Faculty of Science and Engineering

School of Geography, Earth and Environmental Sciences

2022-04

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

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

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- 1 CO₂ uptake decreased and CH₄ emissions increased in first two years of peatland seismic line
- 2 restoration
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18 Abstract:

- 19 Oil and gas exploration has resulted in over 300,000 km of linear disturbances known as seismic lines,
- 20 throughout boreal peatlands across Canada. Sites are left with altered hydrologic and topographic
- 21 conditions that prevent tree re-establishment. Restoration efforts have concentrated on tree recovery
- 22 through mechanical mounding to re-create microtopography and support planted tree seedlings to
- 23 block sightlines and deter the use of lines by wolves, but little is known about the impact of seismic line
- 24 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
 CO₂ 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 $6 21 \text{ g } \text{CO}_2 \text{ m}^{-2} \text{ d}^{-1}$, 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
- 32 other sites with varied conditions, will help determine if these restoration practices are effective at
- 33 restoring carbon cycling.
- 34 Keywords: Peatland restoration, seismic lines, mounding, carbon dioxide, methane
- 35 **Acknowledgements:** The authors would like to thank Bin Xu for leading restoration planning and
- 36 groundwork on the site and Felix Nwaishi for input on study design. We would also like to thank Taylor
- 37 Vodopija and Miranda Hunter for their help in the field and Blake Haskell for laboratory help.
- 38 This study took place on the unceded territories of the peoples of the Treaty 6 region and Métis Nation
- 39 of Alberta, Region 4, and the authors live and work on the unceded traditional territory of the
- 40 Attawandaron (Neutral), Anishinaabeg and Haudenosaunee peoples.
- 41 **Declarations:**

- 42 **Funding:** This work was supported by the Government of Canada Environmental Damages Fund, project
- 43 number EDF-AB-2018c009. Research support was also provided via a Canada Research Chair to Maria
- 44 Strack.
- 45 **Conflicts of interest/competing interests:** The authors declare no conflicts of interest.
- 46 **Availability of data and material:** The datasets generated during and/or analysed during the current
- 47 study are available at https://doi.org/10.5683/SP3/CHCPWR, Scholars Portal Dataverse
- 48 **Code availability:** Not applicable
- 49 **Authors' contributions**: All authors contributed to the study conception and design. Material
- 50 preparation, data collection and analysis were performed by Megan Schmidt. The first draft of the
- 51 manuscript was written by Megan Schmidt and all authors commented on previous versions of the
- 52 manuscript. All authors read and approved the final manuscript.

53 1. Introduction

54 The northern half of the province of Alberta, Canada is known for its vast underground oil and gas 55 reserves and is covered by over 134,000 km² of boreal peatlands (AEP 2018). Extraction and exploration 56 of oil and gas deposits has left a network of linear clearings, known as seismic lines, crisscrossing the 57 boreal region even decades after their creation (Lee and Boutin 2006). Recent estimates put the total 58 length of seismic lines through Alberta peatlands at over 345,000 km or 1900 km² (Strack et al. 2019), 59 reaching mean density of up to 10 km/km² (Lee and Boutin 2006) or as high as 40 km/km² (Schneider 60 2002). Historically, seismic lines were created using heavy machinery during summer months, removing 61 trees and vegetation, as well as roots and surface soil layers, to a width of 5 - 10 m (Bliss and Wein 62 1972); these are often referred to as "legacy lines". Over time clearing began to take place over winter 63 months and bulldozer blades were raised to reduce disturbance of the peat (Bliss and Wein 1972). 64 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 65 (Dabros, Pyper, and Castilla 2018). Undisturbed, peatlands sequester large amounts of carbon (C) (e.g., 66 67 Loisel et al. 2014) and provide important habitat for species such as the threatened woodland caribou 68 (Rangifer tarandus caribou) (Filicetti, Cody, and Nielsen 2019). Alteration of hydrologic and topographic 69 conditions on seismic lines prevents the re-establishment of trees, contributing to habitat fragmentation 70 and increased predation of caribou, which has spurred efforts to restore them. The focus of restoration 71 has, until recently, been on tree recovery, but little is known about the effects of linear disturbances and 72 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

76 properties and C cycling. Not only are hummock tops physically removed in preparation for seismic 77 surveying, but repeated passes of heavy machinery further compress the peat surface (Stevenson et al. 78 2019; Strack et al. 2019), resulting in rutting from machinery and an increase in hollow coverage (Lovitt 79 et al. 2018; Stevenson et al. 2019). Any microforms that do persist on seismic lines tend to be highly 80 suppressed, varying only a few centimeters from mean elevation, compared to up to a meter in natural 81 conditions (e.g., Caners and Lieffers 2014). Microform development in disturbed peatlands has been 82 shown to be resistant to natural formation processes such as Sphagnum growth and fire (van Rensen et 83 al. 2015; Stevenson et al. 2019), likely due to shallow water tables and increasing surface water pooling 84 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 85 86 communities (Lee and Boutin 2006, van Rensen et al. 2015; Strack et al. 2018). Moss cover may be lower 87 on lines and exhibit slower growth overall, potentially due to increased light levels (Pouliot et al. 2011), 88 while hummock forming Sphagnum species must compete with other moss species, resulting in slowed 89 succession that often stalls in early stages (Caners and Lieffers 2014; van Rensen et al. 2015). Loss of 90 hummocks removes oxic zones that support higher rates of gross ecosystem productivity (GEP) and 91 methane (CH₄) oxidation, and in turn creates conditions that favour CH₄ production (Chimner et al. 92 2016; Strack et al. 2018). The net C uptake of the new plant community will determine the C balance on 93 the line; how it compares to the adjacent forested peatland will depend on the ability of a more 94 productive understory to compensate for the loss of C uptake by trees and increased CH_4 emissions. 95 Restoration of disturbed peatlands has the potential to return them to C sinks, as has been seen in 96 peatlands used for horticultural peat extraction (e.g., Strack et al. 2016; Nugent et al. 2018). Until 97 recently, restoration of seismic lines has been largely ignored due to the assumption that such linear 98 disturbances would recover naturally with time. What has been done has focused mainly on structural 99 restoration related to caribou habitat, such as reducing sightlines and access by predators (e.g., Filicetti

100 et al. 2019); full mitigation of the impacts of seismic lines requires a shift in this focus to include 101 restoration of ecological functions for a return to successional peatland pathways. As previous 102 restoration has been targeted at tree recovery, the focus has been on creating artificial mounds to 103 replace microtopography, namely hummocks, lost during construction, and the use of these elevated 104 microsites for tree establishment (Lieffers et al. 2017; Filicetti et al. 2019). Mounding is a mechanical 105 process that scoops soil from one spot and places it nearby to create raised areas (hummocks) and leave 106 low areas (hollows) that mimic natural microforms (Dabros et al. 2018). Studies on mounding have been 107 ongoing for some time to restore peatland oil sands exploration well sites, and mounds on these sites 108 are often large, high, and inverted to expose deeper peat and/or mineral soil (Echiverri et al. 2020; 109 Murray et al. 2021). Not only does inversion eliminate any potential uptake of CO₂ by buried vegetation, 110 but peat surfaces are also exposed to erosion (Shuttleworth et al. 2014) and increased decomposition 111 (Smolander and Heiskanen 2007; Lieffers et al. 2017). Meanwhile, large, flooded hollows have the 112 potential to become CH₄ emission hotspots (Murray et al. 2021). While mounding on seismic lines has 113 become more common (Echiverri et al. 2020; Murray et al. 2021), knowledge of impacts on GHG 114 production and emissions remains limited; this paper is the first to capture detailed, plot-scale CO₂ and 115 CH₄ fluxes on unrestored and restored seismic lines.

116 In this study, CO₂ and CH₄ fluxes were measured in the first two summers immediately following 117 restoration work. The objectives of this study were to: (1) quantify CO₂ and CH₄ fluxes of two types of 118 mounding - inline mounding (IM), in which mounds and hollows were created on the seismic line and 119 not inverted, and hummock transfer (HT), in which established natural hummocks just off the edges of 120 the line were targeted and placed vegetated side up on the line, leaving the associated hollow in the 121 bordering undisturbed peatland and these treatments were compared to untreated lines and adjacent 122 undisturbed peatland; (2) assess the environmental factors contributing to variation in CO_2 and CH_4 123 fluxes on and off the lines.

124 2. Methods

125 **2.1 Study site**

126 The study area is located in central Alberta, Canada approximately 11 km southwest of Brazeau Dam 127 (52.889326, -115.549173; Figure 3.1). The region is classified as Boreal Plains Ecozone (Environment 128 Canada 2001) and Central Mixedwood Natural Subregion (AEP 2006). Average temperatures range from 129 14 – 17 °C in June, July, and August to -15 – -12 °C in December, January, and February with an average 130 of 97 frost free days per year (ECCC 2021). The area receives an average 462 mm precipitation per year, 131 with most occurring during the growing season, May to August (ECCC 2021). Wetlands comprise a large 132 portion of the region, dominated by wooded and shrubby fens, in a mosaic of aspen and spruce upland 133 (AEP 2006). The study site consists of ~3 km of peatland seismic line running north-south and east-west 134 (Figure 1). Although the precise date of creation is unknown, satellite images show seismic lines present 135 as far back as 1982. At time of groundwork in March 2019 the line edges were still clearly visible, with 136 little regeneration of woody vegetation or trees. A hydrological gradient exists longitudinally, becoming 137 drier as the center point is approached from both north and south ends. This corresponds to a gradient 138 in vegetation and fen type, transitioning from rich fen at both north and south ends to poor fen in the 139 center that extends the length of the east-west line. Dominant vegetation in the rich fen consists of Larix 140 laricina (Du Roi) K. Koch, Picea mariana (Mill.) Bitton, Sterns & Poggenburg, Betula spp., Carex spp., 141 Menyanthes trifoliata L., and Polytrichum spp. Dominant vegetation in the poor fen consists of P. 142 mariana, L. laricina, Rhododendron groenlandicum (Oeder) K.A. Kron & W.S. Judd, Salix spp., M. 143 trifoliata, Vaccinium oxycoccos L., Sphagnum fuscum (Schimp.) H. Klinger., and Sphagnum magellanicum 144 Brid. To capture these differences the site was divided into three subsites: north from the center point, 145 south from the center point, and the entire east-west section. For this study only the north and central 146 sections were chosen to create an equal number of sampling points in the poor and rich fen sections.



147

Inline Mounding

Natural

Figure1 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).

151 2.2 Groundwork and project design

- 152 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.

163 **2.3 Carbon dioxide (CO₂) flux**

164 We measured CO₂ fluxes approximately once per week from June to August 2019 and July and August 165 2020 via the closed chamber method (Griffis et al. 2000), in which acrylic chambers are placed onto 166 metal collars installed in the peat. Pairs of 60×60 cm steel collars were permanently installed at each 167 sample plot in corresponding hummocks and hollows. Two pairs were installed in each treatment in 168 both sections, resulting in four replicate plots per microform per treatment. A clear acrylic chamber 169 measuring $60 \times 60 \times 30$ cm was set into a groove along the top of the collars that created a seal when 170 filled with water. Air in the chamber was continuously circulated with a small battery-operated fan. CO_2 171 concentration in the chamber was measured at 15 second intervals for 105 - 120 seconds (~ 2 minutes) 172 using a portable infrared gas analyzer (EGM-4, PP Systems, Massachusetts, USA), along with air 173 temperature, relative humidity, and photosynthetically active radiation (PAR) within the chamber. An 174 opaque tarp was used to create fully dark conditions, enabling ecosystem respiration (ER) to be 175 measured. Order of sampling plots was changed daily to account for different light levels and solar 176 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 R² > 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.

185 **2.4 Methane (CH₄) flux**

186 We measured CH_4 fluxes from the same paired collars used for CO_2 with an opaque chamber and fans to 187 maintain circulation and reduce chamber heating from June to August 2019 and July to August 2020. In 188 2019 we collected CH₄ by extracting 20 ml gas samples from the chamber via syringe at 5-, 10-, 15-, and 189 25-minutes post-closure that were injected into pre-evacuated Exetainers (Labco Ltd.) and analyzed via 190 gas chromatography on a flame ionization detector (Shimadzu GC2014, Mandel Scientific) at the 191 University of Waterloo. Atmospheric samples were collected via syringe and exetainer two to three 192 times throughout each sampling day to provide CH₄ concentration at time zero. Fluxes were inspected 193 for linearity and outlying points associated with potential ebullition were removed to control for R²> 194 0.75, resulting in a 6% loss of fluxes.

195 We changed methods in 2020 to continuous measurement every second for 300 seconds (5 minutes) 196 with a CH₄/CO₂/H₂O Trace Gas Analyzer (LI-7810, LI-COR, Nebraska, USA). Per-second measurements 197 allowed for ebullition (bubble) events to be parsed from diffusive fluxes; however, to capture total CH₄ 198 contribution more accurately from the site, ebullitive fluxes were included in the data presented here. 199 Fluxes were inspected for linearity following the same rules as CO₂ and ebullition events identified when 200 concentration change was >15ppb s⁻¹ for minimum 3 seconds. Diffusive fluxes were calculated from the 201 average of the linear changes before and after ebullition events (Goodrich et al. 2011); when more than 202 one ebullitive event was present during the chamber closure, the linear slope between all ebullition 203 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.

208 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-

210 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

214 fluxes from either method as we did not directly compare between years.

215 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.

221 Vegetation surveys were conducted once in July of each summer. Each collar was assessed visually for

222 percent cover of functional groups: Sphagnum spp., all other mosses (e.g., brown and feathermoss),

223 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.

228 2.6 Data analysis

229 All data analysis was done in the statistical analysis program R (R Core Team 2013). As the main focus of 230 this study was the impact of treatments on CO_2 and CH_4 exchange, we did not investigate the effect of 231 fen type but rather treatment, microform, and their interaction as fixed effects in a separate linear 232 mixed effects model for each flux component in the package *nlme* (Pinheiro et al. 2014). To account for 233 repeated measures, collar was included as the random factor in each model. We used these models to 234 assess treatment and microform impacts on GEP, ER, NEE, and CH₄ in each of 2019 and 2020, separately. 235 The years were separated a priori due to differences in sample size and methods for CH4 measurement 236 between the years. Differences were considered statistically significantly when p < 0.05 using the anova 237 output command for each model. When a significant effect was present, post-hoc Tukey tests via the 238 *Ismeans* package (Lenth 2016) were performed for pairwise comparisons. CH₄ data was log transformed 239 to improve normality of the residuals and a value of 1.6 added prior to transformation to adjust for 240 negative values.

241 To understand how shifts in environmental conditions between treatments influence productivity, we 242 created additional linear mixed effects models using combined 2019 and 2020 data with either water 243 table or soil temperature as fixed effects along with treatment and interaction of water table or soil 244 temperature with treatment to evaluate whether response to environmental variables differed between 245 treatments; plot was included as a random effect. We also investigated whether vegetation cover 246 explained variation in C flux components. As vegetation was measured only once in July of each study 247 season, C flux components were expressed as a seasonal average for each plot. Linear regressions, using 248 the *Im* function were used to evaluate the effect of vascular or moss cover on variation in C fluxes.

249 **3. Results**

250 **3.1 Environmental conditions**

251 In general, average water table in the rich fen section was higher than the poor fen section at 0 cm and -252 10 cm, respectively. Although the difference was significant ($F_{1,44}$ =62.2, p<0.0001), we were mainly 253 interested in the overall treatment effects and how they affected ecosystem function across the range 254 of hydrological variation across the entire fen and therefore focused on treatment and microform 255 effects (Table 1). Across the fen, microform alone significantly impacted water table (2019: F_{1.20}=47.7, 256 p<0.0001; 2020: $F_{1,24}=20.6$, p<0.0001); treatment and the interaction with treatment were not 257 significant. Water table followed the surface elevation gradient of microforms from hummocks to 258 hollows, with the largest difference between HT hummocks (highest surface, deepest water table) and 259 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; 260 261 2020: $F_{1,24}$ =25.7, p<0.0001). Soils were warmest in hummocks and coolest in hollows, but differences 262 were minimal aside from HT and IM, where hummocks were on average 3 °C warmer than hollows in 263 both years (Table 1).

265 Table 1 Mean (SE) environmental conditions and vascular/moss cover for each treatment across both

266 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).

	Soil temp 2	10 cm (°C)	Water ta	ter table (cm) Moss cover (%)		Vascular cover (%)		
Treatment	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)

269	Moss cover was higher in natural and untreated sections than IM and HT, and both treatment (2019:
270	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:
271	F _{1,24} =151.9, p<0.0001)) were significant factors. Moss cover on natural and untreated hummocks ranged
272	from 88 – 100 % and remained at 89 – 100 % in HT and IM (Table 1). In hollows, moss cover dropped
273	from 32 – 93 % in natural and untreated to 1 – 4 % in HT and IM resulting in a significant treatment-
274	microform interaction (2019: F _{3,20} =13.4, p<0.0001; 2020: F _{3,24} =21.1, p=<0.0001). Similarly, vascular plant
275	cover was higher in natural and untreated areas (Table 3.1); treatment was significant in 2019
276	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).
277	Conversely, microform was not significant in 2019 but was in 2020 (2019 F _{1,20} =1.4, p=0.2388; 2020
278	$F_{1,24}$ =5.0, p=0.0347) with higher cover on hummocks. The difference in vascular plant cover between
279	microforms varied with treatment, with greater differences in HT and IM, but the interaction of
280	treatment and microform was significant in 2019 only (2019: $F_{3,20}$ =4.1, p=0.0197; 2020: $F_{3,24}$ =0.5,
281	p=0.6535). Vascular cover dropped from 51 $-$ 80 % on natural and untreated hummocks to 35 $-$ 64 % on
282	HT and IM hummocks. In hollows vascular cover dropped from 43 – 89 % in natural and untreated to 15
283	– 48 % in HT and IM.
284	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,
285	p=<.0001; 2020 F _{1,24} =67.2, p=<.0001) both had significant effects on moss cover in both years. Overall,
286	moss cover decreased with shallower water tables (i.e., water closer to, or above, surface) and
287	increased with warmer soil temperatures. These relationships were steepest in HT, followed by IM,
288	natural, and untreated (WT:treatment 2019: F _{3,20} =1.5, p=0.2356; 2020: F _{3,24} =5.2, p=0.0065; soil
289	temperature:treatment 2019: F _{3,20} =3.3, p=0.0381; 2020: F _{3,24} =3.4, p=0.0336). Treatment alone was
290	significant (F _{3,20} =6.1, p=0.0041) for explaining variation in vascular plant cover in 2019. In 2020, water
291	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 andincreased with warmer soils.

294 **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;

298 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)

299 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
- 302 d⁻¹ in untreated. Respiration rates in hollows were roughly half that of hummocks except in untreated,
- 303 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 CO_2 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,
- 307 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 CO₂ and CH₄ flux.

	2019			2020				
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: Microform	3,22	3.3	0.0362	Treatment: Microform	3,24	4.8	0.0088
ER								
	Treatment	3,22	1.5	0.2334	Treatment	3,24	1.5	0.2254
	Microform	1,22	5.8	0.0238	Microform	1,24	52.7	<.0001
	Treatment: Microform	3,22	1.3	0.293	Treatment: Microform	3,24	8.2	0.0006
NEE								
	Treatment	3,22	9.4	0.0003	Treatment	3,24	6.0	0.0033
	Microform	1,22	0.0	0.9535	Microform	1,24	0.3	0.5335
	Treatment: Microform	3,22	3.0	0.0505	Treatment: Microform	3,24	2.4	0.0844
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: Microform	3,22	1.6	0.2123	Treatment: Microform	3,270	1.9	0.1241

312	In 2020, 247 CO_2 fluxes were included in the dataset. Productivity was significantly lower in IM (-20.5 g
313	$CO_2 m^{-2} d^{-1}$) than untreated (-38.4 g $CO_2 m^{-2} d^{-1}$). Natural and HT were intermediate and did not
314	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
315	effect on GEP (Table 2), except for IM hollows, which were much less productive (i.e., less negative GEP)
316	than any other microform or treatment (Figure 2). Although treatments did not significantly differ in ER
317	(Table 2), hummocks generally respired more than hollows. Respiration from IM and HT hollows (6.0 g
318	$CO_2 m^{-2} d^{-1}$; 9.5 g $CO_2 m^{-2} d^{-1}$) was significantly lower than from natural and untreated hollows (13.6 g CO_2
319	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
320	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
321	(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 (-
322	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
323	than IM. Only IM hollows had significantly lower net uptake among all microform-treatment
324	combinations, following a similar pattern to 2019 (Figure 2).



325

Figure 2 Fluxes of carbon dioxide (CO₂; g CO₂ m⁻² d⁻¹) (NEE = net ecosystem exchange, ER = ecosystem 326 respiration, GEP = gross ecosystem productivity) and methane (CH₄; mg CH₄ m^{-2} d⁻¹) from restoration 327 328 treatments in 2019 (left) and 2020 (right) across both hummocks and hollows. Seasonal means are 329 significantly different if they do not share letters, as per post-hoc Tukey tests. Capital letters refer to the 330 significance of treatments overall; lower case letters refer to differences between microforms. Boxes 331 represent the upper and lower quartiles in which 50% of data points lie and whiskers represent 1.5 332 times the interquartile range. Median is shown by a horizontal line within the box. Statistical analyses 333 for CH₄ were performed on log transformed data but are shown here as raw flux concentrations.



334

335 Figure 3 Relationship between water table and CO₂ and CH₄ fluxes across both hummocks and hollows in 336 2019 and 2020 combined. Linear mixed effects models were used to determine relationships and post-337 hoc Tukey tests via the *Ismeans* package (Lenth 2016) were performed on anova results to determine 338 significant difference between treatments. Lines represent trends and letters represent significant 339 differences in slopes; treatments are significantly different if they do not share letters. 340 A total of 110 CH₄ fluxes passed quality control and were included for 2019. Fluxes ranged from 65.8 mg $CH_4 m^{-2} d^{-1}$ in natural to 775.7 mg $CH_4 m^{-2} d^{-1}$ in IM and were generally higher from hollows than 341 hummocks (Figure 2). However, CH₄ flux was not significantly different between treatments or 342 343 microforms and there was no significant treatment and microform interaction (Table 2). In the 2020 344 dataset, 306 CH₄ fluxes were included in the analysis. There was no significant effect from treatment 345 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. Ebullition was captured in 23 flux 346

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 CH₄ m⁻² d⁻¹ in untreated, 349.0 mg CH₄ m⁻² d⁻¹ in HT, and
 1752.2 mg CH₄ m⁻² d⁻¹ in IM, with no measured ebullition at natural plots.

350 Across the full 2019-20 dataset, higher productivity (i.e., more negative GEP) was significantly related to 351 deeper water tables both alone ($F_{1,293}$ =34.0, p<0.0001) and in interaction with treatment ($F_{3,293}$ =3.4, 352 p=0.0163). There were no significant differences in slope between HT and IM, which had the steepest 353 slopes and were significantly less productive than untreated and natural when water tables were above 354 -20 cm (Figure 3). Ecosystem respiration significantly decreased with shallower water tables (F_{1,293}=96.6, 355 p<0.0001) and although treatment did not have a significant effect on respiration, the interaction with 356 treatment did ($F_{3,293}$ =4.6, p=0.0033), resulting in similar, steeper slopes for HT and IM, followed by 357 untreated and natural (Figure 3). Overall, the effect of water table ($F_{1,293}$ =4.4, p=0.0356) on NEE was 358 significant, resulted in increasing net CO₂ uptake (i.e., more negative) in natural and untreated with 359 shallower water table, and decreasing net productivity in HT and IM. Slopes were very shallow and only 360 untreated and IM were significantly different (Figure 3). Water table position did not significantly predict 361 CH₄ 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). 362

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_{3,269}=1.8, p=0.1383). Higher soil temperatures also resulted in significantly higher CH₄ fluxes

369 (F_{1,349}=12.2, p=0.0005) with no significant treatment interaction.



370

371 Figure 4 Relationships between moss (left) and vascular (right) percent cover on net ecosystem 372 exchange (NEE) and CH₄. Linear mixed effects models were used to determine relationships and post-373 hoc Tukey tests via the *Ismeans* package (Lenth 2016) were performed for pairwise comparisons 374 between slopes when a significant interaction with treatment was present. 375 Both moss and vascular plant cover significantly explained variation in C fluxes (Figure 4), with no 376 significant interactions with treatment in any case. GEP significantly increased (i.e., became more 377 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, 378 p<0.0001) and vascular plant cover (F_{1,52}=19.4, p<0.0001). NEE also significantly increased with 379 380 increasing moss (F_{1,52}=15.4, p=0.0002) and vascular plant cover (F_{1,52}=21.1, p<0.0001). Across all

381	treatments, CH ₄ fluxes decreased with increasing moss cover (F _{1,52} =16.0, p=0.0001; Fig 4). Emission of
382	CH₄ generally decreased with increasing vascular cover although there were no significant effects.
383	4 Discussion
384	4.1 Environmental conditions
385	The creation of seismic lines through our study site altered water tables, vegetation composition, CO_2
386	release and uptake, and CH_4 emission. Though not significantly different, average water tables in
387	untreated sections were approximately 5 cm closer to the surface than in the undisturbed fen, leading
388	to wetter conditions overall (Table 1). This was as expected as previous studies have shown depression
389	in elevation and flattening of hummocks on lines (Stevenson et al. 2019) and concurrent rise in water
390	table (e.g., Lovitt et al., 2018). Additionally, the study area received more than the average total annual
391	precipitation from May 1 to August 31 in both years (551 mm and 510 mm for 2019 and 2020
392	respectively; ACIS 2020). These wet conditions were most noticeable in 2019, when water levels were
393	near or above surface much of the season. Both mounding treatments increased the difference in water
394	table between hummocks and hollows compared to untreated, as was intended. Surprisingly, there was
395	very little difference among treatments in soil temperature at 10 cm below surface in both years,
396	contrary to what was found by Strack et al. (2018) who reported warmer soils on seismic lines than the
397	adjacent forested peatlands. The wet conditions and surface water flow through the fen likely limited
398	the development of local temperature differences.
399	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

403 water table dropped from 6 cm above surface to 4 cm below. Contrary to previous findings (Strack et al.

404 2018; Davidson et al. 2021), vascular plant cover was not significantly higher on the lines than the

405 surrounding natural fen (Table 1), except for low spots on untreated lines in 2019, when they were 406 wetter than hummocks in both untreated and natural but drier than natural hollows. This may be a 407 result of differences in peatland types across studies, highlighting the need for further trials of these 408 techniques. Murray et al. (2021) also found an increase in vascular plant cover versus natural peatland 409 conditions at wet, unmounded control sites on oil sands exploration (OSE) well pads but not on drier 410 sites. Lower vascular plant cover can partially be attributed to the lack of woody vegetation, primarily 411 shrubs, on the line, although increased light availability typically leads to higher cover of herbaceous 412 species (Caners and Lieffers 2014; Strack et al. 2018; Murray et al. 2021). An increase on the line of 413 primarily graminoids (e.g., Carex spp. and Juncus spp.) that have small stem diameter and lack large, 414 spreading leaves (e.g., Menyanthes trifoliata, Caltha palustris), may explain some of the decrease in 415 cover, as may the difference in precipitation and water levels between the two years. As water levels 416 drop hummocks become drier, and the deepest water tables may not be accessible for some plants, 417 while hollows become less flooded, allowing for species that prefer drier conditions to flourish while still 418 sustaining water-loving species with shallow water tables. In natural areas, this maintains a steady 419 cover, though it fluctuates over time; on untreated lines, however, consistently shallow water tables 420 may prevent survival of species that prefer elevated microsites even in drier years.

421 Moss cover did not significantly change from natural and untreated areas on hummocks in either 422 mounding treatment, but both increased from 2019 to 2020 (Table 1). While vascular plant cover was 423 approximately half that of natural plots in 2019, it had recovered to similar levels as natural and 424 untreated by 2020. Both the decrease and recovery can be attributed to our restoration techniques; 425 disturbance during restoration work caused the loss of some plant cover, while not inverting the 426 mounds maintained much of the existing vegetation, roots, and seedbank, allowing for better growth 427 the following year. This contrasts with previous mounding techniques that invert the mounds and leave 428 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.

436 **4.2 Changes in carbon exchange on untreated seismic lines**

437 Carbon dioxide fluxes were not significantly different on the untreated lines compared to natural 438 reference plots in either year (Table 2), but there was generally greater productivity and net CO₂ uptake 439 on the line (Figure 2). Higher GEP (i.e., more negative) on the lines could be due to the lack of 440 microtopography and resulting water tables; following high precipitation flooding was shallower in 441 untreated sections than natural hollows, and during drier periods water table was not as far below 442 ground. The difference in water table depths between years on untreated lines was also much less than 443 in natural, leading to a steadier water supply for plants. Removal of trees and shrubs likely removes 444 some of the competition for water on the lines as well. Moreover, removal of tree and shrub cover 445 increases solar radiation reaching the peat surface on the seismic line and higher temperature and 446 longer growing seasons have been shown to increase GEP (Lund et al. 2010). Davidson et al. (2021) 447 found that peatland seismic lines greened up earlier and had higher GEP than natural reference sites. 448 Although ER was higher from untreated lines, there was no significant difference from the undisturbed 449 fen in either year, or between microforms. ER decreased as water levels rose and increased with 450 warmer temperature; an increase in soil temperature of as little as 1° C has been shown to increase 451 respiration (Walker et al. 2016). Warmer soils on the line spurs photosynthesis, the associated 452 autotrophic respiration from roots and mycorrhizae (Ryan and Law 2005) and heterotrophic respiration

453 of microbial and enzyme communities (Pendall et al. 2004). Rising water levels slow microbial growth 454 and activity, leading to a decrease in respiration, as in natural hollows in 2019 (Figure 3). ER rates in 455 hummocks also increase as CH_4 diffuses through the acrotelm and is converted to CO_2 by 456 methanotrophs (Robroek et al. 2015). Overall, this site is consistent with results from Strack et al. (2018) 457 and Davidson et al. (2021) who both found significantly higher productivity and NEE on seismic lines 458 with increased total vascular cover; however, as total vascular cover was not significantly different from 459 natural, change to species composition or functional groups may be a may be more important for 460 altering carbon cycling than changes to cover alone.

461 Although neither soil temperature nor distance of water table from surface were significantly different 462 in natural or untreated areas, warmer, wetter soils on the untreated lines favoured the production of 463 CH₄ as methanogenic microbial activity increased under anoxic conditions. In addition to reaching the 464 atmosphere via diffusion, some plants, including many sedge and rush species, including *Eriophorum* 465 *vaginatum* found on the site, mediate the movement of CH₄ directly from peat to atmosphere 466 (Carmichael et al. 2014), bypassing much of the potential oxidation in the peat profile. Ebullition events 467 have also been shown to increase with rising soil temperatures (Fechner-Levy and Hemond 1996) and 468 shallower water table depths (Lai 2009) but were only captured in 2/82 (2.4 %) fluxes of untreated lines 469 and never from natural areas. Shallower water tables on the untreated lines reduced the depth or 470 presence of an oxic layer, likely increasing methanogen activity and production of CH₄, while the lack of 471 woody roots and organic matter led to a change in peat composition or density that allows for increased 472 diffusion of CH₄ to the atmosphere. These results reflect those of Strack et al. (2018), who found GEP 473 and NEE improved, ER varied little, and CH₄ increased on a winter road versus the adjacent natural 474 wooded fen, although with more significant differences than on our site.

475 **4.3 Impacts of restoration treatments on CO₂ and CH₄ exchange**

476 There were few significant differences between IM and HT or between HT and natural and untreated 477 areas in either year, although GEP approximately doubled from 2019 to 2020 in both mounding 478 treatments (Figure 2), mirroring the recovery of vascular plant cover (Table 1). Low productivity on IM 479 hummocks was largely the result of disturbance during restoration and root systems that were adapted 480 to saturated conditions suddenly being moved above the ground surface, resulting in water level 481 differences of ~10 cm. Hummocks may not have yet fully integrated and connected with the underlying 482 surface, preventing water from rising up the peat column to the rooting zone. Removal of vegetation in 483 HT hollows was the leading factor in low GEP; however, because we targeted established natural 484 hummocks to transfer, resulting hollows were much shallower, with water levels and soil temperatures 485 in line with those of natural hollows. Many roots, small plants, and some mosses were left behind, and 486 both mosses and vascular species were able to move in from the edges. ER was not significantly 487 different in IM or HT than natural or untreated, and only differed significantly between microforms in 488 2020. Hollows in IM and HT had decreased vegetation cover and water levels at or above surface both 489 years, which favours CH₄ production (Chimner et al. 2016; Strack et al. 2018). ER from hummocks in IM 490 and HT, however, was higher than natural and untreated in both years (Figure 2), likely due to a thicker 491 oxic layer in the rooting zone (Pypker 2013) that resulted in increasing microbial activity and organic 492 matter decomposition. Overall, decreased NEE (i.e., less negative) in the two mounding treatments is 493 due to the loss and disturbance of vegetation and surface peat soils, shown by lower productivity and 494 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⁻²d⁻¹ in 2019 and 775 mg CH₄ m⁻²d⁻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.

502 4.4 Implications for management

503 When considering C fluxes measured in the study plots, these results include only CO₂ and CH₄ fluxes 504 from the hummocks and hollows created during the treatment and not the entire treatment area. By 505 assessing the density and area of hummocks and hollows and treatment area length and width, mean 506 treatment CO₂ and CH₄ fluxes can be estimated (Table 3). As the entire area is not mounded within a 507 treatment, we used the mean flux value from across all collars in untreated sections to represent the flat 508 (i.e., untreated) parts of HT and IM areas. This is not a prediction of annual or seasonal emissions, but 509 rather an estimate of peak season carbon exchange based on proportions of hummocks, hollows and 510 untreated areas within each treatment type. Total treatment uptake of CO₂ (NEE) increased in all four 511 areas from 2019 to 2020, and untreated had the highest net CO_2 uptake in both years at 20 and 21 g CO_2 512 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 513 treatments in 2020 than 2019 despite it being a drier year. Both years followed the same pattern, with 514 natural emitting the least CH₄ (64 and 139 mg CH₄ m⁻² d⁻¹), followed by untreated (135 and 234 mg CH₄ 515 $m^{-2}d^{-1}$), 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 516 517 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

- 524 moderately rich fen captured an additional 50 g C m⁻² y⁻¹ through total above and belowground
- 525 productivity. Future research should aim to determine seasonal to annual carbon balance of mounding
- 526 treatments including the contribution of trees to carbon uptake.
- 527 Table 3 Proportion of treatment areas covered by hummocks, hollows, and flat (i.e., untreated),
- estimated net ecosystem exchange^a (g CO₂ m⁻² d⁻¹), and estimated total methane flux (mg CH₄ m⁻² d⁻¹).
- 529 Proportions are based on our on-site measurements of the cover of hummocks, hollows and untreated
- 530 ground within each treatment, and line width and length. Estimated NEE and CH₄ fluxes were weighted
- by area and average flux cover type from each treatment from 2020 data.

	Proportion of treatment area (%)				
Treatment	Hummocks	Hollows	Flat	Estimated Total NEE ^a	Estimated Total CH₄
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

^aNet 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 allcollars in the untreated area.

535 These results indicate that mounding results in a decline in CO₂ uptake and increase in CH₄ emission, at

least in the first two years post-restoration, largely due to loss of productivity and high CH₄ in the

537 created hollows. However, given the low density of mounds applied in these treatments, changes across

538 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.,

540 Golder 2015; Murray et al. 2021). In comparison, our mounds were not inverted and were applied at a

541 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 CO₂ and CH₄, but also maintain
- 543 uptake of CO₂ from the plant community established on the lines, although the best density is as yet
- 544 uncertain and may vary between peatland types. Over time C uptake through the re-establishment and
- 545 growth of trees and woody species on the line may compensate for increased ER and CH₄, 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) 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.

551 The results from our study help to fill the knowledge gap on C exchange on peatland seismic lines and 552 the potential outcomes of restoration. As this is the first study of its kind, measurements will need to be 553 collected in the coming years to determine the long-term impacts on C emissions from the restoration 554 treatments and which method results in the best outcomes for tree establishment and growth as well as 555 ecosystem C and greenhouse gas exchange. Although untreated areas had the highest NEE (i.e., most 556 negative) both years, by the second year post-restoration the difference between untreated, natural, 557 and HT had declined, suggesting that with time treated areas may have the same or greater C sink 558 strength as untreated and natural areas. The treatments will also need to be applied and studied across 559 more peatlands of differing hydrologic conditions and vegetation types to develop best management 560 practices across a range of peatland conditions. This may help satisfy the goals of restoration from 561 multiple angles, including reduced time and cost for industry, restoration of habitat and connectiveness 562 for conservation, and reduction of GHG production. Ideally, these practices would put restored seismic 563 lines on pathways to improved function, ultimately helping to mitigate climate change while protecting 564 biodiversity.

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