

# *Sensitivity of peatland litter decomposition to changes in temperature and rainfall*

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# 1 Sensitivity of peatland litter decomposition to changes in 2 temperature and rainfall

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## 4 AUTHORS AND AFFILIATIONS

5 Michael C. Bell<sup>a,\*</sup>, Jonathan P. Ritson<sup>b,c</sup>, Anne Verhoef<sup>a</sup>, Richard E. Brazier<sup>d</sup>, Michael  
6 R. Templeton<sup>c</sup>, Nigel J.D. Graham<sup>c</sup>, Chris Freeman<sup>e</sup>, Joanna M. Clark<sup>a</sup>

7 <sup>a</sup> Soil Research Centre, Department of Geography and Environmental Science, School  
8 of Archaeology, Geography and Environmental Science, The University of Reading,  
9 Whiteknights, PO Box 227, Reading RG6 6AB, UK

10 <sup>b</sup> Grantham Institute for Climate Change, Imperial College London, South Kensington,  
11 London SW7 2AZ, UK

12 <sup>c</sup> Department of Civil and Environmental Engineering, Imperial College London, South  
13 Kensington, London SW7 2AZ, UK

14 <sup>d</sup> Geography, College of Life and Environmental Sciences, University of Exeter, EX4  
15 4RJ, UK

16 <sup>e</sup> Wolfson Carbon Capture Laboratory, School of Biological Sciences, Bangor  
17 University, Bangor, Gwynedd LL57 2UW, UK

18 <sup>\*</sup> Corresponding author: Soil Research Centre, Department of Geography and  
19 Environmental Science, School of Archaeology, Geography and Environmental Science, The  
20 University of Reading, Whiteknights, PO Box 227, Reading RG6 6AB, UK. E-mail address:  
21 m.c.bell@pgr.reading.ac.uk

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23

## 24 Abstract

25 Changes to climate are projected over the next 50 years for many peatland areas. As  
26 decomposition of peat-forming vegetation is likely to be intrinsically linked to these changes in  
27 climate, a clear understanding of climate-peat dynamics is required. There is concern that  
28 increased temperature and decreased precipitation could increase the rate of decomposition  
29 and put the carbon sink status of many peatlands at risk, yet few studies have examined the  
30 impact of both climatic factors together. To better understand the sensitivity of peatland  
31 decomposition to changes in both temperature and precipitation and their interaction, we  
32 conducted a short-term laboratory experiment in which plant litters and peat soil were  
33 incubated, in isolation, in a factorial design. Treatments simulated baseline and projected  
34 climate averages derived from the latest UK climate change projections (UKCP09) for Exmoor,  
35 a climatically marginal peatland in SW England. Regular carbon dioxide flux measurements  
36 were made throughout the simulation, as well as total mass loss and total dissolved organic  
37 carbon (DOC) leached. The largest effect on carbon loss in this multifactor experiment was  
38 from substrate, with *Sphagnum*/peat releasing significantly less C in total during the  
39 experiment than dwarf shrubs/graminoids. Climate effects were substrate specific, with the  
40 drier rainfall treatment increasing the DOC leaching from *Calluna*, but decreasing it from  
41 *Sphagnum*. Partitioning between CO<sub>2</sub> and DOC was also affected by climate, but only for the  
42 peat and *Sphagnum* samples, where the future climate scenarios (warmer and drier) resulted  
43 in a greater proportion of C lost in gaseous form. These results suggest that indirect effects of  
44 climate through changes in species composition in peatlands could ultimately turn out to be  
45 more important for litter decomposition than direct effects of climate change from increased  
46 temperatures and decreased rainfall.

47 **Keywords:** decomposition; Exmoor; blanket peat; uplands; DOC; UK

48

## 49 1. Introduction

50 Northern peatlands are an important carbon store, holding around one third of the  
51 global soil carbon stock (Gorham, 1991). For peatlands to accumulate organic matter, and thus  
52 sequester carbon from the atmosphere, the overall loss of carbon from the system through  
53 the combined decomposition of plant litter and peat must be lower than C input via litter  
54 production and vascular plant root exudation (Frolking et al., 2010; Limpens et al., 2008).  
55 Decomposition is the breakdown of organic matter. It can be due to one or a number of  
56 physical, chemical or biological processes. Carbon dioxide (CO<sub>2</sub>), dissolved organic carbon  
57 (DOC), nutrients and stable hummus are among the principal final products of decomposition  
58 (Bragazza et al., 2009). Carbon is primarily lost as CO<sub>2</sub> in peatlands (Billett et al., 2010).  
59 However, additional losses in the form of methane (CH<sub>4</sub>), produced during anaerobic  
60 decomposition, and DOC to aquatic systems can be an important component of the peatland  
61 carbon balance in some areas (Köehler et al., 2011). The speed and final product of the  
62 decomposition process is dependent on the chemistry of the organic matter, and the  
63 environment in which it decomposes (Limpens et al., 2008).

64 Understanding how peat and plant litter decomposition could be affected by climate  
65 change, specifically changes in temperature and rainfall, is key to an improved understanding  
66 of how the peatland carbon source/sink status could change during the 21<sup>st</sup> century. Northern  
67 peatlands are typically dominated by remains of mosses of the genus *Sphagnum*, a bryophyte  
68 (Turetsky, 2003). *Sphagnum* has played an important role in peatlands becoming such a large  
69 carbon sink. Not only does the recalcitrance of its litter mean that it keeps C in the system for  
70 longer, but the presence of *Sphagnum* maintains acidic, nutrient poor conditions which make  
71 it difficult for plants with more labile litter to establish themselves (van Breemen, 1995).

72 Climate change may alter vegetation types in peatlands with vascular plants becoming more  
73 dominant (Buttler et al., 2015; Dieleman et al., 2015; Fenner et al., 2007; Weltzin et al., 2003).  
74 However, many ongoing catchment management programmes aim to encourage *Sphagnum*  
75 species over vascular plants as part of peatland restoration programmes (Grand-Clement et  
76 al., 2013).

77 While some authors have studied the effects of climate variables on the treatability of  
78 water from different peatland plant species (e.g. Ritson et al., 2014; Tang et al., 2013), little  
79 work has so far been conducted on the total flux of carbon and the partitioning between  
80 gaseous and fluvial losses from different peatland plant species. This partitioning is an  
81 important parameter to include in carbon cycle models, particularly for climate change  
82 modelling as there is likely to be a lag between DOC release from the soil and its incorporation  
83 into the atmospheric pool, as in-stream processing can lead to the temporary storage of C  
84 within the aquatic system (Hope et al., 1994). An understanding of partitioning of carbon  
85 losses between aquatic and gaseous fractions from different vegetation sources is also  
86 important for predicting whether catchment management programmes aimed at restoring  
87 certain species have the desired holistic effects of improving drinking water quality (including  
88 enhanced C sequestration) and do not result in environmental problem-shifting.

89 Decomposition of litters in the field is traditionally measured using the litter bag  
90 technique (Bragazza et al., 2009; Wieder and Lang, 1982). Litter of a known mass is enclosed  
91 in mesh bags with openings large enough for decomposers to access the food source, but  
92 small enough to prevent the physical loss of litters. Samples are incubated in the field, either  
93 on, or just below the soil surface (Johnson and Damman, 1991; Moore et al., 2007) and  
94 decomposition rate is quantified via mass loss or changing nutrient quotients of the litter  
95 (Keuskamp et al., 2013). *In situ* litter decomposition studies, using this technique, have

suggested multiple drivers for increased decomposition rates in peatland systems including: elevated nutrient additions (Bubier et al., 2007), water-table drawdown (Straková et al., 2012), litter quality (Limpens and Berendse, 2003) and temperature (Moore et al., 2007). However, these field-based studies are limited to a single measure of decomposition; mass loss. Mass loss gives a valuable measure of net decomposition, which can be useful for comparing between litters, but does not provide information on the final product of the decomposition process. Carbon dioxide and DOC not only represent different flux pathways (gaseous versus aquatic), but can also be indicative of different processes and stages of decomposition. CO<sub>2</sub> is in effect representative of fully utilised carbon, whereas DOC could be stored in the peat column or utilised by microbes and exported to the atmosphere via respiration (Pastor et al., 2003; Turetsky, 2003).

Laboratory incubations of peatland litters to examine the relationship between decomposition and climate have been limited to date. Studies that have been conducted have tended to concentrate on a single measure of decomposition, have focussed solely on peat rather than vegetation, or have just looked at one environmental variable, such as temperature (e.g. Neff and Hooper 2002; Moore et al. 2008) or water table (e.g. Freeman et al. 1993), in isolation. The effect of interactions between changes in temperature and rainfall on the decomposition of peatland litter is not currently well understood (Breeuwer et al., 2008). A gradient based mesocosm study measured increased C soil respiration in mesocosms transplanted to a warmer, drier location (Bragazza et al., 2016). However, isolating the effects of climatic variables such as temperature and rainfall from other confounding variables can be difficult in the field as these factors naturally co-vary. It is also difficult to separate the effects on different litters and measure the relative decomposition from different sources in the field or in intact mesocosms. To address this, previous studies have incubated soils or

litters from wetland sites in isolation, notably Moore and Dalva (2001), Neff and Hooper (2002) and Wickland et al. (2007). This was the approach taken here.

A better understanding of the sensitivity of different peatland litters to changes in temperature and rainfall regime is needed to enable predictions of the impact of climate change on the stability of peatland carbon stocks, and to support better parameterisations of models that simulate past, current and future climate (Frolking et al., 2010; Smith et al., 2010). Therefore, the objectives of this research were as follows: (1) assess how litter decomposition rates varied between common ombrotrophic peatland plant species (*Calluna vulgaris*, *Molinia caerulea*, *Sphagnum* moss, and mixed litter) compared to peat; (2) determine the relative importance (or partitioning) of gaseous (i.e. CO<sub>2</sub>) versus aquatic (DOC) fluxes during decomposition with respect to plant species and peat; (3) evaluate the importance of simulated temperature and rainfall changes on controlling total decomposition, C fluxes and partitioning between CO<sub>2</sub> and DOC. This final objective links to another study (Ritson et al., 2014), which used the same experimental treatments to examine the impact of climate change on the treatability (for potable water supply) of dissolved organic matter.

135

## 2. Methods

### 2.1. Field site and sample collection

In the south-west UK, blanket peat covers large parts of the upland areas and constitute an important store of C (Parry and Charman, 2013). These blanket peatlands represent the southern-most blanket peat in the UK and are considered climatically marginal (Clark et al., 2010). The shallow peat of Exmoor was heavily damaged by intensive drainage during the 19<sup>th</sup> and 20<sup>th</sup> centuries, and recent efforts have been made to restore large areas



143 through a programme of ditch blocking (Grand-Clement et al., 2014, 2013). The field sites are  
144 dominated by *Molinia caerulea* (purple moor grass) (Gatis et al., 2015) and are classified as UK  
145 National Vegetation Classification M25: *Molinia caerulea* - *Polentilla erecta* mires (Rodwell,  
146 1991). Paleoecological studies indicate that purple moor grass expansion corresponds with  
147 the industrial revolution and prior to this, *Sphagnum* was more dominant species within this  
148 region (Chambers et al., 1999). Exmoor typically receives precipitation of around 1800–2600  
149 mm yr<sup>-1</sup> with mean winter and summer temperatures of between 4.5–5.5 °C and 10–12 °C,  
150 respectively (Met Office, 2014). Exmoor receives a relatively high input of nutrients through  
151 atmospheric deposition, and the mean deposition of total nitrogen and oxidised sulphur for  
152 the period 2008–2012 was estimated to be 15.5–16.16 kg N h<sup>-1</sup> yr<sup>-1</sup> and 4.41–4.86 kg S h<sup>-1</sup> yr<sup>-1</sup>,  
153 respectively (DEFRA, 2015).

154 Samples of vegetation and peat were collected from two catchments, Aclands (51°  
155 07'54.2" N 3° 48'43.3" W) and Spooners (51° 07'23.3" N 3° 45'11.8" W), within Exmoor  
156 National Park, UK, during July 2013. A further description of these field sites, including maps,  
157 can be found in Grand-Clement et al. (2014) and Luscombe et al. (2015). Five different  
158 substrates were collected: *Calluna*, *Molina* and *Sphagnum*, mixed litter and peat. Fresh leaves  
159 and branches were collected from *Calluna vulgaris* and *Molinia caerulea*, avoiding any plants  
160 which had been partially eaten by herbivores. Mixed litter (predominantly *Molinia caerulea*  
161 but also some *Eriophorum vaginatum*, both at the early stages of decomposition) was collected  
162 from the bog surface. Intact *Sphagnum* spp. sods (entire plants) were collected from the  
163 centre of stands and peat was collected from the top ca. 30 cm of the soil profile using a screw  
164 auger. The samples were transported back to the laboratory in cool boxes within 8 hours of  
165 collection, and stored at 4 °C prior to their preparation. Sample collection is described fully in  
166 Ritson et al. (2014).

167

## 168 2.2. Sample preparation

169 Samples from each vegetation class (*Calluna vulgaris*, *Molinia caerulea*, mixed litter  
170 and *Sphagnum* spp.), were cut into 2-3 cm sections and mixed thoroughly within their class.  
171 Any material not from the target group was removed. Once sorted, cut and homogenised, the  
172 vegetation samples were left to air-dry in the laboratory to constant mass (Moore et al., 2007).  
173 The structure of peat is permanently altered when dried, becoming hydrophobic (Holden and  
174 Burt, 2002). For this reason, the peat was treated differently to the other substrates and was  
175 kept refrigerated and at field moist conditions prior to the experiment.  
176 Air-dried sub-samples of each plant species and field moist peat ( $n = 5$ ) were oven dried at 70  
177 °C for 24 h to determine an air-dry to oven-dry conversion factor. This conversion factor was  
178 applied to estimate the oven-dry weight of samples at the beginning of the simulation. These  
179 oven-dry samples were analysed for litter quality (see methods below; Table 1).

180

## 181 2.3. Experimental design

### 182 2.3.1. Experimental set up

183 Polypropylene Buchner funnels (Thermo Scientific, USA) were filled with  
184 approximately 2 g dry-weight of air-dry vegetation or 10 g homogenised peat. The stem of the  
185 funnel was packed with acid washed glass wool to ensure no loss of particulates, and each  
186 Buchner funnel was placed above an amber-glass bottle to collect leachate, for separate  
187 analysis (see Ritson et al., 2014). The climate control facilities comprise two versatile  
188 environmental test chambers (Panasonic MLR-352H). The incubators were set to cycle  
189 between 12 h at the mean daily maximum and 12 h at the mean daily minimum temperature.

Deionised water was applied in amounts equivalent to the monthly rainfall, with a dispenser set to the required volume based on the area of the Buchner funnel top. This was applied gradually and evenly, over approximately 30 seconds, taking care not to disturb the samples.

### 2.3.2. *Experimental design*

The experiment was set up as a factorial ANOVA design, where substrate, temperature and rainfall were the treatment factors in a 5 x 2 x 2 design. Five replicates of each treatment were conducted, giving a total of 100 samples (see Ritson et al., 2014). Climate treatments were based on (i) baseline (1961-1990; Met Office, 2013) mean minimum and mean maximum daily temperature and mean rainfall totals for July and August for Exmoor National Park, UK, and (ii) a projection from the UKCP09 high emissions scenario for 2080s (2070-2100). All four possible combinations of baseline and projected temperature and rainfall were applied as treatments. Mean minimum daily temperatures were set to 11.0 °C ( $\pm 0.3$  °C for test chamber error) for the baseline, and 15.4 °C ( $\pm 0.3$  °C) for the UKCP09 treatment. Mean maximum daily temperatures were set to 18.2 °C ( $\pm 0.3$  °C) for the baseline treatment and 23.7 °C ( $\pm 0.3$  °C) for the UKCP09 treatment. Rainfall totalled 100.9 and 117.5 mm for the baseline treatment for July and August, respectively, and 75.4 and 67.2 mm for the UKCP09 treatment. Overall, this was equivalent to a ca. 5.0 °C increase in temperature and ca. 35 % decrease in rainfall for the UKCP09 treatments. The relative humidity (to program the test chambers) and number of rain days were set at mean values from the baseline period for all samples (for relative humidity: 80 % in July, 79 % in August; for rain days: 12 in July and 14 in August). The rainfall total for each month was split evenly between the rain days (resulting in irrigation amounts ranging between 4.2 mm and 8.4 mm per event, or 19 and 38 ml, depending on treatment and month),

213 and the amounts were applied every 2-4 days. The conversion from mm to ml was based on  
214 the area of the funnel.

215

## 216 2.4. Measurements

### 217 2.4.1. Carbon dioxide evolution

218 At eight points during the experiment (on days 2, 7, 10, 17, 31, 45, 50, and 60 after the  
219 start of the incubation) samples were analysed for CO<sub>2</sub> efflux. Samples were taken out of the  
220 incubators individually and the top part of the Buchner funnel, containing the sample, was  
221 removed and placed into an airtight container (Lock&Lock, South Korea) which was connected  
222 to an infra-red gas analyser, IRGA (8100A, LI-COR, USA). Samples were removed for  
223 measurement in a random order to avoid the introduction of systematic measurement error.  
224 Observations with the IRGA lasted for 2 minutes during which time the concentration of CO<sub>2</sub>  
225 in the chamber (water vapour compensated) was measured every second, and the samples  
226 were out of the incubators for a maximum of 5 minutes. The rate of change in CO<sub>2</sub>  
227 concentration in the headspace during the 2-minute measurement period was used to  
228 calculate the instantaneous flux by applying a linear regression. The IRGA system was allowed  
229 to purge for a minute between samples, and a 30 second deadband (a period where the IRGA  
230 is running without the measurements being used in the flux estimate) was introduced at the  
231 beginning of each measurement to allow for adequate mixing of air in the headspace. The CO<sub>2</sub>  
232 time series for each sample was analysed separately, and the deadband was adjusted where  
233 necessary. All flux estimates were derived from a minimum of 60 consecutive CO<sub>2</sub>  
234 concentration measurements (i.e. 60 seconds). Individual flux estimates were converted to  
235 mg CO<sub>2</sub>-C g C<sup>-1</sup> day<sup>-1</sup>. Cumulative CO<sub>2</sub> flux over the course of the two month experiment was

236 calculated for each sample based on a linear interpolation of fluxes between the successive  
237 measurements (Neff and Hooper, 2002), expressed as  $\text{mg CO}_2\text{-C g C}^{-1}$ .

238

#### 239 2.4.2. *Dissolved organic carbon*

240 DOC analysis was performed on the cumulative leachate collected at the end of the  
241 simulation period. Samples were filtered through a GF/F filter (Whatman, UK) with a nominal  
242 pore size of  $0.7\ \mu\text{m}$ , acidified to  $\sim\text{pH } 2$  with HCl and stored at  $4^\circ\text{C}$  prior to analysis. Organic  
243 carbon content was measured as non-purgeable organic carbon (NPOC) using a total organic  
244 carbon analyser (Shimadzu TOC-V, Japan). Calibration and quality control checks were run  
245 daily from freshly prepared standards and samples were analysed within one week of the end  
246 of the simulation. Organic carbon concentration ( $\text{mg L}^{-1}$ ) was converted to DOC leached (mg)  
247 based on the volume of leachate collected. As temperature and rainfall treatment affected  
248 the volume of sample collected, this was then reported as  $\text{mg DOC-C g C}^{-1}$ , based on the  
249 starting dry mass and initial C content of each sample.

250

#### 251 2.4.3. *Mass loss*

252 At the end of the two-month incubation period, samples were removed from the  
253 incubators, dried at  $70^\circ\text{C}$  for 48 h and weighed. C loss was estimated based on the starting  
254 mass and starting C content, as well as the end mass and end C content. This was expressed  
255 as  $\text{mgC g C}^{-1}$ .

256

## 257 2.5. *Litter quality analysis*

258 The oven-dry subsamples from the beginning, and all samples from the end of the  
259 experiment were ground using a laboratory disc mill with an agate barrel (Tema Machinery  
260 Ltd, UK) and analysed for total C and N (Thermo FLASH 2000 elemental analyser). The handling  
261 and analysis of samples were undertaken together with reference material (Pahokee Peat,  
262 International Humic Substances Society) in order to verify the calibration and quality of the  
263 results. Subsamples from the start of the experiment were also analysed for C character using  
264 nuclear magnetic resonance (NMR) analysis. Cross polarisation-magic angle spinning (CP-  
265 MAS)  $^{13}\text{C}$  NMR analysis was undertaken at the University of Reading chemical analysis facility  
266 (CAF) on a Bruker AV500 instrument based on the method of Wong et al. (2002). This  
267 technique is used widely in environmental studies and is good for looking at relative changes  
268 in carbon distribution across a series of samples (Simpson et al., 2011). For the vegetation  
269 samples 1024 transients (executions of the pulse sequence) were taken, however due to the  
270 presence of paramagnetic material in the peat soil transients were increased to 4096 to  
271 improve the signal to noise ratio. Magic angle spinning (MAS) spin rate was set at 10 kHz to  
272 shift spinning side bands out of the region of interest and all signals were externally referenced  
273 to adamant. The aromatic peak between 110-140 ppm was integrated and its percentage of  
274 the total signal calculated to give an indication of the differences between relative aromaticity  
275 between samples (Peuravuori et al., 2003), which can be used as an alternative qualitative  
276 measure of the recalcitrance of the litter (see Table 1). For further reading on using NMR  
277 analysis for environmental research, see Simpson et al. (2011).

278

## 279 2.6. Statistical analysis

280 All statistical analyses were carried out using R version 3.0.2 (R Core Team, 2015) and  
281 plots were generated using the ggplot2 package (Wickham, 2009). The experiment involved a  
282 three-way analysis of variance (ANOVA) design where temperature, rainfall and substrate  
283 were the experimental factors. The effect of these factors, and their interaction, on response  
284 variables of CO<sub>2</sub> flux, mass loss, DOC leached and CO<sub>2</sub>:DOC ratio production was analysed  
285 (Table 2). A Fligner-Killeen test was performed to determine that there was equal variance  
286 between groups, and residuals were checked for normality. Carbon dioxide flux, DOC flux and  
287 CO<sub>2</sub>:DOC ratio were all square-root transformed prior to inclusion, in order to meet the  
288 assumption of normally distributed residuals. A critical value of  $P = 0.05$  was used for statistical  
289 significance. For multiple comparisons, the Tukey honest significant differences (HSD) test was  
290 used. The variance explained by each factor was estimated using omega-squared ( $\omega^2$ ), a test  
291 which is suitable for estimating effect size from small sample sizes (Keselman, 1975).

292

## 293 3. Results

### 294 3.1. Variability of decomposition rates between different substrates

295 Decomposition was greatest for the *Molinia* samples and slowest for the peat samples  
296 (Fig. 1). The same substrate rankings were evident in terms of the following: the estimated  
297 cumulative CO<sub>2</sub> flux (means ranged from 942.87 mgC gC<sup>-1</sup> for *Molinia* to 41.26 mgC gC<sup>-1</sup> for  
298 peat; Fig. 1); the DOC flux (means ranged from 20.95 mgC gC<sup>-1</sup> for *Molinia* to 1.07 mgC gC<sup>-1</sup>  
299 for peat; Fig. 1); and the overall total measured mass loss (means ranged from 511.21 mgC  
300 gC<sup>-1</sup> for *Molinia* to 27.43 mgC gC<sup>-1</sup> for peat; Fig. 2). Substrate was a significant explanatory  
301 factor in the ANOVA for all of these response variables ( $P < 0.001$ ; Table 2). Post-hoc tests

suggested significant differences in the order *Molinia* > *Sphagnum* = Mixed litter > *Calluna* > peat for CO<sub>2</sub> flux; *Molinia* > *Calluna* = Mixed litter > *Sphagnum* > peat for DOC flux; and *Molinia* > *Calluna* > *Sphagnum* = Mixed litter > peat for mass loss.

There was a steady decrease in CO<sub>2</sub> release over the entire incubation period, for all substrates and treatments. The decrease in CO<sub>2</sub> flux was greatest for the *Molinia* samples, with a reduction in estimated flux between the first (day 1 to 30) and second half (day 31 to 60) of the experiment of 64.27%, compared to peat where the difference was 25.13% – the lowest decline recorded.

In terms of total mass balance, the overall loss of C measured directly through mass loss of the substrate and by the total fluxes (DOC + CO<sub>2</sub>) were not equal. Whilst DOC fluxes were less than the total mass loss, the cumulative CO<sub>2</sub> flux estimated by integration across the experimental period (*sensu* Neff and Hooper, 2002) was greater than the total mass loss.

### 3.2. Carbon dioxide to DOC partitioning

The cumulative flux of CO<sub>2</sub> was greater than DOC for all of the samples (Fig. 1). Mean CO<sub>2</sub>:DOC quotients ranged from 31.56 for *Molinia* to 68.67 for *Sphagnum* (all climate treatments pooled), indicating that proportionally more carbon was lost as DOC from *Molinia* than *Sphagnum*. Substrate was again the strongest factor explaining the variation in partitioning between gaseous and aquatic fluxes ( $P < 0.001$ ). There was a significant relationship between total CO<sub>2</sub> flux and DOC leached between groups when all data were pooled, indicating that DOC leaching increases with CO<sub>2</sub> flux. However, this relationship was not observed within substrate groups.

The general trend was for an increase in the CO<sub>2</sub>:DOC quotient under UKCP09 future climate scenarios (Fig. 1). There was a statistically significant interaction between rainfall and



326 substrate ( $P < 0.001$ ), with the proportion of C lost as CO<sub>2</sub> generally increasing under the  
327 UKCP09 (drier) rainfall scenario. Post-hoc tests revealed that drier conditions were only  
328 significant in changing the partitioning between gas and aquatic carbon fluxes for the peat ( $P$   
329  $< 0.001$ ) and *Sphagnum* ( $P < 0.001$ ) samples, the mean of which increased from 24.91 ( $\pm 5.74$ )  
330 to 78.64 ( $\pm 13.85$ ) and from 47.34 ( $\pm 2.41$ ) to 101.78 ( $\pm 16.25$ ) when comparing the baseline  
331 temperature and rainfall to the UKCP09 temperature and rainfall treatments. Partitioning  
332 between CO<sub>2</sub> and DOC in other substrates, *Molinia*, *Calluna* and mixed litter, was unaffected  
333 by temperature and rainfall treatments ( $P > 0.05$ ).

334

335 3.3. *Relative importance and interaction between temperature and rainfall in controlling*  
336 *decomposition*

337 Increasing temperature had a significant effect on decreasing DOC flux ( $P < 0.001$ ) and  
338 increasing mass loss ( $P < 0.001$ ). In addition, significant interactions between temperature and  
339 substrate ( $P < 0.001$ ) and rainfall and substrate ( $P = 0.001$ ) were seen for DOC flux and for  
340 mass loss ( $P = 0.007$  and  $P = 0.035$ , respectively), indicating that the effect of simulated climate  
341 change depended on substrate type. For DOC, post hoc tests suggested that the interaction  
342 was only significant between temperature and *Calluna*, whereas for rainfall, there was a  
343 significant interaction between rainfall treatment and *Calluna* and rainfall treatment and  
344 *Sphagnum*. This interaction between rainfall and substrate was not unidirectional, however,  
345 with the drier rainfall treatment increasing DOC in *Calluna* samples, but decreasing it in  
346 *Sphagnum*. With mass loss, only *Sphagnum* was close to being significant on its own.  
347 Differences in cumulative CO<sub>2</sub> flux between climate treatments (both temperature and  
348 rainfall) were non-significant ( $P > 0.05$ ), though rainfall was close to the critical threshold ( $P =$

0.075). No significant interaction between temperature and rainfall for any of the measures of decomposition was observed (CO<sub>2</sub> flux, DOC flux or mass loss).

## 4. Discussion

### 4.1. *Carbon fluxes and decomposition rates between different substrates*

The fluxes of CO<sub>2</sub> and DOC, and the mass loss, were lowest from the peat samples compared to litter sources, consistent with previous work (Moore and Dalva, 2001; Tang et al., 2013). Decomposition in peatland ecosystems progresses along a continuum from freshly senesced plant litter to peat. Part of the reason that peat decomposes more slowly than fresh litter is because more labile compounds are preferentially utilised by decomposers (Wieder and Lang, 1982). This results in older peat being composed of progressively more recalcitrant materials (Broder et al., 2012). In the field, these differences are likely to be even greater as decomposition of peat is further inhibited by anoxic and acidic conditions which prevail for longer at greater depths. Here, we used measures of C:N ratio and aromaticity to characterise the quality of the substrates. A high C:N ratio usually suggests recalcitrant litter, as N is limiting for microbes and can also stimulate the activity of certain enzymes (Bragazza et al., 2012). Here peat had the lowest C:N ratio, which could be due to retention of N containing compounds during the decomposition process (Kalbitz and Geyer, 2002). Aromaticity is another potential measure of decay resistance as aromatic polymers are characterised by structural and chemical recalcitrance. In this instance, peat had the highest estimated proportion of aromatic compounds (Table 1), which could in part explain its low decomposition rate relative to the other substrates.

371 In terms of specific fluxes, *Sphagnum* produced the least DOC in line with previous  
372 observations of high retention of carbon in this species (Fenner et al., 2004). *Sphagnum* also  
373 produced slightly more CO<sub>2</sub> than *Calluna*, which was unexpected given the well documented  
374 recalcitrance of *Sphagnum* litter which has been shown to decompose much more slowly than  
375 other peat forming vegetation (van Breemen, 1995). This could have been a result of  
376 environmental conditions, specifically water content, as the poor water holding capacity of  
377 *Calluna* litter meant that lack of water supply could have limited its decomposition during  
378 laboratory simulation experiments (Blok et al., 2015; Toberman et al., 2008).

379 The estimates of cumulative CO<sub>2</sub> flux were approximately double the amount  
380 estimated from C mass loss, suggesting an overestimation of cumulative CO<sub>2</sub> flux by  
381 integrating point measurements recorded here. This could partly be due to the measurement  
382 of some autotrophic respiration from peat roots in the earlier stages of the experiment, or an  
383 inadequate representation of night-time respiration rates. However, given that all samples  
384 were treated in the same way, and were well mixed to begin with, it is reasonable to assume  
385 that the observed differences between groups are robust.

386

#### 387 4.2. Relative importance of gaseous and fluvial fluxes

388 The release of DOC during organic matter decomposition to surface waters is an  
389 important component of the peatland carbon budget (Billett et al., 2010). Little is known  
390 about partitioning between CO<sub>2</sub> and DOC fluxes in peatlands (Bonnett et al., 2006). Herein, it  
391 was shown that *Molinia* produced the most DOC as a proportion of its starting mass, and peat  
392 produced the least. However, in terms of the ratio of CO<sub>2</sub> to DOC, *Calluna* and mixed litter had  
393 the lowest ratios, signalling that a higher proportion of mass is lost as DOC than CO<sub>2</sub> for these  
394 substrates, when compared with the others. Again, absolute values for CO<sub>2</sub>:DOC may not be

395 accurate due to overestimation of measured CO<sub>2</sub> fluxes, but comparisons between groups  
396 should still be robust.

397 Previous studies have reported DOC fluxes equivalent to over 20% of net ecosystem  
398 exchange (NEE) (e.g. Dinsmore et al., 2010; Koehler et al., 2011). In some instances, DOC  
399 exported to streams has been estimated to exceed NEE (Billett et al., 2004), highlighting the  
400 importance of including this flux when estimating source/sink status of a peatland. Strong  
401 correlations have been found between gross primary productivity (GPP) and DOC  
402 concentrations in peatland catchments (Dinsmore et al., 2013; Harrison et al., 2008),  
403 suggesting a large amount of DOC is produced directly by living plants, indirectly by  
404 rhizosphere priming or by the decay of fresh plant litter. The difference in the amount of DOC  
405 released from the peat and litter samples in this study during decomposition supports the  
406 theory that the decay of fresh litter is a substantive DOC source (Evans et al., 2007; Palmer et  
407 al., 2001). In the field, a lot will depend on the regulating effect of water table depth on  
408 decomposition, flow pathways of runoff (Wallage and Holden, 2011), as well as substrate.

409 We observed a lower DOC flux but no significant change in CO<sub>2</sub> flux under the UKCP09  
410 (warmer) temperature scenario. Combined with the lower DOC flux under UKCP09 (drier)  
411 rainfall scenario for some of the substrates (*Sphagnum*), this resulted in an increase in the  
412 partitioning coefficient between CO<sub>2</sub> and DOC flux for some of the substrates (*Sphagnum* and  
413 peat), which is in accordance with the findings of Neff and Hooper (2002) in their arctic soil  
414 samples. This implies that a greater proportion of C released from *Sphagnum* and peat would  
415 be lost in the form of CO<sub>2</sub> than DOC under warmer and drier conditions, with the effect of  
416 rainfall being slightly stronger than temperature. The strength of the rainfall effect suggests  
417 that it is due more to the decreased flux of DOC than due to a reduction in total  
418 decomposition. The total DOC measured in this study was in effect net DOC production; a

function of release and transport, as well as its mineralisation. The decreased DOC flux could be explained by a suppression of DOC mobility by drought-induced acidification (Clark et al., 2006; Clark et al., 2012). DOC production could be affected by both microbial enzymatic activity and physical leaching. It could be that DOC was produced, but lack of water supply limited physical leaching and so it was subsequently consumed by microbes present in the samples (Moore and Dalva, 2001). The findings presented here, that CO<sub>2</sub>:DOC partitioning from *Sphagnum* and peat could be affected by climate change, but that the partitioning from other substrates was not affected, will be of interest to those modelling the response of peatland C balance as it indicates that DOC flux cannot be estimated from ecosystem respiration alone. It is also useful information for water treatment engineers who are trying to plan future climate-proof investments in treatment works, as these results imply that DOC could increase or decrease in the future, depending on the proportion of different plant species in the catchments, and the pattern of rainfall.

432

#### 4.3. *Effects and interactions between temperature and rainfall*

Results indicate that simulated temperature changes have a greater effect on the loss of DOC (generally decreasing the flux) and on mass loss (generally increasing the flux) than on the production of CO<sub>2</sub>. The effect varied between substrates, for example, drier conditions increased the DOC flux from *Calluna*, but decreased it from *Sphagnum*. When looking at the main drivers of increased decomposition in peatlands, other laboratory based studies have also found mixed results. Higher temperatures led to increased mass loss (between 5, 10 and 15 °C) in a one year incubation of Scots pine needles below a living moss layer (Domisch et al., 2006), supporting the results found here. Neff and Hooper (2002) also found that temperature controlled the CO<sub>2</sub> flux from their incubated soil samples, but the response depended largely

443 on vegetation community. However, in their study on DOC production across a range of spatial  
444 scales, Preston et al. (2011) found that temperature had no effect on DOC production in their  
445 peat microcosm experiment, regardless of moisture treatment. This contrasts with Clark et al.  
446 (2006) who found increased DOC release at 20 °C relative to 10 °C in another laboratory study  
447 incubating peat mesocosms.

448         Whilst measured mass loss did increase positively with temperature, measured CO<sub>2</sub>  
449 fluxes did not. Investigations at lower temperatures than those used here have suggested  
450 *Sphagnum* becomes more productive, growing faster, with increases in temperature from a  
451 low base (Breeuwer et al., 2008), but that the opposite can occur for warmer temperatures  
452 (17.5 and 21.7 °C) (Breeuwer et al., 2009). Results herein indicate that decomposition,  
453 measured through mass loss, increased with temperature, suggesting that the strength of the  
454 peatland C sink could be reduced during summer months.

455         Rainfall treatment was just above the  $P = 0.05$  critical threshold for cumulative CO<sub>2</sub> ( $P$   
456 = 0.075) and DOC flux ( $P = 0.070$ ), with lower values seen under the UKCP09 (lower rainfall)  
457 conditions. For DOC in peat, this agrees with the findings of Tang et al. (2013) who found that  
458 DOC export was greater in the wetter treatments, and declined in the drought treatments.  
459 Observed responses of DOC export to drought or experimentally lowered water tables have  
460 been varied, with suggestions that DOC export could increase due to drought (Worrall et al.,  
461 2006). However, there are mechanisms which could explain the decrease in DOC observed  
462 here. Decreased DOC during droughts could be due to a suppression of DOC transport by  
463 drought-induced decrease in DOC solubility (Clark et al., 2012, 2006), or an increase in  
464 hydrophobicity (Worrall et al., 2008) rather than a change in the production of DOC through  
465 organic matter decomposition. A difference here is that this study simulated rainfall whereas

those examining the mechanisms have tended to focus on water table effects, either through manipulations in the laboratory or event-based analysis in the field.

#### 4.4. *Limitations of the study*

The data presented here are a useful indication of the relative importance of different substrates as sources of DOC and CO<sub>2</sub> fluxes, decomposition and the potential for changes in partitioning between CO<sub>2</sub> and DOC under future climate conditions. However, these are laboratory based simulation experiments and are subject to constraints that must be taken into account when interpreting the results. Samples may have been exposed to more extreme drying conditions than under field conditions as moisture levels were not regulated by living vegetation and/or underlying water table, and the lack of a peat substrate beneath the samples from the vegetation groups could also have increased the loss of water and so negated any rainfall treatment effect. Ritson et al. (2017) found that any exposure to oxygen (during laboratory experiments) increased DOC from peat and changed the quality of the C. The differences with the results presented here suggest a high sensitivity to the amount and frequency of the rainfall treatments. The findings of these laboratory experiments now need to be tested under field conditions.

#### 4.5. *Wider implications for our understanding of peatland carbon balances*

While predictions about future rainfall patterns are variable, with both increases and decreases projected for northern latitudes (IPCC, 2014), models are in strong agreement that temperatures at mid-to-high latitudes will increase due to climate change (Murphy et al., 2009), and that drought events are likely to become more frequent (Meehl and Tebaldi, 2004).

489 While there are differences in both magnitude and direction of change between the  
490 substrates, the data here suggest that warmer summer temperatures and decreased rainfall  
491 could increase overall decomposition, reduce the flux of DOC, and subsequently alter the  
492 partitioning of carbon released during decomposition from DOC to CO<sub>2</sub>.

493 Other studies have suggested that the effect of climate change will be more through  
494 indirect effects on the composition of vegetation, with graminoids and dwarf shrubs being  
495 favoured over *Sphagnum* mosses in warmer, drier conditions (Kapfer et al., 2011; Weltzin et  
496 al., 2003). Given the greater lability of the litter produced by these plant functional types, seen  
497 here in the greater decomposition of the *Molinia* samples, this could be a concern for climate  
498 change mitigation. However, in SW England, where these samples were collected, graminoids  
499 (*Molinia*) are already highly prevalent (Chambers et al., 1999) and so the effect may not be as  
500 severe, as climate-induced vegetation change may already have occurred. The shallow peat in  
501 this area suggests the area could have been sequestering C at a lower rate than other blanket  
502 peatlands, or even losing C, though further work looking at the entire C balance is needed to  
503 verify this.

504 In the field, it is likely that climate change may increase productivity, and therefore litter  
505 production, alongside any increase in decomposition rates. Studies have already provided  
506 evidence that warmer temperatures can increase rates of carbon sequestration at some  
507 northern latitude peatlands, due in part to longer and warmer growing seasons enhancing  
508 plant growth (Loisel and Yu, 2013). Therefore, when interpreting the differences in  
509 decomposition rates of the different substrates in the context of *in situ* actual conditions, it is  
510 also important to consider their respective litter production rates and substrate supply (Ritson  
511 et al., 2016). While *Molinia* showed the fastest decomposition in this experiment, it also tends  
512 to produce more litter than *Calluna* (e.g. Aerts, 1989), so is likely to be a more significant



513 source of C at the catchment scale. Also, while both substrates generally produce less “litter”  
514 than *Sphagnum* (Bragazza et al., 2012), this may change in a warmer, drier climate (Breeuwer  
515 et al., 2009). Further work is needed to take into account the effect of climate change on  
516 productivity and litter production, as well as on decomposition, to assess the likely impacts of  
517 climate change on the future carbon balance of peatlands.

518

## 519 5. Conclusions

520 This study has used short-term laboratory incubations of peat and peat-forming  
521 vegetation from a climatically marginal blanket peatland in SW England in order to assess the  
522 relative importance of different temperature and rainfall scenarios for the main C fluxes (CO<sub>2</sub>  
523 and DOC) and their partitioning during short-term litter decay. The largest variation in C fluxes  
524 and total mass loss was seen between substrates, with smaller fluxes (cumulative CO<sub>2</sub> and  
525 DOC) and mass lost from *Sphagnum*/peat compared to higher fluxes and mass loss from dwarf  
526 shrub/graminoids. The CO<sub>2</sub>:DOC ratio was lowest for *Molinia* and highest for *Sphagnum*.  
527 Climate factors were important too, but the effect depended largely on the substrate. For  
528 example, while the future (drier) rainfall scenario increased the DOC flux from *Calluna*, it had  
529 the opposite effect for *Sphagnum*. Therefore, indirect effects on species composition in  
530 peatlands could ultimately turn out to be more important than direct effects of climate change  
531 from increased temperatures and decreased rainfall.

532 Upscaling of this work from laboratory to field is needed in order to confirm these  
533 laboratory-based findings and improve understanding of the likely impacts of climate change  
534 on C fluxes and decomposition in peatlands. Further work across natural climatic gradients is  
535 one way of addressing this research need, deploying a space for time substitution to  
536 understand future climate change.

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546

547

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801 Tables and figure captions

802 **Table 1.** Organic matter properties for peatland vegetation and peat classes used in the  
803 experiment. Values are means ( $n = 5$ ) of subsample values, numbers in brackets are the  
804 standard error of the mean. Aromaticity is a qualitative measure and was calculated from  
805 cross polarization-magic angle spinning nuclear magnetic resonance (CP-MAS NMR).

Class	Dominant substrate	C (%)	N (%)	C:N ratio	Moisture content (g g dry wt <sup>-1</sup> )	Aromaticity (%)
<i>Calluna</i>	<i>Calluna vulgaris</i>	49.3 (0.02)	1.35 (0.01)	36.4 (0.22)	1.90 (0.01)	8
Mixed litter	Senesced <i>Molinia c</i>	45.9 (0.01)	1.29 (0.01)	35.6 (0.28)	5.38 (0.11)	5
<i>Molinia</i>	<i>Molinia caerulea</i>	45.5 (0.02)	2.34 (0.02)	19.5 (0.13)	3.56 (0.05)	5
Peat	Peat	29.1 (0.02)	1.64 (0.01)	17.7 (0.07)	6.06 (0.06)	11
<i>Sphagnum</i>	<i>Sphagnum</i> spp.	41.8 (0.03)	1.00 (0.01)	42.1 (0.28)	15.020.15)	7

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817 **Table 2.** ANOVA model results, where the model fitted is *response* = Temperature \* Rainfall \*  
818 Substrate (Cumulative CO<sub>2</sub>, DOC and CO<sub>2</sub>:DOC ratio are square-root transformed data).  
819 Significant differences ( $P < 0.05$ ) are highlighted in bold and effect sizes ( $\omega^2$ ) are stated in  
820 brackets. “Temp”, “Rain” and “Sub” are short for Temperature, Rainfall and Substrate,  
821 respectively).

Variable	Factor	<i>F</i>	df	<i>P</i> value
Cumulative CO <sub>2</sub> (mgCO <sub>2</sub> -C gC <sup>-1</sup> )	Temperature	1.04	1	0.311
	Rainfall	3.25	1	0.075
	Substrate	631.08	4	<b>&lt; 0.001</b> (0.960)
	Temp:Rain	0.06	1	0.804
	Temp:Sub	1.27	4	0.288
	Rain:Sub	1.68	4	0.164
	Temp:Rain:Sub	1.27	4	0.290
Mass loss (mgCO <sub>2</sub> -C gC <sup>-1</sup> )	Temperature	20.10	1	<b>&lt; 0.001</b> (0.005)
	Rainfall	0.01	1	0.927
	Substrate	946.23	4	<b>&lt; 0.001</b> (0.966)
	Temp:Rain	0.01	1	0.915
	Temp:Sub	3.80	4	<b>0.007</b> (0.003)
	Rain:Sub	2.72	4	<b>0.035</b> (0.002)
	Temp:Rain:Sub	0.56	4	0.690
DOC (mgCO <sub>2</sub> -C gC <sup>-1</sup> )	Temperature	18.19	1	<b>&lt; 0.001</b> (0.015)
	Rainfall	3.38	1	0.070
	Substrate	247.14	4	<b>&lt; 0.001</b> (0.838)
	Temp:Rain	0.04	1	0.837
	Temp:Sub	5.12	4	<b>0.001</b> (0.014)
	Rain:Sub	11.34	4	<b>&lt; 0.001</b> (0.035)
	Temp:Rain:Sub	4.43	4	<b>0.003</b> (0.012)
CO <sub>2</sub> -C:DOC-C	Temperature	15.57	1	<b>&lt; 0.001</b> (0.056)
	Rainfall	26.16	1	<b>&lt; 0.001</b> (0.097)
	Substrate	19.96	4	<b>&lt; 0.001</b> (0.292)
	Temp:Rain	0.01	1	0.921
	Temp:Sub	3.27	4	<b>0.015</b> (0.035)
	Rain:Sub	8.71	4	<b>&lt; 0.001</b> (0.119)
	Temp:Rain:Sub	2.24	4	<b>0.072</b>

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824 **Figure 1.** Treatment effects on measures of carbon fluxes, and their ratio; (a) Cumulative CO<sub>2</sub>  
825 flux, (b) DOC flux, (c) Cumulative CO<sub>2</sub> to DOC ratio (Error bars indicate one standard error).  
826 Different letters denote statistically significant differences in means between substrates and  
827 asterisks denote significant differences between treatments for substrates based on Tukey  
828 HSD test ( $P < 0.05$ ).

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830 **Figure 2.** Treatment effects on C loss estimated from mass and C quotient measurements  
831 (Error bars indicate one standard error). Different letters denote statistically significant  
832 differences in means between substrates and asterisks denote significant differences  
833 between treatments for substrates based on Tukey HSD test ( $P < 0.05$ ).

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