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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
25 and compared carbon dioxide and methane fluxes to untreated lines and natural reference areas of a
26 wooded fen in the first two years post-restoration. We found no significant differences in net ecosystem
27 CO₂ exchange, but untreated seismic lines were slightly more productive than natural reference areas
28 and mounding treatments. Both restoration treatments increased ecosystem respiration, decreased net
29 productivity by 6 – 21 g CO₂ m⁻² d⁻¹, and created areas of increased methane emissions, including an
30 increase in the contribution of ebullition, of up to 2000 mg CH₄ m⁻² d⁻¹ over natural and untreated lines.
31 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

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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.

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46 **Availability of data and material:** The datasets generated during and/or analysed during the current
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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
65 (LIS) lines of 1.5 – 5.5 m, cleared with light-weight equipment, had become the preferred method
66 (Dabros, Pyper, and Castilla 2018). Undisturbed, peatlands sequester large amounts of carbon (C) (e.g.,
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.

73 The complexity of peatland systems makes them especially vulnerable to disturbance and prone to long,
74 slow recovery, if they recover at all. Vegetation removal and surface alteration during seismic line
75 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
85 reduces suitable sites for tree establishment and shifts seismic lines towards more *Carex* dominated
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 CH₄ 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

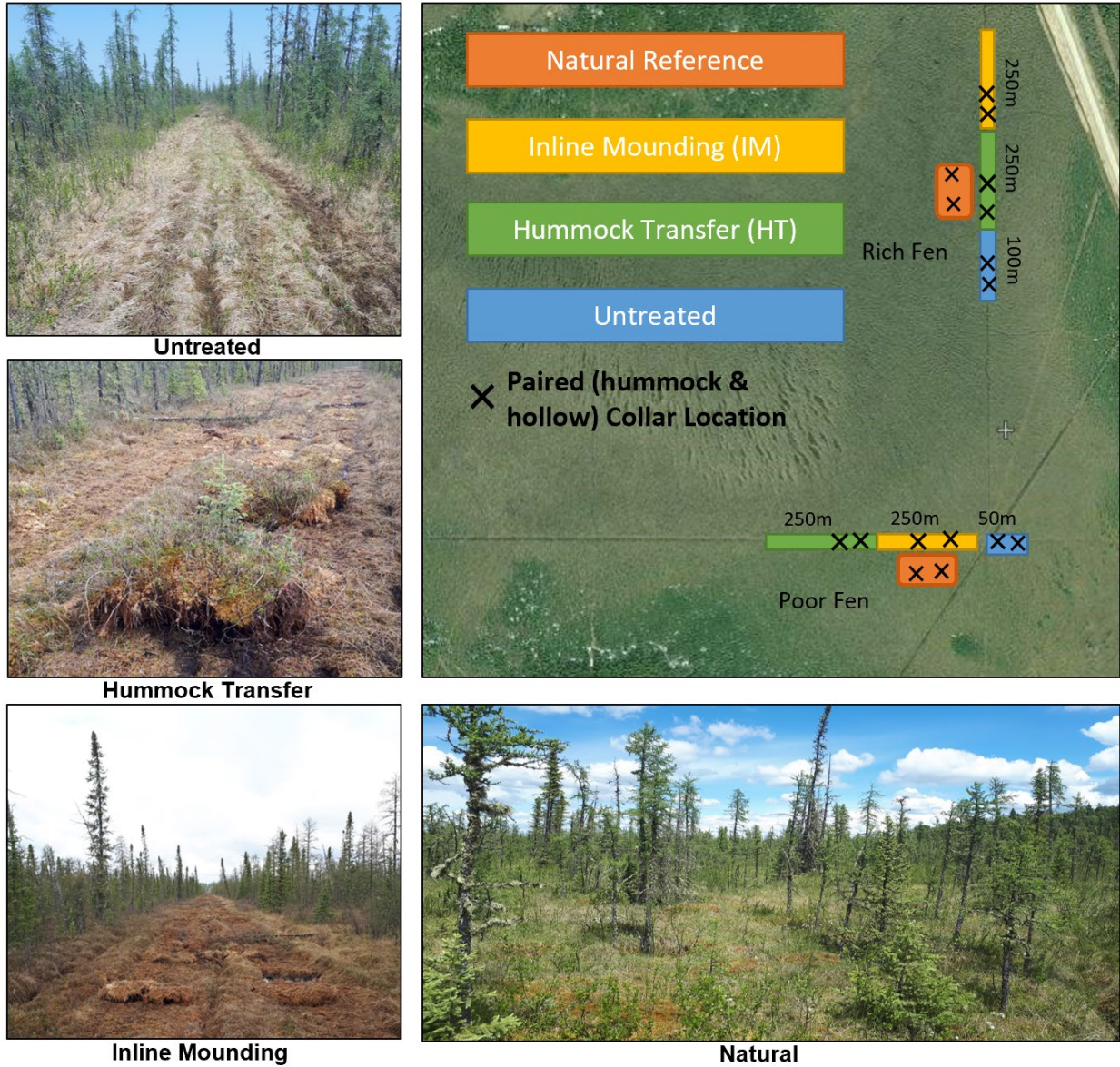
100 et al. 2019); full mitigation of the impacts of seismic lines requires a shift in this focus to include
101 restoration of ecological functions for a return to successional peatland pathways. As previous
102 restoration has been targeted at tree recovery, the focus has been on creating artificial 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 CO₂ and CH₄
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.) B.S.P., *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

148 Figure1 Study site with treatments, locations of collars, and photographs of the four treatments. Each
 149 treatment was applied in both the poor and rich fen, and two pairs of collars (hummock and hollow)
 150 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
 153 place in May 2019, and data collection began June 2019. Mounding was done by a backhoe with
 154 toothed digging bucket, moving an intact scoop of soil from within the seismic line and placing it nearby
 155 in the same orientation (i.e., vegetated side up) for IM, and targeting naturally formed hummocks just

156 off the line for HT. Hummocks with established woody shrubs and small trees were intentionally
157 targeted (Figure 1). Hummocks for both treatments averaged a height of 20 cm, while hollows were an
158 average depth of 10 cm in HT and 19 cm in IM. Single trees from the edges were randomly pulled down
159 onto the line (often referred to as stem-bending) to provide additional microsites and tree seedbank
160 inputs. Treatments were replicated on both poor fen and rich fen sections with some length of line left
161 untreated as controls, and a natural site was selected for each section approximately 20 m away from
162 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 × 60 × 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. CO₂
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.

177 Net ecosystem exchange (NEE) is the overall exchange and direction of C movement between the
178 atmosphere and an ecosystem, measured under full sun. Fluxes under dark condition capture ecosystem
179 respiration (ER), and gross ecosystem productivity (GEP) is calculated as the difference between NEE and

180 ER (Chapin et al. 2006). In this study, we use the sign conventions that C uptake from the atmosphere is
181 negative and emission to the atmosphere is positive (Ryan and Law 2005). Raw data were inspected for
182 linearity of fluxes, controlling for fit of $R^2 > 0.75$, except for fluxes that were relatively unchanging,
183 representing a flux close to zero. Processing resulted in a data loss of 43% (due to issues with collar
184 sealing in the newly formed hummocks) in 2019 and 0.5% in 2020.

185 **2.4 Methane (CH₄) flux**

186 We measured CH₄ fluxes from the same paired collars used for CO₂ 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 $R^2 >$
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 $>15\text{ppb s}^{-1}$ 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

204 additional CH₄ contribution of the ebullition events. The sum of the diffusive and ebullitive fluxes were
205 used as the total flux; diffusive flux slopes were controlled for $R^2 > 0.75$ and in cases with an ebullition
206 event(s) and no acceptable diffusive flux present, the ebullitive flux was used as the total flux. These
207 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
209 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
211 than the LI-COR but well within the range of fluxes from the research area (Supplementary table S3). In
212 addition to the small sample size, differences could be due to longer chamber closure times (15-20 mins
213 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**

216 Additional environmental factors were measured every time C fluxes were measured. Soil moisture was
217 measured with a ML3 ThetaProbe Soil Moisture Sensor (Delta-T Devices) at five points in the hummock
218 or hollow to achieve an average for the feature. A thermocouple soil temperature probe was used to
219 establish a temperature profile at -2, -5, -10, -15, -20, -25, and -30 cm. Water table was measured in a
220 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
224 relationships between C flux and individual vascular or moss plant functional groups so final analysis was
225 based on data combined into two groups: all mosses and all vascular species.

226 To understand how shifts in environmental conditions between treatments influence productivity, we
227 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 CO₂ and CH₄ 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 *lsmeans* 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 *lm* 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
260 variation in soil temperature at 10 cm below surface in both 2019 and 2020 (2019: $F_{1,20}=11.9$, $p=0.0025$;
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).

264

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 =
 267 4/microform/treatment) while soil temperature and water table were measured concurrent with fluxes
 268 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)

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

292 treatment. As with moss cover, vascular plant cover decreased with shallower water tables and
293 increased with warmer soils.

294 **3.2 Carbon exchange**

295 Average values for all C fluxes in each sampling year are given in the Supplementary Material (Table S1).
296 After processing, 84 CO₂ fluxes were included for 2019. Average productivity across the study plots was
297 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
300 significant in HT (Table 2, Figure 3.2). Ecosystem respiration did not differ significantly between any
301 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
304 positive (i.e., release to atmosphere) in IM and did not significantly differ from HT (IM 0.9 g CO₂ m⁻² d⁻¹;
305 HT -4.8 g CO₂ m⁻² d⁻¹); similarly, natural and untreated did not significantly differ (natural -14.9 g CO₂ m⁻²
306 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).

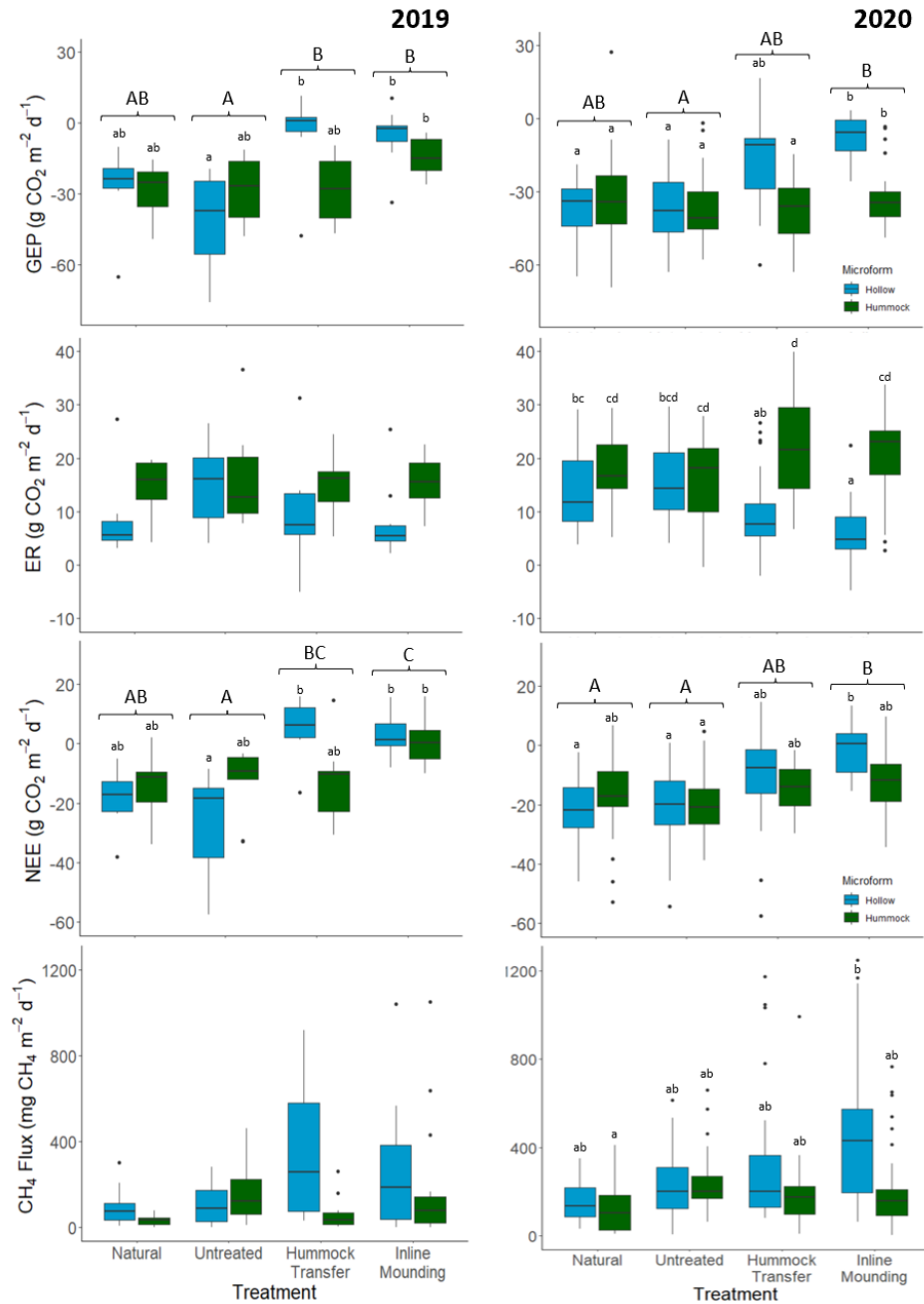
308

309 Table 2 Results from the linear mixed effects models describing effects of treatment, microform, and
 310 interactions on CO₂ and CH₄ flux.

Flux Component	2019				2020			
	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
CH₄								
	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

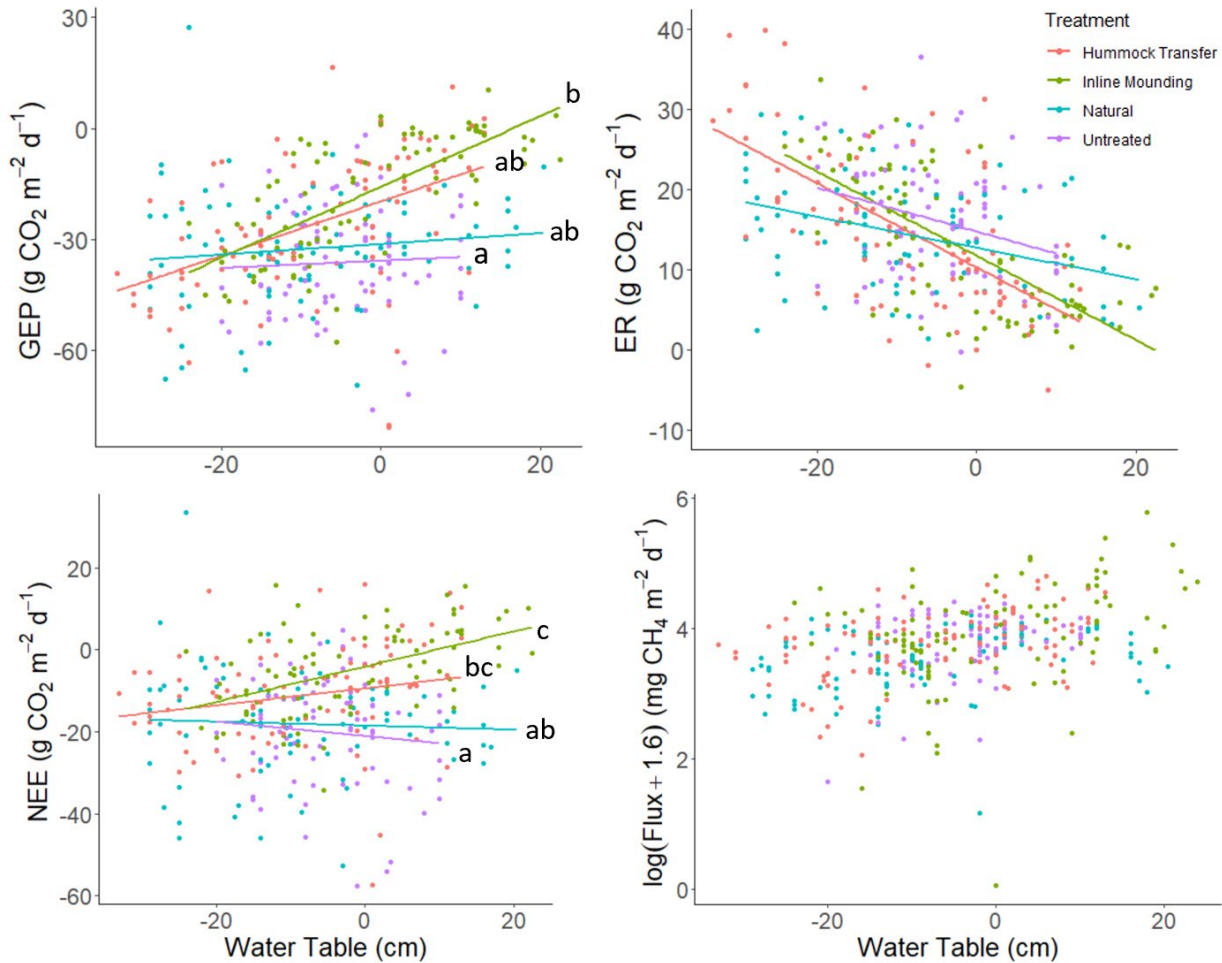
311

312 In 2020, 247 CO₂ fluxes were included in the dataset. Productivity was significantly lower in IM (-20.5 g
313 CO₂ m⁻² d⁻¹) than untreated (-38.4 g CO₂ m⁻² d⁻¹). Natural and HT were intermediate and did not
314 significantly differ at -34.9 g CO₂ m⁻² d⁻¹ and -28.9 g CO₂ 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 CO₂ m⁻² d⁻¹; 9.5 g CO₂ m⁻² d⁻¹) was significantly lower than from natural and untreated hollows (13.6 g CO₂
319 m⁻² d⁻¹; 15.4 g CO₂ m⁻² d⁻¹) and similarly higher from IM and HT hummocks (20.4 g CO₂ m⁻² d⁻¹; 22.2 g CO₂
320 m⁻² d⁻¹) than natural and untreated (17.8 g CO₂ m⁻² d⁻¹; 17.6 g CO₂ m⁻² d⁻¹). NEE was significantly lower
321 (i.e., less negative, less uptake) in IM (-7.4 g CO₂ m⁻² d⁻¹) than natural (-19.3 g CO₂ m⁻² d⁻¹) or untreated (-
322 21.0 g CO₂ m⁻² d⁻¹), while HT (-13.0 g CO₂ 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

326 Figure 2 Fluxes of carbon dioxide (CO_2 ; $\text{g CO}_2 \text{ m}^{-2} \text{ d}^{-1}$) (NEE = net ecosystem exchange, ER = ecosystem
 327 respiration, GEP = gross ecosystem productivity) and methane (CH_4 ; $\text{mg CH}_4 \text{ m}^{-2} \text{ d}^{-1}$) from restoration
 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 CH_4 were performed on log transformed data but are shown here as raw flux concentrations.



334

335 Figure 3 Relationship between water table and CO_2 and CH_4 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 *lsmeans* 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 CH_4 fluxes passed quality control and were included for 2019. Fluxes ranged from 65.8 mg

341 $\text{CH}_4 \text{ m}^{-2} \text{ d}^{-1}$ in natural to 775.7 mg $\text{CH}_4 \text{ m}^{-2} \text{ d}^{-1}$ in IM and were generally higher from hollows than

342 hummocks (Figure 2). However, CH_4 flux was not significantly different between treatments or

343 microforms and there was no significant treatment and microform interaction (Table 2). In the 2020

344 dataset, 306 CH_4 fluxes were included in the analysis. There was no significant effect from treatment

345 and across all microform-treatment combinations only natural hummocks ($122.2 \text{ mg CH}_4 \text{ m}^{-2} \text{ d}^{-1}$) and IM

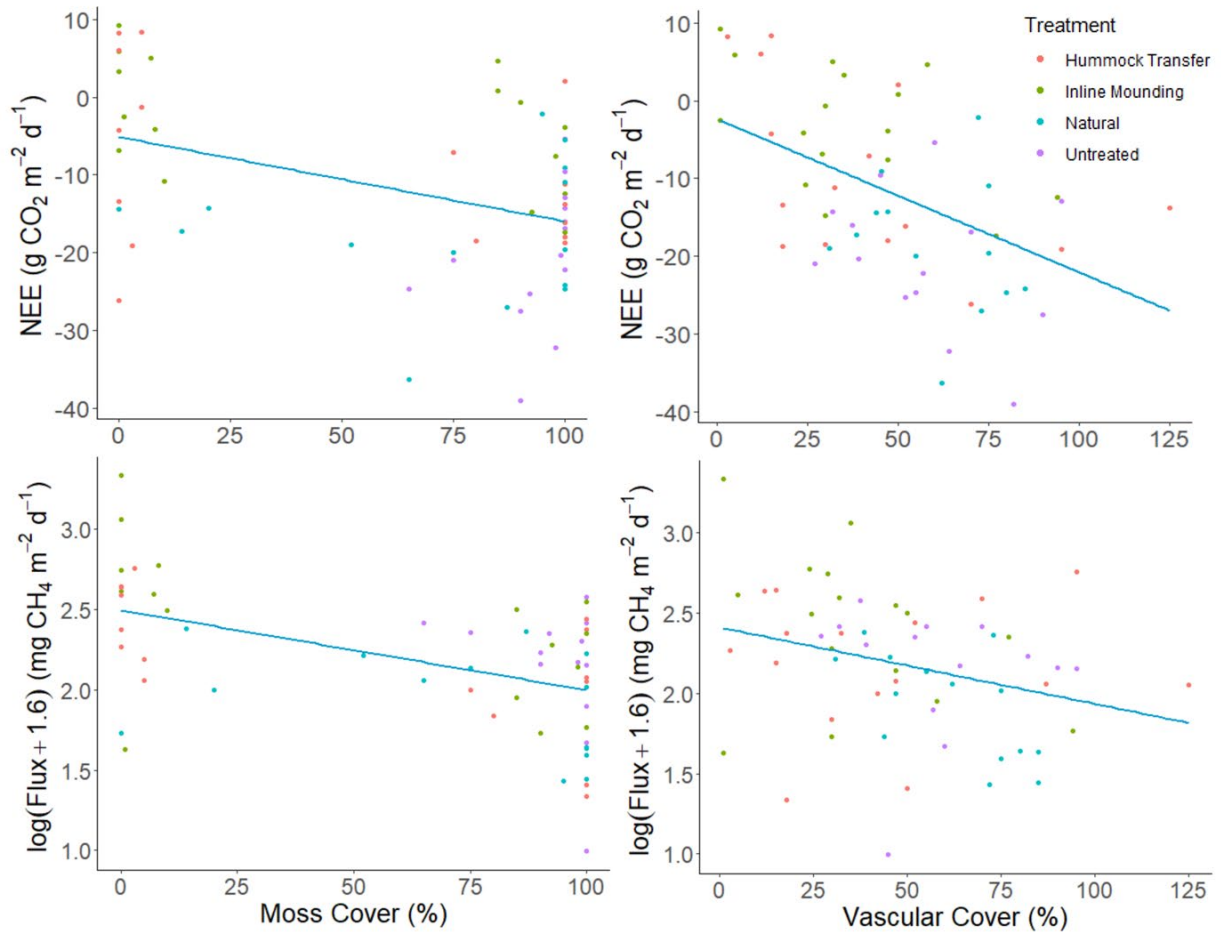
346 hollows ($1282.2 \text{ mg CH}_4 \text{ m}^{-2} \text{ d}^{-1}$) were significantly different. Ebullition was captured in 23 flux

347 measurements (Table S2): 2 in untreated, 1 HT hummocks, 5 HT hollows, 1 IM hummocks, and 14 IM
348 hollows. Average ebullitive flux was $337.9 \text{ mg CH}_4 \text{ m}^{-2} \text{ d}^{-1}$ in untreated, $349.0 \text{ mg CH}_4 \text{ m}^{-2} \text{ d}^{-1}$ in HT, and
349 $1752.2 \text{ mg CH}_4 \text{ m}^{-2} \text{ d}^{-1}$ 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 CO_2 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 CH_4 flux, but flux did generally increase with shallower water table ($F_{1,378}=2.8$, $p=0.0944$); there was also
362 no significant interaction between treatment and water table ($F_{3,378}=0.7$, $p=0.5258$).

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

365 Soil temperature alone ($F_{1,269}=36.5$, $p<0.0001$) also had a significant effect on ER, leading to higher
366 respiration with warmer soils. The overall impact of soil temperature on NEE was significant ($F_{1,269}=8.5$,
367 $p=0.0038$) with greater net uptake from warmer soils, and there was no significant interaction with
368 treatment ($F_{3,269}=1.8$, $p=0.1383$). Higher soil temperatures also resulted in significantly higher CH_4 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 *lsmeans* 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
 378 ($F_{1,52}=28.6$, $p<0.0001$). Respiration significantly increased with both increasing moss ($F_{1,52}=52.5$,
 379 $p<0.0001$) and vascular plant cover ($F_{1,52}=19.4$, $p<0.0001$). NEE also significantly increased with
 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, CH₄ 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, CO₂
386 release and uptake, and CH₄ 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
400 on the untreated lines than in natural. Moss cover had a significant negative relationship with increasing
401 water tables, and flooding has been shown to negatively impact some moss species (Granath et al.
402 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, unrounded 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

429 plant cover on inverted mounds five and six years following mounding. Moss cover was near zero in
430 hollows in both years, but approximately one third the vascular plant cover of natural was maintained in
431 IM hollows and two thirds in HT hollows in the first year. By 2020, vascular cover in HT hollows was
432 similar to natural and untreated areas; after five years vascular cover was still only 15 % in Murray et
433 al.'s (2021) hollows. The shallow depth of our hollows, particularly in HT where the natural hummock
434 accounted for most of the height of the removed peat, not only left shallower pools of water, but also
435 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 CH₄ diffuses through the acrotelm and is converted to CO₂ 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.

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

500 only 6/79 (7.6 %) HT fluxes contained ebullition events, possibly due to the shallower hollows formed in
501 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 CO₂ uptake in both years at 20 and 21 g CO₂
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
513 higher than HT and IM at 19, 18, and 17 g CO₂ m⁻² d⁻¹, respectively. CH₄ emission was also higher across
514 treatments in 2020 than 2019 despite it being a drier year. Both years followed the same pattern, with
515 natural emitting the least CH₄ (64 and 139 mg CH₄ m⁻² d⁻¹), followed by untreated (135 and 234 mg CH₄
516 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
517 et al.'s (2019) modeling of increased CH₄ emissions on seismic lines due to shallower water tables.

518 Notably, we did not look at the contribution of trees to C emissions and uptake, which could potentially
519 shift fluxes, although some tree and shrub roots inevitably contributed to ER of natural collars and of HT
520 hollows, which were located outside of the defined edges of the lines. Nor did we look at the potential
521 influences of tree seedlings planted across the site; however, planting avoided any features with collars,
522 so they did not interfere either. Taking tree and shrub cover into account, the natural peatlands actually
523 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),
 528 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
 531 by area and average flux cover type from each treatment from 2020 data.

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

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

535 These results indicate that mounding results in a decline in CO₂ uptake and increase in CH₄ emission, at
 536 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
 539 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
 542 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

546 needed for this transition remains unclear largely due to the young age of most restoration projects.
547 Therefore, regardless of tree establishment, increased CH₄ emissions from hollows are likely to persist
548 for years. For example, Murray et al. (2021) measured CH₄ fluxes of 14.3 g CH₄ m⁻²y⁻¹ (39.2 mg CH₄ m⁻²d⁻¹)
549 from hollows 9 years after restoration, compared to 1.4 g CH₄ m⁻²y⁻¹ (3.8 mg CH₄ m⁻²d⁻¹) from natural
550 and 6.2 g CH₄ m⁻²y⁻¹ (16.8 mg CH₄ m⁻²d⁻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.

565

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