The Sustainable Carbon Management of Moorlands: spatial distribution and accumulation of carbon on Dartmoor, southwest England

Submitted by Lauren Elizabeth Parry to the University of Exeter

as a thesis for the degree of

Doctor of Philosophy

In May 2011

I certify that all material in this thesis which is not my own work has been identified and that no material has previously been submitted and approved for the award of a degree by this or any other University.

Signature: …………………………………………………………..

Acknowledgements

Throughout the course of my PhD I have had the good fortune to receive support from many kind and generous people and I would like to give each of them my wholehearted and sincere thanks. I would like say a big thank you to my supervisor Prof. Dan Charman, who has provided me with consistent, kind, supportive and wise advice throughout the course of my PhD. I have been very fortunate to have been supervised by someone that I have come to respect so much. Dr Ian Bailey has given me consistently great supervision and has had his door open whenever I needed advice and I am very grateful this. I am also thankful for the support from Dr Will Blake and Prof. Geoff Millward and other members of the CoRIF laboratory in the radionuculide dating facility.

During the past three years I have spent much time in the field and the laboratory, much of this work would not have been possible without the help of the technical staff at the School of Geography at Plymouth University. Also, my many days spent on Dartmoor gathering data have benefited greatly from numerous volunteers, particular thanks go to Jonny Noades, Stef Honeywill and Brett Metcalfe who have given up substantial periods of their free time and shown considerable enthusiasm, even in the pouring rain and howling wind!

During the course of my PhD I transferred from Plymouth University to the University of Exeter. This transition would not have been so smooth without the thoughtfulness of several members of the University of Exeter’s School of Geography particularly Huw Vasey and Thomas Roland. From the University of Plymouth Heather Davies, Jessie Woodbridge, Sally Murrall-Smith and Louise Callard have been great friends. I am forever grateful to my parents Sarah and Richard Parry for being incredibly supportive, particularly during the penultimate three months when they allowed their grown up daughter to move back home!

I would like to thank Great Western Research, The Duchy of Cornwall, Dartmoor National Park Authority, the National Trust and Natural England for the funding provided for my PhD. Additionally thanks go to the Seale-Hayne Educational Trust who generously provided additional funding for radionuculide dating. This funding network has provided me with many stakeholders, who have each provided many ideas, advice, knowledge and perspectives for me to consider in my research. This has helped me to understand the issues faced by moorland managers in the real world and I would like to thank them all for this valuable experience.

Abstract

Peatlands are unique habitats that have absorbed large amounts of carbon dioxide and locked it away as carbon buried in peat for millennia. In the UK, blanket peatlands form one of the largest terrestrial stores of carbon (Milne and Brown, 1997). Recent research suggests that the carbon sequestering potential and carbon stores of UK blanket peatlands are at risk from changes in land use practices and climate. Although, to date, little research has considered blanket peatland at a landscape scale and a comprehensive understanding of land use and degradation impact upon carbon sequestration has not been gained.

This thesis presents a study of Dartmoor, a blanket peatland in south west England vulnerable to climate change (Clark *et al*, 2010). A landscape scale carbon inventory, using a methodology designed for blanket peatlands is presented. Nearly 1000 peat depths and 30 cores were taken using stratified sampling across Dartmoor's landscape. Functional relationships between peat depth, bulk density and carbon content and topographic parameters were found. In *arc* GIS 9.3 these were used to model landscape scale carbon, this estimates that Dartmoor contained 9.7 (-2.91 + 2.97) Mt of carbon, a value similar to that of the national inventory (Bradley *et al,* 2005).

The thesis then considers the impact of drainage and degradation on carbon accumulation. Fifteen cores were dated from a drained, degraded site with a history of burning and control site using Spheroidal Carbonaceous Particles (SCPs) and radionuclide techniques. Previous studies have raised concern surrounding accuracy dating recent peats. Results indicate that although dating was largely successful, some discrepancies existed related to poor calibration of SCPs and mobility of radionuclides. To avoid error in dating, it was concluded that multiple dates should be used per core. With consideration of this, carbon accumulation was found to be active but significantly lower in the degraded site and unchanged in the drained site. Further analysis suggested that this outcome may vary with changing management and topographic situations. Future carbon accumulation at a landscape scale was calculated under different scenarios. This found degradation could potentially reduce carbon sequestration on Dartmoor by up to 32%. Economic valuation of accumulation values was used to demonstrate how this data could be used to inform management.

This thesis provides an insight into the carbon storage and threats to Dartmoor, an under investigated, yet threatened blanket peatland environment. This helps broaden the spatial understanding of the response and value carbon stored in UK blanket peatlands.

Contents

List of Tables

List of figures

1 Introduction and Rationale

Terrestrial stores of carbon are of considerable importance in the global carbon cycle, both for storing large quantities of carbon and for actively exchanging it (Houghton, 2003). Soils globally form the largest terrestrial store of carbon, and alterations in their storage and exchange with the atmosphere could significantly influence the global climate (Jobbágy and Jackson, 2000). Despite covering only 3% of the earth's land area, peatlands contain roughly a third of the planet's soil carbon (Gorham, 1991) and, consequently, have a substantial role to play in this exchange (Moore *et al*, 1998). The importance of this role is evident as the spread of peatlands globally since the early Holocene has been linked with alterations in the climate (Blunier *et al*, 1995, Yu, 2011).

The United Kingdom (UK) contains approximately 15% of the world's blanket peatland resource (Tallis, 1995) and constitutes nearly 50% UK's terrestrial store of carbon (Milne and Brown, 1997). For this reason, blanket peatlands are a valuable environment to the UK. Blanket peatlands require wet and temperate environmental conditions to form and accumulate (Charman, 2002). In the UK these conditions are mainly found within upland areas (Bradley *et al*, 2005; Milne and Brown, 1997).

The narrow environmental limits in which blanket peatlands accumulate and sustain themselves make them highly sensitive to change. British blanket peatlands are located close to human population centres, and have been pressured as a result of rising population and industrialisation. Pressures faced by blanket peatlands take a number of forms which can be divided into extrinsic pressures, including climate change and atmospheric pollution, and intrinsic pressures, resulting from increasing land use. These factors have caused the UK's blanket peatlands to become among the world's most threatened peatland environments (Holden *et al*, 2007a). Accordingly, there is now concern that the conditions required to retain and sequester carbon within blanket peat in the UK may have been altered in ways that are leading to large-scale releases of stored carbon into the atmosphere.

The ability of peatlands to influence the climate has drawn much research attention, particularly over the last fifteen years. This has led to advances in the understanding of the links between peatland functioning, carbon cycling and climate change. At the same time, policy makers, land managers and conservation organisations have become increasingly active in attempts to preserve the UK's degrading blanket peatlands. If conservation activity such as this is to be successful in mitigating changes occurring on blanket peatlands, closer connections between research and the needs of conservationists must be made. This thesis intends to provide a body of work which will build upon current scientific understanding, develop and assess current methodologies and help build up information which can be used to inform carbon friendly management on the following themes:

- *The Landscape Scale Resource*

  The UK benefits from one of the most detailed national soil carbon inventories in the world (Bradley *et al*, 2005; Milne and Brown, 1997). However, the national carbon inventory does not have a high spatial resolution and isn't specifically tailored to peatlands. As a result, this inventory is of limited use in informing land use and management at a landscape scale. Although a few small scale (<2500ha) carbon inventories have been carried out by Frogbrook *et al* (2009) and Garnett *et al* (2001) among others, no standardised and easily replicable methodology exists which can accurately quantify and map carbon distribution at a landscape scale (>10,000ha).

  This study develops a methodology for establishing a high resolution carbon inventory at a landscape scale. It is specifically designed for blanket peatlands and uses a methodology which has been developed to allow easy replication. As a result, it has the potential to provide a representative and more accurate account of a large area of UK blanket peatland, which can be replicated elsewhere. This can be used for the verification of national inventories, to demonstrate the importance of UK moorlands as a carbon store, and to contribute towards informing more effective and targeted management of upland peatland areas.

- *Management and carbon accumulation*

  The UK's blanket peatlands are semi-natural environments, where anthropogenic activities have contributed to the spread and accumulation of peat over millennia. Since the industrial revolution, however, there has been a considerable shift in the role human activity has had to play, and practices such as burning, grazing and drainage have increased significantly as a result of population pressure and mechanisation. It is thought that these increases have significantly altered the dynamics of many UK blanket peatlands and have negatively impacted upon their ability to retain and accumulate carbon.

  This balance must be redressed if the valuable carbon stores are to be retained. In order to do this effectively an understanding of the impacts management and degradation can have upon peatland carbon stocks and of appropriate long-term management techniques must be gained. Despite recent advances through studies such as Ward *et al* (2007) and Rowson *et al* (2010), the complex relationship between management systems, change in peatland condition and carbon cycling remains only partially understood. An aim of this thesis is to contribute to this understanding in order to help land managers on Dartmoor and managers of other blanket peatland areas nationally to manage valuable carbon resources more effectively. Additionally, the thesis will provide an insight into the challenges and advantages of dating recent peat, one of the key methodologies used for understanding long-term carbon accumulation processes in peatlands.

- *Scenario Planning at the landscape scale*

  As noted earlier, two primary datasets are presented in this thesis, the first considers carbon distribution at a landscape scale, and the second reviews the impact of different land-management practices and peat condition at a local scale. The final aim is to explore ways in which the data generated can be used in conjunction to produce information that can be applied practically by land managers and policy makers. In doing this an initial understanding of the magnitude of the impacts of future management choices and past degradation.

Research Location

Blanket peatlands are found throughout the UK and although they possess many common characteristics, each has subtly different environmental, land-management and socio-economic conditions. These may in turn cause variability in the response of peatland carbon dynamics to particular land-management practices and may require more tailored actions by local land managers.

This study focuses on the blanket peatland of Dartmoor National Park in the South West of England. Dartmoor has potential for advancing understanding of the blanket peatland carbon response outside of the usual areas. Firstly, most research into the UK’s threatened blanket peatland resource is carried out in the Pennines, especially on Moorhouse National Nature Reserve (NNR) in the North Pennines and in the Peak District. Although the focus on these areas is understandable due to the need for long-term controlled experiments and the rich dataset that exists as a result for Moorhouse NNR, and the fact that some of the UK’s most heavily damaged blanket peatlands are in the Peak District (Bonn *et al*, 2009a), there is a need to broaden our understanding outside of these areas. In particular, more work is needed on blanket peatlands that are subject to less severe damage in different climatic settings. Despite being an important wildlife habitat and carbon resource, Dartmoor’s blanket peatland has not been investigated in any detail. Studies on Dartmoor and other contrasting areas of UK peatlands are required to understand the broader carbon dynamics of blanket peatlands throughout the UK.

Second, Dartmoor is on the southern most limit of blanket peat distribution in Western Europe (Lindsay, 1995) and has been identified as a highly vulnerable blanket peatland to the impact of climate change (Clark *et al*, 2010). As a result, research and monitoring on Dartmoor may provide an early warning of the response of blanket peatlands to climate change. A greater understanding of Dartmoor may allow managers of UK moorlands to take pre-emptive action and will help inform Dartmoor's managers of how to respond to the threat of climate change. This study provides an assessment of the carbon resource of Dartmoor blanket peat, identifies impacts of management practices on the resource and assesses the extent to which management could mitigate the impacts of future climate change on carbon storage.

As a result of the lack of previous research carried out on Dartmoor, the emphasis on landscape scale analysis, and the need for establishing long-term changes in carbon accumulation changes, this thesis takes an intentionally broad approach both spatially and temporally. It aims to provide an initial insight into the values and threats faced by blanket peatland carbon in a threatened location.

Aims:

The main aims of the thesis are based firmly on the three thematic areas identified above and are broken down into a series of stages reflecting the necessary steps in developing a landscape scale understanding of carbon dynamics for a blanket peat region.

The Peatland Carbon Resource:

1. To develop a methodology for calculating and mapping the distribution of carbon stored within peat soils at a landscape scale (>10,000ha).

2. To estimate the amount of carbon stored within Dartmoor's peat soils, its distribution and relationship with key landscape characteristics.

Peatland carbon accumulation and management

3. To assess the strength of methodologies for dating recent peat used for calculating carbon accumulation rates.

4. To establish the relationship between key management practices and carbon conservation and sequestration.

Scenario Planning and synthesis of data

5. To explore ways in which the data generated in this thesis can be used to assess the magnitude and impact of future management decisions.

Thesis structure

The thesis is presented in eight chapters which are broadly structured around the aims and objectives outlined above. A literature review is presented in Chapter Two to provide context for the thesis. This outlines what peat is, how it accumulates and discusses the processes of, and conditions necessary for blanket peatland development. It then goes on to consider why blanket peatlands are valuable carbon stores and how they are threatened. Finally, it provides an introduction to Dartmoor, considering its climate and soils, land management and previous research.

The main results section is written up in four core chapters; these will form the basis of manuscripts for journals and are edited in this format. Chapters Three and Four form the elements of a landscape scale carbon inventory: Chapter Three presents the results of modelling peat depth and Chapter Four that of bulk density and carbon content. These elements are then brought together in Chapter Four to produce a map and inventory of carbon. Chapters Three and Four cover aims one and two, found within the ‘Peatland Carbon Resource’ theme and are edited as two linked manuscripts. The next stage of the thesis considers the ‘Peatland carbon accumulation and management’ theme. Aim three is covered in Chapter Five where radionuclide and palaeo-environmental methods are analysed for their ability to accurately date peat. In Chapter Six, with consideration of the results of aim three, the differences in carbon accumulation as a result of management are presented and covers aim four.

Chapter Seven provides a synthesis of the data presented in this thesis using scenarios (aim five). This chapter illustrates how landscape scale and accumulation data can be used to identify future changes in peatland carbon under differing management scenarios. Finally, the overall conclusions are summarised and directions for future research are identified in chapter eight.

2 Literature Review

2.1 Peat and peatlands

2.1.1 What is peat?

Many soil types occur in moorland environments, and several different peat soils are included within the National Soil Research Institutes (NSRI) soil classification system. Peat is primarily comprised of undecomposed or partially decomposed organic matter accumulating in areas with positive water balance, where excess of water can restrict decay (Rydin and Jeglum, 2006). Peat contains little or no mineral matter compared with other soil types. The water content of peat is high and can vary greatly. Hobbs (1986) suggests this value is between 200 – 2000% of dry weight over small distances, whilst Lindsay (2010) suggests water content is generally 98% (wet weight). Due to the high proportion of water, peat has very low bulk densities (Hobbs, 1986). There is some discrepancy in the literature regarding how thick peat needs to be to become classified as a peat deposit. Avery (1980) suggests a value of 30cm, if overlying bedrock and 40cm if overlying other soil. There is no formal classification system for peat in England and Wales and depths used vary by country, which creates difficulty for policy makers and researchers in identifying the limits to peatland environments.

2.1.2 Peatlands

Many of the concepts in this thesis consider the peatland at a ‘landscape scale’. As a result it is necessary to discuss the processes which cause peatlands to form and the controls upon their formation. Many types of peatland exist, but all conform to the same criteria determining their development. The maintenance of a positive water balance and low decay rate are essential, as peat will only begin to form when productivity exceeds decay and the environment must allow this (Charman, 2002).

- Generally temperatures will be low enough to prevent rapid decay but high enough to maintain productivity.
- The climate will be sufficiently wet to maintain anaerobic conditions.
- Geology will be impermeable, preventing the loss of water and maintaining a positive water balance.

Once peat formation has been initiated it will grow vertically, spread laterally and gradually change form over time (Charman, 2002). Once established peat begins to control its own hydrological and vegetative processes, these autogenic controls cause peat to spread laterally (Foster *et al*, 1988). For example, peatlands influence moisture and humidity in the surrounding area, inducing peat forming conditions and, consequently, causing lateral spread (Charman, 2002). However, the rate and extent of spread of a peatland are a function of allogenic environmental factors and time. Changes in the surrounding topography and climate will influence when and where peat spreads. For example, basal carbon dates were found to be younger on freely drained areas underlying blanket peat by Charman (1992), thus demonstrating the allogenic control from topography.

2.1.3 Landform Units

Peatlands are not homogenous units; they are made up of a number of peat landforms which are influenced and defined by the hydrological functioning around them. These individual landform units were classified by Ivanov (1981) and can be applied to peatlands throughout the world. Lindsay (1995) describes how Ivanov's classification takes a hierarchical form, with landform units occurring within one another: microform > microtope > mesotopes > macrotope. These classifications represent the 'building blocks' of a peatland and increase in scale from microform to macrotope.

- Microforms - These are the individual structural elements found on a peatland, hummocks and hollows are the most commonly referred to but other types exist (Lindsay, 2010)

- Microtope - An area where plant cover and other physical characteristics are uniform. Microtopes can be made from a complex of microforms, such as a hummock hollow complex (Charman, 2002).
- Mesotope - Individual mire units which expand from a distinct centre, these are delineated by their flow patterns (Ivanov, 1981).
- Macrotope – The fusion of the individual mesotopes to form the macrotope (Ivanov, 1981). The boundaries of a macrotope define the edges of the peatland (Lindsay, 2010).

Ivanov’s landform units come from the Russian scientific literature and were only introduced to the UK in the 1980s. There is currently little focus upon Ivanov’s landform units in the UK and few peatland maps have been made which consider these classifications. Lindsay *et al* (1988) is the one exception, where Ivanov’s landform units were identified by aerial photography. The development of a UK wide resource identifying Ivanov’s landform units would be extremely beneficial for research being applied at a landscape scale in the UK, as they help describe peatland functioning and development at different scales.

2.1.4 Blanket Peat

Blanket peatlands can be characterised by their ability to cover sloping landscapes, as they form in hyperoceanic environments where the excess of water available to them means that their development is not restricted by topography (Charman, 2002). Blanket peatlands begin to form when individual mesotopes, which have been forming as individual peatlands, begin to spread laterally up a slope by a process of paludification. Eventually these mesotopes fuse together to form a blanket peatland macrotope. This process results in blanket peatlands being highly heterogeneous in nature (Charman, 1992). Typically blanket peatlands are ombrotrophic environments, this with their need for hyperoceanic conditions, means that their development is closely controlled by climate (Evans and Warburton, 2007).

The fusion of individual mesotopes and the evolution of blanket peatlands over time have been shown by Charman (1995) to cause a complex hydrological system to evolve. Consequently mesotopes are important indicators of a blanket peatlands functioning. The primary mesotopes on blanket peats are watershed mires, saddle mires and spur mires which can be defined by flow patterns (Lindsay, 1995). At present these are rarely considered in the UK literature.

The UK's blanket peatlands form a substantial portion of the world's blanket peatland resource and consequently are an important environment for the UK (Tallis, 1995). Blanket peats began to develop throughout the UK during the Holocene from approximately 8000 radiocarbon years ago, although the date of initiation is spatially variable (Smith and Cloutman, 1988; Tallis, 1991). Three main causes of blanket peat initiation are hypothesised: changes in the climate to cooler, wetter conditions (Conway, 1954); soil maturation though podsolization which leads to peat formation (Charman, 1992) and anthropogenic activity, including forest clearance (Simmons, 1964; Moore, 1975) and burning (Smith and Cloutman, 1988). Moore (1975) suggests that a combination of these may lead to peat formation, although it is often difficult to separate the causes of blanket peat initiation in individual peatlands (such as observed by Charman, 1992). However, it is thought that the nearer to the climatic limits of peat growth, the more likely that blanket peat formation is related to human activity (Moore, 1984).

There is little recent literature considering the processes of development of blanket peatlands and the blanket peat landform (with exception of erosion and degradation work such as Evans and Lindsay, 2010a). Additionally, the work carried out has a regional bias towards the Pennines (Tallis, 1995). Charman (1995) highlights that landscape scale research is essential for hydrological management and understanding and this assertion can reasonably be extended to understanding carbon dynamics. Peatland development and landscape research would greatly benefit from the further development and an expansion of the spatial area where these studies are carried out.

2.1.5 The Vertical Structure of Peat

Physical changes occur throughout the peat profile and these changes can be used to understand the processes of peatland accumulation and functioning. Ingram (1978) classified the physical changes using the water table level and aeration throughout a peat profile; this system is now the standard classification. The upper level of peat, in the zone of water table fluctuation, where living and partially decayed vegetative matter is present, is known as the acrotelm. The lower level of peat is the catotelm, which consists of more decayed plant matter. This layer is constantly waterlogged and is considerably more anoxic than the acrotelm. It is not always easy to determine the transition between the acrotelm and the catotelm due to the variable nature of the water table (Clymo, 1984). However, Ingram (1978) suggests the mean lowest level of the water table can be taken as the acrotelm to catotelm boundary and this is a widely accepted definition.

In a recent review Lindsay (2010) highlighted an additional structural composition to the traditional catotelm / acrotelm system. Lindsay describes the 'haplotelmic mire' which consists of only one layer; a haplotelmic mire occurs when there has been destruction or modification of the acrotelm. In disturbed peats the upper layer becomes either as dense or denser than the underlying materials. This upper layer is called the 'haplotelm' and Linsday (2010) proposes that its differing structural and chemical characteristics from the acrotelm may cause different processing of carbon in the upper layers of peat. However this classification is in its early stages and it is not yet clear how, or if, the haplotelm varies from the acrotelm. As a result acrotelm processes will be the main focus within this thesis.

2.1.6 The process of peat accumulation

The processes of peat accumulation are closely related to carbon accumulation and cycling. Clymo (1984) describes the processes causing peat accumulation from its start point. Peats accumulation begins when vegetation begins to grow in aerobic conditions, these plants begin to die and additional vegetation is added above this layer. Some mass is lost from the dead vegetation as a result of decay, but the remaining proportion loses its structural integrity under the growing weight and collapses causing the bulk density to increase. The increased bulk density decreases hydraulic conductivity, which in turn causes the water table level to rise because of reduced lateral and vertical water flow. The water table rarely rises into the live vegetative matter as hydraulic conductivities are higher in the surface layers and water runs off more rapidly laterally. Live vegetative matter is continually added to the surface causing the further collapse of dead vegetation below. The water table rises with the vertical growth of the surface because of the low hydraulic conductivity of the collapsing vegetation below it. This results in the dead layer of water logged vegetative matter below decaying very slowly and gradually accumulating in the catotelm. Decay will continue in the catotelm, although not as rapidly as in the acrotelm. Peat will continue to accumulate in the catotelm, whilst the acrotelm remains largely the same depth, until the quantity of peat in the catotelm causes the slower anaerobic decay to equal the rate of matter being transferred from the catotelm to the acrotelm. After this point peat growth is limited, although it can take several thousand years to occur and has not yet been reached for any peatlands where there are suitable data available. Clymo (1984) points out that the older a peat body becomes the slower the rate of carbon accumulation as the ratio of decay in the catotelm to the proportion entering the catotelm is gradually increasing.

Clymo (1992) describes structural and functional layers within which the processes described above occur. This demonstrates how peat accumulation in established peats can be altered by variability in the water table (Figure 2.1). Functional layers within the peat profile largely determine the type of decay occurring and are controlled by the level of the water table; whilst structural layers determine the amount of decay occurring. If the functional layers alter for an extended period of time the position of the structural layers will in turn alter, which will impact upon peat accumulation (Clymo, 1992). For example a lowering of the water table will cause a deeper aerobic layer (acrotelm) and consequently a lower mass will reach the transitional zone and be transferred into the catotelm. However, Belyea and Clymo (2001) have suggested a feedback mechanism, where increased productivity causes a rise in the height of the peatland surface, which in turn deepens the acrotelm, the deeper acrotelm results in more decay and reduces accumulation to a similar level to that in less productive areas regulating the amount entering the catotelm. Thus, a deepening acrotelm may not always mean increased decay rates.

2.1.7 Function and processes of decay in the acrotelm

Near surface peat accumulation is investigated within this thesis, and understanding the function and processes occurring in the acrotelm are essential to enable these results to be put into context. The acrotelm has little direct impact upon the long term accumulation of peat, as much of the mass within it is subject to further decay. Clymo (1984) reviewed the proportion of productivity lost in the acrotelm finding that figures ranged between 80 – 90% loss of mass. Despite this, the fact that peat accumulates demonstrates that some mass must avoid decay in the acrotelm and enter the catotelm, where decay rates are nearly 1000 times slower (Belyea and Clymo, 2001). The catotelm is the real accumulator of peat (Clymo, 1984) and the acrotelm should be considered as the source that feeds peat accumulation (Clymo *et al*, 1998).

The dynamics influencing the balance between the productivity and decay within the acrotelm are essential for determining the mass of peat entering the catotelm. Belyea (1996) considered the patterns of decay throughout the profile of the acrotelm, finding that decay rate decreased with depth, largely due to a decreasing redox potential, although it was found that decay increased again in the zone of water table fluctuation. Belyea (1996) suggests that litter type and microhabitat are important determinants for the rate of decay. Vascular species are less resistant to decay than the genus *Sphagnum* (Clymo, 1984), which is commonly regarded as a key building block for peat. Johnson and Damman (1991) found that decay rates are variable between *Sphagnum* species. For example, the hollow species *S.cuspidatum* decayed 1.5 times as fast as the hummock species *S.fuscum*. However, it is important to note that both *Sphagnum* and vascular species can be transferred into the catotelm (Clymo,1984).

2.1.8 The value of peatlands

Peatlands are valuable environments due the unusual properties and processes of formation and functioning as discussed above. In the UK, blanket peatlands are becoming increasingly valued by scientists, policy makers and the general population for this. The benefits they provide to society in terms of ecosystem services are highly varied. For instance, blanket peatlands can be used to reveal valuable information on past climates, vegetation histories and patterns of human activity as their ombrotrophic nature and low decay rates preserve records over millennia (Charman, 2002). Their location in the uplands and receipt of considerable quantities of water mean that blanket peatlands are the source for much of the UK's drinking water supply (Bonn *et al*, 2009b). Peatlands also have a unique combination of flora and fauna contributing to the UK's biodiversity. Finally, the low decay rates and consequent gradual build up of organic matter mean that blanket peatlands are the UK's largest terrestrial store of carbon (Milne and Brown, 1997). This thesis is most concerned with carbon storage and the climate regulation provided by peatlands and as a result this will be discussed in greater detail in the following section.

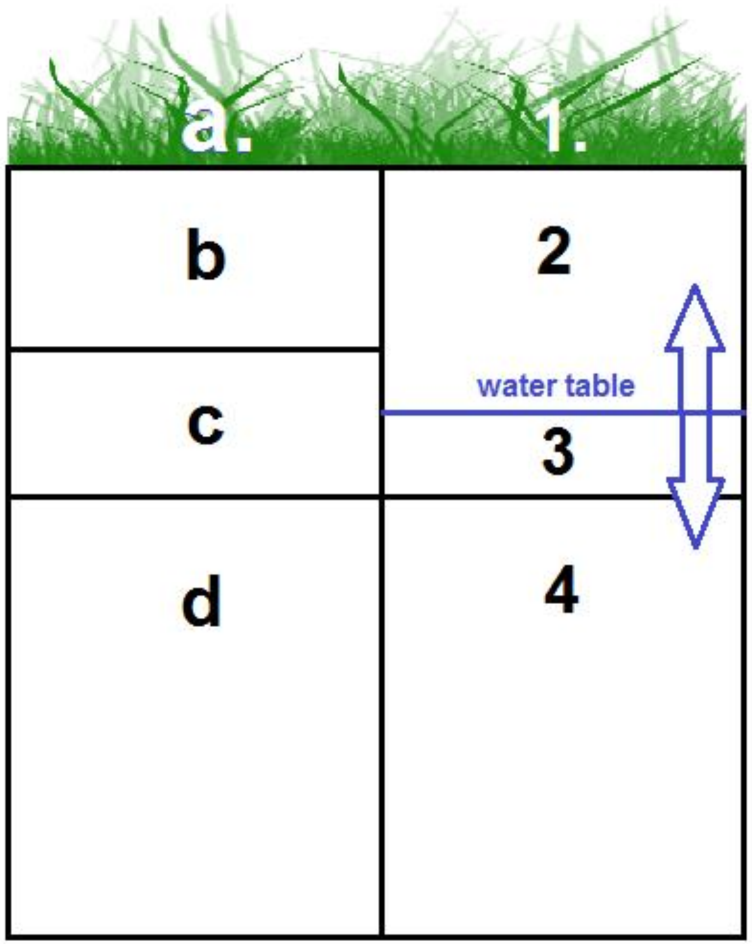

Figure 2.1Peat structural and functional layers (Clymo, 1992)

Structural Layers:

a) *The Euphotic Layer:* the living productive layer, which adds organic matter to the environment (acrotelm*)*.
b) *Second layer:* organic matter becomes buried by the growth of plants and aerobic decay begins to act upon this. Peat bulk density remains low and hydraulic conductivity high (acrotelm*)*.
c) *Collapse Layer*: due to the organic matter becoming weak from decay, the peat collapses from the weight of the upper layers. Bulk density rapidly rises and hydraulic conductivity is much reduced (acrotelm – catotelm *transition*).
d) *Lower Peat Layer*: conditions are anoxic with slow anaerobic decay occurring. Hydraulic conductivity is low and bulk density is high, enabling these conditions to be retained (catotelm).

Functional layers:

1. *The Euphotic Zone*: the same as the structural layer (acrotelm*)*
2. *O_2 Rich Layer*: area within the peat located above the water table, where decay occurs rapidly due to aerobic conditions *(*acrotelm*)*.
3. *Transitional Zone*: area slightly below the water table, where conditions change from aerobic to anaerobic (acrotelm – catotelm *transition*).
4. *Anoxic Zone*: area below the water table, where anaerobic conditions prevail and decay is slow (catotelm).

2.2 Carbon in Peat

2.2.1 Carbon accumulation

One approach for understanding the carbon balance of a peatland is to calculate the rate of carbon sequestration using a combination of carbon, bulk density and dating analysis. However, as peat (and consequently carbon) travels through a peat profile it can go through a number of different processes (see section 2.1.6). Therefore, the carbon at a given point within the profile may be in a different phase of its journey. This has important implications for the interpretation of carbon sequestration data. A number of definitions for these phases are discussed in Clymo (1992) and Clymo *et al* (1998) and in using these better understanding of carbon sequestration rates can be gained.

The accumulation rate of carbon in the acrotelm is called the 'recent rate of carbon accumulation' (RERCA). Although decay rates in the acrotelm are rapid, decay has had little time to take effect, and as a result RERCA sequestration rates are amongst the highest within the peat profile. Carbon accumulation calculated as RERCA does not reflect the actual carbon accumulation because the carbon has not yet transferred into the catotelm and is subject to potential future decay (Belyea and Clymo, 2001). RERCA represents the function the acrotelm provides (see section 2.1.7). RERCA is calculated by taking a date from the acrotelm and dividing by the dry mass and carbon content above it. Examples of calculation of RERCA can be seen in Billett *et al* (2010) which summarises RERCA for a number of sites throughout the UK, with values ranging between 35.1 and 209.1 g C m^{-2} yr^{-1} for various time periods over the last 150 years.

The 'long term rate of carbon accumulation' (LORCA) represents the average sequestration rate throughout the complete peat profile, accounting for decay in both the acrotelm and catotelm. It is calculated by the total mass and carbon content of a profile against its basal age. LORCA values from peat profiles of differing ages may be at differing stages of their development and therefore ratios of decay to input may be different (Clymo, 1992). As a result LORCA will decrease the older a peat profile becomes due to the increasing depth of the catotelm (and resulting increased decay). Consequently a limiting factor when comparing LORCA for sequestration potential is that peat profiles should be of similar ages (Clymo *et al*, 1998). Examples of the use of LORCA are most commonly found in Finnish studies where LORCA has been applied for a variety of reasons such as: to provide a comparison of the sequestering potential of different mire types (Turunen *et al*, 2002; Tolonen and Turunen, 1996) and to understand the role of events, such as long term fire dynamics (Pitkanen *et al,* 1999).

Clymo (1984) explains that the true accumulator of peat is the catotelm, therefore the balance between inputs into the catotelm and the slow long term decay in the catotelm gives the ARCA 'actual rate of carbon accumulation' (Clymo *et al*, 1998). The proportion of carbon entering the catotelm from the acrotelm is also an important value and Clymo *et al* (1998) defines this as p* (the rate of addition of dry mass on an area basis).

Clymo *et al* (1998) states that if constant rates of p* and decay are assumed, age vs depth profiles in peatlands produce a concave curve as a result of long-term decay. This also applies to the carbon sequestration rate. Using the above carbon accumulation terms (RERCA, LORCA and ARCA) assumes a distinct boundary between each functional layer, however the non-linear relationship described by Clymo *et al* (1998) demonstrates that this is not the case. Consideration of this is needed when interpreting values calculated for LORCA / ARCA / RERCA.

2.2.2 Carbon Cycling

Carbon cycling within peatlands is a complex process with many routes and mechanisms whereby carbon is transferred, transformed and removed from a peatland. Although this thesis does not deal specifically with individually quantifying or identifying these processes, understanding their role within the peatland system is needed in order to put its findings into context.

Within a peatland the balance between inputs and outputs of carbon during a given time needs to be considered. Inputs and outputs of carbon occur via fluvial, gaseous and aeolian routes in peatlands (Dawson and Smith, 2007). Carbon can be gained, transferred and removed within and between each of these routes. In this section these processes will be explained by discussing each form of carbon and its production and transfer (Table 2.1 provides a simplified review of each transfer route).

Carbon form	**Route**	**Input / output**	**Process**
CO_2	Gaseous	Input	CO_2 uptake via photosynthesis
	Gaseous	Output	Mineralisation of carbon stored within peat
	Gaseous	Output	Oxidation of CH_4
	Gaseous	Output	Respiration
	Fluvial	Output	Dissolved
CH_4	Gaseous	Output	Methanogenesis, plant transfer
	Fluvial	Output	Dissolved
POC	Fluvial	Output	Erosion
	Aolian	Output	Erosion
DOC	Fluvial	Output	Decay and root exudation

Table 2.1 Overview of primary routes and processes of carbon cycling in a peatland environment

Gaseous Exchange

Net Ecosystem Exchange (CO_2)

Peatland carbon is fixed from the atmosphere by uptake of CO_2 via photosynthesis. However, in ombrotrophic peatlands plant growth is restricted by low nutrient availability. As a result, uptake of CO_2 is low in comparison to other high carbon environments, such as forests (Ruimy *et al*, 1995). Low decay rates cause peatland carbon accumulation, not high CO_2 uptake. However, photosynthesis is the principal source of carbon for blanket peatlands and maintaining optimal conditions for plant growth is still important for the long term build up of carbon content.

CO_2 is also exported from peatlands; root respiration and mineralisation of organic matter are the principal processes by which this occurs (Moore *et al*, 1998). Most CO_2 production occurs within the acrotelm and a number of studies have investigated this process. A commonly held assumption is that CO_2 production will increase with a deeper water table, due to the increased availability of O_2 throughout the profile. This hypothesis is supported by Moore and Dalva (1997) who found a linear relationship between water table depth and CO_2 production. However some studies do not reflect this pattern. For example, Aerts and Ludwig (1997) found high CO_2 emissions in zones of fluctuating water table. Increased temperature is also a cause of CO_2 output, due to both increased respiration and mineralisation (Blodau and Moore, 2003). Additionally vegetation composition plays a role, as more readily decomposed species have been related to elevated CO_2 output, due to more rapid decay (Johnson and Damman, 1991). Oxidation of CH_4, produced in the catotelm which passes through the acrotelm, is another source of CO_2; Frenzel and Karofeld (2000) found that most CH_4 produced in a peat profile was oxidised to CO_2 in the zone of water table fluctuation, whilst Bubier *et al* (1993) showed that deeper water tables were associated with lower production of CH_4 and linked this to oxidation. Although uncertainty in these relationships provides many issues in modelling carbon budgets, the basic processes are understood. Further work is needed to reduce uncertainty and understand the proportional roles of each cause of CO_2 output from a blanket peatland.

The balance between uptake and export of CO_2 gives the Net Ecosystem Exchange (NEE) of CO_2. This balance forms a substantial proportion of carbon budgets. In the UK the NEE of peatlands is generally negative, indicating net drawdown of CO_2 from the atmosphere (Worrall and Evans, 2009). For example, studies which have used static chambers (Ward *et al*, 2007; Rowson *et al*, 2010), extrapolated relationships (Worrall *et al*, 2009a) and eddy covariance (Billett *et al*, 2004) have all found negative NEE, even when the catchment as a whole was found to lose carbon (Rowson *et al*, 2010).

Atmospheric Methane (CH_4) Loss

CH_4 production is largely controlled by microorganisms found within peat (Blodau, 2002). Methanogens produce CH_4 via the digestion of organic matter and methanotrophs consume CH_4 as an energy source, and controls upon the presence of these bacteria types largely determines the CH_4 efflux (Moore and Dalva (1993). Methanogens are anaerobes and thus low redox conditions are required for optimal CH_4 production. The catotelm is therefore the primary producer of CH_4, although the acrotelm can produce some CH_4 (Lindsay, 2010). The main controls upon CH_4 production are water table, temperature and vegetative composition (Laine *et al*, 2007). Whilst these controls are similar to those for CO_2, the direction of the water table relationship reversed. For example, Moore and Dalva (1993) found a strong relationship between shallow water tables and increased CH_4 emissions, and this trend was reflected in the field by Bubier *et al* (1993). The type of vegetation determines the amount and quality of substrates available for Methanogens to produce CH_4 and vascular plants in particular can facilitate the release of CH_4 (Ström *et al*, 2003). Seasonal variations in CH_4 flux were observed by Laine *et al* (2007) who suggested that these were related to temperature (although this could also be related to water balance). This is supported Dunfield *et al*, (1993) who suggested that the sensitivity of methanogens to temperature is greater than methanotrophs, therefore at higher temperatures considerably more CH_4 must be produced than consumed. It is known that CH_4 production is subject to many temporal and spatial controls; however the importance of each and the relationships between them are not fully understood.

Fluvial exchange

Gaseous Supersaturation and degassing

CH_4 and CO_2 can become dissolved within the water contained within the peat mass beyond what would be expected in equilibrium with the atmosphere (Worrall *et al*, 2009a). When water is released from the peat mass into the fluvial network, the dissolved CO_2 and CH_4 can become 'degassed' if the water comes into contact with the atmosphere due to partial pressure differences (Billett and Moore, 2008). This is of concern because degassing is an additional route for atmospheric carbon loss. The rate of degassing is closely related to the proportion of open flowing water in the catchment and can be highly variable between peatlands (Billett and Moore *et al*, 2008). Both Hope *et al*, (2001) and Billett *et al*, (2004) have demonstrated that degassing can make a significant contribution to the carbon budgets of peatlands. Furthermore, Hope *et al* (2001) found that the proportion of dissolved CO_2 was much greater than dissolved CH_4. The large spatial variability observed in the rate of degassing along peatland channels (Billett and Moore *et al*, 2008; Hope *et al*, 2004) causes difficultly in accurately representing supersaturation and degassing in modelled carbon budgets.

Dissolved Organic Carbon (DOC)

DOC is a major source of carbon loss from the peatland system, constituting approximately 10% of carbon release in some peatlands (Holden *et al*, 2006b). DOC represents a wide range of compounds of varying size and characteristics, including fulvic and humic acids and a variety of polysaccharides (Wallage *et al*, 2006). These are produced by the decomposition of organic matter within peats (Evans *et al*, 2005). Due to the wide range of DOC compounds, identifying the processes of DOC production is difficult, although these processes are assumed to be largely biological (Holden *et al*, 2006b). A range of environmental variables have been associated with the production of DOC and much debate exists surrounding their relative importance, as a result predicting future DOC release is difficult. Most DOC exported from peat catchments has been found to be young in age, indicating that DOC is primarily produced in the acrotelm, although some studies do suggest older DOC is exported (ECOSSE, 2007). Temperature has been consistently associated with DOC production, primarily due its effect upon biological functioning (Evans *et al*, 2005), although an alternative explanation for this relationship is that DOC production is enhanced with increased CO_2 levels (Freeman *et al,* 2004). Hydrological changes are also thought to influence DOC dynamics for example changes in flow path may flush out DOC otherwise stored for the long term (Pastor *et al*, 2003). The relationship of DOC with drought and water table drawdown is also complex. A number of studies have shown increases in DOC production with water table draw down a relationship which is commonly related to the activity of phenol oxidase identified by Freeman *et al* (2001a). Although Clark *et al* (2005) highlight that not all studies have exhibited a increasing DOC release relationship with decreasing water table. Temporary acidification due to changes in the sulphur cycle has also been shown to suppress DOC loss during drought years (Clark *et al*, 2005). The literature currently paints a complex picture of controls on DOC production. No single process is has been identified to control DOC production, it is most likely to be a complex combination of all (Worrall and Burt, 2007). Therefore direct measurement or at least locally derived relationships are needed to understand DOC production in individual peatland environments.

Freeman *et al* (2001b) identified that DOC export had increased by 65% in UK uplands catchments over a 12 year period. This trend has been corroborated by both Worrall *et al* (2004a) and Evans *et al* (2005), who demonstrate consistent increases in DOC over varying time periods throughout the UK. Several causes of these increases are considered by Evans *et al* (2005) who conclude the increase is primarily a response to increasing temperature and declining acid deposition. Both Evans *et al* (2002) and Worrall *et al* (2004a) suggest that this increase is due to more factors than temperature alone. Evans *et al* (2005) suggest that predicting the future trends in DOC increase will be difficult until a there is greater understanding of the processes of DOC production.

Particulate Organic Carbon (POC)

POC is an important, but often overlooked, component of peatland carbon cycling. POC is produced and transported in peatlands by the processes of weathering and erosion. It is primarily lost from the peatland system by the fluvial network (Evans and Warburton, 2007). Fluvial losses of POC can be highly variable both within (Warburton *et al*, 2003) and between peatlands (Evans *et al*, 2006). POC losses from intact peatlands are minimal (Worrall and Evans, 2009) but have been shown to be very high from degraded sites (Evans *et al,* 2006). Consequently the importance of POC in a peatland carbon budget varies between sites (Evans *et al*, 2006).

Weathering of the peat surface, via frost action and desiccation, are the principal sources of POC yield in peatlands (Evans and Warburton, 2007). However, POC yield within the peatland does not necessarily control POC flux; the POC catchment output is related to the linkages enabling POC produced to move through channels and eventually exit the peatland (Evans and Warburton, 2005). The presence of vegetation in intact peatland prevents POC from moving within a channel, whereas in degraded peats with sparse vegetation POC can move more freely and is more readily lost from the peatland (Evans *et al,* 2006). The degree of POC loss is highly dependent upon the condition of the peatland, but can make up a substantial proportion of a carbon budget in many cases in the UK (Evans and Warburton, 2007). Although this thesis does not deal directly with erosion and POC losses an awareness of these processes of these additional influences is needed to put the findings into context.

Aolian Losses

Particulate Organic Carbon

The low bulk density of peat and the exposed nature of some peatland surfaces can lead to POC being lost by aeolian processes (Warburton, 2003). Aeolian erosion in peatlands occurs in environments of bare peat, where fluvial processes do not dominate (Evans and Warburton, 2007). Wind erosion of POC is an under-investigated process within peatland carbon dynamics, and as a result there is relatively little understanding of its importance (Evans and Warburton, 2007). However, Evans and Warburton (2005) demonstrate that fluvial fluxes are greater than aeolian losses in the catchment they studied. Greater research is needed to understand the role of aeolian processes, including the ultimate fate of POC, to allow a fuller understanding of peatland carbon dynamics.

2.2.3 Individual components of the carbon cycle and climate change

The different forms of carbon discussed above are not all equal in their influence on the climate. Gaseous emissions are directly atmospherically active and have an immediate effect. However CO_2 and CH_4 vary in their potential to cause atmospheric warming. CH_4 has a shorter atmospheric residence time, but is much more potent than CO_2 (Worrall *et al*, 2009b). Although CH_4 values are small in many carbon balance studies, this increased relative importance should be accounted for. DOC and POC form significant parts of many peatland carbon budgets but in this form are typically considered neutral in terms of climate change. However, both POC and DOC may ultimately contribute significantly to atmospheric flux. For example, Evans *et al* (2006) suggest that a proportion of POC may be oxidised within a peatland sediment budget and Worrall *et al* (2009b) suggest that up to 40% of DOC and POC may become atmospherically active. There is very little research into the processes, timescales and efficiency of transfer into the atmosphere of DOC and POC, despite their important role in the peatland carbon cycle. This is an area in need of further research and is relevant to any study considering the carbon balance of a peatland, over the long or short term.

Conclusion

There are several important components of the carbon cycle in peatlands and much research is being carried out to fully understand the processes and controls on each. This knowledge can be applied to understand how human activity and other factors are affecting peatland carbon balance. However, carbon cycling within a peatland is complex and as a result a full understanding of each component has not yet been gained. There are a number of sources of uncertainty in these studies, for example many are carried out in the laboratory which may not reflect the natural environment, or relationships may be variable between different peatland types. Despite this, significant advances are being made in the understanding of a peatland carbon cycle and this knowledge is now being applied in a number of ways, such as the creation of full carbon budgets for individual peatlands.

2.2.4 Carbon balance

In order to understand the role that a peatland is playing in the global carbon cycle, a full carbon budget can be calculated. This involves quantifying the balance between the individual components of peatland carbon cycling discussed in section 2.2.2. Budgets such as these allow an understanding of the routes by which carbon is lost, gained and transferred within a peatland. This breakdown is valuable information for understanding the climate change potential of a peatland and significant routes of carbon loss.

Carbon balance studies must incorporate the fluxes of carbon discussed in section 2.2.2 with as much accuracy as possible. These fluxes are complex and can be expensive, logistically difficult and time consuming to measure. As a result few measured complete carbon balance studies exist. To compensate, many studies use measurements, where available, in combination with literature values to fill gaps, or use established relationships to interpolate components of the budget (Billett *et al*, 2010).

Carbon balance studies have been carried out in both the UK and elsewhere. In Canada, a greater number of measured budgets have been calculated. The most comprehensive study by Roulet *et al* (2007) provides a measured six year carbon balance of a peatland which measures primary carbon fluxes, but omits POC. Much inter-annual variability in carbon balance was observed by Roulet *et al* (2007), highlighting the need for caution to be applied when interpreting short term carbon balance measurements, as they may not be representative of the long-term average balance.

Although international studies are of use, they are not from a similar peatland environment to the blanket peatlands of the UK. Consequently, the carbon dynamics occurring may differ and policy makers and scientists should primarily focus on studies from the UK. The calculation of carbon budgets in the UK has primarily been carried out using a combination of measured data supplemented by gap filling (Billet *et al*, 2010). A budget for Moorhouse National Nature Reserve (NNR) was carried out by Worrall *et al* (2003), using a mixture of measurement, with gap filling for fluvial losses in some months, and interpolation and secondary data for gaseous losses. For the same site Worrall *et al* (2009a) extended this study, using direct measurement of all components beside CH_4, This data was then extrapolated over a 13 year period. Long term measurements at Auchencorth Moss, Scotland have also been carried out. This site has produced two carbon budgets, both with a particular focus on losses from peatland surface waters. The first Billett *et al* (2004) provides a 2 year budget using discontinuous measurement of carbon fluxes, however DOC and POC were not distinguished between. Billett *et al* (2004) found that the site was either a small carbon source or was carbon neutral. The second study from the same site by Dinsmore *et al* (2010) provides more continuous measurement, again over a 2 year period, using eddy covariance and did distinguish between DOC and POC. In both studies it was concluded that fluvial losses were considerably important parts of the budget. Work on these sites is still ongoing and ultimately it is hoped that a fully measured carbon budget for a site in the UK can be gained (Billett *et al*, 2010). Both Moorhouse NNR and Auchencorth Moss are not heavily managed, however many of the UK's blanket peatlands are heavily affected by management and as a result these studies may not be wholly representative of a typical UK blanket peatland. More recently, Rowson *et al* (2010) used a similar Carbon budget technique in a catchment affected by drainage finding a positive carbon balance in the area measured. More sites studying such as Rowson *et al* (2010) are needed to provide a rounded understanding of UK blanket peatland carbon budgets.

The use of gap-filled carbon budgets is valuable, as there are considerable difficulties with creating meaningful fully measured budgets over appropriate timescales. However, gap-filled carbon budgets such of these will always have room for improvement, as the understanding of controls on blanket peatland carbon cycling is improved. For example, Billett *et al* (2010) highlight that the calculation of the carbon balance in Moorhouse NNR by Worrall *et al* (2003) and Worrall *et al* (2009a) has considerably changed with improvement in understanding of the processes controlling carbon budgets. If UK budgets are used with consideration of the limitations of gap-filling they are a very useful tool to be used by policy makers and researchers.

2.2.5 Carbon budgets and carbon accumulation methodologies

Calculating carbon balance (section 2.2.4) and carbon accumulation (section 2.2.1) are the two primary methodologies for understanding the role of peatlands in the carbon cycle. Each methodology aims to achieve a similar output, but has different attributes, allowing them to be used for different purposes (Table 2.2).

	Accumulation	**Carbon Balance**
Spatial representation	Local area near core sampling point, if only a single core is taken	Catchment scale
Temporal representation	Long term (10-10,000 years) carbon budget (using LORCA other values such as RERCA do not account for all loss)	Daily / Seasonal /Annual balance over short study periods (<1 year to 10 years)
Break down individual components of carbon cycle?	No	Yes

Table 2.2 Attributes of the carbon balance and carbon accumulation methodologies

Carbon budgets such as Worrall *et al* (2003; 2009a) account for the inputs and outputs of carbon on a peatland for the duration of a study. These budgets should be applied when a contemporary and detailed understanding of peatland carbon dynamics is needed. Carbon budgets are able to break down the components of the carbon cycle and can record net losses and net gains of carbon. Carbon budgets give values for entire catchments but can be very expensive and time consuming to carry out. Often these studies can only account for small time period (often 2 or less year). Roulet *et al* (2007) highlighted high inter-annual variability in peatland carbon budgets and this suggests that some studies are potentially unrepresentative of the long term trend.

Carbon accumulation studies account for the long term average annual carbon balance. Accumulation is calculated when a long term understanding of carbon dynamics is needed. Carbon accumulation is easier and cheaper to record, relative to a carbon budget. These studies account for the carbon balance at the point where the core was taken, not for the whole site. Multiple cores can be taken over large areas, such as in Clymo *et* al, (1998) and this can be a cheap way of understanding carbon dynamics over a large area. Carbon accumulation studies provide an end value only; they are unable to represent the routes of carbon loss and gain and do not account for losses of POC. Nor, is carbon accumulation calculation able to account for a positive carbon balance, as this value will be amalgamated into the long term average. As discussed in section 2.2.1 values of carbon accumulation represent different aspects of the overall carbon budget dependent upon where in the profile the dating was carried out. For example the calculation of LORCA accounts for all losses and gains, besides POC, and provides an average carbon balance over 1000s years, whilst the RERCA accounts for recent accumulation in the acrotelm and not the total carbon budget.

If all inputs and outputs are calculated in a carbon balance study and the site is a net carbon sink, the end value will closely relate to the carbon accumulation for that given year (see section 2.2.1). Assuming POC does not form a large proportion of the budget. Nilsson *et al* (2008) and Roulet *et al* (2007) both carried out contemporary carbon balance studies in addition to measuring long term peat accumulation. Both found little difference between contemporary accumulation rates and the LORCA / ARCA values. This helped suggest that their balance values were representative of the long term trend. This demonstrates how the use of carbon budgets and carbon accumulation methodologies can be used together to estimate changes in carbon dynamics.

It would be beneficial for carbon balance studies to be corroborated with a comparison of LORCA values, in order to gauge whether the gap filling is working (with consideration that the long term values do not account for losses of POC). Any substantial difference in LORCA and carbon budget values would either represent inaccuracy in the budget or a deviation in contemporary carbon balance from long term values. Lindsay (2010) provides a comparison of Worrall *et al* (2003) to LORCA values throughout the UK, finding that the values are broadly similar. Although these were not the LORCA from the site Worrall *et al* (2003) carried out his study, this broad comparison demonstrates that UK gap filled carbon budgets are generally reasonable.

2.2.6 Threats to peatland carbon

British blanket peatlands are sensitive environments, the conditions discussed in section 2.1.2 must remain in order for them to continue accumulating carbon. In the UK, blanket peatlands are semi-natural environments, existing within a dynamic equilibrium of upland landscapes (Evans, 2009). There is now increasing concern that anthropogenic activity is tipping the balance from this semi-natural state, towards conditions which are unfavourable for carbon accumulation and retention in blanket peatlands. A number of causes for this switch have been identified, ranging from extrinsic pressures such as climate change and atmospheric pollution, to intrinsic pressures such as increased land use. It is thought that the degradation of blanket peatlands in the UK is likely to be a result of a combination of all of these factors (Smith *et al*, 2007), each with varying influence depending on the environmental setting. Little is currently known about the response and resilience of carbon accumulation in blanket peatlands to these pressures.

Extrinsic Pressures

Extrinsic pressures on peatlands are those which do not occur on the peatland itself, these can be global, national and regional issues. The regulation of extrinsic pressures specifically to protect peatlands is inherently difficult, as commonly they have many sources, a range of impacts on other systems, and require action at a global, national or regional scale. The primary extrinsic threats to peatlands are the impact of climate change and atmospheric pollution.

Climate change

As discussed, blanket peatland condition is particularly determined by climate (see section 2.1). The climatic conditions required for blanket peatlands formation have been outlined by Lindsay *et al* (1988). Areas with the relevant climatic conditions required for blanket peatland formation are known as blanket peatland bioclimatic space (Clark *et al,* 2010). Lindsay *et al* (1988) is the most commonly quoted set of bioclimatic limits for blanket peatland formation, which include the following:

- Precipitation above 1000mm annually
- More than 160 rain days a year
- The warmest month has a mean temperature of less than 15^{o}C
- A limited variability in temperatures between seasons

Further assessment of thresholds for blanket peat formation by Clark *et al* (2010) revealed that using maximum annual temperature and the balance between potential evapo-transpiration and precipitation improved predictions of blanket peat bioclimatic space.

The fourth assessment report of the Intergovernmental Panel on Climate Change (IPCC, 2007) highlights that peatlands are one of the most sensitive environments to climate change. This is largely because peatland presence is so strongly controlled by climatic conditions; if climate change alters precipitation patterns and temperatures the long term impact upon water balance and decay rates in peatlands will be great.

Until recently, very little research had been carried out considering how climate change may impact upon blanket peatland bioclimatic space in the UK. However recent research by Clark *et al* (2010) and Gallego-Sala *et al* (2010) assessed the impact climate change may have under a number of bioclimatic rules (including a modified version of Lindsay *et al*, 1988) on the bioclimatic space of blanket peatlands in Britain. It was found by both papers that bioclimatic space associated with peatlands would significantly decline under the UKCIP02 climate change scenarios. Clark *et* al (2010) applied a range of bioclimatic rules for blanket peatland formation (not just Lindsay *et al*, 1988) and found these did not always predict similar spatial patterns of decline. Further work to refine the criteria defining bioclimatic space of British blanket peatlands would be greatly beneficial to helping target and monitor the worst affected blanket peatlands to climate change.

The movement and reduction of bioclimatic space as a result of climate change will have a number of effects on carbon cycling in peatlands. For example, where suitable bioclimatic space has moved away, a peatland may no longer be able to actively accumulate carbon, as the conditions required for peat accumulating vegetation will no longer be present and increased temperatures and drier environments may lead to increased mineralisation. These changes are largely dependent on the relationships between individual carbon forms with changes in water balance and temperature discussed in section 2.2.2. Uncertainty in predicting the consequences of climate change is associated with gaps in knowledge related to carbon cycling response to changing environmental condition. However, the effects of climate change upon a blanket peatland environment as a whole indicate that climate change will cause a gradual decline in the condition of the UK's blanket peatlands.

The direct effect of climate change on water balance, decay rates and other physical processes may not be the only cause of carbon loss related to climate change, as human activity will also change in response to climate. For example, climate change may cause an increase in recreational use of upland areas of the UK, which in turn will put increasing pressure on blanket peatlands through enhanced risk of wildfire (due to accidental burns) and peat erosion caused by visitors (Albertson *et al*, 2010). The effect of land management upon peatland carbon will be discussed in greater detail in the next section and in subsequent chapters of this thesis.

Atmospheric Pollution

Due to the UK's high population density, moorlands are never very far from significant human population centres. Consequently the UK's moorlands have been subjected to high levels of atmospheric pollution from a range of sources, particularly industry and transport emissions (Holden *et al*, 2006b). The type of pollutants deposited has varied over time; heavy metals and SO_2 were heavily emitted in the industrial revolution but have declined in the last two decades, whilst emissions of NO_x associated with transport have recently been the dominant emission source (Holden *et al*, 2006b).

The full effects of atmospheric pollution on peatland carbon cycling are manifold and discussing each of these is beyond the scope of this thesis. However, these are largely related to changes in species composition and production and decay rates. For example, the deposition of SO_2 has been associated with decline of *Sphagnum* mosses across many UK moorlands which in turn reduces peat accumulation (Ferguson and Lee, 1983). Not all moorlands in the UK are subjected to the same level of emissions, for example, the Peak District, which is surrounded by several large industrial cities has suffered immensely from atmospheric deposition (Crowle, 2007), whilst moorlands such as Dartmoor are under much less threat as a result of prevailing westerly winds and smaller less industrialised cities.

Intrinsic Pressures

Humans have a long history of interaction with British blanket peatlands and over time have helped shape the landscapes we see today (see section 2.1.4). However, blanket peatlands are sensitive and require certain conditions to retain and continue to accumulate carbon (section 2.1.2). Changes in management and pressures put upon upland environments especially during the past 200 years, as a result of increasing population and mechanisation, has caused substantial changes in blanket peatland functioning and carbon dynamics. These pressures are intrinsic and could be directly controlled for the benefit of blanket peatlands by policy and good stewardship, but only if it is understood how to strike the right balance. Blanket peatlands throughout the UK each have their own unique management patterns, pressures and histories. Nonetheless, there are three management activities that consistently cause concern about their effects upon carbon dynamics; grazing, burning and drainage. There is now an emphasis on understanding the action which needs to be taken to manage these activities to maintain blanket peatlands as carbon sinks and stores. This thesis is primarily concerned with the impacts of peatland degradation and drainage upon carbon dynamics and previous research into the effects of each of these will be discussed in greater detail in chapter 6 and to avoid repetition will not be reproduced here. However to put drainage, degradation and burning into context the effect of grazing and any other activities upon carbon dynamics must be reviewed.

Grazing

The last 100 years has seen considerable changes in the patterns of grazing on British blanket peatlands; perhaps the most notable change being the push for self sufficiency following World War Two, and the subsequent introduction of agricultural subsidies through the UK's entrance into the Common Agricultural Policy (CAP) in 1975. Both policies encouraged increased production through market manipulation, thus making it economically beneficial for graziers to maximise upland stocking rates due to guaranteed prices. The sudden increase in stocking levels put pressure on blanket peatland environments and it is thought this may be linked to some of the degradation of peatlands observed throughout the UK.

Most research into grazing on blanket peatlands has considered the effect upon vegetation patterns and the peatland surface, which in turn will impact upon carbon storage through the processes discussed in section 2.2.2. Grazing can initiate areas of bare peat through the weakening of vegetation cover where animals favour particular types vegetation and also through trampling, which then can be exacerbated by the effect of frost, wind and rain action (Evans, 1998). Linkages between bare peat areas and channel systems will be created by increases in bulk density reducing infiltration and causing overland flow along tracks generated by animals (such as observed in Meyles *et al*, 2006 on shallow peats).

Only a small number of papers have considered the impact of grazing directly on carbon dynamics. Most of these studies have been carried out on the Hard Hill experimental exclusion plots in Moorhouse National Nature Reserve (NNR). Garnett *et al* (2000) compared the RERCA of a grazed site and a controlled site, finding that grazed RERCA was not significantly different between the two plots. Ward *et al* (2007) considered the impact of grazing on multiple factors related to carbon cycling, finding that grazing significantly altered above ground storage of carbon, changed the vegetation community composition away from peat forming vegetation, increased CO_2 flux and slightly increased CH_4 flux and DOC export. Ward *et al* (2007) links most of these changes to an alteration of vegetation community composition. The differing results of Ward *et al* (2007) and Garnett *et al* (2000) may relate to the methodologies used: excluding grazing may have taken time to take effect upon the vegetation within the Hard Hill control enclosure and consequently carbon dynamics may have altered equally slowly. Garnett *et al* (2000) recorded the cumulative changes as the control site adjusted to grazing being excluded over a long period, whilst Ward *et al* (2007) was recording the contemporary effect on carbon after grazing had been excluded for a substantial period.

The studies of Ward *et al* (2007) and Garnett *et al* (2000) are of great use, however they are only able to reveal a certain amount about grazing and carbon dynamics. Local grazing patterns surrounding the grazing plots may have a significant effect upon the results, as stock do not graze evenly, having a tendency to group in areas with good grazing or shelter (Meyles *et al*, 2006). Therefore studies of other areas would be beneficial, to establish whether these observations apply elsewhere. Additionally the techniques used in Ward *et al* (2007) and Garnett *et al* (2000) and the use of the Hard

Hill plots would not have represented how grazing initiated erosion or changed hydrology, as discussed above, as the exclusion plots on Moorhouse NNR are small and therefore would not represent the landscape scale impact of grazing.

Recent changes in policy, events such as foot and mouth disease, and economic pressures faced by upland farmers have began to reduce stocking densities in the UK's uplands. Although this reduces some of the pressure on blanket peatlands, policy makers and scientists need to remain aware of the effects grazing can have and what kind of impact the current changes could have. Further work would be greatly beneficial to broaden understanding of this. A key problem in undertaking such studies is the poorly documented and controlled levels of grazing on blanket peatlands, with a general lack of experimental control sites.

2.3 Dartmoor

The study area for this thesis is the peat soils of Dartmoor, located in Devon, south west England. This section aims to outline the environmental setting of Dartmoor and reviews research on Dartmoor's blanket peat. Peat soils on Dartmoor have been taken to be those classified within the National Soil Research Institute (NSRI) soil series mapping as peat. In Dartmoor seven types of peat soil are present, ranging from peaty gley soils to true blanket peat. The blanket peat soils Crowdy and Winterhill are the primary focus of this thesis. Both Crowdy 2 and Winter Hill soils are found throughout British blanket peatlands (Avery *et al*, 1980). Dartmoor's blanket peatland consists of two distinct blanket peatland macrotopes, in the north and the south moor.

2.3.1 Environmental characteristics of Dartmoor

Dartmoor's blanket peatland is situated on a large impermeable granite batholith, meaning little of the precipitation Dartmoor receives is lost to the underlying geology. The intrusion of granite caused Dartmoor's landscape to be elevated from the surrounding countryside approximately 280 million years ago (Mercer, 2009). Dartmoor generally rises in elevation from south to north, reaching a maximum elevation of 621m at Higher Willhays. The increasing elevation results in decreasing temperature and increasing rainfall; as temperature and water balance are critical components of peat formation (section 2.1.2) this gradient has important implications for the distribution of Dartmoor's blanket peat formation. Due to Dartmoor's elevation and positioning in the south west of England Atlantic depression systems are first subjected to orographic uplift at Dartmoor leading to high rainfall levels (Simmons, 2003). Dartmoor has a hyperoceanic and temperate climate which largely fulfils all of the conditions outlined by Lindsay *et al* (1988) required for blanket bog formation (Table 2.3).

Climate characteristic	Lindsay *et al* (1988) requirements	Met Office (2010b) Dartmoor climate (average 1971 – 2000)
Annual rainfall (mm)	>1000	1974
Minimum number of wet days (>1mm rainfall)	160	181
Mean temperature for warmest month	<15^{o}C	Records show average maximum temperature of 17.7^{o}C in July (no mean made available)

Table 2.3 A comparison between Lindsay *et al* (1988) climatic conditions for blanket bog and long term climate statistics for Princetown (elevation 420m) mid Dartmoor (Met Office, 2010b)

2.3.2 Anthropogenic activity

Although Dartmoor fulfils all of the environmental and climatic criteria for blanket peatland formation, it is located very near the edge of the blanket peat bioclimatic envelope (Clark *et al*, 2010). Environmental factors alone may not have been enough to initiate blanket peat spread on Dartmoor. Moore (1984) suggests that the further south a blanket peatland is in the UK, the more likely anthropogenic activity played a role in blanket bog initiation. This hypothesis is supported in the palaeo-environmental record found within Dartmoor's peat. Initiation of the spread of blanket peat has concerned much of the research work carried out on Dartmoor (Caseldine, 1999). The work of Simmons (1964) corroborated by Simmons *et al* (1983); Caseldine and Maguire (1986); Caseldine and Hatton (1993; 1996) all identify an increase in microscopic charcoal with a decrease in tree pollen between 7000 – 6100 BP. Although there is little archaeological evidence to support this (Caseldine, 1999), these records have been associated with the activity of Mesolithic humans. In each record charcoal is found within the transition from woodland to open ground. This trend has been interpreted as representing a hunting strategy by Mesolithic populations who burnt woodland to flush out game, gradually reducing the tree line (Caseldine, 1999). Despite a long history of research into blanket peat initiation, little is known about the spatial and temporal timing of blanket peat initiation on Dartmoor (Caseldine, 1999). Caseldine (1999) highlights the issues of finding sites which can be dated to the Mesolithic. This research demonstrates that Dartmoor formed under similar influences to other peatlands in England and Wales (as discussed in section 2.1.4). A greater understanding of the spread and causes of initiation of Dartmoor's blanket peat would provide a valuable insight into the peatland landforms and carbon accumulation the Dartmoor.

2.3.3 Peatland functioning and carbon research on Dartmoor

Despite Dartmoor's long recognised environmental value, very little peer reviewed research has directly considered Dartmoor's peats beside the palaeo-environmental research discussed above. Notable exceptions include Charman *et al* (1999) and Meyles *et al* (2006). Charman *et al* (1999) is the only previous research considering carbon in Dartmoor's peatland. This study considered the movement of CH_4, CO_2 and DOC within a peat profile. It found that there was significant transport of carbon (primarily DOC) throughout a peat profile. The slow movement of water over 20 - 30 years throughout the profile is suggested to be the principle source of carbon transport. Transport such as this, may be an additional explanation for uncertainty found in relationships between DOC, CO_2 and CH_4 production with conventional mechanisms such as water table and temperature. This study was carried out on raised mire and not the blanket peat environment considered in this thesis.

Meyles *et al* (2006) considered the impacts of grazing on hill slope hydrology and stream discharge. The catchment had a small amount of blanket peat, but was mainly covered in 'peaty gley' soils. Meyles *et al* (2006) found that grazing significantly altered soil properties and the hydrological response to flood events. This study demonstrates that grazing on Dartmoor could potentially play a role in altering the processes and dynamics of peat covered environments, and these changes may also affect carbon dynamics. However, a very small proportion of the catchment studied was on blanket peat and therefore limited conclusions can be drawn from this with regard to the effect of grazing.

2.3.4 Dartmoor and national research

At present Dartmoor is poorly represented in national and international peatland carbon research, although references to Dartmoor exist in studies considering DOC and blanket peatland resilience to climate change at a national scale. These studies highlight why Dartmoor may be a very interesting environment to study peatland carbon dynamics. Worrall and Burt (2007) collated national DOC and water colour records for 315 catchments throughout the UK covering 10 or more years. In the majority of the catchments studied DOC levels were found to increase, whilst in the southwest catchments a decline in DOC was found. Long term DOC trends from the Rivers Axe, Dart and Tamar were considered in greater detail in this study. Declines in DOC in the River Tamar and Axe may not necessarily be related to peatland processes, as both catchments drain from land with a large proportion of mineral soils (especially the River Axe). However, significant declines were observed in the river Dart which does have a primarily peat-based catchment in Dartmoor. This may be indicative that Dartmoor may have been less affected by processes influencing DOC production outlined in section 2.2.2. Although further, more detailed research is necessary to support this theory.

Dartmoor is one of the blanket peatlands most threatened by climate change (Clark *et al,* 2010). Based on UKCP02 (Hulme *et al,* 2002) climate change predictions, the climate space associated with blanket peat growth will move away from Dartmoor over this century. However, Clark *et al* (2010) also found that Dartmoor is not sensitive to all changes in the climate and is more resilient to moisture changes than other variables. This reflects Table 2.3 which highlights precipitation levels far in excess of those required for blanket peat formation by Lindsay *et al* (1988). Considering Dartmoor's position in the far south western limits for blanket peat formation and its likelihood to be amongst the first blanket peatlands affected by climate change, it would be valuable to use Dartmoor to monitor peatland response to climate change. The quantity of research which has been carried out to date does not reflect this, and as a result, this thesis aim to be broad to provide a starting point to develop a base for future research into Dartmoor's blanket peat.

2.3.5 Management on Dartmoor

Dartmoor, like other moorlands in the UK, is affected by anthropogenic activity and management, which is thought to greatly influence peatland carbon storage (section 2.2.6). Contemporary Dartmoor is subject to a number of anthropogenic activities, some of which have long histories, whilst others are a more recent development. These management practices include burning, drainage, peat cutting, grazing, recreation and military activity. Although a number of these activities are similar to those in other blanket peatlands of the UK, subtle differences in tradition, socio-economics and the environment may have important implications for carbon management, which must be considered in order for the context of this thesis to be fully outlined.

Burning

Burning Dartmoor's moorland is an ancient practice which has always been, and still is, subject to much controversy (Greeves, 2006). On Dartmoor the practice of burning the moorland is commonly referred to as '*swaling*', an ancient term which has maintained a strong hold in the south-west region (Greeves, 2006). Both managed burns and unplanned burns have occurred on Dartmoor and during the 20th century there was concern that Dartmoor's blanket bog may have been burnt too often (Mercer, 2009). The motivation for managed burning is not for grouse management, as in many other areas of the UK, as on Dartmoor there are no commercial shooting estates. The practice of *Molinia* burning is instead more prevalent (Yallop *et al*, 2006) which has traditional agricultural origins related to encouraging fresh bite for grazing stock (Mercer, 2009). Attitudes to burning have constantly changed over the years (Greeves, 2009). Currently managed burning on blanket bog is not advised by the Heather and Grassland Burning code (Natural England, 2007) and is heavily discouraged by local management authorities causing more regulation over the practice in Dartmoor than there has ever been (Greeves, 2006). However, unplanned fires do still occur on the blanket bog, as recorded by the Dartmoor National park Authority (DNPA) (see Figure 2.2). Sources of ignition for unplanned fires include arson, poorly managed burns (Mercer, 2009, pp 115) and accidents due to recreation and military activities (as recorded in DNPA fire GPS records, Figure 2.2). The practice of burning blanket peat is currently a matter of great debate amongst the land users and managers on Dartmoor.

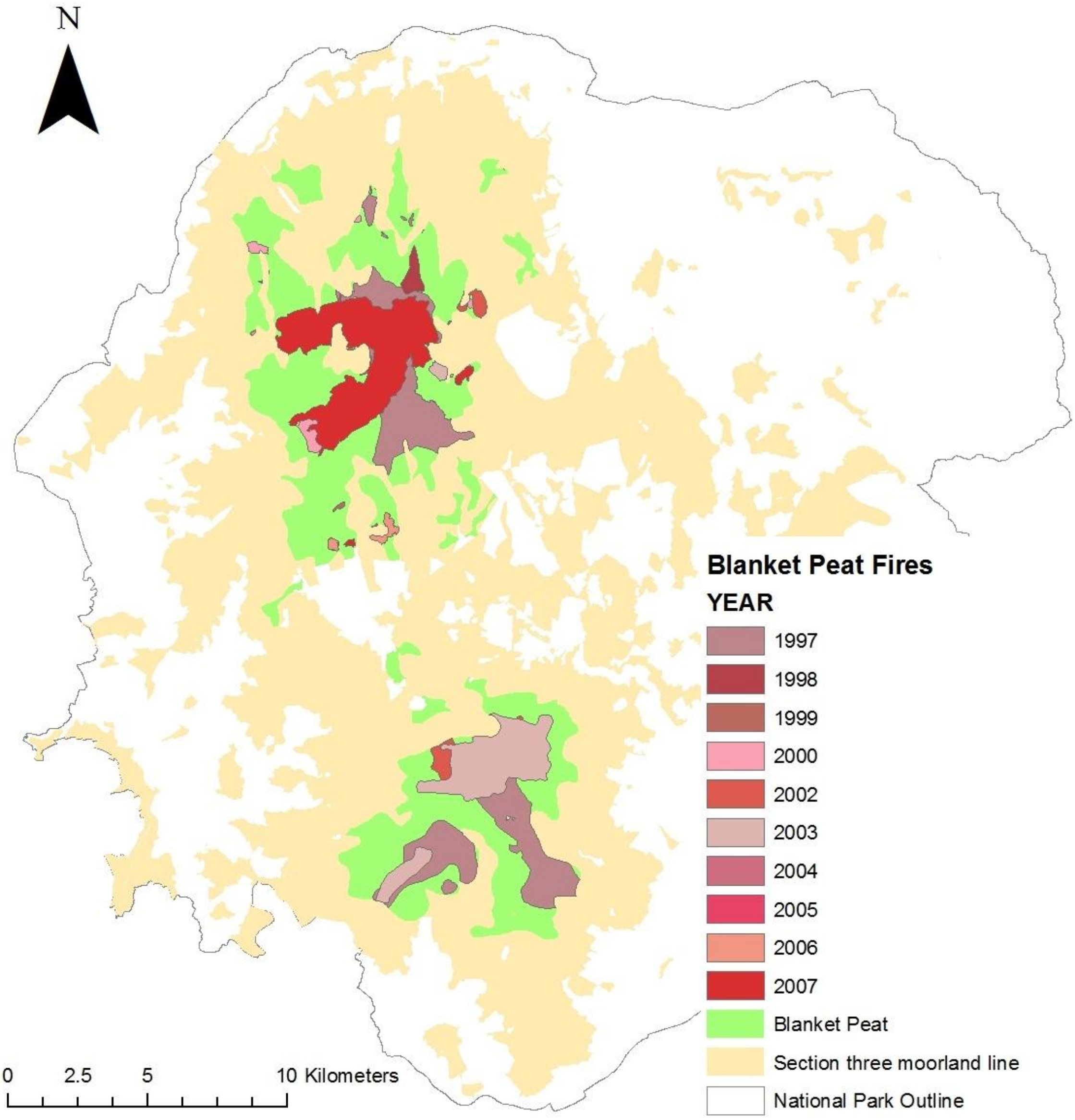

Figure 2.2 Burning patterns on blanket peat between 1997 and 2007 from DNPA GPS readings. Blanket peat areas are defined by the Crowdy 2 and Winter Hill soil series mapped by NSRI natmap

Drainage and peat-cutting

Little is known about the timing and extent of drainage, turbary and commercial peat extraction on Dartmoor. However, Duchy of Cornwall records reveal that all have occurred to varying extents in the past. Turbary has been a right of the commoners on Dartmoor for at least 1000 years (Mercer, 2009) and signs of extraction are evident in a number of easily accessible sites on the moor. Despite this, the right of turbary is no longer practiced and has not been so in living memory of many of the commoners on Dartmoor (Newman, 2010). The cutting of drainage channels is also evident in a number of locations; it is thought that these were largely to facilitate commercial peat extraction, such as on Amicombe Hill, or agricultural improvement, such as on Prison Farm new-take.

Grazing

Grazing of sheep, cattle and ponies has always occurred on Dartmoor, at least in historic times and probably well before. However, several considerable changes in grazing regime and practice have occurred across the entire moor since the beginning of the 19th century. At this time, the commoners of Dartmoor began to over winter stock on the Moor with more hardy breeds (Mercer, 2009). This change put more pressure on Dartmoor's vegetation, as over-wintering did not allow recovery of vegetation in the spring (Mercer, 2009). Following this in the mid to late 20th century Dartmoor's grazing levels again rose when Dartmoor, like other moorlands in the UK, became subject to national agricultural subsidy (Mercer, 2009). Grazing levels on Dartmoor during this period were some of the highest in the UK (Sansom, 1999). Over grazing has been recorded by conservationists as a concern until relatively recently on both the south (Colston *et al*, 2007) and north (Natural England, 1999) of Dartmoor. Local intervention measures were put into place as a response to overgrazing 1990s (Mercer, 2009), but subsequent changes in national policy and the consequences of foot and mouth again changed the situation, causing the numbers of graziers to diminish from the moor

(Mercer, 2009). The frequent and substantial changes to grazing regime in the last 100 years may have resulted in change in peatland functioning, like that observed by Meyles *et al* (2002) on thin peat soils. However due to the large common lands of Dartmoor, a complex land management pattern and variable grouping of animals, the levels of stocking in individual areas are largely unknown, therefore reconstructing this effect is difficult.

Military Activity

Dartmoor's open moorland, particularly in the north of the moor, has been used by the military for no less than 200 years (Mercer, 2009). Military activity in the past has put considerable pressure upon the fragile blanket peat. Firing of shells and mortars into the peat has caused disturbance (Figure 2.3 demonstrates the 864 craters found within a 1km area) which can cause great upheaval of the peat (such as pictured in Mercer, 2009). These activities may have initiated some of the erosion, disturbance and desiccation of the peat presently evident on north Dartmoor. However formal links between the Ministry of Defence (MoD) and DNPA have seen a turn towards more considerate use of the land. The MoD ceased to fire shells in 1996 and has actively funded many environmental surveys since this time (Mercer, 2009).

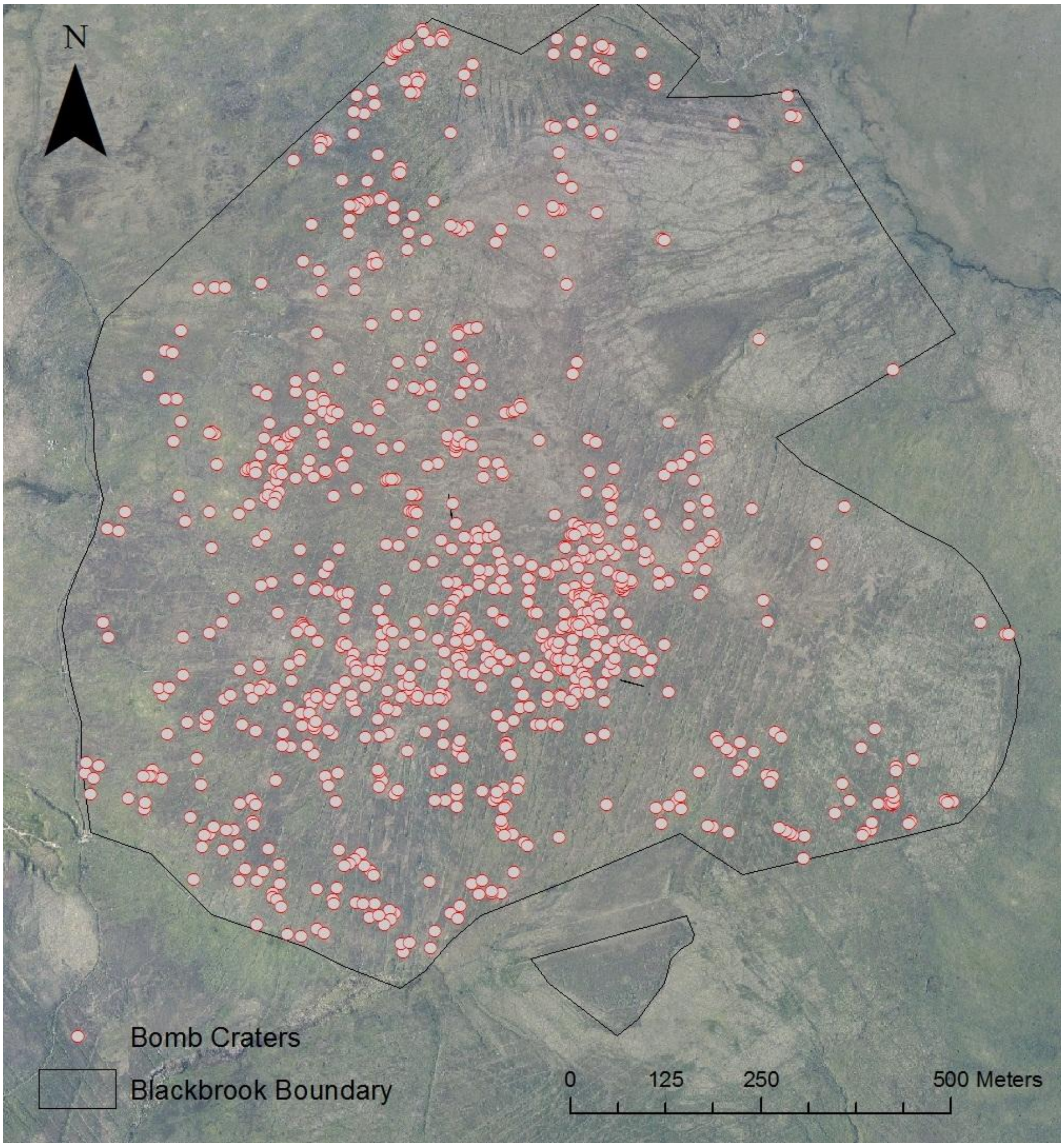

Figure 2.3 Bomb craters found within the Blackbrook Head area

3 Methods for modelling peat depth in blanket peatlands

A manuscript based on this chapter is accepted subject to revision, in Soil Use and Management (6/12/10).

3.1 Introduction

For a long time, British blanket peatlands were considered to be extensive, inaccessible marginal environments of low economic and agricultural value (Bevan, 2009). Mapping of peat depth and extent was restricted to small areas of economic value, such as those accessible for peat cutting (Chapman *et al*, 2009) and little attention was paid to understanding blanket peatland as a landform. However, recently attitudes have changed; they are now considered one of the UK's most valuable natural environments, due to their provision of ecosystem services such as water provision, carbon storage and biodiversity (Bonn *et al*, 2009b). As activity to understand and protect blanket peatlands has increased, there is a greater need for improved baseline data, to guide policy makers, land managers and researchers.

Many of the processes and ecosystem services that peatlands provide occur beneath the surface, such as carbon cycling and storage and hydrological processes, and extend throughout the whole profile of the peat. Blanket peatlands are highly variable in depth (Bragg and Tallis, 2001) and as such, the value and function of each ecosystem service is affected by this. Therefore understanding a peatland as a three dimensional landform is equally important as mapping its spatial extent. At present there are is no standardised methodology for mapping blanket peat landforms in three dimensions. Most studies must use unreliable estimates of extent (Worrall *et al*, 2003), treat peatlands as homogenous environments (Gorham, 1991) or do not include full depth profiles (Bradley *et al*, 2005; Milne and Brown, 1997) and are therefore limited in their accuracy.

Blanket peatlands form in hyperoceanic climates. Cool, moist conditions throughout the year allow peat to form on slopes and summits where runoff would normally prevent it (Lindsay, 1995). Their formation is consequently influenced to a greater extent by topography than many other peatland types. Peat forms earliest and deepest on flatter areas such as valleys and summits, with shallower peat developing on sloping ground. Elevation also has a considerable role to play, as decreasing temperature and increasing rainfall at increasing elevation produce more favourable conditions for peat formation. This study investigates whether relationships between peat depth, topographic variables and vegetation characteristics can be used to model blanket peat depth and spatial extent. By utilising these relationships a better understanding of blanket peatlands landform can be gained, providing land managers and scientists with baseline data needed for ecosystem services evaluation and management. The subsequent chapter will deal with the implications of this for estimating carbon storage at similar spatial scales.

3.2 Methodology

3.2.1 Study Site

Dartmoor is an area of isolated moorland in South West England, which contains a significant area of hyper-oceanic blanket bog. The geology of the area is predominantly impermeable granite. Frequent frontal systems and orographic uplift lead to a high average annual rainfall of 1974mm per year (Met Office, 2010a) in a temperate environment. Many of the characteristics of Dartmoor's blanket peat and factors controlling its development are considered similar to other UK blanket peatlands.

The study focuses on areas of open moorland on Dartmoor (471 km^2), more specifically the area classified by DEFRA as the 'Section Three Moorland Line' within which all of Dartmoor's blanket peat is located. All peat soil associations were modelled, to ensure all peat was covered, including the shallow peat in heathland areas. The following soil associations were included: Crowdy, Winter Hill, Princetown, Wilcox, Hafren, Hexworthy and Laployd, each of which occurs in other areas of UK moorland (Findley *et al*, 1984). These associations vary from raw acid peats to humic gleys with a peaty topsoil (Findley *et al*, 1984). Mire, heath and acid grassland are the principal vegetation communities found within the moorland line.

3.2.2 Spatial Units

Spatial units within the moorland line were defined to provide a spatial framework within which peat depth could be modelled. These spatial units allow the influence of soil and vegetation types upon peat depth to be represented. To produce these units the National Soil Research Institute (NSRI) soil series map and Dartmoor National Park land cover map were intercepted within *arc*GIS 9.3, creating spatial units known as 'Carbon Unit Areas' (CUAs) (Figure 3.1). Initial interception led to over 200 CUAs, which was too many to allow a reasonable field sampling strategy, as a result these CUAs were reclassified to create larger spatial units. Similar soil types were grouped into 'soil units' (Table 3.1) and within the vegetation dataset any polygons under 1 ha in size were merged with their nearest neighbour. This substantially reduced the number of CUAs. Each CUA has a 'soil unit' and a 'vegetation classification' it is found within. Descriptive statistics for the resulting CUAs can be seen in Table 3.2.

NSRI Soil series	**Soil Units in model**
Hafren, Hexworthy, Laployd	Peat to Loam
Winterhill, Crowdy	Blanket Peat
Princetown, Wilcocks	Shallow Peat

Table 3.1 Regrouping of NSRI soil series into soil units

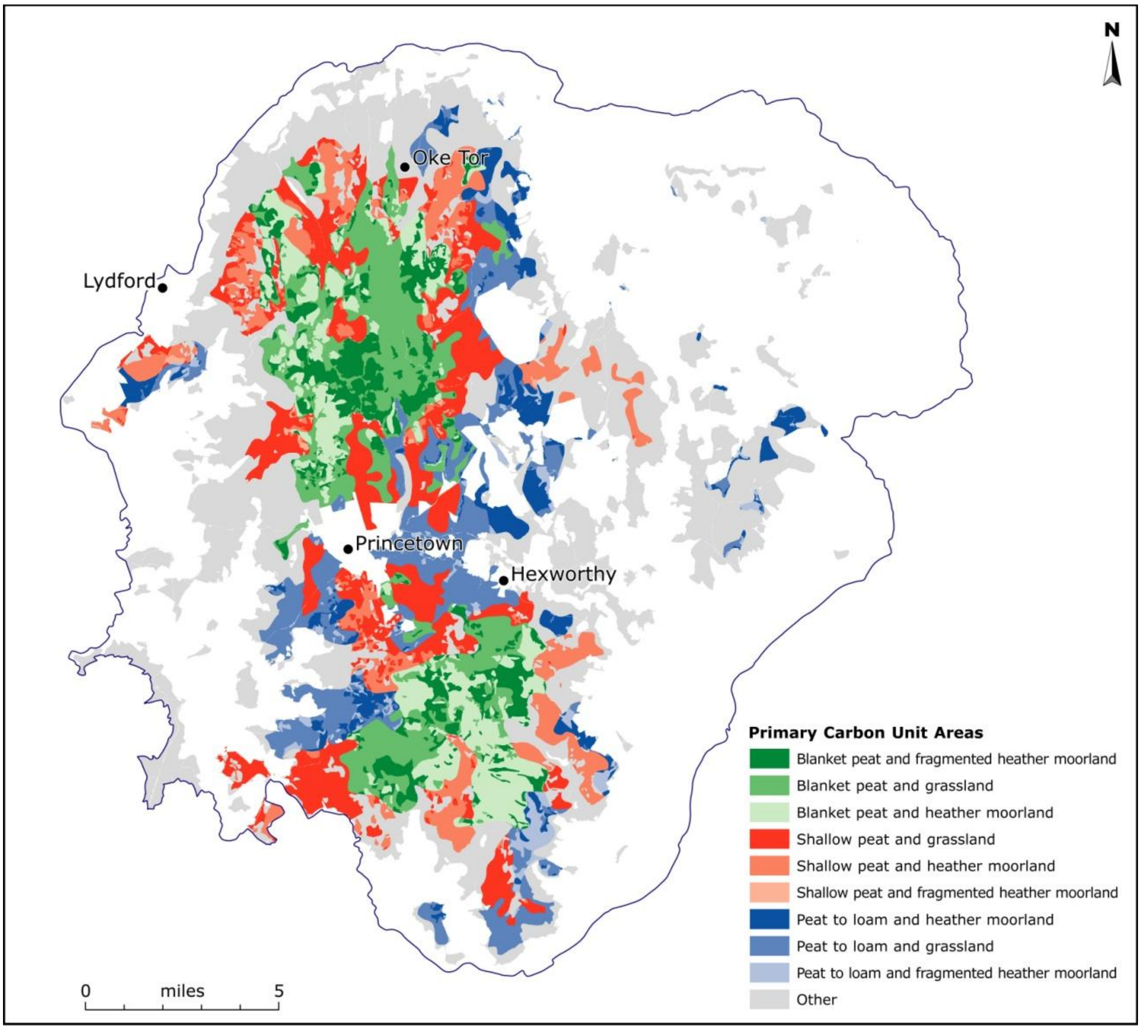

Figure 3.1 Carbon Unit Area (CUA) distribution within the moorland line

CUA	Coverage (ha)	Elevation (m)			Slope °		
		Mean	Min	Max	Mean	Min	Max
Blanket Peat and Heather (BPH)	3587	473	281	609	4	0	38
Blanket Peat and Grassland (BPG)	5055	473	292	599	4.4	0	44
Blanket Peat and Fragmented Heather (BPFH)	2810	492	294	601	4.1	0	41
Shallow Peat and Heather (SPH)	3709	429	155	618	5.9	0	39
Shallow Peat and Grassland (SPG)	5869	407	155	581	6.5	0	40
Shallow Peat and Fragmented Heather (SPFH)	1637	411	185	592	6	0	41
Peat to Loam and Heather (PLH)	2022	364	94	494	5.7	0	42
Peat to Loam and Grassland (PLG)	4477	354	136	528	5.9	0	48
Peat to Loam and Fragmented Heather (PLFH)	895	352	174	529	6.2	0	32
All	30061	417	94	618	5.4	0	40.6

Table 3.2 Descriptive statistics for peat based CUAs

3.2.3 Depth Sampling

A field sampling strategy was developed to identify statistical relationships between slope, elevation and peat depth within each CUA. To ensure representative sampling of slope and elevation a stratified sampling technique was used. 1000 sampling points were identified, these were area weighted for each CUA. Within each CUA the sampling points were distributed incrementally at 15 – 30m increases in elevation class

(Table 3.3). Variation in increment between CUAs occurred to allow the correct number of points to be allocated according to their different area weighting. The number of points assigned within an elevation class was proportional to spatial coverage of that elevation, allowing representation of spatial variation. Samples allocated to each elevation class were then stratified by slope. Table 3.3 outlines sample point allocation and Figure 3.2 maps peat depth sampling points. A Next Map 5m Digital Elevation Model (DEM) was used which is derived from Interferometric Synthetic Aperture Radar (IFSAR), the *arc*GIS Spatial Analyst extension was used to create a slope model from this DEM.

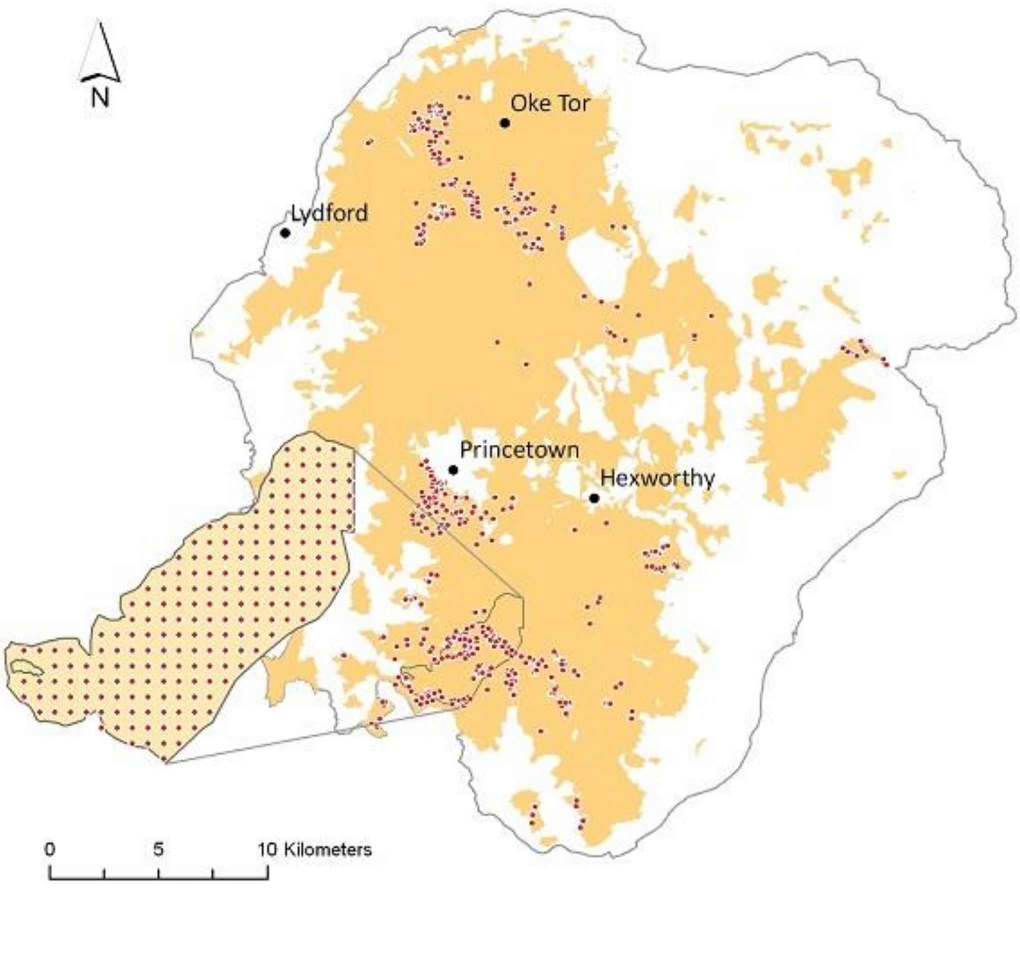

Figure 3.2 Sample point allocation within the moorland line with inset of validation points

Spatial Unit	Sample points (proportionally assigned)	Sample elevation increase (m)	Mean (cm)	Std Dev	Min depth (cm)	Max depth (cm)
Moorland Line	1001		38.1	44.2	0	329
Blanket Peat	382	20	80.7	79.8	0	329
Blanket Peat and Grassland	168	20	88.1	82.1	0	329
Blanket Peat and Heather	120	20	70.0	64.7	9.8	296
Blanket Peat and Fragmented Heather	94	20	75.8	85.4	0	299
Shallow Peat	374		22.6	26.0	0	181
Shallow Peat and Heather	124	20	18.1	14.5	1.8	96
Shallow Peat and Grassland	195	15	24.9	29.1	0	181
Shallow Peat and Fragmented Heather	55	20	25.6	31.8	3	154.9
Peat to Loam	246		12.2	7.7	0	55.9
Peat to Loam and Heather	67	30	15.5	9.19	5	55.9
Peat to Loam and Grassland	149	15	11.3	7.02	0	32.9
Peat to Loam and Fragmented Heather	30	60	12.29	8.26	2.5	40.4

Table 3.3 Peat depth characteristics within each spatial unit

3.2.4 Field methodology

Each point was located using a Trimble Geo XS differential Global Positioning System (GPS), accurate to 30cm real time. This ensured the data recorded matched the DEM used within the sampling strategy. Peat depth was recorded using an extendable steel probe and pushed in until the point of resistance. Five peat depths were recorded at each point, one central point and four at right angles four meters from the centre (Figure 3.3). This allowed localised variability in peat depth to be accounted for as much as was feasibly possible. For each location the average of the five peat depths was used for regression. A very small number of points were inaccessible or severely altered; in this case the point was either reallocated, or removed if the representative sampling strategy was not affected and an accessible replacement point could not be found. Additionally slope was recorded in the field using a clinometer and observations were made about the site, including scale of erosion and evidence of present or historical management.

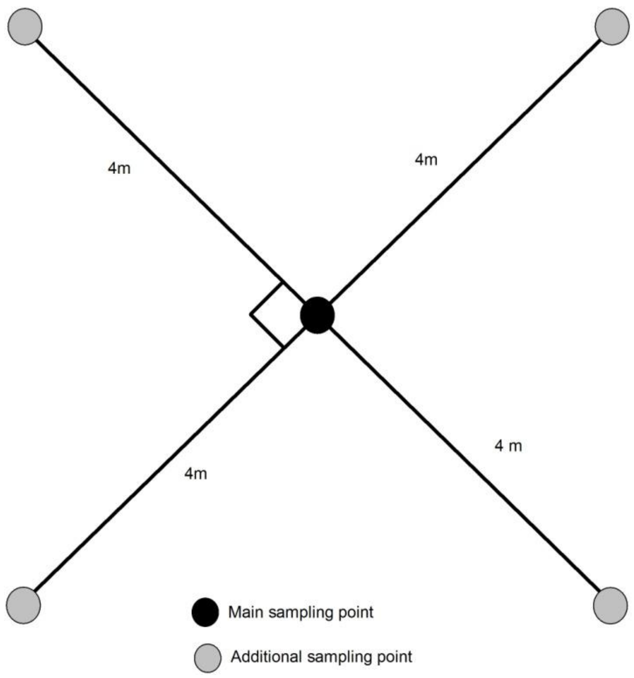

Figure 3.3 Field peat depth sampling design

3.2.5 Validation dataset

To validate the final model and investigate other methods of mapping peat depth, an independent dataset was obtained (Figure 3.2 inset). Using a grid pattern 200 depths at 250 meter intervals were recorded in a southern area of the moor by Jonathan Noades, University of Plymouth, which covered all CUA types. This area also contained a number of the original sampling points, which were used to cross validate a kriging based depth map.

3.3 Results

3.3.1 Descriptive depth statistics

A maximum peat depth of 329 cm was recorded and a minimum of 0 cm (Table 3.3). Greatest average peat depths and standard deviations were found within the blanket peat CUAs. Average peat depths in shallow peat and peat to loam CUAs are below 30cm, less than the depth regarded as a true peat deposit (Avery, 1980). However, in the shallow peat CUAs an upper quartile close to 30cm and a maximum peat depth of 56cm within the peat to loam CUAs suggests areas of true peat deposit do exist within these soil units, potentially in the form of valley mires or small areas of deeper peat within shallower peat soils.

3.3.2 Peat depth relationship with topographic parameters

Non linear univariate relationships between peat depth and slope, and peat depth and elevation can be seen in the blanket peat CUAs (Figure 3.4). Similar relationships can also be seen in the shallow peat CUAs (Figure 3.4) although these relationships are weaker. Poor relationships were found in the peat to loam soil unit when slope and elevation are considered separately (Figure 3.4). When slope and elevation are considered together in a multivariate relationship they demonstrate a similar good relationship with peat depth in the blanket peat soil unit (Figure 3.5), a relationship is also evident in the shallow peat soil unit but is not as strong, and no multivariate relationship is seen in the peat to loam soil unit (Figure 3.5). This indicates that the approach of modelling peat depth using both slope and elevation will be most successful when considering raw blanket peat soils.

Multivariate regression was used to identify if these relationships were statistically significant (Table 3.4). These relationships were investigated within different spatial units: the moorland line as a whole, within soil units and within individual CUAs. This allowed understanding of the necessity to represent soil unit and vegetation type within the model. It was found that if the model was not split into spatial units the R^2 values were greatly reduced (see Table 3.4). The model which has been split into CUAs consistently has the highest R^2 values, demonstrating that splitting the model into CUAs was a worthwhile exercise.

During the field survey a number of the points were noted to be disturbed as a result of peat cutting, tin mining or erosion. It is possible that these may have a great influence upon the strength of the regressions, column b of Table 3.4 provides R^2(adj) values when these disturbed points had been removed from the dataset. This removal reduced the R^2(adj) statistic for blanket peat CUAs but increased it slightly for shallow peat CUAs. Scatter plots in Figure 3.4 indicate that relationships are non-linear for both slope and elevation in the blanket peat CUAs and possibly also in shallow peat CUAs. Log transforming the data was carried out and much improved R^2(adj) values in both blanket peat and shallow peat CUAs (column c of Table 3.4). Log transforming was not necessary for the peat to loam CUAs, as the relationship was very weak and linear (Figure 3.4 and Figure 3.5). Stepwise regression was used to determine which combination of variables was best for predicting depth in each area (Table 3.5). Using both slope and elevation is the best combination in most cases. The importance of slope and elevation varies between each spatial unit. In blanket peat, slope and elevation are equally good predictors of peat depth in all CUAs, apart from Blanket Peat and Heather (BPH) where elevation demonstrates no relationship with peat depth, probably as BPH recorded several shallow peat depths at high elevations and the sampling strategy did not pick up deeper peat (Figure 3.4).

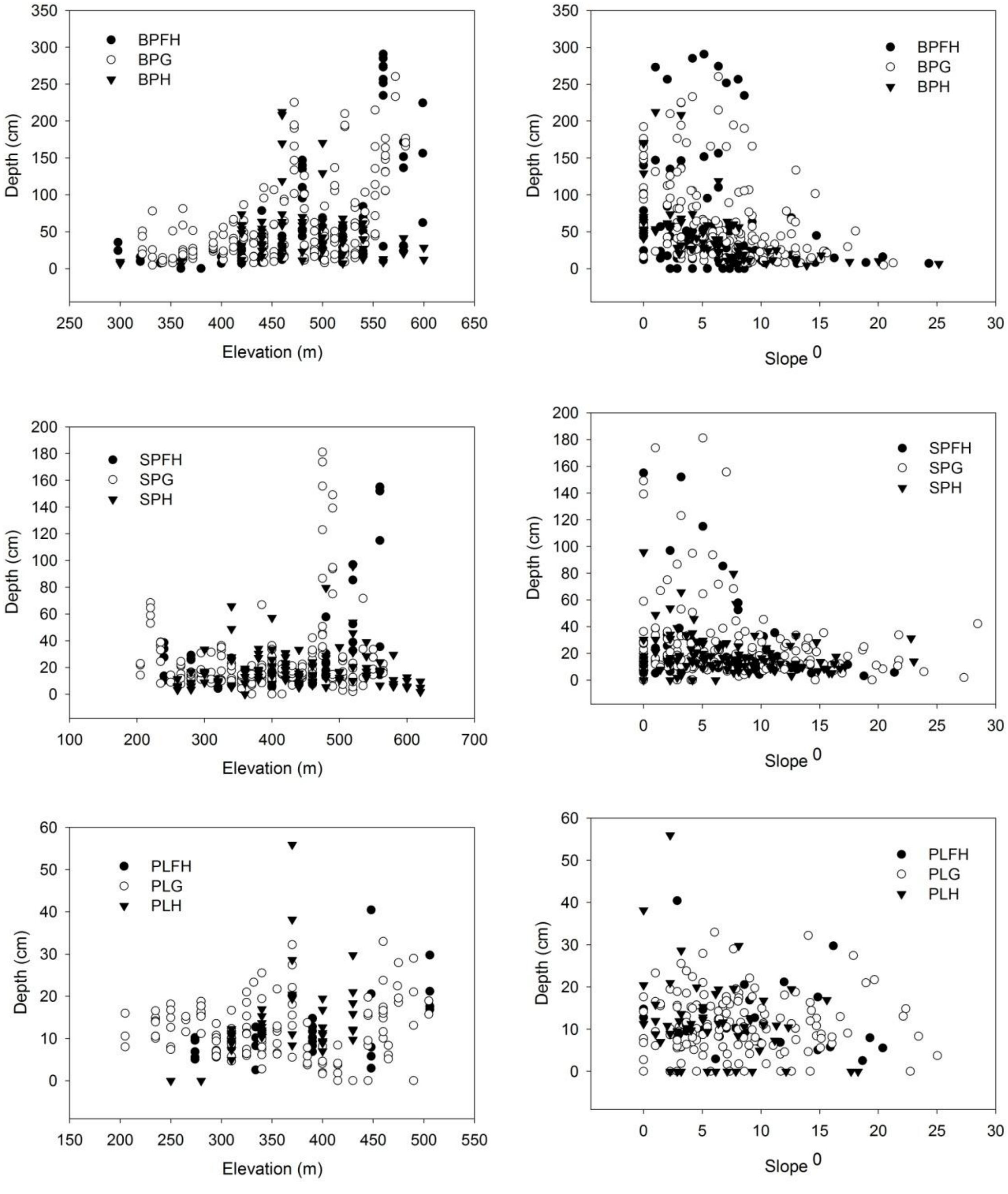

Figure 3.4 The relationship of slope and elevation individually to depth within each soil unit and CUA

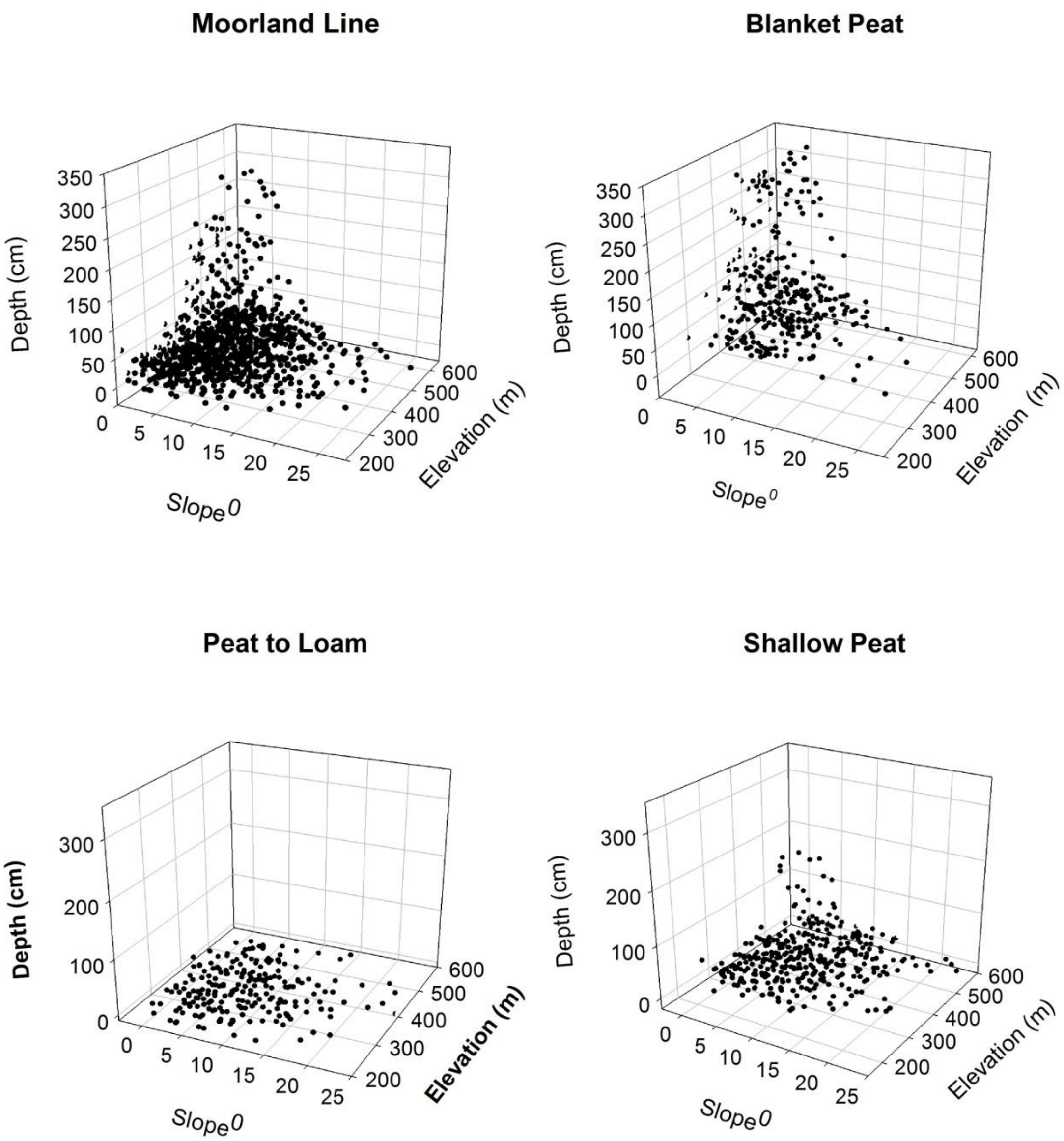

Figure 3.5 Multivariate relationships between peat depth, slope and elevation between each soil unit.

Spatial Unit	a.		b.		c.	
	All data R^2 (adj) %	All data RMSE (cm)	Disturbed removed R^2 (adj) %	Disturbed removed RMSE (cm)	All transformed data R^2 (adj) %	All transformed data RMSE (cm)
Moorland line	21.9	39	16.3	39.9	27.4	9.43
Blanket Peat	32.4	65.6	25.4	53.1	36.5	21
Blanket Peat CUA	40.4	59.5	31.8	48.7	51.6	13.9
Shallow Peat	7.3	25	9.4	25.3	10.2	6.21
Shallow Peat CUA	13.2	23.1	16.5	23.6	19.8	5.5
Peat to Loam	5.8	7.2	None		Linear	
Peat to Loam CUA	P>0.05		P>0.05			

Table 3.4 Multivariate relationships between slope, elevation and peat depth: column a. relationships including all data; b. data points marked as disturbed removed; and c. all data transformed using a natural logarithm

Spatial Unit	Regression to include	Dominant variable	R^2(adj) values (%)		
			Bivariate	Slope	Elevation
Moorland line	Bivariate	Elevation	27.4	8.4	19.1
Blanket Peat	Bivariate	Elevation	36.5	24.9	24.9
Blanket Peat and grassland	Bivariate	Elevation	44.5	24.6	30.9
Blanket Peat and heather	Bivariate	Slope	56.3	53.6	0
Blanket Peat and fragmented heather	Bivariate	Elevation	57.8	24.2	39.7
Shallow peat	Bivariate	Slope	6.5	5.3	0.7
Shallow peat and grassland	Bivariate	Slope	5.8	4.7	P>0.05
Shallow peat and heather	Univariate	Slope	P>0.05	1.9	P>0.05
Shallow peat and fragmented heather	Bivariate	Elevation	31.9	14.0	14.2
Peat to loam	Bivariate	Elevation	5.8	P<0.05	4.7

Table 3.5 Stepwise regression indicating which variables to include in peat depth model, with R^2(adj) values demonstrating the difference between bivariate and univariate relationships with slope and elevation.

3.3.3 Local depth variability

To account for local variation in depth as a result of variability in the underlying geology and surface vegetation, measurements were replicated five times at each sample point. These replicates reveal the extent of local depth variability and the necessity of replicate points at each site. For each point the standard deviation of the replicates was calculated. It was found that the average standard deviation for each soil unit and CUA was small (between 3.8 and 12cm). However at isolated points large variation in local depth was recorded (Figure 3.6). The greatest local variability in depth was recorded in the Blanket Peat CUA; this is possibly as a result of a greater relative depth.

Points with high local depth variability may be outliers and may weaken the relationship of slope and elevation with depth. As a result, points with a standard deviation above the 95th percentile were removed from the blanket peat dataset and the relationship reinvestigated. This treatment did little to improve the model's regression with an average CUA R^2(adj) value of 50% when log-transformed. This is no greater than the average log-transformed blanket peat CUA value of 51.6% (Table 3.4, column c.). As a result removal of points with considerably variable depth locally was not applied within the final model.

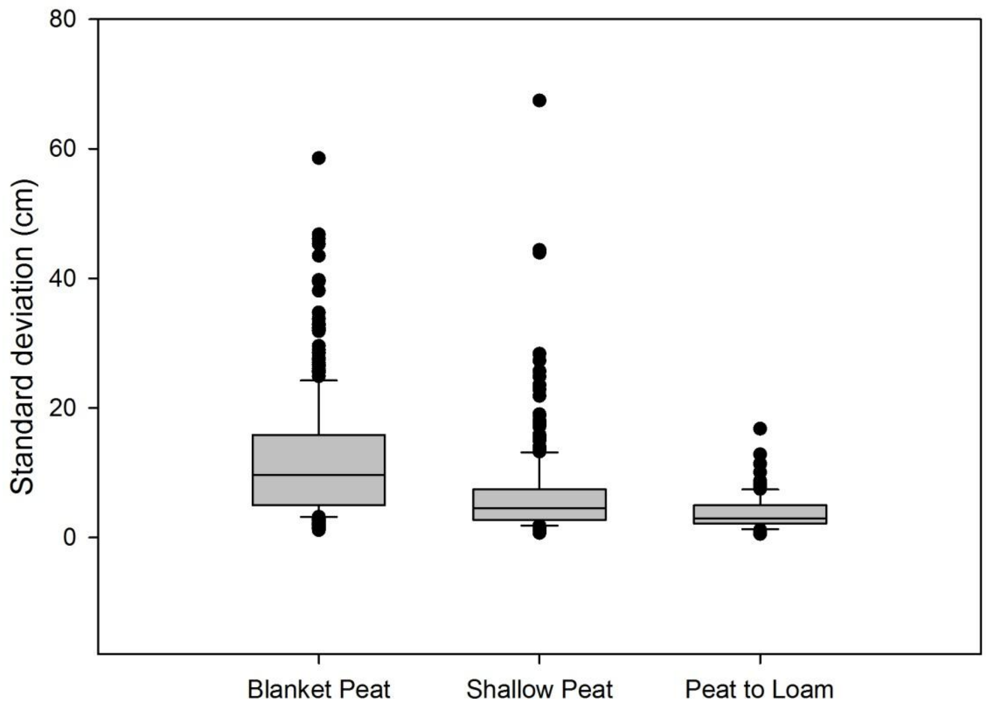

Figure 3.6 Box plot demonstrating the variability in local peat depth between points sampled in each soil grouping

3.4 Modelling

Using the relationships found in section 4.4, three peat depth models were created in *arc*GIS using the following spatial frameworks: the moorland line; soil units and the CUAs. The relationships used in these models were selected according to the variables identified by stepwise regression (generally both slope and elevation, see Table 3.5) and the datasets with the highest R^2(adj) and lowest RMSE from Table 3.4. In the shallow peat and blanket peat areas log-transformed data was used and relationships without disturbed points for shallow peat and with disturbed points for blanket peat (see Table 3.4). Each depth model was subsequently tested against the validation dataset (Figure 3.2) and assessed for ability to fit the distribution of the original datasets (Table 3.6). In blanket peat, the regression caused a constant under estimation in depth, when compared to the original data points as a result of the log transformation. To correct this, the equation of the straight line between the original fit and a perfect fit was added to the model, resulting in a better model fit (Figure 3.8).

Subsequently data in Table 3.6 was reviewed to identify the best-performing model; those highlighted indicate elements included in the final peat depth model. The models' predictive abilities were substantially improved when CUAs were considered separately in shallow peat and blanket peat soil units. As a result individual depth models using the CUA spatial framework were used. The final model was developed by applying the relevant regression to each spatial unit, using grid based map algebra. Following this each model was brought together using the mosaic to new raster function in *arc*GIS 9.3. The model's final output is illustrated in Figure 3.7, the fit with original data is outlined in Figure 3.8, which achieves an R^2(adj) of 53.3% and RMSE of 54.4.

Spatial Unit	Dataset used	Observed depths: Mean (cm)	Standard deviation (cm)	Min depth (cm)	Max depth (cm)	Model depths: Mean (cm)	Standard deviation (cm)	Min depth (cm)	Max depth (cm)	Validation dataset (Spearman's Rho)	Model fit against observed points R^2(adj)
Moorland line	All	31.9	44.2	0	329	21	10	0	70	0.6	0.02 (p>0.05)
Blanket Peat	All	80.7	79	0	329	79	41	0	276	0.6	0.39
Blanket Peat with CUAs	All	80.7	79	0	329	***93***	***50.6***	***0***	***377***	***0.56***	***0.53***
Shallow Peat with CUAs	Disturbed removed	22.6	26	0	181	***23.9***	***7.9***	***0***	***109***	***0.32***	***0.12***
Peat to Loam	All	12.86	55	0	7.4	***17***	***4***	***1.4***	***12.7***	***NA***	***0.06***

Table 3.6

A comparison of observed and modelled peat depth for different spatial units, allowing for a comparison of depth distribution, relationship with original and validated data. Bold and italic areas indicate the submodels which are selected for inclusion

Spatial Unit	CUA	Equation	R^2
Blanket Peat CUA	BPFH	d = ((0.875 + [0.00758e - 0.0903s])-25)+(0.5(0.875 + [0.00758e - 0.0903s]))	57.80%
	BPH	d = ((5.99 – [0.00267e - 0.140s])-25)+(0.5(5.99 – [0.00267e - 0.140s]))	56.30%
	BPG	d = ((-11.7 + [2.76e - 0.585s])-25)+(0.5(-11.7 + [2.76e - 0.585s]))	40.60%
Shallow Peat	SPFH	d = 2.02 + (0.00387e - 0.0969s)	47.2%
	SPH	d = 2.43 + (0.00112e - 0.0342s)	6.4%
	SPG	d = 2.73 + (0.00101e - 0.0318s)	5.8%

Table 3.7 Peat depth model equation and fits, where d= depth (cm) e = elevation and s = slope (degrees)

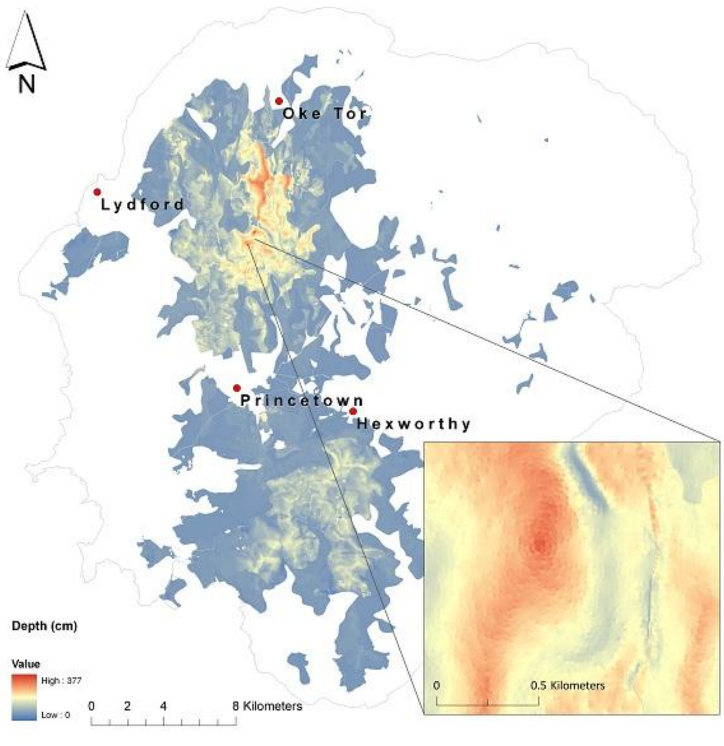

Figure 3.7 Peat depth mapping with inset of detail

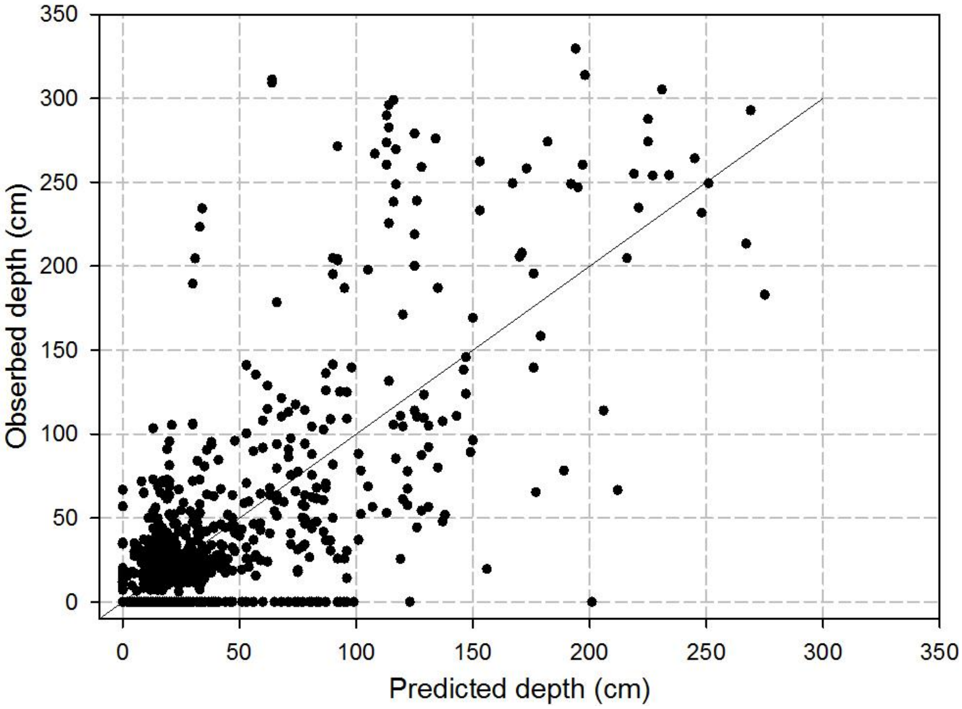

Figure 3.8 Relationships between observed and predicted peat depth in the blanket peat soil unit. Line shows 1:1 relationship.

3.4.1 Modelled depth distribution

The model output was analysed to assess the distribution of estimated depths across Dartmoor's blanket peat; the model calculates that 90% of the peat is less than 160cm deep.

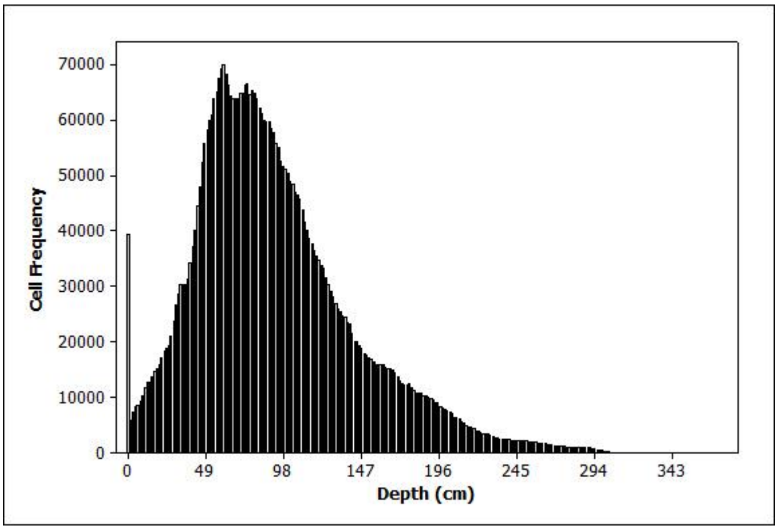

Figure 3.9 Modelled blanket peat depth distribution

3.5 Discussion

3.5.1 Using slope and elevation to predict peat depth

Chapman *et al* (2009) suggest that slope and elevation may be able to predict peat depth. This chapter demonstrates that functional relationships do exist between slope, elevation and peat depth and a model based on these relationships can be developed. Univariate relationships between slope, elevation and peat depth exist individually (Figure 3.4), but when considered together, as a multivariate relationship, they have significantly more explanatory power (Table 3.5). This may be due to slope and elevation reflecting different influences on blanket peat formation. Slope represents the influence of the rate of runoff, whilst elevation represents the gradients produced by increased orographic rainfall and decreased temperature with increased elevation. In blanket peats, slope and elevation have a similar ability to predict depth (Table 3.5) and both variables are therefore important components to blanket peatland development and the model. The strength of slope and elevations relationship with depth is not reflected across all peat types. The relationship is much weaker in soil units with shallower peat, where variability in depth is minor. In future it is worth reducing sampling in these areas and focusing attention on blanket peat areas.

Whilst the relationship between slope, elevation and depth is strongest in true blanket peat, R^2(adj) values of 51.6% indicate that slope and elevation are not the only controlling variables. Furthermore, Figure 3.8 demonstrates increased scatter in the relationship as depth increases, indicating that the ability of slope and elevation to predict peat depth declines as depth increases. The processes of blanket peatland genesis may provide an indication why this may be. Paludification and lateral spreading of blanket peatlands during their development means that there is local variability in timing of blanket peat initiation (Tallis, 1991). Therefore, some areas of blanket peatland have been developing for longer periods of time and will be deeper than equivalent younger sites. Despite this explanation, blanket bog first forms on flatter areas and then spreads upslope by back paludification (Charman, 1995) and slope may be able to indicate age variability to a certain extent. As discussed in Chapter 2, section 2.1.4 blanket peatlands develop as a result of pre-existing peat landforms, the continued presence of these under the blanket peat may be a cause of undetected variability in depth (Charman, 1995). Furthermore, Damman (1979) suggests that as peat gets older the climate and allogenic processes become more important than topographical factors. Therefore older deeper peats, which have been developing for a longer time, have a weaker relationship with slope and elevation. The processes of blanket peat formation form several other plausible controls on peat depth, but other factors may not have been represented within the model.

The greater susceptibility of deeper peat to disturbance of the original surface, by erosion, peat cutting or shrinkage may be a control on depth. This effect was considered in analysis and it was found that when areas which had been visibly subjected to disturbance were excluded from the dataset R^2(adj) values did not improve (Table 3.4). This suggests that disturbance is not an additional control; however, some of the disturbance may not have been detected. Disturbance maybe more of a problem on blanket peatlands which have been subject to more severe erosion, extensive peat cutting and drainage than Dartmoor.

Slope and elevation are not the only topographic variables which may influence, or represent, changes in peat depth. Graniero and Price (1999) found that blanket peat location was determined by a combination of topographic variables, including aspect, upslope contributing area and curvature. It is therefore likely that these are also determinants of blanket peatland depth. Like slope and elevation, additional topographic controls, such as aspect and upslope contributing area may control depth as a result of their effect upon local hydrology and temperature. For example, aspect may determine the amount of precipitation an area of a blanket peatland may receive and contributing upslope area the degree of wetness in a given area. These variables could be incorporated into future sampling strategies to investigate their potential influence. For example, Topographic Index ($\ln(a/\tan\beta)$) as an indication of wetness which uses upslope contributing area, could be used to improve the model. However, topographic index is designed for a smaller scale catchments (Zhou, 2010) and there is concern that it will not work well on a large areas covering multiple catchments. Representative sampling is very important to account for the heterogeneous nature of peat depth distribution. Previous investigations have used secondary data, this often focuses on deep or easily accessible peat and in the past unrepresentative depth maps may have been generated as a result. The sampling strategy of this study designed to be as representative of slope and elevation, to provide a better approximation of depth. The sampling strategy was not designed to be representative of other topographic variables such as aspect. This allowed for sampling sites to be located in areas within walking distance of one another (Figure 3.2), this would not have been possible if other variables were included. However, future investigations could consider other variables as part of their representative sampling to investigate the role of other topographic variables. Finally, stratigraphic studies of blanket peatland show that underlying topography is often more complex than surface topography, giving rise to local variability in depth (Charman 1992; 1995 and Tallis, 1991). As surface topographic values were used the underlying variability would not have been represented in this study and this may be an additional source of error.

Proportionally few of the peat depths estimated by the model can be considered 'deep' as the topographic variability on Dartmoor dictates this. As a result only a small area of the peat can be considered affected by the greater inaccuracy with increasing depth.

3.5.2 Spatial variation in peat depth

The use of slope and elevation to predict peat depth is a valuable tool, but other factors must be taken into account. Blanket peatlands are heterogeneous environments, they form in fused 'complexes' of hydro-topographical units which ultimately cloak the landscape with peat (Lindsay, 1995). The long-term ‘cloaking’ of the landscape gives the relationship between slope, elevation and depth. However subtle traces of previous landforms, or changes in peatland hydrology, may result in slightly different relationships being observed, as discussed above. In the literature, changes in mire morphology such are represented by the landform units developed by Ivanov (1981) particularly as microtopes and mesotopes. These would be ideally suited to representing these changes within the model. However mapping of UK peatlands in this way is not commonly available.

Instead of using microtopes and mesotopes, changes in peat depth were represented by soil unit and vegetation type in the form of CUAs. This methodology improved the models’ ability to predict peat depth, as R^2(adj) values and model accuracy improved when CUAs were considered (Table 3.6). However, the number of vegetation classifications used to form the CUAs was small (Figure 3.1) and not fully representative of habitat types occurring on each soil grouping. Therefore, they may be less effective in representing change in small scale variation in vegetation that could indicate change in peat depth. This data was selected as it was the only vegetation dataset which was fully able to cover the entire moorland line. It was considered a trail for imputing such vegetation datasets, in future greater consideration of how representative each dataset is could be made.

Despite the limitations of the vegetation dataset, the improvement in predictive ability of the model when it is included within CUAs (soil types > CUAs) suggests that the vegetation may reflect some property that also affects peat depth. Although this improvement may simply be a result of the data modelling a smaller spatial area. It is not possible to conclude whether soil series or vegetation are equally valuable controls when representing landform change, as the sampling strategy considered vegetation (CUAs) within soil type (soil units) and therefore their individual relationships cannot be separated. Whilst splitting the model into spatial units improves its predictive ability, it also causes a modelling problem. The use of discrete datasets does not allow reflection of gradual changes in depth that occur between spatial units; this effect is seen in Figure 3.9. Using digital soil mapping techniques, such as fuzzy soil inference schemes, could reduce this problem (Zhu *et al,* 2001).

A further problem with the stratified sampling approach is that samples for each CUA are distributed across a wide range of geographical locations. As a result, any variability related to local factors other than slope and elevation, such as wetness or disturbance, will remain in the model and not be diluted by further local sampling; this may explain the scatter of some points in Figure 3.4. If depth was sampled within single locations for each CUA, the model would undoubtedly have performed better for those locations but would probably have given less robust estimates for other areas of the moor. As a result it is worthwhile maintaining the geographical spread of the sampling strategy. However, if stratified sampling is applied at too great a spatial scale, variables other than slope and elevation will cause an increasing amount of noise in the dataset causing the predictive ability of slope and elevation to reduce (as seen in Table 3.6).

Small scale, local variation in depth is another factor which could influence the model. Undetectable changes in underlying geology, vegetation and hummock hollow surface topography could all influence local depth if not included in the sampling strategy. In blanket peat local variability is small with an average standard deviation of 12cm; this is unlikely to cause a large deviation in the model outcome. However, the extreme local variation did exist (Figure 3.6) and may cause substantial error at a smaller scale if not accounted for with replicates. As a result, it is worthwhile including all five measurements at each point, in order to reduce the occurrence of anomalous results. Repetitive sampling such as this did not remove all anomalous results (see Figure 3.6), but nor did exclusion of these extreme values improve the model, indicating that repetitive local sampling is sufficient in reducing extreme values to a manageable level.

Representing the heterogeneous nature of a peatland, through representative sampling and mapping of major landform units is of considerable importance to the accuracy of this model. Currently the use of soil and vegetation mapping is the only commonly available, reliable and widely available information on landform units for UK peatlands. These maps should be considered the best substitute until further advances mapping Ivanov's landform units are made, potentially through techniques such as remote sensing.

3.5.3 Alternative techniques

The use of statistical relationships between soil and topographic indices is an established technique in mapping mineral soils (Gessler *et al*, 2000). A number of techniques such as cokriging, regression kriging and linear mixed models have been used with a reasonable degree of success (Rawlins *et al*, 2009). In blanket peatlands there have been few attempts to use these techniques. However, alternative techniques which do not require the use of landscape properties to map depth have been used. Ordinary kriging, a univariate technique which statistically interpolates variables by assuming spatial autocorroletion, is the most accessible of these. This technique has been used by Beilman *et al* (2008) and Sheng *et al* (2004) to model peat depth at a regional scale and at a small scale by Frogbrook *et al* (2009).

Using the data available to the study, a comparison between ordinary kriging and the slope and elevation model was carried out in a small area of 1325 ha. The validation depth data had significant spatial autocorroletion (Moran's I 0.36, $P<0.01$) and as a result was a suitable dataset for methodological comparison. A fitted kriged map of depth was created (Figure 3.9) at the same resolution as the slope and elevation model discussed above. The slope and elevation model was extracted from Figure 3.7 for the same 1325 ha area. Both models were subsequently validated using the other model's data points as a comparison. Significant Pearson's correlations of 0.71 for the ordinary kriging and 0.56 for the topographic model were observed.

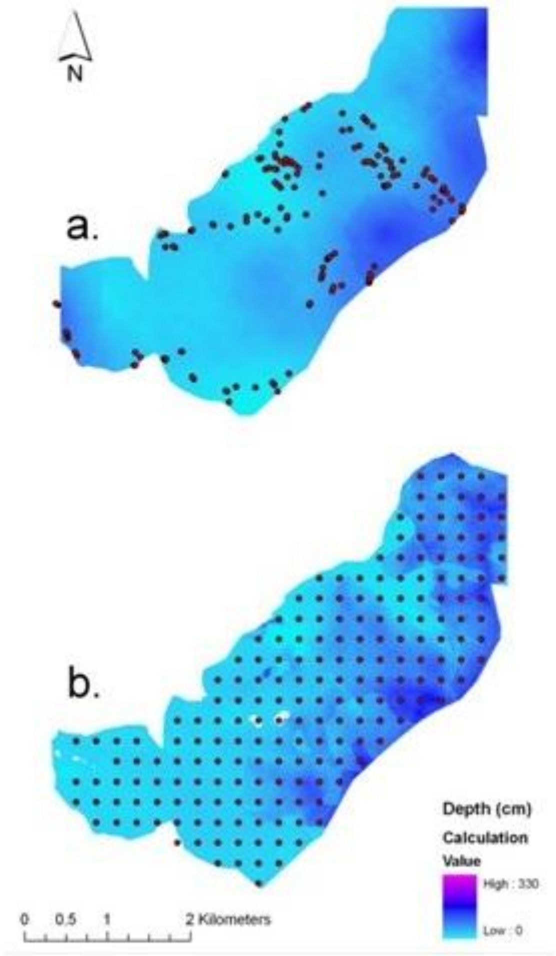

Figure 3.10 A comparison of topographic regression model (map a.) and kriging (map b.) and techniques. Red points identify the sampling points within the national trust estate which created each model

Results indicate that at a small scale kriging was more effective and should be the preferred methodology. Also, kriging is more capable of representing spatial autocorrelation than the slope and elevation model, as all data was locally sourced. However, at a landscape scale the sampling required to achieve significant spatial autocorrelation (semivariogram lag of 437m) would be time consuming and labour intensive. If kriging was used, a reduction in sampling intensity would have to be employed, which would reduce the Moran's I and the accuracy of kriging. Additionally kriging will not allow statistical relationships between peat depth and topographic variables to be revealed, potentially reducing understanding of the blanket peat landform. This analysis demonstrates that when choosing a methodology for modelling peat depth careful consideration of scale and peatland characteristics should first take place, to ensure the most efficient and effective technique is chosen.

3.6 Conclusion

With careful consideration of blanket peatland morphology, it is possible to use the topographic indices, slope and elevation, to develop an effective model of blanket peat depth at a landscape scale. The model could be used for a number of functions, such as landscape scale (>10,000ha) carbon inventories, targeting remediation and monitoring of blanket peatlands, large scale hydrological assessment and identifying target areas for paleo-environmental sampling sites.

The controls used are effective, readily available and the methodology provides a realistic and accessible approach which could be applied to UK blanket peatlands. Results suggest that further developments in mapping of peatland characteristics, understanding of blanket peatland development and accumulation could improve the model further and that there is potential for developing landscape scale mapping of blanket peatland characteristics.

4 Carbon inventory: Methods for modelling soil organic carbon distribution in blanket peatlands at a landscape scale: a case study on Dartmoor, southwest England

4.1 Introduction

Blanket peat and organic 'moorland' soils cover large parts of upland Britain and are a dominant component of the soil organic carbon (SOC) resource in the United Kingdom (Milne and Brown, 1997). To manage soil SOC effectively, a landscape scale understanding of peatland soil organic carbon (SOC) distribution is needed, to provide policy makers and land managers with information on the location and vulnerability of soil carbon within individual land holdings. However, current understanding of the quantity and distribution of SOC throughout British peatlands is limited. Despite the disproportionate ability of blanket peat to store carbon, the UK's national soil carbon inventories (Milne and Brown, 1997 and Bradley *et al*, 2005) do not focus on blanket peatlands and their carbon storage characteristics. This has led to a number of studies questioning the accuracy and applicability of these large generic datasets.

Chapman *et al* (2009) used pre-existing data for Scotland's peatland soils to provide an inventory which was more relevant to peatland environments. Although improving national carbon estimates, Chapman *et al* (2009) used pre-exisiting data which is only applicable to Scotland and was unable to provide the representation of small scale variability of carbon stocks at a landscape scale. Garnett *et al* (2001) and Frogbrook *et al* (2009) have both produced small scale inventories of peatland environments (2200ha and 600ha respectively), but neither of these studies provided methodologies which could be easily be up-scaled to a landscape scale of >10,000 ha. The sampling strategy of Frogbrook *et al* (2009) was unrealistically labour intensive for a landscape scale study and Garnett *et al* (2001) was reliant upon data which are not commonly available across the UK's blanket peatlands, such as detailed vegetative and soil mapping. A more applicable methodology needs to be developed in order to fully understand the quantity and spatial distribution of carbon in the UK's blanket peatlands at a larger landscape scale.

Both Garnett *et al* (2001) and Frogbrook *et al* (2009) provided a comparison of their inventory with the national datasets of Milne and Brown (1997) and Bradley *et al* (2005) respectively. Both found large discrepancies with the national inventory, however due to the small coverage of Garnett *et al* (2001) and Frogbrook *et al* (2009) only very small areas of the national inventory have been validated. Consequently these validations may not provide a representative understanding of the accuracy of the national dataset. A validation of a full blanket peatland is needed in order to corroborate the findings of Frogbrook *et al* (2009) and provide a greater understanding of the true accuracy of Bradley *et al* (2005).

This study develops a methodology for mapping SOC storage within UK blanket peatlands, building on the work of chapter three. The methodology aims to be easily replicable, use commonly available equipment and simple laboratory and field practices. It is hoped that this methodology will enable a standardised resource for land managers, policy makers and scientists at both a national and landscape scale. In addition to providing a landscape scale SOC inventory for Dartmoor, a moorland in the South West of England and a full blanket peatland validation of the national inventory.

4.2 Study site

The carbon inventory is carried out on the same location as the peat depth survey, see chapter three, section 3.2.1 for a fuller site description.

4.3 Model structure

Understanding spatial variation of peat depth, bulk density and carbon content is key to generating an accurate carbon inventory (Bhatti *et al*, 2002). Mapping of peat depth was carried out in chapter 3 and forms stage one of the carbon inventory (see Figure

4.1). Using the work of chapter 3, it was investigated whether statistical relationships could be found between bulk density, carbon and peat depth within each of the peat based soil units of the moorland line, forming stage 2 of the inventory methodology (see Figure 4.1). Peat depth was thought to have an impact on carbon content and bulk density as a result of the increasing proportion of the peat being affected by the mineral layer the shallower the peat. These relationships were then used in *arc*GIS 9.3 for each soil unit to create a map of bulk density and carbon content, which with the peat depth model from Chapter 3 ultimately could be brought together to form the carbon inventory.

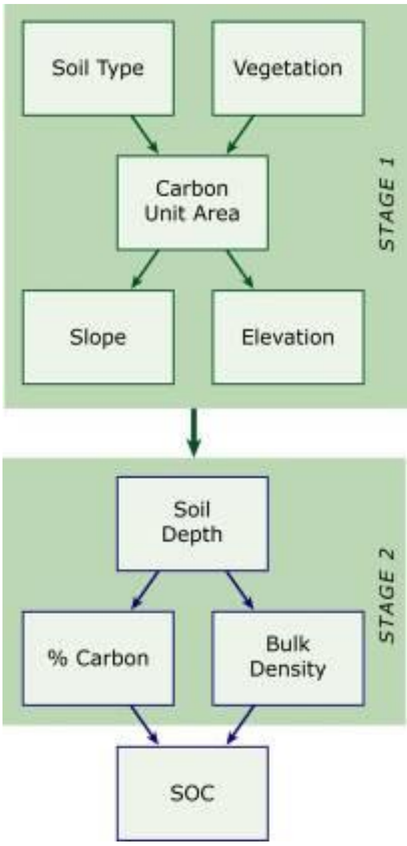

Figure 4.1 Schematic representation of the model structure. Stage one represents processes of chapter three and stage two the processes in chapter 4.

4.4 Sampling strategy and field methodology

Thirty cores were sampled from a range of depths within the blanket peat, shallow peat and peat to loam soil units (Figure 4.2). To ensure a representative sampling strategy the number of cores was area-weighted to represent the distribution of each of the soil series occurring within each soil type, such as Crowdy 2 and Winter Hill in the blanket peat soil unit. Large and small Russian corers were used in blanket peat (as used in Clymo *et al*, 1998 and Buffam *et al*, 2010 for sampling bulk density) and a large gouge auger and pitman tins were used in other soil units. Cores were only accepted if they fully sampled to the mineral layer, as not all cores retrieved the bottom level of peat. Avoiding compaction of cores was a key priority, prior to extraction of cores incisions were made with a serrated knife to ease the corer in with minimum compaction. Before extracting cores the peat depth was recorded, enabling any compaction to be quantified. Due to unavoidable circumstances in the field the bottom 30 cm section of one blanket peat core was lost.

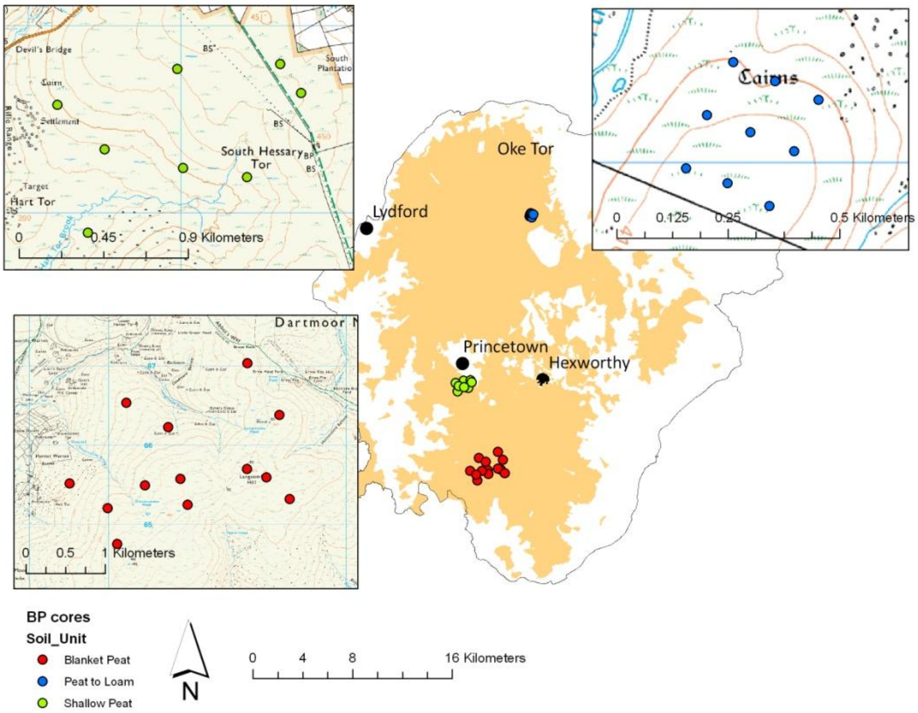

Figure 4.2 Coring sites for each soil unit. Blanket Peat n=13, Shallow Peat n=8 and peat to loam n=9

4.5 Laboratory methodology

Cores were analysed for bulk density and carbon content in the lab. All shallow peat and peat to loam cores and three of the blanket peat cores were sub-sampled at 2cm resolution, to allow analysis of in core variability. The seven other blanket peat cores were sub-sampled at 2 cm resolution for the upper and lower 20% of the core and at 5cm resolution for the remaining 80%, as a result of the low variability observed in the mid sections of the 2cm resolution blanket peat cores. Data was available for an additional three blanket peat cores sampled which had been sampled at 5cm resolution. Each sub-sample was then weighed and dried in an oven at 105^{0}C until it reached a constant weight and was then placed in a desiccator until reweighed to obtain the dry weight. From this bulk density and moisture content were calculated using equations 4.1 and 4.2 respectively.

Equation 4.1 $$\mathrm{BD\ g\ cm^{-3}} = \frac{\mathrm{Dry\ \ weight\ \ (g)}}{\mathrm{Volume\ \ (cm\,3)}}$$

Equation 4.2 $$Moisture\ content\ (ml\ cm^{-3}) = \frac{Wet\ weight\ (g) - dry\ weight\ (g)}{Volume\ \ (cm^{-3})}$$

Organic matter was calculated for each sample by loss on ignition (LOI) at 550^{0}C for 4 hours (Allan, 1989). LOI was subsequently converted to % carbon, using equation 4.3, a regression derived by Bol *et al* (1999) for British peat and organic soils (R^2adj = 98%). Where C is % carbon and L is loss on ignition value (%).

Equation 4.3 $$C = 0.526\,L - 0.167$$

4.6 Results

4.6.1 Core properties

Statistics for bulk densities and carbon contents were calculated for each full core (Table 4.1). Blanket Peat has lower bulk density and higher carbon contents than both the peat to loam and shallow peat soil units. Bulk density and carbon values for the shallow peat and peat to loam soil units were very similar to one another, however differences in bulk density and carbon content between soil units were significant (oneway ANOVA, $F=7.14$, $P = 0.03$) and carbon content (Kruskal-Wallis $H=20.48$, $P = 0.000$). It is expected that this is as a result of the blanket peat soil unit having such different values. Within each soil unit standard deviations in bulk density and were high, indicating large spatial variability of bulk density of variables within soil units.

	Bulk Density (gcm^3)		Carbon Content (%)	
Soil Unit	Mean	Standard deviation	Mean	Standard deviation
Blanket Peat	0.12 ± 0.007 (12)	0.02	50.87 ± 0.116 (12)	1.19
Shallow Peat	0.205 ± 0.01 (8)	0.06	41.50 ± 0.879 (8)	5.94
Peat to Loam	0.191 ± 0.012 (9)	0.07	43.68 ± 0.822 (9)	6.3

Table 4.1 Average bulk density and % carbon for each soil unit from full core values. Values represent the mean, ± SE and bracketed values the number of cores.

4.6.2 Down core variability

The variability of both carbon and bulk density throughout the profile will enable an insight into the patterns of both carbon and bulk density within a mass of blanket peat. It will also provide future awareness of the degree of sub-sampling required in studies measuring bulk density and carbon. Full profiles of both bulk density and carbon are presented for each soil unit in Figure 4.4, Figure 4.5 and Figure 4.6. It is evident that down core variability in bulk density is great throughout most blanket peat core profiles (Figure 4.4), this is supported by a consistently high mean coefficient of variation (COV) of 33% ± 2.8 SE of bulk density throughout each profile. Despite this, the bulk density profiles for each of the blanket peat cores show little consistent trend in their variation from the upper to lower areas of the core. Moreover, in the blanket peat soil unit the COV shows no little relationship with the depth of peat (regression output P=0.194). This demonstrates that accurately predicting the variability in bulk density within a mass of peat may be difficult, and consideration of this heterogeneity must be made to prevent incorrect assumption being made in bulk density studies. There is a similar in core variability of bulk density found within the shallow peat and peat to loam profiles (Figure 4.5 and Figure 4.6 respectively).

Carbon levels throughout the blanket core profiles display more of a trend than bulk density. Stable levels of carbon can be seen until the lower section of peat, where mineral matter begins to dilute the organic content of the peat (Figure 4.4). This is reflected in the low COV levels consistently calculated for each blanket peat core (mean COV of 5.6% ± 1.61 SE). Again the variability in carbon levels throughout each profile does not change with depth (regression output P=0.349). Greater variability in carbon was observed in the shallow peat (Figure 4.5) and peat to loam cores (Figure 4.5 and Figure 4.6), most likely due to their shallower depth and greater contact with the mineral layer.

4.6.3 Compaction

Compaction calculated when sampling each core ranged between 2 and 6 cm, with a mean of 3.4cm ± 0.74cm. Most compaction was in the upper area of the core, due to pressure as the corer was inserted. This is understandable as this is the zone of lower bulk density and fibrous plant material which is less easily penetrated by the corer, although this was minimised by cutting through the surface layers with a knife. However, the overall compaction is small by comparison with overall core depth and was corrected for when bulk density values were calculated.

4.6.4 Core average bulk density and carbon relationship with depth

Average bulk density and carbon content was calculated for each core (Table 4.1), the relationship of these values with peat depth was investigated (Figure 4.3) to determine if a relationship existed which could be used to model both bulk density and carbon. In the blanket peat soil unit it was found that bulk density demonstrates a strong negative relationship with depth (Figure 4.3) with significant regressions (R^2adj 69.9%, $P<0.001$); whilst, carbon content has a weaker positive relationship with depth (Figure 4.3) but still with significant regressions of R^2adj 50.7%, $P<0.001$. The shallow peat soil unit showed no relationship with depth (Figure 4.3) for either carbon or bulk density (R^2adj 16.0%, $P=0.17$ and R^2adj 0.0%, $P=0.97$ respectively). Due to the lack of variability in depth in the peat to loam soil unit (chapter 3, Figure 3.5) no relationship was investigated.

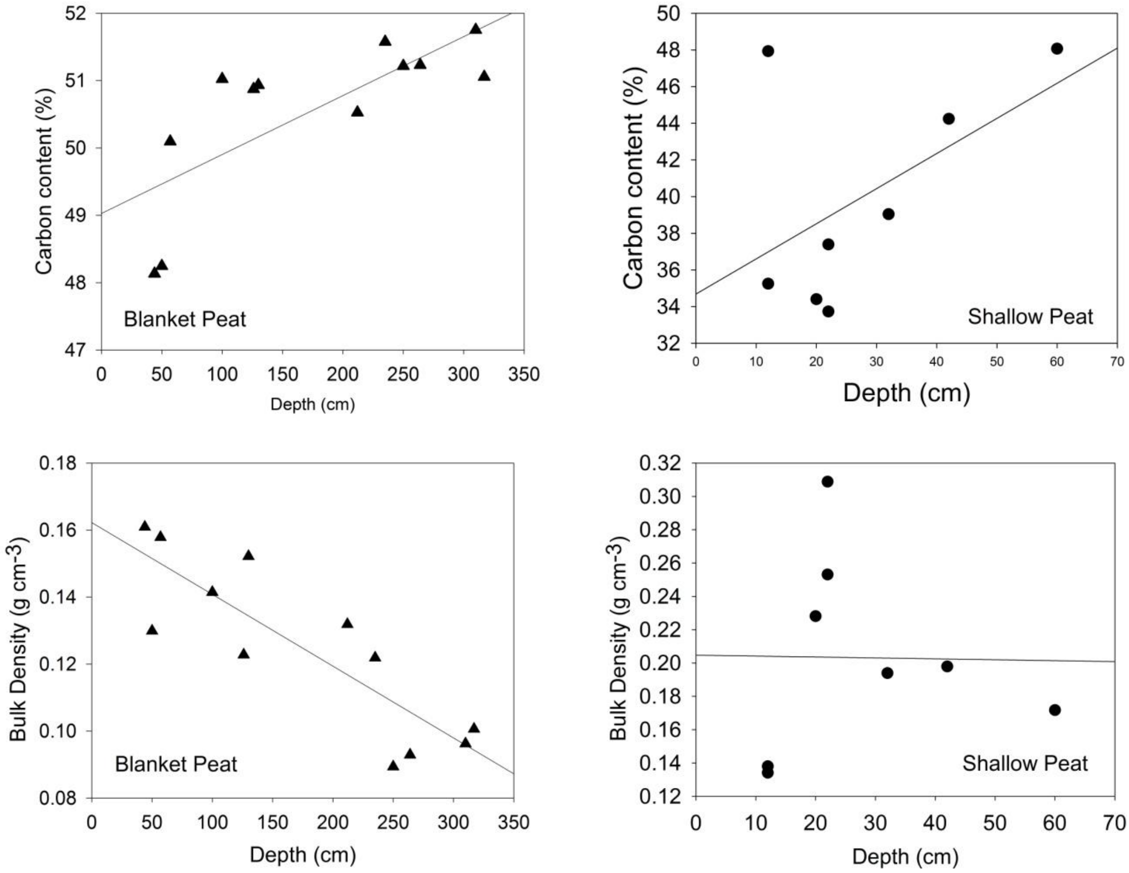

Figure 4.3: Relationships between bulk density and depth and carbon and depth in Blanket Peat and Shallow Peat

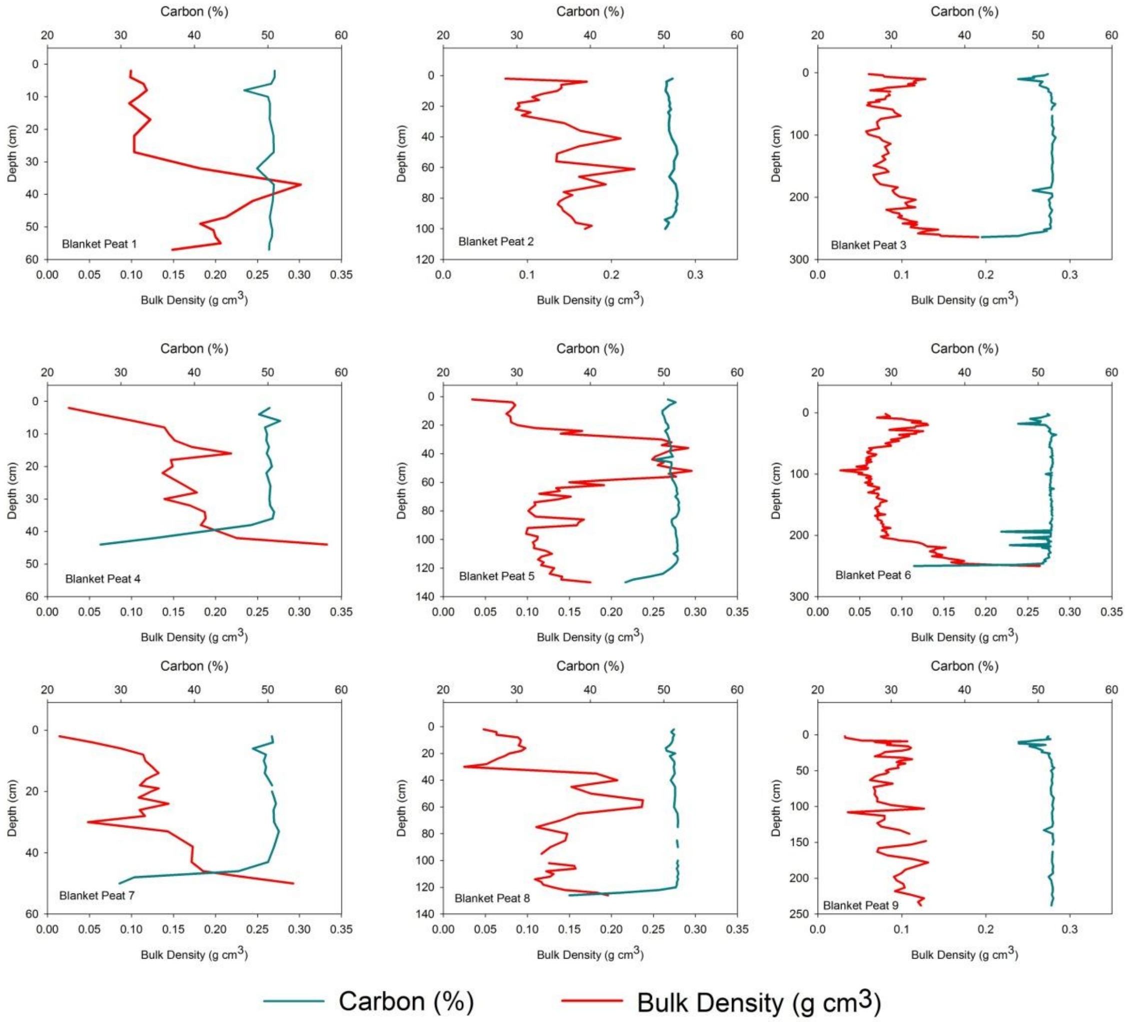

Figure 4.4 Blanket peat and carbon profiles for blanket peat cores. Note differing Y axis values.

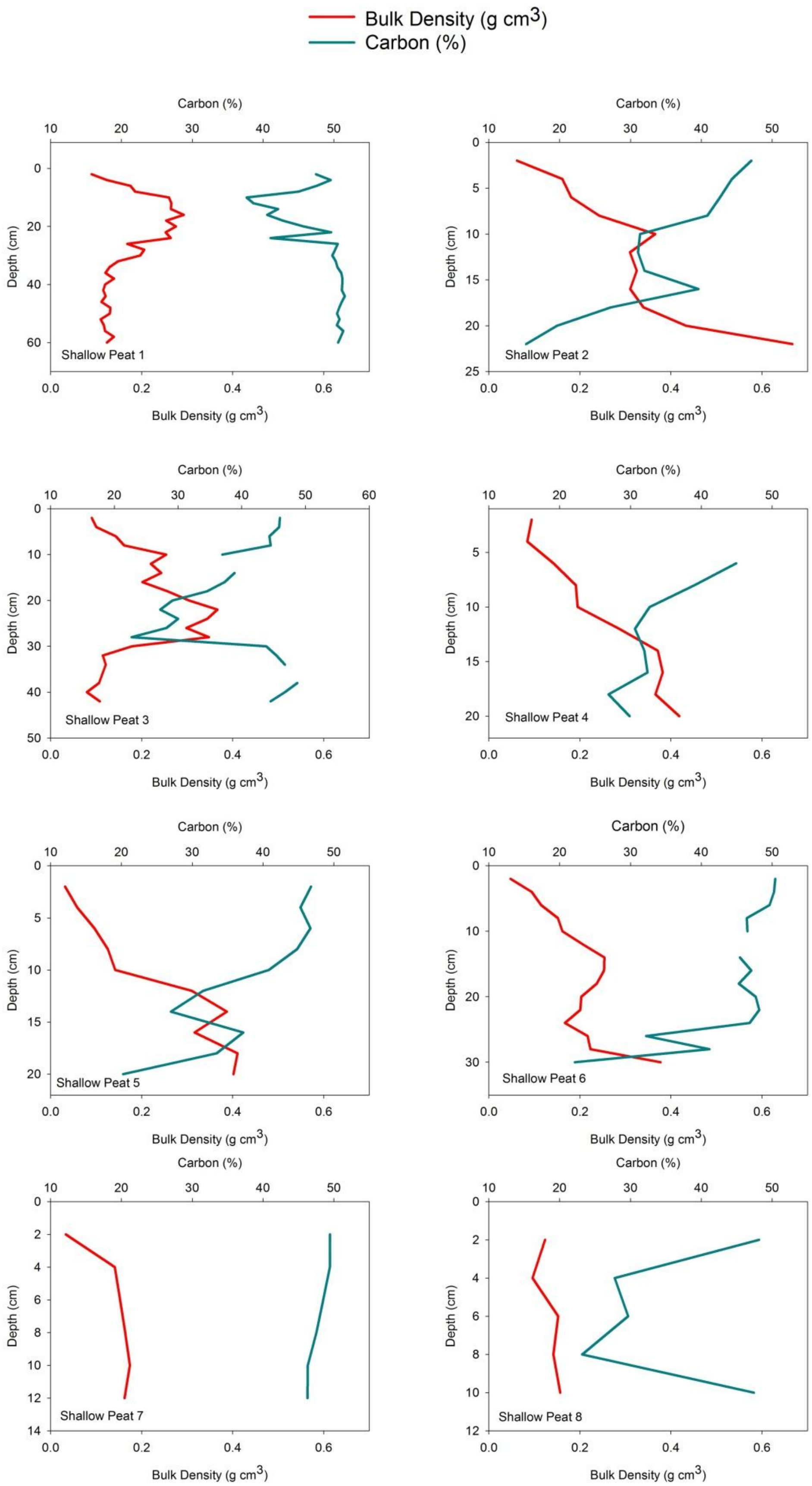

Figure 4.5 Shallow peat carbon and bulk density profiles. Note differing Y axis

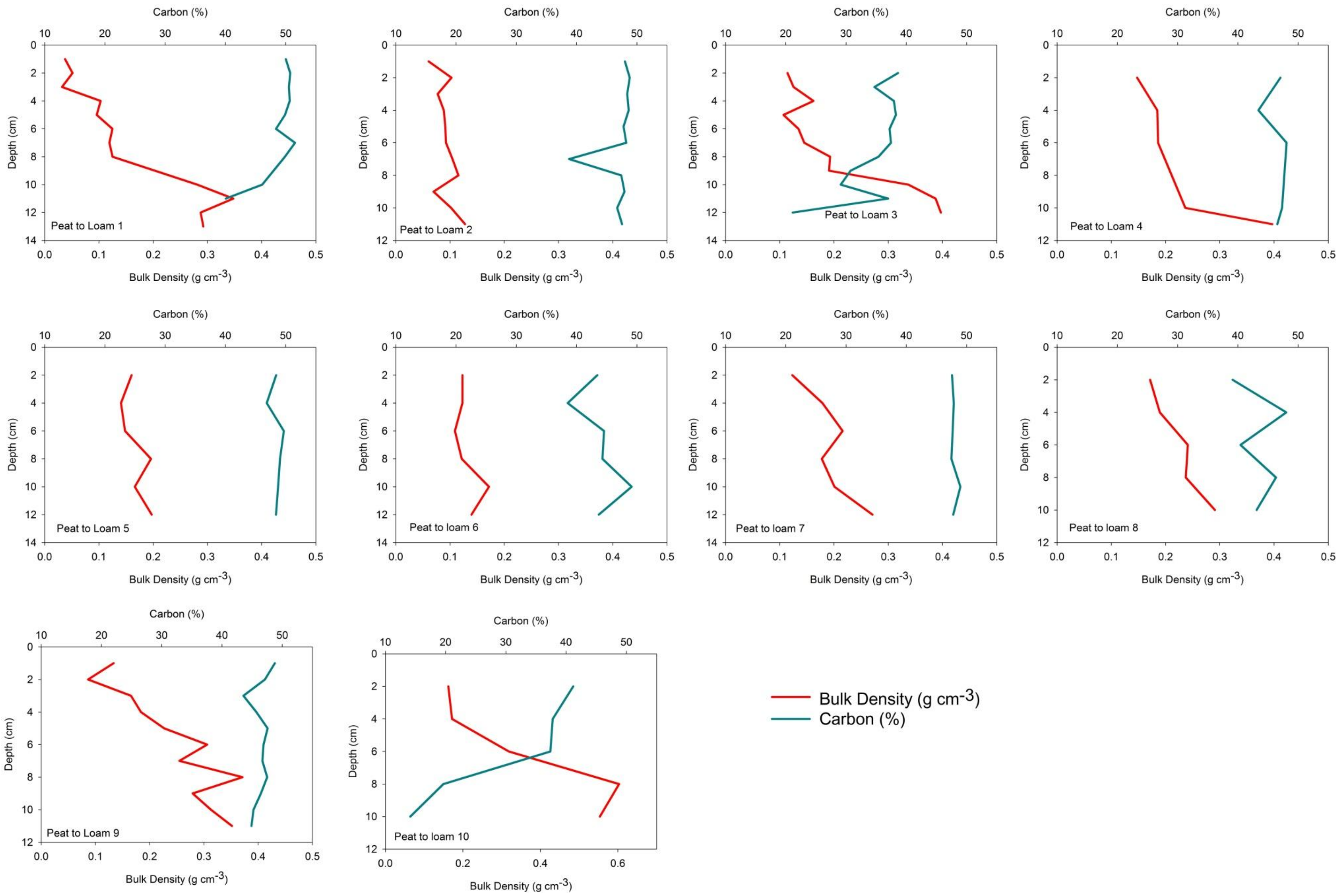

Figure 4.6 Carbon and bulk density profiles for peat to loam cores

4.6.5 Bulk density and moisture content

The relationship between peat bulk density and moisture content was investigated as a parameter for modelling (Figure 4.7). It was found that the relationship between bulk density and moisture content was not significant (R^2adj 5.0%, P = 0.101). Therefore this relationship was disregarded from the model.

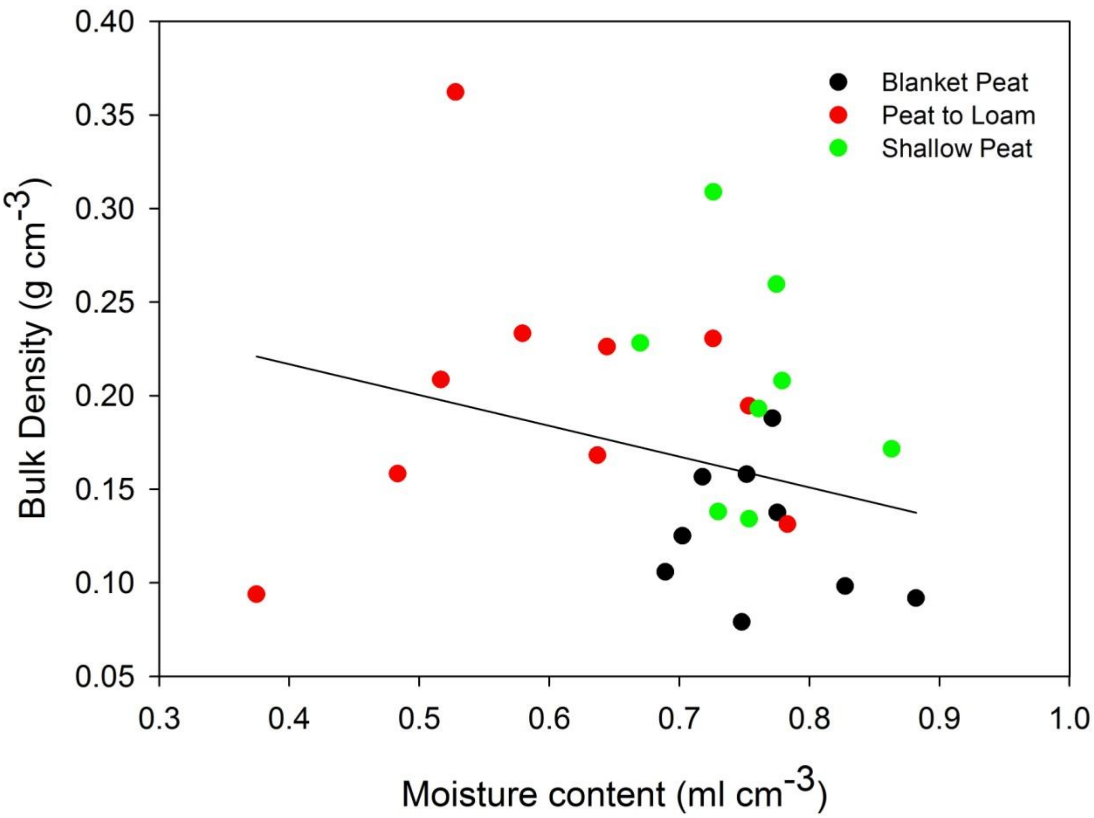

Figure 4.7 Relationship between core total bulk density (g cm^{-3}) and moisture content (ml cm^{-3})

4.7 GIS modelling

Using grid-based map algebra the relationships between bulk density, carbon content and depth were used to create maps of bulk density and carbon content spatial variability within blanket peat areas. Where no relationship was observed between depth and bulk density and carbon (in the shallow peat and peat to loam soil units) the mean value was applied across the whole soil unit. Using map algebra each of the components of the carbon inventory was brought together, to create a map of total carbon storage (equation 4.2). Carbon content maps for each unit were then joined using the mosaic to new raster function in *arc*GIS 9.3 to create a complete carbon content map.

Equation 4.2 $\mathbf{Carbon\ quantity} = ((\mathbf{Depth} * \mathbf{cell\ area}) * \mathbf{BD}) * \mathbf{carbon\ content}$

4.7.1 Modelling results

Total storage within peat based soils of the moorland line is 9.7 (-2.91 + 2.97) Mt carbon. The greatest proportion of which is stored in the blanket peat soil unit (Table 4.2). Blanket peat is also the most effective store of carbon, with more than four times the ability of shallow peats to store carbon per unit area (Table 4.2). The map created of carbon distribution is in Figure 4.8.

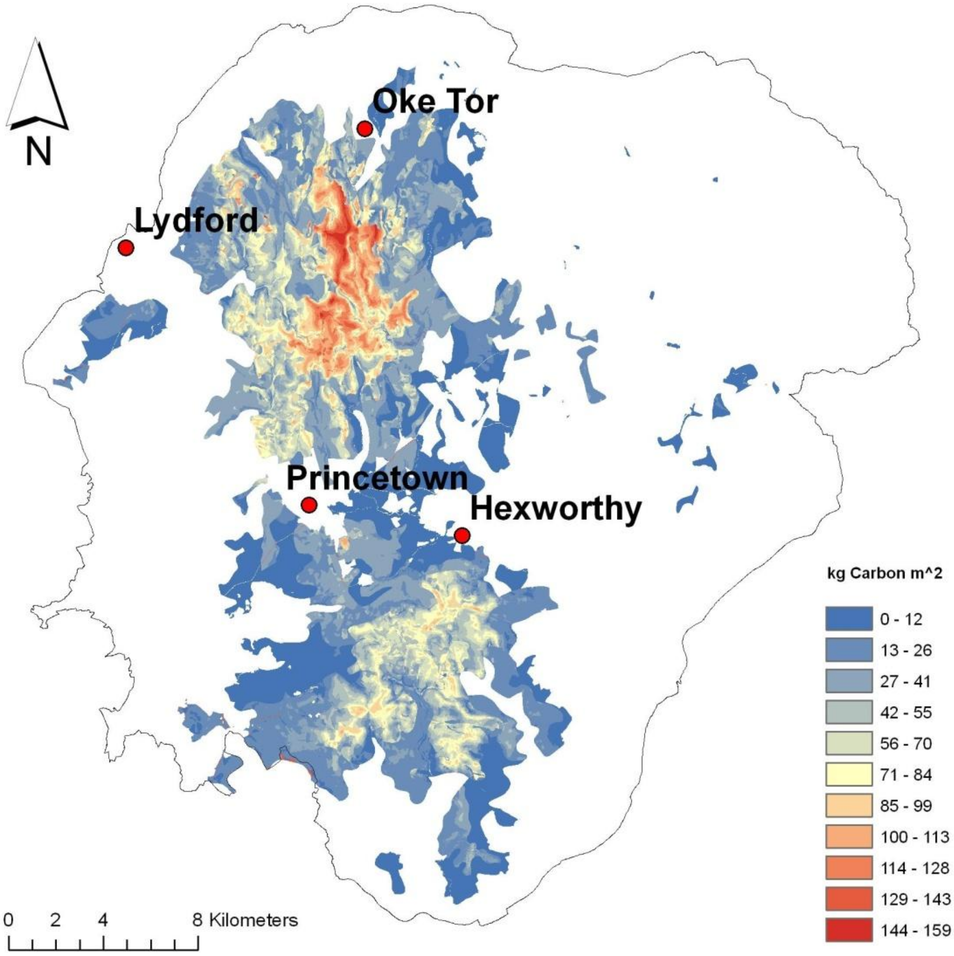

Figure 4.8 Distribution of carbon stocks within the peat soils of Dartmoor National Park

Soil Unit	Total Storage (Mt)	Spatial Distribution	
		Tonnes Per Hectare	Standard Deviation
Blanket Peat	7.22 (-1.51 + 1.71)	629 (-130 + 152)	481
Shallow Peat	1.71 (-0.63 + 0.71)	153 (-56 + 63)	185
Peat to Loam	0.77 (-0.46 + 0.55)	110 (-68 + 70)	34
Total	9.7 (-2.91 + 2.97)		

Table 4.2 Characteristics of carbon storage within soil units of the moorland line. Bracketed values calculated error

Error was calculated by rerunning the model including the RMSE for the upper and lower limits of each regression. Where regression was not used the standard error was incorporated. Dissimilar upper and lower limits are reported in Table 4.2 as RMSE led to negative depths in some areas, as this is not possible these depths were changed to 0cm. Blanket peat error is proportionately less than in the shallower peat units, this is due to regression relationships being stronger in blanket peat areas and no standard errors having to be used. To partition the source of error in blanket peats, the model was calculated using each error source separately in blanket peats (for example a model generated with the upper peat depth error limits but the normal bulk density and carbon equations). Peat depth was the greatest cause of error (7.22Mt +0.85 -0.84), bulk density was responsible for a similar amount of error (7.22 ± 0.62) and carbon was the smallest error term by a substantial margin (7.22 ± 0.14).

4.7.2 How important is each component of the inventory?

To establish the importance of each of the variables contributing to the carbon inventory (bulk density, peat depth and carbon content), several models were generated in the blanket peat soil unit using a combination of variables which were allowed to fluctuate (using their statistical relationships) and other variables were held constant (see Table 4.3). Average values values for peat depth, bulk density and carbon content in blanket peat were used as constants (Table 4.1). The carbon storage calculated by each of these models is in Table 4.3. If the carbon inventory were to be calculated using only constant values (model b in Table 4.3) the value of carbon calculated would be 18.5% lower than that of the model incorporating all statistical relationships which allow fluctuating variables (model a in Table 4.3). When peat depth was the only fluctuating variable and bulk density and carbon were held constant (model c in Table 4.3) the carbon quantified reduced by 9% from the model with all variables fluctuating (model a. in Table 4.3). This level of reduction (not equal to the 18% reduction observed by holding all values constant) indicates that peat depth is not the only important variable for accurately quantifying carbon storage. When bulk density and depth are allowed to vary and carbon is held constant (model e in Table 4.3) the model accounts for nearly all the carbon storage observed where bulk density, carbon and depth are allowed to vary in model a. This shows that bulk density is responsible for the other 9% of the carbon quantified in model a of Table 4.3) and that carbon as a variable has little impact on the quantity of carbon calculated.

Model treatment (which components of the inventory were allowed to fluctuate or were held constant)	Storage (Mt)
a. *Fluctuating:* depth, bulk density and carbon	7.22
b. *Constants:* depth, bulk density and carbon	4.65
c. *Fluctuating*: depth *Constants:* bulk density and carbon	5.29
d. *Fluctuating:* depth and carbon *Constants:* bulk density	5.28
e. *Fluctuating:* depth and bulk density *Constants:* carbon	7.21

Table 4.3 Change in blanket peat carbon storage using different model parameters. Average values used as constant in each model: depth 80.7cm (Chapter 4), bulk density 0.099 g cm^{-3} and 50.87 % carbon

4.7.3 Variables and carbon distribution

Each variable in the carbon inventory (carbon content, bulk density and peat depth) may have a differing influence on how carbon is distributed throughout the peatland. To demonstrate this Figure 4.9 shows the influence of each model in Table 4.3 upon carbon storage per unit area against depth. When bulk density is included as a fluctuating variable the distribution of carbon calculated per unit area changes considerably, with bulk density causing a non linear relationship against depth. However, when carbon is included as a variable, there is little change in the pattern of distribution observed with depth. Carbon content therefore does not influence the distribution of carbon throughout the peatland. Bulk density causes greater carbon storage per hectare in depths less than 270cm, after this the importance of bulk density declines and depth becomes the more dominant variable. However, peat depth is not normally distributed across Dartmoor's peatland (chapter 3); therefore the trend in total carbon storage does not reflect that of tonnes per unit area as seen in Figure 4.10. When cumulative carbon storage is plotted both with and without fluctuating bulk density there is little impact upon the distribution of carbon across Dartmoor's blanket peat as a whole. As peats above 250cm in depth represent little of the total carbon stock, the change in distribution caused by bulk density in Figure 4.9 has little impact.

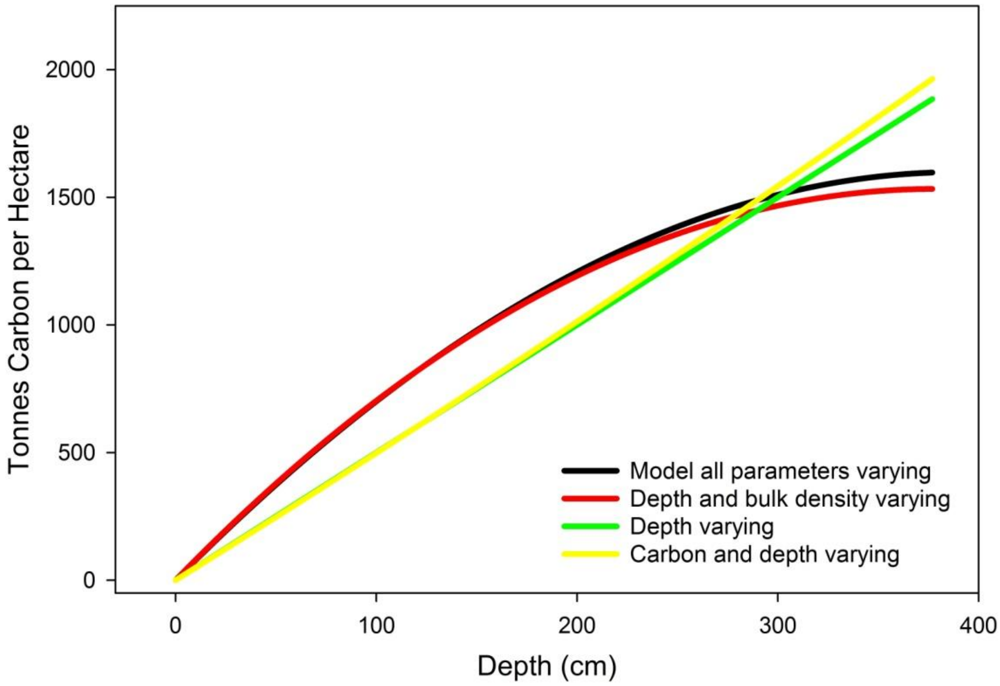

Figure 4.9 Cumulative plot indentifying total carbon stored

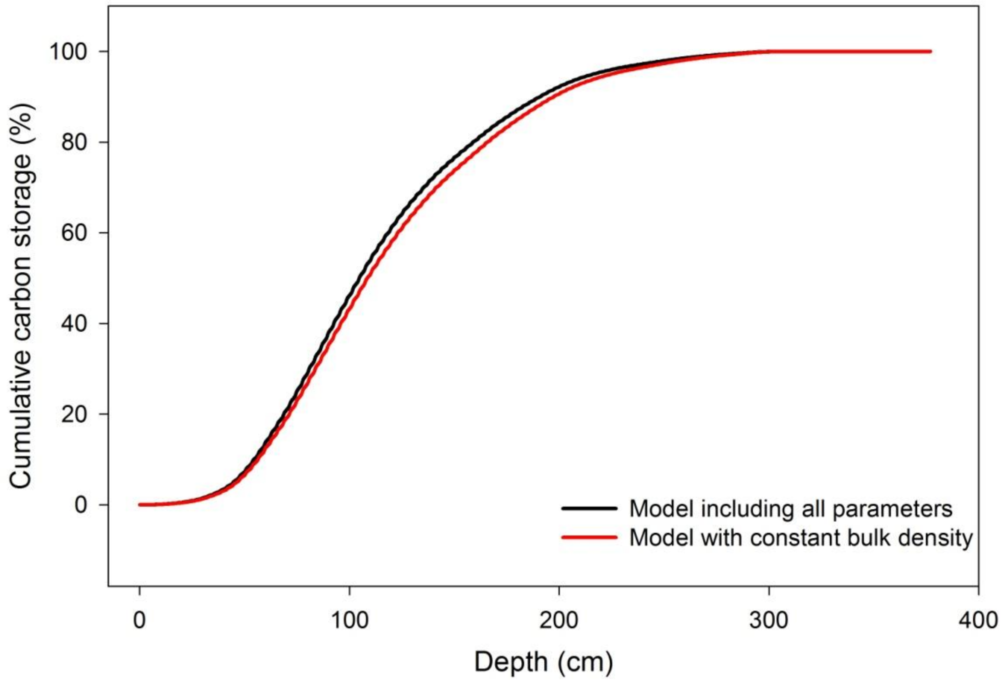

Figure 4.10 Cumulative total storage of carbon throughout depths of the moorland line

4.8 Comparison with the national inventory

The Dartmoor carbon inventory (Figure 4.8) was used to validate the national inventory of Bradley *et al* (2005). The national inventory is split into a grid of 1 km^2 cells and only the national inventory cells which entirely fitted the Dartmoor inventory were considered, a total of 178km^2. The Dartmoor inventory was aggregated into 1km^2 cells and aligned to the national inventory cells. The total quantities of carbon stored within each of these inventories were then compared: the national inventory calculated storage of 9.06Mt compared to 8.36Mt calculated by the Dartmoor inventory; the national inventory calculates 5.83% more carbon.

The inventories were then compared on a cell by cell basis, the cells from the Dartmoor inventory quantified on average 9.48 % more carbon than the national inventory, however the differences observed between cells from each inventory were highly variable (Figure 4.12). The maximum disagreement reached 265% and the minimum 53.8% with a standard deviation of 58.78%. The spatial patterning of these deviations can be seen in Figure 4.11.

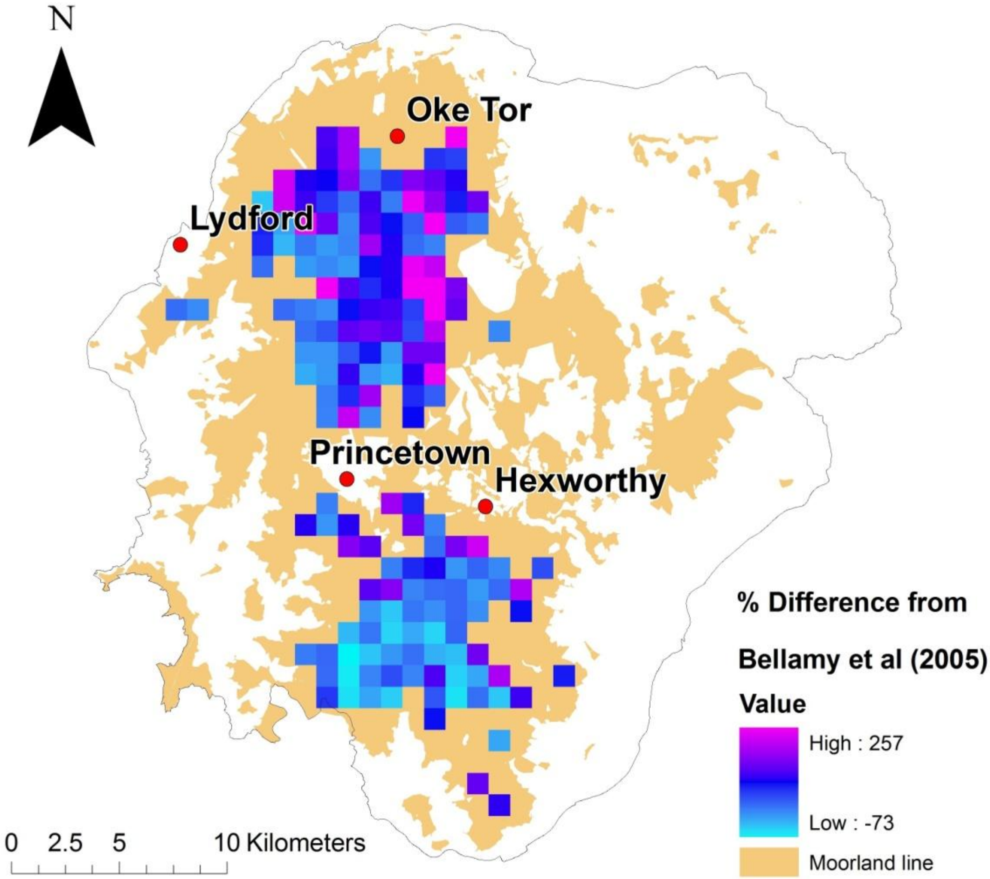

Figure 4.11 Differences found between Bradley *et al* (2005) and the Dartmoor carbon inventory

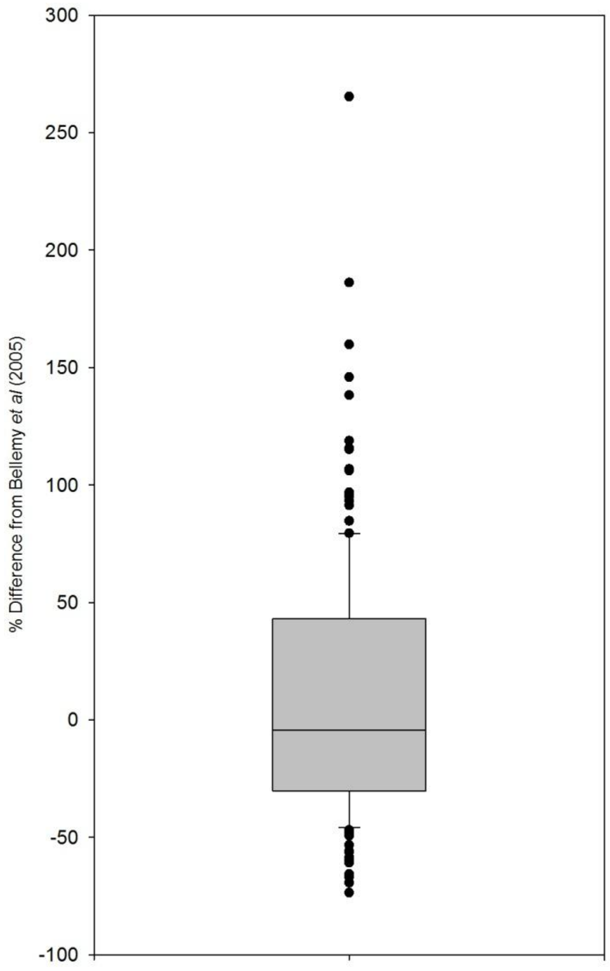

Figure 4.12 Distribution of cell differences (%) of landscape scale carbon inventory from Bradley *et al* (2005)

To determine if agreement between the inventories was influenced by the peat depth and bulk density values used within Bradley *et al* (2005), the peat depth and bulk density maps used within the Dartmoor inventory were also aggregated to 1km^2 cells, reflecting the average value modelled on Dartmoor for each national inventory cell. Additionally the percentage cover of each soil unit within the 1km^2 national inventory cells was determined. Cells were then categorised into pixels with over two thirds coverage of a single soil unit; cells with two soil units with over a third coverage; and, mixed cells with no soil units over a third coverage.

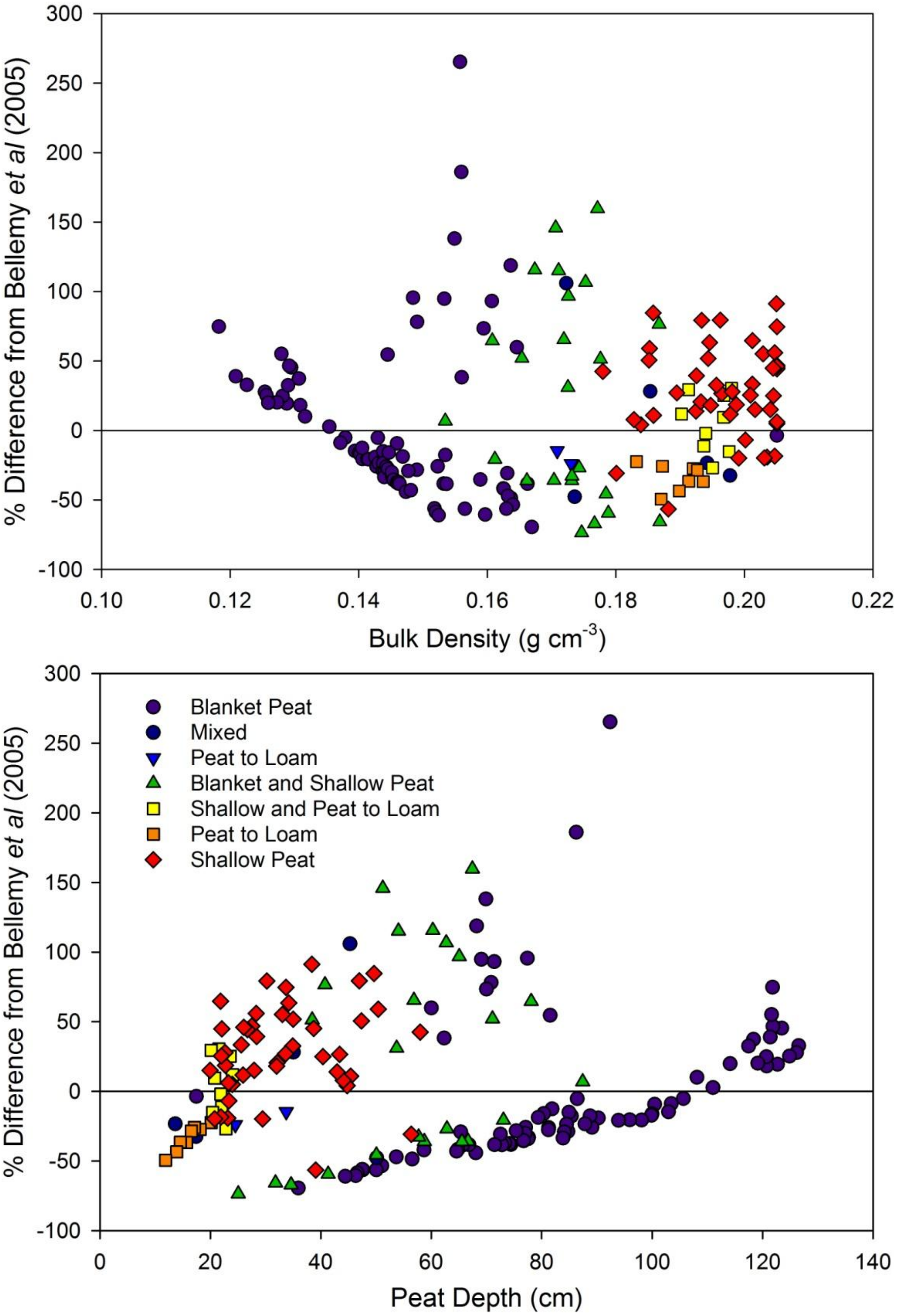

Figure 4.13 Agreement between Bradley *et al* (2005) and the Dartmoor inventory in relation to the average peat depth and bulk density values calculated for the Dartmoor inventory and soil classification

There is some evidence that soil unit plays a role in the extent to which the national and landscape scale inventory agree (Figure 4.13). Some grouping of soil units above and below the 0% difference line can be seen, particularly in the blanket peat and shallow peat and peat to loam dominated cells. This indicates that the bulk density and carbon values used by Bradley *et al* (2005) do not match the ones calculated for these soil units within the Dartmoor inventory for these soil types. Or, that the mix of soil coverage is not fully represented, this is more likely the case in the mixed shallow peat and blanket peat dominated cells. It can also be seen that there is a relationship in agreement between the inventories and the bulk density and peat depth value used, particularly in blanket peat dominated cells. This may be due to bulk density and peat depth variability being great in blanket peat dominated cells, this causes large deviations from the values in used Bradley *et al* (2005) in local situations. In other soil units lower localised variability in peat depth and bulk density (Table 4.1) causes less deviation from the values used in Bradley *et al* (2005), meaning cells from these soil types largely agree.

4.9 Discussion

4.9.1 Characteristics of Carbon Storage and considerations for carbon inventories

Carbon content, bulk density and peat depth are the three primary variables which allow calculation of carbon stocks within peatlands (Bhatti *et al*, 2002). Despite the simplicity of measuring these variables, surprisingly little is known about them. This lack of understanding, and quantification, has lead to error in previous national inventories (as found by Garnett *et al*, 2001 and Frogbrook *et al*, 2009). For example Milne and Brown (1997) used an erroneously high bulk density value for blanket peatlands, as stated in Bradley *et al* (2005), a potential cause of over estimation of the blanket peatland carbon stock within England found by Garnett *et al*, (2001). Problems such as this are encountered, as the data used in large scale inventories is commonly from pre-existing sources, allowing little consistency in methodologies, large gaps in data (Chapman *et al*, 2009) and datasets that are unable to represent carbon and bulk density contents at a localised scale. A clear and consistent understanding of the distribution and quantities of variables influencing carbon storage is needed in order to achieve reliable high spatial resolution estimates of carbon quantities in British blanket peatlands.

If a carbon inventory is to be used as a management tool the representation of spatial variability of carbon distribution is a necessity. Frogbrook *et al* (2009) compared a very high resolution carbon inventory, covering 6 km^2, to the national carbon inventory of Bradley *et al* (2005). It was found that there was poor agreement between the two inventories and the primary cause of this was likely to be a lack of representation of spatial variability at the local scale. The Dartmoor inventory presented in this chapter uses relationships between the variables bulk density, carbon content and peat depth to account for this problem over a much larger spatial area than Frogbrook *et al* (2009). Although the local scale accuracy is unlikely to be as great as Frogbrook *et al* (2009), it provides a means by which local variability can be accounted for. Using this methodology the spatial variability of carbon storage was revealed to be very large for Dartmoor; in some areas the standard deviation of carbon storage per unit area was nearly equivalent to the mean (Table 4.2). It was found that if spatial variability was not represented by these factors the total carbon quantification would be inaccurate, corroborating the finding of Frogbrook *et al* (2009).

As a result of the evident importance of representing carbon distribution in carbon inventories, it is necessary to identify the ways in which the spatial resolution can be further improved using the methodology applied in this chapter. Results indicate that peat depth has the largest influence on carbon distribution in Dartmoor. Improving mapping of peat depth therefore is a crucial component if carbon stocks and distributions are to be better understood.

As noted in Chapter Three blanket peatlands are three dimensional environments and bulk density and carbon may vary throughout the peatland profile. These three dimensional variations can considerably influence a carbon inventory, and as a result changes in the variables bulk density and carbon content throughout a profile are considered within the national soil inventories of Bradley *et al* (2005) and Milne and Brown (1997). However, due to the lack of data available and large spatial area, a number of assumptions are made. In Bradley *et al*, (2005) the profiles of peat soils are divided into generic layers of 0 – 30 cm and 30 – 100cm and it is assumed that bulk density and carbon values used within these classifications do not vary further. However, in the literature a variety of trends of bulk density and carbon content throughout peat profiles have been reported, Frogbrook *et al* (2009) identified a decrease in bulk density with depth and Howard *et al* (1995) identified an increase (which was used in the subsequent national inventories). As a result subdividing peat profiles and applying generic values, assuming a pattern is likely to cause error. As a result of this, it was decided that profiles in the Dartmoor methodology would not be sub-divided and the true average bulk density and carbon content of each core were instead calculated for the entire length of each core. In doing this any trend in bulk density and carbon throughout a number of peat profiles could be presented. It was found that variation of bulk density was high in all cores (COV 30% ± 2.8), also the profiles revealed little regular pattern of bulk density variability (Figure 4.4, Figure 4.5, and Figure 4.6). This has shown that broad subsampling (as in Frogbrook *et al*, 2009) and assuming a trend (as in the national inventories) may be a large source of error. Future inventories should therefore include full representation of bulk density quantities in order to allow for an improved accuracy. Less variation in carbon content was observed throughout profiles. As a result sub-sampling will have introduced little error, unless the change in value at the bottom section in the transition to the underlying mineral ground (as seen in Figure 4.4) was included.

By including each of the variables that influence total carbon storage separately in the inventory, it was possible to understand how each impacts upon the total carbon quantity and distribution throughout Dartmoor's peatland. Results identify show that variability in bulk density and peat depth both account for a similar degree of variation in carbon storage, while variability in carbon accounted for an insignificant amount (Table 4.3). This has important implications for the emphasis of future studies. Both Buffam *et al* (2010) and Natural England (2009b) have correctly highlighted peat depth as being a key part of increasing understanding carbon stocks in peatlands. However, bulk density variability is equally important in terms of the ultimate quantity of Carbon estimated. Frogbrook *et al* (2009) also noted that bulk density variability may have an impact upon carbon calculations. In future, greater emphasis should be given to improving understanding spatial variability of bulk density, in addition to improving the peat depth methodology presented in chapter 4.

Spatial and three dimensional variations are evidently key components of a carbon inventory. Therefore alternative controls on bulk density and carbon variability should be investigated. Clymo *et al* (1998) suggested a relationship between bulk density and moisture content throughout the peat profile. As a result moisture content, represented through topographic index, may be able to explain bulk density variability across a peatland. However, no relationship was found here between total core bulk densities and moisture contents (Figure 4.7) and was not considered a valid variable to include within the inventory and as a result this inclusion may not be as powerful as first thought.

4.9.2 Carbon inventory methodologies

The need for peatland inventories at the landscape scale is increasing, as management decisions at this scale influence the land use and eventually carbon storage and fluxes (Buffam *et al*, 2010). At present for blanket peats, no landscape scale inventories have been carried out depicting spatial variability at a small scale for blanket peats. As a result, effective carbon management is limited. A number of techniques can be used to create peatland carbon inventories and the choice of technique is can be tailored to the characteristics of peatland studied, the scale and the planned use of the inventory. To date, kriging techniques have been the primary approach (Sheng *et al* 2004, Beilman *et al,* 2008 and Frogbrook *et al,* 2009), but pre-existing data has also been used (Chapman *et al*, 2009 and Bradley *et al*, 2005), and constant rates of bulk density and peat depth have been applied across a peatland using primary data (Garnett *et al*, 2001). Few of these inventories are able to predict local variability over large areas (>10,000ha) and their methodologies provide problems at the landscape scale in accuracy due to the availability of data or the demands of sampling at a large scale. Most geostatistical techniques require unmanageable sampling strategies to achieve a high resolution at a landscape scale (e.g. Frogbrook *et al*, 2009 Weishample *et al*, 2009). Other studies rely on datasets that which have been built up over considerable periods of time or are not commonly available (Garnett *et al*, 2001 and Chapman *et al*, 2009). There is a great need for a methodology which provides high resolution data, which represents spatial variability at a landscape scale.

This study takes advantage of one of the unique features of a blanket peatland environment; they are influenced to a great extent by topography and therefore the variables of carbon storage are directly or indirectly related to this. A number of benefits have resulted from using this approach at a landscape scale. The use of map algebra and regression, rather than spatial autocorrelation allows a similar amount of data to be applied over a greater spatial area, resulting in reduced sampling demand. In addition, only simple field, laboratory and GIS techniques are needed and the technique requires commonly available datasets on soils and major vegetation types. This results in a methodology which is replicable and can be readily applied to blanket peats in other areas to provide landscape scale data at a national scale. The resolution of the dataset is determined by the Digital Elevation Model (DEM), not the field sampling strategy, as in geostatistical studies, although error is introduced by non-local sampling. The high resolution coverage of large areas of land allows carbon content and distribution for individual land units to be estimated. This could potentially provide landowners with a mechanism for helping landowners to quantifying the amount of carbon on their land, for example as maybe required by the Uplands Environmental Stewardship Scheme for example (Natural England, 2010).

The methodology is able to calculate sources of statistical error (see section 4.7.2) and from this it is possible to identify areas where the inventory could be improved. In blanket peats, depth and bulk density introduced similar degrees of error into the model and carbon content introduced little. This finding reflects the influence each variable had on the total inventory. However, error in the peat depth mapping is of greatest concern, as this map forms the foundation of the bulk density and carbon maps and any error introduced at this stage will be followed through to the bulk density and carbon mapping. Furthermore, peat depth has the greatest degree of influence upon carbon distribution and as a result it is most important to reduce error at the peat depth stage. Mechanisms by which this can be improved are outlined in Chapter 3.

Reducing the degree of error caused by bulk density is also important to the accuracy of the inventory. A larger sample may reduce the RMSE if the trend identified within the regression observed remains true. However, further sampling may reveal other sources of uncertainty from sampling error. The blanket peat sampling site was an area of relatively pristine peatland and as badly damaged peat tends to have higher bulk densities as a result of humification and shrinkage, error may have been introduced in not fully representing areas such as these. Additionally, all blanket peat cores were sourced from one area of blanket peat, due to time restrictions and despite the cores being widely distributed within this region (Figure 4.2) an element of spatial autocorrelation may have been introduced. In future, sampling should take into account a broader range of peat conditions and have a larger spatial coverage. The relationship found between bulk density and peat depth in this study was linear, in contrast to the non linear relationship found by Frogbrook *et al* (2009). Further sampling should be carried out to check this relationship is not repeated in Dartmoor if further samples are taken. However Frogbrook *et al* (2009) found this relationship when considering both peat and stagnopodzols together and the change to a non linear relationship was only observed in depths <50cm. As peats of less than 50cm account for little of the carbon inventory (see Figure 4.9) any error may have little effect on the estimate of total carbon. Although compaction in cores did occur when sampling, it was to a minimal degree and was unlikely to cause significant degrees of error in the model. Garnett *et al* (2001) also monitored for compaction whilst sampling and found that it also influenced the carbon calculation minimally. In order to resolve this problem an investigation into peatland corer types should be carried out to assess the degree of compaction caused.

The final source of uncertainty lies in the soil mapping used; firstly, error could be introduced on the ground when maps were produced (Chapman *et al*, 2009) and secondly, the mapping scale was 1:250000 which will result in small areas of a soil unit becoming merged with a surrounding soil unit. The use of peat depth within the inventory will resolve this issue to a certain extent, as differences between these soil types are largely defined by the depth of organic matter (Findley *et al*, 1984); any smaller units which have not been picked up by coarse NSRI mapping with shallower depths will be identified and therefore effectively included within the inventory if the depth mapping is accurate enough. The NSRI soil maps were the only soil maps available covering the whole of the moorland line and as a result it was necessary to use them; additionally they provide coverage of the whole of England and Wales and therefore they can be used in other blanket peatlands, therefore fulfilling the criteria for the methodology that it is easily replicable elsewhere.

4.9.3 National inventory

The national soil carbon database of Bradley *et al* (2005) is one of the most comprehensive in the world and has been utilised in much further research, for example in the carbon flux model RothC (Bradley *et al*, 2005). Bradley *et al*, (2005) however remains largely unvalidated, particularly for blanket peatlands. This study has provided a case study against which the national inventory can be tested against an entire blanket peatland. It was found that there was a reasonably good agreement between Dartmoor's peat soils and Bradley *et al* (2005) in terms of total quantity calculated. The previous national soil carbon inventory by Milne and Brown (1997) was tested using the inventory of Garnett *et al* (2001). This study found using a comparison with 22 km^2 cells that Milne and Brown (1997) had a threefold over estimation in the national inventory. The over estimation was largely attributed to incorrect bulk density values and assigning 1km^2 each cells a single soil unit rather than a combination. As blanket peats are three to four times more effective at storing carbon than other peat types (Table 4.2) this was a cause of considerable error in the database. Recognising this problem Bradley *et al* (2005) adapted Milne and Brown's (1997) methodology to include proportional coverage of soil types within each cell. This adaptation appears to have made a significant improvement in the database and may have been a cause of the substantial improvement in agreement when compared with the Dartmoor inventory. Bradley *et al* (2005) did little to the blanket peat bulk density figures and noted they may be a significant source of error. Frogbrook *et al* (2009) validated six peat and organic soil cells from Bradley *et al* (2005) and found large variation in agreement. However, Frogbrook *et al* (2009) noted that the quantity of cells validated was not sufficient to draw conclusions on the overall performance of the database in blanket peats. The Dartmoor carbon inventory provides a comparison with 178 km^2 cells from the national inventory, nearly an entire blanket peatland. Although the methodology does not provide the detailed direct measurements of Frogbrook *et al* (2009), it is able to represent the spatial variability in the variables needed for carbon stock calculation that Frogbrook *et al* (2009) identified as being the largest source of error and uses direct measurement from the peatland studied. As a result this validation can be considered a good indication of the accuracy of the national inventory for blanket peatlands as a whole.

Bradley *et al* (2005) applied a depth limit of 100cm and did not estimate carbon stored in deeper peat, however peatlands vary greatly in depth, and carbon stored in peats over this threshold can be substantial (Frogbrook *et al*, 2009). The spatial variability in peat depth was a factor considered in the Dartmoor inventory and although total peat depth exceeds 100cm in a number of locations, average peat depth when amalgamated into a 1km^2 grid cell, like in Bradley *et al* (2005), only occasionally exceeds 100cm with several of the cells have average peat depths under 100cm. As a result, the peat depth limit of Bradley *et al* (2005) may not have caused a large disagreement between the national and Dartmoor inventories. Bradley *et al* (2005) suggests that the bulk density value which they have used for blanket peat is high (although does not state the value) and the Dartmoor inventory has a larger proportion of 1km^2 cells under 100cm depth than over, as a result it may be that the use of a high bulk density in Bradley *et al* (2005) compensates for the under estimation of average peat depth in the national inventory for Dartmoor. Nevertheless, cells with peat depths over and under 100cm can cause a disagreement between the inventories stocks at a small scale (Figure 4.11). In UK blanket peatlands where average peat depth is not as close to the threshold used by Bradley *et* al (2005) as Dartmoor, larger degrees of error could be observed in the national inventory.

Similar to Frogbrook *et al* (2009) a large variation in cell agreement between the case study and the national inventory cells were was observed, despite the inventories largely agreeing in total quantity. Disagreement in cells was found to be largely related to the spatial variability of peat depth and bulk density (Figure 4.13). As noted in both chapter 3, and this chapter, the variables which make up carbon storage in blanket peats are spatially variable and this can have a large impact upon the carbon stored within a unit area. The national inventory is unable to represent this variability due to its use of non site specific data, which can cause disagreement in individual cells. The national inventory must be treated with caution when information is needed for a specific blanket peat area; for more accurate information at smaller spatial scales, an inventory which uses methodologies such as the one presented here should be used.

4.10 Conclusion

Dartmoor, which makes up a small proportion of the UK's total blanket peat, stores 9.7 (-2.91 + 2.97) Mt of carbon. This is twice the average annual CO_2 emissions from the UK's agricultural sector (Mackintosh, 2010). Blanket peats have also been found to be highly effective stores of carbon per unit area in Dartmoor, a trend which is likely to be reflected in other blanket peat areas. In addition to this, blanket peats are vulnerable to carbon loss (see chapter 2). As a result, it may be most effective in terms of soil carbon retention for agricultural policy and funding to focus on blanket peat areas.

A large scale coarse-resolution national scale inventory has been shown to be largely accurate in estimating the total amount of carbon stored in Dartmoor's peatland, but inaccurate at in accurately predicting the distribution of carbon across Dartmoor. It is essential to effectively understand the distribution of carbon at the scale where it will be managed (Buffam *et al*, 2010), particularly as carbon distribution varies so considerably in blanket peats. The methodology presented here and applied to a case study on Dartmoor is suitable for providing datasets at a scale useful for land management. Improved mapping of peat depth and bulk density is particularly important but the approach used here can provide a very effective methodology for broadening our understanding of landscape scale carbon storage and distribution within the UK's blanket peats.

5 Comparative dating of recent peat deposits: the fallout radionuclide and Spheroidal Carbonaceous Particle techniques at a local and landscape scale

5.1 Introduction

Biological and geochemical proxies stored in peat deposits can provide valuable information about recent environmental changes, including variability of the peatland environment (e.g. Hendon and Charman, 2004), the broader landscape (e.g. Chambers *et al*., 1999) and regional changes in climate and pollution loads (e.g. Charman, 2007, Shotyk *et al*., 1998). Adequate chronological control is a critical part of these studies, especially where rates of change are important, for example in assessing changes in environmental pollutant deposition and rates of carbon accumulation (e.g. Garnett *et al*., 2000). The accuracy of dating these deposits is of crucial importance to the validity of these studies and may have a significant impact upon the detection of changes with implications for environmental management. The absolute magnitude of acceptable errors in dating is small, as the relative errors are larger for recent peats than for older deposits, so that they generally require high chronological resolution and accuracy. Standard dating techniques, such as calibrated ^{14}C ages are not generally applicable in recent peats (Belyea and Warner, 1994) and as a result many alternative dating techniques have been developed. No single dating technique is able to provide a high level of certainty and as a result a multi-technique approach is recommended to provide robust dating outputs (Oldfield *et al*, 1995; Turetsky *et al*, 2004).

Radionuclide dating and Spheroidal Carbonaceous Particles (SCPs) are dating methodologies which are commonly applied in these circumstances (e.g. Yang *et al,* 2001; Garnett *et al*, 2000; Wieder *et al*, 1994). 'Bomb-spike' radiocarbon ages are also increasingly used to date recent peats (e.g. van der Linden *et al*., 2008; Piotrowska *et al*., 2010), although this is an expensive process compared to radionuclide and SCP analyses. SCPs are pollutants produced from the combustion of fossil fuels (Rose, 2001). SCPs have been emitted into the atmosphere and deposited on peatland surfaces since the industrial revolution and the trends in this deposition can be used as date markers. Fallout radionuclides consist of the naturally produced ^{210}Pb, from which sediments can be dated using rate of decay (22.26 years) and the artificially produced ^{137}Cs, ^{241}Am and ^{207}Bi from nuclear weapons testing, which leave datable peaks in sediments (Appleby, 2001). Each of these dating techniques has characteristics which complement each another and when used together provide valuable chronologies which can be applied in a number of circumstances. SCPs are very cheap to analyse and can provide several relative dating features, artificial radionuclides provide definite dating peaks for specific events from weapons testing, and only ^{210}Pb is able to produce a continuous chronology for perhaps 100-150 years. However, each technique also has a number of uncertainties associated with it. Radionuclides such as ^{137}Cs and ^{210}Pb are potentially mobile in peats (Urban *et al,* 1990 and Oldfield *et al*, 1979) and a continuous ^{210}Pb chronology relies on assumptions of fallout rates and sediment accumulation. SCP dates rely on calibration from sediments dated with ^{210}Pb and industrial pollution data, and specific changes are not well calibrated for all regions (Rose *et al*., 2005). As a result using these techniques does not always prove successful and results from peat cores are variable (Oldfield *et al*., 1995).

In this study multiple cores from the same study area have been dated, using SCPs and radionuclides. By using cores from the same study area and peatland type, fallout histories for SCPs and radionuclides can be assumed to be the same for all cores. The differing local environmental conditions of the sampling sites is the only systematic difference between the cores, in this case related to management history. The results will provide an indication of the performance and consistency of each technique and the overall accuracy of the multiple dating approach. The aim of this study is to improve understanding of the validity of the techniques commonly employed for dating recent peats.

5.2 Study site

Cores were taken from three sites in the northern area of blanket peatland in Dartmoor National Park (see chapter 2, section 2.3.1). These sites were selected as part of a corresponding study investigating the impacts of management and peat condition upon carbon accumulation rate. Each site was similar in with elevations between 496 and 577, slopes of less than 4 degrees and aspects in the range of 244 and 287 degrees and were all located on the Crowdy 2 NSRI soil series. Each site however, was subject to either a drained, degraded or control conditions. The degraded site is in poor condition and is desiccated and hagged with largely vascular vegetation; the drained site is in better condition, but is dominated by vascular vegetation; and, the control site is in good condition with high species diversity. Greater detail regarding the characteristics and condition of each site is discussed in the subsequent chapter 6.

5.3 Methodology

5.3.1 Field methodologies

Two fifteen meter long transects were located at each site. Along the first transect five 30cm monolith cores were extracted. Three of these from each site were subject to full SCP and radionuclide analysis and form the basis for the comparison between dating techniques. Two other cores were also sampled but were not subject to radionuclide dating. Results of SCP analysis on these additional cores are presented here for completeness. Dip-wells were placed at points along both transects in monolith locations and in parallel positions along the second transect.

Monolith cores were extracted with great care to so that bulk density was not disturbed. A pit approximately 40cm deep was dug, ensuring the sampling area was not trampled. The sampling face of the pit was cut back using a knife and the monolith tin was placed against the peat face where pilot incisions using the knife were made into the peat. The monolith tin was gently pushed into the peat face and carefully cut out. The dip-wells were subsequently visited once a month to record the water table level.

5.3.2 Laboratory methodologies

Three cores from each site were dated using radionuclides. Each core was cut into 1cm sections with care being taken to ensure that the surface area of each sample was calculated without compression. Sections were freeze-dried until a constant pressure was reached. Dried samples were packed into 50ml petri dishes, weighed and sealed. Each section was counted for ^{210}Pb, ^{137}Cs, ^{241}Am, ^{207}Bi and ^{226}Ra by gamma assay for approximately 24 hours, at least three weeks from being sealed (allowing equilibration of $^{222}Rn/^{226}Ra$), at the Consolidated Radio-isotope Facility (CORiF), Plymouth University using a planar detector under ISO9001 standards. ^{210}Pb was detected for γ rays at 46.5 keV, ^{137}Cs at 662 keV, ^{241}Am at 59.2 keV, ^{207}Bi at 1063.6 keV and ^{226}Ra at 295 keV and 352 keV (via daughter isotope ^{214}Pb). Detector efficiency was calculated by creating a standard of known ^{210}Pb, ^{137}Cs, ^{241}Am and ^{226}Ra value.

5.3.3 SCPs and charcoal

Fifteen cores were dated using SCPs following the methodology of Rose (1994). Counts were also made for charcoal on the SCP slides. Each sample was weighed and then digested using HNO_3 for 1 hour 30 minutes, centrifuged at 3000rpm for 3 minutes, rinsed with deionised water and centrifuged again. As an adaptation to Rose (1994), for each sample a *Lycopodium* tablet was dissolved in 0.5% HCl and added as a known concentration marker after the samples had been thoroughly rinsed. The samples were then transferred to vials, centrifuged and glycerol was added. Samples were mounted on a slide and SCP and charcoal concentrations were counted under a light microscope against a frequency of 50, 75 or 100 *Lycopodium* following the protocols of

Rose (2008). SCP and charcoal frequency were calculated as numbers per gram dry mass.

5.4 Results

5.4.1 SCP dating

SCPs were present within each core, with the exception of considerably low counts in core 6 and 10 (control cores). A further two cores were sampled to replace these. Dating features outlined in Rose *et al* (1995) and Rose and Appleby (2005) were identified. The take off, rapid increase and peak features were present in each core, with the exception of a peak not being present in degraded core 15 where a peak was not present (Figure 5.1). The take off and rapid increase features are largely consistent throughout the UK and have been identified as 1860 ± 25 and 1950– 1960 ± 15 dates respectively (Rose *et al,* 1995; Rose and Appleby, 2005). The peak date varies though out the UK and is identified as 1970 ± 5 in south and central England (Rose and Appleby, 2005). Few reliably ^{210}Pb dated SCP records are available for the south west of England in the CARBYDAT database (http://www.ecrc.ucl.ac.uk/index.php/content/view/299/112/), although these provide greater detail to Rose and Appleby (2005).

A major assumption of SCP dating is that accumulation and bulk densities are constant (Garnett *et al*, 2000). If accumulation rates change rapidly, false dating features could be formed (such as a false peak due to increased bulk density). This is a particularly relevant consideration in peats, as bulk densities can be highly variable (Carbon inventory 4) and accumulation rates may change abruptly when subjected to management changes such as drainage. However, the magnitude of change in SCP concentration is normally sufficiently large for temporal patterns in SCP deposition rates to be reliably identified.

Assuming that all dating features are as a result of true trends in SCP deposition, error in these dates was calculated. This error consists of two sources; a) sampling error, cores were sampled at a 1cm resolution and the dates estimated for a given sample could be at any point in the range of years included represented in the sample slice, and b) error in the ages for dating features calculated by Rose *et al* (1995) and Rose and Appleby (2005) for the South and Central England region (Table 5.1). These errors were derived by Rose *et al* (1995) and Rose and Appleby (2005) from absolute error recorded from the corresponding ^{210}Pb dates from each of the mineral cores found within the CARBYDAT database.

Peak concentrations varied widely with an average of 43,130 SCPs gDM^{-1} and ranged between 9692 – 73,825 SCPs gDM^{-1}. It is not possible to compare concentrations to lake profiles in the south west region as differing accumulation rates, SCP catchments and sampling resolutions will affect the total concentration. SCPs are largely deposited through precipitation (Rose, 2001), and as Northern Dartmoor receives the highest levels of precipitation in the South West (Met Office, 2010a) it is unsurprising that SCP concentrations are relatively high in comparison to the lowland lake sites of the South West on the CARBYDAT database.

Core	Peak Depth (cm)	1970 Peak Error (years)	1955 Take off depth (cm)	Take off error (years)	1860 Start Depth (cm)	Start error (years)
Drained 1	7	± 9	11	± 28	16	± 44
Drained 2	10	± 12.5	12	± 27	20	± 37
Drained 3	7	± 11	9.5	± 23	20	± 33
Drained 4	6	± 7.5	12	± 24	24	± 33
Drained 5	6	± 9	10	± 23	20	± 31.5
Control 7	12	± 9	16	± 29	20	± 49
Control 8	16	± 12.5	18	± 28	20	± 72.5
Control 9	15	± 20	16	± 28	25	± 44
Control 6(2)	6	± 8	11	± 24	18	± 39
Control 10(2)	6	± 10	10	± 26.5	16	± 49
Degraded 11	2	± 10	5	± 33	8	± 52
Degraded 12	2	± 10	5	± 30	10	± 44
Degraded 13	4	± 12.5	6	± 45	8	± 72
Degraded 14	2	± 11	4.5	± 26	12	± 38
Degrade 15	None unidentifiable				6	± 44

Table 5.1 Depth of datable features with their associated error

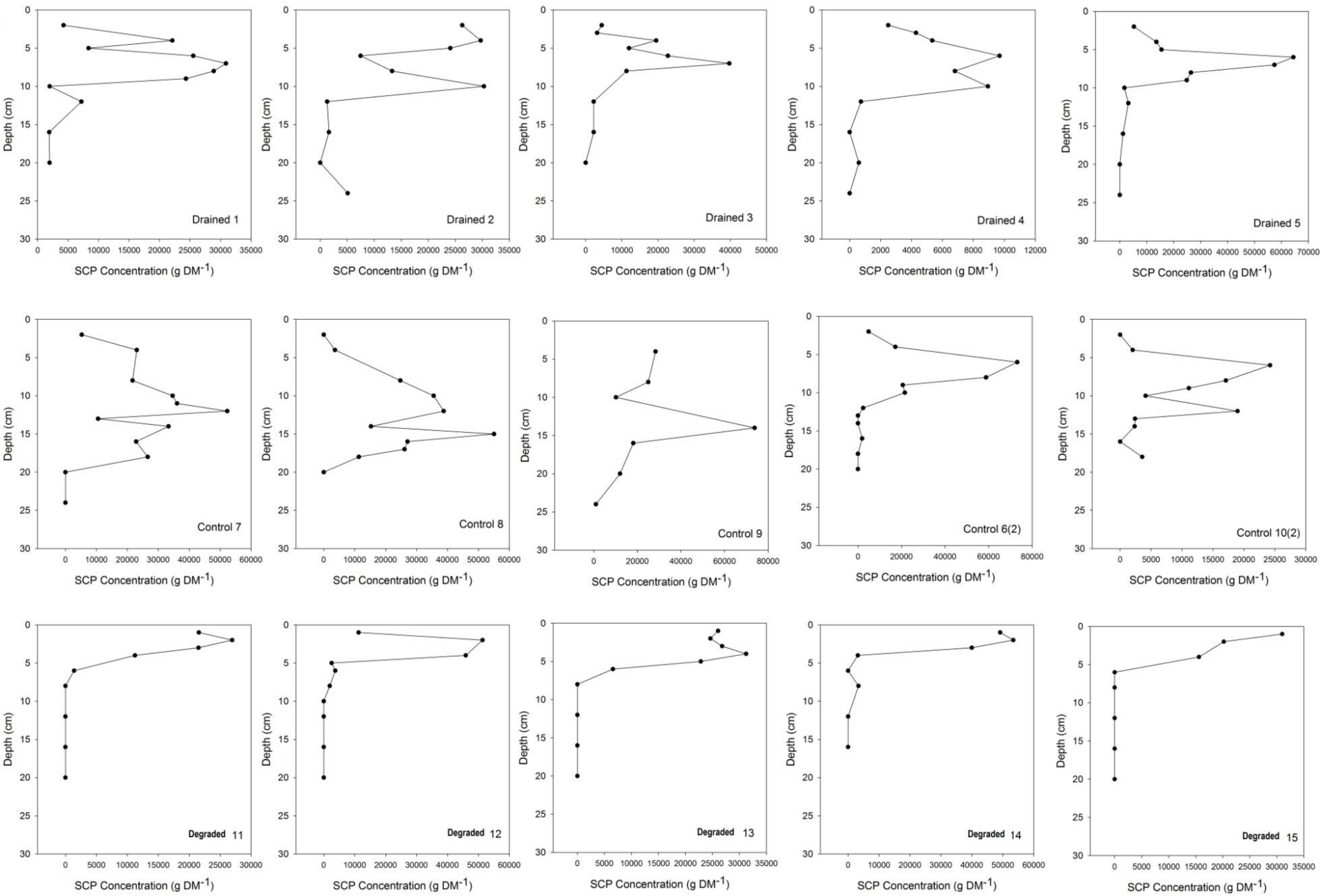

Figure 5.1 SCP concentration profiles plotted against core depth

5.4.2 Radionuclide inventories

^{210}Pb

Total inventories of ^{210}Pb in all cores were high, most of which was unsupported (atmospheric) ^{210}Pb with considerably low levels of supported ^{210}Pb decay from the decay of ^{226}Ra *in situ*. The cores are from an ombrotrophic rainfed peatland, so this result is unsurprising. Fallout inventories of each core for ^{210}Pb are given in Table 5.2. ^{210}Pb fallout across the UK has been found to be relatively consistent (Smith *et al*, 1997) and the inventories are similar to those presented in Smith *et al* (1997) and MacKenzie *et al* (1997) from both peat and mineral cores. Annual fallout was calculated (Table 5.2) using equation 5.1:

$$F = \lambda A \qquad \text{Equation 5.1}$$

Where λ is the ^{210}Pb decay constant 0.03114 year^{-1}, A is the total inventory of the core and F is the annual flux. The mean annual ^{210}Pb in the UK is 77 ± 14 Bq m^{-2} year^{-1} per 1000mm of rainfall (Smith *et al*, 1997). Annually Princetown, in the centre of Dartmoor, received an annual average of 1974.2 mm of metres of rainfall between 1971 and 2000 (Met Office, 2010b) and thus an estimated 152 ± 28 Bq m^{-2} year^{-1} of ^{210}Pb . The annual inventory values for all cores are within these limits (Table 5.2).

All sites are located in areas with similar elevation and aspect and therefore it can be assumed that they receive similar levels of rainfall and consequently comparable levels of annual ^{210}Pb fallout. No statistical differences were observed between annual fluxes in each of the management types (one way ANOVA, P = 0.251). Annual fluxes for cores within each management sampling site were also within the error of one another, indicating that there was no localised leaching of atmospheric ^{210}Pb input. Core control 6(2) had the highest estimated annual flux (197 ± 33 Bq m^{-2} yr^{-1}); although this was still within error of control 8 and several other cores in other management types.

Inventory profiles are able to reveal information about the validity of ^{210}Pb dating. An exponential decrease with depth would be expected, given constant accumulation and steady deposition. This sort of profile is seen in each of the degraded sites, however non-monotonic features are seen in control and drained cores (Figure 5.2). A common feature present in these cores is a dip in ^{210}Pb activity in the upper level of the core (Figure 5.2).

Core	**Total ^{210}Pb fallout (Bq m^{-2})**	**Annual ^{210}Pb fallout (Bq m^{-2} yr^{-1})**	**Total ^{137}Cs fallout (Bq m^{-2})**
Degraded 11	4242 ± 664	132 ± 21	1396 ± 121
Degraded 13	4387 ± 546	137 ± 17	1335 ± 99
Degraded 14	3488 ± 555	109 ± 17	840 ± 102
Degraded 5	5399 ± 671	168 ± 21	795 ± 147
Degraded 3	4458 ± 660	139 ± 20	698 ± 117
Drained 1	5552 ± 757	173 ± 24	807 ± 93
Control 7	4103 ± 674	128 ± 21	1066 ± 131
Control 6(2)	6326 ± 1071	197 ± 33	1459 ± 198
Control 8	4714 ± 603	147 ± 19	739 ± 100

Table 5.2 Core total inventories and annual fallout

^{137}Cs

The artificial fallout radionuclide ^{137}Cs was present in all cores and concentrations were high, but clear peaks were not present in many of the profiles (Figure 5.2). Typically ^{241}Am maximum profiles correspond well with the first ^{137}Cs peak from weapons testing (Appleby *et al*, 1991), but there are no matches between depths of ^{137}Cs peaks and ^{241}Am profiles here (see Figure 5.3) despite remnants of ^{137}Cs peaks being present in 5 and 11. However, no significant difference was observed for total ^{137}Cs loads (and therefore retention) between the management types (one way ANOVA, P= 0.224).

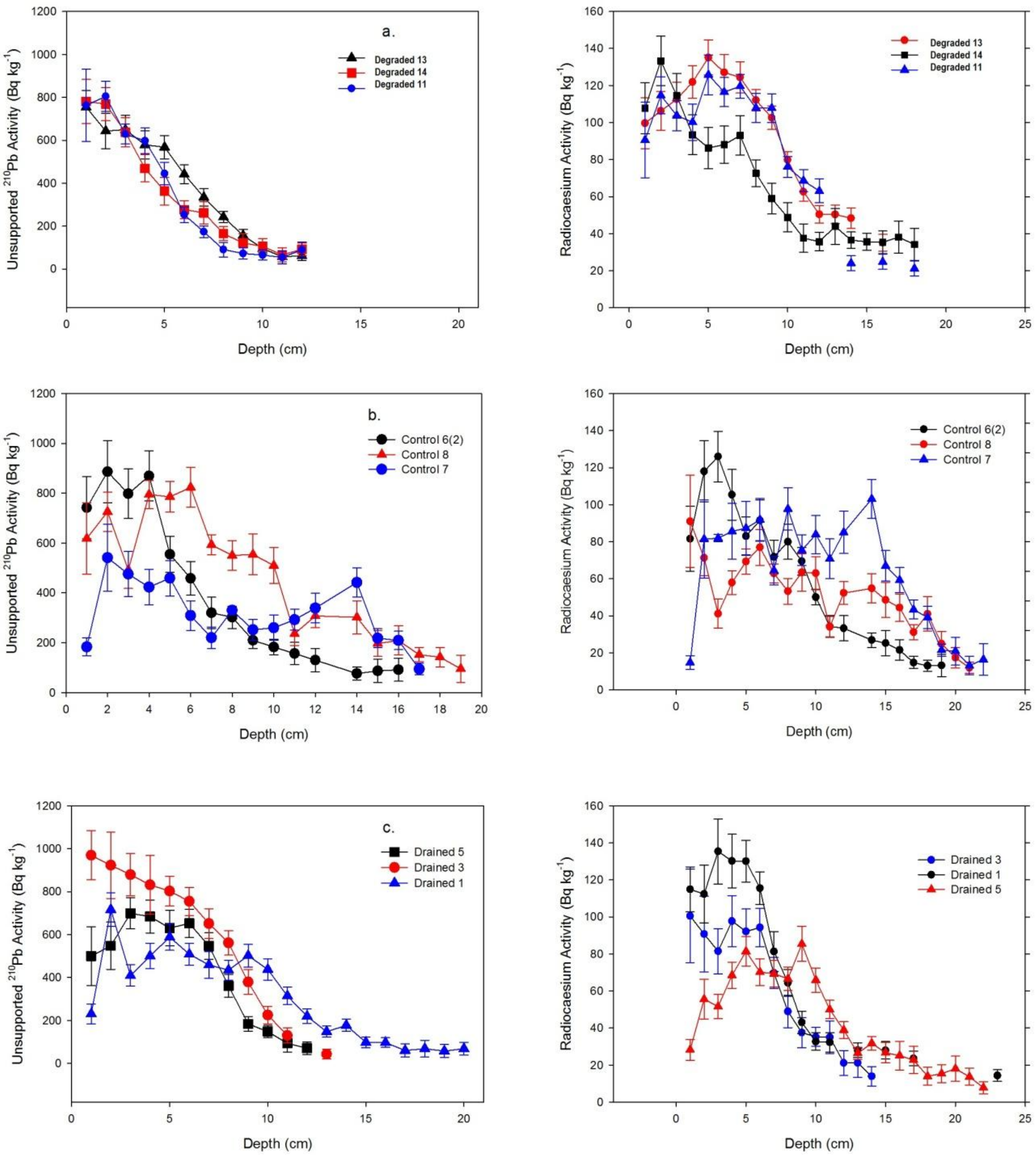

Figure 5.2 ^{137}Cs and ^{210}Pb activities

^{241}Am

Statistically significant levels of ^{241}Am were present in seven of the nine cores. Lower levels of ^{241}Pu, from which ingrown ^{241}Am is derived were released from weapons testing (Appleby *et al,* 1991) and it is more difficult to detect than ^{137}Cs. Different patterns of ^{241}Am fallout were detected within each core profile (Figure 5.3). In two of the seven cores ^{241}Am was only present in one depth, however in five of the seven cores ^{241}Am was present in two or more samples. No clear peaks are present as would be expected due to the nature of ^{241}Am fallout (Figure 5.3) and ^{241}Am is detectable within several cm of the peak.

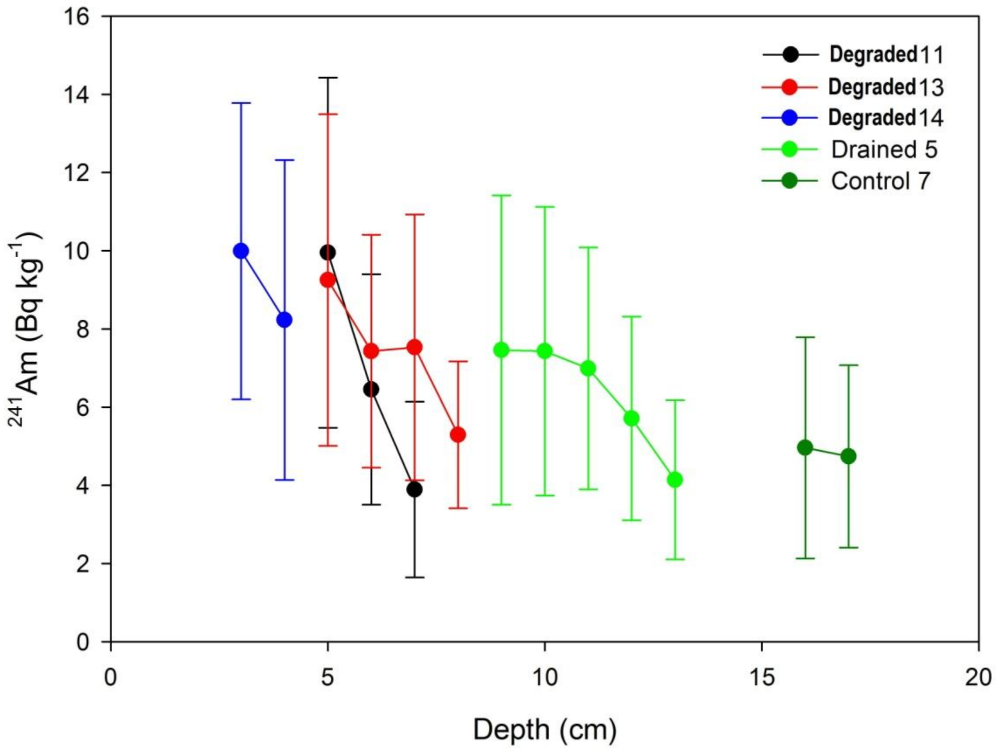

Figure 5.3 ^{241}Am concentration depth profiles for cores where ^{241}Am was detectable in more than one sample

5.4.3 ^{210}Pb dating

The Constant Rate of Supply (CRS) model was used to date unsupported ^{210}Pb of each core (Figure 5.4). In this model a constant supply of ^{210}Pb to the surface is assumed and changes in sedimentation rate are accounted for (Appleby, 2001). The CRS model is considered the most suitable in ombrotrophic peats as ^{210}Pb inputs are dominated by atmospheric inputs (Turetsky *et al*, 2004; Appleby, 2008). The CRS model is calculated by the equation 5.2:

$$T = (1/\lambda)\ln(A_0/A) \qquad \text{Equation 5.2}$$

Where T is age, λ is the ^{210}Pb decay constant 0.0307, A_0 the inventory of unsupported ^{210}Pb under a specified depth and A the total inventory. Error for each of the ^{210}Pb dates was calculated as in Vile *et al* (1995), from the error produced during counting and the error propagated from using the CRS model. The CRS model is considered a robust model for dating in peats (Urban *et al*, 1990) and is used by the majority of studies. It relies on three primary assumptions; a constant supply of ^{210}Pb to the surface, rapid transfer of ^{210}Pb to sediments, and immobility of ^{210}Pb once deposited (Appleby, 2001). The final assumption is the only factor that casts doubt upon the validity of this model and its impact upon dating results in this study will be discussed in greater detail later. Output from cores dated using the CRS model are shown in Figure 5.4.

5.4.4 Dating comparisons

Multiple methodologies must be used when dating recent peat deposits, due to uncertainties involved with each dating technique (Turetsky *et al*, 2004). This will enable conclusions to be drawn about the performance of each dating methodology, help to identify the cores which have reliable chronologies and recognise any patterns which may indicate the causes of agreement and disagreement. For each core, the independent dates based on SCPs and ^{241}Am have been plotted against the continuous CRS ^{210}Pb dates to clarify this (Figure 5.4). This plot shows that there is a broad general agreement between ^{210}Pb and other chronological markers, but that a number of discrepancies do exist. This is further confirmed when these dates are plotted against each other (Figure 5.5). All independent dates are plotted in this graph besides the 'start' SCP date. This was omitted as it is very difficult to detect reliably because of the very low numbers of SCPs in the mid-19th century, although the degree of agreement can still be seen in Figure 5.4.

The pattern of agreement at each site may reveal the causes of agreement / disagreement between each dating methodology according to the sites characteristics. Each of the cores in the degraded site demonstrates the most consistent pattern of agreement between independent (SCP and ^{241}Am) and ^{210}Pb dates. In all degraded cores the ^{210}Pb dates are younger than the SCP dates for peak, take off and start. This pattern does not follow for ^{241}Am in cores 11 and 13, where ^{241}Am is a close match with ^{210}Pb. The control site demonstrates a reasonable agreement between SCP peak and take-off in cores 6(2) and 7 and ^{241}Am where present in core 7. Similar to the degraded cores SCP peaks are dated as younger in these cores, whilst SCP take-off dates are instead older. SCP dates from control 8 however demonstrate a large miss match with ^{210}Pb, this which may be a result of the poor SCP profile (Figure 5.1). The drained site demonstrates a close ^{241}Am match where present, crossing the ^{210}Pb date in both cores 5 and 3 (Figure 5.5). The drained SCP plots do not show a consistent trend. Drained 5 shows a similar pattern to the degraded cores with ^{210}Pb ages younger than the SCP ages, whilst drained cores 1 and 3 peak ages are a good match, but with the take-off ^{210}Pb ages being older. No site shows a consistent pattern of agreement besides the degraded site.

Individual dating features may also reveal consistent patterns. In each site the SCP peak was always a good agreement with the ^{210}Pb date or dated younger by the ^{210}Pb. Whilst the take-off date was variably younger or older than under the ^{210}Pb date. The greatest discrepancies are observed using the SCP start date. In nearly all sites ^{241}Am showed a better agreement with the ^{210}Pb than the SCP dating features.

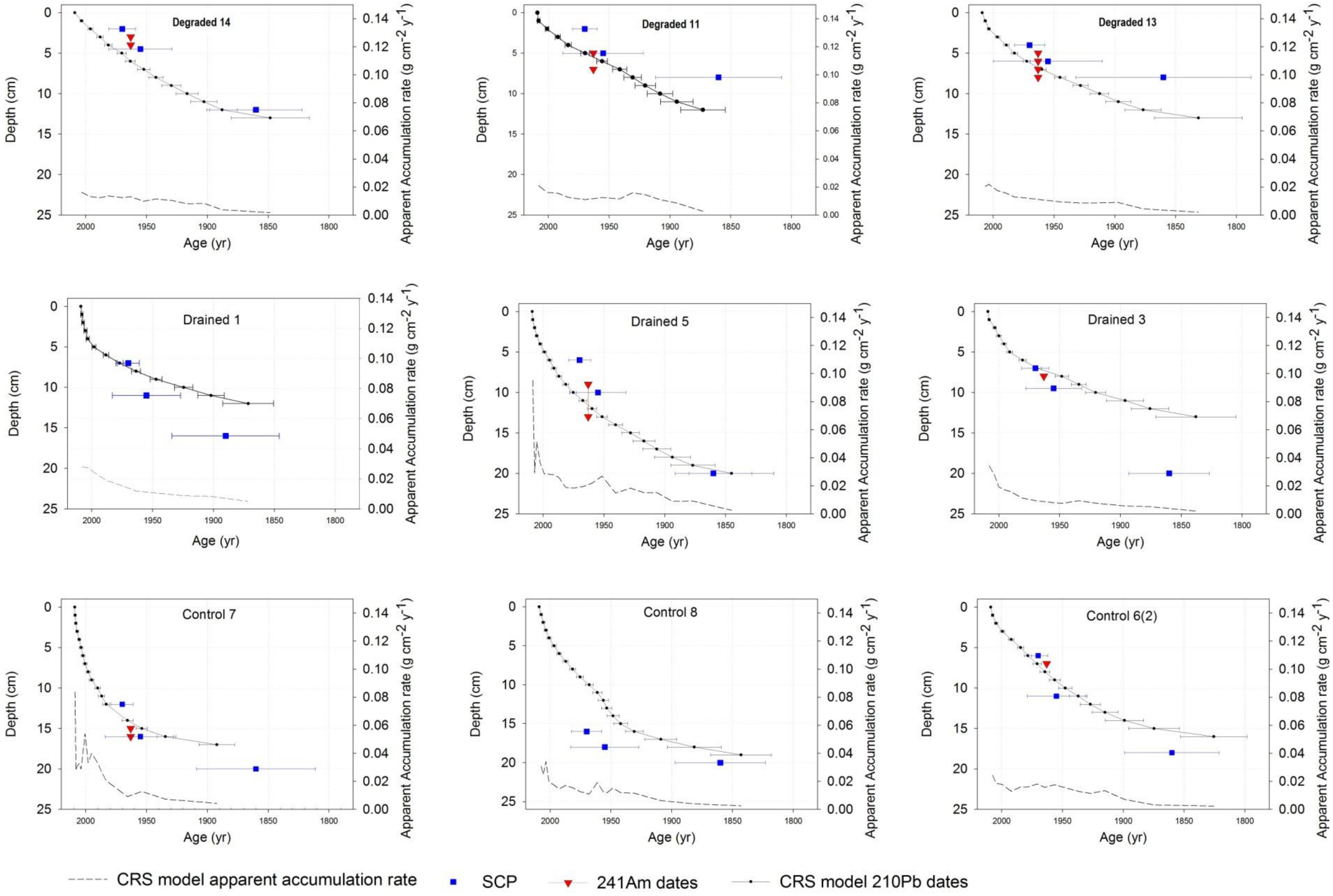

Figure 5.4 CRS 210Pb dates plotted with independent dating markers

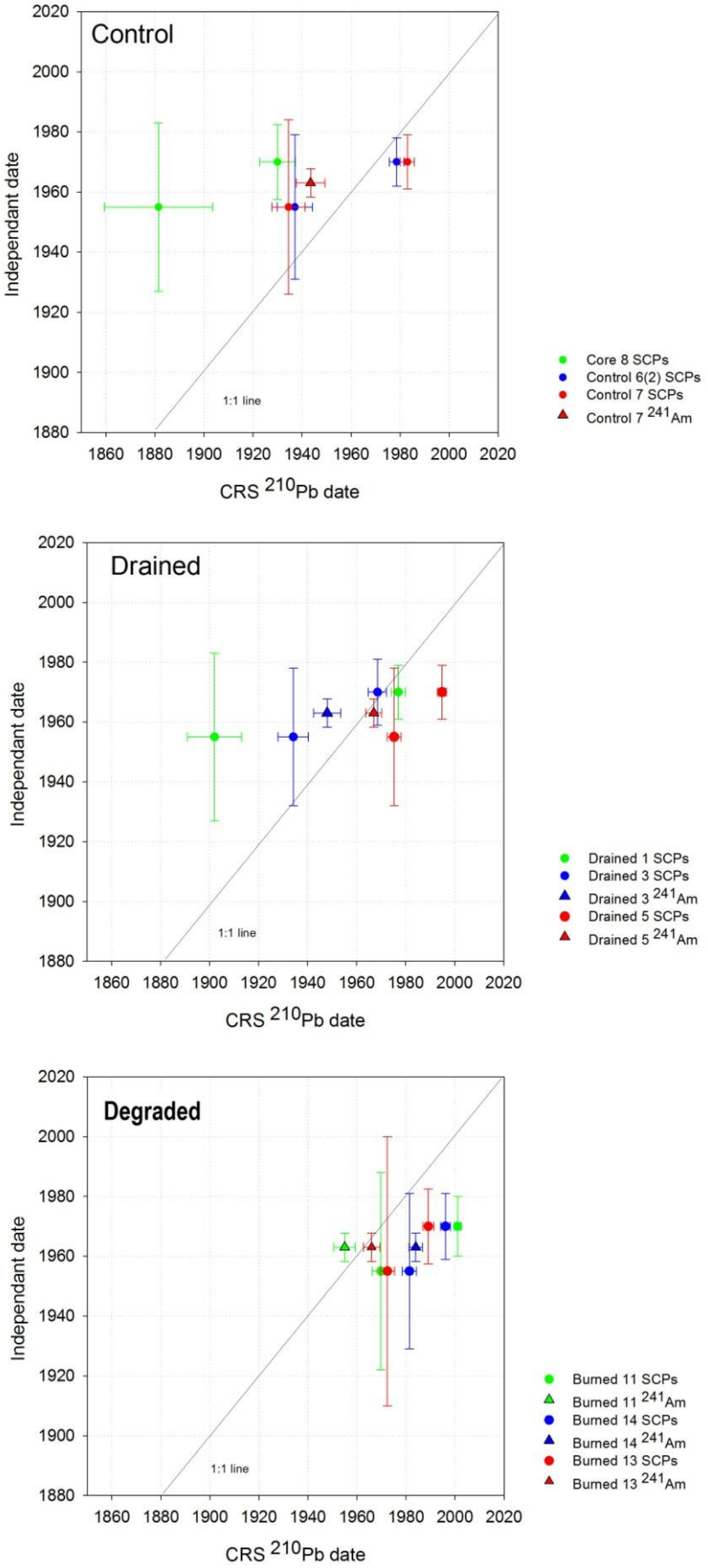

Figure 5.5 Comparison of independent and CRS ^{210}Pb dates. Errors relate to CRS error and gamma counting error for ^{210}Pb dates and Sampling error and SCP error (where applicable) for independent dates. Straight line represents 1:1 agreement.

5.4.5 *Water table levels*

The water table adjacent to the site of each core is presented in Table 5.3. Although water table was recorded at ten points in each management scenario only the water table levels near the cores are included as these are the most relevant for understanding radionuclide mobility. Full water table results will be presented in chapter six. The degraded site has a less comprehensive record due to difficulty regularly accessing the site. No adjacent water table is available for control 6(2) due to resampling of the core.

Treatment	Core	Average depth (cm)	Standard deviation (cm)
Drained	1	22.5 (n=8)	5.7
	3	15.8 (n=8)	9.8
	5	9.8 (n=8)	5.2
Control	7	3.4 (n=8)	4.2
	8	6.5 (n=8)	3.8
Degraded	11	32 (n=3)	
	13	28.5 (n=3)	
	15	29.5 (n=3)	

Table 5.3 Water table levels corresponding to each core

5.5 Discussion

5.5.1 SCPs

SCPs have been used for the purpose of calculating peatland carbon accumulation rates by Garnett *et al* (2000) and Billet *et al* (2010). This technique displays much potential as SCP profiles are cheap and easy to produce (Garnett *et al*, 2000) and SCPs are subject to relatively little vertical mobility (Yang *et al*, 2001). However the use of SCPs in peats has only been carried out in a limited number of profiles and there are very few profiles where SCP and short-lived radioisotope chronologies are available from the same cores. Despite these advantages SCP dating, like other techniques for dating recent peat accumulation, is subject to a number of limitations.

The majority of cores within this study contain datable concentrations of SCPs, with the exception of cores 6 and 10, which were subsequently re-sampled. High levels of rainfall on Dartmoor facilitating the large fallout of SCPs mean they can be relied upon to be present as dating features despite the South West peninsular being less industrialised than other regions of the UK. Probable causes of low levels of SCPs in cores 6 and 10 are either the subsequent disturbance of peat or a location sheltered from high levels of SCP fallout.

In many of the cores, most notably 5, 6(2) and 3 SCP patterns followed the trends outlined in Rose *et al* (1995), meaning that dating features could easily be identified. However, in some profiles, such as cores 7 and 8, the trends did clearly not follow trends identified in Rose *et al* (1995), causing dating features to be more problematic to identify. These cores have double peaks or deviation from the expected trend above and below the peak, this is most evident in core 8. Similar deviation from the expected trend can be seen in a number of the SCP profiles published in the CARBYDAT database, this deviation is a common problem observed in SCP dating. Rose *et al* (1995) outlines protocols to follow in these circumstances, for example the lower peak is always to be considered the true peak. Despite following these, identifying dating features in profiles which deviate from the expected trend relies upon subjectivity, leading to some uncertainty to the dates allocated.

Deviation from the expected trend may be as a result of a number of factors. Variable accumulation rates within a core may cause the dilution and concentration of SCPs, leading to false SCP fallout patterns being interpreted. The CRS ^{210}Pb accumulation rates, plotted in Figure 5.4, demonstrate accumulation variability in cores 7 and 8, where SCP plots also deviate from the expected trend (Figure 5.1). This suggests that variability in accumulation may have caused deviation SCP trends. However the ^{210}Pb CRS accumulation rate calculations rely directly on the immobility of ^{210}Pb, as the immobility of ^{210}Pb is uncertain caution should be applied to CRS accumulation rate calculations. An alternative explanation for variability in SCP trend profiles maybe as a result of local fluctuation in SCP deposition; this is unlikely given the similarity of the environmental setting of each of the management sites and the close proximity of cores within each site between which trends in SCP profiles vary. The cause of the observed deviations from expected trends outlined in Rose *et al* (1995) is unknown and the difficulty caused in correctly identifying dating features therefore must be supported by additional dating techniques.

Less subjective means of dating SCP profiles have been developed to counter these problems. Rose and Appleby (2005) provide additional dating features to Rose *et al* (1995) using SCP profiles plotted on a cumulative curve. However 100% SCP accumulation is taken from the ‘peak’ feature, and as the peak is not present in core 15 and has a double peak in cores 2, 4 and 10(2), this technique was not used as many of the cores were considered unsuitable. Renberg and Wik (1985) used cumulative SCP plots from the full SCP profile, and ^{210}Pb dates from the same core, to transfer dates for cores dated with only SCPs. This method could have been used to transfer ^{210}Pb dates in this study to SCP only dated cores if ^{210}Pb chronologies were considered reliable. However, unlike Renberg and Wik (1984) who used mineral cores, the cores reliability of the ^{210}Pb chronologies is uncertain, therefore this technique was not used to date the SCP only cores. Garnett *et al* (2000) who used SCPs to calculate carbon accumulation did not use cumulative methodologies or Rose *et al* (1995), instead a common horizon was identified when each core reached over a set number of SCPs. Although this technique removed subjectivity of identifying dating features, the SCP concentrations may have been affected by variability in accumulation, diluting or concentrating SCPs in addition to the trend in SCP fallout. The methodology of Garnett *et al* (2000) also would not have provided dates by which the ^{210}Pb and artificial radionuclide dates could have been validated.

SCP records are spatially variable throughout the country (Rose and Harlock, 1998). Regions received different levels of SCP fallout and peak and take-off dating features occurring at different times, due to differing levels and timing of industrialisation. As a result, Rose *et al* (1995) and Rose and Appleby (2005) have been bringing together a database of cores with both SCP profiles and ^{210}Pb chronologies, the 'CARBYDAT' database has helped to identify differing SCP chronologies in different regions. Dartmoor falls within one of the largest regions 'the south'. Few of the cores used for calibrating this region are located in the southwest of England, and those which are in this region provide chronologies close to the coast. Most of the cores used for calibration are instead found within the southeast. Prevailing southwest winds mean that Dartmoor is upstream from SCP production in the southeast. As the southwest was less industrialised, SCP trends in the south west may be different to those in the south east and therefore may be poorly calibrated. This may be a cause for disagreement between ^{210}Pb, ^{241}Am and SCP dates observed in section 5.4.4.

Presence of SCPs in these peat cores proved successful in dating Dartmoor's recent peat accumulation. However issues with calibration and consistency in the trend of SCP profiles have been identified. Yang *et al* (2001) argues that SCPs are an alternative to ^{210}Pb dating in peats. However, the problems identified in this study demonstrate that this is not always the case. In addition to this, SCP dating cannot provide the continuous chronologies that ^{210}Pb can. This makes the two dating techniques incomparable.

5.5.2 Artificial fallout radionuclides

^{241}Am

The fallout radionuclide ^{241}Am is considered a reliable artificial fallout radionuclide in peats, due to its immobility relative to ^{137}Cs (Appleby *et al*, 1988). ^{241}Am decays from ^{241}Pu fallout *in situ* and is becoming increasingly valuable and a dating tool as further decay of ^{241}Pu will increase detectable levels of ^{241}Am until 2037 (Appleby *et al*, 1991). ^{241}Am is not widely reported in peat studies and as a result relatively little is known about it in this context. The patterns and variability in these cores may give an indication of the radionuclide's reliability.

Levels of ^{241}Am detected were very low with large detection errors (Figure 5.3), therefore the presence and absence of ^{241}Am may be to do with variability in ^{241}Pu fallout and proximity to detection limits. Alternatively, ^{241}Am presence and absence may relate to mobility of ^{241}Am. In cores 5, 13 and 11, ^{241}Am is present over a large section of the profile (SCP and ^{210}Pb dates indicate that in core 5 this spread represents a period of 40 years) and no clear peak in ^{241}Am was present (Figure 5.3). Appleby *et al* (1991) state that a clear peak should be present as deposition of ^{241}Pu was very rapid after bomb testing and declined equally quickly; 90% of cumulative deposition would have occurred at this stage and as a result ^{241}Am peaks in sediments should also follow this trend. Even taking into consideration the increasing decay of $^{241}Pu > ^{241}Am$ and errors in SCP and ^{210}Pb dating the ^{241}Am profiles of cores 5, 11 and 13 would not normally be observed. An explanation is that ^{241}Am has been mobilised and has diffused to these concentrations at depth. If this is the case, an explanation for ^{241}Am not being present in 6(2), 8 and 3 (other than it still being below detectable limits) is that it has diffused to undetectable levels. This is a plausible explanation given ^{241}Am presence in local cores less than 1.5m away. In cores 14 and 7 where ^{241}Am is only present in one to two cm slices, either the ^{241}Am has remained immobile, or is in the process of diffusing throughout the cores. Strong peaks of ^{241}Am with good agreement with other dating techniques have been reported in the following studies: Smith *et al,* (1997); Appleby *et al*, (1988); Clymo *et al*, (1990); Oldfield *et al*, (1995). None of these studies have detected the presence of ^{241}Am over a range of depths with no clear peak, as was found here. Oldfield *et al* (1995) and Appleby *et al* (1991) suggest some potential for ^{241}Am mobility and Mitchell *et al* (1992) found evidence for mobility, although this was corroborated against ^{210}Pb dates, which in itself is suspected of mobility. However, to date the ^{241}Am peaks presented in other studies are generally in good agreement with independent dates. Similarly, in this study the presence of ^{241}Am when taken from the median depth has some agreement with both SCPs and ^{210}Pb, when it does occur (Figure 5.5). This indicates that if the trends observed here are related to mobility, it is insufficient to fully invalidate ^{241}Am as a dating tool. The long half life (432 years) of ^{241}Am and improvements in its detection make ^{241}Am a valuable future potential dating tool when ^{210}Pb is no longer available for the industrial period. As ^{241}Am becomes more detectable and the demand for ^{241}Am dating increases further, investigation into the scale of ^{241}Am mobility should be carried out. However, although useful ^{241}Am should not be relied upon as a dating feature, as these results have shown that its presence is not always consistent.

^{137}Cs

Although ^{137}Cs was present in each of the profiles, clear dating features were not present. This is a commonly reported in peat studies (Gerdol *et al*, 1994 and Oldfield *et al*, 1995). Cs^+ is not as strongly exchanged as other cations and hence is mobile in peats which have high cation exchange capacities (MacKenzie *et al*, 1997). Moreover, clay is a key binding site for Cs^+ (Shand *et al*, 1994), and as ombrotrophic peats contain no clay, Cs^+ remains mobile. Evidence for mobility is clear, ^{241}Cs concentrations are still high at levels below the 1963 depth identified by ^{241}Am, SCP dating markers and CRS ^{210}Pb ages (Figure 5.2). Furthermore, any peak in ^{137}Cs does not relate to the ^{241}Am peaks. This data provides further confirmation that ^{137}Cs is not a reliable marker in peats. Appleby *et al* (1997) found that ^{137}Cs retention was greater in a heavily burnt site with a high ash concentration. However this was not the case in this study; although the degraded site, which has high levels of charcoal (see Chapter 7) had slightly higher retention of ^{137}Cs (Table 5.2) it was not statistically significant to deem the use of ^{137}Cs on the heavily burnt sites, such as this, more reliable. Also, only one of the three dated degraded cores contained slight peaks ^{137}Cs peaks (Figure 5.2), neither of which corresponds with CRS ^{210}Pb, SCP or ^{241}Am dates. If charcoal content does have an effect on ^{137}Cs retention it therefore may only be a minor influence in this study.

^{207}Bi

The fallout peak of ^{207}Bi corresponds with the fallout peak of ^{241}Am and the first fallout peak of ^{137}Cs (Bossew *et al*, 2006). ^{207}Bi is not a commonly considered artificial radionuclide, although Turetsky *et al* (2004) discuss ^{207}Bi potential as an additional artificial radionuclide in peats. ^{207}Bi was not found in these cores, this may be for a number of reasons. ^{207}Bi fallout is from bomb testing in the former Soviet Union. Fallout of ^{207}Bi may not have been high enough to leave a detectable signal over Dartmoor. Although ^{207}Bi was detected by Kim *et al* (1997) in salt marsh sediment in the United States further from the emission source than Dartmoor. Also, due to the very low concentrations of ^{207}Bi counting times of 2 – 6 days are needed (Kim *et al,* 1997). It was felt that the presence of ^{241}Am (another low concentration fallout radionuclide) and the large size of samples may have made the 24 hour counting time sufficient to detect a ^{207}Bi signal in this study. The lack of ^{207}Bi found in these samples does not suggest that ^{207}Bi is not a useful marker in peats, although it does indicate that long counting times are needed as suggested by Kim *et al* (1997) despite large bulk samples. Counting all samples for this length of time would be costly, but longer count time could be targeted at samples near the ^{241}Am peak to detect the presence of ^{207}Bi.

5.5.3 *^{210}Pb and ^{210}Pb mobility*

Dating using the fallout radioisotope ^{210}Pb is considered one of the most valuable techniques for dating recent sediments (Turetsky *et al*, 2004). As a natural radionuclide its constant fallout allows continuous chronologies to be calculated (Urban *et al*, 1990). ^{210}Pb dating has been applied in a number of circumstances in peats and is now a well established technique. Despite this there is still a large degree of uncertainty about the mobility of ^{210}Pb in peats and thus the validity of age-depth models based solely on it. Geochemically, ^{210}Pb is a relatively inert radioisotope and its mobility is not an issue in most sediment types (Vile *et al,* 1999). However, Damman (1978) found evidence of Pb mobility in peats and hypothesised that this was due to immobile PbS oxidising to form mobile $PbSO_4$ and Pb union with dissolved organic matter (DOM) causing fluvial loss, both of which could occur in areas of fluctuating water table. It was suggested this would impact upon the reliability of ^{210}Pb dating, an assertion which was subsequently corroborated by Oldfield *et al* (1979) who found poor ^{210}Pb dating agreement with independent markers. As a result, caution is now applied when dating peats using ^{210}Pb. A number of techniques have been used to establish if mobility is occurring; anomalies in concentration profiles, differential ^{210}Pb concentration burdens, and disagreement with independent date markers (Belyea and Warner, 1994). These methodologies have resulted in a number of different conclusions to be drawn regarding the mobility of peat. This study has ^{210}Pb dated several cores, with independent date markers at a high resolution. Several cores located less than 3m apart were dated using ^{210}Pb from three different environmental settings. As a result it may be possible to identify if ^{210}Pb mobility has occurred in these samples and if so the extent to which it may be occurring.

^{210}Pb Fallout inventories

A comparison of total ^{210}Pb inventories to measured fallout is used to establish if ^{210}Pb has been mobilised and leached from the system. This technique can use directly measured Pb inventories such as in Urban *et al* (1990) or indirect calculations of ^{210}Pb flux such as Smith *et al* (1997) and Appleby *et al* (1997). Urban *et al* (1990) found that Pb retention was variable (with losses of up to 75% of the input in some instances) and that this was a function of the characteristics of a site; more loss occurred in hollows with high water tables, than in hummocks with low water tables. Fallout has not been directly measured on Dartmoor and as a result it is not possible to establish mobility in this manner. However, no significant difference is seen between the total ^{210}Pb burdens between any of the sites (Table 5.2) despite significant differences in water table depths. This indicates that large-scale mobilisation due to specific site characteristics has not occurred, in contrast to the findings of Urban *et al* (1990) and Belyea and Warner (1994). Fallout rates have been shown to be relatively consistent throughout the UK (Smith *et al,* 1997) and these are also in good agreement with those from Dartmoor (Table 5.2). This indicates that from all these sites there has not been a significant total loss of ^{210}Pb from the system. This outcome is similar to Appleby *et al* (1997) and Smith *et al* (1997) and demonstrates that in this instance significant losses have not occurred at a large scale.

Fallout profiles

Even minor degrees of mobility could impact upon ages calculated from ^{210}Pb inventories, as immobility is a major assumption within the CRS model (Ali *et al*, 2008). There is a body of evidence that suggests small scale Pb mobility may occur (Oldfield *et al*, 1979; Damman, 1978; Urban *et al*, 1990). Irregularity in expected inventories is a method which Damman (1978) employed to first identify the movement of elements in peat. It was found that Pb was not mobile in the aerobic acrotelm, but accumulated in the zone of water table fluctuation. Vile *et al* (1999) tested this hypothesis in the laboratory, no significant change in inventory was found in peats with high or fluctuating water tables and no evidence was found for Damman's sulphide hypothesis. This indicates that the problem, if any, is not as great as first thought, although Vile *et al* (1999) carried out experiments over a period of five months and mobility may take longer than this to be significant. Inventories of ^{210}Pb are available for each core in this study (Figure 5.2); however assumptions about ^{210}Pb mobility cannot be made from these. It is assumed that each year similar levels of ^{210}Pb are deposited on a core and inventories will exponentially decrease with depth (Smith *et al,* 1997). Deviations from this decrease could be used to indicate ^{210}Pb mobility as in Vile *et al* (1999) and Damman (1978). However this involves the assumption that accumulation rates are constant. Accumulations rates are calculated in the CRS model which relies on deviations from the same expected exponential decrease in accumulation (Ali *et al*, 2008). As a result, the accumulation rates calculated, in this instance, cannot be relied upon as they are from the same proxy from which ^{210}Pb mobility is being assumed. It is therefore not possible to determine if the irregular profiles in cores 7, 8 and 5 (Figure 5.2) are due to changing accumulation rates, ^{210}Pb mobility of a mixture of both (as suggested by Smith *et al,* 1997) but maybe indicative of some mobility. This also applies to the dates produced by the CRS model. If an alternative mechanism for calculating whether peat has accumulating steadily can be developed, ^{210}Pb inventories may be able to provide a more useful insight into ^{210}Pb mobility.

A feature that is present in 5 of the 9 cores is a dip in ^{210}Pb activity at the upper level of the core (Figure 5.2). This feature is present in Appleby *et al* (1997) and has been commented upon in Ali *et al* (2008). Ali *et al* (2008) suggests the dip may be caused by compaction, or that the secular equilibrium of ^{210}Po > ^{210}Pb had not been attained. Neither of these explanations is valid in these cores: equilibrium of ^{210}Po > ^{210}Pb needs to be attained in alpha counted cores, this study used gamma counting which measures ^{210}Pb directly and consequently this explanation does not apply. Moreover, the use of monolith tins allowed for little or no downward compaction, therefore this explanation is also not sufficient. Belyea and Warner (1994) note that in several cores the upper level ^{210}Pb dates do not match the independent date. They suggest that uncompacted peat is an ineffective scavenger of ^{210}Pb, this may be a cause of the dip in this the Dartmoor cores. However, bulk densities in the upper limits of the degraded cores with no dip are similar to those in the drained cores where the dip is present (discussed in greater detail in Chapter 6, Figure 6.4), causing doubt about this being the cause of the dip. The dip is a common feature and the processes causing it may impact upon the application of the CRS model as an exponential decline is expected.

5.5.4 Independent dates

The use of independent date markers has been one of the most common sources of evidence for ^{210}Pb mobility. Although studies which use this technique can only provide an indirect observation of potential ^{210}Pb mobility or immobility (Ali *et al*, 2008) the methodology has been seen as a useful way to validate and constrain ^{210}Pb dates (Oldfield *et al*, 1995). From this, information about trends in accuracy can be gained. However, to date, no standardised methodology exists which defines if an independent date agrees well with the ^{210}Pb CRS date and as a result many studies rely on conjecture (such as Belyea and Warner, 1994 and Boa *et al*, 2010). Moreover, independent date markers can be subject to as much error as ^{210}Pb dating, for example MacKenzie *et al* (1997) used the onset of industrial Pb pollution as an independent date and Urban *et al* (1990), Bao *et al* (2010) and Ali *et al* (2008) used the mobile ^{137}Cs to test for ^{210}Pb mobility, giving rise to circular reasoning. Consideration of these factors must be taken into account when using this technique. A number of different conclusions have been reached from comparisons with independent age markers. El-Daoushy *et al* (1982), Ali *et al* (2008), Clymo *et al* (1990), MacKenzie *et al* (1997), Vile *et al* (1995), Piotrowska *et al* (2010), Bao *et al* (2010) and Appleby *et al* (1997) all found good agreement with independent markers, whilst Urban *et al* (1990), Belyea and Warner (1994), Oldfield *et al* (1979) and Oldfield *et al* (1995) found some disagreement. Often these studies only date a few cores, for example Piotrowska *et al*

(2010) dated only one. Studies such as Oldfield *et al* (1995) and Clymo *et al* (1990) which use multiple cores and Urban *et al* (1990) and Belyea and Warner (1994) who recorded site characteristics are more likely to reveal the extent and cause of ^{210}Pb mobility. A common finding of these is studies is that agreement is least consistent in hollows, which are areas of high or fluctuating water table.

SCP dates and ^{241}Am, where present, were used as independent date constraints in section 5.4.4. As discussed in the earlier stages of this chapter the independent date markers also have uncertainty attached to them, as a result in this section potential inaccuracy in ^{210}Pb, SCPs or ^{241}Am will be discussed. As a whole, individual dating features are not consistently accurate or inaccurate and it is difficult to identify a regular pattern of agreement or disagreement in the data (section 5.4.4). However, considering the findings of Urban *et al* (1990) and Belyea and Warner (1994) it should be a priority to look at patterns of agreement in relation to water table level. Although few patterns were clearly evident, it can be noted that in general, the greater the depth of the date marker the less agreement between SCP, ^{241}Am and ^{210}Pb dates. In most of the cores the 1860 SCP and a number of ^{210}Pb dates were in regions below the water table or in the zone of water table fluctuation (Figure 5.4 and Table 5.3). Although this finding may be related to the large error margins being greatest on ^{210}Pb and SCP dates at this age, it may also be as a result of ^{210}Pb becoming mobilised in areas of high redox, as suggested by Urban *et al* (1990) and Belyea and Warner (1994). Despite this evidence of mobility, ^{210}Pb date agreement with the independent date markers was no better on the degraded site with a lower water table; the degraded cores demonstrate a consistent disagreement with ^{210}Pb dates (Figure 5.5). However, in the degraded site the ^{210}Pb ages are consistently younger than the SCP ages. In order for the degraded cores disagreement to be related to ^{210}Pb mobility, ^{210}Pb would have had to migrate up the core. This is unlikely as water table levels are much lower on the degraded site. Instead the cause maybe that the SCP dates are incorrect; as the trend of younger ^{210}Pb dates is consistent throughout all degraded cores. The consistency between these cores indicates that the error is unlikely to be due to poor placing of the peak, take off and start date marker, but instead more indicative of poor calibration of SCP dates in the south west of England as discussed in section 5.5.1. This theory is supported by the ^{241}Am dates being closer to the ^{210}Pb date than the SCPs in two of the three degraded cores (Figure 5.5). Additionally ^{241}Am dates in all cores more consistently cross ^{210}Pb dates than SCP dates in Figure 5.5. In further support of this, the southwest SCP dated profiles from the CARBYDAT database from Slapton Ley (on the south Devon coast) and Pinkworthy Pond (on the north coast near Exmoor) have a mid 1980s peak and mid 1960s take off respectively. These dates are later than those assigned by Rose *et al* (1995). This evidence calls for better calibration of southwest SCP dates. However, this pattern is not present in all cores dated, and as the evidence is not conclusive and as the reliability of corresponding ^{210}Pb dates is uncertain, for this thesis the original SCP dates will be applied. With consideration of uncertainty related to SCP dates it is difficult to relate agreement of the full ^{210}Pb chronology to water table or any other factor. However in future if recalibration of SCP dates in the southwest does occur this data could be revisited, as younger SCP dates may change patterns of ^{210}Pb agreement and reveal trends related to water table.

5.5.5 Conclusion

This study has provided an example of some of the challenges faced when using methodologies to date recent peats. Although each technique displayed a level of success, it also has identified that each of the methodologies used had a number of uncertainties associated with it. Using multiple techniques has proved a successful approach, as together these techniques are able to provide a coherent understanding of the quality of a chronology. At the very least it is possible to identify profiles where the chronology is most uncertain and distinguish these from profiles which display internal consistency. Until further advances are made in improving understanding the processes behind the variable quality of fallout radionuclide and SCP dates in peats, the use of two or more dating techniques together is essential to test the validity of chronologies. Moreover sampling several cores from a locality should become standard practice in order to ensure that variability in the quality of chronologies is accounted for in interpretation.

6 Managing the peatland carbon resource: the effect of degradation and drainage on carbon accumulation rates

6.1 Introduction

Peatlands store considerable quantities of carbon, an ecosystem service which is of particular significance given the problem of future climate change (Bonn *et al*, 2009b). Peatland carbon stores develop as organic matter slowly accumulates over millennia, as a result of a positive water balance, anaerobic conditions and the resulting low decay rates (Lindsay, 2010). However, this valuable store of carbon is under threat from both future climate change and increasing anthropogenic disturbance in some regions of the world.

The UK's peatlands make up 10-15% of the global blanket peat resource (Tallis, 1998), and are amongst the most badly affected by anthropogenic impacts and pressures (Holden *et al*, 2007a). Blanket peatlands have been forming since the early Holocene in the British uplands, under the influence of processes of natural environmental change and anthropogenic activity such as prehistoric forest removal (Smith and Cloutman, 1988). However, due to processes of industrialisation and increasing population, British blanket peatlands have been subjected to elevated environmental pressures since the eighteenth century (Holden *et al,* 2007a). Many areas of British blanket peatland are now in a degraded state and may have a reduced ability to store carbon. Several forms of anthropogenic activity are assumed to have caused peatland degradation. Pressures such as climate change and atmospheric pollution are thought to degrade blanket peatlands, but these extrinsic threats have multiple external causes and are difficult to regulate. Land management such as burning, drainage and grazing are also sources of pressure. These are intrinsic and can be directly managed to benefit peatland carbon. There is now an emphasis on managing pressures by the UK's upland managers and understanding how degradation may be impacting upon the peatland carbon store. This chapter focuses on the change in carbon accumulation at a drained and degraded site, which has a recent history of burning.

The practice of peatland drainage is carried out to facilitate the lowering of water tables (Holden *et al*, 2004; 2007b). Typically drainage occurred to prepare peat for the process of peat cutting or to improve the agricultural productivity of land. Gullies, which can form as a result of erosion triggers related to land management, are also a significant cause of drainage in many British blanket peat environments (Evans and Warburton, 2007). Drainage has been occurring for several centuries (Holden *et al*, 2004), but the rate and extent of drainage increased markedly following the advent of more effective tools for draining peatlands and the drive for British agricultural self sufficiency after the 1940s (Holden *et al*, 2004). The creation of drainage ditches has now largely ceased in the UK, but the extensive network created in the past is still having a significant impact upon British blanket peatlands.

Many areas of British blanket peatland are now considered to be in a degraded state. Although, in some cases, features of degradation are part of the natural processes occurring on a blanket peatland (Evans and Warburton, 2007), degradation is often attributed to anthropogenic pressure. As a result, there is considerable interest in the role geomorphological degradation, such as desiccation, hagging and gullying, maybe having upon peatland carbon accumulation and storage. Fire is thought to be a major cause of degradation of British blanket peatlands. Burning is deeply ingrained into upland management traditions (Yallop *et al*, 2009) and is still carried out in many areas. Fire is used to alter the ecology, either to benefit grouse shooting practices or to improve grazing (Yallop *et al*, 2009). The use of fire varies significantly across the country, with some areas experiencing regular prescribed burning and others only irregular burning or wildfire as a result of accidental ignition or arson (Davies *et al*, 2008). Although burning is regulated by the Heather and Grass Burning Code, which has a ‘strong presumption’ against burning on blanket bogs (Natural England, 2007), records from Dartmoor National Park Authority and Yallop *et al* (2006) indicate that burning on blanket bog still occurs, often as a result of wildfire. There is concern that change in peatland conditions leading to degradation, together with management, such as burning is considerably restricting the accumulation of carbon on Dartmoor's peatland.

Two main approaches can be used to assess management impact on carbon dynamics. Contemporary carbon budgets have been used to analyse the carbon dynamics of whole catchments. These can involve long term measured budgets such as in Lafleur *et al* (2001; 2003), Roulet *et al* (2007) and Nilsson *et al* (2008) and those which are measured over short time scales and then extrapolated to estimated flux over longer periods such as Worrall *et al* (2009a). The majority of these studies assess the carbon budget for undamaged peatlands, but more recently, Rowson *et al* (2010) measured the carbon budget of a drained peatland. A second methodology involves dating carbon accumulation in peat, a methodology which can be applied over long times scales as in Clymo *et al* (1998); Turunen *et al* (2002), and Tolonen and Turunen (1996) using ^{14}C dating, and short timescales such as in Garnett *et al* (2000), Billett *et al* (2010) and Wieder *et al* (1994), using methodologies for dating recent peats. There is debate surrounding the most appropriate methodology to assess carbon budgets in a peatland. Worrall *et al* (2009a) and Rowson *et al* (2010) suggest that the accumulation technique cannot be used to estimate carbon loss from a system and does not provide detail on greenhouse gas exchange. On the other hand, Turetsky *et al* (2004) state that the large spatial and temporal variability in carbon cycling, which has been observed in Roulet *et al* (2007), can lead to error in extrapolating short-term budgets. In addition to this Turetsky *et al* (2004) argue that global warming is most likely to affect near-surface peat accumulation, due to changes in water table height and soil temperature. As a result, they suggest it is most important to focus research efforts on carbon cycling in near-surface peat. Although all of these points can be considered valid, it can be argued that the techniques are complementary to one another. Peat accumulation takes account of the long term inter-annual variability in carbon exchange, and is relatively cheap and simple allowing broader spatial coverage of carbon dynamics in peatlands. In contrast, full budgets provide comprehensive details of individual components of the carbon balance over short periods of time. Nilsson *et al* (2008) and Roulet *et al* (2007) have successfully used both of these techniques to provide a comprehensive understanding of carbon budgeting in a peatland. These techniques should therefore be seen as separate, but complementary to one another, as each can provide answers to questions where the other cannot.

In this study, carbon accumulation on three sites with contrasting conditions and management histories were investigated. The sites are an artificially drained site, an undrained site (control), and a degraded site with no artificial drainage but records of recent burning. Carbon accumulation on was measured using a range of techniques for dating recent peat deposits. The aim is to improve understanding of the comparative impact of varying management and peatland condition on carbon accumulation.

6.2 Site selection

Dartmoor

Three sites with contrasting management patterns and conditions were identified in the northern area of Dartmoor's blanket peatland. Chapter 2, section 2.3.1 provides a greater explanation of the landscape of northern Dartmoor. To identify these sites an intensive investigation into the records of management were carried out. This included searching historical aerial photography, GIS records held by Dartmoor National Park, published reports and information from land managers. It was important to ensure that each of these sites had otherwise similar conditions for peat formation.

Three sites were selected: a degraded site, not artificially drained but with a recent history of burning; an artificially drained site, with only recent light burning; and, a control site with only very light burning and no drainage.

Originally the intention of this investigation was to examine the impacts of burning, drainage and grazing upon peat accumulation. However, changes were made to this structure following extensive investigation into the long term management records on Dartmoor. Very few sites exist on Dartmoor with long term records of grazing and as a result, it was deemed unreliable to identify sites with enough certainty for grazing to be included as a treatment. Several burnt sites were originally identified using mapping from the 1960s and recent GPR records, these records were filtered to include only those with three or more burns. An area representative of typical drainage was selected in the north of Dartmoor.

Following a review of the historical records, topographical and hydrological setting maps were generated for the areas with appropriate management records. As both hydrology and topography are important controls on peat accumulation rate, it was important to keep these factors as similar as possible on each treatment. Areas with relevant management histories were digitised and using *arc*GIS 9.3 sites were analysed using a 5m DEM to ensure that they were located in similar topographic settings (Table 6.1). Sites were selected in areas where reasonably high levels of accumulation could be expected (low slope, high elevation and greater than 100cm peat depth), to ensure a record of accumulation could be obtained. Care was taken to ensure mesotope and local hydrological features were as similar as possible on each site.

Ultimately very few sites with reliable records of burning in the past 50 years existed, the only site with suitably similar topographic conditions to the control and drained site was heavily degraded. As it is uncertain whether the degradation was caused by burning or other events, the burnt site was reclassified as a degraded site with a recent history of burning which was supported using charcoal analysis. Table 6.1 outlines the key features of each of the sites for comparison.

Site	**Slope (°)**	**Elevation (m)**	**Aspect (°)**	**Known Burns**	**Peat Condition**	**Mesotope**	**Vegetation**
Control	2.8	534	244	0	Intact	Watershed mire (nr. Saddle Mire)	Blanket mire vegetation inc Sphagnum, Eriophorum etc
Drained	3.1	496	287	0	Intact besides drainage	Watershed mire	Vascular species some Sphagnum
Degraded	2.5	577	244	3	Hagged with small vegetated gullies	Watershed mire	Vascular species

Table 6.1 Topographic parameters at each treatment

Also, additional care was also taken to ensure that other management practices which may potentially influence peat accumulation, were kept to a minimum.

- All sites were located on a military firing range, and were subjected to similar levels of military activity (pers comm. DMNP Authority).
- Aerial photography and visual inspection indicates that neither the control nor degraded site has ever been subjected to artificial drainage.
- There have been no recorded burns on the drained and control site. To confirm this charcoal analysis was used to check the burning history.

- As discussed, no continuous records of grazing are available and it was difficult to attain whether the sites had been similarly grazed. However, all of the sites were located on the Forest of Dartmoor, as stocking levels are regulated at a common level each will have received largely similar numbers. Furthermore, Meyles *et al* (2006) identified that stock tend to group in localised areas of moorland. Very few stock were observed during any of the site visits and it can therefore be assumed that none of the locations are heavily grazed.

Degraded site

Black Hill is an area of degraded peatland in the north of Dartmoor, the area is covered by extensive hagging and much of the peat is considerably desiccated and humified. A few small gully channels are present, however much of the area is vegetated with grammoid species and linkages for loss of Particulate Organic Carbon (POC) are minimal. The original cause of the degradation is unknown, it is possible that it was initiated several centuries ago. Black Hill has been subjected to at least three large fires in the last fifty years. Continuous records are not available for this period, however the Dartmoor National Park (DMNP) records of burning events in the past 13 years identify that Black Hill was the most frequently burnt area of blanket peat on Dartmoor. This trend is reflected in aerial photography interpretation and mapping from Ward *et al* (1969) which highlights Black Hill as a burnt site in the in the 1960s (see chapter 2, Figure 2.2 for burning records). High rates of burning are confirmed using the elevated levels of charcoal found at Black Hill, outlined in figure 6.9. Fires in this area have been large between 485 – 1552 ha and are recorded as unplanned, indicating that they have been wildfire or arson. Vascular plants such as *Molinia caerulea, Trichophorum cespitosus* and *Juncus squarrosus* are dominant, and no *Sphagnum* is evident. Accumulation rates on Black Hill can largely be to the degraded nature of the site possibly occurring before the burning records begin. However, the recent records of burning may have a level of contribution to the carbon accumulation rate.

Drained site

Blackbrook head is a 100 ha area drained for agricultural improvement and peat cutting in northern Dartmoor. More than 30km drains and cutting have taken place on this site and little of the area is unaffected (Figure 6.1). Not much is known about the timing or causes of peat cutting on this site or across Dartmoor. The site forms part of the Dartmoor Blanket Bog Restoration Project, although the coring site is not in close proximity to the restoration works. The site was selected as it is considered typical of the drainage occurring on Dartmoor. The area has drains set 12 meters apart, the drains are 75 cm wide and are in an advanced stage of recovery, with vegetation encroachment into the drains (Figure 6.2). Vegetation between drains primarily consists of vascular species, and some *Sphagnum* is present in the drains.

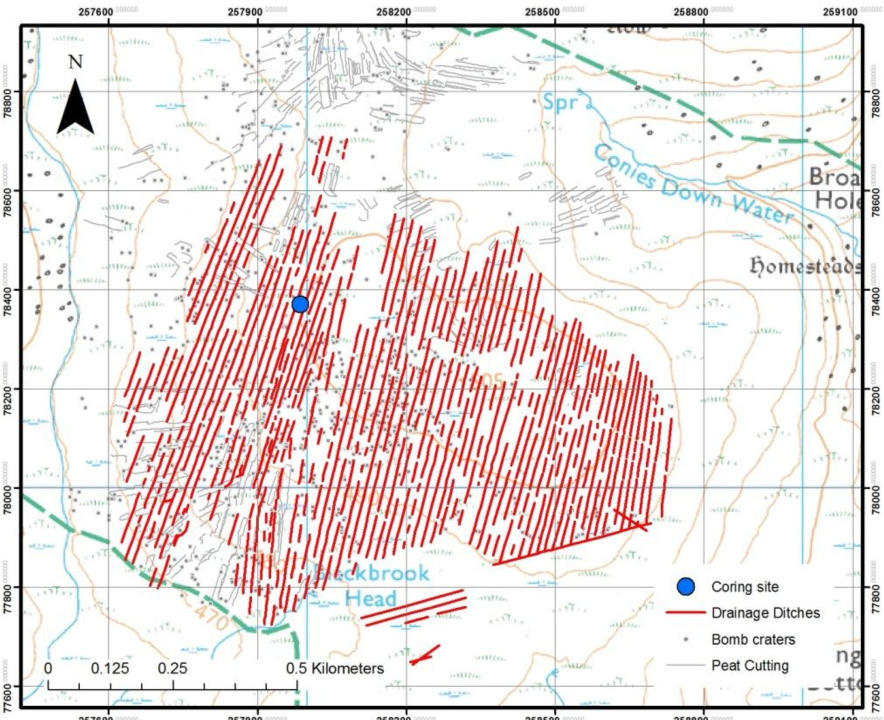

Figure 6.1 Blackbrook Head drainage site, GIS dataset from Fyfe (2008)

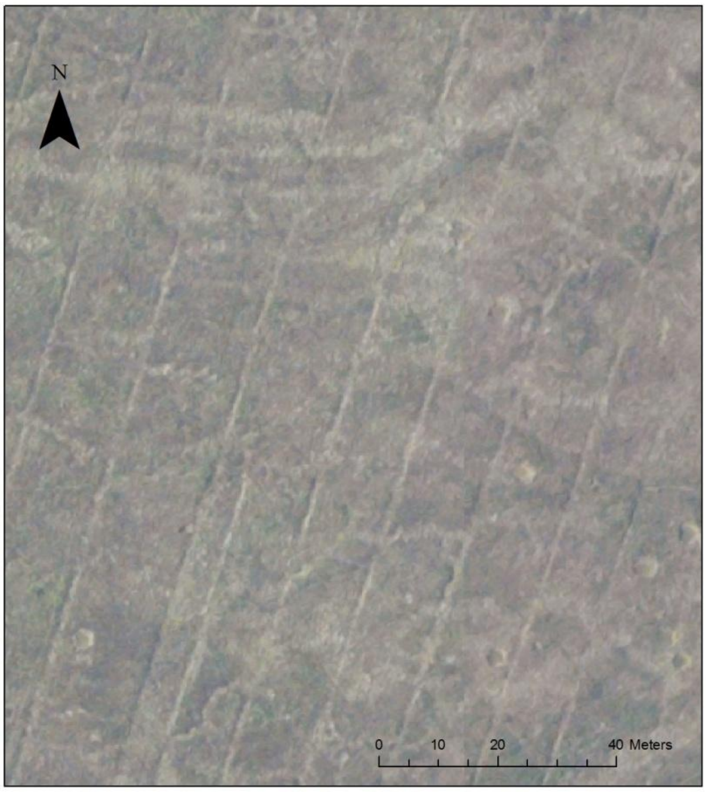

Figure 6.2 Aerial photography of drainage coring site

Control site

Maiden Hill is a site where no visual or recorded evidence of drainage or burning was identified. Aerial photography, fire records and charcoal records were used to verify this. Charcoal levels found confirmed the low rate of burning, discussed in greater detail in section 6.4.7. The site has a healthy mix of blanket mire plants, including *Sphagnum papillosum*, *S. capillifolium*, *Calluna vulgaris, Eriophorum angustifolium*, *E. vaginatum* and *Drosera rotundifolia*.

6.2.1 Water table levels

Water tables were measured for eight months between March 2009 and February 2010 at the control and drained sites, and for three months at the degraded site between September 2009 and March 2010 (lower sampling was due to logistical problems accessing the site). This analysis was carried out to clarify site conditions. Differences between water table levels at each site can be seen in Figure 6.3. The control site has a shallower water table than the drained site (one tailed two sample t-test, $P<0.01$) and the water table rises above the surface level of the peat in some cases (Figure 6.3). The water tables of the drained sites do not demonstrate any clear relationship with their relative distance to the drainage channel (Figure 6.4). The degraded site has the deepest water tables. However as water table levels are strongly controlled by climatic conditions (Clay *et al*, 2009), these results may not be representative of seasonal variation and a statistical comparison could not be drawn.

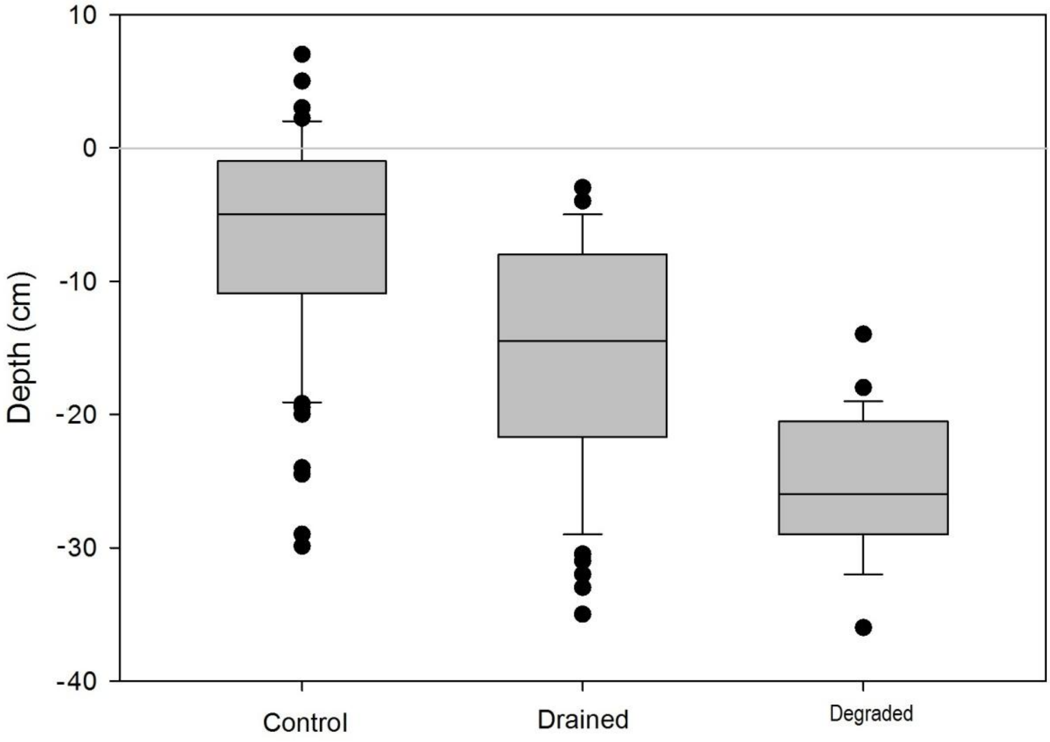

Figure 6.3 Water table levels at each treatment (control n=80, drained n=73 and degraded n=29). Grey line indicates peat surface.

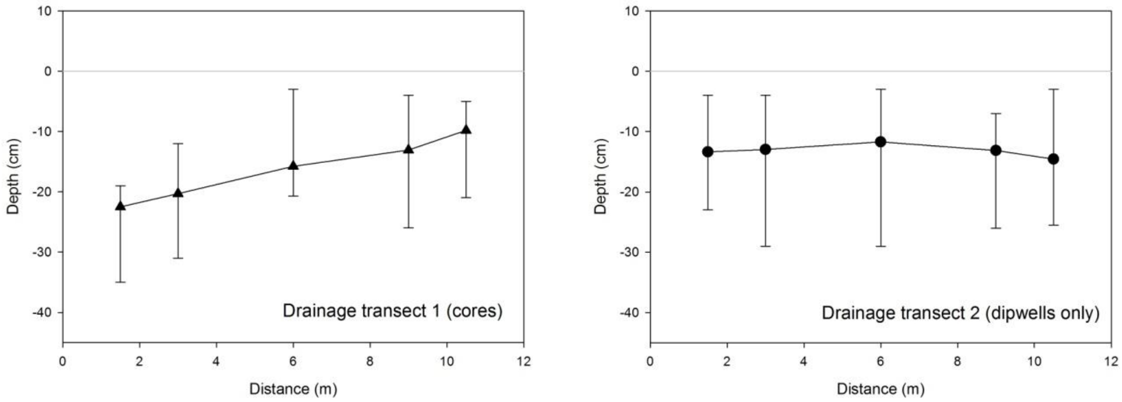

Figure 6.4 Water table depth between drainage channels for each transect, 12m represents upslope on drained transects, grey line represents the peat surface and error bars denote range of water table levels observed.

6.3 Methodology

6.3.1 Field methodology

Two twelve meter long transects were located at each site. Along the first transect five 30 cm monolith cores were extracted at 1.5m or 3m intervals. Dip-wells were placed at points along both transects. Cores were extracted from the lower transect (see Figure 6.5).

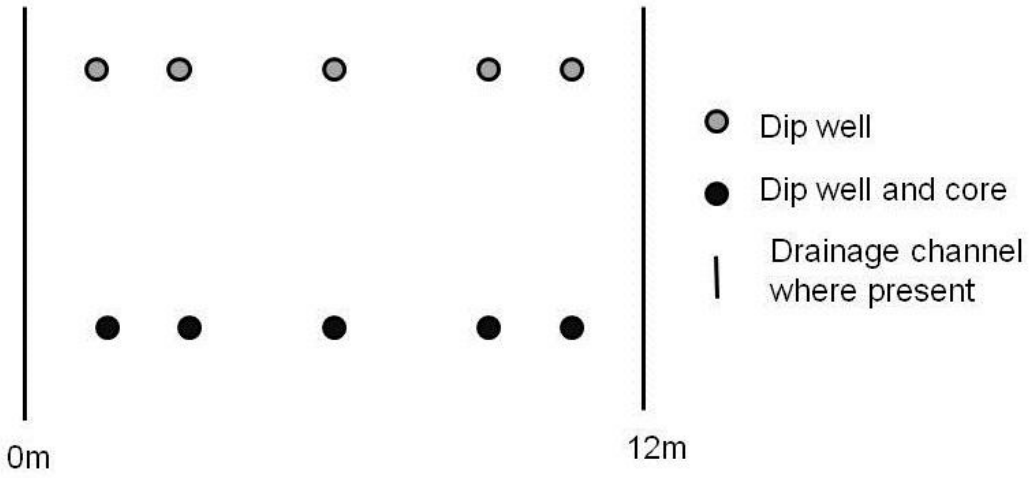

Figure 6.5 Sampling strategy for each site

6.3.2 Laboratory methodology

Dating and charcoal

Each core was dated using Spheroidal Carbonaceous Particles (SCPs) and radionuclide dating. Chapter 6 covers the methodologies used in greater detail.

Bulk density

In the field the monolith cores were extracted with great care, to ensure bulk densities were not affected by sampling (see Chapter 5, section 5.3.1). In the laboratory each core was cut into 1cm slices with care being taken so that the surface area of each sample was calculated without compression. Sections were freeze-dried until a constant pressure was achieved. Bulk density was calculated using equation 4.1 in Chapter 4.

Loss on Ignition

Organic matter was calculated for each sample by loss-on-ignition at 550^0C for 4 hours (Allan, 1989). LOI was then converted to % carbon, using regression derived by Bol *et al* (1999) for British peat and organic soils (R^2adj = 98%). This is the same methodology as applied in chapter 4.

6.4 Results

6.4.1 Bulk density and carbon contents

Bulk density and carbon contents were recorded at in 1cm increments throughout each core profile. These values were used to calculate the carbon content of each core and form the basis of calculating accumulation rates. Carbon and bulk density profiles can also reflect trends between each management type (Figure 6.6 and Figure 6.7); the control site has low bulk density in the upper profile that gradually increases with depth, whilst the burned and drained sites generally have bulk densities that are stable throughout the profile but higher than the control.

Differences in bulk density between the treatment areas may also enable understanding of peatland response to management activities. Increased disturbance and potential for water table draw down may lead to higher bulk densities in the management burned and drained sites than in the control site. Bulk densities for each treatment type, above 20cm depth, were tested using a one tailed two sample t-test the hypothesis that the management site has greater bulk density than the control site.

Results in Table 6.2 show that the null hypothesis can be rejected and bulk densities on both drained and burned sites are significantly greater than those on the controlled site.

Treatment	Average	St dev	P-Value
Drained	0.14	0.02	0.001
Burned	0.12	0.01	0.015
Control	0.10	0.01	

Table 6.2: Variation in bulk density between managed and control sites. Results of one tailed two sample t-test.

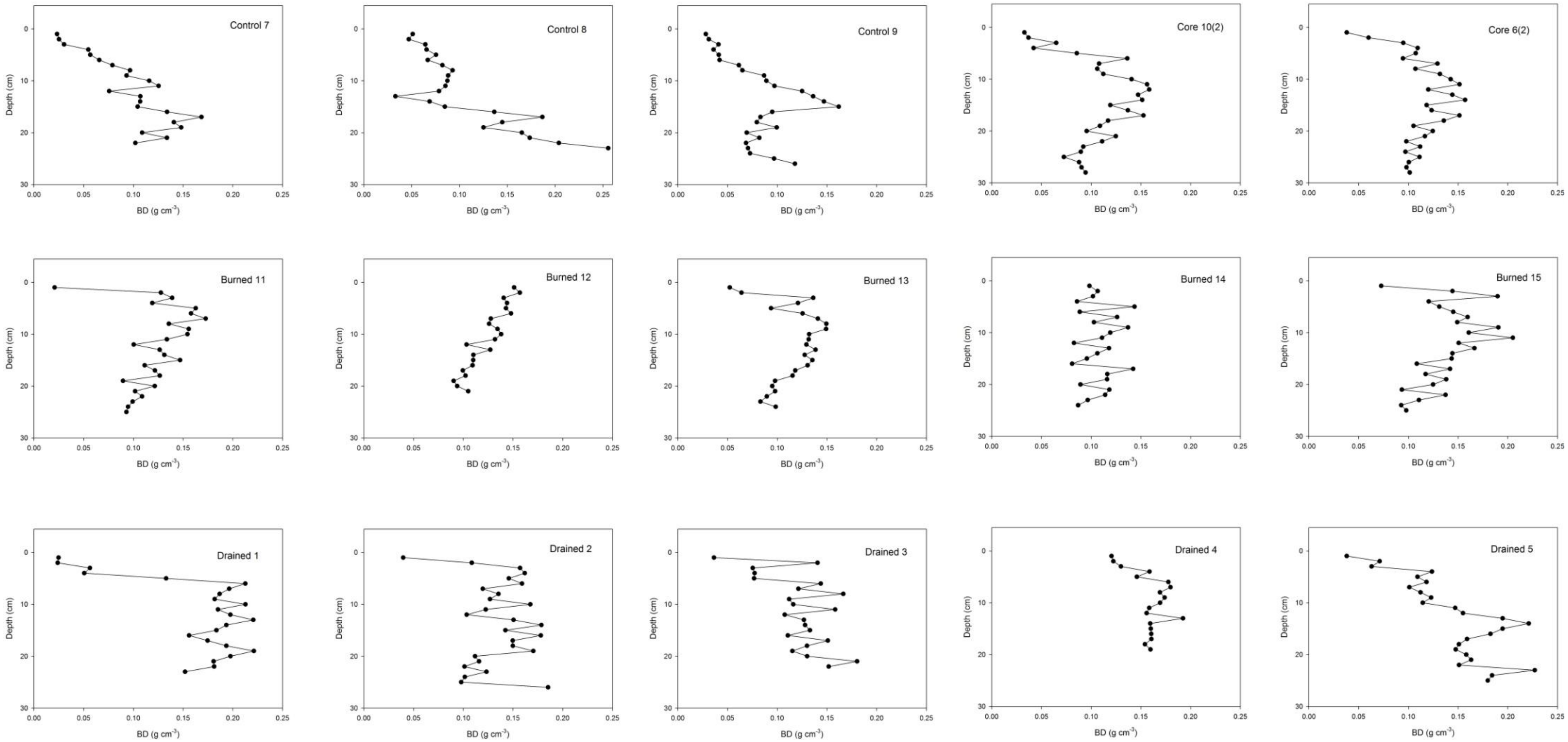

Figure 6.6 Bulk density profiles for degraded, drained and control sites

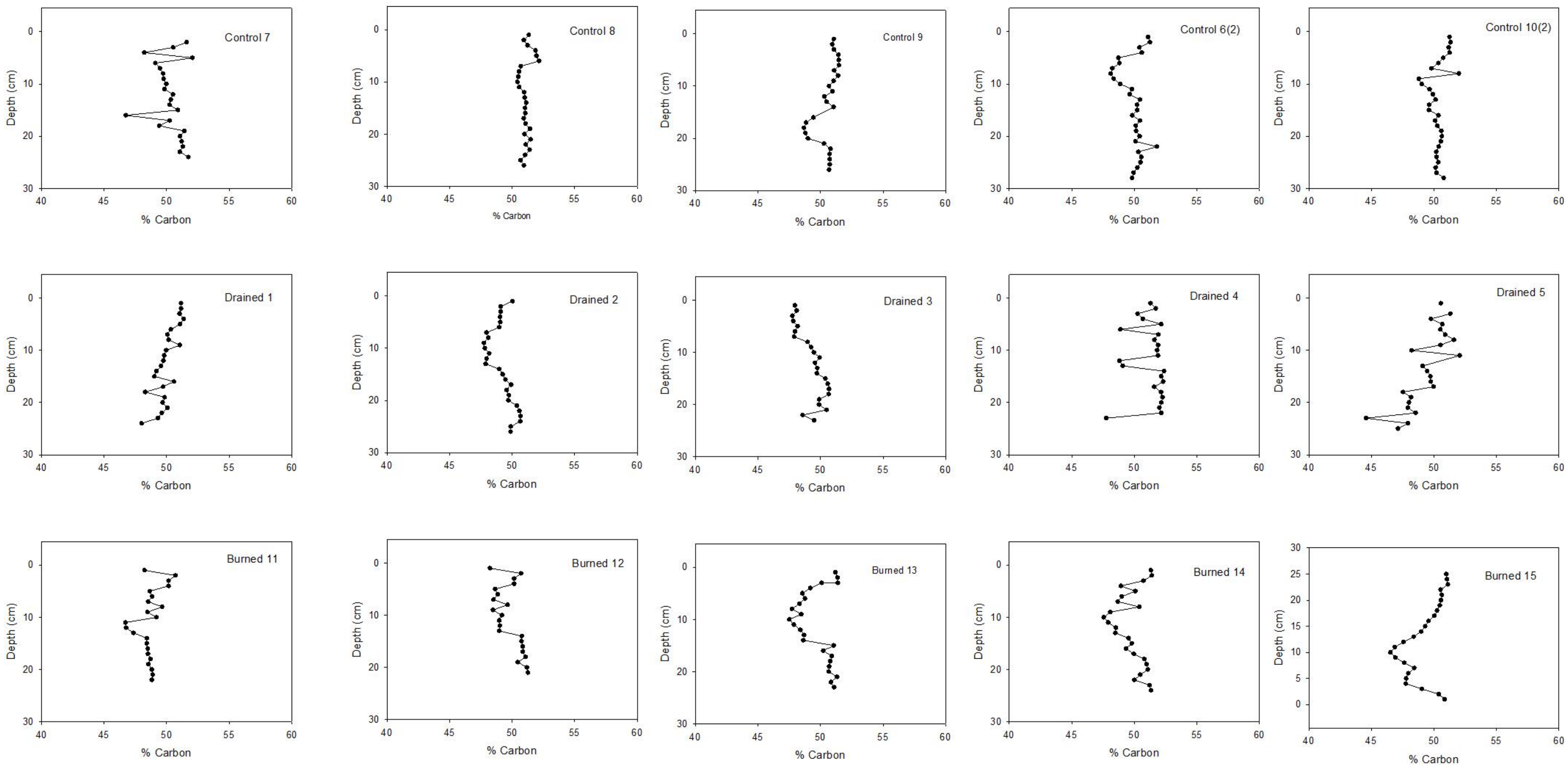

Figure 6.7 Carbon % profiles for degraded, drained and control sites

6.4.2 Impact on carbon accumulation

Carbon density (g C cm^{-3}) for each core slice was calculated by multiplying bulk density and carbon contents. Total carbon accumulation above benchmark dates was then calculated by summing carbon densities in each slice above its depth. These benchmarks were treated as common horizons only; absolute dates were not applied to circumvent inaccuracy in assigning dates to marker horizons. The following benchmarks were used:

- ^{241}Am peak when present
- CRS ^{210}Pb date of 1970 (accuracy was considered greatest at this date in chapter 6)
- The SCP peak, take off and start depths
- The depth of each core at which SCPs reached 40% of the total deposition

Carbon accumulation above each benchmark feature is presented in Table 6.3 and box plots in Figure 6.8. In this, it can be seen that the degraded sites have consistently less carbon accumulation than the controlled site. As a result, degraded cores carbon accumulation was compared to the control site using a one tail two sample t-test to test the following hypothesis.

H_0 : Control cores do not have higher levels of accumulation than degraded site

H_1 : Control sites have greater accumulation rates than degraded site

The t-tests show that the difference between the accumulation rates at the degraded sites and the control are statistically significant for each SCP benchmark ($p<0.05$). The ^{241}Am and ^{210}Pb benchmark dates however are not significant ($P = 0.237$ and 0.103), although this is probably due to low sample numbers (n). As nearly all of the benchmarks demonstrated a difference in accumulation statistically and Figure 6.8 demonstrates less accumulation in burned sites in the ^{241}Am and ^{210}Pb date, it is considered unlikely that dating uncertainty could explain the difference. As a result, the null hypothesis was rejected. Accumulation rates are statistically lower in the degraded site.

Drained sites did not reflect the pattern of the degraded site when compared to the control (Figure 6.8). Accumulation rates of the drained and control sites appear to be very similar, with neither demonstrating consistently higher carbon accumulation (Figure 6.8). As a result, two tailed two sample t-tests were applied, to test the following hypothesis:

> *H_0 : Control cores and drained sites have similar carbon accumulation rates*
>
> *H_1 : Control sites and drained sites carbon accumulation rates are dissimilar from one another*

The drained treatment was not significantly different from the control site for every bench mark (Table 6.4). This resulted in the null hypothesis being accepted: there is no significant difference between the drained and the control carbon accumulation.

Treatment	Core	Accumulation g C m^{2}					
		^{241}Am	^{210}Pb 1970	SCP			
				Start	Take off	Peak	40% Accum
Drained	1		3528.2	12094.5	7378.1	3528.2	5393.2
	2			13579.4	7508	6421.5	4363.8
	3	4035.5	3220.4	10979.9	4874.3	3220.4	3220.4
	4			14933.1	9249.7	5572.3	3732.3
	5	5668.5	5668.5	13324.4	4902.3	2635.8	
Control	6(2)	3142.9	3143	10513	5746.9	2519	4759.8
	7	5781.1	5293.8	9323.3	6453.8	4212.7	8443.2
	8		3684.8	15227.2	7856.5	6166.2	4473.4
	9			10104.2	6492.3	6028.3	4295.3
	10(2)			8691.4	4354.3	2027.5	5920.9
Degraded	11	3605.4	2833.5	5117	2833.5	747.7	2041.8
	12			6947.2	3647.9	1524.6	2952.6
	13	3316	2972.5	4381.8	2972.5	1902.0	2372.4
	14	1778.5	2707.7	6432.4	2348.1	1052.1	1568.5
	15			3926.2			2603.6

Table 6.3 Total carbon accumulation since different time horizons in each core

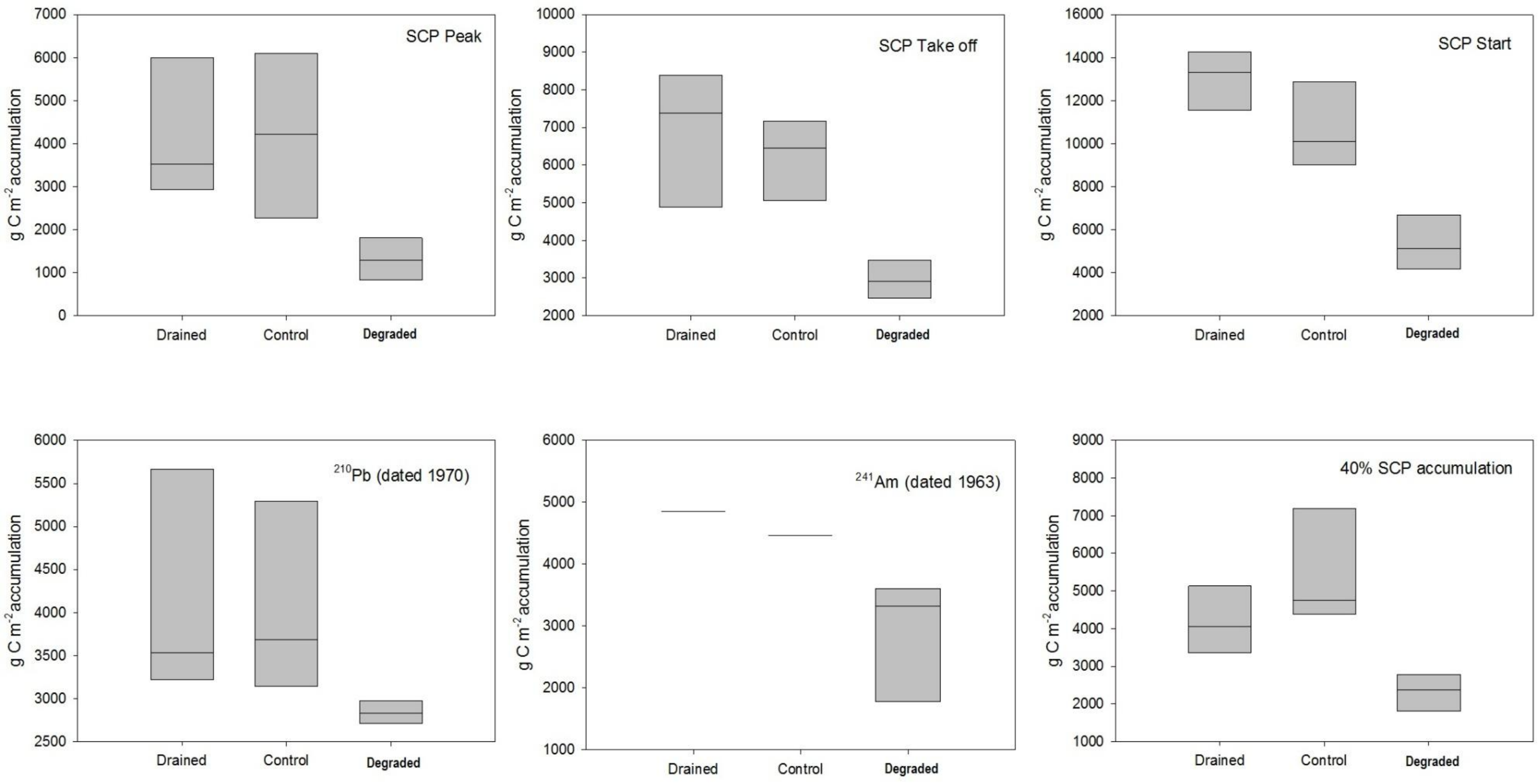

Figure 6.8 Box plots showing accumulation rates calculated at each treatment using each dating technique

Treatment	Dating benchmark	N	P-value
Degraded	^{241}Am	5	0.237
	^{210}Pb (1970)	6	0.103
	SCP Start	10	0.004
	SCP Take off	9	0.002
	SCP Peak	9	0.016
	SCP 40%	10	0.008
Drained	^{241}Am	4	0.843
	^{210}Pb (1970)	6	0.928
	SCP Start	10	0.150
	SCP Take off	10	0.572
	SCP Peak	10	0.942
	SCP 40%	9	0.171

Table 6.4 Two sample t-test results between management treatments and control using benchmark dates (degraded = one tailed, drained = two tailed)

6.4.3 Positioning of the local drainage system

The effects of drainage upon carbon accumulation may be related to the distance from a drain as a result of differential water table draw down. In this study, cores were taken from a 12m long transect between two drainage channels. The accumulation response using each dating technique across this transect is plotted in Figure 6.9. No pattern in accumulation according to distance from drainage channels is apparent.

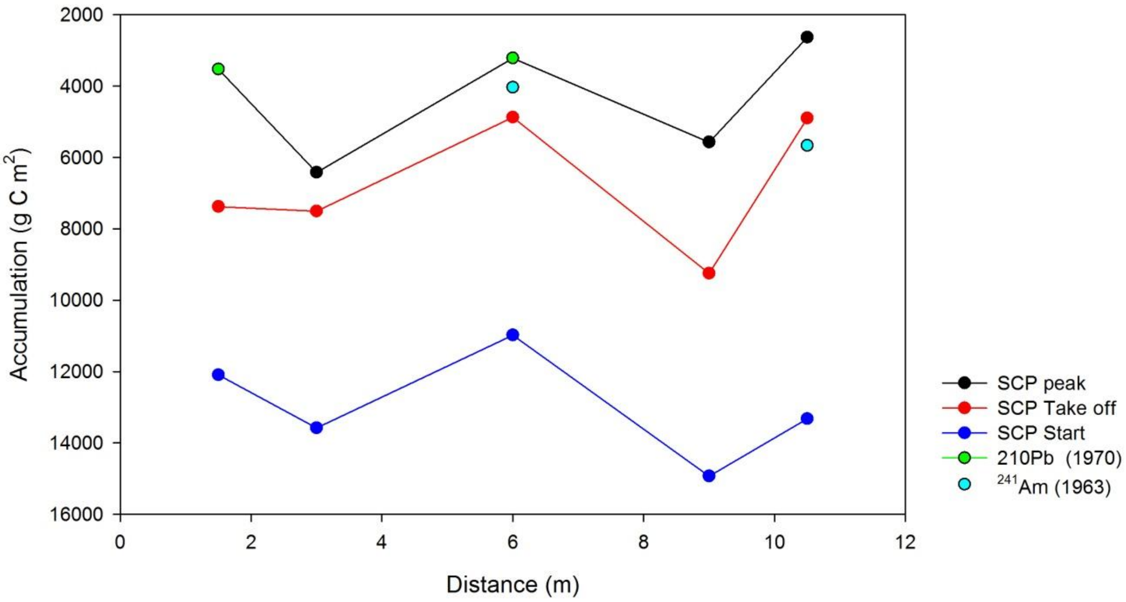

Figure 6.9 Carbon accumulation between two drainage channels. Drains are located at 0 and 12m along the transect. Slope runs downwards from 12m.

6.4.4 Drainage flow direction

In order to understand the efficiency of drainage on Blackbrook Head, a flow direction vector plot was created in *arc*GIS 9.3 using the eight direction flow model (Jenson and Domingue, 1988) within the Spatial Analyst tool of *arc*GIS 9.3. This plot identifies that the natural direction of flow often runs perpendicular to the orientation of drainage (Figure 6.10).

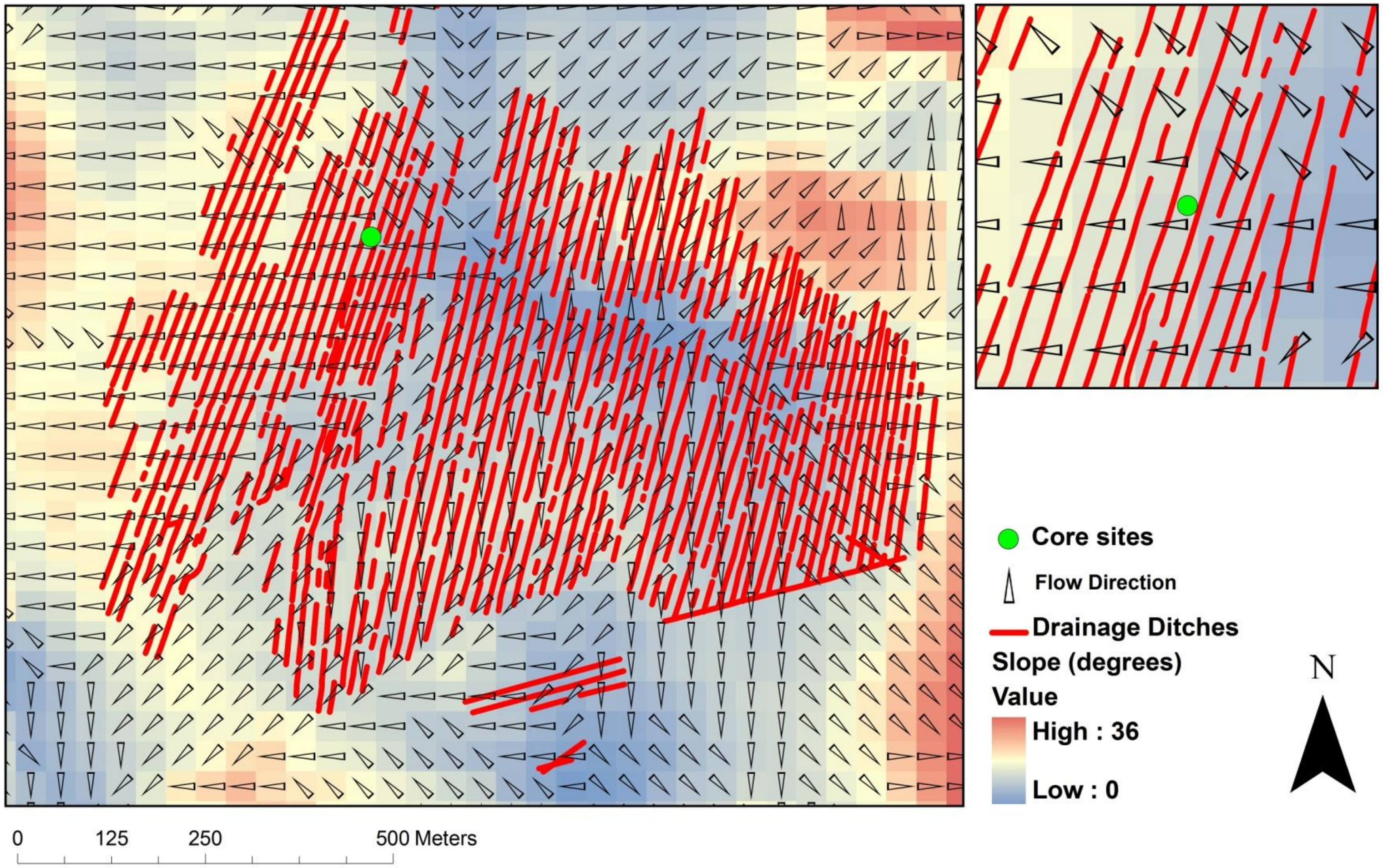

Figure 6.10 Flow directions and slope in proximity to drainage channels

6.4.5 Annual carbon accumulation

The calculation of an annual accumulation of carbon is of use to land managers, policy makers and scientists to understand the potential impacts of management upon carbon sequestration rates and for comparison with other peatland environments. Average annual carbon accumulation rates were calculated using each dating technique using equation 6.1. The average annual accumulation from all dating techniques for each treatment was then calculated (Table 6.5). Due to large potential for error in the SCP start date (see chapter 6) these values were discounted for the mean value.

Equation 6.1: Annual carbon accumulation (g C m^{-2} yr^{-1}) $= \frac{\text{g C m}^2 \text{ above benchmark}}{\text{years since bench mark}}$

Uncertainty in each of the dating techniques has been discussed in chapter 6 and must be considered as a potential cause of error. However annual accumulation rate for each benchmark is presented to indicate potential variability in accumulation rate throughout the acrotelm. Upper and lower error limits in annual carbon accumulation were calculated for each core using error from the dating technique (i.e. SCP error and the CRS counting error) and sampling error. This value was added to each date (see Chapter 5, Table 5.1). Upper and lower limits of accumulation were calculated for each core using equation 6.2.

Equation 6.2: Accumulation error = $\frac{\text{Total accumulation above benchmark (g C m}^2\text{)}}{\text{years since benchmark date} + \text{or} - \text{error}}$

Treatment	Benchmark	Average accumulation (g C m^{-2} yr^{-1})	Maximum accumulation (g C m^{-2} yr^{-1})	Minimum accumulation (g C m^{-2} yr^{-1})
Drained	^{210}Pb (1970)	103.5	132.7	82.2
	^{241}Am (1963)	103.2		
	SCP Peak (1970)	106.9	143	85.7
	SCP Take off (1955)	123.3	241.4	84.6
	SCP start (1860)	86.5		
	Mean	**105.2**	**172.4**	**84.1**
Control	^{210}Pb (1970)	101	132.8	84.7
	^{241}Am (1963)	94.9		
	SCP Peak (1970)	104.7	185.1	89.1
	SCP Take off (1955)	112.4	245.1	80.4
	SCP start (1860)	71.8		
	Mean	**105.2**	**187.7**	**84.7**
Degraded	^{210}Pb (1970)	70.9	91.8	57.5
	^{241}Am (1963)	61.7		
	SCP Peak (1970)	32.7	45.2	31.6
	SCP Take off (1955)	53.6	161.2	33.5
	SCP start (1860)	35.7		
	Mean	**55**	**75.1**	**40.8**

Table 6.5 Annual carbon accumulation rates for each treatment

6.4.6 Local variability within management sites

The variability in carbon accumulation rate between cores from the same management site may reveal additional information surrounding the response of peatlands to management practices. The average annual carbon accumulation rate for each core was calculated using every benchmark date available. Using these values the covariance and standard deviation of annual carbon accumulation for each management site is presented in Table 6.6. This table suggests that local carbon accumulation on both the drained and the control site was much higher than the degraded site.

Treatment	**Standard deviation (g C m^2)**	**Covariance %**
Drained	30.7	29
Control	31.6	30
Degraded	7.4	13

Table 6.6 Variability in annual carbon accumulation rate between cores from the same management site

6.4.7 Charcoal frequency:

Trends in the charcoal plots indicate the fire frequency at each site (Figure 6.11). All of the degraded cores record considerably more charcoal throughout their profile than both the drained and control sites. This suggests that the records of burning on Black Hill are representative of this site receiving a higher than average frequency of burns in the past 50 years. Several of the degraded cores have more than one peak, indicating that the site has been subjected to fire on more than one occasion. The dating models from chapter 6 imply that a peak in charcoal was consistently present in the 1960s, suggesting that the aerial photography interpretation was accurate for this time. Also substantially higher levels of charcoal observed in the degraded peaks indicate that the fires sustained at this site may have been more severe. The drained site has much lower charcoal concentrations, although each core has one definite peak. The peaks in drained one and five indicate a burn in the late 1990s. However the National Park records, which have comprehensively covered burning since 1997, do not show a burn here at this date. This peak may instead be from a charcoal brought in by the wind from a nearby fire or error in recording. The control site demonstrates low levels of charcoal, any present at this site is likely to be from windblown sources or in the worst case light burning. Although the degraded site has greater levels of charcoal and therefore perceived higher rates of burning, this cannot be seen as the sole cause of slower accumulation at the site. It is possible that the degraded conditions found at Black Hill are as a result of a combination of several factors, such as past changes in climate which may have triggered degradation. As a result caution must be applied when interpreting these results.

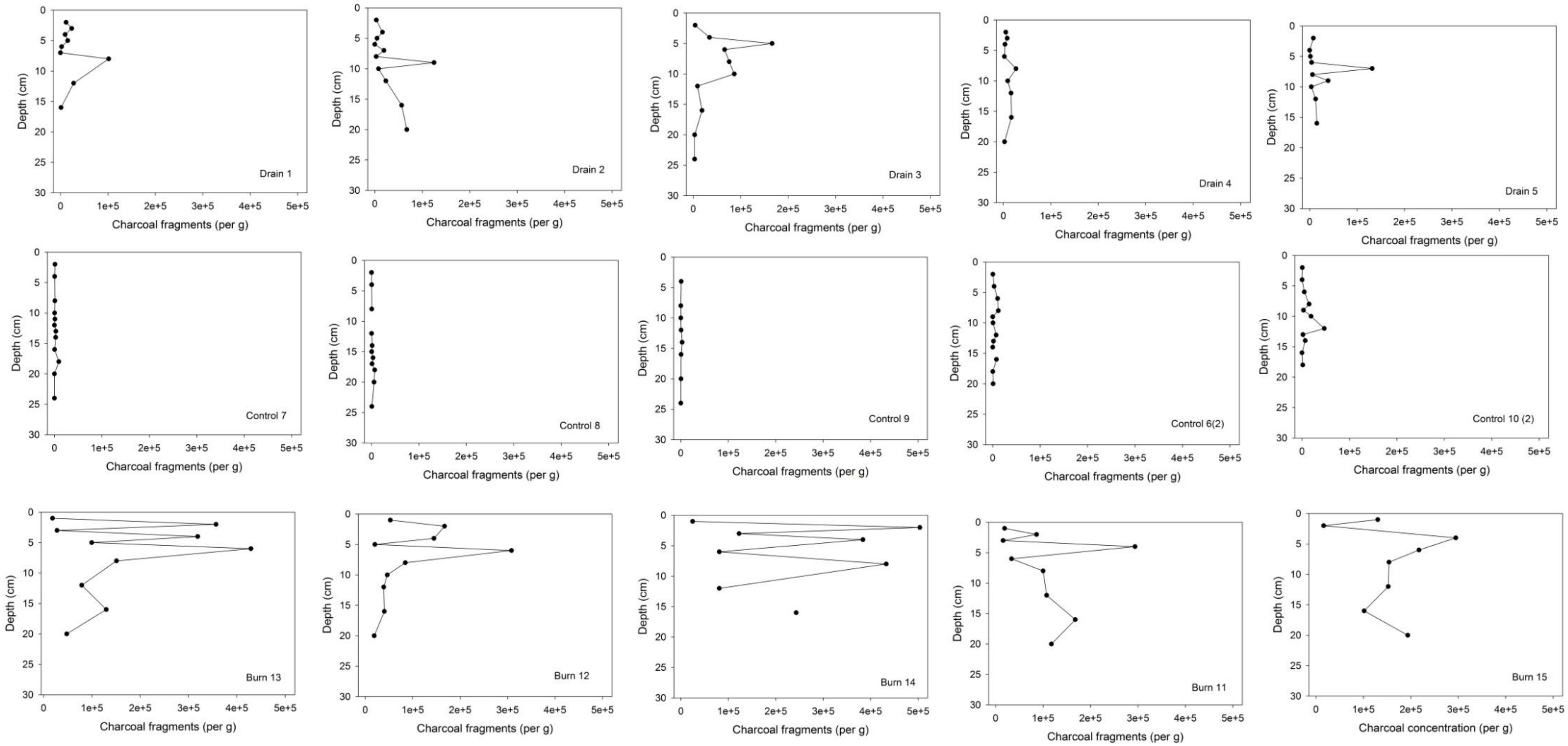

Figure 6.11 Charcoal concentration profiles in each core

6.5 Discussion:

6.5.1 Drainage

The lowering of water tables as a result of drainage could potentially cause a peatland to switch from a carbon sink to a carbon source, by altering a number of natural peatland processes. If drainage is successful in sufficiently reducing water tables increased levels of oxygen and the resulting deeper acrotelm would result in higher rates of carbon mineralisation (Holden *et al,* 2004) and CO_2 respiration (Komulainen *et al*, 1999) causing increased losses of carbon accumulated. Drainage, where effective, has also been shown to alter vegetation community composition away from peat accumulating species such as *Sphagnum* and towards vascular species (Coulson *et al*, 1990; Komulainen *et al*, 1999). Changes in vegetation towards vascular species have been linked with enhanced decay rates (Clymo, 1984) and increased CO_2 flux (Ward *et al*, 2007) and ultimately may be linked with reduced carbon retention. Additionally, accelerated DOC production has been associated with the lowering of water tables (Freeman *et al,* 2001a) and thus enhanced loss of carbon accumulated. The combination of each of these factors is anticipated to also cause a reduction in carbon accumulation rate as a result of drainage.

However, in this study drainage had no impact upon carbon accumulation rates. This demonstrates that drainage activity does not always have a negative impact upon carbon accumulation as is commonly expected. It is important to put this result into context in order to understand how this finding should impact upon policy and land management practices. As a result, this discussion aims to understand why no difference in carbon accumulation occurred, how representative this result is, and how much this might relate to overall carbon budgets.

Drainage efficiency

The efficiency of an artificial drainage network in reducing the water table level, and consequently carbon accumulation, is greatly affected by its surrounding topographic and hydrological characteristics (Holden *et al*, 2004). As a result, artificial drainage systems which have been cut across large areas of blanket peatland, such as that at Blackbrook Head are not all of similar efficiency (Holden *et al*, 2004). The effectiveness of individual drainage channels is often related to the natural hydrological dynamics within the catchment (Holden *et al,* 2006a) in combination with the orientation, spacing and age of the drainage (Holden *et al*, 2004). Consequently, it is important to characterise the drainage system in which the samples were taken from, to understand how these findings may relate to carbon accumulation in other drained catchments.

In the natural system prior to drainage there would have been a small hydraulic gradient which would have caused a westerly flow direction (Figure 6.10). Like many drained peatland catchments in the UK, drainage was cut running parallel to contour lines. This orientation of cutting aimed to reduce the saturation of peat by intercepting upland flows into the drain (Holden *et al*, 2006a). This orientation is maintained throughout much of the Blackbrook Head drainage system. Holden *et al* (2007b) identified that drainage sediment production and recovery rates were closely related to slope, finding that slopes of less than 4 degrees were more able to naturally infill and recover. Similarly, Holden *et al* (2006a) suggests that areas such as the one studied, with a dense ditch network, but shallow slope are perhaps least effective in reducing water table levels. The area surrounding where the samples were taken from fits these characteristics. Although the drainage aimed to intercept flow and lower water tables, it is likely that it was not greatly effective and consequently carbon accumulation remained unaffected before the drain began to recover. The topographic characteristics surrounding the site would not have enabled the drainage system to be very effective as slopes were low in the immediate area and along each of the drainage channels (see Figure 6.10) and the ditch itself is located in an upper area of the slope where not much natural flow would accumulate.

Much of the Blackbrook Head drainage site is an area of similarly low slope with drainage ditches running perpendicular to flow direction (see Figure 6.10). Comparable conditions to the recovering site sampled. However there are a number of sites, namely in the south and north east (Figure 6.10), where slopes are steeper and flow runs parallel to the drainage ditches; it maybe in these areas that drainage is more effective. Recovery is not as evident in these sites and some are still open drainage systems. These sites were not investigated, as they did not match the control and degraded site hydrological and topographical characteristics and were subject to restoration works which may have impacted water table readings. Further research is needed to understand if carbon accumulation rates vary in areas of differing topographic and drainage characteristics.

Local drainage efficiency

Although, as a whole, drainage caused no reduction in carbon accumulation, there is some variability in carbon accumulation levels locally in the drained site (Table 6.6). Research has shown that drainage, at a local scale, may have variable effects upon water table draw down and as a result this may be a cause of the inconsistency. Work by Boelter (1972) and Dunn and Mackey (1996) found that the water table lowered most at the edge of the drain and recovered with increasing distance. More specifically Stewart and Lance (1991), who studied a UK drainage site, found drainage reduced water table level and caused fluctuation a only a few meters from the edge of the drain, and after this point water table levels were found to recover. Whilst, Holden *et al* (2006a) found that the impact of drainage on local water table level was largely controlled by the topographic characteristics of the area, with greater water table lowering immediately down slope of a drain and the water table beginning to recover at approximately 2m before the next drain. Holden *et al* (2006a) suggests that this pattern only applies on sloped areas and that the findings of Stewart and Lance (1991) may hold true for areas of flat peat.

Cores in the drained site were taken from a transect between two drains (Figure 6.5). Given the slight slope between the drains running from 12m to 0m it might be expected that carbon accumulation is reduced to the greatest extent from 12m and increases towards the down slope drain as a result of a long term reduction of water table, reflecting the findings of Holden *et al* (2006a). This pattern is not evident in Figure 6.9. As the slope is only shallow the effect observed by Stewart and Lance (1991) may instead have taken place. Again, no pattern is evident where carbon accumulation is reduced closer to the drain edges. As a result it cannot be concluded that drainage has had any effect at the local scale variability in accumulation observed. Instead this variability can be assumed to be natural, similar to that of the control cores (Table 6.6). Water table levels in transect two reflect no relationship with drainage channel position, whilst transect one represents a lowering of water table down slope (towards 0m) (see Figure 6.4). This does not follow the trend of Holden *et al* (2006a) but may reflect to the local effect of the down slope drain at 0m. However this lowering of water table is not reflected in the carbon accumulation rates calculated in its respective core (Figure 6.9 and Figure 6.4), indicating that the water table lowering on this side of the drain was not enough to affect carbon accumulation.

Drainage processes and local change

Although the carbon accumulation rate at the drained site was not different to the control site, bulk densities were significantly higher in the drainage site. This can be interpreted in terms of the processes which may have occurred in the local area surrounding the coring site after drainage, assuming the bulk density difference is as a result of the drainage activity. Holden *et al* (2004) states that the removal of water from peat removes support from the peat matrix, which then causes consolidation leading bulk densities to increase. The elevated bulk density therefore may suggest that there was an initial water table draw down following drainage. The consolidation of the peat will have caused the peat surface to fall and could have brought the new water table level closer to the surface (Lindsay, 2010). Although there is a significant difference between control and drainage water table levels, this is not great, only around 10 cm (see Figure 6.3). Consolidation of the peat in combination with gradual recovery of drainage channels could be an explanation for why water table differences are not larger. As there is no difference in carbon accumulation levels between the control and the drained site it is likely that only consolidation occurred following drainage, with little, if any, reduction in carbon accumulation. The restriction in water table draw down, due to consolidation causing a lowering of the peat surface and subsequent recovery, may never have been enough to deepen the aerobic acrotelm and cause increased carbon loss at this site. Additionally, the water table may have been close enough to the surface to maintain peat forming vegetation therefore maintaining accumulation rates. Although, in this circumstance, drainage has not reduced the water table level enough to impact carbon accumulation rates, there may be a threshold where topographic and drainage circumstances cause water table levels to decline rapidly and consolidation is no longer able to compensate for this. Thresholds for loss of carbon and substantial changes in vegetation may occur at this point. This again highlights the need for further research into carbon accumulation responses in different drainage circumstances.

Rowson *et al* (2010) carried out a complete carbon budget from a drained peatland in the UK and found that the site was a losing carbon as a whole. However, the breakdown of components of the carbon budget revealed that the exchange of CO_2 was negative (i.e. a net sink). Although this exchange was low in comparison to other elements of the carbon budget (Rowson *et al*, 2010), it supports the observations at Blackbrook Head of continued accumulation of peat following drainage. Rowson *et al* (2010) showed that Particulate Organic Carbon (POC) and Dissolved Organic Carbon

(DOC) were the greatest release of carbon from the system. The importance of POC and DOC loss is also highlighted in Holden *et al* (2007b) who found high sediment production in a number of drains. Carbon accumulation techniques cannot account for POC losses and it is important to consider this factor when interpreting these results. However, as flow may never have been greatly enhanced directly in these channels, it is less likely that losses of POC and DOC were ever greatly increased in this site.

Due to the processes of peatland function this site has retained a pre-drainage level of carbon accumulation. This demonstrates that not all drainage has a negative impact upon carbon sequestration. Drainage is not always effective in reducing water tables sufficiently (Holden *et al*, 2004) and this is the possible cause of the maintenance of accumulation rate. However, other studies have shown drainage impacts upon carbon dynamics (Rowson *et al,* 2010), sediment production (Holden *et al*, 2007b) and hydrology (Holden *et al*, 2006a). As a result, this finding cannot be considered a definitive answer to the carbon response to drainage. The results presented here indicate that the impacts of drainage on carbon sequestration are not always irreversible without human intervention and suitable sites for remediation should be carefully targeted with consideration of these findings; a similar conclusion is also drawn in (Holden *et al.,*2004; 2007b). Further investigation should be carried out into carbon accumulation response in drainage of different characteristics and topographical circumstance to establish if and when a threshold is reached where carbon accumulation is reduced.

6.5.2 Degradation

Degradation considerably alters peatland hydrology and ecology and both of which are variables that determine the rate of accumulation and storage of carbon in peatlands (see chapter 2). However, little is known about how degradation impacts upon peatland accumulation and carbon dynamics. In this study it was found that degradation reduced the rate of carbon accumulation by approximately half. Little is currently understood about the total carbon response of blanket peatland to degradation, these findings can be used to better understand why carbon accumulation was reduced significantly on degraded sites. Additionally, there is evidence to suggest that Black Hill has been subject to considerable burning in the past 50 years (see charcoal records in figure 6.11); this may be an additional cause for reduction in accumulation in recent years and therefore will be given consideration within the discussion. Many areas of blanket peatland are found in similar hagged and desiccated conditions as those of Black Hill and therefore these results are valuable in informing management of areas in a similar condition.

Changes to vegetation and hydrology

Carbon dynamics in peatlands are significantly affected by vegetation community structure (Ward *et al*, 2007). Therefore, the changes in vegetation composition initiated by degradation could impact upon carbon dynamics. Degraded areas, particularly those which have been heavily burnt, commonly have increased prevalence of gramminoid species and reduction in bryophytes such as *Sphagnum* (Ward *et al*, 2007; Hobbs, 1984). Species such as *Sphagnum* are more resistant to breakdown in the acrotelm than graminoids (Blodau, 2002) and alteration in CO_2 fluxes have been observed at a site with graminoid prevalence (Ward *et al*, 2007). Black Hill has a dominance of graminoid species (see section 6.2). As a result, alteration in vegetation could be a cause of the reduction in carbon retention and accumulation within the peat on Black Hill. Additionally, sites which have been subjected to burning, such as Black Hill as suggested by recent charcoal records (figure 6.11) may also have restricted Net Primary Productivity (Worrall *et al*, 2009b). Farage *et al* (2009) showed that a considerable quantity of above ground biomass was lost to fires and Ward *et al* (2007) found that the resulting vegetation stored 60% less carbon than an unburnt plot.

Combustion of surface vegetation, and reduction in NPP following the burn, could be an additional cause for the reduction in rate of carbon accumulation at Black Hill in the past 50 years, in addition to the degradation occurring at the site.

The degraded site in this study is surrounded by hagged and desiccated peat and although the water table results are incomplete, they suggest the degraded site is drier than the drained and control sites (Figure 6.3). Tallis (1987) and Maltby *et al* (1990) have both suggested large wildfires, such as those recorded at Black Hill, as the potential cause of hagging and gullies. Although, it is not possible to attribute the degradation found on Black Hill directly to the high level of burning in the past 50 years as indicted by the charcoal records of figure 6.11, it is likely that these burns, at the very least, exacerbated this problem. Geomorphological features, such as hags and gullies and changes in the physical structure of the peat at the site are a possible cause of the reduced water table. Reduced water table levels will cause increased mineralisation of carbon. This would have resulted carbon previously accumulated being lost, again accounting for the reduced accumulation rate found at the degraded site.

Fire type

As discussed, recent charcoal records and reports suggest that the degraded site at Black Hill has had several large burns in the last 50 years (figure 6.11). These fires may have contributed towards the reduced carbon accumulation level at Black Hill in addition to the degraded nature of the site. However, as with drainage, the response of carbon accumulation to burning may depend upon a number of factors besides whether a site is burned or unburned. No two fires are the same; some fires may alter or disrupt an area more than others (Davies *et al*, 2008) and may therefore have differing carbon accumulation response. Also, the return time of fire will influence the ability of the environment to withstand the consequences of fire and have an additional impact upon carbon accumulation rate (Kuhry, 1994). Fires on moorland environments fall into the categories of planned and unplanned fires. Planned fires are prescribed to an environment by the land managers and aim to produce agricultural benefit. These fires are controlled in size, severity and return time. Unplanned fires occur as a result of wildfire or arson, and their size, severity and return times are not controlled. The different characteristics of planned and unplanned fires may result in varying impacts upon carbon accumulation. In addition, the topographic situation of a fire may have an impact upon the peatland response to burning. For example, burning on a steep slope may initiate erosion and have longer recovery times than a burn on a shallow slope (Maltby *et al*, 1990), consequently causing lower carbon accumulation rates. As a result, it is clear that the causes of the reduced accumulation found at Black Hill cannot be simply defined and caution should be applied when using these results to inform management in other degraded and burned sites.

Unlike drainage, other data are available for a site which has been recently burnt, Garnett *et al* (2000) carried out an investigation into the impact of planned burning at the Hard Hill experimental sites in the North Pennines. The Hard Hill experimental plots occur in controlled conditions and are located on intact blanket bog, therefore reduction in accumulation can be associated with burning with greater certainty. Carbon accumulation was significantly reduced in both studies to similar levels; Garnett *et al* (2000) found a reduction in carbon accumulation of 43% in a planned burn site and in this study carbon accumulation was reduced by 48% as a result of degradation and unplanned burning at Black Hill (in comparison to their respective control plots). These sites can be considered a valid comparison, as the Hard Hill plots are largely similar to Black Hill in characteristics which influence peat accumulation rate (see Garnett *et al*, 2000 and Table 6.1), except for mean annual temperature, which is 2.3^{0}C lower at the Hard Hill plots. Increased temperatures cause greater soil respiration and mineralisation rates (Blodau, 2002) and may be an additional cause of the slightly greater losses of carbon observed on Dartmoor but seems unlikely to make a significant difference. The results of Garnett *et al* (2000) and this study indicate that prescribed burning, and degradation with recent unplanned burning have had a similar impact on carbon accumulation rates.

Other carbon losses

Another major consequence of degradation is the potential loss of DOC and POC via fluvial pathways, as a result of gully networks being initiated. As with carbon accumulation rates, losses via erosion may vary according to the extent and type of degradation. Consequently, the relative importance of change in carbon accumulation rate to the overall carbon budget may vary between settings, depending on whether erosion has or has not occurred. When considering the consequences of degradation upon carbon dynamics losses via fluvial and erosion pathways must also be taken into account.

The results from Black Hill indicate that degradation may have had a considerable impact upon carbon accumulation rates across the UK. However, further work must be carried out considering the full impact of degradation including erosion rates, characteristics and recovery rates. Despite these caveats, the evidence available indicates that degradation can be considered a significant threat to peatland carbon dynamics.

Measuring short term accumulation

As discussed in the introduction, various methodologies can be used to monitor carbon dynamics within a peatland. It is therefore essential to put what has been measured in this study into context to allow these findings to be interpreted correctly in the bigger picture. The short term accumulation of peat has been calculated in this study (the RERCA, see Chapter 2, section 2.2), which considers the balance between production and decay in the acrotelm. The acrotelm supplies the catotelm with carbon, therefore this carbon will be subject to further slow decay in the catotelm and should be considered ‘unsafe’ as a store (Belyea and Clymo, 2001). As a result RERCA calculations such as these do not account for the total quantity of carbon lost throughout the full profile (the LORCA) and RERCA short term calculations cannot be directly related to a full carbon budget (such as that of Rowson *et al*, 2010), although they may approximate net ecosystem CO_2 exchange. Instead this methodology represents whether a peatland is still actively accumulating carbon and allows comparisons between the relative rates of carbon accumulation between locations. The ultimate fate of this carbon is unknown, but can be predicted. Approaches to overall estimates of carbon accumulation based on RERCA will be dealt with in greater detail in Chapter 7.

There are two other important points to consider in the use of RERCA and benchmark dates to compare the individual responses of management types. First, the nature of management will cause the vegetative composition of a site to alter (such as Ward *et al*, 2007). Vascular species are more easily decomposed than species such as S*phagnum and a*lthough this difference will be represented in the difference between RERCA carbon accumulation values between management sites, the preferential decomposition of vascular species will may continue in the catotelm, further reducing the accumulation potential of sites with vascular species prevalence. Secondly, as discussed, RERCA carbon accumulation calculations only account for losses in the acrotelm. This study uses benchmark dates to compare between management types, the water table levels presented in section 6.2.1 suggest that the depth of the acrotelm maybe deeper in the degraded site than the control and drained sites (due to a deeper maximum water table level). Therefore elevated acrotelm decay rates may occur for longer periods of time in the burned site with a deeper acrotelm than the control site. As a result of the problems associated with this technique, the reduction in carbon accumulation caused by burning should be seen as a conservative estimate. Despite this, the use of the RERCA methodology provides a useful insight into the comparative impact of different management activities.

6.6 Conclusion

This chapter has discussed the impact of drainage and degradation upon carbon accumulation of peat in two blanket peat sites in Northern Dartmoor and compared these to a control site. It found that degradation had an impact upon carbon accumulation, but drainage did not. In the discussion, it was highlighted that responses to management could be variable according to environmental setting and characteristics of management occurring. In this study, the drainage occurring on the site could be considered relatively small-scale, with low slopes and potentially with ineffective drainage positioning, whilst the degraded site could be considered heavily damaged, with evidence for recent large unplanned fires and desiccated and hagged peat. However, conditions at the degraded and drained sites should be taken into account, as it is difficult to claim that these sites typify drained and degraded sites throughout the UK. Further work could investigate the potential variability of carbon accumulation response under different management characteristics and environmental settings. Also, in this study each site was discussed as though only factor was impacting upon accumulation. In reality, on these sites and in moorlands across the UK, multiple activities occur simultaneously and may vary over time. For example, all of the sites on in this study will have received some grazing management in addition to the degradation and drainage specified. Further work on moorland response to multiple management situations, such as a drained site with heavy grazing, would be beneficial for providing realistic scenarios for land managers to use. The Hard Hill plots on Moorhouse NNR are able to cover this to some degree as some plots have multiple management treatments, but do not consider drainage as an activity. To date, most research in the UK into the impact of management upon blanket peatland carbon dynamics has focused on the Hard Hill plots. Although this study on Dartmoor cannot provide the rigor of Hard Hill's experimental design, it does provide an insight site into the response of UK peatlands to management and change in condition in other areas of the UK.

7 Scenario Planning

7.1 Introduction

The role peatlands play in the global carbon cycle and climate change is now recognised by policy makers, land managers and scientists. All of which, have a common aim to allow peatlands to retain their ability to sequester carbon and retain stocks already accumulated. However, global agreements, such as the United Nations Framework Convention on Climate Change (UNFCC), which aim to reduce greenhouse gas emissions, have previously given very little direct consideration to peatlands (Joosten, 2010). As a result, peatlands threatened, yet highly valuable, store of carbon was largely omitted from some of the most powerful legislation. However, following the Conference of Parties (CoP) 16 in Cancún, Mexico, an agreement to consider including wetland 'rewetting' in Land Use Land Use Change and Forestry (LULUCF) section for Annex I parties was made (UNFCCC, 2010). Although this agreement currently only considers 'rewetting' of peatlands, this step forward may lead to peatlands taking a more prominent place in international climate change policy. It is hoped that this inclusion will filter down into national and local projects and policies aimed at protecting peatland carbon. In order for this to be achieved transfer of knowledge from scientist to policy maker must be made.

A number of local, national and European funded projects with the aim of protecting blanket peatlands are already active. For example, Natural England's 'Ecosystem Services Pilot Project' aims to provide economic valuation of upland ecosystem services in the South West and will incorporate upland peat carbon storage and sequestration (Traill-Thomson and Bloomfield, 2009). Commitments to carbon protection agreed at an international, national and local level must be achieved by applying best estimates using current science. It is therefore necessary for current understanding of peatland dynamics and response to climate change to be used to model potential future scenarios to consider what may happen to peatland carbon stores.

The aim of this PhD has been to develop new datasets and insights to inform and assist actions to increase carbon sequestration and protection. It has done this through the creation of a map of carbon distribution and by assessing the carbon accumulation response to different land management practices and peatland conditions on Dartmoor. Although there is still a large amount of uncertainty and limitations within these findings, these datasets can be used, in combination with previously available understanding, to identify the likely changes in peatland carbon under a number of different scenarios. The intention of this chapter is not to state with certainty the outcomes of these scenarios, but to illustrate how data from the PhD could be used to inform management of upland environments. Assumptions and sources of uncertainty will be identified during the chapter where appropriate and caution should be used in interpreting the results, especially the absolute figures on changes in carbon.

7.2 Scenarios

Two scenarios were developed to make spatially explicit projections of carbon sequestration under future management and peatland condition scenarios across Dartmoor, these include:

a) Calculating the carbon accumulation across Dartmoor to 2100 assuming current patterns of degraded and 'control' condition peat. Burning maps will be used as a theoretical guide for degraded areas of Dartmoor in the absence of other datasets.
b) No degradation has occurred on Dartmoor, all sites reflect the conditions found on the control site of Chapter 6.

Each scenario involves a number of assumptions, some of which are generic for all scenarios and some of which are relevant to specific scenarios.

7.3 General methodology

Carbon accumulation rates were applied spatially across Dartmoor to calculate total carbon accumulation under each scenario. The calculations use data from the previous chapters relating to blanket peat distribution and accumulation rates under control and degraded scenarios. Additional sources of data were UKCP09 scenarios (Murphy *et al*, 2009, downloaded from www.ukclimateprojections.defra.gov.uk); National Park digitised GPS records and estimates of long term carbon accumulation (LORCA) rates that implicitly include the long-decay in deeper peat (Clymo, 1984).

7.3.1 Peatland areas

The spatial coverage for these scenarios includes all areas of blanket peat above 100cm in depth, an area of 4184 ha (Figure 7.1). This limit was applied as carbon accumulation rates in chapter 6 were calculated in peats of approximately 130 – 200cm. Carbon accumulation on peats may be lower in peats shallower than 100cm, shown by the stronger influence of topography on peat depth (see chapter 3) and, potentially, carbon accumulation rate. A major assumption is that accumulation rates in peats >100cm do not vary greatly across the area calculated, despite the argument made in chapter 6 that there may be some natural spatial variability in blanket peatland accumulation rates due to hydrological processes. Data from chapter 3 suggest that only 10% of blanket peat consists of peats above 160cm and as a result, applying accumulation rates from peats of intermediate depths should be representative of the main areas used in the scenarios here.

7.3.2 Carbon accumulation

Two forms carbon accumulation (g C m^{-2} yr^{-1}) were used to calculate total carbon accumulation spatially. Firstly, the accumulation rates at the top of each core (RERCA), by using data from Chapter 6, Table 6.5. These were applied across the spatial area and time period required by the scenario and upper and lower limits error of accumulation were also calculated. Secondly, long term decay (LORCA) was calculated as RERCA does not account for long term losses of carbon in the catotelm and it was thought that including LORCA should provide a more accurate estimate of carbon accumulation for the whole peat profile.

7.3.3 Calculating LORCA values

The rate of carbon accumulation throughout peats profile is not linear. Carbon accumulation curves are concave in shape, even if productivity and decay are constant over time (Clymo, 1984). As peat depth increases overall accumulation rates become lower as a result of the cumulative decay occurring above in the catotelm (this is visually represented in Figure 7.2). Most of the accumulated carbon within a peat profile is held within the catotelm (Clymo *et al*, 1998). The actual rates of carbon accumulation (RERCA) calculated in chapter 6 are averages for the acrotelm only and do not take into account the decay of carbon that takes place in the catotelm peat (Figure 7.2). As a result, RERCA calculated rates of accumulation over-estimate the long-term carbon sequestration rates. In consideration of this, to provide better estimates of carbon accumulation for the whole peat profile, RERCA needs to be corrected for the proportion of carbon lost in the transition of material through the acrotelm to the catotelm. One simple way of estimating this is to compare RERCA with LORCA for profiles from intact peat sites. The ratio between the average RERCA for the acrotelm and the average LORCA for the catotelm provides a first-order estimate of the proportion of carbon sequestered in the peat over longer timescales (Figure 7.2).

LORCA in the catotelm (catotelm accumulation rate) was calculated using available radiocarbon dates from blanket peat sites selected from a Dartmoor radiocarbon database (Fyfe, unpublished). The radiocarbon dates were only selected if they were basal dates, because LORCA may decrease non-linearly throughout a peat profile and therefore standardisation to ensure that each represented the ultimate point of development was needed (Clymo *et al*, 1998). The dates were additionally checked for their suitability using the original publications to avoid erroneous ages being included. It was assumed that none of the radiocarbon ages have been subjected to long term anomalous conditions, which could affect accumulation rate (e.g. long term burning) and that all are representative of LORCA in intact blanket peat similar to the control site, at least during the formation of the catotelm.

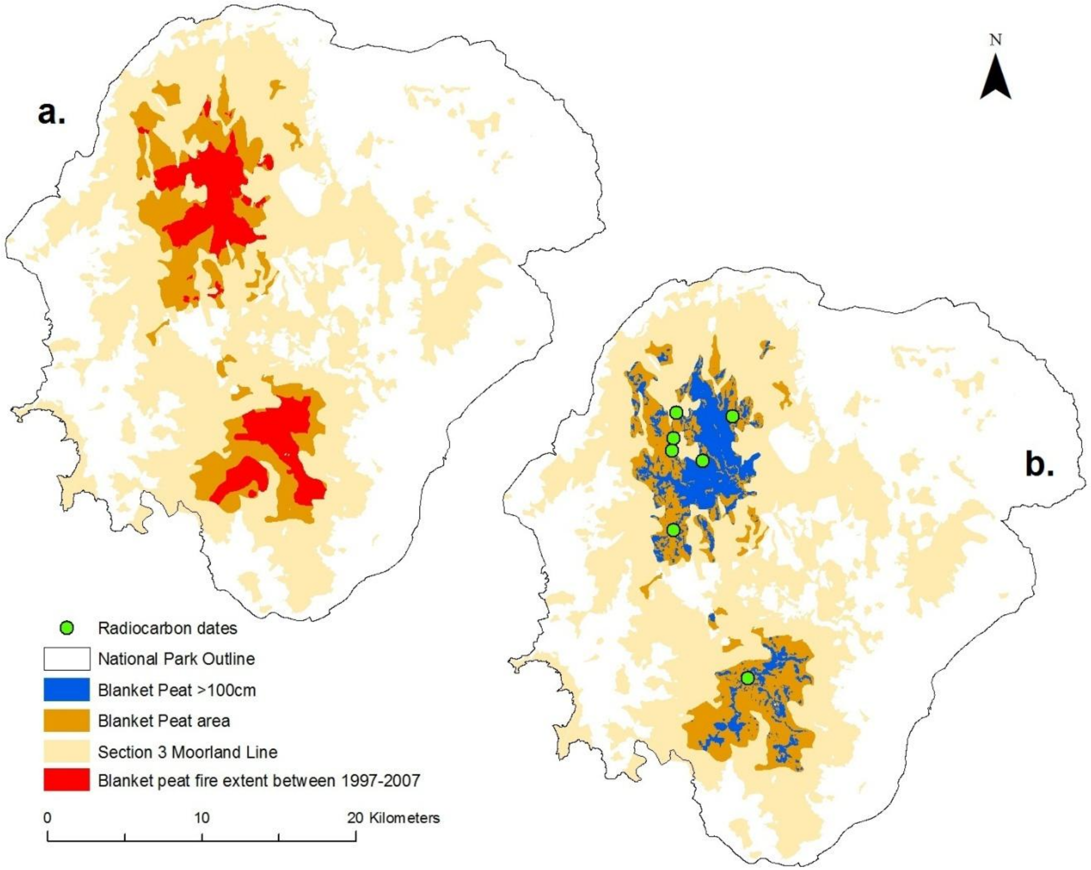

Figure 7.1 Scenario planning datasets a. Wildfire within blanket peat (area used to denote spatial extent of degraded peat) b. Peat > 100cm with location of radiocarbon cores (Fyfe, unpublished)

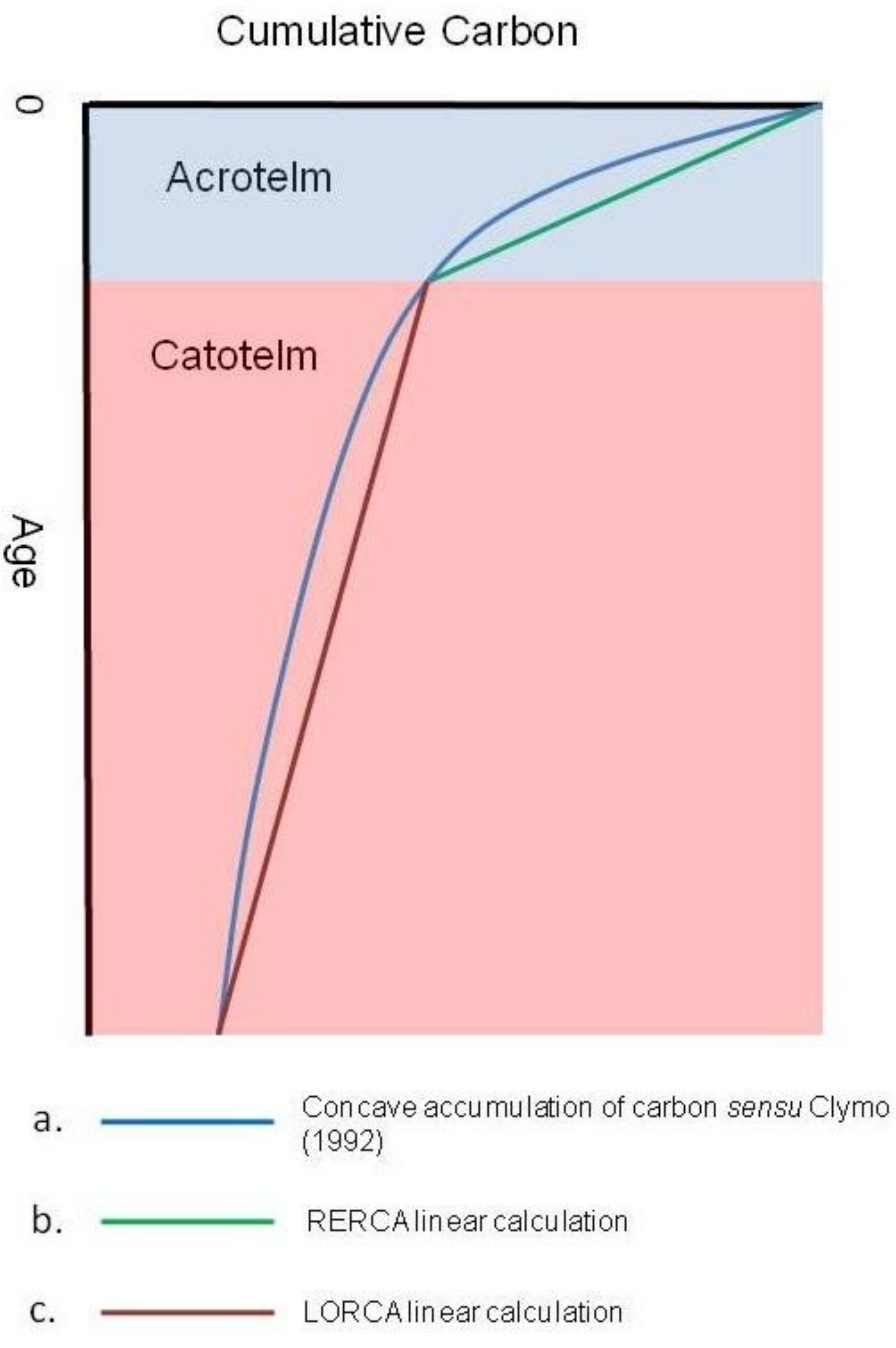

Figure 7.2 Schematic diagram demonstrating the nature of carbon accumulation in a peat profile. The difference in gradient between lines b. and c. represents mass lost as carbon is transported through the catotelm. Please note not all loss occurs at the acrotelm / catotelm boundary

To estimate catotelm LORCA the following equation was applied to each blanket peat core with a radiocarbon age:

Equation 7.1: $$Catotelm\ decay\ (g\ C\ m^2\ yr^{-1}) = \frac{(a-b)(cd)}{f-g}$$

Where a = total peat depth (cm), b= assumed acrotelm depth (cm), c= bulk density (g cm^{-3}), d= % carbon (/100), f= calibrated radiocarbon age (yr), g = assumed max acrotelm age (yr).

Bulk density and % carbon is needed to calculate LORCA (Turunen *et al*, 2002), but these data were unavailable for cores in the radiocarbon database. Instead, regression equations calculated in chapter 4 for bulk density (g cm^{-3}) and % carbon were applied to the total depth of the peat quoted for each radiocarbon core to estimate total carbon.

The maximum acrotelm age and acrotelm depth were unknown for the dated cores in Table 7.1 and therefore were estimated from the control cores (chapter 6). This assumes that the radiocarbon dated sites were similar to the control site. Acrotelm depth was estimated as the average maximum water table depth recorded in the control site (17.3cm), as the maximum water table level represents the limit of the acrotelm (Ingram, 1978). Age for the base of the acrotelm was estimated from the CRS ^{210}Pb age at the point of maximum water table depth for each core. Although these ages have errors associated with counting and ^{210}Pb mobility (chapter 5), these are the only dates available at these depths. On this basis the average acrotelm depth was estimated at 155 years.

The error associated with catotelm LORCA was calculated for each radiocarbon core, using equation 7.2. Error values were calculated using RMSE for bulk density (c*) and % carbon calculation (d^*), radiocarbon age error (f^*) and acrotelm age sampling and CRS error (g^*) and where h = catotelm LORCA of radiocarbon cores.

Equation 7.2: $$Catotelm\ decay\ error\ (g\ C\ m^2\ yr^{-1}) = \left(\frac{(a-b)((c+c^*)(d+d^*))}{(f-f^*)-(g+g^*)}\right) - h$$

For each radiocarbon dated core the loss of acrotelm carbon in the catotelm was calculated by calculating the percentage difference between catotelm and acrotelm accumulation rates (Chapter 6, Table 6.5). This calculation assumes that the radiocarbon dated cores are representative of the catotelm LORCA where the acrotelm RERCA was calculated (the control site). Although there are uncertainties with the method of estimating catotelm LORCA values, they are mostly similar across Dartmoor (Table 7.1), varying between 13.27 ± 2.32 and 27.8 ± 4 g C cm^{-3}. Calculated values of carbon lost in transferring material from acrotelm to the catotelm in Table 7.1 largely correspond with the values reviewed in Clymo (1984) of between 80 – 94%. This supports the idea that the data used are representative of long-term accumulation rates in peatlands. The average proportion lost from RERCA in transferring to the catotelm in an intact peatland was then applied to the RERCA values for the degraded site (Table 7.2). This allowed for long term losses to be estimated for ‘degraded’ areas of peatland. In doing this a number of assumptions were made. Firstly, the degraded site has a deeper acrotelm than the control site and as a result, acrotelm decay may continue for a longer period than the control site and the percentage of carbon lost may be greater. However, even within the acrotelm, decay decreases with depth as the environment gets more anoxic (Belyea, 1996) and this effect may not be large. Secondly, using control site values assumes that the catotelm has not been damaged and catotelm decay remains unchanged by degradation. As a result, the effect of changes in the catotelm, such as peat piping which may increase decay may not be accounted for. Consequently, the degraded accumulation values adjusted for catotelm decay should be seen as conservative for the characteristics of the site.

Source	Depth (cm)	Basal Age (Cal yr BP)	Estimated Bulk Density (g cm^{-3})	Estimated % Carbon	Catotelm LORCA (g C cm^{-3})	% difference between acrotelm RERCA and catotelm LORCA		
						Average RERCA value (105.2 g C cm^{-2})	Upper RERCA value (187.7 g C cm^{-2})	Lower RERCA value (84.7 g C cm^{-2})
Caseldine and Maguire (1986)	201	7905±295	0.11	50.7	13.27±2.32	87.4	92.9	84.3
Simmons *et al* (1983)	167	6623±395	0.12	50.5	14.00±2.72	86.7	92.5	83.5
Fyfe (2008)	115	2520±60	0.13	50	27.80±4.00	73.6	85.2	67.2
Caseldine and Hatton (1993)	186	5420±100	0.11	50.6	18.58±2.77	82.3	90.1	78.1
Fyfe (2007)	181	4934±86	0.12	50.6	20.09±2.95	80.9	89.3	76.3
Fyfe (2006)	102	3375±275	0.14	49.9	18.14±3.86	82.7	90.3	78.6

Table 7.1 Blanket peat cores with radiocarbon ages on Dartmoor, with catotelm LORCA values and loss of RERCA in the catotelm

.

	Acrotelm RERCA ($g\ C\ m^{-2}\ yr^{-1}$)	Long-term accumulation ($g\ C\ m^{-2}\ yr^{-1}$)
Mean	55	9.75
Maximum	75.1	16.53
Minimum	40.8	4.05

Table 7.2 Carbon accumulation rates in the degraded site, expressed as actual (RERCA) rates in the acrotelm and long-term rates, taking into account decay losses in the transfer from acrotelm to catotelm.

7.4 Scenario A

Carbon accumulation for Dartmoor's deep blanket peat (>100cm depth), if degradation extent remains the same and there is no change in carbon accumulation rate.

This scenario is designed to consider accumulation of carbon under current conditions until 2100. This involves calculating accumulation within the degraded areas using accumulation values from the degraded site (using the 2413ha of burnt areas in figure 7.1 to represent potentially degraded areas), and areas with control site conditions (1771ha) with control accumulation rates. Values using both RERCA and catotelm LORCA are calculated in ten year time periods until 2100 (Table 7.3). Total carbon sequestration under this scenario by 2100 is 55,919 tonnes of carbon using the adjusted LORCA rates and 287,141 tonnes carbon using the acrotelm accumulation only.

This scenario uses burning records recorded by Dartmoor National Park Authority to represent spatial patterns of degradation. Consequently it will not be a true representation of degradation across Dartmoor, however in the absence of peatland condition maps these records can be used as a proxy. The degraded site selected in chapter 6 was heavily damaged, with desiccation and hagging of peat. From personal observation of the condition of Dartmoor's peatland little of the area recorded as burnt in figure 7.1 is in a similar condition, so that carbon sequestration may not be as low as predicted in this scenario.

The calculation also assumes all the degraded area has a similar history to Black Hill. To achieve a similar response, other sites would have to have had the same onset of degradation, extent and condition. This is a cause of uncertainty, but to make the calculation, it must be assumed that the degraded areas across the Moor are in a similar condition as Black Hill.

Acrotelm RERCA									
Date	Degraded accumulation (2413ha) tonne C			'Control' area accumulation (1771ha) tonne C			Total accumulation (4184ha) tonne C		
	Mean	Maximum	Minimum	Mean	Maximum	Minimum	Mean	Maximum	Minimum
2020	13269	18118	9674	18635	33250	15004	31905	51368	24678
2030	26538	36237	19349	37271	66499	30008	63809	102736	49357
2040	39808	54355	29023	55906	99749	45012	95714	154104	74035
2050	53077	72474	38698	74541	132998	60016	127618	205472	98713
2060	66346	90592	48372	93177	166248	75020	159523	256840	123392
2070	79615	108711	58047	111812	199497	90024	191427	308208	148070
2080	92884	126829	67721	130447	232747	105027	223332	359576	172749
2090	106154	144948	77396	149083	265996	120031	255236	410944	197427
2100	119423	163066	87070	167718	299246	135035	287141	462312	222105
Long-term carbon loss included									
2020	2352	3989	978	3861	9214	1405	6213	13204	2383
2030	4705	7979	1956	7722	18429	2810	12426	26408	4766
2040	7057	11968	2934	11582	27643	4216	18640	39612	7150
2050	9410	15958	3912	15443	36858	5621	24853	52816	9533
2060	11762	19947	4890	19304	46072	7026	31066	66020	11916
2070	14114	23937	5868	23165	55286	8431	37279	79223	14299
2080	16467	27926	6846	27025	64501	9836	43492	92427	16683
2090	18819	31916	7824	30886	73715	11242	49705	105631	19066
2100	21171	35905	8802	34747	82930	12647	55919	118835	21449

Table 7.3 Scenario A accumulation rates with both acrotelm RERCA and adjusted long-term accumulation rates

Finally, this scenario does not consider the potential for loss of carbon via particulate sources, which are a potentially great cause of carbon loss as a result of degradation (see chapter 6). These values should only be interpreted as changes in the rate of carbon sequestration, as a result of NPP and subsequent decay, and not as a full carbon budget.

This scenario should be considered the 'worst case' due to the poor condition of the site that the data is taken from (Black Hill) and the low likelihood that the other sites have similar histories or will develop them in future.

7.5 Scenario B

No degradation has occurred on Dartmoor, all sites reflect the conditions found on the control site

In this scenario values of carbon sequestration taken from the control site for acrotelm RERCA and adjusted LORCA accumulation rates were applied across the entire extent of the blanket peat greater than 100cm depth. This scenario represents a situation past pressure on Dartmoor's peatland is minimal, allowing for maximum carbon accumulation on Dartmoor. It assumes that the control site and radiocarbon dated sequences used to calculate LORCA is representative of 'unmanaged' accumulation across the moor. Under this scenario, a total of 361,141 tonnes of carbon will have been sequestered by 2100 in the acrotelm under these conditions, reduced to 82,071 tonnes of carbon when long-term decay losses in the catotelm are taken into account (Table 7.4). The figures in Table 7.4 provide greater detail of decadal changes.

	Acrotelm RERCA (tonnes carbon)			Long-term carbon loss included (tonnes carbon)		
Year	Mean	Maximum	Minimum	Mean	Maximum	Minimum
2020	44016	78534	35438	9119	21764	3319
2030	88031	157067	70877	18238	43528	6638
2040	132047	235601	106315	27357	65292	9957
2050	176063	314135	141754	36476	87056	13276
2060	220078	392668	177192	45595	108820	16595
2070	264094	471202	212631	54714	130584	19914
2080	308110	549736	248069	63833	152348	23233
2090	352125	628269	283508	72952	174112	26552
2100	396141	706803	318946	82071	195876	29871

Table 7.4 Scenario B accumulation of carbon until 2100 with both acrotelm RERCA and catotelm adjusted long-term accumulation rates.

7.6 The consequences of degradation

Values for carbon sequestration have been calculated under two scenarios, each shows the potential future sequestration of carbon under differing management situations. By calculating the differences between scenario A and B the reduction in potential for carbon sequestration as a result of degradation can be calculated. Calculations demonstrate that scenario A, which accounts for degradation, has 32% less potential to sequester carbon than when Dartmoor's blanket peatland is in pristine condition (when average values of accumulation are used for both scenarios). Greater details about the decadal values of potential carbon storage lost out on as a result of burning Dartmoor's blanket peat are outlined in Table 7.5. Differences in mass accumulated until 2100 can be seen in Figure 7.3. This figure highlights an overlap between the two scenarios, and shows that if mean accumulation values used are not representative for one scenario then the difference between the two management types maybe more similar. This demonstrates potential error in the scenario planning method.

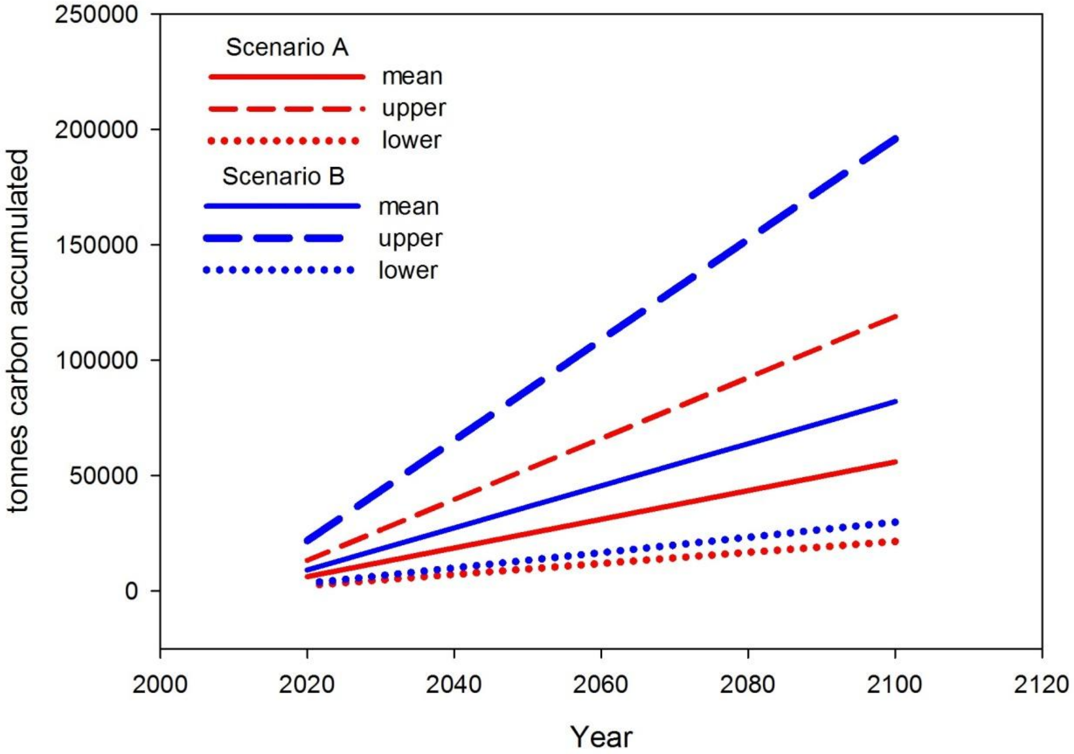

Figure 7.3 Difference in total accumulation between scenario A and scenario B (using LORCA adjusted accumulation rates)

	a. Acrotelm RERCA reduction (tonnes carbon)			b. Long-term accumulation reduction (tonnes carbon)		
Year	Mean	Upper	Lower	Mean	Upper	Lower
2020	12111	27166	10760	2906	8560	936
2030	24222	54331	21520	5812	17120	1872
2040	36333	81497	32280	8717	25680	2807
2050	48445	108663	43040	11623	34240	3743
2060	60556	135828	53801	14529	42800	4679
2070	72667	162994	64561	17435	51360	5615
2080	84778	190160	75321	20341	59920	6550
2090	96889	217325	86081	23246	68480	7486
2100	109000	244491	96841	26152	77040	8422

Table 7.5 The difference in sequestration between scenario A and scenario B. This represents the reduction in potential for carbon accumulation as a result of continued degradation.

7.7 Carbon sequestration and the carbon economy

Economic valuation is used by policy markers as a mechanism for prioritising and justifying changes in policy and land management. For example, the findings of Natural England's ecosystem services pilot project will be applied to form an economic valuation of upland ecosystem services (Traill-Thomson and Bloomfield, 2009). Global carbon markets are a well developed example of economic valuation of an environmental asset. Carbon prices can be used by government bodies such as Natural England to aid the economic valuation of upland ecosystem services (Natural England, 2009a). This thesis has produced values of the total quantity of carbon stored in Dartmoor's peatland and changes in carbon sequestration. These could contribute to Natural England and other policy making bodies by improving understanding of the relative value of Dartmoor's blanket peat carbon, providing that limitations and context of the dataset are considered thoroughly.

Carbon accounting must include values which are directly relevant to contemporary climate change. Consequently, only atmospherically active carbon can be considered. For peatlands this means that only sequestration rates will be valued (see chapter 2, section 2.2.3 for further discussion). Dartmoor's carbon stocks calculated in chapter 4 do have a value, but the ultimate fate of the carbon stored is unknown, and is relevant to valuation only insofar as it may react to differences in future management. Atmospherically active carbon comes in a number of forms (mainly CO_2 and CH_4), each of which have a different global warming potential (see chapter 2, section 2.2.3). To represent this carbon markets work in tonnes of CO_2 equivalent (tCO_2e); a conversion factor is applied to each component relative to the greenhouse potential of CO_2 (DECC, 2009). When a kilogram of carbon is burned completely it produces 3.67kg of CO_2 (Lyons *et al*, 1985). This conversion factor was applied to carbon stocks for all valuation purposes.

There is no single market price for carbon as there are several trading schemes, each with different and floating carbon prices. In the UK, the main market price to apply would be the value for the European Union Emission Trading Scheme (EU ETS) (Natural England, 2009a). However, currently the EU ETS does not consider emissions outside of industry (Natural England, 2009a), so EU ETS prices are not strictly directly applicable to peat or other natural managed carbon stores. The UK government publishes a shadow price of carbon, which is a proxy price calculated according to the costs of meeting abatement targets and the social cost of carbon, and is designed to provide standardisation throughout government departments (DECC, 2009). This can be used on a non-industrial basis (DECC, 2009), and as a result can be used for upland carbon emissions. This price is used by Natural England (2009a). The prices from DECC (2009) are used to value carbon sequestration in this study are outlined in

Table 7.6. Linear interpolation can be used to estimate prices between 2010 and 2050 (DECC, 2009).

Date	Mean price	Lower price	Upper price
2020 (non traded)	£60	£30	£90
2030	£70	£35	£105
2050	£200	£100	£300

Table 7.6 DECC (2009) prices of carbon per tCO_2e

7.7.1 Value of carbon remaining in the atmosphere

As discussed in section 7.6, a considerable amount of CO_2 will remain in the atmosphere instead of Dartmoor's peatland as a result of scenario A. An estimate of the value of this carbon using the shadow price can be used to compare the value of lost sequestration potential against other policy activities where a similar economic valuation is applied. Values were only calculated for the adjusted LORCA figures, as despite greater potential error, this value is more representative of the actual carbon balance over the whole peat profile and avoids false accounting. Values were calculated for the area of Dartmoor's peat over 100cm, using section b of Table 7.5. The amount of carbon not sequestered as a result of degradation was calculated annually as 290.6 tonnes carbon (93.6 tonnes lower and 856 tonnes upper) and when converted annually this equates to 1066.4 tCO_2e (343.4 tCO_2e lower and 3141.5 tCO_2e upper). The economic value of the carbon not sequestered annually is presented in Table 7.7, using the pricing in Table 7.6. Upper valuations were calculated from the upper estimates of lost sequestration potential and lower valuations were calculated from lower estimates of lost sequestration potential, to produce the broadest range of prices which could apply.

Year	Mean value	Lower value	Upper value
2020	£63,984	£10,303	£282,738
2030	£74,650	£12,020	£329,861
2040	£143,968	£29,191	£636,161
2050	£213,285	£34,343	£942,461

Table 7.7 Annual value of lost potential sequestration as a result of degradation (using scenario A)

Additionally the cumulative values lost as a result of degradation in scenario A were calculated until 2050, using interpolation of the values in Table 7.6 (it was assumed that relationships were linear between 2010 – 2030 and 2030 – 2050). The total values of CO_2 not sequestered as a result of burning in scenario A was £4,292,369 with a range of £694,585 to £18,935,608.

The above values are totals based on the current account for the spatial patterns across Dartmoor, but for management purposes, it is useful to know the cost of degradation to individual units of land. The annual per hectare value of carbon not sequestered due to degradation is presented in Table 7.8, assuming that the degradation site is similar over the long term to Black Hill (see section 6.2, Chapter 6). However, correction using these values should not be applied if restoration strategies are applied, as they assume that the site will immediately return to a ‘control’ accumulation rate, which will not happen. Instead this should be seen as the reduced potential of a degraded site than a ‘control’ site in the same time period.

Year	Mean value	Lower value	Upper value
2010	£22.10	£3.60	£97.70
2020	£26.50	£4.30	£117.20
2030	£30.90	£5.00	£136.70
2040	£59.70	£12.10	£263.70
2050	£88.40	£14.20	£390.60

Table 7.8 Value of carbon per hectare remaining in the atmosphere as a result of scenario A.

7.7.2 Value of carbon sequestered

Dartmoor's peatland above 100cm depth is still accumulating carbon in both scenarios and this accumulation can also be valued. The total value of carbon sequestered annually under scenarios A and B is presented in Table 7.9. This represents the value of CO_2 being sequestered by Dartmoor's peatland from the atmosphere, assuming all assumptions in section 7.3 are correct. The cumulative value of carbon sequestration on Dartmoor until 2050 under scenario A is £6,544,272 with a range of £1,255,126 to £20,861,324. Under scenario B the cumulative value is calculated as £13,470,313 with a range of £2,463,569 to £48,143,822. The upper limit of scenario B represents the maximum possible value of Dartmoor's carbon sequestration on deep peat areas from now until 2050 assuming costing, sequestration and spatial information is correct. The value of sequestration annually per unit area may be useful to know for accounting in individual units of land, such as an area of a common, these figures are presented in Table 7.10.

Scenario A				Scenario B		
	Mean	Upper	Lower	Mean	Upper	Lower
2010	£114,012	£363,438	£21,866	£167,333	£599,053	£30,452
2020	£136,814	£436,125	£26,240	£200,800	£718,863	£36,542
2030	£159,616	£508,813	£30,613	£234,266	£838,674	£42,633
2040	£182,419	£581,500	£34,986	£451,799	£1,613,449	£82,830
2050	£205,221	£654,188	£39,359	£669,332	£2,392,218	£122,417

Table 7.9 Annual value of carbon accumulated in each scenario

Date	Degraded			Control		
	Mean	Upper	Lower	Mean	Upper	Lower
2020	£21	£55	£4	£48	£172	£9
2030	£25	£64	£5	£56	£200	£10
2040	£48	£123	£13	£108	£388	£25
2050	£72	£182	£15	£160	£573	£29

Table 7.10 Value of carbon sequestration per hectare for degraded and ‘control’ condition sites

7.8 The future impact of climate change and increased degradation on Dartmoor

Past degradation of blanket peatlands has been associated with changes in the climate (Tallis, 1987). This degradation is caused by either change in factors such as the water balance, alteration in plant phenotypes or increased occurrence of wildfire as a result of drought conditions (Albertson *et al*, 2009). This change potentially can greatly influence carbon sequestration rates and accordingly there is now concern that climate change may be a significant threat to British Blanket Peatlands (McMorrow *et al*, 2009). For example, Yallop *et al* (2006) suggested that the extent of burning is already increasing in the uplands, although the cause of this may not be related to solely climate change.

In combination with changes in other drivers, such as policy and land ownership, the climate change threat can either be enhanced or controlled. It may be useful to model to the potential future impact on climate change upon British Blanket Peatland, allowing for anticipation of this threat. This may involve following methodologies such as Albertson *et al* (2010) who applied long term records of wildfire incidence in the Peak District to model increases in wildfire under the UK Climate Projections 2009 (Murphy *et al*, 2009). If understanding of carbon response to management and degradation is increased, models such as Albertson *et al* (2010) could be applied to calculate changes in accumulation rates. It is currently not possible to precisely calculate how carbon sequestration may be altered in the future as a result of climate change. However, scenario A calculates carbon accumulation using data from a site that is heavily damaged. Although not all of Dartmoor's peatland is currently in this condition, this scenario becomes more likely under the influence of climate change. Further research to better identify the impact of degradation on carbon sequestration would be a significant improvement in understanding the impact of management on future carbon balance. Even in the absence of specific knowledge, awareness raising and preparation for the potential increase in degradation on Dartmoor should be considered.

7.9 Conclusion

This chapter demonstrates how the data and understanding of peatland carbon accumulation gathered in the field and lab can be applied practically to develop policy through financial mechanisms, such as carbon accounting, to aid environmental decision-making. It also demonstrates that Dartmoor is able to sequester considerable quantities of carbon and shows peatland condition can have a significant impact upon this potential. Carbon sequestration is a valuable ecosystem service and care should be taken when considering management regimes for blanket peatlands. However, these scenarios are intended to provide an indication about of how such data may be used and they do not necessarily provide accurate predictions of actual change. A number of assumptions are applied which do not allow consideration of the spatial heterogeneity of blanket peatlands and the variation within individual management practices. Caution should therefore be applied and these findings should be used as in indication rather than a precise quantification of changes to carbon balance. Nor does economic valuation allow credit to be applied to peatlands for the longevity of the carbon stored once 'safe' within the catotelm. Despite these caveats, there is great potential for techniques such as this to be built upon in the future and for data such as this to be applied in economic valuation of ecosystem services.

8 Conclusion

The aim of this chapter is to provide a synthesis and conclusion to this thesis. It will discuss the outcomes, limitations and potential areas of further work. Additionally, it will suggest recommendations for the future management of Dartmoor National Park's peatland carbon store and some potential uses of the data and knowledge gathered during the course of this research.

This thesis originally set out to provide an initial insight into the landscape scale carbon resource and threats to carbon on Dartmoor, with the aim of providing an understanding which could be transferred to, and applied by, land managers on Dartmoor. The thesis was intentionally broad, as little research had previously been carried out considering Dartmoor's blanket peatland and even less was known about its carbon storage and sequestration potential. In doing so, the research aimed to provide a platform by which further research could be carried out into this under investigated peatland region.

The research was structured into three main themes:

- The peatland carbon resource (chapters three and four)
- Peatland carbon accumulation and management (chapters five and six)
- Scenario planning and the use of data (chapter seven)

It is within this framework that the primary outcomes, limitations and further work will now be discussed. Following this, the outcomes and themes which are common to all of the above sections will be considered. Finally, the implications of the research carried out in this thesis will be examined, to provide land managers and policy makers with guide for future use of the data.

8.1 The Peatland Carbon Resource

8.1.1 Outcomes

British blanket peatlands provide a wide range of ecosystems services, including the storage of considerable quantities of carbon. To manage this carbon effectively an understanding of its distribution is needed at a scale which is useful to land managers and policy makers. Previously, few methodologies existed where carbon could be mapped at a useful scale, nor was there a full understanding of the spatial variability of peatland carbon storage. This thesis set out to develop a methodology by which the distribution of peatland carbon storage could be mapped and quantified at a landscape scale (>10,000ha) using a methodology which was easily replicable in other similar blanket peatlands. Additionally, it aimed to provide a full carbon inventory for use by Dartmoor's land managers. The outcome demonstrated that carbon quantity and distribution could be successfully modelled and relationships were revealed between blanket peatland form and external topographic factors.

The study encompassed an area of 30,000ha and included all of the soils classified as 'peat' by NSRI within the moorland line of Dartmoor National Park. The components which make up a carbon inventory, namely peat depth, bulk density and carbon content (Bhatti *et* al, 2002), were each mapped individually. To do this each was measured in the field using a stratified sampling strategy, to investigate whether they could be statistically linked to previously mapped datasets, including soil type, vegetation, slope and elevation. It was found that functional relationships largely did exist and these were used to map peat depth, bulk density and carbon content. These relationships were subsequently used to successfully generate a full carbon inventory of peat soils within Dartmoor National Park.

It was found that a total of 9.7Mt (range 6.63 -12.61 Mt) of carbon was stored in the peat soils of Dartmoor, most of which was found in the blanket peat soils represented by the Crowdy 2 and Winter Hill soil series. The distribution of carbon was spatially was highly variable, meaning that the application of constant values of peat depth, bulk density and carbon content, even if representative of the whole area, would reveal little useful information for land managers. Individual maps of peat depth, bulk density and carbon content were generated, each of which is useful in its own right for providing an insight into the blanket peat landform and in allowing further analysis of the carbon inventory. This further analysis revealed that bulk density was as important as peat depth in a carbon inventory, but most carbon distribution was explained by peat depth variability. It was found that Carbon content varies little and has only a minor influence on in the inventory. Additionally, the carbon inventory allowed a validation test of the national inventory of Bradley *et al* (2005) across a large area of blanket peatland. This analysis revealed that the national inventory performed provided reasonable estimates for total soil carbon on Dartmoor as a whole, but performed poorly in estimating the spatial variability of carbon, on a cell by cell basis there were large discrepancies between the inventories corroborating the findings of Frogbrook *et al* (2009).

8.1.2 Limitations

If this methodology is to be applied appropriately in the future, improved upon, and the output is to be interpreted correctly, a number of limitations and sources of error and limitations must be made clear:

- The relationships used within for the inventory are extrapolated over large areas. Although this approach reduces the sampling required and reveals information about the likely form of a peatland, it may introduce undetectable error in areas where unaccounted other controls on carbon distribution become more dominant. An example of this would be where a previous peat landform exists, such as a deep in filled depression in underlying ground. Although it has been shown that other geostatistical methods such as kriging can account for this error when sampling is carried out at high resolution, it may not be practical to use these techniques at a landscape scale. When choosing a methodology for a carbon inventory, the scale, time available and accuracy required should all be taken into account to choose the appropriate methodology and avoid this error as much as is possible.

- Secondly, the relationships identified for this carbon inventory are largely for a blanket peatland which is in good condition. Many blanket peatlands in the UK are in a considerably more degraded state than Dartmoor. The processes of erosion, cutting and drainage may cause changes to in peat depth and bulk density which may significantly reduce the strength of relationships returned between peat depth and other parameters in comparison to Dartmoor. Before carrying out this methodology in other areas, consideration must be given to the condition of the blanket peatland and whether the blanket peatland morphology is likely to be subject to the same controls to those on Dartmoor.
- The third limitation is related to the data which that are used in the model. This takes two forms; first, representation of the controls of an inventory will only be as good as the secondary data that are used. For example, in this inventory the vegetation classifications were not specifically related to the vegetation found on a blanket bog and the NSRI soil series mapping was at a very coarse scale. Second, bulk density was found to be an important component of the carbon inventory, yet this is subject to some error in measurement for example that introduced when sampling bulk density using a Russian corer (Lindsay, 2010). Other techniques for bulk density sampling must be investigated.

8.1.3 Further Work

This methodology has been developed as part of this thesis and it is the first time, to the author’s knowledge, that such a technique has been used on a blanket peatland to develop a carbon inventory. The technique of using targeted and representative sampling of topographic indicators to model carbon stored has proved to be a success. Both during the process of developing this methodology, and from the outcomes of the model, a number of potential adaptations and routes for further work have been identified.

This carbon inventory encompassed all of the peat soils within Dartmoor's moorland line, from raw blanket peats to humic gleys. The methodology worked well for the raw blanket peat soils, but was less effective for the humic gleys and shallower peats. Despite the limitations of the approach for shallow peats and peaty soils, it was clear that the raw blanket peats were by far the largest stores of carbon. It would therefore be most worthwhile for future inventories to focus on the true blanket peat soils. This would allow greater scope for further improvements of the inventory in the most carbon dense soil unit, with the highest carbon density. Some improvements could involve the inclusion of additional controls on peat depth and bulk density within the sampling strategy. These could include aspect, plan curvature and potentially Topographic Index ($\ln(a/\tan\beta)$). Additionally, the soil unit and vegetation secondary datasets could be improved upon. There is potential for both peat depth and bulk density to be better represented through mapping Ivanov's landform units (such as in Lindsay *et al*, 1988) or through remote sensing. If these units were mapped prior to the sampling strategy being designed, the carbon inventory potentially may increase in accuracy.

In the UK, blanket peats have been shown to be one of the largest terrestrial stores of carbon and have a number of carbon storage characteristics which differ from mineral soils (such as high carbon storage throughout the whole peat profile). Despite this, the national soil carbon inventory (Bradley *et al*, 2005) considers all soil types equally throughout the UK. This treatment can cause large inaccuracies in individual cells of the national inventory for blanket peatlands. A methodology such as this, or similar to the carbon inventory presented here, could produce a more realistic map of blanket peatland and carbon distribution throughout the wider UK. Such a map would aid policy makers and land managers in decision making and would help academics in the larger scale modelling of peatland carbon.

In contrast, there is increasing demand for an understanding of peatland three dimensional carbon storage at a very fine scale, for example for use in building in to carbon budgets and carbon offsetting schemes for peatland wind farms (Nayak *et al*, 2008). At a very small scale, the underlying geology of a blanket peatland can be highly variable (Chapter 3). Although this does not affect larger scale inventories which require less accuracy, underlying variability must be accounted for at this fine scale. Conventional manual probing and Digital Elevation Models may not be able to reasonably estimate changes in depth at a high enough spatial resolution. Further investigation using geophysical techniques, such as ground penetrating radar, which are able to continuously detect underlying variability, may be an improved approach in such circumstances.

8.2 Peatland carbon accumulation and management

8.2.1 Outcomes

Anthropogenic pressure upon blanket peatlands in the UK has increased considerably since the industrial revolution. This has lead to concern that degradation and management such as drainage is causing blanket peatlands to switch from a carbon sink to a carbon source. A first step in informing management of peatlands to preserve peat and soil carbon would involve providing land managers and policy makers with greater information about how various land management practices and peatland conditions may affect carbon storage. Carbon cycling within peatlands is complex and can be recorded and budgeted for using a number of different methodologies. The two primary methodologies include contemporary carbon budgets and monitoring estimates of long- term peat accumulation. This thesis investigated the impact of degradation and drainage over a long term of several decades using the estimates of past peat accumulation technique. The sampling strategies and methodologies used provided for an analysis of the impact on carbon accumulation of each two peatland conditions (a drained site and a heavily degraded site, with a recent history of burning), and also an assessment of the dating methodologies used to calculate carbon accumulation rates; thus allowing for an understanding of the power of the methodology to be gained.

Fifteen peat cores were dated, five from each management area (control, drained and degraded) using a combination of SCP and radionuclide techniques. Previous work by Urban *et al* (1990) and Belyea and Warner (1994) had raised concerns regarding the validity of using radionuclide dating in organic soils (particularly ^{210}Pb and ^{137}Cs). However, little work had been carried out using SCPs in the south west of England. Both dating techniques produced dates, with the exception of two cores that did not contain SCPs. High levels of SCPs were recorded, datable ^{210}Pb profiles were obtained and the artificial radionuclides ^{137}Cs and ^{241}Am were detected. Analysis of the patterns produced by each dating technique showed that although some dating techniques agreed, there were a number of discrepancies between dating features. From this it was concluded that multiple dating techniques, as in this study, should be used if recent carbon accumulation rates in peat are to be calculated reliably. Additionally, the results showed a great deal of mobility in ^{137}Cs, suggesting it is not a valid as a dating technique for peat and corroborating the findings of Gerdol *et* al (1994) and Oldfield *et al* (1995). There was also found to be potential for a small degree of mobility in ^{241}Am, as something which had previously been speculated by Appleby *et al* (1991) and Oldfield *et al* (1995) but never identified. However, this mobility is not great enough to invalidate its use as a relative age marker for carbon accumulation. Finally, it was noted that the published estimates of calibrated ages for SCPs in the south west of England may be incorrect and further calibration maybe needed in the region.

The accumulation of carbon was then calculated above dated horizon features to enable a comparison of carbon accumulation rates between each of the sites. Carbon accumulation was found to be significantly reduced by degradation, but was unchanged in the drained site. It is however important to note that peat accumulation was not fully prevented by degradation and that accumulation still occurred despite the lack of species such as *Sphagnum* which are typically associated with peat accumulation. High bulk density levels in the drained site did indicate that an alteration of the physical peat structure had occurred. How representative these sites were of the impact of management on Dartmoor's blanket peatland was discussed.

8.2.2 Limitations

The purpose of chapter five was to identify the reliability of the dating techniques increasingly used to calculate recent carbon accumulation rates. In this way, Chapter five considered one of the major limitations of using recent carbon accumulation as a methodology. In doing this, the carbon accumulation rates of chapter six was based on the best possible data and calculated using a methodology which avoided as much of the error sources as possible. In addition, Chapter 6 allowed a number of limitations of the chronologies to be identified, which unavoidably would have been followed through to Chapter 6. These limitations include the subjectivity of identifying SCP ages, and issues involved in calculating annual carbon accumulation rates, such as the calibration of the south west SCP dates.

Notwithstanding the issues with dating, the major limitation of this study lies in the identification of appropriate sampling sites. Although every effort was made to verify the management history of each site, there was still a degree of uncertainty regarding these histories. For example, the degraded site was selected due to its recent high level of burning records, initially to provide an insight into carbon accumulation response to fire. However, it is uncertain at the degraded site whether the reduced carbon accumulation is as a result of recent burning events, as recorded in the charcoal record, as a result of degradation at the site unrelated to burning, or most likely a mixture of the two. Although, the sites used in this study do not have the same level of control and understanding of sites such as the Hard Hill plots at Moorhouse NNR (Garnett et al., 2000), they do have the advantage of being representative of natural management and typical conditions found on Dartmoor.

Finally it was highlighted in the discussions in Chapter 2 and Chapter 6 that RERCA represented recent carbon accumulation and not the full decay occurring within the catotelm. Also, that carbon accumulation values cannot fully account for changes in all aspects of the carbon budget, specifically POC. As a result, these data should not be used as a definitive to provide absolute estimates of changes in carbon sequestration under different management regimes. However, it is an excellent approach for estimating differences between treatments and the carbon sequestering ability of a peatland under different management scenarios, which can be used as a guide to the management activities relative impact of different management practices.

8.2.3 Further work

The technique to measure and compare carbon accumulation using multiple dating techniques has provided successful results for comparing carbon accumulation response to sites with differing management and peat conditions. However, like most current work into peatland management, this research has raised several questions as well as answering others. Opportunities for further work from this take two forms: firstly, to improve the methodologies used for assessing carbon impact in peatlands; and, secondly, to better understand the response of a peatland to different management regimes and peatland conditions.

As discussed previously, accurate dating is a critical element of calculating and assessing carbon accumulation rates. Consequently, it is very worthwhile to develop dating methodologies that are as rigorous as possible. For instance, it would be very beneficial to develop a best practice methodology for dating recent peats (such as a requirement always to use two different dating methodologies) and a standardised methodology for assessing the success of dating recent peats. This could involve using multiple dating techniques, as used within this thesis. This would ensure that all data output would represent what it was initially intended to and not the error some methodologies for dating recent peats are prone to. It is clear that dating errors need to be carefully and explicitly taken account of in estimating carbon accumulation in peatlands.

This thesis has also highlighted potential error in individual dating techniques. It was shown that the calibration for SCP dates in the south west of England maybe incorrect. This may suggest that the spatial areas for dates used in Rose and Appleby (2005) are too general. As the UK SCP dataset grows (CARBYDAT see http://www.ecrc.ucl.ac.uk/index.php/content/view/299/112/), it may be beneficial to reassess these boundaries and, if necessary, to recalibrate certain areas. ^{241}Am demonstrates considerable potential as a date marker within peat. Although the data from this thesis suggests that a certain degree of mobility may have occurred throughout the peat profile. Further investigation would be useful to provide an indication of the degree of ^{241}Am mobility and the conditions which might cause this. Finally, this thesis analysed the ^{210}Pb dataset for potential mobility. The variability between cores suggests that some mobility does occur but it is still not clear why this occurs in some situations and not others. Although it was possible to draw some conclusions regarding the validity of the dataset, using independent dating techniques and ^{210}Pb inventories, it was not able to provide a definitive answer to the degree of mobility which occurred. The use of ^{210}Pb dating as a tool still requires further investigation by using long term laboratory and field experiments to better identify the conditions and controls.

On a number of occasions in this thesis, the relative advantages and disadvantages of the contemporary carbon budget and the carbon accumulation approaches have been discussed. Both methodologies have their merits for example budgets providing a breakdown of flux and accumulation providing a long term value, but, so far, they have rarely been used in a complimentary fashion (a notable exception includes Evans and Lindsay, 2010b). The future use of both techniques in concordance with one another would provide much additional benefit to the understanding of the carbon dynamics of a peatland and the response to management activities.

The impact of the natural heterogeneity of blanket peatlands on carbon accumulation is highlighted as the major area for further research. As noted in Chapter 6 the sites chosen to represent management scenarios can only possibly represent the carbon accumulation response in a particular hydrological, management, topographical and peatland condition situations. For example, management techniques can vary widely; i.e. drainage channels can vary in size and spacing, and fire return times and intensities are highly variable. As a result, further investigation is necessary into the carbon accumulation response under differing management combinations and topographic circumstances in order for managers to identify the most appropriate response to management in a given area.

As a whole the data provided in this thesis has provided a useful insight into carbon accumulation response to a number of different management and peat condition scenarios. This research also highlighted that peatlands were capable of accumulating carbon even in degraded conditions. It has also demonstrated that dating recent peat accumulation has considerable potential as a methodology increasing understanding of peatland carbon dynamics over large spatial areas and considerable timescales.

8.3 Scenario Planning

The primary intention of the scenario planning chapter was to demonstrate ways in which the data gathered for this thesis could be used to inform future land management practices on Dartmoor. Calculations demonstrated how Dartmoor, as a whole, was able to sequester considerable quantities of carbon and that degradation may reduce this figure considerably. It then went onto investigate what the potential financial value of the carbon sequestration maybe under degraded and un-degraded scenarios when input into the government policy. In theory, this system allows policy makers and land managers to assess the impact of changing management practices against other economically assessed methodologies for management of the natural costs and benefits.

The scenario planning used within this chapter considered the impact of degradation only, as the data suggested that, in the area investigated, drainage management was unlikely to be an important influence on carbon sequestration rates. The scenario calculations used a large amount of secondary data and included some assumptions which lead to a degree of uncertainty in the outcome. However, the approach used was considered to be a reasonable approach, given that there is a growing need by policy makers and land managers to understand how management activities, and the resulting degradation may impact upon carbon sequestration, despite limited current knowledge of the detail of the processes involved.

Scenario planning displays much potential and with future improvements in sequestration will be a very useful tool for management and policy makers. This could include: using actual basal carbon dates to calculate the carbon lost as it is transferred through the catotelm; to generate a better understanding of the variability in management patterns; and widening the range of scenarios to other management activities, if they are found to effect carbon for management and climate variability. This chapter provided an introduction to the applied usage of the research in chapters three to six and provides a good basis for informing future management.

8.4 Dartmoor

There is not currently a balanced geographical spread of research into carbon in blanket peatlands in the UK. An advantage of this study is that it broadens the spatial understanding of blanket peat carbon storage and response to management. It also provides a better understanding of a region with different historical, climatic and management settings. This thesis has provided baseline information for a blanket peatland on the limits of western blanket bog formation which is threatened by climate change (see Clark *et al*, 2010). This was achieved by providing an improved understanding of the level of the peatland carbon store in on Dartmoor, its current spatial distribution, as well as and providing an assessment of the level of threat to the carbon store from management activities. This baseline data will encourage and inform future research on Dartmoor, an important and potentially indicative, yet under investigated peatland area.

8.5 Outcomes for Dartmoor's Upland Managers

In addition to advancing scientific understanding of blanket peatland carbon, this thesis aimed to produce insights that could assist the management of carbon stored within Dartmoor's blanket peat. This was achieved through the development of a detailed landscape-scale map of peatland carbon that incorporated bulk density, depth and carbon density. These maps can be used by moorland managers for a number of purposes, such as targeting areas for conservation and providing justification for remediation work. With the important caveat that land managers and policy makers must pay careful consideration of their limitations. Furthermore, the study has provided Dartmoor's managers with a direct understanding of carbon accumulation rates for specific areas of Dartmoor's blanket peatland (Black Hill, Blackbrook Head and Maiden Hill) in relation to past management practices and peatland condition. These accumulation rates have been used to calculate the potential financial value of sequestering soil carbon on Dartmoor. The results provide general estimates of carbon storage and sequestration and are the best information now available to inform decision-making. These values are indicative and are perhaps most useful in terms of comparing relative valuations rather than as attempts to provide precise evaluations. When moorland managers are deciding upon a management plan for an area, the local topographic situation, history and characteristics of management should also be taken into account in addition to these values.

At present our understanding of the carbon dynamics of blanket peatlands in the UK and their response to management is rapidly improving. This thesis has contributed to this by providing an understanding in a way that can be used practicably by land managers and improving upon previous methodologies for quantifying carbon storage and measuring carbon accumulation. There is still a long way to go before carbon management of blanket peatlands can be undertaken with any full certainty, however this thesis provides a basis upon which moorland managers can base take decisions and the beginnings of a cost-based approach to evaluating outcomes.

9 References

Aerts, R., and Ludwig, F. (1997). 'Water changes and nutritional status affect trace gas emissions from laboratory columns of peatland soils'. *Soil Biology and Biochemistry.* **29**, 1691 – 1698

Albertson, K., Aylen, J., Cavan, G., and McMorrow, J. (2009). 'Forecasting the outbreak of moorland wildfires in the English Peak Disrict'. *Journal of Environmental Management*. **90**, 2642 – 2651.

Albertson, K., Aylan, J., Cavan, G., McMorrow, J. (2010). 'Climate change and the future occurnace of moorland wildfires in the Peak District of the UK'. *Climate Research.* **45**, 105 – 118. Doi: 10.3354/cr00926

Ali, A. A., Ghaleb, B., Garneau, M., Asnong, H., and Loisel, J. (2009). 'Recent peat accumulation rates in minerotrophic peatlands of the Bay James region, Eastern Canada, inferred by ^{210}Pb and ^{137}Cs radiometric techniques'. *Applied Radiation and Isotopes*. **66**, 1350 – 1358.

Allan, S. E. (1989). *Chemical Analysis of Ecological Materials.* Blackwell Scientific Publications, Oxford

Appleby, P. G. (2001). Chronostratigraphic techniques in recent sediments. In. Last, W. M., and Smol, J. P. (eds). *Tracking environmental change using lake sediments,* *Volume 1 – Basin analysis, coring and chronological techniques*'. Kluwer Academic Publishers: London.

Appleby, P. G. (2008). 'Three decades of dating recent sediments by fallout radionuclides: a review'. *The Holocene*. **18**, 83 – 93.

Appleby, P. G., Nolan, P. J., Oldfield, F., Richardson, N., and Higgitt, S. R. (1988). '^{210}Pb dating of lake sediments and ombrotrophic peats by gamma assay'. *The Science of the Total Environment*. **69**, 157 – 177

Appleby, P. G., Richardson, N., and Nolan, P. J. (1991). '^{241}Am dating of lake sediments'. *Hydrobiologica*. **214**, 35 – 42

Appleby, P. G., Shotyk, W., and Fankhauser, A. (1997). 'Lead-210 age dating of three peat cores in the Jura mountains, Switzerland'. *Water, Air and Soil Pollution*. **100**, 223 – 231

Avery, B. W. (1980). *Soil classification for England and Wales (higher categories) - Technical Monograph No. 14*. Soil Survey of England and Wales: Harpenden

Bao, K., Xia, W., Lu, X., and Wang, G. (2010). 'Recent atmospheric lead deposition recorded in an ombrotrophic peat bog of Great Higgan Mountains, Northeast China, from ^{210}Pb and ^{137}Cs dating'. *Journal of Environmental Radioactivity*. **101**, 773 – 779

Beilman, D. W., Vitt, D. H., Bhatti, J. S., and Forest, S. (2008). ‘Peat carbon stocks in the southern Mackenzie River Basin: uncertainties revealed in a high-resolution case study’. *Global Change Biology*, **14**, 1 – 2.

Belyea, L. R. (1996). ‘Separating the effects of litter quality and microenvironment on decomposition rates in a patterned peatland’. *Oikos.* **77**, 529 – 530

Belyea, L. R., and Warner, B. G .(1994). ‘Dating of the near surface layer of a peatland in Northwestern Ontario, Canada’. *Boreas*. **23**, 259 – 269.

Belyea, L. R., and Clymo, R. S. (2001). ‘Feedback control of the rate of peat formation’. *Proceedings of the Royal Society of London B*. **268**, 1315 – 1321

Bevan, B. (2009). Moors from the past. In. Bonn, A., Allott, T., Hubacek., Stewart, J. (eds.) *Drivers for Change in the Uplands*. Routledge: London

Bhatti, J. S., Apps. M. J., and Tarocai, C. (2002). ‘Estimates of soil organic carbon stocks in central Canada using three different approaches’. *Canadian Journal of Forest Research*. **32**, 805 – 812

Billett, M. F., and Moore, T. R. (2008). ‘Supersaturation and evasion of CO_2 and CH_4 in surface waters at Mer Bleue peatland, Canada’. *Hydrological Processes*. **22**, 2044 – 2054

Billett, M.F., Palmer, S.M., Hope, D., Deacon, C., Storeton-West, R., Hargreaves, K.J., Flechard, C., Fowler, D. (2004). ‘Linking land-atmosphere-stream carbon fluxes’. *Global Biogeochemical Cycles.* **18**(1).GB1024, doi:10.1029/2003 B002058

Billett, M.F., Charman, D.J., Clark, J.M., Evans, C.D., Evans, M.G., Ostle, N.J., Worrall, F., Burden, A., Dinsmore, K.J., Jones, T., McNamara, N.P., Parry, L., Rowson, J.G., Rose, R. (2010). ‘Carbon balance of UK peatlands: current state of knowledge and future research challenges’. *Climate Research*. **45**, 13 – 29. Doi: 10.3354/cr00903

Blodau, C. (2002). ‘Carbon cycling in peatlands – A review of processes and controls’. *Environmental Review* **10**, 111 – 134.

Blodau, C., and Moore, T. R. (2003). ‘Experimental response of peatland carbon dynamics to water table fluctuation’. *Aquatic Sciences*. **65**, 47 – 62

Blunier, T., Chappellaz, J., Schwander, J., Stauffer, B., and Reynaud, D .(1995). ‘Variations in atmospheric methane concentration during the Holocene epoch’. *Nature*. **374**, 46 – 49

Boelter, D. H. (1972). ‘Water table drawdown around an open ditch in organic soils’. *Journal of Hydrology*. **15,** 329 – 340

Bol, R. A., Harkness, D. D., Huang, Y., Howard, D. M. (1999). 'The influence of soil processes on carbon isotope distribution and turnover in the British Uplands'. *European Journal of Soil Science.* **50**, 41 – 51

Bonn, A., Allott, T., Hubacek, K., and Stewart, J. (2009a). Introduction drivers for change in upland environments: concepts, threats and opportunities. In. Bonn, A., Allott, T., Hubacek., Stewart, J. (eds.). *Drivers for Change in the Uplands.* Routledge: London

Bonn, A., Rebane, M., and Reid, C. (2009b). Ecosystem services: a new rationale for conservation of upland environments. In. Bonn, A., Allott, T., Hubacek., Stewart, J. (eds.). *Drivers for Change in the Uplands.* Routledge: London

Bossew, P., Lettner, H., and Hubmer, A. (2006). 'A note on ^{207}Bi in samples'. *Journal of Environmental Radioactivity.* **91**, 160 – 166

Bradley, R. I., Milne, R., Bell, J., Lilly, A., Jordan, C., and Higgins, A. (2005). 'A soil carbon and land-use database for the United Kingdom'. *Soil Use and Management.* **21**, 363 – 369

Bragg, O. M., and Tallis, J. H. (2001). 'The sensitivity of peat covered upland landscapes'. *Catena* **42**, 345 – 360

Bubier, J., Costello, A., Moore, T. R., Roulet, N. T., and Savage, K. (1993). ‘Microtopography and methane flux in boreal peatlands, Northern Ontario, Canada’. *Canadian Journal of Botany.* **71**, 1056 – 1063

Buffam, I., Carpenter, S. R., Yecj, W., Hanson, P. C., Turner, M.G. (2010). ‘Filling the holes in regional carbon budgets: predicting peat depth in a temperate lake district’. *Journal of Geophysical Research*. **115.** doi: 10.1029/2009JG001034

Burke, E. J., Perry, R. H. J., Brown, S. J. (2010). ‘An extreme value analysis of UK drought and projections of change in the future’. *Journal of Hydrology*. 388, 131 – 143

Caseldine, C. J. (1999). ‘Archaeological and Environmental Change on Prehistoric Dartmoor – Current Understanding and Future Directions’. *Quaternary Proceedings*. **7**, 575 – 583

Caseldine, C.J. and Maguire, D.J. (1986). ‘Late glacial/early Flandrian vegetation change on northern Dartmoor, South West England’. *Journal of Biogeography.* **13**, 255-264

Caseldine, C. J., and Hatton, J. (1993). The development of high moorland on Dartmoor: fire and the influence of Mesolithic activity on vegetation change. In. Chambers, F. M. (ed). *Climate change and human impact on the landscape*. Chapman and Hall: London

Caseldine, C. J., and Hatton, J. (1996). Vegetation history of Dartmoor – Holocene development and the impact of human activity. In. Charman, D. J., Newnham, R. M., and Croot, D. G. (eds). *Devon and East Cornwall Field Guide.* Quaternary Research association: Cambridge

Chambers, F. M., Mauquoy, D., Todd, P. (1999). 'Recent rise to dominance of *Molinia caerulea* in environmentally sensitive areas: new perspectives from palaeoecological data'. *Journal of Applied Ecology*. **36**(5), 719 – 733.

Chapman, S. J., Bell, J., Donnelly, D., and Lilly, A. (2009). 'Carbon stocks in Scottish Peatlands'. *Soil Use and Management*. **25**, 105 – 112

Charman, D. J. (1992). 'Blanket mire formation at Cross Lochs, Sutherland, northern Scotland'. *Boreas*. **21**, 53 – 72

Charman, D. J. (1995). 'Patterned fen development in northern Scotland: hypothesis testing and comparison with ombrotrophic blanket peats'. *Journal of Quaternary Science*. **10**, 327 – 342

Charman, D. J. (2002). *Peatlands and environmental change*. Wiley: London

Charman, D.J. (2007). 'Summer water deficit controls on peatland water table changes: implications for Holocene palaeoclimate reconstructions'. *The Holocene.* **17**(2), 217-227

Charman, D. J., Avavena, R., Bryant, C. L., Harkness, D. D. (1999). ‘Carbon isotopes in peat, DOC, CO_2, and CH_4 in a Holocene peatland on Dartmoor, southwest England’. *Geology*. **27**(6), 539 – 542

Clark, J. M., Chapman, P. J., Adamson, J. K., Lane, S. N. (2005). ‘Influence of drought-induced acidification on the mobility of dissolved organic carbon in peats’. *Global Change Biology*. **11**, 791 – 801

Clark, J. M., Gallego-Sala, A. M., Allott, T. E. H., Chapman, S. J., Farewell, T., Freeman, C., House, J. I., Orr, H. G., Prentice, I. C., Smith, P. (2010). ‘Assessing the vulnerability of blanket peat to climate change using an ensemble of statistical bioclimatic envelope models’. *Climate Research*. **45**, 131 – 150. DOI: 10.3354/cr00929

Clay, G. D., Worrall, F., Clark, E., and Fraser, E. D. G. (2009). ‘Hydrological responses to managed burning and grazing in upland blanket bog’. *Journal of Hydrology*. **376**, 486 – 495

Clymo. R, S. (1984). ‘The limits to peat bog growth’. *Philiosophical transactions of the Royal Society of London B*. B303, 605 – 654

Clymo, R. S. (1992). Productivity and decomposition of peat-land ecosystems. In. Bragg, O. M., Hulme, P. D., Ingram, H. A. P., and Robertson, R. A. (eds). *Peatland Ecosystems and Man: An Impact Assessment*. International Peat Society: Finland

Clymo, D., Turunen, J., and Tolonen, K. (1998). ‘Carbon accumulation in peatland’. *Oikos*. **81**, 368 – 388

Clymo, R. S., Oldfield, F., Appleby, P. G., Pearson, G. W., Ratnesar., and Richardson, N. (1990). ‘The record of atmospheric deposition on a rainwater-dependant peatland’. *Philosopical Transactions of the Royal Society of London B*. **327**(1240), 331 – 338

Colston, A., Blaylock, S., Lister, J., Holley, S., Fergusson, G. (2007). The National Trust’s Upper Plym property, south west Dartmoor – a 50 year vision to restore favourable condition’. The National Trust: Devon

Conway, V. M. (1954). ‘Stratigraphy and pollen analysis of southern Pennine blanket peats’. *Journal of Ecology*. **42**, 117 – 147

Coulson, J. C., Butterfield, J. E. L., and Henderson, E. (1990). ‘The effect of open drainage ditches on the plant and invertebrate communities of moorland and decomposition of peat’. *Journal of Applied Ecology*. **27**, 549 – 561

Crowle, A. (2007). ‘Letting our carbon go free: the sustainable management of carbon and blanket peat in the English uplands’. *British Wildlife* **19**, 29 – 34

Damman, A.W.H. (1979). Geographic patterns of peatland development in eastern North America. In. Kivinen, E., Heikurainen, L., Pakarinen, P. (Eds.). *Classification of Peat and Peatlands*. Proc. International Peat Society, Finland.

Damman, A. W. H .(1978). 'Distribution and movement of elements in ombrotrophic peat'. *Oikos*. **30,** 480 – 495

Davies, M. G., Gray, A., Hamilton, A., and Legg, C. J. (2008). 'The future of fire management in the British uplands'. *International Journal of Biodiversity Science and Management*. **4**, 127 – 147

Dawson, J. J. C., and Smith, P. (2007). 'Carbon losses from soil and its consequences for land-use management'. *Science of the Total Environment*. **382**, 165 – 190

DECC. (2009). Carbon valuation in UK policy appraisal: A revised approach. Department of Energy and Climate Change: London. Available at < http://www.decc.gov.uk/publications/basket> [Accessed 15/10/2010]

Dinsmore, K. J., Billett, M. F., Skiba, U. M., Rees, R. M., Drewer, J., and Helfter, C. (2010). 'Role of the aquatic pathway in carbon and greenhouse gas budgets of a peatland catchment'. *Global Change Biology*. **16**, 2750 – 2762

Dunfield, P., Knowles, R., Dumont, R., Moore, T. (1993). 'Methane productions and consumption in temperate and subartic peat soils: response to temperature and pH'. *Soil Biology and Biogeochemistry.* **23**, 321 – 326

Dunn, S. M., and Mackay, R. (1996). 'Modelling hydrological impacts of open ditch drainage'. *Journal of Hydrology*. **179**, 37 – 66

ECOSSE. (2007). *Estimating carbon in organic soils sequestration and emission*. Scottish Executive: Edinburgh. Available at < http://www.scotland.gov.uk/Publications/2007/03/16170508/1> [Accessed 20/11/2007]

Evans, C. D., Freeman, C., Monteith, D. T., Reynolds, B., and Fenner, N. (2002). 'Terrestrial export of organic carbon'. *Nature*. **415**, 861 – 862

Evans, C. D., Monteith, D. T., Cooper, D. M. (2005). 'Long-term increases in surface water dissolved organic carbon: Observations, possible causes and environmental impacts'. *Environmental Pollution*. **137**, 55 – 71

Evans, M.G. (2009). Natural Changes in Upland Landscapes. In. Bonn, A., Allott, T., Hubacek., Stewart, J. (eds.). *Drivers for Change in the Uplands*. Routledge: London

Evans, M. G., and Lindsay, J. (2010a). 'High resolution quantification of gully erosion in upland peatlands at the landscape scale'. *Earth Surface Processes and Landforms*. **35**, 876 – 866

Evans, M. G., and Lindsay, J. (2010b). 'The impact of gully erosion on carbon sequestration in blanket peatlands'. *Climate Research.* **24**, 31 – 41. DOI: 10.3354/cr00887

Evans, M.G., and Warburton, J. (2005). 'Sediment budget for an eroding peat-moorland catchment in northern England'. *Earth Surface Processes and Landforms*. **30**, 557 – 577

Evans, M. G., and Warburton, J. (2007). *Geomorphology of upland peat: Erosion, form and landscape change*. Wiley Blackwell: Oxford

Evans, M., Warburton, J., Yang, J. (2006). 'Eroding blanket peat catchments: Global and local implications of upland organic sediment budgets'. *Geomorphology*. **79**, 45 – 57

Evans, R. (1998). ' The erosional impacts of grazing animals'. *Progress in Physical Geography*. **22**, 251 – 268

Farage, P., Ball, A., McGenity, T. J., Whitby, C., and Pretty, J. (2009). 'Burning management and carbon sequestration of upland heather moorland in the UK'. *Australian Journal of Soil Research.* **47**, 351 – 361.

Ferguson, N. P., and Lee, J. A. (1983). 'Past and present sulphur pollution in the southern Pennines'. *Atmospheric Environment*. **17**, 1131 – 1171

Findley, D. C., Colborne., Cope, D. W., Harrod, T. R., Hogan, D. V. and Staines, S. J. (1984). *Soils and their use in South West England*. Soil Survey of England and Wales: Harpenden

Foster, D. R., King, G. A., Glaser, P. H., and Wright, H. E. (1988). 'Bog development and landform dynamics in central Sweden and south east Labrador, Canada'. *Journal of Ecology*. **76**, 1164 – 1185

Freeman. C., Ostle. N., Kang, H. (2001a). 'An enzymic 'latch' on a global carbon store – a shortage of oxygen locks up carbon in peatlands by restoring a single enzyme'. *Nature*. **409**, 149.

Freeman, C., Evans, C. D., Monteith, D. T. (2001b). 'Export of organic carbon from peat soils'. *Nature*. **412**, 785

Freeman, C., Fenner, N., Ostle, N. J., Kang, H., Dowrick, D. J., Reynolds, B., Lock, M. A., Sleep, D., Hughes, S., and Hudson, J. (2004). 'Export of dissolved organic carbon from peatlands under elevated carbon dioxide levels'. *Nature.* **430**, 195 – 198

Frenzel, P., and Karofeld, E. (2000). 'CH_4 emission from a hollow ridge complex in a raised bog: the role of CH_4 production and oxidation'. *Biogeochemistry*. **51**, 91 – 112

Frogbrook, Z. L., Bell, J., Bradley, R. I., Evans, C., Lark, R. M., Reynolds, B., Smith, P., and Towers, W. (2009). 'Quantifying terrestrial carbon stocks: examining the spatial variation in two upland areas in the UK and a comparison to mapped estimates of soil carbon'. *Soil Use and Management* **25**, 320 – 332

Fyfe, R. M. (2006). *White Horse Hill phase 1 - Unpublished interim report for DNPA* School of Geography, University of Plymouth: Plymouth

Fyfe, R.M. (2007). *Cut Hill phase 2 - Unpublished interim report for DNPA*. School of Geography, University of Plymouth: Plymouth

Fyfe, R. M. (2008). *Archaeological and palaeological assessment – Blackbrook Hill (Unpublished interim report for DNPA).* School of Geography, University of Plymouth: Plymouth

Fyfe, R. M. (unpublished). *Dartmoor radiocarbon database*. School of Geography, University of Plymouth: Plymouth

Gallego-Sala, A. V., Clark, J. M., House, J. I., Orr, H. G., Prentice, C. I., Smith, P., Farewell, T., Chapman, S. J. (2010). 'Bioclimatic envelope model of climate change impacts on blanket peatland distribution in Great Britain'. *Climate Research*. **45**, 151 – 162. DOI: 10.3354/cr00911

Garnett, M. H., Ineson, P., and Stevenson, A. C. (2000). 'Effects of burning and grazing on carbon sequestration in a Pennine blanket bog, UK'. *The Holocene*. **10**(6), 729 – 736

Garnett, M. H., Ineson, P., Stevenson, A. C., and Howard, D. C. (2001). 'Terrestrial organic carbon storage in a British moorland'. *Global Change Biology* **7**, 375 – 388

Gerdol, R., Degetto, S., Mazzotta, D., and Vecchiati, G. (1994). 'The vertical distribution of the Cs-137 derived from Chernobyl fall-out in the uppermost *Sphagnum* layer of two peatlands in the southern Alps (Italy)'. *Water, Air and Soil Pollution*. **75**, 93 – 106

Gessler, P. E., Chadwick, O. A., Chamran, F., Althouse, L., and Holmes, K. (2000). 'Modeling soil – landscape and ecosystem properties using terrain attributes'. *Soil science society of America* **64**, 2046 – 2056

Gorham, E. (1991). 'Northern peatlands: role in the carbon cycle and probable responses to climatic warming'. *Ecological Applications*. **1**(2), 182 – 195

Graniero, P. A., and Price, J. S. (1999). 'The importance of topographic factors on the distribution of bog and heath in a Newfoundland blanket bog complex'. *Catena*. **36**, 223 – 254.

Greeves, T. (2006). *Swaling on Dartmoor: an historical survey - Report for Dartmoor National Park Authority and Natural England.* Dartmoor National Park Authority: Bovey Tracy

Hendon, D. and Charman, D,J. (2004). 'High resolution peatland water table changes for the past 200 years: the influence of climate and implications for management'. *The Holocene.* **14**, 125-134

Hobbs, N. B. (1986). ‘Mire morphology and the properties and behaviour of some British and foreign peats’. *Quarterly journal of Engineering Geology and Hydrogeology*. **19**, 7 – 80

Hobbs, R. J. (1984). ‘Length of burning rotation and community composition in high level Calluna-Eriophorum bog in Northern England’. *Vegetatio.* **57**, 129 – 13

Holden, J. (2005). ‘Peatland hydrology and carbon release: why small scale processes matter’. *Philosophical transactions of the Royal Society A*. **363**, 2891 – 2913

Holden, J., Chapman, P., Labadz, J. C. (2004). ‘Artificial drainage of peatlands: hydrological and hydrochemical process and wetland restoration’. *Progress in Physical Geography*. **28**(1), 95 – 123

Holden, J., Evans, M. G., Burt, T. P., and Horton, M. (2006a). ‘Impact of Land Drainage on Peatland Hydrology’. *Journal of Environmental Quality*. **35**, 1764 – 1778

Holden, J., Chapman, P., Evans, M., Hubacek, K., Kay, P., and Warburton, J. (2006b). *Vulnerability of organic soils in England and Wales – Project SP0532*. DEFRA: London

Holden, J., Shotbolt, L., Bonn, A., Burt, T.P., Chapman, P. J., Dougill, A. J., Fraser, E. D.G., Hubacek, K., Irvine, B., Kirkby, M. J., Reed, M. S., Prell, C., Stagl, S., Stringer, L. C., Turner, A. and Worrall, F.(2007a). ‘Environmental change in moorland landscapes’. *Earth Science Reviews.* **82**, 75 – 100.

Holden, J., Gascoign, M., and Bosanko, N. R. (2007b). 'Erosion and natural revegetation associated with land surface drains in upland peatlands'. *Earth Surface Processes and Landforms*. **32**, 1547 – 1557

Hope, D., Palmer, S. M., Billett, M. F., Dawson, J. C. (2001). 'Carbon dioxide and methane evasion from a temperate peatland stream'. *Limnology and Oceanography*. **46**(4), 843 – 857

Hope, D., Palmer, S., Billett. M.F., and Dawson J.J.C. (2004). 'Variations in dissolved CO_2 and CH_4 in a first order stream and catchment: an investigation of soil-stream linkages'. *Hydrological Processes, **18**: 3255-3275*

Houghton, R. A. (2003). The Contemporary Carbon Cycle. In. Schlesinger, W. H.(Ed). *Biogeochemistry - Volume 8*. Elsevier: Oxford

Howard, P. J. A., Loveland. P. J., Bradley, R. I., Dry, F. T., Howard, D.M., and Howard, D. C. (1995). 'The carbon content of soil and its geographical distribution in Great Britain'. *Soil Use and Management*. **11**, 9 – 15

Hulme, M., Jenkins, G.L., Lu, X., Turnpenny, J.R., Mitchell, T.D., Jones, R.G., Lowe, J., Murphy, J.M., Hassell, D., Boorman, P., McDonald, R., and Hill, S. (2002). *Climate Change Scenarios for the United Kingdom: The UKCIP02 Scientific*

Report. Tyndall Centre for Climate Change Research, School of Environmental Sciences, University of East Anglia, Norwich, UK

Ingram, H. A. P. (1978). 'Soil layers in mires: function and terminology'. *Journal of Soil Science*. **29**, 244 – 227

IPCC. (2007). *Intergovernmental Panel on Climate Change: Forth Assessment Report*. Available at <http://www.ipcc.ch> [Acessed 23/10/10]

Ivanov, K. E. (1981). *Water movement in mirelands*. Translated from Russian by Thompson, A., and Ingram, H. A. P. Academic Press: London

Jenson, S. K., and J. O. Domingue. (1988). 'Extracting Topographic Structure from Digital Elevation Data for Geographic Information System Analysis'. *Photogrammetric Engineering and Remote Sensing*. **54** (11), 1593–1600

Jobbágy, E, J. and Jackson, R. B. (2000). 'The vertical distribution of soil organic carbon and its relation to climate and vegetation'. *Ecological Applications*. **10**(2), 423 – 436

Johnson, L. C., and Damman, A. W. H. (1991). 'Species controlled decay on a south Swedish raised bog'. *Oikos*. **61**, 234 – 242

Joosten, H. (2010). *Conference proceedings - Importance of peatlands for climate change mitigation and adaption*. IUCN Peatlands and Climate Change Conference: Durham

Kim, G., Hussain, N., Church, T. M., Carey, W. L. (1997). 'The fallout isotope ^{207}Bi in a Delaware salt marsh: a comparison with ^{210}Pb and ^{137}Cs as a geochronological tool'. *The Science of the Total Environment.* **196**, 31 – 41

Kinako, P. D. S., and Gimingham, C. H. (1980). 'Heather burning and soil erosion on upland heaths in Scotland'. *Journal of Environmental Management*. **10**, 277 – 284

Komulainen, V., Tuittila, E., Vasander, H., Laine, J. (1999). 'Restoration of drained peatlands in southern Finland: initial effects on vegetation change and CO_2 balance'. *Journal of Applied Ecology*, **36**. 634 – 648

Kuhry, P. (1994). 'The role of fire in the development of *Sphagnum* dominated peatlands in western boreal Canada'. *Journal of Ecology*. **81**, 899 -910

Lafleur, P. M., Roulet, N. T., and Admiral, S. W. (2001). 'The annual cycle of CO_2 exchange at a boreal bog peatland. *Journal of Geophysical Research*. **106**, 3071 - 3081

Lafleur, P. M., Roulet, N. T, Bubier, J. L., Frolking, S., and Moore, T. R. (2003). ‘Interannual variability in peatland-atmosphere carbon dioxide exchange at an ombrotrophic bog’. *Global Biogeochemical Cycles*. **17**(2). DOI:10.1029/2002GB001983

Laine, A., Wilson, D., Kiely, G., Byrne, K. A. (2007). ‘Methane flux dynamics in an Irish lowland blanket bog’.*Plant Soil*. **299**, 181 – 193

Lindsay, R. A. (1995). *Bogs: the Ecology, Classification and Conservation of Ombrotrophic Mires.* Scottish Natural Heritage: Edingburgh

Lindsay, R. (2010). *Peatbogs and carbon: a critical synthesis to inform policy development in oceanic peat bog conservation and restoration in the context of climate change*. Royal Society for the Protection of Birds: London. Available at < http://www.rspb.org.uk/Images/Peatbogs_and_carbon_tcm9-255200.pdf> [accessed on 10/09/10]

Lindsay, R. A., Charman, D. J., Everingham, F., O’Reilly, R. M., Palmer, M. A., Rowell, T. A., and Stroud, D. A. (1988). *The Flow Country: the Peatlands of Caithness and Sutherland*. Nature Conservancy Council: Peterborough

Lyons, G. J., Lunny, F., Pollock, H. P. (1985). ‘A procedure for estimating the value of forest fuels’. *Biomass*. 8, 288 – 300

MacKenzie, A. B., Farmer, J. G., Sugden, C. L. (1997). 'Isotopic evidence of the relative retention and mobility of lead and radiocaesium in Scottish ombrotrophic peats'. *The Science of the Total Environment*. **203**, 115 – 127

Mackintosh. J. (2010). *UK climate change sustainable development indicator: 2009 greenhouse gas emissions, provisional figures and 2008 greenhouse gas emissions, final figures by fuel type and end-user*. Department of Energy and Climate Change, London. Available at <http://www.decc.gov.uk/en/content/cms/statistics/climate_change/gg_emissions/uk_emissions/2009_prov/2009_prov.aspx> [assessed on 15/11/10]

Maltby, E., Legg, C. J., and Proctor, M. C. F. (1990). 'The ecology of severe moorland fire on the North York moors: effects of the 1976 fires, and subsequent surface and vegetation development'. *Journal of Ecology*. **58**, 490 – 518

McMorrow, J., Lindley, S., Aylan, J., Caven, G., Albertson, K., and Boys. D. (2009). 'Moorland wildfire risk, visitors and climate change: patterns, prevention and policy'. In. Bonn, A., Allott, T., Hubacek., Stewart, J. (eds.) *Drivers for Change in the Uplands*. Routledge: London

Mercer, I. (2009). *Dartmoor*. Collins: London

Met Office. (2010a.). 'South West England: climate'. Available from <http://www.metoffice.gov.uk/climate/uk/sw/> [accessed on 12/08/10]

Met Office. (2010b.). 'Princetown 1971 – 2000 averages'. Available from <http://www.metoffice.gov.uk/climate/uk/averages/19712000/sites/princetown.html>[accessed on 12/08/10]

Meyles, E. W., Williams, A. G., Ternan, J. L., Anderson, J. M., and Dowd, J. F. (2006). 'The influence of grazing on vegetation, soil properties and stream discharge in a small Dartmoor catchment, southwest England, UK'. *Earth Surface Processes and Landforms* **31**, 622 – 631

Milne, R., and Brown, T. A. (1997). 'Carbon and Vegetation of Soils of Great Britain'. *Journal of Environmental Management* **49**, 413 – 433

Mitchell, P. I., Schell, W. R., McGarry, A., Ryan, T. P., Sanchez-Cabeza, J. A., Vidal-Quadras, A. (1992). 'Studies of the vertical distribution of ^{134}Cs, ^{137}Cs, ^{238}Pu $^{239,240}Pu$, ^{241}Pu, ^{241}Am and ^{210}Pb in ombrogenous mires at mid-latitudes'. *Journal of Radioanalytical and Nuclear Chemistry*. **156**(2), 361 – 387

Moore, P. D. (1975). 'Origin of blanket mires'. *Nature*. **256**, 267 – 269

Moore, P. D. (1984). *European Mires*. Academic Press: London

Moore, T. R., and Dalva, M. (1997). 'Methane and carbon dioxide exchange potentials of peat soils in aerobic and anaerobic laboratory conditions'. *Soil Biology and Biochemistry*. **29**, 1159 – 1164

Moore, T. R., Roulet, N. T., and Waddington, J. M. (1998). 'Uncertainty in predicting the effect of climatic change on the carbon cycling of Canadian peatlands'. *Climatic Change*. **40**, 229 – 245

Murphy. J., Sexton, D., Jenkins, G., Boorman, P., Booth, B., Brown, K., Clark, R., Collins, M., Harris, G., Kendon, L. (2009). *UK Climate Projections Science Report: Climate Change Projections*. Met Office: Hadley Centre

Natural England. (1999). *Environmental cross compliance stage two overgrazing assessment: Okehampton common, Dartmoor*. Natural England: Exeter

Natural England. (2007). The Heather and Grass Burning Code: 2007 (Report: PB12650). Natural England: London. Available at < www.naturalengland.org.uk/publications> [accessed on 15.05.10]

Natural England. (2009a). Economic valuation of upland ecosystem services (Report: NECR029. Natural England: London. Available at < www.naturalengland.org.uk/publications> [accessed on 15.05.10]

Natural England. (2009b). England's Peatlands: Carbon Storage and Greenhouse Gases (Report NE297). Available at <http://naturalengland.etraderstores.com/NaturalEnglandShop/NE257> [accessed on 17.05.10]

Natural England. (2010). Entry level stewardship – Environmental stewardship handbook, third edition – Feburary 2010 (report NE226). Natural England: London. Available at < www.naturalengland.org.uk/publications> [accessed on 14.04.10]

Nayak, D. R., Miller, D., Nolan, A., Smith, P., and Smith, J. (2008). *Calculating carbon savings from wind farms on Scottish Peat lands – A new approach.* Abeerdeen: University of Edinburgh and Macauley Land Use research institute

Newman, P. (2010). Historical survey of Dartmoor's peat workings: stage two progress. Dartmoor National Park Autority, Bovey Tracey, Devon

Nilsson, M., Sagerfors, J., Buffam, I., Laudon, H., Erikkson, T., Grelle, A., Klemdtssons, L., Weslein, P., Lindroth, A. (2008). 'Contemporary carbon accumulation in a boreal oligotrophic minerogenic mire – a significant sink accounting for all C-fluxes'. *Global Change Biology*. **14**, 2317 – 2332

Oldfield, F., Richardson, N., and Appleby, P. G. (1995). 'Radiometric dating (^{210}Pb, ^{137}Cs, ^{241}Am) of recent ombrotrophic peat accumulation and evidence for changes in mass balance'. *The Holocene*. **5**(2), 141 – 148

Oldfield, F., Appleby, P., Cambray, R., Eakins, J., Barber, K., Batterby, R., Pearson, G., and Williams, J. (1979). 'Pb-210, Cs-137, and Pu-239 profiles in ombrotrophic peat'. *Oikos*. **33**, 40 – 45

Pastor, J., Solin. J., Bridgham, S. D. (2003). 'Global warming and the export of dissolved organic carbon from boreal peatlands'. *Oikos*. **100**, 380 – 386

Piotrowska, N., De Vleeschouwer, F., Sikorski, J., Pawlyta, J., Fael, N., Le Roux, G., Pazdur, A. (2010). 'Intercomparison of radiocarbon bomb pulse and ^{210}Pb age models. A study in a peat bog core from North Poland'. *Nuclear Instruments and Methods in Physics Research B*. **268**, 1163 – 1166

Pitkanen, A., Turunen, J., and Tolonen, K. (1999). 'The role of fire in the carbon dynamics of a mire, eastern Finland'. *The Holocene*. **9**(4), 453 – 462

Rawlins, B. G., Marchant, B. P., Smyth, D., Scheib, C., Lark, R. M., and Jordan, C. (2009). 'Airborne radiometric survey data and a DTM as covariates for regional scale mapping of soil organic carbon across Northern Ireland'. *European Journal of Soil Science* **60**, 44 – 54

Renburg, I., and Wik, M. (1985). 'Soor partical counting in recent lake sediments: an indirect dating method'. *Ecological bulletins*. **37**, 53 – 57

Rose, N. L. (1994). ‘A note on further refinements to a procedure for the extraction of carbonaceous fly-ash particles from sediments’. *Journal of Paleolimnology*. **11**, 201 – 204

Rose, N. L. (2001). *Fly Ash Particals*. In. *Tracking Environmental Change Using Lake Sediments – Volume 2: Physical and Geochemical Methods*. Last, W. M., and Smol, J. P. (eds). Kluwer Academic Publishers: London

Rose, N. L. (2008). ‘Quality control in the analysis of lake sediments for spheroidal carbonaceous particals’. *Limnology and Oceanography: Methods*. **6**, 172 - 179

Rose, N. L., and Appleby, P. G. (2005). ‘Regional applications of lake sediment dating by spheroidal carbonaceous partical analysis I: United Kingdom’. *Journal of Paleolimnology*. **34**, 349 – 361

Rose, N. L., and Harlock, S. (1998). ‘The spatial distribution of characterised fly-ash particles and trace metals in lake sediments and catchment mosses in the United Kingdim’. *Water, Air and Soil Pollution*. **106**, 287 – 308

Rose, N. L., Harlock, S., Appleby, P. G., and Battarbee, R. W. (1995). ‘Dating of recent lake sediments in the United Kingdom and Ireland using spheroidal carbonaceous particle (SCP) concentration profiles’. *The Holocene*. **5**(3), 328 – 335

Roulet, N., Lafleur, P. M., Richard, P, J, H., Moore, T. R., Humphreys, E. R., and Bubier, J. (2007). 'Contemporary carbon balance and late Holocene carbon accumulation in a northern peatland'. *Global Change Biology*. **13**, 397 – 411

Rowson, J. G., Ginson, H. S., Worrall, F., Ostle, N., Burt, T. P., Adamson, J. K. (2010). 'The complete carbon budget of a drained peat catchment'. *Soil Use and Management*. **26**, 261 – 273

Ruimy, A., Jarvis, P.G., Baldocchi, D. D., and Saugier, B. (1995). 'CO_2 fluxes over plant canopies and solar radiation: a review'. *Advances in Ecological Research*. **26**, 1 -68

Rydin, H., and Jeglum, J. (2006). *The Biology of Peatlands*. Oxford: Oxford University Press

Sansom, A. L. (1999). 'Upland vegetation management: the impacts of overstocking'. *Water Science Technology*. **39**(12), 85 – 92

Shand, C. A., Cheshire, M. V., Smith, S., Vidal, M., and Rauret, G. (1994). 'Distribution of radiocaesium in organic soils. *Journal of Environmental Radioactivity*. **23**, 285 – 302

Sheng, Y., Smith, L. C., MacDonald, G. M., Kremenetski, K. V., Frey, K. E., Velichko, A. A., Lee, M., Beilman, D. W., and Dubinin, P. (2004). 'A high- resolution GIS-based inventory of west Siberian peat carbon pool'. *Global Biogeochemical Cycles*. **18**, doi:10.1029/2003GB002190

Shotyk, W., Weiss, D., Applyby, P. G., Cheburkin, A. K., Frei, R., Gloor, M., Kramers, J. D., Reese, S., Van der Knapp, W. O. (1998). 'History of atmospheric Lead deposition since 12, 370 ^{14}C yr BP from a peat bog, Jura, Switzerland'. *Science*. **281**, 1635 – 1640

Simmons, I. G. (1964). 'Pollen diagrams from Dartmoor'. *New Phytologist*. **65**, 165 – 180

Simmons, I. G. (2003). *The moorlands of England and Wales: An Environmental History 8000BC – AD2000*. Edinburgh University Press: Edinburgh

Simmons, I.G., Rand, J.I., and Crabtree, K. (1983). 'A further pollen analytical study of the Blacklane peat section on Dartmoor, England'. *New Phytologist.* 94, 655-667

Smith, A. G., and Cloutman, E. W. (1988). 'Reconstruction of Holocene vegetation history in three dimensions at Waun-Fignen-Felen, an upland site in south Wales'. *Philosophical transactions of the Royal Society of London. Series B Biological*. **322**(1209), 159 – 219

Smith, J. T., Appleby, P. G., Hilton, J., and Richardson, N. (1997). 'Inventories and Fluxes of ^{210}Pb, ^{137}Cs and ^{241}Am Determined from the Soils of Three Small Catchments in Cumbria, UK'. *Journal of Environmental Radioactivity*, **37**(2). 127 – 142

Smith, P., Chapman, S. J., Scott W. A., Black, H. I .J., Wattenbach, M. N, Milne, R., Campbell, C. D., Lilly, A., Ostlez, N., Levy, P .E., Lumsdown, D. D. G., Millard, P. , Towers, W., Zaehle. S., and Smith, J. (2007). 'Climate change cannot be entirely responsible for soil carbon loss observed in England and Wales, 1978 – 2003'. *Global Change Biology.* **13,** 1 – 5

Stewart, A. J. A., and Lance, A. N. (1991). 'Effects of moor draining on the hydrology and vegetation of northern Pennine blanket bog'. *Journal of Hydrology*. **28,** 1105 – 1117

Ström, L., Ekberg, A., Mastopanov, M., and Christensen, T. (2003). 'The effect of vascular plants on carbon turnover and methane emissions from a tundra wetland'. *Global Change Biology*. **9**, 1185 – 1192

Tallis, J. H. (1987). 'Fire and flood at Holme Moss: erosion processes in an upland blanket mire'. *Journal of Ecology*. **75**, 1099 – 1129

Tallis, J. H. (1991).'Forest and moorland in the south Pennine uplands in the mid-Flandrian period. III. The spread of moorland – local, regional and national'. *Journal of Ecology*, **79**, 401 – 415

Tallis, J. H. (1995). Blanket Mires in the Upland Landscape. In. Wheeler, B. D., Shaw, S. C., Fojt, W., and Robertson, R. A. (eds). *Restoration of temperate wetlands*. John Wiley: London

Tallis, J. H. (1998). The southern Pennine experience: an overview of blanket mire degradation. In. Tallis, J. H., Meade, R. And Hulme, P. D. (eds.). *Blanket Mire Degradation. Causes, Consequences and Challenges.* British Ecological Society: Aberdeen

Tolonen, K and Turunen, J. (1996). 'Accumulated rates of carbon in mires in Finland and implications for climate change'. *The Holocene*. **6**(2), 171 – 178

Triall-Thomson, J., and Bloomfield, D. (2009). 'The South West uplands public benefits project: working in partnership to find new ways to value all the services that uplands provide'. <Available at http://www.naturalengland.org.uk/Images/Jo%20Traill%20Thomson%20-%20SW%20Pilot_tcm6-15897.pdf.> [Accessed on 21.09.10]

Turetsky, M. R., Manning, S., Wieder, R. K. (2004). 'Dating Recent Peat Deposits'. *Wetlands*, **24**(2), 324 – 356

Turunen, J., Tomppo, E., Tolonen, K., and Reinikainen, A. (2002). 'Estimating carbon accumulation rates of undrained mires in Finland – application to boreal and subartic regions'. *The Holocene*. **12**, 69 – 80

UNFCCC. (2010). Ad Hoc working group on further commitments for Annex I parties under the Kyoto Protocol, fifteenth session, Cancun, 29 November. Available at http://maindb.unfccc.int/library/view_pdf.pl?url=http://unfccc.int/resource/docs/2010/awg15/eng/crp04r04.pdf> [Accessed on 21/12/2010]

Urban, N. R., Eisenreich, S. J., Grigal, D. F., and Schurr, K. T. (1990). 'Mobility and diagenesis of Pb and ^{210}Pb in Peat'. *Geochimica et Cosmochimica Acta*. **54**, 3329 – 3346

van der Linden, M., Barke, J., Vickery, E., Charman, D.J. and van Geel, B. (2008) Late Holocene human impact and climate change recorded in a North Swedish peat deposit. *Palaeoclimatology, Palaeoecology and Palaeogeography*. **258**, 1–27

Vile, M. A., Novák, M. J. V., Břizovà., Wiedar, R. K., Schell., W. R. (1995). 'Historical rates of atmospheric ^{210}Pb dated peat cores: corroboration, computation and interpretation'. *Water, Air and Soil Pollution.* **79**, 89 – 106

Vile, M. A., Wiedar, R. K., Novák, M. (1999). 'Mobility of Pb in *Sphagnum*-derived peat'. *Biogeochemistry*. **45**, 35 – 52

Wallage, Z. E., Holden, J., and McDonald, A. T. (2006). 'Drain bocking: An effective treatment or reducing dissolved organic carbon loss and water discoloration in a drained peatland'. *Science of the total environment*. **367**, 811 – 827

Warburton, J. (2003). 'Wind-splash erosion of bare peat on UK moorlands'. *Catena*. **52**, 191 – 207

Warburton, J., Evans, M. G., and Johnson, R. M. (2003). 'Discussion on 'The extent of soil erosion in upland England and Wales'. *Earth Surface Processes and Landforms.* **28**, 219 – 223

Ward, S. D., Thomson, A. G., Davis, P. S. (1969). A review on swaling on Dartmoor. In. The Nature Conservancy Montane Grassland Habitat Team. *Report on Dartmoor Ecological Survey*. NERC: London

Ward, S. E., Bardgett, R. D., MsNamara, N. P., Adamson, J. K., and Ostle, N. J. (2007). 'Long-term concequences of grazing and burning on northern peatland carbon dynamics'. *Ecosystems*. **10**, 1069 – 1083

Weishample, P., Kolka, R., King, J.Y. (2009). 'Carbon pools and productivity in a 1-km^2 heterogeneous forest and peatland mosaic in Minnesota, USA'. *Forest Ecology and Management*. **257**, 747 – 754

Wieder, R. K., Novák, M., Schell, W. R., Rhodes, T. (1994). 'Rates of peat accumulation over the past 200 years in five *Sphagnum*- dominated peatlands in the United States'. *Journal of Paleolimnology*. **12**, 35 – 47

Worrall, F., and Burt, T. P. (2007). ‘Trends in DOC concentration in Great Britain’. *Journal of Hydrology*. **346**, 81 – 92

Worrall, F. and Evans, M. G. (2009). The carbon budget of upland peat soils. In. Bonn, A., Allott, T., Hubacek., Stewart, J. (eds). *Drivers for Change in the Uplands.* Routledge: London

Worrall, F., Burt, T. P., and Adamson, J. (2004a). ‘Can climate change explain increases in DOC flux from upland peat catchments’. *Science of the Total Environment*. **326**, 95 – 112

Worrall, F., Armstrong, A., and Adamson, J. K. (2007). ‘The effects of burning and sheep-grazing on water table depth and soil water quality in upland peat’. *Journal of Hydrology*. **339**, 1 – 14

Worrall, F., Reed, M., Warburton, J., and Burt, T. (2003). ‘Carbon budget for a British upland peat catchment’. *Science of the Total Environment*. **312**, 133 – 146

Worrall, F., Burt, T. P., Rowson, J. G., Warburton, J., and Adamson, J. K. (2009a). ‘The multi-annual carbon budget of a peat covered catchment’. *Science of the Total Environment*. **407**, 4084 – 4094

Worrall, F., Evans, M. G., Bonn, A., Reed, M. S., Chapman, D., and Holden, J. (2009b). 'Can carbon offsetting pay for upland ecological restoration'. *Science of the Total Environment*. 408, 26 – 36

Worrall, F., Harriman, R., Evans, C. D., Watts, C. D., Adamson, J., Neal, C., Tipping, E., Burt, T., Grieve, I., Monteith, D., Nadan, P. S., Nisbet, T., Reynolds, B., and Stevens, P. (2004b). 'Trends in dissolved organic carbon in UK rivers and lakes'. *Biogeochemistry*. **70**, 369 – 402

Yallop, A. R., Clutterbuck, B., and Thacker, J. I. (2009). Burning Issues: the history and ecology of managed fire in the uplands. In. Bonn, A., Allott, T., Hubacek., and Stewart, J. (eds). Drivers of Environmental Change in Uplands. Routledge: London

Yallop, A. R., Thacker, J. I., Thomas, G., Stephens, M., Clutterbuck, B., Brewer, T., and Sannier, A. D. (2006) 'The extent and intensity of management burning in the British uplands'. *Journal of Applied Ecology.* **43**, 1138 – 1148

Yang, H., Rose, N. L., Boyle, J. F., Battarbee, R. W. (2001). 'Storage and distribution of trace metals and Spheroidal carbonaceous particles (SCPs) from atmospheric deposition in the catchment peats of Lochnagar, Scotland'. *Environmental Pollution*. **115**, 231 – 238

Yu, Z.C. (2011). Holocene carbon flux histories of the world's peatlands: global carbon-cycle implications. *Holocene* **21***,* DOI: 10.1177/0959683610386982

Zhou, C. (2010). ‘Mapping soil organic matter using the topographic wetness index: A comparative study based on different flow-direction algorithms and kriging models’. *Ecological Indicators*. **10**(3), 610 – 619

Zhu, A. X., Hudson, B., Burt, J., Lubich, K., and Simonson, D. (2001). ‘Soil mapping using GIS, Expert Knowledge, and Fuzzy Logic’. *Soil Science Society of America Journal*. **65**, 1463 – 1472