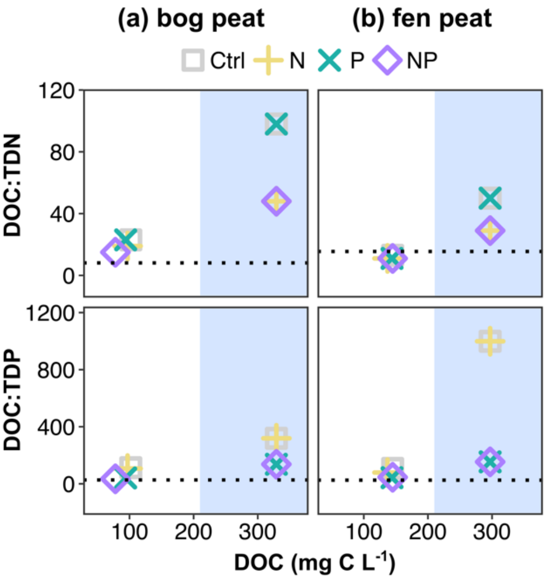


Figure 4: Porewater dissolved organic carbon and stoichiometric ratio change before and after the incubation. The blue shade~~d area~~ indicates where the initial (i.e., at the start of the incubation experiment) compositions ~~porewater ratios~~ are marked in comparison to the compositions at the end of the incubation. The dotted lines represent the initial microbial biomass atomic ratios (bog peat MBC/MBN = 8.1, MBC/MBP = 26.9; fen MBC/MBN = 15.5, MBC/MBP = 25.6).

Despite the aerobic headspace setting of incubation jars, the water saturated peat environments initiated anaerobic microbial metabolism, as indicated by the net CH_4 production and release (Fig. 2). The increase of porewater dissolved iron (Fe) and manganese (Mn) is also indicative of the reductive dissolution of sediment iron and manganese oxides (Table 4). As the DRP concentrations increased in the fen samples, we interpret that some of this DRP production may have been associated with the release of DRP that was bound to Fe and Mn oxides (Table 4). To attribute the net DRP production to Fe and Mn oxide reduction versus biological sources of DRP production such as enzymatic release, we subtracted the amount of DRP release that was possible from Fe and Mn oxide reduction using the maximum molar ratio of 1 mol Fe and Mn to 1 mol DRP (Orihel et al., 2017). As a result, the largest biological DRP release was measured in the N addition and the smallest amount was released in the NP addition. In the bog peat, net DRP uptake was estimated (Table 4), implying the inherent limitation of available P for microbial communities.

Table 4: Porewater phosphorus changes from the initial to the final analysis.

Porewater		Initial		Final		Final – Initial				Approx. DRP production[a] (μmol L^{-1})
		DRP (μmol L^{-1})	TDP (μmol L^{-1})	DRP (μmol L^{-1})	TDP (μmol L^{-1})	ΔDRP (μmol L^{-1})	ΔTDP (μmol L^{-1})	ΔFe (μmol L^{-1})	ΔMn (μmol L^{-1})	
Bog	Ctrl	67.9	86.0	64.8	75.2	-3.1	-10.8	1.05	0.43	Net uptake
	N	67.9	86.0	56.9	75.6	-11.0	-10.4	0.89	0.41	Net uptake
	P	182	200	146	193	-36.0	-7.0	0.79	0.46	Net uptake
	NP	182	200	160	187	-22.0	-13.0	0.80	0.44	Net uptake

Fen	Ctrl	22.9	24.8	98.2	112	75.3	87.2	15.5	6.54	53.3
	N	22.9	24.8	135	146	112	121.2	12.6	6.85	92.7
	P	159	161	217	282	58.0	121.0	13.8	6.11	38.1
	NP	159	161	205	261	46.0	100.0	14.0	5.33	26.7

[a] Hypothetical DRP production that could be attributed to enzymatic release; for the bog porewater measurements, net DRP uptake was estimated, and thus the DRP production was not determined.

These observations support the critical role of microbial metabolism in regulating nutrient forms and concentrations in peat porewater and imply the role of peatlands in large scale on the watershed nutrient fluxes and water quality (Liu et al., 2023). Relying on the internal supply of C substrates from peat decomposition, the fen porewater ended up accumulating additional dissolved inorganic P, possibly in the process of balancing for the bioavailable substrate stoichiometry by microbial activities. In contrast, the net uptake of P by the bog peat in all four of the treatments suggests that bog peatlands might have some capacity to sequester P inputs via microbial assimilation because P is the inherently limiting nutrient for microbial growth (Worrall et al., 2016). However, the stability and time scale of this P sequestration into microbial biomass in the bog soil remains uncertain.

3.5 Limitations

The experimental results reported herein should be considered in the light of some limitations, mainly derived from the small sample size. First, the number of replicates for this type of experiment is recommended three or more, rather than two in this experiment, but this was limited by the amount of soil samples from these remote peatland sites. To test the hypothesis, four different treatments with subsampling of multiple aliquots before and at the end of the incubation were also needed to measure key soil property changes. Despite sample-size limitations, results from the duplicate samples were consistent in the direction of changes, within instrumental error ranges, and thus the results can be used to infer and discuss environmental implications. Also, in this article, experimental procedures for potential future replication studies are described in detail, so that additional samples and replicates can be tested to assess the generality of the current findings. Second, there was a methodological artefact in estimating the microbial biomass of the post incubation samples from one of the two study sites. It may have been possible to mitigate this issue by measuring additional aliquots of the incubated samples if prepared for the chloroform treatment at the time. Although this artefact was considered insignificant for the main discussion of this study results, it is encouraged to improve the experimental procedure, adapted in particular for peat soil samples, to avoid this issue reoccurring even for small sample experiments if necessary.

4 Future research suggestions

The experiment in this study tested the instantaneous responses of peat microbial community to a one-time, or pulsed, nutrient nitrogen and phosphorus addition. In a real-world situation, multiple possible scenarios could occur in terms of the variability in the concentrations, ratios, timing, and duration of such nutrients that may be delivered to peatlands, while the combined effects of permafrost thaws and frequent wildfires are likely to increase dissolved nutrients in subarctic peatlands, previously considered remote and less affected by industrial inputs. Future experiments should more systematically investigate various nutrient supply scenarios across different time scales and peatland types for conducting large-scale studies. Understanding the microbial regulation of porewater stoichiometry in response to increased nitrogen or phosphorus availability may be crucial in explaining sudden changes in CO_2 and CH_4 emissions from peatlands. We recommend future work to explore the existing inherent nutrient limitation of the peatland microbial communities and their response to both pulsed and continuous nutrient inputs with a wide range of representative seasonal temperature fluctuations. Consequently, peatland carbon cycle models should also account for the alteration of the temperature sensitivity parameter by shifts in nutrient concentrations and ratios.

Data availability

The data that support the findings of this study are available at https://doi.org/10.20383/102.0712 in Federated Research Data Repository (FRDR).

Author contribution

Eunji Byun: Conceptualization, Methodology, Investigation, Writing - Original draft preparation. **Fereidoun Rezanezhad**: Conceptualization, Supervision, Writing - Review & Editing. **Stephanie Slowinski**: Conceptualization, Methodology, Validation, Writing - Review & Editing. **Christina Lam**: Methodology, Writing - Review & Editing. **Saraswati Saraswati**: Validation, Writing - Review & Editing, **Stephanie Wright**: Resources, Validation, Writing - Review & Editing. **William L. Quinton**: Resources, Validation, Writing - Review & Editing. **Kara Webster**: Writing - Review & Editing. **Philippe Van Cappellen**: Conceptualization, Supervision, Writing - Review & Editing.

Competing interests

The authors declare that they have no conflict of interest.

Acknowledgements

Funding was provided by the Canada Excellence Research Chair (CERC) in Ecohydrology, the Advancing Climate Change Science in Canada program (ACCPJ 536050-18), the Winter Soil Processes in Transition project within the Global Water Futures (GWF) program funded by the Canada First Research Excellence Fund (CFREF), and a Natural Sciences and Engineering Research Council (NSERC) Discovery Grants (RGPIN-2015-03801and RGPIN-2022-03334) and the Can-Peat: Canada’s peatlands as nature-based climate solutions project (https://uwaterloo.ca/can-peat) undertaken with the financial support of the Government of Canada. Partial financial support for the first author was provided by the Yonsei University Research Fund (2023-22-0433). The authors wish to acknowledge the support of ArcticNet through the Dehcho Collaborative on Permafrost.

References

Ackley, C., Tank, S. E., Haynes, K. M., Rezanezhad, F., McCarter, C., and Quinton, W. L.: Coupled hydrological and geochemical impacts of wildfire in peatland-dominated regions of discontinuous permafrost, Sci Total Environ, 782, 146841, 10.1016/j.scitotenv.2021.146841, 2021.

Amador, J. A. and Jones, R. D.: Nutrient limitations on microbial respiration in peat soils with different total phosphorus content, Soil Biol. Biochem., 25, 793-801, 10.1016/0038-0717(93)90125-U, 1993.

Aspila, K. I., Agemian, H., and Chau, A. S.: A semi-automated method for the determination of inorganic, organic and total phosphate in sediments, Analyst, 101, 187-197, 10.1039/an9760100187, 1976.

Bona, K. A., Shaw, C., Thompson, D. K., Hararuk, O., Webster, K., Zhang, G., Voicu, M., and Kurz, W. A.: The Canadian model for peatlands (CaMP): A peatland carbon model for national greenhouse gas reporting, Ecological Modelling, 431, 109164, 10.1016/j.ecolmodel.2020.109164, 2020.

Brookes, P. C., Powlson, D. S., and Jenkinson, D. S.: Phosphorus in the soil microbial biomass, Soil Biology and Biochemistry, 16, 169-175, 10.1016/0038-0717(84)90108-1, 1984.

Burd, K., Estop-Aragonés, C., Tank, S. E., Olefeldt, D., and Naeth, M. A.: Lability of dissolved organic carbon from boreal peatlands: interactions between permafrost thaw, wildfire, and season, Canadian Journal of Soil Science, 100, 503-515, 10.1139/cjss-2019-0154, 2020.

Burd, K., Tank, S. E., Dion, N., Quinton, W. L., Spence, C., Tanentzap, A. J., and Olefeldt, D.: Seasonal shifts in export of DOC and nutrients from burned and unburned peatland-rich catchments, Northwest Territories, Canada, Hydrology and Earth System Sciences, 22, 4455-4472, 10.5194/hess-22-4455-2018, 2018.

Byun, E., Rezanezhad, F., Fairbairn, L., Slowinski, S., Basiliko, N., Price, J. S., Quinton, W. L., Roy-Léveillée, P., Webster, K., and Van Cappellen, P.: Temperature, moisture and freeze–thaw controls on CO2 production in soil incubations from northern peatlands, Scientific Reports, 11, 23219, 10.1038/s41598-021-02606-3, 2021.

Byun, E., Rezanezhad, F., Slowinski, S., Lam, C., Saraswati, S., Wright, S., Quinton, W., Webster, K., Van Cappellen, P.: Dataset for Examining the Effects of Nutrient Pulses on Biogeochemical Cycling in Subarctic Peatlands in the Context of Permafrost Thaw and Wildfires. Federated Research Data Repository. 10.20383/102.0712, 2024.

Carpino, O., Haynes, K., Connon, R., Craig, J., Devoie, É., and Quinton, W.: Long-term climate-influenced land cover change in discontinuous permafrost peatland complexes, Hydrol. Earth Syst. Sci., 25, 3301-3317, 10.5194/hess-25-3301-2021, 2021.

Chasmer, L., Kenward, A., Quinton, W., and Petrone, R.: CO2 Exchanges within Zones of Rapid Conversion from Permafrost Plateau to Bog and Fen Land Cover Types. Arctic, Antarctic, and Alpine Research, 44, 399–411. 10.1657/1938-4246-44.4.399, 2012.

Connon, R. F., Quinton, W. L., Craig, J. R., and Hayashi, M.: Changing hydrologic connectivity due to permafrost thaw in the lower Liard River valley, NWT, Canada, Hydrological Processes, 28, 4163-4178, 10.1002/hyp.10206, 2014.

Emelko, M. B., Stone, M., Silins, U., Allin, D., Collins, A. L., Williams, C. H., Martens, A. M., and Bladon, K. D.: Sediment-phosphorus dynamics can shift aquatic ecology and cause downstream legacy effects after wildfire in large river systems, Glob Chang Biol, 22, 1168-1184, 10.1111/gcb.13073, 2016.

Emmerton, C. A., Cooke, C. A., Hustins, S., Silins, U., Emelko, M. B., Lewis, T., Kruk, M. K., Taube, N., Zhu, D., Jackson, B., Stone, M., Kerr, J. G., and Orwin, J. F.: Severe western Canadian wildfire affects water quality even at large basin scales, Water Res, 183, 116071, 10.1016/j.watres.2020.116071, 2020.

Geyer, K. M., Kyker-Snowman, E., Grandy, A. S., and Frey, S. D.: Microbial carbon use efficiency: accounting for population, community, and ecosystem-scale controls over the fate of metabolized organic matter, Biogeochemistry, 127, 173-188, 10.1007/s10533-016-0191-y, 2016.

Gibson, C. M., Chasmer, L. E., Thompson, D. K., Quinton, W. L., Flannigan, M. D., and Olefeldt, D.: Wildfire as a major driver of recent permafrost thaw in boreal peatlands, Nat Commun, 9, 3041, 10.1038/s41467-018-05457-1, 2018.

Giesler, R., Esberg, C., Lagerström, A., and Graae, B. J.: Phosphorus availability and microbial respiration across different tundra vegetation types, Biogeochemistry, 108, 429-445, 10.1007/s10533-011-9609-8, 2011.

Haynes, K. M., Connon, R. F., and Quinton, W. L.: Permafrost thaw induced drying of wetlands at Scotty Creek, NWT, Canada, Environmental Research Letters, 13, 10.1088/1748-9326/aae46c, 2018.

Hill, B. H., Elonen, C. M., Jicha, T. M., Kolka, R. K., Lehto, L. L. P., Sebestyen, S. D., and Seifert-Monson, L. R.: Ecoenzymatic stoichiometry and microbial processing of organic matter in northern bogs and fens reveals a common P-limitation between peatland types, Biogeochemistry, 120, 203-224, 10.1007/s10533-014-9991-0, 2014.

Hoyos-Santillan, J., Lomax, B. H., Turner, B. L., Sjögersten, S., and Austin, A.: Nutrient limitation or home field advantage: Does microbial community adaptation overcome nutrient limitation of litter decomposition in a tropical peatland?, Journal of Ecology, 106, 1558-1569, 10.1111/1365-2745.12923, 2018.

Hugelius, G., Loisel, J., Chadburn, S., Jackson, R. B., Jones, M., MacDonald, G., Marushchak, M., Olefeldt, D., Packalen, M., Siewert, M. B., Treat, C., Turetsky, M., Voigt, C., and Yu, Z.: Large stocks of peatland carbon and nitrogen are vulnerable to permafrost thaw, Proc Natl Acad Sci U S A, 117, 20438-20446, 10.1073/pnas.1916387117, 2020.

Jenkinson, D. S., Brookes, P. C., and Powlson, D. S.: Measuring soil microbial biomass, Soil Biology and Biochemistry, 36, 5-7, 10.1016/j.soilbio.2003.10.002, 2004.

Joergensen, R. G.: The fumigation-extraction method to estimate soil microbial biomass: Calibration of the kEC value, Soil Biology and Biochemistry, 28, 25-31, 10.1016/0038-0717(95)00102-6, 1996.

Koch, J. C., Bogard, M. J., Butman, D. E., Finlay, K., Ebel, B., James, J., Johnston, S. E., Jorgenson, M. T., Pastick, N. J., Spencer, R. G. M., Striegl, R., Walvoord, M., and Wickland, K. P.: Heterogeneous Patterns of Aged Organic Carbon Export Driven by Hydrologic Flow Paths, Soil Texture, Fire, and Thaw in Discontinuous Permafrost Headwaters, Global Biogeochemical Cycles, 36, 10.1029/2021gb007242, 2022.

Lin, X., Tfaily, M. M., Green, S. J., Steinweg, J. M., Chanton, P., Imvittaya, A., Chanton, J. P., Cooper, W., Schadt, C., and Kostka, J. E.: Microbial metabolic potential for carbon degradation and nutrient (nitrogen and phosphorus) acquisition in an ombrotrophic peatland, Appl Environ Microbiol, 80, 3531-3540, 10.1128/AEM.00206-14, 2014.

Liu, C., Tian, H., Gu, X., Li, N., Zhao, X., Lei, M., Alharbi, H., Megharaj, M., He, W., and Kuzyakov, Y.: Catalytic efficiency of soil enzymes explains temperature sensitivity: Insights from physiological theory, Sci Total Environ, 822, 153365, 10.1016/j.scitotenv.2022.153365, 2022.

Liu, H., Zak, D., Zableckis, N., Cossmer, A., Langhammer, N., Meermann, B., and Lennartz, B.: Water pollution risks by smoldering fires in degraded peatlands, Sci Total Environ, 871, 161979, 10.1016/j.scitotenv.2023.161979, 2023.

Lu, F., Wu, J., Yi, B., Xu, Z., Wang, M., Sundberg, S., and Bu, Z.-J.: Long-Term Phosphorus Addition Strongly Weakens the Carbon Sink Function of a Temperate Peatland, Ecosystems, 10.1007/s10021-022-00754-9, 2022.

Mahdianpari, M., Brisco, B., Granger, J., Mohammadimanesh, F., Salehi, B., Homayouni, S., and Bourgeau-Chavez, L.: The Third Generation of Pan-Canadian Wetland Map at 10 m Resolution Using Multisource Earth Observation Data on Cloud Computing Platform, IEEE Journal of Selected Topics in Applied Earth Observations and Remote Sensing, 14, 8789-8803, 10.1109/jstars.2021.3105645, 2021.

Manzoni, S., Taylor, P., Richter, A., Porporato, A., and Agren, G. I.: Environmental and stoichiometric controls on microbial carbon-use efficiency in soils, New Phytol, 196, 79-91, 10.1111/j.1469-8137.2012.04225.x, 2012.

Mekonnen, Z. A., Riley, W. J., Randerson, J. T., Shirley, I. A., Bouskill, N. J., and Grant, R. F.: Wildfire exacerbates high-latitude soil carbon losses from climate warming, Environmental Research Letters, 17, 10.1088/1748-9326/ac8be6, 2022.

Moore, T. R., Knorr, K.-H., Thompson, L., Roy, C., and Bubier, J. L.: The effect of long-term fertilization on peat in an ombrotrophic bog, Geoderma, 343, 176-186, 10.1016/j.geoderma.2019.02.034, 2019.

Orihel, D. M., Baulch, H. M., Casson, N. J., North, R. L., Parsons, C. T., Seckar, D. C. M., and Venkiteswaran, J. J.: Internal phosphorus loading in Canadian fresh waters: a critical review and data analysis, Canadian Journal of Fisheries and Aquatic Sciences, 74, 2005-2029, 10.1139/cjfas-2016-0500, 2017.

Orlova, J., Olefeldt, D., Yasinski, J. H., and Anderson, A. E.: Effects of Prescribed Burn on Nutrient and Dissolved Organic Matter Characteristics in Peatland Shallow Groundwater, Fire, 3, 10.3390/fire3030053, 2020.

Porter, T. J., Schoenemann, S. W., Davies, L. J., Steig, E. J., Bandara, S., and Froese, D. G.: Recent summer warming in northwestern Canada exceeds the Holocene thermal maximum, Nat Commun, 10, 1631, 10.1038/s41467-019-09622-y, 2019.

Post, E., Alley, R. B., Christensen, T. R., Macias-Fauria, M., Forbes, B. C., Gooseff, M. N., Iler, A., Kerby, J. T., Laidre, K. L., Mann, M. E., Olofsson, J., Stroeve, J. C., Ulmer, F., Virginia, R. A., and Wang, M.: The polar regions in a 2 degrees C warmer world, Sci Adv, 5, eaaw9883, 10.1126/sciadv.aaw9883, 2019.

Qualls, R. G. and Richardson, C. J.: Phosphorus enrichment affects litter decomposition, immobilization, and soil microbial phosphorus in wetland mesocosms, Soil Sci. Soc. Am. J., 64, 799-808, 10.2136/sssaj2000.642799x, 2000.

Quinton, W., Berg, A., Braverman, M., Carpino, O., Chasmer, L., Connon, R., Craig, J., Devoie, É., Hayashi, M., Haynes, K., Olefeldt, D., Pietroniro, A., Rezanezhad, F., Schincariol, R., and Sonnentag, O.: A synthesis of three decades of hydrological research at Scotty Creek, NWT, Canada, Hydrology and Earth System Sciences, 23, 2015-2039, 10.5194/hess-23-2015-2019, 2019.

Schillereff, D. N., Chiverrell, R. C., Sjöström, J. K., Kylander, M. E., Boyle, J. F., Davies, J. A. C., Toberman, H., and Tipping, E.: Phosphorus supply affects long-term carbon accumulation in mid-latitude ombrotrophic peatlands, Communications Earth & Environment, 2, 10.1038/s43247-021-00316-2, 2021.

Schuur, E. A. G., Abbott, B. W., Commane, R., Ernakovich, J., Euskirchen, E., Hugelius, G., Grosse, G., Jones, M., Koven, C., Leshyk, V., Lawrence, D., Loranty, M. M., Mauritz, M., Olefeldt, D., Natali, S., Rodenhizer, H., Salmon, V., Schädel, C., Strauss, J., Treat, C., and Turetsky, M.: Permafrost and Climate Change: Carbon Cycle Feedbacks From the Warming Arctic, Annual Review of Environment and Resources, 47, 343-371, 10.1146/annurev-environ-012220-011847, 2022.

Sinsabaugh, R. L., Manzoni, S., Moorhead, D. L., and Richter, A.: Carbon use efficiency of microbial communities: stoichiometry, methodology and modelling, Ecol Lett, 16, 930-939, 10.1111/ele.12113, 2013.

Sinsabaugh, R. L., Turner, B. L., Talbot, J. M., Waring, B. G., Powers, J. S., Kuske, C. R., Moorhead, D. L., and Follstad Shah, J. J.: Stoichiometry of microbial carbon use efficiency in soils, Ecological Monographs, 86, 172-189, 10.1890/15-2110.1, 2016.

Talucci, A. C., Loranty, M. M., and Alexander, H. D.: Siberian taiga and tundra fire regimes from 2001–2020, Environmental Research Letters, 17, 10.1088/1748-9326/ac3f07, 2022.

Treat, C. C., Kleinen, T., Broothaerts, N., Dalton, A. S., Dommain, R., Douglas, T. A., Drexler, J. Z., Finkelstein, S. A., Grosse, G., Hope, G., Hutchings, J., Jones, M. C., Kuhry, P., Lacourse, T., Lahteenoja, O., Loisel, J., Notebaert, B., Payne, R. J., Peteet, D. M., Sannel, A. B. K., Stelling, J. M., Strauss, J., Swindles, G. T., Talbot, J., Tarnocai, C., Verstraeten, G., Williams, C. J., Xia, Z., Yu, Z., Valiranta, M., Hattestrand, M., Alexanderson, H., and Brovkin, V.: Widespread global peatland establishment and persistence over the last 130,000 y, Proc Natl Acad Sci U S A, 116, 4822-4827, 10.1073/pnas.1813305116, 2019.

Van Beest, C., Petrone, R., Nwaishi, F., Waddington, J. M., and Macrae, M.: Increased Peatland Nutrient Availability Following the Fort McMurray Horse River Wildfire, Diversity, 11, 10.3390/d11090142, 2019.

Vance, E. D., Brookes, P. C., and Jenkinson, D. S.: An extraction method for measuring soil microbial biomass C, Soil Biology and Biochemistry, 19, 703-707, 10.1016/0038-0717(87)90052-6, 1987.

Webster, K. L., Bhatti, J. S., Thompson, D. K., Nelson, S. A., Shaw, C. H., Bona, K. A., Hayne, S. L., and Kurz, W. A.: Spatially-integrated estimates of net ecosystem exchange and methane fluxes from Canadian peatlands, Carbon Balance Manag, 13, 16, 10.1186/s13021-018-0105-5, 2018.

Worrall, F., Moody, C. S., Clay, G. D., Burt, T. P., and Rose, R.: The total phosphorus budget of a peat-covered catchment, Journal of Geophysical Research: Biogeosciences, 121, 1814-1828, 10.1002/2016jg003375, 2016.

Wright, S. N., Thompson, L. M., Olefeldt, D., Connon, R. F., Carpino, O. A., Beel, C. R., and Quinton, W. L.: Thaw-induced impacts on land and water in discontinuous permafrost: A review of the Taiga Plains and Taiga Shield, northwestern Canada, Earth-Science Reviews, 232, 10.1016/j.earscirev.2022.104104, 2022.

Wu, Y., Xu, X., McCarter, C. P. R., Zhang, N., Ganzoury, M. A., Waddington, J. M., and de Lannoy, C. F.: Assessing leached TOC, nutrients and phenols from peatland soils after lab-simulated wildfires: Implications to source water protection, Sci Total Environ, 822, 153579, 10.1016/j.scitotenv.2022.153579, 2022.