

Drain blocking has limited short-term effects on greenhouse gas fluxes in a *Molinia caerulea* dominated shallow peatland

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ABSTRACT

Drained peatlands dominated by purple moor grass (*Molinia caerulea*) are widespread in the UK and Western Europe. Although substantial carbon stores may be present in these peatlands, in this degraded state they are not currently acting as carbon sinks. Therefore, *M. caerulea* dominated peatlands have been identified as potential sites for ecohydrological restoration to tackle the current climate emergency. However, at present little is known about whether ditch blocking can raise water tables and promote the recovery of bog plant species, and the subsequent effects on carbon sequestration in these peatlands.

To investigate the potential for restoration, we measured changes in water table depth, vegetation composition, photosynthesis at 1000 $\mu\text{mol Photons m}^{-2} \text{s}^{-1}$ (P_{G1000}), ecosystem respiration (R_{Eco}) and partitioned below-ground respiration in two *M. caerulea* dominated peatlands in which drainage ditches had been blocked located in Exmoor National Park, southwest England. Measurements were made in two headwater catchments at $\frac{1}{8}$, $\frac{1}{4}$ and $\frac{1}{2}$ of the distance between adjacent drainage ditches at four control-restored paired sites, during the growing seasons pre- (2012) and post- (2014, 2016 & 2018) restoration.

Restoration had a small but significant ($p = 0.009$) effect on water table depths however, this did not result in a significant change in vegetation composition ($p > 0.350$). Ecosystem respiration increased in both the control and restored locations following restoration however, this increase was significantly smaller ($p = 0.010$) at the restored locations, possibly due to a similarly reduced increase in photosynthesis, although this change was not significant ($p = 0.116$). Below-ground respiration showed no significant changes following restoration.

This research illustrates how degraded these shallow peatlands are, and raises concerns that ditch blocking alone may not bring about the high and stable water tables required to perturb the existing *Molinia caerulea*-dominated ecosystem and substantially alter the carbon balance. Additional restoration measures may be required.

1. Introduction

As local and national governments worldwide declare a “climate emergency” (e.g. House of Commons Canada, 2019; Senado Argentina, 2019; Welsh Government 2019) people are searching for solutions. Peatlands have long been recognised as large carbon stores (Gorham 1991) and more recently as increasing important carbon sinks (Yu et al. 2011), with naturally functioning peatlands accumulating carbon over time due to a small productivity excess (Clymo 1984). However, degraded peatlands commonly act as carbon sources and may globally

emit more than 1.2 GtC annually (Joosten 2010), further exacerbating the “climate emergency”. Wetland rewetting has been recognised as an activity (United Nations Framework Convention on Climate Change 2012) that can reduce carbon emissions and potentially sequester carbon.

Across Europe, degraded upland and lowland *Sphagnum* peatlands have been encroached by graminoid, *Molinia caerulea*, as a result of nutrient deposition, climate change and/or poor management practices. This has altered their carbon cycling (Leroy et al. 2019, Leroy et al., 2017), potentially shifting these peatlands from carbon sinks to carbon

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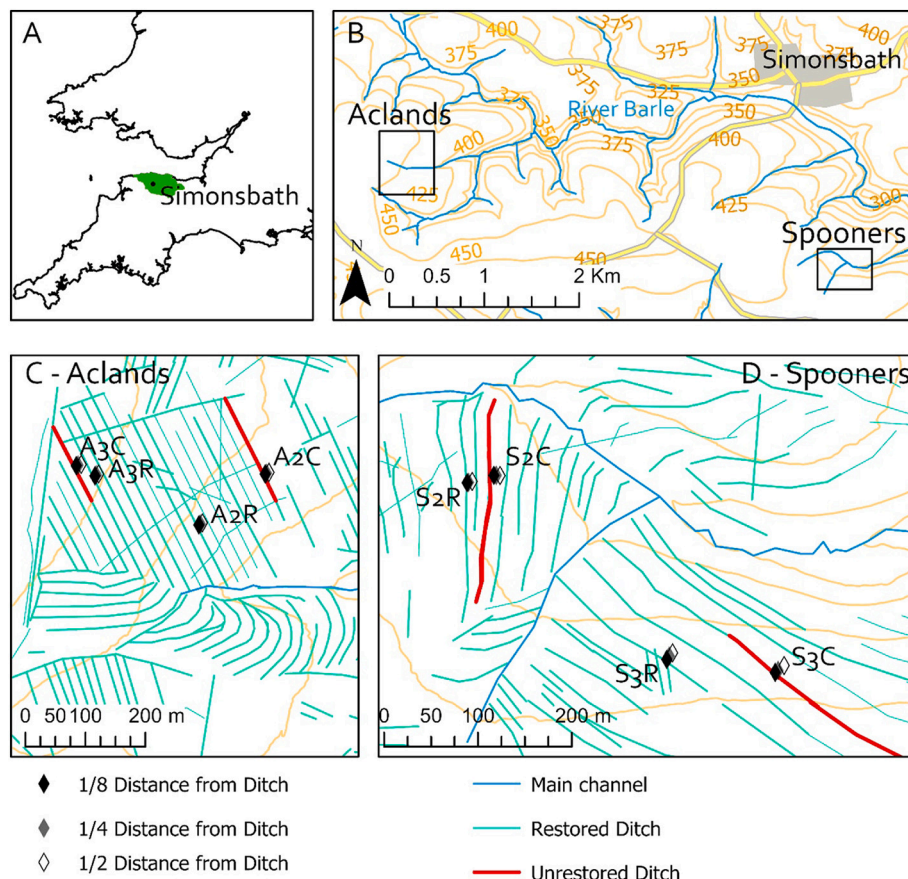


Fig. 1. Location of Exmoor National Park within the southwest of England (A). Location of Aclands and Spooners catchments within Exmoor (B). Location of control (C) and restored (R) study sites within Aclands (C) and Spooners (D) study catchments. Shapefiles (Strategi, 2012; Terrain, 2016).

sources (Gatis et al. 2019; Nieveen et al. 1998). In England it is estimated that 440 km² of blanket bog now has $\geq 50\%$ *Molinia caerulea* cover with higher cover occurring in the south west of England and South Pennines (Glaves 2016).

Although a range of restoration techniques are used to ameliorate different forms of degradation (Parry et al. 2014), there are two main aims of ombrotrophic bog restoration; 1, the re-establishment of near- or at-surface water tables and 2, the re-colonisation of peat forming species especially *Sphagnum* (Holden et al. 2004). Re-establishing *Sphagnum*-rich vegetation communities is particularly important when carbon sequestration is a key aim of restoration (Lunt et al. 2010).

The temporal and spatial effects of restoration on CO₂ fluxes can be better understood by using a combination of gas flux chambers and soil collars to measure both ecosystem and partitioned below-ground fluxes at a scale that is attributable to processes (Tuittila et al. 1999). Partitioning below-ground respiration enables the separation of the more variable autotrophic respiration, anticipated to respond slowly as vegetation composition changes, from heterotrophic respiration associated with decomposition of the peat store. At present very little is known about the short-term effects of restoration on below-ground heterotrophic respiration even though conserving the existing peat store is the first step towards reinstating carbon sequestration. Re-establishing a *Sphagnum*-rich vegetation community can take decades (Haapalehto et al. 2011) however, measuring photosynthesis, ecosystem respiration and below-ground autotrophic respiration should reveal the short-term (1–5 year) effects of restoration on *Molinia caerulea* and the processes driving these changes.

Research is needed to understand the direction of the response of CO₂ fluxes following restoration (Ballantyne et al. 2014) and to indicate if restoration weakens *Molinia caerulea*, a species adapted to live where

water table depths fluctuate (Jefferies 1915), and therefore whether future vegetation change is probable in *Molinia caerulea* dominated peatlands.

Studies on the effect of ditch blocking on GHG fluxes have focused on areas representative of different microtopography and vegetation cover (Komulainen et al. 1999; Urbanová et al. 2012) or fluxes from the ditches themselves (Green et al. 2018). Where the spatially distributed response of CO₂ fluxes has been investigated (Clay et al. 2012) this was from erosional gullies which were much deeper and wider than those typically cut for drainage, had exposed bare peat prior to restoration and required different restoration techniques (e.g. seeding and liming, heather brush cover) therefore they provide a poor comparison with the work proposed here. Research is needed to understand the response of CO₂ fluxes to ditch blocking in *Molinia caerulea* dominated peatlands.

To investigate the potential for restoration in *Molinia caerulea* dominated peatlands we measured changes in water table depth, vegetation composition and CO₂ fluxes in control/restored pairs subject to ditch blocking. It was hypothesised that ecohydrological restoration would raise water tables and reduce all CO₂ fluxes measured (ecosystem respiration; gross photosynthesis; total below-ground respiration; heterotrophic respiration of soil organic matter and below-ground autotrophic respiration including root respiration and microbial respiration of root exudates). More specifically it was hypothesised that:

1. Restoration would raise water tables with the greatest effect observed closest to the ditch nearer the pools formed upslope of the peat dams (Laine et al. 2011; Wilson et al. 2010).
2. Restoration would increase the presence of non-*Molinia caerulea* species due to increased wetness (Bellamy et al. 2012; Haapalehto et al. 2017).

3. Restoration would reduce CO₂ fluxes with the greatest effect observed;
 - a. on ecosystem respiration and below-ground heterotrophic restoration as raised water tables will reduce decomposition rates, shifting the ecosystem towards carbon sequestration (Silvola et al. 1996; Urbanová et al. 2012),
 - b. closest to the ditch, controlled by water table depth, primary productivity and vegetation composition.
4. Restoration would increase CH₄ fluxes with the greatest effect observed closest to the ditch, controlled by water table depth, primary productivity and vegetation composition (Cooper et al. 2014; McNamara et al. 2008; Urbanová et al. 2012).

These hypotheses were tested in two drained, temperate maritime peatlands in the United Kingdom that had been restored by ditch blocking using peat dams.

2. Materials and methods

2.1. Study sites

The study sites (Fig. 1) were located in Exmoor National Park in the southwest England in two *Molinia caerulea* dominated headwater catchments subject to drainage (Aclands 51°7'51.3 N 3°48'44.4 W and Spooners 51°7'21.9 N, 3°44'52.9 W). These catchments were selected to be characteristic of the widespread drained peatland areas which are currently being restored across Exmoor. Restoration occurred in March and April 2013 at Spooners and March 2014 at Aclands. Restoration involved taking a small amount of peat from adjacent to the ditch and placing it in the ditch to create a peat dam (Grand-Clement et al. 2015). Dams were spaced along ditches so that the pond formed behind a dam would reach to the base of the next dam (i.e. closer spaced in steeper areas). In particularly steeper/wider drainage ditches, wood may also have been used (not on the studied ditches). For further details on the study sites see (Gatis 2015; Gatis et al. 2019, Gatis et al., 2015).

2.2. Experimental design

Within each catchment, two pairs of sites were chosen to encompass the expected variation in altitude, aspect, slope, peat depth, ditch dimensions and initial wetness, while minimising the variation between the paired replicates (Table 1). In each pair, one site remained un-restored to act as a control (C) whilst the other site was restored through ditch blocking (R) with peat dams. As ecosystem respiration exhibits significant climate driven inter-annual variability (Lafleur et al. 2003), comparison with control sites is essential for the effects of restoration to be quantified. Investigations into the effects of drainage ditches on water table depths commonly use transects perpendicular to the ditch (e.g. Wilson et al. 2010). As the ditches in these sites were

largely unevenly spaced (Fig. 1), plots were located 1/8, 1/4, and 1/2 of the distance between the ditch being monitored and the adjacent ditch to evaluate the spatial effect of restoration on CO₂ fluxes across all the peatland. This meant plots 1/8 of the distance between adjacent ditches ranged from 1.5 to 4 m from the ditch, 1/4 ranged from 3.1 to 8.2 m and 1/2 ranged from 6.3 to 16.4 m (Table 1).

2.3. Measurements and data analysis

2.3.1. Photosynthesis and ecosystem respiration measurements

A 55 cm × 55 cm × 25 cm Perspex gas flux chamber was rested on permanently installed 50 cm tall legs with a plastic skirt weighted down to the soil surface (Gatis et al. 2019, Gatis et al., 2015). An EGM-4 infrared gas analyser (PP Systems, Hitchin, UK) measured accumulated CO₂ every 10 s for 2 min concurrently with chamber temperature, photosynthetic active radiation (PAR) (Skye Instruments, Llandrindod Wells, UK), soil temperature at a depth of 5 cm (Electronic Temperature Instruments, Worthing) and the water table depth below the peat surface (manual measurement in a perforated tube).

Measurements were made approximately monthly over the *Molinia caerulea* growing seasons (May to September) of 2012, 2014 and 2018. Measurements were taken at the R and C pairs on the same day whenever possible to minimise variation in environmental conditions. Net CO₂ exchange (μg m⁻² s⁻¹) was calculated from the linear change in CO₂ concentration in the chamber at 100, ~60–40, ~10 and 0% light levels using a combination of shade cloths. Linear regressions with r² of < 0.7 were either excluded or allocated a zero flux. Temperature and pressure were not measured as part of the baseline data (2012) so for consistency were assumed to be 273 K and 100 kPa throughout. Comparisons with temperature and pressure corrected data showed a mean difference of 3.1 ± 2.7 μg m⁻² s⁻¹ similar to the level of detection, 3.4 to 4.4 μg m⁻² s⁻¹, dependent on the chamber volume. Ecosystem respiration (R_{Eco}) and photosynthesis at 1000 μmol Photons m⁻² s⁻¹, light saturated conditions, were calculated from light response curves measured at each R and C site (each site having plots at three distances from the ditch) during 10 separate campaigns.

To account for climatic variability (e.g. temperature, precipitation etc.) between the baseline year and subsequent years, P_{G1000} and R_{Eco} at the restored locations was divided by equivalent values measured at the paired control locations (P_{G1000}R/C and R_{Eco}R/C).

2.3.2. Methane

A Perspex collar 30 cm diameter, 10 cm tall was inserted at each location 6 cm into the peat in April 2015. A 30 cm diameter 50 cm tall chamber was attached to the collar using a collar adaptor consisting of a Polyvinyl Chloride (PVC) ring with a groove fitting on one side and a gas-tight foam seal on the other. The chamber was this height to allow for the expected mid-summer vegetation height of the *M. caerulea*. A Los Gatos Ultra-Portable Greenhouse Gas Analyser (San Jose, California,

Table 1
Experimental site properties; peat depth, ditch width, depth, direction and spacing, altitude, slope and aspect.

Site	Mean peat depth ¹ (cm)	Mean ditch width ³ (cm)	Mean ditch depth ² (cm)	Distance from ditch to adjacent ditch ³ (m)	Altitude ⁴ (m)	Slope ⁴ (°)	Aspect	Direction of ditch ⁴ (°)	Ditch direction w.r.t slope
S2R	71	84	42	12.5	397	5	NE	2	Down slope
S2C	56	80	45	15.4	395	6		0	
S3R	29	38	24	29.3	405	5	N	302	Down-slope
S3C	29	42	24	32.8	407	5		300	
A2R	40	40	26	19.1	446	4	SE	154	Down slope
A2C	43	43	21	19.1	448	4		150	
A3R	30	50	18	18.3	463	3	SE	144	Down slope
A3C	36	42	18	18.7	461	3		151	

¹ Measured during dipwell installation from base of peat (n = 3).

² Measured from base of ditch to tussock shoulder.

³ Measured using tape measure.

⁴ From LiDAR.

USA) measured variation CH₄ and CO₂ concentrations every 2 s over 2 min. Methane fluxes were calculated from the linear change in CH₄ concentration in the chamber. Linear regressions with r^2 of < 0.7 were either excluded or allocated a zero flux if the change in concentration was < 0.01 $\mu\text{gC m}^{-2} \text{s}^{-1}$, the level of detection. Measurements were collected approximately monthly over the 2016 growing season. Soil temperature and water table depth were measured as above. CO₂ measurements are not reported here due to the different sampling footprint and therefore vegetation present compared to sampling in other years.

2.3.3. Below-ground respiration measurements

At each location, four PVC collars (16 cm diameter, 8 cm height) were sealed to the peat using non-setting putty (Evo-Stik 'Plumbers Mait') in March 2012. All collars had above-ground vegetation removed by clipping, following CO₂ exchange measurement, so they measured below-ground fluxes only. In addition, circular 20 cm deep and 56 cm diameter trenches were dug around half the collars. Thus, live roots were excluded enabling below-ground heterotrophic respiration to be measured. The collars with only above-ground vegetation removed were used to measure total below-ground respiration (including autotrophic and heterotrophic components). CO₂ measurements were taken approximately monthly between May and September within as few days as possible of the NEE measurements (section 2.3.1). CO₂ flux was measured over 2 min using an EGM-4 infra-red gas analyser and a CPY-4 canopy assimilation chamber (PP Systems, Hitchin, UK). At each plot, the two replicates of each treatment were averaged to produce a single value for total and heterotrophic respiration. Autotrophic respiration was then calculated from the difference between total and heterotrophic respiration.

To account for climatic variability the ratio of respiration at the restored location to that at the paired control location was calculated ($R_{\text{TotR/C}}$, $R_{\text{HetR/C}}$ and $R_{\text{AutR/C}}$).

2.3.4. Vegetation composition and annual net primary productivity

Annual net primary productivity (ANPP) was measured in late July/August 2012, 2014, 2016 and 2018 by destructive harvest of a 55 × 55 cm plot < 4 m down-slope of each flux measurement plot ($n = 24$).

Vegetation composition including leaf litter (% cover) of each NEE plot was estimated by visual inspection in August 2012, 2014 and 2018. As several species occurred at only one location, the percentage coverage of non-*Molinia* species was calculated. The number of species present at each location was counted to derive the species richness for each location. Ellenberg's Moisture Indicator Values (Hill et al. 1999) were determined for each location. Vascular species have been classified according to their ecological niche on a 12-point scale ranging from 1 (extreme dryness) to 12 (submerged plants). The classification values for the vascular species identified in this study were looked up and the average value for the species present at each location was weighted by species coverage.

2.3.5. Ancillary data

Water table depths were measured in dipwells installed to the base of the peat soil profile. Measurements were taken concurrently with both NEE and soil efflux measurements. Again, to account for climatic variability between years the water table depth at the restored location was normalised by dividing by the water table depth at the paired control location (WTD_{R/C}).

Rainfall data were collected every 15 min using a 0.2 mm tipping bucket rain gauge and a NOMAD Portable Weather station (Casella, USA) in each catchment. Rainfall was summed for the preceding 0, 7, 14 and 28 days to describe antecedent conditions prior to sampling. A previously derived (Gatis et al. 2019) linear relationship ($r^2 = 0.87$) between instantaneous Photosynthetic Photon Flux Density (PPFD) measurements and global irradiation at Liscombe Met Station (15 km

away) (UK Meteorological Office, 2014) was used to derive total PPFD in the preceding hour and day. Soil temperature was measured at a depth of 15 cm every 15 min (Gemini Data Loggers, Chichister, UK) at the location ¼ distance between ditches, from this soil temperature (maximum and mean) in the preceding hour and day was determined.

2.3.6. Statistical analysis

Repeated measures (yearly average due to varying sample numbers) ANOVA was carried out on WTD, ANPP, P_{G1000} , R_{Eco} , R_{Tot} , R_{Het} , R_{Aut} with proportional distance from the ditch, year and restoration as factors. Prior to analysis WTD, ANPP, R_{Tot} , R_{Het} and R_{Aut} were square root transformed to obtain normality.

To account for climatic variability the ratio of the measured parameter at the restored location to its paired control location was also analysed. Repeated measures (yearly average due to varying sample numbers) ANOVA was carried out on WTD_{R/C}, ANPP_{R/C}, $P_{\text{G1000R/C}}$, $R_{\text{EcoR/C}}$, $R_{\text{TotR/C}}$, $R_{\text{HetR/C}}$ and $R_{\text{AutR/C}}$ with year and proportional distance from the ditch as factors. Prior to analysis WTD_{R/C} was power (-2.1) transformed, Richness_{R/C}, $R_{\text{EcoR/C}}$, $R_{\text{TotR/C}}$ and $R_{\text{HetR/C}}$ were natural logarithm transformed, $P_{\text{G1000R/C}}$ and ANPP_{R/C} square root transformed $R_{\text{AutR/C}}$ sin transformed to obtain normality. A non-parametric Wilcoxon signed ranks test was carried out to test for differences in paired (2012–2018) Eilenberg's moisture index R/C data.

A stepwise multiple regression was carried out to test for significant relationships between CO₂ ($P_{\text{G1000R/C}}$, $R_{\text{EcoR/C}}$, $R_{\text{TotR/C}}$, $R_{\text{HetR/C}}$ and $R_{\text{AutR/C}}$) and CH₄ fluxes and independent variables (WTD, soil temperature, PPFD and soil temperature (max and mean) in the preceding hour and day and total rainfall in the preceding 1, 7, 14 and 28 days) pre- and post-restoration. To test for differences in the relationship between CO₂ or CH₄ fluxes and variables pre- and post-restoration a sum-of-squares F-test was carried out for identified significant relationships.

A related samples Wilcoxon signed rank test was used to test for variation in CH₄ fluxes at control and restored sites post-restoration. A related sample Friedman's two-way analysis of variance by ranks test was used to test for spatial variation in CH₄ fluxes. All statistical analyses were performed with SPSS 19 (SPSS Inc., Chicago, Illinois, US).

3. Results

3.1. The effect of restoration on water table depths

The regional (southwest England) seasonal rainfall for the hydrological summer (June, July, August) of 2012 was 234 mm greater than the 20-year average, whilst the summers of 2014, 2016 and 2018 were 4, 17 and 243 mm drier than the long-term average, respectively (UK Meteorological Office 2019). The maximum summer temperature of 2012 was also 1.1 °C cooler than average and 2018 notably warmer (2.3 °C than average). Due to climatic variation, a cool wet baseline year (2012) compared to warmer drier post-restoration years, water tables fell following restoration (Fig. 2).

Water tables varied significantly between years ($p < 0.001$) with 2012 the shallowest (wettest) year and 2014 and 2018 the deepest (driest) (Fig. 2). Water tables were significantly deeper ($p = 0.045$) in the restored locations both before and after restoration. Looking at individual years water tables were statistically shallower in the control locations in 2012 and again in 2016. Once climatic variability had been accounted for by comparing the ratio of water table depths at the restored location to their paired control locations, WTD_{R/C} was significantly greater in 2012 (pre-restoration) compared to all other years (Fig. 3A), with the most obvious difference in 2014. Water tables rose in the restored locations compared to the control locations following restoration as hypothesised (hypothesis 1). However, the change in estimated marginal means of WTD_{R/C} was small, from 1.5 (95% confidence interval of 1.2 to 2.1) in 2012 to 1.2 (95% confidence interval 1.0 to 1.4) in 2014 with considerable spatial variability.

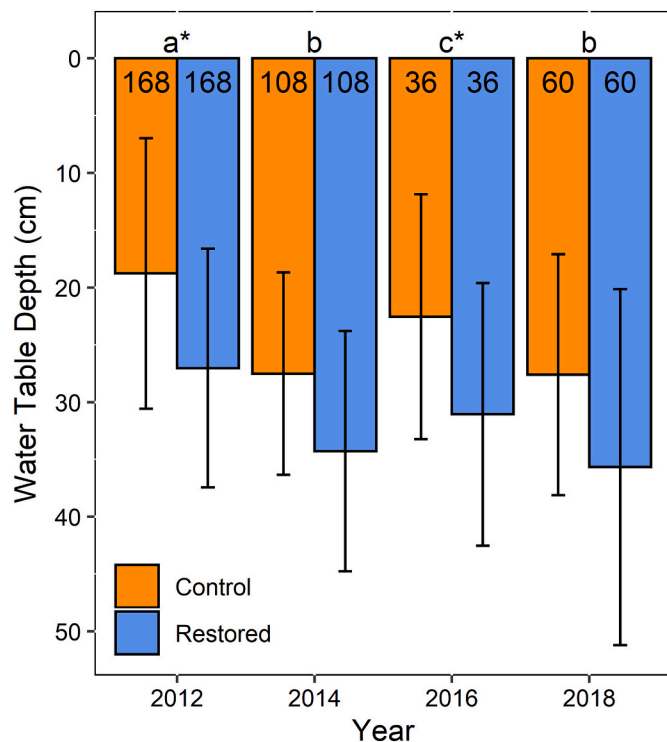


Fig. 2. Mean water table depth (cm) at the control and restored locations over time. Number of measurements are indicated; error bars are 1 standard deviation. Letters a,b,c denote statistically different years, * denotes statistical differences between control and restored locations within a year.

Prior to restoration water tables were deeper nearer to the ditch, but not significantly so (Gatis et al. 2015). Following restoration WTD_{R/C} nearer the ditch was closer to 1 (restored locations equal to control locations) when compared to further away from the ditch (supplementary material Fig. 1) however, this difference was not significant (Table 2). There was also no distance-year interaction which might have been expected if the effects of restoration had been stronger nearer to the ditch.

3.2. The effect of restoration on vegetation composition

ANPP was significantly ($p = 0.002$) greater in 2018 ($551 \pm 130 \text{ g m}^{-2}$) than 2012 ($290 \pm 63 \text{ g m}^{-2}$) and 2016 ($415 \pm 217 \text{ g m}^{-2}$) with 2014 ($435 \pm 179 \text{ g m}^{-2}$) intermediate. However, repeated measures ANOVA indicates no significant difference (Table 3) in ANPP_{R/C} between pre- (2012) and post-restoration (2014, 2016, 2018) years. Other vegetation indices also showed no significant change between pre- (2012) and post-restoration (2018) (Table 3) contrary to hypothesis 2. Although the ratio of *Molinia caerulea* coverage at the restored location to the control pair was significantly greater ($p = 0.037$) nearer the ditch, there was no distance-year interaction suggesting this pattern was pre-existing and not effected by restoration.

3.3. The effect of restoration on carbon dioxide fluxes

Post-restoration (2014 and 2018) P_{G1000} and R_{Eco} fluxes were significantly ($p \leq 0.001$) larger than pre-restoration (2012) at both the control and restored locations (Fig. 5A & B). This coincides with greater ANPP values for 2012 and 2018 potentially reflecting better growing conditions.

There were no significant differences in P_{G1000} and R_{Eco} fluxes between control and restored locations within years. However, for R_{Eco} the interaction between restoration and year was marginally non-

significant ($p = 0.052$) (full ANOVA results in supplementary material Table 1) with R_{Eco} showing a greater increase at the control locations between 2012 and 2016 than at the restored locations (Fig. 5A).

Similar to water table depths, climatic variation i.e. the variation between years within the control locations, was larger than variation due to restoration (Fig. 4). Climatic variation can be accounted for by looking at the difference in fluxes at the control-restored pairs. $R_{EcoR/C}$ and $P_{G1000R/C}$ both show a decrease between 2012 and subsequent years (Fig. 3C and E) potentially supporting hypothesis 3. However, this difference is only significant ($p = 0.010$) for $R_{EcoR/C}$ (Table 2), with R_{Eco} increasing in restored locations less than the control locations (Fig. 4A) resulting in a decrease in $R_{EcoR/C}$. There is no obvious spatial pattern in the response of $R_{EcoR/C}$ or $P_{G1000R/C}$ to restoration (supplementary material Fig. 1) and neither $R_{EcoR/C}$ nor $P_{G1000R/C}$ show a significant year-distance from the ditch interaction (Table 2).

Below-ground respiration varied significantly between years (Fig. 5) with the pattern varying between the different respiration sources. Although total respiration was significantly greater ($p = 0.040$) at the control locations overall, and appeared higher for all years this was only significant in 2016 ($p = 0.020$). Contrary to this heterotrophic and autotrophic respiration were not significantly different between control and restored locations overall ($p = 0.184$ and $p = 0.185$ respectively) but showed significant differences within 2012 ($p = 0.015$) and 2016 ($p = 0.016$) respectively (Fig. 5).

Heterotrophic respiration had a significant year-restoration interaction ($p = 0.012$) with respiration decreasing more rapidly in the control locations between 2012 and 2016 (Fig. 5). Conversely autotrophic respiration increased more rapidly between 2012 and 2018 at the control locations, this year-restoration interaction was marginally non-significant ($p = 0.094$).

Allowing for climatic variability by looking at the ratio of respiration at the restored locations to the control locations (Fig. 3B, D & F), below-ground respiration was not significantly different pre- (2012) and post-restoration (2014 onwards). $R_{HetR/C}$ and $R_{AutR/C}$ both appear to show increasing trends and $R_{TotR/C}$ a decreasing trend however, only $R_{HetR/C}$ was significantly different between 2016 and the other years.

None of the below-ground respiration sources showed any significant spatial patterns with increasing distance from the ditch nor any distance-year interactions (Table 2, supplementary material Fig. 1 and Table 1).

Investigating the effects of restoration on the relationship between drivers and gas fluxes indicates variation in photosynthesis ($P_{G1000R/C}$) is most strongly related to PPFD in the preceding hour both pre- ($r^2 = 0.15$, $p = 0.021$) and post-restoration ($r^2 = 0.13$, $p = 0.002$) with a significant difference in the relationship (sum-of-squares F-test) pre- and post-restoration ($p = 0.001$). The $P_{G1000R/C}$ - PPFD relationship had a steeper slope (0.002 pre- and 0.001 post-) but smaller intercept (0.026 pre- and 0.550 post-) pre-restoration compared to post-restoration. However, if 2 outliers ($P_{G1000R/C} > 5$) are removed then there is no significant change post-restoration ($p = 0.107$).

Of all the variables tested, pre-restoration variation in ecosystem respiration ($R_{EcoR/C}$) was only significantly related to $P_{G1000R/C}$ ($r^2 = 0.33$, $p < 0.001$) with $R_{EcoR/C}$ increasing as $P_{G1000R/C}$ increased. Following restoration $R_{EcoR/C}$ was still significantly related to $P_{G1000R/C}$ ($r^2 = 0.32$, $p < 0.001$) with no significant change in the relationship between $R_{EcoR/C}$ and $P_{G1000R/C}$ ($p = 0.104$) pre- and post-restoration.

Variation in below-ground heterotrophic respiration ($R_{HetR/C}$) had a significantly ($r^2 = 0.15$, $p = 0.007$) positive relationship to soil temperature; $R_{HetR/C}$ increased during warmer weather. However, the $R_{HetR/C}$ -soil temperature relationship was not significantly different ($p = 0.162$) pre- and post-restoration. Below-ground autotrophic respiration ($R_{AutR/C}$) had a significant ($r^2 = 0.14$, $p = 0.009$) positive relationship to PPFD the preceding day; $R_{AutR/C}$ increased following sunnier weather, this relationship was not significantly altered

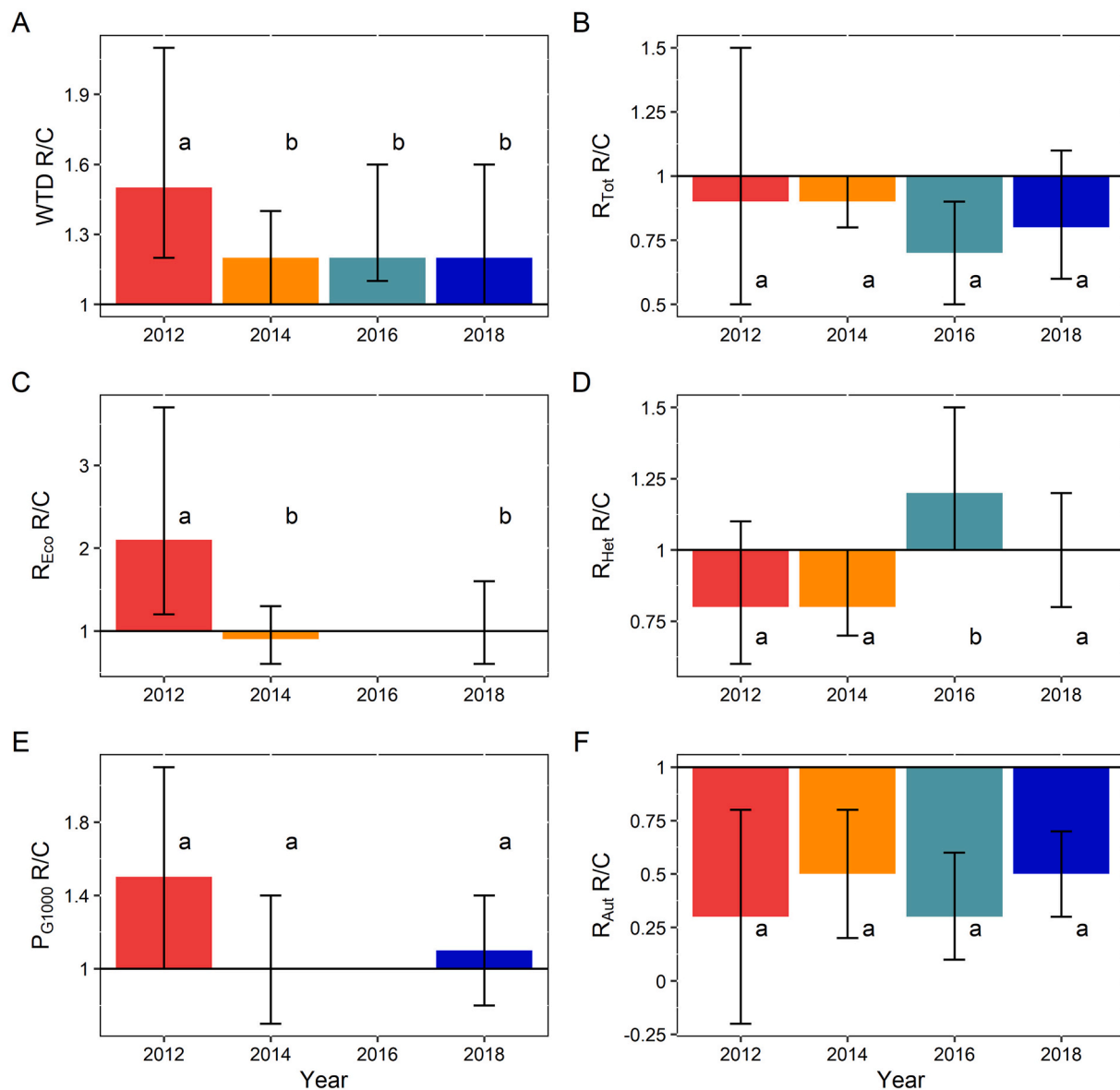


Fig. 3. A, Estimated marginal means for water table depth (WTD R/C), C, ecosystem respiration (R_{Eco} R/C), E, photosynthesis at 1000 $\mu\text{mol Photons m}^{-2} \text{s}^{-1}$ (P_{G1000} R/C), B, total (R_{Tot} R/C), D, heterotrophic (R_{Het} R/C) and F, autotrophic below-ground respiration (R_{Aut} R/C) at the restored location as a proportion of the variable at the controlled location pre- (2012) and post-restoration (2014, 2016 and 2018). Values above 1 indicate depth to water table or CO_2 flux from restored location greater than depth to water table or flux from control location. Significantly different ($p < 0.05$) years are marked by letters. Error bars are 95% confidence intervals, $n = 12$.

($p = 0.418$) by restoration. Total below-ground respiration (R_{Tot} R/C) was not significantly related to any of the variables tested.

3.4. The effect of restoration on methane fluxes

Most methane measurements (213/242) were within the levels of detection ($< 0.01 \mu\text{gC m}^{-2} \text{s}^{-1}$) and therefore allocated a zero-flux value. Once averaged by location and date, 3 results were positive (methanogenesis) reaching a maximum of $0.04 \mu\text{gC m}^{-2} \text{s}^{-1}$ and 35 were negative (methane oxidation) reaching a minimum of $-0.02 \mu\text{gC m}^{-2} \text{s}^{-1}$. It appears (Fig. 6) that methane is oxidised more at the restored locations compared to the control locations contrary to hypothesis 4, however this difference is not significant ($p = 0.071$). There are also no significant differences between locations closer to the ditch than those further away ($p = 0.472$ and $p = 0.105$ for the control or restored locations respectively).

Multiple linear regression indicates methane fluxes were driven by a

combination of PPFd in the preceding 24 h (coefficient = -6.51×10^{-8}) and WTD (coefficient = -1.14×10^{-5}); $r^2 = 0.17$, $p \leq 0.001$ (Table 4). Fluxes were more negative (methane oxidation) following sunnier weather and during drier conditions. No spatial patterns associated with vegetation composition were observed.

4. Discussion

4.1. The effect of restoration on water table depths

Within these shallow peats the effect of drainage ditches on water table depths extended to around 8 m downgradient of a ditch (Luscombe et al. 2016). All the locations $\frac{1}{3}$ and $\frac{1}{4}$ of the distance between ditches (Table 1) lie within this zone and would therefore reasonably be expected to be impacted by restoration. However, restoration had a very small but significant impact on water tables (Fig. 3, Table 2) with considerable spatial variability both between sites

Table 2
Repeated measures (year) ANOVA results with proportional distance (1/8, 1/4, 1/2) from the ditch as a between subject factor.

Variable	Factor	Degrees of freedom	F-ratio	Significance
WTD R/C	Year	2.12	5.927	0.009*
	Year x distance	4.24	1.034	0.418
	Distance	2.00	1.776	0.224
R _{Eco} R/C	Year	1.46	7.668	0.010*
	Year x distance	2.92	0.418	0.738
	Distance	2.00	0.145	0.867
P _{G1000} R/C	Year	1.69	2.560	0.116
	Year x distance	3.38	0.271	0.866
	Distance	2.00	0.034	0.966
R _{Tot} R/C	Year	2.11	1.468	0.256
	Year x distance	4.22	1.772	0.174
	Distance	2.00	1.339	0.310
R _{Het} R/C	Year	1.69	3.911	0.048*
	Year x distance	3.39	1.949	0.160
	Distance	2.00	0.828	0.468
R _{Aut} R/C	Year	1.57	0.543	0.551
	Year x distance	3.14	0.706	0.570
	Distance	2.00	0.145	0.867

* Significant $p < 0.050$.

and with increasing distance from the ditch.

It has been suggested restoration may have a limited effect on water tables due to subsidence of the drained peat near to the ditch (Dolan et al. 2017). In their study although the water table was lowered by drainage (compared to a datum), it remained relatively shallow compared to the ground surface and at a similar depth to those observed in functioning blanket bogs (Holden et al. 2017, Holden et al., 2011; Wilson et al. 2010) reducing the scope for restoration to raise water levels. This explanation is unlikely in this case as pre-restoration water tables were not near the ground surface across these catchments (Luscombe et al. 2016).

Daniels et al. (2008) found for a drained peatland, where the pre-restoration water tables fluctuated over 0.8 m that water tables were only raised to within 0.05 m of the surface by the largest of storms or following particularly wet antecedent conditions and that water tables rapidly fell following rainfall events. They suggest this was due to rapid throughflow occurring through a subsurface macropore/pipe network formed by regular drying. It is probably that a similar mechanism is occurring in our shallow, degraded peatland sites. In addition, the dams were made from this dried-out peat and, despite compaction by the digger, are likely to allow throughflow. Further supporting this theory, pools were only observed behind the dams in the wettest conditions and rapidly dried out following rainfall. Although continuous monitoring of different ditches within the same catchments found restoration to have raised water tables in some areas (Luscombe, pers. Comm.) the effects were highly temporally and spatially variable.

Table 3
Vegetation indices for control ($n = 12$) and restored ($n = 12$) pairs in 2012 (pre-restoration) and 2018 (post-restoration).

Year	Treatment	<i>Molinia caerulea</i> coverage (%)	non- <i>Molinia caerulea</i> coverage (%)	Species richness	ANPP (g m^{-2})	Ellenberg's moisture indicator value
2012	C	85 (4)	7 (2)	2.9 (0.5)	290 (63)	8.0 (0.0)
	R	72 (3)	6 (3)	2.4 (0.4)	338 (72)	7.9 (0.1)
2014	C				435 (179)	
	R				447 (149)	
2016	C				415 (217)	
	R				410 (120)	
2018	C	85 (4)	33 (7)	3.2 (0.5)	515 (130)	7.9 (0.0)
	R	75 (3)	24 (4)	3.8 (0.5)	586 (285)	7.7 (0.1)
Year ^a		0.546	0.580	0.350	0.901	0.812
Distance ^a		0.037*	0.356	0.425	0.604	
Year x distance ^a		0.517	0.151	0.192	0.991	

Standard deviation shown in brackets.

* Significant $p < 0.050$.

^a Significance values of repeated measures ANOVA or paired Wilcoxon signed rank test for Ellenberg's moisture indicator value.

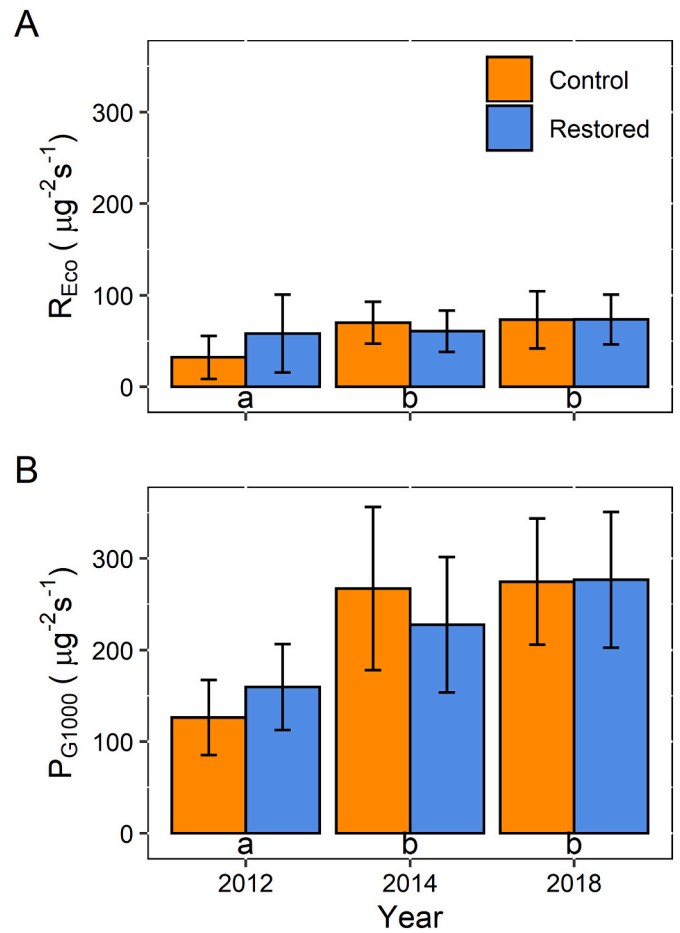


Fig. 4. Average ecosystem respiration (a) and photosynthesis at 1000 $\mu\text{mol Photons m}^{-2} \text{s}^{-1}$ ($\mu\text{gC m}^{-2} \text{s}^{-1}$) (b) at the control and restored locations for each monitored growing season. Error bars are 1 standard deviation. Letters denote statistically different years. There are no statistical differences between control and restored locations within years. Average of location yearly average ($n = 12$).

Water table depth measurements were biased towards the drier conditions required for gas flux monitoring. It may be that restoration had a greater impact on water tables during and shortly after rainfall events, and the timing of sampling failed to fully capture this effect. Consequently, this study does not completely represent the temporal dynamics of hydrology and gaseous carbon exchange at the site, further study including during rainfall events is required. However, it has been

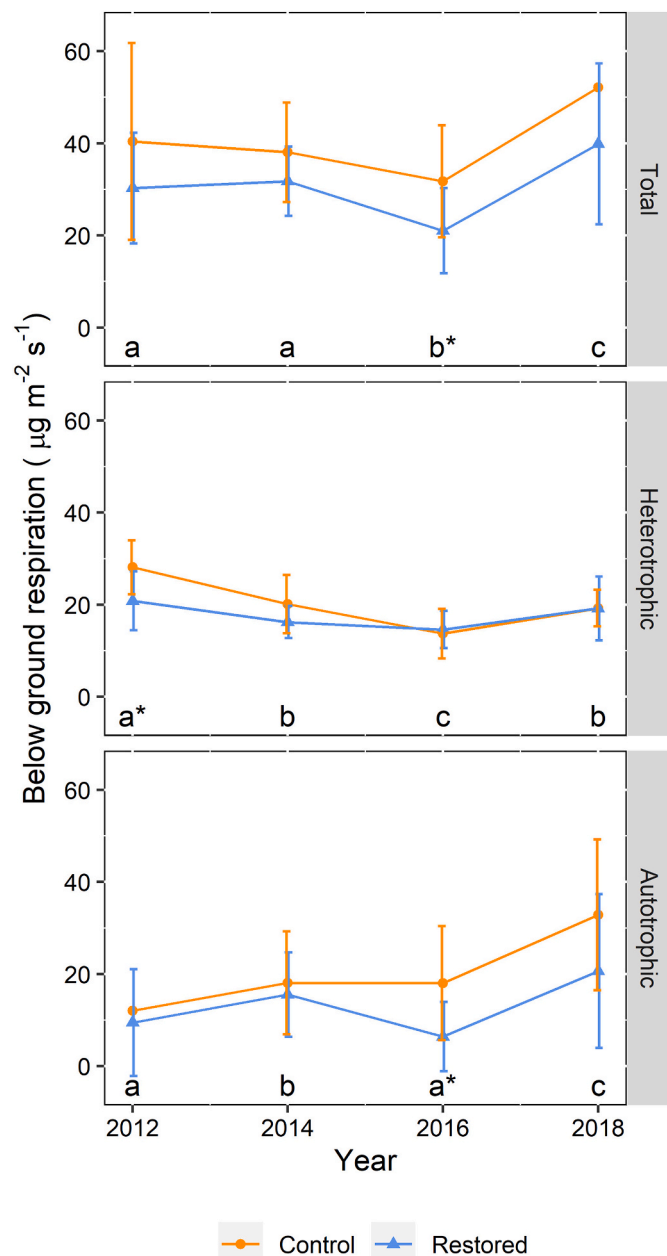


Fig. 5. Average total, heterotrophic and autotrophic below-ground respiration ($\mu\text{g C m}^{-2} \text{s}^{-1}$) at the control and restored locations for each monitored growing season. Error bars are 1 standard deviation. Letters denote statistically different years, * denote statistical differences between control and restored locations within a year.

argued that for a significant change in vegetation community to occur water tables must be at or near the peat surface (Holden et al. 2004; Schouwenaars 1993). Restoration by peat dams has clearly not created these conditions across the studied sites. To persistently raise water tables, impermeable/very low permeability barriers would be required regularly throughout the peat, however, these are unlikely to be either economically or socially acceptable.

4.2. The effect of restoration on vegetation composition

Six years is a relatively short time to monitor for vegetation change. Vegetation surveys across Exmoor suggest a significant increase in *Sphagnum* cover does not occur for at least 7 years post-restoration and even then, not in all locations (Hand 2019). More widely studies of

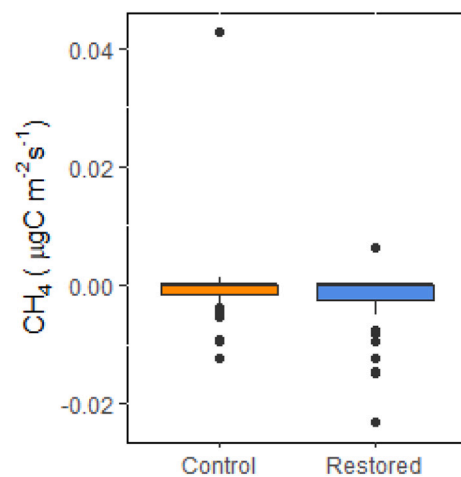


Fig. 6. Box and whisker plot of methane flux (CH_4) ($\mu\text{g C m}^{-2} \text{s}^{-1}$) at the control and restored locations. A negative flux represents methane drawdown from the atmosphere. The box stretches from the 25th to the 75th percentile, with a horizontal bar at the 50th percentile. The whiskers stretch to the largest and smallest values within 1.5 times the interquartile range. Dots show outlying measurements.

similar or shorter length have observed change (Haapalehto et al. 2011; Jauhainen et al. 2002; Komulainen et al. 1999), minimal change (Dolan et al. 2017; Green et al. 2017) or no change (Urbanová et al. 2012). No significant change in vegetation composition (Table 2) was observed in this study contrary to hypothesis 2.

To initiate a return to a *Calluna vulgaris* dominated ecosystem containing *Sphagnum*, which was previously present on Exmoor (Chambers et al. 1999), it has been suggested that the water table must be maintained near the peat surface without great fluctuation (Holden et al. 2004; Schouwenaars 1993). *Eriophorum* sp. have been shown to flourish and *Molinia caerulea* decline when water stagnates (Jefferies 1915), requiring high water tables and slow flow. Water table depths measured at the restored locations in this study still reached as deep as 1.01 m post-restoration (at S2R1 on 19/07/2018). If water tables are insufficiently high and stable following restoration it may be that *Molinia caerulea* adapts its morphology (root distribution and tussock height) to suit the new water table depth conditions (Rutter 1955) rather than being outcompeted by other species. It appears that ditch blocking alone will not perturb the existing *Molinia caerulea* ecosystem sufficiently to reduce its dominance. Additional restoration techniques, such as impermeable bunds, mowing or *Sphagnum* re-seeding, may be required.

4.3. The effect of restoration on carbon dioxide fluxes

The relative difference in ecosystem respiration between the control and restored locations significantly declined (i.e. decrease in $R_{\text{EcoR/C}}$) following restoration (Fig. 3C, Table 2). The wetter conditions pre-restoration may have caused a reduction in photosynthesis and in turn ecosystem respiration (Gatis et al. 2019) however, the observed reduction in $P_{\text{G1000R/C}}$ (Fig. 3E, Table 2) was not significant. Considerable spatial and temporal variability in CO_2 fluxes could have masked any effects of restoration, suggesting a much larger sample size would be required to confidently observe effects within these ecosystems. Alternatively these results (Table 2) may reflect the insubstantial effect restoration has had due to a minimal change in water tables (Fig. 2) and no significant change vegetation composition (Table 3). Much of the observed (spatial) variation in CO_2 fluxes most likely reflects the individual locations response to climatic conditions due to different initial conditions (wetness and vegetation composition) as the relationships between $R_{\text{EcoR/C}}$ and $P_{\text{G1000R/C}}$, $R_{\text{HetR/C}}$ and soil temperature and

Table 4

Parameters with a significant linear regression with methane fluxes. Variables tested were PPFd concurrent with flux measurement, in the last hour and past day (PPFD day), water table depth (WTD) concurrent with measurement, rain in the last hour, 1 (Rain 1 day), 7, 21 (Rain 21 days) and 28 (Rain 28 days) days, NDVI, soil temperature concurrent with flux measurements and average and maximum soil temperature in the last hour and day.

Parameter	R ²	Sig.	F	Constant	Coefficient 1	Coefficient 2
PPFD day	0.10	0.001	11.370	3.67E-04	-6.43E-08	
WTD	0.06	0.011	6.728	1.78E-04	-1.13E-05	
Rain 21 days	0.05	0.024	5.288	2.57E-04	-3.98E-06	
Rain 28 days	0.05	0.026	5.107	3.74E-04	-3.78E-06	
Rain 1 day	0.04	0.046	4.079	-1.70E-04	4.09E-05	
PPFD day & WTD	0.17	< 0.001	9.769	6.67E-04	-6.51E-08	-1.14E-05

R_{Aut}R/C and PPFd on the preceding day did not change pre- and post-restoration.

It is possible that limited effects on CO₂ were observed due to the bias towards drier conditions (unavoidable due to the monitoring equipment used). However, for a significant change in gaseous carbon fluxes to occur as a result of restoration, water tables would have to be consistently different (not just during rainfall events) to enable changes in vegetation and soil microbial communities to occur.

Other published restoration studies have found significant changes in CO₂ associated with a rise in water tables and subsequent changes to the vegetation community present (e.g. Clay et al. 2012; Komulainen et al. 1999; Soini et al. 2010; Strack et al. 2015; Waddington et al. 2010). This reflects either the underreporting of null hypotheses and/or the unusual nature (within the literature) of these shallow *Molinia caerulea* dominated peatlands.

Where net ecosystem exchange has been measured in *Molinia caerulea* dominated peatlands these have consistently been found to be growing season sinks (Gatis et al. 2019; Urbanová et al. 2013) but both CO₂ sources (Nieveen et al. 1998) and sinks (Laggoun-Défarge et al. 2016) over the whole year. Mesocosm experiments found *Sphagnum* and *Molinia caerulea* to be a greater carbon sink than *Sphagnum* alone (Leroy et al. 2019) however, this did not account for the longer-term effects on respiration due to increased labile organic material. It is still unknown how *Molinia caerulea*, a species adapted to survive prolonged periods of waterlogging (Taylor et al. 2001) would respond in the longer-term if water tables were eventually raised and stabilised and whether this would result in the anticipated carbon benefits.

The authors are unaware of any studies reporting the effect of restoration on either heterotrophic or autotrophic below-ground respiration. Therefore, this study is the first to investigate the effects of restoration on partitioned below-ground respiration rates. Given there was limited change in water table depths (Fig. 2) or photosynthesis (Fig. 4) following restoration it is unsurprising that there was also no significant change in below-ground heterotrophic or autotrophic respiration (Fig. 5). Previous work in these catchments had raised concerns that an increase in wetness may increase heterotrophic respiration rates (Gatis et al. 2019) and a non-significant increase in heterotrophic respiration was observed (Fig. 3D). Given reducing heterotrophic respiration and therefore conserving the existing peat store is the first stage in returning a system to carbon sequestration this raises concerns about the efficacy of ditch blocking as a means to protect the existing carbon store in these peatlands.

4.4. The effect of restoration on methane fluxes

Fluxes observed in this study were low and often negative (-0.003 to 0.003 µgC m⁻² s⁻¹) compared to values reported for other blanket bogs in similar maritime climatic conditions; 0.104 to 0.177 µgC m⁻² s⁻¹ (Cooper et al. 2014); 0.114 to 0.146 µgC m⁻² s⁻¹ (Koehler et al. 2011) and undetectable to 0.292 µgC m⁻² s⁻¹ (Chapman and Thurlow 1996). It is unlikely these low emissions are due to the vegetation present as *Molinia caerulea* mesocosms have been found to emit 1.129 µgC m⁻² s⁻¹ (Leroy et al. 2017).

Methane fluxes measured for other sites on Exmoor with varying times since restoration also found both methanogenic and methane oxidation conditions (range of -0.012 to 0.959 µgC m⁻² s⁻¹) (McAleer 2016). A strong association was found between CH₄ fluxes and the presence of vegetation with higher Ellenberg moisture values, especially *Sphagnum* mosses. Additionally lower water tables were found to correlate strongly ($r = 0.43$) with the number of methanogens across a drained, rewet and pristine bog (Urbanová and Bárta 2020). In the drained bog (WTD 20 ± 8 cm) methanogens made up < 5% of the archaeal community present at 0–10 cm depth. This suggests the low and negative CH₄ emissions found in this study could be due to low and fluctuating water levels (Fig. 2) preventing the development of an anaerobic microbial community required for methanogenesis.

The apparent difference in CH₄ fluxes between control and restored sites observed in this study (Fig. 6) most likely reflects pre-existing variation in vegetation composition and wetness conditions rather than an effect of restoration as a concomitant change in water table depths (Fig. 2) or vegetation community (Table 3) required to drive this change was not observed.

5. Conclusion

Undertaken in usual climatic conditions this study found ditch blocks in shallow peats had a limited effect on water tables which did not result in significant changes to the vegetation present or reduce CO₂ fluxes (gross photosynthesis; total below-ground respiration; heterotrophic respiration of soil organic matter and below-ground autotrophic respiration including root respiration and microbial respiration of root) as hypothesised with the exception of ecosystem respiration.

This illustrates how degraded these shallow peatlands were and raises concerns about the ability of ditch blocking by peat dams alone to promote the high and stable water tables required to perturb the existing *Molinia caerulea*-dominated ecosystem. It remains unknown how *Molinia caerulea* would respond in the longer-term if water tables were raised and stabilised and whether this would result in the anticipated carbon benefits. Additional restoration techniques, such as impermeable bunds, mowing or *Sphagnum* re-seeding, may be required to bring about the desired ecosystem change.

Data Availability

The research data supporting this publication are openly available from the University of Exeter's institutional repository at: <https://doi.org/10.24378/exe.2723>

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.ecoeng.2020.106079>.

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