Carbon storage in Pennine moorland and response to change

By Mark H Garnett

Thesis submitted to the University of Newcastle-Upon-Tyne
for the degree of Doctor of Philosophy

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Abstract

The climate of the Earth is expected to warm due to increasing atmospheric concentrations of greenhouse gases, particularly carbon dioxide (CO_2). The impact of this change on terrestrial ecosystems is uncertain, yet these ecosystems contain three times more carbon (C) than the atmosphere and could considerably augment anthropogenic CO_2 emissions. This thesis aimed to quantify the terrestrial C stored in an area of Pennine moorland and predict its response to changes in climate and management.

The area of soil and vegetation types was determined using existing maps of the study site and a geographical information system. Vegetation C was assessed using results from published productivity studies, and soil C was evaluated from a programme of field sampling. Most C was contained in peat soils and comparison of the results with the UK's national C inventory identified inaccuracies in the national values.

The impact of sheep grazing and moorland burning on C storage was investigated by utilising long-term experiments at the site. Areas where sheep had been excluded for forty years were compared to grazed areas and burning was investigated using a randomised block experiment. Grazing had little impact on terrestrial C storage, but regular burning significantly reduced C sequestration in peats.

The response of the C store to climate change was investigated by i) determining the impact of past climate on C storage and ii) developing a predictive model by measuring the underlying processes of C input and output in the ecosystem. Both approaches illustrated the sensitivity of the ecosystems C balance to climate, suggesting that global warming will reduce C accumulation at the site and, therefore, augment atmospheric CO_2 concentrations.

The novel use of near infrared reflectance spectroscopy to characterise peat samples was also demonstrated, indicating opportunities for application of this technique in palaeoecological studies of peats.

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Contents

1. Introduction

1.1 Climate change and the greenhouse effect

1.1.1 The Earth's climate

The Earth's climate has not been constant through time (Nicholls *et al.*, 1996). During the last 2 million years it has been characterised by cycles of colder (glacial) and warmer (inter-glacial) conditions, with the last 10,000 years (the Holocene) being relatively warm and stable (Roberts, 1989; Ditlevsen *et al.*, 1996).

The main causes of past climatic change are believed to have been changes in incoming solar radiation (*e.g.* changes in the Earth's orbit), atmospheric dust, ocean circulation and the gaseous composition of the atmosphere (Street-Perrott and Roberts, 1993; Mann *et al.*, 1998). These factors affect the Earth's energy balance, which controls the climate system.

Solar radiation provides the main source of radiative energy for the Earth (Trenberth *et al.*, 1996) and since the sun is an extremely hot body, the radiation it emits is short wave. Short wave radiation is not absorbed by most gases within the Earth's atmosphere and passes straight through it and warms the ground, although a proportion is reflected back to space by clouds and atmospheric particulates. The ground re-emits radiation at longer wavelengths since the Earth is cooler. However, certain gases within the atmosphere (greenhouse gases) are able to absorb radiation of longer wavelengths, and retain the energy emitted from the Earth, causing the atmosphere to warm (Shine *et al.*, 1995). Clouds and atmospheric particulates also reflect radiation back to the Earth (Shine *et al.*, 1995; Trenberth *et al.*, 1996).

In an unperturbed system, the amount of energy entering the Earth's atmosphere would be equal to the amount leaving it, and therefore annual global temperatures would remain the same. However, the variability of past global climate indicates that changes in the energy balance can occur, and that future changes in this balance are also likely to cause global climatic change.

1.1.2 Increasing atmospheric CO_2 and the global carbon cycle

The energy balance of the Earth's atmosphere is widely believed to be changing as a consequence of increasing concentrations of greenhouses gases in the atmosphere (IPCC, 1995, 1996). With increased concentrations of greenhouse gases the Earth becomes more efficient at trapping out-going radiation, causing warming of the atmosphere. Whether the atmosphere has warmed as a result of the increased greenhouse gas concentration is still a matter of debate, although assessments by the Intergovernmental Panel on Climate Change have continually linked increasing global temperatures with rising greenhouse gas concentrations (IPCC, 1995, 1996).

Although concentrations of several greenhouse gases are currently increasing in the atmosphere, most concern focuses on the increased concentration of carbon dioxide (CO_2) (Figure 1.1), since this gas is considerably more abundant than other greenhouse gases. CO_2 occurs naturally in the atmosphere where it forms an important component of the global carbon (C) cycle.

There are three main pools in the global C cycle (Figure 1.2). The oceans are the largest pool, with approximately fifty times more C than the atmospheric C pool, whereas the terrestrial C pool (soils and vegetation) contains about three times more C than the atmosphere. Large amounts of C are transferred between these pools, with at least one third of the atmospheric C store annually exchanged between terrestrial and oceanic ecosystems and the atmosphere (Schimel, 1995).

In an unperturbed world, equal amounts of C are transferred between these pools such that the atmospheric C store remains approximately constant. For example, annual growth of vegetation in the world removes C from the atmosphere through photosynthesis. However, the amount of C released to the atmosphere through respiration by plants and animals (including decomposition) is about equal to the amount fixed by terrestrial ecosystems, so that the size of the atmospheric C pool stays the same. Similar balanced processes occur between the ocean and atmosphere.

However, anthropogenic activities have disturbed the global C cycle, resulting in an inbalance in fluxes of C to and from the atmosphere (Schimel *et al.*, 1996). Firstly, human activities are causing the release of fossil C which had been removed from the global C cycle. This C is contained within fossil fuels such as oil, coal and gas, with the burning of these fuels creating a significant flux of C into the atmosphere. Secondly, cement production releases large amounts of C to the atmosphere as CO_2. Thirdly, human disturbance of terrestrial ecosystems has resulted in the release of C from the terrestrial pool to the atmosphere, through processes such as deforestation. These three anthropogenic activities have perturbed the global C cycle and caused the atmospheric CO_2 concentration to rise, as shown from the records derived from ice cores and historical measurements (atmospheric CO_2 concentrations have risen from 280 ppm to *ca.* 360 ppm over the last 200 years).

The transfer of C between these pools has been summarised by Schimel (1995), Schimel *et al.*, (1995) and Schimel *et al.*, (1996). Schimel (1995) calculated that, in the

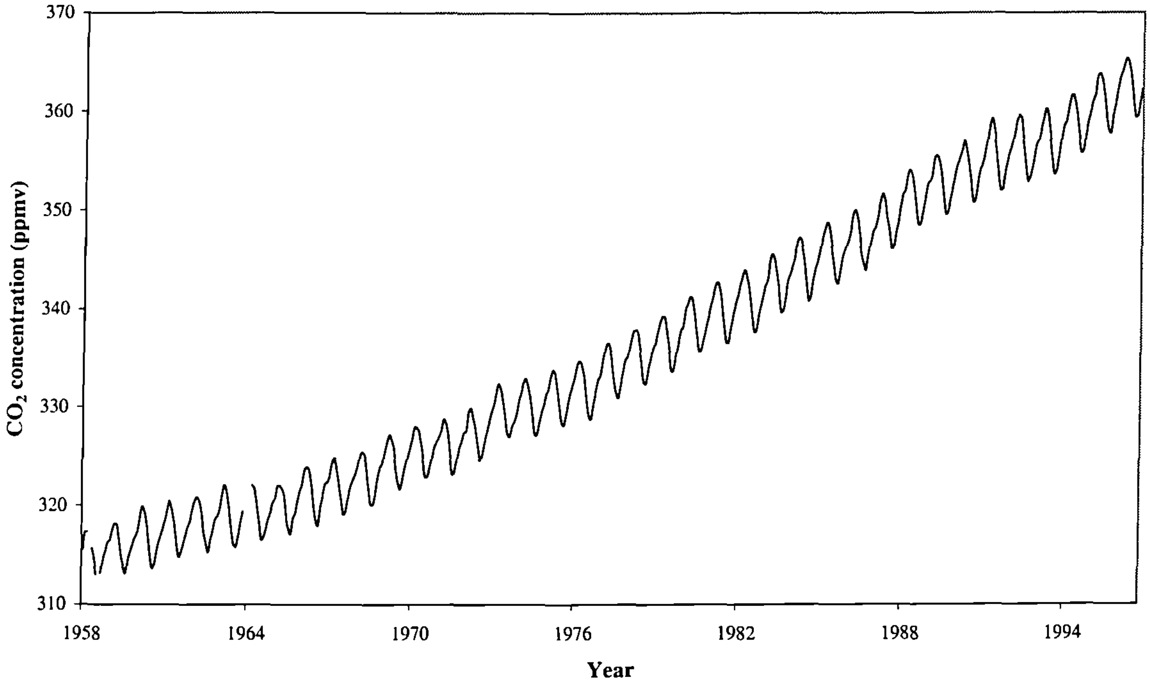

Figure 1.1 Atmospheric CO_2 concentration at Mauna Loa, Hawaii 1958-1996. Data source: C.D. Keeling and T.P. Whorf, Scripps Institution of Oceanography, University of California, La Jolla, California USA 92093-0220

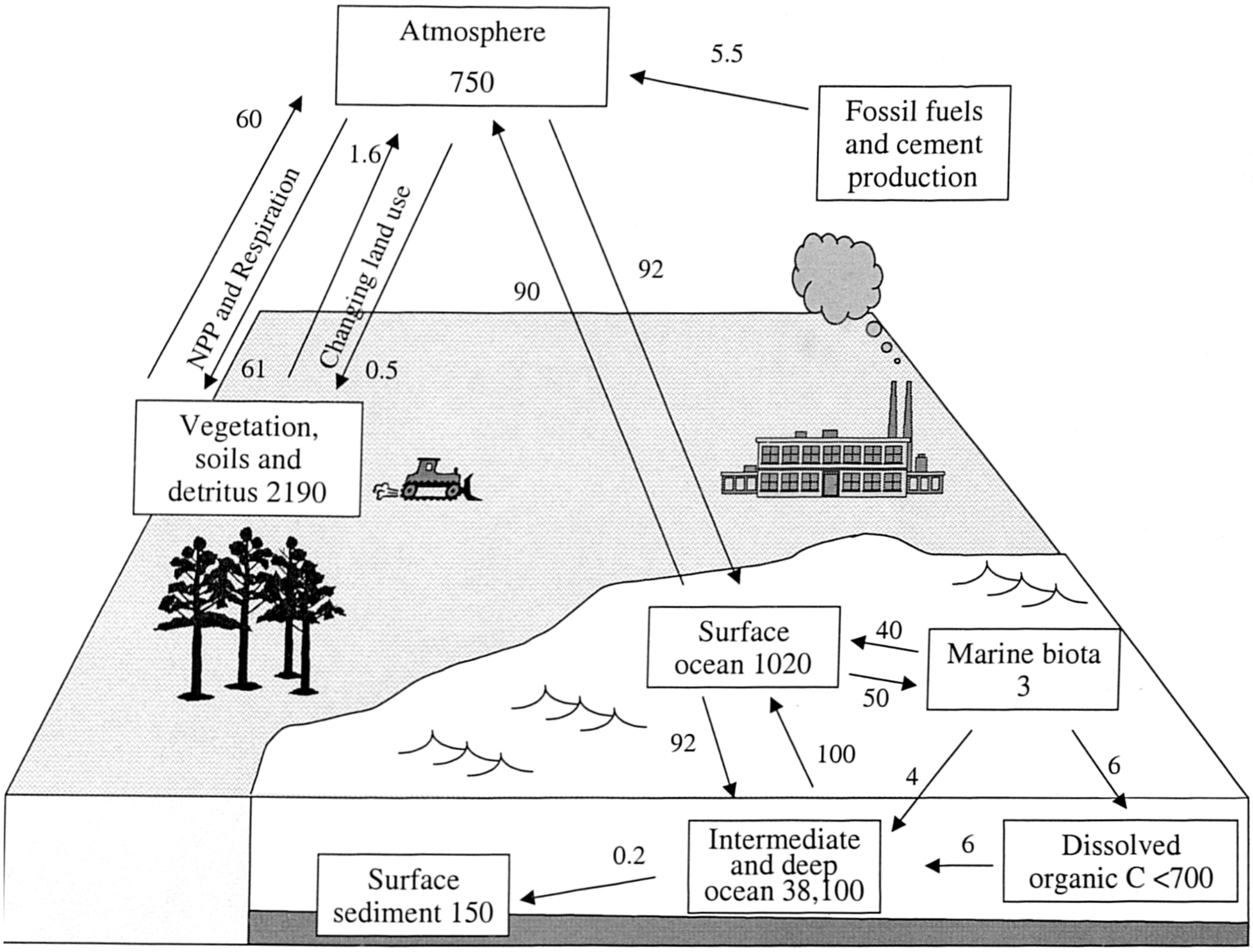

Figure 1.2 The global C cycle. All stores in kg x 10^{12} and fluxes in kg x 10^{12} per year (from Schimel *et al.*, 1996)

1980's, the annual release of C from fossil fuel burning and cement production was 5.5 x 10^{12} kg (5.5 Gt), while CO_2 produced from deforestation of tropical rainforests was 1.6 Gt. However, the atmospheric pool has only increased by an average rate of 3.2 Gt per year, meaning that 3.9 Gt of C have been transferred to other stores. The oceans have probably gained 2 Gt per year according to Schimel (1995), and regrowth of northern forests are thought to have sequestered about 0.5 Gt. However, 1.4 Gt of C remains unaccounted for, and since the global C cycle can be considered a closed system, this C must be contained within one of the stores.

This 'missing C sink' may be located in terrestrial ecosystems with suggestions that increased forest regrowth, CO_2 fertilisation of plant growth and nitrogen (N) deposition (Schimel, 1995) could account for the failure in the balance. It is important to determine the location of this missing C since the reduced rate that the atmosphere's C store has increased in CO_2 concentration compared to the anthropogenically released C, is partially a consequence of this sink; if the sink is only temporary, the mitigating effect on anthropogenic CO_2 release could be lost.

Attempts to predict the future climate of the Earth have been made assuming various scenarios for anthropogenic CO_2 release. Some of the most recent predictions of future global temperature, based on current and anticipated levels of CO_2 emissions, project the Earth's mean temperature to rise by between 1 and 4.5°C by the year AD 2100 (Kattenberg *et al.*, 1996).

However, there are considerable uncertainties in these predictions mainly due to questions of how the C cycle will respond to warmer climates. Since the terrestrial C store holds a large amount of C which is easily transferred to the atmosphere, there is concern that a reduction in terrestrial C storage, caused by global warming, may further increase levels of CO_2 in the atmosphere. The possible feedbacks to climate change such as this pose severe obstacles in predicting the future climate of the Earth. However, in order to estimate possible future fluxes of C between terrestrial ecosystems and the atmosphere under a changing climate, accurate information on the present fluxes and quantity of current terrestrial C stores are needed.

1.2 Terrestrial C stores

1.2.1 Estimates of global terrestrial C storage

The importance of quantifying terrestrial C stores was recognised at the Rio Earth Summit in 1992 when signatories to the Framework Convention on Climate Change committed nations to quantifying terrestrial C stores (United Nations, 1993). Several national C inventories have already been published (see below) against a background of global attempts to quantify the C stored in the Earth's vegetation and soils.

Terrestrial C stores are usually quantified by classifying the vegetation and soil and multiplying the area of the classes with estimates of mean C storage (Smith and Shugart, 1993). Estimates for vegetation C storage are likely to be more accurate and reliable than soil C stores since vegetation is easier to map and sample. In particular, remote sensing of vegetation is feasible and allows large areas to be mapped at high detail. Table 1.1 shows several estimates of global vegetation C stores, indicating that the total amount of C stored in the living vegetation of the Earth has been estimated to be *ca.* 600 Gt.

Table 1.1 Estimates of global vegetation C storage

Global vegetation C pool (Gt; kg x 10^{12})	Reference
610	Schimel (1995)
560	Ajtay *et al.* in Woodwell (1984)
557	Olson *et al.* in Woodwell (1984)
827	Whittaker and Likens (1973)

Table 1.2 shows estimates for the total vegetation C within the different vegetation types of the Earth's terrestrial ecosystems determined by Whittaker and Likens (1973). The table shows that forest ecosystems contain by far the greatest amount of C per unit area, with for example, tropical rainforests and boreal forests containing an average of 20 and 16 kg C m^{-2} respectively. In terms of total C storage, tropical rainforests hold by far the greatest amount of C with an estimated 340 Gt C. However, these estimates for vegetation C storage were based on area estimates made in the 1950's and deforestation of tropical rainforests has greatly reduced this store. For example, it is estimated that 20 x 10^9 m^2 yr^{-1} of Brazilian rainforest alone were deforested throughout the 1980's (Schimel, 1995), resulting in a reduction of up to 0.4 Gt C per year (see below).

Table 1.2 Estimates of total vegetation C stored in the world's terrestrial ecosystems (from Whittaker and Likens, 1973)

Ecosystem type	Area (m^2 x 10^{12})	Mean plant biomass (kg C m^{-2})	Total vegetation C (Gt; kg x 10^{12})
Tropical rainforest	17.0	20.0	340.0
Tropical seasonal forest	7.5	16.0	120.0
Temperate evergreen forest	5.0	16.0	80.0
Temperate deciduous forest	7.0	13.5	95.0
Boreal forest	12.0	9.0	108.0
Woodland and shrubland	8.0	2.7	22.0
Savanna	15.0	1.8	27.0
Temperate grassland	9.0	0.7	6.3
Tundra and alpine meadow	8.0	0.3	2.4
Desert scrub	18.0	0.3	5.4
Rock, ice and sand	24.0	0.01	0.2
Cultivated land	14.0	0.5	7.0
Swamp and marsh	2.0	6.8	13.6
Lake and stream	2.5	0.01	0.02
Total continental	149.0	5.55	827.0

Soil C storage has been estimated by using classifications of vegetation or soil, and multiplying the area of the class by its estimated mean soil C storage. For example, Schlesinger (1984) estimated soil C storage based on a land-use classification system while Batjes (1996) used the FAO-UNESCO soil classification. Estimates have also been based on other classification schemes; for example, the Holdridge life-zone classification system (Post *et al.*, 1982).

There is greater range in the estimates for soil C storage of the Earth (Table 1.3) than for vegetation C, although most recent studies appear to be converging at *ca.* 1,500 Gt C. Uncertainty in estimates of global soil C storage arise for several reasons: (i) the very high spatial variability in the C content of soils; (ii) unreliable estimates of area occupied by the classes of soils; (iii) lack of reliable field data, particularly bulk density; and (iv) the confounding effects of vegetation and land use changes (Eswaran *et al.*, 1993). Despite the many attempts at estimating global C stores, the estimates still contain large uncertainties (Schimel, 1995; Batjes and Sombroek, 1997).

Table 1.3 Estimates of global soil C storage

Global soil C pool (Gt; kg x 10^{12})	Reference
1,462-1,548	Batjes (1996)
1,576	Eswaran *et al.* (1993)
1,220	Sombroek *et al.* in Batjes (1996)
1,576	Eswaran *et al.* (1993)
1,515	Schlesinger (1984)
1,477	Buringh (1984)
1,395	Post *et al.* (1982)
2,070	Ajtay *et al.* in Post *et al.* (1982)
1,456	Schlesinger in Post *et al.* (1982)
1,392	Basilevich in Post *et al.* (1982)
1,080	Baes *et al.* in Post *et al.* (1982)
2,946	Bohn in Post *et al.* (1982)
700	Bolin in Post *et al.* (1982)

The soils with the greatest amount of C are peats (histosols), largely due to the high amount of C stored per square metre (Table 1.4; Eswaran *et al.*, 1993). Although, inceptisols cover at least ten times the area of histosols in the world, total C storage is slightly less than histosols due to their lower C per unit area. There is however, considerable uncertainty in the amount of C stored in peat soils, yet Gorham (1991) has estimated that northern peatlands alone contain *ca.* 450 Gt, and therefore approximately one third of the Earth's soil C. Other estimates for C stored in peatlands have been collated by Immirzi *et al.* (1992) and show a large range with values from 42 Gt to 1,500 Gt; this latter value is similar to other estimates for the total soil C storage of the world and clearly illustrates the uncertainty in estimates of terrestrial C pools.

Table 1.4 Mass of organic C in the world's soil (from Eswaran *et al.*, 1993)

Order	Area (m^2 x 10^9)	Mean soil C (kg C m^{-2})	Total soil C (Gt; kg x 10^{12})
Histosols	1,745	204.6	357
Andisols	2,552	30.6	78
Spodosols	4,878	14.6	71
Oxisols	11,772	10.1	119
Vertisols	3,287	5.8	19
Aridisols	31,743	3.5	110
Ultisols	11,330	9.3	105
Mollisols	5,480	13.1	72
Alfisols	18,283	6.9	127
Inceptisols	21,580	16.3	352
Entisols	14,921	9.9	148
Misc. land	7,644	2.4	18
Total	135,215	11.7	1,576

1.2.2 Terrestrial C storage in the United Kingdom

Several estimates of terrestrial C storage have been made for Great Britain (Howard *et al.*, 1995; Milne and Brown, 1997), and Cruikshank *et al.* (1996) have evaluated stores in Northern Ireland. Other estimates for terrestrial C storage in the United Kingdom (UK) include Harrison *et al.* (1995a) for forest soils in the UK, and Cannell and Milne (1995) who also reported estimates for the size of British vegetation C stores.

The estimates of vegetation and soil C storage in the UK are shown in Figures 1.3 and 1.4 respectively (figures provided by R. Milne, ITE Bush and derived from Milne and Brown, 1997 and Cruikshank *et al.*, 1996). Most vegetation C is stored in woodlands in Britain (*ca.* 80 %; Cannell and Milne, 1995), and the map of vegetation C storage largely reflects the distribution of these ecosystem types (Figure 1.3).

Soil C storage in the UK shows a distinct spatial pattern, with greatest C storage in the north and west, and least in the south and east. In Great Britain, Howard *et al.* (1995) estimated that 87 % of soil C was in Scottish soils, despite the smaller area of Scotland compared to England and Wales. Furthermore, the importance of peat soils for C storage, shown to be important globally, is also the case in Britain; 75 % of all soil C was reported to

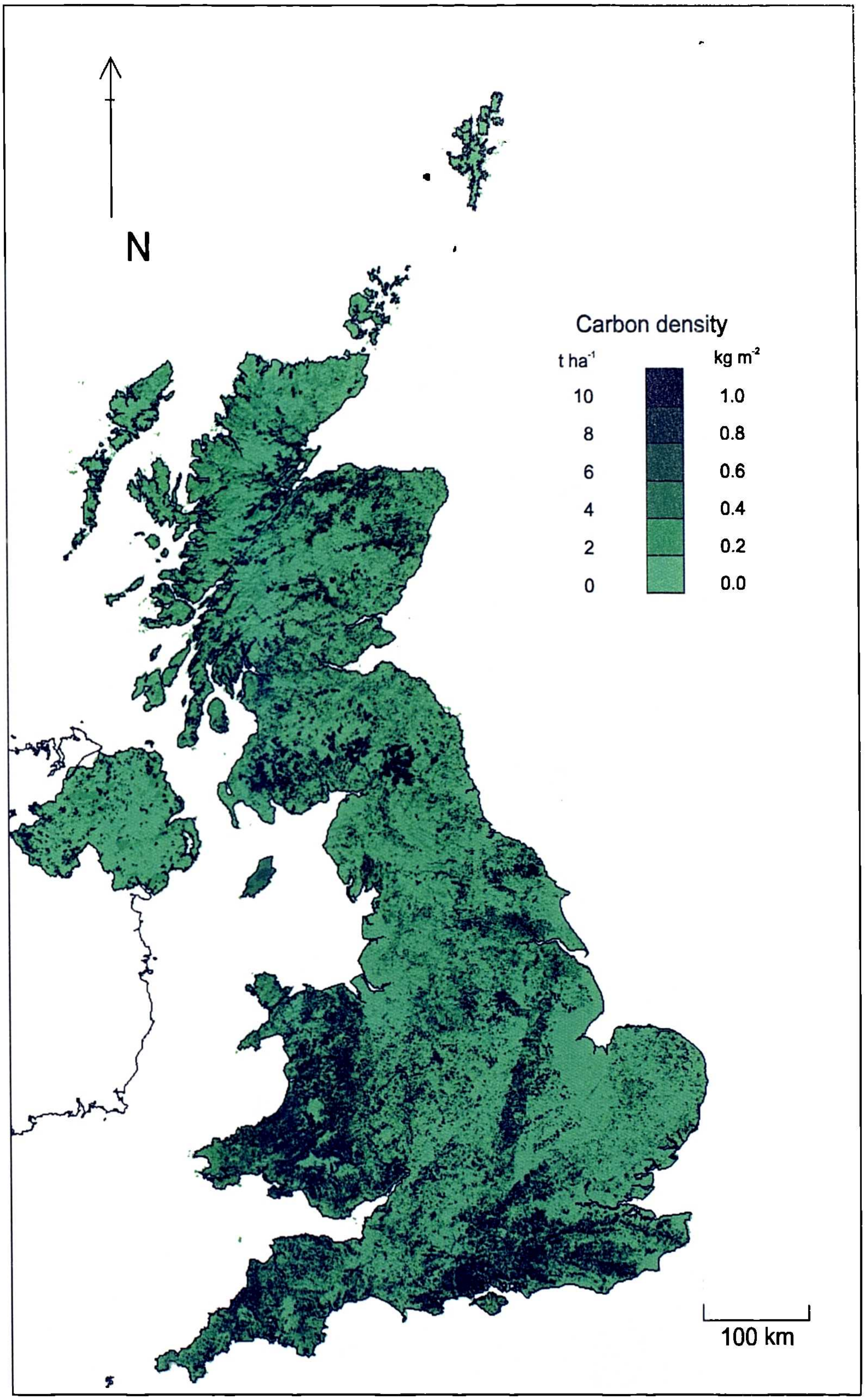

Figure 1.3 Vegetation C storage in the UK (provided by R. Milne, ITE Edinburgh)

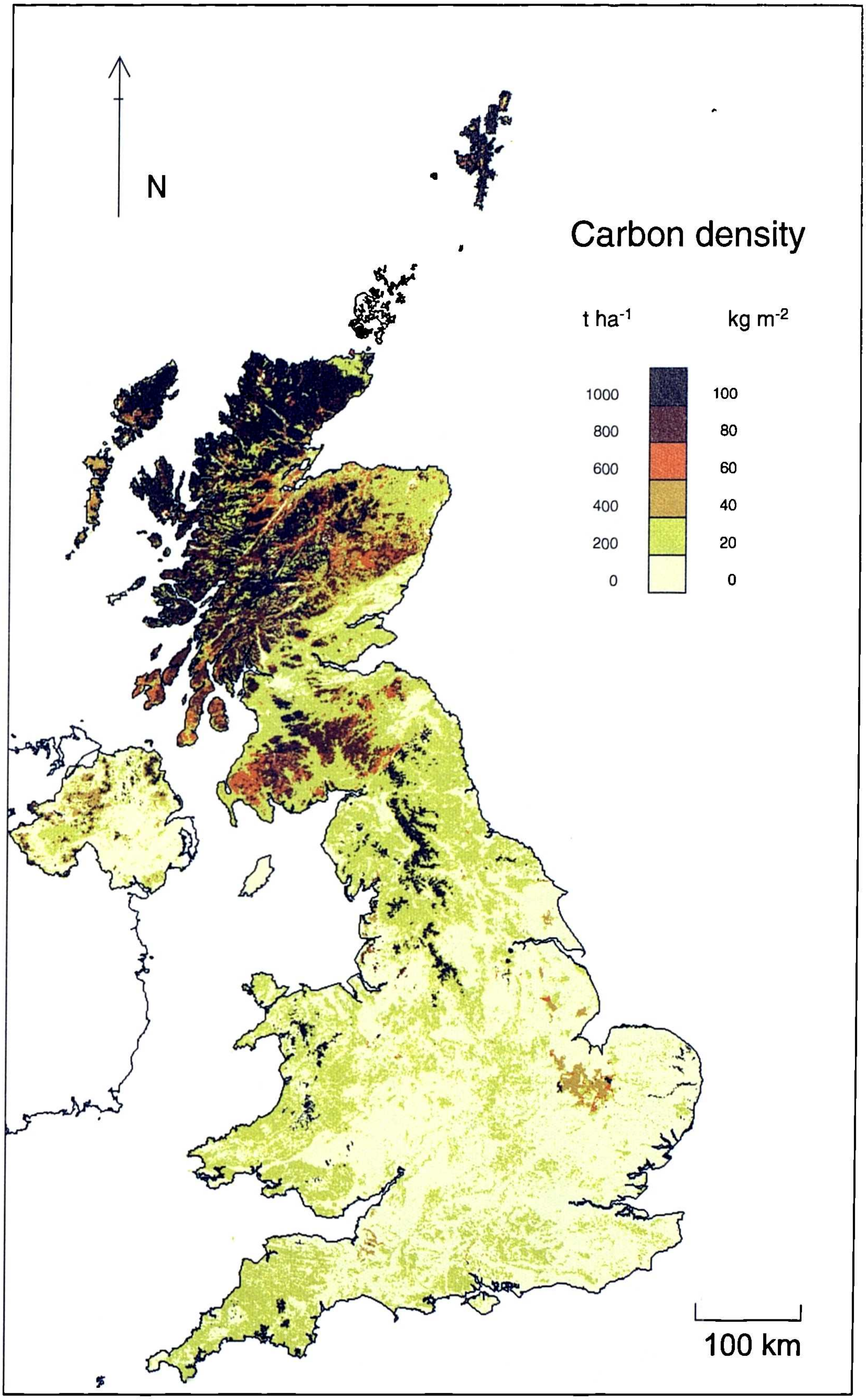

Figure 1.4 Soil C storage in the UK (provided by R. Milne, ITE Edinburgh)

be in Scottish peat soils by Howard *et al.* (1995). This is clearly evident from Figure 1.4 with peat soils in Britain being most dominant in the north and west.

The initial estimates for soil C storage made by Howard *et al.* (1995) were considerably higher than the estimates of Milne and Brown (1997) and this discrepancy clearly illustrates the present uncertainties in inventories of soil C stores. For Great Britain, Howard *et al.* (1995) estimated a total soil C storage of 21,784 x 10^9 kg (21,784 Mt), while Milne and Brown (1997) estimated 9,838 Mt, less than half the value given by Howard *et al.* (1995). The main reason for this discrepancy in estimates is considered to be in the quantification of the Scottish peatland C stores.

Milne and Brown (1997) used a mean bulk density for Scottish peats of 0.11 g cm^{-3} which was derived from analyses undertaken on Scottish peats by the Forestry Commission. In contrast, Howard *et al.* (1995) had no bulk density information for Scottish peats and assumed a bulk density value of 0.35 g cm^{-3} which was unrealistically high compared to Milne and Brown's (1997) data, and other estimates for peat bulk density (*e.g.* Clymo 1983). Therefore, for a one metre peat depth, the C content derived using Milne and Brown's (1997) bulk density value would be less than one third that derived by Howard *et al.* (1995). Since blanket peat soils alone comprise over 30 % of the area of all Scottish soils and have some of the highest C storage per square metre values, recalculation of the soil C for Britain using the revised bulk density value appears to be the major cause of the discrepancy in estimates of soil C. This example clearly illustrates some of the problems associated with making C inventories and that, despite improved methods of quantification, revised C inventories need to be validated.

1.3 Impact of change on terrestrial C stores

On their own, static national inventories of vegetation and soil C storage are of little use since it is the impact of change on the size of fluxes which affects atmospheric C concentration. However, in order to quantify the size of fluxes within the C cycle the accurate estimates of the size of C stores and the rate of C transfer are required. The previous section has highlighted the uncertainty in estimates of terrestrial C storage, however, the response of the C fluxes to change are also poorly understood.

1.3.1 Climatic impacts

The impact of climate on terrestrial C storage is complex because many factors which influence terrestrial C storage are influenced by climate in different ways (Melillo *et al.*, 1996). Furthermore, interactions between these processes exist which make predictions of the response of terrestrial C stores to climate change complex.

It has been suggested that vegetation C stores will increase following global warming since biological processes such as photosynthesis and plant growth generally increase with increasing temperature (Schimel *et al.*, 1995). Furthermore, the increased temperatures would mean longer growing seasons and a longer time for plants to grow and sequester C. However, since decomposition increases with increasing temperature it has been suggested that global warming could cause a reduction in soil C stores (Schimel *et al.*, 1995). Furthermore, water-logged soils (peats) contain large amounts of C because decomposition is suppressed and it is suspected that global warming may increase water table depths in these soils and cause increased rates of decomposition and consequent reductions in soil C stores (Gorham, 1991).

The impact of climate change is further complicated because increased rates of decomposition following warming will increase rates of CO_2 evolution from soils, and decomposition will also release nutrients available for new plant growth. In many ecosystems nutrient availability strongly limits plant growth and in these ecosystems, the increased recycling of nutrients may cause a fertilisation effect and increase plant growth rates and, consequently, C fixation from the atmosphere. This fertilisation effect may ameliorate or even reverse the overall loss of C (Schimel *et al.*, 1995).

The effects of climate change on terrestrial C storage have been investigated by studying past responses of the terrestrial C store to change, manipulating contemporary ecosystems to simulate climate change, and by investigating the underlying processes and modelling the impact of future climate change. The results of recent studies are reviewed here.

Several studies of soils along gradients of altitude and latitude (both simulating climate) provide evidence that increased global temperatures may result in a net loss of soil C storage:

Tate (1992) used a range of spatially distributed sites at different altitudes in the South Island of New Zealand to investigate the importance of climate on soil C storage, and to infer the impact of global warming. He found that soil C content was curvi-linearly related to the ratio of annual temperature / annual precipitation, and suggested that east of the Southern Alps, global warming would cause increased decomposition of soil C and a resultant loss of C from soils. Tate (1992) suggested that the loss of C could result in a 10 % reduction of the soil C storage of New Zealand by AD 2050, and cause a positive feedback to climate change.

Ineson *et al.* (in prep) simulated the impact of climate change on soils using a transplant approach (see Ineson *et al.*, 1998b). In this experiment, intact lysimeters containing three different soil types (peaty gley, brown earth and a podzol) were transplanted to various locations along an altitudinal transect at Great Dun Fell, northern England (see Chapter 6), with the reduced altitude corresponding to an increased annual temperature of up to 4.6°C. The authors found a significant reduction in soil C for peaty gley soil after being incubated for three years at a site with temperatures greater than 2.8°C warmer than usual, suggesting that global warming of this magnitude would cause a reduction in the amount of C stored in these soils. Furthermore, the amount of C lost in leachates was measured and suggested that increased mineralisation of soil C was responsible for most of the reduction in C storage.

The impact of past climate change on past C accumulation in peats has been investigated by Oldfield *et al.* (1997) in a study of C accumulation rates in a peatland (Stor Åmyrân) in northern Sweden. Oldfield *et al.* (1997) determined C accumulation rates in the mire for the last millennia using close interval radiocarbon dating and tephra analysis (see Chapter 5), and compared the results with a proxy record of past climate preserved in tree rings. Accumulation rates of C between AD 1400 and AD 1800 were found to be lower than for periods before and after this time period. Oldfield *et al.* (1997) acknowledged that the C accumulation rates near the surface of the mire would be high due to incomplete decomposition, though the lowest rates of C accumulation in the mire coincided with a time when tree ring evidence implied that spring-summer temperatures were lower. The results suggested that, for this site, net primary production of vegetation was more responsive to climate change than decomposition, with the implication being that global warming would lead to increased rates of C accumulation at this site, and therefore, contrary to Tate (1992) and Ineson *et al.* (in prep), provide a negative feedback to climate change.

The conclusion from the results of Oldfield *et al.* (1997) that C accumulation in organic soils may increase with increasing temperature is supported by evidence from Marion and Oechel (1993). They investigated rates of C accumulation in profiles along a latitudinal transect of tundra soils in Alaska, and observed that rates of C accumulation were significantly higher in the warm mid-Holocene than in the colder late-Holocene. These results suggested that following global warming, the rate of C accumulation in these sites would increase.

However, the results of Oldfield *et al.* (1997) and Marion and Oechel (1993) contrasted with those of Oechel *et al.* (1993) also working on Arctic tundra. Oechel *et al.* (1993) measured CO_2 fluxes from the tundra in the 1980's and 1990's and found that despite these soils having been long-term net accumulators of C, they had recently become CO_2 sources. This change from a net sink to a source was coincident with recent climatic warming yet Oechel *et al.* (1993) did not consider this shift to be a direct result of increased temperatures, but rather a consequence of lowered water tables caused indirectly by warming. Therefore, Oechel *et al.* (1993) suggested that, following global warming, these ecosystems would provide a positive feedback to climate change, and release significant amounts of soil C to the atmosphere.

The results of Oechel *et al.* (1993) may represent a transient response to climate change since over the long term, species changes may occur which could increase the rate of C input through increased rates of primary production. For example, in a study of 1,028 dated peat profiles from Finland, Tolonen and Turunen (1996) observed that C accumulation rates were nearly double in southern mires than northern boreal mires. If, following global warming, there is a northerly species shift as these authors suggest, *Sphagnum* bogs with higher C accumulation rates would spread north and cause C accumulation rates in northerly mires in Finland to increase. However, this would be a transitory response since it would take time for species to spread north following the warming.

A transitory response of terrestrial C storage to climate change may explain the long-term apparent positive response of C storage to warmer climates suggested by Oldfield *et al.* (1997) and Marion and Oechel (1993), but these interactions could have important short-term implications for atmospheric CO_2 concentrations.

The transitory response of terrestrial ecosystems to global change has also been investigated using models by Smith and Shugart (1993). They modelled the impacts of climate change caused by a doubling of the atmospheric CO_2 concentration and noted that although the long-term affect was to increase terrestrial C stores, the processes which resulted in a loss of C from the ecosystems (*e.g.* conversion of forest to grassland) responded more rapidly than the processes responsible for fixing more C (*e.g.* conversion of tundra to forest). The consequence is that, in the short-term, significant amounts of CO_2 would be released to the atmosphere causing a short-term positive feedback to warming.

Schimel *et al.* (1994) attempted to investigate the sensitivity of soil C storage to climate change using the CENTURY soil model. Temperature was found to be very important for soil organic C storage and Schimel *et al.* (1994) predicted that global soil C would decrease following global warming and despite increases in decomposition of soil organic matter leading to N fertilisation, these increases in productivity would only partially offset the loss of C caused by increased decomposition. However, other factors were found to be important for C storage and although not directly modelled by Schimel *et al.* (1994), it was noted that climate change could, if it results in a drying of soils, lead to a large loss of soil C from peats.

Other attempts to predict the impact of future climate change on soil C stores have been made using the CENTURY model by Parton *et al.* (1995). They investigated the potential impact of climate change on grassland production and soil C stores for 31 temperate and tropical sites and found that climate change would result in a decrease in soil C storage.

However, the increased CO_2 concentration of the atmosphere increased vegetation production within the models and reduced the size of the C loss from global grasslands, suggesting that tropical grasslands would become small sinks for CO_2. However, an important finding of this study was that the predicted changes in plant productivity would be difficult to measure in the field, due to their small size and the high annual variations in plant productivity.

Thus, the range of studies investigating the impact of global warming on terrestrial C stores currently provide conflicting predictions for the response of terrestrial ecosystems to global warming. The results from some studies clearly suggest that terrestrial C storage will increase following warming, while other results suggest the opposite and that, as a result, terrestrial C stores will augment fossil fuel CO_2 emissions following global warming. The literature suggests that the responses of terrestrial C stores to global change are likely to be complicated, with important transient responses.

Despite the conflicting results about whether terrestrial ecosystems will be net sources or sinks for CO_2 following warming, the literature suggests any such changes could be globally significant, and that further investigation of the impact of climate change on terrestrial C storage is required with the results being incorporated into models attempting to predict future climate change (Henderson-Sellers and McGuffie, 1995).

1.3.2 Direct anthropogenic impacts

Direct anthropogenic impacts on terrestrial C stores are largely a consequence of changing land use frequently resulting in reductions in terrestrial C storage. However, there is interest in investigating whether terrestrial ecosystems could be managed to increase the size of terrestrial C stores to offset industrial emissions of CO_2.

Houghton (1995) investigated the impact of global land use change between 1850 and 1980 on terrestrial C stores. He estimated that the increased area of croplands, pastures and shifting cultivation reduced the area of forest by 6×10^6 km^2 and caused a net transfer of *ca.* 100 Gt C from terrestrial stores to the atmosphere. In addition, this land use change resulted in 23 Gt of C being moved from live vegetation to dead plant material and wood products over the same period. At the same time, 10.4×10^6 km^2 of woodland were estimated to have been logged, causing a net loss of 23 Gt C to the atmosphere between 1850 and 1980. Houghton (1995) states that, although land use changes and associated rates of CO_2 emissions have decreased in temperate and boreal areas in recent decades, tropical land use change has increased and accounts for most of the CO_2 currently emitted from terrestrial ecosystems. Houghton (1995) estimated the annual net rate of transfer of C from the terrestrial pool to the atmosphere as 1.6 ± 0.7 Gt C yr^{-1} over the 1980's.

The greatest impact of anthropogenic activities on the vegetation C store has been through deforestation. Destruction of tropical rainforest has been estimated to have released an average of 1.6 Gt of C per year during the 1980's (Schimel, 1995), which is equal to the global rate of C lost from land-use change estimated by Houghton (1995), clearly illustrating the uncertainty in these estimates of C fluxes, but stressing the importance of rainforest destruction in anthropogenic CO_2 emissions. Furthermore, it is likely that large amounts of C are lost from tropical forest soils since Eswaran *et al.* (1993) showed that these soils contain a large proportion of the Earth's total soil C store.

Other anthropogenic influences on terrestrial C stores include the build up of organic matter in northern hemisphere forests as a result of fire suppression (Schimel, 1995) while regrowth of some forests following clearing is allowing terrestrial C to accumulate (Schimel, 1995). Furthermore, changes in agricultural practice also affect the size of terrestrial C stores, with for example, increased C storage in areas of agricultural set-a-side in Britain (Milne *et al.*, 1997).

Terrestrial C inventories have illustrated the importance of peatlands for terrestrial C storage (see above). These ecosystems are particularly sensitive to change and, consequently, anthropogenic activities which affect peatlands may also have important influences on the total terrestrial C storage. However, relatively little is known about the impact of management activities on the C dynamics of these ecosystems, although global peatlands have been estimated to release 182-272 Mt of C to the atmosphere on an annual basis (Immiriz *et al.*, 1992). These fluxes represent the equivalent of between 11 and 17 % of the annual rates of C emitted from deforestation of tropical rainforest (Schimel, 1995).

The main land use impacts on peatlands are drainage, afforestation and exploitation for fuel and horticultural peat. Drainage of peatlands has been undertaken by humans for centuries in an effort to improve the agricultural usage of peatlands. Immirzi *et al.* (1992) have reviewed the areas of drained mire in the world and estimated the total loss of C caused by C mineralisation after drainage, reporting that two thirds of the C transferred from peatlands to the atmosphere has been caused by drainage, particularly in Florida and tropical areas.

Afforestation has also been a major land use change affecting peatlands, normally being accompanied by drainage operations prior to plantation. An estimated *ca.* 120 x 10^6 km^2 of peatland have been drained prior to afforestation (Immirzi *et al.*, 1992). However, there is uncertainty whether this land use change represents a net loss of C from peatlands through increased decomposition of peat, or a net gain due to C sequestration by trees. For example, Cannell *et al.* (1993) investigated the impact of conifer plantations on drained peatlands in Britain, and suggested that the gain in terrestrial C storage from tree growth could be exceeded by the long-term loss of C from peat, particularly if the planted areas contained deep peats. However, Harrison *et al.* (1997) found, by comparing forested and non-forested parts of a Scottish raised mire, that C apparently accumulated more rapidly in the peat following afforestation, possibly through increased C accumulation on the peat surface from needle litter. When C sequestered by the growth of the trees was included, the total terrestrial C storage increased in the afforested areas by an average rate of between 0.6 and 0.8 kg C m^{-2} yr^{-1}.

The area of peatland affected by exploitation for fuel and horticultural usage has also been reviewed by Immirzi *et al.* (1992). Peat has been used for fuel for many centuries although domestic usage has greatly declined. The industrial use of peat has, in contrast, rapidly increased particularly for the generation of electricity (Immirzi *et al.*, 1992), although is still lower than extraction for horticultural usage. Despite the large amounts of peat being exploited for fuel and horticultural usage, there is little useful information about how these practices affect global emissions of CO_2, although Cruikshank and Tomlinson (1997) have recently estimated that around 0.2 Mt of C is annually emitted to the atmosphere in the UK through these processes.

Potential to increase terrestrial C stores through management options

Smith *et al.* (1997) considered the potential of five different management options for increasing terrestrial C storage in European soils as a way of mitigating increases in atmospheric CO_2. Their investigation was based on the extrapolation of long-term studies and examined the following scenarios: (i) amendment of arable soils with animal manure; (ii) amendment of arable soils with sewage sludge; (iii) incorporation of cereal straw into the soil; (iv) afforestation of surplus arable land through natural woodland regeneration; and (v) extensification of agriculture through ley-arable farming (Smith *et al.*, 1997). Of these options, only extensification of agriculture and afforestation were found to offer any significant benefits for increasing terrestrial C storage, with a possible increase in European soil C stocks of *ca.* 8 % and 17 % over a century, respectively. However, the authors concluded that, despite these potential increases in soil C storage, the actual amounts of C sequestered by adopting these management options would not greatly contribute to mitigating anthropogenic emissions of CO_2, and that the most practicable means to reducing CO_2 concentration increases was through halting tropical forest destruction and reducing fossil fuel burning.

Afforestation has been investigated as a possible land use approach for sequestering more C in terrestrial ecosystems. Harrison *et al.* (1995a) investigated this for the UK by calculating the amount of C which would be sequestered if all the land currently under crops and on soils with low C contents was afforested with deciduous woodland. The result was an increase in vegetation C storage of 255 Mt, or equivalent to about 2 years fossil fuel emissions, within the UK. Clearly, afforestation on this scale is not feasible in Britain and it is unlikely that this management option could be used to significantly offset the amount of C released through other anthropogenic activities.

Other options have been proposed for ways in which anthropogenic activities could be modified in order to reduce the loss of C from the terrestrial C pool. However, changes in land use to sequester more C do not appear to offer great potential to mitigate anthropogenic CO_2 emissions and it is clear that reductions in the present land use activities which are currently responsible for the transfer of large amounts of C from the terrestrial C pool to the atmosphere should be restricted (Dixon and Krankina, 1994; Smith *et al.*, 1997). However, the impacts of many land uses on the terrestrial C store are poorly understood and need to be investigated to enable better quantification of current C fluxes and enable appropriate management options to minimise greenhouse gas emissions.

1.4 Objectives of the thesis

It is apparent from the above that considerable uncertainties exist about the impact of the anticipated global warming on terrestrial C stores. The uncertainty about whether terrestrial C stores will act as a sink or source for atmospheric CO_2 is increased by the considerable uncertainties in estimates of the quantity of C stored in terrestrial ecosystems. In response, this thesis aimed to address these gaps in our present understanding through a detailed investigation of terrestrial C storage in an area of typical British moorland. The specific objectives were:

1) ***To quantify the C stored in the terrestrial ecosystem of the study site***

2) ***To predict the impact of the anticipated global warming on the size of this terrestrial C store***

3) ***To investigate the potential of management options for increasing terrestrial C storage at the site.***

Besides these principal objectives there are other, additional objectives which also form important components of the present investigation:

4) ***To test the accuracy of the UK's national C inventory***

5) ***To investigate the impact of past climate change on C accumulation at the study site***

6) To measure contemporary fluxes of C at the site and develop a predictive C balance model

7) To quantify present C fluxes at the site

1.5 Introduction to the study site

The site chosen for the present investigation was Moor House National Nature Reserve (NNR; Figure 1.5; National Grid ref. NY70 30), an area of Pennine moorland representative of the uplands of Britain (Heal *et al.*, 1975). It was also considered an area of high C storage having been intensively studied in the past. Consequently, a large amount of data were already available for the site which could be used in the present investigation (*e.g.* detailed digital soil and vegetation maps; see Table 1.5).

Table 1.5 Details of digitized maps used in the present investigation

Coverage	Original source	Digitized by	Digitised map supplied by
Topography	Ordnance Survey	R. Pilbeam, Institute of Terrestrial Ecology	Environmental Change Network
Vegetation	Eddy *et al.* (1969)	R. Pilbeam, Institute of Terrestrial Ecology	Environmental Change Network
Soil	Johnson and Dunham (1963)	R. Pilbeam, Institute of Terrestrial Ecology	Environmental Change Network

Moor House is located in the northern Pennines of England and is approximately 15 km south east of Penrith and 30 km west of Durham. The Reserve is divided along a north-south axis by a ridge of high ground known as the Dun Fells (Great and Little Dun Fell; altitudes 847 m and 842 m, respectively), with the highest peak of the Pennines (Cross Fell; altitude 893 m) being just outside the northern boundary of the Reserve (Figure 1.5). This ridge forms a major watershed between rivers flowing westwards to the Irish Sea, and east flowing rivers draining to the North Sea.

The ridge of high ground traversing the Reserve serves to divide the contrasting nature of the east and west sides. The east is typified by gently sloping hills of the Pennine dip slope (Johnson and Dunham, 1963), while the west contains steeper slopes being the

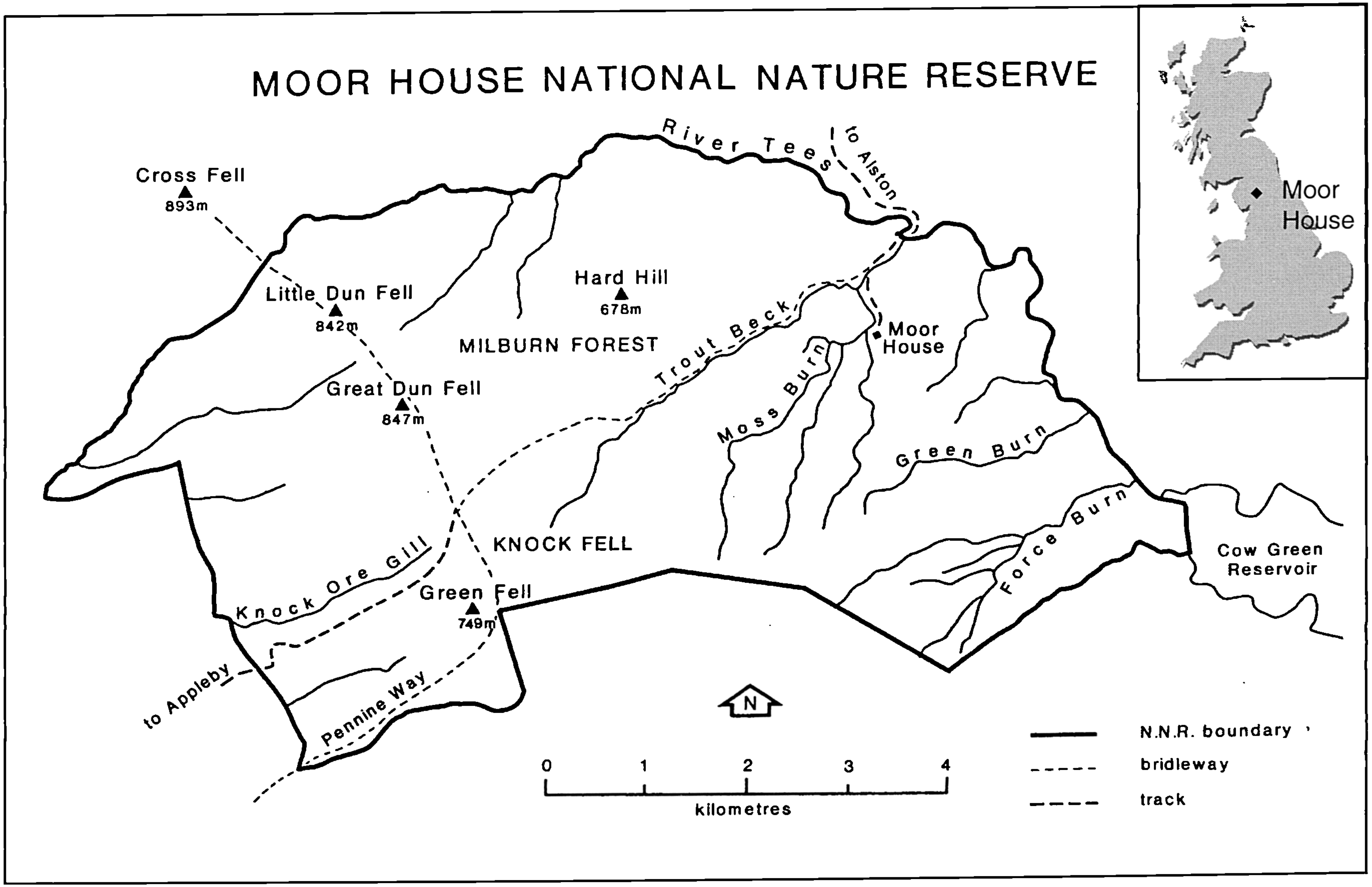

Figure 1.5 Map of Moor House NNR showing location within Great Britain

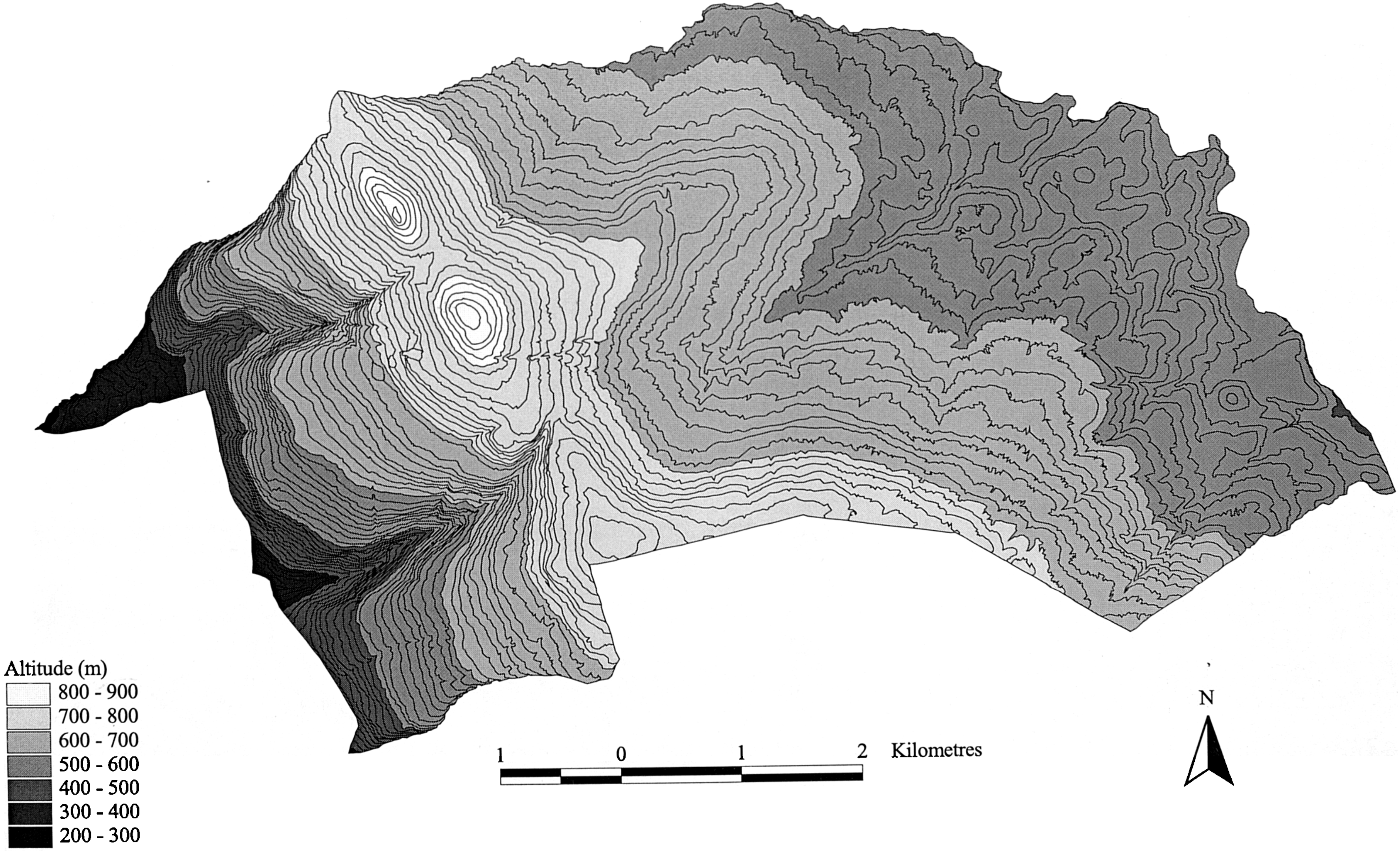

Figure 1.6 Topography of Moor House National Nature Reserve

scarp slope of the Pennines (Figure 1.6). The differences in topography across the Reserve are pronounced, due to previous glacial activity; the gentle hills of the east are mainly covered and smoothed by glacial till, although till is rarer to the west of the Pennine ridge. The contrasting geomorphology of the two sides of the Moor House Reserve has greatly influenced the pattern of soil and vegetation within the site (see Chapters 2 and 3 for detailed descriptions).

The research undertaken at Moor House over the last forty years has included hydrology (*e.g.* Conway and Millar, 1960), nutrient dynamics (*e.g.* Harrison *et al.*, 1994), effects of sheep grazing (*e.g.* Rawes and Hobbs, 1979), vegetation productivity (see Smith and Forrest, 1978) and decomposition of organic matter (*e.g.* Latter *et al.*, 1998). Moor House was the main moorland ecosystem in the UK during the International Biological Programme (IBP; Heal and Smith, 1978) and was a major site within the Natural Environment Research Council's (NERC's) Terrestrial Initiative in Global Environmental Research (TIGER) Programme (*e.g.* Ineson *et al.*, 1998a). Moor House is a founder site of the UK Environmental Change Network (ECN) and a wide range of ecological variables are currently being monitored at the site (see Garnett and Adamson, 1997).

Storage and fluxes of C at Moor House

The main stores and fluxes of C at Moor House, have been identified and are illustrated in Figure 1.7.

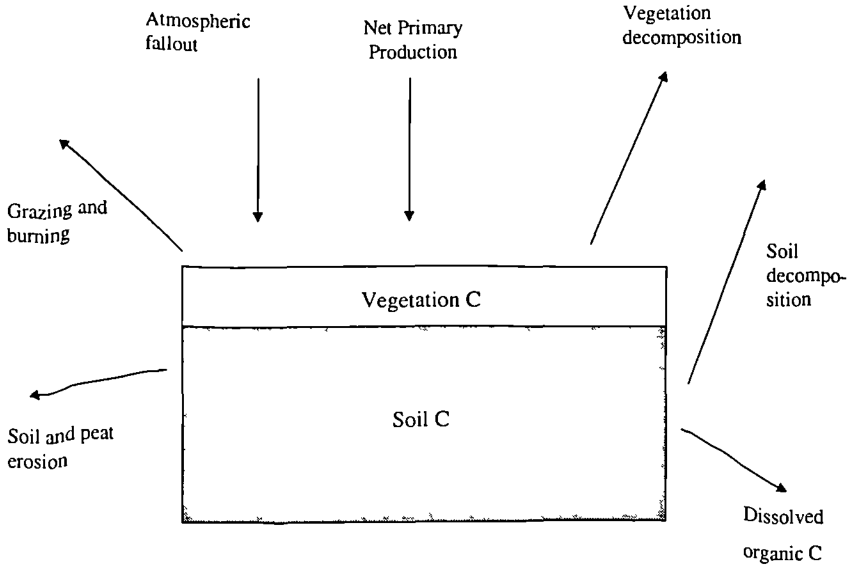

Figure 1.7 Stores and fluxes of C at Moor House NNR

2. Present carbon storage in vegetation at Moor House

2.1 Introduction

To predict the impact of future climates and management on the size of terrestrial C stores, accurate estimates of the present size of these stores are necessary. This was realised at the Rio Earth Summit in 1992 when signatories to the Framework Convention on Climate Change committed nations to evaluating terrestrial sources and sinks of C (United Nations, 1993). In accordance with the Convention, the UK now has a national inventory of terrestrial C stores (Milne and Brown, 1997). The accuracy of this national C inventory has been investigated within the present study by comparison with a detailed quantification of terrestrial C stores at Moor House NNR; this chapter describes quantification of the vegetation component.

The national C inventory has been produced at a one-kilometre scale, with each National Grid kilometre square of the Ordnance Survey in Great Britain being assigned an estimate of vegetation C storage. To test the accuracy of this inventory, vegetation C has been evaluated for each kilometre square which is entirely contained within the NNR; a total of 22 one-kilometre squares (Figure 2.1) representing approximately two thirds of the area of the Reserve are included. The C storage values for these squares, derived in this case study, have been compared directly with the estimates for the same squares in the national inventory.

The quantification of vegetation C stores at Moor House required i) a classification of the vegetation, ii) determination of the area of each of these classes using digital maps, and iii) estimation of the C stored per square metre (C density[1]) for each class. The vegetation C density values were derived from published results from previous work mainly carried out at Moor House. Consideration of the importance of deriving C storage estimates based on traditional two-dimensional (2d) map projections is described for producing inventories of vegetation stores. Estimates of C stores are usually based on 2d areas (*e.g.* the UK's national C inventory) and will always be underestimates of the true surface area, unless it is perfectly flat. The consequences of ignoring this effect in estimating vegetation C stores are evaluated within this chapter.

The vegetation of Moor House has been studied intensively for several decades being a major UK site within the International Biological Programme (Heal and Perkins, 1978) and

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[1] The term 'C density' has been used to describe the C stored per square metre, as adopted by Milne and Brown (1997) and others, though strictly speaking the values are not true densities

other projects. Consequently, a large resource of published and unpublished material exists on the vegetation of the site, including the results of extensive studies of the productivity of the vegetation and a detailed map of vegetation classes within the Reserve (Eddy *et al.*, 1969). Evaluation of vegetation C storage has been made using published data from these studies, with no additional field sampling needed.

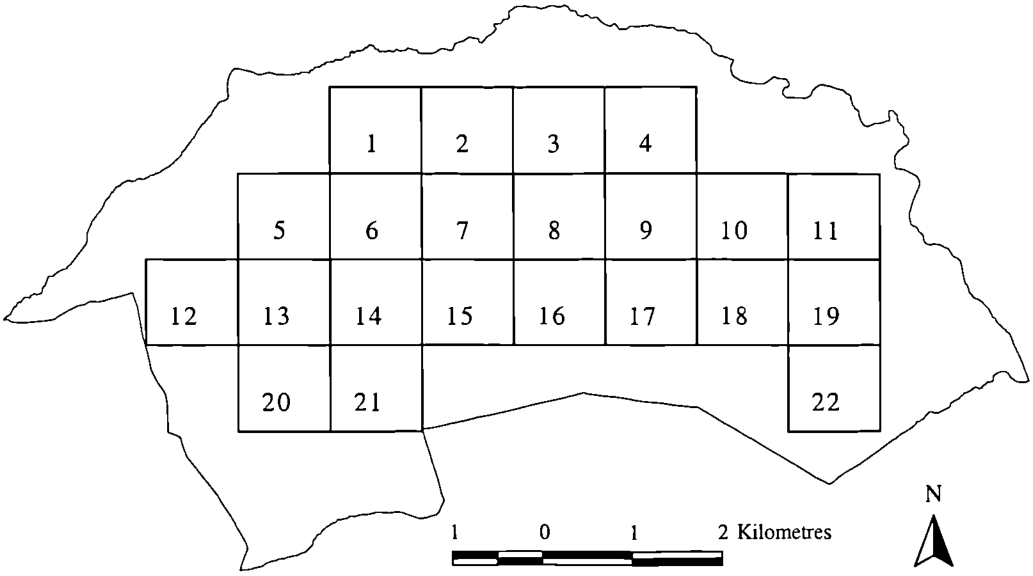

Figure 2.1 The 22 one-kilometre squares of the study site contained within the boundary of Moor House NNR. Reference numbers for each kilometre square are shown and National Grid references for each square are provided in the Appendix (Section 10.1).

2.1.1 Description of the vegetation at Moor House

The Moor House vegetation has been described in detail and classified by Eddy *et al.* (1969) while Heal and Smith (1978) provide a more general description. The vegetation coverage of the 22 one-kilometre squares which form the study site, derived from a digitised version of the vegetation map (see Table 1.5 for details) of Eddy *et al.* (1969) is shown in Figure 2.2. The original map was produced from extensive surveys of the Reserve in the 1960's, and the vegetation parcels appear to be largely unchanged today (personal observation); the site has been designated a Nature Reserve since 1952 and no major changes in management have occurred.

The Reserve is dominated on the eastern side by vegetation classes associated with blanket bog while extensive areas of upland grassland occur on the western side where slopes are steeper (Figure 1.6). The main vegetation units represented within the study area, together

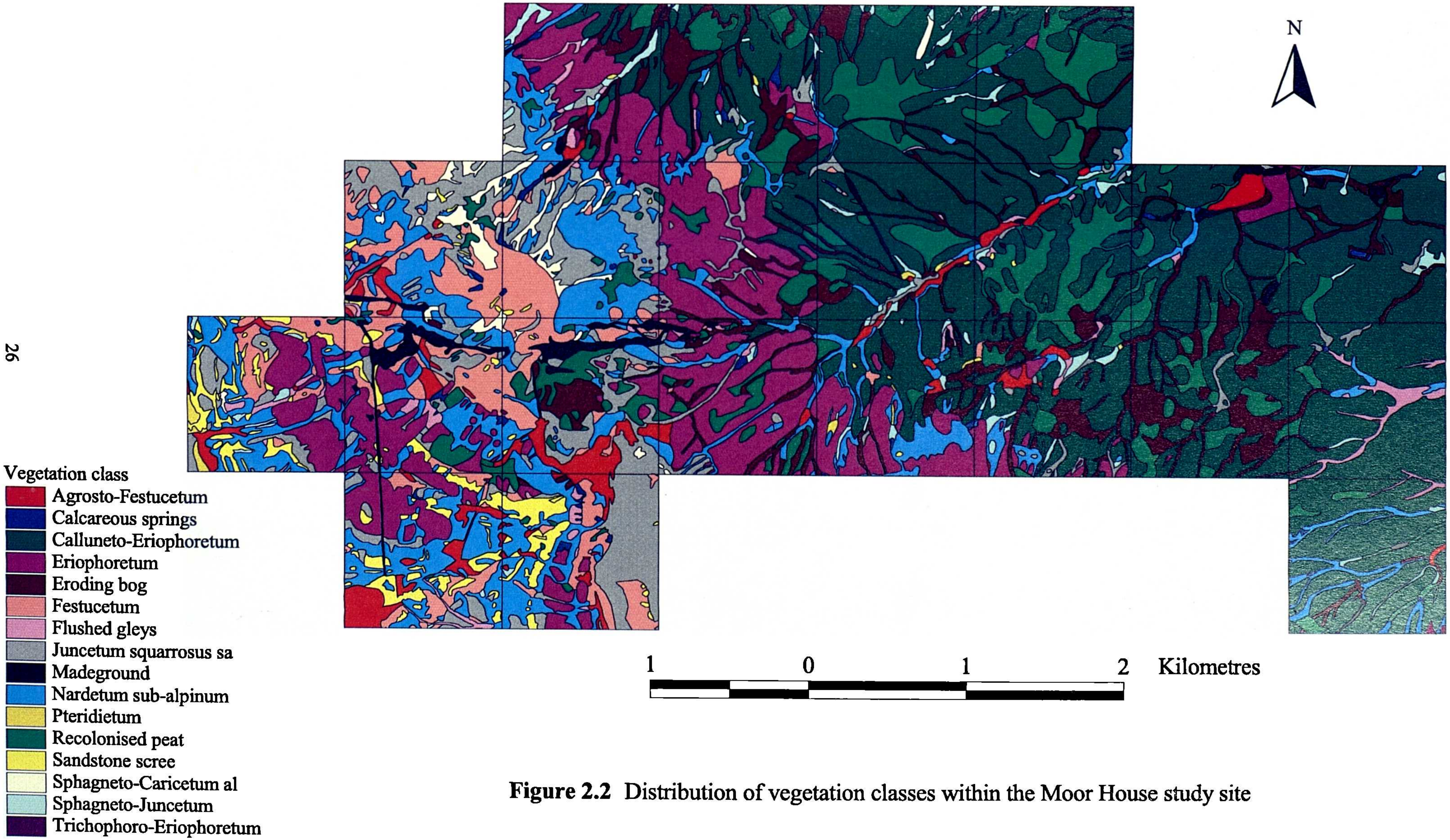

Figure 2.2 Distribution of vegetation classes within the Moor House study site

with major characteristics reported by Eddy *et al.* (1969), are briefly described in Table 2.1 and further details can be found in the original article by Eddy *et al.* (1969).

Table 2.1 Vegetation classes represented at Moor House

Vegetation class	Characteristics
Calluneto-Eriophoretum	A vegetation class dominated by *Calluna vulgaris* and *Eriophorum vaginatum*, though normally with *Sphagnum* sp., *E. angustifolium* and other species. It occurs on uneroded blanket bog below 630 metres altitude.
Eriophoretum	*C. vulgaris* appears to reach an altitudinal limit within the Reserve and seems unable to tolerate the intense grazing on the western side of the Reserve. Under these conditions *E. vaginatum* becomes the dominant species, while other species associated with this class include *Empetrum nigrum* and *Juncus* sp.
Recolonised peat complexes	Where peat has eroded and subsequently been recolonised by moorland vegetation communities, the vegetation has been classed as recolonised peat by Eddy *et al.* (1969). This class does not include areas where the peat has been completely eroded, but occurs where the vegetation was Calluneto-Eriophoretum, but the original mire surface was no longer present.
Eroding bog	This vegetation unit represents blanket bog in which more than about 30% of the original surface has been removed by erosion, and in which recolonisation has occurred on less than half of the newly exposed surface (Eddy *et al.*, 1969).
Juncetum squarrosi sub-alpinum (sa)	This species-poor vegetation class forms on wet gleyed soils and is characterised by *J. squarrosus*, though *Nardus stricta* is also well represented.
Nardetum sub-alpinum	This class is dominated by the grass *Nardus stricta*, though other species such as *J. squarrosus* and those normally associated with the Festucetum and Agrosto-Festucetum classes are present.
Festucetum	This class occurs mainly on the central ridge of the Reserve and consists of *Festuca* sp. and other grasses.
Agrosto-Festucetum	This vegetation unit is characterised by grasses and represents dry species-rich grasslands occurring mainly on limestone outcrops around the Reserve.
Sandstone scree	Bare rock and the vegetation growing between rocks form the 'Sandstone scree' class. It comprises a range of different vegetation species due to the wide variety of habitats within the class, though is frequently similar to the Festucetum class.

Table 2.1 (continued)

Madeground	Vegetation associated with man-made soils on spoil heaps, the result of past mining activity on the Reserve have been mapped as 'Madeground'. Based on the description of Eddy *et al.* (1969) the vegetation is similar to Agrosto-Festucetum.
Pteridietum	This class mainly occurs in small areas at the extreme west of the study site and is dominated by bracken (*Pteridium aquilinum*), though frequently there is an understorey of grasses.
Others	A few other vegetation classes occur within the study area, although they cover only minor areas. They are: Flushed gleys, Calcareous springs, Sphagneto-Caricetum alpinum (al), Sphagneto-Juncetum effusi (eff), Trichophoro-Eriophoretum. For descriptions of these minor classes, again, see Eddy *et al.* (1969).

2.2 Method

C stored within each vegetation class was evaluated by multiplying the area of the class by its C density, as shown in equation 2.1:

$$C_v = A_v \times U_v \qquad \textit{[equation 2.1]}$$

Where: C_v = C stored in vegetation class v (kg)

A_v = area of vegetation class v (m^2)

U_v = C density of vegetation class v (kg m^{-2})

The C density of vegetation classes was derived by multiplying measurements of above-ground biomass by the percentage C concentration of vegetation (see Section 2.2.3). The total vegetation C within the study site was calculated by summing the C storage values of each vegetation class. The vegetation C within each kilometre square was determined by multiplying the area of each vegetation class, within each kilometre square, by the C density value, then summing the storage values for each class represented in the square. The determination of the area and derivation of C density values are described in the following sections.

2.2.1 Area of vegetation classes at Moor House

Usually, estimates of C storage are based on 2d area projections (*e.g.* Howard *et al.*, 1995; Milne and Brown, 1997) and not the true, three-dimensional surface area. This has largely been through convenience since traditional maps provide only 2d projections and three-dimensional modelling of surfaces has only recently become widely available with developments in computer technology. However, a 2d area is an underestimate of the true surface area unless the area represented is perfectly flat (ESRI, 1991). This is rarely the case, and, as illustrated by the topography coverage (Figure 1.6), is certainly not the case for Moor House.

The national C inventory is based entirely on 2d area projections and may consequently underestimate the size of vegetation C stores. To investigate this possible error within the national inventory, the 2d and true surface area of the vegetation classes within the study site have been determined and the consequences of using these different area estimates on the final vegetation C storage values investigated.

Vegetation C stocks should, unlike soil C stores (see Section 3.2.2), be derived using true surface areas. The difference in area projection used for the vegetation and soil C storage estimates is largely a consequence of the method of field measurement used to obtain C density. The vegetation C densities were derived from studies where plots of vegetation were clipped, and the dry weight of organic matter determined after oven-drying. Above-ground biomass was calculated by dividing the weight of organic matter by the area of the plot. The size of the plots used for determining biomass was typically 0.5 m^2 (Forrest, 1971; Forrest and Smith, 1975) with the area measured along the surface of the ground. The plot area contained was therefore the true surface area, which, if on sloping ground, would be greater than if the plot was projected onto a 2d surface (see Figure 2.3).

In Figure 2.3, a vegetation plot of dimensions b multiplied by the width of the plot (*e.g.* 1 metre; plot area = 1 x b m^2) has been sampled from an area of uniform slope (*s* degrees). The plot is assumed to be representative of the vegetation on the whole slope (as in, for example, a field) with an area represented as T multiplied by several width units (*e.g.* 10 metres; *i.e.* 10 x T m^2), which is the true surface area of the field. The total vegetation biomass would therefore be the product of multiplying the vegetation biomass clipped in the plot (per unit area) by the area of the field. However, most studies simply ignore the difference between the true surface and 2d area and instead multiply the vegetation biomass per unit area, determined in the plot (which is based on the true surface area), by the 2d area provided by traditional maps. Hence, this results in an underestimate of the true vegetation biomass (and also C storage) for the field (since $F < T$; Figure 2.3). Therefore, vegetation C storage estimates should be determined using true surface areas.

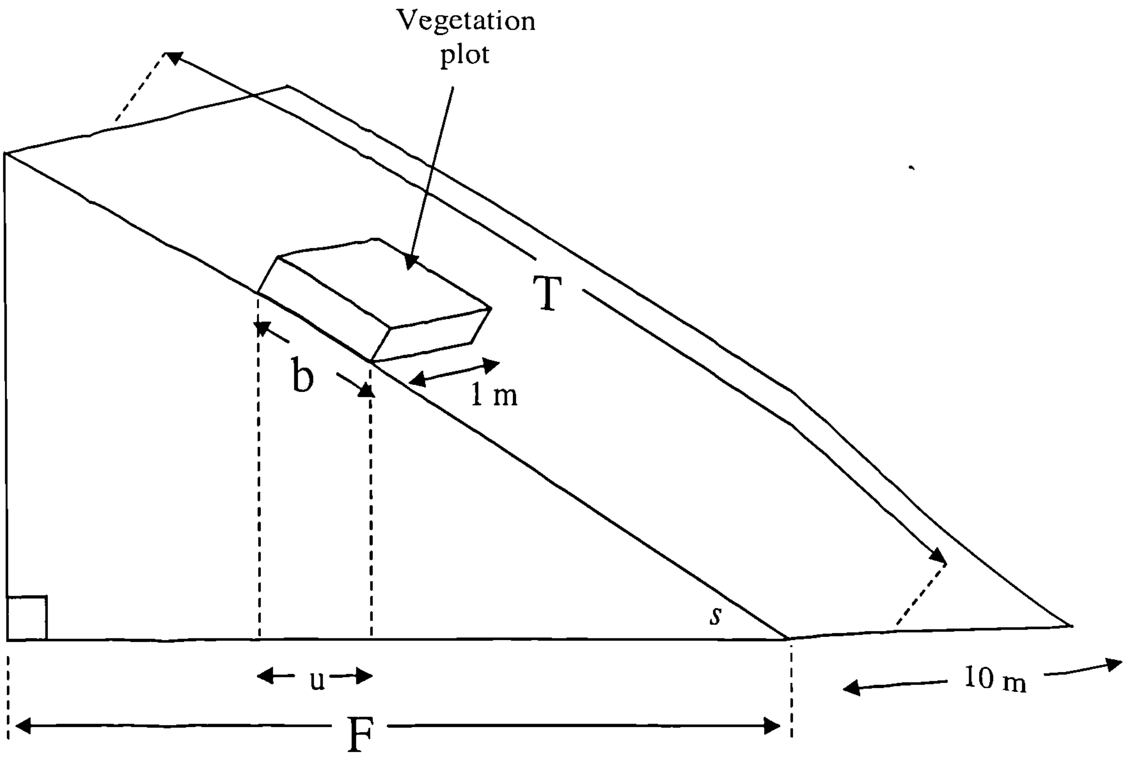

Figure 2.3 Theoretical parcel of vegetation on sloping ground showing a vegetation plot used to determine vegetation biomass and C storage (see text for details)

For example, we can assume that the vegetation parcel in Figure 2.3 is a linear field of 10 metres width, and therefore the size of the field is 10 x T metres. If T is 50 metres, the total area is 500 m^2. Next, we assume that the size of the clipped plot was 1 metre by 0.5 metres, giving an area of 0.5 m^2. If the weight of vegetation clipped was 0.5 kg, then the mass of vegetation per unit area would be 1 kg m^{-2}. The actual mass of vegetation for the whole field, as stated above, is simply the true surface area of the parcel multiplied by the vegetation mass per unit area, *i.e.* 500 x 1 = 500 kg. Therefore, the correct mass of vegetation in the field is 500 kg. However, in most studies the weight of vegetation determined from plots is multiplied by the 2d area. If we assume that the angle of slope *s*, is 10 degrees, then using trigonometry, the 2d equivalent of the distance T (represented as F on Figure 2.3) is equal to T x COS 10°, i.e. 49.24 metres. The 2d area is therefore 49.24 x 10 = 492.4 metres, and the estimated weight of vegetation for the slope is 492.4 kg. The example serves to illustrate that by using 2d areas for estimating vegetation biomass (or C storage), an underestimate of the true value will be made, the extent of which will depend on the severity of the slopes within the study area. Therefore, the best estimates of C storage for Moor House determined in this case study are based on true surface areas of vegetation classes.

2.2.2 Determination of true surface areas for the vegetation classes

True surface areas of the vegetation classes represented in the study area were estimated by constructing a three-dimensional model of the study site from the contours of the topography coverage of Moor House (Figure 1.6). The method outlined below was undertaken using Arc/Info© GIS and also allowed the average percentage slope for each vegetation class to be determined. The values can only be considered estimates of the true surface area since the surface area of a landscape will depend directly upon the scale it is being investigated (see below).

A Triangular Irregular Network (TIN) of Moor House was created, based on the 10 metre contours. A TIN is a three-dimensional model of a surface where triangles (satisfying the requirements of Delaunay triangulation; ESRI, 1991) represent the shape of the surface. The corners of the triangles have x and y co-ordinates which represent their 2d location, like co-ordinates on a map. The triangles also have a z value, which is the third-dimension and represents altitude in three-dimensional surfaces of landscapes, such as in the TIN of Moor House. Using these three values, the area and angle of slope for each triangle can be calculated (ESRI, 1991).

The TIN was then 'flattened' using a procedure in Arc/Info©, which reduced the triangles back to 2d, though the true surface area and slope attributes for each triangle were retained. This 'flattened' TIN was intersected with the vegetation coverage for the study area (see Figure 2.2), giving vegetation parcels (or polygons) containing the 2d area of the parcel but also providing the true surface area and slope of the parent triangle in which the vegetation parcel was located.

An example of a TIN being intersected by a vegetation coverage is provided in Figure 2.4. This diagram shows part of a 'flattened' TIN which has been intersected so that some of the triangles forming the TIN have been intersected by the boundary between two vegetation classes; the original (parent) triangle has been represented after the intersection by two polygons (1 and 2), of different vegetation classes. The true surface area of the new polygons can be calculated since the ratio of the 2d area to the true surface area, in both the original triangle and the newly-created polygons, are the same. Therefore, the true surface area of each vegetation parcel was determined using:

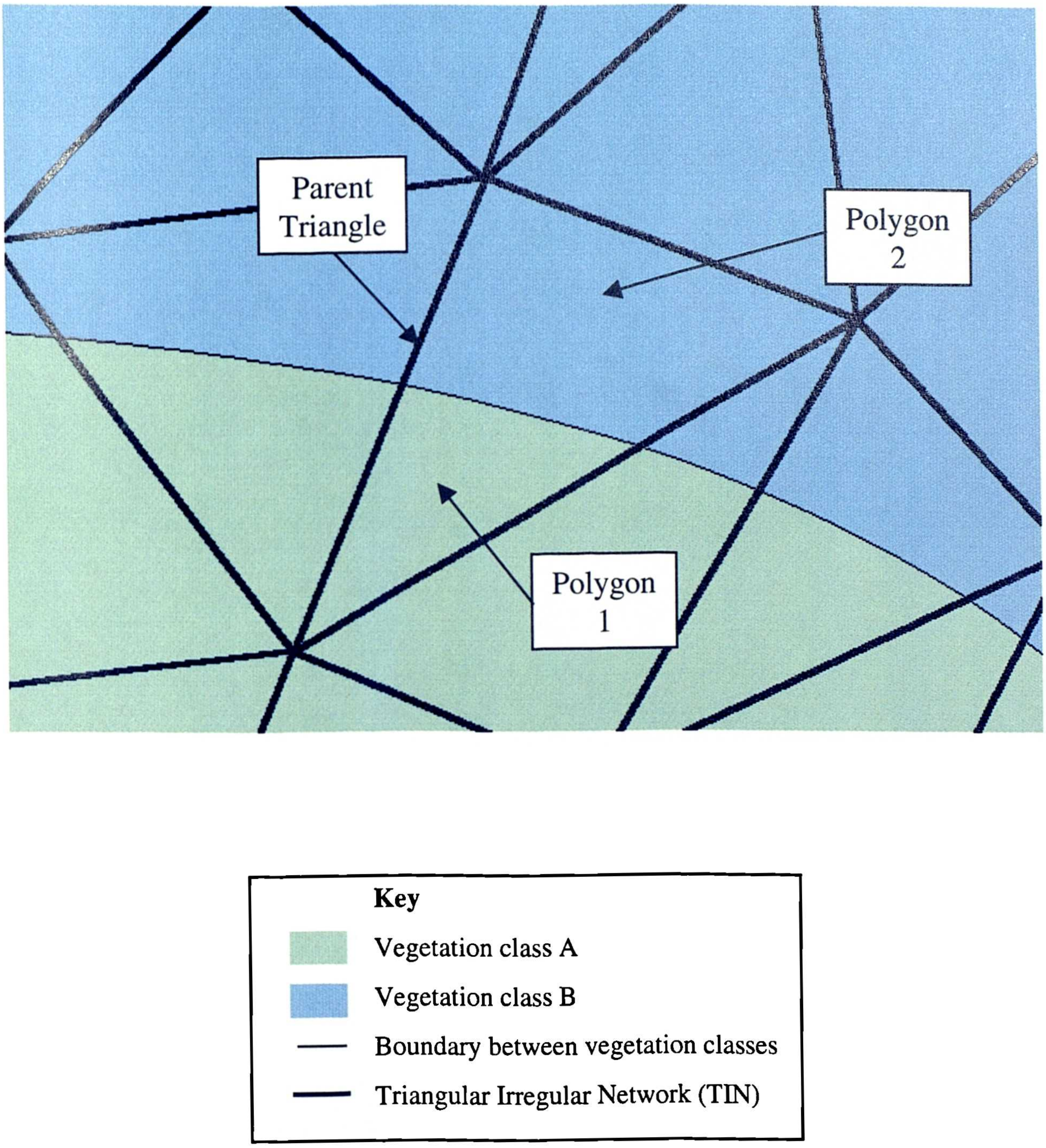

Figure 2.4 Diagram illustrating the intersection of a 'flattened' TIN by a vegetation coverage (see text)

$$\textit{True surface area} = (P_{2d} / T_{2d}) \times T_{sa} \quad \textit{[equation 2.2]}$$

Where:

P_{2d} = 2d area of the vegetation parcel produced after intersecting the 'flat' TIN and vegetation coverage (*e.g.* polygon 1)

T_{2d} = 2d area of the parent triangle in which the vegetation parcel was located

T_{sa} = True surface area of the parent triangle in which the new vegetation parcel was located

In reality, many of the original triangles were not dissected by the vegetation coverage so that their true surface area was the same as prior to the intersection, though they now represented a vegetation class. Therefore, the total true surface area of a vegetation class (*e.g.* if vegetation class A was Eriophoretum) was determined:

$$\textit{Total true surface area} = \Sigma T_{csa} + \Sigma\,((P_{2d} / T_{2d}) \times T_{sa}) \quad \textit{[equation 2.3]}$$

Where:

Total true surface area = True surface area of Eriophoretum in the study area

T_{csa} = True surface area of complete triangles not affected by intersection and containing Eriophoretum vegetation class

P_{2d} = 2d area of the resultant polygon formed from the intersection and containing Eriophoretum vegetation class (*e.g.* polygon 1)

T_{2d} = 2d area of the parent triangle in which the vegetation parcel was located

T_{sa} = True surface area of the parent triangle in which the new vegetation parcel was located

This procedure was used to determine the true surface area of each vegetation class within the Moor House study area (a detailed description of the procedure is provided in the Appendix; Section 10.2). The true surface area was also determined for each vegetation class within each individual kilometre square. The equivalent 2d area of each true surface area was also derived.

2.2.3 C stored in vegetation classes

Various terms have been used in the literature to describe mass of vegetation such as standing crop (Pearsall and Gorham, 1956; Rawes and Welch, 1969; Forrest and Smith, 1975), above-ground biomass (Smith and Forrest, 1978) and total biomass (Rawes and Welch, 1969; Forrest and Smith, 1975). Standing crop and above-ground biomass values have been used in this study to determine C storage, since these data represent the total dry weight of vegetation above ground in late summer, when biomass values are highest; for convenience, the term 'above-ground biomass' has been used here to encompass all of these measures.

The division between vegetation and soil C is not clear-cut because plant roots penetrate the soil system to varying depths, and can either be included as part of the soil or vegetation stores. However, estimations of plant root biomass are notoriously difficult and imprecise (Whittaker and Marks, 1975) and most production studies have concentrated on above-ground vegetation only. In evaluating the C storage of vegetation at the study site, only vegetative parts above the soil surface have been included and roots have been included in the soil stores.

By definition, above-ground biomass refers to the vegetation above the *Sphagnum* carpet where present (Smith and Forrest, 1978). Most workers have adopted this approach due to the difficulties in distinguishing between *Sphagnum* that is dead, alive or part of the peat (Clymo, 1970). *Sphagnum* was recorded as part of the soil C store, and in any case, was a small component in terms of the total C quantities in most vegetation classes.

Variation among published above-ground biomass values, even for the same vegetation class, can be great (see Forrest and Smith, 1975) and this variation is considered to be a reflection of the high natural heterogeneity of the vegetation. To provide a 'best estimate' of C in vegetation, mean values have been derived where available. Similarly, to obtain an idea of the range or maximum C storage within a vegetation class, the highest published biomass values have been found, providing some indication of an upper boundary for estimated values. The two estimates made in this way have been termed the mean estimate and the maximum estimate.

Where above-ground biomass values were unavailable from research carried out at Moor House itself, published results for the same vegetation class elsewhere have been used. For classes with no published above-ground biomass estimates, the values used have been derived from figures for similar communities, although this was only necessary for the rarest vegetation classes. More accurate assessments would require specific sampling of these classes, but it became obvious during the study that such sampling would not significantly improve the final C estimates. Fortunately, the more extensive vegetation classes have been the subject of intensive measurement (*e.g.* Calluneto-Eriophoretum; Forrest, 1971).

Allen (1989) reported the %C contents of vegetative parts of plants at between 43% and 47% of dry mass and, therefore, the values for above-ground biomass were converted to C density by assuming the mean C concentration of all vegetation was 45%. The estimate for maximum vegetation C assumed a C concentration of 47%.

2.3 Results

2.3.1 Area of vegetation classes at Moor House

Table 2.2 shows the 2d and true surface areas of the different vegetation classes for the study site. The Calluneto-Eriophoretum class was most dominant, covering about one third of the study site.

The true surface area of the entire study site was found to be only 1.24% greater than the 2d area, though the percentage increase in area varied with different vegetation classes (Figure 2.5). The proportional increase in area was directly related to the severity of slopes on which the vegetation existed (Figure 2.6). For example, the vegetation classes on blanket peat had a true surface area only slightly greater than the 2d area, reflecting the fact that blanket peat does not form on steep slopes with good drainage. However, the grassland classes associated with steeper well-drained slopes showed a large increase in area (5.42% for Agrosto-Festucetum) when the true surface and 2d areas were compared.

Table 2.2 Comparison of 2d and true surface area for vegetation classes within the study area

Vegetation class	2d area (m^2 x 10^3)	True surface area (m^2 x 10^3)	Difference (m^2 x 10^3)
Calluneto-Eriophoretum	7,376	7,415	39
Eriophoretum	2,896	2,919	22
Nardetum sub-alpinum	2,571	2,634	63
Recolonised peat	2,172	2,181	9
Eroding bog	1,827	1,834	8
Juncetum squarrosus sa	1,708	1,731	23
Festucetum	1,224	1,258	35
Agrosto-Festucetum	602	634	33
Sandstone scree	431	455	24
Sphagneto-Juncetum eff	291	294	3
Madeground	278	282	4
Flushed gleys	248	252	4
Sphagneto-Caricetum al	226	229	3
Calcareous springs	98	100	2
Pteridietum	52	53	1
Trichophoro-Eriophoretum	2	2	0
Total	22,000	22,273	273

The true surface and 2d areas of individual one-kilometre squares are shown in Table 2.3, and the percentage increases in area when the true surface was compared to the 2d area are displayed in Figure 2.7. Though the true surface area of the total study site was only slightly greater than the 2d area, there were substantial differences between individual kilometre squares (range 0.25 to 5.71% greater). This variation is attributed to the topography of the kilometre squares with the ones showing the greatest increases, after considering the true surface area, being located on the steeper western side of the study site (*i.e.* squares 5, 12, 20 and 21).

The 2d and true surface areas of each vegetation class for individual kilometre squares used to derive these summary tables and charts are provided in the Appendix (Section 10.3).

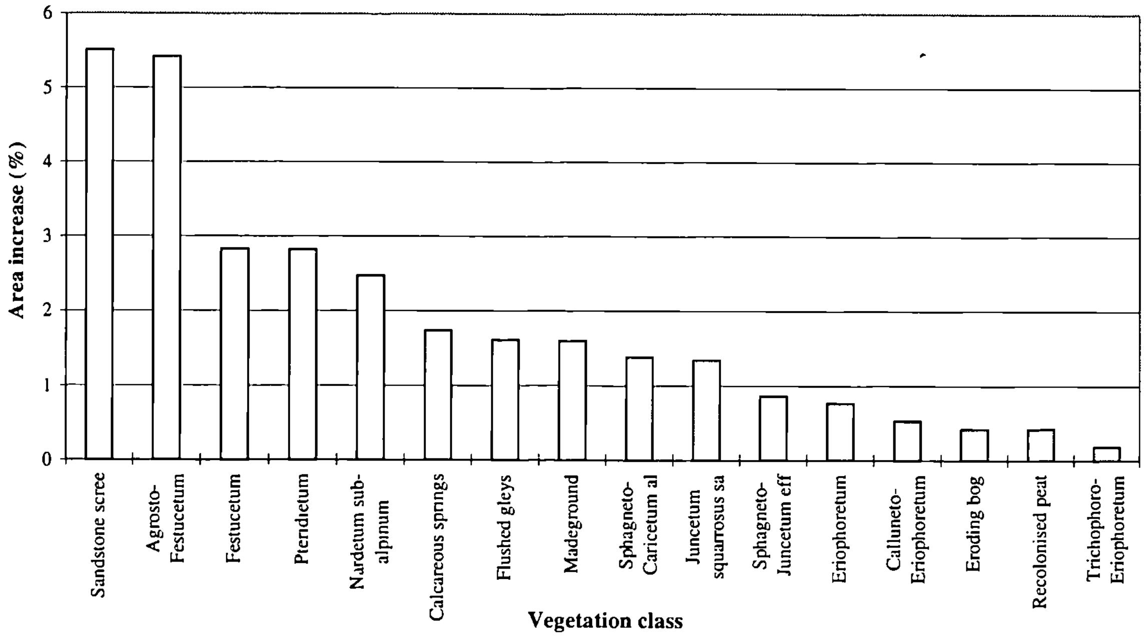

Figure 2.5 Percentage increase in area of true surface to 2d area by vegetation class

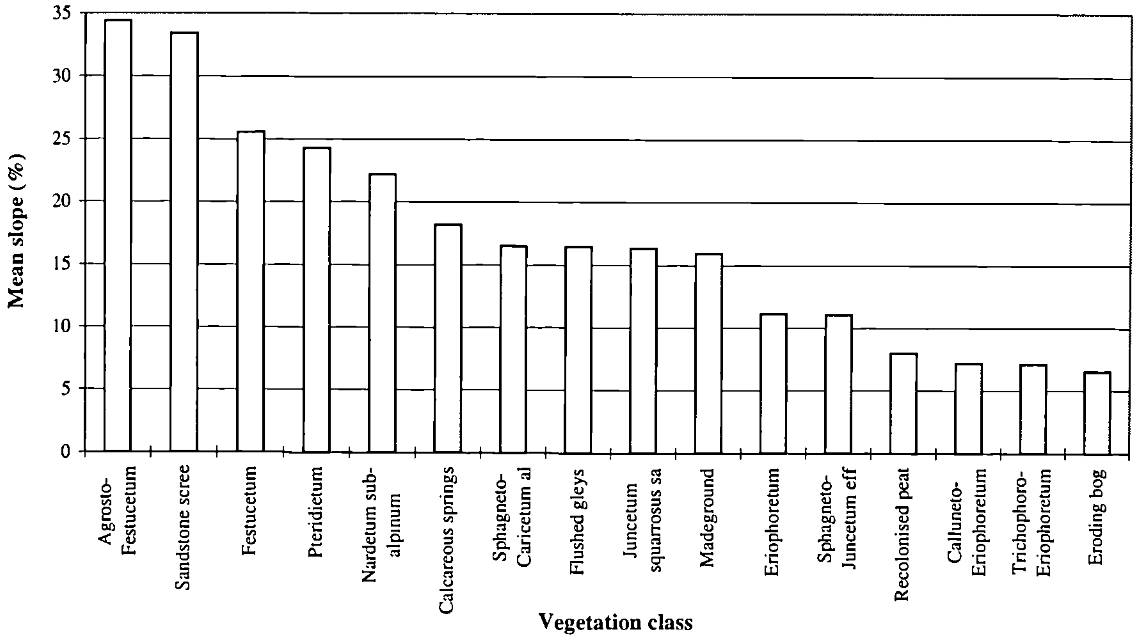

Figure 2.6 Mean percentage slope of the vegetation classes

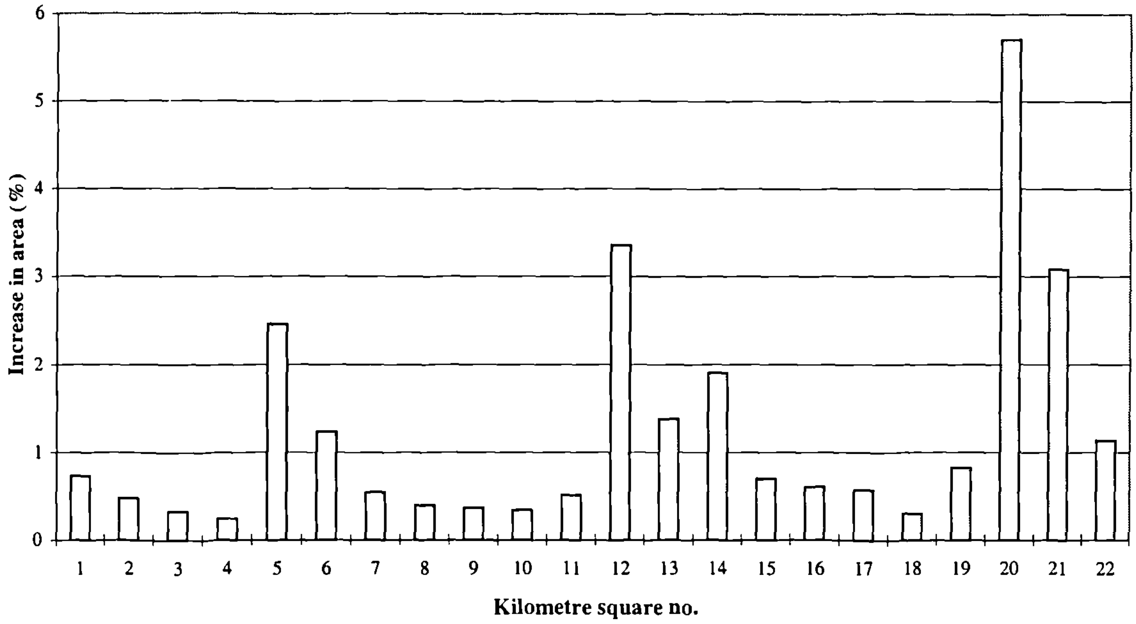

Figure 2.7 Percentage increase in area when comparing true surface and 2d areas, by kilometre square no. (see Figure 2.1 for square locations)

Table 2.3 Comparison of 2d and true surface areas by kilometre square

Kilometre square no. (See Fig. 2.1)	2d area (m^2 x 10^3)	True surface area (m^2 x 10^3)	Difference (m^2 x 10^3)
1	1,000	1,007	7
2	1,000	1,005	5
3	1,000	1,003	3
4	1,000	1,003	3
5	1,000	1,025	25
6	1,000	1,012	12
7	1,000	1,006	5
8	1,000	1,004	4
9	1,000	1,004	4
10	1,000	1,004	3
11	1,000	1,005	5
12	1,000	1,034	34
13	1,000	1,014	14
14	1,000	1,019	19
15	1,000	1,007	7
16	1,000	1,006	6
17	1,000	1,005	6
18	1,000	1,003	3
19	1,000	1,008	8
20	1,000	1,057	57
21	1,000	1,031	31
22	1,000	1,011	11
Total	22,000	22,273	273

2.3.2 C stored in vegetation classes of Moor House

The sources for mean and maximum above-ground biomass values used to determine C density are provided in Tables 2.4 and 2.5 respectively. These tables show which vegetation classes had C densities derived from measurements taken directly from the vegetation class at Moor House, and the classes for which there were no published biomass values available; these classes were assigned the same values as similar classes which had been studied at Moor House. The remaining minor classes, with no appropriate results from Moor House, were assigned biomass values determined from elsewhere, and are also shown. Table 2.6 presents the mean and maximum values of above-ground biomass and C density for each vegetation class represented within the study area.

Table 2.4 Description and sources of mean above-ground biomass values

Vegetation class	Reference	Details
Agrosto-Festucetum	Smith and Forrest (1978)	Biomass measurements made at Moor House under grazed conditions
Calcareous springs	Pearsall and Gorham (1956)	No specific information was available for this class. The mean value of four sedge communities found in similar habitats from across Britain were therefore used
Calluneto-Eriophoretum	Forrest (1971) in Smith and Forrest (1978)	Biomass measurements made at Moor House under grazed conditions
Eriophoretum	Forrest and Smith (1975) in Smith and Forrest (1978)	Biomass measurements made at Moor House under grazed conditions
Eroding bog	Forrest (1971) in Smith and Forrest (1978)	No specific information was available for this class, however, the class is similar to Calluneto-Eriophoretum (Eddy *et al.*, 1969). The values of above-ground biomass for that class were used
Festucetum	Rawes and Welch (1969)	Biomass measurements made at Moor House under grazed conditions. The value represented the mean from two grazed plots studied in 1962
Flushed gleys	Pearsall and Gorham (1956)	No specific information was available for this class. The mean value of four sedge communities found in similar habitats from across Britain were therefore used
Juncetum squarrosus sa	Rawes and Welch (1969)	Biomass measurements made from a single grazed location at Moor House in 1964
Madeground	Smith and Forrest (1978)	No specific information was available for this class, however, the class is similar to Agrosto-Festucetum (Eddy *et al.*, 1969). The values of above-ground biomass for that class were used
Nardetum sub-alpinum	Smith and Forrest (1978)	Biomass measurements made at Moor House under grazed conditions
Pteridietum	Pearsall and Gorham (1956)	No information on the biomass of this vegetation class was available for Moor House. The estimate used was derived from studies in New Forest and Bowland Forest.
Recolonised peat	Forrest (1971)	No specific information was available for this class, however, the class is similar to Calluneto-Eriophoretum (Eddy *et al.*, 1969). The values of above-ground biomass for that class were used
Sandstone scree	Rawes and Welch (1969)	No specific information was available for this class, however, the class is similar to Festucetum (Eddy *et al.*, 1969). The values of above-ground biomass for that class were used
Sphagneto-Caricetum al	Pearsall and Gorham (1956)	No specific information was available for this class. The mean value of four sedge communities found in similar habitats from across Britain were therefore used
Sphagneto-Juncetum eff	Pearsall and Gorham (1956)	No specific information was available for this class. The mean value of four sedge communities found in similar habitats from across Britain were therefore used
Trichophoro-Eriophoretum	Forrest and Smith (1975)	Direct measurements made at Moor House in 1970 (Cottage Hill site A)

Table 2.5 Description and sources of maximum above-ground biomass values

Vegetation class	Reference	Details
Agrosto-Festucetum	Rawes and Welch (1969)	Biomass measurements made at Moor House under grazed conditions. The value represents the highest encountered from several sites over five years
Calcareous springs	Pearsall and Gorham (1956)	No specific information was available for this class. The highest value from a sedge community found in similar habitats in Britain was therefore used
Calluneto-Eriophoretum	Forrest and Smith (1975)	Biomass measurements made at Moor House under grazed conditions. The measurement was made at Sike Hill in 1968
Eriophoretum	Forrest and Smith (1975)	Biomass measurements made at Moor House under grazed conditions. Only one source was found for this class so a maximum estimate was made by adding the reported standard error to the mean value
Eroding bog	Forrest and Smith (1975)	No specific information was available for this class, however, the class is similar to the Calluneto-Eriophoretum class (Eddy *et al.*, 1969). The values of maximum above-ground biomass for that class were used
Festucetum	Rawes and Welch (1969)	Biomass measurements made at Moor House. The value represents the highest encountered from several sites studied in 1962
Flushed gleys	Pearsall and Gorham (1956)	No specific information was available for this class. The highest value from a sedge community found in similar habitats in Britain was therefore used
Juncetum squarrosus sa	Rawes and Welch (1969)	Direct measurements made at Moor House from a site in 1964
Madeground	Rawes and Welch (1969)	No specific information was available for this class, however, the class is similar to Agrosto-Festucetum (Eddy *et al.*, 1969). The values of maximum above-ground biomass for this class were used
Nardetum sub-alpinum	Rawes and Welch (1969)	Biomass measurements made at Moor House under grazed conditions. The measurement was the highest recorded between 1965 and 1966
Pteridietum	Pearsall and Gorham (1956)	No data specifically available for Moor House, therefore published values derived from other locations were used
Recolonised peat	Forrest and Smith (1975)	No specific information was available for this class, however, the class is similar to the Calluneto-Eriophoretum class (Eddy *et al.*, 1969). The values of maximum above-ground biomass for that class were used
Sandstone scree	Rawes and Welch (1969)	No specific information was available for this class, however, the class is similar to the Festucetum class (Eddy *et al.*, 1969). The values of maximum above-ground biomass for that class were used
Sphagneto-Caricetum al	Pearsall and Gorham (1956)	No specific information was available for this class. The highest value from a sedge community found in similar habitats in Britain was therefore used
Sphagneto-Juncetum eff	Pearsall and Gorham (1956)	No specific information was available for this class. The highest value from a sedge community found in similar habitats in Britain was therefore used
Trichophoro-Eriophoretum	Forrest and Smith (1975)	Only biomass values from one site were available for this class, therefore the maximum estimate was the same as the mean

Table 2.6 Mean and maximum estimates of above-ground biomass and C density for vegetation classes

Vegetation class	Mean above-ground biomass (g m^{-2})	Mean C density (g m^{-2})	Maximum above-ground biomass (g m^{-2})	Maximum C density (g m^{-2})
Agrosto-Festucetum	46	21	102	48
Calcareous springs	490	221	630	296
Calluneto-Eriophoretum	776	349	979	460
Eriophoretum	432	194	676	318
Eroding bog	776	349	979	460
Festucetum	71	32	122	57
Flushed gleys	490	221	630	296
Juncetum squarrosus sa	275	124	550	259
Madeground	46	21	102	48
Nardetum sub-alpinum	250	113	522	245
Pteridietum	1,010	455	1,410	663
Recolonised peat	776	349	979	460
Sandstone scree	71	32	122	57
Sphagneto-Caricetum al	490	221	630	296
Sphagneto-Juncetum eff	490	221	630	296
Trichophoro-Eriophoretum	188	85	188	88

The class of vegetation with the highest C density was Pteridietum with mean and maximum above-ground biomasses of 1,010 and 1,410 g m^2, respectively. The vegetation classes having the lowest C densities (*e.g.* Agrosto-Festucetum) were mainly dominated by grasses.

The values of total vegetation C storage are shown in Figure 2.8, which provides mean estimates for C based on 2d and true surface areas, and a maximum estimate of C based on the true surface area and the maximum vegetation C values shown in Table 2.6. The raw data used to produce this figure are provided in the Appendix (Section 10.3). Figure 2.9 shows the spatial distribution of C in the vegetation of Moor House, using the mean values.

The vegetation C storage for each kilometre square of the study area is displayed in Figure 2.10, illustrating the differences for estimates of vegetation C stores with 2d and true surface areas. The figure also shows the maximum estimated vegetation C for each kilometre square.

2.3.3 Comparison of the national inventory vegetation C estimates for Moor House with this study

Table 2.7 shows the mean estimate of vegetation C for individual kilometre squares produced by this case study, together with the national C inventory values for the same squares. On the whole, the values derived from this case study were higher than the national database estimates, though there was no consistent pattern in the differences. In a few squares (*e.g.*

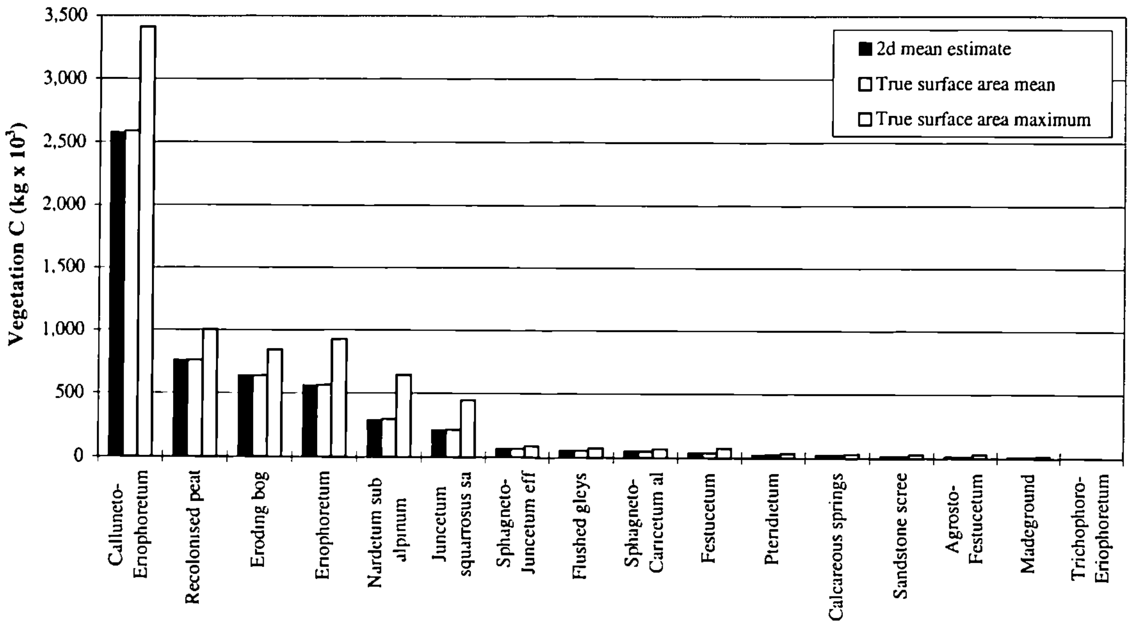

Figure 2.8 Mean and maximum vegetation C estimates based on 2d and true surface areas, by vegetation class

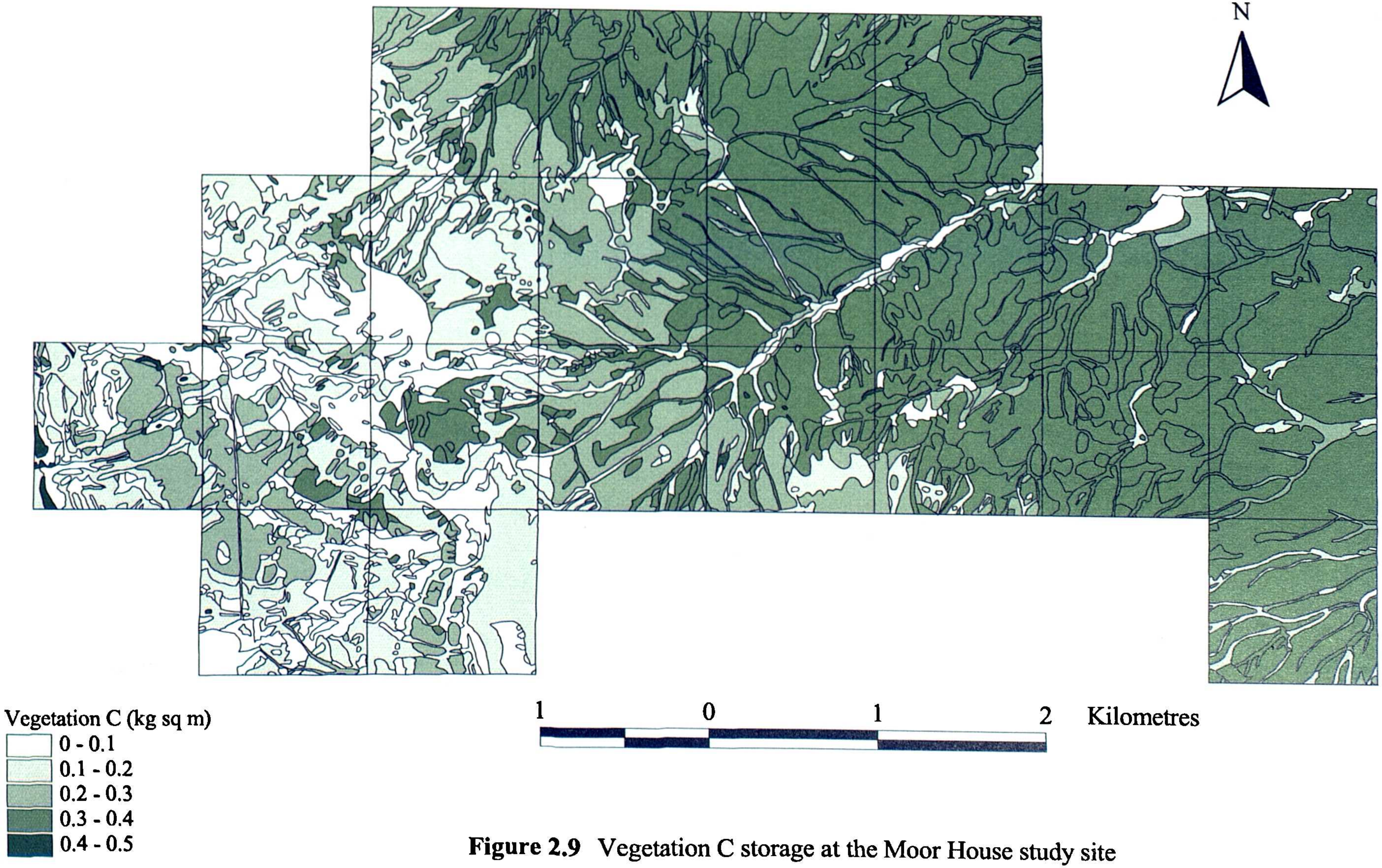

Figure 2.9 Vegetation C storage at the Moor House study site

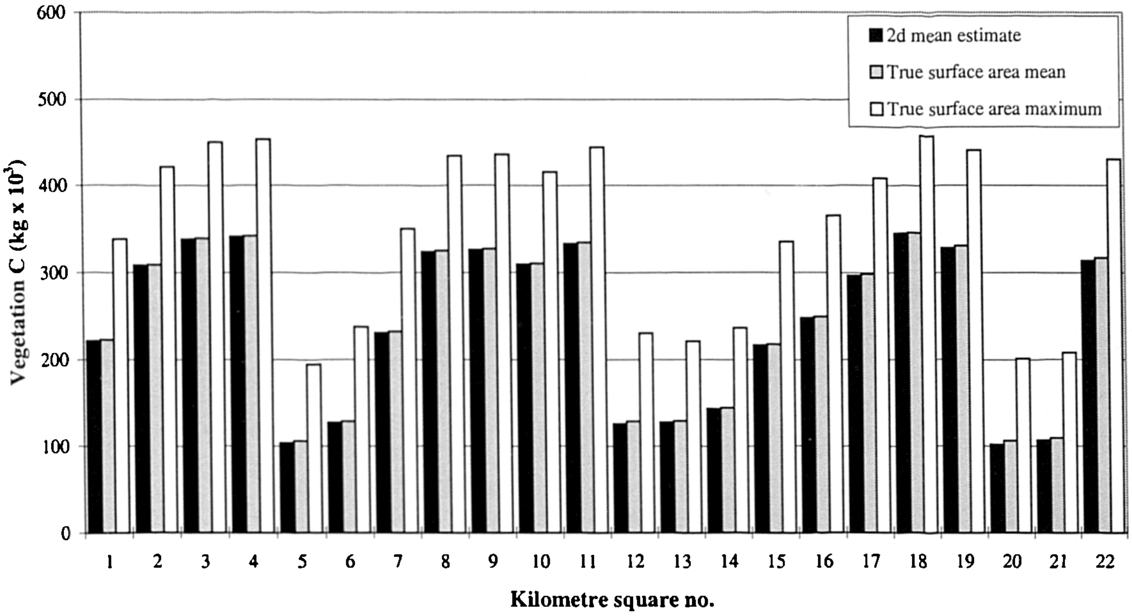

Figure 2.10 Mean and maximum vegetation C estimates based on 2d and true surface areas, by kilometre square (see Figure 2.1)

squares 5 and 13) the estimates were very similar, though in other squares the national inventory estimate was less than a third of the value provided by this study (*e.g.* squares 3 and 8).

Table 2.7 Vegetation C in each kilometre square of the study site, showing comparison between this study's values and the national database estimate. National database values provided by R. Milne, Institute of Terrestrial Ecology

Kilometre square no. (see Fig. 2.1)	Mean case study estimate of vegetation C (kg x 10^3)	National database estimate of vegetation C (kg x 10^3)	Difference (kg x 10^3)
1	223	122	101
2	310	137	173
3	340	98	242
4	343	227	116
5	106	115	-9
6	129	107	22
7	232	140	92
8	326	86	240
9	328	199	129
10	311	391	-81
11	335	256	79
12	129	156	-27
13	129	122	7
14	145	113	32
15	218	132	86
16	249	142	107
17	299	136	163
18	346	189	157
19	332	142	190
20	107	131	-24
21	109	135	-26
22	317	212	105
Total	5,360	3,488	1,872

2.4 Discussion

Reliable estimates of vegetation C storage are required in order to assess the importance of terrestrial ecosystems in the global C cycle, particularly with the current need to understand the influence of vegetation on atmospheric CO_2 concentrations. This fact has been widely recognised and, hence, signatories to the Climate Change Convention of 1992 have been required to produce estimates of national vegetation C stocks.

However, the usefulness of these vegetation C inventories depends on their accuracy and quantification of vegetation C stores within the Moor House study area provided an opportunity to test the accuracy of the existing national C inventory.

Errors associated with estimating C storage in vegetation are principally caused by inaccuracies in both determining the area of each vegetation class and in assigning a reliable value for vegetation biomass for that class. However, the high resolution of the Moor House vegetation map when combined with the detailed biomass values determined from studies carried out on the Reserve, provides an ideal case study for vegetation C estimates. The case study values are therefore considered to be a significant improvement on the national database estimates for the study area. Furthermore, an indication of the accuracy of this case study's vegetation C values has been provided by calculating a maximum value of C storage.

The problems associated with evaluating vegetation C stores, which may have lead to inaccuracies in the estimates, are discussed below.

2.4.1 Area of vegetation classes at Moor House

To estimate the vegetation C storage at Moor House a classification of the vegetation was necessary and then the area and C density of these classes determined. However, any classification system is a simplification of reality and, even though there are strong differences between the vegetation classes mapped at Moor House, variations in species composition and C density will have existed within classes; large variations in the published estimates of above-ground biomass for the same class indicate this (*e.g.* Calluneto-Eriophoretum; Forrest and Smith, 1975).

The vegetation maps produced by Eddy *et al.* (1969) were remarkably detailed for this area of upland moorland. These maps were produced in the 1960's and it has been assumed that the distribution of vegetation classes has not changed since then. Though small changes in climate, management and atmospheric deposition may have taken place over this time, no evidence for major changes in the pattern of vegetation was identified during recent fieldwork (personal observation). However, there is still the possibility of errors in the original maps together with inaccuracies introduced during digitisation of the maps.

It has been shown that true surface, and not 2d, areas should be used when estimating vegetation C storage (Section 2.2.1). However, there were inaccuracies in the values of true surface area because any three-dimensional model is only an approximation of the real landscape. The model used was a TIN, representing the surface of Moor House as triangles, and the closeness of the model to the real surface largely depended on the resolution of the model. A topography coverage containing 10 metre contours was used for Moor House, though a more representative model of the study site could have been obtained if topographical information had been available at higher resolution.

The increased area of the true surface area compared to the 2d area was largely dependent on the scale of the model. The model used to derive the true surface area for the C storage estimates in this work was at the highest resolution practical. Models could have been created at higher resolutions, though the increases in computing time and storage space precluded this. However, lower resolution models were investigated and showed that the increase in area was strongly influenced by the resolution of the model. When less triangles were used in the model, the resulting surface had fewer small scale variations in topography, with a reduced surface area (Figure 2.11).

The true surface area of the study site was calculated to be only 1.24% greater than the 2d area. The increase in area differed between vegetation classes, though the classes which increased proportionally the most tended to be ones with the lowest C densities. Therefore, the increase in the estimate for vegetation C storage, after considering true surface area, was a lower percentage than the increase in area alone (C in vegetation based on true surface areas was only 0.74% higher than the 2d estimate). Though using true surface areas to quantify the Moor House vegetation may have improved the overall C storage estimate, the increased accuracy was small compared to the errors associated with other aspects of evaluating vegetation C stores (the maximum estimate for vegetation C was 30% higher than the mean estimate, suggesting the accuracy of C density values was a greater source of error). Based on the findings of this investigation, it is unlikely that the national inventories of vegetation C would be greatly improved by using true surface areas instead of the usual 2d areas when compared to other sources of error. However, in situations where a vegetation class of high C storage (*e.g.* woodland; see Milne and Brown, 1997) is confined to steep slopes (areas of woodland are sometimes located on steep slopes which have least agricultural value) the correction for true surface area may be more important.

2.4.2 C stored in vegetation classes at Moor House

The measurement of above-ground biomass and subsequent derivation of C density for the classes of vegetation represented at Moor House was a relatively straightforward procedure, yet inaccuracies may have been introduced in several ways.

The values of above-ground biomass used in this study were determined by vegetation clipping and are thought to be accurate since the figures were, in the main, derived from numerous representative plots at Moor House (*e.g.* Forrest, 1971; Forrest and Smith, 1975). However, slight differences in the time of year when the harvests took place will have introduced inaccuracies in the estimates. Most workers attempted to quantify the maximum biomass by harvesting in late summer, though slight year to year variations in the actual day of clipping may have affected the results obtained. Furthermore, the values used in this study were derived from the research of different workers, with individual methods which may have affected the results.

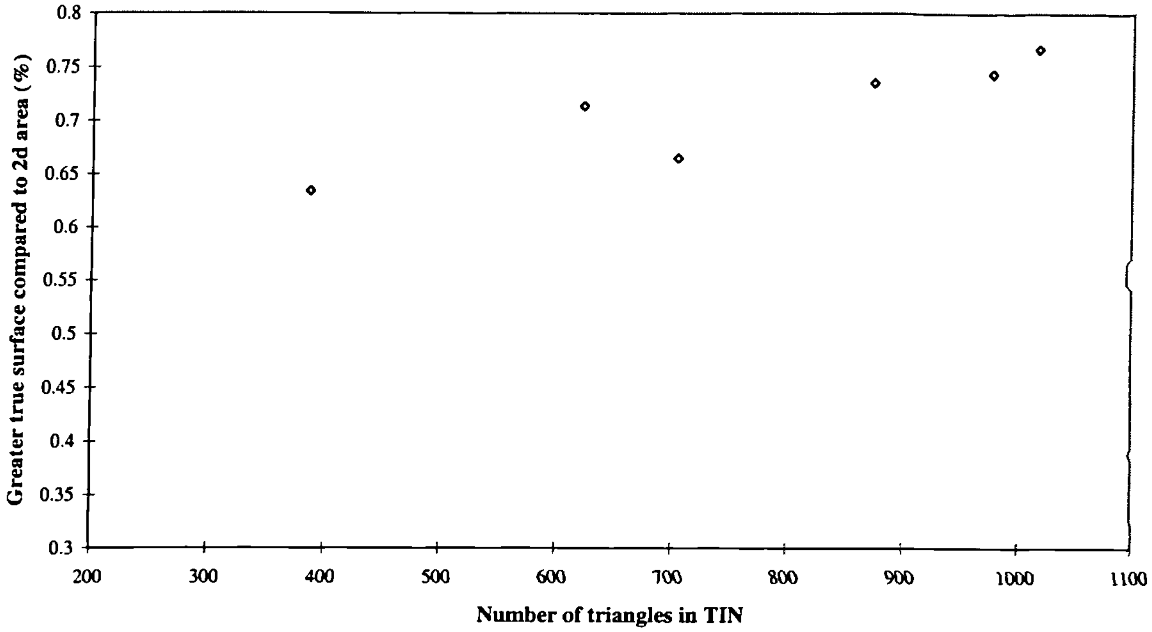

Figure 2.11 Percentage increase of true surface compared to 2d area, when using TINs represented by different amounts of triangles.

It is fortunate that so much research on above-ground biomass has been undertaken on many of the vegetation classes at Moor House, allowing accurate estimates of vegetation C density to be made. However, for the vegetation classes with no published estimates of above-ground biomass the vegetation was assumed to have the same C density as similar vegetation classes for which there were values. This will have lead to inaccuracies, but the area of these vegetation classes was low, and it is believed that the error introduced was small.

Many of the estimates of above-ground biomass were determined from studies carried out in the 1960's and 1970's (*e.g.* Forrest, 1971; Forrest and Smith, 1975) and it has been assumed that these estimates are still applicable today. Management of this Nature Reserve has been constant over the intervening period and we know that temperatures at the Moor House field station have not greatly changed over the last few decades (Garnett *et al.*, 1997a).

This study assumed that for the mean estimate of vegetation C stocks the concentration of C in vegetation was 45%. This seemed a reasonable assumption since Allen (1989) reported that the vegetative parts of plants have a C concentration in the range 43%-47%, though because the middle value of the range was adopted for the mean estimate, some small inaccuracies will have been introduced.

The total amount of C stored in the vegetation of the 22 one-kilometre squares of the study site was 5,360 x 10^3 kg, whereas the maximum estimate was 7,716 x 10^3 kg. Since the maximum estimate was derived from the highest C density values found for the different vegetation classes, it provides an indication of the accuracy of the mean estimate. The maximum estimate was only 30% higher than the mean and therefore in terms of total terrestrial C storage at the study site, the error associated with the vegetation component is small (see Chapter 3).

The vegetation class with the greatest vegetation C storage was Calluneto-Eriophoretum since it covered the greatest area, and had a higher C density than many of the other vegetation classes. The other vegetation classes associated with the blanket peat also had high C storage due to their large area and high C density. The greater C storage of the moorland vegetation classes when compared with the grasslands was partly a result of sheep grazing on the Reserve; the grasslands are more intensively grazed than the moorland vegetation classes and, therefore, do not accumulate large amounts of above-ground biomass The impact of sheep grazing on the vegetation C stores in the study site is considered in more detail in Chapter 4.

2.4.3 Comparison with the UK's national C inventory

The national database estimate for total C storage in the vegetation at Moor House was 3,488 x 10^3 kg (Milne, personal communication), which is 35% lower than the case study value. The estimates of vegetation C in each kilometre square in the case study ranged from 106 x 10^3 kg to 346 x 10^3 kg, while the national database estimates for the same squares ranged from 86 x 10^3 kg to 391 x 10^3 kg. Clearly, these estimates overlap, although there is a wider spread of values in the national database.

The national database showed an increase in the amount of C in vegetation from the west to east at the study site, but the trend was not as distinct as in the current case study. Case study vegetation C values had higher estimates than the national database values for most of the kilometre squares, and, with the exception of square 10, all the eastern squares had lower estimates in the national database than the case study values. However, the total amount of C in vegetation for both estimates was small compared to that in soil (see Chapter 3).

3. Present organic carbon storage in the soil at Moor House

3.1 Introduction

Soil C[1] is a major component of the global C inventory and contains approximately two thirds of the Earth's terrestrial C store (Schimel *et al.*, 1994) and about twice as much C as the atmosphere (Schimel, 1995). Soils have an important role in regulating the C cycle (see Chapter 1) and there is concern that future climate change may alter rates of C sequestration and emission from soils (IPCC, 1995). These processes directly influence the atmospheric C content and, since soils contain much larger amounts of C than the atmosphere, a small reduction in soil C stores could significantly increase atmospheric CO_2 concentration.

Reliable estimates of soil C stores are required to produce accurate and quantitative predictions of future C fluxes under climate change. This was acknowledged at the Rio Earth summit in 1992 when signatories to the Framework Convention on Climate Change, including the UK, committed nations to evaluating soil C stocks (United Nations, 1993). As a direct result, an inventory of soil C storage in the UK has been produced (Milne and Brown, 1997).

This chapter describes an evaluation of soil C stores in the upland ecosystem at Moor House NNR. The importance of factors such as soil type and topography have been investigated and soil C was determined to a range of soil depths for each of the National Grid kilometre squares fitting entirely within the Reserve boundary (Figure 2.1). The values obtained from this case study have been used to test the accuracy of the UK's national C inventory by comparison with estimates for the same kilometre squares.

3.1.1 Description of soil types represented at Moor House

The soils of Moor House have been studied less than the vegetation, though an exceptional amount of information is available. The soils have been classified and mapped by Johnson and Dunham (1963) and a digital version of their map has been investigated in this study using a GIS (see Table 1.5 for details). Figure 3.1 shows the distribution of soils within the 22 one-kilometre squares of the study site derived from Johnson and Dunham's map. Many of the soils within the Reserve have also been investigated by Hornung (1968), while a general description of them is provided by Heal and Smith (1978).

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[1] Soil C in this study refers only to soil organic carbon

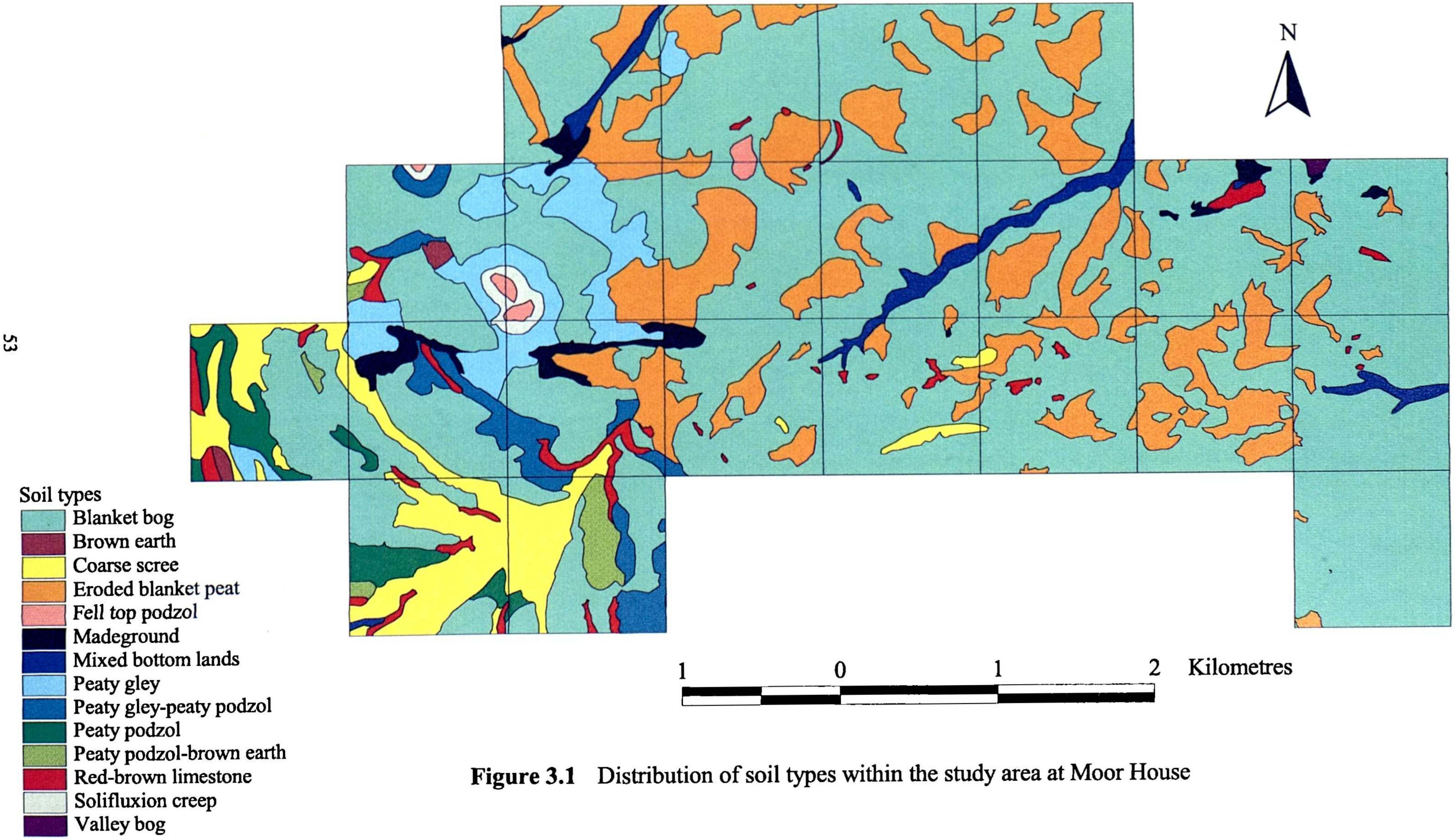

Figure 3.1 Distribution of soil types within the study area at Moor House

The dominant soil type at Moor House is blanket bog[1] which overlies glacial till and is believed to have started accumulating at *ca.* 7500 BP (Heal and Smith, 1978). However due to the variation in topography and geology at the site, a wide range of other peat and mineral soils are present. Based largely on the descriptions of Johnson and Dunham (1963), characteristics of each soil type represented in the study site are given in Table 3.1.

Table 3.1 Soil types represented at Moor House

Soil type	Characteristics
Blanket bog	This soil type occurs where peat depths are greater than 0.3 m, though the deep valley peats on the Reserve have been classified separately (see below). It occurs on flat and gentle slopes and covers most of the eastern side of the study site.
Brown earth	This soil occurs on freely draining slopes and is characterised by a uniformly coloured brown horizon.
Coarse scree	This soil type is found mainly on the western escarpment and summit ridge and consists of thin, stony skeletal soils.
Fell top podzol	The Fell top podzol, is restricted to level ground on sandstone outcrops at, or near, the summits of the fells.
Peaty gley	This soil contains a gleyed horizon covered by peat of less than 0.3 m thick. It occurs on gentle or level slopes where drainage is impeded.
Peaty podzol	Peaty podzols occur in the blanket peat zone in areas of better than average subsoil drainage. These podzols have a water-logged peaty surface and are restricted to the upper slopes of the summit ridge and western escarpment.
Red-brown limestone	This soil has developed on limestone outcrops and is characterised by a red-brown coloured clay-loam.
Valley bog	This soil type is restricted to a single area within the study site and represents a deep peat filled basin which developed from a small lake.

Where two or more soil types were repeatedly present but changed from one to the other over distances too short to map, they have been classified as soil complexes (Table 3.2; Johnson and Dunham, 1963).

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[1] In this chapter, the first letter of a soil type has been capitalised to distinguish it from other words used to describe the soils at Moor House

Table 3.2 Soil complexes represented at Moor House

Soil complex	Characteristics
Eroded blanket peat	A complex formed where the blanket bog has been eroded to leave a mosaic of blanket peat, bare eroded peat, bare gleyed soil and peaty gleyed soil (Johnson and Dunham, 1963).
Madeground	This is a complex of man-made soils developed on spoil heaps and other relicts of past mining activity on the Reserve.
Mixed bottom lands	This soil complex occurs beside larger streams on the Reserve and contains alluvium, peaty alluvium, blanket peat and areas of bare drift and bedrock (Johnson and Dunham, 1963).
Peaty gley-peaty podzol	A soil complex comprised of Peaty gley and Peaty podzol soils (see Table 3.1).
Peaty podzol-brown earth	A soil complex containing Brown earth and Peaty podzol (see Table 3.1).
Solifluxion creep	These soils show evidence of solifluxion activity and are orange-yellow in appearance. They have similar characteristics to Brown earth soils, though have developed on steep slopes at greater altitudes.

3.2 Method

The C storage of individual soil types was calculated in a similar way to the vegetation C whereby the area of each soil type was multiplied by its C density[1]. Soil C storage was determined for individual soils and each kilometre square of the study site, with the latter providing a comparison with the national C inventory.

However, there were important differences between the methods used to determine soil and vegetation C. Firstly, total soil C storage estimates were based on 2d areas and not the true surface areas used for vegetation C (see Section 3.2.2). Also, unlike the vegetation, the data required to produce accurate estimates of soil C density were not available from existing records, and consequently, a programme of field sampling was undertaken. Since the study site was dominated by blanket bog which was expected to contain the greatest amounts of soil C, and was known to be very variable in depth (Johnson and Dunham, 1963), this soil was stratified in an attempt to improve the evaluation of soil C stores (see Section 3.2.1).

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[1] Soil C density, like vegetation C density, refers to the C per square metre and is not a true density. However, the term is used here for convenience following its use by Milne and Brown (1997)

3.2.1 Stratification of the Blanket bog soil type

The Blanket bog soil type was stratified by vegetation class because it was thought the C density of this soil may vary with vegetation; as there was a digital version of the vegetation map by Eddy *et al.* (1969) available the strata were easy to define. Furthermore, in the soils map of Johnson and Dunham (1963), areas of eroding blanket bog had been grouped together as a soil complex named Eroded blanket peat which contained eroded and uneroded areas. However, in the vegetation map of Moor House areas of 'Eroding bog' had been mapped by Eddy *et al.* (1969; see Figure 2.2). Therefore, it was likely that the true area of eroding blanket bog was more accurately represented by the vegetation map. As a result, the soil types Blanket bog and Eroded blanket peat in the soils coverage were amalgamated using Arc/Info . This new soil type was then intersected with the vegetation coverage separating it into seven sub-types of Blanket bog, one of which represented the distribution of eroding blanket bog (since it was based on the vegetation maps 'Eroding bog' class). Five of the other sub-types were defined by vegetation class (Calluneto-Eriophoretum, Eriophoretum, Nardetum, Juncetum and Recolonised peat; described in Table 2.1) while a final sub-type, called 'Other blanket bog', represented the minor vegetation classes existing on blanket bog. In this new classification of the Moor House soils, the non-blanket bog soil types remained unchanged, though for convenience all new classes have been referred to as sub-types. Figure 3.2 shows how the soil sub-types in this new classification were derived.

3.2.2 Area of soil types

Unlike the vegetation C storage estimates, the 2d area and not the true surface area was used to evaluate soil C stores. Since soil profiles are normally investigated by sampling vertically down the soil profile, if the soil is on a slope, the depth will be slightly greater than if the soil was measured perpendicular to the slope (*i.e.* the minimum soil thickness). This increase compensates for the underestimation of area caused by using a 2d area, and is explained below with reference to Figure 3.3.

Soil type (after Johnson and Dunham, 1963)	**Soil sub-type** (used in this study)
Blanket bog Eroded blanket peat	Calluneto-Eriophoretum bog Eriophoretum bog Eroding bog Juncetum bog Nardetum bog Recolonised peat Other bog
Brown earth	Brown earth
Coarse scree	Coarse scree
Fell top podzol	Fell top podzol
Madeground	Madeground
Mixed bottom lands	Mixed bottom lands
Peaty gley	Peaty gley
Peaty gley-peaty podzol	Peaty gley-peaty podzol
Peaty podzol	Peaty podzol
Peaty podzol-brown earth	Peaty podzol-brown earth
Red brown limestone	Red brown lime tone
Solifluxion creep	Solifluxion creep
Valley bog	Valley bog

Figure 3.2 Soil types and the origin of soil sub-types used in this study

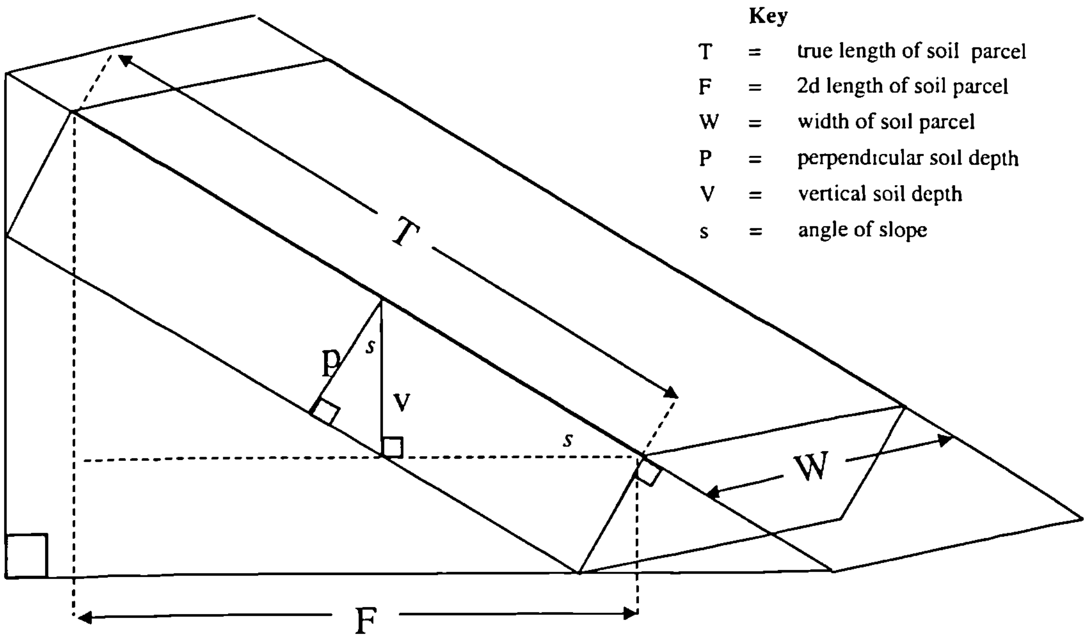

Figure 3.3 Theoretical soil parcel on sloping ground.

The shaded box represents a compartment of soil where volume can be determined in two ways. Firstly, the perpendicular soil depth (P) can be multiplied by the true surface area (T x W). Alternatively, the vertical soil depth (V) can be multiplied by the 2d area of the soil compartment (F x W). This can be shown using simple trigonometry:

$$Volume = T \times P \times W \qquad \textit{[equation 3.1]}$$

Using trigonometry:

$$COS\ s = F / T \qquad \textit{[equation 3.2]}$$

Rearranged:

$$T = F / COS\ s \qquad \textit{[equation 3.3]}$$

Also, using trigonometry:

$$COS\ s = P / V \qquad \textit{[equation 3.4]}$$

Rearranged:

$$P = V\ COS\ s \qquad \textit{[equation 3.5]}$$

Equations 3.3 and 3.5 can be substituted into equation 3.1:

$$\begin{aligned} Volume &= T \times P \times W \\ &= F / COS\ s \times V\ COS\ s \times W \\ &= F \times V \times COS\ s / COS\ s \times W \end{aligned}$$

Since, $COS\ s / COS\ s = 1$, then:

$$Volume = F \times V \times W \qquad \textit{[equation 3.6]}$$

Because soil profiles are traditionally investigated vertically and not perpendicular to the slope, volumes of soil parcels should be determined by multiplying the 2d area by the vertical soil depth measurement, as in equation 3.6. This differs to the estimates for vegetation C storage which were based on true surface areas (Section 2.2.1).

The 2d areas of the different soil sub-types were determined using Arc/Info GIS.

3.2.3 Determination of soil C density

Soil C density was determined for 164 individual locations across the study site between October 1995 and September 1997, with samples being retrieved from each soil sub-type except Valley bog and the soil complexes. The sampling locations for most soil sub-types were determined using random numbers and a gridded overlay on the soils map of Moor House. Random sampling locations were obtained and discarded if not located within the correct soil type. This process continued until the required number of sampling locations had been obtained.

However, sampling locations were determined differently for a few soils. Since the Brown earth soil only occurred in two small areas within the study site and at different altitudes, both compartments were investigated. This soil was the first to be studied and was used to evaluate several different approaches to determining the C density value. Similarly the Peaty gley soil was studied at two altitudes and attempts were made to investigate the importance of slope and aspect by determining the variation in soil C density along transects of varying slope and aspect. The Red-brown limestone and Solifluxion creep soils, which occupy small areas within the study site, were investigated as part of a study into the effects of sheep grazing on C stores (Chapter 4). The sampling points chosen for these soils were not randomly located and were chosen for the reasons outlined in Chapter 4. All other locations for soil sampling were randomly selected.

Due to the different characteristics of the soils at Moor House, no single method was suitable to obtain samples required for estimating C density. The methods employed are described below.

3.2.3.1 Sampling from soil pits

Several soils were sampled by digging pits and collecting soil samples with bulk density tins. These metal tins were square (typically 10 cm x 10 cm across and 5 cm deep) and open at either end allowing them to be pushed into the soil causing minimal compaction. The tins were then carefully dug out and contained a known volume of soil. The soils sampled this way were: Brown earth, Coarse scree, Fell top podzol, Madeground, Peaty gley, and Peaty podzol. For the shallow soils this method was found to be convenient, but for the Brown earths and Peaty gleys, which were frequently quite deep (up to 1 metre), this method was time-consuming and, therefore, the Peaty gley was also sampled using a soil coring method.

Where distinct horizons were evident in the soil, as in the Brown earth, samples were retrieved with bulk density tins from each horizon. The thickness of each horizon was measured and the samples returned to the laboratory for analysis. Soil C density was determined by calculating the mass of soil in each horizon and multiplying by the %C of the horizon, which was determined using loss-on-ignition (%LOI; see below).

Where distinct horizons were not evident and the soil was quite deep, several samples were collected from different depths. However, uniformity in the soil was normally associated with a consistent organic C content, therefore these samples tended to have similar %LOI and bulk density values.

Where the soil being investigated was shallow, representative samples were removed from the centre of the soil profiles using a bulk density tin. If possible the tin was pushed into the soil sideways allowing a sample covering a wide depth range to be collected. This sample was assumed to be representative of the soil profile and the C density value was obtained by calculating a C density value for the soil layer sampled, and multiplying by the depth.

3.2.3.2 Coring

Coring was used to sample both mineral and peat soils, with several different corers being required due to variations in the characteristics of soils being sampled.

All Blanket bog sub-types were investigated using a peat corer which took samples of dimensions 5 x 5 x 100 cm (modified Cuttle and Malcolm, 1979). This corer could only sample the top 1 metre of peat and therefore a gouge auger was also employed to take deeper samples and to determine the total peat depth. Frequently the peat base lay directly on the bedrock, though in most cases the peat was underlain by a gleyed layer, as described by Heal and Smith (1978). It was not possible to accurately sample this because it was difficult to push even a narrow corer into this dense layer, and even harder to measure its total thickness, therefore the C density values refer only to the actual depth of peat. Analysis of samples from this gleyed layer, using %LOI, suggested only small amounts of organic C existed in this layer.

Each peat core was sectioned immediately after coring and the samples placed in sealed plastic bags. The top 30 cm was first separated, then the next 20 cm (allowing C in the top 50 cm of the peat profile to be determined). When possible the next 50 cm was separately collected to allow estimates of C density in the top 1 metre to be calculated. If the profile was greater than about 150 cm depth, a representative sample of known volume was selected in the middle of the remaining depth using the gouge augar. The bulk density and %LOI from this sample were used to determine the C content of the parts of the profile not sampled.

Several soil corers were used because certain corers performed well only on certain soils. For example, a short stainless steel corer of length 26 cm and with an internal diameter of 3.6 cm was suitable for sampling the thin Solifluxion creep soil. This soil contained many boulders which prevented the use of soil pits and bulk density tins. However, this corer could only sample to a depth of 26 cm and was restricted to thin soils. Therefore a larger and more robust corer was used for the Peaty gley and Red-brown limestone soils; this had a chamber of length 36 cm and an internal diameter of 5.2 cm, and could be forced below the soil surface enabling complete soil profiles to be sampled.

Unlike the peat corers, the soil corers tended to compress the soil being sampled. However, because the whole profile was being sampled, a known amount of soil was being removed from a known volume (the internal diameter of the corer was known and the depth of the soil profile was measured after extracting the core). Therefore, the compaction did not affect the values of C density determined using these corers.

After retrieval, the samples were placed in sealed plastic bags and returned to the laboratory. If the samples were not immediately analysed they were stored in a refrigerated room until required.

3.2.3.3 Laboratory analysis

Samples were weighed while fresh and then oven-dried at 105°C for 24 hours (until all moisture was removed), and weighed again after cooling in desiccators. Samples were then sieved through a 2 mm mesh, thoroughly mixed, and returned to the oven for further drying at 105°C. After cooling in desiccators the samples were re-weighed to obtain the dry-weight on a stone-free basis. Moisture content of fresh soil and bulk density (on a stone-free basis) were calculated using the following equations:

$$\textit{moisture content (\%)} = \left((\textit{wet weight} - \textit{dry weight}) / \textit{wet weight} \right) \times 100 \qquad \textit{[equation 3.7]}$$

$$\textit{bulk density} = \textit{dry weight / volume (when sampled)} \qquad \textit{[equation 3.8]}$$

Representative sub-samples were then used to determine %LOI using the method of Allen (1989); samples of at least 1 g dry-weight were oven dried and weighed to determine dry mass, and then placed in a muffle oven at 550°C for 2 hours. After allowing to cool in desiccators, the samples were re-weighed and the %LOI calculated. The %LOI values were converted to %C using a regression equation which had been developed from a range of soils taken from Moor House. The regression, provided by Bol *et al.* (in press), had a correlation coefficient (r^2) of 0.984 and was:

$$\%C = (\%LOI \times 0.5265) - 0.1672 \qquad \textit{[equation 3.9]}$$

Some of the dried peat samples were found to be extremely difficult to sieve using the 2 mm mesh size, as the peat became extremely hard with drying. Therefore, these samples were forced through a 4 mm mesh sieve and larger sub-samples were used for %LOI.

3.3 Results

3.3.1 Area of soil types at Moor House

The 2d area of different soil types at Moor House are shown in Figure 3.4, while Figure 3.5 displays the area of the soil sub-types after stratifying the combined Blanket bog and Eroded

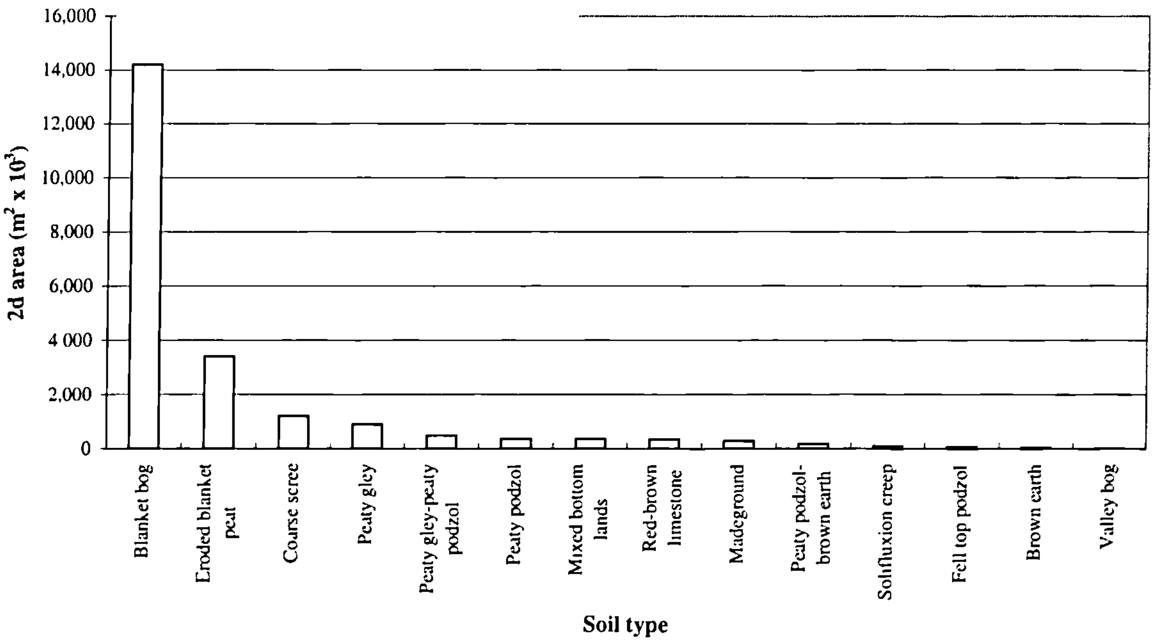

Figure 3.4 Area of soil types within the study site

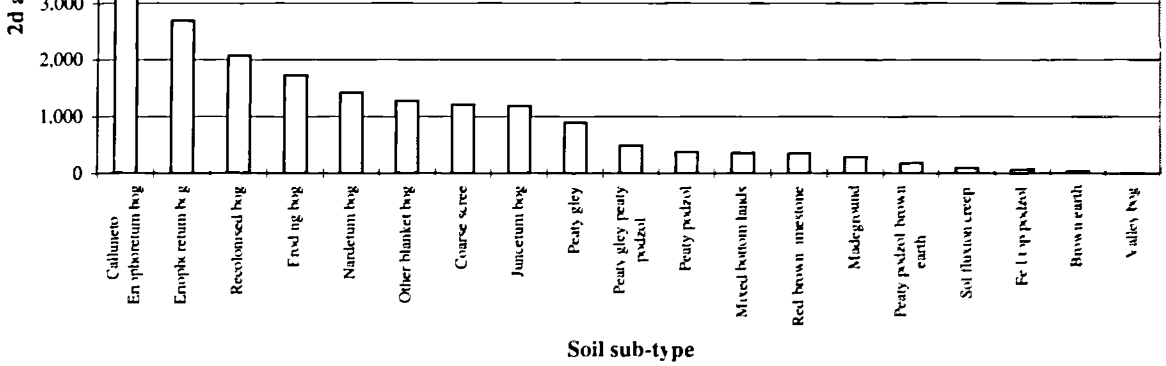

Figure 3.5 Area of soil sub-types within the study site

blanket peat classes with the vegetation coverage. The distribution of the Blanket bog sub-types is shown in Figure 3.6. The dominance of blanket bog within the study site is clear, with Calluneto-Eriophoretum bog being the sub-type with the greatest area. The 2d areas of each soil type within each kilometre square used to derive these summary charts are provided in the Appendix (Section 10.4).

The mean percentage slope of each soil type was determined by creating a three-dimensional model of the study site which had also been used to obtain slopes of vegetation classes (Section 2.2.2). These data are displayed in Figure 3.7.

3.3.2 C stored in the soil types at Moor House: sampling strategies

An outline of the sampling strategy used for each soil type is described below, providing an explanation for the final sampling strategy chosen.

3.3.2.1 Brown earth

The Brown earth soil type was the first to be investigated for C density during the programme of field sampling and was used to assess several methods for deriving soil C density values. Since only two parcels of this soil occurred within the study area at high and low altitudes (760 m and 490 m respectively), both locations were investigated to determine whether C density was influenced by the conditions at the different altitudes. To determine whether slope or aspect affected soil C density the high altitude site was sampled along two transects which covered a range of slopes and aspects. No significant correlation between soil C density and either slope or aspect was found, suggesting neither variable strongly influenced C density. Consequently, the lower altitude site was sampled from random locations.

To determine whether soil C density at the two sampling locations was significantly different a *t*-test was performed on the results (PROC TTEST, SAS Institute 1990). The results of this test showed that the soil C densities obtained were not significantly different ($p > 0.07$), implying that altitude did not strongly influence the soil C density of this soil type. Therefore, the soil C density values for this soil were calculated by taking the mean from all sampling points.

3.3.2.2 Peaty gley

Similarly, the Peaty gley soil was investigated at high and low altitudes in order to investigate whether this, and other variables, significantly influenced soil C density. At the lower altitude site (Eller Gill; 540 m) random samples were taken, while at the upper site (800 m), in the parcel of peaty gley soil surrounding the summit of Great Dun Fell (Figure 3.1), four transects were sampled with aspects of north, south, east and west respectively.

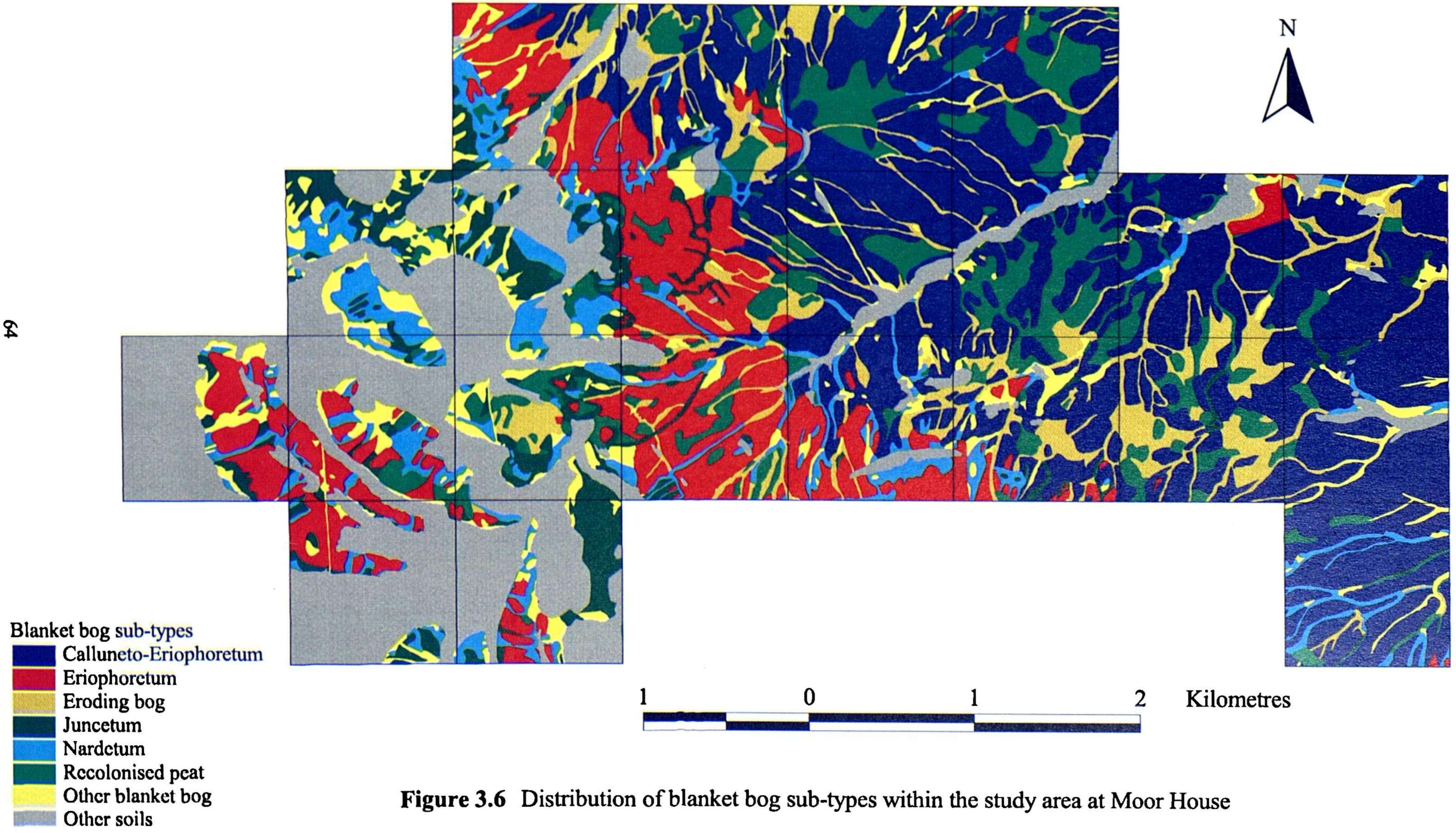

Figure 3.6 Distribution of blanket bog sub-types within the study area at Moor House

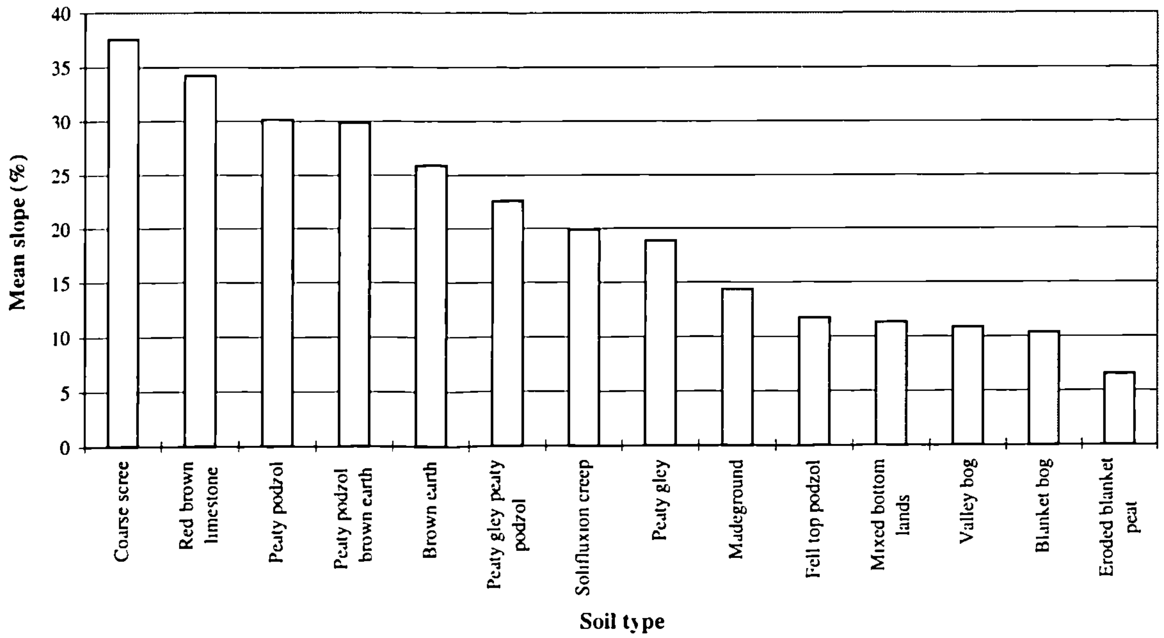

Figure 3.7 Mean percentage slope of the soil types

To determine whether the soil C density values from the two sampling locations were significantly different, a *t*-test was performed on the results (PROC TTEST, SAS Institute 1990). The results of this test revealed no significant difference ($p > 0.09$) in the soil C densities between the sites. Consequently, the estimates of soil C density for this soil type have been determined using the mean from all sampling locations.

3.3.2.3 Coarse scree

Preliminary investigations suggested that this soil type would contain only small amounts of soil C because it had been described as a thin skeletal soil (Johnson and Dunham, 1963). An intensive study of this soil was considered unimportant since it would not greatly increase the accuracy of the soil C storage estimates for the study site and, consequently, this soil was only sampled from three randomly chosen locations. Soil samples were collected using bulk density tins and the mean from the three locations used to provide the soil C density estimates; because the soil depth at each sampling point was less than 30 cm, the soil C density to each depth was identical.

3.3.2.4 Fell top podzol

This soil type covers only small areas at Moor House and was sampled at three random locations by digging soil pits and collecting samples in bulk density tins. The means from each sampling point were used to determine the final soil C density estimates.

3.3.2.5 Madeground

Due to the small area classified as Madeground and based on the description of this soil by Johnson and Dunham (1963), it was thought this soil would contribute only small amounts to the total soil C storage at Moor House. Therefore, randomly located samples were taken from three locations using bulk density tins to determine soil C density.

3.3.2.6 Mixed bottom lands

This soil complex was not specifically sampled since it consisted of blanket peat and areas of alluvium and bedrock (Johnson and Dunham, 1963). No information was available describing the proportions of the components in this complex and, therefore, the soil C density was estimated by assuming equal proportions of blanket bog and bedrock. The soil C density was consequently estimated to be half the blanket bog value, for all depths.

3.3.2.7 Peaty podzol

The soil C density of the peaty podzol was estimated from five randomly located sampling points. The means of all sampling locations have been used to provide C density values to different depths, for this soil.

3.3.2.8 Peaty gley-peaty podzol

This soil complex was not specifically sampled since its component two soil types were sampled independently. However, the proportions of Peaty gley and Peaty podzols in this soil complex was unknown and, therefore, the estimates of soil C density assumed equal proportions of both soils. The C density values were therefore the mean of the two soils C density values.

3.3.2.9 Peaty podzol-brown earth

This complex was not specifically sampled and consequently the soil C density values were based on the values determined for the component soils forming the complex. No information was available on the proportions of Peaty podzol and Brown earth soils and, consequently, equal proportions were assumed and the C density derived by taking the mean of the Brown earth and Peaty podzol values.

3.3.2.10 Red-brown limestone

The Red-brown limestone soil was investigated as part of the study of the impact of sheep grazing on terrestrial C stores (see Chapter 4).

3.3.2.11 Solifluxion creep

The C density of this soil type was also determined from the study of the impact of sheep grazing on terrestrial C stores (see Chapter 4).

3.3.2.12 Valley bog

Apart from the soil complexes, this was the only soil type not sampled during the current study. However, although its C density was expected to be high, being deep peat, the soil type covered only a very small area within the study site. Furthermore, reliable published results were available from Jones and Gore (1978) for this soil and have been used to determine C density.

Jones and Gore (1978) retrieved peat cores down to a depth of 3 metres from Valley bog and determined bulk density in 10 cm thick contiguous samples. Their results showed a general increase in bulk density with depth until, at about 2 metres depth, it stabilised at a mean value of 0.063 g cm^{-3}. The total peat depth at Valley bog was 8.7 metres (Johnson and Dunham, 1963) and therefore the C density of this soil has been calculated by using the values of Jones and Gore (1978) for the C density estimates of 30, 50 and 100 cm depth, and by assuming the bulk density of the peat between 3 metres and the base was 0.063 g cm^{-3}. The peat has been assumed to have a C concentration of 50% (Heal and Smith, 1978; Allen, 1989; Immirzi *et al.*, 1992).

However, the C density value for the total depth of this soil is likely to be an over estimate because the 8.7 metres peat depth quoted from Johnson and Dunham (1963) represents the maximum depth they obtained. Since no information was available on the range of peat depths in this soil type, the C storage of the soil is based on this depth.

3.3.2.13 Blanket bog sub-types

Using the stratification of the blanket bog, samples were randomly chosen to determine soil C density for each blanket bog sub-type. The number of sampling locations was proportional to the area of the sub-type, with most samples being taken from the Calluneto-Eriophoretum sub-type. A total of 50 locations were sampled for C density for the blanket bog as a whole, with limitations imposed due to inaccessibility of sites and the need to carry coring equipment and samples over long distances and difficult terrain.

Each blanket bog sub-type was sampled except for 'Other blanket bog' which had C density values derived from the mean of all Blanket bog sampling locations.

3.3.3 Soil C in the soil sub-types to different depths

The values of C density for each soil, to the various depths, are shown in Figures 3.8 to 3.11, showing the mean C density in the top 30, 50, 100 cm and complete soil profile, respectively. Where possible, the standard errors have been determined and are shown as error bars. The charts show that the range of C density for soils at the site increased when considering soil C to greater depth. Whereas soil C to 30 cm for most soil sub-types was between 10 and 20 kg m^{-2}, soil C contained within the total soil depth ranged from below 10 kg m^{-2} to greater than 100 kg m^{-2}. In particular, the peaty soils had the highest soil C densities when considering the total depth (*e.g.* Valley bog and Calluneto-Eriophoretum bog), whereas some of the shallow mineral soils had considerably lower C densities (*e.g.* Solifluxion creep). The actual values of C density, number of sampling locations and standard errors are provided in the Appendix (Section 10.4).

Figures 3.12 to 3.15 show the estimated C stored within each soil sub-type for the entire study site, to the different soil depths, and Figure 3.16 shows the distribution of total soil C for the site. When considering the soil C density to all depths, the blanket bog sub-

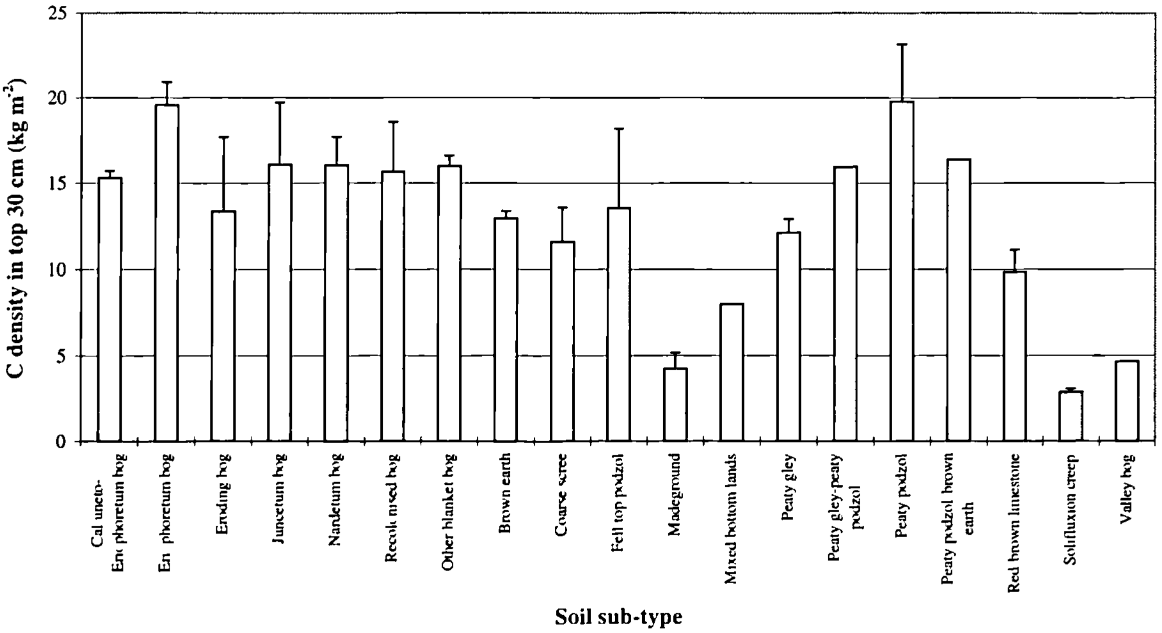

Figure 3.8 Soil C density in top 30 cm. Error bars indicate standard error (no confidence limits for Valley bog or soil complexes).

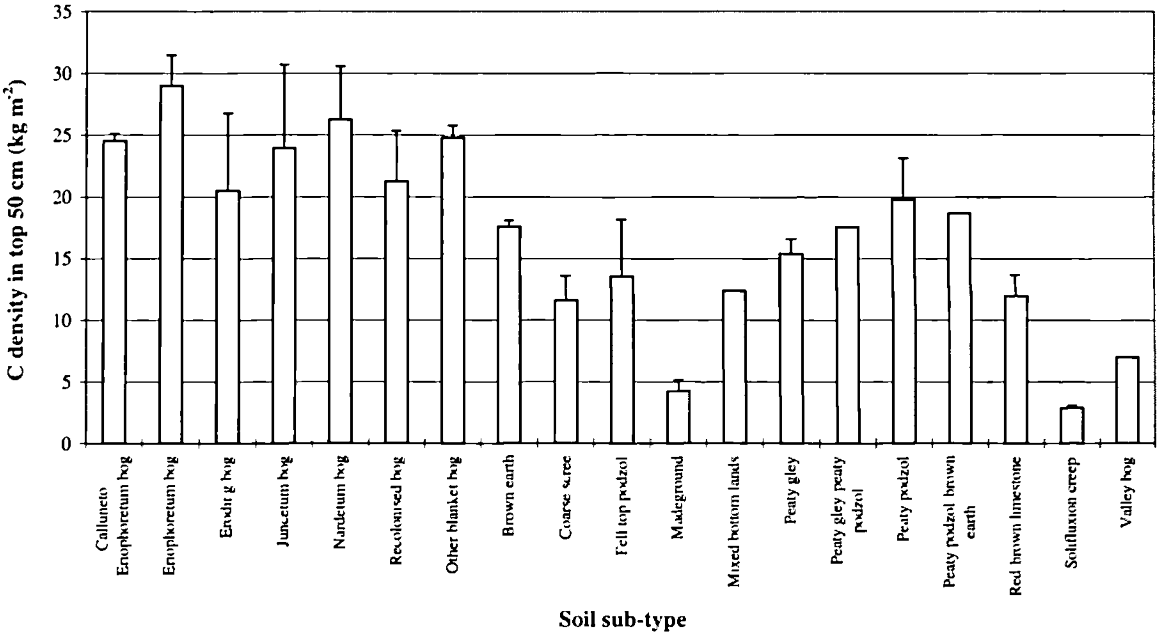

Figure 3.9 Soil C density in top 50 cm. Error bars indicate standard error (no confidence limits for Valley bog or soil complexes).

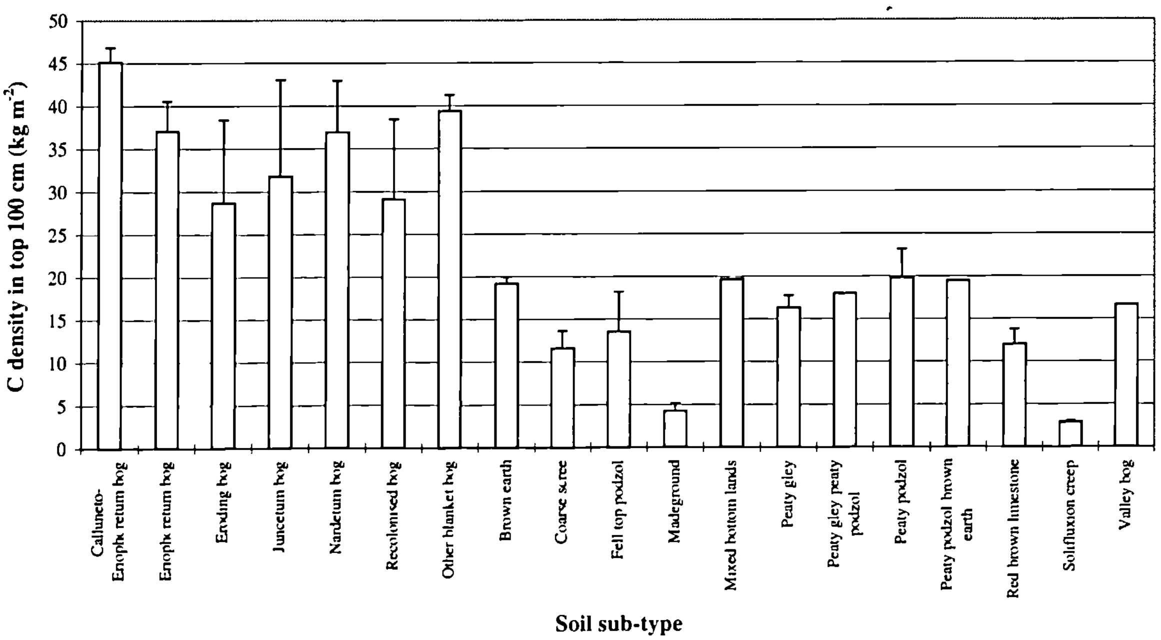

Figure 3.10 Soil C density in top 100 cm. Error bars indicate standard error (no confidence limits for Valley bog or soil complexes).

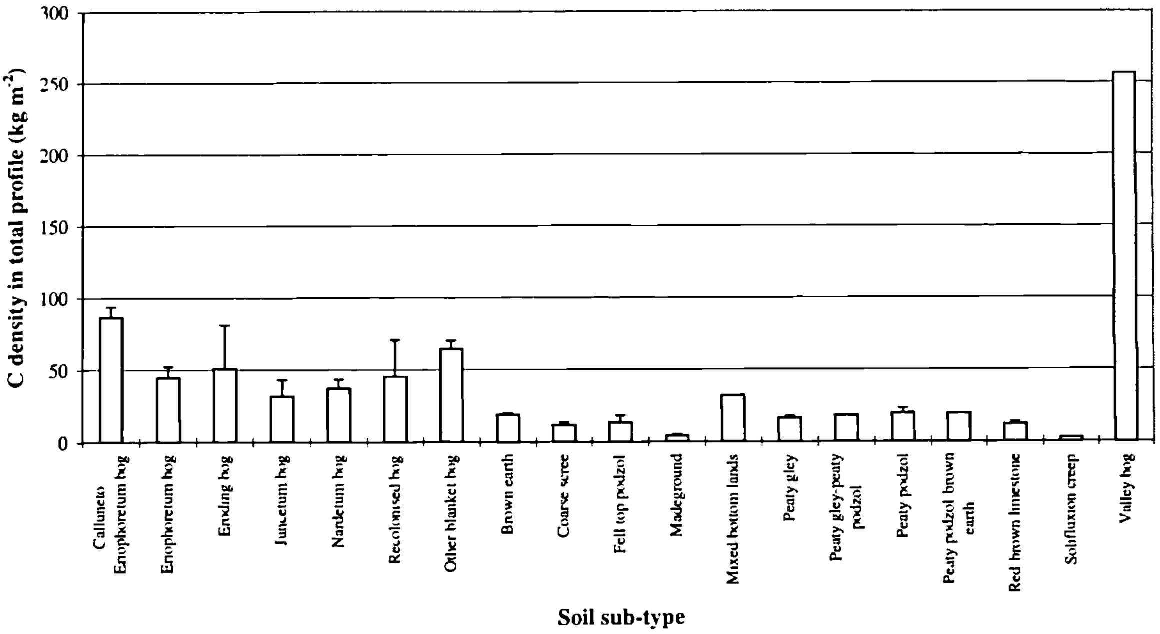

Figure 3.11 Soil C density in total soil profile. Error bars indicate standard error (no confidence limits for Valley bog or soil complexes).

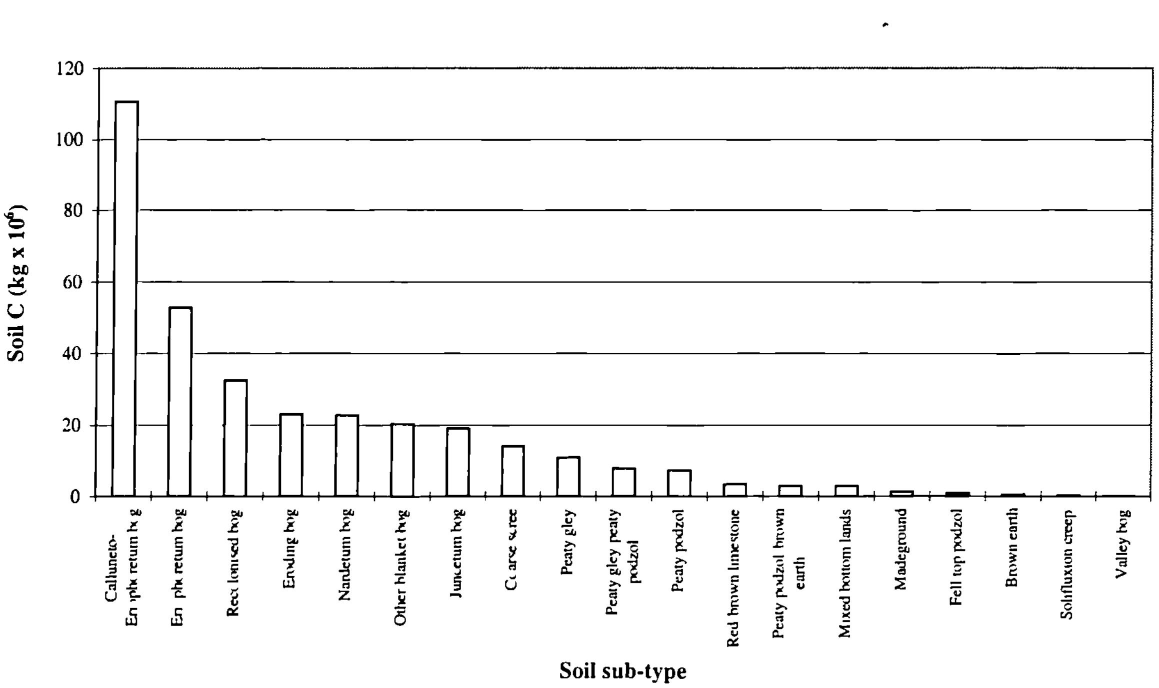

Figure 3.12 C stored in top 30 cm of Moor House soil sub-types

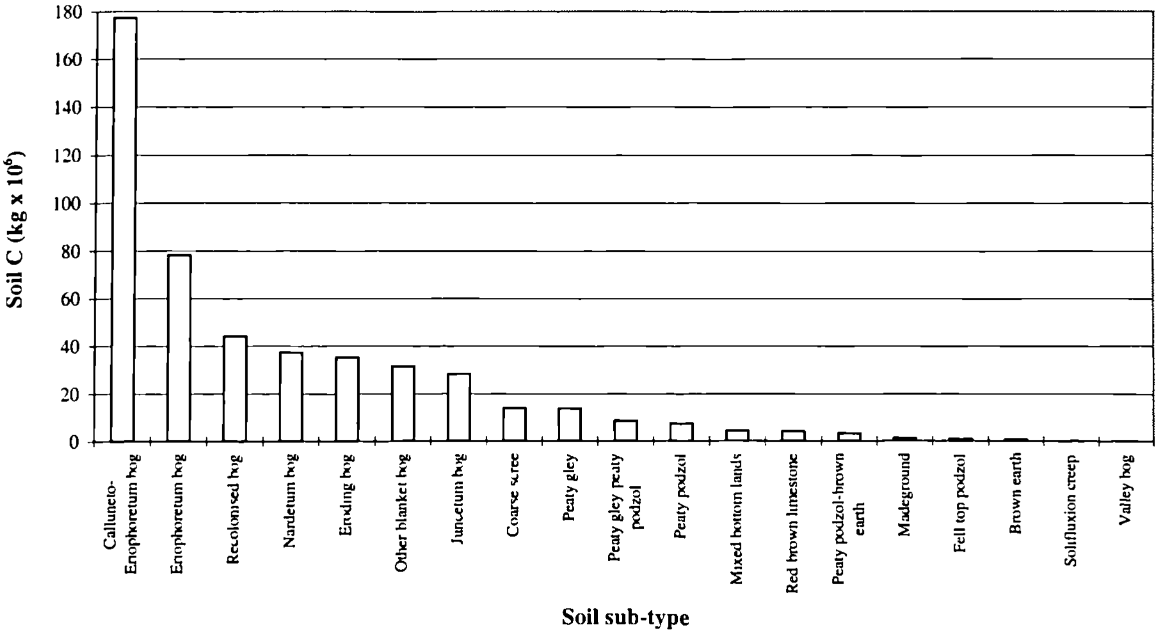

Figure 3.13 C stored in top 50 cm of Moor House soil sub-types

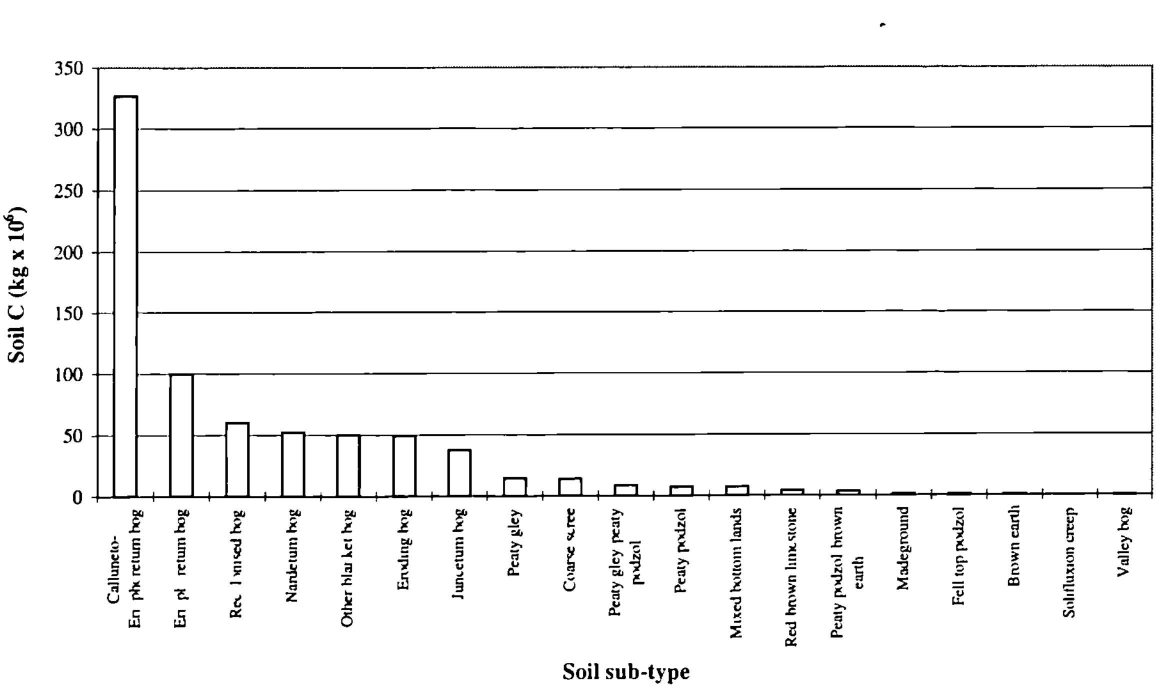

Figure 3.14 C stored in top 100 cm of Moor House soil sub-types

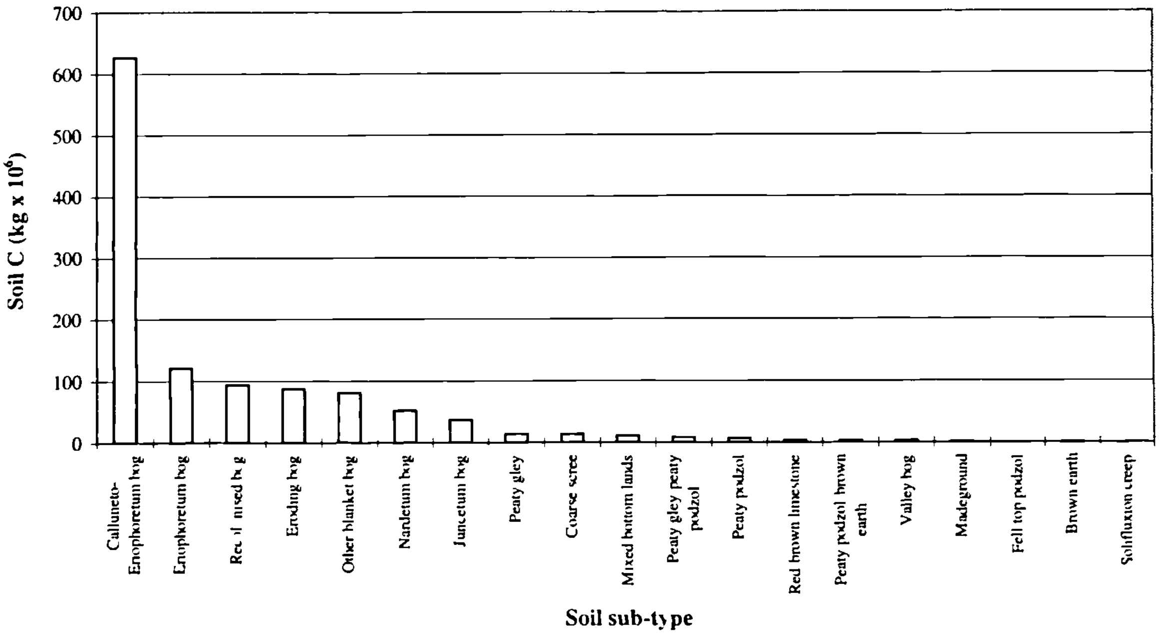

Figure 3.15 Total C storage in Moor House soil sub-types

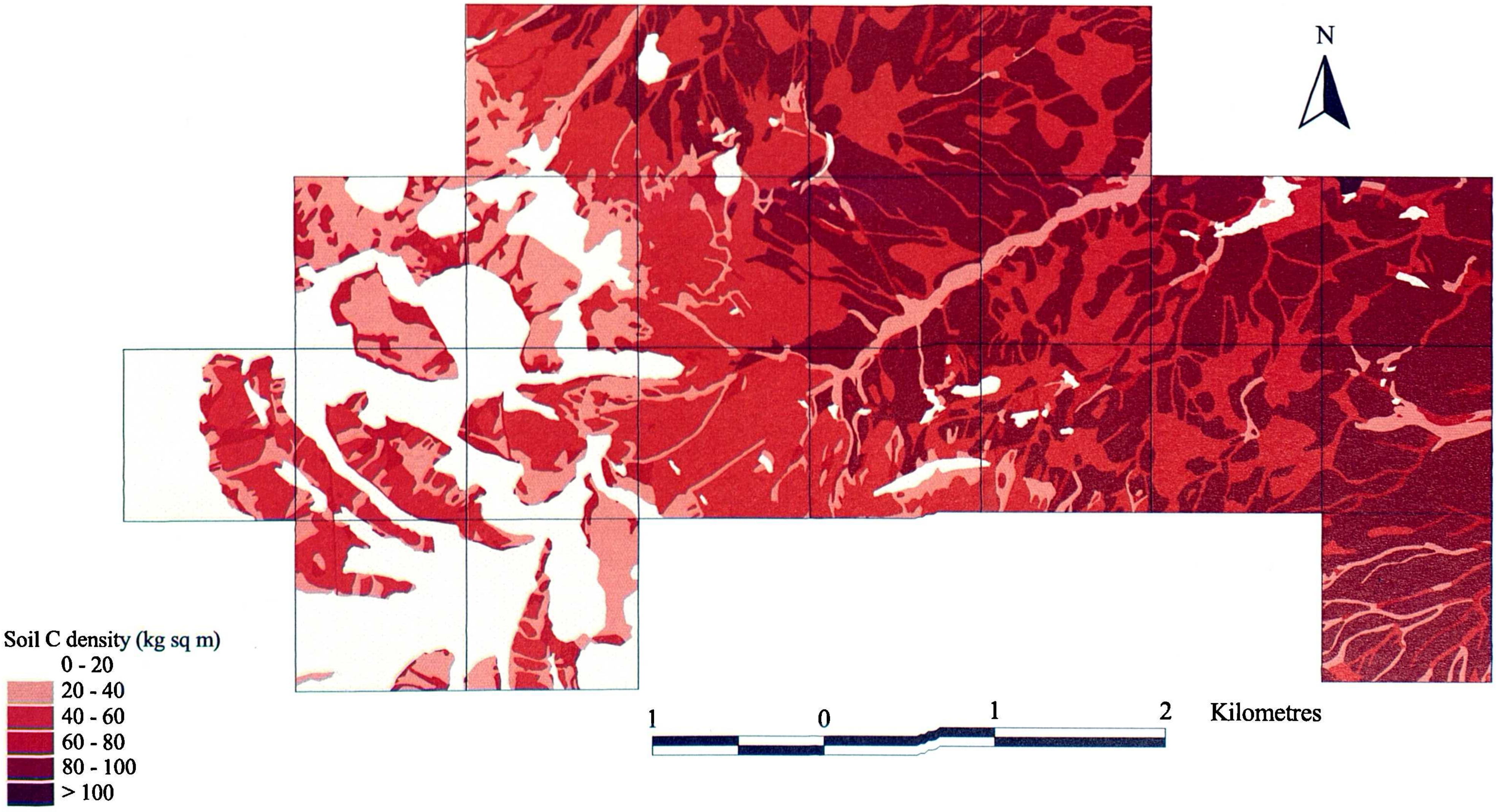

Figure 3.16 Soil C storage within the study area at Moor House

types contained the most C (particularly Calluneto-Eriophoretum). Though the Valley bog soil type had by far the greatest C density in its total depth (Figure 3.11), the total amount of C it contained was extremely small compared to other soil types.

Table 3.3 displays the soil C storage values to different depths for each kilometre square within the study site. The table shows a consistent trend of increasing C storage in each kilometre square when considering increasing soil depth. However, some kilometre squares showed a greater increase than others, for example, square 19 contained 15 x 10^6 kg in the top 30 cm, yet had 78 x 10^6 kg within the total soil depth. However, square 20 had 14 x 10^6 kg in the top 30 cm, though only had slightly more C when considering the total soil depth (22 x 10^6 kg).

The values of C density and soil C storage of each soil in each kilometre square of the study site, used to produce these summary figures, are contained within the Appendix (Section 10.4).

Table 3.3 Soil C in each kilometre square to depths of 30, 50, 100 cm and to total soil depth. All units are kg C x 10^6

Kilometre Square (see Fig. 2.1)	Top 30 cm	Top 50 cm	Top 100 cm	Total depth
1	16	24	34	48
2	15	23	37	63
3	15	23	40	72
4	15	23	39	71
5	14	21	27	33
6	14	20	26	29
7	17	26	36	50
8	15	23	39	72
9	15	22	37	66
10	15	23	38	69
11	15	23	41	80
12	16	20	24	28
13	15	20	25	31
14	14	19	25	32
15	17	25	34	44
16	16	24	37	56
17	15	24	38	64
18	15	23	36	66
19	15	24	42	78
20	14	17	20	22
21	15	19	23	26
22	15	24	43	76
Total	334	492	741	1,174

3.4 Discussion

To determine the importance of soil C storage within the global C cycle and to assess the influence of global soil C storage on climate, reliable estimates of contemporary soil C stocks are required. However, the value of these soil C inventories largely depends on their accuracy. The aims of this chapter have been i) to accurately quantify the soil C storage for an area of upland Britain, and, ii) to test the accuracy of the UK's national soil C inventory by comparing the estimates derived in this case study with the national database estimates, for the same area.

Soil C inventories, such as the UK's national C inventory, are produced by adopting a classification of soils, determining the area of each soil type in the study site, and quantifying the C storage by multiplying the area of the soil types by their soil C density. Errors in the inventories may be introduced in many ways, for example in determining the area of soil types or deriving representative values of soil C density. This case study used a detailed map of soil types and involved an extensive programme of field sampling to determine soil C density. Since the national inventory used soil maps of lower resolution and used soil C density values derived from other areas, it is believed that the case study provides a far more accurate estimate of soil C storage for the study site.

Many of the reported inventories of soil C have calculated C densities to different depths. For example, Batjes (1996) gives soil C densities for the soils of the world for the following depths: 0-30 cm, 0-50 cm, 0-100 cm and 0-200 cm. The UK's national C inventory uses soil C densities for the top 100 cm of soil for England and Wales, and the total soil depth for Scotland (Milne and Brown, 1997). Other inventories have quantified soil C in the top 30 cm of soil only, since this is believed to be the most sensitive to climate change. In the present study, soil C has been calculated to a range of depths (30 cm, 50 cm, 100 cm and total depth) to enable the quantification of the underestimate made when determining C densities to depths less than the total soil depth.

3.4.1 Area of soil types at Moor House

Possible inaccuracies associated with determining the area of soil types were similar to the sources of error encountered when deriving the area of vegetation classes. They have already been described (Section 2.4.1) and are only briefly discussed here.

The classification of soils, like vegetation, was a simplification of reality and therefore will have been a source of error. Soils are rarely defined by clear boundaries (though areas of eroding bog may be an exception) but typically merge from one soil to another over several metres. Furthermore, soil types are not consistent within areas mapped as a certain soil because small scale differences occur in the factors which determine the soil type. There may be alternative ways of classifying soils which lead to better estimates of C storage.

Soils are more difficult to map than vegetation because the identifying features of a particular soil are mainly below-ground and consequently difficult to observe. It is therefore not surprising that the soil map of Moor House (Johnson and Dunham, 1963) is less detailed than the vegetation map of Eddy *et al.* (1969), although the resolution of the soil map is still exceptional for an area of upland Britain. In practice, the boundaries of soil types are defined by changes in vegetation composition since this tends to be strongly influenced by soil type. However, because other factors affect the distribution of vegetation classes, error will have been introduced to the soil map.

The soil map of Moor House was published in 1963 yet it has been assumed that the distribution of soil types has not changed. This is reasonable for most soils, although the extent of peat erosion or recolonised blanket bog may have changed over the last few decades.

As discussed for the vegetation coverage of Moor House, inaccuracies in the estimate of the area of different soil types will have been introduced due to possible errors in the original mapping of soil types, which may have been augmented by small errors introduced during digitisation of the map.

An initial investigation of the area of soil types revealed that Blanket bog covered a far greater area than any other soil type. Since this soil type was anticipated to have a high and variable soil C density, it was stratified using the vegetation coverage in an attempt to improve the overall estimate of soil C storage for the study site. It was thought that C density of the Blanket bog may vary with vegetation class, reflecting variations in the depth of peat or hydrological conditions. Furthermore, the vegetation class of Eroding bog was believed to be a more accurate estimate of the true area of peat erosion at the site than the soil complex of Eroded blanket peat which contained eroded and uneroded areas. However, errors will have been introduced with inaccuracies in the vegetation map (Section 2.4.1).

3.4.2 C stored in soil types at Moor House

Unlike the case study of vegetation C at Moor House, detailed estimates of C density were not available for soil types and a programme of field sampling was considered necessary; the errors associated with this sampling and the results obtained are discussed below.

The representativeness of samples used to derive soil C densities for the soil types of Moor House depends on the number of samples taken and the extent that sampling locations were typical for the soil type. The number of samples obtained during the programme of field sampling was largely limited by time constraints, yet all soils were sampled with the exception of the soil complexes and the Valley bog soil type, and a total of 164 locations were investigated across the study site. However, in an attempt to obtain the best possible estimate of soil C storage, the soil types which were expected to contain the greatest amounts of C were investigated the most intensely. A preliminary study based on literature values for soil C density was conducted which indicated that the Blanket bog would be the most important soil type for C storage (Garnett *et al.*, 1996, 1997b). Consequently, the greatest amount of time was spent investigating this soil type.

The determination of soil C density of individual samples involved measuring the %LOI of sub-samples and converting these values to %C using a regression which had been previously derived for Moor House soils. However, this regression will have been a potential source of error even though the correlation between the two variables was high ($r^2 = 0.984$). %C is frequently estimated from %LOI (Ball, 1964; Allen, 1989; Jain *et al.*, 1997), although more accurate methods of deriving %C exist (Allen, 1989). Unfortunately, none of these methods for determining C concentration were available for this study.

Several soil complexes had been mapped within the study area; their component soil types were sampled and had C densities calculated. The soil complexes were not specifically sampled and C densities for them were determined from the values of the composite soil types. However, since no information was available about the proportions of the different soils within the complex, they were assumed to be equal, and the mean of the two soil types forming the complex was used to provide a C density value. This will have introduced error, and although specific sampling of the complexes would have provided a more representative value, time constraints precluded this.

The Solifluxion creep and Red-brown limestone soils were sampled as part of a study investigating the impact of grazing on soil C stores. The sampling locations were not randomly located, but were selected directly outside and within fenced sheep exclosures (see Chapter 4). Therefore, the sampling locations were not randomly selected, though have been assumed to be representative of the soil type.

It was necessary to use different methods to collect soil samples due to the range of characteristics in different soils sampled. The soils with the highest C densities, the Blanket bog sub-types, were relatively easily sampled since they could be cored using conventional peat corers. However, in order to obtain representative samples, sampling locations were randomly chosen and were widely scattered across the study site. Since the coring equipment and samples were of considerable weight, and due to the inaccessibility of the sampling points, the number of locations able to be investigated was limited. The other soils were generally more accessible though frequently more difficult to sample, and both soil corers and bulk density tins were used. This use of different methods to obtain samples was unavoidable yet may have introduced error into the estimation of soil C density of different soils, though tests undertaken to compare different methods indicated that any differences in sampling method were probably small when compared to other sources of error in evaluating soil C stores.

Initial attempts at determining soil C investigated the importance of slope, aspect and altitude by studying the variation in soil C density with each variable. These studies were conducted on a mineral (Brown earth) and peaty soil (Peaty gley), and it was thought that, if soil C density was strongly influenced by any of these variables, a more accurate estimate of soil C could be produced by using a three-dimensional model of the site. Since rainfall and temperature vary with altitude (see Chapter 6), the difference in soil C density between the two altitudes should reflect the importance of climate on the C density of these soils. It was anticipated that analysis of the same soil type at different altitudes would produce different estimates of C density because climate has been shown to strongly influence soil C density (Post *et al.*, 1982; Harrison *et al.*, 1995a). However, the results of *t*-tests showed no significant difference in soil C density with altitude for the two soils investigated. There was a high degree of variation in soil C density at both sites and there are many factors which influence soil C density which may have obscured the influence of climate on C density (*e.g.* Post *et al.*, 1982).

Similarly, no significant correlation was found between either slope or aspect with C density, for either soil type, though aspect and topography are believed to influence soil C density (Post *et al.*, 1982). It may be that, due to the large number of factors which determine soil C density (*e.g.* climate, topography, aspect, parent material, age of soil profile *etc.*) the importance of a single variable is obscured by other factors.

However, characteristics were observed from the soil sampling at Moor House which have been recorded elsewhere. For example, the depth of peat in the blanket bog was significantly correlated and inversely related to slope ($p < 0.01$; Figure 3.17), a relationship Tallis (1987) observed in work undertaken in the southern Pennines. Since peat accumulates under water-logged conditions where decomposition is suppressed (Clymo, 1984), areas of the poorest drainage, such as the lowest slopes, are likely to accumulate the greatest amounts of peat. Although C density in the total peat profile was strongly related to peat depth (Figure 3.18), C density varied with the surface vegetation class (Figure 3.11), and therefore, stratification of the blanket bog soil type by vegetation has been used in the estimates of C storage.

When considering only the top 30 cm, the Moor House soils had relatively similar C densities. However, three soils had markedly lower values than the others, with Madeground

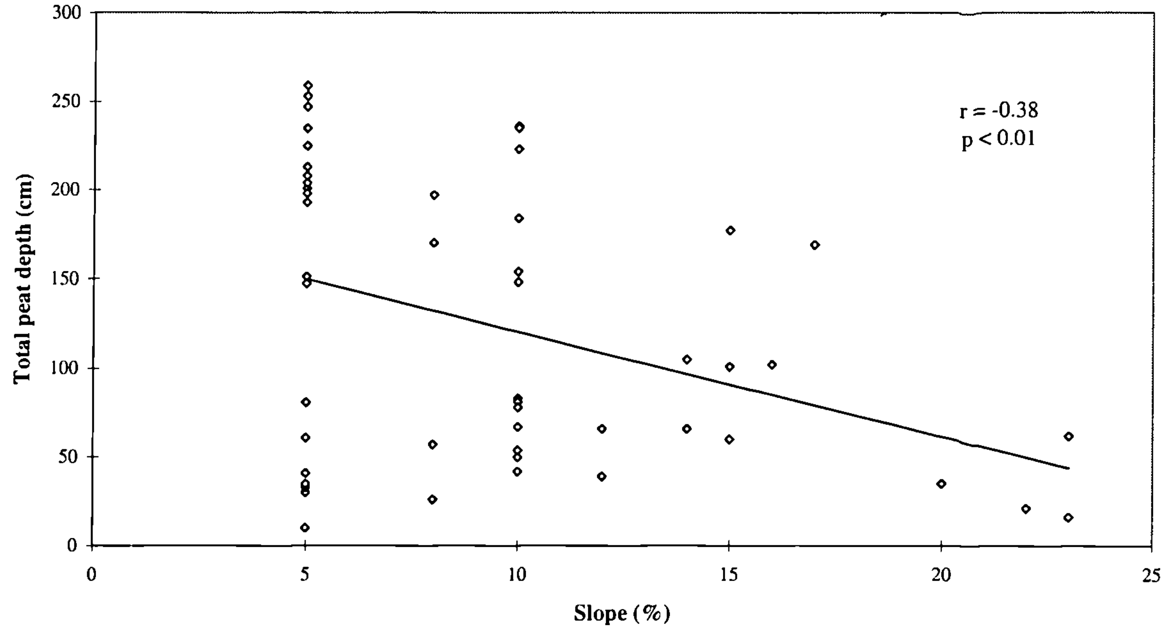

Figure 3.17 Plot of total peat depth against percentage slope, for all blanket bog sampling points

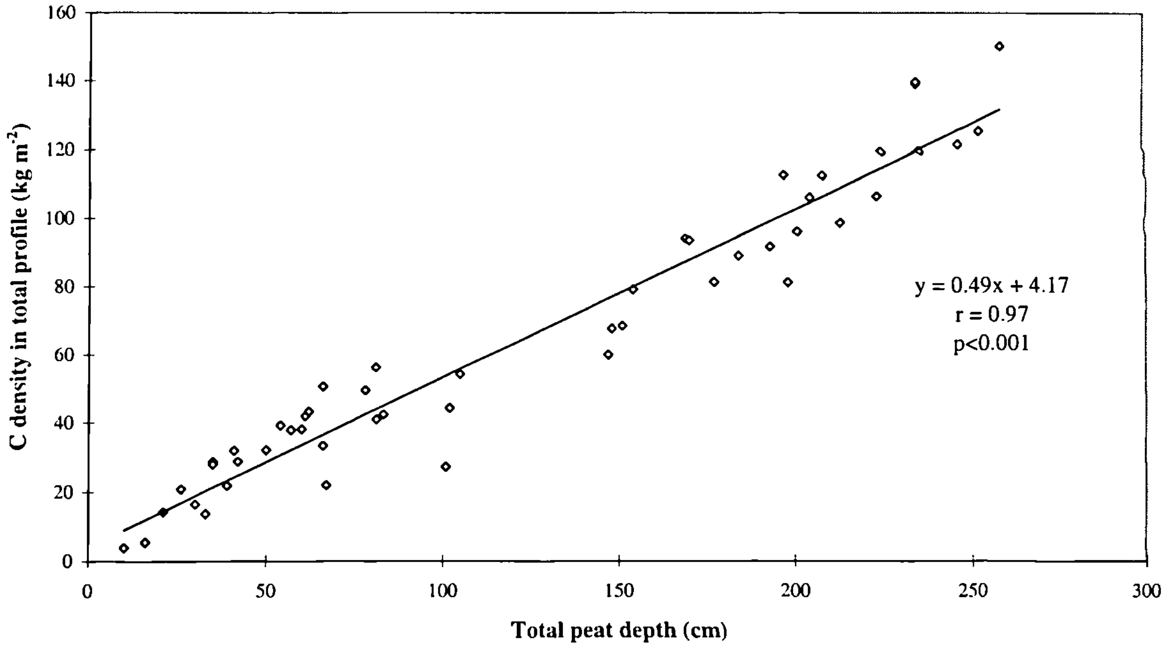

Figure 3.18 Plot of C density in total profile against total peat depth, for all blanket bog sampling points

and Solifluxion creep having low C densities due to the soils being less than 30 cm deep and having a low C concentration. The Valley bog soil type had a low C density in the top 30 cm because its bulk density was low near the surface (although its C concentration was high). When calculating C densities to greater depths, the range in values increased because some soils were less than 30 cm deep and therefore their C density did not change, though other soils extended deeper (*e.g.* the Blanket bog sub-types) and their C densities increased. The non-blanket bog soil types, by definition, had a peaty layer of less than 30 cm, and since non-peaty layers had a lower C concentration, the non-blanket bog soil types did not increase their C densities as much as the Blanket bog sub-types, when considering greater soil depths. The soil type with the highest value of C density when considering the total soil depth, was Valley bog, since this soil had a reported depth of 8.7 m and had a high C concentration.

It was clear from the results that a high natural variability in soil C density occurs both between soil types and within the same type. The error bars (indicating standard error) for the sampled soils reflect this, but because only a few sampling locations were investigated for some soils in the current study, it is likely that considerable reductions in the size of the error terms could be achieved with further sampling.

The total amount of C stored in the soils of the 22 one-kilometre squares of the study site was 1,174 x 10^6 kg, although there was a wide range in the amount of C stored in different soil types. Since the range of C densities was highest when considering the total soil depth, the range in amount of total C stored in each soil type was greatest when considering the total depth values. The soil sub-type containing the most C in the study site was the Calluneto-Eriophoretum bog, a result of its large area and high C density. Though the Valley bog soil type had by far the highest C density, it contained only a very small proportion of the study sites soil C since it covered only a very small area.

The soil C contained within individual kilometre squares in the study site varied greatly, being mainly a reflection of the proportion of Blanket bog sub-types in each square. For example, square 20 contained a relatively small area of Blanket bog and had a total soil C storage of 22 x 10^6 kg. However, square 22 was almost completely covered by Blanket bog and was estimated to contain 76 x 10^6 kg of soil C.

The results confirmed that the Blanket bog contained by far the greatest amount of C at this site, and that in order to obtain an accurate estimate of soil C storage reliable estimates of C storage for this soil are required. The importance of peat soils for C storage has been identified in many other studies (see Post *et al.*, 1982; Batjes, 1996; Milne and Brown, 1997) with Scottish peats alone being estimated to contain between 46% (Milne and Brown, 1997) and 75% (Howard *et al.*, 1995) of the total soil C in Great Britain.

3.4.3 Comparison with the UK's national C inventory

The national soil C inventory contains an estimate for soil C within each kilometre square of the UK (Milne and Brown, 1997). These estimates were made from a classification of the soils of the UK, with the soil covering the dominant area within each kilômetre square being selected as representative of the square. The whole square was therefore assumed to comprise this single soil type and the C storage determined from the C density of this soil type. This approach contrasted with the case study reported here since variation in soil types within the kilometre square was included. Furthermore, the estimates of soil C density in the national database were derived from samples collected elsewhere in Britain, though the case study values were derived from samples taken within the site. Consequently it is believed the case study provides a far more accurate estimate for soil C for this site.

Table 3.4 shows the values of soil C storage for each kilometre square determined by this case study and the national inventory values for the same kilometre squares. The case study values used in this comparison were the values to a depth of 100 cm as these were comparable to the national inventory values (the national inventory calculated soil C in England and Wales to a depth of 100 cm).

In all kilometre squares the case study values were lower than the national inventory values, with considerable differences between the two estimates for most squares; in most squares the national database value was about three times higher than the case study value and was over five times higher in square 20. The square with the lowest difference between the two estimates was square 21, the only square which had been assigned, by the national inventory, a C storage value lower than the 113 x 10^6 kg estimated for the other squares.

When considering the complete 22 squares of the study site, the national database estimate was over three times higher than the case study value. There are several reasons which may account for these differences:

The national inventory is based on soil maps of lower resolution than the one used in the case study. Furthermore, the resolution was even lower in the national database because only the dominant soil from each kilometre square was used to derive C storage estimates (Milne and Brown, 1997). For example, most kilometre squares in the study site were classed as Winter Hill blanket bog because the maps used by the national inventory indicated that this was the dominant soil series in the squares. The C content of each kilometre square classed as Winter Hill was derived using an estimate of C density for this soil type. Since 21 of the 22 squares at Moor House had been classified as Winter Hill, all but one of the squares had a C content of 113 x 10^6 kg. Square 21 was classified as Wilcocks peaty gley because this soil was dominant in this square, and had a lower C content of 34 x 10^6 kg. Since the national database simplified each kilometre square so that it was represented by only one soil type, variation within the square will have been overlooked, resulting in inaccuracies.

Table 3.4 Comparison of case study and national inventory soil C estimates by kilometre square. All units are kg C x 10^6

Kilometre Square (see Fig. 2.1)	Case study 100 cm	National database	Difference
1	34	113	79
2	37	113	76
3	40	113	74
4	39	113	74
5	27	113	86
6	26	113	87
7	36	113	77
8	39	113	74
9	37	113	76
10	38	113	75
11	41	113	72
12	24	113	89
13	25	113	88
14	25	113	88
15	34	113	80
16	37	113	76
17	38	113	76
18	36	113	77
19	42	113	71
20	20	113	93
21	23	34	11
22	43	113	71
Total	741	2,412	1,671

The kilometre square with the most similar C storage value in the two inventories was square 21. This suggests that the process of generalising the kilometre squares to a single soil type, as in the national database, resulted in a mis-classification of squares, or that the original soil maps were inaccurate. Alternatively, the value of C density for the Winter Hill soil series may be too high and unrepresentative of the Moor House soils.

In the national database, the C density of the Winter Hill soil series, the main blanket bog soil type in England and Wales, has been estimated to be 113 x 10^6 kg per km^2, or 113 kg C m^{-2}. Since the national database estimates soil C in England and Wales only to a depth of 100 cm, this amount of C stored in the Winter Hill series must be to a depth of 100 cm. If it is assumed that the C content of peat is 50% as reported elsewhere (Heal and Smith, 1978; Allen, 1989; Immirzi *et al.*, 1992), then the mean bulk density of the Winter Hill soil series must have been about 0.23 g cm^{-3}. Since this is considerably higher than most published measurements of peat bulk density (*ca.* 0.1 g m^{-3}; Clymo, 1983), it may be that the Winter Hill soil C density is untypical of blanket bog, and that an estimate derived from the sampling at Moor House would be more representative. Assuming this to be the case, a revised estimate for the C storage of England and Wales can be made because the Winter Hill soil series covers an area of 2,575 km^2 (based on the Legend of the 1:250,000 Soil Map of England and Wales, 1983). The mean value of C stored in the 21 kilometre squares classified as Winter Hill was 34 x 10^6 kg km^{-2} using this case studies values. Therefore, the new estimated C content of the Winter Hill blanket bog for England and Wales was 88 x 10^9 kg (*i.e.* 2,575 x 34 x 10^6 kg) based on the Moor House values, and 292 x 10^6 kg based on the national database estimate for C storage in the Winter Hill soil series (*i.e.* 2,575 x 113 x 10^6 kg). Applying the Moor House values of C storage for the Winter Hill, the estimated total amount of soil C in England and Wales is reduced from 2,890 x 10^9 kg to 2,687 x 10^9 kg, a reduction of 7%.

The large difference in estimated soil C storage between this case study and the national database illustrates the difficulties in deriving inventories of soil C. Although Moor House contains soils of high C storage and high variability, the discrepancies in the estimates for this site mirror the uncertainty in both national and global estimates of soil C. For example, Howard *et al.* (1995) estimated the soil C content of Great Britain to be 21,784 x 10^9 kg, while a revised estimate produced by Milne and Brown (1997) of 9,838 x 10^9 kg was half the previous estimate. Estimates of global soil C have ranged from 700 x 10^{12} kg to 2,946 x 10^{12} kg (see Post *et al.*, 1982) and although many recent estimates have fallen between these values (*e.g.* Batjes, 1996), the accuracy of these estimates is uncertain since they were derived from less detailed investigations than the UK national C inventory.

4. The impact of management on terrestrial carbon storage at Moor House

4.1 Introduction

The quantity of C stored in terrestrial ecosystems is determined by many factors, many of which are influenced by anthropogenic activities, such as changes in land use. These activities affect terrestrial C stores by influencing rates of primary production and decomposition, or more directly through the removal of biomass, for example, by burning or harvesting vegetation. Any decrease in terrestrial C storage is likely to be balanced by an increase in atmospheric CO_2 which will add to the rising concentration of this greenhouse gas. Consequently, land uses or land management activities affecting terrestrial C stores offer the potential to mitigate anthropogenic emissions of CO_2. Furthermore, by encouraging land uses which increase soil and vegetation C storage, it may be possible to offset a proportion of the CO_2 derived from industrial sources and, therefore, reduce the rate of increasing atmospheric CO_2 concentration (IPCC, 1996).

The previous two chapters have described the large amounts of C held in the vegetation and soil at Moor House. Since Moor House can be considered representative of Pennine moorland and upland areas in Britain (Heal *et al.*, 1975), the C stored in these areas represents a considerable C reservoir which, if transferred to the atmosphere, could cause a significant increase in atmospheric C concentrations. It is therefore important to understand the C balance in these ecosystems and to determine whether current anthropogenic activities influence the extent to which these systems are a source or sink for C.

There is a long history of human influence at Moor House (Johnson and Dunham, 1963) although anthropogenic impacts have been minimal at the site since it was designated a Nature Reserve in 1952. However, in common with most upland areas in Britain, sheep grazing is widespread within the Reserve; apart from a small open-cast mine to the west of Great Dun Fell (Figure 1.5), the site is largely used only for scientific research and recreation.

The impact of several management activities currently practised in the upland areas of Britain has been investigated on the Reserve over several decades (Rawes and Hobbs, 1979). The influence of sheep grazing on botanical composition and biomass production of vegetation communities at Moor House has been studied extensively (*e.g.* Rawes, 1983; Forrest and Smith, 1975) and the effects of rotational burning, a technique used to improve the grazing value of moorland, have also been investigated (*e.g.* Rawes and Hobbs, 1979).

The research undertaken at Moor House, and other locations, has shown that grazing and burning can have considerable effects on the standing biomass, and therefore, the vegetation C content of certain communities (Smith and Forrest, 1978; Perkins *et al.*, 1978). However, the impact of these activities on total terrestrial C storage is poorly quantified and although it is known that rates of vegetation productivity can be affected by grazing, very little research has been carried out into the effects of grazing on rates of decomposition. Similarly, the impact of burning on terrestrial C storage is poorly understood despite the recognised importance of this management activity (Hobbs and Gimingham, 1987) in locations with the highest C storage in Britain (Rowell, 1988).

This chapter describes studies into the influence of sheep grazing and moorland burning on terrestrial C stores by utilising existing long-term experiments at Moor House. The effects of sheep grazing on vegetation and soil C stores have been studied at two upland grassland and one blanket bog site, while the impact of moorland burning has been investigated at the blanket bog site only, since this management activity is not commonly practised on upland grasslands. Since the investigations at the grassland and blanket bog sites required very different sampling and laboratory techniques, the blanket bog study has been described separately.

4.1.1 Study 1: Impact of sheep grazing on the terrestrial C storage of two grassland sites

The impact of sheep grazing has frequently been studied by contrasting areas where sheep have been allowed to graze with plots where they have been excluded. This has normally been performed by selecting an area of uniform grazed land and constructing a stock-proof exclosure. The impact of removing grazing has been investigated by comparing characteristics, such as botanical composition, within the exclosure, with surrounding areas or with a nearby unenclosed control plot. Long-term studies into the effects of sheep grazing have been performed at Moor House using exclosures in a variety of locations and with differing vegetation communities; several of these exclosures have been used in the present work.

In the current study the impact of sheep grazing on terrestrial C storage was investigated by quantifying vegetation and soil C within and adjacent to sheep exclosures. The aims were to determine whether sheep grazing had significantly affected the C stores at Moor House and, consequently, whether management options associated with sheep grazing could be employed to increase terrestrial C storage at this and similar sites.

4.1.1.1 Description of sites

Sheep exclosures located at Little Dun Fell and Knock Fell (Figure 4.1) were chosen for this study because they had been established on different vegetation and soil types, and had been maintained for over four decades; any short or mid-term affect of removing sheep on C storage in these areas should be evident after this length of time. Furthermore, the exclosures contained soils which could be consistently sampled and, therefore, accurate estimates of soil C could be derived for the different treatments.

The exclosures were located on the main ridge of high ground on the western side of Moor House; characteristics of the two sites are provided in Table 4.1. The areas surrounding the exclosures have both been subjected to high grazing pressure relative to the rest of the Reserve, since the vegetation has higher grazing value than the blanket bog vegetation classes.

Table 4.1 Characteristics of the sheep exclosure at Little Dun Fell and Knock Fell

	Little Dun Fell	Knock Fell
Grid reference	NY 705331	NY 718312
Altitude (m)	830	750
Vegetation prior to exclosure	*Festuca ovina* grassland	Agrostis-Festuca grassland
Soil type	Solifluxion creep	Red-brown limestone
Geology	Sandstone	Limestone
Date of exclosure	1954	1955

The Little Dun Fell exclosure is situated on steeply sloping ground and has an easterly aspect and dimensions of *ca.* 30 m x 30 m. The Knock Fell exclosure is of similar aspect and size, although is located on a gentler slope.

4.1.2 Study 2: Impact of burning and sheep grazing on C accumulation in blanket bog

Since blanket peat contains by far the greatest amount of C at Moor House (Chapter 3) and peatlands hold most of the C in Great Britain (Howard *et al.*, 1995; Milne and Brown, 1997), it is particularly important to determine the impact of land use on these soils. Additionally, since mires can be major sinks for C, practical measures may exist to increase terrestrial C storage through increasing blanket bog C stores.

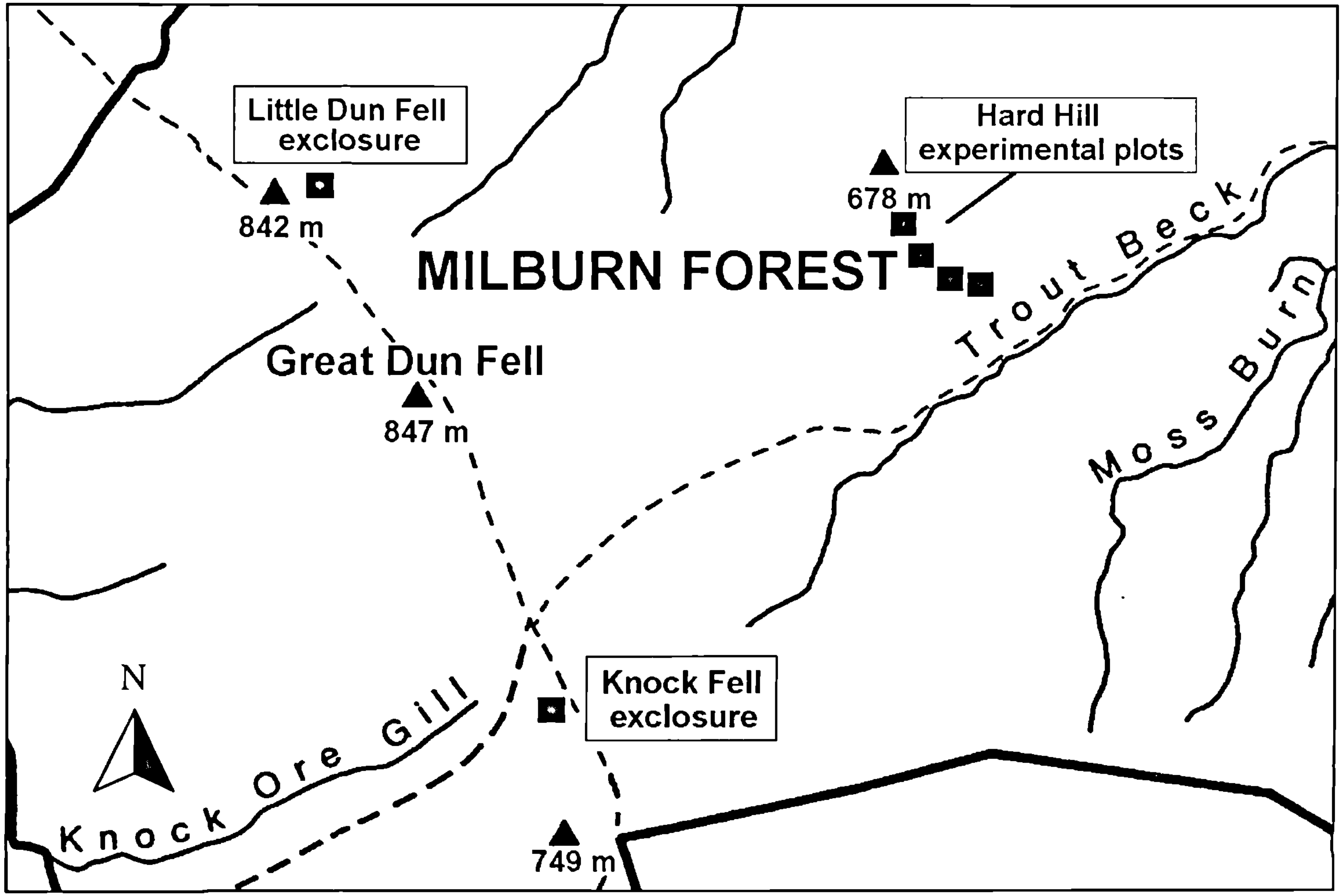

Figure 4.1 Location of Hard Hill experimental plots and Knock Fell and Little Dun Fell exclosures

As mentioned above, sheep grazing may affect soil C storage by reducing C inputs from plant litter (due to removal by grazing) or by influencing rates of primary production and decomposition. These influences are likely to apply to the blanket bog, although there may be impacts of sheep grazing which are particularly important to peats. For example, trampling may be of greater consequence for peats than mineral soils because peats typically have a low density, and high water content, making them easily compressed in the surface layers. Since peat accumulation, and therefore C accumulation, is strongly controlled by hydrological conditions (Clymo, 1984), any disruption of surface hydrology caused by trampling may influence C accumulation rates.

Large areas of moorland are traditionally burnt in upland Britain to provide uneven-aged stands of heather *(Calluna vulgaris)* which provide suitable conditions for red grouse (Hobbs and Gimingham, 1987). Regular burning is believed to improve the grazing value of heather moorland for sheep, although Hobbs (1984) has questioned this assumption. Few studies have investigated the impact of rotational burning on blanket bogs (Hobbs, 1984), although results obtained from experimental studies at Moor House indicate that this practice caused major changes in the species composition of plant communities (Hobbs, 1984); the impacts on rates of C accumulation in peats are poorly understood.

In the current study, existing randomised block experiments at Moor House were used to investigate the impacts of these two management activities. The aim was to establish whether grazing or burning influenced C accumulation in peats and the potential of these management options for modifying terrestrial C stores.

The study involved identifying a chronologically synchronous layer within the stratified peat profiles and determining the amount of C contained above this layer, in each of the treatments. For this technique a 'fixed point' layer of peat, formed at the same time between plots was required. This layer had to have been formed before the start of the experiment and, to allow small changes in C accumulation caused by the treatments to be identified, the amount of C represented above the 'fixed point' and below the start of the experiment had to be small to reduce natural variability in C accumulation.

There were several potential chronological markers preserved in the peat which could have been used for fixed points in the present study, such as pollen signals (*e.g.* the rise in *Pinus* pollen associated with plantations of the 18th-19th centuries; *e.g.* Clymo *et al.*, 1990), or tephra layers. Alternatively, radiometric techniques could have been employed; ^{210}Pb dating (Oldfield *et al.*, 1995; Appleby *et al.*, 1997) may have been particularly useful as absolute dates could have been provided for many depths, allowing a synchronous layer to be accurately determined.

The present study utilised the record of spheroidal carbonaceous particles (SCPs) preserved within peat profiles, since these particles can be rapidly analysed at low cost (Rhodes, 1996). These particles are formed from oil and coal combustion and records of their deposition follow the historical fuel consumption of the area where the particles were deposited (Wik and Renberg, 1996). Therefore, in lake sediments and stratified peat profiles a rapid increase in the number of SCPs reflects industrialisation and the increase in fuel consumption in a region. However, the precise date of this 'take-off' will vary between sites since it may have been produced by localised industrialisation which commenced at different times in different regions (Wik and Renberg, 1996). However, preliminary analyses of SCPs at Moor House suggested that the SCP 'take-off' represented a pre-1940 date (Chapter 5), and therefore, the technique would be suitable to provide a fixed point for the present study. Furthermore, since the Hard Hill experimental plots were located within metres of each other, it was reasonable to assume that each had received the same historical load of SCP deposition. Therefore, a 'take-off' in SCPs would indicate a peat layer of synchronous age, allowing any treatment effects on C accumulation to be distinguished. However, without an accurate date for the 'take-off', absolute rates of C accumulation (*e.g.* in g m^{-2} yr^{-1}) could not be calculated, although this was beyond the principal aim of the study. Records of SCPs derived in the present study are discussed further in Chapter 5.

4.1.2.1 Description of site

The Hard Hill burning experiment was established in 1954 to investigate the effects of grazing and rotational burning on blanket bog vegetation (Plate 4.1). The experiment contains three burning treatments, with a fenced and unfenced version of each burning treatment enabling the interactive impacts of grazing also to be studied. Each treatment is replicated in four blocks and located on a uniform and generally uneroded slope to the south east of Hard Hill (Figure 4.1). Burning of the plots is undertaken manually using the traditional methods employed for moorland burning. Characteristics of the site are provided in Table 4.2.

Plate 4.1 Hard Hill burning and grazing experiment (Photograph by J.K Adamson, Institute of Terrestrial Ecology)

Table 4.2 Site characteristics of Hard Hill burning experiment (from Marrs *et al.*, 1986)

	Hard Hill plots
Grid references:	
Block A	NY 743330
Block B	NY 740330
Block C	NY 736330
Block D	NY 738331
Altitude (m)	600-630
Original vegetation	Calluneto-Eriophoretum
Soil type	Blanket bog

4.2 Method

4.2.1 Study 1: Impact of sheep grazing on the terrestrial C storage of two grassland sites

The present work investigated the effects of sheep grazing on C storage by analysing soil and vegetation within and immediately outside two exclosures where sheep grazing has been excluded for over 40 years. Field sampling was carried out in September 1996 when above-ground biomass values were at a maximum.

4.2.1.1 Sampling procedure

Sampling points for vegetation and soil were located on all four sides of the exclosures and at regular distances to allow for inherent natural variability. Unfortunately, this was not possible at the Knock Fell site since the Pennine Way footpath runs along one side of the exclosure, and a sink hole is located immediately adjacent to another. Therefore, one complete side and part of a second side were not sampled.

At the Little Dun Fell exclosure, three sampling points were located along each side, with paired samples being taken inside and outside the exclosure at a slight distance from the fence to avoid areas which may have been influenced by excess trampling by people and sheep. Sampling points were chosen to have similar aspects and slopes. Twelve paired samples (*i.e.* a total of 24) were taken.

At the Knock Fell exclosure, sampling points were located in the same way as for Little Dun Fell. However, due to the problems outlined above, only eight paired sampling points were chosen, with samples being taken inside and outside the exclosure at these locations.

Vegetation samples were taken by clipping an area, 25 cm x 25 cm, with shears and scissors. All living and dead vegetation within this area was removed down to the surface of the soil litter layer, and soil samples were also taken from these same points by coring to bedrock; a short circular corer (internal diameter 3.6 cm and length 26 cm) was used for thin soils and a larger corer (internal diameter 5.2 cm and chamber length 36 cm) which was able to be pushed below the soil surface, was used where the soil was thicker. Although different coring instruments were used depending on the type of soil being sampled, each technique was performed to achieve the same end point. However, so as not to bias the results, each pair of grazed and ungrazed samples were taken using the same corer.

Where distinct pedological horizons could be recognised, the soil cores were divided along these boundaries, though not all samples had the same horizons (the variation in soils at the Knock Fell site was particularly high). Details of the depth of the soil, and vegetation composition, were noted for each sample.

All soil samples were returned to the laboratory and analysed for C content. Samples were oven dried at 105°C to constant weight and the dry weight determined. Sub-samples were taken from each horizon (where they had been separated) and the %LOI determined (Allen, 1989; see Section 3.2.3.3 for method). Existing regressions between %C and %LOI derived for soils at Moor House (Bol *et al.*, in press; equation 3.9) were used to determine the C concentration of the samples. The C density was then calculated from the values of soil dry weight and %C. Vegetation was oven dried at 105°C until constant weight and C concentration assumed to be 45% of the vegetation dry weight (Allen, 1989).

During the field sampling it was evident that for the Little Dun Fell exclosure the proportion of exposed bare rock was greater outside the exclosure than within, and that this may have been an important aspect in terms of C storage. Measurements of the area of rock cover were therefore made to improve estimates of the soil and vegetation C density.

4.2.1.2 Statistical analyses

Paired *t*-tests were performed on the data for each exclosure to determine whether there were statistically significant differences in the C held in the vegetation and soil components of grazed and ungrazed areas. The SAS procedure UNIVARIATE was used (SAS Institute, 1990).

4.2.2 Study 2: Impact of burning and sheep grazing on C accumulation in blanket bog

The randomised block experiment at Hard Hill consists of three burning treatments: i) burnt 1954 and every 10 years since, ii) burnt 1954 and every 20 years since, and iii) burnt 1954 only. Each treatment had a fenced and unfenced plot (30 m x 30 m) within each block to study the impact of sheep grazing. In the present study two burning treatments have been investigated; the 10 year rotation ('burnt plot') and the '1954 burnt only' control (referred to as the 'unburnt plot'). Grazed and ungrazed sub-plots within the unburnt plots have been sampled to determine the influence of grazing on C accumulation in peats. A key to the different treatments investigated in this study is provided in Table 4.3.

Table 4.3 Experimental layout of Hard Hill burning experiment used in this study, showing treatment reference codes

Block	Burning treatment	Grazing treatment	Reference
A	Burnt 1954 and every 10 years	Grazed	A/GB
A	Burnt 1954 only	Grazed	A/G
A	Burnt 1954 only	Ungrazed	A/U
B	Burnt 1954 and every 10 years	Grazed	B/GB
B	Burnt 1954 only	Grazed	B/G
B	Burnt 1954 only	Ungrazed	B/U
C	Burnt 1954 and every 10 years	Grazed	C/GB
C	Burnt 1954 only	Grazed	C/G
C	Burnt 1954 only	Ungrazed	C/U
D	Burnt 1954 and every 10 years	Grazed	D/GB
D	Burnt 1954 only	Grazed	D/G
D	Burnt 1954 only	Ungrazed	D/U

4.2.2.1 Sampling procedure

Short peat cores were extracted from the central part of each experimental plot in September 1997 by pushing a plastic tube into the peat surface, after first carefully incising an outline of the core through the top few centimetres of peat with a knife. The core was dug out with a spade and the base of the core severed using a knife. The cores were then placed in labelled plastic bags and sealed. The aspect, slope, vegetation composition and total depth of peat were recorded at each coring location.

4.2.2.2 Laboratory analyses

On returning to the laboratory, each core was stored in a refrigerated room until analysed. The cores were extracted, one centimetre at a time, by gently forcing the core vertically out of the plastic tubing. One-centimetre thick horizontal sections were taken using a sharp knife, weighed while fresh and wrapped in aluminium foil. A sub-sample of approximately 20-40 g was removed to determine the C content of each section. This sub-sample was weighed while fresh and oven-dried (105°C) for 24 hours, cooled in desiccators and weighed again to determine the moisture content (%; using equation 3.7). The sub-sample was assumed to be representative of the whole section, and therefore, the total dry mass of the complete section could be calculated using:

$$D_t = W_t \times (100 - \%M_s) / 100 \qquad \textit{[equation 4.1]}$$

Where:

D_t = dry weight of total section (g)
W_t = wet weight of total section (g)
$\%M_s$ = moisture content of sub-sample (%)

The C concentration of peat was assumed to be 50% dry mass since this had been consistently derived for blanket bog samples analysed when determining the soil C storage of Moor House (Chapter 3), and has frequently been reported for the C concentration of peat elsewhere (Heal and Smith, 1978; Allen, 1989; Immirzi *et al.*, 1992).

To determine the industrial 'take-off' in deposition of SCPs, samples were initially selected at intervals of 4 cm down the profile of each core. The SCP concentration of these 'range-finder' samples was determined and then a second batch of samples selected by choosing the part of the profile which was expected, based on the 'range-finders', to contain the industrial SCP 'take-off'. Determination of the SCP concentration of every sample from each core was not possible (and unnecessary) due to time constraints.

The method used to quantify the SCPs in each sample was based on Rhodes (1996) and involved the following stages:

1) 0.2 g of air-dried peat was placed in a 250 ml conical flask. 20 ml of distilled water was added and the solution covered and left for 24 hours to allow the peat to re-hydrate.
2) 20 ml of 6% hydrogen peroxide was added to the solution, which was again covered and placed in an oven at 50°C for 48 hours.

3) The solution was filtered through a Whatman Number 1 filter paper; the filter paper contents were retained and the liquid discarded.
4) The filtrate was carefully washed into a 9 cm diameter plastic Petri-dish using distilled water and returned to the oven (at 50°C) to evaporate the excess liquid.
5) 20 ml of 6% hydrogen peroxide was then added and the lid placed on the Petri-dish. The solution was replaced in the oven at 50°C for 48 hours.
6) The lids to the Petri-dishes were then removed allowing the liquid to evaporate after which they were ready to count using a stereo microscope (Wild M3Z, Heerbrugg, Switzerland) at x40 magnification.
7) Eight transects, covering more than half the Petri-dish, were scanned under the microscope and the total number of identifiable SCPs counted; it was found from a preliminary investigation that the total amount of SCPs in samples could be accurately estimated by scanning these 8 transects (see Chapter 5).

4.2.2.3 Statistical analyses

The quantity of C contained in the peat above the SCP 'take-off' in each core was calculated and an analysis of variance (ANOVA) performed to establish whether there were significant treatment effects. The ANOVA was undertaken using the SAS procedure ANOVA (SAS Institute, 1990) with *a posterori* multiple treatment comparison (Tukey).

4.3 Results

4.3.1 Study 1: Impact of sheep grazing on the terrestrial C storage of two grassland sites

Tables 4.4 and 4.5 show the dominant vegetation components at each sampling location under grazed and ungrazed treatments at the Little Dun Fell and Knock Fell exclosures, respectively. The results indicate that the vegetation composition under grazed conditions was frequently different to the ungrazed at both sites, particularly at Little Dun Fell.

Figures 4.2 and 4.3 show the C density of vegetation and soil at Little Dun Fell, under the two treatments. The total terrestrial C, the sum of the soil and vegetation components, is shown for each sampling location in Figure 4.4. Figures 4.5 to 4.7 display characteristics of the soil within the different treatments at this site; Figure 4.5 shows the total soil depth and Figures 4.6 and 4.7 display the soil bulk density and %LOI respectively.

Table 4.4 Little Dun Fell. Vegetation at sampling points

Sampling point	Grazed	Ungrazed
1	*Festuca ovina*	*Carex bigelowii*
2	*Festuca ovina*	*Carex bigelowii* and *Deschampsia flexuosa*
3	*Festuca ovina*	*Carex bigelowii* and *Deschampsia flexuosa*
4	*Festuca ovina*	*Carex bigelowii* and *Deschampsia flexuosa*
5	*Festuca ovina*	*Carex bigelowii* and *Deschampsia flexuosa*
6	*Festuca ovina*	*Carex bigelowii* and *Deschampsia flexuosa*
7	*Festuca ovina*	*Deschampsia flexuosa*
8	*Festuca ovina*	*Deschampsia flexuosa*
9	*Festuca ovina*	*Carex bigelowii*
10	*Festuca ovina*	*Carex bigelowii*
11	*Festuca ovina*	*Carex bigelowii*
12	*Festuca ovina*	*Carex bigelowii* and *Deschampsia flexuosa*

Table 4.5 Knock Fell. Vegetation at sampling points

Sampling point	Grazed	Ungrazed
1	Agrosto-Festucetum	Agrosto-Festucetum
2	Agrosto-Festucetum	Agrosto-Festucetum
3	Juncetum squarrosus	Agrosto-Festucetum
4	Agrosto-Festucetum	Agrosto-Festucetum
5	Agrosto-Festucetum	Agrosto-Festucetum
6	Nardetum sub-alpinum	Agrosto-Festucetum
7	Nardetum sub-alpinum	Agrosto-Festucetum
8	Agrosto-Festucetum	Agrosto-Festucetum

The results of the C storage determination for each sampling location at Knock Fell are shown in Figures 4.8 to 4.10, displaying vegetation, soil and total terrestrial C density. Figure 4.11 provides the soil depth measurements for each sampling location at Knock Fell.

The mean values of C storage for both the Little Dun Fell and Knock Fell sites are shown in Table 4.6. The table displays the mean vegetation, soil and terrestrial C, and gives

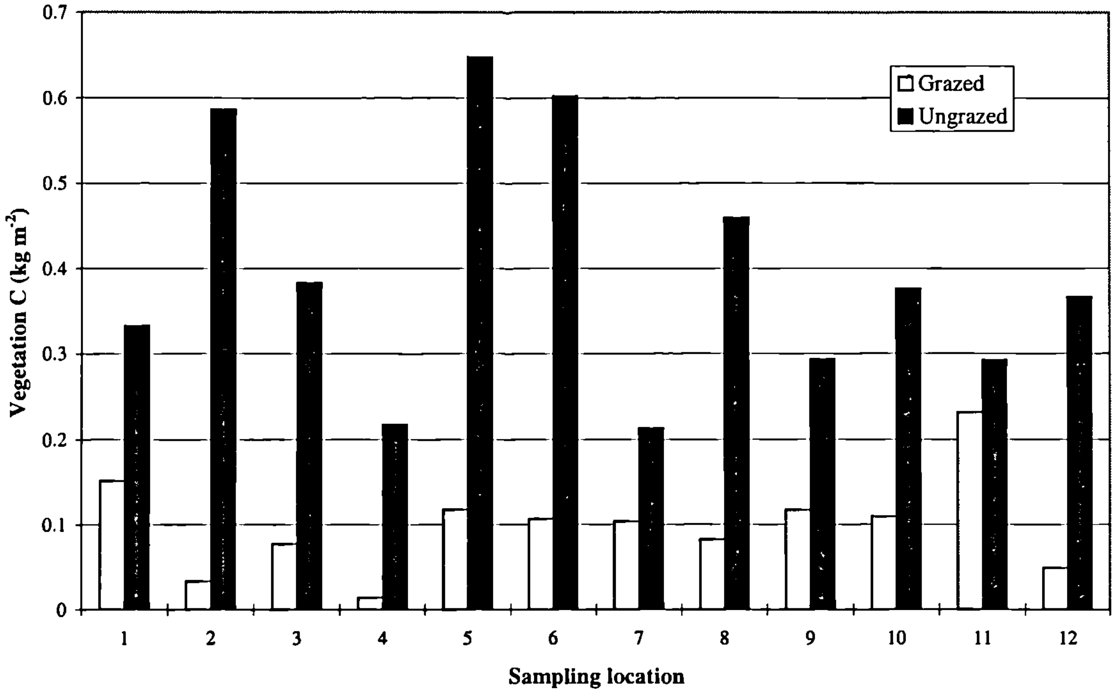

Figure 4.2 Little Dun Fell. Vegetation C under grazed and ungrazed treatments

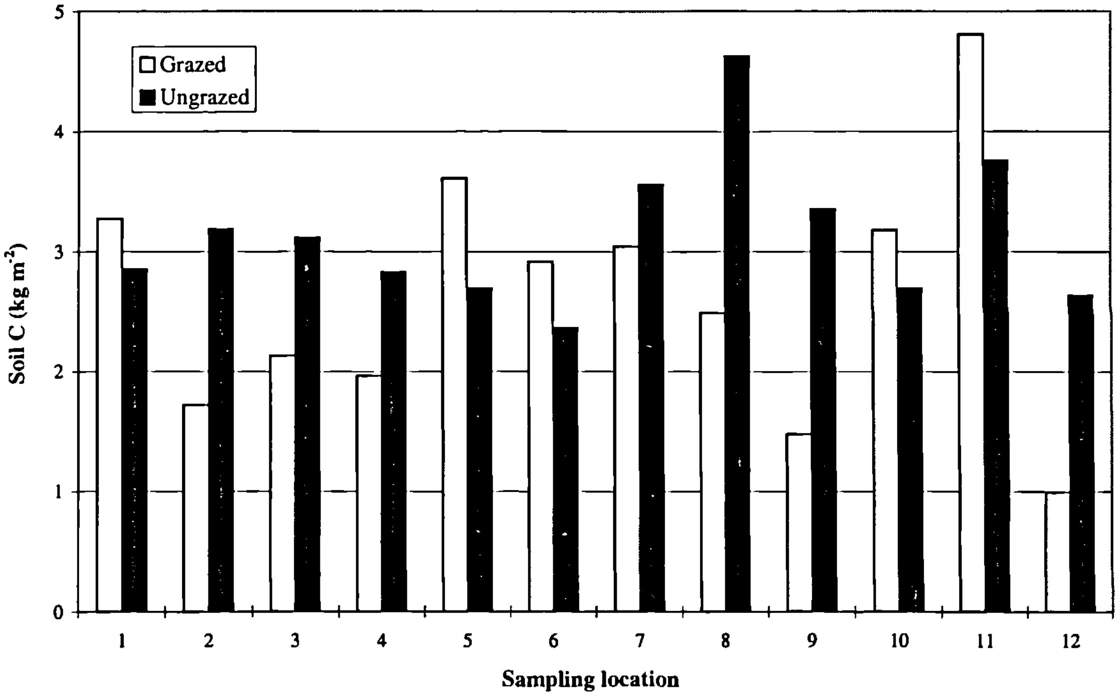

Figure 4.3 Little Dun Fell. Soil C under grazed and ungrazed treatments

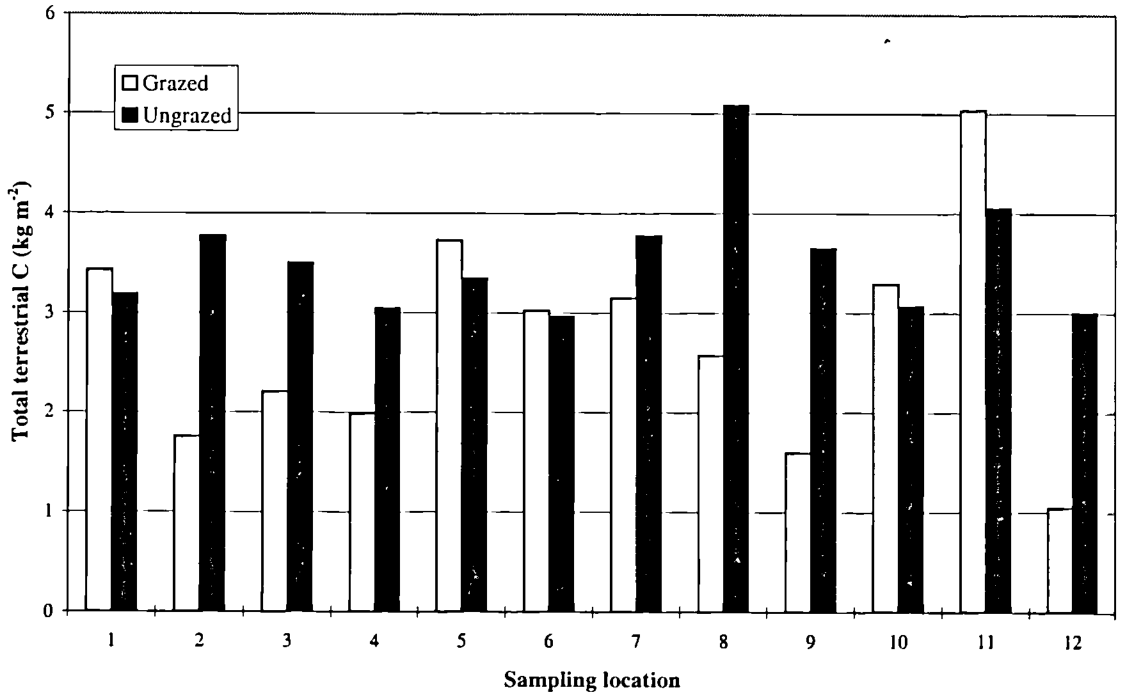

Figure 4.4 Little Dun Fell. Total terrestrial C under grazed and ungrazed treatments

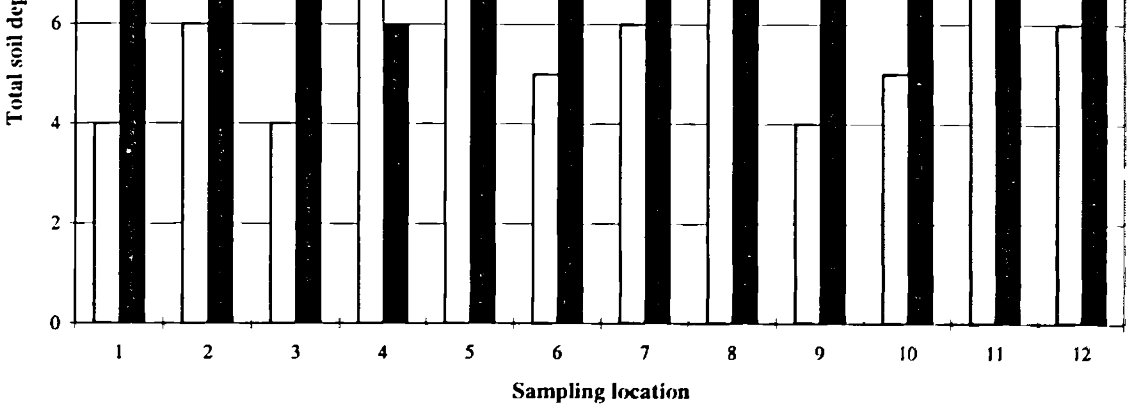

Figure 4.5 Little Dun Fell. Soil depth under grazed and ungrazed treatments

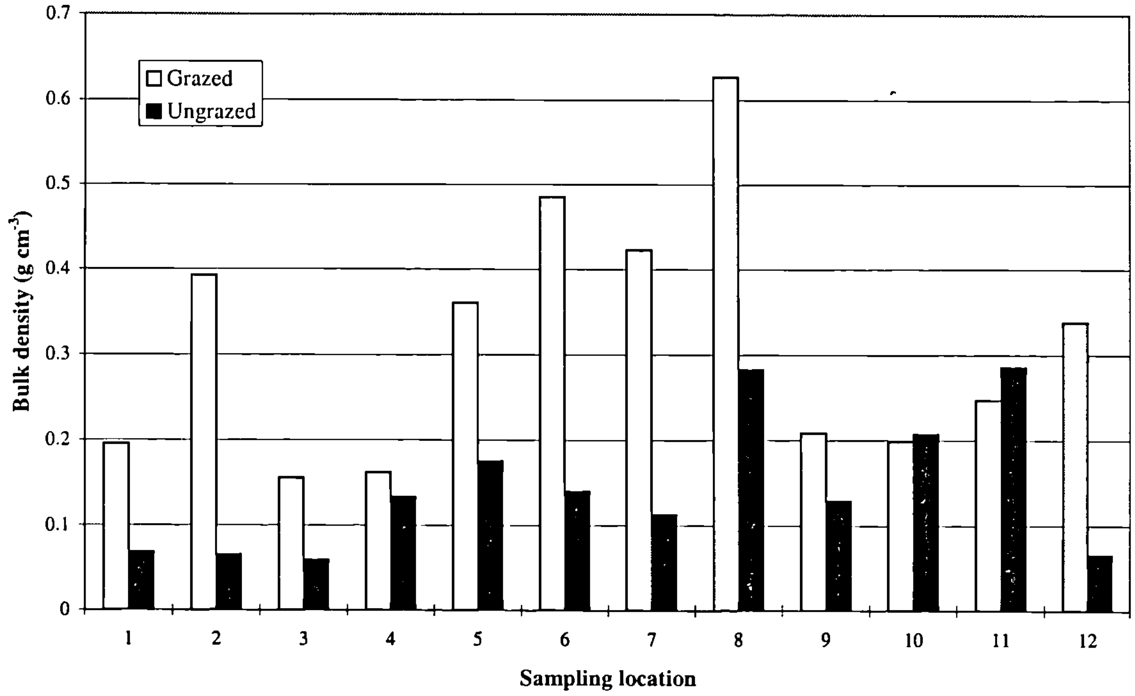

Figure 4.6 Little Dun Fell. Soil bulk density under grazed and ungrazed treatments

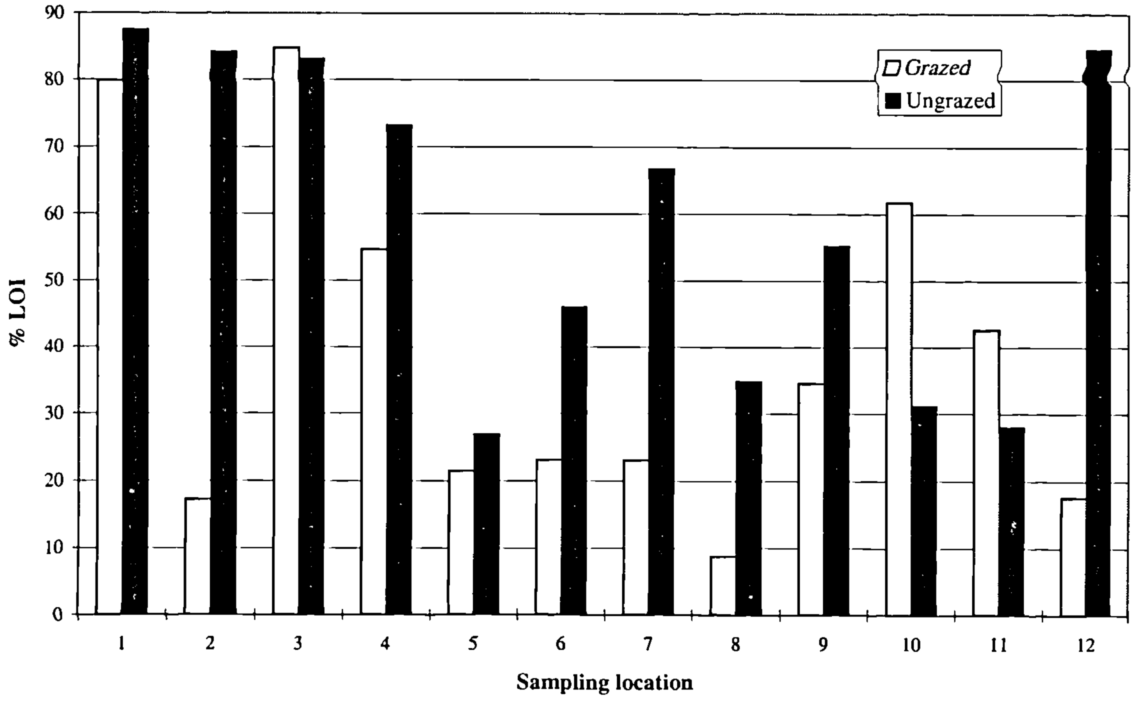

Figure 4.7 Little Dun Fell. % LOI of soil under grazed and ungrazed treatments

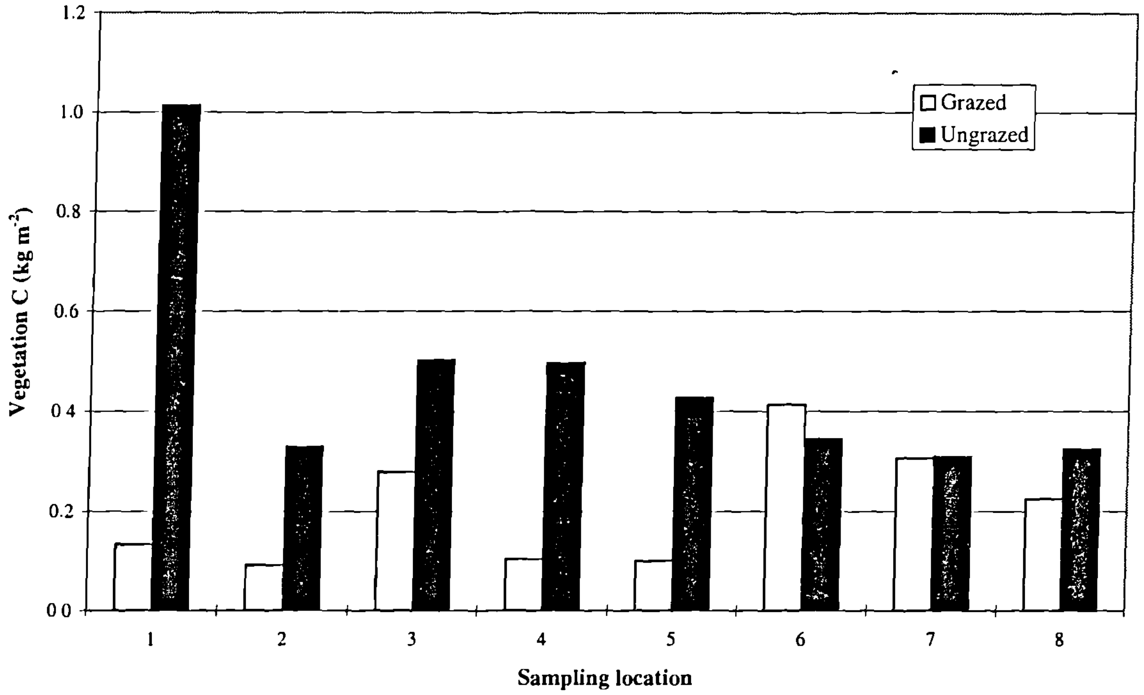

Figure 4.8 Knock Fell. Vegetation C under grazed and ungrazed treatments

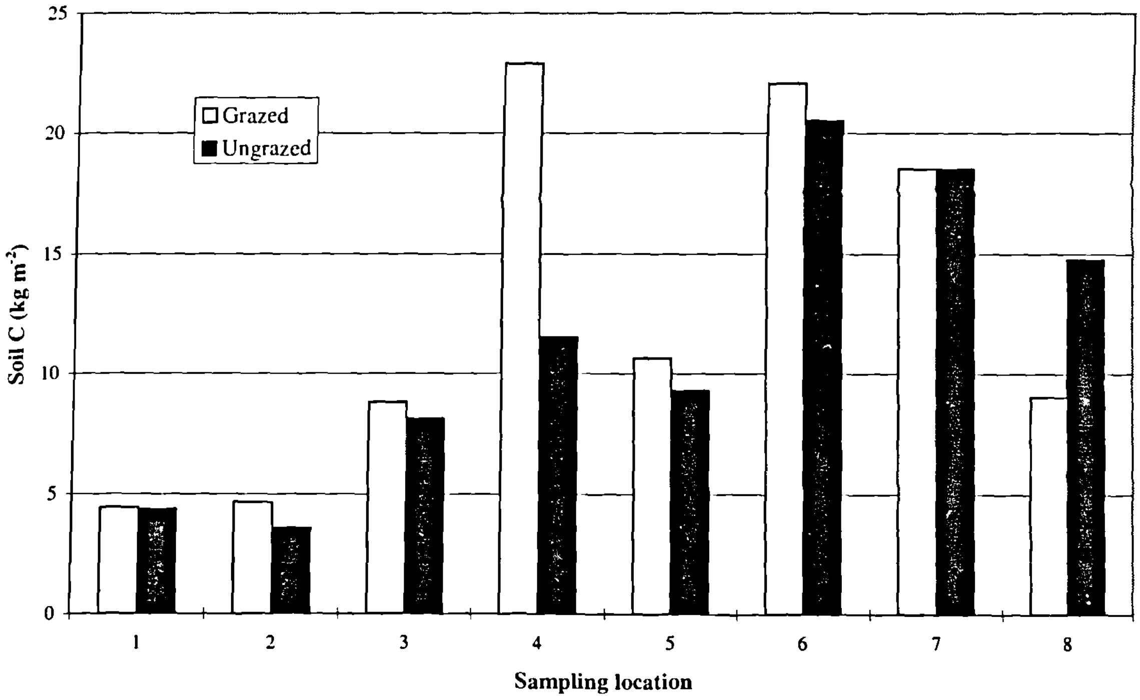

Figure 4.9 Knock Fell. Soil C under grazed and ungrazed treatments

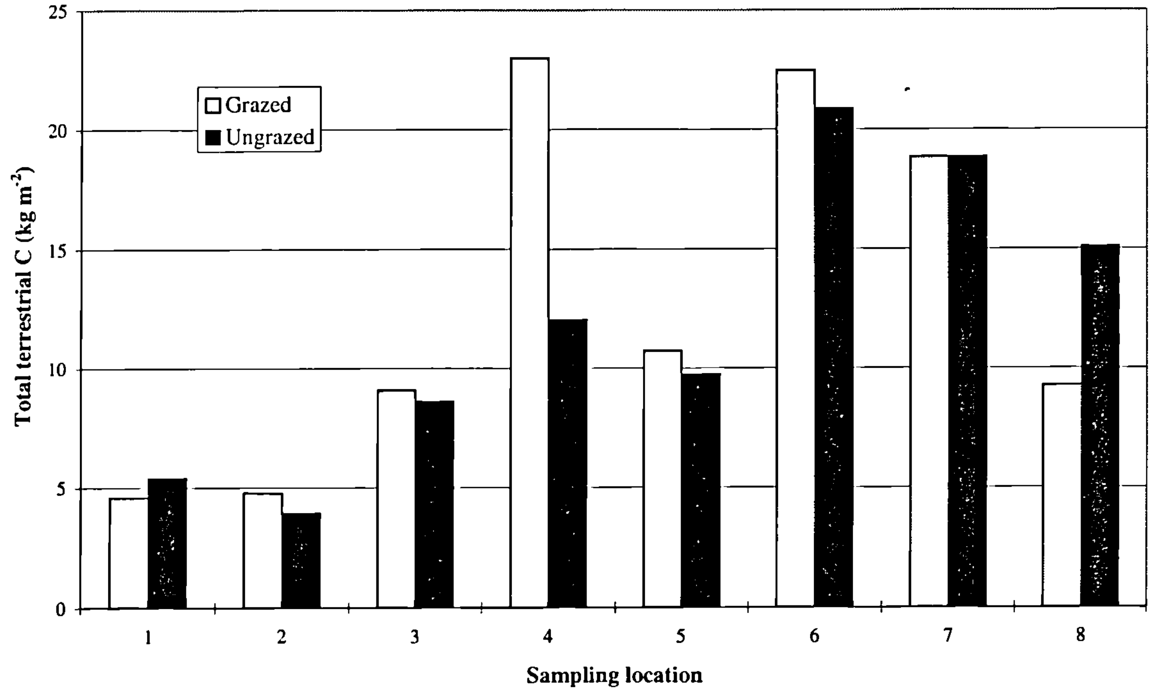

Figure 4.10 Knock Fell. Total terrestrial C under grazed and ungrazed treatments

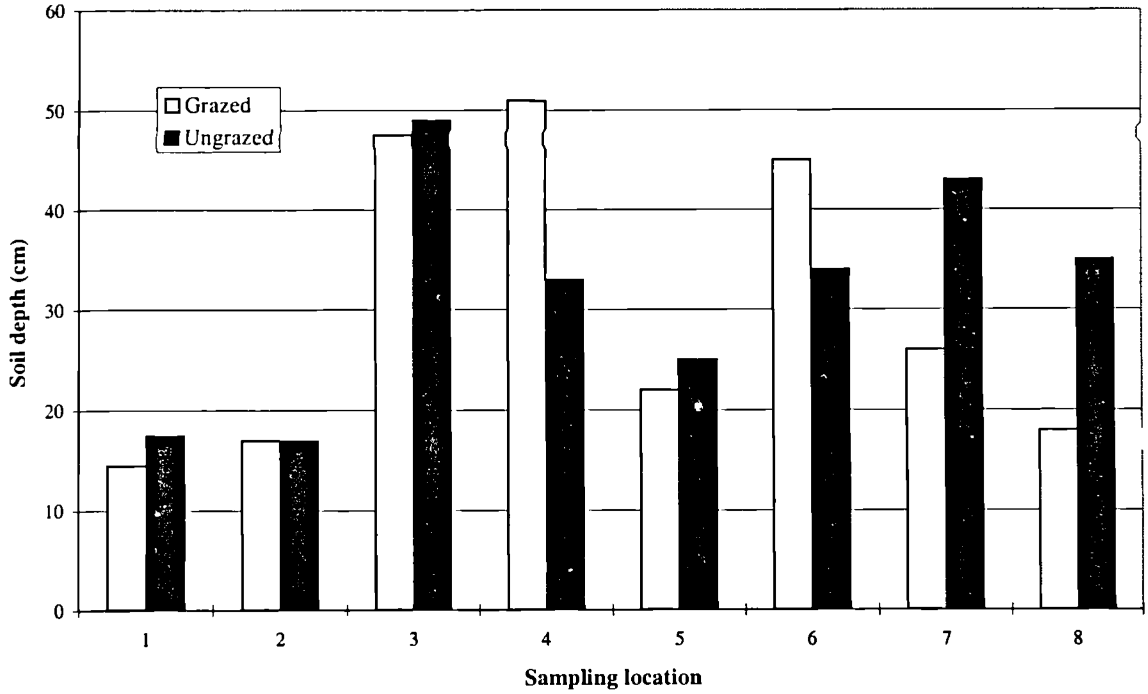

Figure 4.11 Knock Fell. Soil depth under grazed and ungrazed treatments

the level of significance for the differences between treatments. At both sites vegetation C was significantly higher when not grazed, although there were no significant treatment effects on the soil C store at either site. Total terrestrial C storage was significantly higher at Little Dun Fell at the 95% level.

Table 4.7 gives the mean values of rock cover, soil depth, soil bulk density and %LOI for each site, where available. The results of the *t*-tests are shown, indicating whether treatment differences were significant.

Table 4.6 Mean values of vegetation, soil and total terrestrial C stored in grazed and ungrazed areas at Little Dun Fell and Knock Fell. Significance of differences was determined using paired *t*-tests (N.S. not significant, * $p<0.05$, ** $p<0.01$, *** $p<0.001$).

	Treatment	Little Dun Fell	Knock Fell
Vegetation C	Grazed	0.10	0.21
(kg m^{2})	Ungrazed	0.40	0.47
	Significance	***	*
Soil C	Grazed	2.64	12.63
(kg m^{-2})	Ungrazed	3.14	11.34
	Significance	N.S.	N.S.
Total C	Grazed	2.74	12.84
(kg m^{-2})	Ungrazed	3.54	11.81
	Significance	*	N.S.

Table 4.7 Mean values of other variables measured under grazed and ungrazed treatments at Little Dun Fell and Knock Fell. Significance of differences was determined using paired *t*-tests (N.S. not significant, * p<0.05, ** p<0.01, *** p<0.001).

	Treatment	Little Dun Fell	Knock Fell
% rock cover	Grazed	11.7	0
	Ungrazed	0.7	0
	Significance	*	N.S.
Soil depth (cm)	Grazed	6.3	30.1
	Ungrazed	9.1	31.7
	Significance	**	N.S.
Bulk density (g cm^{-3})	Grazed	0.32	-
	Ungrazed	0.14	-
	Significance	**	
% LOI	Grazed	39.1	-
	Ungrazed	58.5	-
	Significance	*	-

4.3.2 Study 2: Impact of burning and sheep grazing on C accumulation in blanket bog

Figure 4.12 shows the profiles of cumulative C content for each peat core investigated. The figure is divided into four charts with each block being represented on a separate chart. The cores generally had similar profiles of cumulative C showing an approximately linear increase with depth.

The profiles of SCP concentration are shown in Figure 4.13 and again illustrate the cores retrieved from each individual block on separate graphs. The values for SCPs represent the actual number of particles counted from 8 transects across the Petri-dish samples, which had been derived from air-dried peat samples weighing 0.2 g. These results show that SCP levels were less than 50 at the base of the core, rising to maximum values of greater than 1000. The 'take-off' in SCP concentration was more distinct in some profiles than others and varied considerably at the depth it occurred. However, in the profiles where the 'take-off' was obvious, it coincided with the depth where the SCP count exceeded 100. Therefore, for

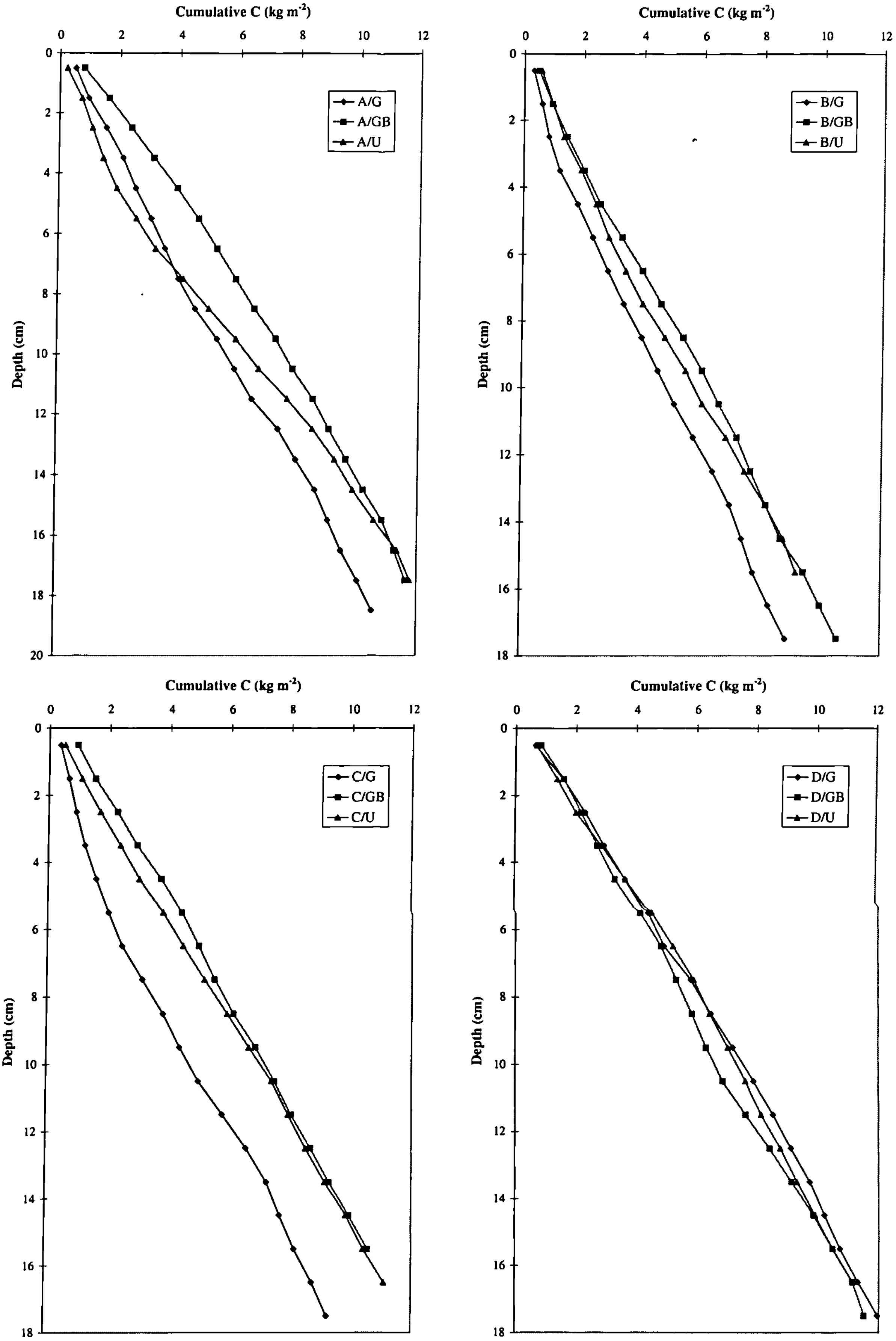

Figure 4.12 Profiles of cumulative C with depth for each treatment and each block. Blocks are represented as A, B, C and D. The treatments are: G = grazed and not burnt; GB = burnt every 10 years and grazed; U = ungrazed and not burnt.

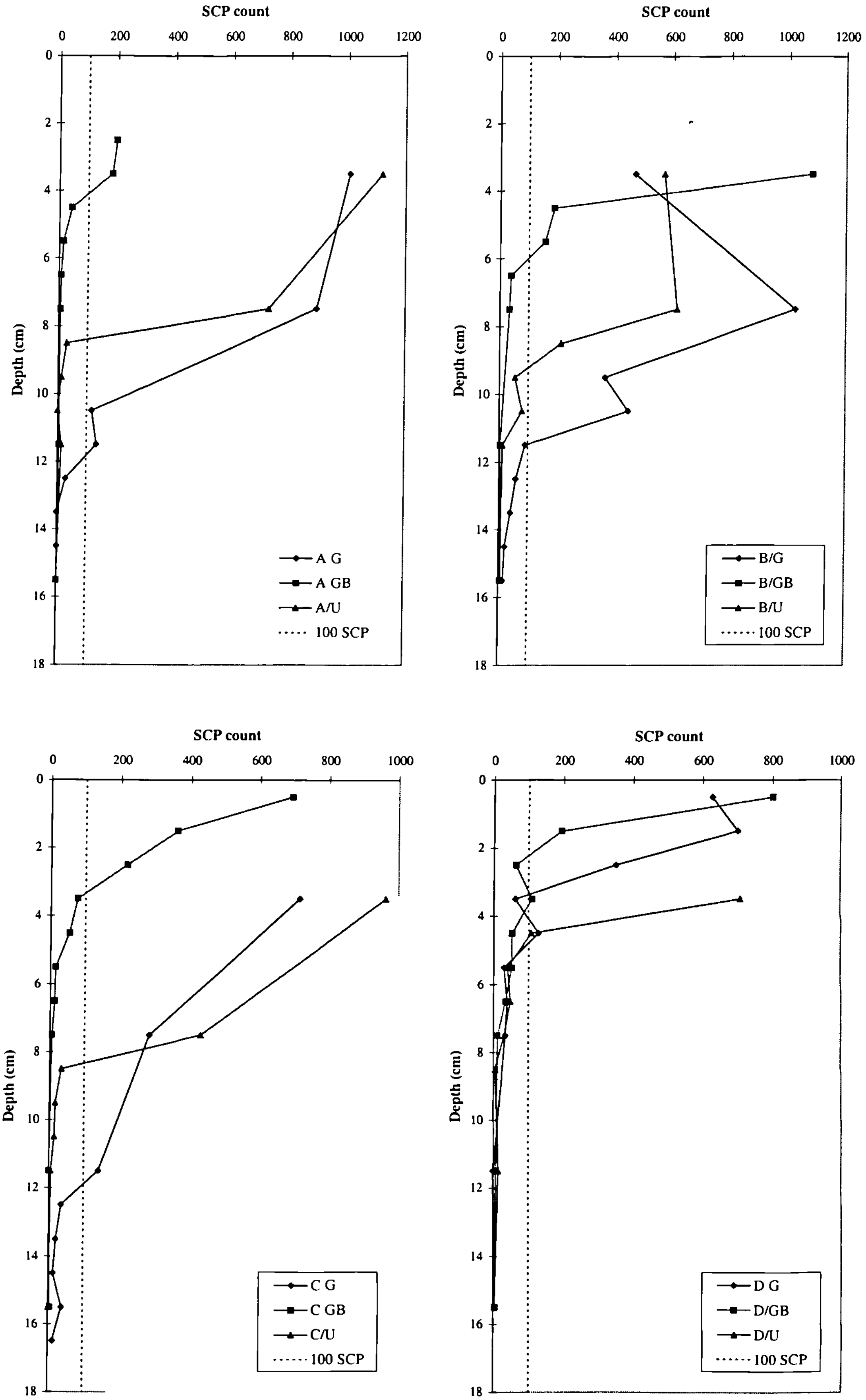

Figure 4.13 Profiles of SCP concentrations for each treatment and each block. Blocks are represented as A, B, C and D. The treatments are: G = grazed and not burnt; GB = burnt every 10 years and grazed; U = ungrazed and not burnt.

the samples where the 'take-off' was not as clear it was defined as the level where the count exceeded 100 SCPs. The 'take-off' was generally at a greater depth for the unburnt treatments (G and U) than the treatment being burnt every 10 years (GB). In three of the blocks the 'take-offs' centre around a depth of 8-12 cm, though in block D, all treatments showed 'take-offs' nearer the surface.

Table 4.8 shows estimated depths of the SCP 'take-offs' derived from Figure 4.13, and the amount of C contained in the peat above this layer, for each treatment in each block. Figure 4.14 displays the mean values of C stored above the SCP 'take-off' for each treatment.

Table 4.8 Depths of SCP 'take-offs' and C contained in the peat above the 'take-offs' for each profile

Block	Treatment	Depth of the SCP 'take-off' (cm)	C in peat above the SCP 'take-off' (kg m^{-2})
A	G	12	6.5
A	GB	4	3.2
A	U	8	4.2
B	G	11	5.1
B	GB	6	3.3
B	U	9	4.7
C	G	12	5.8
C	GB	3	2.3
C	U	8	5.2
D	G	5	3.7
D	GB	4	2.7
D	U	5	3.7

4.4 Discussion

4.4.1 Study 1: Impact of sheep grazing on the terrestrial C storage of two grassland sites

In a review of research undertaken at Moor House, Smith and Forrest (1978) described studies showing that the productivity and above-ground biomass of different vegetation communities were strongly influenced by the presence, absence or intensity of grazing. The effects were probably due to the direct removal of vegetation by these animals, although, may partly reflect the strong influences sheep have on the species composition of high altitude grasslands (Rawes, 1981). However, very little is known about the impact of grazing on the decomposition of soil organic matter and, therefore, the impact of grazing on soil C storage is

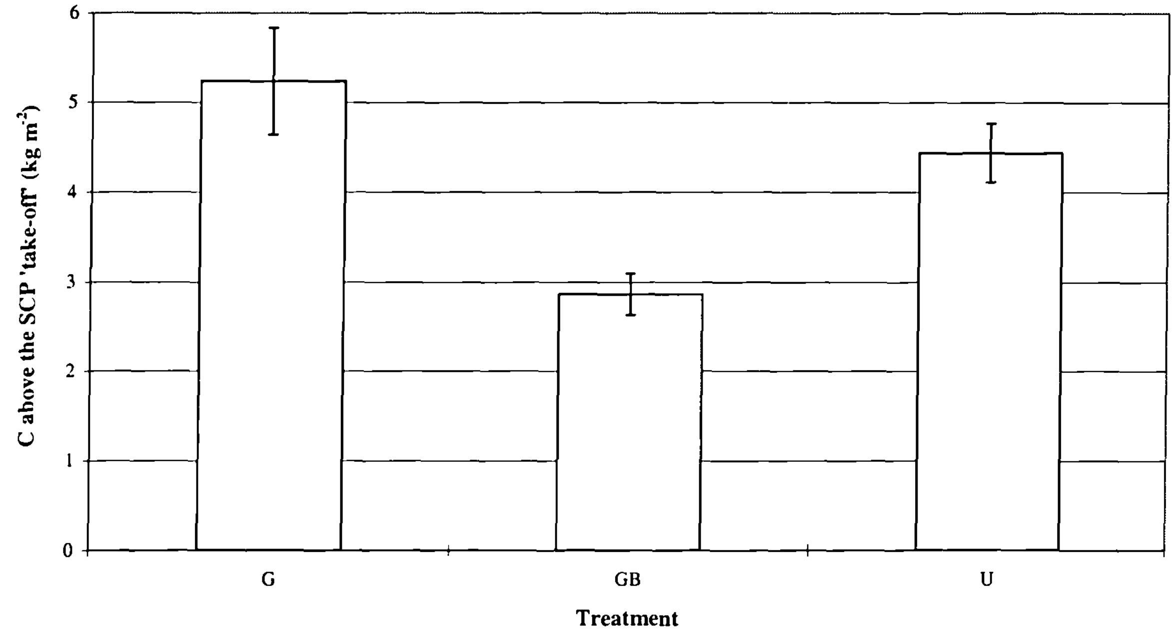

Figure 4.14 Mean values of C above the SCP 'take-off' layer in peat profiles under different treatments. The only significant difference was between treatments G and GB ($p<0.01$). Error bars indicate standard error. The treatments are: G = grazed and not burnt; GB = burnt every 10 years and grazed; U = ungrazed and not burnt.

poorly understood. Since it has been shown that soil contains far greater amounts of C than vegetation at this site (Chapters 2 and 3), management activities which affect soil C storage also need to be investigated in order to understand the impact of these activities on total terrestrial C storage.

The exclosure method used to investigate the impact of grazing on terrestrial C storage at the two grassland sites has been widely used at Moor House and other locations. Rather than indicating the influence of grazing by sheep, exclosures show the effects of excluding sheep from areas which have normally already been grazed. This is directly relevant to the aim of the present investigation which was to determine whether cessation or reduction in sheep grazing in these upland grasslands would increase terrestrial C storage.

However, there were a number of problems encountered in the present investigation which may have affected the accuracy of the results obtained: Although the exclosures were large (*ca.* 30 m x 30 m) they may have created micro-climate effects which were uneven inside and outside the exclosures, therefore, the observed differences may not have been entirely due to the presence or absence of grazing. For example, the sheltering provided by the fences causes drifting snow to accumulate. However, this is not likely to have affected the results because samples for both treatments were taken from similar distances away from the fence, and from most sides; any micro-climate effect was likely to have occurred under both treatments.

Problems in sampling vegetation and soils have been discussed previously (Chapters 2 and 3) and are clearly of relevance here. However, because identical sampling techniques were applied inside and outside the exclosures, the error in these estimates was unlikely to have been biased.

4.4.1.1 Little Dun Fell

Significantly greater values of vegetation C were measured within the Little Dun Fell exclosure (*i.e.* ungrazed) than outside ($p<0.001$), with mean C amounts of 0.4 and 0.1 kg C m^{-2} for the ungrazed and grazed treatments respectively. Besides the direct removal of vegetation by sheep grazing, there were differences in the species composition under the different grazing conditions which may have lead to the increased vegetation C within the exclosure; it was clear that *Carex bigelowii* had become an important component of the ungrazed vegetation, although was mainly absent outside the exclosure (where *Festuca ovina* dominated). This parallels the findings of Rawes (1981).

Soil C density in the grazed and ungrazed areas was not significantly different at the Little Dun Fell site, although the mean soil C content was slightly higher in the ungrazed samples. However, there were significant differences in soil characteristics: Soil depth was significantly greater within the exclosure ($p<0.01$); soil C concentration (%C) was higher where grazing had been prevented ($p<0.05$); and soil bulk density was higher under grazed conditions ($p<0.01$).

Although the total depth of the soil was higher under ungrazed conditions, the soil had a lower bulk density and a higher %C, resulting in a similar soil C density under both treatments. The lower bulk density may have been partly due to the absence of the trampling effect by sheep, though there was probably a greater amount of litter produced in the ungrazed treatment. The inverse relationship between organic matter concentration and bulk density (Figure 4.15) has been identified previously with Harrison and Bocock (1981) providing a method to determine soil bulk density from %LOI based on this relationship.

It is possible that *Carex bigelowii* decomposes more slowly than *Festuca ovina*, retaining structural strength longer. This may have resulted in greater amounts of standing dead vegetation, a higher amount of C in the litter layer, and a soil with lower density but higher %C. Some support for this hypothesis was provided by Cornelissen (1996) who found that *Festuca ovina* decomposed more rapidly than *Carex flacca,* a sedge species not dissimilar to *Carex bigelowii.* The fact that the soil C content was not significantly higher in the ungrazed area could be due to the low rate at which dead vegetation was incorporated into the soil.

The total terrestrial C stored under the different treatments at Little Dun Fell was significantly greater within the exclosures ($p<0.05$).

4.4.1.2 Knock Fell

A significant difference ($p<0.05$) was found between grazed and ungrazed vegetation C at the Knock Fell exclosure, with greater vegetation C in the ungrazed vegetation. However, the vegetation C was not as consistently greater in the ungrazed areas as it had been at Little Dun Fell. For example, at sampling location 6, the total C in vegetation in the grazed area was higher than the ungrazed (Figure 4.8). This is believed to be predominantly due to small-scale differences in the species composition of the vegetation, and in particular, to the palatability of the species to grazing. High vegetation C density was found at grazed sampling points 3, 6, 7 and 8, all of which contained large amounts of *Juncus squarrosus*, a species which is far less palatable to sheep than the Agrosto-Festucetum species. Therefore, where the vegetation was mainly composed of the *Juncus squarrosus*, the impact of excluding sheep has caused little effect, at least in terms of the C held in the vegetation.

There was no significant difference in soil C between the grazed and ungrazed treatments at the Knock Fell site. There was a large natural variability in the soil over short distances with, for example, soil depth ranging from 15 to over 50 cm. This made

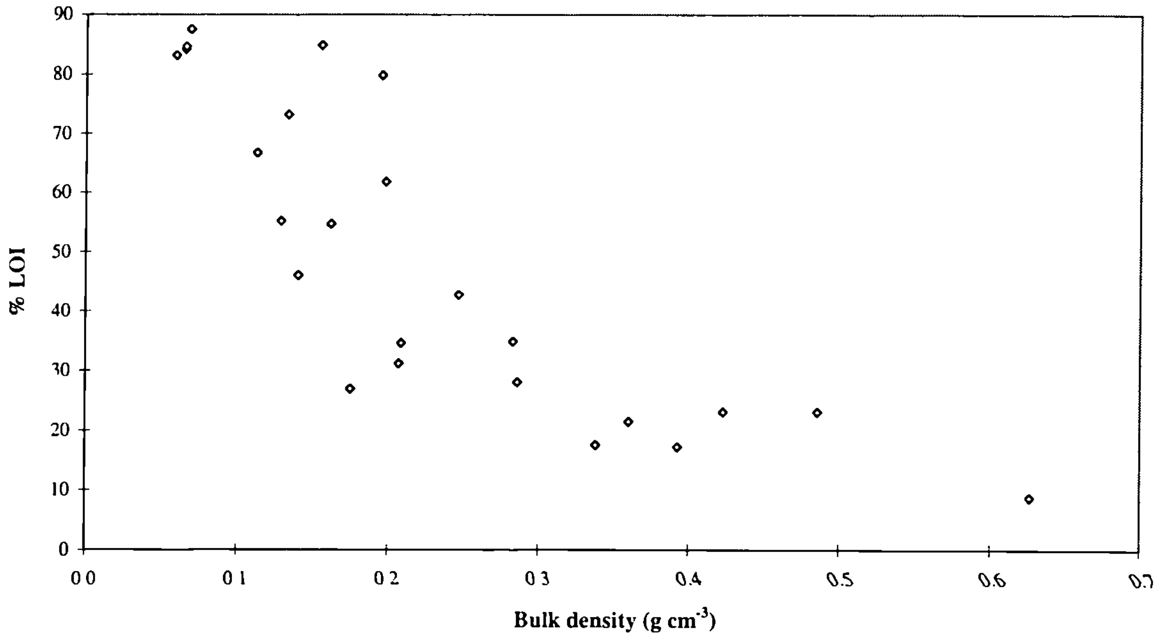

Figure 4.15 Little Dun Fell. Plot of soil %LOI against bulk density under grazed and ungrazed treatments

it difficult to determine whether the amount of soil C changed due to sheep exclusion; it is not possible to say whether observed differences between treatments at the sampling points were a result of the treatment or not, because the variability of the soil was so high, despite sampling areas of similar characteristics (*e.g.* slope). However, the results indicate that after 40 years of excluding sheep, the amount of soil C at this site has not been greatly affected.

No significant difference was found in the depth of soil between treatments, again probably due to the high natural variability of the soil, although there has been the development of a thicker litter layer within the exclosure (personal observation; Rawes, 1981).

Since the vegetation C was marginally significantly different and soil C not different between treatments, and because total C was dominated by the soil component, there was no significant difference in the total terrestrial C stored at this site.

4.4.1.3 Impact of sheep grazing on terrestrial C storage in upland grasslands

The exclusion of sheep from these high-altitude grasslands significantly increased the vegetation C density in both the Little Dun Fell and Knock Fell sites. This result was not surprising as the sheep clearly remove vegetation by grazing and other studies have shown that grazing can reduce above-ground biomass (*e.g.* Rawes and Welch, 1969; Smith and Forrest, 1978). However, other factors appear to have also been involved, such as the palatability of individual plant species, with different species responding differently to the absence of grazing (Rawes, 1981).

The amount of C stored in the soils appears not to have changed markedly in either exclosure. This could be that grazing, although affecting the inputs and outputs of C, has not affected the **net** amount of C stored. Howard and Howard (1976), studying part of the Knock Fell exclosure, found no evidence to suggest C was accumulating when grazing was removed, contrary to original suggestions made by Welch and Rawes (1964). Alternatively, the methods employed to analyse the amount of C stored may not have been sufficiently sensitive to determine small grazing effects, given the high natural soil variability.

There are surprisingly few published studies on the impact of grazing on soil C storage although, in a similar study using two sheep exclosures situated on an upland area in New Zealand, Basher and Lynn (1996) found few significant differences between grazed and ungrazed soils. They found total C was greater within one exclosure when compared to the adjacent grazed land, although the opposite was found at their other site. As observed at Little Dun Fell, Basher and Lynn (1996) found a lower soil bulk density and greater soil depth within one of their exclosures.

Consequently, although the prevention of sheep grazing at the Moor House exclosures resulted in differences in certain soil properties, the total amount of terrestrial C stored was only significantly higher at the Little Dun Fell site. Although the changes in total C stored were small, especially over a 40 year period, grazing appears to modify the input, output and C storage in these grasslands. However, these investigations at Moor House, and other sites, suggest that altering grazing regimes does not appear to offer a short or even medium-term management option for greatly increasing terrestrial C stores.

4.4.2 Study 2: Impact of burning and sheep grazing on C accumulation in blanket bog

The impact of sheep grazing and moorland burning on C accumulation in blanket bog has been investigated by utilising an existing long-term experiment which was based on a randomised block design. This experimental design reduces the confounding influence of natural variations in site characteristics, minimising any bias in observations caused by non-treatment effects.

Although the Hard Hill site had a more robust experimental design than the grassland sites, there were several problems encountered in the investigation which may have introduced error into the results. Firstly, during coring it was noticeable that a small amount of compression occurred when inserting the plastic cores. This compaction will have been exacerbated during extraction and slicing of the cores. Although this will not have affected the estimates of C contained above the SCP 'take-off', it will have affected the cumulative C profiles of each coring location. Furthermore, to allow the depth of the SCP 'take-off' to be determined to within 1 cm, the cores were sectioned at 1 cm intervals by slicing with a sharp knife. However, sectioning cores is, in itself, not very accurate because the slices can only be cut to an accuracy of *ca.* 1 mm. Since the thickness of the samples was 10 mm, this represented an error of around 10%, which is incorporated into the estimate of C content for each slice; since the cumulative C profiles were based on these values, the trends apparent in Figure 4.12 partly reflect the accuracy which individual samples were sliced.

Consequently, no attempt has been made to investigate the effects of the treatments on the trends of cumulative C in the profiles, although it was noticeable while sectioning that the cores removed from the burnt plots (GB) appeared to have a higher bulk density. Accurate estimates of bulk density were not made directly for the cores in this study, but a lower moisture content in the top part of the cores from the burnt plots was noticeable (Figure 4.16), and a strong correlation, and inverse relationship, had been found between bulk density and moisture content (%) from blanket bog samples (Figure 4.17; Chapter 3). Therefore, the lower moisture contents in the surface of the cores subjected to the burning

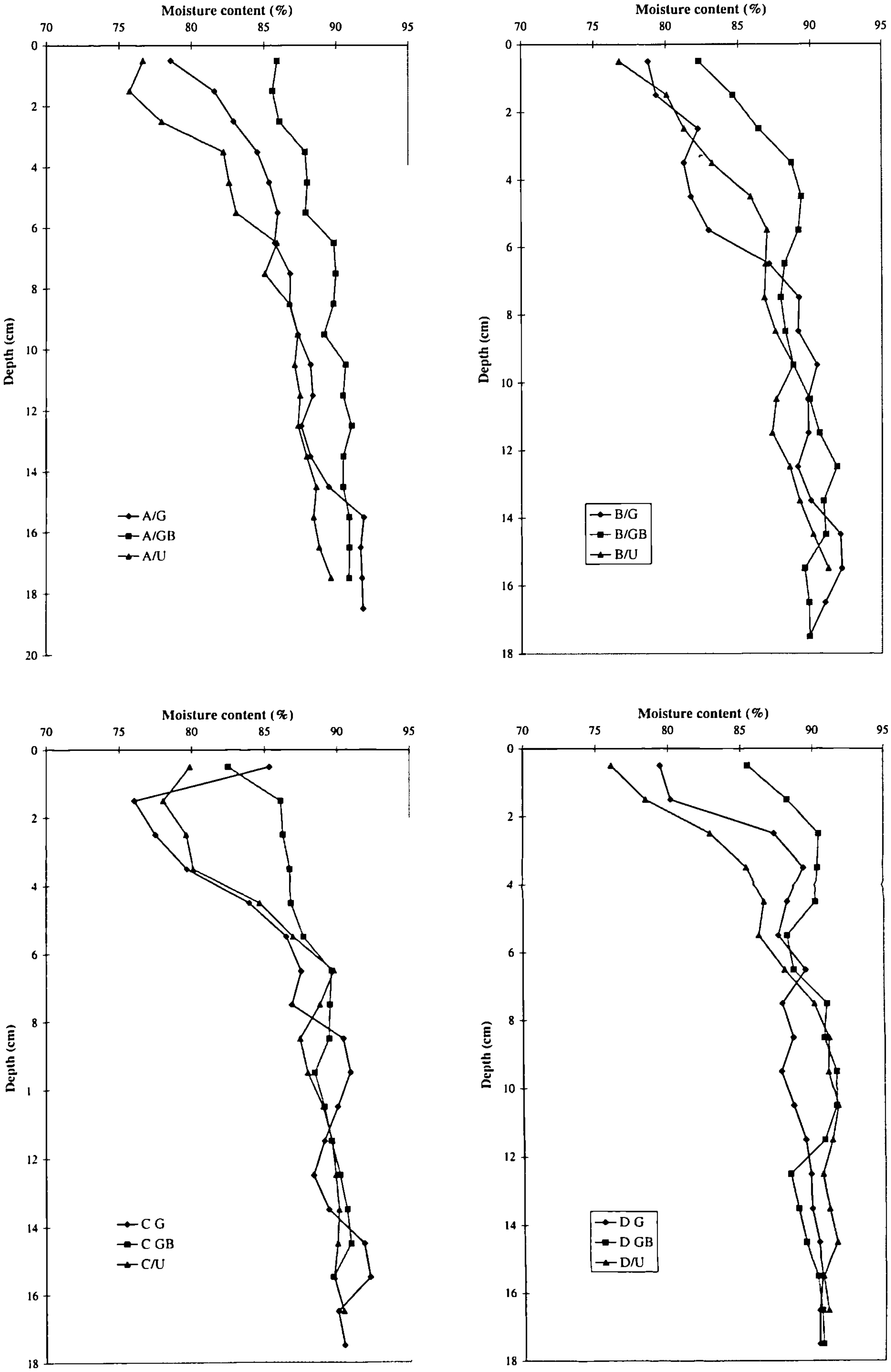

Figure 4.16 Profiles of moisture content with depth for each treatment and each block. Blocks are represented as A, B, C and D. The treatments are: G = grazed and not burnt; GB = burnt every 10 years and grazed; U = ungrazed and not burnt.

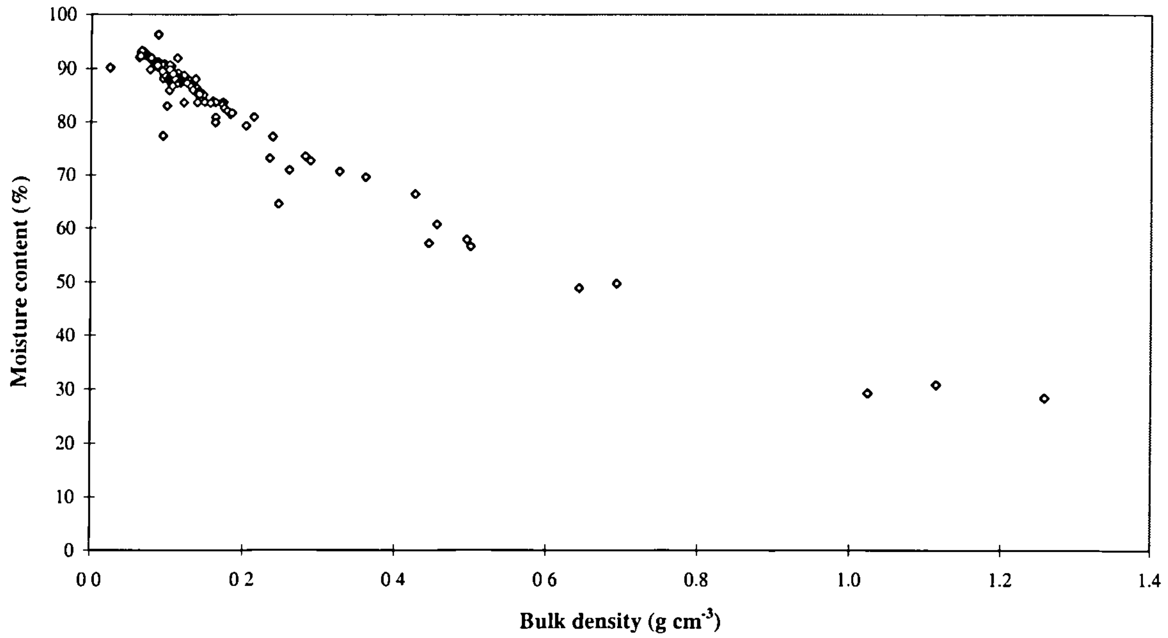

Figure 4.17 Plot of moisture content against bulk density for blanket bog samples collected for determining soil C density in Chapter 3

treatment imply that the treatment probably caused an increased bulk density in the surface layers of peat. Rowell (1988) states that burning can permanently reduce the moisture content and cause crusting of the surface layers of peat.

Rose (1990) provided a methodology for the extraction of SCPs from lake sediments which was shown to retrieve greater numbers of SCPs than the method of Renberg and Wik (1985). The latter method is most similar to the method used to extract SCPs in the present study, and therefore, a modification of Rose's (1990) method for peats (omitting the stages for the removal of minerogenic material which are unnecessary for peats), as performed by Gedye (1998), would have allowed a more accurate quantification of the SCP concentration in the samples. However, this method was more time consuming which may have reduced the number of samples able to be investigated.

There is an element of subjectivity involved in defining the SCP 'take-off' since the trend of SCP concentration in profiles rather than actual concentrations are used for chronological reference (Wik and Renberg, 1996). There are several factors which determine the concentration of SCPs (and other atmospheric pollutants) in peat: Deposition rate of SCPs; rate of peat accumulation and; compaction of peat (Clymo *et al.*, 1990). In the present study, the SCP 'take-off' was believed to reflect the increasing deposition of SCPs caused by industrialisation. Although variations in compaction will have influenced the profiles of SCPs in the cores, most of the 'take-offs' were located in the middle part of the cores where compaction during coring and extraction would have been least. Since the basal layer of the peat in the cores had SCP counts of about 20-30 and the 'take-off' tended to be located where the SCP counts exceeded 100, compaction equivalent to squashing 4 cm of peat into 1 cm would have been required to cause an apparent SCP 'take-off'. The compaction that was observed during coring and core extraction was certainly considered lower than this and was confined mainly to the surface layers of peat.

Similarly, the rate of peat accumulation would have needed to have been reduced by at least a quarter in order to produce the concentrations of SCPs which could have been mistaken for an industrial 'take-off'. However, there was no evidence in the stratigraphy to suggest that the accumulation rate of peat had changed rapidly at the same time as the SCP 'take-offs', such as distinct layers of increased humification or changes in plant species composition of the peat (although a detailed investigation of the humification and macrofossil composition was not undertaken). Furthermore, the experimental plots were situated on uniform slopes and some distance away from areas of peat erosion which could have been responsible for a rapid reduction in peat accumulation (through alteration of hydrological conditions). The SCP concentrations below the 'take-off' in each peat core were also relatively constant, with counts of typically 0-30. In several cores, concentrations within this range existed over a large depth interval (*e.g.* from 6-10 cm and the base of the core). The rate of peat accumulation over this depth is likely to have fluctuated with variations in the factors affecting peat accumulation (*e.g.* small changes in climate) but, the relatively constant concentrations of SCPs suggest that changes in peat accumulation did not have a major affect on the SCP concentrations.

Differences in the amount of C contained above the SCP 'take-off' could have been produced before the experimental plots were established as a result of natural variability in peat accumulation. That the sites have historically accumulated peat at different rates (at least in terms of height increment) is evident since measurement of the peat depth at each coring location showed total peat depth ranged from 1 m to 2 m (Figure 4.18). However, although peat depth varied considerably between different blocks, there were predominantly only small differences in total peat depth within the same block under the different treatment plots. Therefore, the past peat accumulation within separate blocks seems to have been even across the different plots. Furthermore, there was no correlation between the depth of the SCP 'take-off' and the total depth of peat for each sample, which would have suggested natural variations in sampling locations were responsible for the different amounts of C contained above the SCP 'take-offs'.

In this study, therefore, the SCP 'take-off' recorded in each profile was believed to represent a take-off in the atmospheric deposition of SCPs, associated with an increased industrial emission of these particles. The precise date of this 'take-off' is unknown since it varies between regions. For example, Scottish lake sediments dated using ^{210}Pb suggest the SCP record began in the early nineteenth century and that a major rise in SCP deposition occurred after the 1940's (Rose, 1990). However, from sites in south Cumbria, Gedye (1998) found 'take-offs' in SCPs to be much earlier, *ca.* 1800 AD, possibly indicating early and localised industrialisation. However, although the actual age of this take-off was unclear, it is reasonable to assume it was synchronous across the different treatment plots due to their close proximity. However, an independent method of dating the SCP 'take-off' from a number of the cores using, for example, ^{210}Pb (Oldfield *et al.*, 1995; Appleby *et al.*, 1997) may have indicated whether the SCP 'take-offs' were synchronous and caused by increased industrial SCP deposition, though was not available for the present study.

4.4.2.1 Impact of sheep grazing on C accumulation in blanket bog

The mean amount of C above the SCP 'take-off' in the grazed treatment was 5.2 ± 0.6 kg m^{-2} (SE), whereas the ungrazed treatment contained 4.4 ± 0.3 kg m^{-2}. Although the grazed treatment contained a greater amount of C, the results of the ANOVA showed this difference

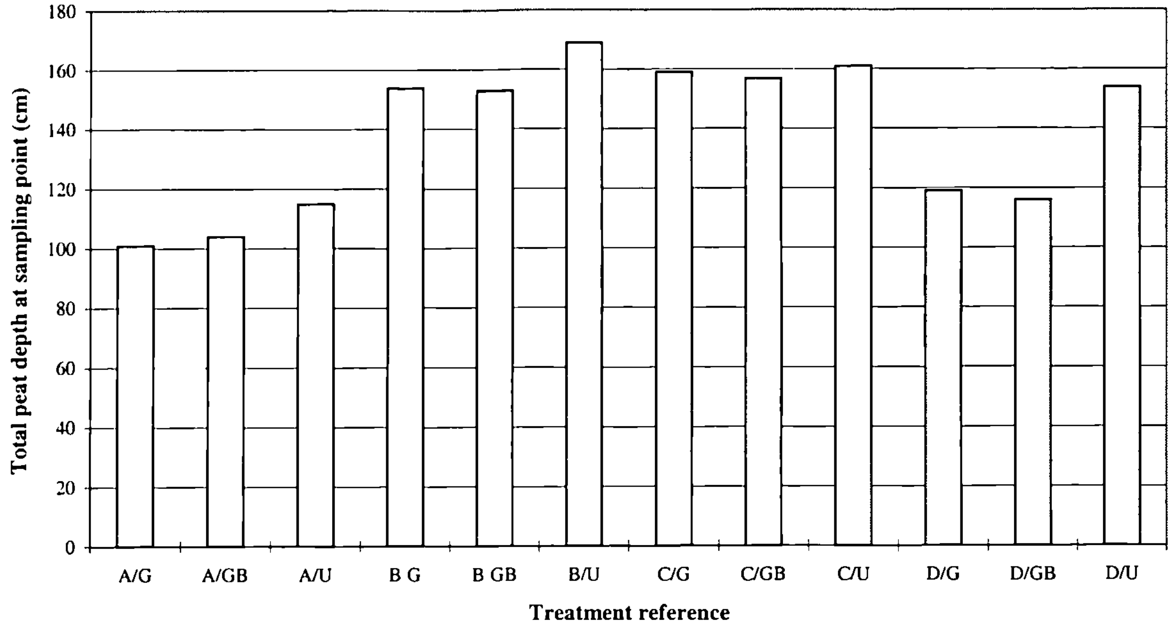

Figure 4.18 Total depth of peat at each sampling point at the Hard Hill experimental plots. Blocks are represented as A, B, C and D. The treatments are: G = grazed and not burnt; GB = burnt every 10 years and grazed; U = ungrazed and not burnt.

was not significant. Consequently, after over 30 years of different management, there was no difference in the C accumulated under the separate treatments.

The results are not surprising since the density of sheep at this site was very low (0.02-0.2 sheep ha^{-1}; Smith and Forrest, 1978) and no significant impact of sheep grazing had been found on the above-ground biomass of blanket bog vegetation at Moor House under this stocking density (Smith and Forrest, 1978). However, Taylor and Marks (1971) found that grazing did reduce the above-ground biomass of *Rubus chamaemorus*, though this species is a minor component of blanket bog vegetation (Forrest and Smith, 1975) and decomposes rapidly (Latter *et al.*, 1998) and, consequently, does not contribute very much to peat accumulation.

4.4.2.2 Impact of burning on C accumulation in blanket bog

The impact of fire on C accumulation in peatlands is poorly understood (Gorham, 1991) yet fire occurs naturally (*e.g.* lightning strikes; Kuhry, 1994) accidentally (*e.g.* in recreational areas; see Phillips *et al.*, 1981) and as a management tool on peatlands (Hobbs, 1984). Furthermore, increased fires are expected with global warming (Overpeck *et al.*, 1990) which could provide a positive feedback to warming through increased release of the large amounts of C contained in peats.

The impact of rotational burning of blanket bog vegetation at Moor House has been investigated in this chapter to determine the effect of this management practice on C accumulation in peats; the impact on vegetation composition has been previously reported by Rawes and Hobbs (1979) and Hobbs (1984). The results of this investigation may be useful in assessing the impact of natural and accidental fires on C accumulation in peats, although there are important differences between these types of fire (*e.g.* heather burning is only allowed between October and April in England and Wales to prevent fires becoming uncontrollable; Rowell 1988), which may limit the application of the results presented in this chapter to other fire types.

That burning strongly affects the above-ground biomass on blanket bog is clear since immediately after a burn there is a reduced biomass on the peat surface, although burns rarely completely remove all former vegetation (*e.g.* woody material and charcoal frequently remain; Rhodes 1996). However, because the vegetation recovers after the burn, possibly at an increased rate due to the high availability of nutrients left in the ash (Allen, 1964), it is unclear from the literature whether burning affects the total C storage in the blanket bog. Furthermore, fire may reduce the C accumulation of blanket bog by directly burning the surface peat (*e.g.* Maltby *et al.*, 1990), or influence properties in the surface layers of peat which may alter rates of decomposition or hydrological characteristics of peat (*e.g.* bulk density; see above and Rowell, 1988). Hobbs (1984) reported that burning increased the dominance of *Eriophorum* sp. over *Calluna vulgaris* which may affect the total amount of C accumulating in vegetation in the plots and the quality of the litter produced.

The results of the investigation at Hard Hill showed that a significantly lower amount of C was contained above the SCP 'take-off' under the treatment which had been burnt every 10 years, than the unburnt treatment (burnt = 2.9 ± 0.2 kg m^{-2} and unburnt = 5.2 ± 0.6 kg m^{-2}) and the results of the ANOVA showed that this difference was significant ($p<0.01$). These results suggest that this management practice contributes to anthropogenic emissions of CO_2 through either i) decreasing the rate of peat accumulation, ii) stopping peat accumulation and/or iii) reducing C stores by burning existing surface peat. It is not possible from the results of the present investigation to establish which of these processes has been in operation at this site, because it is not possible to establish whether peat formed before the burning treatments began has subsequently been burnt. However, a reduction in peat accumulation since burning commenced must have at least taken place.

Allen (1964) conducted laboratory experiments to simulate heather burning and found that during the burn, 60.5-67.5% of the original vegetation C was transferred to the atmosphere, and therefore, it is probable that a large amount of the vegetation C stored in the burnt plots was rapidly transferred from the vegetation biomass to the atmosphere during the burn. However, other factors may have increased the rate of loss of C from the burnt plots. Decomposition may have been accelerated since the moisture content of the cores from the burnt plots was lower in the surface layers of the peats. Since water-logged conditions suppress decomposition rates, the reduced moisture content in the burnt treatment may have subjected a larger volume of peat to aerobic decomposition, which is considerably more rapid than decomposition under anaerobic (water-logged) conditions (Clymo, 1984). Therefore, increased decomposition of the surface layers of peat may have reduced the C stored above the SCP 'take-off'.

The main purpose of burning as a management tool is to improve the grazing value of moorland vegetation. Areas of vegetation recovering from burning will therefore have been favoured by sheep and more intensively grazed. Consequently, a large proportion of vegetation C may have been removed from the burnt (and unenclosed) plots by sheep, which would otherwise have formed new peat. Although there was no significant effect of sheep grazing on the blanket bog vegetation C on unburnt plots (see above), this is not directly comparable because recently burned areas would have had increased grazing value than unburnt areas and probably received a greater intensity of grazing; at the two grassland sites, where the vegetation is of higher grazing value, grazing was found to significantly reduce the vegetation C density (see Section 4.4.1).

Since the date of the industrial SCP 'take-off' was uncertain (based on other studies it may lie between *ca.* 1750 and *ca.* 1850 AD; Rose 1990; Gedye, 1998), actual rates of C accumulation under the different treatments could not be made. However, comparative estimates of the average rate at which C stored above the SCP 'take-off' across the different treatments could be made because the dates of the start of the experiment and application of treatments were known.

The experiment at Hard Hill was established in 1954 and initially the area covering all plots was burnt (Hobbs, 1984). The plots were then defined and different treatments applied. Therefore, although the experiment commenced in 1954, the burnt and unburnt treatments have only had a different management treatment from the time of the first 10 year burn on the burnt plots in 1965 (Hobbs, 1984). Therefore, the differences in C above the SCP 'take-off' represent the impact of burning over a period of 32 years (since samples for the present study were collected in 1997). The difference in the amount of C above the 'take-off' layer in the different treatments is equivalent to a reduced C sequestration in the burnt treatment of 74 g m^{-2} yr^{-1} (calculated by dividing the difference between the mean estimates of C stored above the SCP 'take-off', by the number of years the treatments had differed). This value represents C accumulation in peat which has not all passed into the catotelm (Ingram, 1978) and therefore cannot be compared to estimates of long-term C accumulation rates (*e.g.* Tolonen and Turunen, 1996). However, the results imply that if the burnt plots and other similarly managed blanket bogs in the UK had not been burnt over the last 32 years, then an average of 74 g C m^{-2} yr^{-1} would have been stored in the blanket bog, and not in the atmosphere contributing to higher atmospheric CO_2 levels.

Studies investigating the impact of burning as a management tool (or even as natural or accidental fire) on the C balance of peatlands are rare in the published literature. However, the observed reduced rates of C accumulation under the burning treatment are in agreement with the results of Kuhry (1994). From a study of peat profiles in the boreal peatlands of Canada he found that peat accumulation was significantly reduced under increasing frequencies of fire, implied from charcoal analysis. Maltby *et al.* (1990) described the complete loss of peat profiles following accidental fires on blanket bog in the North York Moors of England, though these fires penetrate deeply into the peat profiles, unlike the carefully controlled burns used for moorland management.

The investigation at Hard Hill implies that regular burning of the blanket bog at this site has resulted in a reduced C storage in the peat when compared to non-burnt areas, and that consequently, the abandonment of this management practice may provide an opportunity to increase terrestrial C storage in similar areas and, therefore, mitigate industrial emissions of CO_2.

5. Past terrestrial carbon storage at Moor House

5.1 Introduction

The size of global terrestrial C stores has varied considerably during the Earth's recent geological history. For example, it has been estimated that global terrestrial C stores are currently double the size of the stores present during the last glacial maximum (Bird *et al.*, 1994; Schimel, 1995). Since the terrestrial biosphere has an important role in regulating atmospheric CO_2 (Woodwell *et al.*, 1978), variations in past terrestrial C storage are likely to have influenced past atmospheric CO_2 concentration; we know from several sources that atmospheric CO_2 concentration varied from *ca.* 200 ppm to greater than 300 ppm during the Holocene (White *et al.*, 1994). Global climate has also varied considerably throughout the Quaternary (Nicholls *et al.*, 1996) and, since a strong correlation has been found between atmospheric CO_2 concentration and temperature during the last 160,000 years (Prentice and Fung, 1990), it seems likely that terrestrial C storage has strongly influenced past climate change (*e.g.* Klinger *et al.*, 1996).

The size of terrestrial C stores is a function of net primary productivity (NPP) and decomposition (Melillo *et al.*, 1996). These two processes are strongly influenced by climate and it is unclear whether their response to the anticipated global warming will result in terrestrial ecosystems storing more or less C (IPCC, 1995). There is particular concern about the impact of global warming on peatlands (Gorham, 1991) since they contain approximately the same amount of C as the atmosphere (Clymo, 1996; Clymo *et al.*, 1998). The implication is that, for example, a 5 % reduction in peatland C stores could potentially cause a 5 % increase in atmospheric CO_2.

Models have been developed to predict the response of terrestrial ecosystems to future climate change (Smith and Shugart, 1993), although there are considerable uncertainties in these predictions since it is difficult to verify the results of such models. However, the response of terrestrial ecosystems to future climate change can also be predicted by investigating how ecosystems have responded to past changes in climate. The results from such investigations can help predict how specific ecosystems will respond, and produce data required to verify and improve models.

It has been shown in Chapters 2 and 3 that the blanket peat contains by far the greatest amount of organic C at Moor House. The blanket peat is believed to have accumulated over the last *ca.* 7,500 years (Heal and Smith, 1978) and has, therefore, been a long-term sink for atmospheric C. However, the rate of CO_2 sequestration is unlikely to have been constant, with natural variations in climate having affected the rate of peat accumulation. The rate of peat growth is directly dependent on rates of NPP and decomposition (Clymo, 1991), processes which are strongly influenced by climate (Leith, 1975; Meentemeyer, 1978). However, it is unclear how future climate change will affect rates of peat accumulation and, therefore, the vast amount of C sequestered annually in global peatlands (Gorham, 1991).

The aim of this chapter is to investigate the impact of past variations in climate on the rate of C accumulation in blanket bog at Moor House. The response of the past peat accumulation to climate change is being used as a surrogate to predict the impact of the anticipated global warming on future C accumulation in peats.

5.1.1 Peat accumulation

Mires accumulate peat because the decomposition of plant material produced by their surface vegetation is impeded (Clymo, 1984). Reduced decomposition is predominantly due to the water-logged conditions associated with mires, but other factors, such as high acidity, cold temperatures and the low quality of plant litter (*e.g. Sphagnum* sp.) also contribute.

To understand the processes occurring in peat accumulation, it is useful to adopt the classification of Ingram (1978). This classification defines two horizontal layers within the vertical peat profile; the acrotelm is the top layer of peat extending from the surface to the lowest point which the water table reaches in summer; the catotelm is the layer beneath, which is continually saturated with water. The conditions in the catotelm are, therefore, permanently anaerobic, whereas the acrotelm experiences fluctuating aerobic and anaerobic conditions, depending on the depth of the water table. Peat accumulates on the surface of mires as litter from plants growing on the surface, (*e.g. Sphagnum* sp.) which are subsequently covered by new plant growth. The litter arriving from new plant growth is later deposited on the top of the existing litter, which only partly decays due to the low rate of decomposition. Subsequent layers of peat are therefore formed directly above older layers, and form stratified deposits, frequently several metres in depth.

Since decomposition of organic matter is much greater under aerobic than anaerobic conditions, rates of decomposition are greater in the acrotelm (*ca.* 10-80 % weight loss per year; Clymo, 1984). Once decaying plant material has passed into the catotelm it decays at a much slower rate and the catotelm is the true site of peat accumulation (*ca.* 0.1 % weight loss per year; Clymo, 1984; Clymo *et al.*, 1998). Although decomposition under anaerobic conditions is extremely slow, microbial production of CH_4 within the catotelm is evidence for its occurrence (Clymo and Pearce, 1995). Eventually, a limit to the depth of peat deposits is believed to occur when the **total** decomposition throughout the peat profile balances the input of new litter (Clymo, 1984).

The factors which determine the extent of litter decomposition before entering the catotelm include the rate of weight loss and the total time the material spends within the acrotelm. Since the acrotelm is, by definition, only water-logged for part of the year, the extent of decomposition largely reflects the proportion of time the decaying plant material experiences aerobic conditions. In fact, the process is probably far more complex since Belyea (1996) found, in a mire in south west Scotland, highest decompositon rates at, or just above, the zone of water table fluctuation.

Climate, therefore, strongly influences the rate of peat accumulation through a number of processes. Firstly, climate controls NPP and, therefore, the rate at which new organic matter enters the system. Decomposition is also affected by climate because, as with other biological reactions, the process is more rapid at higher temperatures (Meentemeyer, 1978). However, a greater influence of climate on decomposition is through control of water table depths. Water tables represent a balance between water inputs in precipitation and outputs as evapotranspiration and drainage, processes governed by climate.

Rates of peat accumulation can be expressed in terms of height increment (*e.g.* cm yr^{-1}) or dry mass accumulation (*e.g.* g m^{-2} yr^{-1}). For the purposes of the present study, the latter is used to avoid errors arriving from variation in the bulk density of peat which occur both between different peat types and within single profiles. However, many studies of peat accumulation have reported changes in the rate of height increment, with bulk density frequently not being determined (Clymo, 1983). Consequently, there are relatively few estimates of peat mass accumulation rates in the literature. In the present study, rates of accumulation refer to mass and not height increases.

The determination of rates of C accumulation in peat requires measurement of the mass of C accumulated over a known time interval. Measurement of C content between two points is a relatively straightforward task, although, accurate determination of the age of peat is more difficult. However, many methods now exist for dating peat samples, such as pollen analysis (*e.g.* Tallis, 1985) and volcanic tephra layers (Dugmore *et al.*, 1995), although radiometric techniques (*e.g.* Oldfield *et al.*, 1997; Appleby *et al.*, 1997) are the most frequently employed. However, all methods of dating peat samples are subject to inaccuracies and the most reliable chronologies of peat profiles are based on a combination of several dating methods. The present investigation was aimed to determine rates of C accumulation in peats by dating a peat profile using a range of techniques, including radiocarbon dating, pollen analysis, correlation with known pollution histories and tephra analyses.

5.1.2 Reconstructing past climates from peat

It is now recognised that raised mires contain a clear proxy-climate signal preserved in their peat stratigraphy (Aaby, 1976; Blackford, 1993; Barber *et al.*, 1994; Tallis, 1995). Additionally, the value of blanket mires as sources of palaeoclimatic information has recently been demonstrated (Blackford & Chambers, 1991; Tallis, 1995; Chambers *et al.*, 1997). Proxy-climate signals can be derived from peats in several ways, such as analysis of preserved macro-fossils (*e.g.* Barber *et al.*, 1994) and degree of peat humification (*e.g.* Blackford & Chambers, 1991). In the present study, degree of humification, plant macro-fossil analysis and rhizopod analysis have been used to reveal the palaeoclimate signal in a peat profile from the blanket bog at Moor House.

5.1.2.1 Degree of peat humification

The degree of humification of peat has been identified as a proxy-climate signal since climate strongly controls rates of decomposition (Section 5.1.1) and, therefore, the proportion of humic acid in peat (Blackford and Chambers, 1993). The degree of humification is thought to relate to the surface wetness of the mire at the time of deposition (Blackford and Chambers, 1993).

Although there are other factors which influence the degree of humification in peat samples, such as species composition (Blackford and Chambers, 1993), synchronous changes in this characteristic have been identified at different sites, suggesting a climate cause (Blackford and Chambers, 1991). For example, Blackford and Chambers (1991) demonstrated a synchronous shift in the degree of peat humification at *ca.* 1400 BP from a range of sites across the British Isles and, subsequently, Blackford and Chambers (1995) found a correlation between solar variability and degree of humification of Irish peats formed during the last 1000 years, implying changes in solar output were recorded in peat stratigraphy.

5.1.2.2 Plant macro-fossil analysis

Although many factors influence species composition of vegetation growing on the surface of mires (*e.g.* grazing; Smith and Forrest, 1978; Grant *et al.*, 1985) possibly the most important factor is climate. Climate influences species composition both directly, through selection of species adapted to particular temperature and precipitation conditions, and indirectly through climate's influence on water tables. That water tables conditions are particularly important for determining the species composition on mires is evident from vegetation found in hummocks and hollows; the water table in hollows is frequently near the mire surface and species tolerant of extremely water-logged conditions (*e.g.* *S. cuspidatum*) are typically present; conversely, hummocks are drier and exhibit characteristically different species assemblages (*e.g. Racomitrium lanuginosum, S. papillosum*; Birks and Birks, 1980). Since peat is formed from layers of partially decomposed vegetation which once grew on the surface of the mire (Clymo, 1984), the plant species composition of stratified peats retains a record of past climatic conditions. Therefore, identification of the species composition of plant macro-fossils in these deposits reveals a record of past climate (Barber, 1981; Barber *et al.* 1994).

However, there are many factors which influence the fossil species composition of peat, which may modify or reduce the climate signal. For example, there are non-climatic factors, such as changes in nutrient availability to plants and nitrogen deposition (Lee, 1998) which influence surface vegetation composition. Furthermore, differences in the relative decomposition rates of plant species may reduce the extent to which the fossil assemblage of plant species preserved in the peat represents previous vegetation assemblages, since decomposition rates vary greatly between mire species (Heal *et al.*, 1978; Latter *et al.*, 1998). Additionally, the identification of plant macro-fossils depends partly on the degree of their preservation such that, species which decay more rapidly are harder to identify. Some plant species, such as *Eriophorum vaginatum* and *E. angustifolium* produce deep roots which are incorporated into layers of peat already formed and, therefore, compromise the original species composition. Therefore, although profiles of macro-fossil composition derived from peats contain a strong climate signal, it is clear that interpretations of profiles should be made with caution due to potential confounding factors (Mauquoy, 1997).

Despite these limitations, macro-fossil analysis of plant remains has increasingly been used in palaeoecological studies of past climate change, although the most convincing recognition of a climate signal is demonstrated when a synchronous shift in climate, in the same direction, is seen at more than one site, or correlates with records derived from other palaeoclimate sources (*e.g.* tree rings); this reduces the possibility that localised changes in, for example land management and pollutant deposition *etc.,* have produced the observed changes in species composition. There are many good examples of possible synchronous shifts obtained from macro-fossil analyses from different sites (*e.g.* Stoneman, 1993; Barber *et al.*, 1994; Mauquoy, 1997).

5.1.2.3 Rhizopod analysis

Testate amoebae (rhizopods) are microscopic unicellular organisms with shells which are preserved in peats (Hendon and Charman, 1997), which inhabit, and are extremely abundant in, the top few centimetres of peats (Heal, 1962). Recent research has shown that species assemblages of testate amoebae are strongly related to peatland hydrology and that transfer functions can be used to make quantitative estimates of past water table levels based on the species assemblage (Hendon and Charman, 1997). For example, the species *Amphitrema flavum* is typically found in high moisture conditions, whereas *Assulina muscorum* prefers drier mire surface conditions (Mauquoy, 1997). Testate amoebae can, therefore, be used to complement plant macro-fossil and degree of humification investigations in order to reconstruct past climate change (Mauquoy, 1997).

5.2 Method

The present investigation was undertaken in collaboration with Dmitri Mauquoy, a PhD student from the Department of Geography, University of Southampton. Mauquoy assisted during fieldwork, and performed the plant macro-fossil, rhizopod and pollen analyses. All other analyses were undertaken by the author, unless stated otherwise.

5.2.1 Sampling location and collection and preparation of core

In order to produce a full C inventory in peat, together with accompanying climatic reconstruction, a large peat sample at each depth was needed. Conventional peat corers do not supply the necessary amount of peat from a single core and, although it may be possible to match closely sampled cores, based on stratigraphy, this may introduce unacceptable errors. Consequently, a peat corer was specifically designed (with the assistance of David Benham, ITE Merlewood) which enabled undisturbed peat samples to be retrieved whilst providing sufficient sample for all the analyses. The corer was a hinged box corer (Plate 5.1) comprising a stainless steel case, open at either end, with a hinge running down the entire length of one corner. The opposite corner was fixed with removable screws permitting extraction of the core. The corer was square (15 cm x 15 cm) and measured 92 cm in length.

During sampling, the corer was pushed into the surface of the blanket bog vertically, after first incising an outline of the corer with a sharp knife in the surface of the peat. Once the corer had been forced to full depth, a hole was dug down one side and a knife used to sever the peat at the base of the corer from the peat below. The corer was then extracted with a spade held against the base of the core to prevent the core from adhering to deeper layers. The coring operation was quite a slow and difficult process, but, the peat core obtained this way provided the necessary amount of peat.

The peat core used in the present study (MH2) was retrieved on the 14th May 1996, from Shaft Hill, Moor House (Figure 5.1; Plate 5.2). The site was water-shedding and away from currently eroding areas. After retrieval, the core was immediately returned to the laboratory and sectioned. Slices of 1 cm thickness were taken using a sharp knife, although

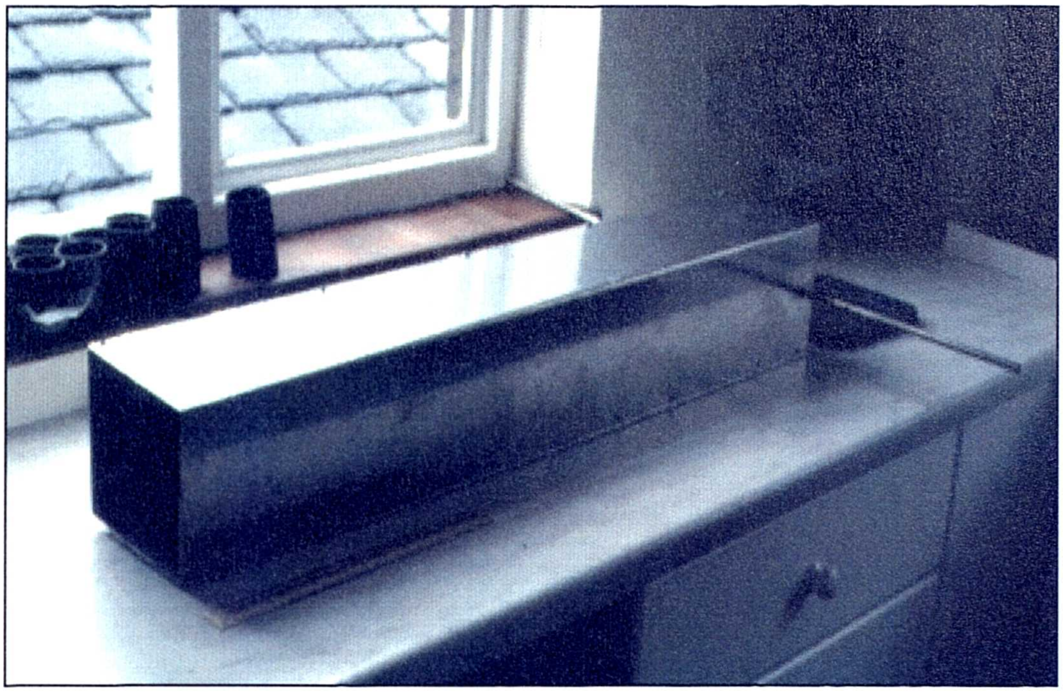

Plate 5.1 Hinged box corer used to take peat core MH2

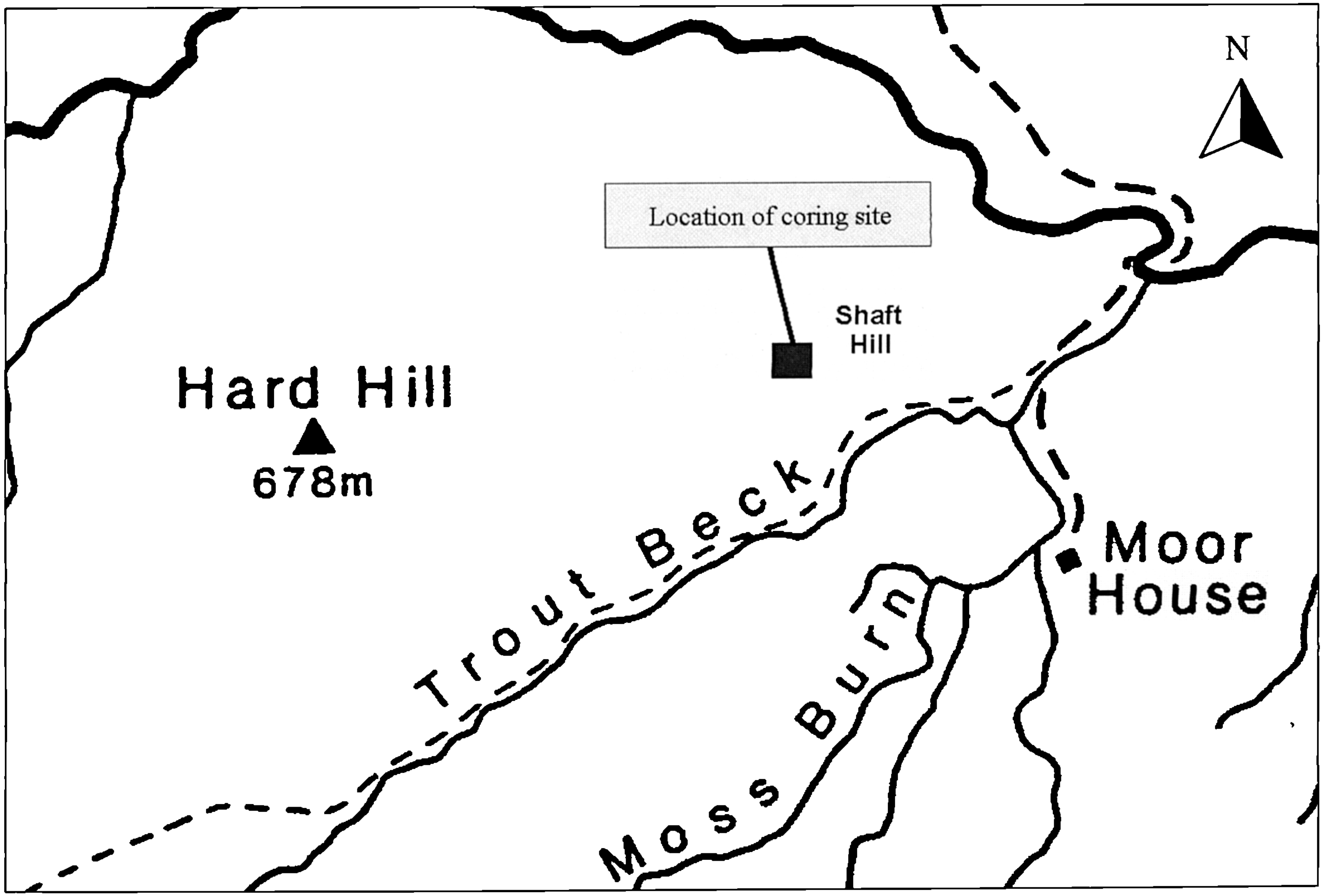

Figure 5.1 Sampling location for core MH2 within Moor House NNR

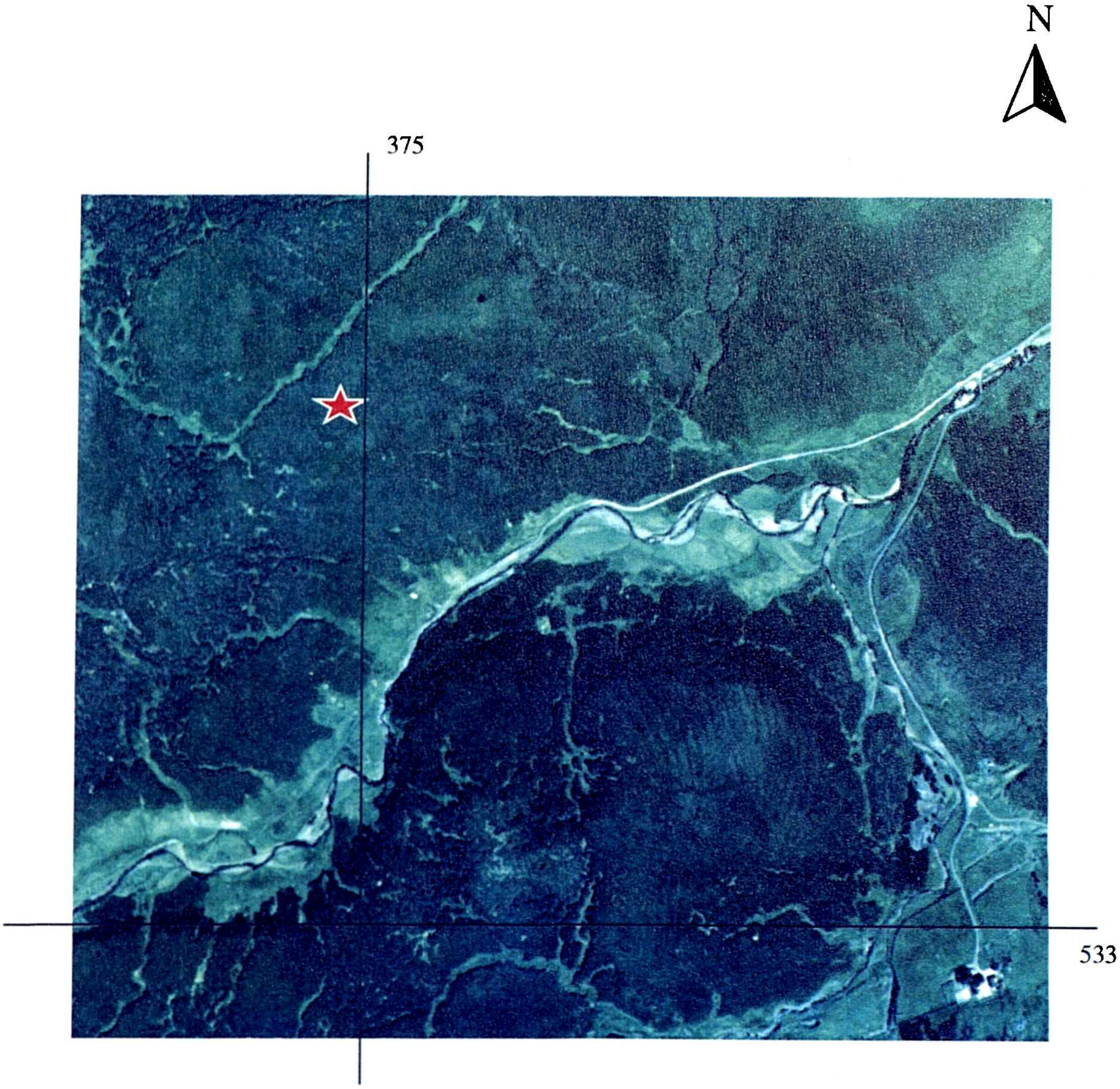

Plate 5.2 Location of the sampling point for core MH2 (★) at Moor House NNR

this was not possible for the surface 20 cm and bottom few centimetres of the core because of a lack of cohesion in the peat; these parts of the profile were therefore sectioned at larger intervals.

Each slice was wrapped in aluminium foil, labelled and then wrapped in polythene to prevent water loss. Samples were placed in plastic bags and stored in a refrigerated room (4°C) until required for analysis, ensuring that samples were not compressed.

5.2.2 Determination of C accumulation rates

Bulk density and moisture content were measured on sub-samples from every slice of the peat core. However, because accurate estimates of bulk density require accurate estimates of the fresh volume of a peat sample, simply measuring the volume of the sub-sample with a ruler was considered too inaccurate. Consequently, the volume of the sub-sample from each slice was determined using a displacement technique similar to that described by Janssens (1983):

The sub-sample had dimensions of *ca.* 15 cm x 2 cm x 1 cm (since most slices were 1 cm thick) and was removed from the centre of the peat slice, and weighed to obtain the wet weight. The sub-sample was carefully placed inside a clean 250 ml measuring cylinder using a narrow ruler, so that it stood vertically in the cylinder. Exactly 200 ml of deionised water was carefully poured into the cylinder, completely submerging the sub-sample. The volume registered by the measuring cylinder was noted and the sub-sample volume calculated.

The contents of the measuring cylinder were then poured over a very fine sieve, retaining the peat but allowing the water to drain away. The peat was then placed in an aluminium tin, oven-dried for 24 hours (to constant weight) at 105°C, cooled in a desiccator and re-weighed to obtain the dry-weight of the sub-sample. Bulk density and moisture content were calculated using equations 3.7 and 3.8.

%LOI was determined for each depth except for the top and base of the core which had been sectioned at larger intervals. Percentage LOI was also determined for most of these thicker slices. For %LOI measurements, sub-samples were dried at 105°C and ground in a ball mill fitted with an agate chamber (Glen Creston). About 1 g dry weight of peat was used for each %LOI determination after Allen (1989; described in Section 3.2.3.3). The %C content of each sample was estimated using the regression derived by Bol *et al.* (in press) specifically for Moor House soils (equation 3.9).

The above analyses provided a record of the cumulative C with depth. However, the study required a chronology for the profile in order to determine rates of C accumulation. Accelerator Mass spectrometry (AMS) radiocarbon dating of individual above-ground plant fragments was believed to be the best available method to devise an accurate and precise chronology and has been undertaken on samples from the core.

In order to guide the selection of samples for AMS dating, the record of SCP preserved in peat (see Chapter 4) was investigated. Samples were prepared using the methods described in Section 4.2.2.2, based on the technique described by Rhodes (1996). The SCP profile for MH2 was determined from contiguous samples down to 35 cm and then approximately every 4 cm to the base (81-83 cm) by counting every SCP particle in the Petri-dish samples. The transect approach used to rapidly quantify SCPs in Chapter 4 was developed by also scanning transects of the Petri-dishes.

Two samples were radiocarbon dated using conventional methods and acted as 'range finders' to aid the selection of sample depths for AMS dating. Samples of dried and ground peat (prepared as for %LOI described above) from depths 58-59 cm and 76-77 cm were dated by the NERC Radiocarbon Laboratory, East Kilbride.

Based on the two 'range-finder' sample dates for core MH2, samples for AMS dating were selected at intervals of 4 cm, starting from 28 cm down the profile. A second, independently collected sub-sample was obtained from depth 56-57 cm to verify the reproducibility of the AMS radiocarbon dates.

The samples for AMS dating were hand sorted so that plant roots, which may have penetrated from further up the profile, were not included. Individual leaves of *Sphagnum* sp. were sought and obtained for most samples. However, when these were not found, stems of *Sphagnum* sp. and leaves of *Calluna vulgaris* were also used for the sample, the material dated being noted. The air-dry weight of resulting material was *ca.* 0.02 g, and samples were placed in clean glass vials and sent for dating to the NERC Radiocarbon Laboratory.

Since the radiocarbon content of the atmosphere has varied in the past, calibration between radiocarbon age and calendar age using radiocarbon dated tree ring records is necessary. The AMS dates were calibrated using the 'Sequence' function in OxCal (version 2.18; Bronk Ramsey, 1995); the procedure calculates the most probable calendar age ranges for samples formed in a chronological sequence, such as in stratified peat deposits, using Bayesian statistics (Bronk Ramsey, 1995). The individual conventional radiocarbon dates were calibrated independently using OxCal.

Pinus pollen counts were made in order to determine the 'pine rise', believed to relate to afforestation with *Pinus sylvestris* in the 18th-19th centuries (Mauquoy, 1997). Preparation of pollen slides followed standard methodologies (Moore *et al.*, 1991) and was performed on sub-samples (volume 5 cm^3) obtained using a volumetric sampler. Counts were made at intervals of 4 cm to a depth of 81 cm in core MH2. The pollen concentrations were calculated using marker grains of Lycopodium (Stockmarr, 1971), and counts continued until 50 *Pinus* pollen or 800 Lycopodium spores had been counted.

Additionally, samples were analysed for volcanic tephra layers in order to provide fixed chronological references. This work was undertaken in collaboration with Jeff Blackford[1] who performed the analyses.

5.2.3 Reconstruction of past climatic change

The degree of peat humification, plant macro-fossil composition and testate amoebae were analysed in an attempt to derive a proxy record of past climate change. Principal component analysis (PCA) was used to convert the results of the analyses for macro-fossils and testate amoebae into a 'surface wetness index', with similar multivariate techniques having been shown to be effective for summarising macro-fossil results into proxy climate records (Barber *et al.*, 1984; Mauquoy, 1997). The SAS procedure PRINCOMP was used for PCA analyses (SAS Institute Inc., 1985).

5.2.3.1 Degree of peat humification

Degree of humification was determined using the method of Blackford and Chambers (1993) in which the absorbance of the alkali-extract of dried and ground peat samples is measured. The analyses were performed on each sample from core MH2.

Contiguous 1 cm thick samples of peat were oven-dried (105°C) to constant weight. The samples were placed immediately into desiccators on removal from the oven and allowed to cool. The samples were then ground to a fine powder using a ball mill (Glen Creston) fitted with an agate chamber and returned to the oven to remove any moisture absorbed from the atmosphere during grinding. The samples were removed and cooled in desiccators again.

Sub-samples of 0.2 g of the prepared peat were placed in 150 ml flasks and 100 ml of freshly prepared 8% NaOH added, the time of mixing being recorded. The flasks were heated on a hot plate (Corning PC-100) in a fume cupboard, until the solutions boiled, removed from the heat and allowed to cool slightly and returned to the hot plate. Samples were then gently simmered for 1 hour after which they were removed from the heat and allowed to cool. A total of eight samples and one internal standard were analysed in a single batch.

The solutions were transferred into separate 100 ml volumetric flasks and deionised water was added to bring the solution up to the 100 ml mark. The flasks were thoroughly shaken and samples filtered through Whatman Qualitative 1 filter paper, into 250 ml volumetric flasks.

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[1] Department of Geography, Queen Mary and Westfield College, London

Exactly 50 ml of the filtered solution was poured into a glass measuring cylinder and diluted 1:1 by topping up to the 100 ml mark with deionised water. This solution was then transferred to a 150 ml volumetric flask ready for colour measurement.

The absorbance of the solution was measured at 540 nm using a linear readout grating spectrophotometer (Cecil Instruments CE373). All measurements were taken four hours after the initial mixing of the peat samples with the NaOH, since fading of the colour extract occurs.

The solution was thoroughly shaken and a sub-sample of 3.3 ml pipetted into a glass cuvette. The cuvette was placed in the spectrophotometer and absorbance measured. The absorbance of a total of three sub samples was derived to obtain a representative value for the sample; if the solution had been thoroughly mixed there was little variation in absorbance values of sub-samples.

5.2.3.2 Plant macro-fossil analysis

Plant macro-fossil analysis was undertaken on samples at 2 cm intervals except for the top and base of the core where slices were greater than 1 cm thickness. These samples were all analysed.

A sub-sample, 1 cm x 1 cm x 4 cm was extracted from each depth investigated with a surgical scalpel and placed on a 125 µm sieve. The sub-sample was sprayed with a high pressure jet of water to remove highly humified material and disaggregate the sample. The remaining material was then transferred to a glass trough and 70 ml of water added, and scanned using a Nikon stereozoom microscope at x10 magnification. The abundance of identifiable *Sphagnum*, unidentifiable organic matter, ericaceous remains, monocotyledons and the proportion of empty space was determined using a 10 x 10 square grid graticule inserted into the eyepiece of the microscope. A score of 1 % was given to each component which covered more than half of a square. A total of 15 quadrats were scanned this way for each sample.

The material was then stained blue by adding toluidine blue and thoroughly shaking the trough. Between *ca.* 80 and 100 *Sphagnum* leaves were then randomly selected and identified to the lowest taxonomic level at x400 magnification. A bryophyte reference collection and the drawings and keys of Smith (1980) and Daniels and Eddy (1990) were used for identification of leaves.

The following formula was used to calculate the percentage of *Sphagnum* leaves in each sample:

$$Ps = (Nli / n) \times Plds \quad [equation\ 5.1]$$

Where:
Ps = percentage of leaves in the sample
Nli = number of leaves of species i counted
n = total number of leaves in sample
$Plds$ = percentage identifiable *Sphagnum* in the sample

The content of non-*Sphagnum* bryophytes (*e.g. Racomitrium lanuginosum*) was also estimated for each sample and expressed as a mean percentage of the peat matrix examined. Fruit and seeds (referred to as seeds here for convenience) were counted in each sample on a 1-5 basis where 1 = one seed, 2 = 2-5 seeds, 3 = 5-20 seeds, 4 = 20-40 seeds, 5 = greater than 40 seeds. Charcoal fragments were also counted while quantifying other peat components with each fragment being assigned a size class (<15,625 μm^2, 15,625-62,500 μm^2, 62,500 μm^2-0.25 mm^2, 0.25-1 mm^2, 1-4 mm^2, and > 4 mm^2), determined by using the square grid graticule.

5.2.3.3 Rhizopod analysis

Testate amoebae and other zoological remains were identified from the same sample depths as the plant macro-fossils. A detailed description of the method is provided by Mauquoy (1997), and summarised below:

A sub-sample of 5 cm^3 of peat was collected from each level using a volumetric sampler and boiled in 50 ml of distilled water for 10 minutes (following the method of Tolonen, 1986 in Mauquoy, 1997). To remove larger material, the peat residue was sieved (mesh = 250 µm) and then centrifuged at 3000 revolutions per minute, for three minutes. The sample was then decanted and 0.5 ml of glycerol added and stained with safranine.

Testate amoebae were identified using a variety of references (see Mauquoy, 1997) and enumerated until between 100 and 150 had been identified. While scanning for testate amoebae, the fossil remains of other zoological remains were identified (*e.g.* parts of Cladocera) and also counted.

5.2.4 Measurement of water table depth at the coring site

The water table was measured at the coring site for a year in order to estimate the depth of the acrotelm-catotelm boundary which is necessary in order to compare rates of C accumulation in surface peats. The dip well used was of a design previously used by Chapman (1986). It consisted of a 2 m square plastic tube (*ca.* 5 cm x 5 cm) which enclosed a glass tube of diameter *ca.* 1 cm and length 1 m fixed vertically inside towards one end. A moveable pointer was attached to the glass tube and the marker attached to a sealed plastic bottle. The dip well was placed vertically in the peat so that the plastic bottle floated on the water table, and the marker therefore indicated the water table height. Since the bottle moved with the water table, the marker also moved along the glass tube, indicating the changing water table. In the original design, maximum and minimum water tables could be measured by fastening two metal springs to the glass tube, above and below the marker; when the water table moved closer to the surface the marker pushed one of the springs upwards which remained at its new location, indicating the maximum height of the water table. However, maximum and minimum water tables could not be measured by this dip well at the Shaft Hill site because strong winds caused the springs to move, and because the springs frequently rusted and broke.

Measurement of water table depth from the dip well was taken on a fortnightly basis for over a year, with a few exceptions when, due particularly to severe weather conditions, the dip well was not measured. The author undertook a few of the measurements, however, the majority were taken by staff and students of the ECN, since they make weekly visits to the site. The dip well was investigated on a total of 24 occasions between October 1996 and November 1997.

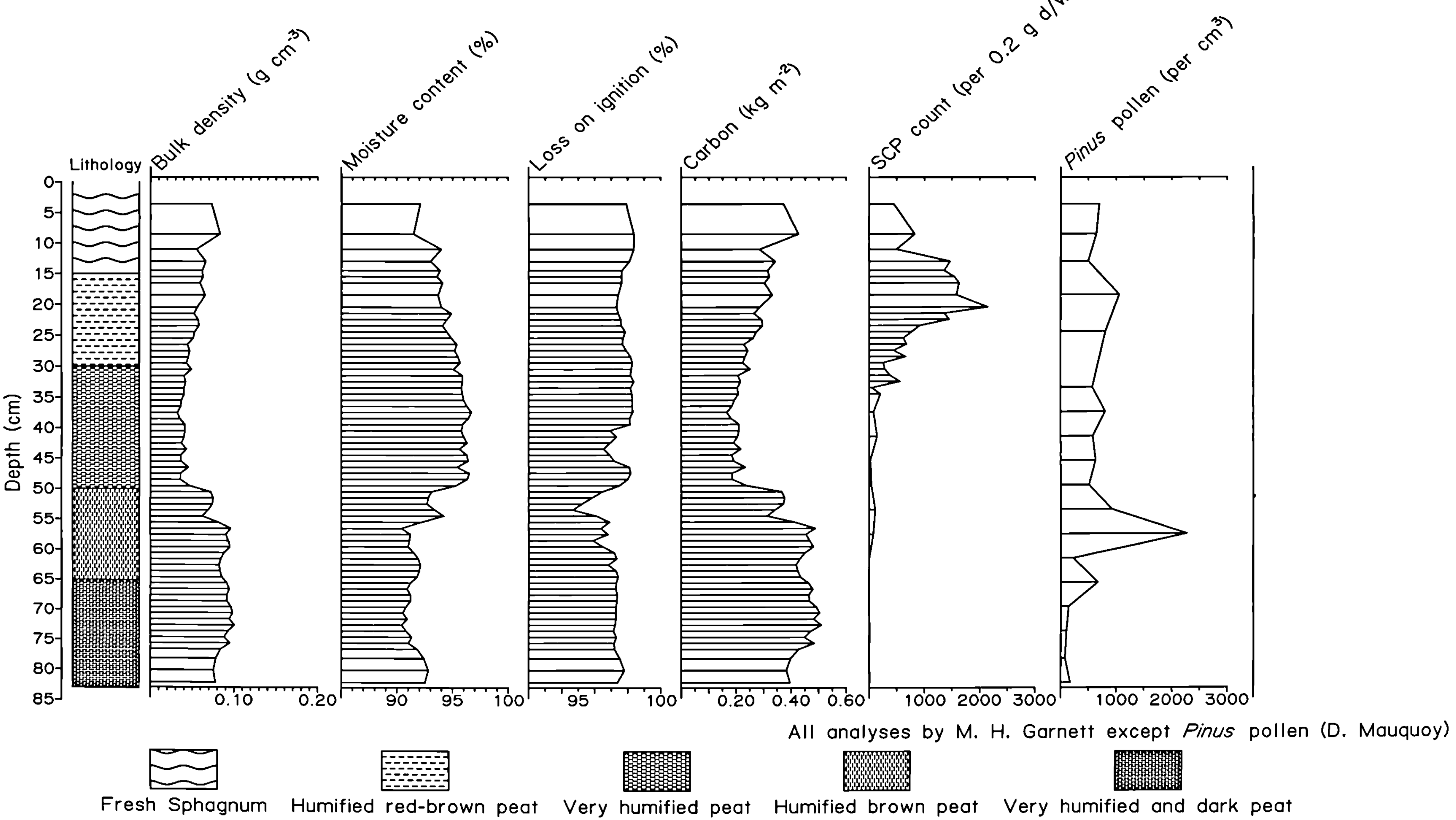
Water table depth was also measured at a nearby site using a buried pressure transducer (PDCR830, Campbell Scientific, Shepshed) connected to a data logger (CR10, Campbell Scientific, Shepshed), by the ECN. The pressure transducer was buried at approximately 80 cm below the peat surface and monitored water tables by measuring the pressure of water above the sensor. Pressure measurements were made every 5 seconds and averaged for each hour. The sensor could be calibrated against a dip well located above the sensor and maintained by the ECN. This logger therefore provided hourly records of water table depth. For the present study, the dip well at the MH2 coring site has been calibrated against the ECN logger results in order to estimate the maximum water table depths encountered at the site, and therefore to help estimate the boundary between the acrotelm and catotelm.

5.3 Results

5.3.1 Determination of C accumulation rates

Figure 5.2 shows the profile of bulk density, moisture content, %LOI, SCP count and *Pinus* pollen concentration for core MH2. Figure 5.3 shows the correlation between counts of SCPs

Figure 5.2 Profile description of MH2 showing bulk density, moisture content, %LOI, C content, SCP count and concentration of *Pinus* pollen



All analyses by M. H. Garnett except *Pinus* pollen (D. Mauquoy)

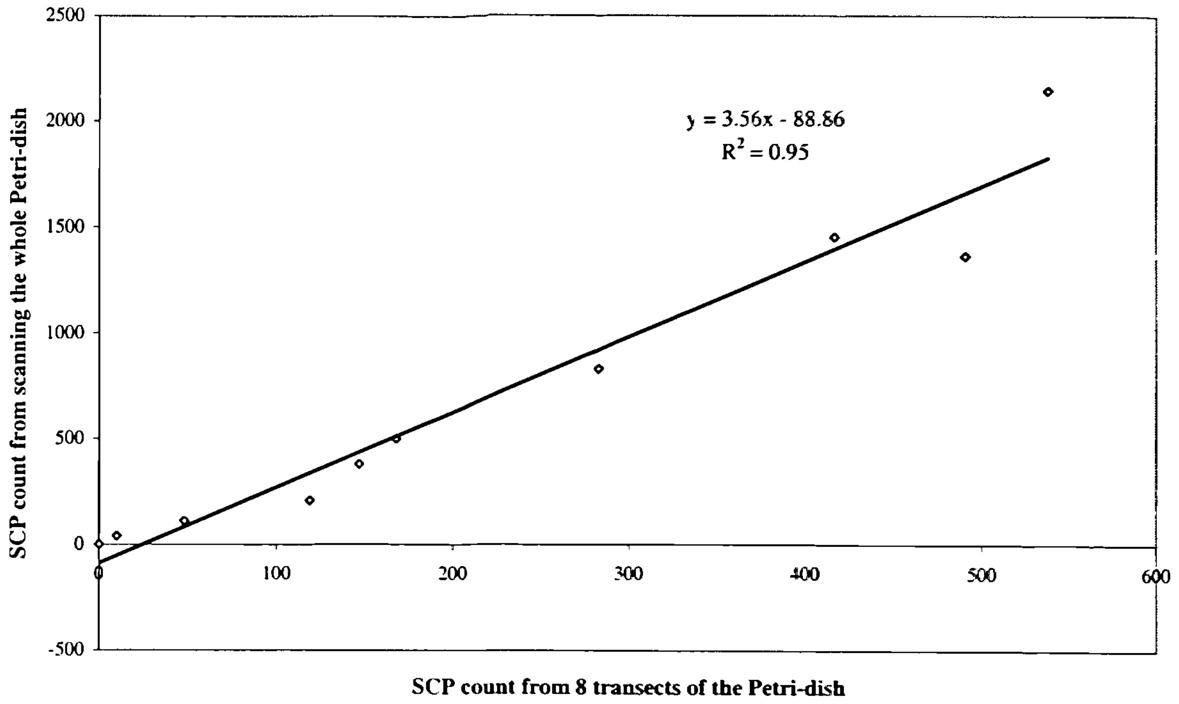


Figure 5.3 Scatter plot showing the comparison of SCP counts on samples after scanning the whole Petri-dish and the 8 transects (n=10)

from the whole Petri-dish and counts based on the 8 transects used to allow a more rapid determination of the SCP profiles for the short cores investigated in Chapter 4.

Bulk density was greatest (*ca.* 0.1 g cm^{-3}) in the lower part of the core, declined rapidly at *ca.* 50 cm depth and then progressively increased from *ca.* 0.05 g cm^{-3} to 0.08 g cm^{-3} towards the surface. Moisture content followed an inverse trend to bulk density, while %LOI ranged between 95 % and 98 %. The profile of C content was derived from the bulk density and %LOI results, and consequently, since %LOI did not vary as much as bulk density, the profile of C content was similar to the bulk density profile.

The concentration of SCPs increased progressively nearer the surface until reaching a peak at 15 cm depth where, above this, concentrations declined. The main 'take-off' in SCP numbers occurred at *ca.* 25 cm.

The profile of *Pinus* pollen concentration showed low concentrations in basal samples, a distinct peak at *ca.* 60 cm depth, and approximately constant counts towards the surface.

Unfortunately, no clear tephra layers were discovered (Blackford, personal communication), although a few fragments were found in several samples (depths 14-16, 51-52, 61-62, 68-69, 75-76 and 81-83 cms). The chronology of the core is, consequently, based only on the SCP profile and radiocarbon results.

Table 5.1 shows the conventional bulk radiocarbon dates, providing the sample reference, depth range, uncalibrated and calibrated ages of the samples.

Table 5.1 Range-finder conventional radiocarbon dates and δ ^{13}C from core MH2. Calibrations were performed using OxCal (version 2.18; Bronk Ramsey, 1995)

Sample reference	Depth (cm)	Radiocarbon age (BP; 1 sigma error)	Calibrated age to 2 δ	Mid-point of calibrated age range	δ ^{13}C ‰
SRR-5900	58-59	820 ± 45	1040-1280 AD	1160 AD	-26.0
SRR-5901	76-77	1675 ± 45	230-450 AD	340 AD	-26.5

Table 5.2 shows the results from the AMS radiocarbon dating of samples. The table provides the sample references, depths, uncalibrated radiocarbon age and delta ^{13}C.

Table 5.2 Details of AMS radiocarbon dates and $\delta\ ^{13}C$ values from core MH2

Sample reference	Depth (cm)	Radiocarbon age (BP ± 1σ)	$\delta\ ^{13}C_{PDB} \pm 0.1$ ‰
AA-27163	28-29	155 ± 50	-25.3
AA-27164	32-33	155 ± 45	-24.4
AA-27165	36-37	220 ± 60	-24.1
AA-27166	40-41	285 ± 45	-23.7
AA-27167	44-45	455 ± 50	-23.8
AA-27168	48-49	205 ± 50	-21.5
AA-27169	52-53	185 ± 50	-25.5
AA-27170	56-57	435 ± 45	-24.1
AA-27171	60-61	405 ± 45	-27.2
AA-27172	64-65	495 ± 50	-27.5
AA-27173	68-69	790 ± 50	-27.3
AA-27174	72-73	865 ± 50	-26.9
AA-27175	76-77	2,200 ± 55	-27.6
AA-27176	56-57 (repeat)	510 ± 55	-23.0

The AMS dates from profile MH2 did not show a fully consistent increase in age with depth. The radiocarbon dates at depths 48-49 cm and 52-53 cm were considerably younger than the dates obtained for samples above, whilst the sample at 44-45 cm had a radiocarbon age that was slightly older than the sample at 60-61 cm. Since ombrotrophic peat accumulates vertically, with younger peat forming on top of older, the radiocarbon ages for these few samples are surprising. In particular, the samples at 48-49 cm and 52-53 cm appear incorrect and possible explanations for these samples having considerably younger radiocarbon ages are presented below. However, the sample at 44-45 cm is only slightly older than the sample at 60-61 cm, and given the error associated with radiocarbon dating, it is possible that this date is correct.

The two AMS dates from the same depth in core MH2 (56-57 cm) showed similar radiocarbon ages (435 BP and 510 BP) with overlapping error values. One sample (76-77 cm) from MH2 was dated using both AMS and conventional radiocarbon dating and the two methods produced different estimates of radiocarbon age; AMS provided an age of 2,200 ± 55 BP and conventional dating an age of 1,675 ± 45 BP.

5.3.1.1 Calibration of the radiocarbon dates

Due to the occasional problems with the AMS dates mentioned above, three different calibrations were derived by omitting dates suspected of being erroneous. Using all the AMS dated depths did not produce an acceptable model according to the OxCal program (Agreement index < 60 %; Bronk Ramsey, 1995), so dates suspected of being erroneous were omitted in further calibrations. Table 5.3 shows the samples omitted to produce the three modelled age series; application of these different age models allowed the effect of dating errors on the calibrated age versus depth profiles to be investigated and the calibrated ages for the three age models are shown in Table 5.4.

Table 5.3 Details of radiocarbon dated samples not used for the three different age models

Age model	Depth of AMS dated samples omitted
1	48-49 cm and 52-53 cm
2	44-45 cm, 48-49 cm and 52-53 cm
3	48-49 cm, 52-53 cm and 56-57 cm

Table 5.4 Calibrated ages for AMS radiocarbon dates. Three different age models have been produced by omitting different dates suspected of being incorrect. All dates are AD unless shown and calibration was performed in OxCal (version 2.18; Bronk Ramsey, 1995).

	Model 1		Model 2		Model 3	
Sample depth (cm)	2 δ range	Mid point	2 δ range	Mid point	2 δ range	Mid point
28-29	1733-	-	1736-	-	1732-	-
32-33	1660-1930	1795	1660-1930	1795	1660-1890	1775
36-37	1530-1810	1670	1530-1810	1670	1530-1820	1675
40-41	1490-1670	1580	1480-1670	1575	1480-1670	1575
44-45	1430-1630	1530	Not included	-	1420-1630	1525
48-49	Not included	-	Not included	-	Not included	-
52-53	Not included	-	Not included	-	Not included	-
56-57	1425-1515	1470	1430-1630	1530	Not included	-
60-61	1415-1485	1450	1415-1510	1463	1414-1505	1460
64-65	1310-1460	1385	1310-1460	1385	1300-1460	1380
68-69	1160-1290	1225	1160-1290	1225	1160-1290	1225
72-73	1030-1240	1135	1030-1240	1135	1030-1240	1135
76-77	390 BC -110 BC	250 BC	390 BC -110 BC	250 BC	400 BC -110BC	255 BC

Figure 5.4 shows the age versus depth of samples from core MH2 derived using the three calibration models. The calibration models show strong agreement in the lower part of the core, though differ slightly over the range where samples have been omitted.

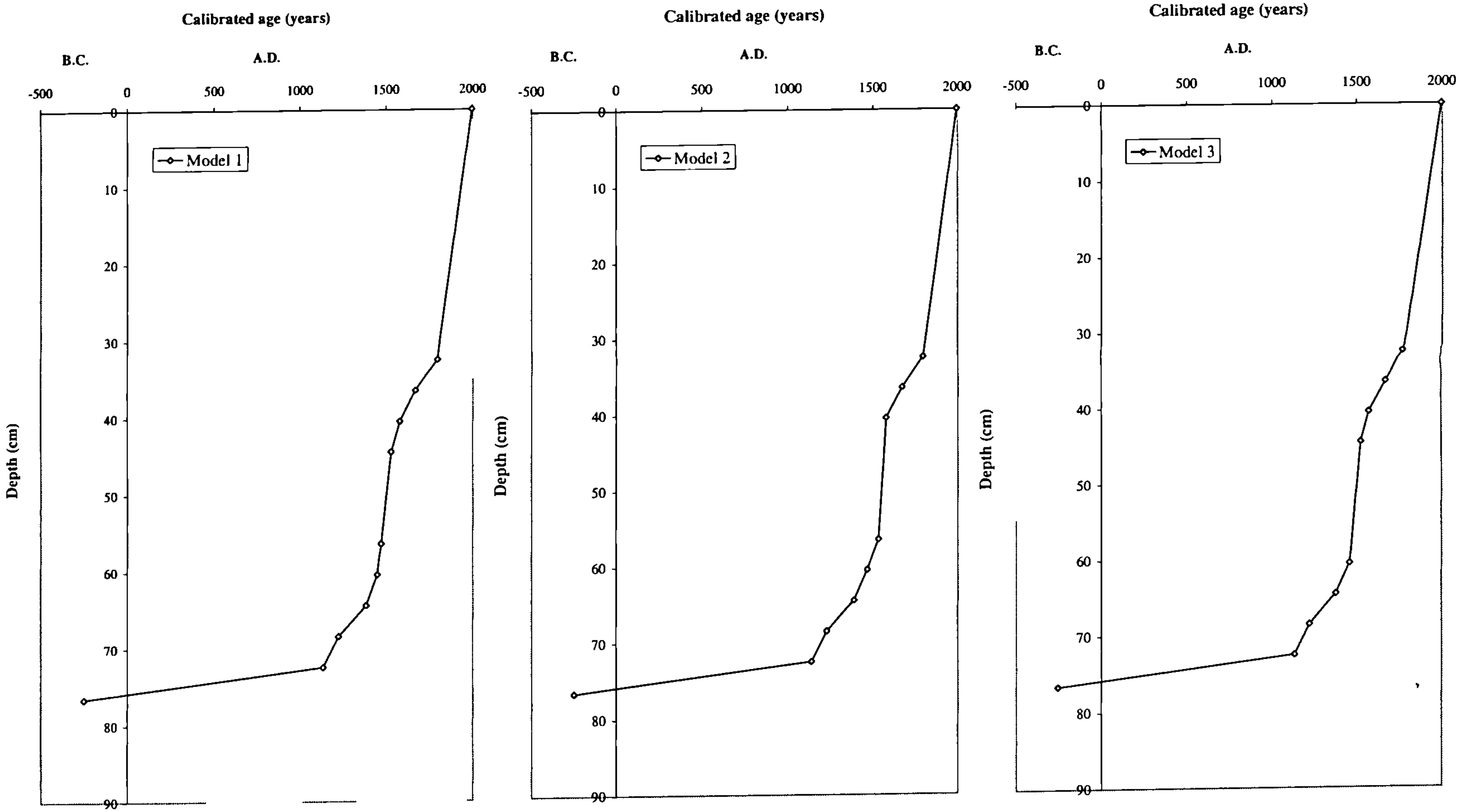


Figure 5.4 Plots of age against depth for core MH2. The plots represent the three age models derived by excluding dates suspected of being incorrect, prior to calibration (see text).

Since the age models showed very similar age-depth relationships after calibration, only the first model which excluded just two dates has been used in the determination of C accumulation rates. However, C accumulation rates were also determined for the other models and showed very similar results, although these results are not presented.

Figure 5.5 shows the calculated rate of C accumulation against depth for the MH2 core, whilst in Figure 5.6, the same rates are plotted against age.

5.3.2 Reconstruction of past climatic change

5.3.2.1 Degree of peat humification

The humification profile for core MH2 is presented in Figure 5.7. The absorbance at the base of the core was very high (*ca.* 0.7) and declined rapidly over the depths 50-70 cm. However, absorbance values remained between around 0.2 to 0.3 throughout the rest of the core. The main feature of the profile suggests a wet shift occurred between 50 and 70 cm.

5.3.2.2 Plant macro-fossil analysis

Figure 5.8 displays the macro-fossil profile for core MH2, showing the peat components, *Sphagnum* leaf counts and number of observed charcoal fragments for each sample.

The base of the core (55 cm-80 cm; Figure 5.8) contained a high proportion of UOM. Above this was a zone of rapid change as UOM declined and Identifiable *Sphagnum* (predominantly *S.* s. Cuspidata) greatly increased; this apparently represents a shift to wetter mire surface conditions. The top part of the core showed little change in UOM, although a notable feature was the change in species composition of *Sphagnum*: Between 37 cm and 53 cm *S.* s. Cuspidata dominated, but at 35 cm this was largely replaced by *S. papillosum*, which is less tolerant of extremely waterlogged conditions (Stoneman, 1993), indicating a drier mire surface. Subsequently the mire appears to have become even drier because at the top of the core *S. papillosum* was superseded by *S. magellanicum*, which is associated with even drier mire conditions (Stoneman, 1993).

5.3.2.3 Rhizopod analysis

Figure 5.9 shows the results of the Rhizopod analyses providing the profile of species composition for testate amoebae shells and other identified faunal parts for core MH2.

Hyalosphehenia subflava is present in large proportions in the base of the core, but is not represented in the upper parts; this species is considered an indicator of dry mire conditions (Mauquoy, 1997) and supports the inferred dry conditions implied by both the plant macro-fossils and degree of humification. Supporting evidence for the wet shift inferred

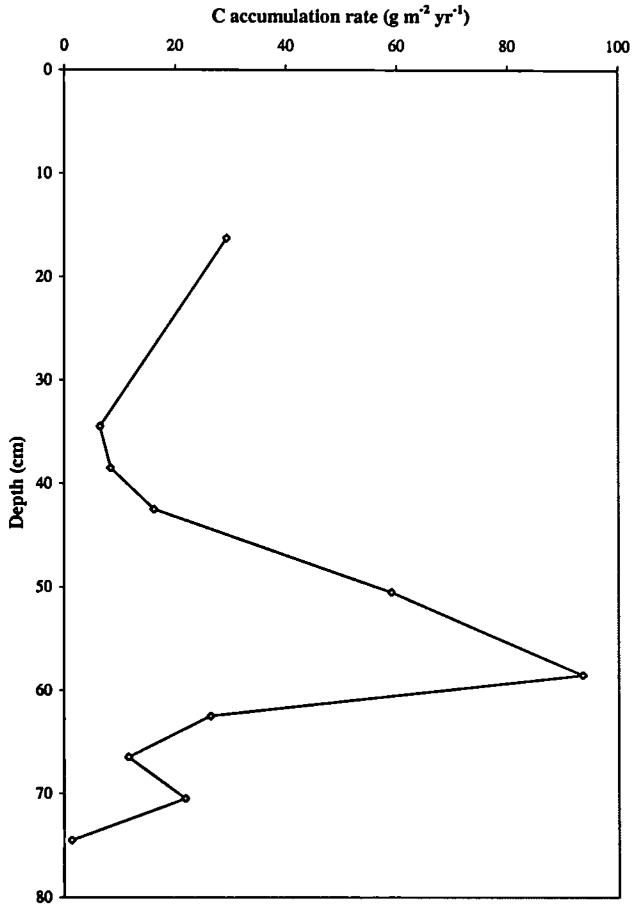


Figure 5.5 Plot showing rate of C accumulation in core MH2 with depth,

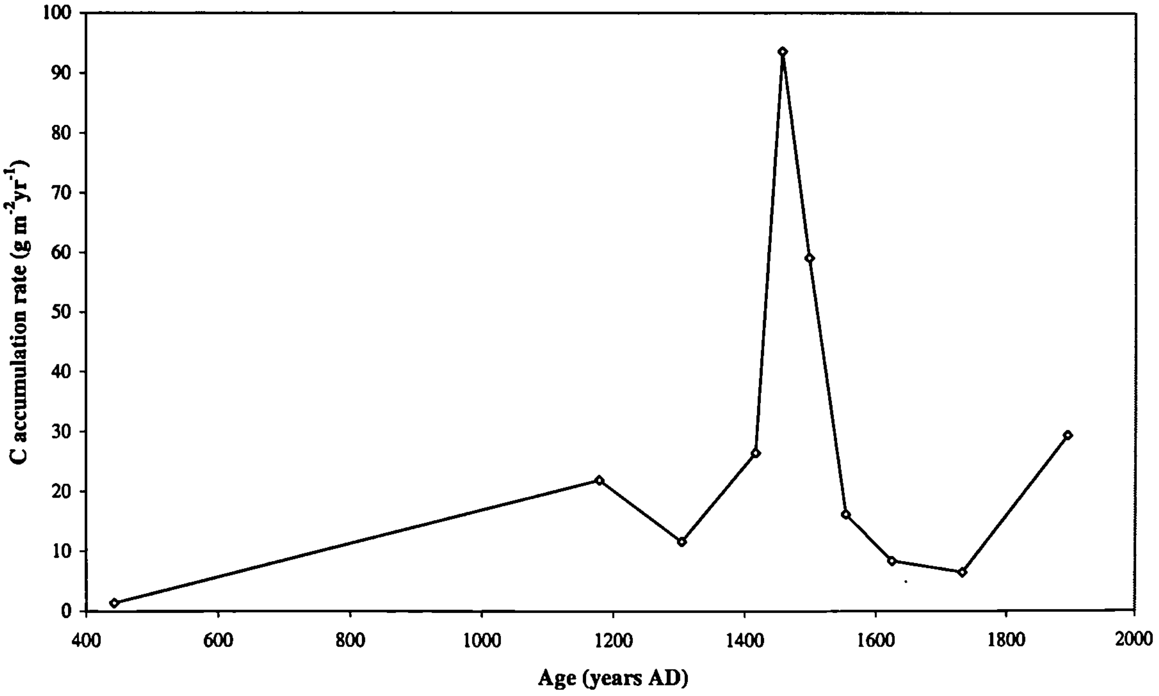


Figure 5.6 Rates of C accumulation against time for core MH2.

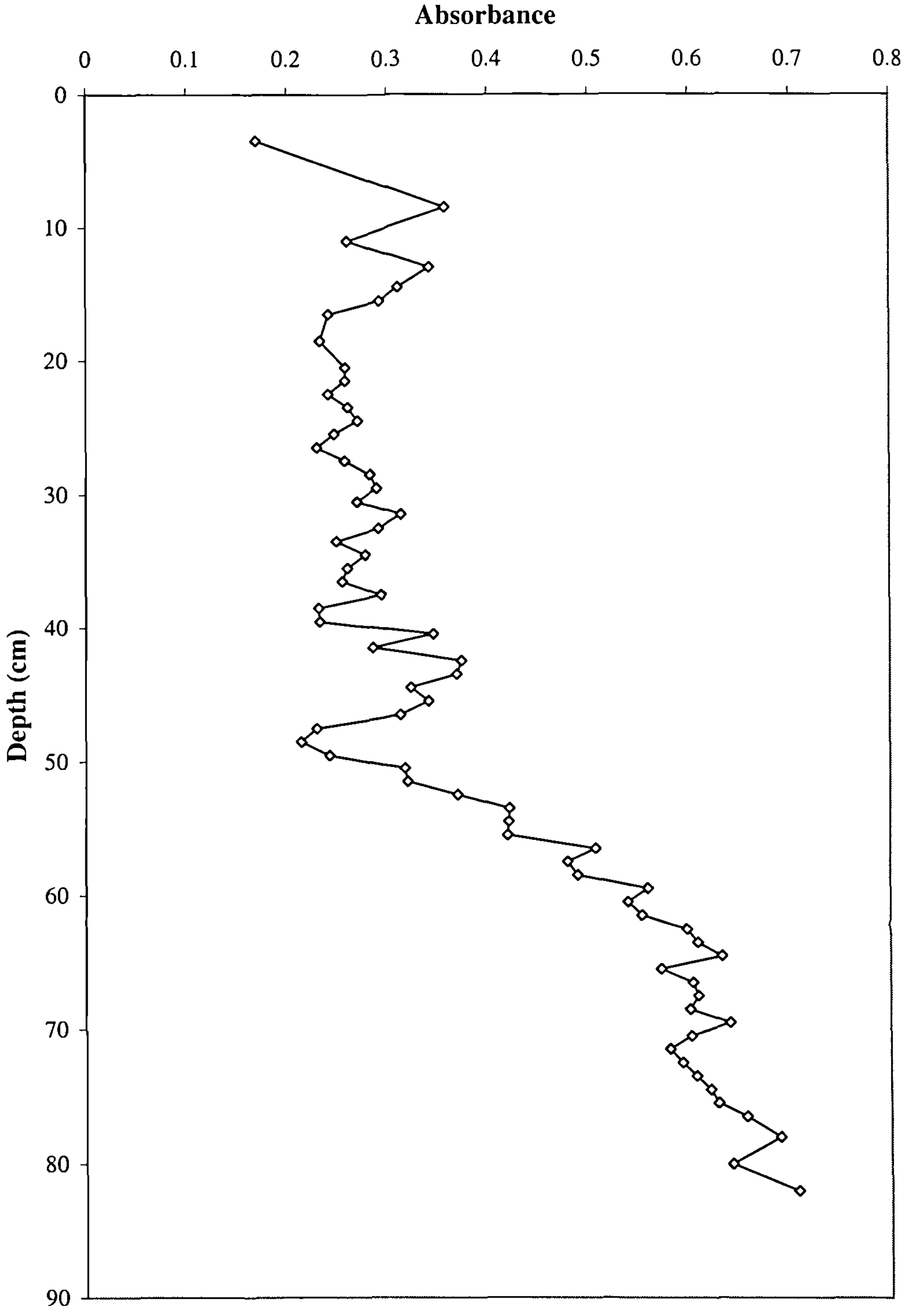
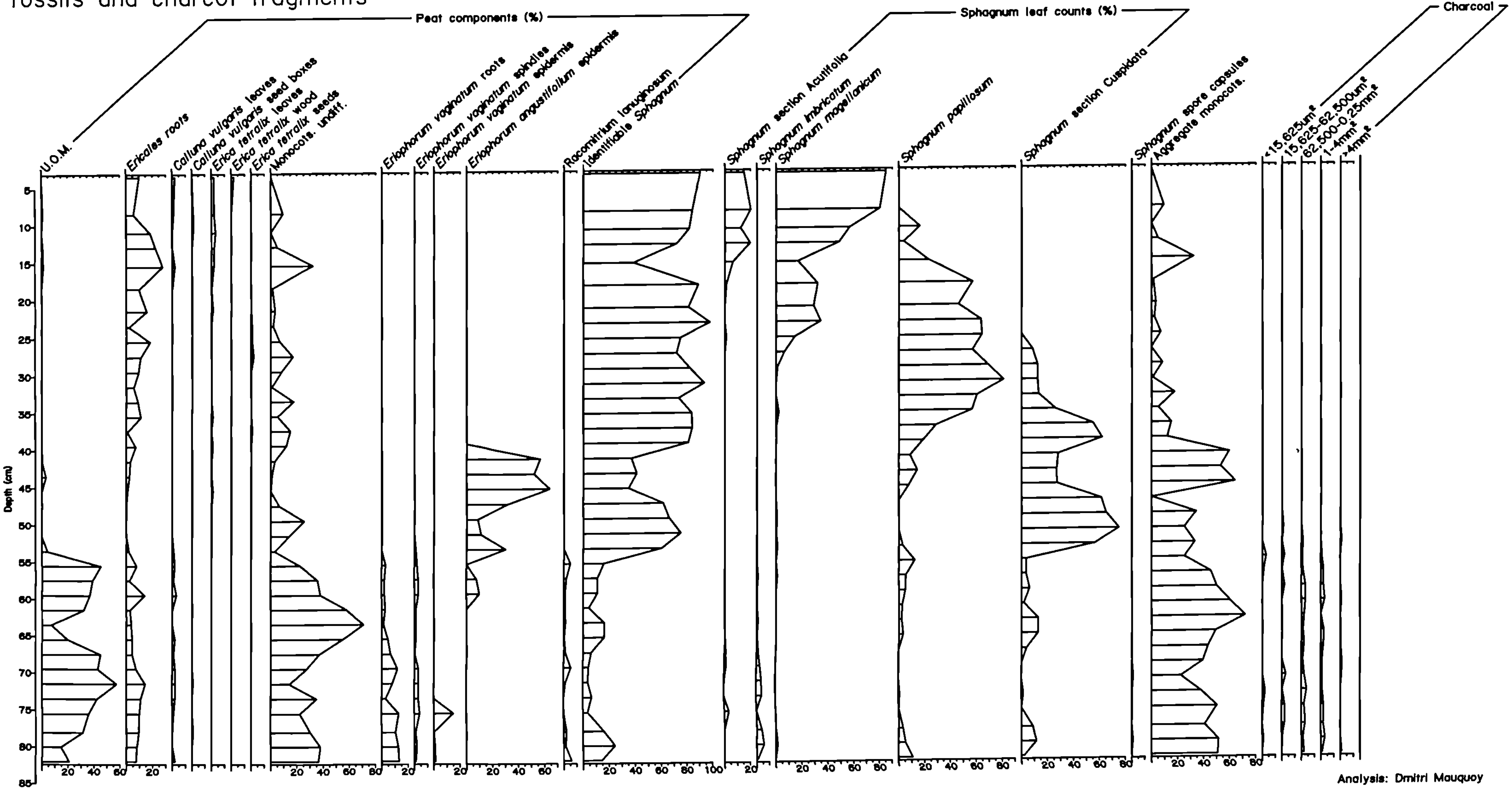


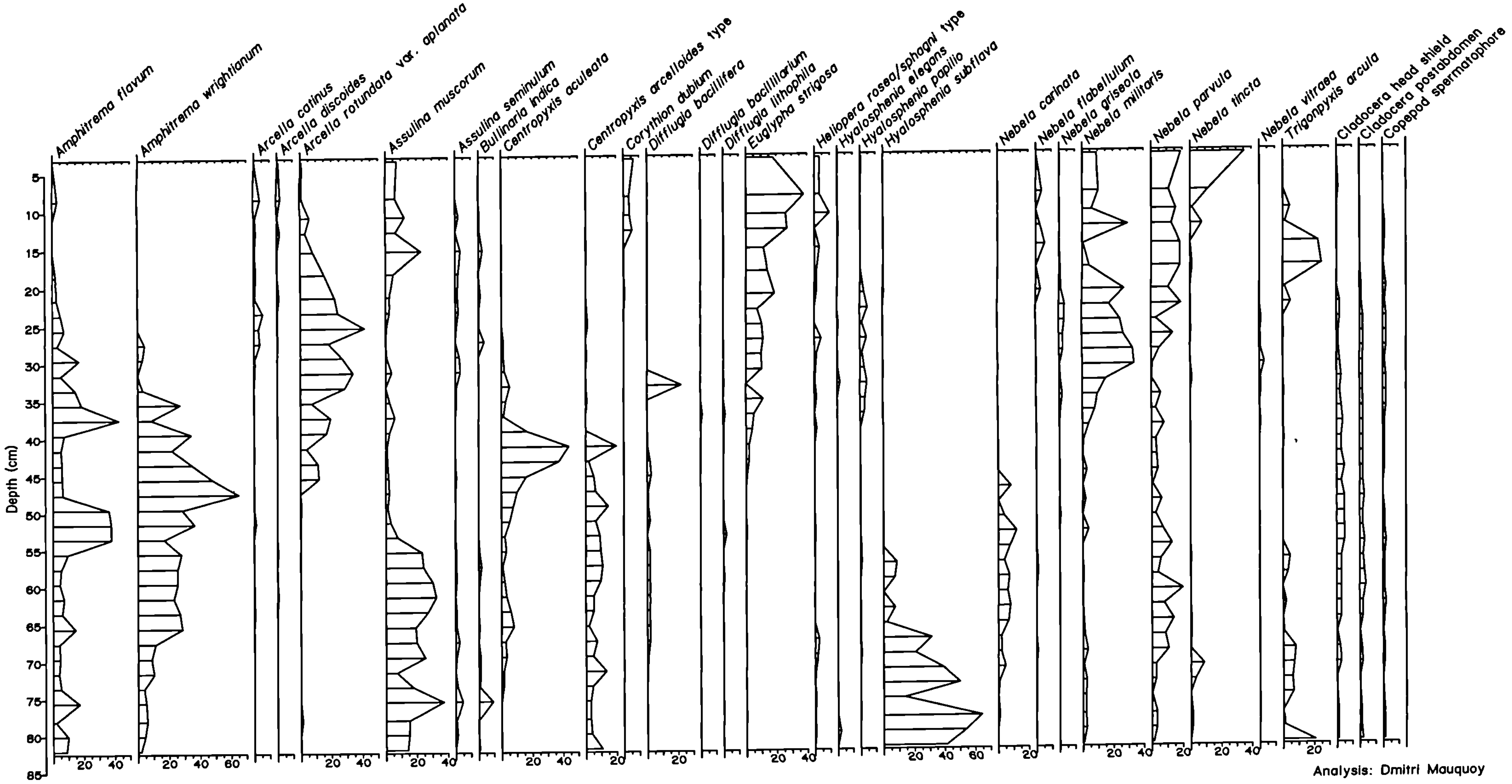
Figure 5.7 Profile of degree of humification (absorbance at 540 nm) against depth for core MH2

Figure 5.8 Profile description of core MH2 showing plant macro fossils and charcol fragments



Analysis: Dmitri Mauquoy

Figure 5.9 Profile description of core MH2 showing testate amoebae



Analysis: Dmitri Mauquoy

by the plant macro-fossils and degree of humification is also present in the testate amoebae profile since, at approximately the same depth, *Amphitrema flavum* and *Amphitrema wrightianum* increase in concentration; these species are typical of high moisture conditions on the mire surface. Further description of the testate amoebae profiles is provided by Mauquoy (1997).

5.3.2.4 PCA of plant macro-fossils and testate amoebae

Figure 5.10 shows a plot of the first and second principal components derived from the plant macro-fossil results. This graph shows the clustering of sample depths with similar plant macro-fossil assemblages. Based on the known surface wetness preferences of modern plant species on mires, it appears likely that the second principal component (PRIN2) refers to a mire surface wetness gradient; *e.g.* sample depths A-D (0-14 cm) refer to the surface samples with high concentrations of *S. magellanicum.* However, more negative values of PRIN2 indicate samples which have a greater proportion of species preferring much wetter surface mire conditions (*e.g.* samples Q to V (41-51 cm) contain large amounts of *S.* s. Cuspidata). The contribution of individual species to the principal components is provided in the Appendix (Section 10.5).

Figure 5.11 provides a similar plot for the first and second principal components derived for the testate amoebae results. In this plot, the first principal component axis (PRIN1) can be related to a surface wetness score; samples with a high value represent drier mire conditions (*e.g.* samples A to D contain dry indicators such as *Assulina muscorum* and *Nebela flabellulum;* Mauquoy, 1997) while samples with very low values for PRIN1 have high proportions of testate amoebae species with preference for wetter mire conditions (*e.g.* samples T to X contain large amounts of *Amphitrema flavum* and *Amphitrema wrightianum,* indicative of very high moisture contents; Mauquoy, 1997). The contribution of individual rhizopod species to the principal components is provided in the Appendix (Section 10.5).

Figure 5.12 is a plot of the wetness scores derived from the PCA of the plant macro-fossils and testate amoebae results, showing variation in wetness score with depth. The wetness profiles derived from the different analyses both show a wet period in the middle depths of the core, surrounded above and below by peat formed under drier conditions.

5.3.3 Measurement of water table depth at the coring site

Figure 5.13 shows the manual readings of water table depth for the dip well located at the coring site on Shaft Hill. The water table stayed within *ca.* 10 cm of the surface for most of the year.

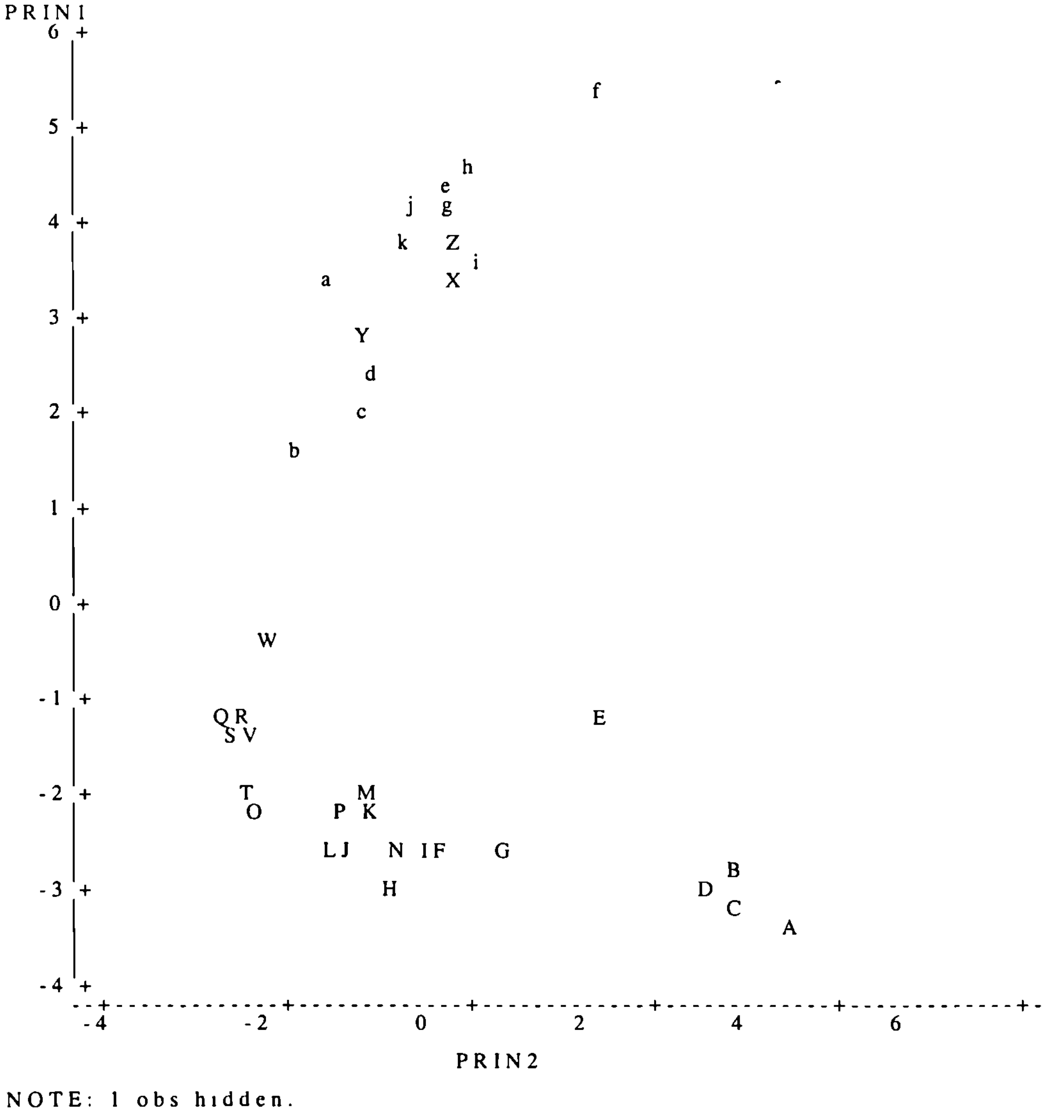


Figure 5.10 Principal component plot of plant macro-fossil results from MH2. Plot of principal component 1 against principal component 2. Letters refer to sample depths; A to Z = 0 to 60 cm and a to k = 60 to 82 cm (see Appendix, Section 10.5 for complete key).

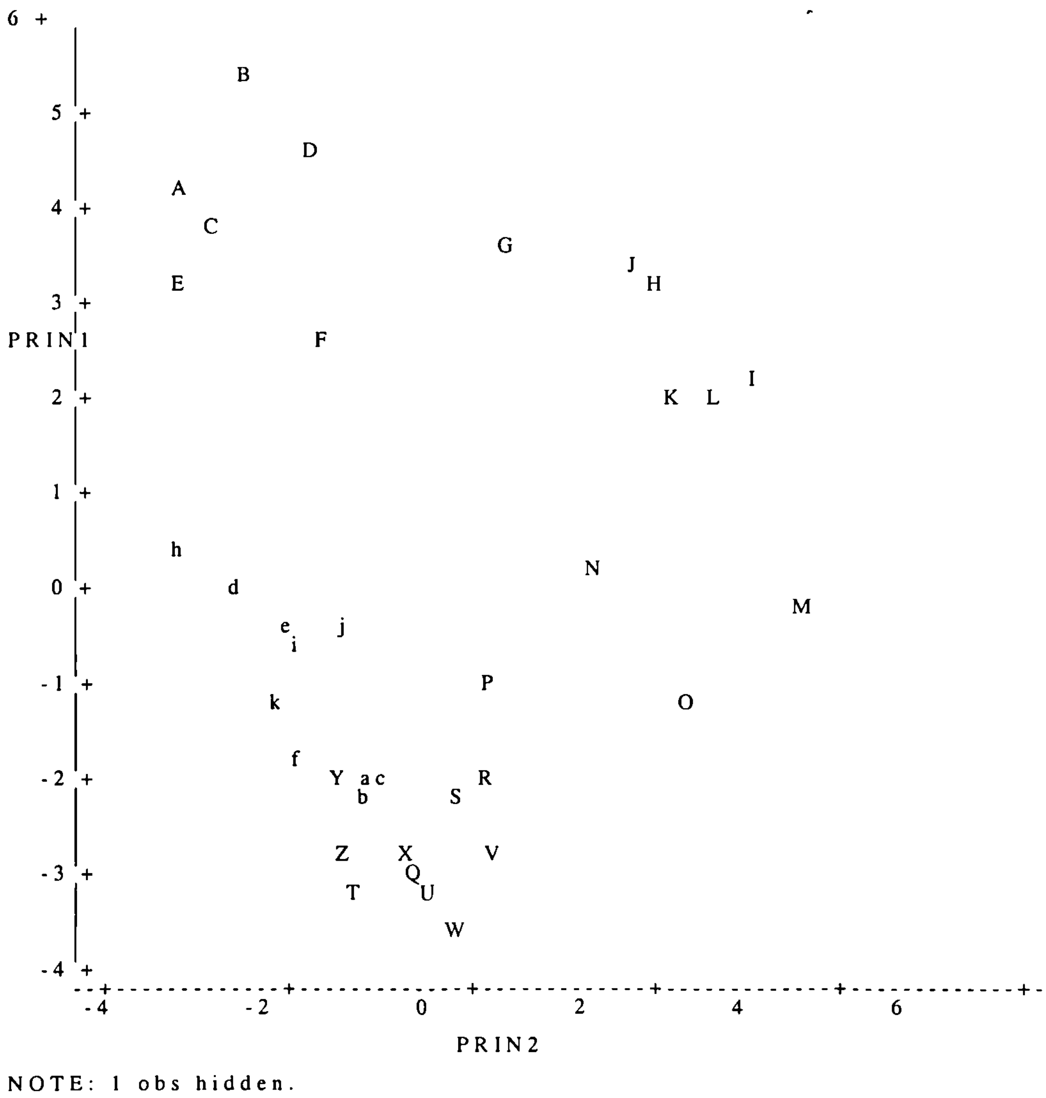


Figure 5.11 Principal component plot of testate amoebae results from MH2. Plot of principal component 1 against principal component 2. Letters refer to sample depths; A to Z = 0 to 60 cm and a to k = 60 to 82 cm (see Appendix, Section 10.5 for complete key).

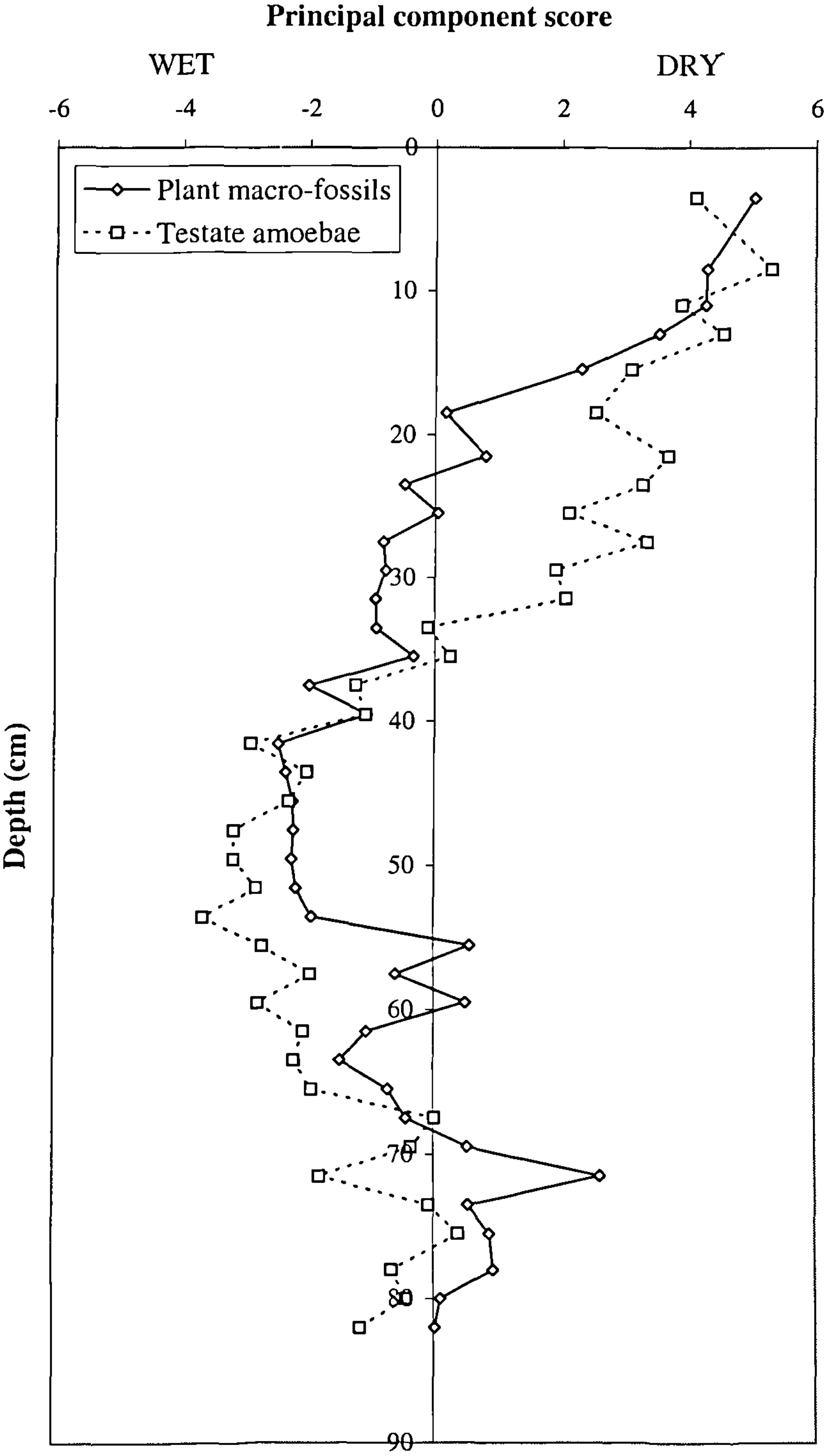


Figure 5.12 Plot of PCA wetness score against depth for core MH2, showing results for plant macro-fossils and testate amoebae

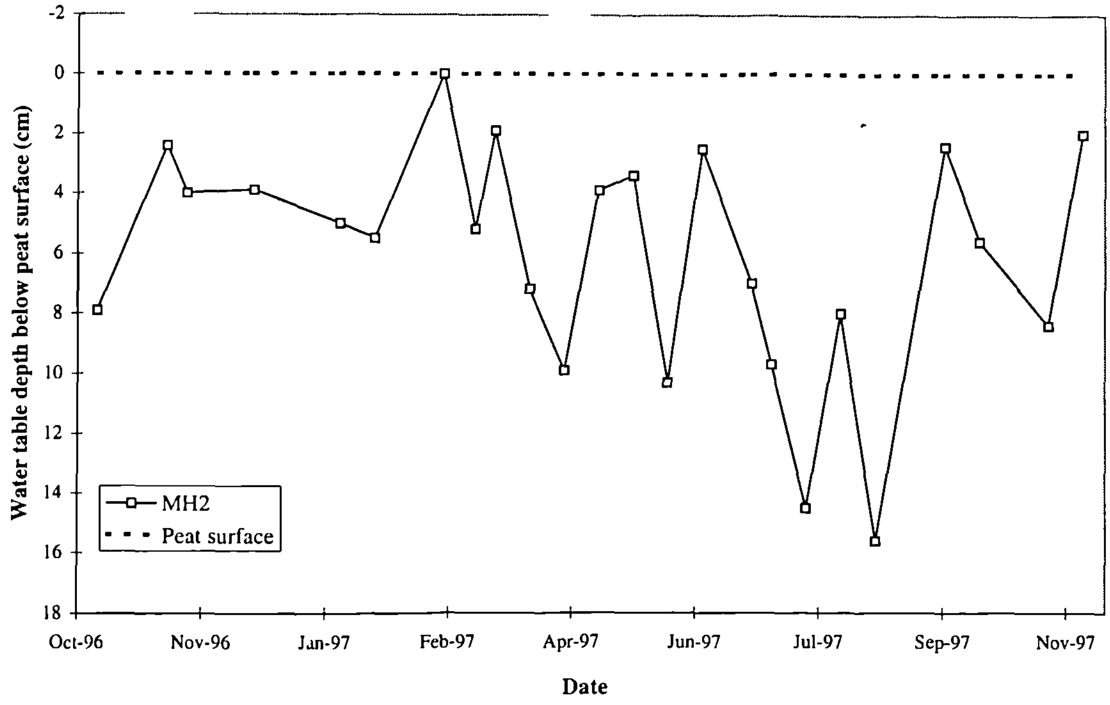


Figure 5.13 Depth of water table at the MH2 coring site, Shaft Hill, between October, 1996 and November, 1997

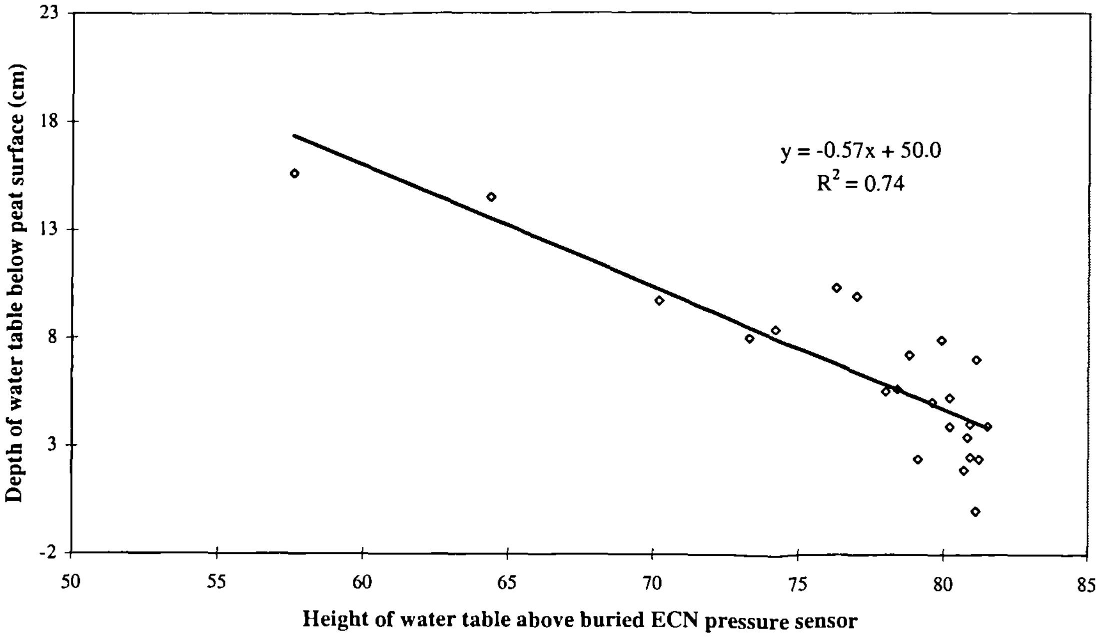


Figure 5.14 Scatter plot showing the correlation between ECN water table logger and water table depths at the MH2 coring site

Figure 5.14 shows a scatter plot of the water table depth of the MH2 dip well plotted against the recorded pressure of the ECN water table data logger. A linear regression has been derived for the correlation and used to predict the water table depth at the coring site from the ECN logger.

Table 5.5 summarises the water table measurements at the MH2 coring site, providing mean, minimum and maximum measured water table depths. Since the maximum water table depth measured by the ECN logger was recorded for July, 1995, the approximate water table depth has been predicted for this period, based on the regression provided in Figure 5.14. This therefore provides an estimate of the maximum water table depth for the coring site since 1994 and is also shown in Table 5.5.

Table 5.5 Mean, minimum and maximum water table depths at the Shaft Hill coring location

Water table depth below surface (cm)	MH2
Mean water table depth	6
Minimum depth	0
Maximum depth	16
Predicted maximum depth 1994-1997	28

5.4 Discussion

5.4.1 Accumulation rates of C

Profiles of C content, SCPs, *Pinus* concentrations and rates of C accumulation have been derived for the peat core investigated in the present study. Before interpreting the results of these analyses, it is important to consider the possible inaccuracies involved in calculating rates of C accumulation. There are essentially two areas of uncertainty: i) in the measurement of C stored in the peat profile, and ii) in the chronology constructed for the profile.

The profile of C content in the core was determined by measuring the bulk density and %LOI of samples from each depth. Considerable care was taken in measuring bulk density which, because dry weight can be reliably determined, required accurate measurement of sample volume. The displacement method used to determine volume of the peat samples is believed to have provided repeatable measurement of sample volume since replicate samples showed consistent values. Furthermore, a strong correlation was found between the wet weight of samples and sample volume (Figure 5.15); since the moisture content of samples was > 94 %, most of the weight, and volume would be water. Water has a density of 1 g cm^{-3} which is not very different to estimates of the dry density of peat (*ca.* 1.6

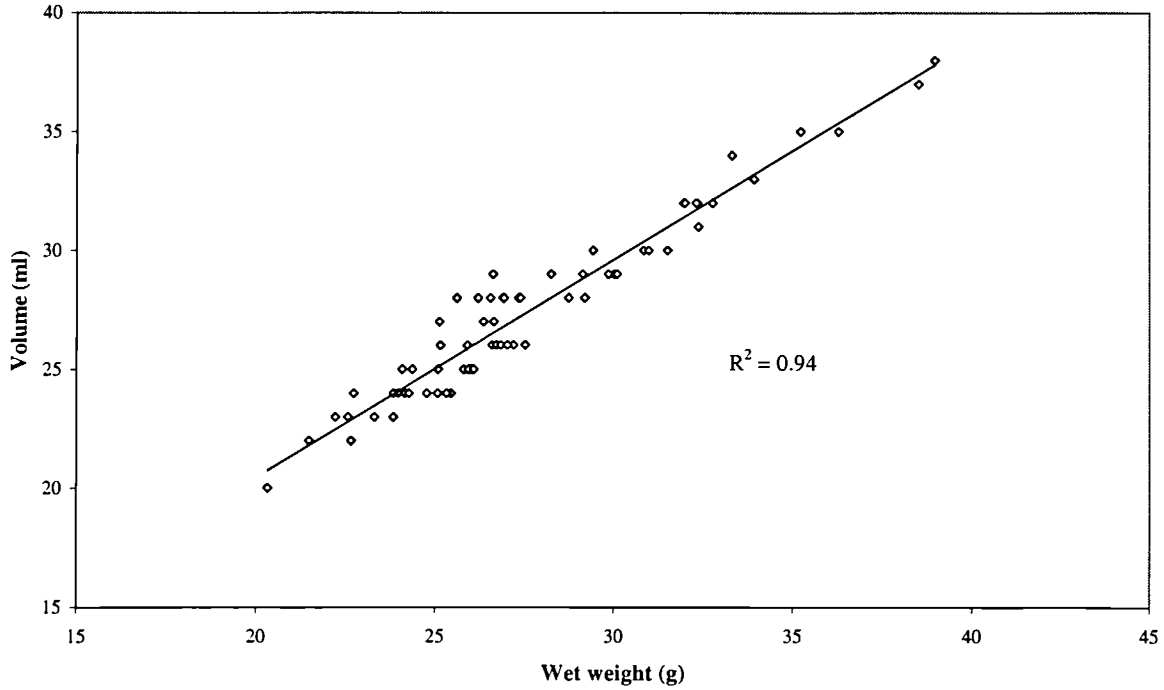


Figure 5.15 Scatter plot showing correlation between volume and wet weight for samples from core MH2

g cm^{-3}) and therefore peat samples with high moisture contents have a wet volume of similar magnitude to wet weight (see Chapter 8).

The C concentration of peat was estimated by measuring the %LOI of samples and converting the organic fraction to C using the regression equation provided by Bol *et al.* (in press; equation 3.9). As discussed in previous sections, there is likely to be a small source of error in using this equation although the approach should provide a more accurate estimate of C content than simply assuming a C concentration of 50 %, as many workers do (*e.g.* Tolonen and Turunen, 1996; Oldfield *et al.*, 1997). However, the %LOI for ombrotrophic peat samples ranged in the present study from 94.7 to 98.4 % corresponding to a C concentration range of 49.7 % to 51.6 % and confirming that a value of 50 % would have provided a good estimate for the C concentration of the peat samples.

Another source of uncertainty in the present investigation was non-horizontal stratigraphy present within the core. A large peat core was needed to provide enough peat for the wide range of analyses performed since the alternative approach of using thicker slices of peat would have reduced the temporal resolution of reconstructions. A non-horizontal stratigraphic boundary was observed in core MH2 and it is possible that peats of slightly different ages and characteristics may have been used for the different analyses.

The main area of uncertainty in the C accumulation rates is considered to be the chronology derived for the peat profile. The most robust chronologies are derived when a range of different and independent dating methods support the same chronology since they are unlikely to show consistent errors. The present study attempted to derive chronologies using several techniques including conventional bulk and AMS radiocarbon dating, records of SCPs, pollen dating and tephra analysis.

Tephra analysis

Very few tephra shards were found in core MH2 (Blackford, personal communication), and therefore, tephra analysis did not contribute to the construction of a chronology for the core. It was disappointing that no distinct tephra layers were found because the geochemical composition of shards could have been determined and matched to tephra layers with known dates (*e.g.* Dugmore *et al.*, 1995; Oldfield *et al.*, 1997). Tephra layers have been found in peats in the north of England (Pilcher and Hall, 1996) although the shards identified were dated to 2120±60 BP and 2310 cal. BC; the radiocarbon dates for MH2 suggest that the latter tephra layer is too old to be present in the MH2 core, while the former tephra layer would relate to the very bottom of the core where there may have been a hiatus in peat accumulation (see below). Very small quantities of tephra were present in the core (Blackford, personal communication) and, although these were too small to be geochemically identified, there occurrence may relate to known tephra layers, such as the AD 1510 tephra observed in Scotland (Dugmore *et al.*, 1995).

Pinus pollen

Mauquoy (1997) summarised the evidence for a rise in *Pinus* pollen concentrations associated with the plantation of Pine trees in northern England *ca.* AD 1800. However, the AD 1800 *Pinus* rise in core MH2 was defined at 57.5 cm depth by Mauquoy (1997), yet this depth is slightly below the AMS radiocarbon date for 56-57 cm of 435 BP (calibrated 1 sigma range AD 1425-AD 1515). Therefore, the *Pinus* rise in core MH2 does not appear to relate to the AD 1800 date used by Mauquoy (1997).

The concentration of pollen in peat deposits is related not only to the rate of deposition of pollen on the mire surface, but also the rate of peat accumulation. For example, if the rate of pollen deposition on a mire was constant over time, and the rate of peat accumulation varied, the quantity of pollen grains in a standard volume of peat would not be constant. The method used by Mauquoy (1997) to determine *Pinus* pollen concentrations in core MH2 involved taking a known volume of peat and counting the *Pinus* pollen grains in this sample. However, large variations in the bulk density of the peat have been shown for the core (Figure 5.2) and peat accumulation rates seem also to have varied greatly with time (see below). Therefore, it is likely that the *Pinus* rise detected by Mauquoy (1997) in core MH2 is at least partly a result of changes in the rate of peat accumulation or bulk density of the peat. Consequently, although the method used by Mauquoy (1997) is rapid, it is not believed to be useful for providing a reliable chronological marker.

The use of the *Pinus* rise for dating may be more successful if other pollen grains were identified and counted, as in traditional pollen analysis approaches. Such an approach would have allowed the concentration of *Pinus* relative to other species, to be determined. Therefore, variations in bulk density and peat accumulation rate would affect all pollen grains equally. A rise in *Pinus* pollen relative to other species would then indicate an increase in the true amount of *Pinus* pollen being deposited on the mire surface and could, therefore, indicate any changes in the regional abundance of this species.

Spheroidal carbonaceous particles

The record of SCPs in core MH2 showed a marked increase in concentration in the surface samples of the profile, reflecting increasing production of the particles following industrialisation. However, like the concentrations of *Pinus* pollen in the profile, the quantity of SCPs will vary with the rate of peat accumulation and bulk density, although, bulk density only varied slightly over the depth range where SCPs were present and changes in peat accumulation rate are unlikely to have caused the increased SCP concentrations (see Section 4.4.2).

The take-off in SCP profiles is usually used as a chronological marker although there are uncertainties in the date assumed for the take-off (see Section 4.4.2.); Rose (1990) dated SCP take-offs in Scottish lake sediments using ^{210}Pb to *ca.* AD 1940, yet, Gedye (1998) dated take-offs in peat profiles from south Cumbria to *ca.* AD 1800. Since Moor House is located only *ca.* 50 km (30 miles) from the south Cumbrian sites used by Gedye (1998), and industrial activity seems to have occurred here quite early due to lead mining (Welch, 1974), it is thought that the SCP take-off in the Moor House core reflects a pre- AD 1940 date. Furthermore, the depth of the take-off in MH2, would mean that peat accumulation would need to have been extremely rapid for the take-off to be buried so deep in the peat profile (*ca.* 1.6 yr^{-1} cm^{-1}). The top 10 cm of the core shows reduced SCP concentrations which may indicate reduced emissions of these particles over the last *ca.* 15-20 years.

However, without an independent date for the SCP take-offs (derived using *e.g.* ^{210}Pb; Appleby *et al.*, 1997) the SCP profile derived for the Moor House core does not provide a very useful chronological reference point, although it is useful for correlating between cores (see Chapter 4).

Radiocarbon dating

Two conventional radiocarbon determinations were performed on core MH2, however, the results of these analyses provide only approximate ages for these depths since bulk radiocarbon dates suffer from inaccuracies caused by root contamination and possible 'reservoir' effects (Kilian *et al.*, 1995). The conventional radiocarbon dates have therefore not been used for deriving C accumulation rates.

AMS radiocarbon dating is believed to be a more accurate method for dating peat profiles since individual above-ground plant fragments can be dated (Kilian *et al.*, 1995) and these can be hand-picked and cleaned to reduce the possibility of contamination with roots. A further advantage of AMS dating is that, since only very small amounts of organic material are required, samples can be obtained from very narrow peat depths. However, despite the increased accuracy of AMS radiocarbon dating several problems were encountered in the current study which raised uncertainties about the age-depth profiles derived.

The main concern with the AMS radiocarbon age profile concerned the depths 48-49 cm and 52-53 cm. AMS radiocarbon ages for these depths were considerably younger than samples lying directly above, which should theoretically not occur on an undisturbed stratified peat profile. No evidence for disturbance of the peat profile was found either in core MH2 or at the coring site and therefore contamination of the samples with younger material seems the most likely explanation. However, during coring, slicing, storage and hand-picking of samples for AMS dating, extreme care was taken at each stage to avoid contamination of samples. Furthermore, the suspect dates are adjacent in the peat profile and other samples appear not to have been contaminated. Consequently, contamination seems unlikely and it is not currently easy to explain why these two dates are apparently too young. Although Wohlfarth *et al.* (1998) have recently suggested that fungi or micro-organisms may contaminate samples with modern ^{14}C, making them apparently younger, the two samples from MH2 were treated identically to the other samples which did not yield apparently incorrect dates.

A second concern with the results of the AMS dating of samples was the date of the deepest sample analysed. The sample from 76-77 cm had an AMS date (2200 BP) which was considerably older than the sample dated from only 4 cm above (72-73 cm; 865 BP). However, the very old date for depth 76-77 cm is not considered erroneous (although this cannot be ruled out) because the duplicated conventional bulk radiocarbon date made for this depth (see below) was also considerably older than the AMS date 4 cm above (1,675 BP). Therefore, there appears either to have been a very slow rate of peat accumulation between the dated samples at 72-73 cm and 76-77 cm or a break in peat accumulation.

The repeatability of the AMS radiocarbon dating of samples was assessed by dating two samples from the same depth. The resulting ages were very close (435±45 BP and 510±55 BP) and indicate that AMS dating of plant fragments gave repeatable estimates of the age for the peat slices. Since the error term associated with the dates is 1 sigma, and the error ranges of the two samples overlap, the two ages can't be considered different.

One peat slice was dated both with AMS and conventional bulk radiocarbon dating. The AMS date for the sample was 2200 BP while the bulk date was 1675 BP. The difference in age may have been because the material used for the conventional bulk radiocarbon date was unsorted, whereas the AMS date was obtained from hand-picked, above-ground plant fragments. Consequently, roots from plants growing in the peat after it had been formed may have been present within the bulk sample, producing a younger radiocarbon age (Kilian *et al.*, 1995).

Due to the problems associated with the AMS dating of samples, three different age models were developed to provide a chronology for core MH2. Calibration (OxCal) using all dates resulted in an unlikely age-depth relationship since the two suspect dates were overlain by older peat. Therefore, the two apparently young dates were not used to construct age-depth profiles (age model 1), whilst a further sample was excluded for the other two models.

However, it is acknowledged that other interpretations of the radiocarbon dates are possible which may result in different age-depth relationships for the MH2 core. Different age-depth models may alter the estimates of C accumulation rate discussed below, and therefore, the results of the present study must be treated with caution. Further investigation (*e.g.* pollen analysis) is currently being undertaken to improve the chronology of the MH2 core.

C accumulation rates

The profiles of C accumulation derived using the three age models resulted in very similar trends of C accumulation with depth and time, and therefore only the results of age model 1 have been presented.

Lowest rates of C accumulation occurred at the base of the core which may have been a result of a break in peat formation. C accumulation rates above this depth ranged from less than 10 g C m^{-2} yr^{-1} to greater than 80 g m^{-2} yr^{-1}, with an average rate of 27 g m^{-2} yr^{-1}.

The mean value of C accumulation rate is very similar to published values from elsewhere: Gorham (1991) estimated C accumulation rates for northern peatlands as 29 g m^{-2} yr^{-1}; Tolonen and Turunen (1996) calculated mean C accumulation rates of 26.1 g m^{-2} yr^{-1} for Finnish mires; Oldfield *et al.* (1997) suggested a range of 10 to 40 g m^{-2} yr^{-1} for recent C accumulation in a Swedish mire.

The range of C accumulation values obtained in the present study is not unusual: Tolonen and Turunen (1996) found C accumulation rates in individual peat cores from Finnish mires to range between 2.8 to 88.6 g m^{-2} yr^{-1}. Since most studies have calculated C accumulation rates over relatively large time periods, the values derived this way mask any short term changes in peat accumulation rate.

The rate of C accumulation increased in the top of core MH2, however, this does not reflect a true increase in the long-term rate of C accumulation since this peat has not entirely passed into the catotelm. The acrotelm-catotelm boundary in the MH2 profile has been estimated from the measurement of water tables at the coring site as being at *ca.* 28 cm depth. All AMS radiocarbon dates used to provide C accumulation rates were located below this depth and therefore were taken from within the catotelm. All the estimates of C accumulation in the profile below *ca.* 30 cm (*i.e.* all but the most recent rate) provide estimates of C accumulation rate in the catotelm, and, since decomposition is very slow within the catotelm, the rates of C accumulation can be directly compared to indicate how they have changed with time. Accumulation rates of C in the lower part of the peat profile will be reduced compared to rates near the surface due to some decay in the catotelm, yet, since the time interval covering the rates is only *ca.* one millennium the influence of catotelm decay on C accumulation rates may be considered negligible (Christen *et al.*, 1995).

The higher rate of C accumulation in the top of MH2, attributed to peat not yet having passed entirely into the catotelm, is predicted by Clymo's (1984) model of peat accumulation and has also been observed in a Swedish mire investigated by Oldfield *et al.* (1997).

5.4.2 Reconstruction of past climatic change

Tallis (1995) states that clear proxy-climate signals have been derived from analysis of the profiles of European raised mires. Other researchers have derived proxy-climate signals from blanket mires very similar to the mire investigated in the present study. However, despite this, investigation of the MH2 peat core and others from Moor House by Mauquoy (1997) led him to question whether these peat profiles were sensitive to climate change. Mauquoy (1997) did not have the AMS radiocarbon dates available to him as in the present study, and from a few conventional radiocarbon dates, the SCP profile and *Pinus* pollen concentration profiles, Mauquoy (1997) suspected a large hiatus in the Moor House cores around the medieval period. Given the chronological evidence available to him, this was a reasonable interpretation to make and helped to explain why the Moor House cores did not appear to contain the same climate signals derived from nearby lowland raised mires by Mauquoy (1997).

However, in the present study, the AMS radiocarbon dates provide strong evidence against a hiatus in peat formation over the last millennium (*cf.* Mauquoy, 1997). Furthermore, the validity of the method used to determine the *Pinus* rise has also been questioned (see above). Uncertainties about the true age of the SCP take-off have also been raised (see above). Consequently, re-analysis of the proxy-climate records from Moor House has been undertaken in the present study.

Plant macro-fossils

The base of core MH2 provides both indicators of dry and wet mire surface conditions. Although there were remains of *Racomitrium lanuginosum* and a high proportion of %UOM, which suggest a high degree of decomposition (and therefore aerobic conditions), important quantities of wet indicators, such as *S.* s. Cuspidata, were also present.

A wet shift was the most distinctive feature of the plant macro-fossil record and was dated to *ca.* AD 1500. However, if the macro-fossils do indeed provide a proxy-record of past climate change, this change in the profile should indicate a major climatic event in the last millennium. There is now a large amount of evidence from various historical and proxy sources for a major climatic deterioration termed the 'Little Ice Age' (LIA) between approximately the fifteenth and nineteenth centuries; Bradley and Jones (1993) found from historical, tree-ring and ice core evidence that the coldest period in the last 560 years was between AD 1570 and AD 1730; Chambers *et al.* (1997) found increased peat humification in a Scottish blanket mire commencing *ca.* AD 1420; Mauquoy (1997) found similar evidence of climatic deterioration from macro-fossil studies of lowland mires in north Cumbria.

The upper part of core MH2 showed a reduced mire surface wetness because *Sphagnum* species typical of drier mire conditions became more dominant. This is consistent with other records for a climatic amelioration after the LIA (Bradley and Jones, 1993; Mann *et al.*, 1998).

Therefore, it seems likely that the plant macro-fossil analysis of core MH2 has revealed a palaeoclimate signal, suggesting the major climate shifts of the last millennium.

Testate amoebae

The profile of testate amoebae composition shows strong variation throughout the core and the main wet-shift indicated by the plant macro-fossils is accompanied by clear changes in rhizopod species. The main wet shift, dated to *ca.* AD 1500, is strongly evident in the testate amoebae record and provides evidence that the rhizopod remains form a proxy-record of climate change. Like the plant macro-fossils, the testate amoebae profile shows a drier climatic trend towards the top of the core with species more representative of drier mire conditions being present.

Changes in testate amoebae composition in peat profiles have been found to be strongly related to water tables with analysis of fossil assemblages in peat profiles being used to reconstruct past water tables (Woodland *et al.*, 1998). Therefore, the proxy-climate signal provided by the testate amoebae may in fact be a record of past water table depth and, since water tables are strongly related to climate, provides a proxy-climate signal.

Degree of humification

Absorbance of samples in the profile of core MH2 showed a major decline between 50 and 55 cm depth. Below this depth the absorbance was relatively high whilst, above it, absorbance remained low and gradually increased towards the surface of the core. The degree of humification down the profile did not provide the same detailed evidence for climatic change as did the plant macro-fossils and testate amoebae. However, the strong wet shift noted in the plant macro-fossils and rhizopods, dated to *ca.* AD 1500 and attributed to the LIA, was clearly represented in the absorbance profile of MH2.

The increase in absorbance towards the surface of MH2 is consistent with drier mire conditions and the climatic amelioration suggested by the other analyses in the current study, and from other published sources. Humification would be expected to decrease towards the surface as peat is still within the acrotelm and decomposing at a rapid rate prior to reaching the catotelm.

Proxy-climate record from PCA results

The three methods for deriving proxy-climate records for the Moor House cores showed synchronous shifts in climate for the same depths. Additionally, PCA of the plant macro-fossil and testate amoebae data has been used to derive quantitative estimates of mire surface wetness, following the reported successful use of similar multivariate techniques for analysing macro-fossil results (Stoneman, 1993; Barber *et al.*, 1994; Mauquoy, 1997).

The PCA of the plant macro-fossil and testate amoebae results show broadly similar trends (Figure 5.12) although there were important differences. Firstly, the PCA of the plant macro-fossils suggested that the second principal component was responding to mire surface wetness. This differed to the testate amoebae where the primary axis represented a wetness gradient. Therefore, this may suggest that the testate amoebae are more sensitive to changes in mire surface wetness than the plant macro-fossils and, therefore, that testate amoebae may provide a more sensitive proxy-record of past climate change.

The testate amoebae surface wetness profile showed an earlier and more gradual shift to wetter conditions than the plant macro-fossils, at a date of *ca.* AD 1400. This is not inconsistent with recent estimates for the start of the LIA; Chambers *et al.* (1997) found the LIA apparently began at Talla Moss, a blanket bog site in southern Scotland at *ca.* AD 1410 (they note that Lamb (1995) now regards the start of the LIA as AD 1420). Mann *et al.* (1998) show a large decrease in northern hemisphere temperature in the mid 15th century.

Both the plant macro-fossils and testate amoebae PCA results suggested greatest mire wetness between 40 and 50 cm in the MH2 peat core. This represents approximately AD 1500 to AD 1600, and overlaps with the period of minimum temperatures derived from various sources by Bradley and Jones (1993; AD 1570 and AD 1730).

The plant macro-fossils and testate amoebae 'wetness' scores both suggest drier mire conditions towards the surface of the core, although the change appears more rapid in the testate amoebae record.

Mean wetness scores were calculated by averaging the scores for samples between radiocarbon dates, for both the plant-macro-fossil and testate amoebae results. The plant macro-fossil were found to be significantly ($p<0.05$) correlated with mean northern hemisphere temperatures derived from proxy sources by Mann *et al.* (1998), for the same time periods. Mean wetness scores increased with northern hemisphere temperature, although unfortunately, the temperature record of Mann *et al.* (1998) only extends from AD 1400 and only five points were available for comparison.

There was no correlation between the testate amoebae mean wetness score and the record of northern hemisphere temperature derived by Mann *et al.* (1998). However, testate amoebae composition may be more strongly influenced by water table depth than temperature and, because water tables are a product of both temperature and rainfall, a strong correlation between the testate amoebae wetness score and temperature may not exist due to variations in past rainfall. Furthermore, the long-term record of temperature for Moor House compares poorly with global mean instrumental records (Section 6.4.4) and, therefore, reconstructed northern hemisphere temperatures are unlikely to accurately reflect historical temperature changes at Moor House.

5.4.3 Impact of past climatic change on C accumulation rates

Figure 5.16 shows the records of C accumulation and proxy-climate for core MH2. The figure shows the rate of C accumulation, mean values of estimated mire surface wetness (average values between AMS dated depths) from plant macro-fossils and testate amoebae, and degree of peat humification. The variables are plotted against time over the last millennia. Even though the figures use mean values of surface wetness and absorbance, general trends in climate are represented by both the plant macro-fossils and testate amoebae profiles, though with some exceptions. All proxy-climate records suggest an increased mire surface wetness between *ca.* AD 1400 and AD 1600, at a time when C accumulation rates reached a distinct maximum.

Figure 5.17 shows a scatter-plot of the rate of C accumulation against the plant macro-fossil surface wetness score for the age model and it is clear from the figure that no correlation exists between the variables. This implies that either climate does not affect C accumulation rates in peats, that the plant macro-fossil composition of core MH2 does not provide a very sensitive proxy-climate signal, or that the C accumulation rates are inaccurate.

However, the scatter plot displayed in Figure 5.18 shows that C accumulation rates are significantly correlated with the testate amoebae surface wetness scores. A significant negative correlation was found ($p<0.05$) suggesting that C accumulation in the MH2 profile has been greater under conditions of increased mire surface wetness, and that reduced C accumulation occurred when the mire surface was dry.

The significant correlations derived between C accumulation rate and mean PCA wetness score for testate amoebae are more striking when considering the possible uncertainties in deriving the high resolution chronology for core MH2, and the uncertainty in the value of rhizopod analysis for a proxy-climate signal. However, several age models were investigated to construct the profile of C accumulation and all produced very similar results. Again, several methods were used to derive proxy-climate signals and, although there were

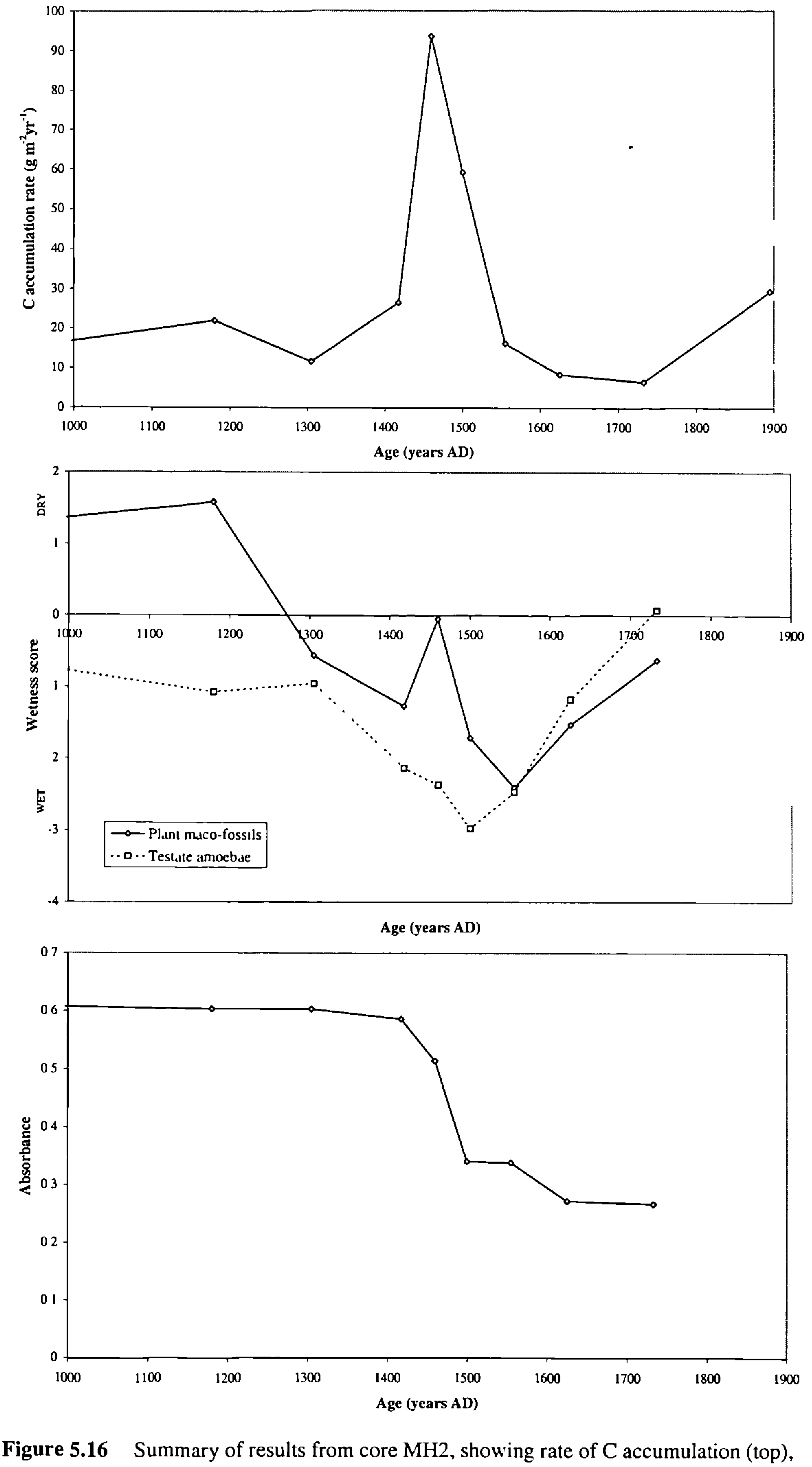


Figure 5.16 Summary of results from core MH2, showing rate of C accumulation (top), mean PCA wetness scores (middle) and mean degree of humification (bottom) against age.

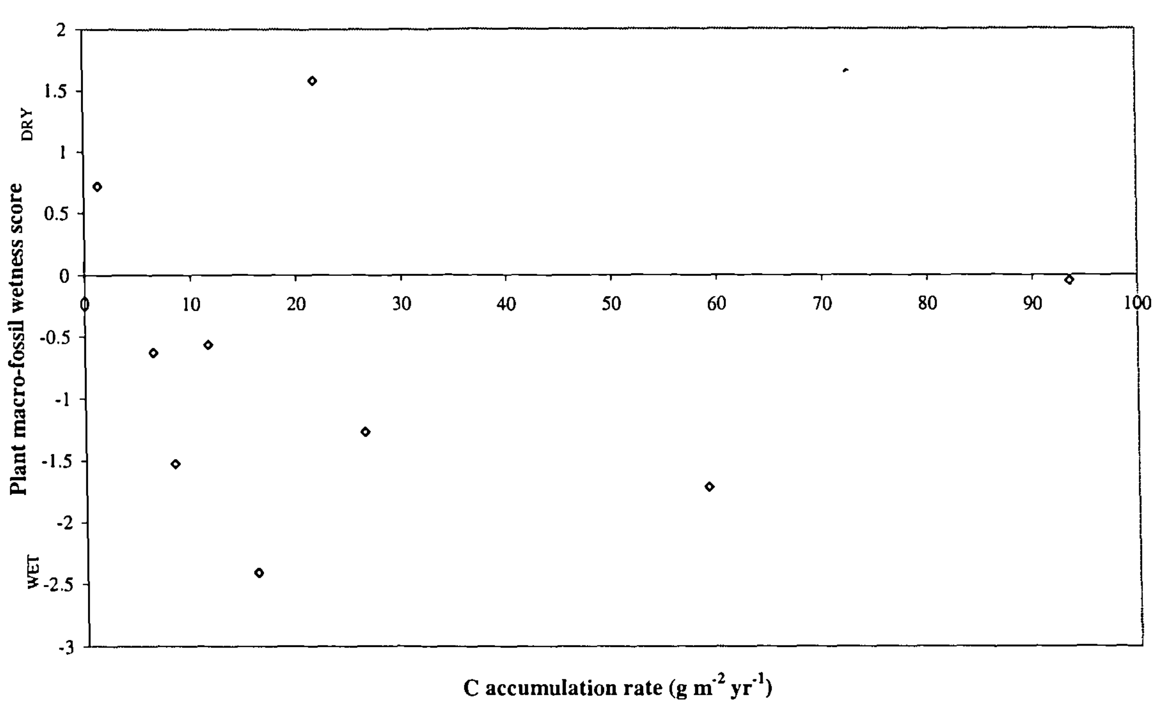


Figure 5.17 Scatter plot between rate of C accumulation and PCA wetness score from plant macro-fossils

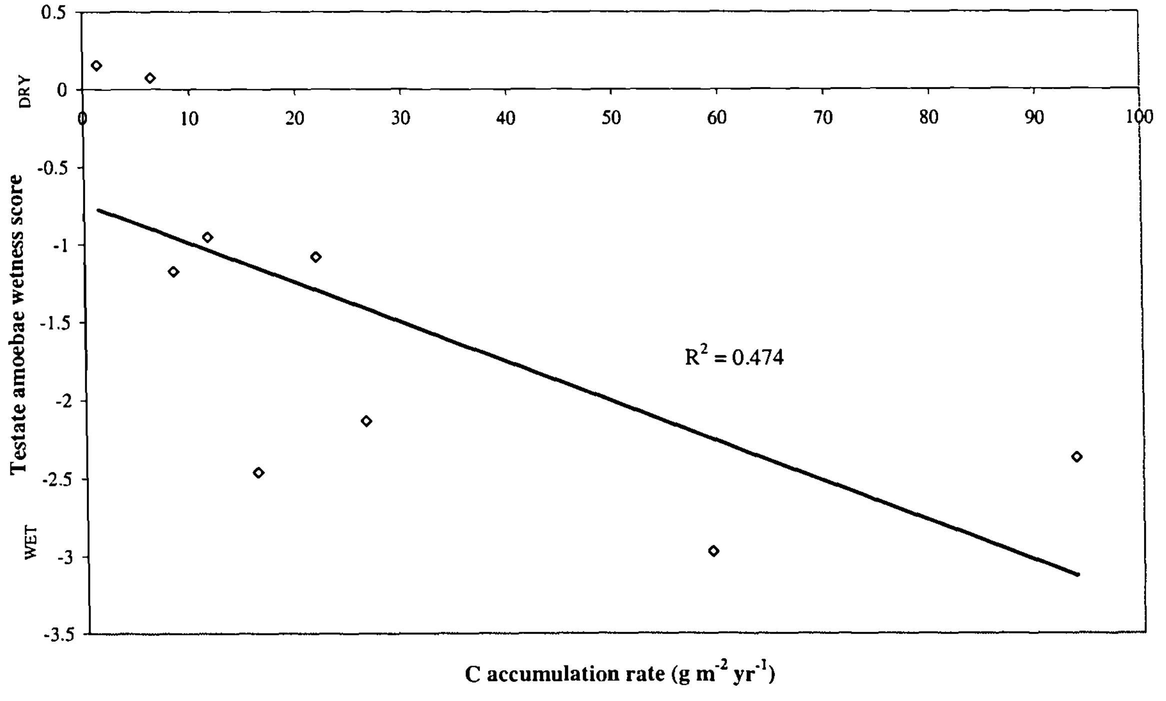


Figure 5.18 Scatter plot between rate of C accumulation and PCA wetness score from testate amoebae

some differences between the records derived, there were strong similarities in major trends which agreed with published records of climate from other sources. Furthermore, there is a theoretical basis for the observation that the C accumulation rate increased following a shift to wetter mire conditions; increased surface mire wetness would increase the proportion of time that the acrotelm experienced anaerobic conditions, therefore reducing initial decomposition rates. High moisture contents would also reduce the depth of the acrotelm-catotelm boundary, and decomposing plant material would enter the catotelm more quickly, increasing net peat accumulation rates.

However, reduced water table depths may reduce rates of NPP (Forrest and Smith, 1975), and therefore, initial inputs of C. Since the amount of C accumulated in the peat is a resultant of NPP and decomposition it may be that, following a shift to wetter conditions, NPP decrease does not offset the reduced decomposition rate, therefore increasing C accumulation under wetter conditions. Alternatively, there may have been a transient response to changing climate, since plant material formed before the wet shift occurred, and then passing through the acrotelm, may have passed more rapidly into the catotelm due to the higher water tables associated with the wet shift.

The previous chapter provided evidence that repeated burning of blanket bog vegetation caused a significant reduction in recent C accumulation at Moor House. This result supported the findings of Kuhry (1994) and Korhola *et al.* (1995) who both provide evidence for reduced rates of C accumulation under higher fire frequency. However, fire does not seem to have affected the rate of C accumulation in the MH2 profile as strongly as climate since, although small amounts of charcoal particles were observed in core MH2, they covered both periods of implied wet and dry mire conditions, and high and low rates of C accumulation.

Changes in atmospheric deposition of nutrients are unlikely to have caused the changes in peat accumulation since Moor House is located some distance from major industrial areas and the C accumulation rates have been derived for periods before the increase in atmospheric pollution associated with industrialisation. No evidence for significant changes in land management at Moor House has been found which may have affected rates of C accumulation in the blanket peat.

There are few equivalent records of C accumulation at such high resolution in peats to compare with the results of this study. Oldfield *et al.* (1997) found rates of C accumulation in a high latitude peat profile from Sweden (Stor Åmyrân) to be relatively high between *ca.* AD 1300 to AD 1400, and to have been much lower between *ca.* AD 1400 to AD 1700. However, the results from Oldfield *et al.* (1997) contrast with those derived in the present study from Moor House. Firstly, Oldfield *et al.* (1997) found lowest rates of C accumulation when the Moor House peat appears to have accumulated C most rapidly. Furthermore, Oldfield *et al.* (1997) implied that cold conditions during this time period (derived from dendrochronological studies) resulted in the reduced C accumulation rates, although the Moor House results suggest that cold conditions, leading to increased mire surface wetness, caused greater C accumulation rates. However, the two areas of peatland are very different; Moor House is an upland blanket mire in the mid-latitudes while Oldfield *et al.* (1997) studied Stor Åmyrân, a lowland 'eccentric' or 'aapa' mire in the high latitudes of Sweden; there are many possible reasons why different mires may respond differently (see Chapter 8).

The results of this investigation suggest that, under wetter surface mire conditions, C has accumulated at a faster rate in the blanket bog at Moor House. The implication is that under a decreased mire surface wetness, in response to the predicted global warming, C sequestration will be reduced in the blanket bog at Moor House.

6. A model of terrestrial carbon accumulation at Moor House

6.1 Introduction

In the previous chapter, the response of the Moor House C store to future climate change was inferred from its response to past change. The impact of climate change on ecosystems can also be predicted by modelling contemporary processes and observing the model response to future climatic scenarios (*e.g.* Schimel *et al.*, 1994). This chapter describes a model which attempts to simulate the C balance of soils which is used to predict the impact of future climate change on the size of terrestrial C stores at Moor House.

Climate influences the size of terrestrial C stores because temperature and moisture influence both C gain, through photosynthesis, and C loss, through respiration (Melillo *et al.*, 1996). Therefore, the impact of climate change on terrestrial C storage can be predicted if the response of these C fluxes can be modelled. The main C fluxes of terrestrial ecosystems are summarised in three equations provided by Melillo *et al.* (1996):

$$NEP = GPP - Ra - Rh \qquad \textit{[equation 6.1]}$$

$$NPP = GPP - Ra \qquad \textit{[equation 6.2]}$$

$$NEP = NPP - Rh \qquad \textit{[equation 6.3]}$$

Where : NEP = Net ecosystem production. The yearly rate of change in C storage in an ecosystem

GPP = Gross primary production. The amount of C fixed by photosynthesis each year

Ra = Autotrophic respiration. The annual amount of C released by plants through respiration

Rh = Heterotrophic respiration. The amount of C released to the atmosphere as CO_2 by respiration of the ecosystems animals and micro-organisms

NPP = Net primary production. The difference between GPP and Rh in one year

Net ecosystem production (NEP) is the rate of C gain or loss from an ecosystem; a positive NEP indicates a sink for atmospheric C, while negative NEP describes a net source. Whether an ecosystem will be a source or sink for atmospheric C following global warming will, as suggested by equation 6.3, depend on the relative response of net primary production (NPP) and heterotrophic respiration (Rh), to climate change (Melillo *et al.*, 1996). Since NPP can be measured using a variety of vegetation harvesting techniques (Whittaker and Marks, 1975), and Rh can be determined by, for example, measuring the weight loss of decomposing litters (Swift *et al.*, 1979), it is possible to determine NEP under contemporary climates.

Annual rates of NPP and decomposition have been shown to correlate with annual actual evapotranspiration (AET), which is a measure of the actual amount of water evaporated or transpired from an area in one year (Shaw, 1994). For example, Leith (1975) demonstrated an exponential relationship between NPP and AET in the results from globally distributed production studies, while Meentemeyer (1971) and later Berg *et al.* (1984) found linear correlations between annual weight loss of plant litters and AET. It is not surprising that correlations exist between these variables since AET is a composite measure of both energy (temperature) and moisture (Ineson and Stevens, 1990), both of which strongly influence plant growth and decomposition. However, the exponential relationship between AET and NPP, and the linear correlation between AET and decomposition, suggest that the two processes respond differently to changing values of AET. Since climate change will affect AET, the relative rates of NPP and decomposition will also change and impact NEP and the size of terrestrial C stores (Ineson and Stevens, 1990).

The relationships between AET, NPP and decomposition form the basis of the C balance model developed in the present study. To test these relationships for Moor House, NPP and decomposition rates have been measured at sites along an altitudinal transect within the Reserve, providing a gradient of AET. NPP was measured to verify the predictions from Leith's (1975) regression, for Moor House vegetation. However, because the relationship between AET and decomposition is species dependent (Berg *et al.*, 1984), decomposition of litter from important plant species at Moor House was investigated to parameterise the decomposition component of the C balance model.

To run the model, long-term records of temperature and rainfall were required for the study site. The construction of long-term records from existing, fragmented records, has also been undertaken.

This chapter describes the measurement of NPP, decomposition and climate variables along an altitudinal transect at Moor House, the construction of long-term records of temperature and rainfall for the site, and the development of a predictive model to simulate terrestrial C storage under different climate scenarios.

6.1.1 An investigation of NPP and decomposition along an altitudinal gradient

The Great Dun Fell (GDF) altitudinal transect extends from Newton Rigg College near Penrith, to the summit of Great Dun Fell, Moor House (Figures 6.1 and 6.2), spanning an altitudinal range of *ca.* 700 m along a horizontal distance of 24 km. The transect was

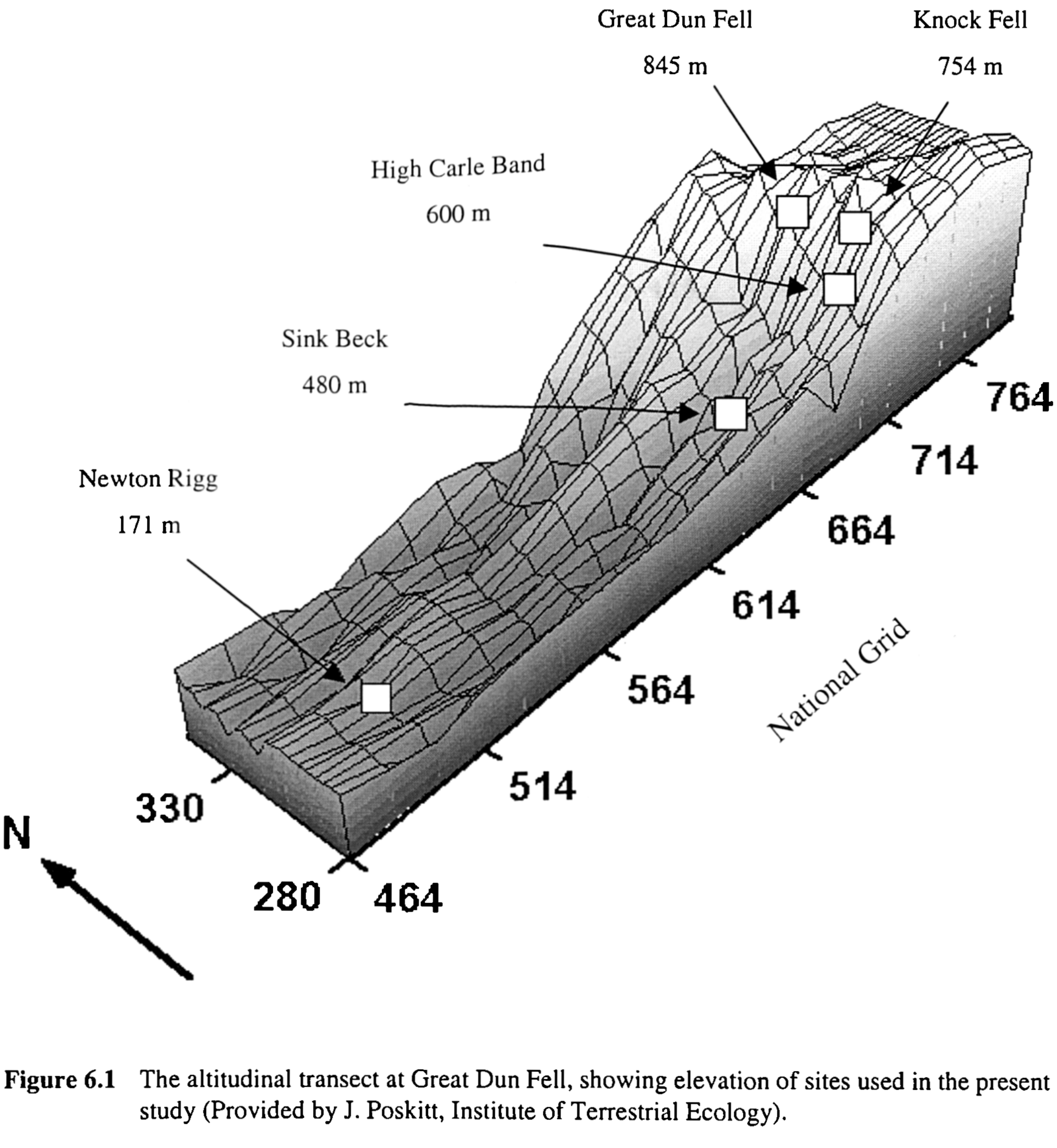


Figure 6.1 The altitudinal transect at Great Dun Fell, showing elevation of sites used in the present study (Provided by J. Poskitt, Institute of Terrestrial Ecology).

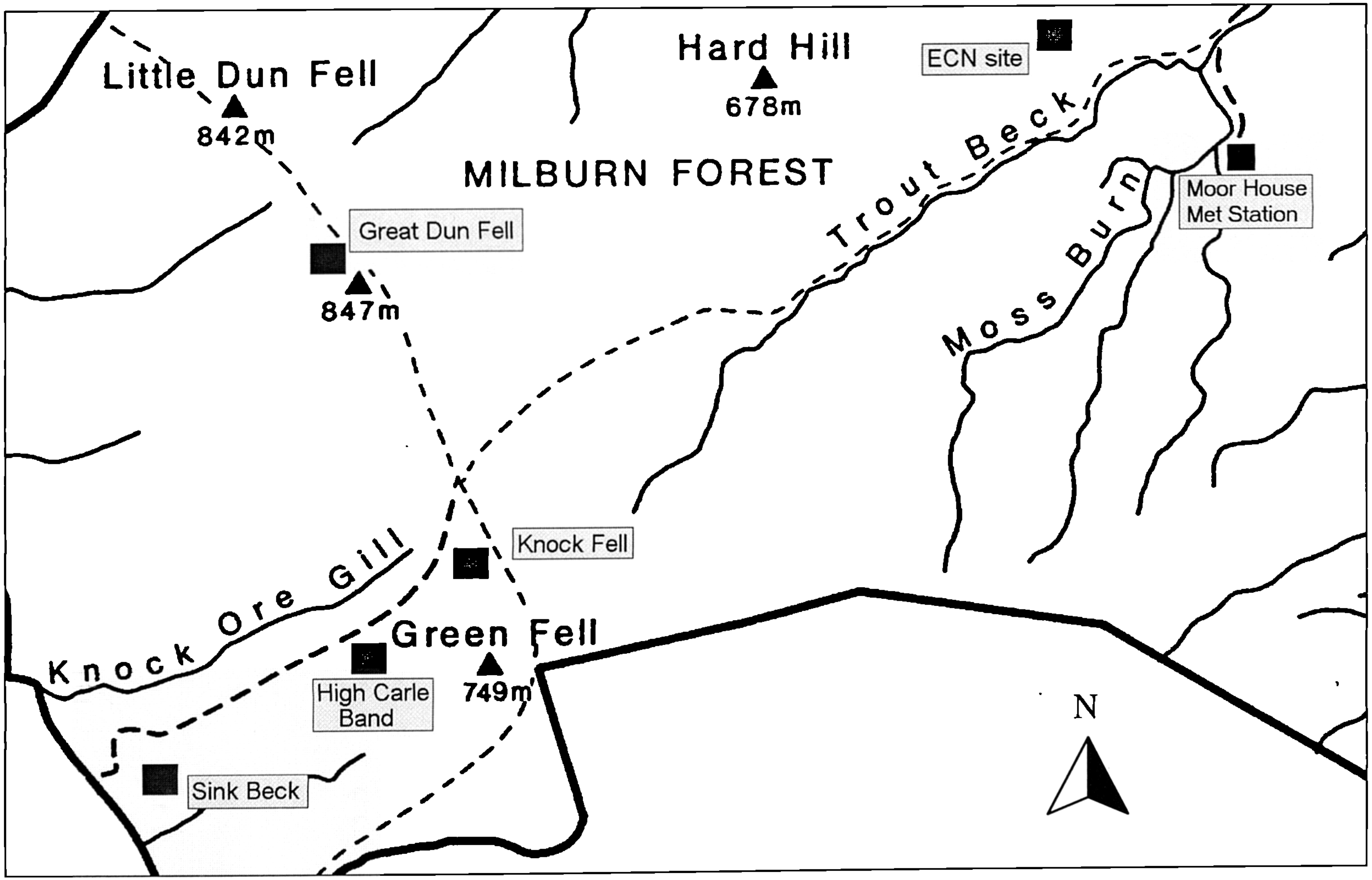


Figure 6.2 Location of the Moor House meteorological station and sites used in the NPP and decomposition studies (Newton Rigg not shown).

established as part of the Natural Environment Research Council's (NERC's) Terrestrial Initiative in Global Environmental Research (TIGER) programme to investigate the impact of climate change on soil processes (see *e.g.* Ineson *et al.*, 1998b).

The transect used during the TIGER programme consisted of four sites: Great Dun Fell (GDF; 845 m), High Carle Band (HCB; 600 m), Sink Beck (SB; 480 m) and Newton Rigg (NR; 171 m). At each site a stock-proof exclosure *ca.* 40 m x 40 m was established and an automatic weather station (AWS) installed to monitor climate variables. In October 1996, an additional AWS was installed at Knock Fell (KF; 754 m; Figure 6.2), and although this was installed after the decomposition study had been established (see below) the site was included in the NPP study.

6.1.1.1 Measurement of NPP

Measurement of NPP was undertaken at the sites along the altitudinal transect in 1997. Two approaches were used to estimate NPP:

i) a transplant approach in which intact soil cores, complete with vegetation cover, were moved to exclosures along the GDF transect,

ii) sampling of fixed plots containing existing vegetation at each exclosure.

In the transplant study, NPP was calculated as the difference between the vegetation C at the start and end of the growing season (Milner and Hughes, 1968). In the fixed plots study, the maximum vegetation C was determined by sampling at intervals throughout the growing season and the difference between the maximum vegetation C and measured vegetation C at the start of the growing season, used as an estimate of NPP (Rawes and Welch, 1969). Since both approaches have different advantages and disadvantages, both were employed in the present study in an attempt to determine independent estimates of NPP.

6.1.1.2 Measurement of decomposition

The rate of decomposition of plant litter can be measured in a variety ways, including weight loss of material in litter bags (*e.g.* Heal *et al.*, 1978; Latter *et al.*, 1998) or CO_2 evolution (Swift *et al.*, 1979); the litter bag method was used in the present investigation.

Sphagnum papillosum was chosen as the litter in the decomposition study since it is a major component of the vegetation at Moor House (Eddy *et al.*, 1969) and had been shown to be such an important component of the blanket peat from the macro-fossil results of core MH2 (Chapter 5; Mauquoy, 1997). Since blanket peat is the largest C store at Moor House

(Chapter 3) the C balance model developed in this study aims to simulate C sequestration in peats and it was desirable, therefore, that a peat-forming species was used to estimate decomposition in the model. Moreover, the decomposition of other major blanket bog species (*Calluna vulgaris*, *Eriophorum vaginatum* and *Rubus chamaemorus*) had already been investigated along the GDF altitudinal transect (Ineson *et al.*, in prep).

In addition to the four TIGER sites along the GDF transect, the ECN target sampling site at Shaft Hill (altitude 560 m), Moor House (Figure 6.2) was used in the decomposition study. The site provides ideal conditions for a decomposition study since a wide range of environmental variables, including water table depths, are being regularly recorded by the ECN. Since the site will be maintained as a research site for many years the opportunity was also taken to establish a long-term litter bag study: Litter bag studies are rarely maintained over a longer period than a few years although Latter *et al.* (1998) have demonstrated the limitations of extrapolating from such short-term decomposition studies. Only the results from the first year of this long-term *Sphagnum* decomposition study were available for inclusion in this thesis although the author has established the ECN sampling strategy for a *ca.* twenty year study.

6.1.2 Long-term records of temperature and rainfall for Moor House

Continuous records of temperature and rainfall are needed as input variables for C balance models, yet most long-term climatic records are for low altitudes due to historical reasons and convenience. These records cannot be simply adjusted to provide records of temperature and rainfall for high altitude sites such as Moor House since variations in lapse rates may have altered the temperature difference between upland and lowland sites, and because rainfall is spatially heterogeneous (see below).

Air temperature has been measured at Moor House since the 1930's and rainfall from at least the 1950's. However, the records do not represent continuous long-term records because of changes in the instruments used to take measurements, small changes in the location of instruments and gaps in the records. To construct continuous long-term records of temperature and rainfall for Moor House these different records need to be amalgamated by comparing periods with overlapping measurements, and using corrected measurements from other meteorological stations to fill the gaps. The record of temperature for Moor House produced in this way is also described in Garnett *et al.* (1997a) and is believed to be the longest record for any upland location in Britain.

6.1.3 A C balance model for mineral and peat soils

Models are simplifications of reality that can be used to simulate natural processes (Clymo, 1993). They are useful for predicting the impact of climate change on ecosystems since they allow results of experimental investigations to be scaled-up. However, predictions from models must be treated with caution since the real processes, and interactions between these processes are normally complex.

Many models have been developed to simulate processes in soils such as the CENTURY (Metherell *et al.* 1993) and SOIL models (Jansson, 1994). A number of models have been developed to simulate peat accumulation, for example, by Clymo (1978, 1984, 1993), Wildi (1978) and Jones and Gore (1978). However, most peat accumulation models have been designed to simply simulate past peat accumulation and are not necessarily useful for predicting the impact of future climate change on the C balance of peats. The predicted impact of climate change on soil C stores has, however, been investigated in several studies using the CENTURY model. For example, Schimel *et al.* (1994) used the model to investigate the impact of a range of factors (including climate) on soil C stores and Parton *et al.* (1995) used the model to predict the impact of increasing atmospheric CO_2 and climate change on global soil C stores.

A model was developed in the present study based on the simple empirical relationships between climate and the major processes controlling the input and output of C in soils. The main aim of the model was to simulate C accumulation in peats and to investigate the impact of climate change on this accumulation since these stores are the greatest at Moor House (Chapter 3).

A trade-off exists in model construction between complexity, which makes models difficult to understand ('black boxes' *sensu* Clymo, 1993) and simple models, which are easy to comprehend, yet may produce less accurate predictions. The main objective of the model described here was not to produce accurate estimates of the response of soil C stores to climate change (though this was a secondary product) but to simulate the major processes which affect the input and output of C in terrestrial stores. It was hoped that this would allow a greater understanding of the net response of these processes to climate change, enabling predictions to be made about the impact of global warming on the C stores at Moor House. It was hoped that the model would also help explain the changes in C accumulation rates observed in core MH2.

A modelling approach is required to determine the influence of climate change on the C balance of peats since it is difficult to experimentally simulate changes in climate. Climatic warming can be simulated by either artificially heating soils (*e.g.* Ineson *et al.*, 1998a) or by transplanting soils down an altitudinal gradient (*e.g.* the transplant approach used to measure NPP, this chapter; Ineson *et al.*, 1998b). However, it is difficult to determine changes in the size of the soil C pool over short time-scales which are likely to be small and virtually undetectable over a few years, although may represent an important C flux if extrapolated to the total soil C store and longer time periods. However, small changes in the rates of C input and output can be detected in experiments simulating climate change because changes in these fluxes are more easily detectable. Models are therefore essential in extrapolating these changes in fluxes to determine the longer-term net impact on C stores.

The model developed in the current work is based on the relationships of NPP and decomposition rate with AET. The model calculates C stores annually and is based on the following equation:

$$S_n = S_o + NPP_d - D_o \qquad \textit{[equation 6. 4]}$$

Where: S_n = Total soil C at the end of the year

S_o = Total soil C at the start of the year

NPP_d = Net primary production during the year

D_o = Decomposition rate of soil C over the year

In the model, NPP_d and D_o are derived from the relationships of NPP and decomposition rate with AET, described above. Since AET is a climatic index based on both temperature and rainfall, the model can be used to simulate the predicted impact of climate change on soil C stores.

Equation 6.4 best describes the model for free draining soils without a water table. However, to simulate C accumulation in peats, the model should incorporate water table depth and the effect of water table on decomposition rates. Development of the model is described in the methods section.

6.2 Method

6.2.1 Measurement of NPP along an altitudinal gradient

Characteristics of the six sites used in the NPP and decomposition studies are provided in Table 6.1. All were stock-proof exclosures with the exception of the ECN site which, although being unfenced, was in a part of the Reserve with low sheep densities.

Table 6.1 Characteristics of the sampling sites used for the NPP and decomposition studies

Site	Great Dun Fell	ECN site	Knock Fell	High Carle Band	Sink Beck	Newton Rigg
Site abbreviation	GDF	ECN	KF	HCB	SB	NR
Variables measured	NPP and weight loss	Weight loss only	NPP only	NPP and Weight loss	NPP and weight loss	NPP and weight loss
Altitude (m)	845	560	754	600	480	171
Vegetation	Festucetum	Calluneto-Eriophoretum	Agrosto-Festucetum	Nardetum sub-alpinum	Nardetum sub-alpinum	Improved grassland
Soil	Fell top podzol	Blanket bog	Red-brown limestone	Peaty podzol	Peaty gley	Brown earth

The AWSs (Type WS01, Delta-T, Cambridge, UK) located at each of the GDF transect sites measured air temperature, precipitation, solar radiation, humidity, wind speed and wind direction and were maintained by the author throughout 1997. Measurements were stored in binary form by a data logger (Type DL2) which was down-loaded every *ca.* 4 weeks using a portable lap-top computer.

6.2.1.1 Transplant approach

Intact vertical cores (diameter 10.2 cm and height 22 cm) of soil were collected from a small plot (1 m^2) near the Knock Fell site (Figure 6.2) in March 1997. The cores were retrieved by carefully hammering plastic tubes into the ground and then digging them out using a spade. The soil chosen for sampling was a micro-podzol (Hornung, 1968) with distinct mineral and litter/organic layers. It was anticipated, therefore, that the below-ground C content could be accurately determined in this soil, enabling any small changes in C content to be detected at the end of the growing season. The vegetation and soil in the cores was undisturbed although, prior to sampling, the plot where cores were collected was clipped to an even height in order to reduce the initial high natural variation in vegetation C.

After collection, the cores were immediately randomised and transported to the incubation sites along the altitudinal transect. Four cores were incubated at each site, making a total of twenty cores. Another four cores were placed in sealed plastic bags, returned to the laboratory and analysed to determine the C content of the cores at the start of the growing season (T_0 samples). At the incubation sites, a rectangular trench was excavated and a thin layer of gravel spread along the base of the hole to allow drainage. The cores were placed directly upon this and adjacent to each other so that the surface of the cores was level with the surrounding soil surface. The trench was refilled to support the cores and restore the surrounding native vegetation.

The cores were left throughout the growing season and sampled in late September 1997, cores simply being lifted out of the incubation holes and placed into labelled plastic bags. All cores were returned to the laboratory and stored in a refrigerated room until analysed.

Laboratory analyses

In the laboratory, the cores were carefully removed from their plastic tubes so that the soil remained intact. A description of the soil profile was made by recording the depth and thickness of individual horizons. The above-ground vegetation was clipped to the surface of the soil using scissors, fresh weight determined and then oven-dried (105°C) for 24 hours and re-weighed to obtain the oven-dry weight.

The soil component of the core was sectioned into the following horizons: Litter and organic layer, a grey eluviated and brown illuviated layer, and a yellow clay at the base of the cores. Each core analysed contained these horizons, although there was variation in horizon thickness between cores.

The soil samples were weighed while fresh and then a sub-sample (5-10 g wet weight) removed by pushing a small corer (diameter 1 cm) vertically through each sample. This sub-sample was weighed and then oven-dried (105°C) and weighed again to obtain the oven-dry weight. The moisture content (%) of the sub-sample was determined using equation 3.7. This sub-sample was assumed to have a moisture content which was representative of the whole sample, and therefore, the dry mass of the complete soil sample was determined using equation 4.1.

The rest of the soil from each sample was washed over a 2 mm mesh sieve in order to extract the roots. By placing a much finer sieve (mesh size 500 microns) beneath it was observed during washing that only very small amounts of vegetation passed through the 2 mm sieve, and therefore, the method provided a suitable way to extract roots from the soil. The vegetative parts extracted this way were washed until all the surrounding soil had been removed, oven-dried (105°C) for 24 hours (until constant weight) and weighed.

The organic content of each vegetation sample was determined using %LOI (Allen, 1989). Where only small amounts of sample had been retrieved (*e.g.* the roots from the yellow clay layer), the complete sample was used for %LOI determination. Larger samples were ground using a mill (Christy-Norris) and a representative sub-sample (>1 g dry weight) used. %LOI was determined following Allen (1989; see Section 3.2.3.3) and calculated assuming the C content of organic matter was 50 % (Jain *et al.*, 1997).

6.2.1.2 Fixed plots

Plots were established at the same sites and at the same time (March, 1997) as the transplant study by defining an area of 1.5 m x 1.5 m of existing undisturbed vegetation in the GDF transect exclosures (Table 6.1). The plots were chosen to have uniform slope, aspect and vegetation composition and were clipped to an even height a few centimetres above the surface of the soil to reduce the high natural variation in vegetation C at the start of the study.

Samples were collected at the start of the study and analysed immediately to enable the vegetative biomass at the beginning of the growing season to be determined (T_0 samples). Subsequently, samples were collected every 1-2 months to provide a total of 4 sampling dates (Table 6.2).

Table 6.2 Sampling dates for the fixed plots study of NPP

Sampling number	Date
1 (T_0)	20th March, 1997
2	25th June, 1997
3	19th August, 1997
4	23rd September, 1997

Each plot was divided into a six column and four row sampling grid and numbered one to twenty-four. Random numbers were used to select the location of sampling points, the sampling areas based on the grid. Four samples were retrieved on each sampling occasion using a soil corer (diameter 11 cm and coring depth 12 cm) by forcing it into the ground and turning. The corer was pushed into the soil to 12 cm, and meant that standard samples were retrieved from each location at each sampling time. Samples were immediately sealed in labelled plastic bags and stored in a refrigerated room until analysed.

Laboratory analyses

The above-ground vegetation of the cores was clipped to the soil surface using scissors, weighed while wet, oven-dried (105°C) and weighed again to determine dry weight. Since the soils types were different at each location, and contained most organic material in the top part of the cores, each core was divided by slicing at 2 cm depth. The samples were weighed while fresh and sub-samples extracted using the small corer described above for the transplant study. Below-ground vegetation was extracted and measured for each sample by washing (as for the transplant study; see above). Similarly, %LOI was used to determine the organic content of each sample for all the vegetation samples and organic material assumed to be 50 % C (Jain *et al.*, 1997).

6.2.2 Decomposition of *S. papillosum* along an altitudinal gradient

S. papillosum plants were collected from an area of blanket bog a few metres away from the coring location at Shaft Hill, Moor House, described in Chapter 5 (Section 5.2.1). The plants were cleaned of any foreign material and the top 5 cm of each plant removed using scissors to standardise the litter; the lower part of each plant was discarded. All plants were air-dried on plastic trays for over a week, turning them every few days to enable even drying. The *S. papillosum* was then placed in a large plastic bag which was enclosed in a cardboard box and subjected to gamma irradiation (Isotron plc, Bradford, UK) to make certain all the plants were dead. This was thought necessary since Heal *et al.* (1978) found, in a similar litter bag experiment, apparently dead litter of *S. acutifolia* grew after being replaced on the blanket bog.

Preparation of litter bags

Litter bags of dimensions 6 x 12 cm were made from nylon net with a maximum aperture of *ca.* 1 mm. Approximately 0.5 g of air-dried *S. papillosum* was placed in each litter bag and sealed using nylon fishing line which extended out of the bag by about one metre and to which was attached a plastic label. The label contained the sample number engraved into the plastic and overwritten in permanent ink, ensuring that the sample could be identified even after extreme weathering. Prior to sealing the litter bags, a small engraved plastic label containing the sample number was placed into each litter bag, in case the label on the fishing line became detached (Plate 6.1).

Litter bags were incubated at the five sites shown in Table 6.1 and retrieved after 6 and 12 months. To investigate the influence of the start time on decomposition, an extra 10 bags were created for the ECN site which were placed out when the first sampling (6 months) took place.

An extra 50 litter bags were produced for the long-term *Sphagnum* litter bag study and another ten were produced to investigate the possible loss of litter during preparation of the samples and placement in the field. These ten bags were placed at the ECN site with the other bags and retrieved immediately, returned to the laboratory and analysed. Therefore, a total of 170 litter bags containing *S. papillosum* were produced, and the weight changes for

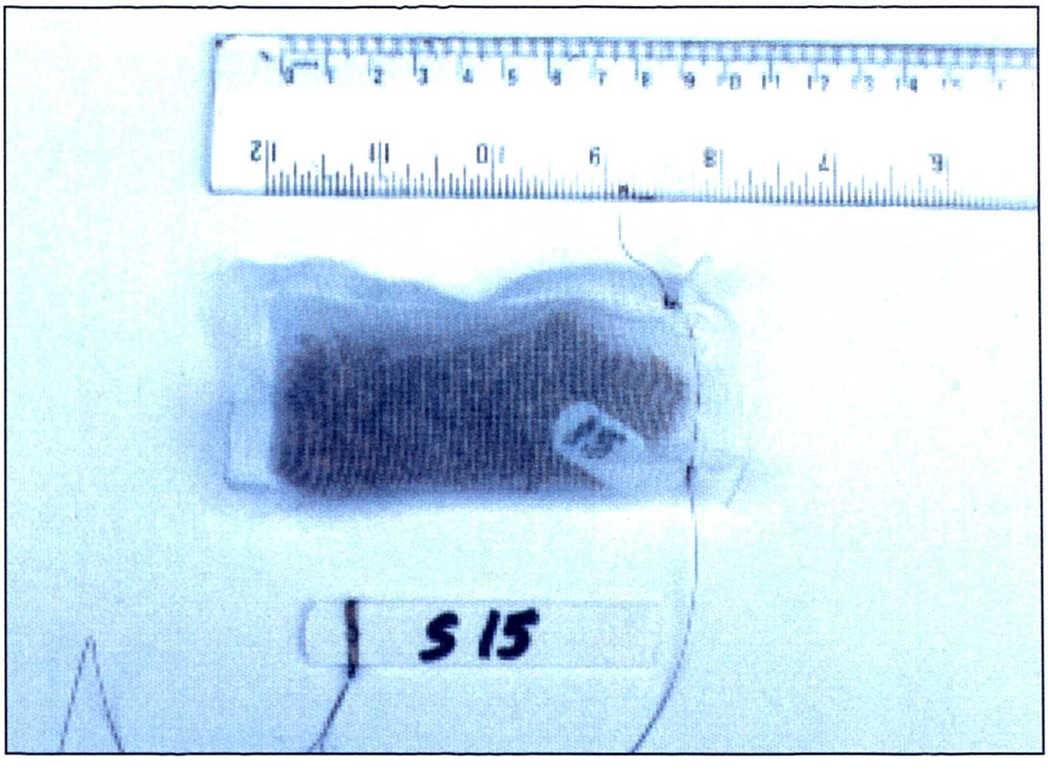


Plate 6.1 Example of a litter bag filled with *S. papillosum*, used in the decomposition study

120 are reported in the present work; the remaining 50 will be sampled by the ECN over the next twenty years.

Ten samples were also required to provide a conversion factor between oven dried and air dried samples. Since accurate measurements of dry-weight loss were needed, all weight losses were calculated from litter at a standardised moisture content. All weight losses were therefore determined after oven drying retrieved samples to constant weight at 40°C. However, initial weights of litter in individual bags was determined using the conversion factor since drying at warmer temperatures may have influenced the litter quality and consequently, decomposition rates (Swift *et al.*, 1979; Ineson *et al.*, in press). Therefore, ten samples of air-dried *Sphagnum* at the start of the experiment were oven-dried to provide the necessary air dry-oven dry conversion factor. This factor enabled estimation of the oven-dry weight of all samples, prior to incubation.

Incubation

In order to simulate the normal conditions of decaying *Sphagnum* litter and to standardise this between sites, the litter bags were placed onto transplanted peat monoliths which had previously been placed within the four exclosures of the GDF altitudinal transect for a decomposition study conducted under the TIGER programme (Ineson *et al.*, in prep). The monolith transplants originally came from a single site on the blanket bog at House Hill, near the meteorological exclosure at Moor House (Figure 6.2) and were 1.5 m x 1 m in area and *ca.* 20 cm in depth. The transplant vegetation was originally typical for the Moor House blanket bog and contained *C. vulgaris*, *E. vaginatum* and *Sphagnum* sp.

Three transects were constructed along the length of the peat monoliths using string tied to wooden stakes at either end of the monolith. The fishing line from each litter bag was then attached to this string. Litter bags had been randomly selected for each site, and were placed onto the peat transplants using random numbers. The litter bags were placed on the surface of the peat transplant and fastened with a stainless steel pin pushed through one corner of the bag, and the fishing line attached to the string (Plate 6.2).

Litter bags were incubated slightly differently at the ECN site since there were more bags, and because the site was undisturbed blanket bog, there was no need for peat transplants. An area of 10 m x 10 m was provided for this study alongside existing ECN monitoring activities. Bags were placed in five randomised blocks with each block containing two litter bags for each sampling occasion. The bags were attached to string, as in the peat transplants, and placed directly onto the blanket bog surface and secured with stainless steel pins.

Plate 6.2 Litter bags placed out on the peat transplant at Great Dun Fell

Litter bags were placed out in September 1996 and collected from all sites after 6 and 12 months of incubation. As mentioned above, additional bags were placed at the ECN site after the first 6 months (March 1997), and left for 6 months to investigate whether the timing of the start of incubation affected the rate of weight loss. Ten samples were collected from each location at each sampling occasion; the bags removed at each sampling had previously been randomly selected before they were placed in the field. On retrieval, litter bags were placed immediately into plastic bags and stored in a freezer until analysed in the laboratory.

Laboratory analyses

The contents of the litter bags were allowed to defrost for a few hours at room temperature and the outside of the bags carefully cleaned. The bags were cut open at one end and the contents removed and carefully examined by eye. In most bags there was apparently very little material which was not the original decomposing *S. papillosum*, although any obviously extraneous material (*e.g.* other vegetation sp.) was removed.

After the contents of the litter bags had been thoroughly checked, the litter was weighed while wet, placed into paper bags and dried in an oven (40°C) to constant weight. The litter was removed from the oven and placed into a desiccator with a maximum of ten samples being placed in a single desiccator since it was found that oven dried *S. papillosum* was extremely hygroscopic. The oven dry weight was measured and the weight loss due to decomposition calculated using equation 6.5:

$$\%W = (S_{40} - E_{40}) / S_{40} \times 100 \qquad [equation\ 6.5]$$

Where: $\%W$ = Percentage weight loss

S_{40} = Estimated oven dry weight before incubation

E_{40} = Measured oven dry weight after incubation

6.2.3 Long-term records of temperature and rainfall for Moor House

Location of meteorological stations

The Moor House meteorological station (Grid Reference NY 757328) is at an altitude of 560 m, and located *ca.* 1 km south of the ECN site where litter bags were incubated in the decomposition study (Figure 6.2). The Widdybank Fell station (Grid Reference NY 818298), used to fill gaps in the Moor House records, lies 6.6 km to the south-east of Moor House at an altitude of 515 m and within the Upper Teesdale National Nature Reserve.

6.2.3.1 Temperature

Available records

Three data sets have been combined to produce the continuous record for 1931-1997, as summarized in Table 6.3. The only published record for air temperature at Moor House is for the period 1931-1978 (Manley 1980). This is an amalgamation of measurements made by Moor House Field Station staff and by Manley himself. Other temperature measurements have been made at the site by the ECN since 1991. The amalgamation of these records has been achieved by comparison of overlapping data sets and by correlation with parallel data collected at the nearby Widdybank Fell meteorological station (data collected by the Nature Conservancy and its successor organisations, the Nature Conservancy Council and English Nature; 1974-1995).

The comparison and correction of the individual data sets is described in the results section (Section 6.3.4). Since most of the available temperature data are for the meteorological enclosure to the east of the former Moor House Field Station, and this is where current readings were being made, the Manley data set for this location acted as the standard against which others were calibrated.

Table 6.3 Temperature records used to produce the Moor House (1931-1997) record (from Garnett *et al.*, 1997a)

Record name	Location	Dates of collection	Instrument	Recording time	Source
Manley	Moor House	1931-1941	Bi-metallic Meteorological Office thermograph in a Stevenson screen	Standardized to 0900 GMT	Manley (1980)
		1942-1947	Mercury-in-steel thermograph in a Stevenson screen	Standardized to 0900 GMT	Manley (1980)
		1952-1978	Standard thermometer in a Stevenson screen	Daily readings at 0900 GMT	Manley (1980)
Widdybank Fell	Widdybank Fell	1974-1995	Standard thermometer in a Stevenson screen	Daily readings at 0900 GMT	Nature Conservancy Nature Conservancy Council English Nature
ECN	Moor House	1991-present	Automatic Weather Station	Daily means	Environmental Change Network (ECN)

6.2.3.2 Rainfall

Rainfall has been measured manually at the Moor House meteorological station (Figure 6.2) from May 1952. However, a large gap in rainfall measurements exists because manual reading ceased in 1979 and measurements (by the ECN) only recommenced in 1991. To extend the Moor House rainfall record, measurements made at the nearby Widdybank Fell meteorological station were used to derive monthly correction factors from periods of overlap in the Moor House and Widdybank Fell records. Details of the available data sets and methods of rainfall measurement are provided in Table 6.4, while the comparison and correction of the data sets are described in the results section.

Table 6.4 Rainfall records used in the Moor House long-term rainfall record

Record name	Location	Dates of collection	Instrument	Source
Moor House	Moor House	1952-1979	Standard Meteorological Office rainfall gauge in a turf wall	Nature Conservancy Nature Conservancy Council
Widdybank Fell	Widdybank Fell	1968-1995	Standard Meteorological Office rainfall gauge in a turf wall	Nature Conservancy Nature Conservancy Council English Nature

6.2.4 A C balance model for mineral and peat soils

The C accumulation model was developed as a spreadsheet computer program (Microsoft Excel 97). The model was developed to run for the period when total annual rainfall and mean annual temperature records were available (*i.e.* 1953 to 1994).

The model calculates C storage on a yearly basis, with an annual input of C from NPP and output through decomposition. There are no other processes included within the model which could affect soil C stores (*e.g.* loss through soil erosion or burning). The inputs of C are considered as 'parcels' which are added annually to the soil C store. Each year a new parcel of C is added to the store which diminishes in subsequent years through decomposition. The total amount of C stored in the soil in any year is a sum of the total amount of C remaining in all parcels.

The model commences by calculating AET from the values of rainfall and temperature in the constructed long-term records for Moor House (this chapter). AET was calculated using Turc's formula (Shaw, 1994) requiring inputs of annual temperature and rainfall. NPP was estimated using Leith's (1975) equation (equation 6.6) and transformed into an estimate of C input for that year by assuming a C concentration of 50%. This initial amount of C represented the total input of C for that year and the initial size of the year's C parcel.

$$NPP = 3000\ (1 - e^{-0.0009695(AET-20)}) \qquad [equation\ 6.6]$$

(From Leith, 1975)

In the model, aerobic decomposition of the organic material occurs in subsequent years at a rate determined by the particular year's AET and the vegetation composition. Different rates of weight loss applied to each individual year because of that year's temperature and rainfall values and the derived AET.

The rate of weight loss for a particular species is determined from the regression of weight loss against AET, derived from the litter bag studies. This regression has been developed for *S. papillosum* in the present study and also for *R. chamaemorus*, *E. vaginatum* and *C. vulgaris* from results of a previous litter bag study on the GDF transect (Ineson *et al.*, in prep); these species are typical blanket bog species and the results have been made available by Ineson *et al.* (in prep) for use in this model.

Single species or combinations of the four species can be used to model decomposition in the C balance model by determining rates of weight loss for selected species or averaging the estimated weight losses derived for different species. This procedure has been undertaken to investigate the impact of changing vegetation compositions on C accumulation in the model, simulating the effect of changing litter quality.

The importance of water table in the process of peat accumulation has been stressed in the previous chapter and a water table function has been included in the C balance model. Water table is very important in controlling C accumulation in peats since decomposition rates below the water table are very slow (Clymo, 1984) and can be considered negligible over the time scale which this C balance model operates (Christen *et al.*, 1995). Therefore, in the model only aerobic decomposition occurs and when the C parcel enters anaerobic conditions, decomposition ceases.

Since the upper layers of peat fluctuate above and below the water table, the proportion of time the C parcel experiences aerobic conditions needs to be calculated. This value is then multiplied by the estimate of weight loss from the regressions with AET to determine the total percentage weight loss.

The proportion of time a particular C parcel is under aerobic conditions depends on its depth in the profile and the depth of the water table. In the C balance model each parcel of C is assumed to increase in depth by 0.5 cm each year. This is similar to the rate which Latter *et al.* (1998) found litter bags to be buried by accumulating peat after originally being placed on the surface of blanket bog at Moor House. After 23 years these litter bags had been covered by accumulating peat at an average rate, relative to the surface, of between *ca.* 0.5 and 0.7 cm yr^{-1}.

The water table depth has been calculated for the C balance model by using the results of the water table monitoring at Shaft Hill, Moor House. The pressure measurements made by the ECN logger in 1996 (see Section 5.2.4) have been calibrated against the manual water table measurements made at the MH2 coring site, using the regression shown in Figure 5.14). The proportion of time each depth experienced aerobic conditions has been estimated by summing the amount of time when the water table was below a certain depth; Figure 6.3 shows the amount of time the water table was calculated to have been at different depths at the MH2 coring point in 1996, and this was assumed to be a typical year for water table depths. Figure 6.3 also shows the total number of hours within the year that each depth experienced aerobic conditions. The proportion of the year that each depth experienced aerobic conditions has been calculated from these results. As the parcels of C pass down the profile they increasingly decompose more slowly because the proportion of time they are under aerobic conditions decreases.

The actual weight of C lost from each C parcel in each year was calculated by multiplying the final weight loss (weight loss predicted from the AET regression multiplied by the proportion of time under aerobic conditions) by the C mass in the parcel at the start of the year.

The model therefore calculates the input of C for each year between 1953 and 1994, and the subsequent size of the C parcel for each year after it was produced. A flow diagram of the stages of calculation is presented in Figure 6.4, while Figure 6.5 shows a sample five year period of the spreadsheet model which calculates C storage.

The impact of different climate scenarios has been investigated using the model by modifying the values of mean annual temperature and total annual rainfall. The impact of different water table regimes has also been investigated by changing the proportion of time each depth experiences aerobic conditions. At one extreme, continually aerobic conditions through the whole profile simulated a freely draining soil where the water table never entered the profile. As mentioned above, the impact of different vegetation species and, therefore, litter quality on C accumulation was also investigated by calculating weight loss using the

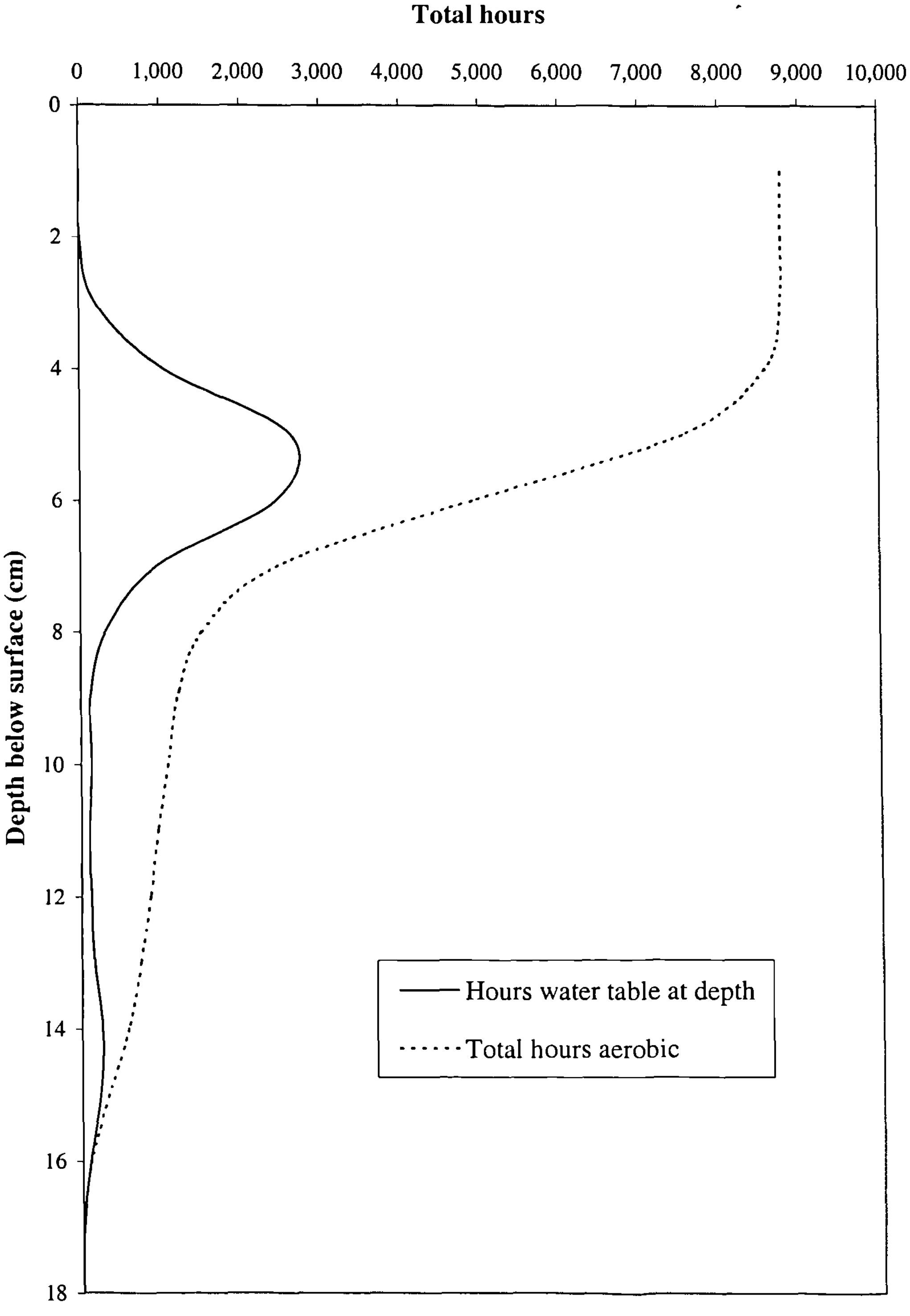


Figure 6.3 Predicted water table at the MH2 coring site in 1996 used to determine the proportion of time aerobic conditions were experienced at different depths for the C balance model

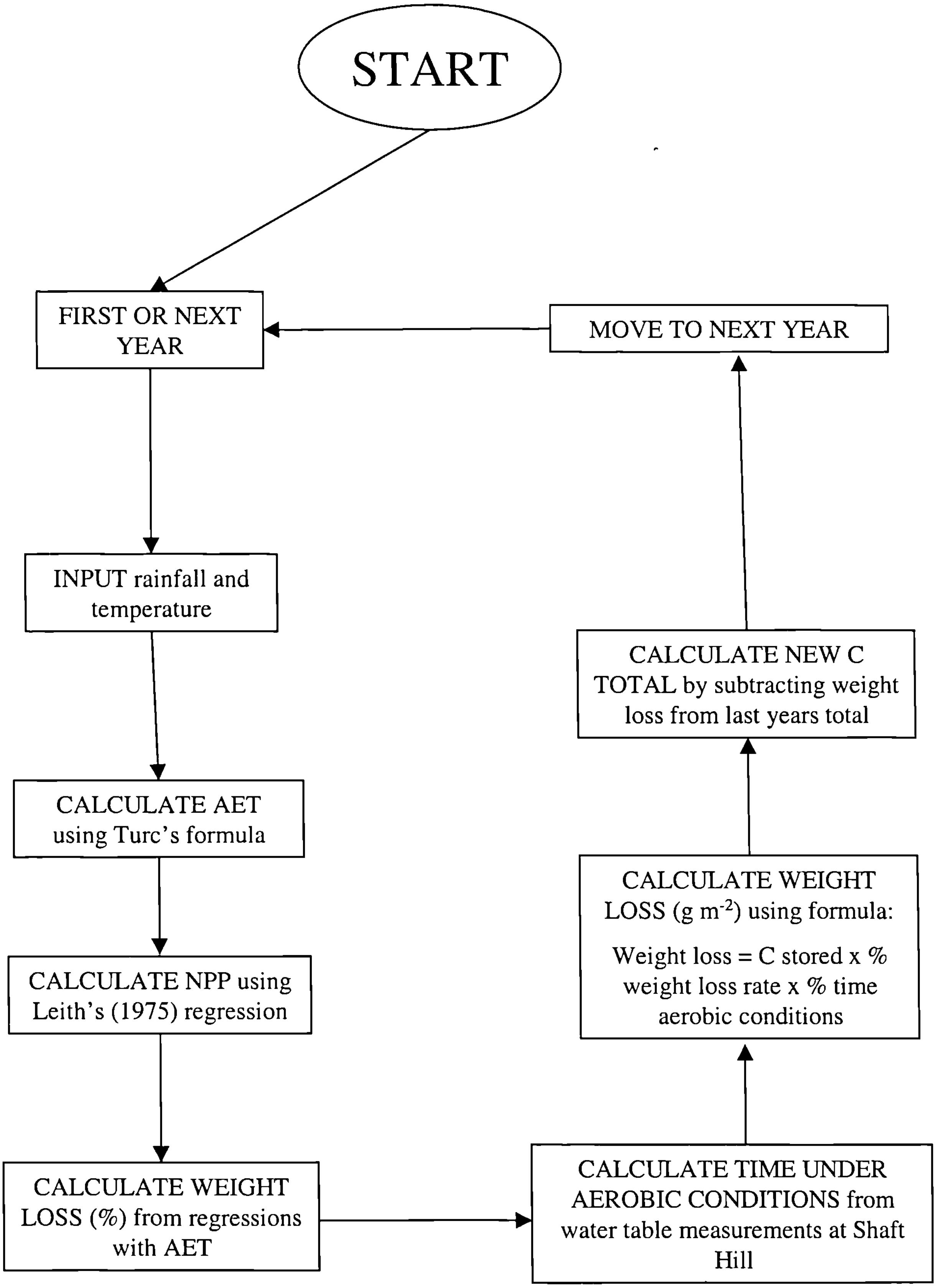


Figure 6.4 Flow diagram of the C balance model for each C parcel

	B	C	D	E	F	G	H
2							
3							
4							
5			Annual	Scenario	Scenario		
6	Year	Total rain	temperature	Total rain	Ann temp	AET	NPP (C)
7	1953	1619	6.2	-C7	D7	Turc's formula (*Eqn 1*)	Leith's regression (*Eqn 2*) * 0.5
8	1954	2612	4.9	C8	-D8	Turc's formula (*Eqn 1*)	Leith's regression (*Eqn 2*) * 0.5
9	1955	1582	4.9	-C9	=D9	Turc's formula (*Eqn 1*)	Leith's regression (*Eqn 2*) * 0.5
10	1956	1805	4.9	-C10	=D10	Turc's formula (*Eqn 1*)	Leith's regression (*Eqn 2*) * 0.5
11	1957	1993	5.8	=C11	=D11	Turc's formula (*Eqn 1*)	Leith's regression (*Eqn 2*) * 0.5
12							

I	J	K	L	M	N
Year	1953	1954	1955	1956	1957
% wt loss	=I7	=I8	=I9	=I10	=I11
% time aerobic	100	100	100	100	100
Ratio aerobic	=J4/100	=K4/100	=L4/100	=M4/100	=N4/100
Mean % wt loss	Parcel	Parcel	Parcel	Parcel	Parcel
Regressions (*Eqn 3*)	=H7	=J7-((K3/100)*J7*K5)	=K7-((L3/100)*K7*L5)	=L7-((M3/100)*L7*M5)	=M7-((N3/100)*M7*N5)
Regressions (*Eqn 3*)		=H8	=K8-((L3/100)*K8*K5)	=L8-((M3/100)*L8*L5)	=M8-((N3/100)*M8*M5)
Regressions (*Eqn 3*)			=H9	=L9-((M3/100)*L9*K5)	=M9-((N3/100)*M9*L5)
Regressions (*Eqn 3*)				=H10	=M10-((N3/100)*M10*K5)
Regressions (*Eqn 3*)					=H11
Total C stored	**=J7+J8+J9+J10+J11**	**=K7+K8+K9+K10+K11**	**=L7+L8+L9+L10+L11**	**=M7+M8+M9+M10+M11**	**=N7+N8+N9+N10+N11**

Figure 6.5 Simplified version of the C balance model, showing calculations for a period of five years from 1953. For convenience the spreadsheet has been divided into two to allow it to fit on the page. The value in each cell can be represented by its column (letters) and row (numbers) as in conventional spreadsheets, for example, cell C7 refers to the total rainfall in 1953 and has a value of 1619. Equations: *Eqn 1* = Turc's formula (Shaw, 1994), *Eqn 2* = equation 6.6 and *Eqn 3* = regressions on Figure 6.22.

different regression equations for each species. The different scenarios described above have been investigated in the model to determine their impact on C accumulation.

6.3 Results

6.3.1 Climatic results for 1997

Figures 6.6 and 6.7 display the mean monthly temperature and total monthly rainfall for each site along the GDF altitudinal transect in 1997. Table 6.5 provides an annual summary of temperature and precipitation recorded by the AWSs at each site along the GDF altitudinal transect. Values for AET are also provided in the table, estimated using Turc's formula, (Shaw, 1994). A more accurate estimate of evapotranspiration could have been derived using Penman's equation (Shaw, 1994), yet this requires humidity measurements which were unavailable. Table 6.6 gives mean annual temperature and rainfall for four of the sites between 1993 and 1995 (annual means provided by Ken Taylor, Institute of Terrestrial Ecology).

Table 6.5 Mean annual temperature and precipitation for sites along the GDF altitudinal transect in 1997

Site	Mean air temperature (°C)	Total rainfall (mm)	AET (mm)
GDF	4.45	1,473	402
KF	5.22	1,947	428
HCB	6.37	1,361	449
SB	7.39	1,158	467
NR	8.99	893	482

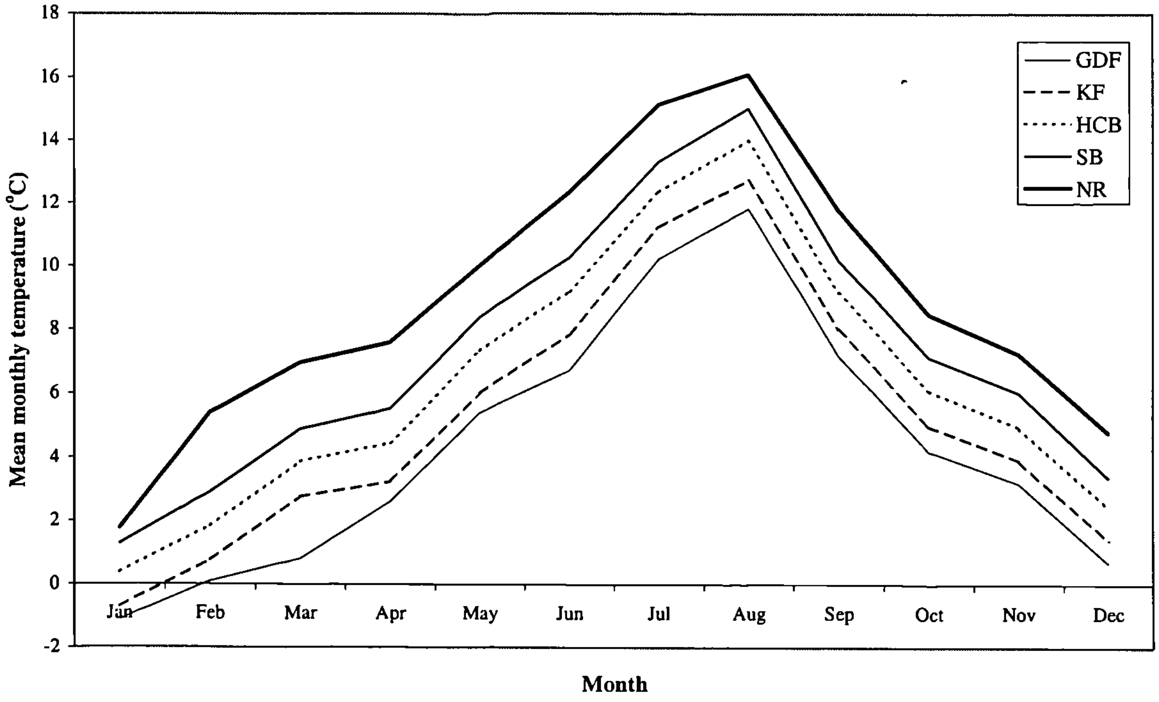


Figure 6.6 Mean monthly temperature for the GDF transect sites in 1997

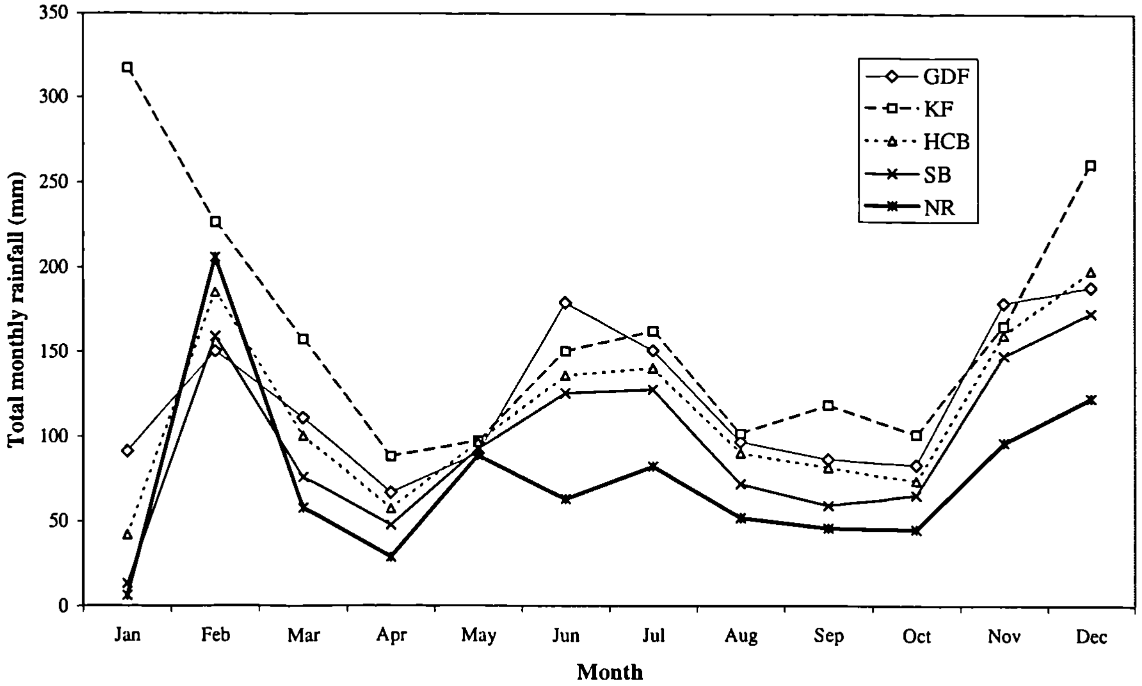


Figure 6.7 Total monthly rainfall for the GDF transect sites in 1997

Table 6.6 Mean annual temperature and rainfall for four of the GDF transect sites for 1993-1995

Site	1993 Mean air temperature (°C)	1993 Total rainfall (mm)	1994 Mean air temperature (°C)	1994 Total rainfall (mm)	1995 Mean air temperature (°C)	1995 Total rainfall (mm)
GDF	3.35	1,758	3.56	1,451	4.14	1,060
HCB	4.97	971	5.20	1,206	6.38	961
SB	6.22	1,242	6.36	1,299	6.99	988
NR	7.81	1,018	8.30	1,240	8.86	873

6.3.2 NPP along an altitudinal gradient

6.3.2.1 Transplants

Plate 6.3 shows examples of the transplant cores from each site after sampling and Figures 6.8 and 6.9 show the results for determining NPP at the altitudinal transect sites. Figure 6.8 provides the mean mass of vegetation C before the 1997 growing season began (T_0 sample) and the mean vegetation C mass at the end of the growing season, for the different locations where the transplanted cores were placed. Besides the total amount of vegetation C from each sample, the above and below-ground components are shown.

Generally, the cores moved to lower altitudes had greater above-ground vegetation C, with the lowest site (NR) having the highest value. However, three of the sites had lower values of above-ground vegetation C than the T_0 sample, and therefore appeared to have lost C over the growing season.

At the end of the growing season there was a consistent trend of reduced below-ground vegetation C with lower altitude, and all sites had lower below-ground vegetation C than the original T_0 samples. The total amount of vegetation C at the different sites therefore showed only a slight increase with increasing altitude at the end of the growing season. Total vegetation C was lower at most sites at the end of the growing season than in the original T_0 samples, which was an unexpected result; a consequence being that NPP was negative for several sites.

Figure 6.9 provides scatter plots between three climatic variables for each site in 1997, against the mean estimates of NPP from the transplant study. Although NPP was estimated to be negative at many sites, for reasons explained in the discussion, NPP may have

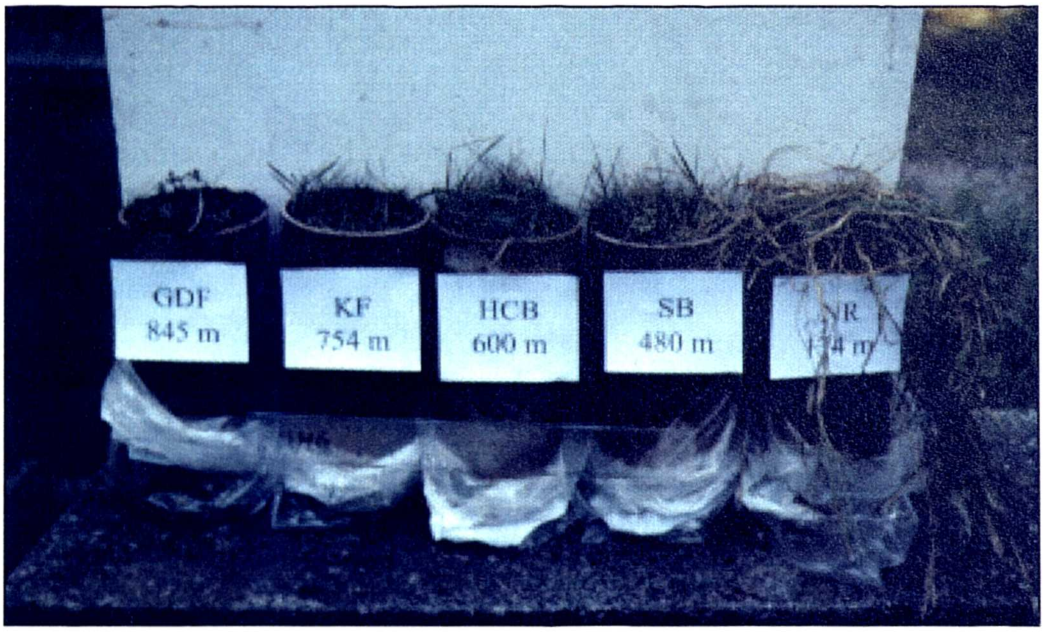


Plate 6.3 Examples of cores retrieved at the end of the growing season from the NPP transplant study.

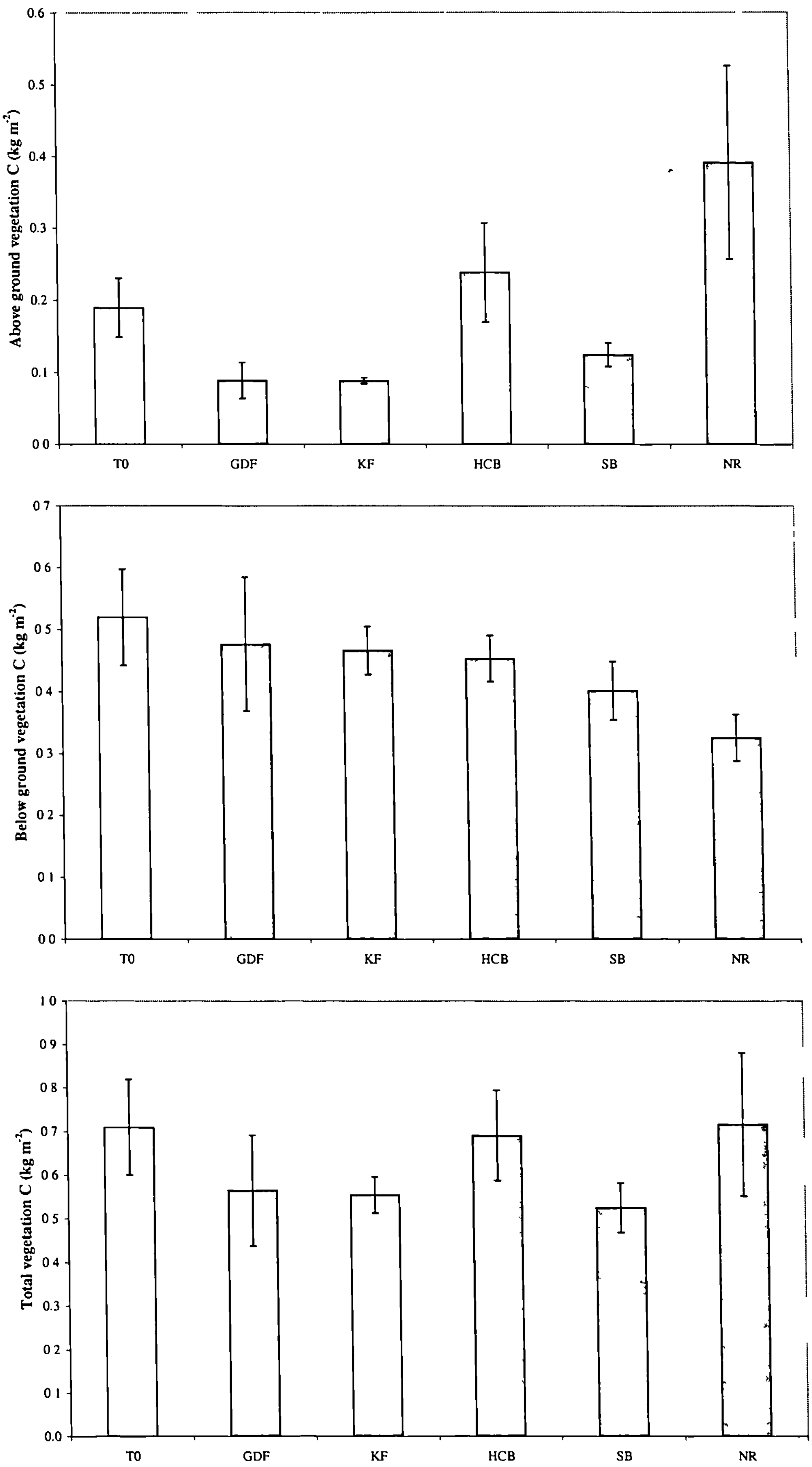


Figure 6.8 Mass of vegetation C in transplants at the start of the growing season (T_0) and after being incubated at different sites throughout the growing season: a) above-ground vegetation C (top), b) below-ground vegetation C (middle) and c) total vegetation C (bottom).

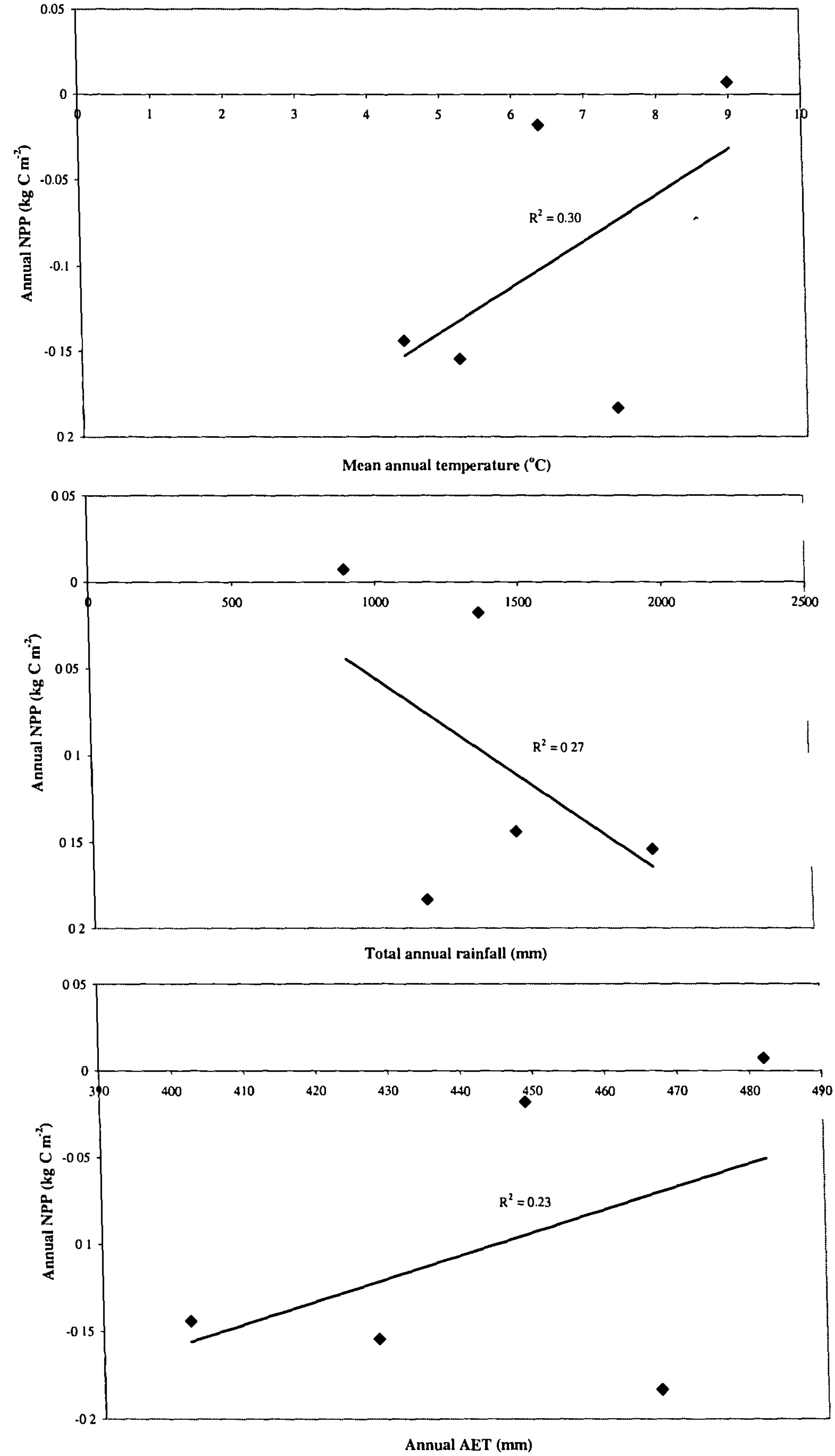


Figure 6.9 Relationships between NPP measured in transplants and a) mean annual temperature (top), b) total annual rainfall (middle) and c) AET (bottom)

been consistently underestimated using the transplant approach. Therefore, the relative differences in NPP may be real and any relationships between climatic variables may still be valid. However, no significant correlations were found between NPP and either mean annual temperature, total annual rainfall or AET.

6.3.2.2 Fixed plots

The values of above-ground, below-ground and total vegetation C measured at the different sampling times in each of the fixed plots along the GDF transect are shown in Figure 6.10. Above-ground vegetation C generally increased throughout the year, although some sites, especially Sink Beck, showed peak values earlier than others. Trends in below-ground vegetation C were less apparent, although the Sink Beck site, as with above-ground vegetation, showed large variability throughout the growing season. At most sites the maximum amount of below-ground vegetation was not much greater than the amount measured at the start of the season. Total vegetation C was less variable during the study period which suggests that the variability observed in above and below-ground compartments may have been partly a result of inconsistencies in defining the boundary between above and below-ground. There was a general increase in total vegetation C during the growing season, although several sites showed reduced values towards the end of the growing season.

NPP was calculated as the difference between the maximum measured total vegetation C and the initial total vegetation C at the start of the growing season. All sites showed positive values for NPP which generally increased with lower altitudes. Figure 6.11 provides scatter-plots between estimated NPP using this approach and several measured climatic variables. Two significant correlations ($p<0.05$) were found; i) between NPP and annual temperature and ii) between NPP and AET. The latter correlation was the best and provided an r^2 of 0.735. As expected, NPP increased with increasing annual temperature and increasing AET.

6.3.3 Decomposition of *S. papillosum* along an altitudinal gradient

The litter bags retrieved immediately after being placed out at the ECN site showed a weight loss of 1.24 ± 0.20 % (SE) and this 'handling loss' was used to correct the initial weights of the other litter bags.

Litter bags were retrieved after 6 and 12 months incubation. Surprisingly, many of the litter samples appeared to have gained weight over these periods even after the litter had been carefully hand-cleaned of foreign material, such as mineral particles.

Examination of the litter bag contents revealed that algae had colonised the *S. papillosum* within the first six months at the lowest site (NR), and at all sites by the end of

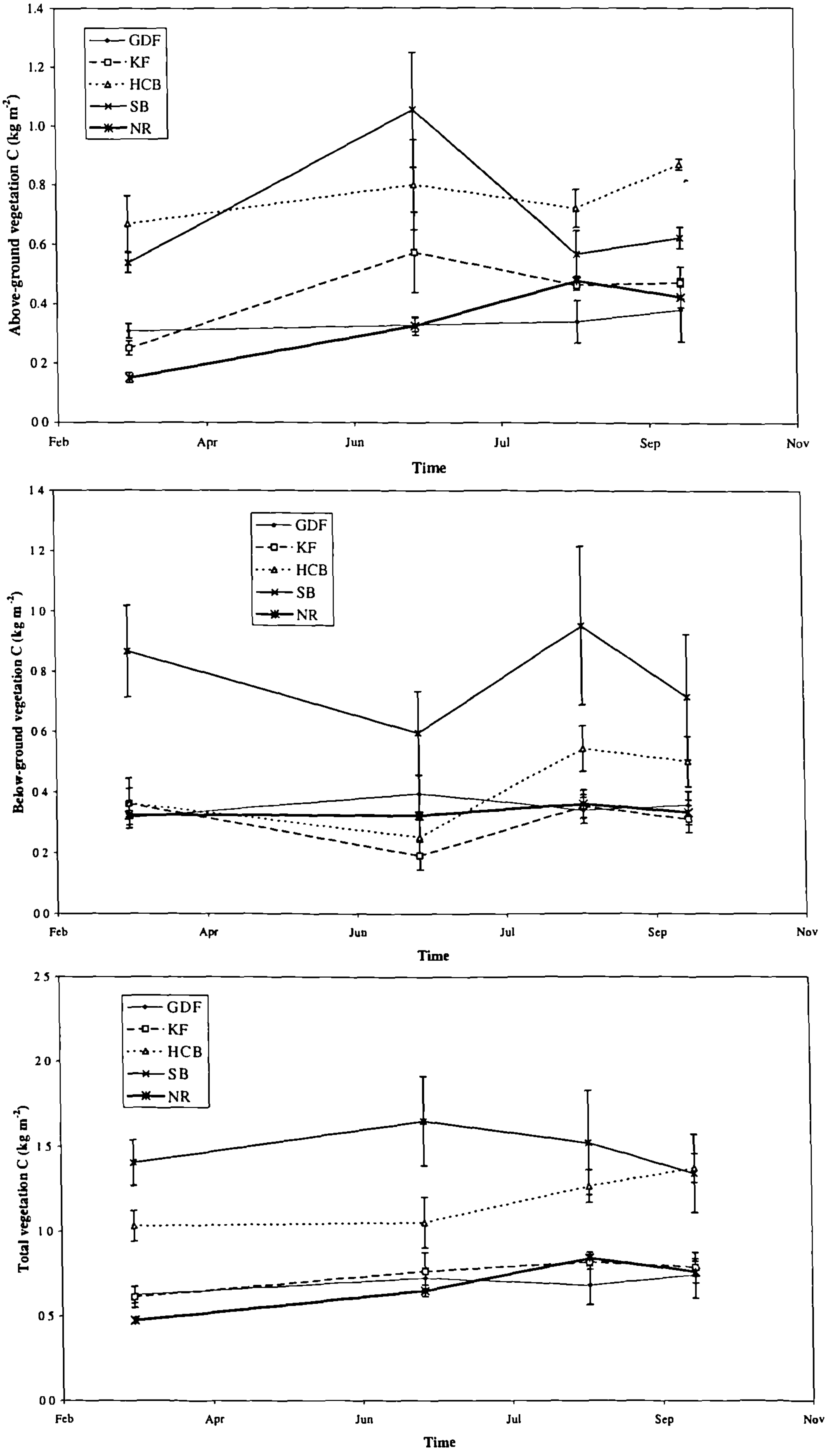


Figure 6.10 Total mass of vegetation C in fixed plots along the GDF altitudinal transect at various sampling times in 1997: a) above-ground vegetation C (top), b) below-ground vegetation C (middle) and c) total vegetation C (bottom). Error bars indicate standard error.

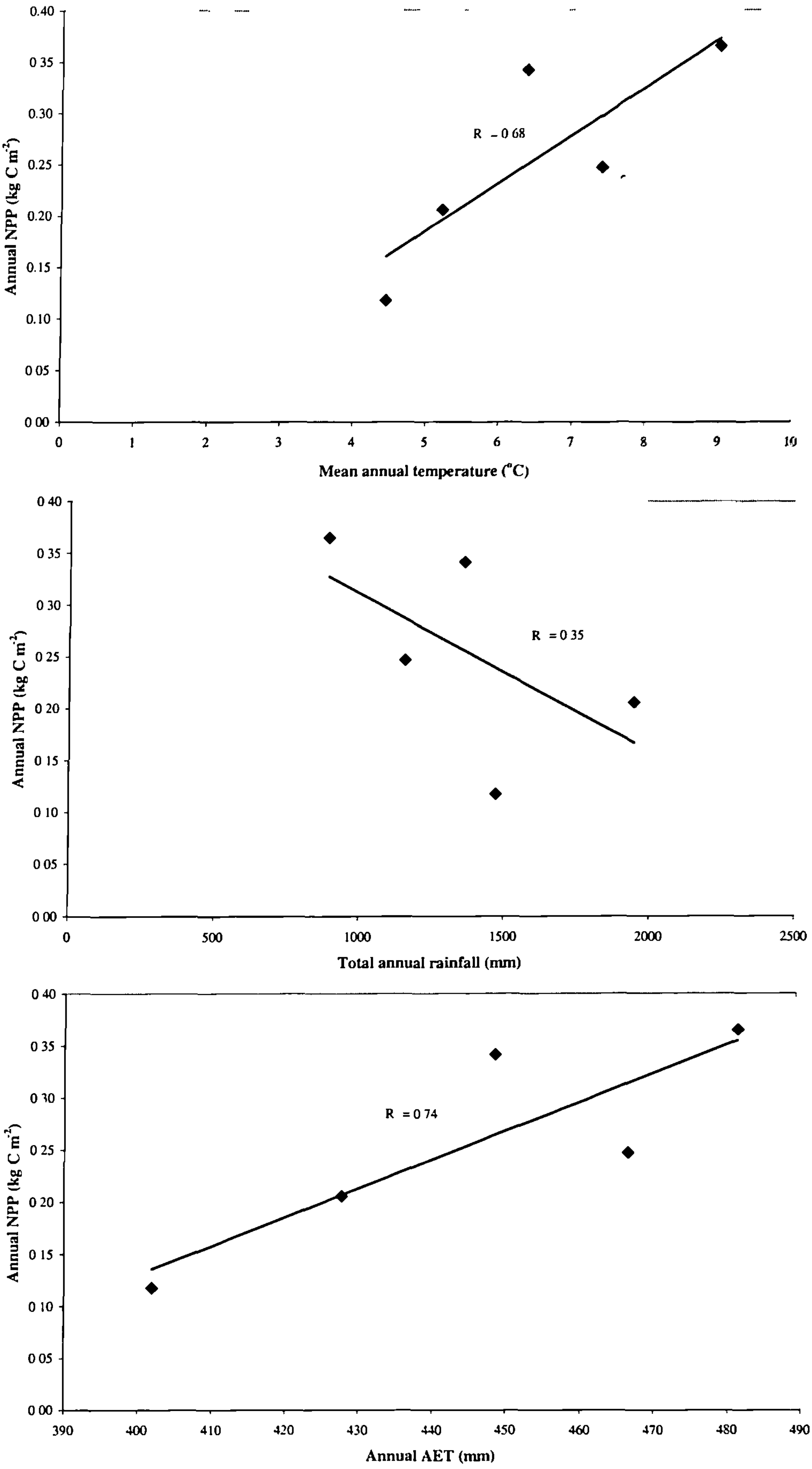


Figure 6.11 Scatter-plots showing relationships between NPP measured in fixed plots and a) mean annual temperature (top), b) total annual rainfall (middle) and c) AET (bottom)

twelve months. Algal growth was recognisable with the naked eye in fresh samples since they were bright green and the decomposing *S. papillosum* litter were a dull yellow. Microscopic examination of the contaminated litter bags confirmed a significant algal growth within the bags.

The algal growth was considered responsible for the gain in weight observed for many of the litter bags, although the author is unaware of this being a problem encountered in other litter bag studies. No suitable method was found to account for the weight increase in bags due to algal growth, and consequently, the results from sites where algal growth was evident have been omitted in the current work. Weight loss was very low for all sites and the increased weight due to the growth of algae is believed sufficient to account for the weight increases observed in certain of the litter bags.

Figure 6.12 shows the weight loss of *S. papillosum* litter after decomposing for six months at four of the sites. The figure shows greatest weight loss occurred in samples incubated at the SB and HCB sites, where weight loss was 6.1 ± 0.5 % and 7.1 ± 0.5 %, respectively. Weight loss of *S. papillosum* at the GDF and ECN sites was *ca.* 4.7 %.

The litter bags placed out in March 1997, when the six month samples were retrieved, did not appear to be contaminated by algal growth and an average weight loss of 4.1 ± 0.4 % was measured for these samples.

Annual weight loss (%) of decomposing *S. papillosum* has been estimated from the results of litter bags incubated for 6 months. Olson's (1963) equation was used to extrapolate results to the longer-term (see Frankland, 1974; Cotrufo, 1994) with the equation assuming that weight loss is exponential. Table 6.7 provides the estimates of annual weight loss derived by extrapolating the results of the six month decomposition study using Olson's (1963) equation.

Table 6.7 Annual weight loss of *S. papillosum* in litter bags and climatic variables for 11th September, 1996 to 10th September 1997. N.B. * climatic data supplied by the ECN.

Site	Annual weight loss (%)	Annual temperature (°C)	Total annual rainfall (mm)	AET (mm)
SB	12.0	6.98	824	428
HCB	13.9	5.96	1242	434
GDF	9.2	3.94	1335	386
ECN*	9.4	6.07	1810	450

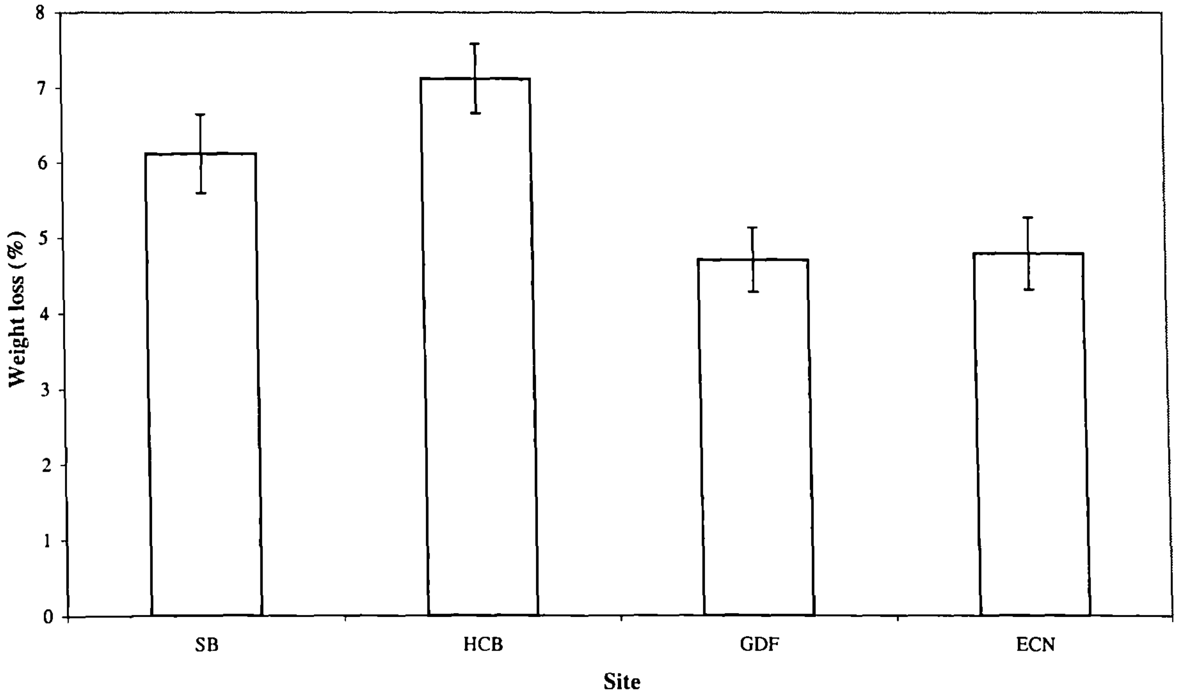


Figure 6.12 Weight loss of *Sphagnum papillosum* litter after six months at four sites at Moor House

The estimates of annual weight loss have been plotted against mean annual temperature, total annual rainfall and AET in Figure 6.13. No significant correlations were found between the variables although, generally, weight loss increased with increasing temperature and AET.

6.3.4 Long-term records of temperature and rainfall for Moor House

6.3.4.1 Temperature

Comparison of Manley and Widdybank Fell records

Mean monthly temperatures recorded concurrently at Moor House and Widdybank Fell between 1974 and 1978 were compared (Figure 6.14). The Widdybank Fell site was, on average, 1.09 °C warmer than Moor House, with the correlation producing an r^2 of 0.987. However, this difference was not consistent throughout the year (Table 6.8) and when differences were calculated for individual months, January to April temperatures were considerably closer to the Moor House temperature than for May to August. These individual monthly temperature differences were applied as corrections to the Widdybank Fell data set, producing a higher correlation (r^2 of 0.992) for the period of overlapping records (Figure 6.15). The corrections were then used for the rest of the Widdybank Fell record to extend Manley's Moor House temperature record up to 1997.

An indication of the uncertainty in the values in the 'extended' Manley record was determined by calculating the standard deviation of the difference between the corrected Widdybank Fell temperatures and values in the Manley record for the period of overlap. These values were used to produce error bars for the final record.

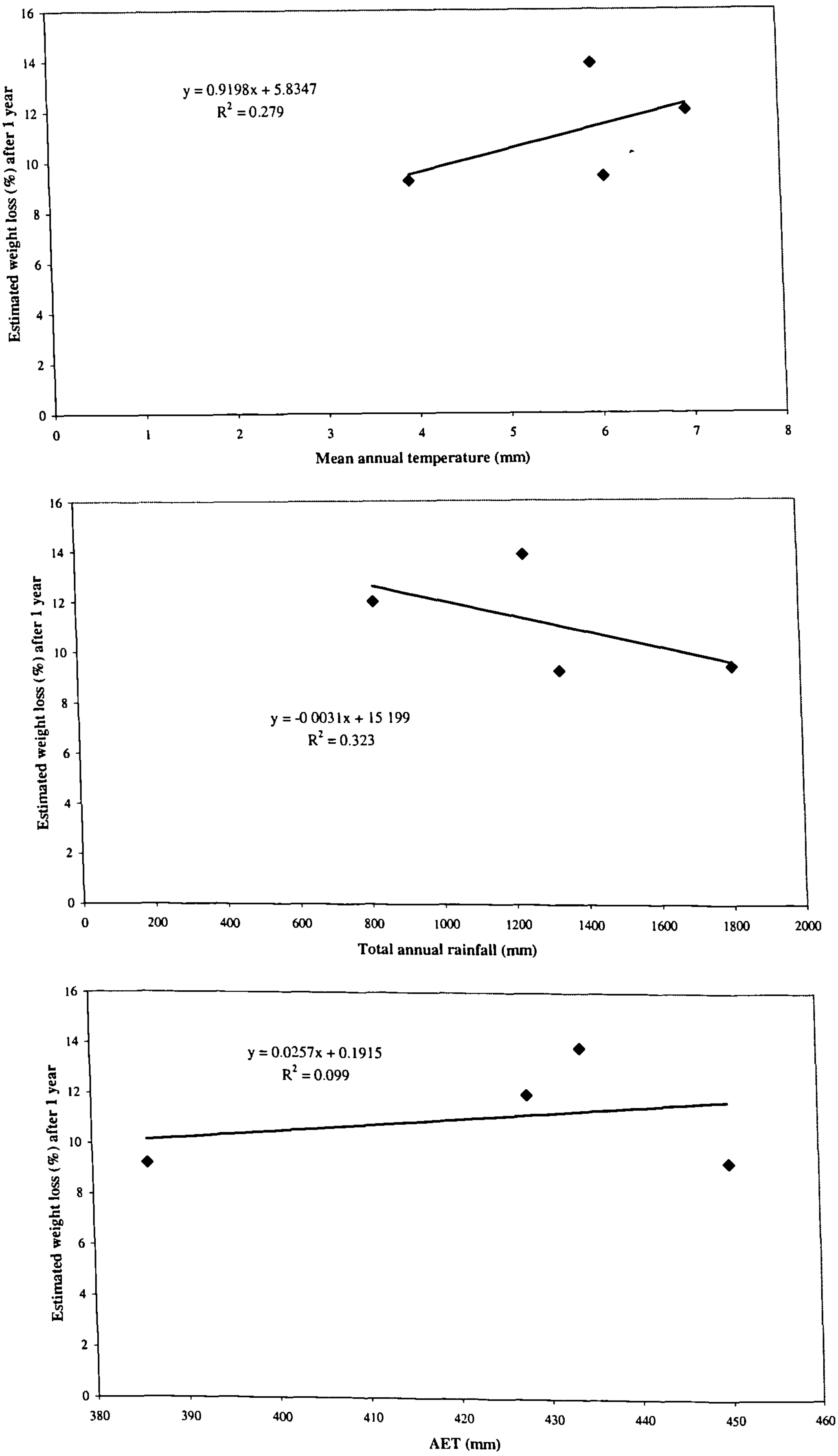


Figure 6.13 Weight loss of *Sphagnum papillosum* against a) mean annual temperature (top), b) total annual rainfall (middle) and c) AET (bottom)

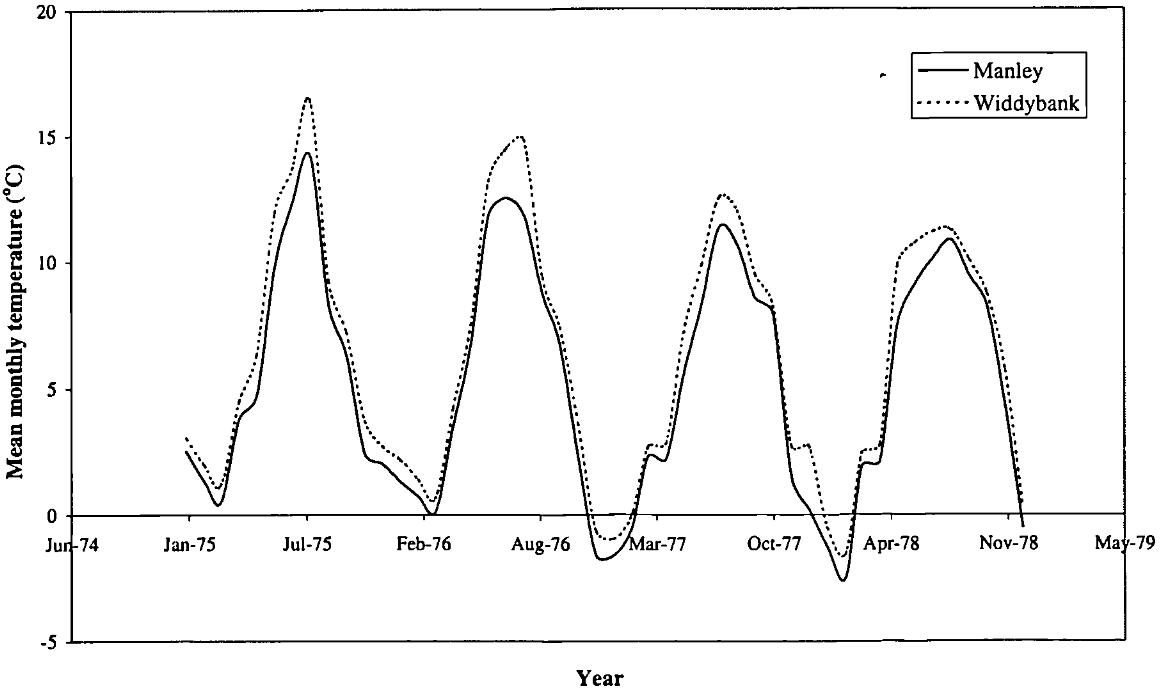


Figure 6.14 Comparison between Manley and Widdybank Fell monthly temperatures between 1975 and 1978

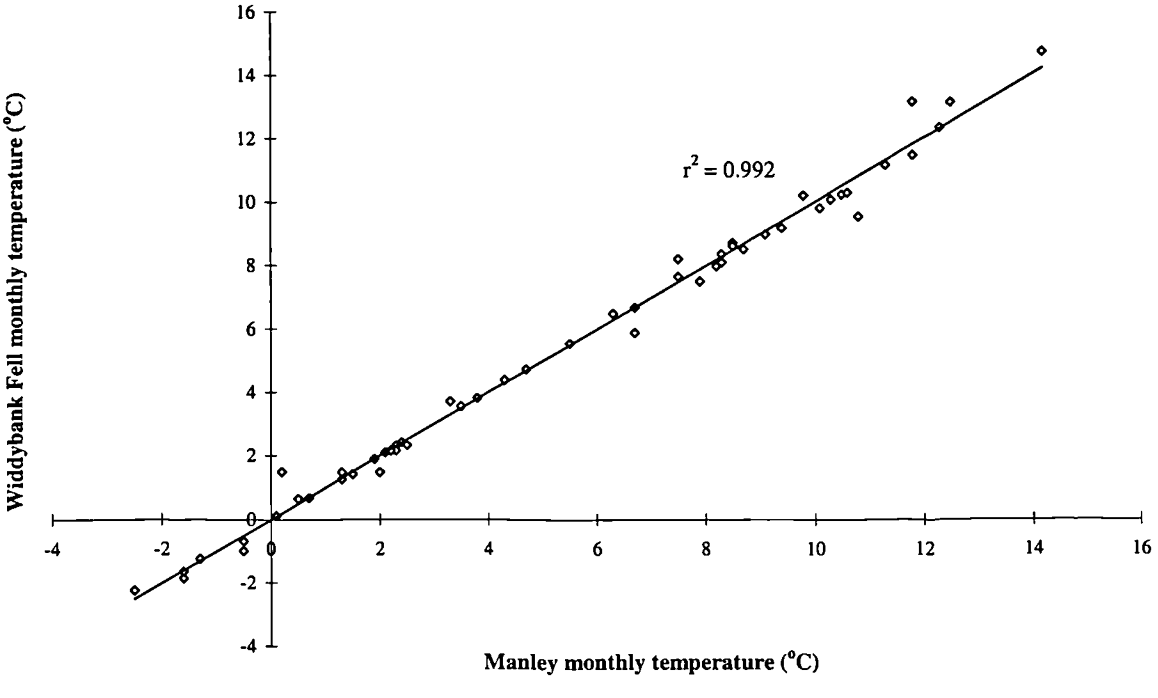


Figure 6.15 Relationship between Manley and Widdybank Fell monthly temperatures between 1975 and 1978

Table 6.8 Mean monthly temperature differences (correction values) between records

Month	Manley-Widdybank Fell	ECN-Widdybank Fell	ECN-Manley
January	-0.69	-0.37	0.32
February	-0.68	-0.33	0.35
March	-0.53	-0.32	0.21
April	-0.67	-0.39	0.28
May	-1.60	-1.05	0.55
June	-1.72	-0.99	0.73
July	-1.33	-0.92	0.41
August	-1.67	-1.02	0.65
September	-0.80	-0.72	0.08
October	-0.68	-0.47	0.21
November	-1.28	-0.50	0.78
December	-1.22	-0.35	0.87
Mean	-1.07	-0.65	0.42

Comparison of ECN and Widdybank Fell records

Recent measurements of temperature made at Moor House by the ECN cannot simply be added to the extended Manley record due to changes in measurement instruments and protocols. However, by determining the monthly temperature differences between the ECN data and overlapping records from Widdybank Fell (producing an r^2 of 0.994), and then subtracting the monthly correction values produced for the Manley and Widdybank Fell records, corrections were derived to calibrate the recent Moor House measurements against Manley's record (Table 6.8). By applying these values, measurements made at Moor House since 1991 are directly comparable with Manley's record, and have been used to produce the final record for Moor House.

Again, an indication of the uncertainty in the corrected values was calculated. This involved determining the standard deviation of the difference between corrected ECN temperatures and the Widdybank Fell values where they overlapped. The additional uncertainty caused by correcting the Widdybank Fell record to the Manley record was used to produce error terms for the ECN values corrected to the Manley record.

The complete record

The complete record of mean monthly and annual temperature derived for Moor House is shown in Figures 6.16 and 6.17, respectively. Strong evidence for warming is not clear from this record, though there are several marked deviations from the long-term mean.

The Moor House record shows mean annual temperatures in the period 1930 to 1950 to be consistently around 5°C but there was a marked decrease in mean annual temperature from *ca.* 1960, continuing to the mid-early 1980's, with a brief reversal of the trend in the mid 1970's. The final part of the record indicates a slight warming trend, with the 10-year running mean returning to the 1931-1997 mean (Figure 6.17).

6.3.4.2 Rainfall

The monthly rainfall totals recorded at the Moor House and Widdybank Fell meteorological stations were also compared. Figure 6.18 shows a comparison between the rainfall values for both sites over the period of overlap of records and Figure 6.19 is a scatter plot showing the correlation between the two sites. The comparison between sites produced an r^2 value of 0.59 although the relationship between monthly rainfall totals for the two sites was closer in months when least rainfall occurred (Figure 6.19).

Since monthly correction values had proven very useful for extending the temperature record at Moor House (see above), monthly correction values were derived to calibrate the Widdybank Fell rainfall record to the Moor House values. The correction values are shown in Table 6.9; greatest differences between the two sites occurred in the summer months, with June, July and August being the months where rainfall differences between the two sites was greatest. Moor House received a greater amount of rainfall than Widdybank Fell for every month.

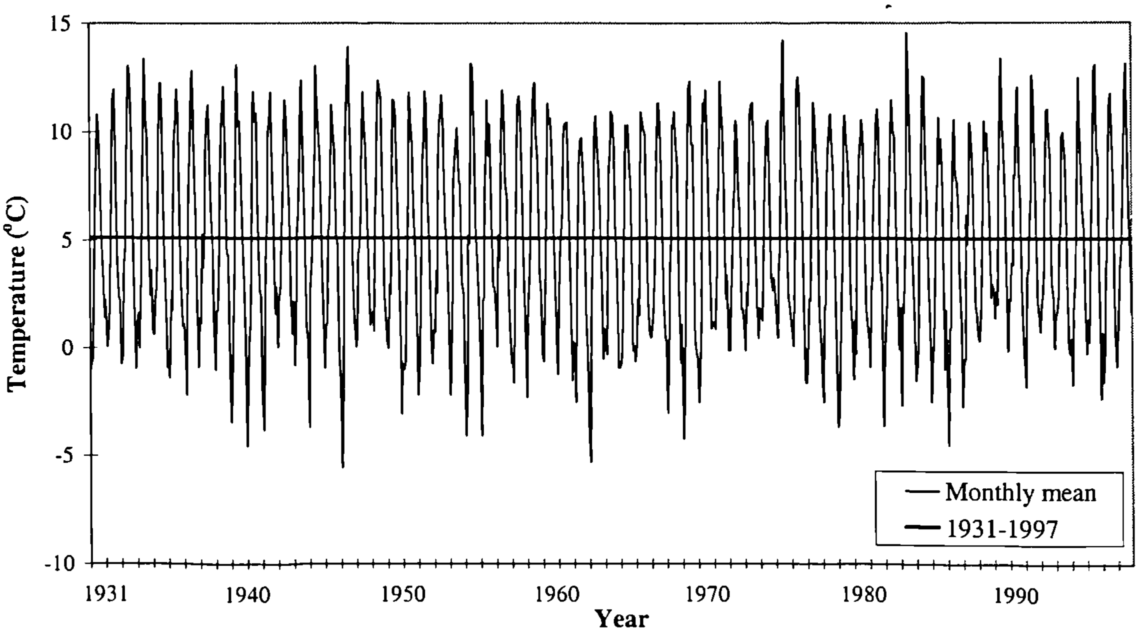


Figure 6.16 Mean monthly temperature at Moor House between 1931 and 1997

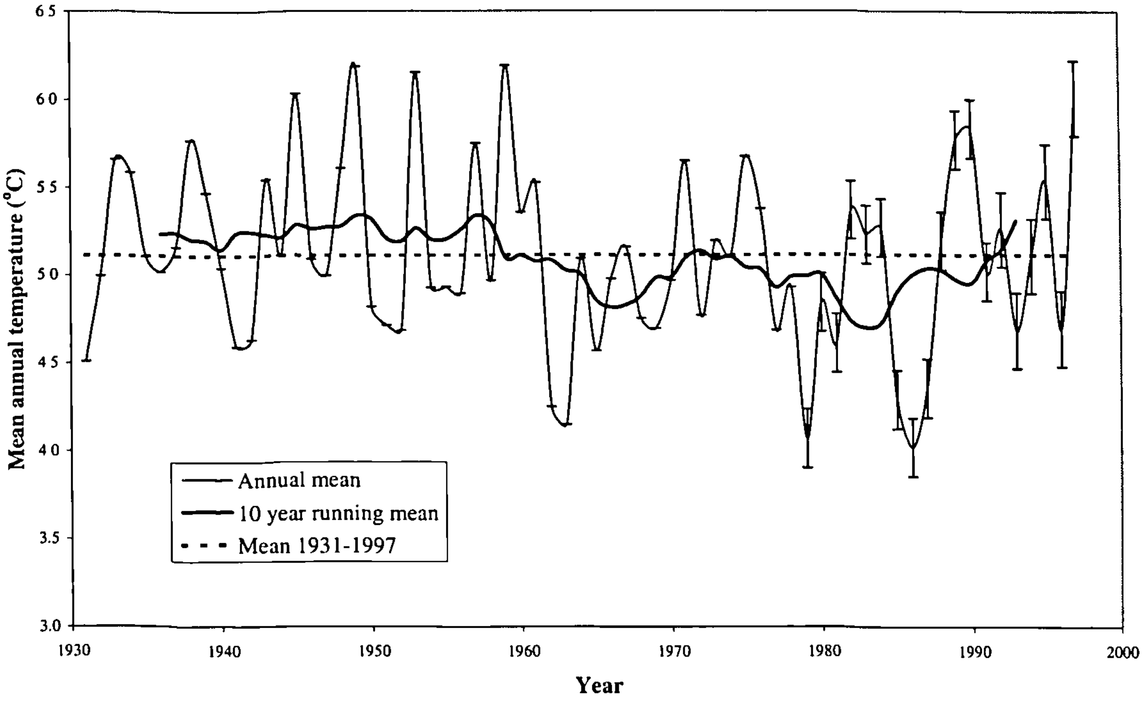


Figure 6.17 Mean annual temperature at Moor House between 1931 and 1997. Error bars indicate 95 % confidence limits for predicted temperatures, showing the uncertainty caused by using correction values

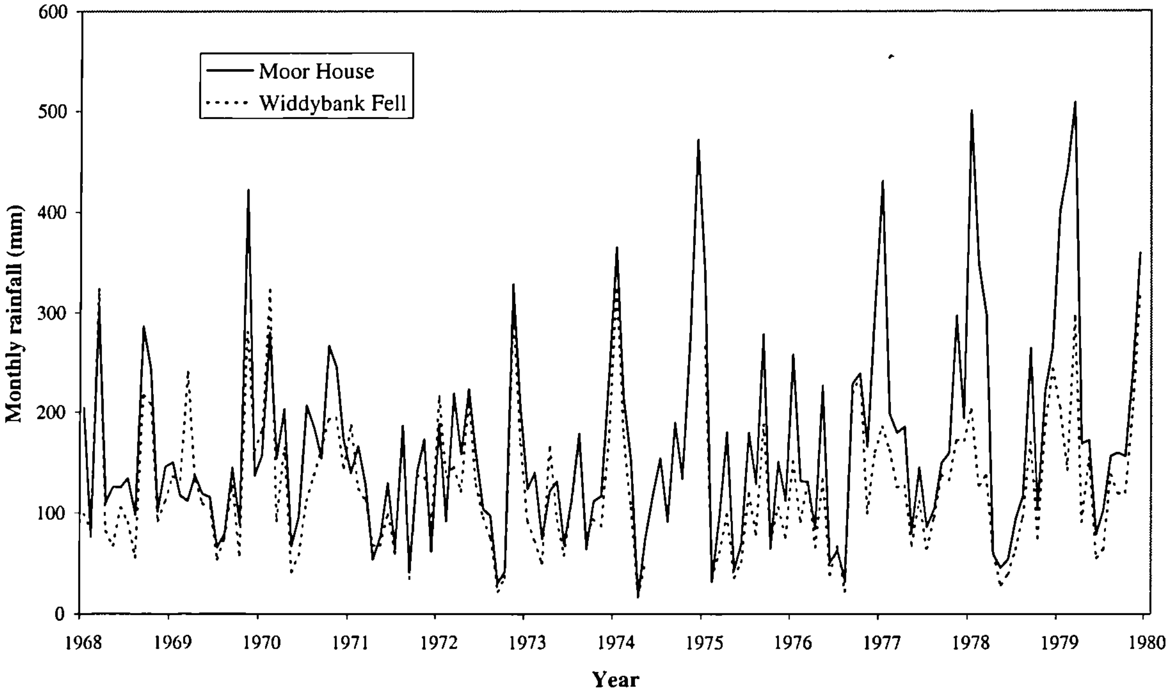


Figure 6.18 Comparison of Moor House and Widdybank Fell monthly rainfall totals for 1968 to 1979

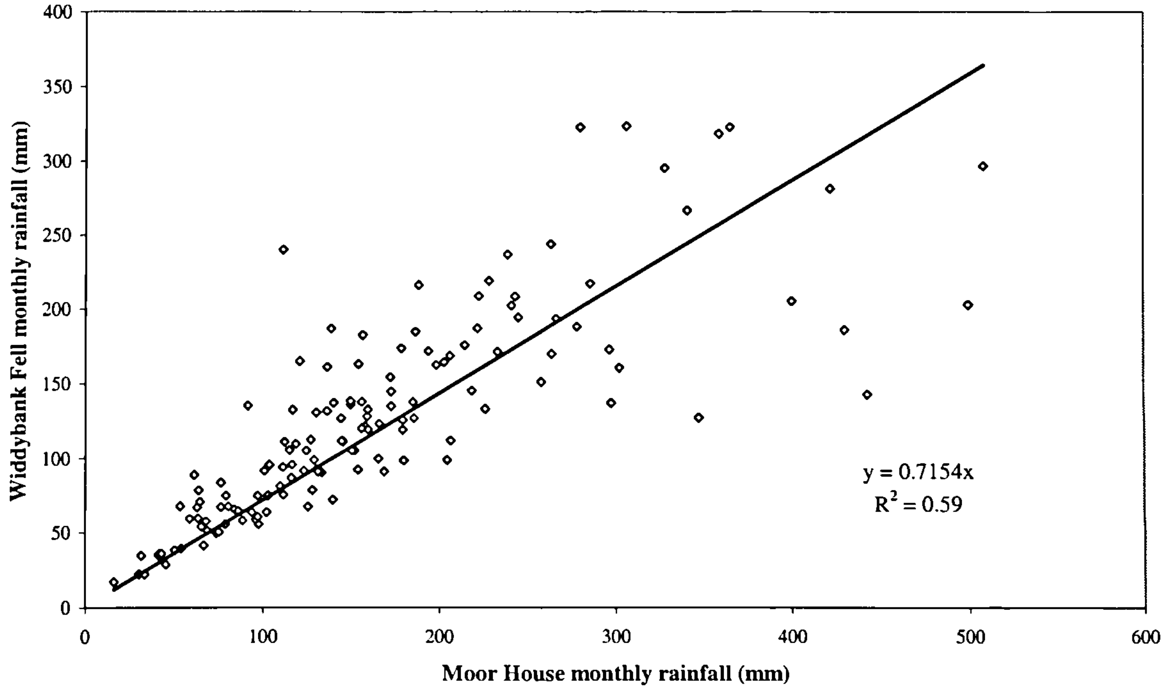


Figure 6.19 Relationship between Moor House and Widdybank Fell monthly rainfall totals, over period of overlap. The regression line has been forced through the origin

Table 6.9 Monthly rainfall differences (correction values; mm) between Moor House and Widdybank Fell

Month	Moor House-Widdybank Fell
January	17
February	24
March	14
April	30
May	37
June	96
July	69
August	53
September	29
October	29
November	37
December	24
Grand Total	458

The correction values shown in Table 6.9 were used to extend the Moor House rainfall record beyond 1979 when rainfall measurements ceased at Moor House. Although rainfall measurements (by the ECN) have begun again at Moor House over the last few years, they have not been used to extend the rainfall record since there have been changes in the location and measurement instruments.

An indication of the uncertainty caused by using the correction values has been determined by calculating the standard deviation of the difference between the monthly corrected Widdybank Fell and Moor House annual rainfall totals, for the period of overlap. These values indicate the uncertainty in the predicted rainfall values for Moor House and are shown as error bars in the final record.

The complete record of rainfall for Moor House is shown in Figures 6.20 and 6.21, with the former providing monthly totals and the latter the annual rainfall totals for Moor House. Distinct trends are not evident in either record, although variability in extremes of monthly rainfall seems to have been reduced over the past twenty years. Annual totals over the last twenty years have been around 2000 mm, whereas in the first half of the record totals appear to have been slightly less.

6.3.5 A C balance model for mineral and peat soils

Figure 6.22 shows the regressions between annual weight loss (%) against AET for the different blanket bog vegetation types. Although the r^2 values show that regressions were not particularly strong, most species decomposed more rapidly with increasing AET.

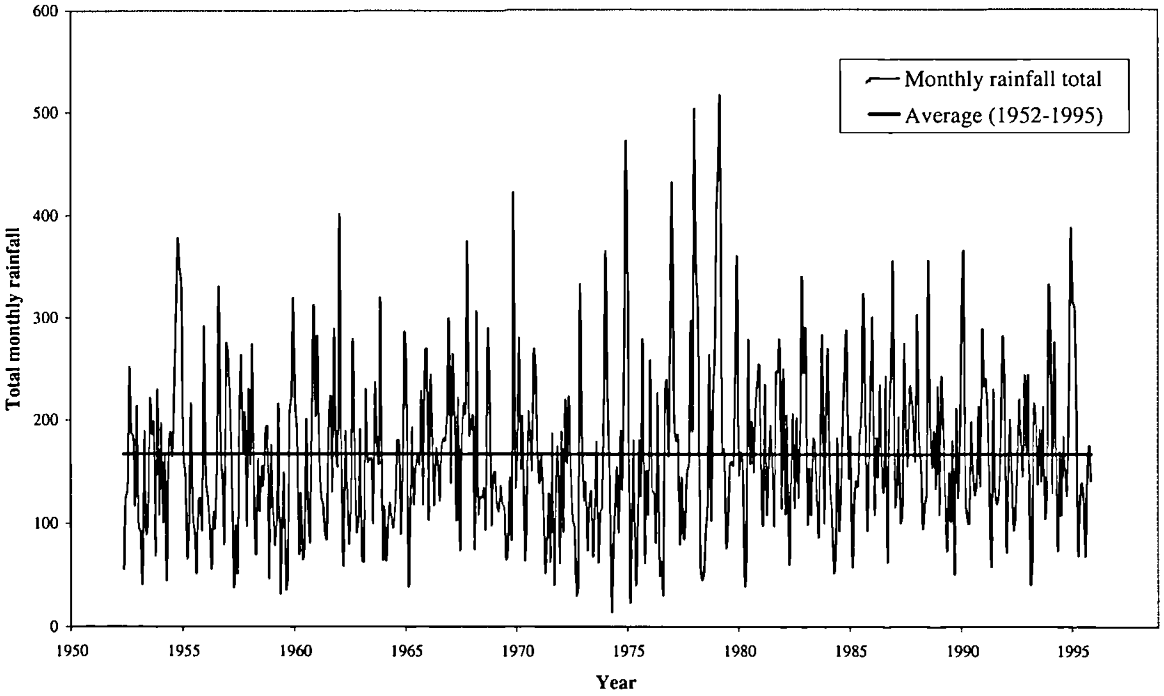


Figure 6.20 Monthly rainfall totals for Moor House between May, 1952 and November, 1995

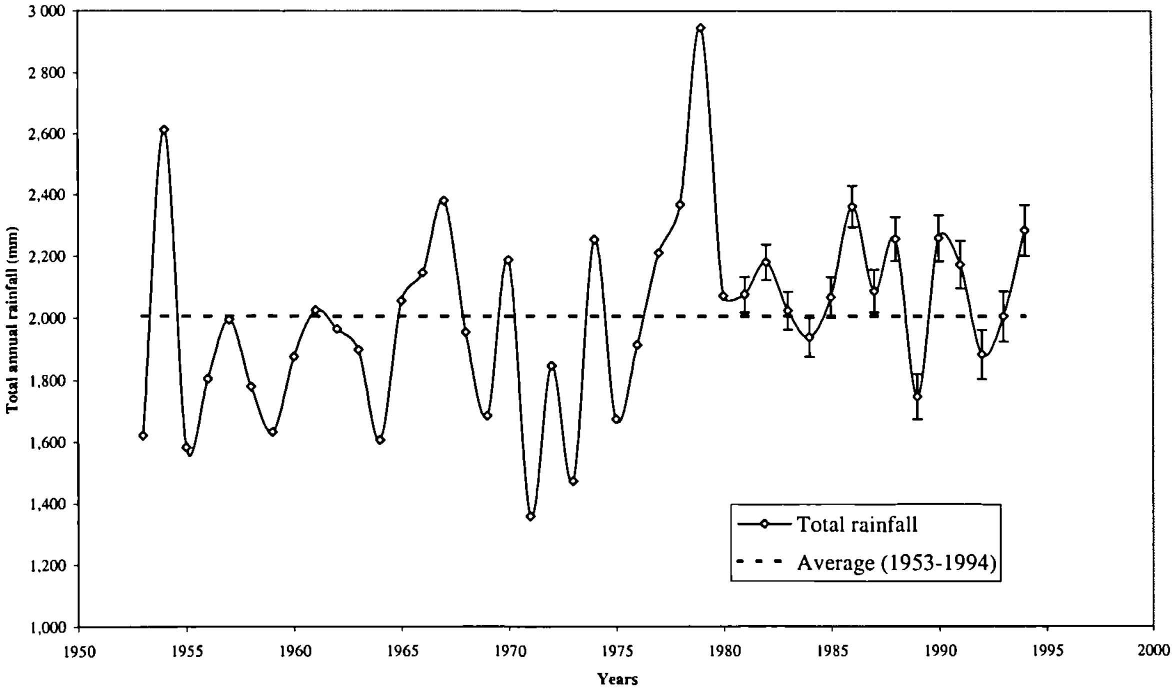


Figure 6.21 Total annual rainfall for Moor House between 1953 and 1994. Error bars indicate 95 % confidence limits for predicted rainfall totals, showing the uncertainty caused by using correction values.

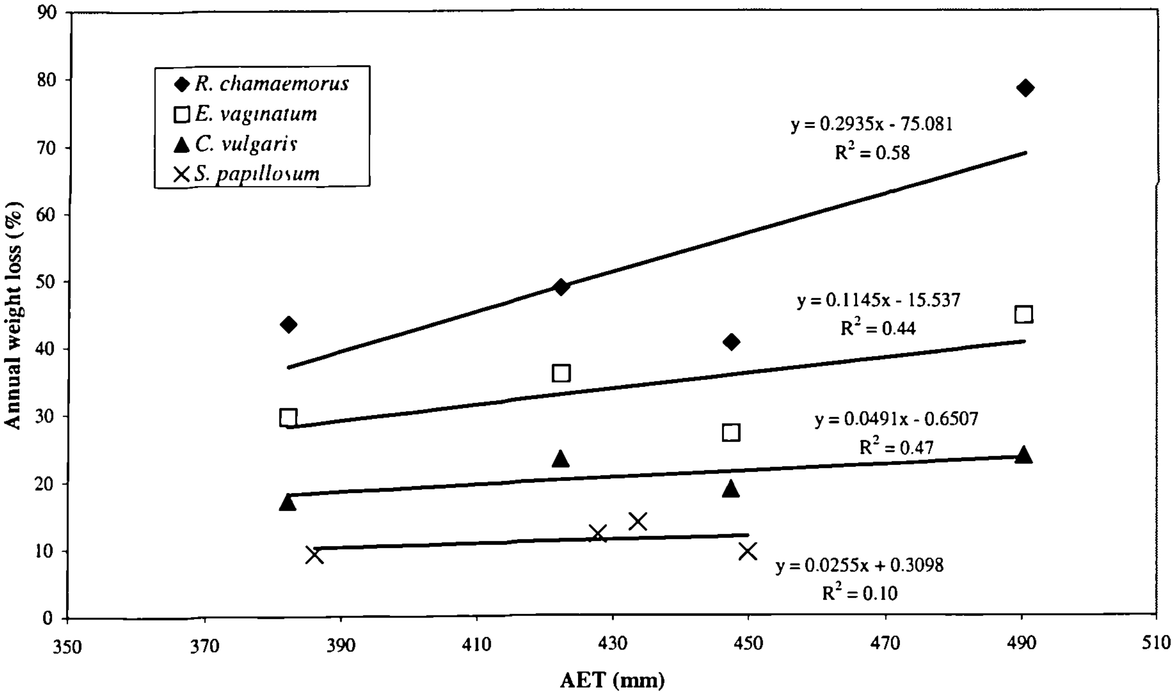


Figure 6.22 Annual weight loss (%) of different species in litter bags against AET (results for *R. chamaemorus*, *E.vaginatum* and *C. vulgaris* from Ineson *et al.*, in prep)

Furthermore, the species which decomposed more rapidly showed the greatest rate of increase in decomposition with AET.

C storage modelling

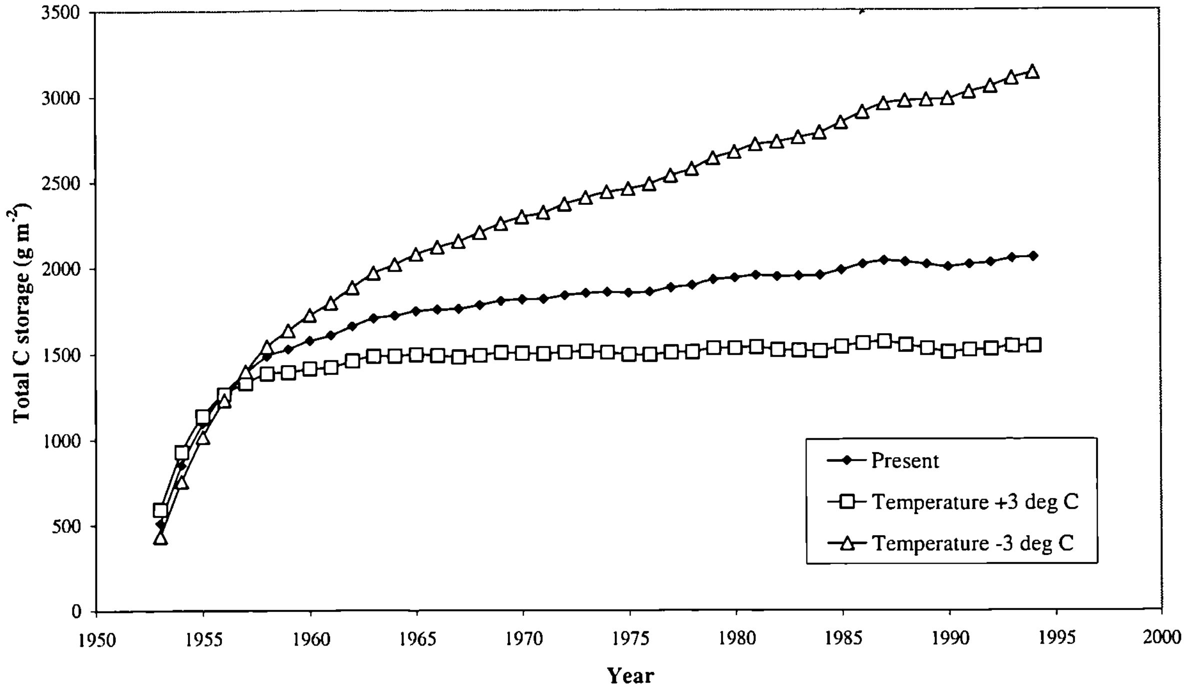
Figures 6.23 to 6.26 show the output from the C balance model for different scenarios of temperature, rainfall, water table depth and plant species composition. The estimate under the actual temperature and rainfall conditions is plotted and termed the 'present' scenario. The 'present' scenario assumes that the four species are equally abundant to determine the decomposition rate and calculates C budgets between 1953 and 1994, the period covered by both the long-term records of temperature and rainfall.

In the figures, the upper graph shows the total amount of C stored per square metre starting in 1953 and running until 1994. The lower graph shows the amount of C remaining from the initial input of NPP for each year, *i.e.* the rate of C accumulation. All figures for total C storage show a rapid rate of increase in the first few years following 1953 due to C stores being initially set to zero. All values of net C remaining from each year are highest for the most recent 'parcels' of C, since weight remaining is time dependent.

Figure 6.23 shows the influence of different mean annual temperatures on C storage rates compared to the 'present' estimate which is based on actual temperature and rainfall data. The 'present' total C storage increased slowly at approximately 10 g C m^{-2} yr^{-1}. However, when mean annual temperatures were elevated by 3°C for the period 1953-1994, C storage stabilised at *ca.* 1.5 kg m^{-2}. Reducing mean annual temperatures by 3°C increased rates of C accumulation to *ca.* 30 g C m^{-2} yr^{-1}. The lower graph in Figure 6.23 shows the higher rate of C accumulation when temperatures were reduced by 3°C, and shows that, although the initial input of C through NPP may be lower under reduced temperatures, the greater reduction in decomposition rates cause the net rate of accumulation to be higher.

In Figure 6.24 the effects of changing total annual rainfall are shown. When rainfall was halved, and other variables unchanged, the C accumulation rate increased. However, doubling the rainfall caused C accumulation to be only slightly reduced when compared against the modelled C storage under actual rainfall values. There was very little difference between the different rainfall scenario's in the amount of C remaining from each year.

The impacts of different water tables on C accumulation are shown in Figure 6.25. Four different scenario's were investigated: i) present conditions based on the measured water table at Shaft Hill in 1996, ii) no water table with aerobic conditions throughout the profile, iii) water tables depths halved, based on the Shaft Hill results, and iv) water table depths doubled, again based on the Shaft Hill results. There was very little difference in C accumulation when water table depth was increased from the present conditions, with



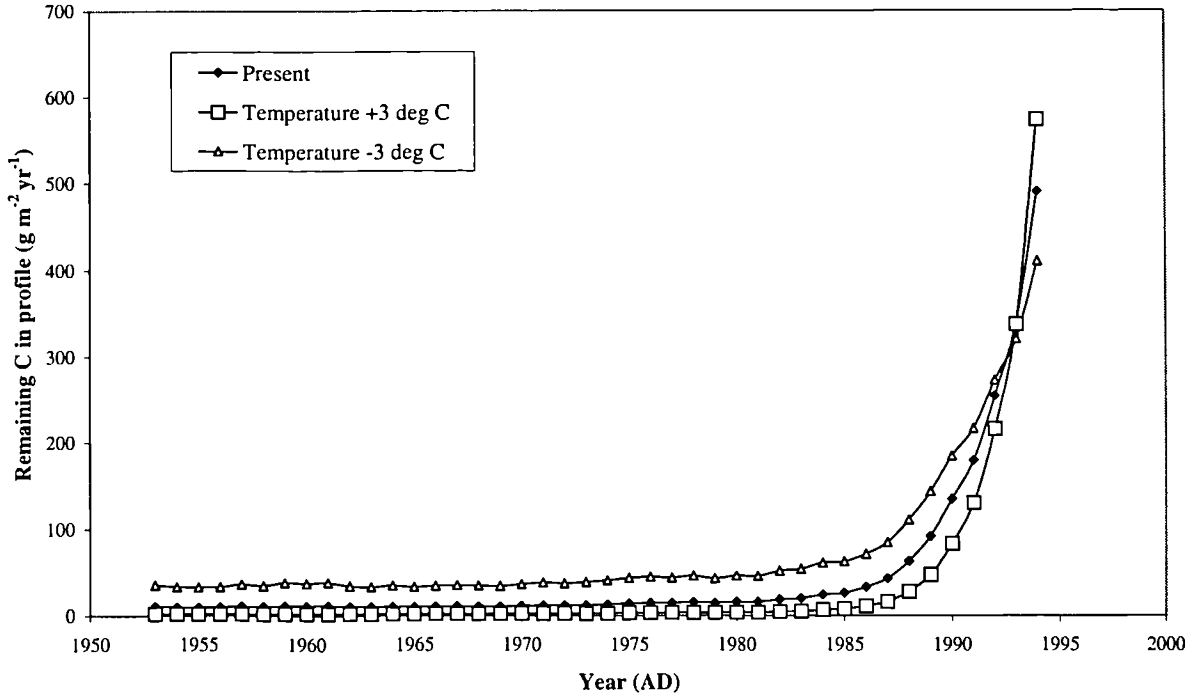
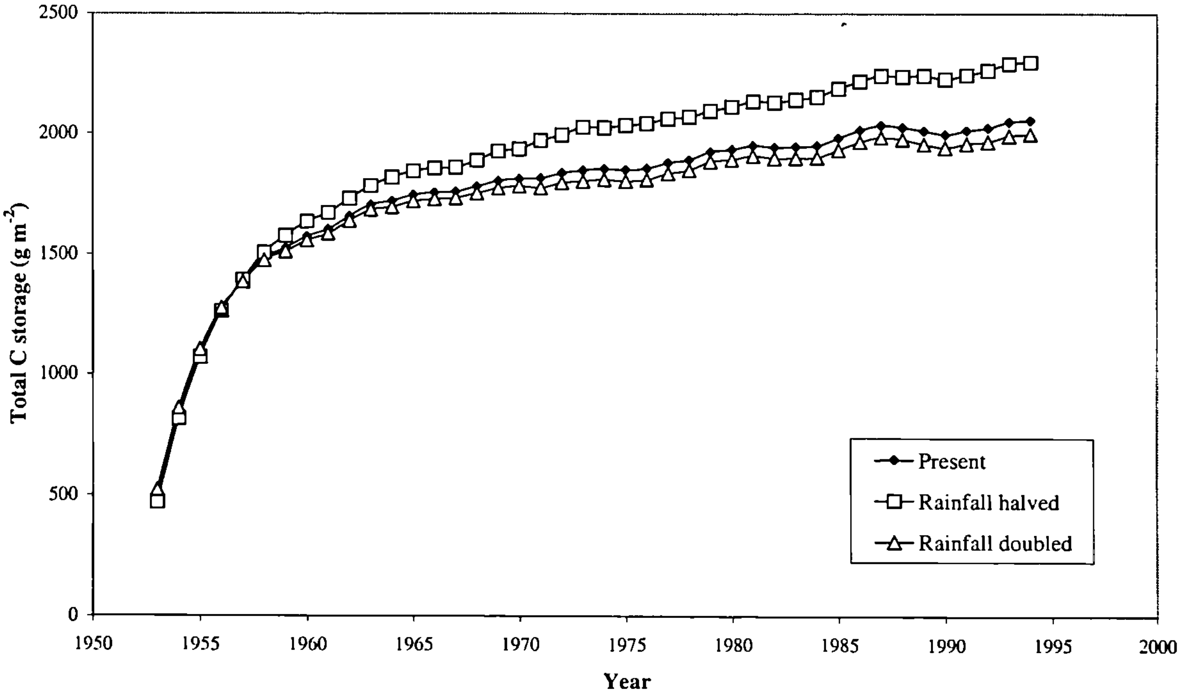


Figure 6.23 Simulated C accumulation in peat under different temperature scenarios. C storage against time (top) and C remaining in profile from each years NPP, in 1994 (bottom).



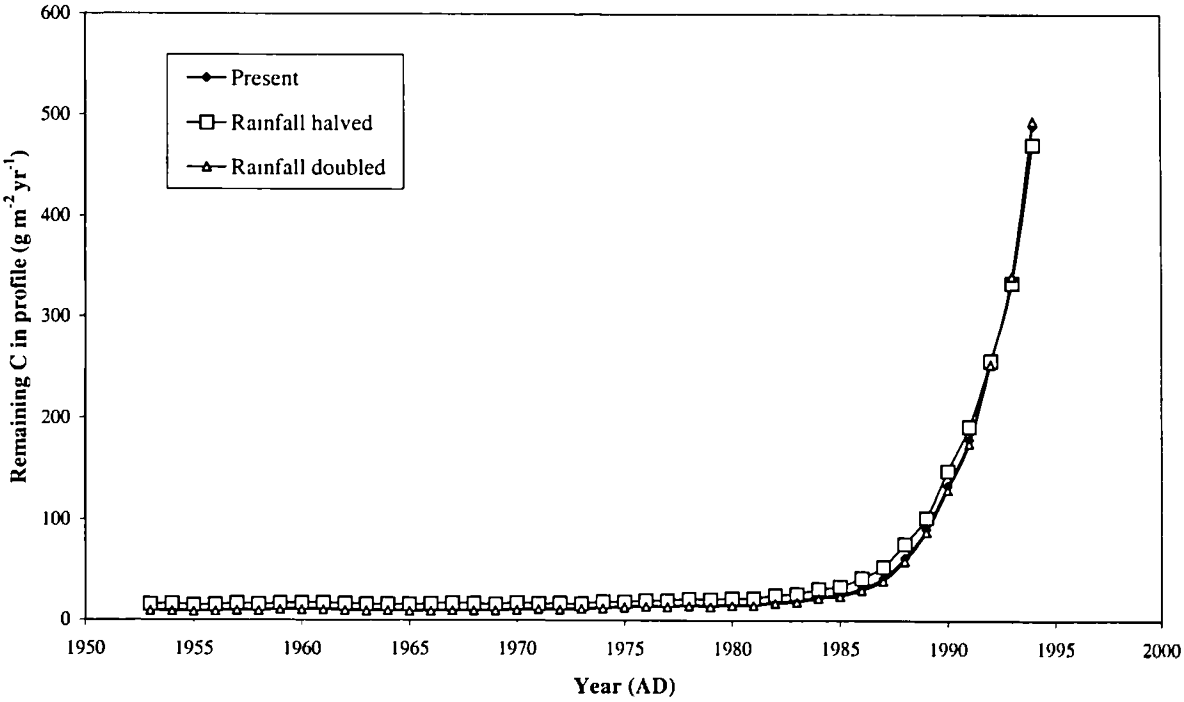


Figure 6.24 Simulated C accumulation in peat under different rainfall scenarios. C storage against time (top) and C remaining in profile from each years NPP, in 1994 (bottom).

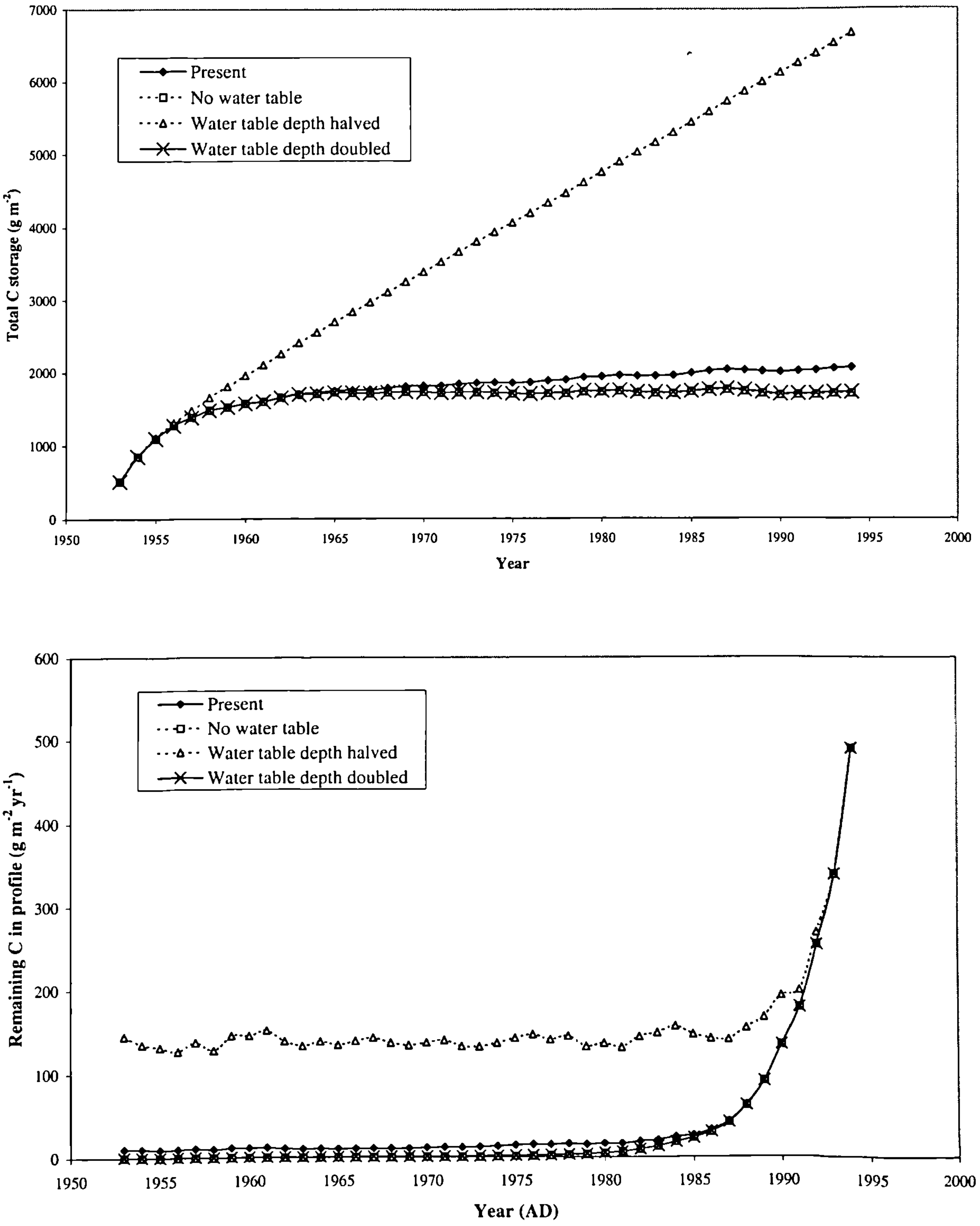
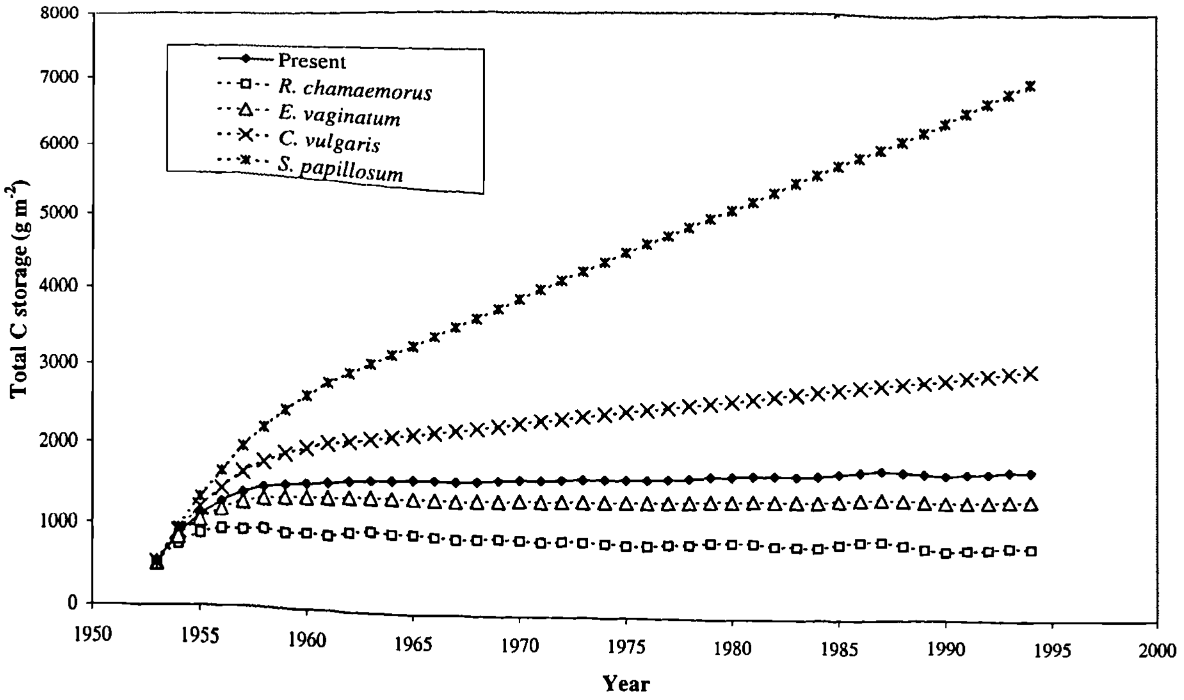


Figure 6.25 Simulated C accumulation in peat under different water table scenarios. C storage against time (top) and C remaining in profile from each years NPP, in 1994 (bottom).



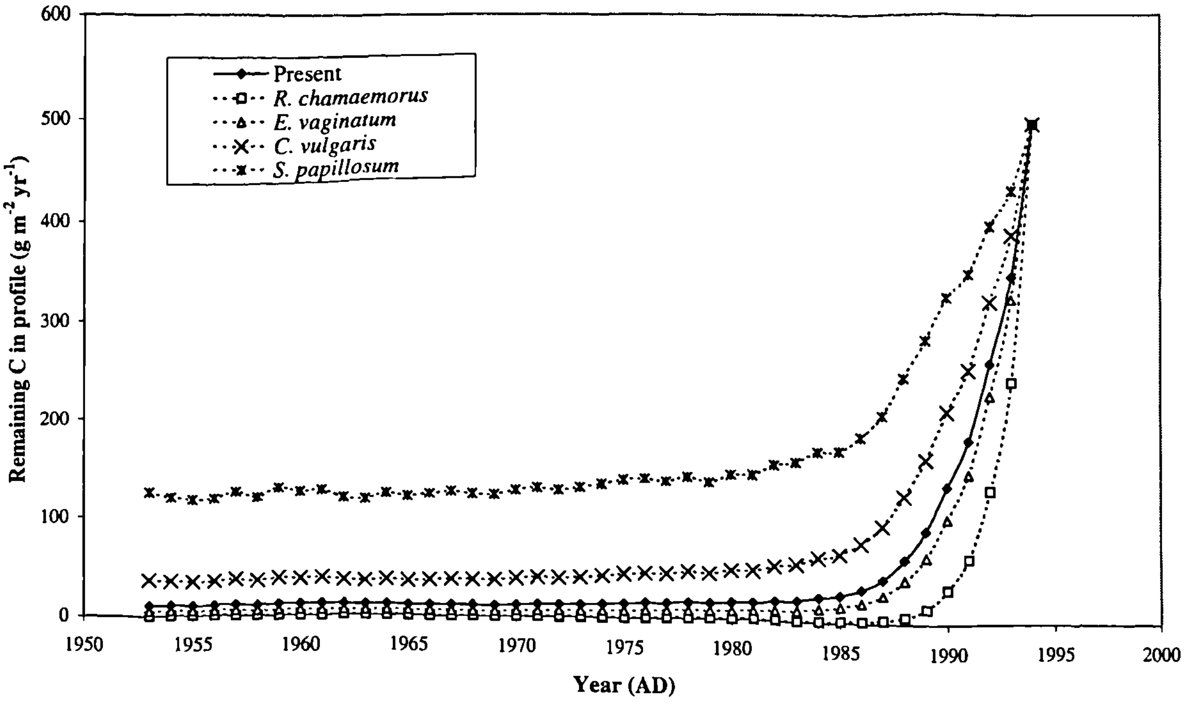


Figure 6.26 Simulated C accumulation in peat formed from different vegetation species. C storage against time (top) and C remaining in profile from each years NPP, in 1994 (bottom).

doubled water table depths and no water table producing very similar total C storage estimates. Both scenarios showed that C did not really increase in the soil and remained at *ca.* 1.5 kg m^{-2}. However, C accumulation rates greatly increased when water tables were halved, reaching a storage of under 7 kg m^{-2} after over 40 years. This was a result of C accumulation rates being maintained at *ca.* 120 g C m^{-2} yr^{-1}.

Figure 6.26 shows the effect of changing the proportions of different vegetation species on C accumulation rates by assuming that the vegetation was entirely composed of a single species. The 'present' conditions assumed equal proportions of the four species. As expected, C storage was lowest when the vegetation was composed of species which decomposed rapidly. *R. chamaemorus* produced a soil of only *ca.* 1.0 kg C m^{-2}. *E. vaginatum* produced a C storage slightly higher, though this stabilised at *ca.* 1.3 kg C m^{-2}. Accumulation of C was more rapid when the vegetation was assumed to be entirely *C. vulgaris* or *S. papillosum*, with rates of *ca.* 30 and 120 g C m^{-2} yr^{-1}, respectively.

6.4 Discussion

6.4.1 Climatic results for 1997

Comparison of the air temperature records from the four sites along the GDF transect with measurements for previous years shows that 1997 was slightly warmer than most years at all locations. The mean annual temperature at the Moor House field station was also one of the warmest in the long-term record, yet rainfall at most sites was within the range of values recorded over the few years of measurement.

AET has been estimated using Turc's equation which, although widely used (Shaw, 1994), is based on empirical relationships and can therefore only be considered an estimate. Therefore, the attempts to correlate NPP and decomposition rate with AET have errors associated with the AET estimates compounded by errors associated with measurements of NPP and decomposition rates.

6.4.2 NPP along an altitudinal gradient

NPP was measured at sites along the GDF altitudinal transect to establish whether the relationship between AET and NPP, as described by Leith (1975), appeared valid for the Moor House study area. Two independent approaches were used, although both estimated NPP as the difference between two measurements of vegetation biomass. However, there were large discrepancies between the estimates made using the two methods and the possible reasons for the differences in the estimates are discussed below.

NPP is defined as gross primary productivity minus autotrophic respiration (equation 6.2; Melillo *et al.*, 1996), *i.e.* the amount of C fixed by plants through photosynthesis minus plant respiration. However, estimates of NPP from the harvest methods used in the present study will have resulted in an underestimation of NPP because unmeasured heterotrophic respiration will have occurred. There was no sheep grazing within the exclosures, but, invertebrate grazing, an important source of error in NPP studies (Milner and Hughes, 1968), will have occurred.

Underestimation of NPP may have been greatest at the high altitude sites since Welch and Rawes (1965) noted that frosts, which could occur in any month of the year at high altitude at Moor House, can kill new vegetation growth. This reduces apparent rates of vegetation growth and causes an increase in the amount of litter available for respiration by decomposers.

At the start of the study both the transplant study cores and the fixed plots were clipped to reduce the natural heterogeneity in the above-ground mass of vegetation. It was thought that without reducing this variation it would be impossible to measure differences in biomass between the start and end of the growing season resulting from NPP. However, Milner and Hughes (1968) recommend that plots are not clipped at the start of NPP studies since it may actually affect rates of NPP. However, this should not affect comparison of results from the transplant study since all cores were clipped, although the different vegetation communities in the fixed plots may have responded differently to clipping. However, prior to the construction of the exclosures a few years ago, all sites had been grazed, and contain species resilient to grazing.

In addition to climate, vegetation growth is also limited by the supply of nutrients from the atmosphere (Schlesinger, 1991). There are different levels of atmospheric deposition of nutrients at different altitudes along the GDF transect (Fowler *et al.*, 1988) with, for example, nitrate levels at the summit of GDF averaging almost double those at the lowest site (Taylor *et al.*, accepted). However, increased nitrate deposition would be expected to increase rates of NPP, and therefore, if nitrate concentrations had a greater influence on NPP than climate, NPP would have increased with altitude. Consequently, differences in atmospheric deposition between sites does not appear to have had a strong influence on rates of NPP.

6.4.2.1 Transplant approach

The transplantation of intact cores of soil, complete with original vegetation, to sites along the GDF altitudinal transect subjected both soil and vegetation to different climatic conditions. At the start of the growing season, samples were standardised by cutting and any difference between them at the end of the study period was considered a consequence of the different climate under which the cores had been placed. The soil type and vegetation cover were specifically chosen for the study, with a soil of high mineral content being selected to enable below-ground vegetation to be accurately determined. The vegetation was dominated by fine grasses and again, could be easily sampled to determine mass of C. The boundary between soil and above-ground vegetation was well defined allowing the above and below-ground C to be easily distinguished.

However, the results suggest that the transplant approach did not appear to provide reliable estimates of NPP with negative values being found for several locations. The highest estimates of vegetation C (7 g C m^{-2} yr^{-1} for the NR site) were very low in comparison with published results for similar vegetation (*e.g.* 22–63 g C m^{-2} yr^{-1}; Rawes and Welch, 1969, assuming a C concentration of 45 %) and there may be several reasons why the estimates of NPP made in this transplant study were questionable: The soil and vegetation used in this study were enclosed in plastic cores which were open at the top and base. Although all estimates of vegetation C used in the transplant study were derived from the same types of cores at different locations, the physical constraints of the cores will have caused, for example, new root growth to have been restricted. Furthermore, hydrological conditions would have been altered because lateral flow of water into and out of the cores was not possible and any compression of soil occurring during core extraction may have further restricted water movement.

The cores were placed in trenches directly on a thin layer of coarse stones to improve drainage. However, it was observed on at least one occasion that the surface of some of the cores was water-logged after severe rainfall, unlike the surrounding areas.

The timing of the start and end of the experiment will have affected the estimates of NPP. It was largely through convenience that the cores were transplanted to the different sites along the altitudinal transect at the same time, and that they were retrieved on the same day at the end of the study period. Since the sites were located at different altitudes and experienced different climates, the start and end of the growing season will have varied from site to site with, for example, the growing season on the summit of GDF starting later and finishing earlier than for the other sites.

More accurate estimates of NPP may have been achieved if larger cores had been used. Larger cores would have meant the mass of vegetation C could have been more accurately measured with less soil compression during coring. Larger cores would have also reduced the initial natural high variance between cores at the start of the study. However, the smaller cores used in the NPP study were inexpensive and easy to install, enabling high replication.

Figure 6.8 shows that although above-ground vegetation C generally increased in the cores moved to the lower altitudes, the variability of above-ground C also increased in samples incubated at the lower sites. The values of above-ground C at most sites were around the range of values published for Moor House vegetation; Rawes and Welch (1969) provide a range of *ca.* 55 to 140 g C $m^{-2} yr^{-1}$ when a C concentration of 45 % is used; Allen, 1989).

Below-ground biomass was reduced in cores which were incubated at lower altitudes, suggesting that decomposition of dead roots within the soil responded more rapidly to the new climates than did the input of C from new root growth. However, Fitter *et al.* (1998) investigated root production and death along the same altitudinal transect and found that neither was strongly related altitude and that consequently, root populations in winter were similar at all sites. The conflicting results observed in the present investigation may have been caused by problems with the soil cores discussed above, possibly affecting root production more than root decomposition.

The estimates of NPP from the transplant study did not correlate with any of the climate variables tested, yet, it is not possible to say from the results of this study whether such correlations do not exist or whether the estimates of NPP made were too inaccurate.

6.4.2.2 Fixed plots

It was expected that the transplant approach would provide accurate estimates of NPP due to the initial standardisation of soil and vegetation types. It was thought that since soil and vegetation varied so markedly between the exclosures where fixed plots were established, rates of NPP would be strongly influenced by the soil and vegetation type. Furthermore, the accuracy which mass of vegetation C could be measured at the different sites was expected to vary since, for example, it is difficult to extract below-ground vegetation from the organic soils at HCB and SB.

However, the fixed plots approach appeared to provide more meaningful NPP estimates than the transplant study because samples were collected throughout the growing season, allowing the maximum mass of vegetation C to be estimated. The maximum value provides a better estimate of NPP than the amount sampled at the end of the growing season because the impact of heterotrophic respiration is minimised. For example, C fixed in the vegetation during the growing season may have been lost through heterotrophic respiration before the vegetation was sampled. Peak values in vegetation C were observed at most sites before the final (September) sampling and, therefore, the reduced amounts of vegetation C in the last samples were either a result of sampling error or heterotrophic respiration. Consequently, the estimates of NPP in the transplant study are likely to have been underestimates since the fixed plots showed greatest vegetation C before the transplants were retrieved. As a result, in the transplants, heterotrophic respiration will have reduced vegetation biomass and caused an underestimate of NPP. The underestimation caused by heterotrophic respiration is not totally removed by taking the maximum value in the fixed plot study but will have been reduced.

A further important factor reinforcing the belief that NPP estimation is more accurate using the fixed plots approach is that vegetation and soil were maintained under natural conditions and there were no changes in hydrological conditions, as in the transplant study. NPP may have varied between exclosures due to differences in individual species growth rates or site fertility, yet, Leith's (1975) regression demonstrates that across a wide range of different ecosystems with different soils these factors are of less importance than climate.

The estimates of NPP in the fixed plot approach appeared more reliable than the transplants results because the estimates were not greatly different to other published studies. However, comparison with other studies was difficult since below-ground production has generally not been measured and, due to difficulties in defining the boundary between soil and vegetation in the fixed plot samples, a direct comparison between above-ground production estimates has not been made. However, estimates of NPP using the fixed plots are slightly higher than the estimates of production given by Rawes and Welch (1969) for the same Moor House vegetation classes, which is considered to be due to including below-ground production in the fixed plot estimates.

The mean estimates for NPP increased with decreasing altitude. When plotted against the three climatic variables, NPP showed the best correlation with AET, although a good correlation was also found with annual temperature.

6.4.2.3 Comparison with Leith's (1975) regression and other estimates of NPP

The values of NPP estimated using the fixed plots are shown in Table 6.10 together with the calculated predicted values of NPP derived simply by using Leith's (1975) regression with AET.

Table 6.10 Estimated NPP for the GDF transect using the fixed plots. Units are g C m^{-2} yr^{-1} and dry mass has been converted to C for the estimate using Leith's (1975) regression assuming a C concentration of 45 % (Allen, 1989).

Site	Fixed plots	Using Leith's regression
GDF	117	418
KF	206	441
HCB	341	459
SB	247	475
NR	365	487

The estimate of NPP from the fixed plots was lower at all sites than the estimate based on Leith's (1975) regression. Furthermore, the difference between the two estimates was greatest in the sites with the lowest AET, and diminished as AET increased.

The lower values of NPP from the fixed plots may simply reflect the underestimation of NPP caused by determining the difference in vegetation mass between two harvests (Rawes and Welch, 1969), which fails to account for heterotrophic respiration during the growing season. The increasing difference between the estimates with lower AET may be due to the reduced accuracy of the difference method to estimate NPP with altitude as discussed by Welch and Rawes (1965) and mentioned above.

Leith's (1975) regression is based on a global data set from sites with AET values ranging from below 100 mm to greater than 1000 mm, which is far greater than the small AET range used in the Moor House study (402 mm - 482 mm). The equation provided by Leith (1975; equation 6.6) best describes his global set of data with considerable scatter in the values of NPP estimated for the different AET values (see Leith, 1975 p. 256); the fixed plot NPP results fit well within the scatter used to derive Leith's (1975) regression.

The fixed plots study has demonstrated that NPP is more strongly related to AET than mean annual temperature or total annual rainfall at Moor House, confirming the general relationship derived from global data sets by Leith (1975). It is believed that the regression between AET and NPP provided by Leith (1975) can be considered adequate for modelling NPP of vegetation at Moor House within a C balance model.

6.4.3 Decomposition along an altitudinal gradient

The litter bag study was intended to determine the influence of climate on the decomposition rate of *S. papillosum*. In particular, the relationship between decomposition rate and AET, reported by authors for other litters (*e.g.* Berg *et al.*, 1984) was investigated. Unfortunately, the experiment was not successful because, unexpectedly, the litter bags were colonised by algae actually increasing the weight of organic matter in many of the bags, and reducing the apparent decomposition weight loss in others.

However, algal contamination seems only to have occurred at the lowest altitude site over the first six months of incubation and weight loss rates were obtained for four sites after six months decomposition. Although it is not possible to be entirely certain that no algal growth occurred on these samples, careful microscopic examination failed to reveal any evidence for algal colonisation.

The results of the six month decomposition studies have been extrapolated using Olson's (1963) equation. In applying Olson's equation an assumption is made that weight loss of decaying plant material is exponential. The calculated values of annual loss from the study ranged between 9 % and 12 % and are similar to other estimates for the annual weight loss for *S. papillosum*. For example, Clymo (1965) calculated weight losses for *S. papillosum* of *ca.* 5.0 %. Belyea (1996) in a similar litter bag study measured weight losses for *S. papillosum* in litter bags placed on the surface of between 6 % and 9 %.

However, Latter *et al.* (1998) found in a 23 year litter bag study that the weight loss of several species on the blanket bog at Moor House, was best described by an asymtotic curve. The apparent cessation of decomposition in the litter bags of Latter *et al.* (1998) was probably due to the bags being buried by new peat and the gradual incorporation of the bags below the water table; bags were at a depth of *ca.* 12-15 cm after 23 years and possibly under anaerobic conditions for much of the year (Latter *et al.*, 1998). Therefore, the results of the present litter bag study have been modelled using Olson's (1963) equation with the impact of the water table being specifically incorporated in the C balance model.

The decomposition study conducted here was designed to determine the affect of AET on decomposition rates. Litter bags were placed at different locations along an altitudinal transect with as many variables as possible, other than climate, being standardised. For example, at most sites the bags were placed on peat transplants typical for the Moor House blanket bog, containing *C. vulgaris*, *E. vaginatum* and *Sphagnum* sp. However, transplantation had caused changes in the vegetation composition and structure of the transplants with, for example, a reduced vegetation cover existing in the transplant at the GDF site. Colonisation of the peat monoliths by non-blanket bog species also occurred at the Newton Rigg site and living *Sphagnum* sp. were largely absent from all the peat transplants. Therefore the micro-site conditions under which the *S. papillosum* litter bags were incubated were different despite the attempts to standardise. If *Sphagnum* sp. had still remained within the peat transplants then the litter bags could have been inserted into the top few cm of living

Sphagnum after Clymo (1965) and Belyea (1996), which may have prevented algal growth in bags.

The results from the decomposition studies of Ineson *et al.* (in prep) using the same peat transplants showed that weight loss increased with increasing AET as shown by Berg *et al.* (1984). However, the rate of increase in decomposition rate with increased AET varied with species type (Figure 6.22). The species which decomposed at the fastest rates also showed the greatest increase in decomposition at higher AET values. Therefore it was perhaps predictable from the results of Ineson *et al.* (in prep) that the weight loss of *S. papillosum* would not vary greatly between the different sites along the GDF altitudinal transect, particularly as Clymo (1965) and others have shown that this species decomposes very slowly.

6.4.4 Long-term records of temperature and rainfall for Moor House

6.4.4.1 Temperature

Ideally, a long-term record of temperature would be produced by the accumulation of temperature measurements using the same instruments at the same location and using identical protocols. Such records are extremely rare and reconstruction of temperature records using sequential and overlapping data series becomes essential. It is believed that such records can be accurate if the appropriate corrections are made for changes in recording methods.

The comparisons of different temperature records were successful in the current work, as illustrated by the r^2 values, allowing reliable correction values to be derived and making concatenation of the records possible. Some uncertainty in the final record is introduced because the monthly corrections used represent average monthly differences between parallel records, ignoring inter-annual variations. However, this uncertainty was small, as indicated by the error bars on Figure 6.17.

When compared with records of global temperature over the same period (*e.g.* the combined global land-surface air and sea temperature; IPCC, 1996) both similarities and discrepancies can be observed. Both records show the 1930's and 40's to be relatively warm, with a cooling trend through the 1960's and early 70's. However, the recent 'record-breaking' years of warmth found in the IPCC assessment were not found in the Moor House record.

There are several reasons which may account for this discrepancy. Firstly, global warming will not be spatially uniform around the world with both global circulation models and maps of recent global temperature change suggesting different responses to global warming over different areas (IPCC, 1996). Indeed, while records indicate global temperature has increased in the last few decades, the extreme west of Europe and north-east Atlantic have only slightly warmed, or even cooled (IPCC, 1996). This may account for the lack of recent warming suggested from the Moor House temperature record.

Alternatively, Moor House may not be changing in the same way as other monitoring stations due to location, particularly altitude. Even the 'record-breaking' years of warmth in the last decade recorded elsewhere in England are not prominent in the Moor House record within the context of the last 60 years. This suggests that lapse rates may be modified under climate change, as discussed by Pepin (1995) with the implication that local changes in climate may be difficult to predict.

The record of temperature produced for the Moor House meteorological station provides a rare long-term description of past temperature for an upland location in Britain. Some similarities with records of global temperature are apparent from this upland record, though recent decades do not appear to have undergone the same increase in temperature seen at other locations, at different altitudes. It is suggested that under changing global climates, the extent of temperature changes in the uplands and lowlands may differ.

6.4.4.2 Rainfall

The record of rainfall for Moor House has been extended by correcting rainfall measurements made at the nearby Widdybank Fell meteorological station. The correction factors, derived by comparing periods of overlapping measurements also appeared to be a successful approach as shown by the error bars in Figure 6.21.

The record of rainfall was not investigated in detail in the present study, although rainfall totals do appear to have increased over the last twenty years at Moor House. Burt *et al.* (in press) have recently made a more detailed investigation of the Moor House rainfall records and concluded that variability has increased over recent years. In particular, these authors suggest that the contrast between winter and summer rainfall has become more extreme.

Although a detailed time series analysis of the long-term rainfall record has not been undertaken in the present work the record of rainfall has, together with the temperature record for the site, provided the necessary long-term climatic data required for the C balance model.

6.4.5 A C balance model for mineral and peat soils

The C balance model described in the present chapter is based on simple empirical relationships between climate and the processes controlling the input and output of C in terrestrial ecosystems. The model contains several assumptions which may limit the accuracy of simulating the C balance in terrestrial ecosystems, but the model allows the sensitivity of different processes which influence C accumulation at Moor House to be investigated.

A major assumption in the model is that NPP can be predicted from AET, since Leith (1975) observed in the results of productivity studies from across the world a relationship between these two variables. The measured values of NPP in the present study increased with increasing AET, as predicted by Leith's (1975) equation, although there were differences in both the actual values of NPP and the rate of increased NPP with increased AET. However, as discussed above, there were many sources of uncertainty in the estimates of NPP from the GDF transect and it is not possible to say whether the observed differences in NPP estimates are due to error in the Moor House NPP results or whether the equation of Leith (1975) is less valid for this site. The NPP data confirmed the conclusion of Leith (1975) that AET correlated better than temperature and rainfall with NPP.

Decomposition rates were determined from relationships between weight loss and AET for different species. However, the litter bag study in the present work failed to produce a strong correlation with AET, as has been observed by others (*e.g.* Meentemeyer, 1971; Berg *et al.*, 1984), largely due to technical difficulties. This may partially have been caused by specific difficulties experienced in using *S. papillosum* litter as Ineson *et al.* (in prep) found stronger relationships between weight loss and AET for other, more decomposable, plant litters (Figure 6.22). However, the model assumes that the relationships between weight loss and AET are valid.

A major assumption of the model is that decomposing litter is buried by accumulating organic matter at a constant rate of 0.5 cm per year. This rate is based on the empirical data of Latter *et al.* (1998) who observed the rate of litter bag burial in accumulating peat at Moor House. However, it is unlikely that the rate of burial of plant material in peats is constantly at 0.5 cm per year since, as organic material accumulates, the increased pressure of overlying material causes compression and increased density of lower layers. Furthermore, the rate of depth increase will also partly be related to the rate of C accumulation at the surface since this will in turn influence the amount of pressure caused by overlying plant material.

The approach to simulation of water table depth was very simple in the C balance model, being derived from a single years measurement at Shaft Hill. Above the water table, decomposition has been assumed to be at a constant rate, although, Belyea (1996) found higher rates of decomposition at or slightly above the water table.

The variables used in the model scenarios were assumed to act independently. This is unlikely to be the case in reality with, for example, increased rainfall totals leading to changes in the water table. However, by manipulating the variables independently the sensitivity to the different parameters with the model was investigated.

Despite the many assumptions made by the model, the actual values of C accumulation and C storage were similar to the observed values from the results at Moor House. For example, when the water table factor was removed, the C storage of the soil was found to stay at about 1.0 kg m^{-2}. This is not much lower than some of the more freely-draining soils at Moor House (*e.g.* Solifluxion creep, 2.9 kg m^{-2}; Madeground, 4.3 kg m^{-2}).

Under the actual temperature and rainfall conditions experienced at Moor House between 1953 and 1994, and with the water table simulated, C was predicted to accumulate at a rate of *ca.* 10 g m^{-2} yr^{-1}. The average rate of C accumulation for the MH2 core in Chapter 5 was estimated to be 27 g m^{-2} yr^{-1}, and rates were calculated between < 10 g m^{-2} yr^{-1} to *ca.* 90 g m^{-2} yr^{-1}. Furthermore, if the proportion of species in the model was altered to reflect the typical vegetation composition at Moor House, *e.g.* as in the surface sample of core MH2 (Figure 5.8), C accumulation rates were even closer to the mean values observed for the core. For example, when the vegetation was assumed to consist of 10 % *R. chamaemorus*, 20 % *E. vaginatum*, 20 % *C. vulgaris* and 50 % *S. papillosum*, C accumulation rates in the catotelm were *ca.* 30 g m^{-2} yr^{-1}.

When temperature was increased in the model by 3°C and other variables remained constant, the rate of C accumulation decreased. This decreased rate of C accumulation was a result of decomposition rates increasing relative to NPP at the new temperature. This is illustrated using Figure 6.27. This figure shows the net C remaining after 1 year at a range of AET values and for different vegetation species. The net C remaining after one year has been calculated by determining the NPP and rate of weight loss for the different species under the different AET values, the net C remaining being calculated as:

$$\textit{Net C remaining} \quad = \quad NPP - (\ NPP \ \ \mathrm{x} \ \ (\ \%wt\ loss\ /\ 100\ )) \qquad \textit{[equation 6.7]}$$

The figure is therefore a plot of the C added to the system after 1 year of decomposition and the curves represent the difference in response of NPP and weight loss to different AET values. For example, the amount of *R. chamaemorus* remaining after 1 year decreases with increasing AET above 300 mm per year, due to the higher rate that decomposition occurs with higher AET for this species. However, *S. papillosum* only decomposed slightly more rapidly under higher AET (Figure 6.22) and, therefore, NPP was more sensitive to the changing conditions and resulted in an increased rate of C accumulation. The diagram shows that the actual response of a terrestrial ecosystem to climate change in terms of being a C sink or source will be strongly influenced by the species

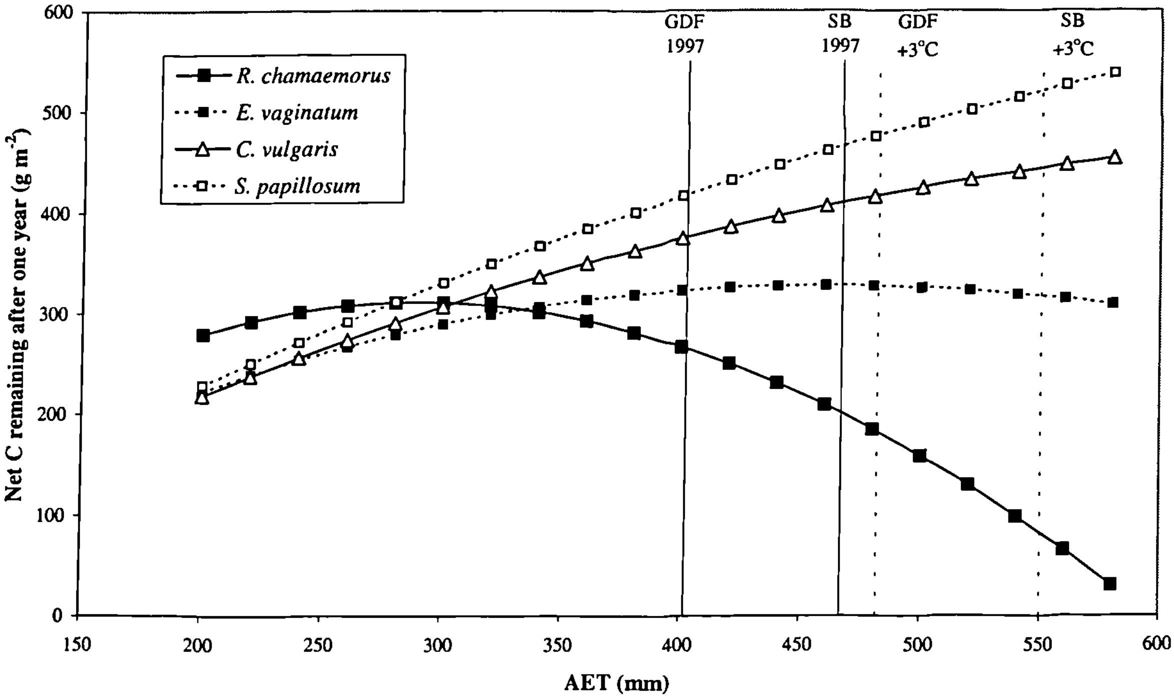


Figure 6.27 Net C remaining after one year for different species under different AET values. Vertical lines indicate AET at GDF and SB in 1997 (solid) and under a climate 3°C warmer than 1997 (dashed).

composition of the vegetation (assuming no changes in other variables, *e.g.* water tables). This suggests that following climate change, ecosystems which increase in the proportion of species with low decay rates, such as *S. papillosum*, will increase rates of C storage.

In Figure 6.27, the values of AET for 1997 at the GDF and SB sites have been identified, together with the anticipated AET for the sites if temperature was raised by 3°C and other variables remained constant. Under a warmer climate the amount of *R. chamaemorus* remaining after one year would be much lower, while *S. papillosum* would increase.

Increased rainfall caused a lower rate of C accumulation in the C balance model, because the AET values, which were partly derived from the rainfall totals, increased. However, the increased rainfall would in reality affect the water table conditions and therefore, the small changes in C balance caused by changing the rainfall totals would probably be of less importance than the impact of changing water tables linked to rainfall (see below).

The consequences of changing water tables on C accumulation was assessed by removing the water table factor altogether, doubling and halving the depths of water tables. Increasing the depth or totally removing the water table caused only a slight change, with reduced C accumulation rates and total C storage remaining approximately constant. However, when water table depths were halved, simulating a shift to wetter mire conditions, C accumulation greatly increased. This effect was caused because, although both NPP and % weight loss were unchanged, the proportion of aerobic time for the litter to decompose was halved, and therefore, more C entered the lower depths sooner. The model response to increased mire surface wetness through halved water table depths supports the observation from core MH2 where the main wet shift was synchronous with a rapid increase in the rate of C accumulation (Figure 5.16).

The impact of different vegetation compositions has already been discussed with reference to Figure 6.27 (see above). However, it is worth noting that a shift to wetter conditions, such as by halving the water table depth is likely to increase the proportion of *Sphagnum* species. Clymo and Hayward (1982) have shown that *Sphagnum* sp. generally have low rates of decomposition and it is likely that they show only slight increases in decomposition rates with increasing AET, as observed for *S. papillosum* in the present study. Consequently, a shift to wetter mire conditions is even more likely to increase C accumulation rates because not only will decomposition be reduced due to plant litter experiencing less time under aerobic conditions, but species composition is likely to contain increased *Sphagnum* (since this genus prefers wetter conditions; Chapter 5), therefore causing a reduction in litter quality and further decrease in decomposition.

Alternatively, a drier mire surface will be less favourable to *Sphagnum* sp. and more favourable to species which decompose more rapidly. Combined with the lower water tables, C accumulation would be expected, as shown in the model, to decline.

Clymo's (1984) model of peat accumulation predicts a limit to peat growth when rates of mass input from vegetation equal the total amount of peat decomposed from the entire peat profile. This is largely a result of decomposition within the catotelm which although occurs very slowly, contributes increasingly as more peat accumulates in the catotelm. The C balance model described in the present study does not reach a limit in most cases because the decay rates in the catotelm are assumed to be nil and because the model only runs for a few decades. Inclusion of decay rates for the catotelm would therefore not significantly affect the response of the model in its present form, although would be necessary if allowed to run over a much greater time scale.

7. Use of near-infrared reflectance spectroscopy in palaeoecological studies of peat

7.1 Introduction

Palaeoecological studies of peat deposits have proven particularly useful for reconstructing vegetation histories (*e.g.* for pollen analysis; Roberts, 1989) and for providing proxy records of past climatic change (see Chapter 5; Barber *et al.*, 1994; Tallis, 1995). However, such investigations have frequently been limited because the analyses are time consuming, relying for example, on manual identification of individual fossils. Consequently, the number of analyses that can be performed have tended to be restricted, resulting in reconstructions with limited spatial and temporal resolution. For example, single peat cores are often used and the results assumed to be representative of wider areas. Furthermore, samples are often analysed at large depth intervals down peat profiles (typically 4 cm; Stoneman, 1993; Barber *et al.*, 1994) reducing temporal resolution of reconstructions. There is a clear need for more rapid methods of analyses for use in these types of palaeoecological investigations.

Near-infrared reflectance spectroscopy (NIRS) is a rapid and non-destructive technique for determining chemical and physical characteristics of organic samples (Malley and Nilsson, 1995). Although the technique has been used extensively in the food and feed industries, application to ecological studies is rare (Malley and Nilsson, 1995; Joffre *et al.*, 1992). However, NIRS has been used for performing leaf chemistry analyses in decomposition studies (Joffre *et al.*, 1992), for predicting methane production from peats (Nilsson, 1991) and for estimating lake water chemistry from surface sediments (Nilsson *et al.*, 1996). In palaeoecological studies the technique appears to have only been applied to lake sediments; Korsman *et al.* (1992) and Bowes (1996) used NIRS to reconstruct the past pH of lakes. Prior to the current work there had been no attempt to use NIRS to quantify the range of variables routinely analysed in palaeoecological studies of peat.

NIRS involves the measurement of energy reflected from samples illuminated by radiation with wavelengths around the near-infrared region of the electromagnetic spectrum. Different molecular bonds within samples account for the variations in the reflected energy at different wavelengths, and therefore, NIRS spectra are characteristic of the chemical composition of samples (Malley and Nilsson, 1995). However, determination of the chemical composition of samples using NIRS requires the development of calibration models between the spectra and reference samples; consequently, NIRS determinations can only be as accurate as the reference values used to derive the calibration models.

The current study was undertaken with the collaboration of Kevin McTiernan[1] who was developing the use of NIRS to determine chemical characteristics of decomposing leaf litters (McTiernan, 1998). The application of NIRS described in this chapter was a collaborative project between Kevin McTiernan and the present author, and some of the results have been co-published in McTiernan *et al.* (in press).

The main objective of the investigation was to determine whether the physical characteristics and plant macro-fossil composition of peat samples could be correlated with NIRS determinations (principal calibrations). A second component of the study was to determine the minimum number of samples which needed to be analysed using conventional techniques in order to produce robust calibration equations.

7.2 Method

7.2.1 Reference values

The peat core (MH2) used to derive C accumulation rates and a proxy record of climate in Chapter 5, was also used in this chapter; details of the coring location, coring method and sectioning of the core can be found in Section 5.2. Many of the variables measured from core MH2 have been investigated in the present chapter to determine the potential for quantification using NIRS. The variables tested included the physical characteristics and plant macro-fossil composition (Table 7.1) of samples; as explained in Chapter 5, physical characteristics were determined by the present author and Dmitri Mauquoy performed the macro-fossil analyses. With the exception of samples used for degree of humification, all analyses were carried out on separate sub-samples from core MH2. Descriptions of the methods used to determine these variables have already been described in Section 5.2.

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[1] Department of Biological Sciences, University of Lancaster

Table 7.1 Characteristics of peat samples investigated with NIRS

Plant macro-fossils (%)	Physical characteristics
Unidentifiable organic matter (UOM)	Radiocarbon age (BP)
Ericales roots	Delta ^{13}C
Aggregate monocotyledons	Degree of humification (absorbance)
Identifiable *Sphagnum*	Bulk density (g cm^{-3})
S. magellanicum	Moisture content (%)
S. papillosum	
S. section Cuspidata	

7.2.2 Acquisition of spectra

Sub-samples (approximately 5 g dry weight) were removed from each section of core MH2, oven dried (60°C) and ground using a ball mill (Glen Creston) fitted with an agate chamber. The samples were wrapped in aluminium foil, followed by a layer of polythene film, and posted to the Centre National de la Recherche Scientifique (Montpellier, France) where Kevin McTiernan performed the NIRS scans.

To obtain the NIRS spectra, the ground samples were packed into sample cells with quartz-glass windows and scanned using a spectrophotometer (NIRSystems 6500, Perstorp Analytical). The instrument measured the energy reflected over a spectral range of 400 to 2500 nm at 2 nm intervals, while the sample cell was being spun at high speed. The full spectral range was scanned for each sample 64 times and the measurements combined. Samples were emptied, repacked into the sample cell, and scanned again. This procedure was repeated once more and the spectra from all scans combined to give a single spectrum for each sample. Reflectance (R) was converted to absorbance (A) using the formula:

$$A = \log(1/R) \qquad \textit{[equation 7.1]}$$

7.2.3 NIRS calibrations

7.2.3.1 Principal calibrations

All calibrations were performed by Kevin McTiernan using Infrasoft International (ISI) software (Infrasoft International, Silverspring, USA). Prior to calibration, various forms of scatter correction were applied to the spectra in order to reduce unwanted variation in the spectral data (McTiernan *et al.*, in press) and to determine whether this correction improved calibration models. After scatter correction the spectra were transformed into first or second order derivatives with the number of data points (1 data point = 2 nm) over which the derivatives were taken and smoothed, being varied. The combination of scatter correction and mathematical treatment which resulted in the best calibration for each treatment was selected.

Stepwise regression calibrations were performed for the macro-fossils, delta ^{13}C and radiocarbon age, since few values were available. This procedure selected the wavelength most highly correlated with the reference values, re-evaluated the remaining spectra, and then added the next most highly correlated wavelength to the calibration equation. The procedure continued until the addition of further wavelengths explained no more variation in the reference values.

Stepwise or modified partial least squares regression (MPLS; Martens and Jensen, 1982) was performed on the physical characteristics (except radiocarbon age) since more samples were available (see McTiernan *et al.*, in press). MPLS is a combination of multiple linear regression (MLR) and principal components analysis (PCA); PCA reduces the spectra to a few combinations of absorbances that explain most of the spectral data and MLR relates the sample reference values to these principal components (PCs; Shenk and Westerhaus, 1991).

To determine the optimal number of principal components for modified partial least squares calibration a cross validation procedure was performed (McTiernan *et al.*, in press). Three quarters of the data set was used to develop a model subsequently used to predict the variable for the rest of the samples. This process was repeated four times, enabling all samples to be used for model development and prediction. The residuals were pooled to provide a standard error of cross validation (SECV) on independent samples. These calculations were performed on the first and then each subsequent additional PC, with the minimum SECV determining the number of terms to be used. The final model was then recalculated using all the samples, with the residual giving the standard error of calibration (SEC). Values for each conventional analysis were then predicted from the spectra, using each model. The predictive ability of the models was determined by calculating the standard error of prediction (SEP), representing the standard deviation (SD) of the difference between reference and NIRS predicted values.

7.2.3.2 Minimum number of samples

The minimum number of samples required to produce acceptable calibration models was determined by developing models on portions of the available reference samples. These models were then tested by comparing the predicted values for the samples not used in the calibration, against their conventionally measured values. The samples used for calibration had spectra which were representative of the entire population and were chosen objectively using NIRS software ('Select' program; Shenk and Westerhaus, 1991); samples were chosen entirely on spectra and independently from reference values. Details of the algorithm used to select the most representative samples can be found in Shenk and Westerhaus (1991) and McTiernan *et al.* (in press).

7.2.3.3 Outliers

Outliers occur when samples contain spectra which are very different from the population (Y-outliers) or because of a large difference between the predicted value of a sample and its reference value (X-outliers). Outliers are not necessarily errors, although it is important to detect them since they can have a strong influence on calibration models. Spectral (Y) outliers were determined using a standardised Mahalanobis distance (H; Mahalanobis, 1936); observations with a H value > 3 were removed from the calibrations, as recommended by Shenk and Westerhaus (1991). Conventional analysis of these samples is therefore necessary even if calibration models can be developed for the other samples. X-outliers are detectable when calibration models are used to predict reference values; if the difference between predicted and reference values of samples was over 2.5 times the SEP the samples were considered outliers and removed from the calibration.

7.3 Results

7.3.1 Conventional analyses

The conventional analyses results from core MH2 have already been presented in Section 5.3. They are also displayed in this section for comparison with the NIRS values (see below). Further description and interpretation of the conventional results can be found in Section 5.3.

7.3.2 Spectra

Examples of the NIRS spectra are provided in Figure 7.1. The figure illustrates the pattern of absorbance over the entire wavelengths scanned for samples from a range of depths. Generally, absorbance increased rapidly with depth in the top 25 cm of the core and subsequently showed little change until 42-43 cm depth. Absorbance then decreased between 42-43 cm and 46-47 cm depth and increased again between 46-47 cm and 51-52 cm. Samples from below 52 cm depth had very different spectra to samples above; absorbance

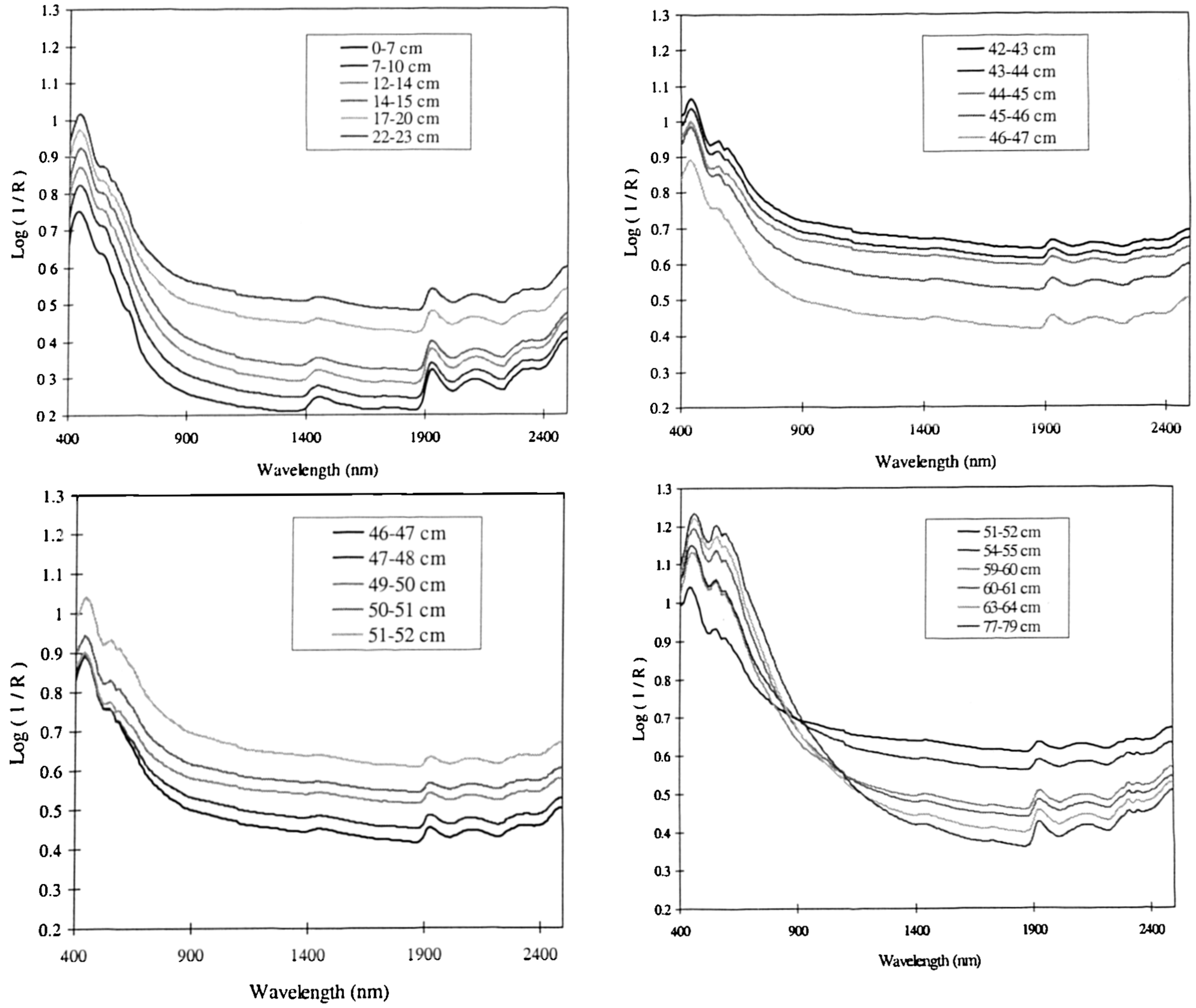


Figure 7.1 Examples of spectra from samples from core MH2

progressively increased with depth between 400 and 700 nm and decreased with depth between 1100 and 2500 nm.

7.3.3 Principal calibrations

The surface sample (0-7 cm) from core MH2 was found to exhibit a spectral signature very different to the other samples ($H > 3$) and, as recommended by Shenk and Westerhaus (1991), was not included in calibrations.

Table 7.2 provides details of the principal calibration models derived for the physical characteristics and plant macro-fossils of core MH2, describing the X-outliers (also removed from calibration), scatter corrections, mathematical treatments and calibration methods used. SEPs and correlation coefficients (r^2) illustrating the strength of the calibration models are also provided for each physical characteristic and macro-fossil in this table. Scatter plots showing the correlations between variables determined using the NIRS models and the original reference values are shown in Figures 7.2 to 7.6. Profiles of NIRS predicted and conventionally determined values are shown in Figures 7.7 to 7.10 and illustrate the variation in estimates with depth; Figures 7.9 and 7.10 present the NIRS predicted and conventional macro-fossil results, respectively, and as conventional macro-fossil plots (Tilia graphs).

Two calibration models were developed for radiocarbon age because the sample from the base of the core, although not believed to be erroneous, had a much older radiocarbon age than the other samples in the core (see Section 5.4). Details of calibration models which include, and omit this sample, are provided in Table 7.2.

Table 7.2 Principal calibrations. The table gives details of the calibration models developed for each conventional variable, including samples not used (outliers), calibration method, scatter correction, math treatment, number of PLS terms, wavelengths used, standard error of calibration (SEC), correlation coefficient (r^2), standard error of cross validation (SECV) and the standard error of prediction (SEP).

Conventional analysis	Outliers	Calibration method	Scatter correction	Math treatment	Number of PLS terms	Wavelengths	SEC	r^2	SECV	SEP
Radiocarbon age (11 samples)	76-77	SW	None	2 10 5	NA	536, 1220	27.8	0.99	62.73	23.71
Radiocarbon age (12 samples)	None	SW	Detrend	1 5 5	NA	1700, 2100	141.67	0.95	278.09	122.69
Delta ^{13}C	None	SW	Detrend	2 10 5	NA	896, 1708	0.44	0.96	0.54	0.38
Bulk density	7-10, 50-51, 51-52, 54-55	MPLS	WMSC	1 5 5	4	NA	0.0041	0.97	0.0043	0.0038
Moisture content	7-10, 50-51, 51-52, 54-55	SW	None	2 5 5	NA	760, 912, 1308, 1500, 1948, 2116	0.41	0.97	0.44	0.39
Degree of humification	None	MPLS	SNVD	2 5 5	5	NA	0.021	0.98	0.027	0.020
UOM	53-54, 63-64, 65-66	SW	Detrend	2 10 5	NA	864, 1780, 2292, 2332	3.94	0.97	4.37	3.63
Ericales roots	7-10, 15-16, 23-24, 59-60	SW	SNV	1 5 5	NA	632, 1196, 2140	2.41	0.85	2.54	2.21
Aggregate monocotyledons	63-64	SW	Detrend	2 5 5	NA	488, 888, 1564, 1572	8.23	0.86	8.77	7.62
Identifiable *Sphagnum*	15-16, 51-52, 53-54	SW	Detrend	2 10 10	NA	800, 1148, 1596	6.37	0.97	6.82	5.97
S. papillosum	None	SW	WMSC	1 10 5	NA	416, 720, 1892, 2396	6.09	0.95	6.73	5.65
S. magellanicum	23-24	SW	None	2 5 5	NA	936, 1180, 2300, 2476	4.74	0.96	4.93	4.39
S. section Cuspidata	53-54	SW	SNVD	2 10 5	NA	688, 904, 1156, 2156	6.48	0.92	7.28	6.00

MPLS - modified partial least squares; SW - Stepwise; MSC - multiplicative scatter correction; WMSC - weighted MSC; SNV - standard normal variance; SNVD - standard normal variance and detrend; NA - not applicable. [1]Math treatment indicates the mathematical transformation of spectral data; the first number is the order of the derivative function, the second is the segment length in data points (1 data point = 2 nm) over which the derivative was taken, and the third number the segment length over which the function was smoothed.

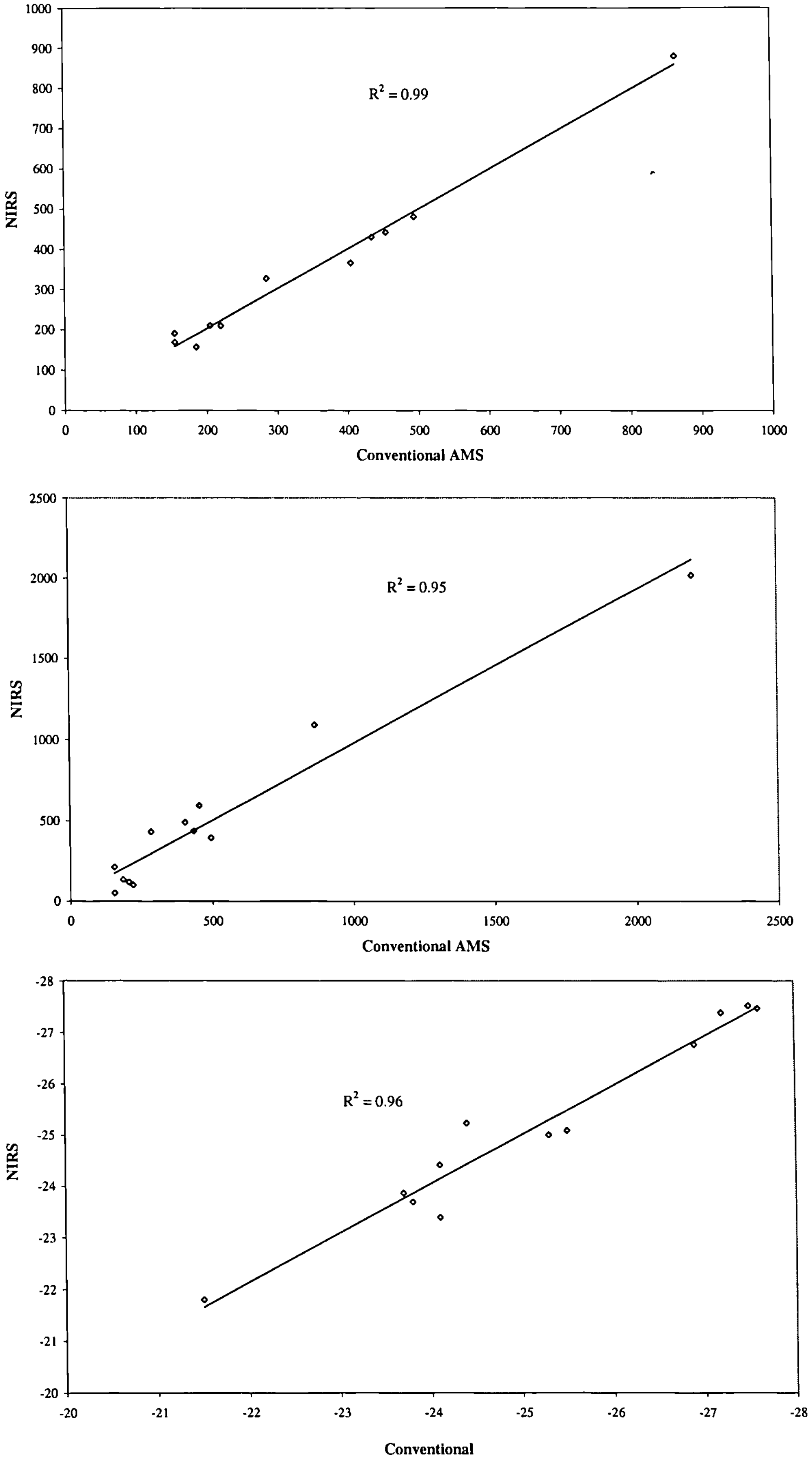


Figure 7.2 Correlations between conventional and NIRS determined values for a) ^{14}C age (years BP; 11 sample calibration model; top), b) ^{14}C age (years BP; 12 sample calibration model; middle) and c) delta ^{13}C (bottom).

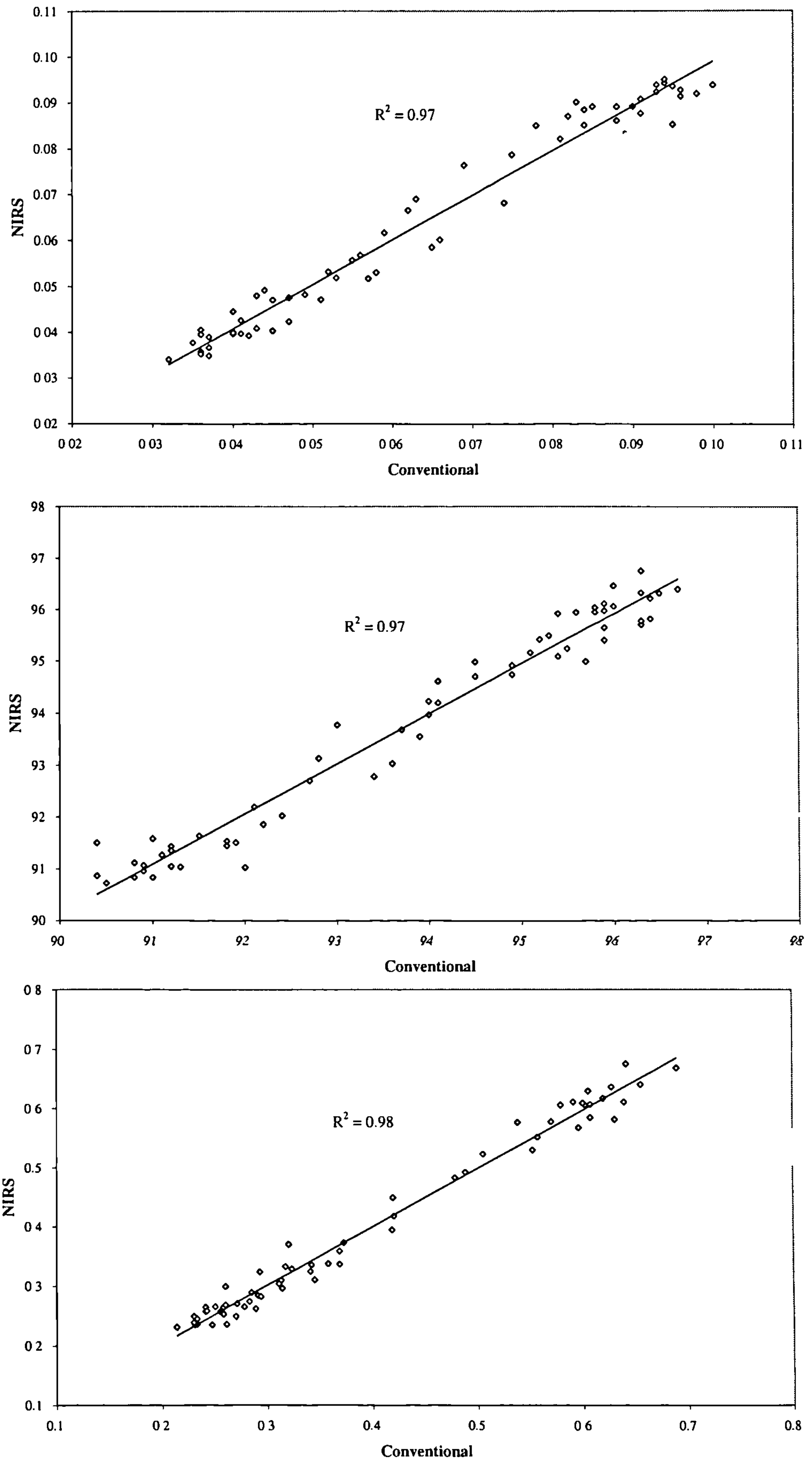


Figure 7.3 Correlations between conventional and NIRS determined values for a) bulk density (g cm^{-3}; top), b) moisture content (%; middle) and c) degree of humification (absorbance; bottom)

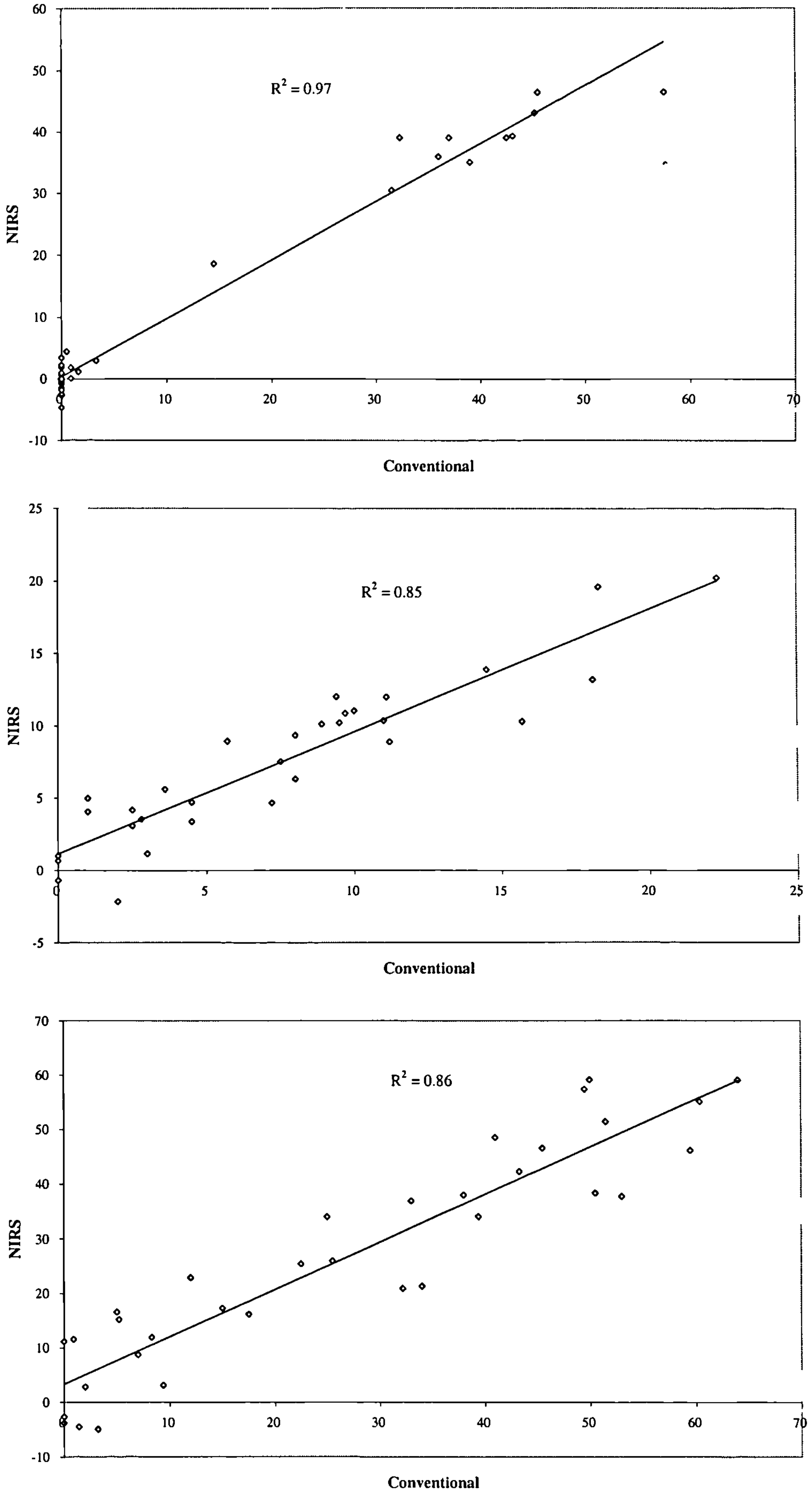


Figure 7.4 Correlations between conventional and NIRS determined values for a) % UOM (top), b) % *Ericales* roots (middle) and c) % aggregate monocotyledons (bottom).

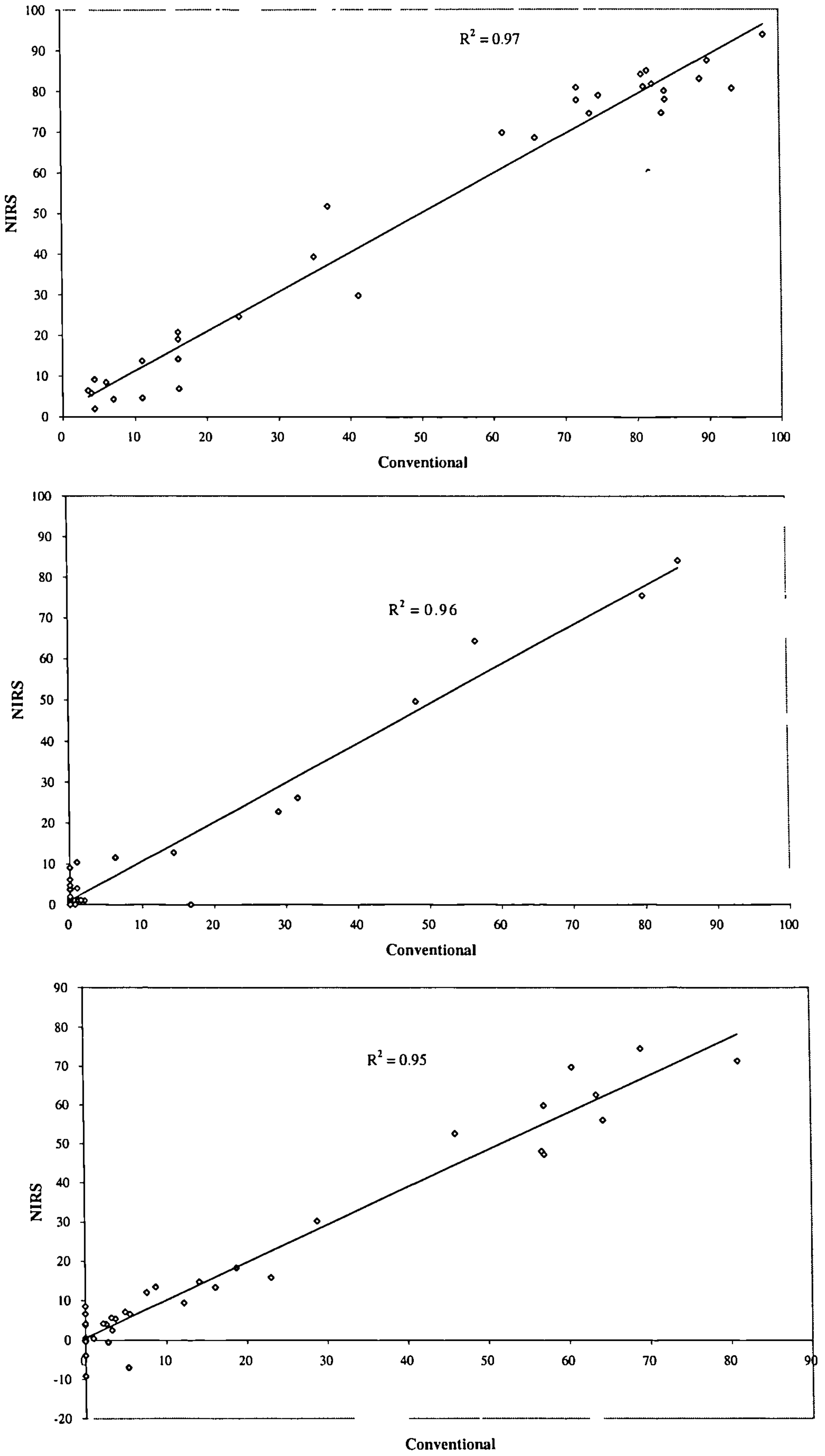


Figure 7.5 Correlations between conventionally and NIRS determined values, showing a) % identifiable *Sphagnum* (top), b) % *S. magellanicum* (middle) and c) % *S. papillosum* (bottom).

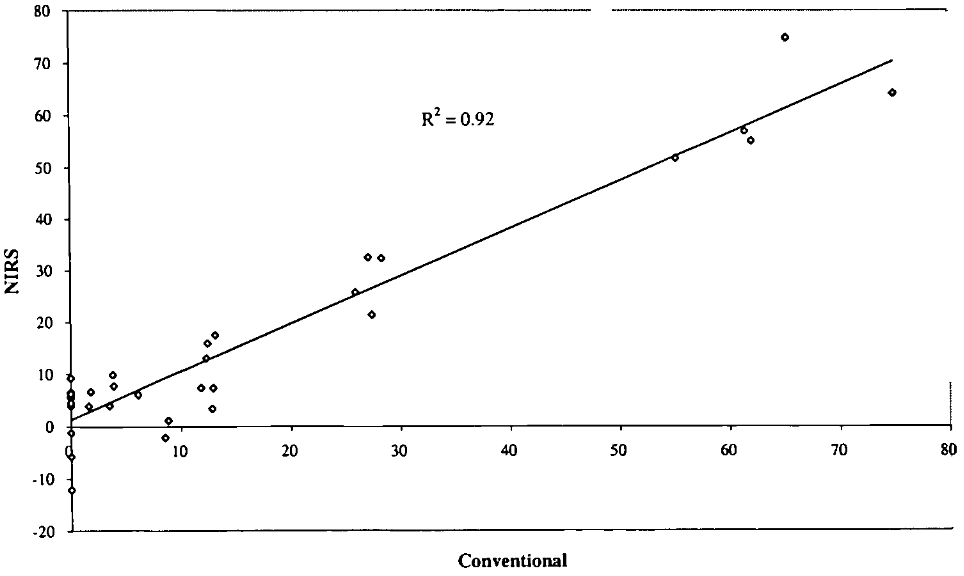


Figure 7.6 Correlation between conventional and NIRS determined values for % *S.* s. Cuspidata

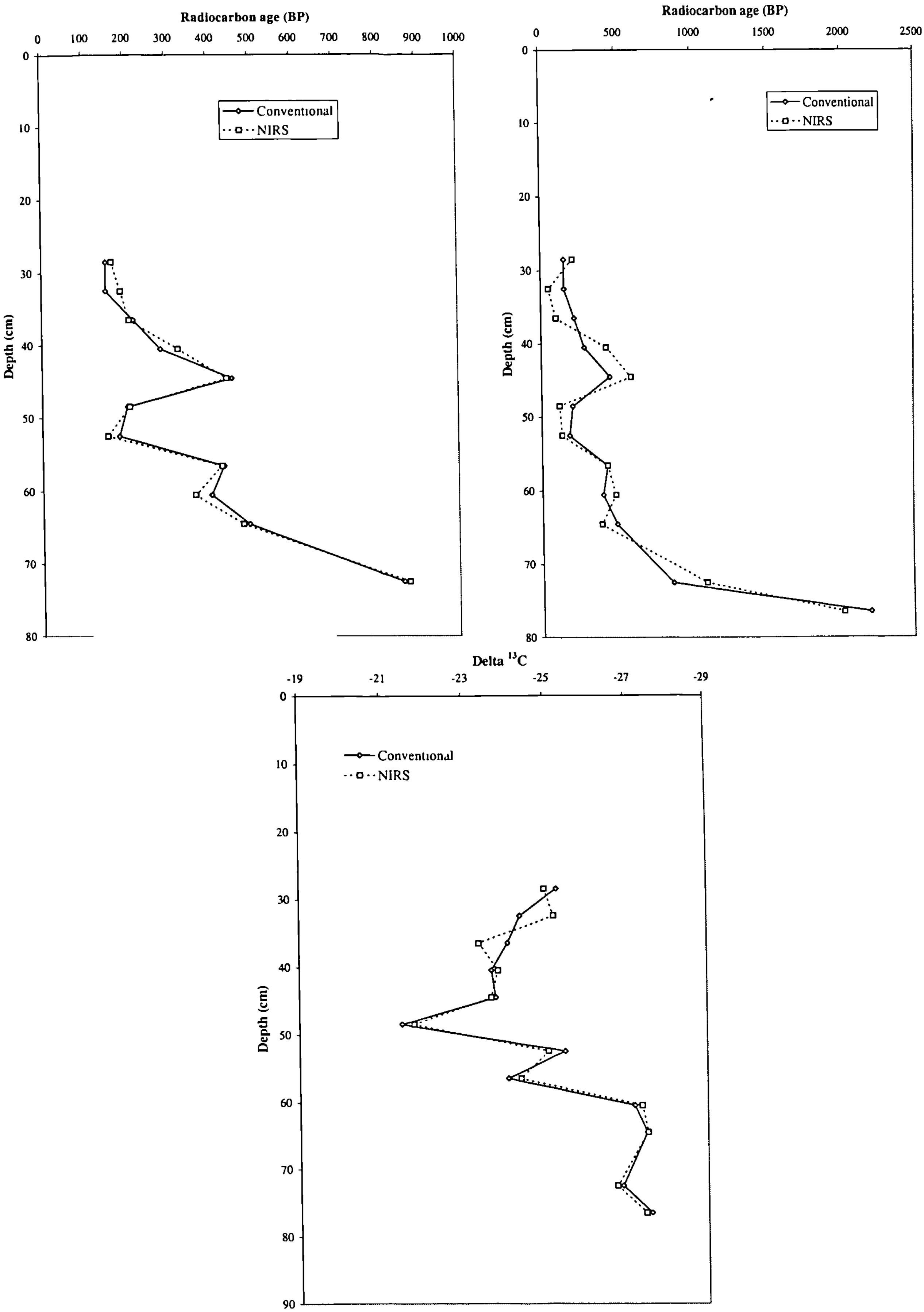


Figure 7.7 Profiles of conventional and NIRS determined values with depth, showing a) ^{14}C age (11 sample model; top left), b) ^{14}C age (12 sample model; top right), and c) delta ^{13}C (bottom).

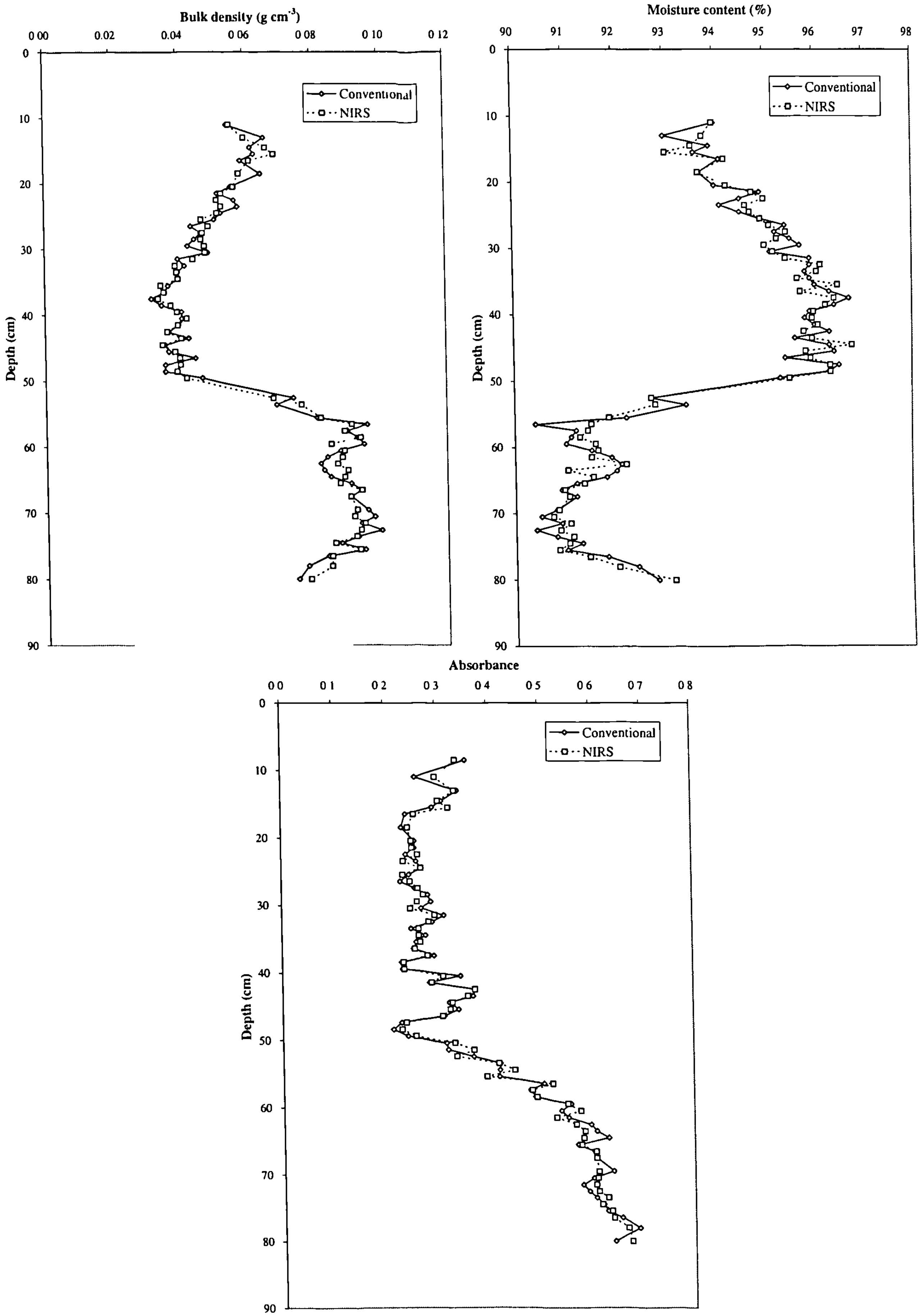


Figure 7.8 Profiles of conventionally and NIRS determined values with depth, showing a) bulk density (g cm^{-3}; top left), b) moisture content (%; top right), and c) degree of humification (absorbance; bottom).

Figure 7.9 NIRS determined macro-fossil results

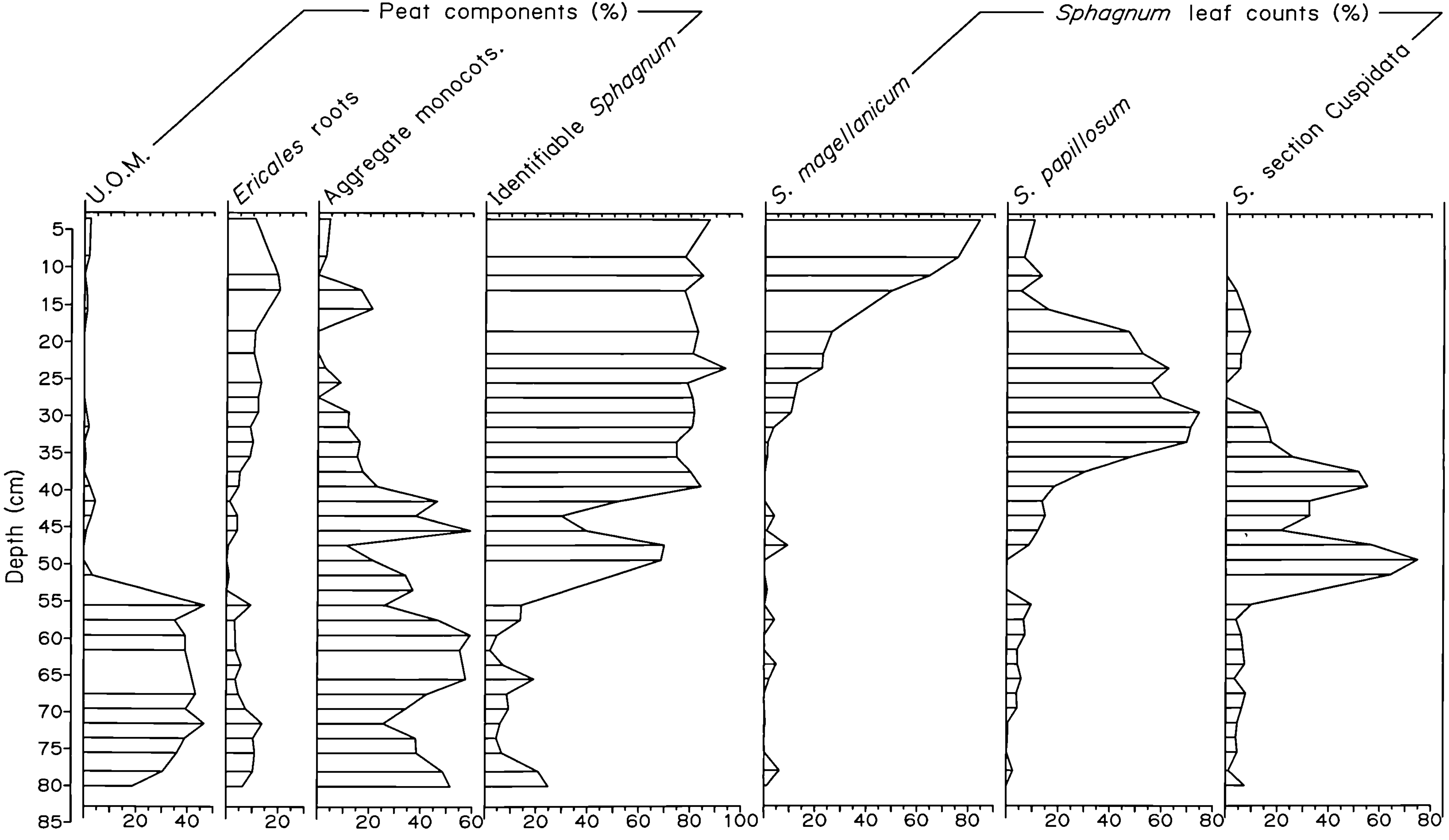
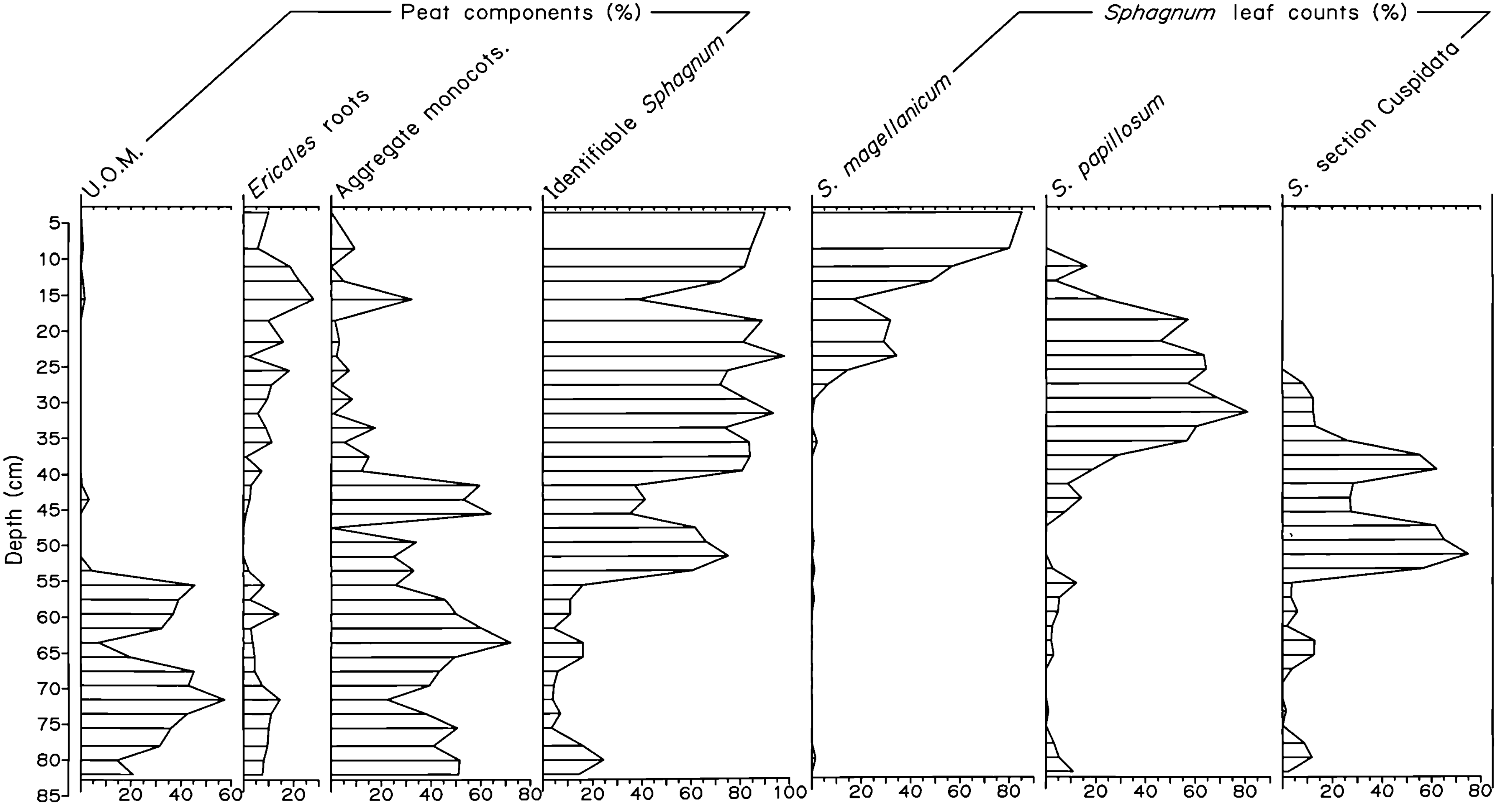


Figure 7.10 Conventional macro-fossil results for core MH2



Analysis: Dmitri Mauquoy

High correlation coefficients and low SEPs were found for most of the variables investigated, with in particular, high r^2 values for radiocarbon age, degree of humification and bulk density. However, calibration models derived for some of the macro-fossils were less successful with the best calibration derived for *Ericales* roots only producing an r^2 of 0.85 between predicted estimates and reference values.

7.3.4 Minimum number of samples

Ten samples were selected (using a minimum H of 0.6 in the 'Select' program), representative of the rest for the macro-fossils, while 14 were chosen for the physical characteristics. Since the samples were chosen purely from their spectra, the same samples were chosen for each physical characteristic and macro-fossil investigated.

Table 7.3 shows details of the calibrations derived using a minimum number of conventionally measured reference samples. Again, degree of humification was very successfully calibrated against NIRS, producing an r^2 of 0.94 when a calibration model, built using just 10 reference samples, was used to predict values for the other samples. Models were produced for both bulk density and moisture content which, with a small proportion of samples, could predict the value of the variable for the other samples accurately: For example, using a third of the samples for the bulk density model, an r^2 of 0.89 was obtained between the NIRS predicted and conventional values for the rest of the samples. Models were determined for radiocarbon age and delta ^{13}C from just 6 samples and resulted in an r^2 of 0.91 for radiocarbon age and 0.93 for delta ^{13}C.

Calibration models using a minimum number of samples were less successful for the macro-fossils with acceptable models for *Ericales* roots, aggregate monocotyledons and UOM not being found. However, using just 13 samples in the calibration resulted in a model which enabled the concentration of *S. magellanicum* in other samples to be predicted with an r^2 of 0.88. A similarly successful model was produced for *S. papillosum* although calibration was less successful for *S.* s. Cuspidata; with 13 samples in the calibration model the comparison between reference and NIRS determined values produced an r^2 of 0.70.

Table 7.3 Minimum number of samples. The number of samples N used to develop each calibration model is provided along with outliers (not used), scatter correction, math treatment, wavelengths used, standard error of calibration (SEC), correlation coefficient (r^2), standard error of cross validation (SECV) and the standard error of prediction (SEP).

Conventional analysis	N	Outliers	Scatter correction[1]	Math treatment[1]	Wavelengths	SEC	r^2	SECV	SEP
Radiocarbon age	6	72-73 76-77	Detrend	2 10 10	2460	9.38	0.91	11.47	43.06
Delta ^{13}C	6	44-45	WMSC	2 10 5	1188	0.36	0.93	0.72	0.56
Bulk density	14	None	WMSC	2 5 5	1332, 2444	0.0058	0.75	0.0068	0.0119
	17	None	Detrend	2 5 5	1064, 1204	0.0061	0.79	0.0068	0.0120
	20	None	WMSC	2 10 10	752, 1780, 2044	0.0061	0.89	0.0070	0.0088
Moisture content	14	None	SNV	2 5 5	568, 1268	0.58	0.42	0.63	1.14
	17	None	None	2 10 10	760, 904	0.64	0.87	0.72	0.87
Degree of humification	10	None	Detrend	2 5 5	584, 1040	0.013	0.94	0.014	0.037
	14	None	Detrend	1 5 10	632, 2108	0.017	0.94	0.019	0.038
Identifiable *Sphagnum*	10	None	SNV	2 10 10	536, 1372	6.01	0.57	7.08	43.46
	13	51-52	SNV	2 10 5	472, 904	5.78	0.83	6.36	15.23
S. papillosum	10	None	None	2 10 5	528, 1900	6.22	0.81	9.78	12.93
	13	None	SNVD	2 10 5	1588, 1940	6.82	0.86	7.34	12.18
S. magellanicum	10	None	None	2 10 5	568, 768	3.14	0.78	4.03	13.93
	13	None	Detrend	2 5 5	696, 1388	2.69	0.88	4.98	7.99
S. section Cuspidata	10	53-54	MSC	1 10 5	416, 2276	4.55	0.50	5.91	16.22
	13	53-54	SNVD	2 5 10	904, 1588	6.97	0.70	8.34	13.54

[1] See Table 7.2 for abbreviations and explanation of math treatment

7.4 Discussion

7.4.1 Principal calibrations

Calibration equations enabling accurate estimates of variables from NIRS spectra have been developed for a range of physical and botanical peat characteristics. The accuracy with which these models were able to estimate the characteristics investigated is indicated by the high correlations found between NIRS predicted and conventionally determined values. However, predictive models were developed using different calibration methods and the accuracy of models differed between the different variables.

The best calibration models were developed for radiocarbon age and degree of peat humification. When the models were used to estimate these variables using only the NIRS spectra, the predicted values were very close to the conventionally measured values producing r^2 values of 0.99 and 0.98 for radiocarbon age and degree of humification, respectively. Accurate predictions were also made for bulk density and moisture content from the NIRS spectra, and although Downey and Byrne (1986) have previously demonstrated that NIRS could accurately determine moisture content and bulk density of air-dried milled peat, their analyses were very different to the ones undertaken on fresh peat reported here.

Calibrations models were less successful for the macro-fossils than for most of the physical characteristics investigated. In particular, *Ericales* roots produced the least successful model but the percentage of *Sphagnum* species were accurately determined using the NIRS approach. Comparison of the profiles of conventionally and NIRS determined values with depth showed strong similarities. Radiocarbon age showed remarkably strong agreement even though there was apparently younger peat lying beneath older material at *ca.* 50 cm depth. The predicted profile of delta ^{13}C also followed the measured pattern of delta ^{13}C obtained from the conventionally measured samples.

The other profiles of values predicted using NIRS versus conventionally determined values show very similar results, yet a notable feature, particularly of degree of humification and macro-fossils, was a much smoother variation in values with depth. This is unlikely to reduce the usefulness of the NIRS estimates for macro-fossils since only the more distinctive trends in these values are used in palaeoclimate reconstructions. It is difficult to state whether small scale variations in these variables represented significant trends or variations in the accuracy of conventional measurements.

The accuracy of NIRS calibrations depends partly on the accuracy of the conventional method of analysis used to provide reference values. Therefore, it is likely that the variables which can be most accurately determined using conventional methods will produce the best correlations with NIRS spectra. Moisture content and degree of humification yielded more accurate calibration models than for the macro-fossils. Moisture content and humification are accurately determined using objective physical and chemical measurements. In contrast, macro-fossils are determined semi-quantitatively (Stoneman, 1993) and, therefore, calibration may have been less successful due to error in conventional macro-fossil analyses.

Outliers were frequently located around the depths of 50-60 cm. This was a zone of transition in the peat profile with a change from very humified to overlying, relatively fresh *Sphagnum* peat. Furthermore, the zone of transition between these peat types was slightly non-horizontal. It is possible that some of the variables measured on separate sub-samples were determined on peat samples different to the material used for the NIRS scans. Outliers for several of the macro-fossils were detected at this depth and may have been caused by different peat samples being used for the NIRS and macro-fossil analyses. Degree of humification was the only variable measured on the same sub-sample used for the NIRS scans and produced one of the best calibration models.

Several of the macro-fossil samples were predicted to have negative values for some variables. The reason may be that different samples were used for the NIRS scans and macro-fossil analysis and that small samples used for macro-fossil quantification may have missed fossils of a species present in the larger sample used to obtain NIRS spectra. In such a case, the calibration model could predict negative values for samples used for NIRS which did not contain any fossils of the species.

There is a likelihood that NIRS is determining values for variables indirectly. For example, if two conventionally measured variables correlate with each other, NIRS would only need to be able to calibrate with one of the variables; the other variable would be predicted because of the correlation between conventional variables. Correlation coefficients have therefore been calculated between each conventional variable measured (Table 7.4). The results indicate that in only one case is there a strong correlation between two conventionally measured variables; bulk density and moisture content. Therefore, all the other variables seem to be predicted directly from the NIRS spectra. However, the correlation coefficient of 0.99 between moisture content and bulk density is higher than the correlation between NIRS predicted and conventional values. This suggests that bulk density could be more accurately determined by taking moisture content measurements to predict bulk density, rather than using NIRS; moisture content is a very simple measurement to take requiring simple weighing and drying of samples.

Perhaps the potentially most useful and surprising calibration model developed was for radiocarbon age. Two models were developed since the oldest radiocarbon date was much older

Table 7.4 Correlation coefficients (r^2) between conventionally determined variables

	Radio-carbon age	Delta ^{13}C	Degree of humification	Bulk density	Moisture content	UOM	*Ericales* roots	Aggregate monocots	Identifiable *Sphagnum*	*S. magellanicum*	*S. papillosum*	*S.* s. Cuspidata
Radiocarbon age	1											
Delta 13C	0.32	1										
Degree of humification	0.48	0.74	1									
Bulk density	0.23	0.57	0.66	1								
Moisture content	0.23	0.54	0.62	0.99	1							
UOM	-	-	0.62	0.63	0.58	1						
Ericales roots	-	-	0.01	0.01	0.02	0.00	1					
Aggregate monocots	-	-	0.49	0.19	0.14	0.21	0.10	1				
Identifiable *Sphagnum*	-	-	0.76	0.50	0.43	0.71	0.00	0.66	1			
S. magellanicum	-	-	0.16	0.00	0.01	0.12	0.11	0.26	0.25	1		
S. papillosum	-	-	0.28	0.33	0.33	0.22	0.04	0.34	0.39	0.00	1	
S. s. Cuspidata	-	-	0.11	0.22	0.25	0.15	0.31	0.00	0.08	0.00	0.02	1

than other dates and therefore strongly influenced the calibration models; this date may have been erroneous and caused by contamination with older material. However, calibration with and without this date produced very good models for predicting radiocarbon age from the NIRS spectra alone and Figure 7.11 shows the predicted radiocarbon age for the rest of the samples in core MH2, based on NIRS spectra, and the two calibration models developed with 11 and 12 reference samples. Since there are no further AMS dates from this profile it is not possible to independently verify the predicted results, although, the estimated dates follow the expected profile of radiocarbon age, given the assumed chronology based on the AMS dates.

Radiocarbon ages are derived from the measurement of ^{14}C isotopes within organic samples. There may be characteristic molecular bonds between ^{14}C isotopes and other atoms which could infer the concentration of ^{14}C isotopes in samples, but it is unlikely that the NIRS could detect this since the concentration of ^{14}C in organic samples is extremely low. However, there may be chemical or physical signals in the peat samples which can be determined by NIRS and increase with age, thereby allowing NIRS to be calibrated with age. For example, chemical changes in organic samples occur with time through decomposition processes, with for example, a decrease in the C:N ratio in peats with depth (*i.e.* age; Malmer and Holm, 1984; Kuhry and Vitt, 1996). There may be presently unknown chemical changes in organic samples which are detectable using NIRS and are strongly related to time.

The profile of delta ^{13}C with depth has also been predicted by estimating the variable for all samples using the calibration model and the NIRS spectra of each sample. The results also follow the trend anticipated from the 12 conventional results.

Despite the above suggestions, there is currently no physical basis to explain why the NIRS correlates with radiocarbon age and delta ^{13}C and, consequently, the results of the present study must be treated with caution. Further investigation is required before it can be stated that NIRS can be used to predict these variables in peat samples.

7.4.2 Minimum number of samples

The minimum number of samples required to give acceptable calibration models depends largely on the accuracy of the results required. For example, trends in the degree of humification of peat profiles have been used to provide proxy records of climate change (Blackford and Chambers, 1993) while accurate measurements of bulk density are necessary for modelling peat and C accumulation in mires (Clymo, 1984).

An acceptable calibration model for degree of humification was constructed using only ten reference samples, producing an r^2 for the remaining samples of 0.94 when NIRS and conventionally determined values were compared. Less than a third of the samples were required

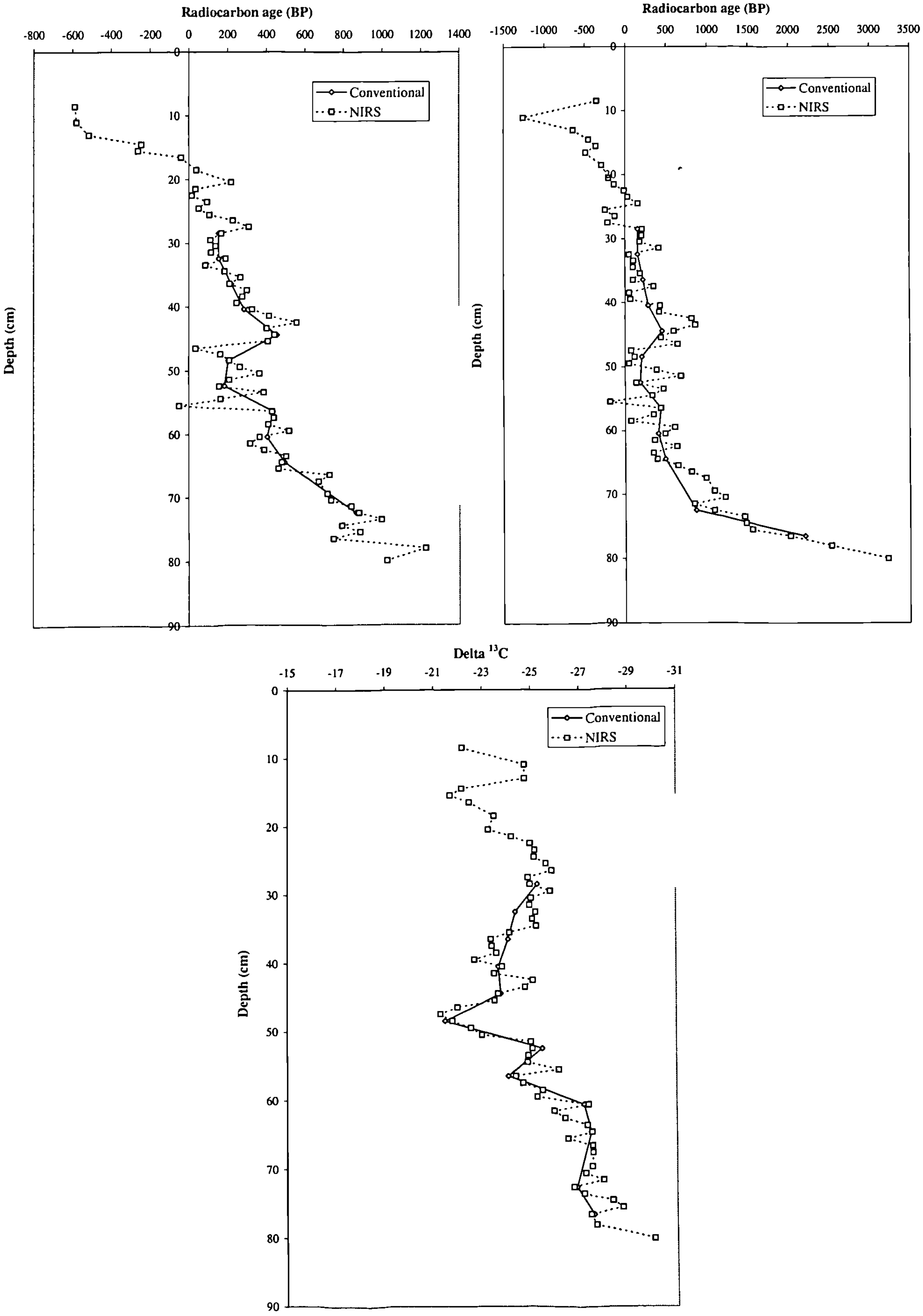


Figure 7.11 Comparison of NIRS predicted values for all depths, against conventionally measured values, for a) ^{14}C age (11 samples calibration model; top left), b) ^{14}C age (12 samples calibration model; top right) and c) delta ^{13}C (bottom).

for bulk density and moisture content to produce models which could accurately determine these variables for the remaining samples using only NIRS spectra. For the *Sphagnum* leaf counts slightly more than a third of samples were required in the calibration model, although it was not possible to produce acceptable models using this approach for UOM, *Ericales* roots and aggregate monocotyledons.

The main reasons why calibration models could not be developed for three of the macro-fossils were high variability and a limited number of samples. When enough samples had been included to produce an acceptable model, not enough samples remained for validation (McTiernan *et al.*, in press). These variables are particularly difficult to quantify using conventional methods and were more likely to have been represented differently in the separate samples used for conventional analysis and NIRS spectra. *Ericales* roots were also very difficult to grind with the ball mill, and therefore, an improved method of grinding may have provided a more representative sample for NIRS scanning.

7.4.3 Potential application of NIRS in palaeoecological studies of peat

The results of the present investigation suggest that NIRS is a potentially useful technique for palaeoecological studies of peat. The time-savings offered by NIRS are perhaps greatest in areas where many samples are analysed for a range of variables. Palaeoecological investigations require many replicate samples to be analysed in order to produce high resolution reconstructions. In most cases it is likely that the resolution and replication of studies would be higher if the analyses could be undertaken more rapidly yet with little reduction in data quality. Therefore, NIRS clearly offers advantages for palaeoecological studies of peat.

The actual time-savings made possible with NIRS depends on the accuracy of the calibration required and how easily the reference values can be calibrated with NIRS spectra. Preparation of samples for scanning is minimal (drying and grinding) and likely to be undertaken if degree of humification is being investigated. Scanning of samples with NIRS is also rapid and can be automated; once spectra have been obtained for each sample the remaining major task is model development. The time required to develop calibration models can vary considerably (McTiernan *et al.*, in press). The most successful models are generally rapidly developed, but where reference values are not easily calibrated with the spectra, model development can take longer since different models should be tried and outliers removed (*e.g.* the degree of humification model was developed in a few minutes, but the *Ericales* roots took 20 to 30 minutes; McTiernan *et al.*, in press). However, the amount of time for model development is small compared to the time necessary for conventional analyses; a trained macro-fossil analyst can only investigate *ca.* 8 samples a day (Mauquoy, personal communication).

In the present work, NIRS has been calibrated against the radiocarbon age of peat samples, suggesting that there is information in the spectra relating to the age of the sample. This is a potentially useful finding since a major constraint in palaeoecological studies is the need for a strong chronology (*e.g.* Chambers *et al.*, 1997; Mauquoy, 1997). Radiocarbon dates are expensive and take several months to process while NIRS may allow the age of a sample to be predicted within minutes.

The full extent to which NIRS could benefit palaeoecological investigations of peat will depend on the extent to which the need for conventional analyses is reduced. The results reported in this study are from a single peat core, from a single location and containing only a limited range of peat types. If the calibrations developed from these samples can be used to accurately determine the variables in other cores and from other locations, then the NIRS approach offers considerable advantages. However, even if new calibrations are required for each core investigated, NIRS is still likely to offer a more rapid analysis of samples with analyses of samples only being required for the development of the necessary calibration models.

8. Discussion: Terrestrial carbon storage at Moor House and response to change

The close coupling between terrestrial ecosystems and the atmosphere has prompted concern that a depletion in terrestrial C stores, in response to global warming, may cause a consequent increase in the rate of rising atmospheric CO_2 concentration (Melillo *et al.*, 1996). As a result, the size of terrestrial C stores and response to climate change have been the subject of intense research over recent years (see Melillo *et al.*, 1996). However, there are still considerable differences in estimates of terrestrial C storage and uncertainties about whether, under a warmer climate, terrestrial ecosystems will be a source or sink for CO_2 (Schimel *et al.*, 1994).

This thesis represents an investigation into the size of terrestrial C stores and response to change for an area of typical British moorland. Terrestrial C stores have been quantified in detail in order to test the accuracy of the UK's national C inventory and predictions for the response of the C store to future climate change have been made by investigating its response to past climate change, and through measurement and modelling of contemporary C fluxes at the site.

Present terrestrial C stores at Moor House and comparison with the UK national C inventory

The total amount of terrestrial C contained within the 22 one-kilometre squares at Moor House has been estimated in the present study to be 1,179 x 10^6 kg, which is similar to the annual C emission (as CO_2) of a coal fired power station (*ca.* 1,700 x 10^6 kg; Immirzi *et al.*, 1992). Only 0.5 % of the Moor House C store was contained within living vegetation, while greater than 94 % was in blanket peat at the site, a consequence of the dominant area of peat coupled with high C density.

The values of vegetation C storage for Moor House (*ca.* 0.1-0.5 kg m^{-2}) were typical for grassland and moorland, but were lower than C densities of British woodlands (broad-leaved *ca.* 6 kg m^{-2}; coniferous *ca.* 2.5 kg m^{-2}; Milne and Brown, 1997). The estimates for C storage in mineral soils are similar to values reported elsewhere (*e.g.* Batjes, 1996) and contrast with the high values for C storage in peatlands emphasised by the present study. However, the values of C per square metre calculated for Moor House (average 64 kg m^{-2}) are considerably lower than some published values for peatlands (*e.g.* 133 kg m^{-2}, Gorham, 1991; 264 kg m^{-2}, Batjes, 1996).

Since the UK's national C database is based on detailed maps of soil and vegetation, with a large database of C densities (Milne and Brown, 1997), it should be a relatively accurate inventory of terrestrial C when compared to previous, less detailed, estimates of terrestrial C stores (*e.g.* Gorham, 1991). It is therefore of considerable concern that the national database soil C estimate for Moor House was three times higher than the value produced in the present study. The C inventory derived for Moor House in this case study utilised maps of even greater detail than those used in the national database, and involved targeted soil C density measurements for the actual soils at Moor House. Consequently, the inventory of soil C derived in the present study is considered to be more accurate than the equivalent figures in the national database. Therefore, given that the UK's national database is a relatively accurate inventory of terrestrial C stores, and apparently over-estimates the C storage at Moor House by a factor of three, how reliable are the existing global estimates of terrestrial C stores?

Clymo (1996) considers that global-scale estimates of terrestrial C storage, such as Gorham's (1991) widely quoted value for northern peatlands, should be treated only as indicative, although there is no doubt of the importance of peatlands for C storage, even with the uncertainties in estimates of peatland area, bulk density and peat depth. However, accurate estimates of terrestrial C storage are required in order to provide quantitative estimates of potential terrestrial CO_2 sources. The so called 'missing C sink' may at least partially be attributed to inaccurate estimates of C storage (Jain *et al.*, 1997). The results presented in this thesis suggest that there is considerable scope for improving inventories of terrestrial C stores, and that current estimates of global stores should be treated only as indications of the true size of the global terrestrial C store.

Other implications for producing terrestrial C inventories

Several national inventories have already been published in accordance with the objectives of the Climate Change Convention (*e.g.* UK; Milne and Brown, 1997; New Zealand; Tate *et al.*, 1997). The results of the verification exercise performed at Moor House have further implications for these C inventories.

The Moor House results suggest that consideration of true surface areas is unlikely to greatly improve estimates of terrestrial C storage since true surface areas were only slightly larger than the 2d areas normally used for C inventories. This was particularly the case for soil C, since soils with the highest C densities were generally located on the flattest slopes, a relationship that has been observed by other researchers (*e.g.* Arrouays *et al.*, 1998). At Moor House the low C storage of soils on steep slopes is probably a result of the better drainage, allowing decomposition rates to be high, despite the high rainfall. However, the lack of a strong relationship between slope and soil C density within some soil types suggests that other factors also strongly influence C storage.

Moor House is an area of upland grassland and moorland with low densities of vegetation C storage. However, if woodland had occupied the steeper slopes, considerable underestimation of vegetation C stores may have occurred if only 2d areas were used in the C inventory (*e.g.* C densities: Calluneto-Eriophoretum = 0.35 kg m^{-2}, Chapter 2; broad-leaved woodland = *ca.* 6 kg m^{-2}, Milne and Brown, 1997). Consequently, it is suggested from the Moor House results that consideration of true surface areas will only significantly improve C storage estimates when soil C is relatively low and vegetation of high C density (*e.g.* woodland) occupies steep slopes.

The lack of reliable field data for soil C inventories is frequently quoted as a major source of uncertainty (*e.g.* Howard *et al.*, 1995; Milne and Brown, 1997; Arrouays *et al.*, 1998). However, several excellent correlations were observed between variables in samples from Moor House and use of these correlations would allow further samples from the site to be analysed more rapidly. Furthermore, since the correlations have a physical basis (see below), the relationships should be suitable for use at other sites.

The correlation observed between moisture content (%) and bulk density has already briefly been discussed (Section 4.4.2). Figure 4.17 shows the correlation observed in the samples collected for the quantification of soil C at Moor House (Chapter 3) while Figure 8.1 shows the correlation between the variables observed in peat samples from core MH2 (Chapter 5). Both graphs show an inverse relationship between the two variables and the difference in shape of the correlation simply reflects the lower range of values in samples from the MH2 core. A simple model has been developed to explain this correlation and the shape of the relationship:

> Peat can be considered to be composed of organic matter and water. Water has a density of 1 g cm^{-3} while organic matter has been estimated, in the present study, to have a dry density of *ca.* 2 g cm^{-3}. For peat samples of known weights and moisture contents, the bulk density of samples can be estimated by calculating the volume of the water and dry peat components of the samples. For example, a sample with a moisture content of 90 % and weighing 100 g contains 90 g of water and 10 g of dry peat. The volume of water is, therefore, 90 cm^3 and the volume of dry peat is *ca.* 5 cm^3 (using volume = mass / bulk density; rearranged equation 3.8 and assuming peat dry density of 2 g cm^{-3}). Therefore the total volume of the wet sample is 90 + 5 = 95 cm^3. Using equation 3.8, the bulk density of the whole sample when wet is calculated as 10 / 95 = 0.11 g cm^{-3}.

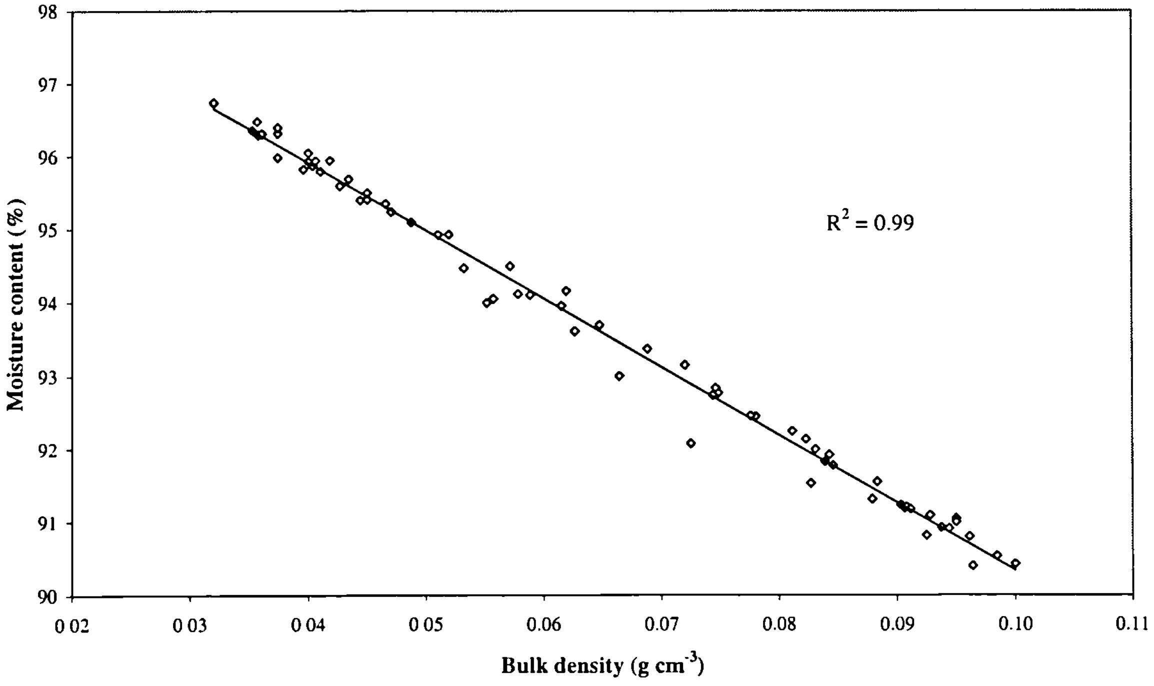


Figure 8.1 Scatter plot showing correlation between bulk density and moisture content for samples from core MH2

Bulk density has been calculated for a range of moisture contents, assuming a dry density for peat of 2 g cm^{-3} and plotted on the same graph as the conventionally measured sample results (Figure 8.2); a strong agreement exists between the sample results and the calculated relationships between moisture content and bulk density.

Similarly, a correlation was observed between bulk density and %LOI in the peat samples from Moor House. This relationship has been observed by other researchers (*e.g.* Jeffrey, 1970) and Harrison and Bocock (1981) used regressions between the two variables to estimate bulk density from %LOI measurements. The model used to explain the correlation between bulk density and moisture content (see above) can also be modified to explain the correlation of bulk density and %LOI:

> Dried peat can be considered to be composed of a mineral and organic fraction, with the organic fraction equal to the %LOI. Assuming the organic fraction, and therefore, the %LOI of a sample, is 80 % of the total by dry weight, then the mineral fraction contains 20 % of the dry weight. For a peat sample of 100 g dry weight, 20 g would be mineral and 80 g would be organic matter. In peatlands, bulk density of organic matter (*e.g.* peat samples with very low %LOI values) is about 0.1 g cm^{-3} (Clymo, 1983; Milne and Brown, 1997), whereas mineral material, which unlike organic matter does not absorb large amounts of water, has a density *ca.* 2-3 g cm^{-3} (Moseley, 1978). Therefore, the volume of the two components can be calculated using volume = mass / bulk density (rearranging equation 3.8); mineral volume is 20 / 2 = 10 cm^3; organic matter volume is 80 / 0.1 = 800 cm^3. Using equation 3.8, the bulk density of the entire sample can be calculated *i.e.* 100 / (10 + 800) = 0.12 g cm^{-3}.

Bulk density has been calculated using the above equations for a range of %LOI values and compares very well with the measured values of %LOI and bulk density from the samples collected for Chapter 3 (Figure 8.3).

A further useful correlation was found between total C storage per square metre of peat and total peat depth for blanket bog samples (Figure 3.18). This correlation may be of potential benefit, particularly if additional measurements for C density at Moor House were required; obtaining the depth of peat is far more rapid than coring for samples, transporting them back to the laboratory and analysing them. A reduction in sampling and analysis time would allow more sampling points to be investigated and provide a more representative estimate of the mean C density for the site. Furthermore, recently developed techniques such as ground-penetrating radar have been shown to be useful for mapping peat depths (Lapen *et al.*, 1996), offering the potential of greatly increasing the detail of C inventories.

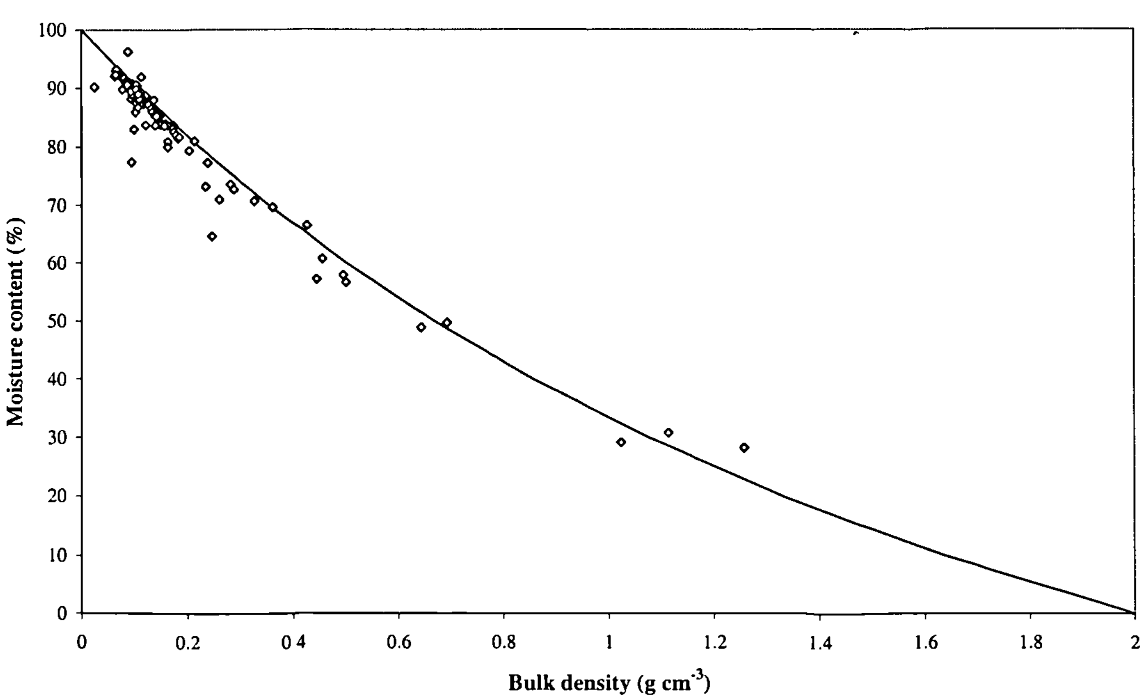


Figure 8.2 Plot of the correlation between moisture content and bulk density obtained from the blanket bog samples (diamonds) and the modelled relationship between the variables (line)

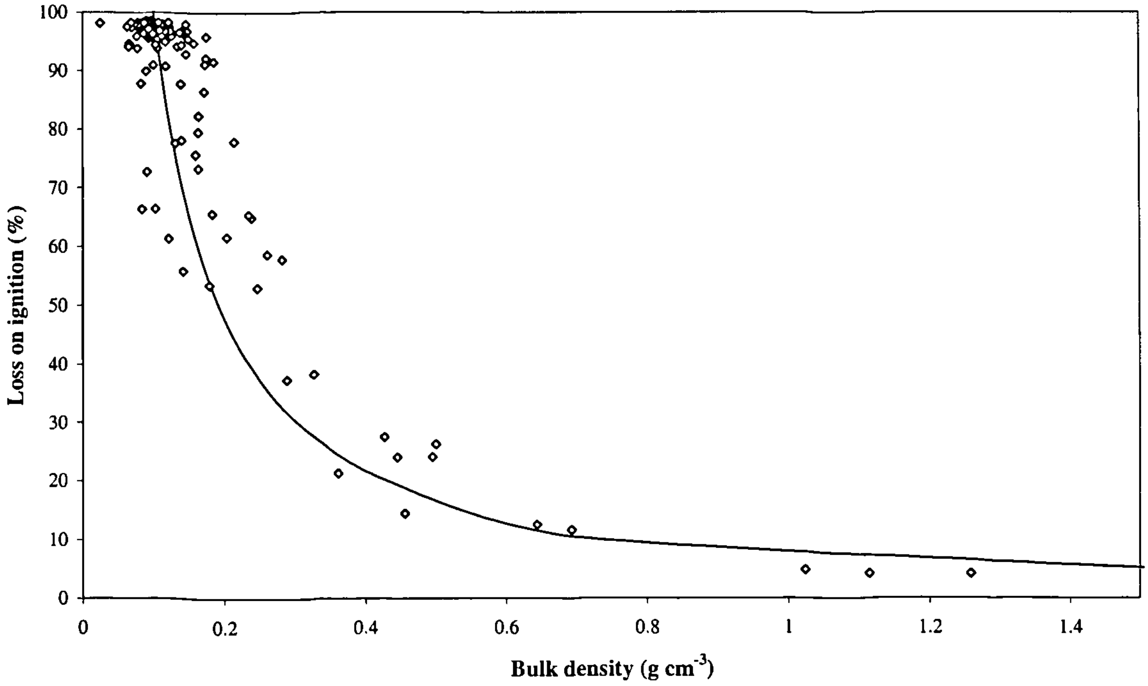


Figure 8.3 Plot of the correlation between loss on ignition and bulk density obtained from the blanket bog samples (diamonds) and the modelled relationship between the variables (line)

Past terrestrial C storage at Moor House

The terrestrial C store at Moor House has varied greatly since the late Pleistocene and provides a useful example of the rapidity and magnitude with which terrestrial C stores can change. The C store at Moor House would have been virtually nil during the last glacial maximum (*ca.* 18,000 BP) since glacial erosion processes would have removed C accumulated in previous inter-glacials and the severe climate at that time is believed not to have supported much, if any, vegetation (Pennington, 1997).

However, evidence from pollen analysis of peat deposits from, for example, Valley bog, Moor House (Chambers, 1978) reveals the history of vegetation colonisation of the area and hence, to some degree the consequent accumulation of C at the site. The pollen evidence suggests that during the first part of the Holocene, the vegetation was predominantly deciduous woodland. Turner and Hodgson (1979) suspect that the composition of these early woodlands was mainly birch, elm, hazel and ash, although whether the highest parts of the study area such as the summit of Great Dun Fell were ever wooded, is subject to debate (Turner, 1987). However, an estimate of the terrestrial C storage of the Moor House study site can be made for the period prior to blanket bog initiation, by assuming that the woodland vegetation had a C density similar to contemporary broad-leaved woodlands, and assuming that the soil had a C density typical of present soils associated with these woodlands. Consequently, with a mean vegetation C density of 6 kg m^{-2} (Milne and Brown, 1997) and a soil C density of 23 kg m^{-2} (for a brown forest soil; Milne and Brown, 1997), the terrestrial C storage of the site at *ca.* 8000 BP would have been *ca.* 640 x 10^6 kg, or about half the size of the present store.

With the onset of blanket bog formation at Moor House there will have been large changes in terrestrial C storage. Several hypotheses for the initiation of blanket bogs have been suggested (Moore, 1975) including climate change, soil development and the activities of man. It is unknown whether blanket mire initiation was synchronous across sites like Moor House, although it is possible that peat started forming on hill summits and slopes at different times. However, the net result was an increase in terrestrial C storage, although since tree remains are present within the basal layers of peat at Moor House and other blanket bog sites (Tallis, 1975) the suggestion is that vegetation C stores must have declined quite rapidly. The woodland would have been replaced by a vegetation composition similar to the present day in terms of C storage. Soil C storage would have increased and, assuming a C accumulation rate of *ca.* 27 g m^{-2} yr^{-1} (mean for core MH2), would have compensated for the reduced vegetation C storage within a couple of centuries.

From the results of the present investigation and others (*e.g.* Oldfield *et al.*, 1997) the rate of C accumulation in the blanket bog at Moor House is unlikely to have been constant over its period of formation. The rates of C accumulation for core MH2 cover only the last 1000 years, and apparently show that the rate of C accumulation was strongly influenced by climate. We know that climate has shown large temporal variability, from a variety of published sources, over the last *ca.* 7000 years (Roberts, 1989) and, therefore during the period of blanket peat formation at Moor House. Consequently there must have been changes in the past rate of C accumulation in the peats at Moor House, with highest rates of increasing C storage when the climate favoured a wet mire surface (see below and Chapter 5).

Further changes in the quantity of C stored at the Moor House site will have been caused by anthropogenic deforestation of well drained areas, where woodland would have persisted (Turner and Hodgson, 1979). The expansion of sheep grazing on the site would have affected vegetation C storage, based on the results of studies at Moor House (Chapter 4, Smith and Forrest, 1978), although grazing did not appear to have a major effect on soil C storage from the present study (Chapter 4).

The impact of soil erosion has not really been considered in the present study, although is likely to have significantly reduced the terrestrial C storage of the Moor House study area. Estimates for total C storage of the study site assuming that no peat erosion has taken place can be obtained by substituting the C density of 'Eroding bog' with the uneroded 'Calluneto-Eriophoretum' blanket bog C density value. The results suggest that, if peat erosion had not occurred at Moor House, the total C storage of the site would have been 14 % higher (1,241 x 10^6 kg).

Present fluxes of C at Moor House

Figure 1.7 introduced the major fluxes of C at the Moor House site. Estimates for the size of these fluxes can be made from both the results of the present work and published results from other researchers working at the Moor House and other similar sites.

The amount of C annually fixed by vegetation at Moor House can be estimated by multiplying the area of vegetation classes by measurements of NPP. From the fixed plots study, the mean NPP was estimated at between 117 and 341 g m^{-2} yr^{-1}. Extrapolation of these results suggests that between 2.5 x 10^6 kg and 7.5 x 10^6 kg of C is fixed annually by vegetation growing within the study site; annual NPP is approximately equal to the C stored in the vegetation at Moor House (5.4 x 10^6 kg). Since the vegetation C storage was calculated from measurements taken when the vegetation classes contained highest storage (*i.e.* late summer), a total NPP equal to the total vegetation C implies that the vegetation C storage at the start of the growing season was nil. However, since sheep grazing is common on the study site, much of the annual NPP is removed by sheep during the growing season. In the absence of grazing, the value of vegetation C storage would be higher and suggests that considerable quantities of vegetation C remain over winter, which is the case at Moor House (*e.g.* T_0 samples in the NPP studies; Chapter 6).

The input of C as SCPs is still occurring at Moor House as shown by the surface samples of the peat cores investigated in Chapters 4 and 5. Furthermore, contemporary fluxes of SCPs have been estimated by filtering rainfall collected in rainfall gauges at various sites within the Reserve. These results are not presented in the current work, but they confirm that SCPs contribute only extremely small amounts of C to the total store due to the small size and density of SCPs; the total annual influx of C has been estimated as *ca.* 0.03 g m^{-2}. However, deposition rates of SCP vary due to factors such as proximity to SCP sources (Wik and Renberg, 1996) and altitude (Rose and Juggins, 1994), and therefore, it is possible that they may provide a larger input of C to other sites. Furthermore, there may be seasonal variations in the deposition of SCPs caused by differences in production rates; weekly monitoring of SCP deposition at Moor House is currently on-going.

The loss of C from Moor House due to decomposition is more difficult to estimate since different vegetation species have been found to decompose at widely different rates (Chapter 6). Furthermore, the total C loss from the site due to decomposition will include C mineralisation form soil organic matter as well as recently produced litter. However, for the following discussion the non-blanket bog soils have been assumed to be C stable since they represent only a very small proportion of the total C store.

NPP must have exceeded decomposition in the blanket bog over the last few millennia since peat has accumulated. An average rate of peat accumulation was calculated in Chapter 5 for core MH2 of 27 g m^{-2} yr^{-1}, which is very similar to mean estimates derived by other researchers for other sites (Gorham, 1991; Tolonen and Turunen, 1996). The range of C accumulation in the core was from less than 10 g m^{-2} yr^{-1} to over 80 g m^{-2} yr^{-1}, and was also within the range of estimates provided by other researchers (*e.g.* Tolonen and Turunen, 1996). These estimates of C accumulation really relate to increases in the catotelm C store and can be viewed as long-term rates since decay in the catotelm is very slow (Clymo, 1984). By assuming that the MH2 core was representative of the uneroding blanket bog at Moor House, and multiplying the C accumulation rates by the area of this soil type, the total amount of C sequestered for the site can be calculated. The results suggest that on average *ca.* 0.4×10^6 kg of C, equal to between 5 % and 17 % of the annual NPP of the entire study site, is being sequestered by the blanket bog at Moor House each year.

As suggested in Figure 1.7, C is being lost from Moor House in streams as dissolved organic C (DOC). This fraction is usually defined as the organic C which passes through a 0.45 μm filter (Eatherall *et al.*, 1998) and has been shown by Eatherall *et al.* (1998) to correlate with the proportional area of Winter Hill blanket bog soil series in a catchment. Based on the correlation provided by Eatherall *et al.* (1998), the total amount of DOC removed from the Moor House blanket bog should be *ca.* 10-20 g m^{-2} yr^{-1}.

The present flux of C through peat erosion from the study site can be estimated by extrapolating the results of Crisp (1966). He quantified the flux of peat transported out of the Rough Sike catchment (a tributary of Moss Burn; Figure 1.5) for a whole year (1963). The catchment of this river is described by Crisp (1966) as having approximately the same proportion of eroding area as the eastern plateau of Moor House where blanket bog is the dominant soil type. Therefore, extrapolation of the loss of peat from this catchment can reasonably be used to estimate the annual loss of C from the study area.

Crisp (1966) estimated a total of 93,064 kg of peat was lost from the Rough Sike catchment in his year of study. The catchment was estimated to cover an area of 83 ha and, assuming that the eroded peat had a C concentration of 50 % (Heal and Smith, 1978; Allen, 1989; Immirzi *et al.*, 1992), the total amount of C loss from Moor House due to peat erosion is estimated at 56 g m^{-2} yr^{-1} (*i.e.* [93,064 x 1,000] / [83 x 100 x 100]). This represents a considerable loss of C and is double the average rate of C accumulation obtained from the MH2 core (27 g m^{-2} yr^{-1}).

Peat erosion is also likely to increase drainage from surrounding areas, lowering the water table and, according to the results of the C balance model and predictions from experiments (Oechel *et al.*, 1998), reduce rates of C accumulation in peats.

These estimated fluxes of C under present conditions for the blanket bog at Moor House are summarised in Table 8.1. The values are based on the assumptions discussed above and, therefore, are only an approximation of the C balance of Moor House.

Table 8.1 Present C balance for the blanket bog area at Moor House

C flux	Rate (g m^{-2})
NPP minus decomposition	+ 27
SCP deposition	+ 0.03
Particulate organic C (POC)	- 56
DOC	- 10 to 20
Net C balance	-39 to -49

Therefore, in terms of total terrestrial C at the site, Moor House is probably losing C at a greater rate than it is gaining it. This statement relies on the assumption that the non-blanket bog areas can be considered to be in C balance; even if these areas did represent sources or sinks, their minor proportion of the total area means their influence on the total flux is small.

Future C storage at Moor House; predictions from model results and analogues from the past

The C balance model developed in Chapter 6 illustrated the sensitivity of terrestrial C stores to changes in climate. Despite only minor changes in C accumulation rates associated with changes in mean annual temperature and total annual rainfall, the depth of the water table and species composition of the peat forming vegetation had potentially large impacts on C accumulation rates; both water tables (Laine *et al.*, 1996) and vegetation species composition of mires (Birks and Birks, 1980) are strongly controlled by climate.

The importance of plant species composition on C accumulation rates in peats may have other implications for understanding peat growth. For example, there are wide variations in plant species composition across mires caused by factors such as micro-topography and proximity to water table depth. Where species which result in greater C accumulation occur, due to lower rates of decomposition, the peat may be expected to accumulate more rapidly. This may account for changes in micro-topography of mires and the formation of pools and hummocks (Belyea, 1996).

The strong influence of plant species composition on C accumulation rates predicted from the C balance model supports the results of other workers. For example, Tolonen and Turunen (1996) predicted that the total amount of C accumulating in Finnish mires will increase following global warming because mire species which decompose more slowly will occupy a greater proportion of Finnish mires. A similar mechanism may have caused the increased C accumulation rates observed by Oldfield *et al.* (1997) in Stor Åmyrân, northern Sweden, whereby C accumulation rates were highest when tree-ring evidence suggested the climate was warmer; perhaps these warmer climatic conditions favoured plant species which decomposed more slowly. A similar mechanism may explain the different response between these northern mires and the results from Moor House, since a warmer climate may favour species which decompose more rapidly at Moor House.

It has been predicted that global warming will lower water tables in peatlands (Gorham, 1991; Laine *et al.*, 1996). The C balance model described in the present study showed that a moderate increase in water table depth would cause greater decomposition of peat, reducing rates of C accumulation and potentially causing C accumulation in peats to cease. Alternatively, the model predicted that higher water tables associated with wetter mire conditions would cause a rapid increase in C accumulation rates. This supports recent evidence from experimental observations in high C storage areas (*e.g.* Oechel *et al.*, 1993, 1998).

The model predictions and rates of C accumulation from the blanket bog profile investigated suggest that C is being sequestered at Moor House at around 20-30 g C m^{-2} yr^{-1}. Assuming that the blanket bog started forming at *ca.* 7500 BP (Heal and Smith, 1978), and that the mean C storage of the blanket bog at Moor House is 87 kg m^{-2}, (Calluneto-Eriophoretum; Chapter 3), then the long-term rate of C accumulation is *ca.* 12 g m^{-2} yr^{-1}. This value is about half the rate of the mean estimate for C accumulation in the MH2 core, at least partly because decomposition continues (although at a slow rate) within the catotelm (Clymo, 1984).

Both the investigation of past C accumulation and the C balance model suggest that a consequence of global warming will be reduced rates of C accumulation in the blanket bog at Moor House. The small areas of Moor House on non-blanket bog areas are also likely to lose C as demonstrated by soil heating and transplant experiments performed along the GDF altitudinal transect (Ineson *et al.*, in prep). These experiments also suggested that C loss as DOC will increase following global warming (Tipping *et al.*, submitted). Furthermore, peat erosion may increase, as Crisp (1966) found that most particulate organic carbon (POC) is transported during extreme events such as floods, and global warming is expected to increase climate extremes (Kattenberg *et al.*, 1996).

Potential to increase terrestrial C storage through management options

The impact of grazing and burning on the C storage at Moor House has been investigated in the present study, being major land management options for Moor House and similar moorland areas.

After over forty years, cessation of sheep grazing on grassland areas was not found to affect soil C storage, although there were small and significant increases in vegetation C stores. Cessation of sheep grazing is therefore not considered a viable method for increasing C storage in upland areas, although it may be that at lower altitudes colonisation by trees could cause larger and more important increases in vegetation C stores. However, in high altitude areas tree growth is currently restricted by climate.

Burning at Moor House was found to significantly reduce the rate of C accumulation in peats, or even nullify C accumulation in peats altogether. Burning is a widespread management technique in upland areas, with small patches of moorland being typically burnt every *ca.* 10-20 years. It was not possible to quantify the impact of burning on long term rates of C accumulation since the sampled peat was within the acrotelm. However, significantly greater amounts of C had accumulated when the blanket bog was not burnt, with an extra *ca.* 74 g C m^{-2} yr^{-1} being sequestered, compared to the burnt bog. In reality, the total difference in C accumulation between burnt and unburnt areas would be reduced since a large proportion of the C would be mineralised through decomposition before reaching the catotelm. However, the results from Moor House suggest that prevention of burning would cause higher rates of C accumulation in the blanket bogs.

Therefore, the management options investigated in the present study do not appear to offer major means for sequestering large amounts of C, although if the extent of moorland burning was reduced there may be some small gains in terrestrial C storage. The Moor House results support the conclusions from other studies which have indicated that changes in the management of terrestrial C stores are unlikely to significantly offset the huge amounts of C emitted from fossil fuel burning (*e.g.* Smith *et al.*, 1997).

Implications for global C cycling and climate change based on the Moor House results

Table 8.1 shows that, despite C accumulating in the blanket bog at Moor House, the net C balance is apparently negative under present conditions, largely due to the loss of C from the site as POC derived from peat erosion. However, as discussed below, the fate of this C may not necessarily be a return to the atmosphere (as CO_2), and therefore, it is uncertain as to whether this loss of C from the Moor House site directly contributes to atmospheric CO_2 levels. Once in the river system, POC may either be transported to coastal seas and estuaries, or stored within the river system in river banks or lakes and reservoirs. Storage in reservoirs may be temporary as these are flushed to minimise the reduced capacity caused by sedimentation (Newson, 1992). The POC may be oxidised in the river system (Schlesinger, 1991) or oxidised in coastal waters and estuaries (Schlesinger, 1991; Holligan, 1994). Alternatively, the POC may be buried in coastal sediments and, therefore, not returned to the atmosphere (Holligan, 1994). However, since Holligan (1994) claims that the flux of C from land to the ocean is greater than the rate of organic matter burial in oceans, the oceans can be considered as a net source for CO_2. It is therefore likely that the increased POC caused by peat erosion will increase the inbalance between ocean C fluxes. The additional transfer of C, due to the respiration of POC within rivers, means the POC lost from Moor House must at least result in some transfer of C from the Moor House terrestrial C store to the atmosphere. However, all blanket bog areas on the eastern slopes of Moor House drain into the Cow Green Reservoir, and it is likely that a large proportion of peat eroded from Moor House in the last 20 years is being, at least temporarily, stored in this reservoir.

Peat erosion is particularly common in blanket bogs and it is unlikely that the vast areas of peatland in the rest of the world are subject to the high erosion rates found at Moor House. However, the rate of C accumulation in the Moor House peats appears to be strongly influenced by climate, and it is reasonable to assume that climate may have similar influences on the rate of C accumulation in other peats. Despite the conflict in results between the investigation of Oldfield *et al.* (1997) at Stor Åmyrån and the results from core MH2, estimates of the impact of changes in the rate of C accumulation can be made. For example, rates of C accumulation in the Moor House core appear to have ranged from less than 10 g m^{-2} yr^{-1} to approximately 90 g m^{-2} yr^{-1}, values similar to those reported by Tolonen and Turunen (1996) for lowland Finnish peats. Given that the area of undrained northern peatlands is *ca.* 334 x 10^6 ha (Gorham, 1991), C accumulation in these mires may range from less than 33 x 10^9 to 300 x 10^9 kg yr^{-1}; the average rate suggested by Gorham (1991) was *ca.* 97 x 10^9 kg yr^{-1}. The range of C accumulation values represents between 1 % and 5 % of the estimated anthropogenic C emissions (*ca.* 5,500 x 10^9 kg C yr^{-1}; Schimel, 1995). Therefore, the responses of C accumulation in mires to global warming are of considerable importance.

Suggestions for future research

The investigations of the application of NIRS which have formed a secondary part of this thesis have shown the potential of this technique for palaeoecological studies (Chapter 7; McTiernan *et al.*, in press). However, the true potential of the technique depends on the extent to which it can reduce the need for conventional analyses. The investigation in the present study has been from a single peat profile from one site and further work is clearly required to determine whether calibrations determined for one type of peat can be used for other types of peat.

The apparent ability of NIRS to correlate with the radiocarbon age of peat is an exciting prospect since radiocarbon dating is an expensive and time consuming process and frequently hinders palaeoecological studies. Further investigation of this correlation is required in order to determine the usefulness of the technique and the inter-site reliability of calibrations.

NIRS was only used to predict a relatively small set of variables within the present work, but based on these results, it is probable the technique could be used to measure many more variables routinely investigated in palaeoecological studies.

The results from the present investigation suggest that the rate of C accumulation in mires will decrease following global warming. This is in agreement with the results of other workers (*e.g.* Oechel *et al.*, 1993, 1998), although conflicts with others (*e.g.* Oldfield *et al.*, 1997; Tolonen and Turunen, 1996). There are possible reasons why some ecosystems may increase C storage rates and some decrease following warming and further investigations are needed to resolve this important question. Further studies investigating the impact of past climate change on the rate of C accumulation in a wide range of peatlands under different climates are required.

The present investigation has illustrated the large uncertainties in current inventories of terrestrial C stores. The estimate of C storage for Moor House was very different from the published national database estimate for the same area, yet it is unclear whether the same level of inaccuracy can be found across the rest of the UK, and within global C inventories. Further investigation is required to test the accuracy of the national database at other sites.

More representative estimates of terrestrial C storage require a greater number of sampling points and estimates of the area of different vegetation and soil types. However, application of remote sensing could allow more accurate inventories of terrestrial C stores. Reid *et al.* (1994) illustrated the potential of using such an approach for mapping the vegetation of south Pennine moorlands. Further research is required in order to investigate whether such an approach can improve estimates of vegetation C storage. Additionally, there may be ways to map soils more accurately using similar techniques, or to estimate peat depth (*e.g.* possibly related to species composition) to improve estimates of soil C stores.

The results of the present investigation indicate that it is unlikely that any management practice on upland areas such as Moor House could sequester the large amounts of C required to significantly offset fossil fuel emissions of CO_2. However, since these areas contain large quantities of C, which could be released to the atmosphere, inappropriate usage of areas like Moor House could significantly contribute to the increasing atmospheric CO_2 concentrations. Therefore, consideration of land uses which occur in areas of high C storage is necessary and should be accounted for in national inventories of C fluxes. Since there are other land use activities which are extensively practised in areas of high C storage such as Moor House, investigation of their impact is necessary. For example, commercial afforestation has taken place on large areas of upland Britain with high C storage (*e.g.* south Scotland and Kielder), yet the assumptions that increased C storage will result have only recently begun to be questioned (*e.g.* Harrison *et al.*, 1995b, 1997). Furthermore, several national forests have recently been created, partly in an effort to increase terrestrial C stores, yet, if this results in a decrease in soil C storage greater than the associated increase in vegetation C stores, such forests will contribute rather than offset CO_2 emissions. Careful consideration of this issue is necessary in formulating policies.

As mentioned above, peat erosion appears to be an important process in reducing the size of the C store at Moor House. Despite this importance, the author is unaware of any published studies of the direct impact of peat erosion on C storage, yet the evidence presented above suggests that peat erosion will contribute to increased CO_2 concentrations, although there are many uncertainties. The possible causes of peat erosion (*e.g.* over-grazing, pollution, climate change) have been well documented (Stevenson *et al.*, 1990) and there may be management options to reduce rates of erosion (*e.g.* lower grazing densities). Further work is required to determine the impact of peat erosion on net terrestrial C storage and whether reduction of erosion is one option for increasing C stores.

The accuracy of the rates of C accumulation derived in the present study for core MH2 were strongly dependent on the constructed chronology. Improved methods of dating are essential for palaeoecological studies of peats as has been recognised by many authors (*e.g.* Mauquoy, 1997; Chambers *et al.*, 1997). However, more confidence in the chronology would have been gained if known tephra layers had been clearly observed in core MH2. Future research on C accumulation in peatlands would benefit from being performed in peats with known tephra layers. Alternatively, pollen analysis can be used for dating purposes particularly when the pollen profile can be matched against documented historical changes in local vegetation (*e.g.* the changes in hemp cultivation illustrated by Gedye, 1998).

Improvements could be made in the interpretation of proxy climate signals in peats. The present investigation relied on PCA of the plant macro-fossils and testate amoebae data, and identification of gradients in the components believed to reflect the surface wetness conditions of the mire. However, although the relative wetness conditions preferred by different species are well known, it is really necessary to reconstruct surface wetness conditions in a more quantitative way. Such an approach has been attempted by Woodland *et al.* (1998) with the use of a transfer function derived from measuring contemporary water table depths and testate amoebae compositions. Using the transfer function water table depths can be predicted from the testate amoebae composition; a similar approach should be undertaken for plant macro-fossils.

Although an attractive feature of the C balance model described in Chapter 6 is its simplicity, allowing the impact of manipulating variables to be clearly observed, there is considerable scope for improvement in the model. Presently, only four species were modelled since only four blanket bog species have known regressions between weight loss and AET. Further litter bag studies could be performed to determine the rates of decomposition of other major peat-forming vegetation with AET (*e.g.* other species of *Sphagnum*).

Water tables were very simply simulated in the C balance model, although were shown to strongly influence the rates of C accumulation. The C balance model would more accurately simulate the impact of climate on C accumulation if water table depth and climate parameters were more closely coupled. This should include consideration of many factors, including the hydraulic conductivity of peat, which are not presently included in the current model.

In conclusion, the present investigation has demonstrated the importance of areas like Moor House for terrestrial C storage. The results suggest that climate has considerable influence on the size of the terrestrial C stores, and although there remain many uncertainties, I must conclude that the anticipated global warming will reduce C stores in areas like Moor House, and therefore, enhance the increasing atmospheric CO_2 concentration.

9. References

Aaby, B. 1976. Cyclic climatic variations in climate over the past 5,500 yr reflected in raised bogs. *Nature*, **263**, 281-284.

Allen, S.E. 1964. Chemical aspects of heather burning. *Journal of Applied Ecology*, **1**, 347-367.

Allen, S.E., ed. 1989. *Chemical Analysis of Ecological Materials.* 2nd ed. Oxford: Blackwell Scientific Publications.

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

Arrouays, D., Daroussin, J., Kicin, J.L. & Hassika, P. 1998. Improving topsoil carbon storage prediction using a digital elevation model in temperate forest soils of France. *Soil Science*, **163**, 103-108.

Ball, D.F. 1964. Loss-on-ignition as an estimate of organic matter and organic carbon in non-calcareous soils. *Journal of Soil Science*, **15**, 84-92.

Barber, K.E. 1981. *Peat Stratigraphy and Climate Change.* Rotterdam: A.A.Balkema.

Barber, K.E., Chambers, F.M., Maddy, D., Stoneman, R. & Brew, J.S. 1994. A sensitive high-resolution record of late Holocene climatic change from a raised bog in northern England. *Holocene*, **4**, 198-205.

Basher, L.R. & Lynn, I.H. 1996. Soil changes associated with the cessation of sheep grazing in the Canterbury High Country, New Zealand. *New Zealand Journal of Ecology*, **20**, 179-189.

Batjes, N.H. 1996. Total carbon and nitrogen in the soils of the world. *European Journal of Soil Science*, **47**, 151-163.

Batjes, N.H. & Sombroek, W.G. 1997. Possibilities for carbon sequestration in tropical and subtropical soils. *Global Change Biology*, **3**, 161-173.

Belyea, L.R. 1996. Separating the effects of litter quality and microenvironment on decomposition rates in a patterned peatland. *Oikos*, **77**, 529-539.

Berg, B., Jansson, P.-E. & Meentemayer, V. 1984. Litter decomposition and climate - regional and local models. In: *State and Change of Forest Ecosystems - Indicators in Current Research*, edited by G.I. Agren, 389-404.

Bird, M.I., Lloyd, J. & Farquhar, G.D. 1994. Terrestrial carbon storage at the Late Glacial Maximum. *Nature*, **371**, 566.

Birks, H.J.B. & Birks, H.H. 1980. *Quaternary Palaeoecology.* London: Edward Arnold.

Blackford, J. 1993. Peat bogs as sources of proxy climatic data: past approaches and future research. In: *Climate Change and the Human Impact*, edited by F.M. Chambers, 47-56.

Blackford, J.J. & Chambers, F.M. 1991. Proxy records of climate from blanket mires: evidence for a Dark Age (1400 BP) climatic deterioration in the British Isles. *Holocene*, **1**, 63-67.

Blackford, J.J. & Chambers, F.M. 1993. Determining the degree of peat decomposition for peat-based palaeoclimatic studies. *International Peat Journal*, **5**, 7-24.

Blackford, J.J. & Chambers, F.M. 1995. Proxy climate record for the last 1000 years from Irish blanket peat and a possible link to solar variability. *Earth and Planetary Science Letters*, **133**, 145-150.

Bol, R., Harkness, D.D., Huang, Y. & Howard, D.M. in press. ^{12}C, ^{13}C and ^{14}C distribution and carbon turnover in three British upland soils. *European Journal of Soil Science.*

Bowes, M. 1996. *Analysis of Lake Sediments by Near-infrared Spectroscopy and Pyrolysis Mass Spectrometry: The Development of New Methods to infer Past Lake Water Chemistry.* BSc Thesis. Department of Geography, University of Newcastle-Upon-Tyne.

Bradley, R.S. & Jones, P.D. 1993. Little Ice Age summer temperature variations: their nature and relevance to recent global warming trends. *Holocene*, **3**, 367-376.

Bronk Ramsey, C. 1995. Radiocarbon calibration and analysis of stratigraphy: the OxCal program. *Radiocarbon*, **37**, 425-430.

Buringh, P. 1984. Organic carbon in soils of the world. In: *The Role of Terrestrial Vegetation in the Global Carbon Cycle: Measurement by Remote Sensing*, edited by G.M. Woodwell, 91-109. Chichester: John Wiley and Sons Ltd.

Burt, T.P., Adamson, J.K. & Lane, A.M.J. in press. Long-term rainfall and streamflow records for north central England: putting the Environmental Change Network site at Moor House in context. *Hydrological Sciences Journal.*

Cannell, M.G.R., Dewar, R.C. & Pyatt, D.G. 1993. Conifer plantations on drained peatlands in Britain: a net gain or loss of carbon? *Forestry*, **66**, 353-369.

Cannell, M.G.R. & Milne, R. 1995. Carbon pools and sequestration in forest ecosystems in Britain. *Forestry*, **68**, 361-378.

Chambers, C. 1978. A radio-carbon dated pollen-diagram from Valley Bog, on the Moor House National Nature Reserve. *New Phytologist*, **80**, 273-280.

Chambers, F.M., Barber, K.E., Maddy, D. & Brew, J. 1997. A 5,500-year proxy-climate and vegetational record from blanket mire at Talla Moss, Borders, Scotland. *Holocene*, **7**, 391-399.

Chapman, K. 1986. *Interaction between Tree Species: Decomposition and Nutrient Release from Litters.* Doctor of Philosophy Thesis. University of Lancaster.

Christen, J.A., Clymo, R.S. & Litton, C.D. 1995. A Bayesian approach to the use of ^{14}C dates in the estimation of the age of peat. *Radiocarbon*, **37**, 431-442.

Clymo, R.S. 1965. Experiments on the breakdown of *Sphagnum* in two bogs. *Journal of Ecology*, **53**, 747-758.

Clymo, R.S. 1970. The growth of *Sphagnum*: Methods of measurement. *Journal of Ecology*, **58**, 13-49.

Clymo, R.S. 1978. A model of peat bog growth. In: *Production Ecology of British Moors and Montane Grasslands*, edited by O.W. Heal & D.F. Perkins, 187-223. Berlin: Springer-Verlag.

Clymo, R.S. 1983. Chapter 4. Peat. In: *Ecosystems of the World*, edited by A.J.P. Gore, Vol. 4a, 440. Amsterdam: Elsevier Scientific.

Clymo, R.S. 1984. The limits to peat bog growth. *Philosophical Transactions of the Royal Society of London B*, **303**, 605-654.

Clymo, R.S. 1991. Peat Growth. In: *Quaternary Landscapes*, edited by L.C.K. Shane & E.J. Cushing, 76-112. London: Bellhaven.

Clymo, R.S. 1993. Models of peat growth. *Suo*, **43**, 127-136.

Clymo, R.S. 1996. Assessing the accumulation of carbon in peatlands. In: *Northern Peatlands in Global Climate Change*, edited by R. Laiho, J. Laine & H. Vasander, 207-212. SILMU: Proceedings of the international workshop held in Hyytiälä, Finland, 8-12 October, 1995.

Clymo, R.S. & Hayward, P.M. 1982. The ecology of *Sphagnum*. In: *Bryophyte Ecology*, edited by A.J.E. Smith, 30-78. London: Chapman and Hall.

Clymo, R.S., Oldfield, F., Appleby, P.G., Pearson, G.W., Ratnesar, P. & Richardson, N. 1990. The record of atmospheric deposition on a rainwater-dependent peatland. *Philosophical Transactions of the Royal Society of London B*, **327**, 331-338.

Clymo, R.S. & Pearce, M.E. 1995. Methane and carbon dioxide production in, transport through, and efflux from a peatland. *Philosophical Transactions of the Royal Society of London A*, **350**, 249-259.

Clymo, R.S., Turunen, J. & Tolonen, K. 1998. Carbon accumulation in peatland. *Oikos*, **81**, 368-388.

Conway, V.M. & Millar, A. 1960. The hydrology of some small peat-covered catchments in the northern Pennines. *Journal of the Institute of Water Engineers*, **14**, 415-424.

Cornelissen, J.H.C. 1996. An experimental comparison of leaf decomposition rates in a wide range of temperate plant species and types. *Journal of Ecology*, **84**, 573-582.

Cotrufo, M.F. 1994. *Effects of Enriched Atmospheric Concentration of Carbon Dioxide on Tree Litter Decomposition*. Doctor of Philosophy Thesis. University of Lancaster.

Crisp, D.T. 1966. Input and output of minerals for an area of Pennine moorland: the importance of precipitation, drainage, peat erosion and animals. *Journal of Applied Ecology*, **3**, 314-327.

Cruikshank, M.M. & Tomlinson, R.W. 1997. Carbon loss from UK peatlands for fuel and horticulture. In: *Carbon Sequestration in Vegetation and Soils*, edited by M.G.R. Cannell. Institute of Terrestrial Ecology: Report to the Department of Environment.

Cruikshank, M.M., Tomlinson, R.W., Devine, P.M. & Milne, R. 1996. Estimation of carbon stores and production of carbon density maps for Northern Ireland. In: *Carbon Sequestration in Vegetation and Soils*, edited by M.G.R. Cannell. Institute of Terrestrial Ecology: Report to the Department of Environment.

Cuttle, S.P. & Malcolm, D.C. 1979. A corer for taking undisturbed peat samples. *Plant and Soil*, **51**, 297-300.

Daniels, R.E. & Eddy, A. 1990. *A Handbook of European Sphagna.* Swindon: Natural Environment Research Council.

Ditlevesen, P.D., Svensmark, H. & Johnsen, S. 1996. Contrasting atmospheric and climate dynamics of the last-glacial and Holocene periods. *Nature*, **379**, 810-812.

Dixon, R.K. & Krankina, O.N. 1995. Can the terrestrial biosphere be managed to conserve and sequester carbon? In: *Carbon Sequestration in the Biosphere*, edited by M.A. Beran, 153-179. Heidelberg: Springer-Verlag.

Downey, G. & Byrne, P. 1986. Prediction of moisture and bulk density in milled peat by near infrared reflectance. *Journal of the Science of Food and Agriculture*, **37**, 231-238.

Dugmore, A.J., Larsen, G. & Newton, A.J. 1995. Seven tephra isochrones in Scotland. *Holocene*, **5**, 257-266.

Eatherall, A., Naden, P.S. & Cooper, D.M. 1998. Simulating carbon flux to the estuary: The first step. *Science of the Total Environment*, **210/211**, 519-533.

Eddy, A., Welch, D. & Rawes, M. 1969. The vegetation of the Moor House National Nature Reserve in the Northern Pennines, England. *Vegetatio*, **16**, 239-284.

ESRI. 1991. *Surface Modeling with TIN.* Redlands, California: Environmental Systems Research Institute.

Eswaran, H., van der Berg, E. & Reich, P. 1993. Organic carbon in soils of the world. *Journal of the Soil Science Society of America*, **57**, 192-194.

Fitter, A.H., Graves, J.D., Self, G.K., Brown, T.K., Bogie, D.S. & Taylor, K. 1998. Root production, turnover and respiration under two grassland types along an altitudinal gradient: influence of temperature and solar radiation. *Oecologia*, **114**, 20-30.

Forrest, G.I. 1971. Structure and production of north Pennine blanket bog vegetation. *Journal of Ecology*, **59**, 453-479.

Forrest, G.I. & Smith, R.A.H. 1975. The productivity of a range of blanket bog vegetation types in the northern Pennines. *Journal of Ecology*, **63**, 173-202.

Fowler, D., Cape, J.N., Leith, I.D., Chourlarton, T.W., Gay, M.J. & Jones, A. 1988. The influence of altitude on rainfall composition at Great Dun Fell. *Atmospheric Environment*, **22**, 1355-1362.

Frankland, J.C. 1974. Decomposition of lower plants. In: *Biology of Plant Litter Decomposition*, edited by C.H. Dickinson & G.J.F. Pugh, 3-36. London: Academic Press.

Garnett, M.H. & Adamson, J.K. 1997. Blanket mire monitoring and research at Moor House National Nature Reserve. In: *Blanket Mire Degradation: Causes, Consequences and Challenges*, edited by J.H. Tallis, R. Meade & P.D. Hulme, 116-117. Aberdeen: Macaulay Land Use Research Institute.

Garnett, M.H., Ineson, P. & Adamson, J.K. 1997a. A long-term upland temperature record: No evidence for recent warming. *Weather*, **52**, 342-351.

Garnett, M.H., Ineson, P. & Howard, D.C. 1996. *Soil and Vegetation Carbon Stores in UK Uplands: Moor House Case Study.* Institute of Terrestrial Ecology: Progress report to the Department of Environment.

Garnett, M.H., Ineson, P. & Howard, D.C. 1997b. *Terrestrial Carbon Stores in UK Uplands: Moor House Case Study.* Institute of Terrestrial Ecology: Final report to the Department of Environment.

Gedye, S. 1998. *Mass Balance in Recent Peats.* Ph.D. Thesis. University of Liverpool.

Gorham, E. 1991. Northern peatlands: Role in the carbon cycle and probable responses to climatic warming. *Ecological Applications*, **1**, 182-195.

Grant, S.A., Bolton, G.R. & Torvell, L. 1985. The responses of blanket bog vegetation to controlled grazing by hill sheep. *Journal of Applied Ecology*, **22**, 739-751.

Harrison, A.F. & Bocock, K.L. 1981. Estimation of soil bulk-density from loss-on-ignition values. *Journal of Applied Ecology*, **8**, 919-927.

Harrison, A.F., Howard, P.J.A., Howard, D.M., Howard, D.C. & Hornung, M. 1995a. Carbon storage in forest soils. *Forestry*, **68**, 335-348.

Harrison, A.F., Jones, H.E., Howson, G., Garnett, J. & Woods, C. 1997. Long term changes in the carbon balance of afforested peatlands. In: *Carbon Sequestration in Vegetation and Soils*, edited by M.G.R. Cannell. Institute of Terrestrial Ecology: Report to the Department of Environment.

Harrison, A.F., Jones, H.E., Howson, G. & Garnett, M.H. 1995b. *Impacts of Forestry on Carbon Balance in Peats.* Report to the Department of Environment: Institute of Terrestrial Ecology.

Harrison, A.F., Taylor, K., Hatton, J.C. & Howard, D.M. 1994. Role of nitrogen in herbage production by Agrostis-Festuca hill grassland. *Journal of Applied Ecology*, **31**, 351-360.

Heal, O.W. 1962. The abundance and micro-distribution of testate amoebae (Rhizopoda: testacea) in *Sphagnum. Oikos*, **13**, 35-47.

Heal, O.W., Jones, H.E. & Whittaker, J.B. 1975. Moor House, UK. In: *Structure and Function of Tundra Ecosystems*, edited by T. Rosswall & O.W. Heal, 295-320. Stockholm: Ecol. Bull.

Heal, O.W., Latter, P.M. & Howson, G. 1978. A study of the rates of decomposition of organic matter. In: *Production Ecology of British Moors and Montane Grasslands*, edited by O.W. Heal & D.F. Perkins, 136-159. Berlin: Springer-Verlag.

Heal, O.W. & Perkins, D.F., eds. 1978. *Production Ecology of British Moors and Montane Grasslands*. Berlin: Springer-Verlag.

Heal, O.W. & Smith, R.A.H. 1978. Introduction and site description. In: *Production Ecology of British Moorlands and Montane Grasslands*, edited by O.W. Heal & D.F. Perkins, 3-16. Berlin: Springer-Verlag.

Henderson-Sellers, A. & McGuffie, K. 1995. Global climate models and 'dynamic' vegetation changes. *Global Change Biology*, **1**, 63-75.

Hendon, D. & Charman, D.J. 1997. The preparation of testate amoebae (Protozoa: Rhizopoda) samples from peat. *Holocene*, **7**, 199-205.

Hobbs, R.J. 1984. Length of burning rotation and community composition in high-level Calluna-Eriophorum bog in N England. *Vegetatio*, **57**, 129-136.

Hobbs, R.J. & Gimingham, C.H. 1987. Vegetation, fire and herbivore interactions in heathland. *Advances in Ecological Research*, **16**, 87-173.

Holligan, P.M. 1994. Carbon sequestration in marine systems. In: *Carbon Sequestration in the Biosphere*, edited by M.A. Beran, 47-68. Berlin: Springer.

Hornung, M. 1968. *The Morphology, Mineralogy and Genesis of some Soils on the Moor House National Nature Reserve*. Ph.D. Thesis. University of Durham.

Houghton, R.A. 1995. Land-use change and the carbon cycle. *Global Change Biology*, **1**, 275-287.

Howard, P.J.A. & Howard, D.M. 1976. *Respiration, Litter Nutrients, and Soil Organic Matter in Grazed and Ungrazed Upland Limestone Grassland.* Institute of Terrestrial Ecology. Merlewood Research and Development Paper Number 67.

Howard, P.J.A., Loveland, P.J., Bradley, R.I., Dry, F.T., Howard, D.M. & Howard, D.C. 1995. The carbon content of soil and its geographical distribution in Great Britain. *Soil Use and Management*, **11**, 9-15.

Immirzi, C.P., Maltby, E. & Clymo, R.S. 1992. *The Global Status of Peatlands and their Role in Carbon Cycling*. London: Friends of the Earth.

Ineson, P., Benham, D.G., Poskitt, J., Harrison, A.F., Taylor, K. & Woods, C. 1998a. Effects of climate change on nitrogen dynamics in upland soils. 2. A soil warming study. *Global Change Biology*, **4**, 153-161.

Ineson, P., Garnett, J.S., Howson, G., Garnett, M.H. & Roberts, J.D. in press. Elevated CO_2 reduces long-term field decomposition rates of *Rubus chamaemorus* L. leaf litter. *Soil Biology and Biochemistry*.

Ineson, P. & Stevens, P.A. 1990. Soil processes. In: *The Greenhouse Effect and Terrestrial Ecosystems of the UK*, edited by M.G.R. Cannell & M.D. Hooper, 14-18. HMSO, London: Institute of Terrestrial Ecology Research Publication No. 4.

Ineson, P., Taylor, K., Harrison, A.F., Poskitt, J., Benham, D.G., Tipping, E. & Woof, C. 1998b. Effects of climate change on nitrogen dynamics in upland soils. 1. A transplant approach. *Global Change Biology*, **4**, 143-152.

Ingram, H.A.P. 1978. Soil layers in mires: Function and terminology. *Journal of Soil Science*, **29**, 224-227.

IPCC. 1995. *Climate Change 1994*. Cambridge: Cambridge University Press.

IPCC. 1996. *Climate Change 1995*. Cambridge: Cambridge University Press.

Jain, T.B., Graham, R.T. & Adams, D.L. 1997. Carbon to organic matter ratios for soils in Rocky Mountain coniferous forests. *Journal of the Soil Science Society of America*, **61**, 1190-1195.

Janssens, J.A. 1983. A quantitative method for stratigraphic analysis of bryophytes in Holocene peat. *Journal of Ecology*, **71**, 189-196.

Jansson, P. 1994. *SOIL model version. 7.5*. Uppsala: Swedish University of Agricultural Sciences.

Jeffrey, D.W. 1970. A note on the use of ignition loss as a means for the approximate estimation of soil bulk density. *Journal of Ecology*, **58**, 297-299.

Joffre, R., Gillon, D., Dardenne, P., Agneessens, R. & Biston, R. 1992. The use of near-infrared reflectance spectroscopy in litter decomposition studies. *Annales des Sciences Forestieres*, **49**, 481-488.

Johnson, G.A.L. & Dunham, K.C. 1963. *The Geology of Moor House*. 1st ed. London: Her Majesty's Stationery Office.

Jones, H.E. & Gore, A.J.P. 1978. A simulation of production and decay in blanket bog. In: *Production Ecology of British Moor and Montane Grassland*, edited by O.W. Heal & D.F. Perkins, 160-186. Berlin: Springer-Verlag.

Kattenberg, A., Giorgi, F., Grassl, H., Meehl, G.A., Mitchell, J.F.B., Stouffer, R.J., Tokioka, T., Weaver, A.J. & Wigley, T.M.L. 1996. Climate models-projections of future climate. In: *Climate Change 1995*, edited by J.T. Houghton, L.G. Meira Filho, B.A. Callander, N. Harris, A. Kattenberg & K. Maskell, 285-357. Cambridge: Cambridge University Press.

Kilian, M.R., van der Plicht, J. & van Geel, B. 1995. Dating of raised bogs: New aspects of AMS ^{14}C wiggle matching, a reservoir effect and climatic change. *Quaternary Science Reviews*, **14**, 959-966.

Klinger, L.F., Taylor, J.A. & Franzen, L.G. 1996. The potential role of peatland dynamics in Ice-Age initiation. *Quaternary Research*, **45**, 89-92.

Korhola, A., Tolonen, K., Turunen, J. & Junger, H. 1995. Estimating long-term carbon accumulation rates in boreal peatlands by radiocarbon dating. *Radiocarbon*, **37**, 575-584.

Korsman, T., Nilsson, M., Ohman, J. & Renberg, I. 1992. Near-infrared reflectance spectroscopy of sediments: a potential method to infer the past pH of lakes. *Environmental Science and Technology*, **26**, 2122-2126.

Kuhry, P. 1994. The role of fire in the development of *Sphagnum*-dominated peatlands in western boreal Canada. *Journal of Ecology*, **82**, 899-910.

Kuhry, P. & Vitt, D.H. 1996. Fossil carbon/nitrogen ratios as a measure of peat decomposition. *Ecology*, **77**, 271-275.

Laine, J., Silvola, J., Tolonen, K., Alm, J., Nykanen, H., Vasander, H., Sallantaus, T., Savolainen, I., Sinisalo, J. & Martikainen, P.J. 1996. Effect of water-level drawdown on global climatic warming: northern peatlands. *Ambio*, **25**, 179-184.

Lapen, D.R., Moorman, B.J. & Price, J.S. 1996. Using ground-penetrating radar to delineate subsurface features along a wetland catena. *Soil Science Society of America*, **60**, 923-931.

Latter, P.M., Howson, G., Howard, D.M. & Scott, W.A. 1998. Long-term study of decomposition on a Pennine peat bog: which regression? *Oecologia*, **113**, 94-103.

Lee, J.A. 1998. Unintentional experiments with terrestrial ecosystems: ecological effects of sulphur and nitrogen pollutants. *Journal of Ecology*, **86**, 1-12.

Leith, H. 1975. Modeling the primary productivity of the world. In: *Primary Production of the Biosphere*, edited by H. Leith & R.H. Whittaker, 237-263. Berlin: Springer-Verlag.

Mahalanobis, P.C. 1936. On the generalised distance in statistics. *Proceedings of the Natural Institute of Science, India*, **2**, 49-55.

Malley, D.F. & Nilsson, M. 1995. Environmental applications of near infrared spectroscopy: Seeing the environment in a different light. *Spectroscopy Europe*, **7**, 8-16.

Malmer, N. & Holm, E. 1984. Variation in the C/N-quotient of peat in relation to decomposition rate and age determination with Pb-210. *Oikos*, **42**, 171-182.

Maltby, E., Legg, C.J. & Proctor, M.C.F. 1990. The ecology of severe moorland fire on the North York Moor: effects of the 1976 fires, and subsequent surface and vegetation development. *Journal of Ecology*, **78**, 490-518.

Manley, G. 1980. The northern Pennines revisited: Moor House, 1932-1978. *The Meteorological Magazine*, **109**, 281-292.

Mann, M.E., Bradley, R.S. & Hughes, M.K. 1998. Global-scale temperature patterns and climate forcing over the past six centuries. *Nature*, **392**, 779-787.

Marion, G.M. & Oechel, W.C. 1993. Mid- to late-Holocene carbon balance in Arctic Alaska and its implications for future global warming. *Holocene*, **3**, 193-200.

Marrs, R.H., Rawes, M., Robinson, J.S. & Poppit, S.D. 1986. *Long-term Studies of Vegetation Change at Moor House NNR: Guide to Recording Methods and the Database.* Institute of Terrestrial Ecology: Merlewood Research and Development paper no. 109.

Martens, H. & Jensen, S.A. 1982. Partial least squares regression: a new two-stage NIR calibration method. In: *Proceedings of the 7th World Cereal Bread Congress*, edited by J. Holas & R. Kratchovil, 607-647. Amsterdam: Elsevier.

Mauquoy, D. 1997. *Testing the Sensitivity of the Palaeoclimate Signal from Ombrotrophic Peat Stratigraphy.* Doctor of Philosophy Thesis. University of Southampton.

McTiernan, K.B. 1998. *The Effect of Climate on the Decomposition of Chemical Constituents of Tree Litters.* Doctor of Philosophy Thesis. University of Lancaster.

McTiernan, K.B., Garnett, M.H., Mauquoy, D., Ineson, P. & Coûteaux, M-M. in press. Use of near-infrared reflectance spectroscopy (NIRS) in palaeoecological studies of peat. *Holocene.*

Meentemeyer, V. 1971. *A Climatic Approach to the Prediction of Regional Differences in Decomposition Rate of Organic Debris in Forests.* Doctor of Philosophy Dissertation. Southern Illinois University.

Meentemeyer, V. 1978. Macroclimate and lignin control of litter decomposition rates. *Ecology*, **59**, 465-472.

Melillo, J.M., Prentice, I.C., Farquhar, E.-D., Schulze, O.E. & Sala, O.E. 1996. Terrestrial biotic responses to environmental change and feedbacks to climate. In: *Climate Change 1995. The Science of Climate Change*, edited by J.T. Houghton, L.G. Meira Filho, B.A. Callander, N. Harris, A. Kattenberg & K. Maskell. Cambridge: Cambridge University Press.

Metherell, A.K., Harding, L.A., Cole, C.V. & Parton, W.J. 1993. *CENTURY Soil Organic Matter Model Environment. Technical Documentation.* Fort Collins, Colorado: United States Department of Agriculture.

Milne, R. & Brown, T.A. 1997. Carbon in vegetation and soils of Great Britain. *Journal of Environmental Management*, **49**, 413-433.

Milne, R., Brown, T.A.W. & Howard, D.C. 1997. Emissions and uptake by UK terrestrial carbon reservoirs, 1990-2020. In: *Carbon Sequestration in Vegetation and Soils*, edited by M.G.R. Cannell. Institute of Terrestrial Ecology: Report to the Department of Environment.

Milner, C. & Hughes, R.E. 1968. *Methods for the Measurement of Primary Production of Grassland.* Oxford: Blackwell Scientific Publications.

Moore, P.D. 1975. Origin of blanket mires. *Nature*, **256**, 267-269.

Moore, P.D., Webb, J.A. & Collinson, M.E. 1991. *Pollen Analysis.* London: Blackwell.

Moseley, F., ed. 1978. *The Geology of the Lake District.* Leeds: Yorkshire Geological Society.

Newson, M. 1992. *Land, Water and Development.* London: Routledge.

Nicholls, N., Gruza, G.V., Jouzel, J., Karl, T.R., Ogallo, L.A. & Parker, D.E. 1996. Observed climate variability and change. In: *Climate Change 1995. The Science of Climate Change*, edited by J.T. Houghton, L.G. Meira Filho, B.A. Callander, N. Harris, A. Kattenberg & K. Maskell. Cambridge: Cambridge University Press.

Nilsson, M. 1991. *Fungi and Bacteria in Peat and Peat Forming Plant Communities*. Ph.D. Thesis. Swedish University of Agricultural Sciences, S-901, 83 Umea, Sweden.

Nilsson, M.B., Dabakk, E., Korsman, T. & Renberg, I. 1996. Quantifying relationships between near-infrared reflectance spectra of lake sediments and water chemistry. *Environmental Science and Technology*, **30**, 2586-2590.

Oechel, W.C., Hastings, S.J., Vourlitis, G., Jenkins, M., Riechers, G. & Grulke, N. 1993. Recent change of arctic tundra ecosystems from a net carbon sink to a source. *Nature*, **361**, 520-523.

Oechel, W.C., Vourlitis, G.L., Hastings, S.J., Ault, R.P. & Bryant, P. 1998. The effects of water table manipulation and elevated temperature on the net CO_2 flux of wet sedge tundra ecosystems. *Global Change Biology*, **4**, 77-90.

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

Oldfield, F., Thompson, R., Crooks, P.R.J., Gedye, S.J., Hall, V.A., Harkness, D.D., Housley, R.A., McCormac, F.G., Newton, A.J., Pilcher, J.R., Renberg, I. & Richardson, N. 1997. Radiocarbon dating of a recent high-latitude peat profile: Stor Åmyrân, northern Sweden. *Holocene*, **7**, 283-290.

Olson, J.S. 1963. Energy storage and the balance of producers and decomposers in ecological systems. *Ecology*, **44**, 322-331.

Overpeck, J.T., Rind, D. & Goldberg, R. 1990. Climate-induced changes in forest disturbance and vegetation. *Nature*, **343**, 51-53.

Parton, W.J., Scurlock, J.M.O., Ojima, D.S., Schimel, D.S., Hall, D.O. & Scopegram group members. 1995. Impact of climate change on grassland production and soil carbon worldwide. *Global Change Biology*, **1**, 13-22.

Pearsall, W.H. & Gorham, E. 1956. Production Ecology: 1. Standing crops of natural vegetation. *Oikos*, **7**, 193-201.

Pennington, W. 1997. Vegetational history. In: *A Flora of Cumbria*, edited by G. Halliday, 42-50. University of Lancaster: Centre for North-West Regional Studies.

Pepin, N.C. 1995. The use of GCM scenario output to model effects of future climatic change on the thermal climate of marginal maritime uplands. *Geografiska Annaler*, **77**, 167-185.

Perkins, D.F., Jones, V., Millar, R.O. & Neep, P. 1978. Primary production, mineral nutrients and litter decomposition in the grassland ecosystem. In: *Production Ecology of British Moors and Montane Grasslands*, edited by O.W. Heal & D.F. Perkins, 304-331. Berlin: Springer-Verlag.

Phillips, J., Yalden, D. & Tallis, J.H., eds. 1981. *Peak District Moorland Erosion Study. Phase 1 Report.* Bakewell: Peak Park joint planning board.

Pilcher, J.R. & Hall, V.A. 1996. Tephrochronological studies in northern England. *Holocene*, **6**, 100-105.

Post, W.M., Emanuel, W.R., Zinke, P.J. & Stangenberger, A.G. 1982. Soil carbon pools and world life zones. *Nature*, **298**, 156-159.

Prentice, K.C. & Fung, I.Y. 1990. The sensitivity of terrestrial carbon storage to climate change. *Nature*, **346**, 48-51.

Rawes, M. 1981. Further results of excluding sheep from high-level grasslands in the north Pennines. *Journal of Ecology*, **69**, 651-669.

Rawes, M. 1983. Changes in two high altitude blanket bogs after the cessation of sheep grazing. *Journal of Ecology*, **71**, 219-235.

Rawes, M. & Hobbs, R. 1979. Management of semi-natural blanket bog in the northern Pennines. *Journal of Ecology*, **67**, 789-807.

Rawes, M. & Welch, D. 1969. Upland productivity of vegetation and sheep at Moor House National Nature Reserve, Westmorland, England. *Oikos*, **11**, 7-72.

Reid, E., Mortimer, G.N., Lindsay, R.A. & Thompson, D.B.A. 1994. Blanket Bogs in Great Britain: An assessment of large-scale pattern and disturbance using remote sensing and GIS. In: *Large-Scale Ecology and Conservation*, edited by P.J. Edwards, R.M. May & N.R. Webb, 1st ed., 375. Oxford: Blackwell Science.

Renberg, I. & Wik, M. 1985. Soot particle counting in recent lake sediments. An indirect dating method. *Ecological Bulletins*, **37**, 53-57.

Rhodes, A.N. 1996. *Moorland Fire History from Microscopic Charcoal in Soils and Lake Sediments*. Doctor of Philosophy Thesis. University of Newcastle Upon Tyne.

Roberts, N. 1989. *The Holocene*. Oxford: Basil Blackwell.

Rose, N.L. 1990. A method for the extraction of carbonaceous particles from lake sediment. *Journal of Paleolimnology*, **3**, 45-53.

Rose, N.L. & Juggins, S. 1994. A spatial relationship between carbonaceous particles in lake sediments and sulphur deposition. *Atmospheric Environment*, **28**, 177-183.

Rowell, T.A. 1988. Burning. In: *The Peatland Management Handbook*, edited by T.A. Rowell, . Peterborough: Nature Conservancy Council.

SAS Institute Inc. 1985. *Users Guide: Statistics*. Cary, USA: SAS Institution Inc.

SAS Institute Inc. 1990. *SAS Procedures Guide, Version 6*. 3rd ed. Cary, NC, USA: SAS Institute Inc.

Schimel, D.S. 1995. Terrestrial ecosystems and the carbon cycle. *Global Change Biology*, **1**, 77-91.

Schimel, D., Alves, D., Enting, I., Heimann, M., Joos, F., Raynaud, D., Wigley, T., Prather, M., Derwent, R., Ehhalt, D., Fraser, P., Sanhueza, E., Vhou, X., Jonas, P., Charlson, R., Rodhe, H., Sadasivan, S., Shine, K.P., Fouquart, Y., Ramaswâmy, V., Solomon, S., Srinivasan, J., Albritton, D., Derwent, R., Isaksen, I., Lal, M. & Wuebbles, D. 1996. Radiative forcing of climate change. In: *Climate Change 1995*, edited by J.T. Hougton, L.G. Meira Filho, B.A. Callander, N. Harris, A. Kattenberg & K. Maskell, 65-131. Cambridge: Cambridge University Press.

Schimel, D.S., Braswell, B.H., Holland, E.A., McKeown, R., Ojima, D.S., Painter, T.H., Parton, W.J. & Townsend, A.R. 1994. Climatic, edaphic and biotic controls over storage and turnover of carbon in soils. *Global Biogeochemical Cycles*, **8**, 279-293.

Schimel, D., Enting, I.G., Heimann, M., Wigley, T.M.L., Raynaud, D., Alves, D. & Siegenthaler, U. 1995. CO_2 and the carbon cycle. In: *Climate Change 1994*, edited by J.T. Houghton, L.G. Meira Filho, J. Bruce, H. Lee, B.A. Callander, E. Haites, N. Harris & K. Maskell, 35-71. Cambridge: Cambridge University Press.

Schlesinger, W.H. 1984. Soil organic matter: a source of atmospheric CO_2. In: *The Role of Terrestrial Vegetation in the Global Carbon Cycle: Measurement by Remote Sensing*, edited by G.M. Woodwell, 111-127. Chichester: John Wiley and Sons Ltd.

Schlesinger, W.H. 1991. *Biogeochemistry. An analysis of Global Change.* San Diego: Academic press.

Shaw, E.M. 1994. *Hydrology in Practice.* London: Chapman and Hall.

Shenk, J.S. & Westerhaus, M.O. 1991. *ISI NIRS-2. Software for Near-Infrared Instruments.* Silverspring, USA: Infrasoft International.

Shine, K.P., Fouquart, Y., Ramaswamy, V., Solomon, S. & Srinivasan, J. 1995. Radiative forcing. In: *Climate Change 1994*, edited by J.T. Houghton, L.G. Meira Filho, J. Bruce, H. Lee, B.A. Callander, E. Haites, N. Harris & K. Maskell, 163-203. Cambridge: Cambridge University Press.

Smith, A.J.E. 1980. *The Moss Flora of Britain and Ireland.* Cambridge: Cambridge University Press.

Smith, P., Powlson, D.S., Glendining, M.J. & Smith, J.O.U. 1997. Potential for carbon sequestration in European soils: preliminary estimates for five scenarios using results from long-term experiments. *Global Change Biology*, **3**, 67-79.

Smith, R.A.H. & Forrest, G.I. 1978. Field estimates of Primary Production. In: *Production Ecology of British Moors and Montane Grasslands*, edited by O.W. Heal & D.F. Perkins, 17-37. Berlin: Springer-Verlag.

Smith, T.M. & Shugart, H.H. 1993. The transient response of terrestrial carbon storage to a perturbed climate. *Nature*, **361**, 523-526.

Stevenson, A.C., Jones, V.J. & Battarbee, R.W. 1990. The cause of peat erosion: a palaeolimnological approach. *New Phytologist*, **114**, 727-735.

Stockmarr, J. 1971. Tablets with spores used in absolute pollen analysis. *Pollen et Spores*, **13**, 615-621.

Stoneman, R. 1993. *Holocene Palaeoclimates from Peat Stratigraphy: Extending and Refining the Model.* Doctor of Philosophy Thesis. University of Southampton.

Street-Perrott, F.A. & Roberts, N. 1993. Past climates and future greenhouse warming. In: *The Changing Global Environment*, edited by N. Roberts. Oxford: Blackwell

Swift, M.J., Heal, O.W. & Anderson, J.M. 1979. *Decomposition in Terrestrial Ecosystems.* Oxford: Blackwell Scientific Publications.

Tallis, J.H. 1975. Tree remains in southern Pennine peats. *Nature*, **256**, 482-484.

Tallis, J.H. 1985. Mass movement and erosion of a southern Pennine blanket peat. *Journal of Ecology*, **73**, 283-315.

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. 1995. Climate and erosion signals in British blanket peats: the significance of *Racomitrium lanuginosum* remains. *Journal of Ecology*, **83**, 1021-1030.

Tate, K.R. 1992. Assessment based on a climosequence of soils in tussock grasslands of soil carbon storage and release in response to global warming. *Journal of Soil Science*, **43**, 697-707.

Tate, K.R., Giltrap, D.J., Claydon, J.J., Newsome, P.F., Atkinson, I.A.E., Taylor, M.D. & Lee, R. 1997. Organic carbon stocks in New Zealand's terrestrial ecosystems. *Journal of the Royal Society of New Zealand*, **27**, 315-335.

Taylor, K. & Marks, T.C. 1971. The influence of burning and grazing on the growth and development of *Rubus chamaemorus* L. in Calluna-Eriophorum bog. In: *The Scientific Management of Animal and Plant Communities for Conservation*, edited by E. Duffey & A.S. Watt, 153-166. Oxford: Blackwell Scientific.

Taylor, K., Woof, C., Ineson, P., Scott, W.A., Rigg, E. & Tipping, E. accepted. Variation in seasonal precipitation chemistry with altitude in the northern Pennines, UK. *Environmental Pollution.*

Tipping, E., Woof, C., Harrison, A.F., Ineson, P., Taylor, K., Rol, R., Harkness, D.D., Benham, D., Poskitt, J., Rigg, E. & Rowland, A.P. submitted. Climatic influences on the leaching of dissolved organic matter from upland UK moorland soils, investigated by a field manipulation experiment. *Environment International.*

Tolonen, K. & Turunen, J. 1996. Accumulation rates of carbon in mires and implications for climate change. *Holocene*, **6**, 171-178.

Trenberth, K.E., Houghton, J.T. & Meira Filho, L.G. 1996. The climate system: an overview. In: *Climate Change 1995*, edited by J.T. Houghton, L.G. Meira Filho, B.A. Callander, N. Harris, A. Kattenberg & K. Maskell, 51-64. Cambridge: Cambridge University Press.

Turner, J. 1987. Pollen diagrams from Cross Fell and their implications for former tree-lines. In: *Lake Sediments and Environmental History*, edited by E.Y. Haworth & J.W.G. Lund, 317-357. Leicester: Leicester University Press.

Turner, J. & Hodgson, J. 1979. Studies in the vegetational history of the northern Pennines. 1. Variations in the composition of the early Flandrian forests. *Journal of Ecology*, **67**, 629-646.

United Nations. 1993. *United Nations Framework Convention on Climate Change*. Geneva: UNEP/WMO.

Welch, D. 1974. *Aspects of the Ecology of the Northern Pennines: History*. Institute of Terrestrial Ecology: Merlewood Research and Development paper no. 7.

Welch, D. & Rawes, M. 1964. The early effects of excluding sheep from high-level grasslands in the north Pennines. *Journal of Applied Ecology*, **1**, 281-300.

Welch, D. & Rawes, M. 1965. The herbage production of some Pennine grasslands. *Oikos*, **16**, 39-47.

White, J.W.C., Ciais, P., Figge, R.A., Kenny, R. & Markgraf, V. 1994. A high-resolution record of atmospheric CO_2 content from carbon isotopes in peat. *Nature*, **367**, 153-156.

Whittaker, R.H. & Likens, G.E. 1973. Carbon in the Biota. In: *Carbon and the Biosphere*, edited by G.M. Woodwell & E.V. Pecan, 281-302. Springfield, USA: United States Atomic Energy Commission.

Whittaker, R.H. & Marks, P.L. 1975. Methods of Assessing Terrestrial Productivity. In: *Primary Productivity of the Biosphere*, edited by H. Leith & R.H. Whittaker, 1st ed., 55-118. New York: Springer-Verlag.

Wik, M. & Renberg, I. 1996. Environmental records of carbonaceous fly-ash particles from fossil-fuel combustion. *Journal of Paleolimnology*, **15**, 193-206.

Wildi, O. 1978. Simulating the development of peat bogs. *Vegetatio*, **37**, 1-17.

Wohlfarth, B., Skog, G., Possnert, G. & Holmquist, B. 1998. Pitfalls in the AMS radiocarbon-dating of terrestrial macro-fossils. *Journal of Quaternary Science*, **13**, 137-145.

Woodland, W.A., Charman, D.J. & Sims, P.C. 1998. Quantitative estimates of water tables and soil moisture in Holocene peatlands from testate amoebae. *Holocene*, **8**, 261-273.

Woodwell, G.M. 1984. The carbon dioxide problem. In: *The Role of Terrestrial Vegetation in the Global Carbon Cycle: Measurement by Remote Sensing*, edited by G.M. Woodwell, 3-17. Chichester: John Wiley and Sons Ltd.

Woodwell, G.M., Whittaker, R.H., Reiners, W.A., Likens, G.E., Delwiche, C.C. & Botkin, D.B. 1978. The biota and the world carbon budget. *Science*, **199**, 141-145.

10. Appendix

10.1 National Grid references for the study site

Square ref no.	National Grid ref
1	NY 371 533
2	NY 372 533
3	NY 373 533
4	NY 374 533
5	NY 370 532
6	NY 371 532
7	NY 372 532
8	NY 373 532
9	NY 374 532
10	NY 375 532
11	NY 376 532
12	NY 369 531
13	NY 370 531
14	NY 371 531
15	NY 372 531
16	NY 373 531
17	NY 374 531
18	NY 375 531
19	NY 376 531
20	NY 370 530
21	NY 371 530
22	NY 376 530

10.2 Method used to calculate true surface areas for the vegetation classes at Moor House

The stages involved in determining true surface and 2d areas for the different vegetation classes within the study site are outlined below. Further details can be obtained from ESRI (1991). The work was performed using Arc/Info version 7.

1) A three-dimensional model of the entire Moor House Nature Reserve was created in Arc/Info. The model was a Triangular Irregular Network (TIN) and was produced using Arc/Info's CREATETIN procedure, using the topology coverage of Moor House for input.

2) This TIN was 'flattened' using the TINARC command in Arc/Info. The 'flattened' triangles in the coverage had attributes of true surface area (Sarea) and 2d area.

3) The area attribute was copied to form a new attribute called 'oldarea' because this variable, the original 2d area of each triangle, was required for later calculations.

4) The vegetation coverage of the 22 one-kilometre squares of the study site was intersected with the 'flattened' TIN, to form a new coverage. This produced polygons which represented different vegetation classes. Each new polygon had attributes of vegetation class and 2d area, and also contained the values of sarea and 2d area belonging to the parent triangle from which it was formed.

5) The sarea of the newly created polygons was calculated within the INFO database part of Arc/Info. This new sarea value was named 'newsarea' and was calculated using the following equation:

$$\mathit{newsarea} = (\mathit{area} / \mathit{oldarea}) \times \mathit{sarea}$$

Where: ***newsarea*** = true surface area of the polygon containing a vegetation attribute
area = 2d area of the new polygon containing a vegetation attribute
oldarea = area of the parent triangle from which the polygon was derived
sarea = true surface area of the parent triangle from which the polygon was derived

The final coverage, therefore, had both 2d and true surface values for each polygon representing a different parcel of vegetation, allowing the 2d and true surface areas of different vegetation classes to be determined for both each kilometre square and the whole of the study site.

10.3 Area and C storage of vegetation at Moor House

Table 10.1 2d area of vegetation classes in each kilometre square

All units in m^2

	Kilometre square (see Fig 2 1)																						
Vegetation class	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	Total
Agrosto-Festucetum	13,800	1,904	0	0	47,029	1,126	153	18 995	29,980	44,834	0	38,149	31,023	113,386	15,238	19,573	31,197	0	0	131,011	58,761	5,461	601,618
Calcareous springs	7,848	16,629	0	0	11,178	13,713	630	1,191	0	3,771	4,064	1,352	6,492	2,506	4,473	4,823	353	0	937	1,854	12,478	4,191	98,485
Calluneto-Eriophoretum	134,018	443,378	623,538	600,737	0	0	133,925	675,067	531,819	623,653	760,948	0	0	0	20,059	348,406	446,732	447,230	831,504	0	0	754,578	7,375,591
Eriophoretum	393,291	96,692	14,270	7,352	0	48,007	523,225	1,503	0	49,774	5,862	237,949	263,880	38,514	548,608	284,290	91,431	1,389	0	194,966	90,935	4,402	2,896,339
Eroding bog	105,401	209,453	82,099	85,598	0	1,060	86,316	94,707	50,575	113,058	117,650	0	4,272	83,358	123,874	77,610	186,466	352,315	23,853	0	7,265	21,663	1,826,593
Festucetum	1,361	7,848	0	0	294,427	161,615	29,628	2,289	0	0	0	123,986	188,651	150,711	6,684	4,766	4,125	0	0	98,183	149,379	0	1,223,653
Flushed gleys	24,610	2,280	2,115	0	6,133	1,430	0	1,647	9,415	3,051	0	21,917	19,002	2,161	7,176	6,277	10,722	0	79,742	18,304	18,187	14,113	248,283
Juncetum squarrosus sa	69,411	37,293	2,118	3,289	256,152	288,006	66,040	25,599	11,766	1,594	9,443	116,401	82,614	217,160	56,662	3,476	29,466	16,416	16,156	88,763	302,042	7,981	1,707,845
Madeground	9,035	0	0	0	15,403	8,040	0	13,462	1,475	35,383	16,129	0	78,669	77,743	14,164	1,925	0	0	0	6,106	0	0	277,536
Nardetum sub-alpinum	130,107	42,312	14,641	17,999	235,632	344,546	53,423	18,731	23,690	19,037	12,336	270,772	206,845	158,694	89,004	188,660	54,504	0	23,932	322,309	232,367	111,321	2,570,862
Pteridietum	1,932	1,651	0	0	564	685	0	5,068	0	0	0	28,667	0	0	0	10,654	2,030	0	0	720	0	0	51,973
Recolonised peat	44,741	123,037	226,630	274,068	30,690	45,342	104,829	117,523	322,078	105,919	44,232	6,243	67,113	144,185	108,902	26,116	97,445	182,513	19,614	4,600	17,339	58,636	2,171,794
Sandstone scree	0	0	0	0	14,384	3,127	0	899	1,951	0	0	129,681	38,141	2,768	0	1,007	0	0	0	130,292	108,799	0	431,049
Sphagneto-Caricetum al	12,629	0	11,694	661	86,901	83,166	1,937	0	0	0	0	4,790	13,198	8,920	1,690	0	0	0	0	0	0	0	225,585
Sphagneto-Juncetum eff	51,848	17,470	22,880	10,476	1,684	0	0	23,304	17,451	146	27,442	20,239	0	0	3,328	22,560	45,269	0	4,415	2,778	2,500	17,473	291,262
Trichophoro-Eriophoretum	0	0	0	0	0	0	0	0	0	0	1,874	0	0	0	0	0	0	0	0	0	0	0	1,874
Total	1,000,029	999,947	999,985	1,000,180	1,000,177	999,863	1,000,106	999,984	1,000,200	1,000,220	999,982	1,000,147	999,900	1,000,106	999,860	1,000,142	999,740	999,862	1,000,154	999,885	1,000,053	999,819	22,000,342

Table 10.2 True surface area of vegetation classes in each kilometre square

All units in m^2

												Kilometre square (see Fig 2 1)											
Vegetation class	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	Total
Agrosto-Festucetum	13,974	1,917	0	0	49,996	1,129	153	19,045	30,217	45,009	0	41 583	32,235	120,575	15,446	19,695	31,288	0	0	143,921	62,475	5,582	634,239
Calcareous springs	7,899	16 782	0	0	11,526	13,909	637	1,196	0	3,776	4 094	1,451	6,644	2,576	4,489	4,874	357	0	937	1,897	12,836	4,314	100,196
Calluneto Eriophoretum	134,475	445,721	625,756	602,423	0	0	134,536	677,978	533,721	625,676	764,941	0	0	0	20,149	350,547	449,514	448,768	837,632	0	0	762,824	7,414,662
Eriophoretum	395,407	97,213	14,275	7,371	0	48,724	526,365	1,503	0	50,178	5,884	239,450	266,511	39,057	552,416	285,848	92,136	1,403	0	198,307	92,026	4,466	2,918,541
Eroding bog	105,815	210,114	82,293	85,746	0	1,097	86,862	95,019	50,734	113,480	118,167	0	4,342	83,639	124,632	78,032	187,316	353,275	23,979	0	7,745	21,961	1,834,248
Festucetum	1,414	7,886	0	0	302,715	163,885	29,724	2,303	0	0	0	130,386	191,549	153,316	6,782	4,815	4,146	0	0	105,981	153,352	0	1,258,254
Flushed gleys	24,773	2,296	2,126	0	6,484	1,469	0	1,653	9,465	3,058	0	22,454	19,159	2,178	7,232	6,354	10,777	0	80,935	19,032	18,653	14,184	252,282
Juncetum squarrosus sa	70,215	37,669	2,132	3,371	260,052	290,997	66,365	25,700	11,933	1,601	9,574	119,833	83,700	219,218	57,272	3,489	29,698	16,438	16,234	91,113	306,102	8,091	1,730,797
Madeground	9,206	0	0	0	16,199	8,346	0	13,517	1,475	35,489	16,171	0	79,764	79,215	14,305	1,952	0	0	0	6,332	0	0	281,972
Nardetum sub-alpinum	132,347	42,611	14,677	18,198	241,424	348,597	53,873	18,806	23,916	19,072	12,382	280,625	209,943	162,510	89,669	190,001	54,954	0	24,557	342,564	240,719	112,836	2,634,280
Pteridietum	1,940	1,660	0	0	594	695	0	5,111	0	0	0	29,982	0	0	0	10,687	2,032	0	0	738	0	0	53,439
Recolonised peat	45,009	123,294	227,382	274,414	31,489	45,698	105,133	117,887	322,804	106,171	44,273	6,495	67,872	145,028	109,419	26,257	97,740	183,007	19,663	4,656	17,968	59,220	2,180,877
Sandstone scree	0	0	0	0	14,697	3,145	0	899	1,953	0	0	135,884	38,506	2,824	0	1,014	0	0	0	139,500	116,377	0	454,799
Sphagneto-Caricetum al	12,792	0	11,739	661	87,874	84,502	1,954	0	0	0	0	4,938	13,493	9,025	1,728	0	0	0	0	0	0	0	228,707
Sphagneto-Juncetum eff	52,117	17,586	22,943	10,498	1,715	0	0	23,386	17,689	147	27,772	20,645	0	0	3,345	22,673	45,527	0	4,455	2,963	2,622	17,697	293,781
Trichophoro-Eriophoretum	0	0	0	0	0	0	0	0	0	0	1,877	0	0	0	0	0	0	0	0	0	0	0	1,877
Total	1,007,383	1,004,750	1,003,323	1,002,681	1,024,766	1,012,193	1,005,603	1,004,002	1,003,908	1,003,657	1,005,136	1,033,726	1,013,718	1,019,161	1,006,883	1,006,239	1,005,486	1,002,891	1,008,392	1,057,004	1,030,874	1,011,174	22,272,951

Table 10.3 C stored in vegetation classes based on 2d areas and mean estimates of vegetation C density, for each kilometre square

All units in kg

	Kilometre square (see Fig 2 1)																						
Vegetation class	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	Total
Agrosto-Festucetum	286	39	0	0	973	23	3	393	621	928	0	790	642	2,347	315	405	646	0	0	2,712	1,216	113	12,454
Calcareous springs	1,730	3,667	0	0	2,465	3,024	139	263	0	832	896	298	1,431	553	986	1,064	78	0	207	409	2,752	924	21,716
Calluneto-Eriophoretum	46,799	154,828	217,740	209,777	0	0	46,767	235,733	185,711	217,779	265,723	0	0	0	7,005	121,663	155,999	156,173	290,361	0	0	263,499	2,575,557
Eriophoretum	76,456	18,797	2,774	1,429	0	9,333	101,715	292	0	9,676	1,140	46,257	51,298	7,487	106,649	55,266	17,774	270	0	37,901	17,678	856	563,048
Eroding bog	36,806	73,141	28,669	29,891	0	370	30,142	33,072	17,661	39,480	41,083	0	1,492	29,109	43,257	27,101	65,114	123,028	8,330	0	2,537	7,565	637,846
Festucetum	44	251	0	0	9,422	5,172	948	73	0	0	0	3,968	6,037	4,823	214	153	132	0	0	3,142	4,780	0	39,157
Flushed gleys	5,426	503	466	0	1,352	315	0	363	2,076	673	0	4,833	4,190	477	1,582	1,384	2,364	0	17,583	4,036	4,010	3,112	54,746
Juncetum squarrosus sa	8,593	4,617	262	407	31,712	35,655	8,176	3,169	1,457	197	1,169	14,410	10,228	26,884	7,015	430	3,648	2,032	2,000	10,989	37,393	988	211,431
Madeground	187	0	0	0	319	166	0	279	31	732	334	0	1,628	1,609	293	40	0	0	0	126	0	0	5,745
Nardetum sub-alpinum	14,637	4,760	1,647	2,025	26,509	38,761	6,010	2,107	2,665	2,142	1,388	30,462	23,270	17,853	10,013	21,224	6,132	0	2,692	36,260	26,141	12,524	289,222
Pteridietum	878	751	0	0	256	311	0	2,304	0	0	0	13,029	0	0	0	4,842	922	0	0	327	0	0	23,622
Recolonised peat	15,623	42,965	79,139	95,705	10,717	15,833	36,606	41,039	112,470	36,987	15,446	2,180	23,436	50,349	38,028	9,120	34,028	63,734	6,849	1,606	6,055	20,476	758,391
Sandstone scree	0	0	0	0	460	100	0	29	62	0	0	4,150	1,221	89	0	32	0	0	0	4,169	3,482	0	13,794
Sphagneto-Caricetum al	2,785	0	2,578	146	19,162	18,338	427	0	0	0	0	1,056	2,910	1,967	373	0	0	0	0	0	0	0	49,741
Sphagneto-Juncetum eff	11,433	3,852	5,045	2,310	371	0	0	5,138	3,848	32	6,051	4,463	0	0	734	4,974	9,982	0	973	612	551	3,853	64,223
Trichophoro-Eriophoretum	0	0	0	0	0	0	0	0	0	0	159	0	0	0	0	0	0	0	0	0	0	0	159
Total	221,683	308,170	338,321	341,690	103,718	127,402	230,933	324,254	326,601	309,458	333,389	125,896	127,783	143,546	216,464	247,699	296,818	345,237	328,996	102,290	106,595	313,908	5,320,851

Table 10.4 C stored in vegetation classes based on true surface areas and mean estimates of vegetation C, by kilometre square

All units in kg

Vegetation class	Kilometre square (see Fig 2 1)																						
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	Total
Agrosto-Festucetum	289	40	0	0	1,035	23	3	394	625	932	0	861	667	2,496	320	408	648	0	0	2,979	1,293	116	13,129
Calcareous springs	1,742	3,700	0	0	2,542	3,067	140	264	0	833	903	320	1,465	568	990	1,075	79	0	207	418	2,830	951	22,093
Calluneto-Eriophoretum	46,959	155,646	218,514	210,366	0	0	46,980	236,750	186,376	218,486	267,118	0	0	0	7,036	122,411	156,970	156,710	292,501	0	0	266,378	2,589,200
Eriophoretum	76,867	18,898	2,775	1,433	0	9,472	102,325	292	0	9,755	1,144	46,549	51,810	7,593	107,390	55,569	17,911	273	0	38,551	17,890	868	567,364
Eroding bog	36,951	73,372	28,737	29,942	0	383	30,332	33,181	17,716	39,627	41,264	0	1,516	29,207	43,521	27,249	65,411	123,364	8,373	0	2,705	7,669	640,519
Festucetum	45	252	0	0	9,687	5,244	951	74	0	0	0	4,172	6,130	4,906	217	154	133	0	0	3,391	4,907	0	40,264
Flushed gleys	5,463	506	469	0	1,430	324	0	365	2,087	674	0	4,951	4,225	480	1,595	1,401	2,376	0	17,846	4,196	4,113	3,128	55,628
Juncetum squarrosus sa	8,693	4,663	264	417	32,194	36,025	8,216	3,182	1,477	198	1,185	14,835	10,362	27,139	7,090	432	3,677	2,035	2,010	11,280	37,895	1,002	214,273
Madeground	191	0	0	0	335	173	0	280	31	735	335	0	1,651	1,640	296	40	0	0	0	131	0	0	5,837
Nardetum sub-alpinum	14,889	4,794	1,651	2,047	27,160	39,217	6,061	2,116	2,690	2,146	1,393	31,570	23,619	18,282	10,088	21,375	6,182	0	2,763	38,539	27,081	12,694	296,356
Pteridietum	882	755	0	0	270	316	0	2,323	0	0	0	13,627	0	0	0	4,857	923	0	0	335	0	0	24,288
Recolonised peat	15,717	43,054	79,402	95,825	10,996	15,958	36,712	41,166	112,723	37,075	15,460	2,268	23,701	50,644	38,209	9,169	34,131	63,906	6,866	1,626	6,274	20,680	761,562
Sandstone scree	0	0	0	0	470	101	0	29	63	0	0	4,348	1,232	90	0	32	0	0	0	4,464	3,724	0	14,554
Sphagneto-Caricetum al	2,821	0	2,588	146	19,376	18,633	431	0	0	0	0	1,089	2,975	1,990	381	0	0	0	0	0	0	0	50,430
Sphagneto-Juncetum eff	11,492	3,878	5,059	2,315	378	0	0	5,157	3,900	32	6,124	4,552	0	0	738	5,000	10,039	0	982	653	578	3,902	64,779
Trichophoro-Eriophoretum	0	0	0	0	0	0	0	0	0	0	159	0	0	0	0	0	0	0	0	0	0	0	159
Total	222,999	309,558	339,459	342,492	105,874	128,936	232,152	325,570	327,689	310,492	335,084	129,143	129,352	145,035	217,870	249,172	298,480	346,287	331,549	106,564	109,291	317,387	5,360,435

Table 10.5 C stored in vegetation classes based on true surface areas and maximum estimates of vegetation C density, by kilometre square

All units in kg

Vegetation class	Kilometre square (see Fig. 2.1)																						
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	Total
Agrosto-Festucetum	669	92	0	0	2,395	54	7	912	1,447	2,156	0	1,992	1,544	5,776	740	943	1,499	0	0	6,894	2,993	267	30,380
Calcareous springs	2,339	4,969	0	0	3,413	4,118	189	354	0	1,118	1,212	430	1,967	763	1,329	1,443	106	0	277	562	3,801	1,277	29,668
Calluneto-Eriophoretum	61,872	205,076	287,910	277,175	0	0	61,900	311,938	245,565	287,874	351,949	0	0	0	9,271	161,287	206,822	206,478	385,395	0	0	350,975	3,411,486
Eriophoretum	125,621	30,885	4,535	2,342	0	15,480	167,226	478	0	15,942	1,869	76,073	84,671	12,408	175,503	90,814	29,272	446	0	63,002	29,237	1,419	927,221
Eroding bog	48,686	96,674	37,863	39,452	0	505	39,965	43,718	23,343	52,212	54,369	0	1,998	38,482	57,343	35,903	86,184	162,542	11,033	0	3,564	10,104	843,937
Festucetum	81	452	0	0	17,346	9,391	1,703	132	0	0	0	7,471	10,976	8,785	389	276	238	0	0	6,073	8,787	0	72,098
Flushed gleys	7,335	680	629	0	1,920	435	0	490	2,803	905	0	6,649	5,673	645	2,141	1,882	3,191	0	23,965	5,635	5,523	4,200	74,701
Juncetum squarrosus sa	18,151	9,738	551	872	67,223	75,223	17,155	6,644	3,085	414	2,475	30,977	21,637	56,668	14,805	902	7,677	4,249	4,197	23,553	79,127	2,091	447,411
Madeground	441	0	0	0	776	400	0	647	71	1,700	775	0	3,821	3,794	685	93	0	0	0	303	0	0	13,506
Nardetum sub-alpinum	32,465	10,453	3,600	4,464	59,221	85,511	13,215	4,613	5,866	4,678	3,037	68,837	51,499	39,864	21,996	46,607	13,480	0	6,024	84,031	59,048	27,679	646,189
Pteridietum	1,286	1,100	0	0	393	460	0	3,387	0	0	0	19,869	0	0	0	7,083	1,346	0	0	489	0	0	35,414
Recolonised peat	20,709	56,728	104,619	126,258	14,488	21,026	48,371	54,240	148,522	48,849	20,370	2,988	31,228	66,728	50,344	12,081	44,970	84,201	9,047	2,142	8,267	27,247	1,003,422
Sandstone scree	0	0	0	0	842	180	0	52	112	0	0	7,786	2,206	162	0	58	0	0	0	7,993	6,668	0	26,060
Sphagneto-Caricetum al	3,788	0	3,476	196	26,019	25,021	579	0	0	0	0	1,462	3,995	2,672	512	0	0	0	0	0	0	0	67,720
Sphagneto-Juncetum eff	15,432	5,207	6,794	3,108	508	0	0	6,925	5,238	44	8,223	6,113	0	0	990	6,714	13,481	0	1,319	877	776	5,240	86,989
Trichophoro-Eriophoretum	0	0	0	0	0	0	0	0	0	0	166	0	0	0	0	0	0	0	0	0	0	0	166
Total	338,873	422,052	449,977	453,865	194,545	237,803	350,311	434,528	436,052	415,892	444,446	230,647	221,214	236,747	336,047	366,085	408,265	457,916	441,256	201,555	207,791	430,500	7,716,367

10.4 Area and C storage of soil sub-types at Moor House

Table 10.6 2d area of soil sub-types in each kilometre square

All units in m^2 x 10^3

Soil sub type	Kilometre square (See Fig.2.1)																						
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	Grand Total
Calluneto Eriophoretum bog	119.5	437.1	620.2	599.1	0.0	0.0	130.1	659.0	519.6	617.7	747.8	0.0	0.0	0.0	20.0	330.7	440.5	447.5	790.2	0.0	0.0	754.6	7,233.7
Eriophoretum bog	372.7	96.5	14.3	7.4	0.0	46.6	522.0	1.5	0.0	49.5	5.7	219.5	196.2	32.4	535.2	275.3	89.7	1.4	0.0	145.2	80.1	4.4	2,695.6
Eroding bog	83.6	173.0	82.0	85.4	0.0	1.1	86.3	87.1	48.6	108.7	111.9	0.0	3.3	79.0	118.9	71.9	183.6	352.5	23.8	0.0	2.4	21.7	1,724.7
Juncetum bog	65.1	32.2	1.9	1.5	194.6	210.8	65.9	10.3	6.4	1.6	9.4	42.2	35.6	159.8	46.2	2.4	27.1	16.4	16.2	60.5	171.5	8.0	1,185.5
Nardetum bog	118.9	30.7	13.0	5.6	184.4	174.8	50.6	9.9	8.5	17.4	8.1	55.9	135.4	77.2	81.9	144.2	53.6	0.0	16.8	42.9	82.0	111.3	1,423.0
Recolonised bog	40.4	121.7	224.7	273.9	15.0	38.1	103.4	117.0	303.5	104.5	43.8	0.9	60.6	128.5	108.0	23.9	92.8	182.5	19.6	2.6	9.9	58.6	2,073.9
Other blanket bog	86.8	44.9	36.6	11.1	196.5	87.0	25.5	32.5	21.4	27.7	41.1	79.2	97.7	109.9	36.0	53.5	82.5	0.0	66.5	33.4	64.1	41.3	1,275.3
Brown earth	0.0	0.0	0.0	0.0	23.6	0.0	0.0	0.0	0.0	0.0	0.0	18.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	41.8
Coarse scree	0.0	0.0	0.0	0.0	17.6	0.0	0.0	0.0	0.0	0.0	0.0	269.2	88.0	24.8	4.3	53.2	11.9	0.0	0.0	496.5	246.2	0.0	1,211.7
Fell top podzol	0.0	19.6	0.0	0.0	12.3	23.6	14.1	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	69.6
Madeground	43.1	0.0	0.0	0.0	0.0	6.0	0.0	0.0	0.0	34.4	11.3	0.0	85.5	74.8	33.4	1.3	0.0	0.0	0.0	0.0	0.0	0.0	289.7
Mixed bottom lands	65.3	1.9	0.0	15.9	0.0	0.0	0.0	82.5	91.9	3.6	0.1	0.0	0.0	0.0	0.5	26.7	0.0	0.0	63.2	9.4	0.0	0.0	361.1
Peaty gley	4.6	36.0	0.0	0.0	190.4	372.6	0.0	0.0	0.0	0.0	0.0	42.5	141.0	112.5	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	899.7
Peaty gley-peaty podzol	0.0	0.0	0.0	0.0	81.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	113.8	123.7	10.7	0.0	0.0	0.0	0.0	0.0	166.2	0.0	495.6
Peaty podzol	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	209.0	17.9	0.0	0.0	0.0	0.0	0.0	0.0	138.2	12.3	0.0	377.4
Peaty podzol-brown earth	0.0	0.0	0.0	0.0	14.1	0.0	0.0	0.0	0.0	0.0	0.0	15.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	15.0	134.0	0.0	178.9
Red-brown limestone	0.0	6.5	7.3	0.0	31.4	0.0	2.4	0.4	0.0	34.7	7.7	48.2	20.4	62.7	4.8	16.9	18.2	0.0	3.8	56.2	31.1	0.0	352.4
Solifluxion creep	0.0	0.0	0.0	0.0	38.8	39.3	0.0	0.0	0.0	0.0	0.0	0.0	4.5	14.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	96.6
Valley bog	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.2
Grand Total	1,000.0	1,000.0	999.8	1,000.0	1,000.0	1,000.0	1,000.1	1,000.2	1,000.0	999.7	1,000.0	999.8	1,000.0	1,000.0	1,000.0	1,000.0	1,000.0	1,000.3	1,000.0	1,000.1	999.8	999.9	21,999.6

Table 10.7 C densities of different soil types to different soil depths

Soil sub-type	No. of sampling locations	Top 30 cm		Top 50 cm		Top 100 cm		Total depth	
		Mean	Standard error	Mean	Standard error	Mean	Standard error	Mean	Standard error
Calluneto-Eriophoretum bog	25	15.30	0.42	24.54	0.57	45.15	1.69	86.66	7.05
Eriophoretum bog	8	19.58	1.34	29.03	2.44	37.05	3.53	44.78	7.35
Eroding bog	4	13.37	4.33	20.51	6.30	28.74	9.64	50.96	30.22
Juncetum bog	4	16.09	3.62	23.94	6.75	31.83	11.23	31.83	11.23
Nardetum bog	5	16.03	1.67	26.27	4.29	36.98	5.99	37.18	6.04
Recolonised bog	4	15.68	2.90	21.28	4.05	29.15	9.30	45.45	25.01
Other blanket bog	ns	16.00	0.60	24.80	0.99	39.380	1.90	64.47	5.68
Brown earth	28	12.97	0.42	17.59	0.52	19.18	0.78	19.18	0.78
Coarse scree	3	11.60	1.99	11.60	1.99	11.60	1.99	11.60	1.99
Fell top podzol	3	13.56	4.62	13.56	4.62	13.56	4.62	13.56	4.62
Madeground	3	4.27	0.91	4.27	0.91	4.27	0.91	4.27	0.91
Mixed bottom lands	ns	8.00		12.40		19.69		32.24	
Peaty gley	34	12.11	0.79	15.39	1.21	16.33	1.40	16.33	1.40
Peaty gley-peaty podzol	ns	15.95		17.59		18.06		18.06	
Peaty podzol	5	19.79	3.36	19.79	3.36	19.79	3.36	19.79	3.36
Peaty podzol-brown earth	ns	16.38		18.69		19.49		19.49	
Red-brown limestone	16	9.84	1.28	11.96	1.71	11.99	1.72	11.99	1.72
Solifluxion creep	24	2.89	0.18	2.89	0.18	2.89	0.18	2.89	0.18
Valley bog	ns	4.66		7.02		16.65		256.41	

Notes

ns Soil types not sampled during the present study

Table 10.8 C stored in top 30 cm of soil sub-types, by kilometre square

All units in kg x 10^3

Soil sub-type	Kilometre square (See Fig.2.1) 1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	Grand Total
Calluneto-Eriophoretum bog	1,828	6,687	9,488	9,165	0	0	1,990	10,081	7,950	9,450	11,440	0	0	0	307	5,059	6,739	6,846	12,088	0	0	11,544	110,661
Eriophoretum bog	7,298	1,890	279	144	0	913	10,223	30	0	969	111	4,298	3,843	635	10,481	5,392	1,757	27	0	2,844	1,570	86	52,791
Eroding bog	1,117	2,313	1,096	1,142	0	14	1,153	1,164	650	1,453	1,496	0	44	1,056	1,590	961	2,455	4,713	319	0	32	290	23,059
Juncetum bog	1,047	518	30	25	3,132	3,392	1,060	166	103	26	152	679	573	2,571	743	38	436	264	260	974	2,759	128	19,078
Nardetum bog	1,906	492	208	90	2,956	2,801	811	158	137	278	129	896	2,171	1,237	1,312	2,311	859	0	269	688	1,314	1,783	22,808
Recolonised bog	634	1,908	3,522	4,294	235	598	1,621	1,834	4,758	1,638	686	14	950	2,014	1,693	374	1,454	2,862	308	41	155	919	32,512
Other blanket bog	1,389	718	586	178	3,143	1,391	408	521	342	444	657	1,267	1,563	1,757	576	857	1,320	0	1,063	534	1,025	661	20,400
Brown earth	0	0	0	0	306	0	0	0	0	0	0	236	0	0	0	0	0	0	0	0	0	0	543
Coarse scree	0	0	0	0	205	0	0	0	0	0	0	3,123	1,021	288	50	617	138	0	0	5,760	2,856	0	14,059
Fell top podzol	0	265	0	0	167	320	191	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	944
Madeground	184	0	0	0	0	26	0	0	0	147	48	0	365	319	143	6	0	0	0	0	0	0	1,236
Mixed bottom lands	523	15	0	128	0	0	0	659	735	29	1	0	0	0	4	213	0	0	505	76	0	0	2,888
Peaty gley	56	436	0	0	2,306	4,514	0	0	0	0	0	515	1,708	1,363	1	0	0	0	0	0	0	0	10,898
Peaty gley-peaty podzol	0	0	0	0	1,296	0	0	0	0	0	0	0	1,815	1,972	171	0	0	0	0	0	2,652	0	7,905
Peaty podzol	0	0	0	0	0	0	0	0	0	0	0	4,136	354	0	0	0	0	0	0	2,735	243	0	7,467
Peaty podzol-brown earth	0	0	0	0	231	0	0	0	0	0	0	246	0	12	0	0	0	0	0	246	2,195	0	2,930
Red-brown limestone	0	64	71	0	309	0	23	4	0	341	76	474	201	617	47	166	179	0	37	553	306	0	3,469
Solifluxion creep	0	0	0	0	112	114	0	0	0	0	0	0	13	40	0	0	0	0	0	0	0	0	279
Valley bog	0	0	0	0	0	0	0	0	0	0	61	0	0	0	0	0	0	0	0	0	0	0	61
Grand Total	15,982	15,305	15,281	15,165	14,397	14,083	17,480	14,618	14,675	14,773	14,858	15,884	14,621	13,884	17,117	15,995	15,339	14,711	14,849	14,452	15,107	15,412	333,989

Table 10.9 C stored in top 50 cm of soil sub-types, by kilometre square

All units in kg x 10^3

	Kilometre square (See Fig 2.1)																						
Soil sub-type	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	Grand Total
Calluneto-Eriophoretum bog	2,931	10,724	15,216	14,699	0	0	3,191	16,168	12,749	15,155	18,347	0	0	0	492	8,114	10,809	10,979	19,387	0	0	18,515	177,479
Eriophoretum bog	10,819	2,802	414	213	0	1,354	15,155	44	0	1,436	164	6,372	5,697	941	15,538	7,993	2,605	40	0	4,216	2,327	128	78,261
Eroding bog	1,714	3,548	1,682	1,752	0	22	1,769	1,786	998	2,230	2,295	0	68	1,620	2,438	1,475	3,766	7,230	489	0	50	445	35,374
Juncetum bog	1,558	771	45	37	4,659	5,046	1,576	247	154	38	226	1,011	853	3,824	1,106	57	649	393	387	1,449	4,105	191	28,381
Nardetum bog	3,124	807	340	147	4,844	4,590	1,329	259	224	456	212	1,469	3,557	2,027	2,151	3,788	1,408	0	441	1,128	2,153	2,923	37,377
Recolonised bog	860	2,590	4,781	5,829	319	812	2,200	2,490	6,459	2,223	932	19	1,290	2,734	2,298	508	1,974	3,885	418	56	211	1,248	44,137
Other blanket bog	2,153	1,113	909	276	4,872	2,157	633	807	530	688	1,019	1,965	2,423	2,724	892	1,328	2,047	0	1,649	828	1,590	1,024	31,625
Brown earth	0	0	0	0	416	0	0	0	0	0	0	320	0	0	0	0	0	0	0	0	0	0	736
Coarse scree	0	0	0	0	205	0	0	0	0	0	0	3,123	1,021	288	50	617	138	0	0	5,760	2,856	0	14,059
Fell top podzol	0	265	0	0	167	320	191	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	944
Madeground	184	0	0	0	0	26	0	0	0	147	48	0	365	319	143	6	0	0	0	0	0	0	1,236
Mixed bottom lands	810	24	0	198	0	0	0	1,022	1,139	45	1	0	0	0	7	331	0	0	783	117	0	0	4,477
Peaty gley	71	553	0	0	2,930	5,735	0	0	0	0	0	654	2,171	1,731	1	0	0	0	0	0	0	0	13,847
Peaty gley-peaty podzol	0	0	0	0	1,429	0	0	0	0	0	0	0	2,001	2,175	188	0	0	0	0	0	2,924	0	8,718
Peaty podzol	0	0	0	0	0	0	0	0	0	0	0	4,136	354	0	0	0	0	0	0	2,735	243	0	7,467
Peaty podzol-brown earth	0	0	0	0	264	0	0	0	0	0	0	280	0	14	0	0	0	0	0	281	2,505	0	3,344
Red-brown limestone	0	77	87	0	376	0	28	5	0	415	92	576	244	750	57	202	217	0	45	672	372	0	4,216
Solifluxion creep	0	0	0	0	112	114	0	0	0	0	0	0	13	40	0	0	0	0	0	0	0	0	279
Valley bog	0	0	0	0	0	0	0	0	0	0	93	0	0	0	0	0	0	0	0	0	0	0	93
Grand Total	24,225	23,274	23,474	23,152	20,591	20,176	26,073	22,829	22,254	22,832	23,429	19,924	20,057	19,191	25,361	24,418	23,614	22,527	23,598	17,242	19,334	24,473	492,050

Table 10.10 C stored in top 100 cm of soil sub-types, by kilometre square

All units in kg x 10^3

	Kilometre square (See Fig.2.1)																						
Soil sub-type	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	Grand Total
Calluneto-Eriophoretum bog	5,395	19,736	28,002	27,051	0	0	5,873	29,754	23,462	27,890	33,764	0	0	0	905	14,932	19,891	20,205	35,677	0	0	34,072	326,609
Eriophoretum bog	13,807	3,576	529	272	0	1,728	19,341	56	0	1,833	210	8,132	7,271	1,201	19,829	10,201	3,325	51	0	5,380	2,969	163	99,875
Eroding bog	2,402	4,972	2,356	2,454	0	31	2,479	2,502	1,398	3,124	3,215	0	95	2,269	3,417	2,066	5,277	10,130	685	0	69	623	49,565
Juncetum bog	2,071	1,025	60	49	6,194	6,709	2,096	329	204	51	300	1,344	1,134	5,085	1,470	76	863	522	514	1,927	5,458	254	37,735
Nardetum bog	4,399	1,136	479	207	6,819	6,463	1,871	365	316	642	299	2,068	5,008	2,854	3,028	5,333	1,983	0	620	1,588	3,032	4,115	52,624
Recolonised bog	1,178	3,547	6,549	7,985	437	1,112	3,014	3,411	8,848	3,045	1,276	26	1,767	3,745	3,148	696	2,704	5,321	572	77	289	1,709	60,456
Other blanket bog	3,419	1,767	1,443	439	7,737	3,425	1,005	1,282	842	1,092	1,617	3,120	3,847	4,326	1,417	2,109	3,250	0	2,618	1,315	2,524	1,626	50,220
Brown earth	0	0	0	0	453	0	0	0	0	0	0	349	0	0	0	0	0	0	0	0	0	0	803
Coarse scree	0	0	0	0	205	0	0	0	0	0	0	3,123	1,021	288	50	617	138	0	0	5,760	2,856	0	14,059
Fell top podzol	0	265	0	0	167	320	191	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	944
Madeground	184	0	0	0	0	26	0	0	0	147	48	0	365	319	143	6	0	0	0	0	0	0	1,236
Mixed bottom lands	1,287	38	0	314	0	0	0	1,624	1,809	71	2	0	0	0	10	525	0	0	1,244	186	0	0	7,110
Peaty gley	76	587	0	0	3,109	6,086	0	0	0	0	0	694	2,303	1,837	2	0	0	0	0	0	0	0	14,695
Peaty gley-peaty podzol	0	0	0	0	1,467	0	0	0	0	0	0	0	2,055	2,233	193	0	0	0	0	0	3,002	0	8,951
Peaty podzol	0	0	0	0	0	0	0	0	0	0	0	4,136	354	0	0	0	0	0	0	2,735	243	0	7,467
Peaty podzol-brown earth	0	0	0	0	275	0	0	0	0	0	0	292	0	15	0	0	0	0	0	293	2,612	0	3,487
Red-brown limestone	0	77	87	0	377	0	28	5	0	416	92	577	244	752	57	203	218	0	45	673	373	0	4,226
Solifluxion creep	0	0	0	0	112	114	0	0	0	0	0	0	13	40	0	0	0	0	0	0	0	0	279
Valley bog	0	0	0	0	0	0	0	0	0	0	219	0	0	0	0	0	0	0	0	0	0	0	219
Grand Total	34,217	36,726	39,505	38,771	27,353	26,013	35,898	39,327	36,879	38,310	41,044	23,861	25,478	24,968	33,669	36,763	37,649	36,230	41,976	19,935	23,427	42,562	740,558

Table 10.11 C stored in total depth of soil sub-types, by kilometre square

All units in kg x 10^3

	Kilometre square (See Fig.2.1)																						
Soil sub-type	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	Grand Total
Calluneto-Eriophoretum bog	10,355	37,880	53,747	51,922	0	0	11,272	57,110	45,034	53,532	64,806	0	0	0	1,737	28,661	38,179	38,782	68,479	0	0	65,398	626,894
Eriophoretum bog	16,687	4,322	639	329	0	2,088	23,375	68	0	2,215	254	9,828	8,787	1,451	23,965	12,328	4,018	62	0	6,503	3,589	197	120,706
Eroding bog	4,258	8,815	4,178	4,352	0	54	4,395	4,436	2,479	5,539	5,701	0	169	4,024	6,058	3,663	9,355	17,961	1,215	0	123	1,105	87,880
Juncetum bog	2,071	1,025	60	49	6,194	6,709	2,096	329	204	51	300	1,344	1,134	5,085	1,470	76	863	522	514	1,927	5,458	254	37,735
Nardetum bog	4,422	1,142	482	208	6,857	6,498	1,881	367	318	646	300	2,079	5,036	2,870	3,044	5,362	1,993	0	624	1,597	3,048	4,137	52,910
Recolonised bog	1,837	5,530	10,210	12,448	681	1,733	4,698	5,317	13,793	4,747	1,990	41	2,754	5,839	4,908	1,085	4,216	8,295	892	119	450	2,664	94,248
Other blanket bog	5,597	2,893	2,363	718	12,667	5,607	1,646	2,098	1,379	1,788	2,648	5,108	6,299	7,083	2,320	3,452	5,322	0	4,286	2,153	4,133	2,662	82,222
Brown earth	0	0	0	0	453	0	0	0	0	0	0	349	0	0	0	0	0	0	0	0	0	0	803
Coarse scree	0	0	0	0	205	0	0	0	0	0	0	3,123	1,021	288	50	617	138	0	0	5,760	2,856	0	14,059
Fell top podzol	0	265	0	0	167	320	191	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	944
Madeground	184	0	0	0	0	26	0	0	0	147	48	0	365	319	143	6	0	0	0	0	0	0	1,236
Mixed bottom lands	2,106	62	0	514	0	0	0	2,658	2,961	116	3	0	0	0	17	860	0	0	2,037	304	0	0	11,640
Peaty gley	76	587	0	0	3,109	6,086	0	0	0	0	0	694	2,303	1,837	2	0	0	0	0	0	0	0	14,695
Peaty gley-peaty podzol	0	0	0	0	1,467	0	0	0	0	0	0	0	2,055	2,233	193	0	0	0	0	0	3,002	0	8,951
Peaty podzol	0	0	0	0	0	0	0	0	0	0	0	4,136	354	0	0	0	0	0	0	2,735	243	0	7,467
Peaty podzol-brown earth	0	0	0	0	275	0	0	0	0	0	0	292	0	15	0	0	0	0	0	293	2,612	0	3,487
Red-brown limestone	0	77	87	0	377	0	28	5	0	416	92	577	244	752	57	203	218	0	45	673	373	0	4,226
Solifluxion creep	0	0	0	0	112	114	0	0	0	0	0	0	13	40	0	0	0	0	0	0	0	0	279
Valley bog	0	0	0	0	0	0	0	0	0	0	3,379	0	0	0	0	0	0	0	0	0	0	0	3,379
Grand Total	47,594	62,599	71,765	70,539	32,564	29,236	49,582	72,388	66,168	69,196	79,522	27,571	30,535	31,838	43,964	56,313	64,303	65,622	78,092	22,065	25,886	76,418	1,173,760

10.5 Principal component analysis of plant macro-fossils and testate amoebae

Table 10.12 Key to PCA sample depths for core MH2

PCA reference	Mid point (cm)	Depth range (cm)
A	3.5	0-7
B	8.5	7-10
C	11.0	10-12
D	13.0	12-14
E	15.5	15-16
F	18.5	17-20
G	21.5	21-22
H	23.5	23-24
I	25.5	25-26
J	27.5	27-28
K	29.5	29-30
L	31.5	31-32
M	33.5	33-34
N	35.5	35-36
O	37.5	37-38
P	39.5	39-40
Q	41.5	41-42
R	43.5	43-44
S	45.5	45-46
T	47.5	47-48
U	49.5	49-50
V	51.5	51-52
W	53.5	53-54
X	55.5	55-56
Y	57.5	57-58
Z	59.5	59-60
a	61.5	61-62
b	63.5	63-64
c	65.5	65-66
d	67.5	67-68
e	69.5	69-70
f	71.5	71-72
g	73.5	73-74
h	75.5	75-76
i	78.0	77-79
j	80.0	79-81
k	82.0	81-83

Table 10.13 Contribution of individual plant macro-fossils to the first three principal components

Plant macro-fossil	PRIN1	PRIN2	PRIN3
U.O.M.	0.31	0.08	0.05
Ericales roots	-0.02	0.35	0.22
Calluna vulgaris leaves	0.19	0.20	-0.10
Calluna vulgaris seed boxes	0.03	0.28	-0.02
Erica tetralix leaves	-0.16	0.37	-0.22
Erica tetralix wood	-0.07	0.22	-0.21
Erica tetralix seeds	-0.10	-0.07	0.39
Monocots. undiff.	0.24	-0.06	-0.03
Eriophorum vaginatum roots	0.27	0.07	0.08
Eriophorum vaginatum spindles	0.29	0.07	0.05
Eriophorum vaginatum epidermis	0.10	0.04	0.03
Eriophorum angustifolium epidermis	-0.05	-0.25	-0.39
Racomitrium lanuginosum	0.21	0.04	0.04
Identifiable *Sphagnum*	-0.32	0.03	0.13
Sphagnum section Acutifolia	-0.11	0.41	-0.22
Sphagnum imbricatum	0.22	0.08	0.12
Sphagnum magellanicum	-0.17	0.39	-0.13
Sphagnum papillosum	-0.19	-0.07	0.50
Sphagnum section Cuspidata	-0.10	-0.32	-0.22
Sphagnum spore capsules	0.13	0.09	0.14
Aggregate monocots.	0.23	-0.18	-0.30
<15,625 μm^2	0.16	0.04	0.10
15,625-62,500 μm^2	0.19	0.11	0.10
62,500-0.25 mm^2	0.29	0.03	-0.01
1-4 mm^2	0.28	0.02	0.02
>4 mm^2	0.15	0.00	-0.05

Table 10.14 Contribution of individual testate amoebae and other faunal remains to the first three principal components

Testate amoebae	PRIN1	PRIN2	PRIN3
Amphitrema flavum	-0.18	0.16	0.05
Amphitrema wrightianum	-0.28	0.00	0.19
Arcella catinus	0.23	0.13	0.09
Arcella discoides	0.21	-0.07	0.32
Arcella rotundata var. aplanata	0.16	0.36	-0.11
Assulina muscorum	-0.08	-0.26	-0.21
Assulina seminulum	0.14	-0.05	-0.42
Bullinaria indica	0.05	-0.12	-0.28
Centropyxis aculeata	-0.16	0.03	0.14
Centropyxis arcelloides type	-0.27	-0.11	0.03
Corythion dubium	0.23	-0.15	0.32
Difflugia bacillifera	-0.05	0.16	-0.03
Difflugia bacillilarium	-0.03	0.12	0.03
Difflugia lithophila	-0.11	0.12	0.08
Euglypha strigosa	0.33	-0.03	0.23
Heliopera rosea/sphagni type	0.22	-0.11	0.04
Hyalosphenia elegans	0.03	0.12	-0.09
Hyalosphenia papilio	0.13	0.33	-0.07
Hyalosphenia subflava	-0.06	-0.17	-0.27
Nebela carinata	-0.23	-0.07	0.13
Nebela flabellulum	0.19	-0.12	0.07
Nebela griseola	0.15	0.30	-0.07
Nebela militaris	0.25	0.27	0.00
Nebela parvula	0.19	-0.12	0.17
Nebela tincta	0.16	-0.14	0.27
Nebela vitraea	0.08	0.26	-0.13
Trigonpyxis arcula	0.08	-0.18	-0.26
Cladocera head shield	-0.30	0.15	0.21
Cladocera postabdomen	-0.26	0.21	0.11
Copepod spermatophore	-0.03	0.32	-0.02