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Title: Peatland wildfire severity and post-fire gaseous carbon fluxes

Shortened Title: Peatland wildfire severity and carbon fluxes

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Abstract

The future status of peatlands as carbon stores/sinks is uncertain given current and predicted environmental change. Several factors can affect the magnitude of the peatland carbon sink including disturbances such as wildfire. There is at present little evidence of how wildfire affects the emission of carbon dioxide (CO_2) and methane (CH_4) via perturbation to aerobic and anaerobic respiration. The greatest effects, which are likely to vary according to wildfire severity, would be expected in the immediate post fire stages when little recovery has taken place. Here, we investigate five UK peatland wildfires (2011-2012) in the immediate post-wildfire period measuring CO_2 and CH_4 fluxes using static chambers. Fire severity was described using a modified form of the Composite Burn Index. A hierarchical partitioning approach indicated time since fire was the most strongly associated variable that fluxes of both CO_2, and CH_4 followed by soil temperature for CO_2 and fire severity for CH_4. Using a liner mixed modelling approach to account for repeated measures; fire severity was a significant term for CH_4 and borderline significant for CO_2. Mean fluxes of CH_4 were consistently lower on burnt sites. In contrast, data from a fire in the north of Scotland appeared to show the opposite relationship for CH_4 with higher fluxes on the burnt sites. These results suggest that wildfire can affect gaseous carbon fluxes but the responses can be variable in both space and time and that disruption to anaerobic processes may be site and/or fire dependent.

Keywords: Methane, Carbon dioxide, *Calluna vulgaris*, Canadian Fire Weather Index System, carbon, Composite Burn Index, UK

Manuscript Highlights:

- We investigated the effects of fire severity on fluxes of CH_4 and CO_2.
- Fire severity affects fluxes of CH_4 more than CO_2 but results are not consistent.

- Carbon flux responses to wildfire are spatially and temporally variable.

Introduction

Peatlands are major global carbon stores (ca. 550 Pg C) that have accumulated at a rate of around 19 g C m^{-2} y^{-1} since the last glacial maximum (Yu, 2011). However, their current and future status as carbon sinks is uncertain because of environmental change and episodic disturbances such as fire. The occurrence of wildfires is projected to increase under a warner drier climate projected due to changes in temperature and rainfall patterns (Krawchuk and others 2009; de Groot and others 2013). Recent estimates for boreal regions suggest a doubling of the mean annual burn area since the 1960s coincident with significant regional warming (Gillett and others 2004; Kasischke and Turetsky, 2006; Turetsky and others 2015).The consequences of global climate change for peatland wildfire frequency and severity are uncertain but there is the potential for positive feedbacks between these processes (McMorrow and others 2009; Turetsky and others, 2015; Grau-Andrés and others 2019). In boreal regions, interactions between climate change and increases in fire frequency have the potential to switch peatlands from acting as net carbon sinks to net sources (Wieder and others 2009). In addition to chronic alterations to peatland C-cycling wildfires can cause significant instantaneous losses of carbon from both above (Davies and others 2016a) and below ground. Smouldering combustion of peat deposits is a particular issue affecting both carbon stocks (Davies and others 2013; Rein, 2013) and ecosystem function more broadly. For example, the long duration of smouldering fires can transfer more and deeper heat to surrounding soils that results in a fuel consumption two orders of magnitude above flaming combustion, and lasting damage to heat-sensitive plant roots and microorganisms (Treseder and others 2004; Hart and others 2005; Turetsky and others, 2015).

To date, the majority of wildfire research in peatland ecosystems has been completed in northern tundra (Turetsky and others, 2015), boreal (Thompson and Waddington, 2013) and tropical ecosystems (Page and others 2002; Turetsky and others, 2015). Temperate peatlands, such as those in the UK have, by contrast, been somewhat overlooked scientifically (Davies and others, 2013; Davies and others 2016b). This is a concern as, in the UK, standardized national monitoring of protected peatlands (Williams, 2006), have classified 42% of blanket bogs, and 78% of lowland raised bogs as being in “unfavourable” condition (i.e. showing one or more indicators of adverse ecological condition or management). Peatlands are the most widespread semi-natural habitat in the UK and store an estimated 4.5 Tg of carbon (Bradley and others 2005) in deep peat deposits (up to several meters) and shallower carbon rich organic soils (e.g. peaty podzols). In the latter systems, estimates suggest around 88 t C ha^{-1} are stored in the soil and up to 2 t C ha^{-1} in the associated dwarf shrubs dominated vegetation (Carey and others 2008; Ostle and others 2009). The role of fire in UK peatland ecology has become a highly controversial subject with substantial debate between scientists, land-managers and conservationists about the effects of managed burning on ecosystem dynamics (Grant and others 2012; Davies and others, 2016a). A substantive evidence gap regarding fire effects on temperate peatlands does not help: the majority of peatland fire studies have been completed on low severity managed burns (e.g. Davies and others 2010; Harris and others 2011) or less often, on wildfires at single sites (e.g. Maltby and others 1990; Clay and Worrall, 2011). The small number of single-site based studies available suggest that wildfires can occur at very high severities and lead to long-term changes to ecosystem function (Maltby and others, 1990; Legg and others 1992; Davies and others, 2013). There is thus a distinct gap in our understanding of the impacts of wildfire under a range of severities particularly in the UK where little data is available and management decisions are being made on the basis of incomplete knowledge (Davies and others, 2016a).

In order to develop an understanding of the ecological effects of wildfire, their interactions with environmental change, and how this could potentially affect ecosystem carbon storage, we need to capture and understand variation in effects across the spectrum of fire severity. This requires consideration of three key processes (Flannigan and others 2009):

1. Direct loss of carbon as CO_2, CO and CH_4 during combustion.
2. Post-fire changes in ecosystem carbon fluxes due to altered sequestration and decomposition processes.
3. Post-fire vegetation dynamics and ecosystem recovery.

The aim of this paper is to addresses the second of the above issues and we thus seek to assess variation in soil (i.e. belowground) carbon fluxes from peatlands subject to wildfires of varying severity. In a previous paper (Davies and others, 2016a) we considered the first issue and, for the same set of fires, estimated above and below ground losses of biomass and organic matter due to combustion. Our focus here is on the immediate post-fire period (first 6-12 months following burning) as we expect this to be the period when peatland ecosystem function displays the greatest sensitivity to variation in fire severity due to impacts on soil hydrology, temperature regimes and the effects of heating, ash deposition and root turn-over on soil microbial communities. Although longer-term data are also sorely lacking, there is also an urgent need to assess the extent to which fire effects can vary between and within landscapes on shorter time scales.

Methods

Main Study Sites

We sampled wildfires that occurred in the springs of 2011 and 2012 (Table 1) and burn across peatland habitats with carbon rich deposits. Sites were selected from information provided by land-managers, public and private landowners, government agencies and regional Fire and Rescue Services. From an initial list of twenty-six fires we selected five sites that, i) based on managers' descriptions and site walk-overs, represented moderate to high fire severity; ii) included a variety of peatland plant communities and soil types; and iii) captured elements of the North-South and West-East range of bioclimatic conditions in Great Britain (Harrison and others 2001). Most sites were broadly classified as mires on deep peat whilst one site, was a dry heathland Finzean with shallow (ca. 10 cm), stony organic soil. Further details on site vegetation and environmental conditions are in Table 1 and can be found in Davies and others (2016a).

Experimental design and estimation of fire severity

Fire severity and fluxes of CO_2 and CH_4 were measured approximately one year 2 months after the fires occurred except at Finzean where they were measured 4 months after the fire. In the case of one fire these variables were measured roughly 4-6 months post-burn. As wildfires are sporadic unpredictable events, we were unable to survey sites prior to the fires. Instead, we used paired plots with burnt and unburnt subplots located across the fire perimeter to estimate the effects of fire (e.g. Kasischke and Johnstone, 2005; de Groot and others 2009; Hollis and others 2011). To avoid potentially confounding pre- and post-fire differences in vegetation structure, subplots were only established where we were confident that pre-fire fuel conditions across the fire-line were similar (see Davies and others, 2016a), and in regions of the fire-line known to have been actively extinguished. Two or three paired plots were located within each fire and chosen to represent the range of burn severities visible during a detailed site reconnaissance with local stakeholders (Table 1).

The Composite Burn Index (CBI) was developed as a method to provide rapid assessment of fire effects and as a tool to validate remotely-sensed indices of fire impact (Key and Benson, 2006). Research in boreal and tundra ecosystems has provided somewhat mixed results regarding its utility. Early results (e.g. Murphy and others 2008) suggested relatively weak relationships with differenced Normalized Burn Ratio (dNBR) values in Alaska but subsequent studies (e.g. Jones and others 2009; Kolden and Rogan, 2013; Loboda and others

2013; Chen and others 2020) have shown significant, though sometimes noisy, relationships between the CBI and a variety of remotely sensed indices. Furthermore, studies have demonstrated significant, though again noisy, relationships between CBI values and changes to soil microbial communities (Whitman and others 2019), and mineral soil exposure, canopy tree damage and proportional reduction in organic layer depth (Kasischke and others 2008). Hudspith and others (2017) potentially shed some light on the reason for weak relationships between ground-based observations of tundra fire effects and the CBI noting that the spatial scales of variation in combustion dynamics and durations of heating tend to be smaller than those associated with the comparatively large CBI plots. To estimate fire severity we adapted the Composite Burn Index (CBI; (Key and Benson, 2006) to account for the ecological characteristics of peat bog ecosystems. This included assessing the specific impact of fire on peat-building *Sphagnum* species. We recorded fire severity in circular plots 20 m in diameter and assessed a variety of characteristics according to two strata – substrates (soil, litter and mosses), and the field layer (dwarf shrubs and graminoids). All variables were rated on a scale of 1-3 with individual ratings averaged within strata and then summed across the strata (Table S1). A full protocol and data collection sheet for using the peatland CBI methodology (pCBI) can be found in Davies and others (2016a).

Creating low severity reference conditions

Untangling the belowground effects of fires on ecosystem carbon dynamics is complicated by the fact that unburnt sites have actively respiring and photosynthesising aboveground vegetation, which is missing or very limited in recently burned locations. Exploring whether there are differences in GPP and NEE between burned and unburned areas is thus comparatively uninteresting where a primary objective is to understand short-term ecosystem responses to disturbance severity. In sum, obvious differences in ecosystem function between unburnt and recently burned areas thus mask more interesting potential differences in belowground processes. As we were interested in assessing how fires affect belowground processes we carefully considered how best to define a low/no severity control to contrast with our burnt plots. Peatland soil carbon fluxes are controlled by a myriad of complex, interacting biotic and abiotic processes. Microbial community composition and activity are known to be key controls on ecosystem carbon fluxes. Above ground vegetation composition and structure is, however, also critical due to coupled above-below ground community composition and the presence of aerenchyma (Jassey and others 2013). Removal of above-ground vegetation by fire is likely to cause changes in the production of root exudates (Basiliko and others 2012)) in addition to other direct and indirect effects of fire on hydrology (Brown and others 2015), chemical (González-Pérez and others 2004) and physical characteristics (Morison and others 2019). To isolate the effects of fire impacts on soil processes from variation in fluxes driven by differences in the abundance of above-ground vegetation in burned and unburnt subplots we removed all surface vegetation to the top of the peat. This included all living vascular and cryptogrammic vegetation in the unburnt chambers, and any small patches of moss and vascular plant resprouts and seedlings in the burned chambers. Cutting treatments were applied as soon as we had gained access to the sites and at the time the chambers were installed (June 2012). This was approximately 14 months after the fires for the majority of sites (2 months post-burn in the case of the Finzean fire). Cutting treatments removed vascular and cryptogamic plant photosynthesis and respiration as a factor in our measured fluxes. Unburned plots thus constituted a “zero severity” control where, similarly to burned subplots, vegetation had been removed and soil microclimatic conditions altered but soils had not been exposed to the direct heating and chemical (González-Pérez and others, 2004; Grau-Andrés and others 2018) effects of the passage of a fire front . The subplots thus differed in the magnitude of consumption and chemical alteration (e.g. charring, ash deposition) of the soil, but also in the time since disturbance (14 months versus 2 months at first measurement) Removal of above ground vegetation by fire or cutting are both likely to influence soil microbial communities (Chen and others 2008). This may be tied to differential production of root exudate production, though previous research (Basiliko and others, 2012) has suggested differences have limited consequences for carbon fluxes. Cutting as opposed to burning to sedges (particularly *Eriophorum* spp.) could have consequences for soil gas fluxes if the disturbances yield differential outcomes for opening up pathways for gas transport through aerenchyma. We therefore hypothesized that, in the absence of a fire severity effect, plots where *Eriophorum* spp. were dominant would show higher methane fluxes in unburned compared to burnt subplots.

Chamber flux measurements

CO_2 and CH_4 fluxes were measured between June and September 2012 with each plot measured across three different months (late June/early July, early-mid August, early September). Measurements were made via the static chamber method (Hutchinson & Mosier 1981). Five gas flux chambers were permanently located at random co-ordinates within each subplot of our burnt/unburnt plots. The chambers were made of black plastic, had a diameter of ca. 38 cm and a volume of ca. 30 l. Chambers were buried 3-5 cm below the top of the peat to ensure a good seal. Chambers were closed using metal lids lined with an insulating foam strip to ensure a good seal. Gas samples were extracted using a 100 ml syringe, which was then used to fill an evacuated vial. Samples were extracted immediately following closure and at ten-minute intervals thereafter to produce five samples per chamber. Both CH_4 and CO_2 concentrations were analysed using a Hewlett Packard gas chromatograph (5890 series II).

For each sequence of gas samples from a chamber, the flux was calculated as:

$$F = \frac{dC}{dt_0} \cdot \frac{pV}{A} \qquad (1)$$

Where: F is gas flux from the soil (μmol m^{-2} s^{-1}), dC/dt_0 is the initial rate of change in concentration with time (μmol m^{-2} s^{-1}), p is the density of air (mol m^{-3}), V is the volume of the chamber (m^3) and A is the ground area enclosed by the chamber in (m^2).

The parameter dC/dt_0 was calculated using linear and non-linear asymptotic regression following the methods established by Levy et al. (2011). The regression method that provided the best fit for the time series of concentration was chosen for each individual measurement using a mixture of goodness-of-fit statistics and visual inspection.

Ancillary measurements: soil temperature and water table depth

Soil temperatures estimates at 10 cm depth were measured alongside all chamber measurement runs using a digital probe thermometer. Water table depth was only recorded for the Forsinard fire where it was measured manually using dip wells (Brooks and Stoneman, 1997). Water table depth is understood to be an important control on both CO_2 and CH_4 fluxes from peatlands (Levy and others 2012). Whilst our data does not currently allow us to explore this important mechanistic control, our data still allowed us to make broad comparisons between burned and unburned areas, different pre-fire vegetation communities and across varying levels of burn severity. In lieu of specifically measured hydrological data we used weather data to calculate indices that could act as proxies for the lack of information on soil moisture conditions (see below).

Fire weather

Estimating variation in burning conditions between the fires was completed using the indices and codes of the Canadian Fire Weather Index System (Van Wagner, 1987). Calculating the values of the FWI System requires daily data on wind speed, temperature and humidity at 12 noon as well as 24-hour accumulated rainfall. These data were extracted from the British

Atmospheric Data Centre database (LINK) for the nearest weather station to each of the wildfires. In some instances rainfall data were available from rain gauges closer to the fire site than the nearest full weather station and so data from the two locations were combined. Available data on 24 hour accumulated rainfall was 09:00-09:00 rather than noon to noon though the difference is unlikely to be of great importance. FWI System values were calculated using the package “fume” (Santander Meteorology Group 2012) in R 3.1.2 (R Development Core Team Team, 2016). Some of the moisture codes and indices of the FWI System have relatively long lag times (52 days for the Drought Code) and so values were calculated starting at least three months prior to the date of each fire. FWI System moisture codes were used as surrogates for soil moisture as we had no way of consistently measuring peat moisture across the main wildfire sites. We calculated the Fine Fuel Moisture Code (FFMC), Duff Moisture Code (DMC) and Drought Code (DC), for each fire on the dates of the flux sampling. Of these the latter has been observed to relate relatively strongly to peat moisture content and smouldering fire activity (Davies and others, 2010). We also included the raw weather data as explanatory variables in subsequent analyses.

Data analysis

Following the approach developed by Fernandes and others (2000), and we analysed data from the wildfires at the plot-level in essence treating each plot as a separate observation of fire effects and burn severity. The authors here are very aware of the challenge of pseudoreplication (Davies and Gray, 2015). We believe our approach is valid as fire behaviour within individual plots can be considered quasi-independent due to: i) substantial variations in fuel structure across the fire ground, ii) changes in fire weather during the course of the burn day; iii) the long firelines and substantial distances between plots; and iv) the different times at which plots were burnt. Previous research in similar vegetation types has revealed this method to be appropriate with greater variation in vegetation structure observed within experimental burn areas than exists, on average, between them (Grau-Andrés and others, 2019). Irrespective of the above points we included fire and plot identify as random effects in our analysing.

Both CO_2 and CH_4 flux data were log transformed and we firstly used a Hierarchical Partitioning (HP) to assess the overall relationships between fire severity (pCBI), FWI variables (mainly as surrogates for soil moisture), weather data and carbon fluxes. We used HP in preference to the other methods as HP has the ability to assess the independent contributions of relationships of each of the variables, even when the variables are strongly intercorrelated, as is the case here (Chevan and Sutherland, 1991; MacNally, 1996; MacNally, 2000; Walsh and MacNally, 2003; MacNally and Walsh, 2004). The independent contributions of each of our explanatory variables were thus determined using HP but the significance of each contribution was not tested due to the grouping structure in the experimental design. Data were analysed using R 3.3.1 (R Development Core Team Team, 2016).

To determine the significance of fire severity we applied a linear mixed model to the same data using the fixed effects of air temperature, soil temperature, relative humidity, rainfall, FFMC, DMC, DC, and pCBI. Random effects represented the time since the fire occurred (accounting for the repeated measures structure of our monitoring and the fact one fire had burned more recently than the others), and the nested structure of our sampling design: chambers were nested within plots, which were nested within fires. Significance was determined by Type II Wald Tests. Including plot as a random effect accounts for the paired burnt-unburnt subplots design of our experiment. In the case of the CH_4 flux data we ran the model twice, firstly with all the data and then with some outlying large fluxes removed (four in total) to gauge their influence on the results. The general outcome of the model was not altered with these outliers removed so we report the full dataset analysis here.

Additional Fire

In addition to the above sites we also include data from a previous field campaign which examined a small wildfire that occurred in March 2004 in the Forsinard area of Caithness in Scotland (Table 1). These data were collected in the summer immediately after the fire occurred using paired unburnt and burnt plots. Fluxes from the Forsinard fire were sampled monthly from July to October 2004 in a similar manner as above but with a slightly larger chamber (0.32 m^2, see Gray, 2006 for details). At the Forsinard fire, gas samples were taken in the field using a similar static chamber approach. Two chamber top designs, were used to assess: i) Gross Primary Productivity (CO_2 flux); ii) Net Ecosystem Exchange (CO_2 flux); and iii) net methane exchange (CH_4). For GPP the chamber sides and top were made of Propafilm-C©; this material has a high transmittance to both light and thermal radiation (Hunt, 2003). For NEE and CH4 exchange a light exclusion chamber was used. In both chamber s a 5-volt fan was located inside ensuring sufficient mixing of chamber air. The basal area of the chambers was 0.32 m^2 and internal volume 0.09 m^3. Three burnt and three unburnt chambers were placed in random locations within a reasonably homogeneous area of vegetation in burnt and unburnt areas dominated by *Trichophorum cespitosum* (L.) Hartm., *Sphagnum capillifolium* (Ehrh.) Hedw. and *Molinia caerulea* (L.) Moench . For estimating CO_2 fluxes, light and dark measurements were recorded over a five-minute period or until concentrations changed by 50 ppm from ambient using an infrared gas analyser (IRGA) (Gascard II Edinburgh Instruments) at the rate of 0.009 m^3 min^{-1}. CH_4 measurements were made using the dark chamber but over a 30-minute period with gas collection every 10 minutes. CH_4 concentration was determined by either a Hewlett Packard 5890 Series II or an Agilent 6890N flame ionisation gas chromatograph. Unfortunately, no fire severity data were available for the Forsinard fire and due to the limited replication for the Forsinard fire data we follow the guidance in Davies & Gray (2015) and only summarise the data graphically and make qualitative comparisons with the results from the other wildfires.

Results

Mean CH_4 fluxes from unburned subplots ranged from 3.89 nmols m^{-2} s^{-1} at Finzean to 10.08 nmols m^{-2} s^{-1} at Loch Doon (Figure 1). Mean CH_4 fluxes for the burnt subplots ranged from 3.45 nmols m^{-2} s^{-1} at Finzean to 7.53 nmols m^{-2} s^{-1} at Marsden (Figure 1). There was thus a general trend for mean fluxes to be higher in unburned subplots than in those which had been burned and a general decreasing trend with increasing pCBI (Figure 1).

Mean CO_2 fluxes from unburned subplots ranged from 0.83 µmols m^{-2} s^{-1} at Angelzarke to 2.01 µmols m^{-2} s^{-1} at Finzean (Figure 1). Mean CO_2 fluxes for the burnt subplots ranged from 1.30 µmols m^{-2} s^{-1} at Star to 1.60 µmols m^{-2} s^{-1} at Wainstalls (Figure 1). There was no clear general trend for mean CO_2 fluxes between the unburned and burnt plots and no trend with increasing pCBI (Figure 1).

Hierarchial partioning

The HP results indicate that time since fire time since fire was the variable most strongly associated with fluxes of both decreasing CO_2 and increasing CH_4 (Figure 2). Fire severity (pCBI score) was the next most associated variable for CH_4 with all other variables having little explanatory power (Figure 2a). CO_2 appeared to be somewhat related soil temperature and DMC with increased fluxes with increased soil temperature and decreasing fluxes with increasing DMC but fire severity appeared to have little association (Figure 2b).

Mixed-effects model results

For CH_4 emissions, the mixed-effects model showed pCBI to be the only significant variable (Table 2), with higher fluxes associated with lower pCBI severities. For CO_2, no significant variables were found at $P < 0.05$, although fire severity (pCBI) displayed a trend ($P < 0.1$) suggestive of slightly reduced fluxes at higher pCBI values (Table 3). However, for both the CH_4 results and the CO_2 results the explanatory power of fixed effects was low (Tables 2 and

3). Random effects (fire and plot identity, and time since fire) accounted for much of the variance in our dataset, and time since fire accounted for over a third of the variance explained by the random effects (Tables 2 and 3).

Fluxes from the Forsinard Fire

For CO_2 at Forsinard there appeared to be no clear differences between burnt and unburnt plots when using light or dark chambers (Figure 2). However, for CH_4 emissions were greater in the burnt plots. These differences do not appear to be driven by differences in soil temperature or water table but burnt plots have slightly more aerenchymatous species (Figure 2) and more *Myrica gale* L. a nitrogen fixing species, although, overall cover of this was never over 15% in any plot.

Discussion

The fires examined here suggest that wildfire may disrupt anaerobic respiration processes in the immediate post fire regeneration phase but that the magnitude of such effects are dependent upon fire severity. Interestingly, the effects of varying severity on aerobic respiration processes seemed to be more modest. The decrease in CH_4 emissions in relation to fire severity suggest that wildfire disrupts the microbial process involved in the production and/or oxidation of CH_4. Whilst, the direct effects of our fires on the microbial community are unknown it seems unlikely that the fire would affect the methanogenic community responsible for CH_4 production as this tends to occurs in deeper more waterlogged peat in the catotelm rather than the aerated acrotelm (c.f. Clymo, 1984; Morris and others 2011) where our effects would be most prevalent. The direct effects of moderately severe fires on peat, as measured by the fire-induced soil heating, are barely noticeable 2 cm below ground during the fire, but may be significant during the longer-post fire period due to induced changes in soil mircoclimate (Grau-Andrés and others, 2018). It therefore seems likely that post-fire oxidation of CH_4 to CO_2 during methanotrophy is somehow promoted (c.f. Wang and others 2012). The precise mechanism remains behind such changes will remain elusive unless further research explores post-fire acrotelm thermal dynamics, water table fluctuations, additions of nutrients from the fire, alterations to root exudates and microbial community changes. The effects of wildfires on the ecosystem function and structure of temperate peatlands remain fundamentally understudied (Davies et al. 2016). Disruption to the C sink potential of, broadly analogous, boreal peatlands by wildfire has however been attributed to fuel combustion, reduced photosynthetic activity and increases in ecosystem respiration arising from increased temperatures and ash fertilization (Turetsky and others 2002).

The fact that the Forsinard site showed the complete opposite reaction in burnt plots, emitting higher CH_4 fluxes, suggests that the effects of wildfire are not simple unidirectional effects.

The substantial amount of variance in fluxes explained by random effects in our models emphasizes the importance of temporal and spatial variability in the effects of wildfires. Individual fires and or sites may behave in different ways spatially and perhaps temporally as was underlined by Davies and others (2016a) examination of variation in fire severity and fuel consumption at the same wildfire sites.

It was unfortunate that no burn severity data was available for the Forsinard fire, but from casual observation it appeared to be of low severity fire as it lacked the large-scale combustion of the bryophytes layer seen at other sites. At Forsinard, there were no differences in soil temperature or water table between burnt and unburnt plots, it is possible that little disruption was caused by the fire and burnt plots were located on areas that just happen to produce more methane. Vegetation was similar in burnt and unburnt plots, except for a slightly higher coverage of species with aerenchyma in burnt plots (Figure 3), this may partly explain the disparity. Research elsewhere has shown that the effects of burning on CH_4 flux are not simple, either reporting no short-term (< 3 years) change (Taylor, 2015) or slightly longer-term (10 years) declines (Ward et al., 2007) following managed fires.

The strongest relationship with CO_2 fluxes appeared to be time since fire but aerobic respiration did not vary with either the presence of fire itself and only weakly with fire severity. Similarly there appeared to be no effect of fire on CO_2 fluxes at the Forsinard site. That CO_2 fluxes under light conditions at Forsinard were relatively similar in burnt and control plots may suggest a low severity fire with rapid vegetation recovery. Comparison of burnt and unburnt plots at the Moor House Nature Reserve in northern England found that ecosystem respiration was similar at least for the short-term (< 18 months) (Clay Gareth and others 2010; Ward and others 2012). Similarly, fire had no short term effect on respiration at three contrasting sites in Scotland (Taylor, 2015). However, respiration was higher in plots burnt on shorter rotations than in those burnt over longer-term at Moor House (Ward and others 2007). However, most of these studies investigated managed burning which by definition is usually of low severity and may not be directly comparable to wildfire.

Conclusions

In the wildfires studied here, fire did not appear to disrupt aerobic respiration processes but anaerobic process especially those that relate to CH_4 emissions could be linked to fire severity. However, the precise mechanism remains elusive and may be related to the condition of individual fires and/or sites. Further research should explore post fire acrotelm thermal dynamics, additions of nutrients from the fire, alterations to root exudates and microbial community change.

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Table Legends

Table 1: Summary of wildfire sites conditions and vegetation reported as National Vegetation Classification (NVC) communities (Rodwell, 1991) with the NVC code in brackets. Paired plots were those placed around the fire perimeter to enable direct comparison of burnt and unburnt gas fluxes.

Table 2: CH_4 mixed model results significance determined by Type II Wald Test. Soil T – soil temperature (°C); Air T – air temperature (°C); RH – relative humidity; Rain (mm); FFMC Fine fuel moisture code; DMC duff moisture code; DC - drought code. pCBI – peatland composite burn index (fire severtity).

Table 3: CO_2 mixed model results significance determined by Type II Wald Test.Soil T – soil temperature (°C); Air T – air temperature (°C); RH – relative humidity; Rain (mm); FFMC Fine fuel moisture code; DMC duff moisture code; DC - drought code. pCBI – peatland composite burn index (fire severtity).

Figure Legends

Figure 1: (a) CH_4 fluxes plotted against fire severity (b) CO_2 fluxes against fire severity. (c) Mean CH_4 fluxes (± 95% condidence intervals) for burnt and unburnt areas for the 2014 wildfire sites. (d) Mean CO_2 fluxes (± 95% condidence intervals) for burnt and unburnt areas for the 2014 wildfire sites.

Figure 2: CH_4 (a) and CO_2 (b) Hierarchical Partitioning results showing the explained variation partitioned into the individual contributions of each of the variables and each contribution is a percentage of the overall R^2. Soil T – soil temperature (ºC); Air T – air temperature (ºC); RH – relative humidity; Rain (mm); FFMC Fine fuel moisture code; DMC duff moisture code; DC - drought code. pCBI – peatland composite burn index (fire severtity); Time – time since fire (days).

Figure 3: Mean (± 95% condidence intervals) CO_2 and CH_4 fluxes (µmol m^{-2} s^{-1}), water table (mm), soil temperature (°C) and the sum of percent cover for plant life forms for the Forsinard fire site.

Table 1

Fire name and Location	**Latitude and Longitude**	**Date of fire**	**Sampling commenced**	**Burned area (ha)**	**Elevation (m)**	**Soil**	**Vegetation type**	**Paired CBI plots**
Anglezarke (N England)	53.658°N 2.569°W	29/Apr/2011	July 2012	4,144	270 - 380	Deep peat	*Calluna vulgaris - Eriophorum vaginatum* blanket mire (M19)	3
Mardsen (N England)	53.596°N 1.976°W	09/Apr/2011	June 2012	316	385 - 480	Deep peat	*Calluna vulgaris - Eriophorum vaginatum* blanket mire (M19) *Calluna vulgaris - Vaccinium myrtillus* heath (H12)	3
Star Moss (SW Scotland)	55.214°N 4.393°W	29/May/2011	June 2012	No data	230 - 250	Shallow peat	*Molinia caerulea - Potentilla erecta* mire (M25a)	2
Wainstalls (N England)	53.777°N 1.928°W	30/Apr/2011	June 2012	82	385 - 420	Deep peat	*Calluna vulgaris - Eriophorum vaginatum* blanket mire (M19) Scattered *Calluna vulgaris - Vaccinium myrtillus* heath (H12)	3
Finzean (NE Scotland)	57.025°N 2.702°W	30/Mar/2012	August 2012	19	320 - 340	Rocky organic	*Calluna vulgaris - Vaccinium myrtillus* heath (H12)	3
Forsinard (NE Scotland)	58.425°N 3.912°W	May 2004	June 2004	2	105	Deep peat	*Scirpus cespitosus Erica tetralix* wet heath (M15)	0

Table 2

CH_4	Estimate	Std. Error	Chisq	Df	P
Soil.Temp	0.019	0.017	1.385	1	0.239
Air.Temp	0.002	0.011	0.019	1	0.891
RH	0.000	0.004	0.000	1	0.999
Rain..mm.	-0.002	0.003	0.232	1	0.630
FFMC	-0.001	0.003	0.063	1	0.802
DMC	0.000	0.019	0.000	1	0.985
DC	0.000	0.001	0.093	1	0.760
pCBI	-0.037	0.009	18.595	1	0.000
R^2 fixed	0.059				
R^2 fixed and random	0.628				

Table 3

CO_2	Estimate	Std. Error	Chisq	Df	P
Soil.Temp	0.012	0.018	0.433	1	0.510
Air.Temp	0.007	0.024	0.098	1	0.755
RH	-0.004	0.008	0.297	1	0.586
Rain..mm.	-0.007	0.007	1.059	1	0.304
FFMC	-0.004	0.005	0.630	1	0.427
DMC	0.010	0.039	0.069	1	0.793
DC	-0.002	0.003	0.402	1	0.526
pCBI	-0.018	0.009	3.656	1	0.056
R^2 fixed	0.059				
R^2 Fixed and random	0.714				

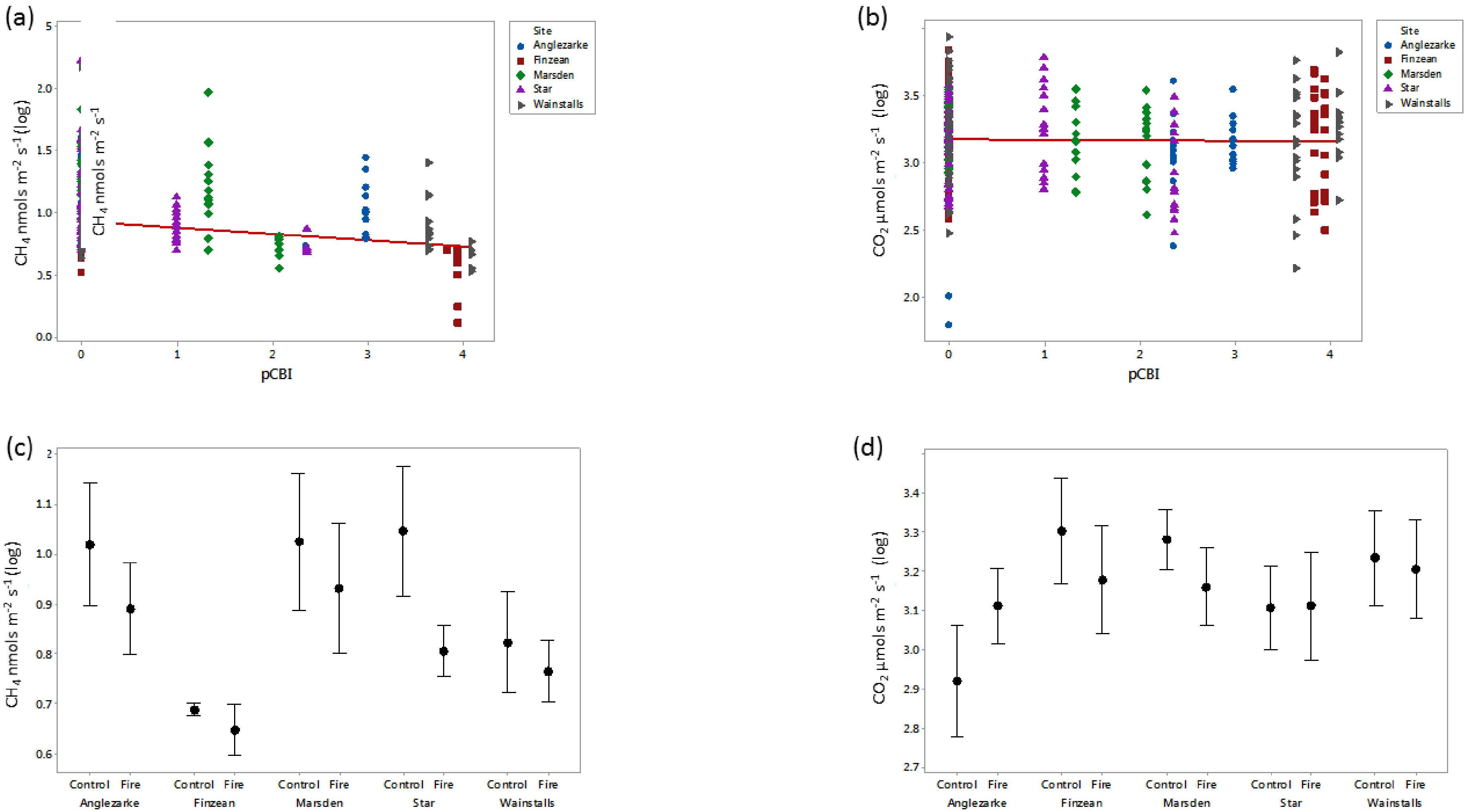
(a)
CH4 nmols m-2 s-1 (log)
pCBI
Site
Anglezarke
Finzean
Marsden
Star
Wainstalls
(b)
CO2 µmols m-2 s-1 (log)
pCBI
(c)
CH4 nmols m-2 s-1 (log)
(d)
CO2 µmols m-2 s-1 (log)
Control
Fire
Anglezarke
Finzean
Marsden
Star
Wainstalls

Figure 1

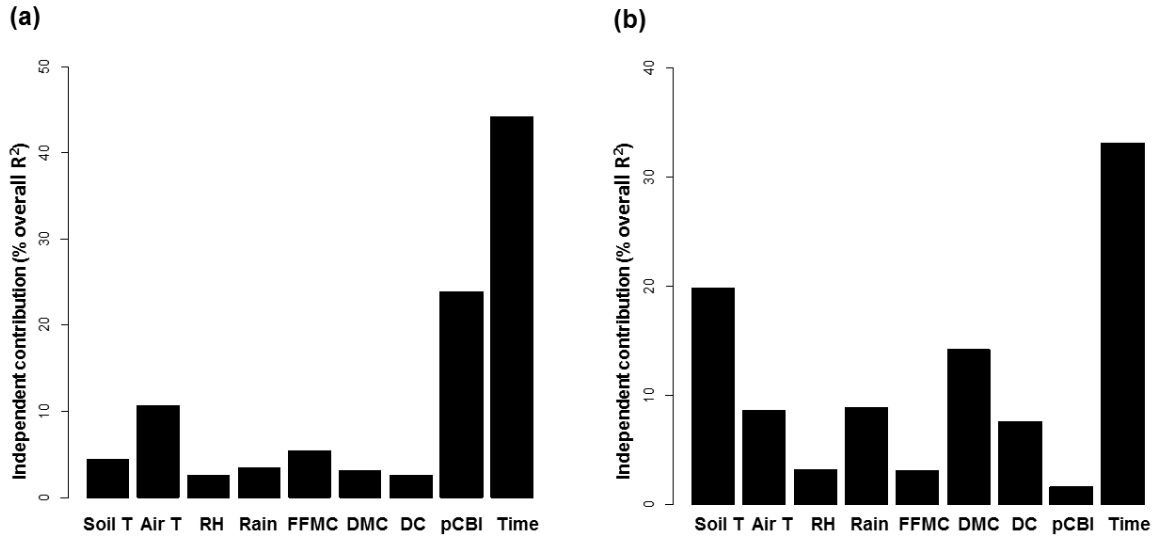

Figure 2

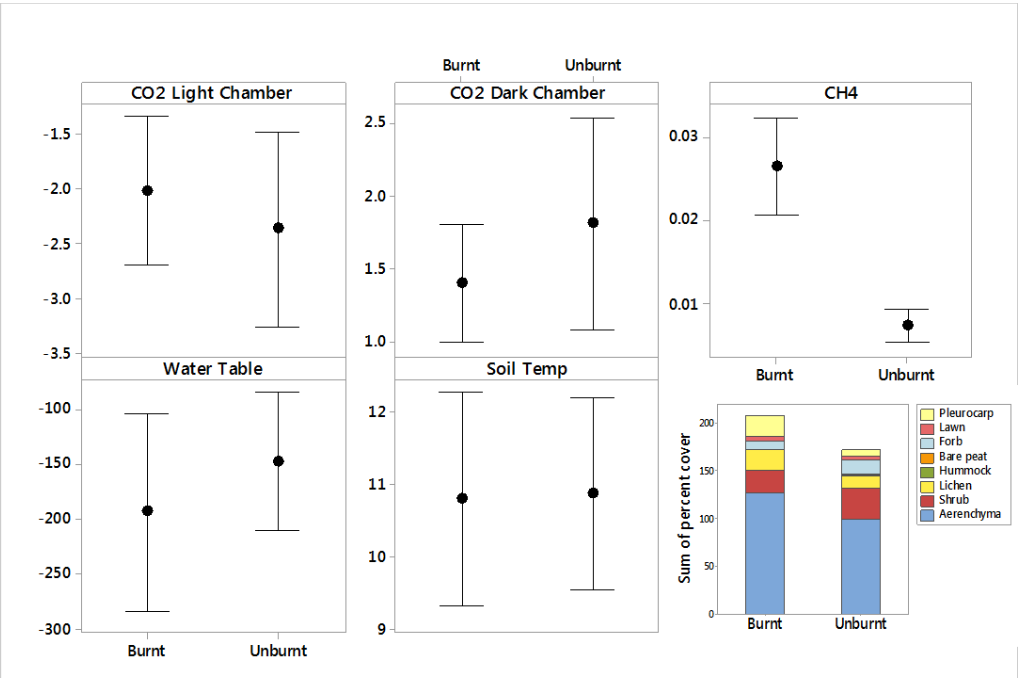

Figure 3

Supplementary Material: Peatland wildfire severity and post-fire gaseous carbon fluxes

Alan Gray[1*], G. Matt Davies[2,3,4], Rut Domènech[3,5], Emily Taylor[1,6] & Peter E. Levy[1]

Table S1: Variables recorded during fire severity assessment using the peatland Composite Burn Index. Table S1: Variables recorded during fire severity assessment using the peatland Composite Burn Index.

Substrate/ground fuel effects	Surface fuel effects
Litter/light fuel consumed	Proportion of plants top-killed
Area showing charred or consumed peat	Fine/Crown fuel consumed
Ash cover	Survival (%) of grass/sedge tussocks
Exposed mineral soil cover	Survival (%) of shrubs
Sphagnum damage (% loss of capitula)	Shrub resprout abundance
Moss scorch/consumption	Potential for new colonizing species
Sphagnum/moss survival	Potential changes in species composition

Figure S1: Mean soil temperature (±95% CI) of burnt (fire) and unburnt (control) plots for each of the wildfire sites.