Carbon stocks and carbon fluxes from a 10 year prescribed burning chronosequence on a UK blanket peat

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**Abstract**

Prescribed burning is a common land management technique in many areas of the UK uplands. However, concern has been expressed at the impact of this management practice on carbon stocks and fluxes found in the carbon rich peat soils that underlie many of these areas. This study measured both carbon stocks and carbon fluxes from a chronosequence of prescribed burn sites in northern England. A range of carbon parameters were measured including: above-ground biomass and carbon stocks; net ecosystem exchange (NEE) ecosystem respiration ($R_{eco}$) and photosynthesis ($P_g$) from closed chamber methods; and particulate organic carbon (POC). Analysis of the $CO_2$ data showed that burning was a significant factor in measured $CO_2$ readings but that other factors such as month of sampling explained a greater proportion of the variation in the data. Carbon budget results showed that whilst all the sites were net sources of carbon, the most recent burn scars were smaller sources of carbon compared to the older burn scars (Burn year 2009: 85 ± 29 gC $m^{-2}$ $yr^{-1}$; Burn year 1999: 152 ± 12 gC $m^{-2}$ $yr^{-1}$). By trading off these smaller sources against larger sources including those on the control plots (no burn scenario), this study ~~shows~~showed, that even accounting for the loss of carbon during a burn itself, burning management ~~shows~~ showed an "avoided loss" of approximately 40 gC $m^{-2}$ $yr^{-1}$ relative to not burning.

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## 1. Introduction

The UK holds approximately 10-15% of the world's blanket bog (Tallis et al., 1998) and approximately 2302 Mt of carbon (C) are stored in peatland settings in the UK (Billett et al., 2010), however, UK upland habitats are intensely managed and have been impacted by a legacy of atmospheric pollution, tourism, overgrazing and wildfire (Holden et al., 2007). In the UK many upland peat catchments are managed for sheep grazing and/or for recreational hunting. One common land management practice is the use of prescribed burning. Here the aim is to remove the older, less productive vegetation and to encourage new growth for livestock grazing and for red grouse production. Patches of vegetation are burned on an 8 - 25 year rotation creating a mosaic of stand ages. There have been numerous studies and reviews on the impact of burning on vegetation and a range of ecosystem services (e.g. Gimingham, 1972; Hobbs and Gimingham, 1984; Ramchunder et al., 2009; Tucker, 2003) though fewer have investigated the impact of burning upon carbon dynamics and the ecosystem service of climate mitigation.

Of those studies that have examined carbon dynamics, most have considered individual components of the carbon (or greenhouse gas) budget of a peatland under prescribed burning especially, dissolved organic carbon (DOC) (e.g. Clay et al., 2009; Yallop and Clutterbuck, 2009), and the $CO_2$ exchange (Ward et al., 2007). Clay et al. (2010) estimated the total C budget of plots under a range of burn managements including unburned controls and showed that while the total C budget on unburned plots was a source of 158 gC $m^{-2}$ $yr^{-1}$ that on burned plots was a total C source of 118 gC $m^{-2}$ $yr^{-1}$. Worrall et al. (2010) carried out a meta-analysis of the existing datasets and showed that there was only a 7% probability of improving the carbon budget and a 40% probability of improving the greenhouse gas budget (GHG) by introducing prescribed burning onto a peatland. An improvement was defined in that study as increased carbon storage relative to the soil e.g. an increase in primary productivity would be an improvement, whilst a decrease in particulate organic carbon (POC) loss would also be an improvement. A distinct feature of the carbon and GHG balance of peatlands under prescribed burning is that the act of management i.e. fire, is itself a carbon (and GHG) transfer e.g. from relatively degradable biomass to more refractory char. Therefore for a more complete understanding of the carbon cycling, it is necessary to consider the change in carbon stocks at the time of burning itself while including the production of refractory black carbon during the burning process (Clay and Worrall, 2011; Forbes et al., 2006).

Prescribed burning on peatlands takes place on rotations of 8 to 25 years (Yallop et al., 2009) but few studies of the impact of prescribed burning carbon fluxes have long-term records; one of the longest records available only considered 33 months of data – 21 months prior to a burn and 12 months afterwards (Clay et al.,

2010). The question therefore is what happens to the carbon fluxes over time between repeat burning i.e. across the entire burn cycle? With many years between burning treatments, adopting a long-term monitoring strategy would pose significant financial and logistical problems while an alternative is to carry out a spatial-temporal substitution and monitor a range of sites with different times since burn across an annual cycle, i.e. consider a chronosequence. The aim of this study is to carry out an assessment of prescribed burning using a combination of field based experiments coupled with probabilistic modelling of the overall impacts of burning on the carbon budgets of peatlands.

## 2. Materials and methods

### 2.1. Study site

The study was conducted across a paired site in Northumberland. The first site was Emblehope Moor (UK Grid Ref: NY 724 972) in Redesdale Forest and the second site was located 23 km away on Ray Demesne moor, near Raylees (UK Grid Ref: NY 941 893). The nearby Coalburn catchment (Robinson, 1999) has a mean annual rainfall of 1350 mm and daily mean temperature of 8 °C (2000-2005). The area is underlain by sequences of Carboniferous calcareous mudstones, limestones and sandstones. Both sites are areas of deep peat (50cm or greater) although there are some areas of thinner soils and this is more pronounced on the Ray Demesne plots. Dominant vegetation types across both sites include: *Calluna vulgaris* (hereafter known as *Calluna), Eriophorum vaginatum, Eriophorum angustifolium, Sphagnum spp., Polytrichum spp., Molinia caerula, and Rubus chamaemorus.* Sheep grazing occurs across both sites and typical summer stocking densities are 1.0 – 1.5 ewe $ha^{-1}$ (Fletcher et al., 2010).

### 2.2. Carbon stocks survey

Survey plots were selected from a sequence of plots burned between 1997 and 2009 from across the two sites (Emblehope and Ray Demesne). The location of each plot along with the year of burning was determined by Game and Wildlife Conservation Trust (GWCT) and land manager records made at the time of the burns. For each available burn year ~~multiple~~ duplicate ~~burned~~ plots were selected where one ~~burn~~ plot is an individual burn, typically 20m by up to 150m (Defra, 2007). Only two sites did not have duplicate burns – 1997 had only one site and 2005 had three sites. The following burn years were sampled where the figure in parentheses is the number of individual ~~burns~~ plots surveyed: 1997 (1); 1998 (2); 1999 (2); 2000 (2); 2002 (2); 2003 (2); 2004 (2); 2005 (3); 2006 (2); 2007 (2); 2009 (2). The ~~burned~~ plots surveyed from each year were selected at random from

a range of possible plots for that year except in case of burns from 1997 where there were so few plots that only one was considered.

Within each ~~burn~~ plot, 6 quadrats (50 cm x 50 cm) were positioned randomly along the longer dimension of any ~~burn~~ plot. Within ~~E~~each quadrat the major ~~was then surveyed by~~ plant functional ~~vegetation~~ type were recorded (vegetation was classified as *Calluna*, sphagnum mosses, non-sphagnum mosses, sedges, grasses and other shrubs); % of quadrat with vegetation cover; height of vegetation cover, presence of litter and presence of bare soil. From each quadrat samples of vegetation, 'stick' and litter were taken for analysis of carbon concentration. Vegetation was taken as the living above-ground biomass including the moss layer and vascular plants; 'stick' is the dead, standing biomass largely composed of the standing remains of burned *Calluna* shrubs; the litter is operationally defined as identifiable, unhumified, dead plant material that is unconsolidated.

Carbon stocks were calculated using the methodology of Clay and Worrall (2011) where literature values for moorland vegetation biomass, and their associated uncertainties~~,~~ (Forrest, 1971), were combined with cover data from the stocks survey and carbon concentration data from experimental analysis. Total carbon stocks for each burn year were calculated from a mean of all the plots for that year (e.g. n = 12) and the standard error estimated from that sample size. The carbon stock in the neighbouring unburned vegetation (termed control plot, see section 2.3) was also calculated allowing the burned plots to be expressed as a percentage of control plot carbon.

**Comment [gdc1]:** R2: The methods used to estimate above-ground carbon stocks seem lazy. The authors have used a crude allometric technique to estimate biomass despite the fact that cover alone is a poor predictor of biomass in shrublands. A range of more accurate non-destructive or destructive methods could have been used. At the very least the empirical equation used to predict biomass should have been tested/calibrated for their site

### 2.2.1. Carbon concentration analysis

Vegetation samples were analysed for their carbon (C) concentration on a Costech ECS 4010 Elemental combustion system with pneumatic autosampler. The elemental analyser was set up for CN analysis. Reactor 1 consisted of chromium (III) oxide/silvered cobaltous-cobaltic oxide catalysts @ 950°C. Reactor 2 consisted of reduced high purity copper wires @ 650°C. Helium was used as the carrier gas at a flow rate of 100 ml $min^{-1}$. A packed (Porous polymer, HayeSep Q) 3m GC column was used for separation of the gases.

## 2.3. Flux monitoring

For the on-going monitoring of carbon fluxes, for each selected year of burning, a pair of burned plots and a control plot were selected for instrumentation. The following burn years were monitored: 1999, 2000, 2002, 2003, 2004, 2005, 2007, and 2009 ~~(see section 2.4 for distribution of burn years between sites).~~ Due to logistcla

reasons at the time of burning plots from burn years 2000, 2004 and 2009 were only available on the Ray Demesne location and not at Emblehope. Plots from 1999, 2002, 2003, 2005, and 2007 were only available on Emblehope. Three controls plots were installed on Emblehope and three on Ray Demesne, in unburned vegetation next to burned plots. Note that this does not preclude the possibility that the control plots have never been burned, rather they have not been burned during the period of the chronosequence (i.e. since 1997) and represent the present state of the vegetation that would have existed locally when a particular burn took place. This design represents a blocked design where the following factors can be considered: burn year (from 1999 to 2009); site (Ray Demesne and Emblehope) and month of sampling (January - December).

**Comment [gdc2]:** The authors need to clearly describe the design of their monitoring strategy here rather than pointing the reader to a subsequent section. Understanding the monitoring strategy is key to assessing if the statistical treatment is valid and to interpreting the results. Crucially year is not independent of site but this is not made clear until later. The strategy for establishing control plots seems odd, the authors seem to suggest that controls are paired with burns as they "...represent the present state of the vegetation that would have existed locally when a particular burn took place." However there are only six control plots (3 on each site) and eight times since fire across the two sites. The authors seem to be assuming that the moor was a blank homogenous canvas of even age heather onto which burns have subsequently been place with the unburnt heather maturing around them. This is unlikely to be the case some burns, or some parts of some burns, may have occurred in heather 25 years old others in heather 15 years old. Again it would be better to consider the "controls" as a part of a wider chronosequence.

### 2.3.1. Plot scale monitoring measurements

Within each ~~burn~~ plot, or control, three dipwells and three crest fall runoff traps were installed. Where possible dipwells were inserted into the peat to a depth of at least 70 cm though as noted earlier not all dipwells could be inserted this deep and thus some dipwells could only be inserted ~40 cm. Depth to the water table was measured upon every visit and a sample of soil water was extracted at the same time - detailed analysis of the soil and runoff water composition data is presented elsewhere (Clay et al., 2012). Three ~~P~~permanently fixed circular gas collars were installed into the upper peat surface in close proximity to the dipwells. Monitoring started in December 2009 and continued until February 2011 i.e. a 15 month sampling window. Poor weather conditions prevented all plots from being visited in January 2010 and some plots during February and March 2010 and December 2010, therefore the maximum number of visits to some sites was 14 and the minimum was 8. The study was extended beyond the planned 12 months to 15 months so that a complete set of months could be sampled upon each plot.

In addition to the monthly monitoring, a meteorological station was installed on Emblehope that monitored air temperature, photosynthetically active radiation (PAR) and rainfall every 15 minutes. The station monitored data between February 2010 and February 2011 and was assumed to be representative of both Emblehope and Ray Demesne.

### 2.3.2. $CO_2$ flux monitoring

Surface exchange of $CO_2$ was measured by a closed chamber method (e.g. Rowson et al., 2010; Wickland et al., 2001). Monthly visits were made to each fixed collar and $CO_2$ was measured with an infra-red gas analyser (IRGA) (PP systems EGM-4, Hitchin, UK) which was fitted with a purpose built clear 20 cm tall, 15 cm

diameter acrylic closed chamber (PP-systems CPY-2 Canopy Assimilation Chamber). The chamber also included an air temperature probe and a photosynthetically active radiation (PAR) gauge. When using a clear chamber the measured flux was considered as the sum of photosynthesis ($P_g$) and net ecosystem respiration ($R_{eco}$), i.e. the net ecosystem exchange (NEE). In order to measure only $R_{eco}$ the chamber was covered to exclude all light - $P_g$ is therefore estimated from the difference between NEE and $R_{eco}$. Each flux measurement was for approximately 2 minutes with air temperature and PAR being recorded simultaneously. This study uses the convention that all fluxes are assessed relative to the atmosphere, therefore, $P_g$ is always negative in sign; $R_{eco}$ is always positive and NEE can be either positive or negative. Processed data was subjected to quality control by removing any $P_g$ values greater than zero; classing all negative flux values for $R_{eco}$ as zero; only allowing $P_g$ values to be calculated if there were two accepted values for $R_{eco}$ and NEE.

#### 2.3.3. Erosion pins

Surface recession was calculated using erosion pins, a commonly used method for determining erosion rates in peatlands (Evans and Warburton, 2007). Total surface change (i.e. either recession or accretion) was estimated from an array of 12 erosion pins. These erosion pins are 600mm lengths of 2mm diameter stainless steel inserted into the peat surface (at least 200mm). An array was installed on each burn year and on three of the control plots. The depth to the surface was measured approximately every 3 months during the monitoring period; however, sub-annual sampling can lead to significant noise thus total surface change was calculated from the difference in surface height after one year. Outliers were removed from the dataset using Grubbs' test at 95% confidence level thus allowing annual rates of erosion (cm $yr^{-1}$) to be calculated.

### 2.4. Statistical methodology

#### 2.4.1. ANOVA

The annual flux monitoring part of this study represents a blocked ANOVA design and as such this study can be considered initially as a three factor experiment: month, site and burn years. The site factor has two levels: Emblehope and Ray Demesne. Finally, the burn year has eight levels: 1999, 2000, 2002, 2003, 2004, 2005, 2007, and 2009. It should be noted that burn year is not independent of location as years 2000, 2004 and 2009 were only available on the Ray Demesne location and not at Emblehope. Likewise 1999, 2002, 2003, 2005, and 2007 were only available and monitored on Emblehope. Therefore, burn year is nested within the site factor.

**Comment [gdc3]:** R2: P6 Ls 10-27: I have concern about the use of erosion pins on peat as there is a risk that what they are actually measuring is peat shrinkage/contraction due to drying following the removal of the heather canopy and moss/litter layer. This should be assessed by taking measurements of bulk density.

**Comment [gdc4]:** P6 Ls 34-45: I'm not sure the statistical approach described here is appropriate. The experimental design seems to really require a mixed model. Site is certainly a random effect whilst month and year could be considered fixed or random depending on the question being asked. The authors have stated that year is nested in site but it's not clear they actually did this during their modelling. Though their results table suggests they did, their description of the results on p. 12 does not make this clear.

I also wonder whether the authors feel confident that they can test for differences between years when each burn year seems to have only been replicated twice? The authors have not tested for any interacting effects. I'm uncertain as to the real benefit of any statistical analysis it might be better to just graph the C flux data for each burn year in each site and provide a descriptive analysis.

The statistical significance of the independent factors and interactions was determined using a general linear modeling approach based on an analysis of variance (ANOVA) using MINITAB v14 software package. The magnitude of the effects, in this case generalized $\omega^2$ (Olejnik and Algina, 2003), of each significant factor and interaction were calculated. Post-hoc testing of the results was made for pairwise comparisons between factor levels using the Tukey test in order to assess where significant differences occurred. ANOVA was performed both with and without covariates to explore the relative importance of the factors and covariates and whether significant differences could be explained by inclusion of covariates. Wherever appropriate, the depth to the water table, temperature, PAR, and $P_g$ were used as covariates within the analysis.

There are several assumptions associated with using the ANOVA approach. Firstly, the Levene test was used to assess homogeneity of variance with respect to the factors in ANOVA; if this test failed then data were log-transformed. It should be noted that ANOVA is robust against the assumptions of homogeneity of variance and normality of the data. Secondly, in order to avoid type I errors all probability values are given even if significance was assessed at the 95% level. Finally, although the power of the design was not tested in this case the number of replicates and length of study are based upon the design of Worrall et al. (2007a) where the same number of replicates per plot, over the same number of plots, and for the same length of study was shown to have sufficient power such as to distinguish between different burn managements.

### 2.5. Carbon budget estimation

In order to calculate a complete annual carbon budget, the following fluxes are needed: $P_g$, $R_{eco}$, POC, DOC, $CH_4$ and dissolved $CO_2$ (Worrall et al., 2009a). The formulation of the budget can be found in Equation 5. Rainfall inputs of carbon are not included as inputs are deemed to either be negligible e.g. particulate organic carbon, or, in the case of DOC, not required as the DOC flux is calculated from direct loss of DOC from the soils and not at the catchment outlet (Worrall et al., 2009a)

#### 2.5.1. Surface exchange of $CO_2$ – $P_g$ and $R_{eco}$

The flux of gaseous $CO_2$ is best estimated by extrapolation methods i.e. estimating fluxes from driving variables such as temperature and water table. A common way of estimating NEE is by calibrating equations for $P_g$ and $R_{eco}$. For $R_{eco}$ the approach of Lloyd and Taylor (1994) was used:

$$R_{eco} = R_{10} e^{E_a\left(\frac{1}{283.15 - T_0} - \frac{1}{T - T_0}\right)} \quad (2)$$

Where: $R_{eco}$ is the respiration rate at temperature T; $R_{10}$ is the respiration rate at 283K; $E_a$ the activation energy; $T_0$ the reference temperature (227·13 K); T the ambient temperature (K).

For $P_g$ the method proposed by Bubier et al. (1998) was used:

$$NEE = \left( \frac{GP_{max} \alpha PAR}{\alpha PAR + GP_{max}} \right) + R \qquad (3)$$

Where α = initial slope of the rectangular hyperbola (also called the apparent quantum yield and a conversion factor between units), PAR = photosynthetically active radiation (µmol $m^{-2}$ $hr^{-1}$), $GP_{max}$±R=NEE asymptote (gC $m^{-2}$ $hr^{-1}$), and R = y-axis intercept (or dark respiration value, R < 0).

Both equations were calibrated against the observed $R_{eco}$ and $P_g$ data measured across the plots. The best-fit equation was then extrapolated to calculate an annual budget by using 15-minute logged temperature and PAR data for $R_{eco}$ and $P_g$, respectively. In line with the approach of Worrall et al. (2009a) and Worrall et al. (2011) the error in an annual budget calculated from an extrapolation of a fitted equation was taken from the error in the fit of that equation. In this case, the fit of Equation (3) gives an error in the annual budget of ±7.7%.

### 2.5.2. Dissolved organic carbon (DOC) flux

In this study DOC is operationally defined as those dissolved organic carbon compounds passing through a 0.45µm filter (Wallage and Holden, 2010). The material that does not pass through is usually defined as the particulate organic carbon (POC). Water budgets for each plot were not calculated as part of this study because each ~~burn~~ plot does not represent a catchment; therefore it is not possible to directly calculate DOC flux from these plots. In order to calculate a flux, an estimate for the amount of water leaving the site is needed. By combining the measured rainfall data from Emblehope with a runoff coefficients from Moor House in the North Pennines (Worrall et al., 2007b), it is possible to estimate the monthly flow of water from the catchment. Combining this water flow with soil water DOC concentration data (Clay et al., 2012), a monthly DOC flux can be calculated. As the rainfall gauge was not installed for Dec 2009 and January 2010, it is not possible to calculate an estimate of the DOC flux for those months. However the period February 2010 to February 2011

**Comment [gdc5]:** P8 Ls 37-40: To estimate DOC flux the authors use run-off coefficients from Moor House a site some 60 miles to the south. This seems hard to justify.

allows sufficient data for an annual flux to be calculated. Missing DOC concentration data was accounted for by fitting a two-way ANOVA to the DOC data using month and burn year as factors. It is then possible to estimate means for each burn year × month combination. This way the seasonal and plot-specific signals are preserved. Previous work on monthly sampling of DOC data suggests that the error in sampling is approximately ±14% (Worrall et al., 2011). However, it should be noted that it is not possible to estimate the error on the runoff coefficient used in these calculations.

### 2.5.3. Particulate organic carbon (POC) flux

Worrall et al. (2011) compared erosion pin data with direct measurement of POC flux through catchments in the Peak District and gave a value of 1 cm erosion is equivalent to a POC flux of 43 (±9) gC $m^{-2}$ $yr^{-1}$. Accreting plots by definition do not have a POC flux. However, accreting sites in the English Peak District have been found to have an average POC flux of 4 gC $m^{-2}$ $yr^{-1}$ but these were all peat-covered catchments where the POC flux was attributable to bank erosion and so no loss is from the soil surface (Worrall et al., 2009b) The estimates of surface recession detailed in section 2.3.3 are combined with this equivalence in order to estimate POC flux for each burn year. Upper and lower estimates of POC flux using the error in the equivalence by Worrall et al. (2011) i.e. ±9 gC $m^{-2}$ $yr^{-1}$, were calculated in order to calculate an error for POC for each burn year.

**Comment [gdc6]:** P8 Ls 1-19: POC flux is estimated using relationships derived for the Peak District, no consideration is given to whether this is justifiable

### 2.5.4. Methane

Methane was not measured as part of this study but it is possible to estimate $CH_4$ fluxes from relationships with potential drivers such as water table depth. Worrall et al. (2009a) detail an empirical relationship between water table depth and $CH_4$ flux calibrated for UK peatland sites:

$$\ln F = 4.12 - 3.9W_D \quad (4)$$

Where: F = molar flux of the molar flux of $CH_4$ (μmol $CH_4$ $m^{-2}$ $hr^{-1}$); and $W_D$ =depth to the water table (m). The error in equation 4 is ± 80% (Worrall et al., 2009a).

**Comment [gdc7]:** P8 Ls 24-41: CH4 is a very important flux, developing a crude estimate from water table depth alone is not appropriate.

### 2.5.5. Dissolved $CO_2$

Dissolved $CO_2$ in the water samples was not measured as part of this study. Dissolved $CO_2$ peatland waters has been shown to be a small proportion of total carbon budgets (e.g. Worrall et al., 2009a). Therefore an estimate of dissolved $CO_2$ was estimated from a stochastic estimate from within published ranges of dissolved $CO_2$ (0.2 – 15.1 gC $m^{-2}$ $yr^{-1}$; Dawson et al., 2004; Worrall et al., 2003; Worrall et al., 2009a). A mean value and error was calculated from 50 randomly selected values from this published range.

**Comment [gdc8]:** P8 Ls 45-55: I really don't know why the authors even bother to include dissolved CO2 as they're now just using random numbers.

### 2.6. Carbon budget

The carbon budget is defined as:

**Comment [gdc9]:** P10 Ls 1-7: Data for large parts of this equation feel, to me at least, like they're no better than guesswork. What we need with regards to peatland management is science - there's far too much policy being made on the basis of guesses and hunches already. The authors have some great data, they should present it for what it is. I really do look forward to seeing a revised paper.

$$F_C = P_g + R_{eco} + POC + DOC + dissCO_2 + CH_4 \quad (5)$$

Where: Where: $F_c$=the total flux (gC $m^{-2}$ $yr^{-1}$); $P_g$ = the annual flux of $CO_2$ through photosynthesis (gC $m^{-2}$ $yr^{-1}$); $R_{eco}$ = the annual flux of DOC through ecosystem respiration (gC $m^{-2}$ $yr^{-1}$); POC=the annual flux of POC (gC $m^{-2}$ $yr^{-1}$); DOC=annual DOC flux (gC $m^{-2}$ $yr^{-1}$); $dissCO_2$=the annual flux of dissolved $CO_2$ (gC $m^{-2}$ $yr^{-1}$); and $CH_4$=the annual methane flux (gC $m^{-2}$ $yr^{-1}$). The overall flux was calculated for the combination of uptake and release pathways given the preferred method and by convention a negative flux is considered as an uptake of carbon by the system. Budgets were also compiled for the control plots and a mean carbon budget for Emblehope and Ray Demesne controls was calculated.

In order to calculate the error of the budget estimate, an upper and lower limit for each pathway in equation 5 was calculated where the upper and lower limit were defined as ± standard error as defined in previous sections. One hundred values were chosen at random from these ranges and combined in equation 5. The resulting distribution of carbon budgets was then examined to give the inter-quartile range (IQR) of the total carbon budget.

### 2.7. Carbon budget meta-analysis

The results of this study were also considered using the Bayesian meta-analysis approach of Worrall et al. (2010). The meta-analysis approach enables separate studies to be considered alongside each other and means that the results can be interpreted in terms of the probability that prescribed burning will result in an increase in either the carbon or GHG budgets. Furthermore, the approach can be used to estimate the equivalent number of total carbon or GHG budgets the data represent even though each published study considered may not include a

consideration of all relevant carbon or GHG fluxes. Since the publication of Worrall et al. (2010) additional studies have been published upon the impact of burning upon peatland carbon and GHG fluxes (Helliwell et al., 2010; Clay et al., 2010; Chapman et al., 2010; Yallop and Clutterbuck, 2009; Yallop et al., 2010) and the results of these studies can be included alongside those from this work in order to update the meta-analysis. As with all previous studies, the results for each burn year for each measured pathway are considered relative to an unburned control and only an improvement relative to carbon or GHG storage is counted as positive, no change relative to the control is considered a negative result as well as a decline in carbon or GHG sink relative to an unburned control.

## 3. Results

### 3.1. Carbon stocks

Mean control above-ground biomass was 860 ± 35 g $m^{-2}$ (± standard error) and control above-ground carbon stock of 437 ± 18 gC $m^{-2}$. The above-ground biomass values are within the range reported in Davies et al. (2008) though somewhat smaller than other studies such as Kayll (1966) which estimate 1840 g $m^{-2}$ for 25-year old *Calluna*. In the burned plots mean above-ground biomass was 409 ± 44 g $m^{-2}$ whilst mean above-ground carbon stock in the burned plots was 201 ± 23 gC $m^{-2}$. Those plots that were covered by the monitoring campaign are shown in Figure 1. The recent burns had the lowest carbon stocks in the above-ground vegetation compared to the older burns; however, there are some burn years that have higher carbon stocks than might be expected. For example, 2004 has a carbon stock of approximately 315 gC $m^{-2}$, higher than the immediately preceding or successive years. Deriving a linear regression ($r^2 = 0.63$, $p < 0.05$, $n = 8$) from Figure 1, there is an average annual increase in above-ground carbon stock of approximately 20 gC $m^{-2}$ $yr^{-1}$ across the monitoring period. Figure 2 plots the total carbon stock for the monitored sites as a proportion of the neighbouring control sites in order to compare between years whilst accounting for site to site variation. At year 0 i.e. at burn, the regression suggests approximately 80% of the biomass is consumed by the burn. This figure is within typical range of biomass consumption in prescribed fires (Legg et al., 2010; Worrall et al., 2013). It should be noted that even the older sites i.e. 1999, do not reach 100% of carbon stock found in the control plots; the point at which 100% of the control carbon stock is recovered is 18.6 years since burning.

### 3.2. Carbon fluxes

#### 3.2.1. Measured $CO_2$ from annual monitoring

Overall 647 $CO_2$ flux measurements were determined (326 ecosystem respiration and 321 net ecosystem exchange) and following quality control 270 determinations of photosynthesis could be calculated. Photosynthesis ranged across all burn years and months from -2.80 to 0 $gCO_2\ m^{-2}\ hr^{-1}$ with 2009 burn year having the highest rate of photosynthesis (Table 1; Figure 3a). Emblehope and Ray Demesne controls had mean $P_g$ values of -0.33 and -0.17 $gCO_2\ m^{-2}\ hr^{-1}$ respectively (Figure 3a). Net ecosystem exchange ranged across the monitoring period from -1.31 to 0.60 $gCO_2\ m^{-2}\ hr^{-1}$. Recent burns (e.g. 2009 and 2007) have a negative NEE whereas older burns have a positive NEE (Table 1; Figure 3b). Net ecosystem exchange on the control plots was -0.03 and 0.09 $gCO_2\ m^{-2}\ hr^{-1}$ for Emblehope and Ray Demesne respectively. Note Emblehope had a negative NEE i.e. net $CO_2$ sink, and Ray demesne control had a positive NEE i.e. net $CO_2$ source (Figure 3b). Ecosystem respiration ranged from 0 to 2.16 $gCO_2\ m^{-2}\ hr^{-1}$. The highest $R_{eco}$ values were found on 2009 and 2000 burn year sites (Table 1; Figure 3c). Control plots $R_{eco}$ values were approximately equal with mean values of 0.224 $gCO_2\ m^{-2}\ hr^{-1}$ and 0.216 $gCO_2\ m^{-2}\ hr^{-1}$ for Emblehope and Ray Demesne respectively (Figure 3c). Applying linear regression to the measured $CO_2$ data showed that only NEE had a significant trend over the chronosequence ($r^2 = 51\%$, $p < 0.05$, $n = 8$) where there is a decrease in NEE towards more recent burn years; neither Pg or $R_{eco}$ showed significant linear regressions.

Photosynthesis and ecosystem respiration failed the Levene test so were log-transformed prior to ANOVA; net ecosystem exchange did not need to be transformed. All of the factors, (site, burn year and month) were significant factors for all of the $CO_2$ pathways (Table 2). Month of sampling explained more variation than burn year or site in all three cases ranging from 26% to 61% variance explained. Whilst the burn year factor explained less than month of sampling it still explained between 3 and 6% of the variation in all the datasets (Table 2). With the addition of covariates the predictive power of the ANOVA model increased (Table 2). Burn year and month of sampling were both significant with the inclusion of covariates; however, site variation was only significant for $lnP_g$ (Table 2b). lnPAR was significant for $lnP_g$ and NEE explaining 14% and 2% of the variation respectively whilst lnWT was significant for NEE explain 14% of the variation in the data. The only significant covariate for $lnR_{eco}$ was $lnP_g$ explaining 33% of the variation in the data (Table 2). Temperature was not found to be significant for any of the pathways.

Post-hoc testing of burn year for $CO_2$ data showed significant differences between burn years though importantly there was no systematic difference over time. For $lnP_g$, 2009 was significantly different from 1999 and 2002 where 2009 had the largest $P_g$ and 1999 and 2002 smallest $P_g$. Additionally 2000 was significantly different to 2002. For NEE, 2009 was significantly different from 2000, 2002 and 2005 where 2009 had the

largest negative $CO_2$ flux whilst 2000, 2002 and 2005 all had positive fluxes with 2002 having the largest positive NEE. Finally for $lnR_{eco}$ there were no significant differences between burn years or control plots at the 95% confidence interval. However, at 90% confidence level, there is a significant difference between 1999 and both 2000 and 2009, where 1999 has the lowest ecosystem respiration values.

### 3.2.2. Modelled $CO_2$ fluxes

Based on equation (2) the ecosystem respiration of $CO_2$ varied from 148 to 308 gC $m^{-2}$ $yr^{-1}$ (Table 3). These values are within the range of previous studies (Worrall et al., 2011; Clay et al., 2010) though the upper end of this range in this study is 50gC $m^{-2}$ $yr^{-1}$ higher than those found in Clay et al (2010). Variation between burn years shows recent burns having some of the highest $R_{eco}$ values and decreasing values over time until an increase in 2000 and 2002; there is no significant linear trend over the chronosequence for $R_{eco}$. Modelled ecosystem respiration values on the control sites are higher than any of the burn years with values of 375 and 347 gC $m^{-2}$ $yr^{-1}$ for Emblehope and Ray Demesne respectively.

Using equation (3), $P_g$ was estimated to be between -73.4 and -342.5 gC $m^{-2}$ $yr^{-1}$ (Table 3). These values again are within the range reported by Worrall et al. (2011) though, similar to the $R_{eco}$ results, the upper $P_g$ is higher than that from other prescribed burning experiments (Clay et al., 2010) by up to 140 gC $m^{-2}$ $yr^{-1}$ Comparing the fluxes between burn years, the $P_g$ data showed a significant linear trend over time with recent burns having the largest sink of carbon and the smallest sink on the oldest burn, 1999 ($r^2 = 0.712$, $p<0.05$, $n = 8$).

### 3.2.3. Surface change and POC flux

In total 460 erosion pin measurements were taken throughout the monitoring period across all burn years resulting estimates of surface change across different burn years. A surface change estimate for burn year 2005 could not be calculated due to missing erosion pins and damage to remaining pins. Mean surface change rates ranged between 0.76 cm $yr^{-1}$ (erosion) to -2.3 cm $yr^{-1}$ (accretion) across the burn years (Figure 4). Erosion was the dominant surface change in recent burns (2004, 2007 and 2009) whilst surface accretion was dominant in the older burns (1999 – 2003). There was a significant trend in surface change over the period ($r^2 = 0.61$, $p < 0.05$, $n = 7$). Post-hoc testing of the surface change data shows that both 2004 and 2009 burn years were significantly different from 2000 and 2003. The 2004 burn year was also significantly different from 2002.

Using the equivalence of 43 gC $m^{-2}$ $yr^{-1}$ per cm erosion, POC fluxes from the eroding sites were between 0 and 32 gC $m^{-2}$ $yr^{-1}$ (Table 3). Control sites on Ray Demesne showed surface accretion i.e. no POC losses, whilst the Emblehope control sites showed a small loss, 5 gC $m^{-2}$ $yr^{-1}$.

### 3.2.4. DOC flux

The DOC flux calculated from soil water DOC concentrations varied between 60.1 and 87.5 gC $m^{-2}$ $yr^{-1}$ (Table 3). These values are within, though at the higher end, of the range reported by Worrall et al., (2009a) when calculating DOC flux from shallow soil water and are also within the range reported by Worrall et al. (2011). Dawson et al. (2004) report DOC exports for a range of peat covered catchments of between 8.3 and 26.2 gC $m^{-2}$ $yr^{-1}$, though this study does not account for in-stream losses so would not be directly comparable to those DOC fluxes calculated here from soil water DOC concentrations. Fluxes of DOC from the control sites are 90.14 and 105.38 gC $m^{-2}$ $yr^{-1}$ for Emblehope and Ray Demesne respectively – higher than for the burned plots. When runoff water was considered as the source of the DOC concentration, then the DOC fluxes for the burned plots range from 31.4 to 71.1 gC $m^{-2}$ $yr^{-1}$ and the control plots in this case have DOC fluxes of approximately between 57.2 and 67.5 gC $m^{-2}$ $yr^{-1}$ (Table 3). Neither the soil water nor runoff water showed any significant trend in DOC flux over time. This is perhaps not unsurprising given that the underlying DOC concentration data also shows no significant trend over the chronosequence (Clay et al., 2012).

### 3.2.5. $CH_4$ emissions

Methane emissions varied between 4.98 and 6.33 gC $m^{-2}$ $yr^{-1}$ with control plot values on Emblehope of 5.77 gC $m^{-2}$ $yr^{-1}$ and on Ray Demesne of 4.23 gC $m^{-2}$ $yr^{-1}$ (Table 3). There was no significant trend across the period for the $CH_4$ emissions. It should be noted that these data were derived from depth to water table measurements that also showed no systematic trend over the monitoring period (Clay et al., 2012). These values are comparable to previous values for $CH_4$ in the carbon budget literature (Worrall et al., 2009b; Worrall et al., 2011). MacDonald et al. (1998) report values for $CH_4$ fluxes from a blanket bog in Scotland of between 0.16 and 13.5 gC $m^{-2}$ $yr^{-1}$.

### 3.2.6. Dissolved $CO_2$

The value selected from the stochastic modelling was 3.78 ± 0.28 gC $m^{-2}$ $yr^{-1}$; this value was applied equally across all the burn years and control plots (Table 3) and therefore is insensitive to differences between burn years.

### 3.3. Carbon budgets

The carbon budgets for all the burn years were all positive i.e. net sources, and ranged from 4 to 265 gC $m^{-2}$ $yr^{-1}$ with an inter-quartile range (IQR) of between 12 and 29 gC $m^{-2}$ $yr^{-1}$ (Table 3). The two most recent burns, 2007 and 2009 were the smallest net sources of carbon. The 2007 and 2009 plots were net gaseous sinks of $CO_2$ (i.e. negative NEE), however, high POC flux on 2007 and the DOC fluxes cancelled out this gain and moved the overall C budget to a net source. By comparing these values to Clay et al. (2010), it can be seen that most of the budgets are within the published ranges for sites managed by prescribed burning (Figure 5). Both the control budgets are higher than the upper limit in Clay et al. (2010). Overall there is a trend towards larger sources the older the stand age, however, this relationship is not significant at the 95% level ($r^2$ =0.434, p = 0.075, n= 8).

### 3.4. Meta-analysis

Combining the results of this study with those in Worrall et al. (2010) and published since means a much strengthened meta-analysis could be performed (Table 4). The data of this study means that the meta-analysis now has data from the equivalent of 16 complete carbon budgets and 11 GHG budgets; the discrepancy between these two is due to the greater importance of $CH_4$ flux to the GHG budget of peat soil and this study could not measure $CH_4$ flux directly. The available results now suggest that prescribed burning has a probability of improvement of 0.55 and 0.45 for C and GHG budgets respectively.

## 4. Discussion

This study has been able to produce an estimate of the carbon budgets from a chronosequence of burned site using a combination of measured and modelled data. Whilst there are some inherent assumptions and error, it does provide an estimate of carbon budgets under prescribed burning and more importantly tracks trends over the chronosequence. Additionally, the approach taken in this study is appropriate for the nature of the plots measured both in terms of scale e.g. the plots are small relative to the footprint of eddy-covariance towers so closed chamber approach was used, and statistical power e.g. multiple measurements over many plots. The carbon budgets for these sites were all sources and ranged from near neutral carbon balance (Burn year 2007:4 gC $m^{-2}$ $yr^{-1}$) to large net sources (Burn year 2002:265 gC $m^{-2}$ $yr^{-1}$). Existing literature shows a range of values for peatland carbon budgets. Worrall et al. (2003) measured a carbon budget of −15·4 gC $m^{-2}$ $yr^{-1}$, that was later updated (Worrall et al., 2009a) to a 13 year average of −56 gC $m^{-2}$ $yr^{-1}$. Elsewhere in the world peatlands have

found to be carbon sinks of approximately -20 gC $m^{-2}$ $yr^{-1}$ (Roulet et al., 2007; Nilsson et al., 2008). However, these values are for what may be termed unmanaged sites i.e. not damaged or restored sites. There are limited carbon budget studies on managed or restored peatlands sites. Rowson et al. (2010) suggested that an artificially drained peatland was a net source of up to 107 gC $m^{-2}$ $yr^{-1}$, while Clay et al. (2010) has shown that plots of peat soils subject to prescribed burning were a source of carbon of up to 118 gC $m^{-2}$ $yr^{-1}$. Recently Worrall et al. (2011) showed that damaged bare peat sites can have carbon budgets as high as 522 ± 3 gC $m^{-2}$ $yr^{-1}$ and that restoration can improve the carbon budget through a reduction in the size of the source i.e. an avoided loss. Other disturbances such as wildfires have been shown to significantly affect carbon stores in peatlands (Wieder et al., 2009). Prescribed fires occur in other ecosystems such as slash and burn in rainforests (e.g. Panosso et al., 2009) or fuel reduction burns in chaparral ecosystems (e.g. Potts et al., 2010) though this study cannot comment on the application of the results from this study to other ecosystems.

Whilst all the sites were carbon sources, the most recent burns (2007 and 2009) had the smallest sources and in both cases were net sinks of gaseous $CO_2$. In fact a small net carbon sink is within the error bounds for 2007. In comparison the control sites were also large sources and were not sinks of gaseous $CO_2$. One potential reason for the recent burn years acting as gaseous sinks could be the rapid growth of vegetation relative to the older vegetation; Johnson and Knapp (1993) found higher rates of photosynthesis along with increased above-ground biomass production on annually burned sites. It is then the hydrological export of carbon from the sites that moves these net gaseous sink sites into a net loss. Despite the level of interest in erosion from wildfire, there is relatively little research on erosion at the plot scale from prescribed burn sites. The key observation on erosion from this study is that the erosive nature observed on recent burns transitions to a non-erosion nature on older burns after approximately 5-6 years. It should also be noted that transfers into or out of plots may also play a significant role in the carbon budget of a site. As with the water table monitoring and DOC data (Clay et al., 2012), whilst burn year is a significant factor for the measured $CO_2$ fluxes, there was not a systematic change in the measured data over time across all pathways. Indeed in a complex natural system with interacting feedbacks, it is perhaps unsurprising that such variation occurs.

Despite all sites (burned plots and controls) being net sources, the net source represented by a burned plot is substantially less than that on a control plot. Therefore it may be possible to see whether any trade-offs exist where burning may represent a net carbon saving relative to 'no burning'. If a best fit line is calculated through Figure 2 it is possible to show that the average carbon stock left on a burned site immediately after the burn was 80 gC $m^{-2}$; that is the action of burning itself represents an 80% loss of above-ground carbon stock or

an average source of 360 gC $m^{-2}$. Furthermore, using Figure 2, the carbon stock will return to a level similar to control conditions after 18.6 years. Equally, the flux data for each burn year shows that the net source is significantly changing with time at an average rate of +15.7 gC $m^{-2}$ $yr^{-2}$; meaning that the net source represented by a burned plot would return to that of the control conditions (taking a mean of Emblehope and Ray Demesne of 291 gC $m^{-2}$ $yr^{-1}$) in 18 years. Over that period the average annual flux from a burned plot would be 192 gC $m^{-2}$ $yr^{-1}$ compared to the average for control, unburned plots of 291 gC $m^{-2}$ $yr^{-1}$. It is therefore clear that the difference in the average annual budget means that the reduced losses from the burned plots each year would very quickly make up for the losses during the burn itself. Clay et al. (2010) proposed that the difference in net sink size between burned and unburned plots was sufficient to offset releases from the burn. By accumulating the difference in the annual average carbon flux across the burn sequence to 18 years and even accounting for the loss of biomass in the burn itself shows that over an 18 year burn cycle the total avoided loss was 744 gC $m^{-2}$, equivalent to 41 gC $m^{-2}$ $yr^{-1}$. Therefore, even if allowing for complete above-ground biomass removal at the time of the burn, the burning still produces an avoided loss.

This study has allowed the meta-analysis of Worrall et al. (2010) to be updated. From the analysis ~~is~~ it would appear that there is a balance as to whether burning would bring about a carbon (or GHG) benefit. A prima facie interpretation could be equivocation and to say that a 50% probability means that we do not know, but a more reasoned explanation would be that the meta-analysis now better and represents the complete burning cycle where over the cycle the plot returns to being similar to the surrounding unburned ground.

This study measured the chronosequence of annual carbon flux and the biomass loss and development confirming that burning of *Calluna*-dominated landscapes avoids a loss of carbon. However, it is important to note that this management intervention leads to an avoided loss as the change is one in the magnitude of the source and not a transition back to sink – it would be better to convert *Calluna*-dominated ecosystems to species that are actively peat forming such as *Sphagnum*-dominated systems.

## 5. Conclusions

By monitoring and sampling a chronosequence of prescribed burning plots, this study has been able to construct complete carbon budgets over a 10 year chronosequence. Total carbon budgets for all sites were carbon sources but the youngest sites had the smallest sources. Gaseous exchange of $CO_2$ at these sites played an important role and key drivers of $CO_2$ exchange were identified. This study adds to the sparse literature on the effects of prescribed burning on peatlands and suggests that there is a more or less 50:50 chance of an improvement of

carbon budgets under prescribed burning. Over the chronosequence it is clear that burn management represents a technique for avoiding losses of carbon in *Calluna*-dominated peatlands. However, this benefit is an avoided loss and at no time was burning observed to lead to peat forming, or net carbon sink, conditions.

Whilst this study adds to the growing literature on the effects of burning on carbon budgets of upland peatlands in the UK, it also highlights a number of areas for further research including updated biomass measurements on UK moorland vegetation.

**Comment [gdc10]:** Added something but could do with some others

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Figure 1. Total above-ground carbon stock across sites surveyed against time since burn (mean ± standard error).

Figure 2. Plot of normalised carbon stocks for the sites monitored as part of the carbon flux monitoring campaign (mean ± standard error)

Figure 3. $CO_2$ fluxes (mean ± standard error) for control plots and burn year plots for a) Photosynthesis ($P_g$), b) Net ecosystem exchange (NEE), c) Ecosystem respiration ($R_{eco}$)

Figure 4. Mean erosion rates (cm $yr^{-1}$) for each burn year ± 1 standard error.

Figure 5. Carbon budgets, as calculated by equation 5, for control plots and each burn year (error bars are ± 1 standard error). Upper (red) and lower (green) limits of carbon budgets measured by Clay et al. (2010).

Table 1. Median values for Pg, $R_{eco}$ and NEE for each burn year. Values in parentheses show the inter-quartile range. n is the sample size

Table 2. ANOVA of ln-transformed $CO_2$ exchange for each factor and covariate a) without covariates b) with covariates. df = degrees of freedom, p = probability of factor being zero, $\omega^2$ = generalized proportion of variance explained and n.s. = covariate not significant. Factors and covariates taken as significant are shown in bold. Only significant covariates are included in the models in b).

Table 3. Summary of each carbon pathway for measured and modelled values of carbon fluxes, gC $m^{-2}$ $yr^{-1}$. Figures in parentheses are the standard error for each pathway in gC $m^{-2}$ $yr^{-1}$. For the total carbon budget, $F_c$, the figures in parentheses is the inter-quartile range (IQR) in gC $m^{-2}$ $yr^{-1}$

Table 4. The summary of studies results included in the meta-analysis for prescribed burning of peatlands.