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Schmidt, M

http://hdl.handle.net/10026.1/19857

10.1007/s11273-022-09858-4
Wetlands Ecology and Management
Springer

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CO$_2$ uptake decreased and CH$_4$ emissions increased in first two years of peatland seismic line restoration

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Abstract:
Oil and gas exploration has resulted in over 300,000 km of linear disturbances known as seismic lines, throughout boreal peatlands across Canada. Sites are left with altered hydrologic and topographic conditions that prevent tree re-establishment. Restoration efforts have concentrated on tree recovery through mechanical mounding to re-create microtopography and support planted tree seedlings to block sightlines and deter the use of lines by wolves, but little is known about the impact of seismic line disturbance or restoration on peatland carbon cycling. This study looked at two mounding treatments and compared carbon dioxide and methane fluxes to untreated lines and natural reference areas of a wooded fen in the first two years post-restoration. We found no significant differences in net ecosystem CO$_2$ exchange, but untreated seismic lines were slightly more productive than natural reference areas and mounding treatments. Both restoration treatments increased ecosystem respiration, decreased net productivity by 6 – 21 g CO$_2$ m$^{-2}$ d$^{-1}$, and created areas of increased methane emissions, including an increase in the contribution of ebullition, of up to 2000 mg CH$_4$ m$^{-2}$ d$^{-1}$ over natural and untreated lines. Further research on this site to assess the longer-term impacts of restoration, as well as application on other sites with varied conditions, will help determine if these restoration practices are effective at restoring carbon cycling.

Keywords: Peatland restoration, seismic lines, mounding, carbon dioxide, methane

Acknowledgements: The authors would like to thank Bin Xu for leading restoration planning and groundwork on the site and Felix Nwaishi for input on study design. We would also like to thank Taylor Vodopija and Miranda Hunter for their help in the field and Blake Haskell for laboratory help.

This study took place on the unceded territories of the peoples of the Treaty 6 region and Métis Nation of Alberta, Region 4, and the authors live and work on the unceded traditional territory of the Attawandaron (Neutral), Anishinaabeg and Haudenosaunee peoples.

Declarations:
Funding: This work was supported by the Government of Canada Environmental Damages Fund, project number EDF-AB-2018c009. Research support was also provided via a Canada Research Chair to Maria Strack.

Conflicts of interest/competing interests: The authors declare no conflicts of interest.

Availability of data and material: The datasets generated during and/or analysed during the current study are available at https://doi.org/10.5683/SP3/CHCPWR, Scholars Portal Dataverse

Code availability: Not applicable

Authors’ contributions: All authors contributed to the study conception and design. Material preparation, data collection and analysis were performed by Megan Schmidt. The first draft of the manuscript was written by Megan Schmidt and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.
1. Introduction

The northern half of the province of Alberta, Canada is known for its vast underground oil and gas reserves and is covered by over 134,000 km² of boreal peatlands (AEP 2018). Extraction and exploration of oil and gas deposits has left a network of linear clearings, known as seismic lines, crisscrossing the boreal region even decades after their creation (Lee and Boutin 2006). Recent estimates put the total length of seismic lines through Alberta peatlands at over 345,000 km or 1900 km² (Strack et al. 2019), reaching mean density of up to 10 km/km² (Lee and Boutin 2006) or as high as 40 km/km² (Schneider 2002). Historically, seismic lines were created using heavy machinery during summer months, removing trees and vegetation, as well as roots and surface soil layers, to a width of 5 - 10 m (Bliss and Wein 1972); these are often referred to as “legacy lines”. Over time clearing began to take place over winter months and bulldozer blades were raised to reduce disturbance of the peat (Bliss and Wein 1972).

Through the 1990s, line width was reduced to approximately 5 m and by the 2000s, low impact seismic (LIS) lines of 1.5 – 5.5 m, cleared with light-weight equipment, had become the preferred method (Dabros, Pyper, and Castilla 2018). Undisturbed, peatlands sequester large amounts of carbon (C) (e.g., Loisel et al. 2014) and provide important habitat for species such as the threatened woodland caribou (Rangifer tarandus caribou) (Filicetti, Cody, and Nielsen 2019). Alteration of hydrologic and topographic conditions on seismic lines prevents the re-establishment of trees, contributing to habitat fragmentation and increased predation of caribou, which has spurred efforts to restore them. The focus of restoration has, until recently, been on tree recovery, but little is known about the effects of linear disturbances and restoration of these features on peatland C exchange; thus, this is the goal of this study.

The complexity of peatland systems makes them especially vulnerable to disturbance and prone to long, slow recovery, if they recover at all. Vegetation removal and surface alteration during seismic line construction likely start a domino effect on other biogeochemical factors, from hydrology to peat
properties and C cycling. Not only are hummock tops physically removed in preparation for seismic
surveying, but repeated passes of heavy machinery further compress the peat surface (Stevenson et al.
2019; Strack et al. 2019), resulting in rutting from machinery and an increase in hollow coverage (Lovitt
et al. 2018; Stevenson et al. 2019). Any microforms that do persist on seismic lines tend to be highly
suppressed, varying only a few centimeters from mean elevation, compared to up to a meter in natural
conditions (e.g., Caners and Lieffers 2014). Microform development in disturbed peatlands has been
shown to be resistant to natural formation processes such as *Sphagnum* growth and fire (van Rensen et
al. 2015; Stevenson et al. 2019), likely due to shallow water tables and increasing surface water pooling
that often sustains flooded conditions through the growing season (Caners and Lieffers 2014). This
reduces suitable sites for tree establishment and shifts seismic lines towards more *Carex* dominated
communities (Lee and Boutin 2006, van Rensen et al. 2015; Strack et al. 2018). Moss cover may be lower
on lines and exhibit slower growth overall, potentially due to increased light levels (Pouliot et al. 2011),
while hummock forming *Sphagnum* species must compete with other moss species, resulting in slowed
succession that often stalls in early stages (Caners and Lieffers 2014; van Rensen et al. 2015). Loss of
hummocks removes oxic zones that support higher rates of gross ecosystem productivity (GEP) and
methane (CH$_4$) oxidation, and in turn creates conditions that favour CH$_4$ production (Chimner et al.
2016; Strack et al. 2018). The net C uptake of the new plant community will determine the C balance on
the line; how it compares to the adjacent forested peatland will depend on the ability of a more
productive understory to compensate for the loss of C uptake by trees and increased CH$_4$ emissions.

Restoration of disturbed peatlands has the potential to return them to C sinks, as has been seen in
peatlands used for horticultural peat extraction (e.g., Strack et al. 2016; Nugent et al. 2018). Until
recently, restoration of seismic lines has been largely ignored due to the assumption that such linear
disturbances would recover naturally with time. What has been done has focused mainly on structural
restoration related to caribou habitat, such as reducing sightlines and access by predators (e.g., Filicetti
et al. 2019); full mitigation of the impacts of seismic lines requires a shift in this focus to include restoration of ecological functions for a return to successional peatland pathways. As previous restoration has been targeted at tree recovery, the focus has been on creating artificial mounds to replace microtopography, namely hummocks, lost during construction, and the use of these elevated microsites for tree establishment (Lieffers et al. 2017; Filicetti et al. 2019). Mounding is a mechanical process that scoops soil from one spot and places it nearby to create raised areas (hummocks) and leave low areas (hollows) that mimic natural microforms (Dabros et al. 2018). Studies on mounding have been ongoing for some time to restore peatland oil sands exploration well sites, and mounds on these sites are often large, high, and inverted to expose deeper peat and/or mineral soil (Echiverri et al. 2020; Murray et al. 2021). Not only does inversion eliminate any potential uptake of CO₂ by buried vegetation, but peat surfaces are also exposed to erosion (Shuttleworth et al. 2014) and increased decomposition (Smolander and Heiskanen 2007; Lieffers et al. 2017). Meanwhile, large, flooded hollows have the potential to become CH₄ emission hotspots (Murray et al. 2021). While mounding on seismic lines has become more common (Echiverri et al. 2020; Murray et al. 2021), knowledge of impacts on GHG production and emissions remains limited; this paper is the first to capture detailed, plot-scale CO₂ and CH₄ fluxes on unrestored and restored seismic lines.

In this study, CO₂ and CH₄ fluxes were measured in the first two summers immediately following restoration work. The objectives of this study were to: (1) quantify CO₂ and CH₄ fluxes of two types of mounding - inline mounding (IM), in which mounds and hollows were created on the seismic line and not inverted, and hummock transfer (HT), in which established natural hummocks just off the edges of the line were targeted and placed vegetated side up on the line, leaving the associated hollow in the bordering undisturbed peatland and these treatments were compared to untreated lines and adjacent undisturbed peatland; (2) assess the environmental factors contributing to variation in CO₂ and CH₄ fluxes on and off the lines.
2. Methods

2.1 Study site

The study area is located in central Alberta, Canada approximately 11 km southwest of Brazeau Dam (52.889326, -115.549173; Figure 3.1). The region is classified as Boreal Plains Ecozone (Environment Canada 2001) and Central Mixedwood Natural Subregion (AEP 2006). Average temperatures range from 14 – 17 °C in June, July, and August to -15 – -12 °C in December, January, and February with an average of 97 frost free days per year (ECCC 2021). The area receives an average 462 mm precipitation per year, with most occurring during the growing season, May to August (ECCC 2021). Wetlands comprise a large portion of the region, dominated by wooded and shrubby fens, in a mosaic of aspen and spruce upland (AEP 2006). The study site consists of ~3 km of peatland seismic line running north-south and east-west (Figure 1). Although the precise date of creation is unknown, satellite images show seismic lines present as far back as 1982. At time of groundwork in March 2019 the line edges were still clearly visible, with little regeneration of woody vegetation or trees. A hydrological gradient exists longitudinally, becoming drier as the center point is approached from both north and south ends. This corresponds to a gradient in vegetation and fen type, transitioning from rich fen at both north and south ends to poor fen in the center that extends the length of the east-west line. Dominant vegetation in the rich fen consists of Larix laricina (Du Roi) K. Koch, Picea mariana (Mill.) Bitton, Sterns & Poggenburg, Betula spp., Carex spp., Menyanthes trifoliata L., and Polytrichum spp. Dominant vegetation in the poor fen consists of P. mariana, L. laricina, Rhododendron groenlandicum (Oeder) K.A. Kron & W.S. Judd, Salix spp., M. trifoliata, Vaccinium oxyccos L., Sphagnum fuscum (Schimp.) H. Klinger., and Sphagnum magellanicum Brid. To capture these differences the site was divided into three subsites: north from the center point, south from the center point, and the entire east-west section. For this study only the north and central sections were chosen to create an equal number of sampling points in the poor and rich fen sections.
Figure 1 Study site with treatments, locations of collars, and photographs of the four treatments. Each treatment was applied in both the poor and rich fen, and two pairs of collars (hummock and hollow) were installed in each treatment-fen type combination (n = 8/treatment; 32 collars total).

2.2 Groundwork and project design

Groundwork was carried out in March 2019 on frozen ground, installation of research equipment took place in May 2019, and data collection began June 2019. Mounding was done by a backhoe with toothed digging bucket, moving an intact scoop of soil from within the seismic line and placing it nearby in the same orientation (i.e., vegetated side up) for IM, and targeting naturally formed hummocks just
off the line for HT. Hummocks with established woody shrubs and small trees were intentionally targeted (Figure 1). Hummocks for both treatments averaged a height of 20 cm, while hollows were an average depth of 10 cm in HT and 19 cm in IM. Single trees from the edges were randomly pulled down onto the line (often referred to as stem-bending) to provide additional microsites and tree seedbank inputs. Treatments were replicated on both poor fen and rich fen sections with some length of line left untreated as controls, and a natural site was selected for each section approximately 20 m away from the seismic line.

2.3 Carbon dioxide (CO₂) flux

We measured CO₂ fluxes approximately once per week from June to August 2019 and July and August 2020 via the closed chamber method (Griffis et al. 2000), in which acrylic chambers are placed onto metal collars installed in the peat. Pairs of 60 × 60 cm steel collars were permanently installed at each sample plot in corresponding hummocks and hollows. Two pairs were installed in each treatment in both sections, resulting in four replicate plots per microform per treatment. A clear acrylic chamber measuring 60 × 60 × 30 cm was set into a groove along the top of the collars that created a seal when filled with water. Air in the chamber was continuously circulated with a small battery-operated fan. CO₂ concentration in the chamber was measured at 15 second intervals for 105 - 120 seconds (~ 2 minutes) using a portable infrared gas analyzer (EGM-4, PP Systems, Massachusetts, USA), along with air temperature, relative humidity, and photosynthetically active radiation (PAR) within the chamber. An opaque tarp was used to create fully dark conditions, enabling ecosystem respiration (ER) to be measured. Order of sampling plots was changed daily to account for different light levels and solar angles throughout the day.

Net ecosystem exchange (NEE) is the overall exchange and direction of C movement between the atmosphere and an ecosystem, measured under full sun. Fluxes under dark condition capture ecosystem respiration (ER), and gross ecosystem productivity (GEP) is calculated as the difference between NEE and
ER (Chapin et al. 2006). In this study, we use the sign conventions that C uptake from the atmosphere is negative and emission to the atmosphere is positive (Ryan and Law 2005). Raw data were inspected for linearity of fluxes, controlling for fit of \( R^2 > 0.75 \), except for fluxes that were relatively unchanging, representing a flux close to zero. Processing resulted in a data loss of 43% (due to issues with collar sealing in the newly formed hummocks) in 2019 and 0.5% in 2020.

### 2.4 Methane (CH\(_4\)) flux

We measured CH\(_4\) fluxes from the same paired collars used for CO\(_2\) with an opaque chamber and fans to maintain circulation and reduce chamber heating from June to August 2019 and July to August 2020. In 2019 we collected CH\(_4\) by extracting 20 ml gas samples from the chamber via syringe at 5-, 10-, 15-, and 25-minutes post-closure that were injected into pre-evacuated Exetainers (Labco Ltd.) and analyzed via gas chromatography on a flame ionization detector (Shimadzu GC2014, Mandel Scientific) at the University of Waterloo. Atmospheric samples were collected via syringe and exetainer two to three times throughout each sampling day to provide CH\(_4\) concentration at time zero. Fluxes were inspected for linearity and outlying points associated with potential ebullition were removed to control for \( R^2 > 0.75 \), resulting in a 6% loss of fluxes.

We changed methods in 2020 to continuous measurement every second for 300 seconds (5 minutes) with a CH\(_4\)/CO\(_2\)/H\(_2\)O Trace Gas Analyzer (LI-7810, LI-COR, Nebraska, USA). Per-second measurements allowed for ebullition (bubble) events to be parsed from diffusive fluxes; however, to capture total CH\(_4\) contribution more accurately from the site, ebullitive fluxes were included in the data presented here. Fluxes were inspected for linearity following the same rules as CO\(_2\) and ebullition events identified when concentration change was >15 ppb s\(^{-1}\) for minimum 3 seconds. Diffusive fluxes were calculated from the average of the linear changes before and after ebullition events (Goodrich et al. 2011); when more than one ebullitive event was present during the chamber closure, the linear slope between all ebullition events were averaged. By subtracting the diffusive flux slope from ebullitive slope we determined the
additional CH₄ contribution of the ebullition events. The sum of the diffusive and ebullitive fluxes were used as the total flux; diffusive flux slopes were controlled for R² > 0.75 and in cases with an ebullition event(s) and no acceptable diffusive flux present, the ebullitive flux was used as the total flux. These processes resulted in only 1% loss of flux data.

We performed a cross-comparison between the two methods by collecting seven fluxes via the periodic sample extraction method immediately before measuring the same collar with the LI-COR. Cross-comparison found that, on average, flux determined by the extraction method was 1.3 times higher than the LI-COR but well within the range of fluxes from the research area (Supplementary table S3). In addition to the small sample size, differences could be due to longer chamber closure times (15-20 mins vs 5 mins) increasing chances of capturing ebullition events. No correction was applied to calculated fluxes from either method as we did not directly compare between years.

2.5 Vegetation community and environmental conditions

Additional environmental factors were measured every time C fluxes were measured. Soil moisture was measured with a ML3 ThetaProbe Soil Moisture Sensor (Delta-T Devices) at five points in the hummock or hollow to achieve an average for the feature. A thermocouple soil temperature probe was used to establish a temperature profile at -2, -5, -10, -15, -20, -25, and -30 cm. Water table was measured in a standpipe installed adjacent to each plot.

Vegetation surveys were conducted once in July of each summer. Each collar was assessed visually for percent cover of functional groups: *Sphagnum* spp., all other mosses (e.g., brown and feathermoss), graminoids (i.e., sedges, reeds, rushes, grasses), forbs, shrubs, and trees. Analysis showed similar relationships between C flux and individual vascular or moss plant functional groups so final analysis was based on data combined into two groups: all mosses and all vascular species.
To understand how shifts in environmental conditions between treatments influence productivity, we looked at the relationships of 2019 and 2020 data combined.

### 2.6 Data analysis

All data analysis was done in the statistical analysis program R (R Core Team 2013). As the main focus of this study was the impact of treatments on CO$_2$ and CH$_4$ exchange, we did not investigate the effect of fen type but rather treatment, microform, and their interaction as fixed effects in a separate linear mixed effects model for each flux component in the package *nlme* (Pinheiro et al. 2014). To account for repeated measures, collar was included as the random factor in each model. We used these models to assess treatment and microform impacts on GEP, ER, NEE, and CH$_4$ in each of 2019 and 2020, separately. The years were separated a priori due to differences in sample size and methods for CH$_4$ measurement between the years. Differences were considered statistically significantly when p < 0.05 using the *anova* output command for each model. When a significant effect was present, post-hoc Tukey tests via the *lsmeans* package (Lenth 2016) were performed for pairwise comparisons. CH$_4$ data was log transformed to improve normality of the residuals and a value of 1.6 added prior to transformation to adjust for negative values.

To understand how shifts in environmental conditions between treatments influence productivity, we created additional linear mixed effects models using combined 2019 and 2020 data with either water table or soil temperature as fixed effects along with treatment and interaction of water table or soil temperature with treatment to evaluate whether response to environmental variables differed between treatments; plot was included as a random effect. We also investigated whether vegetation cover explained variation in C flux components. As vegetation was measured only once in July of each study season, C flux components were expressed as a seasonal average for each plot. Linear regressions, using the *lm* function were used to evaluate the effect of vascular or moss cover on variation in C fluxes.
3. Results

3.1 Environmental conditions

In general, average water table in the rich fen section was higher than the poor fen section at 0 cm and -10 cm, respectively. Although the difference was significant ($F_{1,44} = 62.2$, $p < 0.0001$), we were mainly interested in the overall treatment effects and how they affected ecosystem function across the range of hydrological variation across the entire fen and therefore focused on treatment and microform effects (Table 1). Across the fen, microform alone significantly impacted water table (2019: $F_{1,20} = 47.7$, $p < 0.0001$; 2020: $F_{1,24} = 20.6$, $p < 0.0001$); treatment and the interaction with treatment were not significant. Water table followed the surface elevation gradient of microforms from hummocks to hollows, with the largest difference between HT hummocks (highest surface, deepest water table) and IM hollows (lowest surface, often inundated). Microform was also the only significant factor explaining variation in soil temperature at 10 cm below surface in both 2019 and 2020 (2019: $F_{1,20} = 11.9$, $p = 0.0025$; 2020: $F_{1,24} = 25.7$, $p < 0.0001$). Soils were warmest in hummocks and coolest in hollows, but differences were minimal aside from HT and IM, where hummocks were on average 3 °C warmer than hollows in both years (Table 1).
Table 1 Mean (SE) environmental conditions and vascular/moss cover for each treatment across both hummocks and hollows. Moss and vascular cover were measured once per year (n = 4/microform/treatment) while soil temperature and water table were measured concurrent with fluxes over the season (n 2019 = 12/microform/treatment; n 2020 = 30/microform/treatment).

| Treatment     | Soil temp 10 cm (°C) | Water table (cm) | Moss cover (%) | Vascular cover (%) |
|---------------|----------------------|------------------|----------------|-------------------|
|               | 2019     | 2020     | 2019     | 2020     | 2019     | 2020     | 2019     | 2020     |
| Natural       | 15 (0.8) | 18 (0.4) | -3 (5.5) | -11 (4.5) | 66 (18.3) | 77 (11)  | 64 (7.3) | 60 (6.9)  |
| Hummock       | 15 (0.7) | 18 (0.6) | -11 (4.3) | -17 (4.7) | 100 (0)   | 96 (3.1) | 80 (2.9) | 69 (8.3)  |
| Hollow        | 14 (1.5) | 17 (0.3) | 6 (7.7)  | -4 (6.9)  | 32 (22.4) | 58 (17.8)| 45 (3.3) | 52 (10.2)|
| Untreated     | 15 (0.6) | 19 (0.4) | -3 (2.7) | -6 (2.2)  | 91 (5.5)  | 96 (3.1) | 71 (8.4) | 47 (5.5)  |
| Hummock       | 15 (1.1) | 19 (0.2) | -8 (2.5) | -6 (3.5)  | 88 (11.7) | 99 (0.5) | 53 (4.4) | 51 (9.3)  |
| Hollow        | 15 (0.9) | 18 (0.6) | 1 (3.3)  | -5 (3.2)  | 93 (3.3)  | 92 (5.9) | 89 (3.8) | 43 (6.8)  |
| Hummock Transfer | 16 (0.6) | 18 (0.4) | -5 (4.8) | -10 (4.1) | 45 (16.7) | 51 (18.7)| 33 (10.4)| 56 (13.2)|
| Hummock       | 17 (0.5) | 19 (0.2) | -16 (3.7) | -20 (3.6) | 89 (6.6)  | 100 (0)  | 35 (7.0) | 64 (20.7)|
| Hollow        | 14 (0.5) | 17 (0.3) | 6.5 (2.7)| 0 (1.1)   | 2 (1.2)   | 1 (1.3)  | 31 (21.4)| 48 (18.2)|
| Inline        |           |           |           |           |           |           |           |           |
| Mounding      | 14 (0.9) | 17 (0.7) | 1 (4.8)  | -3 (3.4)  | 46 (16.7) | 51 (17.7)| 31 (7.6) | 42 (10.4)|
| Hummock       | 16 (0)   | 19 (0.6) | -11 (2.1)| -11 (2.5) | 90 (3.5)  | 98 (1.8) | 46 (5.9) | 62 (14.4)|
| Hollow        | 13 (1.2) | 16 (0.8) | 13 (3.6) | 5 (2.1)   | 2 (1.9)   | 4 (2.5)  | 15 (8.5) | 23 (6.1) |
Moss cover was higher in natural and untreated sections than IM and HT, and both treatment (2019: $F_{3,20}=13.1$, $p<0.0001$; 2020: $F_{3,24}=20.4$, $p<0.0001$), and microform (2019: $F_{1,20}=121.6$, $p<0.0001$; 2020: $F_{1,24}=151.9$, $p<0.0001$) were significant factors. Moss cover on natural and untreated hummocks ranged from 88 – 100 % and remained at 89 – 100 % in HT and IM (Table 1). In hollows, moss cover dropped from 32 – 93 % in natural and untreated to 1 – 4 % in HT and IM resulting in a significant treatment-microform interaction (2019: $F_{3,20}=13.4$, $p<0.0001$; 2020: $F_{3,24}=21.1$, $p<0.0001$). Similarly, vascular plant cover was higher in natural and untreated areas (Table 3.1); treatment was significant in 2019 immediately following restoration, but not in 2020 (2019: $F_{3,20}=7.7$, $p=0.0012$; 2020: $F_{3,24}=0.8$, $p=0.5059$). Conversely, microform was not significant in 2019 but was in 2020 (2019 $F_{1,20}=1.4$, $p=0.2388$; 2020 $F_{1,24}=5.0$, $p=0.0347$) with higher cover on hummocks. The difference in vascular plant cover between microforms varied with treatment, with greater differences in HT and IM, but the interaction of treatment and microform was significant in 2019 only (2019: $F_{3,20}=4.1$, $p=0.0197$; 2020: $F_{3,24}=0.5$, $p=0.6535$). Vascular cover dropped from 51 – 80 % on natural and untreated hummocks to 35 – 64 % on HT and IM hummocks. In hollows vascular cover dropped from 43 – 89 % in natural and untreated to 15 – 48 % in HT and IM.

Soil temperature (2019 $F_{1,20}=9.8$, $p=0.0052$; 2020 $F_{1,24}=43.9$, $p=<.0001$) and water table (2019 $F_{1,20}=123.0$, $p=<.0001$; 2020 $F_{1,24}=67.2$, $p=<.0001$) both had significant effects on moss cover in both years. Overall, moss cover decreased with shallower water tables (i.e., water closer to, or above, surface) and increased with warmer soil temperatures. These relationships were steepest in HT, followed by IM, natural, and untreated (WT:treatment 2019: $F_{3,20}=1.5$, $p=0.2356$; 2020: $F_{3,24}=5.2$, $p=0.0065$; soil temperature:treatment 2019: $F_{3,20}=3.3$, $p=0.0381$; 2020: $F_{3,24}=3.4$, $p=0.0336$). Treatment alone was significant ($F_{3,20}=6.1$, $p=0.0041$) for explaining variation in vascular plant cover in 2019. In 2020, water table ($F_{1,24}=16.0$, $p=0.0005$) and soil temperature ($F_{1,24}=5.1$, $p=0.032$) were significant, regardless of
treatment. As with moss cover, vascular plant cover decreased with shallower water tables and increased with warmer soils.

### 3.2 Carbon exchange

Average values for all C fluxes in each sampling year are given in the Supplementary Material (Table S1). After processing, 84 CO₂ fluxes were included for 2019. Average productivity across the study plots was significantly reduced (i.e., less negative GEP) in the two restoration treatments (HT -16.9 g CO₂ m⁻² d⁻¹; IM -10.8 g CO₂ m⁻² d⁻¹) compared to natural (-24.8 g CO₂ m⁻² d⁻¹) and untreated (-36.2 g CO₂ m⁻² d⁻¹) sections. Generally, hummocks were more productive than hollows; however, this difference was only significant in HT (Table 2, Figure 3.2). Ecosystem respiration did not differ significantly between any treatments or microforms (Table 2) although it ranged from 9.9 g CO₂ m⁻² d⁻¹ in natural to 16.4 g CO₂ m⁻² d⁻¹ in untreated. Respiration rates in hollows were roughly half that of hummocks except in untreated, where hollow ER was slightly higher than hummocks. On average across the study plots, NEE was positive (i.e., release to atmosphere) in IM and did not significantly differ from HT (IM 0.9 g CO₂ m⁻² d⁻¹; HT -4.8 g CO₂ m⁻² d⁻¹); similarly, natural and untreated did not significantly differ (natural -14.9 g CO₂ m⁻² d⁻¹; untreated -19.8 g CO₂ m⁻² d⁻¹). Both IM hummocks and hollows acted as sources while natural, untreated, and HT acted as sinks under the measured conditions (Figure 2).
Table 2 Results from the linear mixed effects models describing effects of treatment, microform, and interactions on CO$_2$ and CH$_4$ flux.

| Flux Component | Effect     | DF | F-value | p-value | Effect     | DF | F-value | p-value |
|----------------|------------|----|---------|---------|------------|----|---------|---------|
| GEP            | Treatment  | 3,22| 7.9     | 0.0009  | Treatment  | 3,24| 5.7     | 0.0041  |
|                | Microform  | 1,22| 1.3     | 0.257   | Microform  | 1,24| 9.7     | 0.0046  |
|                | Treatment: | 3,22| 3.3     | 0.0362  | Treatment: | 3,24| 4.8     | 0.0088  |
|                | Microform  | 1,22| 1.3     | 0.293   | Microform  | 3,24| 8.2     | 0.0006  |
|                | Microform  | 1,24| 9.7     | 0.0046  | Microform  | 1,24| 52.7    | <.0001  |
|                | Treatment: | 3,24| 4.8     | 0.0088  | Microform  | 3,24| 8.2     | 0.0006  |
|                | Microform  | 3,24| 6.0     | 0.0033  | Microform  | 1,24| 0.3     | 0.5335  |
|                | Microform  | 1,22| 0.0     | 0.9535  | Microform  | 1,24| 2.4     | 0.0844  |
|                | Treatment: | 3,24| 8.2     | 0.0006  | Microform  | 3,24| 8.2     | 0.0006  |
|                | Microform  | 3,24| 2.4     | 0.0844  | Microform  | 3,24| 8.2     | 0.0006  |
| CH$_4$         | Treatment  | 3,22| 1.0     | 0.3812  | Treatment  | 3,28| 2.9     | 0.0503  |
|                | Microform  | 1,22| 4.3     | 0.0478  | Microform  | 1,270| 2.8   | 0.0926  |
|                | Treatment: | 3,22| 1.6     | 0.2123  | Treatment: | 3,270| 1.9   | 0.1241  |
|                | Microform  | 1,24| 9.7     | 0.0046  | Microform  | 1,270| 2.8   | 0.0926  |
|                | Microform  | 1,270| 2.8 | 0.0926  |
In 2020, 247 CO$_2$ fluxes were included in the dataset. Productivity was significantly lower in IM (-20.5 g CO$_2$ m$^{-2}$ d$^{-1}$) than untreated (-38.4 g CO$_2$ m$^{-2}$ d$^{-1}$). Natural and HT were intermediate and did not significantly differ at -34.9 g CO$_2$ m$^{-2}$ d$^{-1}$ and -28.9 g CO$_2$ m$^{-2}$ d$^{-1}$, respectively. Microform had no significant effect on GEP (Table 2), except for IM hollows, which were much less productive (i.e., less negative GEP) than any other microform or treatment (Figure 2). Although treatments did not significantly differ in ER (Table 2), hummocks generally respired more than hollows. Respiration from IM and HT hollows (6.0 g CO$_2$ m$^{-2}$ d$^{-1}$; 9.5 g CO$_2$ m$^{-2}$ d$^{-1}$) was significantly lower than from natural and untreated hollows (13.6 g CO$_2$ m$^{-2}$ d$^{-1}$; 15.4 g CO$_2$ m$^{-2}$ d$^{-1}$) and similarly higher from IM and HT hummocks (20.4 g CO$_2$ m$^{-2}$ d$^{-1}$; 22.2 g CO$_2$ m$^{-2}$ d$^{-1}$) than natural and untreated (17.8 g CO$_2$ m$^{-2}$ d$^{-1}$; 17.6 g CO$_2$ m$^{-2}$ d$^{-1}$). NEE was significantly lower (i.e., less negative, less uptake) in IM (-7.4 g CO$_2$ m$^{-2}$ d$^{-1}$) than natural (-19.3 g CO$_2$ m$^{-2}$ d$^{-1}$) or untreated (-21.0 g CO$_2$ m$^{-2}$ d$^{-1}$), while HT (-13.0 g CO$_2$ m$^{-2}$ d$^{-1}$) remained lower than natural and untreated but higher than IM. Only IM hollows had significantly lower net uptake among all microform-treatment combinations, following a similar pattern to 2019 (Figure 2).
Figure 2 Fluxes of carbon dioxide (CO₂; g CO₂ m⁻² d⁻¹) (NEE = net ecosystem exchange, ER = ecosystem respiration, GEP = gross ecosystem productivity) and methane (CH₄; mg CH₄ m⁻² d⁻¹) from restoration treatments in 2019 (left) and 2020 (right) across both hummocks and hollows. Seasonal means are significantly different if they do not share letters, as per post-hoc Tukey tests. Capital letters refer to the significance of treatments overall; lower case letters refer to differences between microforms. Boxes represent the upper and lower quartiles in which 50% of data points lie and whiskers represent 1.5 times the interquartile range. Median is shown by a horizontal line within the box. Statistical analyses for CH₄ were performed on log transformed data but are shown here as raw flux concentrations.
Figure 3 Relationship between water table and CO₂ and CH₄ fluxes across both hummocks and hollows in 2019 and 2020 combined. Linear mixed effects models were used to determine relationships and post-hoc Tukey tests via the *lsmeans* package (Lenth 2016) were performed on *anova* results to determine significant difference between treatments. Lines represent trends and letters represent significant differences in slopes; treatments are significantly different if they do not share letters. A total of 110 CH₄ fluxes passed quality control and were included for 2019. Fluxes ranged from 65.8 mg CH₄ m⁻² d⁻¹ in natural to 775.7 mg CH₄ m⁻² d⁻¹ in IM and were generally higher from hollows than hummocks (Figure 2). However, CH₄ flux was not significantly different between treatments or microforms and there was no significant treatment and microform interaction (Table 2). In the 2020 dataset, 306 CH₄ fluxes were included in the analysis. There was no significant effect from treatment and across all microform-treatment combinations only natural hummocks (122.2 mg CH₄ m⁻² d⁻¹) and IM hollows (1282.2 mg CH₄ m⁻² d⁻¹) were significantly different. Ebulition was captured in 23 flux
measurements (Table S2): 2 in untreated, 1 HT hummocks, 5 HT hollows, 1 IM hummocks, and 14 IM hollows. Average ebullitive flux was 337.9 mg CH$_4$ m$^{-2}$ d$^{-1}$ in untreated, 349.0 mg CH$_4$ m$^{-2}$ d$^{-1}$ in HT, and 1752.2 mg CH$_4$ m$^{-2}$ d$^{-1}$ in IM, with no measured ebullition at natural plots.

Across the full 2019-20 dataset, higher productivity (i.e., more negative GEP) was significantly related to deeper water tables both alone (F$_{1,293}=34.0$, p<0.0001) and in interaction with treatment (F$_{3,293}=3.4$, p=0.0163). There were no significant differences in slope between HT and IM, which had the steepest slopes and were significantly less productive than untreated and natural when water tables were above -20 cm (Figure 3). Ecosystem respiration significantly decreased with shallower water tables (F$_{1,293}=96.6$, p<0.0001) and although treatment did not have a significant effect on respiration, the interaction with treatment did (F$_{3,293}=4.6$, p=0.0033), resulting in similar, steeper slopes for HT and IM, followed by untreated and natural (Figure 3). Overall, the effect of water table (F$_{1,293}=4.4$, p=0.0356) on NEE was significant, resulted in increasing net CO$_2$ uptake (i.e., more negative) in natural and untreated with shallower water table, and decreasing net productivity in HT and IM. Slopes were very shallow and only untreated and IM were significantly different (Figure 3). Water table position did not significantly predict CH$_4$ flux, but flux did generally increase with shallower water table (F$_{1,378}=2.8$, p=0.0944); there was also no significant interaction between treatment and water table (F$_{3,378}=0.7$, p=0.5258).

Productivity was significantly greater (i.e., became more negative) with warmer 10 cm soil temperature (F$_{1,269}=28.4$, p=<0.0001) and there was no significant interaction with treatment (F$_{3,269}=2.5$, p=0.0581).

Soil temperature alone (F$_{1,269}=36.5$, p=<0.0001) also had a significant effect on ER, leading to higher respiration with warmer soils. The overall impact of soil temperature on NEE was significant (F$_{1,269}=8.5$, p=0.0038) with greater net uptake from warmer soils, and there was no significant interaction with treatment (F$_{1,269}=1.8$, p=0.1383). Higher soil temperatures also resulted in significantly higher CH$_4$ fluxes (F$_{1,349}=12.2$, p=0.0005) with no significant treatment interaction.
Figure 4 Relationships between moss (left) and vascular (right) percent cover on net ecosystem exchange (NEE) and CH$_4$. Linear mixed effects models were used to determine relationships and post-hoc Tukey tests via the `lsmeans` package (Lenth 2016) were performed for pairwise comparisons between slopes when a significant interaction with treatment was present. Both moss and vascular plant cover significantly explained variation in C fluxes (Figure 4), with no significant interactions with treatment in any case. GEP significantly increased (i.e., became more negative; $F_{1.52}=36.3$, $p<0.0001$) with higher cover of mosses and increasing vascular plant cover ($F_{1.52}=28.6$, $p<0.0001$). Respiration significantly increased with both increasing moss ($F_{1.52}=52.5$, $p<0.0001$) and vascular plant cover ($F_{1.52}=19.4$, $p<0.0001$). NEE also significantly increased with increasing moss ($F_{1.52}=15.4$, $p=0.0002$) and vascular plant cover ($F_{1.52}=21.1$, $p<0.0001$). Across all
treatments, CH4 fluxes decreased with increasing moss cover (F_{1,52}=16.0, p=0.0001; Fig 4). Emission of CH4 generally decreased with increasing vascular cover although there were no significant effects.

4 Discussion

4.1 Environmental conditions

The creation of seismic lines through our study site altered water tables, vegetation composition, CO2 release and uptake, and CH4 emission. Though not significantly different, average water tables in untreated sections were approximately 5 cm closer to the surface than in the undisturbed fen, leading to wetter conditions overall (Table 1). This was as expected as previous studies have shown depression in elevation and flattening of hummocks on lines (Stevenson et al. 2019) and concurrent rise in water table (e.g., Lovitt et al., 2018). Additionally, the study area received more than the average total annual precipitation from May 1 to August 31 in both years (551 mm and 510 mm for 2019 and 2020 respectively; ACIS 2020). These wet conditions were most noticeable in 2019, when water levels were near or above surface much of the season. Both mounding treatments increased the difference in water table between hummocks and hollows compared to untreated, as was intended. Surprisingly, there was very little difference among treatments in soil temperature at 10 cm below surface in both years, contrary to what was found by Strack et al. (2018) who reported warmer soils on seismic lines than the adjacent forested peatlands. The wet conditions and surface water flow through the fen likely limited the development of local temperature differences.

Moss cover was higher than vascular plant cover in both years, and although not significant, was higher on the untreated lines than in natural. Moss cover had a significant negative relationship with increasing water tables, and flooding has been shown to negatively impact some moss species (Granath et al. 2010), likely explaining the increase of moss cover in 2020 vs 2019, especially in natural hollows where water table dropped from 6 cm above surface to 4 cm below. Contrary to previous findings (Strack et al. 2018; Davidson et al. 2021), vascular plant cover was not significantly higher on the lines than the
surrounding natural fen (Table 1), except for low spots on untreated lines in 2019, when they were wetter than hummocks in both untreated and natural but drier than natural hollows. This may be a result of differences in peatland types across studies, highlighting the need for further trials of these techniques. Murray et al. (2021) also found an increase in vascular plant cover versus natural peatland conditions at wet, unmounded control sites on oil sands exploration (OSE) well pads but not on drier sites. Lower vascular plant cover can partially be attributed to the lack of woody vegetation, primarily shrubs, on the line, although increased light availability typically leads to higher cover of herbaceous species (Caners and Lieffers 2014; Strack et al. 2018; Murray et al. 2021). An increase on the line of primarily graminoids (e.g., Carex spp. and Juncus spp.) that have small stem diameter and lack large, spreading leaves (e.g., Menyanthes trifoliata, Caltha palustris), may explain some of the decrease in cover, as may the difference in precipitation and water levels between the two years. As water levels drop hummocks become drier, and the deepest water tables may not be accessible for some plants, while hollows become less flooded, allowing for species that prefer drier conditions to flourish while still sustaining water-loving species with shallow water tables. In natural areas, this maintains a steady cover, though it fluctuates over time; on untreated lines, however, consistently shallow water tables may prevent survival of species that prefer elevated microsites even in drier years.

Moss cover did not significantly change from natural and untreated areas on hummocks in either mounding treatment, but both increased from 2019 to 2020 (Table 1). While vascular plant cover was approximately half that of natural plots in 2019, it had recovered to similar levels as natural and untreated by 2020. Both the decrease and recovery can be attributed to our restoration techniques; disturbance during restoration work caused the loss of some plant cover, while not inverting the mounds maintained much of the existing vegetation, roots, and seedbank, allowing for better growth the following year. This contrasts with previous mounding techniques that invert the mounds and leave bare peat and/or mineral soil exposed. Murray et al. (2021) found only 19 % moss and 21 % vascular
plant cover on inverted mounds five and six years following mounding. Moss cover was near zero in
hollows in both years, but approximately one third the vascular plant cover of natural was maintained in
IM hollows and two thirds in HT hollows in the first year. By 2020, vascular cover in HT hollows was
similar to natural and untreated areas; after five years vascular cover was still only 15% in Murray et
al.’s (2021) hollows. The shallow depth of our hollows, particularly in HT where the natural hummock
accounted for most of the height of the removed peat, not only left shallower pools of water, but also
left some roots, rhizomes, and seedbank in place for recovery to begin immediately.

4.2 Changes in carbon exchange on untreated seismic lines
Carbon dioxide fluxes were not significantly different on the untreated lines compared to natural
reference plots in either year (Table 2), but there was generally greater productivity and net CO2 uptake
on the line (Figure 2). Higher GEP (i.e., more negative) on the lines could be due to the lack of
microtopography and resulting water tables; following high precipitation flooding was shallower in
untreated sections than natural hollows, and during drier periods water table was not as far below
ground. The difference in water table depths between years on untreated lines was also much less than
in natural, leading to a steadier water supply for plants. Removal of trees and shrubs likely removes
some of the competition for water on the lines as well. Moreover, removal of tree and shrub cover
increases solar radiation reaching the peat surface on the seismic line and higher temperature and
longer growing seasons have been shown to increase GEP (Lund et al. 2010). Davidson et al. (2021)
found that peatland seismic lines greened up earlier and had higher GEP than natural reference sites.
Although ER was higher from untreated lines, there was no significant difference from the undisturbed
fen in either year, or between microforms. ER decreased as water levels rose and increased with
warmer temperature; an increase in soil temperature of as little as 1°C has been shown to increase
respiration (Walker et al. 2016). Warmer soils on the line spurs photosynthesis, the associated
autotrophic respiration from roots and mycorrhizae (Ryan and Law 2005) and heterotrophic respiration
of microbial and enzyme communities (Pendall et al. 2004). Rising water levels slow microbial growth and activity, leading to a decrease in respiration, as in natural hollows in 2019 (Figure 3). ER rates in hummocks also increase as CH$_4$ diffuses through the acrotelm and is converted to CO$_2$ by methanotrophs (Robroek et al. 2015). Overall, this site is consistent with results from Strack et al. (2018) and Davidson et al. (2021) who both found significantly higher productivity and NEE on seismic lines with increased total vascular cover; however, as total vascular cover was not significantly different from natural, change to species composition or functional groups may be more important for altering carbon cycling than changes to cover alone.

Although neither soil temperature nor distance of water table from surface were significantly different in natural or untreated areas, warmer, wetter soils on the untreated lines favoured the production of CH$_4$ as methanogenic microbial activity increased under anoxic conditions. In addition to reaching the atmosphere via diffusion, some plants, including many sedge and rush species, including *Eriophorum vaginatum* found on the site, mediate the movement of CH$_4$ directly from peat to atmosphere (Carmichael et al. 2014), bypassing much of the potential oxidation in the peat profile. Ebullition events have also been shown to increase with rising soil temperatures (Fechner-Levy and Hemond 1996) and shallower water table depths (Lai 2009) but were only captured in 2/82 (2.4 %) fluxes of untreated lines and never from natural areas. Shallower water tables on the untreated lines reduced the depth or presence of anoxic layer, likely increasing methanogen activity and production of CH$_4$, while the lack of woody roots and organic matter led to a change in peat composition or density that allows for increased diffusion of CH$_4$ to the atmosphere. These results reflect those of Strack et al. (2018), who found GEP and NEE improved, ER varied little, and CH$_4$ increased on a winter road versus the adjacent natural wooded fen, although with more significant differences than on our site.

### 4.3 Impacts of restoration treatments on CO$_2$ and CH$_4$ exchange
There were few significant differences between IM and HT or between HT and natural and untreated areas in either year, although GEP approximately doubled from 2019 to 2020 in both mounding treatments (Figure 2), mirroring the recovery of vascular plant cover (Table 1). Low productivity on IM hummocks was largely the result of disturbance during restoration and root systems that were adapted to saturated conditions suddenly being moved above the ground surface, resulting in water level differences of ~10 cm. Hummocks may not have yet fully integrated and connected with the underlying surface, preventing water from rising up the peat column to the rooting zone. Removal of vegetation in HT hollows was the leading factor in low GEP; however, because we targeted established natural hummocks to transfer, resulting hollows were much shallower, with water levels and soil temperatures in line with those of natural hollows. Many roots, small plants, and some mosses were left behind, and both mosses and vascular species were able to move in from the edges. ER was not significantly different in IM or HT than natural or untreated, and only differed significantly between microforms in 2020. Hollows in IM and HT had decreased vegetation cover and water levels at or above surface both years, which favours CH₄ production (Chimner et al. 2016; Strack et al. 2018). ER from hummocks in IM and HT, however, was higher than natural and untreated in both years (Figure 2), likely due to a thicker oxic layer in the rooting zone (Pypker 2013) that resulted in increasing microbial activity and organic matter decomposition. Overall, decreased NEE (i.e., less negative) in the two mounding treatments is due to the loss and disturbance of vegetation and surface peat soils, shown by lower productivity and increased respiration.

While CH₄ emissions in both treatments were not significantly different than natural or untreated plots in either year, mounding increased CH₄ emission overall (Figure 2). At 585 mg CH₄ m⁻² d⁻¹ in 2019 and 775 mg CH₄ m⁻² d⁻¹, emissions were five to eight times higher in IM than natural. Presumably, some of the increase came from diffusion and plant mediated transport; however, ebullition events were much more frequent, occurring in 15/70 (21 %) of IM fluxes in 2020, predominantly from hollows. In contrast,
only 6/79 (7.6 %) HT fluxes contained ebullition events, possibly due to the shallower hollows formed in this treatment.

4.4 Implications for management

When considering C fluxes measured in the study plots, these results include only CO2 and CH4 fluxes from the hummocks and hollows created during the treatment and not the entire treatment area. By assessing the density and area of hummocks and hollows and treatment area length and width, mean treatment CO2 and CH4 fluxes can be estimated (Table 3). As the entire area is not mounded within a treatment, we used the mean flux value from across all collars in untreated sections to represent the flat (i.e., untreated) parts of HT and IM areas. This is not a prediction of annual or seasonal emissions, but rather an estimate of peak season carbon exchange based on proportions of hummocks, hollows and untreated areas within each treatment type. Total treatment uptake of CO2 (NEE) increased in all four areas from 2019 to 2020, and untreated had the highest net CO2 uptake in both years at 20 and 21 g CO2 m^-2 d^-1. In 2019, natural, HT, and IM all averaged 15 g CO2 m^-2 d^-1, and in 2020 natural was only slightly higher than HT and IM at 19, 18, and 17 g CO2 m^-2 d^-1, respectively. CH4 emission was also higher across treatments in 2020 than 2019 despite it being a drier year. Both years followed the same pattern, with natural emitting the least CH4 (64 and 139 mg CH4 m^-2 d^-1), followed by untreated (135 and 234 mg CH4 m^-2 d^-1), HT (169 and 260 mg CH4 m^-2 d^-1), and IM (280 and 438 mg CH4 m^-2 d^-1). This is in line with Strack et al.'s (2019) modeling of increased CH4 emissions on seismic lines due to shallower water tables.

Notably, we did not look at the contribution of trees to C emissions and uptake, which could potentially shift fluxes, although some tree and shrub roots inevitably contributed to ER of natural collars and of HT hollows, which were located outside of the defined edges of the lines. Nor did we look at the potential influences of tree seedlings planted across the site; however, planting avoided any features with collars, so they did not interfere either. Taking tree and shrub cover into account, the natural peatlands actually have a higher NEE than measured; Murray et al. (2021) estimated trees and shrubs in a wooded
moderately rich fen captured an additional 50 g C m$^{-2}$y$^{-1}$ through total above and belowground productivity. Future research should aim to determine seasonal to annual carbon balance of mounding treatments including the contribution of trees to carbon uptake.

Table 3 Proportion of treatment areas covered by hummocks, hollows, and flat (i.e., untreated), estimated net ecosystem exchange$^a$ (g CO$_2$ m$^{-2}$ d$^{-1}$), and estimated total methane flux (mg CH$_4$ m$^{-2}$ d$^{-1}$).

| Treatment            | Proportion of treatment area (%) | Estimated Total NEE$^a$ | Estimated Total CH$_4$ |
|----------------------|---------------------------------|-------------------------|-------------------------|
|                      | Hummocks | Hollows | Flat |                      |                        |                      |
| Natural              | 50.0     | 50.0    | -    | -19.3                | 138.8                  |
| Untreated            | 100.0    | -       | -    | -21.1                | 234.3                  |
| Hummock Transfer     | 1.3      | 2.4     | 96.3 | -20.7                | 237.1                  |
| Inline Mounding      | 0.6      | 1.6     | 97.8 | -20.7                | 252.8                  |

$^a$Net ecosystem exchange represents full sun conditions and includes only understory plant community. It should not be interpreted as a growing season rate. Data for flat areas taken from the average of all collars in the untreated area.

These results indicate that mounding results in a decline in CO$_2$ uptake and increase in CH$_4$ emission, at least in the first two years post-restoration, largely due to loss of productivity and high CH$_4$ in the created hollows. However, given the low density of mounds applied in these treatments, changes across the entire treatment area were small. Restoration of seismic lines and exploration well pads has, until this point, typically involved inverted mounds at very high densities of 1000 – 1200 mounds/ha (e.g., Golder 2015; Murray et al. 2021). In comparison, our mounds were not inverted and were applied at a density of 155 mounds/ha. It is important that lower mounding densities are considered going forward to maintain untreated surfaces and manage not only production of CO$_2$ and CH$_4$, but also maintain uptake of CO$_2$ from the plant community established on the lines, although the best density is as yet uncertain and may vary between peatland types. Over time C uptake through the re-establishment and growth of trees and woody species on the line may compensate for increased ER and CH$_4$, but the time
needed for this transition remains unclear largely due to the young age of most restoration projects. Therefore, regardless of tree establishment, increased CH₄ emissions from hollows are likely to persist for years. For example, Murray et al. (2021) measured CH₄ fluxes of 14.3 g CH₄ m⁻² y⁻¹ (39.2 mg CH₄ m⁻² d⁻¹) from hollows 9 years after restoration, compared to 1.4 g CH₄ m⁻² y⁻¹ (3.8 mg CH₄ m⁻² d⁻¹) from natural and 6.2 g CH₄ m⁻² y⁻¹ (16.8 mg CH₄ m⁻² d⁻¹) from untreated sections of an oil sands exploration well-pad. The results from our study help to fill the knowledge gap on C exchange on peatland seismic lines and the potential outcomes of restoration. As this is the first study of its kind, measurements will need to be collected in the coming years to determine the long-term impacts on C emissions from the restoration treatments and which method results in the best outcomes for tree establishment and growth as well as ecosystem C and greenhouse gas exchange. Although untreated areas had the highest NEE (i.e., most negative) both years, by the second year post-restoration the difference between untreated, natural, and HT had declined, suggesting that with time treated areas may have the same or greater C sink strength as untreated and natural areas. The treatments will also need to be applied and studied across more peatlands of differing hydrologic conditions and vegetation types to develop best management practices across a range of peatland conditions. This may help satisfy the goals of restoration from multiple angles, including reduced time and cost for industry, restoration of habitat and connectiveness for conservation, and reduction of GHG production. Ideally, these practices would put restored seismic lines on pathways to improved function, ultimately helping to mitigate climate change while protecting biodiversity.
References:

Alberta Climate Information Service (ACIS) (2020) Current and Historical Alberta Weather Station Data Viewer. Government of Alberta. https://www.acis.alberta.ca/acis. Accessed 09 August 2021.

Alberta Environment and Parks (AEP) (2018) Alberta Merged Wetland Inventory. https://geodiscover.alberta.ca/geoportal/catalog/search/resource/details.page?uuid=%7BA73F5AE1-4677-4731-B3F6-700743A96C97%7D. Edmonton, Alberta. Accessed 24 October 2019.

Alberta Environmental Protection (2006) Natural Regions and Subregions of Alberta: Summary. Publ. No 1/531. Alberta Environmental Protection, Edmonton, Alberta.

Bliss LC, Wein RW (1971) Plant community response to disturbances in the western Canadian Arctic. Can J Bot 50:1097-1109

Caners RT and Lieffers VJ (2014) Divergent pathways of successional recovery for in-situ oil sands exploration drilling pads on wooded moderate-rich fens in Alberta, Canada. Restoration Ecology 22(5):657-667. 10.1111/rec.12123

Carmichael MJ, Bernhardt ES, Brauer SL, Smith WK (2014) The role of vegetation in methane flux to the atmosphere: should vegetation be included as a distinct category in the global methane budget? Biogeochemistry 119(1-3):1-24. 10.1007/s10533-014-9974-1

Chapin FS, Woodwell GM, Randerson JT et al (2006) Reconciling carbon-cycle concepts, terminology, and methods. Ecosystems 9:10411050. 10.1007/s10021-005-0105-7

Chimner RA, Pypker TG, Hribljan JA, Moore PA, Waddington JM (2016) Multi-decadal changes in water table levels alter peatland carbon cycling. Ecosystems 20(5):1042-1057. 10.1007/s10021-016-0092-x
Dabros A, Pyper M, Castilla G (2018) Seismic lines in the boreal and arctic ecosystems of North America: environmental impacts, challenges, and opportunities. Environ Rev 26:214-229. 10.1139/er-2017-0080

Davidson SJ, Goud EM, Malhotra A et al. (2021) Linear disturbances shift boreal peatlands plant communities toward earlier peak greenness. 10.1002/essoar/10506838.1

Echiverri LFI, Macdonald SE, Nielsen SE (2020) Disturbing to restore? Effects of mounding on understory communities on seismic lines in treed peatlands. Can. J. For. Res. 50:1340-1351. 10.1139/cjfr-2020-0092

Environment and Climate Change Canada (ECCC) (2021) Canadian Climate Normals 1981-2010 Station Data: Wabasca RS. https://www.climate.weather.gc.ca/climate_normals. Accessed 08 July 2021

Fechner-Levy EJ, Hemond HF (1996) Trapped methane volume and potential effects on methane ebullition in a northern peatland. Limnology and Oceanography 41(7):1375-1383. 10.4319/lo.1996.41.7.1375

Filigetti AT, Cody M, Nielsen SE (2019) Caribou conservation: restoring trees on seismic lines in Alberta, Canada. Forests 10:185-195. 10.3390/f10020185

Granath G, Strengbom J, Rydin H (2010) Rapid ecosystem shifts in peatlands: linking plant physiology and succession. Ecology 91(10):3047-3056. 10.1890/09-2267.1

Griffis T, Rouse WR, Waddington JM (2000) Scaling net ecosystem CO$_2$ exchange from the community to landscape-level at a subarctic fen. Global Change Biology 6:459-473. 10.1046/j.1365-2486.2000.00330.x

Golder Associates (2015) Boreal Caribou Habitat Restoration Operational Toolkit for British Columbia. Report No. 1313720037. BC Oil and Gas Commission. https://www.bcogc.ca/files/application-manuals/ABA-Guidance/2015golderrestoration-toolkit.pdf
Goodrich JP, Varner RK, Frolking S, Duncan BN, Crill PM (2011) High-frequency measurements of methane ebullition over a growing season at a temperate peatland site. Geophysical Research Letters 38. 10.1029/2011GL046915

Lai DYF (2009) Methane dynamics in Northern peatlands: a review. Pedosphere 19(4):409-421.

Lee P, Boutin S (2006) Persistence and developmental transition of wide seismic lines in the western Boreal Plains of Canada. Journal of Environmental Management 78:240-250. 10.1016/j.jenvman.2005.03.016

Lenth RV (2016) Least-Squares Means: The R Package lsmeans. Journal of Statistical Software, 69(1), 1-33. 10.18637/jss.v069.i01

Lieffers V, Caners R, Ge H (2017) Re-establishment of hummock topography promotes tree regeneration on highly disturbed moderate-rich fens. Journal of Environmental Management 197:258-264. 10.1016/j.jenvman.2017.04.002

Lieffers VJ, Caners RT, Ge H (2017) Re-establishment of hummock topography promotes tree regeneration on highly disturbed moderate-rich fens. Journal of Environmental Management 197:258-264. 10.1016/j.jenvman.2017.04.002

Loisel J, Yu Z, Beilman DW, Camill P, Alm J, Amesbury MJ et al (2014) A database and synthesis of northern peatland soil properties and Holocene carbon and nitrogen accumulation. The Holocene 24:1028–1042.

Lovitt J, Rahman MM, Saraswati S, McDermid GJ, Strack M, Xu B (2018) UAV remote sensing can reveal the effects of low-impact seismic lines on surface morphology, hydrology, and methane (CH4) release in a boreal treed bog. Journal of Geophysical Research: Biogeosciences 123:1117–1129. https://doi.org/10.1002/2017JG004232
Lund M, Lafleur PM, Roulet NT, Lindroth A et al (2010). Variability in exchange of CO2 across 12 northern peatland and tundra sites. Global Change Biology 16(9):2436-2448. 10.1111/j.1365-2486.2009.02104.x

Murray KR, Bird M, Strack M, Cody M and Xu B (2021) Restoration approach influences carbon exchange at in-situ oil sands exploration sites in east-central Alberta. Wetlands Ecol Manage. Doi: 10.1007/s11273-021-09784-x

Nugent KA, Strachan IB, Strack M, Roulet NT, Rochefort L (2018) Multi-year net ecosystem carbon balance of a restored peatland reveals a return to carbon sink. Global Change Biology 24:5751-5768. 10.1111/gcb.14449

Pendall E, Bridgham S, Hanson PJ, Hungate B, Kicklighter DW, Johnson DW, Law BE, Luo YQ, Megonigal JP, Olsrud M, Ryan MG, Wan SQ (2004) Below-ground process responses to elevated CO2 and temperature: A discussion of observations, measurement methods, and models. New Phytol 162:311-322.

Pinheiro J, Bates D, DebRoy S, Sarkar D (2016) Linear and nonlinear mixed effects models, R package version 3.1–128. http://CRAN.R-project.org/package=nlme

Pouliet R, Rochefort L, Karofeld E, Mercier C (2011) Initiation of Sphagnum bryophyte hummocks in bogs and the presence of vascular plants: Is there a link? Acta Oecologica 37:346-354. 10.106/j.act.ao.2011.04.001

Pypker TG, Moore PA, Waddington JM, Hribljan JA, Chimner RC (2013) Shifting environmental controls on CH4 fluxes in a sub-boreal peatland. Biogeosciences 10:7971-7981. 10.5194/bg-10-7971-2013

R Core Team (2013) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. Online at: http://www.R-project.org/
Robroek BJM, Jassey VEJ, Kox MAR et al (2015) Peatland Vascular plant functional types affect methane dynamics by altering microbial community structure. Journal of Ecology 103(4):925-934. 10.1111/1365-2745.12413

Ryan MG, Law B (2005) Interpreting, measuring, and modeling soil respiration. Biogeochemistry 73:3-27. 10.1007/s10533-00405167-7

Schneider RR (2002) Alternative futures: Alberta’s boreal forest at the crossroads. Federation of Alberta Naturalists. Edmonton, AB, Canada.

Shuttleworth E, Evans MG, Hutchinson SM, Rothwell JJ (2015) Peatland restoration: controls on sediment production and reductions in carbon and pollutant export. Earth Surf. Process. Landforms 40:459-472. 10.1002/esp.3645

Smolander A and Heiskanen J (2007) Soil N and C transformations in two forest clear-cuts during three years after mounding and inverting. Canadian Journal of Soil Science 87(3):251-258. 10.4141/S06-028

Stevenson CJ, Filicetti AT, Nielsen SE (2019) High precision altimeter demonstrates simplification and depression of microtopography on seismic lines in treed peatlands. Forests 10:295-3017. 10.3390/f10040295

Strack M, Cagampan G, Hassanpour Fard G, Keith AM, Nugent K, Rankin T, Robinson C, Strachan IB, Waddington JM, Xu B (2016) Controls on plot-scale growing season CO₂ and CH₄ fluxes in restored peatlands: Do they differ from unrestored and natural sites? Mires and Peat 17(5):1-18. 10.19189/MaP.2015.OMB.216

Strack M, Softa D, Bird M, Xu B (2018) Impact of winter roads on boreal peatland carbon exchange. Glob Change Biol 24:201-212. 10.1111/gcb.13844
Strack M, Hayne S, Lovitt J, McDermid GJ, Rahman MM, Saraswati S, Xu B (2019) Petroleum exploration increases methane emissions from northern peatlands. Nature Communications 10:2804-2811. 10.1038/s41467-019-10762-4

Van Renson CK, Nielsen SE, White B, Vinge T, Lieffers VJ (2015) Natural regeneration of forest vegetation on legacy seismic lines in boreal habitats in Alberta’s oil sands region. Biol Conserv 184:127-135.

Walker TN, Garnett MH, Ward SE, Oakley S, Bardgett RD, Ostle NJ (2016) Vascular plants promote ancient peatland carbon loss with climate warming. Glob Change Biol 22:1880–1889