Land Management Impacts on the Carbon Cycle in UK Blanket Peats

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ABSTRACT

Peatlands occupy a mere 3 % of the world's land mass, but store up to one third of terrestrial carbon stocks. Peatlands are widely regarded as carbon sinks owing to their ability to sequester more carbon than is released. Carbon cycling in peatlands is driven by environmental conditions e.g. water table levels, temperature and pH; substrate quality i.e. the ease with which microbes can synthesise the carbon; nutrient availability and the composition of the microbial community.

Peatlands are valued not only for their ability to sequester carbon, but also for the range of ecosystem services which they provide including the provision of food, recreation and leisure, a source of income for rural communities, water supply and as habitats for a range of flora and fauna. As a result, management of peatlands is widespread, with the four most common methods of management of upland blanket bogs being afforestation, drainage, grazing and burning. To date, little work has been carried out on the effects of such management practices on carbon losses or drivers of the carbon cycle. The aim of this research was to identify how these management practices influenced losses of carbon from peatlands as well as the chemical and physical drivers of the peatland carbon cycle. A combination of field and laboratory work was carried out on managed peats with an unmanaged site at the Moor House National Nature Reserve in Cumbria. Field monitoring involved measurement of dissolved organic carbon (DOC) in the peat solution, water table levels and carbon dioxide gains and losses. Laboratory analysis was carried out on cores of peat to examine nutrient concentrations, the structure of the peat in terms of porosity and density; carbon stocks and the quality of the carbon.

The results of this research demonstrated that all sites including the unmanaged site acted as carbon sources. Greatest losses occurred from the afforested site, where losses of DOC were significantly higher than all other sites and some of the highest losses of carbon dioxide were found. In contrast, the site that was burnt on a 10 year rotation was found to be a very slight carbon sink, held the most carbon within the peat and lost the least amount of DOC.

Few significant differences in the chemical composition of the peat were observed between the sites, however, lignin, the most recalcitrant fraction was found to be

significantly lower in the burnt (every 10 years) site, which had the highest carbon content. Lignin was identified as the dominant constituent of the peat for all the sites, with highest concentrations present in the afforested site. The high lignin content of the peats from all the sites indicated that the peats are in the latter stages of decomposition, and are thus fairly recalcitrant. The higher lignin content in the afforested site, coupled with the highest losses of DOC, some of the highest CO_2 losses through ER (ecosystem respiration), however, suggest that the chemical composition of the peat is not a strong a driver of the peatland carbon cycle.

Temperature was found to be the dominant driver of ER, accounting for between 54 and 92 % of variation in the data. The afforested site was the only treatment where a significant relationship between temperature and ER was not identified. Rates of primary productivity were highest in the burnt and grazed sites indicating that regeneration of the vegetation through management is of key importance in terms of sequestering carbon. The lowest primary productivity was identified at the drained site, where concentrations of nitrogen were also lowest. In terms of the structure of the peat, the air filled porosity of the burnt and grazed (every 20 years) site was greatest, however no linkages were established between the structure of the peat and gaseous carbon losses.

This thesis has provided a unique insight into the effects of land management on the drivers of the peatland carbon cycle, carbon dioxide gains and losses, and DOC production. Further work should focus on examining the effects of the intensity of land management practices on peatland carbon budget for example, comparing low and high temperature burns, or closely spaced drains with drains that are located far apart.

The results of this thesis suggest that future management needs to focus on encouraging increased PP by managing water table levels and promoting growth of peat forming species of vegetation such as *Sphagnum*. Light burning was also found to increase water table levels and peat solution acidity, thus reducing losses of DOC into the peat solution. The results demonstrated that temperature is the most important control on ER, and under climate change losses are likely to increase, therefore, the need to conserve carbon through increased PP is unquestionable. DOC

was found to be strongly linked to water table levels, pH and the carbon quality, with higher concentrations of holocellulose resulting in reduced losses of DOC.

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1.1 Climate Change and Global Carbon Stocks

Climate change is one of the most significant environmental problems facing the world today and moves are afoot to at least reduce, if not fully mitigate, its potential effects. Global concentrations of carbon dioxide in the atmosphere have risen from 280 ppm in 1750 to 379 ppm in 2005, and have been linked to a rise in global temperatures of 0.6°C since the end of the 19^{th} century (IPCC 2007). The onset of the industrial and agricultural revolutions has been cited as being responsible for this rise, alongside degradation of terrestrial carbon stocks, which has contributed to increased concentrations of atmospheric carbon dioxide (IPCC 2001).

Efforts are being made to preserve existing carbon stocks that are not stored in the atmosphere, and to expand the potential for carbon sinks (carbon stores where accumulation of carbon exceeds losses) to amass more atmospheric carbon (Walker & King 2008). Predictions of future emissions are based on a number of assumptions including estimates of how much carbon is currently held in different stores and what the impact of climate change might be on these stores (Manning et al. 2011). Global carbon stocks can be sub-divided into three components: terrestrial, atmospheric and oceanic, which in the absence of anthropogenic activity, approximately balance in terms of losses and gains across the carbon cycle (Lal 2004). During the 1980s, anthropogenic activity such as fossil fuel burning, cement production and land use changes accounted for losses of terrestrial carbon into the atmosphere of approximately 7.1 Pg C yr^{-1}, of which 5.3 Pg C yr^{-1} were due to annual emissions from fossil fuel burning (IPCC 2001). Based on data presented by Lal (2004) relating to carbon storage (Figure 1-1), annual losses of carbon due to fossil fuel burning accounted for 0.011 % of global carbon stocks (including terrestrial, oceanic and atmospheric stores). More recently, losses of carbon due to fossil fuel burning have risen, in 2009, 8.4 Pg C yr^{-1} were lost, which represented a 1.3 % decrease from 2008. The decline has been attributed to the downturn in the global economy and hence industrial outputs (Friedlingstein et al. 2010). Despite this apparent fall in carbon losses, there have been significant rises in carbon emissions from developing economies in Asia such as China, where economic output increased by 9.1 % in 2009 (Friedlingstein et al. 2010). The rise in fossil fuel burning since the 1980s has resulted in annual carbon emissions from fossil fuel burning, increasing by 58 %, with annual emissions now accounting for 0.018 % of global carbon stocks.

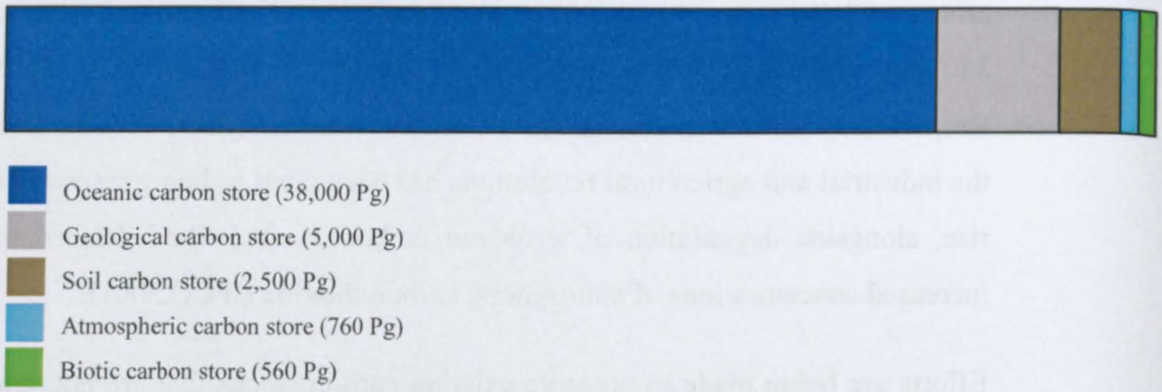


Figure 1-1 Schematic of the quantity of carbon held in each of the five global carbon stores (after Lal, 2004)

1.2 The Importance of Terrestrial Carbon Stocks

Emissions of carbon need to be reduced by finding more sustainable methods of industrial production and energy sources. Fossil fuel burning alone, however, is not the only cause of increased carbon emissions to the atmosphere. During the 1980s land use change accounted for losses of 1.7 Pg C yr^{-1} (IPCC 2001) from soils and the biosphere. Agriculture, tropical forest removal, ploughing soils, draining wetlands and burning biomass have all been cited as causes for increased losses of carbon from terrestrial stocks (Lal 2004). In each case, land management has caused reductions in terrestrial carbon stocks by reducing the capacity of soils to take up and retain carbon, and changes in vegetation species, have resulted in a decrease in the quantities of carbon sequestered (Lal 2004).

Carbon is fixed from the atmosphere by plants during photosynthesis and forms part of the soil organic matter once the plant has died and been decomposed (Wild 1993). The decomposition of plant material by microbes represents the main pathway through which carbon dioxide is returned to the atmosphere as a result of microbial respiration. This release of carbon has been cited as one of the dominant fluxes in the carbon cycle (Schlesinger & Andrews 2000). The sensitivity of carbon stored in the organic matter of soils to increases in temperature has been the focus of much research into the effects of climate change on soils. Davidson and Janssens (2006) clearly illustrated the potential for positive and/or negative feedback cycles to occur within the soil carbon cycle. Summaries of these feedback loops are presented in Figure 1-2 and Figure 1-3.

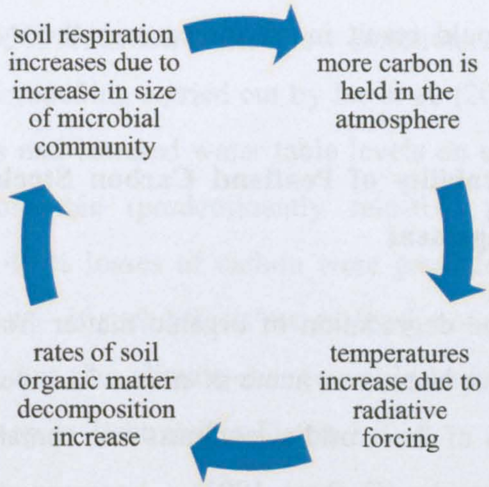


Figure 1-2 Positive Feedback Loop in Response to Warming under Climate Change (after Davidson & Janssens 2006)

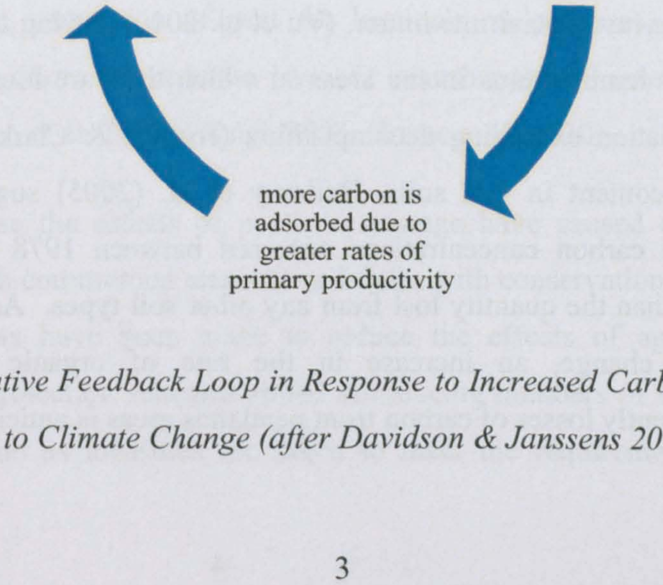


Figure 1-3 Negative Feedback Loop in Response to Increased Carbon Dioxide in the Atmosphere due to Climate Change (after Davidson & Janssens 2006)

Increased temperatures result in faster rates of microbial decomposition (Kirschbaum 2006), with the most labile (easily degradable) substances being synthesised more rapidly than complex structures which have a higher activation energy (Davidson & Janssens 2006). The importance of substrate quality and the existence of three pools of carbon (fast, intermediate and very slowly degradable carbon) has been highlighted by many authors (e.g. Knorr et al. 2005, Powlson 2005, Fang et al. 2005). Each has suggested that the more recalcitrant (i.e. slowly degradable pools) are more sensitive to temperature rises, and thus, increases in temperature under climate change could result in an increase in the synthesis of previously stable carbon stocks.

1.3 The Vulnerability of Peatland Carbon Stocks to Climate Change and Land Management

Concerns about the degradation of organic matter from soils and the response to climate change have been most acute in areas where peatlands are found. Peatlands cover a mere 3 % of the world's landmass but contain an estimated one third of terrestrial carbon stocks (Gorham 1991). Approximately 96.5 % of peatlands are located in northern Europe and North America (Taylor 1983). In the UK, 2.2 million hectares of blanket bog exist (Shepherd et al. 2010), comprising between 10 and 15% of the world's blanket bogs (Tallis 1998). In total, it has been estimated that approximately 2,302 Mt of carbon are stored in UK peatlands (Billett et al. 2010).

Peatlands store large quantities of carbon as peat is predominantly comprised of organic matter. Rates of organic matter decomposition are typically very low in northern peatlands (which have stored approximately 89 % of global peatland carbon since the last glacial maximum, (Yu et al. 2010)) owing to the saturated conditions and low temperatures in the areas in which they are located, resulting in rates of accumulation exceeding decomposition (Joosten & Clarke 2002). In a survey of carbon content in UK soils, Bellamy et al. (2005) suggested that decreases in peatland carbon concentrations occurred between 1978 and 2005; and were far greater than the quantity lost from any other soil types. As temperatures rise due to climate change, an increase in the rate of organic matter breakdown and consequently losses of carbon from peatlands areas is anticipated.

Bio-climatic envelope modelling is typically used to predict the effects of climate change on the spatial distribution of ecological species (Pearson & Dawson 2003). Recent work carried out by Gallego-Sala et al. (2010) used bio-climatic envelope modelling to predict the impacts of climate change on the areal extent of peatlands in the UK under high and low emissions scenarios. Data taken from the UK Climate Impacts Programme 2002 projections for the UK were run with a bio-climatic envelope model. The results indicated that by 2080, there would be an 84 % reduction in areas suitable for peatland development under a high emission scenario (worst case) and a 53% reduction under a low emissions scenario (Gallego-Sala et al. 2010). Simulation modelling carried out by Ise et al. (2008) looked at the effects of higher temperatures and lowered water table levels on carbon losses from northern hemisphere, ombrotrophic (predominantly rain-fed) peatlands. Under a 4°C warming scenario, 40 % losses of carbon were predicted from shallow peats, and 80 % from deep peats. If such effects are realised, not only would valuable carbon stocks be degraded, but also climate change would be exacerbated as a result of the positive feedback loop as illustrated in Figure 1-2.

Upland areas of the UK including peatlands have been influenced by humans since Palaeolithic times (Simmons 2003), as efforts have been made to forge a living from these areas (Maltby 2010). Today management of peatlands is extensive, with estimates of the total managed area exceeding 82 % of UK peatlands (Bragg & Tallis 2001). Much concern exists over the current state of peatlands as a result of land management practices (Holden et al. 2007b). Initiatives to increase and diversify agricultural production have resulted in damage to peatlands including erosion, changes in vegetation, and increased losses of carbon. Furthermore, over-grazing, drainage of peat, extraction of peat for horticultural purposes, recreational use, burning, and windfarm construction have all contributed to the deterioration of UK peatlands (Holden et al. 2007b, Haigh 2003, House et al. 2010).

Efforts to reverse the effects of peatland damage have caused conflicts between stakeholders with commercial interests and those with conservation interests (Maltby 2010). Attempts have been made to reduce the effects of agriculture through environmental protection schemes aimed at reducing numbers of livestock, and this is set to continue as measures are taken to meet the requirements of the Water

Framework Directive (Reed et al. 2009). In addition, protection schemes have been awarded to many UK peatlands such as National Parks, Areas of Outstanding Natural Beauty, National Nature Reserves, Ramsar Sites and Sites of Special Scientific Interest (Orr et al. 2008). Ecosystem services approaches are at the forefront of recent efforts to manage peatlands (Ostle et al. 2009). Such approaches aim to combine the interests of multiple stakeholders and acknowledge the competing demands placed on upland areas including provision of food, recreation and leisure, a source of income for rural communities, water supply and a sensitive habitat in need of environmental protection (Maltby 2010).

Carbon budget calculations for blanket peatlands in the UK have primarily focussed on unmanaged (or relatively undisturbed) peatlands (e.g. Worrall et al. 2009, Dinsmore et al. 2010). Such calculations have indicated that blanket peatlands are net carbon sinks i.e. more carbon is captured than released; however, losses vary on a seasonal basis. Few calculations have been carried out on more intensively managed sites (e.g. Clay et al. 2010b, Rowson et al. 2010), and those that have, have not compared the four most common methods of blanket peatland management (burning, grazing, drainage and afforestation) within one study. Furthermore, the effects of management on the drivers of the carbon cycle have not been fully investigated on blanket peatlands. Through a better understanding of the causes for variations in carbon budgets due to land management, scientific knowledge can be used to inform and guide future conservation efforts, policy development and stakeholder dialogue and decision making.

1.4 Aims and Objectives

1.4.1 Aim

The aim of this thesis is therefore to understand how the key drivers of the carbon cycle vary between differently managed peatlands. The work primarily focuses on the four main methods of management of blanket peatlands in the UK (grazing, burning, drainage and afforestation); but also considers burning frequency and combinations of burning and grazing. Differences in the chemical and physical properties of managed peatlands will be used to aid understanding of losses of carbon dioxide and dissolved organic carbon (DOC) from peatlands.

Land management practices are commonly described as being detrimental to peatlands, resulting in degradation such as erosion, discoloration of water, loss of habitat, and pollution (Haigh 2003, Holden et al. 2007b, House et al. 2010). As a consequence, much emphasis has been placed on peatland restoration. Much of the peatland restoration work carried out to date has aimed to protect peatlands from the effects of climate change and to restore peatlands from an environmental perspective. More recently, the focus has turned to identifying the value of peatlands from an ecosystem services perspective (Bonn et al. 2009).

Carbon cycling in peatlands is driven by a combination of chemical, biological and physical processes. Carbon losses occur as a result of microbial synthesis of organic matter resulting in the release of carbon dioxide through respiration, and partially degraded organic compounds into the peat solution in the form of DOC. The biological component of the cycle is thereby represented by the composition of the microbial community (Laiho 2006). In order to ensure the microbial community is appropriate for organic matter decomposition, the correct physical and chemical conditions to support that community need to be present (Blodau 2002).

The physical blanket peatland environment is primarily water-logged and cool, owing to the location of these peats in upland areas with low temperatures and heavy rainfall. The cool, saturated conditions limit rates of microbial activity (Blodau 2002), while the saturated conditions can also impair the ability of gases to diffuse through peat profile due to low porosity (Iiyama & Hasegawa 2005) .

The chemical composition of the peatland carbon cycle is represented by nutrient status and the composition of the peat substrate. Peatlands are renowned for being nutrient poor ecosystems upon which only selected plants can survive owing to the paucity of nutrients, and the cold, waterlogged and acidic environment (e.g. Gorham 1991, Charman 2002). In order to synthesise the organic matter inputs into the peat from decaying plants, microbes rely on a supply of nutrients. The degree to which the organic matter can be synthesised depends on the composition of the substrate. Labile substrates are deemed to be of high quality and are easily synthesised (e.g. holocellulose); while recalcitrant substances (e.g. lignin) have larger and more complex chemical structures, and as such, are more difficult for microbes to degrade

(Berg 2000). Substrate quality is a direct function of the plant community growing on peatlands, which in turn is dependent on the availability of nutrients and to some extent the acidity of the local environment (Laiho 2006).

Land management has unquestionably influenced the chemical, physical and biological characteristics of the peatland carbon cycle as well as losses of carbon. How losses of carbon compare between differently managed peats is poorly understood and little is known about how land management affects the chemical and physical drivers of the carbon cycle. By concentrating on the main carbon loss pathways, information can be attained which will enable future conservation efforts to identify which management practices are the most damaging from a carbon cycling perspective. Analysis of the physical and chemical drivers of the peat carbon cycle will enable an understanding of the dominant drivers of the carbon cycle in managed peatlands and to assist in identifying whether they are different from those in unmanaged peatlands. Furthermore, by understanding which properties are most affected by management, future restoration work can focus on addressing these parameters, and as such, seek to promote carbon storage in the future.

The aim of this thesis is to be achieved through six objectives which are detailed below alongside a rationale for each.

1.4.2 Objectives

Objective 1. *To establish how concentrations of nutrients required in the carbon cycle for synthesis of carbon stocks compare between differently managed peatlands within the upper 50 cm of the peat profile.*

The role of nutrients (both micro and macro nutrients) in peatland carbon cycling is twofold. Firstly, nutrients are essential to plant growth, and secondly they are required by microbes to synthesise organic matter (Blodau 2002). Examining the differences in nutrient concentrations between differently managed peats and a an unmanaged site will allow the effects of land management to be determined. By understanding how changes in land management affect nutrient concentrations, differences in carbon losses may be accounted for and the data provided will aid future conservation efforts. A comparison of surface concentrations with those at depth (to the top of the permanently saturated zone) permits an assessment to be made as to whether land management has influenced the whole of the saturated zone or just the surface of peatlands. Such data could be useful in the future when identifying the extent of restorative works required to increase carbon sequestration and reduce carbon losses.

Objective 2. *To investigate what differences exist in the carbon stocks of differently managed peatlands and to identify how carbon quality varies as a result of peatland management, with a focus on establishing which peats are the most recalcitrant.*

While measurements of carbon losses have been made for some managed peats and comparisons made between one or two treatments (e.g. Ward et al. 2007, Clay et al. 2010b, Rowson et al. 2010); few data exist on how the quantity of carbon stored in peats is influenced by management. Such data will allow updates to be made to estimates of carbon stocks within the UK for areas where data on management are available. In addition, efforts to restore peatlands can focus on those where carbon stocks are evidently depleted, and those where there is most carbon, can be conserved. While identification of the effect of land management on carbon stocks is important, understanding how the quality of carbon varies is vital. Less recalcitrant carbon species are likely to be rapidly depleted through microbial decomposition (Kirschbaum 1995), resulting in greater losses of carbon in both fluvial and gaseous forms. At present, data on substrate quality between differently managed peatlands have not been published. Studies to date have focussed on either the composition of the carbon stored in the vegetation or on the rate of degradation of substrate materials within different peatlands i.e. the response of substrate to environmental conditions has been analysed (Laiho 2006) but the composition of the actual peat itself has rarely been assessed.

Objective 3. *To identify the effect of land management on the physical drivers of the peatland carbon cycle.*

The physical structure of peat is partially responsible for regulating rates at which carbon is lost. The physical structure of peat in terms of bulk density and porosity is rarely considered, and as such, few data exist in published literature on the effects of land management on the physical structure of peats. The structure of managed peats is anticipated to be affected by trampling and compaction caused by grazing sheep (Evans 2005), inputs of ash (Mallik & FitzPatrick 1996) and compression due to drains (in both drained and afforested sites) and the presence of trees (Minkkinen & Laine 1998). Changes in the structure are anticipated to affect the transport of fluvial and gaseous forms of carbon through the peat profile. By understanding the nature and extent of the changes to the physical environment, the causes for any differences in carbon loss rates between managed peats should be better understood.

Objective 4. *To establish how peatland management affects carbon dioxide losses and environmental controls on carbon dioxide losses.*

Carbon dioxide is one of the dominant forms of carbon lost from peatlands, and accounts for between 56 % (Worrall et al. 2003b) and 77 % (Dinsmore et al. 2010) of the carbon budget in UK blanket peatlands. To date, research in the field on losses of carbon dioxide from differently managed peats has been confined to comparisons of one or two treatments, sometimes with an unmanaged site. Comparisons between burnt and grazed sites have been contradictory, some studies have suggested that management reduced carbon dioxide losses compared to unmanaged sites (Ward et al. 2007) while others found that less carbon dioxide was lost from unmanaged sites (Clay et al. 2010b). Clarity needs to be sought through additional measurements of burnt and grazed sites, but also comparisons need to be made with drained and afforested sites to provide an overview of the impact of the four main methods of land management on peatland carbon dioxide losses. Whilst examining losses of carbon dioxide through respiration, gains should also be measured to identify whether sites that are losing the most carbon dioxide are adsorbing more or less, thus providing an insight into the carbon dioxide balance of managed peats. Thus objective four provides critical information on the impact of management on carbon gains and losses and consequently allows the impact of peatland management on climate change to be considered.

Objective 5. *To determine how concentrations of DOC in peat solution varies with depth and between managed sites.*

DOC is lost from peatlands by throughflow, overland flow and through soil pipes (Holden et al. in review) before being released into streams and rivers. In recent decades, significant increases in DOC losses from peatlands have been recorded (Evans et al. 2005). Management of peatlands has been cited as one possible cause for observed increases, yet comparisons between the peat solutions of differently managed sites have not been carried out for the four main methods of peatland management in the UK and compared to an unmanaged site. By examining differences between managed and unmanaged sites, a baseline can be established to identify how DOC concentrations are affected by land management. In addition, by analysing variation with depth, the influence of management on different parts of the peat profile can be identified.

Objective 6. *To examine changes in the water chemistry of managed peatlands, with a focus on the properties that are relevant to DOC loss.*

Of the many theories that have been proposed to explain increased levels of DOC in peatland streams, alterations to the peat solution chemistry appear to be one of the most credible and favoured (Evans et al. 2006a). Reductions in the water table have been observed to cause the onset of sulphur reduction to sulphate, and subsequently cause the pH of the peat solution to fall. Increasingly acidic conditions reduce rates of microbial activity and hence losses of carbon. Once water table levels recover, however, the pH and rates of microbial activity increase, and subsequent losses of DOC have been observed to exceed those recorded prior to the lowering of the water table (Clark et al. 2006). Modifications to the peatland environment have been recorded as a result of land management, in particular, water table levels (Worrall et al. 2007a, Holden et al. 2011) which are likely to simulate drought conditions and/or cause changes to the chemistry of the peat solution. The effect of land management on peat solution chemistry, in particular sulphate and pH is relatively unknown, but could provide valuable knowledge to aid understanding of the causes for observed increases in DOC concentrations, and potentially identify where losses might be greatest in the future.

1.5 Proposed Methodology to Address Aims and Objectives

This thesis provides an assessment of carbon losses and drivers of the carbon cycle across four differently managed sites (burnt, grazed, drained and afforested) with a unmanaged site for comparative purposes. The selection of one field location where all four key management practices are employed provided a unique opportunity to make comparisons between treatments without confounding factors such as differences in climate and geology influencing the results.

To fulfil the aims and objectives described in Section 1.4, a combination of field monitoring, peat core collection and laboratory analyses were carried out. The drivers of carbon cycling were studied through analyses of the chemical properties of the peat and peat solution (objectives 1 and 2) and the physical properties of the peat (objective 3). Data on environmental conditions in the field were collected through a combination of on-site monitoring and through the provision of data from the local weather station. Monitoring of gaseous carbon gains and loss (objective 4) and peat solution chemistry and aqueous carbon losses (objectives 5 and 6) was carried out in the field.

1.6 Thesis Outline

Chapter 2 provides a review of published literature on the formation and location of peats, the carbon cycle and its drivers, a comparison of peatland carbon budgets and details of peatland management in the UK. Chapter 3 introduces the chosen field site and provides a rationale for the selection of the site, and background information on the location, climate geology, soils and management practices of the site. Details of the fieldwork, sampling design and equipment that was installed to address objectives 1 to 6 are also provided.

The results of analysis of nutrients and acidity in peat samples collected from differently managed peats to fulfil objective one are presented in Chapter 4. A discussion of differences in concentrations between and within each management practice is provided, and differences with depth are discussed. The effects of combining burning and grazing and altering the frequency with which peatlands are burnt are also examined.

Chapter 5 addresses objective two by providing an assessment of variations in carbon stocks between differently managed peatlands and with depth. The results from experiments to characterise the quality of the carbon in samples collected from the surface of the four main management practices and unmanaged site are presented and discussed to determine whether management affects the recalcitrance of carbon stocks in peats.

Differences in losses and gains of carbon dioxide between differently managed sites are presented in Chapter 6 in order to address objective four. The results of analyses of the peatland environment in terms of temperature, water table and the density and porosity of the peat are presented and discussed to fulfil objectives three and four; and to provide some interpretation of the results of the measurement of carbon dioxide losses and gains.

Data on DOC concentrations in the peat solution in managed peats are presented in Chapter 7, alongside results from analysis of water samples for pH and sulphate, and data on water table depth which are used to provide an explanation of differences in DOC concentrations in the peat solution. Differences in peat solution chemistry between differently managed sites are discussed and incorporate objectives five and six.

Chapter 8 provides a summary of the key findings of each of the research chapter and discusses the implications for carbon budgets and drivers of the carbon cycle. The importance of seasonal fluctuations in carbon losses and the limitations of the generic diplotelmic peatland model are also discussed.

Chapter 9 summarises the findings of the work presented in Chapters 4 to 7, and offers some final conclusions and recommendations for further work.

2 THE EFFECTS OF LAND MANAGEMENT ON THE DRIVERS OF CARBON CYCLING IN PEATLANDS

2.1 Introduction

This chapter provides a critical review of research on carbon cycling in peatlands with a focus on the drivers of the carbon cycle. The main pathways through which carbon can be lost are examined, and the factors which determine rates of loss considered. The most common methods of peatland management in upland Britain are introduced and attention is given to the research that has been carried out for each management practice. Finally, the impacts of each management method on the key drivers of the carbon cycle are noted.

2.2 The Formation of Upland Peats

Peats are highly organic, nutrient poor, acidic soils formed from degraded plant materials under waterlogged conditions. Peat formation occurs by one of two methods: hydroseral succession (also known as terrestrialisation) or paludification (Charman 2002). Terrestrialisation occurs when surface water bodies infill with organic material and the ecosystem is transformed from being an aquatic to a terrestrial peatland. Initially, mats of peat develop before the entire basin becomes infilled with organic matter. The final transition into a peatland depends on the water table remaining sufficiently high enough to allow organic matter to continue accumulating at a faster rate than it is decomposed (Rydin & Jeglum 2006).

Paludification refers to the formation of peat over mineral strata without the presence of water-logged conditions prior to initiation. It is the most common method of peatland formation, and is often found in areas that have been previously afforested (Charman 2002). There are four means by which paludification can occur: local climatic change, upslope paludification, the presence of iron pans, and anthropogenic activity. Changes in local climatic conditions which cause subsequent changes in local hydrological conditions are the most common causes for paludification. Such conditions allow the accumulation of organic matter and debris at a rate that exceeds decomposition (Charman 2002). Upslope paludification is the process under which peatlands expand across mineral soils, and is enabled by rising water table levels at the same time as the peat rises. Under such conditions, adjacent mineral soils become water-logged, thus creating suitable conditions for further peat growth. Paludification may also be instigated due to the formation of oxide pans within mineral soil profiles. Pans create an impermeable layer within the soil due to the binding of mineral soil particles with aluminium, iron or manganese oxides. The layer prevents infiltration of water, which in turn induces saturated conditions under which peats can form, provided there are sufficient inputs of vegetation. Anthropogenic activity may also cause paludification. Tree clearance in areas where rainfall rates are sufficient to increase the wetness of soils, can lead to succession with plant species that are commonly associated with peats. This method of paludification is thought to be responsible for the development of many of the UK's peatlands (Rydin & Jeglum 2006).

Peatlands may be classified on the means through which they receive inputs of water: ombrotrophic and minerotrophic. Ombrotrophic peatlands rely almost entirely on precipitation for inputs of water, nutrients and minerals. Minerotrophic peatlands are not hydrologically isolated from the underlying strata and are hydrologically connected to other strata. Ombrotrophic peats have a high water content due to their location in areas with high rates of precipitation (Bragg & Tallis 2001). Minerotrophic peats receive water not only from precipitation but also from telluric sources. The connection between minerotrophic peat and underlying strata results in peats with a greater nutrient content than ombrotrophic peats. Ombrotrophic peats are typically referred to as bogs, which are acidic (pH<4) and often dominated by *Sphagnum* mosses with a combination of sedges, herbs and woody plants. Minerotrophic peats are typically referred to as fens, with a pH ranging from 4.0 to 6.0 i.e. acidic to slightly acidic. Fens support a wider variety of plant species including grasses, sedges, herbs, mosses and woody species. The nutrient content of fens is often greater than bogs owing to inputs of nutrients within groundwater, in addition to the atmospheric inputs on which bogs rely on (Wheeler & Shaw 1995).

Within the ombrotrophic bog category, there are two sub-categories: raised bogs and blanket bogs. Raised bogs have a distinctive convex profile that has a dome-like appearance. They can only develop in areas where precipitation exceeds rates of evaporation and runoff. Blanket bogs cover the entire landscape of a particular area, from mounds and slopes to valley bottoms (Charman 2002).

Many peatlands feature distinctive microforms, resulting in what is termed as a patterned peatland. The three most common microforms are: hummock, lawn and hollow. Hummocks are dry mounds which feature a thick aerobic surface layer that is described as fairly decay resistant. Hollows are water filled depressions, and lawns occupy the area in between hummocks and hollows (Belyea & Clymo 2001). Hollows have sparse vegetation dominated by mosses while lawns often feature graminoids and bryophytes (Rydin & Jeglum 2006).

Figure 2-1 Location of Blanket Bogs in Britain (Holden 2005b)

Blanket peatlands occupy approximately 8 % of the UK landmass (Figure 2-1). Owing to the climatic conditions required for peatland formation, most UK peats are confined to areas of the north and west, with 10.4 % of Scotland and 7.7 % of Wales covered by peatlands. These areas of the country are also suitable for peatland formation due to the presence of substantial areas of flat or slightly concave land situated at high altitude (Taylor 1983). In upland Britain, blanket peats tend to be located across plateaux or in gentle depressions, but rarely extend on to steep slopes (Moore 1975). Approximately 87 % of UK peatlands are blanket peats (Baird et al. 2009).

Peatlands are characterised by the presence of a thick layer of organic material that has accumulated over hundreds to thousands of years. The organic matter content of peats is often 50 to 60 %, in some cases more (Shepherd et al. 2010) resulting in soils with a high carbon content. Initially decay takes place under aerobic conditions, but as the structure of the plant material begins to collapse, the bulk density increases, and the partially decomposed matter is pushed beneath the water table (Clymo 1984). UK peatlands grow at an average rate of approximately 1 mm a year, depending on rates of decay (Charman 2002). Peat typically grows as the organic matter from decaying plants accumulates at faster rates than the plant materials are degraded due to the water-logged environment (Moore 2002) and to the inability of the microbial community to decompose peat at a rate that matches or exceeds primary production (Moore 1975).

Peat profiles are often sub-divided into two sections, the upper layer is referred to as the acrotelm. The acrotelm is an aerobic layer where most plant degradation takes place. This layer has a fluctuating water table, high hydraulic conductivity, a plentiful supply of micro-organisms synthesising inputs of litter and has live plants growing on it (Ingram 1978). The catotelm is located beneath the acrotelm, and is permanently saturated, consequently, anaerobic conditions prevail. Rates of degradation in the catotelm are much slower than those found in the acrotelm. The boundary between the two layers rises as the peat accumulates (Clymo 1984).

The division of peats into two distinctive layers (the acrotelm and –catotelm) is described as a diplotelmic model, that has been widely accepted by peatland scientists since it was first proposed approximately 60 years ago (Morris et al. 2011). The model provides a useful basis on which to begin to explain concepts relating to peatland ecohydrology, and appears to be widely accepted amongst the peatland researchers, however, suggestions have been made that the model is oversimplified (Holden & Burt 2003, Morris et al. 2011). The model suggests that a simple division exists between saturated and unsaturated zones, without considering the role of peat pipes providing linkages between the profile, or the role of preferential flow pathway in governing infiltration and runoff (Holden & Burt 2003). In addition, the diplotelmic model assumes that the boundary between the two zones is static, however much evidence exists to suggest that this is not the case, and the surface of the peatland fluctuates seasonally in response to changes in the volume of water stored in the peat. The presence of microforms across patterned peatland is ignored by the diplotelmic model yet they have a significant bearing on peatland hydrology and decomposition (Morris et al. 2011), thus caution needs to be taken when interpreting data using the traditional diplotelmic model.

2.3 Carbon in Peatlands

Owing to their high organic matter content, peatlands store vast quantities of carbon. Despite only occupying 3 % of the world's landmass, peats store one third of global terrestrial carbon stocks (Gorham 1991). Carbon is lost from peatlands in gaseous and aquatic forms. Where accumulation of carbon exceeds carbon losses, the peatland is referred to as a carbon sink, and where carbon losses exceed carbon gains, the peatland becomes known as a carbon source.

Peatland carbon budgets provide an indication of whether a peat is a source or a sink of carbon. Calculations of carbon budgets vary greatly depending on the site, the components of the budget that are measured, the scale of the measurement and the methods of measurement used, resulting in some sites being considered as both sinks and sources between different assessments. A summary of the main components of peatland carbon budgets is presented below, to underpin a discussion of budget calculations that have been published to date.

2.3.1 Main Components of the Peatland Carbon Budget

Figure 2-2 highlights the main pathways through which carbon may be lost from upland peats.

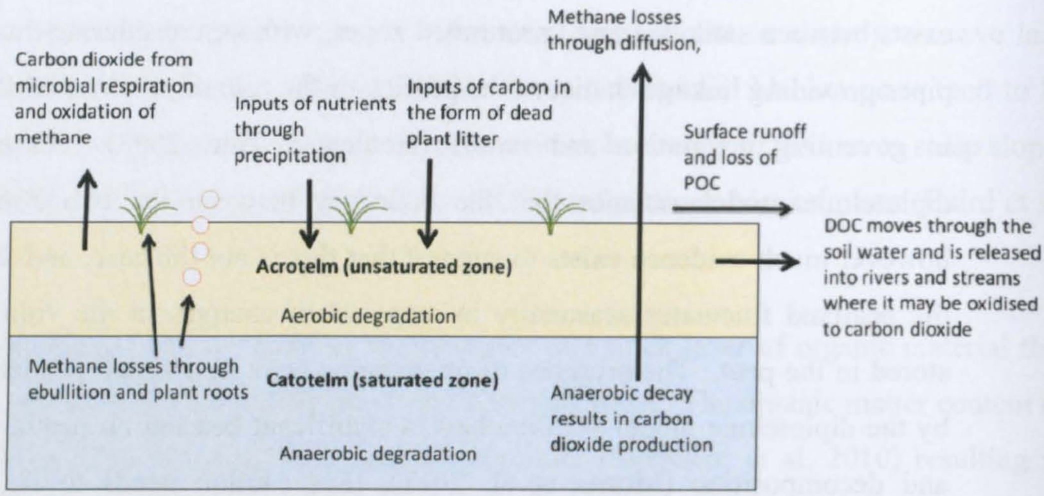


Figure 2-2 Simplified Carbon- Loss Pathways from an Ombrotrophic Peatland

2.3.1.1 Carbon Dioxide

Carbon dioxide is primarily lost from peats as a result of the decomposition of organic matter by micro-organisms and respiration from roots (Moore et al. 1998). The rate of decomposition depends on a number of factors including substrate quality, aeration, peat chemistry, pore-water chemistry and the community of microbes involved in the breakdown (Yavitt et al. 2000). In addition, environmental controls such as temperature and moisture content are also key drivers of carbon cycling (Laiho 2006). Net Ecosystem Exchange (NEE) is the difference between the amount of carbon gained through primary production and that lost through ecosystem respiration. Further details of the drivers of carbon dioxide production in peats are presented in Section 2.4.1.

2.3.1.2 Methane

Methane is produced within the anaerobic zone of the peat, and moves up through the profile and is released into the atmosphere through diffusion, ebullition or vascular plant roots. The rate of methane production is controlled by methanogen and methanotroph activity within the peat profile, as well as rates of methane transport (Moore et al. 1998). Methane emissions are controlled by environmental conditions such as temperature and the level of the water table, and the composition of the peat itself (Bellisario et al. 1999). Lower water table levels are commonly associated with decreases in methane emissions (Gorham 1991). Methane may be oxidised to carbon dioxide as it passes through the acrotelm. Despite comprising only a small part of the carbon budget (see section 2.3.2), methane has a much greater global warming potential (GWP) than carbon dioxide. Losses of methane have been calculated as having a 25 times greater GWP than carbon dioxide over a 100 year timescale (Baird et al. 2009). Methane is however less persistent in the atmosphere than carbon dioxide (Walker & King 2008).

2.3.1.3 Dissolved Organic Carbon (DOC)

DOC may be described as a complex collection of organic carbon molecules produced as a result of plant decay (Moore et al. 1998). DOC is lost from the peat profile and into the peat solution as a result of microbial synthesis of carbon, root exudation, leaching during storm events and as a consequence of erosion of soil organic matter (Hope et al. 1994). The release of DOC from peatlands results in the waters of peatland catchments having a characteristic brown colour, which is sometimes used as a surrogate measure of DOC concentration (Wallage & Holden 2010). Causes of DOC losses remain poorly understand and are the subject of much speculation (Blodau 2002). Further discussion of possible causes of DOC losses are provided in Section 2.4.2.

2.3.1.4 Particulate Organic Carbon (POC)

POC is lost from peatlands as a result of erosion. Carbon adheres to the eroding particles that are removed as overland flow. Numerous gaps exist in current understanding of POC transport from peatlands into watercourses, thus assessments of the contribution of POC to carbon budgets are often prone to errors (Evans & Warburton 2007). The release of carbon from land into riverine environments is thought to be dependent on the nature of the catchment, local climatic conditions, rates of discharge and the nature and presence of vegetation growing on the peat (Hope et al. 1997). Rates of runoff in response to rainfall events are also important (Worrall et al. 2007d).

Soil pipes provide an additional pathway through which POC can be lost from peatlands. Up to 10 % of streamflow can pass through soil pipes before entering the stream channel (Holden & Burt 2002), during which time, additional carbon in the form of POC may be lost as suspended sediment (Holden 2005a). In heavily eroded catchments, POC has been identified as one of the most significant loses of fluvial organic carbon, with losses exceeding DOC. Significant spatial variations in POC concentrations have been reported; therefore highly detailed sampling of eroded peatlands is required where carbon budgets are to be calculated accurately (Pawson et al. 2008). The high degree of spatial variability will make general predictions for carbon loss in the future prone to error. POC is thought to be oxidised and transformed into gaseous carbon, thus POC could provide a significant feedback to climate change (Pawson et al. 2008, Evans et al. 2006b).

2.3.1.5 Dissolved Inorganic Carbon (DIC)

DIC can be a product of weathering of the parent material (Worrall et al. 2007c) or the product of rainfall inputs of carbon (Worrall et al. 2005). In peatland stream environments DIC often takes the form of hydrogen carbonates, carbonate ions or dissolved free carbon dioxide (Dawson et al. 2002). Causes of variations in DIC releases from peatlands have not been identified, but are likely to be linked to variations in carbon dioxide concentrations. As will be seen in section 2.3.2, DIC is only a minor component of the peatland carbon cycle, owing to the acidity of the peatland environment (Dawson et al. 2002).

2.3.2 Carbon Budget Calculations

Much effort has been put into identifying how much carbon is being lost from peats, and most studies focus on one or more of the components of the carbon budget described above. A few studies have used a combination of in-situ monitoring and modelling techniques (based on published data, or values from which actual losses can be estimated) to calculate whole carbon budgets in an attempt to ascertain whether peats are sinks or sources of carbon. For ease of reference, whole carbon budgets are considered to be those where both gaseous and fluvial losses of carbon were included. A summary of published whole carbon budgets is presented in Table 2.1, those selected for the table contain fluvial and gaseous losses of carbon, but did not necessarily measure all five components of the carbon budget detailed above. It is acknowledged that this is not a finite list of all carbon measurements carried out to date, and that many studies exist that have measured just one of the carbon loss pathways. Studies where just one pathway was measured for a managed peatland are noted later in this section.

Table 2.1 Summary of annual peatland carbon budget calculations completed to date which incorporate both fluvial and gaseous pathways

Author	**Site**	**Net Ecosystem Exchange**	**CH4**	**DOC**	**POC**	**DIC**	**Exports in downstream**					**Overall Budget Estimate**	**Comments**
							CO_2	CH4	DOC	POC	DIC		
(Worrall et al. 2003b)	Moor House Northern England (Unmanaged)	*-40 - 70*	*1.5 – 11.3*	9.4 - 15	2.7 – 31.7	4.1 – 5.9						13.8 ± 15.6	1, 3, 6
(Worrall et al. 2007c)		-107.4	3.90	16.6	20.3							-11.2 to -20.9	1, 3, 6
(Worrall et al. 2009)		*53.6*	*6.2*	14.5	15 (49.2 in soil water)							-20 to -91	1, 3, 6
(Clay et al. 2010b)	Moor House (Grazed)	-25.2	*5.4*	*61.5*	*49.6*							144.15	2, 4, 6
	Moor House (Burnt)	-14.6	*6.0*	*63.2*	*27.0*							113.01	2, 4, 6
	Moor House (Unmanaged)	-83.5	*5.7*	*60.6*	*16.7*							169.19	2, 4, 6
(Billett et al. 2004)	Auchenforth, Scotland (Unmanaged)	-27.8	4.1				0.9	<0.01	28.3*		1.2	8.3	2, 3, 5
(Dinsmore et al.)		-136	0.29	-1.26			1.58	<0.01	32.2	5.46	0.39	-69.5	2, 3, 6
(Rowson et al. 2010)	Hexhamshire Common N England (drains blocked at start of study, no unmanaged site used for comparative purposes)	-17.7	1.6	29-4 – 85.8	1.9 – 7.8							63.8 to– 106.8	1, 3, 6
Nilsson et al. (2008)	Degerö Stormyr mire, Sweden	-55 ±1.9	9 ±1.7	20.4 ±1.7 (total stream organic C)			6.0 ±0.8	0.4±0.06				-24±4.9	1, 3, 5
(Roulet et al. 2007)	Mer Bleue, Canada (unmanaged)	-40.2	3.7	14.9								50 to -150**	2, 3, 5

All units are expressed as g C m^{-2} yr^{-1}. Negative values = a sink, positive = a source. Values in italics are based on interpolation and/or values from published literature rather than direct measurement. Blank – no data * Measured total organic carbon, therefore includes POC as well as DOC. ** based on 6 years data. 1. Range of values reflects the annual variations for the time period over which the budget was calculated. 2. Values represent the mean for the time period over which the study was conducted 3. Catchment scale study 4. Plot scale study 5 – Seasonal variations identified 6. Seasonal variations not presented.

Early work carried out by Garnett et al. (2001) identified the Moor House National Nature Reserve (NNR) in the North Pennines to be a carbon sink based on measurements of peat depth. Garnett et al. (2001) however suggested that work on the UK carbon inventory using two dimensional models is insufficient, and that further work is required using data on variations in soil thicknesses. More recent work has focussed on measurement of actual fluxes either at plot or catchment scale. Carbon budget calculations made by Worrall et al. (2003b) for Moor House NNR (an intact peatland) were among the first to be carried out for a UK peatland. The measurements included fluvial carbon fluxes and used data from previous studies to calculate releases of carbon dioxide and methane. Loses of carbon through subsurface flow were not calculated. The results of the study showed the site to be a small carbon sink (Worrall et al. 2003b). The budget was updated using information on primary productivity and inputs from dry and wet deposition to provide a more realistic carbon budget. The results indicated that the catchment was considered to be a source of carbon and within the next 10 years (using current climate change estimates), that source could double in size (11.2 to 20.9 g C m^{-2} yr^{-1}) (Worrall et al. 2007c). Further updates to the budget included more measurements in the field, and concluded that the site was a sink; Net Ecosystem Exchange (NEE) comprised the largest part of the budget, followed by losses of DOC (Worrall et al. 2009).

Elsewhere in the UK, studies have supported the findings of this most recent carbon budget for Moor House. At Auchenforth, Scotland, measurements of carbon losses from streams were incorporated into budget calculations and suggestions made that losses in streams could become greater than the amount of carbon absorbed from NEE (Billett et al. 2004). Further work at Auchenforth has included measurement of all major pathways, including in-stream losses of greenhouse gases. The findings demonstrated that downstream losses and losses in surface waters were an integral part of the carbon budget and should be incorporated in future calculations to provide a complete carbon budget. Excluding losses from surface waters would have resulted in the carbon sink appearing to be much greater than the one that was calculated (Dinsmore et al. 2010).

Much of the work carried out in the UK has been irrespective of the micro-topography of the peat and yet the exact points at which gaseous fluxes were occurring may impact on the budget values as there maybe hotspots of gaseous carbon loss (Morris et al. 2011) which result in the under or over-estimation of fluxes. Thus, detailed characterisation of gaseous losses from UK peatlands with a patterned micro-topography is needed. Studies carried out on Finnish boreal mires identified that hummocks were releasing the most carbon dioxide (206.4 g C $m^{-2}yr^{-1}$) compared to lawns (140.4 g C m^{-2} yr^{-1} for *Eriophorum* species and 164.4 g C $m^{-2}yr^{-1}$ for *Carex* species) and hollows (110.4 g C m^{-2} yr^{-1}). Diurnal variations reflected changes in climatic conditions, and note was made that budgets can vary greatly from day to day (Alm et al. 1997). Further work carried out in Finland examined the effects of an exceptionally warm and dry summer on losses of carbon, and found the peat to be a source of carbon during warmer drier conditions. Greatest losses of carbon were from hummocks, and least from the hollows (Alm et al. 1999b).

The importance of variation in local climate was also noted in calculating the carbon budgets of peatland streams in Scotland and Wales. DOC was the largest component of the budget, and losses were greatest during the summer and early autumn. Losses from the Scottish site (Brocky Burn) were greater than those at the Welsh site (Upper Hafren) due to the thicker accumulation of organic matter and lower annual precipitation (Dawson et al. 2002). Climatic variation controls seasonal variations in carbon losses, and typically the winter months are assumed to lose minimal amounts of carbon dioxide owing to reduced microbial activity caused by lower temperatures and a greater proportion of the peat being saturated. Work in Canada found carbon sequestration rates during the non-growing season to be smaller than the growing season, whilst carbon dioxide continued to be emitted, albeit at lower concentrations (Roehm & Roulet 2003). This study concluded that seasonal variations in carbon cycling need to be taken into account when determining carbon budgets.

Turetsky et al. (2002) suggested that peatland disturbances have had major impacts on carbon stocks and are likely to turn peatlands into sources of carbon rather than sinks. In light of such suggestions some studies have endeavoured to calculate complete peatland carbon budgets for managed peats.

As efforts are made to reduce the potential effects of climate change on uplands, more information is needed to understand the influence of management techniques on carbon budgets from peatlands. Efforts can then be made to determine which methods might enable carbon stocks to be preserved in the future. To date, work on carbon budgets carried out on managed peats has typically focussed on single management practices, and in some cases, reference has been made to an unmanaged site for comparative purposes. To date, only two attempts have been made to produce full peatland carbon budgets in the UK. Further details of management of peatlands are provided in Section 2.5.

The significance of DOC and losses of carbon dioxide identified in studies of pristine peats (Moor House and Auchenforth) was mirrored during studies of two recently blocked, drained catchments in Hexhamshire, northern England. Both recently blocked, drained catchments were identified as sources (Rowson et al. 2010). Carbon budget calculations for managed peatlands at Moor House in the North Pennines found grazed, burnt and unmanaged peats to be sources of carbon, with greatest losses coming from the unmanaged site (Clay et al. 2010b) contrary to the findings of previous studies at Moor House, where unmanaged parts of the reserve have been identified as carbon sinks (Worrall et al. 2009). In each case, management appears to be causing carbon sinks to become carbon sources. The effects of management are not irreversible however, as shown by Bortoluzzi et al.. (2006), who reported the findings of respiration monitoring carried out on a restored peatland in France. Restoration work had been ongoing for 20 years. Areas where *Sphagnum* had regenerated sequestered the most carbon (122 to 183 g C $m^{-2}a^{-1}$), compared to areas of bare peat were identified as sources of carbon (carbon exchange = 19 to 32 g C $m^{-2}a^{-1}$).

Although carbon budgets are inherently useful in understanding the extent to which carbon stocks are augmenting or diminishing, consideration needs to be given to the stage of vegetation growth that has been achieved. Clay et al (2010b) identified greater losses of carbon from unmanaged peats compared to burnt and grazed sites, where rates of primary productivity were higher, and thus rates of carbon sequestration were greater. Afforested peats are often described as carbon sinks, however, once the trees have reached maturity, their ability to sequester carbon reduces (Cannell et al. 1993).

2.4 Drivers of Carbon Cycling in Peatlands

As detailed in the section above, carbon dioxide and DOC are the main pathways through which carbon is lost from managed peatlands, and therefore shall be the focus of the remainder of this review. A summary of the main drivers of carbon dioxide and DOC loss from peat is presented in the sections below. A summary of the linkages between the drivers of the carbon cycle and losses of carbon is presented in Figure 2-3.

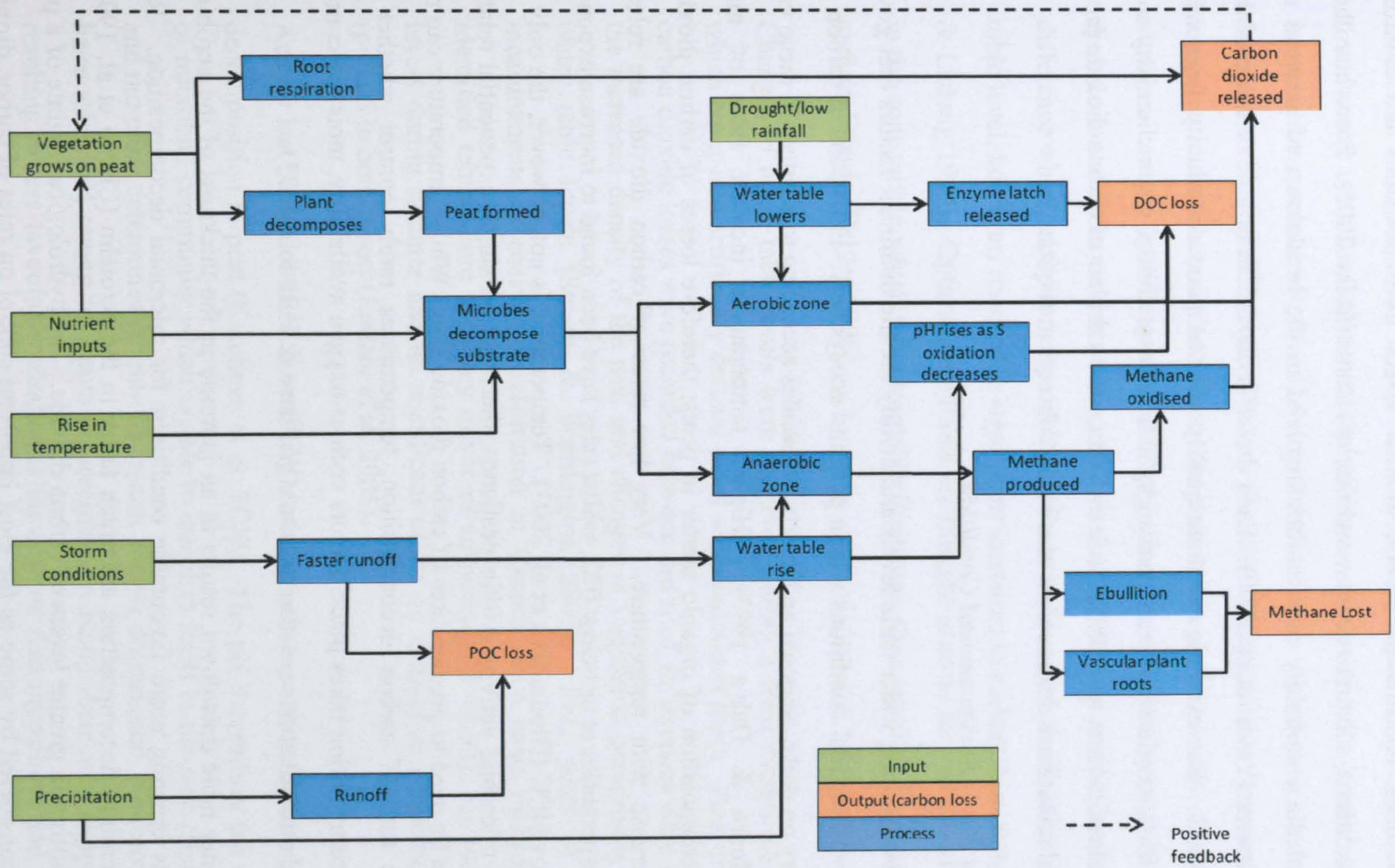


Figure 2-3 Conceptual Model Illustrating the Principal Linkages between the Drivers of the Peatland Carbon Cycle

2.4.1 Controls on Carbon Dioxide Losses from Peat

Carbon dioxide is released from peatlands due to microbial respiration. The dominant controls on carbon dioxide losses from peatlands are environmental conditions, substrate quality, and nutrient status (Laiho 2006). Porewater chemistry, and the community of microbes involved in the breakdown of material are also relevant (Yavitt et al. 2000). Each driver is dependent on one or more of the other drivers. For example substrate quality and the plant community from which the substrate is derived, are governed by nutrient availability, water chemistry and water table (Glaser et al. 1990). Each of the three key drivers of carbon dioxide production and release are discussed in turn in the subsequent sections.

2.4.1.1 Environmental Conditions

Temperature, water table levels and acidity are the dominant factors that govern the environmental conditions within peatland ecosystems. Emissions of carbon dioxide vary on daily, seasonal and yearly timescales according to changes in these variables (Moore & Dalva 1993). Higher temperatures increase rates of microbial decomposition of organic matter in peats; therefore losses of carbon dioxide also increase with temperature. Very low rates of carbon dioxide are released at temperatures at or below 0°C, whilst rates have been found to increase exponentially above 0°C (Dioumaeva et al. 2003). Temperature is not, however, the only control on microbial activity within peatlands, therefore no single exponential relationship can be used to predict rates of carbon dioxide loss. Whilst temperature can regulate the rate of carbon decomposition, temperatures rarely control whether or not decomposition takes place, factors such as oxygen availability, moisture content, pH and nutrient status are also relevant (Williams & Crawford 1983).

Water table drawdown results in an increase in the thickness of the aerobic layer, thus creating more favourable conditions for microbial decomposition. Rates of microbial decomposition are much faster in the acrotelm (Clymo et al. 1998) thus resulting in greater losses of carbon dioxide. The hydrological regime of a peatland is considered by some as the most important control on rates of carbon dioxide loss (Nilsson & Bohlin 1993). Scanlon and Moore (2000), however, suggested that a combination of lower temperatures and increasingly anoxic conditions were required to reduce rates of substrate degradation.

Studies of mesocosms on which water table levels and temperatures were varied found water table level to have little effect on carbon dioxide losses. Increases in temperatures accounted for 80 % ($p<0.001$)of the variation in carbon dioxide emissions (Updegraff et al. 2001). As the decomposition process advanced, temperature had less influence on the rate of degradation, and factors such as nitrogen content have a greater bearing (Berg & Meentemeyer 2002). Carbon dioxide losses from columns of peat studied in laboratory conditions showed little difference when a drop in water table of 10 cm was instigated. Fluctuations in water table level, however, resulted in significant increases in carbon dioxide losses (Aerts & Ludwig 1997). Optimal conditions are thought to exist for carbon mineralisation in the zone within which the water table fluctuates, thus resulting in greater losses of carbon dioxide (Belyea 1996).

Changes in water table levels were observed during a field experiment in Canada which compared artificially drained peats with undrained peats. Few differences in carbon dioxide losses were recorded between the sites, differences were attributed to the increased density of the peat and changes in vegetation community rather than water table levels (Strack & Waddington 2007). Five years of continuous measurements of ecosystem respiration in Canada on a large ombrotrophic bog identified temperature as a key driver of respiration ($r^2=0.62$), but reduced water levels during summer had no significant effect on ecosystem respiration ($r^2=0.11$) ($p<0.05$ in both cases) (Lafleur et al. 2005).

Acidity has been identified as exerting a strong influence on rates of organic matter decomposition in peat (Eskelinen et al. 2009). The pH determines the composition of microbial community which is able to establish itself in the peat (Bardgett 2005) and therefore determines the rate of organic matter degradation. Ombrotrophic bogs have a low pH owing to inputs of precipitation being their only supply of water, resulting in less favourable conditions for organic decomposition, and the possible presence of toxic soluble compounds (Aerts et al. 1999).

2.4.1.2 Substrate Quality

Substrate may be defined as the litter that has entered the peat profile and is undergoing decomposition. Substrate quality is a measure of the ease with which the material can be decomposed by microbes. Easily degraded compounds are considered to be of "high quality", whilst more recalcitrant materials are thought of as "low quality" (Berg 2000). Organic matter components such as "sugars, amino acids, starches, proteins and some hemicelluloses" are the most readily degraded compounds (Waksman & Stevens 1928 p 120). Celluloses, oils, some fats and other hemicelluloses are also easily degraded however the process takes longer than the aforementioned substrates. The most slowly degraded compounds include "lignins, waxes, cutins and some hemicelluloses" (Waksman & Stevens 1928 p 120). The quality of the substrate is directly linked to the plant from which it was derived, e.g. shrubs have been found to be recalcitrant owing to their high lignin content (Hobbie 1996). In addition, carbon may be present in the peat in the form of black carbon, commonly referred to as char (Clay & Worrall 2011). Inputs of char occur on burnt peatlands and aer a result of vegetation being converted to pieces of black carbon which remains in the peat post-burning. This material is considered to make a positive contribution to the carbon stores in burnt peatlands (Clay & Worrall 2011)

The carbon balance of peatlands is determined by the decomposability of the plant matter inputs to the peat (Limpens et al. 2008). Peats featuring easily degradable substrate are more likely to become carbon sources than sinks (Moore et al. 2007). However, where *Sphagnum* mosses comprise a significant proportion of the litter entering peat, the composition of the moss does not necessarily reflect the ease with which the substrate may be decomposed. *Sphagnum* is typically found in saturated and therefore anoxic areas of peatlands, demonstrating that whilst the plant material may be labile, if the environmental conditions within the bog do not favour decomposition, the material will not be degraded (Moore et al. 2007).

Easily decomposable substrates tend to be found in the upper layers of the peat profile and are typically fully degraded before the organic matter moves down the profile/more peat accumulates. Organic matter becomes increasingly recalcitrant with depth owing to the increase in its age and therefore the length of time it has been subjected to microbial decomposition (Hilli et al. 2008). Studies of different plant litters found that organic matter dominated by *Sphagnum* as opposed to sedges had a far greater carbon content (Updegraff et al. 1995). Other studies on *Sphagnum* have attributed low rates of decomposability to the poor nutrient content, acidity, and the very wet conditions that *Sphagnum* is commonly associated with (Heal and French, 1982). Studies of litter at Moor House, northern England found heather shoots and stems to be the most recalcitrant, whilst *Rubus* leaves were the most easily degraded (Latter et al. 1998). High concentrations of acid insoluble materials in litter are generally associated with low rates of carbon dioxide production (Shaver et al. 2006). Species with higher concentrations of water soluble carbon, e.g. needle litter tend to have higher rates of carbon dioxide loss (Domisch et al. 1998).

Exponential models describing rates of litter decomposition often suggest that a point will be reached where the supply of litter approaches exhaustion. Rates of decay are expected to slow sufficiently with time, so that rates become negligible owing to both the recalcitrance of the organic matter and the absence of favourable conditions under which decomposition can take place (Latter et al. 1998). An adequate supply of utilisable substrate often limits microbial activity in peatlands as demonstrated by experiments in which additions of labile substrate (e.g. glucose) were made, generally resulted in increased carbon dioxide production (e.g. Dettling et al. 2006). Studies of microbial activity in peatlands in North America, however, failed to identify a significant increase in microbial activity following additions of substrate (Fisk et al. 2003). The lack of consensus can be attributed to the influence of environmental conditions within the peat being more favourable to decomposition in Dettling et al. (2006) compared to the study by Fisk et al. (2003)

2.4.1.3 Nutrients

Nutrients play a vital role in the peatland carbon cycle not only by supporting plant growth but also to enable microbes to synthesise organic material. Ombrotrophic bogs typically only receive inputs of nutrients from rainfall, nutrients released from root exudates and those held in the decomposing plants that form the peat. Nitrogen and phosphorus are typically in limited supply and therefore plant growth is inhibited in many peatlands (Charman 2002, Rydin & Jeglum 2006). Atmospheric inputs of nitrogen, however, reduce the effects of low nitrogen availability on net primary production (Blodau 2002). Studies in boreal peats have identified the binding of nitrogen to humus as a cause of nitrogen limitation which in turn reduces rates of decomposition (Prescott 2005).

The presence and availability of nitrogen not only govern the plant species growing on a bog, but also determines the rate at which carbon is mineralised. Low concentrations of nitrogen contribute towards a high carbon to nitrogen (C:N) ratio which impedes rates of microbial activity and hence rates of carbon dioxide loss. Experiments involving fertilisation of peats found that additional nutrients had little effect on rates of respiration (Bubier et al. 2007), suggesting that nutrient availability is not the sole driver of carbon dioxide loss from peats. The addition of nitrogen to peats has been found to result in microbes converting the nitrogen into ammonia, consequently reducing the pH of the peat environment, and limiting carbon mineralisation further (Aerts & Toet 1997). Studies carried out over long periods of time have identified increased carbon dioxide losses from fertilised peats. At Mer Bleue, Canada, Basiliko et al. (2006) identified a decrease in losses in the first year, but as new plant materials entered the system in the second year, carbon dioxide losses increased. Peats with a high initial nitrogen content have been found to feature more recalcitrant material in the latter stages of decomposition, because high concentrations of nitrogen limit the formation of lignolytic enzymes, which are required to decompose the most recalcitrant compounds, which typically contain lignin (Berg & Meentemeyer 2002). The impact of nitrogen additions does, however, depend on the plant species concerned. Increased concentrations of nitrogen to bogs where *Sphagnum* mosses are present can result in not only reduced *Sphagnum* growth, but also increased decomposition and subsequently greater losses of carbon dioxide (Gerdol et al. 2007).

Primary production in ombrotrophic bogs is thought by many to be controlled and limited by nutrient availability. Evidence exists, however, to suggest that rates of mineralisation of nitrogen and phosphorus are higher in bogs compared to fens (Verhoeven et al. 1990). This evidence does not concur with rates of carbon dioxide loss which are generally reported as being higher in fens than bogs owing to the more favourable conditions for decomposition. Limitations on primary productivity and therefore microbial degradation once the plant species have died must be attributable to factors other than nutrient supply. Suggestions have been made that the presence of a lower pH or properties of water chemistry e.g. the presence of toxic compounds could be relevant (Aerts et al. 1999).

Strong links have been identified between nutrient concentrations and methane production in peatlands (Nilsson & Bohlin 1993, Basiliko & Yavitt 2001). Such linkages could have an indirect impact on carbon dioxide production as methane is oxidised to carbon dioxide in the acrotelm. Much work on nutrients has assessed differences between different types of peatland e.g. ombrotrophic versus minerotrophic gradients have been examined between the different forms of peat. Keller et al. (2006) looked at the effects of nutrient additions on anaerobic respiration and methane production along an ombrotrophic – minerotrophic gradient in the USA. Changes in vegetation community and increased methane production were observed in response to additions of nitrogen and phosphorus in bog peats, however fen peats did not respond. The authors concluded that whilst nutrients exert controls over anaerobic carbon loss from peats, the effects were also dependent on the time-scale that was studied, and the type of peat examined.

2.4.2 Controls on DOC in Peatlands

DOC losses from peatlands are driven by a number of factors. Over the past four decades, DOC concentrations in rivers and lakes have increased rapidly (Worrall et al. 2004b). Much effort has been spent on identifying the underlying causes of these increases (Evans et al. 2006a). There is still much dispute as to the exact causes, but the main factors identified include environmental conditions (temperature changes, water table fluctuations), land management, the enzyme latch mechanism, reductions in sulphur deposition and subsequent reductions acidification and increases in riverine flow rates (Tranvik & Jansson 2002, Clark et al. 2005, Monteith et al. 2007, Freeman et al. 2001b, Freeman et al. 2001a, Holden et al. 2007b, Evans et al. 2006a).

2.4.2.1 Environmental Conditions

As with losses of carbon dioxide, increased temperatures provide micro-organisms with a more favourable environment in which to synthesise carbon, thus rates of activity increase with temperature (Fenner et al. 2007). On account of such rises in activity and subsequent DOC production, concentrations in peat solution vary seasonally, with highest concentrations during the summer months (Bonnett et al.

2006, Koehler et al. 2009). Temperature changes have also been noted as a key driver of DOC losses (Evans et al. 2005).

Variations in water table levels have been associated with fluctuations in DOC concentrations in peats. Concentrations and rates of loss increase as the water table level falls owing to increased microbial synthesis of carbon stocks under aerated conditions (Freeman et al. 2001a). Experimental work on peat columns identified a significant relationship between greater DOC concentrations and lower water table levels (Pastor et al. 2003).

Debate exists as to the extent of linkages between DOC loss and water table levels. Laboratory simulations of reduced water levels (Blodau et al. 2004) and drought conditions (Freeman et al. 2004a) failed to identify significant DOC losses over periods of 7 and 36 months respectively. Studies of long term DOC records by Worrall et al. (2003a) identified a time lag between lowered water tables caused by drought conditions and increased DOC losses. The time lag was attributed to the hydrophobic nature of peats; as time is required for peats to re-wet after drought periods before organic compounds can become dissolved into the soil solution and exported (Worrall et al. 2003a).

Artificial drains have been found to cause even greater DOC concentrations, particularly in areas where drainage networks are dense (Mitchell & McDonald 1995). Blocking drains in a bid to reduce DOC concentrations is becoming increasingly common, with the majority of studies suggesting that blocking results in reduced DOC concentrations. Detailed studies carried out by Wallage et al. (2006) identified lower concentrations of DOC in the peat solution adjacent to blocked drains compared to unblocked, and areas that had never previously been drained. Monitoring of a series of blocked and unblocked catchments over a period of two years in Allendale and Upper Teesdale, northern England also identified reduced DOC concentrations in blocked drains (Gibson et al. 2009).

Drain blocking schemes have been found to successfully raise water table levels, although the extent of such rises varies between studies. Despite the rise in water table levels, not all studies have identified a corresponding decline in DOC concentrations. Work carried out in Canada has involved the re-introduction of moss species in an attempt to facilitate re-wetting of the peat (Waddington et al. 2008). Blocking drains with heather bales has been found to lead to small increases in water table levels with an expectation that further recovery will occur with time (Wilson et al. 2010). Studies carried out in Wharfedale by Wallage et al. (2006) found indications that blocking of drains not only resulted in the raising of water table levels, but also reductions in DOC concentrations (of between 60 and 70 %) and the colour observed in water samples. Significantly higher water table levels were observed in an extensive study of numerous peatland sites across Scotland and northern England, and note was made that the method of drain blocking did not make a significant difference to DOC concentrations or the colour of the peat solution (Armstrong et al. 2010). Wilson et al. (2010) showed water table level recovery to be gradual and to vary between catchments. Their work indicated that post-drain blocking a decrease in DOC concentrations occurred, although the fluxes were dependent on rainfall.

2.4.2.2 The Enzyme Latch Mechanism

The release of additional DOC during periods of water level drawdown has been attributed to the increase in acrotelm thickness which provides microbes with aerobic conditions which favour organic matter degradation. The so-called "enzyme-latch mechanism" introduced by Freeman et al. (2001b) has also been cited as a cause for increased DOC losses during and after water level draw down. As the water table is lowered, the phenol oxidase enzyme is provided with a supply of oxygen which it uses to breakdown phenolic compounds. The phenolic compounds ordinarily provide a barrier to the decomposition of the most recalcitrant organic compounds. Once the phenolic compounds have been decomposed, hydrolase enzymes can then break down the more recalcitrant molecules, which even if only partly broken down, can be released as DOC. This process is assumed to continue after water table levels have risen again, provided sufficient phenolic compounds have been degraded. Without a lowering of the water table, the phenol oxidase enzyme cannot degrade phenolic compounds and thus acts as a "latch" that safeguards carbon stocks in peatlands from decomposition (Freeman et al. 2001b).

2.4.2.3 *Sulphate Deposition and Acidification*

Since the early 1980s, deposition of sulphur from industrial sources has reduced in line with a fall in emissions of sulphur dioxide (Fowler et al. 2005) brought about by legislation and international pressure (Morecroft et al. 2009, Jenkins 1999). As a result, the formation of acid rain has declined and consequently deposits of sulphur and acid waters onto peatlands have decreased. Less acidic conditions within peatlands have enabled microbes to synthesis organic compounds more easily, thus resulting in an increase in DOC losses from peats (Evans et al. 2005).

The onset of drought conditions (i.e. lower water tables) has been linked to the oxidation of sulphur stored in peats (Chapman et al. 2005). Lower water tables result in an increase in the thickness of the acrotelm, and thus provide suitable conditions under which sulphur can be oxidised to sulphate. The oxidation process results in losses of hydrogen ions into the peat solution, thus lowering the pH. Increased acidity limits microbial activity, and so less carbon is metabolised and DOC production decreases (Clark et al. 2005, Clark et al. 2006). Following the period of drought, water tables recover, the acidity of the peat solution decreases and DOC concentrations increase, in some cases to values higher than those observed prior to the drought (Clark et al. 2009). Recovery from acidification does not occur immediately. Long term monitoring of sites belonging to the UK Acid Waters Monitoring Network (AWMN) showed increases in DOC concentrations but no associated changes in pH or sulphate concentrations within the first 10 years of monitoring (Evans et al. 2005). Further monitoring over the next five years did find a link between increasing DOC production and lower sulphate concentrations and associated rises in pH values (Evans et al. 2005). Concentrations of organic acids such as DOC have been found to decrease in soil solution in the presence of strong acids such as sulphuric acid as the acidity results in a decrease in their solubility (Krug & Frink 1983)

Rates of recovery from sulphur deposition have been found to vary depending on the location and management of the site in question. Studies of records of water quality across Scotland by Harriman et al. (2001) identified a reduction in sulphur deposition. The study found that in areas where unmanaged peatlands were present, the recovery was almost immediate. Rises in pH values were also seen at many of the study sites; afforested sites were not, however, reported to be recovering as well in terms of acidity (Harriman et al. 2001). Conversely studies of sulphur loss from peats in the southern Peak District found recovery rates from sulphur deposition to be poor. During storm events, losses of DOC were found to be low in catchments with many gullies, whilst losses were high in catchments with few gullies. The results have been attributed to concentrations of sulphate being greater in drained catchments and thus suppressing DOC production (Daniels et al. 2008).

2.4.2.4 Flow Rates

The hydraulic conductivity of both the water flowing through the peat, and flow rates in the streams and rivers into which runoff and throughflow discharge into, have a bearing on DOC concentrations and fluxes. During storm events, rates of DOC export rise rapidly but subside and even decrease to lower levels than those preceding the storm owing to exhaustion of DOC supplies (Worrall et al. 2002). Often DOC production coincides with recent climatic conditions. During dry periods concentrations in the peat solution may rise, whilst during wet periods, they may fall due to dilution with precipitation (Waddington & Roulet 1997). Increased fluxes of DOC in rivers and lakes in Sweden have been attributed to a combination of raised temperatures which increase DOC production coupled with greater precipitation and runoff rates which have resulted in a rise in the amount of carbon being exported from terrestrial carbon stores (Tranvik & Jansson 2002).

2.5 Management of Upland Peats

While much effort has been focussed on calculating carbon budgets for unmanaged/pristine peatlands, as witnessed above, only limited research has been carried out on the carbon budgets of managed peatlands, and even less on the drivers of managed peatlands. Before considering how management might affect the peatland carbon cycle and its drivers, it is worth considering how peatlands in the UK are managed.

Approximately 9,000 years before present (BP) the whole of the UK was covered with trees following the retreat of the ice sheets and the establishment of climax vegetation (Evans 2009). Pollen records have indicated that the onset of peat formation began between 9,000 and 5,000 years ago, during a period when the climate became wetter (Tallis 1991). At the same time, Neolithic people began to clear small areas and commenced farming (Yallop et al. 2008) including upland areas (Simmons 2003). These landscapes have continued to be modified because of human induced activities, without which, it is believed that the heather would have disappeared and the trees returned (Dodgshon & Olsson 2006). The absence of trees however enabled greater quantities of precipitation to reach the soils, causing waterlogging, an increase in anoxic conditions and consequently a decline in microbial activity. Succession of species such as *Sphagnum* mosses would have resulted in increasingly acidic conditions, with higher rates of cation binding and therefore lower nitrification rates, causing reduced nutrient cycling and thereby further decreasing rates of microbial activity, and thus the formation of peats (Simmons 2003).

Present day upland management is confined to rough grazing, afforestation, burning to create suitable habitats for grouse shooting and the provision of recreational areas for tourism (Dawson & Smart 2006). In recent years, there has been a decline in shepherding practices due to falls in the number employed on upland farms (Backshall et al. 2001). Changes in management are set to continue as efforts to restore damaged peat ecosystems take place, and alterations to current management practices occur (Holden et al. 2007b).

A summary of the most common land management practices used on upland peats in the UK is provided below. A brief outline of the research carried out to date on peats managed by each method is also given.

2.5.1 Peatland Burning

Burning of peatlands in the UK began in earnest in the early 1800s when grouse shooting increased in frequency (Holden et al. 2007b). Records of burning in Scotland, however, date as far back as the 1400s when the first references to “muirburn“ were made (Dodgshon & Olsson 2006). Burning controls heather in areas where grouse shooting and sheep rearing take place. Of the estimated 6,780 km^2 of managed peatlands in the England, Natural England (2010) suggest that 30 % of blanket bogs in England have been subjected to deliberate burning.

Studies of aerial photographs for selected areas of the English uplands have suggested that burning has become more frequent and intense since the 1940s (Yallop et al. 2006). Work by Hester and Sydes (1992) also found evidence for such an increase through an examination of trends in burning on Scottish grouse estates between the 1940s and 1980s. Their work aimed to assess the hypothesis that burning was becoming less common, but they found no substantive evidence to support this hypothesis.

Burning aims to encourage heather shrubs to regenerate, which prevents them from becoming too large and woody. Older shrubs tend to be unpalatable to sheep and are often difficult for livestock to access. Short, tender heather stems are an important food source for grouse chicks on shooting estates. Larger plants provide the grouse with areas to shelter and nest. Burning is typically carried out on strips and patches of heather, resulting in a range of heather stands of different height and age. Burning is carried out on a rotational basis across peatlands to provide a varied habitat; intervals between burning of between eight and 25 years are recommended (Tucker 2003). The Heather and Grass Burning Code in England and Wales (DEFRA 2007) and the Muirburn Code in Scotland (Scottish Executive 2008) are used as a guide for regulation of burning practices. The codes determine the timing and conditions under which burning may be carried out.

The impact of burning on peatlands depends on the vegetation type and cover, the intensity and frequency with which the burn is carried out, the time of year and recent climatic conditions. Spring and summer burns are more intensive than autumn and winter burns due to the lower moisture content of litter and vegetation (Shaw et al. 1996). Burning has reportedly caused a number of changes to peatlands, a summary of which is provided below.

Early studies on the effects of peatland burning focussed on changes in vegetation type and re-growth. Vegetation surveys conducted by Hobbs (1984) and Rawes and Hobbs (1979) identified changes in vegetation as a result of burning peats. Hobbs (1984) found that the frequency of burning had an effect on the species present, frequent burning led to the dominance of *Eriophorum vaginatum*, which favours grazing sheep, whilst infrequent burning led to a dominance of *Calluna vulgaris* which favours grouse.

Davies et al. (2010) investigated the effects of fire intensity on the regeneration of *Calluna vulgaris* on heathlands on the borders of the Cairngorms National Park. The authors concluded that burning of older plants is not recommended due to the fire hazards associated with burning very woody plants. Additionally, seedling survival and consequently establishment were found to be poor at sites where older heather plants were located. Legg and Davies (2009) noted that many burnt peatlands did not develop as *Calluna* dominated peat, instead species such *Eriophorum* and *Sphagnum* were prevalent.

Nutrients have been found to volatilise during burning (Allen 1964, Forgeard & Frenot 1996). Dikici and Yilmaz (2006) compared two Turkish peatland sites which had been burnt 36 years apart. The results of the study identified higher concentrations of nutrients in the more recently burnt site; which the authors suggested demonstrated that sites do not recover from the effects of burning over short to medium timescales. Studies of montane forests have also identified significant losses of nutrients during fires, yet nutrient availability in the first year post burning was found to increase due to inputs from ash (DeBano 1990).

To date, some studies of carbon stocks and losses from burnt peatlands have been carried out. Studies in Finland (Pitkanen et al. 1999) and Canada (Kuhry 1994) identified reduced carbon stocks accumulating in peats that were burnt. As of yet, a complete carbon budget (measuring both fluvial and gaseous carbon losses at the catchment scale) for burnt peatlands has not been carried out. Meta-analysis carried out by Worrall et al. (2010a) assessed the probability of a reduction in burning resulting in an improvement in carbon budget terms i.e. a reduction in carbon losses. The results indicated that the cessation of burning would result in a 93 % improvement in the carbon budget, with a 60 % improvement in greenhouse gas emissions i.e. less carbon could be lost to the atmosphere and more would be sequestered. Farage et al. (2009) carried out a study of burning in the Yorkshire Dales, in which they suggested many carbon inventories underestimate the carbon stored in uplands peats, given the quantity of carbon stored in the biomass. The study site was found to range from source of 34 g C m^{-2} yr^{-1} to a sink of 146g C m^{-2} yr^{-1}, the variations accounted for the range of estimates of fluvial fluxes (only respiration was measured, all other flux estimates were based on data from other sites) and climatic fluctuations. The findings of this study however have been subject to criticism by Legg et al. (2010) who stated that estimates of biomass post-burn are grossly over-estimated and carbon losses underestimated compared to their own findings of work carried out on 26 burnt moorland sites (including peatlands and other sites with highly organic soils). Farage et al. (2009) noted that the work was preliminary, small-scale, that the impact of burning is heavily dependent on local climatic conditions at the time of the burn and that whilst using data from other sites was not ideal, the data did allow an initial estimate of the carbon budget to be made.

As noted previously (in Section 2.3.2), only one study has attempted to calculate a carbon budget for a burnt peatland in the UK at a plot scale (as of yet, a catchment scale, whole carbon budget has not been published). The work was based on a combination of measurements taken in the field and predictions using existing data (Clay et al. 2010b). The calculations imply that burnt peatlands are carbon sources; however, the unmanaged site was identified as an even greater carbon source contrary to previous carbon budget calculations for unmanaged sites. This result does not support the findings for Moor House of Ward et al. (2007), who found burning resulted in greater losses of carbon than unmanaged sites. Differences between the sites could be due to differences in the stage of the burn cycle during which field monitoring was carried out. Suggestions have been made that the presence of charred materials in the soils could impede rates of microbial activity (Haslam et al. 1998), which would result in lower rates of carbon dioxide loss. Consequently, recently burnt sites are likely to release carbon dioxide, but, the overall carbon balance would depend on rates of primary productivity and therefore inputs of carbon into the system.

Holden et al. (2007b) noted that there is a lack of studies on the effects of burning on peatland hydrology, sediment release and water quality. Studies on the effect of burning on peat solution chemistry have failed to identify any significant differences in dissolved organic carbon (DOC) concentrations. Early work carried out by Worrall et al. (2007a) suggested that burning resulted in a decrease in DOC. Subsequent studies by Clay et al. (2009b) looked at a longer period of time and discovered that significant differences only occur immediately after the burn. Work by Clutterbuck and Yallop (2010) and Yallop and Clutterbuck (2009) refutes these findings; their work examined catchments in the Yorkshire Dales and the south Pennines where increased concentrations of DOC in streams strongly correlated with burning of peatlands. The disagreement between the two sites could be attributable to the latter studying stream water concentrations and the former peat soil concentrations collected within a metre of the peat surface. In addition, the strong correlations identified by the latter were for peatlands burnt within two to three years of the study being carried out. Differences in vegetation composition between the sites might also be a significant factor, as suggestions have been made that vegetation is a key driver of DOC loss in burnt peats (Worrall et al. 2010b).

Changes to the physical properties of peats post-burning have been noted such as reduced rates of infiltration due to clogging of pores by ash particles (Mallik et al. 1984b); cracking, desiccation and surface instability (Maltby et al. 1990). The formation of crusts on the surface has also been cited as a cause for reduced rates of infiltration (Tucker 2003). Holden (2005b) identified a link between the presence of heather and an increase in soil pipe frequency. Pipes are associated with changes in peatland hydrology and provide a conduit through which carbon can be lost. Given the link between heather and peat pipes it is feasible to suggest that the regeneration of heather through burning could increase the number of pipes present, thereby increasing the amount of carbon lost from peatlands that have been burnt.

2.5.2 Grazing of Peatlands

Grazing is one of the most common management approaches on peatlands, and ensures vegetation levels are kept in check whilst providing a source of income for rural communities. Grazing of uplands is believed to have commenced between 4,000 and 5,000 years before present, following the clearance of woodlands for agricultural purposes (Backshall et al. 2001). Traditionally grazing was carried out at low intensities, and the consequences for upland soils and vegetation were limited. Increases in grazing intensity occurred as a consequence of the introduction of subsidies and support schemes for hill farms (Adamson & Gardner 2004). A post-war peak in grazing livestock in upland areas was witnessed during the 1980s when payments were received per head of livestock under the Common Agricultural Policy. The introduction of schemes which focus on environmental outcomes and the area of land owned by farmers such as the English Environmental Stewardship and the Single Payment Scheme have resulted in reduced numbers of livestock and consequently a decline in over-grazing (Gardner et al. 2008). Decreases in the numbers of people employed in agriculture in upland communities has resulted in fewer shepherds controlling grazed areas, leading to overgrazing of some areas, damage to the peat and vegetation, and changes in vegetation composition (Holden et al. 2007b).

Few studies have looked at the effects of grazing on peats. The key changes that have been reported are defoliation, trampling and changes in the nutrient status of the peats (Crofts & Jefferson 1994). Changes in plant species have been reported as a result of grazing, the changes observed varied depending on the stocking density of the sheep (Lance 1983, Pakeman et al. 2003, Hope et al. 1996). Evidence exists to suggest that grazing depletes phosphorus concentrations in heathland soils owing to the removal of vegetation by livestock, where much of the phosphorus is stored (Hardtle et al. 2009). Reductions in infiltration rates have been reported by Shaw et al. (1996) as a result of trampling and possibly stocking densities that are too high, which have also been cited as the cause of erosion of upland peat soils (Evans 2005). High stocking densities have also been cited as a cause of flooding, for example heavily grazed areas of Dartmoor were found to have a lower saturation threshold resulting in higher rates of runoff and the onset of flooding (Meyles et al. 2006).

Whilst changes in the floral composition and physical properties of peat soils have been observed, very little work has related these to changes in carbon losses from peatlands. Shifts in the properties of the peat will however influence the hydrological regime and the quality of substrates entering the peat soils. These factors in turn are likely to cause further alterations to the peat i.e. to the availability of nutrients, and the ability of gases to move through the soil profile. A combination of some or all of these factors will influence carbon cycling in peat soils.

Studies at Moor House carried out by Ward et al. (2007) identified slight increases in carbon dioxide losses from grazed peats, however, the grazed plots were found to act as carbon dioxide sinks based in the NEE data and sequestered more carbon dioxide than the unmanaged site. In contrast, Clay et al. (2010b) found all plots at Moor House to be sources of carbon. Their work considered all aspects of the peatland carbon budget, rather than just gaseous fluxes. They did not however, measure all elements of the carbon budget; instead, estimates for POC, methane and DIC were provided based on the work of others. In addition, an assumption was made that catchment losses of DOC would be the same as the concentrations found in the peat solution, despite the authors noting that in-stream processing and dilution are likely to reduce concentrations at the catchment outlet.

2.5.3 Peatland Drainage

Extensive drainage of the UK uplands using open ditches took place in the 1960s and 1970s to facilitate enhanced agricultural productivity primarily for sheep and grouse (Adamson & Gardner 2004). In many cases the aims of drainage have not been achieved, grouse populations have not thrived, and peatlands have not been found to be able to support increased livestock populations (Holden et al. 2007b). More recently, attempts have been made to reverse the negative effects of drainage (e.g. erosion) by blocking drains. The success of such schemes depends on the extent of the degradation of the peat prior to inserting the drain (Holden et al. 2007b), and often, additional measures are required for example, the use of mulch and open pools to restore the moisture content of the peat (Price 1997).

The thicker acrotelm in drained peatlands results in the prevalence of more aerobic conditions within the peat (Laiho 2006) and consequently greater losses of carbon dioxide and lower emissions of methane (Blodau et al. 2004, Blodau & Moore 2003) as the increased aeration of the peat results in faster microbial activity and therefore decomposition (Holden et al. 2004). Peats that have been oxidised for long periods of time, however, are unlikely to release significant quantities of carbon dioxide, as they will have become resistant to decomposition (Hogg 1993). Oxidation of peat soils has been identified as a cause for the "enzyme latch mechanism" (Freeman et al. 2001b) as described in Section 2.4.2.2, whereby an increase in the thickness of the acrotelm results in increased losses of DOC.

Chapman et al. (2005) identified reduced concentrations of DOC from peatlands during periods when water tables were also observed to be low. Low pH values and higher concentrations of sulphate were found to coincide with such conditions and thus were proposed as being significant factors in explaining trends in DOC concentrations. Laboratory simulations were carried out to identify the effects of lower water tables i.e. draining the peat on DOC and sulphate concentrations. The findings of the simulation supported their interpretation of the results observed in the field.

Changes in the physical properties of peatland soils subjected to drainage have included increased bulk density (Rosenberry et al. 2006), a flashier hydrological regime and increases in rates of throughflow (Holden 2006, Holden et al. 2007a, Holden et al. 2006). Intensively drained peatlands have been reported to cause rates of water movement through the peat to increase, which have been cited as a cause for increased colour intensity in peatland waters leaving the catchment (Mitchell & McDonald 1995).

Rowson et al. (2010) provided the first complete carbon budget for a drained peatland in the UK. The study compared two catchments immediately after blocking (0.75 and 0.24 hectares) and found both to be sources of carbon, with losses of between 63.8 and 106.8 Mg C km^{-2} yr^{-1}. The sites were found to be very small sinks in terms of carbon dioxide exchange, values were lower than those recorded by others for British peats (Cannell et al. 1993, Clymo 1995). DOC fluxes were found to vary widely between 29 and 85 Mg C km^2 yr^{-1}. The study focussed on highly disturbed catchments but indicated that draining a peat under these conditions can result in a switch from a carbon sink to a source. Due to the nature of the experimental set-up, comparisons were not made with an undrained catchment.

Increased rates of respiration have been recorded at sites where drainage has successfully resulted in lower water tables. A study of a homogenous fen in northern Finland by Jaatinen et al. (2008) focussed on an artificial drainage gradient that has been present since 1959. The driest areas of the site showed a three-fold increase in respiration, which was attributed to the presence of increased fungal and bacterial biomass in these areas. Carbon stocks were found to accumulate in drained, ombrotrophic Finnish peats, however, losses of carbon dioxide also increased owing to decreases in acidity and increases in microbial decomposition (Minkkinen et al. 1999). Alm et al (1999a) estimated carbon dioxide losses increased by 24% in drained peats compared to undrained, resulting in the drained sites representing a carbon source. These findings are also supported by a study of gullies in northern

England in which carbon dioxide emissions from gullies were found to account for 21.6 % of the peatland carbon dioxide fluxes (McNamara et al. 2008).

DOC concentrations are typically found to be higher in artificially drained catchments compared to those which are undrained. The effects of blocking drains on DOC concentrations has been widely studied and found to result in reduced DOC concentrations. Lower DOC concentrations were attributed to reductions in catchment flow rates in a study by Gibson et al. (2009) study. Fieldwork carried out by Worrall et al. (2007b) showed that no one method of drain blocking is preferable over another, all methods studied resulted in rising water tables in the drain. The colour and DOC concentrations, however, also increased within the individual drains, though no increases were detected at the catchment scale.

Wallage et al. (2006) identified lower DOC concentrations in peat solution samples collected from blocked catchments compared to sites that had never been drained, and those that had been drained but were not blocked. Mixed findings were reported by the survey of Armstrong et al. (2010) which compared 32 drained and blocked catchments across northern England and Scotland. Significant differences in DOC were identified between blocked and unblocked sites in the majority of cases, however at an intensively monitored site no significant differences between DOC concentrations in drained and undrained sites were found, showing that site specific factors can mask broader, regional patterns.

Changes in plant species have also been observed at drained peatlands typically downslope of the drain (Stewart & Lance 1991, Holden et al. 2007a) thereby influencing substrate quality. Work carried out by Updegraff et al. (2001) identified lower methane emissions as a result of drainage, owing to the thicker acrotelm, which not only resulted in less methane production but also methane was oxidised to carbon dioxide in the aerobic zone. The results of the study did not however identify a relationship between vegetation and carbon dioxide emissions.

Drain spacing is a key factor in determining the impact of drainage in peatlands (David & Ledger 1988), in some cases a maximum drain spacing of 2 m is recommended to have any impact on water levels (Hudson & Roberts 1982). This view was supported by Stewart and Lance (1991) as a result of a study of water table fluctuations on a drained peatland. Monitoring wells located furthest from the drains were found to be most responsive to rainfall and had the highest water table levels. Increased drainage densities have been associated with greater discolouration of water within the catchment due to the more rapid release of water containing organic compounds (Mitchell & McDonald 1995). Coulson et al. (1990) found drains had little effect on water levels at sites with rainfall over 1,200 mm a^{-1} and significant differences in peat moisture content were not identified.

2.5.4 *Afforestation of Peatlands*

Afforestation has been one of the main causes of peatland habitat loss in the UK over the past century. It is estimated that 315,000 hectares of shallow peat and 190,000 hectares of deep peat have been afforested in Britain (Cannell et al. 1993). Losses of carbon from deep afforested peats are expected to be greater than the quantity of carbon sequestered by the trees in these areas (Cannell et al. 1999). Since the 1980s, planting in deep peats has declined in an attempt to conserve these wetland habitats (Hargreaves et al. 2003).

Increases in the thickness of the acrotelm occur as a result of drainage prior to planting and the additional water requirements of tree species (Anderson et al. 2000). Shrinkage and desiccation have been reported due to drainage and uptake of water by tree roots (Pyatt 1993). As cracks form, a network of hydrological conduits can arise which can cause the water table to drop even further. Peak flows and rates of evaporation have also been noted to increase in afforested peats (Anderson et al. 2000).

Summer time soil temperatures tend to decrease in response to afforestation of peatlands due to shade, and could potentially create limiting conditions for soil microbes (Silvola et al. 1996, Trettin et al. 2006). Studies of streams in afforested catchments compared to moorland streams in the Yorkshire Dales identified lower temperatures in the afforested streams during summer months, but found little difference in temperatures during winter months (Brown et al. 2010). In terms of carbon cycling, lower temperatures within afforested catchments could result in lower emissions of carbon dioxide during the summer compared to unmanaged sites. Strong seasonal trends in the peatland carbon cycle were identified by Byrne and

Farrell (2005), the greatest losses of carbon dioxide occurred during the summer months. Losses from spruce plantations were greater than pine owing to greater root respiration from spruce sites, which was due to the increased fine root biomass present at the spruce sites, however, losses from afforested plots were not always found to be significantly different to undrained plots that had not been planted.

To date, a complete carbon budget has not been published for afforested peatlands. Decomposition of carbon is expected to be greater in afforested peats due to the increased aerobic zone brought about by lowering the water table. Inputs of carbon into afforested peats are often considered greater than peatlands that have not been planted with trees, and this can compensate for increased carbon losses. The quantity of carbon sequestered however depends on the age of the tree stand, with trees in excess of 100 years sequestering very little carbon compared to peatlands which can accumulate carbon for thousands of years (Byrne et al. 2004).

The presence of drains in afforested peatlands is often cited as a cause for increased carbon losses. Studies of afforested peats in Finland have found that conditions mirror those found in hummocky peats. Increased acidity, decreased temperatures and litter quality created conditions that were not favourable for decomposition, thus carbon losses were not as great as expected (Laiho et al. 2004b). Cannell et al. (1993) suggested that the more oxic conditions present in drained and afforested peats could result in increased microbial activity and subsequently peatlands will ultimately (over a period of hundreds of years) comprise only the most recalcitrant organic matter fractions as sequestration rates slow with time and the ability of afforested peats to continue to act as sinks is brought into question.

Increased DOC losses of between five and ten percent have been recorded in afforested catchments (Grieve 1994) and the composition of DOC has been found to be affected by afforestation, with increased quantities of non-humic substances recorded (Miller et al. 1996). Even higher concentrations of DOC have been recorded for felled peatland sites along with slightly less acidic conditions (Cummins & Farrell 2002, Neal et al. 1998, Reynolds 2007). Felling results in higher water tables and increased peat temperatures due to increased insolation (Trettin et al. 2006)

Studies of peatlands planted with Scots pine found that the inputs of litter were more important in terms of carbon cycling than nutrient status and water table level (Domisch et al. 2000). Suggestions have been made that increased litter inputs in afforested peats compensates for increased rates of decomposition and subsequent carbon dioxide losses (Martikainen et al. 1995).

Trettin et al. (2006) noted that there are many gaps in knowledge in relation to afforested peatland carbon cycling, in particular with reference to the biochemical controls on organic matter turn over, the relationship between the quality of organic matter, peatland hydrology, temperature and nutrient stores. A study of peatland afforestation in Finland by Laiho et al. (1999) identified increased concentrations of nitrogen and phosphorus over time as a result of drainage compared to undrained sites. Little effect was noted for potassium concentrations (Laiho et al. 1999). Studies of the effects of drainage on treed bogs in Finland, demonstrated that nutrient depletion did not occur owing to a rise in bulk density of the peat (Westman & Laiho 2003). The use of fertilisers when establishing forests on peatlands has, however, been recorded as having caused significant increases in peatland nutrient cycling (Anderson 2001).

2.6 Linking Carbon Losses with Land Management Practices in Upland Peats

To date, only one complete carbon budget where all carbon loss pathways were measured has been published for a drained peatland in the UK. Budgets for grazed and burnt sites have given indications of how land management affects the ability of a peatland to gain or lose carbon. As of yet, no complete carbon budgets based on either measurement or models have been developed for an afforested peat. Predictions of the effects of management using meta analysis and carbon modelling were presented by Worrall et al. (2010a). Afforested and burnt sites were predicted to be sources of carbon whilst grazed and drained sites were expected to be carbon sinks. The predicted effects of management on the drivers of the carbon cycle are summarised in Figure 2-4. A summary of the predicted effects of management on carbon losses from peatlands based on evidence collected to date is presented in Table 2.2.

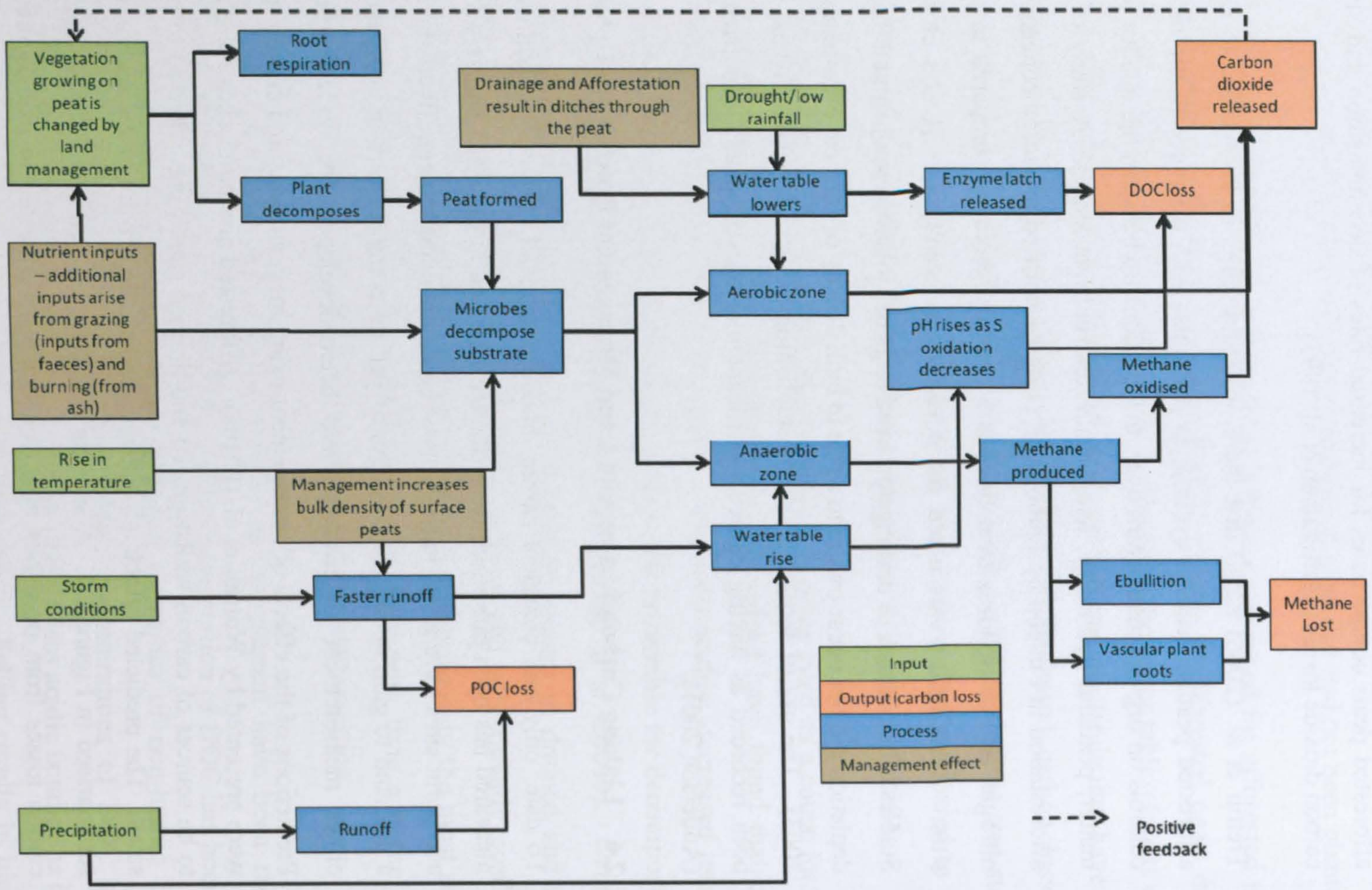


Figure 2-4 Predicted Changes to the Principal Linkages within the Carbon Cycle Due to the Effects of Land Management

Table 2.2 Predicted Impacts of Land Management on Carbon Gains and Losses from Peat based on evidence presented in Sections 2.4 and 2.5

	CO_2 (emissions)	CO_2 (inputs)	CH_4	DOC	POC
Drainage	**Increase** due to increase in acrotelm thickness resulting in greater aerobic activity and oxidation of methane to carbon dioxide (limited by lower moisture content).	**Unknown** Values comparing drained and unmanaged sites for the UK were not found, the impact is likely to depend on the effects of drainage on the vegetation community	**Decrease** due to increase in acrotelm, where methane is likely to be oxidised to carbon dioxide	**Increase** owing to greater microbial activity in thicker acrotelm.	**Increase** (theoretically, no evidence from actual measurement)
Afforestation	**Increases and decreases** – depending on time since drainage and planting, thickness of the acrotelm and the potential for methane to be oxidised to carbon dioxide	**Increase and decrease** – conflicting evidence between studies	**Unknown** – possibly decreases owing to greater thickness of acrotelm resulting in oxidation of methane to carbon dioxide.	**Seasonably variable**, lower water tables could give rise to increased losses, however increase in acidity of peat waters could limit microbial activity.	**Unknown**
Burning	**Decreases and increases** (dependent on local conditions and burn severity). Published literature found both increases and decreases (Ward et al. 2007, Clay et al. 2010b)	**Decrease** during the period when vegetation is absent from the site, **Increase** owing to regeneration of heather.	**Decrease**	**No change** (Temperature and water table level suggested to be more influential). Evidence to data suggests that concentrations increases immediately after burning.	**Increase** (theoretically, based on evidence of increase erosion, though no actual POC measurements made)
Grazing	**Unknown**. Conflicting evidence between studies as to whether increases or decreases	**Unknown.** Conflicting evidence between studies, with some suggesting primary productivity increases due to heather regeneration (Clay et al. 2010b)	**Unknown**. Conflicting evidence between studies as to whether increases or decreases	**Unknown**. Little change observed in studies to date.	**Potential increase** due to erosion caused by trampling – no actual measurements made

Whilst calculating complete carbon budgets is important, and further work must be carried out in this area, little attention has been given to the drivers of the carbon cycle in managed peats. Changing environmental conditions seem to be the most commonly studied driver of carbon cycling. Most work however has focussed on pristine/unmanaged peats, despite the majority of British uplands being subject to management.

Studies of substrate have tended to focus either on the composition of the peat litter or have assessed how different types of litter respond to environmental conditions. Hardly any studies have looked at the composition of the substrate itself, despite recognition that substrate is a key driver of carbon cycling. There is no evidence of work having been carried out to compare differences in substrate quality between managed peatlands. It seems apparent that all four key peatland management practices (burning, grazing, drainage and afforestation) in the UK have a significant effect on vegetation community and consequently will affect substrate quality. Understanding the effect of land management on substrate quality will enable carbon budgets that consider future carbon losses to be modified to account for differences in rates of decomposition.

Changes in the plant community will affect not only substrate quality but also nutrient supply and availability. Substrates of varying quality will release nutrients into the peat in differing concentrations. The nutrient demand of different plants growing on managed peats will also vary. The outcome of such differences is expected to affect the carbon cycle by governing rates of decomposition.

Changing environmental conditions are the most obvious impact of land management on peatlands. Lower water tables in drained and afforested peats have been found to augment losses of carbon dioxide and DOC. Whilst studies exist which have measured losses of DOC from peats managed in one or two ways, a cross comparison of the four key methods of peatland management in the UK has yet to be carried out. Doing so would allow the benefits of one management method over another to be determined.

Much debate exists over the drivers of DOC losses from peatlands. In particular, the increase in concentrations in freshwater streams, rivers and lakes has caused much concern. How most commonly cited drivers of DOC loss are influenced by land management is yet unknown.

It is feasible to suggest that the main drivers of carbon cycling will vary because of land management. It is also likely that the degree to which each driver varies will differ according to the management strategy employed. Such variations are anticipated to have a significant effect on peatland carbon losses. Future research needs to identify how each driver of the carbon cycle varies with land management as well as comparing differences in carbon losses between management methods.

As noted in Chapter 1, the aim of this thesis is to investigate the fluxes of carbon dioxide and DOC production in managed peatland and to identify how land management influences the drivers of the peatland carbon cycle. This work will seek to address some of the gaps in current knowledge that were identified in this chapter.

3 STUDY SITE SELECTION, SAMPLING AND FIELDWORK

3.1 Introduction

The aim of this chapter is to provide the framework to be used to investigate the overall aim of this thesis and the associated objectives set out in Chapter 1; which were derived from identifying the gaps in research presented in Chapter 2. This chapter also outlines how a field site was selected, along with background information on the selected site and details of the fieldwork carried out. Details of the laboratory methods used are provided in each of the following chapters for which those methods are relevant.

The overall methodology aimed to investigate the effect of land management on the drivers of the peatland carbon cycle and on key carbon losses from peatlands. Laiho (2006) suggested that carbon cycling in peats is governed by four drivers: substrate quality; nutrient availability; environmental conditions (e.g. water table, temperature); and microbial population.

This study has been designed around the first three drivers as these will be directly affected by peatland management, whereas the composition of the microbial community will be determined by these three drivers. In addition to substrate quality, environmental conditions and nutrients, the physical properties of the peat were examined, as changes in the structure of the peat are considered to be of relevance to the transport of gaseous carbon through the peat profile into the atmosphere. The study focused on gaseous carbon dioxide fluxes and dissolved organic carbon production as these account for the greatest proportion of carbon losses from upland peats in the UK (Worrall et al. 2007c).

3.2 Site Selection

The criteria for site selection included choosing an upland peatland that had areas which have been subjected to different land management practices. The site also had to have similar climatic, geological and topographical characteristics between each land management type. In addition, a site was required where permission to collect peat cores, install monitoring equipment and carry out monitoring would be granted.

The site had to be reasonably accessible from Leeds to allow frequent monitoring visits.

A shortlist of the sites considered is presented in Table 3.1. The table demonstrates that Moor House-Upper Teesdale National Nature Reserve (NNR) in Cumbria was selected as the most suitable site because it is the only site which fulfilled all of the selection criteria. In addition to all four management practices of interest being present (burning, grazing, drainage and afforestation), sites where combinations of burning and grazing were found, as well as sites that are burnt on two different burning cycles (every 10 and every 20 years). Confounding factors such as climate, altitude and geology were minimised by selecting Moor House as the managed sites were within 3.5 km of one another. The site could be easily accessed from Leeds and permission was granted by Natural England for access to the site to carry out research. In addition, all the differently managed sampling locations at Moor House had the same aspect (east facing). The site had further benefits in that it had an automatic weather station which was regularly maintained by the Environmental Change Network (ECN), and the site had a long history of research into upland peat. Further details of the site are provided in Section 3.3.

Table 3.1 Land Management Practices at Short-Listed Sites

Location	Land Management Practice					Accessible from Leeds	Access Possible
	Burning	Grazing	Afforested	Drained	Unmanaged site		
Moor House NNR	✓	✓	✓	✓	✓	✓	✓
Wharfedale – Oughtershaw		✓	✓	✓	✓	✓	✓
Nidderdale - Scarhouse	✓	✓				✓	✓ (from multiple parties)
Lake Vyrnwy				✓			✓
Kielder			✓			✓	

3.3 Site Description

3.3.1 Location

Moor House-Upper Teesdale NNR is located in Upper Teesdale, Cumbria (54°65’N 2°45’W), as shown in Figure 3-1. The site is situated approximately 6 km south of the village of Garrigill. The site covers an area of approximately 7,500 ha, and includes Great Dunn Fell, the highest point in the north Pennines (848 m AOD). The River Tees forms the northern boundary of the reserve.

Moor House NNR was established as a nature reserve in 1952 and was subject to much research under the International Biological Programme. A field station was located in the former shooting lodge until 1999. The site was merged in 1999 with the Upper Teesdale NNR to form Moor House-Upper Teesdale NNR, The two sites are divided by Cow Green Reservoir which was constructed between 1967 and 1971 (Adamson 2009). The site is one of the ECN's terrestrial and freshwater monitoring sites. The ECN's programme of work was established in 1992 and aimed to monitor environmental change over time, through measurements of air, water, ecological and soil quality (ECN 2010a). Moor House is one of the largest areas of blanket bog in upland England and became a SSSI to allow studies of moorland ecology and change to be carried out (Heal & Smith 1978). The site is also a UNSECO Biosphere Reserve and a European Special Protection Area (ECN 2010b) due to the unique combination of vegetation (arctic, alpine and continental) present at the site.

Within the boundaries of the nature reserve, there are areas which have been subjected to different management practices. A summary of these locations is provided below; the locations of the managed areas are shown on Figure 3-2.

3.3.1.1 The Hard Hill Managed Plots – Burnt, Grazed and Unmanaged

The Hard Hill Experiment Plots were set up in 1954 to establish the effects of burning and grazing on plant species. There are four experimental blocks, which are subdivided into six plots. The experimental plots are centred on 54°41'N 29°93'W, at approximately 678 m AOD. Each plot measures approximately 30 by 30 m. Three plots within each block are fenced to prevent access from grazing animals. Within the fenced area, one plot is burnt every 10 years, one every 20 years and one is not burnt at all. Outside of the fenced area, the same burning treatments are replicated. Grazing was mainly from sheep and is reported to be light (noted as 0.04 sheep ha^{-1}) Ward et al. (2007). The burnt (every 10 years) plot was last burnt in February 2007, the burnt (every 20 years) site was last burnt in 1995. The layout of the managed plots is shown on Figure 3-3. The site that has been burnt every

20 years in Block A at Hard Hill is depicted in Figure 3-4; the northern end of Block A is depicted in Figure 3-5.

The majority of work on Hard Hill was carried out on Block A, however samples were also collected on the burnt (every 10 years) sites on Blocks B (B10B) and C (B10C) to identify the extent of variation between sites that have been subjected to the same treatment/land management practice. Due to limitations in terms of time and resources, Blocks B and C could not be used for all analyses.

3.3.1.2 The Afforested Site

The afforested plot is situated immediately north of Great Dodgen Pot Sike, and is centred on 54°41'N 2°21'W, 550 m AOD. The plot covers an area of 60 m by 100 m and was planted in the 1950s. Prior to planting, a series of drains were installed to lower the water table, creating a series of ridges and furrows. Sitka spruce trees were planted on the ridges. Little management of the plot has taken place since planting and as a result the trees have not been "thinned", which would be normal practice on such a plantation. The afforested site is depicted in Figure 3-6.

3.3.1.3 The Drained Site

Burnt Hill is located at 54°41'N 2°22'W and is situated at 570 m AOD. The site was drained in 1952; the drains were spaced at approximately 10 to 15 m intervals, were 0.5 m deep and ran perpendicular to the slope of Burnt Hill. Two years prior to draining, a fire occurred at the site; however, the site has not been burnt since. (Stewart & Lance 1991). The drained site is depicted in Figure 3-7.

Fieldwork was carried out at the three locations (Hard Hill, Burnt Hill and the afforested site) described above; aerial photographs of the selected sites are shown on Figure 3-8 to 3.10.

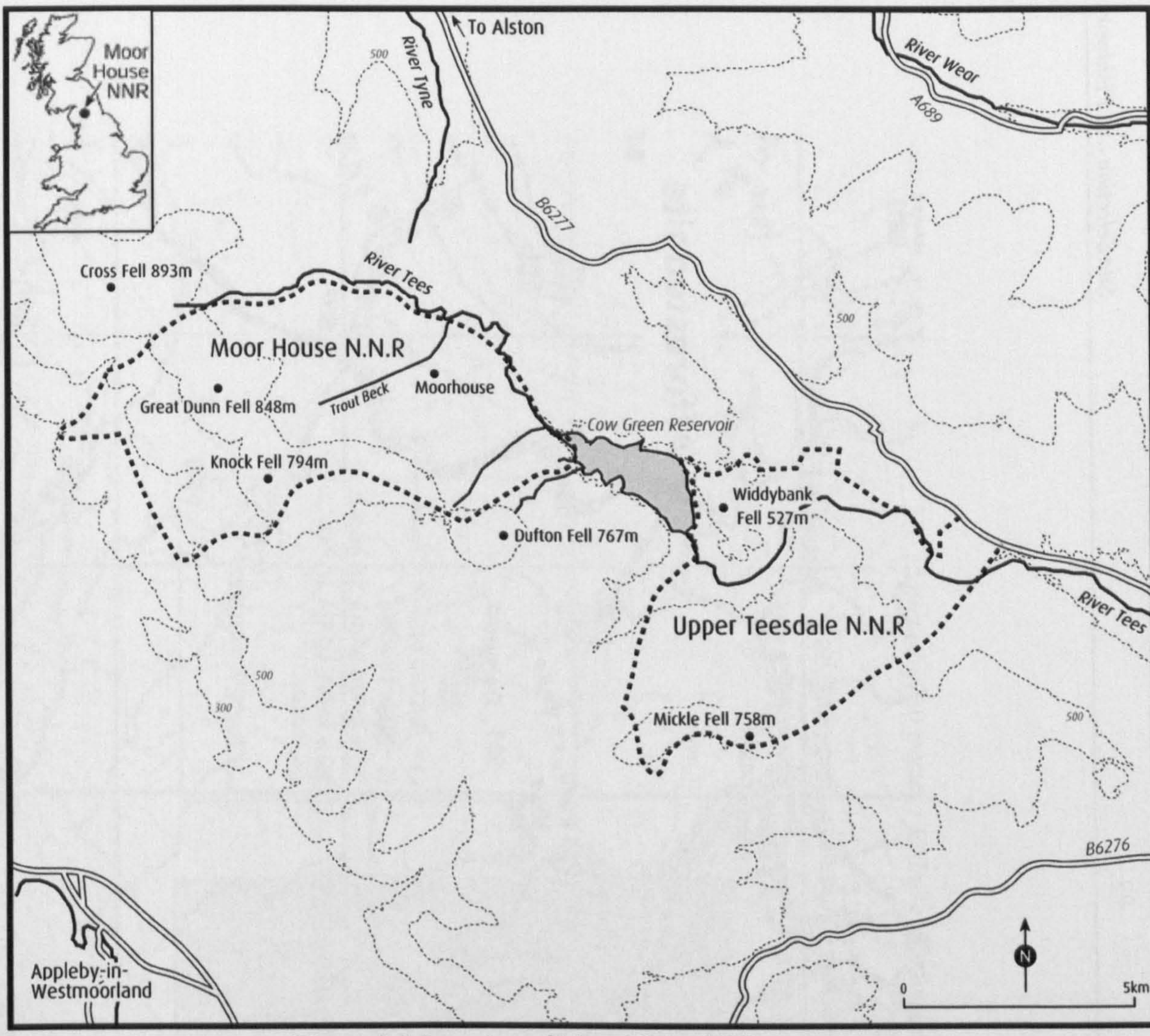


Figure 3-1 Location of Moor House NNR

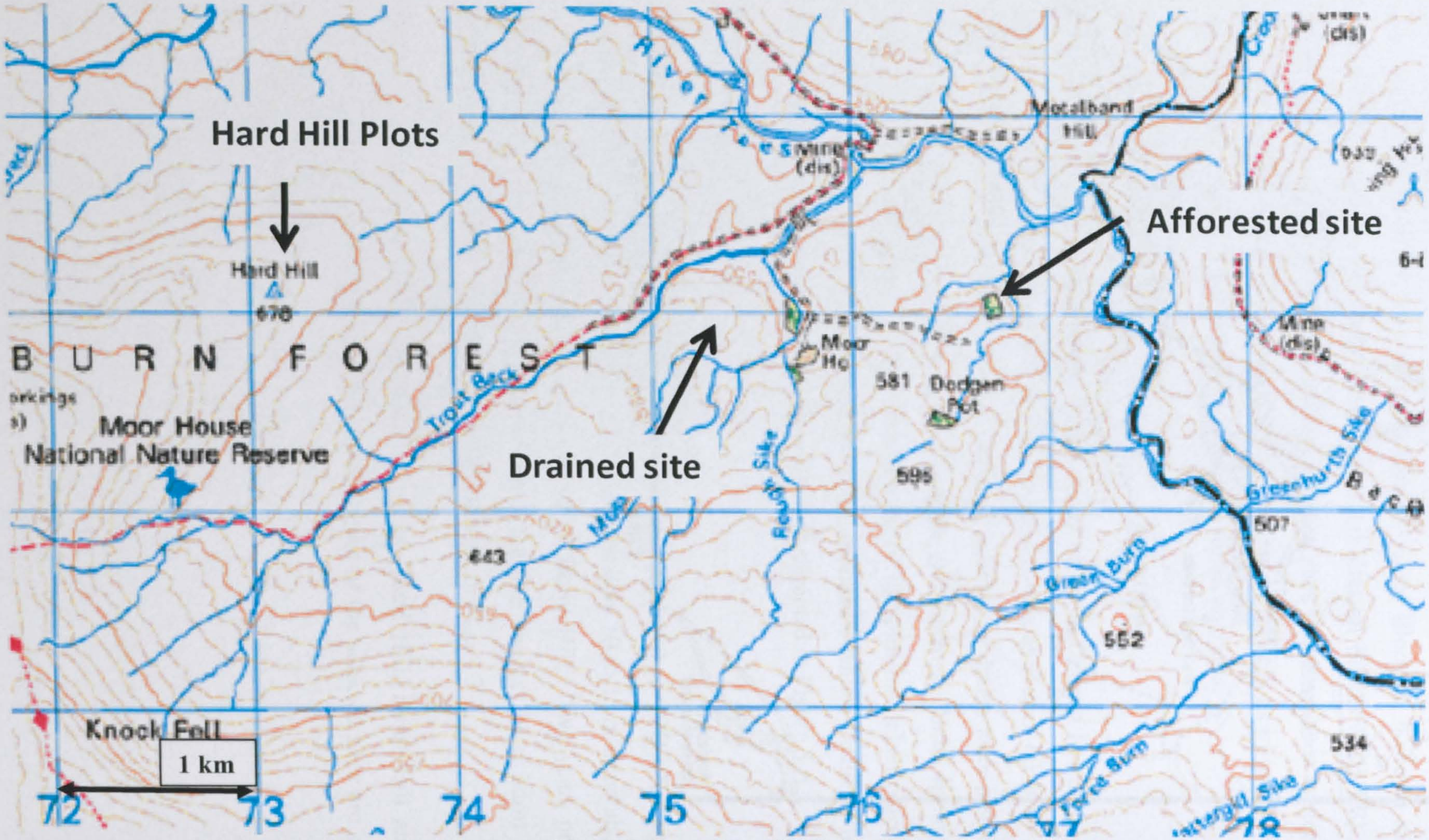


Figure 3-2 Location of Each Monitoring Site

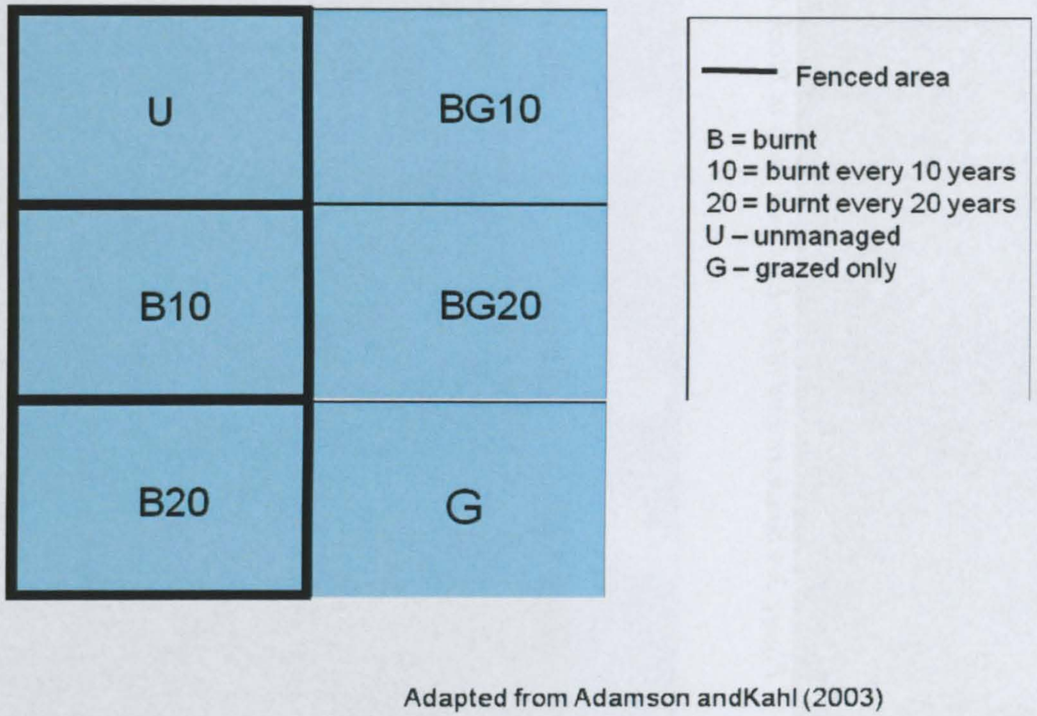


Figure 3-3 Experimental Setup of Block A at Hard Hill

Figure 3-4 Southern End of the Enclosed Plots in Block A at Hard Hill

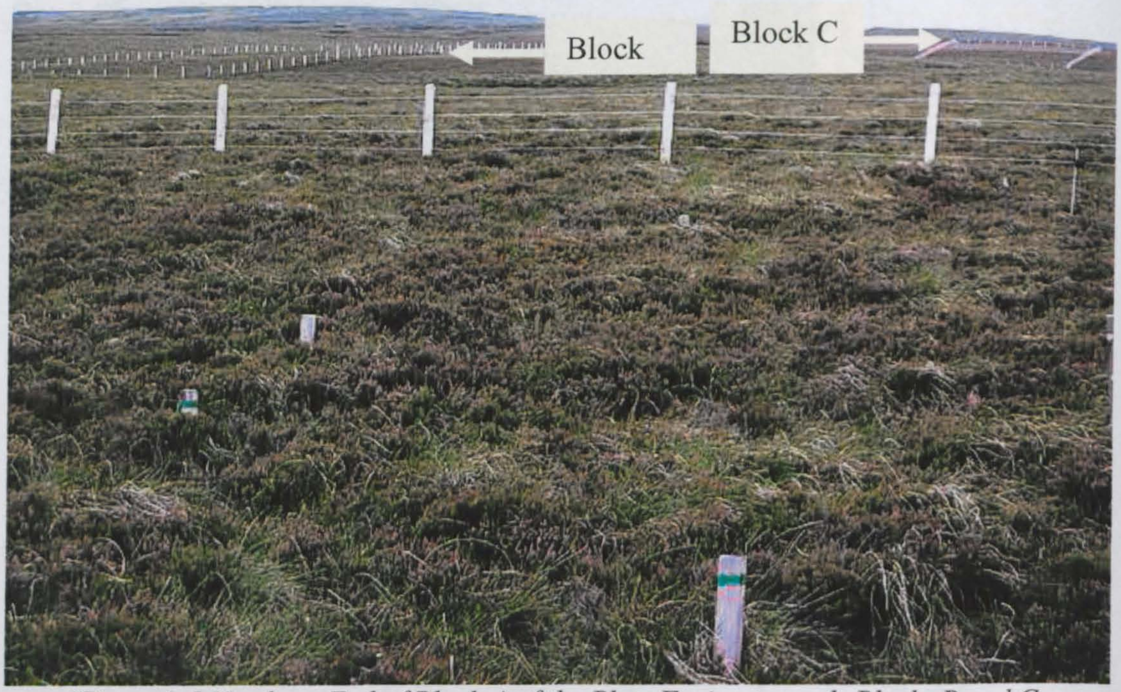


Figure 3-5 Northern End of Block A of the Plots Facing towards Blocks B and C

Figure 3-6 Afforested Site Pictured from the East

Figure 3-7 The Drained Site

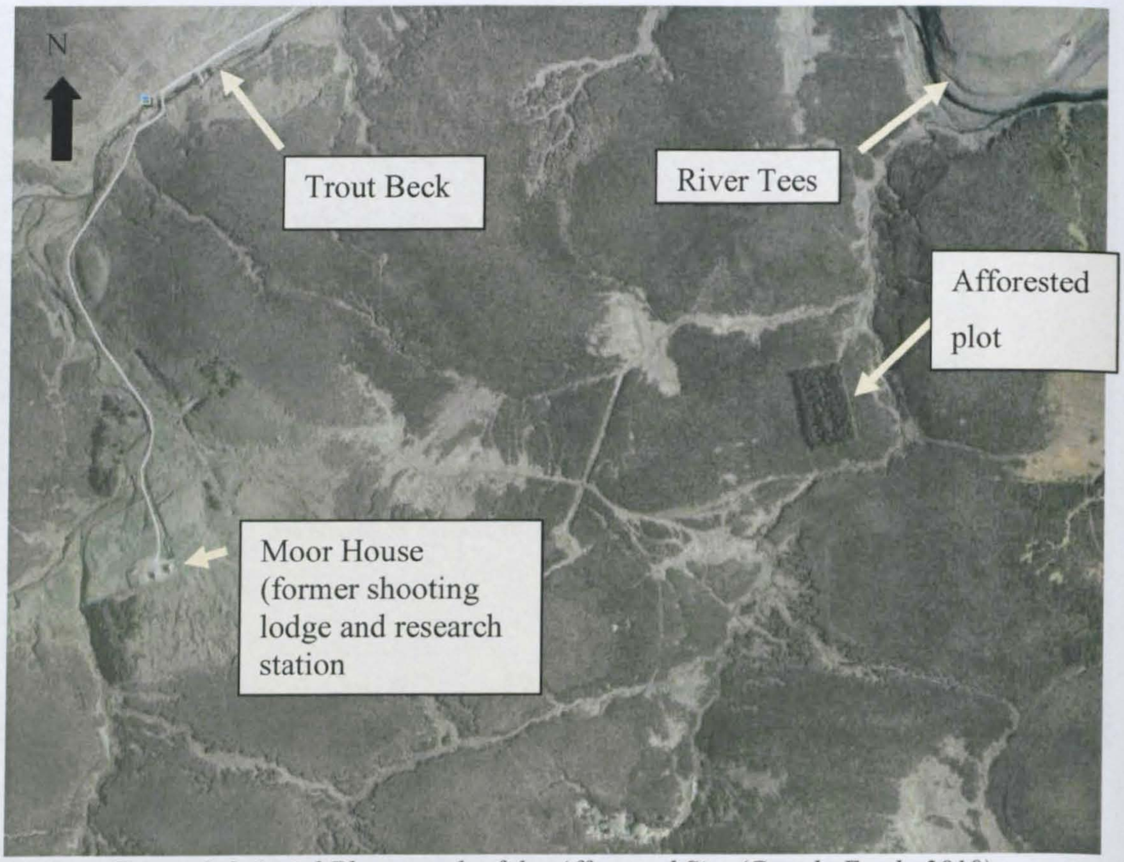


Figure 3-8 Aerial Photograph of the Afforested Site (Google Earth, 2010).

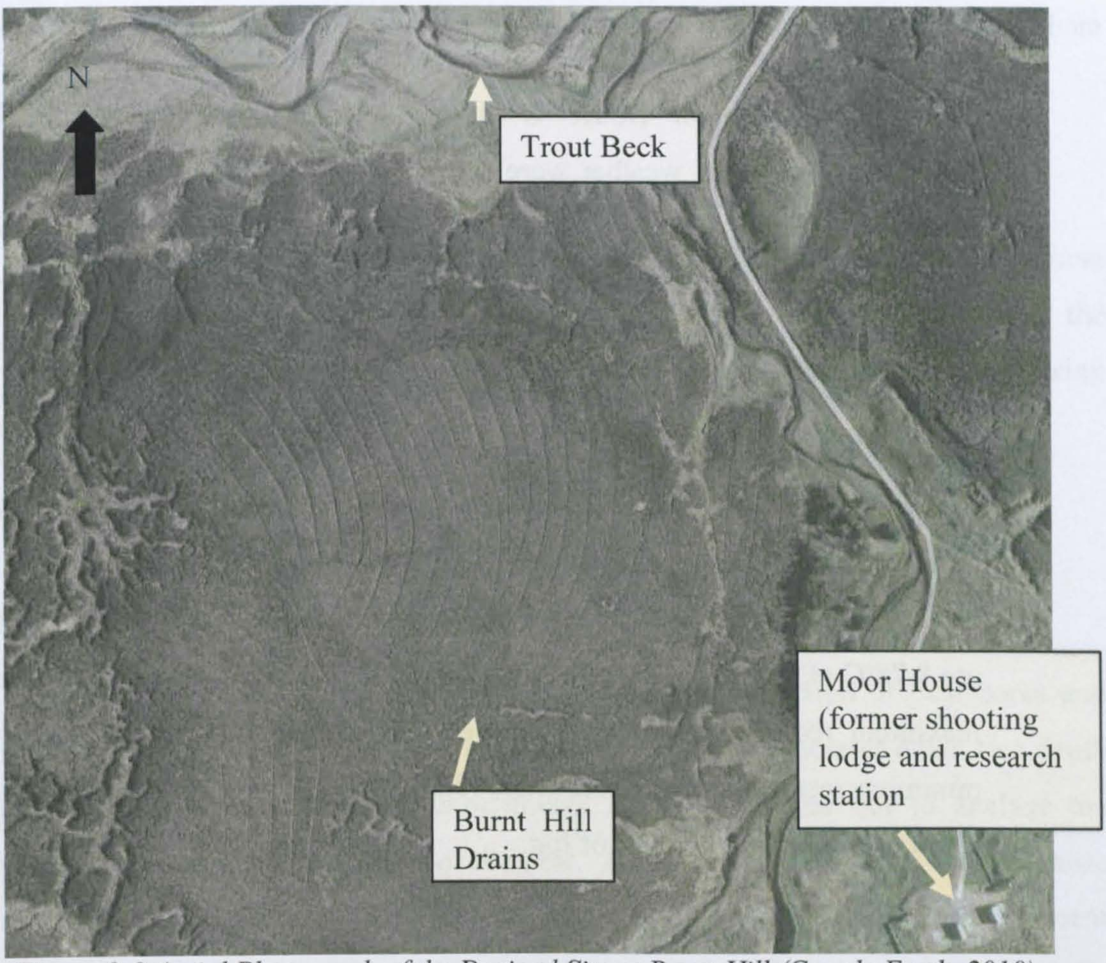


Figure 3-9 Aerial Photograph of the Drained Site at Burnt Hill (Google Earth, 2010)

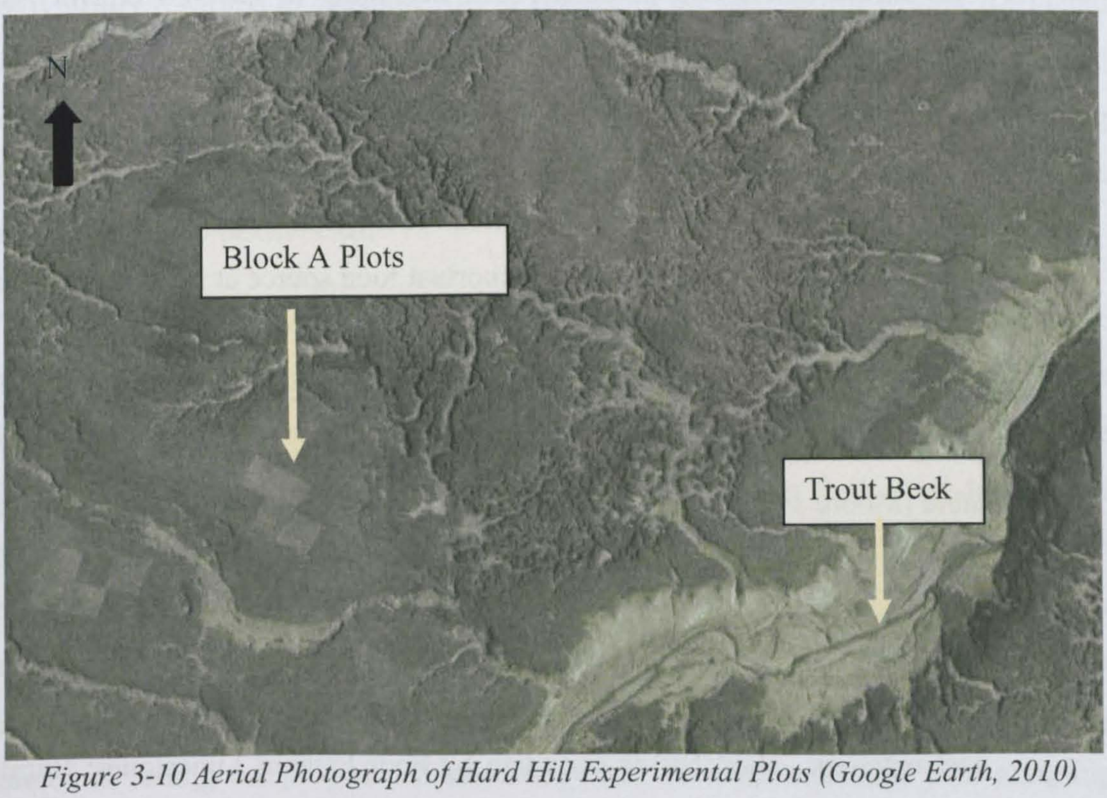


Figure 3-10 Aerial Photograph of Hard Hill Experimental Plots (Google Earth, 2010)

3.3.2 Climate

The climate at Moor House is classed as sub-Arctic/oceanic (ECN 2010b). Instruments to record weather were first installed by Gordon Manley in 1932; and were used up until 1952. The Nature Conservancy Council recorded weather data adjacent to the shooting lodge between 1952 and 1980, whilst operating the meteorological station. In 1991 an Automatic Weather Station (AWS) was installed and is operated by the ECN. Records between 1980 and 1991 are not available, but work by Holden and Rose (2010) used data from the Widdy Bank weather station (6.6 km southwest of Moor House) to interpolate values for this period of time. Mean annual precipitation at Moor House is 2012 (±470) mm, with an average of 15 days a year when snow is lying on the ground. Mean daily maximum temperatures are 8.78°C (±5.64 °C) with mean daily minima of 2.87 °C (±4.78 °C). The absolute maximum recorded temperature between 1931 and 2006 was 27.6°C; whilst the minimum recorded temperature was -18.5°C. On average, air frosts are present for 99 days a year, with 52 days of fog.

3.3.3 Geology and Soils

The site is overlain by blanket peats, belonging to the Winter Hill Association. These soils are described by the Soil Survey of England and Wales (1980) as deep, acid peat soils that are frequently saturated. Blanket peats in the Pennines are believed to have begun forming between 7,500 and 5,000 years ago at the time when human activities intensified (Charman 2002) and trees were cleared to allow improved hunting of red deer – an important food source at the time. The removal of trees from the North Pennines has been associated with shallower water tables as interception reduced the amount of rainfall reaching the soil surface, whilst transpiration and uptake of water through tree roots reduced the level of the water table (Moore 1975). Following on from the removal of trees and an increase in the level of the water table, peats would have developed through the processes described in Chapter 2.

The solid geology at Moor House comprises a series of limestones with occasional coal outcrops. Hard Hill is underlain by Four Fathom Limestones; Burnt Hill and the afforested plot are underlain by Tyne Bottom Limestone (Johnson & Dunham 1963).

3.3.4 Vegetation

The vegetation at Moor House is predominantly a combination of heather (*Calluna vulgaris)*, sedges (e.g *Eriophorum)* and moss (e.g. *Sphagnum*) at all sites with the exception of the afforested site where Sitka spruce were planted on the ridges during the 1950s.

3.4 Fieldwork

3.4.1 Introduction

The fieldwork carried out at Moor House comprised collection of peat cores and field monitoring. The peat cores were collected to provide samples down to a depth of 0.5 m on which laboratory experiments could be carried out to analyse the physical and chemical properties of the peat. A value of 0.5 m was selected because a) peat grows at a rate of 1 mm a year, therefore the effects of land management since the 1950s will be most evident in the upper layers of the peat; b) the acrotelm is typically regarded as the upper 0.3 m portion of the peat profile (Tallis 2001) and it is within this zone where changes are most likely to be seen due to the increased rates of biogeochemical cycling in the aerobic zone (Belyea 1996). By measuring the 0.2 m beneath the acrotelm, comparisons of the effects of management on the two zones could be assessed.

The field monitoring was carried out to collect data on carbon losses and water table fluctuations during monitoring periods. Instrumentation was installed at each of the differently managed sites to allow field monitoring to be carried out to observe losses of carbon from peat in both gaseous and aqueous form, and to monitor groundwater levels. Figure 3-11 provides an overview of the methods used in the field and demonstrates how they link to the laboratory work. Details of the methods used in the laboratory are detailed in Chapters 4, 5, 6, and 7 where the results are presented and discussed. The initial phase of fieldwork comprised a pilot study aimed at testing the proposed methodology and devising a suitable peat sampling regime. Further details of the pilot study are provided below

3.4.2 Sample Design

The sampling strategy was based on the results of a pilot study. The aim of the pilot study was to test the proposed methodology and to determine the number of samples to be collected at each location. The fieldwork for the pilot study was undertaken between 13th July and 4th August 2008. Two plots from Block A of the Hard Hill experimental plots at Moor House were studied – ungrazed and burnt every 10 years and the plot which is grazed and burnt every 20 years. Peat cores were collected using a Russian peat sampler. The cores were placed in plastic guttering, and wrapped with cling-film in order to secure them during transit back to the laboratory. Surface cores were collected using a plastic bulk density tube. Each surface core was wrapped in a plastic sampling bag.

The sampling design on the burnt plot was based on guidance published by Sykes and Lane (1996) for the ECN's target sites recommends sampling every five metres on a regular grid. When plotted out, and excluding samples taken on the boundaries of the site, this gave a total of 25 samples to be collected.

A random sampling design was chosen rather than a systematic grid to prevent bias, but was modified to ensure that representative coverage of the sites was achieved. The plots were sub-divided into four areas; six sampling locations were selected in each area using grid co-ordinates created using a random number generator. One additional sample was collected from a randomly selected location. Samples located in the centre of the site were re-located to avoid sampling within a permanent quadrat set-up to record ecological change on the plots through time.

A total of 10 samples were collected at the plot which is grazed and burnt (every 20 years), five surface samples and five cores. Results from the laboratory work carried out on samples taken during the pilot study have been incorporated into Chapters 4, 5 and 6.

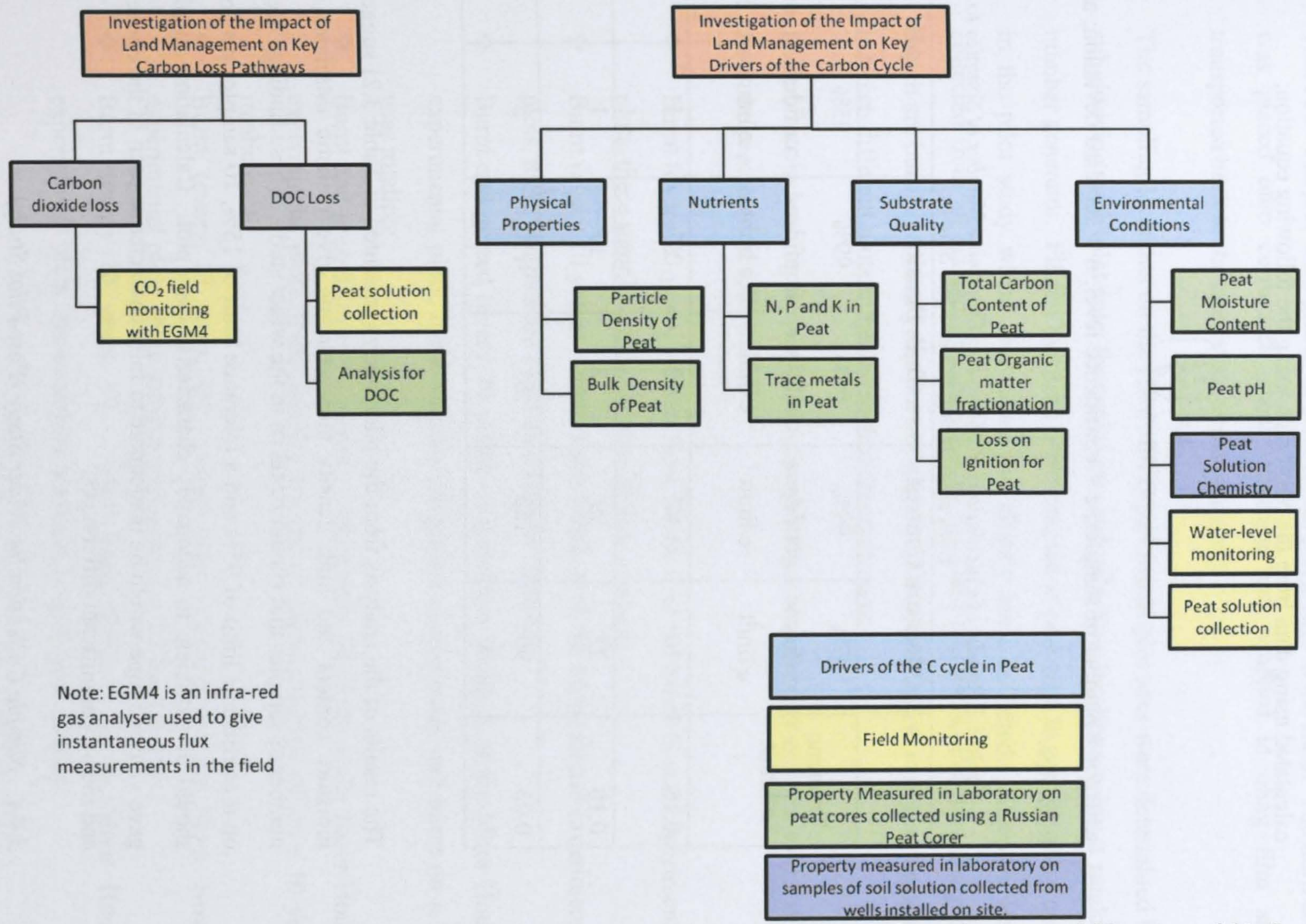


Figure 3-11 Organogram Demonstrating Properties to be Analysed and Measurements to be Made to Achieve the Aims and Objectives of this Research

3.4.3 Calculation of Number of Samples

Estimates of the number of samples to be used in the wider field study were calculated using data from the burnt plot, using the following equation:

$$n = \left(\frac{z\,\sigma}{d}\right)^2$$

where n = number of samples; z = confidence level; σ = standard deviation; and d = tolerance.

Table 3.2 Results of Calculations Performed to Determine Number of Samples to be Collected During Wider Field Study

	Moisture Content		Bulk Density		pH	
Tolerance Limit	95% confidence limit	99% confidence limit	95% confidence limit	99% confidence limit	95% confidence limit	99% confidence limit
0.15	10	17	10	17	1	1
0.10	22	39	22	37	1	1
0.05	90	155	87	150	2	3

The results of the analysis from the pilot study (presented in Table 3.2) suggest that moisture content and bulk density measurements give realistic estimates of the number of samples that should be taken in the wider study. These findings are based on a confidence limit of 95% and a tolerance limit of 15%, 10 samples per location should be sufficient to adequately characterise each plot. Calculation on pH data gave values of one would be inadequate to fully characterise each of the chosen sites and identify significant differences.

3.4.4 Sample Collection for Wider Study (Post-Pilot Study)

Samples were collected from ten managed sites at Moor House, on each site there were fifteen sampling locations. Samples were collected using a Russian peat corer.

Three cores were collected at each sampling location; two cores were divided into five sections measuring 10 cm in length to allow changes with depth to be analysed. These sub-samples were placed in labelled sampling bags in the field. The third core was placed onto corrugated plastic sheeting and wrapped in cling film and transported back to the laboratory as a whole core.

The sampling locations on the Hard Hill experimental plot sites were determined by sub-dividing each plot into four sections, and selecting locations using a random number generator. Fifteen locations were sampled at each plot, except for those used in the pilot study where five cores had already been collected. Samples were collected from all six plots on Block A of the Hard Hill plots, and from the plots which are burnt every ten years but not grazed on Blocks B and C to assess variation between different plots subjected to the same treatment.

Differences in burning frequency and the plot from which burnt samples were taken from are presented as follows:

- ❖ Burnt (every 20 years) – taken from Block A of the Moor House experimental plots, these samples are burnt on a 20 year rotation;
- ❖ Burnt (every 10 years) – taken from Block A of the Moor House experimental plots, these samples are burnt on a 10 year rotation;
- ❖ Burnt and grazed (every 20 years) – taken from Block A of the Moor House experimental plots, these samples are grazed continuously and burnt on a 20 year rotation;
- ❖ Burnt and grazed (every 10 years) – taken from Block A of the Moor House experimental plots, these samples are grazed continuously burnt on a 10 year rotation;
- ❖ Burnt (every 10 years, B) – taken from Block B of the Moor House experimental plots, these samples are burnt on a 10 year rotation;
- ❖ Burnt (every 10 years, C) – taken from Block C of the Moor House experimental plots, these samples are burnt on a 10 year rotation.

In addition, samples were collected from Burnt Hill (drained in 1952), fifteen cores were taken along three transects, spaced at five metre intervals either side of a drain close to the brow of the hill. More samples were collected in the afforested site as sampling locations comprised a mixture of ridges and furrows. Samples were collected from a total of nineteen locations in the afforested site, along four transects from both ridges and furrows through the site. In order for the afforested site to be planted, drains were installed to reduce the water table and prevent the tree roots from rotting. The afforested site will only be referred to as being afforested for the remainder of the text.

Note should be made that only the sites on Hard Hill that were not enclosed by fences were grazed, sheep at Moor House are excluded from both the drained and afforested sites. During sample collection note was taken of the class of vegetation present at each of the sampling locations. The locations at which cores were collected are presented on Figures 3.12 to 3.21.

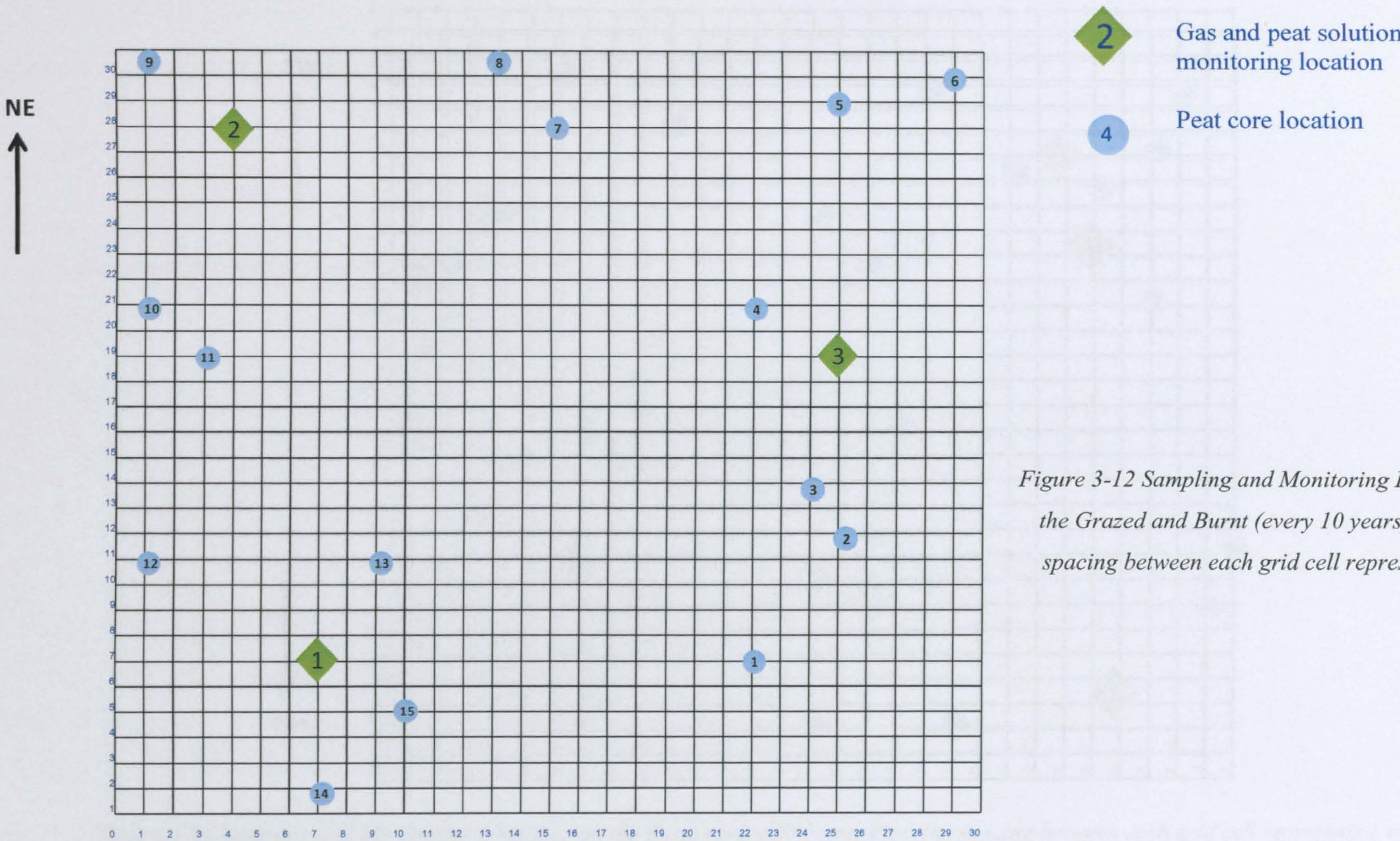


Figure 3-12 Sampling and Monitoring Locations on the Grazed and Burnt (every 10 years) Plot (the spacing between each grid cell represents 1 m)

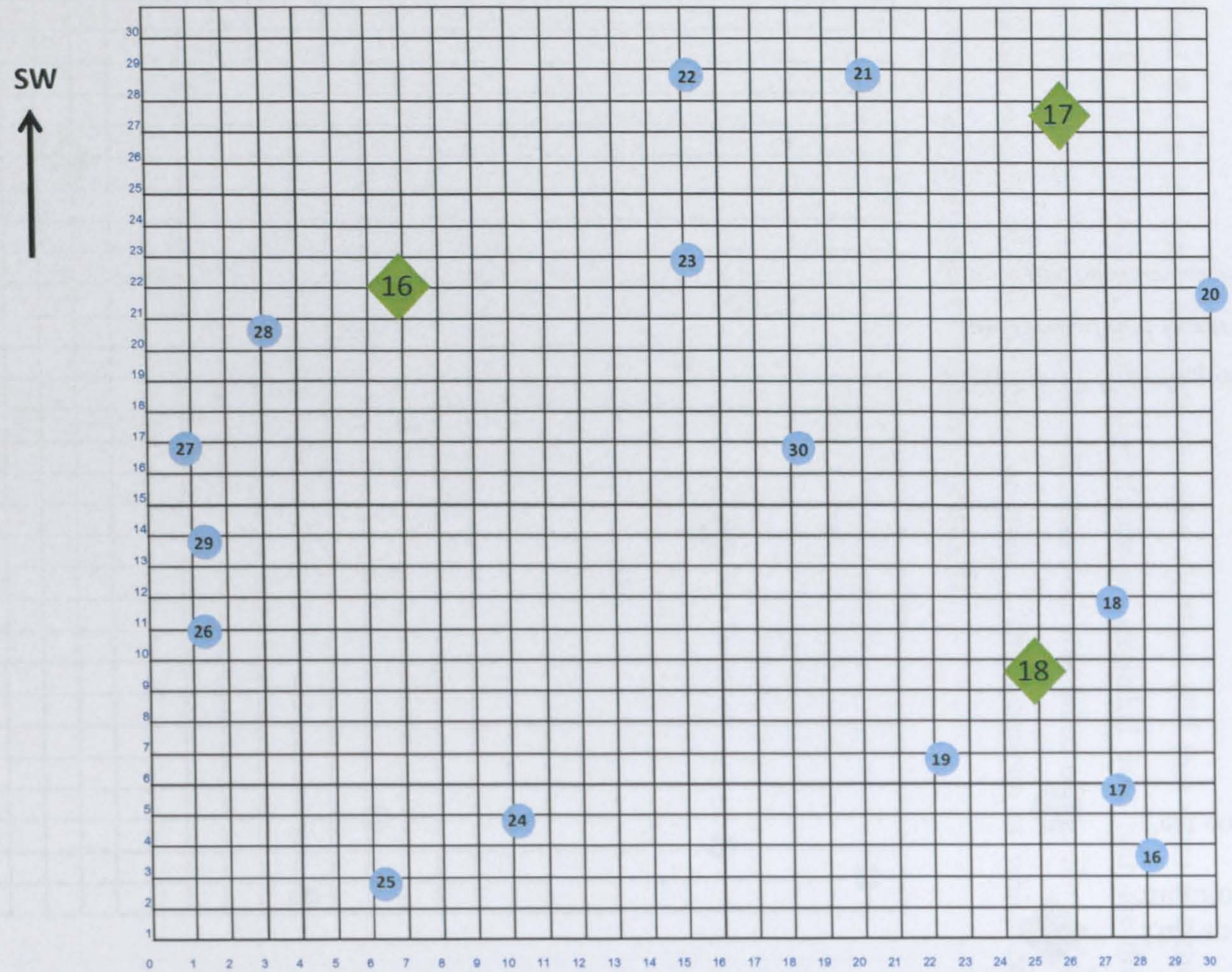


Figure 3-13 Sampling and Monitoring Locations on the Unmanaged Plot (the spacing between each grid cell represents 1 m)

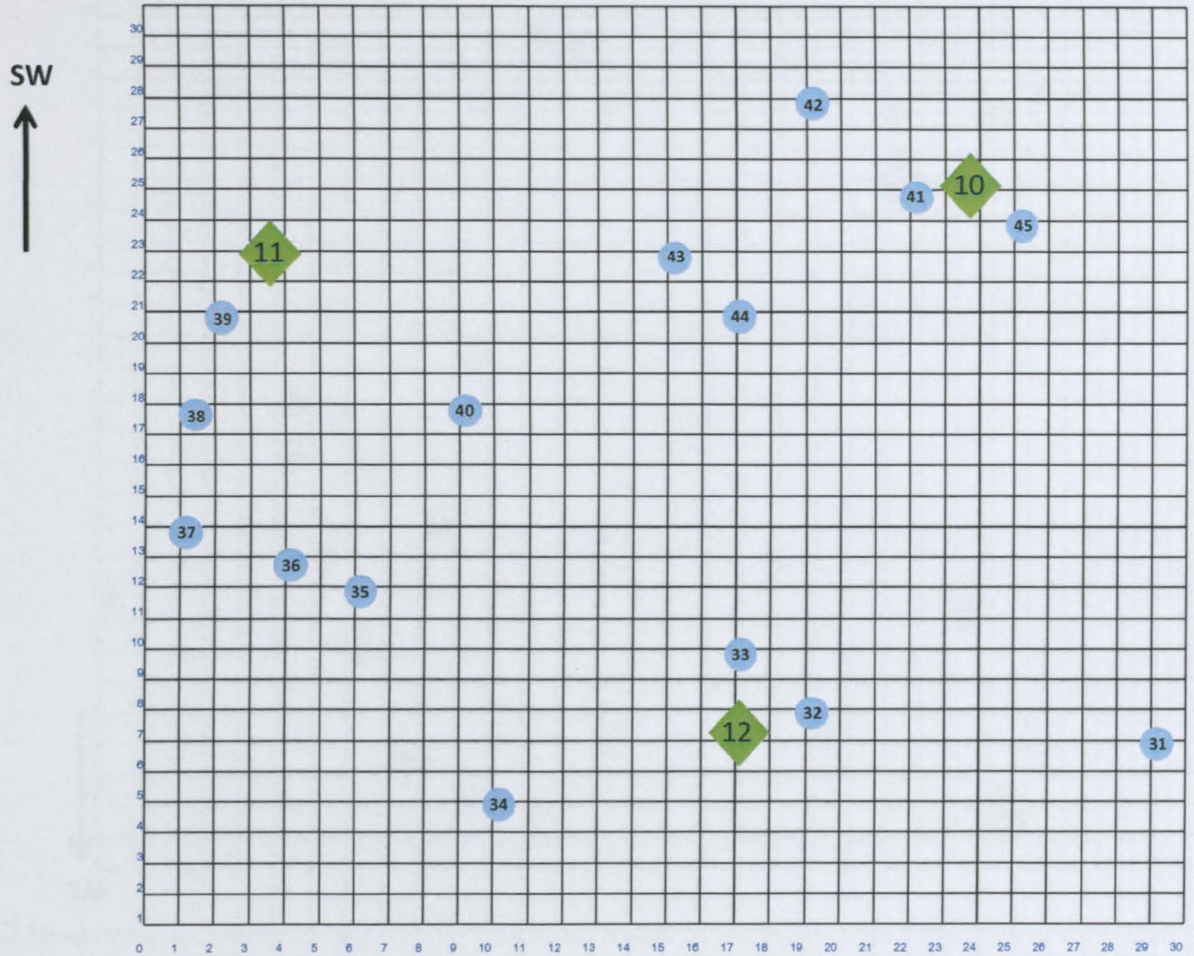


Figure 3-14 Sampling and Monitoring Locations on the Burnt (every 20 years) Plot (the spacing between each grid cell represents 1 m)

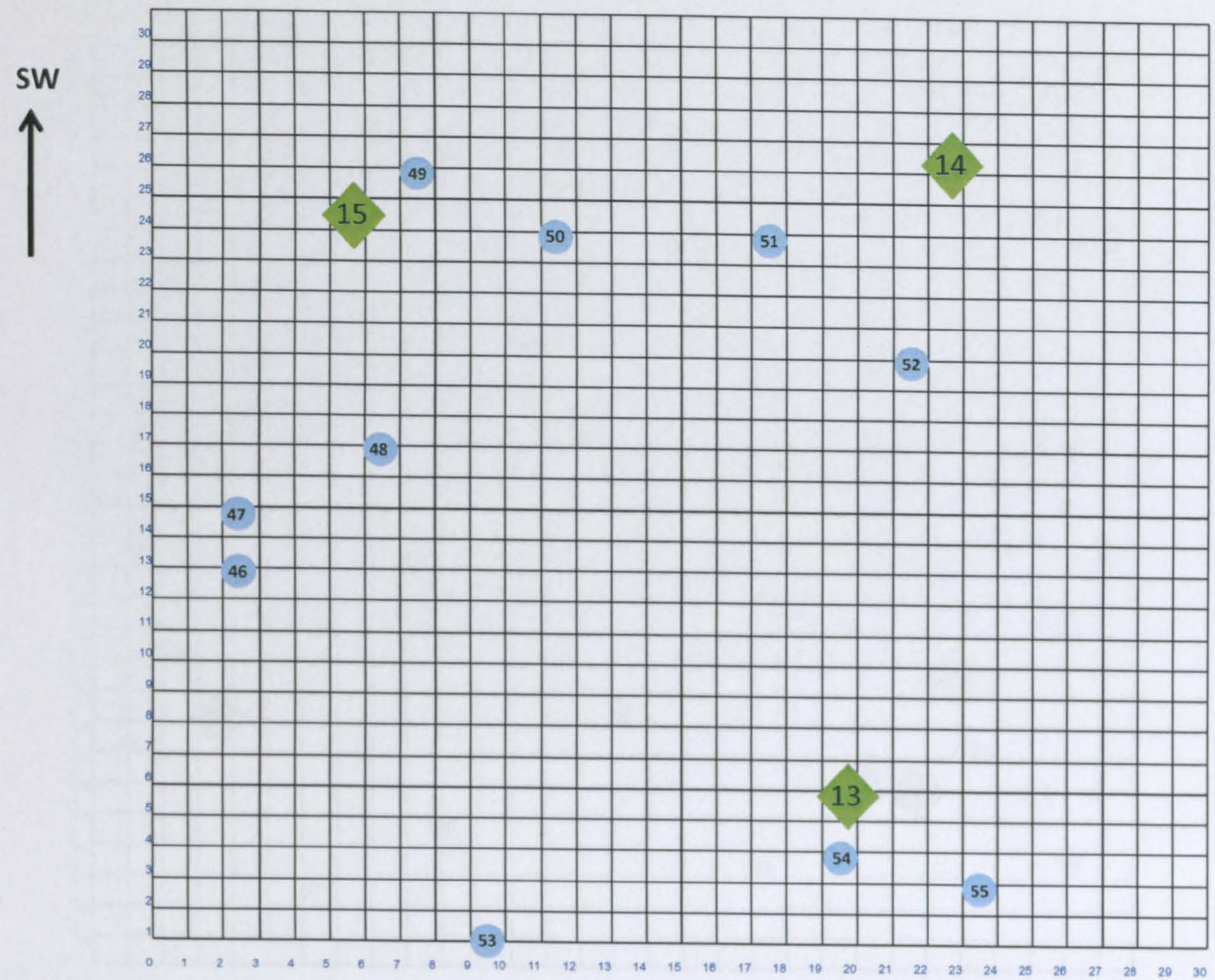


Figure 3-15 Sampling and Monitoring Locations on the Burnt (every 10 years) Plot (the spacing between each grid cell represents 1 m)

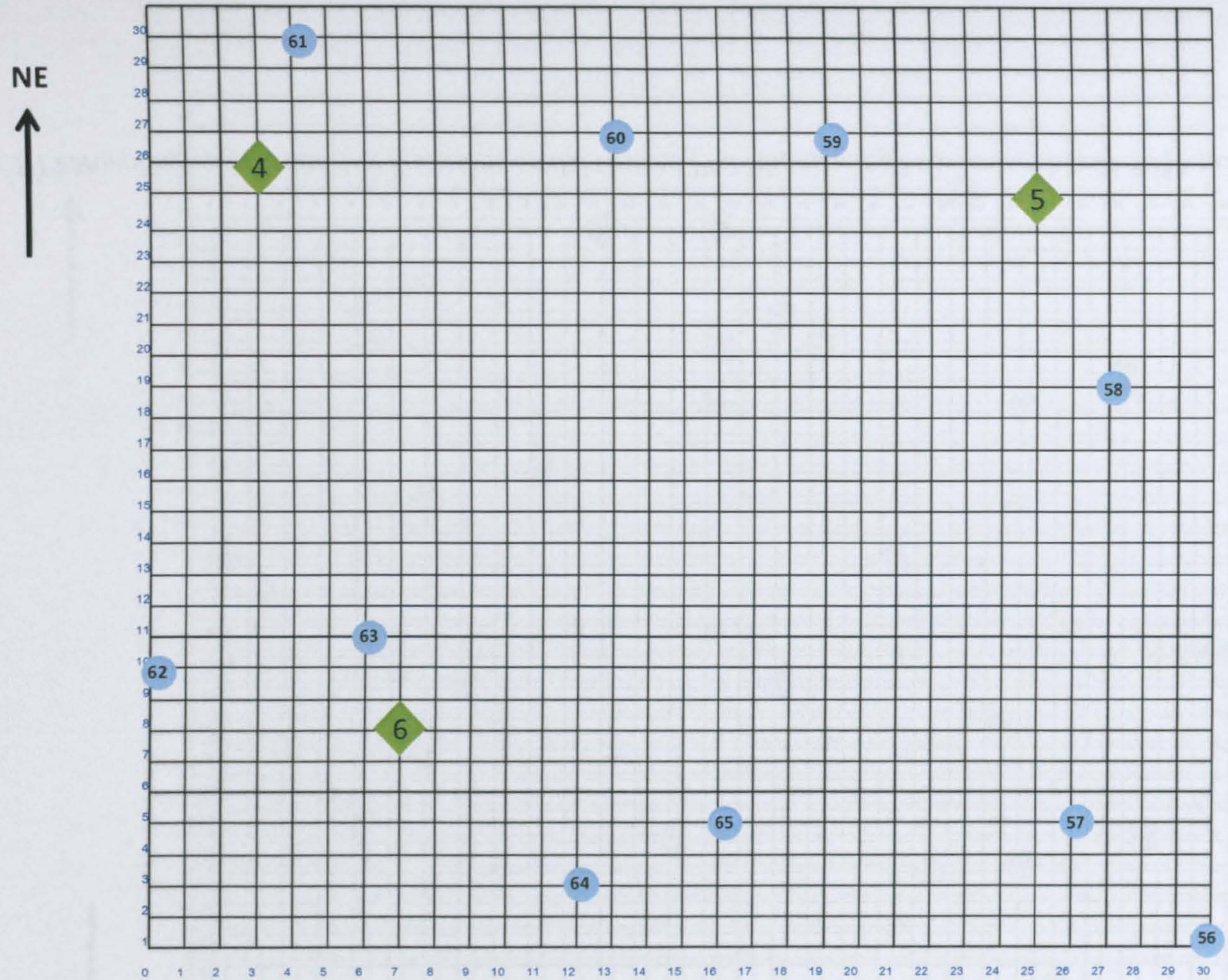


Figure 3-16 Sampling and Monitoring Locations on the Grazed and Burnt (every 20 years) Plot (the spacing between each grid cell represents 1 m)

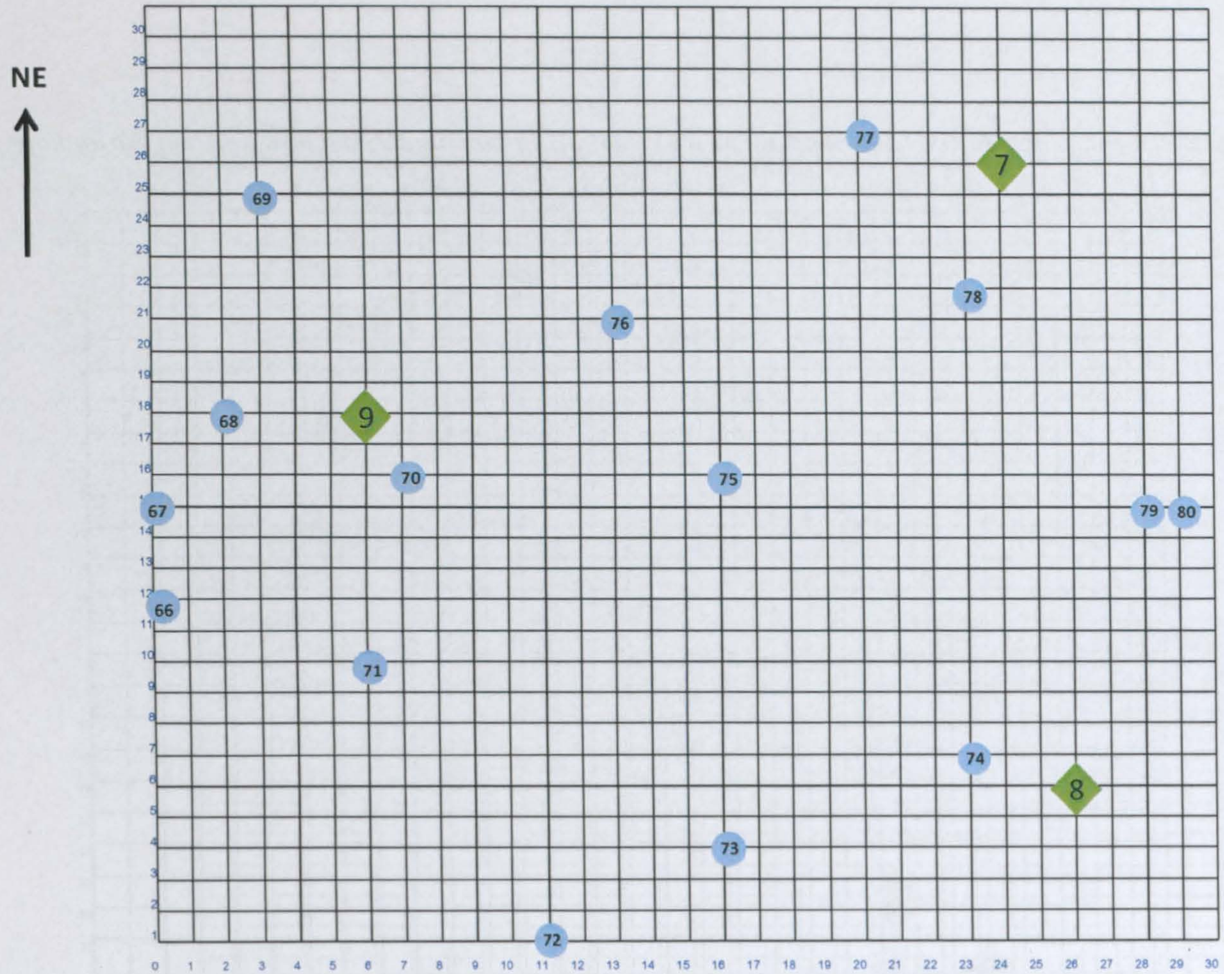


Figure 3-17 Sampling and Monitoring Locations on the Grazed Plot (the spacing between each grid cell represents 1 m)

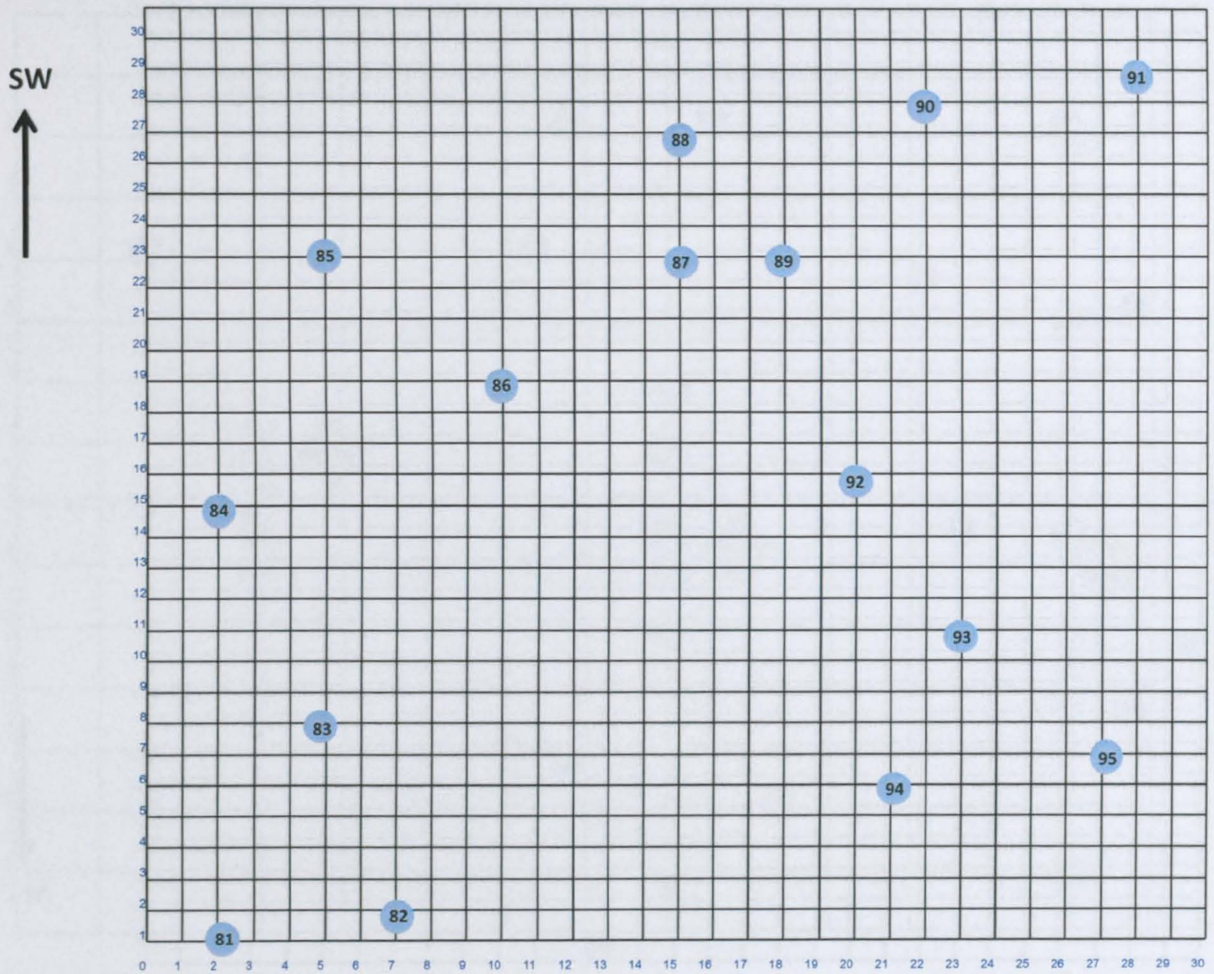


Figure 3-18 Sampling and Monitoring Locations on the Burnt (every 10 years) Plot on Block B (the spacing between each grid cell represents 1 m)

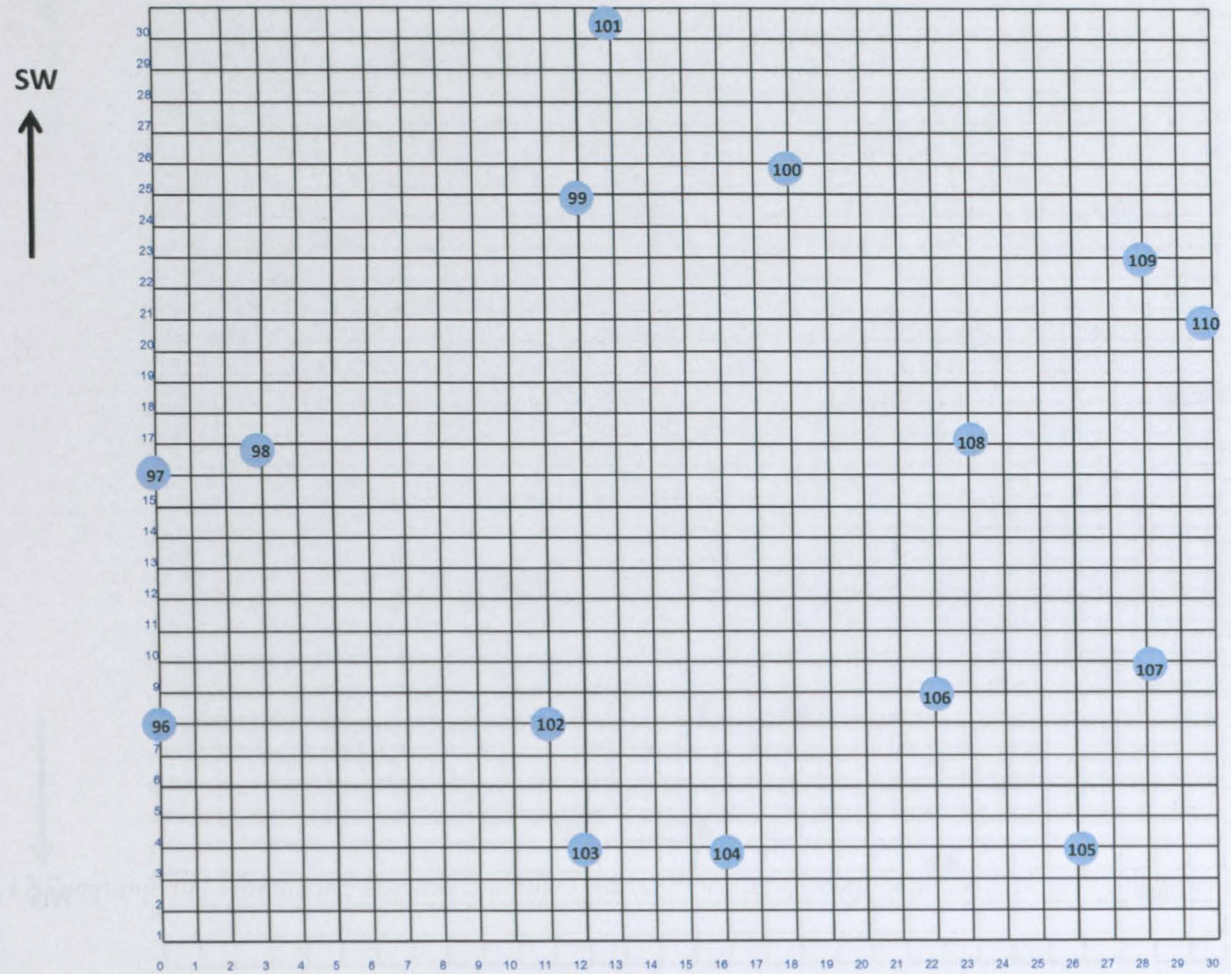


Figure 3-19 Sampling and Monitoring Locations on the Burnt (every 10 years) Plot on Block C (the spacing between each grid cell represents 1 m)

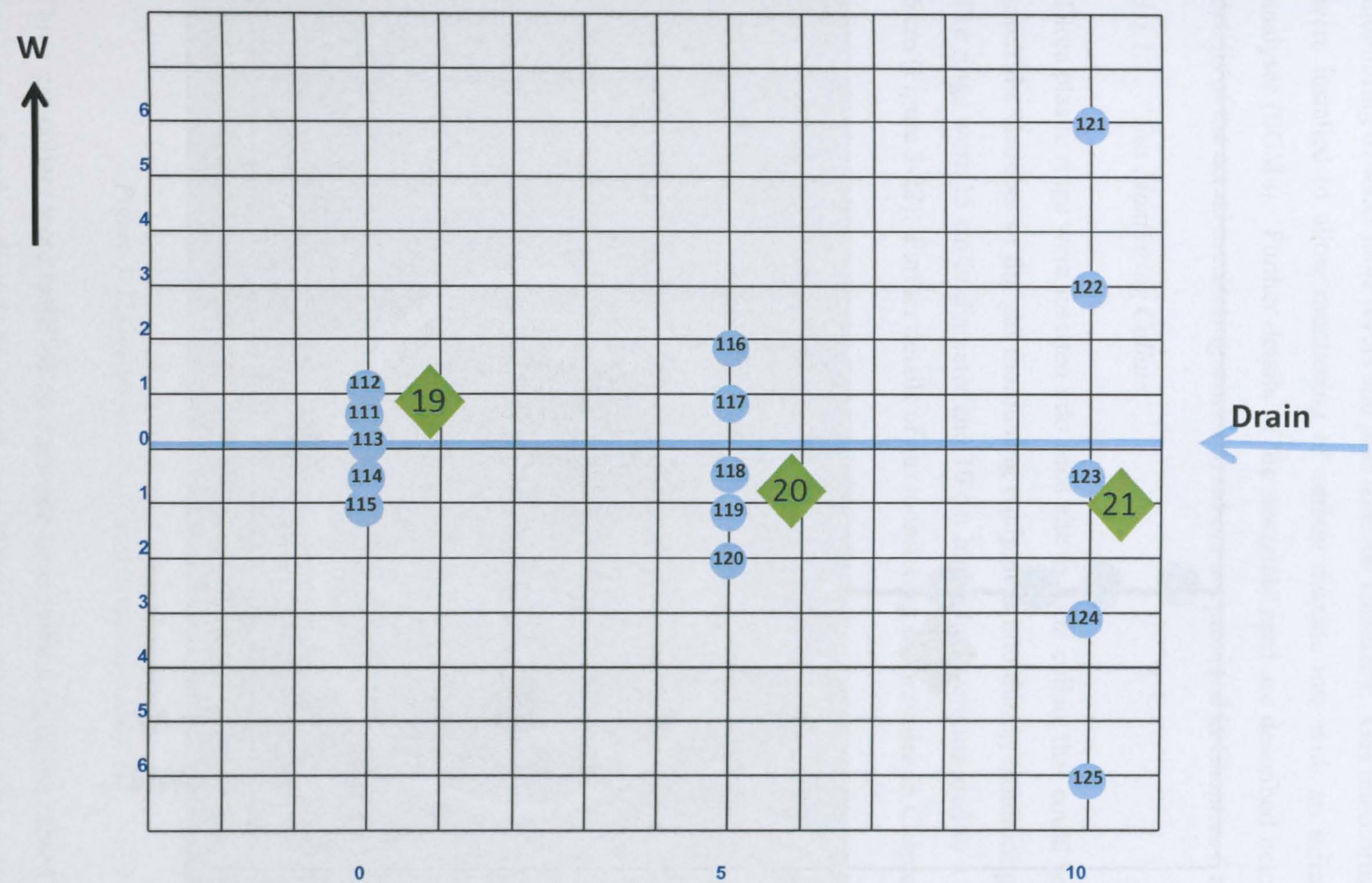


Figure 3-20 Sampling and Monitoring Locations on the Drained Site

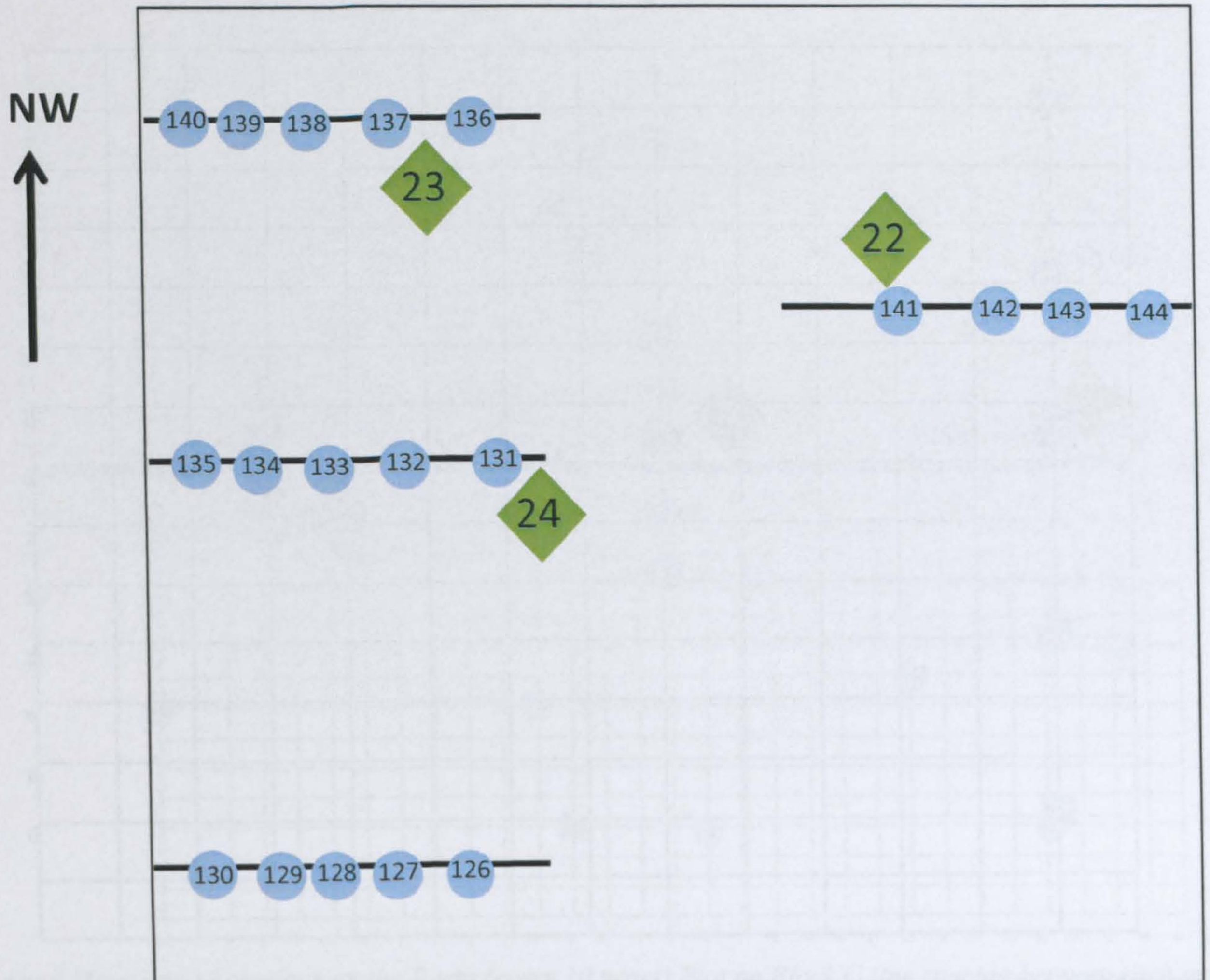


Figure 3-21 Sampling and Monitoring Locations on the Afforested Site

3.5 Field Monitoring Installations

Peat solution wells were installed on each differently managed site to allow monitoring of water table levels and peat solution chemistry. Gas monitoring collars were installed to allow monitoring of carbon dioxide loss with an infra-red gas analyser (EGM4). Further details of the materials used are described below. Full details of the actual monitoring work carried out are provided in Chapters 6 and 7.

3.5.1.1 Gas Monitoring Collars

Three plastic rings were inserted into each site to form collars that could be used to insert the chamber of the gas monitoring equipment into during monitoring rounds. The rings were 15 cm in diameter and 10 cm high, they were inserted to a depth of 5 cm (Figure 3-22). Further details of gas monitoring are provided in Chapter 6.

Figure 3-22 Gas Monitoring Collar in the Afforested Site

Three gas collars were installed on each site to provide a triplicate record of gains and losses of carbon dioxide from peat. In addition, having three collars on each site provided a number of collars that could be monitored on a practical timescale with the resources available. Analysis of monitoring data between collars presented in chapter 6 suggested little variation was found between the collars within each treatment.

3.5.1.2 Peat Solution Monitoring Wells

Monitoring of water table levels was carried out in specially designed wells made from 2.5 cm diameter plastic pipe with holes drilled at 10, 20, 30 and 40 cm; a rubber stopper was placed in the end of each well to prevent the pipe filling with peat during insertion into the peat (Figure 3-23). Rubber bungs were placed in the top of each pipe to prevent rainwater entering the well.

Additional wells were installed at the site to collect peat solution samples for chemical analysis at specific depths within the peat profile: 10 cm, 20 cm and 40 cm below the surface of the peat. The wells were formed from 2.5 cm diameter plastic pipe with holes drilled at the target depth, a rubber stopper in the base of the well and a rubber bung in the top of the well. Peat solution monitoring wells to monitor levels and chemistry were installed at three locations on each site.

Figure 3-23 Peat Solution Level Well

The overall design of the study aimed to reduce bias introduced through pseudo-replication by selecting a field site where variations in geology, soil, climate, aspect and altitude and vegetation were kept to a minimum. In addition, the order in which monitoring work was carried out was varied as well as the time of day during which measurements were taken in a bid to minimise bias.

3.6 Summary

The selection of Moor House NNR as a field site enabled samples of peat to be collected and analysed for the chemical and physical properties of peat that are relevant to the carbon cycle. Details of the laboratory methods used are presented in Chapters 4 to 6. In addition, measurements of carbon losses (both carbon dioxide and DOC) from managed peatlands were made using the field monitoring equipment described in section 3.5. The results of the on-site monitoring are presented in Chapters 6 and 7

4 THE EFFECTS OF LAND MANAGEMENT ON THE CHEMICAL PROPERTIES OF PEATS

4.1 Introduction

As discussed in section 2.4, carbon cycling in peat is dependent on four key factors: substrate quality; environmental conditions, the composition of the microbial community and the nutrient content of the peat (Laiho 2006). In upland areas of the U.K ombrotrophic bogs are the dominant type of peatland. These nutrient poor ecosystems rely on water and nutrient inputs from the atmosphere and releases from decaying plant materials. Nutrients are not only required for plant growth on peatlands (Rydin & Jeglum 2006) but are also required by microbes engaged in the mineralisation of organic carbon into carbon dioxide and methane (Keller et al. 2006). In addition to nutrients, some metallic elements (primarily iron, nickel and cobalt) have been found to be of importance to microbes involved in the breakdown of substrate (Basiliko & Yavitt 2001). Laiho (2006) notes that nutrient availability is determined by environmental conditions and substrate quality.

The degree of saturation and temperature are the two main environmental conditions that are assessed when studying in-situ carbon dynamics in upland peatlands. Decreases in water levels and increases in temperatures have been associated with increased losses of carbon dioxide (Davidson & Janssens 2006). Additionally, other environmental factors such as pH are related to carbon dynamics. Bergman et al. (1999) identified low pH values as a cause for restricted carbon mineralisation rates, peats with pH values of 4.3 had much lower rates of C mineralisation compared to values of 6.8. Changes in pH and peat moisture content will regulate the environment in which microbes live, and in turn will alter the composition of the microbial community.

Despite the significance of the various geochemical factors that drive the carbon cycle that are detailed above, little work has been carried out to determine how land management affects the chemical properties that drive carbon cycling in peat.

4.2 Aim and Objectives

4.2.1 Aim

This chapter aims to identify how land management affects the chemical properties of peats which are relevant to carbon cycling. The chapter provides data that allow a comparison to be made for the first time between different land management practices commonly utilised on UK peatlands; and can be linked to losses of carbon dioxide and dissolved organic carbon which are discussed in Chapters 6 and 7 of this thesis. The chapter examines how moisture content, pH nutrient and metal concentrations vary between for peatland management practices. The chapter will look at how nutrient concentrations vary with depth from the surface, examine any interaction effects with surface vegetation, identify if there is greater variation within one treatment than between treatments and investigate if the chemical properties vary spatial across each treatment. For example, the effect of burning frequency on the chemical properties of the peat will be investigated and the combined influence of burning and grazing will be compared to peatland sites that are only burnt of grazed since these two practices are often combined in upland areas.

4.2.2 Hypotheses

❖ Land management will impact on the chemical properties of differently manage peats in the following ways:

4.2.2.1 Burning

Burning is expected to result in an increase in nutrient concentrations in shallow peats as a result of inputs of ash (Allen 1964), however, as of yet, field studies on burnt peats have not been carried out, neither have comparisons been made to the impacts of other land management practices. Burning has been associated with changes in vegetation community (e.g. Rawes & Hobbs 1979, Hobbs 1984, Ward et al. 2007) which this is also likely to affect the available substrate for mineralisation and consequently the nutrients which are released into the peat as well as the nutrient demands of the vegetation. Moisture content changes are anticipated as the peat dries out due to burning and becomes hydrophobic (DeBano 2000), however, work carried out by Mallick (1986) found the moisture content of the peat to increase after burning. Slight increases in pH values can be expected (Forgeard & Frenot 1996).

4.2.2.2 Grazing

There are two possible options in terms of the effect of grazing on peatland soils: i) concentrations could increase due to inputs of nutrients from sheep faeces; ii) despite the increased inputs of nutrients from faeces, the overall concentrations of nutrients in the peat could decline as an increase in nutrient demand by plants is placed on the peats. Changes in the plant community are likely to occur on grazed peatlands (Alonso et al. 2001), this could affect the substrate available for mineralisation and consequently the nutrients which are released into the peat as well as the nutrient demands of the vegetation. The extent of such effects are noted to depend on stocking density, however, Grant and Hunter (1968) noted that sheep management is much more important than the actually stocking rate, with regular and even grazing resulting in heather regeneration Moisture content could change as a consequence of changes in vegetation species e.g. to species with a greater or lesser water demand. The effects of grazing on pH are unknown.

4.2.2.3 Drainage

Heathwaite (1990) provided data on differences in nutrient contents between drained and undrained peats taken from a lowland site in south-west England. In general, the results suggested a decrease in nutrients concentrations in the drained site, although significance testing was not carried out by the author. This study is the only one found in the UK to provide data on nutrient concentrations in drained peats, no studies appear to exist for upland peatlands. Depleted concentrations of nutrients are expected at drained sites owing to a greater unsaturated zone in which mineralisation rates will be quicker than those found at undrained sites (Gorham 1991). In addition, there is greater potential for nutrients to be leached out into the drains, thus decreasing nutrient concentrations. Throughflow of rainwater is also likely to be quicker, resulting in a more flashy regime which would prevent nutrients from rainfall being adsorbed into the peat. Changes in plant community could also occur on drained peatlands (Coulson et al. 1990, Faubert 2004), which would affect the substrate available for mineralisation and consequently the nutrients which are released into the peat, as well as the nutrient demands of the plants growing. Drained peatlands are expected to be drier, however the extent will depend on drainage spacing (Hudson & Roberts 1982). Drained peatlands are also expected to be more acidic (Laiho et al. 2004b), therefore resulting in decreased rates of substrate mineralisation and hence release of nutrients into the soil.

4.2.2.4 Afforestation

Afforestation is expected to result in a decrease in peat nutrient concentrations owing to the demands of the trees for nutrients as well as an expected decrease in soil pH (Byrne & Farrell 2005), which is also likely to cause in a decline in nutrients held by the peat. Furthermore, inputs of nutrients from rain are likely to be lower due to interception by the tree canopy; and those that do reach the peat, are less likely to be retained because through-flow rates are faster on drained (and hence afforested) sites. Rates of mineralisation might be reduced in afforested peatlands owing to the lower temperatures caused by shading (Silvola et al. 1996), which in turn might result in slower rates of nutrients release into the peat. Conversely, rates of microbial decomposition might be greater due to a thicker acrotelm leading to greater release of nutrients (Byrne & Farrell 2005). Pyatt (1993) suggested that afforestation of peat resulted in a reduced moisture content due to drainage and demand for water by trees; a view supported by Anderson et al. (2000).

❖ Land management will not influence peats at depth.

Concentrations of nutrients and metals are expected to decrease with depth, and variation between treatments will reduce as the impacts of land management lessen, as peats that were formed prior to management practices are encountered. The effect of root depth may affect the concentrations of nutrients, thereby causing changes between treatments – based on an assumption that vegetation changes will occur between treatments (see below).

pH is expected to change with depth, and could vary between treatments for the whole of the 0.5 m of the peat profile examined if the peat solution chemistry is affected by land management. The moisture content of the peats is expected to increase with depth and vary less between the sites with depth as increasingly saturated conditions are encountered. The afforested site is expected to be the driest.

- ❖ Land management will have an impact on the species of vegetation that grow on peats, and thus the nutrient content of the peats will vary depending on the species of vegetation growing on the peat.

Changes in peat chemistry as a result of changes in vegetation have been documented by Alonso and Hartley (1998), Shaver (2006) and Cuttle (1983). Conflicting evidence exists as to whether the presence of mosses results in higher (Gorham et al. 1986) or lower concentrations (Cuttle 1983) of nutrients.

- ❖ The frequency with which peats are burnt will impact on the chemical properties of the peat

Differences in the frequency of burning are expected based on the results of work carried out in Turkey by Dikici and Yilmaz (2006). The greater recovery time between burns is expected to result in the burnt (every 20 years) site to having lower nutrient concentrations.

- ❖ Combining burning and grazing will result in peats with chemical properties that differ from those that are subjected to just one treatment.

Combinations of burning and grazing are expected to have impacts on the nutrient concentrations in peats as an increase in the number of sources of nutrient inputs will have occurred. Changes in moisture content and pH are less likely to be affected.

- ❖ No differences are expected to exist between the three plots from blocks A to C that are subjected to the burning every 10 years.

Samples collected from plots which are managed in the same way are expected to have very similar properties, significant differences are not expected to exist between plots managed in the same way.

In addition, the potential existence of edge effects will be examined to determine identify whether the close proximity of the treatments to one another could result in one treatment impacting upon the another. Fences however exist between those plots that are grazed and not grazed. In addition, as the experimental plots are carefully managed, therefore differences should not exist within one treatment in terms of the nutrient concentrations in the peats.

Table 4.1 provides an overview of the chemical properties to be examined in this chapter along with a rationale for the choice of the different properties.

Table 4.1 Chemical Parameters to be Analysed to Investigate the Effects of Land Management on the Moisture Content and the Chemical Properties of Peat.

Property	Rationale for Study
Nitrogen	Nitrogen is needed for plant growth, and will (in part) determine which plants are able to grow at a site.
Phosphorus	Phosphorus is needed for plant growth, and will (in part) determine which plants are able to grow at a site.
Potassium	Potassium is needed for plant growth, and will (in part) determine which plants are able to grow at a site.
Iron	Iron has been linked with carbon dioxide production and is also required by methanogens. Methane can be oxidised to carbon dioxide and released into the atmosphere (Basiliko & Yavitt 2001).
Selenium	Selenium is toxic to methanogens and may ultimately result in decreased carbon dioxide loses from peat (depending on how much methane would be produced and oxidised to carbon dioxide.)
Cobalt	Cobalt has been linked with carbon dioxide production and is also required by methanogens for growth. Methane can be oxidised to carbon dioxide and released into the atmosphere (Basiliko & Yavitt 2001)
Molybdenum	The presence of molybdenum can result in carbon dioxide being reduced to methane if the correct environmental conditions (anaerobic) are present.
Nickel	Nickel has been linked with carbon dioxide production and is also required by methanogens for growth. Methane can be oxidised to carbon dioxide and released into the atmosphere (Basiliko & Yavitt 2001).
Moisture content	The moisture content can control the nature and extent of microbial decomposition and plant growth in a peatland.
pH	The pH not only affects microbial decomposition but also the composition of the vegetation growing on a bog.

4.3 Methodology

4.3.1 Fieldwork

Fieldwork was carried out at the Moor House National Nature Reserve (NNR) as detailed in Chapter 3. A total of 144 cores measuring 0.5 m in length were collected and divided into five 10 cm sections on site and stored in clean, sealed, labelled plastic bags. Samples were transported back to the laboratory and stored at 4°C as soon as possible. Samples remained in cold storage until required for analysis. The method of sample preparation used for each determinand is detailed below. The surface vegetation present at each core location was recorded. Details of which site each core was collected from is provided in

Table 4.2 Treatments From Which Samples Were Collected

Treatment	**Sample Numbers**
Burnt and Grazed (every 10 years) (BG10)	1-15
Unmanaged (U)	16-30
Burnt (every 20 years) (B20)	31-45
Burnt (every 10 years) (B10) – Block A	45-55 (plus C1-C5 from pilot study)
Burnt and Grazed (every 20 years) (BG20)	56-65
Grazed (G)	66-80
Burnt (every10 years, block B) (B10B)	81-95
Burnt (every10 years, block C) (B10C)	96-110
Drained (D)	111-125
Afforested (F)	126-144

4.3.2 Laboratory Work

4.3.2.1 Soil Moisture Content

The moisture content of the samples was determined by weighing approximately 10 g (±0.1g) of wet soil into a pre-weighed pre-labelled crucible which was placed in an oven at 105°C for 24 hours. After drying samples were placed in a desiccator to cool, and were then re-weighed. The gravimetric moisture content of the sample was calculated using the following equation:

$$\theta = \frac{\text{water lost (g)}}{\text{mass of oven dry soil (g)}}$$

(Rowell 1994)

4.3.2.2 Total Nitrogen

Total carbon and nitrogen analysis was carried out using a Eurovector EA3000 Elemental Analyser. Tin cups measuring approximately 0.5 mm by 0.5 mm were taken, and 5 mg of oven dried soil was added with 1.5 mg of vanadium pentoxide. Vanadium pentoxide was added as a catalyst to ensure that complete combustion occurred once the samples were placed in the analyser[1]. Energy Peat (*Sphagnum*) Reference Material NJV 942 was used as a standard reference material. Once prepared, the samples were analysed on the Elemental Analyser, which was calibrated using a range of sulphanilic acid standards. Results were expressed as a percentage of the weight of the sample.

4.3.2.3 Soil pH

pH was measured by placing 15 g of field moist peat in a plastic beaker to which 30 ml of deionised water was added. Samples were stirred and left to stand to equilibrate with atmospheric carbon dioxide for 30 minutes before being analysed with a pH meter (Mettler-Toledo).

4.3.2.4 Analysis of Metals and Nutrients

Sample Preparation Trial

Traditionally, analysis of metals and nutrients in soils has been carried out on air dried soils as recommended by good practice guidance e.g. ISO 11464 (2006). Peat, however, has very different characteristics from other soil types. For example, air drying peats is not only time consuming (taking up to several weeks) but may also result in the chemical properties of the peats being altered. In addition, the hydrophobicity of peats can result in a material that is difficult to mix into an

[1] A trial carried out in July 2008 indicated that without a catalyst, incomplete combustion occurred on samples of peat.

extracting solution, thereby resulting in only partial extraction of the metal of interest. Conflicting evidence exists as to whether peats should be prepared differently from conventional soils or not. For example, Wieder et al. (1996) noted that previous works made use of wet peat samples to carry out biogeochemical analysis in order to prevent alteration to the chemical composition of the samples. They recommend that samples are freeze dried to prevent oxidation of the elements of interest, and state that analysis of wet soils is problematic in light of the likelihood of different samples having different moisture contents. This notion raises questions as to whether analysis of wet samples provides an accurate reflection of the elemental composition of the peat or the peat pore water solution. Alternatively, the presence of water in the sample could result in a dilution or even an increase in concentrations of elements present in the sample. Canfield et al. (1986) carried out a trial on estuarine peats and sediments and suggested that no differences exist between air-dried, freeze-dried and wet samples. In contrast, Amaral et al. (1989) found that elements were lost as a result of freeze-drying. As a lack of clarity exists as to which method of sample preparation is best for peats, a trial was conducted to identify which method resulted in the highest concentrations of elements.

Samples of both peat and non-peat soils (clays and sands provided courtesy of Fugro Engineering Services) were analysed in the metals trial. Traditionally non-peat samples are air dried prior to extraction; and their inclusion in the trial aimed to identify whether significant differences occur between the four methods of soil preparation used in the trial. Samples of non-peat soils from natural strata were provided courtesy of Fugro Engineering Services Ltd from an undisclosed site in Hampshire. A total of five samples were analysed – two peat and three non-peat, and five repetitions were carried out on each sample. Each sample was subjected to the following methods of preparation:

- Air-drying
- Freeze-drying
- Oven-drying (70°C)
- No preparation – wet soil/peat was used.

Following drying, the samples (with the exception of the wet samples) were ground in a small mill and passed through a 2 mm sieve. Samples were analysed in triplicate.

Samples were extracted with 5 ml of 2M nitric acid following the method detailed by Whitton et al. (1991). The acid and soil mixture was placed in a 15 ml PTFE tube in a water bath at 100°C for 45 minutes. Following extraction the samples were filtered through Waterman No. 44 filter papers into 25 ml volumetric flasks, and deionised water was used to ensure all extractant left the PTFE tube and was rinsed through into the volumetric flask. The solutions were made up to 25 ml with deionised water. Within each batch of samples, three blank samples were analysed to ensure that contamination of the samples had not occurred. The samples were analysed by an Inductively Coupled Plasma - Optical Emission Spectrophotometer (Perkin Elmer 5300DV ICP-OES) for cobal,t iron, selenium and molybdenum as these are known to be important in terms of carbon cycling. In addition, lead, cadmium, copper, chromium and arsenic were included to provide a broad range of elements commonly found in abundance in UK soils, which would provide additional data on which a decision could be made.

Concentrations of metals and nutrients were calculated as follows:

$$\text{Sample} * \left(\frac{25}{\text{concentration}}\right)$$

Whitton et al. (1991)

Minimum detection limit was determined by calculated as follows:

(Standard Deviation of Blank Samples*3) + (Average of blank samples)

Results of Sample Preparation Trial

The results for each sample were used to determine which method of sample preparation achieved the greatest recovery for each element, defined as the highest concentration in the comparison of the sample preparation types. The results are presented in Table 4.3

Table 4.3 Mean Values (mg/Kg) for Each Sample Preparation Method and Element Analysed in the trial

Sample	Method of Preparation	Arsenic	Cadmium	Cobalt	Chromium	Copper	Iron	Molybdenum	Nickel	Lead
Detection Limit		*0.008*	*0.001*	*0.001*	*0.001*	*0.007*	*0.117*	*0.002*	*0.001*	*0.004*
A	Air dry	3.21	0.28	2.82	2.77	12.35	5554.71	0.04	10.27	507.87
A	Freeze dry	3.37	0.28	1.60	1.60	12.52	5806.18	0.03	7.69	180.61
A	Oven dry	16.34	1.14	4.08	7.11	12.37	18585.82	BD	14.69	714.34
A	wet	3.24	0.68	1.16	3.11	7.73	3139.32	0.55	2.82	175.97
B	Air dry	2.82	0.2	0.36	2.10	12.17	1328.77	0.69	3.85	146.00
B	Freeze dry	0.86	BD	BD	0.90	8.62	414.4	0.17	2.11	23.66
B	Oven dry	2.35	BD	BD	1.45	6.98	956.81	0.36	2.10	40.55
B	Wet	1.58	BD	BD	1.82	5.85	187.90	0.36	0.93	13.99
C	Air dry	5.55	BD	10.41	7.46	15.94	21233.89	BD	14.82	12.27
C	Freeze dry	6.91	1.39	11.07	6.10	13.82	22423.27	BD	15.10	13.76
C	Oven dry	5.61	BD	11.00	5.07	13.35	20911.17	BD	15.44	13.10
C	Wet	6.51	BD	9.71	4.66	10.68	18876.17	BD	12.09	11.25
D	Air dry	4.77	0.01	9.93	7.36	15.11	14585.13	BD	16.39	11.33
D	Freeze dry	5.56	0.29	11.58	7.15	14.98	18626.14	BD	15.73	14.14
D	Oven dry	4.35	0.78	10.82	7.57	13.18	16405.19	BD	14.19	13.58
D	Wet	5.47	BD	8.53	4.90	12.83	15023.23	BD	12.40	11.06
E	Air dry	2.29	BD	2.39	2.69	10.25	5679.52	BD	3.54	3.52
E	Freeze dry	4.22	BD	4.57	3.40	7.99	10133.23	BD	6.36	5.87
E	Oven dry	3.94	BD	4.69	3.37	9.33	8687.77	BD	7.68	8.45
E	Wet	2.83	BD	2.30	1.78	4.94	7545.24	BD	3.16	4.05

Cells shaded in grey highlight the highest value found for each sample for each element analysed. A and B were peats, C-E were non-peat soils BD= below the detection limit. N=5.

The results demonstrated that some form of sample treatment prior to extraction of metals and nutrients with acid is preferable, as the lowest values tended to be found in samples where no preparation had taken place. The lowest concentrations recorded in the untreated soils could have been caused by one or more of the following factors:

- Wet samples could not be sieved; which would firstly this would result in particles > 2 mm being included in the analysis. Soil particles of > 2 mm are considered to be chemically inert, thus reducing the concentrations of metals that would be recorded in each sample. Secondly, it is more difficult to ensure that samples are thoroughly mixed with the acid, making it less likely that all of the available nutrients will have been extracted;

- The presence of water in the sample might have diluted the acid being used to extract the metals and nutrients resulting in less effective extraction;

- The presence of water in the sample would have resulted in extraction of metals and nutrients from both the soil and water phase;

- The weight of the water would have been included in the sample weight, but if there were no nutrients or metals in the water phase then the results would be lower than if the water had been removed prior to extraction. Given the low concentrations identified in the wet samples, it is unlikely that higher concentrations of nutrients and metals were present in the soil solution.

The preparation of samples prior to extraction provided increased recovery of metals. In 15 cases, air drying samples resulted in the highest concentrations for a given element, whereas in 14 cases freeze drying resulted in the highest rates of metal recovery and in a further 14 cases oven drying resulted in the highest concentrations being recovered. Figure 4-1 illustrates concentrations of nickel in peat and non-peat soils. The result demonstrates that oven drying resulted in the highest concentrations of nickel being extracted. In all cases, the lowest concentrations were identified in the wet samples.

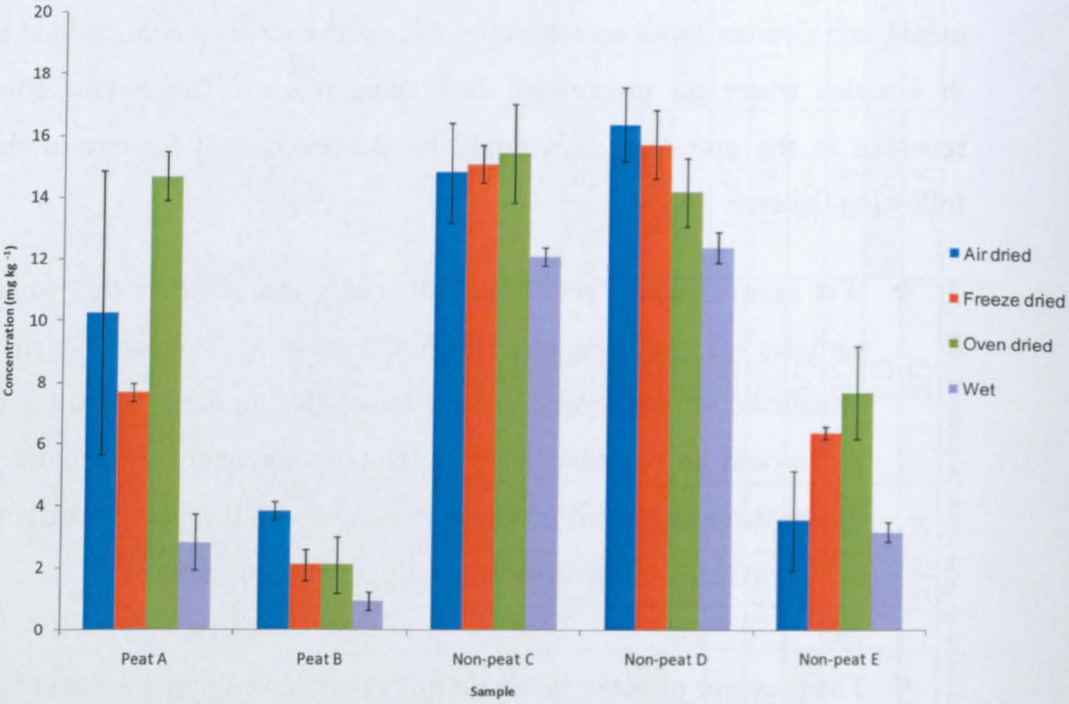


Figure 4-1 Concentrations of Nickel in Peat and Non-Peat Soils Subjected to Different Methods of Preparation Prior to Acid Extraction

Two way ANOVA was carried out to identify whether differences existed between the preparation methods, and to ascertain whether soil type had a significant effect on the outcome of the analysis. The results demonstrated that there were not significant differences between the methods of preparation ($p=0.456$) and that interaction with soil type was not significant ($p=0.690$)

In all instances, despite the lack of a significant difference between the methods of preparation, the wet samples had the lowest values. The results showed little difference between freeze-drying, air drying and oven drying of samples. Owing to the length of time required to air dry samples (in some cases up to several weeks) and the costs involved in freeze-drying samples, oven drying was chosen as the preferred option for preparing samples of peat soils for analysis of nutrients and metals. Samples from all sites investigated at Moor House were analysed for nutrients and metals using the procedure outlined above by Whitton et al. (1991) on oven-dried peats. Two blank samples and one certified reference sample was included in every batch of 20 samples.

4.3.3 Statistical Analysis

To determine if the differences between the treatments were significant, data were analysed to determine if a normal distribution were present to enable Analysis of Variance (ANOVA) to be carried out. A normal distribution was not found in the majority of data sets. The following methods of transforming data were utilised in an attempt to normalise the data: logarithmic; square; square root; and reciprocal. The results of the Anderson-Darling normality test demonstrated that none of the transformations produced a normal distribution. The absence of a normal distribution suggest that either non-parametric tests should be utilised, or, ANOVA could be carried out, but caution must be exercised when interpreting the results, given the data do not conform to the assumptions underlying ANOVA.

The Kruskall-Wallis test requires data to fulfil the following requirements: to be selected at random; with independent populations, and populations with the same variance and distributions (Rumsey 2007). As the data do not meet the requirements of either test, ANOVA was performed as the test is more robust than non-parametric tests and is relatively insensitive to non-normal data. Tukey's post-hoc test was used to identify where significant differences ($p<0.05$) existed between treatments

4.4 Results

4.4.1 Spatial variation in the chemical properties of peats within treatments

The results of the analysis of peats for chemical elements, moisture content and pH were plotted spatially for each treatment to check for edge effects. Samples were also divided into edge and non-edge locations. Samples within 2 m of the plot boundary were considered to be edge locations and those further away from the boundary non-edge locations. No visual trends were identified at any of the sites within the surface peats, with the exception of moisture content in the afforested site which decreased with distance from the perimeter of the forest and was lower on ridged areas of the forest compared to furrowed areas.

There were no statistically significant differences in the results for any of the parameters analysed along the edges of the plots compared to those further than 2 m from the edges ($p>0.05$). The results of this analysis suggest the experimental set-up was unlikely to be influenced by edge effects, thereby reducing any bias caused by the proximity of the sites to one another.

4.4.2 Moisture Content

Differences in the moisture content of the peat samples analysed are presented on Figure 4-2. Maximum values tended to be found between 20 and 30 cm beneath the surface with the exception of the unmanaged site, the burnt (every 10 years) and the drained sites. The maximum values for these sites were found 30 to 40 cm beneath the surface.

The afforested site was the driest, with values ranging between 360 % and 902 % moisture by dry mass. In the surface soils, the burnt (every 20 years) site had the highest values (average 940 %), beneath the surface layer, maximum values were identified in the drained site (average values ranged between 963 % and 1,092 %).

Analysis of Variance (ANOVA) demonstrated significant differences between the moisture contents of the sites ($p<0.0001$, Table 4.4). The afforested site was significantly drier than all other treatments, whilst the drained site was significantly wetter than all treatments except the burnt and grazed (every 10 years) site and the burnt (every 10 years, C) site.

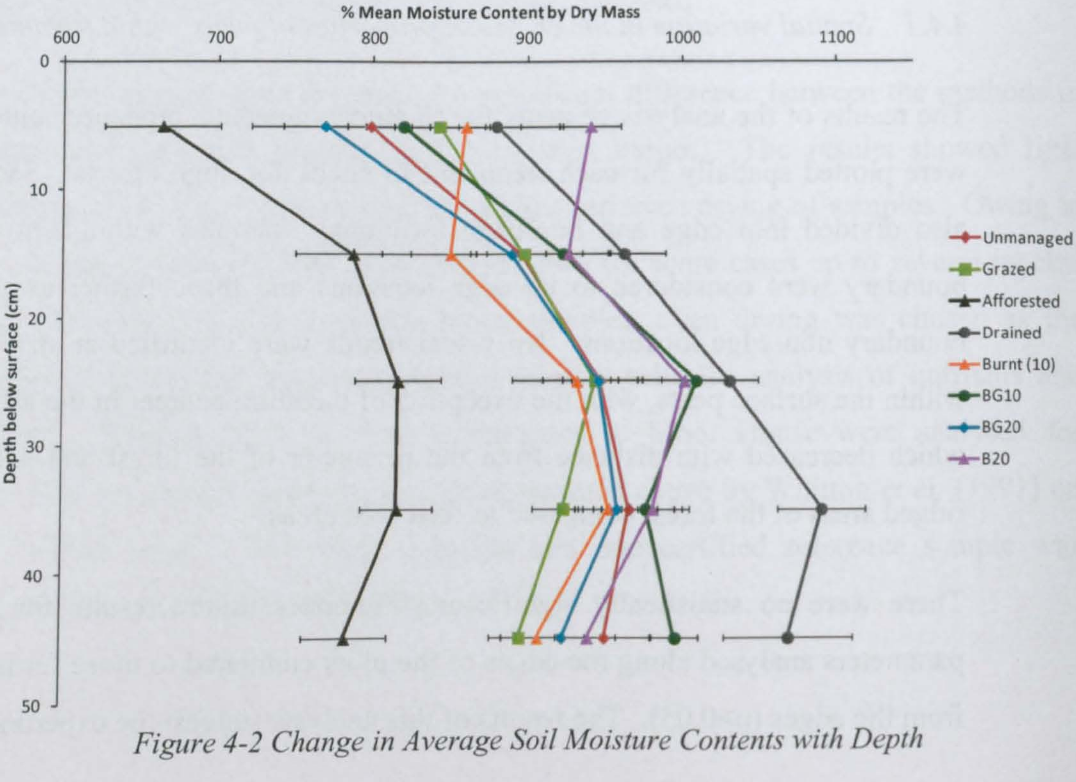


Figure 4-2 Change in Average Soil Moisture Contents with Depth

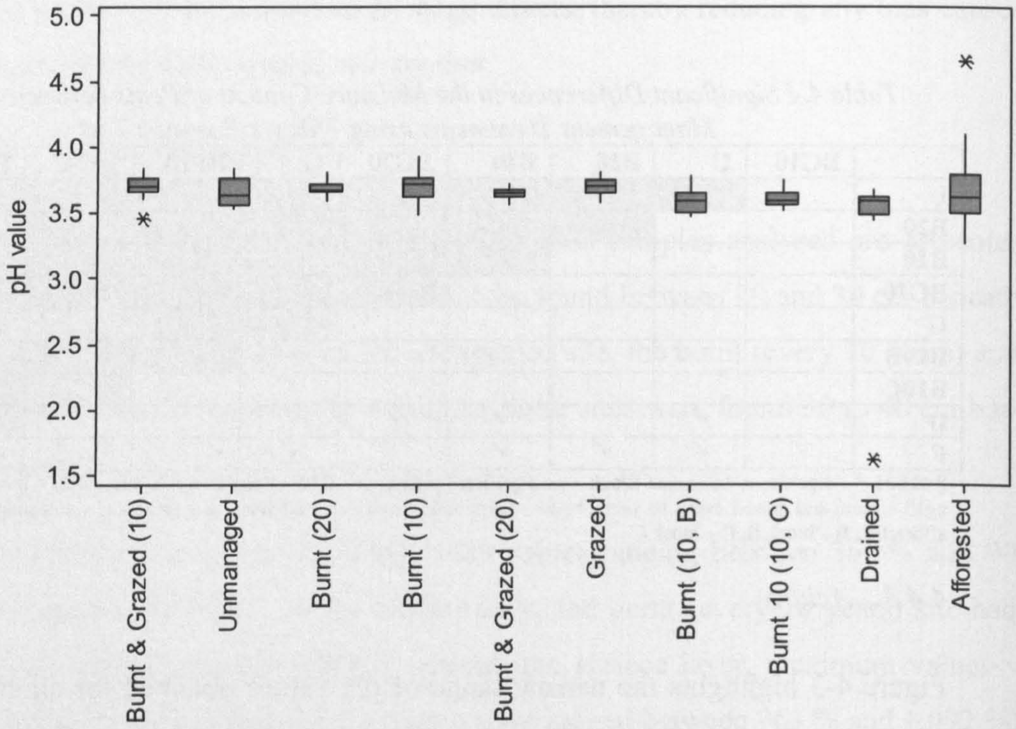
Table 4.4 Significant Differences in the Moisture Content of Peats between Different Management Treatments using Tukey's Post-hoc Test

	BG10	U	B20	B10	BG20	G	B10B	B10C	D
U									
B20									
B10									
BG20									
G									
B10B									
B10C									
D		✓		✓	✓	✓	✓		
F	✓	✓	✓	✓	✓	✓	✓	✓	✓

P<0.001 ✓ - significant difference Blank – no significant difference. B10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, g – grazed, u – unmanaged, d- drained, f– afforested; B – block B; C – block C

4.4.3 *Acidity*

Figure 4-3 highlights the narrow range of pH values obtained for all sites with the exception of two outliers in the drained and afforested sites. All sites were found to have low pH values, becoming less acidic with depth, with the exception of the afforested site. The afforested site was the least acidic in the surface peats (3.6). The unmanaged site was the least acidic site between 10 and 50 cm beneath the surface (3.8 to 4.4). ANOVA demonstrated significant differences between treatments, as illustrated in Table 4.5. The drained, afforested and burnt sites on Blocks B and C were found to be significantly more acidic than all other treatments.



The extent of the box represents the interquartile range, the horizontal line within the box indicates the median of the data. The upper and lower 25% of the data (excluding outliers) are shown by the lines extending out of the box. Outliers are represented with an *.

Figure 4-3 Box and Whisker Plot of pH Values for the Surface Layer Peats

Table 4.5 Significant Differences in the pH Value of Peats between Different Management Treatments According to Tukey's Post-hoc Test

	BG10	U	B20	B10	BG20	G	B10B	B10C	D
U									
B20									
B10									
BG20									
G									
B10B	✓	✓	✓	✓	✓	✓			
B10C	✓	✓	✓	✓	✓	✓			
D	✓	✓	✓	✓	✓	✓			
F	✓	✓	✓	✓		✓			

P<0.001 ✓ - significant difference Blank – no significant difference. B10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, g – grazed, u – unmanaged, d- drained, w – afforested; B – block B; C – block C

4.4.4 Nitrogen

No clear trends in the nitrogen values were identified with depth down the peat profile within each treatment (Figure 4-4). The greatest concentrations of nitrogen (average 2.69 %) were identified at the burnt (every 20 years) site whilst the lowest (average 1.26 %) were found in the afforested site in the surface layer. The burnt site (every 20 years) was found to have the highest concentrations of nitrogen between 0 and 40 cm below the surface of the peat. Between 40 and 50 cm beneath the surface the burnt (every 10 years, C) site was found to have the highest concentrations of nitrogen (2.19 %) whilst the lowest average concentration (1.22 %) was found at the burnt (every 10 years, B) site.

Significant differences in the nitrogen content between treatments were identified ($p<0.001$) as illustrated in Figure 4-4. The burnt (every 20 years) site had a significantly higher nitrogen content than the grazed, the drained, the unmanaged, the afforested, the burnt (every 10 years) site and the sites where burning and grazing were combined. The unmanaged site had a significantly lower nitrogen content than the burnt (every 20 years) site, the burnt site on block C and the burnt and grazed (every 20 years) sites.

Table 4.6 Significant Differences in the Nitrogen Content of Peats between Different Management Treatments According to Tukey's Post-hoc Test

	BG10	U	B20	B10	BG20	G	B10B	B10C	D
U									
B20	✓	✓							
B10			✓						
BG20		✓							
G			✓		✓				
B10B			✓						
B10C		✓				✓			
D			✓					✓	
F			✓		✓			✓	

P<0.001 ✓ - significant difference Blank – no significant difference. B10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, g – grazed, u – unmanaged, d- drained, f –afforested; B – block B; C – block C

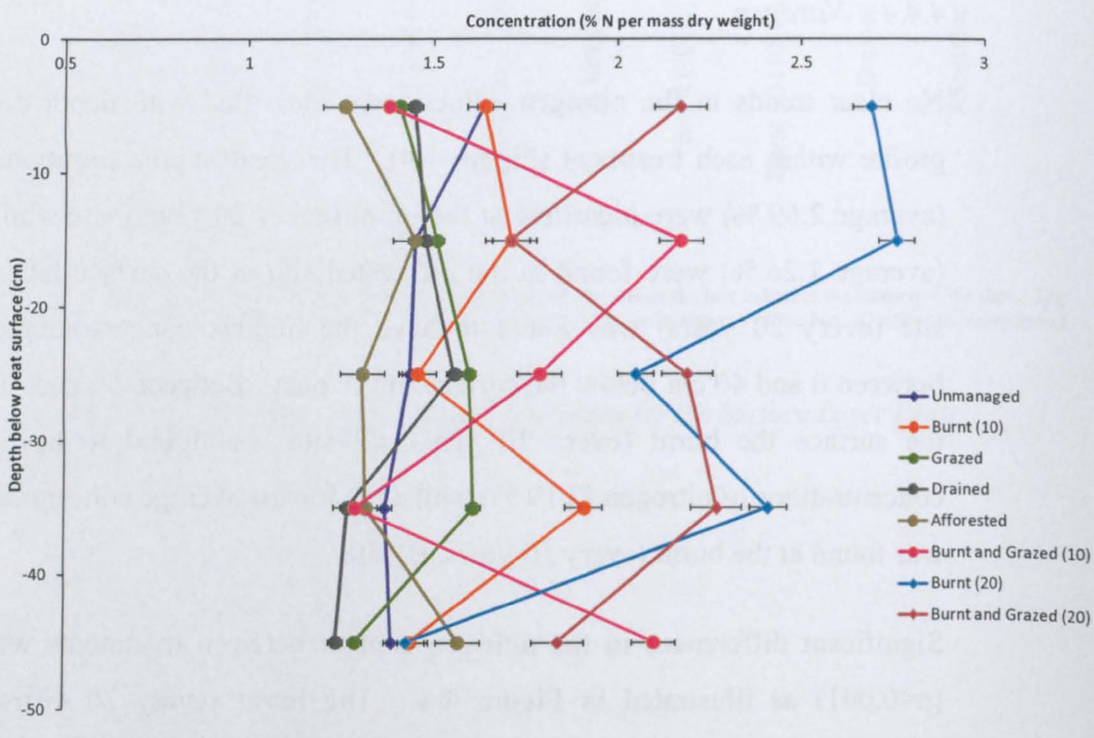


Figure 4-4 Variations in Mean Nitrogen Concentrations with Depth

BG10 n=73 U n=75 B20 n=75 B10 n=72 BG20 n=74 G n=72 D n=73 F n=95

4.4.5 Phosphorus

Contrasts between surface values and the base of the peat profile are illustrated in Figure 4-5, demonstrating that higher concentrations were found in the surface peats (0-20 cm). In the top 30 cm, the highest average concentration was found in the burnt (every 20 years) site whilst the lowest average concentrations were found in the afforested site. The burnt site (every 10 years) had the highest concentrations between 30 and 50 cm beneath the surface (185.5 – 166.5 mg kg^{-1}). The lowest concentrations between these depths were found in the drained site (140.6 mg kg^{-1}) and the unmanaged site (117.9 mg kg^{-1}). No site was identified using ANOVA as being significantly different to the unmanaged site. The burnt site (20 years) site peats had a significantly higher phosphorus content than the afforested site.

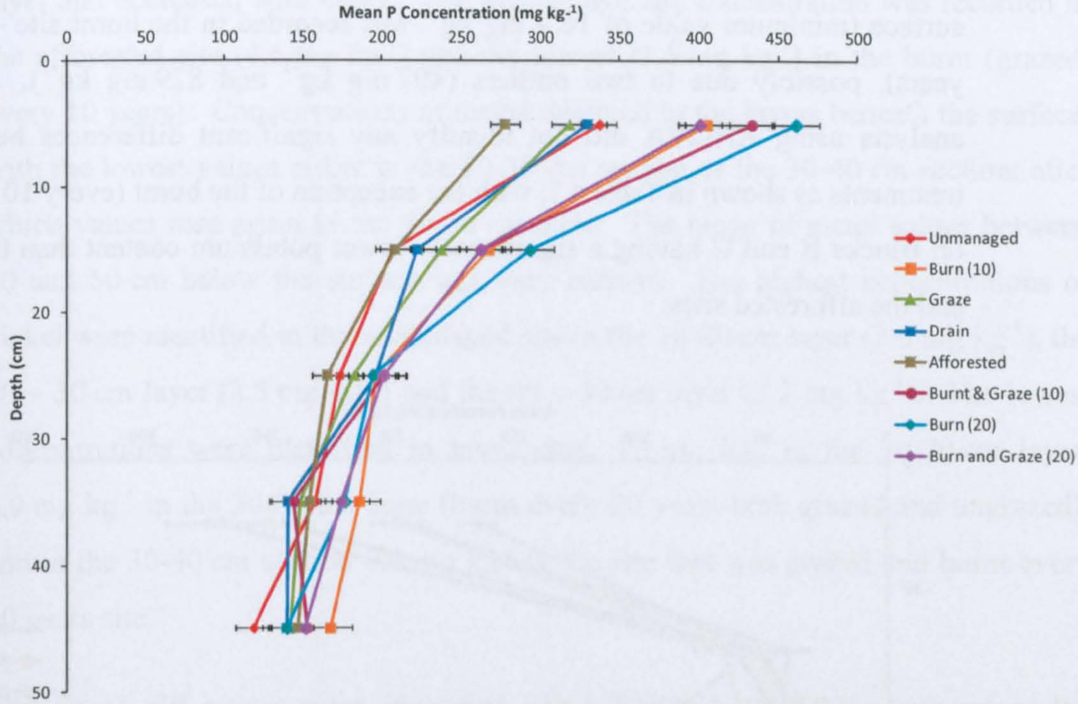


Figure 4-5 Phosphorus Concentrations with Depth

BG10 n=75 U n=74 B20 n=74 B10 n=68 BG20 n=72 G n=71 D n=69 F n=94

Table 4.7 Significant Differences in the Phosphorus Content of Peats between Different Management Treatments According to Tukey's Post-hoc Test

	BG10	U	B20	B10	BG20	G	B10B	B10C	D
U									
B20									
B10									
BG20									
G									
B10B									
B10C									
D									
F			✓						

✓ - Significant difference Blank – no significant difference. B10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, g – grazed, u – unmanaged, d- drained, f – afforested; B – block B; C – block C

4.4.6 Potassium

Concentrations of potassium were at their highest in the surface layer (0-10 cm) with the highest values in the drained (274.3 mg kg^{-1}) site and the lowest (86.7 mg kg^{-1}) in the burnt (every 10 years, C) site. Large ranges were identified in this layer with the greatest (113.6 – 761.9 mg kg^{-1}) recorded for the afforested site. The values for potassium decreased in the layers below the 0-10 cm zone, with the lowest values recorded between 20 and 40 cm beneath the surface, before increasing in the 40-50 cm layer of the profile, as shown in Figure 4-6. Average concentrations of potassium greatly increased in the drained site between 20 and 30 cm beneath the surface (minimum value of 16.4 mg kg^{-1} was recorded in the burnt site – every 20 years), possibly due to two outliers (407 mg kg^{-1} and 829 mg kg^{-1}). Statistical analysis using ANOVA did not identify any significant differences between the treatments as shown in Table 4.7; with the exception of the burnt (every 10 years) site on Blocks B and C having a significantly lower potassium content than the drained and the afforested sites.

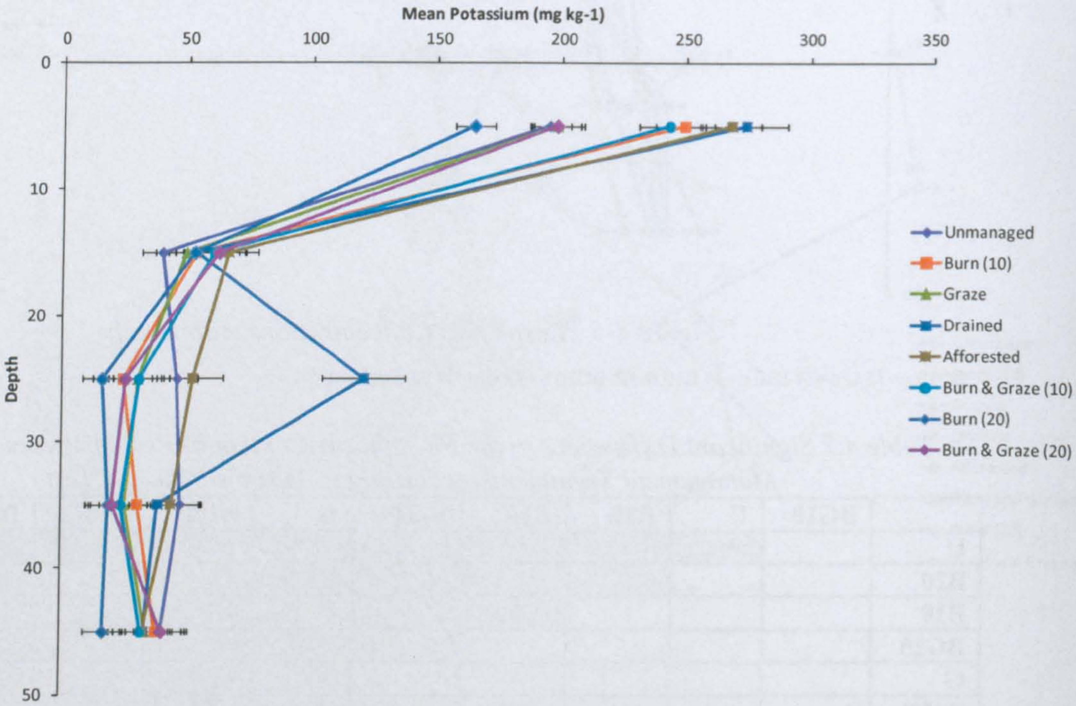


Figure 4-6 Potassium Concentrations with Depth

BG10 n=75 U n=72 B20 n=74 B10 n=66 BG20 n=71 G n=67 D n=68 F n=92

Table 4.8 Significant Differences in the Potassium Content of Peats between Different Management Treatments According to Tukey's Post-hoc Test

	BG10	U	B20	B10	BG20	G	B10B	B10C	D
U									
B20									
B10									
BG20									
G									
B10B									
B10C									
D							✓	✓	
F							✓	✓	

P=0.001 ✓ - significant difference Blank – no significant difference. B10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, g – grazed, u – unmanaged, d- drained, f – afforested; B – block B; C – block C

4.4.7 Nickel

Figure 4-7 demonstrates that concentrations of nickel were highest in the surface layer and decreased with depth. The highest average concentration was recorded in the afforested site (4.6 mg kg^{-1}) and the lowest (1.6 mg kg^{-1}) in the burnt (grazed, every 10 years). Concentrations of nickel declined in the layers beneath the surface, with the lowest values either in the 20-30 cm section or the 30-40 cm section; after which values rose again in the 40-50 cm zone. The range of mean values between 10 and 50 cm below the surface was very narrow. The highest concentrations of nickel were identified in the unmanaged site in the 10-20 cm layer (2.5 mg kg^{-1}), the 20 – 30 cm layer (2.5 mg kg^{-1}) and the 40 – 50 cm layer (3.2 mg kg^{-1}). The lowest concentrations were identified in burnt sites, 1.1 mg kg^{-1} in the 10-20 cm layer, 0.9 mg kg^{-1} in the 20-30 cm zone (burnt every 20 years both grazed and ungrazed); and in the 30-40 cm section 0.9 mg kg^{-1} in the site that was grazed and burnt every 20 years site.

Significant differences were identified using ANOVA ($p<0.001$) between nickel concentrations in the unmanaged site and all burnt treatments with the exception of the burnt and grazed (every 10 years) site. Significantly higher concentrations of nickel were identified in the drained and the afforested sites and sites burnt on a 20 year rotation, and Blocks B and C.

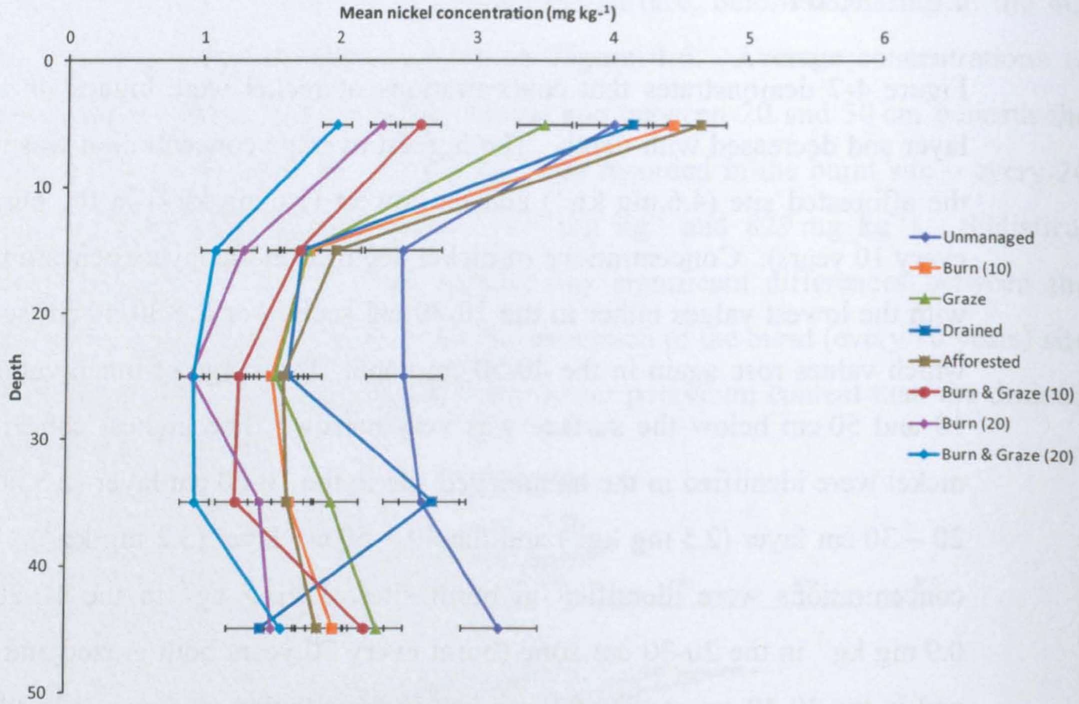


Figure 4-7 Changes in Nickel Concentrations with Depth

BG10 n=75 U n=74 B20 n=74 B10 n=68 BG20 n=71 G n=71 D n=67 F n=92

Table 4.9 Significant Differences in the Nickel Content of Peats between Different Management Treatments According to Tukey's Post-hoc Test

	BG10	U	B20	B10	BG20	G	B10B	B10C	D
U	✓								
B20		✓							
B10									
BG20		✓		✓					
G		✓			✓				
B10B		✓		✓		✓			
B10C		✓							
D			✓		✓		✓		
F			✓		✓		✓	✓	

$p<0.001$ ✓ - significant difference Blank – no significant difference. B10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, g – grazed, u – unmanaged, d- drained, f – afforested; B – block B; C – block C

4.4.8 Cobalt

Little variation was apparent in the concentrations of cobalt for each site or with depth. The range of values within each treatment for each depth was very small. Maximum mean concentrations tended to be found in the unmanaged site (0.71 mg kg^{-1} at 0-10cm; 1.22 mg kg^{-1} at 10-20 cm; 1.34 mg kg^{-1} at 30-40 cm and 1.25 mg kg^{-1} at 40-50 cm). The unmanaged site had the largest range of values detected (0.37 – 2.42 mg kg^{-1} in surface peats). No clear trend in values was observed within the treatments over the different depths examined. Unlike other metals and nutrients, values did not decrease with depth, however maximum values were not detected in the 40-50 cm zone. The unmanaged site was found to have significantly higher concentrations of cobalt than all sites other than the burnt (every 10 years) site. Table 4.10 illustrates the sites between which significant differences were identified. Where differences were significant, concentrations of cobalt were higher in the grazed and burnt (every 10 years) sites compared to the sites.

Table 4.10 Significant Differences in the Cobalt Content of Peats between Different Management Treatments According to Tukey's Test

	BG10	U	B20	B10	BG20	G	B10B	B10C	D
U	✓								
B20		✓							
B10	✓		✓						
BG20		✓		✓					
G			✓		✓				
B10B		✓		✓		✓			
B10C		✓		✓	✓	✓			
D		✓		✓		✓			
F		✓		✓		✓		✓	

$p<0.001$ ✓ - significant difference Blank – no significant difference. B10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, g – grazed, u – unmanaged, d- drained, f – afforested; B – block B; C – block C

4.4.9 Iron

Concentrations of iron were highest in the surface layer and declined with depth until the 30-50 cm layer where values increased again, as illustrated in Figure 4-8. The maximum average concentration in the surface layer was 1,857 mg kg^{-1} in the burnt (every 10 years) site. The lowest average concentration was 871 mg kg^{-1} in the burnt (every 10 years C) site. The highest concentrations for each depth examined were found in the unmanaged site from 10-20 cm downwards to 50 cm. The greatest ranges of values were found in the unmanaged site's peat at these depths. Minimum values were consistently found in the burnt (every 10 years, C) site for depths between 20 and 50 cm (914, 1,080 and 1,205 mg kg^{-1} respectively). Following the sharp drop in values for iron in the 10 to 20 cm layer, values steadily rose in all treatments and peaked in the 40 to 50 cm zone, with a maximum value of 6,303 mg kg^{-1} recorded in the unmanaged site and a minimum value of 729 mg kg^{-1} recorded in the afforested site.

Significant differences in the iron content of the peat samples were identified between the different land management practices as illustrated in Table 4.11. All treatments with the exception of the burnt (every 10 years) site and the grazed site had significantly lower iron concentrations than the unmanaged site.

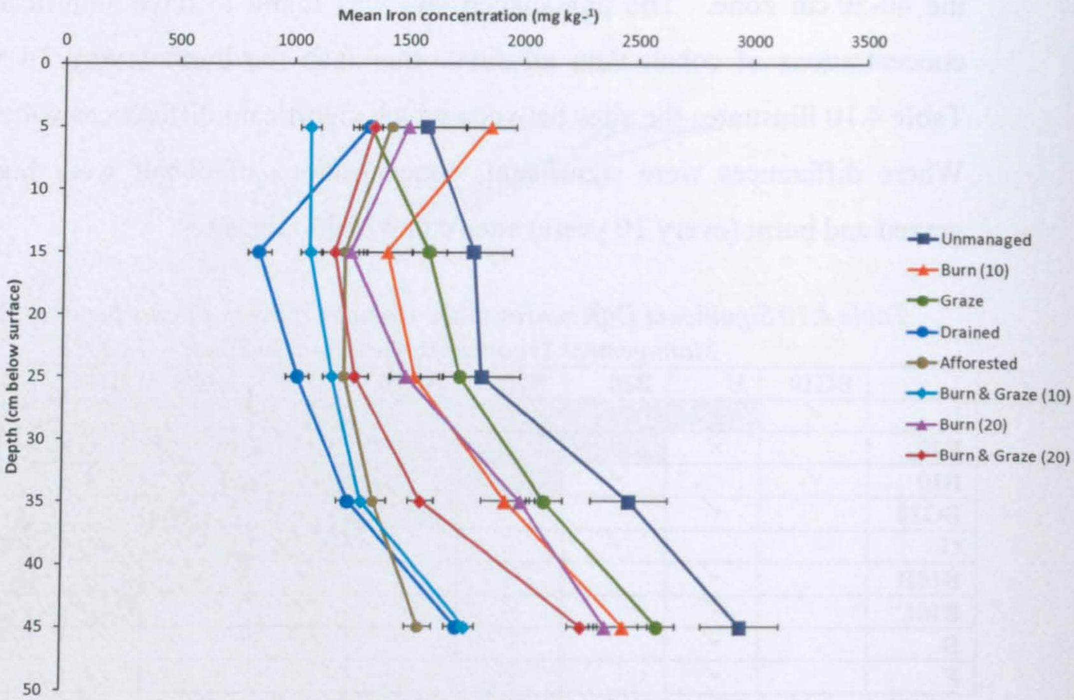


Figure 4-8 Iron Concentrations with Depth

BG10 n=75 U n=73 B20 n=74 B10 n=68 BG20 n=72 G n=71 D n=69 F n=94

Table 4.11 Significant Differences in the Iron Content of Peats between Different Management Treatments According to Tukey's Post-hoc Test

	BG10	U	B20	B10	BG20	G	B10B	B10C	D
U	✓								
B20	✓	✓							
B10	✓								
BG20		✓							
G	✓				✓				
B10B		✓	✓	✓		✓			
B10C		✓	✓	✓	✓	✓			
D		✓	✓	✓		✓			
F		✓	✓	✓		✓		✓	

$p<0.001$ ✓ - significant difference Blank – no significant difference. B10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, g – grazed, u – unmanaged, d- drained, f – afforested; B – block B; C – block C

4.4.10 Molybdenum

All values for molybdenum were low regardless of depth or treatment, with the majority of values below 2.0 mg kg^{-1}. Maximum values tended to be found in the surface peats with the exception of the grazed and the drained sites where all samples had concentrations below the minimum detection limit. In general, concentrations decreased with depth to the 30 to 40 cm layer, and rose slightly in the 40 to 50 cm zone. Exceptions to this trend were identified in the burnt (every 20 years and every 10 years replicates B and C) sites. A peak value was identified in the drained site between 20 and 30 cm of 58,047 mg kg^{-1}, the value has been omitted from the data set on the grounds that it is five orders of magnitude greater than all other values in the data set. Checks were carried out to determine if errors had occurred in blank and reference samples but none were identified, neither were errors found in calculating the value. The value was therefore considered to be an outlier.

The highest average concentration of molybdenum was identified in the unmanaged site with a value (the only value in the data set for this depth) of 6.05 mg kg ^{1}, which appears to be an outlier based on the trends observed across the data set (see Figure 4-9). The lowest mean concentration was 0.38 mg kg^{-1} from the burnt (every 10 years, C) site. At 40 to 50 cm below the surface, the average maximum concentration was identified in the drained site (2.58 mg kg^{-1}) whilst the minimum (0.21 mg kg^{-1}) was found in the burnt (every 10 years, C) site.

Significant differences in molybdenum concentrations were identified between the treatments as shown in

Table 4.12. Due to the low number of samples within each population, qualitative analysis of whether molybdenum concentrations tallied with vegetation type could not be performed.

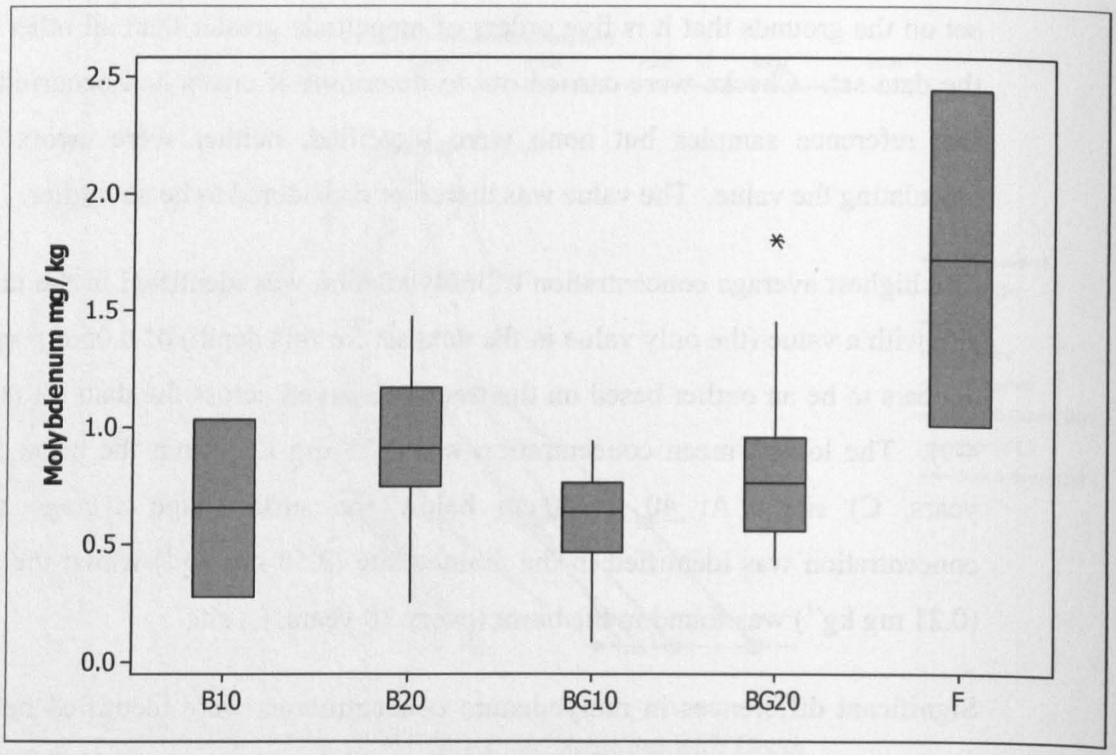


Figure 4-9 Molybdenum Concentrations in Surface Peats

The extent of the box represented the first and third quartiles, with the median represented by the line in the centre of the box. The whiskers stretch to the upper and lower limits within the first and third quartiles, values outside of this range are represented with an *.

Table 4.12 Significant Differences in the Molybdenum Content of Peats between Different Management Treatments According to Tukey's Post-hoc Test

	BG10	B20	B10	BG20	B10B	B10C
B20						
B10						
BG20						
B10B		✓		✓		
B10C	✓	✓		✓		
F						✓

P<0.001 ✓ - significant difference Blank – no significant difference. B10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, g – grazed, u – unmanaged, d- drained, f – afforested; B – block B; C – block C

4.4.11 Selenium

As shown in Figure 4-10, concentrations of selenium were highest in surface peats. Concentrations of selenium were lower between 10 and 50 cm, and in all but one case were less than 3 mg kg^{-1}. The unmanaged site had the highest average concentration in this layer (6.64 mg kg^{-1}) in the surface layer whilst the lowest value (2.71 mg kg^{-1}) was identified in the burnt (every 10 years replicate C) site.

The results of ANOVA analysis demonstrated that there were significantly higher concentrations of selenium in the unmanaged site compared to the burnt (every 20 years) site, and the grazed and burnt (every 10 years B) site. Peats from the afforested site had significantly higher concentrations of selenium than the burnt (every 20 years) site, grazed and burnt (every 10 years B and C) sites.

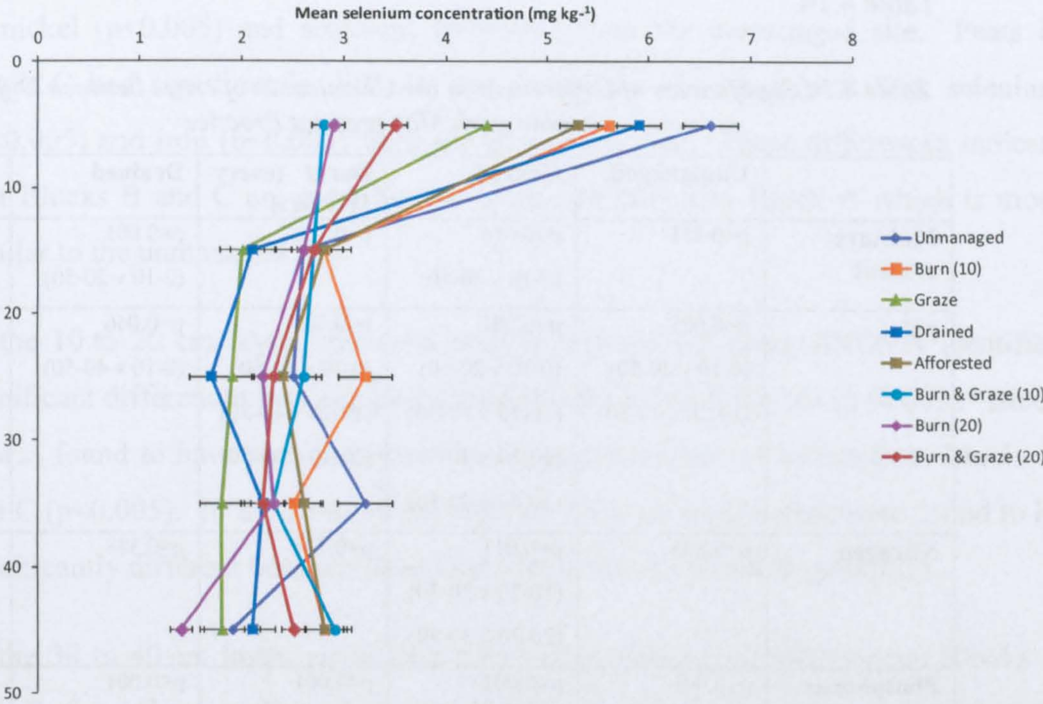


Figure 4-10 Concentrations of Selenium at Selected Depths below the Peat Surface

BG10 n=75 U n=71 B20 n=74 B10 n=66 BG20 n=72 G n=67 D n=69 F n=93

Table 4.13 Significant Differences in the Selenium Content of Peats between Different Management Treatments According to Tukey's Post-hoc Test

	BG10	U	B20	B10	BG20	G	B10B	B10C	D
U									
B20		✓							
B10			✓						
BG20									
G		✓		✓					
B10B		✓		✓					
B10C									
D									
F			✓			✓	✓	✓	

$p<0.001$ ✓ - significant difference Blank – no significant difference. B10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, g – grazed, u – unmanaged, d- drained, f – afforested, B – block B; C – block C

4.4.12 Variation in Key Chemical Properties with Depth

ANOVA was used to identify if significant differences existed in the chemical properties of the managed peats with depth. A summary of the results is presented in Table 4.14.

Table 4.14 Significance of Differences in the Chemical Properties between Different Depths within each Management Practice

	Unmanaged	**Grazed**	**Burnt (every 10 years)**	**Drained**	**Afforested**
Moisture content	p=0.451	p=0.016 (0-10 v 20-40)	p=0.101	p<0.001 (0-10 v 20-50)	p=0.005 (0-10 v 10-40)
pH	p=0.005 (0-10 v 40-50) (10-20 v 40-50)	p<0.001 (0-10 v 20-50) (10-20 v 30-50) (20-30 v 40-50) (30-40 v 40-50)	p<0.001 (0-10 v 40-50) (10-20 v 40-50) (20-30 v 40-50)	p=0.046 (0-10 v 40-50)	p=0.041 (10-20 v 40-50)
Nitrogen	p=0.805	p=0.011 (10-20 v 20-30) (20-30 v 30-40)	p=0.774	p=0.388	p=0.196
Phosphorus	p<0.001 (0-10 v 10-50) (10-20 v 40-50)	p<0.001 (0-10 v 20-50) (10-20 v 30-50)	p<0.001 (0-10 v 10-50) (10-20 v 40-50)	p<0.001 (0-10 v 10-50) (10-20 v 30-50)	p<0.001 (0-10 v 10-50) (10-20 v 40-50)
Potassium	p<0.001 (0-10 v 10-50)	p<0.001 (0-10 v 10-50)	p<0.001 (0-10 v 10-50)	p<0.001 (0-10 v 10-50)	p<0.001 (0-10 v 10-50)

p = p value from ANOVA analysis, the depths between which differences were identified based on the results of Tukey's Test are presented in brackets. <0.05 = significance value.

4.4.13 Analysis of Variation in Nutrient and Metal Concentrations between Three Plots Subjected to the same Treatment

The experimental set-up was designed to include a triplicate of one treatment – burning every 10 years for Blocks A, B and C, to ascertain whether significant differences exist between plots that have been subjected to the same treatment. The plots at Moor House are divided into four blocks, with six plots within each block as described in Chapter 3. Values for each parameter were compared between the triplicate treatments using ANOVA. Differences were not expected to be found as all three plots have been managed in the same way since the early 1950s.

For the surface soils (0 – 10 cm beneath the peat surface) plots that are burnt every 10 years from Blocks B and C at Moor House NNR were found to be significantly different from Block A in terms of their iron ($p<0.005$), nickel ($p<0.005$), selenium ($p<0.005$) and potassium ($p<0.005$) concentrations. Notably, Block A burnt peats were not found to have significant differences from the unmanaged peats for any of the properties analysed. Peats in Block B had significantly different concentrations of nickel ($p<0.005$) and selenium ($p<0.005$) from the unmanaged site. Peats in Block C had significantly different concentrations of nickel ($p<0.005$), selenium ($p<0.005$) and iron ($p<0.005$) from the unmanaged site. These differences indicate that Blocks B and C are more similar to one another than Block A which is more similar to the unmanaged site.

In the 10 to 20 cm layer, statistical analysis carried out using ANOVA identified significant differences between peats from Blocks A and B for pH ($p<0.005$). Block A was found to have significantly different concentrations of cobalt from Blocks B and C ($p<0.005$). In the 20 to 30 cm layer concentrations of cobalt were found to be significantly different between peats taken from Blocks A and B ($p=0.001$).

In the 30 to 40 cm layer, significant differences were identified between Blocks A and B for pH as well as between Blocks A and C ($p<0.005$). Significantly differences were identified between Blocks A and C for cobalt ($p<0.005$) and iron ($p<0.005$). In the 40 to 50 cm layer, significant differences were found between Blocks A and B for pH ($p<0.005$) and selenium ($p<0.002$). Blocks A and C had significantly different concentrations of iron ($p<0.005$).

4.4.14 Vegetation Analysis

This section aims to establish whether the vegetation growing at the location where a core was collected, affects the nutrient content, moisture content and/or pH of peats collected from the burnt, the grazed or the unmanaged plots. A qualitative analysis of the relationship between vegetation and concentrations of key elements was carried out; trace metals were not included in this analysis due to the low concentrations identified at each site. The results of the analysis are presented in Table 4.15

Table 4.15 Relationships between Mean Values for Chemical Properties which Drive the Carbon Cycle in Peats and Vegetation Type for each Managed Site

Site	Burnt and Grazed (10)	Unmanaged	Burnt (20)	Burnt (10)	Burnt and Grazed (20)	Grazed
			Moisture ($g\ H_2O$ per g dry mass)			
Moss	8.95 (n=6)	10.85 (n=4)	9.24 (n=7)	9.26 (n=6)	n/a	n/a
Heather	7.30 (n=1)	7.92 (n=5)	9.69 (n=6)	7.77 (n=1)	8.06 (n=7)	7.62 (n=4)
Sedge	8.01 (n=4)	7.63 (n=3)	n/a	8.94 (n=3)	7.64 (n=2)	8.55(n=3)
Heather and Sedge	6.73 (n=1)	7.19 (n=2)	9.10 (n=2)	n/a	7.83 (n=1)	8.60 (n=6)
Heather and Moss	7.51 (n=1)	9.93 (n=1)	n/a	n/a	n/a	9.34 (n=2)
Sedge and Moss	7.83 (n=2)	n/a	n/a	n/a	n/a	n/a
			pH			
Moss	3.70 (n=6)	3.65 (n=4)	3.67 (n=7)	3.70 (n=6)	n/a	n/a
Heather	3.75 (n=1)	3.62 (n=5)	3.74 (n=6)	3.48 (n=1)	3.65 (n=7)	3.69 (n=4)
Sedge	3.72 (n=4)	3.70 (n=3)	n/a	3.73 (n=3)	3.68 (n=2)	3.66 (n=3)
Heather and Sedge	3.67 (n=1)	3.54 (n=2)	3.63 (n=2)	n/a	3.65 (n=1)	3.71 (n=6)
Heather and Moss	3.60 (n=1)	3.64 (n=1)	n/a	n/a	n/a	3.72 (n=2)
Sedge and Moss	3.73 (n=2)	n/a	n/a	n/a	n/a	n/a
			Nitrogen (%)			
Moss	1.41 (n=6)	1.46 (n=4)	3.16 (n=7)	1.61 (n=6)	n/a	n/a
Heather	1.62 (n=1)	1.55 (n=5)	2.16 (n=6)	1.66 (n=1)	2.44 (n=7)	1.53 (n=4)
Sedge	1.19 (n=4)	1.80 (n=3)	n/a	2.78 (n=3)	1.49 (n=2)	1.37 (n=3)
Heather and Sedge	1.16 (n=1)	1.66 (n=2)	1.83 (n=2)	n/a	1.43 (n=1)	1.45 (n=6)
Heather and Moss	1.73 (n=1)	2.02 (n=1)	n/a	n/a	n/a	1.53 (n=2)
Sedge and Moss	1.67 (n=2)	n/a	n/a	n/a	n/a	n/a
			Potassium ($mg\ kg^{-1}$)			
Moss	257.56 (n=6)	211.55 (n=4)	151.88 (n=7)	279.03 (n=6)	n/a	n/a
Heather	582.95 (n=1)	196.19 (n=5)	192.47 (n=6)	n/a	209.17 (n=7)	213.76 (n=4)
Sedge	153.41 (n=4)	186.01 (n=3)	n/a	367.73(n=3)	209.17 (n=2)	213.76 (n=-3)
Heather and Sedge	286.66 (n=1)	144.96 (n=2)	123.65 (n=2)	n/a	286.81 (n=1)	235.58 (n=6)
Heather and Moss	219.62 (n=1)	251.82 (n=1)	n/a	n/a	n/a	227.40 (n=2)
Sedge and Moss	199.76 (n=2)	n/a	n/a	n/a	n/a	n/a
			Phosphorus ($mg\ kg^{-1}$)			
Moss	422.63 (n=6)	341.56 (n=4)	445.99 (n=7)	444.53 (n=6)	n/a	n/a
Heather	745.6 (n=1)	361.97(n=5)	502.63 (n=6)	n/a	441.68 (n=7)	369.76 (n=4)
Sedge	363.93 (n=4)	319.63 (n=3)	n/a	527.28 (n=1)	345.49 (n=2)	292.37 (n=3)
Heather and Sedge	511.94 (n=1)	252.85(n=2)	378.86 (n=2)	n/a	389.89 (n=1)	372.30 (n=6)
Heather and Moss	430.17 (n=1)	414.65(n=1)	n/a	n/a	n/a	425.30 (n=2)
Sedge and Moss	407.80(n=2)	n/a	n/a	n/a	n/a	n/a

Wetter sites were associated with mosses as anticipated, whilst grasses tended to be associated with drier sites. No clear trends were identified between pH values and vegetation type, mosses were expected to dominant the most acidic sites. Trends may not be evident due the very narrow range of pH values identified at Moor House.

The results indicate (with the exception of the burnt site – every 10 years) that where sedges were present, lower concentrations of elements were identified, and where heather or a combination of heather and moss are present, values are higher. At the burnt site (every 10 years) higher nutrient concentrations were identified at locations where sedges were recorded; the lowest concentrations were found where mosses and heathers were noted. These results contrast with the other sites considered and may reflect the recent burn at this site. Sedges are likely to recover most quickly and thus will be the first to breakdown and form new inputs of substrate into the peat.

The results demonstrate the importance of vegetation type on upland peats, and indicate that vegetation represents an important input of nutrients and has a notable effect on moisture content but not pH.

The results of the analysis show that vegetation composition is a potentially important control of nutrients and moisture content in upland peats, although not pH. Changes in vegetation on upland moorlands have been linked to land management (Chapman & Rose 1991). Future land management practices need to consider how management will influence the species of vegetation present and hence the chemical properties of the peat that drive the carbon cycle. Drier, more nutrient rich heather sites could result in an increase in carbon mineralisation and subsequently carbon loss. Further discussion of carbon losses from peat is presented in Chapter 6.

4.5 Discussion

4.5.1 Impact on Trace Metals

Concentrations of trace metals (nickel, molybdenum, cobalt and selenium) were typically found to be low in all instances i.e. within an order of magnitude of the detection limit. These elements are generally required in the production of carbon dioxide and/or methane gases (Basiliko & Yavitt 2001). The concentrations identified are sufficiently low that the effects of land management are difficult to separate out from background concentrations i.e. concentrations typically found in peats in the Moor House area. Published data on background concentrations of trace metals are not available for UK peatlands. Data presented in Table 4.16 show the differences between values for the unmanaged site at Moor House compared to the values attained for managed sites.

Table 4.16 Comparison of Trace Element Concentrations between Managed and Unmanaged Sites at Moor House

Element	Average at Unmanaged Site (mg/kg)	Range of Average Concentrations for Managed Sites (mg/kg)	Range of Values Attained for Managed Sites (mg/kg)
Molybdenum	all below detection limit (0.01)	0.38 – 1.73**	0.05 – 2.45 **
Nickel	4.00 (n=15)	1.6 – 4.6	0.1 – 7.1
Selenium	6.64 (n=15)	2.71 – 5.90	0.13 – 9.26
Cobalt	0.71 (n=15)	0.39 – 0.66	0.04 – 1.91

*only one value was identified for the unmanaged site (all others were below the limit of detection). ** No values for drained and grazed sites (all values were below the limit of detection).

Whilst these trace elements are required for the production of greenhouse gases, the quantities that each micro-organism requires for survival are unknown. It is therefore not possible to conclude whether deficiencies might be occurring in the peats examined. No significant differences in the surface concentrations of cobalt or molybdenum were identified using ANOVA ($p=0.036$ and <0.005 respectively). The unmanaged site was found to have significantly different ($p<0.005$) concentrations of selenium to the burnt sites (every 10 years, both grazed and ungrazed) and the grazed site. Strong correlations between trace elements and gas production identified by Basiliko and Yavitt (2001), correlation data between gaseous carbon loss and nutrient and metal concentrations are present in chapter 8.

4.5.2 The Effect of Land Management on the Chemical Properties of Peat

4.5.2.1 Impact of Afforestation on Peat Chemistry

As hypothesised, the afforested site was found to be significantly drier than all other treatments, perhaps owing to the greater water demand of the trees compared to heather, mosses, grasses and sedges. In addition the trees provided the peat with shelter from precipitation through interception. Thirdly, the site was drained prior to the planting of trees and cores taken from ridges were generally found to be drier than cores collected from furrows.

The pH of the afforested site was found to be significantly lower than the unmanaged, burnt and grazed sites as predicted. Byrne and Farrell (2005) identified lower pH values in afforested peats, and work carried out by Laiho et al. (2004b) found drained sites to have a lower pH (3.0-3.1) than those that are not drained (4.1). The presence of tree and drains appears to have contributed to the results observed at Moor House.

The afforested site had some of the lowest concentrations of nitrogen at Moor House (1.26 %). The demand for nitrogen by trees is likely to be greater than the vegetation species found on all other sites. Concentrations of nitrogen were significantly lower than those found in the burnt (every 20 years) site. Concentrations of phosphorus were among the lowest concentrations identified at Moor House, although the differences were not significant (with the exception of the burnt (every 20 years) site) having a significantly higher phosphorus content). Afforested peats frequently have a more flashy hydrological regime owing to the presence of drains which allow rates of flow to be higher, and the movement of water out of the system to be quicker (Pyatt 1993). As a result of a flashier regime, nutrients could potentially either leach out faster or may not be retained at all.

Concentrations of potassium were among the highest at the afforested site, potassium is considered to be a highly mobile nutrient (Westman & Laiho 2003), therefore, the high concentrations are surprising. Rydin and Jeglum (2006), however, noted that the highest concentrations of potassium tend to be found in treed mires, as under anoxic conditions potassium is regularly flushed and/or leached out. Under aerobic conditions however potassium may be strongly bound to rootlets and retained by micro-organisms, which could explain the trends observed at Moor House. This notion is supported by Bragazza et al. (2005) who found greater concentrations in hummocks on a Swedish mire than in hollows.

Published data on iron could not be found for afforested peats, however, work carried out by Heathwaite (1990) indicated that peatland drains cause iron concentrations to lower. The afforested site was found to have significantly lower concentrations of iron than the unmanaged, burnt and grazed treatments, and it is possible that this is attributable to the presence of drains at the site.

Of the trace metals analysed, molybdenum and nickel concentrations at the afforested site were found to be among the highest observed across all treatments; while selenium and cobalt concentrations were among the lowest recorded at the site. Although significant differences were found to exist between the afforested site and the grazed, unmanaged and burnt sites, the differences are sufficiently small that it is difficult to determine whether the differences are great enough to influence gaseous losses of carbon from the sites.

The results suggest that afforestation of peat results in drier more acidic peats, however, the effects on nutrient concentrations are less clear. No significant differences in nutrient concentrations were identified between the unmanaged and afforested sites; suggesting that the demands from trees combined with and water tables differences between the sites were not great enough to have a significant effect. On the whole, concentrations of nutrients at the afforested sites were found to be significantly different from those at the burnt sites, in the majority of cases (for five of the eight elements) the burnt (every 20 years) site had significantly higher concentrations of nutrients than the afforested site, although the burnt (every 10 years) site was only significantly different to the afforested site for two of the eight elements analysed. The differences are likely to be caused by higher inputs of nutrients entering the burnt plots from ash deposits, and whilst the additional nutrients are utilised by young, rapidly growing plants on the burnt (every 10 years) site, the more mature vegetation of the burnt (every 20 years) site has a lower nutrient demand, and thus nutrients are stored in the peats. Additionally, it is possible that the nutrients released during the most recent burn were leached from the burnt (every 10 years) site rather than adsorbed due to the presence of a crust on the surface of the plot which would have increased overland flow rates and reduced the potential for nutrients to infiltrate.

4.5.2.2 The Impact of Drainage on Peat Chemistry

The drained site was expected to have drier peats than all other sites except for the afforested site, yet the wettest peats were identified at all depths at the drained site. The moisture content of the peats on the drained site were found to be significantly higher than the unmanaged, grazed, afforested, burnt (every 10 years) and burnt and grazed (every 20 years) sites. The spacing of the drains could be one reason why the moisture content of the drained site is higher than expected. Work carried out by Hudson and Roberts (1982) on tile drains in the Plynlimon catchment in mid-Wales found that for a significant reduction in the moisture content of peat to occur, drains should be spaced no more than two metres apart. The drains on Burnt Hill are narrow and spaced between 10 and 15 m apart, therefore may not be effective in lowering the moisture content of peats, although the findings of Hudson and Roberts (1982) might not apply outside of their study area. Data presented in Chapter 5 on total organic matter content (Figure 5-4) demonstrate that the drained site has a much higher total organic matter content which could account for the higher moisture content.

The nitrogen content of the drained site was the second lowest of the treatments examined, although only the peats from the burnt (every 20 years) site had a significantly higher nitrogen content than the drained site. The differences could be attributable to leaching of nitrogen into the drains, vegetation on the drained site having a higher nitrogen demand than the burnt (every 20 years) site, or lower inputs of nitrogen into the drained site. Westman and Laiho (2003) found nitrogen concentrations fluctuated not only with time since drainage but also with vegetation type, with the greatest concentrations being associated with herb-rich vegetation and lowest concentrations found where dwarf shrubs are present. The absence of differences between the drained and unmanaged site at Moor House may be a result of the similarities in the vegetation between the two sites. Work carried out by Laiho et al. (1999) identified significant differences in vegetation community between recently drained sites, and sites that had been drained between 41 and 56 years ago had twice as much nitrogen as undrained sites.

Slightly higher phosphorus concentrations were identified at the drained site compared to the unmanaged site, contrary to the findings of Heathwaite (1990) who found nearly twice as much phosphorus in the drained site compared to undrained peats. The differences between this study at Heathwaite's could be attributed to the differences in the two sites. Heathwaite (1990) studied a lowland fen that supported grasses on neutral peats compared to the acidic blanket bog found at Moor House. The study, however, represents the only other comparison of the properties of a drained and an undrained peat in the UK. Laiho et al. (1999) found small increases in phosphorus concentrations in drained sites compared to undrained site. The time since drainage was seen as a significant factor – the longer the time since drainage, the higher the concentrations of phosphorus. A cause for this observation is not presented, but it is possible that drainage coincided with changes in vegetation to species with a greater phosphorus demand which decreased over time. Differences in the development of vegetation at Moor House could be responsible for the contrast in the observations made. Further work by Laiho et al. (2004a) showed that the phosphorus concentrations in peatlands differ little from undrained peats except in the cases where peat has been drained for a long period of time, in this case, sites drained between 1961 and 2004 had almost identical phosphorus concentrations to the undrained site, the site drained in 1937 had 22 % more phosphorus than the undrained site. Such differences may be attributable to variations in the depth and spacing of drains between these studies and/or the demands of the vegetation at each site and/or differences in the environmental conditions that microbes are subjected to. The lack of a significant difference between the unmanaged and the drained sites at Moor House therefore may reflect the spacing of the drains and/or lack of difference in vegetation species or time since the drains were installed.

The potassium concentrations in the drained site were the highest found at Moor House. The results may be a reflection of the oxic conditions present, as potassium is often flushed out under anoxic conditions. While this notion would be plausible on a site where the water table had been lowered and the thickness of the acrotelm increased, the peats on the drained site were among the wettest found at Moor House, indicating that conditions were far from oxic. The total organic matter content of the drained site was the highest of any of the treatments at Moor House (as illustrated in Chapter 5), and is possible that this provided cation exchange sites for potassium to be adsorbed to and retained within the peat.

The drained site was identified as the most acidic of the sites examined, the pH results were found to be significantly lower than the unmanaged, burnt and grazed sites. Work carried out by Laiho et al. (2004b) also found drained sites to have a lower pH (3.0-3.1) than those that are not drained (4.1). Mean values obtained in this study were 0.6 pH units lower at the drained site compared to the unmanaged site. A casual mechanism for the reduced pH values of drained sites is not clear, however, given the additional organic matter content of the drained peats, it is possible that cations have become adsorbed to the organic matter and displaced hydrogen ions in the process, thus decreasing the pH of the peat.

Concentrations of iron were significantly lower than the burnt, grazed and unmanaged sites, a finding that is compatible with that of Heathwaite (1990). Although the reason for lower differences in values is not evident, the concentrations are sufficiently large that it is unlikely that the differences will have a significant effect on carbon cycling.

Of the trace metals analysed, molybdenum and selenium concentrations were found to be among the lowest observed, whilst cobalt and nickel concentrations were among the highest recorded at the site. Although significant differences were found to exist between the drained site and the grazed (cobalt) burnt every 10 years (cobalt and molybdenum) burnt every 20 years (nickel, molybdenum), burnt and grazed every 20 years (nickel and molybdenum), burnt and grazed every 10 years (molybdenum), afforested (molybdenum) and unmanaged (cobalt) sites, the differences are sufficiently small that it is difficult to determine whether the differences are great enough to influence gaseous losses of carbon from the sites.

4.5.2.3 Burning

This section of the discussion relates to the burnt (every 10 years) site which as previously noted was burnt in 2007. A discussion of the effects of burning frequency is presented in section 4.5.7, whilst the impact of combining burning and grazing is presented in section 4.5.6.

No significant differences in the moisture content of the burnt site were found compared to the unmanaged site. The drained site was the only site to have a significantly different (higher) moisture content from the burnt site. Burnt peats are often associated with the formation of a crust on the surface which reduces rates of infiltration and therefore potentially reduces the moisture content of the peat. Conflicting evidence exists however as to whether burning increases infiltration rates as proposed by Imeson (1971) who found increases in throughflow as a result. Conversely, Mallick et al. (1984a) found that rates of infiltration decreased due to pores becoming clogged with ash particles from the burn (the structure and porosity of the peat is considered in Chapter 6). Debano (2000) notes that (wild) fires cause water-repellency due to the formation of a crust on the post-fire soils. The size of the crust depends on the temperature of the fire, its extent, and the properties of the soil. The absence of a significant difference in the moisture content of the burnt sites compared to the unmanaged site could signify that a crust did not form during or after the burn, and that infiltration rates were not affected. The results may also reflect the length of time between burning (February 2007) and the collection of the peat cores (September 2008). Whilst evidence of burning was noted in the field (dry peats, charred vegetation and reduced vegetation size compared to other sites as witnessed in Figure 4-11) recovery of the plot was under way, therefore, it is possible that water repellency had decreased and conditions were returning to their pre-burn state.

Figure 4-11 Heather Burnt in Spring 2007 - the heather has evidently been charred whilst sedges have recovered.

The nitrogen, phosphorus and potassium contents of the peat samples collected from the burnt site were not significantly different from any other treatment. The burnt (every 10 years) site featured some of the highest phosphorus concentrations, whilst the nitrogen and potassium concentrations were in the middle of the range of concentrations found in the samples collected across all treatments. The results suggest that increasing inputs of nutrients from ash during the burn did not make a significant difference to nutrient concentrations, contrary to the hypotheses presented in section 4.2.2.

Work carried out by Ward et al. (2007) on the managed plots at Moor House did not demonstrate a significant difference between the nitrogen content of peats collected from burnt and unburnt plots. The work carried out by Ward et al. (2007) was completed before the 2007 burn, and provides a benchmark with which to compare peats prior to burning and post-burning. Given the nature of the 2007 burn ("a cool, quick burn" (R.Rose; pers comm.)), it is possible, that damage to the vegetation was not extensive, and therefore, the demand for nitrogen by regenerating plants was not sufficient to make a difference to the nitrogen, phosphorus and potassium content of the peats. Work carried out by Forgeard and Frenot (1996) found that burning results in a small increase (as little as 0.1%) in the amount of nitrogen found in heathland soils burnt at 150°C whilst a decrease of 0.05% was recorded in soils burnt at 300°C. Their result corresponds with that of Dikici and Yilmaz (2006) who identified higher concentrations of phosphorus and potassium in burnt peats than unburnt peats. The concentrations are much higher than those found at Moor House owing to the different nature of the peats which included use for agriculture (and hence were drained as well as burnt) and are located on the Gavur Lake Peatland, in Turkey. Despite the differences in location and prior management, the Turkish study focussed on comparing the effects of burning on peats, and concluded that peatlands do not recover from burning in the long term.

Peats from the burnt site were found to be significantly more alkaline than the drained and afforested sites, but no differences were identified between the burnt and unmanaged sites. The lack of differences might be attributable to the temperature of the 2007 burn, which was suggested as being "cool" (R.Rose; pers comm.). Forgeard and Frenot (1996) found low temperature burns did not affect the pH value of heathland soils, but they identified a decrease of 0.2 pH unit was identified in the peat burnt at 300°C. Increases of 0.1 pH units were recorded in sites burnt every 10 and 20 years at Moor House, however the differences were not significant. Dikici and Yilmaz (2006) also found burning resulted in an increase in pH values of peat, sites burnt most recently had higher values than those not burnt since 1965.

Iron concentrations at the burnt site were higher than all treatments except for the unmanaged site. Concentrations were significantly higher than the drained and afforested site, but not significantly different to any other treatment. Published data on the effects of burning on iron concentrations in peat could not be found. The higher pH of the burnt and unmanaged sites might have resulted in iron being retained in the peat matrix rather than being leached. Santelmann and Gorham (1988) state that *Sphagnum* mosses have a much greater capacity to retain iron than other vegetation species found on peatlands. Moss species were found to be more prevalent on the burnt sites than other treatments, which could in part, explain the increased iron concentrations found at the burnt site.

Of the trace metals analysed, cobalt and selenium concentrations were found to be among the lowest observed, whilst molybdenum and nickel concentrations were in the middle of the range of values recorded at Moor House. While significant differences were found to exist between the burnt site and: the burnt and grazed (every 20 years) site (nickel and cobalt), the drained site (cobalt and molybdenum), the afforested site (cobalt), the burnt (every 20 years) site (cobalt and selenium), the burnt and grazed (every 10 years) site (cobalt), and the grazed sites (selenium). The differences are sufficiently small that it is difficult to determine whether the differences were great enough to influence gaseous losses of carbon from the sites. Linkages between gaseous losses of carbon and nutrient and metal concentrations are presented in chapter 8.

4.5.2.4 Grazing

The grazed site had one of the highest moisture contents of the treatments examined only the drained site was significantly wetter. The lack of difference in the moisture content of the grazed and unmanaged sites, could be attributable to the low grazing density at the site - noted as 0.04 sheep ha^{-1} by Ward et al (2007), which would not only reduce the potential for vegetation change which was hypothesised as a reason for variation in moisture content, but also compaction due to trampling would be less likely to occur – which was proposed as a second mechanism through which moisture content changes might occur in grazed peats.

Contrary to expectations, the nitrogen content of the grazed site was the lowest of the treatments examined. The results could reflect a lack of inputs of nutrients from sheep urine and faeces due to the low stocking density, combined with an increased nitrogen demand of the plants growing on the grazed site. Only peats from the burnt (every 20 years) site had a significantly different nitrogen content from the unmanaged site. Marrs et al. (1989) also found concentrations of nitrogen to be lower in grazed plots (0.59 %) than in unmanaged plots (1.13 %); as did Ward et al. (2007) who found fractionally higher values for an unmanaged site (1.34 kg m^{-2}) compared to grazed plots (1.32 kg m^{-2}).

Work carried out by Alonso et al. (2001) also identified low stocking densities as a cause for low nitrogen concentrations. Alonso et al. (2001) found that areas of the Southern Cairngorms without fencing had higher concentrations of nitrogen in grazed than ungrazed sites, suggesting that higher concentrations ought to be present in grazed peats. The contrasting results between the present study and that of Alonso et al. (2001) might be as a result of the higher grazing intensity (3.6 sheep ha^{-1}) and the presence of deer at the Southern Cairngorms site. Grant and Hunter (1968) noted that the frequency with which sheep return to particular areas, and the way in which their movements are controlled are important factors in determining vegetation re-growth and state e.g. young shoots or woody stems. Carefully managed flocks would be kept away from recently grazed areas allowing plant time to re-generate, but would not be kept away sufficiently long enough for the heather to become woody and unpalatable.

Heal and Smith (1978) found average concentrations of nitrogen to be 1.04 % in surface litter and 1.64 % in dark brown peats. These data are compatible with those of Allen (1964) who suggested typical concentrations of nitrogen in peats at Moor House were 1.1 %. These comparisons indicate that the results found in this study are within the typical range of values found at Moor House NNR and that little change in concentrations has occurred over time. Concentrations of potassium and phosphorus were also at the lower end of the range of results recorded for all sites, however, the differences were not found to be significant. The results indicate that the site either does not benefit from additional nutrients from sheep faeces, or, if additional nutrients are added, they are fully utilised, potentially due to the additional nutrient demands by plants needing to continually re-generate themselves to recover from grazing.

Lower concentrations of phosphorus in the peat were also identified in the grazed plots at Moor House by Marrs et al. (1989) than in the other experiment plots. The differences in mean phosphorus concentrations were much greater than those witnessed in the present study – a difference of 330 mg kg^{-1} was found between enclosed and the grazed plots, with the unmanaged site having a mean value of 770 mg kg^{-1}. The considerable drop in values for the unmanaged site between the two studies is surprising but could possibly be account for by the fall in pH values between the study carried out by Marrs et al. (1989) of 4.3 and the value identified in this study of 3.6. The fall in pH could have resulted in phosphorus ions being replaced by hydrogen ions. Work carried out by Hardtle (2009) also found grazing to reduce phosphorus concentrations.

In contrast, Marrs et al. (1989) found a difference in mean potassium concentrations from 370 mg $kg\text{-}^{1}$ in unmanaged sites to 210 mg kg^{-1} in the grazed plot. Such a difference between the two data sets may be due to changes in vegetation composition and/or substrate resulting in either less nutrients being released from the organic matter, less capacity for the peat to retain the nutrients or greater demand for potassium by plant species. Marrs et al. (1989) suggest less litter accumulates at grazed than ungrazed sites and cited this mechanism as a cause for decreased concentrations of plant nutrients in peats.

The pH of the grazed site was significantly higher than that of the drained and afforested sites, and although lower than the unmanaged site, the difference was not significant. The addition of faeces from sheep was expected to have lowered the pH of the grazed site, but the absence of a significant difference between the pH of the grazed and unmanaged sites, adds further support to the theory that additions are not significant, and therefore the nutrient content of the grazed site has not been significantly altered.

Iron concentrations were found to be significantly higher at the grazed site compared to the drained, afforested, and the burnt and grazed (every 10 and 20 years) sites. Published data on the effects of grazing on iron concentrations could not be found.

Given the lack of apparent impact of additional nutrient inputs from sheep urine and faeces on other nutrients, the differences are unlikely to be attributable to the actual grazing process itself.

Of the trace metals analysed, cobalt and molybdenum concentrations at the grazed site were found to be among the lowest observed, whilst selenium and nickel concentrations were in the middle of the range of values recorded at Moor House. Whilst significant differences were found to exist between the grazed site and: the burnt and grazed (every 20 years) site (nickel, molybdenum and cobalt), the drained site (cobalt), the afforested site (cobalt and molybdenum), the burnt (every 20 years) site (cobalt), the unmanaged site (nickel and selenium), the burnt (every 10 years) site (selenium), and the burnt and grazed (every 10 years) site (molybdenum). The differences are sufficiently small that it is difficult to determine whether the differences are great enough to influence gaseous losses of carbon from the sites. Linkages between gaseous losses of carbon and nutrient and metal concentrations are presented in chapter 8.

4.5.3 Changes in the Chemical Properties with Depth for Differently Managed Peats

The sites studied at Moor House have been managed in a similar way since the 1950s. Peatlands are reported to grow at approximately 1 mm a^{-1} (Charman 2002), and so, the impact of management on the peat therefore was expected to be most visible in the 0 to 10 cm layer studied, with fewer differences expected between the remaining layers examined. An examination of differences with depth in the concentrations of the main nutrients (nitrogen, phosphorus and potassium), pH and moisture contents of the differently managed peats is presented below. As noted previously, changes in these properties were expected with depth owing to the needs of the plants growing on the peat, changes in water table levels and due to the fact that most of the peat was formed during times when land management practices were not in place.

pH values were expected to increase with depth as the amount of humified material tends to increase with depth in peats (Stewart & Wheatly 1990). In the case of pH values, all sites showed an increase in value with depth. For all sites, the increase in alkalinity between the surface layer and deeper layers was found to be significant. In addition an increase in the range of pH values between sites occurred, indicating greater variation in pH with depth. Statistical analysis using ANOVA identified significant differences between the pH values for the unmanaged site and the drained site at all depths ($p<0.005$) and the afforested site between 30 and 50 cm below the surface ($p<0.005$). The results support data published by Updegraff *et al.* (1995) where a slight increase in pH values were identified with depth in a treed *Sphagnum* bog.

Phosphorus concentrations decreased gradually with depth. The range of phosphorus values within each depth category also decreased between the 0 and 10 cm layer and the layers below. The decrease in concentrations with depth between the surface layer and deeper layers was found to be significant for all sites. No significant differences were identified between treatments in any of the layers using ANOVA. This result is consistent with Cuttle (1983) who identified declines in phosphorus values with depth, with lowest values at sites where *Sphagnum* mosses were identified. The decrease in concentrations with depth can be attributed to inputs stemming from precipitation and degradation of plant matter in the surface layers, where the nutrients would either be taken up by plant roots or adsorbed onto the surfaces of the organic matter.

Potassium concentrations decreased with depth until the 30-50 cm layers where increases in values were observed. The sharp decline in values with depth indicates that inputs are from the surface of the peat and are either utilised or leached with depth. Differences in the concentrations of potassium between the surface layers of the managed sites and the deeper layers were found to be significant. Laiho et al. (1999) demonstrated little variation in values with depth in Finnish mires for unmanaged sites, but identified decreases with depth in drained sites. Similarly Heathwaite (1990) found concentrations of potassium declined with depth, as did Basiliko et al. (2006) at Mer Bleue in Canada.

Nitrogen concentrations fluctuated greatly with depth as illustrated on Figure 4-4. Variation in the range of values decreased slightly with depth but not sufficiently to reduce the number of significant differences identified between the unmanaged site and the managed plots. Differences between the surface peat and deeper layers were not significant within any of the treatments except for the grazed plot. The lack of trends with depth (i.e. the results increased and decreased within each treatment with depth) contrasts with work carried out by Laiho et al. (1999) where concentrations of nitrogen were found to decrease with depth. Variations in nitrogen content could reflect the varied vegetation identified in the Moor House peats, with roots of differing depths and hence different demands for nitrogen at different points within the profile.

Moisture contents were found to increase with depth in all plots down to the 30 to 40 cm layer, reflecting increases in the saturation of the peat. In all cases a small decrease in the moisture content of the peat was observed between 40 and 50 cm beneath the surface. Whilst unexpected (previously moisture contents were hypothesised to increase with depth), it is possible that this observation could be accounted for by water demand from the vegetation peaking at this depth. Significant differences in the moisture content of the surface peats and deeper layers were observed for the grazed, drained and afforested sites. In addition, the afforested site was found to be significantly drier than the unmanaged site between 20 and 50 cm.

In general, as predicted, moisture content and concentrations of nutrients tended to decrease with depth (with the exception of nitrogen) and pH increased slightly. Overall the results suggest there is as much variation between treatments in the peats at depth as there is at the surface of the profile. Such variations may be attributable to the different methods of management used at the site, and could reflect one or more of the following:

- ❖ The lowering of the water table on sites that have been drained could give rise to elements being leached out of the system;
- ❖ The depth at which roots extend to could vary between sites (especially between the unmanaged site and the afforested site). Such variation could explain changes in nutrient demands at different depths;
- ❖ Inputs of ash could leach through the profile at the burnt site, giving rise to variations in concentrations of nutrients throughout the profile;

- ❖ Plants recovering from either grazing or burning could draw up nutrients from different parts of the profile, resulting in differences when compared to other sites.

4.5.4 The Influence of Vegetation Type on the Chemical Properties of Peats

The vegetation present on peatlands dictates the substrate quality of the peat and influences chemical properties such as nutrients which are a key requisite for vegetation growth. Changes in land management have been associated with changes in the vegetation found growing on peats. Grazed sites studied near Braemar in the southern Cairngorms typically have more grasses and less heather than unmanaged sites (Alonso et al. 2001); while burnt sites have an array of vegetation types which change over time depending on when the most recent burn took place, and what vegetation was destroyed during the burn.

Heather was expected to produce the highest concentrations of nutrients based on work carried out by Alonso and Hartley (1998) and Shaver et al. (2006). Conflicting evidence exists in the literature as to whether mosses have high concentrations of nutrients or not. Work carried out by Gorham et al (1986) noted that mosses have a higher capacity to adsorb elements than other species of vegetation on moorlands, yet Buttler et al. (1994) demonstrated that mosses have a lower nitrogen content compared to other vegetation species. The results presented in this study (Table 4.15) suggest that mosses have higher concentrations of nitrogen, whilst heather and grass combinations featured the lowest concentrations for nitrogen.

Cuttle (1983) found *Sphagnum* mosses to have low concentrations of phosphorus whilst *Calluna* species had higher concentrations, and species of sedges and grasses (*Eriophorum and Molinia* varieties) had the greatest range of values. In this study at Moor House, concentrations of phosphorus and potassium were neither high nor low compared to other values in the data set.

Mosses were expected to be present at the wettest sites, and the results demonstrate that this was the case. No clear trends were identified between pH values and vegetation type, despite mosses being expected to be at the most acidic sites. Trends in the data may not be evident due the very narrow range of pH values identified at Moor House.

The moisture content of the peat however is determined by micro-topography and inputs of rainfall, and therefore, is responsible for determining where mosses develop. Changes in vegetation on upland moorlands have been linked to land management (Chapman & Rose 1991). Future land management practices need to consider how management will influence the species of vegetation present and hence the chemical properties of the peat that drive the carbon cycle. Drier, more nutrient rich heather sites could result in an increase in carbon mineralisation and subsequently carbon loss. Further discussion of carbon losses from peat is presented in Chapter 6.

4.5.5 Variation in Chemical Properties within One Treatment

The results of the triplicate study suggest that there is variation within one method of management of upland peatlands. The differences might be attributable to differences in the burning intensity between the plots (data on the temperatures reached during the most recent burn are unavailable) or may reflect heterogeneity in the properties analysed. The ECN hold records made at the time of the burns suggest the burning on Block A was a light burn, and that much of the vegetation was still frozen before the burn. The burns in Blocks B and C were much faster, and this was attributed to the frost having melted by the time these burns commenced (R.Rose, pers. comm.). Research carried out by Forgeard and Frenot (1996) supports the notion that the temperature of the burn has an impact on the chemical properties of moorland soils. Work carried out under laboratory conditions found that hotter burns result in chemical properties that are less similar to unmanaged sites than cooler burns. The results indicate that significant variation exists within each treatment for cobalt, pH, nickel and iron and therefore variations between treatments cannot be attributed to land management alone. The results demonstrate that natural heterogeneity exists within peatland environments and indicate how difficult it is to replicate data.

4.5.6 The Effect of Combining Burning and Grazing on Peatland Chemical Properties

The Hard Hill plots at Moor House NNR afforded the opportunity to study combinations of burning and grazing treatments on peat. Plots which were both burnt and grazed (on 10 and 20 year cycles) were studied to determine how their properties differed from plots that are solely burnt or grazed. Burnt and grazed plots were expected to have higher nutrient concentrations owing to inputs from two potential sources – sheep faeces/urine and ash after burning.

The moisture content of the burnt and grazed (every 10 years) plot was found to be higher than the separately burnt and grazed plots alone at all depths considered, yet statistical analysis did not find these differences to be significant. Little change was noted in the pH values, and no significant differences were identified as a result of analysis using ANOVA.

Concentrations of nitrogen were very similar to those found on the grazed site in the surface soils. The highest concentrations, however, were found in the burnt and grazed site in the 10 to 20 cm zone. No significant differences, however, were identified between the treatments. Phosphorus and potassium values were higher in the burnt and grazed plots in surface peats than burnt and grazed plots alone. Concentrations of trace elements (with the exception of nickel) were lowest in the burnt and grazed plots. Statistically significant differences (using ANOVA) were not identified between the treatments for phosphorus, potassium or trace metals at any of the depths considered.

In general, burning and grazing at one location within the Moor House managed plots appeared to give rise to slightly higher concentrations of key nutrients in shallow peats, however, the differences were not significant. This pattern may be attributable to inputs of nutrients from three sources – rainfall, ash during burning and waste products from sheep. The grazed plot was found to have lower concentrations of nutrients than the unmanaged site, and this was attributed to the low density grazing that occurs at Moor House. While this maybe the case, it is probable that when sheep are in the vicinity of the Hard Hill plots, the burnt and grazed plot is given preference owing to the young, more tender plants that are available for grazing as Shaw et al. (1996) noted that sheep tend to avoid woody, unpalatable species in favour of younger vegetation.

4.5.7 The Influence of Burning Frequency on the Chemical Properties of Peat

The Hard Hill plots afforded the opportunity to make comparisons between sites that were burnt every 10 years and those burnt every 20 years. Comparisons of data and indicated that in all cases the 20 year burn resulted in wetter conditions but there was no effect on pH values. Little change in pH values was detected between all treatments considered; therefore it is unsurprising that significant differences were not noted between the burning cycles.

Data for all metals (iron, nickel, selenium and cobalt) at all depths showed greater concentrations to be present in the burnt (every 10 years) site. The results confirm the theory suggested earlier that burning results in greater inputs of elements into the peat. The 10 year site has not only been burnt more recently resulting in fresher inputs of metals from the ash into the site, but also is burnt more frequently, therefore, greater concentrations of elements might have entered these peats over time.

In all cases, concentrations of nitrogen were much higher in the 20 year burn site than the 10 year burn site. Such trends are likely to be attributable to the higher nitrogen demand of new plants growing on the most recently burnt site. Concentrations of potassium were found to be higher on the 10 year burn site, but phosphorus levels were greater in the 20 year site in the top 20 cm. The lower phosphorus levels in the burnt every 10 years site may reflect increased demand by new plants for this nutrients, especially as the lower concentrations were detected in the root zone.

Dikici and Yilmaz (2006) also found significantly higher concentrations of potassium in a plot burnt in 2001 compared to the site burnt in 1965; although the concentrations were still much higher than those found in the unmanaged site, in contrast to the findings at Moor House. Allen (1964) also found higher concentrations of potassium in burnt peat (272 mg kg $^{-1}$) than unburnt (141 mg kg $^{-1}$) following a burning experiment carried out in the laboratory.

4.6 Conclusions

4.6.1 Summary of Findings

The aim of this chapter was to identify the impact of land management on the chemical properties of peatland soils that are responsible for driving the carbon cycle. Six hypotheses were devised to investigate this aim, the hypotheses are re-stated below along with a summary of the findings.

1. *Land management will impact on the chemical properties of differently managed peats*

Burning was expected to result in higher concentrations of nutrients and this was supported by the results which provided evidence that burning at Moor House has resulted in peats with higher concentrations of nutrients, that are slightly drier and fractionally less acidic. Afforestation was expected to result in drier peats owing to the presence of drains, with a lower pH and reduced concentrations of nutrients due to greater nutrient demand of the trees compared to traditional blanket bog vegetation. The results confirmed the hypothesis as the afforested peats were found to be drier, more acidic peats with lower concentrations of nutrients. The grazed site was expected to either have a higher nutrient content owing to inputs of nutrients from sheep faeces and urine, or a lower nutrient input due to greater nutrient demand from plants recovering from grazing. The results suggested that few differences exist in the properties of the grazed site compared to the unmanaged site. The drained site was expected to be drier, more acidic with lower nutrient concentrations. The drained site however was found to be wetter, with slightly higher nutrient concentrations. The results were attributed to the wide spacing of the drains, and the higher organic matter content of the peats at the drained site.

Concentrations of trace metals in very low. Whilst these metals might be of importance for the production of methane and/or oxidation of methane to carbon dioxide, the concentrations were so low, that land management is unlikely to have impacted upon them.

2. *Land management will not influence peats at depth*

Concentrations of nutrients were not expected to vary beneath the surface, as the effects of land management were not thought to influence the chemical properties of the peat at depth. Differences in the chemical properties between management treatments however were found to continue with depth down the peat profile. The results indicate that land management not only affects the surface layers of the peat, but to a minimum depth of 0.5 m beneath the surface.

3. *Land management will have an impact on the species of vegetation that grow on peats, and thus the nutrient content of the peats will vary depending on the species of vegetation growing on the peat*

Previous work has indicated that a relationship exists between the nutritional content of vegetation and inputs of nutrients into peat; and that grazing alters the nutrient concentrations of both the peat and vegetation. Work on vegetation change on other types of managed sites had not been investigated. In this study, linkages were identified between vegetation type and concentrations of key properties in the surface peats for the unmanaged, burnt and grazed plots. These results suggest that careful management of vegetation is required if the uplands are to be managed in a way which ensures carbon losses are minimised in the future. Changes in vegetation are one of the key results of changes in land management practices, therefore vegetation management should be considered when selecting management practices to reduce carbon losses. Correlations between nutrient concentrations and carbon fluxes will be considered in Chapter 8.

4. *The frequency with which peats are burnt will impact on the chemical properties of the peat*

The frequency with which sites are burnt was predicted to impact on nutrient concentrations, with more frequently burnt sites having higher nutrient concentrations. The results of the investigation demonstrated less frequent burning produces wetter soils with higher concentrations of phosphorus and nitrogen but lower concentrations of potassium.

5. *Combining burning and grazing will result in peats with chemical that different from those that are subjected to just one treatment*

Combinations of burning and grazing were found to have an effect on the chemical properties of the peat, with slightly higher concentrations of key elements.

6. *No differences are expected to exist between the three plots from Blocks A to C that are subjected to burning every 10 years*

Repeat measurements of properties such as nitrogen have been carried out on the Hard Hill plots in the past (e.g. Ward et al. 2007), however, here the results of a range of chemical properties have been combined to present a holistic view of differences between three sites that were subjected to burning on a 10 year cycle. By investigating differences between the plots, variations in peat chemistry were identified; which can be attributed to the intensity of the burns at these sites.

4.6.2 Recommendations for Further Work

The results demonstrate that differences exist in the chemical properties of differently managed peatlands. This work could be further by looking at differences in chemical properties between unmanaged peatlands and at greater depths to confirm whether the variations identified at depth are due to land management or a reflection of the heterogeneity of peats. Further work could also investigate other combinations of management for example burning and drainage. The intensity of management practices has arisen as an important consideration and further work should be carried out to identify at what temperatures burning impacts on the chemical properties of the peat. Additionally, trials could be performed to ascertain what grazing intensities make a difference to the properties of the peat. Further analysis should also be carried out to investigate the immediate impact of the burn and to determine the nature and extent of changes in the chemical properties of the peat in the ensuing months. This would enable an understanding to be gained of whether the effects of burning are immediate or if the changes take effect over time.

5 THE EFFECTS OF LAND MANAGEMENT ON CARBON STOCKS AND QUALITY IN PEATS

5.1 Introduction

Peatlands are renowned for their ability to sequester and store carbon (e.g. Gorham 1991, Blodau 2002, Limpens et al. 2008). Estimates of carbon stocks in peatlands have been carried out in recent decades (e.g. Milne & Brown 1997, Billett et al. 2010, Worrall et al. 2009), and efforts have been made to establish the influence of land management on these stocks (e.g. Armentano & Menges 1986, Worrall et al. 2010a). Although estimating overall peat carbon stocks is important, understanding the composition of the carbon is of greater importance when assessing the impact of climate change on peatland carbon stocks. The ability of carbon stocks to withstand mineralisation is essential if judgements are to be made as to whether one method of managing peat is better from a carbon storage perspective than another. Evidence exists to suggest that more recalcitrant species of plant litter decompose more slowly and therefore less carbon lost through respiration (Yavitt et al. 2005). Peats formed from more recalcitrant species of vegetation are considered to be of lower quality than peats formed from labile plant species (Berg 2000).

When conditions are favourable for microbial activity to take place, the more easily degradable substrates are decomposed by micro-organisms. Increased temperatures in laboratory simulations were found to cause higher rates of organic matter decomposition (Kirschbaum 2006). The breakdown of plant material results in the formation of new organic molecules of differing recalcitrance, compounds with complex structures tend to have low decomposition rates and require a high activation energy in order for decomposition to commence (Davidson & Janssens 2006). Debate over whether different types of organic matter have differing temperature sensitivities to one another has existed for quite some time, with some authors proposing that organic matter is not temperature sensitive such as Giardina and Ryan (2000) whose paper has sparked much controversy. Subsequent work has proposed that more than one pool of carbon exists within soils, and that the single pool model used by Giardina and Ryan (2000) was insufficient to identify such differences.

The importance of substrate quality and the existence of three pools of carbon (fast, intermediate and very slowly degradable carbon) have been highlighted by many authors (e.g. Knorr et al. 2005, Powlson 2005, Fang et al. 2005). All have suggested that the more recalcitrant (i.e. slowly degradable) peats are more sensitive to temperature rises, and thus, increases in temperature under climate change could result in a rapid increase in the synthesis of previously stable carbon stocks. In addition to identifying that rapid synthesis of previously slowly degradable carbon stocks under climate change is likely, suggestions have been made that micro-organisms involved in organic matter decomposition will acclimatise to warmer temperatures, thus enabling decomposition to continue, and losses of carbon dioxide to the atmosphere to continue (Jarvis & Linder 2000).

The aim of this chapter is to identify how land management influences carbon stocks and the carbon quality of peat. To achieve this aim, the following two hypotheses will be investigated:

i) Land management has a significant effect on carbon stocks in peat.
ii) Land management has a significant effect on carbon quality.

Chapter 2 highlighted the paucity of published data available on the effects of peatland management on substrate quality. The chapter examines the four key methods of peatland management used in the UK – burning, grazing, afforestation and drainage with an unmanaged site acting as a control site. Land management influences the environmental conditions that prevail within a peatland, the species of vegetation and hence the inputs of litter into the peat and finally, the availability of nutrients (Laiho 2006). These three factors control rates of organic matter decomposition within peats, and therefore the amount of carbon and quality of carbon present.

The quality of the carbon present within a peatland is governed by the chemical composition of the litter inputs and rates of decomposition. Land management affects the vegetation species growing on the peat, and hence litter quality. Species such as *Sphagnum* have been reported to decompose much more slowly than *Carex* species (Verhoeven & Toth 1995). To date, much consideration of carbon quality has focussed on the litter quality rather than the quality of the peat itself (Bragazza et al. 2009). Investigating the peat itself is important because it is from there that carbon dioxide is lost into the atmosphere and it is this valuable carbon store that needs to be preserved. The rate of degradation of this carbon reserve will depend in part on the chemical composition of the peat. This chapter undertakes a novel method of examining the carbon stocks of managed peatlands, by conducting a modified carbon fractionation analysis, details of the modifications and rationale are provided in section 5.2.3 and analysis of total carbon stocks in managed peats.

Compared to the unmanaged site, carbon stocks are anticipated to be lowest in the drained and afforested sites, where environmental conditions will favour more rapid rates of organic matter decomposition (Holden et al. 2007b). Work carried out at the drained sites at Moor House found drainage to have a limited effect on vegetation composition, (Coulson et al. 1990, Stewart & Lance 1991), whereas changes in the environmental conditions (i.e. more aerobic) on the drained site are likely to impact on rates of decomposition (Laiho 2006). Afforestation of peatlands results in different inputs of litter into the peat, with differing decomposition rates (Domisch et al. 2000).

Rates of carbon accumulation within sites subjected to moorland burning might be higher owing to regular regeneration of plant species, or could be lower due to the absence of litter inputs into the peat post-burning. Little is known about the effects of managed burns on the carbon content of the peat beneath the vegetation and litter layers (Legg et al. 2010), however, changes in vegetation composition have been identified (Hobbs & Gimingham 1984) and are likely to influence carbon stocks. Grazing has been noted to change vegetation composition and structure (Hope et al. 1996). Observations made by Rawes and Welch (1969) at Moor House suggest that sheep are selective grazers, and this can cause changes in the species of vegetation present on grazed sites compared to ungrazed areas. Changes to carbon stocks are therefore likely on grazed sites.

Changes to the structure of the vegetation community and to the prevailing environmental conditions in peatlands as result of management are therefore anticipated to impact on carbon stocks and carbon quality. To date, little data have been published on the whether the changes are significant, and thus the aim of this chapter is to provide an initial baseline assessment of the effect of peatland management on peatland carbon stocks and quality.

5.2 Methodology

5.2.1 Total Organic Matter Content

The total organic matter content of the samples was calculated using the loss on ignition method (Rowell, 1994). Oven dried samples were weighed and placed into pre-weighed crucibles which were placed in a furnace at 550°C for 24 hours. Samples were immediately placed in a desiccator after being removed from the furnace, placed in a desiccator and were weighed once cool. The organic matter content was calculated using the following equation:

$$\text{Loss on Ignition} = \frac{(\text{mass of oven dry soil} - \text{mass of ignited soil}}{\text{mass of oven dry soil}}$$

(Rowell, 1994)

5.2.2 Total Carbon and Nitrogen Content

Total carbon and nitrogen analysis was carried out using a Eurovector EA3000 Elemental Analyser. Tin cups measuring approximately 0.5 mm by 0.5 mm were used, and 5 mg of oven dried soil was added with 1.5 mg of vanadium pentoxide. Vanadium pentoxide was added as a catalyst to ensure that complete combustion occurred once the samples were placed in the analyser[2]. Energy Peat (*Sphagnum*) Reference Material NJV 94-2 was used as a standard reference material. Once prepared, the samples were analysed on the Elemental Analyser, which was calibrated using a range of sulphanilic acid standards. Results were expressed as a percentage of the sample weight. The C:N ratio was calculated by dividing the carbon content by the nitrogen content.

[2] A trial carried out in July 2008 indicated that without a catalyst, incomplete combustion of samples of peat occurred.

5.2.3 Organic Matter Fractionation

Following the method presented by Wieder and Starr (1998), analysis of the different organic fractions of the samples was carried out on selected freeze-dried samples. The method entails a five stage fractionation process which is summarised in Figure 5-1.

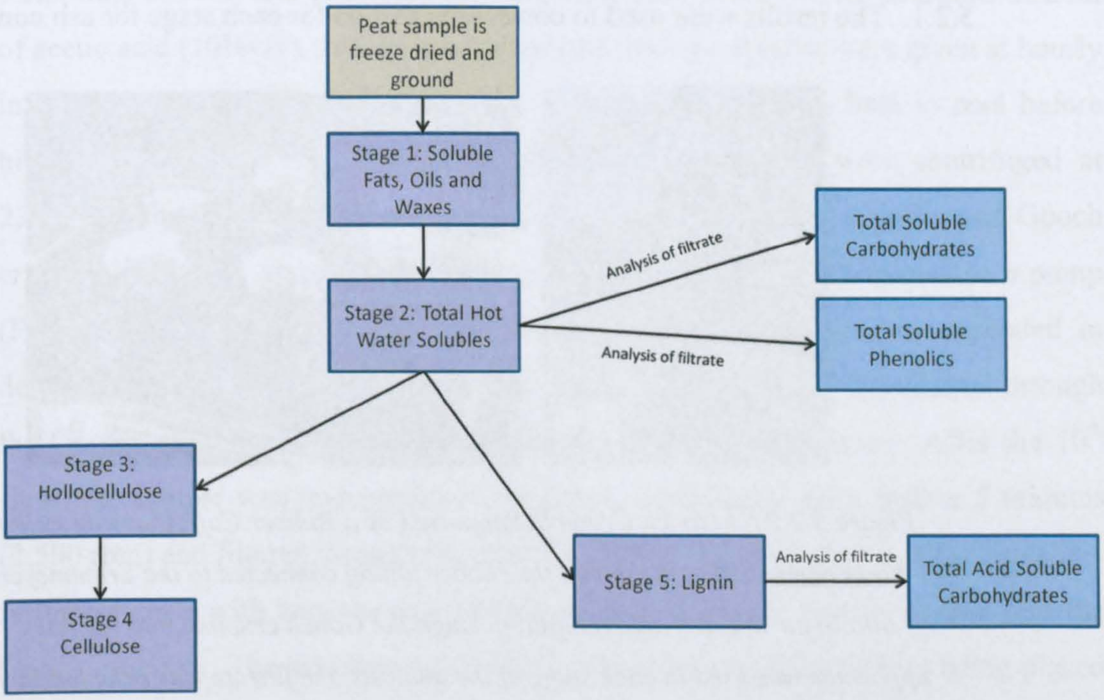


Figure 5-1 Schematic Representation of the Sequential Extraction Procedure used to determine how management affects substrate quality in peatlands (after Wieder and Starr, 1998)

Several trials were carried out on a bulk sample of peat collected from Moor House to test the method and optimise it. The trials found that the use of porous bottomed Gooch crucibles resulted in filtering took excessively long periods of time, therefore Gooch crucibles with perforated bases lined with glass wool were used as an alternative for stages two to five. Rubber cuffs were used to support the Gooch crucibles above the Erlenmeyer flasks (Figure 5.2a). Each filtration stage required the use of a pump to create suction, running the pump at the lowest speed possible was found to be preferable to prevent the crucible bases from becoming clogged with peat residue. Where a water bath and/or a sonicator were required, a lead weight was placed around the conical flask to prevent the contents from spilling (Figure 5.2b). The original method specifies leaving the samples in the oven overnight after stage 2 (total solubles); however, this did not adequately dry the samples. Trials were done to determine whether 72 hours would be preferable, however no further water loss was identified between 48 and 72 hours. Stage 4b of the original method (determination of hemicellulose) was omitted as samples took in excess of five days to filter. After each stage, a sub-sample of the peat was taken and analysed for total organic matter content following the method described in 5.2.1. The results were used to correct the results for each stage for ash content.

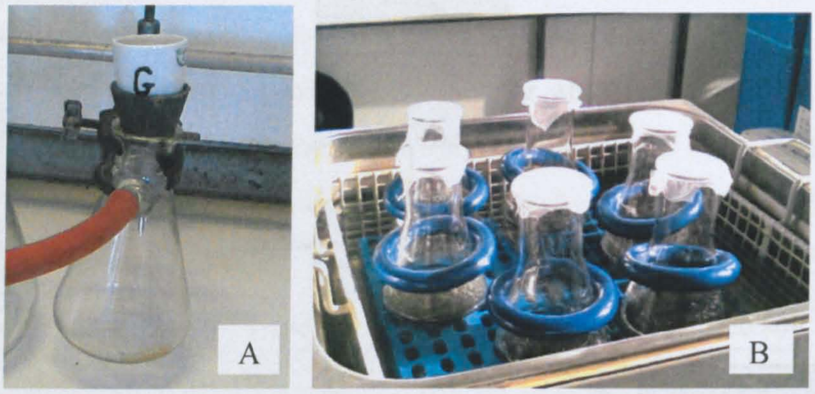


Figure 5-2 A) A Gooch Crucible Supported by a Rubber Cuff Above an Erlenmeyer Flask connected to the pump, the rubber tubing connected to the Erlenmeyer flask allows air to draw excess liquid through the Gooch crucible into the flask. This apparatus was used in each stage of the analysis, the filtrate was reserved in stages 2 and 5 for further analysis. B) Peat samples in Conical Flasks Covered with Parafilm and Supported by Lead Ring Weights to prevent toppling. The flasks are placed in a water bath at 20°C for two hours as part of stage 4.

Soluble fats, oils and waxes (SFOW) were measured by sonicating 2 g of peat in 100 ml of dichloromethane for one hour. The solution was filtered through Gooch crucibles with coarse porous bases. Samples were placed in a vacuum oven at 60°C overnight, cooled in a desiccator and weighed.

Hot water soluble substances were measured by placing the remaining peat into a conical flask with 50 ml of deionised water. Samples were left to boil gently for 3 hours in a sand bath. The solution was filtered through a perforated Gooch crucible lined with glass wool. The filtrate was retained and analysed for soluble carbohydrates and phenolics (details provided below). The Gooch crucibles containing the samples were placed in a vacuum oven at 60°C for 48 hours, cooled in a desiccator and weighed.

Holocellulose was determined by placing three-quarters[3] of the remaining sample into a conical flask and adding 30 ml of deionised water, 2 ml of acetic acid (10%v/v), and 0.6 g of anhydrous sodium chlorite. The samples were placed in a water bath at 75°C and covered with a watch glass. Three further additions of 2 ml of acetic acid (10%v/v), and 0.6 g of anhydrous sodium chlorite were given at hourly intervals. After four hours the samples were placed in an ice bath to cool before being transferred to 50 ml centrifuge tubes. The samples were centrifuged at 2,500 rpm for 5 minutes. The supernatant was filtered through a perforated Gooch crucible lined with glass wool set above an Erlenmeyer flask connected to a pump (Figure 5-3 shows the set-up of the flasks). The sample was re-suspended in deionised water, centrifuged for a further 5 minutes (2,500 rpm) and filtered through the Gooch crucible; this process was repeated a further nine times. After the 10th rinse, the sample was re-suspended in acetone, centrifuged for a further 5 minutes (2,500 rpm) and filtered through the Gooch crucible; this process was repeated twice. A further rinse with acetone was performed and the whole sample poured into the Gooch crucible. The samples were rinsed with petroleum ether before being placed in the vacuum oven for 30 minutes at 105°C. Samples were cooled in a desiccator and weighed.

[3] The method proposed by Wieder and Starr (1998) suggested that two-thirds of the sample is used for stages 3 and 4, however, this resulted in insufficient sample being available for stage 4. Using three-quarters of the sample resulted in sufficient sample being available for stages 3 to 5 inclusive.

Figure 5-3 Set-up of Erlenmeyer Flasks and Gooch Crucibles Used to Filter Samples at Each Stage of the Organic Fractionation Experiment

The sample remaining after holocellulose had been determined was used to measure cellulose. The sample was placed in a conical flask with 20 ml of 4.3M potassium hydroxide and covered with Parafilm. The sample was left at room temperature for 2 hours before being filtered through a perforated Gooch crucible lined with glass wool. The sample was rinsed with 5 ml of acetic acid (5% v/v) followed by acetone and then petroleum ether. Samples were placed in a vacuum oven at 60°C overnight, cooled in a desiccator and weighed.

The remaining quarter of the sample set aside after the determination of total soluble carbon, was used to measure lignin. The samples were placed in test tubes and 4 ml of sulphuric acid (72%) were added. The test tubes were placed in a water bath at 30°C for one hour. The samples were removed and 12 ml of deionised water added. The solution was transferred to a conical flask with an additional 44 ml of deionised water. Samples were placed in an autoclave for one hour at 17 psi. The solution was filtered through a perforated Gooch crucible lined with glass wool. The filtrate was retained and analysed for acid soluble carbohydrates (details provided below). The samples were placed in a vacuum oven at 60°C overnight, cooled in a desiccator and weighed.

The filtrate retained from stage 2 was analysed for soluble carbohydrates, soluble phenolics, whilst the filtrate retained from stage five was used to determine acid soluble carbohydrates. Analysis of soluble fractions was carried out colourimetrically. Soluble carbohydrate solutions were diluted to 1:19 using deionised water, whilst acid soluble carbohydrates were diluted to 1:49 with deionised water. In both cases, 5 ml of solution were taken and 75 μl of liquefied phenol and 5 ml of sulphuric acid were added. Samples were left for 20 minutes and absorbances were read on a spectrometer, with a wavelength set at 490 nm. Glucose standards were used to calibrate the results. Standard solutions were made by creating a 1M solution of glucose, and devising standards that were within the range of absorbance values (1 to 20 ml L^{-1}) identified during pilot trials of the method. Where necessary, a ten fold dilution was carried out on samples exceeding the range of the standards. Liquid phenol and sulphuric were added to the standards in the same quantities added to the samples, and were left to stand for 20 minutes prior to analysis, to enable the colour change to stabilise.

Soluble phenolics were determined by placing 5 ml of the filtrate in a 50 ml volumetric flask, adding distilled water followed by 2.5 μl of Folin-Denis reagent, and 10 ml of 1.6 M sodium carbonate. The solution was diluted to volume and left at room temperature for 20 minutes. Absorbances were read on a spectrometer, with the wavelength set at 760 nm. Tannic acid standards were used to calibrate the results, and were created from a 0.1M solution of tannic acid. Standards were devised that were within the range of absorbance values (1 to 10 ml L^{-1}) identified during pilot trials of the method. Where necessary, a x10 dilution was carried out on samples exceeding the range of the standards. Folin Dennis reagent and sodium carbonate were added to the standards in the same quantities added to the samples, and were left to stand for 20 minutes prior to analysis, to enable the colour to change to stabilise.

The results of the organic fractionation analysis were calculated following the equations presented by Wieder and Starr (1998).

5.2.3.1 Bulk Density

Intact cores (total of 111) were collected from all 8 treatments as described in Chapter 3 and transported to the laboratory on corrugated, plastic sheeting. Bulk density was calculated by pushing a metal cylinder with a volume of 1 cm^3 into the core. The samples were weighed, and placed in a pre-weighed crucible in an oven for 24 hours at 105°C. The samples were cooled in a desiccator, re-weighed and bulk density calculated by dividing the mass of oven dry soil by the volume of the cylinder. Three tests were performed on each 10 cm section of the cores, resulting in a total of 555 tests.

5.2.4 Carbon Stock Calculation

Carbon Stocks were calculated using the method presented by Guo and Gifford (Guo & Gifford 2002) as follows:

$$C_t = BD * C_c\% * D$$

where C_t = total carbon stock (t ha^{-1}), BD = bulk density C_c = carbon concentration (%) and D = depth (cm).

5.2.5 Statistical Analysis

Data for loss on ignition, total carbon and C:N ratio followed a normal distribution based on the results of analysis carried out using the Anderson-Darling test to check for a normal distribution, and comparisons for significance were thus made between the five treatments of interest using analysis of variance (ANOVA). Data from the organic matter fractionation experiment did not have a normal distribution, and the data could not be transformed to create a normal distribution using the following methods: log, square, square root and reciprocal. The Kruskall-Wallis H test and Mann-Whitney U test were therefore used to analyse the organic matter fractionation data.

Analysis of variance with co-variance was carried out to identify differences between the carbon stock content of each of the managed treatments and to determine if there was significant interaction between the drivers of carbon stocks.

5.3 Results

5.3.1 Loss on Ignition

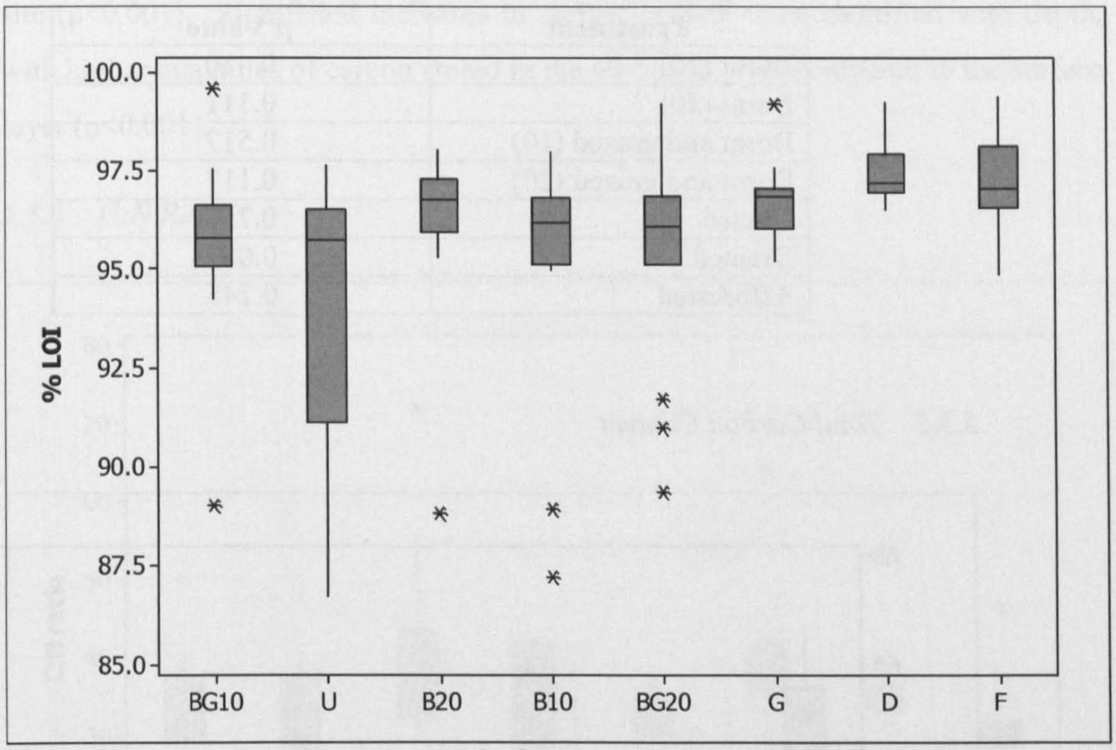


Figure 5-4 Loss on Ignition Results for Samples Collected between 0 and 10 cm

The extent of the box represented the first and third quartiles, with the median represented by the line in the centre of the box. The whiskers stretch to the upper and lower limits within the first and third quartiles, values outside of this range are represented with an *. b10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, g – grazed, u – unmanaged, d- drained, f – afforested.

Loss on ignition values reflect the total organic matter content of the peats (Figure 5-4). ANOVA identified significant differences between treatments (p=0.002). The afforested and drained sites were found to have significantly greater organic matter content than the unmanaged site in both the 0-10 cm and 40-50 cm layers (p<0.001). No significant differences were found between any other treatments. The drained site had the smallest range of values (96.5 – 99.2 %) and the unmanaged site had the greatest (86.7 – 97.7 %).

Small increases in the total organic matter content of the peat occurred with depth, however these increases were not significant. A summary of the significance values is presented in Table 5.1. The drained site had a significantly (p=0.047) higher ash content in the 40-50 cm layer than the 10-20 cm layer.

Table 5.1 Significance values from ANOVA testing carried out to identify whether total organic matter content changed significantly with depth

Treatment	p Value
Burnt (10)	0.410
Burnt (20)	0.111
Burnt and grazed (10)	0.517
Burnt and grazed (20)	0.117
Grazed	0.757
Drained	0.047
Afforested	0.245

5.3.2 Total Carbon Content

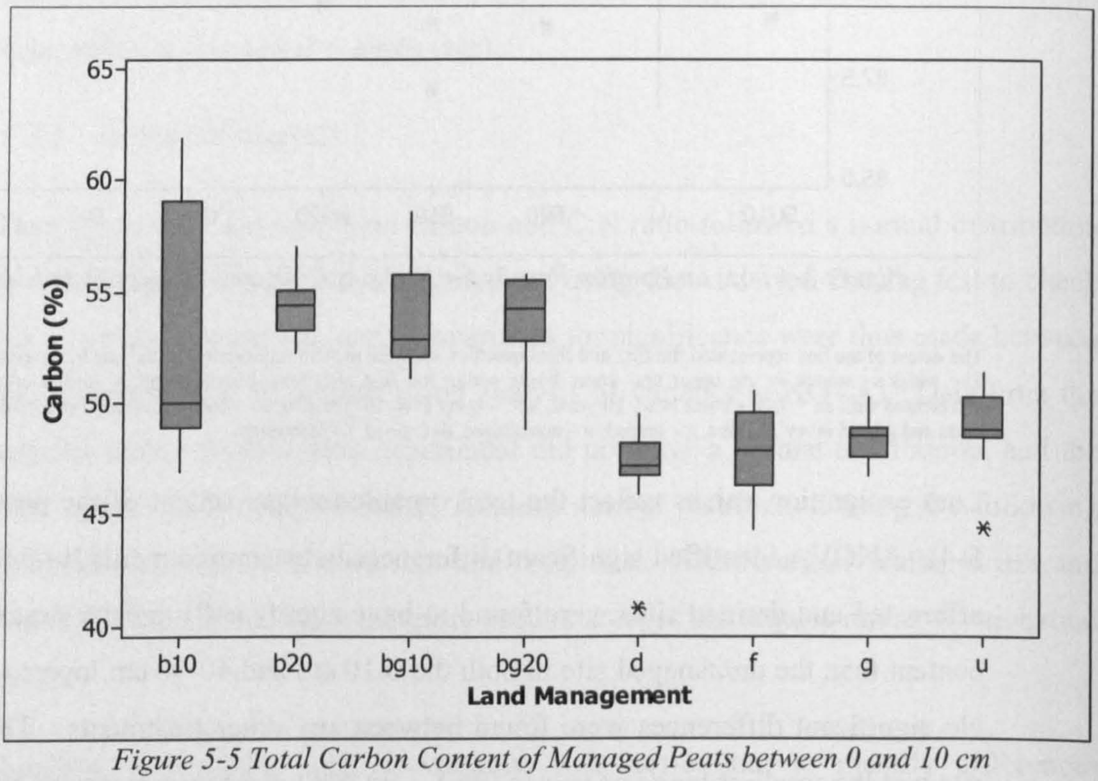


Figure 5-5 Total Carbon Content of Managed Peats between 0 and 10 cm

b10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, g – grazed, u – unmanaged, d- drained, f – afforested. The extent of the box represented the first and third quartiles, with the median represented by the line in the centre of the box. The whiskers stretch to the upper and lower limits within the first and third quartiles, values outside of this range are represented with an *.

Significant differences in the carbon content of peats were found between treatments ($p<0.001$). The burnt sites were found to have a significantly greater carbon content than all other treatments examined in the 0-10 cm layer ($p<0.001$). The drained site had the lowest quantity of carbon (46.9 %) whilst the burnt site (every 20 years) had the greatest (54.3%) (Figure 5-5). There were no significant differences between treatments at the base of the profile (p=0.123). In the 10-20 cm zone, the burnt sites (every 10 and 20 years) had significantly higher carbon stocks than the drained and afforested sites. The drained site had significantly less carbon than the unmanaged site ($p<0.001$). Significant increases in carbon content were identified with depth, with higher quantities of carbon stored in the 40-50 cm layer compared to the surface layer ($p<0.001$).

5.3.3 C:N Ratio

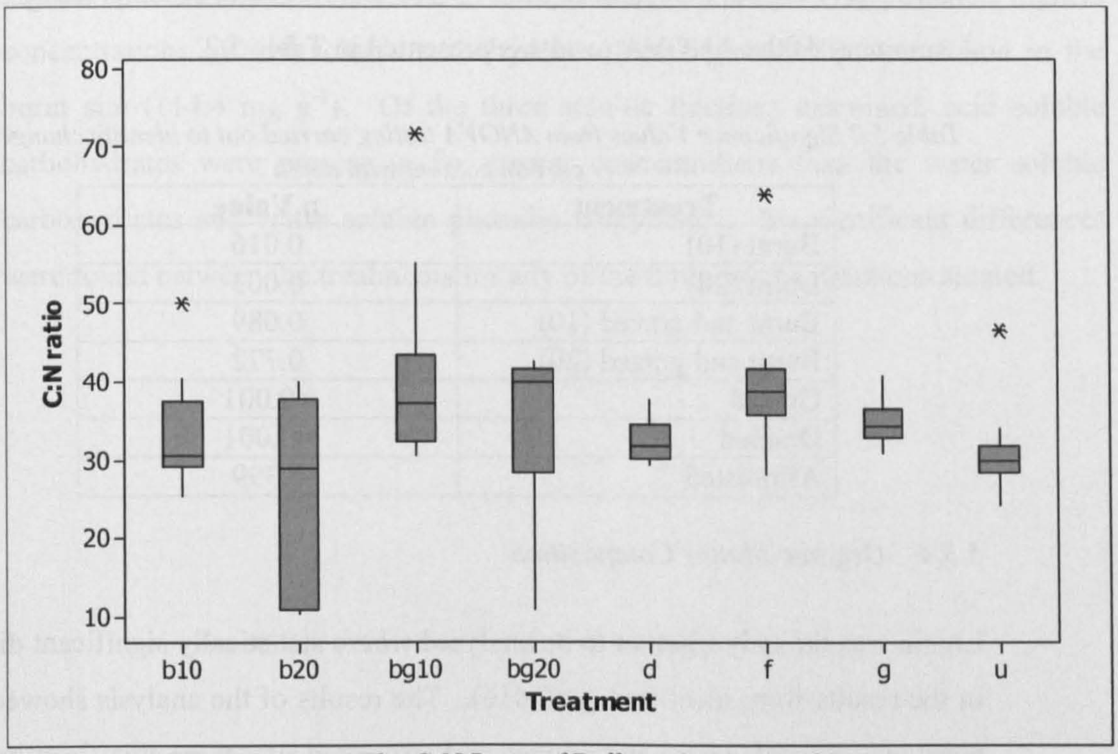


Figure 5-6 The C:N Ratio of Differently Managed Peats

b10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, g – grazed, u – unmanaged, d- drained, f – afforested. The extent of the box represented the first and third quartiles, with the median represented by the line in the centre of the box. The whiskers stretch to the upper and lower limits within the first and third quartiles, values outside of this range are represented with an *

ANOVA testing identified significant differences in the C:N ratio of differently managed peats between 0 and 20 cm beneath the surface. In the 0-10 cm layer, the burnt site (every 20 years) was found to have a significantly lower C:N ratio than the burnt and grazed (every 10 years) site and the afforested site (p=0.001) (Figure 5-6). Within the 10 to 20 cm layer, the burnt site (every 20 years) had a significantly lower C:N ratio than the afforested and unmanaged sites ($p<0.001$). No other significant differences were identified between treatments. The afforested site had greatest variation in values for the surface layer (28.2 – 35.9); the drained site had the smallest (29.3 - 37.7).

The C:N ratio increased with depth at all sites except the burnt and grazed site (every 10 years). The greatest increase was in the burnt site (every 20 years) (16.7), the smallest increase was in the afforested site (2.7). The drained, grazed, burnt (every 10 and 20 years) sites had a significantly higher C:N ratio in the surface peats (0-20 cm) than the base of the profile (40-50 cm). The unmanaged site had a significantly higher C:N ratio in the 0-10 cm than the base of the profile (40-50 cm). A summary of the ANOVA results is presented in Table 5.2.

Table 5.2 Significance Values from ANOVA testing carried out to identify changes in total carbon content with depth

Treatment	p Value
Burnt (10)	0.016
Burnt (20)	0.003
Burnt and grazed (10)	0.089
Burnt and grazed (20)	0.772
Grazed	<0.001
Drained	<0.001
Afforested	0.399

5.3.4 Organic Matter Composition

Lignin was the only fraction to be analysed where statistically significant differences in the results were identified ($p=0.016$). The results of the analysis showed that the burnt and afforested sites and drained and afforested sites were significantly different to one another in terms of lignin content. No other significant differences between treatments were identified, including comparisons with the unmanaged site. Lignin formed the greatest component of each sample analysed, as shown in Figure 5-7. Values for lignin were greatest in the afforested site (1,173.0 mg g^{-1}) and least in the burnt site (747.3 mg g^{-1}).

The smallest of the four main fractions analysed were the soluble fats, oils and waxes (SFOW) and the total soluble fractions. Analysis using the Kruskall-Wallis H test showed that there were no significant differences between the treatments for these two fractions ($p=0.825$ and $p=0.296$ respectively).

Holocellulose values did not differ significantly between any of the treatments examined. Of the four main fractions, hollocellulose was the second most abundant. The greatest range of values was identified for the drained site (279.7 – 810.1 mg g^{-1}). The smallest range of values was identified in the grazed site (322.1 – 609.9 mg g^{-1}).

For the water soluble carbohydrates (see Figure 5-8), the drained site had the highest concentrations (1.03 mg g^{-1}). For the water soluble phenolics (see Figure 5-9), the highest concentrations were found in the unmanaged site (0.04 mg g^{-1}). The highest concentrations of acid soluble carbohydrates (see Figure 5-10) were found in the burnt site (114.4 mg g^{-1}). Of the three soluble fractions examined, acid soluble carbohydrates were present in far greater concentrations than the water soluble carbohydrates and water soluble phenolic compounds. No significant differences were found between the treatments for any of the three soluble fractions studied.

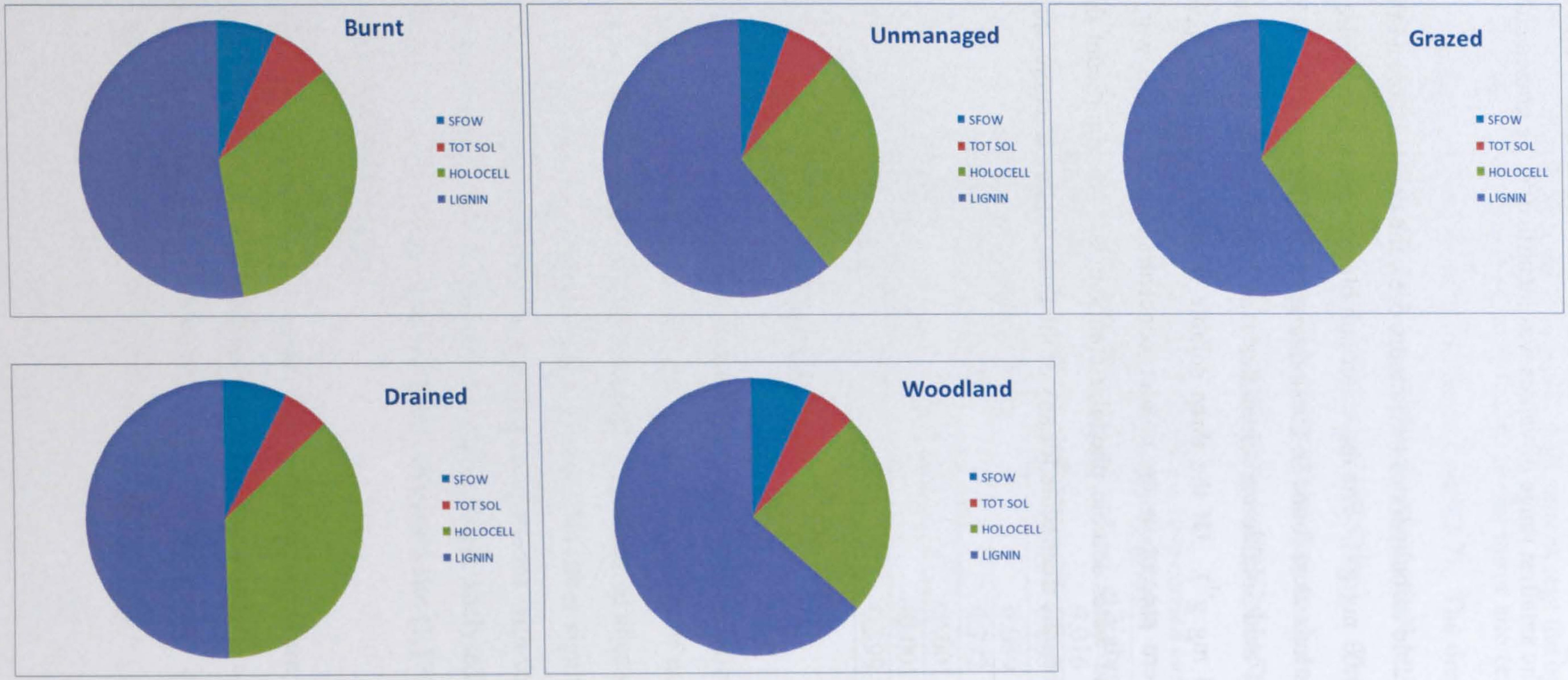


Figure 5-7 Proportion of total organic matter which each fraction comprised
(SFOW = Soluble Fats, Oils and Waxes; Tot Sol – Total Water Soluble Component; Holocell = Holocellulose)

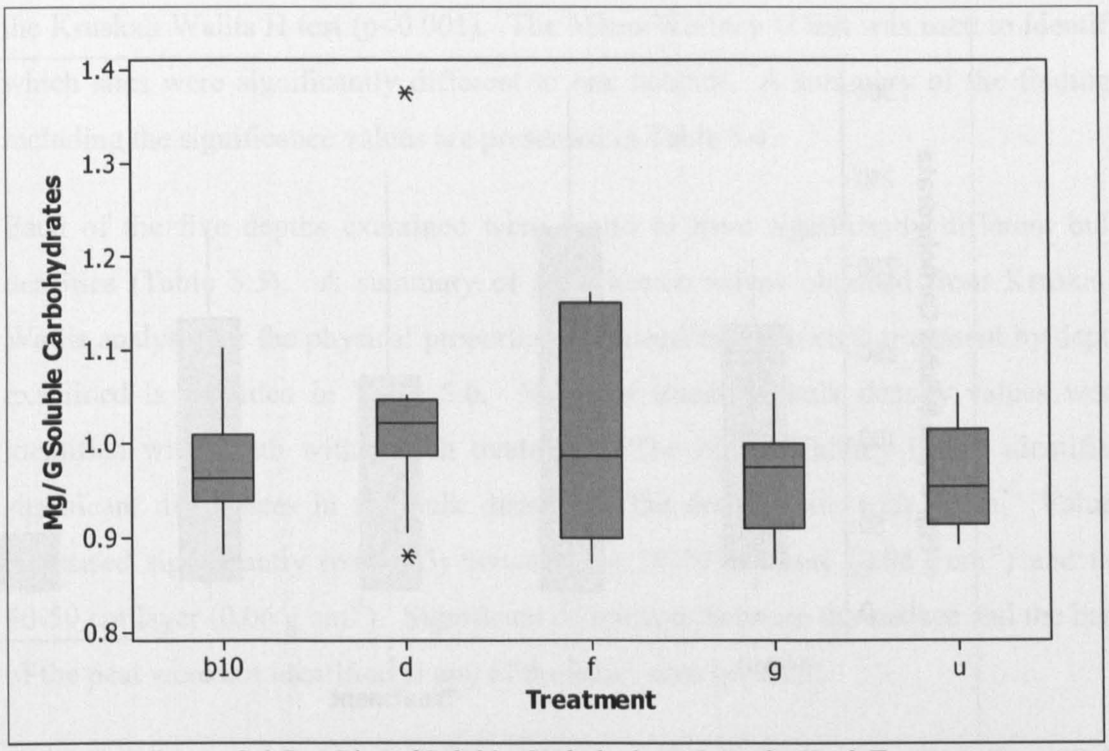


Figure 5-8 Box Plot of Soluble Carbohydrate Data for Each Treatment

b10 – burnt every 10 years (n=9), g – grazed (n=10), u – unmanaged (n=9), d- drained (n=14), f – afforested (n=11). The extent of the box represented the first and third quartiles, with the median represented by the line in the centre of the box. The whiskers stretch to the upper and lower limits within the first and third quartiles, values outside of this range are represented with an *.

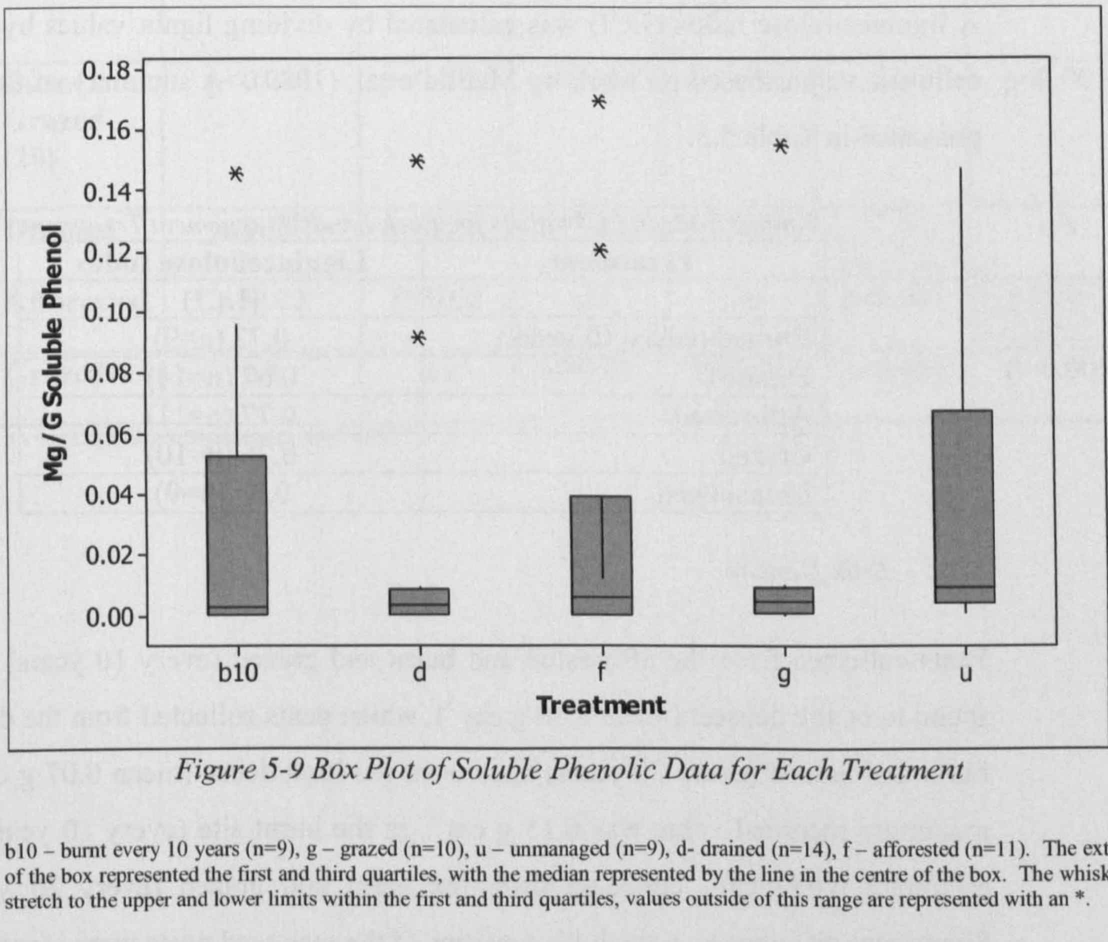


Figure 5-9 Box Plot of Soluble Phenolic Data for Each Treatment

b10 – burnt every 10 years (n=9), g – grazed (n=10), u – unmanaged (n=9), d- drained (n=14), f – afforested (n=11). The extent of the box represented the first and third quartiles, with the median represented by the line in the centre of the box. The whiskers stretch to the upper and lower limits within the first and third quartiles, values outside of this range are represented with an *.

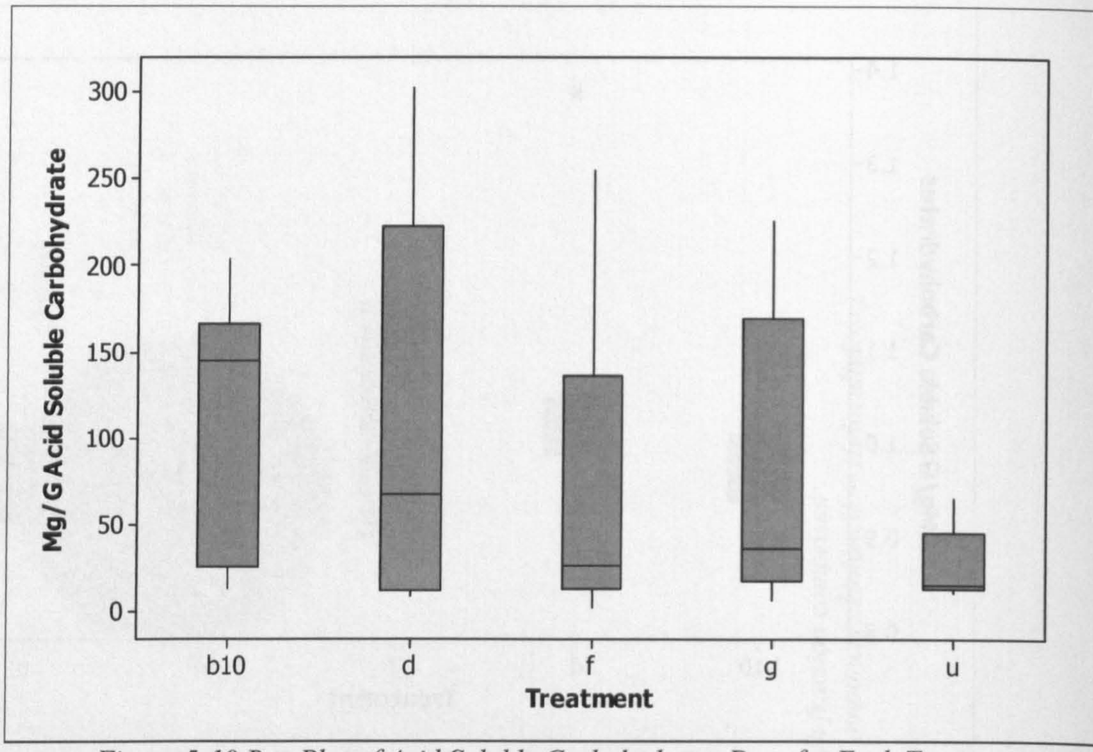


Figure 5-10 Box Plot of Acid Soluble Carbohydrates Data for Each Treatment

b10 – burnt every 10 years (n=9), g – grazed (n=10), u – unmanaged (n=9), d- drained (n=14), f – afforested (n=11). The extent of the box represented the first and third quartiles, with the median represented by the line in the centre of the box. The whiskers stretch to the upper and lower limits within the first and third quartiles, values outside of this range are represented with an *.

A lignincellulose index (LCI) was calculated by dividing lignin values by lignin and cellulose values based on work by Melillo et al. (1989). A summary of the results is presented in Table 5.3.

Table 5.3 Mean LCI Values for Each Land Management Treatment

Treatment	Lignincellulose Index (LCI)
Burned (every 10 years)	0.72 (n=9)
Drained	0.67 (n=14)
Afforested	0.77 (n=11)
Grazed	0.76 (n=10)
Unmanaged	0.77 (n=9)

5.3.5 Bulk Density

Peats collected from the afforested and burnt and grazed (every 10 years) sites were found to be the densest (mean 0.10 g cm^{-3}), whilst peats collected from the drained and burnt and grazed (every 20 years) sites were the least dense (mean 0.07 g cm^{-3}). The maximum recorded value was 0.15 g cm^{-3}, in the burnt site (every 10 years) and the minimum 0.02 g cm^{-3} collected from the burnt and grazed (every 20 years) site. Significant differences in the bulk densities of the managed peats were identified using the Kruskall Wallis H test ($p<0.001$). The Mann-Whitney U test was used to identify which sites were significantly different to one another. A summary of the findings including the significance values are presented in Table 5.4.

Each of the five depths examined were found to have significantly different bulk densities (Table 5.5). A summary of significance values obtained from Kruskall-Wallis analysis for the physical properties examined between each treatment by depth examined is provided in Table 5.6. No clear trends in bulk density values were identified with depth within each treatment. The Mann-Whitney U test identified significant differences in the bulk density of the drained site with depth. Values decreased significantly ($p=0.015$) between the 10-20 cm zone (0.08 g cm^{-3}) and the 40-50 cm layer (0.06 g cm^{-3}). Significant differences between the surface and the base of the peat were not identified at any of the other sites ($p>0.05$).

Table 5.4 Significance Values Indicating which Treatments had a Significantly Different Bulk Density when Compared to other Treatments

	Unmanaged	**Burnt (20)**	**Burnt (10)**	**Burnt and Grazed (10)**	**Drained**	**Afforested**
Burnt and Grazed (20)	p<0.001	p<0.001	p<0.001	p<0.001	n/s	p<0.001
Drained	p<0.001	p<0.001	p<0.001	p<0.001		n/s
Afforested	p=0.02	P=0.02	n/s	n/s	p<0.001	
Grazed	n/s	n/s	p=0.03	P=0.04	p<0.001	P=0.001

n/s – not significant. Significance level <0.05

Table 5.5 Mean and Significance Values for Bulk Density Values for the Different Treatments within Each Layer

Depth	**Burnt and grazed (every 10 years)**	**Unmanaged**	**Burnt (every 20 years)**	**Burnt (every10 years)**	**Burnt and grazed (every20 years)**	**Grazed**	**Drained**	**Afforested**
0-10 cm	0.09 (BG20 p=0.047)	0.09 (BG20 p=0.047)	0.08	0.08	0.06 (bg10, U, G, F p=0.047)	0.08 (BG20 p=0.047)	0.08	0.09 (BG20 p=0.047)
10-20 cm	0.10 (BG20. G and D, p<0.001)	0.09	0.10 BG20. G and D, p<0.001)	0.09	0.07 (B20, BG10, G, F, U p<0.001;)	0.08 (B20 BG10, p<0.001)	0.08 (B20 BG10, BG20, F p<0.001)	0.10 (BG20, D, p<0.001)
20-30 cm	0.10	0.08	0.09	0.10	0.07 (G, F , p<0.001)	0.09 (D, BG20, p<0.001)	0.07 (B20, BG10, B10, G, F, p<0.001)	0.10 (D, BG20, p<0.001)
30-40 cm	0.10 (BG20, D p<0.001)	0.10 (BG20, D p<0.001)	0.09 (BG20, D p<0.001)	0.10 (BG20, D p<0.001)	0.07 (U, B10, B20, BG10, G, F p<0.001)	0.09 (BG20, D p<0.001)	0.06 (U, B10, B20, BG10, G, F p<0.001)	0.10 (BG20, D p<0.001)
40-50 cm	0.09 (D, p<0.001)	0.09 (D, F p<0.001)	0.10 (D, BG20 p<0.001)	0.10 (D, BG20 p<0.001)	0.07 (B10, B20, G p<0.001)	0.10 (D, BG20 p<0.001)	0.07 (BG10, B10, B20, U, F, G, p<0.001)	0.11 (D, U p<0.001)

Values brackets indicate which treatments were significantly different with the significance value. BG10- burnt and grazed (every 10 years), U – unmanaged, B20 – burnt (every 20 years), B10- burnt (every 10 years), G – grazed, D – drained, F – afforested.

Table 5.6 Significance values from Kruskall-Wallis Analysis Indicating Whether Significant Differences Existed between Treatments for Each of the Depths Considered.

	Bulk Density	Particle Density	Total Porosity	Air-filled Porosity
0-10	0.047	n/s	0.016	0.022
10-20	<0.001	n/s	<0.001	<0.001
20-30	<0.001	n/s	<0.001	0.023
30-40	<0.001	n/s	<0.001	0.002
40-50	<0.001	n/s	<0.001	0.164

Significance level – 0.05 n/s – not significant

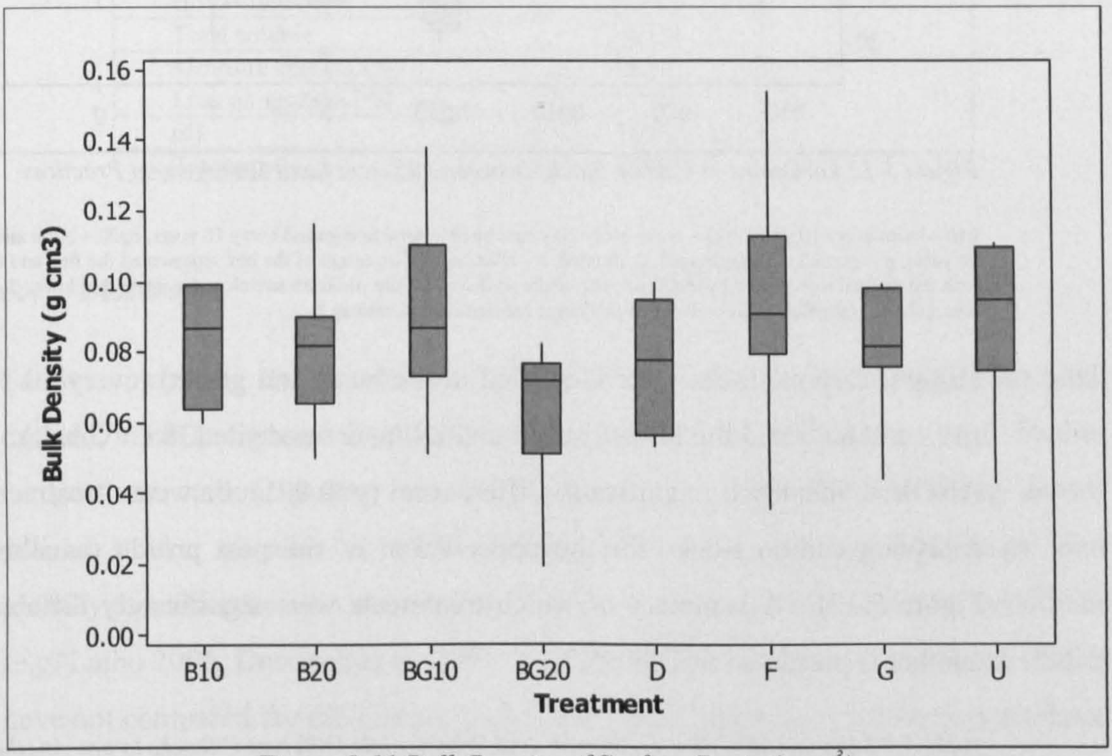


Figure 5-11 Bulk Density of Surface Peats (g cm^3).

b10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, g – grazed, u – unmanaged, d- drained, f – afforested. The extent of the box represented the first and third quartiles, with the median represented by the line in the centre of the box. The whiskers stretch to the upper and lower limits within the first and third quartiles.

5.3.6 Carbon Stocks

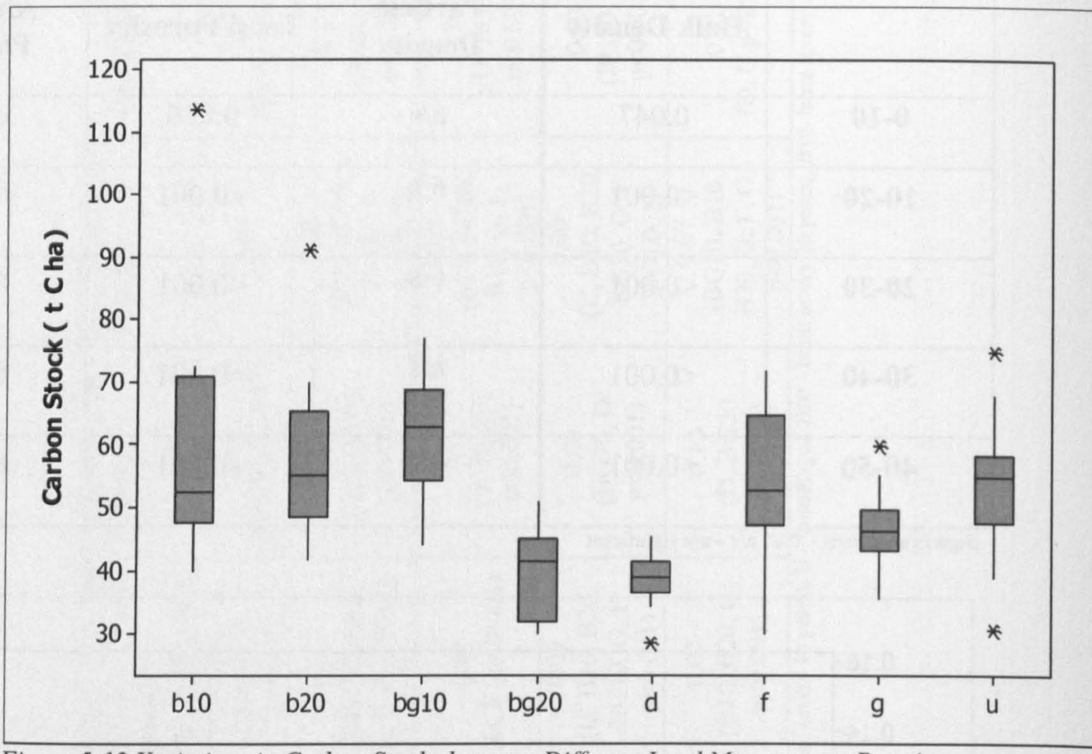


Figure 5-12 Variations in Carbon Stocks between Different Land Management Practices

b10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, g – grazed, u – unmanaged, d- drained, f – afforested. The extent of the box represented the first and third quartiles, with the median represented by the line in the centre of the box. The whiskers stretch to the upper and lower limits within the first and third quartiles, values outside of this range are represented with an *.

Highest carbon stocks were identified in the burnt and grazed (every 10 years) site (61.1 t C ha^{-1}) and the lowest was found in the drained site (38.8 t C ha^{-1}). One way ANOVA identified significant differences ($p<0.001$) between treatments when analysing carbon stocks for the upper 0.5 m of the peat profile (as illustrated in Figure 5-12). A summary of which treatments were significantly different to one another is presented in Table 5.7.

Table 5.7 Summary of Which Treatments had Significantly Different C Stocks to one Another.

	BG10	**U**	**B20**	**B10**	**BG20**	**G**	**D**
U							
B20							
B10							
BG20	✓		✓	✓			
G	✓						
D	✓	✓	✓	✓	✓		
F							✓

✓ - significant difference Blank – no significant difference. B10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, g – grazed, u – unmanaged, d- drained, f – afforested;

Further analysis using one way ANOVA for just the surface samples (i.e. those most likely to have been affected by land management) did not identify any significant differences in the carbon stocks of the differently managed sites (p=0.218). Similarly, ANOVA with co-variance using treatment, the organic fractions and nitrogen as co-variables did not identify significant differences between the treatments, and the absence of significant differences indicated that the absence of differences could be due to confounding factors. A summary of the results is presented in Table 5.8.

Table 5.8 Results of Multiple Way ANOVA with Co-variance to Determine the Effect of Land Management on Carbon Stocks and the Interdependence of Different Drivers on Carbon Stocks

Factor	P value
Lignin:n	0.758
SFOW	0.136
Holocellulose	0.457
Total soluble	0.236
Moisture content (%)	0.269
Loss on ignition (%)	0.876
pH	0.501
Treatment	0.581

5.4 Discussion

Studies comparing the quantity and quality of carbon in peats between afforested, drained, burnt and grazed sites do not appear to have been published to date. Studies on peatland carbon quality have focussed on litter decomposition using above-ground stocks, or rates of decomposition below ground using litter bags, thus identifying differences in decomposition due to varying environmental conditions (e.g. Laiho 2006, Domisch et al. 2000, Yavitt et al. 2005). Furthermore, such studies have not compared the differences in the four land management practices considered here. Differences wee expected to exist in soil carbon stocks and quality between management practices owing to changes in plant community, environmental conditions and nutrient supply which were expected to exist between treatments. These expectations are based on suggestions that peatland management affects nutrient status (e.g. Allen 1964), environmental conditions and substrate quality (e.g. Laiho 2006).

The results presented above did not identify significant differences in the carbon stocks between the different treatments, neither were significant differences in substrate quality found, with the exception of differences in the lignin content. The lack of significant differences in the carbon concentration and stocks and ash content at the base of the profile examined indicate that the peat at this depth has been untouched by land management. It is feasible therefore to suggest that the quality of the carbon at these depths is unlikely to vary between treatments. Peat in the UK uplands forms at an approximate rate of 1 mm a year (Charman 2002). Management practices at Moor House began in earnest during the 1950s, thus it is feasible to suggest that the top 50 to 60 mm represent peat accumulation since the initiation of such management practices. Higher concentrations of carbon were identified in all the burnt treatments within the top 10 cm, however, when the density of the peat was taken into consideration to calculate carbon stocks, the differences were not found to be significant. No significant differences in terms of carbon concentration or carbon stocks were identified between treatments between 10 and 50 cm beneath the surface, suggesting that land management has not had a significant effect on carbon stocks in the surface or at depth within the peat profile.

Work carried out by Melillo et al. (1989) identified two distinctive decomposition stages: the first a steady rate of loss of soluble carbohydrates e.g. celluloses; the second a period of very slow decomposition. Using the LCI, the final stages of decomposition commence when a value of 0.7 is reached. The notion that the samples from each site are recalcitrant is upheld by the results of LCI calculations. The early stages of litter decomposition have passed, which accounts for the small quantities of SFOW and water soluble fractions within each sample which are decomposed first (Berg 2000). The high volume of lignin (a minimum of 50 % of each sample) is also indicative that the peats are in the latter stages of decomposition. Significant differences in the LCI were identified between the drained site and the afforested, grazed and unmanaged sites ($p=0.041$) which all had higher lignin contents than the drained site.

The findings demonstrate that much of the labile fractions have already been decomposed, and only small amounts of these fractions remain. During decomposition, the water soluble fractions are the most rapidly decomposed and quantities typically decrease in the first few months before stabilising to a fairly constant level (Berg 2000). Lignin comprised the highest proportion of the organic matter in each sample, which implies that the carbon is recalcitrant (Reiche et al. 2010), and thus losses of carbon into either the atmosphere or through hydrological pathways are likely to be limited. Alternatively, it is possible that the rate of microbial degradation of labile compounds matched the rate at which these compounds entered the peat.

Data on the composition of litter collected from Moor House published by Heal et al. (1978) suggested that fresh litter has a different composition from that of the peat samples collected in this study. This supports the theory presented above using the LCI that the peat sampled is in the latter stages of decomposition. Heal et al. (1978) found litter samples (*Calluna vulgaris, Eriophorum sp.* and *Sphagnum sp.*) to comprise between 34 %and 69 % of holocellulose, with a mean of 58 % compared a mean of 29.6 % calculated for the peat samples in this study. Mean lignin values for the litter samples were 25.8 % compared to a mean of 57.4 % lignin for the peats in this study. The results give further support to the theory that most of the labile fractions of carbon in the peats have already been decomposed and the carbon is in the latter stages of decomposition.

Work carried out by Gunnarsson et al. (2008) cited high rates of nitrogen deposition as a cause for reduced carbon accumulation. In this study of Moor House, there were no significant differences in the nitrogen content of surface peats among the key sites (as noted in Chapter 4), and nitrogen was not found to be a significant covariatiant during ANOVA between the treatments. Furthermore, the highest carbon stocks were identified in the burnt (every 10 years) site which had the highest nitrogen content as did the unmanaged site, but the latter had lower carbon stocks. The lowest nitrogen stocks were identified in the afforested site which had one of the lowest percentage carbon contents of the five sites examined (47.8 %) in contrast to the work of Gunnarsson et al. (2008).

Previous studies of carbon quality (e.g. Moore et al. 2007, Valentine et al. 1994) have used the lignin:nitrogen ratio to identify labile and recalcitrant peats. Based on the data collected at Moor House, the most recalcitrant carbon was identified in the afforested site (lignin:N = 704.7) and the burnt (every 10 years) site had the least (lignin:N = 490.4), the drained site had the second most labile peats (lignin:N = 552.7). The results imply that imply that while the afforested site contains less carbon than treatments subjected to burning, examined, the carbon within afforested peats is less likely to undergo microbial decomposition, especially given the comparatively lower nitrogen content of the afforested peats.

Work published by Armentano and Menges (1986) suggested that rates of carbon accumulation in peats are lower at more acidic sites, and Bergman et al. (1999) found that carbon synthesis rates are also lower in peats with lower pH values. No significant differences in the pH values of the peats were found between the treatments analysed either in the surface layers or at the base of the profile examined (as noted in Chapter 4), and pH was not found to be a significant covariant during ANOVA. Changes in the acidity of the peats therefore cannot explain differences in carbon stocks and the chemical composition of the peats in this instance.

More rapid plant growth on the burnt sites may account for the greater carbon stocks, as greater inputs of plant material are likely. Higher carbon stocks are also consistent with the higher water table levels recorded at the burnt site. Shallower water tables create a thinner acrotelm and therefore more anaerobic conditions which limit the rate of carbon decomposition. In contrast to this study, Garnett et al. (2000) identified reductions in carbon stocks in burnt peats at Moor House using surrogate measures of carbon, two years after the 1995 burn. This finding is not, however, corroborated by that of Clay et al. (2010b) who identified higher rates of primary productivity on burnt sites, which would result in greater inputs of carbon into the peat[4]. Additionally, Ward et al. (2007) identified higher carbon stocks in burnt peats at Moor House compared to grazed and ungrazed sites.

Dikici and Yilmaz (2006) studied burnt and unburnt peats in Turkey and found less carbon in burnt peats. They attributed the changes in carbon stocks to the time taken to recover from burning as well as volatilisation of carbon during the burning process. Variations in the temperature of the burns between this study and Moor House are likely to be the cause of differences in carbon stocks, as well as differences in the peat types and therefore vegetation between the two studies. Data on the temperature of the burns at Moor House in 2007 were not recorded, but

[4] Data on primary productivity are to be presented in Chapter 6, and demonstrate that higher rates of primary productivity were found at the burnt sites.

anecdotal evidence recorded by the site manager suggested that some of the vegetation was still frozen after the burn, indicating a cool burn (R.Rose, pers. comm.). The fires studied by Dikici and Yilmaz (2006) were described as catastrophic (1965 fire) and large scale (2001 fire) although temperature data were not presented. Between 75 and 100 % of the peat was burnt during the 2001 fire, which the authors suggest must have been caused by temperatures approaching 490°C. Cooler fires have been shown to have less affect on carbon losses, Forgeard and Frenot (1996) found no significant difference in the carbon contents of soils burnt at 150°C and 300°C in a laboratory study. Whilst the maximum temperatures attained during a fire are undoubtedly important, fire intensity, however, depends not only on the temperature at the time of the burn but also on the moisture content of the peat and litter, and whether the fire is planned or accidental wildfire (Legg & Davies 2009).

Farage et al. (2009) suggested the carbon content of burnt peats from Mossdale Moor, Upper Wensleydale is approximately 9.9 kg m^{-2}, which is approximately a fifth of the quantity found in this study and that of Ward et al. (2007) at Moor House. The disparity could indicate that significant differences in the carbon stocks of burnt peats exist between different locations. Further research is required to identify whether this is the case. Farage et al. (2009) did note that their site had been subjected to poor husbandry up until the late 1980s; which might have been a cause for reduced carbon stocks.

Analysis of the effects of wildfires on peatlands carried out in the Peak District by Clay and Worrall (Clay & Worrall 2011) identified higher quantities of carbon in burnt areas compared to non-burnt areas. The findings were attributed to the existence of black carbon remaining in the peat after the fire, which was described as refractory. The absence of significant differences between the burnt and unburnt sites at Moor House could be attributable to the burn at Moor House being cooler than the wildfires studied in the Peak District.

The carbon stocks, C:N ratio and total organic matter content of the grazed site were not significantly different to the unmanaged site. The chemical composition of peats collected from the grazed site was almost identical to those collected from the unmanaged site. These results are consistent with proposals made by Garnett et al.

(2000) that grazing has no effect on carbon stocks in peats. Studies of burning and grazing on peatlands carried out by Ward et al. (2007) did not identify any significant differences in carbon stocks at a depth of 1 m compared to the unmanaged site. A small increase in carbon was recorded in the grazed site compared to ungrazed, however the significance of this difference was not commented on by the authors.

The drained site had the lowest carbon stocks and the lowest proportion of lignin (50 %), suggesting that drainage does not favour carbon preservation. Work carried out by others has suggested that the impacts of peatland drainage on carbon stocks are contradictory. Laiho (2006) reviewed numerous studies of peatland drainage and found evidence that drainage can increase carbon stocks at some sites, whilst decreases or no change were recorded at others (e.g. Laiho et al. 2004a, Minkkinen et al. 1999). The causes of variation were cited as differences in nutrient content, climate, type of bog, vegetation type and consequently substrate quality. Laiho (2006) suggested that oxygen availability, temperature and acidity were the most important controls on litter decomposition. Given the proximity of the Moor House sites to one another, significant differences in local air temperatures are unlikely, with the exception of the afforested site where soil temperatures may have been comparatively lower during summer months and higher during winter months due to the shelter provided by the tree canopy. No significant differences in peatland acidity were identified between the treatments, however, differences in water table levels (presented in Chapters 6 and 7) and therefore potentially oxygen availability were recorded. All the sites studied were ombrotrophic bogs, consequently the differences that have been attributed to bog type elsewhere, are not applicable in this case.

The lower lignin content of the drained site could be attributed either to differences in litter inputs or environmental conditions. A low lignin content indicates higher rates of decomposition which could account for the low carbon content (Turetsky 2004). The drained site had the highest quantities of soluble carbohydrates and acid soluble carbohydrates. Turetsky (2004) found soluble carbon fractions (e.g. soluble carbohydrates) to be have a strong, positive correlation with carbon dioxide losses, which were attributed to the more labile nature of soluble carbohydrates.

The lack of a significant difference in the carbon quantity or quality between the drained and unmanaged sites is consistent with the findings of vegetation surveys at Moor House. Coulson et al. (1990) found no evidence of changes in vegetation composition on drained sites at Moor House and concluded that drainage has little effect on the vegetation species or rates of decomposition at upland sites, where rainfall exceeds 1,200 mm per annum. Based on this evidence it is therefore unlikely that changes in the incoming litter composition on the drained site could account for differences in litter quality compared to the unmanaged site.

Comparisons of different types of drained bog in Sweden carried out by Strakova et al. (2010), however, did reveal differences in litter decomposition (increases in herbaceous species) between the sites in contrast to this study. The differences could be attributed to the water table levels at Moor House differing from the unmanaged site by only 2 to 7 cm, whereas at the Swedish site they were 10 to 15 cm deeper. Strakova et al. (2010) concluded that the changes in above ground litter are highly likely to influence below-ground litter inputs. Given that previous studies have not identified a significant difference in vegetation composition on drained peats, and that the environmental conditions did not vary significantly from the unmanaged site, it is unsurprising that significant differences in carbon stocks were not identified at Moor House.

Differences in litter inputs between the drained and afforested sites could account for the increased lignin content in the afforested site owing to the woody nature of forest litter (Hobbie 1996). The shade offered by the canopy in the afforested site would have reduced the temperature of the peat and hence rates of microbial activity would have decreased (Silvola et al. 1996). The surface layer of the afforested site held the most lignin, which was significantly higher than the burnt and drained sites. The nitrogen content of the afforested site was significantly lower than all other treatments (as discussed in Chapter 4). Differences in the nitrogen content could explain the higher lignin content of the afforested site. The decreased nitrogen content could be attributable to (a) the greater nitrogen demand from the trees; (b) leaching into the ditches; (c) lower inputs of nitrogen from litter and/or (d) reduced inputs from atmospheric deposition due to tree interception. Lower concentrations of nitrogen observed could have limited lignin degradation the afforested site (Berg

2000). Nitrogen is needed by microbes to synthesise carbon, and a high C:N ratio is often associated with low rates of decomposition (Eskelinen et al. 2009).

Of the five sites examined for carbon quality, the highest C:N ratio was identified in the afforested site where the highest lignin content was found. Low C:N ratios are associated with greater amounts of undegraded litter at the end of the decomposition process, which is consistent with the lower carbon stocks identified in the afforested site (Berg & Meentemeyer 2002). The high lignin content of the afforested peats however should serve to prevent further degradation of the peat in the future, whilst peats with more labile fractions will continue to decompose, such as those at the burnt site (Updegraff et al. 1995). Lignin has a more complex molecular structure than labile fractions, which requires a higher activation energy for the substrate to be broken down (Hartley & Ineson 2008). Studies on labile and recalcitrant soil fractions however failed to identify differences in rates of decomposition between the fractions in response to increases in temperature. Moreover, labile and recalcitrant fractions were both found to be sensitive to changes in temperature (Fang et al. 2005).

Under current conditions the afforested peats are unlikely to decompose as rapidly as the other treatments owing to the greater carbon content of afforested peats and their increased recalcitrance. If temperatures rise sufficiently under climate change, however, the most recalcitrant fractions in the forest may begin to degrade (Kirschbaum 2006). Additionally, the afforested site had the lowest moisture content (mean 660 %), indicating that conditions were favourable for the decomposition of labile substances. Holocellulose is the most labile component of organic matter (Yavitt et al. 2005), and the lowest values of holocellulose were identified at the afforested site.

Changes in bulk density due to land management were expected on the grazed, drained and afforested site. The grazed site was expected to have a higher bulk density due to trampling by sheep. No significant differences were identified in the bulk density, however, this could be due to the light grazing intensity at the managed plots. The drained and afforested sites, however, did have significantly different bulk densities compared to the unmanaged site. It was anticipated that the bulk density of both sites would increase owing to the presence of drains. This was found to be the case in the afforested site, which had a significantly higher bulk density than the unmanaged site. This was not the case, however, at the drained site where the bulk density was significantly lower than the unmanaged site. It is possible that the added weight of the trees had caused an increase in the bulk density of the afforested site. There are no clear indications as to why the burnt and grazed (every 20 years) site had a significantly lower bulk density compared to the other sites. The lower bulk density identified at the drained site is consistent with the expectations for dry peats which are reported to have a lower bulk density (Evans 2005)

Significant changes in the bulk density of peats have previously been associated with peatland where *Sphagnum* species are dominant. Due to the fragile structure of the species, it's structure collapses as the water table rises (Clymo 1984). Much of the peat at Moor House is dominated by a combination of sedges, grass and heather which could account for the lack of significant changes in the bulk density within the top 50 cm of the profile.

Significant differences in the physical properties of the peats were expected to be identified with depth, as the transition from the acrotelm to the catotelm is witnessed. The drained site was the only site, however, where a significant change in the bulk density was identified with depth. Mean bulk density values were fairly constant with depth in all other treatments. Only in the burnt (every 10 years), grazed and afforested sites were steady increases in the bulk density identified with depth. The results from the afforested site supported the findings of studies carried out in Finland where the increasing weight of trees resulted in increased compaction of the peat and a rise in bulk density (Minkkinen et al. 1999). Studies carried out by the Forestry Commission in the UK between 1974 and 1981 however found little difference in bulk density with depth (Cannell et al. 1993). The results of bulk density analysis for all sites fell within the range expected for UK deep peatlands of between 0.07 and 0.15 g cm^3 (Cannell et al. 1993).

5.5 Conclusions

5.5.1 Summary of Findings

The aim of this chapter was to identify how land management influences carbon stocks and the carbon quality of peat. To achieve this aim, the following two hypotheses were investigated:

i) Land management has a significant effect on carbon stocks in peat.
ii) Land management has a significant effect on carbon quality.

The results provided a unique assessment of the variations in carbon stocks and quality across differently managed peatland sites and suggested that different management practices applied to peats within one nature reserve have not affected carbon stocks but did influence carbon quality. Carbon stocks were observed to be greatest for the burnt treatments, however litter quality was poorer here than that found in other treatments (excluding the drained site). Carbon stocks were smallest in the drained site, which also had the lowest quantity of lignin and consequently poor litter quality. Afforestation resulted in the most recalcitrant organic matter, with a high C:N ratio, although carbon stocks were lower than those found at the burnt sites. The lignin content of the drained and burnt sites was found to be statistically significantly different to the afforested peats. No site was found to be significantly different from the unmanaged site in terms of carbon quality or carbon stocks. The lignincellulose index identified all peats sampled as being in the latter stages of decay, and therefore rates of decomposition are likely to be low. The labile fractions of the organic matter in each sample analysed are small and provide further evidence that the peat is highly decomposed. It is feasible to suggest however, that the rates at which the more labile materials decompose varied between treatments, according to nutrient supply and environmental conditions.

5.5.2 Further Work

Further examination of carbon quality might identify differences between the treatments, if more detailed analyses were carried out. The proximate analysis used in this study divides the organic matter up into large groups according to their decomposition potential, differences between each of the managed sites could occur at a smaller scale than this. Analysis of samples using pyrolysis, chemolysis and/or nuclear magnetic resonance spectroscopy might be able to identify such differences, if they exist. Although the total carbon content of the peat was not found to vary with depth, Hogg (1993) identified deeper peats as being more recalcitrant, suggesting further examination of peats under different forms of management could provide valuable information on which predictions of the future of peatlands under climate change could be based.

Changes in carbon stocks as a result of land management such as forestry and agriculture have been widely reported by others (e.g. Singh 2008). The intensity of the management practices at Moor House has not been recorded however, making comparisons between treatments problematic. For instance, records indicate that the burn in 2007 was light, but how this compares to burning on other sites is unknown. If we are to truly understand the impact of land management on carbon stocks and quality, studies need to be carried out to look at both ends of the scale in terms of intensity (e.g. comparing severe burns with light burns, heavily grazed sites with lightly grazed sites). In addition, the timescales over which sites have been managed needs to be taken into consideration and comparisons made. Some work on changes in vegetation and peat properties over time since drainage has been carried out in Finland (e.g. Minkkinen et al. 1999), but additional work is needed to examine the other land management practices of interest in the UK.

6 CARBON DIOXIDE GAINS AND LOSSES FROM MANAGED PEATS

6.1 Introduction

Carbon budget calculations for uplands peats in the UK suggest that carbon dioxide losses represent the main pathway through which carbon is lost from peat (e.g. Dinsmore et al. 2010, Worrall et al. 2009). Much effort has been focussed on measuring losses of carbon dioxide from peats that have not been intensively managed, some studies have compared different types of peat, for example, ombrotrophic compared to minerotrophic (e.g. Bubier et al. 1998) whereas others have looked at the effect of micro-topographic features such as hollows and hummocks (Bubier et al. 2003b). To date, limited work has been carried out on managed peats, and no study exists which compares field data for the four main methods of peatland management (burning, grazing, drainage and afforestation) in the UK with a unmanaged site.

The principal aim of this chapter is to analyse the effects of land management on the carbon balance of managed peats focussing on carbon dioxide losses and gains. In addition, the effect of management on the physical properties of the peat that influence gaseous diffusion will be examined. The following hypotheses will be tested:

i) Land management has a significant effect on losses of carbon dioxide from peat
ii) Land management has a significant effect on net ecosystem exchange
iii) Land management has a significant effect on the porosity of the peat, thereby altering the potential for carbon dioxide to diffuse through the peat

The rationale for each hypothesis is presented in Table 6.1. As well as testing the effects of land management on carbon dioxide adsorption and loss from managed peatlands, environmental controls (water table levels and temperature) will also be considered.

Carbon dioxide losses from peatlands occur as a result of microbial and root respiration. As described in Chapter 2, microbes synthesise the organic matter in the peat and release carbon dioxide. Losses of carbon dioxide from both roots and microbial decomposition are collectively termed ecosystem respiration (ER). Rates of ER are controlled by the nature and quantity of substrate available, nutrient availability and environmental conditions. Water table levels, temperature and pH are the dominant environmental controls on ER. ER rates are typically lower than rates of carbon dioxide adsorption by plants, thus resulting in peatlands acting as carbon sinks. Net ecosystem exchange (NEE) represents the balance between carbon dioxide gains and losses within the ecosystem.

NEE and ER are expected to be affected by land management owing to changes in nutrient concentrations in peats, altered environmental conditions and changes in substrate quality (Table 6.1). The porosity of the peat is also expected to be affected by management owing to changes in the bulk density of the peat.

Table 6.1 Expected Effects of Land Management on Carbon Dioxide Gains and Losses and Porosity in Peatlands

	NEE and ER	**Porosity**
Burnt	Burnt peats have been found to become greater carbon sinks owing to increased primary production following burning (Ward et al. 2007), although in some instances, burning has been found to result in a carbon source (Clay et al. 2010b).	The effect of burning on the porosity of the peat is unclear. Some suggest it decreases owing to inputs of ash following burning causing the pores of the peat to become clogged (Mallik et al. 1984b) whilst other have found the porosity of the peat to increase for up to three years post-burning (Mallik & FitzPatrick 1996).
Grazed	Grazed peats have been found to become greater carbon sinks owing to increased primary production due to continual removal of grasses by grazing sheep (Ward et al. 2007), although in some instances, grazing has been found to result in a carbon source (Clay et al. 2010b).	Increases in the bulk density of the peat in relation to sheep grazing have been identified in upland peats (Zhao 2008). The porosity of grazed peats is therefore expected to increase on grazed sites.
Drained	The increased thickness of the acrotelm is expected to result in greater microbial activity and hence grater losses of carbon dioxide. In addition, the phenol oxidase enzyme is known to breakdown phenolic compounds following water table drawdown, and consequently allows rates of litter decomposition to increase (Freeman et al. 2004b). Work on drained sites carried out immediately after blocking by Rowson et al. (2010) identified drained sites as carbon sinks.	The insertion of drains into peats has been associated with a decrease in bulk density owing to structural collapse in the peat (Minkkinen & Laine 1998).
Afforested	Lower water tables owing to drainage and increased water demand by trees are expected to result in greater losses of carbon dioxide compared to unmanaged peatlands (Anderson et al. 2000). The more recalcitrant nature of the substrate (Hobbie 1996), decreased temperatures (Silvola et al. 1996) and lower pH values (Minkkinen et al. 1999) however are expected to compensate for the effects of the lowered water table by reducing rates of microbial activity.	The insertion of drains into peats has been associated with a decrease in bulk density owing to structural collapse in the peat. Furthermore tree planting causes the bulk density to increase even more owing to the weight of the trees (Minkkinen & Laine 1998).

6.2 Methodology

6.2.1 Laboratory Analysis

6.2.1.1 Particle Density

Particle density was measured on 555 air dried samples. Approximately 5 g of sample were taken and placed in a glass beaker with a known quantity of de-gassed, de-ionised water. The sample was stirred and placed on a hotplate for 10 minutes. The solution was cooled in a water bath and the contents transferred to a 100 ml volumetric flask of known weight, ensuring that every particle was transferred. The volume of the flask contents was made up to 100 ml using deionised water. The weight of the flask was recorded. The particle density was calculated using the following equation:

$$\mathrm{Dp} = \frac{\text{mass of soil (g)}}{\text{Volume of soil minus air spaces (cm}^3\text{)}}$$

Where:
Dp = particle density (g cm^3)
Volume of soil = 100-volume of water (cm^3)
Volume of water was calculated as follows:
Volume of water = final flask weight – (soil + initial flask weight)

6.2.1.2 Total Porosity and Air Filled Porosity

Total porosity and air filled porosity were calculated based on the particle density and bulk density values for each sample. Total porosity was calculated using the following equation:

$$\mathrm{St} = 1 - \frac{\mathrm{Db}}{\mathrm{Dp}}$$

Where: St = total porosity (cm cm^{-3}), and Db = bulk density (g cm^{-3})
(Carter & Gregorich 2007)

Air filled porosity was calculated using the following equation:

$$\mathrm{Fa} = \frac{S_t - \theta_w D_b}{D_w} * 100$$

where: Fa = air filled porosity (cm^{-3} cm^{-3}), St = total porosity (cm^{-3} cm^{-3}), θw = gravimetric moisture content ($g\ g^{-1}$), Db = bulk density ($g\ cm^{-3}$) and Dw = water density (Carter & Gregorich 2007)

6.2.2 Field Monitoring

6.2.2.1 Measurement of Carbon Dioxide Loss and Net Ecosystem Exchange

ER was monitored on 12 occasions and NEE on 19 occasions. Details of the dates during which each monitoring round was carried out are provided in Table 6.2. the dates chosen allowed a combination of both winter and summer monitoring to be carried out. More emphasis was placed on monitoring during a warmer period as microbial activity was expected to be greater during warmer periods, therefore, an intensive period of monitoring was carried out in August 2010.

Table 6.2 Dates and Climate Data for Each Monitoring Round

Monitoring Round	Date	Rainfall during Monitoring Day (mm)	Maximum Temperature on Monitoring Day (°C)	Average Temperature on Monitoring Day (°C)	ER Measured	NEE Measured
1	10/03/09	4.0	4.6	2.7	✓	✓
2	10/05/09	0.0	3.4	1.8		✓
3	19/05/09	1.4	8.4	5.0		✓
4	02/06/09	0.0	4.7	1.9		✓
5	16/06/09	0.0	4.4	1.9		✓
6	02/07/09	2.0	6.7	3.5		✓
7	17/07/09	68.0	9.5	6.0		✓
8	29/07/09	10.8	6.9	4.4		✓
9	07/10/09	0.0	6.1	3.3	✓	✓
10	23/10/09	4.0	3.6	2.6	✓	✓
11	16/11/09	26.2	9.1	6.0	✓	✓
12	07/12/09	32.6	8.7	6.2	✓	✓
13	12/08/10	4.0	14.6	10.5	✓	✓
14	13/08/10	14.5	11.1	9.5	✓	✓
15	14/08/10	1.0	13.6	10.5	✓	✓
16	15/08/10	0.0	20.2	13.4	✓	✓
17	16/08/10	0.5	17.2	12.1	✓	✓
18	17/08/10	3.5	15.4	11.6	✓	✓
19	18/08/10	2.0	12.7	9.8	✓	✓

ER and NEE fluxes were monitored using an Environmental Gas Monitor (EGM4 from PP Systems, Hitchin, Hertfordshire, Figure 6-1). The soil respiration chamber of the EGM4 (PP Systems' CPY2 canopy assimilation chamber) was fitted into a plastic 15 cm diameter collar (inserted to a depth of 5 cm into the peat) and a latex band placed around the outside of the chamber and inside of the collar to ensure that a gas-tight seal was obtained. The chamber was fitted with a fan which allowed air to circulate within the chamber. Monitoring was carried out for 124 seconds, with gas measurements being recorded by the instrument's infra-red gas analyser (IRGA) every

4 seconds to allow a flux to be calculated. NEE was measured as the total of ER and primary productivity (PP). Following completion of each NEE measurement, ER was recorded by placing a dark cover over the CPY2 chamber and using the IRGA to record the flux again. Details of the collar locations are presented in Table 6.3.

Figure 6-1 EGM4 Attached to the CPY2 Chamber Fitted with a Rubber Seal, Located Adjacent to a Monitoring Collar.

Table 6.3 Locations of the Gas Monitoring Collars

Collar Number	Site
1-3	Grazed and burnt (every 10 years)
4-6	Grazed and burnt (every 20 years)
7-9	Grazed
10-12	Burnt (every 20 years)
13-15	Burnt (every 10 years)
16-18	Unmanaged
19-21	Drained
22-24	Afforested

6.2.2.2 Groundwater Levels

Groundwater levels were measured in the field during the site visits listed in Table 6.2. Measurements were made by inserting a dip-meter into each groundwater monitoring well adjacent to the gas monitoring collar as detailed in Chapter 3.

6.2.2.3 Weather Data

Data collected from the Moor House Automatic Weather Station (AWS) were provided by the Environmental Change Network (ECN) for the time period over which monitoring work was carried out. The AWS is situated at 54.690° N, 2.375°W and is located 556m AOD. Data from the AWS are downloaded weekly by the ECN and subject to quality control checks prior to release.

6.2.3 Statistical Analysis

Data from environmental monitoring were compared using Analysis of Variance (ANOVA) and subsequently Tukey's test was performed to identify where differences occurred. Results from analysis of the physical properties of the peat were compared using the Kruskall-Wallis H test and Mann-Whitney U test as the data did not have a normal distribution.

6.3 Results

6.3.1 Ecosystem Respiration

No significant differences in ER were found between the managed sites ($p=0.359$). Significant differences were, however, identified between collar 9 on the grazed site and collars: 2 (burnt and grazed every 10 years), 4 and 6 (burnt and grazed every 20 years), 13 (burnt every 10 years) and 18 (unmanaged). The average values for all these locations (except collar 13) were found to be higher than the mean value for collar 9 (0.032 g CO_2 m^{-2} h^{-1}). As shown on Figure 6-2, no clear trends in data were evident to indicate which site lost the most carbon dioxide and which lost the least.

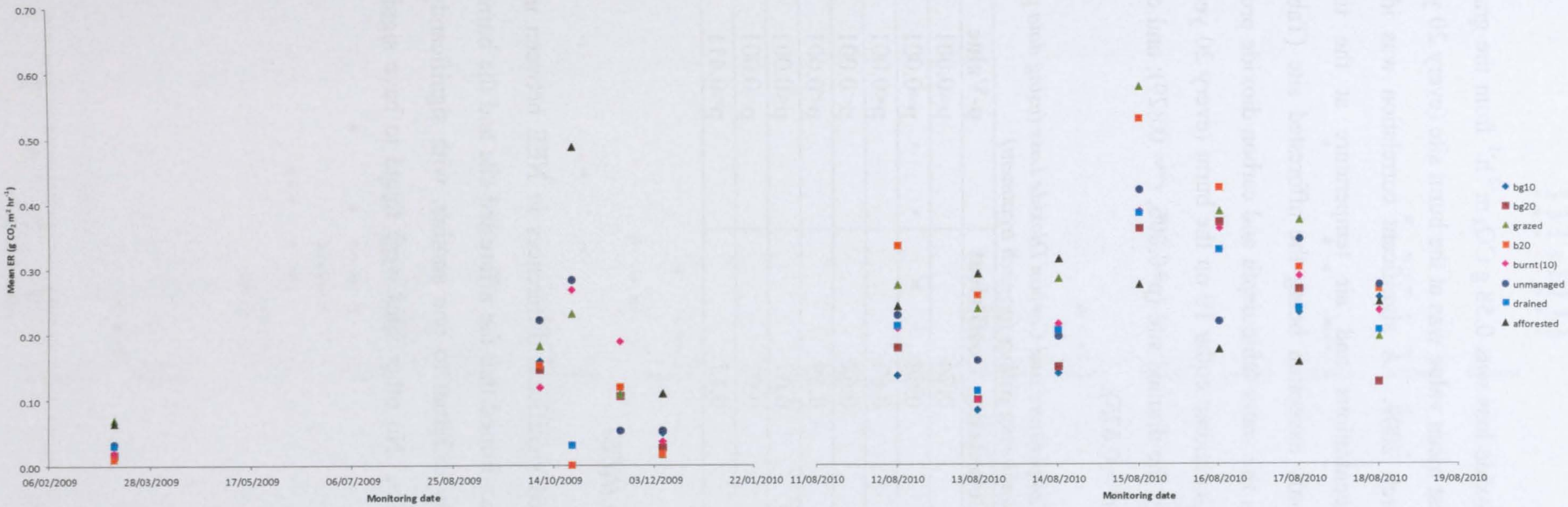


Figure 6-2 Mean Ecosystem Respiration for the Managed and Unmanaged Sites

BG10 – burnt (every 10 years) and grazed; BG20 – burnt (every 20 years) and grazed, B20 – burnt every 20 years. Each data point represents the mean of 3 recorded values.

The maximum mean carbon dioxide loss was 0.58 g CO_2 m^{-2} h^{-1} from the grazed site on 15th August 2010. The lowest mean value was at the burnt site (every 20 years) of 0.01 g CO_2 m^{-2} h^{-1} on 10th March 2009. A significant correlation was identified between carbon dioxide concentrations and air temperature at the time the measurement was taken, the only exception being the afforested site (Table 6.4). Significant correlations between the water table depth and carbon dioxide production were only found for following locations: collar 10 on the burnt (every 20 years) site ($p=0.033$ $r^2 = 0.64$); collar 21 on the drained site ($p=0.006$, $r^2 = 0.829$); and collar 23 on the afforested site ($p=0.032$, $r^2 = 0.675$).

Table 6.4 Correlation between Temperature and Carbon Dioxide Loss (using data pooled from the 3 monitoring collars for each treatment)

Site	**Correlation Coefficient**	**p-Value**
Burnt and grazed (10)	0.74	p<0.001
Unmanaged	0.54	p=0.001
Burnt (20)	0.83	p<0.001
Burnt (10)	0.62	p<0.001
Burnt and grazed (20)	0.59	p<0.001
Grazed	0.67	p<0.001
Drained	0.92	p<0.001
Afforested	0.17	p=0.411

6.3.2 Net Ecosystem Exchange (NEE)

Results from ANOVA identified significant differences in NEE between managed peats ($p=0.023$). Tukey's test confirmed that the afforested site and the burnt (every 10 years) site were significantly different to one another, with significantly lower values produced at the burnt site. No other sites were found to have significantly different NEE from one another.

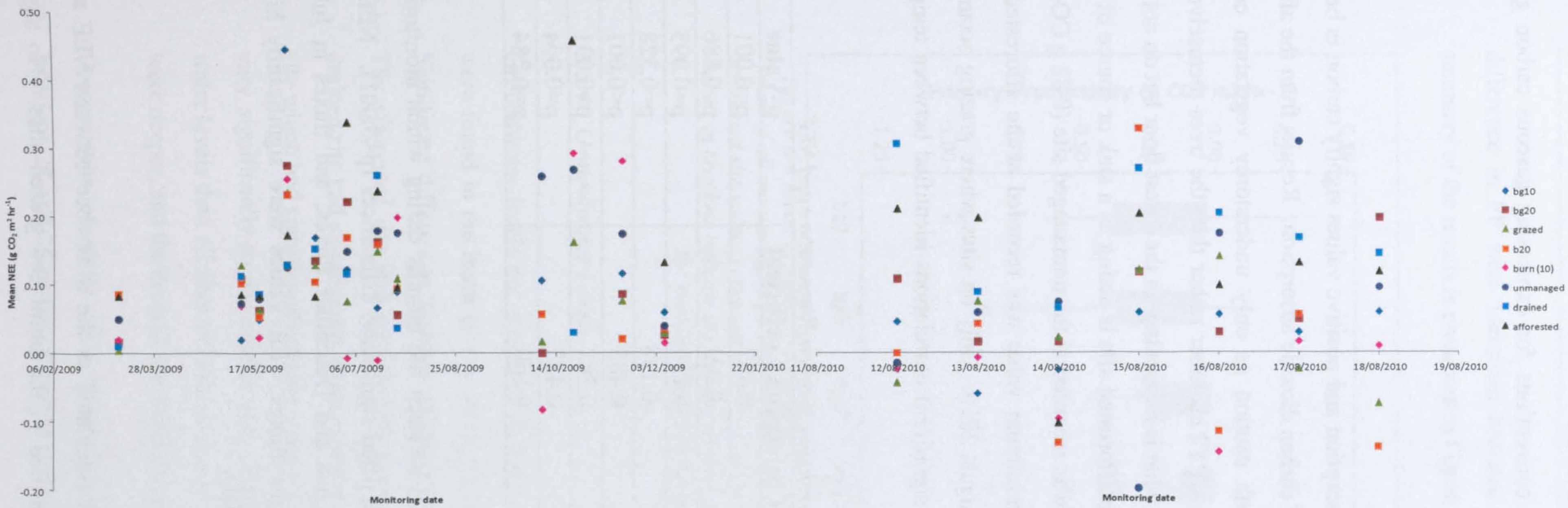


Figure 6-3 Mean NEE for the Managed and Unmanaged Sites (note measurements for the afforested site did not include the tree canopy)

BG10 – burnt (every 10 years) and grazed; BG20 – burnt (every 20 years) and grazed, B20 – burnt every 20 years. Each data point represents the mean of 3 recorded values.

In line with commonly used conventions for expressing gaseous carbon gain and losses, negative values (

Figure *6-3*) indicate carbon adsorption and positive values signify carbon is being lost as respiration outstrips rates of carbon dioxide adsorption. Results from the afforested site should be interpreted with caution as only understorey vegetation could be monitored within the EGM4's CPY2 chamber rather than the trees themselves. The results give an indication as to what is happening on the forest floor but do not give an accurate picture of whether the afforested site is acting as a sink or source of carbon. The maximum value for NEE was recorded at the unmanaged site (0.82 g CO_2 m^{-2} hr^{-1}) on 17th August 2010; the minimum value was recorded at the afforested site (-1.01 g CO_2 m^{-2} hr^{-1}) on 14th August 2010. Only on sites where grazing occurred and at the drained site were there significant correlations identified between temperature and NEE (Table 6.5).

Table 6.5 Correlation between Temperature and NEE

Site	Correlation Coefficient	p-Value
Burnt and grazed (10)	0.457	p=0.001
Unmanaged	-0.021	p=0.886
Burnt (20)	0.234	p=0.095
Burnt (10)	-0.138	p=0.328
Burnt and grazed (20)	0.485	p<0.001
Grazed	0.288	p=0.031
Drained	0.427	p=0.004
Afforested	0.160	p=0.284

Analysis of differences in NEE between the months during which monitoring was carried out at Moor House identified significant differences ($p<0.001$). NEE fluxes were higher in June than August and December fluxes, and fluxes in July were significantly higher than August fluxes. ER values were significantly higher in October than December ($p<0.001$).

6.3.3 *Primary Productivity*

Primary productivity (PP) was calculated as the difference between NEE and ER. Greatest productivity was identified in the burnt and grazed sites. No significant difference in PP were identified between the different treatments (p=0.123). A summary of the results is presented in Figure 6-4.

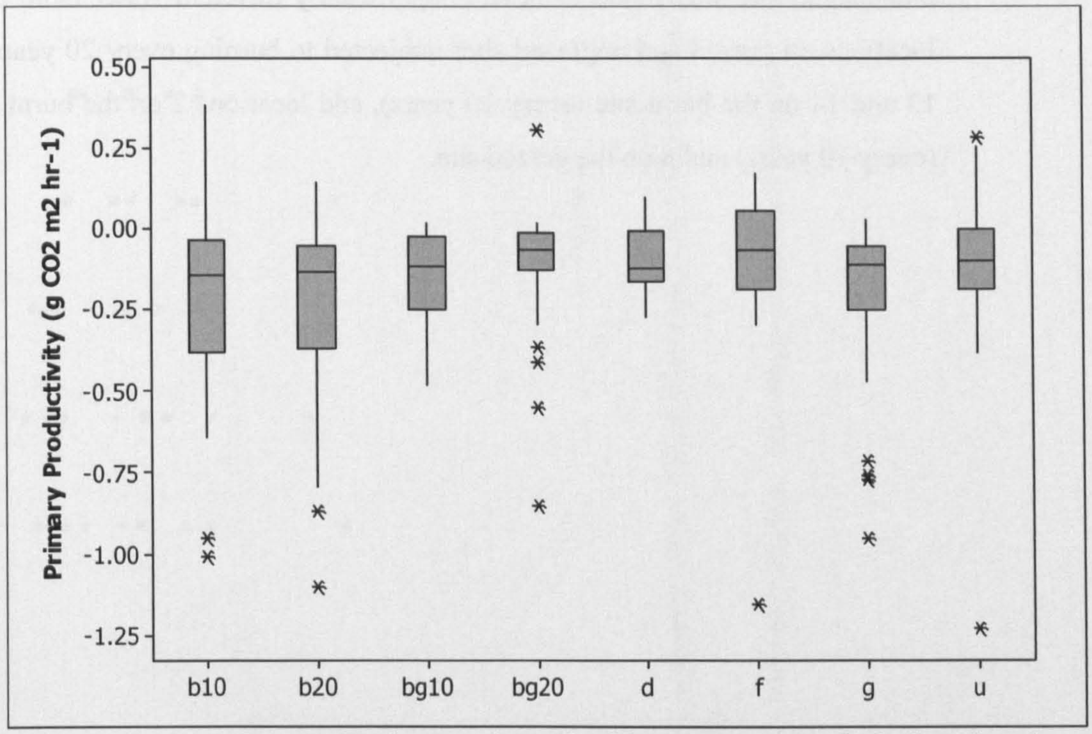


Figure 6-4 Mean Primary Productivity for each of the Treatments Studied, - based on the mean of all monitoring trips where ecosystem respiration was measured. The values for the afforested site values represent the PP of understorey vegetation and do not include the trees. Values in brackets indicate the number of years between managed burns. Error bars indicate the standard deviation of measurements at each site.

6.3.4 Groundwater Levels

Groundwater levels are presented in Figure 6-5. On average the shallowest levels were found in the burnt (every 10 years) site and the deepest at the afforested site. Significant differences between treatments were analysed using ANOVA ($p<0.001$). The afforested site had significantly lower water levels than all other sites. The drained site had significantly lower water levels than all sites except for the afforested site which had significantly deeper water levels, and the unmanaged site which did not vary significantly to the drained site. The unmanaged site had significantly deeper water levels than all sites with the exception of the afforested site, where water levels were deeper, and the drained site, which was not significantly different.

Comparisons of water table levels between each monitoring location identified locations 22 and 24 in the afforested site as having significantly deeper water tables than all other locations with the exception of locations 16 and 17 on the unmanaged site which were only significantly different to location 24. Location 17 on the unmanaged site was found to have a significantly different water table depth to all locations on grazed and ungrazed sites subjected to burning every 20 years, locations 13 and 14 on the burnt site (every 10 years), and locations 2 on the burnt and grazed (every 10 years) and 8 on the grazed site.

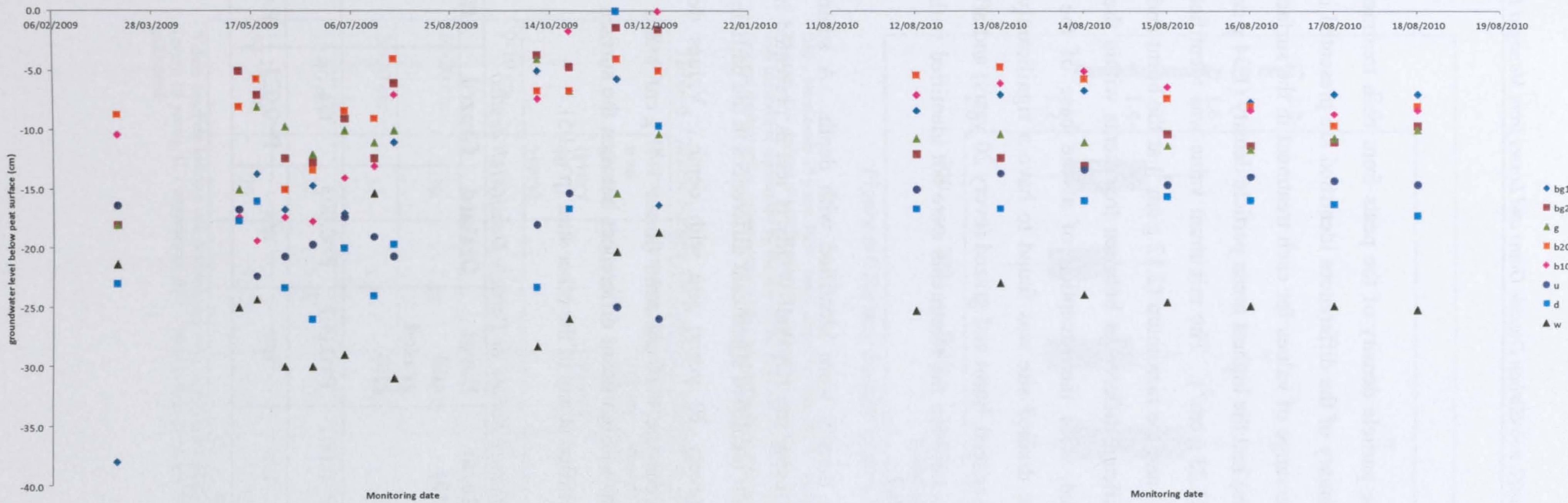


Figure 6-5 Variation in Mean groundwater levels between the different management practices. Mean Values Were Calculated from the Three Wells Installed on Each Site

BG10 – burnt (every 10 years) and grazed; BG20 – burnt (every 20 years) and grazed, B20 – burnt every 20 years. Each data point represents the mean of 3 recorded values.

6.3.5 Particle Density

Significant differences in the particle density of the peats from each treatment were identified (p=0.002). A summary of the differences identified are presented in Table 6.6, Figure 6-6 illustrates the range of values for each treatment in the surface peats. The burnt site (every 10 years) had the highest mean particle density (1.34 g cm^{-3}) and the unmanaged the lowest (1.23 g cm^{-3}). The minimum value was identified at the unmanaged site (0.85 g cm^{-3}) and the maximum (2.12 g cm^{-3}) at the burnt and grazed site (every 20 years). Significant differences between treatments within the depths analysed were not identified, with the exception of at the base of the profile investigated (p=0.014). The drained site was found to have a significantly higher particle density than the unmanaged, burnt and grazed (every 20 years) and afforested sites. Significant differences between the treatments were not identified within each layer, as illustrated in Table 6.7.

No clear trends in particle density were identified with depth. A summary of significance-values obtained using the Kruskall-Wallis H test is presented in Table 5.4. The Mann-Whitney U test identified significant differences in the particle density of the burnt and grazed (every 20 years) site with depth. Values decreased significantly (p=0.017) between the 0-10 cm zone (mean 1.41 g cm^{-3}) and the 40-50 cm layer (mean 1.19 g cm^{-3}). Significant differences between the surface and the base of the peat were not identified at any of the other sites (p>0.05).

Table 6.6 Significance Values for Particle Density (all depths)

	Burnt and grazed (10)	**Burnt (10)**	**Burnt and grazed (20)**	**Drained**	**Grazed**	**Afforested**
Unmanaged	P=0.002	P<0.001	P=0.007	P=0.003	n/s	n/s
Burnt (10)	n/s	n/s	n/s	n/s	P=0.021	P=0.028

n/s – not significant <0.05=significance level

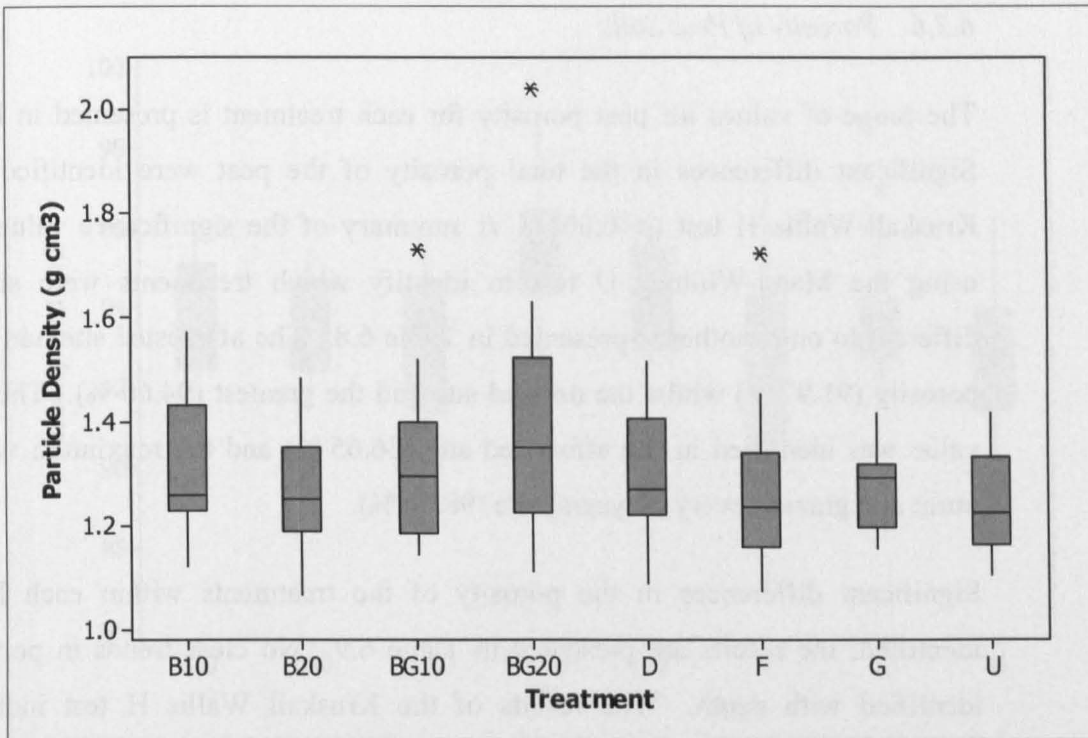


Figure 6-6 Particle Density of Acrotelm (0-10 cm) Peats (g cm^3).

B10 – burnt every 10 years, B20 – burnt every 20 years, BG10 – burnt and grazed every 20 years, BG20 burnt and grazed every 20 years, D – drained, F – afforested, G – grazed, U- unmanaged. The extent of the box represents the first and third quartiles, with the median represented by the line in the centre of the box. The whiskers stretch to the upper and lower limits within the first and third quartiles, values outside of this range are represented with an asterisk.

Table 6.7 Mean Values for Particle Density for the Different Treatments within Each Layer

Depth	Burnt and grazed (every 10 years)	Unmanaged	Burnt (every 20 years)	Burnt (every10 years)	Burnt and grazed (every20 years)	Grazed	Drained	Afforested
0-10 cm	1.32	1.25	1.26	1.31	1.41	1.28	1.30	1.27
10-20 cm	1.30	1.24	1.29	1.31	1.28	1.29	1.30	1.36
20-30 cm	1.34	1.25	1.31	1.32	1.31	1.24	1.30	1.27
30-40 cm	1.27	1.22	1.25	1.46	1.43	1.26	1.25	1.27
40-50 cm	1.29	1.17	1.26	1.25	1.19	1.28	1.30	1.21

Values brackets indicate which treatments were significantly different with the significance value. BG10- burnt and grazed (every 10 years), U – unmanaged, B20 – burnt (every 20 years), B10- burnt (every 10 years), G – grazed, D – drained, F – afforested.

6.3.6 Porosity of Peat Soils

The range of values for peat porosity for each treatment is presented in Figure 6-7. Significant differences in the total porosity of the peat were identified using the Kruskall-Wallis H test ($p<0.001$). A summary of the significance values obtained using the Mann-Whitney U test to identify which treatments were significantly different to one another is presented in Table 6.8. The afforested site had the lowest porosity (91.97 %) whilst the drained site had the greatest (94.60 %). The minimum value was identified in the afforested site (86.65 %) and the maximum value in the burnt and grazed (every 20 years) site (98.52 %).

Significant differences in the porosity of the treatments within each layer were identified; the results are presented in Table 6.9. No clear trends in porosity were identified with depth. The results of the Kruskall Wallis H test indicated that significant differences in porosity exist with depth ($p<0.025$). Statistical analysis using the Mann-Whitney U test identified the 10-20 cm layer as having a significantly lower porosity than the 20-30 cm ($p=0.031$), 30-40 cm ($p=0.003$) and 40-50 cm ($p=0.017$) layers.

Table 6.8 Significance Values for Total Porosity

	Burnt and grazed (10)	**Burnt (10)**	**Burnt (20)**	**Grazed**	**Afforested**	**Unmanaged**
Burnt and grazed (20)	0.001	0.001	0.001	0.001	0.001	0.001
Drained	<0.001	<0.001	0.001	0.001	0.001	0.001
Afforested	0.05	0.05	0.01	0.001		0.05

n/s – not significant <0.05=significance level

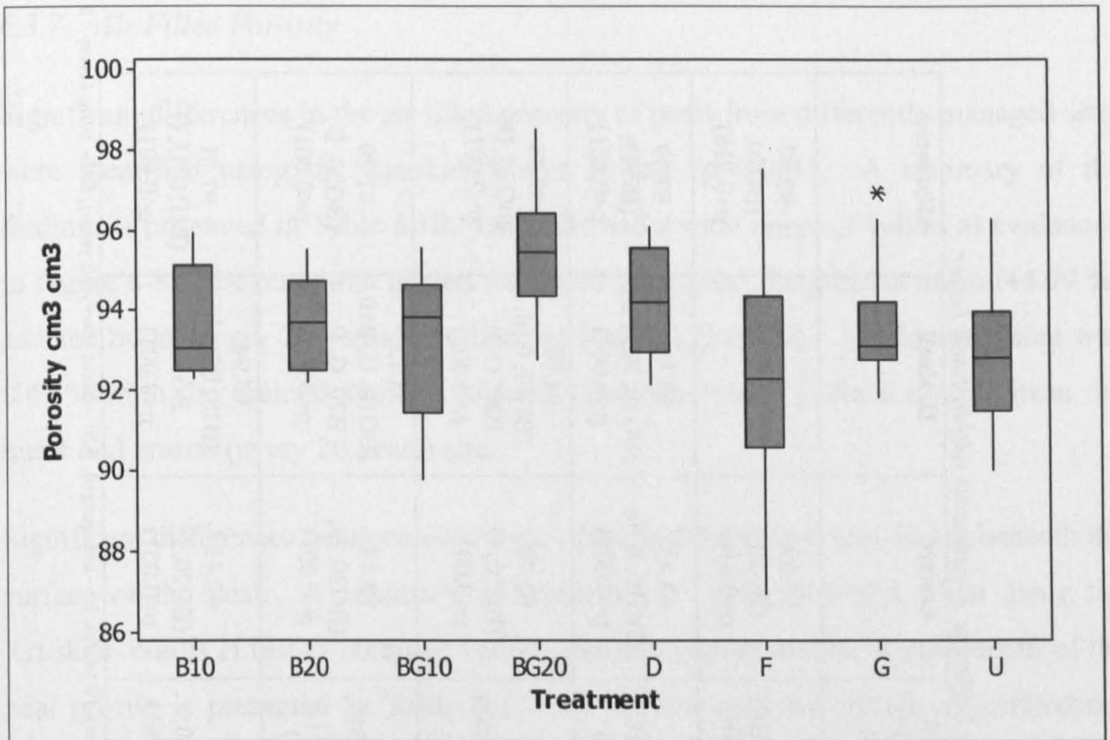


Figure 6-7 Total Porosity of Surface Peats (cm^3 cm^3).

B10 – burnt every 10 years, B20 – burnt every 20 years, BG10 – burnt and grazed every 20 years, BG20 burnt and grazed every 20 years, D – drained, F – afforested, G – grazed, U- unmanaged. The extent of the box represents the first and third quartiles, with the median represented by the line in the centre of the box. The whiskers stretch to the upper and lower limits within the first and third quartiles, values outside of this range are represented with an asterisk.

Table 6.9 Mean Values and Significant Differences in Porosity Values for the Different Treatments within Each Layer

Depth	Burnt and grazed (every 10 years)	Unmanaged	Burnt (every 20 years)	Burnt (every 10 years)	Burnt and grazed (every 20 years)	Grazed	Drained	Afforested
0-10 cm	93.17 (BG20, p=0.016)	92.86 (BG20, p=0.016)	93.64 (BG20, p=0.016)	93.73	95.42 U, B20, BG10, G, F p=0.016	93.69 (BG20, p=0.016)	94.19	92.47 (BG20, p=0.016)
10-20 cm	91.95 (BG20, G, D; p<0.001)	92.37 (BG20, D; P<0.001)	92.59 (BG20, G, D; p<0.001)	93.10	94.46 (BG10, U, B20 p<0.001)	93.63 (BG10, B20; p<0.001)	93.82 (BG10, B20, U; p<0.001)	92.39 (BG20; p<0.001)
20-30 cm	92.83 (D; p<0.001)	93.48	93.24 (D; p<0.001)	92.49 (D; p<0.001)	94.43 (G, F; p<0.001)	92.70 (BG20, D; p<0.001)	94.87 (BG10, B10, B20, G, F; p<0.001)	92.13 (BG20, D; p<0.001)
30-40 cm	92.08 (BG20, D; p<0.001)	92.10 (BG20, D; p<0.001)	92.59 (BG20, D; p<0.001)	92.92 (BG20, D; p<0.001)	95.11 (BG10, U, B10, B20, G, F; p<0.001)	93.14 (BG20, D; p<0.001)	95.12 (BG10, U, B10, B20, G, F; p<0.001)	91.89 (BG20, D; p<0.001)
40-50 cm	92.94 (D, F; p<0.001)	92.68 (F, D; p<0.001)	92.37 (BG20, D; p<0.001)	92.23 (BG20, D; p<0.001)	94.36 (G, F, B10, B20; p<0.001)	92.41 (BG20, F; p<0.001)	94.94 (B10, B20, U, BG10; p<0.001)	91.01 (BG10, U, G ; p<0.001)

Values in brackets indicate which treatments were significantly different with the significance value. BG10- burnt and grazed (every 10 years), U – unmanaged, B20 – burnt (every 20 years), B10- burnt (every 10 years), G – grazed, D – drained, F – afforested.

6.3.7 Air Filled Porosity

Significant differences in the air filled porosity of peats from differently managed sites were identified using the Kruskall-Wallis H test ($p<0.001$). A summary of the findings is presented in Table 6.10. Each site had a wide range of values as evidenced in Figure 6-8. The burnt and grazed (every 20 years) had the greatest mean (44.99 %) and the burnt (every 20 years) site had the lowest (25.68 %). The lowest value was 0.47 % from the drained site, the highest value was 86.92 % for a sample from the burnt and grazed (every 20 years) site.

Significant differences between sites were identified between 0 and 40 cm beneath the surface of the peat. A summary of significance values obtained when using the Kruskall-Wallis H test to compare values between each treatment at each depth of the peat profile is presented in Table 6.11. A summary of the significant differences between treatments is presented in Table 6.10.

The air filled porosity values for each treatment decreased with depth. Significant differences between the surface and base of the profiles examined were found for the burnt and grazed (every 10 years) site ($p=0.018$) and the grazed ($p=0.013$) site.

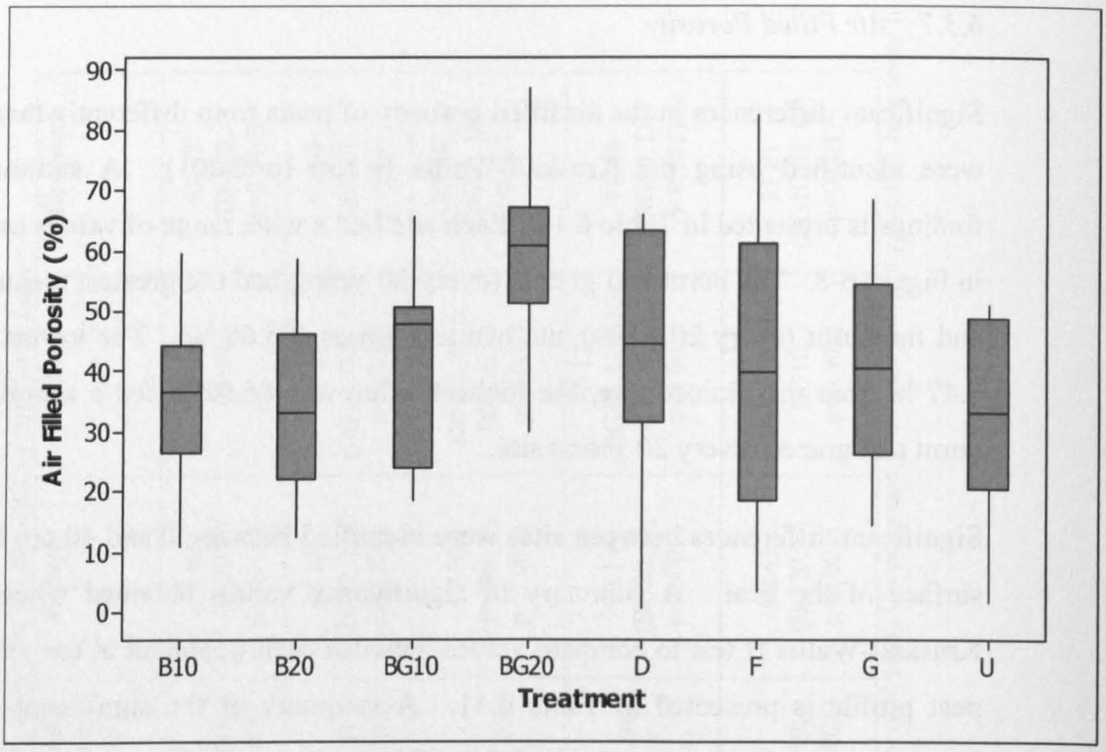


Figure 6-8 Air Filled Porosity of Surface Peats (%).

B10 – burnt every 10 years, B20 – burnt every 20 years, BG10 – burnt and grazed every 20 years, BG20 burnt and grazed every 20 years, D – drained, F – afforested, G – grazed, U- unmanaged. The extent of the box represents the first and third quartiles, with the median represented by the line in the centre of the box. The whiskers stretch to the upper and lower limits within the first and third quartiles, values outside of this range are represented with an asterisk.

Table 6.10 Significance Values for Air filled Porosity

	Burnt and grazed (10)	Burnt (10)	Burnt (20)	Grazed	Afforested	Unmanaged
Burnt and grazed (20)	p<0.001	p<0.001	p<0.001	p<0.001	p<0.001	n/s
Drained	p<0.001	p<0.004	p<0.001	p<0.001	p<0.001	p<0.001

n/s – not significant <0.05=significance level

Table 6.11 Mean and Significance Values in Air-filled Porosity Values for the Different Treatments within each Layer

Depth	Burnt and grazed (every 10 years)	Unmanaged	Burnt (every 20 years)	Burnt (every10 years)	Burnt and grazed (every20 years)	Grazed	Drained	Afforested
0-10 cm	40.67 (BG20; p=0.022)	32.48 (BG20; p=0.022)	33.38 (BG20; p=0.022)	35.68 (BG20; p=0.022)	59.30 (U, B10, B20, BG10; p=0.022)	41.55	43.11	42.59
10-20 cm	20.88 (BG20, G, D; p<0.001)	27.36	24.26 (BG20, G, D; p<0.001)	35.49	44.53 (BG10; p<0.001)	36.50 (BG10; p<0.001)	34.33 (BG10; p<0.001)	33.52
20-30 cm	24.06 (D; p=0.023)	33.25	27.34 (D; p=0.023)	23.83 (D; p=0.023)	37.07	24.11 (D; p=0.023)	41.99 (BG10, B10, B20, G, F; p<0.023)	28.80 (D; p=0.023)
30-40 cm	19.16 (D, BG20; p=0.002)	20.49 (D, BG20; p=0.002)	20.93 (D, BG20; p=0.002)	26.08 (D, BG20; p=0.002)	44.61 (BG10, B10, B20, U, F; p=0.002)	29.57	42.02 (BG10, B10, B20, U, F; p=0.002)	26.53 (D, BG20; p=0.002)
40-50 cm	22.35	26.67	21.94	27.58	39.50	24.52	39.66	23.01

Values in brackets indicate which treatments were significantly different with the significance value. BG10- burnt and grazed (every 10 years), U – unmanaged, B20 – burnt (every 20 years), B10- burnt (every 10 years), G – grazed, D – drained, F – afforested.

6.4 Discussion

6.4.1 Introduction

Comparisons of carbon dioxide gains and losses in the field between multiple peatland management practices have not been published to date. At best, two common methods of management have been compared with an unmanaged site (e.g. Ward et al. (2007), Clay et al (2010b)). These studies provide a useful baseline on which further research can be developed, yet do not allow decisions to be made as to which of the four most common methods of land management is preferable from a carbon storage and release perspective. Differences were expected to exist in the losses and gains of carbon dioxide owing to the effect of land management on the key drivers of carbon cycling: environmental conditions, nutrients and substrate quality. In addition, changes were expected to occur to the physical properties of the peat as a result of land management, consequently affect the ability of gases to diffuse through the peat profile.

6.4.2 Effect of Land Management on Carbon Dioxide Exchange in Peatlands

In general, no clear trends existed in the data to identify which site lost or gained the most carbon dioxide during the course of monitoring. Significant differences in ER rates between managed peatland sites were not identified. Significant differences between individual monitoring locations were, however, found, demonstrating the heterogeneity of peatland ecosystems. The variation gives support to the notion of possible hotspots existing across sites put forward by McClain et al. (2003) as opposed to assuming homogeneity exists across each site as suggested by the classic yet simpler acrotelm-catotelm model (Morris et al. 2011). More detailed analysis of ER would be required to verify whether "hotspots" truly exist.

In terms of ER, one of the grazed monitoring locations (collar 9) had significantly higher ER than collars location on the burnt and grazed (every 10 years) site (collar 2); the burnt and grazed (every 20 years) site (collars 4 and 6); the burnt (every 10 years) site (collar 13) and the unmanaged site (collar 18) ($p=0.003$). Although collar 9 did not have a significantly difference groundwater levels to these collars, it is worth noting the mean water table level at collar 9 was lower than those noted above. Despite no significant correlation being identified between water level and carbon dioxide at collar 9, it is plausible to suggest that the water level would have had some influence on carbon dioxide losses. As noted in Chapter 2, debate exists in published literature as to the importance of the water table in controlling losses in carbon dioxide, with some authors (e.g. Nilsson & Bohlin 1993) suggesting deeper water table levels result in greater losses of carbon dioxide, and others finding no relationship between water table depth and carbon dioxide losses (e.g. Strack & Waddington 2007).

NEE data showed that the majority of the sites were net carbon sources throughout the monitoring period, indicating that the sites were emitting more carbon than is being sequestered. Only the burnt and afforested sites were found to have significantly different NEE values, with the forest appearing to emit more carbon dioxide than the burnt (every 10 years) site. These results do not however include the tree stand which would be the main carbon sequestering plants in the forest. Measurement of the forest NEE would only be possible with an eddy-covariance flux tower. Data published by Measurements of ER using eddy covariance methods were carried out Fowler et al. (1995) in May and June 1994 in Caithness. The results suggested that whilst losses of carbon dioxide were lower from the afforested site during the day (-0.086 g CO_2 m^{-2} h^{-1}) compared to an unmanaged site (-0.0004 g CO_2 m^{-2} h^{-1}),, losses of carbon dioxide were greater from the afforested site at night time than the unmanaged site - 0.002 g CO_2 m^{-2} h^{-1} and 0.0003 g CO_2 m^{-2} h^{-1} respectively.

Carbon sinks were identified at individual collar locations across the managed sites, as described below. A significantly lower value for NEE was identified for collar 10 on the burnt (every 20 years) compared to collar 21 on the drained site ($p<0.001$). Average NEE values for collar 10 indicated that this area of the burnt (every 20 years) site is a carbon sink, whereas the peat at collar 21 was identified as a carbon source. On the burnt site (every 10 years), collar 15 was found to be a sink, with a significantly ($p<0.001$) lower NEE flux than collars 7 (grazed); 12 (burnt and grazed every 20 years); 14 (burnt every 10 years) 16 and 17 (unmanaged) 21 (drained) and 23 and 24 (afforested) which were all found to be sources of carbon dioxide. Despite the absence of a significant correlation between NEE and water table level at collar 15, water levels at collar 15 were, however, frequently shallower than 10 cm beneath the surface (12 out of 19 monitoring visits) and the peat around the collar was often observed to be saturated with evidence of ponding at the surface. These aspects may have contributed to this location acting as a sink. Additionally the frequently saturated conditions would have prohibited sufficient variation in water table depth from being measured, and therefore a significant relationship between NEE and water table depth at this site could not be found using statistical analysis. The presence of saturated conditions and a high water table would have created anaerobic conditions, which could have resulted in reduced rates of microbial respiration, therefore creating a stronger carbon sink.

The majority of monitoring collars were located in areas where a combination of mosses and grass/sedge species were present, with the exception of the afforested site where moss and pine needles were identified. The consistency in the type of vegetation between managed sites could account for the lack of significant differences in NEE and ER values. On patterned peatlands, the different microforms tend to be characterised by different vegetation species, which are adapted to the water table found in each microform. *Sphagnum* species are typically associated with hummocks and hollows, whilst shrubs dominate lawns (Laine et al. 2007). Strong relationships have been identified between species and respiration and photosynthesis rates. Sites dominated by mosses are likely to continue to adsorb carbon during the winter months whilst shrubs lose their leaves and therefore, are unable to adsorb carbon but continue to lose carbon through ER (Laine et al. 2007).

No overall correlation was found between temperature or water level and NEE. Moderately weak correlations were identified between temperature and NEE on the burnt (every 10 years) site (r^2= -0.36, p=0.020) and drained (r^2=0.45, p=0.016) sites, indicating that temperature has some influence on NEE. Much of the variation in values however remains unaccounted for. Seasonal variations in the data were evident, with higher rates of carbon sequestration during warmer months when plant growth and photosynthesis would be expected to be greater. The lack of correlation between water table and NEE (r^2= 0.16, p=0.011)could be attributable to differences in plant species in the collars within each treatment because different plant species sequester carbon at different rates (Blodau 2002). Blodau (2002) also notes that nutrient availability affects rates of plant productivity, and thus rates of carbon sequestration.

No significant differences in primary productivity were identified between the managed sites using ANOVA ($p=0.123$). Primary production values for the afforested site represent the understorey vegetation only, and not the tree canopy. The drained site had the smallest mean primary productivity (-0.084 g CO_2 m^{-2} h^{-1}) whilst the burnt site (every 20 years) had the greatest (-0.21 g CO_2 m^{-2} hr^{-1}). The results of work carried out by Clay et al. (2010b) also found that the burnt (every 20 years) site had greatest primary productivity of the Hard Hill plots; and the unmanaged site the lowest. In the study at Moor House presented in this thesis, the burnt and grazed site (every 20 years) was found to have the lowest rates of primary productivity (-0.11 g CO_2 m^{-2} h^{-1}) of the Hard Hill plots, closely followed by the unmanaged site (-0.12 g CO_2 m^{-2} h^{-1}). Clay et al. (2010b) did not comment on whether significant differences exist between management practices for either ER or NEE, instead consideration was given to the carbon budget as whole, the results of which indicated significant differences existed between treatments. Although it was not possible to gain access to Moor House in all seasons for practical reasons, the strong relationship between temperature and ER identified in this study suggest that seasonal variation does have a strong bearing on ER, and possibly more so than land management, owing to the absence of significant differences in ER between treatments. In all cases, the sites were found to be carbon sources, however, as shown in Figure 6-2, lower ER was identified in winter months.

Ward et al. (2007) also concluded that seasonal variations were greater than variations in fluxes due to land management. A study carried out on a sub-Arctic fen has also highlighted the importance of temperature as a driver of respiration rates. The study looked at a particularly hot and dry summer identified lower rates of photosynthesis compared to a typical summer whilst respiration rates remained high. Conditions were not favourable for photosynthesis to match respiration during early summer when the site was a source; during mid summer the site was a sink then during late summer there was insufficient light for photosynthesis to match respiration (Schreader et al. 1998).

Monitoring of peat cores has also identified reduced rates of respiration at sites where the moisture content of the surface has been lowered. Published results indicate that lowering of the water table does not automatically cause an increase in respiration

(Lafleur et al. 2005). The role of water table in governing respiration rates is further complicated by the presence of microforms. During dry summers hummocks have been found to be unresponsive to lower water table levels whilst rates of respiration from hollows have increased (Bubier et al. 2003a).

The findings from the burnt and grazed sites are consistent with those of Clay et al. (2010b) who identified the burnt and grazed sites at Moor House as emitting significantly less carbon dioxide compared to the unmanaged site on Hard Hill. Such trends were attributed by Clay et al. (2010b) to the deeper water tables and lower rates of primary productivity in the unmanaged site. It was suggested that rates of new vegetation growth on the burnt and grazed sites are higher compared to the unmanaged site, hence carbon sequestration rates were greater. As a result of higher rates of primary productivity, it was suggested that rates of carbon accumulation would be greater in the burnt sites. Data presented above in this thesis also identified higher rates of primary productivity in the burnt (every 10 years) (-0.21 g CO_2 m^{-2} h^{-1}) and grazed (-0.20 g CO_2 m^{-2} h^{-1}) sites compared to the unmanaged site (-0.12 g CO_2 m^{-2} h^{-1}).

Evidence presented by Garnett et al. (2000) using radiocarbon methods to calculate the total carbon stores is not supported by the work of Clay et al. (2010b) which used measurements of primary productivity, and indicated that greater carbon was sequestered on the burnt site. The results of total carbon content analysis are presented in Chapter 5 and support the theory presented by Clay et al. (2010b) that burning increases peatland carbon stocks (although the differences in this study at Moor House were not found to be significant). Ward et al. (2007) also identified significantly higher carbon stocks in burnt peats. In addition, monitoring work showed significantly higher rates of respiration and photosynthesis in burnt and grazed treatments compared to unmanaged. The burnt and grazed sites were found to act as greater sinks than the unmanaged site. These findings are supported by Clay et al. (2010b); when looking at primary productivity data, the unmanaged site is less of a sink than the burnt and grazed plots. When looking at NEE from Clay et al. (2010b) however, it becomes apparent that most of the managed plots at Moor House are sources, with only the grazed sites and burnt sites acting as very small sinks.

No other studies in the UK have looked at NEE for burnt and grazed sites. A study comparing carbon dioxide release from burnt and unburnt sites in Mossdale, North Yorkshire did not identify any significant differences between the sites. The study, however, only looked at rates of release during April 2004, and NEE was not calculated, therefore conclusions cannot be drawn as to whether the site was a source or sink (Farage et al. 2009).

Studies on peatlands burnt by wildfires in Canada identified burnt sites as sources of carbon following burning, although the sites became sinks with time. In addition, seasonal differences were recorded, with sites acting as sources during winter and sinks during summer (Wieder et al. 2009). Although significant correlations were not identified between the sites for either NEE or PP, a visual analysis of NEE data indicates that the burnt site (every 10 years) did act as a sink during the majority of summer monitoring rounds and a source during winter monitoring. This result supports the findings of Roehm and Roulet (2003) who stressed that peats act as carbon sinks during the growing seasons and sources during the non-growing seasons, even during cold temperatures, when microbial activity is likely to become limited. Although this was the case at Moor House, the data collected were very low during the December 2009 monitoring trip when temperatures ranged between 3.8 °C and 5.3°C.

A study of a recently restored (by ditch blocking) catchment in Hexhamshire, northern England carried out by Rowson et al (2010) identified the site as a gaseous carbon sink (-0.007 g CO_2 m^{-2} yr^{-1}). The authors considered the sink to be smaller than expected, and attributed the difference to increased peat temperatures following water table drawdown. The effect of lower water table levels causing increased respiration was discounted based on the lack of a relationship being identified between water table levels and respiration by others (e.g. Lafleur et al. 2003). The study of the Hexhamshire drained catchment did not, however, include measurements of water table level to verify the absence of such a relationship. The low carbon exchange values for the site are also attributed to low rates of primary productivity. Low rates of productivity could have been symptomatic of the species growing on the site (potentially drainage resulted in less productive species growing), or the size of the chamber used to measure fluxes at the site (Rowson et al. 2010). Studies of the botanical composition of the drained site at Moor House were carried out in 1986 and did not identify any significant differences between areas located close to the drains and those furthest from the drains (Coulson et al. 1990).

The phenol oxidase enzyme is responsible for breaking down phenolic compounds, thereby making the substrate suitable for the hydrolase enzyme to synthesise, and so causing further decomposition in peats. The reduction of water table levels results in the creation of aerobic conditions and facilitates phenol oxidase activity (Freeman et al. 2004b). Aerobic conditions are not, however, the only requirement for the phenol oxidase enzyme. Low temperatures (Freeman et al. 2001b) and acidic conditions (Williams et al. 2000) can suppress rates of activity. Furthermore, the botanical composition of the peat also affects phenol oxidase activity, resulting in differences in rates of activity between different peatlands (Laiho 2006). It is possible that at Moor House the initial lowering of the water table during the 1950s at the drained site resulted in a rise in rates of enzyme activity. It is likely, however, that the supply of phenolic compounds has been exhausted sufficiently so as to cause the available substrate to have been fully utilised. Furthermore, significant differences in water levels were not identified between the drained site and other treatments except for the afforested site. The absence of significant differences in water table levels suggests that additional decomposition due to an thicker acrotelm, and/or the release of the enzyme latch mechanism is unlikely to occur. The absence of a thicker acrotelm can in part explain the lack of a significant difference between the drained site and other treatments.

Comparisons of Sitka spruce and lodgepole pine forests with felled sites on Irish peats found greater losses of carbon dioxide occurred from spruce sites than pine. The lower water table in the pine site did not result in greater losses of respiration owing to the decay resistant nature of the peat. In clearfelled areas, lower respiration rates were recorded than those found in the mature Sitka sites; the difference was attributed to the contribution of respiration from the roots of the Sitka spruces (Byrne & Farrell 2005).

6.4.3 Environmental Controls

6.4.3.1 Moisture

Studies of moisture gradients across peatlands have identified lower rates of carbon dioxide adsorption at the driest end of the spectrum (Bubier et al. 2003a). Studies of drought conditions in peatlands have identified the onset of drier conditions as causing peatlands to switch from sinks to sources of carbon owing to reduced rates of plant production (Alm et al. 1999a). Qualitative comparison of the data suggests that it is possible that increased losses of carbon dioxide through ER may be attributable to the higher moisture content found in the afforested site, however, further analysis would need to be carried out to verify this proposal.

6.4.3.2 Temperature

Fluctuations in carbon dioxide emissions were clearly linked to air temperature, with lowest concentrations emitted during the coldest monitoring rounds (e.g. 7th December 2009 round 12). The afforested site was the only site not to show a significant correlation with temperature, with a very weak correlation coefficient (r^2=0.17; p=0.411). The results may be due to temperatures in the forest being cooler during the summer months and warmer during the winter months relative to other treatments, owing to the shade and shelter provided by the canopy (Silvola et al. 1996). Whilst the afforested site was found to have significantly deeper groundwater levels (especially at monitoring locations 22 and 24 which were situated on ridges), no significant correlation was identified between water depth and ER from the afforested site.

Suggestions have been made that the relationship between temperature and carbon dioxide release increases exponentially (Dioumaeva et al. 2003). Whilst rates of carbon dioxide production did increase with temperature, and rates of production were low during cold periods, the relationship was found to be linear rather than exponential. The differences in the findings between the studies may be attributable to differences in the types of peat – blanket bog at Moor House compared to boreal peat studies carried out under laboratory conditions.

The findings for the environmental controls on carbon dioxide losses are also consistent with those of Updegraff et al. (2001) who identified temperature as a major control on carbon dioxide, accounting for up to 80 % of variation and Nieveen et al. (2005) who found temperature explained up to 93 % of variance. Results from the managed sites at Moor House indicated that temperature accounted for between 54 and 91 % of variation (excluding the results from the afforested site; p=0.001).

6.4.3.3 Water table effects

Overall, the depth to the water table level was not a significant control on ER. No significant relationship was found between ER and depth to water table for any site, thus contradicting suggestions made by Nilsson and Bohlin (1993) that water level is the most important control on carbon dioxide losses. Even when all the data for all sites were pooled together, a significant relationship was not identified (r^2= -0.031; p=0.627).

During monitoring events in autumn 2009, shallower water table levels were recorded in October than those found during monitoring in November and December. Rates of carbon dioxide loss were, however, higher in October when temperatures were higher, compared to November and December when temperatures were lower as well as groundwater levels being deeper.

The results of the dipwell monitoring indicated that the peats on managed sites at Moor House are rarely saturated. Mean groundwater levels in the afforested peat were always 15 cm or more beneath the surface of the peat, even after heavy rainfall had occurred prior to a monitoring round. Shallower water levels were found in sites subject to burning and grazing compared to the unmanaged site. Such elevated levels did not, however, result in lower ER or significant differences in the porosity of the peat. Clay et al. (2009a) also identified deeper water table levels in the unmanaged site compared to those subject to burning and/or grazing at Moor House. Comparisons with the drained and afforested sites were not made.

Elsewhere, sites where drains have been installed were found to have higher losses of carbon dioxide than undrained sites owing to an increase in the unsaturated zone (Hogg et al. 1992). A review of drained sites carried out by Laiho (2006), however, suggests there is evidence for and against carbon stores being affected by changes in water table levels. Experimental work carried out on a Canadian sub-Arctic fen found rapid increases in carbon dioxide losses following lowering of the water table. Subsequent lowering of the water table did not result in any further losses of carbon dioxide. The presence of easily labile carbon in the newly aerated peat was attributed as the cause for the increase in the flux following drainage (Chimner & Cooper 2003). Peatlands that have been exposed to lower water table levels for longer periods of time have been found to be highly resistant to decomposition (Bridgham & Richardson 1992) thus resulting in no differences in carbon losses, as found in the results of this thesis. Nieveen et al. (2005) demonstrated that dry periods and/or grazing livestock can reduce losses of carbon dioxide from peatlands, following water table drawdown and afforestation. Scottish peatlands sites were identified as sources of carbon for two to four years post-drainage. Eight years after being drained, the sites returned to acting as carbon sinks (Hargreaves et al. 2003).

Despite the establishment of relationships in some studies between water table and losses of carbon dioxide owing to more rapid mineralisation; evidence exists to suggest that peat becomes more decay resistant with depth (Hogg et al. 1992). Lowering of the water table will therefore expose peat which cannot easily be degraded, and thus will not contribute significantly to additional carbon dioxide losses. The results provide evidence that carbon mineralisation is controlled by more than just the water table level and thickness of the aerobic zone of the peat, hence the lack of a significant relationship between ER and water table level, as found by others e.g. Strack and Waddington (2007).

6.4.3.4 pH

The pH of peats often increases following water table drawdown owing to an increase in organic compounds being oxidised, and in afforested sites, due to an increase in base cation uptake by trees (Laiho 2006). Falling pH values due to afforestation are thought to counteract the effects of the increase in oxic conditions in the peats (Minkkinen et al. 1999) and thus reduce respiration rates. The afforested site at Moor House did have a significantly lower pH in both peats (as detailed in Chapter 4) and peat solution (as detailed in Chapter 7), which might explain the absence of significant differences in respiration rates between the afforested site and the other treatments considered.

6.4.4 Physical Properties

A discussion of the effects of land management on bulk density is presented in section 5.4.

Wide variation in the air-filled porosity of the peats within each treatment reflects variation in the bulk density and moisture content within each treatment. Whilst Moor House does not feature many hummocks and hollows as would be found on a patterned peatland, areas of the site were found to significantly wetter than others. These findings add to the notion presented earlier that hotspots might exist across peatlands reflecting the heterogeneity of the peat which is often over simplified through the acrotelm-catotelm model (Morris et al. 2011), further discussion of the application of the diplotelmic model is presented in Chapter 8.

Burning was expected to result in a reduction in porosity owing to the clogging of air-filled pores with ash particles (Mallik et al. 1984b). Porosity and air-filled porosity values were not however significantly different from the unmanaged site. These findings might be due to insufficient quantities of ash being produced to cause blocking of pores or could be due to the ash being washed away due to runoff in the time period between burning (February 2007) and sample collection (September 2008).

Changes in soil porosity as a result of burning have been identified as a cause for increased soil moisture content on burnt sites due to improved moisture retention (Mallik et al. 1984b). No significant differences in moisture content were identified in this most recent study at Moor House (as noted in Chapter 4). Elsewhere, studies of burnt podzols in Scotland identified ash particles in the soil pores, resulting in reduced porosity and consequently reduced infiltration (Mallik et al. 1984b) and heating soils was found to result in increased in soil water repellancy and thus decrease infiltration rates (DeBano 1990). The contrast between these studies and those at Moor House might be a reflection of the cool burns which were carried out at the Moor House site in 2007.

The lower bulk density and higher total porosity and air-filled porosity identified in the drained site demonstrated that drainage improves the structure of the peat, resulting in a more porous, less saturated peat, however, ER from the drained site was not significantly different to any other treatment. Moisture content analysis (presented in Chapter 4), however, identified the moisture content of the drained site as being significantly higher than most other sites. The significantly higher loss on ignition value for the drained site compared to the unmanaged site ($p=0.02$) as presented in

Chapter 5, however, could have resulted in a better structure for the peats at the drained site resulting in higher porosity and lower bulk density values.

6.4.5 Effect of Burning Frequency

No significant differences were identified between the burnt (every 10 years) and burnt (every 20 years) sites. Differences in the rate of vegetation recovery might have resulted in differences in ER, however, this did not appear to be the case. Firstly, the burn carried out in February 2007 was reported to be light, and so the impact on vegetation was not severe. Secondly, the site had made a good recovery from any damaged caused to the vegetation. It is possible that differences would be evident immediately after a burn compared to both the unmanaged site and one that had been burnt 10 years ago. Work carried out by Clay et al. (2010b) also found little difference in ER and PP values between the two sites. The impact of burning frequency is therefore difficult to discern, given the possible variations in burn severity between the 10 and 20 year burn cycles. Data on actual temperatures reached during burning were not recorded, but records kept by the site manager suggest the most recent burn was of a very cool nature.

6.4.6 Effect of Combining Burning and Grazing

Combining burning and grazing methods did not result in any significant differences to the observed sequestration or release of carbon dioxide. The physical properties of the burnt and grazed (every 20 years) site were, however, significantly different to most other treatments in terms of bulk density, total porosity and air-filled porosity. The burnt and grazed (every 20 years) site had a significantly lower bulk density than all other sites where burning took place and compared to the unmanaged site. Similarly total porosity and air filled porosity values at the burnt and grazed (every 20 years) were significantly higher than those found on other plots on Hard Hill.

Significant differences in the porosity of the peat were identified with the burnt and grazed (every 20 years) site and the drained site being more porous than most other treatments. The results however do not appear to have had an effect on diffusion of carbon dioxide through the peat profile. Similarly the air filled porosity values varied significantly between the drained and burnt and grazed (every 20 years) sites and most other treatments but did not result in significant differences in the release of carbon dioxide.

6.5 Conclusions

6.5.1 Summary of Findings

The principal aim of this chapter was to analyse the effects of land management on the carbon balance of managed peats focussing on carbon dioxide losses and gains. In addition, the effect of management on the physical properties of the peat that influence gaseous diffusion was examined. The following hypotheses were tested:

i) Land management has a significant effect on losses of carbon dioxide from peat
ii) Land management has a significant effect on net ecosystem exchange
iii) Land management has a significant effect on the porosity of the peat, thereby altering the potential for carbon dioxide to diffuse through the peat

6.5.1.1 Effects of Land Management on ER and NEE

Land management was not found to have a significant effect on NEE or ER contrary to the hypotheses proposed above. The frequency with which sites were burnt had no observed effect on carbon dioxide fluxes, neither did combining burning and grazing. The absence of significant differences in water table levels between the sites compared to the unmanaged site (with the exception of the afforested site), suggest that no difference exists in the thickness of the acrotelm of the managed sites, therefore, the potential for greater or lesser rates of microbial decomposition through aerobic activity within the treatments did not exist. Furthermore, rates of primary production between the sites were not significantly different, suggesting that in spite of the application of different management practices, the ability of the vegetation on the sites to sequester carbon was not significantly different. Further, more powerful analysis would be required to verify these suggestions.

Water table levels were significantly lower in the afforested site compared to burnt, grazed, drained and unmanaged sites. No other significant differences in water levels were identified between treatments. Significant differences in the physical properties of managed peatlands were found, however there was no indication that these differences affected transfusion of carbon dioxide through the peats.

All sites were found to be sources of carbon dioxide, with greatest losses during the summer monitoring rounds. Temperature was found to be a much stronger driver of ER than land management, with moderate to strong relationships (r^2=0.54 – 0.92) identified between temperature and ER for all sites except the afforested site. These results suggest that the alterations peats undergo as a result of management intervention investigated are insufficient to cause significant changes to the drivers of the carbon cycle. Further consideration of the linkages between losses of carbon and drivers of the carbon cycle is given in Chapter 8.

No significant relationships were identified between water table and NEE or ER. Whilst high water tables are very important for carbon accumulation, no evidence was found to suggest that water table drives carbon dioxide losses from managed peats. In the future (under climate change), lower water table levels might not give rise to increased respiration rates, however, they could limit carbon accumulation, thus depleting the quantity of carbon held within peatlands.

6.5.1.2 Effects of Land Management on the Physical Properties of Managed Peats

Significant differences in the physical structure of the peat were anticipated between different treatments owing to the effects of drains (in both afforested and drained peats), inputs of ash from the burnt sites and trampling on grazed sites. The results demonstrated that only the afforested and drained site had significantly different porosities; with the afforested becoming less porous owing to a higher bulk density, and the drained site becoming more porous.

6.5.2 Recommendations for Further Work

A full investigation of the carbon balance of the afforested site was not possible due to the limitations of the NEE measurements. To fully quantify the flux, an eddy-covariance flux tower would be needed, and to make comparisons with other sites, flux towers would need to be on each site. Accurate measurements at the Moor House plots might, however, be problematic owing to their small size. In addition, future work needs to focus on managed sites outside of the Moor House NNR, and variations in management intensity should be examined such light burns compared to heavy burns, light grazing compared to heavy grazing, and different drain spacing.

7 THE EFFECTS OF LAND MANAGEMENT ON DOC PRODUCTION AND PEAT SOLUTION CHEMISTRY

7.1 Introduction

Dissolved Organic Carbon (DOC) comprises a complex mixture of humified plant materials that have been dissolved in water (Dillon & Molot 1997). Evans et al. (2005, p55.) define DOC as "any organic compound that can pass through a 0.45 µm filter". Given the range of particle sizes containing organic matter, the definition is somewhat arbitrary. No specific chemical formula exists for DOC as the molecules present depend on the degree of organic matter degradation, and hence vary through time. In general terms DOC typically includes carbohydrates, amino acids and humic substances. Once produced within the pores of the peat, DOC is released from the peat solution during periods of rainfall and is transported into rivers and streams (Charman 2002).

DOC is the dominant form of aqueous carbon released from upland peats. The means through which DOC is formed and released are poorly understood (Holden 2005a). Work carried out by Hope et al. (1997) identified strong links ($r^2=0.83$, $p<0.001$) between the amount of carbon stored in peats and concentrations of DOC leaving the peat. This relationship was confirmed by work carried out by Aitkenhead et al. (1999) who also explored relationships between catchment size, peat cover and DOC output. They found small catchments (less than 5 km^2) exported the greatest fluxes of DOC.

DOC gives rivers and streams in upland areas their characteristic brown coloured water (Urban et al. 1989). DOC molecules can react with chlorine to form trimethohalogens which are known carcinogens (Pereira et al. 1982). Water supply companies are obliged to ensure DOC is removed from water within the treatment works before being supplied to customers (Hsu et al. 2001, Sharp et al. 2006). Such treatment is costly, therefore water companies are keen to seek methods of reducing DOC concentrations within the catchment prior to entering the treatment works (Worrall et al. 2003a).

Studies of concentrations of DOC over the last 40 to 50 years have shown significant increases in concentrations of DOC leaving upland catchments across northern Europe (Evans et al. 2005). Monteith et al. (2001) examined data from the Acid Waters

Monitoring Network (AWMN) for 22 sites in the UK. The sites included 11 lakes and 11 streams. The data showed that over a 10 year period, concentrations of DOC rose at all sites. Further work carried out by Freeman et al. (2001a) identified a 65 % increase in DOC concentrations in watercourses that drained from peatland catchments in the UK over a period of 12 years. Evans et al. (2005) reviewed studies carried out across Europe and North America which showed increases in DOC concentrations of between 10 and 91 %. Much debate has taken place over the causes of the rises in DOC concentrations and subsequently fluxes. The key arguments were presented in Chapter 2, and are briefly summarised below. Note should be taken that some studies have focussed on concentrations within the peat solution, while others have focussed on fluxes in peatland catchments, thus making direct comparisons difficult.

7.1.1 Drought and Water Levels

Changes in water table levels have been linked with increased DOC concentrations in peat solution, however, debate exists however as to the extent of such linkages. A laboratory mesocosm study was carried out to test the effects of warming and changes in water table level on DOC, from which a strong link was established between increasing water table levels and decreasing DOC concentrations ($p<0.001$) (Pastor et al. 2003). Work by Blodau et al. (2004) found no significant changes in DOC losses from peatlands where water table levels were altered. Simulation of drought conditions by Freeman et al. (2004a) also failed to demonstrate a rise in DOC concentrations. Studies by Worrall et al.(2004a), however, demonstrated that where increased drought conditions and hence lower water table levels were recorded, DOC concentrations subsequently rose but were subject to a time lag which was attributed to the hydrophobic nature of peats. Whilst drawdown of water levels may impact on DOC concentrations in peat solution, the required level of drawdown to match the increases in DOC concentrations witnessed in recent decades, have not been recorded. Thus water table drawdown alone cannot explain the increased concentrations of DOC identified in catchments.

7.1.2 Enzyme Latch Mechanism

Freeman et al. (2001b) suggested that the presence of anaerobic conditions presented when water levels were high, has prevented the phenol oxidase enzyme from operating. The enzyme latch mechanism as described in Section 2.4.2.2 inhibits the degradation of complex carbon molecules under anaerobic conditions, but once water tables have been lowered, degradation of the most recalcitrant molecules can begin.

7.1.3 Temperature

Freeman et al. (2007) identified recent temperature rises as a cause for increased losses of DOC, due to enhanced rates of microbial activity (Freeman et al. 2001a) resulting in more rapid breakdown of organic matter and thus losses of carbon into the aqueous phase.

The arguments posed above as possible causes for rises in DOC concentrations have been refuted by authors such as Evans et al. (2006a) who noted that the rises in temperatures have been insufficient to match the rise in DOC witnessed over the past 50 years, and certainly not the rise (65 % increase) witnessed in the 12 year study carried out by Freeman et al. (2001a). Whilst changes in temperatures and water levels can be linked to rates of DOC losses, they do not explain the whole story.

7.1.4 Sulphates and Changes in Acidity

Clark et al. (2005) proposed that droughts caused the onset of sulphur oxidation in peats. Aerobic conditions within the peat allow sulphur to be reduced to sulphate, thus releasing hydrogen ions into the peat solution. The release of hydrogen ions causes an increase in acidity thus inhibiting rates of microbial activity and carbon mineralisation. Rates of DOC loss are subsequently reduced. Once water levels recover and pH values increase, DOC production increases again, as the less acidic conditions favour microbial decomposition of carbon, and the absence of sulphates allows DIC to be released. A significant correlation was identified between DOC and sulphate losses ($p<0.001$), with the mechanism described above accounting for a greater proportion of the variance in DOC losses ($r2=0.81$) than temperature alone could ($r2=0.58$).

7.1.5 Vegetation

Limpens et al. (2008) provided preliminary data that suggest that the species of vegetation present has an impact on DOC production, with greatest concentrations released from *Calluna* and least from *Eriophorum*. Changes in land management are often associated with changes in vegetation, as noted in Chapter 4. Work carried out on tundra and boreal peats identified greater losses of DOC from shrub vegetation compared to sedges (Neff & Hooper 2002).

7.1.6 Land Management

Whilst complex arguments exist for the increases in DOC losses from upland peats, it is clear that land management alone cannot have caused such changes – almost all upland areas affected have been subjected to different forms of land management. No one land management practice can therefore be held accountable for the rises and it would seem that the changes have occurred regardless of land management. The influence of land management on the drivers of DOC production and loss in peatland has not however been investigated, and doing so, would enable a better understanding of the role of land management in DOC production and loss.

Work on highly organic soils (podzols) in Wales identified higher concentrations of DOC in the soil solution of afforested podzols compared to grazed (Hughes et al. 1990). A comparison of runoff waters from felled and afforested sites across Wales found felled sites to have higher concentrations of DOC and to be less acidic than afforested sites (Neal et al. 1998). Conversely work by Evans et al. (2005) on causes for long term increases in DOC losses from sites belonging to the AWMN did not find any significant differences between afforested sites and nearby moorland sites.

Holden et al. (2007b) noted that to date there is little evidence to link the impacts of burning on water quality. More recent work carried out by Clay et al (2009b) and Ward et al. (2007) has begun to provide some initial data to start to address this gap, with an emphasis on peat solution studies. Analysis of peat solution samples collected by Ward et al. (2007) prior to the 2007 burn found burning did not impact upon DOC concentrations, however, small increases were noted in grazed sites. In contrast, work carried out by Clay et al (2009b) found grazing did not impact upon DOC concentrations in soil solution. The difference in these findings is surprising as both studies used Moor House NNR Hard Hill plots as their study site. The study by Clay et al. (2009b) study found no differences between burnt and unmanaged sites in the months preceding a burn or a few months post burn. The study did however find that DOC concentrations rose in the weeks immediately following the burn before falling to similar levels to those found in the unmanaged plots. Yallop and Clutterbuck (2009) also identified a much stronger correlation between recent burns and DOC concentrations (r^2=0.62 as opposed to r^2=0.37 for sites that have not newly burnt).

Work carried out by Mitchell and McDonald (1995) identified that increased colour (and hence DOC) was associated with catchments featuring a higher drainage density. Wallage et al. (2006) identified higher concentrations of DOC in the pore waters of peats in drained catchments compared to undrained catchments, and found that blocked catchments (previously drained) had even lower concentrations of DOC than undrained catchments. Blocking drains has been found to reduce rates of surface flow in catchments, thereby reducing the rate at which DOC is exported from peats into watercourses (Gibson et al. 2009). Predictions for climate change in upland areas of the UK suggest that increased rainfall will occur, thus increasing rates of flow in streams and rivers. Such increases are predicted to reduce concentrations of DOC in watercourses as a result of dilution caused by higher rates of flow under climate change (Clark et al. 2008). Holden et al. (2007b) noted that catchment characteristics can result in decreased DOC concentrations compared to undrained nearby sites, as found in studies in Canada, USA and Scotland.

7.2 Approach

7.2.1 Aim

The aim of this chapter is to determine how concentrations of DOC in peat solution vary with depth and between managed sites; and to examine changes in the water chemistry of managed peatlands, with a focus on the properties that are relevant to DOC loss.

7.2.2 Hypotheses

1. Land management affects concentrations of DOC in the peat solution

- As discussed in Section 7.1.6, land management has some effect on DOC concentrations at a local scale (Wallage et al. 2006, Ward et al. 2007, Clay et al. 2010a, Gibson et al. 2009), but comparisons have not yet been made between several different treatments and an unmanaged site.

2. Land management affects concentrations of sulphate in peat solution

- Changes in water table levels are expected to occur between differently managed sites (Worrall et al. 2007a, Holden et al. 2007b, Cannell et al. 1999). The potential for sulphur reduction to occur between treatments is expected to vary as a result, thus affecting sulphate production, which has been found by some to influence DOC concentrations.

3. Land management affects the acidity of peat solution

- Changes in acidity are expected between different land management practices owing to changes in water table levels thus creating differences in the thicknesses of the acrotelm and catotelm at each site. In addition, the planting of trees is known to reduce pH values (Miller et al. 1990), thus the afforested site is expected to be the most acidic. Significant effects are not expected at the burnt or grazed sites (Clay et al. 2009b). Changes in plant community due to management (Ward et al. 2007) however may cause changes to the substrate entering the peat, which once decomposed could release different forms of acid compounds, thus altering the pH of managed sites.

4. Water chemistry changes with depth down the peat profile

- Changes in the peat solution chemistry with depth are expected as environmental conditions change between the acrotelm and the catotelm (Clymo 1984). Concentrations of DOC in the managed sites are expected to decrease with depth as the peat becomes more recalcitrant and rates of groundwater flow decrease (Hughes et al. 1990, Ward et al. 2007).

5. Burning frequency effects peat solution chemistry.

- Burning frequency is not expected to have a significant effect on DOC concentrations based on previous studies (Clay et al. 2010b). Sulphate concentrations and acidity may be significantly different however owing to different substrate inputs and groundwater levels.

6. Combinations of burning and grazing affect peat solution chemistry.

- Limited changes are expected to occur between sites that are solely burnt, solely grazed and those where a combination of burning and grazing take place (Clay et al. 2010b, Ward et al. 2007).

7. Land management impacts on the key drivers of DOC production in peats.

- A number of drivers of DOC production in peats were discussed in Section 7.1.6. Given that land management is expected to have an effect on some, if not all of these drivers, the strength of the relationships between DOC and these drivers might also change.

7.3 Methodology

7.3.1 Fieldwork

Fieldwork was carried out at the monitoring sites detailed in Chapter 3. Three sets of wells were monitored on each of the Hard Hill plots, and at the drained and afforested sites. Each set of monitoring wells included a well to monitor groundwater levels, and wells to collect peat solution from the following target depths: 10 cm, 20 cm and 40 cm. Between March and December 2009, 14 rounds of peat solution sampling were carried out on the following dates: 10/03/09; 07/04/09; 21/04/09; 10/05/09; 19/05/09; 02/06/09; 16/06/09; 02/07/09; 15/07/09; 29/07/09; 07/10/09; 23/10/09; 16/11/09 and 07/12/09. Samples were collected using a clean 50 ml syringe with

Tygon tubing attached to the syringe. The Tygon tubing was lowered into the well and the sample collected in the syringe. The syringe was rinsed with deionised water between samples. The wells were emptied after each sample was collected. Samples were collected in clean, 50 ml polypropylene screw cap containers and labelled with a permanent marker pen. Samples were stored in the dark at 4°C until required for analysis. Samples were analysed for pH, DOC and sulphate using the methods detailed below. Water table levels were measured using a dipmeter as detailed in section 6.2.2.2.

7.3.2 Acidity

The pH value for each solution sample was measured using a pH probe (Mettler-Toledo).

7.3.3 Dissolved Organic Carbon

Samples were filtered into clean plastic vials using a 0.45 μm PTFE filter and analysed for DOC by Thermal Oxidation (Thermalox). The analyser was calibrated using standards produced by diluting a solution of potassium hydrogen phthalate containing 1,000 ppm carbon with deionised water into an appropriate working range. A blank and a certified reference sample (VKI QC WW4a) were analysed every 15 samples. The results were corrected for drift using the certified reference material results.

7.3.4 Sulphate

Samples were filtered using a 0.45 μm PTFE filter and 2 ml of each sample was placed into clean screw cap vials with a perforated cap. Samples were analysed for sulphate concentrations by ion chromatography (ICS-3,000 High Performance Ion Chromatography System). Standards were made from a 1,000 ppm stock solution of sulphate diluted into an appropriate working range with deionised water.

7.3.5 Statistical Analysis

The data sets examined did not have a normal distribution. Attempts to normalise the data by log transformations failed to produce a normal distribution. As the requirements of the Kruskall-Wallis H test (a non-parametric test which does not require a normal distribution) were not fulfilled either, ANOVA was selected to identify where significant differences between datasets occurred, as the method is considered to be more robust. The relationships between the key drivers of DOC cycling were examined using the Pearson Product Moment Test. Values for each site presented in the results section are taken from an average of each of the three duplicate wells located on the different sites studied.

7.4 Results

Where there are gaps in the data, either the monitoring well was empty at the time of sampling or access to the site could not be gained (e.g. due to thick snow). On some occasions, access across Moss Burn could not be achieved to reach the drained site due to the high discharge rates in the surrounding streams.

7.4.1 Environmental Conditions

The air temperatures recorded by the Automatic Weather Station (AWS) at Moor House at midday on each of the sampling days are presented in Figure 7-1. The highest temperatures were achieved in July 2009, whilst the lowest were in March, April and December. Groundwater levels for the main treatments are presented on Figure 7-2. On average the shallowest levels were found in the grazed site and the deepest at the afforested site. Significant differences were identified in the treatments' groundwater levels, as summarised in Table 7.1.

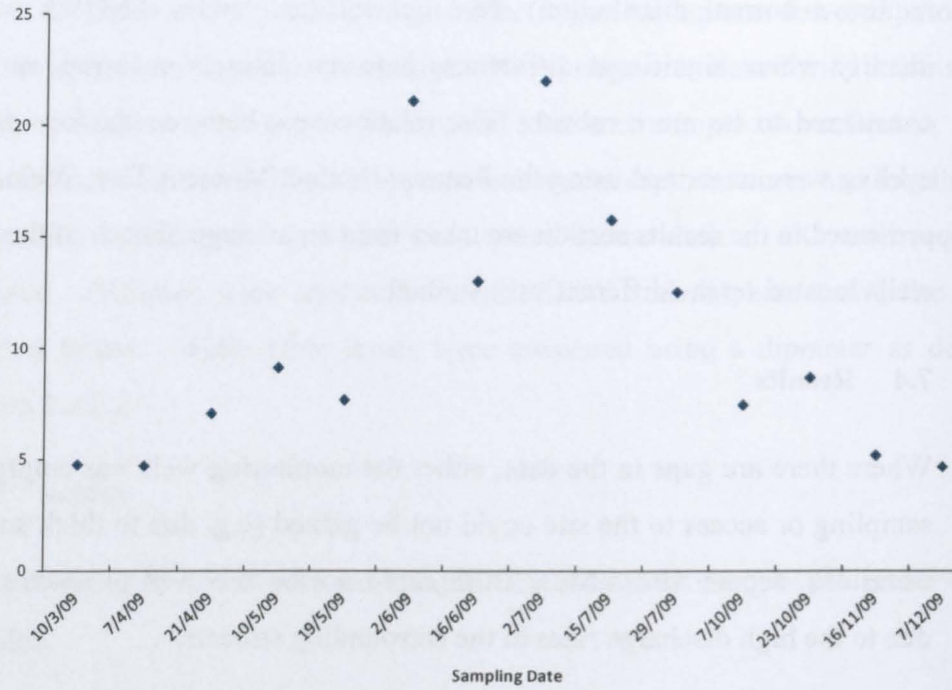


Figure 7-1 Temperatures at Midday for Each Sampling Date

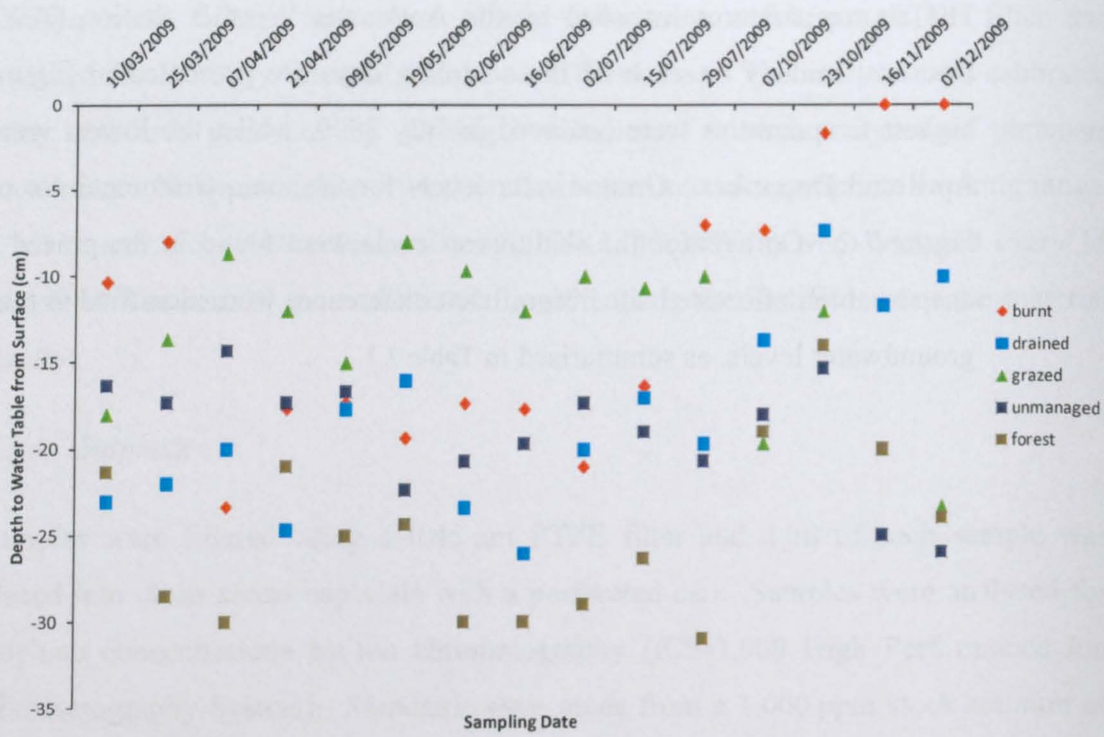


Figure 7-2 Mean Groundwater Levels Below the Surface of the Peat for Each Site Based on the Mean of the Three Wells Located on Each Site

Table 7.1 Locations Where Significant Differences Were Identified between the Groundwater Levels between Treatments

	Forest	**Drained**	**Unmanaged**	**Grazed**	**Burnt (every 20 years)**	**Burnt and grazed (every 10 years)**	**Burnt and grazed (every 20 years)**
Burnt (every 10 years)	✓	✓	✓			✓	
Burnt (every 20 years)	✓	✓	✓			✓	
Burnt and grazed (every 20 years)	✓	✓	✓				
Burnt and grazed (every 10 years)	✓						
Drained	✓						
Unmanaged	✓						
Grazed	✓		✓				

✓ - Significant difference between treatments

7.4.2 *Water chemistry results for all sites*

Table 7.2 provides a summary of the water chemistry and levels measured at all sites, grouping all management practices together. The data were used to identify if correlations exist between the measured parameters when all land management practices were combined, the results are presented in Table 7.3. The results demonstrate that significant, positive relationships existed between pH and DOC, pH and sulphate, DOC and temperature, DOC and sulphate, sulphate and water table levels, sulphate and temperature, and pH and water table levels. Significant, negative relationships were identified between water table and pH, sulphate and water table, and temperature and water table.

The pH data were found to be more alkaline that expected for a blanket bog, however when compared with data collected by others for Moor House e.g. Worrall *et al.* (2007a) where pH data for the managed plots ranged between 4 and 7.

Table 7.2 Summary of Values for Each of the Parameters Measured in the Field when all Land Management Practices and Depth from which the Data were Collected were Grouped Together and for Each Treatment

Parameter	Mean Value	Minimum Value	Maximum Value
DOC (mg L^{-1})	52.25	0.07	241.85
Sulphate (mg L^{-1})	4.54	0.17	32.74
pH	5.8	3.1	7.2
Water table level (cm below peat surface)	15.2	0.0	50.0
Burnt (every 10 years) (n=90)			
DOC (mg L^{-1})	42.84	9.80	141.08
Sulphate (mg L^{-1})	4.03	0.26	22.89
pH	5.59	4.19	6.52
Water table level (cm below peat surface)	-12.4	0.00	-50.00
Burnt (every 20 years) (n=65)			
DOC (mg L^{-1})	44.21	16.71	85.40
Sulphate (mg L^{-1})	3.36	0.27	11.51
pH	5.70	4.09	6.88
Water table level (cm below peat surface)	-8.14	0.00	-19.00
Grazed (n=45)			
DOC (mg L^{-1})	50.67	15.38	112.97
Sulphate (mg L^{-1})	4.61	0.45	10.27
pH	6.21	5.08	7.05
Water table level (cm below peat surface)	-10.53	-0.00	-32.00
Burnt and Grazed (every 10 years) (n=45)			
DOC (mg L^{-1})	59.76	0.83	132.33
Sulphate (mg L^{-1})	4.83	0.83	11.12
pH	6.28	5.50	7.20
Water table level (cm below peat surface)	-17.02	-0.00	-42.00
Burnt and Grazed (every 20 years) (n=54)			
DOC (mg L^{-1})	57.54	22.48	138.74
Sulphate (mg L^{-1})	5.39	0.17	15.38
pH	5.9	4.6	7.1
Water table level (cm below peat surface)	-8.02	0.00	-18.00
Drained (n=70)			
DOC (mg L^{-1})	53.88	0.94	126.90
Sulphate (mg L^{-1})	4.77	0.35	32.74
pH	5.92	3.90	7.12
Water table level (cm below peat surface)	-19.62	0.00	-33.00
Afforested (n=61)			
DOC (mg L^{-1})	72.23	22.67	241.85
Sulphate (mg L^{-1})	5.97	0.23	24.76
pH	5.06	3.08	6.66
Water table level (cm below peat surface)	-27.50	-5.00	-50.00
Unmanaged (n=62)			
DOC (mg L^{-1})	42.78	14.61	105.43
Sulphate (mg L^{-1})	3.9	0.28	14.43
pH	5.9	4.7	7.0
Water table level (cm below peat surface)	-18.68	-6.00	-36.00

Table 7.3 Correlation Coefficients with Significance Values for all Sites Regardless of Treatment or Depth

	pH	**DOC**	**Sulphate**
DOC	0.187 $p<0.001$		
Sulphate	0.097 $p=0.046$	0.454 $p<0.001$	
Water table	0.168 $p<0.001$	-0.084 $p=0.065$	-0.165 $p=0.001$
Temperature	-0.05 $p=0.26$	0.153 $p=0.001$	0.171 $p<0.001$

p = significance of each correlation, value underneath the p value is the correlation coefficient (R^2).

7.4.3 Water Chemistry Variations at the Burnt (every 10 years) Site

Concentrations of DOC (Figure 7-3) did not differ significantly (Table 7.4) between the acrotelm (0-10 cm) and catotelm (20-40 cm) ($p=0.851$) and were found to increase during periods when temperatures were higher, although a significant relationship between temperature and DOC was not identified. Positive relationships were identified between DOC and both sulphate and pH (Table 7.5) indicating less acidic conditions, with greater concentrations of sulphate promoted greater losses of DOC. No significant changes in sulphate concentrations were identified with depth, despite a reduction in mean concentrations (Figure 7-4), however the peat solution was found to be significantly less acidic with depth (Figure 7-5). Although DOC concentrations were not significantly different to the unmanaged site, concentrations were significantly lower in samples collected from the burnt (every 10 years) site compared to all other treatments.

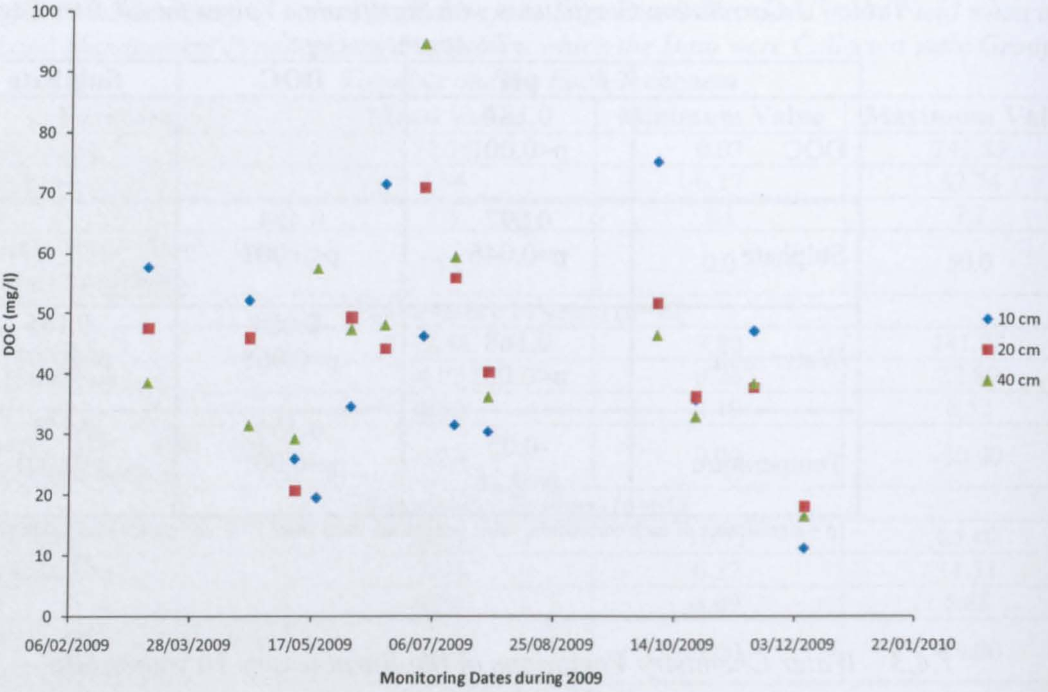


Figure 7-3 Mean DOC Concentrations in Peat Solution Samples Collected from the Burnt (every 10 years) Site for the Three Sampling Depths

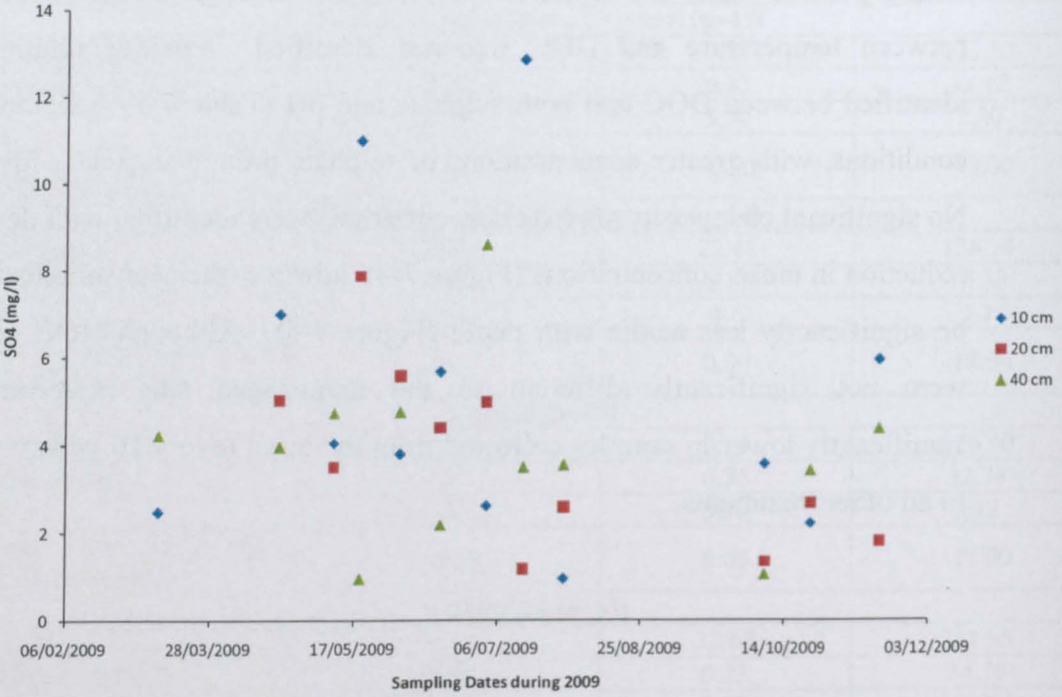


Figure 7-4 Sulphate Concentrations in Peat Solution Samples Collected from the Burnt (every 10 years) site for the Three Sampling Depths

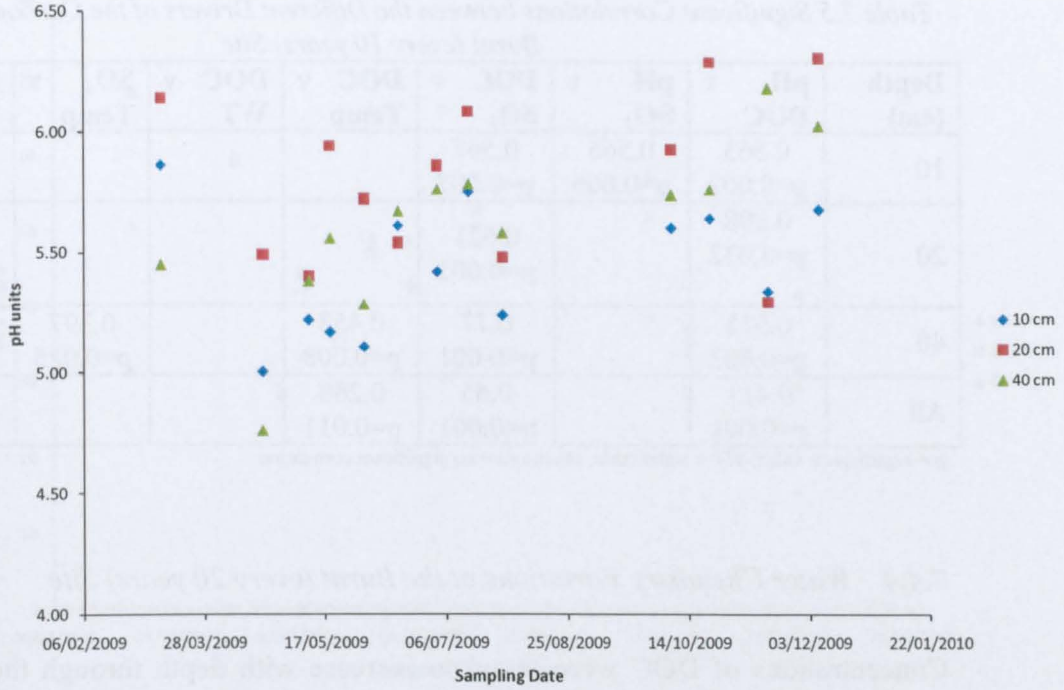


Figure 7-5 Variations in the pH of Peat Solution Samples Collected from the Burnt (every 10 years) Site from Three Depths

Table 7.4 Significance Values Indicating where Significant Differences in Values Occur between Treatments and the Burnt (every 10 years) Site

	B20	G	BG10	BG20	D	F	U
DOC all depths	0.039	0.013	<0.001	<0.001	<0.001	<0.001	
DOC (10 cm)			0.049	0.024		<0.001	
DOC (20 cm)	0.011	<0.001	<0.001	<0.001	<0.001	<0.001	
DOC (40 cm)			0.018		0.001	<0.001	
SO_4 (all depths)		0.031	0.005	0.021			0.012
SO_4 10 cm							
SO_4 20 cm			0.011	<0.001			
SO_4 40 cm		0.014					
pH (all depths)		<0.001	<0.001	0.003	<0.001	<0.001	<0.001
pH 10 cm		0.002	<0.001	0.004	<0.001	0.006	0.002
pH 20 cm		<0.001	<0.001	<0.001		<0.001	
pH 40 cm		<0.001	<0.001				0.003
Water table levels			0.0148		<0.001	<0.001	<0.001

Blank – no significant difference. b20 – burnt every 20 years, g – grazed, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, d- drained, f – afforested, u – unmanaged.

Table 7.5 Significant Correlations between the Different Drivers of the Carbon Cycle for the Burnt (every 10 years) Site

Depth (cm)	**pH v DOC**	**pH v SO_4**	**DOC v SO_4**	**DOC v Temp**	**DOC v WT**	**SO_4 v Temp**	**SO_4 v WT**
10	0.563 p=0.002	0.566 p=0.006	0.567 p=0.007				
20	0.398 p=0.032		0.621 p=0.003				
40	0.515 p=0.003		0.77 p<0.001	0.453 p=0.008		0.397 p=0.025	
All	0.411 p<0.001		0.65 p<0.001	0.268 p=0.011			

p = significance value, WT = water table, blank cell = no significant correlation

7.4.4 Water Chemistry Variations at the Burnt (every 20 years) Site

Concentrations of DOC were found to increase with depth through the peat profile (Figure 7-6), however significant differences in concentrations were only identified between 20 and 40 cm beneath the surface (p=0.002). No significant relationships between temperature, or water chemistry with DOC were identified, as illustrated in (Table 7.7). Similarly, concentrations of sulphate increased with depth (Figure 7-7), although the differences between each depth were not statistically significant. At 20 cm beneath the surface of the peat, the peat solution was found to be more alkaline than at 10 cm and 40 cm (Figure 7-8). Mean DOC concentrations were found to be significantly lower than all other treatments with the exception of the grazed and unmanaged sites, where concentrations were not significantly different (Table 7.6). Mean sulphate concentrations were found to be significantly lower than the afforested and the burnt and grazed (every 20 years) sites. Mean peat solution was found to significantly less acidic than the afforested site (p<0.001) and more acidic than the grazed (p<0.001), burnt and grazed (every 10 years) (p<0.001) and drained (p=0.05) sites. Less acidic conditions were found to coincide with increased DOC and sulphate concentrations.

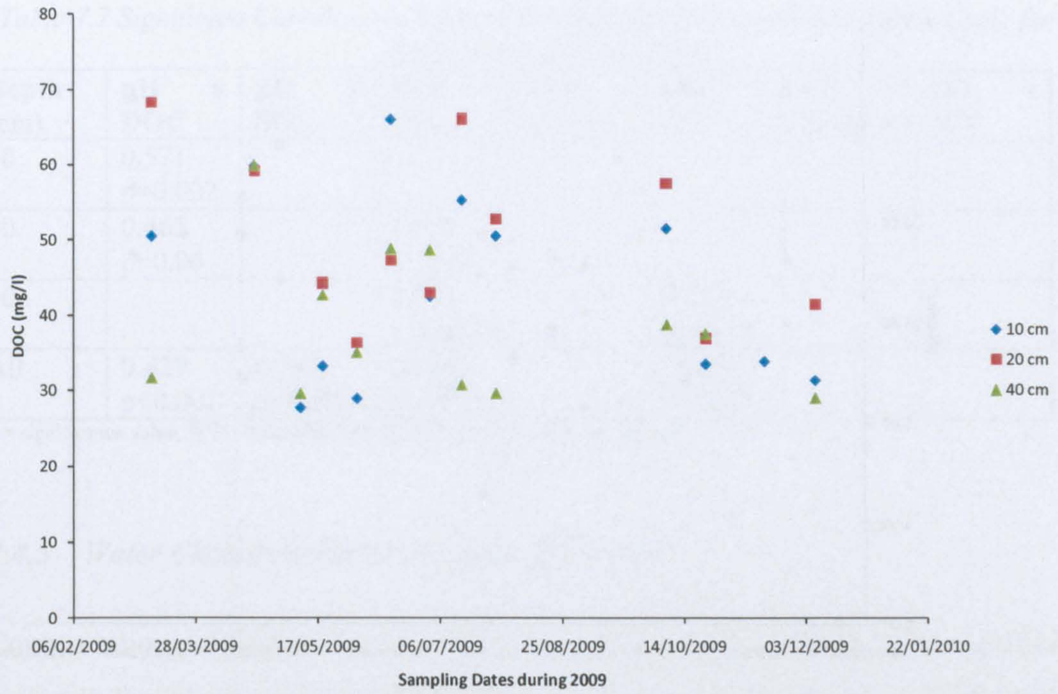


Figure 7-6 Variations in DOC Concentrations from Peat Solution Samples Collected from the Burnt (every 20 years) Site from Three Depths

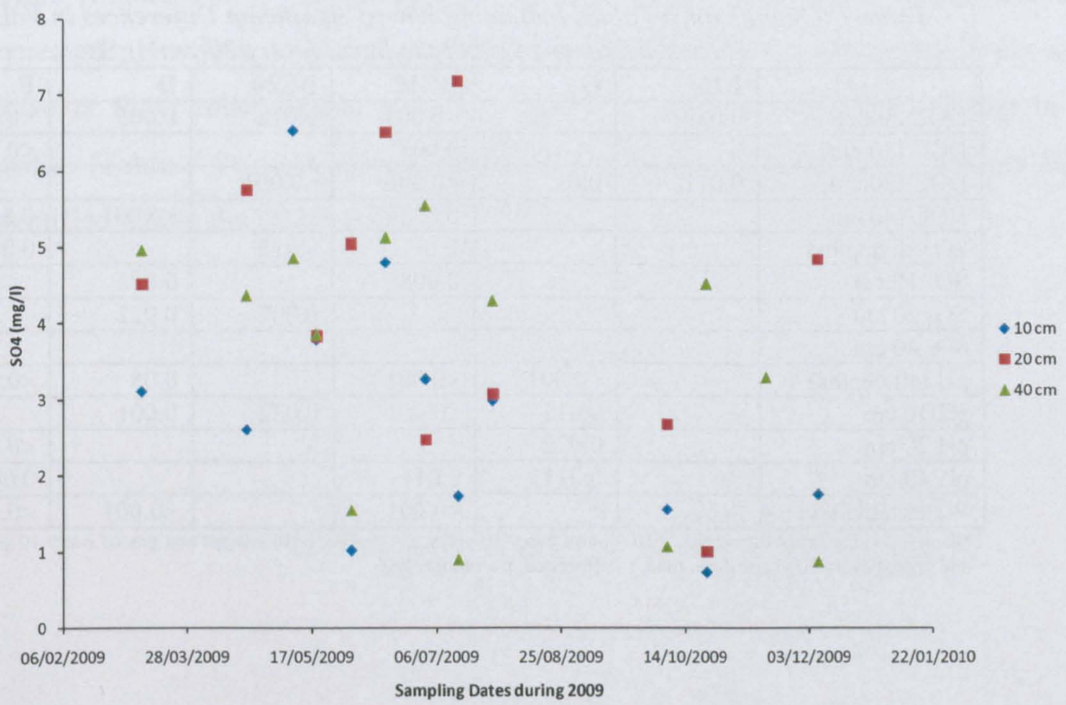


Figure 7-7 Variations in Sulphate Concentrations from Peat Solution Samples Collected from the Burnt (every 20 years) Site from Three Depths

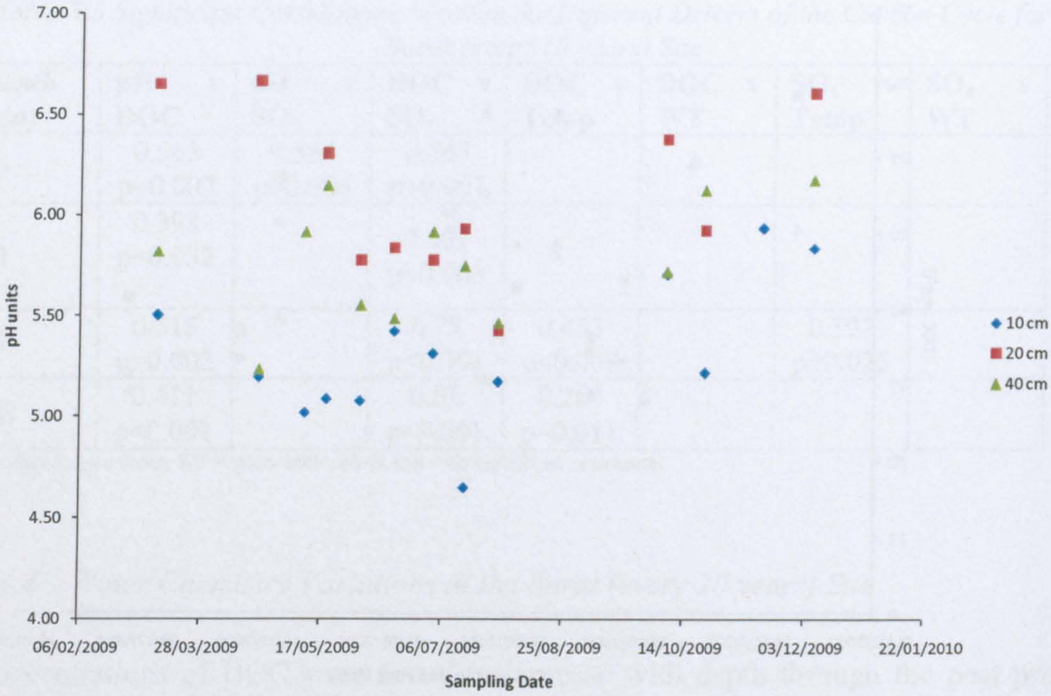


Figure 7-8 Variations in the pH of Peat Solution Samples Collected from the Burnt (every 20 years) Site from Three Depths

Table 7.6 Significance Values Indicating Where Significant Differences in Values Occur between Treatments and the Burnt (every 20 years) Site

	B10	**G**	**BG10**	**BG20**	**D**	**F**	**U**
DOC all depths	0.039		<0.001	0.018	0.005	<0.001	
DOC (10 cm)			0.007			<0.001	
DOC (20 cm)	0.011	0.01	<0.006	<0.001			
DOC (40 cm)			0.003		<0.001	0.006	
SO (all depths)				0.016		0.007	
SO_4 10 cm			0.008		0.023		
SO_4 20 cm				0.007	0.022		
SO_4 40 cm							
pH (all depths)		<0.001	<0.001		0.05	<0.001	
pH 10 cm		0.011	<0.001	0.022	0.001		0.006
pH 20 cm		0.022				<0.001	0.050
pH 40 cm		0.011	0.014			0.008	
Water table level			<0.001		<0.001	<0.001	<0.001

Blank – no significant difference. b10 – burnt every 10 years, g – grazed, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, d- drained, f - afforested, u – unmanaged

Table 7.7 Significant Correlations between the Different Drivers of the Carbon Cycle for the Burnt (every 20 years) Site

Depth (cm)	**pH v DOC**	**pH v SO_4**	**DOC v SO_4**	**DOC v Temp**	**DOC v WT**	**SO_4 v Temp**	**SO_4 v WT**
10	0.571 p=0.002						
20	0.465 p=0.06		0.624 p=0.01				
40			0.639 p=0.002		0.555 p=0.011		
All	0.429 p<0.001	0.343 p=0.009	0.528 p<0.001		-0.302 p=0.015		

p = significance value, WT = water table, blank cell = no significant correlation

7.4.5 Water Chemistry Variations at the Grazed Site

Concentrations of DOC (Figure 7-9) at the grazed site were found to be significantly higher in the 20 cm zone than the 10 and 40 cm zones as illustrated in Table 7.8. No significant differences in sulphate concentrations were identified with depth (Figure 7-10), however the 20 cm zone was found to be significantly less acidic than other layers (Figure 7-11). Increased DOC concentrations were found to coincide with increased pH values, sulphate concentrations and air temperatures in the 10 cm zone, however these relationships were not identified as being significant deeper in the profile (Table 7.9), with the exception of increased sulphate concentrations being identified within the 20 cm zone.

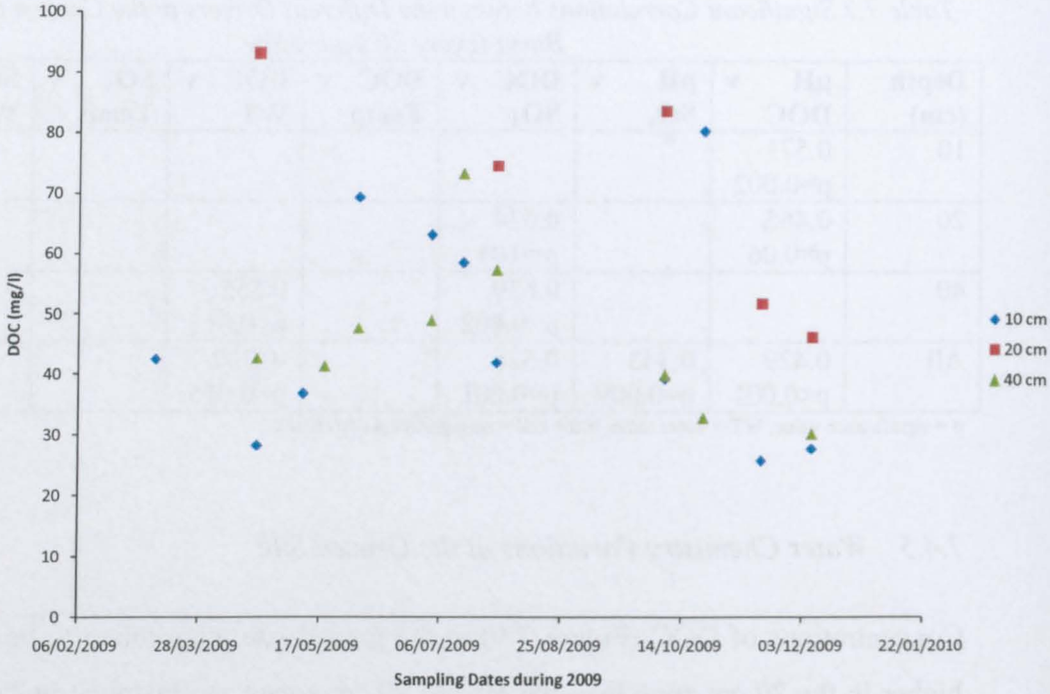


Figure 7-9 Variations in DOC Concentrations from Peat Solution Samples Collected from the Grazed Site from Three Depths

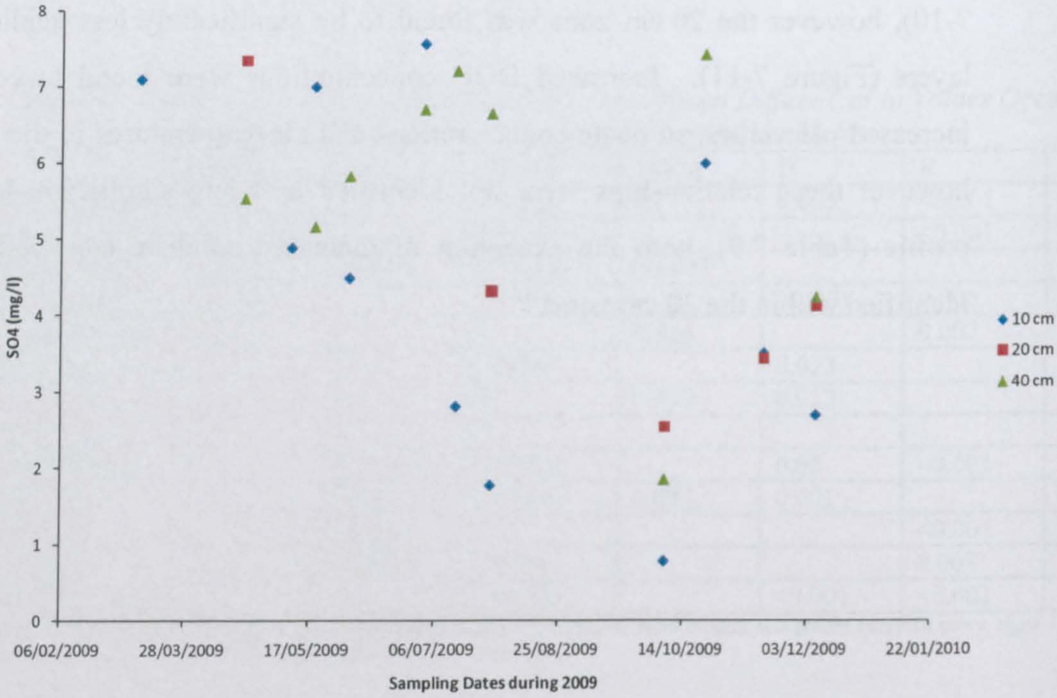


Figure 7-10 Variations in Sulphate Concentrations from Peat Solution Samples Collected from the Grazed Site from Three Depths

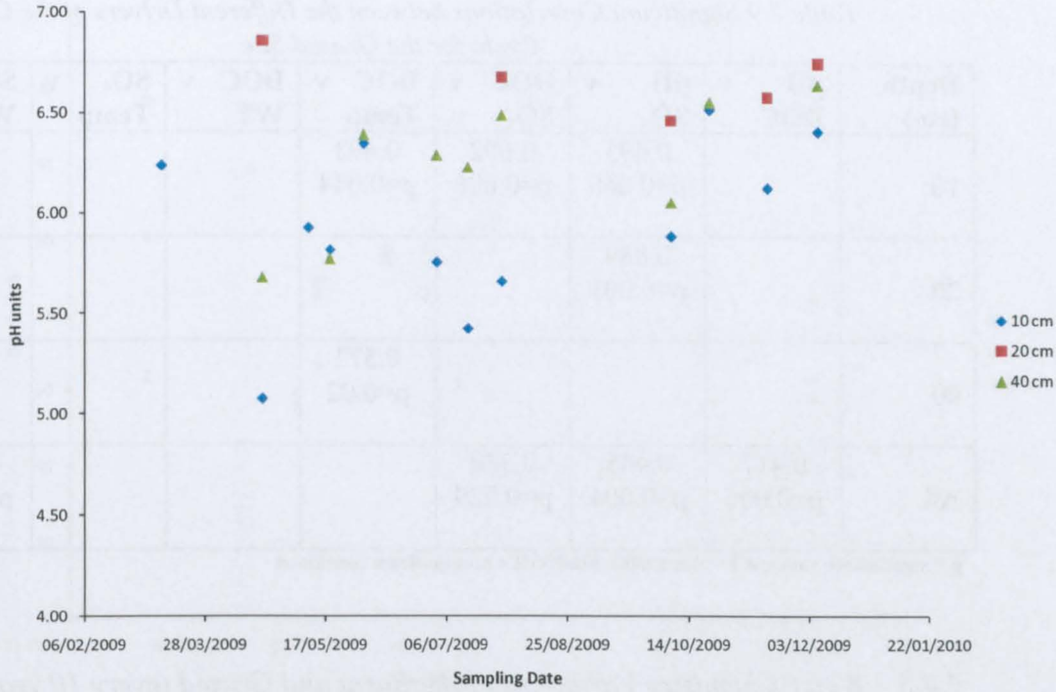


Figure 7-11 Variations in the pH of Peat Solution Samples Collected from the Grazed Site from Three Depths

Table 7.8 Significance Values Indicating where Significant Differences in Values Occur between Treatments and the Grazed Site

	B10	B20	BG10	BG20	D	F	U
DOC all depths	0.013		0.027			<0.001	
DOC (10 cm)			0.015			0.001	
DOC (20 cm)	<0.001	0.01					
DOC (40 cm)			0.048		0.015	<0.001	
SO_4 (all depths)	0.031						
SO_4 10 cm							
SO_4 20 cm				0.003			
SO_4 40 cm	0.014	0.007			0.043		0.057
pH (all depths)	<0.001	<0.001		0.002	0.007	<0.001	0.05
pH 10 cm	0.002	0.011	0.007			<0.001	
pH 20 cm	<0.001	0.022			<0.001	<0.001	0.002
pH 40 cm	<0.001	0.011			0.001	<0.001	
Water table level			0.005		<0.001	<0.001	<0.001

Blank – no significant difference. b10 – burnt every 10 years, b20 – burnt every 20 years, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, d- drained, f - afforested, u – unmanaged.

Table 7.9 Significant Correlations between the Different Drivers of the Carbon Cycle for the Grazed Site

Depth (cm)	pH v DOC	pH v SO_4	DOC v SO_4	DOC v Temp	DOC v WT	SO_4 v Temp	SO_4 v WT
10		0.493 p=0.066	0.692 p=0.006	0.493 p=0.044			
20		0.889 p=0.001					
40				0.573 p=0.02			
All	0.417 p=0.006	0.445 p=0.004	0.350 p=0.029				-0.348 p=0.024

p = significance value, WT = water table, blank cell = no significant correlation

7.4.6 Water Chemistry Variations at the Burnt and Grazed (every 10 years) Site

Significantly lower concentrations of DOC were identified in the 40 cm zone compared to the 10 cm and 20 cm zones (Figure 7-12). The highest concentrations of DOC were identified in the 20 cm zone (mean 77.43 mg L-1) where conditions were least acidic (mean 6.6). No significant relationship was identified between pH and DOC at this depth however (r2=0.-0.183, p=0.694). Concentrations of sulphate were also greatest in the 20 cm zone (Figure 7-13), however no significant relationship was found to exist between sulphate and DOC (r2=0.200, p=0.606). A positive correlation was identified between DOC and sulphate, and pH and temperature in the 40 cm zone (Table 7.11).

Concentrations of DOC were found to be significantly higher than the burnt (every 10 and 20 years) sites, grazed, and unmanaged sites, but were significantly lower than those recorded for the afforested site (Table 7.11). Concentrations of sulphate were found to be lower than the burnt (every 10 years) site, otherwise no significant differences in concentrations were identified when compared to other treatments. The peat solution was found to be significantly less acidic than all other treatments except for the burnt and grazed (every 20 years) and grazed sites, which did not have significantly different pH values.

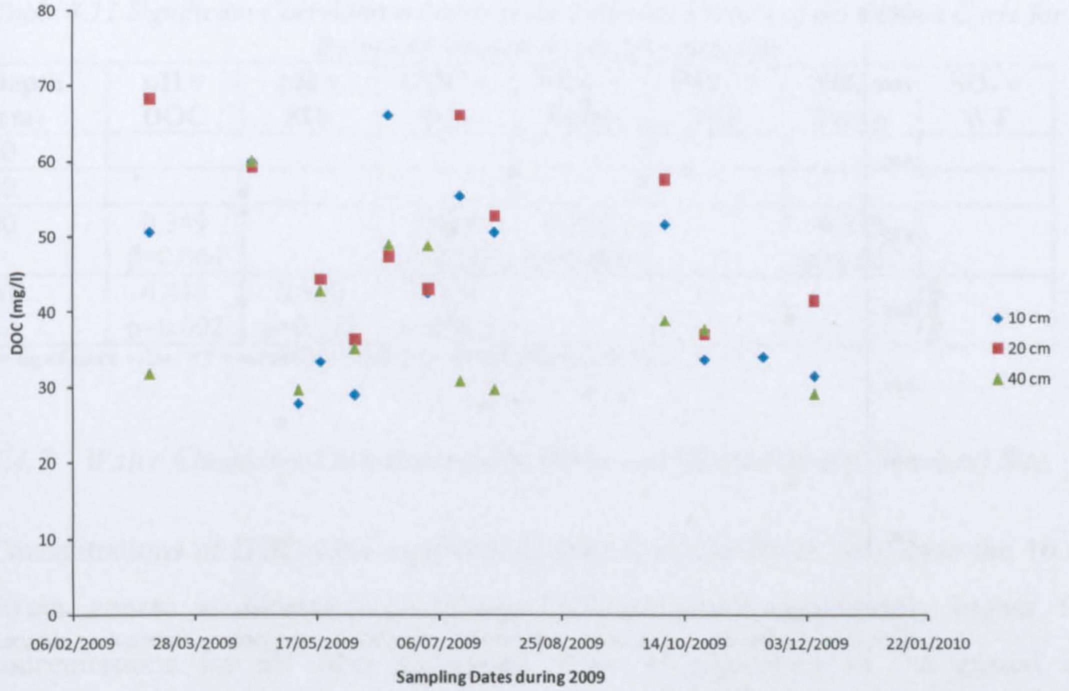


Figure 7-12 Variations in DOC Concentrations from Peat Solution Samples Collected from the Grazed and Burnt (every 10 years) Site from Three Depths

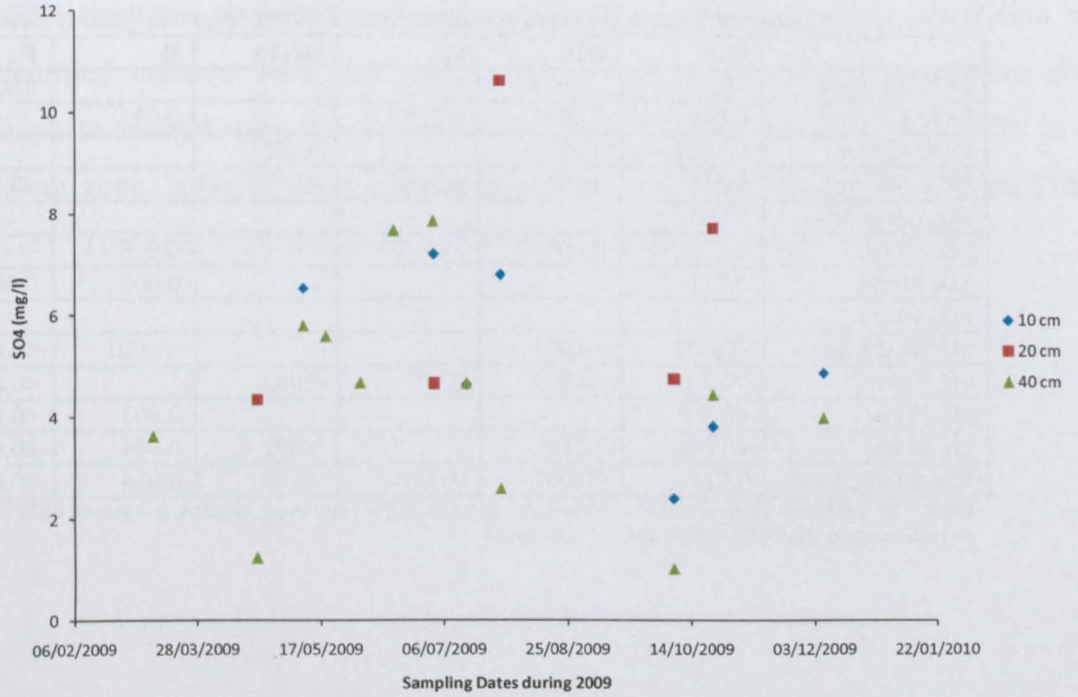


Figure 7-13 Variations in Sulphate Concentrations from Peat Solution Samples Collected from the Grazed and Burnt (every 10 years) Site from Three Depths

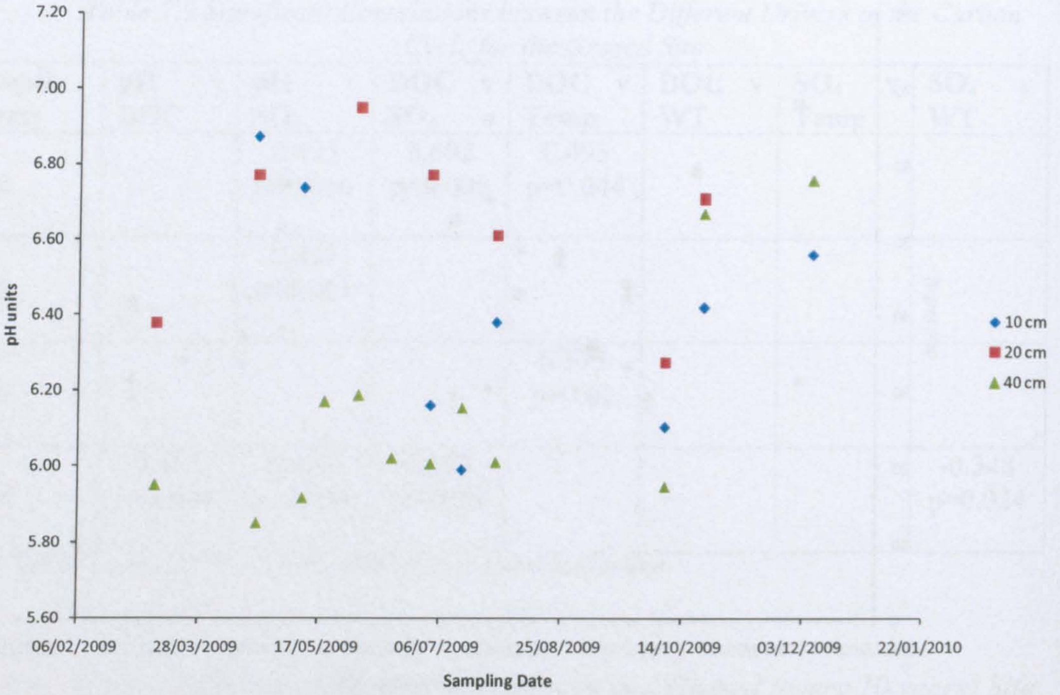


Figure 7-14 Variations in the pH of Peat Solution Samples Collected from the Grazed and Burnt (every 10 years) Site from Three Depths

Table 7.10 Significance Values Indicating where Significant Differences in Values Occur between Treatments and the Grazed and Burnt (every 10 years) Site

	B10	**B20**	**G**	**BG20**	**D**	**F**	**U**
DOC all depths	<0.001	<0.001	0.027			0.033	<0.001
DOC (10 cm)	0.049	0.007	0.015		0.031		0.011
DOC (20 cm)	<0.001	0.006		0.052			0.004
DOC (40 cm)	0.018	0.003	0.048	0.039		<0.001	0.007
SO_4 (all depths)	0.005						
SO_4 10 cm		0.008			0.023		
SO_4 20 cm	0.011				0.003	0.001	<0.001
SO_4 40 cm							
pH (all depths)	<0.001	<0.001			<0.001	<0.001	0.007
pH 10 cm	<0.001	<0.001	0.007	0.003		<0.001	
pH 20 cm	<0.001				0.003	<0.001	0.003
pH 40 cm	<0.001	0.014		0.001	0.004	<0.001	
Water table level	0.014	<0.001	0.005	<0.001	0.014	<0.001	

Blank – no significant difference. b10 – burnt every 10 years, b20 – burnt every 20 years, g – grazed, bg20 – burnt and grazed every 20 years, d- drained, f - afforested, u – unmanaged

Table 7.11 Significant Correlations between the Different Drivers of the Carbon Cycle for the Burnt and Grazed (every 10 years) Site

Depth (cm)	pH v DOC	pH v SO_4	DOC v SO_4	DOC v Temp	DOC v WT	SO_4 v Temp	SO_4 v WT
10							
20							
40	0.349 p=0.064		0.445 p=0.029	0.483 p=0.008		0.386 p=0.057	
All	0.418 p=0.002	0.322 p=0.033	0.364 p=0.016				

p= significance value, WT = water table, blank cell = no significant correlation

7.4.7 Water Chemistry Variations at the Burnt and Grazed (every 20 years) Site

Concentrations of DOC were significantly higher in the 20 cm zone than the 10 and 40 cm zones, as illustrated in Table 7.12 and were significantly higher than concentrations for all other treatments with the exception of the grazed site. Concentrations of sulphate were also found to be significantly ($p<0.001$) higher in the 20 cm zone (mean 11.09 mg L^{-1}) compared to the 10 and 40 cm zones (3.68 and 4.61 mg L^{-1} respectively). The peat solution in the 20 cm zone was significantly more acidic than the 10 and 40 cm zones ($p=0.01$). A strong positive correlation was identified between DOC and pH, sulphate and temperature in the 20 cm zone. Sulphate and pH were also found to have a significant correlation with DOC in the 10 cm zone, however such relationships were not found in the 40 cm zone (Table 7.13). Trends in DOC, sulphate and pH are presented on Figures 7.15 to 7.17.

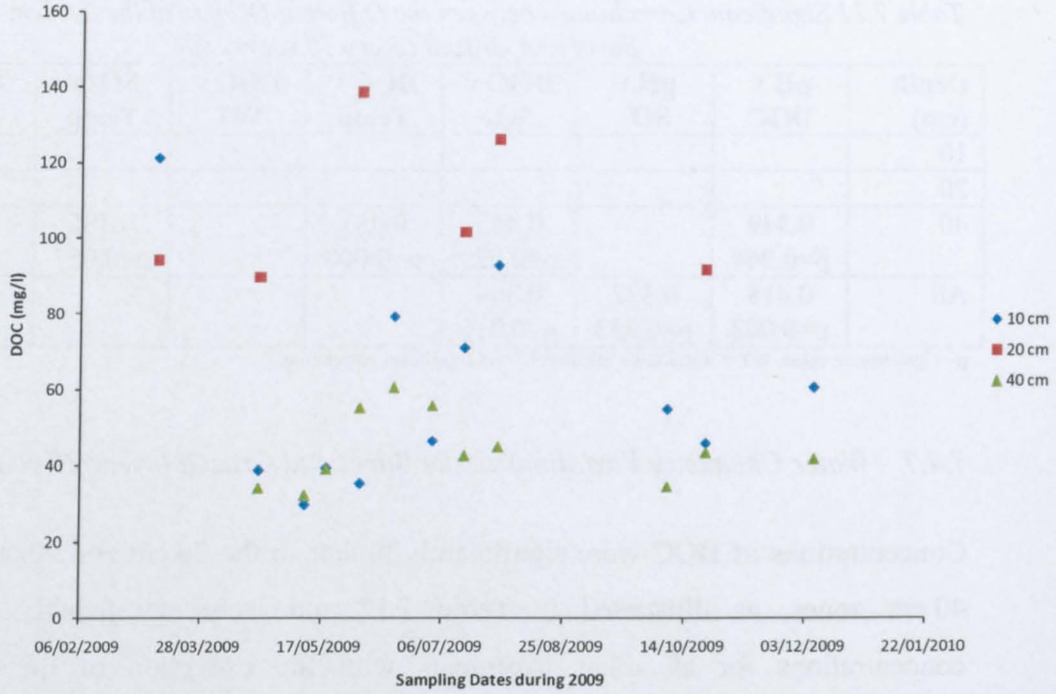


Figure 7-15 Variations in DOC Concentrations from Peat Solution Samples Collected from the Grazed and Burnt (every 20 years) Site from Three Depths

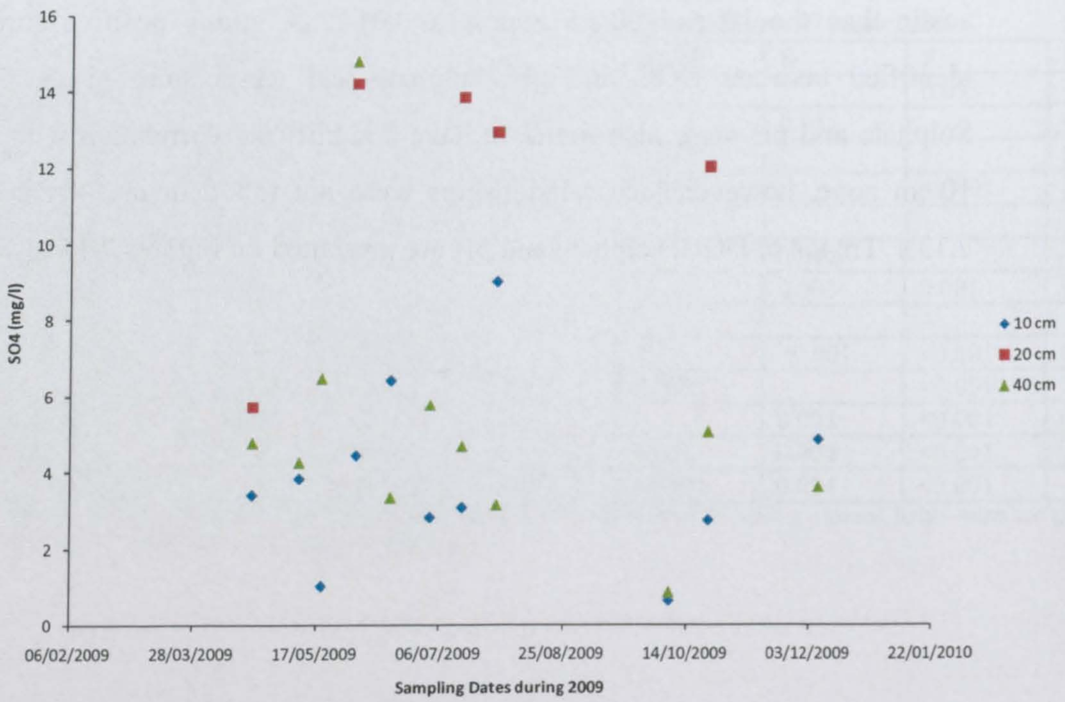


Figure 7-16 Variations in Sulphate Concentrations from Peat Solution Samples Collected from the Grazed and Burnt (every 20 years) Site from Three Depths

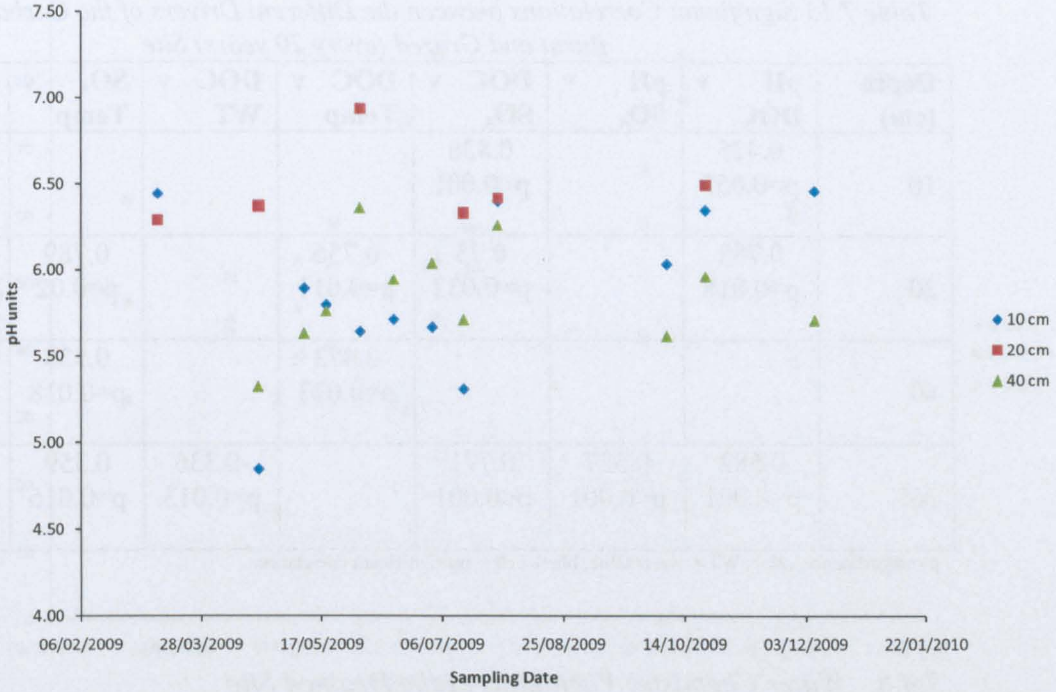


Figure 7-17 Variations in the pH of peat solution samples collected from the grazed and burnt (every 20 years) site from three depths

Table 7.12 Significance Values Indicating where Significant Differences in Values Occur between Treatments and the Grazed and Burnt (every 20 years) Site

	B10	**B20**	**G**	**BG10**	**D**	**F**	**U**
DOC all depths	<0.001	0.018				<0.001	<0.001
DOC (10 cm)	0.024					0.013	
DOC (20 cm)	<0.001	<0.001		0.052	0.001	0.001	<0.001
DOC (40 cm)				0.039	0.008	<0.001	
SO_4 (all depths)							
SO_4 10 cm							
SO_4 20 cm							
SO_4 40 cm							
pH (all depths)	0.003		0.002	<0.001		<0.001	
pH 10 cm	0.004	0.022		0.003		<0.001	
pH 20 cm	<0.001		0.022		0.039	<0.001	0.035
pH 40 cm			0.001	0.001		0.013	0.05
Water table level				<0.001	<0.001	<0.001	<0.001

Blank – no significant difference. b10 – burnt every 10 years, b20 – burnt every 20 years, g – grazed, bg10 – burnt and grazed every 10 years, d- drained, f- afforested, u – unmanaged.

Table 7.13 Significant Correlations between the Different Drivers of the Carbon Cycle for the Burnt and Grazed (every 20 years) Site

Depth (cm)	**pH v DOC**	**pH v SO_4**	**DOC v SO_4**	**DOC v Temp**	**DOC v WT**	**SO_4 v Temp**	**SO_4 v WT**
10	0.425 p=0.055		0.836 p<0.001				
20	0.758 p=0.018		0.75 p=0.032	0.756 p=0.011		0.789 p=0.02	
40				0.473 p=0.023		0.536 p=0.018	0.478 p=0.039
All	0.582 p<0.001	0.527 p<0.001	0.771 p<0.001		-0.336 p=0.013	0.359 p=0.016	

p = significance value, WT = water table, blank cell = no significant correlation

7.4.8 Water Chemistry Variations at the Drained Site

Concentrations of DOC (Figure 7-18) were significantly higher in the 20 cm zone than the 10 cm zone (p=0.04). The peat solution became increasingly acidic with depth, differences between the 10 and 40 cm zones and 20 and 40 cm zones were identified as significant (p=0.006 and p=0.012 respectively). Concentrations of sulphate (Figure 7-19) were highest in the 20 cm zone (mean 4.93 mg L-1), however the differences were not significant (p=0.558). The 20 cm zone was also the least acidic (Figure 7-20). Concentrations of DOC were significantly higher than the burnt (every 10 and 20 years) and unmanaged sites, and significantly lower than the afforested site. Concentrations of sulphate were not significantly different to other treatments. The burnt sites (every 10 and 20 years) sites had a significantly more acidic peat solution than the drained site. The grazed, burnt and grazed (every 10 years) and afforested sites had a significantly less acidic peat solution than the drained site (Table 7.14). For all three depths examined, no relationship was identified between pH and DOC, however increased sulphate concentrations were associated with increased DOC concentrations, although the strength of the correlation decreased with depth as demonstrated in Table 7.15

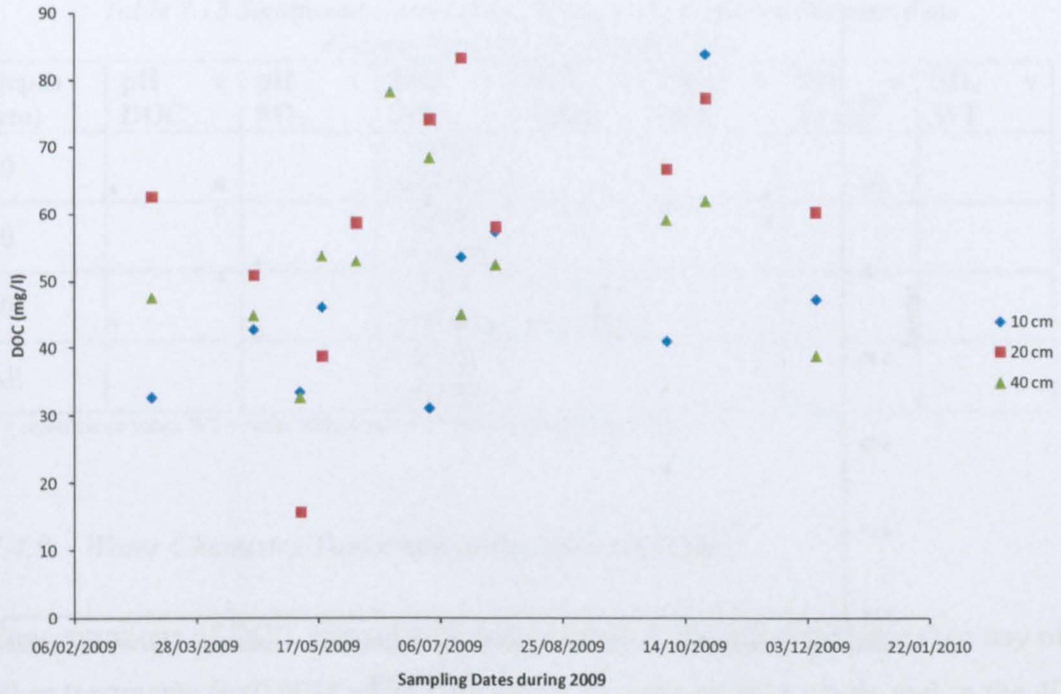


Figure 7-18 Variations in DOC Concentrations from Peat Solution Samples Collected from the Drained Site from Three Depths

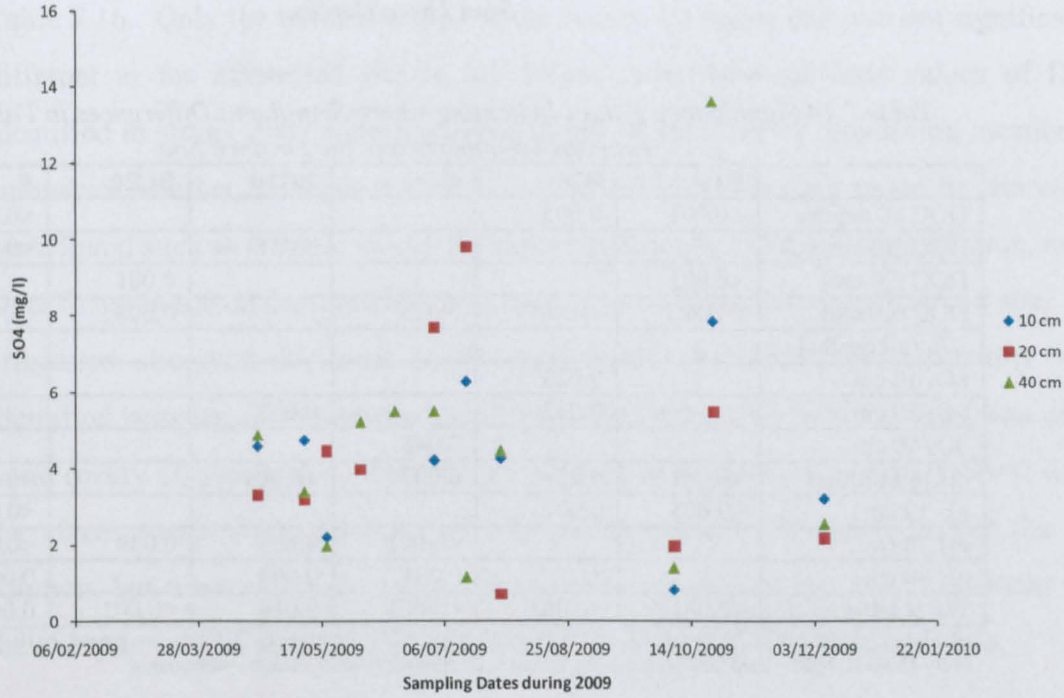


Figure 7-19 Variations in Sulphate Concentrations from Peat Solution Samples Collected from the Drained Site from Three Depths

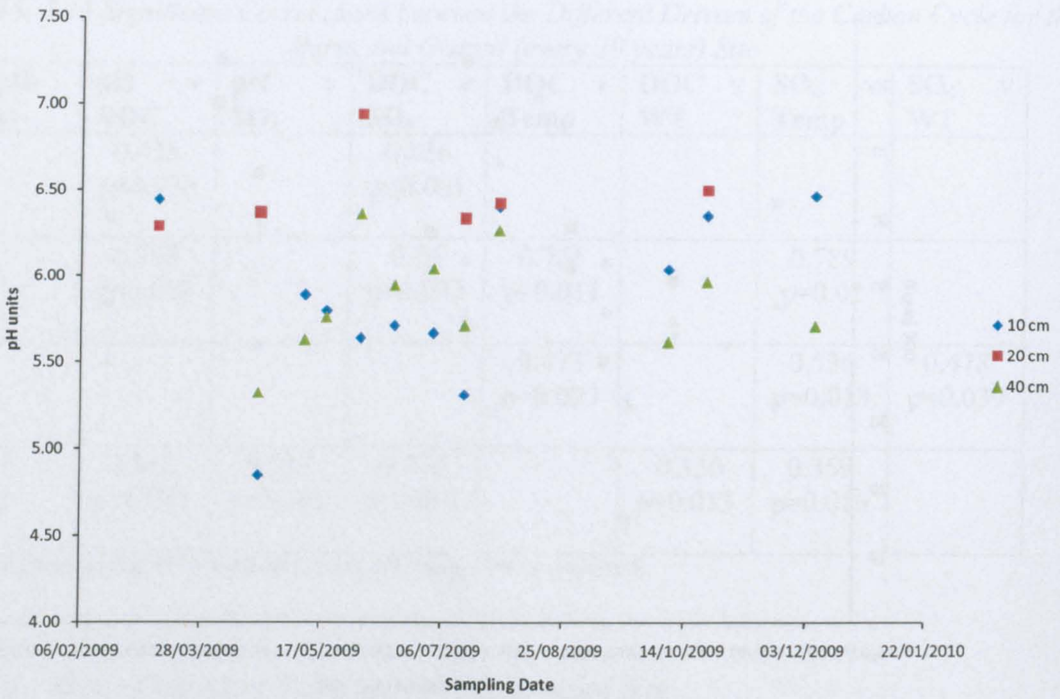


Figure 7-20 Variations in the pH of Peat Solution Samples Collected from the Drained Site from Three Depths

Table 7.14 Significance Values Indicating where Significant Differences in Values Occur between Treatments and the Drained Site

	B10	B20	G	BG10	BG20	F	U
DOC all depths	<0.001	0.005				<0.001	<0.001
DOC (10 cm)				0.031		<0.001	
DOC (20 cm)	<0.001				0.001		0.018
DOC (40 cm)	0.004	<0.001	0.015		0.008	0.002	0.003
SO_4 (all depths)							
SO_4 10 cm		0.023					
SO_4 20 cm		0.022		0.003			
SO_4 40 cm			0.043				
pH (all depths)	<0.001	0.05	0.007	<0.001		<0.001	
pH 10 cm	<0.001	0.001				<0.001	
pH 20 cm			<0.001	0.003	0.039	<0.001	
pH 40 cm			0.001	0.004		0.004	
Water table level	<0.001	<0.001	<0.001	0.014	<0.001	0.041	

Blank – no significant difference. b10 – burnt every 10 years, b20 – burnt every 20 years, g – grazed, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, d- drained, f - afforested, u – unmanaged

Table 7.15 Significant Correlations between the Different Drivers of the Carbon Cycle for the Drained Site

Depth (cm)	pH v DOC	pH v SO_4	DOC v SO_4	DOC v Temp	DOC v WT	SO_4 v Temp	SO_4 v WT
10			0.804 $p<0.001$				
20			0.602 $p=0.008$				
40			0.566 $p=0.004$	0.398 $p=0.033$			
All			0.524 $p<0.001$				

P = significance value, WT = water table, blank cell = no significant correlation

7.4.9 Water Chemistry Variations at the Afforested Site

Concentrations of DOC were significantly higher in the afforested site than any of the other treatments ($p<0.001$) when comparing the data set as a whole and in the 40 cm zone (although differences were not identified in the 10 and 20 cm zones) as shown in Table 7.16. Only the burnt and grazed site (every 10 years) site was not significantly different to the afforested site in the 10 cm zone.Mean elevated values of DOC identified in spring 2009 were identified in the 40 cm zone of monitoring location 22 compatred to other locations within the afforested site. No clear cause for the values were found such as unusual values for water table depth, sulphate concentration, or pH were found based on comparisons with other data collected for the afforested site. The afforested site was the most acidic, and significant differences in acidity were identified between all treatments for all the layers examined with the exception of the burnt (every 10 years) site within the 40 cm zone and the burnt (every 20 years) site in the 10 cm zone. Concentrations of sulphate increased with depth through the peat solution, but were not found to be significantly different to any other treatment with the exception of the burnt and grazed (every 20 years) site for the 20 cm zone.

Correlation data presented in Table 7.17 indicate that a negative correlation exists between DOC and sulphate within the 10 cm zone, however, no other correlations were identified within this zone. Within the 20 cm zone, a negative correlation was identified between pH and sulphate. A summary of trends in DOC, sulphate and pH data for all three depths is presented in Figure 7-21, Figure 7-22 and Figure 7-23 respectively.

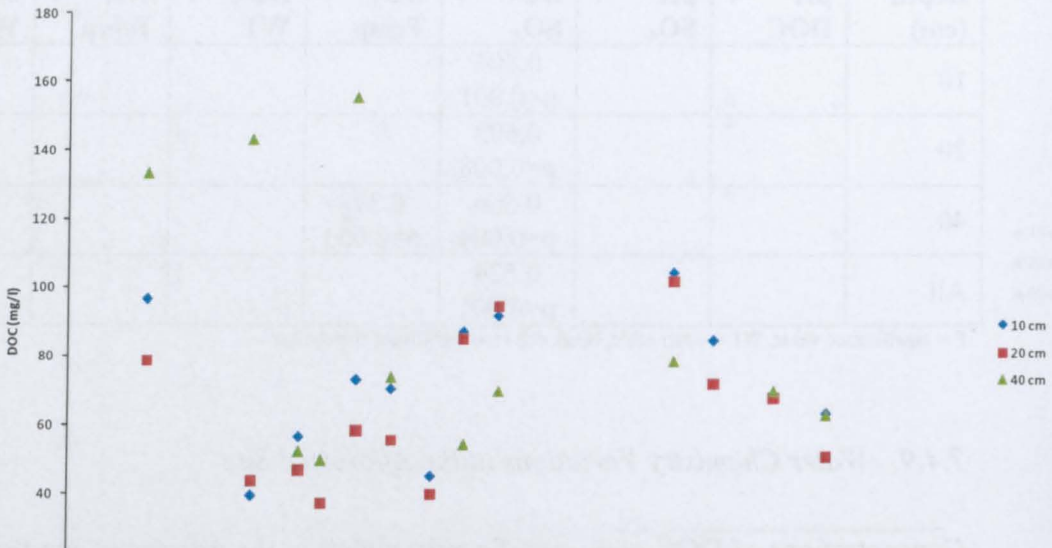


Figure 7-21 Variations in DOC Concentrations from Peat Solution Samples Collected from the Afforested Site from Three Depths

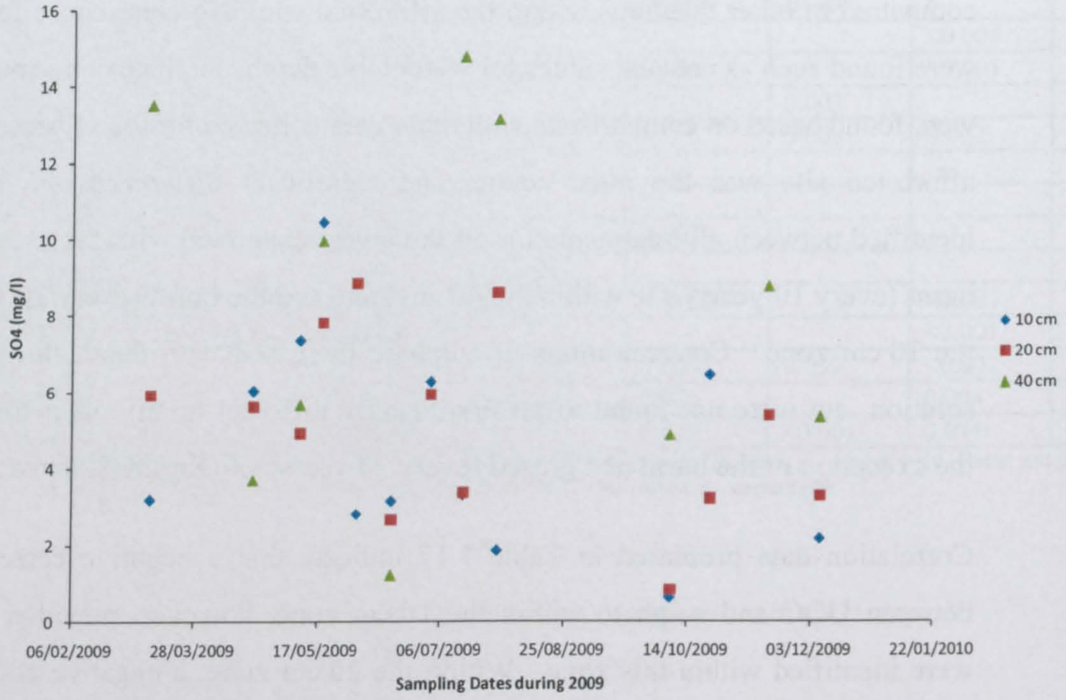


Figure 7-22 Variations in Sulphate Concentrations from Peat Solution Samples Collected from the Afforested Site from Three Depths

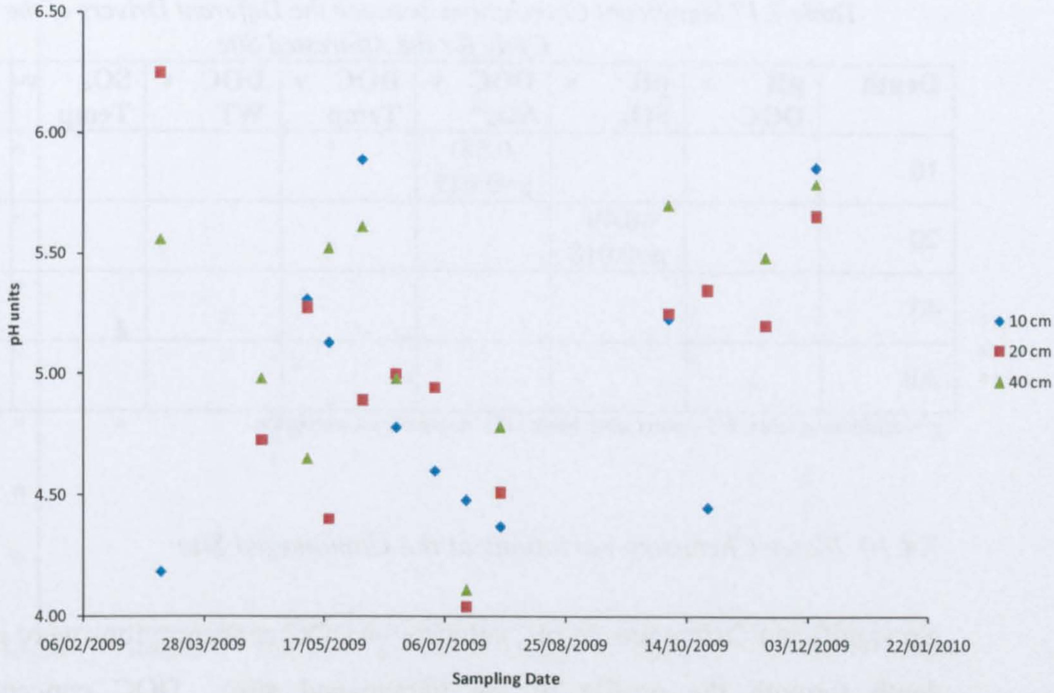


Figure 7-23 Variations in the pH of Peat Solution Samples Collected from the Afforested Site from Three Depths

Table 7.16 Significance Values Indicating where Significant Differences in Values Occur between Treatments and the Afforested Site

	B10	B20	G	BG10	BG20	D	U
DOC all depths	<0.001	<0.001	<0.001	0.033	<0.001	<0.001	<0.001
DOC (10 cm)	<0.001	<0.001	0.001		0.013	<0.001	<0.001
DOC (20 cm)	<0.001				0.001		0.007
DOC (40 cm)	<0.001	0.006	<0.001	<0.001	<0.001	0.002	<0.001
SO_4 (all depths)	0.012	0.007					0.028
SO_4 10 cm							
SO_4 20 cm				0.001			
SO_4 40 cm							
pH (all depths)	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
pH 10 cm	0.006		<0.001	<0.001	<0.001	<0.001	<0.001
pH 20 cm	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
pH 40 cm		0.008	<0.001	0.013	<0.001	<0.001	0.004
Water table level	<0.001	<0.001	<0.001	<0.001	<0.001	0.041	0.013

Blank – no significant difference. b10 – burnt every 10 years, b20 – burnt every 20 years, g – grazed, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, d- drained, u – unmanaged

Table 7.17 Significant Correlations between the Different Drivers of the Carbon Cycle for the Afforested Site

Depth	**pH v DOC**	**pH v SO_4**	**DOC v SO_4**	**DOC v Temp**	**DOC v WT**	**SO_4 v Temp**	**SO_4 v WT**
10			-0.581 p=0.047				
20		-0.49 p=0.018					
40							0.547 p=0.023
All							-0.410 p=0.003

p = significance value, WT = water table, blank cell = no significant correlation

7.4.10 Water Chemistry Variations at the Unmanaged Site

No significant differences in pH, sulphate or DOC concentrations were identified with depth through the profile of the unmanaged site. DOC concentrations were significantly lower in the drained, afforested and both of the burnt and grazed sites (Table 7.18). Significant differences in sulphate concentrations were not identified between treatments for the separate layers with the exception of the burnt and grazed (every 10 years) which featured significantly higher concentrations of sulphate in the 20 cm zone. Increased DOC concentrations were found to coincide with higher air temperatures and sulphate concentrations within the 10 cm zone. A positive correlation was found between increased DOC concentrations and sulphate and higher pH values (Table 7.19). A summary of trends in DOC, sulphate and pH data for all three depths is presented in Figure 7-24, Figure 7-25 and Figure 7-26 respectively.

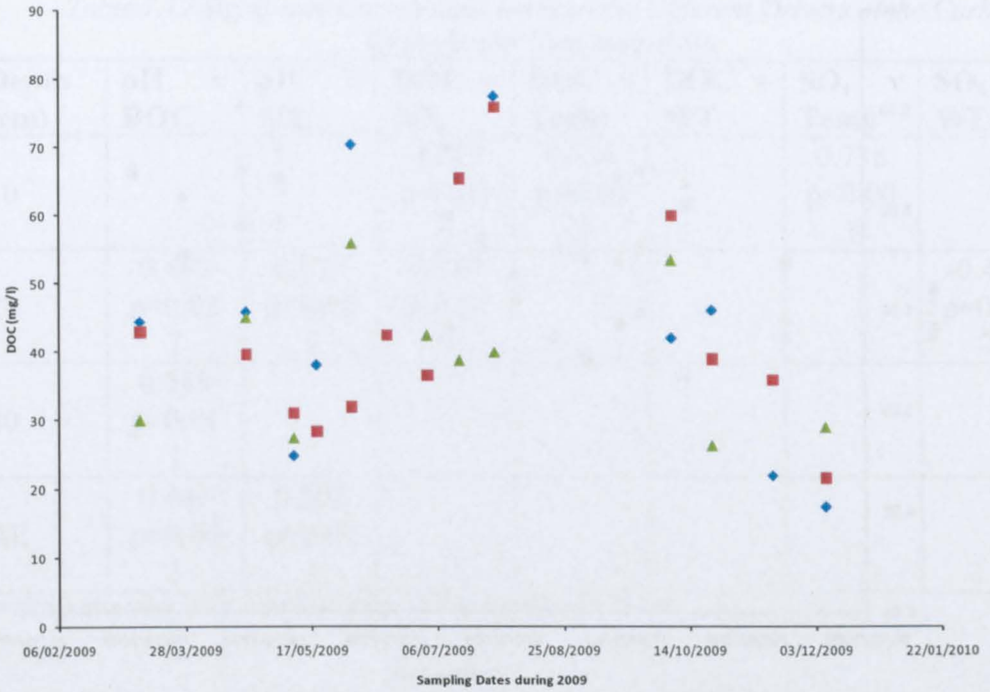


Figure 7-24 Variations in DOC Concentrations from Peat Solution Samples Collected from the Unmanaged Site from Three Depths

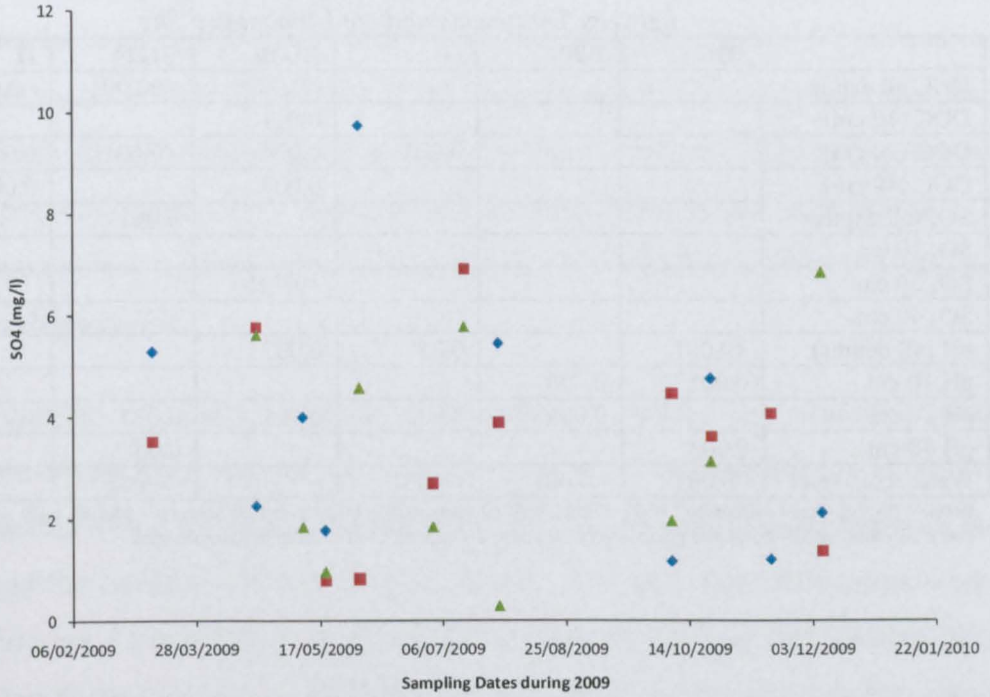


Figure 7-25 Variations in Sulphate Concentrations from Peat Solution Samples Collected from the Unmanaged Site from Three Depths

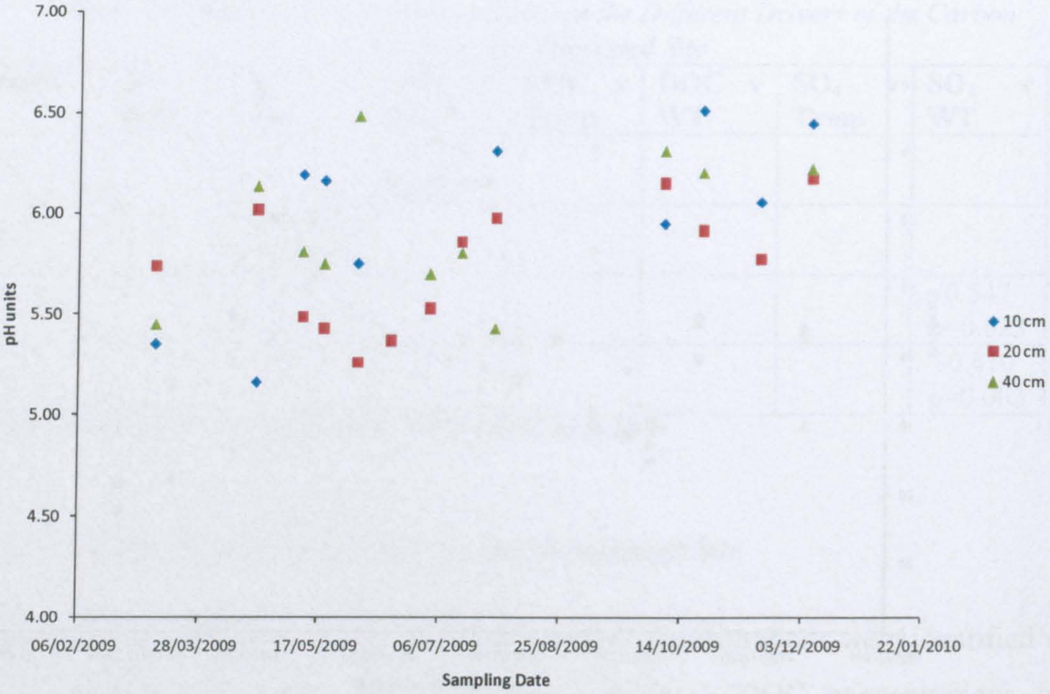


Figure 7-26 Variations in the pH of Peat Solution Samples Collected from the Unmanaged Site from Three Depths

Table 7.18 Significance Values Indicating where Significant Differences in Values Occur between Treatments and the Unmanaged Site

	B10	B20	G	BG10	BG20	D	F
DOC all depths				<0.001	<00.001	<0.001	<0.001
DOC (10 cm)				0.011			<0.001
DOC (20 cm)			0.003	0.004	<0.001	0.018	0.007
DOC (40 cm)				0.007		0.003	<0.001
SO_4 (all depths)					0.041		0.028
SO_4 10 cm							
SO_4 20 cm				<0.001			
SO_4 40 cm							
pH (all depths)	<0.001		0.05	0.007			<0.001
pH 10 cm	0.002	0.006					<0.001
pH 20 cm		0.005	0.002	0.003	0.035		<0.001
pH 40 cm	0.003				0.05		<0.001
Water table level	<0.001	<0.001	<0.001		<0.001		0.013

Blank – no significant difference. b10 – burnt every 10 years, b20 – burnt every 20 years, g – grazed, bg10 – burnt and grazed every 10 years, bg20 – burnt and grazed every 20 years, d- drained, f - afforested, u – unmanaged

Table 7.19 Significant Correlations between the Different Drivers of the Carbon Cycle for the Unmanaged Site

Depth (cm)	pH v DOC	pH v SO_4	DOC v SO_4	DOC v Temp	DOC v WT	SO_4 v Temp	SO_4 v WT
10			0.659 p=0.004	0.596 p=0.007		0.738 p<0.001	
20	0.483 p=0.027	0.777 p<0.001	0.547 p=0.019				-0.427 p=0.077
40	0.545 p=0.011						
All	0.447 p=0.003	0.502 p=0.001					

p= significance value, WT = water table, blank cell = no significant correlation

7.4.11 Influence of Soil Solution Sampling Depth on Soil Solution Chemistry

Soil solution samples were collected from depths of 10, 20 and 40 cm beneath the surface of the peat. No significant differences were identified between the different depths in the unmanaged site (p=0.690), the afforested site (p=0.213) and burnt (every 10 years) (p=0.851) sites for DOC. Significant differences were not identified with depth for pH values in either the afforested site (p=0.255) or the unmanaged site (p=0.429) sites where some of the deepest water levels were found. Table 7.20 summarises where significant differences were identified between depths sampled for each treatment.

Analysis of water chemistry with depth for differently managed peats has not previously been carried out. The data allow an examination of trends within three depths within the peat – the acrotelm (10 cm), the mixing zone between the acrotelm and the catotelm (20 cm) and the catotelm (40 cm). Few differences were identified between 10 and 20 cm or 10 and 40 cm beneath the peat, but between 20 and 40 cm significant decreases in pH values were identified for six of the eight treatments. In all cases, average pH values rose between 10 and 20 cm beneath the surface, as did DOC concentrations. Concentrations of DOC and pH values then fell below those values found at 10 cm and those in the 40 cm layer. Significant differences were not found in sulphate concentrations between the three depths examined (p=0.217). The results imply that rates of biogeochemical cycling are greatest in the 20 cm zone, which is area of the profile where fluctuations in the water table were also the greatest (Sundh et al. 1997).

Table 7.20 Significance Values for Changes in Water Chemistry between the Depths Analysed

Depths Beneath Peat (cm)	Element	Burnt and Grazed (20)	Grazed	Burnt and Grazed (10)	Drained	Burnt (20)	Burnt (10)
10 and 20	DOC	p<0.001	p=0.005	N/S	p=0.040	N/S	N/S
	Sulphate	N/S	N/S	N/S	N/S	N/S	N/S
	pH	p=0.008	p<0.001	N/S	N/S	p=<0.001	p=0.003
10 and 40	DOC	N/S	N/S	p=0.036	N/S	N/S	N/S
	Sulphate	N/S	N/S	N/S	N/S	N/S	N/S
	pH	N/S	N/S	p=0.026	p=0.006	p=0.045	N/S
20 and 40	DOC	p<0.001	p<0.001	p=0.001	N/S	P=0.002	N/S
	Sulphate	N/S	N/S	N/S	N/S	N/S	N/S
	pH	p<0.001	p=0.004	p=0.003	p=0.012	p=0.023	p=0.023

N/S – not statistically significant

7.5 Discussion

7.5.1 The Effect of Land Management on DOC Production

Peatland management was expected to have a significant effect on concentrations of DOC in peat solution owing to changes in pore water chemistry and the peatland environment brought about by land management. Significant differences in DOC concentrations in peatlands were found to occur due to land management. Concentrations were measured in the peat solution to ensure that factors such as dilution and residence time within the catchment did not affect comparisons between the different treatments. Few studies have considered the actual concentrations of DOC in the peat solution, (with the notable exceptions of Wallage et al (2006), Wallage and Holden (2010), Ward et al. (2007), Worrall et al. (2007a) and Clay et al. (2010b)); most studies have looked at differences between concentrations in catchment streams. For the purposes of comparing the findings of this study and others, both peat solution and catchment studies will be referred to. Note should be taken that differences observed in catchments where fluxes were measured are not directly comparable to peat solution studies, but provide some indication of the effects of management. A discussion of the influence of land management on DOC concentrations in peat will follow the effects of land management on the drivers of DOC concentrations in peats and causes for differences in concentrations between treatments, will be discussed in section 7.5.2.

The results suggest that the peats at the drained and afforested sites had significant increases in DOC concentration compared to the unmanaged site. No significant differences in DOC concentration were found between the unmanaged site and the burnt site or the grazed site. Sites where combined burnt and grazed took place were found to be significantly different from the unmanaged site, and are discussed in section 7.5.5.

The findings for the grazed site supported those of Worrall et al. (2007a) who did not find any significant differences between grazed and ungrazed plots at Moor House during a study of monitoring wells placed at depths up to 90 cm beneath the peat. The grazed site was found to have significantly higher DOC concentrations than the burnt (every 10 years) site, in line with the findings of Clay et al. (2010a). Published comparisons of DOC concentrations for blanket peat with treatments other than burning or no management were not available.

Significant differences in pore water concentrations of DOC were not identified between the burnt and unmanaged site, in line with the findings of Ward et al. (2007) who completed prior to the 2007 burn. Work carried out by Clay et al. (2009b) demonstrated that burning only made a significant contribution to DOC concentrations in the period immediately following the burn, and that after one year, no significant differences can be found between burnt and unburnt sites. Yallop and Clutterbuck (2009) however, argued that burning was responsible for the increases in DOC concentrations observed in recent years in peatland catchments. Their study identified higher concentrations of DOC in streams where peats have been burnt recently. The authors argued that the removal of vegetation and drying of the peat results in greater microbial activity and hence increased DOC concentrations. Yallop and Clutterbuck

(2009) identified newly burnt peats as those having been burnt in the past four years. Whilst this has not been the case at Moor House as evidenced by this study and that of Ward et al (2007) and Clay et al. (2009b), little detailed data on burning has been published for other sites. This needs to be addressed in order to verify whether the findings for pore water samples at Moor House are a unique case. It is plausible that differences in the intensity of burning could be responsible for the disparity in the results. The catchments studied by Yallop et al. (2009) could have been burnt more intensively than the plots at Moor House which were only lightly burnt. This notion is supported by laboratory trials carried out by McDonald et al (1991) who identified higher concentrations of colour loss from peats that were burnt at higher temperatures.

The results showed that significantly higher concentrations of DOC existed in the pore waters of afforested peats compared to all other treatments. Larger concentrations of DOC in afforested peat were also observed by Grieve (1994) with concentrations ranging between 9.2 and 10.8 mg L^{-1} in streams flowing through afforested catchments compared to the unmanaged sites where concentrations averaged 7.6 mg L^{-1}. The concentrations are far lower than those measured at Moor House, but they were from streamwaters rather than the peat pore solution. The results however support the notion that afforestation increases DOC concentrations in peat solution. The results of a study of Scottish lakes and streams by Harriman et al (2003) also identified higher concentrations of DOC in afforested sites than moorland sites, however the results were attributed to the moorlands being located at higher altitude with less peat than the afforested site. In a previous study (Harriman et al. 2001), afforested sites were found to have lower DOC concentrations than moorland sites. The present study at Moor House involved collection of samples from similar elevation and sites within 3.5 km of one another, thus minimising confounding factors such as differences in climate, geology and altitude, therefore providing a fair basis for comparison between sites

The drained site was found to have significantly greater concentrations of DOC than the burnt and unmanaged sites, yet concentrations were significantly lower in the pore waters of the drained site compared to the afforested site. Wallage et al. (2006) also identified higher concentrations of DOC in drained peat pore waters (median 42.9 mg C L^{-1}) compared to intact peats (median 27.6 mg C L^{-1}). The differences between the sites at Moor House were much smaller – with an average concentration of 44.01 mg L^{-1} for the unmanaged site compared to 47.58 mg L^{-1} at the drained site, for shallow samples. For samples collected at 40 cm, values were 39.71 mg L^{-1} compared to 52.17 mg L^{-1} for the drained site. Wallage et al. (2006) also found concentrations to increase with depth, however the range of values only increased slightly between drained and undrained sites whereas at Moor House the range of values decreased with depth.

Gibson et al. (2009) also compared drained, blocked and pristine peats in northern England. The data were collected from streams by automatic samples, therefore would also have been diluted, thus giving a less clear picture of the actual effects of management on concentrations of DOC in the peat solution. High concentrations of DOC were identified in waters standing in the blocked drains, and a strong correlation was found between DOC concentration and both catchment size and discharge rates. The study concluded that drain blocking reduced rates of DOC release into catchments but has no influence on DOC production. Concentrations of DOC were compared between sites but comparisons of concentrations between drained and undrained peats were not possible as whole catchments were studied rather than pore water concentrations.

A short-term study at Whitendale, Forest of Bowland by Worrall et al. (2007b) did not identify significant differences in the DOC concentrations between blocked and unblocked drain systems. Concentrations in the drain waters ranged between <10 to 70 mg C L^{-1} for the blocked site and <10 to 55 mg C L^{-1} for the unblocked site. Rowson et al. (2010) studied the carbon budget for two recently dammed, drained catchments in Hexhamshire and reported average concentrations for DOC to be between 29.4 and 85.8 mg L^{-1} in stream samples. The concentrations extend beyond the mean range identified at Moor House, despite the measurements being taken within the stream, and the drains being blocked immediately prior to the commencement of the sampling regime.

7.5.2 Drivers of DOC in Peat Solution

In order to understand the causes for significant differences to have occurred between the treatments, statistical analysis was carried out to examine the strength of the relationships between the individual parameters studied, to determine which properties had most influence on DOC concentrations. Concentrations of DOC were expected to rise with increased temperatures, decreased water table levels, increased pH values and decreased concentrations of sulphate, as outlined in Section 7.1. With the exception of temperature, these parameters were expected to vary with land management, as discussed in Chapter 2.

7.5.2.1 Temperature

Statistical analysis of all samples regardless of depth and land management practice identified a significant, positive relationship between DOC and temperature. This relationship indicates that as the temperature increases, microbial activity increases, however, it was not strong (r^2=0.153, $p<0.001$), indicating that other factors are likely to play a part in controlling DOC production.

Ward et al (2007) found DOC concentrations to be significantly greater in the grazed plot compared to the unmanaged site. Significant correlations were identified by Ward et al (2007) between temperature, water level and DOC concentrations, and the authors noted that seasonal differences were evident. Seasonal trends were not identified during this study (p=0.188) at Moor House, despite average summer DOC concentrations for each treatment being higher than average winter values for all sites except the afforested site. The assessment of seasonal differences was based on seven monitoring rounds carried out during the winter months and seven during the summer months. More frequent monitoring might have identified a significant seasonal effect i.e. if the data set had included data during the months when access to the site was not possible (mid-December to late February).

7.5.2.2 Water Table

Previous research has shown that a positive relationship exists between water table and DOC (Wallage et al. 2006), however, this was not found to be the case across the sites at Moor House. Previous work has tended to focus on drained sites, where the water table has been artificially lowered. The success of drainage schemes has been found to depend on the spacing of the drains, precipitation at the site (Stewart & Lance 1991) and the orientation of the drain relative to the hillslope. The absence of a significant relationship between water table and DOC production indicates that it might only be artificial lowering of the water table that increases DOC production. No correlation, however, was identified between water table depth and DOC in the drained or the afforested sites at Moor House. Work carried out by Clark et al. (2009), however, did find a significant relationship between water table level and DOC. Their experiment was carried out on mesocosms under controlled laboratory conditions. The results showed a highly significant correlation between DOC and water table drawdown ($p<0.001$). The differences identified in the results could be attributable to a number of factors. Firstly, the mesocosm water tables might have been lowered further than on any previous occasion, resulting in an initial flush of DOC leaving the peat. Secondly, differences may have existed in the composition of the peat, with possibly more labile substances being present in the peats studied by Clark et al. (2009) than those examined in this study. Thirdly, it is possible, that the water table on the drained site was not lowered for long enough for increased DOC production to occur, thus resulting in no correlation being found between DOC concentration and water table levels at the drained site. Alternatively, the drained site might have been drained for so long (since 1955) that labile carbon would have been synthesised and flushed out of the system prior to the sampling over which this study took place.

In the case of the drained site, whilst the water table was found to be significantly lower than some of the burnt and grazed sites, levels were not significantly lower than the unmanaged site, and as noted in Chapter 4, the drained site had a higher peat moisture content than any other site. The higher moisture content suggests that the drains are not only ineffective, but also that whilst the thickness of the acrotelm might be greater than sites that have been subjected to burning or grazing, conditions are not necessarily drier and therefore more favourable to microbial decomposition. The afforested site was found to be significantly drier than all sites, with significantly deeper water table levels. It is possible that the absence of a significant relationship between DOC and water table at the afforested site is a reflection of the water level monitoring wells at the afforested site being dry on several of the visits, thus, the full variation in water table levels for the afforested site could not be accounted for.

Wallage et al. (2006) identified significant differences in water table levels between drained and undrained sites, and attributed these significant differences to increased DOC concentrations in the drained site compared to the undrained. In the case of

Moor House, the drains run perpendicular to the slope and are very narrow and heavily vegetated with shrubs such as *Calluna vulgaris*. In the case of the Wharfedale site studied by Wallage et al. (2006), the drains are much wider than those at Moor House, the banks are vegetated with grasses and sedges, and the peat is shallower. The variations between the sites could have contributed to the differences in DOC concentrations and water table levels, as the composition of the peat would have differed, and thus the ease with which the carbon could be mineralised and DOC produced. These factors could explain the correlation between DOC production and water table levels at Wharfedale and the absence of a correlation at Moor House.

Despite the absence of a relationship between water table levels and DOC concentrations, the burnt site had the significantly lower DOC concentrations than all other sites, and significantly higher water table levels than the unmanaged, the drained and the afforested sites. The absence of a statistically significant difference might be owing to the lack of data on water table variability data caused by the water table levels regularly being at the surface during monitoring (13 out of 42 measurements). The pattern in the data however could indicate that a relationship exists between the two variables.

7.5.2.3 Acidity

The burnt site was significantly more acidic than all other sites, except for the afforested site and had a significantly higher water table than all other treatments. The combination of highly acidic, anaerobic conditions is likely to have resulted in reduced microbial activity, and consequently lower DOC production. The pore waters at the grazed site had a significantly higher DOC content than the burnt site, as well as being significantly more alkaline, which indicates that the conditions were more favourable for microbial decomposition and could be responsible for the higher DOC concentrations.

In contrast, the afforested site was found to have lower water table levels and even more acidic conditions than the burnt site, yet the highest concentrations of DOC were found in the pore waters of the afforested peat. The difference in results could indicate that aerobic conditions were a stronger driver of DOC production than acidity. Highly acidic conditions tend to be associated with afforested sites, as reported by

Miller et al. (1990). Comparisons of felled peats with unfelled found less acidic conditions in felled sites, supporting the notion that afforested sites are more acidic (Neal et al. 1998, Cummins & Farrell 2002). Hughes et al. (1990) compared afforested with grazed sites and identified slightly lower pH values and lower concentrations of DOC to be present in grazed sites compared to afforested site, as found at Moor House, where all other treatments were less acidic than the afforested site. Afforested sites tend to be more acidic due to the uptake of cations by trees, resulting in an increase in the concentrations of H^+ ions remaining in the soil (Miller et al. 1990). The afforested site at Moor House was found to be significantly more acidic than all other treatments for all depths except for the burnt site (every 20 years) in the 10 cm zone, and the burnt site (every 10 years) in the 40 cm zone.

Work by Evans et al. (2005) suggested that under such acidic conditions, DOC production should be low, as the acidic conditions will limit microbial activity, thus reducing the amount of DOC released into the peat solution. Adamson et al. (2001) noted that hydrogen ion concentrations at Moor House increased as a result of water table drawdown thus implying drainage causes pH to decrease. Most studies however have reported higher concentrations of DOC in afforested peat (e.g. Grieve 1994, Harriman et al. 2003) and those that have measured the pH of afforested peat have identified conditions that are more acidic than unmanaged peats. The contradiction within the literature over the role of acidity in DOC production suggests that acidity is not the only factor controlling DOC production. Very weak negative correlations between pH and DOC were identified for each of the three depths examined, and were not found to be significant; therefore reinforcing the notion that pH is not a dominant driver of DOC loss in managed peatlands.

7.5.2.4 The Tree Stand

The higher concentrations of DOC in the afforested site might in part be explained by the damage to the tree stand at Moor House, which has occurred owing to a lack of tree thinning at the afforested plot. The density of the trees has resulted in strong competition between trees for space to spread roots and to reach sunlight. Consequently many of the trees are leaning, and have fallen in high winds due inadequate root support, as illustrated in Figure 7-27.

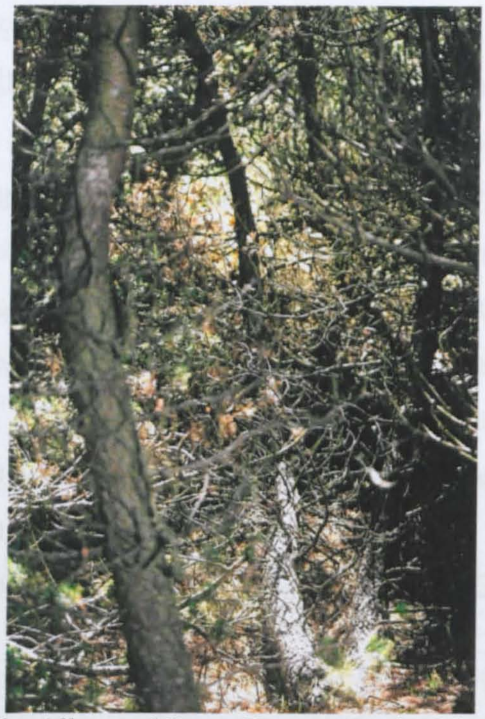


Figure 7-27 The Afforested Site – Showing Lack of thinning of Trees

Zech et al. (1994) noted that afforested sites with damaged tree stands have increased concentrations of DOC being lost into local watercourses. The rationale for this is that increased radiation can penetrate through the canopy and ground temperatures increase, thus resulting in higher rates of microbial activity and hence production of DOC. Greater rates DOC production from felled catchments were found by Neal et al.(1998) in afforested catchments across Wales, whereby higher concentrations of DOC were identified in catchments where tree felling had taken place, compared to unfelled catchments. Cummins and Farrell (2002) studied afforested peats in Ireland and also concluded that felling resulted in increased DOC losses, particularly during the summer months when losses peaked. The higher concentrations found in the felled sites still followed a strong seasonal cycle, but it was the top end of the cycle i.e. summer months, that was most pronounced, and was not mirrored in the falls in the cycle i.e. the winter months.

7.5.2.5 Sulphate

The relationship between sulphate and DOC concentrations was found to be one of the strongest, however, the results contradict those found in published literature on the effects of sulphate on DOC production. A positive correlation was identified between sulphate and DOC concentrations in many of the peat solution samples analysed from

Moor House, suggesting that increased sulphate occurred when increased DOC concentrations are present. The afforested site was the only one to follow the trend presented in published literature i.e. decreased sulphate concentrations resulted in increased DOC concentrations in the pore waters within the 10 cm zone. This trend could be attributable to the deeper water levels at the afforested site, which also had significantly greater sulphate and DOC concentrations in the pore waters. Thus, it is possible that water levels within the other treatments were not deep enough to allow the commencement of sulphate oxidation and acidification to occur. Alternatively falls in the water table might not have been for a long enough period of time to allow the process to commence. The positive relationships identified between sulphate and DOC in all treatments could either occur due to another peatland biogeochemical cycle, or, could be a reflection of background fluctuations in the peat solution.

Miller et al. (1996) reported that the introduction of trees to peats in Caithness resulted in slight increases in concentrations of sulphate being released into local catchments. Neal et al. (1998) however found average concentrations of sulphate to be fractionally higher in felled catchments than afforested sites during a survey of water quality in afforested catchments across Wales. These studies support the findings at Moor House that few differences occur in sulphate concentrations at afforested sites; the differences identified in this study were not significant.

The underlying causes for the significantly greater concentrations of DOC in the drained site compared to the unmanaged site are unclear. The drained site did not have significantly deeper water levels than the unmanaged site, and a comparison of the data on Figure 7-2 demonstrates that the maximum and minimum values were similar for both sites albeit not always at the same time. It is therefore not possible to suggest that the water levels for the drained site might have reached deeper levels than those in the unmanaged site, therefore instigating the enzyme latch mechanism. Whilst less acidic than the unmanaged site, the drained site pore waters did not have significantly different pH values. A strong relationship was identified between sulphate concentrations and DOC values, however, sulphate concentrations were not significantly different to those found in the unmanaged site. Furthermore, the sulphate concentrations cannot be associated with drought conditions in line with the proposals of Clark et al. (2006), as the water table levels were not significantly different from the unmanaged site. Daniels et al (2008) noted that drain density had a strong bearing on DOC and sulphate losses, with greater losses of sulphate from low density networks, where fluxes of DOC were found to be higher. It is therefore possible that the drainage network is not sufficiently dense to warrant a significant difference in DOC production at Moor House. Work carried out by Armstrong et al. (2010) found that whilst blocking caused water tables to rise, in some cases DOC concentrations did not fall, suggesting that higher water tables do not always result in a reduction in DOC losses, as evidenced in this study. It is plausible that the composition of the organic matter in peat at the drained site may have been responsible for the increased concentrations of DOC in the pore waters of the site. Further discussion of this possibility is provided in Chapter 8.

7.5.3 Variations in Peat Solution Chemistry with Depth

Concentrations of DOC and pH values were expected to decrease uniformly with depth. The results of the analysis indicated that in many cases, mean values in the 20 cm zone were greater than in the 10 and 40 cm zones. Significant differences in mean values were identified between the three zones examined for one of more of the properties analysed for each treatment with the exception of the afforested site. The results demonstrate the importance of sampling at specific depths rather than sampling a column of pore water that includes waters from several depths. Further work could be carried out to characterise the properties of pore waters from depths beneath 40 cm, especially in the case of the afforested site.

The lack of significant differences with depth in the afforested site for all three parameters studied could be due to the maximum water table depths being below the maximum sampling depth, thus creating a uniform acrotelm which is thicker than the other managed peats. A study of afforested peat solution chemistry carried out in mid-Wales identified decreases in DOC concentrations with minor increases in pH with depth (Hughes et al. 1990). The lack of significant differences in concentrations of DOC and sulphate and pH values within the afforested site could be attributed to the low water table in this site, mean groundwater levels were 26.4 cm but ranged between 5 and 50 cm beneath the surface. Further studies should look at different depths to compensate for this e.g. study 60 cm, 80 cm and 100 cm beneath the surface in addition to the depths examined in this study.

7.5.4 The Effects of Frequency of Peatland Burning on Peat Solution Chemistry

The frequency of burning was not expected to have a significant effect on water chemistry based on the findings of previous work carried out by Clay et al. (2009b). Yallop and Clutterbuck (2009), however, argued that differences can occur up to four years after burning. Sites burnt on a 10 year rotation at Moor House were last burnt in February 2007, and this study focussed on samples collected in 2009 i.e. two years after burning. Sites burnt on a 20 year rotation were last burnt in 1995.

No significant differences were identified between the burnt (every 10 years) and burnt (every 20 years) treatments in terms of water chemistry, with the exception of DOC concentrations at 20 cm, where the burnt every 20 years concentrations were found to be significantly higher ($p=0.01$). Clay et al. (2009b) found that increases in DOC only occurred at the start of the burning cycle, therefore it is unsurprising that differences in DOC concentrations were not identified with burning frequency as sample collection began two years after the most recent burn. The results of Clay et al. (2009b) and the present study contrast with the findings of Yallop and Clutterbuck (2009) whose work suggested significantly higher concentrations of DOC are lost from sites that have been burnt within the last four years.

7.5.5 The Effects of Combining Burning and Grazing on Peat Solution Chemistry

DOC concentrations reported at all depths for the burnt and grazed sites (both every 10 years and every 20 years) were found to be significantly greater than sites where either grazing or burning took place. In terms of pH values and sulphate concentrations, the values obtained were found to lie between those attained for sites that were solely grazed or solely burnt.

Few data have been published on the effects of combining burning and grazing. Worrall and Adamson (2007) found that grazing had no effect on the chemical composition of soil water. Increased concentrations of elements associated with a lower pH value were found on burnt sites, peat water level depth was found to be shallower on burnt sites compared to unburnt sites. These results suggest that combinations of burning and grazing make a difference to DOC concentrations and pH values by creating less acidic, more aerobic conditions that promote the release of DOC into the peat solution.

7.6 Conclusions

7.6.1 Summary of Findings

The findings of this chapter have suggested that land management has a significant effect on peat solution chemistry and the drivers of DOC losses from peatlands. Previous studies have focussed on one or two methods of management, sometimes with an unmanaged site for comparative purposes. This chapter aimed to investigate how concentrations of DOC in peat solution vary with depth and between managed sites; and to examine changes in the water chemistry of managed peatlands, with a focus on the properties that are relevant to DOC loss. The investigation was carried out by testing a series of hypotheses which are presented below along with key findings.

Hypotheses

1. Land management affects concentrations of DOC in the peat solution

A review of published literature demonstrated that land management typically causes variations in DOC concentrations compared to unmanaged sites. The results of this work demonstrated that significant differences exist in DOC concentrations between the four most commonly used management practices in the UK. Afforestation was found to result in increased losses of DOC which were far greater than those produced on burnt, grazed or drained sites. The lowest concentrations were identified in the burnt and unmanaged sites. Concentrations in the burnt (every 10 years) site were found to be significantly lower than other sites subjected to burning, and the drained and afforested sites.

2. Land management affects concentrations of sulphate in peat solution

Sulphate concentrations in the afforested site were found to be significantly higher than the unmanaged site. Significant differences in sulphate concentrations were identified between the burnt sites and all other treatments. The highest concentrations of sulphate were found in the afforested site and the lowest in the burnt (every 10 years) and unmanaged sites.

3. Land management affects the acidity of peat solution

Changes in acidity were expected between different land management practices owing to changes in water table levels thus creating differences in the thicknesses of the acrotelm and catotelm at each site. In addition, the planting of trees is known to reduce pH values (Miller et al. 1990), thus the afforested site is expected to be the most acidic. Significant effects are not expected at the burnt or grazed sites (Clay et al. 2009b). Investigation of this hypothesis found that land management makes a significant difference to the pH values of peatland pore waters. The burnt treatments, grazed and afforested sites were found to be have significantly different pH values from the unmanaged site. The afforested site was significantly more acidic than all other sites. The unmanaged site was more alkaline than all other treatments except for the grazed and burnt and grazed (every 10 years) sites.

4. Water chemistry changes with depth down the peat profile

Changes in the peat solution chemistry with depth were expected as environmental conditions change between the acrotelm and the catotelm (Clymo 1984). Concentrations of DOC in the managed sites were expected to decrease with depth as the peat becomes more recalcitrant and rates of groundwater flow decrease (Hughes et al. 1990, Ward et al. 2007). Evidence from this study demonstrated that the depth from which samples were collected made a significant difference to the results, indicating that water chemistry varies with depth. The two exceptions to this were the unmanaged and afforested sites. Highest concentrations of DOC were found in samples collected at 20 cm, where samples tended to be least acidic. It was in this part of the profile that samples were found to be significantly different to the unmanaged site, whereas fewer differences were noted between samples collected at 10 and 40 cm. Such differences possibly reflected greater groundwater movement at this depth within the peat profile, as groundwater levels tended to be below 10 cm; hence all treatments had similar water chemistry at this depth. Most sites were saturated from 40 cm and lower, hence fewer significant differences were identified at depth.

5. Burning frequency effects peat solution chemistry.

Burning frequency was not expected to have a significant effect on DOC concentrations based on previous studies (Clay et al. 2010b). The effect of burning frequency on sulphate concentrations and acidity was unclear from published literature. The results of this investigation provided evidence that the frequency with which peats are burnt did not have a significant effect on water chemistry.

6. Combinations of burning and grazing affect peat solution chemistry.

Limited changes were expected to occur between sites that are solely burnt, solely grazed and those where a combination of burning and grazing take place (Clay et al. 2010b, Ward et al. 2007). The data demonstrated that this was the case as combinations of burning and grazing resulted in greater concentrations of DOC and more alkaline solution chemistry compared to the unmanaged site or to sites that were only subjected to burning or to grazing.

7. Land management impacts on the key drivers of DOC production in peats.

The findings of this chapter have demonstrated that land management has a significant effect on peat solution chemistry and the drivers of DOC losses from peatlands. Previous studies have focussed on one or two methods of management, sometimes with an unmanaged site for comparative purposes. The results demonstrate that of the most commonly used management practices in the UK, afforestation resulted in increased losses of DOC which are far greater than those produced on burnt, grazed or drained sites.

The mechanisms driving such losses at the afforested site match the findings of Clark et al. (2006) in that lower water table levels coincided with increased sulphate concentrations and thus reduced DOC concentrations prior to water table levels rising. This mechanism was not however found to be applicable at all sites, suggesting that management alters the dynamics of DOC production and loss from the peat solution. Acidity and water table levels were found to be the dominant drivers of DOC production, however, sulphate concentrations had little effect, and where significant correlations were identified between DOC and sulphate, the relationship tended to be positive.

DOC production at the grazed site was not significantly different to those at the unmanaged site, and qualitative comparisons of the data for the two sites, found the results to be very similar. The burnt site was significantly more acidic than all other treatments, with higher water table levels, this combination of conditions is likely to have suppressed rates of microbial activity, hence DOC production was significantly lower than other treatments.

The drained site had significantly higher concentrations of DOC in the pore solution compared to the unmanaged site, however, factors such as acidity, sulphate concentrations and water table levels were not significantly different to the unmanaged site, thus, the trends remain unexplained.

7.6.2 Further Work

Further work on the dynamics of DOC production are clearly needed to fully understand recent increases in DOC concentrations in both managed and unmanaged peatlands. To investigate the linkages between falling water table levels and increased sulphate concentrations further, additional studies could focus on identifying whether the depth to which the water table is lowered has a significant effect, as well as the length of time for which the water table is lowered.

The lack of significant differences in concentrations of DOC and sulphate and pH values within the afforested site could be attributed to the low water table in this site, mean groundwater levels were 26.4 cm but ranged between 5 and 50 cm beneath the surface. Further studies should look at different depths to compensate for the low water table levels e.g. study 60 cm, 80 cm and 100 cm beneath the surface in addition to the depths examined in this study. The results might be able to identify a significant relationship between DOC and water table depth.

Further work should be carried out to look at the effects of afforestation on peat solution chemistry at deeper depths than examined in the present study, as well as with different species of tree. A study by Miller et al. (1996) found that DOC concentrations varied depending on the species of tree planted. Lodgepole pines released greater concentrations of DOC released compared to Sitka spruce sites.

The findings from the drained site found different trends to those published in studies such as Wallage et al. (2006). The findings could be attributable to the differences in the physical characteristics of the two sites and further work should be carried to identify if the following have a significant effect on DOC concentrations: drain width and depth, peat thickness, and vegetation type. In addition, further work must be carried out on burnt sites outside of Upper Teesdale to ensure that the trends identified i.e. that only the most recent of burns affects DOC concentrations are not specific to the one site that has been examined in detail.

8 SYNTHESIS

8.1 Introduction

This chapter aims to bring together the findings of the previous four chapters in order to discuss the results collectively. A brief summary of the key findings of each chapter is presented below, followed by a discussion of the implications of the findings for peatland carbon budgets, and drivers of the carbon cycle. The relevance of seasonal changes and the traditional diplotelmic model are also discussed.

8.2 Summary of the key findings of each chapter

8.2.1 Chapter 4 – the effects of land management on the chemical properties of peat

Chapter 4 aimed to identify if land management had a significant effect on the chemical properties of peats that are responsible for carbon cycling. The results provided evidence that burning results in higher concentrations of nutrients with slightly drier, fractionally less acidic peats. Afforestation was found to result in drier, more acidic peats with lower concentrations of nutrients. Little difference was found in the properties of the grazed sites compared to the unmanaged site, whilst drainage resulted in wetter, more acidic peats with slightly higher concentrations of nutrients. Concentrations of trace metals were found in very small concentrations. Whilst these metals might be of importance for the production of methane and/or oxidation of methane to carbon dioxide (Basiliko & Yavitt 2001), the concentrations were so low, that land management is unlikely to have impacted upon them.

8.2.2 Chapter 5 – impact of land management on peatland carbon stocks and quality

The results suggested that different management practices applied to peats within one nature reserve have affected carbon quality but not carbon stocks. Carbon stocks were observed to be greatest for the burnt treatments (although not significantly different to other treatments), however litter quality was poorer here than that found in other treatments (excluding the drained site). Carbon stocks were smallest in the drained site, which also had the lowest quantity of lignin and consequently poor litter quality. The afforested site exhibited the most recalcitrant organic matter, and a high C:N ratio, carbon stocks were lower than those found in sites subjected to burning but were higher than all other treatments. The lignin content of the drained and burnt sites was found to be significantly lower than the afforested peats. The lignincellulose index identified all peats sampled as being in the latter stages of decay, and therefore rates of decomposition are likely to be low. The labile fractions of the organic matter in each sample analysed were small and provided further evidence that the peat is highly decomposed.

8.2.3 Chapter 6 – effects of land management on carbon dioxide gains and losses

Land management was not found to have a significant effect on NEE or ER. The frequency with which sites were burnt had no observed effect on carbon dioxide fluxes; neither did combining burning and grazing. The absence of significant differences in water table levels between the sites compared to the unmanaged site (with the exception of the afforested site), suggest that no difference exist in the thickness of the acrotelm of the managed sites, therefore, the potential for greater or lesser rates of microbial decomposition within the treatments did not exist. Furthermore, rates of primary production between the sites were not significantly different; suggesting that in spite of the application of different management practices, the ability of the vegetation on the sites to sequester carbon was not significantly different.

8.2.4 Chapter 7 – effects of land management on peat pore solution chemistry and DOC loss

The results demonstrated that of the most commonly used management practices in the UK, afforestation resulted in increased DOC production which was far greater than those produced on burnt, grazed or drained sites. The mechanisms driving such losses at the afforested site matched the findings of Clark et al. (2006) in that lower water table levels coincided with increased sulphate concentrations and thus reduced DOC concentrations prior to water table levels rising. This mechanism was not however found to be applicable at all sites, suggesting that management alters the dynamics of DOC production and loss from the peat solution. pH and water table levels were found to be the dominant drivers of DOC production, however, sulphate concentrations had little effect, and where significant correlations were identified between DOC and sulphate, the relationship tended to be positive.

8.3 Implications for Carbon Budgets

Table 8.1 summarises the carbon gains and losses that were measured as part of this study. As discussed in Chapter 6, significant differences in ER were not identified between the individual treatments, however, significant differences were identified between one of the monitoring locations at the grazed site and selected collars on the Hard Hill plots from the burnt and grazed, burnt (every 10 years) and unmanaged plots. Significant differences in NEE were identified between the afforested site and the burnt (every 10 years) site, however, caution must be taken when interpreting this result, as the NEE measurements at the afforested site only included the understory vegetation and not the tree canopy. No significant differences in the PP of each treatment were identified; however, differences were identified between individual monitoring locations on the Hard Hill plots. Significant differences in the DOC content of the peat solution were identified across the treatments, with greatest concentrations being identified in the afforested site, and the lowest in the burnt (every 10 years) site. Predictions were of the impacts of land management on carbon losses and gains from peats were presented in Table 2.2 based on a review of published literature. Based on the notion that unmanaged sites tend to act as carbon sinks, management was predicted to result in an increase in gaseous carbon losses from the afforested and drained sites owing to increasingly aerobic conditions, whilst the expected effects of management on burnt and grazed sites gaseous carbon losses were uncertain. Based on the data presented in Chapter 6, the only site to act as a carbon sink was the burnt (every 10 years) site. As illustrated in Table 8.1, greatest losses of carbon dioxide were reported from the afforested site, and least from the burnt (every 10 years) site.

Table 8.1 Mean Carbon Gains and Losses from Each Treatment

Treatment	NEE ($g CO_2 m^{-2} hr^{-1}$)	ER (g CO2 m-2h r-1)	PP ($g CO_2 m^{-2} hr^{-1}$)	DOC (mg/L)	Carbon stock 0-10cm ($t C ha^{-1}$)
Burnt (Every 10 years)	-0.004 (49)	0.202 (35)	-0.206 (35)	42.84 (90)	65.7 (10)
Burnt (Every 20 years)	0.021 (50)	0.240 (32)	-0.219 (32)	44.21 (64)	53.65 (14)
Burnt and Grazed (Every 10 years)	0.042 (47)	0.194 (30)	-0.148 (30)	59.76 (51)	50.3 (14)
Burnt and Grazed (Every 20 years)	0.042(51)	0.169 (33)	-0.148 (33)	57.54 (54)	33.2 (10)
Grazed	0.095 (52)	0.212 (36)	-0.124 (36)	50.67 (42)	37.9 (15)
Drained	0.125 (40)	0.173 (24)	-0.084 (23)	53.88 (70)	38.4 (12)
Afforested	0.133* (47)	0.210 (27)	-0.096 (27)	72.23 (60)	49.0 (19)
Unmanaged	0.095 (48)	0.212 (32)	-0.124 (32)	44.01 (70)	47.1 (15)

*Note the measurements for the afforested site do not include the tree canopy. Numbers in brackets indicate number of samples used to calculate the mean value

8.3.1 Afforested Carbon Budgets

Work carried out on afforested peats in Scotland by Hargreaves et al. (2003) identified newly planted peats as sources of carbon, and plantations over 8 years old as sinks. Further measurements at Moor House using eddy-covariance measurements would provide a clearer indication of the carbon budget of the afforested site by providing data on carbon sequestration by both the trees and the understory vegetation. Regardless of the ability of the afforested site to sequester carbon, significantly higher concentrations of DOC were recorded from the site in response to the reduced pH, increased sulphate concentrations and lower water tables. The lower water tables caused the afforested site to have a significantly lower moisture content than all other sites, thus indicating that rates of microbial activity are likely to be higher at the afforested site, and hence the site had greater concentrations of DOC.

ER rates at the afforested site were not the highest recorded during this study, which could possibly be linked to the cooler temperatures within the afforested site, or be due to differences in the chemical composition of the afforested peats, which had the highest lignin content. Lignin was the most recalcitrant fraction analysed on the peats, and was highest in the afforested site, whereas holocellulose, the most labile fraction analysed, was lowest in the afforested peats, although the differences between the afforested and other treatments were not significant. Linkages have been found between ER in afforested sites and high quantities of water soluble carbohydrates (Domisch et al. 1998). No significant difference was identified in the quantities of water soluble carbohydrates or ER between the differently managed sites. The afforested site lost the most carbon in through ER and had the highest concentrations of DOC in the peat solution. The carbons stocks of the afforested site were not significantly different to any of the other treatments, suggesting that the afforested site is losing carbon the most rapidly.

8.3.2 Drained Carbon Budgets

The drained site had the second lowest rate of NEE owing to having the lowest primary productivity of the sites studied, despite having some of the lowest rates of ER. Losses of DOC were in the middle of the range of values obtained for all sites, with losses being significantly higher than the unmanaged site but lower than the afforested site. The drained site had the highest moisture content, despite having significantly deeper water table levels than most of the Hard Hill plots except the burnt and grazed (every 10 years) and unmanaged sites. The high moisture content and low pH of the peats could account for the reduced rates of ER, because microbial activity would be impeded by these conditions.

The drained site had the lowest carbon stocks, with a lower lignin content than most treatments except the burnt (every 10 years) site. The drained site had the highest holocellulose content of all the sites examined. Concentrations of soluble carbohydrates were highest at the drained site and concentrations of soluble phenolics the lowest, the latter could suggest that the enzyme latch mechanism (detailed in Chapter 2) had degraded phenolic compounds when the level of the water table was lowered. Significant differences in the chemical composition of the drained sites and other sites were not found. The low primary productivity rates at the drained site might be a reflection on the site having the lowest nitrogen content of all the sites studied, possibly causing limitations to plant growth.

Work carried out on peat monoliths under laboratory conditions by Freeman et al. (1993) identified higher rates of ER upon lowering the water table, which decreased once the water table levels were raised. The results indicated the lowering of the water table resulted in increasingly aerobic conditions which allowed ER to increase. In this study of Moor House, ER at the drained site was lower than most other treatments, yet overall NEE was higher (a carbon source) owing to low rates of PP. Lower rates of PP might be due to the lower nitrogen content of the peat, which was also found to be one of the most acidic of the sites studied, thus restricting plant growth. Only the burnt (every 20 years) site had a significantly higher nitrogen content, however, PP results were also highest at the burnt (every 20 years) site. The drained site had the third lowest C:N content suggesting that the carbon at the drained site could be synthesised more easily than other sites, but the differences compared to other sites were not significant. In addition, the lowering of the water table might have caused a rapid flush of carbon being lost as a result of the water table never having been so low before, thus causing mineralisation of labile compounds (Freeman et al. 1993). Furthermore, it has been suggested that carbon losses increase when peatland ecosystems are disturbed (Laiho 2006). Given the time since drainage however, it is likely that the peatland ecosystem equilibrium has been restored.

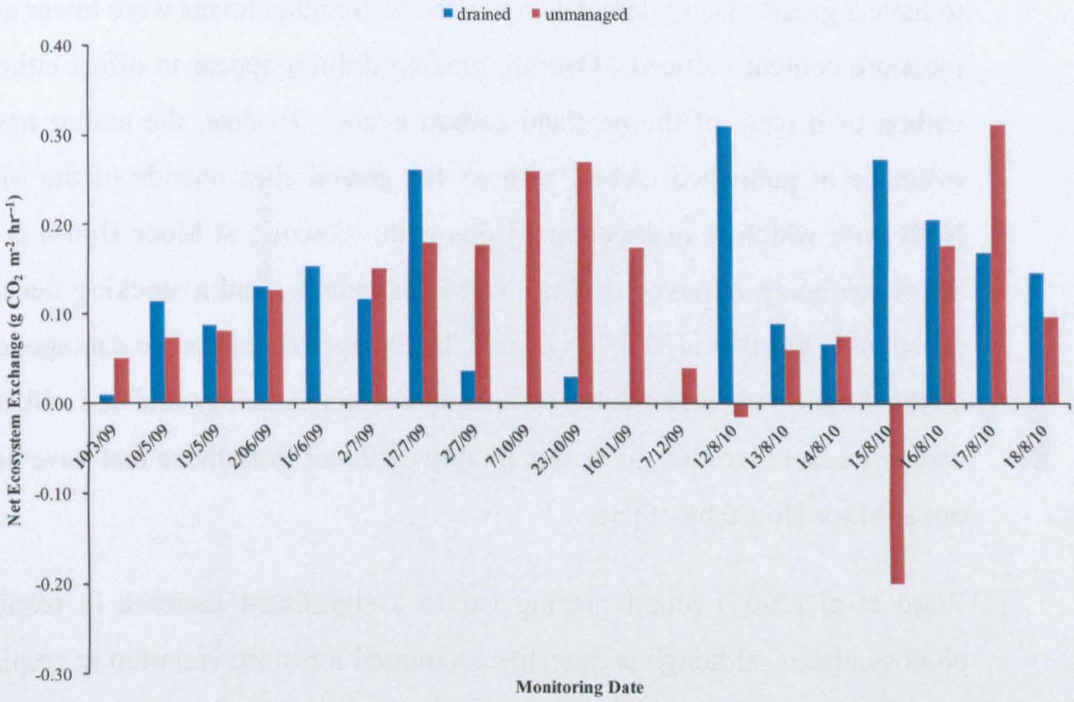


Figure 8-1 Comparison of Net Ecosystem Exchange Values for the Unmanaged and Drained Sites over Time

Overall the drained site was found to be a carbon source (Figure 8-1), in line with the findings of Rowson et al. (2010) who analysed a drained catchment that had been very recently blocked. Comparisons with an unmanaged site were not made by Rowson et al. (2010); therefore, the impact of drainage on the peatland carbon cycle for their study site could not be determined. Results from this study indicated that losses of carbon dioxide from the drained site were less than those from the unmanaged site; however, less carbon was stored at this site, suggesting less carbon was available to be mineralised, although losses of DOC were greater.

8.3.3 Grazed Carbon Budgets

Losses of carbon at the grazed site were typically very similar to the unmanaged site (as illustrated in Figure 8-2), and significant differences were not identified between the grazed and unmanaged site. Similarly, no differences existed in the carbon content or composition of the carbon in the peats between the two sites. With the exception of two trace elements (nickel and selenium), no significant differences in the nutrient content of the two sites were found either. The water table levels were found to be significantly higher at the grazed site than the unmanaged site. Field notes taken during monitoring suggested that the coverage of heather was greater on the unmanaged site, with larger bushes compared to the grazed site, which are likely to have a greater water demand, hence the water table levels were lower and the peat moisture content reduced. Overall, grazing did not appear to affect either losses of carbon or drivers of the peatland carbon cycle. To date, the author has found no evidence of published carbon budgets for grazed sites outside of the Moor House NNR with which to make comparisons with. Grazing at Moor House is light, with the sheep being removed during the winter months, and a stocking density of 0.04 sheep ha^{-1} (Ward et al. 2007). Evans (2005) reported extensive damage to peatlands in the South Pennines owing to intensive sheep grazing, and it is likely that the carbon budget there would be remarkably different from those that have been carried out at Moor House by others.

Ward et al (2007) found grazing led to a significant increase in respiration and photosynthesis, although seasonality accounted for more variation in respiration than landuse across all the sites studied, which were found to be a sink for carbon dioxide during the summer when primary production peaked. Overall, the grazed site was found to sequester more carbon dioxide than was lost (Ward et al. 2007), which was not the case in this study, or that of Clay et al (2010b). Clay et al (2010b) found NEE to fluctuated annually, with the grazed site acting as a source, as well as the unmanaged site (Figure 8-2). The discrepancies between the studies could be down to the methodology, Ward et al (2007) used a static chamber and took gas samples with a syringe, which were then analysed by gas chromatography. In this study, and that of Clay et al (2010b), measurements were taken using an EGM4, which takes measurements every four seconds over a period of 120 seconds. The former method has been noted to be more vulnerable to errors, (Norman et al. 1997) whereas the latter is considered to underestimate fluxes compared to those measured by steady state flow through chambers (Pumpanen et al. 2004). Comparisons of eddy-covariance techniques and a closed dynamic system (e.g. EGM4), have however been found to be in close agreement (Norman et al. 1997), thus suggesting that the EGM4 provides a reasonably accurate estimate of carbon dioxide fluxes. The method used by Ward et al (2007) involved flushing the syringes three times before collecting a sample, which could have reduced the concentrations of carbon dioxide near the surface (Norman et al. 1997), thus underestimating losses of carbon dioxide from the site.

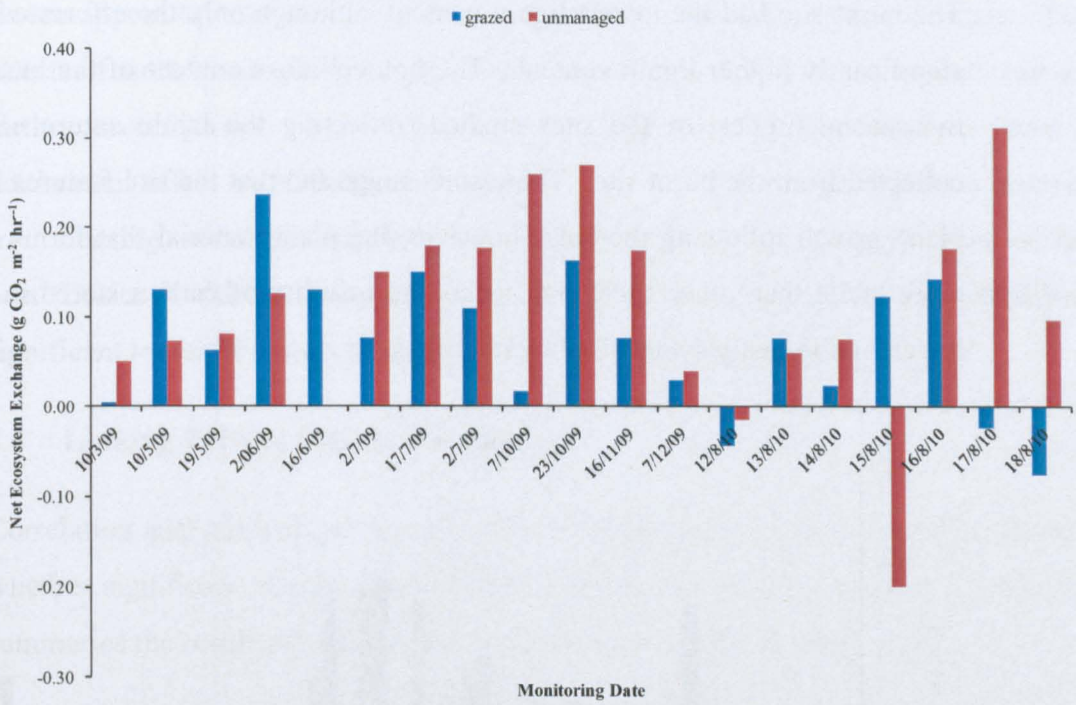


Figure 8-2 Comparison of Net Ecosystem Exchange Values for the Unmanaged and Grazed Plots over Time

In shallow peat, Ward et al (2007) found grazed peats to have significantly more DOC at 10 cm than the ungrazed peat solution, but notes the effect was not large or consistent over time. In this study, significant differences were not found in the 10 cm or 40 cm zones, but they were at 20 cm. Differences in the carbon losses calculated by Ward et al (2007) and this study could be attributable to seasonal differences; as noted by Roulet et al (2007), carbon budgets can fluctuate significantly between different years. Work carried out by Clay et al. (2010b) identified little difference in DOC concentrations between the grazed and unmanaged sites.

8.3.4 Burnt Carbon Budgets

The burnt site was the only site for which a mean carbon sink was recorded (Figure 8-3), as well as having the lowest DOC concentrations and the second highest rate of primary productivity. Concentrations of nutrients were in the middle of the range of values recorded for the site. Carbon stocks were highest in burnt peats however the differences were not found to be significant.

The burnt site had the lowest lignin content, although only the afforested site had a significantly higher lignin content. The holocellulose content of the burnt site was the second highest of the sites studied, reflecting the labile nature of the peats collected from the burnt site. The results suggested that the site features more rapid plant growth following the burn, however, the plant material that forms the peat is more labile than other sites, and hence, the quanlity of carbon stored in the site is lower, and is easily synthesised by microbes.

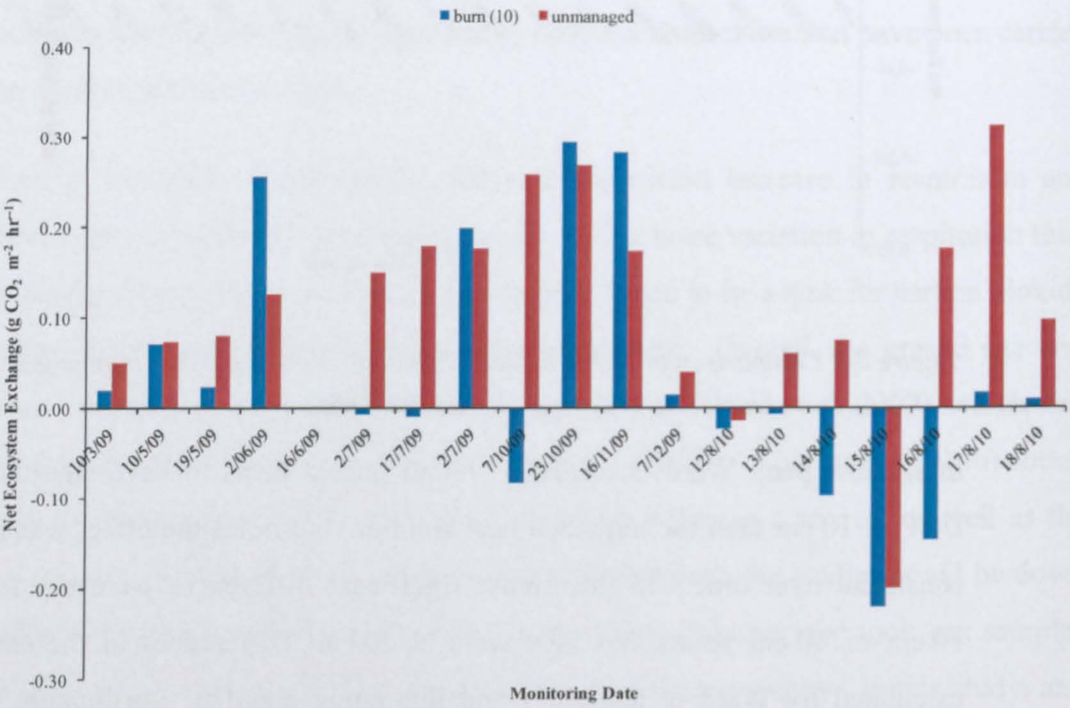


Figure 8-3 Comparison of Net Ecosystem Exchange Values for the Unmanaged and Burnt Plots over Time

Burning was also found to result in an increase in the carbon sink capacity of the managed plots by both Ward et al. (2007) and Clay et al. (2010b). In both cases, the authors attribute the differences in NEE to increased primary productivity at the burnt site. In addition, Clay et al. (2010b) suggested the lowered water tables were responsible for additional ER from the unmanaged site, however, as stated in Chapter 6, no correlation was found between water table depth and ER, possibly reflecting the number of occasions during which the water table was at the surface during monitoring on the burnt site. No significant difference was found in DOC concentrations by Ward et al. (2007), however, an increase was recorded by Clay et al. (2010b) during 2007 when burning took place. Overall, the light burns at Moor

House appear to have had a positive impact on the peatland carbon balance. The effects of more intense burning on UK peatlands are unknown however, and warrant further investigation. Severe burning on Turkish peats was found to have a detrimental effect on nutrient concentrations and the ability of the peat to recover from burning, which resulted in a significant decrease in the organic carbon of the peat (Dikici & Yilmaz 2006). Laboratory studies on burning Cambisols identified significant losses of carbon at higher temperatures (Fernandez et al. 1997).

8.4 Linking Drivers with Carbon Losses

Correlation analysis was performed on the different parameters measured to identify whether significant relationships existed between the variables studied. Table 8.2 summaries the results where significant differences were identified.

Table 8.2 Significant Correlations between Mean Values for Carbon Losses and Carbon Drivers

	Lignin	Holocellulose	Total soluble fraction	Cellulose	Temperature	DOC
DOC	0.679 p=0.005	-0.522 p=0.046	ns	ns	ns	
CO_2	ns	ns	ns	ns	0.303 p<0.001	-0.458 p=0.024
PP	ns	ns	ns	ns	-0.204 p=0.001	ns
Total C	0.566 p=0.028	ns	ns	ns	ns	ns
C stock						0.43 P=0.036
pH	ns	ns	0.57 p=0.027	ns	ns	ns
P	ns	ns	ns	0.543 p=0.03	ns	ns

ns – no significant correlation

DOC has been defined as a complex mixture of humified plant materials that are derived from plants and animals, and have been dissolved in water (Dillon & Molot 1997). The results of the correlation analysis detailed in Table 8.2 demonstrate that where higher concentrations of lignin are found, greater losses of DOC occur. This notion is supported by the findings at the afforested site which had both the highest lignin content and highest DOC concentrations, whilst the burnt (every 10 years) site had the lowest lignin content and lowest DOC concentrations. Lignin is the most recalcitrant of the fractions analysed for, and is therefore typically the most difficult for micro-organisms to degrade. The presence of higher lignin concentrations associated with higher losses of DOC, suggest that the enzyme latch mechanisms described by Freeman et al (2001b) may be responsible for the observed results. The enzyme latch mechanism enables phenolic compounds to be degraded, thus allowing hydrolase enzymes to mineralise more recalcitrant organic compounds which can then be released into the peat solution as DOC. Concentrations of soluble phenolics were very low in all of the samples analysed (0.02 – 0.04 mg g^{-1}) suggesting that phenolic compounds had possibly been degraded. No significant correlation was identified between soluble phenolic concentrations and DOC.

To support this argument further, low pH values have been found by Pind et al. (1994) to inhibit the activity of the phenol oxidase enzyme. Data collected during this study demonstrated that a positive correlation existed between pH and total soluble organic compounds from the peat. This result indicates that less acidic conditions resulted in greater concentrations of soluble compounds in the peat substrate, which in turn, could contribute to greater losses of DOC.

Additionally, a negative correlation was identified between holocellulose and DOC, indicating that the presence of less recalcitrant compounds reduces the concentration of DOC in the peat solution. It is possible, that the more labile compounds are completely degraded, and the bi-products are lost as gaseous carbon i.e. carbon dioxide or methane. Notably, increased concentrations of DOC tended to coincide with lower concentrations of carbon dioxide losses, suggesting that different fractions of the peat substrate might be responsible for losses of carbon through different phases. The results suggested that whilst sites that had a higher lignin content also had higher carbon concentrations, this carbon was also more likely to be lost in the form of DOC, as a significant correlation was identified between carbon stocks and DOC.

As noted in Chapter 6, temperature was found to be the dominant driver of carbon dioxide losses. No significant correlations were identified between carbon dioxide and the other drivers of the carbon cycle analysed.

8.4.1 Seasonality

Previous research has suggested that seasonal differences must be considered when examining fluctuations in the peatland carbon cycle (Roehm & Roulet 2003). Previously, only limited work has looked at the importance of seasonal effects on carbon losses from managed peatlands. Both Ward et al. (2007) and Bonnett et al. (2006) identified seasonal fluctuations in DOC concentrations, whilst seasonal variations in gaseous carbon losses were identified by others e.g. Alm et al. (1999b).

In order to assess whether seasonal differences existed in the data collected at Moor House, the data were split into two seasons, a summer season spanning from May to September inclusively, and a winter season spanning from October to April inclusively. The division into these seasons allowed the number of monitoring trips to be split evenly between the two seasons, as well as capturing differences in the temperatures (Figure 7-1). ANOVA was carried out on the data sets to identify if significant differences existed in the data between the two seasons. A summary of the results is presented in Table 8.3.

Table 8.3 Significant Differences in Seasonal Carbon Gains and Losses within each treatment based on ANOVA

Treatment	ER Significant seasonal difference? (p value where significant)	NEE Significant seasonal difference? (p value where significant)	PP Significant seasonal difference? (p value where significant)
Burnt and grazed (every 10 years)	Y (p=0.017)	Y (p=0.024)	Y (p<0.001)
Burnt and grazed (every 20 years)	Y (p=0.015)	N	N
Grazed	Y (p=0.01)	N	Y (p=0.001)
Burnt (every 20 years)	Y (p<0.001)	N	Y (p=0.002)
Burnt (every 10 years)	Y (p=0.008)	Y (p=0.004)	Y (p<0.001)
Unmanaged	Y (p=0.029)	N	Y (p=0.016)
Drained	Y (p<0.001)	N	Y (p=0.01)
Afforested	N	N	Y (p=0.038)

N – no Y - yes

Significantly higher rates of ER were recorded for all sites in the summer compared to the winter with the exception of the afforested site. For NEE, significant differences between summer and winter were only found for the burnt and grazed (every 10 years) site and the burnt (every 10 years) site. In the case of the burnt and grazed (every 10 years) site, the site was a source of carbon dioxide during both seasons. In the case of the burnt (every 10 years) site, during winter the site acted as a source of carbon dioxide, but was a sink during the summer. Despite significant differences existing between winter and summer values for carbon loss and storage, significant differences did not exist within the treatments for summer or winter rates of ER. In summer, significant differences existed in NEE for the burnt (every 10 years) site and the drained and afforested sites. During winter, the afforested site was found to be significantly different to the burnt and grazed (every 10 and 20 years) sites, the grazed site and the burnt (every 20 years) site ($p=0.003$). During summer, the PP of the burnt (every 20 years) site was found to be significantly higher than the drained site ($p=0.005$), whilst during winter, the grazed site had significantly higher PP than the understory of the afforested site ($p=0.015$). The results reinforce the notion that temperature is one of the key drivers of gaseous carbon dioxide losses and gains from managed peats, and that few significant differences existed between the sites as a result of management.

Table 8.4 Mean Summer and Winter Carbon Gains and Losses from Each Treatment

Treatment	ER Summer	ER Winter	NEE Summer	NEE Winter	PP Summer	PP Winter
Burnt (every 10 years)	0.258	0.117	-0.067	0.083	-0.325	-0.340
Burnt (every 20 years)	0.039	0.075	0.006	0.047	-0.328	-0.028
Burnt and grazed (every 10 years)	0.230	0.110	0.025	0.073	-0.205	-0.016
Burnt and grazed (every 20 years)	0.223	0.074	0.072	0.033	-0.151	-0.042
Grazed	0.336	0.123	0.036	0.054	-0.300	-0.069
Drained	0.242	0.008	0.147	0.003	-0.096	-0.005
Afforested	0.245	0.091	0.124	0.154	-0.1215*	0.028
Unmanaged	0.265	0.112	0.070	0.117	-0.195	0.002

* note that the PP and NEE measurements for the afforested site are based on measurements of the understory vegetation and did not include the trees. All units are expressed in g CO_2 m^{-2} hr^{-1}

The results presented in Table 8.4 provide mean values for gaseous carbon gains and losses across the managed sites investigated. The results demonstrated that during both the summer and winter months, most of the sites were carbon sources as opposed to carbon sinks. Previous studies of unmanaged peatlands have shown such sites to be carbon sinks based on gaseous carbon emissions alone (e.g. Dinsmore et al. 2010). Previous estimates of the carbon budget at Moor House have found the site to act as a carbon sink (Worrall et al. 2003b, Worrall et al. 2009), however plot scale work carried out by Clay et al. (2010b) identified the unmanaged plot as a greater carbon source than the grazed and burnt treatments. In the case of the drained site, the source appears to be due to low rates of PP compared to the other sites. During the summer months, little variation existed in the ER of the different treatments, especially the Hard Hill plots; however, variations did exist in NEE owing to differences in PP, with lower rates of PP at the unmanaged site compared to the other treatments. The results demonstrate that management results in the vegetation being regenerated, thus increasing the amount of carbon stored in the vegetation and subsequently the peat, and therefore, reducing the overall carbon source. Note should also be taken that the sites with the highest PP, also had the highest concentrations of phosphorus, and higher concentrations of nitrogen than most other sites, which possibly supplemented plant growth and thus PP. Furthermore, the burnt sites were found to have significantly higher carbon contents than the unmanaged site, suggesting that the higher PP had a positive effect on carbon stocks. Overall, despite the existence of seasonal differences within treatments, all sites were sources in both summer and winter months, although the sources tended to be greater in the winter months. Significant differences within treatments according to season were not identified, however, in both summer and winter the afforested site was found to be more acidic and to lose more DOC than all other treatments.

8.4.2 Evaluation of the Diplotelmic Model

The vast majority of peatland studies have used the diplotelmic model to describe peatland processes, particularly those concerning carbon cycling. The model is commonly accepted by peatland scientists as a foundation on which complex processes can be explained. Recent publications (e.g. Morris et al. 2011), however, have added to the criticisms of the model made by Holden and Burt (2003). In this section, alternatives to the diplotelmic model are presented, before the model's relevance of the model to the findings of this study of managed peatlands at Moor House is considered.

The presence of a single boundary between the aerobic and anaerobic zone was defined by Ingram (1978) as the boundary between the acrotelm and catotelm, and occurs at the depth the water table reaches during drought years. Morris et al. (2011), however, noted that no definition of drought is provided, nor is consideration given to the time over which the drought should be defined. Furthermore, the definition of what constitutes a drought varies between different disciplines (Smith 2011). On this basis, the application of the diplotelmic model has been inconsistent. The application of terms such as oxic/anoxic as suggested by Ise et al. (2008) provides greater clarity when explaining peatland ecohydrological processes, and allows fluctuations in the system to be accounted for. Furthermore, such terms allow a gradient to be considered between the saturated and unsaturated layers, rather than insisting on the presence of an abrupt change in conditions which is rarely witnessed in the field. Furthermore, beneath the water table, conditions may not be wholly saturated, as the presence of bubbles of gas (in particular methane) may be present (Rosenberry et al. 2006).

One advance on the diplotelmic model would be to use a polytelmic model involving many layers including a mesotelm layer between the acrotelm and catotelm as suggested by Clymo and Bryant (2008) and possible additional layers e.g. reflecting a low permeability close to the surface, and layers representing different stages of peat decomposition. Such a model has been criticised by authors such as Morris et al. (2011) for failing to provide sufficient flexibility. Whilst the model provides a more accurate reflection of a peatland, and is simple to use and understand, it still involves the use of rigid horizontal boundaries that cannot reflect spatial variation and areas of a peatland where disturbances have occurred e.g. erosion due to anthropogenic activities.

Morris et al. (2011) proposed a "hot and cold spot" model as an alternative to the diplotelmic model. This model would encompass the variability of peatlands whilst still being applicable to multiple sites. Terms such as acrotelm and catotelm would be replaced with oxic and anoxic, high and low decay zones which would provide gradients rather than fixed boundaries. The model would encompass the differences that exist in boundaries between the layers depending on the property being examined i.e. it would allow for the saturated and unsaturated zones being located at different levels within the vertical profile compared to the oxic and anoxic zones. Areas where high rates of activity and/or cycling would be highlighted as hot spots, a term regularly used by geochemists in other fields e.g. brownfield regeneration. Areas with particularly low rates of activity and/or cycling would be termed cold spots. McClain et al. (2003) used the terminology to identify zones where many processes converge and create high rates of activity, when such events occur over a limited period these are referred to as hot moments. The resultant model would demonstrate vertical and horizontal variation in ecohydrological and geochemical processes and cycling in peatlands.

The results from this study indicate that the ever popular diplotelmic model is over-simplified, and does not provide sufficient detail to encompass the variations found in the properties of the peats between the different sites. The polytelmic model would also not be applicable, as such a model would merely add layers of complexity to account for variations in the vertical profile, but would still be broadly split into aerobic and anaerobic zones.

The afforested site at Moor House was found to have a higher soil moisture content at the edges than in the centre of the site, thus demonstrating that horizontal variation exists as well as vertical variation. In addition, ridges within the forest were found to be drier than furrows, thus adding additional complexity to the trends observed at the site, and any models that might subsequently made for the site.

Elsewhere, significant differences between water table levels and carbon losses within individual sites were found. For example, on the burnt (every 10 years) site, collar 13 was found to have significantly higher water table levels than other locations at Hard Hill plots, however, collars 14 and 15 on the same site did not. The variation in water table levels demonstrates that dividing the site into two zones (aerobic and anaerobic) does not reflect the conditions at the site, and thus cannot be used to explain the results found. Furthermore, whilst the drained site had significantly deeper water table levels than the grazed site and some of the burnt plots, the drained site had the highest peat moisture content of all the sites. These findings indicate that despite lower water table levels, the area above the water table might not be as aerobic as one would expect, and consequently the diplotelmic model could not be used to explain trends in the data, as evidenced in Chapter 7.

The results in this thesis showed significant differences in soil nutrient and carbon concentrations (Chapters 4 and 5) and DOC concentrations (Chapter 7) with depth. The pH was found to decrease with depth, however nutrient concentrations tended to fluctuate with values decreasing from the surface peats downwards before rising again in the 40 to 50 cm zones. In the case of DOC concentrations, values tended to be significantly higher in the 20 cm zone than the 10 and 40 cm zones. Research by Sundh et al. (1997) suggested that the boundary between aerobic and anaerobic peats tends to be an optimum zone for microbial activity. The presence of such a zone favours the polytelmic model; however the fluctuations in nutrient concentrations would be better represented by a hotspot model.

Literature published on alternative models since the commencement of this study (e.g. Morris et al. 2011) does suggest that future research should be designed with alternative and more complex models in mind than the diplotelmic model. Whilst the diplotelmic can be useful in explaining initial findings, the heterogeneity that exists within the peatland environment is sufficient that a more detailed model is required. The variations and trends observed at Moor House warrant the use of a more complex model such as the hotspot model proposed by Morris et al. (2011) to describe and explain the results found, however, it is plausible to suggest that such a model might not be applicable to sites elsewhere, thus resulting in a site specific model only.

8.5 Conclusions

All sites were found to lose more carbon dioxide than was gained, burning had most beneficial impact on the carbon balance, while afforestation was the most detrimental to the carbon balance. The results indicate that nutrient status is not an important driver of carbon losses from managed peats (based on a lack of a significant correlation between nutrient concentrations and carbon losses), however, sites with low primary productivity tended to have low nitrogen concentrations, whilst higher primary productivity was observed at sites with greater concentrations of nitrogen. Substrate quality did not vary significantly between the sites studied, with the exception of lignin which was significantly higher in the afforested site compared to the burnt site. A strong correlation was identified between DOC concentrations and substrate quality. Sites which had higher concentrations of DOC tended to have a higher lignin content e.g. the afforested site, whereas sites with high quantities of more labile carbon tended to have lower concentrations of DOC in the peat solution.

Seasonal differences were found to exist in gains and losses of carbon within each treatment, with higher rates of ER in the winter months and lower NEE in the summer months owing to higher PP. Temperature was found to be the key driver of ER, with no significant correlations existing between ER and the other parameters measured. Differences noted in the properties of the peats responsible for carbon cycling were observed with depth and across selected sites, demonstrating that the traditional diplotelmic model is over-simplified. The hotspot model proposed by Morris et al (2011) clearly has advantages, however, there is a risk of the model becoming overly complex, and potentially too site specific in order for models to be derived that can be applied to multiple peatland sites. Further investigation using a field experiment designed to test the practicality of the model would be useful.

9 CONCLUSIONS

9.1 Overview

Peatlands occupy a mere 3 % of the world's land mass but store up to one-third of terrestrial carbon stocks (Gorham 1991). Formed in cold, upland areas with high rates of precipitation, they act as carbon sinks, storing more carbon than is released through microbial synthesis of the organic matter inputs that form the peat. As climate change scenarios develop, the future of peatlands looks uncertain, and with this in mind, efforts are being made to identify the extent of the carbon sink or source in peatlands, and to better understand peatland carbon cycling. At present, little is known about how management affects carbon stocks, and whether one strategy might be favoured over another in the future, from a carbon stock preservation perspective. As the need to safeguard carbon stocks rises up the political agenda, questions are being asked about how peatlands should be managed to limit carbon losses (UK National Ecosystem Assessment 2011).

British peatlands have historically been managed in many different ways to provide an income for rural communities. Such practices involve heather burning on grouse shooting estates, sheep grazing, drainage to increase the area of land available for agriculture and afforestation. Carbon budget calculations for unmanaged peatlands have demonstrated that peatlands are carbon sinks (e.g. Worrall et al. 2009).

Carbon budget calculations carried out by Worrall et al. (2003 and 2009) indicated that carbon dioxide and dissolved organic carbon (DOC) account for the greatest losses of carbon from peatland systems. If climate change predictions (increases in temperatures and declines in rainfall) are realised, peatlands are expected to become sources of carbon as rising temperatures and falling water tables will result in increased rates of carbon mineralisation and subsequent losses of carbon. By investigating the influence of land management on these key carbon loss pathways, more accurate predictions of the effects of climate change on UK peatlands can be made.

Carbon cycling in peat is governed by four drivers (Laiho 2006): environmental conditions (e.g. temperature, water table level), substrate quality (e.g. how recalcitrant the peat is), nutrients (e.g. nitrogen required to synthesise the carbon stocks) and microbial community (e.g. whether the microbes present are able to utilise the available substrate). Changes in one or more of these drivers will influence the carbon budget of a peatland. How land management influences these drivers has previously been unclear.

The research presented in this thesis aimed to examine the effects of land management on the environmental conditions, nutrient status and substrate quality of managed peatlands. The work involved studying small managed plots at the Moor House NNR in Cumbria, where all four of the main methods of peatland management in the UK were present – burning, grazing, drainage and afforestation. The investigation comprised a combination of fieldwork and laboratory work, and the detailed results were presented in Chapters 4 to 7, with a discussion of the implications of the findings presented in Chapter 8. This final chapter aims to summarise the key findings of the research with respect to the aims and objective set out in Chapter 1, and to identify limitations in the data and to provide suggestions of the directions in which future research could go.

9.2 Findings in Relation to Aims

Aim: - to understand how the key drivers of the carbon cycle vary between differently managed peatlands, focussing primarily on the four main methods of management of blanket peatlands in the UK; but also considering burning frequency and combinations of burning and grazing. Differences in the chemical and physical properties of managed peatlands will be used to aid understanding of losses of carbon dioxide and dissolved organic carbon (DOC) from peatlands.

Overall, the results of this research have indicated that environmental factors have the strongest bearing on carbon losses from peatlands, although substrate quality is not irrelevant. The nutrient concentrations present in peatlands do not appear to be significant, although evidence of potential linkages between PP and nitrogen were identified. In terms of carbon losses from managed peatlands, this research has indicated that the only one of the main four treatments studied was a carbon sink – the burnt (every 10 years) site. Interestingly, the site that was burnt on a 20 year rotation was not found to be a carbon sink, despite having slightly higher rates of PP.

Overall, the differences in carbon losses between the burnt sites were minimal, and few significant differences were identified between the treatments. In contrast, losses of carbon were much greater in the drained and afforested site, with greatest losses of both carbon dioxide and DOC being identified in the afforested site. Caution should be urged when examining the data from the afforested site however, as NEE measurements could not take into account the tree canopy, and thus reflect the understorey vegetation only.

Few significant differences in the chemical composition of the peat were observed between the sites in Chapter 5. Lignin was identified as the dominant constituent of the peat for all the sites, with highest concentrations present in the afforested site, and least in the burnt (every 10 years) site. The high lignin content of the peats from all the sites indicated that the peats are in the latter stages of decomposition, and are thus fairly recalcitrant. The higher lignin content in the afforested site, coupled with the highest losses of DOC, some of the highest CO_2 losses through ER, however, suggest that the chemical composition of the peat is not as strong a driver as originally thought. Carbon stocks were lowest in the drained site and highest in the burnt sites.

The assessment of NEE in Chapter 6 indicated that all the sites acted as carbon sources, with the exception of the burnt (every 10 years) site which was a very small carbon sink. Temperature was found to be the dominant driver of ER, accounting for between 54 and 92 % of variation in the data ($p=0.001$). The afforested site was the only treatment where a significant relationship between temperature and ER was not identified. Average PP were highest in the burnt and grazed sites indicating that regeneration of the vegetation through management is of key importance in terms of sequestering carbon. The lowest averagePP was identified at the drained site, where concentrations of nitrogen were also lowest. In terms of the structure of the peat, the air filled porosity of the burnt and grazed (every 20 years) site was greatest, however no linkages were established between the structure of the peat and gaseous carbon losses.

Losses of DOC were greatest from the afforested site, and it was at this site that the findings of Clark et al. (2006) were supported, in that lower water table levels were found to coincide with higher concentrations of sulphate and more acidic conditions, causing a reduction in DOC losses, which increased once the water table rose. On other sites, this relationship was not identified, and other drivers were found to control losses of DOC e.g. pH and water table levels.

9.3 Findings in Relation to Objectives

Objective 1. *To establish how concentrations of nutrients required in the carbon cycle for synthesis of carbon stocks compare between differently managed peatlands within the upper 50 cm of the peat profile.*

The results of the investigative work into the influence of land management on peatland nutrient concentrations demonstrated that land management does not cause significant differences in nutrient concentrations compared to unmanaged sites. The moisture content of afforested peats increased compared to the unmanaged site, yet unexpectedly, the moisture content of the drained site was significantly higher than the unmanaged site, suggesting that the drains were ineffective. These results imply that the tree demand for water is more important than the depth of the drains in terms of moisture content. The drains at Moor House are spaced between 10 and 15 m apart, and previous work e.g. Hudson and Roberts (1982) has suggested that a maximum spacing of 2 m is required to have a significant effect on peat moisture content. The drained and afforested sites were found to have significantly more acidic conditions than the unmanaged site.

Previous work by Allen (1964) suggested that burning causes nutrient concentrations in peat to increase due to inputs of ash. These findings were supported by work carried out on heathland soils by Forgeard and Frenot (1996), however, in both cases the results were based on laboratory studies not field studies. The only field study to published to date on the effects of burning on peatland nutrient concentrations was that of Dikici and Yilmaz (2006) who looked at severe burns on a Turkish peatland that had also been drained. Their results indicated that nutrient concentrations increased significantly compared to unburnt areas of the same peatland. The research carried out at Moor House did not identify significant increases in nutrient concentrations due to burning. It is plausible to suggest that nutrient concentrations may have increased immediately after the burn, but when the peat cores were collected two years post-burn, no significant differences were identified, and consequently the implications for the carbon cycle are not significant.

Previous research on the effects of grazing on peats has suggested that the effects of nutrient concentrations in peats are unclear. This most recent research has indicated that for the lightly grazed peat at Moor House, no significant differences in nutrient concentrations, moisture content or pH were evident. The findings demonstrate that under light grazing, inputs of nutrients from sheep urine and faeces are not sufficient to make a significant difference to the nutrient content of the peat, or possibly were leached out of the peat.

Studies of the nutrient concentrations in drained upland peats have not previously been studied, although the results of a study at a lowland site in Somerset suggested that draining peats results in a decrease in nutrients (Heathwaite 1990). In the case of this study, the drained site was found to be the most acidic of all the treatments considered, and was significantly more acidic than the unmanaged site. The drained site also had a significantly higher moisture content than the unmanaged site. The nitrogen content of the drained site was the lowest of all the sites, although concentrations were not significantly lower than the unmanaged. Furthermore, the potassium content of the drained site was higher than at all the other sites, but was not significantly different from the unmanaged site. The lack of significant differences could be attributable to the drain spacing.

The impacts of afforestation on peatland nutrient concentrations have not been previously published, although suggestions as to the potential impacts have been made e.g. Jandl et al. (2007). Given the additional demand for nutrients by trees, concentrations of nutrients were expected to be lower, the pH reduced (Miller et al. 1996) and moisture content reduced (Pyatt 1993). The results of this research confirmed the hypothesis in that reduced moisture content and pH were observed but there were no significant differences in the nutrient content of the peats were observed.

The effects of management on the nutrient status alone are not thought to be significant enough to impact on the carbon cycle, suggesting that future efforts to preserve peatland carbon stocks do not need to focus on manipulating the nutrient balance in order to encourage carbon sequestration. Although no statistically significant correlations were identified, the nitrogen content of the site burnt on a 20 year rotation was found to be higher than any other site, and this site also had highest rates of PP, suggesting that nitrogen concentrations could be limited, and thus from a carbon storage perspective, additions of nitrogen would be beneficial in order to promote carbon sequestration.

The implications for higher PP are not as straightforward as might first appear. The burnt (every 20 years) site had the lowest C:N ratio, suggesting that carbon was more rapidly broken down than at other sites, although the C:N ratio was only significantly different to the burnt and grazed (every 10 years) and afforested sites. PP was not significantly different from other sites. The carbon content however, was the highest of all sites, and significantly more than the drained, afforested, grazed and unmanaged sites, suggesting that the higher nitrogen content was valuable in terms of capturing additional carbon by promoting rates of PP.

The burnt (every 20 years) site was found to be a slight carbon source based NEE data alone with some of the lowest concentrations of DOC loss. The high C:N ratio however means it is possible the site would become a bigger source, particularly during periods when temperatures are high, given the correlation between temperature and ER.

Objective 2. *To investigate what differences exist in the carbon stocks of differently managed peatlands and to identify how carbon quality varies as a result of peatland management, with a focus on establishing which peats are the most recalcitrant.*

Analysis of carbon stocks in managed peats at Moor House identified the burnt and grazed (every 10 years) site as having the most carbon and the drained site containing the least. Statistical analysis revealed no significant differences in the carbon stocks of the managed peats within the upper 10 cm of the profile. In terms of carbon quality, lignin formed the dominant fraction, within which statistically significant differences were identified. Lignin concentrations were highest in the afforested site, implying that afforested peats were the most recalcitrant. Lignin concentrations in the drained and burnt (every 10 years) sites were found to be significantly lower than the afforested site.

Despite having the most recalcitrant carbon, the afforested site also had the lower carbon stocks than burnt sites, potentially due to low rates of PP. Measurements of PP in the afforested site were limited as only the understorey could be measured as opposed to the tree canopy and understory vegetation. Measurements based on understorey vegetation suggested no significant differences existed compared to the unmanaged site, neither did the carbon stock or lignin content. Despite the high lignin content and recalcitrant nature of the afforested peats, losses of both carbon dioxide and DOC were greatest from the afforested site. Only losses of DOC were significantly greater than other sites. The opposite trend was identified in the burnt (every 10 years) site, where lignin concentrations were lowest but carbon stocks were among the highest and losses of DOC the lowest. Rates of ER were not significantly different to other sites. NEE for the burnt (every 10 years) site was negative, indicating that the site is a sink – the only one of the sites examined, owing to the high rates of PP at the site. The results indicate that rather than highly recalcitrant carbon composition favouring carbon stores and reducing losses, substrates with greater concentrations of more labile compounds e.g. holocellulose and SFOW promote carbon storage, possibly because the microbes have begun to synthesise these compounds and before they have become bound into the organic matter.

Overall the results demonstrated that management has a significant effect on carbon quality and the C:N ratio but not on carbon stocks. Although significant differences in the chemical composition were only found for the lignin content between treatments, linkages were however identified between the chemical composition and DOC production, demonstrating that substrate plays an important role in peatland carbon cycling.

Objective 3. *To identify the effect of land management on the physical drivers of the peatland carbon cycle.*

The impact of land management on the physical structure of peatlands has not been previously investigated. The results of this study have demonstrated that management has a significant effect on the density and porosity of peats. The afforested site was found to have significantly denser peats whilst the drained site had the least dense peats. In terms of the carbon cycle, no significant linkages were identified between the physical structure of the peat and either DOC or NEE losses of carbon.

Objective 4. *To establish how peatland management affects carbon dioxide losses and environmental controls on carbon dioxide losses.*

Previously, measurements of ER on managed peats have been published for one or two types of land management within one study, but a comparison of multiple management practices with an unmanaged site has not been published at a single site. The results demonstrated that significant differences in ER and PP do not exist between differently managed sites or in relation to the unmanaged site. Much debate exists in published literature as to the causes for fluctuations in ER, with some authors suggesting that water table levels are of key importance e.g. Nilsson and Bohln (1993), and others finding no relationship between water table and ER (e.g. Updegraff et al. 2001). Temperature was clearly the dominant driver of ER, accounting for between 54 and 92 % of variation ($p=0.001$), with the exception of the afforested site.

All sites were found to be carbon sources with the exception of the burnt (every 10 years) site which acted as a very slight carbon sink. Seasonal differences were identified in all the sites, with greater ER rates in the summer than winter, supporting the notion that temperature is the dominant control on ER. Sites that were burnt on a 10 year rotation were found to have significantly greater NEE values in winter than summer.

Previous studies of ER from managed sites have found grazed and burnt sites to act as both sinks (Ward et al. 2007) and sources (Clay et al. 2010b). Additional inputs of nutrients were expected to increase rates of PP and thus alter the carbon balance from these sites. Given the absence of significant differences in terms of nutrient content and substrate quality between the grazed and burnt sites with the unmanaged site, it is unsurprising that no significant differences were identified in terms of ER. Furthermore, whilst significant differences were not identified in water table levels between the sites, water table levels were not identified as a significant driver of the ER.

The afforested and drained sites were expected to have greater losses of ER owing to the presence of reduced water table levels, however, as water table levels were not identified as a significant driver of the ER, significantly higher ER rates were not observed in these sites compared to the unmanaged site.

Objective 5. *To determine how concentrations of DOC in peat solution varies with depth and between managed sites.*

Previously the effects of land management on DOC concentrations have focussed on one or two management practices, and as yet, comparisons between a range of treatments have not been carried out. This study provided data on concentrations of DOC in peat solution wells from four differently managed sites and comparisons were made to an unmanaged site. The data collected enabled a comparison to be made of the effects of land management without confounding factors such as dilution and collection of data from different catchments.

The drained and afforested treatments were expected to lose the most DOC owing to the reduced water table levels, the actions of the phenol oxidase enzyme and/or sulphur oxidation to sulphate, which has been reported as a cause for increased DOC losses (Clark et al. 2006). The afforested site was found to lose significantly more DOC than all other treatments. The results were attributed to the deeper water table levels and as a consequence, oxidation of sulphates, as described under the findings for objective 6. The drained site also had significantly higher losses of DOC compared to the unmanaged site; however water table levels and acidity were not significantly different to those at the unmanaged site. The burnt (every 10 years) site lost the least DOC, which was attributed to the more acidic conditions compared to all other sites, and significantly higher water table levels resulting in anaerobic conditions. The results from the grazed site indicated that no significant difference existed in DOC concentrations between the grazed and unmanaged site, in line with the findings of Clay et al. (2010b).

In terms of the peatland carbon budget, light burning, light grazing and no management at all appeared to be the most beneficial in terms of reducing the amount of carbon lost. The burnt (every 10 years) site was significantly more acidic than most other treatments and had higher water table levels than other treatments, indicating that it represents the best option for reducing microbial synthesis of organic carbon.

Objective 6. *To examine changes in the water chemistry of managed peatlands, with a focus on the properties that are relevant to DOC loss.*

Significant increase in DOC concentrations have been observed from peatland catchments over the last 40 to 50 years across northern Europe (Evans et al. 2005). The causes for such increases has been the subject of much debate, with increases in temperature (Fenner et al. 2007), changes in water table levels (Pastor et al. 2003), the enzyme latch mechanism (Freeman et al. 2001b), oxidation of sulphur to sulphate and subsequent acidification (Clark et al. 2006) all discussed by peatland scientists. This study provided a unique opportunity to assess the potential impacts of land management on DOC losses from peats, and to determine what might be the underlying causes for such variations.

The mechanisms driving DOC losses at the afforested site match the findings of Clark et al. (2006) in that lower water table levels coincided with increased sulphate concentrations and thus reduced DOC concentrations prior to water table levels rising. This mechanism was not however found to be applicable at all sites, suggesting that management alters the dynamics of DOC production and loss from the peat solution. pH and water table levels were found to be the dominant drivers of DOC production, however, sulphate concentrations had little effect, and where significant correlations were identified between DOC and sulphate, the relationship tended to be positive. Significant differences in the pH of the peat solution were identified in most sites (except the burnt and grazed (every 20 years) and drained treatments). Significant differences in sulphate concentrations were not identified however.

9.4 Data Limitations and Future Research Recommendations

This research has provided an insight into the effects of land management on the carbon cycle. The results have provided a unique comparison of the four most common peatland management practices in the UK with an unmanaged site. The data are not without limitations however, and a summary of these is provided below.

The main limitation in the data was the measurement of NEE in the afforested site, where only understorey vegetation could be measured, and thus the measurements are not representative of afforestation as a whole, and do not allow judgements to be made about the overall carbon balance of afforested peats.

The management practices at Moor House are fairly light, the burn was not intense (records suggest that the hummocks were still frozen post-burning), grazing is at 0.04 sheep ha^{-1} (Ward et al. 2007) and the drains are spaced at 10 to 15 metres apart.

The installation of monitoring wells to 0.5 m beneath the surface of the peat was not sufficient for samples to be recovered during every monitoring visit. Deeper wells would allow better monitoring of peat solution chemistry and water table levels, particularly in the afforested site.

The potential options for future research are numerous, although budgetary and time constraints are more than likely to limit the extent to which further work could be achieved. The following options would provide additional information that would build on the foundation of knowledge that has been achieved during this study:

- Investigate carbon budgets using eddy-covariance techniques for each treatment, which could provide year round, continuous data that could be supplemented with peat solution sampling
- Study managed sites outside of the Moor House NNR to identify whether the data are replicable or whether Moor House is a unique case
- Study differences in management intensities to determine the effects on both carbon losses and drivers of the carbon cycle. For example, to compare closely spaced drains to widely spaced drains, light burning compared to heavy burning, light grazing compared to heavy grazing, and to compare the effects of different species of trees. The results of such work could be compared to the results of this study which has examined sites that have been subjected to light management practices.
- Carry out further research into the chemical composition of peats using analysis such as pyrolysis to look at the macro-molecular structure of the peat to identify

if differences occur in the chemical composition of the managed peats at a finer scale. This could enable a better understanding of why some managed peats lost or stored more carbon than others.

- Examine the effects of time since management practices were implemented e.g. time since burning, using a carefully managed set of sites that allow differences between sites that have been burnt for several years compared to sites that have been recently burnt. The work carried out in this study has provided a insight into the effects of land management on carbon losses and drivers of the carbon cycle over a short period of time, rather than looking at transitional changes, which could provide valuable information on whether management over time causes fewer or greater differences between sites.
- Further research could also aim to measure all components of the peatland carbon budget, and where possible, to measure fluvial fluxes, although doing so would require measurements at sites where the whole catchment is subjected to one method of management, and in doing so, would make comparisons to an unmanaged site difficult. Furthermore, issues of scaling up the results are likely to arise, unless a very intensive monitoring regime is executed, which typically is not feasible.

9.5 Concluding Remarks

The results presented in this thesis demonstrate that land management affects both the drivers of the peatland carbon cycle and carbon gains and losses. Both managed and unmanaged sites were found to be sources of carbon, with afforestation resulting in greatest losses of carbon both within the peat, through ER and DOC losses.

Future management needs to focus on encouraging increased PP in particular by increasing water table levels to encourage *Sphagnum* growth and thereby peat bog growth. Light burning was also found to increase water table levels and peat solution acidity, thus reducing DOC concentrations into the peat solution. The results demonstrated that temperature is the most important control on ER, and under climate change losses are likely to increase, therefore, the need to conserve carbon through increased PP is unquestionable. DOC was found to be strongly linked to water table levels, pH and the carbon quality, with higher concentrations of holocellulose resulting in reduced DOC concentrations.

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