The Impacts of Heather and Grassland Burning in the Uplands: Creating Sustainable Strategies

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"igne natura renovatur integra"

through fire, nature is reborn whole

- Alchemical aphorism

THE IMPACTS OF HEATHER AND GRASSLAND BURNING IN THE UPLANDS: CREATING SUSTAINABLE STRATEGIES

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Abstract

Both nationally and globally, UK upland peat is an important store of carbon as well as a source of other important ecosystem services. However, concerns have been raised regarding the stability of these stores. Significant increases in water colour and dissolved organic carbon (DOC) from catchments draining upland peat have been observed across the UK.

Unlike many boreal peats, the peat soils of UK uplands are heavily managed for sheep grazing and recreational shooting. Productivity of these landscapes has been increased through managed burning of the vegetation. Burning has been linked with increases in water colour and inappropriate burning can lead to ‘unfavourable’ conditions in these landscapes.

This thesis presents the results from a monitoring programme at Moor House National Nature Reserve. Results show that burning does not lead to dramatic increases in DOC and that longer rotations may have benefits for carbon by reducing water colour. Increases in the occurrence and changes in the quality of runoff water following burning could help explain changes in water quality parameters such as DOC.

Experimental studies into biomass loss during burning, combined with a survey of a wildfire, have shown that the production of char is an important carbon store that should be accounted for in fire prone upland settings. Modelling studies suggest that rotation lengths of 15 years are suitable for char production and that on these longer rotations char becomes a more important carbon store than any remaining unburnt biomass or litter.

Therefore this work would suggest that longer rotations may have benefits for carbon storage and water quality. Longer rotations may be sustainable in some areas but that this is unlikely to be appropriate across the entire of the UK. The caveats to this work should always be presented and local knowledge be consulted when drawing up management plans.

Declaration and Copyright

I confirm that no part of the material presented in this thesis has previously been submitted by me or any other person for a degree in this or any university. Where relevant, material from the work of others has been acknowledged.

Signed:

Date:

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1.1 Overview and project rationale

A commonly used management strategy in the UK uplands is managed burning in order to maintain levels of heather and grass for grazing and grouse management. This preserves a landscape that has high amenity and resource value. However, is this sustainable? Many upland habitats e.g. heathland, are now considered unfavourable and declining due to overgrazing and/or burning (Natural England, 2008). This increases the loss of sediment, changes water colour, and affects terrestrial carbon storage. So, what is the best way to manage burning and grazing of the uplands?

For many decades botanists and ecologists have been investigating the effects of prescribed burning on floristic changes and community succession in moorland habitats (e.g. Gimingham, 1972; Hobbs, 1984; Webb, 1986). In recent years the concept of “ecosystem services” has been proposed as a way to value landscapes for the services they provide (Millennium Ecosystem Assessment, 2005):

- Provisioning e.g. water supply
- Regulating e.g. carbon sequestration
- Supporting e.g. nutrient cycling
- Cultural, e.g. recreational experiences

UK uplands support all these categories of ecosystem service and so upland land management is likely to affect one or more of these services. Recent reviews on the consequences of moorland burning (Glaves and Haycock, 2005; Tucker, 2003) found that there are few studies that investigate the consequences of burning on water quality, hydrology, or soil quality.

In order to predict future changes in UK peatlands, a clear understanding of the processes at work is needed. Future drivers, such as climate change and economic fluctuations, are likely to affect these ecosystems through both direct effects e.g. increasing temperatures, and indirect effects such as changes in the rural populations leading to progressive rural collapse. In order to manage the UK uplands for the future an integrated understanding of these landscapes will be required. This work will assess the impact of managed burning and grazing on peatland hydrology and carbon balance and will investigate the social implications of this management technique.

1.2 Research Review

1.2.1 Peatlands

Peatlands develop in waterlogged conditions that create anaerobic conditions which inhibit decomposition of plant material. The combination of oxygen-poor conditions coupled with low temperatures and low nutrient availability, lead to the ongoing accumulation of organic matter. Definitions for peat vary, depending on country-specific definitions, though a common definition is any soil with an organic content greater than 50% and greater than 30cm in depth (Johnson and Dunham, 1963).

Worldwide, peatlands cover between 386 and 409 million ha (Immirzi et al., 1992) and are principally found in the northern latitudes. Interest in peatlands has increased in recent years due to their importance in storing carbon. Approximately 455 Pg of carbon, one-third of the world's soil carbon, is sequestered in peatlands (Gorham, 1991) although they only occupy 2% of the Earth's surface (Updegraff et al., 2001).

It is estimated that there is between 14,000 and 29,000 km^2 of peat in the UK (Milne and Brown, 1997; Tallis et al., 1998) which contributes around 13% of the global blanket bog resources (Ratcliffe and Oswald, 1988). In the UK some 3 billion tonnes of carbon are stored in peat deposits (Cannell et al., 1993). There are limited areas of lowland peat concentrated in the Cambridgeshire and Norfolk Fens and the Somerset Levels. The remaining peat is located in upland areas where it overlies many headwater catchments. As such, peat's hydrological characteristics play an important role in influencing stream flow and water quality in these catchments.

1.2.1.1 Hydrology

Changes in the water balance of a peatland have profound consequences for other biogeochemical cycles at work within the peat profile. The simplest expression for the water balance of a peatland is given by (Eggelsmann et al., 1993):

$$\text{Influx} - \text{efflux} - \text{changes in storage} = 0 \qquad \text{(Eq. 1.1)}$$

However, simple 'black box' expressions like equation 1.1 do not satisfactorily show the complexities in hydrology in peatlands. Changes in precipitation, groundwater inflow and evapotranspiration, amongst others, will affect the water balance in peat and consequently peat profiles.

Ingram (1978) proposed a two-layer model to explain the sharp transition in peat profiles from loose, decaying vegetation to more humified denser peat. The upper layer is termed the acrotelm and the lower layer is the catotelm. The definition for the boundary is commonly taken as the deepest point the water table reaches in its annual cycle (Charman, 2002). The acrotelm is the zone where aeration and microbial activity is high leading to some authors to term it the 'active' layer. Variations in water table, within the acrotelm, can lead to changes in carbon fluxes within this zone. By increasing depth to the water table, a greater depth of peat is exposed to aeration and therefore potentially carbon export (Worrall et al., 2004). By raising water table, microbial activity can be reduced, leading to lower dissolved organic carbon (DOC) export and CO_2 emissions, though this is at the expense of CH_4 production.

Variations in water table have shown to play an important role in controlling enzyme activity. Freeman et al. (2001a) show that following the lowering of the water table, phenol oxidase activity increases which in turn destroys phenolic compounds that would otherwise repress hydrolase enzymes. These enzymes are important in the restriction of decomposition, and therefore DOC production, in peat. As water tables rise again, decomposition can continue due to the destruction of the phenolic compounds. Another mechanism for DOC production following water tables lowering include Clark et al. (2005) who make a link between soil sulphate and the suppression of DOC concentrations. Therefore, severe droughts appear to be an important mechanism for DOC production (Worrall and Burt, 2008). However, the regular cycle of water tables and other hydrology variations will also play a part in the carbon dynamics of peatlands.

1.2.2 Carbon dynamics in peatlands

There are many definitions of carbon cycling in ecosystems (for a review see Chapin et al., 2006); however, the underlying model is the change in carbon over time:

$$NECB = \frac{dC}{dt} \qquad \text{(Eq. 1.2)}$$

where NECB is the net ecosystem carbon balance, dC is change in carbon fluxes and dt is change in time.

Early work on carbon accumulation in ecosystems often defined the net rate of carbon accumulation as the difference between photosynthesis and ecosystem respiration (Woodwell and Whittaker, 1968). However, when considering different ecosystems over varying timescales, other carbon fluxes such as dissolved, volatile and depositional can also be important. Therefore, the difference between ecosystem primary productivity and respiration does not necessarily equal carbon accumulation.

In calculating carbon accumulation in peatland settings, two methodological approaches have been employed. The first uses radiocarbon dating to calculate ages of peat, and therefore calculate the rate of carbon accumulation (Belyea and Warner, 1996; Kilian et al., 2000). However, there are problems associated with this method. This technique can only calculate average accumulation rates above a horizon and not periods of carbon loss from the peat. Short term (10 – 20 years) changes in carbon cycling may also be poorly represented (Hilbert et al., 2000). The second main approach is to calculate present day carbon budgets based on fluxes of carbon through various pathways. Typically, fluxes are measured directly from a site or catchment and calibration curves created between them and readily and continuously measured variables e.g. temperature, stream flow.

Worrall et al. (2003a) provide the first comprehensive study of the carbon balance of an UK upland peat by using a North Pennine catchment, Trout Beck, as the study site. Pathways included in calculating the carbon budget are: rainfall DIC and DOC; CO_2 exchange; CH_4 emissions; DOC export; POC export; dissolved inorganic carbon and dissolved CO_2 and input from weathering of underlying strata. Results from this work show that the site in 1999 was a net sink of 14.9 gC m^{-2} yr^{-1}. Further work this site improved the method and was able to predict future changes by modelling the catchment on forecasted rainfall and temperature data (Worrall et al., 2007a). More recently, Worrall et al. (2009a) present a multi-annual carbon budget of the Trout Beck catchment and show that over the period 1993-2005, the site was a net sink of between 20-91 gC m^{-2} yr^{-1}. However, it should be noted that no full carbon budget exists for sites under managed burning.

When calculating carbon budgets of peat, it is important to understand the different pathways by which carbon may be lost or gained. The following sections detail some existing research on aspects of the peatland carbon cycle.

1.2.2.1 Dissolved Organic Carbon (DOC)

Dissolved organic carbon (DOC) is a collective term for dissolved and colloidal organic compounds in various stages of decomposition therefore consisting of a variety of molecules from simple acids to complex humic substances (Wallage et al., 2006). DOC is operationally defined as carbon that passes through a 0.45μm syringe filter (Roulet and Moore, 2006)

There is growing evidence for large increases in the DOC concentration of terrestrial water draining boreal and sub-Boreal peat soils (Monteith et al., 2007). These increases have been observed in Europe (Hejzlar et al., 2003) and North America (Driscoll et al., 2003). For the UK, Worrall and Burt (2007) have shown that for 315 sites across the UK, 68% showed a significant increase in DOC concentration over time (8 to 42 years) while 18% of records showed a significant decline. The catchments showing a decline were predominantly the peat-covered catchments of the south-west of England.

Water supply companies, in particular, are interested in better learning the mechanisms whereby DOC arrives in water as DOC discolours the water and is costly to remove especially in regions with peat-covered catchments. Not only does DOC discolour water, leading to low aesthetic quality, it also increases the potential for biological contamination as it consumes the free residual chlorine. Finally, it can form potentially carcinogenic tri-halomethanes whose concentration in drinking water is limited by law in the UK (Hsu et al., 2001).

There are several possible mechanisms causing the observed increases in DOC these include: increasing air temperature (Freeman et al., 2001b); changes in pH (Lofts et al., 2001); change in the amount and nature of flow (Tranvik and Jansson, 2002); increases in atmospheric CO_2 (Freeman et al., 2004); changes in atmospheric deposition (Evans et al., 2005); occurrence of severe drought (Worrall and Burt, 2004); eutrophication (Harriman et al., 1998) and these factors could be enhanced by local land management (Wallage et al., 2006). It is likely that some or all of these drivers have contributed to increases in DOC concentrations.

1.2.2.2 POC

Fluvial export of carbon from upland catchments has received much attention (e.g. Hope et al., 1997; Worrall et al., 2003b) with the aim of understanding how these stores will respond in light of potential changes to climate and the impact of carbon sequestration that that might bring. Research into fluvial carbon often focuses on dissolved organic carbon (DOC) and particulate organic carbon (POC). Many studies have shown that DOC plays the most significant contribution to fluvial carbon export. Dawson et al. (2002), in a study of two catchments in Wales and Scotland, that DOC contributed 69 and 88% of fluvial carbon export respectively. Hope et al. (1997) show that in two Scottish rivers, POC contributes between 10 – 25% of total organic carbon flux. While POC fluxes may represent a small portion of fluvial carbon export in some areas and over long time scales, POC export is episodic in nature and responds to high flow conditions (Hope et al., 1997). Pawson et al. (2008) show that for a degraded catchment in the South Pennines that POC export constituted 80% of the organic carbon export. This study also showed that during high flow events 95% of POC export occurs within 8% of the event period, again indicating the periodic nature of POC export.

1.2.2.3 CO_2

Climate change is predicted to increase global surface temperatures by up to 4°C by the end of the 21st century (IPCC, 2007). One of the contributors to this effect is the increase in global CO_2 concentrations which have garnered much interest since the 1950s. CO_2 concentrations continue to rise and current estimates put CO_2 concentrations in the atmosphere at 382.7ppm in 2007 (Levinson and Lawrimore, 2008).

Understanding the feedback mechanisms in the terrestrial-atmospheric carbon cycle is an important area of research due to the large amount of carbon in terrestrial stores. Soil carbon is one of the largest stores with estimates ranging from 1110 to 2200 Pg C (Batjes, 1996). Peat soils are estimated to contain 455 Pg of carbon (Gorham, 1991) and as such understanding the drivers behind CO_2 exchange from these areas is increasing important. There have been several studies that investigate CO_2 emissions from UK peats (e.g. Kechavarzi et al., 2007; Lloyd, 2006) and many studies have investigated drivers for these changes; soil moisture (Glatzel et al., 2006); water table (Oechel et al., 1998; Silvola et al., 1996); and soil temperature (Updegraff et al., 2001).

1.2.2.4 Methane

Methane is an important driver of climate change, as although it accounts for a smaller proportion of carbon in the atmosphere, 1,774 ppb compared to 379 ppm for CO_2 in 2005 (IPCC, 2007), it has a greater global warming potential than CO_2; it is 62 times more effective than CO_2 over a 20-year time scale (Hargreaves and Fowler, 1998).

Measurements from peatland settings are made either using a closed chamber approach (Bortoluzzi et al., 2006) or using an eddy covariance method (Fowler et al., 1995). MacDonald et al. (1998) report CH_4 fluxes for a blanket bog in Scotland between 0.16 and 13.5 gC m^{-2} yr^{-1}. In a study on a

Finnish mire Hargreaves et al. (2001) report annual emissions of 5.5 ± 0.4 $gCH_4\ m^{-2}\ yr^{-1}$.

1.2.2.5 Dissolved CO_2

Excess dissolved CO_2 is commonly defined as the amount of dissolved CO_2 above which would be expected to be present if the water was in equilibrium with the atmosphere. There are three main methods used in calculated excess CO_2: titration based methods where the CO_2 can be calculated based on alkalinity and pH (Neal et al., 1998); direct measurements of dissolved CO_2 in solution by headspace analysis (Hope et al., 1995); and direct measure of CO_2 from the stream surface using floating chambers (Billett et al., 2006).

In their study of a partial peat-covered catchment Dawson et al. (1995) estimate the dissolve CO_2 flux to be 0.4 $gC\ m^{-2}\ yr^{-1}$. Studies have found that dissolved CO_2 represents a small percentage of the total carbon budget (e.g. Worrall et al., 2009a) though it can form a significant proportion of fluvial carbon flux (e.g. 24%, Worrall et al., 2007b)

1.2.3 Upland land use

Upland areas of the UK host many important ecosystems and unique species. One of these is moorland which often develops on the poor, acidic soils found in upland settings. In the UK, moorland covers 38% of Scotland, 5.5% of England and Wales and 8% of Northern Ireland (Holden et al., 2007). These communities are characterised by small shrubs such as heather (*Calluna vulgaris* (L) Hull) or bilberry (*Vaccinium myrtillus* (L)), and sedges such as cotton grass (*Eriophorum spp*.). In wetter conditions such as peat bogs, the main peat-forming species are often bog mosses (*Sphagnum spp*.).

Various drivers alter the community composition across the UK including: north-south and altitudinal climate variation; east-west precipitation variation; local drainage conditions; prescribed fire management; wildfires; differing grazing pressures; other management e.g. afforestation, gripping; and acidic deposition (Holden et al., 2007). In addition to supporting a diverse and globally important ecosystem, upland areas also support multiple land uses: (1) water supply; (2) agriculture; (3) commercial forestry; (4) sport and leisure; (5) tourism (Bonn et al., 2009a; Reed et al., 2009a).

An example of a highly valued ecosystem service in the UK is the provision of clean water. Over 70% of the fresh water in the UK is sourced from upland catchments and this is of particular importance in areas of high demand such as the reservoirs in the Peak District which provide 450 million litres of waters a day to the neighbouring towns and cities (Bonn et al., 2009b). The removal of water colour, or dissolved organic carbon (DOC), is a major cost to water companies so many are looking for alternatives to this ‘end of pipe’ removal and are investigating the impact of altering land management in the catchment in order to reduce colour at source.

With these different land uses the peat soils of the UK have been heavily and extensively managed for many centuries (Lovat, 1911). Drainage (Worrall et al., 2007c) and grazing (Palmer et al., 2004) are common features of the uplands along with prescribed fire. In order to increase productivity of these ecosystems for sheep and grouse, managed burning of has been a common feature of these landscapes (Yallop et al., 2009).

1.2.3.1 Managed burning

Fire has been a common part of the uplands of the UK for many hundreds, even thousands, of years. Whilst there is evidence for that fire may have been used to clear land since Neolithic times (Fyfe et al., 2003), it was not until the late mediaeval period when burning started to become a common management practice. Records show that burning, or ‘swaling’ was a common practice on Exmoor in the 1300s to improve pasture (Rackham, 1986) and records in Scotland show the term ‘muirburn’ occurs in an Act of Scottish parliament of 1400 (Dodgshon and Olsson, 2006). The use of managed burning for habitat maintenance for grouse spread rapidly during the middle of the 19th century. Prior to this burning was carried out predominantly for sheep grazing where the priority was for large areas of regenerating vegetation. These burns were often larger and carried out using rough and ready methods. The current method of strip burning was known to occur in the 19th century, however, it was not until an inquiry into grouse disease in 1911 (Lovat, 1911) that the practice started to become codified.

Early research into the practice looked at rejuvenation following fire (Fritsch and Salisbury, 1915) though much of the seminal work on burning in heathland settings was carried out in the 1980s where again the vegetation responses to burning were examined (Hobbs and Gimingham, 1984b; Hobbs, 1984; Mallik and Gimingham, 1983). The burning of peatland vegetation promotes the development of grass-dominated communities especially on shorter burning rotations (Hobbs, 1984). This vegetation response improves grazing for sheep and is reflected in higher sheep performance on burnt plots (Lance, 1983). Grouse production has also been correlated with the density of burnt areas (Picozzi, 1968). However, recent reviews of the consequences of heather and grassland burning, including that on peat (DEFRA, 2005; Tucker, 2003), found there were very few studies that examined the consequences of burning for hydrology, water or soil quality.

Currently, up to 40% of English moorland is under burning management (Yallop et al., 2005) and in the year 2000 17% of upland heath and bog in the North Pennines Area of Outstanding Natural Beauty (AONB) had been burnt within the past 4 years (Yallop et al., 2006). Burning is regulated by the UK Department for Environment, Food and Rural Affairs (DEFRA). DEFRA recommends (DEFRA, 2007a) that individual burns should; not exceed 2 hectares with a maximum width no greater than 30m (DEFRA, 2007a); burning that is no more frequent than once every 12 years; and finally, that burning is restricted to being between 1st October and 15th April. The aim in restricting burning practice is to ensure a 'cool' burn by not allowing the peat and vegetation to have dried out during hotter summer months. A 'cool' burn aims to remove the vegetation layer without damaging the underlying peat.

1.2.3.2 Fuel load reduction

Fire suppression is often adopted when fire can be seen as detrimental to the environment e.g. forestry. However, the build up of fuels, both fine and coarse, over time can lead to catastrophic fires occurring. Prescribed burning is often used around the globe to reduce fuel loading in the particular area in order to reduce wildfire risk (Agee and Skinner, 2003; Fernandes and Botelho, 2003). Some studies from the United States investigating wildfire risk in forests, suggest that some reduction of surface fuels through prescribed burning of forest may lead to reduced fireline intensity (Vaillant et al., 2009) and reduced wildfire risk through increased return intervals (Shang et al., 2004).

Research on prescribed burning of shrubland for the purposes of fuel reduction is limited, though it is a commonly held belief amongst many land managers that this works. Of those that have investigated this hypothesis, Baeza et al. (2002) suggest that frequent low-intensity prescribed fires are able to reduce the risk of catastrophic fires in Mediterranean gorse shrublands. A benefit from regular managed burning of upland moors may be the reduction of wildfire risk through the removal of old woody material, though research into this claim for UK settings is sparse.

1.2.3.3 Sheep Grazing

The uplands of the UK have been grazed by sheep for hundreds of years. Early woodland clearance was for the improvement of grazing land and between the 12th and 14th centuries, the monasteries developed an extensive use of sheep grazing to produce saleable wool. However, it has only been in recent times that changes to agricultural subsidies and economic pressures has altered the way sheep are managed in the uplands. Most moorland cannot support grazing above two sheep per hectare. European Union Common Agricultural Policy (CAP) subsidies in the 1970s along with the 1975 Less Favoured Area (LFA) Directive, which favoured livestock production, led to major increase in sheep numbers during the 1980s (Fuller and Gough, 1999). This change in policy led to increased stocking rates during the 1970s and 1980s; by 1977, 29% of moors were stocked above 2 sheep per hectare and by 1987 71% were above this level (Holden et al., 2007).

Grazing in the uplands of the UK has received attention in the literature for several reasons: conversion of grassland to shrubland and woodland (Hope et al., 1996); increased soil erosion (Evans, 1996); decrease in soil infiltration (Langland and Bennett, 1973); and increase in surface runoff (Burt and Gardiner, 1984). However, the number of studies focussing on the effect sheep-grazing has on soil and water quality are limited (Shand and Coutts, 2006; Worrall et al., 2007a). Common techniques to investigate grazing include the use of sheep exclosures to examine changes after exclusion of animals from an area (Ball, 1974; Hill et al., 1992) or by calculating stocking rates of animals across an area (Pakeman and Nolan, 2009).

Managed burning historically has also been used to improve the land for grazing (Rackham, 1986). Some studies have looked at the effect the interaction between grazing and managed burning has on: vegetation (Grant and Hunter, 1968); DOC (Worrall et al., 2007d); and carbon accumulation (Garnett et al., 2000).

1.2.4 Wildfires

Wildfires are common phenomena within many ecosystems with vegetation fires present in tropical, temperate and boreal regions. Worldwide, vegetation fires are estimated to burn between 530 and 555 × 10^6 ha per year (Gonzalez-Perez et al., 2004). The majority of burned areas, 80-86%, are located in the grasslands and savannas of Africa and Australia (Mouillot and Field, 2005) though biomass burning is a global-scale and continuously occurring activity with fires taking place year round in both hemispheres (Carmona-Moreno et al., 2005). Andreae (1991) estimates average annual biomass consumption through fires to be approximately 8.68 Pg. Crutzen and Andreae (1990) estimate the annual release of carbon from wildland fires and other biomass burning to be 1.8 – 4.7 Pg C. Wildfires not only represent a loss of biomass and ecosystem carbon stocks but also have damaging effects upon human health and well being, the economy and biodiversity (Lohman et al., 2007). For example, the devastating wildfires in Victoria, Australia in February 2009 led to the death of over 170 people, serious injury to 500 more and the destruction of 2,000 homes (Hill, 2009).

The UK routinely experiences wildfires and vegetation fires. In the period between 1974 and 2005, the Fire Service attended on average 71,700 vegetation fires a year with more frequent fires in periods of drought e.g. in 1995 and 2003, 174,600 and 152,700 fires respectively were recorded

(McMorrow et al., 2009). Many wildfires in the UK occur on moorlands and peatlands.

Current research would support the view that in a warmer climate an increase in area burned (e.g. Flannigan and Vanwagner, 1991) or increases in fire occurrence (e.g. Li et al., 2000) will be likely, although there is likely to a good deal of spatial variability with some areas of no change and some predicted decreases in occurrence and area. For a comprehensive review of the implications of a changing climate for global wildfires see Flannigan et al. (2009).

1.2.5 Socio-economic factors

In two recent major reviews of the effects of moorland burning (DEFRA, 2005; Tucker, 2003), neither investigated the effect on the socio-economic factors of burning e.g. impact of available rural labour, expertise of keepers. Whilst not in the scope of these reviews, which primarily investigated biophysical and hydrological changes, they highlight that work has been carried out on these other factors and that social factors should be taken in account when changing regulations or policy. In making any recommendations about burning practices, an understanding of socio-economic problems and current policies in the uplands should be considered.

1.2.5.1 Economic value of the uplands

The uplands of the UK support a diverse economy with perceived traditional incomes such as agriculture and grouse shooting sitting alongside manufacturing, real estate and retail trade. The economic profile and activities in rural areas are often similar to those in urban areas often with higher rates of self employment (Hubacek et al., 2009).

Grouse moors support many direct jobs i.e. grouse keepers, and also many secondary jobs e.g. local bed and breakfasts. McGilvray (1995) calculated grouse shooting provided £14.7 million in wages in Scotland in the early 1990s and it supported 904 full-time jobs in the hotel industry. The larger sporting shooting industry has been calculated to be worth £1.6 billion to the UK economy, with 12% or £120 million, spent on grouse-shooting in good grouse years (PACEC, 2006).

Upland grazing on moorland is often a low-input and extensive exercise that often yields little or no return on investment. The change from headage payments to agri-environment schemes has meant that public payments often account for most of the farm income. Gardner et al. (2009), in an assessment of 14 grazing regimes, show that without the addition of payments from Single Payment Scheme or Higher Level Stewardship, the net margin of all regimes is an overall loss. With the addition of these payments, most regimes return a net profit though intensive al-year sheep regimes still remained a net loss.

Tourism is a large income source for upland areas. Visitors to the Peak District National Park reach 22 million visitor days annually (Peak District National Park Visitor Survey, 1998) and the estimated overall business turnover arising from tourism in the Peak District National Park was £75 million. Within the National Park, the estimate for visitor spending in 1998 was £185 million, which supports over 3,400 jobs, representing 27% of total employment (Peak District National Park Visitor Survey, 1998).

One untapped potential for these upland areas is in carbon sequestration schemes. The ability of peat to sequester large amounts of carbon has been suggested as a mechanism for carbon offsetting in upland areas (Worrall et al., 2009b). If all bogs in England and Wales were in pristine condition they could absorb 400,000 tonnes carbon a year (Worrall et al., 2003a). At the Sixth Conference of Parties to the United Nations Framework Convention on Climate Change (UNFCCC) it was agreed that countries could use carbon sequestration from human activities on grazed land, crop land revegetation or by forest management to help meet emissions targets (Worrall et al., 2003a). Much of the uplands of the UK are grazed and could be used to meet emissions targets.

1.2.5.2 Upland Policy

Management of the uplands is often changed or modified by changes to local, national or international policy. The CAP reform replaced subsides based on production with Single Farm Payments that reward farmers for using sustainable agricultural practices that help to promote wildlife (Lowe et al., 2002); however, the long-term effects on rare and uncommon species has been questioned (Kleijn et al., 2006).

Agricultural incentives following the Second World War led to the mass drainage of many peatland areas in order increase yields from forestry and agriculture (Holden et al., 2004). However, by the late 1970s and early 1980s studies showed that there was no evidence to show that the promised benefits had emerged (Stewart and Lance, 1983; Watson and Ohare, 1979). Many of these drains, or grips, are now being blocked up as part of large investments by land owners, water companies and other upland stakeholders (Armstrong et al., 2009). Changes in policy can lead some stakeholders to question the reasoning behind the changes and why local knowledge is not included in the process:

> "No one on the conservation side has explained to me yet why their view of the world will be anymore correct (whatever correct is) than the Forest Commission's was in 1976 when we were all taught to…plough up heather moorland, and yet everybody now assumes

that they're right…I've spent thirty years managing land and I've seen all these things come and go. So when you tell me as a very sincere young man with a great deal of credentials, that your prescription is right, you just listen to me: the guy who gave me 100% grant aid…to plough heather moorland also believed he was right because heather moorland was “waste”. “Why keep heather moorland? Why not grow Sitka Spruce on it?” They weren't all liars and cheats and thieves and incompetents. That was not the case. And they all look at you in absolute amazement.”

Anonymous grouse moor agent (Holden et al., 2007)

1.3 Aims and objectives

- To conduct a sustained fieldwork campaign monitoring burnt and grazed land and to use the results to investigate the effects of burning on carbon dynamics in a peatland setting.
- To examine the differences between burning regimes and suggest mechanisms for these differences.
- To calculate the carbon budget before and after managed burning.
- To understand the immediate effects of burning on surface carbon stocks.
- To investigate spatial variability of a moorland wildfire.
- To investigate with stakeholders their current views on managed burning.
- To explore the public's views on managed burning.

1.4 Scope and organisation of the thesis

The research can be broadly divided into three sections:

- Chapter 2 deals with data obtained from a long-term, plot scale experiment to monitor hydrological parameters and carbon pathways. It allows an assessment of the long-term impact of burning and grazing on hydrology, carbon and water. A managed burn that occurred midway though the monitoring allows changes before and after burning to be assessed. Chapter 3 constructs complete carbon budgets for the Hard Hill plots by combing the hydrology data with measurements of gaseous carbon exchange.

- Questions were raised through the field site monitoring as to how fire affects the ecosystem in the short term i.e. what is produced during a fire itself. Chapter 4 and 5 look at laboratory experiments and field examples of biomass survival and char production during fires of differing intensities. Modelling studies draw the data together to assess the importance of fires in carbon accumulation. This can then be tied to field observations from Hard Hill.

- Finally, the project aims to sit theses results in the wider issues surrounding the uplands of the UK. Chapter 6 assess the perceptions of managed burning by stakeholders and the general public.

Chapter 7 draws the results from the thesis together and suggests possible areas for future research. By adopting a multidisciplinary approach to investigating the effects of burning on peatland settings and by combining it with an appreciation of the wider social context, an interdisciplinary understanding of this unique environment can be gained.

Chapter 2:

Hard Hill field experiment – effects of managed burning on water quality and hydrology

2.1 Introduction

The peat deposits of the UK are the largest terrestrial carbon pool and are a significant store of carbon (Cannell et al., 1993). However, there is increasing concern at how these import carbon stores are affected by the effects of climate change e.g. predicted temperature changes, and how management and land use within these areas either mitigates or exaggerates these effects. Climate change is predicted to increase global surface temperatures by up to 4°C by the end of the 21st Century (IPCC, 2007). Increased temperatures are likely to enhance rates of photosynthesis, decomposition and CH_4 emission; however, these effects are expected to be overshadowed by changes in hydrology (Gorham, 1991) and that changes to the hydrological behaviour, specifically water table position, will affect the carbon balance of a peatland (Silvola et al., 1996). Understanding the relationship between water table and carbon is important in order to avoid sites turning from a sink of carbon to a source (Lloyd, 2006).

There are many pathways by which carbon may be lost from a peat soil and one of these routes is via dissolved organic carbon (DOC). There is growing evidence for large increases in the DOC concentration of terrestrial water draining boreal and sub-Boreal peat soils (Monteith et al., 2007). There is a range of possible reasons for these changing concentrations of DOC and water supply companies in particular are interested in better learning the mechanisms whereby DOC arrives in water as DOC discolours the water and is costly to remove especially in regions with peat-covered catchments. Not only does DOC discolour water, leading to low aesthetic quality and costly end-of-pipe removal, it also increases the potential for biological contamination as it consumes the free residual chlorine.

Given the consequences of increased losses of DOC from peat soils for carbon storage and water treatment, is it possible to manage these changes? It is unlikely that any discernable change to climate change can be made in short timescales (<10s of years), and in order to reduce water treatment costs, it would therefore seem prudent to manage these vulnerable environments in order to minimise DOC losses through good land-use practices.

Unlike many boreal and sub-boreal peatlands, the peat soils of the UK uplands are heavily and extensively managed for livestock and recreational shooting. In order to increase productivity, managed burning of vegetation has been a common feature of the UK uplands. Recent reviews of the consequences of heather and grassland burning, including that on peat

(DEFRA, 2005; Tucker, 2003), found there were very few studies that examined the consequences of burning for hydrology, water or soil quality, let alone DOC and the review recorded only one study of interactions with other management (Ball, 1974). Existing studies have tended to focus on infiltration rates (Mallik et al., 1984) or water repellency (Doerr et al., 2006; Mallik and Rahman, 1985). Burning in other settings has been associated with the development of water repellency that limits infiltration (e.g. for Californian wildlands, DeBano 2000). However, Mallik and Rahman (1985) demonstrated that water repellency in regularly burnt peat peaked within the first month after burning then declined to a minimum. Immediately after burning, Mallik et al. (1984) showed that infiltration decreased by up to 74% in burnt compared to unburnt peat but moisture retention increased. These twin phenomena can be explained by the presence of fresh ash particles in the surface layers of the peat. Mallik and Fitzpatrick (1996) used thin section studies to show that porosity increased in recently, intentionally burnt soils but that any difference disappeared within 2-3 years of burning.

As for soil and water quality, Garnett et al. (2000) record a significant reduction in carbon accumulation on plots that had been regularly burnt. Ward et al. (2007) have shown significant increases in gross ecosystem CO_2 fluxes in burned and grazed treatments relative to the control plots where these observed differences are thought to be related to changes in vegetation community structure. Worrall et al. (2007d) have shown significant differences between burning regimes in terms of: depth to the water table; soil water conductivity; pH and dissolved organic carbon – with significant declines in interstitial soil water DOC concentration on burnt areas. This link between DOC and managed burning remains an important area of research in the UK with studies finding both decreases (Worrall et al., 2007d) and increases (Yallop and Clutterbuck, 2009) in DOC with burning. Worrall et al. (2007d) suggested that the observed pattern of soil water composition could be due to structural changes in the soil under different management regimes. In a follow up study, Worrall and Adamson (2008) showed that burning, but not grazing, caused significant changes in soil water composition. Specifically, they found increased interactions between incoming rain water and peat soils that led to loss of interaction with deeper water. However, no evidence was found for structural change in the soils even after long-term (50 years) grazing and burning management.

Previous studies (Ward et al., 2007; Worrall et al., 2007d) have been limited to the effect of burning and grazing at the end of the burn cycle but not the effect of burning itself or the consequence in the early part of the cycle. The work in this chapter considers the end of the burning cycle, the consequences of the burn itself and into the year after the burning.

2.2 Chapter Objectives

The objectives of this chapter are to:

- Examine the effects of managed burning and grazing on DOC concentration;
- Examine the variations in water table and other hydrological parameters in response to burning and grazing;
- Investigate the mechanisms for any changes in water quality and hydrology.

This chapter is formed from three papers that have been published in the *Journal of Hydrology*:

- Clay, G.D., Worrall, F. and Fraser, E.D.G. 2009. Effects of managed burning upon dissolved organic carbon (DOC) in soil water and runoff water following a managed burn of a UK blanket bog. Journal of Hydrology, 367(1-2): 41-51.
- Clay, G.D., Worrall, F., Clark, E and Fraser, E.D.G. 2009. Hydrological responses to managed burning and grazing in an upland blanket bog. Journal of Hydrology, 376(3-4): 486-495
- Clay, G.D., Worrall, F. and Fraser, E.D.G. 2010. Compositional changes in soil water and runoff water following managed burning on a UK upland blanket bog. Journal of Hydrology, 380 (1-2): 135-145

2.3 Materials and Methods

2.3.1 Study site

The site used for the field experiments in this chapter was Moor House National Nature Reserve (NNR) in the North Pennines. The North Pennines are an area of high moorland and broad upland dales at the northern end of the Pennine chain in England. The area is an Area of Outstanding Natural Beauty (AONB) and has recently been award UNESCO European Geopark status. This latter accolade reflects the rich geological heritage of the area that includes Carboniferous successions of limestone, sandstones, shale and coal seams (Johnson and Dunham, 1963).

Moor House reserve was first designated as a National Nature Reserve in 1952 and later approved as a World Biosphere Reserve in 1976. The site covers around 75 km^2 and includes, within its boundaries, Great Dun Fell, High Force Waterfall and a large part of the headwaters of the River Tees catchment. Within the reserve lies Trout Beck, a headwater tributary of the River Tees with the entire catchment lying within the NNR. The Trout Beck catchment lies largely above 500 m O.D (Figure 2.1). 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.

Meteorological measurements began at Moor House in 1930s and continue today through an automatic weather station set up 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),(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).

The vegetation is dominated by *Eriophorum sp*. (cotton grass), *Calluna vulgaris* (heather) and *Sphagnum sp.* (moss). The catchment is grazed by sheep at a density of between 0.6 – 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 catchment area has not been burnt since 1954 (Garnett et al., 2000)

In 1954, an experiment was set up within the Trout Beck catchment at Moor House Nature Reserve to examine the ecological effects of traditional heather burning (National grid ref. NY 756326 - Figure 2.1).

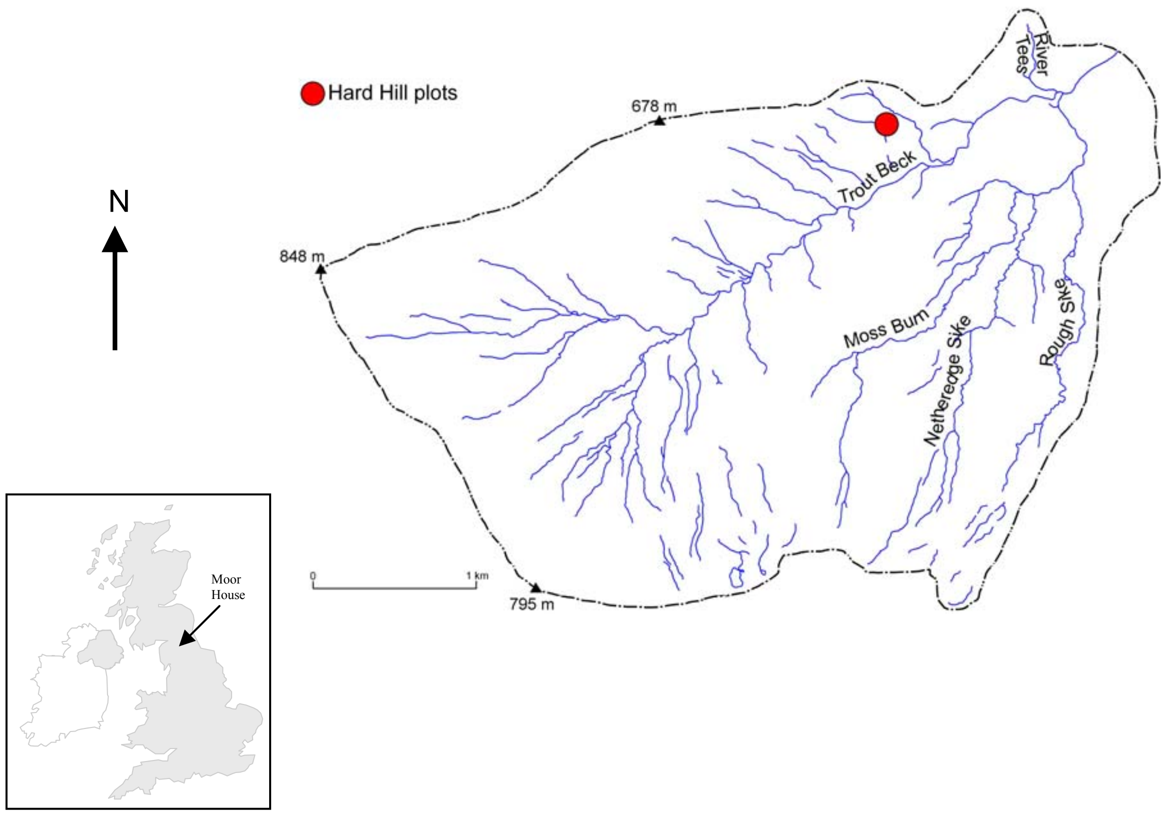

Figure 2.1. Location map of Hard Hill plots.

As part of the design, grazing was also included. Four blocks were set up, with each block sub divided 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 assigned a burning regime – no further burning, burnt every 10 years or burnt every 20 years (Figure 2.2).

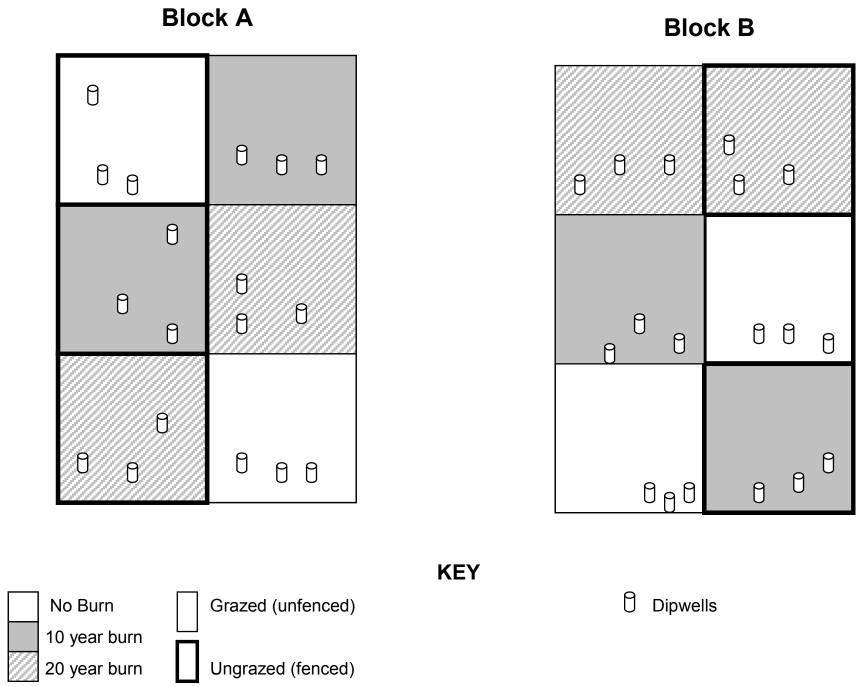

Figure 2.2. Plan of the Hard Hill plots

The 10-year cycle plots were burnt on 6th February 2007 by staff trained and experienced in heather burning. The conditions of the day were conducive to a cool burn (SEERAD, 2001a) – clear day, low moisture on ground except frost, light northerly winds down the slope. The plots were prepared by starting a small width (<1m), well controlled back burn to act as a fire break. The managed burns started off as cool light burns but as the vegetation dried out, the burns got a little hotter but nothing that could be described as a "hot" burn. The pre-burn stand height was low so this may also have helped the cool nature of the fire.

2.3.2 Monitoring regime

This study focused on two of the four experimental blocks meaning that all management combinations were examined in duplicate. Soil water was accessed via a series of dipwells. Soil water sampling started on 6th April 2005 and initially included no burning and 20-year rotations plots for grazed and ungrazed plots. This dipwell network was later extended to include the 10-year burning rotation plots on 1st June 2005. In each plot, three dipwells were placed to at least 90 cm depth. Depth to water table and soil water was measured at least once a month until the managed burn of the 10 year plots on 6th February 2007. The plastic field kit was removed from these plots on the day of the burn and returned to the same positions immediately afterwards. The monitoring continued at least monthly from then until January 29th 2008; therefore, the study considered 33 months of data with at least one year of sampling before and after a burn; in total there were 59 sampling visits to the site though not all visits sampled soil water.

One question raised by Worrall et al. (2007d) was the relationship between observed soil water concentrations of DOC and the DOC concentration that would occur in runoff. As a result, crest-fall runoff traps (Holden, 2000) were installed in late October 2006 in order to intercept surface flow from the plots. These 20 cm pipes were sunk into the peat surface with seals at both ends but with holes at the peat surface to allow in any surface runoff, holes were aligned with and perpendicular to the local slope. As with the dipwells, the runoff traps were placed in triplicate in each of the experimental plots being considered. Traps were inspected at least once a month until January 2008 and, if water was present, it was sampled. The samples were then analysed using the same techniques as for the soil water samples collected from the dipwells. The runoff record is more intermittent than soil water as runoff frequency varied at each runoff trap. From October 2006 to January 2008, the runoff traps were inspected 19 times; during this period some traps had 18 samples compared to one with only 6 samples for the same period. The reason for this diversity of possible sampling frequency may be due to the siting of the traps in areas of low runoff or due to the differing runoff proportions and flow pathways of differently managed plots.

In addition to the analysis of water table depth and runoff occurrence, the hydraulic conductivity of the soil at each dipwell location in each plot was measured. In total, six site visits were made in the period from September 2006 to November 2006. To calculate the hydraulic conductivity of the peat, dipwell slug tests were carried out using a similar methodology to that outlined in Baird et al., (2004). The theory behind dipwell slug tests was noted by Hvorslev (1951) and is detailed in Baird et al. (2004). It was not always possible to measure the hydraulic conductivity of each dipwell on each of the six surveys and so, in total, 149 hydraulic conductivity measurements were made.

Water samples from the dipwells were analysed for pH, Conductivity, absorbance at 400 nm (Abs_{400}) and DOC concentration. The pH and conductivity were measured using electrode methods (pH meter, HI-9025; Conductivity meter, HI-9033, Hanna instruments). Absorbance was measured at 400 nm for a basic colour reading (Thurman, 1985). DOC concentrations were measured colorimetrically using the method of Bartlett and Ross (1988). By measuring both absorbance at 400 nm and DOC, specific absorbance can be evaluated and thus the nature of the DOC can be tested. Specific absorbance can be used to look at the relative proportions of coloured humic substances and uncoloured non-humic substances. This can be used as an indication of microbial activity within the peat (Wallage et al., 2006). It is often assumed that absorbance is a good proxy for DOC and a calibration curve between the two is all that is needed. However, Watts et al. (2001) have shown that DOC/absorbance relationships are site-specific and also show seasonal variation. Worrall et al. (2007d) have shown for this site that there is no clear relationship between DOC and absorbance.

Water samples were analysed for major cations and anions: aluminium (Al), iron (Fe), calcium (Ca), magnesium (Mg), potassium (K), sodium (Na), silicon (Si), fluoride (F^-), chloride (Cl^-), bromide (Br^-), nitrate (NO_3^-) phosphate (PO_4^{3-}) and sulphate (SO_4^{2-}). The cations were analysed by inductively coupled plasma optical emission spectroscopy (Perkin Elmer Optima 330 RL

ICP-OES). Analysis was conducted on filtered samples (<0.45μm, cellulose acetate syringe filters). Mixed standards (including blanks) were run prior to the analysis and the 50 and 25 mg l^{-1} standards were reanalysed as samples approximately every 20 samples as a manual check for drift; all standards were reanalysed at the end of each run. Two wavelengths were collected for each element except K, and all calibration curves used for data processing had $r^2 > 0.99$ for all elements. Instrumental drift was corrected during data post processing using the internal standard method. Yttrium (Y) was selected for the internal standard as it was not found at detectable levels in any samples. All standards and samples were Y-spiked. Optical sensor output counts for each element are converts into milligram per litre concentrations by comparing counts for Y between samples and standards.

The anions were analysed by ion chromatography (Metrohm, Compact IC 761). Analysis was again conducted on filtered samples (<0.45μm). Mixed standard (including blanks) were run prior to the analysis and the 50 and 25 mg l^{-1} standards were reanalysed as samples approximately every 20 samples as a manual check for drift; all standards were reanalysed at the end of each run.

2.3.3 Statistical Methodology

2.3.3.1 ANOVA

Statistical analysis was undertaken on the raw data and normalised data. The data were normalised to minimise the effect of differences due to different sampling days. The normalisation was performed by considering the grazed and unburnt plots as the control. On each sampling day, the average of all the measured variables for the two grazed, unburnt plots was calculated. This value was taken as the value that would be normal for this particular catchment and was used to normalise all other measurements on that day.

The sampling survey design represents a factorial approach to the problem of understanding the consequences of heather burning and grazing (Winer, 1971). This study can be considered initially as a three-factor experiment: time, burning regime, and grazing, where the time factor is the difference between each day of monitoring and henceforward referred to as day of sampling. Secondly, the burning regime has three levels: no burning; 10 years and 20 years. Thirdly, the grazing has only two levels: grazed and ungrazed. Wherever possible the depth to the water table, pH and conductivity were used as covariates within the analysis. The design was sufficient that interactions between factors could be considered. The statistical significance of the independent factors was determined using a general linear modelling approach based on an analysis of variance using the commercially available MINITAB v13 software package. The magnitude of the effects of each significant factor and interaction were calculated. Post-hoc testing of the results is made for pairwise comparisons between factor levels using the Tukey test in order to assess where significant differences lie between factor levels. There are several problems associated with using the ANOVA approach. Firstly, the Levene test was used to assess homogeneity of variance with respect to the three factors in ANOVA; if this test failed, then data were log-transformed. Secondly, in order to avoid type I errors all probability values are given even if significance is assessed at the 95% level. Thirdly, statistical power (probability of a type II error at a given level of significance) was calculated to estimate each of these effects using the method of (Winer, 1971) and the non-centrality factor (Koele, 1982).

Testing of the data was done in stages on both soil water and runoff water quality:

1. All the ‘pre-burn’ data were analysed. This test examines the end of a burn cycle and is essentially an extended dataset to that reported in Worrall et al. (2007d). This is an improvement for the rigour of the results as the pre-burn data now cover 33 sampling days across more than one entire year compared to 16 sampling days in the previous study.
2. The effects before and after the burn were investigated on the 10-year plots for those treatments. The ANOVA approach used in this

case was modified as burn regime could no longer be a factor. The modified model included a before and after burning factor (PrePost) with two levels; pre-burn and post-burn. As a complete year was sampled before and after burning, the month of the year could be considered as a factor with 12 levels (Jan = 1, Feb = 2 etc).

3. The total dataset was analysed. The data span nearly three years so any seasonal effects can be investigated with month as a factor in the analysis. The previous study only looked at the effects during the summer of 2005. However, analysis of this dataset is limited as the 10-year plots were burnt part way through the study and there were no 10-year controls left unburnt at that point.

In addition, in the DOC dataset, runoff water compositions were compared to soil water compositions. This was done by combining the two datasets and applying ANOVA with factors of day of sampling, burn, grazing and type of water sampled, where this latter factor had two levels, soil water and runoff water.

Hydraulic Conductivity

Hydraulic conductivity was analysed as a three-factor ANOVA; however, it was not sampled after the controlled burning so is only indicative of conditions at the end of a 10-year burn cycle. It was not considered appropriate to include any covariates in the analysis of water table results as it was decided that normalisation and the choice of factors was sufficient. However, for the ANOVA of the hydraulic conductivity data, the initial water table depth, H_0 (Hvorslev, 1951), was included as a covariate.

2.3.3.2 Runoff Occurrence – χ^2 test

Due to the intermittent nature and the non-uniform spatial distribution of production, runoff frequency was assessed using a χ^2 test. A runoff event was defined as any time a runoff trap had a measurable amount of sample present. The total number of events per plot was divided by the total possible number of events to give a runoff proportion, e.g. if three out of the six traps were full, a runoff proportion of 0.5 was recorded. The data were analysed using the different factors and their combinations: by burning (no burn, 10 year and 20 year), by grazing (no grazing, grazing) and by management (all burning and grazing combinations). To calculate the test statistic, the method outlined in Fleiss (1981) for the comparison of *m* proportions from several independent samples was used. Using this method, each sample e.g. burn regime, is characterised by the presence or absence of a characteristic, in this case the presence of runoff. The test statistic is derived from difference between each sample proportion and the overall proportion in the whole dataset. A more complete working of the method is given in Appendix 1. To investigate any significant difference in the runoff proportions, the data were partitioned into two groups and post-hoc testing was carried out to investigate if any significant differences occurred between groups and also within groups (Fleiss, 1981)

Analysis was undertaken on data from the pre-burn and post-burn period as well as the total dataset. Due to the limited number of sampling days of the runoff prior to the burn in February 2007, conclusions from the pre-burn analysis must be treated with caution.

2.3.3.3 Runoff Occurrence - Binary Logistic Regression

To investigate the differences in runoff mechanisms between management treatments, an event analysis of the data was conducted using binary logistic regression (Worrall et al., 2007e). Binary logistic regression converts a binary observation, in this case the presence or absence of runoff water in a trap, to continuous variables, such as total rainfall in the preceding period. This means that logistic regression can provide a model to predict the probability of a runoff event given values of X (antecedent rainfall conditions) where the logistic regression equation has the form:

$$\ln\left(\frac{\theta}{1-\theta}\right) = \beta_0 + \beta_1 X + \ldots \qquad \text{(Eq. 2.1)}$$

where θ is the probability of a runoff event.

The study hypothesizes that the occurrence of runoff on peat will be related to rainfall in the preceding sample period and that the threshold for runoff may change or that the critical component of the rainfall responsible for runoff may change between management treatments. Rainfall events between sampling dates were identified and characterised by duration, intensity and total rainfall in each event. Rainfall events were defined as a period with a rainfall intensity greater than 1mm hr^{-1} with periods of no rain in the hour preceding or following it. To compare rainfall events with different conditions a dimensionless term for each rainfall event was calculated, DI/T (Heppell et al., 2002)

$$DI / T = \frac{Duration \times Intensity}{\text{Total Rainfall}} \quad \text{(Eq. 2.2)}$$

In each period between runoff sampling dates, the 'runoff window', the following characteristics were used as predictors in the logistic regression:

- Maximum Duration Event, hr
- Maximum Intensity Event, mm hr^{-1}
- Maximum Rainfall Event, mm
- Total Rainfall in period, mm
- Total number of events in period
- Maximum DI/T event
- Average DI/T over period

All terms were initially included in the analysis and insignificant terms were removed until only significant variables ($p < 0.05$) remained. The data from the pre-burn period were not included in the logistic regression so that the analysis did not include the unburnt portion of the 10-year record. This allowed for comparison between plots without bias from the pre burn period. The data were considered as a whole, and also split into the different burning regimes (no burn, 10 year, and 20 year) in order to see if there were different conditions that controlled runoff on these plots.

2.3.3.4 Principal Component Analysis

Principal component analyses (PCAs) were performed on the data from this study. The data were considered in a number of ways. In the first case, pre-burn data for all the measured anions and cations from the Hard Hill plots were analysed in raw and sea salt corrected forms (Krauskopf, 1982); and then also with the inclusion of observations of pH, conductivity and DOC concentration. In the second case, pre-burn and post-burn data from the 10 year plots were combined with Environmental Change Network (ECN) precipitation data to investigate changing flowpaths over a managed burn.

For the purposes of end-member analysis, precipitation data from ECN were combined with the data from Hard Hill. All precipitation samples were analysed for conductivity, pH, alkalinity, Ca, Mg, K, Na, Fe, Al, Cl, NO_3, PO_4,

SO_4, DOC and total nitrogen. The methods of analysis are detailed in Sykes and Lane (1996).

The inclusion of ECN data means that the following species could be included in a combined analysis of raw data: Al, Ca, Fe, K, Mg, Na, Cl, NO_3, PO_4, and SO_4. Even though all the anion and cation species were analysed on a milligram per litre scale, the PCAs were performed using the correlation matrix in order to ensure that any differences in scale did not distort the result. The number of components to retain was based on the rule to include all those with an eigenvalue >1.

2.4 Results

The results are discussed in the following order – DOC data, hydrology and finally chemistry data

2.4.1 DOC Results

2.4.1.1 Soil Water DOC

Pre-burn data

The depth to water table, pH and conductivity were not found to be a significant covariate for any of the DOC-related parameters (Abs_{400}, DOC and specific absorbance). For Abs_{400} there is a significant difference between day of sampling and burn regime. However, no significant differences between grazing treatments were found (Table 2.1).

	Abs_{400}			DOC			Specific Absorbance		
Factor	df	p	ω^2	df	p	ω^2	df	p	ω^2
Day	23	**0.000**	0.218	19	**0.000**	0.137	19	**0.000**	0.059
Burn	2	**0.000**	0.047	2	0.226	0.001	2	0.265	0.001
Grazing	1	0.569	0.000	1	0.959	0.000	1	0.375	0.000
Day*Burn	46	**0.000**	0.075	38	0.285	0.007	38	0.664	0.000
Day*Grazing	23	0.230	0.004	19	0.749	0.000	19	0.750	0.000
Burn*Grazing	2	0.056	0.003	2	0.763	0.000	2	0.319	0.000
Error	690		0.653	495		0.855	495		0.939

Table 2.1. The ANOVA of the pre-burn soil water data. Values of $p<$ 0.05 are highlighted and ω^2 = proportion of variance explained.

The post-hoc testing shows that it is the 20-year burn treatment that is significantly different from the other burn treatments and that no other significant differences exist at the 95% level. Here, the 20-year plots have the lowest absorbance values – an average 7% lower than control plots. No significant difference is observed between grazing or burning treatments for either DOC or specific absorbance values where only the day of sampling is a significant factor. For absorbance a significant interaction was found between day of sampling and burning regime which implies potential seasonal changes between treatments.

Soil water quality before and after managed burn

Month of sampling was a significant ($p<0.05$) factor for Abs_{400}, DOC and specific absorbance (Table 2.2).

	Abs_{400}			DOC			Specific Absorbance		
Factor	df	p	ω^2	df	p	ω^2	df	p	ω^2
Grazing	1	**0.016**	0.008	1	0.300	0.000	1	0.440	0.000
PrePost	1	0.675	0.000	1	0.832	0.000	1	0.052	0.006
Month	11	**0.000**	0.219	11	**0.000**	0.060	11	**0.000**	0.059
Grazing*PrePost	1	0.713	0.000	1	0.063	0.006	1	0.125	0.003
Grazing*Month	11	0.480	0.000	11	0.560	0.000	11	0.291	0.005
PrePost*Month	11	**0.000**	0.073	11	**0.000**	0.054	11	**0.002**	0.044
Error	419		0.700	348		0.880	348		0.883

Table 2.2. The ANOVA of the pre-/post-burn soil water data. Values of $p< 0.05$ are highlighted and ω^2 = proportion of variance explained.

The difference between months explained 6% of the variance shown in DOC and specific absorbance but 22% of the variance in Abs_{400}. Grazing was only found be significant on the normalised absorbance data but it explains less than 1% of the variance in the data. The PrePost factor was only found to be significant on the un-normalised data for Abs_{400} and specific absorbance but upon normalisation it is no longer significant. Concentrations of DOC showed an average rise of approximately 5% following the burn but this difference was not found to be significant (Table 2.2)

The only significant interaction found was that between the PrePost factor and the month factor and this interaction was found to be significant across all parameters, i.e. the seasonal cycle after the burn was significantly different from that before the burn. This increase is influenced by a peak just after the burn and can be seen most clearly on a time series plot of specific absorbance (Figure 2.3).

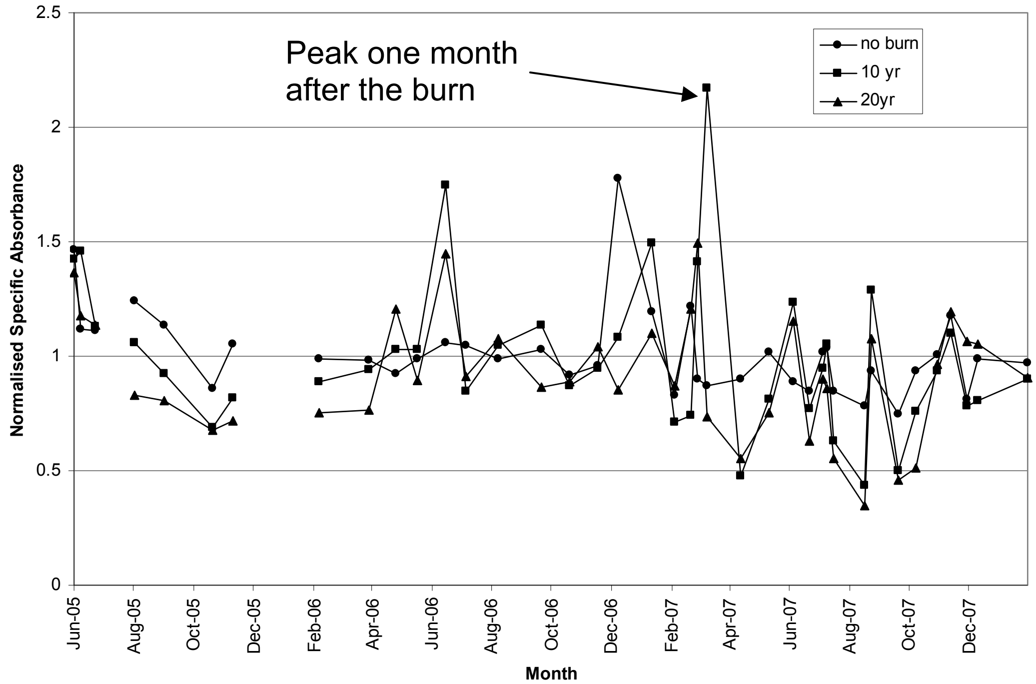

Figure 2.3. Specific absorbance record of soil water at Hard Hill. Labelled peak is 4 weeks after the burn

In the weeks after the burn, normalised specific absorbance of the 10-year plot peaks at above twice the control plots. Absorbance at 400nm also shows a similarly timed peak after burning; however, this is not a unique event as similarly sized peaks also occur before burning in this record. The DOC record does not show a distinct peak on the recently burnt plots (10 year) in the months following the burn.

All soil water data

Seasonality trends have been noted in DOC records from peat catchments (Naden and McDonald, 1989) with labile carbon produced during the summer months being flushed out during the autumn period (Worrall et al., 2002).

Once the data had been normalised, water table was not a significant covariate for any of the parameters. Month was a significant factor explaining up to 23% of the variation in the data (Table 2.3).

	Abs_{400}			DOC			Specific Absorbance		
Factor	df	p	ω^2	df	p	ω^2	df	p	ω^2
Month	11	**0.000**	0.125	11	**0.000**	0.040	11	**0.000**	0.030
Burn	2	**0.000**	0.017	2	**0.000**	0.012	2	**0.018**	0.005
Grazing	1	0.506	0.000	1	**0.010**	0.004	1	0.448	0.000
Month*Burn	22	**0.000**	0.048	22	0.145	0.006	22	0.193	0.005
Month*Grazing	11	**0.043**	0.005	11	0.223	0.003	11	0.150	0.004
Burn*Grazing	2	0.058	0.002	2	0.540	0.000	2	0.537	0.000
Error	1345		0.803	1125		0.935	1124		0.957

Table 2.3. The ANOVA of the seasonal data for soil water. Values of $p < 0.05$ are highlighted and ω^2 = proportion of variance explained.

DOC data showed an elevation in values during the summer with the highest value occurring in September. Specific absorbance and Abs_{400} also showed distinct seasonal trends with higher values during the autumn. Although the peaks occur more or less at the same time (September) Abs_{400} values appear to lag behind DOC values by a couple of months – the highest Abs_{400} values occur when DOC values are starting the fall during the autumn and winter months.

Unlike the pre-burn period, where burn treatment was only significant for Abs_{400} (Table 2.1), Month is also significant for DOC and specific absorbance. This is true for both raw and normalised data. This is most likely due to inclusion of further data. Upon normalisation, the proportion of variance explained increases for both Abs_{400} and DOC though burn treatment only explains around 1% of the variation.

Post hoc comparisons show the highest values for Abs_{400} on 10-year plots though this effect may be due in part to the inclusion of the post-burn data in this analysis. Burning was also a significant factor for DOC unlike the pre-burn period alone. Finally, a significant difference lay between 20-year treatment and no burn for specific absorbance (Table 2.3). Burning is now significant across more factors. This could be due to the extended dataset or the grouping of day of sampling into months. Additionally the inclusion of burnt plots is likely to have influenced these results.

Grazing was also a significant factor for DOC concentration though this explains less than 1% of the variation in the data and is not significant once the data have been normalised.

Only Abs_{400} values show any significant interaction. Here, interactions between month and burn and between month and grazing are statistically significant ($p<0.05$). The former interaction displays a clear divide in the overall pattern whereby values are clustered together for no-burn plots but display a clear split once any burning treatment has occurred. In the 10-year and 20-year plots, those months early in the year (January-April) have higher normalised Abs_{400} values than those later months. The latter interaction also shows a similar split, this time on the application of grazing. Those plots that have been grazed split in a similar fashion in that those months early in the year (January-June) have higher Abs_{400} values than later months. These effects coincide with when the sheep are on the reserve. With sheep on the reserve, often during the summer months, those plots that are not enclosed experience defoliation and removal of vegetation. Sheep will also preferentially eat grasses and younger heather rather than the unpalatable 50-year heather found on the unburnt control plots

How the sheep affect the Abs_{400} values is not clear. One possible mechanism is through limiting the vegetation growth by defoliation and trampling by the sheep. Evapotranspiration is reduced and, consequently, the water table is not drawndown as much. This leads to shallower water tables. With shallower water tables, DOC production is limited and Abs_{400}, an indicator of DOC, is lower. This vegetation removal effect has been suggested for these plots on long-term scales (Worrall et al., 2007d) though whether the water table responds as quickly as month-long timescales is unclear at present.

2.4.1.2 Runoff Water DOC

Pre-burn runoff water data

The only major differences that can be assessed from Figure 2.4 are differences in absorbance values; absorbance displays higher values on 10-year rotations and also higher on ungrazed sites. DOC and specific absorbance show no obvious differences upon a visual inspection.

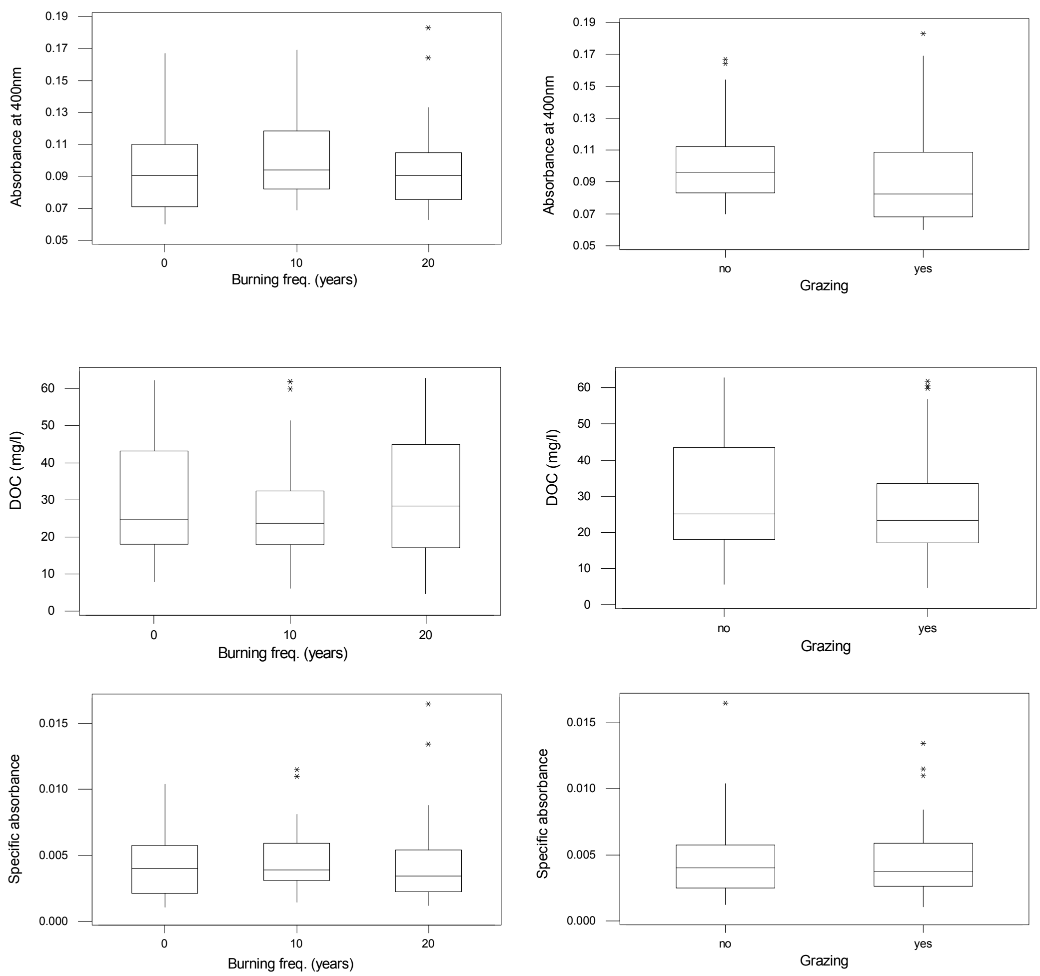

Figure 2.4. A box-and-whisker plot of carbon parameters of runoff water quality relative to burning and grazing treatments

To investigate changes following the burn, it is necessary to first look at the data from before the burn. Runoff traps were installed in October 2006 and the managed burn occurred in February 2007 leaving only a few months in which to collect data. When this is compared to almost two years pre-burn for soil water, the relatively small number of observations should be considered with care when interpreting the results.

The day of sampling was a significant factor for all parameters (Table 2.4). Burning and grazing were not significant factors for any of the parameters except for Abs_{400}. Here burning and grazing explained 3% and 6% of the variation in the data respectively. Post-hoc comparisons showed that a significant difference lay between no burn and 10-year rotations but no other differences were found between the other burn treatments. The effect of this difference is that Abs_{400} values are higher on 10-year plots. The only significant interaction was between burning and grazing and only for Abs_{400} (Table 2.4).

		Abs$_{400}$			DOC			Specific Absorbance	
Factor	df	p	ω^2	df	p	ω^2	df	p	ω^2
Day	4	**0.000**	0.169	4	**0.001**	0.165	4	**0.001**	0.147
Burn	2	**0.042**	0.031	2	0.727	0.000	2	0.430	0.000
Grazing	1	**0.002**	0.063	1	0.405	0.000	1	0.513	0.000
Day*Burn	8	0.296	0.012	8	0.814	0.000	8	0.265	0.021
Day*Grazing	4	0.486	0.000	4	0.992	0.000	4	0.493	0.000
Burn*Grazing	2	**0.001**	0.093	2	0.117	0.021	2	0.856	0.000
Error	75		0.633	71		0.813	71		0.832

Table 2.4. The ANOVA of the pre-burn runoff data. Values of $p< 0.05$ are highlighted and ω^2 = proportion of variance explained.

Runoff water quality before and after a managed burn

Once the data had been normalised, month of the year is a significant factor for all the parameters (Table 2.5) and in each case it is the most important factor explaining up to 30% of the variation in the normalised data. The PrePost comparison was found to be significant for Abs$_{400}$ and DOC with both the absorbance and DOC decreasing in the runoff after the burn. The un-normalised data showed an increase in DOC upon burning but during the post-burn period the other treatments also experienced a similar increase so the relative trend of DOC on the 10-year treatment was downwards – there was a 35% decrease in DOC observed and a 14% decrease in Abs$_{400}$ values.

		Abs$_{400}$			DOC			Specific Absorbance	
Factor	df	p	ω^2	df	p	ω^2	df	p	ω^2
Grazing	1	0.998	0.000	1	0.443	0.000	1	0.777	0.000
PrePost	1	**0.023**	0.049	1	**0.002**	0.077	1	0.618	0.000
Month	3	**0.001**	0.179	3	**0.000**	0.302	3	**0.002**	0.183
Grazing*PrePost	1	0.557	0.000	1	0.175	0.008	1	0.842	0.000
Grazing*Month	3	0.805	0.000	3	0.394	0.000	3	0.814	0.000
PrePost*Month	3	0.096	0.041	3	**0.010**	0.080	3	0.720	0.000
Error	54		0.731	51		0.533	51		0.817

Table 2.5. The ANOVA of the pre-/post-burn runoff water data of 10 year plots on runoff water. Values of $p< 0.05$ are highlighted and ω^2 = proportion of variance explained.

The only significant interaction was between the PrePost and Month factors and only for the DOC concentrations and explained 8% of the variance. As with the soil water concentrations, this could suggest a change in seasonal cycle after burning; however, it is more likely a temporary spike following the burning. In both soil water and runoff, this peak in absorbance (both Abs$_{400}$ and specific absorbance) and DOC occurs not immediately after the burning but between three and seven weeks later (Figure 2.5).

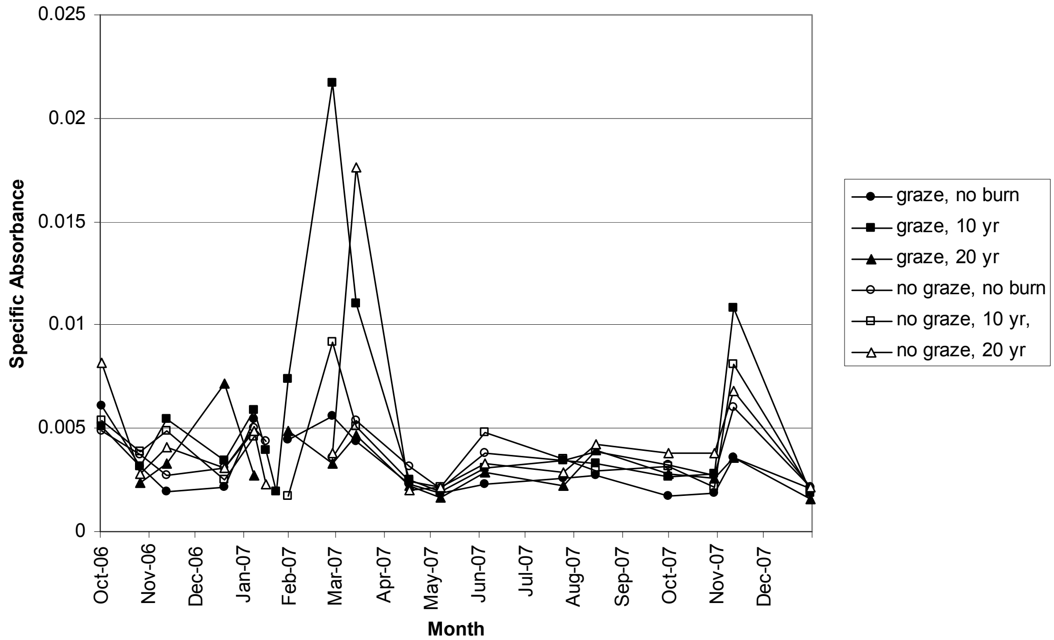

Figure 2.5. Specific absorbance record of runoff water at Hard Hill.

This delayed response to burning may be due to the presence of snow during the early months on 2007. Following burning, the reduction in vegetation cover would allow for a greater interaction between rainwater and soil leading to any burning-related DOC peaks to be seen. However, with snow lying on the ground this interaction would be reduced only returning to normal once the snow had melted.

2.4.1.3 Soil water and runoff water comparisons

The DOC-related parameters (Absorbance at 400nm, DOC and specific absorbance) of the soil water appear, on visual examination, to display similar ranges and average values between treatments; however, there are some parts of the data that are worth noting. Average absorbance values are lower on plots that have been burnt every twenty years (Figure 2.6) and absorbance values show a greater range in those plots that have never been burnt. There is also a larger range of DOC values on unburnt plots.

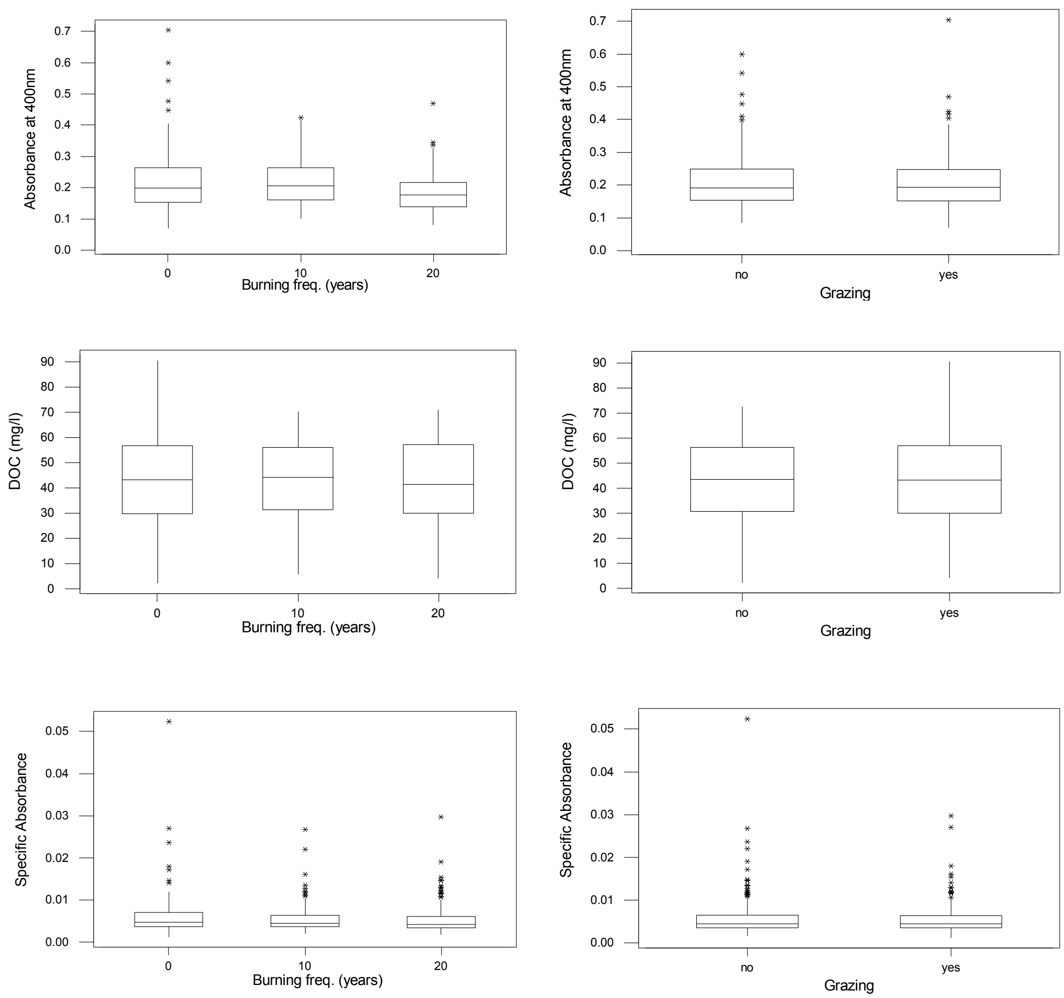

Figure 2.6. A box-and-whisker plot of water colour parameters of soil water quality relative to burning and grazing treatments

By comparing Figure 2.4 (runoff water) with Figure 2.6 (soil water), differences between the two types of water can be seen. Runoff water has lower values for the three carbon parameters e.g. average runoff DOC concentration is approximately 25 mgC l^{-1} whereas soil water DOC concentration is approximately 45 mgC l^{-1}.

In order to investigate any statistical differences between soil water and runoff, the two datasets were combined and analysed using ANOVA. In addition to sampling day, burn and grazing as factors, type of sample was included where this latter factor had two levels: soil water and runoff water Only those months that had soil water and a runoff sample were analysed so that the model would be balanced.

Table 2.6 shows the type of water analysed is a significant factor for the three carbon parameters. Three and four-way interactions were not significant and are omitted from the table to aid clarity. Runoff water is significantly 'lighter' than soil water; absorbance values, DOC concentration and specific absorbance are lower in runoff samples than soil water.

Factor	df	Abs_{400}	DOC	Specific Absorbance
Day	11	0.000	0.000	0.000
Grazing	1			
Burn	2	0.000		
Type	1	0.000	0.000	0.000
Day*Grazing	11			
Day*Burn	22			0.003
Day*Type	11			
Grazing*Burn	2			
Grazing*Type	1	0.018		
Burn*Type	2			

Table 2.6. Soil water and runoff comparison. Only those factors and interactions that are significant ($p<0.05$) are shown.

2.4.2 Hydrology Results

2.4.2.1 Water Table

The depth to water table during the study period for all dipwells varied from 0 mm (peat surface) to 671 mm below the peat surface (Figure 2.7).

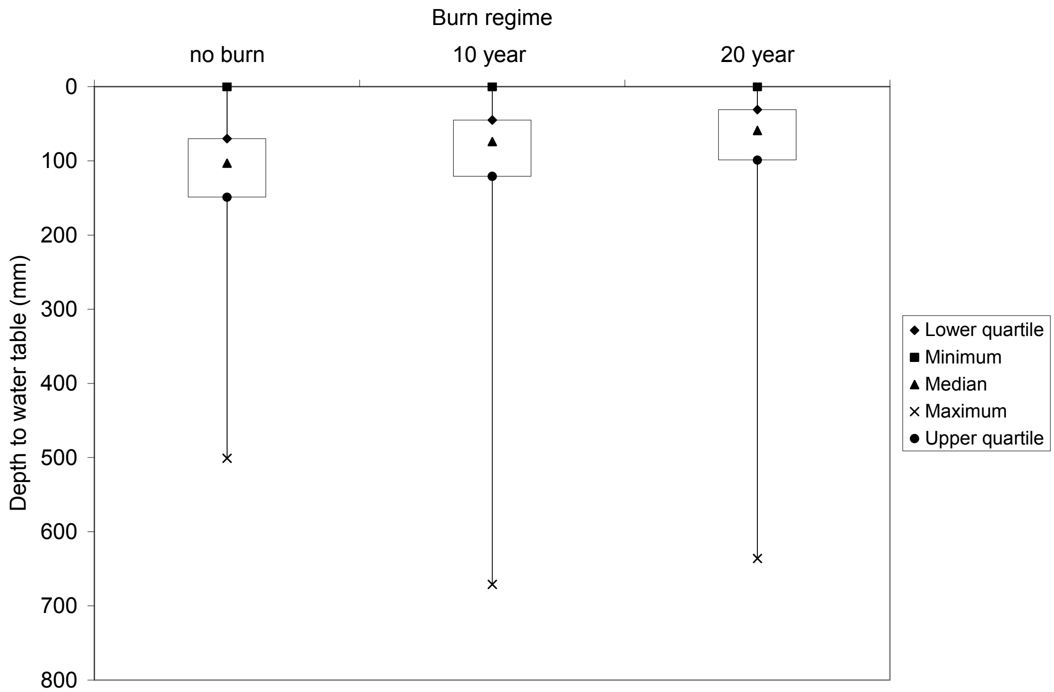

Figure 2.7. Range of depth to water table for each burn regime.

Pre-burn data

There are significant differences between burning and grazing regimes. The day of sampling explains the largest proportion of variance, around 73%, but after normalization it becomes the difference between burning regimes that is the most important factor, explaining 14% of the variation in the data (Table 2.7). Grazing is a significant factor, but it explains less than 1% of the variation within the data.

Raw Data				Normalised Data			
Factor	df	p	ω^2	Factor	df	p	ω^2
Day	32	**0.00**	0.73	Day	32	**0.00**	0.05
Burn	2	**0.00**	0.03	Burn	2	**0.00**	0.14
Grazing	1	**0.00**	0.00	Grazing	1	**0.00**	0.01
Day*Burn	64	0.34	0.00	Day*Burn	64	0.35	0.00
Day*Grazing	32	0.80	0.00	Day*Grazing	32	0.63	0.00
Burn*Grazing	2	**0.00**	0.01	Burn*Grazing	2	**0.00**	0.01
Error	1000		0.23	Error	1000		0.79

Table 2.7. ANOVA Results for water table in the pre-burn period, p = probability that the factor or interaction is zero; and ω^2 = the proportion of the variance explained by the factor or interaction. Values of $p< 0.05$ are highlighted.

Post-hoc testing shows that significant differences exist between all burn cycles for both raw and normalized data. The shallowest water tables can be found on 20-year plots and the deepest water tables can be found on no burn plots. Shallower water tables can be found on grazed compared to ungrazed plots. The shallowest water tables can therefore be found on 20 year plots that have been grazed. The effect size shows that burning on a 10-year cycle will decrease depth to water table by 26% while burning on a 20-year cycle will decrease depth to water table by 35% (the average normalized decrease for both grazed and ungrazed plots) (Table 2.8). Grazing has the effect of decreasing depth to water table by 25%.

	Grazed			Ungrazed		
Burning	0	10	20	0	10	20
Pre Burn	1	0.67	0.59	0.96	0.79	0.71
Post Burn		0.58			0.75	

Table 2.8. The average normalized value for each factor level for water table in the pre-burn and post-burn period, where the average value for a grazed, unburnt plot = 1.00 (Only 10 year plots in post-burn period are shown for comparison).

The interaction between grazing and burning is the only significant interaction in the data, explaining less than 1% of the variation in the data. Water tables are shallower on those plots that have experienced both managed burning and grazing.

Water table before and after managed burn

The 10-year plots allow for direct comparison in conditions before and after managed burning. The month of sampling is a significant factor explaining 20% of the raw data which decreases to 7% upon normalisation (Table 2.9).

Raw Data				Normalised Data			
Factor	df	p	ω^2	Factor	df	p	ω^2
Grazing	1	**0.00**	0.01	Grazing	1	**0.00**	0.04
PrePost	1	**0.00**	0.01	PrePost	1	**0.03**	0.00
Month	11	**0.00**	0.20	Month	11	**0.00**	0.07
Grazing*PrePost	1	0.87	0.00	Grazing*PrePost	1	0.40	0.00
Grazing*Month	11	1.00	0.00	Grazing*Month	11	0.94	0.00
PrePost*Month	11	**0.00**	0.24	PrePost*Month	11	**0.00**	0.04
Error	650		0.55	Error	650		0.84

Table 2.9. ANOVA results for PrePost water table comparison p = probability that the factor or interaction is zero; and ω^2 = the proportion of the variance explained by the factor or interaction. Values of p< 0.05 are highlighted.

Grazing is also a significant factor and explains up to 4% of the variation in the data. The PrePost factor, although significant, explains less than 1% of the variation in the original dataset. After the managed burn, water tables were significantly shallower compared to those before the burn. The effect size data for the 10-year plots shows a decrease in depth to water table in the post burn period of nearly 7% (Table 2.8).

The interaction between month and the PrePost factor is also significant explaining around 24% of the variation in the raw data. When comparing the water table in similar months before and after the burn, the biggest differences occur in summer months with the greatest change occurring in July, when water tables were, on average, 200mm.

All soil water data

Month is a significant factor when analysing the entire dataset (Table 2.10) and water tables were deepest during the summer months. Burning was another significant factor in the entire dataset; it explains 4% of the variation in the data but upon normalisation this increases to 15% and becomes the most important factor. Grazing is a significant factor only in the normalised data and explains less than 1% of the variation. The interaction between burning and grazing is the only significant interaction in the analysis but explains approximately 1% of the variation in the original data. This interaction shows that the shallowest water tables across the entire dataset can be found on sites burnt every 20 years and grazed. The deepest water tables can be found on unburnt, grazed sites.

	Raw Data				Normalised Data		
Factor	df	p	ω^2	Factor	df	p	ω^2
Month	11	**0.00**	0.19	Month	11	**0.00**	0.01
Burn	2	**0.00**	0.04	Burn	2	**0.00**	0.15
Grazing	1	0.20	0.00	Grazing	1	**0.00**	0.00
Month*Burn	22	0.98	0.00	Month*Burn	22	0.11	0.00
Month*Grazing	11	1.00	0.00	Month*Grazing	11	0.92	0.00
Burn*Grazing	2	**0.00**	0.01	Burn*Grazing	2	**0.00**	0.02
Error	2002		0.77	Error	2002		0.82

Table 2.10. ANOVA results for seasonal water table comparison p = probability that the factor or interaction is zero; and ω^2 = the proportion of the variance explained by the factor or interaction. Values of $p < 0.05$ are highlighted.

2.4.2.2 Hydraulic conductivity

The hydraulic conductivity of the peat varied from 1.3×10^{-8} - 1.4×10^{-3} cm s^{-1}. The data from this study are consistent with other hydraulic conductivities found in the literature (Figure 2.8).

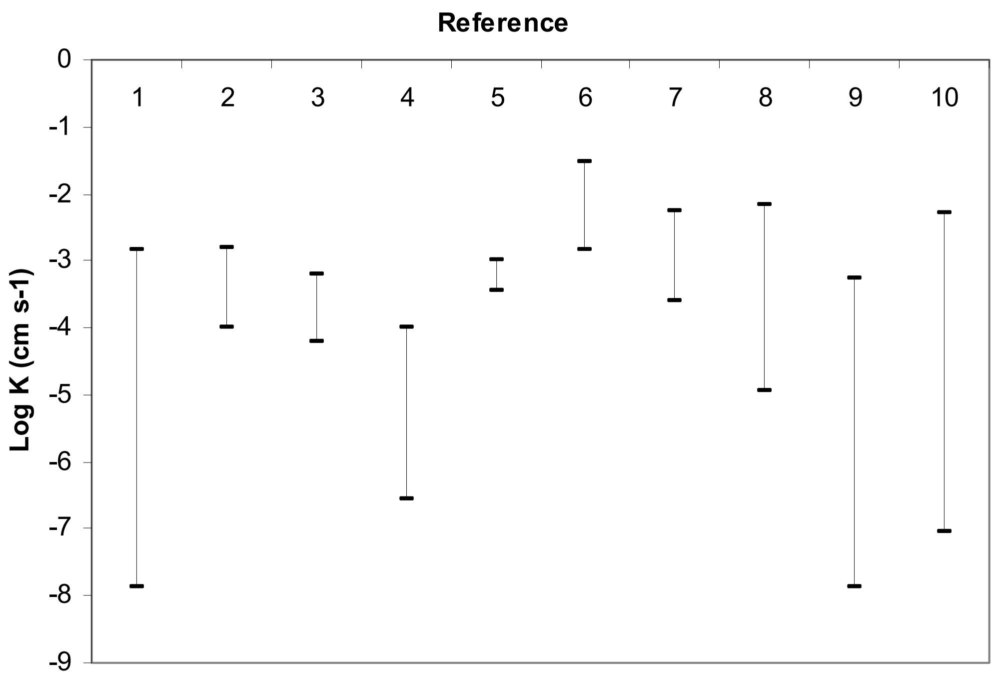

Figure 2.8. Ranges of published hydraulic conductivities, including data from this study: 1, this study. 2, Surridge et al., 2005. 3, Baird and Gaffney, 2004. 4, Almendinger and Leete, 1998. 5, Koerselman, 1989. 6, Baird et al., 2004 . 7, Chason and Siegel, 1986. 8, Rosa and Larocque, 2008. 9, Holden et al., 2001. 10, Rycroft et al., 1975.

Figure 2.9 shows the variation in hydraulic conductivity measurements with initial water table depth.

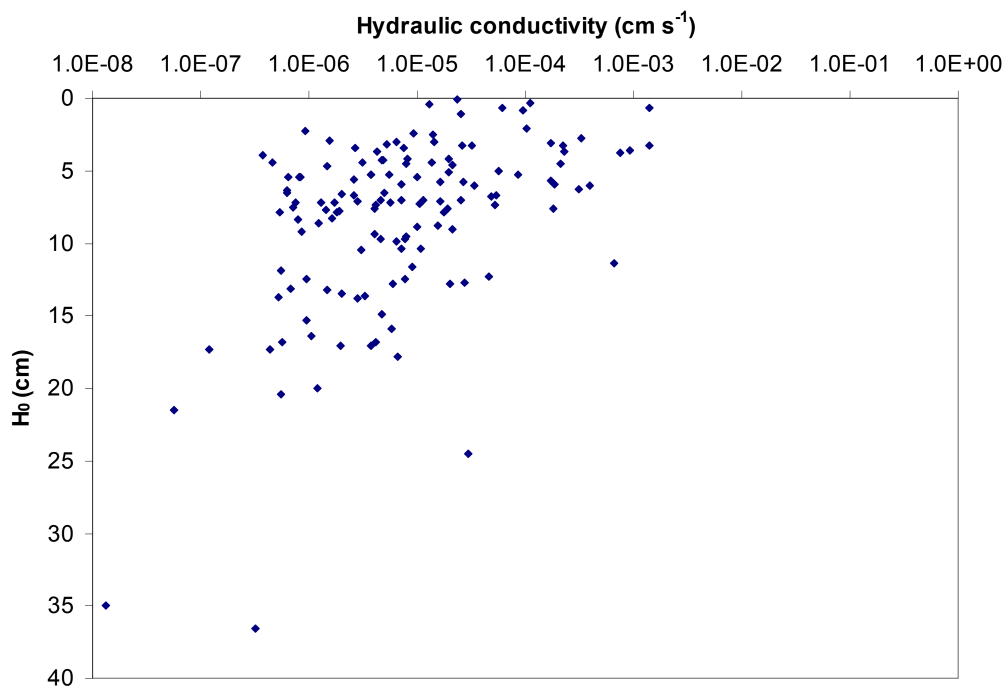

Figure 2.9. Plot of H_0 vs. hydraulic conductivity, K

There are four different set of results from the ANOVA for the hydraulic conductivity (Table 2.11).

(a) Raw data (K)				(b) Log K			
Factor	df	p	ω^2	Factor	df	p	ω^2
H0	1	0.01	0.20	H0	1	0.00	0.19
Day	5	0.00	0.67	Day	5	0.01	0.06
Burn	2	0.59	0.00	Burn	2	0.10	0.01
Grazing	1	0.62	0.00	Grazing	1	0.95	0.00
Day*Burn	10	0.95	0.00	Day*Burn	10	0.71	0.00
Day*Grazing	5	0.85	0.00	Day*Grazing	5	0.96	0.00
Burn*Grazing	2	0.22	0.13	Burn*Grazing	2	0.00	0.09
Error	107		0.00	Error	107		0.65
(c) Raw data (K)				(d) Log K			
Factor	df	p	ω^2	Factor	df	p	ω^2
Log H0	1	0.00	0.29	Log H0	1	0.00	0.14
Day	5	0.00	0.71	Day	5	0.01	0.06
Burn	2	0.32	0.00	Burn	2	0.04	0.02
Grazing	1	0.94	0.00	Grazing	1	0.42	0.00
Day*Burn	10	0.96	0.00	Day*Burn	10	0.86	0.00
Day*Grazing	5	0.85	0.00	Day*Grazing	5	0.96	0.00
Burn*Grazing	2	0.50	0.00	Burn*Grazing	2	0.00	0.06
Error	107		0.00	Error	107		0.71

Table 2.11. ANOVA of hydraulic conductivity data p = probability that the factor or interaction is zero; and ω^2 = the proportion of the variance explained by the factor or interaction.

The data were log transformed and analysed alongside the unaltered data. At the same, time initial water table (H_0) was used as a covariate again in unaltered and log transformed form. Initial water table was a significant covariate ($p<0.05$) in both raw and logged form and explained between 14% and 28% of the variation (Table 2.11, a and c). Day of sampling was another significant factor explaining between 66% and 70% of the variation in the non-transformed data. Upon log transformation, day of sampling explained around 6% of the variation. The only other significant factor was burning. This occurred in the log-transformed hydraulic conductivity data with log-transformed initial water table as a covariate (Table 2.11d). Here, it explains around 2% of the variation in the data. Looking at the main effects plots, 20-year plots have lower conductivities than both 10-year plots and unburnt plots though the post-hoc testing does not show a significant difference between factor levels.

The only significant interaction is between burning and grazing in the log transformed data (Table 2.11b and Table 2.11d). Here, it explains between 6% and 9% of the variation in the data. This interaction explains more of the variation than burning alone, suggesting it is the additive effects of burning and grazing that affects hydraulic conductivity. The burnt plots (10 years and 20 years) showed an increase in hydraulic conductivity when combined with grazing. However, this effect is reversed on the unburnt plots. When combined with grazing, unburnt plots showed a decreased in hydraulic conductivity.

2.4.2.3 Runoff generation

χ^2 tests

The χ^2 tests for the total dataset showed that there are significant differences in the proportions when the data are analysed by burning regime. However, no significant differences were observed for grazing or management (Table 2.12). The post-hoc tests show that the significant difference in burning regimes lies at the 10-year burn regime i.e. higher runoff proportions on most frequently burnt plots. This higher value could either be due to pre-burn or post-burn vegetations conditions so, in order to investigate this result further, the data were split into pre and post-burn datasets.

	Total dataset	Pre-burn dataset	Post-burn dataset
Burning regime (no burn, 10 year, 20 year)	***7.87***	0.15	***9.58***
Grazing regime (no grazing, grazing)	0.01	0.41	0.27
Management regime (all combinations)	1.49	0.20	1.36

Table 2.12. χ^2 test statistics for runoff proportions. Significant results are highlighted.

Runoff proportions in the pre-burn period did not significantly differ for the management combinations or by considering burning and grazing independently. This result will have to be treated with caution, however, due to the limited amount of data prior to the burn in February 2007.

In the post-burn period, grazing and management showed no significant differences in proportions; however, like the total dataset, there were significant differences when considering burning regime (Table 2.12). The post-hoc testing shows a significant difference occurs between no-burn plots and 10-year plots. There is no significant difference between 10-year and 20-year plots with the 20-year plots lying in an intermediately position between no-burn and 10-year regimes. This pattern of significance suggests that the similar result in the total dataset is dominated by post-burn changes on the 10-year plots. Figure 2.10 shows that although runoff coefficients were higher in the post-burn period than in the pre-burn period, higher rates of runoff on 10-year plots compared to no-burn plots can be seen. The runoff proportions on the 10-year plots after the burn are greater than 80% when compared to 65% and 70% for no-burn and 20-year plots respectively.

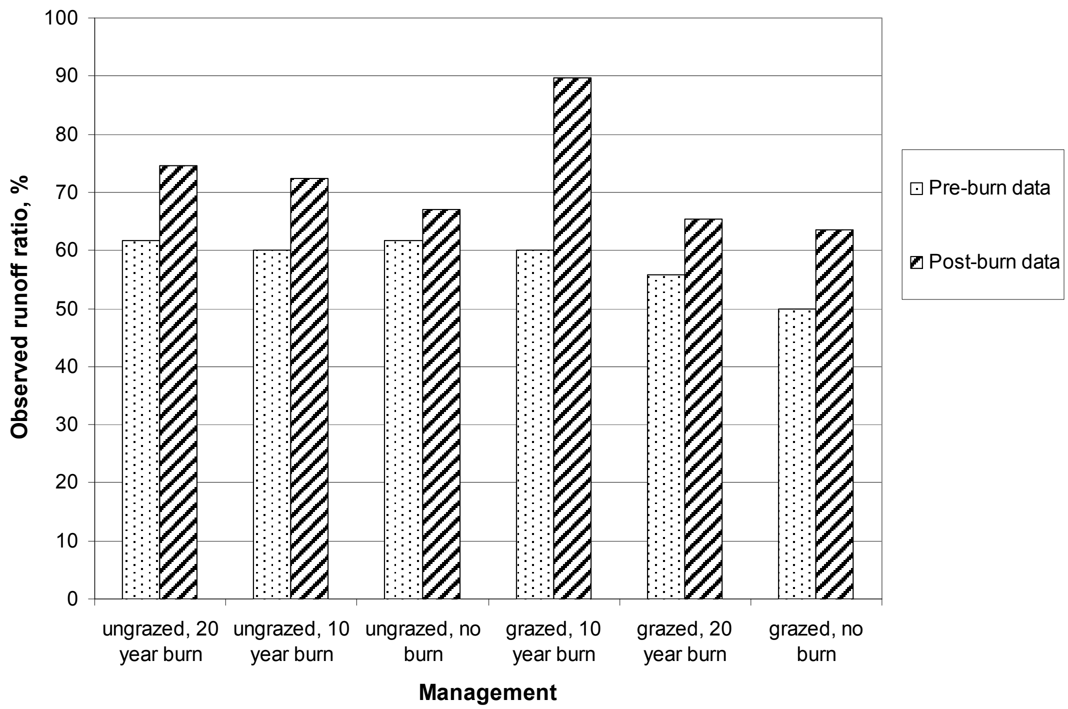

Figure 2.10. Plot of runoff return ratios for pre-burn period and post-burn period relative to management.

Rainfall Event Analysis

Analysis of the rainfall events during the post-burn period showed that Maximum DI/T was the only significant predictor that could be used to model runoff events. The logistic regression equations for the total dataset and for each burning regime can be given by:

$$\ln\left(\frac{\theta}{1-\theta}\right) = 0.8482Max(DI/T)_{Total} - 1.8001 \qquad \text{(Eq. 2.3)}$$

$$\ln\left(\frac{\theta}{1-\theta}\right) = 0.8271Max(DI/T)_{noburn} - 2.1566 \qquad \text{(Eq. 2.4)}$$

$$\ln\left(\frac{\theta}{1-\theta}\right) = 0.9913Max(DI/T)_{10year} - 1.4638 \qquad \text{(Eq. 2.5)}$$

$$\ln\left(\frac{\theta}{1-\theta}\right) = 0.8541Max(DI/T)_{20year} - 1.9943 \qquad \text{(Eq. 2.6)}$$

where θ is the probability of a runoff event and Max(DI/T) is the maximum DI/T value in the preceding period and after the last observation. This parameter, used in other studies (Heppell et al., 2002; Worrall et al., 2002), allows the rainfall event to be classified better than if each component were considered separately and it can also be used to assess the peakedness of a rainfall event.

By rearranging equations 2.3-2.6, the threshold value of DI/T can be calculated whereby a runoff event has a greater than 50% probability of occurring on the given management type. The threshold value is lowest for

10-year plots followed by 20-year plots and finally no-burn plots (Total dataset – 2.12; no burn, - 2.61; 10 year – 1.48; 20 year – 2.34).

2.4.3 Chemistry Results

An examination of the concentration data from soil water and runoff water from the experimental plots show differences in concentrations between species and also between water types. Soil water anions are generally dominated by Cl, which is higher than SO_4 and PO_4, with minor amounts of NO_3, Br and F (Table 1). These observations are in line with other data from this site (Worrall and Adamson, 2008). Cations are dominated by Ca and Na with lower concentrations of the other elements. Runoff water shows high values of Cl and SO_4 with minor contributions from PO_4, NO_3, Br and F. Calcium shows the highest concentration of the measured cations, which is higher than K and Na, with minor amounts of the other cations (Table 2.13).

By carrying out ANOVA with month, burning regime, grazing regime and also type of water, comparisons between the two water types can be made. Table 2.14 shows that the type of water analysed is a significant factor in all but three of the cations and anions. Post-hoc testing shows that runoff water has generally lower concentrations of cations and anions; only Ca and SO_4 show higher concentrations in runoff water.

Soil Water	Burning Rotation			**Runoff Water**	Burning Rotation		
Species	0	10	20	Species	0	10	20
Al	0.06	0.09	0.08	Al	0.01	0.00	0.00
	(0.04 - 0.11)	(0.06 - 0.14)	(0.04 - 0.14)		(0.00 -0.02)	(0.00 - 0.02)	(0.00 - 0.02)
Ca	1.69	1.07	0.84	Ca	6.28	6.14	4.23
	(0.90 - 4.07)	(0.61 - 1.67)	(0.54 - 1.36)		(2.65 - 9.70)	(2.03 - 9.59)	(1.31 - 8.79)
Fe	0.46	0.45	0.39	Fe	0.00	0.00	0.00
	(0.29 - 0.72)	(0.28 - 0.64)	(0.25 - 0.63)		(0.00 - 0.06)	(0.00 - 0.07)	(0.00 - 0.06)
K	0.56	0.55	0.46	K	0.80	0.80	0.62
	(0.30 - 2.03)	(0.22 - 1.14)	(0.21 - 0.99)		(0.34 - 2.03)	(0.30 - 2.29)	(0.34 - 1.48)
Mg	0.54	0.49	0.43	Mg	0.42	0.40	0.39
	(0.32 - 1.14)	(0.30 - 0.83)	(0.28 - 0.75)		(0.29 - 0.68)	(0.24 - 0.67)	(0.26 - 0.66)
Na	4.24	3.36	3.42	Na	2.87	3.11	3.18
	(2.75 - 7.88)	(2.62 - 6.62)	(2.42 - 6.28)		(2.06 - 4.04)	(2.33 - 4.13)	(2.24 - 4.29)
Si	0.06	0.05	0.04	Si	0.00	0.00	0.00
	(0.01 - 0.16)	(0.01 - 0.14)	(0.00 - 0.13)		(0.00 - 0.01)	(0.00 - 0.00)	(0.00 - 0.00)
Fl	0.00	0.00	0.00	Fl	0.00	0.00	0.00
	(0.00 - 0.04)	(0.000 - 0.10)	(0.00 - 0.07)		(0.00 - 0.00)	(0.00 - 0.00)	(0.00 - 0.00)
Cl	3.73	3.60	3.61	Cl	3.97	3.77	4.41
	(2.57 - 4.97)	(2.64 - 4.56)	(2.73 - 5.00)		(2.31 - 5.31)	(2.57 - 5.07)	(2.65 - 5.52)
Br	0.00	0.00	0.00	Br	0.00	0.00	0.00
	(0.00 - 0.00)	(0.00 - 0000)	(0.00 - 0.00)		(0.00 - 0.00)	(0.00 - 0.00)	(0.00 - 0.00)
NO_3^-	0.00	0.00	0.00	NO_3	0.00	0.00	0.00
	(0.00 - 0.00)	(0.00 - 0.02)	(0.00 - 0.04)		(0.00 - 0.00)	(0.00 - 0.24)	(0.00 - 0.00)
PO_4	0.37	0.00	0.00	PO_4	0.00	0.00	0.00
	(0.00 - 1.54)	(0.00 - 0.30)	(0.00 - 0.21)		(0.00 - 0.27)	(0.00 - 0.34)	(0.00 - 0.29)
SO_4	0.48	0.50	0.50	SO_4	1.16	1.05	1.02
	(0.06 - 0.70)	(0.15 - 0.79)	(0.19 - 0.68)		(0.86 - 1.81)	(0.74 - 1.54)	(0.75 - 1.61)

Table 2.13. Median values (inter-quartile range) for concentration, mg l^{-1}, for each species in the total dataset.

	df	Al	Ca	Fe	K	Mg	Na	Si	F	Cl	Br	NO_3	PO_4	SO_4
Month	11	0.000	0.002	0.000			0.000	0.009	0.019	0.000				0.000
Burn	1	0.039	0.000										0.002	
Grazing	2		0.021											0.005
Type	1	0.000	0.000	0.000		0.038	0.000	0.000	0.003		0.003		0.007	0.000
Month*Burn	22		0.002											0.004
Month*Grazing	11								0.006					
Month*Type	11	0.000	0.000	0.000			0.006	0.043		0.004				0.000
Burn*Grazing	2			0.000										
Burn*Type	2												0.040	
Grazing*Type	1													0.001

Table 2.14. ANOVA for each chemical species. df = degrees of freedom. Only those factors or interaction that are significant ($p<0.05$) are shown.

2.4.3.1 Analysis of variance

Soil Water

Pre-burn soil water data

At the end of a burning cycle, all but two of the chemical species that were analysed, showed significant differences between months, with or without sea salt correction (Table 2.15).

Significant differences occur between burning treatments for Ca, Mg, Na, PO_4, and Fe. No significant differences were found for grazing in either the original data or the sea salt corrected form. Significant interactions between burning and grazing were found for Fe, Mg, and Cl. Post-hoc testing, of those species showing significant effects of burning treatments, shows a decline of all species with burning. All have lowest concentrations of the species on the 20-year plots with Ca and PO_4 show significant decrease with burning regardless of the rotation while Mg and Na on 20-year plots are significantly different from the unburnt plots but not from 10-year plots. Finally, Fe shows a significant decrease only on 20-year plots. When pH and conductivity are included as covariates, only Ca and Fe remain significant and K is now significantly different between burning regimes. Both Ca and Fe both show declines on burning with the lowest values on 20-year plots. Potassium, on the other hand, shows an increase on burning with both burning treatments showing significant increases relative to unburnt sites.

	Without covariates					With covariates						
Species	Month	Grazing	Burning	G*B	log WT	Month	Grazing	Burning	G*B	log WT	pH	Conductivity
df	11	1	2	2	1	11	1	2	2	1	1	1
Al	0.000					0.000			0.035			0.001
Ca	0.000		0.000			0.000		0.033				0.000
Fe	0.000		0.003	0.003		0.000		0.004	0.003			
K	0.010							0.005			0.000	0.000
Mg	0.000		0.021	0.014		0.000			0.013			0.000
Na	0.000		0.001			0.000			0.037			0.000
Si	0.000				0.000	0.000				0.000		0.000
Fl	0.000					0.000						0.015
Cl	0.000			0.029		0.000			0.028		0.000	
Br	0.000					0.000						0.000
NO												
PO_4			0.000			0.002						0.000
SO_4	0.000					0.000					0.049	
	SEA SALT CORRECTED					SEA SALT CORRECTED						
Species	Month	Grazing	Burning	G*B	log WT	Month	Grazing	Burning	G*B	log WT	pH	Conductivity
Al	0.000					0.000			0.035			0.001
Ca	0.000		0.000			0.000		0.033				0.000
Fe	0.000		0.003	0.003		0.000		0.004	0.003			
K	0.010					0.000		0.005			0.000	
Mg	0.000		0.023	0.015		0.000			0.011	0.030		0.000
Na	0.000		0.003	0.019	0.023	0.000			0.005	0.027	0.025	0.000
Si	0.000				0.000	0.000				0.000		0.000
Fl	0.000					0.000						0.015
Cl	na	na	na	na	na	na	na	na	na	na	na	na
Br	0.000					0.000						0.000
NO_3												
PO_4			0.000			0.002						0.000
SO_4	0.000					0.000						

Table 2.15. ANOVA for Pre-Burn Soil Water Quality, with and without covariates, with and without sea-salt correction. G*B = grazing/burning interaction; df = degrees of freedom

Soil water quality before and after managed burn

When considering the managed burn of the 10-year plots, six species show significant differences after the managed burn (Table 2.16). Increases are observed in Al, Fe, Na and decreases are seen in Ca, Cl and Br. With the addition of pH and conductivity as covariates, Fe, Na and K still show significant increases following the burn.

a) Soil Water					b) Runoff water			
Species	Month	Grazing	PrePost	log Water Table	Species	Month	Grazing	PrePost
df	11	1	1	1	df	11	1	1
Al		0.0000	0.13		Al	0.005		0.000
Ca	0.006	0.018	0.043		Ca			0.000
Fe	0.000		0.016		Fe	0.006		0.001
K					K	0.029		
Mg	0.000	0.006			Mg	0.000		
Na	0.000		0.000		Na	0.000		
Si	0.039				Si	0.000		
F	0.002		0.03		F			
Cl	0.000				Cl	0.009		
Br	0.032		0.045		Br			
NO_3					NO_3			0.028
PO_4	0.028				PO_4	0.048		
SO_4	0.000				SO_4	0.049		

SEA SALT CORRECTED					SEA SALT CORRECTED			
Species	Month	Grazing	PrePost	log Water Table	Species	Month	Grazing	PrePost
Al	0.000		0.013		Al	0.006		0.000
Ca	0.008	0.018	0.043		Ca			0.000
Fe	0.000		0.016		Fe	0.006		0.001
K					K	0.023		
Mg	0.000	0.01			Mg	0.000		
Na	0.000		0.000		Na	0.000		
Si	0.039				Si	0.000		
Fl	0.002		0.03		Fl			
Cl	na	na	na	na	Cl	na	na	na
Br	0.033		0.045		Br			
NO_3					NO_3			0.028
PO_4	0.028				PO_4	0.048		
SO_4	0.000				SO_4	0.022		

Table 2.16. PrePost comparisons for water quality and runoff water. df = degrees of freedom

Runoff Water

Runoff pre-burn data

Most of the species show significant differences between months in the pre-burn data; however, F, Br and PO_4 show no significant seasonal trends (Table 2.17). Significant differences between burning treatments are observed for Al, Ca and Fe. Both Al and Fe show increases in concentration on the 10-year plots relative to no-burn and 20-year plots whilst Ca shows a significant decrease on 10-year plots relative to the other burning treatments. Unlike the soil water data, which showed no significant differences with grazing, Ca, Cl and SO_4 showed differences with grazing. Post-hoc testing shows Ca and SO_4 concentrations increased with the presence of grazing while Cl showed a decrease upon grazing. With the addition of extra covariates only Fe shows significant differences with burning where the highest values are found on 10-year burning regimes.

	Without covariates					With covariates						
Species	Month	Grazing	Burning	Grazing*Burning	Species	Month	Grazing	Burning	Grazing*Burning	pH	Conductivity
df	11	1	2	2	df	11	1	2	2	1	1
Al	0.000		0.03	0.005	Al	0.000				0.006	0.048
Ca	0.002	0.004	0.018	0.008	Ca	0.003					
Fe	0.000		0.016	0.013	Fe	0.000		0.023		0.000	
K	0.017				K	0.001	0.031				0.006
Mg	0.000				Mg	0.000					0.006
Na	0.000				Na	0.000					0.013
Si	0.017				Si						
F					F						
Cl	0.000	0.018			Cl	0.000	0.007				
Br					Br						
NO_3	0.044				NO_3						
PO_4				0.007	PO_4				0.009		
SO_4		0.04		0.018	SO_4	0.039					0.034
	SEA SALT CORRECTED					SEA SALT CORRECTED					
Species	Month	Grazing	Burning	Grazing*Burning	Species	Month	Grazing	Burning	Grazing*Burning	pH	Conductivity
Al	0.000		0.03	0.005	Al	0.000				0.006	0.048
Ca	0.002	0.003	0.018	0.008	Ca	0.003					
Fe	0.000		0.016	0.013	Fe	0.000		0.023		0.000	
K	0.016				K	0.001	0.039			0.048	0.008
Mg	0.000				Mg	0.000				0.034	0.027
Na	0.000				Na	0.000				0.018	
Si	0.014				Si						
Fl					Fl						
Cl	na	na	na	na	Cl	na	na	na	na		na
Br	0.000	0.018			Br	0.000	0.007				
NO_3	0.045				NO_3						
PO_4				0.007	PO_4				0.008		
SO_4		0.008		0.013	SO_4						

Table 2.17. ANOVA for Pre-Burn Runoff Water Quality, with and without covariates, with and without sea-salt correction. df = degrees of freedom

Runoff water quality before and after managed burn

Table 2.16 shows that Al, Ca, Fe and NO_3 showed significant differences following burning. The largest difference is seen in Ca concentrations where following the burn concentrations increased nearly five fold. Interestingly Al, Fe and NO_3 show significant decreases in the year following the managed burn. With the addition of covariates (pH and conductivity) Al, Mg, Na and NO_3 show significant decreases post-burn whilst Ca shows an increase.

2.4.3.2 Principal component analysis

PCA1 - Raw, Sea-salt corrected and with covariates

Soil Water

The soil water data, whether raw or sea-salt corrected soil water, show similar patterns of loadings across the species in the PCAs (Table 2.18a). The first three components explain 67% and 70% of the raw and sea-salt corrected datasets respectively. The first principal component, PC1, shows high loadings for Ca, K, Mg, Na and PO_4, elements that showed differences between burning regimes in the ANOVA. The second principal component (PC2) shows high loadings for Al and Fe. Finally, the third component (PC3) has high loadings for SO_4 and Cl. With the addition of PO_4 in PC1 and Cl in PC3, these results broadly reflect similar trends shown in Worrall and Adamson (2008). When pH, conductivity and DOC are included, the first principal component has high loading for Ca, K, Mg, Na, PO_4 and conductivity. The second component has high positive loadings for Al and

Fe suggesting a shallower soil water component. The third component has low loading for pH and conductivity and high loadings for Cl and SO_4. A high loading of DOC in the fourth principal component could explain the lack of a high loading for DOC in PC2.

Runoff Water

The runoff water dataset shows different results in the PCAs when compared to soil water data (Table 2.18b). The raw and sea-salt corrected data show similar magnitude effects though the direction of the effects is not always the same. The first three components explained 73% and 76% of the variation in the raw and sea-salt corrected datasets respectively. The first principal component (PC1) is relatively evenly weighted except for NO_3 that is associated with PC3. For PC1, Al and Fe show high positive loadings in comparison to the negative loading for other species. The second principal component shows high loadings for K, Mg, and Na. The third principal component is dominated by a large loading for NO_3; Na and Cl also have high values. Again, further analysis of the data is possible when pH, conductivity and DOC are included. The first component has positive loading for Al, Fe and also DOC The second principal component has strong negative loading for K and Mg and low loadings for pH and conductivity. The strong loading for K and Mg, terrestrial derived species, could indicate a source deeper in the peat profile for this component. The third component has high positive loadings for Na, Cl, NO_3 and a high negative loading for conductivity. The strong sea-salt component (Na and Cl) suggests a rainwater influence to this component.

		a) Soil Water				b) Runoff Water			
	Species	PC1	PC2	PC3	PC4	PC1	PC2	PC3	PC4
a)	Al	0.007	0.591	-0.017		0.394	0.326	-0.052	
	Ca	0.392	-0.260	0.330		-0.390	0.132	-0.311	
	Fe	0.192	0.559	0.106		0.353	0.377	-0.039	
	K	0.404	-0.001	-0.261		-0.277	0.449	-0.143	
	Mg	0.480	0.037	0.178		-0.307	0.493	-0.105	
	Na	0.471	0.221	-0.034		-0.281	0.400	0.325	
	Cl	0.186	-0.168	-0.600		-0.322	-0.173	0.363	
	NO_3	0.040	0.220	0.324		-0.051	0.051	0.741	
	PO_4	0.387	-0.325	0.086		-0.218	-0.260	-0.278	
	SO_4	0.086	0.194	-0.553		-0.405	-0.170	-0.067	
	Variance Explained	0.359	0.546	0.667		0.368	0.613	0.732	
b)	Al	0.046	0.577	-0.040		0.338	0.400	0.107	
	Ca	0.415	-0.317	0.068		-0.487	0.017	0.126	
	Fe	0.236	0.520	0.119		0.301	0.436	0.123	
	K	0.378	-0.040	-0.113		-0.379	0.343	0.116	
	Mg	0.507	0.000	-0.009		-0.383	0.429	0.078	
	Na	0.471	0.237	-0.019		-0.233	0.466	-0.181	
	NO_3	0.056	0.201	0.721		-0.033	0.049	-0.926	
	PO_4	0.375	-0.374	0.005		-0.213	-0.290	0.202	
	SO_4	0.061	0.240	-0.668		-0.409	-0.208	-0.080	
	Variance Explained	0.371	0.583	0.700		0.335	0.639	0.760	
c)	Al	0.041	0.560	0.046	0.175	0.345	-0.330	-0.163	0.016
	Ca	-0.372	-0.200	0.298	0.083	-0.356	-0.163	-0.110	-0.287
	Fe	-0.135	0.520	0.038	-0.301	0.313	-0.376	-0.058	0.161
	K	-0.361	0.096	-0.265	-0.011	-0.253	-0.457	-0.037	0.047
	Mg	-0.417	0.112	0.177	0.050	-0.284	-0.491	0.011	0.032
	Na	-0.389	0.291	-0.032	-0.105	-0.240	-0.379	0.367	0.137
	Cl	-0.129	-0.152	-0.602	-0.179	-0.250	0.182	0.455	0.234
	NO_3	-0.042	0.247	0.304	-0.331	-0.027	-0.031	0.460	-0.335
	PO_4	-0.355	-0.246	0.175	0.202	-0.217	0.247	-0.215	0.356
	SO_4	-0.048	0.191	-0.488	0.337	-0.371	0.154	-0.054	-0.306
	pH	-0.288	-0.033	-0.278	-0.260	-0.355	0.076	-0.154	-0.074
	Conductivity	-0.395	-0.163	0.066	0.204	-0.180	-0.070	-0.567	-0.158
	DOC	0.023	0.247	0.000	0.673	0.217	-0.013	0.092	-0.672
	Variance Explained	0.342	0.501	0.605	0.699	0.332	0.523	0.644	0.722

Table 2.18. The loadings on the principal components for PCA for a) raw data, b) sea-salt corrected data and c) with pH, conductivity and DOC data. Both soil water and runoff water results are shown.

PCA2 – 10 year plots before and after managed burning

When the data from Hard Hill are combined with the ECN rainwater data, a comparison of PC1 versus PC2 shows a clear pattern of behaviour (Figure 2.11).

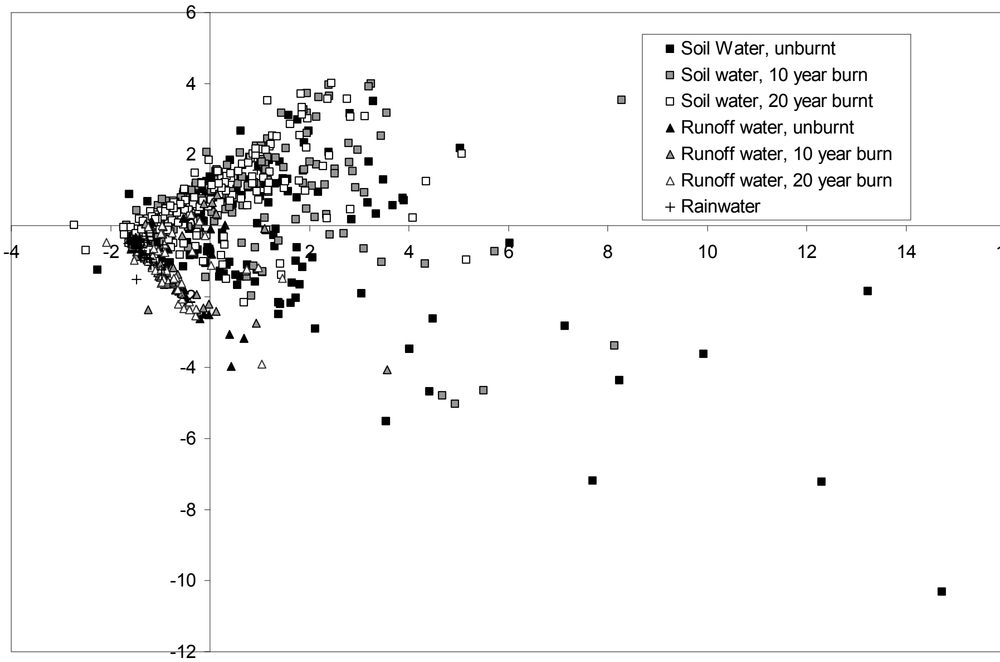

Figure 2.11. Comparison of PC1 and PC2 for sea-salt corrected data from Hard Hill and ECN precipitation data.

The majority of the data are bound by two trends; one formed from rainwater samples and the other soil water samples from 20-year burn plots. Water tables on the 20-year plots are closest to the surface on these sites so this latter trend can be interpreted as a shallower water trend. Runoff water samples occur dominantly, but not exclusively, along the rainwater trend.

Changes in water chemistry following the managed burn are able to help trace water sources contributing to soil water and runoff water. Though there is scatter in the data, soil water following managed burning shows a rotation towards more shallow water dominated trends (Figure 2.12).

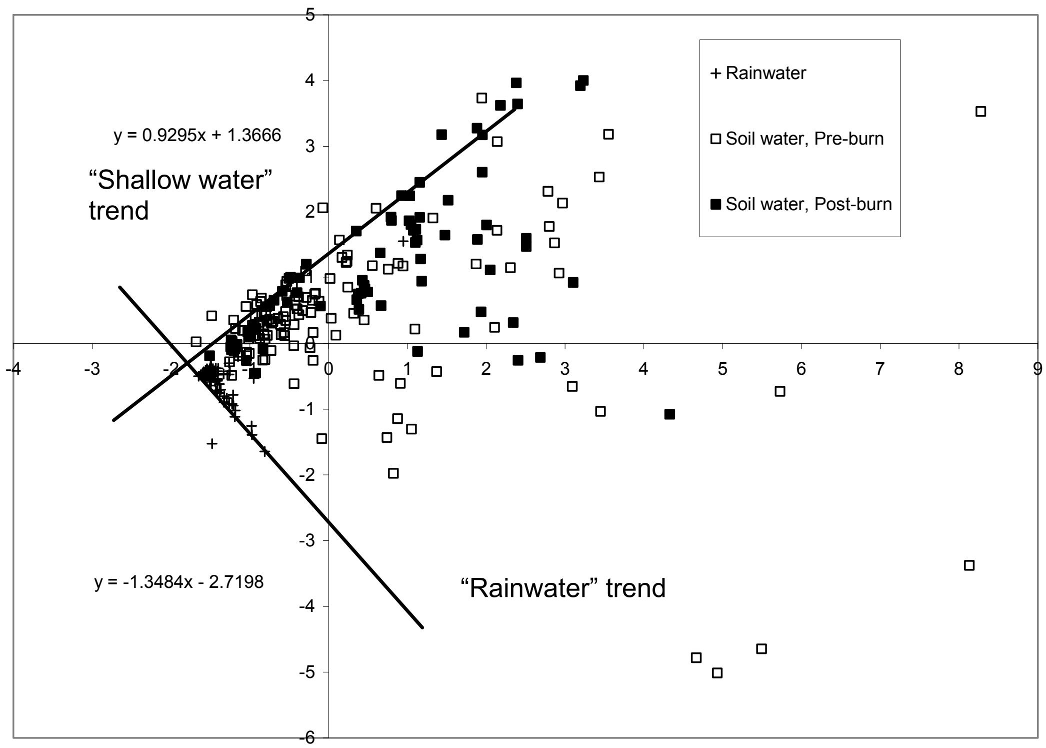

Figure 2.12. Plot of PC1 and PC2 for soil water from 10 year plots, pre-burn and post-burn

Runoff samples generally occur along the rainwater trend (Figure 2.10); however, a proportion of the pre-burn runoff water on the 10-year plots has a component associated the shallow water trend (Figure 2.13). Following the managed burn, runoff water on the 10-year plots is almost exclusively along the rainwater trend.

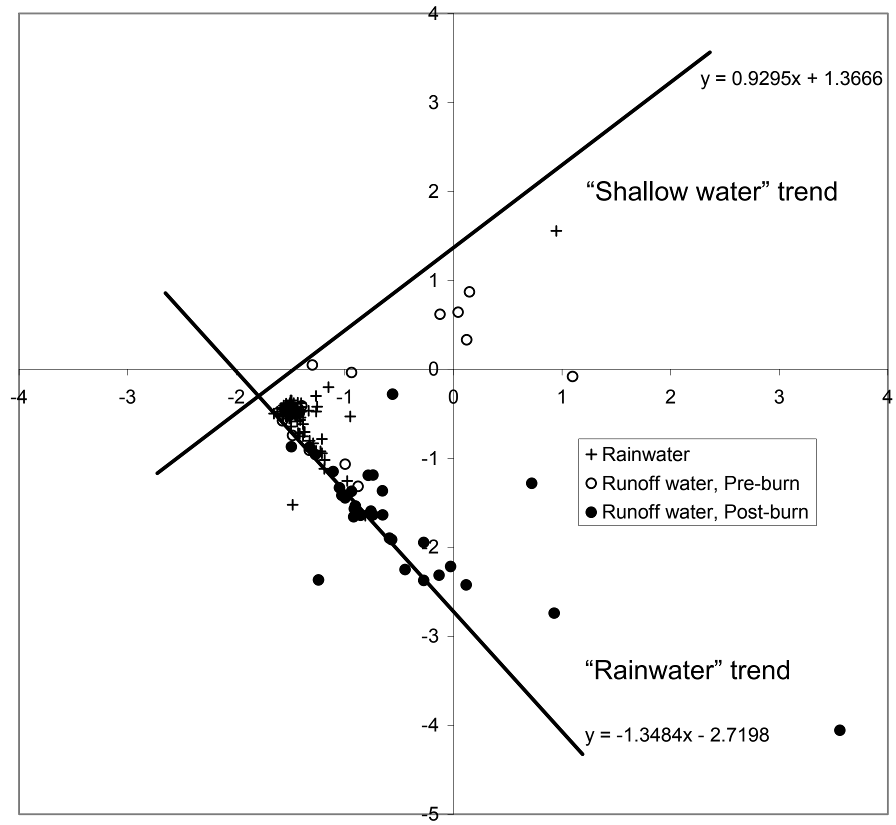

Figure 2.13. Plot of PC1 and PC2 for runoff water from 10 year plots, pre-burn and post-burn

Un-mixing PCA2 trends

The axes of theses trends are almost perpendicular suggesting that the behaviour of shallow soil water is independent from rainwater. By referencing the co-ordinates of the data to these new axes, changes before and after the managed burn can be quantified. The angles between the co-ordinates and the two axes were calculated. In order to identify any significant rotations following burning differences in the angles before and after burning were tested using a t-test. Rain water shows no significant changes before and after managed burning with rainwater samples lying close to the rainwater trend (Table 2.19).

	Average angle to "Rainwater" axis			Average angle to "20 year" axis		
Water Type	Rainwater	Soil Water	Runoff Water	Rainwater	Soil Water	Runoff Water
Pre-burn	21.6 (1.8)	68.0 (1.1)	36.9 (4.1)	72.9 (1.6)	26.4 (1.1)	57.4 (3.8)
Post-burn	23.9 (2.0)	76.2 (1.2)	10.2 (1.7)	71.9 (2.0)	17.4 (1.3)	78.6 (1.4)

Table 2.19. Average angle between co-ordinate and new axes (± standard error)

In the soil water data, a significant ($p<0.05$) change in the angle to soil water axis exists. This rotation towards the soil water axis can be seen by an approximate 8° change in angles between axes. Runoff samples show a significant change in the angle to the rainwater axis. This change can be seen by a rotation towards the rainwater axis with an approximately 20° rotation away from the 20-year axis.

2.5 Discussion

2.5.1 DOC results

Has this study improved our understanding of how different management practices impact on water quality in peat catchments? This study has a number of advantages on the previous work on DOC for this site (Ward et al., 2007; Worrall et al., 2007d). The study: includes more samples; runs over more than one complete seasonal cycle; it sampled both soil and runoff water; and includes both the beginning and the end of a burn cycle.

However, unlike Worrall et al. (2007d), this study could find little or no significant effect of controlled burning upon measured DOC in soil water or runoff water although Abs_{400} values are lower in soil water on 20-year cycles. Why could this be the case? Worrall et al. (2007d) studied only 7 months over one summer whereas this study included 3 summers and the best part of three complete years. Figure 2.14, illustrates seasonal cycles in the Absorbance (400nm) record and also highlights the period of sampling used in Worrall et al. (2007d). Although there are seasonal differences, these may cancel each other out. Indeed, although one might expect differences between burning regimes in terms of DOC concentrations, it might appear that these effects are not significant if a complete burn cycle is considered

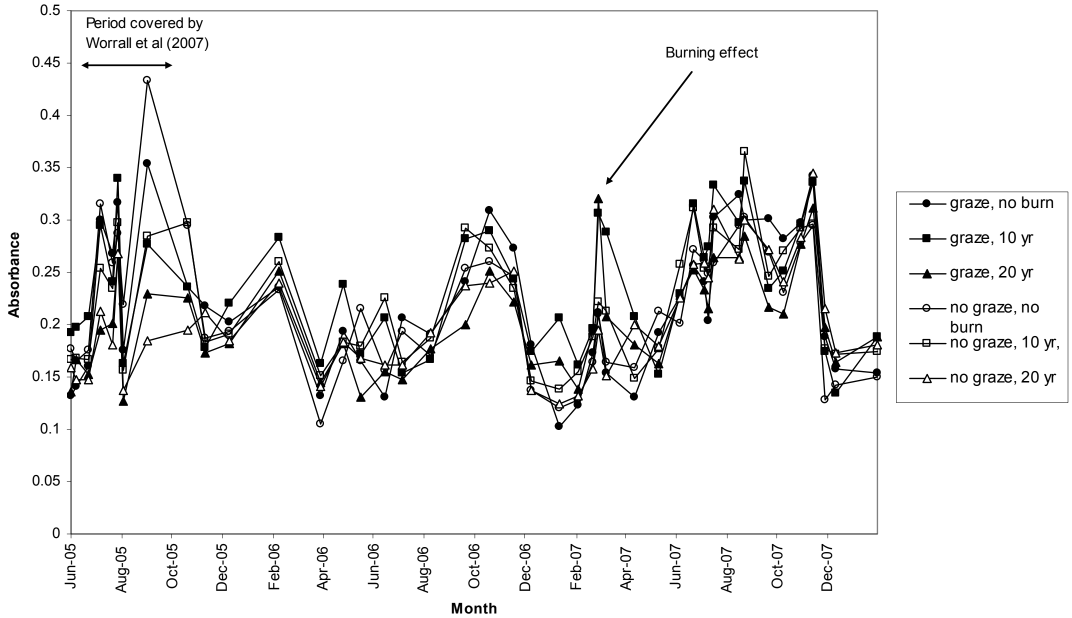

Figure 2.14. Absorbance record (Abs_{400} raw data) of soil water at Hard Hill

Burning does appear to have an effect upon absorbance (Abs_{400}) of both runoff and soil water and also appears to affect runoff water quality even at the end of a burn cycle. The 10-year plots had the highest Abs_{400} values when compared to the other burn treatments even if differences in DOC concentrations are not significant. There are several possible reasons for the elevated colour. Firstly, char produced in the original fire could still be on the site and interacting with the runoff. This is unlikely due to long time period between fire and sampling in which erosion and transportation of easily removable material may have occurred. Erosion in UK peat landscapes is a well documented process (Warburton, 2003; Warburton et al., 2003). Any high carbon burn products are likely to be removed after a period of time; however, it is possible for the re-establishment of vegetation to arrest erosion from these bare soils within a few years (Kinako and Gimingham, 1980) potentially leaving these charred products in the litter layer. However, if char were responsible, then one would expect the effect to be larger after the burn rather than before it. Furthermore, char could also be expected to absorb DOC and it could be that the highly-coloured DOC is less absorptive than other DOC fractions; however, to achieve this differentiation without a significant change in DOC concentration would be impossible. The presence of char could alter soil pH which would in turn change Abs_{400} but pH was found not to be a significant covariate for either Abs_{400} or DOC. Therefore, it can be assumed that differences in Abs_{400} are due to differences in DOC production or site of production.

One effect on production could be changes to the microfauna. It has been observed that prescribed burning increases the numbers of Enchytraeid worms (Mallik and FitzPatrick, 1996) and these worms have been linked with DOC production in the surface layers of peat (Cole et al., 2002). Second, managed burns also make long-term changes in vegetation and this could have several effects. The lack of mature vegetation on burnt sites means that the soil temperature experienced could be quite different; however, once again temperature contrasts due to vegetation would be most extreme immediately after a burn rather than at the end of a 10-year burn cycle. Alternatively, managed burning does reset the vegetation structure and after

10 years *Calluna vulgaris* has yet to dominate and it might be the absence of the type and quantity of litter from mature *Calluna* vegetation that is causing a difference. The variation may not be due to a change in production but a change in site of production or the access of mobile water to that DOC. The formation of water-repellent layers following fires and their relationship with runoff is well documented (DeBano, 2000). Water-repellent layers can potentially alter flow paths across and through the peat and limit the interaction of rain with the soil. This could lead to surface peat rich in ash and char being mobilised. Worrall and Adamson (2008) have shown that for this site at the end of the burn cycle, burning did lead to increased interaction between incoming water and soil, but not with deeper soil water (>1m).

How does burning affect the plots in the year after the managed burn? Along with the long-term effects following fire, there are also the transient effects in the weeks and months following a fire. Both soil water and runoff water experienced peaks in colour (Abs_{400}) and DOC in the weeks following the fire. DOC was slightly elevated in soil water in the year after the fire though this effect was not found to be significant. Runoff experienced a relative decrease though actual values increased in the year following the fire. These peaks only occurred in the weeks following the fire so it is possible that while these peaks are large deviations from the normal, they are relatively short-lived occurrences in the long-term trends and thus no significant differences were observed when comparisons were made between years.

Grazing also had a significant influence upon $Abs_{400}$and its seasonal cycle. During the time sheep were on the reserve (i.e. spring-mid autumn), Abs_{400} values were lower than those taken from the winter months. The depth to water table did not display any significant interaction between grazing and month suggesting that another mechanism is required to explain the lower Abs_{400} values. This effect is surprising considering the historical grazing pressure of the catchment is 0.6 – 1 sheep per hectare and at the Hard Hill plots grazing is estimated to less the 0.1 sheep per hectare (Adamson and Kahl, 2003). Nevertheless, sheep have been observed in small groups in close proximity to the plots on several occasions and evidence in the form of faeces indicates that they are regular visitors to the sites.

This study is by no means a definitive answer to the question as to whether managed burning leads to changes in DOC concentrations. Although this study has shown no significant effect of burning on DOC in soil or runoff water, it has not shown what the effect would be at a catchment scale, i.e. would stream water leaving a catchment under burn management have a lower DOC concentration than one not under burn management? The study has shown that there are significant differences between soil and runoff water and so the question becomes one of how do these flowpaths combine?

2.5.2 Hydrology results

This study has found that for water table depth, there are significant differences between burning treatments at the end of the burning cycle. However, unlike Worrall et al. (2007d), who showed that 10-year, grazed plots had the shallowest water tables, it is the 20-year, grazed plots in this study that have the shallowest water tables. One possible reason why the two studies give different results is that this study examines data from 59 sampling days compared to 16 in Worrall et al. (2007d). This extended dataset, which covers complete seasonal cycles, may capture seasonal variations that Worrall et al. (2007d) could not capture. Worrall et al. (2007d) was conducted during a single summer period and it may be that conditions during that period were such that 10-year plots generated the shallowest water tables.

The hydraulic conductivity of the site was also investigated at the end of the burning cycle. The results from the ANOVA show that variations in hydraulic conductivity could be reflecting the depth in the peat profile at which the measurement was made i.e. H_0 was found to be a significant covariate. Figure 2.9 shows a decrease in hydraulic conductivity with increasing initial water table depth. This variation is likely to be due to the degree of macropores in the peat. The upper layers are mainly composed of litter and partially decomposed vegetation with a larger amount of pores and interconnectivity leading to a relatively high hydraulic conductivity. At greater depths, compaction and humification lead to fewer spaces and consequently a lower hydraulic conductivity. These variations in hydraulic conductivity in the peat profile have been accounted for in the ANOVA by using both H_0 and log H_0 as covariates so any pattern of significance should be reflective of the management rather than variations in H_0.

Burning was found to be a significant factor after log-transformation of the data with the lowest hydraulic conductivities found on the 20-year plots. The occurrence of the lowest hydraulic conductivities on plots with the shallowest water tables raises the question, why? With lower hydraulic conductivities, flow through the peat on 20-year plots is impeded leading to sites where water tables exhibit a buffered response to water table fluctuations. It is, therefore, the low hydraulic conductivity that leads to the shallow water tables found on the 20-year plots. However, what is the mechanism leading to the low hydraulic conductivities on 20-year plots?

One possible explanation for the variations in hydraulic conductivity is the development of shrubby vegetation at the site. *Calluna vulgaris* has been linked to an increase in soil piping in peatlands which consequently increases macropore flow through the peat (Holden, 2005). It is, therefore, possible that, through the development of vegetation, and specifically its root systems, hydraulic conductivities may be affected. The hypothesis is that when *Calluna* reaches its mature phase, the root system will occupy a maximum space. As a consequence there is reduced macropore development, limited by the root system, which leads to lower fluid flow and low hydraulic conductivities. At Moor House National Nature Reserve, the age of the *Calluna* vegetation contributing most to the biomass is around 19 years (Forrest, 1971). This age of 19 years is similar to the 20 year plots where the hydraulic conductivities were lowest and would seem to support the hypothesis that the development of root systems plays a role in hydraulic conductivity. Following the mature phase *Calluna* moves into a degenerate phase after 30 years, where it stops growing and eventually dies (Webb, 1986). At this time, the root system is also likely to go through a similar degenerate phase whereby the roots die back but leave a network of macropores. This network of macropores will lead to increased fluid flow and higher hydraulic conductivities. At the experimental site, the highest hydraulic conductivities were found on the no-burn plots. These plots have not been burnt since 1954 and therefore could have a high degree of interconnectivity in the sub-surface from the development of the mature and degenerate heather on the plots. The 10-year plots have hydraulic conductivities somewhere between the two end members (no burn plots and 20 year plots). The vegetation on these plots is not yet mature so the below ground root system has not yet fully developed to occupy any pre-existing macropore network.

The effects on hydrology in the weeks and months following a managed burn are poorly understood. This study found that in the year following a managed burn, the depth to water tables on the recently burnt plots (10-year plots) reduced by 7%, i.e. water tables became significantly shallower. This is likely to be associated with the removal of vegetation in the burning process. A loss of vegetation would lead to a decrease in evapotranspiration and associated drawdown of the water table. This mechanism has been cited as a possible cause for shallow water tables on regularly burnt sites that have young or immature vegetation (Worrall et al., 2007d).

Runoff generation varied across the study period and also between treatments types. In the pre-burn period, runoff generation was not significantly different between factors (burning, grazing, and management) although the limited number of sampling days may have influenced this analysis. During the post-burn period, the plots that had the highest occurrences of runoff were the 10-year plots i.e. those that had been recently burnt. This higher runoff on burnt plots could be due to a combination of several factors. Firstly, the water table record showed that the water table had risen. Evans et al. (1999) have shown that the rapid generation of near-surface or surface runoff in peat-covered catchments occurs when water tables are close to the surface. Therefore, any further rise in water tables would lead to a greater amount of surface saturation-excess and consequently a greater proportion converted to saturation overland flow (Burt, 1992). Secondly, burning removes the vegetation cover and this loss of vegetation would lead to a greater proportion of rainfall reaching the surface. Soto and Diaz-Fierros (1997) show, in a gorse shrubland in Spain, that throughfall is significantly increased on burnt plots in the 2 years following fire. Thirdly, hydrophobic compounds generated by burning may reduce interaction with the soil leading to a greater degree being partitioned into surface flow (DeBano, 2000). Fourthly, intensification of soil crusting following a fire can lead to physicochemical changes within the soil creating feedback mechanisms that lead to greater runoff (Mills and Fey, 2004).

One aim of the analysis was to see if runoff generation occurred through different trigger mechanisms under different burning regimes. The results from the event analysis suggest that this is not the case; rather, it is an issue related to the sensitivity of the plots to each rainfall event. All plots showed that the only significant descriptor was Max DI/T, but that the 10-year plots show the lowest threshold value for Max DI/T compared to the other burning rotations. This indicates runoff will occur on the 10-year plots during more uniform rainfall events.

Changes in the hydrological behaviour of peatlands have important consequences for other ecosystem functions. The work on this study site was stimulated by questions of the effect of burning upon DOC concentrations draining peat soils and whether this land management had a detrimental effect for the UK situation (Worrall and Burt, 2007). This study has showed no significant effect of burn treatment upon this site for DOC concentration in either soil water (section 2.4.1.1) or runoff water (section 2.4.1.2), but showed that runoff water had significantly lower concentration of DOC relative to soil water (section 2.4.1.3). With a greater amount of precipitation partitioned into runoff following a fire, changes to degree of interaction between precipitation and ground water in the surface of peat may have implications for DOC dilution and export. A shift to greater runoff could cause DOC concentrations in the stream water of peat-covered catchments to decrease following burning.

2.5.3 Chemistry results

In order to understand how source waters vary with burning, this study has used a multiple tracers approach. The use of water chemistry as a tracer of source waters is well established (Christophersen and Hooper, 1992; Rice and Hornberger, 1998). Many studies often use only one tracer (Katsuyama et al., 2001) though it is common to use multiple tracers (Worrall et al., 2006a). Conservative tracers should ideally be used (Christophersen and Hooper, 1992) but in a study of the effect of burning and grazing the use of non-conservative tracers will provide information on changes in the environment above as well as information upon hydrological flowpaths. This study uses a principal component analysis which has several advantages over traditional chemical tracer studies. Firstly, it does not assume the number of end-members, nor does it assume that end-member compositions are fixed over time. Secondly, the study can cover a broad period of time, in the case of this study several years. Thirdly, it is not restricted to a few events but many. Fourthly, the study includes information from many tracers.

How do concentrations of chemical species vary between burning regimes at the end of a burning cycle? The results show significant differences between months though this pattern is to be expected as it reflects differences across seasonal cycles. This study has also found significant differences in composition of soil water and runoff water between burning treatments.

Soil Water – end of burning cycle

Principal component analysis is able to show three different water sources that go to make up soil water compositions from this site. The first component shows high loadings (i.e. Ca, K, Mg, Na PO_4 and conductivity) which can be interpreted as base-rich groundwater which is similar to that reported in Worrall and Adamson (2008) and also to the end-member identified within the catchment (Worrall et al., 2003c). At Hard Hill the local topography may lead to some down-slope flow from the sub-surface geology further up the slope. The second indicates a shallow water component though it is not matched by high loadings of DOC observed in the previous work on this site nor by high negative loadings for pH that might be expected to correlate with these species. Finally, the third component, which has high loading for Cl and SO_4, suggests a rainwater composition. These different water sources can be used to help calculate difference in likely source water under different management conditions.

The presence of burning leads to lower concentrations, in soil water, of those species associated with base-rich ground waters e.g. Ca, Mg (Worrall et al., 2003c). Worrall and Adamson (2008) suggest that it is the development of mature, shrubby vegetation on unburnt sites that draws in deeper groundwater through increased evapotranspiration. Although it is unlikely the evaporation alone is able to draw groundwater from the sub-peat zone, increased evapotranspiration on the older sites could be accessing a deeper water source within the peat that is enriched in base-rich water from down-slope flow. This mechanism is likely to be limited on sites where shrubby vegetation is limited i.e. burnt sites. Indeed lower concentrations were found on burnt sites with the lowest concentrations occurring on 20 year plots. When covariates are considered Mg, Na and PO_4 are no longer significant, suggesting they can be explained by changes in pH or conductivity

Runoff Water – end of burning cycle

Runoff water shows an increase in Al and Fe on the 10-year plots suggesting on these plots mixing with shallow water at the surface may be occurring. The positive loadings for Al and Fe in the first component of the runoff water analysis suggest it is reflecting a shallow water component (Worrall et al., 2006a) again indicating mixing at the surface. However, only Fe shows significant changes with burning when covariates are considered. The high positive loadings for Al and Fe in the principal component analysis, in contrast to the negative loadings for other species, suggests a pH control on this component as it contrasts base and acidic cations. .

Unlike soil water, there were differences in runoff composition when grazing is considered. Here a decrease in Cl and increase in SO_4 could be seen as an increased importance of rainwater on grazed plots. Grazing by sheep would reduce the level of interception by vegetation leading to an increased mixing with rainwater.

Post-burn period

Increases in Al and Fe in soil water following burning suggests an increasing importance of a pH-controlled shallow water component and a decrease in base-rich groundwater influence shown by a decrease in Ca concentrations in the post-burn period.

Runoff water, in the post-burn period, shows decreases in species that are associated with shallow soil water e.g. Al. With the inclusion of covariates there are still four species showing significant decreases suggesting these changes are linked to burning rather than to changes in any other water quality parameters. The only species to show a significant increase following burning was Ca. This rise is not linked with pH or conductivity and is likely to be due to burning. Increases in calcium concentration following fires have been reported around the world (Belillas and Rodà, 1993; DeBano et al., 1979) and have been attributed to post-fire ash inputs. However, low concentrations in the pre-burn period coupled with the limited number of samples may influence this result.

Principal component analysis

Principal component analysis would seem to confirm the idea that in the post-burn year soil water becomes more like shallow soil waters and less like rainwater. If this is the case, what is the rainwater mixing with? Analysis of the runoff samples suggest that runoff is mixing with incoming rainwater and therefore the effect of burning is to increase the differences between pathways as soil water and runoff water compositions diverge. There are several possible mechanisms that could lead to changes in flow pathways that would alter the mixing of rainwater. The generation of hydrophobic compounds during a fire may reduce interaction with the soil leading to a greater degree being partitioned into surface flow (DeBano, 2000).

Alternatively, an intensification of soil crusting following burning could can lead to changes within the soil creating feedback mechanisms alter the amount of runoff (Mills and Fey, 2004). Finally, organic matter contributes to surface soil structure and the quantity and quality of organic matter can be affected significantly by fires (Gonzalez-Perez et al., 2004). It is this alteration than can lead to degradation in soil structure and changes in parameters such as porosity and infiltration (Neary et al., 1999)

What does this mean for DOC? Surface runoff water typically shows lower concentrations of DOC compared to soil water but what happens to the composition of source water following a managed burn? Following burning, runoff water displays a trend similar to that of rainwater suggesting that runoff becomes increasingly influenced by rainwater inputs. Precipitation coming into this catchment during the monitoring period had DOC concentrations typically less than 2mg l^{-1}. Therefore, if runoff water has a greater proportion of rainwater as a source, then DOC concentrations in runoff following a managed burn should be significantly lower. Indeed a relative decrease in the year following a managed burn has been observed at this site. The data suggest that soil water following the burn became more like shallow soil water. If soil water is becoming less mixed with rainwater DOC concentrations should be expected to rise. This work showed that DOC concentrations rose in the year following burning but that this effect was not significant.

Furthermore, this work shows that following a managed burn there is an increase in the frequency of runoff detection in crest fall traps which was taken as an increased importance of runoff water. By suggesting that flowpaths shifted towards a lower DOC water source, this would then predict that burning could act to lower DOC at scales greater than these study plots. However, this point cannot be tested from these data and will rely on an examination of these data in conjunction with local stream data to understand how flowpaths scale up.

Care must be taken in extrapolating these results to other locations. Moor House is often seen as a pristine peat site and the burning carried out in February 2007 was carefully controlled and at a relatively small scale, neither which may be typical of the UK situation. Finally, the mechanisms, or critical thresholds, leading to flow portioning following managed burning in this setting may not occur in other areas. This may explain why a variety of responses by DOC to burning have been observed: increases (Yallop and Clutterbuck, 2009); decreases (Worrall et al., 2007d); and no difference (This study; Ward et al., 2007).

2.6 Conclusions

UK uplands are heavily managed through the use of rotational burning and grazing by sheep. This study has investigated the effects of these management practices on blanket peat in northern England and has found a series of complex interactions between management practice, the development of long-term macroporosity and rainfall-runoff processes, and the concentration of DOC.

There are two themes about DOC concentration and hydrology that come through from the results of this work: what happens towards the end of a burning cycle and what happens in the period immediately after the fire.

At the end of the burning cycle, differences in the water table were found between the burning regimes; the shallowest on the 20-year plots and the deepest on the unburnt sites. One likely driver for this difference in water table position is the differences in hydraulic conductivity on the plots. A low hydraulic conductivity on the 20-year plots may create areas of buffered water table response leading the shallow water tables seen there. This shallow water table may also explain, in part, the significantly reduced water colour observed on these plots. This study suggests that it is the development of root systems and associated macroporosity that acts as the control on the hydraulic conductivity of the sites. The study looked at DOC concentration at the end of the burning cycle and found no clear evidence for either increases or decreases between burning regimes. At the end of a burn cycle, burning does not significantly affect DOC concentration in either soil water or runoff water. This study would suggest that longer burning rotations may be beneficial in order to reduce water colour in upland peat, but that burning in itself does not lead to dramatic increase in DOC in soil water or runoff water. Overall, runoff water has significantly lower values for the three carbon parameters; lower absorbance, DOC and specific absorbance.

The second theme is concerned with the period following the managed burns in February 2007. Following burning, short-lived peaks in DOC concentration and water colour were recorded but no significant effect was observed in the year following the fire. During the same period results show a shift towards greater proportions of water transmitted as runoff following managed burning. There are several reasons that could explain changes in runoff on recently the burnt sites (i.e. 10-year plots) but it is this increased sensitivity to rainfall events that leads to the increased runoff occurrence. Further analysis of the soil water and runoff water showed that it is not only the quantity of water that changed following burning but the nature of the waters also changed. Water samples from before and after the managed burn on the 10-year plots were analysed and analysis showed rotations in the trends after burning. Soil waters following burning became less mixed with rainwater and more like soil water. Runoff waters became more dilute with decreased interaction with soil water and increased influence of rainwater. The divergence of runoff and soil-water compositions show that flowpaths are altered by managed burning and this may explain changes in important water quality parameters such as DOC. This increased importance of a water source with lower DOC than soil water has important implications for the export of DOC.

These changes could be interpreted as increased importance of soil water and decreased importance of rainwater and groundwater compositions on burnt peat soils, i.e. there was evidence of changes in flow and water mixing pathways within the soil upon burning. Hydrological parameters (e.g. frequency of runoff, depth to water table and hydraulic conductivity) are affected by managed burning and showed an increased importance of runoff water following a managed burn. With lower DOC concentrations in runoff water compared to soil water, changes in flowpath upon burning may have important consequences to the export of DOC from a catchment following managed burns i.e. it could lead to lower DOC export even if burning caused increased DOC concentration in the soil water.

Chapter 3:

Hard Hill field experiment – Carbon Budgets

3.1 Introduction

Worldwide, peatlands cover between 386 and 409 million ha (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. Northern peatlands are a store of carbon and current estimates suggest that approximately 455 GtC, one-third of the world's soil carbon, is sequestered in peatlands (Gorham, 1991). The rate of accumulation of carbon during the Holocene has been estimated at 0.96 MtC yr^{-1} (Worrall et al., 2009a) making these sites significant sinks of carbon. However, under a predicted warming climate (IPCC, 2007), these sensitive areas could be converted from sink to source of carbon.

The responses of peatlands to increasing air temperatures are numerous: increased release of CO_2 from ecosystem respiration (Dorrepaal et al., 2009); greater number of droughts leading to enzymic-latch mechanisms (Freeman et al., 2001a); and increased water table drawdown (Christensen et al., 1998). The importance of peatland hydrology to carbon dynamics has already been demonstrated (Chapter 2) and an increased drawdown of water tables can lead to greater DOC export (Strack et al., 2008) or, when combined with enhanced CO_2 levels, could lead to further destabilisation of the carbon store (Ellis et al., 2009). Due to 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.

In calculating carbon budgets in peatland settings, two methodological approaches are often used. The first uses radiocarbon dating to calculate ages of peat in order to estimate the rate of carbon accumulation (Schlesinger, 1990; Tolonen and Turunen, 1996). However, this technique can only calculate average accumulation rates above a horizon and does not account for periods of carbon loss from the peat.

The second main approach is to calculate present-day carbon budgets based on fluxes of carbon through various pathways. For an upland peatland setting, Figure 3.1 shows the different pathways carbon may be lost or gained.

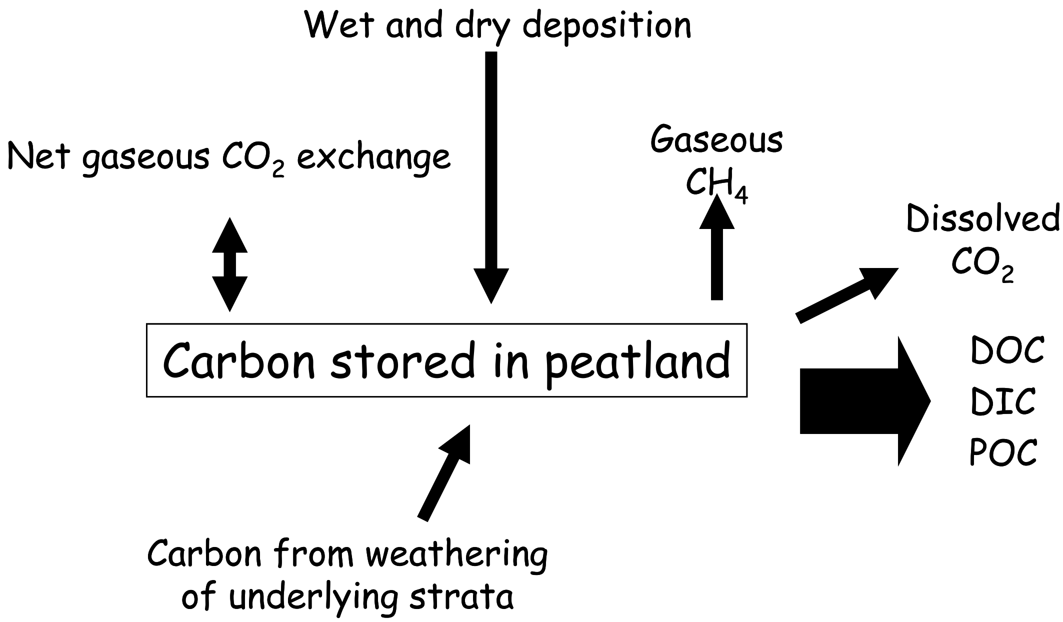

Figure 3.1. Carbon uptake and release pathways for upland peat

Worrall et al. (2003a) provide the first comprehensive study of the carbon balance of an UK upland peat by using a North Pennine catchment, Trout Beck, as the study site. Pathways included in calculating the carbon budget are: rainfall DIC and DOC; CO_2 exchange; CH_4 emissions; DOC export; POC export; dissolved inorganic carbon and dissolved CO_2 export and input from weathering of underlying strata. Results from this work showed that the site in 1999 was a net sink of -14.9 gC m^{-2} yr^{-1}.

Further work on this site improved the method and was able to predict future changes by modelling the catchment based on forecasted rainfall and temperature data (Worrall et al., 2007a). More recently, Worrall et al. (2009a) presented a multi-annual carbon budget of the Trout Beck catchment and showed that over the period 1993-2005, the site was a net sink of between -20 and -91 gC m^{-2} yr^{-1}.

Studies focussing on carbon fluxes from peat catchments tend to focus on gaseous exchange, mainly CO_2 but sometimes in association with CH_4; however, fluvial pathways are often left out. Evans et al. (2006) show the importance of fluvial carbon loss in upland peat settings and Worrall et al. (2006b) show that the degradation of DOC and POC releasing CO_2 to the atmosphere is a significant process. Therefore, fluvial carbon export must be considered when making carbon budget estimations.

This chapter presents measurements of many of the carbon pathways in order to make the best possible estimate of carbon budgets under burning and grazing regimes; where this is not possible data, have been drawn from the most applicable sources.

3.2 Chapter Objectives

The objectives of this chapter are to:

- Estimate carbon pathways throughout the monitoring period
- Calculate a complete carbon budget for the Hard Hill plots for each treatment.

3.3 Materials and Methods

3.3.1 Study site

The data for this chapter come from the Hard Hill plots at Moor House National Nature Reserve, North Pennines. For a full description of these plots refer to chapter 2. From these plots the following data have been measured in this study: DOC concentration and water table depth (Chapter 2.3.2); and surface exchange of CO_2. Two other carbon pathways have been studied at the Hard Hill plots in other work: POC (Clement, 2005); and CH_4 (Ward et al., 2007).

The UK Environmental Change Network (ECN) maintains a flow gauging station within the Trout Beck catchment with river discharge measured hourly (Figure 3.2). A meteorological station is situated within the catchment with hourly recording of rainfall, air and soil temperature and solar radiation. Continuous water table measurements (every 15 minutes) have been made since 1994 using pressure transducers and are also calibrated weekly.

ECN collects weekly samples of precipitation, stream waters and soil waters (Chapter 2.3.3.4). Details of values measured can be found in Chapter 2 and methods are detailed in Sykes and Lane (1996).

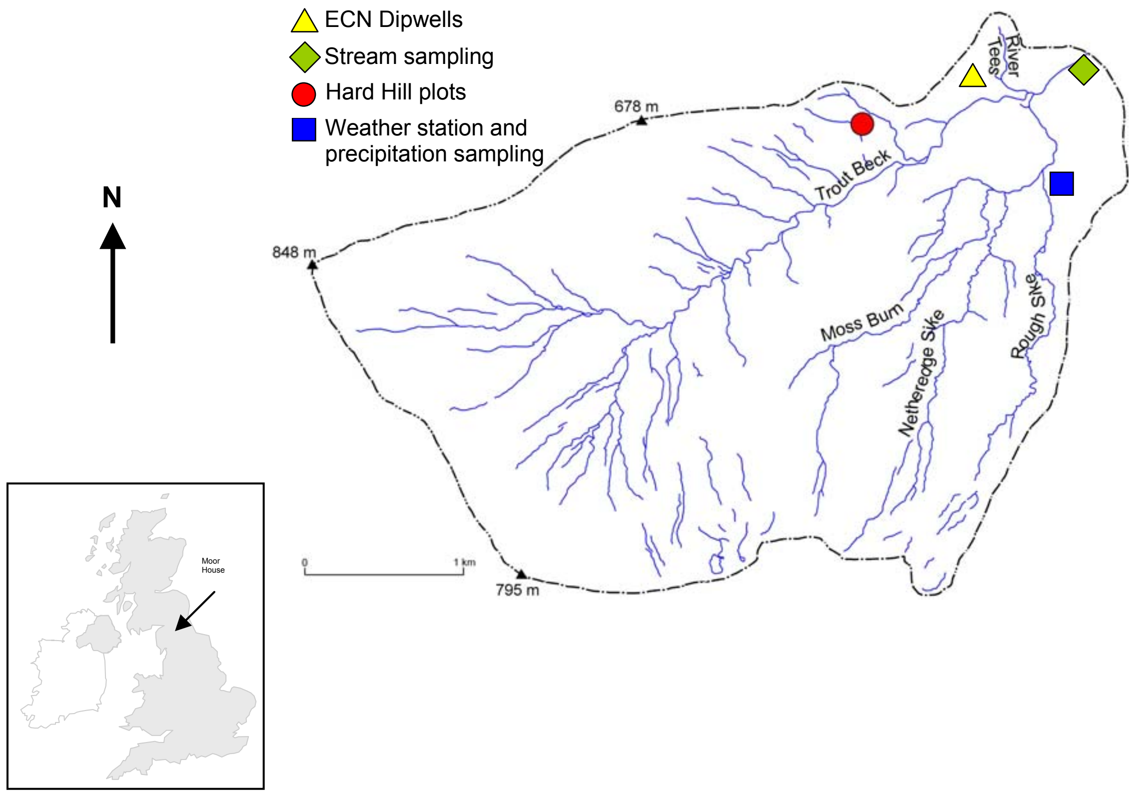

Figure 3.2. Location of Trout Beck catchment.

3.3.2 Budget calculation

Carbon budgets can be calculated for each management regime by taking values from those plots and extrapolating to the catchment i.e. what would the carbon budget be if that management regime were adopted across the reserve.

Fluxes are 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). A commonly used approach in carbon budgets of upland peats (Rowson, 2007; Worrall et al., 2003a; Worrall et al., 2009a) is 'Method 5' of Littlewood et al. (1998):

$$L_5 = \left(\frac{K \sum_{i=1}^{n} (Q_i C_i)}{\sum_{i=1}^{n} Q_i} \right) \overline{Q} \qquad \text{(Eq. 3.1)}$$

where K = conversion factor for period of sampling (24 ×3600× N i.e. number of seconds in *N*-day year); C_i = concentration of determinand in sample i; Q_i = flow corresponding to sample taken on day i; $\overline{Q}$ = mean river discharge over the period; and n = number of samples

Extrapolation methods take measured driver variables and use them to predict fluxes, sometimes beyond the range of current observations. If the drivers are known over a period of time, and provided there has been calibration between drivers and determinand, then annual fluxes can be calculated. Strong relationships exist between monitored variables and net ecosystem respiration, primary productivity and methane that allow extrapolation methods to be used.

Carbon budgets are calculated for the period April 2005 to December 2007 thus 2005 is not a complete calendar year. 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 was calculated. This 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 three months.

3.3.3 DOC

DOC concentration (mgC l^{-1}) was measured for each treatment type and for soil and runoff water; however, for the purposes of carbon budget modelling, and in order to calculate the upper limits of DOC export, only soil water is used in this study. For further details of these measurements refer to chapter 2. This approach is probably an overestimate as it assumes no dilution from groundwater or any in-stream conversion processes (Worrall et al., 2006b) though it has shown to be a suitable, conservative method that adequately describes the source of the DOC leaving the catchment (Worrall et al., 2003c)

3.3.4 POC

No direct POC measurements were taken at Hard Hill. However, a previous study (Clement, 2005) measured suspended sediment concentrations (SSC) from rainfall simulation studies on the Hard Hill plots.

3.3.5 Dissolved CO_2

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 was calculated using the method in Neal and Hill (1994) and based on alkalinity, pH, calcium concentration and temperature. Aluminium concentration is also commonly used in speciation models. Calcium and aluminium concentration and pH were measured as part of the monitoring of the Hard Hill sites (Chapter 2) whilst ECN soil temperature was used as a proxy for water temperature. Alkalinity measurements were made at Hard Hill on 16th August 2007. Data from this date, in conjunction with additional data from Bleaklow, Peak District (F. Worrall, pers. comm.) were used to construct a model to predict dissolved CO_2 based on existing water quality parameters e.g. pH. A significant model was developed for pH and aluminium concentration:

$$dissCO_2 = 0.398pH - 0.69\log(Al) - 1.31 \qquad \text{(Eq. 3.2)}$$

$r^2 = 0.38 \qquad n = 33$

where: $dissCO_2$ is dissolved CO_2 concentration in mgC l^{-1} and Al is aluminium concentration in mg l^{-1}.

3.3.6 Surface exchange of CO_2

In order to gain a better understanding of the carbon fluxes at Hard Hill, CO_2 monitoring has been in place since late October 2006. Permanently fixed gas collars were installed on the sites at the same time as the crest-fall runoff traps. Initially two per plot were installed due to resources limitations; however, this was expanded to three per plot in during late spring 2007. Collars were inserted into the upper peat surface in close proximity to the dipwells and runoff traps.

Gaseous flux was defined using a micrometeorological approach where CO_2 release to the atmospheres is positive and CO_2 uptake is negative. The ecosystem respiration (ER) is defined as the total amount of CO_2 released from the peat surface in $gCO_2\ m^{-2}\ hr^{-1}$ and is always positive. Gross primary productivity (GPP) is the total amount of CO_2 (in $gCO_2\ m^{-2}\ hr^{-1}$) taken up by the plants at the peat surface and is always negative. 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 flux can be either positive or negative but has the unit $gCO_2\ m^{-2}\ hr^{-1}$.

The methods used mean that primary productivity is hard to measure, however, ER and NEE are easily measured therefore GPP is calculated as:

$$NEE = GPP + ER \qquad \text{(Eq. 3.3)}$$

where GPP is always negative and ER is always positive.

Sampling is done monthly by taking an infra-red gas analyser (IRGA) (PP Systems, EGM-4, Hitchin, UK) placing it on the collar, ensuring a tight fit, and letting the IRGA take a reading of the change in CO_2 concentration over the period of 124 seconds. Fluxes were then calculated from changes in these concentrations. For details of the method, see Rowson (2007). Measurements are taken in the dark (with cover) in order to calculate ecosystem respiration and taken in the light (no cover) to measure net ecosystem exchange.

Due to a limited number of readings prior to the burn in February 2007, the calibration of the respiration and primary productivity, described in the following sections, is based primarily on data from 2007 and on the 10-year plots, from after the burn.

3.3.6.1 Respiration

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 water table measurements of dipwells were also taken at the same time.

In order to extrapolate CO_2 fluxes a long-term temperature and water table record were needed. ECN monitors air temperature each hour. 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 six dipwells instrumented with pressure transducers on the Moor House site (Figure 3.2). The six calibration curves (6 treatments) had r^2 values of between 0.75 and 0.84 (Figure 3.3; Table 3.1)

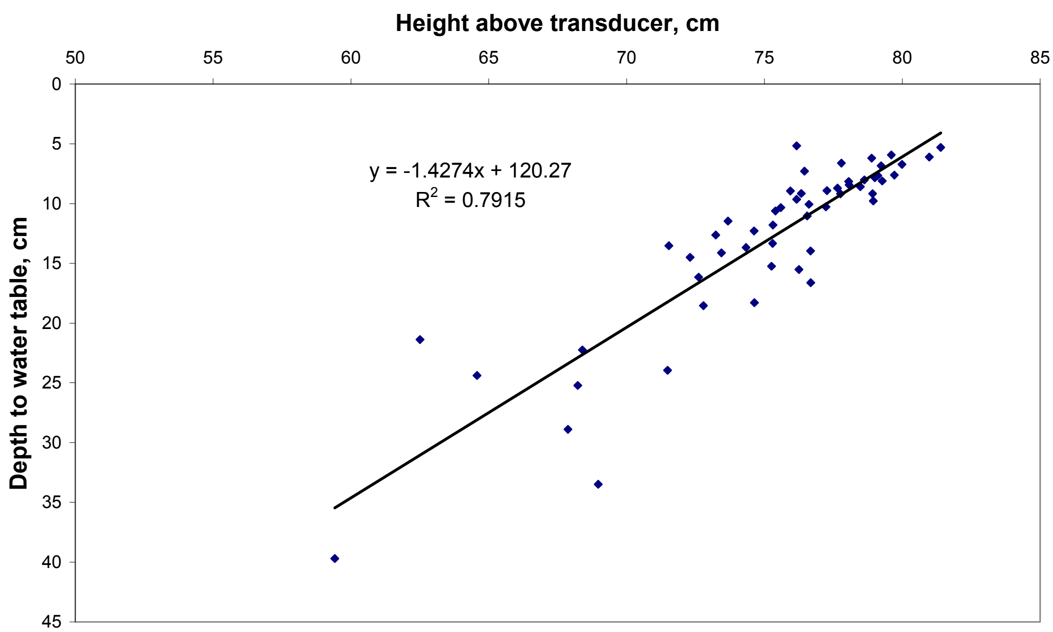

Figure 3.3. Example of a water table calibration curve – grazed, no burn

	graze, no burn	graze, 10 yr	graze, 20 yr	no burn, no graze	no graze, 10 yr,	no graze, 20 yr
r^2	0.79	0.84	0.80	0.76	0.83	0.75

Table 3.1. r^2 values for water table calibration curves

In order to calculate ecosystem respiration, an iterative approach was used to solve the Lloyd and Taylor (1994) method of predicting gross flux:

$$R = R_{10} e^{E_0\left(\left(\frac{1}{283.15-227.13}\right)-\left(\frac{1}{T_{soil}}-227.13\right)\right)} \qquad \text{(Eq. 3.4)}$$

where: R = gross flux value ($gCO_2\ m^{-2}\ hr^{-1}$); R_{10} = respiration rate of a collar at 10°C ($gCO_2\ m^{-2}\ hr^{-1}$); E_0 = unitless constant; and T_{soil} = soil temperature (K).

This equation can be modified to include a water table function (Lloyd, 2006; Rowson, 2007).

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

(Eq. 3.5)

where: (A×WTD + B) is effectively R_{10}; WTD is depth to water table (mm) and A and B are constants for that treatment.

Calibration of equation 3.5 was achieved with measurements from dark chamber readings. Respiration flux for each collar was calculated from the dark chamber reading. Temperature and water table depth at time of IRGA measurement were also measured. R_{10} values could then be calculated for each treatment based on an iterative solving solution. By combing R_{10} values for each treatment, water table record and temperature record, an estimate of the ecosystem respiration can be made.

3.3.6.2 Primary Productivity

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:

$$PP = \left(\frac{GP_{\max} \alpha PAR}{\alpha PAR + GP_{\max}} \right) \qquad \text{(Eq. 3.6)}$$

where α = 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}).

In order to predict the constants, α and GP_{max}, primary productivity was plotted against PAR for each treatment. By plotting a rectangular hyperbola,

GP_{max} is defined as the maximum amount of CO_2 taken up and α as the linear part of the hyperbola (Figure 3.4)

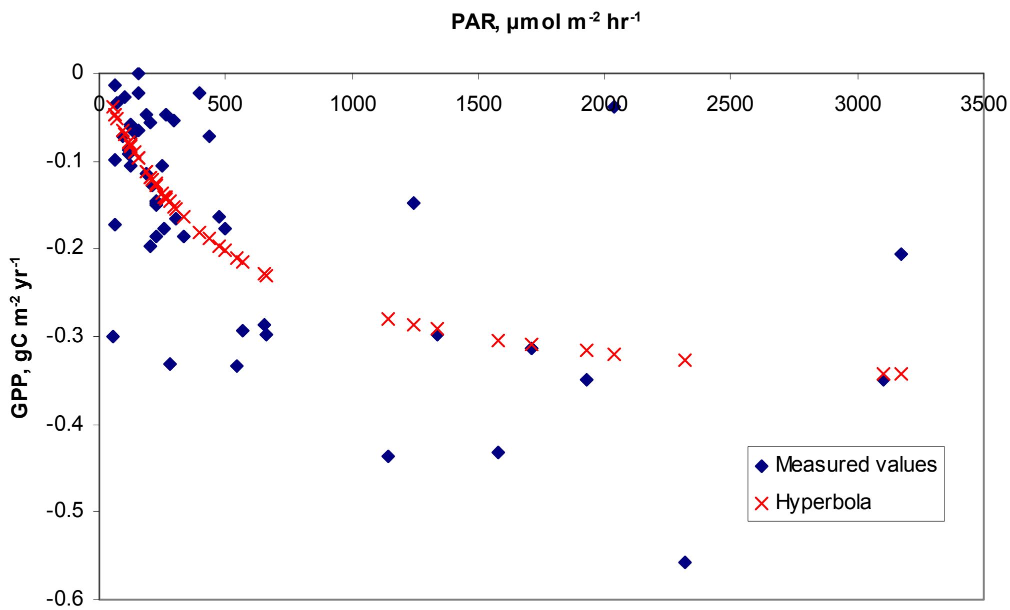

Figure 3.4. Plot of PAR and GPP for grazed, no burn plots

A long-term record of PAR was not maintained at the site so, for the purposes of flux prediction, a best-fit calibration between PAR and solar radiation was used (Worrall et al., 2009a). Solar radiation records are measured every 15 min and calibrated against hourly records of solar radiation in the form:

$$PAR = 19.39 + 1.79S \qquad r^2 = 0.82 \; n = 8760 \qquad \text{(Eq. 3.7)}$$

where: S = solar radiation (W m^{-2}); PAR = photosynthetically active radiation (μmol m^{-2} hr^{-1}).

3.3.7 Rainfall carbon

The annual input of carbon from precipitation was calculated from DOC measurements from rainfall samples collected as part of the ECN monitoring at Moor House and from 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.

3.3.8 Methane

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 (Reed and Mitchell pers. comm. cited in Worrall et al., 2009a):

$$\ln F = 4.12 - 3.9W_D \qquad \text{(Eq. 3.8)}$$

where F = the molar flux of CH_4 (μmol CH_4 m^{-2} hr^{-1}) and W_D = depth to water table (m).

3.3.9 Carbon budget

To calculate the carbon budget of the Hard Hill plots, the method of Worrall et al. (2009a) was used. The total magnitude of the carbon sink can be calculated thus:

$$F_c = PP + ER + POC + DOC + dissCO_2 + CH_4 \qquad \text{(Eq. 3.9)}$$

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

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., 2009a).

3.4 Results

3.4.1 DOC

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

3.4.2 POC

Table 3.2 shows the mean suspended sediment concentrations (SSC) from rainfall experiments conducted on the Hard Hill as part of Clement (2005) Assuming 50% of the sediment is in the form of carbon, then values for POC range from 11 mg l^{-1} to 38 mg l^{-1} (Table 3.2)

Treatment	Grazed, no burn	Grazed, 10 yr	Grazed, 20 yr	No graze, no burn	No graze, 10 yr	No graze, 20yr
Mean SSC (mg l^{-1})	66.05	44.08	27.33	22.22	35.7	75.38
POC (mg l^{-1})	33.02	22.01	13.67	11.11	17.85	37.69

Table 3.2. Suspended sediments concentrations (SSC) from Clement (2005) and estimated POC concentrations.

Clement (2005) shows that, although runoff production was increased on the burning plots, burning had little effect on sediment yield in these plots experiments. This work also only considers larger rainfall events with high intensities (12 – 24 mm hr^{-1}) in order to consider those events that contribute most to peat loss (Evans and Warburton, 2005). Most of the events at Moor House over the study period were less than 12 mm hr^{-1}; therefore in order to assess the POC loss, a lower intensity calibration curve would be required.

The POC flux, based on these data and flow from the catchment outlet, ranges from 14 – 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., 2009a); however, values from that study were based on the flux of POC from the catchment outlet. This 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.

3.4.3 Dissolved CO_2

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}. This is within then range of previously published results (Worrall et al., 2003a)

3.4.4 Surface exchange

3.4.4.1 Ecosystem Respiration (ER)

Respiration calibration resulted in a range of fits to the measured data though they are all significant at $p<0.01$ (Table 3.3). Coefficients A and B from Equation 3.6 are also presented in Table 3.3.

	graze, no burn	graze, 10 yr	graze, 20 yr	no burn, no graze	no graze, 10 yr,	no graze, 20 yr
A	-0.0003	0.0000	0.0000	-0.0006	0.0002	0.0005
B	0.1657	0.1089	0.0874	0.1949	0.1031	0.0974
r^2	0.63	0.62	0.70	0.48	0.44	0.47

Table 3.3. R_{10} values for A and B for the combined approach and r^2 values of fit to measured values.

Using these coefficients along with the calibrated water tables and ECN temperature record, respiration was modelled for the study period (2005-2007) with the assumption that the coefficients did not change during the period. Ecosystem respiration varied across the treatments from 136.6 to 258.7 gC m^{-2} yr^{-1}. This is higher than reported from Moor House (Worrall et al., 2009a) though similar to results from other areas of the North Pennines measured using similar techniques (Rowson, 2007).

3.4.4.2 Primary Productivity (PP)

Primary productivity calibration resulted in significant ($p<0.05$) fits between measured and modelled values (Table 3.4)

	graze, no burn	graze, 10 yr	graze, 20 yr	no burn, no graze	no graze, 10 yr,	no graze, 20 yr
α	-0.000835	-0.000900	-0.001301	-0.001704	-0.000928	-0.001203
GP_{max}	-0.394330	-0.365593	-0.217564	-0.127970	-0.437923	-0.362194
r^2	0.43	0.29	0.20	0.18	0.12	0.19

Table 3.4. α and GP_{max} values used in modelling and r^2 of fit with measured values.

Taking these values to be representative of the sites during the study period, primary productivity varied between -137.8 and -222.5 gC m^{-2} yr^{-1}, which is within the range of reported values for upland peat (Worrall et al., 2009a).

3.4.5 Rainfall DOC

Rainfall DOC concentration varied from 0 – 5.4 mgC l^{-1} over the study period and are similar to values presented in other studies (Worrall et al., 2003a). Inputs from rainwater DOC varied over the three years from -0.9 to -2.1 gC m^{-2} yr^{-1} which is a similar to ranges reported elsewhere (Worrall et al., 2007a; Worrall et al., 2009a). 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}.

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.

3.4.6 Methane

The values of CH_4 flux from the plots ranged from 5.25 to 6.86 gC m^{-2} yr^{-1}. This range is similar to that reported in Worrall et al. (2009a). 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}.

3.4.7 Carbon budget

Table 3.5 details the different carbon pathways, measured or estimated during this study, and the estimates for the carbon flux for each year of the study period.

2005	graze, no burn	graze, 10 yr	graze, 20 yr	no burn, no graze	no graze, 10 yr,	no graze, 20 yr
PP	-188.39	-189.85	-173.77	-140.41	-209.25	-219.89
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
CH4	5.58	6.51	6.86	5.81	6.18	6.53
DissCO2	2.47	1.85	1.79	2.35	1.99	1.98
Total	**115.16**	**63.68**	**37.18**	**134.97**	**64.06**	**92.89**
2006	graze, no burn	graze, 10 yr	graze, 20 yr	no burn, no graze	no graze, 10 yr,	no graze, 20 yr
PP	-191.81	-192.90	-174.51	-139.82	-213.05	-222.52
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
DissCO2	3.57	3.17	2.91	3.35	2.72	3.02
Total	**126.56**	**103.48**	**70.06**	**129.72**	**112.35**	**182.18**
2007	graze, no burn	graze, 10 yr	graze, 20 yr	no burn, no graze	no graze, 10 yr,	no graze, 20 yr
PP	-186.67	-187.96	-171.15	-137.76	-207.35	-217.33
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
CH4	5.46	6.36	6.73	5.71	6.03	6.40
DissCO2	1.28	1.72	1.91	2.17	1.77	1.80
Total	**139.92**	**96.72**	**80.59**	**157.02**	**106.18**	**147.19**
Mean total budget	127.21	87.96	62.61	140.57	94.20	140.75
Standard Deviation	12.39	21.30	22.64	14.49	26.28	44.99

Table 3.5. Summary of each carbon uptake and release pathway for each year (2005-2007) for measured and modelled values.

Examining the data shows that all the treatments are net sources of carbon, 37.2 – 182 gC m^{-2} yr^{-1}, during the study period, though some sites are smaller sources than others (Table 3.5). Over the study period, unburnt sites were, on average, a source of 133.89 gC m^{-2} yr^{-1} compared to a source of 91.1 gC m^{-2} yr^{-1} and 101.7 gC m^{-2} yr^{-1}on the 10-year and 20-year plots respectively. Figure 3.5 shows the data in an alternative format with the ranges for carbon flux for each treatment across the study period.

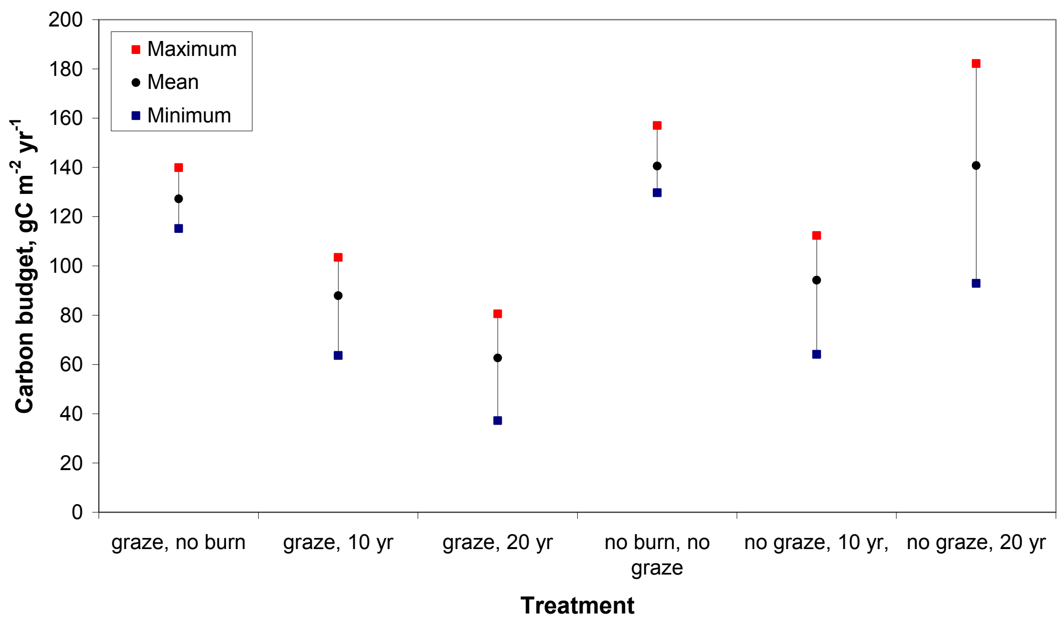

Figure 3.5. Range of carbon budget for each treatment

If only gaseous emissions (ecosystem respiration, primary productivity, and methane) are considered, then grazed and burnt plots are sinks of carbon throughout the period and ungrazed, burnt plots are occasionally sinks of carbon. When hydrological export of carbon (DOC, POC and dissolved CO_2)

is included the sites become a source of carbon. The DOC flux in this study is at the higher end of reported values (Worrall et al., 2009a) and this study shows an upward trend over the period (Figure 3.6) suggesting that this may be one possible reason for why the carbon budgets indicate that the plots are carbon sources.

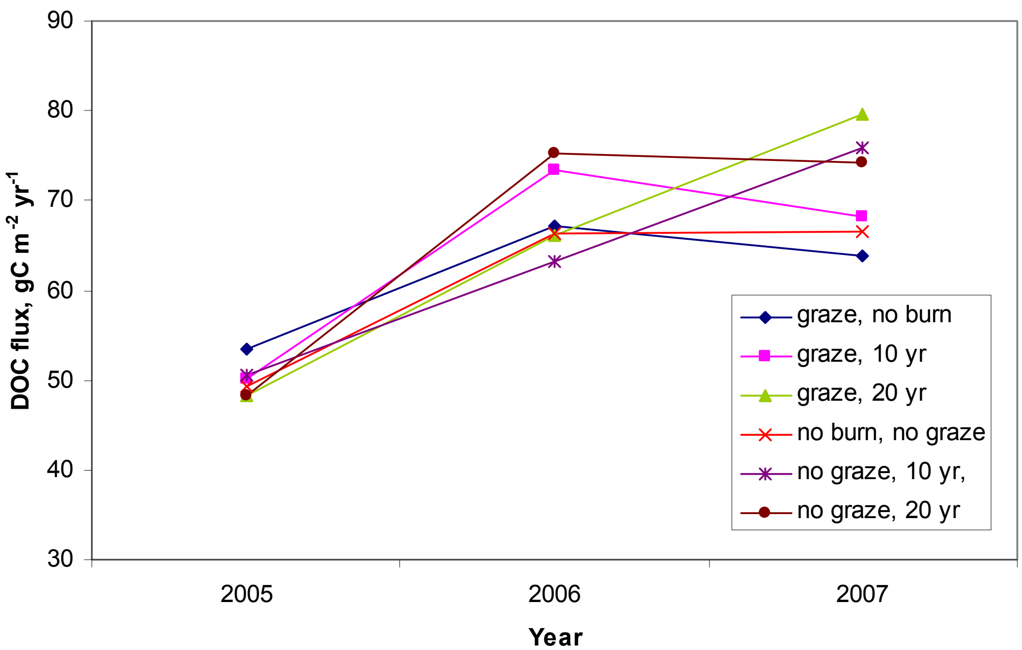

Figure 3.6. DOC flux for each treatment over the study period

To assess any significant differences between treatments, ANOVA was carried out with Year, Burning regime and Grazing regime as factors and post-hoc testing was carried out to investigate where the significant differences lay. Burning and grazing regimes, along with year, were significant factors in the carbon budgets (Table 3.6).

Factor	df	p	ω^2
Burn	2	**0.014**	0.23
Grazing	1	**0.008**	0.19
Year	2	**0.017**	0.21
Burn*Grazing	2	**0.021**	0.18
Burn*Year	4	0.226	0.04
Grazing*Year	2	0.642	0.00
Error	4		0.14

Table 3.6. ANOVA of the total carbon budgets. Values of $p < 0.05$ are highlighted and ω^2 = proportion of variance explained.

Inter-annual variation accounted for 21% of the variation in the data with 2006 and 2007 being significantly greater sources than in 2005. Grazing explained 19% of the variation with grazed sites having significantly lower sources than ungrazed sites. Finally, burning accounted for the largest proportion of the variance, 23%. Here, the presence of burning rather than a specific regime led to significantly smaller sources. The interaction term between burning and grazing was also significant explaining 18% of the variation in the data.

3.5 Discussion

This study has shown that for some small-scale plots under different management the carbon budget is positive i.e. net source of carbon. This is opposite in sign to other studies based on Moor House (Worrall et al., 2003a; Worrall et al., 2009a) which poses the question, why?

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-year and 20-year plots show significantly lower overall budgets than unburnt plots. The main reason for this difference is the combined effect of higher primary productivity on some burnt sites and lower ecosystem respiration which leads to a negative gaseous carbon balance. 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 above ground biomass production, inflorescence density and plant height in annually burnt sites. As the vegetation becomes older and degenerate, its ability to sequester CO_2 reduces. Bond-Lamberty et al. (2004) show, for a fire-prone forest chronosequence in Canada, that net primary productivity in young stands were net sources, middle aged stands were net sinks whilst the oldest stands were carbon neutral. However, it must be remembered that although burnt sites may be smaller sources, the loss of carbon during combustion of the biomass could outweigh any benefits to the carbon budgets. In Chapter 5, in a survey of a wildfire, up to 90% of biomass of was lost during combustion and in managed burns this loss may be approximately 60% (Allen, 1964).

On the whole, the largest sources in this study were the unburnt sites, in particular the unburnt, ungrazed, sites; however ungrazed, 20 year plots are also large source due in part to large POC concentrations from these sites. On these sites there are very high respiration values and the lowest primary productivity 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 (Chapter 2) allowing a greater depth of aerobic decomposition leading to greater respiration values (Moore et al., 1998).

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).

Another possibility for the source-sink discrepancy is that the hydrological export of carbon has changed since the study period of Worrall et al. (2009a). If considering only gaseous exchange, then the site is net sink of carbon but when other pathways are included the site is a net source. This is a similar situation to Worrall et al. (2007a) that while the site overall was estimated to be a net source of 11.2 to 20.9 gC m^{-2} yr^{-1} the gaseous components were a net sink.

Previous studies (Worrall et al., 2007a; Worrall et al., 2009a) have scaled results from Moor House to the larger UK scale. By taking the extent of peat in the UK to be between 14,790 km^2 (Tallis and Meade, 1998) and 29,209 km^2 (Milne and Brown, 1997), estimates of the area of UK peat were randomly selected from this range and combined with randomly a selected carbon budget from the ranges in this study. This is repeated 100 times for each treatment. This results in an estimate of the carbon budget of the UK under each management regime of between 1.27 ± 0.38 Tg yr^{-1} and 3.21 ± 0.6 Tg yr^{-1} (Figure 3.7). These ranges are larger than results presented in Worrall et al. (2007a).

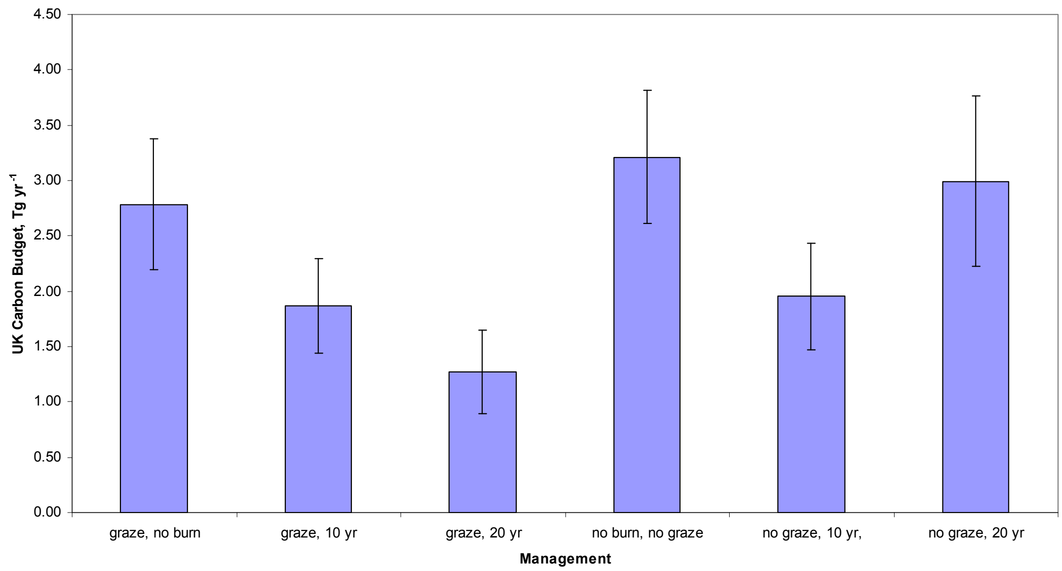

Figure 3.7. UK carbon budgets based on management regimes from this study

3.6 Conclusion

By using the best combination of experimental and modelling approaches, the treatments at Hard Hill are shown to be net sources of carbon of between 37.2 and 182 gC m^{-2} yr^{-1}. However, the presence of burning and grazing appears to limit the magnitude of this source by reducing ecosystem respiration and enhancing net ecosystem exchange. When considering the carbon budgets of upland peat managed by fire, the loss of biomass and carbon through combustion must be considered in order to assess where carbon benefits lie.

The largest part of this budget are two components the gaseous exchange of carbon, primary productivity (-223 to -138 gC m^{-2} yr^{-1}) and net ecosystem exchange (137 to 259 gC m^{-2} yr^{-1}). Hydrological export of carbon via DOC is the next largest component of the carbon budget (48 to 80 gC m^{-2} yr^{-1}) and turns the sites, which are gaseous sinks of carbon, into an overall source of carbon. By extrapolating these ranges across the UK, the carbon budget of UK peat soils would be a net source of up to 3.2 TgC yr^{-1}

Chapter 4:

Loss and transformation of carbon from vegetation

4.1 Introduction

It is estimated that the burning of biomass releases between 2 and 6 x 10^{15} gC yr^{-1} (Crutzen and Andreae, 1990; Wittenberg et al., 1998). This burning of biomass leads to production of black carbon (or char) on the scale of 270 x 10^{12} gC yr^{-1} (Kuhlbusch and Crutzen, 1995). Black carbon has been considered refractory in the environment and thus represents a input of carbon (Kuhlbusch, 1998). These additional inputs of carbon have been ignored in the carbon budgets of fire-affected peatlands; neither Garnett et al. (2000) nor Harden et al. (2000) consider char inputs, and both show that fire limits the magnitude of sink.

A common management technique used on peat soils of the UK uplands is the burning of vegetation on a regular cycle. The burning is undertaken to promote new vigorous vegetation growth that provides improved forage for sheep grazing. Furthermore, the rotational burning practice over cycles from five to 25 years means that a patchwork of vegetation is created that provides both forage and cover for the ground-nesting red grouse. Presently the UK Government Department of Environment, Food and Rural Affairs (DEFRA) restricts managed burns in terms of timing, frequency and size of burnt area (DEFRA, 2007a). The timing restrictions on burning are to ensure it takes place while the ground and vegetation are sufficiently moist to ensure a “cool burn” and thus reduce damage to the underlying peat. The size of burnt areas is restricted to no more than 30m wide by 150 m long in order to limit the possibility of runaway wildfire, but this width of burn strips coincides with preferred foraging distance of the red grouse.

In a recent review of the consequences of heather and grassland burning, including that on peat (DEFRA, 2005; Tucker, 2003), there were very few studies reported that examined the consequences of burning for carbon storage. Worrall et al. (2007d) has shown significant differences between burning regimes in soil water conductivity, pH and the depth to the water table. Chapter 2 found no significant difference in dissolved organic carbon (DOC) concentrations for a year either side of a managed burn and for the same site showed a significantly higher water table with increased frequency of burning. Ward et al. (2007) have shown significant increases in gross ecosystem CO_2 fluxes in burned and grazed treatments relative to the control plots. Intentional and catastrophic burns have been linked to increased peat erosion and therefore mass losses of carbon and particulate organic carbon (POC) (Mackay and Tallis, 1996).

Garnett et al. (2000) examined peat accumulation of carbon under three treatments (grazed/unburnt, grazed/burnt, and ungrazed/unburnt) using spherical carbon particles to define a common horizon representing peak production of soot particles with changes in industrialisation in the region. Peat accumulation and carbon accumulation were calculated above this common horizon. Garnett et al. (2000) report a mean difference between burnt and unburnt treatments of 2.3 kg m^{-2} . This difference represents a reduced carbon accumulation with burning of 73 Mg C km^{-2} yr^{-1}. This study only covers a 10-year burning frequency. Other studies have considered the role of wildfire rather than managed burning on the accumulation of peat. Kuhry (1994) 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, i.e. they do not consider the presence of different carbon types.

In managed burning, the biomass is turned into fumes, smoke and char. Fumes and smoke represent the airborne fraction of the combustible biomass with the former representing the vapours from burning and the latter representing the airborne solid and liquid particulates; char is the solid material that remains following burning. Some biomass remains as unburnt but possibly dead material and therefore represents an additional litter input. The fumes and smoke represent a loss of carbon from the ecosystem and the loss of live vegetation also means the loss of litter production in years subsequent to the burn until there is full recovery of the vegetation. In opposition to the losses, the production of char and of dead biomass litter represents an input of carbon into the peat. This means that at the time of the fire litter input is substituted for char inputs. Litter is a high-volume, low-carbon content, labile organic matter relative to char that is a low-volume, high-carbon content material. Char has mean residence times of up to 10,000 years in soils (Swift, 2001) while typical turnover times of soil organic matter in soil surface layers is between 6 and 20 years (Torn et al., 2005). The question becomes whether sufficient refractory carbon and/or sufficient dead biomass can be produced during a fire that can offset the loss of biomass by burning and the subsequent loss of litter production.

This chapter seeks to understand the impact of burning of vegetation on loss of carbon to the atmosphere and the production of char products. Laboratory studies of soil carbon loss and transformation have been conducted by a number of workers (e.g. Almendros et al., 2003), and studies of experimental burns have occurred in the field (e.g. Fearnside et al., 2001). However, no study has measured the loss and conversion of biomass in the laboratory in order to inform the estimation of the carbon budgets of fires.

4.2 Chapter Objectives

The objectives of this chapter are to:

- Conduct experimental burning in laboratory conditions to investigating the loss of biomass and production of char;
- Model the production of char over a different range of fire conditions;

- Investigate the carbon storage of peat under different burning regimes.

4.3 Materials and methods

This series of experiments aims to simulate the burning conditions experienced by the three most common vegetation types found in upland peat of the north Pennines. All samples were collected fresh from the Moor House National Nature Reserve in Upper Teesdale (N 54:41:45 W 2 2:24:46) used in Chapter 2 and 3, though none of the samples came from within the experimental plots on Hard Hill. The vegetation types chosen were heather (*Calluna vulgaris*), cotton grass (*Eriophorum spp*.) and sphagnum mosses (*Sphagnum spp*.) The vegetation and soil samples from Moor House were collected fresh and placed in sealed plastic bags so that moisture loss was limited prior to experimentation.

4.3.1 Obtaining samples

4.3.1.1 Production of experimental samples

To replicate a range of burning conditions, the samples of vegetation and soil were treated in a factorial designed experiment. The factors considered were:

i) Burning temperature – For the burning of heather, fire temperature between 220°C and 886°C have been observed (Hamilton, 2000; Whittaker, 1961); however, the lower reported temperature is below the reported ignition temperature for vegetation (Pyne et al., 1996). Therefore, the following burning temperatures were chosen for vegetation - 400, 600 and 800°C.

ii) Burning time – the samples were exposed to two different burn times – two and five minutes. These times were chosen to cover typical fire speeds of upland burns (SEERAD, 2001a). Shorter exposure times would be difficult to replicate in the laboratory, as the shorter the time, the greater the proportion of that exposure time that is represented by placing or removing the sample in the furnace.

iii) Initial temperature – burning in the field takes place at range of ambient temperatures set by the weather conditions on the day of the burn and so the samples are stored before exposure to the furnace at three different temperatures – room temperature (22°C), refrigerated (4°C) and frozen (-5°C). Samples were left at these respective temperatures overnight before being burnt in the furnace.

iv) Return temperature – as stated above, the ambient temperature experienced by burnt vegetation varies and so it is useful to consider materials at different starting temperatures, but also, once the fire front has passed over vegetation in the field, the burn products experience different temperatures. It is possible that in particularly cold conditions the effects of burning are effectively quenched and smouldering is restricted, that in turn limits the loss of carbon or production of burn products. Thus, samples in this experiment having been in the furnace are returned to conditions at a known temperature overnight before analysis. The temperatures used were room temperature (22°C), refrigerated (4°C) and frozen (-5°C). However, it should be noted that this is a complete factorial design and so samples were not necessarily returned to the initial temperature from which they came, rather samples experienced all possible combinations of factors.

v) Vegetation – the study considered the behaviour of each of three vegetation types under every combination of the factor levels above.

The initial moisture content of the samples was not considered as a factor within the experiment as it was considered too difficult to manipulate to set levels between the differing vegetation. However, the moisture content was measured on subsets of vegetation samples that were about to experience a given set of experimental factors. The moisture content of sub-samples were measured by weighing samples prior to being left at 105°C overnight and then being reweighed. The moisture content of samples is then considered as a covariate within the analysis of the experimental data.

Samples of vegetation were placed in ceramic crucibles and weighed before and after burning under their particular combination of experimental factors. Samples were also dried in a drying oven following the flash burning in order to calculate whether any moisture remained in the sample. Therefore, both wet and dry weight loss could be considered in the analysis.

On the basis of the first set of experiments, a second set of experiments were then performed for vegetation types in which the following additional burn temperatures were considered – 450, 500, 550, 650, 700 and 750°C. For this second set of experiments, the design only considered room temperature values for the initial and return temperatures but was otherwise factorial with respect to burning time and vegetation.

4.3.1.2 Field samples – Peat cores

To assess the carbon storage under different management regimes, peat cores were taken from the Hard Hill plots (Chapter 2; Figure 2.1). Initial sampling of cores was prior to the managed burn on 6th February 2007 using a Dutch auger. The aim was to sample as deep as possible i.e. 1 metre; however, this did not always occur so any short cores or damaged cores were rejected. Observations on the cores were recorded in the field and show that finer material and fewer identifiable plant remains were seen from approximately 15 cm to the base of the core which reflects enhanced peat humification (Givelet et al., 2004).

Cores were cut into smaller sections in the field before transport to the laboratory. Peat between the surface and 20 cm depth was sampled in 2 cm intervals whilst 20 – 50 cm was sampled in 5 cm intervals. Samples were transported back to the lab in individual bags where they were stored at 4°C until time to prepare the samples. Samples were dried in a drying oven before crushing and removal of any roots. Samples were also weighed to determine their dry weight (DW) therefore the dry bulk density (DBD). Using the dry bulk density, and by determining the carbon content of each section, the carbon stored in each horizon could be calculated.

Following initial analysis of the results, a further set of cores was obtained on 3rd November 2008. Only the top 20 cm was sampled on this occasion.

4.3.2 Analytical techniques

In addition to mass loss (both wet and dry), the products of burning were also analysed. The burn products were analysed in two ways – CHN (Carbon, Hydrogen and Nitrogen) analysis and pseudo-thermogravimetric analysis (pTGA)

4.3.2.1 Wet and dry mass loss

To evaluate the role of fire characteristics on vegetation loss, the total mass loss was calculated for each sample. Firstly, the mass of the sample was calculated before and after the flash burning by:

$$S_{start} = C_{start} - C \quad \text{(Eq. 4.1)}$$

and $$S_{flash} = C_{flash} - C \quad \text{(Eq. 4.2)}$$

where S_{start} is the sample weight before flash burning, S_{flash} is the sample weight after flash burning, C is the weight of the crucible, C_{start} is the weight of sample and crucible before flash burning and C_{flash} is the weight of sample and crucible after flash burning.

Thus the percentage total mass loss through flash burning,

$$Totalmassloss = 100 - \left(\frac{S_{flash}}{S_{start}} \times 100 \right) \quad \text{(Eq. 4.3)}$$

However, this does not take into account changes to the dry mass of the sample. A 50% total mass loss in *Sphagnum*, which has very high moisture content, will not affect dry mass as much a 50% loss in *Calluna*, which has a lower moisture content. Therefore the total dry mass loss needs to be known and is calculated by:

$$drymassloss = \frac{D_{flash} - D_{start}}{D_{start}} \times 100 \quad \text{(Eq. 4.4)}$$

where D_{flash} is the dry mass after flash burning and D_{start} is dry mass before flash burning

The dry mass of the sample before flash burning is given by

$$D_{start} = \frac{S_{start}}{\left(\frac{M_{start}}{100}\right)+1} \qquad \text{(Eq. 4.5)}$$

where M_{start} is the initial moisture content calculated by:

$$M_{start} = \frac{S_{start} - S_{105}}{S_{105}} \qquad \text{(Eq. 4.6)}$$

where S_{105} is the sample weight after 24 hrs at 105°C

The dry mass of the sample after flash burning is calculated in a similar fashion and is given by:

$$D_{flash} = \frac{S_{flash}}{\left(\frac{M_{flash}}{100}\right)+1} \qquad \text{(Eq. 4.7)}$$

where M_{flash} is the moisture content of material after flash burning calculated by:

$$M_{flash} = \frac{S_{flash} - S_{flash105}}{S_{flash105}} \times 100 \qquad \text{(Eq. 4.8)}$$

where $S_{flash105}$ is the weight of the flash burnt material after 24hrs at 105°C

4.3.2.2 CHN analysis

The one of the techniques used to analyse burn products was carbon, hydrogen and nitrogen (CHN) analysis. It would be expected that, as vegetation or soil is burnt, the mass would decline but the percentage carbon content of the remaining sample increase. CHN analysis of moorland vegetation (Chapter 5) typically gives carbon content values of approximately 45 - 50%. These values are similar to other values reported for moorland vegetation: *Calluna vulgaris* (Lageard et al., 2005); *Eriophorum vaginatum* (Thormann and Bayley, 1997); *Sphagnum spp* (Hall and Louis, 2004). A subset of the burn experiment samples were saved and analysed for their percentage C content.

Samples were analysed for their carbon, hydrogen and nitrogen (CHN) content on a Costech ECS 4010 Elemental combustion system with pneumatic autosampler in the Department of Geography, Durham University. It was set up for CHN analysis. Reactor 1 consisted of chromium (III) oxide/Silvered cobaltous-cobaltic oxide catalysts @ 950°C. Reactor 2 consisted of reduced high purity copper wires @ 650°C. Helium was used as the carrier gas at a flow rate of 95 ml min^{-1}. This was filtered for hydrocarbons upstream of the instrument. A packed (Porous polymer, HayeSep Q) 3m GC column was used for separation of the gases. This was replaced approximately every 120 samples though depending on circumstances e.g. cracked column, it was replaced more frequently. A thermal conductivity detector (TCD) was used to calculate the signal of each sample. Laboratory standards used were BBOT (a calibration standard, COSHTECH Analytical Ltd) and a high organic standard (b2150, Elemental Microanalysis Ltd.). For the purposes of quality control, laboratory standards and repeats were included in each run (approximately 30 samples per run). Calibration curves were based on a linear regression with an r^2 of 0.995 or better. Sample runs failed quality control if the AQC or high organic standard was outside 95% confidence level. If this was the case, the sample batch was re-run.

4.3.2.3 pTGA analysis

The pseudo-thermogravimetric approach uses the loss on ignition at a range of temperatures in order to assess the amount of black carbon that has been formed in the experimental process. The theory is that, while vegetation and soil carbon burn at temperatures less than 400°C, the opposite is true for pyrolysed and black carbon. This technique is a version of thermal oxidation technique often used to analyse the presence of black carbon in soil and sediment samples (Lopez-Capel et al., 2005). Batches of burn products from the burning experiments coupled with suitable standards are placed in crucibles and then heated from 105°C to 805°C in 50°C increments with each heating step lasting at least 4 hours and the samples reweighed after every step. Within each batch of experimental burn samples analysed by pTGA, duplicate samples of wood charcoal from a crushed and homogenized reserve were analysed in order to act as internal standard and as a check upon the method. In addition to samples of wood charcoal, samples of cellulose (ash-free filter papers), lignin (Aldrich) and fresh *Calluna vulgaris* were also analysed.

4.3.3 Statistical analysis

The experimental design as presented represents a complete factorial design and as such is analysed by analysis of variance (ANOVA) with five factors and moisture content as the covariate. Because the experimentation is considered in triplicate, it is possible to consider not only the significance of individual factors but also all possible two-way interactions between factors. The magnitude of the effects of each significant factor and interaction were calculated using the method of Howell (1996). Post-hoc testing of the results is made for pairwise comparisons between factor levels using the Tukey test in order to assess where significant differences lie between factor levels.

4.3.4 Modelling approaches

In order to answer the question of whether char production offsets the loss of biomass, the results from ANOVA are used to guide the development of equations for: loss of biomass; the percentage carbon (*%C*) of the burn products; and the percentage char produced (*%char*). These equations can then be combined to assess the possibility of carbon benefit in the products of a managed burn.

Equations derived for *%drymass*, *%C* and *%char* are applied across a range of feasible burn temperatures (400 to 800°C); a range of burn times (1 to 6 minutes); and at average moisture content (31%). Given equations for *%drymass*, *%*C and *%char*, the proportions of litter and char C in the remaining dry mass from the fire can be calculated. The proportion of litter and char C after a range of years (up to 50 years) are then calculated and compared to the amount of C that would have existed if there had been 100% biomass survival and 0% char production.

4.4 Results

In the initial run 486 samples were analysed for their total and dry mass loss in a fully factorial design

4.4.1 Total and dry mass loss

Dry mass loss of the vegetation types across all experiments ranged from 0 to 100%. The ANOVA of the burning of the vegetation samples show that all individual factors are significant at the 95% level except vegetation type although this factor is significant at a probability of 93% (Table 4.1).

Factor (or interaction)	df	p	Percentage of variation explained
Moisture content	1	0.00	9.6
Vegetation	2	0.07	0.1
Initial temperature	2	0.00	1.0
Burn temperature	2	0.00	57.3
Burn time	1	0.00	9.4
Return temp	2	0.01	0.2
Vegetation*initial temp.	4	0.101	0.1
Vegetation *burn temp.	4	0.00	2.2
Vegetation *burn time	2	0.03	0.2
Vegetation *return temp.	4	0.54	0.0
Initial temp.*burn temp.	4	0.00	4.9
Initial temp.*burn time	2	0.00	1.0
Initial temp.*return temp.	4	0.49	0.0
Burn temp.*burn time	2	0.00	6.5
Burn temp.*return temp.	2	0.17	0.1
Burn time*return temp.	2	0.99	0.0
Error	445		7.5

Table 4.1. The significance (probability of factor, interaction or covariate = 0) and percentage of the original variance explained for the percentage dry mass loss (*%dryloss*).

The vegetation factor is significant if the moisture content is not considered as a covariate, i.e. the differences between vegetation can be partially explained by the differences in moisture content between the vegetation types. *Sphagnum spp.* samples always had higher water content than the other vegetation types.

By far the most important individual factor is the burn temperature explaining 57.4% of the variation. Post-hoc testing shows significant differences between all burn temperatures but the biggest difference is between 400 and 600°C, but with an average dry mass loss of 84.2% at 800°C. The second most important single factor is the burn time (explaining 9.4%) of the variation in the dataset. The difference between a burn time of two and five minutes is an average dry mass loss of 27.6 to 59.7%.

The initial temperature of the sample between frozen and room temperature explains only 1% of the variation in the dataset but nevertheless is a significant factor. The post-hoc testing shows that a significant difference, at the 95% level, only lies between the -5°C and the 4 and 22°C factor levels but not between 4 and 22°C, i.e. the effect of initial temperature is an effect of freezing conditions. The difference between the frozen and room temperature conditions is an average difference of 10.6%, i.e. there would be 10.6% less dry mass loss if the burn took place on a frozen day rather than in summer conditions. The effect of the return temperature explains an even smaller proportion of the variation in the original dataset but nevertheless was significant. A significant effect due to the return temperature does imply there is a quenching effect and smouldering of vegetation does occur and does lead to continued mass loss. However, the effect is smaller than that due to the initial temperature of the sample and the post-hoc comparison shows that only the difference between -5°C and 22°C is significant. The difference between these two extremes is the difference of 4.5%, i.e. there would be 4.5% less mass loss if the burning took place on a day when the air temperature was below freezing than on a summer's day.

The most important interaction effect is that between the burn temperature and the burn time (6.5%) of the original variance. It is perhaps not surprising that there would be greater mass loss with vegetation exposed to higher temperatures for longer times. The significance shown to exist between these two factors shows that there is disproportionately higher dry mass loss in moving from a two-minute to a five-minute burn time at a burn temperature of 600°C than if this increase in burn time occurs at either 400 or 800°C. The second most important interaction effect is between the initial temperature of the sample and the burn temperature – explaining 4.9% of the original variance (Table 4.1). There is little difference between dry mass loss between initial temperature levels at 400 and 800°C; the biggest difference between levels of initial temperature temperatures occurred when the burn temperatures were 600°C. Initial temperature is also significant in interaction with the burn time although this explains only 1% of the original variance. There are no significant interactions with any other factor and the return temperature.

Although the importance of vegetation as a single factor is greatly reduced by the inclusion of moisture content as covariate, vegetation as a factor does significantly interact with the burn temperature and the burn time, the interaction with the temperature explains the most variance. *Calluna vulgaris* and *Eriophorum spp.* behave most like each other at 400 and 800°C with *Sphagnum spp*. being distinctive. However, at 600°C the *Calluna vulgaris* shows a distinctively higher dry mass loss. The interaction between burn time and vegetation shows that the largest difference between the two and five minute burn times exists for *Sphagnum spp*. and the smallest for *Eriophorum spp.*

The error term in the optimising model represents all the variance in the original dataset that is not explained by the factors and covariates chosen within the experimental design. In the ideal case it would represent only the proportion of the variance explained by the measurement error, i.e. the irreducible error due to performing the burning experiment. In this case the measurement error (7.5% of the original variance – Table 4.1) could be large due to the constraints upon placing samples in the furnace in an efficient, swift and repeatable manner. There could also be a sampling error involved in selecting samples of vegetation where it could, for example, easily be possible to select variable amounts of woody material within a sample of *Calluna vulgaris*.

4.4.2 Optimising Burning Conditions

For the purposes of this modelling exercise 126 samples were analysed by pTGA and 21 samples were analysed by CHN analysis. The best-fit regression for predicting the dry mass loss for each individual species was calculated.

For *Calluna vulgaris* the best-fit equation was:

$$\%drymassloss = 0.19T_B + 7.7t_B - 0.12\theta - 85.7 \qquad n = 191,\ r^2 = 0.72 \qquad \text{(Eq. 4.9)}$$

0.03 1.0 0.04 7.5

For *Sphagnum spp.,* the best-fit equation was:

$$\%drymassloss = 0.11T_B + T_B t_B - 0.7\theta \qquad n = 191,\ r^2 = 0.69 \qquad \text{(Eq. 4.10)}$$

0.06 0.001 0.2

For *Eriophorum spp.,* the best-fit equation was:

$$\%drymassloss = 0.18T_B + 5.1t_B - 0.46\theta - 65 \qquad n = 191,\ r^2 = 0.81 \qquad \text{(Eq. 4.11)}$$

0.008 0.8 0.04 6.0

where: T_B = temperature of the burn (°C); t_B = time of burning (minutes); θ = moisture content (%). Only those variables found to be significant at least at the 95% level are included and figures below the equations are the standard errors in the coefficients. Note that in equation 4.10 that the interaction term ($T_B t_B$) was significant whereas the constant term was not found to be significant

Pseudo-TGA (pTGA)

The analysis of standard materials showed that for fresh *Calluna vulgaris* all the mass loss had occurred before 405 °C, but for wood char samples the sharp decline in mass loss did not occur until 555°C. By comparing changes in mass loss at these two temperatures, a ratio can be formed where *Calluna vulgaris* Δ405: Δ555 = 1 while charcoal had an average Δ405: Δ555 = 0.239.

This distinct relationship reflects the fact that fresh vegetation mass losses occur at low temperatures relative to charcoal. Based on these average values for *Calluna* and char, it was possible to interpret each experimental burn sample as linear mix of these two organic matter types. Results show that the char composition of the experimental samples ranged from zero to 87% char and that burn temperature was the most important factor.

The main effects plot with respect to burn temperature suggests that char content rose steadily from 450°C. Therefore, it is possible to get a linear equation for %char produced, the best-fit equation was:

$\%char = 1.5\log_{10} T_B - 3.9$ n = 57, r^2 = 0.52 (Eq. 4.12)

0.2 0.5

CHN analysis

The CHN analysis was performed on 21 samples of *Calluna vulgaris* from across the range of temperatures and burn times. The ANOVA (Table 4.2) shows that both burn time and burn temperature were significant and, by using multiple regression to predict the *%C* of the burn products, the best-fit equation was:

$$\%C = 0.08T_B + 9.9t_B - 0.1T_Bt_B \qquad n = 21,\ r^2 = 0.85 \qquad \text{(Eq. 4.13)}$$

0.01 2.3 0.004

Here the interaction term is significant and has a negative effect upon the *%C*.

Factor	df	p	Percentage of variation explained
Burn temperature	2	0.00	50.0
Burn time	1	0.00	0.6
Initial temperature	2	0.00	36.7
Return temperature	2	0.00	2.0
Burn temperature * burn time	2	0.00	2.7
Error	11		8.0

Table 4.2. The significance (probability of factor, interaction or covariate = 0) and percentage of the original variance explained for CHN content.

Optimisation of burn conditions

It is possible to consider the question of whether there are conditions when the production of char by burning is greater than the loss of carbon within the burning. For the case of *Calluna vulgaris* this question becomes a trade off between equations 4.9, 4.12 and 4.13. The results of the stochastic modelling of these equations over time scales of 5 to 50 years shows that up to 10 years after the fire there is no advantage from the production of char during the burn and the amount of carbon remaining is in line with litter decomposition (Fig. 4.2).

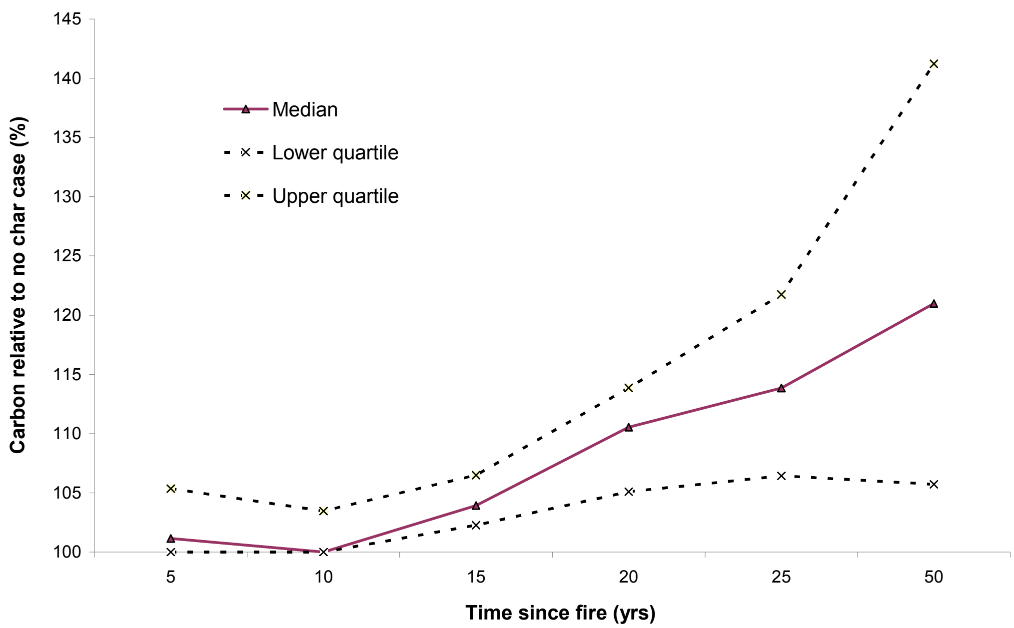

Fig. 4.1. The percentage present 5 to 50 years after the fire expressed relative to the amount that would be present for a fire with no char production

However, after 15 years there is a distinct advantage and by 50 years since the fire the amount of carbon present from the fire is 20% higher than if no fire had occurred. The median burn times under which these optimal conditions is never greater than 1 minute, but the median burn temperature at which optimal conditions occurred and would lead to a carbon benefit, i.e. at 15 years or greater after the fire, varied between 450 and 600°C.

It is possible to go a step further: Hobbs and Gimingham (1984b) have found a significant relationship between the vegetation height, in this case the height of the *Calluna vulgaris*, and the fire temperature

$$T = 11.1H + 12.8W - 3.6S \qquad \text{(Eq. 4.14)}$$

where: T = burn temperature (^{o}C); H= height of *Calluna vulgaris* (cm); W= width of the burn (m); and s = wind speed (ms^{-1}).

Unfortunately, no errors on this equation were cited. The height of *Calluna vulgaris* can be predicted from research on the growth rates of *Calluna vulgaris* for the study site (Hobbs and Gimingham, 1984b):

$$H = 4age - 0.08age^2 \qquad \text{(Eq. 4.15)}$$

where: *age* = age of *Calluna vulgaris* (years).

Therefore combining equations 4.14 and 4.15:

$$T = 44.4age - 0.89age^2 + 12.8W - 3.6S \qquad \text{(Eq. 4.16)}$$

From the above empirical results it is possible to suggest that burn temperatures should be between 450 and 600°C. From regulations within the UK the permitted widths of managed burns is between 10 and 30 m, the age of heather in burns is between 5 and 25 years and for this study site the interquartile range on the wind speed is 2.2 to 6.1 ms^{-1}. Given these ranges it is possible to examine the range of conditions under which the required burn temperatures would be achieved. This analysis shows that the required burn temperatures would be achieved for wind speeds between 2.2 and 6.1 ms^{-1}, for widths between 10 and 29.8 m; and for *Calluna vulgaris* ages between 5 and 15 years. That is to say that the window of opportunity of burn conditions is not sensitive to wind speed or burn width over the permitted and observed ranges but is sensitive to the age of the *Calluna vulgaris*. The experimental results suggest that it is fast burns nearer 600°C that will generate the maximum amount of char.

4.4.3 Field data

The dry bulk density of all samples ranged from 0.053 to 0.355 g cm^{-3}. Analysis of the results show increasing density with depth broadly consistent with other data reported from Moor House (Holden and Burt, 2003). The average carbon content across the samples was 49.2%, which is similar to reported values elsewhere (Immirzi et al., 1992), though the range of values was from 35% to 61%.

Table 4.3 shows carbon contained in peat above common horizons (50 cm and 20 cm) from both sampling campaigns. The carbon stored above SCP 'take off' depth identified in Garnett et al. (2000) is also presented along with results from that study. Using plots A3, A4, A6, B2, B5 and B6, a direct comparison between this study and previous work of Garnett et al. (2000) can be undertaken. This study shows slightly higher amounts of carbon storage in these plots than the previous work at this site. Although higher values are reported here, by plotting results of the two studies, it can be shown that the relative sizes of each plots are similar to Garnett et al. (2000) (Figure 4.2)

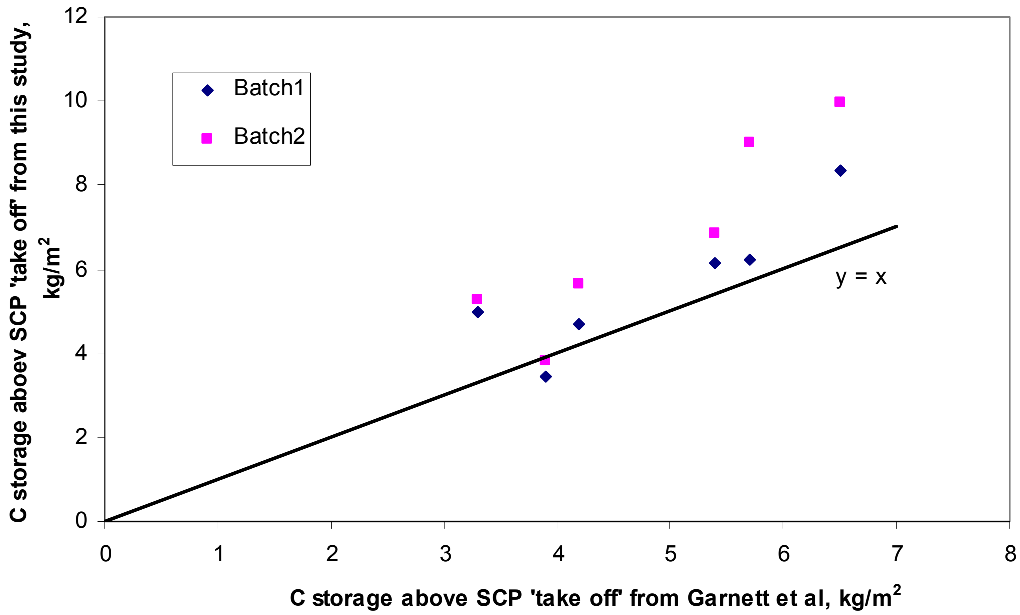

Figure 4.2. Comparison of results from this study and Garnett et al. (2000).

Plot Code	Management	C in peat above 50 cm (kg m^{-2})	C in peat above 20 cm (kg m^{-2})	C in peat above SCP depth (kg m^{-2})	Garnett et al., 2000: C in peat above SCP depth (kg m^{-2})
		Batch 1 - February 2006			
A1	No grazing, 20 year	30.8	12.1		
A2	No grazing, 10 year	36.5	14.6		
A3	No grazing, no burn	31.5	12.5	4.7	4.2
A4	Grazing, 10 year	30.9	12.5	3.5	3.9
A5	Grazing, 20 year	35.4	14.5		
A6	Grazing, no burn	39.1	13.5	8.4	6.5
B1	No grazing, 10 year	29.4	10.5		
B2	No grazing, no burn	33.9	13.0	6.2	5.4
B3	No grazing, 20 year	37.9	17.4		
B4	Grazing, 20 year	37.7	14.1		
B5	Grazing, 10 year	44.8	19.3	5.0	3.3
B6	Grazing, no burn	34.3	12.0	6.2	5.7
		Batch 2 - November 2008			
A1	No grazing, 20 year		13.2		
A2	No grazing, 10 year		15.7		
A3	No grazing, no burn		13.8	5.7	4.2
A4	Grazing, 10 year		14.4	3.8	3.9
A5	Grazing, 20 year		14.2		
A6	Grazing, no burn		15.9	10.0	6.5
B1	No grazing, 10 year		13.0		
B2	No grazing, no burn		13.8	6.9	5.4
B3	No grazing, 20 year		14.1		
B4	Grazing, 20 year		12.4		
B5	Grazing, 10 year		15.4	5.3	3.3
B6	Grazing, no burn		14.9	9.0	5.7

Table 4.3. Carbon in peat above different horizons at Hard Hill

By combining data from the two field campaigns, the average carbon storage above common horizons for each management regimes can be calculated. The average carbon storage above SCP 'take off' cannot be estimated for all management regimes so instead the average carbon stored in the top 20 cm is calculated.

Figures 4.3 and 4.4 show the average carbon content above 20 cm for the grazing and burning regimes.

Figure 4.3. Average carbon content above 20 cm for each grazing regime. Standard deviations are given.

Carbon storage shows no significant difference between grazing treatments though grazed plots have a slightly higher mean than ungrazed plots. This is result is similar to that seen in Garnett et al. (2000). The grazing intensity at Hard Hill may be too low to lead to a detectable difference in carbon storage.

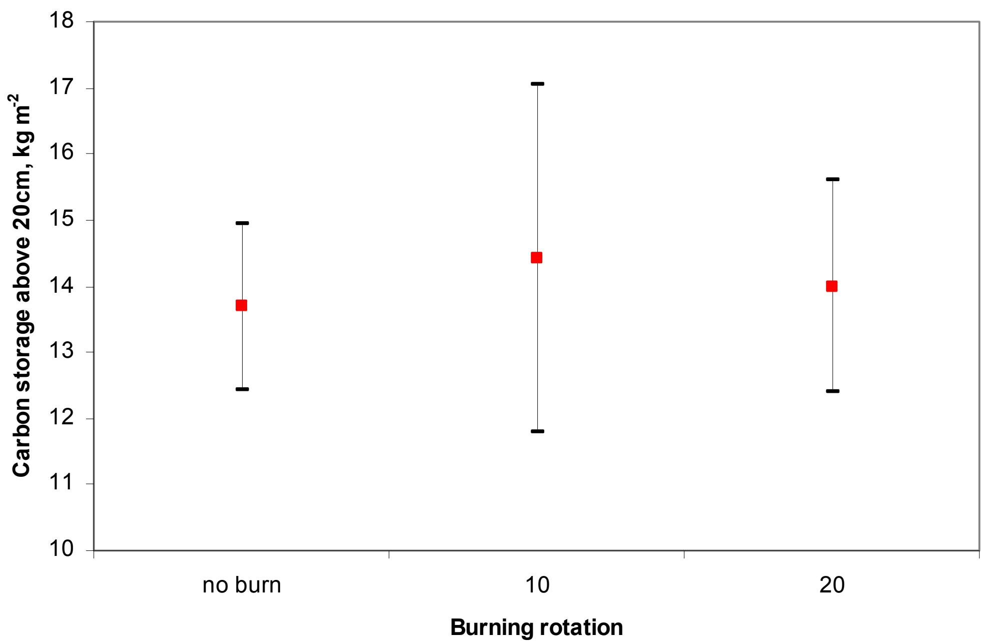

Figure 4.4. Average carbon content above 20 cm for each burning regime. Standard deviations are given.

When considering carbon storage under burning regimes, again no statistically significant difference occurs between burning regimes. Unlike Garnett et al. (2000) who showed a decrease in carbon storage under 10-year rotations compared to no burn, this study shows no difference between no-burn and 10-year rotations. In fact the burning plots (10-year and 20-year rotations) appear to have slightly higher carbon storage than no burn plots.

4.5 Discussion

The process of burning can never result in an increase in the carbon of the ecosystem at the time of the burn itself; the amount of carbon present in the biomass before the fire is the maximum that can be present in the remaining biomass or produced char. However, char produced in the fire will have almost zero turnover while the litter produced by the fire will turnover like any other litter (Latter et al., 1997). Therefore, the advantage of char production lies in the time after a fire when the litter will decompose but the char will not.

These results have provided evidence that the production of char during a fire can lead to greater carbon storage than if no char was produced. This occurs even when allowing for the greater biomass loss in order to achieve a greater proportion of char in the burn products. However, this result is not sufficient to say that some fires may lead to greater carbon storage than if no fire occurred.

As has been demonstrated, the production of char depends heavily on the meteorological conditions and fire intensity. Char, or black carbon, production also depends heavily on the parent material and the degree of woody material. Forbes et al. (2006) examined the black carbon production following fires in different ecosystems around the world. They found that, when expressed as a percentage of carbon consumed during the fire, black carbon production varied from 0.1% - 0.5% for Australian grassland to 10.5%

for coarse woody debris burnt in North American boreal forests. Moorland communities have a mix of species ranging from grasses and forbs through to woody shrubs suggesting that black carbon production may vary depending on the parent community. Of the studies examined in Forbes et al. (2006), few were of non-forest origin and none took place in Europe highlighting the need for further investigation in European shrublands and grasslands. Results in chapter 5 provide the first estimate of black carbon production following a moorland wildfire and suggest that black carbon is approximately 4% of the carbon consumed. This lies within the range of values given in Forbes et al. (2006).

The models predict an increase in carbon stocks through the production of char after approximately 15 years when the surviving char outweighs the litter input. However, burn characteristics are optimised to produce the maximum amount of char on rotations of 5 -15 years. Therefore longer rotations, 15 years, appear to be best for char production.

It is possible to theorise that this study could go one step further and propose that, if char production were sufficient, then it could be that some fires may lead to more carbon storage in the environment than if no fires occurred at all. In order to consider this, the loss of biomass during the fire and the reduction in litter production after the fire are outweighed by the survival of litter and char during the fire.

Recent work by Worrall et al. (Durham Carbon Model, unpublished data) has developed a model to explore the total ecosystem carbon stocks over time. This work expands on this work by including stocks in the soil, char and vegetation carbon pools. It investigates the question of where the carbon trade-off occurs between a 'business as normal' scenario (i.e. regular litter input into peat formation) and a fires scenarios (i.e. char input to peat but reduced litter input over the following years). In this work under longer rotations (>25 years) 6-7% more carbon was stored than if there had been no fires.

If burning, and specifically longer rotations, stores more carbon than if no burning had taken place, is there any evidence for this in the field? The answer to this is equivocal. Average carbon storage in the upper layers is slightly higher on burnt plots; however, this is not significant. This increase in carbon storage under burning may be reflecting increased char input in the peat as predicted by section 4.4. One might have expected to find the greatest carbon storage under the longest rotation; however, this was not the case.

When considering only those plots studied in Garnett et al. (2000), this study also shows a decrease in carbon storage under burning. However, by using this approach it is not fully factorial in that ungrazed, ungrazed plots were not included and as such the analysis does not use all the available data to calculate the effect of burning. Recalculating the data of Garnett et al. (2000) shows that the mean difference between burnt and unburnt treatments is 1.975 kg m^{-2} (not 2.3 as reported); this gives a mean effect of burning of 62 Mg C km^{-2} yr^{-1} (not 73 Mg C km^{-2} yr^{-1} as reported). Discrepancies between this study and the previous work could also come from different measures of carbon content. This study uses direct measurements of carbon content unlike Garnett et al. (2000) who assume a 50% carbon concentration of the dry mass.

This modelling study is limited to being a laboratory studied and although the study could build upon empirical field relationships (Hobbs and Gimingham, 1984b), it could not reflect the variety of field behaviour. Variations in wind speed and topography across the burn area are likely to affect the range of temperatures experienced during the fire (Pyne et al., 1996) and could therefore lead to spatial variations in char production even within a single burn. Future modelling work on this topic will need to include a 'risk' factor to allow for spatial distribution of 'hot' and 'cool' spots.

One methodological problem of this study uses cores taken from either side of the managed burn in February 2007. The cores taken in November 2008 are over 18 months older than the first set of cores and therefore there is likely to be additional carbon accumulation in the surface layer. This is unlikely to be a large error as peat accumulation shows growth of around 1 or 2 mm yr^{-1} (Borren et al., 2004; Ukonmaanaho et al., 2006). Whilst this equates to a 10% error for the surface sample (0 – 2 cm), its overall contribution to the carbon stocks in the upper 20 cm should be minimal.

An additional problem arises on the 10-year plots where the addition of char from the managed burn may affect the carbon content and therefore storage in the upper layers. Analysis of the carbon content and carbon storage in the upper two centimetres before and after the fire across the 10-year cores showed no significant difference at the 95% level, suggesting char input was not large enough to affect carbon stocks in the cores on a short timescale.

4.6 Conclusions

One question posed by this study is, how are these results likely to affect management decisions in the future? This study has shown that managed moorland burning may be able enhance carbon accumulation under certain conditions. The conditions suggested by this study (e.g. fast burns at 600°C) will be hard to create precisely in the field; however, simple steps can be taken to maximise the char production from each burns. The choice of burning day is important as cold conditions have been shown to reduce the dry mass loss though both initial moisture content and a post-fire quenching effect.

The modelling results show that the optimal window for burning is not sensitive to burn width. This suggests that current guidelines (DEFRA, 2007a), which suggest burn strips are no more than 30m, are appropriate for char production. The model also shows that char production is not sensitive to wind speed up to 6.1ms^{-1}. Current guidelines suggest that burning is most effective and easily controlled in wind speeds between 3.4 ms^{-1} and 5.4ms^{-1} (DEFRA, 2007a; SEERAD, 2001a) and advise avoid burning in winds greater than 6.7ms^{-1} (SEERAD, 2001b). For the purpose of char production, current best practice for burning techniques is suitable.

The modelling results from this study suggest that burning on a timescale exceeding 15 years may be best for char production and therefore carbon accumulation. In order to keep the heather from becoming too tall for grouse to feed on, land managers will aim to burn heather once it reaches about 30 cm (Watson and Miller, 1976). A 15-year cycle is within the range of suggested rotations lengths at which heather reaches this height (Tucker, 2003; Watson and Miller, 1976). This study suggests that overall, current practice is suitable for char production and suggest that benefits for carbon accumulation may occur.

Chapter 5:

Black carbon production and spatial variability of the Grindsbrook Wildfire, May 2008

5.1 Introduction

Wildfires are a common occurrence in the UK (McMorrow et al., 2009) and can have important effects on different ecosystem services in upland landscapes. The ecological impact of fire is particularly important as it has the potential to be very harmful to habitats (Legg et al., 1992; Maltby et al., 1990) and their associated fauna e.g. ground-nesting birds. Wildfire can also lead to a long-term change in the ecology of an area. In a wildfire in the Derbyshire Dales, the destruction of the vegetation and humus layer led to change from acid grassland to limestone grassland (Grime, 1963). The direct physical effects of wildfire have also been investigated. Vegetation cover plays an important role in binding the peat surface together. If the vegetation layer is weakened by fire, physical erosion of the peat may occur (Evans, 2009). Erosion following wildfires could have important implication for the release of heavy metals from the peat surface to the fluvial system (Rothwell et al., 2007).

Wildfire also has an important impact on carbon stores and fluxes. Furthermore, the fire can burn through the above ground biomass and burn the litter layer and the soil organic matter below. In addition to the carbon that is lost through combustion, fire could lead to enhanced export of carbon via a range of pathways, e.g. post-fire erosion could be enhanced due to the loss of vegetation leading enhanced losses particulate and dissolved organic carbon from peat to streams (Evans, 2009). With the peatlands of the UK storing around 3 billion tonnes of carbon (Cannell et al., 1993), the consequence of wildfire and post-fire erosion could have important impacts on this nationally, and internationally important carbon store. Many studies consider the impact of managed burning on carbon (e.g. Garnett et al., 2000; Ward et al., 2007) though few studies have been carried out for carbon fluxes following wildfires in the UK. There are no papers that considered the carbon balance of a moorland fire itself, be that for either a managed fire or wildfire.

During the process of combustion, carbon is released to the atmosphere in the form of various gases and particulates with most of the carbon in the form of CO_2 (Lobert et al., 1991). However, depending on fire conditions, a percentage of the original biomass is converted to charred products and remains on the site. These charred materials, often referred to as black carbon (BC), are the product of incomplete combustion of vegetation and fossil fuels. Novakov (1984) defines BC as “combustion-produced black particulate carbon having a graphitic microstructure”. In the field, however, BC can be thought of as a continuum of products ranging from slightly charred degradable biomass through to highly graphitized soot spheroids

(Hedges et al., 2000; Masiello, 2004). The interest in BC primarily comes from its importance to the global carbon cycle and its potential role as a carbon sink (Kuhlbusch, 1998). Due to its long mean residence time, often on the millennial time scale (Lehmann et al., 2008), and its high degree of resistance to chemical agents (Bird and Gröcke, 1997), BC may have the potential to remove significant amounts of carbon from the short-term bio-atmospheric system and transfer it to the longer geological carbon cycle. Thus BC production could help mitigate the losses of carbon during the fire itself.

Forbes et al. (2006) discuss the many problems associated with the definition and quantification of BC and how it is expressed relative to other components of the carbon cycle. They propose a standard way to express BC by expressing it as a percentage of the amount of carbon consumed by the fire (BC/CC). By using this method, BC formation in forest fires ranged from 5% to <3% BC/CC and in savannah and grassland fires a value of <3% BC/CC is common. The studies included in Forbes et al. (2006) are from a narrow range of ecosystems, and of the grassland and savannah studies included, two were from Africa and four were from Australia. This work is the first to estimate the carbon budget and the BC production of wildfire in a European moorland setting.

Any large-scale heterogeneity in black carbon production is likely to be related to variations in processes and characteristics of the fire. This gives rise to areas of intensely burnt "hot spots" and area of relatively little damage, "cool spots". In order to help target fire suppression during fires and also to help direct post-fire restoration work it is important to know where these hot spots occur.

The nature of large fires has led to the development of many techniques to remotely determine fuel and fire characteristics. Active thermal monitoring during a fire has been used to locate active fire fronts (Roy et al., 1999) and to estimate energy release during burning (Wooster et al., 2003). Spectral reflectance measurements following a fire are often compared to pre-burn measurements to give an estimate of the areas burnt. These techniques often require the equipment to be available at short notice e.g. aircraft, or are expensive to acquire data e.g. satellite time. Few studies have looked at estimating fire characteristics from post-burn products (Smith et al., 2005). Those that have investigated the post-burn products have often used field spectroscopy techniques that can often be affected by noise from atmospheric water and require later corrections.

Lab based measurements, such as near-infrared reflectance spectroscopy (NIRS), offer an alternative technique to field methods. Using NIRS offers a rapid, non-destructive method that requires little or no sample preparation

(Guerrero et al., 2007). While mid-infrared reflectance spectroscopy (MIRS) is often associated with more intense vibration fundamentals (Ludwig et al., 2008) MIRS often requires additional preparation (Guerrero et al., 2007), and work has shown that overtones present in the near-infrared can confidently predict soil properties (Ben-Dor and Banin, 1995; Fritze et al., 1994). NIRS has a wide range of uses as it can provide repeatable measurements of chemical constituents in organic materials (Norris et al., 1976). The application of NIRS in soil science is wide ranging: litter decomposability (Gillon et al., 1999); palaeo-ecological studies (McTiernan et al., 1998); soil carbon distribution (Barthès et al., 2008); soil fertility (Du and Zhou, 2009); forage quality (Norris et al., 1976); hydrocarbon pollution in soils (Schwartz et al., 2009).

Few studies have used lab-based NIRS in predicting fire characteristics. Guerrero et al. (2007) use NIRS as a method to accurately estimate the maximum temperature reached on burned soils. They found that the spectra produced are independent of heating duration; however, the minimum duration samples were exposed to was 10 minutes which is greater than most fire residence times in moorland fires (SEERAD, 2001a). Work presented in this chapter will investigate the use of NIRS as a method to constrain fire severity across the Grindsbrook wildfire. Studying a managed burn would have been ideal; however, by studying this wildfire important lessons can be learnt.

5.2 Chapter Objectives

The objectives of this chapter are two fold:

- Survey and analyse above ground biomass loss and black carbon production from a UK moorland wildfire;
- Investigate spatial variability in fire severity using several analytical methods including near-infrared reflectance spectroscopy.

5.3 Materials and Methods

5.3.1 Field Survey

On 26th May 2008 a wildfire was reported on moorland near Edale, Peak District, UK (Figure 5.1; UK Grid Ref: SK 104 873). The fire burnt for three days and covered an area of approximately 10 ha, crossing several major gullies and numerous small gullies. The fire was attended by the Fire and Rescue Services, Peak District National Park rangers, National Trust rangers, and a helicopter for transport of personnel and water bombing. A police helicopter was on attendance and an RAF helicopter was on standby. It was brought under control on the 29th May.

The surrounding vegetation is dominated by bilberry (*Vaccinium myrtillus*), heather (*Calluna vulgaris)*, and cotton grasses (*Eriophorum vaginatum and Eriophorum angustifolium*) with areas of *Sphagnum spp*. The area is one of deep peat soils (organic layer greater than 50cm depth) and are underlain by gritstones and shales of the Millstone Grit series that are exposed as the

Kinder Scout Grits and underlying Grindslow shales (Gluyas and Bowman, 1997; Stevenson and Gaunt, 1971)

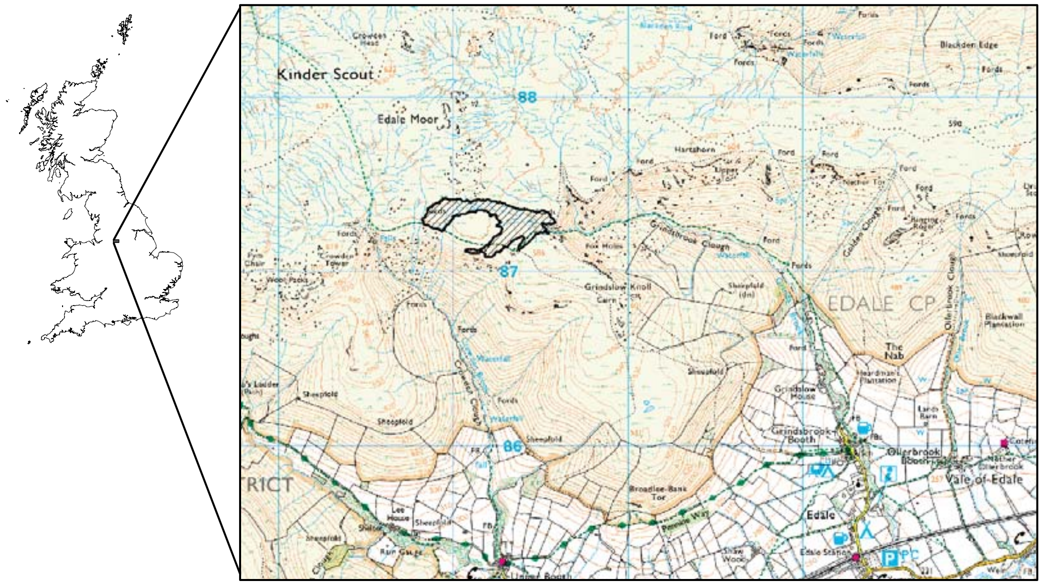

Figure 5.1. Location of fire scar (hashed area). © Crown Copyright Ordnance Survey 2009. An Ordnance Survey/EDINA supplied service

In order to assess the loss of biomass and the production of char, a survey of the burnt area was carried out three weeks after the fire (16^{th}-19^{th} June 2008). The primary field data were gathered through a series of $0.5m^2$ quadrat surveys in the fire scar and surrounding unburnt vegetation. An obvious limitation of studies of wildfires is that it impossible to know where the wildfire will occur prior to happening and thus it has to be assumed that the unburnt area surrounding the fire represented the area of the burn before the fire. The survey was conducted in a semi-stratified manner in order that the following key regions were surveyed: fire length; fire width; a back burn area i.e. burnt against the principal wind direction; and spur from the main fire.

A total of 65 quadrats (42 burnt, 23 unburnt) were surveyed. At each quadrat spot location various data were collected including: GPS location, altitude, vegetation types and cover, and vegetation heights. Samples of vegetation, litter and char were removed from each quadrat whenever possible for later laboratory analysis. Vegetation was clipped to peat surface to remove representative samples for later analysis. Litter and char were removed by scraping areas clean within the quadrat. Samples were placed in sealed bags in the field and stored in air-tight desiccating chambers prior to lab analysis.

5.3.2 Laboratory Analysis

5.3.2.1 Carbon analysis

Samples were dried at 70°C for 48 hours before being ground and homogenised. Large roots and plant matter were removed from the soil samples and litter and vegetation samples were cut to suitable sizes before analysis. Samples were stored in an airtight desiccating chamber between preparation and analysis.

Samples are analysed for their carbon, hydrogen and nitrogen (CHN) content on a Costech ECS 4010 Elemental combustion system with pneumatic autosampler in the Department of Geography, Durham University. Further details of the methods and quality control can be found in section 4.3.2.2.

5.3.2.2 Near-infrared reflectance spectroscopy (NIRS)

To examine the field samples in more details, near-infrared reflectance spectroscopy was used. Samples of soil and char were crushed to increase the degree of homogeneity within the sample and to allow the material to fit in the sample cell. Vegetation samples were cut so that they would also fit into the cell.

Spectral reflectance measurements were made on a Varian spectrometer (Cary 5E Varian UV-Vis) fitted with a Praying Mantis Diffuse Reflectance Accessory (Figure 5.2)

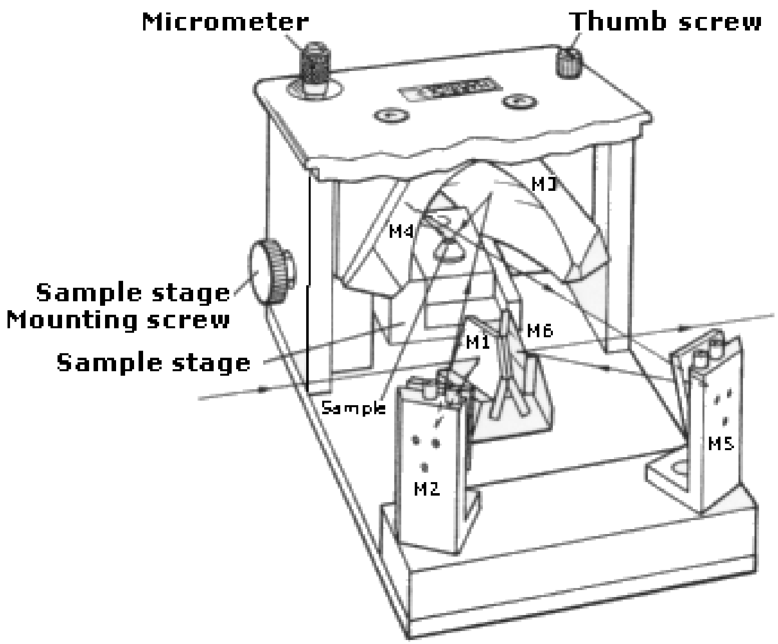

Figure 5.2. Praying mantis accessory.

The instrument was set up for reflectance readings. Each sample was scanned over the range of 250 nm – 3000 nm at 1nm intervals at a rate of 900 nm min^{-1}. In each batch of samples repeats and control samples were included for quality control purposes.

To reduce the noise and help identify absorbance features, spectral data were post-processed by applying a 3rd order polynomial using the Savitsky-Golay method with a 9nm moving window (Vasques et al., 2008). To identify peaks in the spectra, the Peak Analyser function in OriginPro 8 (OriginLab) was used and local maxima and minima were calculated.

In addition to the char samples from the Grindsbrook wildfire, a series of char samples from controlled experimental burns (Chapter 4) was also analysed using NIRS. A range of samples had been retained from previous experiments; *Calluna* was used as the experimental material in this section as it was decided that this fitted the general vegetation cover at the field site.

To replicate a range of burning conditions, samples of *Calluna* were treated in a factorial design. The factors considered were:

i) Burning temperature – Initially 3 burning temperatures were chosen – 400, 600 and 800°C and later extended to include 450, 500, 550, 650, and 700°C

ii) Burning time – the samples were exposed to two different burn times – 2 and 5 minutes.

For the purposes of this study, the start and return temperature of the material was room temperature (~22°C).

5.3.3 Biomass and carbon stock calculations

An estimation of the pre-burn biomass was calculated from the surrounding unburnt vegetation. The total biomass in each quadrat was calculated by multiplying the percentage cover of each species by its published biomass value. Biomass values for *Sphagnum spp.*, *Eriophorum spp.* and *Calluna vulgaris* come from work done at Moor House in the North Pennines

(Forrest, 1971). However, no values are available for *Vaccinium spp.* and so this study considers this ‘shrub’ to be equivalent to *Calluna vulgaris.*

$$\text{Biomass} = \frac{1}{n}\sum_{i=1}^{n} Co \times B \qquad \text{(Eq. 5.1)}$$

where *Co* is percentage cover for each species, and *B* is its published biomass value (g m^{-2})

The carbon content of each vegetation type, as measured by CHN analysis, was used to calculate the pre-burn above-ground carbon stock in each quadrat (Eq. 5.2)

$$\text{Carbon stock} = \frac{1}{n}\sum_{i=1}^{n} Co \times B \times C \qquad \text{(Eq. 5.2)}$$

where *C* is the carbon content of the material

The mass and carbon stock of the post-burn products were calculated in the same fashion to pre-burn biomass with the addition of char and stick data.

Char samples from each quadrat were analysed for their carbon content and based upon collection of samples from the field, a weight per unit area for char was conservatively estimated to be approximately 25 g m^{-2}. This was estimated by scraping all the char from a known area (10cm × 10 cm quadrat cell).

Burnt stick was defined as upright woody stems from shrubs that had been burnt but were still attached to the soil. The biomass for stick was calculated by modelling the stick as a cylinder with uniform density. To calculate the biomass of stick using this approach, two pieces of information were needed: a relationship between height and mass of stick; and the average number of sticks per unit area. A relationship between height and mass was determined from laboratory measurements of sticks taken from the field. The average number of sticks per m^2 was derived from field records and photographs. Using this, it is then possible to calculate an average biomass for stick (Eq. 5.3):

Biomass = (Stick height × 0.0265 – 0.9758) × average stick density (Eq. 5.3)

where stick height is in mm and average stick density was 88 sticks m^{-2}. Using an average carbon content of *Calluna* stems of ~ 52% (Lageard et al., 2005) the carbon stock in the sticks could be calculated.

In order to enable a comparison with other studies, black carbon is expressed as a percentage of carbon consumed (Forbes et al., 2006) where black carbon is defined, in this study, as the char and charred stick left on the site following the wildfire. Black carbon production can be calculated for each quadrat by using equation 5.4:

$$BC/CC = \frac{BCFraction}{(PreBurn) - (PostBurn)} \quad \text{(Eq. 5.4)}$$

where: BCFraction is the mass of carbon in the BC fraction post burn (i.e. char and stick) in each quadrat (gC m^{-2}); PreBurn is the pre-burn estimation of carbon for the site calculated from Eq. 5.2 (gC m^{-2}); PostBurn is mass of post burn carbon in each quadrat (gC m^{-2}).

5.3.4 Extent of wildfire

In order to understand the wider importance of any of the biomass loss or BC production estimated in the study, it is important to consider how common and extensive wildfires and managed burning is within English peat uplands. Reports compiled by Peak District National Park rangers upon the timing and size of wildfires have been recorded since 1976 and data have been extracted from these reports.

5.4 Results

5.4.1 Black carbon results

5.4.1.1 Field and Laboratory results

Figure 5.3 shows the average ground cover of the site by burnt and unburnt status. Shrubs and grasses occupy approximately 46% and 33% respectively in unburnt sections and in burnt stands char and exposed soil occupy similar proportions. Moss occupies similar amounts of area (5%), in both burnt and unburnt areas suggesting moss was little affected by this fire. Mosses have higher water content than other vegetation types and at low fire temperatures moss merely dries out rather than becoming burnt.

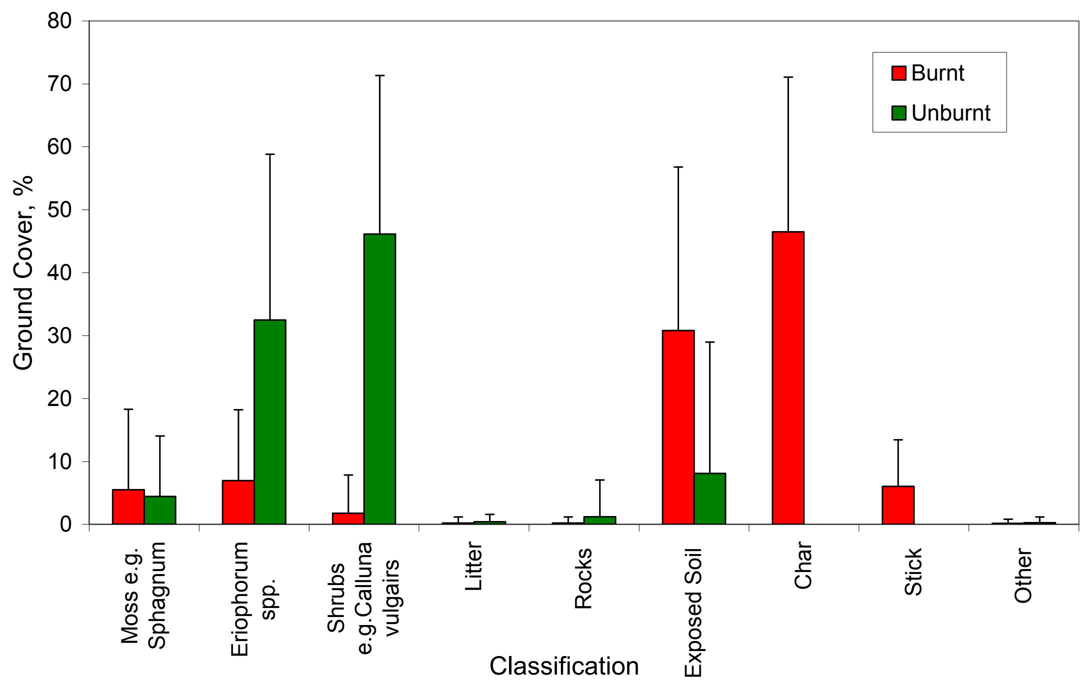

Figure 5.3. Distribution of ground cover for field classes used during initial survey (16th-18th June 2008). ± 1 standard deviation shown.

Variations in carbon content occurred between sample types and also within each sample type (Figure 5.4). Char samples displayed the greatest range of carbon contents ranging from 30% to over 70% carbon content. Litter and vegetation samples display similar carbon contents that are approximately 45%. These vegetation carbon content values are similar to others published in the literature: *Eriophorum vaginatum – 43.5% to 45.3%* (Thormann and Bayley, 1997); *Sphagnum capillifolium – 43.08* ± *0.13 %* (Vingiani et al., 2004); *Sphagnum spp.* – 46.6 ± *0.3 %* (Hall and Louis, 2004); *Calluna vulgaris* – 52-54% (Lageard et al., 2005).

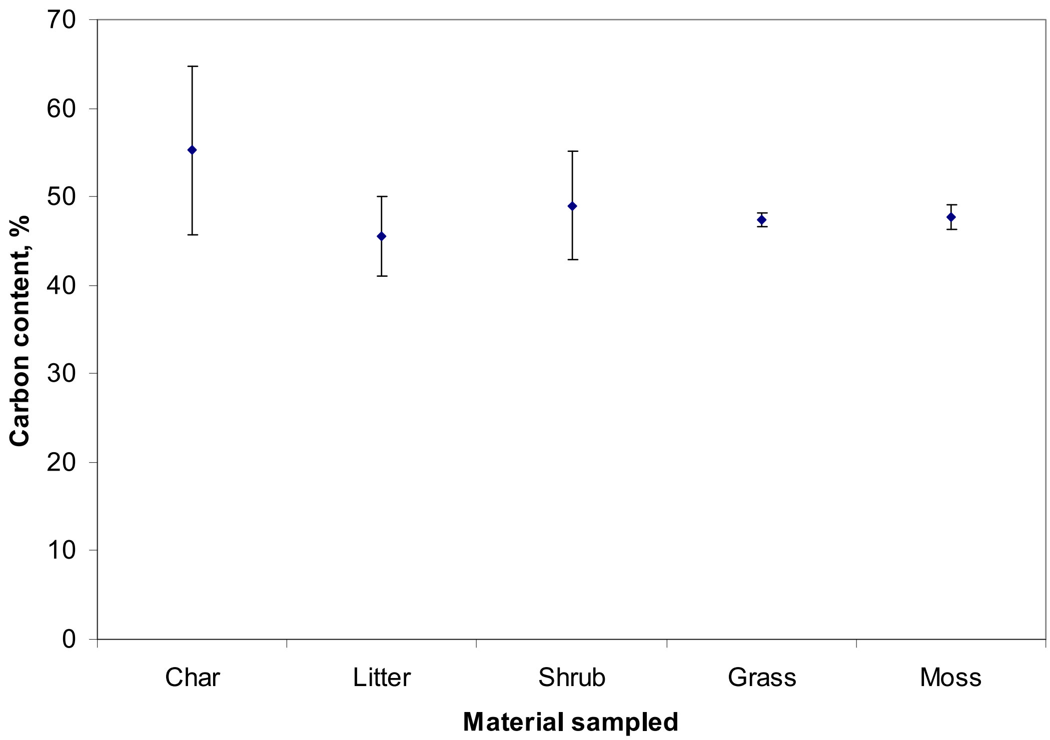

Figure 5.4. Distribution of measured carbon contents of sampled materials (± 1 standard deviation).

5.4.1.2 Carbon losses and black carbon production

The mean pre-burn biomass was 566 ± 232 g m^{-2} and pre-burn carbon was 230 ± 91 gC m^{-2}. The biomass value is within the range of values reported for other heathland settings though it is at the lower end of these estimates (Forrest, 1971; Gimingham, 1972).

The mean mass of post-burn products was 61 g m^{-2} with an average carbon stock in the post-burn products of 28 gC m^{-2} (Table 2). Using these estimates of pre-burn and post-burn products, the total biomass loss is 89% and carbon losses are 88%. Unburnt biomass was the largest component of post-burn products with BC contributing around 24% of the total mass or 7.87gC m^{-2}. BC contributes to approximately 28% of post-burn carbon.

	Pre-Burn Biomass g m^{-2}	Post Burn Products g m^{-2}	Post burn products - components Biomass g m^{-2}	Char g m^{-2}	Stick g m^{-2}
Biomass	566.23	61.67	47.13	11.62	2.92
	Pre-Burn Carbon gC m^{-2}	Post Burn Carbon gC m^{-2}	Post burn products - components Biomass - carbon gC m^{-2}	Char - carbon gC m^{-2}	Stick – carbon gC m^{-2}
Carbon stock	230.35	28.60	20.73	6.35	1.52

Table 5.1. Mean pre-burn and post-burn biomass and carbon stocks with individual components given

An estimate of black carbon produced during the fire was calculated for each quadrat to enable variations in BC production to be investigated (Eq. 5.4). Mean black carbon production (BC/CC) for this fire is 3.98 ± 1.99%.

5.4.1.3 Extent and distribution of wildfires

Given the extent of biomass survival and BC production from this wildfire, it is then important to ask how important is this additional carbon input compared to the loss during the burn itself? Furthermore, how important is this carbon input across a region?

The ranger reports suggests that there were 341 wildfires in the National Park between 1976 and 2004, varying from 81 in 1976 and zero in 1979, with an average of 12 per year. The size distribution of the fires is log normal and so the geometric mean wildfire size is 670 m^2, but the fires range in size from 1 m^2 to the largest that was 5.5 km^2. The total area under wildfire each year averaged 1.2km^2 with a maximum of 5.5 km^2. Alternatively, Worrall et al. (2009b) have surveyed the National Park and found 186 km^2 within an area of 725 km^2 showed evidence of managed burning. If it is assumed that burn cycle in the area is between 10 and 20 years, then between 9.3 and 18.6 km^2 of land in the park are burnt each year – effectively 3 times the area of wildfire even in the worst year. Furthermore, of the 341 reported wildfires, 41 have an attributed cause and of those 41, 10 have been attributed to managed burns, i.e. a little under 25% of wildfires are due to managed burning. However, when the area of the wildfires is considered, of the 41 fires with an assigned cause, those due to managed burns represented 51% of the burnt area, i.e. fires from managed burns appear to have been bigger when they did occur.

5.4.2 NIRS results and fire severity

5.4.2.1 Overall spectral shape

Initial analysis of the spectra showed that the spectra were somewhat consistent with previous work (Smith et al., 2005; Vasques et al., 2008) and a number of common features were observed, e.g. water absorption peaks (Figure 5.5). However, the amplitude and overall percentage reflectance was lower than in published works. This may be due to various reasons including problems with sampling, preparation, or instrumental. There is also the question of differences in study site compared to published work. This work deals with a moorland fire on a peat soil whereas many other fires deal with dry Mediterranean-type settings or American forest.

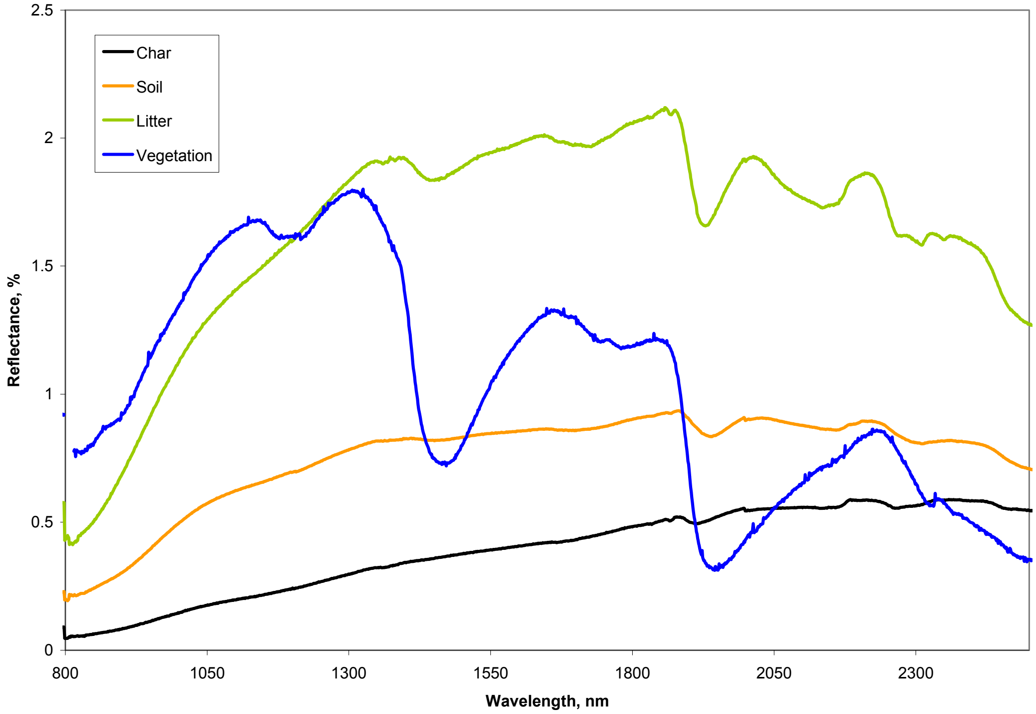

Figure 5.5. Sample spectra (averages of total spectra)

Qualitative description of spectra

Vegetation and litter samples have a greater number and more pronounced absorption features than soil or char. These are related to different structures within the plant material e.g. water, lignin and cellulose. In turn the spectra for soils show more spectral features than the char spectra such as a defined water absorption feature at around 1900 nm (Figure 5.5). The char spectra have some features but their amplitude is greatly reduced.

Quantitative description of spectra

Char samples are on the whole featureless; black ash is typically spectrally featureless (Smith et al., 2005). Char has a low degree of reflectance in the range below 2000 nm and reflectance continues to decrease below this point. Conversely, soil spectral reflectance maintains a level from around 2400 nm until 1400 nm wherein it also decreases continually. By assigning key points to the spectra it is possible to define the shape of the spectra. The portion from 2400 nm to 1390 nm is the bulk of the change in reflectance and is labelled α on figure 5.6.

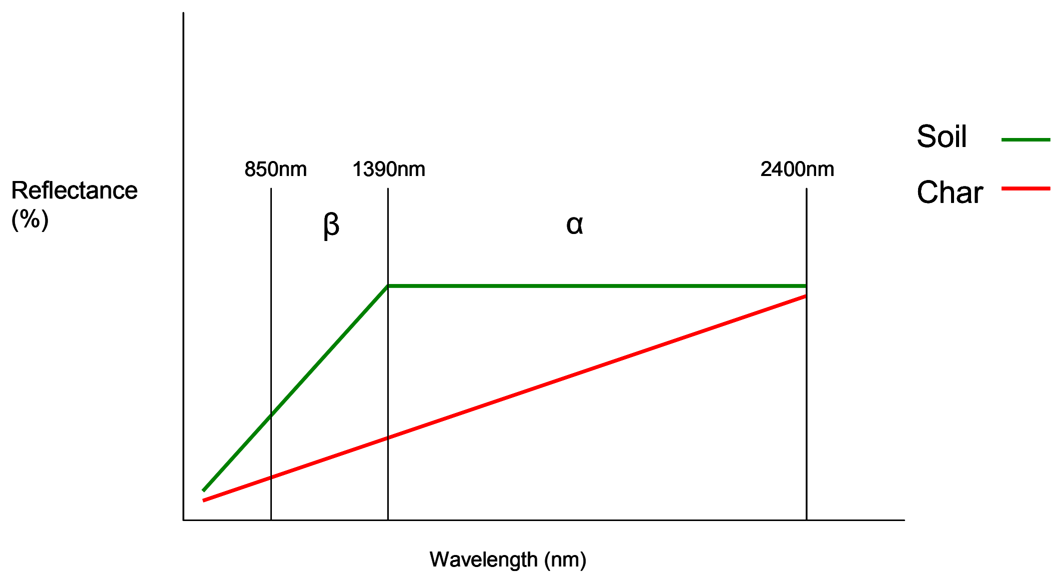

Figure 5.6. Simplified view of char and soil spectra with key point labelled.

The portion from 1390 nm to 850 nm is the end of the slope and is known as β. The gradient of β in this simplified view is non-zero and α is zero. In the char samples the gradient of α and β are essentially the same and are typically less than β in the soil samples. These definitions have little use by themselves so by using ratio of α and β, a number can be calculated to describe the overall shape. If α / β ratio equals one, then the spectrum has the same gradient throughout and is not horizontal; this is indicative of a char sample. If, however, the value is much higher than one, the sample is more soil-like in character where the large gradient of the α section of the soil spectra and a low gradient of the β section leading to this large value (Table 5.2).

	Average ratio	Maximum ratio
Char	1.93	9.28
Soil	3.63	230

Table 5.2. Average and maximum shape factors

By using these defined gradients and the overall shape it is possible to discriminate between sample types by their spectral characteristics. Figure 5.7 plots the gradients of α and β of the sample types from Grindsbrook including litter samples.

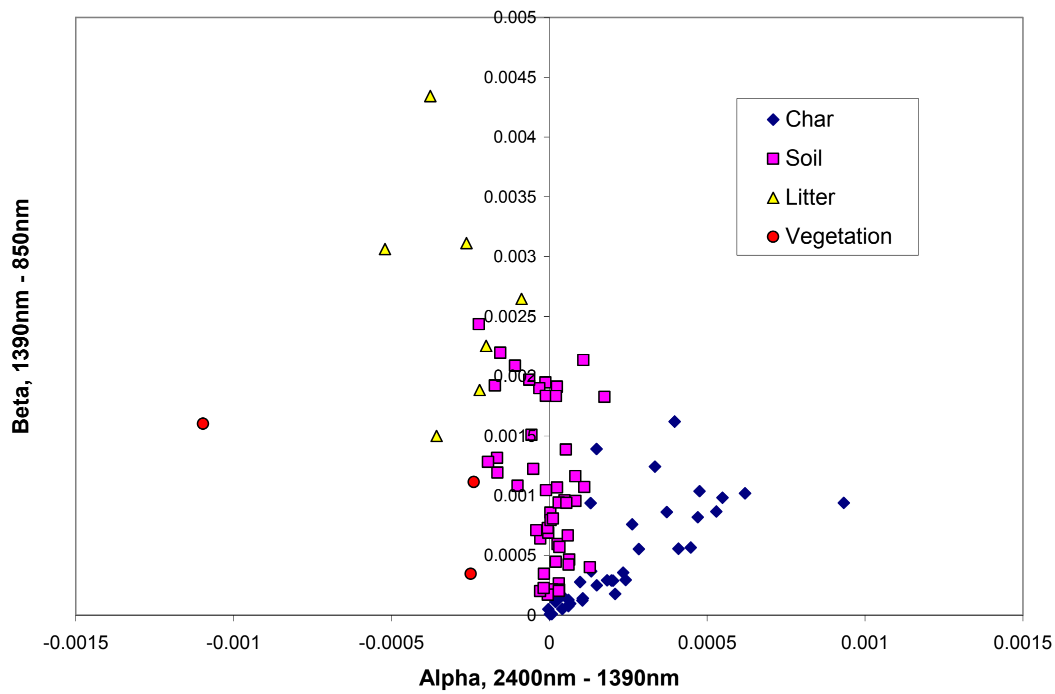

Figure 5.7. Plot of α vs. β showing clustering of sample types

The data plot in district clusters. Fresh vegetation and litter plot in the negative area of the x-axis due a positive gradient from 2400 nm and 1390 nm. Soil samples plot around the y-axis, which supports the initial descriptions of a flat section from 2400 nm to 1300 nm. Char samples plot close to, but slightly above, a 1:1 line. These changes in gradient are likely to reflect changes in organic compounds that occur during decomposition. As vegetation and litter become more homogeneous through peat formation, β rotates to a level gradient. During the process of heating vegetation (or litter), organic compounds are destroyed or altered leading to a further flattening of the spectra.

5.4.2.2 Absorbance features

By looking at the shape of key absorbance features it may be possible to calculate the temperature at which the sample was formed. Char samples from the experimental burns (Chapter 4) were analysed using the same method at the samples from the wildfire. One of the clearest features in the spectra was the feature close to 1900 nm.

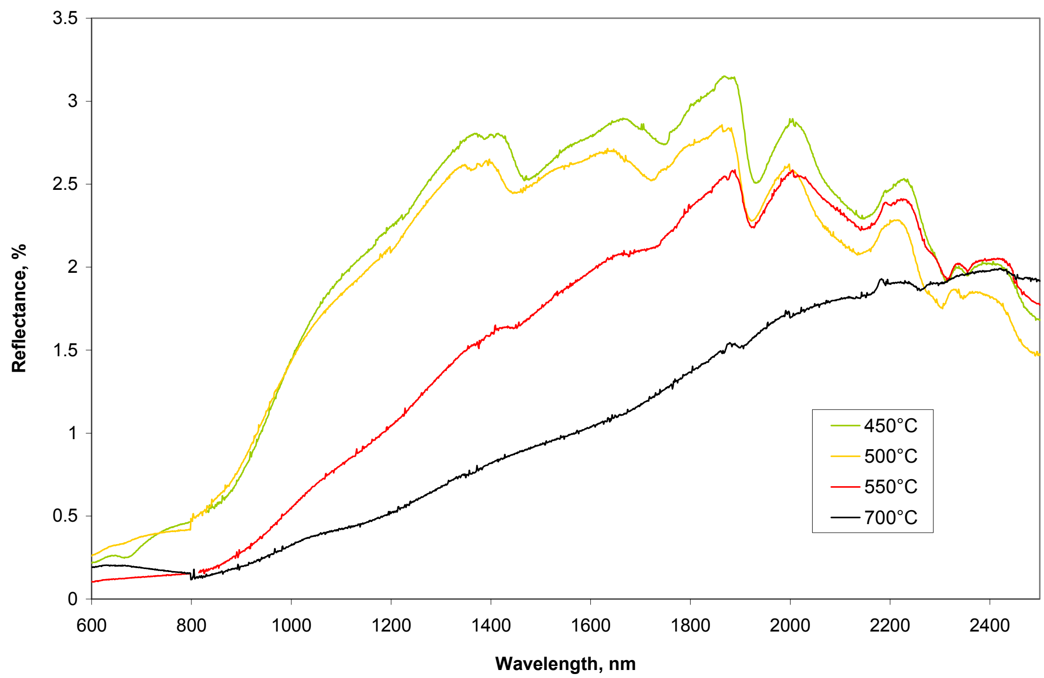

Figure 5.8. Example spectra from burning experiments

This feature, which has two distinct peaks either side of a deep trough, was characterised by measuring the different dimensions of the peak e.g. height, width. Samples burnt at 5 minutes did not show clear features, possibly due to degree of alteration over time therefore due to the integrated nature of burn temperature and burn duration (Smith et al., 2005) it was decided to focus on the samples burnt at 2 minutes duration. Figure 5.9 shows the measurements taken.

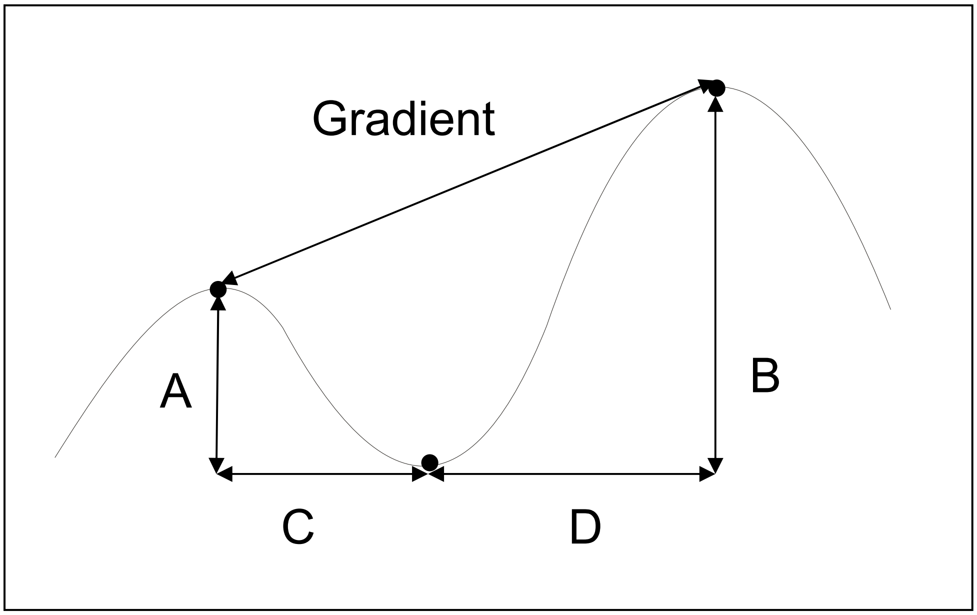

Figure 5.9. Measurements chosen to parameterise the 1900nm feature.

Using these parameters, regression equations could be derived using the known temperature of the experimental burn. Table 5.3 shows that not all parameters were good estimators of burn temperature; however, there are three measurements that resulted in a high r^2 with linear regression. Other curves were fitted, where appropriate, and improvements to the r^2 can be seen in table 5.3.

Measurement	Linear regression r^2	Significant at 95% level	Other fitting curves	Best fit r^2
A	0.62	No		
B	0.26	No		
C	0.90	Yes	2^{nd} order Polynomial	0.99
D	0.40	No		
Gradient	0.83	Yes	2^{nd} order Polynomial	0.98
A/B	0.81	Yes	Boltzmann curve	0.95

Table 5.3. Parameters used to predict burn temperature

The best fit curve is the second order polynomial with measurement "C" given by equation 5.5

$$Temperature = 8.66X - 0.32X^2 + 641 \qquad \text{(Eq. 5.5)}$$

where X is the measurement "C" in nm.

By using equation 5.3 to back calculate temperatures, average temperatures of the Grindsbrook wildfire was 597 ± 100°C. The range was from 258°C to 700°C which is in range of reported values (Hamilton, 2000; Whittaker, 1961). Figure 5.10 shows the spatial distribution of these temperatures.

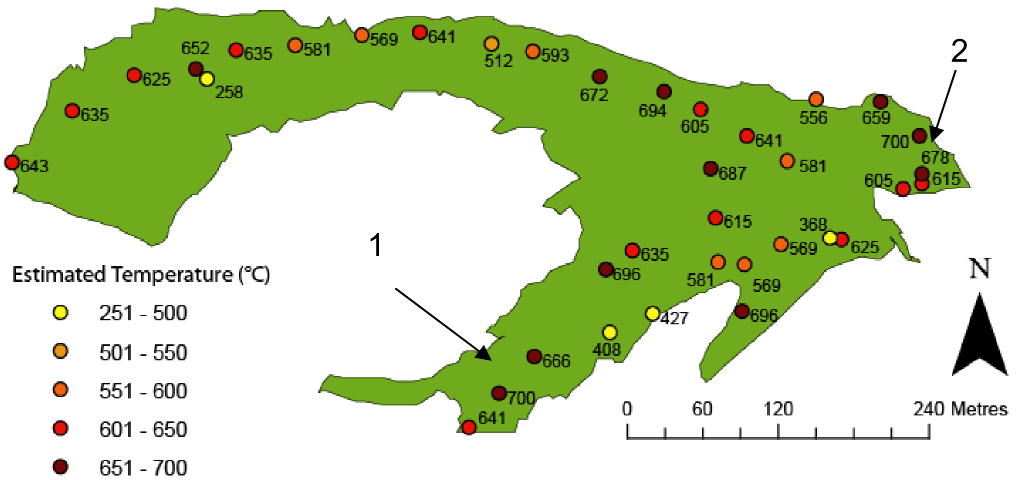

Figure 5.10. Estimated fire temperatures of the Grindsbrook burn scar. 'Hot' spots labelled, 1 and 2.

Examination of spatial distribution of estimate temperatures shows clusters of hot and cool spots. Hot spots occur in the back burn area and down slope area, labelled one and two on Figure 5.10, respectively. This matches closely with qualitative descriptions taken during the surveys in June 2008. Cool spots occur throughout the burn scar, sometimes close to hot spots, suggesting fine-scale heterogeneity in fire combustion.

By using a selection of the quadrats, Moors for the Future (MFF) surveyed the sites over the course of the following year to track the vegetation re-growth (Moors for the Future, unpublished data). Out of those quadrats surveyed by this study, analysed using NIRS, and then surveyed by MFF, only four quadrats can be used. By plotting the percentage recovery of grasses, shrubs and total vegetation with the estimated temperature, a negative trend can be found i.e. those sites that were predicted to experience the highest temperatures, had the smallest amount of re-growth. However, this trend is only based on four results and is not significant.

One aim of this work was to investigate whether simple field measurements could be used to estimate fire severity. Unfortunately, none of the field measurements gave a good relationship with estimated temperature. The only field measurement that gave any relationship to estimated temperature was percentage char cover; however, this only yielded an r^2 of 0.2.

5.5 Discussion

5.5.1 Black carbon production

The production of black carbon during moorland wildfires has important implications for carbon stocks in these ecosystems. This study has shown that though biomass and carbon losses were very high, there was not complete combustion in the fire. In areas of the fire there was a very high survival of unburnt or slightly charred biomass with up to 50% in some individual quadrats. Survival of biomass following a fire will have important implications for any carbon balance or carbon accumulation models following fire.

This study is the first to estimate black carbon production in a heathland setting. In order to calculate pre-burn biomass, biomass values from Forrest

(1971) were used. However, this study was based on *Calluna* from the North Pennines and considered a uniform stand. To more accurately measure biomass, relationships between the height or age of the stand and its biomass are needed. Much data has been gathered on the variation in biomass with stand age (Gimingham, 1972 and references therein) though few studies have directly linked height and biomass.

Black carbon production for this wildfire is approximately 4% BC/CC which is similar to other ecosystems (2.9-7.8% for Amazon Basin rainforest, (Forbes et al., 2006)) though higher than other non-forested ecosystems (1.0-1.1% for African savannas, (Forbes et al., 2006)). This high BC production rate relative to other non-forest settings may be due to the high percentage cover of shrubby vegetation that has a high wood content. When scaled up, black carbon production of 7.87 MgC km^{-2} (7.87 gC m^{-2}) is similar to some components of carbon fluxes from peatland settings (e.g. CH_4, dissolved CO_2 (Worrall et al., 2009a)) and therefore may be a significant component in carbon dynamics of peatlands in fire prone settings.

Given the estimates of fires, both wildfires and managed burning in the Peak District, it is possible to scale BC production across the Peak District National Park. Assuming a uniform BC production of 7.87 gC m^{-2}, across all wildfires, 1.2 km^2 yr^{-1}, approximately 9.4 Mg of BC is produced in the National Park each year during wildfires. If the same production rates of BC occur in areas that show evidence of managed fires (9.3 – 18.6 km^2 yr^{-1}), then between 73 and 146 Mg BC yr^{-1} is produced in the Park during managed burning. By combining these values the total BC produced in the Peak District National Park each year is between 82.4 and 155.4 Mg BC. This value is a potential BC production and does not consider factors that may remove it from the landscape.

There are several pathways that the carbon may be lost from a site during and following a fire: converted to gaseous products during combustion e.g. CO, CO_2; formation of airborne particles; and erosion of char from surface following the fire. Approximately 90% of biomass and carbon was lost from the site during the Grindsbrook wildfire. An understanding of these pathways and their relative contribution to carbon loss is important to help mitigate their loss following fire.

As vegetation regrows, CO_2 is removed from the atmosphere through primary productivity and incorporated into the new vegetation growth, that some authors suggest may balance out the emissions due through combustion leading to a balance in carbon stocks (Levine et al., 1995). The rapid regeneration of vegetation following fires occurs from those surviving stem bases and surviving seed bank (Legg et al., 1992). Higher photosynthetic rates have been observed in grassland settings on burnt sites in the months following a fire (Feldman et al., 2004) and higher rates have also been found on annually burnt communities relative to unburnt plots (Johnson and Knapp, 1993). This enhanced rate of photosynthesis and primary productivity following fires may contribute to offsetting carbon losses through combustion.

Black carbon in an airborne form can be transported and deposited large distances from its formation source (Ming et al., 2009; Shindell et al., 2008) so whilst it represents a direct loss of carbon at the site of the fire, on regional or global terms it contribute to carbon accumulation elsewhere. However, if BC is deposited on snow or ice it has the potential to affect solar absorption by affecting the albedo (Warren and Wiscombe, 1980).

Following wildfires, increases in erosion rates and sedimentation rates in streams draining burnt areas have been well documented (Johansen et al., 2001; Moody and Martin, 2001; Wondzell and King, 2003). The erosion of charred materials from a fire scar could potentially lead to a significant portion of any BC produced being removed from the site. Erosion is often arrested within a few months or years through re-vegetation that helps to stabilise the soil surface (Kinako and Gimingham, 1980) and reduce char losses.

The relative importance of surviving biomass and BC to carbon accumulation is dependant on the relative decomposition rates of these materials. Black carbon is often assumed to be inert to degradation though it is likely to degrade over time (Schmidt and Noack, 2000). Possible mechanisms for loss of BC over time include: oxidation by subsequent fires; slow chemical oxidation; biological degradation; and physical fragmentation (Preston and Schmidt, 2006, and references therein). Although BC may degrade over time, many studies have shown that it has a very long mean residence time, often on the order of 1000s of years (Kuzyakov et al., 2009; Lehmann et al., 2008; Schmidt et al., 2002).

By treating the surviving biomass as litter and the char as black carbon, the decay of the burn products can be modelled as an exponential decay with decay constant, *k*, representing the fractional weight loss each year (Eq. 5.6).

$$M_t = M_0 e^{-kt} \qquad \text{(Eq. 5.6)}$$

where M_t is the mass of litter at time *t*, M_o is the initial mass of litter, *t* time, and *k* the fractional weight loss per year.

Rate of weight loss of material at the near surface varies from 0.01 yr^{-1} to 0.8 yr^{-1} (Clymo, 1984 and references therein) and are influenced by original plant material and external factors. Figure 4 models the decay of the post-burn products from the Grindsbrook wildfire and shows the losses from each carbon pool over time. For this simple model, typical values for *k* were chosen. The surviving biomass was modelled as a litter material with a decay rate of 0.1 yr^{-1}; this is similar to rates of other *Calluna* dominated litters (Heal et al., 1978; Van Meeteren et al., 2007). BC was modelled using a decomposition rate of 0.005 yr^{-1} for BC (Kuzyakov et al., 2009). By calculating the actual size of the carbon pool over time, it is possible to calculate how long it would take before surviving BC is greater than surviving litter. Figure 5.11 shows that this point occurs at around 10 years following the fire. This simple model would tend to support the results in Chapter 4 where after 15 years char outweighs litter.

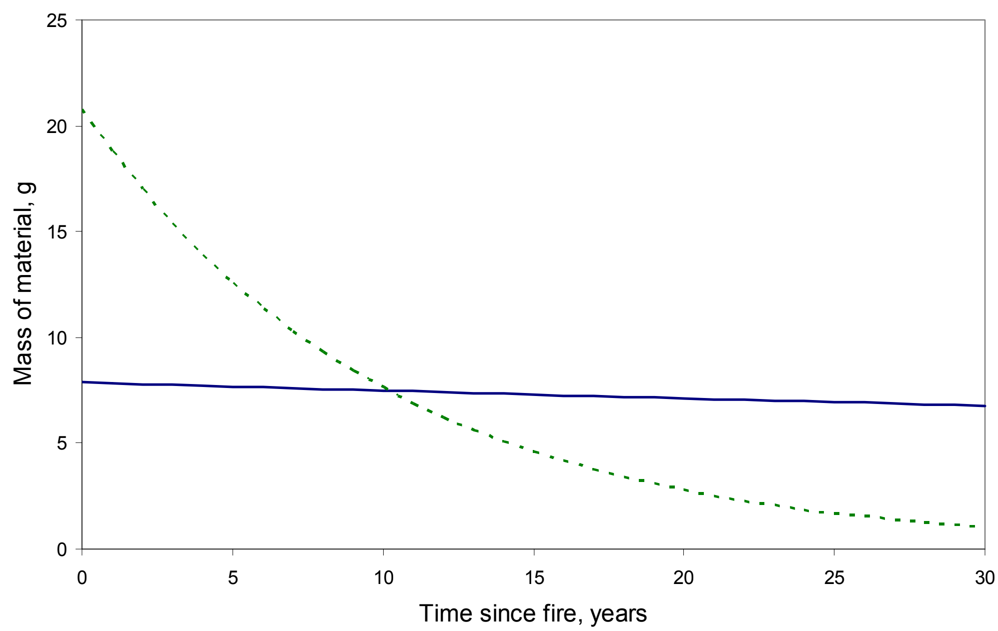

Figure 5.11. Exponential decay of post-burn litter (dashed line) and black carbon (solid line) over time.

However, there are a few caveats to this result. Decomposition rates are strongly controlled by temperature and soil moisture content (Heal and French, 1974; Van Meeteren et al., 2007) so may vary both spatially and temporally depending on local conditions. The decomposition rate of litter in this model is assumed to be exponential; however, this may not be the best regression to accurately model the changes over time. Latter et al. (1997) suggest that exponential curves derived from short-term litter bag experiments may overestimate the loss of litter and that an asymptotic curve, rather than exponential, best describes the long-term decomposition pattern of moorland litter.

When considering the effects of wildfires, the spatial variability of fire intensity needs to be taken into account. This study has considered the black carbon production of the Grindsbrook wildfire as a whole. However, in reality, wildfires will have a mix of “hot” and “cool” spots. This variation in intensity will have important consequences for the consumption of above-ground biomass, termed fire severity (Keeley, 2009), and therefore black carbon production. This wildfire showed variations in fire severity i.e. variations in biomass survival and char production, that could be used to indicate the intensity of the fire. An understanding of fire intensity is important for understanding the effect on seed banks and for regeneration strategies aimed at re-vegetating a site following wildfires. Superficial burns may regenerate naturally, whereas intensive burns may need restoration treatment to re-vegetate and prevent further loss of carbon stores through erosion.

5.5.2 Spatial variability

Results from the NIRS works and field data suggest that the fire had ‘hot’ and ‘cool’ spots. Predicted hot spots coincided with areas thought to be hotter burns from field evidence and fire rescue service reports i.e. back burn area.

There are a few caveats to this work though. Firstly, the calibration curve used assumes a fire duration of two minutes, which although is within range of reported durations, may be too short a duration. This also has consequence for predicted temperatures; a fire with a longer duration but same temperature will appear hotter on the map. The integrated nature of fire duration and fire temperature has been commented on before (Smith et al., 2005) with authors often finding it difficult to disentangle the two aspects. Secondly, the work only addresses the 1900nm absorbance feature; other absorbance features may show different changes with increasing temperatures. Finally, this laboratory method used pure char whereas field-based radiometry will observe a greater degree of heterogeneity. Therefore any up-scaling between lab and field scale must be treated with caution.

5.6 Conclusions

Wildfires are a major source of carbon to the atmosphere and at this wildfire, around 90% of the above-ground biomass was lost through combustion. Of the remaining products, 72% (20.8gC m^{-2}) was surviving biomass. By using the loss of biomass during the fire and associated changes in carbon stocks, black carbon production has been calculated to be approximately 8gC m^{-2}. Alternatively this is 4% of carbon consumed during the fire. By extrapolating this across the Peak District National Park, up to 155 Mg of black carbon may be produced per year.

This study presents the first estimate of black carbon production from a moorland wildfire in an upland setting of the U.K. Results show that whilst black carbon is an important carbon sequestration mechanism on the long-term scale, on short-medium term scales fires lead to net carbon emissions.

There was evidence of a larger degree of spatial variability in the fire severity during the Grindsbrook wildfire as evidenced from field observations and ranger reports. By using NIRS, a quantitative approach has been developed that may provide insights into fire conditions at time of char formation.

Chapter 6:

Perceptions of managed burning - Conflict in the uplands?

6.1 Introduction

The interconnectivity of ecosystem services and multiple uses that the uplands provide means that there can often be competition for land between different stakeholders (Bonn et al., 2009c). This competition can lead to a potential for conflict to occur in upland areas. These conflicts, often about an aspect of biodiversity e.g. raptors and game birds (Redpath et al., 2004; Thirgood et al., 2000), are usually conflicts *between* people *about* wildlife, rather than conflicts where an action by humans or wildlife has an adverse effect on the other (Conover, 2002).

Many of these upland areas of the UK are protected areas e.g. Special Protection Areas (SPA), in order to safeguard these habitats and to ensure ongoing conservation. One the most recognisable of these designations is the National Parks. The National Parks and Access to the Countryside Act of 1949 led to the creation of National Parks that cover a significant proportion of UK uplands (9.3 per cent and 7.2 per cent of England and Scotland respectively (Association of National Park Authorities, 2009)). The statutory requirements of each National Park in the UK, set out in Section 61 of the Environment Act 1995, are:

a) To conserve and enhance the natural beauty, wildlife and cultural heritage; and
b) To promote opportunities for the understanding and enjoyment of the special qualities of those areas by the public.

If these purposes conflict, then the “Sanford Principle” applies and conservation comes first. This is given statutory force in section 62 where if conflict arises then any relevant authority “shall attach greater weight to the purpose of conserving and enhancing the natural beauty, wildlife and cultural heritage of the area comprised in the National Park”. Even at the heart of the National Park Act there is the potential for conflict between users in these upland settings.

Another contentious issue in the uplands is that of managed burning of moorland (Glaves and Haycock, 2005). The discord between stakeholder groups is exemplified in a recent public consultation on the heather and grass burning code (DEFRA, 2007b) where stakeholder groups split along traditional division lines, making discussion on the issues difficult. At the same time the literature on managed burning has often provided contradictory research on many aspects of burning e.g. burning and lichens (Davies and Legg, 2008; Vandvik et al., 2005), leading some stakeholders to argue that without a clear grasp of the science, any changes to policy may be foolhardy. Therefore, any addition to the literature on managed burning should take into consideration the wider implications of the work.

One of these wider implications is the impact burning has on the general public and visitors to these areas. Heather moorlands are sought after for recreational use due to their perceived tranquillity, with 75% of visitors to the Peak District National Park citing the landscape and 'naturalness' of the landscape as the primary reason for visiting (Moors for the Future, 2004). These landscapes are a heavily managed landscape (Holden et al., 2007; Thirgood et al., 2000) and many visitors may not realise the 'wilderness' they are visiting is in fact a semi-natural environment.

In order to increase productivity in these landscapes, managed burning of vegetation is a common feature of the UK uplands and has been carried out for several centuries (Yallop et al., 2009). However, there is ongoing interest in finding suitable alternatives to managed burning. This may be due to changes in rural labour leading to a loss of management skills (Eadie, 1984; Hubacek et al., 2009), or sensitive areas where burning is restricted e.g. SSSIs (DEFRA, 2007a). Proposed alternatives include cutting by mechanical means, flailing or 'rolling' back the vegetation. Several authors have investigated the impacts of adopting such methods (Cotton and Hale, 1994) but there are currently few data available on the impacts of adopting such a method on ecosystem services.

In their paper, Cotton and Hale (1994) investigate the effectiveness of cutting as an alternative to burning. One of the drivers for this research was to “reduce public hostility to the management programme [managed burning]”. However, as the research was focussed on the natural sciences problem, no further information was given about this hostility. Public surveys are commonly used in the uplands of UK (PDNP), though studies on the public perception of managed burning are limited in the UK. Studies in America and Australia have investigated attitudes towards prescribed fire management in forest settings (Bell and Oliveras, 2006).

6.2 Chapter Objectives

The objectives of this chapter are two fold:

- Examine stakeholders’ opinions on natural science study (results from chapters 2 and 4) and look at the consequence of using this as a basis of management decisions;
- Investigate the public’s perceptions to managed burning to see if there is ‘public hostility’ towards managed burning.

6.3 Materials and methods

In order to assess the impact managed burning has on stakeholders and the general public, two separate studies were carried out: one focussing on semi-structured interviews with key stakeholders and the other a survey carried out at National Park visitor centres.

Initially a structured workshop had been designed in which multiple (>20) stakeholders would discuss the results of the natural science study, share concerns, and to comment on the data. This would have stimulated debate about managed burning with the aim of drawing out potential policy strategies. This workshop was to be run at a conference as a side session in order that key stakeholders would be present. However, due to problems external to this study, this proposed session was cancelled at short notice.

The workshop was then redrafted into an interview format. Due to the late cancellation of the workshop, and also due to the start of the grouse hunting season in August 2009, only a small selection of stakeholders were able to take part.

6.3.1 Semi-structured interviews

The multi-user environment of the uplands means that indentifying who is a stakeholder can often be challenging. Welp (2001) defines a stakeholder as: "one who: (a) is affected by or affects a particular problem or issue and/or (b) is responsible for problems or issues and/or (c) has perspectives or knowledge needed to develop good solutions or strategies, and/or (d) has the power and resources to block or implement solutions or strategies." This definition therefore includes not only those who affect outcomes but also those who are affected by them.

Previous research in the uplands has identified eight stakeholder categories with interests in the uplands: water companies; recreational groups; tourism-related enterprises; agriculture; conservationists; grouse moor interests; foresters; and statutory bodies (Prell et al., 2007; Prell et al., 2008). It is these categories that were used to help select relevant people to interview. Individual stakeholders were selected by using existing networks of contacts within the moorland community and also through a “snowball” sampling technique (Reed et al., 2009b). Six semi-structured interviews were carried out with members of the following categories:

- Conservation (n = 3)
- Statutory body (n = 2)
- Tourism related (n = 1)

The primary aim of the interviews was to investigate the implication of a future scenario where longer burning rotations were favoured. This reflects the results in chapters 2 and 4 where possible carbon benefits may exist for longer rotations. It was felt important that stakeholders were included in the process of evaluating these results in the wider picture, as the issue of managed burning is often a contentious issue and is seen as the most pressing land management issue in the uplands (Dougill et al., 2006).

A prepared list of questions and notes were on hand to guide the conversations (Appendix 2) and some key questions were used in all the interviews:

- How does your role link with managed burning in the uplands?
- What would be the ideal rotation length to manage for your interests?
- If policy were changed so that longer rotations were the only ones allowed, what would the implications be for you and/or your organisation?

During the interviews simple scenarios were used to investigate possible futures for managed burning in the uplands e.g. what would happen if policy favoured longer rotations, what would be the implications of a shooting ban? In an ever changing world, the use of scenarios to help anticipate and plan for the future, has been advocated due to constraints on using traditional modelling approaches (Rotmans et al., 2000). The use of scenarios in upland settings, to help understand future changes, have been widely used across the UK (Reed et al., 2009a).

The average interview length was approximately 40 minutes partly to keep the interview focussed and also to keep the respondent's interest and minimise disruption to their normal working life. Interviews were recorded and qualitative results were based on transcriptions from the interviews. Transcripts were coded using key words and phrases (Table 6.1). The method roughly followed the process used in Grounded Theory, a qualitative method to systematically collect and analyse data to construct theoretical models on social phenomena (Corbin and Strauss, 1990; Glaser and Strauss, 1967). Coding was done both during and after the transcription process and collated using NVivo v7 software package (QSR International).

Coding words	
Code	**Description**
Conflict	This relates to a feeling of conflict between stakeholders
Education	Relates to comments about educating others about managed burning
Critical of management	Any comment critical of managed burning as a management tool
Burning management plans	Relates to mention of burning management plans drawn up between statutory bodies and land owners/managers
Money	Mention of money or finance related to moorland management
Wildfire	Mention of wildfire (uncontrolled fires)
Sheep	Mention of sheep and/or impact of sheep on landscape
Politics	Pertaining to politics related to working in the uplands

Table 6.1. Examples of word and phrases used in the coding process with descriptions

6.3.2 Public perceptions survey

6.3.2.1 Study areas

Public surveys were carried out in three National Parks in northern England – North York Moors National Park, Yorkshire Dales National Park and Peak District National Park (Figure 6.1; Table 6.2). Within each national park managed burning is used as a management technique and wildfires have also occurred in recent years in these areas (North York Moors (Maltby et al., 1990); Peak District (Albertson et al., 2009)). The survey was carried out at National Park visitor centres during summer 2009.

National Park	Area (km^2)	Population	Visitor Numbers (millions)	Visitor Centre
North York Moors	1434	25,000	6.3	The Moors Centre, Danby
Yorkshire Dales	1769	19,654	9.5	National Park Centre, Reeth
Peak District	1437	38,000	10.1	Moorland Centre, Edale

Table 6.2. Statistics for National Parks used in this survey (Association of National Park Authorities, 2009)

Figure 6.1. National Parks used during the public survey (reproduced with permission from the Association of National Park Authorities)

6.3.2.2 Survey questions

The survey was written using plain language, avoided using technical words and consisted of mainly closed questions rather than open questions. The aim was to keep the length of time to a minimum, typically 5-10 minutes, to avoid losing the attention of the participants. As the survey was administered rather than a postal questionnaire, definitions could be given and any questions could be clarified during the survey.

The following definitions were used during the course of the surveys:

- Managed burns are planned fires, wildfires are unplanned fires
- Managed burning on uplands can be used for a variety of reasons including (but not limited to) habitat maintenance for grouse and sheep, and fuel reduction to mitigate fire risk.
- Uplands are areas above 300m often with poor soils, rough grazing, often with moorland type vegetation e.g. heather, bilberry, grasses

The questions used in the survey were based on a similar survey in Wombat State Forest, Victoria, Australia that aimed to understand perceptions of prescribed forest burning (Bell and Oliveras, 2006). The survey consisted of 18 questions (Table 6.3; Appendix 3). The first four aimed to establish if the respondent had seen any activity of fire in the uplands. Only respondents who said they had seen managed burning or evidence of managed burning in the UK (question 3) or the local area (question 4) were asked to continue answering the remaining burning questions in the survey (questions 5-11). The section related to burning asked specific questions about managed burning and how the respondent is affected by managed burning. Photos were used to illustrate landscapes described in question five and additional information was given if requested for all these questions. The remaining set of questions (12-18) were a set of data-gathering questions and also aimed to establish reasons for visiting the park on that day.

The survey was administered in a face-to-face interview with members of the public at each of the visitor centres. The survey was conducted during the main summer school holidays and was completed on both weekdays (Yorkshire Dales and North York Moors) and weekends (Peak District). In total 88 surveys were completed across the three Parks. Only a few people (< 10) approached did no wish to take part. It was made clear to the visitors that the survey was part of a PhD research at Durham University and visitors were asked if they could spare a few moments to answer a series of questions on fire in the upland environment. Contact details were available should any visitor want to follow up with questions at a later date.

Question Number	Question
1 and 2	Are you aware of any past wildfires in the UK uplands/this local area in the past 10 years?
3 and 4	Have you seen managed burning activities or evidence of managed burning in the UK uplands/this local area?
5 a-c)	How would you rate the appearance of the landscape following managed burning? a) Immediately after a burn? b) one year after a burn? C) ten years after a burn?
6	How much does the smoke from managed burning affect your experience in the uplands?
7	After managed burning, plants and animals recover in different ways. If you have seen examples of recovery following a fire please write them here.
8	How often do you think managed burning should occur?
9	How would you rate the effectiveness of current managed burning practices for habitat management?
10	How would you rate the effectiveness of current managed burning strategies for fuel reduction and fire management?
11	Home town/Postcode (first part)
12	Male/Female
13	Age
14	How many times in the past year have you been to this area?
15	What activities do you/are going to undertake whilst here?
16	Do you work in the National Park?
17	Are you a member of any environmental related groups?

Table 6.3. Questions used in the survey

6.3.2.3 Data analysis

Statistical comparisons, where used, were made using the same approach outlined in Bell and Oliveras (2006) by using a χ^2 analysis assuming a null hypothesis of an equal chance for yes/no questions (1:1 ratio) and graded questions (1:1:1:1:1 ratio). The χ^2 test uses absolute numbers rather than the percentages shown.

6.4 Results

6.4.1 Interview results

Several specific issues were raised in response to moving to a longer rotation scenario: increased fuel loading; potential access problems; impacts on bird populations and its use in policy.

By moving to a longer rotation, the increase in fuel loads on the moors was highlighted as a major concern: *"what owners would say straight away, and is a difficult thing to argue, is this build up of fuel in the canopy"* and *"from a fire behaviour perspective, the longer the rotation the more woody material, therefore potentially the more intense the burn could be".* This build up of woody material would lead to more intense fires, be they managed burns or wildfire. The implications of 'hot' fires would be that the risk of burning into the litter and peat layers would increase and as one respondent said: *"if it does get into the peat, then your carbon's gone out the window".* Hotter burns were also linked with less successful re-growth and slower regeneration.

Access issues were raised as a response to a longer rotation scenario: *"I suppose potentially one of the outcomes is that we might have longer heather beds across certain public rights of way"* and also *"in open access areas where people are discouraged from walking through deeper heather".*

From the bird conservation point of view, longer rotations would have mixed results. For sites on deeper peats, the general feeling was that the grouse numbers would not be significantly altered. However *"if it's rolled out onto the drier heath at lower altitude, then it might have a significant effect on grouse"*. The effects suggested were *"lower population densities and lower breeding success".* The underlying reason for this lower success was that longer rotations would reduce the amount of 'edge' preferred by grouse (Watson and Miller, 1976; Watson and Moss, 2008). The effect on waders such as the golden plover and curlew was also highlighted with the hypothesis that numbers would decrease with longer rotations as *"things like golden plover like the shorter vegetation after fires"* and *"do really well on the mosaic provided by burning"* so any shift to less frequent burning would have a big impact on these species.

Those in the policy sector also added that the results from this thesis would go some way to support existing feelings within the policy bodies*: "in some ways we welcome the results…we've felt that long rotations is the way forwards for a long time"*. Another said: *"in some ways it's going to help us…persuade, or give us another angle to persuade the intensive grouse moors that they shouldn't be as intensive"*. Whilst this may be a benefit for those who would favour longer rotations, care must be taken to describe the limitations of the study and its applicability to other settings: *"people always take out of scientific data what they want and the danger would be your findings will get portrayed as what should be done on heather moor"*.

By analysing the occurrence and source of the coding words, key topics that emerged from the interviews can be quantified (Table 6.4).

Coding word	Number of References	Number of Sources
Money	17	5
Wildfire	14	5
Responses to longer rotation scenario*	13	6
Conflict	12	4
Burning management plans	12	4
Rotation lengths	11	3
Politics	11	5
Fuel loading	9	3
General public	8	4
Unclear/mixed results of managed burning	8	3

Table 6.4. Top ten code words used.

To uncover key themes and contrast the opinions of these different stakeholders, it is possible to quantitatively assess the abundance and diversity of comments made in these categories. If a topic is an important issue across all the stakeholder groups, one might expect to see an even distribution of comments but if one person is the only discussant it may only pertain to that stakeholder category. In order to determine the evenness, with which topics were discussed, the Shannon Index was applied. Used primarily in the field of biodiversity, it, along with species evenness, is used to measure diversity in categorical data (Krebs, 1972).

The Shannon index is calculated thus:

$$H' = -\sum_{i=1}^{S} p_i \ln p_i \qquad \text{Eq. 6.1}$$

where S is the total number of respondents (n = 6), and p_i is the relative abundance of comments calculated as proportion of the total number of comments.

To calculate evenness:

$$E = \frac{H'}{H'_{\max}} \qquad \text{Eq. 6.2}$$

where H'_{max} = ln S and E is between zero and one.

In biodiversity terms, the less variation there is in communities, the higher the value of E. When used in this context, the higher the value of E the more evenly distributed a topic is across the respondents. Figure 6.2 shows that evenness values for those topics in table 6.4. The three highest scoring topics are “money”, “responses to longer rotation scenario” and “politics”.

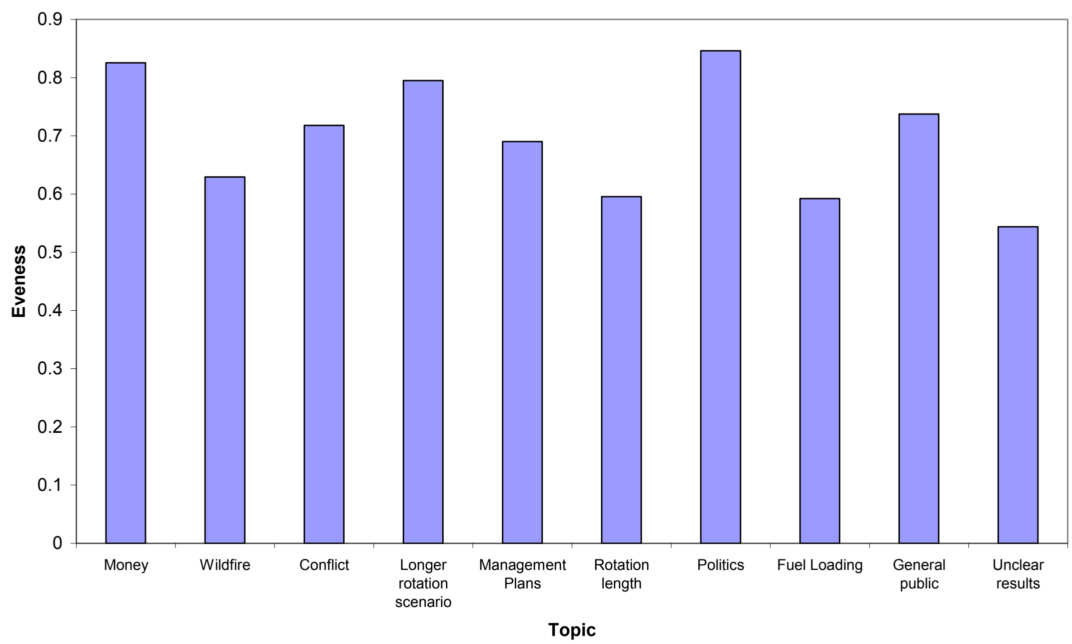

Figure 6.2. Evenness values for topics in table 6.4

In both measures of topic importance, “responses to longer rotation scenario” had a high number of references and sources. This is not surprising as this was the main point of the interviews and was a prompted question. These comments were about the general reaction to the work rather than specific issues which were assigned additional codes. Overall, the reactions were generally positive but with reservations about the caveats attached to work on the Moor House with one respondent noting: *"my gut reaction to that [research results] is that you've got a bit of a problem, in that your whole thesis is based upon one small part of one small moor… how representative is that situation? I would say that it's very dangerous to base anything solid around that".*

Aside from the responses to the longer rotation scenario, money and politics ranked highly. The main money issue that was raised was that of "new money" coming into the grouse shooting: *"a lot of new money is coming in now and that's the nouveau riche if you like and they are very much business men… and they want to see a return straight away"*. This contrasts with the "older money" like "*the Duchy land or something in the same ownership since the 1300s"* where *"it's a rich man's hobby"* and where *"money isn't an object to people who have grouse moors. They want to have fun [and] shoot grouse".* It is not uncommon for grouse moors to make losses and in years where the grouse population crashes, these losses can be very high (Sotherton et al., 2009).

Politics also played an important part of the interviews. This refers to both the wider stakeholder conflicts and also internal politics within organisations. The wider debate was exemplified by one respondent who said: *"I think at the moment it's like the raptor debate. You've got two sides of it"*. The perception of the statutory body, Natural England was questioned: *"everyone* *thinks all Natural England wants from their grouse moors is for it to look like Moor House"*. Internal politics was also highlighted as an issue when dealing with the burning regulations: *"the local team feels that we have some sort of special conditions that we can allow sometimes more burning than the national team would be happy with"*.

6.4.2 Survey Results

In total 88 visitors to the three National Parks were interviewed, 35% (n = 31) from the North York Moors National Park, 28% (n = 25) from the Yorkshire Dales National Park, and 36% (n = 32) from the Peak District National Park.

Who made up the sample?

Table 6.5 and Table 6.6 shows the ages of the visitors and the numbers of visits they had made to the local area in the previous year. Nearly half of those surveyed (47%) were over 55 and over 40% were visiting for the first time.

Age, years	< 25	25-34	35-44	45-54	55-64	>65
Proportion, %	5	7	23	20	30	16

Table 6.5. Proportion of respondents by age group (n = 88).

Number of visits	First time	1	2-5	6-10	10+
Proportion, %	42	13	16	8	22

Table 6.6. Proportion of respondents by number of visits to the area (n = 88).

The type of activities undertaken in each of the parks varied slightly; nearly all the visitors to the Peak District went walking compared to less than two-thirds for the North York Moors (Table 6.7). This perhaps reflects that the Pennine way starts in Edale so most visitors would be starting either this long-distance walk or doing a section of it as a day walk. The Moors Centre at Danby, on the other hand, had fewer walkers and a greater number of sightseers. These visitors were perhaps more likely to be casual walkers with dogs or visiting with families.

Activity	North York Moors	Yorkshire Dales	Peak District	All Parks
Walking	64.5	84.0	93.8	80.7
Climbing	3.2	0.0	9.4	4.5
Draw/paint/photography	25.8	4.0	3.1	11.4
Picnic	25.8	24.0	37.5	29.5
Sightseeing	71.0	40.0	28.1	46.6
Cycling/mountain biking	0.0	12.0	0.0	3.4
Bird watching	9.7	0.0	6.3	5.7
Horse riding	0.0	4.0	3.1	2.3
Other	0.0	36.0	21.9	18.2

Table 6.7. Percentage of visitors undertaking activities in the National Parks

Many visitors were part of organisations that interact with the upland environment with 36% (n = 32) of visitors saying they belonged to an environmental related group. Organisations included the National Trust, RSPB, CPRE, BASC, Woodland Trust, and local wildlife trusts.

Awareness of fire in the uplands: Questions 1 – 4

Many of the visitors surveyed were unaware of wildfires in the UK (57%, χ^2 = 1.64, NS (not significant), Figure 6.3) or within the local area i.e. National Park they were visiting (81%, χ^2 = 33.1, p < 0.01, Figure 6.3). A greater proportion of those surveyed were aware of managed burning compared to wildfire. Nearly 70% of those surveyed had seen managed burning or evidence of managed burning in the UK (69%, χ^2 = 13.1, p< 0.01, Figure 6.3) and over two-fifths had seen in within the local area (44%, χ^2 = 1.13, NS, Figure 6.3).

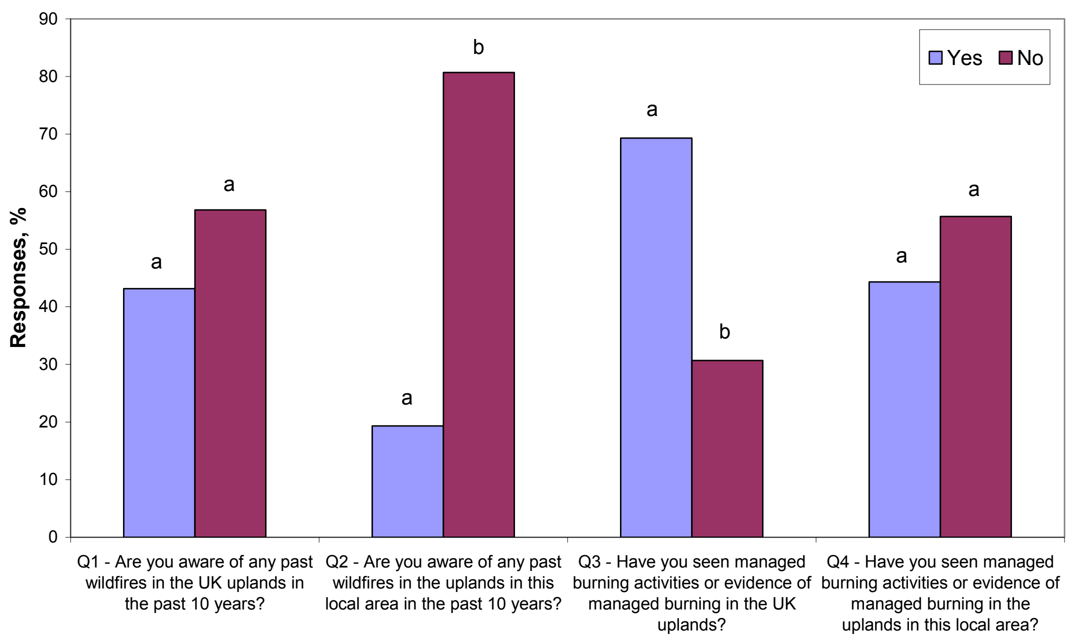

Figure 6.3. Proportion of yes/no responses to questions 1 to 4. (n = 88, Different letters above a bar denotes values that were found to be significantly different, p<0.01)

Approximately 31% (n = 27) of those surveyed had not seen managed burning in the UK. Whilst overall 31% had not seen burning, there were some regional differences in this figure. Visitors to the North York Moors were perhaps more aware of managed burning with only 16% of visitors expressing no knowledge on the subject. Of the visitors to the Peak District, 38% said they had not seen managed burning and a similar proportion, 40%, occurred within the Yorkshire Dales.

Perceptions of managed burning: Questions 5 – 11

Responses for question 5 showed a general trend of increased proportions of positive ratings i.e. "good" or "very good", with increasing time since burning (Figure 6.4). Only "good" or "very good" responses were recorded for a landscape ten years after burning. Though most respondents thought a landscape immediately after burning was unsightly (80% rated it "poor" or "very poor", $\chi^2 = 67.6$, $p<0.01$), many (26%, n = 16) made similar comments saying they understood why it had to be done so in the short term they wouldn't mind it. One visitor said that they liked seeing burnt patches as it "*varied the landscape for a while*".

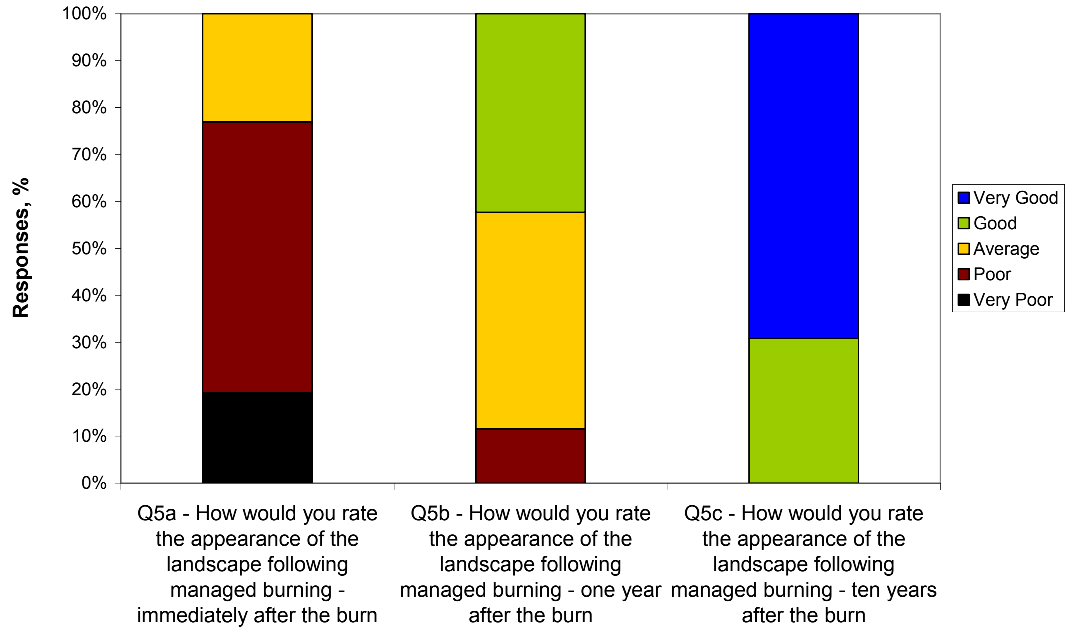

Figure 6.4. Proportion of responses to question 5 (a-c) (n = 61)

Of those who expressed an opinion in question 6, a high proportion of the visitors stated that they were not affected by the smoke from managed burning (83%, $\chi^2 = 68.4$, $p<0.01$), with one visitor actually saying they liked it. Comments from those who were affected by smoke were due to either medical reasons e.g. asthma, or that it would "*spoil the day*" if they were to come across it directly.

Question 7 was left intentionally open ended to try to capture the broad range of local conditions of moorlands around the country. More than 50% (n = 33) gave some comment on regeneration they had seen. Many visitors (n = 20) either said that they had seen new growth/re-growth or *"green shoots"* whilst out walking or driving. Some gave more detailed responses such as *“bilberry comes back first in my part of the dales”* or *“heather grows better* “after fire. Whilst most comments were on the vegetation recovery, some visitors gave some examples of the effect on animals. Birdlife was commented on with some people noticing *“nesting birds”* coming back though one visitor had noticed burnt eggs following a fire. Snakes were the other animal that elicited comments. One visitor suggested *“adders come back to burnt areas”* and another visitor, in a different park, went on to say that “*buzzard numbers reflect snake population which is changed when habitats are changed through burning”.*

When asked about the frequency of burning a large proportion of visitors (44%) did not know enough to comment on the timing of burning. Of those who chose one of the other options, most chose the “fine at present” option (85%, χ^2 = 42.4, p<0.01). The remaining 15% thought burning should occur more frequently with bracken control being cited as a reason for wanting additional burning. No-one thought burning should be less frequent.

Perceptions towards the effectiveness of managed burning for habitat management and fuel load management were on the whole positive though a large percentage of visitors did not know enough about burning and links to habitat and fuel management (41% and 30% respectively; Figure 6.5). Approximately 50% of visitors stated that managed burning was “good” or “very good” for habitat management. A higher proportion, nearly 70%, said managed burning was "good" or "very good" for fire management. This may be a falsely high result as the question required a certain degree of information to be given to the respondent before they understood the premise of fuel loading on moors. Once they understood many said that they *"hadn't thought of that before*" and that it was *"a good idea".*

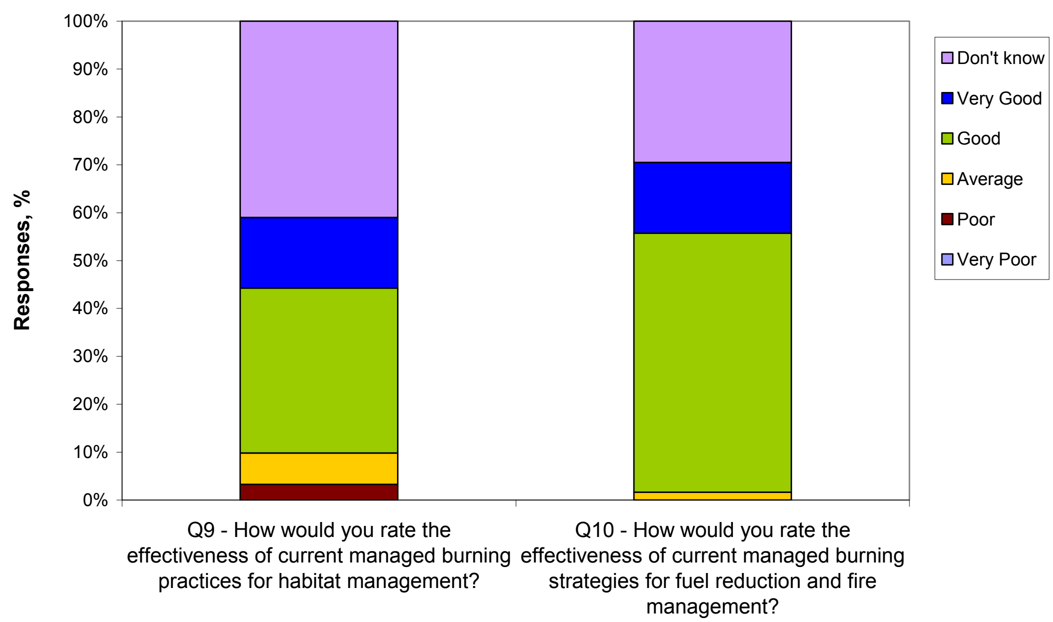

Figure 6.5. Proportion of responses to questions 9 and 10 (n = 61).

Interaction between demographics and perceptions

In order to see if there are any differences between groupings in the awareness of wildfire and managed burning, the responses to questions 1 - 4 were broken down into gender, age and familiarity with the area (number of visits). To test for any significant differences a χ^2 test of a 2 × N contingency table was used (Table 6.8). The number of categories determined N where gender N = 2, age N = 6, and visits N = 5. Table 6.8 shows an example of this.

	Male	Female	Total
Yes	21	17	38
No	27	23	50
Total	48	40	88

Table 6.8. Contingency table of responses, by gender, to question 1 - "Are you aware of any past wildfires in the UK uplands in the past 10 years?

Data from Table 6.8 gives a p-value of 0.906 suggesting that there is insufficient evidence to suggest that there is a strong link between gender and the awareness of wildfires nationally. Table 6.9 shows the results from different questions and factors.

	Gender	Age	Visits
Question 1 – Wildfires nationally	×	×	×
Question 2 – Wildfires locally	×	×	×
Question 3 – Managed burning nationally	×	0.032	×
Question 4 – Managed burning locally	×	0.034	0.001

Table 6.9. Significance of each factor in responses to questions 1 - 4. p-values of significant results given, × = not significant.

There is no significant difference between gender and the awareness of fire in the uplands be that wildfire or managed burning. There is also no difference between age or familiarity when considering awareness of wildfires (Table 6.9).

Age is, however, a significant factor when discussing managed burning. For question 3 the results are influenced by the under-25 age group who had not seen any managed burning nationally unlike the other age groups. When this class is removed, there is no significant relationship between age and awareness of managed burning nationally. A similar effect is seen for question 4 that looks at managed burning locally. With all classes included there is evidence for a relationship between awareness of managed burning and age, however, when the under 25s are removed, there is no significant relationship.

There is a significant relationship between number of visits and awareness of managed burning in the local area (Table 6.9). In this instance those people who were visiting for the first time or had only been once before, had not seen any burning locally. However, when these classes (<25 and 25-34) are removed from the χ^2 test, there is no significant relationship between number of visits and awareness of managed burning in the local area.

6.5 Discussion

6.5.1 Interview Discussion

This primary aim of this work was to investigate the impacts of using the results of this thesis as a basis for future policy. The overall feeling of the stakeholders interviewed was that of cautious restraint. Several of those interviewed sounded a note of caution about scaling up results from a site-specific experiment to national policy. Moor House, as a research location, may carry criticism for being an atypical upland moor due to its pristine nature and having been primarily used for scientific purposes for the last 55 years; though these are also its strongest features. There are very few long running experimental sites that deal with ecological and environmental change in the UK (e.g. The Park Grass Experiment, Silvertown et al., 2006) so to have a resource like Moor House Nature Reserve with its rich history of experiments is invaluable. However, with many caveats placed on data obtained from plot scale experiments, such as this study, should these types of results be discarded as nothing more than trivial at best?

The translation of research findings into public policy is not always straightforward with researchers and policymakers often working to different timescales and audiences (Brownson et al., 2006). There are several models for the way in which research is viewed by both researchers and practitioners (Fox, 2003):

- Practitioners know best and should be left to get on with it without interference from professional researchers (the conservative view)
- Practitioners are lacking in key knowledge and ways must be found to re-educate them into effective service delivery (the evidence based practice approach)
- The model of research in academia, which does not normally engage with end users, must be altered so that the researcher alters their way of working in order to provide meaningful research to practitioners

It is this last scenario, where interaction between the partners is key, that leads to a more robust approach to problem based research. The engagement of stakeholders in research in order to create a sense of ownership over research outcomes is being widely advocated (e.g. Dougill et al., 2006; Fraser et al., 2006)

From the interview discussions there are several possible consequences in adopting longer rotations as a standard policy. One of the main implications of moving to longer rotations would be increases in the fuel loading of the moorlands. The issue of fuel loading is an important one as little work has been carried out to quantify the effect of burning on wildfire occurrence. Increases in fuel loads will lead to hotter fires be they managed fire or wildfires and if longer rotations are adopted larger fire breaks would need to be put in to prevent large wildfires and these breaks would "*from a biodiversity point of view [would have] a lot less value because they'd be huge".* Fuel loading is an important part of wildfire prediction and prevention so an understanding of how fuel loading changes with heather stand is important (Hobbs and Gimingham, 1984a). Estimation techniques must also be quick and cheap in order to effectively obtain the data without costly and time consuming destructive biomass techniques (Davies et al., 2008).

There are many anecdotal examples from farmers and land managers about the link between managed burning and wildfire reduction. However, if we are to consider the future, and the threat of increased wildfires that climate change is predicted to bring (Albertson et al., 2009; Wotton et al., 2003), we need good evidence-based research to present to governments and funding bodies for future fire service funding. One respondent said: *"but getting the powers that be to appreciate that money spent now is better in the future, requires us to have some really good modelling so we can determine how much we are potentially saving by putting in these measures* [fire breaks, managed burning etc]*"*

Another issue that was brought up as part of moving to longer rotations was the impact on birds. The UK has many internationally important bird species that are present in the uplands. Like much of the uplands there are several drivers for change in upland bird populations (Pearce-Higgins et al., 2009) one of which is grouse moor management. The average number of grouse shot on estates have been found to correlate with the density of burnt patches (Picozzi, 1968). Some experiments have attempted to look quantitatively at the density of moorland birds, in particular grouse, on large scales (Pearce-Higgins et al., 2006; Sotherton et al., 2009) though other factors such as the level of keepering play an important and complementary role (Hudson, 1992). The effect of burning and moorland management on waders such as curlew and golden plover has also been investigated (Pearce-Higgins et al., 2009 and references therein). Few of these studies, however, have investigated changes related to burning rotation length, rather the presence of moorland management and burning.

6.5.2 Survey discussion

In general, visitors to the three National Parks were aware of the practice of managed burning both nationally and also on the local scale, but their knowledge of the practice varied greatly. Whilst nearly 70% of those surveyed had seen managed burning, only a quarter of those said that it “had to be done”. First-time visitors and young people were those who had not seen managed burning either nationally or in the local area.

During the course of the study, there were no majorly antagonistic comments towards managed burning. The negative comments encountered were one relating to asthma and smoke and one towards game bird shooting rather than burning *per se*: *"if it's good for wildfire reduction then yes, but if it's for rich Londoners to shoot birds then no, absolutely no".*

Overall this lack of averse comments would seem to differ from the "public hostility" cited in Cotton and Hale (1994). However, this hostility may not be as severe or as frequent as first thought. Public hostility in their paper (Cotton and Hale, 1994) mainly arose from a lack of knowledge about managed burning amongst those who used Ilkley Moor for recreational purposes. Complaints to the local council (Bradford Metropolitan District Council) were mainly a result of staining of clothing following walking though charred remains (Dr D. Cotton, pers. comm.). In a recent visitor survey in the Peak District National Park, out of over 200 comments of spoiling factors, only one negative comment was about burning (Peak District National Park Visitor Survey, 2005).

The hostility towards burning could in part be driven by a lack of knowledge on the subject. Indeed, although most of the visitors recognised managed burning, many said they knew little about the reasons behind the practice. One visitor would be wary of seeing smoke and ringing up the Fire and Rescue Services said that it would be *"useful to perhaps learn more about it*

... as it would put my mind at ease if I know more about it". This was echoed by other visitors across the different National Parks. The smoke from burning may be seen as a signal to some that access on to land is allowed or that setting fires is acceptable. One visitor in the North York Moors said that *"some visitors may have been tempted to go onto the hills during foot and mouth year* [2001] *when they saw burning and asked why farmers were allowed up and they weren't".* Another posed the question of whether the smoke from managed burning in the spring might in fact encourage arsonists out onto the moors during the spring and early summer that are the periods that often see higher numbers of wildfires (Albertson et al., 2009). If visitors understand what is going on, they may be more willing to take notice of any access limitations or increased fire risk ratings.

Education was highlighted as a way of improving levels of understanding within the visitors to these areas. Several comments suggested that it could *"be covered more in the media"* and that it would be *"good to keep public informed*". One visitor said that burning may have a *"poor public image but if keepers explained the case people would understand*". One possible outlet for disseminating information about why burning occurs is the through exhibitions and displays that are often present at visitor centres. Information about burning was present at each of the visitor centres though the scale and prominence of these displays differed. Allen (2004) advises that exhibits should have areas that are "immediately comprehendible" so that visitors do not have to work out what to do in order to access the exhibit. Learning happens throughout people lives and that this kind of leisure time learning is motivated by their interests (Falk et al., 2007) so these underlying reasons for wanting to learn more must be taken into account when designing displays and exhibitions.

Many studies have focussed on the public's perception of wildfire especially at the wildland-urban interface and other fire prone areas (McGee and Russell, 2003; Mendez et al., 2003). Vining and Merrick (2008) suggest that poor communication between forest managers and the public is often though be a key contributor to disputes and misunderstandings. Other studies in the United States suggest that for meaningful dialogue about prescribed fire the flow of communication between land managers and the public must be a two-way process (Bell and Oliveras, 2006; Jacobson et al., 2001). Fraser and Kenny (2000), in their study of perceptions of urban forests in Canada, suggest that the best strategy to adopt in order to develop successful urban forest plans was an educational programme. They conclude that although it takes more time and effort, a programme like this may ultimately generate support and awareness of the planner's aims. Public education programmes may be also beneficial as they allow the public to weigh up the pros and cons of any management and to make suitably informed decisions (Vining and Merrick, 2008).

6.6 Conclusions

One of the future research requirements suggested by Gray and Levy (2009) is "what are the social consequences of either using or not using prescribed fire in peatlands?". This work has made an initial attempt at trying to understand how research within this thesis be taken in the stakeholder community and has also looked at the public knowledge and understanding of managed burning.

The stakeholders interviewed raised several specific changes that may occur in a longer rotation scenario; however, a note of caution was raised in all interviews about the transferability of the results to other areas of the country. The results in this thesis are a significant addition to a sparse literature on burning and carbon, so care should be taken when using them as a basis for policy or management plans as caveats to the work need to be highlighted in order to avoid an incorrect interpretation and subsequent miscommunication.

The public's perception of burning was, overall, fairly well informed. Most had seen some aspect of burning whether the smoke or the burnt ground afterwards and some said they knew that it had to be done. However, many did not quite understand the full nature of the reasons behind the practice. Further education through schools, TV, or displays could help in the dissemination of a message that moorland burning is an intrinsic part of the physical and social landscape of the uplands.

Conflicts, when they arise, are often a result of a disconnection between stakeholder groups and a possibly limited understanding of the other point of view. By bringing together stakeholders barriers can be broken and trust built up. During the scoping study for the RELU Sustainable Uplands project (2004-2009), stakeholders from different backgrounds were brought together during workshop settings. The view of many of the participants was summed up by one participant who said:

> *"This is the first time all these people have sat round the same table with each other. Until this project came along, I don't think any of us would have believed we'd be sitting here."* (Dougill et al., 2006)

This positive step forward has allowed further work and collaborations to carry on (e.g. Reed et al., 2009b) and to build up networks of partners in the Peak District National Park. By incorporating stakeholders into the process of evidence gathering, a greater ownership of the work can be gained. Transparency and reasoned debate must be allowed if stakeholders and end-users are to trust each other and go on to implement any change to policy (Hajkowicz, 2008). This multi-stakeholder approach needs to be used effectively and fed into decision-making process, otherwise the process will seem irrelevant to those parties involved (Fraser et al., 2006). Conflict management by necessity calls for approaches that address the 'social' nature of the conflict. However, for an improved understanding of these conflicts, an overarching view of the social, economic and ecological factors should be taken when trying to address the issues (White et al., 2009).

Chapter 7:

Conclusions

7.1 Introduction

The need for joined-up thinking when it comes to understanding complex environmental problems is increasingly important and the ability to understand the interlinked nature of socio-environmental problems is of particular importance in rural upland areas. Many authors already collaborate with others outside their own immediate sphere of work on upland issues (Bonn et al., 2009a). An interdisciplinary approach is needed to join up natural and social science.

Defining what is multidisciplinary work or interdisciplinary work can sometimes be difficult with these words often being used interchangeably, however, interdisciplinary work can be thought of as work that transcends traditional subject boundaries to allowing researchers to view the entire end product. This thesis has aimed to take an interdisciplinary approach to investigating the issue of managed burning of upland peat soils and its perceptions by stakeholders and visitors alike.

7.2 Review of objectives

- In chapter 2, the long-term and short-term effects of managed burning on soil water and runoff water quality i.e. DOC, were assessed. In addition to DOC and other water quality parameters, changes to hydrological parameters and the nature of runoff response was also investigated.
- Chapter 3 aimed to combine additional carbon pathways to the DOC data in order to create a complete carbon budget for each of the Hard Hill treatments.
- Chapter 4 saw the study move into the laboratory where the factors affecting biomass loss and char production were tested.
- Chapter 5 investigated the effects of wildfire on a moorland biomass and its implications for above-ground carbon stocks. Spatial variability of fire severity was also investigated.
- Chapter 6 took the results of the natural science study and sat them in a wider context. The implications for stakeholders were examined and the public's perceptions of managed burning were tested.

7.3 Findings and conclusions

7.3.1 Water quality before and after burning

Chapter 2 investigated the effect burning has on dissolved organic carbon (DOC) concentration both in the long-term and also the short-term post-fire changes. Prior to the burn in February 2007, sites showed no significant difference in DOC concentration between burning treatments. Water colour was lower on the 20-year plots and at the same time water tables were shallowest on these sites. Following burning of the 10-year plots, no significant increase was observed on these sites; there was a peak in concentration but this was short lived. These results would seem to suggest that, for wet blanket peat, burning is not a significant factor in DOC changes.

Results show that, following burning, there is an increased importance of runoff on recently burnt sites. When combined with the shift in source waters observed through chemical tracing, both the quantity and quality of runoff water appears to have changed following burning. This will have implications for the export of soil water and runoff water from catchments following burning.

This study adds significant data to a field where there is currently ongoing debate over the link between burning and DOC export. The mechanisms, or critical thresholds, leading to changes in flow pathways following managed burning in this setting, may not occur in other areas. This may explain why a variety of responses by DOC to burning have been observed: increases (Yallop and Clutterbuck, 2009); decreases (Worrall et al., 2007d) ; and no difference (this study; Ward et al., 2007)

7.3.2 Carbon budgets

In chapter three carbon budgets were calculated for the management combinations at Hard Hill. When only considering gaseous exchange, some burnt sites were net sinks of carbon; however, with the inclusion of hydrological export of carbon all sites are sources of carbon. The sources ranged from 37 to 182 gC m^{-2} yr^{-1}; however management e.g. burning, appeared to reduce the magnitude of this source.

This budget calculation does not include losses of carbon at the time of the burn through combustion of above-ground vegetation or any combustion of litter and surface peat layers. In the Grindsbrook wildfire (Chapter 5) the loss of carbon from above-ground biomass was approximately 200 gC m^{-2} suggesting that any benefits from burning will be offset for many years following the fire.

7.3.3 Biomass loss and char production

Experiments in chapter 4 describe the investigation into the controlling factors of biomass loss and char production. From experiments on over 570 samples, significant factors were found in the dry mass loss of vegetation. These were: burn temperature (ω^2 = 57.3%); moisture content (ω^2 = 9.6%); burn duration (ω^2 = 9.4%); initial temperature (ω^2 =1%) and return temperature (ω^2 = 0.2%). Significant interactions were also found. By using significant factors, from ANOVA, equations could be formed to predict dry mass loss. When combined with data from pTGA results and also with data from CHN analysis, it is possible to optimise all the equations to calculate the conditions that maximise char production. Short-duration burns between 450°C and 600°C would maximise char production. Burns in wind speeds of between 2.2 and 6.1 ms^{-1}, on strip widths between 10 and 30m and on heather between 5 and 15 years would generate the ideal temperatures. Land managers will aim to burn heather once it reaches about 30cm (Watson and Miller, 1976) and a 15-year cycle is within the range of suggested rotations lengths at which heather reaches this height (Tucker, 2003; Watson and Miller, 1976). Therefore this study suggests that current practice is suitable for char production.

7.3.4 Wildfire

The work on the Grindsbrook wildfire, Chapter 5, is the first study to address the question of black carbon production in a moorland wildfire in UK upland setting. It has helped in the formation of models that explore the fate of carbon in peatland ecosystems by providing field evidence for char production and biomass survival. Although 90% of the above-ground biomass was consumed during the fire, a high amount of dead biomass survived the fire leading to questions about how much this surviving biomass may play in carbon stocks. The black carbon, or char, was found to be around 4% of the carbon consumed (BC/CC) which is similar in scale to other wildfire settings around the world (Forbes et al., 2006).

By using novel techniques such as near-infrared reflectance spectroscopy, a possible measure of fire severity has been defined. This matches well with field observations in some areas, however, it is variable in others.

7.3.5 Impact on stakeholders and visitors

Chapter 6 detailed two studies looking at the wider implications of managed burning. The first looked at the implications of adopting some of the results from the natural science work where longer rotations may have benefits for carbon. Qualitative and quantitative methods were used to analyse the data. Results from the semi-structured interviews suggest that further work needs to be done before stakeholders will be willing to accept the use of data from Moor House due to its caveats. One of the main issues highlighted by moving to longer rotations was the build up of fuel load on the moors.

The second study, which looked at visitors perceptions of managed burning, generated some interesting results. On the whole the most of the visitors surveyed were aware of managed burning nationally (69%) though they often did not know quite understand why it was done. Further education was suggested as way to inform the public about this traditional management skill and as young people and first time visitors were groups that had perhaps not seen burning, education could be tailored for these audiences.

7.4 Data limitations

As with all experimental studies one area that can always be improved is that of repeat measurements. This study has aimed to have a high number of measurements (59 sampling days at Hard Hill; 570 samples for the burning experiments; 65 quadrats at Grindsbrook); however, more measurements are always welcome.

There are several problems associated with instrumenting a site with field equipment:

- Dipwells and gas collars inserted into the peat will naturally create disturbance that could affect readings. This was overcome by allowing a period of settling time though there will inevitably be some destruction to vegetation and/or peat.
- Where the equipment is sited may not be representative of the site or the management as a whole. A randomised design was used to help overcome this though sub-plot variations may influence the results.
- Each treatment plot was instrumented with three dipwells per plot. More dipwells will increase the number of data points and increase the reliability of the results. Allott et al. (2009) suggest using minimum of 15 dipwells to accurately monitor fluctuations in water table across a 30 by 30 m plot.

The social science investigation was small in scale so any conclusions from it should be treated with caution. Question and survey design will introduce a level of bias whether from how a question was asked or how something was explained.

7.5 Recommendations for future work

The findings in this study and their limitations can be used to direct a number of future research objectives. The present study on the Hard Hill plots examines the 10-year plots at the end of a burning cycle and the first year after burning. Following burning the importance of runoff increases and flow pathways diverge leading to changes in source water composition. However, how long does this effect last? Further study is needed to investigate what happens in the middle of burning cycle i.e. years 2 – 9.

One area for improvement to this study is to investigate how sheep grazing on blanket bogs affect carbon dynamics. Many authors have considered the effects of changing the grazing intensity, often investigating effects after exclusion of animals from an area and some studies have looked at the effect the interaction between grazing and managed burning has on vegetation (Grant and Hunter, 1968), DOC (Worrall et al., 2007d) and carbon accumulation (Garnett et al., 2000). These types of plot-scale experiments have inherent problems in being able to put numbers on effects. That is they investigate an “on-off” grazing system i.e. grazed or ungrazed, so are unable to quantify the effect for a known number of sheep per hectare.

The processes by which sheep grazing affects carbon fluxes is not entirely clear though physical processes such as compaction seem likely to be areas for investigation. Compaction by sheep or cattle increases soil bulk density with higher bulk densities found on higher stocking densities (Willatt and Pullar, 1984). Other physical changes include decreased air permeability, reduced infiltration, and changed bearing capacity (Willatt and Pullar, 1984). Pengthamkeerati et al. (2005) show CO_2 efflux is significantly reduced with increased bulk density and Hynst et al. (2007) observe lower CO_2 emissions on severely grazed plots. Further work investigating these observed effects is needed in blanket peat settings.

Chapters 4 and 5 show the importance of the addition of char and charred biomass to carbon stocks. It is an important carbon store due to its long mean residence time (Lehmann et al., 2008), however, processes such as wind erosion and fluvial processes can remove char from a site.   Whilst char is often well preserved in peat bogs, the rates of black carbon loss in peat bogs is poorly constrained (Preston and Schmidt, 2006).   Therefore, understanding how much char persists in the environment over time is an important question that warrants further study.

In chapter 5 a pilot study was carried out to investigate the use of near-infrared reflectance spectroscopy (NIRS) to predict fire conditions. The results show that there is potential to use this technique to calculate fire intensity; however, separating the effect of temperature and duration proved difficult. In order to address this problem, additional techniques could be used to investigate parameters that are insensitive to either temperature or time. Mid-infrared reflectance spectroscopy (MIRS) is one potential technique that has been used to analyse black carbon (Bornemann et al., 2008) and has benefits in that stronger vibration fundamentals are present (Ludwig et al., 2008) that could lead to greater understanding of changes to the functional groups present in the char and charred biomass. Carbon isotope values ($\delta^{13}C$) have been shown to vary during different degrees of carbonisation (Bird and Gröcke, 1997; Turney et al., 2006) and work on Eucalyptus wood has shown $\delta^{13}C$ values to be insensitive to heating duration (D. Gröcke, pers. comm.) suggesting that this could be a complementary technique to investigate compositional changes on heating.

Chapter 6 posed the “what if?” questions to stakeholders. While the sample size was relatively small, it did provide grounding for the results of the previous chapters. Based solely on the results of the natural science study, one suggestion that could be made to policy makers would be to lengthen the burning rotation so that it was in excess of 20 years. However, from the discussions with the stakeholders in the social science study, there may some significant problems in rolling out a policy like this across the uplands. As more burning management plans are taken up, more work must be done to engage with stakeholders in order to formulate suitable policy at a local and national level.

In order to gain a more complete understanding of the public's knowledge on managed burning, the surveys should be repeated during the burning season i.e. October to April, to determine if visitors to the uplands are more or less aware of why burning is happening.

Finally, it is also important to consider the wider scale and the external pressures that may occur in coming years and how the UK uplands and specifically the carbon stored in the peat responds to them. If changes to land management policy occur, such as a blanket ban on burning, what will be the impact on carbon storage? If under environmental schemes farmers are paid to manage their land for carbon, how will the decreases in sheep numbers affect CO_2 exchange? Will a warming climate encourage more visitors to our uplands leading to a greater number of wildfires?

Those who manage the uplands of the UK are facing continual change and will need to be prepared to adapt to these changes. With an increasing concern about carbon stores, they will need to balance the services the upland provides and traditional management techniques with wider issues such as climate change. This requires an understanding of the current impact of management on carbon stores in order to be able to predict future trends. In this context there is a need to open up new avenues of discussion in order to create sustainable strategies and policies that will adapt to any future changes.

Appendix 1:

Comparison of proportions from several independent samples

This appendix works through the method and an example taken from Fleiss (1981).

In this example, *m* independent samples of subjects are studies each with characterized by the presence or absence of some characteristic.

Sample	Total in sample	Number with characteristic	Number without characteristic	Proportion with characteristic
1	$n_{1.}$	n_{11}	n_{12}	p_1
2	$n_{2.}$	n_{21}	n_{22}	p_2
.				
.				
m	$n_{m.}$	n_{m1}	n_{m2}	p_m
Overall	$n_{..}$	$n_{.1}$	$n_{.2}$	$\bar{p}$

Table A1.1. Proportions from m independent samples

The proportion with characteristic is calculated thus:

$$p_i = \frac{n_i 1}{n_i} \qquad \text{(Eq. A1.1)}$$

and $$\overline{p} = \frac{\sum n_i . p_i}{\sum n_i}$$

(Eq. A1.2)

The formula for the test statistic is

$$\chi^2 = \frac{1}{\overline{p}\overline{q}} \sum_{i=1}^{m} n_i \left(p_i - \overline{p}\right)^2 \qquad \text{(Eq. A1.3)}$$

where $\overline{q} = 1 - \overline{p}$

To test the significance, reference is made to chi square tables with *m-1* degrees of freedom

An example set of data is given in Fleiss (1981) from a lung cancer study by Dorn (1954).

Study	Number of patients	Number of smokers	Proportion of smokers
1	86 (=$n_{1.}$)	83	.965 (=p_1)
2	93 (=$n_{2.}$)	90	.968 (=p_2)
3	136 (=$n_{3.}$)	129	.949 (=p_3)
4	82 (=$n_{4.}$)	70	.854 (=p_4)
Overall	397	372	.937 (=p_1)

Table A1.2 Smoking status among lung cancer patients in four studies

From these data, the value of χ^2 is

$$\chi^2 = \frac{1}{.937 \times .063}\left[86 \times (.965 - .937)^2 + 93 \times (.968 - .937)^2 + 136 \times (.949 - .937)^2 + 82 \times (.854 - .937)^2\right]$$
$$= 12.56$$

which, with three degrees of freedom, is significant at the 0.01 level.

Having found the proportions differ significantly, the next stage is to identify the samples or groups of samples that contributed to the significant difference i.e. post-hoc testing.

Suppose that the *m* samples are partitioned into two groups, the first containing m_1 samples and the second m_2 where $m_1 + m_2 = m$. Define

$$n_{(1)} = \sum_{i=1}^{m_1} n_{i.} \quad \text{(Eq. A1.4)}$$

to be the total number of subjects in the first group of samples and

$$n_{(2)} = \sum_{i=m_1+1}^{m_1} n_{i.} \quad \text{(Eq. A1.5)}$$

to be the total number of subjects in the second group.

Let the proportion in the first group be denoted $\overline{p}_1$, where

$$\overline{p}_1 = \frac{\sum_{i=1}^{m_1} n_i . p_i}{n_{(1)}} \qquad \text{(Eq. A1.6)}$$

and that the second group be denoted by $\overline{p}_2$, where

$$\overline{p}_2 = \frac{\sum_{i=m_1+1}^{m_1} n_i . p_i}{n_{(2)}} \qquad \text{(Eq. A1.7)}$$

Then

$$\chi^2{}_{diff} = \frac{1}{\overline{p}\overline{q}} \times \frac{n_{(1)} n_{(2)}}{n_{..}} \left(\overline{p}_1 - \overline{p}_2\right)^2 \qquad \text{(Eq. A1.8)}$$

with 1 degree of freedom, may be used to test for the significance of the difference between $\overline{p}_1$ and $\overline{p}_2$

The statistic

$$\chi^2{}_{group1} = \frac{1}{\overline{p}\overline{q}} \sum_{i=1}^{m_1} n_{i.} \left(p_i - \overline{p}_1\right)^2 \qquad \text{(Eq. A1.9)}$$

with *m_1-1* degrees of freedom, may be used to test the significant of the difference among the m_1 proportion in the first group, and the statistic

$$\chi^2{}_{group2} = \frac{1}{\overline{p}\overline{q}} \sum_{i=m_1+1}^{m} n_{i.}\left(p_i - \overline{p}_2\right)^2 \qquad \text{(Eq. A1.10)}$$

with m_2-*1* degrees of freedom, may be used to test the significance of the difference among the m_2 proportion in the second group.

For data presented in table A1.2, the first set of m_1=3 studies consists of

$$n_{(1)} = 86 + 93 + 136 = 315$$

lung cancer patients of whom the proportion who smoke is

$$\overline{p}_1 = \frac{83+90+129}{315} = .959$$

The second set of m_2 =1 consists of $n_{(2)}$ = 82 patients of whom the proportion is $\overline{p}_2$ = .854.

The significance of the difference between $\overline{p}_1$ and $\overline{p}_2$ is assessed by the magnitude of $\chi^2{}_{diff}$.

$$\chi^2{}_{diff} = \frac{1}{.937 \times .063} \times \frac{315 \times 82}{397}\left(.959 - .854\right)^2 = 12.15$$

The significance of the difference amongst p_1, p_2, and p_3 i.e. all from group 1, is assessed by $\chi^2{}_{group1}$.

$$\chi^2_{group1} = \frac{1}{.937 \times .063}\left[86 \times (.965 - .959)^2 + 93 \times (.968 - .959)^2 + 136 \times (.949 - .959)^2\right] = 0.41$$

Because group 2 only consists of a single study, χ^2_{group2} is not applicable.

Both χ^2_{diff} and χ^2_{group1} must be referred to the critical value of chi square with *m* -1= 4 -1 = 3 degrees of freedom. As the critical value for a significance level of .05 is 7.81, the conclusion would be that the proportion of smokers among the patients in study 4 differed from the proportions in studies 1 to 3 as χ^2_{diff} =12.15>7.81, but that there were no difference among the proportions in studies 1 to 3 (χ^2_{group1} =0.41<7.81).

In the runoff results presented in chapter 2, the sample (or study) was the burning, or grazing regime and the number with characteristic was the presence of a runoff sample in the runoff traps.

Appendix 2:

Managed burning interview questions

Managed burning Interview

PART ONE – General Burning Questions

	The first few questions I will ask you are about your role and how managed burning plays a part in this.	
1)	Can you explain your role in the uplands?	
2)	How long have you worked on in the uplands? (years)	
3)	How long have you worked in this [local to the interview location] area? (years)	
4)	How does your role link with managed burning in the uplands?	
5)	How does managed burning play a part in upland management in this area?	
6)	What are the main drivers behind managed burning in this area? [grouse, sheep, ?wildfire risk reduction, other]	
7)	What positive and/or negative impacts do you perceive burning to have in the local area? Are there any specific issues in this area?	
8)	How do other stakeholders perceive burning in the local area?	
9)	Do different stakeholder groups have much communication on the issue of burning?	
10)	How do the local community view burning?	
11)	How do visitors view burning?	

PART TWO – Burning techniques and rotation lengths In this section I'll ask you some specific questions about how land is managed in the local area/on your land		
12)	How do you carry out burning on your land?	
13)	What indicators after a burn are used to decide whether a burn has been successful?	
14)	What is length of a typical burning rotation on your land? If you are not directly involved in the land, what is the typical rotation length in your area of interest?	
15)	What would be the <u>ideal</u> rotation length to manage for your interests? [Not what is actually done, but what would like to be done]	
16)	If simple classifications of 'short', 'long' and 'never' were used to broadly group rotation lengths, what would the typical durations be for you/your area?	
17)	Which of these lengths of rotation produce a landscape ideal for grouse?	
18)	Which of these lengths of rotation produce a landscape ideal for sheep?	
19)	Which of these lengths of rotation produce a landscape ideal for wildlife? [Species other than grouse or sheep i.e. not reared or farmed species]?	
20)	Do 'short rotations' produce a landscape that is sought after for it aesthetic appeal? How about 'long rotation'? And 'never burnt' stands?	
21)	Would you like short, long or never burned landscapes to become more common across the UK uplands?	

PART THREE – Results of the natural science investigation One of the reasons for introducing the broad classifications in the previous section is to link with the results of the natural science investigation. In this study rotations of 10 yrs, 20 yrs and never burnt were compared. Results suggest longer rotations may be beneficial for carbon. (Explain results)		
22)	If policy were to be changed so that longer rotations were the only ones allowed, would this be a positive or negative change? Why?	
23)	What would benefit/loose out?	
24)	Who would benefit? Who would loose out?	
25)	Are there any incentives that would favour longer rotations?	
26)	What incentives would allow this future scenario to happen (e.g. financial, regulatory, cultural)	
27)	Could current policy allow these incentives to happen?	
28)	If not, what could be altered/changed?	
29)	Do you think these suggestions would be mirrored by any other stakeholder/ stakeholder group?	
30)	Where would any difference of opinions lie if these suggestions were adopted?	

Thank you for your time!

Appendix 3:

Visitor Survey

As part of a PhD project, I'm carrying out a survey of visitors and their knowledge of fire in the uplands. Please could you spare a few minutes to answer a few questions?

1. **Are you aware of any past wildfires in the UK uplands in the past 10 years?**

   Yes   No

2. **Are you aware of any past wildfires in the uplands in this local area in the past 10 years?**

   Yes   No

3. **Have you seen managed burning activities or evidence of managed burning in the UK uplands?**

   Yes   No

4. **Have you seen managed burning activities or evidence of managed burning in the uplands in this local area?**

   Yes   No

If Question 3 or 4 is answered yes, then continue. Otherwise go to question 12

5. **How would you rate the appearance of the landscape following managed burning? (please circle one and only one response)**

Immediately after a burn?	Very Poor	Poor	Average	Good	Very Good	Don't Know
One year after a burn?	Very Poor	Poor	Average	Good	Very Good	Don't Know
Ten years after a burn?	Very Poor	Poor	Average	Good	Very Good	Don't Know

6. How much does the smoke from managed burning affect your experience in the uplands? (please circle one and only one response)

It affects me a lot	It affects me a little	It doesn't affect me	Don't know

7. After managed burning, plants and animals recover in different ways. If you have seen examples of recovery following a fire please write them here.

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<u>Timing of burning</u>

Managed burning can only take part during certain times of the year (October – April), in order to reduce fire risk, and the frequency of burning typically ranges from 8-25 years

8. How often do you think managed burning should occur?

It's fine at the moment	Less frequently than present	More frequently than present	Don't know

9. How would you rate the effectiveness of CURRENT managed burning PRACTICES for habitat management?

Very Poor	Poor	Average	Good	Very Good	Don't Know

10. How would you rate the effectiveness of CURRENT managed burning STRATEGIES for fuel reduction and fire management?

Very Poor	Poor	Average	Good	Very Good	Don't Know

11. Are there any further comments about managed burning you would like to make?

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12. **Home town/Postcode (first part)**……………………………………………

13. **Male/Female (please circle one)**

Male Female

14. **Age (please circle one)**

< 25 25 - 34 35 - 44 45 - 54 55 - 64 > 65

15. **How many times in the past year have you been to this area?**

First time 1 2-5 6-10 10+

16. **What activities have you/are going to undertake whilst here?**

Walking	Climbing/bouldering	Draw/paint/photography
Picnic	Sightseeing	Cycling/mountain biking
Bird watching	Horse riding	Other (please specify)…………

17. **Do you work in the National Park?**

Yes No

18. **Are you a member of any environmental related groups?** E.g. National Trust, Wildlife Trust, etc

Yes No

Thank you for your time today in taking part in this survey. Your answers remain confidential.

For administrator

Location………………………… Questionnaire Number ………….

Date……………………………… Time……………………………..

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