CENTRE FOR ECOLOGY AND HYDROLOGY
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TITLE: **A REVIEW OF CARBON FLUX RESEARCH IN UK PEATLANDS IN RELATION TO FIRE AND THE CAIRNGORMS NATIONAL PARK**

Final Report

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Abstract

Peatland areas within the UK as a whole and on a more local level within the Cairngorms National Park (CNP) hold major stocks of terrestrial carbon. These areas are subjected to land management practices, such as burning, drainage, and grazing. Climate change is leading to increased fire frequency in many ecosystems including peatlands as a result of summer droughts. In addition, land use change, e.g. reduction in grazing and management fires or increases in recreational use, have resulted in an increased fire hazard. Particularly problematic are large intense fires which impact upon biodiversity, livelihoods and human life. These problems are directly relevant to the CNP which holds large areas of moorland that includes peatland and heathland ecosystems. Concern over the environmental impact of changes to fire regimes in upland areas of the UK and CNP (i.e. frequency, intensity, size, season) is, compounded by a lack of data on the impacts of fire regime on biodiversity, soil erosion, water quality, carbon sequestration and other ecosystem services. There is therefore a need to take stock of carbon flux research on peatland ecosystems in the UK in general and relate this to the CNP on a more local level. This is required to assess the level of spatial and temporal resolution of the current data, determine whether peatlands are currently sinks or sources to the atmospheres and to understand the implications of fire on peatland ecosystems in order to protect the large carbon stores from being lost to the atmosphere. This is also required within the CNP so that the degree of any knowledge gap can be assessed in order that policy and management decisions can be addressed based on evidence.

Approach

Here we review peatland carbon flux research in the UK and relate its applicability to the CNP in order to:

1. summarise previous work,
2. provide evidence of how fire influences carbon fluxes in UK peatlands,
3. indicate areas of study where research is needed for model parameterisation and
4. indicate opportunities for applied peatland research in relation to fire.

Conclusions

At present, there is insufficient data to draw firm conclusions on the effects of fire on the peatland carbon cycle and to sufficiently parameterise models at a UK level or a morel local one i.e. Cairngorms National Park.

Much of the evidence is from single site (Moor House in the Pennines) and the applicability of the results to the wider UK and CNP peatland landscape is open to question. We suggest that an inter-disciplinary approach is required to include all stakeholders such as, landowners, politicians and scientists.

1. Introduction

In contrast to their global land area, peatland ecosystems store large amounts of terrestrial carbon (Clymo *et al*. 1998); the interactions between peatlands and the atmosphere are important to global climate change research (Gorham 1991). Important peatland carbon processes include the sequestration of carbon dioxide (CO_2) through photosynthesis, the loss of CO_2 to the atmosphere through respiration and atmospheric methane (CH_4) emissions from anaerobic decomposition in the waterlogged conditions (Figure 1). To a lesser extent, there are also losses of dissolved organic carbon (DOC) to stream waters (e.g. Freeman et al. 2001).

Research on gaseous carbon fluxes is important for the parameterisation of climate change models, understanding ecosystem response to climate change and informing government policy (Grace 2004) and in addition to understand the effects of land management practices on carbon cycling. Inventories of greenhouse gases are produced annually by governments including the UK (Baggott *et al*. 2007) but at present despite the large store of carbon there is no account of change to peatlands carbon stocks due to land use change in the UK, although, peat extraction and use for fuel are included. Nevertheless management practices are likely to play a central role in the maintenance of carbon and nutrient budgets, biodiversity, erosion and drinking water quality at a national and a local level. For example, prescribed (and wild) fire can have the potential to become uncontrollable and under certain conditions may cause smouldering peat fires causing emission of carbon to the atmosphere. Conversely, it may be possible through the use of prescribed burning to manipulate peatland vegetation to maximise the benefits of fire and manage the threat of wildfires (see Davies *et al*., 2008). Current climate change model predictions, warmer and drier summer periods in particular, (Hulme et al. 2002) indicate potential severe wildfire conditions may be reached more often in the future (2006 may be a prime example). However, our present ability to define and predict such conditions is limited. Concern over the environmental impact of changes to fire regimes (i.e. frequency, intensity, size, season) is, compounded by a lack of data on the impacts of fire regime on biodiversity, soil erosion, water quality, carbon sequestration and other ecosystem services in peatlands despite over a century of research into this ecosystem in the UK (Field 1981). There is currently debate about how much burning takes place and further concerns that burning has increased in recent years (Yallop *et al.* 2006) the debate seems likely to continue until estimates for the total geographical extent of peatland burning and grazing for the UK are apparent (Gray 2006). This debate is now also widening to speculation over the consequences of these practices on the peatland carbon cycle (see Pearce 2006, Davies et al. 2008) again at both national and local levels.

1.1 Cairngorms National Park

The Cairngorms National Park was the second National Park to be established in Scotland in 2003 and covers an area of 3800 km^2. The park has its own planning authority and is a working part of Scotlands' heritage, land use includes farming, crafting, game management, recreation and in addition has several designations associated with nature conservation e.g. Special Sites of Scientific Interest (SSSI) Special Areas of Conservation (SAC) and Special Protection Areas (SPA). The main peatland type in the area is blanket bog (*Calluna – Eriophorum* dominated and generally 0.5-3.0m deep) which is the second most extensive habitat type in the park often grading into wet upland heath or located within heathland mosaics (Cosgrove 2002). Approximately 41% of the Cairngorms Partnership area is upland heathland (Cosgrove 2002), this habitat also stores significant amounts of carbon. Consideration of fire and peatland habitat therefore should be integrated within the management of heathlands in the Cairngorms area.

1.2 Review Aims

In order to protect the large carbon stores from being lost to the atmosphere there is a need to take stock of carbon flux research on peatland ecosystems in areas where such stocks are significant e.g. CNP. This is required to assess the level of spatial and temporal resolution of the current data, determine whether they are currently sinks or sources to the atmosphere and to understand the implications of management actions such as burning on peatland ecosystems. Here we review peatland carbon flux research in the UK and relate its applicability to the CNP in order to:

1. summarise previous work,
2. provide evidence of how fire influences carbon fluxes in UK peatlands,
3. indicate areas of study where research is needed for model parameterisation and
4. indicate opportunities for applied peatland research in relation to fire.

We have deliberately not restricted this review to the CNP and include research from the UK as a whole in order to maximise the likelihood of obtaining relevant data.

2. Background and General Effects of Fire on Peatlands

To aid interpretation, definitions of some common peatland and fire related terms are given in Table 1.

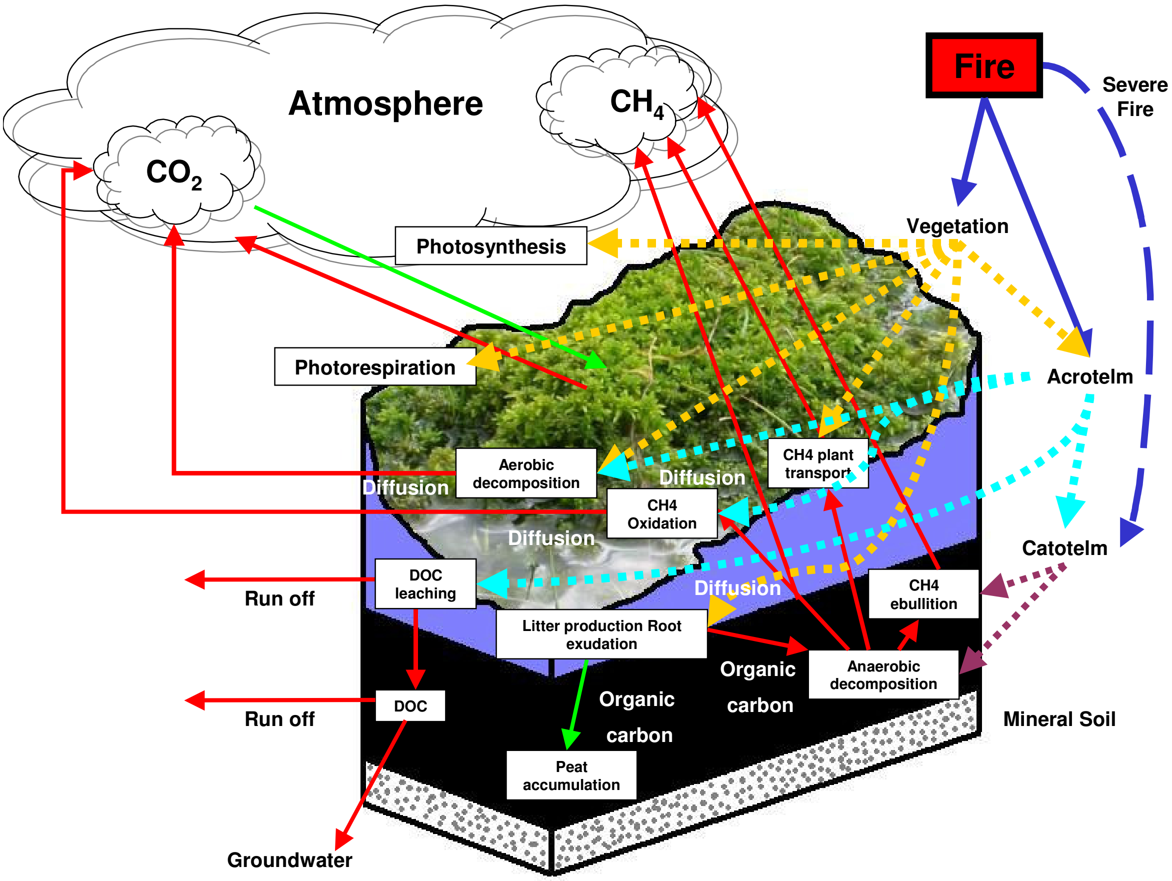

Figure 1: Simplified representation of the peatland carbon cycle showing the principal peatland components that fire can affect. Green lines indicate carbon uptake, red lines carbon losses; solid blue line indicate direct fire effects, dotted lines indicate indirect fire effects and the dashed blue line (on the far right) indicates the potential of severe fires to ignite catotelmic peat (adapted from Gray 2006).

Table 1: Definitions of common peatland and fire terminology (adapted from Davies et al. 2008, Gray et al. 2008).

Type	***Term***	***Definition***
Peatland	**acrotlem**	The upper aerobic peat and vegetation above the water table
	bog	Peat-forming wetland, low pH
	blanket bog	Peatland on gently sloping ground with depth greater than 0.5 m; can be made up of a mosaic of bogs such as valley bogs and saddle bogs.
	carpet	Usually associated with *Sphagnum* spp. where the vegetation is uniformly flat occurs on loosely consolidated peat, extending only slightly above the water table
	catotelm	The lower anaerobic peat below the water table
	eutrophic	Used for vegetation and peat enriched in nutrients, typically nitrogen or phosphorus
	hollow	Depression usually associated with *Sphagnum* spp. where water table is usually at or above the vegetation for most of the season
	hummock	A small raised mound formed by the upward growth of *Sphagnum* or *Racomitrium* spp.
	lagg	Margin of a raised bog, typically supports vegetation dominated by sedges and/or shrubs (incl. dwarf shrubs), usually higher pH than raised bog.
	lawn	Synonymous with carpet.
	mesotrophic	Vegetation and peat, where nutrients are derived from groundwater sources but not as enriched as eutrophic.
	minerotrophic	Vegetation and peat, where nutrients are derived from groundwater sources.
	mire	A generic term for peatlands
	oligotrophic	Vegetation and peat, where nutrients are derived from atmospheric sources
	ombrogenous	Vegetation and peat, where nutrients are derived from atmospheric sources (see below)
	ombrotrophic	Vegetation and peat, where nutrients are derived from atmospheric sources; usually synonymous with above but they are subtly different ('trophic' = nourishment 'genous' = origin)
	pool	Depression usually associated with *Sphagnum* spp. where water table is usually above the vegetation for most of the season
	raised bog	A bog shaped like a dome or elevated above the surrounding land and therefore not accessible to adjacent ground water.
Fire	**fire danger**	General term used to express an assessment of both fixed and variable factors of the fire environment that determine the ease of ignition, rate of spread, difficulty of control and fire impact.
	fire hazard	Measure of that part of the fire danger contributed by the fuels available for burning. Note: this is worked out from the relative amount, type, and condition, particularly the moisture contents.
	fire regime	Pattern of fire occurrence, size, and severity – and sometimes also vegetation and fire effects – in a given area or ecosystem. It integrates various fire characteristics. The classification of fire regimes includes variations in ignition, fire intensity and behaviour, typical fire size, fire return intervals, and ecological effects.
	fire risk	Probability of fire initiation due to the presence and activity of a causative agent.
	fire severity	Degree to which a site has been altered or disrupted by fire.
	fireline intensity	Rate of heat release per unit time per unit length of fire front. Numerically, the product of the heat of combustion, quantity of fuel consumed per unit area in the fire front, and the rate of spread of a fire, expressed in kW m−1. Often referred to simply as ‘intensity’ or ‘fire intensity’.
	prescribed fire	A management-ignited wildland fire or a wildfire that burns within prescription, i.e. the fire is confined to a predetermined area and produces the fire behaviour and fire characteristics required to attain planned fire treatment and/or resource management objectives. (cf. prescribed burning).
	wildfire	Any unplanned and uncontrolled wildland fire that, regardless of ignition source, may require suppression response, or other action according to agency policy.

2.1 Habitat

Peatlands are the most extensive semi-natural habitat in the UK with estimates of the geographical extent of peat greater than 1 m deep in the region of 1.5 Mha (Lindsay 1995). The most common peatland communities in the UK are M17 *Scirpus cespitosus – Eriophorum vaginatum* blanket mire, M18, *Erica tetralix – Sphagnum papillosum* raised & blanket mire M19, *Calluna vulgaris – Eriophorum vaginatum* blanket mire M20 *Eriophorum vaginatum* blanket & raised mire and M25 *Molinia caerulea – Potentilla erecta* mire (Gray et al. 2008). The most common peatland type in the Cairngorms is the M19 often within heathland mosaics and the most predominant community of this vegetation is the H12 *Calluna vulgaris - Vaccinium myrtillus* heath (Johnson and Morris 2000).

The 'traditional' perception of a peatland is of a fire free wet *Sphagnum* dominated area, however in the UK, many deep peat areas are covered by dense canopies of *Calluna vulgaris* (L.) Hull. Though less common in the UK than in Europe or North America, there are also significant areas of peatland patterning that are of significant conservation interest (e.g. hummocks, hollows and pools) particularly in the north of Scotland. Formation of these patterns appears to involve many processes (e.g. climate and hydrology). Fire has rarely been considered as a contributory factor to this patterning but limited evidence does suggest that a differential response to fire may be involved (Benscoter et al. 2005a, Benscoter et al. 2005b). In the UK as a whole and within the Cairngorm National Park the Lammermuir Hills and the Pentland Hills in Scotland, there are also significant peat deposits with a high-carbon content of a lesser depth than is usually used to classify peatland (generally between 0.5 - 1 m, see NCC 1990, Lindsay 1995, Rydin and Jeglum 2006). These may be at more risk from wildfire than the deeper, wetter deposits. In terms of assessing fire effects on carbon cycle processes in peatlands, we agree with Davies et al. (2008) that removing the variable depth distinction to define a peatland would give a more holistic approach to soil carbon; a classification of soils based on carbon content would be helpful (see Davies et al. 2008).

2.2 Anthropogenic Fire

Fire has been used for millennia, certainly evidence stretches as far back as Mesolithic period (Shaw *et al*. 1996) and some authors believe that anthropogenic fire may have been responsible for the initiation of blanket bog in some areas (Moore *et al*. 1984). In the UK prescribed fire has been used as a management tool on moorland and peatlands for more than two centuries leading to the use of the term cultural landscapes to describe them (Thomson et al. 1995); this is also true of the CNP. The goal of these managed fires is to remove and regenerate vegetation to improve food quality and vegetation structure, for example, *Calluna* for red grouse or grass and sedges for the 'early bite' (Shaw et al. 1996, Hamilton et al. 1997, Hamilton 2000, Tucker 2003). This latter strategy is used particularly on blanket bog in the north west of Scotland (Hamilton et al. 1997, Hamilton 2000) but we know of no data detailing how widespread this particular practice is in the Cairngorms.

Guidance on the use of fire in Scotland is contained in the Muirburn Code (Anon 2001), generally the burning of peatlands is not recommended because of the perceived detrimental effect it can have on the characteristic species and the risk of peat ignition. The exception to this policy is where *Calluna* constitutes more than 75% of the vegetation (Anon 2001) but these should be on long rotations (Shaw *et al*., 1996, Tucker, 2003, and references therein). However, *Sphagnum* species are not as sensitive as perhaps is assumed and do not always do badly under fire management (Hamilton 2000, Tucker 2003). The removal of a dense shrub canopy by fire has been observed to benefit the recovery of *Sphagnum* species in some bogs (A Gray personal observation), though this may also be brought about by other mechanical means. There can also be interactions between fire and drainage because the water level can influence the effects of the fire, moist peat is insulated and severe burning may lead to increased peak flows in drainage ditches (Shaw *et al.* 1996). Prescribed burn temperatures are generally low enough to avoid peat combustion, with recorded temperatures 1 cm below the surface not exceeding 100°C (Hobbs and Gimingham 1984). Canopy temperatures though greater are short lived, (250- 840°C), the higher temperatures are usually associated with older *Calluna* containing a greater amount of woody material (Hobbs and Gimingham 1984; Nilsen *et al.* 2005). However, of more long term interest than canopy temperatures is the duration of ground (moss/litter) surface temperature above 50, 100, 400 degrees. These roughly correspond to damaging, lethal and combustion temperatures however more researched is required in this area for peatlands (see Hamilton 2000, Davies 2005). In extreme cases intense fires can ignite the peat removing the vegetation, produce a hard bitumen surface that can lead to increased runoff. Fire intensity may be one control on the initiation of smouldering fire but under more 'normal' conditions this type of fire will be difficult to initiate. Ignition is controlled by the power of the ignition source (i.e. temperature and time), the balance between oxygen in-flow (not usually a limiting factor) and heat out-flow (i.e. aeration v insulation). Micro-sites for initiation are probably important too e.g. smouldering twigs, animal dung, or cracks in dry peat may all be good mechanisms for the initiation of smouldering fires. These types of fire can lead to increased exposure, increased evaporation, increased temperature, decreases in soil organic matter and nutrients, and seed bank destruction making plant establishment difficult leading to erosion. This type of fire is more likely when ignition is accidental or malicious (Maltby et al. 1990, Legg et al. 1992, Tucker 2003). Smouldering combustion is not well studied but if the peat is ignited these fires are extremely difficult to extinguish. Ignition of these smouldering fires appears to be at least partly controlled by moisture content (Rein et al. 2008, Rein et al. 2009).

2.3 Fire effects on species composition

In the 13 years since Shaw et al., (1996) reviewed the effects of fire on blanket bog, peatland fire research does not appear to have moved on. The majority of the work to date investigating fire still relates to grouse moors or lowland heaths, and therefore to a drier type of habitat than peatland. However, there are generalisations that are applicable; the effects of fire are dependant on vegetation, intensity and frequency of the fire, timing of the burn and the wetness of the habitat. Summer fires are likely to be most damaging for wildlife interest. There will be indirect effects through changes in the physical habitat characteristics, plant species composition and vegetation structure and consequently microclimate. Tucker (2003) summarised the impact of fire on selected upland species and the impacts on those species more prevalent in peatlands are reproduced in Tables 2 and 3.

However despite much information the effects of fire on vegetation still remain speculative as evidence for successional pathways (Figure 2) is still lacking (Thompson et al. 1995, Shaw et al. 1996, Gray 2006). Where conditions are conducive, regular burning can produce a monoculture of *Calluna vulgaris* (Hobbs and Gimingham 1984). On wetter peatland areas burning can alter the balance between shrubs and graminoid species such as *Eriophorum* spp. or *Molinia caerulea,* microhabitats or lead to bare peat but the outcome is dependant on the severity and/or fire regime (Taylor and Marks 1971, Rawes and Hobbs 1979, Maltby et al. 1990, Benscoter et al. 2005a). Fire can also cause changes to hydrology, (Charman 2002) incident light, soil temperatures (Hobbs and Gimingham 1984) and nutrients

(Allen 1964, Tucker 2003). Early research suggested that there may be long-term depletion of N, P and K (Elliott 1953), however, subsequent research concluded that losses were replaced from precipitation (Allen 1964, Robertson and Davies 1965, Tucker 2003). Shaw *et al.,* (1996) state that when burning (and grazing) are carried out indiscriminately these management practices are likely to be damaging to the wildlife interests of blanket bog and may even lead to loss of habitat. However, if conducted sensitively, both burning and grazing can have beneficial effects to some species of these habitats (though not all). As suggested by Davies et al. (2008) a diverse and adaptive approach is required to fire management in peatlands. We have in the timeframe of this report been unable to obtain information on current burning practices within the CNP. Nevertheless, much of the above information, although general, is still directly relevant to the CNP and we suggest that an adaptive approach to fire management is adopted if not already in place.

2.4 Greenhouse gas production

As with most ecosystems the peatland carbon cycle is dominated by photosynthesis and respiration; in addition because of the anaerobic conditions of the waterlogged peat, methane production is also significant (Figure 1). The immediate consequence of most fires, including prescribed fire, affects primarily the vegetation and acrotelm. However, severe fires (either accidental or malicious) may have the potential to ignite deeper catotelmic peat leading to smouldering combustion and large carbon losses. The effects to carbon cycling will depend on the fire regime (frequency, intensity and season). An additional loss of carbon and other chemicals through fire will be to the atmosphere in smoke and ash. The consequences of fire on carbon balance will also be scale dependent. While the immediate consequences of fire are the loss of carbon to the atmosphere and death of important peat-forming species such as *Sphagnum*, in the intermediate term, the removal of shrub cover and litter may permit rapid recovery and expansion of *Sphagnum* and peat formation and hence carbon sequestration. In the long term, fire may promote increased *Calluna* dominance and changes to the hydrology of the bog that result in desiccation and oxidation of peat (Hamilton 2000). Evidence for the effects of fire on the microbial community are scarce but the perturbation of fire may stimulate microbial activity within peat and increase the rate of decomposition (Maltby *et al*. 1990). Effects on the microbial community may be persistent (Zenova et al. 2008) and involve change to methane oxidation processes (Jaatinen et al. 2004) and substrate use by the soil microbial community (Bergner et al. 2004), though more general patterns are more difficult to ascertain due to the lack of evidence. Rates of peat accumulation have also been noted to be lower in areas that are burnt (Kuhry 1994, Garnett 1998, Garnett et al. 2000) suggesting that in terms of carbon sequestration burning may not be beneficial.

As the above suggests there is little evidence at either global, UK or CNP levels to derive a general evidence based approach to fire management within the CNP in relation to the peatland carbon cycle and a more holistic approach is required.

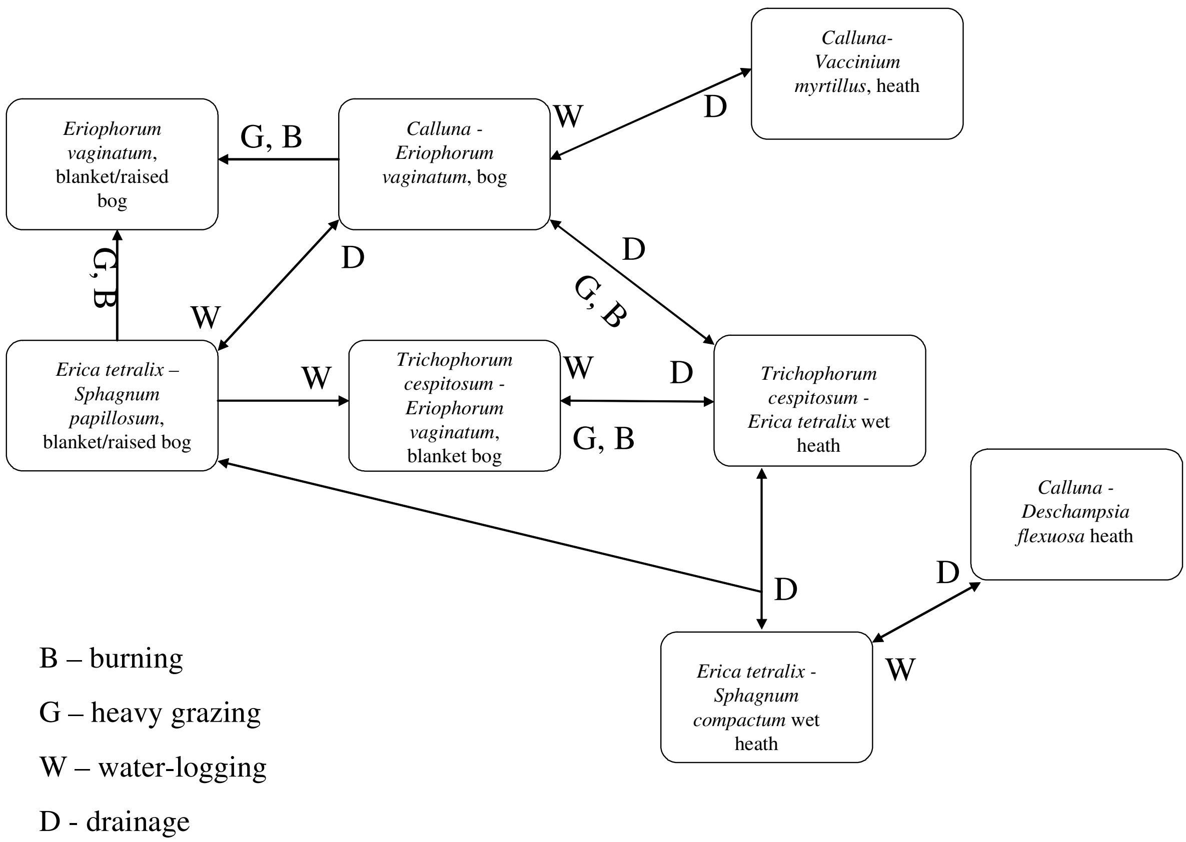

Figure 2: Simplified successional changes between bog and heath communities as affected by burning grazing and water table alteration (re-drawn from Thompson *et al.,* 1995).

Table 2: Extent of the practice of burning and advantages and disadvantages of this type of management on the blanket bog habitat (Tucker 2003)

Habitat	**Extent of burning**	**Advantages**	**Disadvantages**
Blanket Bog	Majority under some sort of burning regime	*Eriophorum* favoured may benefit black grouse and large heath butterfly if abundance low. Some carefully selected controlled burning may be necessary to reduce fuel loads and risk of wild fire	Potential loss of fire sensitive species; can become dominated by *Eriophorum* on short rotations, or *Calluna* on long rotations. Nutrient loss may be significant. Risk of reduced peat formation and significant risk of erosion and combustion of peat. Peat combustion and drying can cause significant losses of carbon. Increased *Eriophorum* may cause increased methane flux.

Table 3: Summary of impacts of burning management on selected peatland species based on Tucker (2003) and Hobbs *et al.,* (1984).

Species	**Perennating organ & fire survival mechanism**	**Impacts**
Calluna vulgaris	Stem bases, protected by litter and persistent seed bank	Regenerates relatively rapidly after typical management fires, if burnt before the late mature phase. Re-establishes by seed from abundant long-lived seedbank if old stands are burnt or if hot fires damage basal stems. But seedling establishment is slow and may allow invasion by rhizomatous species. May not re-establish if burning is too frequent. Generally increases in abundance with long burning rotations (e.g. > 15 years) on bogs.
Empetrum nigrum	Buried branches	May be susceptible to fires but if prostrate stems are not destroyed then may gain temporary dominance in heathlands until overtopped by *Calluna*.
Erica tetralix	Stem bases, protected by litter and persistent seed bank	Similar to *Calluna*, but favoured by shorter burning rotations of 6-10 years. May also be able to regenerate better in wetter habitats because its semi-prostrate lower branches are protected by *Sphagnum* and litter layers.
Eriophorum angustifolium	Rhizomes	Often benefits from periodic fires, as can rapidly recolonise burnt areas from rhizomes, but is later out competed. May not survive post-fire conditions if significant changes in moisture and pH.
Eriophorum vaginatum	Tiller apices within leaf sheaves	Rapidly regenerates after fire and probably resistant to hot fires due to tussocky growth form. Temporarily dominates after fires in blanket bogs and can remain dominant if burning rotations are less than 10 years.
Molinia caerulea	Tiller apices within leaf sheaves	Can regenerate rapidly after fire and often dominates (sometimes with *E. vaginatum*) under frequent burning regimes.
Sphagnum mosses	-	Often thought to be fire sensitive, but little evidence for this. Wet conditions may protect species from fires and some can regenerate from deep buried fragments. Most impacts probably from peat damage and trampling, or due to exposure to drying or algal growth after removal of vegetation cover.

3. Review of peatland carbon flux research in the UK

There are a number of published studies reporting fluxes of CO_2 and CH_4 from the UK (see; Clymo and Reddaway 1971, 1972, Hogg et al. 1992, Choularton et al. 1995, Clymo and Pearce 1995, Fowler et al. 1995a, Fowler et al. 1995b, Nedwell and Watson 1995, Beverland et al. 1996, Chapman and Thurlow 1996, Fowler et al. 1996, Gallagher et al. 1996, Beswick et al. 1998, Chapman and Thurlow 1998, Daulaut and Clymo 1998, Hargreaves and Fowler 1998, Lloyd et al. 1998, MacDonald et al. 1998, Moncrieff et al. 1998, Hughes et al. 1999, Freeman et al. 2002, Gauci et al. 2002, Hargreaves et al. 2003, Beckmann et al. 2004, Ward et al. 2007, McNamara et al. 2008). However, none appear to have been directly conducted in the CNP or are done in relation to fire. The most detailed evidence for the effects of fire on the peatland carbon cycle in the UK comes from Moor House in the Pennines and these papers are reviewed in a separate section below (Garnett et al. 2000, Ward et al. 2007, Worrall et al. 2007); a further recent Moor House paper on methane production from peatland gullies is not included in the Moor House section as the study does not examine fire effects (McNamara et al. 2008). Following convention, negative fluxes denote a carbon sink.

3.1 Carbon dioxide

Excluding the Moor House work a total of eight papers reporting fluxes of CO_2 were examined (Table 4). There is a bias towards Scotland none of which are within CNP boundaries with only that of Clymo and Reddaway (1971 & 1972) from England. The methodologies employed are split between static chambers, peat cores and eddy covariance/conditional sampling. These methods encompass a variety of scales from < $1 m^2$ (chambers cores) to > $1 km^2$ (eddy covariance). Six sites were included meaning some sites have been re-sampled, not always by the same authors. There are double the number of blanket bog sites (4) compared to raised bog (2) sampled. None of the studies stated whether fire or any other practices were carried out within the sites and winter appears to have been included in only half of the cases. Respiration fluxes range from 0.06 to 1.389 μmol CO_2 m^{-2} s^{-1} and net ecosystem exchange (NEE) from -5.556 to 0.704 μmol CO_2 m^{-2} s^{-1}. Ultimately, the data show a lack of spatial and temporal replication and an absence of evidence in relation to fire or other management practices such as grazing.

Table 4: Number and characteristics of gaseous CO_2 and CH_4 flux studies excluding Moor House conducted in the UK from a review of papers (Choularton et al. 1995, Clymo and Pearce 1995, Fowler et al. 1995a, Fowler et al. 1995b, Nedwell and Watson 1995, Beverland et al. 1996, Chapman and Thurlow 1996, Fowler et al. 1996, Gallagher et al. 1996, Beswick et al. 1998, Chapman and Thurlow 1998, Daulaut and Clymo 1998, Hargreaves and Fowler 1998, Lloyd et al. 1998, MacDonald et al. 1998, Moncrieff et al. 1998, Hughes et al. 1999, Freeman et al. 2002, Gauci et al. 2002, Hargreaves et al. 2003, Beckmann et al. 2004) using a keyword searches of bibliographic databases. * Note: does not necessarily sum to total number of studies because some papers used multiple methods. N/S - not stated.

Gas	**No. Studies**	**Country**	**Sites sampled**	***Management/Fire***	**Winter included**
CO_2	8	7 Scotland	Ellergower Moss	8 N/S	3 included
	3 respiration only	1 England	Loch More		4 not included
			Glensaugh		2 not stated
			Auchencorth Moss		
CH_4	19	17 Scotland	Bad a Cheo	19 N/S	5 included
		1 England	Caithness		8 not included
		1 Wales	Cerrig-yr-Wyn		6 not stated
			Ellergower Moss		
			Loch Calium		
			Loch More		
			Moidach More		
			North Scotland		
			Potree to Wick		
			Strathy Bog		

3.2 Methane

A total of nineteen papers (excluding Moor House) were found reporting fluxes of CH_4 (Table 4). Seventeen in Scotland, none are within the CNP boundary, one in England and one in Wales reflecting a similar country bias towards Scotland as the CO_2 data. There are an array of methods and scales from < $1m^2$ (chambers) to almost the entire north of Scotland (aircraft) (Fowler et al. 1996, Gallagher et al. 1996, Beswick et al. 1998). The numbers of sites used are again less than the number of papers indicating re-use of sites for subsequent research; some sites are more frequently reported than others. Eight of the published results come from Loch More, four from Ellergower Moss, three from Caithness and Strathy Bog and the rest of the sites are reported once. A much higher proportion of blanket bog is represented with nine sites; one raised bog, one soligenous gully mire and a gully system also sampled. As with the CO_2 studies there is no information on site management or fire, winter is also under represented with only five of the nineteen reporting winter measurements. Values range from 0.013 µmol CH_4 m^{-2} s^{-1} in Scotland to 0.131 µmol CH_4 m^{-2} s^{-1} at Cerrig-yr-Wyn in Wales. The values reported from Cerrig-yr-Wyn in Wales, and Moor House by McNamara (2008) are high in comparison to the rest but these are from gully mires a slightly different and rarer vegetation type than 'typical' bog. Nevertheless, methane fluxes from these systems may account for up to a third of the CO_2 sequestered in terms of global warming potential of this peatland (McNamara et al. 2008).

Table 5: global warming potential (GWP) of methane relative to carbon dioxide (IPCC 2007)

gas (chemical formula)	GWP time horizon (years)		
	20	100	500
carbon dioxide (CO_2)	1	1	1
methane (CH_4)	72	25	7.6

Peatlands are a major natural source of methane but the importance of methane lies in its greater warming potential relative to carbon dioxide (Table 5) which decrease over time due to oxidation in the atmosphere. Perturbation of the peatland ecosystem by fire has the potential to influence this major flux to the atmosphere. However, other than Hogg et al. (1992) and the Moor House data below, there appears to be little peer reviewed literature on the response of peatland methane fluxes to fire. Hogg et al. (1992) found that burning increased methane fluxes in peat cores, in agreement with immediate post fire methane flux data from Forsinard in Caithness and Sutherland (Figure 3). This suggest that burnt peatlands may emit more methane than unburnt but the use of peat cores is an artificial situation (Hogg et al. 1992) and the Forsinard data is only from one post burn season and one site (Gray 2006). Interestingly though the data from Moor House appears to contrast with these findings; this may be explained by differences in the time since burn that the sampling occurs (see below).

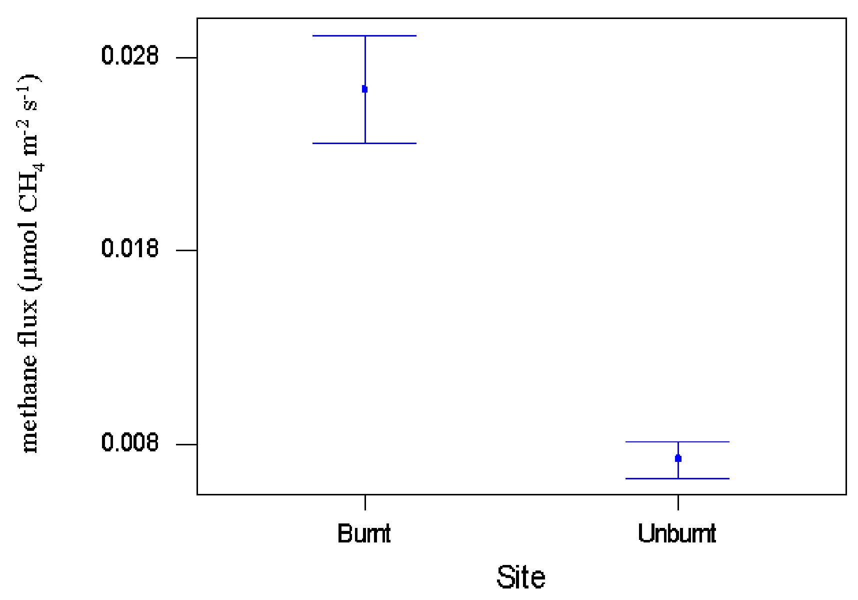

Figure 3: Mean methane fluxes (+/- SE) recorded from chambers every month for 4 months post fire. The paired burnt and unburnt site is located near Forsinard in northern Scotland (Gray 2006).

3.3 Carbon sink or source

From the available evidence, when both CH_4 and CO_2 are taken account of (excluding DOC), UK peatlands appear to be sinks for carbon (Gray 2006). However, there are severe limitations to the data including inadequate replication both in space and time. In the majority of the papers above, management practices, including fire, are barely mentioned, despite the majority of peatlands in the UK being subjected to some form of management particularly grazing and burning (Gray 2006).

Ultimately, one needs to be extremely cautious when interpreting these data and we suggest that at present it is insufficient to provide any firm conclusions on UK peatland carbon cycle dynamics. Therefore, it is also insufficient to provide any firm conclusions peatland carbon cycle dynamics within the CNP.

3.4 Moor House

Presently the most detailed evidence for the effects of fire on the peatland carbon cycle in the UK comes from Moor House NNR in the Pennines (Figure 4) at the Hard Hill site by Ward *et al*, (2007) but Garnett (2000) and Worrall *et al*., (2007) offer some further evidence aiding interpretation.

The Hard Hill experiment at Moor House is a split plot burning and grazing with unburnt, 10 year and 20 year burning treatments. The experiment is on vegetation classified using the National Vegetation Classification, as M19 b *Calluna vulgaris-Eriophorum vaginatum* blanket mire *Empetrum nigrum* ssp. *nigrum* sub-community (see Rodwell 1991). In relation to burning Ward et al. (2007) only examined a comparison between unburnt and the 10-year burn rotation (they also include grazing) and not the 20-year rotation. The results in relation to burning are summarized in Figure 4. From 1 m cores, the majority of the carbon stored in the ecosystem is located in the deeper organic peat (99%). No difference was found between the amount stored in the unburnt plots and the 10 year burning treatments. However, there were significant differences between the amount of carbon stored in the vegetation and the upper peat layer (termed the *F* and *H* horizons in Ward *et al.* (2007)). This is undoubtedly due to the differences in vegetation composition resulting from the burning treatment (Figure 5). In unburnt plots there is a greater propensity for Ericoid shrubs, hence more woody material and more stored carbon. The upper layers of the peat are derived from the immediate above ground vegetation and are therefore also likely to share comparable difference between carbon storage values. Nevertheless, the values for carbon storage in the vegetation and upper layers are dwarfed in comparison to the carbon stored in the peat layer.

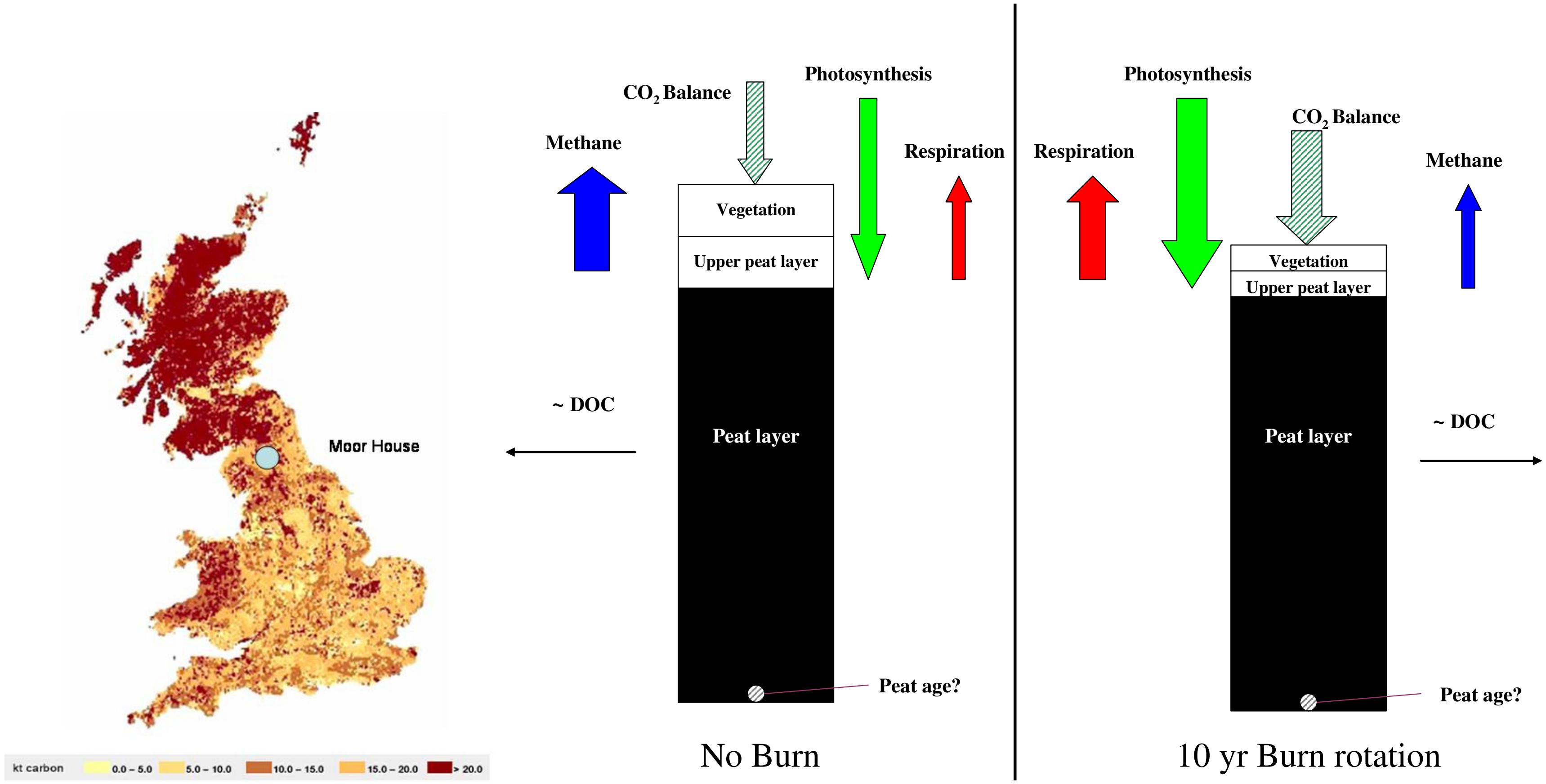

Figure 4: Location of Moor House NNR superimposed on the soils carbon map for the UK (left) and an illustrative summary of the findings at Moor House in relation to 10 year burning cycle using the results of Ward *et al*., (2007) (right). Boxes illustrate carbon stores; light green arrows indicate uptake of CO_2 through photosynthesis; red arrows indicate losses of CO_2 through respiration, hatched dark green represents the balance between photosynthesis and respiration and blue arrows indicate the loss of methane (CH_4) by anaerobic decomposition. The size of both boxes and arrows are a relative not quantitative representation i.e. they show the general pattern only. The age of the basal peat from the 1 m peat cores is unknown.

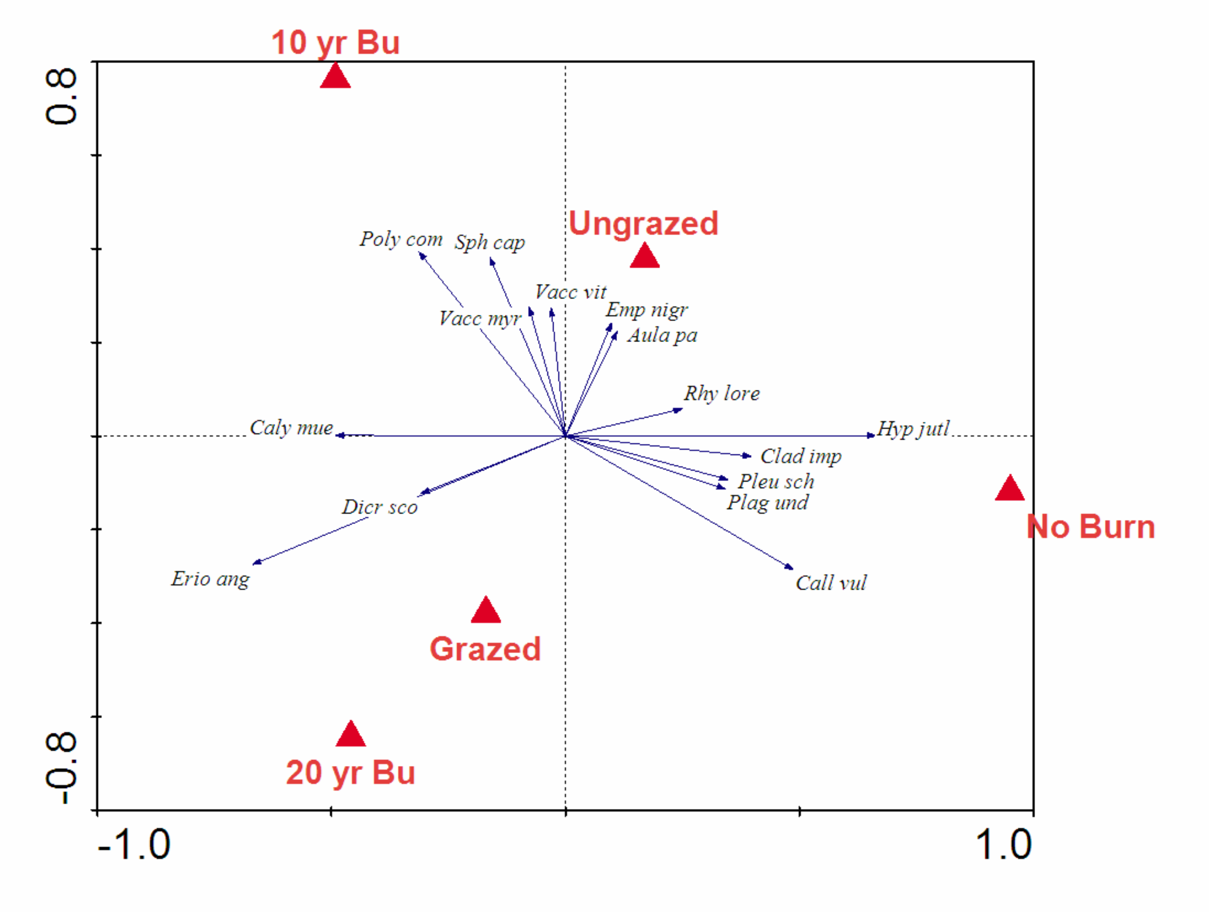

Figure 5: Axes 1 and 2 of a redundancy analysis of vegetation (% cover) from the Hard Hill experimental plots at Moor House, Pennines, England, with 0, 10 and 20-year fire rotations and grazed and un-grazed treatments. Axis 1 $p < 0.001$ (Gray 2006).

DOC rates appeared to be similar for both treatments and were more correlated to climatic variables than management treatments. Values for gaseous fluxes are a little more difficult to interpret since no actual flux rates are given. Photosynthetic and respiration rates appear greater in burnt plots but on balance this treatment still sequesters more CO_2 than the unburnt treatment (Figure 3). In addition, methane fluxes were lower in the burnt treatment than the unburnt. This appears to be curious since microbial measurements did not correlate with this pattern. Also, Worrall *et al.* (2007) reported a raised water table in burnt plots suggesting that methane fluxes should be higher, this was however, combined with a decrease in pH. Peatland microbial communities show sensitivity to temperature and moisture (Whalen and Reeburgh 1996) but evidence of pH effects are equivocal and may depend on the type of vegetation (Bergman et al. 1998, Bergman et al. 1999). Whether changes in pH account for the depressed methane flux in burnt plots is still open to question. Ward et al. (2007)

conducted the research 9 years into a 10 year burning rotation; given the different results shown in Figure 3 it is possible that methane fluxes may show a dynamic temporal response in relation to fire.

Data on the age of the basal peat core would be useful to help clarify whether burning leads to a release of peat from the burning treatments a neutral response or an increase in carbon stored. Garnett *et al* (2000) used the industrial 'take-off' in the deposition of spheroidal carbonaceous particles (SCP) as a chronological marker in the Hard Hill plots. They found that the burning treatments appeared to have accumulated less carbon in the period since the industrial 'take-off'.

3.5 Wider applicability of the Moor House studies

The above evidence is one of the best systematic fire studies for the UK and the value of a unique 50 year experiment cannot be overstated. Nevertheless, the wider applicability of the results is questionable at present and in particular to the peatlands in the CNP. Although the vegetation sampled is one of the most common mire communities in the CNP (M19 see above) Moor House is one site and as a National Nature Reserve one of the top tier of sites of conservation importance in the UK; is on a limestone substrate and is therefore likely to be atypical not only for the Pennine Region but also the UK as a whole and may not be indicative for peatlands in the CNP. The peatland vegetation at Moor House is also considered not to require burning to keep *Calluna* in a 'younger' phase (Rawes and Hobbs 1979). The growth in *Sphagnum* overtops the older growth of *Calluna* leaving the younger stems of *Calluna* as the above ground material (Rawes and Hobbs 1979). As far as we are aware this type of vegetation response has not been reported at any other site.

Again, one needs to be extremely cautious when interpreting data from one site. We suggest that at present the data on peatland carbon cycle responses to fire are insufficient to draw firm conclusions either for the CNP and the UK as a whole.

4. Peatland Carbon Modelling in Relation to Fire

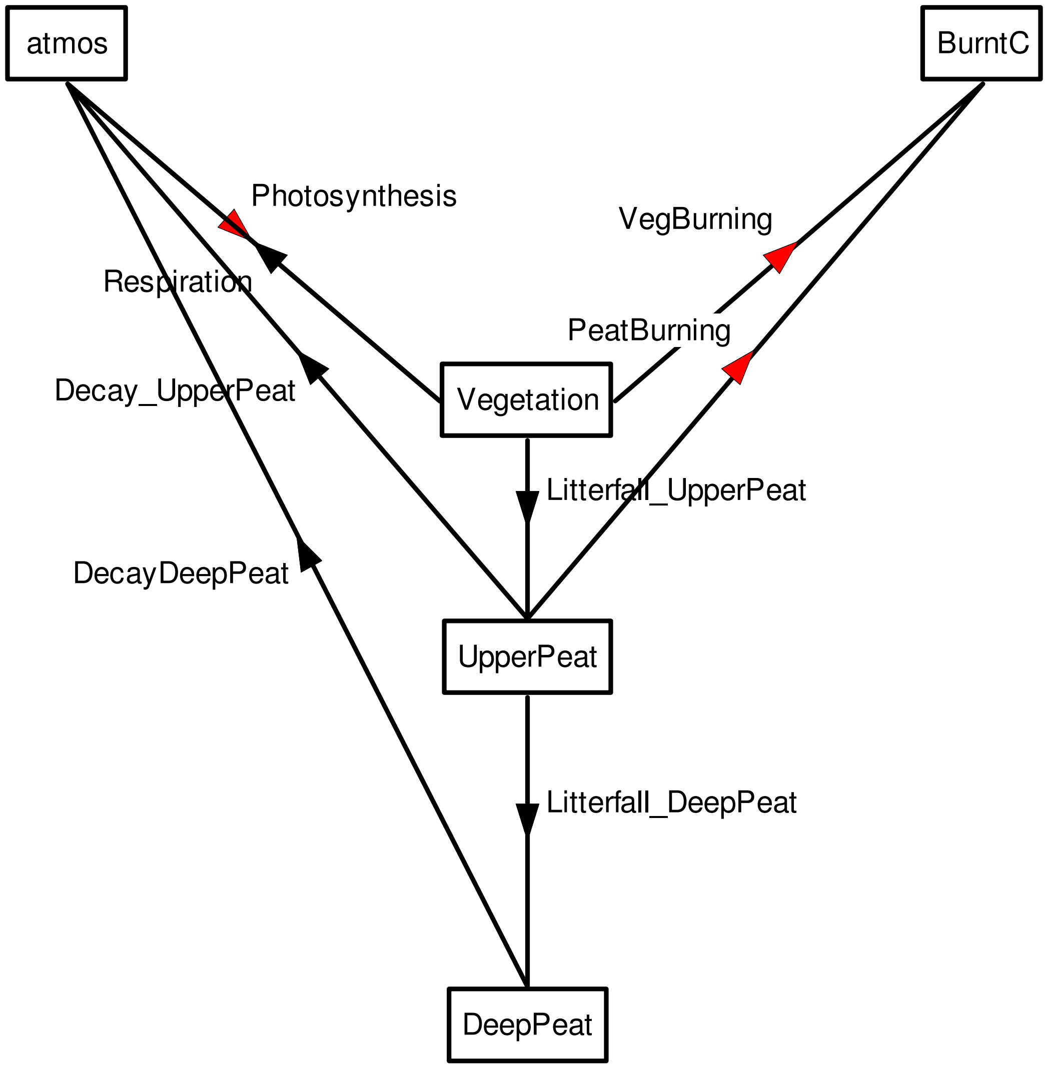

Figure 6: Diagram of simple model of peatland carbon balance, showing the pools of carbon in vegetation, the upper peat / litter layer, and a deep peat layer. Fluxes between these pools and the atmosphere are: gross photosynthesis, plant respiration, litterfall, decomposition of the peat organic matter, and burning.

Here, we develop a simple model of peatland carbon balance as affected by fire, to illustrate the potential for modelling and to analyse some of the key sensitivities. The development of user friendly models to aid land managers in decision making process within the CNP may be one of the tools that could be made available in a future project.

The model is illustrated in Figure 6, and represents the pools of carbon in vegetation, the upper peat / litter layer, and a deep peat layer. Fluxes between these pools and the atmosphere are: gross photosynthesis, plant respiration, litterfall, and decomposition of the peat organic matter, as well as burning. Burning acts to remove a fraction of the carbon in the vegetation and upper peat layer to the atmosphere, at a prescribed periodicity. The fluxes in plant respiration, litterfall, and decomposition are all represented as proportional to the pool size, with a simple turnover constant. Gross photosynthesis is prescribed based on the data of Ward et al. (2006) from the Moor House long-term burning experiment and so the above caveats apply to the CNP. These show a high value of 1.4 kg C m^{-2} y^{-1} in the 10-y burning treatment after the ten years following fire, falling to roughly half this value in the unburnt treatment, 50 years following fire. This was modelled as an increase over the first ten years following fire, followed by a linear decrease to their measured value at 50 years. The pool sizes were taken from the unburnt treatment of Ward et al. (2006), and turnover coefficients calculated assuming the system was in equilibrium in the absence of fire. Simulations were run for 200 years with a range of fire periodicities, to examine the influence of this on total carbon storage within the peatland system.

The results show a peak in carbon storage with a fire period around 30 years (Figure 7). Carbon storage declines sharply when fire period is decreased less than 15 years, and also decreases with longer fire periodicities, but to a much lesser degree. We emphasise that these results are illustrative only, and are not presented as predictions, for several reasons: (i) The shape of this curve depends on the assumed time course of gross productivity following fire, and this is linearly interpolated using two data points (and the origin). We take no account of variations in burn severity or burn depth, which affect the recovery of vegetation following fire, and hence influence the time course of gross productivity following fire. (ii) The data are from a single site, possibly atypical of moors managed for grouse in Scotland. (iii) The absolute numbers depend on the assumed fractions of vegetation biomass and litter lost in fire, although the pattern is relatively insensitive to these values. (iv) We assume the system was in equilibrium in the absence of fire, which may not be the case if there are also changes in grazing, climate, or pollutant deposition. The results do, however, illustrate the potential for a simple model, if calibrated with appropriate measurements, to synthesis the available data so as to predict the net affect of different fire management practices on total carbon storage. Some simple analysis shows that the response of carbon storage to fire period is a simple product of the growth curve following fire. All else being equal, carbon storage is maximised by burning at the point when growth is close to its maximum, analogous to the yield curve approach developed in forestry.

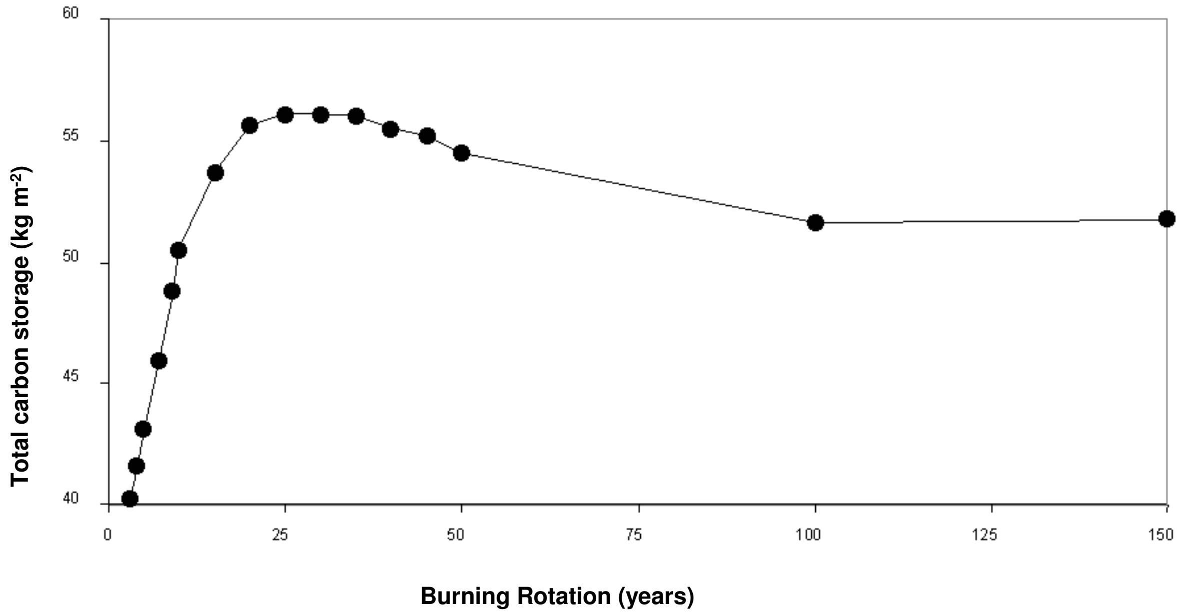

Figure7: Response of total peatland carbon storage to fire period (interval between burns) as predicted by the simple peatland carbon balance model calibrated with the data of Ward et al. (2006) from the Moor House long-term burning experiment.

In order to parameterise such a model for the CNP, data is required on gross photosynthesis, plant respiration, litterfall, decomposition of the peat organic matter, and burning. We are currently unaware of the availability of such data and no complete dataset appears to have been published at present. The ECN Cairngorms site has data on the carbon content of the soils at the site but at present the other data appear to be lacking. However, it is possible that other data sources exist, institutions such as the Macaulay Land Use Research Institute may have data relevant to the Cairngorms; access to such data would require either the initiation of a collaborative project or the funds to pay the necessary costs for access to the data. Retrieving data on burning regimes in the CNP area will undoubtedly require close collaboration with estate managers.

5. Discussion

5.1 Modelling

As the information available on gaseous carbon flux data in the UK and CNP is sparse, it is tempting to try and incorporate data from others areas e.g. continental Europe or North America. However, fire research appears to be lacking globally (but see Turetsky et al. 2002, Turquety et al. 2007, Shelter et al. 2008, O'Donnell et al. 2009). In addition, the UK and CNP have an oceanic climate unlike the more continental climate of Europe and North America. Also, unlike European and North American continental and northern boreal peatlands, UK peatlands have been subjected to deliberate management practices for many centuries; consequently UK peatland ecosystems are in no way pristine or undisturbed this is also the case for the CNP. Therefore, the magnitude and variation of peatland responses in the CNP are likely to differ not only on terms of biology but also physically from those on the continent. This may have important consequences for any modelling approach. Nevertheless, we have demonstrated that a simple modelling approach can be an informative tool in relation to fire although further research is required to parameterise models sufficiently.

5.2 Fire and carbon flux research

Peatlands are a fundamental aspect of the global carbon balance (Matthews and Fung 1987, Gorham 1991, Immirizi et al. 1992). Crucially the carbon cycling processes, in particular gaseous fluxes, that determine carbon balance for peatlands are spatially and temporally variable. Clymo and Reddaway (1971 & 1972) made what may have been the first ever attempt at balancing gaseous fluxes at Moor House. However, peatland gaseous flux research in the UK has lapsed somewhat in recent years. In the north American and European continents, research has continued and have helped elucidate the relationships between environmental controls, the impacts of forestry, drainage and restoration on gas fluxes in peatlands (Billings et al. 1982, Crill et al. 1992, Dise 1992, Martikainen et al. 1992, Oechel et al. 1993, Whiting and Chanton 1993, Bubier 1995, Christensen et al. 1996, Waddington et al. 1996, Bridgham et al. 1999, Christensen et al. 1999, Joabsson et al. 1999, Komulainen et al. 1999, Tuittila 2000, Aurela et al. 2001, Aurela et al. 2002, Blodau 2002). However globally, research on the disturbance due to fire is lacking. Fire may be a key interaction between peatland carbon balance and the climate and requires to be investigated on large scales (Turetsky *et al*. 2002). Although techniques for measuring continuous CO_2 have been in use for a number of years, techniques for the continuous measurement of CH_4 are only just becoming cost effective and more widely available. Previously campaign measurements were only possible (Beverland *et al.* 1996). Now, tunable diode lasers (TDL) are available that make fast automatic measurements, so that CH_4 can be measured by eddy covariance. This means that the investigation of the balance between CO_2 sequestered and CH_4 emitted can be examined at high temporal and spatial scales. The integration of innovative technology with the investigation of peatland disturbance on fluxes will is likely to be a prime area of research. Future carbon flux research in the UK needs to address fire on a reasonable spatial and temporal scale to successfully parameterise models. There is no reason why this approach cannot be applied to peatland ecosystems within the CNP and we suggest that research within this National Park is encouraged if an informed evidence based approach to peatland carbon and fire is to be adopted.

5.3 Relevance of previous studies to Cairngorms National Park

At present it is difficult to gauge the relevance of the above information to the Cairngorms National Park, fundamentally because of the paucity of applied research. In addition, we do not currently have access to CNP estate level information on the history and current extent of fire use, the fire regimes implemented, i.e. at what frequency, intensity, size, and seasonality of prescribed burn practiced within the CNP. Details on the distribution of soil types and vegetation are also required. Access to this information would greatly help interpretation. Implementation of an integrated research programme involving landowners, scientists and other stakeholders would also alleviate some of the key knowledge gaps this would not only be relevant to CNP but also to the UK.

6. Conclusions

There is insufficient data at present for either firm conclusions to be drawn on the effects of fire on the carbon cycle of CNP peatlands or to sufficiently parameterise local or national models.

Much of the evidence is from a single site (Moor House in the Pennines) and whether the patterns found there are applicable within the CNP is open to question.

The use of prescribed fire is currently discouraged on UK peatlands (Anon 2001) but there is an urgent need for applied research to inform management practice and policy, particularly in the light of climate change to address key knowledge gaps. Research into the effects of fire (and herbivores) on peatland carbon balance should be a priority to identify options for the reduction of losses of carbon from peatland store to the atmosphere. We suggest that an inter-disciplinary approach to the problem is required and the CNP would be good place to start. However, funding remains a pervasive issue but given the policy implications of the required research it should undoubtedly involve government funds. We would strongly encourage that a combined approach should involve seeking funds for all stakeholders including landowners as practitioners, scientists and other stakeholders to approach a diversity of funding opportunities towards a common goal of addressing the current knowledge gap.

6.1 Research Requirements

Research on prescribed fires within CNP peatlands might include such questions as:

1. When, and at what frequency, is prescribed fire good and when is it better to risk accidental wildfire?
2. What are the effects of using prescribed fire on UK peatland biodiversity?
3. How are other ecosystem services affected by the use of prescribed fire?
4. How does the interaction between prescribed fire and grazing impact on peatlands?
5. How does fire affect microbial activity and community composition?
6. What are the long and short-term implications for the use of fire on peatland carbon cycle processes?
7. What are the social consequences of either using or not using prescribed fire in peatlands?

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