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Carbon budgets of an upland blanket bog managed by prescribed fire

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[1] This study presents the carbon budget of a blanket bog, North Pennines, UK, subject to grazing and prescribed burning for vegetation management. The budget considers both fluvial and gaseous carbon fluxes and the following uptake and release pathways: dissolved organic carbon, particulate organic carbon, excess dissolved CO_2, release of methane (CH_4), net ecosystem respiration of CO_2, and uptake of CO_2 through primary productivity. Measurements of CH_4 were not directly measured as part of this study but were estimated from hydroclimatic variables measured within the study. The results show that, if management combinations were extrapolated across the catchment, then over a 3 year period, the catchment would be a net source of carbon of between 62 and 206 gC m^{-2} yr^{-1}. The action of both burning and grazing was to significantly decrease the magnitude of the carbon source relative to unburnt controls. Over the study period burnt sites were a mean source of approximately 117.8 gC m^{-2} yr^{-1} compared to unburnt sites with a mean source of 156.7 gC m^{-2} yr^{-1}. Even when including the loss of carbon during the vegetation combustion, there are conditions under which the long-term loss of carbon is less than if no burning had occurred. If total combustion of vegetation occurs, provided burning occurs at cycles longer than 32 years, then less carbon is predicted to be lost than in a no-burn scenario.

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1. Introduction

[2] Worldwide peatlands cover between 3.86 and 4.09 million km^2 [*Immirzi et al.*, 1992], and many are found in the northern latitudes. Interest in peatlands has increased in recent years due to their importance in preserving and enhancing stores of terrestrial carbon. Carbon accumulation in these northern peatlands is a balance between primary productivity and decomposition of organic matter. Historically, northern peatlands have been a store of carbon, and *Turunen et al.* [2002] estimate that approximately 270–370 GtC is stored in boreal and subarctic mires. The long-term rate of accumulation of carbon during the Holocene has been estimated at 15–30 gC m^{-2} yr^{-1} [*Roulet et al.*, 2007; *Turunen et al.*, 2002; *Vitt et al.*, 2000]. However, under a predicted warming climate [*IPCC*, 2007], these sensitive areas could be converted from net sinks to net sources of atmospherically active carbon.

[3] Increases in air temperature as a result of climate change could lead to increased release of CO_2 from ecosystem respiration [*Dorrepaal et al.*, 2009], a greater number of droughts leading to enzymic latch mechanisms [*Freeman et al.*, 2001], and increased water table drawdown [*Christensen et al.*, 1998]. An increased drawdown of water tables can lead to greater dissolved organic carbon (DOC) export [*Strack et al.*, 2008] or when combined with enhanced CO_2 levels could lead to further destabilization of the carbon store [*Ellis et al.*, 2009]. Because of these complex and often interacting processes, it is increasingly important, for the long-term stability of peatlands, to gain an understanding of carbon budget dynamics.

[4] Many areas of the Northern Hemisphere peatlands are subject to land management systems that have not always been conducive to carbon storage [*Holden et al.*, 2007]. Land management practices represent both a threat and an opportunity with respect to the carbon budgets of peat soils. A threat because the management may damage the peat, e.g., enhance erosion, and cause a decrease in the magnitude of the carbon sink or even convert the peat soil to a net source of carbon. Conversely, land management can also represent an opportunity to improve carbon uptake. Management practice can be optimized to help mitigate the consequences of climate change. In the UK upland peat catchments are managed predominantly for sheep grazing and/or for recreational hunting. One common management practice used in order to increase productivity for sheep and/or grouse is the use of managed burning. In an assessment of the extent of managed burning in the English uplands, *Yallop et al.* [2006] showed that approximately 40% of the uplands surveyed showed evidence of managed burning

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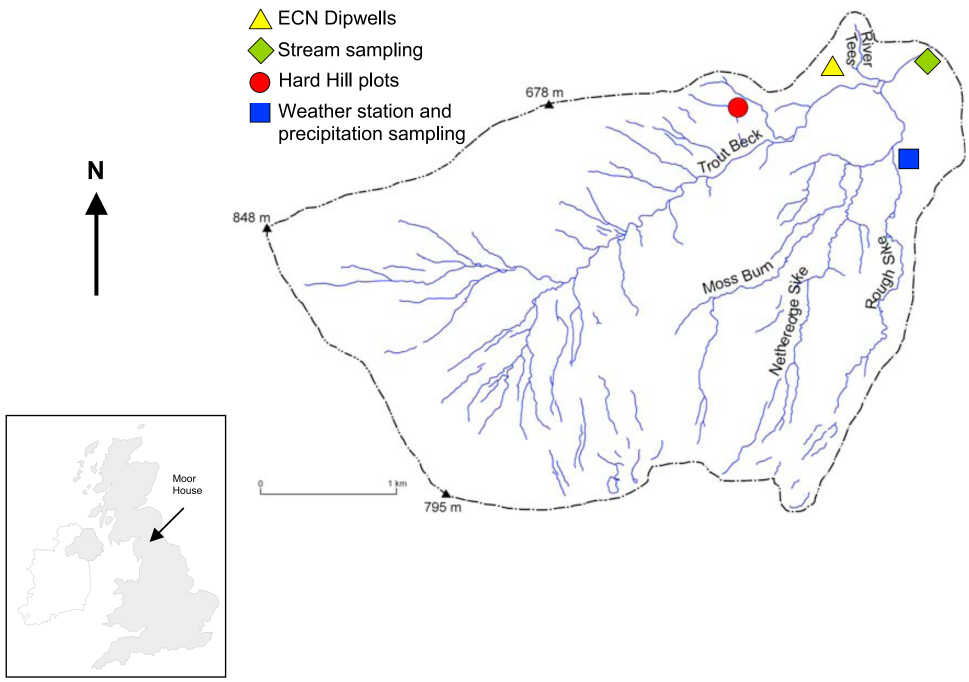

Figure 1. Location of Trout Beck catchment and monitoring locations used in this study.

and, that in the 4 years leading up to the year 2000 over 17% of English uplands had been burnt. Therefore, it is important to understand how managed or prescribed burning may alter the carbon budget of peat soils and how it might be manipulated to enhanced carbon storage.

[5] To calculate carbon budgets in peatland, two methodological approaches are often used. The first uses a range of dating methods to calculate ages of peat to estimate the rate of carbon accumulation [*Schlesinger*, 1990; *Tolonen and Turunen*, 1996]. Using this method *Garnett et al.* [2000] examined peat accumulation of carbon in three treatments (grazed/unburnt, grazed/burnt, and ungrazed/unburnt) on blanket bog vegetation. Recalculating the data of *Garnett et al.* [2000] based upon all of their data shows that the mean difference between burnt and unburnt treatments was 2.3 kg m^{-2} (not 2.48 as reported), this gives a mean effect of burning of 55 Mg C km^{-2} yr^{-1} (not 73 Mg C km^{-2} yr^{-1} as reported). *Kuhry* [1994] has suggested reduced peat accumulation in Boreal forests due to natural wildfires. However, these studies are not carbon accumulation studies rather they are peat accumulation studies, and they do not consider the presence of different carbon types, e.g., char versus humic carbon. Indeed, this first approach to measuring carbon budgets of peatlands technique can only calculate average accumulation rates above a horizon and does not account for periods of carbon loss. Furthermore, such long-term accumulation studies cannot account for greenhouse gas emissions because they cannot judge between the different forms of carbon exchange.

[6] The second approach is to calculate present-day carbon budgets based on fluxes of carbon through various pathways. *Worrall et al.* [2003] provide the first carbon balance for a UK upland peatland by using a North Pennine catchment, the Trout Beck, as their study site. Pathways included in calculating the carbon budget are rainfall dissolved inorganic carbon and DOC, CO_2 exchange, methane (CH_4) emissions, DOC export, particulate organic carbon (POC) export, dissolved inorganic carbon and dissolved CO_2, and input from weathering of underlying strata. A number of studies of contemporary carbon budgets have now been published [*Nilsson et al.*, 2008; *Roulet et al.*, 2007; *Worrall et al.*, 2009]. However, all these studies were for intact, or what might be considered active peat forming areas, i.e., these budgets were not for managed, damaged, or restored peat soils, and no budget exists for managed, damaged, or restored sites.

[7] Therefore, this study aimed to measure, or estimate, all of the carbon pathways as possible for areas under managed burning and grazing in order to make the best possible estimate of carbon budgets under burning and grazing regimes.

2. Materials and Methods

2.1. Study Site

[8] The field experiments in this study were conducted at Moor House National Nature Reserve (NNR) in the North Pennines. Within the reserve lies the Trout Beck catchment, a headwater of the River Tees with the entire catchment lying within the NNR. The Trout Beck catchment lies largely above 500 m OD (Figure 1). The underlying geology is a succession of Carboniferous limestones, sands, and shales with intrusions of the doleritic whin sill [*Johnson and Dunham*, 1963]. This solid geology is covered by glacial till whose poor drainage facilitated the development of blanket peat.

[9] Meteorological measurements began at Moor House in 1930s and continue today through an automatic weather

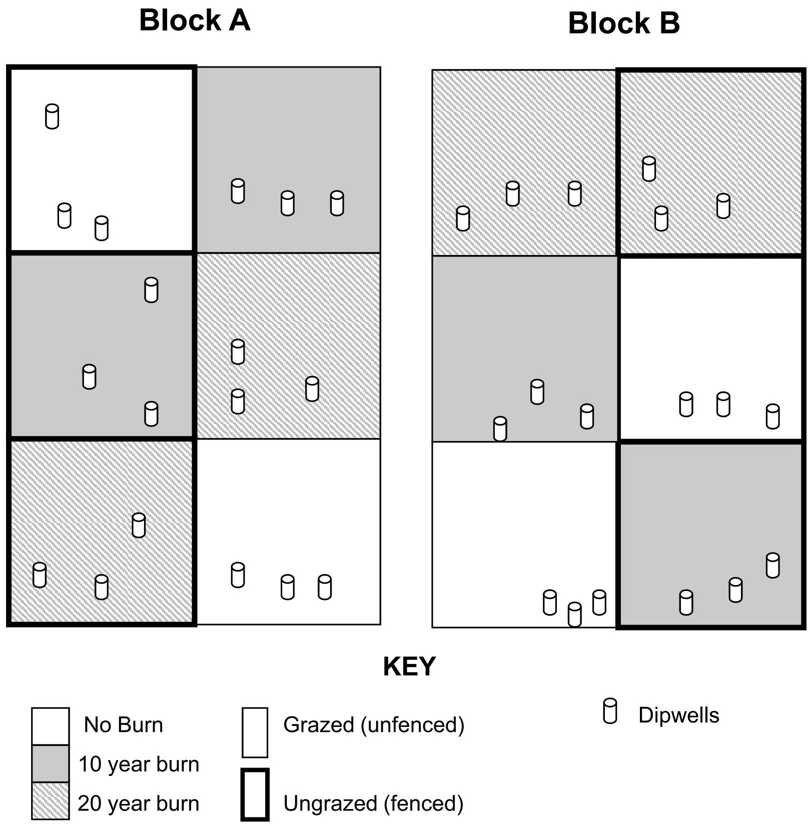

Figure 2. Plan of the Hard Hill plots.

station setup in 1991. The mean annual temperature (1931–2000) is 5.2°C; air frosts are recorded on over 100 days in a year (1991–2000 data, *Holden* [2001]). Mean annual precipitation (1953–1997) is 1953 mm [*Burt et al.*, 1998] with snow representing a noteworthy proportion: annual average snow cover at 500 m is 55 days [*Archer and Stewart*, 1995].

[10] In 1954, an experiment was set up at Hard Hill within the Trout Beck catchment at Moor House Nature Reserve to examine the ecological effects of traditional heather burning (national grid ref. NY 756326, N54:41:18 W2:22:45; Figure 1). As part of the design, grazing was also included as an experimental factor. Four blocks were set up, with each block subdivided into six plots, three of which were fenced off to prevent grazing and three left open to grazing. For practical reasons the fencing of exclosures was not randomly assigned. All plots were burnt in 1954 and then randomly

Table 1. Number of Pin Hits in the Plots in Blocks A and B at the Hard Hill Burning Experiment in 2001

	Block A					
Species	Graze, No Burn	Graze, 10 Years	Graze, 20 Years	No Burn, No Graze	No Graze, 10 Years,	No Graze, 20 Years
Litter	237	148	120	193	160	155
Calluna vulgaris	159	61	70	158	99	106
Eriophorum vaginatum	60	67	60	61	106	94
Rubus chamaemorus	5	13	5	5	22	1
Eriophorum angustifolium	5	83	89	14	66	113
Empetrum nigrum ssp *nigrum*	0	2	1	9	11	26
Others	35	155	107	75	101	94
Total	501	529	452	515	565	589
	Block B					
Species	Graze, No Burn	Graze, 10 Years	Graze, 20 Years	No Burn, No Graze	No Graze, 10 Years	No Graze, 20 Years
Litter	149	147	146	152	160	132
Calluna vulgaris	136	76	63	148	91	58
Eriophorum vaginatum	72	79	69	72	82	51
Rubus chamaemorus	27	0	3	13	4	5
Eriophorum angustifolium	2	95	95	3	49	74
Empetrum nigrum ssp *nigrum*	9	0	0	1	13	0
Others	98	93	67	49	51	84
Total	493	490	443	438	450	404

assigned a burning regime: no further burning, burnt every 10 years, or burnt every 20 years (Figure 2). This study focuses on two of the four experimental blocks; therefore, all management combinations were examined in duplicate.

[11] The vegetation is dominated by *Calluna vulgaris* (heather) and *Eriophorum vaginatum* (cotton grass), with *Sphagnum* spp. (a moss genus) present in some areas. The vegetation at the Hard Hill burning experiments have been surveyed several times since being set up in 1954, and *Adamson and Kahl* [2003] report on the methods used in the vegetation surveys. Data from the 2001 surveys (J. K. Adamson, personal communication, 2007) show that across the four plots, only seven higher plants were present, and on blocks A and B, the treatment blocks used in this study, only five higher plants are observed (*Calluna vulgaris*, *Eriophorum vaginatum*, *Rubus chamaemorus*, *Eriophorum angustifolium*, *Empetrum nigrum* ssp. *nigrum*; Table 1). The "Others" category included mosses, liverworts, and lichens.

[12] The catchment is grazed by sheep at a density of between 0.6 and 1 sheep per hectare though at the experimental plots grazing is estimated to be less than 0.1 sheep per hectare [*Adamson and Kahl*, 2003]. The entire study area has not been burnt since 1954 [*Garnett et al.*, 2000].

[13] The 10 year cycle plots were burnt for the fifth time since the experiment was set up on 6 February 2007 by a staff trained and experienced in heather burning. The conditions of the day were conducive to a cool, quick burn [*Scottish Executive Environment and Rural Affairs Department*, 2001], i.e., a burn where only aboveground vegetation is burnt without damage to litter or soil layers: clear day, low moisture on ground except frost, light northerly winds down the slope.

[14] Within these plots the following data have been measured: DOC concentration [*Clay et al.*, 2009a; *Worrall et al.*, 2007b], water table depth [*Clay et al.*, 2009b], soil water composition [*Clay et al.*, 2010] and acidity for dissolved CO_2, and surface exchange of CO_2. Two other carbon pathways have been studied at the Hard Hill plots in other works: POC [*Clement*, 2005] and CH_4 [*Ward et al.*, 2007].

[15] Soil water was accessed via a series of dipwells. Soil water sampling started on 6 April 2005 and initially included no burning and 20 year rotation for both grazed and ungrazed plots. This was later extended to include the 10 year burning rotation plots on 1 June 2005. In each plot, three dipwells were inserted in the peat to a depth of at least 90 cm. Depth to water table was measured at least once a month until the managed burn on the 10 year plots on 6 February 2007. The dipwells were removed from these plots on the day of the burn and returned to the same positions immediately after the burn. The monitoring continued at least monthly from then until 29 January 2008; therefore, the study considered 33 months of data with at least 1 year of sampling before and after a burn; in total, there were 59 sampling visits to the site.

[16] In order to gain a better understanding of the carbon fluxes at Hard Hill, CO_2 monitoring was initiated in October 2006 when permanently fixed gas collars were installed on the sites. Initially, two collars per plot were installed due to resources limitations; however, this was expanded to three per plot in late spring 2007. Gas collars were inserted into the upper peat surface in close proximity to the dipwells. No readings were taken for the 2 weeks after installation in order to allow the disturbed peat to settle back. Gas measurements were then taken at least once a month between October 2006 and January 2008.

[17] Environmental Change Network (ECN) maintains a flow gauging station within the Trout Beck catchment with river discharge measured hourly and a meteorological station is situated within the catchment with hourly recording of rainfall, air and soil temperature, and solar radiation (Figure 1). Continuous water table measurements of six dipwells (every 15 min) have been made since 1994 using pressure transducers that are calibrated weekly. ECN collects weekly samples of precipitation and stream waters; soil solution samples are taken at 2 week intervals. These samples are analyzed for major anions and cations, and methods of analysis are detailed by *Sykes and Lane* [1996].

2.2. Budget Calculation

[18] Carbon budgets can be calculated for each management regime by taking values from those plots and extrapolating to the catchment scale.

[19] Fluxes were calculated in two ways: interpolation and extrapolation. Interpolation constructs new data points within the range of known discrete data. There are many interpolation methods available [e.g. *Littlewood*, 1995]; however, a commonly used approach in carbon budgets of upland peats [*Rowson*, 2007; *Worrall et al.*, 2003; *Worrall et al.*, 2009] is "Method 5" of *Littlewood et al.* [1998]:

$$L_5 = \left(\frac{K \sum_{i=1}^{n} (Q_i C_i)}{\sum_{i=1}^{n} Q_i} \right) \overline{Q}, \tag{1}$$

where K is the conversion factor for period of sampling, C_i is the concentration of determinand in sample i, Q_i is the flow corresponding to sample taken on day i, $\overline{Q}$ is the mean river discharge over the period, and n is the number of samples. This interpolation method was used for calculating the flux of DOC, POC, and dissolved CO_2.

[20] Extrapolation methods rely on their being strong, significant relationships between the measured component of the carbon flux and one or more readily measurable environmental driver variables (e.g., air temperature). The significant relationship can then be applied to predict fluxes beyond and within the range of current observations. If the drivers are known over a period of time and provided there has been calibration between drivers and the determinand, then annual fluxes can be calculated. Strong relationships exist for net ecosystem respiration (NER), primary productivity (PP), and CH_4 that allow extrapolation methods to be used.

2.3. Dissolved Organic Carbon

[21] For each treatment type DOC concentration (mgC L^{-1}) in soil water was measured colorimetrically using the method of *Bartlett and Ross* [1988]. For further details of these measurements refer to the study by *Clay et al.* [2009a]. In order to estimate the annual flux of DOC, the DOC concentrations measured within each study treatment is assumed to be true for the whole catchment and then the

equivalent flux can be calculated by assuming that all the water flow measured at the catchment outlet has passed through the peat soil as if it was under each specific study treatment: this is the approach adopted by *Worrall et al.* [2009] in order to account for in-stream losses of DOC. This is probably an overestimate as it assumes no dilution from groundwater or rainwater bypassing the soil profile or any in-stream conversion processes though it has been shown to be a suitable method that adequately describes the source of the DOC leaving the catchment [*Worrall et al.*, 2006].

2.4. Particulate Organic Carbon

[22] No direct POC measurements were taken at Hard Hill; however, a previous study on the plots [*Clement*, 2005] measured suspended sediment concentrations (SSC) from rainfall simulation studies. By assuming 50% of the estimated values for SSC is in the form of carbon, values for POC can be estimated.

2.5. Dissolved CO_2

[23] Excess dissolved CO_2 is defined as the amount of dissolved CO_2 found in the water above that which would be expected to be present if the water were in equilibrium with the atmosphere. Excess dissolved CO_2 content of soil waters was calculated using the method by *Neal and Hill* [1994] and based on acidity; pH; calcium, aluminum, and DOC concentration; and temperature. Calcium and aluminum concentration and pH were measured as part of the monitoring of the Hard Hill sites [*Clay et al.*, 2010], whereas ECN soil temperature was used as a proxy for water temperature. Acidity measurements were made at Hard Hill on 16 August 2007. Data from this date, in conjunction with additional data from Bleaklow, Peak District, UK [*Billett et al.*, 2010], were used to construct a model to predict dissolved CO_2 based on existing water quality parameters, e.g., pH. A significant linear regression model was developed that found effects with only pH and aluminum concentration:

$$\mathrm{dissCO_2} = 0.398\mathrm{pH} - 0.69\log(\mathrm{Al}) - 1.31 \quad r^2 = 0.38 \quad n = 33, \tag{2}$$

where $\mathrm{dissCO_2}$ is dissolved CO_2 concentration in mgC L^{-1} and Al is aluminum concentration in mg L^{-1}. The annual flux of dissolved CO_2 for each of the study treatments is then calculated as for the DOC flux, i.e., scaled as if measured concentrations were true for the entire Trout Beck catchment and then flows at the catchment outlet were used to calculate flux.

2.6. Surface Exchange of CO_2

[24] By convention, gaseous fluxes were defined as release to the atmosphere as being a positive flux and uptake from the atmosphere as being a negative flux. The net ecosystem respiration (NER) is therefore defined as the total amount of CO_2 released from the peat surface (gCO_2 m^{-2} hr^{-1}). Primary productivity (PP) is the total amount of CO_2 (in gCO_2 m^{-2} hr^{-1}) taken up by the plants at the peat surface. The difference between these two fluxes is the net ecosystem exchange (NEE) and is the overall release or uptake of carbon from the peat system: the NEE flux can be either positive or negative. The NEE is defined as

$$\mathrm{NEE} = \mathrm{PP} - \mathrm{NER}. \tag{3}$$

CO_2 fluxes were measured using an infrared gas analyzer (IRGA) (PP Systems, EGM-4, Hitchin, UK) placing upon each fixed collar, ensuring a tight fit, and letting the IRGA take a reading of the change in CO_2 concentration over a period of 2 min. Measurements are taken in the dark (with cover) in order to measure NER and taken in the light (no cover) to measure NEE.

[25] Because of a limited number of readings prior to the burn in February 2007, the calibration of the NER and PP, described in sections 2.6.1. and 2.6.2., is based primarily on data from 2007 and on the 10 year plots, from after the burn.

2.6.1. Respiration

[26] In order to estimate the fluxes of CO_2 a commonly used approach is that of *Lloyd and Taylor* [1994] who link net ecosystem respiration to temperature. This study, however, uses the approach of *Lloyd* [2006] and *Rowson* [2007] who have identified depth to the water table as a significant factor in controlling net ecosystem respiration. Air temperature is measured at the time of CO_2 reading by the IRGA, and coincident water table measurements were made using the dipwell network installed on the treatment plots.

[27] In order to extrapolate CO_2 fluxes for the annual flux a long-term temperature and water table record were needed. Environmental Change Network monitor air temperature each hour; however, in order to create a continual water table record, water table measurements from Hard Hill were calibrated against ECN dipwell data. These dipwell data are measured hourly at dipwells instrumented with pressure transducers on the Moor House site (Figure 1).

[28] In order to calculate NER, the approach of *Lloyd and Taylor* [1994] has been modified to include a water table function [*Lloyd*, 2006; *Rowson*, 2007]:

$$R = (A \times \mathrm{WTD} + B)e^{E_0\left(\left(\frac{1}{283.15-227.13}\right)-\left(\frac{1}{T_{\mathrm{soil}}-227.13}\right)\right)}, \tag{4}$$

where R is the gross flux value (gCO_2 m^{-2} hr^{-1}), E_0 is a unitless constant, and T_{soil} is the soil temperature (K); ($A \times$ WTD + B) is the "R_{10} value" where WTD is depth-to-water table (mm) and A and B are constants for that treatment.

[29] By combining a calibrated equation (4) for each treatment, water table record, and temperature record, an estimate of the annual NER for each treatment could be made.

2.6.2. Primary Productivity

[30] One of the most commonly used techniques to calculate primary productivity is to link it to photosynthetically active radiation (PAR). *Bubier et al.* [1998] show a relationship between PP and PAR in the form

$$\mathrm{PP} = \left(\frac{\mathrm{GP_{max}}\alpha\mathrm{PAR}}{\alpha\mathrm{PAR} + \mathrm{GP_{max}}}\right), \tag{5}$$

where α is the initial slope of the rectangular hyperbola (also called the apparent quantum yield), GP_{max} = NEE asymptote (gC m^{-2} yr^{-1}), and PAR = photosynthetically active radiation (μmol m^{-2} hr^{-1}).

[31] Strong relationships exist between PAR and solar radiation [*Ross*, 1981] and as a long term record of PAR was not maintained at the site, for the purposes of flux prediction a best fit calibration between PAR and solar radiation was used [*Worrall et al.*, 2009]. Solar radiation records area measured every 15 min and calibrated against PAR readings taken at the time of the IRGA readings:

$$\begin{array}{ll} \text{If} & S > 0, \mathrm{PAR} = 19.39 + 1.79S \quad r^2 = 0.82 \quad n = 8760 \\ \text{Else} & S = 0, \mathrm{PAR} = 0, \end{array} \tag{6}$$

where S is the solar radiation (W m^{-2}) and PAR is the photosynthetically active radiation (μmol m^{-2} hr^{-1}).

2.7. Rainfall Carbon

[32] The annual input of carbon from precipitation was calculated from in rainfall samples collected as part of the ECN monitoring at Moor House and totaled using rainfall volumes. It was assumed that rainfall was in equilibrium with the atmosphere, i.e., no excess dissolved CO_2, and negligible amount of POC would be present.

2.8. Methane

[33] Methane was not measured directly as part of this study. One common approach to calculate CH_4 flux is by considering its relation to water table depth, a common driver of methane emissions [*Moore and Dalva*, 1993; *Moore and Roulet*, 1993; *Roulet et al.*, 1993]. For methane measurements from peat, a statistically significant relationship between water table depth and CH_4 flux has been found [*Worrall et al.*, 2009]:

$$\ln F = 4.12 - 3.9\mathrm{WTD}, \tag{7}$$

where F is the molar flux of CH_4 (μmol CH_4 m^{-2} hr^{-1}) and WTD is the depth to water table (m). This study used the long-term, calibrated water table record used in the calculation of the NER flux in combination with equation (7) in order to calculate the annual CH_4 flux for each treatment.

2.9. Carbon Budget

[34] To calculate the carbon budget of the Hard Hill plots, the method of *Worrall et al.* [2009] was used. The total magnitude of the carbon sink can be calculated thus:

$$F_c = \mathrm{PP} + \mathrm{NER} + \mathrm{POC} + \mathrm{DOC} + \mathrm{dissCO_2} + \mathrm{CH_4}, \tag{8}$$

where F_c is the total flux of the catchment (gC m^{-2} yr^{-1}), PP is the primary productivity within the catchment (gC m^{-2} yr^{-1}), NER is the net ecosystem respiration within the catchment (gC m^{-2} yr^{-1}), POC is the annual flux of POC (gC m^{-2} yr^{-1}), DOC is the annual flux of DOC (gC m^{-2} yr^{-1}), dissCO_2 is the annual flux of dissolved CO_2 (gC m^{-2} yr^{-1}), and CH_4 is the annual methane flux (gC m^{-2} yr^{-1}). By convention a negative flux is an uptake of carbon by the system.

[35] In calculating the loss of carbon from the peat soils within a catchment, this budget does not include rainfall DOC or inorganic carbon flux. If the loss of DOC from peat soils is estimated by using shallow water soil composition rather than using catchment outlet samples, it is not necessary to consider the rainfall input separately and only the excess dissolved CO_2 is necessary in inorganic carbon flux estimation [*Worrall et al.*, 2009].

[36] The carbon budgets calculated above are for total carbon. Therefore, the following steps were carried to convert budgets to metric ton CO_2 equivalent yr^{-1}: (1) mass C were converted to mass CO_2, (by multiplying by a factor of 3.67); (2) CH_4 is a stronger greenhouse gas than CO_2 and has a different atmospheric residence time. This study uses a factor of 24 to convert methane fluxes to CO_2 equivalents [*Houghton et al.*, 1995]; (3) allowance made for in-stream losses based from *Worrall et al.* [2006] where this study has assumed a 40% loss of the DOC flux but that the POC flux is not atmospherically active. Therefore, this study takes a conservative viewpoint and the following is proposed

$$\mathrm{CO_{2equi}} = \mathrm{CO_{2resp}} + \mathrm{CO_{2CH_4}} + 0.4\mathrm{CO_{2DOC}} + \mathrm{CO_{2dissCO_2}} - \mathrm{CO_{2PP}}, \tag{9}$$

where CO_{2equi} is the total carbon budget of the area (g equivalent CO_2 m^{-2} yr^{-1}), CO_{2PP} is the annual primary productivity (g equivalent CO_2 m^{-2} yr^{-1}), CO_{2resp} is the annual net ecosystem respiration of CO_2 (g equivalent CO_2 m^{-2} yr^{-1}), CO_{2CH4} is the annual methane respiration (g equivalent CO_2 m^{-2} yr^{-1}), CO_{2DOC} is the annual DOC production (g equivalent CO_2 m^{-2} yr^{-1}), and $CO_{2dissCO2}$ is the annual dissolved CO_2 flux (g equivalent CO_2 m^{-2} yr^{-1}).

[37] It should be noted that nitrous oxide (N_2O) is a powerful greenhouse gas released from peats that is not estimated in this approach.

[38] Carbon budgets are calculated for the period April 2005 to December 2007, i.e., there was not a complete year of data available for 2005. Therefore, in order to compare 2005 with the complete years of 2006 and 2007, the contribution of the period January–March to the yearly budgets of 2006 and 2007 were calculated. This contribution was then used to scale up the existing data for 2005. This method was chosen as it would capture the seasonal variations rather than a simple scaling by 3 months.

[39] In order to assess any significant differences in the total carbon budget between treatments, analysis of variance (ANOVA) was carried out. The field site design and sampling used in this study represents a factorial approach to the problem of understanding the consequences of heather burning and grazing [*Winer*, 1971]. The statistical significance of the independent factors was determined using a general linear modeling approach based on an analysis of variance using the commercially available MINITAB v13 software package. This study can be considered as a three factor experiment: year, burning regime, and grazing. The first factor, year of carbon budget, has three levels, 2005, 2006 and 2007. The second factor, the burning regime, has three levels: no burning, 10 years, and 20 years. The third factor, the grazing, has only two levels: grazed and ungrazed. Post hoc testing of the results is made for comparisons between factor levels using the Tukey test in order to assess where significant differences lie between factor levels.

2.10. Carbon Fluxes Versus Carbon Stocks

[40] If long-term carbon and greenhouse gas budgets for managed burns are to be estimated, then account must be made of the carbon losses during the burns themselves. The magnitude of this loss depends on the severity of the fire and

Table 2. Summary of Each Carbon Uptake and Release Pathway for Each Year (2005–2007) for Measured and Modeled Values of Total Carbon (gC m^{-2} yr^{-1})

2005	Graze, No Burn	Graze, 10 Years	Graze, 20 Years	No Burn, No Graze	No Graze, 10 Years	No Graze, 20 Years
PP	−171.33	−171.57	−148.92	−111.52	−190.31	−195.83
ER	200.44	167.30	136.58	203.95	191.76	209.87
DOC	53.44	50.12	48.17	49.25	50.46	48.21
POC	41.63	27.75	17.56	14.01	22.93	46.19
CH_4	5.58	6.51	6.86	5.81	6.18	6.53
DissCO_2	2.47	1.85	1.79	2.35	1.99	1.98
Total	132.22	81.96	62.03	163.86	83.01	116.95
2006	**Graze, No Burn**	**Graze, 10 Years**	**Graze, 20 Years**	**No Burn, No Graze**	**No Graze, 10 Years**	**No Graze, 20 Years**
PP	−174.89	−174.78	−149.92	−111.27	−194.26	−198.68
ER	188.43	177.76	146.69	176.28	224.43	258.68
DOC	67.16	73.39	66.14	66.23	63.30	75.26
POC	53.95	35.96	22.33	18.15	29.16	61.58
CH4	5.25	6.11	6.49	5.52	5.79	6.17
DissCO_2	3.57	3.17	2.91	3.35	2.72	3.02
Total	143.48	121.60	94.65	158.26	131.14	206.02
2007	**Graze, No Burn**	**Graze, 10 Years**	**Graze, 20 Years**	**No Burn, No Graze**	**No Graze, 10 Years**	**No Graze, 20 Years**
PP	−169.82	−169.92	−146.66	−109.33	−188.63	−193.59
ER	202.63	172.91	141.40	202.50	200.95	221.27
DOC	63.90	68.16	79.64	66.46	75.95	74.19
POC	53.31	35.54	22.07	17.94	28.82	60.85
CH_4	5.46	6.36	6.73	5.71	6.03	6.40
DissCO_2	1.28	1.72	1.91	2.17	1.77	1.80
Total	156.76	114.76	105.08	185.45	124.89	170.93
Mean total	144.15	106.11	87.25	169.19	113.01	164.63
Standard deviation	12.29	21.19	22.46	14.36	26.17	44.87

also the total biomass prior to the burn. The biomass loss during burns has not been measured for this site. However, in a survey of a wildfire in a peat ecosystem in the Peak District, UK, *Clay and Worrall* [2010] have shown that approximately 85% of biomass was lost during combustion, and in managed burns, this loss may be as little as 60% [*Allen*, 1964].

[41] By applying these estimates of biomass loss to existing biomass values for total aboveground blanket bog vegetation from Moor House [*Forrest*, 1971] and assuming a carbon content of the biomass of 50%, then carbon losses during fires can be calculated for different burn rotation lengths. The losses during fires are then added to the interburn flux of carbon in order to calculate the cumulative carbon losses or gains over a 100 year period. Losses through combustion were calculated by scaling the biomass calculated to be present for each rotation length by the loss during combustion, e.g., 90% loss of the biomass at the time of the burn.

3. Results

3.1. Dissolved Organic Carbon

[42] The DOC flux, based on soil water concentrations and flow at the catchment outlet, varied from 48 to 80 gC m^{-2} yr^{-1} (Table 2). This is at the top end of ranges reported for upland peat [*Worrall et al.*, 2009] and closer to DOC fluxes reported for peat drains [*Gibson et al.*, 2009] but is much higher than DOC flux values calculated from samples collected at the catchment outlet [*Worrall et al.*, 2007a, 2009].

3.2. Particulate Organic Carbon

[43] Mean suspended sediment concentrations from *Clement* [2005] were between 22.2 and 75.4 mg L^{-1}. Assuming 50% of the sediment is in the form of POC, then values for POC range from 11 to 38 mg L^{-1}.

[44] The POC flux, based on these data and flow from the catchment outlet, ranges from 14 to 62 gC m^{-2} yr^{-1}. The upper end of this range is larger than other estimates from Moor House (7 and 22.4 gC m^{-2}) [*Worrall et al.*, 2009]; however, values from that study were based on the flux of POC from the catchment outlet. That value from the catchment outlet does not account for account for in-stream processes therefore may not be indicative of POC leaving the peat soils that might be expected to be higher than that measured at the catchment outlet. Additionally, periods of high erosion following rainstorms are not captured here and, by using an interpolation method, may actually underestimate POC export.

3.3. Dissolved CO_2

[45] The concentration of dissolved CO_2, calculated from pH and Al concentration ranged from 0.80 to 2.70 mg C L^{-1}. *Hope et al.* [2004] report values for dissolved CO_2 in a first-order stream in an upland peat of 2.8–9.8 mg L^{-1}. The flux of dissolved CO_2 leaving the catchment, based on soil water compositions and stream flow, varied from 1.27 to 3.57 gC m^{-2} yr^{-1} (Table 2). This is within the range of previously published results [*Worrall et al.*, 2003]. Any errors in estimation are likely to have little impact on the overall budget as dissolved CO_2 is often the smallest component of peatland carbon budgets.

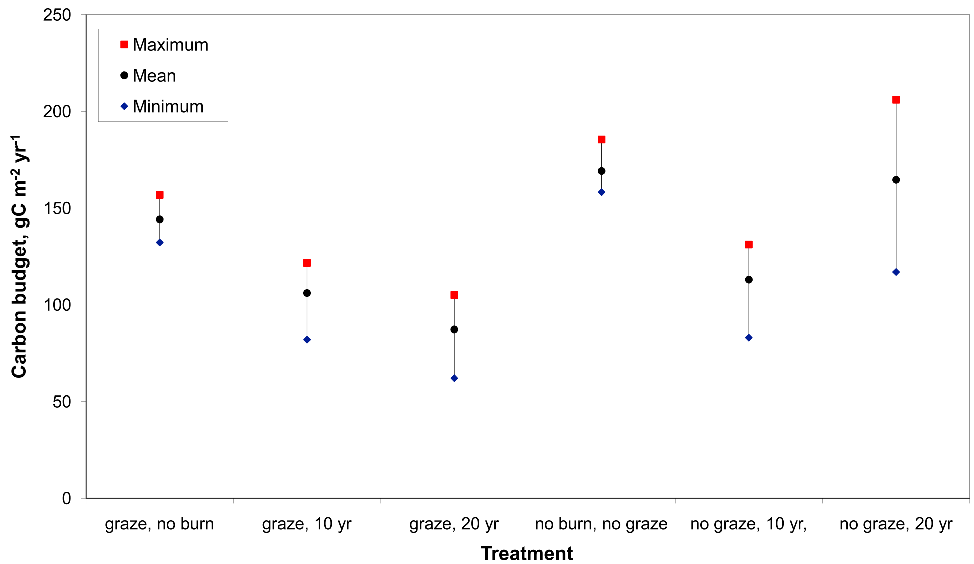

Figure 3. Range of carbon budgets for each treatment.

3.4. Surface Exchange of CO_2

3.4.1. Net Ecosystem Respiration

[46] Respiration was modeled for the study period (2005–2007) with the assumption that the coefficients did not change during the period. Net ecosystem respiration varied across the treatments from 136.6 to 258.7 gC m^{-2} yr^{-1} (Table 2). This is higher than reported from Moor House [*Worrall et al.*, 2009] though similar to results from other areas of the North Pennines measured using similar techniques [*Rowson*, 2007].

3.4.2. Primary Productivity

[47] Taking these values to be representative of the sites during the study period, primary productivity varied between −109.3 and −198.7 gC m^{-2} yr^{-1} (Table 2) that is the range of reported values for upland peat [*Worrall et al.*, 2009].

3.5. Rainfall DOC

[48] Rainfall DOC concentration varied from 0 to 5.4 mgC L^{-1} over the study period and are similar to values presented in other studies [*Worrall et al.*, 2003]. Inputs from rainwater DOC varied over the 3 years from −0.9 to −2.1 gC m^{-2} yr^{-1} (Table 2) that is a similar to ranges reported elsewhere [*Worrall et al.*, 2007c, 2009]. In a study of global precipitation input of DOC, *Willey et al.* [2000] estimated an input of 0.4×10^9 MgC yr^{-1}, of which 70% fell on land. This is equivalent to an input to land of −1.88 gC m^{-2} yr^{-1}.

Table 3. Equivalent Carbon Budget for Each Management Regime (Gram Equivalent CO_2 m^{-2} yr^{-1})

2005	Graze, No Burn	Graze, 10 Years	Graze, 20 Years	No Burn, No Graze	No Graze, 10 Years	No Graze, 20 Years
PP	−628.79	−629.67	−546.53	−409.26	−698.42	−718.71
ER	735.61	613.99	501.24	748.50	703.75	770.22
DOC	196.13	183.96	176.78	180.75	185.19	176.93
CH_4	133.89	156.23	164.60	139.53	148.28	156.80
DissCO_2	9.05	6.78	6.55	8.63	7.30	7.28
Total	328.22	220.91	196.57	559.70	234.98	286.36
2006	**Graze, No Burn**	**Graze, 10 Years**	**Graze, 20 Years**	**No Burn, No Graze**	**No Graze, 10 Years**	**No Graze, 20 Years**
PP	−641.84	−641.44	−550.22	−408.36	−712.92	−729.17
ER	691.55	652.37	538.37	646.95	823.66	949.34
DOC	246.47	269.34	242.75	243.06	232.30	276.21
CH_4	126.10	146.68	155.73	132.52	138.85	148.11
DissCO_2	13.11	11.62	10.68	12.29	9.98	11.07
Total	287.51	276.95	251.66	480.63	352.49	489.82
2007	**Graze, No Burn**	**Graze, 10 Years**	**Graze, 20 Years**	**No Burn, No Graze**	**No Graze, 10 Years**	**No Graze, 20 Years**
PP	−623.25	−623.60	−538.25	−401.25	−692.27	−710.49
ER	743.67	634.58	518.93	743.17	737.50	812.08
DOC	234.52	250.14	292.27	243.92	278.75	272.29
CH_4	131.04	152.66	161.45	137.11	144.70	153.68
DissCO_2	4.69	6.31	7.00	7.97	6.48	6.61
Total	349.96	270.00	266.04	584.58	307.91	370.80

Table 4. ANOVA of the Total Carbon Budget and Individual Components[a]

Component	Year	Burn	Graze	Year * Burn	Year * Graze	Burn * Graze
Total budget	0.018 (0.19)	0.011 (0.26)	0.006 (0.23)			0.028 (0.14)
Primary productivity	0.008 (0.00)	0.000 (0.36)	0.033 (0.00)			0.000 (0.64)
Net ecosystem respiration			0.001 (0.42)			0.004 (0.38)
DOC	0.006 (0.73)					
POC	0.031 (0.06)	0.045 (0.04)				0.000 (0.84)
CH_4	0.000 (0.11)	0.000 (0.78)	0.000 (0.02)	0.003 (0.00)		0.000 (0.09)
DissCO_2	0.003 (0.78)					

[a]Only P values < 0.05 are shown and figures in brackets indicate the proportion of variance explained by that factor or interaction.

[49] As the soil concentrations of DOC are used in the carbon budget of Hard Hill, rainfall DOC is not needed but is included here for completeness.

3.6. Methane

[50] The calculated values of CH_4 flux from the plots ranged from 5.25 to 6.86 gC m^{-2} yr^{-1} (Table 2). This range is similar to that reported by *Worrall et al.* [2009]. In one of the few studies on methane from UK peats, *Macdonald et al.* [1998] report CH_4 fluxes between 0.16 and 13.5 gC m^{-2} yr^{-1}.

[51] It has to be noted that this component of the carbon budget has not been directly measured; however, it does reflect the significant differences observed in the depth to the water table observed between treatments over long periods of observation [*Clay et al.*, 2009b].

3.7. Carbon Budget

[52] Table 2 details the different carbon pathways estimated in this study and the estimates for the carbon flux for each year of the study period.

[53] Examining the data shows that all the treatments are net sources of carbon with source size varying from 62 to 206 gC m^{-2} yr^{-1}, during the study period, though some sites are smaller sources than others (Table 2). Over the study period, unburnt sites were, on average, a source of 156.7 gC m^{-2} yr^{-1} compared to a source of 109.6 and 125.9 gC m^{-2} yr^{-1} on the 10 and 20 year plots, respectively. Figure 3 shows the data in an alternative format with the ranges for carbon flux for each treatment across the study period. When expressing the budgets as CO_2 equivalent, the sites are net sources of up to 585 gCO_2 m^{-2} yr^{-1} (Table 3).

[54] Results from the ANOVA show that burning and grazing regimes, along with year, were significant factors in the total carbon budgets (Table 4). Interannual variation accounted for 19% of the variation in the data with 2006 and 2007 having significantly higher sources than that in 2005. Grazing explained 23% of the variation with grazed sites having significantly lower budgets than ungrazed sites; over the study period grazed sites were on average a source of 112.5 gC m^{-2} yr^{-1} (average of all grazed sites) compared to 149 gC m^{-2} yr^{-1} for ungrazed sites (average of all ungrazed sites). Finally, burning accounted for the largest proportion of the variance, 26%. Here the presence of burning rather than a specific regime led to significantly smaller sources.

[55] Individual components of the budget were also analyzed using ANOVA. Like the total budget, interannual variability was a significant factor in all but one component, and in the case of DOC and dissolved CO_2, it was the only significant factor (Table 4). Burning was a significant factor for primary productivity and accounted for 36% of the variation in the data.

[56] Grazing was a significant factor in primary productivity though it accounted for less than 1% of the variation in

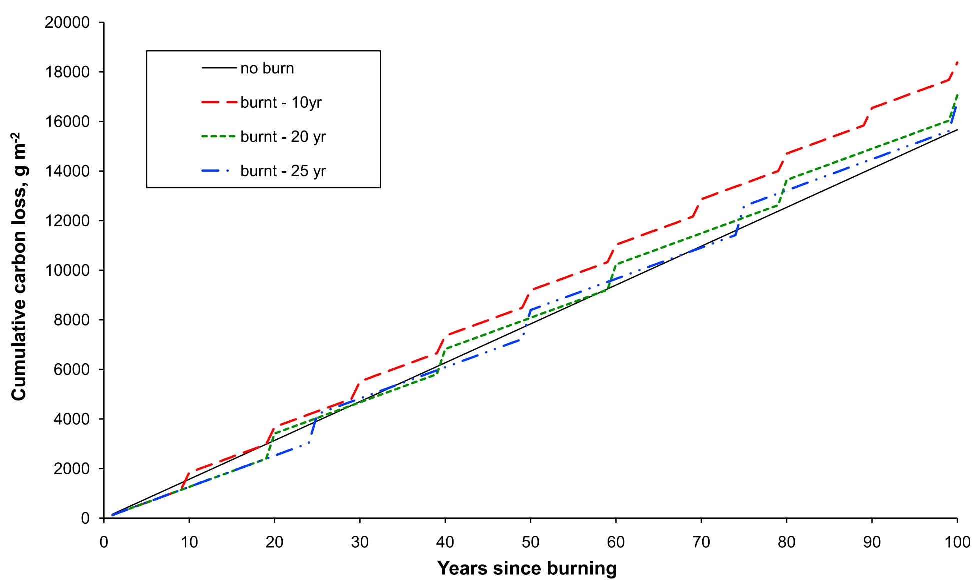

Figure 4. Cumulative carbon losses of no burn and different rotation lengths. Biomass loss = 85%.

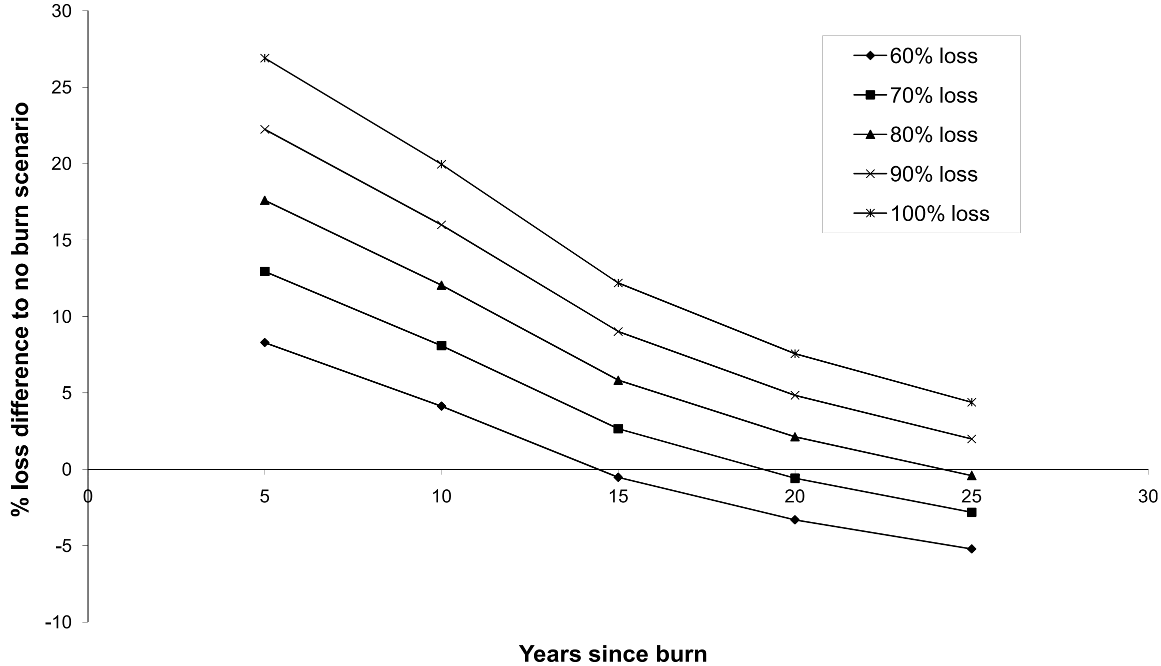

Figure 5. Difference (%) in carbon losses between burning rotations and no-burn scenario for different burning efficiencies.

the data. Grazing was also a significant factor in net ecosystem respiration explaining around 42% of the variation in the data. Here the presence of grazing led to lower net ecosystem respiration (Table 4).

[57] The interaction term between burning and grazing was significant in the surface exchange of carbon, i.e., primary productivity and net ecosystem respiration, explaining 64% and 38%, respectively, of the variation in the data (Table 4).

[58] Methane shows a high number of significant factors with all but the year and grazing interaction being significant. As methane in this study is modeled from water table measurements, it should be expected that any significant factors in methane flux will strongly follow significant trends found in the water table record; and indeed, this is the case [*Clay et al.*, 2009b].

3.8. Stocks and Fluxes

[59] In order to model the long-term fluxes of carbon, including those through combustion, average annual carbon fluxes calculated from the Hard Hill plots were combined with losses through combustion. For the no-burn scenario, the sites are taken as an annual source of 156.7 gC m^{-2} yr^{-1} (average from all the no-burn plots) (Table 2) with no losses through combustion. For the burnt scenario, the sites are assumed to be an annual source of 125.9 gC m^{-2} yr^{-1} (average of all the 20 year plots), i.e., the upper value measured (Table 2).

[60] The cumulative carbon losses from different rotation lengths are shown in Figure 4 with a biomass loss of 85%. After 100 years, the cumulative losses on the 10 and 20 year rotations were greater than if no fire had occurred. However, by extending the rotation length to 25 years, cumulative losses break even with the no-burn scenario. This scenario is based on a high combustion rate of biomass, 85%, so further runs were conducted to see whether cumulative losses on shorter rotations could ever be less than the no-burn scenario. Figure 5 shows percentage difference to the no-burn scenario for different rotation lengths (5–25 years) and different combustion factors (60% combustion through to total combustion of the biomass). Burning on 5 and 10 year rotations do not show any carbon benefit; there is a greater cumulative carbon loss than if no burning had taken place. All rotations longer than 15 years show lower cumulative carbon losses when combustion is set at 60%, i.e., a negative difference, but when combustion of the biomass is increased, this negative difference becomes smaller and in many cases becomes positive, i.e., greater cumulative carbon loss than no-burn scenario. The thresholds for combustion below which cumulative carbon losses are lower than the no-burn scenario are 61% for 15 years, 72% for 20 years, and 82% for 25 years.

[61] However, it must be noted that this model is based on a "cool" burn scenario where only aboveground carbon is affected. If the fire burns into the peat, then it is expected that any carbon benefit would be negated.

4. Discussion

[62] This study has shown that for some small-scale plots under different management, the carbon budget is positive, i.e., net source of carbon. Previous work at the Moor House NNR has shown that when the entire catchment is considered, the area is a net sink of carbon [*Worrall et al.*, 2009]. Why then do the plots show the peat to be a source of carbon? First, there is a scaling issue between plot scale experiments and catchment scale approaches. This study extrapolates the results from peat soils and applies the results as if the entire management was taken across the entire catchment. By doing this, in-stream losses and transforma-

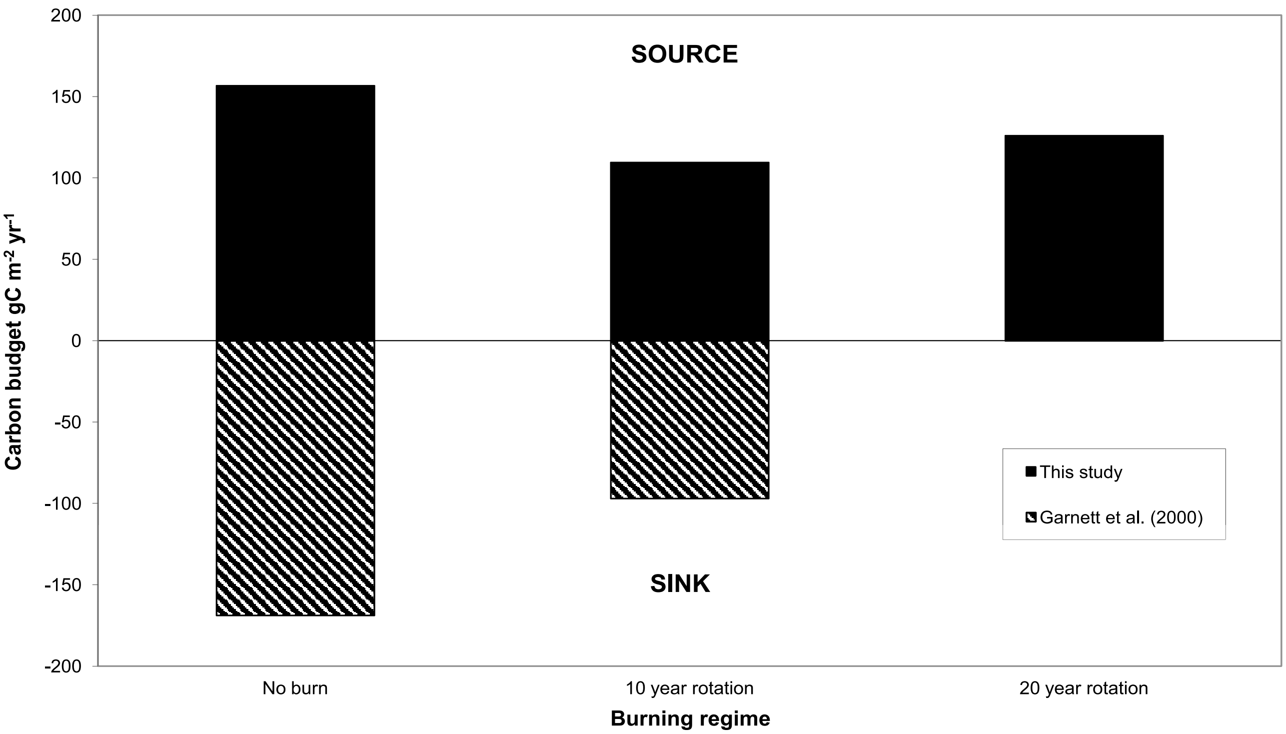

Figure 6. The overall carbon budget for each burning regime in this study and from *Garnett et al.* [2000].

tions may not be accounted for. For example, *Worrall et al.* [2006] show that 32% of the DOC flux transported may reequilibrate with the atmosphere.

[63] Second, it is likely that the management of the plots plays a significant role in contributing to the nature of the carbon budget. This is the first study to calculate total carbon budgets for upland management combinations and previous studies only considered the Trout Beck catchment as a whole. Results from ANOVA show that burning and grazing are significant factors in the carbon budgets. Burnt sites, i.e., 10 and 20 year plots show significantly lower overall budgets than unburnt plots. The main reason for this difference is the combined effect of significantly higher primary productivity on some burnt sites and lower net ecosystem respiration. This in turn reduces the losses seen in the hydrological carbon budget. These higher rates of primary productivity are likely to be due to higher photosynthetic rates found in younger vegetation. *Johnson and Knapp* [1993] found higher photosynthetic rates along with increased aboveground biomass production, inflorescence density, and plant height in annually burnt sites. As the vegetation becomes older and more degenerate, its ability to sequester CO_2 lessens. *Bond-Lamberty et al.* [2004] showed for boreal forest that middle aged stands were net sinks, whereas the oldest stands were carbon neutral.

[64] On the whole, the largest sources in this study were the unburnt sites, in particular, the unburnt, ungrazed, sites. On these sites, there are very high NER values and the lowest PP values. The low primary productivity could be driven by the lower rates of CO_2 uptake in older vegetation as described previously. The higher ecosystem respiration could be due in part to the position of the water table on these plots. The deepest water tables are found on the unburnt plots [*Clay et al.*, 2009b] allowing a greater depth of aerobic decomposition leading to greater respiration values [*Moore et al.*, 1998].

[65] Grazing also plays a significant role in the carbon budgets of these sites. The effect of grazing is similar to burning in that new vegetation growth is encouraged leading to negative NEE and lower sources. Increased CO_2 exchange efficiency has been observed on grazed prairie grasslands and has been linked to the presence of young, highly photosynthetic leaves [*Owensby et al.*, 2006].

[66] These results, which show that this site is a source of carbon, would appear to be at odds with the study of *Garnett et al.* [2000] who indicate that the sites have accumulated carbon since the 1960s. Additionally, *Garnett et al.* [2000] indicate that burning limits the amount of carbon accumulation, whereas this paper shows that burning reduces the losses. Figure 6 shows the nature of the difference; note the relative size of burn regime to no burn is approximately 75% in this study and 60% in the study by *Garnett et al.* [2000].

[67] There are several possible reasons for this difference. First, this study makes use of carbon budget approach that, unlike accumulation studies, is able to determine periods of carbon loss from a system. Therefore, it is plausible that over decades of accumulation, small periods of losses are not captured but the overall trend is of accumulation. Second, interannual variation in component of total carbon fluxes can lead to a range of values for the same site [*Worrall et al.*, 2009], and this study took place between 2005 and 2007 that is nearly a decade after the sampling of *Garnett et al.* [2000]. Third, the hydrological component of the carbon budget can turn sites that are gaseous sinks of carbon into carbon sources [*Worrall et al.*, 2007c]. In recent years,

increasing trends in DOC records across the United Kingdom have been noted [*Worrall and Burt*, 2007]. It is possible that in the years between studies DOC export has increased. Although this cannot explain all of the discrepancy, it may go some way to accounting for the difference. Finally, it should be noted that *Garnett et al.* [2000] measured depth of peat accumulation between treatments and had assume a constant carbon concentration across the range of management types studies here. Carbon concentration and bulk density, factors not presented in the study by *Garnett et al.* [2000], are important factors determining carbon stocks in peat and may vary significantly between treatments affecting any conclusions around long-term carbon accumulation.

[68] The carbon budget of these sites cannot be viewed without some consideration of fluxes associated with the burns themselves. These are point source losses at discrete periods in time so may not be captured during the life of an experiment or study. This work has shown that carbon sources are smaller on burnt sites than on unburnt sites by approximately 25%. However, the emission of carbon during combustion is likely to outweigh this in the short term, so where do trade-offs occur? Modeling suggests that there are windows of opportunity under which less carbon is lost than if no burning had taken place. Taking the example shown in Figure 4, which models a fire in which 85% of the biomass is lost through combustion, 10 year rotations have a greater cumulative carbon loss than if no burning occurred. However, cumulative losses are lower on 25 year rotations. Here the loss of carbon through combustion does not outweigh the gains though lower annual carbon fluxes.

[69] Further analysis shows that for each rotation length there is a critical threshold for the losses of carbon during combustion in order for a burning scenario to remain a smaller source of carbon than a no-burn scenario. For example, if burning on a 15 year rotation, losses of carbon from fires must not exceed 61%. Longer rotations are likely to give rise to hotter burns due to a greater amount of biomass and woody material leading to greater carbon losses. Given the total loss of biomass carbon, what is the minimum rotation that would still lead to a smaller cumulative loss than if no burning had occurred? By examining Figure 5, this trade-off occurs at rotations longer than 32 years.

[70] This study shows that there are conditions under which less carbon is lost under burn management than if no burning had taken place. In this assessment of changes to aboveground carbon stocks during regular burns, a conservative view has been taken. It has been assumed that the biomass that is left behind after a fire (15% of the biomass in Figure 4) is not an addition to the peat store but could of course add to the NER fluxes measured between burns. In reality, this biomass left behind is often dead and does represent an additional litter input to the peat soil. Furthermore, a proportion of the biomass left behind by the fire will actually have been converted to char. Post burn products such as char are more resistant forms of carbon [*Lehmann et al.*, 2008], unlike peat, and will therefore add to long-term carbon storage by increasing the size of refractory carbon pool.

[71] There is currently much debate about the use of managed burning in the UK uplands [*Reed et al.*, 2009] and many of these discussions relate to the issue of carbon. By monitoring the effects of managed burning on an upland peat, this study has been able to start to understand some of the effects of burning on carbon. However, given the debate surrounding the impacts of managed burning in the UK uplands, the results from this research need to be clearly understood in their context. First, the field site is in the North Pennines on blanket bog that is considerably different to many areas where managed burning is routinely carried out, e.g., drier heaths of the North York Moors and some of the degraded peats of the Peak District. Second, the burn was carried out in ideal conditions with little damage to underlying litter and peat. Third, the burn was carried out as part of an experimental program at a plot scale. Fourth, monitoring was carried out over 3 year monitoring period, but this may not reflect the longer-scale peat-forming time scale. Finally, some of the carbon flux pathways, e.g., methane, were estimated from the best available data but were not directly measured on this field site.

5. Conclusion

[72] By using the best combination of experimental and modeling approaches, the treatments at Hard Hill are shown to be net sources of carbon of between 62 and 206 gC m^{-2} yr^{-1}. However, the presence of burning and grazing appears to limit the magnitude of this source by reducing net ecosystem respiration by up to 14% and increasing primary productivity by up to 29%.

[73] The largest part of this budget are two components: the gaseous exchange of carbon, primary productivity (−198 to −109 gC m^{-2} yr^{-1}) and net ecosystem respiration (137–259 gC m^{-2} yr^{-1}). Fluvial export of carbon via DOC is the next largest component of the carbon budget (48–80 gC m^{-2} yr^{-1}) and turns the sites, which are net sinks of gaseous carbon, into an overall net source of carbon.

[74] An assessment of the trade-offs between annual carbon fluxes and loss of carbon through combustion has shown that there may be opportunities for reducing carbon losses from managed sites. Longer rotations have lower cumulative carbon losses than unburnt of short rotations sites though this depends on the level of combustion during fires.

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